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PRE-DECISIONAL WORKING DRAFT

# Biological Status of the Olympic Peninsula Steelhead Distinct Population Segment:

## Report of the Status Review Team

October 2024

U.S. DEPARTMENT OF COMMERCE

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

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# Biological Status of the Olympic Peninsula Steelhead Distinct Population Segment: Report of the Status Review Team

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## Abbreviations

CFS cubic feet per second

CMS cubic meters per second

DIP demographically independent population

DPS distinct population segment

ESA U.S. Endangered Species Act

ESU evolutionarily significant unit

IP intrinsic potential

IPM integral projection model

MARSS multivariate autoregressive state-space

NFH National Fish Hatchery

NMFS National Marine Fisheries Service

NPS National Park Service

ONP Olympic National Park

OP Olympic Peninsula

PVA population viability analysis

QET quasi-extinction threshold

SAS smolt-to-adult survival

SaSI Salmonid Stock Inventory

SPOIR significant portion of its range

SRT Status Review Team

SSH summer-run steelhead

USFWS U.S. Fish and Wildlife Service

VSP viable salmonid population

WDFW Washington Department of Fish and Wildlife

WSH winter-run steelhead

# Executive Summary

In response to a petition to the Secretary of Commerce to list the Olympic Peninsula Steelhead (*Oncorhynchus mykiss*) Distinct Population Segment (DPS) as a threatened or endangered species under the Endangered Species Act, the National Marine Fisheries Service (NMFS) convened a biological review team (SRT) to reassess the configuration and status of this DPS. The SRT was set with five specific tasks:

1. Evaluate the DPS configuration
2. Complete a demographic risk analysis
3. Review and comment on the threats analysis compiled by the West Coast Region
4. Complete the extinction risk synthesis
5. Conduct a Significant Portion of Its Range (SPOIR) analysis, depending on the outcome of (4), evaluate whether either ESU is at moderate or high risk of extinction in a significant portion of its range

The SRT reviewed information relevant to the configuration (boundaries) and risk of extinction for this DPS, including: the biological and demographic status of natural-origin Olympic Peninsula steelhead, past and current harvest and hatchery operations, watershed habitat conditions, past and present fisheries harvest and past and present land use. In addition, observed and predicted environmental effects due to climate change were assessed.

The SRT met several times (virtually) with representatives from the Washington Department of Fish and Wildlife (WDFW), the Northwest Indian Fisheries Commission, and Tribal Nations within the Olympic Peninsula Steelhead DPS or with treaty/management interests within the DPS. In addition, there were presentations by other state and federal agencies, and non-governmental entities on habitat conditions and restoration actions.

## DPS Configuration

The first task of the SRT was to review the configuration of the DPS as defined by Busby et al. (1996). The current DPS includes both winter- and summer-run steelhead populations in the Olympic Peninsula west of the Elwha River and from the Copalis River northward (Figure 1). There was limited new (post-Busby) genetic and life history information available specific to steelhead populations in the Olympic Peninsula and adjacent areas. In general, what information was available did not suggest a plausible alternative DPS configuration. For example, the SRT considered separate DPSs for winter-run and summer-run steelhead populations, as was suggested by the petitioners, but did not find the life history differences warranted a reconfiguration. This decision was also informed by the results of Waples et al. (2022), who examined using run timing as a primary factor in creating distinct DPS and ESU for West Coast salmonids, and concluded that in most cases run-timing alone was not a compelling factor in distinguishing major conservation units. Finally, the SRT considered existing listing unit delineations for other anadromous salmonid species (coho salmon: Weitkamp et al. 1995, Chinook salmon: Myers et al. 1998); these boundaries comported with some or all of the geographic/ecological delineations identified by Busby et al. (1996). The SRT was unanimous in maintaining the existing DPS configuration.


Figure 1. Olympic Peninsula Steelhead DPS as identified in Busby et al. 1996.

## Demographic Risk Analysis

The NMFS had previously reviewed the coastwide status of steelhead (anadromous *Oncorhynchus mykiss*) in 1996, and at that time identified 15 DPSs[5] within the contiguous United States, including the Olympic Peninsula Steelhead DPS, and it was the conclusion of the Status Review Team (SRT) at the time that the Olympic Peninsula Steelhead DPS was not a risk of extinction now or in the foreseeable future (Busby et al. 1996).

Analysis of data relevant to the status of the Olympic Peninsula Steelhead DPS was limited by the varying levels of data quantity and quality for each of the 39 steelhead populations (29 winter-run, 10 summer-run populations) identified in SASSI (WDF et al. 1993). There was very little information available for summer-run populations with which to evaluate their status using the viable salmonid population (VSP) categories. Intermittent snorkel surveys of prespawning adults represented the primary indicators of abundance, with little or no information on harvest, spawning distribution, genetics or productivity. Information on winter-run populations was more complete overall, but some river systems were still lacking in spawner abundance data (e.g. redd[6] counts). Even where redd surveys were undertaken, only redds created after March 15$^{th}$ were included in the natural spawner abundance estimates provided by the co-managers. The use of the March 15$^{th}$ cutoff date to distinguish natural and hatchery-origin spawning winter-run steelhead was a key source of SRT uncertainty in the accuracy of population abundances.

The SRT reviewed information relevant to the relationship between hatchery-origin and natural-origin steelhead, especially because the broodstocks for many hatchery programs originated from, or were strongly influenced by, sources outside of the DPS. These out-of-DPS hatchery stocks were apparently selected by local resource managers because of differences in run and spawn timing between the hatchery broodstocks and the native populations. This temporal separation was the basis for harvest strategies that targeted hatchery-origin steelhead and assumed limited genetic introgression between hatchery spawning naturally and native steelhead. Overall, the SRT concluded that there is evidence for substantial overlap between returning hatchery and native winter-run steelhead, and that contrary to management intent, non-selective harvest has a considerable adverse effect on natural-origin winter-run population abundance.

One consequence of the harvest strategy targeting earlier returning hatchery-oriented winter steelhead is the removal of early-returning native winter-run steelhead and a gradual shift in the overall run timing of native fish to later dates. The continued harvest of this early returning natural-origin component may ultimately be expressed as changes in the geographic distribution of spawners and a shift in spawn timing. Further, it is also likely that there is a continued opportunity for interbreeding between hatchery and native steelhead, although the necessary genetic studies to evaluate this have not been undertaken. Currently, there is no direct harvest of native-summer-run fish in the Olympic Peninsula DPS, although there is a fishery for hatchery-origin summer-run fish on the Quillayute River. The SRT was unable to

[5] Initially, the listing unit for steelhead was the Evolutionarily Significant Unit (ESU), but under a later joint agreement with the U.S. Fish and Wildlife Service, the current ESA listing unit for *O.mykiss* is the Distinct Population Segment (DPS).

[6] Redds are gravel areas in stream where salmonids build "nests" to deposit their eggs for incubation.

establish from the harvest data provided by co-managers whether there was a by-catch of summer-run steelhead harvest in the summer and fall salmon fisheries or in the on-reservation recreational fishery. Historical estimates of the summer-run fisheries in the DPS prior to the initiation of the Bogachiel Hatchery summer-run program in the Quillayute Basin suggest much higher summer-run abundances than are currently roughly estimated. Many of the risks identified by the SRT were related to the direct and indirect consequences of existing harvest and hatchery policies.

The SRT found that habitat conditions in the DPS have improved since the Busby et al. (1996) review. Habitat improvements were ascribed to improvements in land use and timber harvest regulations and policies and widespread restoration efforts, but the legacies of earlier practices are still limiting habitat quality. Land management provided by the Olympic National Park has, and will continue to, provide habitat protection to many headwater areas. Lastly, although there is still some uncertainty in the overall effects of climate change on freshwater and ocean habitat, since Busby et al. (1996) there have already been marked decreases in glacial coverage, increases in summer stream temperatures, decreases in summer stream hydrology, and deleterious changes in ocean conditions (NWIFC 2020). These trends are expected to continue, and within the 40-50 years “foreseeable future” identified by the SRT, will increasingly be a threat to steelhead populations in the DPS.

Of the 39 steelhead populations identified in the DPS, there was sufficient information to calculate abundances and trends for 15 populations, all of which were winter run. While the number of populations examined was numerically small, they do account for the vast majority of steelhead abundance in the DPS. The SRT also considered the effect of past and present (to 2022) hatchery operations and harvest, as well as other relevant data. Following a review and discussion of the information available, SRT members evaluated the viability of individual steelhead populations in the DPS using the four Viable salmonid population (VSP) categories: abundance, productivity, spatial structure, and diversity (McElhany et al. 2000). Where possible, each category was assigned a risk of extinction level from 1-5 (1: low risk, 5-high risk). In addition, SRT members estimated the relative effect of the ESA factors for decline (threats): habitat loss and destruction, over-utilization, disease and predation, inadequacy of existing regulatory mechanisms, hatchery effects, climate change. The threat (risk) from each of these factors was rated similarly to VSP categories. Individual population assessments for VSP parameters and threats provided a basis for assessing the overall risk of extinction to the DPS and subsequent evaluations of risk to significant portions of the range (SPOIR). In addition to assessing the risk status of the DPS as a whole, the team also evaluated whether there were *significant portions of the range* (SPOIR) of the DPS that are at a higher risk of extinction than the DPS. In doing this, the team followed advice from the NMFS WCR and NMFS Office of Protected Resources on how to interpret the phrase “significant portion of its range” in light of the 2014 joint Fish and Wildlife and NOAA SPOIR policy (79 FR 37578) and subsequent legal rulings.

## Population VSP Evaluation

In their evaluation of the VSP parameters for all populations within the DPS, the unweighted averages were in the moderate-range for: a) abundance (2.2) and b) productivity (2.9) and c) diversity (2.3), and low for d) spatial structure (1.3). The scores for winter run steelhead populations from the four major coastal tributaries[7] ["Big Four"]: were much lower than the DPS population average, which reflects that fact that these larger rivers contain the numerical majority of steelhead in the DPS, only in diversity did they have risk scores higher than the overall average for the DPS. Each of the Big Four rivers contains large winter-run hatchery programs. Risk scores were also very high collectively for summer-run steelhead populations in the DPS, due to their low population abundances, limited habitat, and susceptibility to hatchery introgression. Similarly, the steelhead populations in the smaller tributaries that drain to the Strait of Juan de Fuca, had relatively high-risk scores in abundance and productivity. Some SRT members did not provide risk scores for some population's VSP categories, because they concluded that there was insufficient information to make an informed score.

## Population Threats Evaluation

The evaluation of threats for all populations identified climate change (3.1/5.0), inadequate regulation (2.9/5.0), and overutilization (2.5/5.0) as the top threats. Habitat Loss or destruction (2.1/5.0), disease/predation (1.1/5.0) and hatchery effects (2.1/5.0) were ranked as lesser threats to the DPS. Climate change was universally seen by the SRT members as the primary threat, with the ongoing and future loss of glaciers and declines in summer flows identified as the major freshwater climate change effects, in addition to projected declines in ocean productivity. Overutilization, inadequate regulation, and hatchery effects were identified as significant threats to winter-run steelhead, especially in the Big Four river systems, where the majority of the DPS abundance resides. High harvest rates, potentially outdated capacity (escapement goal) estimates, use of non-native hatchery stocks, and lack of adequate marking of hatchery fish influenced these higher risk scores. For summer-run steelhead, the absence of any comprehensive management or monitoring plan (e.g. inadequate regulation) for these low abundance, niche-specific, populations is seen as a major threat, as was climate change. The SRT consensus was that three of these threats (overutilization, inadequate regulation, and hatchery effects) to steelhead viability in Olympic Peninsula DPS could be directly addressed through management and operational changes.

## Olympic Peninsula DPS Risk Evaluation

The SRT reviewed and discussed the VSP category scores and the Threats scores in developing their final DPS risk scores. In examining the risk of this DPS, the SRT considered not only the current status, but how the status had changed since the last review by Busby et al. (1996), and concluded that the Olympic Peninsula Steelhead DPS is a t moderate risk of extinction because;

1) Escapement had declined in most populations since the previous Status Review (1996). In 1996, of the 12 populations for which trends could be calculated (1991-1995), 7 populations were found to be declining, and 5 populations increasing (Busby et al. 1996). In contrast, currently of the 14 populations for which five-year trends could be calculated (2018-2022), no populations were increasing, 1 population was stable and 13 populations were declining (10 of which had trends that were significantly different from 0).

[7] The Quillayute River, Hoh River, Queets River, and Quinault River

2) Run size (escapement + harvest), which is only available for winter-run populations in the four major rivers (Quillayute, Hoh, Queets, and Quinault rivers), has declined by 42%, from 32,556 (1991-1995, the time of the previous status review) to 18,821 (2018-2022).
3) Kelt survival rates went from ~20% to ~12% since 1996 (in the four larger basins). This likely has had a negative effect on overall population reproductive potential as kelts have a disproportionate influence on population productivity, spawning multiple times and with a higher fecundity than maiden (first-time) spawners (Jenkins et al. 2018).
4) Harvest rates on natural-origin steelhead have been excessive for many winter-run populations (Quillayute, Hoh, Queets, and Quinault basins with harvest rates averaging 20%-45% from 1996-2020). At these harvest rates, populations in the four major basins are below replacement. In recent years (2020-2022), harvest rates were lowered (~10% on average in the 4 major basins) due to low forecasted returns, near or below escapement goals. Some SRT members expressed concern in the uncertainty that harvest rates would remain relatively low, while other members were concerned that the populations had not responded more positively to the decrease in harvest rate. It was noted by the SRT that the period of decrease in harvest rates has been relatively short (less than a generation) and with only two or three years of data it is too early to evaluate the demographic responses by these populations to this change.
5) Summer-run populations are effectively unmonitored for escapement and direct or indirect harvest are likely persist at low abundance levels. Available information suggests that summer-run populations are at a level where the risks of catastrophic events and demographic processes (i.e. Allee effects) are of concern. The summer-run life history was viewed as an important diversity characteristic.
6) None of the summer or winter steelhead hatchery stocks in the DPS were considered as part of the DPS; hatchery effects (introgression and reduced fitness) from these hatchery stocks are largely unmonitored and likely deleterious to the natural-origin steelhead populations, due to maladapted (non-native or domestication-related) life-history traits.
7) Climate change has and will continue to have a deleterious effect on DPS viability. This decline in habitat quality may outweigh improvements in land management and restoration efforts given the current rate of climate change effects (higher temperatures, changes in flow, melting of glaciers) and the estimated timeline for recovery of existing habitat degradations could range between 100 and 225 years (Stout et al. 2018, Martens and Devine 2023).
8) The negative trends in run size observed were in spite of the moderate to good conditions the SRT noted in river and riparian habitat, especially those rivers with substantial portions being located within the Olympic National Park. Further, protections provided by State and Federal forest lands provide some assurance of continued stable habitat protection. Other watersheds were still predominantly forested and despite recent habitat improvement efforts, the legacy of past industrial logging practices will continue to negatively affect

> steelhead productivity in a number of rivers for the foreseeable future. There have been widespread habitat restoration actions to address legacy land-use effects, although the benefits of these may not have manifested themselves.

Six of the eight SRT members placed the majority of their risk likelihood points in the moderate risk of extinction, one member placed the majority of their risk likelihood point in the low-risk category (6/10), and one member was evenly split between low and medium risk.

The final conclusion of the SRT was that the Olympic Peninsula Steelhead DPS was at a moderate risk of extinction.

## Olympic Peninsula Significant Portion of the Range Assessment

Following the determination of overall risk to the DPS, the SRT identified presumptive "significant portions" of the DPS to evaluate as part of the Significant Portion Its Range (SPOIR) risk analysis. The SRT ultimately decided on evaluating two SPOIR scenarios . One scenario was based on major life history traits, specifically using run-timing portions: populations exhibiting summer-run (stream-maturing) or winter-run (ocean-maturing) life histories. In deciding upon the significance of each portion, the majority of the SRT members place the majority of their likelihood points in the "not significant" category for summer-run steelhead populations and "significant" for winter-run populations.

The SRT also discussed and assessed a SPOIR scenario based on biogeography In this case, the geographic units included: 1) steelhead populations in rivers that drained to the Strait of Juan de Fuca, and 2) steelhead population in rivers that drained to the Pacific Ocean. These two regions were identified as potential portions due the hydrological and geographic distinctiveness or the rivers supporting Strait populations and coastal populations. The majority of the SRT members assigned a majority of their likelihood points in the "not significant" category for populations draining to the Strait of Juan de Fuca. The coastal populations were considered a "significant" portion under SPOIR policy.

For the winter-run steelhead population and coastal population portions identified as significant, the risk of extinction was determined to not be higher than that of the entire DPS.

In summary, the Olympic Peninsula Steelhead SRT concluded that the DPS was at moderate risk of extinction throughout its range. The team also reviewed potentially significant portions of the DPS, identified SPRs based on run timing and biogeography, and concluded that none of the significant portions was at a higher risk of extinction than the DPS, and therefore not change in risk status was prescribed.

# Introduction

## Petition to List

On 1 August 2022, the Conservation Angler and the Wild Fish Conservancy petitioned the Secretary of Commerce to list the Olympic Peninsula Steelhead (*Oncorhynchus mykiss*) Distinct Population Segment (DPS) as a threatened or endangered species under the Endangered Species Act (TCA and WFC 2022).

The Petition asserts that the biological status of the DPS has declined such that it warrants protection under the Endangered Species Act (ESA). The petitioners point to the four viability components framed by McElhany et al (2000) for viable salmonid populations (VSP): abundance, productivity, diversity, and spatial structure. Further, the petitioners identified multiple examples of the ESA section 4(a)(1) listing factors that may be threatening the DPS:

1. Present of threatened destruction, modification, or curtailment of its habitat or range;
2. Overutilization for commercial and recreational purposes;
3. Disease and predation;
4. Inadequacy of existing regulatory mechanisms; and
5. Other natural or anthropogenic factors.

The Petition does not request a reevaluation of the definition of the Olympic Peninsula Steelhead DPS.

### Petitioners' Risk Assessment

The petitioners presented information on steelhead demographics, management, and marine and freshwater ecosystem conditions, based on published and unpublished sources. The petitioner's assert that there are 30 steelhead populations (26 winter run and 4 summer run populations) in the Olympic Peninsula DPS. Recent abundance information was presented for approximately half (15) of the populations, all of which were winter run. Of those populations, only 20% (3) exhibited increasing trends from 1980-2013 based on Cram et al. (2018). Summer run steelhead populations have not been systematically monitored, although the petitioners presented summer snorkel data that suggests summer-run abundance is very low (<100) for most populations. Furthermore, the petitioners presented historical (circa 1950s) estimates of abundance that suggest population declines since that time have been substantial (61-81%) for the four largest winter-run populations. The petitioners also cited a number of diversity risks related to hatchery operations, the release of out-of-DPS stocks of fish, and the incidental harvest of naturally-produced[8] fish co-occurring with returning "early-returning" hatchery-origin fish. The petitioners assert that hatchery operations have resulted in the dilution of native genetics and a reduction in run timing diversity through the harvest of natural-origin fish late autumn and early winter.

[8] In this document we have not used the term "wild" to describe naturally-produced steelhead. Wild can suggest the absence of any anthropogenic influences (hatchery origin or introgression, direct or indirect selection). In the absence of a historical genetic and phenotypic baseline and present-day sampling it is not possible to make that determination. Where other authors have used the term we have retained "wild" in quotes.

## NMFS 90-Day Finding and Initiating the Olympic Peninsula Status Review

The National Marine Fisheries Service (NMFS) concluded that the petition presented substantial scientific and commercial information indicating the petitioned actions may be warranted. [Federal Register 50 CFR 223-224, 88 FR 8774-8785 (10 February 2023)].

In response to the petition on January 6, 2023, the NMFS West Coast Region (WCR) requested that the Northwest Fisheries Science Center (NWFSC) conduct an analysis and review of the petition's claim that Olympic Peninsula Steelhead DPS is at risk of extinction and warrants listing as a threatened or endangered species under the ESA. The SRT was set with five specific tasks:

1. Evaluate the DPS configuration
2. Complete a demographic risk analysis
3. Review and comment on the threats analysis compiled by the West Coast Region
4. Complete the extinction risk synthesis
5. Conduct a Significant Portion of Its Range (SPOIR) analysis, depending on the outcome of (4), evaluate whether either ESU is at moderate or high risk of extinction in a significant portion of its range

The Northwest Fisheries Science Center (NWFSC) convened a SRT in 2023, with scientists from the NWFSC, Southwest Fisheries Science Center (SWFSC), West Coast Region of the NMFS (WCR NMFS), and National Park Service (NPS). The SRT reviewed information relevant to the configuration (boundaries) and risk of extinction for this DPS, including: the biological and demographic status of natural-origin Olympic Peninsula steelhead, past and current harvest and hatchery operations, watershed habitat conditions, past and present fisheries and land use regulations, in addition to estimates of the effects of climate change. The SRT utilized information from published sources (peer-reviewed articles and agency and tribal reports), information submitted by State, Tribal, and Federal agencies, information presented to the SRT in technical meetings, and traditional ecological knowledge (TEK), in developing their risk analysis. The SRT met several times (virtually) with representatives from the Washington Department of Fish and Wildlife (WDFW), the Northwest Indian Fisheries Commission, and Tribal Nations within the Olympic Peninsula Steelhead DPS or with treaty/management interests within the DPS. In addition, there were presentations by other state and federal agencies, and non-governmental entities. This report presents the information reviewed and analyzed by the SRT, as well as the process by which they made their DPS configuration and risk determinations.

# DPS Configuration

## NMFS DPS Policy

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

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

## Establishment of the DPS

The Olympic Peninsula steelhead DPS[9] was established in 1996 (61 Fed Reg 41544 (Aug. 9, 1996)), based on a review of geographic, ecological, life history, and genetic data (Busby et al. 1996). The DPS included rivers west of the Elwha River and south to, but not including the rivers that flow into Grays Harbor (Figure 1). This DPS includes Water Resource Inventory Areas (WRIA) 19 (Lyre-Hoko), 20 (Sol Duc-Hoh), and 21 (Queets-Quinault) (Phinney and Bucknell, 1975). The rivers and streams in these WRIAs extend from the US EPA Ecoregion III Coast Range (#1) to the North Cascades (#77), and their basins include several Level IV Ecoregions (Figure 2). The Olympic Peninsula Steelhead DPS was further characterized by

[9] Initially steelhead conservation units were based on the Evolutionarily Significant Unit (ESU) policy (Waples et al. 1991), NMFS subsequently adopted a joint DPS policy with the USFWS to list anadromous *O. mykiss* (NMFS 2006), based on earlier DPS policy (1996; 61 FR 4722).

habitat, climatic, and zoogeographical characteristics that distinguished it from its neighboring DPSs (Busby et al 1996). Zoogeographic patterns support ecological separation of the Olympic Peninsula from adjacent areas. West of the Cascades pygmy whitefish (*Prosopium coulteri*) and longnose sucker (*Catostomus catostomus*) are only known from previously glaciated areas to the north of the Chehalis River (McPhail and Lindsey 1986, p. 631). The distribution of several amphibian species also appears to change at the Chehalis River Basin (Stebbins 1966, Cook 1984, Leonard et al. 1993).

Further, Busby et al. (1996) stated:

> Genetic data collected by WDFW support the hypothesis that, as a group, steelhead populations from the Olympic Peninsula are substantially isolated from those in other regions of western Washington. The Olympic Peninsula ESU is further characterized by habitat, climatic, and zoogeographical differences between it and adjacent ESUs. The Olympic Peninsula includes coastal basins that receive more precipitation than any other area in the range of west coast steelhead. Topography on the Olympic Peninsula is characterized by much greater relief than that to the south (Willapa Hills); the Olympic Mountains range from 1,200 to 2,400 m above sea level. This affects precipitation quantity and river-basin hydrography. The result is "copious amounts of rain and over 100 inches of snow during the winter months" as well as substantial summer precipitation (Jackson 1993, p. 50-51) [Figure 3, Figure 4]. One manifestation of the ecological difference between Puget Sound and the Olympic Peninsula is the shift in vegetation zone, respectively, from western hemlock (*Tsuga heterophylla*) to Sitka spruce (*Picea sitchensis*) (Frenkel 1993).

In describing the Olympic Peninsula Steelhead DPS, Busby et al. (1996) reported that life history and abundance information was limited for most populations and there were no historical estimates of (pre-1960s) abundance for populations in the DPS. Winter-run steelhead represented the predominant life history type, with several rivers also supporting summer runs. Of the 31 stocks/populations identified within the DPS, sufficient abundance information to assess demographic status was only available for 12 populations, all of which were winter run. Information on summer run was limited to the presence of populations identified in the Salmon and Steelhead Stock Inventory (SASSI) (WDF et al. 1993).

The steelhead ESU/DPS boundary between the Olympic Peninsula and Puget Sound also corresponded with an ESU boundary for Coho salmon (Weitkamp et al. 1995), Chum salmon (Johnson et al. 1997), and Chinook salmon (Myers et al 1998). These other status reviews similarly relied on species specific genetic and life history data as well as ecological conditions.

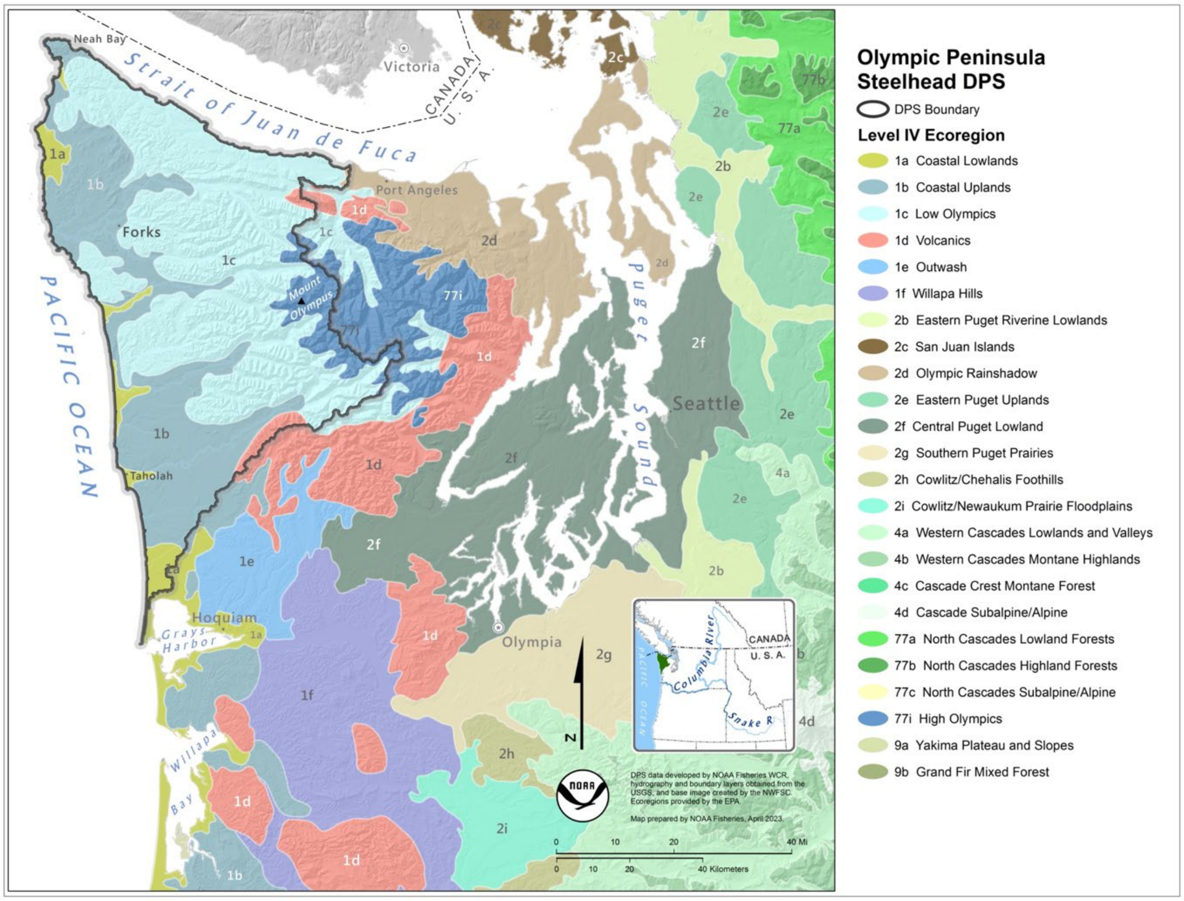



Figure 2. US EPA Level IV Ecoregions. 1 Coast Range: 1a Coastal Lowlands, 1b Coastal Uplands, 1c Low Olympics, 1d Volcanics; 77 North Cascades: 77i High Olympics (Pater et al. 1998). Ecoregions identify areas with distinct climatic, geologic, and vegetative characteristics.

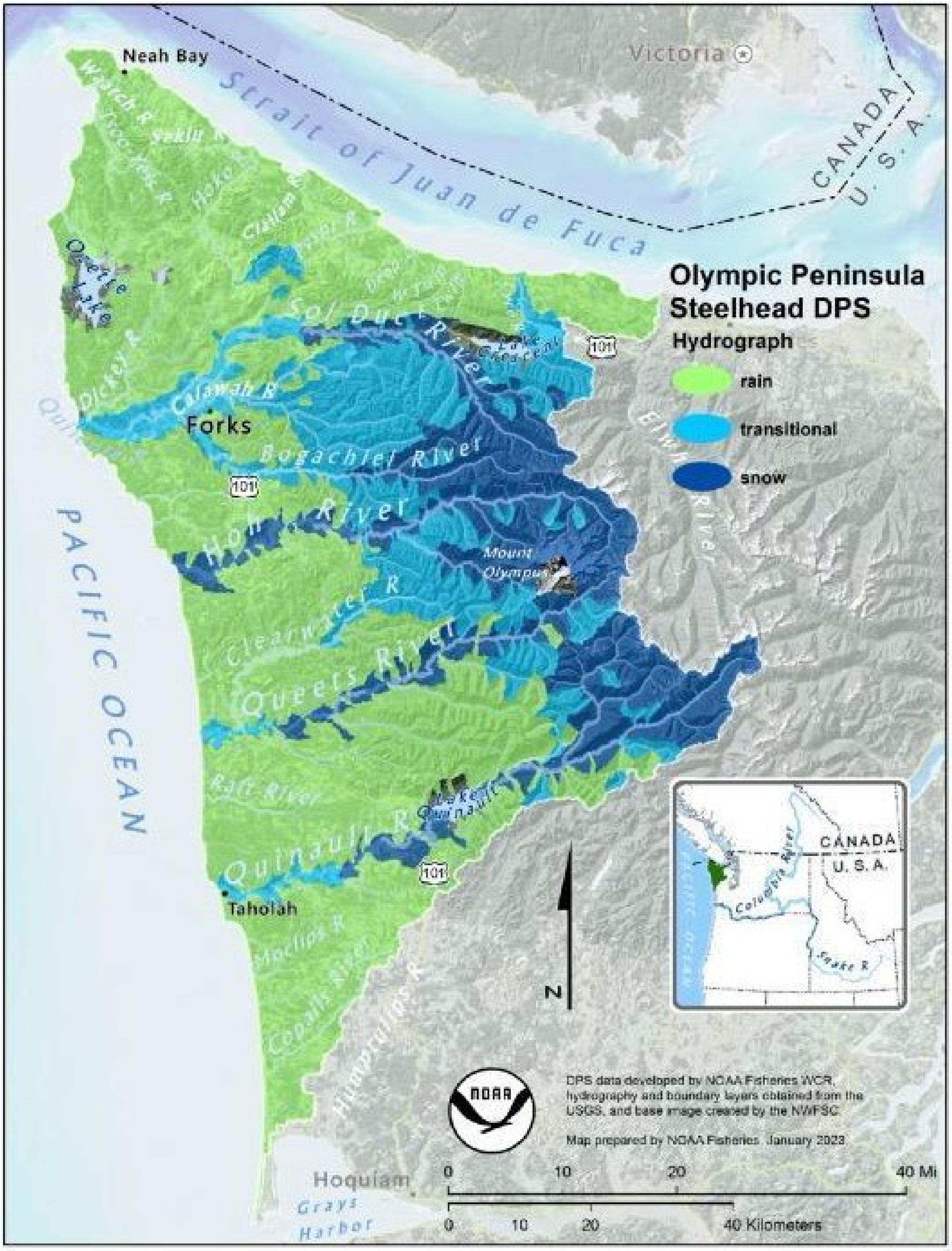



Figure 3. Hydrographic regions within the Olympic Peninsula DPS.

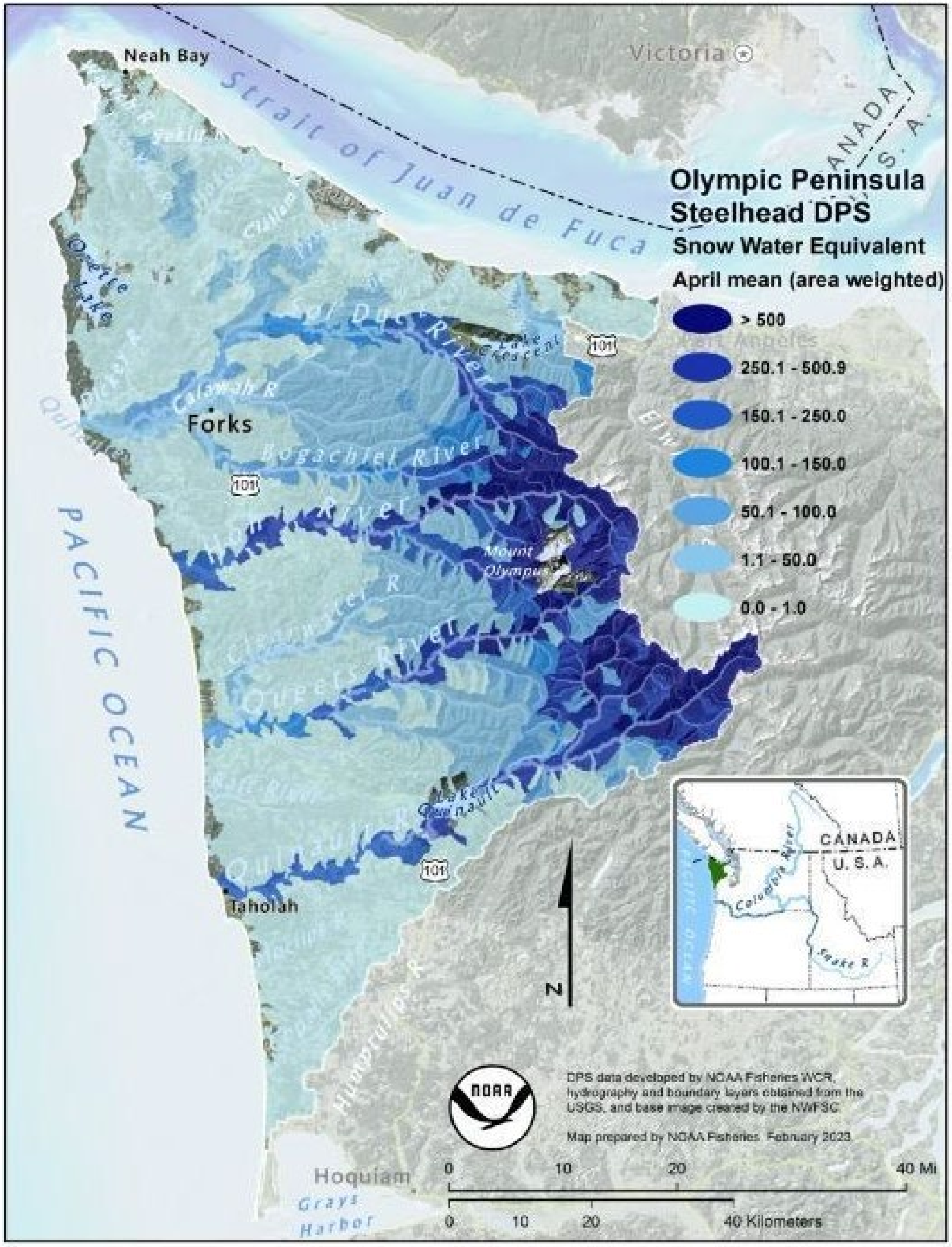



Figure 4. Average April mean snow water equivalent regions in the Olympic Peninsula Steelhead DPS.

# Biology of Steelhead, Anadromous Oncorhynchus mykiss, in the Olympic Peninsula

Migratory Pacific salmonids vary considerably in timing of river entry and spawning, both within and among populations, and this variation in life-history supports local adaptation to specific river environments (Healey 1991; Quinn et al. 2016; Prince et al. 2017; Waples et al. 2022). Populations of *O. mykiss* often manifest multiple life-history pathways, providing a “portfolio effect” that stabilizes mortality risk and increases the likelihood of population-level persistence (Shapovalov and Taft 1954; Busby et al. 1996; Moore et al. 2014; Kendall et al. 2015; Hodge et al. 2016; Jonsson et al. 2019). This variation involves numerous differences in age at emigration to the sea, differences in age of return migration and spawning, and differences in degree of iteroparity (repeat spawning), but sorts into three overall life-history types: resident trout, winter steelhead, and summer steelhead (Kendall et al. 2015). This diversity in life history expressions enable *O.* mykiss to exploit available habitats in each basin.

Steelhead in the Olympic Peninsula exhibit two distinct anadromous life history strategies: summer- and winter-run migrations, in addition to estuarine and freshwater resident life histories (Kendall et al. 2015). Winter-run steelhead, also known as ocean-maturing steelhead, return to freshwater to spawn during the winter and early spring months, November to June(Table 1). Alternatively, summer-run, stream-maturing, steelhead return to freshwater during late spring and early summer in a relatively immature state (bright) and hold, commonly in pools, until spawning from January to April, although the spawn timing for specific populations is not well documented (Table 1). The management period for summer-run steelhead is legally defined as May 1 to October 31 (WDG 1984). Generally, but not necessarily, summer-run steelhead return-timing is coordinated with river flow patterns that allow access to headwater spawning areas, thus summer-run steelhead access spawning and rearing habitat that is unavailable to winter-run steelhead. Winter-run steelhead, presently and historically, are more abundant and ubiquitous than summer-run steelhead in the Olympic Peninsula (Houston and contour 1984, Scott and Gill 2008, Cram et al. 2018). Resident trout spend their entire life-cycle in freshwater, although some “resident” trout may spawn in freshwater and then undertake an anadromous life history. Further, the ability of *O. mykiss* to persist in freshwater alone allows them to persist if marine migration is blocked, sometimes for extended periods (Winans et al. 2018; Fraik et al. 2021).

Steelhead generally spawn in moderate gradient sections of rivers and streams. In contrast to semelparous Pacific salmon, steelhead females do not guard their redds (nests), but return to the ocean following spawning, although they may dig several redds in the course of a spawning season (Burgner et al. 1992). Spawned-out fish that return to the sea are referred to as “kelts.” Adult male steelhead will remain in freshwater to mate with multiple females; however, this increased activity (including fighting amongst males) reduces the likelihood of males returning to the ocean and surviving to become repeat spawners in subsequent years (McGregor 1986, McMillan et. al 2007). Analyses of scale patterns are often used to identify life history trajectories: years of juvenile rearing in fresh water, years in the ocean, frequency of spawning. Recent data suggests there is a genetic component to summer and winter steelhead and that specific alleles are strongly associated with differences in migration timing (Waples et al. 2022). There is also a region of the genome in *O. mykiss* that has been shown to be associated with anadromy/residency in some populations, in particular those in California

(Pearse et al. 2014, Pearse et al. 2019), but this association is not often found in inland and northern populations (Pearse et al. 2019, Clare et al. 2023), including in the Elwha River (Fraik et al. 2022).

Winter steelhead are found throughout the Olympic Peninsula and occur in smaller independent streams that drain directly into the Strait of Juan de Fuca and in larger rivers and their tributaries that drain into the Pacific Ocean (Figure 5). The smaller drainages generally experience rain-dominated hydrological and thermal regimes, while the larger rivers are influenced by rain and snow-transitional or snow-dominated (glacial) hydrological regimes. Larger basins with higher elevation headwaters drain to the Pacific Coast. It is likely that differences in habitat conditions are reflected in the diversity of life history characteristics (i.e. migration and spawn timing) of winter steelhead inhabiting these two types of basins. For example, it appears that steelhead spawn earlier in smaller lowland streams where water temperatures are generally warmer than in larger rivers with higher elevation headwaters. In contrast, the summer-run migration timing is associated with barrier falls or cascades. These barriers may temporarily limit passage in different ways. Some are velocity barriers that prevent passage in the winter during high flows, but are passable during low summer flows, while others are passable only during high flows when plunge pools are full or side channels emerge (Withler 1966).

In the Olympic Peninsula winter-run steelhead predominate (Table 2), in part, because there are relatively few basins with the geomorphological and hydrological characteristics necessary to create the temporal and/or physical barrier features that establish and sustain the summer-run life history. Summer-run steelhead are currently reported for portions of the Big Four streams draining into the Pacific Ocean (Figure 6): Quinault (East Fork, North Fork, and main stem), Queets (mainstem, Clearwater), Hoh (South Fork Hoh), and Quillayute (Bogachiel, Sol Duc, Sitkum, and Calawah) (Cram et al. 2018). Summer-run steelhead are not currently reported for rivers along the Strait of Juan de Fuca, although historically there was a population in the Lyre River (McHenry et al. 1996; Goin 2009). Its current status is unknown. The adaptive basis for the early (pre-maturation) adult run-timing is hypothesized to stem from two complementary selective pressures: the advantages of escaping higher predation risk in the marine environment, and the advantages of utilizing habitats inaccessible to winter runs due to seasonal flow patterns (Quinn et al 2015; Busby et al. 1996). The latter is supported for summer-run steelhead by evidence that they typically spawn further upstream than winter steelhead, in some instances above seasonal hydrologic barriers (Withler 1966; Hard 2007). The summer-run strategy is observed in anadromous fish and is also known as premature migration (Quinn et al 2015), so called because the adult summer-run adults migrate from the ocean to freshwater before sexual maturation, which is fundamentally distinct physiologically from the mature-first-then-migrate strategy of winter steelhead. The ability to migrate with immature gametes, and hold without feeding in freshwater while gametes mature, allows this phenotype to sustain plastic responses of run-timing to large-scale changes in hydrologic conditions, such as shifts in numerical dominance of spring, summer, or fall migrations in response to anthropogenic flow alteration (McEwan 2001). During the summer-run steelhead's extended freshwater residence prior to spawning, the fish normally hold in deep pools which exposes the fish to

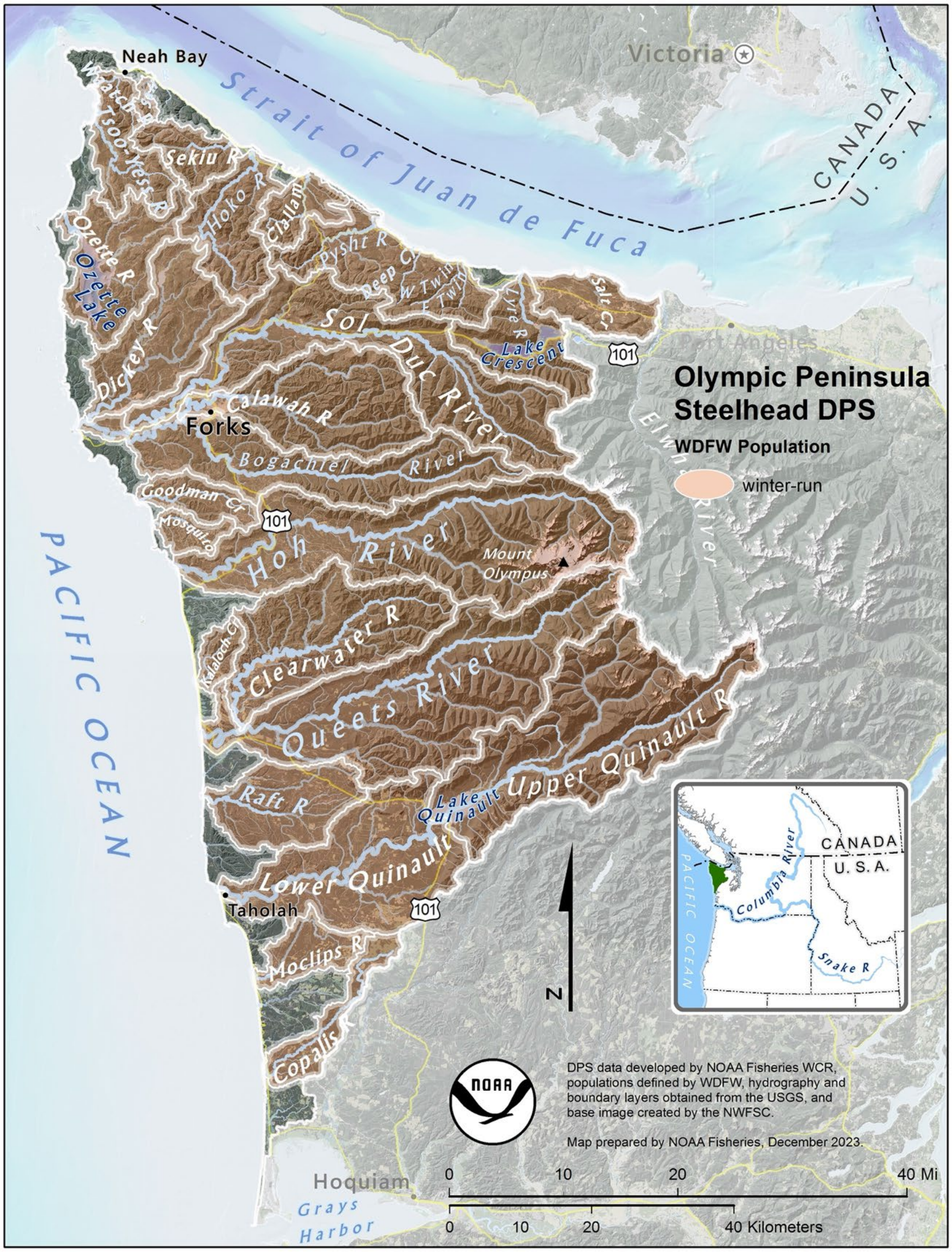



Figure 5. Winter-run steelhead populations in the Olympic Peninsula Distinct Population Segment. Based on presence and hydrographic basin.

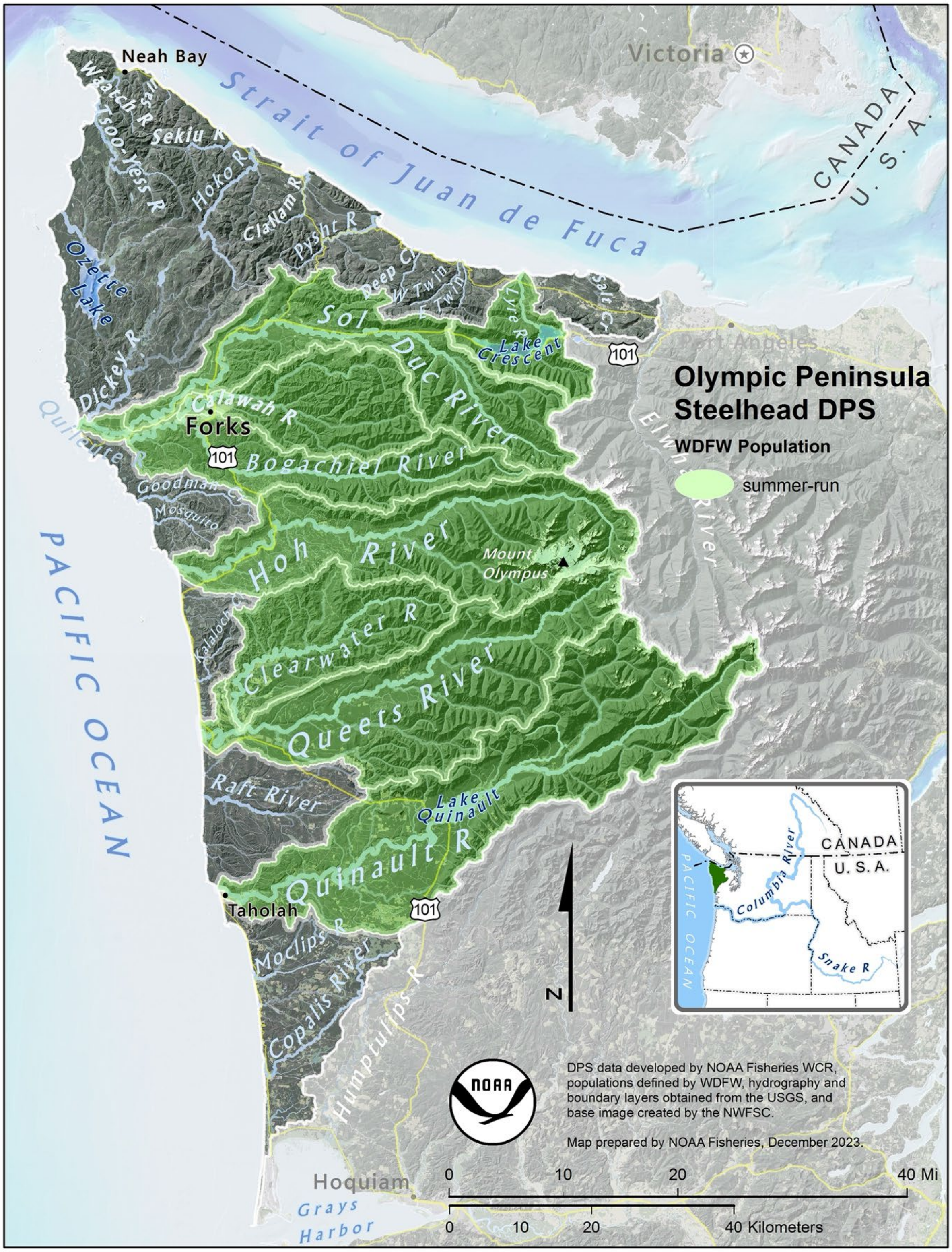



Figure 6. Summer-run steelhead populations in the Olympic Peninsula Distinct Population Segment. Based on presence and hydrographic basin.

prolonged predation, harvest, and poaching risk and seasonal environmental extremes, which likely results in higher prespawning mortality relative to winter-run steelhead. Further, land development, logging, and other human activities can remove large wood from in-stream areas, remove large wood that would eventually recruit into streams, and increase sediment in the stream; all of which reduce or eliminate holding pools.

Table 1. Presumptive run and spawn timing for winter and summer-run steelhead populations in the Olympic Peninsula DPS based on WDF et al. (1992). Shaded areas indicate run timing (green–winter run, red–summer run), s – indicates spawning period, where no "s" is present spawn timing was designated as unknown. Water Resource Inventory Areas (WRIAs) are watershed area defined by the Washington Department of Ecology.

| Steelhead | River | Run | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sept | Oct | Nov | Dec |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| WRIA 19 | Salt Creek | Winter | | s s | s s s | s s s | s s s | | | | | | | |
| | Lyre River | Winter | | s s | s s s | s s s | s s s | s | | | | | | |
| | Pysht River | Winter | | s s | s s s | s s s | s s s | s s | | | | | | |
| | Clallam River | Winter | | s s | s s s | s s s | s s s | | | | | | | |
| | Hoko River | Winter | | s s | s s s | s s s | s s s | s s | | | | | | |
| | Seiku River | Winter | | s s | s s s | s s s | s s s | | | | | | | |
| WRIA 20 | Tsoo Yes River | Winter | | | | | | | | | | | | |
| | Ozette River | Winter | | | | | | | | | | | | |
| | Dickey River | Winter | | | | | | | | | | | | |
| | Sol Duc River | Winter | | s s | s s s | s s s | s s s | s s | | | | | | |
| | Sol Duc River | Summer | | | | | | | | | | | | |
| | Bogachiel River | Winter | | s s | s s s | s s s | s s s | s s | | | | | | |
| | Bogachiel River | Summer | | | | | | | | | | | | |
| | Calawah River | Winter | | s s | s s s | s s s | s s s | s s | | | | | | |
| | Calawah River | Summer | | | | | | | | | | | | |
| | Hoh River | Summer | | | | | | | | | | | | |
| | Goodman Creek | Winter | | | | | | | | | | | | |
| | Mosquito Creek | Winter | | | | | | | | | | | | |
| | Hoh River | Winter | | s s | s s s | s s s | s s s | s | | | | | | |
| | Kalaloch Creek | Winter | | | | | | | | | | | | |
| WRIA 21 | Queets River | Winter | | s s | s s s | s s s | s s s | s s s | | | | | | |
| | Queets River | Summer | | | | | | | | | | | | |
| | Clearwater River | Winter | | s s | s s s | s s s | s s s | s s s | | | | | | |
| | Clearwater River | Summer | | | | | | | | | | | | |
| | Raft River | Winter | | s s | s s s | s s s | s s s | s | | | | | | |
| | Quinault (Upper) River | Winter | | s s | s s s | s s s | s s s | s s s | | | | | | |
| | Quinault/Lake Quinault | Winter | | s s | s s s | s s s | s s s | s s s | | | | | | |
| | Quinault River | Summer | | | | | | | | | | | | |
| | Moclips River | Winter | | s s | s s s | s s s | s s s | s | | | | | | |
| | Copalis River | Winter | | s s | s s s | s s s | s s s | s | | | | | | |

## Populations

We have relied upon SASSI (WDF et al. 1993), Busby et al. (1996) and co-manager reports (COPSWG 2023); to provide a provisional population list (Table 1) of winter-run (Figure 5) and summer-run (Figure 6) "populations" for analysis. The primary purpose of this process is to establish fundamental units for statistical analysis for this risk assessment. The current SRT identified 10 summer-run populations, and 29 winter-run steelhead populations. Based on our assessments, steelhead from individual smaller independent streams may not constitute a demographically independent population (DIPs) as described in McElhany et al. (2000), but would ultimately be combined with other watersheds to create an appropriate DIP. Similarly, the SRT considered that larger watersheds may contain multiple populations, based on run timing (i.e. winter and summer run) or geography (Lower and Upper Quinault rivers). In some cases, the demographic population was defined by coverage of the data sets provided by co-managers. Prior population studies with steelhead will also be used to inform the identification of provisional populations (Myers et al. 2006, Myers et al. 2015).

Table 2. Presumptive populations of winter- and summer-run steelhead in the Olympic Peninsula DPS based on WDF et al. (1992), arranged by Water Resource Inventory Area (WRIA): East to West (WRIA 19), and North to South (WRIA 20 and 21). Winter steelhead also occur in numerous smaller independent tributaries. Similarly, summer*, summer run steelhead have been observed in the Hoko, but it is unclear if it represents an independent population. Dickey, Sol Duc, Calawah, and Bogachiel rivers are tributaries to Quillayute River.

| **Olympic Peninsula Steelhead DPS** | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| WRIA 19 | | | WRIA 20 | | | WRIA 21 | | |
| Marine | Stream | Run Timing | Marine | Stream | Run Timing | Marine | Stream | Run Timing |
| Strait of Juan de Fuca | Salt Creek | Winter | Pacific Coast | Waatch River | Winter | Pacific Coast | Kalaloch Creek | Winter |
| Strait of Juan de Fuca | Lyre River | Winter | Pacific Coast | Tsoo-Yess River | Winter | Pacific Coast | Queets River | Winter |
| Strait of Juan de Fuca | Lyre River | Summer | Pacific Coast | Ozette River | Winter | Pacific Coast | Queets River | Summer |
| Strait of Juan de Fuca | West Twin River | Winter | Pacific Coast | Quillayute River | Winter | Pacific Coast | Raft River | Winter |
| Strait of Juan de Fuca | East Twin River | Winter | Pacific Coast | Quillayute River | Summer | Pacific Coast | Clearwater River | Winter |
| Strait of Juan de Fuca | Deep Creek | Winter | Pacific Coast | Dickey River | Winter | Pacific Coast | Clearwater River | Summer |
| Strait of Juan de Fuca | Pysht River | Winter | Pacific Coast | Sol Duc River | Winter | Pacific Coast | Quinault River | Winter |
| Strait of Juan de Fuca | Clallam River | Winter | Pacific Coast | Sol Duc River | Summer | Pacific Coast | Quinault River | Summer |
| Strait of Juan de Fuca | Hoko River | Winter | Pacific Coast | Calawah River | Winter | Pacific Coast | Upper Quinault River | Winter |
| Strait of Juan de Fuca | Hoko River | Summer* | Pacific Coast | Calawah River | Summer | Pacific Coast | Upper Quinault River | Summer |
| Strait of Juan de Fuca | Sekiu River | Winter | Pacific Coast | Bogachiel River | Winter | Pacific Coast | Moclips River | Winter |
| Strait of Juan de Fuca | Sail River | Winter | Pacific Coast | Bogachiel River | Summer | Pacific Coast | Copalis River | Winter |
| | | | Pacific Coast | Lonesome Creek [10] | Winter | | | |
| | | | Pacific Coast | Goodman Creek | Winter | | | |
| | | | Pacific Coast | Mosquito Creek | Winter | | | |
| | | | Pacific Coast | Hoh River | Winter | | | |
| | | | Pacific Coast | Hoh River | Summer | | | |

[10] Lonesome Creek is likely too small a watershed to support an independent population, but is listed here to account for hatchery releases in the watershed.

## Genetics

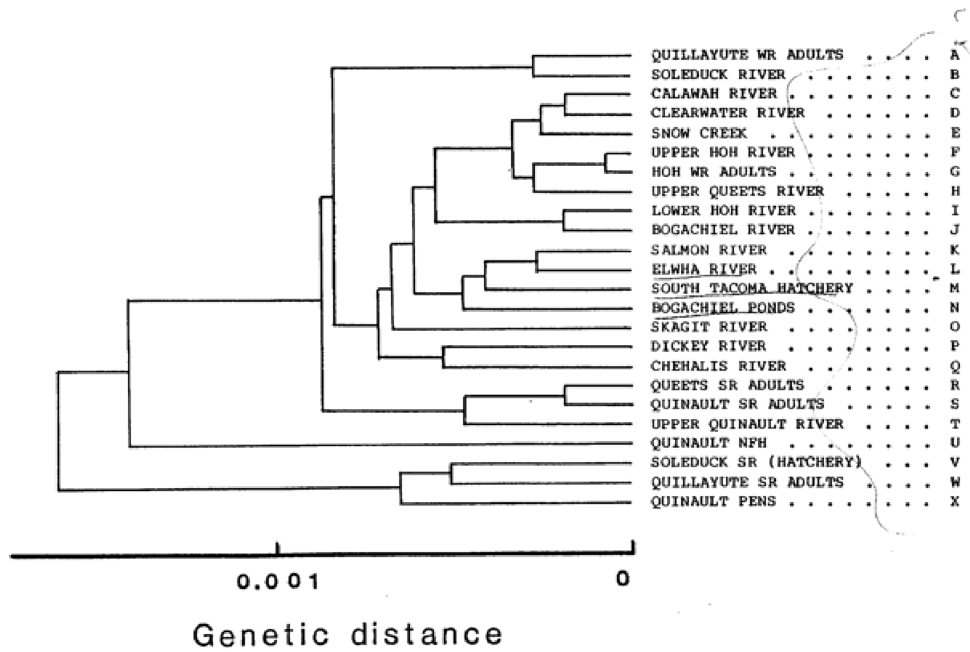
### Genetic studies

There are a limited number of genetics studies that included steelhead samples from Olympic Peninsula watershed and hatcheries. Samples were representative of both large and small populations and those that drain to the Strait of Juan de Fuca and the Pacific Ocean. Two of the earliest, allozyme-based studies examined the relationships between steelhead sampled from rivers compared with hatchery stocks being released into the Olympic Peninsula DPS (Figure 7, Figure 8) (Reisenbichler and Phelps, 1989; Phelps et al. 1995). Reisenbichler and Phelps (1989) analyzed 27 collections of steelhead from five major drainages on the Olympic Peninsula and Washington Coast using an allozyme analysis (Figure 7). Their study revealed that genetic differentiation within and among drainages was not significant, and genetic variation among drainages was much less than that reported in British Columbia (Parkinson 1984). Reisenbichler and Phelps (1989) suggested that the lack of differentiation of the natural-origin collections may be the result of hatchery influence into each of the tributaries. At the time, early-returning winter run steelhead hatchery broodstocks (Cook Creek (Quinault National Fish Hatchery (NFH) stock, Bogachiel Hatchery stock, Makah NFH stock) appeared to be heavily influenced by introductions of early-winter steelhead from Chambers Creek Hatchery (Puget Sound). Additionally, transfers to other hatcheries for release in the OP DPS and off-station releases appear to have influenced the genetic composition of winter steelhead collected from in-river sampling in many rivers (Figure 9, Figure 12) (Kassler et al. 2011, Seamons and Spidle 2023). Similarly, the rearing and release of early summer-run steelhead (originating from the Skamania Hatchery, Lower Columbia River DPS) from the Quinault NFH and Calawah Ponds facility (Quillayute River) appear to have influenced the composition of steelhead sampled from their respective rivers. In spite of widespread releases of non-native hatchery origin steelhead, early genetic studies indicated that there were clusters of native winter-run steelhead distinct from hatchery stocks and populations adjacent to the OP (Figure 7, Figure 8). Although the early genetic studies provided incomplete coverage of the OP DPS, this information was important in supporting the DPS boundaries established by Busby et al. (1996).

More recent studies using microsatellite DNA analysis (Kassler et al. 2010, Kassler et al. 2011, Seamons et al. 2017), show a similar pattern of introgression by non-native early-winter steelhead and early-summer run steelhead into presumed natural-origin population samples, with the natural-origin fish still being distinct from hatchery broodstock (Figure 9, Figure 10, Figure 12). For the limited number of streams investigated, among natural-origin samples exhibited relatively little differentiation.

Kassler et al. (2010) investigated genetic relationships among adult winter steelhead from natural origin populations in five coastal rivers and four Olympic Peninsula hatcheries. The natural-origin stocks from the Bogachiel, Calawah, Sol Duc River, Hoh, and South Fork Hoh rivers were not genetically differentiated from one another, consistent with findings reported by Reisenbichler and Phelps (1989) for OP Steelhead. Hoh natural-origin steelhead exhibited the highest allelic richness value among the natural-origin populations analyzed.

Hatchery-origin collections were differentiated from one another based on pairwise $F_{ST}$ values and had lower measures of genetic diversity (heterozygosity and allelic richness) than the natural-origin collection. In the Hoh River, the evaluation of unclipped and clipped collections did not reveal genetic introgression at the population-level, however, at the individual level, there was evidence of hatchery-origin ancestry within natural-origin collections indicating that some natural origin steelhead spawned with hatchery fish (Kassler et al. 2011). Additionally, the analysis of hatchery steelhead collected from the Hoh River revealed straying from nearby coastal hatcheries. A majority of the samples originated from Cook Creek National Fish Hatchery (released into Hoh) followed by Bogachiel Hatchery (Quillayute Basin) and Salmon River Tribal Hatchery (Queets Basin). A small portion (2.1%) of the steelhead sampled above the Highway 101 Bridge assigned to the Skamania River summer-run hatchery collection.



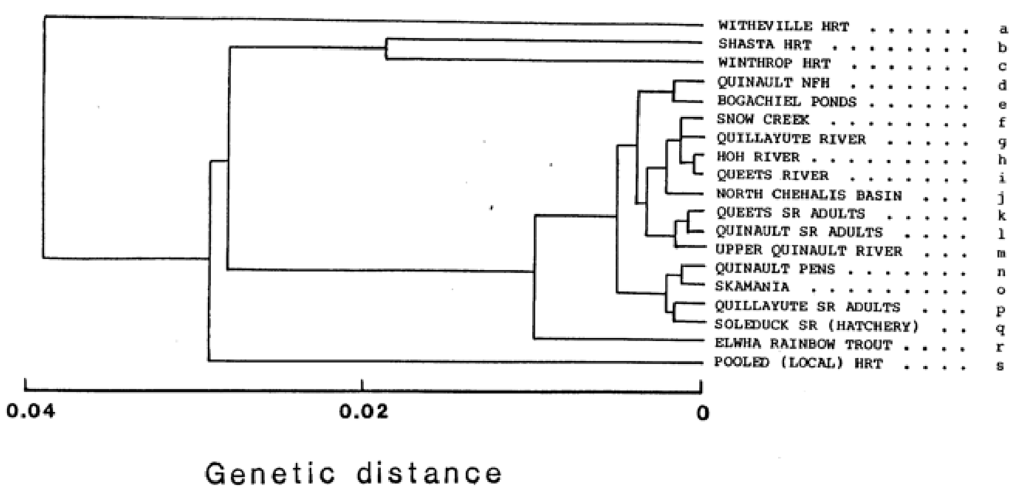




Figure 7. Dendrograms showing results of UPGMA analysis of genetic similarities among samples of steelhead collected (top) and steelhead and hatchery stocks (bottom). Similarities are based on 19 variable loci. WR-Winter Run, SR-Summer Run, NFH-National Fish Hatchery, HRT-Hatchery Rainbow Trout. From Reisenbichler and Phelps, 1989.

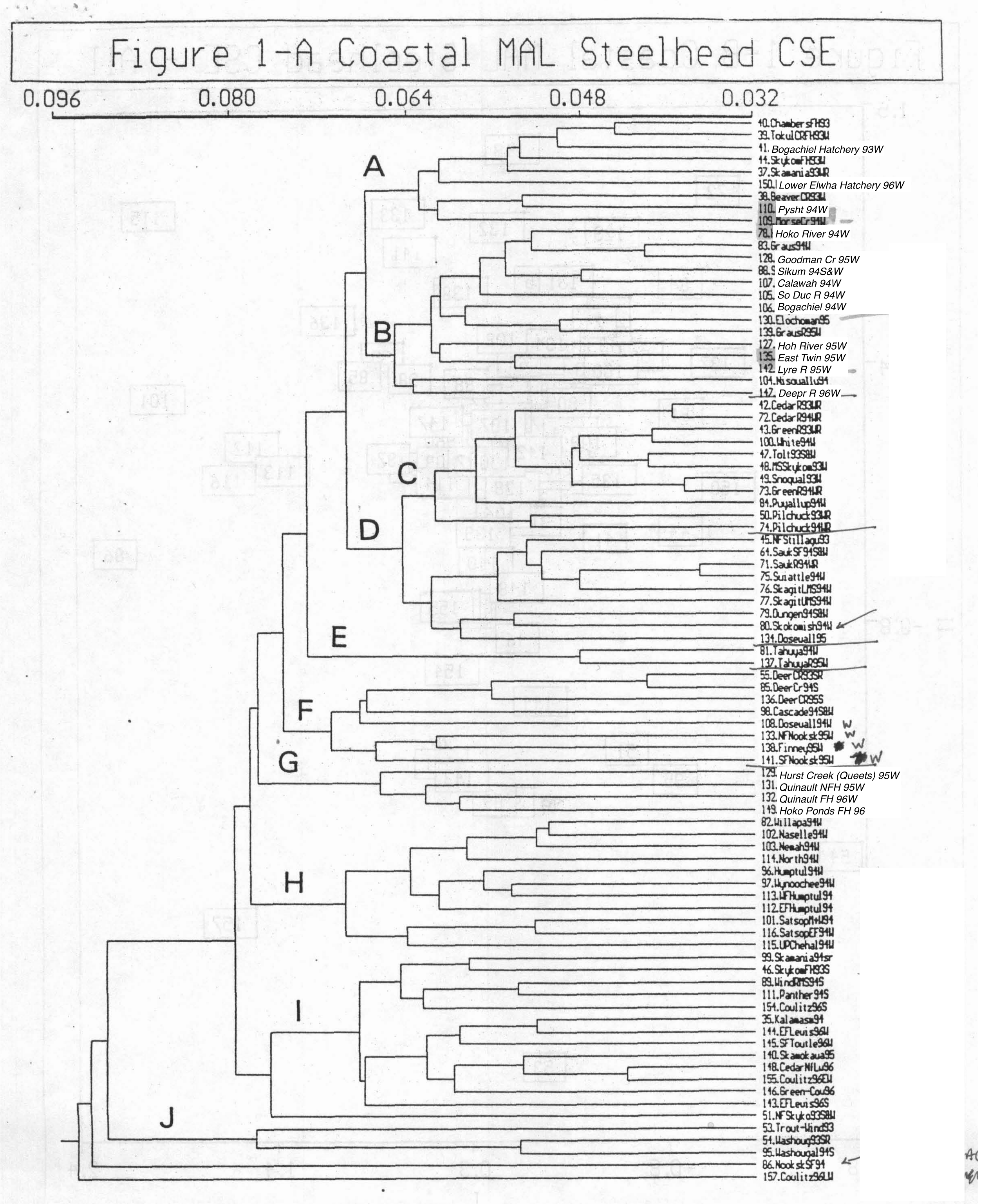



Figure 8. *O. mykiss* populations in Washington State. From Phelps et al. 1997 – OP samples are in "clear" type.

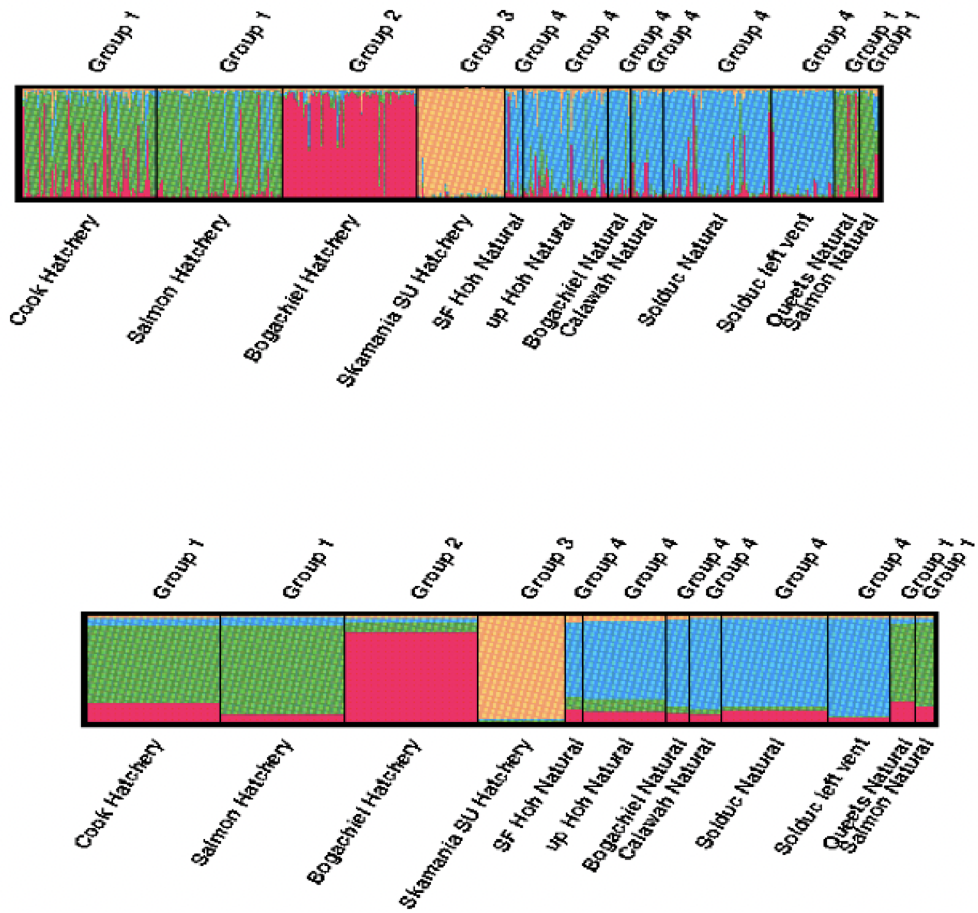



Figure 9. Structure plot showing percent membership of each individual steelhead (top) and the population average (bottom) into the groups that STRUCTURE found in the dataset. Individuals with more than one color in the bar likely have mixed ancestry. The group number identifies the collections with similar ancestry. (Note that Bogachiel Hatchery (winter-run) and Skamania Hatchery (summer-run) were founded from non-native (out-of DPS populations). From Kassler et al. (2011).

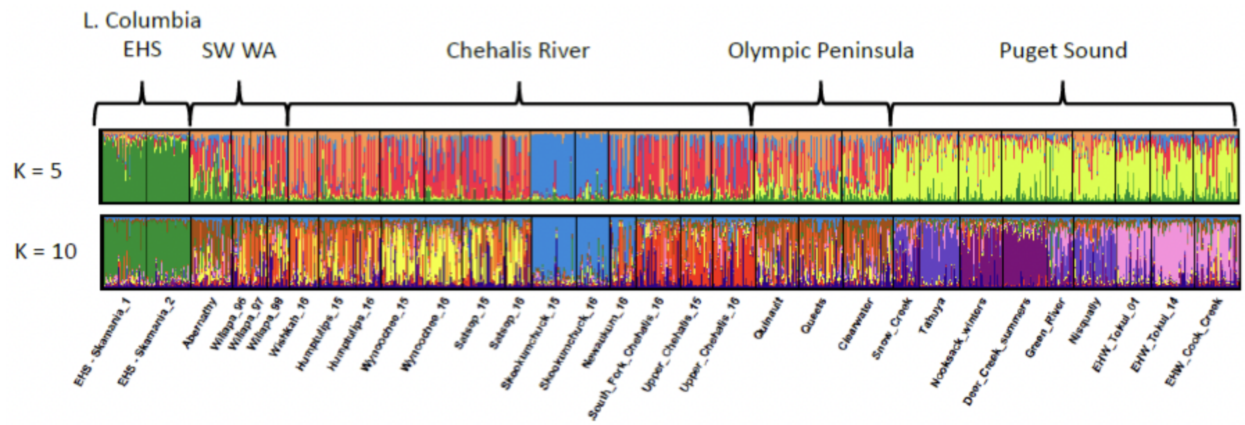



Figure 10. Plots of the results of STRUCTURE analysis of Coastal Lineage O. mykiss collections including Chehalis River collections at K (number of inferred clusters) = 5 and K = 10. The ΔK method of Evanno et al. (2005) supported K = 5 but the mean ln(K) plot supported K = 10, so both are shown. With K = 5, most of the Chehalis samples cluster with other Washington Coast collections, with the Lower Columbia early hatchery summers (green) and Puget Sound (yellow) clustering separately. In the Chehalis, upper/South Fork Chehalis loosely cluster with Wynoochee/Satsop, and Wishkah/Humptulips loosely cluster with Willapa River collections. With K = 10, Puget Sound is split roughly into three clusters and the Chehalis collections are split roughly into 4 clusters: upper/South Fork Chehalis, Skookumchuck/Newaukum, Wynoochee/Satsop, and Wishkah/Humptulips. The Newaukum collection appears to be a mixed collection of Skookumchuck and upper Chehalis individuals. The Skookumchuck collections (blue) formed a separate very distinct cluster no matter the makeup of the rest of the analyzed collections for almost all values of K, including K = 5 and K = 10. (From Seamons et al. 2017).

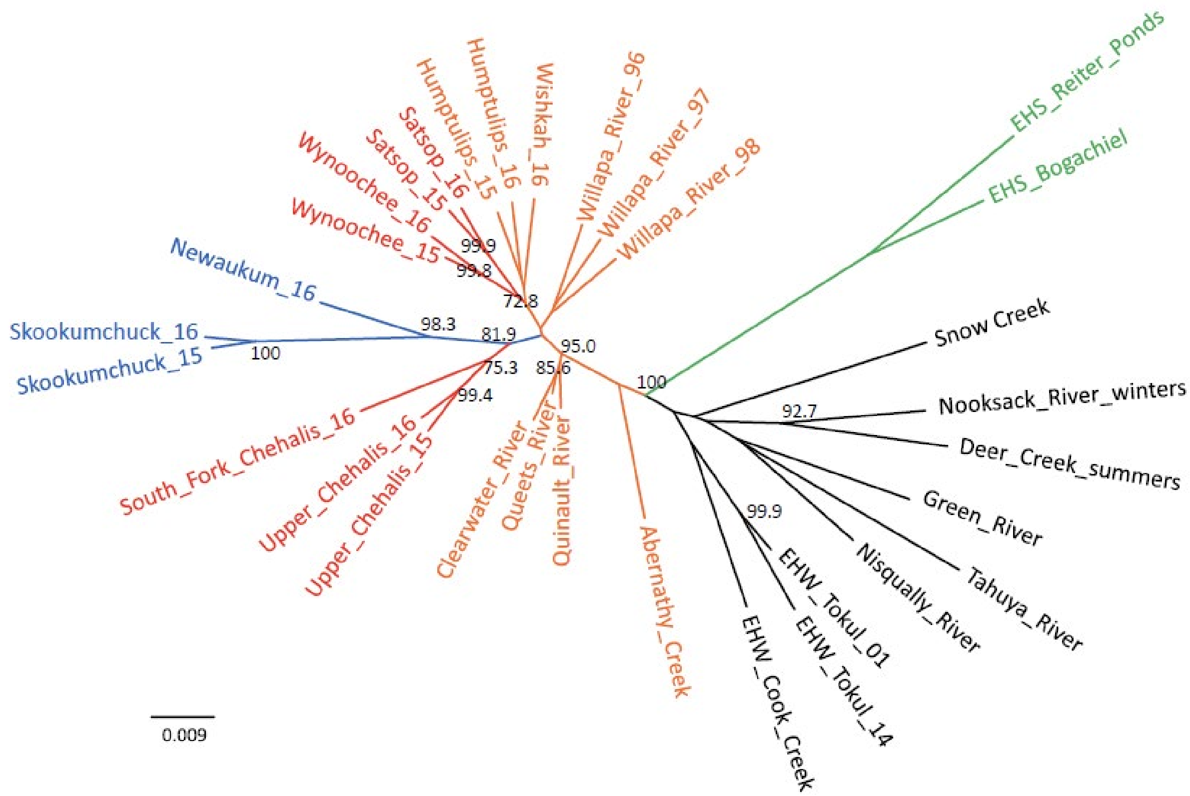



Figure 11. Unrooted neighbor-joining dendrogram (from Seamons et al. 2017) constructed from Cavalli-Sforza genetic distance matrix calculated using PHYLIP (Felsenstein 1993). The dendrogram is color coded to roughly match K = 5 of Figure 5: lower Columbia River in green, Puget Sound in yellow (black), Skookumchuck/Newaukum in blue, upper Chehalis/SF Chehalis/Satsop/Wynoochee in red, and lower Chehalis/Willapa/Olympic Peninsula/Abernathy in orange. With the exception of the Abernathy Creek collection, collections generally clustered with other members of their DPS. Strong bootstrap support was evident separating Chehalis River and Willapa River collections from all other collections. Moderate to strong bootstrap support existed separating the Willapa River from Chehalis collections. (from Seamons et al. 2017)

## New Genetic Analysis and DPS Configuration

In response to data requests by the SRT, WDFW and the NWIFC embarked on an updated analysis of all genetic (single nucleotide polymorphism (SNP) markers) data that has been collected to date, on the Olympic Peninsula DPS *O. mykiss* (Seamons and Spidle 2023). Samples that were analyzed by Seamons and Spidle (2023) ranged from collections taken from 1994 through 2021, and included both hatchery and natural origin steelhead, and many collections that had been previously analyzed (Phelps et al. 1997, Kassler et al. 2010, 2011). Though the major coastal streams on the Olympic Peninsula are represented in the data, many of the collections used for analyses are decades old, and some of the smaller streams located on the coast and in the Strait of Juan de Fuca, are not represented. Generally speaking, the OP steelhead collections show very little genetic differentiation from one another ($F_{ST}$ within the OP DPS 0.008). The major coastal streams, which have the best coverage of samples, in particular, show little to no genetic differentiation supporting the idea that there is genetic exchange between populations on the coast. This is consistent with results from Kassler et al. (2010, 2011), which used some of the same collections in their analysis with microsatellite loci, and with Phelps et al. (1997) and Reisenbichler and Phelps (1985), which used allozymes.

Very few samples from within the OP DPS have been analyzed for the small streams draining into the Strait of Juan de Fuca; only the Pysht and Lyre River collections from within the OP DPS from the 1990s have been used for genetic analyses. The southern boundary of the OP steelhead DPS is supported by genetic differentiation from populations in the southwest Washington DPS (pairwise $F_{ST}$ 0.042). More recent collections are needed to get a definitive understanding of the genetic differentiation among steelhead in the Olympic Peninsula DPS, and in particular, the genetic differentiation in the Strait of Juan de Fuca between the streams to the west of the Elwha River, and the Elwha River and east (in the Puget Sound DPS); though there is clear differentiation between the OP steelhead DPS and the Puget Sound DPS overall ($F_{ST}$ 0.026). One small population of *O. mykiss*, which is resident in Lake Crescent, is notably different from all other *O. mykiss* sampled, and is a known endemic, local form of resident rainbow trout known as the Beardslee trout (see Brenkman et al. 2014 for a review).

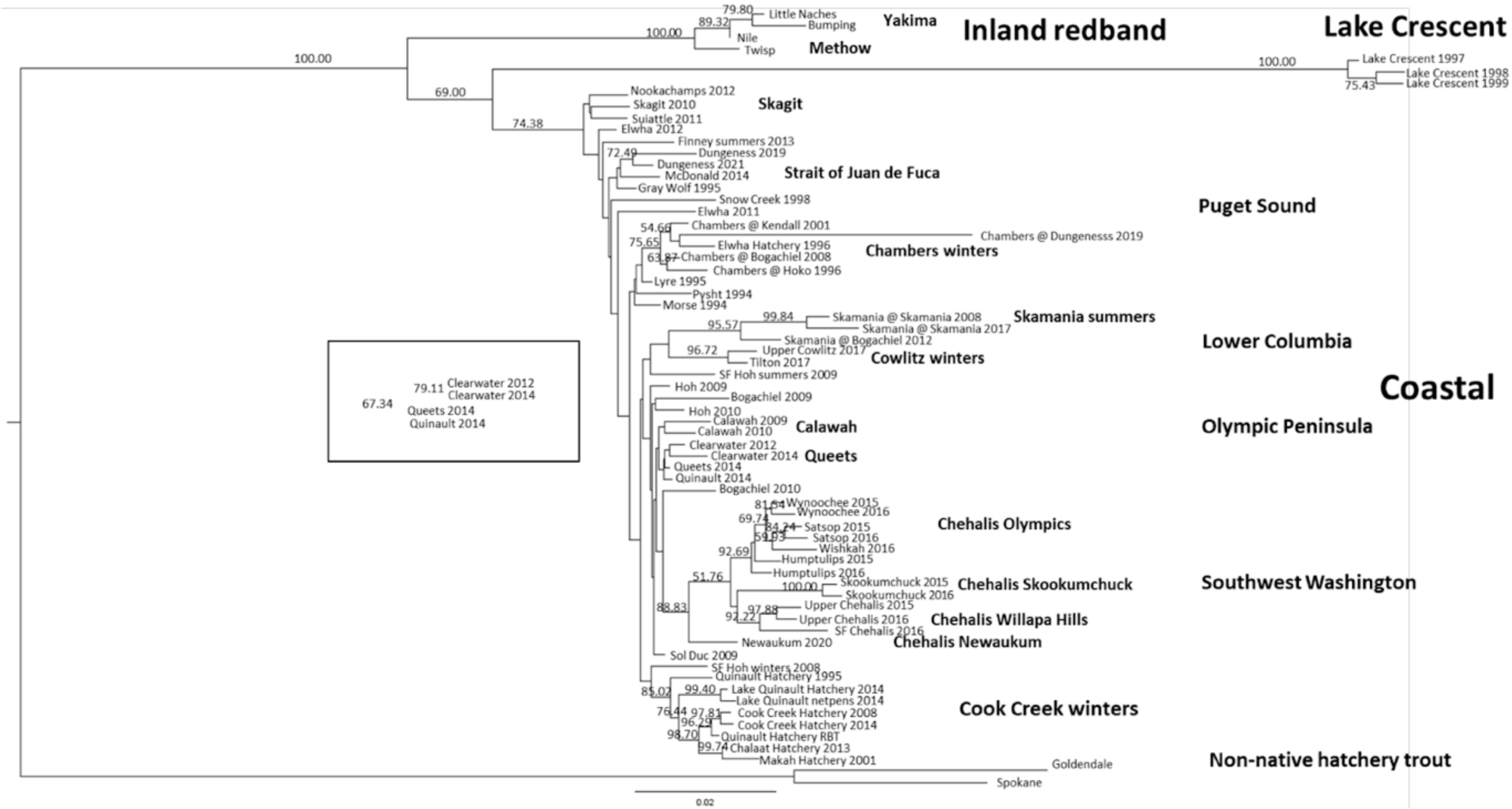



Figure 12. Unrooted Neighbor-joining dendrogram based on pairwise Nei's genetic distance of native and non-native Washington *O. mykiss* from Seamons and Spidle (2023).

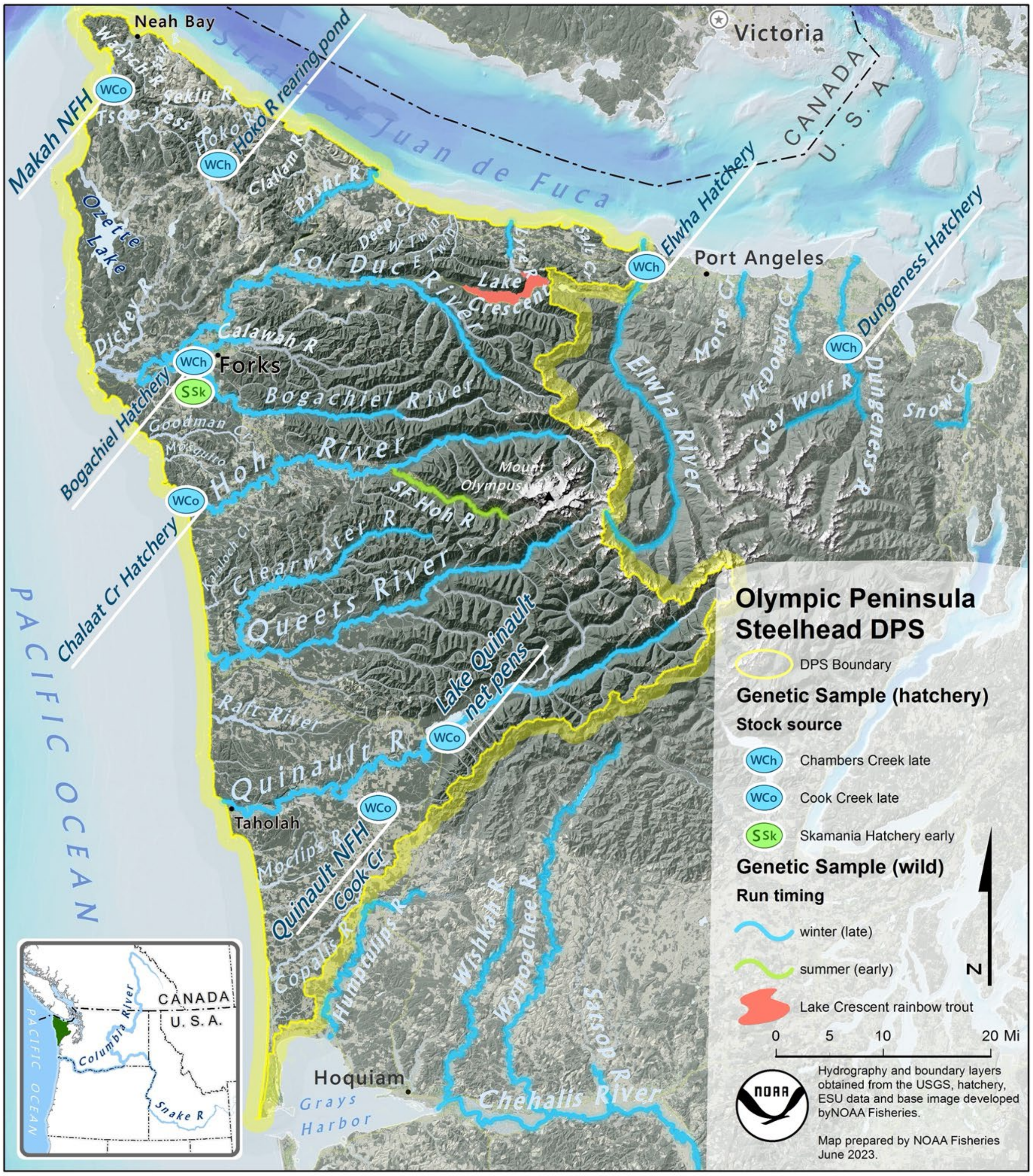



Figure 13. Olympic Peninsula steelhead collections (highlighted in blue or red) used for genetic analysis in Spidle and Seamons (2023). Collections were made from 1994-2021, and were genotyped with single nucleotide polymorphism (SNP) markers.

<u>Genetic information and life history diversity</u>. Busby et al. (1996) first highlighted the paucity of information on summer-run and winter-run steelhead differentiation, but did note that the two life history forms are not monophyletic. Busby et al. (1996) also noted that much of the

information on genetic diversity in Olympic Peninsula steelhead is from winter-run steelhead, and the same is currently true. The most recent genetic analysis by Seamons and Spidle (2023), includes only a small number of known summer steelhead from the South Fork of the Hoh River (Figure 12**Error! Reference source not found.**), which is not enough to evaluate genetic diversity and similarity to winter run steelhead, or to evaluate specifically whether summer run steelhead on the OP possess the 'summer-run haplotypes at a locus that has been shown to be associated with adult return timing in other steelhead (Hess et al. 2016, Fraik et al. 2021), and Chinook salmon populations (Prince et al. 2017; Narum et al. 2018; Thompson et al. 2020; Willis et al. 2021). Although McMillan (2022) and Olympic National Park files (S. Brenkman, pers. comm.) report summer steelhead observations from snorkel surveys in a number of coastal OP streams, focused efforts on sampling and evaluation of the genetic diversity and genetic differentiation between summer and winter steelhead have not been conducted. The contribution of resident *O. mykiss* to the productivity and genetic diversity in anadromous steelhead is currently unknown in the OP steelhead DPS. Currently there is no existing information on the genetic diversity and differentiation of resident vs. migratory *O. mykiss* in the OP steelhead DPS.

Prior studies in *O. mykiss* in the southern portion of the species range have identified a major genome region (on *O. mykiss* chromosome 5) associated with migration and residency in *O. mykiss*, but diversity at this region of the genome has not been examined in OP *O. mykiss*. In the Elwha River, polymorphism at this region of the genome is not associated with migration and residency in *O. mykiss* (Fraik et al. 2021), and the association of this locus with migration and residency is not consistent across the range northward and inland, where the 'resident' or 'rearranged' haplotype for this genome region increases in frequency in both anadromous and resident *O. mykiss* (Pearse et al. 2019; Weinstein et al. 2019).

<u>Hybridization with coastal cutthroat trout</u>. Hybridization between *O. mykiss* and *O. clarki* can occur where the two species co-occur. Spidle and Seamons (2023) specifically evaluated evidence for hybridization with coastal cutthroat trout (*O. clarki clarki*) within the *O. mykiss* collection that they genotyped, using a single diagnostic SNP marker, together with genotypes of Tokul coastal cutthroat trout. Only four individuals were categorized as hybrids in the >3000 *O. mykiss* genotyped, but the authors note that to fully study hybridization between these two species, both species and their putative hybrids should be sampled. The *O. mykiss* collection, in general, intentionally excluded presumptive hybrids being sampled in the field, so as to avoid *O. clarkii clarkii* and potentially hybrid individuals. Martens and Dunham (2021) note significant overlap in the occurrence of steelhead and coastal cutthroat trout in the OP DPS, but little is known about whether or not there is introgression between the species, or the influence of introgression on genetic diversity and productivity in OP steelhead.

<u>Artificial propagation</u>. At the time, Busby et al. (1996) cited widespread production of hatchery steelhead within this ESU, derived from only a few stocks from out of basin. There is a long history of steelhead releases into the Olympic Peninsula DPS. Duda et al. (2018) reported a total of 44.7 million winter steelhead were released into the Quillayute, Hoh, Queets, and Quinault River systems through 2014. The first recorded releases of steelhead into the major coastal drainages were: Quinault Basin (1922), Quillayute Basin (1933), Hoh Basin (1959), and Queets Basin (1978).

A few focused studies have been undertaken to specifically evaluate the influence of hatchery stocks on natural origin steelhead on the OP. Reisenbichler and Phelps (1989) used protein electrophoresis to evaluate allozyme variation in hatchery and naturally-produced fish. Kassler et al. (2010, 2011) used microsatellite markers in a focused evaluation of the genetic diversity among natural and hatchery origin steelhead from coastal collections of OP steelhead, including the Hoh, South Fork Hoh, Sol Duc, Calawah, and Bogachiel rivers, as well as hatchery-origin steelhead from four Olympic Peninsula hatcheries. For the most part, Kassler et al. (2010, 2011) failed to find significant introgression of hatchery steelhead with wild OP steelhead, except in the 2008 South Fork Hoh River winter collection, which shows evidence of interbreeding with the Cook Creek hatchery collection; the same finding was reported by Seamons and Spidle (2023) in a reanalysis of the samples with newer SNP data. Alternatively, the 2009 and 2010 Hoh River winter collections, these steelhead were more similar to other OP natural origin steelhead collections (see Seamons and Spidle (2023) Figure 2, and Figure 12 above). Kassler et al. (2010, 2011) also determined population-of-origin for hatchery-origin winter steelhead captured in sport and commercial fisheries in the Hoh River, finding straying of adult hatchery steelhead released as juveniles in the Bogachiel River to the Hoh River.

Seamons and Spidle (2023) included three hatchery stocks that are currently propagated at Olympic Peninsula hatcheries: Chambers Creek early winter steelhead (Puget Sound origin), Skamania early hatchery summer steelhead (Lower Columbia River origin), and Cook Creek early winter steelhead ('putatively' Olympic Peninsula origin), none of these hatchery stock samples clustered with samples taken from presumptive natural PO steelhead (Figure 12).

There is some evidence for hatchery influence on the native steelhead in Olympic Peninsula streams in these historical collections. Individuals collected from the Lyre and Pysht Rivers (in 1995 and 1994, respectively) in the Strait of Juan de Fuca are more similar to Chambers Creek hatchery winter steelhead, and individuals collected from the Hoh River in 2008, appear to have been influenced by Skamania summer steelhead hatchery individuals (see Kassler et al. 2010, 2011, Seamons and Spidle 2023).

Newer collections would be needed in the OP DPS to assess the influence of past and current hatchery releases on the genetic diversity and provenance of naturally-produced *O. mykiss* in the system, particularly since the termination of, or modification of, hatchery programs and releases that occurred relatively recently. Finally, the effective numbers of breeders, calculated by Seamons and Spidle (2023) were in the hundreds to thousands (considering uncertainty in the estimates) for coastal OP naturally-produced steelhead collections, but were very small in the hatchery populations, the few collections from streams that drain to the Strait of Juan de Fuca, and in the Lake Crescent rainbow trout (resident *O. mykiss* not considered by the SRT).

## Summary: DPS Boundary

The SRT considered new information and analyses relevant to the designation of the Olympic Peninsula DPS boundary since the original ESU was determined by Busby et al. (1996). There were a limited number of new steelhead genetic studies pertinent to the DPS configuration question; however, the SRT concluded that patterns of genetic variation and differentiation reported do not warrant a revision to the DPS boundaries for OP steelhead at this time.

However, extant genetic data on steelhead in streams that drain into the Strait of Juan de Fuca are nonexistent, sparse, or decades old. The team recommends continued evaluations of genetic diversity within and among OP DPS steelhead with new collections to further evaluate the genetic relationships between streams within the DPS, and with the adjacent Puget Sound DPS, in addition to understanding genetic exchange with populations in streams in Canada. In the absence of data to the contrary, the SRT concluded that there was no justification in altering the current configuration. Finally, with changes in hatchery practices after many of the existing genetic collections were made, the SRT recommends an updated study evaluating hatchery influences on the genetic diversity of naturally-produced OP steelhead to enable future evaluation of the threats of hatchery practices to the productivity and genetic diversity of natural origin steelhead in the OP DPS.

# DPS Risk Analysis

## Abundance and Productivity

### Previous assessments

In an assessment of salmonid stocks (WDF et al. 1993), 31 stocks were identified within the Olympic Peninsula DPS, of which 23 were considered to be native with predominantly natural production. Of these 11 were identified as healthy with the remaining 12 as unknown. All four native summer-run stocks identified (Bogachiel River, Hoh River, Queets River, and Quinault River) were of unknown status. Of the 12 independent steelhead populations (all winter run) that Busby et al (1996) reviewed, 7 were found to be declining, and 5 increasing. The maximum decrease was 8% per year, with the maximum increase at 14% per year. Busby et al. (1996) estimated that total run size (escapement + harvest) for the DPS was 54,000, with a total natural spawning escapement of 20,000.

During a recent review by WDFW, Cram et al. (2018) reported escapement abundance was only available for 15 populations (all of which were winter run populations) of the 31 identified populations (48%) in the DPS. Of the 15 populations for which there was data, two (Calawah River winter run and Upper Quinault River winter run) exhibited positive abundance trends (1980-2013), with many of the remaining populations having negative trends. Analysis of larger rivers (Quillayute, Hoh, Queets, Quinault) draining to the Pacific (WRIA 20 and 21) indicated that total run sizes had nearly halved from the late 1970s and 1980s to 2022, while the trends in escapements was slightly declining or stable (Harbison et al. 2022) (Figures 15-18). Based on harvest and escapement information from the co-managers (COPSWG 2023), the run size declined for all four major rivers from 32,556 at the time of the Busby review (1991-1995), to 18,821 (2018-2022), a 42% decline.

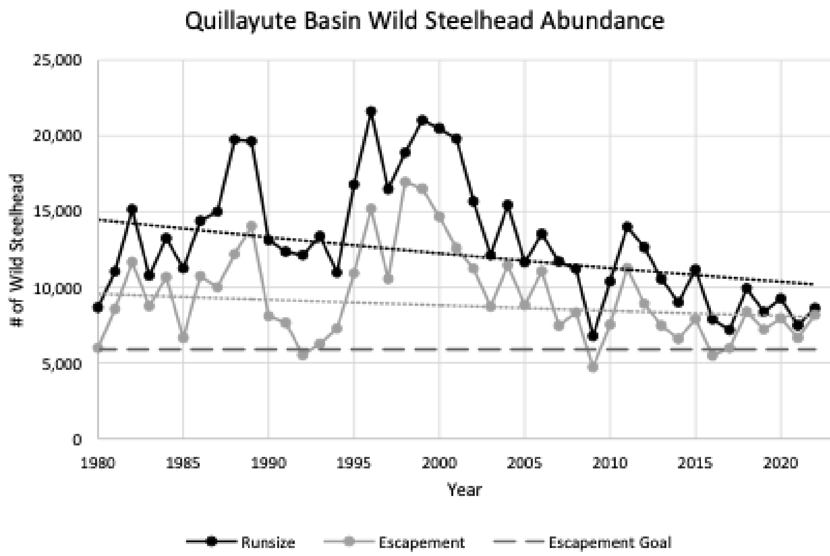



Figure 14. Quillayute Basin naturally-produced steelhead run size and escapement from the 1979/1980 and 2021/22 recreational steelhead fishery seasons, including the Dickey, Calawah, Bogachiel, and Sol Duc rivers. The dashed line indicated the 5,900 steelhead escapement goal. The dotted curves show fitted exponential trends. (From Harbison et al. 2022).

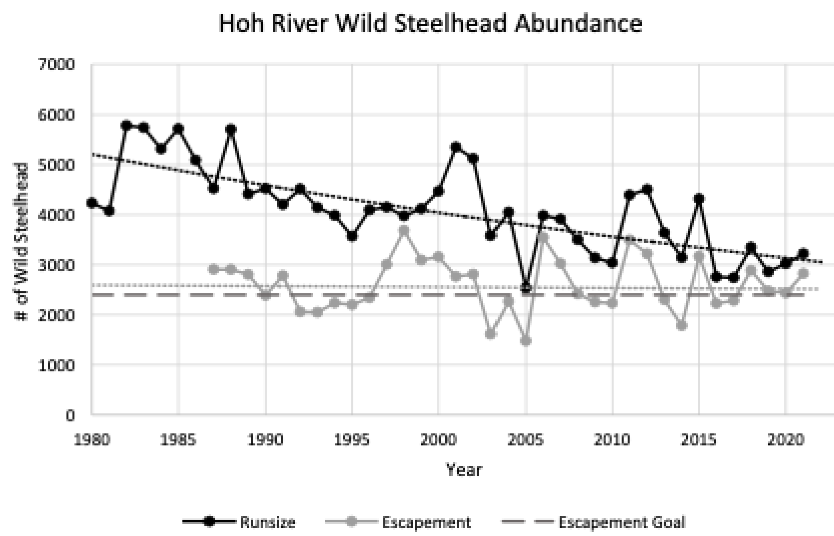



Figure 15. Hoh River naturally-produced steelhead run size and escapement from the 1979/80 to 2021/22. The dashed line indicated the WDFW 2,400 steelhead escapement goal. The dotted curves show fitted exponential trends. (From Harbison et al. 2022).

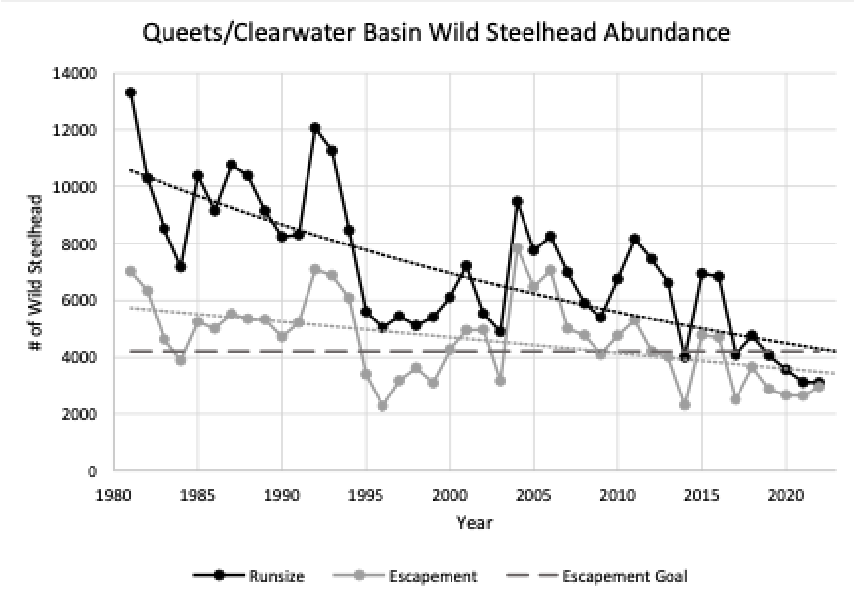



Figure 16. Queets/Clearwater basin naturally-produced steelhead run size and escapement between the 1980/81 and 2021/22 recreational steelhead fishery seasons. The dashed line indicates the 4,200 steelhead WDFW escapement goal, the tribal goal is 2,700. The dotted curves show fitted exponential trends. (From Harbison et al. 2022).

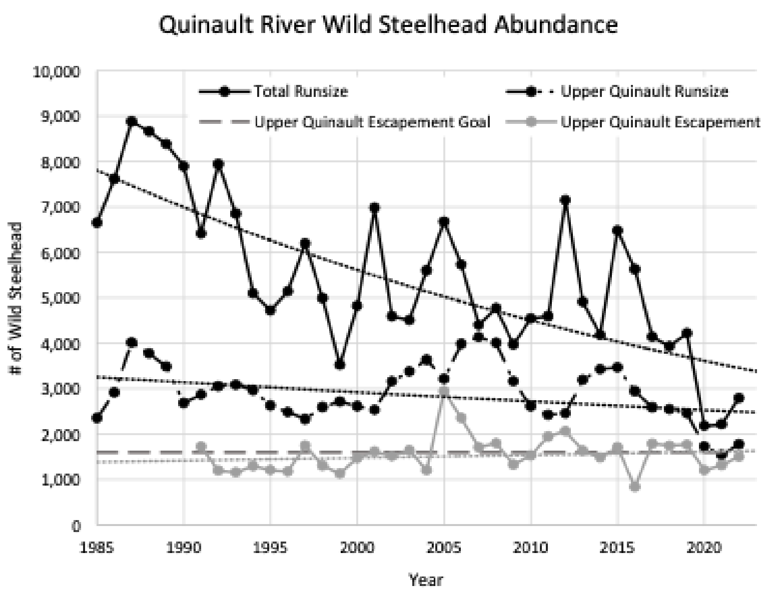



Figure 17. Total run size, Upper Quinault run size, and Upper Quinault escapement from the 1984/85 to 2021/2. The Upper and Lower Quinault River areas are separated because the State of Washington manages a recreational steelhead fishery in the upper river, while the Quinault Tribe manages steelhead in the lower river. The dashed line depicts the 1,200 steelhead escapement goal for the Upper Quinault River. The dotted curves show fitted exponential trends. (From Harbison et al. 2022).

### Hatchery-Origin Steelhead Survival

A recent study comparing the smolt survival of hatchery and natural-origin smolts underscores the complex nature of hatchery and natural interactions (Harbison et al. 2022). Although limited in scope, this analysis suggests that the survival of hatchery smolts is substantially less than that of natural origin smolts and further that it has diminished in recent years. The potential consequences of this decrease in hatchery survival extend beyond the normal considerations of hatchery-natural interactions. Effectively, while hatchery releases have remained relatively unchanged in the major watershed, the returning run of hatchery-origin adults is decreasing, putting further harvest pressure on natural-origin fish. Additionally, if reduced hatchery smolt survival is caused by genetic (as opposed to rearing environment) effects of hatchery propagation, then introgression could result in a degradation of population viability. It should be underscored that this study only tracked the survival of natural-origin smolts in two watersheds, and it does not address any changes in spawning success, incubation survival, and presmolt juvenile survival, that could influence population productivity.

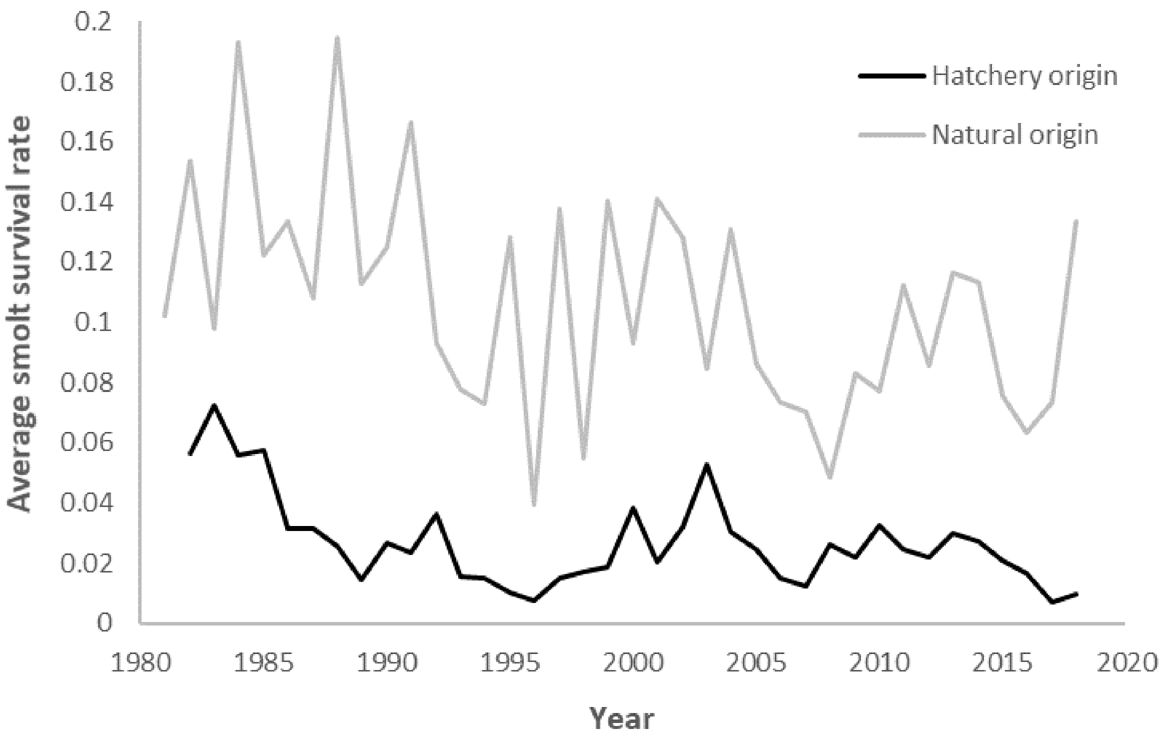



Figure 18. Average coastal Washington steelhead smolt survival rate for the 13 hatchery-origin stocks and 2 natural-origin populations between the early 1980s and 2018. (From Harbison et al. 2022).

### Harvest Rates

Finally, Cram et al. (2018) reported that harvest rates in the OP Steelhead DPS were the highest in Washington State, 36.5% in the four major basins from the 1980s to 2013. Although, it was emphasized that sport fishers were no longer allowed to retain naturally-produced (unmarked) steelhead, continued harvest in the commercial net fisheries could potentially influence size and run-timing selection (Quin et al. 2007, Kendall and Quinn 2011). Further Cram et al. (2018) underscored that non-retention (hooking) mortality and net drop-out rates had been quantified for only a few populations. Finally, the SRT was not aware of any estimates of the level of indirect harvest (bycatch) of steelhead in the commercial or recreational salmon fisheries or the recreational harvest of steelhead on reservation. These data gaps suggest that the Cram et al. (2018) harvest rate estimates may be underestimates.

Table 3. Average annual harvest rates by population and run type. Harvest rate is for winter run natural-origin recruits (NORs) and includes hooking and net drop-out mortality (Hoh River only). Data from Cram et al. (2018).

| WRIA | Population | Run | Average Annual Harvest Rate | Years |
|---|---|---|---|---|
| | | | Cram et al. (2018) | |
| **19** | Clallam River | W | 00.7% | 1999-2013 |
| **19** | Pysht River/Independents | W | 14.0% | 1999-2013 |
| **19** | Salt Creek/Independents | W | 03.9% | 1995-2013 |
| **20** | Quillayute River System | W | 29.6% | 1978-2013 |
| **20** | Goodman Creek | W | 06.8% | 1995-2009 |
| **20** | Hoh River | W | 36.7% | 1980-2013 |
| **21** | Queets River System | W | 35.5% | 1981-2011 |
| **21** | Quinault River System | W | 48.2% | 1991-2013 |

### Repeat Spawner Rate

In contrast to the Pacific salmon, steelhead are capable of iteroparity. The ability to repeat spawn, often within a year of the initial spawning, provides steelhead populations with added productivity and a buffer against decline. Because repeat spawners are larger than first-time spawners, they are able dig deeper, more secure redds, and they have a higher fecundity (repeat spawners are predominantly female). Spawning across multiple brood years ensures gene flow among cohorts and therefore increases genetic variation. Information provided by co-managers indicates that repeat spawning rates (kelt survival rates) were variable and have decreased among the four major coastal rivers over the period of record (Figure 19). The decline in kelts (repeat spawners) would decrease the reproductive potential of a population, for this reason the SRT discussed repeat spawning in the context of productivity rather than as a life history trait.

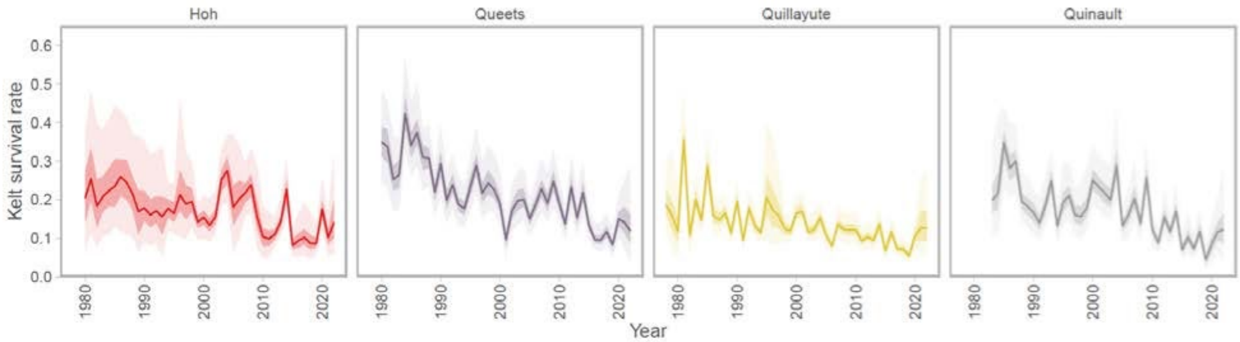



Figure 19. Estimated kelt survival rates by outmigration year for Olympic Peninsula winter steelhead populations. Thick lines with dark and light bands are medians with 50% and 90% confidence intervals. (Originally Figure 13 from COPSWG 2023).

# Status Review Team Analyses for Olympic Peninsula Steelhead

## Introduction

This section provides an overview of demographic data and trends for Olympic Peninsula winter-run steelhead based on the data provided by co-managers (May 15, 2023, COPSEG 2023). Escapement and catch time series data are not available for summer-run steelhead, and their status is discussed separately.

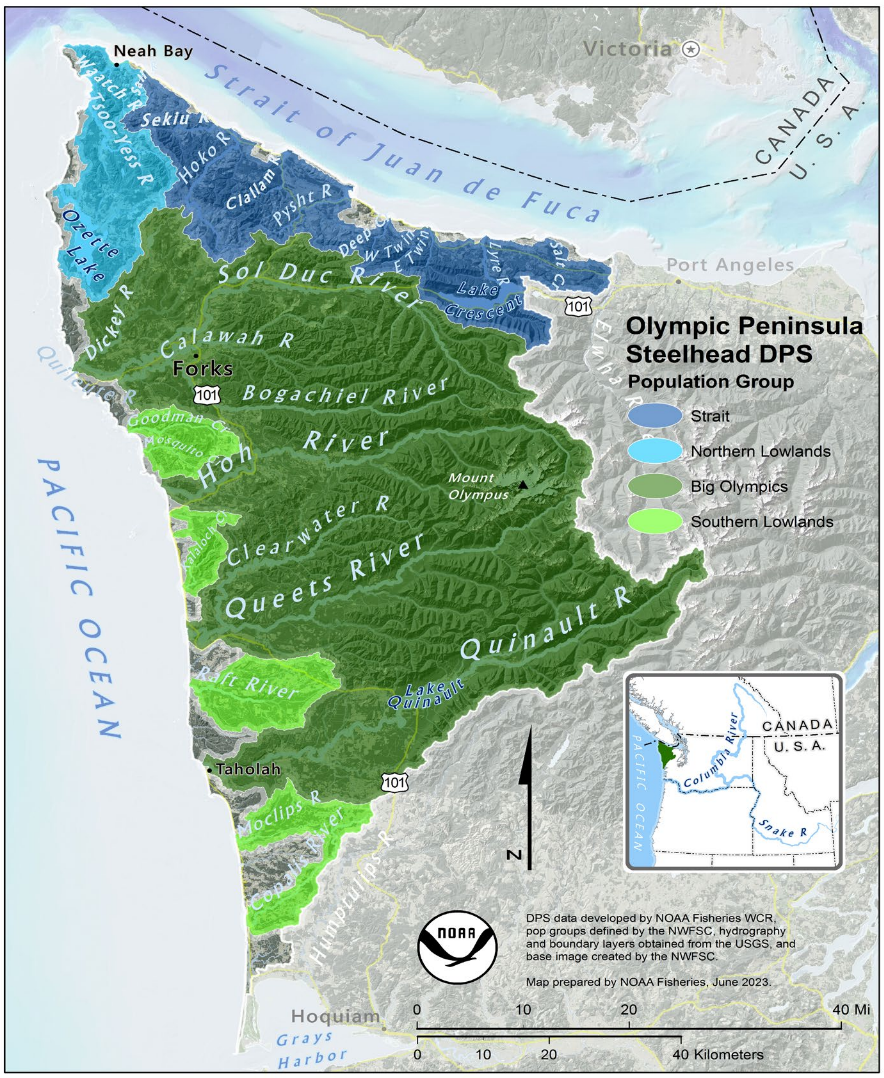



Figure 20. Map of Olympic Peninsula steelhead geographic regions used in demographic analyses.

## Population and data description

### Natural-origin and hatchery-origin steelhead contribution to escapement

The "wild"[11] escapement is based on a set cut-off date not on a survey-based proportion of natural vs hatchery-origin adults. In most cases, the date after which all escapement is categorized as "wild" is March 15 (COPSWG 2023). However, this varies by river (Table 3). Thus, the "wild" escapement is technically "escapement after cutoff date." Some hatchery escapement is included in this number and some naturally-produced escapement occurring before the cutoff date is not included.

### Escapement data summary

Table 4. Summary of the escapement data. Note that the naturally-produced estimates are based on a calendar cut-off date denoted as 'Proportion of spawning season included in annual estimate' in data provide by co-managers. This is typically March 15 but varies by region and year. The escapement goals are from the SCoRE public database. SS – Summer steelhead, WS – Winter steelhead.

| Population | start | end | run | cutoff | goal |
|---|---|---|---|---|---|
| Calawah River | 2012 | 2012 | Summer | | No established goal |
| Calawah River | 1978 | 2022 | Winter(*) | after March 15 | EG=1,740 |
| Clallam River | 1999 | 2022 | Winter | after March 15 | No established goal |
| Deep Creek | 1995 | 2022 | Winter | after March 15 | |
| Dickey River | 1978 | 2022 | Winter(*) | after March 15 | EG=123 |
| East Twin River | 1995 | 2022 | Winter | after March 15 | |
| Goodman Creek | 1995 | 2022 | Winter | after March 15 | Index EG=206 |
| Hoh River | 1976 | 2022 | Winter | after March 15 | EG=2,400 |
| Hoko / Little Rivers | 1985 | 2022 | Winter | after March 15 | |
| Moclips River | 1988 | 2000 | Winter | various dates in March | No established goal |
| Mosquito Creek | 2016 | 2016 | Winter | after March 16 | No established goal |
| Pysht / SF Pysht rivers | 1984 | 2022 | Winter | after March 15 | Index EG=200; Index EG=103; Index EG=86 |
| Queets River (incl. Clearwater) | 1980 | 2022 | Winter | various dates in March | WDFW goal= 4,200 |
| Quillayute-Bogachiel River | 1978 | 2022 | Winter(*) | after March 15 | EG=1,127 |
| Quinault River | 1978 | 2022 | Winter | various dates in March | |
| Salt Creek and Tributaries | 1995 | 2022 | Winter | after March 15 | Index EG=137 |
| Sol Duc River | 1978 | 2022 | Winter(*) | after March 15 | EG=2,910 |
| West Twin River | 1995 | 2022 | Winter | after March 15 | |

Winter(*) – Assumed winter-run, but may include some summer run.

[11] "Wild" was retained here to reflect the language used by the co-managers. As noted previously, NMFS has not otherwise used the term "wild" to describe naturally-produced steelhead, as it can suggest the absence of anthropogenic influences (hatchery-origin or hatchery introgression, direct or indirect selection).

**Escapement goals**

Harvest and escapement levels of Olympic Peninsula steelhead have been largely governed by the principle of maximum sustainable yield (MSY), in large part because it was established by the landmark 1974 Federal court case (*United States v. Washington* , 384 F. Supp. 312 (W.D. Wash. 1974) (" *Boldt Decision* ")). The theoretical underpinning of MSY is that there exists a maximum level of harvest for any given population which can be sustained in perpetuity (Ricker 1975). In theory, if one understands the underlying productivity of a population, this harvest level can be calculated and used to establish management objectives that will ensure a stock's persistence over time. Generally, the management objectives are either expressed in terms of "escapement goals" (number of adult fish which survive to spawn) or "harvest rates" (proportion of the total population which may be harvested)" (Duda et al 2018). The co-managers have established escapement goals for wild steelhead in several rivers of the OP DPS (Table 4).

The river systems throughout the Olympic Peninsula DPS support sport fishing and commercial, ceremonial, and subsistence gill-net fisheries, with Pacific salmonid populations subjected to fishing pressure and harvest during most months of the year. The recreational fisheries, which include guided and non-guided sport fishing for Pacific salmon and steelhead are economically important to local communities. Commercial catches of Pacific salmonids are integral to the tribal fisheries, and fish are sold to local, regional, and national markets. Subsistence catch is for personal consumption and ceremonial catch occurs for cultural events. There is no direct ocean harvest of steelhead. Adult steelhead that "escape" harvest in recreational and commercial fisheries contribute to the abundance of populations. Busby et al. (1996) reported different escapement goals from different sources, and we have summarized these and included the actual goals used by the co-managers (Table 4).

Table 5. Escapement goals listed in the SCoRE database versus those in Busby et al. 1996 and from WDFW (R. Cooper) for the Strait group. QIN: Quinault Indian Nation.

| Population | WDFW SCoRe | WDFW R. Cooper | Busby et al 1996 (total) | Busby et al. 1996 (natural) | Current Goal |
|---|---|---|---|---|---|
| Moclips River | | | 400 | 250 | -- |
| Quinault River | | | 6,300 | 3,400 | 1,600 (Up.Quinault) |
| Queets River | | | 7,400 | 5,900 | 4,200 WDFW<br>2,500 QIN |
| Hoh River | 2,400 | | | 2,300 | 2,400 |
| Goodman Creek | 206 | | | | |
| Mosquito Creek | | | | | |
| Quillayute-Bogachiel River | 1,127 | | 8,300 | 6,900 | 1,127 |
| Calawah River | 1,740 | | | | 1,740 |
| Sol Duc River | 2,910 | | | | 2,910 |
| Dickey River | 123 | | | | 123 |
| Hoko / Little River | | 440 | | 550 | |
| Clallam River | | 144 | | | |
| Pysht / SF Pysht rivers | 389 | 185 | 400 | 250 | |
| Deep Creek | | 99 | | | |
| West Twin River | | 103 | | | |
| East Twin River | | 86 | | | |
| Salt Creek and Tributaries | 137 | 137 | | | |

## Trend Analysis

### Correlation structure

The correlation plot (**Error! Reference source not found.**) shows how the escapement time series are correlated across the rivers. Based on clustering, they fall into clusters of smaller systems with tributaries to the Quillayute River while the 3 other large watershed systems (Hoh, Queets, and Quinault) being independent of one another.

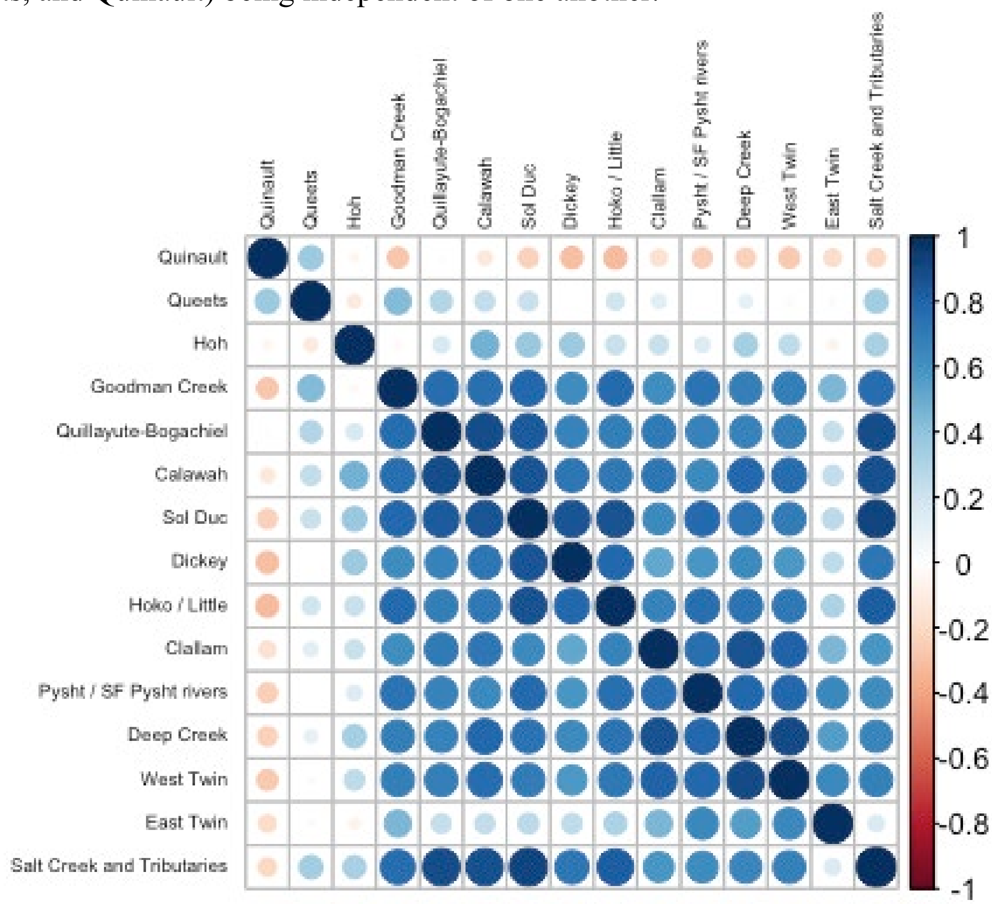



Figure 21. Correlation across the observed naturally-produced escapement after the cutoff dates in March. In this plot, a clustering algorithm is applied that finds similar clusters. Populations are arranged south (top) to north (bottom).

### Smoothed escapement estimates

To understand trends in the escapement trends of Olympic Peninsula steelhead, we follow Ford et al. (2022) and use multivariate dynamic linear modeling (DLM) to estimate population-specific trends. The DLMs provide an estimate of the smoothed spawner counts after accounting for observation and process errors (see Ford et al. 2022 and citations therein for details). For all component populations, we calculate smoothed time-series of spawner abundances, geometric-mean abundances for each 5-year window, and population trends over 15-year window of the time-series.

In addition, we sum the component population abundances to provide a time-series of aggregate abundance across the individual winter-run populations. We use a Bayesian DLM (ref. O. Shelton) using the statistical software *Stan* 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 constructed a DLM model using total escapement data for each river and separately for estimated total run size (escapement plus harvest) where available (four rivers). We used a single observation variance for all winter-run populations and a single process variance and single covariance for the process covariance (equivalent to the *MARSS* options: R = "diagonal and equal", Q = "equalvarcov", respectively). No information on the fractions of natural-origin spawners is available for populations; for the purposes of these analyses, the fraction naturally-produced was assumed to be 1 for the escapement data. This follows the assumption from the co-managers that escapement after the March 13-30 cut-off date (Table 3) are almost exclusively natural-origin spawners.

We present 15-year trends derived from linear regressions of year against log-transformed escapement estimates from the DLM against years (Figure 24, Table 8). We calculated geometric means for each 5-year period for each population using output from the MARSS model (Table 10, Table 11).

**Escapement estimates**

**By population**
These represent the DLM estimates from the data summarized (Table 3) and only concern winter-run populations (Figure 22).

**Aggregate (entire region)**
We combined the escapement estimates for each stock to provide an aggregate time-series for the total spawner abundance (Figure 23). The Bayesian DLM provides smoothed estimates of the abundance of each stock in each year (replicate draws from the posterior distribution of abundance in river in each year) and we summed across stocks to arrive at an estimate of total spawner abundance within each stratum as well as across all Olympic Peninsula winter-run stocks (Figure 22).

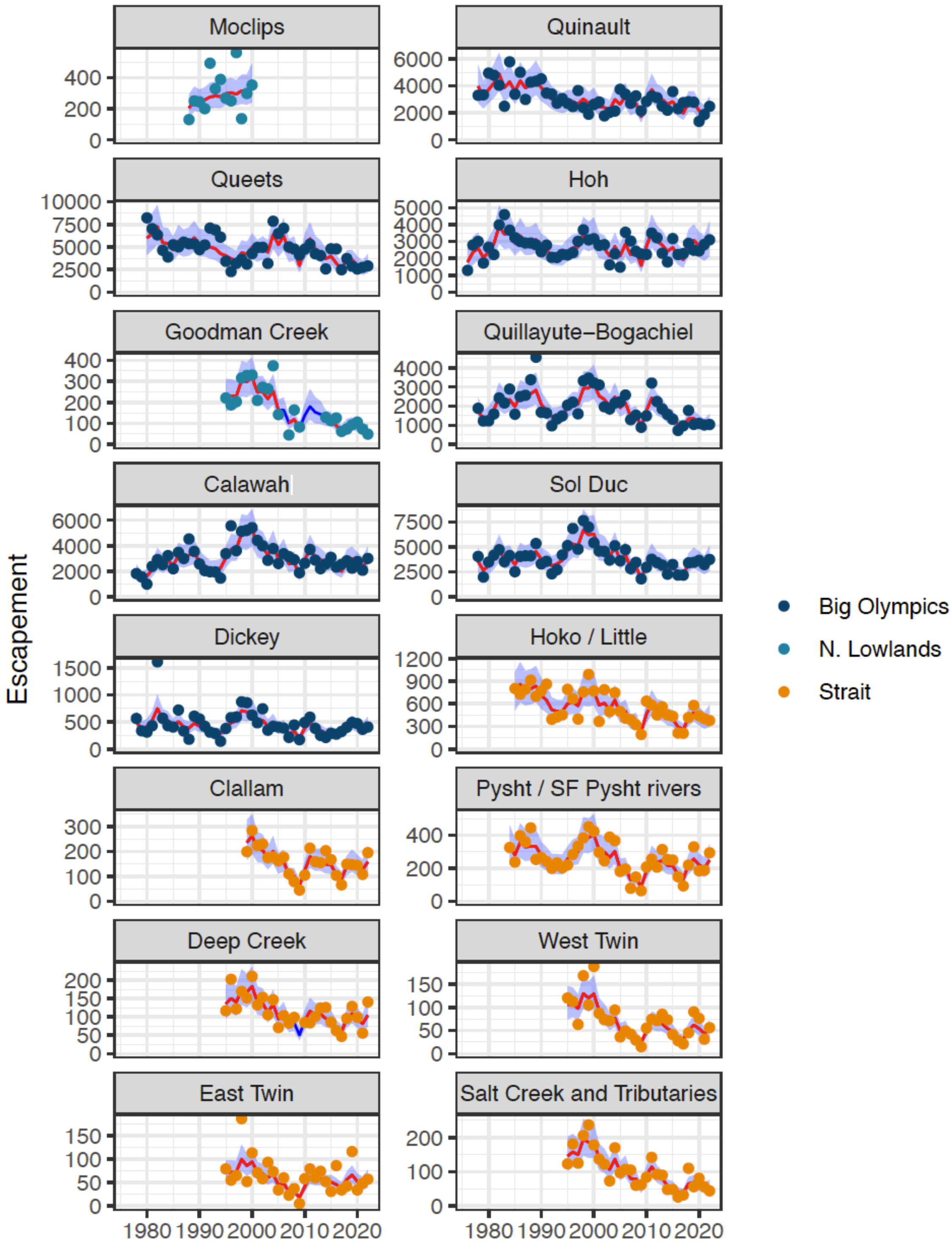



Figure 22. Total escapement after calendar cut-off (assumed primarily natural origin) for winter-run populations in the Olympic Peninsula. Points show observations, blue line and shaded area shows model predictions of abundance and 95% CI. There was no information to determine hatchery contribution, thus plots simply show total escapement after the cutoff dates (Table 3).

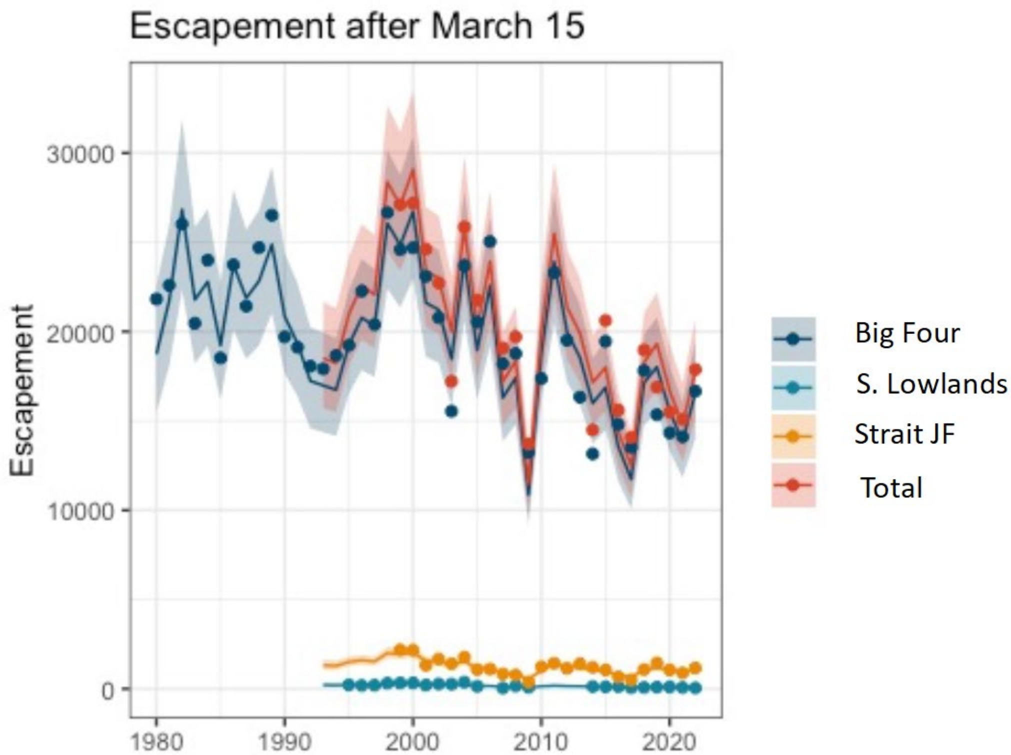



Figure 23. Escapement time-series summed across all rivers and creeks for winter-run steelhead, 1993-2021 (excluding Moclips because that population has no data after 2000). Aggregate is the sum of the smoothed estimates from the DLM within each spatial stratum. Total shows the combined abundance across all strata (1993-2022) Points show observed abundance estimates for years in which all populations within a stratum have observed counts.

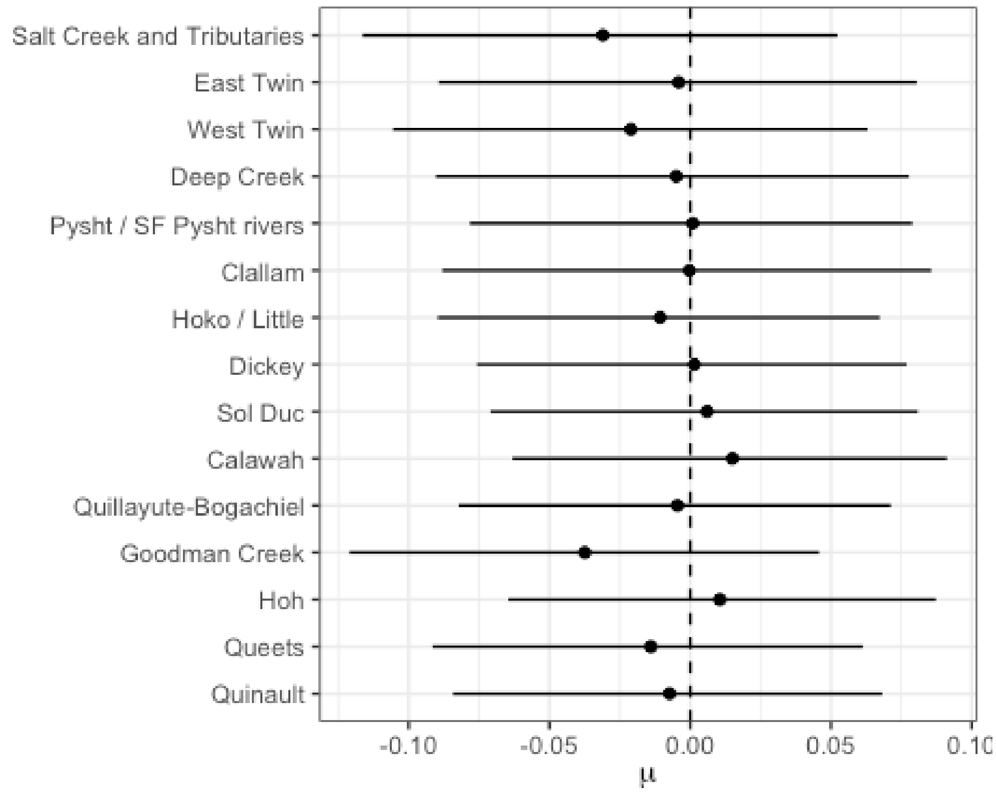



Figure 24 Estimated trend for each population over the time-series estimated by the DLM.

**Summer-run escapement data**

There have been several efforts to examine the status and trends, of summer-run steelhead in the Olympic Peninsula (Nehlsen et al. 1991; Cooper and Johnson 1992; WDF et al. 1993, and McHenry et al. 1996). In most cases summer-run steelhead were either not identified or their overall abundance and associated status and trends categorized as "unknown" (WDF et al. 1993, and McHenry et al. 1996). SASSI (WDF et al. 1993) did identify summer steelhead populations as being present in the Sol Duc, Bogachiel, and Calawah watersheds. Summer-run steelhead in the Queets, Quinault, Hoh, Sol Duc, Bogachiel, and Calawah rivers were all described as distinct stocks from winter steelhead, based on run timing and geographical isolation of the spawning areas (WDF et al. 1993). Escapement was categorized as unknown and not monitored, and the status of summer runs was unknown with the exception of the Queets population, which was judged healthy based on combined sport and tribal harvest of wild steelhead (WDF et al. 1993), although harvest can be a misleading indicator of status. McHenry *et al.* (1996) identified summer steelhead populations in the Hoh, and Queets/Clearwater, but reported no actual population estimates. SASSI (WDF et al. 1993) did not recognize summer-run steelhead in any of the watershed draining to the Strait of Juan de Fuca (WRIA 19).

Busby *et al.* (1996) found very little information on the abundance and status of summer steelhead in this region and the degree of interaction between hatchery and natural stocks. Since

1996, several efforts have produced data on the number of adult summer steelhead in streams of the Olympic Peninsula, but none used methods to produce statistically unbiased estimates of breeding population size, commonly equated to the number of holding adults prior to spawning season. Cram et al. (2018) identified summer steelhead populations in the Clearwater, Hoh, Queets, Bogachiel, Quinault, and Sol Duc systems, but reported no trends, extinction risk, status relative to an abundance goal, or overall risk rating due to insufficient data (Cram et al. 2018).

Because there are no spawner surveys done that specifically target summer-run steelhead redds, the only available estimates of abundance come from harvest data and a limited number of snorkel surveys that counted steelhead adults during prespawn holding. Snorkel surveys of holding adults can validly indicate trends in breeding population size, if based on a statistically sound sampling design for reaches, and if bias-corrected for imperfect detection rate (Boughton et al. 2022).

Staff of Olympic National Park have collected summer steelhead information as part of their fish assemblage monitoring program since 2004. They conducted snorkel surveys at ~5-km reference sites in several coastal rivers from June to September, 2004 to 2012 using methods described in Brenkman and Connolly (2008). The monitoring objectives were to determine seasonal and annual trends in: (1) fish species composition, (2) migration timing of adult fish, (3) relative abundance, and (4) relative extent of hatchery fish. These surveys were temporally intensive, but spatially limited to reference sites that were not necessarily representative of the encompassing river system, and so cannot be expanded to make inference about total breeding population size. Although not designed specifically to assess trends in summer steelhead, comparisons among years provide information on trend (Figure 25). The time-series must be interpreted with caution, because reference sites are often selected from better-than-average habitat, and can therefore mask downward trends if habitat selection is density-dependent.

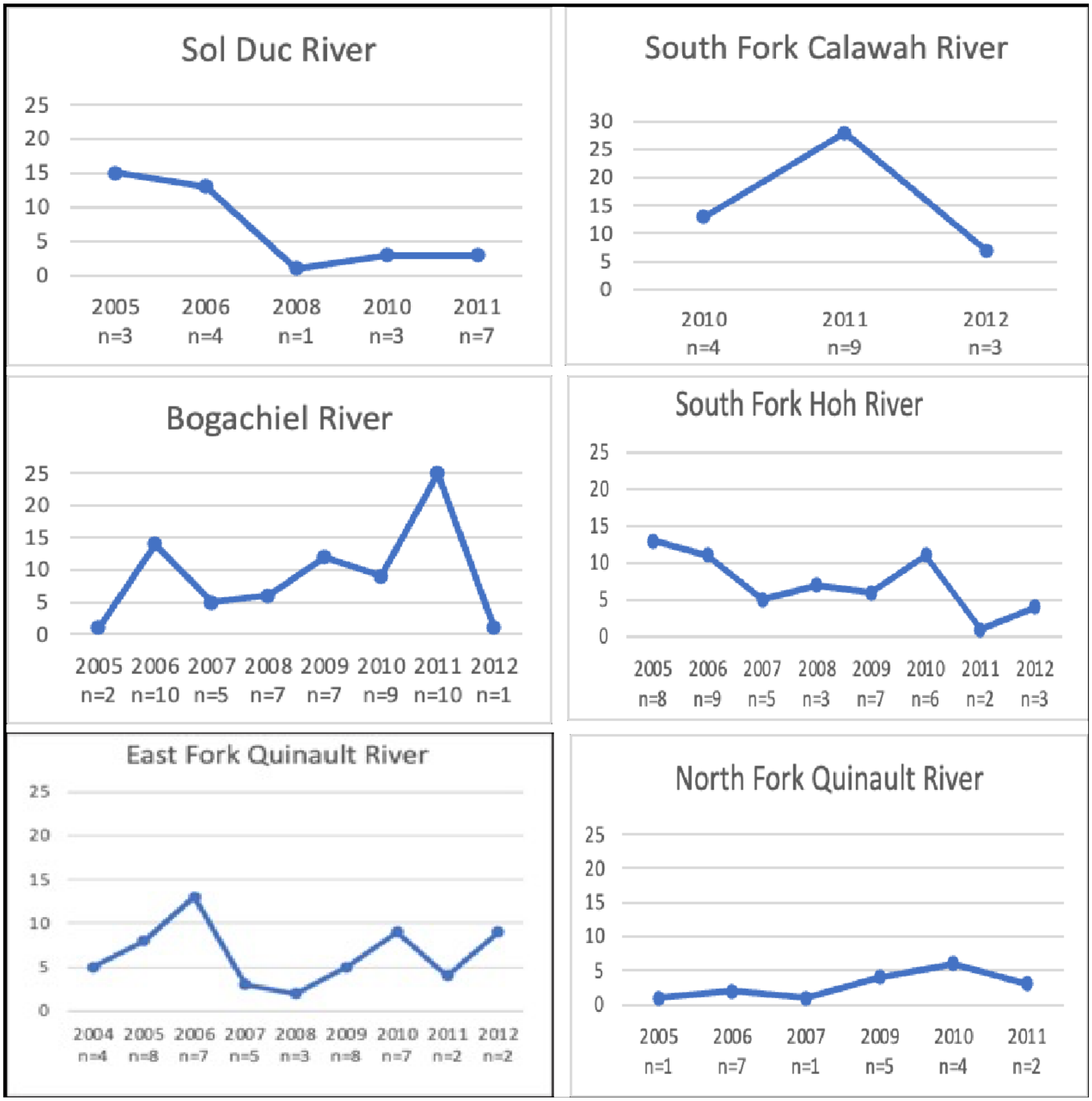



Figure 25. Annual peak counts of adult summer steelhead per 5-km, from snorkel surveys conducted during summer months at reference sites. Reference sites were in each of six rivers draining from Olympic National Park, with the annual number of repeat surveys *n* reported in the labels of the X-axis. Counts include both natural- and hatchery-origin adults (see Table 5). See Brenkman and Connolly (2008) for details.

The counts (Table 5) show no consistent trends up or down, but do show a presence, albeit at consistently low numbers: almost always fewer than 3 holding adults per kilometer of stream channel, but occasionally as high as 5.6. These reference sites were used mostly by wild-origin summer steelhead, but hatchery-origin fish were observed in all except the Sol Duc, and outnumbered the wild-origin fish in the South Fork Hoh River (Table 5).

Table 6. Proportions of hatchery-, natural-, and unknown origin adult steelhead observed in reference sites during summer months, for six rivers draining Olympic National Park. Unmarked fish are assumed to be natural origin.

| Population | Hatchery Origin | Natural Origin | Unknown Origin | Total Observed |
|---|---|---|---|---|
| Sol Duc River | 0% | 65% | 35% | 55 |
| SF Calawah River | 22% | 74% | 4% | 144 |
| Bogachiel River | 16% | 71% | 13% | 189 |
| SF Hoh River | 46% | 39% | 14% | 142 |
| EF Quinault River | 8% | 69% | 23% | 180 |
| NF Quinault River | 34% | 57% | 9% | 35 |

More recently, snorkel surveys have been conducted using a more spatially extensive "riverscape" approach. Scientists from Olympic National Park, as well as the United States Geological Survey (USGS), United States Fish and Wildlife Service (USFWS), NOAA, and WDFW conducted riverscape surveys in the Olympic Peninsula using methods described by Brenkman et al. (2012) and Duda et al. (2021). These surveys (Table 6), while using snorkeling methods similar to the reference site surveys, covered entire river systems and obtained information on the spatial extent, relative abundance among rivers, and relative proportion of hatchery and wild summer steelhead, for the larger rivers of the Olympic Peninsula (Brenkman et al. 2012; Duda et al. 2021).

Table 7. Statistics from riverscape surveys of summer steelhead in key coastal watersheds. Spatial extent of observed summer run steelhead, numbers observed by origin (hatchery-, natural-, unknown-origin steelhead) and totals for surveys conducted by staff of ONP, WDFW, NOAA, USGS, USFWS, Treaty Tribes, and other project partners (ONP files, unpublished).

| Survey Coverage | | | | Adult Summer Steelhead Counts | | | |
|---|---|---|---|---|---|---|---|
| **River** | **Survey dates** | **Distance Surveyed (Rkm to Rkm)** | **Spatial Extent (Rkm)** | **Hatchery Origin** | **Natural Origin** | **Unknown Origin** | **Total Observed** |
| Bogachiel River | 8/1-8/4/2016 | 0-55.6 | 6.2-45.1 | 4 (15%) | 16 (62%) | 6 (23%) | 26 |
| SF Hoh River | 9/13-9/15/2016 | 0-22.3 | 1.2-22.3 | 3 (5%) | 19 (33%) | 35 (61%) | 57 |
| SF Hoh River | 9/23/2003 | 0-21.0 | NA | 33 (54%) | 28 (46%) | 0 (0%) | 61 |
| SF Hoh River | 10/1/2002 | 0-21.0 | NA | 21 (27%) | 56 (73%) | 0 (0%) | 77 |
| Quinault River | 8/17-8/21/2009 | L. Quinault -51.4 | 0.5-48.5 | 1 (1%) | 108 (95%) | 5 (4%) | 114 |
| Sol Duc River | 8/18-8/21/2014 | 0-99.3 | 3.0-88.6 | 38 (26%) | 55 (37%) | 54 (37%) | 147 |

Similar to the reference sites in the riverscape surveys the rivers were used mostly by wild-origin summer steelhead, but hatchery-origin fish were observed in all rivers, this time including the Sol Duc, and outnumbered the wild-origin fish in the South Fork Hoh in one of the three years it was surveyed (Table 5, Table 6).

Assuming that observation probability was high (~1.0), and that unknown-origin fish had the same wild proportion as known-origin fish, breeding population size of the wild component was generally less than 120 summer steelhead per river, and often much less (Table 6). The average across rivers was 66 breeding fish per year, or roughly a breeding population size of ~260 per river assuming a 4-year generation time. This modest number of adults indicates high risk of population-level extinction using the rating scheme of Allendorf et al. (1997), and is close to the threshold for very high risk (<250). One would expect very high levels of genetic drift in these populations, and thus loss of wild genetic diversity and inbreeding depression over time (Allendorf et al. 1997), although modest levels of gene flow among the various summer steelhead populations would counteract this tendency. Even a very low observation probability (say, 50%) would not change this conclusion very much, implying an average breeding population of ~520, which is still high risk (Allendorf et al. 1997).

These snorkel surveys also characterized the spatial extent and patterns of relative abundance of adult hatchery and wild summer steelhead, including georeferenced data in the major rivers in the coastal portion of the DPS (Figure 26). As with the reference sites, densities were low, averaging about 1.6 adults/km and always less than 2.4. Moreover, wild-origin and hatchery-origin fish often co-occurred within the same kilometer of river channel (Figure 26), increasing their likelihood of interbreeding, and, depending on the degree of interbreeding, impacting fitness of offspring via maladaptation. Hatchery adult summer steelhead were detected as high as Rkm 59.1, 45.1, and 21.2 in the Sol Duc, Bogachiel, and South Fork Hoh Rivers, respectively. Notably, the last recorded hatchery plantings of summer steelhead occurred in the Hoh system in 1983, and there have been no reported hatchery outplantings of summer steelhead into the Quinault system, so the recent observations of hatchery-origin adults in these systems imply straying of hatchery-origin summer steelhead. Houston and Contor (1984) similarly noted that hatchery summer steelhead were straying from unknown release locations to the Hoh, Queets, and Quinault since 1979. The low level of monitoring for the presence of hatchery adults in natural spawning areas prevents any quantification of this risk; however, based on available information this risk is not negligible.

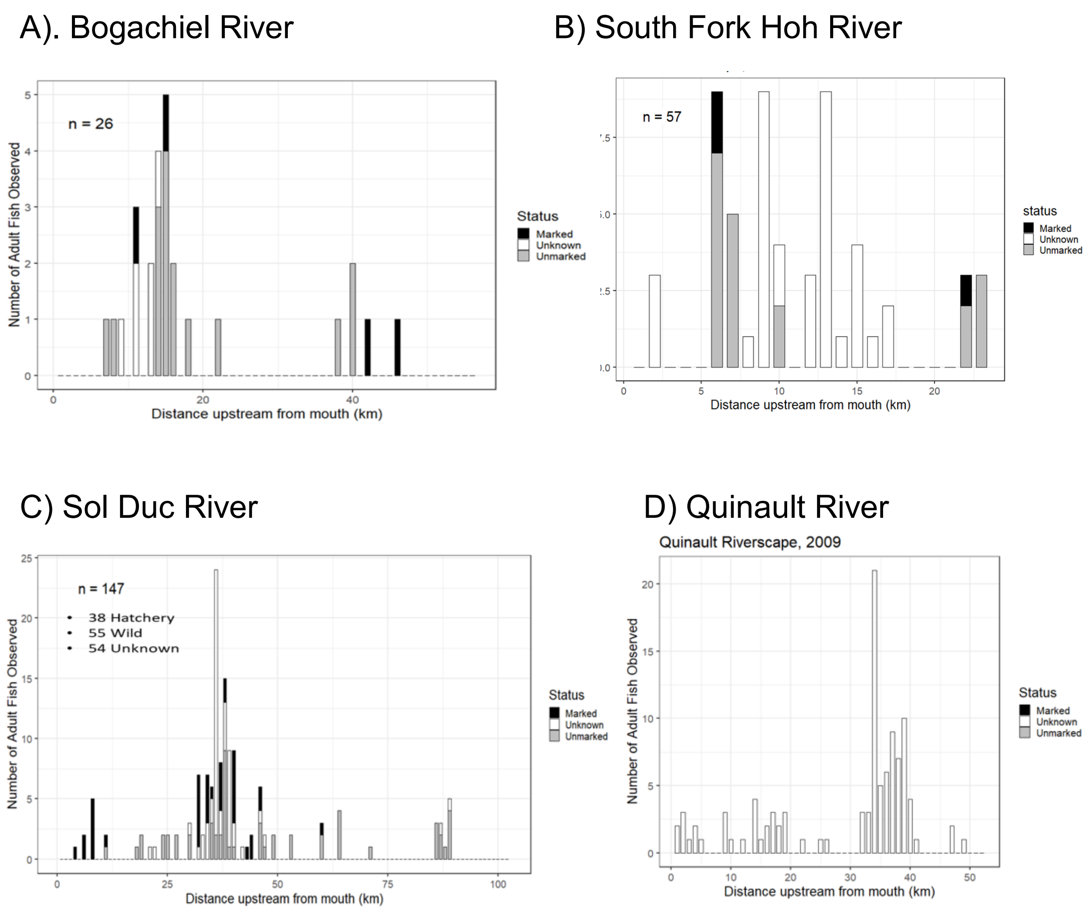



Figure 26. Distribution and relative abundance of adult summer steelhead counted in the continuous snorkel surveys ( see Table 6). Longitudinal profiles of adult steelhead were plotted at 1 km spatial scale indicated as bin lengths.

The petition for ESA-listing used these and other data from the Olympic Peninsula to assert that almost all summer steelhead populations are at critically low levels, while noting that there is no formal analysis of historical catch and no monitoring by the co-managers. The petitioners provided rough estimates of peak historical abundance for summer-run steelhead on the Olympic Peninsula, using harvest data for the larger systems (Quinault, Hoh, Quillayute, and Queets). They estimated that total abundance of summer-run steelhead in these systems ranged from 848 to 1,788 adult spawners from the late 1940s/early 1950s to the late 1970s.

Using the snorkel survey information from Brenkman *et al.* (2012) and McMillan (2022), the petitioners estimated recent numbers of adult summer steelhead returning to spawn each year in several different streams (Calawah River system, North Fork Calawah River, South Fork Calawah River, Sitkum River, and South Fork Hoh River for Brenkman *et al.*, 2012; Bogachiel River, Sol Duc River, South Fork Hoh River, East Fork Quinault River, and North Fork Quinault for McMillan, 2022). Mean estimates ranged from 3 to 303 individuals. The Calawah River is at the upper end of this range, but most of the returning summer steelhead were hatchery-origin (89 native-origin, 214 hatchery-origin). For other rivers, the mean proportion of hatchery-origin spawners ranged from 3 to 43 percent. According to the petitioners, McMillan (2006) estimated that summer steelhead abundance in the Queets River and Clearwater River was no more than 100 fish based on catch data.

Utilizing the snorkeling surveys from ONP and the petitioners, the co-managers developed an alternative set of abundance estimates for summer steelhead populations in the Hoh, Quillayute, and Quinault River systems (Table 7). Their analysis utilizes both the index survey data from the fish assemblage data collected by ONP, the riverscape surveys efforts of the ONP, as well as the petitioners' efforts in the South Fork Calawah to develop an estimate of both estimated unclipped escapement and terminal run size (Co-Manager Olympic Peninsula Steelhead Working Group 2023). The use of expanded index surveys generally introduces considerable uncertainty into population estimates, as reflected in the broad range for estimates (Table 7). Furthermore, because index areas are generally selected for their likelihood of occupancy, expansions into total habitat are biased to overestimate abundance.

Table 8. Median co-manager's estimates of naturally-produced summer-run steelhead populations in the Olympic Peninsula Steelhead DPS (COPSWG 2023)

| River | Estimate | Range |
|---|---|---|
| Hoh River | 210 | 123-516 |
| Bogachiel River | 90 | 53-221 |
| SF Calawah/Sitkum River | 330 | 193-809 |
| Sol Duc River (2009) | 545 | 320-1,337 |
| Sol Duc River (2014) | 552 | 324-1,355 |
| Quinault River | 545 | 320-1,337 |

## Escapement trends

### 15-year trends

The DLM escapement estimates were used to calculate 15-year trends (Figure 27, Table 8). A minimum of two observations (escapement estimates) in the first 5 years of the 15-year window and two observations in the last 5 year of the window were required to report a trend estimate. This was to ensure that we did not report trend estimates when there was no data to constrain the beginning and end of the 15-year segment. Populations in the Strait group have considerably smaller abundances (Figure 22) than in the Big (Four) Olympics group which includes the larger river systems (Quillayute, Hoh, Queets, and Quinault) and the effect of each population's trend to overall DPS viability will be proportional. The smaller basins at Cape Flattery and along the Strait of Juan de Fuca contain rain-driven streams with limited year-round rearing habitat. Martin (2023) reports that even pre-contact, the run sizes of steelhead in this area were never very large.

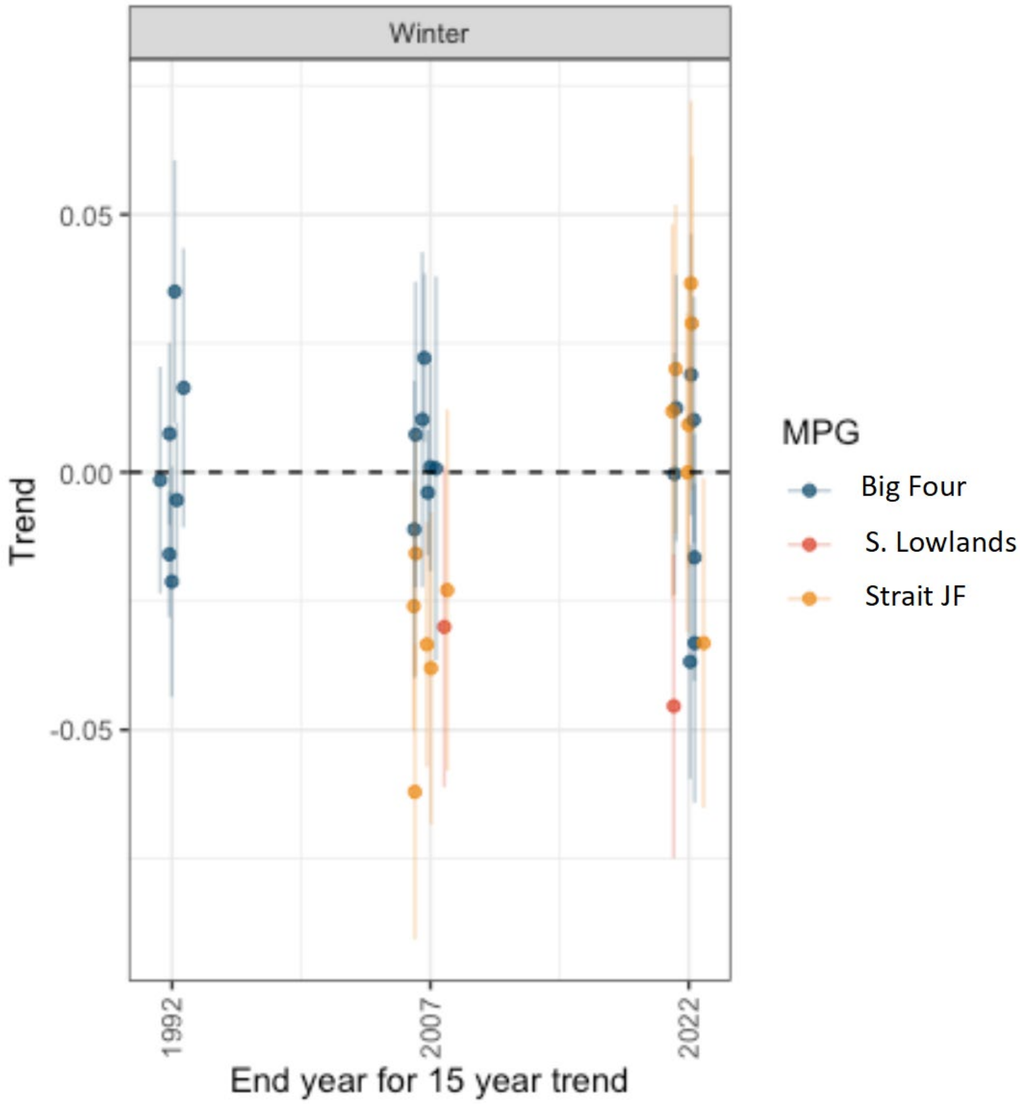



Figure 27. 15-year escapement trends estimated for winter-run stocks (total escapement after cut-off). Points show estimated trend through time and 95% CI for individual stocks. The end of the 15-year window is the year in the x axis. Only 15-year windows where at least 2 observations (data points) are in the first 5 years and 2 observations are in the last five years are shown. Note that the populations in the Strait JF group are considerably smaller (Figure 22).

Table 9. 15-year trends (slope) in log total spawner abundance for winter-run stocks. In parentheses are the upper and lower 95% CIs. Only populations with at least 2 data points (observations, not estimates) in the first 5 years and last 5 years of the 15-year ranges are shown. Populations are ordered south to north.

| Population | MPG | 1978-1992 | 1993-2007 | 2008-2022 |
|---|---|---|---|---|
| Moclips River | Southern Lowlands | | | |
| Quinault River | Big Olympics | -0.01(-0.02,0.01) | 0(-0.02,0.01) | -0.02(-0.04,0.01) |
| Queets River | Big Olympics | -0.02(-0.03,0) | 0.02(0.01,0.04) | -0.04(-0.06,-0.01) |
| Hoh River | Big Olympics | 0(-0.02,0.02) | 0(-0.02,0.02) | 0.01(-0.01,0.03) |
| Goodman Creek | Southern Lowlands | | -0.03(-0.06,0) | -0.05(-0.07,-0.02) |
| Quillayute-Bogachiel River | Big Olympics | 0.02(-0.01,0.04) | 0.01(-0.02,0.04) | -0.03(-0.06,0) |
| Calawah River | Big Olympics | 0.04(0.01,0.06) | 0.01(-0.02,0.04) | 0(-0.02,0.02) |
| Sol Duc River | Big Olympics | 0.01(-0.01,0.03) | -0.01(-0.04,0.02) | 0.01(-0.01,0.04) |
| Dickey River | Big Olympics | -0.02(-0.04,0) | 0(-0.04,0.04) | 0.02(-0.01,0.05) |
| Hoko / Little River | Strait | | -0.02(-0.04,0.01) | 0(-0.03,0.03) |
| Clallam River | Strait | | | 0.02(-0.01,0.05) |
| Pysht / SF Pysht rivers | Strait | | -0.02(-0.06,0.01) | 0.03(0,0.06) |
| Deep Creek | Strait | | -0.03(-0.05,0) | 0.01(-0.02,0.04) |
| West Twin River | Strait | | -0.06(-0.09,-0.03) | 0.01(-0.02,0.05) |
| East Twin River | Strait | | -0.04(-0.07,-0.01) | 0.04(0,0.07) |
| Salt Creek and Tributaries | Strait | | -0.03(-0.06,-0.01) | -0.03(-0.07,0) |

**Pre- and post-Busby trends**

In addition to the 15-year trends, the trends for 1977-1994 corresponding to the years used in considered by Busby et al. (1996) were compared to the most recent trends (1995-2021) (Figure 28). For those winter run populations where trends could be calculated the overall trend was more negative than at the time of the Busby et al. (1996) review. This decline in trend was especially prevalent in steelhead populations in the major (Big Four) basins where the majority of the DPS abundance lies. Differences between the Busby and post-Busby periods for Strait of Juan de Fuca populations are due to the inclusion of additional populations in the more recent interval and the termination of recreational harvest.

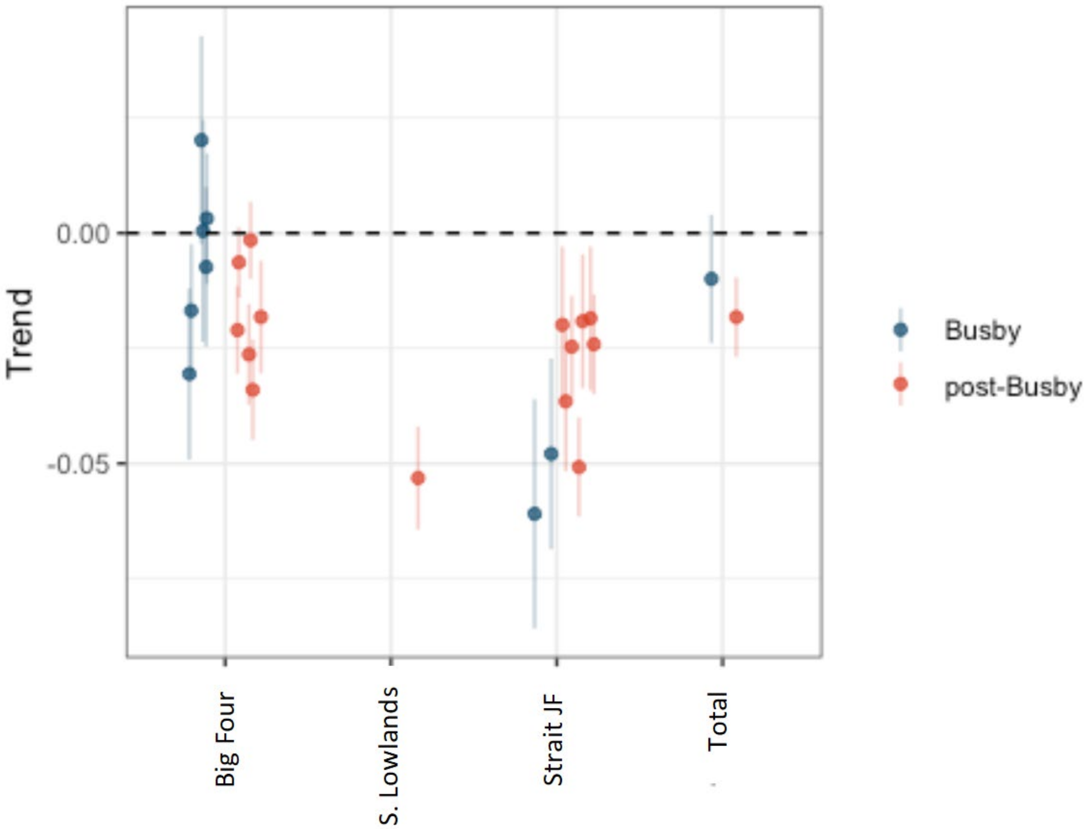



Figure 28. Escapement trends estimated for winter-run stocks (total escapement after cut-off) for the Busby (1977-1994) and post-Busby (1995-2022) periods. Points show estimated trend and 95% CI.

Table 10. Trends in log total escapement for winter-run stocks in the Busby period (1977-1994) and post-Busby period (1995-2021). In parentheses are the upper and lower 95% CIs. Only the range from the first year with data and last year with data were used. Populations are ordered south to north. The first row shows the aggregate (sum over all stocks) trends.

| Population | MPG | year range | Busby et al. | year range post | post-Busby et al. |
|---|---|---|---|---|---|
| Aggregate | Total | 1977-1994 | -0.01 (-0.02,0) | 1995-2022 | -0.02 (-0.03,-0.01) |
| Moclips River | S. Lowlands | 1988-1994 | 0.05 (0.02,0.07) | 1995-2000 | 0.03 (0.01,0.05) |
| Quinault River | Big Four | 1978-1994 | -0.02 (-0.03,0) | 1995-2022 | -0.01 (-0.01,0) |
| Hoh River | Big Four | 1977-1994 | -0.01 (-0.02,0.01) | 1995-2022 | 0 (-0.01,0.01) |
| Goodman Creek | S. Lowlands | 1977-1994 | | 1995-2022 | -0.05 (-0.06,-0.04) |
| Quillayute-Bogachiel River | Big Four | 1978-1994 | 0 (-0.02,0.02) | 1995-2022 | -0.03 (-0.04,-0.02) |
| Calawah River | Big Four | 1978-1994 | 0.02 (0,0.04) | 1995-2022 | -0.02 (-0.03,-0.01) |
| Sol Duc River | Big Four | 1978-1994 | 0 (-0.01,0.02) | 1995-2022 | -0.03 (-0.04,-0.02) |
| Dickey River | Big Four | 1978-1994 | -0.03 (-0.05,-0.01) | 1995-2022 | -0.02 (-0.03,-0.01) |
| Hoko / Little River | Strait JF | 1985-1994 | -0.06 (-0.09,-0.04) | 1995-2022 | -0.02 (-0.03,-0.01) |
| Clallam River | Strait JF | 1977-1994 | | 1999-2022 | -0.02 (-0.04,0) |
| Pysht / SF Pysht rivers | Strait JF | 1984-1994 | -0.05 (-0.07,-0.03) | 1995-2022 | -0.02 (-0.03,0) |
| Deep Creek | Strait JF | 1977-1994 | | 1995-2022 | -0.02 (-0.04,-0.01) |
| West Twin River | Strait JF | 1977-1994 | | 1995-2022 | -0.04 (-0.05,-0.02) |
| East Twin River | Strait JF | 1977-1994 | | 1995-2022 | -0.02 (-0.03,0) |
| Salt Creek and Tributaries | Strait JF | 1977-1994 | | 1995-2022 | -0.05 (-0.06,-0.04) |

## Means and geomeans of escapement

### 15-year mean

The DLM escapement estimates are used to calculate 15-year means. A minimum of 2 years in the first 5 years of the 15-year window and 2 years in the last 5 year of the window were required to report an estimate (Figure 29). The graph indicates a decline in abundances in each of the geographic regions, although none is significant.

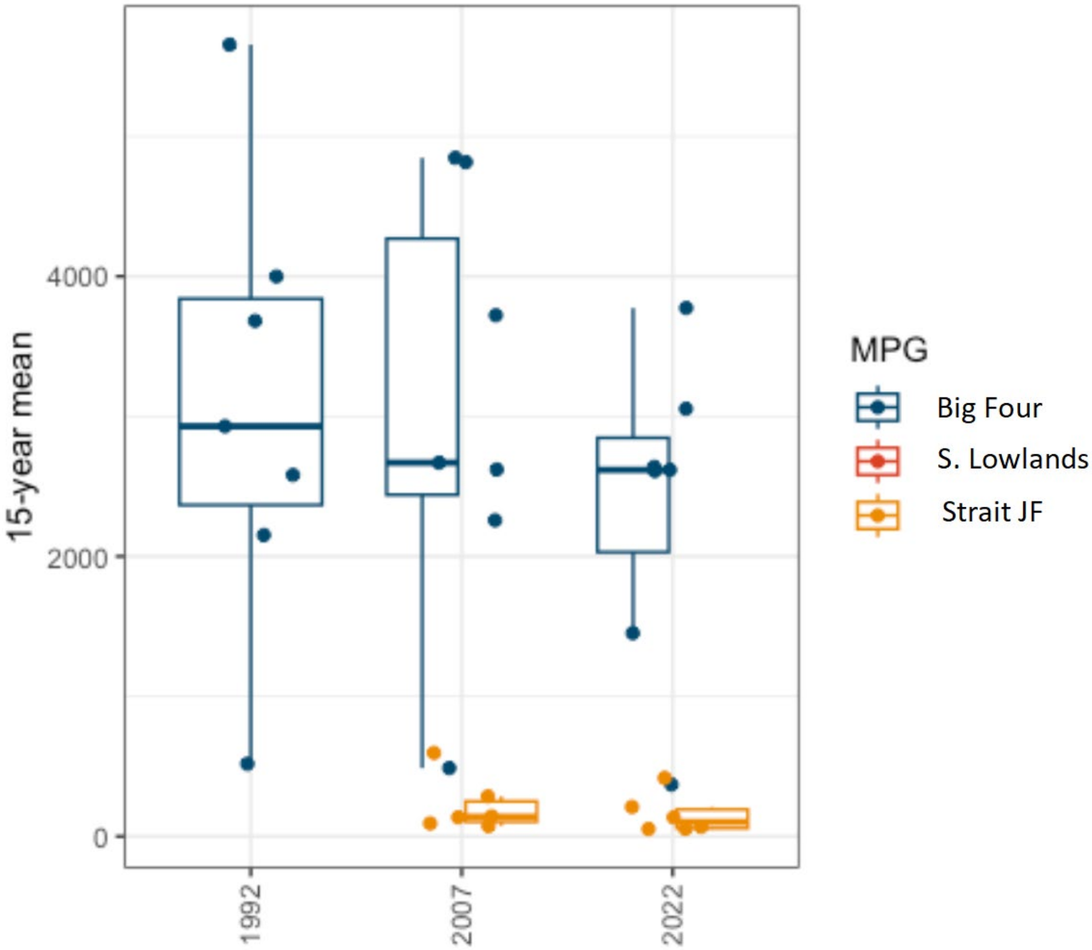



Figure 29. 15-year mean escapement estimated for winter-run stocks (total escapement after cut-off). Points show estimated mean for individual stocks for the 15-year period ending at the year in the x axis. Only 15-year windows where at least 2 years are in the first 5 years and 2 years are in the last five years are shown. The year on the x-axis is the end year of the 15-year period.

### Pre- and post-Busby Abundance Means

The mean of the estimated escapement (from DLM) for the pre- (1988-1993) and post-Busby (2018-2023) periods was also calculated (Figure 30). The decreases in the regional and overall DPS-wide abundances from those considered by the previous SRT in the 1990s and those considered by the current SRT indicate a degradation in status. Differences in mean escapement in the Strait of Juan de Fuca are likely biased by the limited number of populations included in the Busby time period, although this would not strongly affect the total abundance comparisons due to the small size of populations in the Strait.

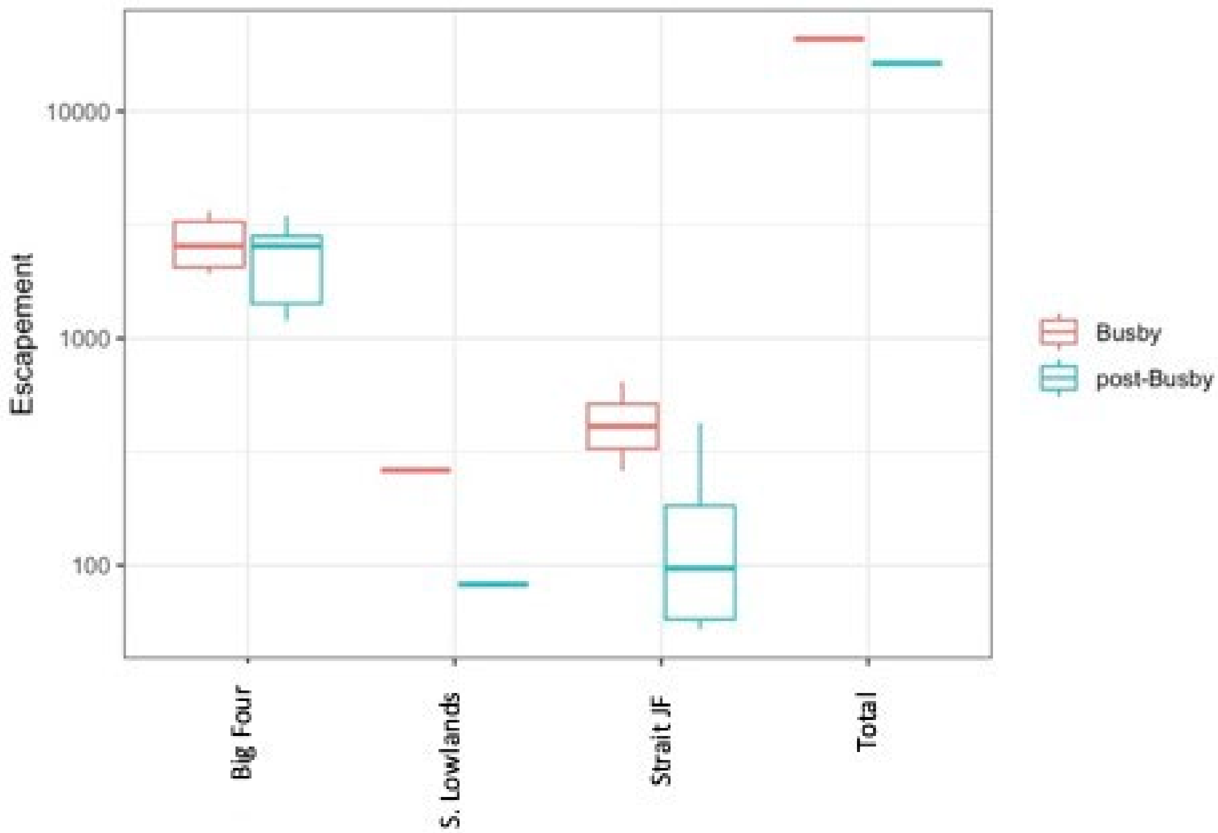



Figure 30. Mean escapement estimated for winter-run stocks (total escapement after 15 March cut-off) for the pre- (1989-1993) and post- (2018-2023) periods. Note that the y axis is on log10 scale

## 5-year geomeans

5-year geometric means were calculated using the observed escapement after the cutoff and using the estimated escapement from the DLM (Table 10, Table 11).

Table 11. 5-year geometric mean of winter-run stocks. Observed escapement after cutoff data are shown first and then in parentheses, the 5-year geometric mean of smoothed total spawners (from the DLM) is shown. Geometric mean was computed as the product of counts raised to the power 1/(number of values in band).

| Population | MPG | 1978-1982 | 1983-1987 | 1988-1992 | 1993-1997 |
|---|---|---|---|---|---|
| Moclips River | Southern Lowlands | | | 239 (245) | 343 (291) |
| Quinault River | Big Four | 4018 (3959) | 3734 (3977) | 3965 (3747) | 2887 (2757) |
| Queets River | Big Four | | 4820 (5267) | 5480 (5404) | 4003 (4029) |
| Hoh River | Big Four | 2613 (2694) | 3430 (3210) | 2569 (2650) | 2348 (2376) |
| Goodman Creek | Southern Lowlands | | | | |
| Quillayute-Bogachiel River | Big Four | 1613 (1725) | 2293 (2281) | 2105 (2097) | 1703 (1800) |
| Calawah River | Big Four | 1810 (1916) | 2842 (2803) | 2779 (2766) | 2862 (2911) |
| Sol Duc River | Big Four | 3523 (3490) | 3597 (3761) | 3577 (3761) | 4526 (4328) |
| Dickey River | Big Four | 527 (497) | 473 (481) | 379 (393) | 350 (385) |
| Hoko / Little River | Strait JF | | | 699 (693) | 526 (557) |
| Clallam River | Strait JF | | | | |
| Pysht / SF Pysht rivers | Strait JF | | | 271 (280) | 251 (254) |
| Deep Creek | Strait JF | | | | |
| West Twin River | Strait JF | | | | |
| East Twin River | Strait JF | | | | |
| Salt Creek and Tributaries | Strait JF | | | | |

Table 12: 5-year geometric mean of winter-run stocks. Observed escapement after cutoff data are shown first and then in parentheses, the 5-year geometric mean of smoothed total spawners (from the DLM) is shown. Geometric mean was computed as the product of counts raised to the power 1/(number of values in band).

| Population | MPG | 1998-2002 | 2003-2007 | 2008-2012 | 2013-2017 | 2018-2022 |
|---|---|---|---|---|---|---|
| Moclips River | Southern Lowlands | | | | | |
| Quinault River | Big Four | 2259 (2683) | 2716 (2673) | 2887 (2770) | 2625 (2508) | 2186 (2356) |
| Queets River | Big Four | 4111 (4532) | 5634 (5285) | 4613 (4531) | 3583 (3507) | 2931 (3140) |
| Hoh River | Big Four | 3088 (3032) | 2254 (2397) | 2677 (2569) | 2314 (2366) | 2735 (2840) |
| Goodman Creek | Southern Lowlands | 287 (283) | | | | 76 (81) |
| Quillayute-Bogachiel River | Big Four | 2957 (2776) | 1972 (1980) | 1710 (1664) | 1221 (1280) | 1166 (1191) |
| Calawah River | Big Four | 4798 (4590) | 3122 (3218) | 2732 (2729) | 2526 (2488) | 2551 (2728) |
| Sol Duc River | Big Four | 5696 (5678) | 3897 (3898) | 2980 (3016) | 2553 (2676) | 3483 (3458) |
| Dickey River | Big Four | 699 (628) | 344 (391) | 384 (344) | 268 (297) | 423 (418) |
| Hoko / Little River | Strait JF | 698 (688) | 494 (490) | 401 (415) | 344 (362) | 438 (420) |
| Clallam River | Strait JF | | 158 (153) | 105 (118) | 129 (128) | 146 (144) |
| Pysht / SF Pysht rivers | Strait JF | 351 (356) | 209 (211) | 160 (170) | 194 (192) | 237 (229) |
| Deep Creek | Strait JF | 162 (160) | 99 (104) | | 83 (83) | 99 (97) |
| West Twin River | Strait JF | 116 (109) | 55 (55) | 42 (46) | 43 (45) | 56 (52) |
| East Twin River | Strait JF | 85 (83) | 50 (48) | 35 (38) | 51 (48) | 54 (55) |
| Salt Creek and Tributaries | Strait JF | 171 (165) | 106 (105) | 84 (82) | 44 (53) | 66 (60) |

## Total Run Size and Estimated Harvest Mortality

The following plots use the “Harvest (Summary)” tab in the data provided by the co-managers. This includes an estimate of run size for natural origin (COPSWG 2023). From this, the total assumed natural origin escapement (escapement after the cutoff date) is subtracted to give an estimate of mortality. This is computed as “harvest/runsize” (Figure 31).

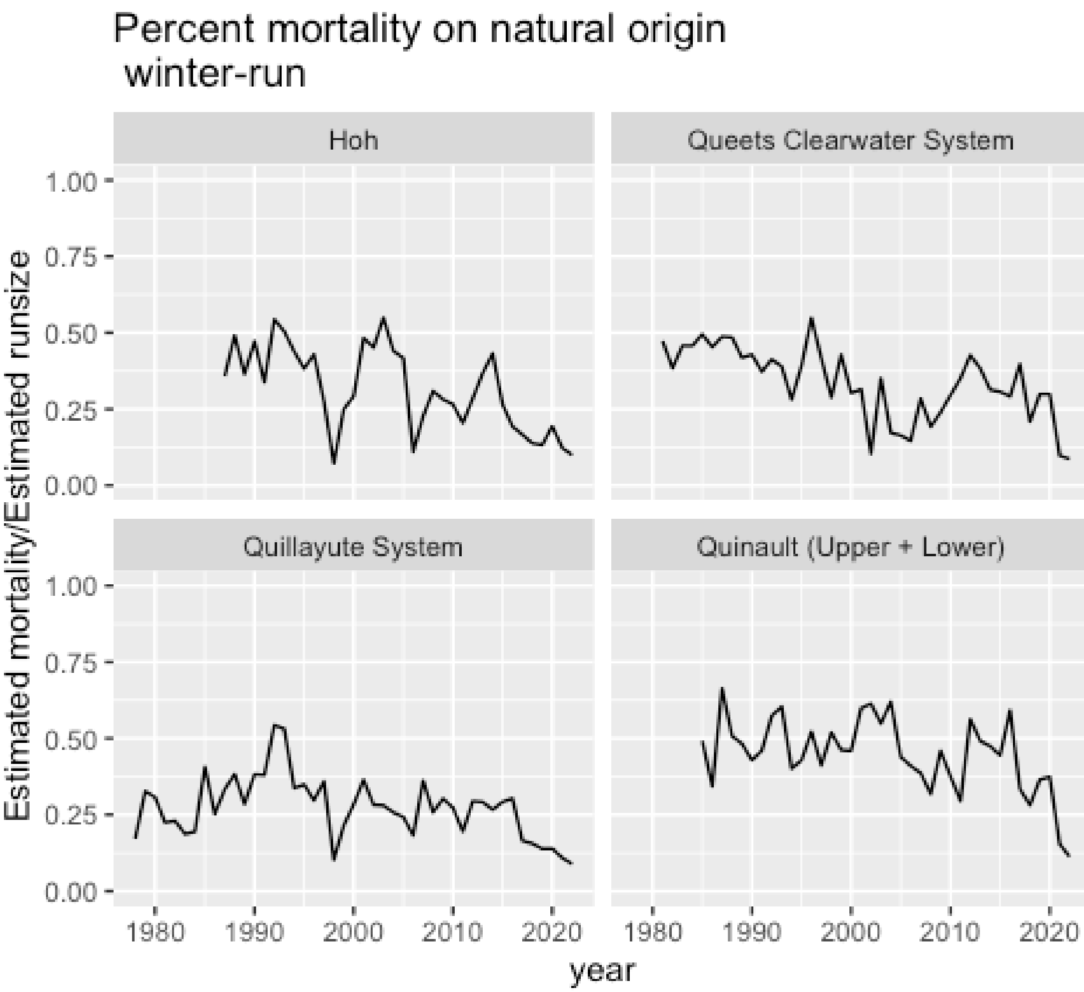



Figure 31. Harvest mortality of natural (escapement after March cutoff) winter-run steelhead reported by co-managers. This is harvest/runsize. Recreational hooking (catch and release) mortality is only included in the Hoh River data.

## Population Growth and Harvest in Strait Populations

The DLM above is a time-series model for escapement alone. We know additional information about the patterns of harvest for the Strait populations and can use that information to examine population trajectories as a function of harvest. Specifically, most populations along the strait experienced a cessation of harvest mortality at some point during the time-series (Table 12) and from the DLM we annual estimates of population change (specifically, population growth rate is equal to $\mu_i + \epsilon_{it}$). We can plot those estimates for each population through time highlighting the time harvest ceased (Figure 32) and as function of their harvest category (Figure 33), to look for obvious signatures of harvest on population growth.

Table 13. Year in which recreational harvest ceased for each population in the Strait of Juan de Fuca, except for the Hoko/Little River, where recreational catch is still allowed.

| Population | No Fishing After |
|---|---|
| East Twin River | 2003 |
| Salt Creek and Tributaries | 2005 |
| West Twin River | 2008 |
| Clallam River | 2017 |
| Deep Creek | 2018 |
| Pysht / SF Pysht rivers | 2019 |
| Hoko / Little River | NA |

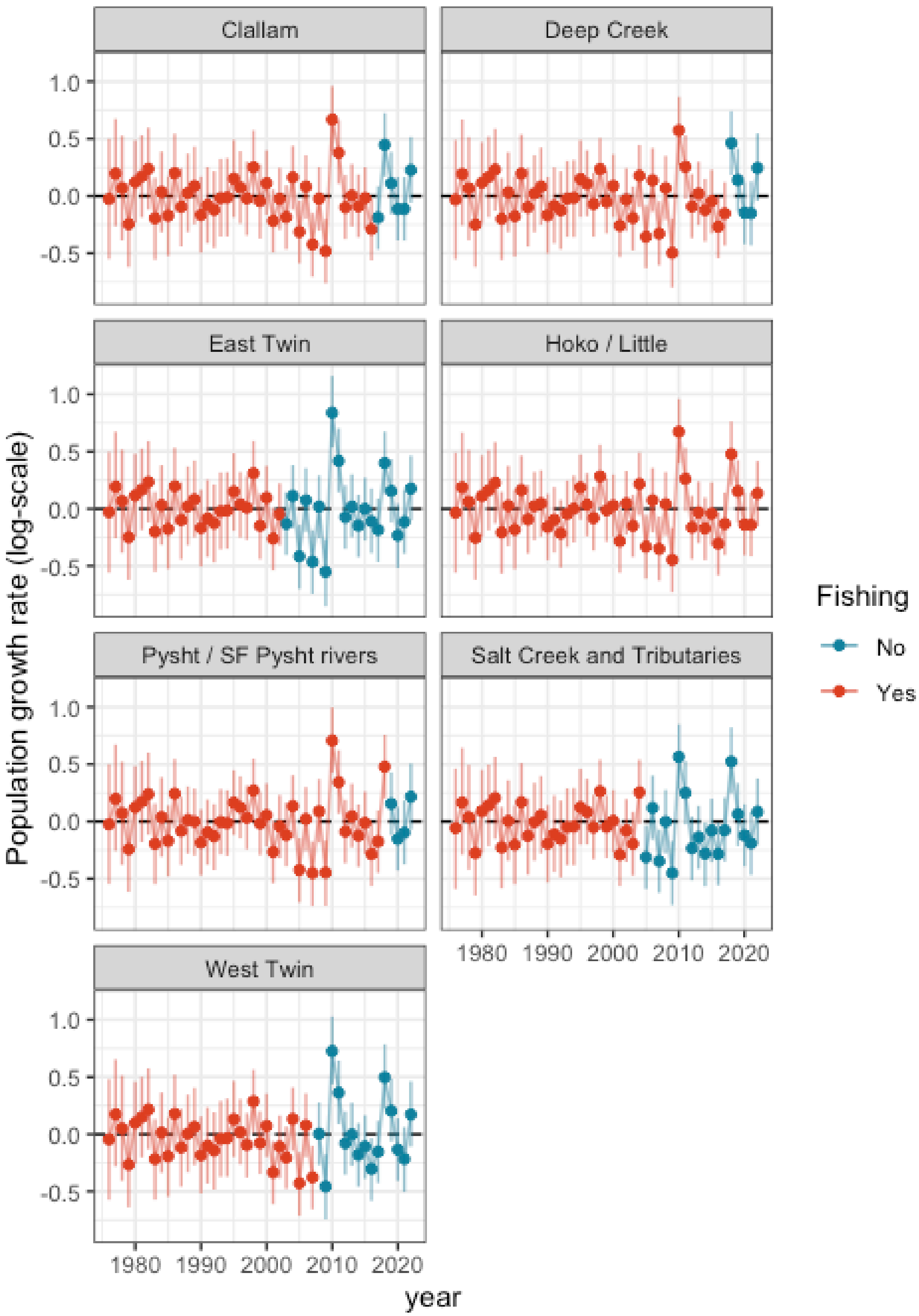



Figure 32. One-year estimates of population growth during period with and without harvest on strait populations of OP steelhead. Estimates are from the DLM output. Mean and 95% CI shown in vertical lines.

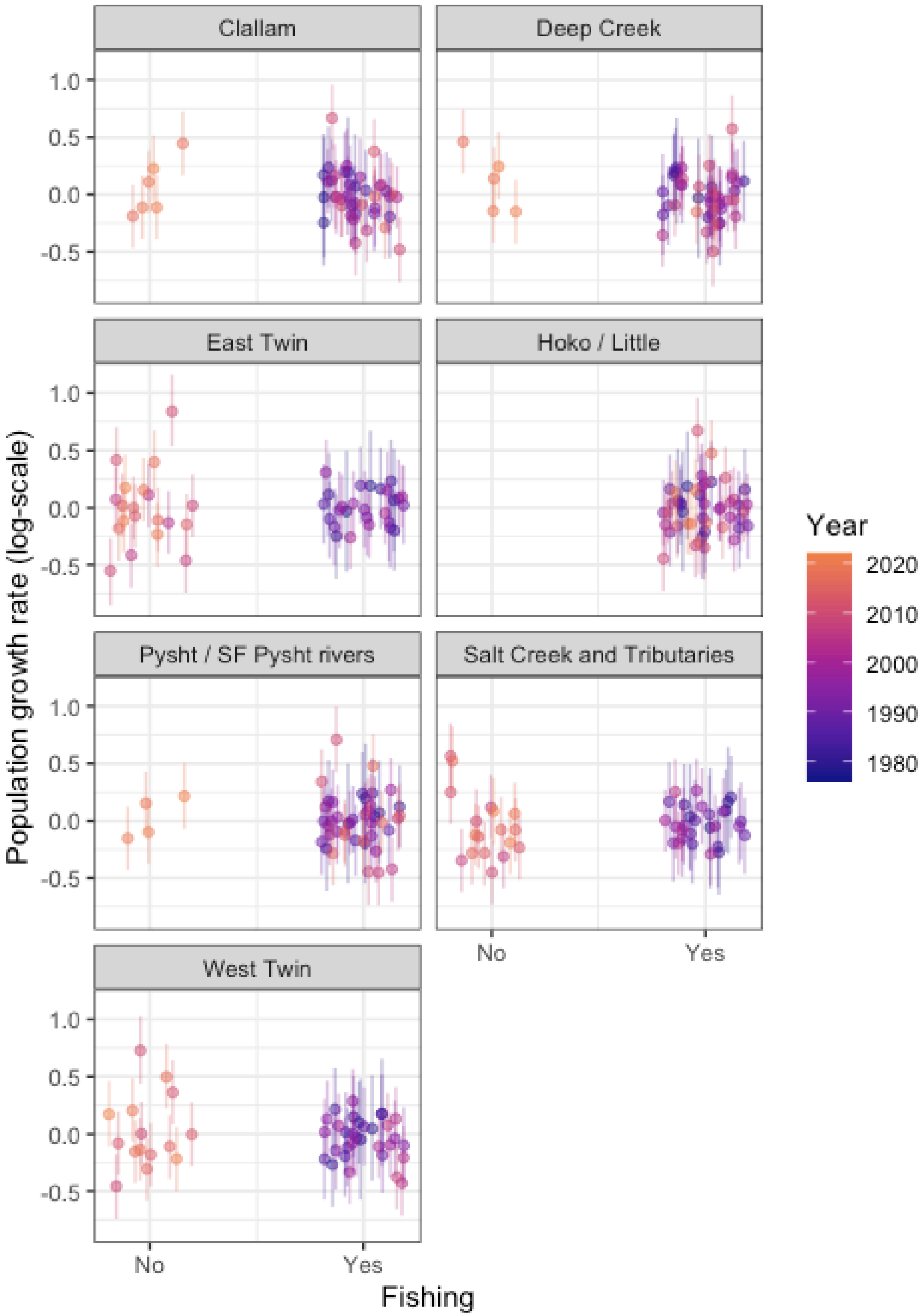



Figure 33. Population growth rates during periods with and without harvest on strait populations of OP steelhead. Estimates are from the DLM output. Mean and 95% CI shown in vertical lines.

# Population Growth and Harvest in Coastal Populations

## A simple lag-1 time series model for catches and escapement

Here is a joint time-series model for escapement and harvest. Let $Z_{it}$ be the observed escapement in numbers of steelhead of population $i$ in year $t$ and $C_{it}$ be the observed catch in numbers in all fisheries. Both $Z$ and $C$ are observed with uncertainty. We can construct a time-series model for the true but unobserved escapement, $X_{it}$, and total run size, $Y_{it}$, from these observations. We assume that the population dynamics can be approximated using a lag-1 time-series model in log-space. We let $F_{it}$ be the instananeous fishing mortality rate, and $\widehat{C_{it}}$ be the predicted catch, then

$$Zit \sim LogNormal\,(\log Yit - Fit - 0.5\sigma^2 R, \sigma^2 R) \tag{1}$$
$$\log Y\,t = \log Xt{-}1 + \mu + \epsilon t \tag{2}$$
$$\epsilon t \sim MVN\,(0, \Sigma Q) \tag{3}$$

and for catches,

$$Cit \sim Normal\,(C\hat{ı}t, \phi C\hat{ı}t) \tag{4}$$
$$C\hat{ı}t = Yit(1 - \exp(-Fit)) \tag{5}$$

Following standard notation, bold symbols indicate vectors. Note that unlike a standard $MARSS$ model, this model is non-linear in log-space and has two likelihood components, one for catch and one for escapement. We estimate the process error covariance $\mathbf{\Sigma}_Q$ with a single variance term $\sigma_Q^2$ and correlation among rivers $\theta$. For a four-population model, $\mathbf{\Sigma}_Q$, is $\sigma_Q^2$ on the diagonal and $\theta\sigma_Q^2$ on the off-diagonal entries. We assume a relatively small observation uncertainty for the catch ($\phi = 0.10$) corresponding to assuming a coefficient of variation of 10% on catch. Allowing catches to be uncertain differs from most uses of catch data in models for Pacific salmon and steelhead.

To improve model estimation, we model $F_{it}$ hierarchically: $F_{it} \sim N(\bar{F}, \tau_F)$. We use diffuse priors on all parameters.

$$input - priors - here. \tag{6}$$

We fit the above model to the harvest and escapement data for the 4 large OP rivers (Figure 34),

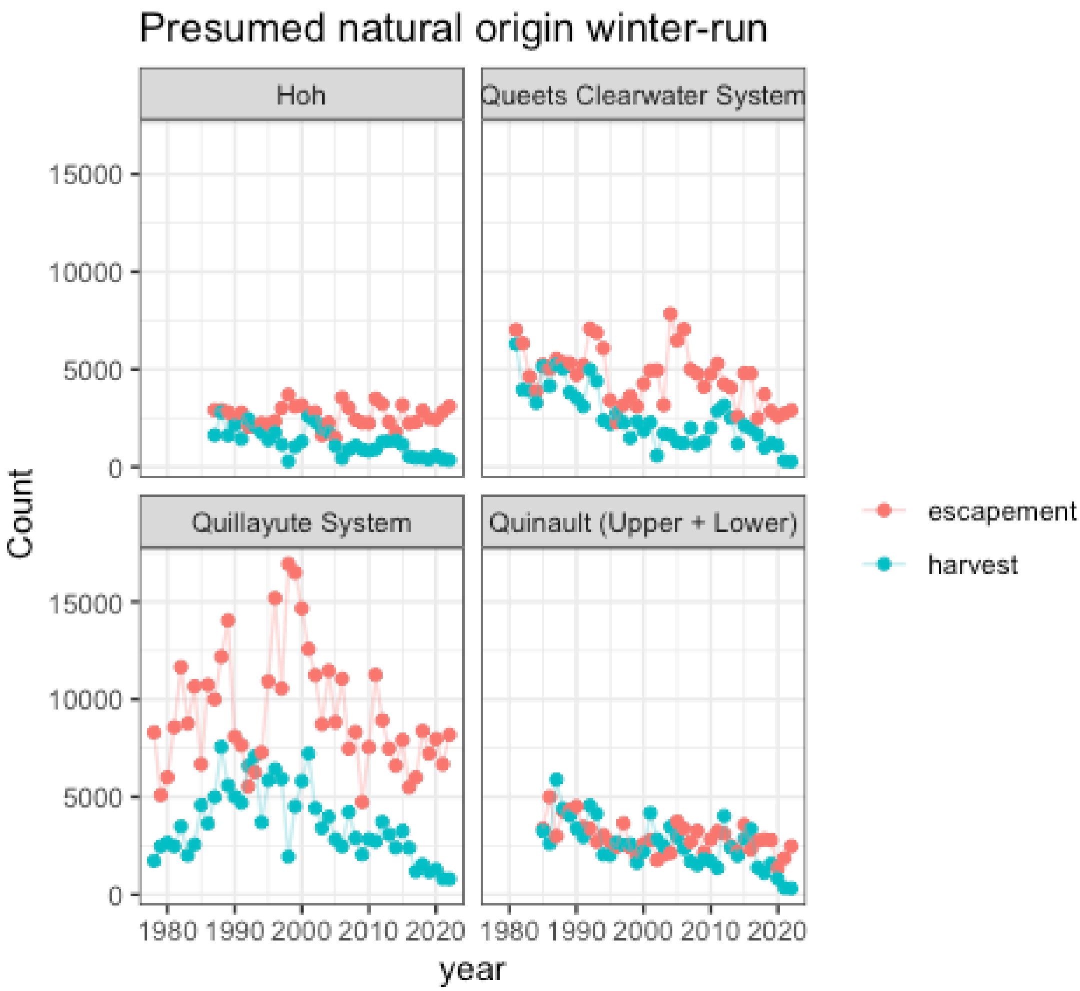



Figure 34. Raw data for escapement and harvest of natural (after March cutoff) winter-run steelhead reported by co-managers.

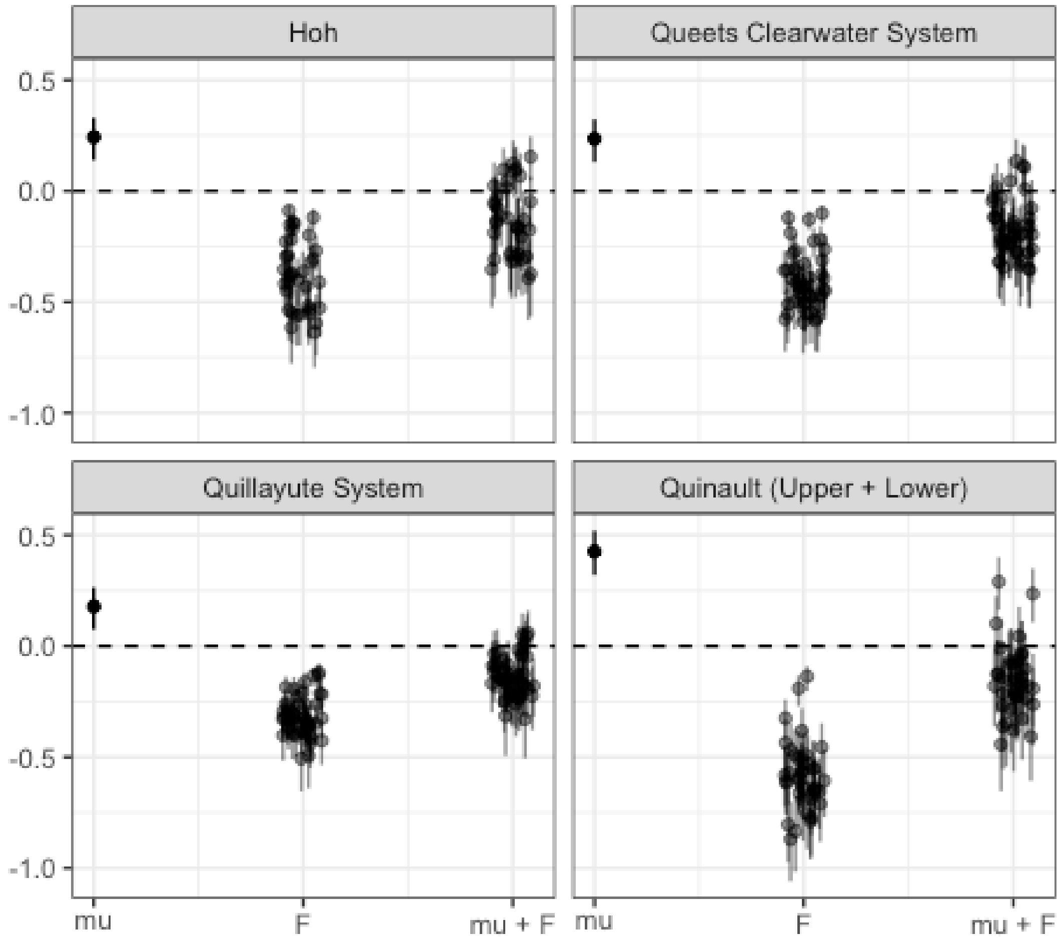



Figure 35. Estimated log-scale population growth rate (mu), estimated annual harvest mortality (F), and the net population growth rate (mu + F). For ‘F’ and ‘mu + F’, each point represents the estimated value in a particular year. For all parameters means and 95% CI shown.

The model fits and produces reasonable estimates of escapement, harvest, and total runsize (Figure 34). Model estimates for this model suggest that these populations largely have an intrinsic population growth substantially greater than zero (point estimates of $\mu_i > 0.15$ for all populations) (Figure 35). However, they are also subjected to substantial fisheries mortality and that in most years this fishing mortality is greater than intrinsic mortality (i.e. generally $\mu_i - F_{it} < 0$; which will result in declining populations (Figure 35). A small minority of years in each population were judged to have population growth greater than 0. Estimates of correlation among populations were positive and large, indicating that all four of these populations fluctuated in unison ($\theta = 0.83[0.62, 0.97]$ mean[95%CI]).

Note that this model is intermediate between the time-series model for escapement (see above) and a full integrated model that accounts for age-structure, greater than 1 year time-lags, density-dependance, and other important processes. However, this does partition out the density-independent component of fishing mortality relative to population growth and uses catch observations in a reasonable way.

**Summer-run Steelhead Population Harvest**

The petitioners utilized harvest data for summer-run steelhead as one measure of abundance. In general, summer-run steelhead were identified by the time of harvest May to October, although it is likely that winter-run kelts or early returning winter-run fish may be harvested during this time window. Further, the timing of river entry for summer-run steelhead overlaps with salmon gill-net harvest targeting summer coho salmon, sockeye salmon, and Chinook salmon; this bycatch data was not available and is not included in harvest estimates. Further, due to the prolonged period in freshwater, up to 10 months prior to spawning, there is an increased probability of incidental catch in the recreational fisheries. Houston and Contor (1984) reported limited sport and commercial catch of wild and hatchery summer steelhead in the Hoh, Quillayute, Queets, and Quinault rivers with generally fewer than 100 fish per year average. The average harvest of summer run fish has historically been a few hundred in most basins, except for the Quillayute Basin, where hatchery-origin summer run steelhead are released and combined hatchery and natural-origin fish catches are in the thousands of fish (Table 13). Recent recorded catches of unmarked summer-run steelhead have been very limited in recent years.

Table 14. Summary of sport and commercial catch of hatchery and wild summer steelhead among Olympic Peninsula DPS rivers.

| River | Sport Harvest (Mean annual, range, period of record) | | Commercial Harvest | Source |
|---|---|---|---|---|
| | Hatchery | Total* | Total* | |
| Quillayute River | 611<br>(27-1,974)<br>1988-2022 | | 179<br>(29-373)<br>1953-1957 | WDG 1984 |
| Hoh River | | 275<br>(38-711)<br>1962-1992 | 291<br>(23-954)<br>1975-1982 | WDG 1984 |
| Queets River | | 222<br>(21-516)<br>1962-1992 | 104<br>(43-171)<br>1975-1982 | WDG 1984 |
| Quinault River | | 132<br>(0 to 452)<br>1962-1992 | | |

* May include hatchery and wild

### Historical abundance

The SRT was not able to find historical (pre-contact) DPS-wide estimates of steelhead abundance for the OP Steelhead DPS. Busby et al. (1996) cited an estimate of 60,000 by Light (1987); however, this estimate was for the 1980s and included hatchery-origin steelhead. Using harvest data from the major coastal tributaries extending back to the first half of the 20th Century, abundance estimates by McMillan et al. 2022 and McMillan et al. 2023 for the major coastal tributaries (Quillayute, Hoh, Queets, and Quinault rivers) suggest a historical cumulative run size of 67,436 winter steelhead and few thousand summer steelhead. This compares with a current (2018-2022) run size estimate of 18,824 winter-run steelhead (WDFW data) for the same basins. In the absence of direct harvest effort and bycatch estimates, there is conservable uncertainty in the expansions of harvest to run size.

## Diversity

### Life History Traits

Life history trait diversity within and among steelhead populations in a DPS allows for the exploitation of diverse habitats and provides a buffer for annual environmental variation and the ability to adapt to long-term climatic changes. While variation in life history traits on a larger geographic scale was reviewed to confirm the DPS configuration, changes in that variation with the DPS were assessed as potential indicators of anthropogenic selection or selection due to changes in the environment.

**Run timing and harvest**

The Petition identified a shift in return and spawn timing due to the loss of the early returning portion of the native winter steelhead run as a factor affecting diversity; this is distinct from the presence of non-native early returning winter steelhead (initially Chambers Creek Hatchery stock) released into multiple basin in the Olympic Peninsula Steelhead DPS. Several studies (McLachlan 1994, McMillan and Gayeski 2004, Cram et al. 2018) have identified this contraction in range of return timing as a concern. Specifically, intensive harvest on the early returning winter steelhead (November through January) which was largely directed at hatchery-origin winter steelhead also intercepted natural-origin native winter steelhead. Hatchery broodstocks for the majority of winter-run steelhead programs in the Olympic Peninsula DPS were established using Chambers Creek Hatchery origin winter steelhead, because it was originally thought that the earlier return timing of the Chambers Creek fish would allow a selective harvest and limit introgression into local populations.

This compression is a loss of diversity in run-timing. Run timing variability likely confers a long-term bet-hedging against environmental effects related to stream accessibility or major storm events. Such loss is expected to reduce population resilience (ability to recover from disturbances) via a portfolio effect, potentially increasing extinction risk (Greene et al. 2010). It may also increase extinction risk by reducing population productivity (mean cohort replacement rate); because the compression stemmed mostly from a loss of early spawners (Nov – Jan), the replacement rate may suffer if conditions for egg and fry survival are higher in mid-winter and/or the earlier spawning allows fry to reach larger sizes before facing the challenging conditions of the summer low-flow high temperature season. If the mean replacement rate drops below one, it produces a downward trend in abundance that eventually leads to extinction (Essington et al., 2006).

McLachlan (1994) attempted to estimate the historical range of winter steelhead return timing in the Quillayute River. He found a contraction in run timing with a decrease in the proportion of the run return before 1 January, from 35% of the run to 20% of the run. Meyer (1994) expressed concern over the harvest related loss of early returning winter run steelhead, and that their upper basin spawning areas would be underseeded. Similarly, McMillan et al. (2022) found an 18% decline in the fraction of natural winter steelhead run in the Quillayute River returned before 1 January. Further, the *q25*[12] of the natural run was delayed by 33 days. McMillan et al. (2022) also found that the early portion of the winter steelhead run in the Hoh River passing before 1 January had declined by 43%, with the *q25* exhibiting a delay of 71 days, compared to historical.

One hypothesis proposed by McMillan et al. (2022) for the compression is the direct and indirect effects of the existing steelhead fisheries in the Quillayute, Hoh, and Queets river systems. The fisheries mainly target hatchery steelhead, which overwhelmingly return from the ocean early to mid-winter (Nov-Jan), but the fishery harvests wild steelhead as well, and the petitioners argue that declines in early wild steelhead coincided with the introduction of these early-returning hatchery-origin steelhead. The compression could come from two mutually-reinforcing processes: 1) Interbreeding of early-returning wild steelhead with the hatchery-origin fish, potentially reducing their fitness relative to late-returning wild steelhead; and 2) higher exploitation (harvest rate) of early-returning wild fish relative to late-returning wild fish, which would also reduce their relative fitness. If within-season run timing has a heritable genetic component, these processes would tend to drive evolution of a compressed run-timing favoring the late returning fish (Quinn et al. 2009). This in turn could increase extinction risk as described above. In addition to the petitioners discussing the loss of the early-returning winter-run steelhead, they emphasized the decline of summer-run steelhead populations.

The SRT discussed the evidence for harvest driven changes in native winter-run steelhead run timing and the potential effect of these changes on overall DPS viability. Changes in winter-run steelhead return timing were assessed by the SRT using catch records prior to the release of non-native early returning "Chambers Creek" winter steelhead in the DPS. Information compiled during the 1940s-1960, indicate that there was significant harvest of steelhead in November and December in the Quinault River (Figure 36) and Queets River (Moore 1960). Later studies, in the Queets and Quinault Rivers indicated that there was a significant overlap in run timing between hatchery and natural steelhead, with little difference in the timing of the start of the run (Figure 37, Figure 38).

[12] *q25*: the Julian date at which 25% of the run has entered the river.

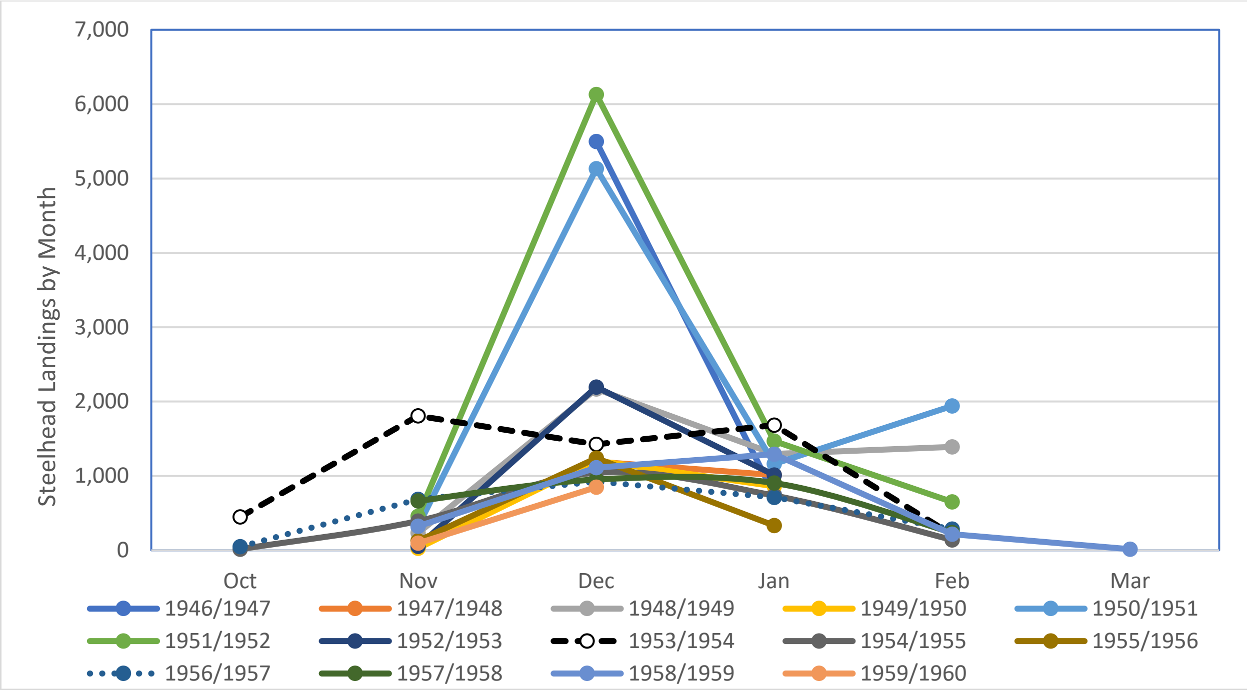



Figure 36. Quinault River steelhead gill and set net harvest from 1946-1960. (From Moore 1960).

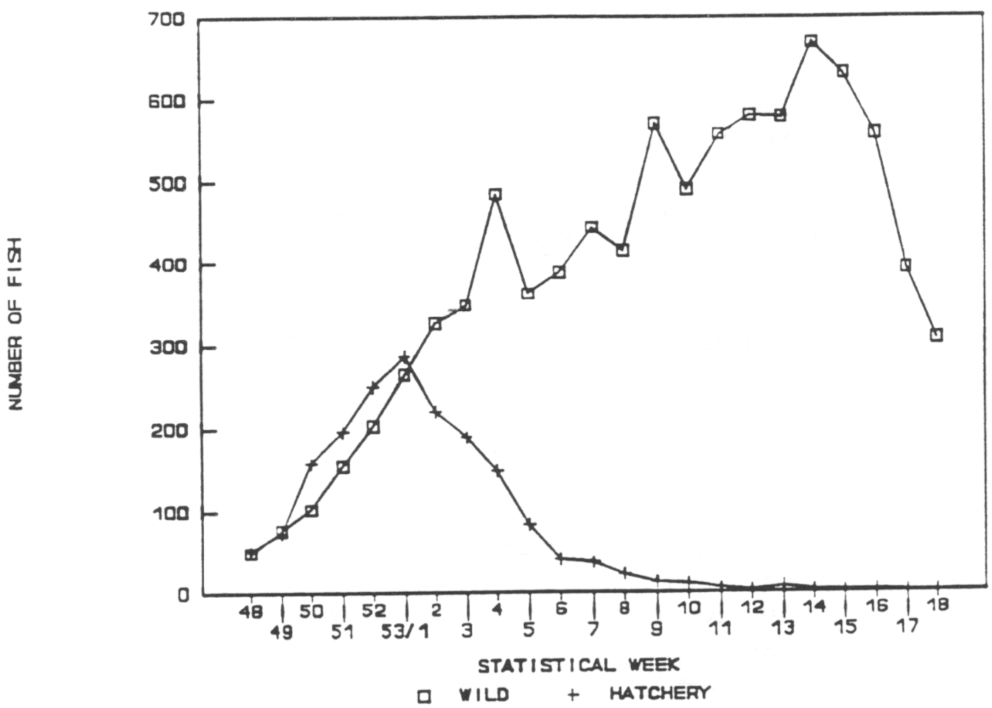



Figure 37. Predicted run timing and magnitude of hatchery and natural winter-run steelhead entering the Queets River during 1993-1994. (From Figure 2.2, page 4, QFD and WDW 1993).

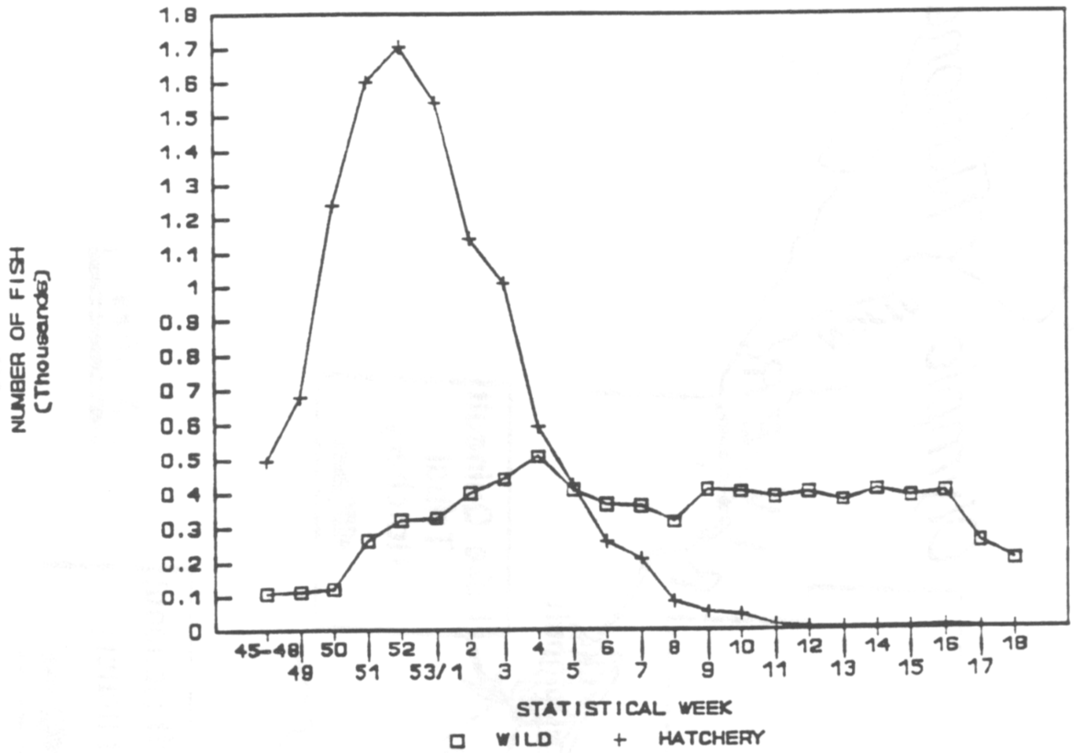



Figure 38. Predicted run timing and magnitude of hatchery and natural winter-run steelhead entering the Quinault River during 1993-1994. (From Figure 2.2, page 4, QFD and WDW 1993).

Alternatively, in the Hoh River it was observed that hatchery-origin early-returning (December to February) winter run steelhead did have a run time that was distinct from the later returning (March and April) natural origin winter run steelhead. However, it was suggested that the underlying reason for this may have been due to management action rather than historical run timing.

> The late timing of the "wild" run may itself be an artifact of the generally early run timing of the hatchery releases, and not an inherent characteristic of the native stock (Houston and Contor 1984). The relatively high harvest rates corresponding to the harvest of a hatchery run may have resulted in over-harvest of the early component of the "wild" run and a shift of the run to a later timing pattern. …Artificial selection against early "wild" returns on the Hoh may have occurred as a result of higher early fishing effort in some years to harvest the early hatchery segment of the run (Hiss et al. 1986, p3).

Meyer (1994) similarly expressed concern that there had been "a shift in abundance and timing of 'wild' stocks toward the later part of the return timing and that this shift is due primarily to the heavy harvest on early-returning hatchery fish." At the time of the initial coastwide status review, WDFW asserted that there was temporal separation between hatchery and naturally-produced populations (Busby et al. 1996).

The SRT finds this argument plausible and requested data from the co-managers that would allow us to further evaluate whether this process is ongoing since the last status review by Busby et al. (1996). McMillan et al. (2022) reached their conclusions by comparing mean conditions for a mid-century period (1948-1960) versus the recent past (1980-2017), but fisheries managers have made changes to hatchery operations in the last two decades to reduce their impacts on wild fish, which ideally would have stalled the ongoing compression of run timing. Genetic data, summarized elsewhere in this status review, suggests the strategy to segregate hatchery stocks has tended to prevent interbreeding with wild stocks (although these genetic data are 10 to 20 years out of date and thus represent a weak test). However, the catch of natural-origin steelhead in the early-winter fishery is appreciable, consistent with the exploitation hypothesis. Meanwhile, the ongoing declines of wild run size over the past three decades, summarized elsewhere in this review clearly indicate that the mean cohort replacement rate is less than one and the populations are trending in a downward direction.

**SRT assessment of winter-run run timing changes**

To test whether the distribution of run-timing of natural-origin steelhead is continuing to compress, and whether harvest is plausibly one driving force, we would need weekly or monthly catch and weekly or monthly run size for each year over a sufficient time period (a few decades), to estimate inter-annual trends in run timing and catch rate. Alternatively, to formally test the hypothesis that run-timing is changing, we could use generalized additive regression models (Wood 2017) to estimate an interaction effect between year and week on number of migrant spawners. This is similar in spirit to standard ANOVA or linear regression, except the effects are estimated as smooth spline curves rather than categories (ANOVA) or straight lines (linear regression), and so makes fewer restrictive assumptions about the shape of interannual or within-season trends. If $s$(X) is a fitted spline curve for predictor X, and $re$(X) is a random effect, the regression model for run-timing has the form:

Weekly spawners = *mean* + *re*(year) + *re*(week) + *s*(year) + *s*(week) + *s*(year, week)

where *mean* is the mean weekly run size across the entire dataset, and the other terms describe random or systematic departures from the mean. The last term is a 2D surface with two predictors, and estimates systematic departure from the predicted main effects *s*(year) + *s*(week). If the run season is not systematically changing over time, there is no interaction effect, and *s*(year, week) = 0, which can be formally tested as a hypothesis. If rejected, inspection of the surface of *s*(year, week) would then reveal the pattern of change—compression, expansion, or shift. Likewise, a similar regression with harvest rate (and a logistic link function) would allow us to use weekly catch and run-size data to test whether exploitation is setting up a selection gradient against early run timing of wild fish.

Datasets with sufficient granularity to test the above compression hypothesis for wild steelhead were not available:

- The petitioners provided daily tribal harvest data (Petitioners Tribal Harvest Data), but the dataset does not distinguish hatchery fish from wild fish.
- The co-managers provided harvest data for wild steelhead (see Harvest Summary tab), but only in summary form, which lumps by year so monthly or weekly totals are not available.

- The comanagers provided weekly recreational harvest data for wild steelhead broken down by year and month (Summer Steelhead Recreational Catch data), but only for summer steelhead (June through October); and includes only the recreational catch-and-release portion of the fishery rather than the harvested.

The McMillan et al. (2022) paper cited by the petitioners has an associated public GitHub site maintained by one of the authors (https://github.com/MartinLiermann/historicalOPsteelhead), with relevant data for the Queets, Hoh, and Quillayute River systems for the period 1980 to 2017 (only 1997-2015 for the Hoh). The site has tribal and recreational catch data at a suitable granularity for the hypothesis (week × year), but not run-size data. Catch-per-unit-effort—from which run size might be reasonably inferred—is also available, but only summarized as weekly averages over many years, so is not useful for testing for change in run timing across years.

For this analysis we relied on the datasets from the public GitHub site associated with a peer-reviewed paper (McMillan et al. 2022) within a publication of the American Fisheries Society (*North American Journal of Fisheries Management*) (given that no other weekly, monthly specified data with wild and hatchery separated was provided to us). Unfortunately, these datasets limit us to only examining changes in the seasonal timing of catch rather than migrant abundance (escapement + catch) or harvest rate (catch / (escapement + catch)). Catch is the outcome of a complex process: Weekly catch tracks weekly run size to some degree, but also reflects weekly fishing effort, as well as environmental conditions conducive to catching fish. In the past two decades, annual run size has declined in these three river systems, and this is another factor affecting the seasonality of catch in addition to any changes in seasonality of the run itself.

Below, we apply the regression approach, developed above for weekly spawners, to weekly catch instead, to get as best estimate of how things have changed over the past three or four decades. We focus on the commercial catch, because unlike the recreational fishery which has undergone a shift to catch-and-release in recent decades[13], this fishery consistently removes hundreds to thousands of natural-origin fish from the population. Because fishery activities target both the hatchery and natural-origin steelhead as a group, we focus our analysis on two questions: 1) changes in total catch, and 2) changes in proportion of wild steelhead in the catch. All analyses were conducted in the statistical package R (R Development Core Team 2021) using the package mgcv (Wood 2017) to fit GAMs and the package gratia (Simpson 2022) to check model assumptions. Catch models used the negative binomial family with an identity link function; proportion wild used a beta family with a complementary log-log link.

[13] Recreational harvest of natural-origin fish did not decrease until the early 2000s, and catch and release rules for unmarked steelhead have only been implemented in the last few years. Analysis of recreational catch is more complex due to the temporal and geographic distribution of effort, but also likely contributed to the loss of early returning fish in the past.

**Wild Steelhead Catch in the Quillayute River**

For both total catch and proportion natural-origin steelhead, the main effects of year and week, and their interaction effects, all had extremely high statistical significance ($p << 10^{-10}$), indicating both changes in total catch across years, changes in the distribution of the catch within a year, and changes in the proportion of natural-origin fish in each weekly catch. Figure 39 shows the estimated spline curves for all these effects, which illustrate the pattern and magnitude of the various associations. In these and the other figures, a partial effect is the effect of one regression term while holding the other regression terms constant at their mean value. These include the random effects of year and week within year, which are normally distributed noise centered on zero, so the spline curves represent nonrandom pattern with this noise removed.

In the Quillayute, the total catch has steadily declined since 1980, while the proportion of wild fish in the catch has steadily increased, to over 70% of the catch in the latest years of record, and over half the catch since 1990 (Figure 39, left-hand panels). During this period the catch of hatchery steelhead was declining about twice as fast as for wild steelhead, but from a total catch about twice as high initially.

Seasonally, on average the catch peaked in December (middle top panel; week 0 = last week in December) but shows a smaller bump in March corresponding to the wild run. However, wild steelhead dominate the catch well before this bump, comprising over half the weekly catch starting in mid-January (middle panel bottom) and over 80% of catch by early February (week).

In the interaction plot for total catch (top right panel), red shows catch lower than expected given the main effects, while blue shows higher than expected. As the total catch has declined over the years, disproportionately more has occurred later in the season (blue after week 3 vs red before week 3). The lower-right interaction plot shows that even as total catch has disproportionately shifted away from this mid-winter period, the proportion of wild fish has increased in it (blue patch for weeks -3 to 3).

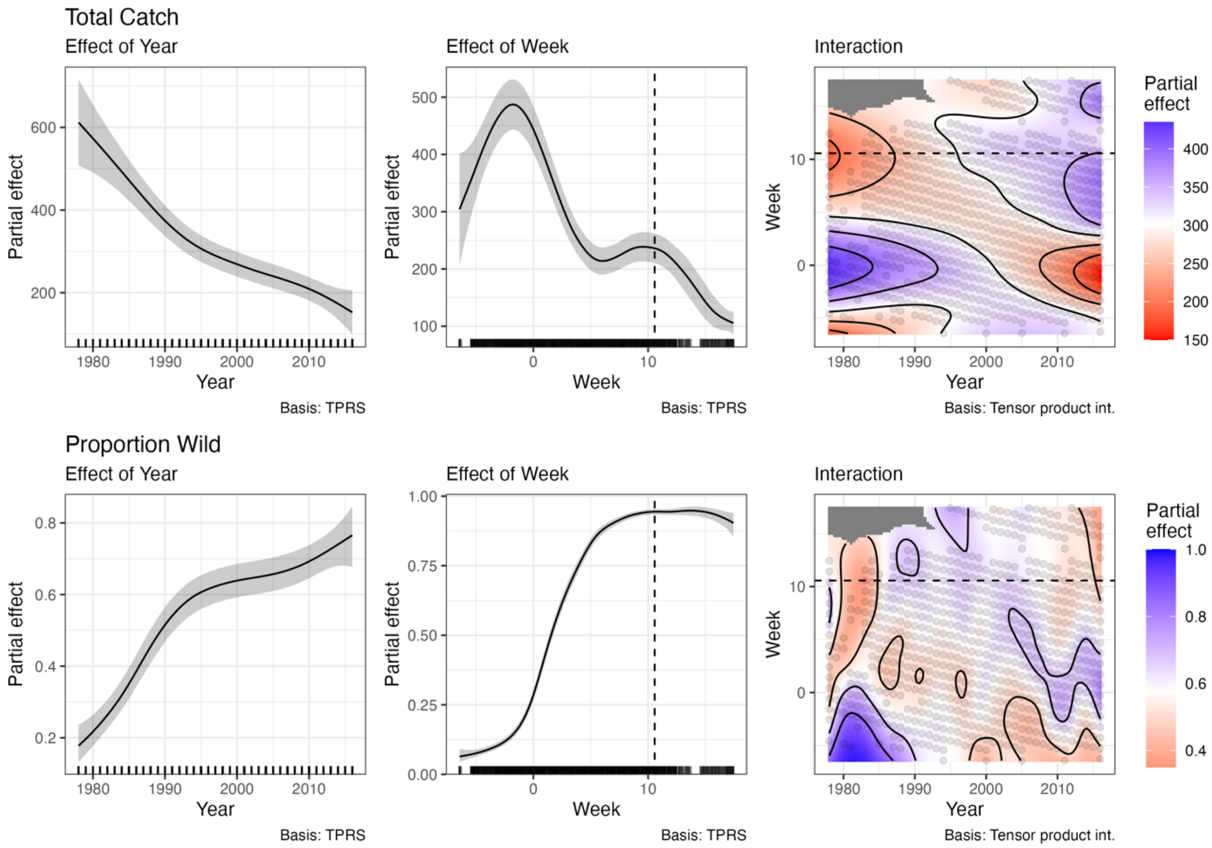



Figure 39. Quillayute River steelhead catch from the gillnet fishery, decomposed into long-term and seasonal trends using generalized additive regression models. Top row shows patterns of total catch, bottom row shows proportion of wild steelhead in total catch. Vertical dashed line marks the week of March 15.

Overall, this suggests the hatchery-origin steelhead component of the fishery is diminishing, and the natural-origin component is becoming a higher proportion of the overall catch, with higher harvest of wild steelhead moving earlier and earlier into the season.

**Wild Steelhead Catch in the Hoh River**

In the Hoh River (Figure 40), the various main and interaction effects had high statistical significance (all $p < 0.0005$, some much lower). The overall pattern is similar to the Quillayute results, but less extreme: Total catch is declining, while proportion wild is increasing, though less steadily than in the Quillayute (Figure 2, left panels). As in the Quillayute, wild steelhead catch is larger than hatchery catch by week 3 (middle bottom panel) and dominates (>75%) by week 5 in early February. By the end of the period of record, the proportion of wild steelhead in the catch is disproportionately increasing in weeks 0 through 5, beyond what is expected from the main effects (blue patch in lower right panel).

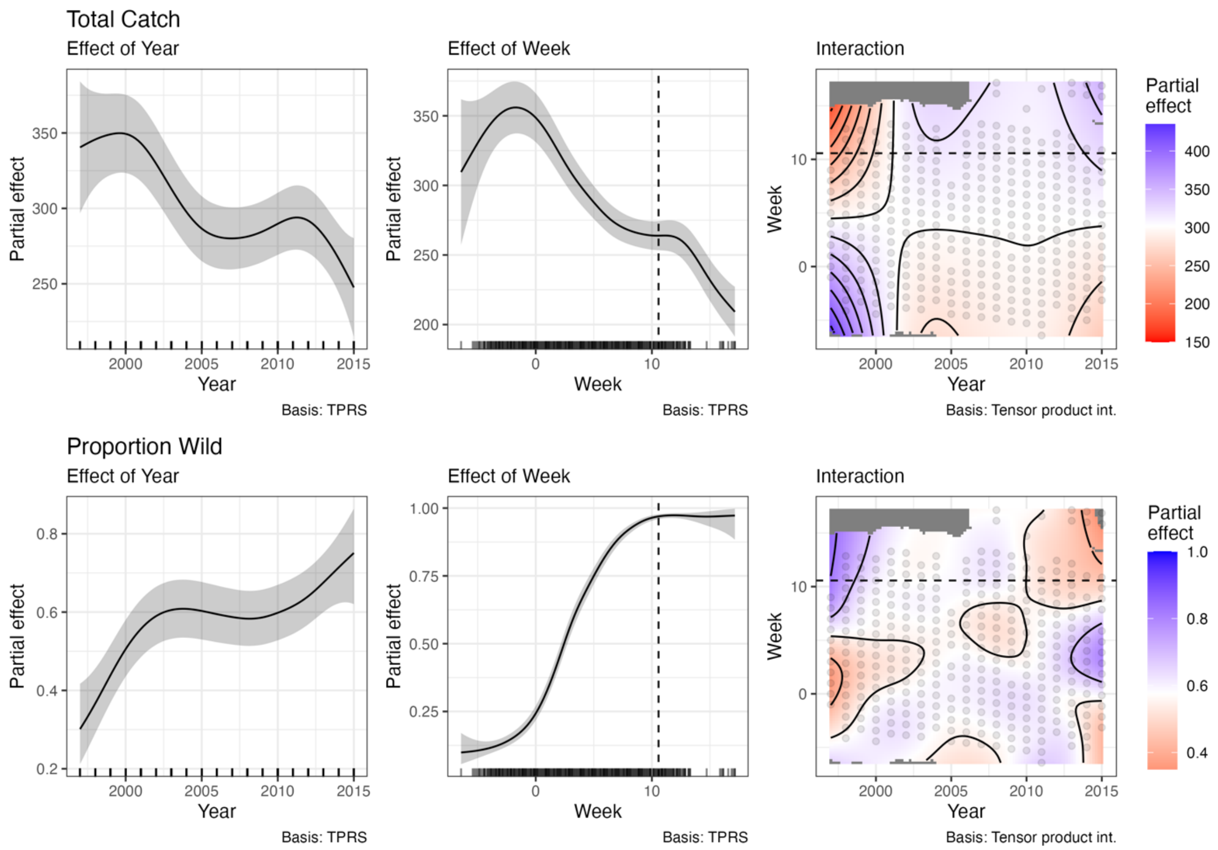



Figure 40. Hoh River steelhead catch from the gillnet fishery, decomposed into long-term and seasonal trends using generalized additive regression models. Top row shows patterns of total catch, bottom row shows proportion of wild steelhead in total catch. Vertical dashed line marks the week of March 15.

**Wild Steelhead Catch in the Queets River**

In the Queets system, both total catch and proportion wild had extremely high statistical significance for all main effects and all interaction effects ($p \ll 10^{-10}$). The various partial effects for the Queets (Figure 41) show similar long-term decline of total catch (upper left panel), but a significantly more complicated pattern for the proportion of wild steelhead in the annual catch (lower left panel). Despite these fluctuations in the proportion of wild steelhead in the gillnet fishery, the proportion has been above half for most of the period of record. Even as total catch, flat for about a decade after the turn of the millennium, started declining slightly after about 2012 (top left panel), the proportion of wild fish in the catch increased (bottom left panel), indicating a diminished role for hatchery fish in the fishery.

As in the other systems, seasonally the catch becomes dominated (>50%) by wild steelhead around week 3 on average (middle bottom panel), and then rises to >75% by around week 6 in mid-February. The interaction plot for proportion wild (bottom right panel) shows that during the recent increase in the proportion wild mentioned above, most of this wild catch came disproportionately from late to mid to early winter as the years progressed from 2005 to 2017 (large blue patch in bottom right panel).

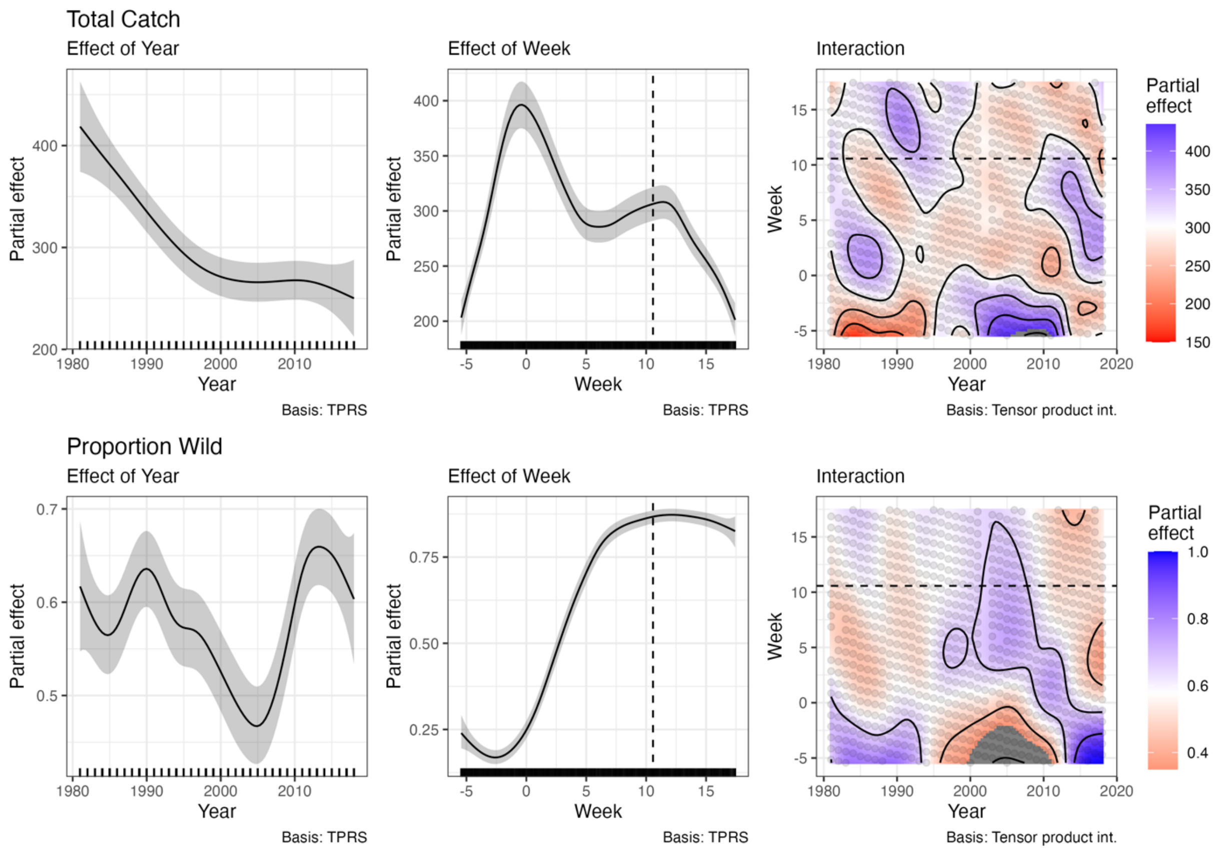



Figure 41. Queets River steelhead catch from the gillnet fishery, decomposed into long-term and seasonal trends using generalized additive regression models.

In all three river systems the wild catch is both declining and flattening out seasonally over time. There is also some change in seasonality of wild catch, shifting disproportionately earlier for the gillnet fishery—becoming disproportionately common in January for the Quillayute and Hoh Rivers, and even earlier in the Queets River.

## Hatchery Operations in the Olympic Peninsula Steelhead DPS

Hatchery operations, especially those utilizing non-native broodstocks, could introduce maladapted life -history traits through introgression . Non-native broodstocks are presumed to be more adapted to the ecology of their watershed of origin, and therefore express life history traits that are not necessarily adapted to the watershed that they were transferred to. In addition, both non-native and native-origin broodstocks can be subjected to directed and inadvertent selection (domestication selection) that can alter major life history traits and reduce the degree of local adaptation (Gow et al. 2011, Hutchings 2014).

There have been a number of studies that report on the deleterious effects on native salmonids from the release of non-native salmonids (e.g., Reisenbichler 1983, Tatara and Berejikian 2012), as well as the short-term and long-term effects of hatchery rearing on reproductive fitness (Naish et al. 2007, Araki et al. 2008; Ford et al. 2016). The relatively long-term nature of steelhead rearing (at least one year in the hatchery) may expose steelhead to stronger domestication selection effects than other salmonids.

Of the nine State, Tribal, and Federal Hatcheries in this DPS (Figure 42), the majority (seven) operate segregated hatchery programs that release non-native fish predominantly of Chambers Creek origin early-winter run or Skamania Hatchery origin early summer-run origin (Table 14). The hatchery propagation and release of winter and summer-run steelhead in this DPS has remained relatively constant since 1980, with the majority of releases being winter-run steelhead smolts (Figure 43, Figure 44, Figure 45, Figure 46). For more details on hatchery releases see Appendix B. Hatchery programs were reviewed by the Hatchery Science Research Group (HSRG) in 2004. Broodstocks for these hatcheries have been maintained on site for a number of generations and some integration with native populations has likely occurred. The two integrated hatchery programs, the Quinault Lake and Salmon River facilities, maintain broodstocks founded by Quinault Lake winter run steelhead, which are of "unknown origin" (Marston and Huff, 2022)[14]. In the Quillayute Basin, a native late winter-run steelhead program (from hook and line caught broodstock) at Snider Creek was operated from 1998 to 2021 with recent releases of 30,000 smolts annually, but his program has been terminated (Marston and Huff 2022). All of the currently operated hatchery broodstock programs appear to have either been founded by out-of-DPS stocks, or have been influenced by transfers of out-of-DPS stocks.

In general, releases of hatchery-reared steelhead have become more centralized since 1980, with the majority of releases being in WRIAs 20 and 21, areas with the largest basins. Beginning in the early 1980s, releases were adipose clipped, to allow for selective harvest management in the recreational fishery, with the exception of tribal releases in the Queets and Quinault River basins, where currently only a small percentage of the hatchery production is marked. Later, the Hatchery Science Review Group specifically recommended adipose clipping all hatchery production in these programs (HSRG 2004), but the overwhelming majority of hatchery-origin steelhead released into the Queets and Quinault basins are still unclipped. In 2014, there were 11 hatchery programs in the OP DPS, which annually released 1,072,781 smolts (2009-2013), of which 61.1% were released off station (Cram et al. 2018). For BY 2018-2020, 1,105,855 juveniles (> 5 grams) were released in the OP DPS, predominantly winter-run and primarily on station releases (RMIS Database 2023). Limiting most hatchery production to onside releases, a major change in hatchery operations since Busby et al. (1996), has likely led to a reduction in hatchery-origin steelhead straying onto natural spawning grounds.

[14] It appears from the genetics studies that early-returning Chambers Creek Hatchery fish have been incorporated into the Quinault Lake Hatchery and Salmon Creek Hatchery broodstocks. Quinault Lake Hatchery broodstock have been transferred to the Salmon Creek Hatchery (see-Hatchery Broodstocks).

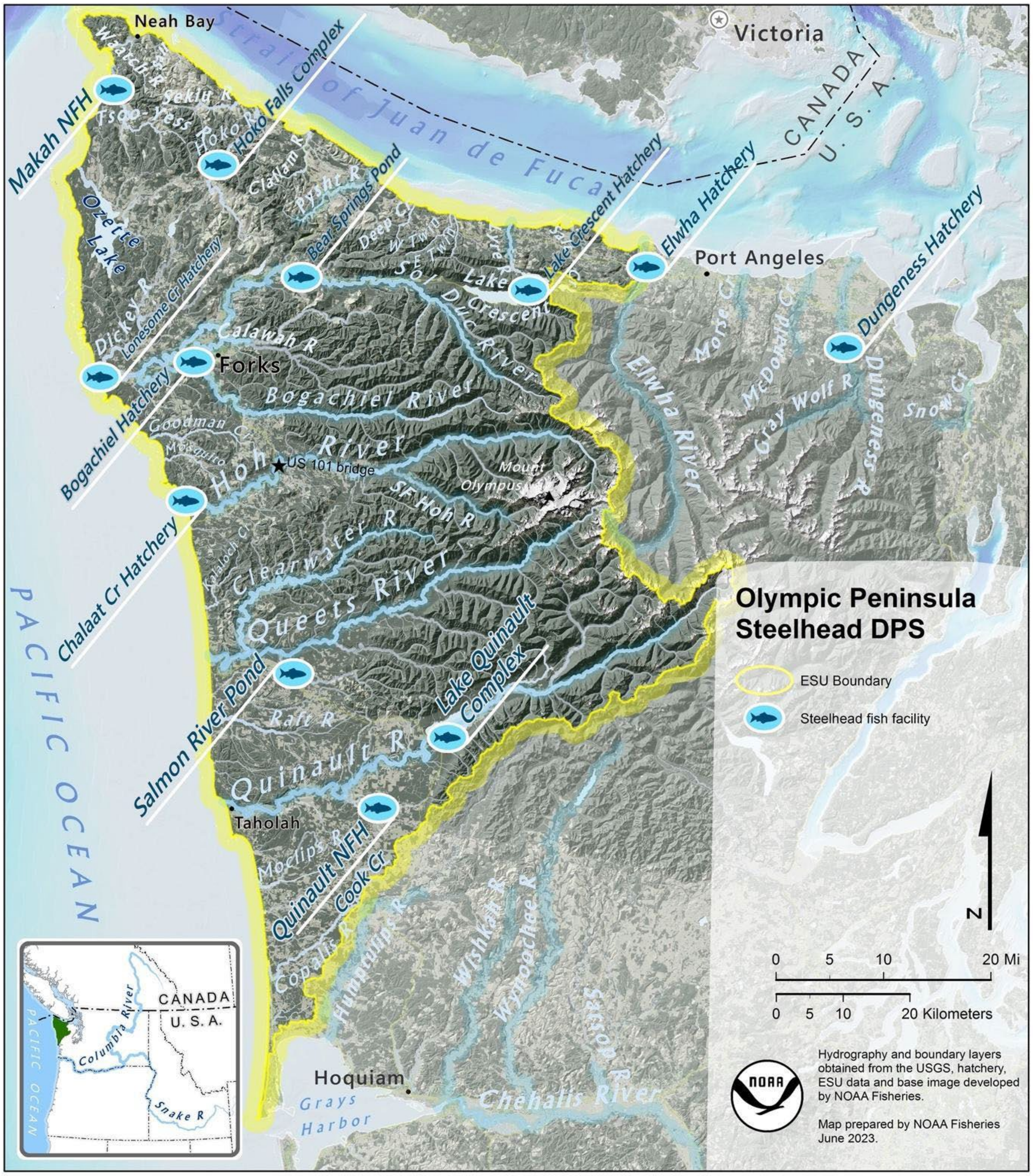



Figure 42. Hatchery facilities in the Olympic Peninsula DPS that currently release winter and/or summer-run steelhead. NFH: National Fish Hatchery. The Lake Crescent Hatchery (Lake Crescent) released resident *O. mykiss* into the 1970s.

Table 15. Current hatchery programs in the Olympic Peninsula Steelhead DPS.

| Hatchery Program | WRIA | Operation | Location of juvenile releases | Run | Program Goal | Release in 2021 | Type of Broodstock Program |
|---|---|---|---|---|---|---|---|
| Hoko Falls Hatchery | 19 | Tribal | Hoko River | Winter | 45,000 EWS | Hoko R: 20,354 EWS<br>Sekiu R: 5,580 EWS | segregated |
| Educket Creek | 20 | Tribal | Waatch River | Winter | 22,000 EWS | | segregated |
| Makah NFH | 20 | FWS/Tribal | Tsoo-Yess River | Winter | 158,000 (EWS) | 128,523 EWS | segregated |
| Lonesome Creek | 20 | Tribal | Bogachiel Hatchery | Winter | 80,000 (EWS) | Transferred to Bogachiel Hatchery | segregated |
| Chalaat Creek | 20 | Tribal | Hoh River | Winter | 100,000 (EWS) | 64,354 EWS | segregated |
| Bear Springs Pond | 20 | Tribal | Quillayute River | Winter | | | segregated |
| Calawah Pond (North and South) | 20 | WDFW | Quillayute River | Summer/Winter | See Bogachiel Hatchery | 56,357 EWS<br>31,486 ESS | segregated |
| Bogachiel Hatchery | 20 | WDFW | Quillayute River | Summer/Winter | 150,000 (EWS) 30,000 (ESS) | 108,281 EWS | segregated |
| Salmon River Fish Culture Facility | 21 | Tribal | Queets River | Winter | 200,000 (WS) | 171,624 WS | integrated |
| Quinault Lake Complex | 21 | Tribal | Quinault River | Winter | 250,000 (WS) | 253,493 WS | integrated |
| Quinault NFH (Cook Creek) | 21 | FWS/Tribal | Quinault River | Winter | 190,000 (EWS) | 225,811 EWS | segregated |

EWS: Early winter steelhead (Chambers Creek Hatchery origin)

ESS: Early summer run steelhead (Skamania Hatchery origin)

Program information from 2022 Future Brood Document Draft, 1108 pg. (https://wdfw.wa.gov/sites/default/files/publications/02295/all_alpha_2022_2nd_draft.pdf)

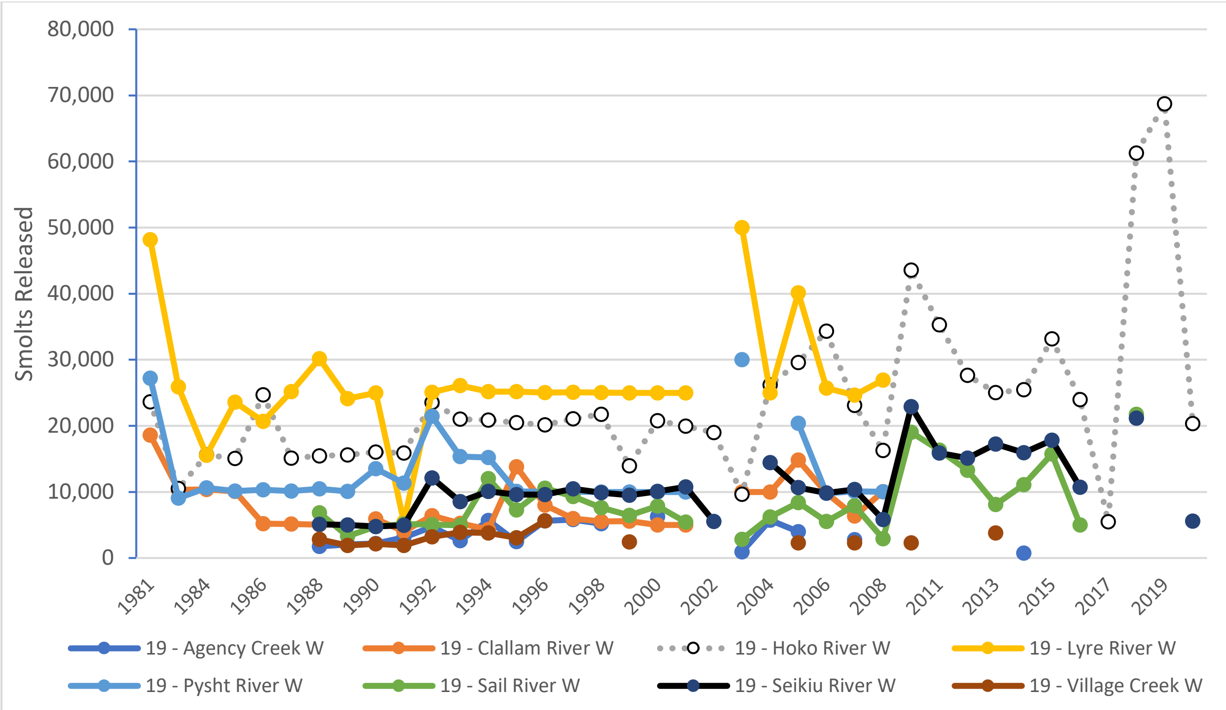



Figure 43. Releases of hatchery-reared winter-run steelhead into Water Resource Inventory Area 19 streams from 1981 to 2021. Releases of juvenile steelhead weighing less than 5 grams are not included. (Data from RMIS database, accessed 23 January 2023).

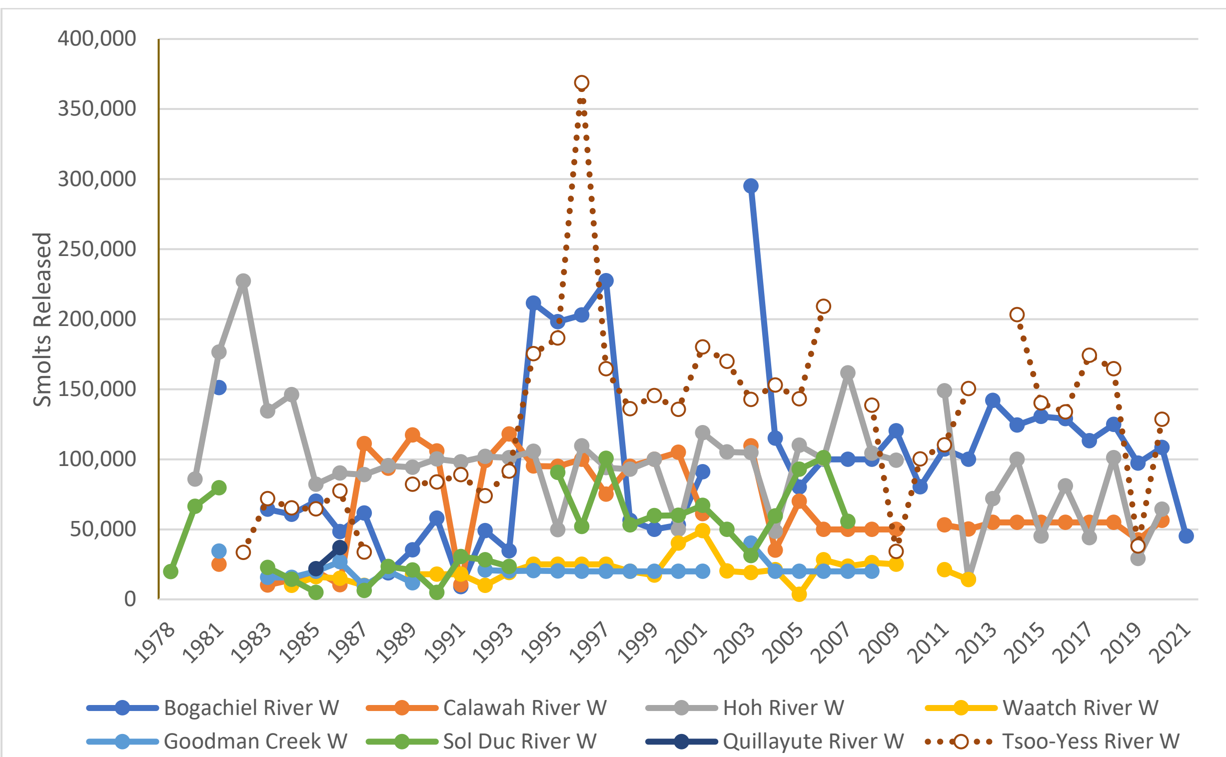



Figure 44. Releases of hatchery-reared winter-run steelhead into Water Resource Inventory Area 20 streams from 1981 to 2021. Releases of juvenile steelhead weighing less than 5 grams are not included. (Data from RMIS database, accessed 23 January 2023).

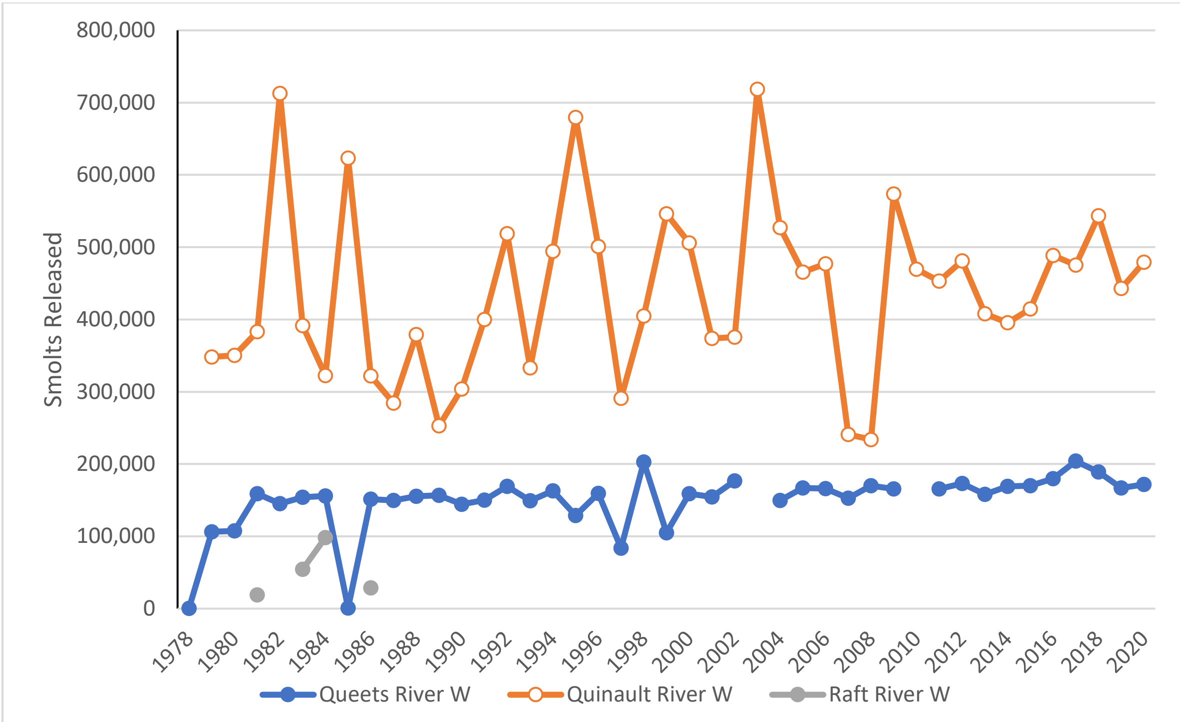



Figure 45. Releases of winter-run hatchery-reared steelhead into Water Resource Inventory Area 21 streams from 1981 to 2021. Releases of juvenile steelhead weighing less than 5 grams are not included. (Data from RMIS database, accessed 23 January 2023).

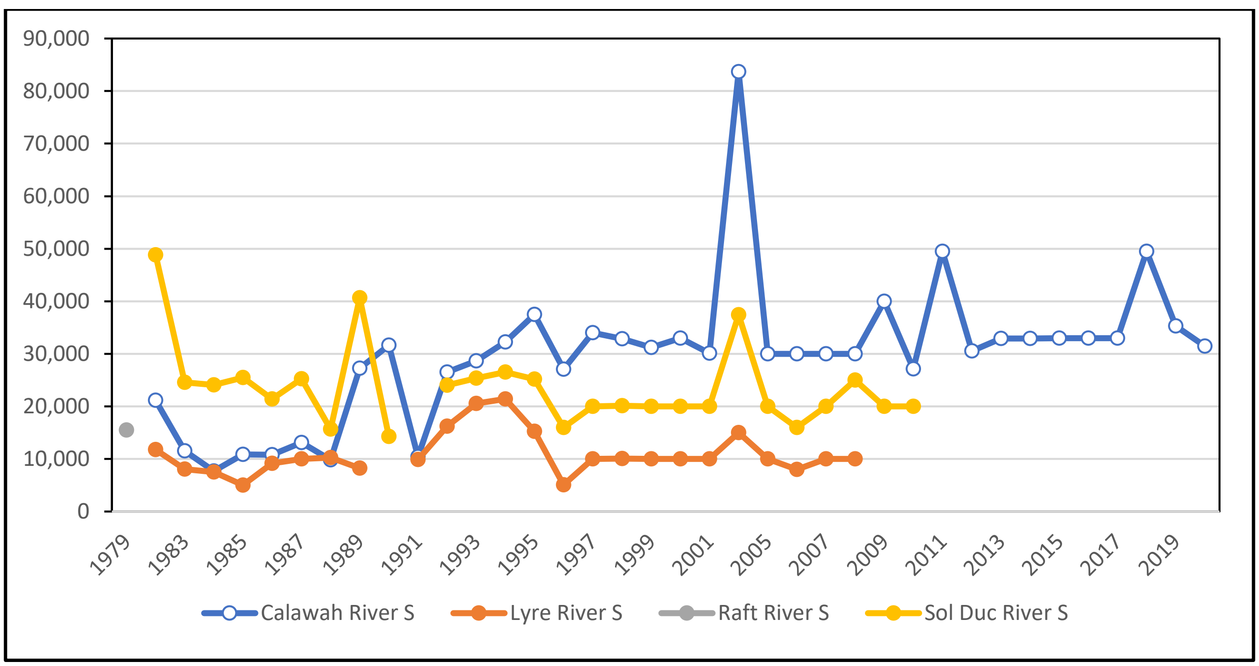



Figure 46. Releases of summer-run hatchery-reared steelhead into Olympic Peninsula streams from 1981 to 2021. Releases of juvenile steelhead weighing less than 5 grams are not included. (Data from RMIS database, accessed 23 January 2023).

While there are multiple hatchery rearing and release sites in the OP DPS for steelhead (Figure 42), these releases are derived from a limited number of broodstocks. In reviewing the relative risks and benefits of hatchery programs, a major concern of the SRT is whether a hatchery stock being used reflects the corresponding natural population, or whether at the time of founding or through subsequent transfers it has been genetically influenced by non-native steelhead introductions. If the broodstock was locally sourced, there are outstanding questions about whether these broodstocks have intentionally or unintentionally been selected for modified life history traits such as run-timing, subject to domestication selection through hatchery rearing practices, or subject to inbreeding through spawning protocol. While many of the details of hatchery operations are not known, there is a relatively complete record of hatchery transfers. In general, hatchery stocks imported from outside of the DPS are assumed to have lower fitness than native populations. For example, the reproductive success of early-winter-run steelhead from the Bogachiel Hatchery (non-native) spawning in Forks Creek (Willapa Basin) was assessed relative to native natural origin winter steelhead; it was reported that the non-native hatchery fish were only 2.3% and 11% as successful as native steelhead in the two brood years studied (McLean et al. 2003). In the coastwide steelhead review, past and present hatchery practices were considered the major genetic threat to the OP Steelhead DPS (Busby et al. 1996). Here the SRT reviewed details on the founding and subsequent operation of hatchery broodstocks in the DPS.

**Current Hatchery Broodstocks**

**Hoko River Hatchery winter-run steelhead:** This hatchery program is operated by the Makah Tribe. The current broodstock was established from returning Hoko River adult winter steelhead since 1990. Prior to 1990, the Hoko River program was stocked by WDFW Bogachiel Hatchery, which was founded by early-returning Chambers Creek Hatchery (South Puget Sound) winter steelhead. WDFW has identified this broodstock as having Chambers Creek origins (Scott and Gill 2008). In addition to the Hoko River, fish from this program have been released in a number of smaller independent tributaries along the Strait of Juan de Fuca over the years, although more recently (post 2010) most of these out-of-basin transfers have been terminated except in Agency, Sekiu, and Village Creeks. This broodstock is currently being operated as a segregated[15] program, and is considered by the SRT as being non-native from outside of the DPS.

**Makah National Fish Hatchery (NFH) winter-run steelhead:** Located on the Tsoo-Yess River, the Makah NFH operates in partnership between the Makah Tribe and the US Fish and Wildlife Service. This is an early-timed, hatchery stock originally founded from Quinault River stock. The hatchery supplies fish to the Educket Creek facility. Broodfish return to the hatchery racks on site. This is a segregated program and only returning hatchery fish are utilized as broodstock. The origins of the Quinault River Hatchery broodstock are unclear but include native Quinault River steelhead and Chambers Creek related stocks, at a minimum this stock is not released in its native watershed. WDFW has identified this broodstock as having Quinault

[15] Segregated hatchery programs do not incorporate natural-origin adults into the hatchery broodstock. This policy is designed to limit the mining of natural populations and the production of hatchery-natural-hybrids that may be more likely to spawn with natural-origin adults in the wild. There is no direct effort to remove hatchery-origin fish from natural spawning grounds.

River origins (Scott and Gill 2008). Genetically, this hatchery stock is closely related to Quinault NFH/Cook Creek early Winter steelhead (Seamons and Spidle 2023). Additionally, there has been some selection for life history traits. This stock is considered significantly distinct from its corresponding natural population and therefore not included in the DPS.

**Quillayute Basin Hatchery summer-run steelhead:** Returning hatchery adults are collected at the North Calawah Pond facility (Calawah River). Spawning and subsequent rearing of juveniles takes place at the WDFW Bogachiel Hatchery, with releases in the Calawah River. This broodstock was founded by transfers from the Skamania Hatchery, Washougal River, Lower Columbia DPS (Scott and Gill 2008). The Skamania stock was established using summer-run steelhead from the Washougal and Klickitat River, and has been in culture since 1963. This broodstock is non-native and is operated as a segregated program. Releases have been confined to the Quillayute Basin. This stock is considered non-native from outside of the DPS.

**Quillayute Basin Hatchery winter-run steelhead:** This is a cooperative program with the Quileute tribe. The founding broodstock for this program was from Chambers Creek-origin stock in 1967. Spawning, incubation, and rearing take place on station at Bogachiel Hatchery. Eggs are also transferred to the Lonesome Creek Hatchery. In addition to on-station releases, fish are also released at Goodman Creek. In the past, releases were more widespread in the Quillayute Basin and independent tributaries to the Strait of Juan de Fuca; the influence of these releases is unknown. Genetically, this stock is distinct from the natural steelhead populations (Seamons and Spidle 2023). This stock is considered non-native from outside of the DPS.

**Hoh River Hatchery winter steelhead:** The Chalaat Hatchery is run by the Hoh Tribe. It releases early-returning winter steelhead, eggs and juveniles have been received from the Quinault NFH since 1984. This program also receives “makeup”eggs from the Bogachiel Hatchery if locally returning adults do not meet egg production needs. This program is designed to support harvest. For further details see Quinault NFH winter steelhead. Both the Bogachiel Hatchery early winter steelhead stock and the Quinault NFH stocks are not included in the DPS; therefore, this hatchery stock would also be excluded.

**Queets River Hatchery winter steelhead:** The Salmon River Hatchery, on the Salmon River, a tributary to the Queets River, currently uses an early-returning broodstock from Lake Quinault (Cook Creek Hatchery). Juvenile steelhead are transferred from the Quinault NFH to the Salmon River Hatchery for final rearing and release. This is an integrated program for harvest use (see Footnote 13). The origins of the Quinault River Hatchery broodstock is unknown, at a minimum this stock is not released in its native watershed. Spawn timing for this broodstock is reported to be temporally distinct from the native population (WDF et al. 1993). Genetic analysis indicates a close affinity of Cook Creek broodstock to Chambers Creek early winter-run steelhead, rather than other OP populations (Seamons et al. 2017, Seamons and Spidle 2023). Juvenile releases are unmarked, increasing the potential for integration with the natural population in the hatchery. This hatchery stock was not considered part of the DPS.

**Quinault River Hatchery Winter Steelhead**: There are two hatcheries currently operating in the Quinault River Basin, the Quinault NFH (Cook Creek), and the Lake Quinault Hatchery. The Lake Quinault Hatchery is operated as an integrated program, while the Quinault NFH is operated as a segregated hatchery[16]; production at both hatcheries is intended for harvest. The Bureau of Commercial Fisheries released steelhead (run unknown) produced from locally returning adults intermittently from 1915 to 1947. There is some uncertainty in the origins of the current Quinault NFH broodstock (HSRG 2004, Scott and Gill 2008). Kassler and Brenkman (2010) suggest that the broodstock originated as a mix of native Quinault River winter steelhead and Bogachiel Hatchery winter steelhead. The Quinault NFH hatchery began operation in 1969, although there has been a salmon hatchery in the basin, operated by various agencies, since the early 1900s. Early-returning winter steelhead return from November to January. Spawning, incubation, rearing and release all take place on site. Genetically, Quinault NFH winter steelhead closely resembles Bogachiel Hatchery winter steelhead (Kassler and Brenkman, 2010). The Lake Quinault Hatchery broodstock similarly has "mixed" origins (WDFW, 2023). Currently the program collects returning adults in set nets in Lake Quinault, spawning, incubation, and early rearing are done at the Quinault NFH Hatchery, with later rearing done in net pens in the Lake. With the exception of 30,000 coded-wire-tag marked juveniles, production releases from this program are unmarked. While the co-managers expressed some uncertainty about the origins these two hatchery stocks, based on the existing stock transfer information, available genetics (Seamons and Spidle 2023), and reported selection within the stocks the SRT did not consider these stocks as part of the DPS.

**Hatchery Interactions**

The percent hatchery contribution to escapement has been estimated for only a few populations in the Olympic Peninsula DPS and for only a few years. In the absence of direct estimates, harvest contribution provides an indicator of the presence of hatchery-origin adults. Royal (1973) reports that winter-run hatchery fish made up 34.0, 19.0, and 73.0% of the sport catch in the Hoh, Sol Duc, and Lyre rivers at a time of non-selective harvest and off-site releases; although the contribution to sport catch most likely overestimates the level of introgression by hatchery-origin steelhead into the native population.

In terms of the effect of hatchery releases into rivers with native steelhead, Royal (1973) wrote:

> One can only conclude from the foregoing that the "wild" winter and probably the summer steelhead populations have declined with the development of the hatchery program involving all stream rearing salmonids including steelhead. In this case "wild" steelhead include both naturally produced hatchery fish, if any, and the original stock of "wild" fish (p 115).

More recently, the Washington Coast Sustainable Salmon Partnership (WCSSP, 2013) estimated the proportion of hatchery-origin adults that were naturally spawning in Olympic Peninsula DPS basins based on the professional opinion of local biologists (WSC 2010). In general, smaller basins with hatchery programs (Tsoo-Yess River, Goodman Creek) and the Quinault River were thought to have higher pHOS levels (26-50%), other basins less so

[16] In the 2023 Co-manager Assessment (COPSWG 2023) it states that all three hatchery stocks [groups of stocks] released into the OP DPS are managed as segregated stocks; using only hatchery origin fish as broodstock. This is in contrast to the 2022 Proviso Plan (Harbison et al. 2022) and Future Brood Document (https://wdfw.wa.gov/sites/default/files/publications/02295/all_alpha_2022_2nd_draft.pdf) statements that the Quinault Lake Hatchery and Salmon River FCF were integrated programs, and does not explain how hatchery-origin and natural-origin fish can be distinguished in the Queets and Quinault rivers, when the majority of hatchery fish released are unmarked.

(>25%); although a number of basins, especially those that drained to the Strait of Juan de Fuca, were not reported; however, changes in hatchery operations since the publication of that report have likely decreased the proportion of hatchery strays on the natural spawning grounds.

### Small population effects

The Petition identified demographic declines in both summer- and winter-run steelhead populations, with summer-run populations being underscored due to the extremely low abundances observed, with population estimates varying from a few hundred adults to near zero. Historical estimates of summer run populations are few. The Sol Duc River was described as having "excellent" winter and summer steelhead, and the Quinault River as having a "fine" summer run of steelhead (Kreider 1948). Meyer (1994) reports that punch card records suggest that summer-run populations have declined since the mid-1970s, and that summer run populations may be at risk.

## Spatial structure

### Barriers

The Olympic Peninsula Steelhead DPS lies in a region of the West Coast that is not affected by major dams or other major in-stream passage blockages. State and County road stream crossings may block or impair passage at culverts, similarly, forest road stream crossing may reduce spatial structure. In general, road culverts block tributary access to relatively small areas of spawning and rearing habitats (Figure 47), but collectively they do not appear to be not a major factor in limiting current productivity. Impassable culverts on State roads are required to be upgraded under the 2013 U.S. District Court Injunction (*U.S. v. WA Culvert Case*), whereas forestry road culverts are covered under the Road Maintenance and Abandonment Plan (RMAP) (Table 15). There has been considerable process in replacing culverts, especially under the RMAP process where over 80% of the culverts are passable (NWIFC 2020).

Table 16. Inventory of culverts, repaired and impassable, in the Olympic Peninsula Steelhead DPS under the Road Maintenance and Abandonment Plan (RMAP) and non-RMAP areas. Data from NWIFC (2020).

| OP Areas[1] | Culverts | Total | Fixed | Fixed (%) | Remaining/ Impassable[2] | Remaining/ Impassable |
|---|---|---|---|---|---|---|
| Makah R | RMAP[3] | 550 | 448 | 81% | 102 | 19% |
| | Non-RMAP[4] | 232 | 75 | 33% | 157 | 67% |
| Quillayute R | RMAP | 691 | 587 | 85% | 105 | 15% |
| | Non-RMAP | 371 | 161 | 43% | 210 | 57% |
| Hoh R | RMAP | 299 | 240 | 80% | 59 | 20% |
| | Non-RMAP | 134 | 67 | 50% | 67 | 50% |
| Quinault[5] R | RMAP | 1,433 | 1,232 | 86% | 201 | 14% |
| | Non-RMAP | 3,108 | 2380 | 77% | 728 | 23% |

1: Tribal Areas as reported in the State of Our Watershed Report (NWIFC 2020)
2: Non-RMAP culverts that are 100% impassable
3: RMAP: Road Maintenance and Abandonment Plan
4: non-RMAP: Culverts on State, County, and other roads

5: Quinault area includes watershed south of the OP DPS

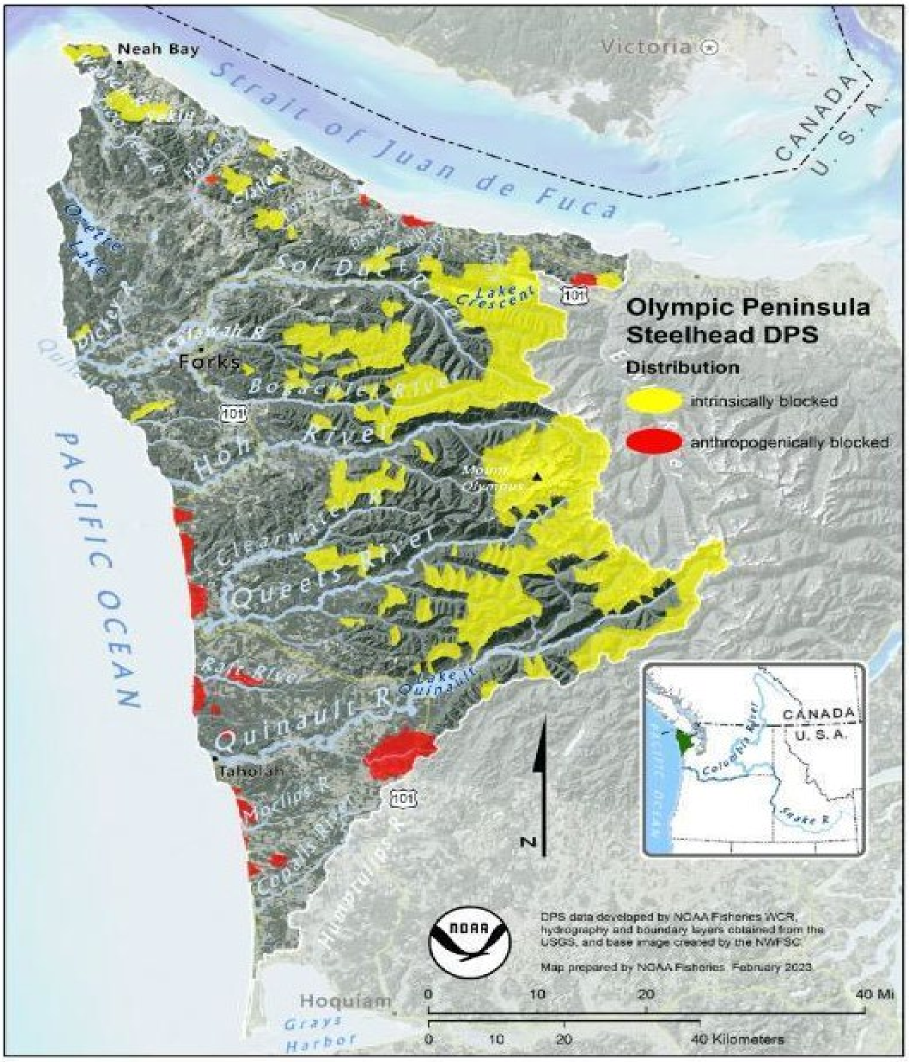



Figure 47. Olympic Peninsula DPS stream basins blocked by natural barriers (yellow) and anthropogenic barriers (red).

The SRT also discussed the potential for future restrictions in spatial structure due to low summer flows that may limit passage to headwater areas. Climate change projections for 2040 and 2080 suggest that low flow and/or high-water temperature barriers (Figure 48) may create temporal passage blockages; these may disproportionately affect summer-run steelhead.

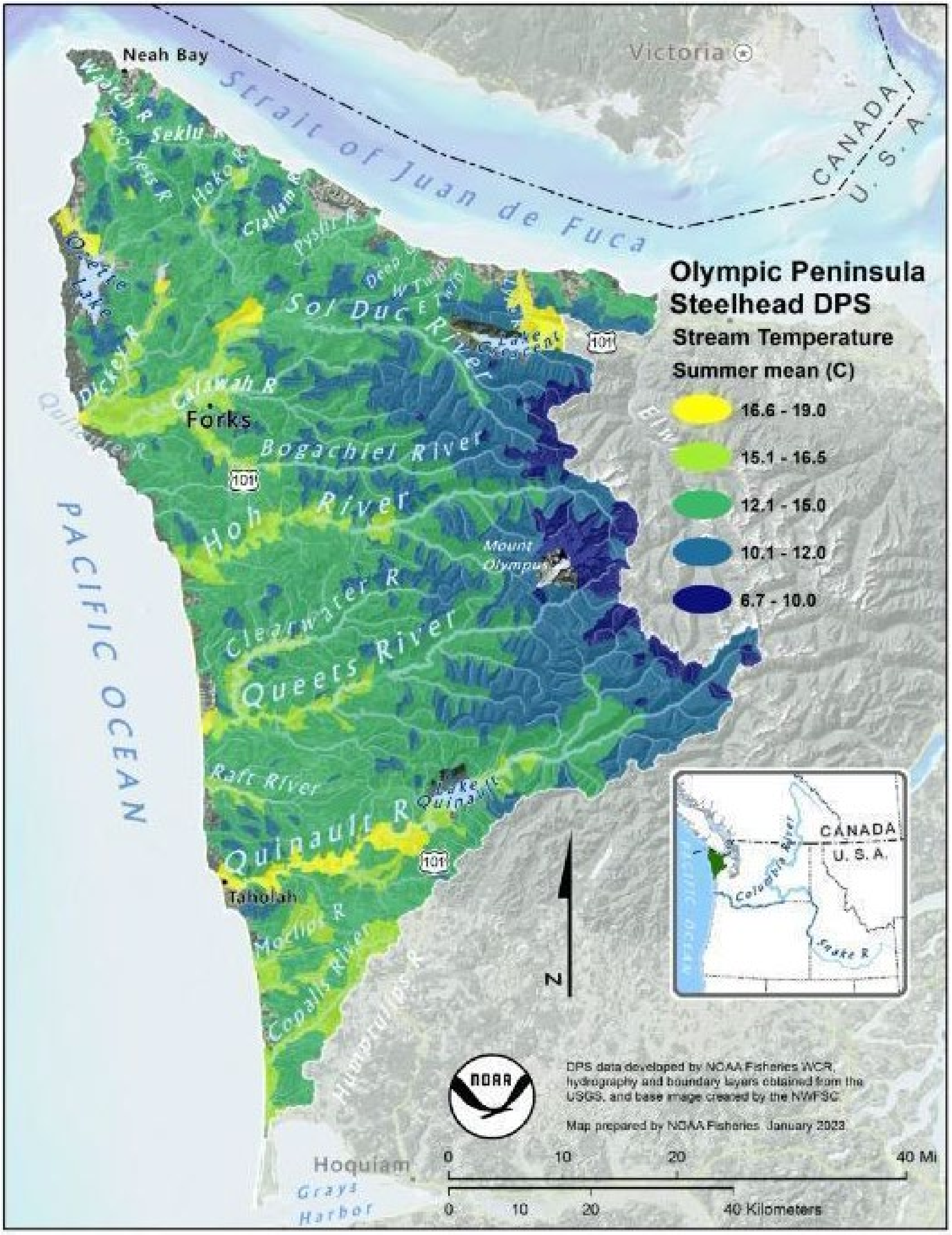



Figure 48. Stream summer mean temperature (°C) for stream reaches in the Olympic Peninsula Steelhead DPS.

## Habitat

The quantity and quality of stream, riparian, and upland habitat can directly and indirectly affect the risk of extinction for the Olympic Peninsula DPS. There have been a number of assessments of salmon and steelhead habitat in this region. Phinney and Bucknell (1975) provided comprehensive stream mapping, with partial and complete barriers, for the Washington Coast. Habitat issues for many OP basins were discussed in Bishop and Morgan (1996) within the context of critical habitat for Chinook salmon. Subsequent analyses were done on streams in WRIA 20 (Smith 2000) and WRIA 21 (Smith and Caldwell 2001). Most recently, the State of Our Watersheds (SOW) reports reviewed conditions throughout much of Western Washington, including basins in the OP DPS (NWIFC 2020). The SRT's assessment of habitat is provided in detail in the threats analysis and habitat appendices. Overall, the SRT found that the majority of the river and riparian habitat was in moderate to good condition, especially those rivers with substantial portions being located within the Olympic National Park (Table 16). Additionally, protections provided by State and Federal forest lands provide some assurance of stable habitat conditions. Other watersheds were still predominantly forested and despite recent habitat improvement efforts, the legacy of past industrial logging practices will continue to negatively affect steelhead productivity in a number of rivers for the foreseeable future.

Table 17. 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 NWIFC 2020.

| 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 $km^2$ | 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. | 769.5 $km^2$ | 388.92 $km^2$ | 51% | 380.61 $km^2$ | 49% |
| Quinault R. | Quinault R. | 1123.48 $km^2$ | 567.02 $km^2$ | 50% | 556.42 $km^2$ | 50% |

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

Pursuant to the ESA and implementing regulations, NMFS determines whether species are threatened or endangered based on any one or a combination of the following Section 4(a)(1) factors: (A) the present or threatened destruction, modification, or curtailment of habitat or range; (B) overutilization for commercial, recreational, scientific, or educational purposes; (C) disease or predation; (D) inadequacy of existing regulatory mechanisms; and (E) other natural or man-made factors affecting the species' existence. We provide a detailed review of ESA Section 4(a)(1) factors, otherwise known as threats, specific to OP steelhead in NMFS 2024b. Here, we provide our main findings for each 4(a)(1) factor, focusing on the time since the last NMFS review of OP steelhead, and present our overall conclusions.

### A. The present or threatened destruction, modification, or curtailment of habitat or range

The current greatest threat to OP steelhead habitat is the legacy impacts from previous land-use practices and to some extent, continued land-use practices. WDFW concluded that legacy impacts of historical (post-contact) land-use resulting in habitat degradation continues to be a threat for wild steelhead, and that these practices include past clear-cut logging, road building, and bank protections that were poorly designed or unmitigated, and floodplain infrastructure impacts (Cram et al. 2018). Pre-contact conditions were influenced by anthropogenic alterations to the habitat (Martin 2023); however, the relative influence of Native Americans on the environment and riparian functions is not comparable to later habitat alterations.

Both logging and agriculture activities result in similar types of impacts to salmonid habitat. It is important to note that the magnitude of impact will vary between agriculture and forestry because of the land conversation that typically occurs with agriculture. Major impacts common to both activities include loss of large woody debris, sedimentation, loss of riparian (streamside) vegetation, and loss of habitat complexity, all of which affect stream channel morphology, environmental conditions (i.e. water quality) and the associated biotic communities. Logging practices prior the 1970s led to "clogged" waterways due to accumulated smaller woody debris that blocked fish migration. Afterwards, actions to remove this woody debris led to too much removal (aka stream cleaning) due to the fact that both smaller and larger material was removed, and resulted in the loss of salmonid habitat (Bottom et al. 1985, CDFG 1994, Botkin et al. 1995) that is likely to persist for another 50 to potentially 200 years (Stout et al. 2018, Martens and Devine 2023). Furthermore, past logging has resulted in the elimination of large trees on streamside areas, so consequently there are very few large-enough trees available for recruitment into streams. Nutrient loading impacts to stream productivity can be caused by mining, livestock, or forest management. Logging has altered stream flows and hydrology, road construction has led to erosion and increased sedimentation, and culverts have blocked access to various spawning grounds and rearing habitat and impacted sedimentation and wood recruitment processes. Alternatively, a portion of OP steelhead habitat is within the Olympic National Park and therefore largely protected from development (see Table 16 for proportion in ONP). However, not all stream/river reach habitat is accessible to steelhead use (see Table 17 below for percent of steelhead habitat used within the ONP). We note that even if steelhead do not utilize portions of a watershed within the ONP, protecting the integrity of the headwater areas provides benefits to the entire system. Restoration projects are occurring, including supplementation of woody debris, and a large percent of culverts that previously blocked migration have been removed, but many also still remain (see NWIFC 2020). Although efforts are underway to address habitat issues, it may take decades for habitat to recover (Martens *et al.*, 2019) and climate change may further delay or prevent recovery (Wade *et al.*, 2013).

Table 18. The percentage of steelhead habitat used that falls within the Olympic National Park (ONP) for various rivers and creeks or basins (for example, "Hoh River" contains subbasins) in coastal Washington that drain directly into saltwater, or in the case of Quillayute – the rivers that comprise the Quillayute system that had more than 0% in the ONP. Any basins/rivers not listed have 0% of steelhead habitat used in the park.[17]

| Basin | Total Length of Steelhead Use (m) | Within ONP (m) | % Within | Outside ONP (m) | % Outside |
|---|---|---|---|---|---|
| **Cedar Creek** | 17,103 | 2,833 | 17% | 14,270 | 83% |
| **Goodman Creek** | 44,652 | 5,443 | 12% | 39,209 | 88% |
| **Kalaloch Creek** | 11,076 | 1,136 | 10% | 9,940 | 90% |
| **Ozette** | 149,053 | 14,113 | 9% | 134,940 | 91% |
| **Mosquito Creek** | 20,269 | 1,710 | 8% | 18,558 | 92% |
| **Upper Quinault** | 183,483 | 119,663 | 65% | 63,821 | 35% |
| **Queets** | 220,090 | 90,816 | 41% | 129,274 | 59% |
| **Hoh** | 276,356 | 103,266 | 37% | 173,090 | 63% |
| **Quillayute:** | | | | | |
| **Bogachiel** | 188,336 | 56,716 | 30% | 131,620 | 70% |
| **Calawah** | 139,831 | 24,264 | 17% | 115,567 | 83% |
| **Sol Duc** | 256,847 | 44,347 | 17% | 212,500 | 83% |

Logging and forestry practices account for the vast majority of land-use impacts that have been, or are, detrimental to OP steelhead habitat; agriculture is also a factor, but to a geographic limited extent. This discussion will mainly concern logging practices.

Strait of Juan de Fuca

The majority of land use on the Strait of Juan de Fuca within river basins in the OP steelhead range is for timber harvest (Table 18). For Salt Creek, state and private forestlands are mostly located in the headwaters (~56%), while agricultural and rural residential lands (42%) are strongly clustered in low gradient stream channel areas in the middle and lower watershed (McHenry, McCoy and Haggerty 2004; North Olympic Peninsula Lead Entity for Salmon (NOPLE) 2015). The Lyre River watershed includes the Olympic National Park (~66%), as well

[17] We attributed the NHD catchments (Hill et al. 2016) with our proto populations (usually inheriting the largest river name) and steelhead distribution (WDFW 2022) by run and use type. These spatial features were then intersected with the land manager polygons from the PAD (USGS 2024) database. From these values we then summarized stream length by steelhead use and population name to determine the quantity and percent of occupied habitat within the Olympic National Park.

as commercial timberlands (31%), and low-density rural residential (~3%) (McHenry, Lichatowich and Kowalski-Hagaman 1996; NOPLE 2015). The East Twin River Basin is mostly forest lands; Washington state Department of Natural Resources lands (WA DNR) and United States Forest Service lands (USFS) comprise over 90% of the ownership (NOPLE 2015). Similarly, for the West Twin River, Deep Creek, and the Pysht River the majority of the land use is for forestry with the majority of the forestlands managed by USFS or WA DNR (~61% for West Twin River, ~50% for Deep Creek, and 75% for Pysht River) followed by 29%, ~43%, and ~24% owned as private timberlands for West Twin River, Deep Creek, and Pysht River, respectively (NOPLE 2015). Washington state timberlands and industrial forest timberlands make up over 95% of the land ownership in the Clallam River basin (Haggerty 2008). The vast majority of land use in the Hoko River is for commercial timberlands; however, portions of the Lower Hoko River and Little Hoko have been converted to open areas or hardwood-dominated areas and purchased by Washington state parks (NOPLE 2015, personal communication with Mike McHenry, Lower Elwha Tribe, December 5, 2023). The Sekiu River predominately contains privately-owned and state-owned timberlands, but is also partially on the Makah Tribal Reservation (NOPLE 2015).

Table 19. Percentage of each landownership type for watershed area by subbasin. Modified from NOPLE 2015. For acronyms: WDNR = Washington State Department of Natural Resources, ONP = Olympic National Park, USFS = United States Forest Service, and Ease./ROW = easements/right of ways.

| Sub-basin | Private | WDNR | ONP | USFS | Reservation | County | Other state land | Other fed land | Ease./ ROW | Other |
|---|---|---|---|---|---|---|---|---|---|---|
| Salt | 50.2% | 44.3% | 0 | 0 | 0 | 1.1% | 0 | 3.1% | 1.34% | 0 |
| Lyre | 10.4% | 17.5% | 65.5% | 5.7% | 0 | 0 | 0 | 0.6% | 0.3% | 0 |
| East Twin | 6.8% | 46.1% | 0.01% | 46.2% | 0 | 0.1% | 0 | 0.5% | 0.3% | 0 |
| West Twin | 29.0% | 9.9% | 0 | 60.9% | 0 | 0 | 0.01% | 0 | 0.2% | 0 |
| Deep | 43.2% | 4.9% | 0 | 50.4% | 0 | 0.6% | 0 | 0.8% | 0.05% | 0 |
| Pysht | 76.7% | 5.9% | 0 | 16.6% | 0 | 0.03% | 0.2% | 0 | 0.5% | 0 |
| Clallam | 49.6% | 47.6% | 0 | 0.1% | 0 | 0.1% | 2.1% | 0.02% | 0.6% | 0.01% |
| Hoko | 72.5% | 24.6% | 0 | 0.9% | 0 | 0.2% | 1.7% | 0 | 0.1% | 0.02% |
| Sekiu | 75.7% | 17.3% | 0 | 0 | 7.1% | 0 | 0.01% | 0 | 0.01% | 0 |
| WSI | 57.1% | 57.1% | 0 | 0 | 16.8% | 0.6% | 0.4% | 1.2% | 1.0% | 0.1% |
| Total WRIA 19 | 51.4% | 22.3% | 11.6% | 9.1% | 3.9% | 0.3% | 0.6% | 0.5% | 0.4% | 0.02% |

Pacific Coast

For the four major river basins on the Pacific Ocean Coast, other than land within the ONP, Olympic National Forest (ONF), or Tribal lands, the remaining land is predominately state or private-owned timberlands. In the case of the Quinault Basin, land ownership varies as a function of whether it is located below and above Lake Quinault. Below Lake Quinault, ownership is predominantly the Quinault Tribal reservation (~80%), followed by Olympic

National Forest (~14%), and private timberlands (~7%). Above Lake Quinault ownership is dominated by Federal lands (~95%), followed by Quinault Tribal reservation (~4.5%), and private lands (<0.5%). See NMFS (2024a) for further descriptions of each individual watershed/river.

NMFS 1996 (Factors for Decline Report) summarizes impacts of logging and agriculture on steelhead habitat by habitat feature - woody debris, sedimentation, riparian vegetation, and habitat complexity/connectivity. We summarized discussion of this in NMFS (2024b) and briefly describe here. Woody debris is important to salmonid habitat because it impacts formation of habitat units, provides shelter (cover and complexity) and protection from peak flows, and acts as substrate (Bisson et al. 1987; Sedell and Maser 1994; Swanson et al. 1976; Hicks et al. 1991), and can produce surfaces for the benthic food web that may be beneficial to salmon (Coe et al. 2009). Loss of woody debris may also reduce the carrying capacity of habitat, increase predation vulnerability for salmonids, lower winter survival rates, reduce food production, and may result in lower species diversity (Hicks et al. 1991). Recent research has shown that there are temporal dynamics of wood and that the status is not necessarily static (see Gregory et al. 2024). In general, 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 useable habitat by scouring and filling of pools and riffles and changing bedload composition; reducing photosynthetic growth and primary production (and thus prey); and affecting intergravel permeability and dissolved oxygen levels (Koski and Walter 1978; Hicks et al. 1991; Suttle et al. 2004; Jensen et al. 2009). Sediment effects on steelhead can be grouped into effects of suspended sediment (turbidity), fine sediment that settles into the bed, and coarse sediment. Egg-to-fry survival asymptotes at only 10% when fine sediment (<0.85 mm) is greater than 25% (Jensen et al. 2009). Reduction in shade canopy from tree loss in the riparian zone can lead to increased water temperatures, and riparian vegetation also protects stream banks from erosion and provides deposition of silt (Bottom et al. 1985; California Department of Fish and Game 1994; Forest Ecosystem Management Assessment Team 1993).

A diverse habitat mosaic is essential for healthy and sustainable salmon and steelhead populations (Brennan et al. 2019; Hilborn et al. 2003). In Pacific Northwest and California streams, habitat simplification has often occurred and led to a decrease in the diversity of anadromous salmonid habitat, salmonid life histories, and overall species complexity (Bisson and Sedell 1984; Hicks 1990; Li et al. 1987; Munsch et al. 2022; Reeves, Everest and Sedell 1993). Reduction of wood in the stream channel, either from past or present activities, generally reduces pool quantity and quality (Wohl 2017), alters stream shading which can affect water temperature regimes and nutrient input (Bowler et al. 2012), and can eliminate critical stream habitat needed for both vertebrate and invertebrate species (Richardson and Danehy 2007).

We summarized land-use practices, as well some specific restoration work, by watershed and river (see NMFS 2024a) relative to the impacts of past land-use practices in the OP. For streams within the Strait of Juan de Fuca watershed, the loss of wood due to systematic removal during the 1950's was widespread, occurring in the Lyre, East Twin, West Twin, Pysht, Clallam, Hoko, and Sekiu rivers. Similarly, the loss of riparian recruitment potential due to previous timber harvest and road development was widespread, and not all streams have had or have ongoing restoration actions (wood treatments) (for example West Twin River but see description of treatment in East Twin River and Deep Creek). Wood treatment to restore woody debris can also be impacted by natural disturbances such as flooding events. There has also been an increase in stream channel incision due to the loss of in-stream obstructions like woody debris and also due to decreased floodplain activity. The frequency of landslides has also increased in the Strait watersheds specifically west of the Lyre River in East Twin, West Twin, Pysht, Hoko, and Sekiu rivers. As we discuss extensively in listing Factor E, related to climate change, increases in winter flow events, decreases in summer flows, and increases in stream temperatures have already been occurring in these watersheds. Finally, the estuarine area has been reduced by almost 50% in the Pysht due to land-use activities and the estuarine mouth of the Clallam River has been blocked due to anthropogenic impacts from channel modifications: log rafting, milling, etc. Restoration efforts in Clallam River have endeavored to reestablish the interface between the river and marine waters. Similarly, in the Pysht River there are plans to restore the estuarine habitat. Thus, for many basins draining to the Strait of Juan de Fuca the legacy of past land use practices continue to influence habitat stream and riparian habitat quality.

Along the west side of the Peninsula there have been similar impacts from previous land-use and logging. Historical (from the last two centuries) land-use practices included: forest harvest without stream buffers, the removal of instream wood, high-density road construction and frequent road use, and harvesting large proportions of watersheds (Martens et al. 2019). Past timber harvest has resulted in changes to sediment supply, wood supply, streamflow, stream temperature, and stream channel morphology. Timber harvest intensity does vary by river; for example, the Calawah River Basin had intensive logging and road building after a fire in 1951, while the Bogachiel River is partially within ONP boundary and has had less timber harvest and road building (Jaeger, Anderson and Dunn 2023). In general, the reduction in wood loadings and instream wood removal have led to the loss of pools, and decreases in stabilizing wood jams which led to the loss of channel complexity (particularly in the Queets) (Abbe and Montgomery 2003; Martens et al. 2019). Wood loadings continue to decrease and the density of large wood in the OP in forests managed by USFS has decreased by ~50% from 2002-2018 (Dunham et al. 2023). Historic logging in the Queets River Basin, even though a large portion of the watershed is in ONP and has a protected floodplain corridor, was intensive and extensive (McHenry et al. 1998). Road construction in the Queets during this time included techniques that are now known to be sub-standard and resulted in road failures, increased landslide rates (which were 168 times those of a natural reference area), reduced stream habitat conditions particularly in some of the tributaries such as the Clearwater River basin, and 2.5 times the instream sediment levels of unclogged OP streams resulting in reduced salmon egg survival and fry emergence from the density of roads (Cederholm and Reid 1987; Cederholm and Salo 1979; McHenry et al. 1998; Tagart 1984). Additionally, the loss of large trees along riparian zones have resulted in greater streambank erosion (Abbe and Montgomery 2003; Martens 2018). Changes to stream channel morphology have resulted from stream channel incision, stream channel widening, and increased bedload movement. Stream width reduction has occurred in the Calawah River Basin since the 1990s, but not in the Bogachiel River (Jaeger, Anderson and Dunn 2023). In the Hoh River, increases in sediment supply (from timber harvest and glacial retreat) has led to an increase in channel width and braiding, and due to the high alpine terrain of the Hoh Basin, its hypothesized that the Hoh could be particularly vulnerable to sediment increases from high-altitude warming

(East et al. 2017). Similar to the Strait, there has been an increase in the magnitude and frequency of flooding events on the west side of the Peninsula. Due to climate change, glacial extent declines have already occurred, with a decline of up to ⅓ of summer critical stream flow from glacial melt) as well as increases in summer water temperatures and decreases in summer flows (Dunham et al. 2023; and see Listing Factor E).

While cumulatively these habitat changes have been large over space and time, the Hoh River Basin, as well as the Queets, Quinault, and Quillayute still exhibit fundamental natural watershed processes and associated habitat characteristics. These include a large forested floodplain that is still intact and functioning. Further a large proportion of these watersheds lie within the ONP, which provides long-term protection from development (Ericsson et al. 2022). Thus, efforts to protect, restore, and increase the overall resiliency of these larger rivers have been implemented to secure core natural assets (Ericsson et al. 2022).

In addition to logging impacts, culverts have blocked or impeded access to spawning grounds and rearing habitat and also restricted downstream recruitment processes for sediment and wood (Kemp 2015; Sullivan et al. 1987). However, restoration actions have occurred and/or are underway to remove culverts and fix fish passage and restore habitat (Table 15). The State of Our Watersheds report (NWIFC 2020) summarizes by major basin the culverts that have already been fixed. Additionally, various projects funded through the Washington State Recreation and Conservation Office since 2000 have led to the protection and restoration of riparian habitat for almost 33,000 acres on the Washington coast (Coast Salmon Partnership 2022 Annual Report - https://coastsalmonpartnership.egnyte.com/dl/VbBakQwmdS). This annual report summarizes various restoration efforts for WRIAs within the OP steelhead DPS boundaries (WRIA 20, 21) including many efforts undertaken by the Tribes. In WRIA 20, there have been 36 fish passage barriers corrected, improvement in sediment transport due to the restoration of almost 450 acres of upland area, 1,353 acres riparian restoration, 11 acres of floodplain reconnection, and 30 miles of restoration instream. In WRIA 21, corrections to 33 fish passage barriers have occurred, improvement in sediment transport due to the nearly 480 acres of upland area restored, 5,939 acres of riparian habitat restored, 14 acres of floodplain reconnected, and 6 miles of restoration instream. For the Pacific Coast Region, that includes watersheds south of the OP, the State of Washington had repaired or replaced 99 fish blocking culverts in the first six years of the program; this however, apparently leaves 226 culverts yet to be replaced by 2034 (NWIFC 2020).

Although efforts are underway to address these issues, it may take decades for habitat to recover (Martens et al. 2019) and climate change may exacerbate conditions (Wade et al. 2013). Even with ~25 years of more protective timber harvest regulations related to riparian zones, important salmonid habitat components such as instream wood and pools have not recovered through natural recruitment of wood (Martens and Devine 2023). The estimated timeline for recovery of these remaining degradations could range between 100 and 225 years (Martens and Devine 2023; Stout et al. 2018).

## B. Overutilization for commercial, recreational, scientific, or educational purposes

Harvest rates for OP steelhead have declined within in the last decade (particularly the last few years) and varies greatly by region (Strait of Juan de Fuca populations vs. the "four major basins" on the coast – Queets, Quinault, Quillayute, and Hoh). We summarize primarily what has occurred since the last NOAA status review (Busby et al. (1996) report), though also provide some information for earlier. Most of the information presented here concerns winter-run natural-origin steelhead in the major four basins (Queets, Quinault, Quillayute, and Hoh – which we refer to as the major four basins) and there are limited data for rivers draining into the Strait of Juan de Fuca (where harvest is mainly terminated) and for summer-run natural-origin steelhead.

Olympic Peninsula steelhead have in the recent past sustained some of the highest harvest rates among Washington state steelhead populations, with an annual harvest rate of 25.6 percent for natural-origin steelhead averaged across rivers for which there was data through 2013 (Cram *et al.*, 2018). The average harvest rate across the major four basins was 36.5% from the 1980s to 2013, including commercial and recreational harvest. Specifically, winter-run natural-origin steelhead in the Hoh, Queets, Quinault, and Quillayute systems have had harvest rates ranging from 7% to >40% annually since the 1980s (till 2013). WDFW stated in Cram et al. (2018)… " These harvest rates are the highest in the state and are of concern given the limited availability of high-quality population-level monitoring data and the recent declines in abundance." WDFW note that harvest rate estimates are only available for one-third of the OP steelhead populations with escapement data and three additional river systems with combined population escapement (Cram *et al.*, 2018), although these populations contain the majority of the overall DPS abundance. Also, although fishing mortality has been relatively high, the declines observed in run size are not likely due to harvest alone, but more likely some combination of factors (yet undetermined) in combination with harvest rate.

Estimates of combined commercial and recreational harvest since the 1980s for winter-run natural-origin steelhead in the four major basins were provided by the co-managers along with estimated run size, which can be used to estimate harvest rates (Figure 32, Table 19). Data from recent years (2014-2022) not included in Cram et al. (2018) show harvest rates in the major four basins ranging from 13.26% to 59.19% through 2020. From 2013 to 2020, average harvest rates were 31% and 42% in the, Queets and Quinault rivers, respectively and 22-23% in the Quillayute and Hoh rivers. In the last two years for which the Team had data (2021, 2022), there has been considerable declines in harvest rates to 8.66% to 15.44% across basins, rate declines of approximately 50-70% (Table 19).

The SRT acknowledges that Indigenous groups have managed fisheries and the landscape since time immemorial (for example see explanation in Martin 2023), during a time when steelhead thrived. A document by Martin (2023) from Makah notes that sustainable harvest management is a core principle of traditional resource management and embedded into Tribe societal roles, salmon and steelhead have been managed since time immemorial (including their habitat), and this management included both traditional hatchery and harvest practices (see further information from that document presented in Factor D).

Recreational and Tribal catch of winter-run population has typically occurred from November to April. In 2004, Olympic National Park implemented catch-and-release regulations for wild steelhead throughout coastal rivers of the DPS within the park. In 2016, WDFW changed the recreational fishing regulations to prohibit retention of natural-origin (unmarked) winter-run steelhead in OP steelhead river basins. Where available, mortality from catch and release data assumes a 10% hooking mortality; however, for most river system the estimates of harvest rates presented here do not include catch and release (hooking) mortality (see below in this section for further information on where included, including for the Hoh River). Additionally, information from Bentley (2017) led to a sport angler encounter rate calculation of 1.14 for wild steelhead, implying some steelhead are caught and released more than once (Harbison et al. 2022). Estimates of the effect of multiple captures on hooking mortality are not available. Overall, given that the SRT did not have a complete estimate of hooking mortality for most populations, it was presumed that available estimates were a minimum at best and hooking mortality could be relatively high in certain systems especially in the last few years when landed catch has been low (in the low hundreds of fish in certain rivers).

Table 20. Calculated harvest rates (commercial and sport) rate for natural-origin steelhead in the Queets, Hoh, Quinault, and Quillayute Rivers from 1978 to 2022 based on total run size and escapement data provided by the co-managers (Tribes and WDFW). Harvest is equal to run size – escapement and percent harvest is equal to harvest / run size[18].

| **Year** | Hoh River | Queets and Clearwater Basin | Quillayute Basin | Quinault (Upper + Lower) River |
|---|---|---|---|---|
| **1978** | N/A | N/A | 17.23% | N/A |
| **1979** | N/A | N/A | 32.67% | N/A |
| **1980** | 0.00% | | 30.73% | N/A |
| **1981** | 0.00% | 47.27% | 22.40% | N/A |
| **1982** | 0.00% | 38.43% | 23.01% | N/A |
| **1983** | 0.00% | 45.78% | 18.68% | N/A |
| **1984** | 0.00% | 45.76% | 19.45% | N/A |
| **1985** | 0.00% | 49.50% | 40.71% | 49.17% |
| **1986** | 0.00% | 45.32% | 25.28% | 34.38% |
| **1987** | 35.76% | 48.71% | 33.31% | 66.33% |
| **1988** | 49.07% | 48.50% | 38.29% | 50.77% |
| **1989** | 36.40% | 41.83% | 28.45% | 48.24% |
| **1990** | 47.18% | 42.84% | 38.24% | 42.83% |
| **1991** | 33.83% | 37.26% | 38.00% | 46.01% |
| **1992** | 54.35% | 41.27% | 54.38% | 57.40% |
| **1993** | 50.46% | 38.97% | 53.10% | 60.41% |

[18] It is possible that steelhead harvested post-spawning (kelts) would be counted in both escapement and harvest; however, the harvest during March-May period (when kelts would be encountered) is relatively low.

| **Year** | Hoh River | Queets and Clearwater Basin | Quillayute Basin | Quinault (Upper + Lower) River |
|---|---|---|---|---|
| **1994** | 43.86% | 28.16% | 33.69% | 40.11% |
| **1995** | 38.28% | 39.20% | 34.89% | 42.85% |
| **1996** | 42.89% | 54.80% | 29.72% | 52.18% |
| **1997** | 27.55% | 41.55% | 35.96% | 41.15% |
| **1998** | 7.24% | 28.87% | 10.30% | 51.93% |
| **1999** | 24.93% | 42.77% | 21.50% | 46.20% |
| **2000** | 29.23% | 30.25% | 28.39% | 45.96% |
| **2001** | 48.29% | 31.48% | 36.48% | 59.85% |
| **2002** | 45.15% | 10.40% | 28.23% | 61.40% |
| **2003** | 54.90% | 35.06% | 28.04% | 54.90% |
| **2004** | 44.04% | 17.22% | 25.74% | 62.01% |
| **2005** | 41.71% | 16.37% | 24.25% | 43.93% |
| **2006** | 10.97% | 14.61% | 18.25% | 41.03% |
| **2007** | 22.69% | 28.43% | 36.14% | 38.63% |
| **2008** | 30.91% | 19.22% | 25.78% | 31.77% |
| **2009** | 28.18% | 23.95% | 30.25% | 45.91% |
| **2010** | 26.56% | 29.56% | 27.32% | 37.54% |
| **2011** | 20.37% | 35.07% | 19.48% | 29.52% |
| **2012** | 28.50% | 42.64% | 29.41% | 56.30% |
| **2013** | 36.76% | 38.28% | 29.16% | 49.12% |
| **2014** | 43.19% | 31.31% | 26.65% | 47.46% |
| **2015** | 26.58% | 30.67% | 29.19% | 44.43% |
| **2016** | 19.31% | 29.16% | 30.34% | 59.19% |
| **2017** | 16.63% | 39.78% | 16.53% | 33.41% |
| **2018** | 13.79% | 20.86% | 15.63% | 28.14% |
| **2019** | 13.26% | 29.90% | 13.90% | 36.51% |
| **2020** | 19.31% | 29.91% | 13.94% | 37.39% |
| **2021** | 12.29% | 9.76% | 10.93% | 15.44% |
| **2022** | 9.96% | 8.66% | 8.93% | 11.31% |
| Average most recent decade (2013-2022) | 21.11% | 26.83% | 19.52% | 36.24% |

Notably, outside of the major four basins, directed steelhead harvest for most rivers along the Strait of Juan de Fuca was terminated in various years since the late 2000s/2010s (see Figure 35, but see Hoko) (see section *Population Growth and Harvest in Strait Populations*). For harvest in rivers along the Strait, estimates of growth rates for each population were plotted through time highlighting when harvest ceased (Figure 32). The growth rate patterns appear highly correlated among streams even for those where fishing has not ceased. Therefore, it appears that other factors (freshwater and/or ocean conditions) may also be influencing trends in Strait populations.

Additional strategies since the 1990s have been employed to support sustainable fishing, including: harvest restrictions, shorter seasons, and gear restrictions (Harbison et al. 2022; COPSWG 2023, and see Listing Factor D). In recent years, WDFW and ONP have shortened or closed the recreational fishing season on winter-run OP steelhead in part due to low returns. WDFW also imposed restrictions on recreational angling by banning the use of boats ("no fishing from a floating device") and bait (see the following: *https://wdfw.wa.gov/publications/02349; https://wdfw.medium.com/changes-to-the-coastal-steelhead-season-67131dd05ba7; https://wdfw.medium.com/frequently-asked-questions-march-2022-coastal-steelhead-closure-364cfa62826f; https://www.peninsuladailynews.com/sports/fishing-olympic-national-park-to-shut-down-fishing-on-west-end-rivers/* ).

In 2022-2023 sport fishing was closed on the Quinault and Queets for December 1st- April 30th because of low returns, and "failure to reach agreement on an acceptable level of wild steelhead harvest". The total number of weeks of Tribal fisheries has declined in recent years (see more information below) specifically in the Queets and Quinault, and as mentioned before, harvest rates have declined. In addition, WDFW added harvest restrictions to protect returns to the Bogachiel Hatchery to ensure broodstock egg take (https://wdfw.wa.gov/newsroom/news-release/wdfw-announces-2022-2023-coastal-fishing-season). WDFW implemented similar gear and floating device restrictions for 2023-2024 and set a bag limit of two hatchery steelhead (https://wdfw.wa.gov/newsroom/news-release/wdfw-announces-2023-2024-coastal-steelhead-season). For the 2023-2024 season, the National Park Service closed Queets and Quinault Rivers within the ONP to sports fishing beginning on November 27th, 2023 (https://www.nps.gov/olym/learn/news/temporary-sport-fishing-closure-necessary-to-protect-declining-populations-of-wild-steelhead.htm).

On January 26, 2024, the co-managers clarified for the SRT in a written response what data are included in estimates of run size and harvest (email correspondence with Jim Scott, on behalf of the co-managers, January 26, 2024). For the Hoh River, run size and total catch of natural-origin steelhead included hooking mortality in the sport fishery dating back to 2003/2004 season. The estimated mortality was based on total estimated encounters from sport creel surveys multiplied by 10%, the presumed hooking mortality rate. For the Quillayute, Queets, and Quinault Rivers, annual run reconstruction and total catch of wild steelhead does <u>not</u> account for hooking mortality in the sport fishery. Therefore, the total number of natural-origin winter steelhead mortalities from sport fisheries was underestimated for those rivers in all years. For the Hoh River and Quillayute River Basins, ceremonial and subsistence fisheries were included in the estimates of total run size. For the Queets and Quinault systems, on reservation hook and line harvest are currently included in the data, although it was not until the 2020/2021 season that the

Tribal-managed (on reservation) non-treaty recreational harvest component for the Queets system was included. Furthermore, there are key differences in estimates of natural-origin steelhead escapement in surveys in Quillayute/Hoh versus Queets systems. The Quillayute/Hoh estimates are based on number of redds x 0.81 female/redd x 2 fish. In the Queets, the estimator is total number of redds x 1 female/redd x 2 fish. Assuming 1,000 redds in a given river, these escapement estimates of natural-origin fish vary by 19%.

Efforts to estimate harvest are also potentially biased because harvest normally occurs from November to May, while escapement is calculated from counts of redds created after March 15th when it is assumed that all the fish present are natural-origin steelhead. Therefore, those natural-origin fish returning and spawning prior to March 15th would not be counted in redd surveys, resulting in potential underestimate of run sizes and an overestimate of harvest (see discussion above in Life History Traits about run-timing of natural-origin steelhead). Harvest rates for winter-run steelhead include any and all steelhead landed in the weeks between week 45 (approximately November 1st) and week 18 in the following year (approximately April), in directed fisheries or as bycatch in other fisheries[19]; however, any steelhead caught in other salmonid fisheries outside this time period were not included.

In Factor D (Inadequacy of Existing Regulatory Mechanisms) both here and in NMFS (2024b), we provide more detail on how fisheries are managed, specifically that OP steelhead fisheries are mainly managed for escapement goals for winter-run steelhead based on freshwater productivity (see Gibbons, Hahn and Johnson 1985). The established escapement goals vary by river system and range from <100 (in smaller rivers on the Strait) to 5,900 natural origin winter steelhead (Table 4). In the Queets River system, the co-managers have differing escapement goals[20]. Each year, specifically for the major four systems, the co-managers develop management plans outline forecasted run sizes, escapement goals, harvest rates, and fishing seasons (both recreational and commercial). For the Quinault River, although escapement was met in the most recent years (Figure 49), escapement was met only 43% of the time since 1970. In recent years (2021-2022) harvest rates were lower (as noted above) because of low returns in certain rivers, but to the extent necessary to meet escapement goals. Specifically, in the Queets, the State-specific escapement goals were not met in 2020-2021 and 2021-2022 even with the lower harvest rates because returns were low. The returns, however, met the Tribal escapement goal, which is lower. For 2023 in the Queets River the projected return was 4,150 (beginning below the State escapement goal), and State and NPS closed fishing, but the harvest rate was set at 16% for the Tribal fishery, leading to an estimated escapement below the State escapement goal but greater than the Tribal escapement goal. This is not the case in each system and each year. For example, in the Quillayute River, the 2022 harvest was managed to provide escapement above the goal (Quileute-WDFW 2022 plan), and, in general, the escapement goal is more consistently met in the Quillayute (Figure 49). Similarly, for Hoh River, in 2020 harvest rates were set to

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[19] Scott, J.B. OP steelhead follow-up questions. Email to Laura Koehn. 17 July 2024

[20] [20] The Tribal escapement goal of 2,500 comes from a calculation for the number of spawners needed for maximum sustainable yield ($S_{msy}$) calculated separately in the 1980s to be 2,500 but with the caveat that more data was needed. In the late 90s, $S_{msy}$ was recalculated based on the best estimate of the stock-recruit relationship (Ricker curve) to be 2,700 with a highest probability range of 2,500-2,900. (Scott, J.B. OP steelhead follow-up questions. Email to Laura Koehn. 17 July 2024).

provide escapement slightly over the goal (2,485 projected natural-origin escapement). Whether escapement goals are met varies depending on which (State or Tribal) escapement goal is considered. Even with lowered harvest rates in recent years, certain system's harvest rates are still leading to adult returns under the State escapement in the Queets (but not the Tribal escapement goal).

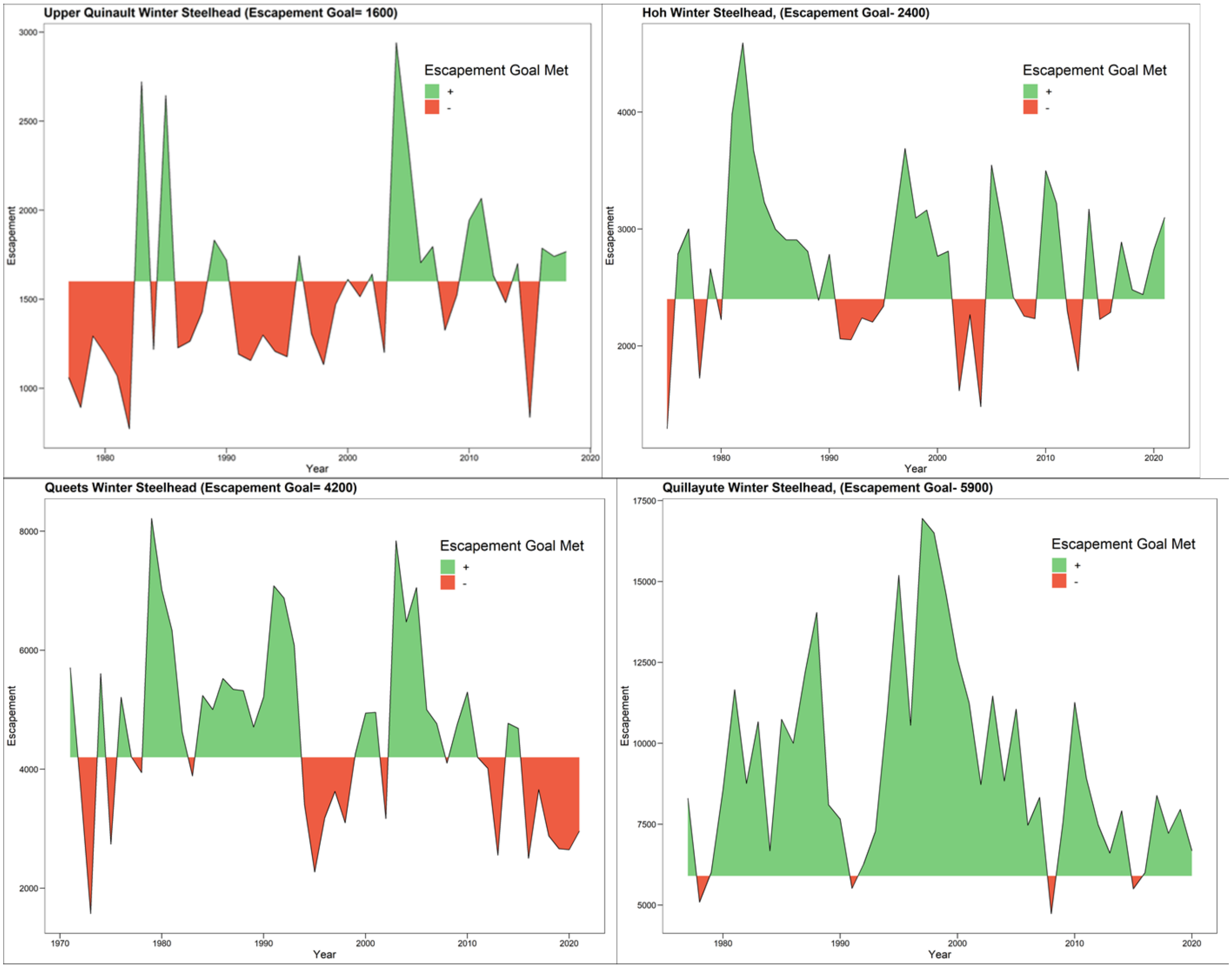


Figure 49. Winter steelhead escapement and escapement goals for the a) Upper Quinault River b) Queets River, c) Hoh River, d. Quillayute. Note that the Washington State escapement goal for the Queets River is 4200, but the Quinault Tribal escapement goal is 2500.

Forecasting accuracy certainly influences whether harvest rates are set to achieve escapement goals in the Olympic Peninsula DPS (Figure 49). In-season harvest monitoring provides some ability to manage escapement. The co-managers state in their 2023 review to the SRT that, "Tribal fisheries are generally shaped by time and area restrictions with in-season management based on monitoring of fishery catches," (Co-Manager Olympic Peninsula Steelhead Working Group 2023). The co-managers provided examples of in-season management and management taken in recent years (Scott, J.B. OP steelhead follow-up questions. Email to Laura Koehn. 17 July 2024). Specifically, for the Quillayute River, in-season fishery catch monitoring led to an earlier closure in February of 2022 given low returns and low harvest, leading to harvest of 385 fish and escapement of 8,516 (above the escapement goal). Since the 2021/22 season which had the lowest run size of recent years, there has been an increase in on river days to 52.7 in 2022/23 and 57.7 in 2023/24 (up from 48.7 in 2021/22) and total run sizes of 9,344 and 9,096 in these years (above escapement, with the 2023/24 escapement still being projected and not a final estimate). For the Hoh river, Tribal fishing has closed in weeks 13-16 since 2015 as this was identified as peak steelhead run time. Harvest was extended in the Hoh River to 17 weeks in 2024, but with fewer fishers participating in the fishery. In the Queets and Quinault rivers, total fishing days has fluctuated through the years during periods of severe changes in ocean conditions. Specifically, in the 1990s to early 2000s, fishing days in the Queets was reduced from an average of 91 to an average of 68 days, and in the Quinault fishing days were reduced from average 106 to 100 days, particularly later in the season (March, April) during natural-origin spawning for both rivers. In the mid 2000s, average days of fishing increased (average 102 days in Queets and average 104 days in Quinault), but with roughly 50% of harvest levels observed in the 1970s. Between 2017/18 and 2020/21 seasons, fishing days were again reduced to 78 days and 88 days on average in the Queets and Quinault, respectively, and early closures were implemented. Finally, in the most recent seasons (2021/2022 and 2023/24), average gillnet days have been reduced to 35 days in each system (Queets and Quinault) with early closures in February and early sport closures as well (in February or early March), leading to catch limits of natural-origin fish of 200 (<10% harvest rates).

The SRT model for harvest mortality fits and produces reasonable estimates of escapement, and harvest (Figure 35). Estimates for this model suggest that populations along the coast (Hoh, Queets, Quinault, and Quillayute) largely have an <u>intrinsic</u> population growth substantially greater than zero (point estimates of $\mu_i > 0.15$ for all populations). However, they are also subject to substantial fisheries mortality and that in most years, this fishing mortality is greater than intrinsic mortality (i.e. generally $\mu_i - F_{it} < 0$), which will result in declining population growth. A small minority of years for each population were judged to have had population growth greater than 0. Estimates of correlations in escapement among populations were positive and large, indicating that all four of these populations fluctuated in unison ($\theta = 0.83[0.62, 0.97]$ mean, [95%CI]) (see section above on Population Growth and Harvest in Coastal Populations.

For summer-run steelhead, directed catch-and-release regulations have been in place from WDFW in state waters and in the ONP since 1992 under the NPS, and there are no set escapement goals. Steelhead fisheries target winter-run steelhead; however, data shows harvest (and/or catch and release mortality) of summer-run steelhead in recent years (NMFS 2024b); above section *Summer-run Steelhead Population Harvest*). It is difficult to interpret the impact of catch when summer-run abundance is more uncertain than winter-run abundance (see section *Summer-run escapement data above*), but available information suggests that the harvest of natural-origin summer-run steelhead has declined since the last NMFS review (NMFS 2024b). Further, the Team did not have information on indirect harvest of summer-run steelhead in fisheries targeting other Pacific salmon (this may be available in fish ticket information). In light of commercial gill-net fisheries and recreational fisheries, adult summer-run steelhead are susceptible[21] to bycatch during their upstream migration to spawn, prespawning holding, or as seaward migrating kelts. Given that summer-run population abundances are inherently smaller, this likely increases the extinction risk for these populations.

[21] By catch rates depend on the specifics of the gear used, timing, and size/age of steelhead.

## C. Disease or predation

Both disease and predation effects on steelhead have not been intensively studied for Olympic Peninsula steelhead. Some outbreaks of infectious hematopoietic necrosis virus (IHNV), reovirus, and Pacific salmon paramyxovirus have been documented in OP steelhead, mainly in hatchery-origin fish, though natural-origin fish are not generally sampled. Breyta *et al.* (2013) summarized previous outbreaks of the M genogroup (group of related viruses) of IHNV in the Hoh, Queets, Quinault, and Quillayute River basins (as well as other coastal areas) between 2007 to 2011. M genogroup IHNV is particularly virulent in steelhead and rainbow trout, with high levels of mortality. Prior to 2007 there was only one detection in Washington coast steelhead, in the Queets watershed at the Salmon River Hatchery (in 1997). Most detections from 2007-2011 were in hatchery-origin fish, but Breyta *et al.* (2013) noted that natural-origin fish are less commonly sampled, and there were detections of this virus in natural-origin fish in the Hoh and Quinault River basins. No IHNV was detected in 2012, but the future risk of IHNV in OP steelhead is unknown given known fluctuations of IHNV incidences in other regions (like Columbia River basin) (Breyta et al. 2013). The effect of IHNV varied across various streams in Washington State and this variation was not fully explained by differences in virulence or hatchery water supplies (Breyta et al. 2014). Even two separate hatchery populations that came from the same ancestral population had variation in mortality after exposure to a MD IHNV strain. Work by Brieuc et al. (2015) suggests that there is a genetic basis for resistance to IHNV and that the population has the ability to adapt, therefore reduction of genetic variation could impact future adaptation and resistance. Exposure may lead to selection of resistance to diseases, but adaptation and the rate that populations become resistant depends on heritability (see Crozier et al. 2008), and Brieuc et al. (2015) showed that resistance to IHNV is likely heritable. Sockeye salmon are frequently infected with IHNV (Traxler et al. 1997; Dixon et al. 2016) so where sockeye could come into contact with steelhead, particularly in hatcheries or in rivers like the Quinault that support a large sockeye run, this could lead to further exposure to steelhead.

Similarly, we obtained data from Tony Capps (WDFW) on instances of disease, parasites, and viruses in steelhead hatcheries (state, federal, and tribal) on the Peninsula. There were four cases of reovirus in winter-run steelhead in December 2002, January 2003, December 2006, and February 2007, all in the Bogachiel system except the 2007 occurrence in the Sol Duc River, with a later occurrence in January 2020 in winter-run steelhead in the Bogachiel. There were eight instances of IHNV in winter-run steelhead on the Bogachiel River in winter 2009-2010, with six in December of 2009 and two in January 2010 (possibly the same as noted in Breyta et al. 2013). Finally, there were two instances of Pacific salmon paramyxovirus in summer-run steelhead in Bogachiel River in summer 2017. Again, most of all known cases are in hatchery fish populations and not a lot of information exists on impacts to natural-origin steelhead in the OP. We note that to accurately assess the potential threat of disease in this population we would need annual pathology reports from each hatchery to effectively assess presence/prevalence of pathogens, viruses, bacteria.

Predation on salmonids can occur among other fishes, particularly during salmonid juvenile life stages, among avian predators, and among marine mammals, including Resident Killer Whales. Public comments on the 90-day finding included mentions of predation by seals, sea lions, otters, eagles, killer whales, cormorants, and/or mergansers on steelhead, including anecdotal accounts of seeing predation by mergansers, otters, and eagles, in the OP steelhead rivers. Invasions of non-native fish species pose threats to native fish fauna but little is known on the extent or effects on OP steelhead. The following nonnative fish species occur in waters of the OP steelhead DPS: Eastern brook trout *(Salvelinus fontinalis*), Atlantic salmon (*Salmo salar*), Westslope cutthroat trout (*Oncorhynchus clarkii lewisi*), yellow perch (*Perca flavescens*), yellow bullhead (*Ictalurus natalis*), largemouth bass (*Micropterus salmoides*), American shad (*Alosa sapidissima),* and Common carp (*Cyprinus carpi*o) (NWIFC 2020).

Avian predators (gulls (*Larus spp.)*, mergansers (*Mergus spp.*), herons (*Ardea spp*.), diving birds like cormorants (*Nannopterum spp.*) and alcids (Family *Alcidae*), including common murres (*Uria aalge*) and auklets (*Althia spp.*) as well as others) have also been shown to impact juvenile salmonids which is summarized in NMFS 1996. More recently, Caspian terns and double-crested cormorants have been documented consuming outmigrating steelhead smolts in the Snake River basin (Hostetter et al. 2015), as well as gulls in the Columbia River (Evan et al. 2019). Avian predation on juvenile salmonids can occur as they enter the ocean as well (Zamon et al. 2014; Tucker et al. 2016). Seabirds are present in the OP watersheds but we are unaware of any unusual or excessive predation events by seabirds or hotspots of seabird predation (based on pers. Comm. with Thomas Good, 15 October 2023, NMFS NWFSC).

The four common marine mammal predators of salmonids in the eastern 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*) (and see the summary in NMFS 1996). Recent research suggests that predation pressure on salmon and steelhead from seals, sea lions, and killer whales has been increasing in the northeastern Pacific over the past few decades (Chasco et al. 2017 a, b; Couture et al. 2024), but this work has been mainly focused on predation on Chinook salmon (Couture et al. also discuss other salmonids but there is limited mention of steelhead). A recent review of pinniped predation in Puget Sound and the Washington Coast concluded that pinnipeds are responsible for reduced abundance of salmon in Washington State waters, but are not likely a primary cause of the lack of salmonid population recovery in these ecosystems (WSAS 2022). Some studies have found that harbor seals can have a significant predation impact on coho salmon and other salmon species of conservation concern (Thomas et al. 2017), as well as steelhead (in Puget Sound; Moore et al. (2021) Moore and Berejikian (2022)) through the consumption of outmigrating juveniles. Given that Moore et al. (2021) showed reduced steelhead smolt survival from Nisqually through Puget Sound out to the Pacific Ocean, and OP steelhead along the Strait of Juan de Fuca would migrate through a portion of this area as well, seals are likely impacting to some extent steelhead smolt survival. Moore et al. (2021) also showed that this impact to smolt survival is higher in years with less anchovy (another similarly-sized harbor seal prey). Work synthesized in Pearson et al. (2015) suggests that marine mammal predators can detect pings emitted by acoustic tags and target those fish, thus biasing survival results. Also, harbor seal predation data specific to coastal tributaries is not currently available, so the extent to which predation by seals in rivers and estuaries is a threat to specific Washington coastal salmon populations is currently unknown.

The relative impacts of marine predation on anadromous salmonids are not well understood. However, it is evident that anadromous salmonids have historically coexisted with both marine and freshwater predators (as well as Indigenous groups) and based on catch data, some of the best catches of coho, chinook, and steelhead along the West Coast of the United States occurred after marine mammals, kingfishers, and cormorants were fully protected by law (Cooper and Johnson 1992). Based on this, it would seem unlikely that in the absence of man's intervention, freshwater or marine predators would extirpate anadromous salmonids. It is likely that historical harvest of harbor seals and other marine mammals by Indigenous communities may have reduced predation on salmonids. Anthropogenic habitat alterations including dams, irrigation diversions, fish ladders, and man-made islands, have led to increased predation opportunities (Antolos et al 2005, Evans et al. 2012, Hostetter et al. 2015, Moore & Berejikian 2022). For OP steelhead, given there are no large dams or barriers, it seems unlikely that the level of predation would have increased from man-made barriers.  There is the possibility that predation has increased given the increase in pinniped predator populations, but we have no specific long-term quantitative information for OP steelhead. Also, the extent of predation on steelhead in the ocean is largely unknown.

## D. Inadequacy of existing regulatory mechanisms

Regulatory mechanisms related to habitat protection and restoration may be inadequate as there continues to be habitat modification and legacy impacts of past habitat modification that are likely impacting OP steelhead. However, progress towards habitat protection is hard to measure as any ongoing efforts related to habitat restoration may take decades (if not longer) to show an effect. Also, there are many existing regulations that provide a generalized protection of freshwater/salmonid habitat, but none specifically directly at steelhead. These include both federal and state forest management plans and here we detail a few of the major existing mechanisms. The Northwest Forest Plan (NWFP) has guided the management of 17 Federal forests in addition to Bureau of Land Management (BLM) lands in the U.S. Pacific Northwest. The Aquatic Conservation Strategy (ACS) part of the NWFP, a regional scale aquatic ecosystem conservation strategy, ensures that Federal land management actions achieve a set of nine ACS objectives, which include salmon habitat conservation. Over 2564 $km^2$ (990 miles $^2$) of the Olympic Peninsula are part of the Olympic National Forest (ONF) (Halofsky et al. 2011). Within the ONF, management is guided by the land and resource management plan (LRMP) which was amended by the NWFP. The ONP created a General Management Plan in 2008 (NPS 2008). This plan set desired outcomes for the Park over the course of the 15-20 years and established management zones within the ONP with goals for resource conditions within those zones (see summary in Halofsky et al. 2011). Additionally, OP steelhead may benefit from the existence of protections for ESA-listed Bull Trout (*Salvelinus confluentus*), Northern spotted owl (*Strix occidentalis caurina*), or marbled murrelet (*Brachyramphus marmoratus*) and associated critical habitat for these species within the OP.

A retrospective on 25 years of the NWFP (Spies et al. 2019) reviewed the scientific literature published since the inception of the NWFP and reports several key findings. It has protected remaining old-growth forests from clearcutting and enabled growth and development of vegetation conditions to support threatened species, including salmonids and riparian-associated organisms (Spies et al. 2018). While the number of ESA-listed salmonid species and population units has increased, the pace of passive restoration, particularly in the face of climate perturbation, is insufficient to improve productivity at a rate necessary to achieve recovery. In addition, existing data are insufficient to determine whether basic survey and management criteria are met, and, management on federal lands alone without parallel efforts on non-federal land is not sufficient to achieve recovery (Reeves et al. 2018).

Numerous Washington State regulations also influence steelhead populations in the Olympic Peninsula DPS. The Forest Practices Act in Washington as well as the Washington State Forest Practices Rules (Title 222 WAC) establish rules and guidelines for forest management on non-federal land in Washington State, and that those lands are to be "managed consistent with sound policies of natural resource protection" (RCW 76.09.010 https://apps.leg.wa.gov/RCW/default.aspx?cite=76.09). Washington State Department of Natural Resources (DNR) states that these rules, "are designed to protect public resources such as water quality and fish habitat while maintaining a viable timber industry" (https://www.dnr.wa.gov/about/boards-and-councils/forest-practices-board/rules-and-guidelines/forest-practices-rules). The statute (RCW 76.09) and the implementing rules and guidelines (WAC 222) govern forest practices on all private forest lands in Washington as well as all non-DNR state-owned forest lands irrespective of ESA listings. Additionally, these protections are monumented in NMFS's Habitat Conservation Plan (HCP) Biological Opinion (NMFS 2006b).

In addition to protections on private and non-DNR state-owned forest lands, DNR's Habitat Conservation Plan (WADNR 2007) addresses compliance with the Federal ESA on state trust lands (NMFS 1997). The HCP covers approximately 1.9 million acres of DNR-owned forest lands within the range of the northern spotted owl, which includes all of the Olympic peninsula. The Department of Ecology has instream flow and water management rules to implement state law requiring that enough water is kept in streams and rivers to protect and preserve instream resources and values such as fish, wildlife, recreation, aesthetics, water quality, and navigation. In 2015, the Washington state legislature created the Fish Passage Barrier Removal Board ((Revised Code of Washington (RCW) 77.95.160) to establish a new statewide strategy for fish barrier removal and administering grant funding available for that purpose.

Other than habitat regulatory mechanisms, regulations related to harvest and hatcheries within Washington State affect the viability of OP steelhead. For background on salmonid fisheries regulations in Washington state and based on the Pacific Salmon Treaty, see the summary in Duda et al. (2018). More recently, the State of Washington has proposed, but not yet implemented, the 2022 WDFW Coastal Steelhead Proviso Implementation Plan (CSPIP) (Harbison et al. 2022). This plan outlines management strategies for the future of OP steelhead as well as other coastal steelhead populations. This was proposed to be partially funded by the Governor, but was not ultimately funded in the Governor's 2024 supplemental budget. The State is pursuing other funding that could begin July 2025. The Proviso Plan is based on existing state policies and does not represent a change in policy. It was developed from the recognition of recent declines in coastal steelhead and the need for adaptive management strategies to address these declines. Additionally, WDFW notes in the Proviso that region-specific Management Plans, including those for the Olympic Peninsula DPS, have yet to be developed (but are planned). The Proviso provides an implementation strategy for addressing monitoring and evaluation, hatchery operations, fisheries, habitat, and human dimensions, but notes that the lack of crucial data is a limiting factor in management of these populations. Specifically, the Proviso Plan identified sport fishery monitoring related to in-season management, summer steelhead monitoring and data collection (including genetic data), SONAR monitoring for more escapement monitoring, marine survival research including estimating smolt/juvenile survival and abundance, and developing tools to link habitat restoration activities and fisheries management as important research needs. Many responses to the 90-Day finding notice on OP steelhead were from fishers who reported that they were not frequently subjected to creel surveys and that recreational fishing monitoring was therefore inadequate. Many of the management deficiencies identified have been known for some time. For example, Busby et al. (1996) specifically identified the near absence of information on summer-run steelhead abundance and status in the Olympic Peninsula, and this situation remains unchanged to date. We also note that the proviso plan is only focused on recreational harvest and state hatchery operations and does not include the Tribal component of harvest nor tribal hatcheries, which currently constitutes the majority of landed catch and hatchery production.

A summary document on Traditional Ecological Knowledge (TEK) provided by the Makah for this status review provides helpful context on fisheries management and biases of certain historic data (Martin 2023). The document from Makah notes that sustainable harvest management is a core principle of traditional resource management and embedded into Tribe societal roles, salmon and steelhead have been managed since time immemorial (including their habitat) and this management included both traditional hatchery and harvest practices. They also highlight that historical documents on harvest from the 1950s-1970s were prepared by non-Tribal entities and contain biases and limitations; not adequately representing historic conditions and biases in reporting of fish. They note that "historical data" may not be reliable. We mainly focus on data since 1996, but note this context for any consideration of more historical data or management information.

Olympic Peninsula rivers support economically important sport fishing, as well as Tribal commercial, ceremonial, and subsistence gill-net fisheries for Pacific Salmon and steelhead. Summer and winter steelhead are collectively managed by WDFW and Treaty Tribes in the Boldt Case Area and also by the Olympic National Park (ONP). WDFW has jurisdiction over recreational fisheries in Washington state waters and outside of the ONP and tribal reservation boundaries. The Treaty Tribes regulate commercial and subsistence gill-net and on-reservation sport and tribal-guided fisheries. ONP has exclusive federal jurisdiction to manage recreational fisheries within the park boundaries. For winter run steelhead, on the Olympic Peninsula in 2016, WDFW changed the recreational fishing regulations to prohibit retention of natural-origin winter-run steelhead in the state waters of OP steelhead river basins. Steelhead fisheries in Olympic National Park are managed for catch and release except for retention of 2 hatchery-origin fish. Additional strategies have been implemented since the 1990s to support sustainable fishing including harvest restrictions (such as bag limits), shorter seasons, and gear restrictions in the face of declining wild steelhead populations (Harbison et al., 2022). In recent years, recreational fisheries have been closed inside and outside of the ONP for certain rivers (see below) due to low returns. Tribal fishing seasons have been shortened in certain recent years as well (depending on river system, see listing Factor B). As noted in factor B, reductions in harvest rates, with large reductions in Tribal harvest rates, have occurred in recent years (2021, 2022). Other regulations related to prohibiting bait, limits on hooks, size limits etc. are listed in Appendix 12.4 of Harbison et al. (2022). Harbison et al. (2022) note that recreational fisheries on tribal lands for the Queets and Quinault do not prohibit the retention of natural-origin steelhead. Additionally, hatchery steelhead released Queets and Quinault rivers are mostly not marked. State regulations allow for retention of steelhead with a dorsal fin height of less than 2 1/8 inches, the height of a credit card, the so named the "credit card rule", because hatchery fish are assumed to have eroded dorsal fins. Finally, for most rivers along the Strait of Juan de Fuca steelhead-directed harvest has been prohibited since the late 2000s/2010s, depending on the river.

Currently, the OP steelhead fisheries are mainly managed for escapement goals for winter-run steelhead based on freshwater productivity (see Gibbons, Hahn and Johnson 1985). Goals are set based on maximum sustainable harvest, which became a priority after U.S. vs. Washington (Boldt decision - Tribes and state will co-manage fisheries and Tribes have the right to half the catch). More specifically, for the term "escapement goal," Harbison et al. (2022) states for WDFW that "In this instance, it refers to the approximate number of fish needed to escape from fisheries to provide enough spawners to perpetuate the run for future generations at maximum sustainable yield (MSY)." Before the Boldt decision, harvest was managed to ensure sufficient returns to the hatcheries for production purposes without regard to returning natural origin fish; WDFW notes that "managers assumed that enough wild fish made it past the fishery to spawn," or in some cases redd counts or abundance counts at dams were used for monitoring and management (see Harbison et al. 2022). Given the lack of data on spawners and recruits for specific watersheds, Gibbons, Hahn and Johnson (1985) developed a Potential Parr Production model to estimate the number of steelhead offspring possible based on habitat, and used this within a modified Beverton-Holt model to determine escapement goals at MSY. Further, while Gibbons et al. is the basis for escapement goals there is some disagreement among co-managers on the escapement goals for some basins (see Table 4 in the Status Review report). For the Queets escapement, the Tribal escapement goal differs from that used by the State. The number of spawners needed for maximum sustainable yield ($S_{msy}$) calculated separately in the 1980s to be 2,500 but with the caveat that more data was needed (Scott, J.B. OP steelhead follow-up questions. Email to Laura Koehn. 17 July 2024). In the late 1990s, $S_{msy}$ was recalculated based on the best estimate of the stock-recruit relationship (Ricker curve) to be 2,700 with a highest probability range of 2,500-2,900. WDFW has yet to reevaluate these escapement goals and the assumptions from Gibbons et al. upon which they are based. WDFW has stated their intention to recalculate escapement goals based individual population models within a management strategy evaluation framework (Harbison et al. 2022).

With the escapement goals and foundation of Boldt, each year the State and the Tribes agree to yearly management plans that detail harvest of natural-origin and hatchery-origin OP steelhead for the upcoming fishing season. These plans consider forecasted returns and escapement goals to set harvest rates. In certain years and depending on the system, escapement goals are not met (see Factor B above). This may be due to errors in projected returns. The co-managers did state in their 2023 submittal to the BRT that, "Tribal fisheries are generally shaped by time and area restrictions with in-season management based on monitoring of fishery catches," so there is some in-season evaluation of the run relative to forecast (Co-Manager Olympic Peninsula Steelhead Working Group 2023) and seasons have been shortened/closed early in recent years in response to monitored catches (see Listing Factor B). Additionally, differing escapement goals (e.g. Queets River) may lead to harvest rates that result in adult returns below the escapement goal, depending on if the State or Tribal escapement goal is considered. Therefore, in certain years and certain systems, projected abundance may be below a certain escapement goal and therefore harvest may not be at MSY and escapement levels may not be at the level to maximize future returns. Note that the information on meeting escapement goals we have is for the major four systems and we do not present information on meeting escapement for rivers along the Strait of Juan de Fuca. For more on harvest that has occurred see Factor B presented above and section *Harvest Rates* above.

Escapement goals and MSY are not directly related to extinction risk, but failure to meet escapement goals suggests a management deficiency or an underlying biological factor that may represent a potential risk to the DPS. In the face of a declining run size, it is unclear if current management goals and strategies will allow for restoration of the runs.

Regulatory mechanisms are very limited for summer-run steelhead. There are no established management goals between Washington State and Treaty Tribes for summer-run steelhead. As referred to above, WDFW's 2022 Proviso specifies critical research needs including summer-run steelhead monitoring and data collection (many of these needs were also identified by Busby et al. 1996). Similarly, Cram et al. (2018) noted that there was insufficient data for all summer-run populations to assess trends or extinction risk. In 1992, WDFW and ONP implemented catch-and-release fishing regulations for summer steelhead (which still results in some harvest mortality). There are no directed commercial gill-net fisheries for summer steelhead in the DPS. The Treaty Tribes develop annual regulations for sport fishing on-reservations and those regulations include daily limits for steelhead that are caught during summer months. Time-series estimates of harvest for summer steelhead are provided above (section Summer-run Steelhead Population Harvest).

WDFW operation of hatcheries is currently regulated by the Statewide Steelhead Management Plan (SSMP) and the Anadromous Salmon and Steelhead Hatchery Policy C-3624 (2021), superseding the policy from 2009 (Hatchery and Fishery Reform Policy C-3619). However, the state and Tribal co-managers are currently working to develop Hatchery Management Plans for hatchery facilities within the Olympic Peninsula (Harbison et al. 2022). Furthermore, the state Coastal Steelhead Proviso Plan (Harbison et al. 2022) aligns with the existing policies, and hatcheries on the west coast are primarily operated for harvest augmentation. We outline current potential impacts of hatcheries below (in Listing Factor E), noting: (1) the use of out-of-DPS origin broodstock, (2) not all hatchery fish are adipose fin clipped, and (3) possible current levels of proportion of hatchery-origin adults spawning (pHOS) with natural origin steelhead that are above desired levels.

## E. Other natural or man-made factors

Other natural or man-made factors that are impacting OP steelhead include (1) hatchery impacts; (2) climate change; and (3) competition among salmonid species.

### Hatchery Impacts

Extensive hatchery programs have been implemented throughout the range of West Coast steelhead. While some programs may have succeeded in providing harvest opportunities and increasing the total number of naturally spawning fish, the programs have also likely increased risks to natural populations. Hatchery programs can affect naturally produced populations of salmon and steelhead in a variety of ways, including competition (for spawning sites and food) and predation effects, disease effects, genetic effects (e.g., outbreeding depression, hatchery-influenced selection (i.e., domestication)), broodstock collection effects (inadvertent selection for run timing or size, or limited numbers of broodstock), and facility effects (e.g., water withdrawals, effluent discharge, blocked streams) (Rand et al. 2012, HSRG 2014, Ohlberger et al. 2018, McMillan et al. 2023), as well as masking of trends in natural populations through the straying of hatchery fish. Additionally, hatchery effects can include reduced genetic diversity and reproductive fitness through interbreeding. Recent research suggests that hatchery introgression can reduce variation in run timing and even despite reduced fitness of hatchery fish, hatchery alleles can quickly assimilate into natural populations (May et al. 2024). State natural resource agencies have adopted or are developing policies designed to ensure that artificial propagation is conducted in a manner consistent with the conservation and recovery of natural, native populations. The role of artificial propagation in the conservation and recovery of salmonid populations continues to be the subject of vigorous scientific research.

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-origin fish while integrated programs incorporate natural-origin broodstock (Harbison et al. 2022). In order to reduce risks from hatcheries, the WDFW Statewide Steelhead Management Plan (SSMP) and the former Hatchery Scientific Review Group (HSRG) (an independent scientific panel that reviewed Pacific Northwest hatcheries), set thresholds of allowable levels of proportion of hatchery origin spawners spawning naturally (pHOS) for segregated programs as well as proportion of natural influence (PNI) for integrated programs. A further consideration in the development of integrated and segregated hatchery programs, is the source of the founding broodstock for the hatchery, and whether it represents the native population, or comes from outside of the basin, or outside of the region.

In the NMFS 1996 review (Busby et al. 1996), NMFS noted the estimated proportion of hatchery stocks on natural spawning grounds ranged from 16 to 44 percent. This proportion was lowest for the two rivers with the largest production of natural-origin steelhead - Queets and Quillayute Rivers. At the time, according to Busby et al. (1996) pHOS level was 43% for the Pysht River, 16% for the Quillayute River, 19% for the Queets River, 44% for the Quinault River, and 37% for the Moclips River. As noted in the status review, more recently, the Washington Coast Sustainable Salmon Partnership (WCSSP, 2013) estimated the proportion of hatchery-origin adults that were naturally spawning in Olympic Peninsula DPS basins based on the professional opinion of local biologists. In general, smaller basins with hatchery programs (Tsoo-Yess River, Goodman Creek) and the Quinault River were thought to have higher pHOS levels (26-50%), with other basins less so (>25%); although a number of basins were not reported. Most summer-run steelhead pHOS is unknown, however the following website was reported by the petitioners from WDFW[22] which shows that for 2009, pHOS for summer-run steelhead for the hatchery program on the Bogachiel River were 23% and 9% for winter-run.

Scott and Gill (2008) showed gene flow of early Winter steelhead from Chambers creek stock into the Hoko, Pysht, and Sol Duc rivers, (5.5-14.5%, 12-75%, and 2.5-6% gene flow respectively). This led to elimination of winter steelhead smolt release into the Pysht river in

[22] (https://fortress.wa.gov/dfw/score/score/hatcheries/hatchery_details.jsp?hatchery=Bogachiel%20Hatchery)

2009, as well as Goodman Creek, Clallam River, and Lyre River. In 2012, the Sol Duc River was designated by WDFW as a Wild Stock Gene Bank, resulting in the cessation of summer smolt releases in 2011 and winter in 2013 (winter-run was local-origin broodstock steelhead) (see Hatchery regulations above).

A recent review by Marston and Huff (2022) looked at the compliance by the WDFW operated Bogachiel Hatchery with standards set in the SSMP. This report also summarized existing hatcheries and then looked at compliance of WDFW operated programs. They found that stray rates by steelhead from Bogachiel-origin programs are unknown; for early winter steelhead they modeled – 6% of hatchery fish spawning in the overlap period when natural-origin fish are spawning, and for summer steelhead – less than 1% of hatchery fish spawning in the overlap period with natural-origin fish. Marston and Huff (2022) recommended assessing the status, spawn timing, and spatial distribution of summer natural-origin steelhead, and also re-evaluating the March 15$^{th}$ hatchery origin/natural origin spawner cut-off date, amongst other recommendations. Recommendations also included specifics for discontinuing or continuing programs and how to manage them.

The recent review of OP steelhead from WDFW (Cram *et al.*, 2018) also identified hatchery operations as "a threat to genetic integrity of wild steelhead populations" in the area occupied by OP steelhead. Cram *et al.* (2018) stated that, as of 2014, there were 11 hatchery programs on the Olympic Peninsula with an average annual release of 1,393,022 smolts from 2000 to 2008 and 1,072,781 from 2009 to 2013. Most hatchery programs (10 of 11) are used for harvest augmentation and most of these were founded by one of two steelhead populations not native to the Olympic Peninsula – Chambers Creek early winter-run steelhead (Puget Sound) or Skamania early summer-run steelhead[23] (Columbia River). Of the hatchery programs in the Olympic Peninsula, five are off-site release programs that transfer smolts from their hatchery to another watershed for release. Cram *et al.* (2018) notes that if returning adults from these programs are not caught by fisheries, they place natural-origin OP steelhead at risk genetically and ecologically. An integrated hatchery program was initiated in the Bogachiel River in 2013 using hook and line caught natural-origin broodstocks, but has since been discontinued, additionally the program on the Sol Duc River ended and steelhead in that river are now managed as a "Wild Steelhead Gene Bank" (Cram *et al.*, 2018).

Above in the section *Hatchery Operations in the Olympic Peninsula Steelhead DPS* we summarize the hatchery programs and hatchery outputs. Hatchery releases have stayed consistent since the late 1970s/early 1980s to the present both for winter-run and summer-run hatchery output. Smolt output depending on the run timing, river, and year can range from <10,000 to >700,000. Additionally, see NMFS (2024a) on Watershed specific information for specific hatchery output for individual systems.

Hatchery-origin winter-run steelhead return migration overlaps with the historical early-run timing of natural-origin winter-run steelhead, so there is likely exposure of the early-run timing to hatchery influence (McMillan et al. 2022). Additionally, McMillan et al. (2022) hypothesize

[23] The use of the Skamania Hatchery and Chambers Creek Hatchery stocks has been eliminated elsewhere on the West Coast due to negative impacts on listed steelhead, see Ford *et al.*, 2022.

that commercial and recreational fisheries targeting hatchery-origin steelhead with early run-timing are harvesting early-run natural-origin steelhead as well, potentially creating directional selection against early run-timing given that run-timing is a heritable trait.

Martin (2023) indicates that stock transfers between watersheds was part of traditional tribal fisheries management. Such movements, most likely between adjacent watersheds, would be akin to returning adult steelhead straying, and would not represent the same level of genetic risk as the cumulative release of millions of steelhead from Puget Sound or the Columbia River hatchery stocks.

Climate Change

Major ecological realignments are already occurring in response to climate change (Crozier et al. 2019). In Washington State, further increases in freshwater temperatures for salmon streams are predicted in addition to large shifts in seasonal hydrology (Climate Impacts Group 2009). Projected changes in climate for the Olympic Peninsula were summarized in (Halofsky et al. 2011), (Dalton 2016), and the 2020 State of Our Watershed Reports from Northwest Treaty Tribes (NWIFC 2020). NWIFC (2020) summarizes potential climate change impacts within the Olympic Peninsula stating, "the observed and projected trends include warmer air temperatures; shrinking glaciers and snowpack; lower summer stream flows; higher winter flood flows; shifts in streamflow patterns and timing; higher stream temperatures; larger and more frequent wildfires; warmer ocean temperatures; rising sea levels; and changing ocean chemistry, including ocean acidification and lower levels of dissolved oxygen." On the OP, warming has already occurred, and is projected to continue during all seasons, with the largest increases during summer. Projected decreases in precipitation in summer in combination with increased summer evapotranspiration will further impact stream flows for both juvenile and adult steelhead. Additionally, increases in winter precipitation quantity but also increases in the intensity of events in the western portion will likely result in redd scouring and habitat degradation (see Halofsky et al. 2011 and references therein). Changes in precipitation and timing of peak streamflow may lead to increased runoff and flood risk, with greater frequency and magnitude of flooding. Warming is likely to reduce snowpack (less winter snow accumulation) which would in turn decrease the risk of floods in springtime. The biggest changes in streamflow are projected where rivers originate from the Olympic Range; where snowpack is likely to decline rapidly, especially for areas that will likely transition from a mix of rain/snow to rain dominated with warming (Yoder and Raymond 2022). Specifically, model projections show up to 30% decline in average summer flow in reaches of low intrinsic potential (<20% in medium to high intrinsic potential) by 2040 (Reeves et al. 2018), and average winter flows of at least 30% higher (Reeves et al. 2016;2018) (Safeeq et al. 2015). Multiple papers have already documented extensive glacier losses (Fountain et al. 2022; Riedel et al. 2015, NWIFC 2020).

Many of these changes have already been observed on the OP. On USFS land within the OP, there has been a decrease in wetted bank extent and increases in August temperatures from <14 °C in 2002 to around 14-18°C in the late 2010s, with data ending in 2018 (Dunham et al. 2023). Additionally, WDOE stream temperature data from Sol Duc shows warming water temperatures in April and May in certain years recently [24]. Peak flows (in winter) have already increased

[24] Washington Department of Ecology. 2023. Freshwater DataStream,

while summer low flows have already decreased. An assessment of peak flood flows between 1976 and 2019 found that peak flows have increased for the Hoko, Hoh, Calawah, and Quinault Rivers, by 5% to 18% with the Hoh River increasing by 18.4% (NWIFC 2020). In both the Calawah and Bogachiel rivers, it is becoming common for peak flows to be at or above flood stage. Examination of the peak discharges for the OP watersheds draining to the Pacific found that the two-year flood event is 10 to 35% greater over the last 40 years, relative to over the entire length of the stream-gage record (East et al. 2017). In the Hoh River basin, the three largest peak flow events recorded have occurred since 2002 (East et al. 2018). The 2-year flood peak calculated for the Hoh River for water years 1978–2013 was 1024 cms, whereas the 2-year flood for the entire period of record at the Hoh River gaging station (12041200) was 924 cms (East et al. 2018). Hoh, Queets, and Quinault rivers have all widened since 1970 consistent with greater flood activity, and Hoh is showing greater braiding likely related to increased sediment loads from retreating glaciers (East et al. 2017). The general increase in flood activity along the OP after the mid-1970s coincided with the onset of a wet phase of the Pacific Decadal Oscillation (PDO, an index of monthly sea-surface temperature anomalies over the North Pacific) (Mantua et al. 1997). This mid-1970s climatic transition has been identified as a major atmospheric and hydrologic shift that affected a large region of the Pacific in both the northern and southern hemispheres (Castino, Bookhagen and Strecker 2016; East et al. 2018). Summer low flows have decreased over time in the Calawah River basin, where the average low flow in the in late 1970s through the 1990s was 2.0cms, while in the 2000s average summer low flow has been 1.5cms.

One of the largest predicted changes, with respect to changing climatic conditions, is the decline in glacial extent (Riedel et al. 2015), particularly for the larger west side watersheds. Over the past several decades, glacier decline in the Olympics was greater than in the Cascades and southern Coast Mountains, and is more comparable with Vancouver Island (Riedel et al. 2015). Riedel et al. (2015) estimate that the glacial contribution to summer streamflow has declined ~20% in the past 30 years, but still remains significant for the Hoh River. In the other westside OP DPS watersheds, glaciers contribute less than 5% to summer streamflow (Riedel et al. 2015). The loss in glaciers over the past 30 years appears to be a result of mean air temperature increases, and illustrates how sensitive these relatively small, thin, and low-elevation glaciers are to climate change (Riedel et al. 2015, East et al. 2018). Continued loss of glaciers will directly impact aquatic ecosystems through higher stream temperatures and lower summer base flows.

Using stream temperature and flow data from the USDA and USFS Rocky Mountain Research Station[25], the SRT reviewed projected changes in temperature, flow, and 25-year flood cfs for individual rivers/streams between now and 2040 and now and 2080 (NMFS 2024b). Changes in summer flow may be dramatic, with declines as large as -70% in summer seasonal mean flow between now and 2080 and mean temperatures may reach near 20°C for certain rivers. Changes in summer flow are more likely to affect returning and holding summer-run steelhead, although juvenile and adult winter-run steelhead in the Upper Quinault and Queets rivers and Salt-creek independents tributaries may also be affected. The highest temperatures experienced now and

https://apps.ecology.wa.gov/ContinuousFlowAndWQ/StationDetails?sta=20A070; provided in a public comment on the 90 day finding from The Conservation Angler and Wild Fish Conservancy

[25] (https://www.fs.usda.gov/rm/boise/AWAE/projects/NorWeST/ModeledStreamTemperatureScenarioMaps.shtml, https://www.fs.usda.gov/rm/boise/AWAE/projects/modeled_stream_flow_metrics.shtml)

likely into the future are predicted to also impact the Lyre winter-run and Clearwater summer-run populations.

For OP steelhead, increases in summer stream temperatures may especially pose risks to juvenile steelhead that spend up to two or three years in freshwater (Halofsky et al. 2011). Adult summer steelhead require cool water holding pools which may be less available with warming temperatures, resulting in higher mortality and/or lower reproductive success (Dalton et al. 2016). Low summer stream flows may affect summer-run steelhead migration by dewatering stream reaches or limiting the accessibility of waterfall or cascades (Halofsky et al. 2011). Future increases in flows at other times of year may displace juvenile fish and/or reduce the availability of suitable slow-water habitat for young fish. However, winter-run steelhead generally spawn after peak flow events and may be less susceptible to their redds being scoured (Halofsky et al. 2011). Still, future changes in streamflow could increase overall stream scouring, impacting eggs and embryos, while warmer temperatures may result in more rapid incubation leading to smaller individuals at emergence (Dalton et al. 2016). Authors note that salmon fry in low gradient streams may be less vulnerable to displacement from high winter stream flows than fish that emerge later in the year in steeper streams (such as summer steelhead) (Dalton et al. 2016). Changes in flows and temperatures could also impact smolt migration timing (Dalton et al. 2016). The Climate Impacts Group (2009) highlighted that salmonids with extended freshwater rearing such as steelhead may experience particularly large increases in temperature and hydrologic stress in summer (from stream temperature increases and lower stream flows), that may result in lower reproductive success. There may be positive impacts from climate change as well, mainly possibly longer growing seasons due to temperature increases, increased productivity within the food-web, and more rapid growth at certain times and life stages (Halofsky et al. 2011, Dalton 2016). Specifically, warmer conditions in summer would likely reduce growth but warmer at other times of year could increase growth rates (Dalton et al. 2016); however, warmer temperatures also potentially increase competition with other species (or predation), through the increased presence of non-native piscivorous species, as well as an increased susceptibility to disease as well.

Within the 2020 State of Watershed Report, the Northwest Treaty Tribes explain that the overall increase in stream temperature leads to salmon being exposed for longer to temperatures outside of their ranges for reproduction and survival NWIFC 2020). Further, increased temperatures along with changes in streamflow lead to lower dissolved oxygen, increased sediment, higher disease susceptibility, competition with other species, and variation in prey for salmonid species. Many of the individual watershed/Tribal reports in the State of Our Watersheds Report note impacts of streamflow and temperature changes on salmon productivity and survival. Within the Quileute report, they note that warmer stream temperatures may lead to accelerated growth and early emergence as well as hydrology impacts on smolting and migration, with overall negative impacts on reproductive success.

A new Climate Adaptation Framework by the Coast Salmon Partnership looked at the resilience to climate change of salmon watershed habitats along the Washington coast (https://www.coastsalmonpartnership.org/current-initiatives/climate-framework/). This work includes a tool to explore the resiliency of various watersheds - https://coast-salmon-partnership.shinyapps.io/CRI_app/. Overall, most of the watersheds on the coast in the OP steelhead DPS range were found to have higher overall resiliency to climate change than watersheds further south. But, certain watersheds in WRIA 20 had lower resiliency, mainly due to metrics around summer low flows. Though this work was made public after the status review teams finalized scoring for the risk assessment, it corroborates that low summer flows will likely impact certain streams within the DPS but there also may be some areas where climate change will be less impactful. See the user guide for the tool (Adams and Zimmerman 2024) for more information on the metrics used.

At the population level, the ability of organisms to genetically adapt to climate change depends on how selection on multiple traits interact, and whether those traits are linked genetically. Factors that affect genetic diversity can thus limit the ability of a population to adapt to climate change. These include, but are not limited to small population size, domestication in hatchery environments, or introgression by introduced non-native stocks. Though populations may be able to adapt to changes if within the range of what they have experienced historically (Waples, Pess and Beechie 2008), it is unknown if Olympic Peninsula steelhead can adapt quickly enough to the rapid pace changing climate and habitat. Further, some SRT members were concerned that diversity loss in some populations will limit their ability to adapt to a changing environment.

Dalton et al. (2016) state that climate change driven changes in freshwater ecosystems will be relatively small by the mid-century, but that additional changes and challenges may occur in the marine environment. A study by Abdul-Aziz, Mantua, and Myers (2011) predicted an 8 to 43 percent contraction of steelhead species' marine habitat due to climate change between the 2020s and 2080s (depending on time period). As stated in the NOAA 2020 Status Review Update (Ford 2022) report – "Historically, ocean conditions cycled between periods of high and low productivity. However, global climate change is likely to disrupt this pattern, in general, leading to a preponderance of low productivity years, with an unknown temporal distribution (Crozier et al. 2019a)."

The assessment by the Co-Manager Olympic Peninsula Steelhead Working Group (2023) suggested that interannual variation in recruitment and kelt survival were both partially explained by summer sea surface temperatures (SST), and also pink salmon abundance; as well as North Pacific Gyre Oscillation for recruitment. In other words, this analysis showed a negative correlation between recruitment and summer SST and a negative correlation between kelt survival and summer SST. Work by Kendall et al. (2017) showed variability in smolt survival consistently for Washington coast and Strait populations (but with less magnitude fluctuations for Washington Coast, on average). There is uncertainty in how smolt survival and recruitment and kelt survival will change overtime but, kelt survival has already declined since the 1980s (see above in section *Repeat Spawner Rate*, Figure 19) This analysis strongly suggests that ocean survivals are likely to decrease in warm years and the frequency of these warm years will increase with climate change.

Competition

OP steelhead may also be affected by competition with other salmonids, particularly Pink salmon. Ruggerone and Nielsen (2004) summarized literature on competition between pink salmon and other salmonids and discussed that pink salmon alter the prey abundance of other species (such as abundance of zooplankton, squid), and that this can then lead to an altered diet, reduced consumption, reduced growth, delayed maturation, and reduced survival depending on the salmon species and location. However, some steelhead specific studies showed that greater abundance of spawning pink salmon can provide greater prey (in the form of pink fry or eggs) to steelhead, including pink salmon eggs enhancing steelhead parr growth and survival. Additional papers have looked at possible connections between pink salmon abundance and other salmonid growth and survival (Ruggerone and Irvine 2018; Ruggerone et al. 2023). As mentioned above, the assessment by the Co-Manager Olympic Peninsula Steelhead Working Group (2023) suggested that interannual variation in recruitment and kelt survival were both partially explained by Pink salmon abundance (and also SST; as well as North Pacific Gyre Oscillation for recruitment). In other words, this analysis showed a negative correlation between recruitment and Pink salmon abundance and a negative correlation between kelt survival and Pink summer abundance. We note that the co-manager analysis however did not sufficiently consider impacts of pinniped predation on kelt survival or smolt survival because of a lack of data for seal/sea lion (pinniped) abundance (shorter time series compared to other factors) and so there is still uncertainty about impacts of predation on survival for steelhead.

## Threats Overview

NMFS last reviewed the status and risk of OP steelhead in the 1996 report, Busby et al. (1996). At that time, the SRT concluded that the "Olympic Peninsula steelhead DPS [ESU] is neither presently in danger of extinction nor likely to become endangered in the foreseeable future." Despite this conclusion, the SRT had several concerns about the overall health of this DPS [ESU] and the status of certain stocks within it related to downward trends in abundance, uncertainty around abundance (especially for summer-run steelhead), and potential impacts of hatchery production and introgression given the use of few parent stocks (see *Previous Risk Assessment and SRT Process* section above).

Since that time, progress has been made to address certain threats. For instance, habitat restoration projects have occurred including the replacement of many culvert barriers (see NWIFC 2020), and Coast Salmon Partnership 2022 Annual Report[26]) in recent years and installation of large wood jams in selected rivers. Additionally, habitat connectivity continues to be maintained in the major river systems largely due to the absence of major blockages. More stringent State and Federal sport fishing regulations have gone into place including catch-and-release restrictions for recreational fishing (since 2016) and area and gear restrictions for natural-origin summer and winter steelhead. Additionally, harvest of steelhead stopped in 2000s/2010s for most rivers on the Strait of Juan de Fuca. More regulatory mechanisms have been established that impact salmonid habitat broadly including: Habitat Conservation Plans that address timber harvest, Northwest Forest Plan and associated Aquatic Conservation Strategy, Land and Resource Management Plan for the Olympic National Forest, Washington Streamflow Restoration law and Fish Passage Barrier Removal Board, 2008 Statewide Steelhead Management Plan, Anadromous Salmon and Steelhead Hatchery Policy C-3624 (see Listing Factor D in NMFS 2024b). Hatchery practices have been modified to reduce off-station releases, in order to increase the proportion of fish returning to the hatchery rack and decrease the number

[26] https://coastsalmonpartnership.egnyte.com/dl/VbBakQwmdS

of hatchery-origin fish straying and spawning naturally[27]. Additionally, disease and predation remain aspects that impact this DPS, but there was a paucity of information in regards to disease in natural-origin populations and limited evidence of any increases in predation since the last review.

Other threats continue to be an issue for this DPS. Legacy impacts from past stream habitat modification were noted as a factor in 1996 and still continue. Although efforts are underway to address habitat issues, it may take decades to centuries for larger rivers to recover (Martens et al. 2019; Stout et al. 2018), especially related to woody debris (which may be most beneficial to steelhead, see Jorgensen et al. (2021)). Moreover, continuing climate change will further exacerbate conditions into the future (Wade et al. 2013). Climate change is currently impacting this DPS and will continue to negatively affect both the freshwater and marine habitat in which these populations reside. In the foreseeable future, projected and modeled climate impacts that may affect steelhead include: prolonged low summer flows, increased frequency and magnitude of winter peaks flows, elevated water temperatures, and continued loss of glaciers (and melt impacts stream flow) (Wenger et al. (2010); Wade et al. (2013); and see Listing Factor E). From a life history diversity perspective, kelt survival has continued to decline in the four major coastal rivers, possibly related to warmer sea surface temperature, pink salmon impacts, and Pacific Decadal Oscillations (but there is uncertainty about other potential contributing factors, including predation).

Furthermore, though harvest and hatcheries operations have been modified as described above, they continue to have an overall negative influence on steelhead populations within the DPS. Prior to 2021, Olympic Peninsula steelhead populations experienced relatively high commercial and recreational fishing pressure (when compared to other DPSs) even while populations declined. There are documented legacy and current impacts associated with harvest. Harvest rates were the highest in the state for the four major OP rivers (13.26%-59.19% depending on year and river between 2014-2020) which contribute the majority of OP steelhead abundance. Since 2016, recreational catch and release for natural-origin steelhead went into place for state and federal management areas, although there is still hooking mortality (10%) and some fish may be caught multiple times. The SRT did observe that in the last 2 years (2021, 2022), harvest rates in the major four OP steelhead basins have been only ~9-15% depending on the basin, but there is no certainty that these rates will remain low, and in certain basins even these reduced rates have resulted in abundances below escapement goals. It is unclear if escapements can be maintained in the future. At the same time, the proportion of harvest that is natural-origin has increased so it is likely that proportionally more natural-origin steelhead are being caught in fisheries that target hatchery-origin steelhead (discussed in section - *SRT assessment of winter-run run timing changes*). There is also evidence of a shifting run timing with later migration for natural-origin winter-run steelhead. Certain hatcheries have produced out-of-DPS stock origin smolts for decades and continue to do so (in the hundreds of thousands annually). Returning hatchery-origin adults overlap in return and spawn timing to some extend with natural-origin winter-run adults, resulting in harvest impacts and the potential for introgression. Finally, though there have been some positive management changes, there continues to be challenges associated with fisheries and hatchery management. Data limitations continue for assessing the current

[27] For example, winter steelhead smolt release into Pysht was eliminated in 2009; Goodman Creek, Clallam River, and Lyre river in 2009, and in Sol Duc, summer smolt releases were terminated in 2011 and winter in 2013.

status and risk of summer-run OP steelhead, an issue identified in the 1996 review and more recently by Harbison et al. (2022)). There continue to be undefined escapement goals for some rivers and differing escapement goals between co-managers for others, and uncertainty if the escapement goals can maintain or restore runs. Certain hatchery fish are not marked in some major rivers on the coast, and there are relatively high redd expansion factors due to challenges in estimating escapement during higher flows and remote terrain. Many threats to Olympic Peninsula steelhead identified by Busby et al (1996) continue today, although some efforts have been made to diminish their effects. However, new threats, such as climate change are beginning to affect steelhead populations in the Olympic Peninsula DPS, and will likely increase in intensity in the future.

# Risk Assessment

## Results and Discussion

### Previous Risk Assessment and SRT Process

In the coastwide steelhead assessment by NMFS (Busby et al 1996), the SRT concluded that the Olympic Peninsula steelhead DPS [ESU] is neither presently in danger of extinction nor likely to become endangered in the foreseeable future". Further, the SRT found

> *Despite this conclusion, the SRT has several concerns about the overall health of this ESU and about the status of certain stocks within it. The majority of recent abundance trends are upward (including three of the four largest stocks), although trends in several stocks are downward. These downward trends may be largely due to recent climate conditions. There is widespread production of hatchery steelhead within this ESU, largely derived from a few parent stocks, and this could increase genetic homogenization of the resource despite management efforts to minimize introgression of the hatchery gene pool into natural populations. Estimates of the proportion of hatchery fish on natural spawning grounds range from 16% to 44%, with the two stocks with the largest abundance of natural spawners (Queets and Quillayute) having the lowest hatchery proportions.*
>
> *These conclusions are tempered by substantial uncertainties. As for with the Puget Sound ESU [DPS], there is very little information regarding the abundance and status of summer steelhead in this region and the degree of interaction between hatchery and natural stocks.* (Busby et al. 1996, pg. 166)

### Risk Assessment

The current SRT has been similarly tasked to assess the status of the Olympic Peninsula Steelhead DPS. Members of the current SRT reviewed and discussed information related to the VSP parameters for individual populations and the DPS as a whole. The team's determination of overall risk to the Olympic Peninsula Steelhead DPS 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 Oregon Coast 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 DPS/ESUs (Ford 2022). They are defined as follows:

> *High Risk: a species or 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/DPS at such a high level of risk may be highly uncertain and strongly influenced by stochastic*

*and/or depensatory processes. Similarly, a species/DPS may be at high risk of extinction if it faces clear and present threats (e.g., confinement to a small geographic area; imminent destruction, modification or curtailment of its habitat, or disease epidemic) that are likely to create such imminent demographic risks.*

*Moderate risk: a species or 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/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 or DPS is more likely than not to become at high risk in the future depends on the 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 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: neither at high or moderate risk of extinction.*

The overall extinction risk determination reflected 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 team considered the foreseeable future as it relates to the moderate risk assessment to be a time period of 40-50 years (roughly ten steelhead generations). Beyond the 40-50-year time horizon, the projected effects on Olympic Peninsula Steelhead 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, review teams have used a “risk matrix” as a method to organize and summarize the conclusions of a panel of knowledgeable scientists. This approach has been used for over 20 years in Pacific salmonid status reviews (Myers et al. 1998; e.g., 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/DPS 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 team also considered the impacts of the environmental threats associated with the listing factors in ESA section 4(a). These include: a) habitat loss and degradation, b) over-utilization for commercial or scientific purposes, c) inadequate regulatory mechanism, d) disease and predation, and e) 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.

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

- Very low risk (1): It is unlikely that this factor contributes significantly to risk of extinction, either by itself or in combination with other factors.
- Low risk (2): 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 (3): 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 (4): 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 (5): This factor by itself indicates danger of extinction in the near future.

In some cases, detailed information was not available at the population level, and in these cases, scores were provided at the level of the entire DPS. The scores were reviewed, and the range of perspectives discussed by the team before making an overall risk determination. 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. For example, a DPS with a single extant sub-population might be at a high level of extinction risk because of high risk to spatial structure/connectivity, even if it exhibited low risk for the other demographic criteria. Another species might be at risk of extinction because of moderate risks to several demographic criteria.

After population-level risks were assessed, each team member assessed the risk of extinction (low, moderate, high) for the DPS 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 correctly reflects the true species status. Thus, if a member were certain that the species was 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 DPS.

**Assessing risk in a significant portion of each DPS’s range**

In addition to assessing the risk status of the entire DPS, the team also evaluated if there were *significant portions of the range* (SPOIR) within the DPS and, if so, were they at either moderate or high risk of extinction. In doing this, the team followed advice from the NMFS WCR and NMFS Office of Protected Resources on how to interpret the phrase “significant portion of its range” in light of the 2014 joint U.S. Fish and Wildlife and NOAA SPOIR policy (79 FR 37578) and subsequent legal rulings.

Based on this advice, this analysis involved identifying and evaluating portions the DPS that are potentially at moderate or high risk of extinction and are important to the overall DPS long-term viability, yet not so important as to be determinative of its current or foreseeable status. In other words, the goal of the SPOIR evaluation was to determine if there are important portions of the DPS that are currently at high or moderate risk, but that are not so important that their status leads to the entire DPS being currently at high or moderate risk. The rationale for this approach is to ensure that there is a clear distinction between a species (or DPS) that is at risk in throughout all of its range and one that is at risk in only a significant portion of its range.

The SRT discussed at length the application of the SPOIR policy, and how it suggested that if a portion was not significant it would not contribute to the immediate or long-term VSP viability status– essentially providing neither risk nor benefit to the DPS. In evaluating the VSP status of the entire DPS, the SRT affirmed the importance of incorporating all populations within the DPS and not just those that could be placed into identifiable SPOIRS. Simply, that populations were still important to the overall risk assessment, even if they were not in an identifiable SPOIR. The team considered and discussed several potential sub-DPS strata that would reasonably meet the criteria of being important to the DPS long-term viability but not so important that their status would drive current or foreseeable DPS-wide risk. After considering multiple possibilities, the team settled on a more detailed evaluation of two potential types of strata based on either geography or adult run-timing. These are discussed in turn below.

Geographic strata:

The Olympic Peninsula Steelhead DPS occupies three WDFW WRIAs, that occupy the Strait of Juan de Fuca (#19), and Washington Coast (#20 and #21). The SRT discussed using the WRIA watershed units as potential “Portions of the Range”, but ultimately decided that the two coastal WRIAs were geographically similar enough to combine. Rivers along the Strait of Juan de Fuca exhibit rain-dominated hydrographs, all draining to the Strait as relatively short rivers that drain low elevation hills. In contrast, the coastal watersheds are dominated by the four major rivers (Quillayute, Hoh, Queets, and Quinault rivers) with higher elevation headwaters that are glacially fed with rain/snow hydrographs. In addition, there are shorter streams that drain directly to the ocean, but these likely interact in a source/sink relationship with the larger rivers. Given the similar abundances in each of the major rivers in the Coastal portion, it was concluded that further division would not fulfill the definition of a "significant" portion.

Adult run-timing strata:

The team also considered whether the variation in adult run-timing might form the basis for identifying alternative portions. Olympic Peninsula steelhead exhibit two distinct life history forms with associated run times. Summer- and winter-run steelhead utilize different freshwater habitats, particularly during the adult freshwater migration and spawning portions of the life-cycle. Generally, summer-run steelhead spawn in the upper portions of river systems, sometimes above temporal flow barriers that are only accessible during high spring flows (Withler 1966; Myers et al. 2015; Waples et al. 2022). SRT concerns about the status of summer run were a major rationale for considering summering as a "portion". Further, the petitioners had highlighted the status and relative importance of summer-run steelhead.

For both the geographic and run-timing approaches to SPOIR, the SRT decided that each member independently evaluate whether the portion identified within the DPS was significant to the long-term viability of the DPS, quantified using the likelihood point method.

## VSP Criteria for Risk Assessment

### Abundance

#### Winter-run Steelhead

In their review of the status of the Olympic Peninsula Steelhead DPS, the SRT considered many different aspects of the information that was available. Escapement abundance was estimated via redd counts, and only those redds observed after March 15th were used in the estimate. This static cutoff date was apparently used by co-managers to ensure that the redd count only reflected production by naturally-produced fish. Steelhead spawn prior to the March 15th and naturally-produced (unmarked) steelhead contribute to this pre-cutoff date production Marsten and Huff 2022). It is also likely that some number of returning unmarked steelhead represent hatchery-origin adults; in the absence of directed genetic studies on this question this proportion is unknown. Alternatively, based on historical harvest data it is clear that native winter-run steelhead returning to rivers in the Olympic Peninsula exhibited a wide range of return timing – so there is little reason to discount the "native" origins of unmarked early steelhead – those spawning before March 15th. Overall, from an abundance perspective, current estimates of escapement likely underestimate natural production and early spawners may represent an additional 10% increase in overall abundance (Marston and Huff 2022). Changes in harvest effort and timing and the intensity, location, and timing of hatchery releases likely have an effect on the relative contribution of hatchery-origin and natural-origin spawners prior to March 15th.

Another effect of the post-March 15th redd count cutoff is the lack of any estimates of the percent hatchery origin spawners (pHOS) among the naturally-spawning steelhead and the potential for hatchery x native introgression. Although hatchery release practices had been modified over a decade ago to eliminate off-station releases, there is considerable uncertainty in the genetic risk to population diversity. While early-winter run hatchery steelhead females may generally spawn earlier than the native females, there is also a tendency for hatchery-origin males to remain on the spawning grounds for extended periods increasing the likelihood of hybridizing with native steelhead. Additionally, the continuation of early-run non-native winter steelhead programs to maintain harvest opportunities also results in the harvest of the early returning portion of the native steelhead population. Given that the recreational fishery is currently managed as no retention for unmarked (unclipped) adults, except in the Queets and Quinault rivers where the "credit card" rule[28] is applied to identify hatchery- origin steelhead in state and ONP waters, the majority of the natural-origin harvest is in the commercial fisheries. In tribal waters of the Queets and Quinault rivers, there is no distinction made between hatchery or natural fish in the tribal guide lead recreational fisheries. Although there was limited information provided to the SRT, it was clear that in most cases by late January the majority of the winter steelhead harvested in the commercial fisheries were of natural origin. The SRT was concerned that the current return timing has been affected by relatively high harvest rates during the early portion of the return timing (November to February) for native winter steelhead. Some members postulated that harvesting the early-returning natural steelhead may affect the spatial distribution of spawners; that the earlier returning steelhead spawners tend to spawn lower in the basin and that harvest may have an effect on spawning spatial structure. Further, this contraction in run timing may remove a run/spawn timing that could be more successful under climate change, with later returning spawners being subjected to higher stream temperatures.

There was some discussion about "historical" run sizes in individual rivers and across the DPS. While there is considerable uncertainty in historical estimates, the SRT did feel that the information submitted and independently assembled reflected that there had been a long-term decline in DPS-wide abundance. Further, the SRT recognizes that the "decline from historical levels" does not directly relate to the risk of extinction but, it does convey that there have been factors that precipitated this decline, and those factors may still be exerting an effect on abundance.

The decline in total winter steelhead run size observed in the four major basins in this DPS was a risk factor cited by members of the SRT. Combined escapement estimates for the four major rivers have decreased by 16% from 18,597 (1991-1995 e.g. the Busby et al. Status Review) to a current level of 15,653 (2018-2022); however, total run size had decreased 42%, from 32,556 (1991-1995) to 18,821 (2018-2022). Additionally, of the 14 populations for which adequate escapement data was available for trend analysis, 1 had a stable trend and 13 were negative (10 significantly so). Historical harvest levels set for these basins do not appear to be sustainable, although there had been a steady reduction in harvest in the last three to five years many of the

[28] Where hatchery-reared fish are not marked, hatchery origin is determined by the height of the dorsal fin. The assumption being that fin wear in the hatchery during juvenile rearing leaves fish with shorter fins. The height of a credit card – 2 1/8 inches (54 mm) – distinguishes hatchery-origin (shorter fins) from natural-origin steelhead (taller fins).

populations had failed to meet their "MSY" escapement levels. There was concern that the productivity estimates from Gibbons et al. (1985) used in setting escapement levels were not appropriate for these large coastal systems. Further, there was considerable uncertainty in the expansion of redd counts, specifically the redd:adult ratio. Similarly, it was unclear if harvest estimates included natural-origin bycatch of summer-run and winter-run steelhead in the salmon fisheries or on-reservation recreational fisheries. These factors were considered likely to lead to a continued decline in abundance of winter run.

### Summer-run Steelhead

There was a paucity of data available for summer-run steelhead in the Olympic Peninsula DPS. Information was limited to past and present harvest (it was implied that steelhead caught between April and October were summer-run) and intermittent snorkel surveys carried out in the last two decades. It is possible that some of the fish caught in the spring are winter-run kelts, and likewise fish caught in October could be very early returning winter run. Summer-run steelhead are present in the Lyre, Quillayute (Sol Duc, Calawah, and Bogachiel rivers), Hoh, Queets, and Quinault rivers. Estimates of abundance vary, although based on summer-run harvest data prior to the releases of hatchery-origin summer run many of the rivers appear to have supported runs of several hundred summer run fish. Based on snorkel surveys, recent abundances likely range from less than a hundred to a few hundred adults, with considerable uncertainty in any estimates. Further, it is unclear if a remnant summer-run population still exists in the Lyre River. In contrast to other river systems on the Pacific Coast, access to summer-run spawning habitat does not appear to be a limiting factor. Similarly, spawning habitat, most of which is located with the Olympic National Park boundaries, is of high quality. Harvest data is very limited, and some members of the SRT were concerned that there was no information on the potential for summer-run steelhead bycatch in the summer/fall salmon commercial harvest, nor was there information on mortality from the recreational fishery. There was a consensus that climate change over the next few decades would result in dramatically reduced summer flows with the potential loss of access to holding and spawning habitat, as well as the loss of the habitat itself due to high summer temperatures and low summer flows.

### Resident O. mykiss

In general, resident *O. mykiss* (rainbow trout) were not considered in the risk assessment. Those resident fish above long-standing natural barriers are excluded from the DPS (based on previous steelhead determinations, see discussions in 70 FR 67131, November 4, 2005; 71 FR 834, January 5, 2006; 71 FR 15666; March 29, 2006). It has been demonstrated that below long-standing barriers, resident fish can contribute to the anadromous population and vice versa; however, despite the incomplete reproductive isolation, resident *O. mykiss* are considered discrete from anadromous *O. mykiss*, and are not considered part of the steelhead DPS (71 FR 15666, March 6, 2009). While resident fish are known to be present in the watersheds of the Olympic Peninsula, there have been no efforts currently to quantify their abundance nor their demographic relationship with the steelhead DPS. Hard et al. (2015) discuss in further detail how resident fish can be included in the viability analysis for steelhead DPSs, but in the absence of information the contribution to the Olympic steelhead DPS viability was considered negligible.

## Productivity

There are few measures of productivity available for natural populations.  The most recent 15-year abundance trend estimates, indicate that 5 of the 15 population analyzed had negative trends, 4 of which were significantly different from 0; including the larger Queets River and Bogachiel River winter run populations.  Positive trends were observed in 8 of the 15 populations, and only 2 of those were significant, specifically the smaller Pysht and East Twin River winter populations.  Analysis of trends in the total run size; however, suggest declining productivity under varying harvest rates, as stated above, for the four major basins the combined 5-year average run size declined by 42% from the time of the Busby et al. (1996) review to present.  Only under the dramatically reduced harvest conditions experienced in the last three years did total run size appear to stabilize or increase slightly in the four major basins.  It is unclear if this "improvement" in total run size is strictly related to harvest changes and/or improvements in freshwater and ocean conditions.  Overall, the population growth rate (mu) for the four major basins appears to be positive in the absence of harvest effects.  Estimates of population growth rate for the smaller populations in WRIA 19 also suggest that, on average, harvest was depressing growth rates, although the effect was more subtle than in the large coastal systems and these populations have not rebounded in the ten or so years since harvest was terminated.  Smolt survival (both natural and hatchery) has decreased since the 1980s (Harbison et al. 2022), although the underlying causes (i.e. marine and/or freshwater conditions) have not been identified.  Similarly, the survival of kelts in the four larger coastal drainages has declined by nearly half since the 1980s.  The reduction in the number of repeat spawners can also affect productivity; larger more fecund repeat spawners can significantly contribute to population productivity.

## Spatial Structure

The Olympic Peninsula Steelhead DPS lies in a region of the West Coast that is not impacted by dams or other in-stream passage blockages on rivers.  State and County road stream crossings may block or impair passage at culverts, similarly, forest road stream crossing may reduce spatial structure.  In general, road culverts block tributary access to relatively small areas of spawning and rearing habitats, collectively they block only a small fraction naturally-accessible habitat. Impassable culverts on State roads are required to be upgraded under the 2013 U.S. District Court Injunction (*U.S. v. WA Culvert Case*), whereas forestry road culverts are covered under the Road Maintenance and Abandonment Plan (RMAP).  There has been considerable progress in replacing culverts, especially under the RMAP process where over 80% of the culverts are passable, but additional culverts exist that are not included within RMAP (NWIFC 2020).  In addition, most of the headwater reaches for the larger rivers are located within the Olympic National Park and are not subject to anthropogenic blockages.

The SRT also discussed the potential for future restrictions in spatial structure due to low summer flows that may limit passage to headwater areas.  Climate change projections for 2040 and 2080 suggest that low-flow or high water-temperature barriers may create temporal passage blockages; these would disproportionately affect summer-run steelhead.

### Diversity

The SRT discussed three major areas of risk regarding diversity. These included harvest-related selection and loss of run timing diversity, introgression and competition with non-native winter-run and summer-run steelhead hatchery stocks, and loss of genetic variability through small population size effects.

The SRT reviewed available historical harvest information that indicated that the winter-run steelhead return run timing was much earlier than is currently expressed. Large numbers of winter-run steelhead were harvested from November to January[29] prior to and following the initiation of hatchery programs in the Olympic Peninsula DPS. With the beginning of hatchery programs in the DPS utilizing early-returning winter-run steelhead (i.e. Chambers Creek Hatchery Stock from south Puget Sound) there was a directed harvest of the early returning portion of the run targeting hatchery fish. As a consequence of this continued harvest, it is likely that a high proportion of the early returning native winter-run steelhead were and continue to be harvested. Further, it is unclear if this selection has affected the geographic or temporal distribution of steelhead spawners in these basins. If so, then there may be a loss in productivity disproportional with the simple harvest rate. The loss of early returning steelhead was also discussed in the context of climate change and that early returning and spawning winter steelhead may be less affected by future conditions. Increased summer temperatures combined with lower summer flows may affect later returning and spawning life histories.

The presence of non-native hatchery-origin steelhead, both winter run and summer run, was a concern in that non-adapted genotypes may be integrated into the naturally spawning native population. The co-managers identified three hatchery stocks utilized in the Olympic Peninsula DPS: early winter-run steelhead (Puget Sound/Chambers Creek Hatchery), early summer-run steelhead (Lower Columbia River/Skamania Hatchery), and Cook Creek (Quinault NFH stock) (COPSWG 2023). While the early winter Chamber Creek Hatchery and early summer Skamania Hatchery stocks are clearly derived from out-of-DPS sources and not considered part of the DPS, the Cook Creek/Quinault NFH stock has a more uncertain origin. Genetic analyses indicates that winter steelhead utilized in the Quinault NFH, Quinault Lake Hatchery, and Salmon River (Queets) Fish Culture Facility are similar (Seamons and Spidle, 2023), and there have been transfers of fish between these facilities and from other facilities both within and outside of the DPS. Furthermore, there has been some effort to select broodstock for life history traits in the past (age, run timing). Although current sampling for genetic analysis provides limited coverage of the DPS, there is some indication that hatchery stocks utilized in the Queets and Quinault rivers are not representative of the natural populations in those watersheds. Therefore, none of the currently released hatchery stocks was considered as part of the DPS.

[29] In some years significant numbers of fish were harvested in October, although these numbers may include summer-run steelhead.

There is a large body of scientific information on the relative reproductive success of hatchery-origin salmonids (McLean et al. 2004, Berejikian et al. 2009, Ford et al. 2016). Domestication selection, non-locally-adapted life history traits, competition, and disease are likely factors that influence the reproductive success of both hatchery-origin fish and the natural-origin fish with which they interact and potentially interbreed. Other than work presented by Marston and Huff (2022), who modelled potential interactions between hatchery and natural-origin fish in the Quillayute Basin, there is little recent specific data on the proportion of hatchery-origin fish on the spawning grounds (pHOS) in Olympic Peninsula rivers. The SRT acknowledged that there have been changes in hatchery operations to reduce off-station releases, in order to increase the proportion of fish returning to the hatchery rack and decrease the number of hatchery-origin fish straying and spawning naturally. The Sol Duc River was established as a “wild steelhead gene bank” in 2012. Further, releases into many tributaries draining to the Strait of Juan de Fuca (WRIA 19) were eliminated almost a decade ago. While many of the hatchery broodstocks were established and or have been influenced by out-of-DPS steelhead stocks and are operated as segregated hatcheries and thus do not directly “mine” natural-origin populations for spawners, there was still considerable concern by the SRT about their effect on the native populations.

The effects of hatchery releases are related to hatchery release protocols and hatchery broodstocks, as well as the status of the natural populations that they interact with. Natural populations along the Strait of Juan de Fuca and Cape Flattery have relatively small abundances and past and continuing releases of hatchery fish are more likely to have a significant effect on natural abundance and genetic composition. Larger rivers draining to the Pacific Ocean have larger natural populations and greater spatial structure, thus despite the large size of many of the corresponding hatchery programs it is possible that there is somewhat limited interaction and introgression between the hatchery and natural populations. Again, in the absence of systematic genetic sampling and spawner surveys it is not possible for the SRT to assess this risk. The SRT is also concerned about the operation of hatcheries in the Queets and Quinault basins, there is some uncertainty regarding the genetic composition of the broodstocks used and whether they are representative of the native population. There are a few representative genetic samples available, taken in different years, and most are of the hatchery populations rather than the natural populations. Hatchery operations in the Quinault and Queets basins were also of concern because only a small proportion of the juvenile releases are marked, on average 30,000 fish are coded-wire-tagged and adipose fin clipped. Hatchery fish in the recreational fishery are nominally identified by the height of the dorsal fish; a process that has been found to be prone to misidentifying smaller natural fish and larger hatchery fish (Harbison et al 2022). Finally, although there are limited releases of summer-run steelhead into the OP DPS, the small population abundances presumed for native summer-runs makes them especially vulnerable to introgression by the non-native summer-run and early-winter-run hatchery releases, including stray hatchery fish released outside of the DPS. Further, this process leads to greater uncertainty in harvest rates, hatchery broodstock collection, and estimates of pHOS. The SRT concluded that hatchery operations pose a risk to DPS diversity, although the level of this risk varies from population to population depending on the specifics of the hatchery program and the natural population.

Another concern voiced by SRT members was the risk to diversity caused by small population abundances. When population abundances are reduced to relatively small numbers, they are subject to a number of demographic processes, including Allee effects. A number of these populations exist in relatively small watersheds and are thus more vulnerable to catastrophic events. Within the context of diversity risk, small populations are more likely to experience a loss of genetic diversity through inbreeding and direct or indirect selection. Tempering diversity concerns for small steelhead populations, is the life history variability: resident *O. mykiss*, multiple spawner ages, and repeat spawners, all contribute to bolstering the number of effective spawners and provide some buffering against inbreeding. Additionally, the relative proximity of populations allows for the continued influx of migrants, even small numbers of migrants, that also helps maintain genetic diversity.

## DPS Risk Assessment

In considering the overall DPS risk of extinction the SRT considered a number of factors. Firstly, contemporary census estimates indicate that there are nearly twenty thousand natural-origin steelhead spawning in the DPS, roughly the same number as were considered by Busby et al. (1996), although the number of populations surveyed has increased. For the four major basins escapement has decreased 16% since the last status review. As before, winter-run are the predominant life history strategy in the DPS, with the abundance of summer-run steelhead populations largely unknown, but clearly at very low levels (likely in the low hundreds). With the exception of the last three or five years, harvest has been maintained at relatively high levels (25.8%)[30] in the four major coastal tributaries since Busby et al. (1996), while elsewhere in the DPS harvest has been eliminated in most of the small tributaries draining to the Strait of Juan de Fuca. The retention of natural-origin (unmarked) fish was prohibited in the recreational fisheries, but not the commercial fisheries, throughout the DPS. There was some concern that in the Queets and Quinault basins only a small proportion of hatchery-origin steelhead are marked and dorsal fin height is used to distinguish natural from hatchery-origin fish, a system that likely results in the recreational harvest of natural-origin steelhead. The continued direct harvest of natural-origin steelhead in the commercial fisheries, in addition to an unknown amount of indirect harvest of natural-origin steelhead in the major coastal tributaries was a source of concern for the SRT. Commercial harvest was purported to target hatchery-origin winter steelhead that return from November to February, but also intercepts large numbers of natural-origin steelhead. This diminution of the early portion of the natural run is thought to have abundance, diversity, and spatial structure aspects that all likely reduce the long-term persistence of these populations and the overall viability of the DPS. The previous SRT based their risk analysis on information that there was sufficient temporal separation between natural and hatchery populations to minimize harvest overlap and the potential for genetic introgression (Busby et al. 1996); whereas, the current SRT was provided with substantial information to conclude that there was considerable overlap between hatchery-origin and natural-origin adults.

The management of co-occurring natural and hatchery winter-run populations in the Quillayute, Hoh, Queets, and Quinault basins have several consequences on the viability estimates for the natural populations. In order to assure that the hatchery contribution to spawner abundance

[30] Average of natural harvest/escapement (post March 15$^{th}$) for Quillayute, Hoh, Queets, and Quinault rivers from 2016-2020 see Figure 31)

estimates is minimized the co-managers use the March 15th threshold for counting natural redds, and while this may exclude hatchery-origin steelhead produced redds it also leads to an undercount of the natural-origin redds. Given the protracted overlap in run timing between hatchery-origin and natural-origin steelhead, it is likely that there is some level of introgression between these populations in each basin, especially given the propensity of hatchery-origin male steelhead to linger on spawning grounds. There was a paucity of genetic information on naturally-produced steelhead, and available samples were taken in intermittent years from different sites. It was not possible to estimate the level of introgression in any basin, but based on available genetics and hatchery transfer records it seems that most of the hatchery broodstocks were founded and/or substantially influenced by hatchery populations outside of the DPS, specifically Chambers Creek Hatchery winter-run steelhead, and Skamania Hatchery summer-run steelhead. The majority of the hatcheries in the DPS are operated as segregated hatcheries, which should eliminate natural-origin steelhead being used as broodstock and reduces the potential for introgression in the hatchery. With the exception of hatcheries in the Queets and Quinault rivers, most hatchery releases are marked. That only a small proportion of the releases in the Queets and Quinault rivers, which constitute about half of the total DPS hatchery production, are marked increases the likelihood that natural-origin adults will be incorporated into hatchery broodstocks. Off-station releases have been largely eliminated to improve returns of non-harvested hatchery-origin fish to the hatchery rack. These efforts (segregated broodstocks and eliminating off-station releases) to minimize the interaction between hatchery-origin and natural-origin steelhead have likely reduced, but not eliminated the potential for genetic degradation of the winter-run populations. There is little monitoring of the interaction between hatchery-origin and natural-origin summer-run steelhead, and it is not possible to quantify hatchery-related effects; given the relatively low abundance of the natural populations and the detection of hatchery summer run fish in natural holding areas, there is a risk of genetic introgression. Broodstock used in the Salmon River Facility (Queets Basin) and Quinault Lake Hatchery have more complicated histories, but have been influenced by transfers of Chambers Creek stock sufficiently to be considered non-native. The operation of the Salmon River and Quinault Lake facilities as integrated hatcheries, incorporating unmarked fish into the broodstock and not marking the majority of releases, creates the opportunity for substantial dilution of the natural genetic diversity. The SRT recognized that elsewhere in the State there had been efforts to transition away from non-native hatchery stocks. Although there have been some improvements in hatchery operations and release protocols, the underlying continued use of non-native broodstocks is a diversity risk to the DPS.

In reviewing the spatial structure and habitat quality of rivers in the DPS the SRT viewed conditions as generally good. Conditions in many of the rivers had likely improved since the review by Busby et al. (1996), although it was recognized that the natural recovery from past timber harvest events and stream cleaning practices.[31] can take decades. Many of the larger basins also had their headwaters in the Olympic National Park, which provided past protection and some assurance of future protection from land development. Some smaller basins are situated in industrial forest lands and are subject to greater harvest effects than State and Federal forest lands. Several programs to retire forest roads and repair culverts were also seen as having

[31] In the past, the presence of large wood in streams was viewed as a barrier to fish migration, and streams were "cleaned" of these blockages. This had little migrational benefit, but more often resulted in the rapid degradation of stream channels.

improved habitat and spatial structure and will continue to do so into the future. In considering habitat and spatial structure changes into the future, the SRT was most concerned about the immediate and long-term effects of climate change. Already, a number of glaciers in the Olympic Mountains have been lost, snow pack has diminished and summer low flows undergone noticeable changes in recent years. Changes in summer flows, with some reaches going dry, directly affect summer-run steelhead in their ability to reach their headwater spawning reaches. Temperature changes in the lower river reaches may improve rearing conditions for juvenile steelhead, but the transitions from snow and rain to rain dominated hydrographs that are predicted for the Olympics in the next 40-60 years will likely lead to river scour and changes in river morphology. Predictions for an increased incidence in atmospheric river events will result in degraded spawning conditions in the foreseeable future. The climate change effects that have already occurred, and those that are predicted, were not incorporated into the risk assessment by the previous SRT (Busby et al 1996) and were a major factor in the analysis by the current SRT.

## SRT VSP Risk Scoring

In the unweighted[32] assessment of VSP criteria for steelhead populations in the Olympic Peninsula DPS (Table 20), the overall highest risk was given to abundance (average 3.4, median 3.5); however, this was largely due to high-risk scores for summer-run populations (average 4.2) and the smaller populations along the Strait of Juan de Fuca (average 3.7). The winter-run steelhead populations in the four major ("Big Four") coastal rivers, which account for the majority of the DPS abundance were given an average risk score of 2.6. Productivity was also scored relatively high by the SRT, with an average risk score of 2.9, median 2.8, for all populations in the DPS. Summer-run steelhead productivity averaged 3.5, with the Strait's winter populations averaging 2.6, and the Big Four winter-run populations averaging 2.5. Diversity risk scores were somewhat lower than Abundance and Productivity scores, with an overall average of 2.3 and similar scores for the Big Four and Strait's winter-run populations and summer-run populations. Finally, spatial structure scores reflected the lack of major anthropogenic barriers and were generally 2.0 or less.

In scoring the relative risks that the Threats pose to the DPS and its populations (Table 21), the SRT considered six types of threats. Foremost amongst the threats was Climate Change, with particular concern for the effects of climate change on summer-run steelhead populations and those larger rivers that currently exhibited a rain/snow hydrograph in the DPS. These effects include low summer flows and increased winter flows, especially the frequency of major winter rainfall, rather than snowfall, events. In addition to concerns related to higher summer stream temperatures, many SRT members concluded that the loss of glaciers would have wide reaching effects on water quality and river conditions throughout the year. The risks related to overutilization and inadequate harvest regulation were often evaluated as a common threat. Relatively high harvest rates were viewed by many SRT members as an indicator of an inadequate system for evaluating the capacity and productivity of the steelhead populations. These harvest rates had continued despite clear evidence that populations could not maintain those rates. Further, the near complete absence of any coordinated summer-run management was considered a threat to the persistence of summer run steelhead. Hatchery Effects and Habitat Loss and Destruction were also identified as threats, but to a lesser extent than harvest

[32] Populations were given equal weighting in the computation of scores for each category regardless of abundance.

overall in the DPS. For Hatchery Effects, the risks are especially low in the Strait of Juan de Fuca populations where most hatchery releases have been terminated, but hatchery legacy effects may continue and higher risks were noted in the "Big Four" where the majority of hatchery fish are released and mixed harvest of hatchery and natural origin fish continues. Current habitat conditions were considered a relatively low risk factor, with most SRT members seeing habitat as generally improving, but due to the relatively long-time frame for habitat recovery legacy effects from stream clearing and timber harvest activities continue. Finally, Disease and Predation was considered a low risk, primarily related to hatchery operations.

Table 21. Status Review Team Risk Scores for Viable Salmonid Population Criteria. "All populations" represents an unweighted mean and median for all populations (see Table 25 for individual population scores), "Big Four Winter Run" mean scores are representative of all winter-run populations in the Quillayute, Hoh, Queets, and Quinault rivers. "Strait Winter run" mean scores represent winter-run populations in those independent rivers and creeks that flow in the Strait of Juan de Fuca. "Summer run" mean scores represent summer-run steelhead populations in the DPS.

| VSP Criteria | Abundance | Productivity | Spatial Structure | Diversity |
|---|---|---|---|---|
| All populations | | | | |
| Mean | 3.4 | 2.9 | 1.4 | 2.3 |
| Median | 3.5 | 2.8 | 1.3 | 2.3 |
| Big Four Winter Run | 2.6 | 2.5 | 1.3 | 2.4 |
| Strait Winter Run | 3.7 | 2.6 | 1.4 | 2.1 |
| Summer Run | 4.2 | 3.5 | 1.5 | 2.5 |

Table 22. Status Review Team Risk Scores for Threats. "All populations" represents an unweighted mean and median for all populations (see Table 26 for individual population scores), "Big Four Winter Run" scores are representative of all winter-run populations in the Quillayute, Hoh, Queets, and Quinault rivers. "Strait Winter run" scores represent winter-run populations in those independent rivers and creeks that flow in the Strait of Juan de Fuca. "Summer run" scores represent summer-run steelhead populations in the DPS.

| Threats | Habitat Loss and Destruction | Over-utilization | Inadequate Regulation | Disease and Predation | Hatchery Effects | Climate Change |
|---|---|---|---|---|---|---|
| All populations | | | | | | |
| Mean | 2.1 | 2.5 | 2.9 | 1.1 | 2.1 | 3.1 |
| Median | 2.1 | 2.7 | 2.8 | 1.0 | 2.2 | 3.0 |
| Big Four Winter Run | 2.1 | 3.0 | 3.1 | 1.4 | 2.6 | 3.3 |
| Strait Winter Run | 2.3 | 1.8 | 2.3 | 1.0 | 1.7 | 2.8 |
| Summer Run | 2.1 | 2.7 | 3.8 | 1.1 | 2.2 | 3.7 |

In their overall evaluation of the DPS status the majority of the SRT members put the majority of their ten likelihood points in the Moderate Risk category, with one member being equally split between Low and Moderate Risk, and another placing the majority of their likelihood points in the Low-Risk category (Table 22). SRT members giving the lowest risk scores concluded that the overall DPS abundance was still somewhat moderate and that the major threats, other than climate change, could be addressed directly through management actions, rather than longer term habitat restoration. In addition, three SRT members placed likelihood points in the High-Risk Category. In discussing their risk scores, all of the members were concerned with the marked decline in run size for all the major populations, and while acknowledging that there have been considerable reductions in harvest to maintain escapement, the populations have not rebounded under reduced harvest pressure. Further, with harvest at already low levels, there were limited options to improve productivity through harvest management. Trends for many of the smaller populations in the Strait of Juan de Fuca were stable, but at absolute abundances that are very low (<100[33]), despite the termination of harvest in most of the basins over a decade ago. While habitat condition was generally good and restoration efforts had made considerable progress in some areas, the SRT considered that the effects of climate change on freshwater and marine conditions already observed are indicative of effects into the foreseeable future and pose a risk to the viability of the DPS. Further, continued hatchery operations with existing stocks and non-selective harvest may reduce life history diversity and limit the ability of these populations to respond to environmental changes.

Table 23. Distribution of SRT member scores for overall DPS risk of extinction.

| SRT Votes | L | M | H |
|---|---|---|---|
| Average | 4 | 5.50 | 0.5 |
| Median | 4 | 5.5 | 0 |
| Range | 2,6 | 4,7 | 0,2 |

## Significant Portion of the Range (SPOIR) discussion

As discussed earlier, the SRT decided on two scenarios for evaluating the portions of the DPS. In evaluating portions based on major life history traits, run-timing portions using summer-run (stream-maturing) and winter-run (ocean-maturing) life histories were identified. In assigning likelihood points for the biological significance of the summer-run portion, the majority of the SRT members placed the majority of their likelihood points in the not significant category, with an average 4.1/10 points in the significant category and 5.9/10 points in the not significant category (Table 23). In the SRT discussion, factors for “not significant” included that: summer-run steelhead currently and historically were not a major contributor to overall DPS abundance, that winter-run and summer-run populations in the same watershed are not completely reproductively isolated and have generally been found to be genetically very similar (thus, there is some possibility for reestablishment if a summer run population is lost), and that summer-run

[33] At population abundance levels of <100, small population effects (inbreeding, demographic depensation) can increase the risk of extinction.

specific habitat (predominantly just for spawning) represents a relatively small fraction of the total accessible spatial structure. Although the majority of likelihood points were assigned to summer-run populations as being “not significant” under the SPOIR policy, most felt that summer-run populations were still relevant in the VSP assessment of the overall DPS viability. A minority of the SRT put the majority of their likelihood points for summer-run steelhead into the “significant” category, with a corresponding high-risk rating. The SRT concurred that the loss of summer-run populations would increase the diversity risk to the DPS and to a lesser extent increase the spatial structure risk, although the level of additional risk varied depending on the SRT member. The SRT discussed at length that summer-run populations were an important consideration in the overall VSP risk scoring, while not being significant. On average, 41% of the SPOIR votes were for the summer-run populations being significant, which reflects this group’s importance, and the need to reflect the status of summer-run populations into the overall DPS risk.

Alternatively, all of the SRT members believed that the winter-run portion of the DPS was significant (9.3/10). In this case, it was envisaged that the loss of winter-run populations would create greatly increased risks to DPS abundance, productivity, diversity, and spatial structure (most rivers in the DPS contain only winter-run steelhead) and loss of winter steelhead would leave any remaining small summer-run populations isolated and susceptible to catastrophic or random demographic events. In evaluating the risk status of the winter-run SPOIR, the SRT identified decreasing population run sizes in the larger rivers, as well as a number of rivers with relatively small abundances. The average point assignment for risks being low 5.6/10, moderate 4.3/10, and high 0.1/10. All of the SRT concluded that the winter-run portion of the DPS was significant, and thus a SPOIR, but the level or risk was not higher than the overall DPS.

Table 24. Scoring for Significant Portion of the Range using portions based on run-timing life history strategies: summer-run and winter-run populations. Members scored each portion for significance and risk level, assigning 10 likelihood points to each question for each portion.

Scenario 1 Run Timing

| Summer-run Significant | | Summer-Run Risk | | |
|---|---|---|---|---|
| Yes | No | L | M | H |
| 4.1 | 5.9 | 0.6 | 4.3 | 5.1 |

| Winter-run Significant | | Winter-run Risk | | |
|---|---|---|---|---|
| Yes | No | L | M | H |
| 9.3 | 0.7 | 5.6 | 4.3 | 0.1 |

The SRT also discussed and assessed a Significant Portion of the Range scenario based on geography. In this case, the geographic units included: 1) steelhead populations in rivers that drained to the Strait of Juan de Fuca, and 2) steelhead population in rivers that drained to the Pacific Ocean. These two regions were identified as potential portions due the hydrological and geographic distinctiveness or the rivers supporting Strait populations and Coastal populations. The majority of the SRT members assigned a majority of their likelihood points in the not significant category (6.0/10) for populations draining to the Strait of Juan de Fuca (Table 24). The SRT considered that populations in the Strait of Juan de Fuca may express distinct life-history strategies and their loss would increase the diversity risk as well as spatial structure risk to the DPS; however, the increased risk in diversity was thought to be tempered by the presence of Coastal winter-run populations in streams ecologically similar to those in the Strait.

Following the hypothetical loss of populations in the Strait, over the long term it was likely that the rivers there could be recolonized. The SRT also assigned the majority of their likelihood points for risk in the moderate category (6/10), 3.6/10 in the low-risk category, and 0.4/10 in the high-risk category. This risk evaluation was primarily influenced by the small population abundances and limited productivity for winter-run populations in the Strait. Because the SRT determined that the Strait populations did not meet the agency's criteria for significance, this population group is not considered to be a SPOIR and the extinction risk assigned to this portion did not supersede the overall DPS extinction risk score. Similar to the VSP assessment of the summer-run "portion", in the overall risk assessment these Strait populations were incorporated into the analysis, while not being considered a "significant portion." Coast populations were determined by the SRT to be significant (8/10). Coastal populations contain more than 90% of the DPS abundance, potentially all of the summer-run populations, and winter-run populations in a wide diversity of river types. Loss of Coastal populations would result in greatly increase risks for abundance, productivity, diversity and spatial structure. Further, given the low population abundances in the Strait it is unlikely that sufficient numbers of fish would be available for recolonization. The low average risk (5/10) received the most likelihood points, the moderate risk likelihood score of 4.7/10 and high-risk score of 0.3/10, suggest a relatively even divide between low and moderate risk. As cited in the main body of this report, abundance (especially for summer-run populations), productivity, diversity (hatchery effects and loss of life histories) concerns influenced the risk score for this SPOIR. Therefore, although the Coastal populations were considered a SPOIR, it was determined not to be at a higher risk level than the DPS overall.

Table 25. Scoring for Significant Portion of the Range using portions based on geography: Strait of Juan de Fuca (JDRF) and Coastal populations. Members scored each portion for significance and risk level, assign 10 likelihood point to each question for each portion.

Scenario 2 Geographic

| Strait JDF Significant | | Strait of Juan de Fuca Risk | | | Coastal Significant | | Coastal Risk | | |
|---|---|---|---|---|---|---|---|---|---|
| Yes | No | L | M | H | Yes | No | L | M | H |
| 4.0 | 6.0 | 3.6 | 6.0 | 0.4 | 8.0 | 2.0 | 5.0 | 4.7 | 0.3 |

The Olympic Peninsula Steelhead SRT concluded that the DPS was at moderate risk of extinction and a subsequent review of the DPS identified two scenarios for identifying significant portions of its range: life history and geographic/ecological distribution. For each of these scenarios, a single SPOIR was identified: winter-run steelhead populations and coastal steelhead populations, respectively. Further, risk analysis for each of these SPOIRs did not result in a risk determination higher than that of the moderate risk assessment for the entire Olympic Peninsula DPS. Following completion of the SPOIR process the SRT reconfirmed the moderate risk of extinction for the Olympic Peninsula Steelhead DPS.

The total abundance of steelhead in the DPS was relatively high compared to other DPSs at moderate risk, but the relatively high risk scores estimated for summer-run populations was a factor in the VSP risk analysis, especially for diversity. Further, analyses by the co-managers and the SRT of run sizes for the four major winter-run populations suggest that over-harvest and other unknown factors were affecting the viability of these populations, and the sustainability of some of these populations is in question. The SRT acknowledged that some hatchery practices had been improved to reduce interactions and introgression; however, the ongoing use of out-of-DPS origin hatchery stocks present a continued risk to the natural populations, and that continued management (harvest, post-March 15th redd surveys) under the concept of temporal separation between hatchery and natural stocks was not supported by available information. Finally, although there is uncertainty in the long-term effects of climate change, climate change has already impacted habitat in the Olympic Peninsula and climatic changes effects in the next 40-50 years will be increasingly deleterious to steelhead populations in the DPS.

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# Appendix A: OP DPS Watershed Summaries

## Strait of Juan de Fuca watersheds

### Salt Creek

Salt Creek is a small watershed in the eastern-most portion of the Olympic Peninsula steelhead DPS, in the Strait of Juan de Fuca (Figure 1). Salt Creek has a drainage area of less than 50 $km^2$ and approximately 40 kilometers of accessible stream habitat for steelhead (McHenry et al. 2004). The entire watershed is influenced by the most recent continental glaciation (Vashon Stade ~25,000 years ago), and as a result over 50% (~59%) of stream habitat is less than 4% gradient, which is conducive for anadromous use (McHenry et al. 2004, NOPLE 2015). Many of the larger scale features, such as glacial striations, determine the stream channel gradients in Salt Creek, and their associated wetland complexes (Tabor and Cady 1978, McHenry et al. 2004). Salt Creek includes an estuarine salt marsh complex (NOPLE 2015).

Salt Creek has a rich cultural history supporting several significant Klallam cultural sites, including: teu' dlt (Agate Point-translates "abounds in mussels"), TL sEent (Crescent Bay translates "deep"), Klte-tun-ut (Salt Creek), Tsatso-Al sEnt (Tongue Point-translates "close by the deep place") (James 1993). Three camp/village sites have been documented in the vicinity (Waterman 1920). Klte-tun-ut was the site of a large permanent village. Present day land ownership patterns in Salt Creek are a complex blend of state and industrial forestland, agricultural, and rural residential uses. State and private forestlands are mostly located in the headwaters (~56%), while agricultural and rural residential lands (42%) are strongly clustered in low gradient stream channel areas in the middle and lower watershed (McHenry et al. 2004, NOPLE 2015).

Current stream habitat conditions represent a juxtaposition of functional areas, including the estuary, combined with simplified stream channels due to historic wood removal and loss of riparian forests along the majority of the stream network (McHenry et al. 2004). Some changes due to the loss of historic wood include stream channel incision of up to 1.5 meters, from river kilometer (RKM) 1.6 to 10.5, resulting in a loss of pools, spawning gravel, and floodplain connection (McHenry et al. 2004, NOPLE 2015). Sections of mainstem Salt Creek and some tributaries have incised down to bedrock (NOPLE 2015). Thirty-five large logjams were reportedly removed from Salt Creek, then clearcut in the early 1950s, followed by cedar salvage (WDF 1953, ref). Riparian conditions reflect the lack of wood and stream characteristics associated with wood in Salt Creek. Near-term LWD recruitment potential is only 18% of its large wood recruitment potential for stream channel less than 2% gradient (McHenry et al. 2004). Almost 15 kilometers of the riparian zone is impacted by adjacent roads which limits the future ability for riparian zones to grow and recruit wood to the channel.

There are other habitat factors affecting stream habitat in Salt Creek that create barriers. Almost 50% of the potential stream available to salmon and steelhead is blocked by fully or partially impassable barriers including human-built ponds (McHenry et al. 2004). Many of these reaches would be habitat for anadromous salmonids if accessible (NOPLE 2015). Because of the natural hydrologic regime, low flows can be naturally limiting. Summer flow conditions can average around 0.06 cubic meters per second (cms) (NOPLE 2015). There are currently 37 water rights for a total of 0.08 cms available for usage (McHenry et al. 2004). Salt Creek retains a relatively high productive potential based upon smolt yields measured in recent years (McHenry et al. 2004). Much of this productivity is due to the high proportion of low-gradient stream habitat in the stream network. Stream and watershed restoration has been implemented focusing on the linear reconnection of stream habitats, the restoration of riparian and wetland functions, wood placement, and the protection of functional habitat through acquisition and easements (McHenry et al. 2004, NOPLE 2015).

The 5-year geometric mean for observed escapement of winter steelhead in Salt Creek has changed: 171 from 1998 to 2002, to 84 from 2008 and 2012, and 66 between 2018 and 2022 (Table A1). The Lower Elwha Klallam Tribe (LEKT) has been monitoring steelhead smolt outmigration in Salt Creek since 2001. There has been an overall decrease in steelhead smolts during this 20-year period, averaging an estimated 1,158 (1,009 – 1,308) between 2001 and 2010, and 594 steelhead smolts (450 – 742) from 2011 to 2022. There has been a slight upward trend in smolt estimates from 2014 to 2022.

Steelhead smolts from hatcheries were planted in Salt Creek for only a limited time between 1962 and 1970, and even then sporadically (Table A2). The average number of releases was 5,818 with a minimum of 422 and a maximum of 10,158 (Table A2).

Since steelhead smolt releases did not occur until 1962 all steelhead harvested prior to that date were considered wild steelhead. The average number of steelhead caught prior to 1962 (1948 to 1961) was 279 steelhead per year with a minimum of 2 and a maximum of 679 (Table A3). During the time of steelhead smolt releases (1962 to 1970), average annual steelhead catch was 291 with a minimum of 75 and a maximum of 697 (Table A3).

After smolt releases were terminated in Salt Creek (1971 to 2005), the average annual steelhead catch was 35 per year with a minimum of 0 and a maximum of 748 (Table A3). The number of steelhead caught decreased each decade. In the 1970s, the average steelhead caught per year was 146, with a minimum of 6 and a maximum of 748 (Table A3). These were all recreational harvest (WDFW and Tribal SJD data June 2023 - NOAA_5_15_23 OP Steelhead and SJD Aggregate Harvest Data. Data sent by WDFW May/June of 2023). The average steelhead catch dropped in the 1980s to an average of 44 steelhead per year, with a minimum of 11 and a maximum of 134 (Table A3). A decrease occurred again in the 1990s when the average was 16 steelhead per year and had a maximum of 39, followed by a decrease in the 2000s to an average of 8 per year and a maximum of 16 (Table A3).

## Lyre River

The Lyre River is a unique watershed, relative to other Strait watersheds, because its headwaters include the outlet of Lake Crescent. The Lyre River has a drainage area of 171 $km^2$ and approximately 27 kilometers of stream habitat (McHenry et al. 1996). The Lyre River was formed approximately 4000 to 5000 years ago when “a complex of several large rockslides descending the north and south valley walls at the eastern end of Lake Crescent resulted in the separation of a larger, ancestral Lake Crescent into two lakes—the modern lakes Crescent and

Sutherland" (Leithold et al. 2019). Leithold et al. (2019) goes onto state the following: "rockslide blockage of the ancestral drainage of Lake Crescent caused its water level to increase by 24 m (difference in surface water elevation between Lake Sutherland and Lake Crescent) until the lake overtopped a low divide connecting it with the Lyre River, which flows north into the Strait of Juan de Fuca".

The Lyre River, relative to nearby watersheds such as Salt Creek, is steeper with an overall stream channel gradient of 2.2% (McHenry et al. 1996). Even though it has a relatively steeper stream channel gradient the Lyre was historically known for numerous logjams, difficult access due to a thick understory in the riparian zone, and limited gravel bars because its water source is Lake Crescent (Goin 2009).

Anadromous access occurs up until rkm 6.1 where a series of falls and cascades blocks further passage. The creation of the falls is likely a combination of natural and anthropogenic causes (Tabor and Cady 1978, McHenry et al. 1996, NOPLE 2015). A permanent Klallam village (*Qhah-qhah ah*) was historically located at the mouth of the Lyre River (Lane 1975, McHenry et al. 1996). The Lyre River watershed includes the Olympic National Park (~66%), as well as commercial timberlands (31%), and low-density rural residential (~3%) (McHenry et al 1996, NOPLE 2015). The lower portion of the Lyre River has been channelized, armored, and the riparian vegetation has been removed (NOPLE, 2015).

Historically the Lyre River had both native winter and summer steelhead (McHenry et al. 1996, Goin 2009). Run-timing for steelhead, identified as winter steelhead in the anecdotal literature, started as early as the middle of October, one of the earliest timings on the Olympic Peninsula (Goin 1990, McHenry et al. 1996, Goin 2009). Peak entry time for native Lyre River winter steelhead occurred from late December and in early March (Goin 1990, McHenry et al. 1996). The size range for winter steelhead was typically between 4.5 and 8.2 Kgs. Wild summer steelhead in the Lyre River were thought to have a historical abundance around 200 adults per year, while surveys conducted in the mid-1990s estimated the population to be approximately 100 adults per year (McHenry et al. 1996). One caveat for this 1990s estimate is that a Skamania summer steelhead hatchery releases of approximately 20,000 juveniles per year started in the early 1980s (McHenry et al 1996). Natural Lyre River summer steelhead ranged in size from 1.4 to 3.2 Kgs (Goin 2009). Goin (2009) notes that "like all native summer runs, they went as far as they could without much loitering" meaning their entrance into the Lyre was swift and they would collect in the canyon reach below the falls/cascades area at approximately rkm 5.0.

Hatchery supplementation and operations for the Lyre River has been ongoing since 1960 (WDG 1972, Goin 1990, McHenry et al. 1996, Goin 2009). Between 1960 and 1972 the 17,849 winter-run steelhead smolt were released annually, with a minimum of 10,071 and a maximum of 35,130 (Table A2). Between 1981 and 2008 the average winter steelhead release was 26,452 (minimum of 5,424 and a maximum of 50,000) winter steelhead smolts (Steelhead releases Washington Coast 1980 to 2021). Skamania summer steelhead were also released between 1981 and 2008 averaging 10,897 (5,029 min. and 21,422 max) (Table A2).

Increased hatchery releases lead to increased fishing pressure on the Lyre River (McHenry et al. 1996, Goin 2009). Fishing pressure was highest starting in the mid-1960s with the initiation of hatchery releases and continued until about 2010, several years after the hatchery releases were terminated. Prior to any hatchery releases the number of wild steelhead caught per year (1949 to 1959) averaged 205 steelhead with a minimum of 20 and a maximum of 347 (WDG 1972). Steelhead smolts started to be released in 1960 and continued to be released until 2008 (Table A2). Average winter steelhead catch during the 1960s was 1046, with a minimum of 312 and a maximum of 1,526, all of which was recreational (Table A2). Average summer steelhead catch during that time period was 23 with a minimum of 8 and a maximum of 60, again all recreationally caught (Table2).

During the 1970s average winter steelhead catch increased to 1,207 annually, with a minimum of 560 and a maximum of 1,744 (Table A3). Summer steelhead annual catch averaged 27 with a minimum of 3 and a maximum of 56. During the 1980s winter steelhead harvest was similar with an average of 1,206, while summer steelhead average catch increased to 112 per year (Table A3). The vast majority of winter steelhead catch and all of the summer steelhead catch was recreational (Table A3). During the 1990s, average harvest of winter steelhead decreased to 619 annually with a minimum of 87 and a maximum of 1,103 (Table A3). Summer steelhead average harvest increased to 109 annually with a minimum of 16 and a maximum of 361 (Table A3). The vast majority of winter steelhead catch and all of the summer steelhead catch was recreational (Table A3).

During the 2000s to 2010, average winter steelhead harvest was 386 with a minimum of 144 and a maximum of 1,037, while summer steelhead average was 69 with a minimum of 6 and a maximum of 164 (Table A3). The vast majority of winter steelhead catch and all of the summer steelhead catch was recreational (WDFW and Tribal SJD data June 2023 - NOAA_5_15_23 OP Steelhead and SJD Aggregate Harvest Data. Data sent by WDFW May/June of 2023). Between 2010 and 2020 average winter steelhead catch was 65 with a minimum of 0 and a maximum of 214 (Table A3). The vast majority of winter steelhead catch and all of the summer steelhead catch was recreational (WDFW and Tribal SJD data June 2023 - NOAA_5_15_23 OP Steelhead and SJD Aggregate Harvest Data. Data sent by WDFW May/June of 2023).

Part of the change in steelhead abundance in the Lyre River potentially had to do with the loss of chum salmon (Goin 1990). Chum salmon spawning escapement estimates for the Lyre ranged close to 10,000 fish annually (Goin 1990). By 1996, abundance levels were down to 500 to 1,000 annually (McHenry et al. 1996). Run timing was from November through January. Chum fry would emerge from the gravels in the spring, similar to timing of outmigrating steelhead smolts, and so chum fry were hypothesized to benefit steelhead smolts as a food resource (refs).

## East Twin River

East Twin River has a drainage area of 35 km$^2$, with approximately 30 km of anadromous habitat (Williams et al. 1975). Streamflow in East Twin averages approximately 1.4 cms and ranges from a low of 0.05 to a high of 52 cms (Hall et al. 2016). Mean daily water temperatures near the mouths of the East Twin and West Twin rivers range from a low of 0°C in winter to a high of 16.7°C in summer (Hall et al. 2016). Almost the entire watershed is identified as forest lands (NOPLE 2015). Washington state Department of Natural Resources lands (WA DNR) and United States Forest Service lands (USFS) comprise over 90% of the ownership (NOPLE 2015).

Logging, removal of in-channel wood, and riparian alteration have simplified and degraded stream habitat conditions over the last 100 years (Bilby et al. 2005, Hall et al. 2016). Furthermore, increased landslide frequency has occurred due to the construction of logging roads on steep slopes as well as timber harvest (DeCilliis 2002, Bilby et al. 2005, Hall et al. 2016).

Historically, the lower East Twin River was a dynamic river and floodplain system, prior to the large scale removal of wood that occurred in the lower 1.5 miles of the East Twin (Kramer 1952, NOPLE 2015). Large conifers once dominated the riparian corridor, while currently there is a lack of large confers in the riparian zone (NOPLE 2015). The combination of a lack of wood and increased sedimentation has led to reduced pool area and simplified stream habitats (NOPLE 2015).

Starting in the mid-1990s the Lower Elwha Klallam Tribe (LEKT) developed and implemented a watershed-scale restoration plan for East Twin and Deep Creek Watersheds (United States Forest Service, Olympic National Forest et al., 2002). The restoration plan focused on reducing the rates of anthropogenic-caused landslides to background levels, recovering riparian forests to provide long-term supplies of in-channel wood, adding wood to offset losses due to land use impacts, and increasing floodplain habitats. These physical habitat objectives were linked to biological factors including fish abundance, growth, and productivity. For example, elevated landslide rates can cause mortality of juvenile salmonids due to scour-and-fill events, degradation of salmonid spawning habitat due to sedimentation, and loss of juvenile rearing habitat due to pool loss, floodplain disconnection, and overall channel simplification (Kemp et al., 2011). Reducing landslide impacts was a necessary first step in restoration to enable habitat-forming processes to recover naturally. Restoration projects began in 1998 and have continued to the present.

In-channel wood placement was a primary tool for restoration treatment because it influences many stream habitat-forming processes that affect salmon life histories (Roni et al., 2008). Large wood is known to form pools, store gravels, and can reverse channel incision and improve floodplain connectivity (Abbe and Brooks, 2011; Wohl and Scott, 2017). Increases in floodplain connectivity may also increase formation of floodplain habitats known to be critical over-winter habitats for juvenile Coho salmon (Martens and Connolly, 2014).

Over half of the 30 projects that were completed during the last 24 years were wood placement efforts. Restoration treatments implemented from 1998 to 2022 were focused on the lower portions of East Twin and Deep Creek. The majority of wood placement focused on increasing low-gradient, mainstem habitat quality and quantity. Initial treatments were in-channel projects constructed of cut logs that relied upon ground-based placement techniques to create features such as log weirs, sills, and logjams. These treatments were generally of small size and of low profile, obstructing a relatively small percentage of the stream channel cross-sectional area. Some wood was placed to protect the toes of deep-seated landslides from further erosion.

In 2002-2003, the first helicopter wood placement projects were implemented using heavy-lift helicopters to fly in key pieces of wood to both previously ground-based treated reaches and into inaccessible habitats. This technology resulted in new or larger jams (adding to ground-based treatments) or individual key pieces. By 2008, there was a shift away from ground-based wood treatments to helicopter placement of wood.

Additional wood treatments did not occur for nearly a decade following the completion of the initial ground-based and initial helicopter treatments. This was due to a combination of factors but wood treatments were renewed from 2013-2022 and exclusively used helicopter placements.

Both East Twin and Deep Creek restoration efforts were affected by natural disturbance events. For example, in upper Deep Creek, a high percentage of the relatively smaller, low-profile ground-based treatments began to degrade or move in response to large floods in the late 2000's (M. McHenry, personal observation). These movements resulted in larger aggregations of wood (i.e., full channel-spanning logjams) that had a greater effect on habitat features (i.e., conversion of stream channel types) downstream. These accumulations of wood now developing in East Twin are similar in type to accounts of historic wood jams, but not generally as massive (Goin 1990, 2009, McHenry et al. 1996, NOPLE 2015).

The 5-year geometric observed escapement mean of winter steelhead in East Twin has changed as follows - 85 from 1998 to 2002, to 35 from 2008 and 2012, and 54 between 2018 and 2022 (Data and Analyses for OP Steelhead Oct 5, 2023). The LEKT has been monitoring steelhead smolt outmigration in East Twin since 2001. There has been an overall decrease in steelhead smolts during this 19-year period, averaging an estimated 1,074 (854– 1,295) between 2001 and 2011, and 703 steelhead smolts (566 – 841) from 2012 to 2021. There has been a slight upward trend in smolt estimates from 2012 to 2021, with a relatively larger increase between 2019 and 2021.

Steelhead harvest in East Twin in the 1950s and 1960s averaged 99 and 62, respectively (Table A1). During the 1970s average winter steelhead catch decreased to 25 with a minimum of 0 and a maximum of 73 (Table A1). The 1980s average winter steelhead catch was 30 with a minimum of 10 and a maximum of 78 (Table A1). In the 1990s average winter steelhead catch was 15, and in the 2000s average winter steelhead catch was three with no harvest after 2002 (Table A1).

## West Twin River

Similar to East Twin River, West Twin River has a drainage area of 33 km$^2$, with approximately 13 km of anadromous habitat (Williams et al. 1975). Streamflow in East Twin averages approximately 1.0 cms and ranges from a low of 0.05 to a high of 30 cms (Hall et al. 2016). Mean daily water temperatures near the mouths of the East Twin and West Twin rivers range from a low of 0°C in winter to a high of 16.7°C in summer (Hall et al. 2016). Almost the entire watershed is identified as forestry (NOPLE 2015). However, the majority of the forestlands in West Twin are USFS (~61%) followed by private timberlands (29%) (NOPLE 2015). Logging, removal of in-channel wood, and riparian alteration have simplified and degraded stream habitat conditions over the last 100 years (Bilby et al. 2005, Hall et al. 2016). Furthermore, increased landslide frequency has occurred due to the construction of logging roads on steep slopes as well as timber harvest (DeCilliis 2002, Bilby et al. 2005, Hall et al. 2016).

Similar to East Twin River, West Twin River historically was affected by the removal of large trees, large conifers dominated the riparian corridor, with current riparian condition being characterized by a lack of large confers in the riparian zone (NOPLE 2015). The combination of a lack of wood and increased sedimentation led to reduced pool area and simplified stream habitats (NOPLE 2015). Unlike East Twin River, West Twin River does not have stream habitat restoration actions associated with it because it is a reference watershed as part of the Intensively Monitored Watershed (IMW) program for the State of Washington (Bilby et al. 2005).

The life-history diversity of steelhead has been recently investigated in West Twin, as well as East Twin and Deep Creeks (Hall et al. 2016). Eighteen life histories were identified in these three watersheds based on the movement patterns of juvenile steelhead with passive integrated transponder (PIT) tags (Hall et al. 2016). There were variations across individuals in age and seasonal migration of juveniles, juvenile use of the Strait of Juan de Fuca prior to migration, the number of years in the ocean, adult return season, and iteroparity (Hall et al. 2016). Age 2 emigrants, followed by age 1 and 3 or older emigrants, were responsible for all the returning adult steelhead, even though the dominant form of juvenile life history was Age 0 emigrants (Hall et al. 2016). The probability of leaving and returning as an adult increased with body length at tagging (Hall et al. 2016). There was evidence of density-dependent growth, with fewer returning adults as a function of decreasing average body size of tagged juveniles (Hall et al. 2016).

The 5-year geometric observed escapement mean of winter steelhead in West Twin has changed as follows - 116 from 1998 to 2002, to 42 from 2008 and 2012, and 56 between 2018 and 2022 (Table A1). The LEKT has been monitoring steelhead smolt outmigration in West Twin since 2001. There has been no obvious trend in the number of steelhead smolts during this 20-year period, averaging an estimated 1,062 (867-1,257) between 2001 and 2011, and 977 steelhead smolts (817 – 1,137) from 2012 to 2022. There has been a slight upward trend in smolt estimates from 2012 to 2022.

Steelhead harvest in West Twin in the 1950s averaged 58 (Table A3). During the 1960s average winter steelhead catch was 50 (127 in the WDG 1972) with a minimum of 2 (22 in WDG report) and a maximum of 125 (300 in WDG 1972) (Table A3). All catch was recreational during this time period. During the 1970s average winter steelhead catch was 31 with a minimum of 0 and a maximum of 100. By the 1980s average winter steelhead catch was 20 and by the 1990s average winter steelhead catch was 10 with a minimum of 0 and a maximum of 23 (Table A3). By the 2000s average winter steelhead catch in West Twin was 3 per year and no harvest of winter steelhead occurred since 2008 (Table A3).

## Deep Creek

The Deep Creek catchment covers an area of 45 $km^2$ and has approximately 24 km of anadromous habitat (Williams et al. 1975, NOPLE 2015). Average daily streamflow is less than 2 cms, but can exceed 40 cms, with peak discharge around 57 cms (W. Ehinger, Washington Department of Ecology, unpublished data). Flows during monitoring were typically less than 2 cms (Pess et al. 2022). Precipitation occurs primarily as rain between October and May and averages 190 cm per year (United States Forest Service et al., 2002). The geology of Deep Creek is characterized by Crescent Formation volcanic rock in the upper catchment, resulting in steep, confined stream channels (Snavely et al., 1980; United States Forest Service et al., 2002). In contrast, glacial deposits, as well as marine sedimentary rocks, both of which are subject to intense erosion, dominate the middle and lower catchment (Snavely et al., 1980; United States Forest Service et al., 2002). Almost the entire watershed is identified as forestry (NOPLE 2015). However, the majority of the forestlands in Deep Creek are USFS (~50%) followed by private timberlands (~43%) (NOPLE 2015).

The estuarine and nearshore conditions of Deep Creek include lateral channel migration changes from the late 1800s to the present, as well as some land use effects such as sedimentation impacts, from upstream sources and effects of roads on and near the delta (Todd et al, 2006, NOPLE 2015). The known disturbance history of the freshwater portion of Deep Creek dates back to a series of fires in ~1308, ~1508, and several fires between 1895 and 1939 (United States Forest Service et al., 2002, Pess et al. 2022). Since the early 1900s, the primary land use in Deep Creek has been industrial forestry (United States Forest Service et al., 2002). During the 1900s, logging road construction and timber harvest increased landslide frequency, while "stream cleaning" activities removed in-channel wood (Pess et al. 2022).

The combination of increased landslide frequency and wood removal resulted in a simplified and degraded stream. Salvage logging following the 1939 fire was particularly intense and resulted in widespread watershed degradation. In the 1980s, poorly constructed midslope roads caused increasing rates of landsliding, including a large dam break flood event that scoured the upper channel network of Deep Creek. By the mid-1990s, when stream restoration began, Deep Creek had little instream wood, lacked mature riparian vegetation, and had experienced a loss of floodplain connectivity, due to stream channel incision from the lack of obstructions such as wood (United States Forest Service et al., 2002, Pess et al. 2022). Starting in the mid-1990s the Lower Elwha Klallam Tribe (LEKT) developed and implemented a watershed-scale restoration plan for East Twin and Deep Creek Watersheds (United States Forest Service, Olympic National Forest et al., 2002). For a detailed description, please see the East Twin section above.

Restoration monitoring of the effects of 23 years of wood additions have shown positive effects on Deep Creek (Pess et al. 2022). In the ~6 km channel with wood placement, there was an increase in wood loading and channel-spanning logjams, which contributed to deeper and more frequent pools, a reduction in particle size, increases in sediment storage, reduced stream width, vegetation re-establishment in the riparian zone, and increased development and maintenance of floodplain channels (Pess et al. 2022). The largest geomorphic changes occurred due to restoration wood effectively storing pieces of wood moving downstream (Pess et al. 2022). There were cumulative habitat restoration actions and associated changes to stream habitat conditions, which demonstrate that wood placement simulates the function of large key, stable pieces and accelerates habitat recovery within basins subjected to historic logging (Pess et al. 2022).

The 5-year geometric observed escapement mean of winter steelhead in Deep Creek has changed as follows - 162 from 1998 to 2002, to 83 from 2013 and 2017, and 99 between 2018 and 2022 (Table A1). The LEKT has been monitoring steelhead smolt outmigration in Deep Creek since 1998. There has been a slight downward trend in steelhead smolts during this 22-year time period, averaging an estimated 1,832 (1,470-1,521) between 1998 and 2014, and 1,204 steelhead smolts (887 – 1,521) from 2014 to 2022. There has been a slight upward trend in smolt estimates from 2014 to 2022.

Steelhead harvest in Deep Creek in the 1950s averaged 103 (Table A3). During the 1960s average winter steelhead catch was 132 (127 in the WDG, 1972 report) with a minimum of 2 (22 in WDG, 1972 report) and a maximum of 125 (300 in WDG, 1972 report) (Table A3). All catch was recreational during this time period. During the 1970s average winter steelhead catch was 51 with a minimum of 0 and a maximum of 91. By the 1980s average winter steelhead catch was 62 and by the 1990s average winter steelhead catch was 14 with a minimum of 0 and a maximum of 37 (Table A3). By the 2000s average winter steelhead catch in West Twin was 4 per year (Table A3).

## Pysht River

The Pysht River has a drainage area of 118 $km^2$, with approximately 67 km of anadromous habitat (McHenry et al. 1996). Average annual flows in the basin are estimated to be around 6.2 cms, while maximum discharge for the 2-year flood event is approximately 57 cms (McHenry et al. 1996). Almost the entire watershed is identified as forestry (NOPLE 2015). Private timberlands comprise over 75% of the basin, followed by USFS and WA DNR, combined to be ~24% of the remainder (NOPLE 2015).

Because of its larger drainage area, the Pysht River has the largest tidal marsh in WRIA 19 (Todd et al. 2006, NOPLE 2015). The estuarine area has been impacted by development and forestry activities, with almost 50% of the tidal marsh either lost or altered through land-use impacts (NOPLE 2015). Suction dredging and channelization has occurred in the lower 2.4 kilometers, and resulted in tidal channels being filled in with dredge material and the main stem being deepened for log piles (Todd et al. 2006, NOPLE 2015). There was a plan underway to assess and restore portions of the estuary, but after five years of planning and over $700,000 invested in engineering and design it was cancelled (Personal communication Mike McHenry, Lower Elwha Tribe, December 5, 2023).

Floodplain habitat access (lateral connectivity) has been limited by barriers in the Pysht River. (Haggerty et al. 2006, NOPLE 2015). Culverts were estimated to represent partial or total barriers to almost 53 percent (~ 12.9 km) of the total length of floodplain habitat (Haggerty et al. 2006, NOPLE 2015). According to Haggerty et al. (2006), of the ~30 hectares of fish-bearing wetlands along the Pysht River floodplain, only 29% was classified as fully accessible to fish (Haggerty et al. 2006, NOPLE 2015).

In-stream and riparian conditions in the Pysht River watershed are similar to the other WRIA 19 watersheds, with many of the same historical riparian impacts. The combination of the systematic removal of in-channel wood, and degraded riparian conditions due to logging practices, have led to a loss of current and future wood recruitment (Kramer 1952, McHenry and Murray 1996, NOPLE 2015). Stream channel conditions such as pool frequency are reduced and residual pool depth is limited (McHenry and Murray 1996, NOPLE 2015). Encroachment of roads, such as highway 112, along the Pysht River is one of the primary impacts to channel migration and floodplain connection, as well as maintaining riparian conditions in a degraded state (Haggerty et al. 2009).

The Pysht River has had relatively higher levels of stream channel aggradation and degradation, as well as elevated fine sediment levels from forestry related sediment inputs that result in degraded steelhead spawning habitat (McHenry et al. 1994, Smith 2000, Haggerty et al. 2009, NOPLE 2015). Elevated fine sediment levels are typically higher in logged watersheds on the Olympic Peninsula; however, once road density and the proportion of a watershed clearcut reach high levels the correlations decrease due to a saturation effect (McHenry et al. 1994, Haggerty et al 2009).

One of the main stories in the Pysht River system is stream channel incision. Below is an excerpt from Mike McHenry, habitat biologist with the Lower Elwha Klallam Tribe on changes to the South Fork Pysht River:

> The SF Pysht is the largest tributary to the Pysht River and is representative of land use histories in the SJF. The watershed was first logged in the early twentieth century by railroad. A railroad line was constructed up the river valley and the SF was crossed at 26 different locations via trestles and bridges. Old growth timber was removed from the floodplain, river valley and likely the river channel. In 1939, the Burnt Mountain fire burned large areas in Deep Creek and the SF Pysht and following the fire large scale salvage occurred via poorly constructed roads. Following this event, the watershed was not replanted, and regenerated as almost all red alder. In the 1980's the alder was extensively clearcut and converted to conifer plantations. As a result of these impacts, the SF Pysht has incised by 1-2 meters below its former elevation, and the channel has been greatly simplified with long reaches of exposed bedrock and plane bed channels. These conditions were documented in the early 1990s by habitat surveys conducted by the Lower Elwha Klallam Tribe. Those surveys found low levels of in-channel wood, sub-optimal pool structure and riparian zones dominated by deciduous trees. In an effort to accelerate natural recovery of habitat, a number of restoration projects were implemented between 1994-2006. Those projects included the insertion of wood using both ground based and helicopter methods as well as manipulations of riparian vegetation in order to increase the recruitment of conifers. These treatments were conducted in ~25% of the total impacted channel length. In 2023, 14 kilometers of the SF Pysht were resurveyed using the IMW wood budget method to assess the scale of stream channel recovery. Those surveys showed that approximately 7 kilometers of the system had recovered as a result of restoration actions and natural recovery particularly in the upper portions of the watershed where the channel is relatively small. In the lower SF Pysht habitat is still quite degraded with at least 5 kilometers of bedrock and plane bed channel types. Also of concern was the documentation of 1.1 kilometers of dewatered channel, ironically in the best quality habitat in the system. This dewatering has not been previously documented to occur in the SF Pysht and is likely the result of sediment oversupply, drought induced by climate change, and changes in hydrology associated with clearcuts. The SF Pysht represents another example of the scale of restoration necessary to support habitat forming processes in managed watersheds on the OP (email dated December 6, 2023).

The 5-year geometric observed escapement mean of winter steelhead in the Pysht River has changed as follows - 351 from 2003 to 2007, to 160 from 2008 and 2012, and 237 between 2018 and 2022 (Table A1). Hatchery supplementation and operations for the Pysht River has been ongoing since 1957 (WDG 1972, Goin 1990, McHenry et al. 1996, Goin 2009). Between 1957 and 1972 the average winter steelhead smolt release was 16,069, with a minimum of 14,220 and a maximum of 20,512 (Table A2). These releases were off-station releases and used Chambers Creek stock (McHenry et al. 1996). Between 1979 and 2008 the average winter steelhead release was 12,722 with a minimum of 9,000 and a maximum of 30,000 winter steelhead smolts (Table A2).

Prior to any hatchery supplementation, winter steelhead catch from 1948 to 1956 averaged 350 fish per year (WDG 1972). There was a minimum catch of 43 in 1950 and a maximum of 639 caught in 1953 (WDG 1972). Winter steelhead catch during the 1960s averaged 713, with a minimum of 307 and a maximum of 995 (Table A3). During the 1970s winter steelhead catch average decreased to 411, with a minimum of 130 and a maximum of 1057 (Table A3). By the 1980s winter steelhead catch averaged 390 with a minimum of 216 and a maximum of 645 (Table A3). Average winter steelhead catch further decreased to 243 in the 1990s and to 48 in the 2000s (Table A3).

## Clallam River

The Clallam River has a drainage area of 80.5 km$^2$ and receives a precipitation range between 203 and 254 cm per year, and approximately 58 km of anadromous habitat (Phinney and Bucknell 1975, Haggerty 2008). Average monthly streamflow is less 4.1 cms, but can exceed 45 cms, with peak discharge around 57 cms (Washington Department of Ecology, unpublished data). Minimum flows average 0.1 cms (Washington Department of Ecology, unpublished data). The Clallam River has had multiple peak flow events occur during the months of October through early January (WA DOE - https://apps.ecology.wa.gov/continuousflowandwq/StationDetails?sta=19H080). There are approximately 25 km of mainstem anadromous habitat, with an additional 27 km of tributary habitat thought to be passable (Haggerty 2008). In total ~85 km of anadromous fish habitat occurs in the Clallam River basin (Haggerty 2008). Washington state timberlands and industrial forest timberlands make up over 95% of the land ownership in the Clallam River basin (Haggerty 2008).

The river mouth of the Clallam River runs parallel to the Strait of Juan de Fuca and, over the decades, has often been blocked by a sand and gravel bar forming and blocking off the connection with the Strait of Juan de Fuca, due to both natural and anthropogenic impacts (NOPLE 2015). The closing of the river mouth affects both outmigrating salmon and steelhead smolts, as well as incoming adult returns such as coho salmon (Haggerty 2008, NOPLE 2015). There have been multiple efforts, over many decades, to allow for consistent direct connection between the mouth of the Clallam and the Strait of Juan de Fuca (Haggerty 2008). The combination of both natural and anthropogenic impacts (i.e. channel modifications, log rafting, milling, etc.) have resulted in large scale juvenile salmonid mortality events due to an inability to emigrate into the marine environment over time (NOPLE 2015).

The Clallam River watershed is large enough to have an agricultural component in its lower portion (Haggerty 2008). In addition, it was heavily forested so the combination led to farming and timber harvest in the late 1800's (Haggerty 2008). Much of the area was initially logged prior to the 1950s, with stand age being reduced to less than 40 years old (Haggerty 2008). In addition to agriculture and timber harvest, there was railroad building, road building (Highway 112), and systematic wood removal that occurred in the lower 10 kilometers of the Clallam (Haggerty 2008). In 1952, a total of 21 log jams were removed to "improve fish passage" (Haggerty 2008).

Current stream habitat conditions differ between the main stem Clallam River and its tributaries (Haggerty 2008). The main stem Clallam River has a minimal number of larger "key pieces" of large wood and subsequently a low number of wood jams associated with it (~0.5 logjam/km), while tributaries average ~14 logjams/km (Haggerty 2008). In contrast, pool frequency in the mainstem Clallam was relatively good in specific sections, while the majority of tributary habitat had relatively lower pool frequencies (Haggerty 2008, NOPLE 2015).

Current stream habitat conditions that have a lack of wood associated with the stream channel are likely to continue into the future, due to the poor condition of the floodplain and riparian zone (Haggerty 2008). In the lower-gradient, moderately to unconfined section of the mainstem, over 70% of the riparian zone, within 60 meters of the stream channel bank, was identified as either "impaired" or "non-functioning" (Haggerty 2008). Tributaries were identified as "impaired" for 50% of the stream length, however unlike the main stem the trajectory is towards "functioning" rather than "non-functioning" (Haggerty 2008, NOPLE 2015). Stream channel substrate size generally coarsens upstream, however, there are sections that change substrate as a function of lithology (i.e. bedrock or glacial deposits) (Haggerty 2008). Substrate size changes with full-spanning logjams in the mainstem Clallam, with finer substrate upstream of logjams, and coarsening occurring below (Haggerty 2008). Passage barriers in the Clallam were also examined and six were identified as total barriers and two were identified as partial barriers, all in tributaries of the Clallam (Haggerty 2008).

The 5-year geometric observed escapement mean of winter steelhead in the Clallam River has changed as follows - 158 from 2003 to 2007, to 105 from 2008 and 2012, and 146 between 2018 and 2022 (Table A1). Hatchery supplementation and operations for the Clallam River has been ongoing since 1981 (Table A2). Between 1981 and 1989 the average winter steelhead smolt release was 9,254, with a minimum of 5,068 and a maximum of 18,590 (Table A2). Between 1990 and 1999 the average winter steelhead smolt release was 6,478, with a minimum of 4,013 and a maximum of 13,820 (Table A2). From 2000 to 2008 the average winter steelhead smolt release was 8,884, with a minimum of 5,000 and a maximum of 14,838 (Table A2).

Prior to any hatchery supplementation, winter steelhead catch from 1962 to 1980 averaged 203 fish per year (Table A3). There was a minimum of 20 caught in 1962 and a maximum of 393 caught in 1971 (Table A3). Winter steelhead catch during the 1960s averaged 264, with a minimum of 20 and a maximum of 337 (Table A3). During the 1970s, the average winter steelhead catch reduced to 164 with a minimum of 63 and a maximum of 393 (Table A3). By the 1980s winter steelhead catch averaged 158 with a minimum of 58 and a maximum of 382 (Table A3). Average winter steelhead catch decreased to 61 in the 1990s and to 24 in the 2000s (Table A3).

## Hoko River

The Hoko River has a drainage area of 184km$^2$ and receives 203 cm of precipitation yearly (NOSC 2019). There are over 130km of anadromous habitat (William et al. 1975). Average monthly streamflow is 11.2 cms, with a peak discharge that ranges between 102 and 549 cms (NOPLE 2015). The largest annual peak flows on record have occurred over the last decade, even though there is gage data back to the early 1980s (NOPLE 2015). Flow during the summer months averages 1.98 m$^3$/s (NOSC 2019). The Hoko River is a rain-dominated watershed, so that flows increase starting in October and decrease starting in March, with low-flows occurring in August through most of September (NOSC 2019). The Hoko River has approximately 38 km of mainstem habitat, plus additional kilometers of tributary habitat (McHenry and Lichatowich 1996).

The vast majority of land in the Hoko River is commercial timberlands, however portions of the Lower Hoko River and Little Hoko have been converted to open areas or hardwood-dominated areas and purchased by Washington state parks (NOPLE 2015, personal communication with Mike McHenry, Lower Elwha Tribe, December 5, 2023). Areas in the riparian zone and floodplain have been impacted by initial conversion to railroad and then roads since the late 1800's (NOPLE 2015, NOSC 2019). Like other watersheds along the Strait of Juan de Fuca, the Hoko River has had a combination of in-stream wood clearing, riparian clearing, splash damming, and hardening and straightening of the main stem Hoko River in the lower portion of the basin (NOPLE 2015, NOSC 2019).

The combination of conversion of native conifer forest, in-stream wood loss, and channel straightening has reduced in-stream habitat conditions, disconnected floodplains, and reduced shade levels particularly during the summer, which has negatively impacted summer stream temperatures (NOPLE 2015, NOSC 2019). The loss of wood and channel simplification has also led to the Hoko River channel incising anywhere between 1.2 and 2.0 m from its historic elevation, particularly in the most impacted reaches of the Lower Hoko (NOSC 2019, Tim Abbe presentation to the Olympic Peninsula Steelhead Biological review team May 15, 2023). During this same time period of in-stream habitat change, upslope watershed conditions have also changed due to mass wasting events from forest practices and associated road networks (McHenry et al. 1994). The increase in sediment supply from such events led to large-scale changes in streambed aggradation and degradation and accompanying stream channel changes that resulted in elevated fine-sediment levels. These fine sediments reduce egg-to-fry survival of salmon and steelhead (McHenry et al. 1994).

The 5-year geometric observed escapement mean of winter steelhead in the Hoko River has changed as follows - 698 from 1998 to 2002, to 401 from 2008 and 2012, and 438 between 2018 and 2022 (Table A1). Hatchery supplementation and operations for the Hoko River have been ongoing since 1981 (Table A2). Between 1981 and 1989 the average winter steelhead smolt release was 16,690, with a minimum of 10,532 and a maximum of 24,700 (Table A2). Between 1990 and 1999 the average winter steelhead smolt release was 19,473, with a minimum of 13,971 and a maximum of 23,546 (Table A2). From 2000 to 2010 the average winter steelhead smolt release was 24,237, with a minimum of 9,658 and a maximum of 43,571 (Table A2). From 2011 to 2020 the average winter steelhead smolt release was 32,639, with a minimum of 5,480 and a maximum of 68,700 (Table A2).

Prior to any hatchery supplementation, winter steelhead catch from 1962 to 1980 averaged 298 fish per year (Table A3). There was a minimum of 66 caught in 1975 and a maximum of 393 caught in 1963 (Table A3). Winter steelhead catch during the 1960s averaged 417, with a minimum of 319 and a maximum of 658 (Table A3). During the 1970s, average winter steelhead catch reduced to 203 with a minimum of 66 and a maximum of 534 (Table A3). By the 1980s annual winter steelhead catch rose again, averaging 572 with a minimum of 356 and a maximum of 996 (Table A3). Average winter steelhead catch then decreased to 421 in the 1990s and to 311 in the 2000s (Table A3).

## Sekiu River

The Sekiu River has a drainage area of 85 $km^2$ , over 80k of anadromous habitat, and receives 213 cm of average annual precipitation (Williams et al. 1975, NOSC 2019). Like other watersheds on the Strait of Juan de Fuca, the Sekiu River is a rain-dominated system with low flows of less than 0.14 cms to a peak of 28 cms (NOPLE 2015). The Sekiu River has approximately 14 km of anadromous mainstem habitat, in addition to tens of kilometers of tributary habitat (NOPLE 2015, McHenry and Lichatowich 1996). The Sekiu River estuary is limited in size, and has also been impacted by infrastructure constraints such as highway 112 (NOPLE 2015)

The vast majority of land in the Sekiu River is commercial timberlands, with ownership dominated by private timber companies and the state of Washington (NOPLE 2015). ). A portion of the watershed is part of the Makah Tribal Reservation (NOPLE 2015). Areas in the riparian corridor and floodplain have been impacted, in large part, by the Sekiu River mainline road, which goes along the main stem up to the North and South Fork Sekiu (NOPLE 2015). Roads, such as the Sekiu River mainline, permanently impact the riparian zone, reduce wood loadings, increase levels of fine sediment inputs into the main stem Sekiu and adjoining tributaries, and cut off floodplain habitats (McHenry et al. 1994; McHenry and Lichatowich 1996; Smith 2000; Currence 2001; NOPLE 2015). The high fine-sediment levels due to surface erosion and landslides, coupled with the lack of in-stream wood, has led to reduced spawning gravel quantity and quality in the mid 1990s and early 2000s (McHenry et al. 1994, Currence 2001). Stream temperatures in specific areas of the Sekiu River have been identified as “impaired” based on the Washington Department of Ecology 303(d) definition (NOPLE 2015).

We do not have information on Sekiu River run size or escapement. Hatchery supplementation and operations for the Sekiu River have been ongoing since 1988 (Table A2). For 1988 and 1989 the average winter steelhead smolt release was 5,047 (Table A2). Between 1990 and 1999 the average winter steelhead smolt release rose to 8,956, with a minimum of 4,773 and a maximum of 12,129 (Table A2). From 2000 to 2010 the average winter steelhead smolt rose further to 11,158, with a minimum of 5,833 and a maximum of 22,912 (Table A2). From 2011 to 2020 the average winter steelhead smolt release was even higher at 14,936, with a minimum of 5,580 and a maximum of 21,175 (Table A2).

Prior to any hatchery supplementation in the Sekiu River, winter steelhead catch from 1962 to 1988 averaged 77 fish per year (Table A3). There was a minimum of 21 caught in 1977 and a maximum of 191 caught in 1986 (Table A3). Summer steelhead harvest during that time was 4 per year (Table A3). Winter steelhead catch during the 1960s average 78, with a minimum of 49 and a maximum of 141 (Table A3). During the 1970s the average winter steelhead catch declined to 43 with a minimum of 16 and a maximum of 116 (Table A3). By the 1980s winter steelhead catch averaged 110 with a minimum of 78 and a maximum of 157 (Table A3). Average winter steelhead catch in the 1990s was 52 and 98 between 2000 and 2010 (Table A3). From 2011 to 2020 average winter steelhead catch was 343, with a minimum of 47 and a maximum of 864 (Table A3).

## Westside watersheds

### Quillayute River Basin

The Quillayute River Basin (1,573km$^2$) consists of four major river basins including the Sol Duc River (603km$^2$), the Calawah River (352km$^2$), the Bogachiel River (395km$^2$), and the Dickey River (223km$^2$), in addition to the mainstem Quillayute River (Williams et al. 1975). Combined, there are over 1,200km of stream drainage, of which over 600km is anadromous habitat (Williams et al. 1975). The over 1,600km$^2$ watershed is relatively lower in elevation and includes approximately 80% of the watershed in the rain dominated and transition dominated hydrography zone. The remaining 20% is in a snow-dominated hydrography. The Quillayute includes portions of the Olympic National Park (34%) and the Olympic National Forest (27%), but is predominantly state and private timberlands, and to a lesser extent the Quillayute Tribal reservation (a combined 39%). The watershed lies within a region of temperate rainforest and is dominated by Sitka spruce (*Picea sitchensis*), red alder (*Alnus rubra*), bigleaf maple (*Acer macrophyllum*), western red cedar (*Thuja plicata*), and Douglas fir (*Pseudotsuga mensiesii*) in the lowlands, with western hemlock (*Tsuga heterophylla*) and silver fir (*Abies amabilis*) in the higher elevations (Smith 2000).

The Quillayute River main stem starts at the confluence of the Sol Duc and Bogachiel Rivers, is approximately 9.0 kilometers in length, and flows in a westerly direction entering the Pacific Ocean near LaPush, WA (Willliams et al. 1975). The average wetted width is approximately 55 meters (Williams et al. 1975). The Dickey River comes into the main stem Quillayute River at Rkm 2.5 and includes an additional 61 km of stream habitat, including the West Fork, Middle Fork, and East Fork Dickey River (Williams et al. 1975). The average wetted width is approximately 23 m near the confluence with the Quillayute (Williams et al. 1975). The Bogachiel River has approximately 53km of main stem anadromous habitat, as well as an additional ~130km of tributary habitat (Williams et al. 1975). Wetted widths range from 18 to 36 m wide in the main stem and tributaries range from 1 to 11 mkm wide (Williams et al. 1975). The Calawah River system meets Bogachiel at ~Rkm 14, adding an additional 84 Rkm of main stem as well as another 78km of tributary habitat (William et al. 1975). Wetted widths on the main Calawah range from 7 meters to 27 meters, while the tributaries range from 1 to 7 meters wetted width (Williams et al. 1975). The Sol Duc River is over 100 km in length and includes over 175 kilometers of tributary habitat (Williams et al. 1975). Winter wetted widths in the main stem Sol Duc range from 13 meters in the headwaters, to 25 to 39 meters in the middle portion of the Sol Duc, to over 40 meters in the lowermost portion (Williams et al. 1975). Summer wetted widths range from 10 to 21 to 27 meters (Williams et al. 1975). Tributary widths range between 2.5 and 10 meters in wetted widths (Williams et al. 1975).

The climate on the western portion of the Olympic Mountains is temperate, with an average annual precipitation of 350 cm between 1980 and 2010 (Jaeger et al. 2023), most of which occurs between November through March as rain or snow events (Jaeger et al. 2023). Peak flows, for example, in the Calawah River are greatest during late fall/early winter months (November, December, and January), when ~80% of all peak flow events have occurred between 1975 and 2021 (https://nwis.waterdata.usgs.gov/nwis/peak?site_no=12043000&agency_cd=USGS&format=html). Peak flows in the Calawah River during those years average approximately 661 cms (https://nwis.waterdata.usgs.gov/nwis/peak?site_no=12043000&agency_cd=USGS&format=html). Monthly low-flow in the Calawah, as well as the other watersheds within the Quileute, occur during August or September. Average low flow in August and September in the Calawah River is approximately 5 cms. (https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&search_site_no=12043000&format=sites_selection_links). Spring snow melt occurs during the months of April through June (Jaeger et al. 2023). Summer low flows has been considered a general limiting factor to salmon and steelhead production in the Quileute river basin (Williams et al. 1975).
Summer low flows have decreased over time in the Calawah River basin, where the average low flow in the in late 1970s through the 1990s was 2.0cms, while in the 2000s average summer low flow has been 1.5cms (https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&search_site_no=12043000&format=sites_selection_links). Summer low flows are predicted to decrease anywhere between 5% and 43% by 2040 in the Quillayute River Basin (Wenger et al. 2010, USBOR 2014, USFS-OSC 2022). The largest changes are predicted to occur in the Sol Duc, Upper Bogachiel, and Quillayute River proper (Wenger et al. 2010, USBOR 2014, USFS-OSC 2022). Peak flows have slightly increased from 1975 to 2010 from a decadal average of 585cms in the 1980s to 721cms average during the first decade of the 2000s (https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&search_site_no=12043000&format=sites_selection_links, Jaeger et al. 2023). In both the Calawah and Bogachiel rivers, it is becoming common for peak flows to be at or above flood stage. These trends could threaten salmon habitat and other aquatic ecosystem functions (NWIFC 2020).

The Quillayute River basin, similar to all the other watersheds in the OP DPS, has a history of timber harvest and the associated impacts, but this varies according to land ownership. The Calawah River basin, as an example, had intensive salvage logging and road building occur after the Great Forks fire of 1951 (Jaeger et al. 2023). In contrast, the Bogachiel River was not impacted by 1951 fire, and has a larger portion of the watershed in Olympic National Park, resulting in less timber harvest and road building (Jaeger et al. 2023).

The sub-basins within the Quillayute vary in terms of current forest cover condition primarily due to land ownership (NWIFC 2020). Forest cover condition in the Dickey River sub-basin is currently rated as moderate, while the upper portion of the Bogachiel, multiple sub-basins of the Calawah including the Sitkum and Elk Creek, and the upper Sol Duc are considered "healthy" in terms of forest cover conditions (NWIFC 2020). According to the State of our Watersheds report (NWIFC 2020) average timber harvest rate has decreased from 2016 to 2019 by approximately 30% relative to 2011 to 2015.

Changes to stream channel morphology and sediment have also resulted due to historic timber harvest and road building practices (Jaeger et al. 2023). Similar to the Hoh River basin, for portions of the Quillayute basin, such as the Calawah River basin, average stream channel width consistently increased from 1935 through the 1980s, with a trend of stream width reduction starting in the 1990s onward (Jaeger et al. 2023). These changes in stream width correlated to increases in peak flows during that time period as well (Jaeger et al. 2023). This was not the case in the Bogachiel River basin (Jaeger et al. 2023).

Road density, in general, follows the same patterns for each of the sub-basins, and is dictated by land ownership. Industrial timberlands (private, state, and national) have the higher road densities, up to 14.2 square kilometers per kilometer in specific private timberlands, averaging over 7.7 $km^2$/km in much of the Dickey, portions of the Calawah, Lower Bogachiel, and Mid to Lower Sol Duc Rivers (NWIFC 2020). Road density decreased in ONP (NWIFC 2020). Road crossings have been a focus and many have been fixed for salmon and steelhead passage (NWIFC 2020). However state, county, and other roads still have impassible culverts that result in a decrease in the amount of available habitat for steelhead to utilize in the Quileute Basin (NWIFC 2020).

The 5-year geometric observed escapement mean of winter steelhead in the Quillayute-Bogachiel River has changed as follows – 2,957 from 1998 to 2002, 1,972 from 2003 to 2007, 1,710 from 2008 to 2012, 1,221 from 2013 to 2017, and 1,166 from 2018 to 2022 (Table A1). The 5-year geometric observed escapement mean of winter steelhead in the Calawah River has changed as follows – 4,798 from 1998 to 2002, 3,122 from 2003 to 2007, 2,732 from 2008 to 2012, 2,526 from 2013 to 2017, and 2,551 from 2018 to 2022 (Table A1). The 5-year geometric observed escapement mean of winter steelhead in the Sol Duc River has changed as follows – 5,696 from 1998 to 2002, 3,897 from 2003 to 2007, 2,980 from 2008 to 2012, 2,553 from 2013 to 2017, and 3,483 from 2018 to 2022 (Table A1). The 5-year geometric observed escapement mean of winter steelhead in the Dickey River has changed as follows – 699 from 1998 to 2002, 344 from 2003 to 2007, 384 from 2008 to 2012, 268 from 2013 to 2017, and 423 from 2018 to 2022 (Table A1).

Hatchery supplementation and operations for the Quillayute-Bogachiel River has been ongoing since 1981 (Table A2). Between 1980 and 1989 the average winter steelhead smolt release was 63,742 (STable 2). Between 1990 and 1999 the average winter steelhead smolt release was 109,671, with a minimum of 9,120 and a maximum of 227,322 (Table A2). From 2000 to 2010 the average winter steelhead smolt release was 113,464, with a minimum of 53,000 and a maximum of 295,000 (Table A2). From 2011 to 2021 the average winter steelhead smolt release was 111,015, with a minimum of 45,000 and a maximum of 130,419 (Table A2).

Hatchery supplementation and operations for the Calawah River has been ongoing since 1981 (Table A2). Between 1980 and 1989 the average winter steelhead smolt release was 50,213 (Table A2). Between 1990 and 1999 the average winter steelhead smolt release was 89,334, with a minimum of 10,293 and a maximum of 117,998 (Table A2). From 2000 to 2010 the average winter steelhead smolt release was 64,500, with a minimum of 35,000 and a maximum of 109,500 (Table A2). From 2011 to 2021 the average winter steelhead smolt release was 53,193, with a minimum of 42,400 and a maximum of 56,357. (Table A2).

Hatchery supplementation and operations for the Calawah River summer run has been ongoing since 1981 (Table A2). Between 1980 and 1989 the average summer steelhead smolt release was 14,023 (Table A2). Between 1990 and 1999 the average summer steelhead smolt release was 29,214, with a minimum of 10,461 and a maximum of 37,480 (Table A2). From 2000 to 2010 the average summer steelhead smolt release was 37,098, with a minimum of 30,000 and a maximum of 83,655 (Table A2). From 2011 to 2021 the average summer steelhead smolt release was 36,115, with a minimum of 31,486 and a maximum of 49,500 (Table A2).

Hatchery supplementation and operations for the Sol Duc River has been ongoing since 1981 (Table A2). Between 1980 and 1989 the average winter steelhead smolt release was 28,253 (Table A2). Between 1990 and 1999 the average winter steelhead smolt release was 21,287, with a minimum of 14,300 and a maximum of 26,507 (Table A2). From 2000 to 2010 the average winter steelhead smolt release was 22,046, with a minimum of 16,000 and a maximum of 37,410 (Table A2).

Hatchery supplementation and operations for the Sol Duc River summer run has been ongoing since 1978 (Table A2). Between 1980 and 1989 the average summer steelhead smolt release was 24,612 (Table A2). Between 1990 and 1999 the average summer steelhead smolt release was 49,194, with a minimum of 4,929 and a maximum of 100,500 (Table A2). From 2000 to 2007 the average summer steelhead smolt release was 64,625, with a minimum of 31,150 and a maximum of 101,000 (Table A2).

Average winter steelhead catch in the Quillayute system during the 1980s was 3,395 with a minimum of 2,013 and a maximum of 7,561 (Table A3). Average winter steelhead catch in the 1990s was 5,178 with a minimum of 1,947 and a maximum of 7,100 (Table A3). Between 2000 and 2010 average winter steelhead catch was 3,831 with a minimum of 2,053 and a maximum of 7,226 (Table A3). From 2011 to 2022 average winter steelhead has been 2,033, with a minimum of 803 and a maximum of 3,719 (Table A3).

Wild summer steelhead sport harvest for the entire Quillayute system in the 1980s was 979 with a minimum of 51 and a maximum of 2,226 (Table A3). Summer steelhead sport catch during the 1990s was 388 with a minimum of 205 and a maximum of 642 (Table A3). Summer steelhead sport catch between 2000 and 2003 was 756 with a minimum of 357 and a maximum of 1,041 (Table A3). Other catch during that time reported as hatchery and wild combined was 102, 180, and 2,946 respectively (Table A3).

## Hoh River

The Hoh River originates at the Hoh Glacier on Mt. Olympus and flows approximately 90 kilometers to the Pacific Ocean (Williams et al. 1975). The 770 $km^2$ watershed includes portions of the Olympic National Park, the Olympic National Forest, state and private timberlands, and the Hoh Tribal reservation. It includes over 350 km of anadromous stream habitat, approximately

89 kilometers of mainstem and the rest in tributaries (Williams et al. 1975). The amount of floodplain habitat is also large but the precise amount has not been quantified. Besides the Hoh Glacier, the drainage includes several other glaciers including the Blue, White, Hubert, and Ice River Glaciers (Williams et al. 1975), all of which serve to sustain summer streamflows. Some notable tributaries include the South Fork Hoh River, Winfield Creek, Elk Creek, Owl Creek, Anderson Creek, Braden Creek, Willoughby Creek, Alder Creek, Pins Creek, and Hell Roaring Creek (Williams et al. 1975). Channel wetted widths along the mainstem Hoh River range between 13 and 27 meters during the summer and 14 to 36 meters in the winter (Williams et al. 1975).

The Hoh watershed has the highest precipitation levels in Washington State (U.S. Weather Bureau 1965, NWIFC 2020). Average annual precipitation ranges from about 225 cm near the Pacific Coast to 600 cm in the Olympic Mountains (U.S. Weather Bureau 1965). Peak flows in the Hoh River are greatest during winter months (e.g., November to February), and average approximately 985 cms (https://nwis.waterdata.usgs.gov/nwis/peak?site_no=12041200&agency_cd=USGS&format=html). Monthly low-flow typically occurs in August or September, averaging approximately 33 cms (https://nwis.waterdata.usgs.gov/nwis/peak?site_no=12041200&agency_cd=USGS&format=html).

The watershed lies within a region of temperate rainforest and is dominated by Sitka spruce (*Picea sitchensis*), red alder (*Alnus rubra*), bigleaf maple (*Acer macrophyllum*), western red cedar (*Thuja plicata*), and Douglas fir (*Pseudotsuga mensiesii*) in the lowlands, with western hemlock (*Tsuga heterophylla*) and silver fir (*Abies amabilis*) in the higher elevations (Smith 2000). The vast majority of land in the Hoh River is forested, with ownership dominated by the Federal Government (Olympic National Park, ONP), state of Washington, and private timber companies (NWIFC 2020). The Hoh Tribal Reservation occupies a portion of the watershed, near the mouth of the Hoh River (NWIFC 2020).

The Hoh River Basin, like many of the watersheds in the OP DPS, has a history of timber harvest and the associated impacts (Cederholm et al. 1981, Logan et al. 1991, NWIFC 2020). Historic land use practices in the Hoh, and the other OP DPS watersheds, included forest harvest without stream buffers, the removal of instream wood, high-density road construction and frequent use, and harvesting large proportions of watersheds (Martens et al. 2019). These practices resulted in deleterious changes to sediment supply, wood supply, the amount and condition of streamflow, and stream channel morphology (Cederholm et al. 1981, Logan et al. 1991, McHenry et al. 1998, Abbe and Montgomery 2003, NWIFC 2020).

Prior to the 1990s, elevated in-stream sediment levels due to the harvest of timber on steeply sloped hillsides constituted the majority of sediments associated with fisheries survival impacts. Impacts occurred in 10 tributaries, the South Fork Hoh, and portions of the mainstem Hoh River (Cederholm et al. 1991, Hatten 1991, McHenry 1991, Logan et al. 1991). In addition, further increases in sediment supply, hypothesized to result from glacial retreat over the last 80 years in the headwaters, is thought to be the primary driver for an increase in mainstem stream channel width and braiding (East et al. 2017). It is important to note that west-side Olympic Peninsula rivers are dynamic systems that have shown spatial and temporal variation in stream migration and channel characteristics between 1939 and 2013 (East et al. 2018). Given the Hoh basin's large proportion of high alpine terrain, it has also been noted that 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 of permafrost, and mass wasting of recently deglaciated valley walls, all drivers of changes to downstream channel characteristics (East et al. 2017).

Wood loadings continue to decrease. Density of large wood in OP streams managed by the United States National Forest Service (USFS) has decreased since 2002 (~3.0 wood pieces greater than 60cm DBH per 100m) by almost 50% in 2018 (~1.5 wood pieces greater than 60cm DBH per 100m) (Dunham et al. 2023). This trend is similar to what occurs on second-growth forests in state timberlands, where wood densities and key pieces per 100 meters are lower in tributary habitats than what is currently observed in unmanaged streams of Western Washington, and below what was measured decades ago (McHenry et al. 1998, Fox and Bolton 2007, Martens et al. 2019).

Timber harvesting reduces hydrologic maturity, and can lead to changes in peak and mean daily flow of streamflow at watershed, sub-basin and basin level, as well as an altered flow regime. All of which are significant habitat factors limiting salmonid production in this basin (NWIFC 2020). There have been changes to the magnitude and frequency of timber harvest activities (NWIFC 2020). Timber harvest magnitude and rate has decreased since 2016 from $162km^2$ to $47km^2$ or $32km^2$/year to $16km^2$/year (NWIFC 2020).

High road densities can also lead to deleterious impacts to both salmon and steelhead spawning and rearing areas due to increased fine sediment levels in spawning areas, as well as road failures and subsequent increased landslide activities (Cederholm et al. 1981, Guthrie 2002). The Hoh River basin has road density values of $7.77km/km^2$ outside the ONP while road density levels are less than $2.59km/km^2$ inside the ONP boundary (NWIFC 2020). The high road densities outside the ONP were built for timber harvest (NWIFC 2020). Barriers due to road culverts blocking both spawning and rearing habitat are always a potential consequence of road construction and a likely impact to steelhead in forested watersheds. There are almost 300 culverts identified by the Hoh River road maintenance and abandonment plan (RMAP) (NWIFC 2020). According to the State of our Watersheds report, 80% of those culverts have been repaired, while 20% remain barriers. There are also an additional 134 barriers outside the plan, of which approximately 50% are impassable (NWIFC 2020).

While cumulatively these impacts have been large over space and time, the Hoh River Basin still has a core of natural watershed processes and associated habitat characteristics. These include a large forested floodplain, relative to other watersheds, that is still intact and functioning, and a majority of the watershed lying within ONP, especially its headwaters (Ericsson et al. 2022). Thus, efforts to protect, restore, and increase the overall resiliency of the Hoh River are being developed and implemented to secure these core natural assets (Ericsson et al. 2022).

Stream flow, in the form of average annual stream discharge, summer low-flows, and peak flood-flows are typically indicators of stream habitat quality as well as a determinant of the amount of habitat quantity (refs). For the Olympic Peninsula as a whole there has been a decline in average annual discharge, more so than other parts of the western USA where United States

National Forest Service (USFS) lands occur (Dunham et al. 2023). Streamflow assessed since 1976 found that the mean low-flow has decreased between 13% and 48% for the Hoko, Hoh, Calawah, and Quinault Rivers, with the Hoh River decreasing an estimated 15% (NWIFC 2023). The proportion of bankfull width that is wetted has decreased on USFS lands across the Olympic Peninsula from almost 70% of bankfull width prior from 2002, to less than 50% of bankfull width in 2018 (Dunham et al. 2023). Summer low-flows (i.e seven-day minimum low-flow) have been documented to be decreasing at a rate of 0.14 cms/year over the last 40 years (NWIFC 2020).

An assessment of peak flood flows between 1976 and 2019 found that peak flows have increased for the Hoko, Hoh, Calawah, and Quinault Rivers, by 5% to 18% with the Hoh River increasing by 18.4% (NWIFC 2023). Examination of the peak discharges for the West side OP DPS watersheds found that the two-year flood event is 10 to 35% greater over the last 40 years, relative to over the entire length of the stream-gage record (East et al. 2017). In the Hoh River basin, the three largest peak flow events recorded have occurred since 2002 (East et al. 2018). The 2-year flood peak calculated for the Hoh River for water years 1978–2013 was 1024 cms, whereas the 2-year flood for the entire period of record at the Hoh River gaging station (12041200) was 924 cms (East et al. 2018). The general increase in flood activity along the OP after the mid-1970s coincided with the onset of a wet phase of the Pacific Decadal Oscillation (PDO, an index of monthly sea-surface temperature anomalies over the North Pacific; Mantua et al. 1997). This mid-1970s climatic transition has been identified as a major atmospheric and hydrologic shift that affected a large region of the Pacific in both the northern and southern hemispheres (Castino et al. 2016, East et al. 2018).

Stream temperatures, particularly during the summer months (i.e. August), have changed on USFS lands in the OP DPS. The seven-day average of maximum daily temperature has increased from below 14° C in the early 2000s to almost 16° C in 2018 (Dunham et al. 2023). Several tributaries in the Hoh River basin have exceeded the 16° C standard for decades, including Winfield Creek, Nolan Creek and Owl Creek (NWIFC 2023). Others such as Elk Creek meet the Washington State Water Quality standard (NWIFC 2023).

One of the largest predicted changes, with respect to changing climatic conditions, is the decline in glacial extent (Riedel et al. 2015), particularly for the larger west side watersheds. Over the past several decades, glacier decline in the Olympics was greater than in the Cascades and southern Coast Mountains, and is more comparable with Vancouver Island (Riedel et al. 2015). Riedel et al. (2015) estimate that the glacial contribution to summer streamflow has declined ~20% in the past 30 years, but still remains significant for the Hoh River. In the other Westside OP DPS watersheds, glaciers contribute less than 5% to summer streamflow (Riedel et al. 2015). The loss in glaciers over the past 30 years appears to be a result of mean air temperature increases, and illustrates how sensitive these relatively small, thin, and low-elevation glaciers are to climate change (Riedel et al. 2015, East et al. 2018). Continued loss of glaciers will directly impact aquatic ecosystems through higher stream temperatures and lower summer base flows. Summer low flows are predicted to decrease anywhere between 25% and 50% by 2040 for the majority of the Hoh River Basin (Wegner et al. 2010, USBOR 2014, USGS-OSC 2022). The largest changes are predicted to occur in the Upper Hoh basin (Wegner et al. 2010, USBOR 2014, USFS-OSC 2022). Peak flows have slightly increased from 1975 to 2010 from a decadal average of 585cms in the 1980s to 721cms average during the first decade of the 2000s (https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&search_site_no=12043000&format=sites_selection_links, Jaeger et al. 2023). Large-scale flood events (i.e. greater than 25 year recurrence interval) are predicted to increase between 10% and 25% by 2040 (Wegner et al. 2010, USBOR 2014, USFS-OSC 2022).

The 5-year geometric observed escapement mean of winter steelhead in the Hoh River has changed as follows – 3,088 from 1998 to 2002, 2,254 from 2003 to 2007, 2,677 from 2008 to 2012, 2,314 from 2013 to 2017, and 2,735 from 2018 to 2022 (Table A1).

Hatchery supplementation and operations for the Hoh River has been ongoing since 1980 (Table A2). Between 1980 and 1989 the average winter steelhead smolt release was 122,072 (Table2). Between 1990 and 1999 the average winter steelhead smolt release was 95,256, with a minimum of 92,845 and a maximum of 101,881 (Table A2). From 2000 to 2010 the average winter steelhead smolt release was 100,256, with a minimum of 48,625 and a maximum of 161,548 (Table A2). From 2011 to 2021 the average winter steelhead smolt release was 69,917, with a minimum of 14,200 and a maximum of 148,765 (Table A2).

During the 1980s winter steelhead catch average was 2,009 with a minimum of 1,607 and a maximum of 2,800 (Table A2). Average winter steelhead catch in the 1990s was 1,544 with a maximum of 2,454 (Table A3). Between 2000 and 2010 average winter steelhead catch was 1,374 with a minimum of 437 and a maximum of 2,584 (Table A3). From 2011 to 2022 average winter steelhead has been 765, with a minimum of 343 and a maximum of 1,358 (Table A3).

Wild summer steelhead sport harvest in the 1980s was 200 with a minimum of 136 and a maximum of 257 (Table A3). Summer steelhead sport catch during the 1990s was 37 with a minimum of 3 and a maximum of 104 (Table A3). Summer steelhead sport catch between 2000 and 2003 was 8 with a minimum of 4 and a maximum of 18 (Table A3). Other catch during that time reported as hatchery and wild was 425, 196, and 118 respectively (Table A3).

## Queets River

The Queets River originates at the Humes Glacier on Mt. Olympus and flows approximately 85 kilometers to the Pacific Ocean (Williams et al. 1975). The 530 $km^2$ watershed includes portions of the Olympic National Park, the Olympic National Forest, state and private timberlands, and the Quinault Tribal reservation. It includes over 640 km of anadromous stream habitat, approximately 83 kilometers of mainstem, 159 km of major tributaries, and over 400 km of smaller tributary habitat (Williams et al. 1975). The amount of floodplain habitat is also large but the precise amount has not been quantified. Besides the Humes Glacier, the drainage includes several other glaciers including the Jeffers and Queets Glaciers (Williams et al. 1975), all of which serve to sustain summer streamflows. Some notable tributaries include the Clearwater River, Salmon River, Matheny Creek, Sams River, and Tshletshy Creek (Williams et al. 1975). Channel wetted widths along the mainstem Queets River range between 16 and 32 meters during the summer and 27 to 46 meters in the winter (Williams et al. 1975).
Peak flows in the Queets River are greatest during October through March with the majority of peak flows occurring November through January since the 1930s

(https://nwis.waterdata.usgs.gov/usa/nwis/peak/?site_no=12040500). Average peak flow is 1,992 cms, with a maximum discharge of 3,766cms and a minimum peak flow of 932 cms (https://nwis.waterdata.usgs.gov/usa/nwis/peak/?site_no=12040500). Monthly low-flow typically occurs in August or September, averaging approximately 27 and 38 cms, respectively (https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&search_site_no=12040500&format=sites_selection_links).

Similar to other West side watersheds, the Queets lies within a region of temperate rainforest and is dominated by Sitka spruce *(Picea sitchensis*), red alder (*Alnus rubra*), bigleaf maple (*Acer macrophyllum*), western red cedar (*Thuja plicata*), and Douglas fir (*Pseudotsuga mensiesii*) in the lowlands, with western hemlock (*Tsuga heterophylla*) and silver fir (*Abies amabilis*) in the higher elevations (Smith 2000). The vast majority of land in the Queets River is forested, with ownership dominated by the Federal Government (Olympic National Park, ONP, 75%), Quinault tribal lands (13%), state of Washington (10%), and private timber companies (2%) (NWIFC 2020).

The Clearwater watershed, a tributary to the Queets River Basin, has a history of timber harvest and the associated impacts (Cederholm and Salo, 1979, Cederholm et al. 1981, Logan et al. 1991, NWIFC 2020). Historic land use practices in the Queets Basin, and the other OP DPS watersheds, included forest harvest without stream buffers, the removal of instream wood, high-density road construction and frequent use, and harvesting large proportions of watersheds (Martens et al. 2019). These practices resulted in deleterious changes to sediment supply, wood supply, the amount and condition of streamflow, and stream channel morphology (Cederholm et al. 1981, Logan et al. 1991, McHenry et al. 1998, Abbe and Montgomery 2003, NWIFC 2020). Historic logging in the Queets River basin, even though a large portion of the watershed is in Olympic National Park and has a protected floodplain corridor, was intensive and extensive (McHenry et al. 1998). By 1971, over 2 billion board feet of timber was harvested from Washington state lands on the Queets and Clearwater Rivers (Brown 1990, McHenry et al. 1998). Road construction during this time, included techniques that are now known to be sub-standard and resulted in road failures, increased landslide rates, and reduced stream habitat conditions particularly in some of the tributaries such as the Clearwater River basin (Cederholm and Salo, 1979, McHenry et al. 1998). During the 1970s and 1980s, landslides rates were 168 times those of natural reference areas (McHenry et al. 1998). In addition, instream sediment levels were 2.5 times the magnitude of unlogged Olympic Peninsula streams and salmon egg survival to fry emergence was reduced due to the relatively high fine sediment levels in streambed spawning gravels due to the density of logging roads (Cederholm and Salo, 1979, Tagart 1984, Cederholm and Reid 1987, McHenry et al. 1998).

Similar to other watersheds in the OP DPS, there has been a large reduction in wood loadings due to riparian harvest, instream wood removal, and logging of floodplain forests. These actions led to changes in stream habitat conditions in tributaries and the mainstem areas, particularly in private and state timberlands (Bilby 1984, McHenry et al. 1998, Abbe and Montgomery 2003, Martens et al. 2014, 2019, 2020). Impacts include the loss of pools in smaller streams and a decrease in stabilizing wood jams in the mainstem, which led to loss of stream channel complexity in larger streams (Abbe and Montgomery 2003, Martens et al. 2019). Even with ~25 years of more protective timber harvest regulations related to riparian zones important salmonid habitat components such as instream wood and pools have not recovered through natural recruitment of wood (Martens and Devine 2023). The estimated timeline for recovery of these remaining wood loading degradations could range between 100 and 225 years (Stout et al. 2018, Martens and Devine 2023).

The Queets River basin still has a significant portion of its main stem, floodplain, and associated habitats intact due to current land ownership and associated protections. As a result, important functions and habitat conditions still exist due to wood debris (LWD) accumulations that arise from the interaction of river and valley bottom (Abbe and Montgomery 2003). Wood debris (WD) accumulations, due to recruitment of historic and current large trees, in the Queets River basin, result in stable in-stream structures that significantly influence river morphology (Abbe and Montgomery 2003, Latererell and Naiman 2007). WD accumulations result in channel anabranching, floodplain topography, and establishment of long-term riparian refugia for old-growth forest development (Abbe and Montgomery 2003). Instream supplies of wood are a mixture of new and old logs from nearby and upstream forests, sustained by the recapture and transport of stockpiled remnant logs during periods when new inputs are low (Latererell and Naiman 2007).

Nevertheless, signals from changes to flood activity still occur in the Queets River basin (East et al. 2017). The 2-year flood recurrence interval (Q2) magnitudes over the most recent ~4 decades have been 12% greater than over the entire length of each stream-gage record in the Queets River basin (East et al. 2017). During this time period there was streambed aggradation (0.33m) that followed a large flood event (Q50) followed by a decrease of 0.2m in the subsequent decade (East et al. 2017). While there has been stream channel widening and narrowing, stream channel widths are similar to what was measured in the early 1900s (BLM 2016, East et al. 2017).

As stated previously, glaciers in the Olympic Mountains have retreated rapidly over recent decades (East et al. 2017). Since 1980, ONP has lost 34% of its glacial ice area and 82 glaciers have disappeared entirely (Riedel et al., 2015), with the Queets losing 9% of its glacial ice area during this time period (Riedel et al. 2015, East et al. 2017). Future consideration, with respect to climatic change in the Queets, and other watersheds on the OP include fewer years with a large snowpack, more rainfall than snow, and short-term intense rainfall potentially resulting in more frequent winter flood activity (East et al. 2017). Models suggest that the Olympic Mountains are especially prone to increased flooding activity (Tohver et al., 2014, East et al. 2017), coupled with the additional hydrologic alterations anticipated from glacial retreat, particularly related to late-summer streamflow (Riedel et al., 2015, East et al. 2017). Summer low flows are predicted to decrease anywhere between 25% and over 50% by 2040 for the majority of the Queets River Basin (Wegner et al. 2010, USBOR 2014, USFS-OSC 2022). The largest changes are predicted to occur in the Upper Queets Basin (Wegner et al. 2010, USBOR 2014, USFS-OSC 2022). Large-scale flood events (i.e. greater than 25-year recurrence interval) are predicted to increase between 10% and 25% by 2040 (Wegner et al. 2010, USBOR 2014, USFS-OSC 2022).

The 5-year geometric observed escapement mean of winter steelhead in the Queets River has changed as follows – 4,111 from 1998 to 2002, 5,634 from 2003 to 2007, 4,613 from 2008 to 2012, 3,583 from 2013 to 2017, and 2,931 from 2018 to 2022 (Table A1).

Hatchery supplementation and operations for the Queets River has been ongoing since 1981 (Table A2). Between 1981 and 1989 the average winter steelhead smolt release was 133,600 (Table A2). Between 1990 and 1999 the average winter steelhead smolt release was 145,527, with a minimum of 83,483 and a maximum of 202,638 (Table A2). From 2000 to 2010 the average winter steelhead smolt release was 162,418, with a minimum of 149,874 and a maximum of 176,580 (Table A2). From 2011 to 2021 the average winter steelhead smolt release was 174,743, with a minimum of 158,204 and a maximum of 204,002 (Table A2).

During the 1980s winter steelhead catch average was 4,535 with a minimum of 3,279 and a maximum of 6,291 (Table A3). Average winter steelhead catch in the 1990s was 2,938 with a minimum of 1,473 and a maximum of 4,977 (Table A3). Between 2000 and 2010 average winter steelhead catch was 1,538 with a minimum of 575 and a maximum of 2,270 (Table A3). From 2011 to 2022 average winter steelhead has been 1,607, with a minimum of 275 and a maximum of 3,155 (Table A3).

Summer steelhead sport catch in the 1960s was 217 with a minimum of 111 and a maximum of 2299 (Table A3). Summer steelhead sport catch during the 1970s was 180 with a minimum of 53 and a maximum of 345 (Table A3). Summer steelhead sport catch in the 1980s was 190 with a minimum of 78 and a maximum of 310 (Table A3). Hatchery and wild summer steelhead were detonated starting in 1986 (Table A3). Summer steelhead sport catch in the 1990s was 32 with a minimum of 6 and a maximum of 69 (Table A3). Summer steelhead sport catch in the 2000s was 40 with a minimum of 8 and a maximum of 84 (Table A33). Summer steelhead sport catch in the 2010s was 30 with a minimum of 12 and a maximum of 72 (Table A3).

## Quinault River

The 490-km$^2$ Quinault River originates in Olympic National Park and flows approximately 111 kilometers to the Pacific Ocean (Williams et al. 1975). It includes over 460 km of anadromous stream habitat and approximately 111 kilometers of mainstem (Williams et al. 1975). Some notable features and tributaries include Lake Quinault, the North Fork Quinault River, Graves Greek, and Cook Creek (Williams et al. 1975). Land ownership varies as a function of the area below and above Lake Quinault. Below Lake Quinault ownership is predominantly the Quinault Tribal reservation (~80%), followed by Olympic National Forest (~14%), and private timberlands (~7%). Above Lake Quinault ownership is dominated by Federal lands (~95%), followed by Quinault Tribal reservation (~4.5%), and private lands (<0.5%),
Peak flows in the Quinault River above the lake are greatest during November through March with the majority of peak flows occurring November through January since the 1930s (https://nwis.waterdata.usgs.gov/usa/nwis/peak/?site_no=12039500). Average peak flow is 697 cms, with a maximum discharge of 1,489 cms and a minimum peak flow of 189 cms (https://nwis.waterdata.usgs.gov/usa/nwis/peak/?site_no=12040500). Monthly low-flow typically occurs in August or September, averaging approximately 26 and 28 cms, respectively (https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&search_site_no=12039500&format=sites_selection_links). Like the other large watersheds draining the west side of the OP, there have been changes to peak flows (East et al. 2017). On the Quinault River the Q2 value for 1978–2013 was 808cms, whereas the Q2 value for the entire Quinault River record, dating back to 1909, is substantially lower at 595cms – 26% increase (East et al. 2017).

A legacy of anthropogenic physical alterations including logjam and tree removal and anthropogenic riverbank disturbances decades ago have likely contributing to an unnaturally dynamic main-stem Quinault above the lake (Bountry et al., 2005; Herrera Environmental Consultants, 2005). Median reach-averaged width of the Quinault over the the same photographic record (299 m) was ~30% greater than that of the Queets (227 m) and 50% greater than on the Hoh (200 m) (East et al. 2017). Wood loadings, relative to the Hoh River and Queets, have also been found to be relatively lower in terms of overall cover (East et al. 2017).

As stated previously, glaciers in the Olympic Mountains have retreated rapidly over recent decades (East et al. 2017). Since 1980, ONP has lost 34% of its glacial ice area and 82 glaciers have disappeared entirely (Riedel et al., 2015), with the Quinault River already losing Anderson Glacier (Riedel et al. 2015). Future consideration, with respect to climatic change in the Queets, and other watersheds on the OP include fewer years with a large snowpack, more rainfall than snow, and short-term intense rainfall potentially resulting in more frequent winter flood activity (East et al. 2017). Models suggest that the Olympic Mountains are especially prone to increased flooding activity (Tohver et al., 2014, East et al. 2017), coupled with the additional hydrologic alterations anticipated from glacial retreat, particularly related to late-summer streamflow (Riedel et al., 2015, East et al. 2017). Summer low flows are predicted to decrease ~50% by 2040 for the majority of Upper Quinault River Basin, and around 26% for Lower Quinault (Wegner et al. 2010, USBOR 2014, USFS-OSC 2022). Large-scale flood events (i.e. greater than 25-year recurrence interval) are predicted to increase between 10% and 25% by 2040 (Wegner et al. 2010, USBOR 2014, USFS-OSC 2022).

The 5-year geometric observed escapement mean of winter steelhead in the Quinault River has changed as follows – 2,259 from 1998 to 2002, 2,716 from 2003 to 2007, 2,887 from 2008 to 2012, 2,625 from 2013 to 2017, and 2,186 from 2018 to 2022 (Table A1).

Hatchery supplementation and operations for the Quinault River has been ongoing since 1979 (Table A2). Between 1979 and 1989 the average winter steelhead smolt release was 397,372 (Table A2). Between 1990 and 1999 the average winter steelhead smolt release was 447,337, with a minimum of 290,865 and a maximum of 679,596 (Table A2). From 2000 to 2010 the average winter steelhead smolt release was 451,067, with a minimum of 234,006 and a maximum of 718,493 (Table A2). From 2011 to 2021 the average winter steelhead smolt release was 458,237, with a minimum of 395,612 and a maximum of 543,613 (Table A2).

Between 1985 and 1989 winter steelhead catch average was 4,045 with a minimum of 2,618 and a maximum of 5,892 (Table A3). Average winter steelhead catch in the 1990s was 2,855 with a minimum of 1,628 and a maximum of 4,560 (Table A3). Between 2000 and 2010 average winter steelhead catch was 2,472 with a minimum of 1,516 and a maximum of 4,177 (Table A3). From 2011 to 2022 average winter steelhead has been 1,796, with a minimum of 316 and a maximum of 4,025 (Table A33). Summer steelhead sport catch between 1962 and 1971 was 321 with a minimum of 197 and a maximum of 463 (Table A3). Summer steelhead sport catch during between 1972 and 2021 is less than 5 annually (Table A3).

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CRC spreadsheet Jim Scott May 2023 - Personal communication Jim Scott WDFW Subject: RE: extension for providing materials regarding OP steelhead petition Date: Friday, July 14, 2023 2:59 PM

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**Tables**

Table A1.– Average escapement (based on expanded redd counts). – Data and Analyses for OP Steelhead Oct 5, 2023

| Watershed | 1998 to 2002 | 2003 to 2007 | 2008 to 2012 | 2013 to 2017 | 2018 to 2022 |
|---|---|---|---|---|---|
| Salt Creek | 171 | | 84 | | 66 |
| East Twin Creek | 89 | | 35 | | 54 |
| West Twin Creek | 116 | | 42 | | 56 |
| Deep Creek | 162 | | | 83 | 99 |
| Pysht River | | 351 | 160 | | 237 |
| Clallam River | | 158 | 105 | | 146 |
| Hoko River | 698 | | 401 | | 438 |
| Quillayute-Bogachiel R. | 2957 | 1972 | 1710 | 1221 | 1166 |
| Calawah River | 4798 | 3122 | 2732 | 2526 | 2551 |
| Sol Duc River | 5696 | 3897 | 2980 | 2553 | 3483 |
| Dickey River | 699 | 344 | 384 | 268 | 423 |
| Hoh River | 3088 | 2254 | 2677 | 2314 | 2735 |
| Queets River | 4111 | 5634 | 4613 | 3583 | 2931 |
| Quinault River | 2259 | 2716 | 2887 | 2625 | 2186 |

Table A2 - Hatchery Supplementation.  Annual juvenile releases

- WDFW and Tribal SJD data June 2023 - NOAA_5_15_23 OP Steelhead and SJD Aggregate Harvest Data. Data sent by WDFW May/June of 2023. Data sent by WDFW May/June of 2023 Personal Communication  From: Harbison, Toby (DFW) Sent: Friday, June 16, 2023 4:13 PM To: Laura Koehn - NOAA Federal <laura.koehn@noaa.gov> for SJD data

| Watershed | Years | Run | Average | Maximum | Minimum |
|---|---|---|---|---|---|
| Salt Creek | 1962 to 1970 | Winter | 5818 | 10158 | 422 |
| Lyre River | 1960 to 1972 | Winter | 17849 | 35130 | 10071 |
| | 1981 to 2008 | | 26452 | 50000 | 5424 |
| | 1981 to 2008 | Summer | 10897 | 21422 | 5029 |
| Pysht River | 1957 to 1972 | Winter | 16069 | 20512 | 14220 |
| | 1979 to 2008 | | 12722 | 30000 | 9000 |
| Clallam River | 1981 to 1989 | Winter | 9254 | 18590 | 5068 |
| | 1990 to 1999 | | 6478 | 13820 | 4013 |
| | 2000 to 2008 | | 8884 | 14838 | 5000 |
| Hoko River | 1981 to 1989 | Winter | 16690 | 24700 | 10532 |
| | 1990 to 1999 | | 19473 | 23546 | 13971 |
| | 2000 to 2010 | | 24237 | 43571 | 9658 |
| | 2011 to 2020 | | 32639 | 68700 | 5480 |
| Sekiu River | 1988 to 1989 | Winter | 5047 | | |
| | 1990 to 1999 | | 8956 | 12129 | 4773 |
| | 2000 to 2010 | | 11,158 | 22912 | 5833 |
| | 2011 to 2020 | | 14936 | 21175 | 5580 |
| Quillayute-Bogachiel | 1980 to 1989 | Winter | 63742 | | |
| | 1990 to 1999 | | 109671 | 227322 | 9120 |
| | 2000 to 2010 | | 113464 | 295000 | 53000 |
| | 2011 to 2020 | | 111015 | 130419 | 45000 |
| Calawah River | 1980 to 1989 | Winter | 50213 | | |

| Watershed | Years | Run | Average | Maximum | Minimum |
|---|---|---|---|---|---|
| Calawah River (cont) | 1990 to 1999 | Winter | 89334 | 117998 | 10293 |
| | 2000 to 2010 | | 64500 | 109500 | 35000 |
| | 2011 to 2020 | | 53193 | 56357 | 42400 |
| Calawah River | 1980 to 1989 | Summer | 50213 | | |
| | 1990 to 1999 | | 89334 | 117998 | 10293 |
| | 2000 to 2010 | | 64500 | 109500 | 35000 |
| | 2011 to 2021 | | 53193 | 56357 | 42400 |
| Sol Duc River | 1980 to 1989 | Winter | 28253 | | |
| | 1990 to 1999 | | 21287 | 26507 | 14300 |
| | 2000 to 2010 | | 22046 | 37410 | 16000 |
| Sol Duc River | 1980 to 1989 | Summer | 24612 | | |
| | 1990 to 1999 | | 49194 | 100500 | 4929 |
| | 2000 to 2007 | | 64625 | 101000 | 31150 |
| Hoh River | 1980 to 1989 | Winter | 122072 | | |
| | 1990 to 1999 | | 95256 | 101881 | 92845 |
| | 2000 to 2010 | | 100256 | 161548 | 48625 |
| | 2011 to 2020 | | 69917 | 148765 | 14200 |
| Queets River | 1981 to 1989 | Winter | 133600 | | |
| | 1990 to 1999 | | 145527 | 202638 | 83483 |
| | 2000 to 2010 | | 162418 | 176580 | 149874 |
| | 2011 to 2021 | | 174743 | 204002 | 158204 |
| Quinault River | 1979 to 1989 | Winter | 397372 | | |
| | 1990 to 1999 | | 447337 | 679596 | 290865 |
| | 2000 to 2010 | | 451067 | 718493 | 234006 |
| | 2011 to 2021 | | 458237 | 543613 | 395612 |

Table A3 - Harvest –1976-2006 Abundance. Date oly_pen_esu.xlsx. Steelhead Historical Database WDFW Fish Mgmt HQ, 600 Capitol Way N, Olympia 98501, (360) 902-2820/2817 https://wdfw.wa.gov/sites/default/files/publications/00150/oly_pen_esu.pdf. CRC spreadsheet Jim Scott May 2023 - Personal communication Jim Scott WDFW Subject: RE: extension for providing materials regarding OP steelhead petition Date: Friday, July 14, 2023 2:59 PM. WDFW and Tribal SJD data June 2023 - NOAA_5_15_23 OP Steelhead and SJD Aggregate Harvest Data. Data sent by WDFW May/June of 2023. Data sent by WDFW May/June of 2023 Personal Communication From: Harbison, Toby (DFW) Sent: Friday, June 16, 2023 4:13 PM To: Laura Koehn - NOAA Federal <laura.koehn@noaa.gov> for SJD data

| Watershed | Years | Run | Average | Maximum | Minimum |
|---|---|---|---|---|---|
| Salt Creek | 1948 to 1961 | Winter | 279 | 748 | 0 |
| | 1962 to 1970 | | 291 | 697 | 75 |
| | 1971 to 1979 | | 146 | 748 | 6 |
| | 1980 to 1989 | | 44 | 134 | 11 |
| | 1990 to 1999 | | 16 | 39 | - |
| | 2000 to ? | | 8 | 16 | - |
| Lyre River | 1949 to 1959 | Winter | 205 | 347 | 20 |
| | 1960 to 1969 | | 1046 | 1526 | 312 |
| | 1970 to 1979 | | 1207 | 1744 | 560 |
| | 1970 to 1979 | Summer | 27 | 56 | 3 |
| | 1980 to 1989 | Winter | 1206 | | |
| | 1980 to 1989 | Summer | 112 | | |
| | 1990 to 1999 | Winter | 619 | 361 | 87 |
| | 2000 to 2010 | | 386 | 1037 | 144 |
| | 2000 to 2010 | Summer | 69 | 164 | 6 |
| | | | | | |

| Watershed | Years | Run | Average | Maximum | Minimum |
|---|---|---|---|---|---|
| Lyre River | 2010 to 2020 | Winter | 65 | 214 | 0 |
| East Twin | 1950 to 1959 | | 99 | | |
| | 1960 to 1969 | | 62 | | |
| | 1970 to 1979 | | 25 | 73 | 0 |
| | 1980 to 1989 | | 30 | 78 | 10 |
| | 1990 to 1999 | | 15 | | |
| | 2000 to2002 | | 3 | | |
| West Twin | 1950 to 1959 | | 58 | | |
| | 1960 to 1969 | | 50 | 125 | 2 |
| | 1970 to 1979 | | 31 | 100 | 0 |
| | 1980 to 1989 | | 20 | | |
| | 1990 to 1999 | | 10 | 23 | 0 |
| | 2000 to 2008 | | 3 | | |
| Deep Creek | 1950 to 1959 | | 103 | | |
| | 1960 to 1969 | | 132 | 125 | 2 |
| | 1970 to 1979 | | 51 | 91 | 0 |
| | 1980 to 1989 | | 62 | | |
| | 1990 to 1999 | | 14 | 37 | 0 |
| | 2000 to ? | | 4 | | |
| Pysht River | 1948 to 1956 | | 350 | 639 | 43 |
| | | | | | |

| Watershed | Years | Run | Average | Maximum | Minimum |
| --- | --- | --- | --- | --- | --- |
| | 1960 to 1969 | | 713 | 995 | 307 |
| | 1970 to 1979 | | 411 | 1057 | 130 |
| | 1980 to 1989 | | 390 | 645 | 216 |
| | 1990 to 1999 | | 243 | | |
| | 2000 to 2020 | | 48 | | |
| Clallam River | 1960 to 1969 | Winter | 264 | 337 | 20 |
| | 1970 to 1979 | | 164 | 393 | 63 |
| | 1980 to 1989 | | 158 | 382 | 58 |
| | 1990 to 1999 | | 61 | | |
| | 2000 to ? | | 24 | | |
| Hoko River | 1960 to 1969 | | 417 | 658 | 319 |
| | 1970 to 1979 | | 203 | 534 | 66 |
| | 1980 to 1989 | | 572 | 996 | 356 |
| | 1990 to 1999 | | 421 | | |
| | 2000 to ? | | 311 | | |
| Sekiu River | 1960 to 1969 | | 78 | 141 | 49 |
| | 1970 to 1979 | | 43 | 116 | 16 |
| | 1980 to 1989 | | 110 | 157 | 78 |
| | 1990 to 1999 | | 52 | | |
| | 2000 to 2010 | | 98 | | |
| | | | | | |

| Watershed | Years | Run | Average | Maximum | Minimum |
|---|---|---|---|---|---|
| Sekiu River | 2011 to 2020 | Winter | 343 | 864 | 47 |
| Quillayute | 1980 to 1989 | | 3395 | 7561 | 2013 |
| | 1990 to 1999 | | 5178 | 7100 | 1947 |
| | 2000 to 2010 | | 3831 | 7226 | 2053 |
| | 2011 to 2022 | | 2033 | 3719 | 803 |
| | 1980 to 1989 | Summer | 979 | 2226 | 51 |
| | 1990 to 1999 | | 388 | 642 | 205 |
| | 2000 to 2003 | | 756 | 1041 | 357 |
| Hoh River | 1980 to 1989 | | 2009 | 2800 | 1607 |
| | 1990 to 1999 | | 1544 | 2454 | |
| | 2000 to 2010 | | 1374 | 2584 | 437 |
| | 2011 to 2022 | | 765 | 1358 | 343 |
| | 1990 to 1999 | Summer | 37 | 104 | 3 |
| Queets River | 1980 to 1989 | Winter | 4535 | 6291 | 3279 |
| | 1990 to 1999 | | 2938 | 4977 | 1473 |
| | 2000 to 2010 | | 1535 | 2270 | 575 |
| | 2011 to 2022 | | 1607 | 3155 | 275 |
| | 1970 to 1979 | Summer | 180 | 345 | 53 |
| | 1980 to 1989 | | 190 | 310 | 78 |
| | 1990 to 1999 | | 32 | 69 | 6 |

| Watershed | Years | Run | Average | Maximum | Minimum |
| --- | --- | --- | --- | --- | --- |
| Queets River | 2000 to 2010 | Summer | 40 | 84 | 8 |
| Queets River | 2011 to 2022 | Winter | 30 | 72 | 12 |
| Quinault River | 1985 to 1989 | | 4045 | 5892 | 2618 |
| | 1990 to 1999 | | 2855 | 4560 | 1628 |
| | 2000 to 2010 | | 2472 | 4177 | 1516 |
| | 2011 to 2022 | | 1796 | 4025 | 316 |

# Appendix B - Review of ESA Listing Factor Threats for the Olympic Peninsula Steelhead Distinct Population Segment: Focus from 1996 to Today

## Analysis of ESA Section 4(a)(1) 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; 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 taking into account efforts to protect such species.

NMFS has reviewed the impacts of various factors contributing to the decline of Pacific salmon and *O. mykiss* in previous listing determinations (e.g., 63 FR 11482, March 9, 1998; 69 FR 33102 June 14, 2004) and supporting documentation (e.g. National Marine Fisheries Service 1996a; National Marine Fisheries Service 1998). These Federal Register notices and technical reports concluded that all of the factors identified in section 4(a)(1) of the ESA had played a role in the decline of West Coast salmonid stocks. Similarly, U.S. Fish and Wildlife Service found that most section 4(a)(1) factors threaten Bull Trout in the Olympic Peninsula and Puget Sound (not disease and predation, though it occurs), but mainly habitat destruction and modification and non-native fish introduction (Factor 5, "Other") (see 64 FR 58910; November 1, 1999). More recently, many ESA 5-year status reviews have summarized new and existing information on these threats as part of the reviews, including for a salmonid population that overlaps geographically with OP steelhead (Lake Ozette sockeye, National Marine Fisheries Service (2022a)) and or other ESA documents for other listed steelhead (e.g. critical habitat for Puget Sound steelhead National Marine Fisheries Service (2015)). This review relies heavily on recent 5-year status reviews, as well as on other NMFS assessments for listed salmonids (such as (Ford 2022)), reports from Washington Department of Fish and Wildlife, reports from Northwest Indian Fisheries Commission and Northwest Treaty Tribes, conversations with managers of OP steelhead, and peer-reviewed literature.

## Summary - current status since last review

NMFS last reviewed the status and risk of OP steelhead in the 1996 report, Busby et al. (1996). At that time, the SRT concluded that the "Olympic Peninsula steelhead ESU [DPS] is neither presently in danger of extinction nor likely to become endangered in the foreseeable future." Despite this conclusion, the SRT had several concerns about the overall health of this DPS and the status of certain stocks within it related to downward trends in abundance, uncertainty around abundance (especially for summer-run steelhead), and potential impacts of hatchery production and introgression given the use of few parent stocks (see Previous assessments section of the status review).

Since that time, actions have been taken to address certain threats. For instance, habitat restoration projects have occurred including the replacement of many culvert barriers (see Northwest Indian Fisheries Commission (2020), and Coast Salmon Partnership 2022 Annual Report[34]) in recent years and installation of large wood jams in selected rivers. Additionally, habitat connectivity continues to be maintained in the major river systems largely due to the absence of major blockages. More stringent State and Federal sport fishing regulations have gone into place including catch-and-release restrictions for recreational fishing and area and gear restrictions for natural-origin summer and winter steelhead. Also, more regulatory mechanisms have been established that protect salmonid habitat broadly, including Habitat Conservation Plans that address timber harvest (others include: Northwest Forest Plan and associated Aquatic Conservation Strategy and Land and Resource Management Plan for the Olympic National Forest, Washington Streamflow Restoration law and Fish Passage Barrier Removal Board, 2008 Statewide Steelhead Management Plan, Anadromous Salmon and Steelhead Hatchery Policy C-3624; see Listing Factor D below). Hatchery practices have been modified to reduce off-station releases, in order to increase the proportion of fish returning to the hatchery rack and decrease the number of hatchery-origin fish straying and spawning naturally[35].

Other threats continue to be an issue for this population. Legacy impacts of stream habitat modification have likely continued to impact this population since 1996 and continue now. Although efforts are underway to address habitat issues, it may take decades or even centuries for larger rivers to recover (Martens et al. 2019; Stout et al. 2018) especially related to woody debris (which may be most beneficial to steelhead, see Jorgensen et al. (2021)). Moreover, climate change will exacerbate conditions into the future (Wade et al. 2013). Climate change is currently impacting this DPS and will continue to negatively affect both the freshwater and marine habitat. In the foreseeable future, projected and modeled climate impacts that may affect steelhead include: prolonged summer low-flows, increased frequency and magnitude of peaks flows, elevated water temperatures, continued loss of glaciers (Wenger et al. (2010); Wade et al. (2013); and see below in Listing Factor E). Also, from a life history diversity perspective, kelt survival has continued to decline in the four major coastal rivers, possibly related to warmer sea surface temperature, pink salmon impacts, and Pacific Decadal Oscillations (but there is uncertainty about other potential contributing factors including predation).

Furthermore, though harvest management plans and hatchery operations have been modified, as described above, they continue to impact steelhead populations within the DPS. Prior to 2021, Olympic Peninsula steelhead populations experienced relatively high commercial and recreational fishing pressure (when compared to other DPSs) even while population run sizes declined. There are documented legacy and current impacts associated with harvest. Harvest four major OP rivers, which make up the majority of OP steelhead abundance was the highest in the state, 13.26%-59.19% depending on year and river between 2014-2020. Though catch and release regulations for natural-origin steelhead went into place in 2016, there is still has an assumed 10% mortality for released natural-origin steelhead, and some fish may be handled more than once. In the last 2 years (2021, 2022), harvest in the major four OP steelhead basins

[34] https://coastsalmonpartnership.egnyte.com/dl/VbBakQwmdS
[35] For example, winter steelhead smolt release into Pysht was eliminated in 2009; Goodman Creek, Clallam River, and Lyre river in 2009, and in Sol Duc, summer smolt releases were terminated in 2011 and winter in 2013.

has declined to ~9-15%, depending on basin, but it is unclear if, and how long, these harvest reductions will continue. In those recent years, even with reduced hargest, escapement goals have not been met in some basins. At the same time, the proportion of harvest that is natural-origin has increased so it is likely that proportionally more natural-origin steelhead are being caught in fisheries that target hatchery-origin steelhead (see section above - *SRT assessment of winter-run run timing changes*). There is also evidence of compressed run timing with harvest disproportionately catching early winter natural-origin winter-run steelhead. Certain hatcheries have for decades continued to release large numbers of out-of-DPS stock smolts (in the hundreds of thousands), and returning hatchery-origin adults overlap to some degree with natural-origin adults. Although some hatchery practices have improved, the naturally-spawning population likely retain genetic legacy impacts of past hatchery practices. Finally, though there have been some positive management changes, there continues to be challenges associated with fisheries and hatchery management. Data limitations continue for assessing the current status and risk of summer-run OP steelhead, an issue identified in the 1996 review and more recently by Harbison et al. (2022). There continue to be undefined escapement goals for some rivers, differing escapement goals between co-managers for others, and uncertainty if the escapement goals can maintain or restore runs. Where escapement goals have been established, there is a need to validate the biological relationships used to developed the goals some 40 years ago. Certain hatchery fish are not marked in some major rivers on the coast. Many threats to Olympic Peninsula steelhead identified by Busby et al (1996) continue today, although some efforts have been made to diminish their effects. However, new threats, such as climate change are beginning to affect steelhead populations in the Olympic Peninsula DPS, and will likely increase in intensity in the future.

# Listing Factor A: The Present or Threatened Destruction, Modification, or Curtailment of Steelhead Habitat or Range

Current habitat conditions within the OP DPS are summarized in multiple documents and reports including the State of Our Watersheds reports from Northwest Indian Fisheries Commission (2020) and reports on specific Water Resource Inventory Areas (WRIAs) (Washington Department of Ecology [WDOE] broke up the state of Washington into 62 WRIAs to delineate major watersheds within Washington and manage activities, where WRIAs 19-21 overlap with OP steelhead DPS range)[36]; see Smith (2000); Smith and Caldwell (2001). We also summarized watershed status within appendix A for the OP steelhead status review (information on specific rivers and watersheds). Here we summarize habitat modifications that have occurred and likely continue to have legacy impacts on OP steelhead, but also touch on restoration efforts that are ongoing to address past destruction and modification. For more general discussion of habitat needs of steelhead and other salmonids, see the following documents: Hicks et al. (1991); National Marine Fisheries Service (1996a); National Marine Fisheries Service (2015).

[36] See also this website for current water quality information from WDOE https://apps.ecology.wa.gov/ApprovedWQA/ApprovedPages/ApprovedSearch.aspx

## Habitat Background

The overall health and likelihood of persistence of salmon and steelhead populations are affected by the abundance, productivity, connectivity/spatial structure, and diversity of the component populations (McElhany et al. 2000). With respect to the habitat requirements for a healthy salmonid DPS, a DPS composed of many diverse populations distributed across a variety of well-connected habitats can better respond to environmental perturbations including catastrophic events (Anderson et al. 2014; Brennan et al. 2019; Greene et al. 2010; Schindler, Armstrong and Reed 2015; Schindler et al. 2010). Additionally, well-connected habitats of different types are essential to the persistence of diverse, locally adapted salmonid populations capable of exploiting a wide array of environments, as well as capable of responding to and surviving both short- and long-term environmental change (e.g., (Groot and Margolis 1991; Wood 1995)). Differences in local flow regime, temperature regime, geological, and ecoregion characteristics correlate strongly with DPS population structure (Beechie et al. 2006; Ruckelshaus et al. 2006).

For winter-run versus summer-run steelhead, while there is some temporal overlap in spawn timing between these forms, in basins where both winter- and summer-run steelhead are present, summer-run steelhead typically spawn farther upstream, often above a partially impassable barrier (Myers et al. 2015). In many cases, it appears that the summer migration timing evolved to access areas above falls or cascades that present velocity barriers to migration during high winter flow months, but are passable during low summer flows (Myers et al. 2015; Narum et al. 2008– genetic indication of separation of habitat by anadromous vs. resident; Withler 1966). Most Olympic Peninsula rivers lack major in-stream velocity barriers, cascades, or falls , and it is unclear what mechanism has provided for the expression of the summer-run life history.

Within the Olympic Peninsula steelhead DPS, major river basins that drain into the Pacific Ocean originate within the Olympic National Park (ONP) where habitat is protected from most detrimental land-use practices such as logging (Figure 50). The lower portions of these watersheds extend outside of the park into Federal, State, and Private forest lands and other developed lands and are subjected to different levels of logging and other land-use practices. Table 25 presents the percent of each population's habitat that falls within the ONP as well as the number of years since last disturbance and overall percent forest cover (data from: Jin et al. (2023)). However, not all stream/river reaches are accessible to steelhead (see Table X below for percent of steelhead habitat used within the ONP). We note that even if steelhead can not utilize portions of a watershed within the ONP, protecting the integrity of the headwater areas provides benefits to the entire system (Table B2).

While populations were generally well described by WDFW, we generated our own polygonal geospatial layer representing both winter and summer groups[37]. This was done by dissolving

[37] For all mapping presented: any summary calculations for the Olympic Peninsula Steelhead DPS climate, hydrological, and landscape data were performed utilizing ESRI's geospatial software. We used the National Hydrography Dataset (NHDPlus) catchments (Moore and Dewald 2016) as the summary unit for all values, including raster products derived from the National Landcover Database (NLCD) (Jin et al. 2023; Yang et al. 2018), and streamflow (Wenger et al. 2010) and temperature (Isaak et al. 2016) metrics provided by the Rocky

catchment features by WDFW population name and run-type. The resulting shapefiles were used for both summarizing environmental data and generating figure maps.. In addition, catchments were attributed with current steelhead use as defined by the Statewide Washington Integrated Fish Distribution (SWIFD) (SWIFD GIS Data 2014) and Streamnet (StreamNet GIS Data 2019). We employed current use for any life cycle stage as a screen for certain summaries within populations in order to more accurately account for steelhead distribution within sub-basins.

We also incorporated land manager status as a supplementary unit of analysis for various outputs. Values showing private or public ownership and manager type by agency name (local, state, tribal or federal) were obtained from the USGS's PAD database (U.S. Geological Survey (USGS) Gap Analysis Project (GAP) 2022). We then attributed catchments to land management type using geoprocessing tools. All catchment values were included in a single spreadsheet from which individual attributes were summarized by population name and/or land manager, with additional assessments done for steelhead life history use type.

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Mountain Research Station (for current, 2040, and 2080 climate predictions for listing factor E). Summaries were developed for individual catchments at the reach level and steelhead population sub-basins.

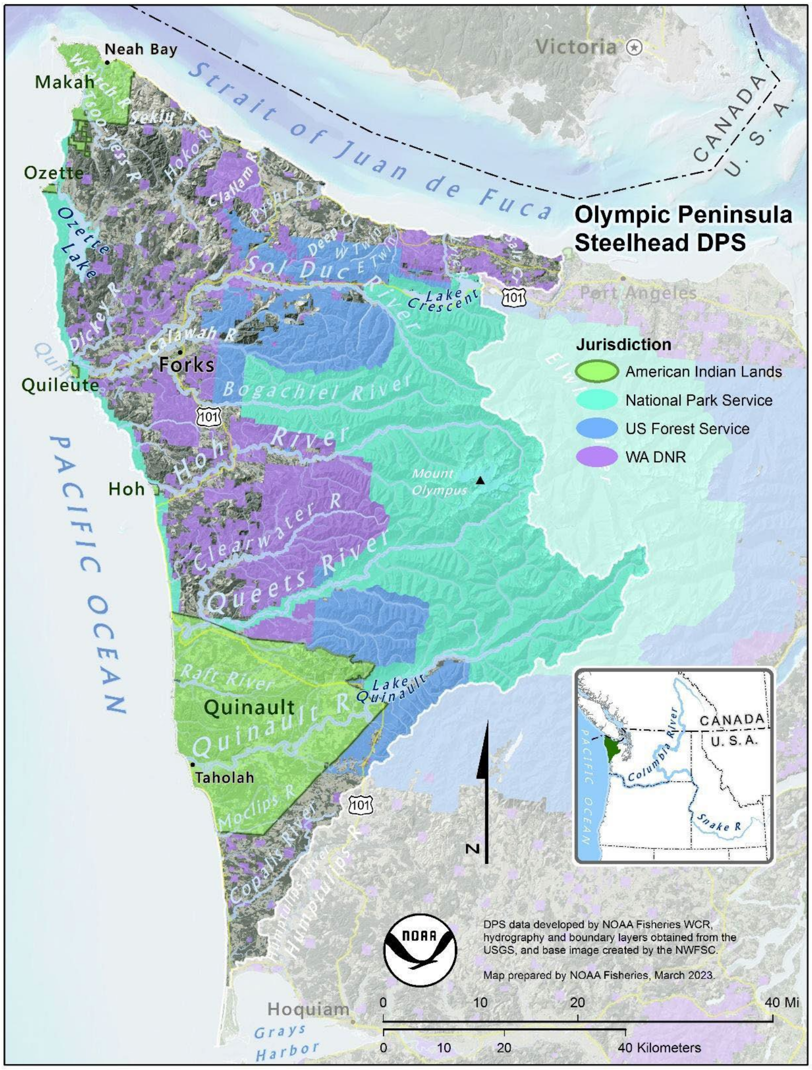


Figure 50. Map of OP steelhead freshwater habitat by jurisdiction

Table 26. Olympic Peninsula steelhead DPS population/run timing and habitat for each in terms of percent that is forest covered, average years since last disturbed based on recent forest disturbance data between 1986-2019 (see Jin et al. 2023) and percent that is within the boundaries of Olympic National Park (ONP). Average across the whole DPS for each habitat metrics is also reported.

| Population | Run | % Forest cover | Average Years Since Disturbed | % in ONP |
|---|---|---|---|---|
| Salt Creek-Independents | winter | 81% | 34.7 | 0 |
| Lyre | winter | 74% | 35.0 | 0 |
| Pysht-Independents (including the Twins) | winter | 93% | 32.6 | 0 |
| Clallam | winter | 87% | 29.5 | 0 |
| Hoko | winter | 96% | 25.8 | 0 |
| Sekiu | winter | 94% | 23.0 | 0 |
| Sail | winter | 98% | 25.5 | 0 |
| Tsoo-Yess-Waatch | winter | 85% | 28.0 | 0 |
| Ozette | winter | 77% | 27.8 | 12% |
| Quillayute-Bogachiel | winter | 87% | 35.6 | 37% |
| Dickey | winter | 86% | 25.4 | 0.1% |
| Sol Duc | winter | 94% | 35.0 | 17% |
| Calawah | winter | 95% | 33.0 | 16% |
| Hoh | winter | 79% | 36.7 | 52% |
| Goodman Creek | winter | 91% | 30.2 | 11% |
| Mosquito Creek | winter | 95% | 26.2 | 18% |
| Kalaloch Creek | winter | 92% | 32.7 | 28% |
| Queets | winter | 79% | 36.9 | 63% |
| Clearwater | winter | 93% | 32.7 | 0.1% |
| Raft | winter | 88% | 30.7 | 0 |
| Lower Quinault | winter | 81% | 31.6 | 0.5% |
| Upper Quinault | winter | 71% | 39.3 | 65% |
| Moclips | winter | 83% | 34.7 | 0 |

| Population | Run | % Forest cover | Average Years Since Disturbed | % in ONP |
|---|---|---|---|---|
| Copalis | winter | 62% | 28.3 | 0 |
| Quillayute-Bogachiel | summer | 82% | 36.3 | 52% |
| Sol Duc | summer | 94% | 36.1 | 25% |
| Calawah | summer | 95% | 32.4 | 14% |
| Hoh | summer | 76% | 37.3 | 61% |
| Queets | summer | 74% | 37.0 | 77% |
| Clearwater | summer | 90% | 32.4 | 0.1% |
| Quinault | summer | 63% | 36.4 | 38% |
| Average | | 85% | 32.2 | 19% |

Table 27. The percentage of steelhead habitat utilized within the Olympic National Park (ONP) for various rivers and creeks or basins (for example, "Hoh river" contains subbasins) in coastal Washington that drain directly into saltwater, or in the case of Quillayute – the rivers that comprise the Quillayute system that had more than 0% in the ONP. Any basins/rivers not listed have 0% of steelhead habitat used in the park.[38]

| Basin | Total Length of Steelhead Use (m) | Within Olympic National Park (m) | % Within | Outside Olympic National Park (m) | % Outside |
|---|---|---|---|---|---|
| Cedar Creek | 17,103 | 2,833 | 17% | 14,270 | 83% |
| Goodman Creek | 44,652 | 5,443 | 12% | 39,209 | 88% |
| Kalaloch Creek | 11,076 | 1,136 | 10% | 9,940 | 90% |
| Ozette | 149,053 | 14,113 | 9% | 134,940 | 91% |
| Mosquito Creek | 20,269 | 1,710 | 8% | 18,558 | 92% |
| Upper Quinault | 183,483 | 119,663 | 65% | 63,821 | 35% |
| Queets | 220,090 | 90,816 | 41% | 129,274 | 59% |
| Hoh | 276,356 | 103,266 | 37% | 173,090 | 63% |
| Quillayute: | | | | | |
| Bogachiel | 188,336 | 56,716 | 30% | 131,620 | 70% |
| Calawah | 139,831 | 24,264 | 17% | 115,567 | 83% |
| Sol Duc | 256,847 | 44,347 | 17% | 212,500 | 83% |

[38] We attributed the NHD catchments (Hill et al. 2016) with our proto populations (usually inheriting the largest river name) and steelhead distribution (WDFW 2022) by run and use type. These spatial features were then intersected with the land manager polygons from the PAD (USGS 2024) database. From these values we then summarized stream length by steelhead use and population name to determine the quantity and percent of occupied habitat within the Olympic National Park.

## Hydropower development

There are no major dams or hydropower development in watersheds within the range of the OP steelhead DPS. The WDFW review of Washington steelhead stated that for the Olympic Peninsula population, that there's been no habitat loss due to large dams or barriers (Cram et al. 2018). But see section on Land-use Practices for discussion of smaller barriers.

## Land-use practices - Logging

Numerous studies have been conducted regarding the impacts of land use activities on salmonid habitat in the states of Washington, Oregon, Idaho, and California. Land use activities associated with logging, road construction, urban development, mining, agriculture, and recreation have significantly altered fish habitat quantity and quality. Associated impacts of these activities include the following: alteration of streambank and channel morphology; alteration of ambient stream water temperatures; degradation of water quality; elimination of spawning and rearing habitat, fragmentation of available habitats; elimination of downstream recruitment of spawning gravels and large woody debris; removal of riparian vegetation resulting in increased stream bank erosion and higher water temperatures; and degradation of water quality (see references in Anderson 1993; Botkin et al. 1995; Bottom, Howell and Rogers 1985; Brown and Moyle 1991; Bryant 1994; California Advisory Committee on Salmon and Steelhead Trout 1988; California Department of Fish and Game 1965; California Department of Fish and Game 1991; California Department of Fish and Game 1994; California State Lands Commission 1993; Hicks et al. 1991including: ; McEwan and Jackson 1996; National Marine Fisheries Service 1996a; Nehlsen, Williams and Lichatowich 1991; Titus, Erman and Snider 2003). The loss of channel complexity, pool habitat, suitable gravel substrate, and large woody debris, and other development activities have caused increased fine sediment input into spawning and rearing areas (cited in NMFS 1996a: Bottom, Howell and Rogers 1985; Forest Ecosystem Management Assessment Team 1993; Higgins, Dobush and Fuller 1992; U.S. Forest Service and U.S. Bureau of Land Management 1994). Creation of splash dams and lumber transport via rivers associated with previous logging practices led to scouring of spawning gravel, clearing/loss of woody debris, degradation of stream beds and floodplain disconnection (summarized in Coast Salmon Partnership 2022 Annual Report[39]). Splash dam structures are mainly gone but their impacts remain. Due to anthropogenic activities such as timber harvest, Bisson et al. (1997) estimated that there was a 2 to 10 times increase in the frequency of major floods, that both debris flows and dam-break floods were 5 to 10 times more frequent, and also that slumps and earth flows were 2 to 10 times more frequent, compared to natural, background conditions.

Both logging and agriculture activities result in many similar impacts on salmonid habitat. Major impacts common to both activities include loss of large woody debris, sedimentation, loss of riparian (streamside) vegetation, increased water temperatures, and loss of habitat complexity, all of which affect water quality and the biotic communities. Nutrient loading impacts to stream productivity can be caused by mining, livestock, or forest management. Recent work by Naman

[39] https://coastsalmonpartnership.egnyte.com/dl/VbBakQwmdS

et al. (2024) showed that across the range of Pacific salmonids, forestry activities led consistently to impacts to stream flow and stream temperature across the range but the magnitude of these impacts varied.

The vast majority of land-use practices in the range of OP steelhead that have been/are detrimental to OP steelhead habitat relate to logging and forestry practices, and only to a limited extent agriculture use, so we focus our discussion mainly to logging practices. The majority of land on the Strait of Juan de Fuca within river basins in the OP steelhead range is timberland (Table 27). For the Salt Creek, state and private forestlands are mostly located in the headwaters (~56%), while agricultural and rural residential lands (42%) are strongly clustered in low gradient stream channel areas in the middle and lower watershed (McHenry, McCoy and Haggerty 2004; North Olympic Peninsula Lead Entity for Salmon (NOPLE) 2015). The Lyre River watershed includes the Olympic National Park (~66%), as well as commercial timberlands (31%), and low-density rural residential (~3%) (McHenry, Lichatowich and Kowalski-Hagaman 1996; NOPLE 2015). For East Twin the majority is forest lands - Washington state Department of Natural Resources lands (WA DNR) and United States Forest Service lands (USFS) comprise over 90% of the ownership (NOPLE 2015). Similarly, for West Twin, Deep Creek, and Pysht the majority of the land is for forestry with the majority of the forestlands managed by USFS or WA DNR (~61% for West Twin, ~50% for Deep Creek, and 75% for Pysht) followed by 29%, ~43%, and ~24% owned as private timberlands for West Twin, Deep Creek, and Pysht respectively (NOPLE 2015).  Washington state timberlands and industrial forest timberlands make up over 95% of the land ownership in the Clallam River basin (Haggerty 2008). The vast majority of land in the Hoko River is commercial timberlands, however portions of the Lower Hoko River and Little Hoko have been converted to open areas or hardwood-dominated areas and purchased by Washington state parks (NOPLE 2015, personal communication with Mike McHenry, Lower Elwha Tribe, December 5, 2023). The Sekiu is also predominately privately-owned and state-owned timberlands, with a portion on the Makah Tribal Reservation (NOPLE 2015).

Table 28. Percentage of each landownership type for watershed area by subbasin. Modified from NOPLE 2015. For acronyms: WDNR = Washington State Department of Natural Resources, ONP = Olympic National Park, USFS = United States Forest Service, and Ease./ROW = easements/right of ways.

| Sub-basin | Private | WDNR | ONP | USFS | Reservation | County | Other state land | Other fed land | Ease./ ROW | Other |
|---|---|---|---|---|---|---|---|---|---|---|
| Salt | 50.2% | 44.3% | 0 | 0 | 0 | 1.1% | 0 | 3.1% | 1.34% | 0 |
| Lyre | 10.4% | 17.5% | 65.5% | 5.7% | 0 | 0 | 0 | 0.6% | 0.3% | 0 |
| East Twin | 6.8% | 46.1% | 0.01% | 46.2% | 0 | 0.1% | 0 | 0.5% | 0.3% | 0 |
| West Twin | 29.0% | 9.9% | 0 | 60.9% | 0 | 0 | 0.01% | 0 | 0.2% | 0 |
| Deep | 43.2% | 4.9% | 0 | 50.4% | 0 | 0.6% | 0 | 0.8% | 0.05% | 0 |
| Pysht | 76.7% | 5.9% | 0 | 16.6% | 0 | 0.03% | 0.2% | 0 | 0.5% | 0 |
| Clallam | 49.6% | 47.6% | 0 | 0.1% | 0 | 0.1% | 2.1% | 0.02% | 0.6% | 0.01% |
| Hoko | 72.5% | 24.6% | 0 | 0.9% | 0 | 0.2% | 1.7% | 0 | 0.1% | 0.02% |
| Sekiu | 75.7% | 17.3% | 0 | 0 | 7.1% | 0 | 0.01% | 0 | 0.01% | 0 |
| WSI | 57.1% | 57.1% | 0 | 0 | 16.8% | 0.6% | 0.4% | 1.2% | 1.0% | 0.1% |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Total WRIA 19 | 51.4% | 22.3% | 11.6% | 9.1% | 3.9% | 0.3% | 0.6% | 0.5% | 0.4% | 0.02% |

For the four major rivers on the West Coast, other than land within the ONP, Olympic National Forest (ONF) or Tribal, the remaining land is state or private- owned timberlands. Land ownership varies as a function of the area below and above Lake Quinault. Below Lake Quinault ownership is predominantly the Quinault Tribal reservation (~80%), followed by Olympic National Forest (~14%), and private timberlands (~7%). Above Lake Quinault ownership is dominated by Federal lands (~95%), followed by Quinault Tribal reservation (~4.5%), and private lands (<0.5%). See Appendix A for further descriptions of each individual watershed/river.

In the OP, past timber harvest practices have resulted in a loss of stream buffers, the removal of instream wood, high-density road construction and frequent use, and harvesting large proportions of watersheds (Martens et al. 2019). These practices resulted in deleterious changes to sediment supply, wood supply, the amount and condition of streamflow, and stream channel morphology (Abbe and Montgomery 2003; Cederholm, Reid and Salo 1981; Logan, Kaler and Bigelow 1991; McHenry et al. 1998; Northwest Indian Fisheries Commission 2020). Forest harvest without stream buffers or minimal streamside buffers, coupled with the removal of instream wood results in stream channels widening due to accelerated erosion, the loss of current and future instream wood important to juvenile steelhead, and overall habitat simplification that results in more variable low and peak flow stream conditions due to the lack of attenuation from instream wood, streambank trees, or more stable streambanks (Abbe and Montgomery 2003; Cederholm, Reid and Salo 1981; Logan, Kaler and Bigelow 1991; McHenry et al. 1998; Northwest Indian Fisheries Commission 2020).

National Marine Fisheries Service (1996a) summarizes impacts of logging (and similar impacts of agriculture) on steelhead habitat by important habitat features - woody debris, sedimentation, riparian vegetation, and habitat complexity/connectivity. Here, we briefly summarize the discussion in that report.

**Riparian vegetation**

The loss of riparian vegetation can also negatively affect steelhead. Reduction in shade canopy from tree loss in the riparian zone can lead to increased water temperatures (see discussion in Hicks et al. (1991)). Riparian vegetation also protects stream banks from exacerbated erosion rates and provides depositional areas for gravel and finer materials, all which create and maintain salmon and steelhead habitat (Bottom, Howell and Rogers 1985; California Department of Fish and Game 1994; Forest Ecosystem Management Assessment Team 1993). The reduction in shade canopy due to logging stands adjacent to rivers has resulted in increased water temperatures, in some instances (Bisson et al. 1987; California Department of Fish and Game 1994; Forest Ecosystem Management Assessment Team 1993), and can increase temperatures by 11.7 to 18°F (7.8 to 11.3 °C) (Reynolds et al. 1993). Riparian vegetation provides important substrates for aquatic invertebrates, cover for predator avoidance, and resting habitat for many fish species. Dead organic matter from the riparian vegetation is an important source of nutrients and contributes to the detrital food web (Bisson and Bilby 1998). Removal of riparian vegetation can change autotrophic production, emergence time of fry, growth rate and age at smolting, survival of juveniles, and increased susceptibility to disease (Hicks et al. 1991). Removal may also result in more needles, bark, and branches in the stream in the short-term, increasing dissolved oxygen demand, increasing organic matter, but also increasing in-river cover - these changes may reduce spawning success but also create short-term increases in food production and juvenile survival (Hicks et al. 1991). Any activities that result in direct riparian or streambank modification or streamflow modification that alters riparian composition can contribute to vegetation loss (Reynolds et al. 1993).

**Woody debris**

Downed trees are important to the functionality of streams and estuaries (Naiman et al. 1992; Sedell and Luchessa 1982; Sedell and Maser 1994; Swanson, Lienkaemper and Sedell 1976) and large woody debris impacts cover, storage of gravel, channel morphology/hydraulic complexity, geometry, pattern, and position, as well as pool formation (Bisson et al. 1987; Hicks et al. 1991). Downstream transport rates of sediment and organic matter are controlled in part by storage of this material behind large wood (Beschta 1979). Woody debris is important to salmonid habitat because it impacts formation of habitat units, provides shelter (cover and complexity) and protection from peak flows, and acts as substrate (Bisson et al. 1987; Hicks et al. 1991; Sedell et al. 1982; Swanson, Lienkaemper and Sedell 1976). Loss of woody debris may also reduce carrying capacity of habitat, increase predation vulnerability for salmonids, lower winter survival, reduce food production, and may result in lower species diversity (Hicks et al. 1991). 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). Logging practices before the 1970s led to clogged waterways due too much woody debris that blocked fish migration. Afterwards, actions to remove woody debris led to excessive removal and resulted in loss of salmonid habitat (Botkin et al. 1995; Bottom, Howell and Rogers 1985; California Department of Fish and Game 1994) that could be expected to persist for 50-100 years. Furthermore, past logging has resulted in the elimination of large trees on streamside areas, so consequently there are very few significant trees available for recruitment into streams. Recent research has shown that there are temporal dynamics of wood and that the status is not necessarily static (see Gregory et al. 2024).

**Sediment effects**

In general, 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 useable habitat by scouring and filling of pools and riffles and changing bedload composition; reducing photosynthetic growth and primary production (and thus prey); and affecting intergravel permeability and dissolved oxygen levels (Hicks et al. 1991; Jensen et al. 2009; Koski and Walter 1978; Suttle et al. 2004). Most forest land-use practices accelerated rates of erosion and supply of both coarse and fine sediment, and road networks from logging are a major source of fine sediment (Forest Ecosystem Management Assessment Team 1993; Gibbons and Salo 1973). Accelerated rates and magnitudes of erosion can results in instream sediment levels being 2.5 times the magnitude of unlogged streams, thus reducing egg survival (Cederholm and Reid 1987; Cederholm and Salo 1979; McHenry et al. 1998; Tagart 1984).

Sediment effects on steelhead can be grouped into effects of suspended sediment (turbidity), fine sediment that settles into the bed, and coarse sediment.

Suspended sediment can have negative physical and biological impacts. Turbidity from continued sediment suspension can decrease photosynthesis of aquatic plants (through light scattering) and clog respiratory and feeding systems of animals (Bash, Berman and Bolton 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; Phillips et al. 1975).

Fine sediment that settles into the stream bed affect both survival of eggs in the gravel and production of benthic invertebrate prey (discussed in Hicks et al. (1991), including Cederholm, Reid and Salo (1981); Cordone and Kelley (1961); Lloyd (1987)). From a more recent study, egg-to-fry survival asymptotes at only 10% when fine sediment (<0.85 mm) is greater than 25% (Jensen et al. 2009). Survival of eyed eggs was >90% until fine percentages increased above 20-25% and then survival decreased (Jensen et al. 2009). Embedded sediment and particles deposited as bedload sediment and unstable spawning gravels may also negatively affect steelhead. Increased sedimentation of gravels and pools can also increase stream temperatures (Hagans, Weaver and Madej 1986).

Coarse sediment (generally small gravels and larger) can fill pools fill in 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. Such changes cause decreased stream stability and increased bank erosion, which exacerbates existing sedimentation problems. Stream widening and reduced depth can increase predation vulnerability for salmonids, and can increase carrying capacity for young fish (age 0) but reduce for age-1 and older fish (Hicks et al. 1991). This can lead to starvation, predation, or reproductive failure of the species. Erosion can also result in increased debris torrents which may decrease cover in some places but increase debris elsewhere; blocking migration and reducing survival or improving habitat where debris is increased (Hicks et al. 1991).

**Habitat complexity**
A diverse habitat mosaic is essential for healthy and sustainable salmon and steelhead populations (Brennan et al. 2019; Hilborn et al. 2003). In Pacific Northwest and California streams, habitat simplification is a common consequence of land use and has led to a decrease in the diversity of anadromous salmonid habitat, life histories, and overall species complexity (Bisson and Sedell 1984; Hicks 1990; Li et al. 1987; Munsch et al. 2022; Reeves, Everest and Sedell 1993). Habitat simplification may result from various land-use activities, including but not limited to timber harvest, grazing, urbanization (California State Lands Commission 1993; Forest Ecosystem Management Assessment Team 1993; Frissell 1992) and agriculture (Forest Ecosystem Management Assessment Team 1993). Timber harvest and range management activities can result in a decrease in the number and quality of pool habitats (Sullivan et al. 1987). Reduction of wood in stream channels, either from past or present activities, generally reduces pool quantity and quality (Wohl 2017), alters stream shading which can affect water temperature regimes and nutrient input (Bowler et al. 2012), and can eliminate critical stream habitat needed for both vertebrate and invertebrate populations (Richardson and Danehy 2007).

**Olympic Peninsula- Watershed Specific Legacy Impacts and Restoration**

Above we summarized the impacts of logging land-use practices and here we provide more details on specifics within the OP steelhead DPS range for specific rivers/watersheds, and we provide even more detail in Appendix A. In addition to the effects of logging, culverts have blocked access to various spawning grounds and rearing habitat and impacted downstream recruitment processes related to sediment and wood (Kemp 2015; Sullivan et al. 1987). Although efforts are underway to address these issues, it may take decades for habitat to recover (Martens et al. 2019) and climate change may exacerbate conditions (Wade et al. 2013). For example, Figure B51 shows currently where barriers exist on the OP due to anthropogenic influence but note below discussion of ongoing barrier removal in the section on *Recent Research on Restoration Potential*. Even with ~25 years of more protective timber harvest regulations related to riparian zones important salmonid habitat components such as instream wood and pools have not recovered through the natural recruitment of wood (Martens and Devine 2023). The estimated timeline for recovery of these remaining degradations could range between 100 and 225 years (Martens and Devine 2023; Stout et al. 2018).

WDFW concluded that the legacy impacts of historical land-use resulting in habitat degradation that continues to be a threat to naturally-produced steelhead, and identified practices including: past clear-cut logging, road building, bank protection mitigations that were poorly designed or unmitigated, as well as floodplain infrastructure impacts (Cram et al. 2018). However, as noted above, most of headwaters for the major river basins occupied by OP steelhead originate within the Olympic National Park (ONP) where these effects should be minimal. WDFW (Cram et al. 2018) identified particular impacts to the Clearwater River, which has headwaters outside of the ONP (unlike other OP steelhead rivers). Specifically, WDFW noted that this river tributary has been extensively logged and this has resulted in increased sediment inputs (from road building and use and tree harvest). Smith and Caldwell (2001) showed that logging in the Clearwater Basin has led to the loss of large woody debris recruitment (see Appendix A for watersheds on the Strait of Juan de Fuca). On the other hand, improvements have been made in the Hoh River basin, where recent land acquisitions (approximately 90 percent of the basin is now owned by state and Federal government or conservation organizations) and subsequent efforts to restore and protect habitat has led to various stages of regeneration across the Hoh River valley rainforest (Cram et al. 2018). Still, between 2016 and 2020 some 51.6 sq. km of timberland were harvested in the Hoh River basin (Northwest Indian Fisheries Commission 2020). According to Cram et al. (2018), “The effectiveness of currently implemented forest practices for minimizing impacts remains uncertain. For example, incorrectly applied or inadequately designed riparian management zones and incorrect stream typing classifications are known problems that impair habitat protection strategies (Hansen 2001). These practices result in loss of large woody material, fish passage impacts, altered hydrology, water quality impacts, mass wasting (landslides), and elevated stream temperatures (Naiman et al. 1998).”

Smith (2005) summarized habitat quality by multiple limiting factors for each WRIA in Washington. Here, we replicate the findings by habitat factor for WRIAs 19-21(Table B28) within the range of OP steelhead, where DG = data gap. Note that across Washington state, most ratings for limiting factors were DG (43%) or poor (38%) and only 20% of ratings were good or fair. See Smith 2005 for more information on each.

Table 29. Summarized habitat quality by multiple limiting factors for each WRIA within the range of OP steelhead (19-21), where DG = data gap. Findings replicated from Smith (2005).

| WRIA | Access (culvert, dams) | Side channel Flood-plain | Sediment quantity | Sediment quality | Road density | Bank/Stream-bed/ Channel stability | Instream large woody debris | Pool habitat | Riparian | Water temp. | Water dissolved O2 | Water other nutrient, toxins, pH | Hydro maturity high flows | Impervious surfaces | Low flows |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 19 | Fair-poor | Poor | Poor | Poor | Fair | Poor | Poor | DG | Poor | Poor-Good | DG | DG | Poor | DG | DG |
| 20 | DG | Poor | Poor | Poor | Fair | Poor | Poor | Good | Poor | Poor | DG | DG | Fair | DG | DG |
| 21 | DG | Poor | DG | DG | Good | DG | Fair | Fair | Fair | Poor | Good | DG | Good | DG | Good |

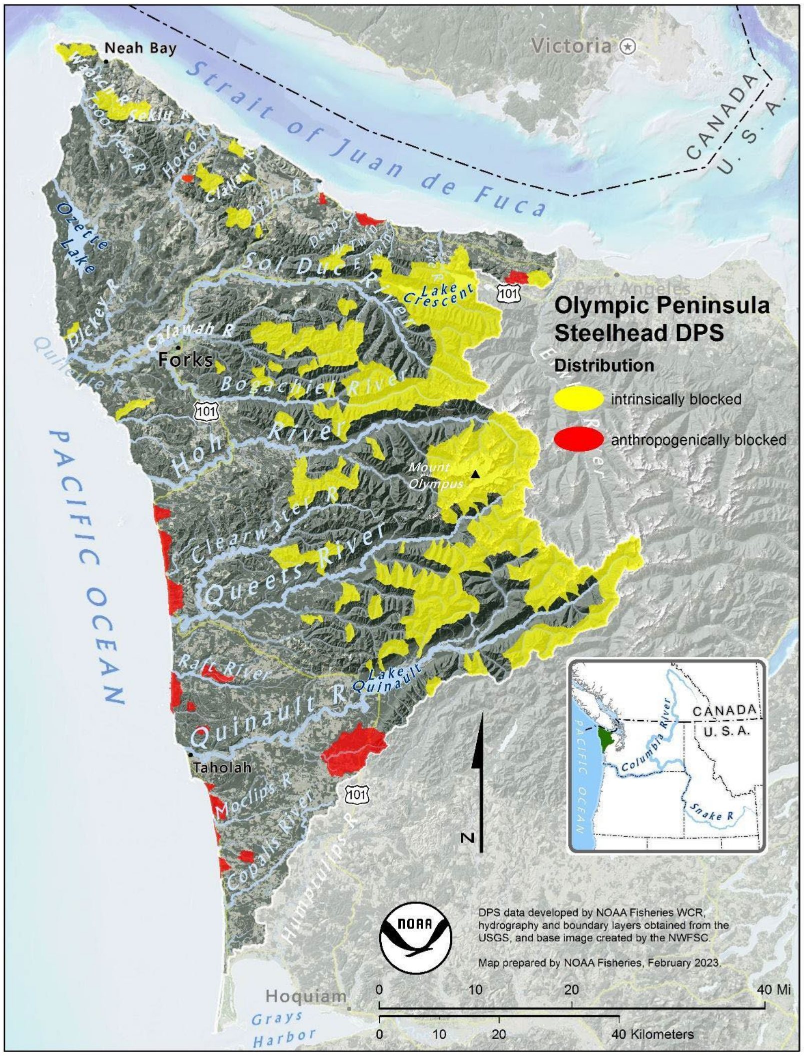


Figure 51. Natural and man-made barriers in the range of the OP steelhead DPS.

We summarized land-use practices, as well as some specific restoration work, by watershed and river (see Appendix A). Here we summarize major themes that were informed by that review about impacts of past and ongoing land-use practices in the OP. For streams within the Strait of Juan de Fuca watershed, the loss of wood due to systematic removal during the 1950s was widespread, occurring in the Lyre, Salt, East Twin, West Twin, Pysht, Clallam, Hoko, and Sekiu basins. Similarly, the loss of riparian recruitment potential due to previous timber harvest and road development was widespread, and not all streams have had or have ongoing restoration actions (wood treatments) (for example West Twin but see description of treatment in East Twin and Deep Creek). Wood treatment to help restore woody debris can also be impacted by natural disturbances such as flooding events. Relatedly, there has been stream channel incision due to the loss of obstructions like woody debris but also due to decreased floodplain activity. The frequency of landslides has also increased in the Strait watersheds specifically west of Lyre River in East Twin, West Twin, Pysht, Hoko, and Sekiu basins. As we cover extensively in listing Factor E related to climate change, increases in winter flow events, decreases in summer flows and increases in stream temperatures have already been observed. Finally, estuarine area has been reduced by almost 50% for the Pysht River due to land-use and estuarine mouth of the Clallam River has been blocked due to anthropogenic impacts from channel modifications, log rafting, milling, etc. Efforts have been made in Clallam River to allow for connection between the river and marine water and in the Pysht River there are plans to restore the estuarine habitat. Thus, for many basins draining to the Strait of Juan de Fuca the legacy of past land use practices continues to influence habitat stream and riparian habitat quality.

Along the West (Pacific Ocean) side of the Peninsula there have been similar impacts of previous land-use and logging. Historical land-use practices included forest harvest without stream buffers, the removal of instream wood, high-density road construction and frequent road use, and harvesting large proportions of watersheds (Martens et al. 2019). This past timber harvest has resulted in changes to sediment supply, wood supply, streamflow, stream temperature, and stream channel morphology. Timber harvest intensity does vary by river; for example, the Calawah River Basin had intensive logging and road building after a fire in 1951, while the Bogachiel River is partially within ONP boundary and has had less timber harvest and road building (Jaeger, Anderson and Dunn 2023). In general, the reduction in wood loadings and instream wood removal have led to the loss of pools, and decreases in stabilizing wood jams which led to the loss of channel complexity (particularly in the Queets) (Abbe and Montgomery 2003; Martens et al. 2019). Wood loadings continue to decrease and the density of large wood in the OP in forests managed by USFS has decreased by ~50% from 2002-2018 (Dunham et al. 2023). Historic logging in the Queets River basin, even though a large portion of the watershed is in ONP and has a protected floodplain corridor, was intensive and extensive (McHenry et al. 1998).

Road construction in the Queets during this time included techniques that are now known to be sub-standard and resulted in road failures, increased landslide rates (which were 168 times those of a natural reference area), reduced stream habitat conditions particularly in some of the tributaries such as the Clearwater River basin, and 2.5 times the instream sediment levels of unclogged OP streams resulting in reduced salmon egg survival and fry emergence from the density of roads (Cederholm and Reid 1987; Cederholm and Salo 1979; McHenry et al. 1998; Tagart 1984). Additionally, the loss of large trees along riparian zones have resulted in greater streambank erosion (Abbe and Montgomery 2003; Martens 2018). Changes to stream channel morphology have resulted from stream channel incision, stream channel widening, and increased bedload movement. Stream width reduction has occurred in the Calawah River basin since the 1990s, but not in the Bogachiel River (Jaeger, Anderson and Dunn 2023).

In the Hoh River, increases in sediment supply (from timber harvest and glacial retreat) has led to an increase in channel width and braiding, and due to the high alpine terrain of the Hoh Basin, its hypothesized that the Hoh could be particularly vulnerable to sediment increases from high-altitude warming and glacial melting (East et al. 2017). There are anthropogenically blocked areas due to culverts (Figure B51). Similar to the Strait, there has been an increase in the magnitude and frequency of flooding events on the west side of the Peninsula. From climate change, glacial extent declines have already occurred (up to ⅓ of summer critical water from ice) as well as increases in the frequency and magnitude of summer low flows and summer water temperatures (Dunham et al. 2023; and see Listing Factor E).

Northwest Indian Fisheries Commission (2020).[40] State of Our Watersheds report summarizes current habitat status (and thus impacts of past and current land use) for the major watersheds within the region. These reports by the Tribes for various watersheds also include useful maps of habitat quality and use and tables with summaries of habitat quality by Tribal indicators. Here, we briefly summarize those descriptions (note more on Climate Change from this report is found in Factor E). Note that most reports provide road density per square mile and densities of greater than 3 miles of road per square mile may impede habitat function (including due to sediment input). Also, there is overlap in area between some of these reports but we summarize conclusions for all that overlap with the OP steelhead DPS.

Quillayute river basin: The Quileute Tribe highlights that the area of interest for Quileute has 75% forest cover and forest cover conditions being good to healthy (NWIFC 2020). Similar to the Hoh watershed, there has been a decline in forest harvest activity with 24.5 square miles per year harvested from 2011-2015 and 17.1 square miles per year from 2016-2019. 56% of land in this region has road density that exceeds 3 miles per square mile. Many culverts that were barriers to fish passage were fixed, but 15% remain that were identified by Road Maintenance and Abandonment Plan (RMAP), while 57% of 371 barriers that were not part of RMAP still remain impassable. Similar to Hoh, peak flows in the region have increased while low flows have decreased. Invasive plant species like scotch broom, reed canarygrass, and herb Robert are problematic and could impact salmon, though knotweed presence has declined as a result of eradication work by the Tribe.

Hoh river basin: The Hob Tribe highlights that 80% of the area outside of the ONP has high road density (>3 miles per square mile). Many culverts that were barriers to fish passage were fixed but 20% remain that were identified by the RAMP, while on non-forestland 50% remain and are impassable. Between 2011-2015, forestlands were harvested at a rate of 12.5 square miles per year but this has dropped to 6.1 square miles per year since 2016. Invasive plant species like scotch broom, reed canarygrass, herb Robert, tansy ragwort, and Canada thistle are prevalent and could impact salmon, though knotweed has been controlled. As discussed under Factor E (Climate Change), there has been an increase in peak flows and decrease in low flows. High water temperatures impair many streams in this system with 14 bodies of water on the water temperature pollution list (303(d) list for water pollution), and for streams monitored by the Hoh Tribe, 8 of 9 have “widespread maximum temperature exceedances”. Though water temperature is a problem there has been improvement in pH and bacteria pollution.

Queets, Quinault, Chehalis river basins: The Quinault Indian Nation highlights that the area of interest for Quinault has 65% forest cover and 51% of the area has forest cover conditions of good to healthy. Areas in the region that are mainly private forestland have moderate to poor forest cover conditions and there was a 10% decline in forest cover from 1992-2016. At the same time, most of the area is not impacted by impervious surface (“good” for impervious surface indicator). Many culverts that were barriers to fish passage were fixed but 14% remain that were identified by RMAP while 23% of 728 barriers that were not part of RMAP still remain impassable. 87% of land in this region has road density that exceeds 3 miles per square mile,

[40] https://nwifc.org/publications/state-of-our-watersheds/#:~:text=The%20State%20of%20Our%20Watersheds,to%20the%20region's%20environmental%20health.

however, higher road density areas are disproportionately outside of OP steelhead DPS near Chehalis and Centralia. Similar to other watersheds already discussed, winter peak flows continue to increase and summer low flows are decreasing (except in Chehalis where low flow has increased but this is a rain dominated river and outside of OP steelhead DPS range). Water temperature pollution[41] and dissolved oxygen continue to be water quality problems but again with the majority of these streams being outside OP DPS range (Chehalis river). The addition of water wells continues in the general area and may negatively affect groundwater supply. Work by the Quinault Indian Nation has continued to control invasive species: knotweed, Scotch broom, tansy ragwort, and herb Robert (in the floodplains of Quinault and Queets rivers).

While cumulatively these habitat changes have been large over space and time, the Hoh River Basin, as well as the Queets, Quinault, and Quillayute still exhibit fundamental natural watershed processes and associated habitat characteristics. These include a large forested floodplain that is still intact and functioning. Further a large proportion of these watersheds lie within the ONP (Ericsson et al. 2022). Thus, efforts to protect, restore, and increase the overall resiliency of these larger rivers have been implemented to secure core natural assets (Ericsson et al. 2022).

Pacific Coast region: For the Pacific region as a whole, 5% of forest cover was removed from 2011-2016, with “properly functioning” riparian forest cover decreasing 34.2% between 2011-2016. Further, road densities that equate to “not properly functioning” accounted for 90% of the area in 2019 (an increase from 86% in 2014). Alternatively, at this time, 85% of RMAP culverts have been fixed; however, 226 blocking culverts (non-RMAP) still need correction. In most watersheds winter peak flows have increased (average increase of 12%) and summer low flows decreased (average -27%). Increases in the number of wells threatens groundwater availability, but with greatest amounts of new and existing wells occurring outside the OP DPS range (in Chehalis basin). Only 3% of this region's stream miles were assessed for water quality in 2014 with 86% of those impaired for one or more parameters identified by Washington Ecology. An intensive eradication effort focused on Knotweed has been successful and efforts have expanded to other invasive plants. Another invasive, the European green crab may spread if action is not taken to abate its expansion.

Northwest Olympic Peninsula: Habitat assessment by the Makah Tribe for this area (along the Strait and around the tip of the Olympic Peninsula), indicates that 81.2% of land has healthy or good forest cover, an improvement from 2011, but an overall decline since 1992. Around 81% of RMAP-identified culverts are not barriers to fish migration, but still more than half of non-RMAP fish culverts on private, federal, and county land remain and are impassable to fish. 83% of the land area has road densities of >3 miles per square mile. Water temperature and dissolved oxygen are the main water pollution problems with 79% of monitored stream length impaired due to temperature and 17% impaired due to dissolved oxygen. Big River is the most degraded with 16.1 miles of impaired stream from temperature, dissolved oxygen, and pH. Similar to the other regions, winter peak flows have increased and summer low flows have decreased

[41] Note that a public comment on the 90 day finding noted that the Washington Department of Ecology’s 2022 Water Quality Assessment shows half of Queets and Clearwater river watershed miles don’t meet water quality standards for temperature (as well as total maximum daily loads) - https://apps.ecology.wa.gov/ApprovedWQA/ApprovedPages/ApprovedSearch.aspx (last retrieved Apr. 10, 2023

(specifically shown for the Hoko River). Additionally, the Makah Tribe continues to work to address the invasive European green crab issue.

Morse Creek to Neah Bay: Lower Elwha Klallam Tribe summarized habitat conditions for portions of WRIAs 18and 19, which includes a portion of the OP steelhead DPS range. They report a net reduction of 1,966 feet of shoreline due to shoreline armoring (903 feet of new shoreline armoring, 475 feet of replacement, and 4,802 feet of removal of armoring). 91.5% of the area has little to no impact from impervious surfaces. Additions of water wells continue in the general area and may affect groundwater supply, but the rate of increase in the number of new water wells has slowed. 71% of the area has good - healthy forest cover with the most damaged conditions are near the town of Sequim (outside the DPS range). However, there were significant negative forest cover changes near the Pysht River. Invasive plants and animals (European Green crab especially) may be impacting species in the area. This report summarizes more specifics for rivers/streams outside of OP DPS range.

In general, urbanization has led to degraded steelhead habitat through stream channelization, floodplain drainage, and riparian damage (Botkin et al. 1995) (see summary inNational Marine Fisheries Service 1996a). Point source and nonpoint source pollution occur due to urbanization, impervious surfaces reduce filtration and increase run-off into the future (and creates flood risk; Leopold (1968)), and flood control and land drainage schemes may also increase flood risk. As the human population increases, additional urbanization and habitat modification are likely to occur. Recently, county populations on the Olympic Peninsula have increased by ~5-12% (see https://usafacts.org/data/topics/people-society/population-and-demographics/, Figure B52) which may continue into the future and would likely lead to continued habitat modification in the region.

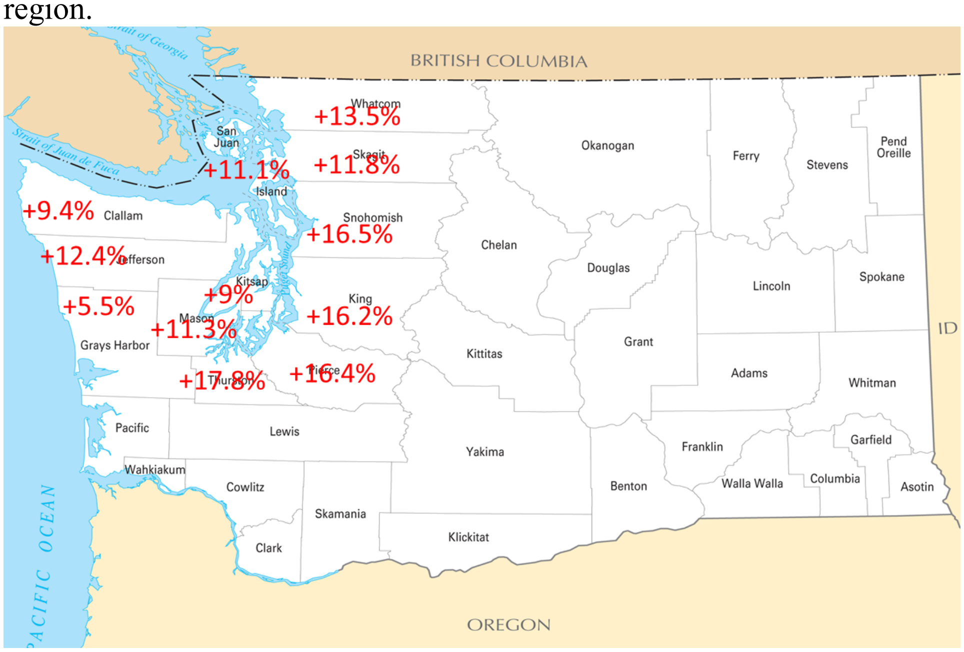

Figure 52. Human population growth 2010-2021 in counties surrounding the OP steelhead DPS

## Recent Research on Restoration Potential

A recent analysis by Beechie et al. (2023) for salmonids in the Chehalis River basin, showed coho are predicted to respond well (higher projected spawner abundance) to modeled restoration efforts across multiple restoration actions even when efforts were relatively low, while species like steelhead required more intensive and extensive restoration efforts to show predictable benefits (needed 75% of restoration potential to have a positive future projection). Similar results were found in simulations by Jorgensen et al. (2021) for Chehalis River basin salmonids, where restoration actions related to migration barriers, fine sediment, shade, floodplain, and large river channel length/bank condition had some positive benefit for steelhead (models showed increases in spawners), but restoration related to wood accumulation had the greatest benefit. Similarly, Pess et al. (2022) looked at wood reintroduction in the Deep Creek watershed and found that 23 years of wood additions led to deeper, more frequent pools, greater sediment storage, smaller sediment particle size, reduced stream width, reestablishment of vegetation in the riparian zone, and greater maintenance of floodplain channels.

Work by Martens and Devine (2023) looked at the last 25 years of modern forest management in a portion of the Olympic Experimental State Forest. Forest management of passive riparian forest restoration (sustainable harvest, riparian forest buffers) has restored stream canopies and reduced stream temperatures (papers cited therein). However, Martens and Devine (2023) found that this type of restoration has not led to improvements in instream wood and pools that are important for salmonid habitat; and stated that it may over time recover but could be on a much longer timeline.

Along with fish passage correction mentioned above, various projects funded through the Washington State Recreation and Conservation Office since 2000 have led to the protection and restoration of riparian habitat for almost 33,000 acres on the Washington coast (Coast Salmon Partnership 2022 Annual Report - https://coastsalmonpartnership.egnyte.com/dl/VbBakQwmdS). This annual report summarizes various restoration efforts for WRIAs within the OP steelhead DPS boundaries (WRIA 20, 21) including many efforts by Tribes. In WRIA 20 (Pacific Coast from Cape Flattery to the Quillayute and Hoh rivers), there have been corrections to 36 fish passage barriers, improvement of sediment transport due to ~450 acres of upland area restoration, 1,353 acres riparian restoration, 11 acres of floodplain reconnection, and 30 miles of restoration instream. In WRIA 21 (predominantly Queets and Quinault rivers), corrections to 33 fish passage barriers have occurred, improvement of sediment transport due to ~480 acres of upland area restoration, 5,939 acres riparian restoration, 14 acres of floodplain reconnection, and 6 miles of restoration instream. For the Pacific Coast Region, that includes watersheds south of the OP, the State of Washington had repaired or replaced 99 fish blocking culverts in the first six years of the program; this however, apparently leaves 226 culverts yet to be replaced by 2034 (Northwest Indian Fisheries Commission 2020).

## Ocean Habitat

Myers (2018) describes in depth what is known of the ocean ecology of steelhead. Across the North Pacific steelhead are sparsely distributed throughout their ocean range and distribution varies by age and maturity group. Although steelhead (*O. mykiss*) is an iconic species found throughout the North Pacific rim, little is known about its ocean ecology. To provide insights into migratory routes and habitats occupied by steelhead in the North Pacific Ocean, Courtney et al. (2022) attached pop-up satellite archival tags (PSATs) to steelhead kelts in 2018 ($n = 16$), 2019 ($n = 12$), and 2020 ($n = 35$) from the Situk River, a robust Alaskan population. PSATs recorded extensive post-spawning migrations extending to the western North Pacific Ocean, and as far north as the central Bering Sea. While at sea, tagged steelhead spent the majority of their time in surface waters (< 5 m) and occasionally dived to 15–20 m, but displayed no observable diel depth-based behaviors. Tagged steelhead kelts experienced a thermal environment of 4–16 °C from June to January, after exiting the Situk River. Results from this project corroborate the limited past research suggesting that steelhead predominantly occupy surface waters and that their distribution is largely influenced by sea-surface temperatures of ~5–15 °C. Additionally, results from this study suggest that the waters near the Aleutian Islands are important feeding grounds for steelhead kelts from the Situk River, and thus may play a critical role in the successful reconditioning of repeat spawners in this population. These results provide the first detailed insights into the ocean ecology of steelhead and may be used for a variety of applications (e.g., niche construction, and forecasting future range dynamics under climate scenarios).

Factors in the marine environment influencing steelhead survival include predation, access to prey (primarily forage fish), contaminants (toxics), disease and parasites, migration obstructions, and degraded habitat conditions which exacerbate these factors.

Studies on tagged steelhead suggest that the fish closely track preferred sea surface temperatures (and likely other conditions) during their marine migrations (Courtney et al. 2022; Hayes et al. 2016). Work summarized in Myers (2018) also points to sea surface temperature as the primary physical factor impacting marine distribution; salinity and currents may also influence distribution. At sea, steelhead tend to travel at depths less than 5 meters (Courtney et al. 2022), and so are more likely to respond to changes in sea surface temperatures than if they traveled at deeper depths with more constant temperatures. However, in certain cases steelhead have been documented remaining off the coast from their natal river and returning to the natal river just a few months after ocean entry. The increased expression of this, more localized, ocean migration life-history strategy may indicate thermally blocked marine migratory corridors or changing ocean conditions. Myers (2018) speculates that prey availability may be the primary biological factor impacting steelhead distribution (see that work for a complete discussion of diet at various life stages).

Work from Myers et al. (2013) summarizes that salmon and steelhead can consume a variety of plastic on the high-seas such as pellets, foams, sheets and that the presence of plastic in the stomach varied by species, age, maturity group, area, and time. They note that potential mechanisms for mortality could be direct through mechanical injury or toxicity, or indirect through impacts to gene expression in offspring and subsequent effects to offspring survival.

A likely threat to marine habitat of steelhead is climate change and this is discussed in the section on listing Factor E.

# Listing Factor B: Overutilization for commercial, recreational, scientific, or educational purposes

The discussion under this heading focuses on the patterns of utilization for commercial, recreational, and Tribal purposes. Current management and regulatory schemes for these activities are discussed more in depth under the heading Listing Factor D: Inadequacy of Existing Regulatory Mechanisms and are only discussed here to give context to the harvest data. Harvest of OP steelhead has declined within in the last decade (particularly the last few years) and varies greatly by region (Strait of Juan de Fuca populations vs. the "four major basins" on the coast – Queets, Quinault, Quillayute, and Hoh). We summarize primarily what has occurred since the last NOAA status review (Busby et al. (1996) report), though also provide some information for earlier. Most information presented here is for winter-run natural-origin steelhead in the major four basins (Queets, Quinault, Quillayute, and Hoh – which we refer to as the major four basins) and there are limited data for rivers draining into the Strait of Juan de Fuca (where harvest is mainly terminated) and for summer-run natural-origin steelhead.

Historically, steelhead were abundant in many western coastal and interior streams of the United States. These steelhead populations supported numerous coastal and inland indigenous tribal fisheries precontact, and Tribal, commercial and recreational fisheries thereafter (Nickelson et al. 1992). In 1932 the newly formed Washington State Game Commission prohibited the commercial catch, possession, or sale of steelhead (Crawford 1979). See the summary of historical fisheries management of steelhead on the Olympic Peninsula and Washington coast in the WDFW Coastal Steelhead Proviso Implementation Plan (CSPIP) (Harbison et al. 2022) available here: https://wdfw.wa.gov/publications/02360.

We do note that Indigenous groups have managed fisheries and the landscape since time immemorial (for example see explanation in Martin 2023), during a time when steelhead thrived. A document by Martin (2023) from Makah notes that sustainable harvest management is a core principle of traditional resource management and embedded into Tribe societal roles, salmon and steelhead have been managed since time immemorial (including their habitat), and this management included both traditional hatchery and harvest practices (see further information from that document presented in Factor D).

The river systems throughout the Olympic Peninsula DPS support sport fishing and commercial, ceremonial, and subsistence gill-net fisheries, with Pacific salmon and steelhead populations subjected to fishing pressure and harvest during most weeks of the year. The highly popular sport fisheries that include guided and non-guided sport fishing are economically important to local communities. Commercial catches of Pacific salmonids are integral to the Treaty Tribes and fish are sold to local, regional, and national markets. Subsistence catch is for personal consumption and ceremonial catch is taken for cultural events by Treaty Tribes. There is no directed ocean harvest of steelhead.

OP steelhead have in the recent past sustained the highest harvest rate among Washington state steelhead populations with an annual harvest rate of 25.6 percent for natural-origin steelhead averaged across rivers where there was data (Cram et al. 2018) and was also highest in the four major basins (Queets, Quinault, Quillayute, and Hoh) – 36.5% from 1980s to 2013. Specifically, until 2013, winter-run OP natural-origin steelhead in the Hoh, Queets, Quinault, and Quillayute systems have had harvest rates ranging from 7% to greater than 48% annually since the 1980s (Table 29. A). Although fishing mortality has been relatively high, it is likely that this factor alone is not the cause of run size declines, and there are other factors in combination with harvest that underlie observed declines.

Estimates of harvest rates since the 1980s for winter-run natural-origin steelhead for the four major systems (Hoh, Queets/Clearwater, Quillayute, and Quinault [upper + lower]) were provided by the co-managers along with estimated run size. This information can be used to estimate percent harvest mortality (Figure 31, Table 29). More recently (2014-2022) harvest rates in the major four basins have ranged from 13.26%-59.19% depending on year and basin up through 2020, with Queets and Quinault continuing to have average harvest rates in the last decade (2013-2022) of 27% and 36%. During this period harvest rates in the Quillayute and Hoh rivers averaged closer to 20%. In the last 2 years for which we have records, 2021-2022, there have been substantial declines in harvest rates, with harvest rates of 8.66% to 15.44% (across basins) (Table 30). Notably, outside of the major coastal basins, harvest in most rivers along the Strait of Juan de Fuca was terminated in the late 2000s/2010s (see Figure 35, but see Hoko) and we present harvest information for populations along the Strait of Juan de Fuca (section *Population Growth and Harvest in Strait Populations* and further in this section).

Sport and Tribal catch of winter-run population has typically occurred from November- April. In 2004, Olympic National Park implemented catch-and-release regulations for wild steelhead throughout coastal rivers of the DPS within the park. In 2016, WDFW changed the recreational fishing regulations (not including streams in tribal reservations and the ONP) to prohibit retention of natural-origin winter-run steelhead in OP steelhead river basins. Sport and Tribal catch of winter-run population has typically occurred from November- April. Tribal harvest targets early returning winter steelhead, which includes both hatchery- and natural-origin steelhead. Most estimates of harvest rates we present here do not include catch and release mortality (hooking mortality), but there is a management assumed 10% hooking mortality (see below in this section for further information on where included, including for the Hoh River). However, information from Bentley (2017) led to a sport angler encounter rate calculation of 1.14 for wild steelhead, implying some steelhead are caught and released more than once (Harbison et al. 2022), and hooking mortality is not known for fish handled more than once and this may be contributing additional mortality. For the Queets and Quinault rivers, regulations allow for retention of steelhead with a dorsal fin of less than 2 1/8 inches, the height of a credit card (so named the "credit card rule"), because hatchery-origin steelhead are assumed to have eroded dorsal fins and majority hatchery fish in these systems are not marked these basins. Other regulations related to prohibiting bait, limits on hooks, size limits etc. are listed in Appendix 12.4 of Harbison et al. (2022). Recreational fisheries on tribal lands do not prohibit the retention of natural-origin steelhead.

On January 26, 2024, the co-managers clarified for the SRT in a written response what data are included in estimates of run size and harvest (email correspondence with Jim Scott, on behalf of the co-managers, January 26, 2024). For the Hoh River, run size and total catch of natural-origin steelhead included hooking mortality in the sport fishery dating back to 2003/2004 season. The estimated mortality was based on total estimated encounters from sport creel surveys multiplied by 10%, the presumed hooking mortality rate. For the Quillayute, Queets, and Quinault Rivers, annual run reconstruction and total catch of wild steelhead does not account for hooking mortality in the sport fishery. Therefore, the total number of natural-origin winter steelhead mortalities was underestimated for those rivers in all years. For the Hoh River and Quillayute River Basins, ceremonial and subsistence fisheries were included in the estimates of total run size. For the Queets and Quinault systems, on reservation hook and line harvest was included, although the sport on-reservation harvest component for the Queets River was not included in the harvest estimates or associated run reconstruction until the 2020/21 season. Furthermore, there are key differences in estimates of natural-origin steelhead escapement in surveys in Quillayute/Hoh versus Queets systems. The Quillayute/Hoh estimates are based on number of redds x 0.81 female/redd x 2 fish. In the Queets, the estimator is total number of redds x 1 female/redd x 2 fish. Assuming 1,000 redds in a given river, these escapement estimates of natural-origin fish vary by 19%. Harvest rates for winter-run steelhead include any and all steelhead landed in the weeks between week 45 (approximately November 1st) and week 18 in the following year (approximately April), regardless of what fishery / what species was being targeted (Scott, J.B. OP steelhead follow-up questions. Email to Laura Koehn. 17 July 2024); however, any steelhead caught in other salmonid fisheries outside this time period were not included.

Table 30. Annual harvest rates for specific winter-run populations (natural-origin and hatchery combined) for the years specified (from Cram et al. 2018).[42]

| Population (winter-run) | Harvest Rate | Years |
|---|---|---|
| Clallam | 0.7% | 1999-2013 |
| Goodman | 6.8% | 1995-2009 |
| Hoh | 36.7% | 1980-2013 |
| Pysht/Independents | 14.0% | 1995-2013 |
| Queets system | 35.5% | 1981-2011 |
| Quillayute system | 29.6% | 1978-2013 |
| Quinault system | 48.2% | 1991-2013 |
| Salt/Independents | 3.9% | 1995-2013 |

[42] Post-spawn steelhead (kelts) may be harvested while returning to the ocean. The redds created by these steelhead (females) would be used to estimate escapement. Thus, there may be double counting of some fish in estimating harvest rates. Given that there is relatively limited harvest in the March-May harvest time, when kelts are emigrating, the bias may not be large.

Table 31. Calculated harvest rate for the four largest basins (Queets, Hoh, Quinault, Quillayute) for years in the late 1970s/early 1980s to 2022, from run size and escapement data provided by the co-managers (Tribes and WDFW), where harvest is equal to run size – escapement and percent mortality is equal to harvest / run size.

| **Year** | Hoh | Queets Clearwater System | Quillayute System | Quinault (Upper + Lower) |
|---|---|---|---|---|
| **1978** | | | 17.23% | |
| **1979** | | | 32.67% | |
| **1980** | 0.00% | | 30.73% | |
| **1981** | 0.00% | 47.27% | 22.40% | |
| **1982** | 0.00% | 38.43% | 23.01% | |
| **1983** | 0.00% | 45.78% | 18.68% | |
| **1984** | 0.00% | 45.76% | 19.45% | |
| **1985** | 0.00% | 49.50% | 40.71% | 49.17% |
| **1986** | 0.00% | 45.32% | 25.28% | 34.38% |
| **1987** | 35.76% | 48.71% | 33.31% | 66.33% |
| **1988** | 49.07% | 48.50% | 38.29% | 50.77% |
| **1989** | 36.40% | 41.83% | 28.45% | 48.24% |
| **1990** | 47.18% | 42.84% | 38.24% | 42.83% |
| **1991** | 33.83% | 37.26% | 38.00% | 46.01% |
| **1992** | 54.35% | 41.27% | 54.38% | 57.40% |
| **1993** | 50.46% | 38.97% | 53.10% | 60.41% |
| **1994** | 43.86% | 28.16% | 33.69% | 40.11% |
| **1995** | 38.28% | 39.20% | 34.89% | 42.85% |
| **1996** | 42.89% | 54.80% | 29.72% | 52.18% |
| **1997** | 27.55% | 41.55% | 35.96% | 41.15% |
| **1998** | 7.24% | 28.87% | 10.30% | 51.93% |
| **1999** | 24.93% | 42.77% | 21.50% | 46.20% |
| **2000** | 29.23% | 30.25% | 28.39% | 45.96% |
| **2001** | 48.29% | 31.48% | 36.48% | 59.85% |
| **2002** | 45.15% | 10.40% | 28.23% | 61.40% |
| **2003** | 54.90% | 35.06% | 28.04% | 54.90% |
| **2004** | 44.04% | 17.22% | 25.74% | 62.01% |
| **2005** | 41.71% | 16.37% | 24.25% | 43.93% |
| **2006** | 10.97% | 14.61% | 18.25% | 41.03% |
| **2007** | 22.69% | 28.43% | 36.14% | 38.63% |
| **2008** | 30.91% | 19.22% | 25.78% | 31.77% |
| **2009** | 28.18% | 23.95% | 30.25% | 45.91% |
| **2010** | 26.56% | 29.56% | 27.32% | 37.54% |
| **2011** | 20.37% | 35.07% | 19.48% | 29.52% |
| **2012** | 28.50% | 42.64% | 29.41% | 56.30% |

| Year | Hoh | Queets Clearwater System | Quillayute System | Quinault (Upper + Lower) |
|---|---|---|---|---|
| **2013** | 36.76% | 38.28% | 29.16% | 49.12% |
| **2014** | 43.19% | 31.31% | 26.65% | 47.46% |
| **2015** | 26.58% | 30.67% | 29.19% | 44.43% |
| **2016** | 19.31% | 29.16% | 30.34% | 59.19% |
| **2017** | 16.63% | 39.78% | 16.53% | 33.41% |
| **2018** | 13.79% | 20.86% | 15.63% | 28.14% |
| **2019** | 13.26% | 29.90% | 13.90% | 36.51% |
| **2020** | 19.31% | 29.91% | 13.94% | 37.39% |
| **2021** | 12.29% | 9.76% | 10.93% | 15.44% |
| **2022** | 9.96% | 8.66% | 8.93% | 11.31% |

Typically, summer-run and winter-run steelhead fisheries in ONP are catch and release for wild fish and anglers can retain up to 2 hatchery steelhead per day during the open season, typically from June 1-March 31. The ONP has required catch-and-release of wild summer steelhead since 1992 and for wild winter steelhead starting in 2005. Additionally, the park has implemented numerous conservation measures including: in-season emergency actions to protect wild steelhead in major coastal watersheds (e.g. closures, reduced seasons, gear restrictions), selective gear regulations and eliminating the use of bait in most sport fisheries directed at wild steelhead; dedication of fly fishing only water (eg. Hoh River); several major habitat restoration efforts including culvert replacements, permitting of steelhead fishing guides with requirements of reporting of daily catch, effort, and CPUE at the end of each season. See the section on Listing Factor D for further regulations within the ONP (and https://www.nps.gov/olym/upload/OLYM_Fish_Brochure_2022-0502-508_all_CHARTs_REMOVED.pdf ).

The number of natural-origin OP steelhead that are encountered in the sport fishery is calculated by WDFW via creel surveys, with a 10 percent catch-and-release mortality rate and is included in the harvest estimates for the Hoh River basin data. WDFW recently reported recreational fishing pressure for two coastal rivers ( https://wdfw.wa.gov/newsroom/news-release/public-invited-oct-25-virtual-town-hall-meeting-coastal-steelhead). In the Hoh River, a total of 666 interviews were conducted with an estimated 57,273 angler hours in 2022-23 fishing season from December 16-March 31. WDFW estimated that 3,575 wild steelhead were caught and released or ~86% of the run. In the Sol Duc River, WDFW conducted 264 interviews with an estimated 28,329 angler hours. Anglers reported catching 2,204 wild steelhead or ~55% of fish that entered the Sol Duc. Work by Bentley (2017) from WDFW presented evidence that coastal creel catch since 2002-2003 has likely underestimated true total catch (specifically in Hoh and Quillayute watersheds), possibly due to a false assumption that all fishing pressure occurs in index effort count reaches (specific regions used to estimate effort). This work highlights multiple needs for improvements to creel surveys and information storage (including the surveys should extend past April since steelhead continue to spawn) (https://wdfw.wa.gov/publications/01918). Finally, Bentley (2017) estimated total harvest and release mortality for lower and upper Hoh River and estimated a total of 2,977 released wild in the lower, and 1,603 released wild in the upper from Dec 2014- April 2015 (total escapement goal is 2,400 for the Hoh). Information from Bentley

(2017) led to a sport angler encounter rate calculation of 1.14 for wild steelhead, implying some steelhead are caught and releases more than once (Harbison et al. 2022). These are no mortality rate estimates for steelhead caught multiple times, but it is likely that this rate would be higher than the first-time encounter rate. Overall, given that the SRT did not have a complete estimate of hooking mortality for most populations, it was presumed that available estimates were a minimum at best, and hooking mortality could even be larger than landed catch in certain systems especially in the last few years when landed catch has been low (in the low hundreds of steelhead).

Many public comments on the 90-day finding voiced concerns that the 10% catch and release mortality rate used by WDFW is too high. Many cited a recent thesis by Lubenau (2022) that found catch and release mortality rates for *O. mykiss* closer to 4% in the Lower Granite Dam area of the Snake River. A more recent publication update by Lubenau et al. (2024) reported a catch and release mortality for wild steelhead in the Snake River of 1.6% (95% credible interval of 0% to 5.2%) and encounter rates of approximately 44% to 47% for wild steelhead (slightly higher for adipose fin clipped steelhead: approximately 47% to 52%). In light of the high encounter rates in the recreational fishery it is likely that many natural-origin steelhead are hooked multiple times, and it is unclear how much this might change the catch and release mortality rate.

Additional conservation strategies since the 1990s have been implemented in commercial and recreational fisheries including: harvest restrictions, shorter seasons, and gear restrictions (Harbison et al. 2022) (see Listing Factor D). In recent years, WDFW has shortened or closed the recreational fishing season on winter-run OP steelhead at least in part due to low returns. WDFW also imposed restrictions on recreational angling by banning the use of boats ("no fishing from a floating device") and bait (see the following*: https://wdfw.wa.gov/publications/02349; https://wdfw.medium.com/changes-to-the-coastal-steelhead-season-67131dd05ba7; https://wdfw.medium.com/frequently-asked-questions-march-2022-coastal-steelhead-closure-364cfa62826f; https://www.peninsuladailynews.com/sports/fishing-olympic-national-park-to-shut-down-fishing-on-west-end-rivers/* ).. In 2022-2023 sport fishing was closed on the Quinault and Queets for December 1st- April 30th because of low returns and because agreement was not reached for natural-origin steelhead harvest level. The total number of weeks for Tribal fisheries has declined in recent years (see more information below) specifically in the Queets and Quinault, and as mentioned before, harvest rates have declined. In addition, WDFW added harvest restrictions to protect Bogachiel Hatchery returns (https://wdfw.wa.gov/newsroom/news-release/wdfw-announces-2022-2023-coastal-fishing-season). See links for additional specifics for gear and other restrictions. WDFW implemented similar gear and floating device restrictions for 2023-2024 and set a bag limit of two hatchery steelhead (https://wdfw.wa.gov/newsroom/news-release/wdfw-announces-2023-2024-coastal-steelhead-season). For the 2023-2024 season, the National Park Service closed Queets and Quinault Rivers within the ONP to sports fishing beginning on November 27th, 2023 (https://www.nps.gov/olym/learn/news/temporary-sport-fishing-closure-necessary-to-protect-declining-populations-of-wild-steelhead.htm).

In Factor D (Inadequacy of Existing Regulatory Mechanisms), we provide more detail on how fisheries are managed, specifically that OP steelhead fisheries are mainly managed for escapement goals for winter-run steelhead based on freshwater productivity (see Gibbons, Hahn and Johnson 1985). The established escapement goals vary much by river system and range from <100 (in smaller rivers on the Strait) to 5,900 natural origin winter steelhead (Table 4). In the Queets River system, the co-managers have differing escapement goals. Each year, specifically for the four major systems, the co-managers develop management plans outline forecasted run sizes, escapement goals, harvest rates, and fishing seasons. For Quinault, though escapement was met in the most recent years (Figure 53), escapement was met only 43% of the time since 1970. In recent years (2021-2022) harvest rates were lowered (as noted above) because of low returns in some systems, but not necessarily to the extent needed to meet escapement goals. Specifically, in the Queets, the State-specific escapement goals were not met in 2020-2021 and 2021-2022 even with the lower harvest rates because returns were low. The returns, however, met the Tribal escapement goal, which is lower. For 2023, in the Queets the projected return was 4,150 (beginning below the State escapement goal), and State and NPS closed fishing but the harvest rate was set at 16% for the Tribal fishery, leading to an estimated escapement of less than the 4,200, lower than the State escapement goal but greater than the Quinault Tribe escapement goal. This is not the case in each system and each year. For example, in the Quillayute River, the 2022 harvest led to an escapement level above the escapement goal (Quileute-WDFW 2022 plan). The escapement goal is more consistently met in the Quillayute River (Figure 53) Similarly for the Hoh River, in 2020 harvest rates were set to achieve an escapement slightly over the goal (2,485 projected natural-origin escapement). However, like other systems, there is continued harvest in cases where escapement goals are not met in these systems. The accuracy of forecasting influences the potential for meeting escapement goals. There is some ability to provide in-season management; “Tribal fisheries are generally shaped by time and area restrictions with in-season management based on monitoring of fishery catches,” (Co-Manager Olympic Peninsula Steelhead Working Group 2023). Whether escapement is met also depends on which (State or Tribal) escapement goal is considered. Even with lowered harvest rates in recent years, certain systems harvest rates (and also likely hooking mortality) are still leading to adult returns under the State escapement in the Queets (but not the Tribal escapement goal).

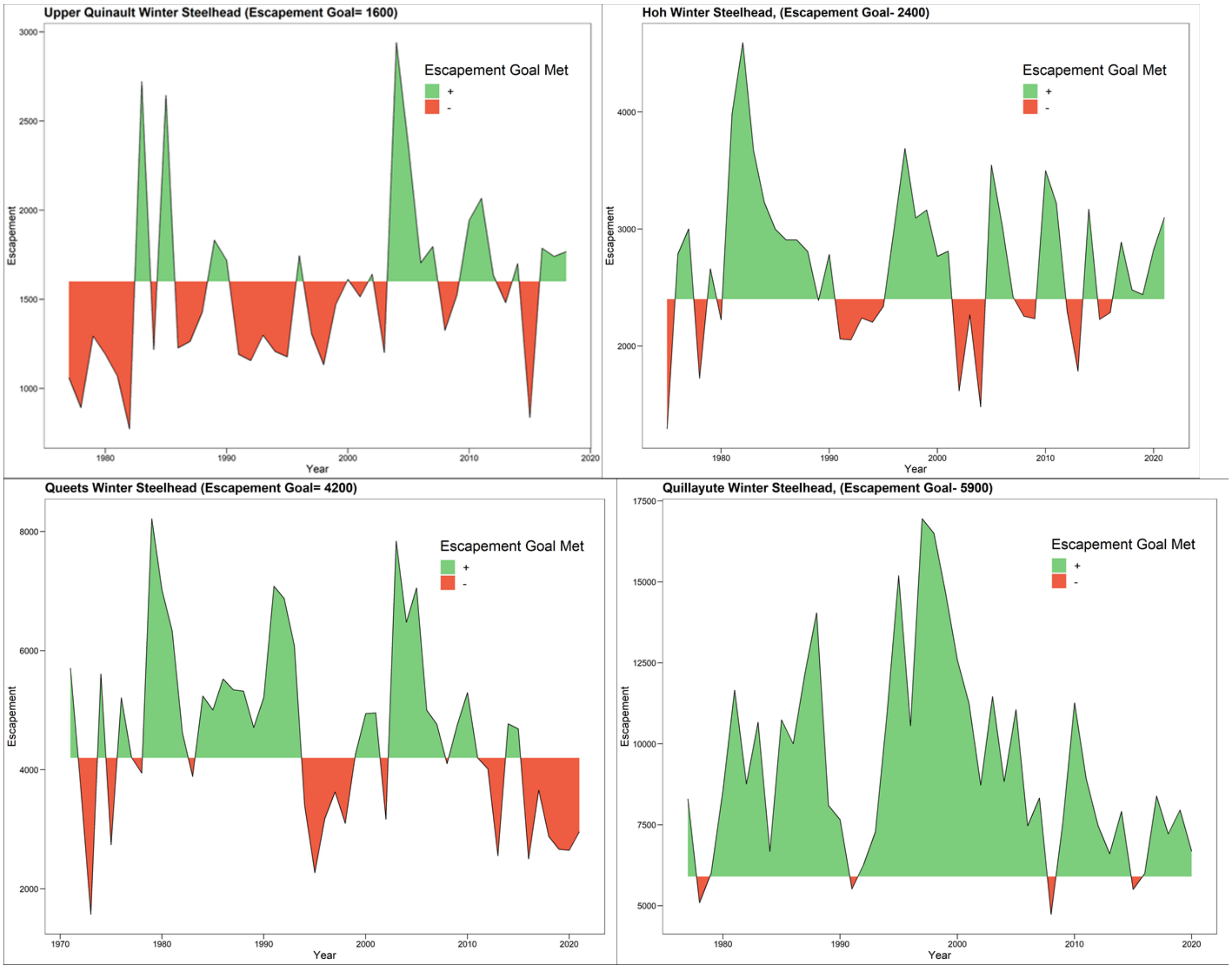

Figure 53. Winter steelhead escapement and escapement goals for the a) Upper Quinault River b) Hoh River, c) Queets River, d. Quillayute. Note that the Washington State escapement goal for the Queets River is 4200, but the Quinault Tribal escapement goal is 2500.

Forecasting accuracy certainly influences whether harvest rates are set to achieve escapement goals in the Olympic Peninsula DPS (Figure 49). In-season harvest monitoring provides some ability to manage escapement. The co-managers state in their 2023 review to the SRT that, "Tribal fisheries are generally shaped by time and area restrictions with in-season management based on monitoring of fishery catches," (Co-Manager Olympic Peninsula Steelhead Working Group 2023). The co-managers provided examples of in-season management and management taken in recent years (Scott, J.B. OP steelhead follow-up questions. Email to Laura Koehn. 17 July 2024). Specifically, for the Quillayute River, in-season fishery monitoring led to an earlier closure in February of 2022 given low returns and low harvest, leading to harvest of 385 fish and escapement of 8,516 (above the escapement goal). Since the 2021/22 season which had the lowest run size of recent years, there has been an increase in on-river days to 52.7 in 2022/23 and 57.7 in 2023/24 (up from 48.7 in 2021/22) and total run sizes of 9,344 and 9,096 in these years (above escapement, with the 2023/24 escapement still being projected and not a final estimate). For the Hoh River, Tribal fishing has closed in weeks 13-16 since 2015 as this was identified as peak steelhead run time. This has increased in the Hoh recently to 17 weeks in 2024 but with less participation in the fishery. In the Queets and Quinault rivers, total fishing days have fluctuated through the years during periods of severe changes in ocean conditions. Specifically, in the 1990s to early 2000s, fishing days on the Queets were reduced from an average of 91 to an average of 68 days, and in the Quinault, days were reduced from average 106 to 100 days, particularly later in the season (March, April) when there natural-origin spawning in both rivers. In the mid 2000s, average days of fishing increased (average 102 days on the Queets and average 104 days on the Quinault), but at roughly 50% of harvest levels observed in the 1970s. Between 2017/18 and 2020/21 seasons, fishing days were again reduced to 78 days and 88 days on average on the Queets and Quinault rivers, respectively, and early closures were implemented. Finally, in the most recent seasons (2021/2022 and 2023/24), average gillnet days have been reduced to 35 days in each system (Queets and Quinault) with early closures in February and early sport closures as well (in February or early March), leading to catch limits of natural-origin fish at around 200 fish (<10% harvest rates).

Cram et al. (2018) indicates that harvest may be affecting diversity in a number of life history traits: body sizes, age at maturation, and run-timing. Analysis of scale samples indicated that Tribal fisheries harvested more of the older fish, whereas the recreational fisheries harvested more of the younger fish (Cram et al. 2018). Additionally, for tribal-treaty fisheries, the number of fishing days per week declines during course of harvest period, possibly leading to greater fishing pressure on early-returning adults (Cram et al. 2018). McMillan et al. (2022) found evidence of a shift in peak run-timing of natural-origin winter-run to 1-2 months later and a contraction in overall run-timing by at most 26 days. One hypothesis provided is that fisheries targeting early-run time hatchery fish may have also overharvested early-run natural-fish (also see analysis in the Status Review in the section *SRT assessment of winter-run run timing changes*). We note that iteroparity (repeat spawning) has declined in the four major coastal rivers, but it is unknown if this is connected to harvest and the assessment by the (Co-Manager Olympic Peninsula Steelhead Working Group 2023) suggests it's related to climatic and biological factors (see Listing Factor E).

Further specifics on harvest in certain watersheds were described in Harbison et al. (2022) and summarized here (but see Listing factor D – Harvest Regulations as well) :

> **Quinault:** State data from 2007-2021 and co-manager knowledge suggest that on any given day there are 10-15 recreational anglers and 2-5 guides in the upper Quinault system during the open season. Note that fishing is managed by the Quinault Indian Nation below Lake Quinault (lower Quinault) and managed above the lake by WDFW (upper Quinault). The escapement goal for upper Quinault is 1,600 steelhead.
> **Queets:** Fishing is managed by the state and Quinault Indian Nation outside of the ONP, and by the National Park Service within the ONP. Recreational fishing is primarily boat angling but on the Salmon River is limited to angling from the bank and portions of the Salmon River are only accessible with a Tribal guide. State data from 2007-2021 and manager knowledge suggest that on any given day there are 55-65 recreational anglers and 13-18 guides in the upper Quinault system during the open season. The escapement goal set by the state is 4,200 steelhead, and by the Quinault Tribe is 2,700.
> **Hoh:** Fishing outside of the ONP is co-managed by WDFW and the Hoh Tribe. Though the Hoh River has both bank and boat access for fishing, since 2016 limits have been placed on use of floating devices in portions or all of the river. Outside of the ONP and not including the South Fork Hoh, there are estimated to be 60-65 recreational anglers

daily on the Hoh River based on creel surveys and 25-30 professional guides, during the open season. The co-managed escapement goal is 2,400 fish.
**Quillayute (including Sol Duc, Bogachiel, Calawah, and Dickey):** Fishing outside of ONP is managed by WDFW and Quileute Indian Nation. Both boat and bank access are used but it is difficult to wade long distances on the Sol Duc and Calawah. On the Bogachiel, there are 20-25 recreational anglers daily and 8-10 guides in the open season based on WDFW data and regional manager information. This is 30-35 recreational and 10-15 guides on the Sol Duc daily, 20-25 recreational and 5-10 guides on the Calawah, 15-20 recreational anglers and 0-2 guides on the Dickey, and 6-8 recreational and 2-4 guides on the mainstem Quillayute.

Harvest over time since the 1980s for winter-run natural-origin steelhead for the four larger systems (Hoh, Queets/Clearwater, Quillayute, and Quinault [upper + lower] rivers) was provided by the Co-managers along with estimated run size which can be used to estimate percent mortality. This information is also presented in the status review (see section Means and geomeans of escapement). Harvest and percent mortality due to harvest have declined in the most recent years (but note that harvest is mainly from November to April while escapement only concerns redds produced after March 15th when it's assumed fish are natural-origin steelhead) (Figure 54). However, analysis of larger rivers (Quillayute, Hoh, Queets, Quinault) indicated that total run size had nearly halved in size from late 1970s and 1980s to 2022, while the recent trend in escapements was slightly declining or stable (Harbison et al. 2022) (see Figures 15-18 in the status review report, section *Abundance and Productivity*).

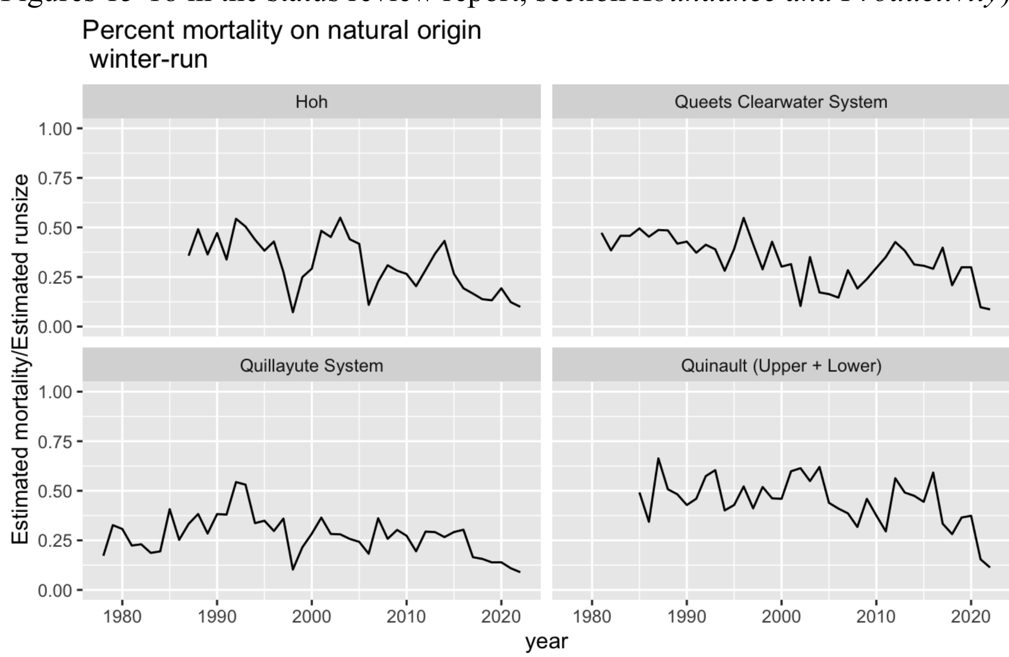


Figure 54. Percent mortality of natural-origin (escapement after March cutoff) winter-run steelhead (harvest mortality divided by estimate run-size) reported by co-managers for the four major coastal rivers. Recreational hooking mortality is only included for the Hoh River.

From a joint time series model for escapement and harvest, we can estimate population growth rate (mu) and harvest mortality for the four major systems:

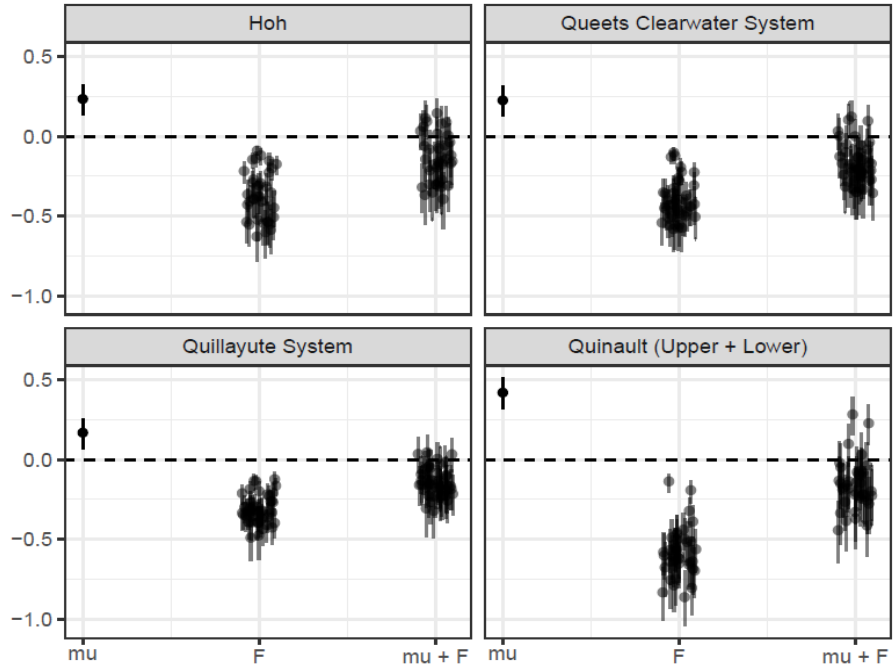


Figure 55. Estimated log scale population growth rate (mu), estimated annual harvest mortality (F), and the net population growth rate (mu + F) and 95% CI for each. For F and mu + F, each point represented the estimated value in a particular year.

The SRT model for harvest mortality fits and produces reasonable estimates of escapement and harvest (Estimates for this model suggest that these populations largely have an intrinsic population growth substantially greater than zero (point estimates of $\mu_i > 0.15$ for all populations). However, they are also subjected to substantial fisheries mortality and in most years this fishing mortality is greater than intrinsic mortality (i.e. generally $\mu_i - F_{it} < 0$; Figure 5), which will result in declining population growth (Figure 55). A small minority of years in each population were judged to have population growth greater than zero. Estimates of correlation among populations were positive and large, indicating that all four of these populations fluctuated in unison ($\theta$ = 0.83[0.62, 0.97] mean[95%CI]).

We also have harvest information for populations along the Strait of Juan de Fuca (the "Strait") (Figure 56). For harvest in rivers along the Strait, we can plot estimates of growth rates for each population through time highlighting the time harvest ceased (Figure 57) though patterns in growth rate appear highly correlated between streams even in ones where fishing has not ceased. Therefore, it appears some other factor (freshwater and/or ocean conditions) is influencing trends in these populations.

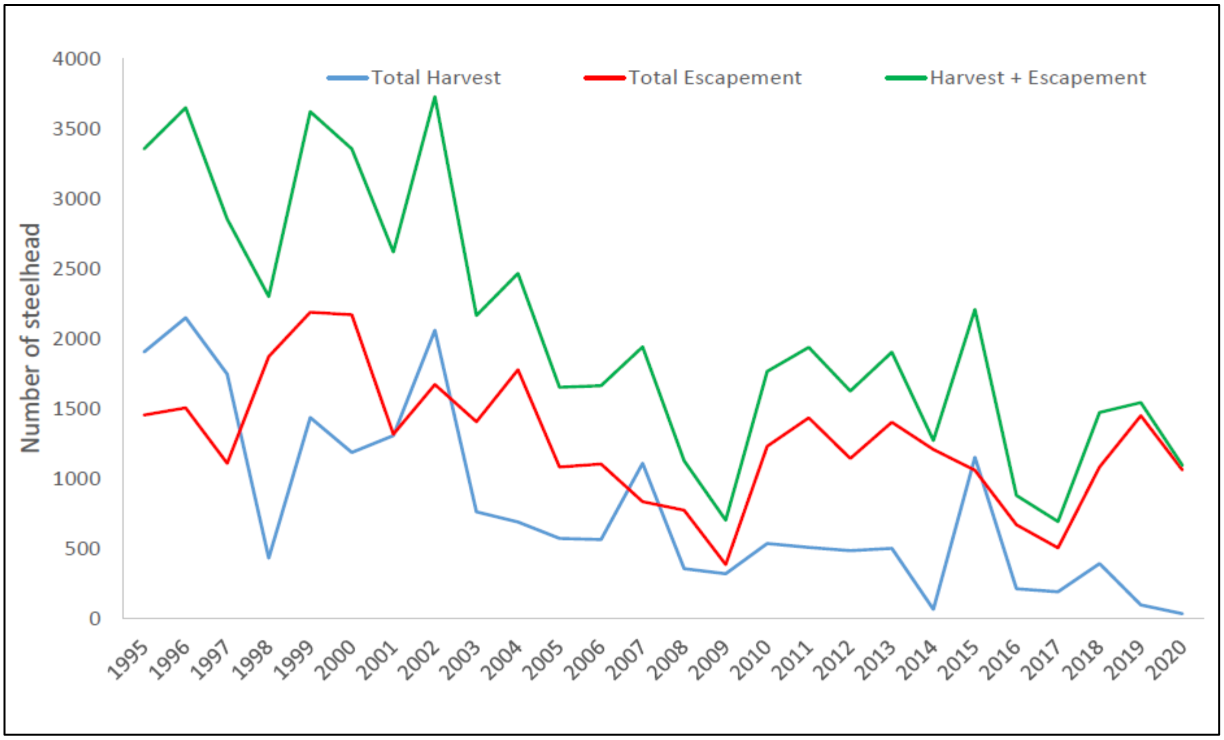


Figure 56. Harvest, escapement, and escapement + hatchery of all steelhead (hatchery and natural-origin) in the Strait system based on catch record cards. Note that in a few years since 1996, harvest has exceeded escapement.

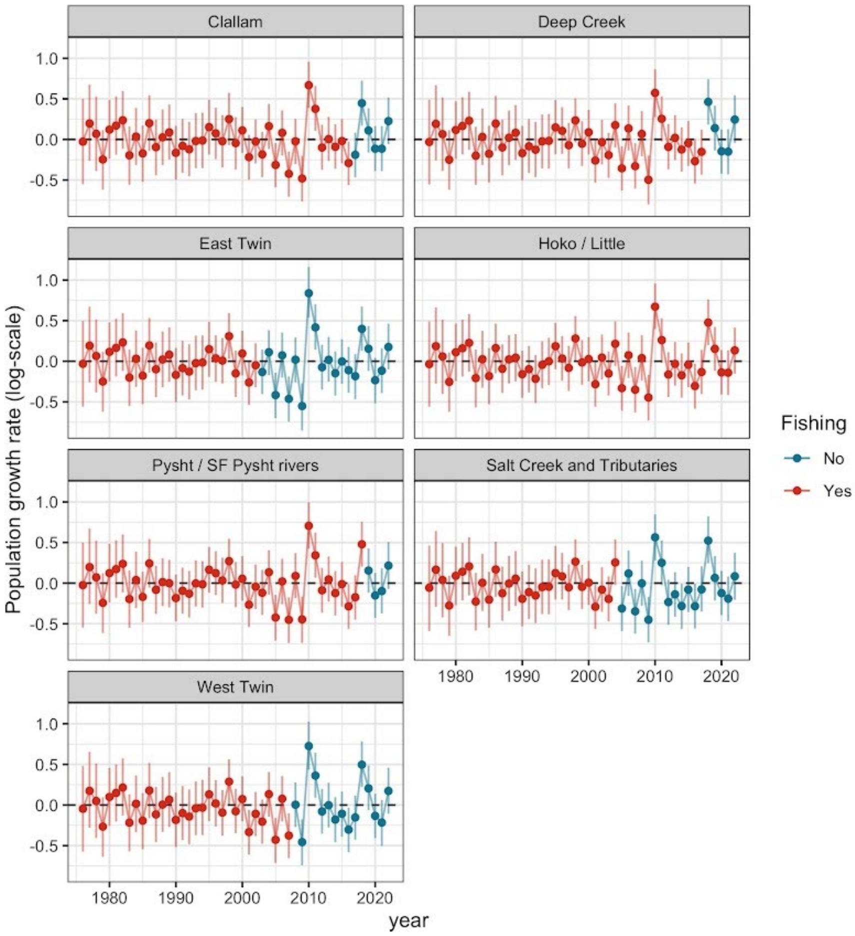


Figure 57. One-year estimates of population growth during period with and without harvest on strait populations of OP steelhead. Estimates are from the DLM output. Mean and 95% CI shown.

There are major caveats to the harvest and run size data presented above. First, we are missing hooking mortalities of natural-origin fish for both on-reservation recreational and other fisheries (except the Hoh River) in these estimates, except in the most recent years (since 2020/21). Alternatively, catch/mortality from fisheries for natural-origin steelhead is based on escapement (redd counts) after March 15th (assuming fish spawning after March 15th are all natural-origin), which may not be an accurate assumption in that natural-origin fish do occur before March 15th. This would lead to underestimates of total run sizes and an overestimate of harvest rates (see discussion in Life History-Traits about run-timing of natural-origin steelhead). For example, older data from 1979-1981 shows that in certain years, roughly on average 30-40% of redds for natural-origin steelhead occur prior to March 15th in certain tributaries of the Queets (figures show more or less for certain creeks in certain years) (Quinault Indian Nation 1981). Additionally, in the Calawah and Sol Duc rivers (Quillayute basin), peak redd abundance is in April/May; however, female natural-origin steelhead can begin spawning in January contributing on average 17% of natural-origin redds created before March 15$^{th}$ (McMillan, Katz and Pess 2007). Alternatively, Marston and Huff (2022) modeling work predicts that approximate 8.4% natural-origin spawning before March 15th (in Calawah and Bogachiel rivers). Since most harvest occurs in December, January, February (Figure 58) the fishery may be selecting against early run-timing of natural-origin steelhead. Also see Quileute and Hoh harvest management plans for 2023 that show predicted catch of natural-origin fish in January, February (Quileute plan provided by co-managers; Toby Harbison on June 22nd, 2023. Hoh plan provided by co-managers).

Catch of natural-origin fish prior to the March 15th date used by management, may also create higher fishing pressure on earlier-returning natural-origin steelhead. Specifically, see the analysis presented in the Status Review in the section *SRT assessment of winter-run run timing changes* that showed that for Hoh, Quileute, and also in Queets but to a lesser extent, that natural-origin fish are being caught disproportionately more than hatchery fish prior to March 15th and that pressure on natural-origin fish has been increasing in recent years while pressure on hatchery-origin has decreased. This analysis also showed that in mid-January, most fish caught are natural-origin and by February basically all fish caught at natural-origin. This could be an effect of poorer survival in hatchery-origin fish, since analysis in Harbison et al. (2022) shows that the survival of hatchery smolts is substantially less than that of natural origin smolt and, further that it has diminished in recent years. The greater proportion of natural-origin caught in January, February corroborates data in the Quileute - WDFW and Hoh- WDFW management plans for 2023 that also shows a high harvest of natural-origin steelhead during January and February.

For summer-run steelhead, catch-and-release regulations have been in place from WDFW and in the ONP since 1992, and there are no established escapement goals. Steelhead fisheries occur during times to target winter-run steelhead. At the same time, data shows some harvest (and/or catch and release mortality) of summer-run steelhead in recent years (Figure 58, Figure 59). It is difficult to interpret an impact of catch when summer-run abundance is unknown (see *Summer-run escapement data* section in the Status Review), but harvest of natural-origin summer-run steelhead has declined since the last NMFS review (see 1980s/early 1990s in Figure 59). Further, we did not have data on the indirect harvest of summer-run steelhead in fisheries targeting other Pacific salmon (this may be reflected in fish ticket information, although the Team did not have that data). In light of commercial gill-net fisheries and recreational fisheries, adult summer-run steelhead are susceptible[43] to bycatch during their upstream migration to spawn, prespawning holding, or as seaward migrating kelts. Given that summer-run population abundances are inherently smaller, this likely increases the potential risk for these populations.

A review by Myers (2018) noted that overall, incidental catch of steelhead in non-target commercial fisheries and illegal catch of steelhead in the marine and estuarine environments is considered low but also not well documented. This includes potential incidental catch in Japanese high sea drift net fisheries targeting other salmon and Asian high sea driftnet fisheries for flying squid. Myers (2018) notes from a study by Pella et al. (1993) that illegal catches in high sea driftnet fisheries in closed areas in the 1980s-1990s may have impacted steelhead in Kamchatka and North America.

[43] By catch rates depend on the specifics of the gear used, timing, and size/age of steelhead.

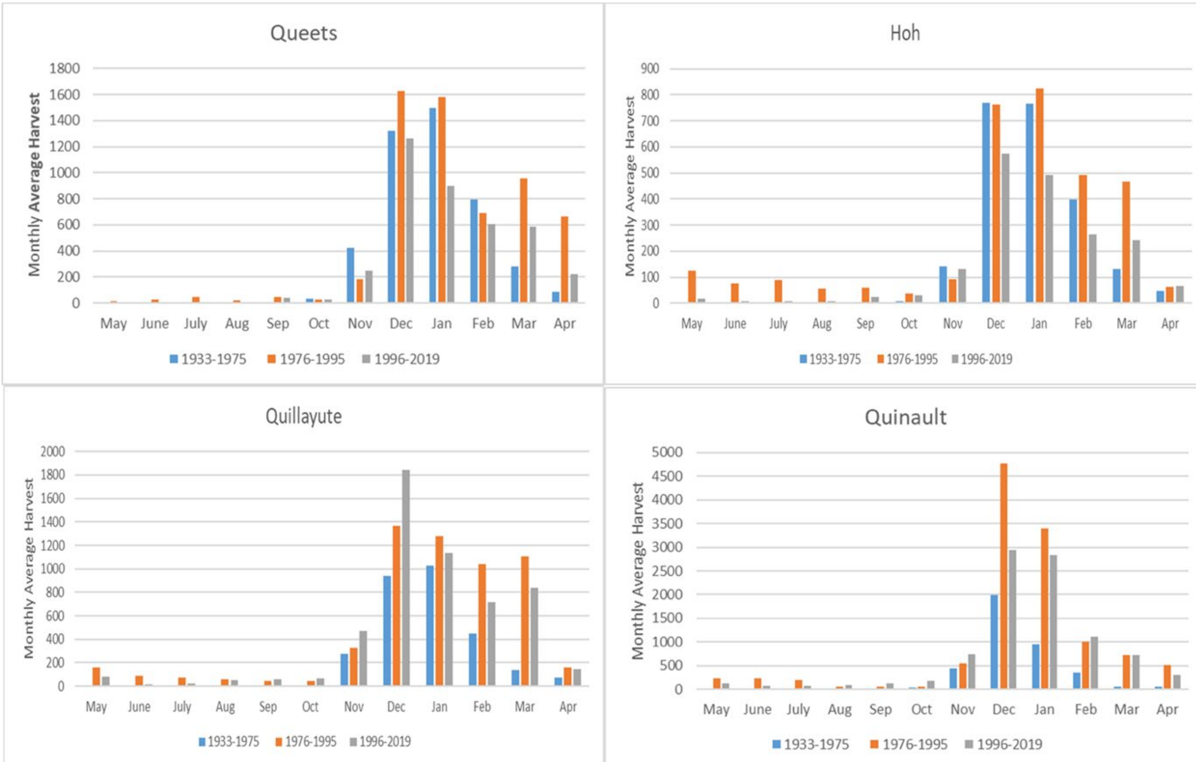


Figure 58. Tribal harvest by month for the four major systems where Nominal Winter Harvest occurs Nov-Apr and Summer Harvest from May-Oct (data provided by the petitioners; from: Rob Kirschner (The Conservation Angler) sent: June 7th, 2023 4:02 PM).

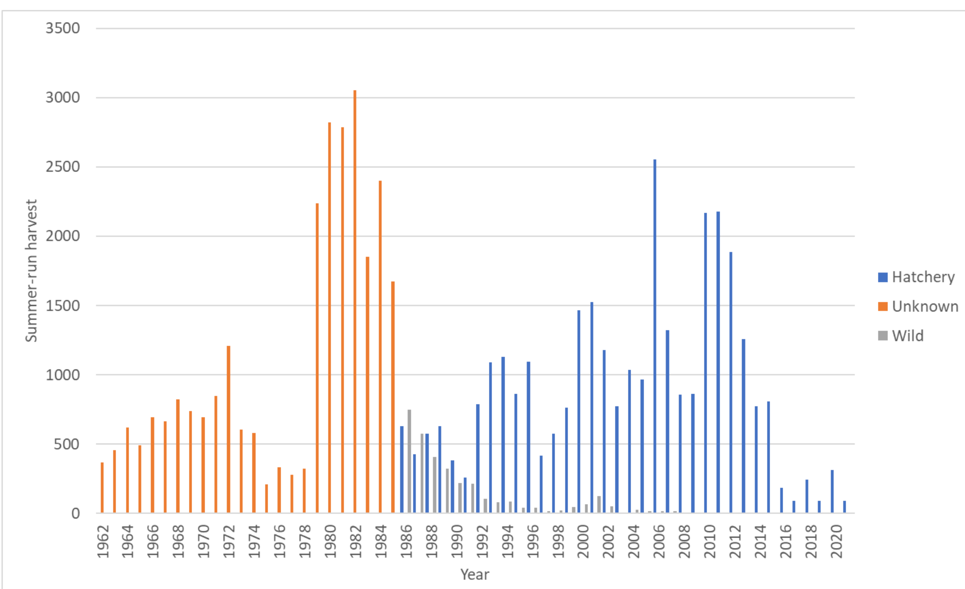


Figure 59. Summer-run harvest broken down by hatchery vs. wild. Prior to 1986 and the marking of hatchery-origin steelhead, hatchery and wild (natural-origin) steelhead harvest was combined. Data provided by the petitioners; from: Rob Kirschner (The Conservation Angler) sent: June 7th, 2023 4:02 PM).

# Listing Factor C: Disease and predation

## Disease

Infectious disease is one of many factors which can influence adult and juvenile survival. Steelhead are exposed to numerous bacterial, protozoan, viral, and parasitic organisms in spawning and rearing areas, hatcheries, migratory routes, and the marine environments. Specific diseases such as bacterial kidney disease (BKD), *ceratomyxosis*, *columnaris*, Furunculosis, infectious hematopoietic necrosis (IHNV), redmouth and black spot disease, Erythrocytic Inclusion Body Syndrome (EIBS), and whirling disease among others are present and are known to affect steelhead and salmon (Foott et al. 1994; Gould and Wedemeyer 1980; Leek 1987; Rucker, Earp and Ordal 1954; Wood and WDFW 1979). Very little current or historical information exists to quantify changes in infection levels and mortality rates attributable to these diseases for steelhead. However, studies have shown that native fish tend to be less susceptible to pathogens than hatchery-reared fish (Buchanan et al. 1983; Sanders et al. 1992). Wild steelhead may contract diseases which are spread through the water column (i.e., waterborne pathogens) (Buchanan et al. 1983). Disease may also be contracted through interbreeding with infected hatchery fish (Evelyn, Ketcheson and Prosperi-Porta 1984; Evelyn, Prosperi-Porta and Ketcheson 1986; Fryer and Sanders 1981). Also, a fish may be infected yet not be in a clinical disease state with reduced performance. Salmonids typically are infected with several pathogens during their life cycle. However, high infection titers (number of organisms per host) and stressful conditions (crowding in hatchery raceways, release from a hatchery into a riverine environment, high and low water temperatures, etc.) usually characterize the system prior to disease expression. At the time of the review by Naish et al. (2007), there were very few cases of direct infectious impacts of hatchery fish to wild stocks, but there are mechanisms by which this could occur. See National Marine Fisheries Service (1996a) for further review of disease cases in other systems.

Another critical factor in controlling disease epidemics is the presence of adequate water quantity and quality during late summer. As water quantity and quality diminishes, and freshwater habitat becomes more degraded, many previously infected salmonid populations may experience large mortalities with added stress triggering the onset of disease. These factors, in combination with high water temperatures common in various rivers and streams, may increase anadromous salmonid susceptibility and exposure to diseases (Holt et al. 1975; McCullough 1999; Wood and WDFW 1979). Furthermore, under most climate change scenarios summer flows will decrease and summer temperatures increasing the susceptibility of summer-rearing juveniles or summer-holding adults to epizootics (Northwest Indian Fisheries Commission 2020).

In the ocean, steelhead may be impacted by ecoparasites. Myers (2018) and studies summarized therein note that salmon lice were highly prevalent, and had mean intensity and abundance of infection in steelhead, pink salmon, and Chinook salmon compared to other Oncorhynchids but that low abundance of steelhead meant that steelhead only hosted a small percentage of the lice *L. salmonis* (Nagasawa 2001). In extreme cases, salmon lice may cause osmoregulatory failure (Nagasawa 1987 cited in Myers 2018; Wootten, Smith and Needham 1982), but natural ocean mortality of steelhead from lice is not known. Myers (2018) discusses that steelhead may have pathogens without any symptoms presenting and that steelhead are more resistant to salmon anemia virus than Atlantic salmon (citing Rolland and Winton 2003).

Some outbreaks of infectious hematopoietic necrosis virus (IHNV), reovirus, and Pacific salmon paramyxovirus have been documented in OP steelhead, mainly in hatchery-origin fish, though natural-origin fish are not generally sampled. Breyta et al. (2013) summarized previous outbreaks of the M genogroup (group of related viruses) of infectious hematopoietic necrosis virus (IHNV) in the Hoh, Queets, Quinault, and Quillayute river basins (as well as other coastal areas) between 2007 to 2011. M genogroup IHNV is particularly virulent for steelhead and rainbow trout, with high levels of mortality. Prior to 2007 there was only one detection in Washington coast steelhead, in the Queets watershed at the Salmon River Hatchery (in 1997). Most detections from 2007-2011 were in hatchery-origin fish, but Breyta et al. (2013) noted that natural-origin fish are less commonly sampled, and there were detections of this virus in natural-origin fish in the Hoh and Quinault river basins. No IHNV was detected in 2012, but the future risk of IHNV in OP steelhead is unknown given known fluctuations of IHNV incidences in other regions (like Columbia River basin) (Breyta et al. 2013). The effect of IHNV varied across various streams in Washington State and this variation was not fully explained by differences in virulence or hatchery water supplies (Breyta, Jones and Kurath 2014). For example, two separate hatchery populations that came from the same ancestral population had variation in mortality after exposure to a MD IHNV strain. Work by Brieuc et al. (2015) suggests that there is a genetic basis for resistance to IHNV and that populations have the ability to develop disease resistance, therefore reduction of genetic variation could impact future adaptation and resistance. Exposure may lead to selection of resistance to diseases, but adaptation and the rate that populations become resistant depends on the heritability of the trait (see Crozier et al. (2008)), and Brieuc et al. (2015) showed that resistance to IHNV is likely heritable. Sockeye salmon are frequently infected with IHNV (Dixon et al. 2016; Traxler et al. 1997) so where sockeye could interact with steelhead, particularly in hatcheries or in rivers like the Quinault or Ozette that support large sockeye runs, this could lead to further exposure to steelhead.

Similarly, we obtained data from Tony Capps (WDFW) on instances of disease, parasites, and viruses in steelhead hatcheries (state, federal, and tribal) on the Peninsula. There were four cases of reovirus in winter-run steelhead in December 2002, January 2003, December 2006, and February 2007, all in the Bogachiel system except the 2007 occurrence in the Sol Duc River. Years later in January 2020 there was another occurrence of reovirus in winter-run steelhead in the Bogachiel. There were eight instances of IHNV in winter-run steelhead in the Bogachiel Basin in winter 2009-2010, with six in December of 2009 and two in January 2010 (possibly the same as noted in Breyta et al. 2013). Finally, there were two instances of Pacific salmon paramyxovirus in Summer-run steelhead in Bogachiel River in summer 2017. Again, most of all known cases are in hatchery fish populations and limited information exists on the incidence of diseases in natural-origin steelhead in the OP. We note that to accurately assess the potential threat of disease in this population we would need annual pathology reports from each hatchery to effectively assess presence/prevalence of pathogens, viruses, bacteria (reports may exist but we were only provided instances of when a pathogen did occur for specific hatcheries)..

## Predation

Predation on salmonids can occur among other fishes, particularly during salmonid juvenile life stages, among avian predators, and among marine mammals, including Resident Killer Whales. Public comments on the 90-day finding included mention of predation by seals, sea lions, otters, eagles, killer whales, cormorants, and/or mergansers on steelhead, including anecdotal accounts of predation in the OP steelhead systems.

In general, predation on juvenile salmon has increased as a result of water development activities which have created ideal habitats for predators and non-native species. More specifically, anthropogenic habitat alterations like dams, irrigation diversions, man-made islands, amongst others have led to increased predation (Antolos et al. 2005; Evans et al. 2019; Hostetter et al. 2012; Moore et al. 2021). However, there are no large dams within the range of OP steelhead; therefore, OP steelhead are not being concentrated by these structures as other salmonids. Predation may significantly influence salmonid abundance in some local populations when other prey are absent and physical habitat conditions lead to the concentration of adult and juvenile salmonids in small areas (Cooper and Johnson 1992). Pearcy (1992) reviewed several studies of salmonids off of the Pacific Northwest coastline and concluded that salmonid survival was influenced by the factional responses of the predators to salmonids and alternative prey.

Invasions of non-native fish species pose threats to native fish fauna but little is known on the extent or effects on OP steelhead. The following nonnative fish species occur in waters of the OP steelhead DPS:  Eastern brook trout *(Salvelinus fontinalis*), Atlantic salmon (Salmo salar), Westslope cutthroat trout (*Oncorhynchus clarkii lewisi*), yellow perch (*Perca flavescens*), yellow bullhead (*Ictalurus natalis*), largemouth bass (*Micropterus salmoides*), American shad (*Alosa sapidissima),* and *Common carp* (*Cyprinus carpi*o), Non-native Brook Trout (*Salvelinus fontinalis*) were dentified as a competing species in the *State of Our Watersheds* report (Northwest Indian Fisheries Commission 2020).

Natural-origin (wild) steelhead likely have greater predator avoidance relative to hatchery-origin fish. Berejikian (1995) found that natural-origin derived fry from the Quinault River had significantly better predator avoidance from prickly sculpin (*Cottus asper*) than hatchery-derived fry, and natural-origin "experienced" fry (visually exposed to sculpin) were eaten less than naïve natural-origin and hatchery-origin. Hostetter et al. (2012) also found that hatchery-origin steelhead in the Snake River were more susceptible than natural-origin to avian predation, and predation was also influenced by steelhead condition and river and rearing conditions. Osterback et al. (2014) found that predation by western gull on steelhead was greatest in intermediate sized juvenile steelhead (compared to small or large) and though natural-origin steelhead had greater predation risk then hatchery, they also had greater survival.

In addition to predation by freshwater fish species, avian predators (gulls, mergansers, herons, diving birds like cormorants and alcids, including common murres and auklets as well as others) have also been shown to impact juvenile salmonids (National Marine Fisheries Service 1996a). More recently, Caspian terns and double-crested cormorants have been documented consuming outmigrating steelhead smolts in the Snake River basin (Hostetter et al. 2012), as well as gulls in the Columbia River (Evans et al. 2019). Avian predation on juvenile salmonids can occur as they enter the ocean as well (Tucker, Mark Hipfner and Trudel 2016; Zamon et al. 2014). Years of higher or lower availability of preferred prey may (inadvertently) increase predation on salmonids (Wells et al. 2017). With the decrease in riverine and estuarine habitat quality, increased predation by avian predators will occur. Salmonids and avian predators have co-existed for thousands of years, but with the decrease in avoidance habitat (e.g., deep pools and estuaries, large woody debris, and undercut banks), avian predation may play a role in the reduction of some localized steelhead stocks. However, Botkin et al. (1995) stressed that overall predation rates on steelhead should be considered a minor factor for their decline. We did not find information documenting an increase in predation by avian predators for OP steelhead since the last time this population was reviewed (1996), and though seabirds are present in the OP watersheds, we are unaware of any unusual or excessive predation events by seabirds or hotspots of seabird predation (based on pers. Comm. with Thomas Good, 15 October 2023, NMFS NWFSC).

The four main marine mammal predators of salmonids in the eastern 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*) (and see the summary in National Marine Fisheries Service (1996a)). Recent research suggests that predation pressure on salmon and steelhead from seals, sea lions, and killer whales has been increasing in the northeastern Pacific over the past few decades; specifically models estimate that consumption of Chinook salmon by marine mammals has increased from 5 to 31.5 million individual salmon since the 1970s (Chasco et al. 2017a; Chasco et al. 2017b, Couture et al. 2024), but this research was focused on Chinook salmon.  Couture et al. (2024) also discuss other salmonids, but there is limited mention of steelhead).  A recent review of pinniped predation in Puget Sound and the Washington Coast concluded that pinnipeds are responsible for reduced abundance of salmon in Washington State waters, but are not likely a primary cause for salmon not recovering in those ecosystems (WSAS 2022).

Some studies have found that pinnipeds like harbor seals can have a significant predation impact on coho salmon and other salmon species of conservation concern (Thomas et al. 2017), as well as steelhead (in Puget Sound; Moore et al. (2021) Moore and Berejikian (2022)) through the consumption of emigrating juveniles. Given that Moore et al. (2021) showed reduced steelhead smolt survival from Nisqually through Puget Sound out to the Pacific Ocean, and OP steelhead along the Strait of Juan de Fuca would migrate through a portion of this area as well, seals are likely impacting to some extent steelhead smolt survival. Moore et al. (2021) also showed that this is impact to smolt survival is higher in years with less anchovy (another harbor seal prey). Work synthesized in Pearson et al. (2015) suggest that marine mammal predators can detect "pings" emitted by acoustic tags and target tagged fish, thus creating a bias in the results. Additionally, harbor seal predation data specific to coastal tributaries is not currently available, so the extent to which predation by seals in rivers and estuaries is a threat to specific Oregon and Washington coastal salmon populations is currently unknown.

Hatchery releases of other salmonids may impact predation pressure on steelhead. A recent paper for steelhead in Puget Sound found a negative correlation between weekly steelhead survival and abundance of hatchery coho smolt releases (but not Chinook salmon smolts) (Malick, Moore and Berejikian 2022). The authors hypothesize that this correlation could be related to either competition between coho and steelhead smolts for prey or shared predators where there is a negative indirect effect to steelhead when predators feed on coho co-occurring with steelhead or predators switch from coho to steelhead. Malick, Moore and Berejikian (2022) voice that the second hypothesis related to shared predators is more likely given that Puget Sound steelhead quickly migrate through the sound and there would be limited time for direct competition (steelhead likely not rearing in Puget Sound).

Environmental and climate conditions may also impact predation risk. Low flow conditions in streams can increase mortality for salmonids (Henderson et al. 2019), with may be related to predation risk and/or predator avoidance ( and see discussion and references in Magoulick and Kobza 2003; Penaluna, Dunham and Andersen 2021). Increased turbidity can decrease predation risk for salmonids from fish piscivores (Gregory and Levings 1998). Warmer water temperatures due to water diversions, water development and habitat modification may affect steelhead mortality from predation directly or indirectly through stress and disease associated with wounds inflicted by pinnipeds or piscivorous predators. Similarly, future climate change (see Factor E) may also increase mortality due to predation. Alternatively, a recent study for Puget Sound steelhead showed that in warmer years (during the heat wave from 2014-2016), steelhead smolt survival probabilities increased, likely has a result of greater alternative prey (anchovy) in warm years for marine mammal predators (Moore et al. 2021).

The relative impacts of marine predation on anadromous salmonids are not well understood. However, it is evident that anadromous salmonids have historically coexisted with both marine and freshwater predators and based on catch data, some of the best catches of coho, Chinook, and steelhead along the West Coast of the United States occurred after marine mammals, kingfishers, and cormorants were fully protected by law (Cooper and Johnson 1992). Based on this, it would seem unlikely that in the absence of man's intervention, freshwater or marine predators would extirpate anadromous salmonids. It is likely that historical harvest of harbor seals and other marine mammals by Indigenous communities may have reduced predation on salmonids. Anthropogenic habitat alterations including dams, irrigation diversions, fish ladders, and man-made islands, have led to increased predation opportunities (Antolos et al 2005, Evans et al. 2012, Hostetter et al. 2015, Moore & Berejikian 2022). For OP steelhead, given there are no large dams or barriers, it seems unlikely that the level of predation would have increased from man-made barriers. There is the possibility that predation effects on steelhead has increased given the increase in pinniped populations, but we have no long term quantitative information on predation on OP steelhead. Also, predation on steelhead in the ocean is largely unknown.

# Listing Factor D: Inadequacy of regulatory mechanisms

Overall, one pending regulatory mechanism that would likely impact OP steelhead is any future implementation of the 2022 WDFW Coastal Steelhead Proviso Implementation Plan, which outlines state management strategies for the future of OP steelhead as well as other coastal steelhead populations. This was proposed to be partially funded by the Governor. Specifically, the Governor’s proposed budget (https://app.leg.wa.gov/billsummary?BillNumber=5950&Initiative=false&Year=2023) states, “$2,139,000 of the general fund-state appropriation for fiscal year 2025 is provided solely for expanded monitoring, evaluation, and management of coastal-river salmonid fisheries to inform decisions focused on the conservation and management of these resources," but was ot ultimately funded in the Governors's 2024 supplemental budget. The State is pursuing, but has not acquired, other funding that would being July 2025.

The Proviso is an application of existing state policies and is not a new policy. It was developed from the recognition of recent declines in coastal steelhead and therefore the need for adaptive management strategies. Additionally, WDFW notes in the Proviso that more region-specific Management Plans, including one for the OP steelhead DPS, have yet to be developed (but are planned). The Proviso provides an implementation strategy for addressing monitoring and evaluation, hatchery operations, fisheries, habitat, and human dimensions, but notes that the lack of crucial data is a limiting factor in management of these populations. Specifically, the Proviso Plan identified recreational fishery monitoring related to in-season management, summer-run steelhead monitoring and data collection (including genetic data), SONAR monitoring for more accurate escapement monitoring, marine survival research including estimating smolt/juvenile survival and abundance, and developing tools to link habitat restoration activities and fisheries management as important research needs.
In the absence any future implementation of the Proviso plan, summer-run steelhead monitoring remains largely unchanged since the time of Busby et al. (1996). Finally, it should be underscored that the Proviso plan is primarily focused on guidelines for management of recreational fisheries in State waters and does not include Tribal commercial or C&S component of harvest, as these are managed by Tribal partners (though the importance of these fisheries are recognized, see Harbison et al. (2022)).

## Habitat-related regulations

Regulatory mechanisms related to habitat protection and restoration may be inadequate as there continues to be habitat modification and legacy impacts of past habitat modification that are likely impacting OP steelhead. However, progress towards habitat protection is hard to measure as any ongoing efforts related to habitat restoration may take decades if not centuries to show an effect. Also, there are many existing regulations that help with the general protection of salmonid habitat (which we summarize below), but none specifically directly at steelhead.

Other existing regulations that may be impacting OP steelhead and are summarized below many of which were initiated after the last review of OP steelhead by NMFS (Busby et al. 1996) or were newly implemented at the time of the last review:

### Federal

The National Forest Management Act of 1976 establishes the development of land management plans by the U.S. Forest Service (USFS) for units of the National Forest System (https://www.fs.usda.gov/emc/nfma/includes/CFR-2018-Title36-Vol2-Part219.pdf).

Since 1994, the Northwest Forest Plan (NWFP) has guided the management of 17 federal forests along with Bureau of Land Management (BLM) lands in the U.S. Pacific Northwest. The aquatic conservation strategy contained in this plan includes elements such as designation of riparian management zones, activity-specific management standards, watershed assessment, watershed restoration, and identification of key watersheds. The NWFP was accompanied by a regional monitoring program and ongoing research. It is a large, multi-agency effort to conserve biodiversity, particularly old-growth forests, northern spotted owl (*Strix occidentalis caurina*), marbled murrelet (*Brachyramphus marmoratus*), and other species associated with older forests on federal lands in western Washington and Oregon, and northwestern California. It is also designed to protect and restore salmonid habitat, and to provide forest products to support local and regional economies. The NWFP was intended to be a 100-year plan and be flexible enough to adapt to new conditions, threats, and knowledge.

The most significant element of the NWFP for anadromous fish is its Aquatic Conservation Strategy (ACS), a regional scale aquatic ecosystem conservation strategy that includes the following: (1) special land allocations, such as key watersheds, riparian reserves, and late successional reserves, to provide aquatic habitat refugia; (2) special requirements for project planning and design in the form of standards and guidelines; and (3) new watershed analysis, watershed restoration, and monitoring processes. These ACS components collectively ensure that Federal land management actions achieve a set of nine ACS objectives, which include salmon habitat conservation.

Relative to forest practice rules and practices on many non-federal lands, the NWFP has large riparian management zones (1 to 2 site-potential tree heights) and relatively protective, activity-specific management standards. A retrospective on 25 years of the NWFP (Spies et al. 2019) reviewed the scientific literature published since the inception of the NWFP and reports several key findings. It has protected remaining old-growth forests from clearcutting and enabled growth and development of vegetation conditions to support threatened species, including salmonids and riparian-associated organisms (Spies et al. 2018). While the number of ESA-listed salmonid species and population units has increased, the pace of passive restoration, particularly in the face of climate perturbation, is insufficient to improve productivity at a rate necessary to achieve recovery. In addition, existing data are insufficient to determine whether basic survey and management criteria are met, and, management on federal lands alone without parallel efforts on non-federal land is not sufficient to achieve recovery (Reeves et al. 2018).

Over 990 square miles of the Olympic Peninsula are part of the Olympic National Forest (ONF) (Halofsky et al. 2011). Within the ONF, management is guided by the land and resource management plan (LRMP) which was amended by the NWFP. Therefore, with the LRMP and the associated establishment of the ACS, the ONF management activities should work to maintain and/or restore watersheds. Additionally, there is a forest strategic plan set for the ONF which helps to prioritize actions related to, “habitat restoration, road decommissioning, forest thinning, and fuel reduction treatments” (Halofsky et al. 2011), and integrating management related to wildlife, aquatics, fire, and silviculture (Halofsky et al. 2011).

According to Halofsky et al. (2011), the ONF is focused on:

> "Managing for native biodiversity and promoting the development of late-successional forests. Restoring and protecting aquatic ecosystems from the impacts of an aging road infrastructure. Managing for individual threatened and endangered species as defined by the Endangered Species Act (ESA) (ESA 1973) and related policies"

The Olympic National Park (ONP) is 1,442 square miles of land encompassing several different ecosystems, from the dramatic peaks of the Olympic Mountains to old-growth forests, beaches, riverine systems, and lakes. The National Park Service carries out its responsibilities in parks and programs under the authority of Federal laws, regulations, and Executive Orders, and in accord with policies established by the Director of the National Park Service and the Secretary of the Interior. The Park sets regulations for access and activities allowed within its boundaries, such as boating and fishing regulations. The National Park Management Policies 2006, has a stated policy for Improving Resource Conditions within the Parks, inclusive of biological resources, as well as responsibility for retaining parks "in their natural condition" – which is defined as the condition of resources that would occur in the absence of human dominance over the landscape.

The ONP created a General Management Plan in 2008 (National Park Service 2008). This plan set desired outcomes for the Park over the course of the 15-20 years and also established management zones within the ONP and goals for resource conditions within those zones (see summary in Halofsky et al. (2011)).

Multiple rivers and streams where OP steelhead occur have been designated as bull trout (*Salvelinus confluentus*) critical habitat (75 FR 63875-63978, October 18, 2010), which may indirectly benefit steelhead. Listed species like Lake Ozette sockeye salmon (*Oncorhynchus nerka*), bull trout, Northern spotted owl (*Strix occidentalis caurina*), and marbled murrelet (*Brachyramphus marmoratus*) occur on the peninsula, and the NMFS and USFWS have conducted biological opinions under section 7 of ESA for Federal actions in this region, including for the Forest Management Activities in the Olympic NF. Therefore, these consultations may help to mitigate federal actions in OP steelhead range that could destroy or adversely modify critical habitat of these other species, but does not prevent actions from potentially adversely affecting these habitats and are not specific to steelhead.

Many nation-wide regulations could have an impact on OP steelhead habitat but is difficult to pinpoint exact repercussions for OP steelhead specifically. The Federal Clean Water Act of 1973 addresses the development and implementation of water quality standards, the development of Total Maximum Daily Loads (TMDLs)[44] 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 Corps of Engineers, and some authority is delegated to the states.

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

[44] A TMDL is a pollution budget and includes a calculation of the maximum amount of a pollutant that can occur in a waterbody and allocates the necessary reductions to one or more pollutant sources. A TMDL serves as a planning tool and potential starting point for restoration or protection activities with the ultimate goal of attaining or maintaining water quality standards.

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 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. "NMFS believes altering and hardening stream banks, removing riparian vegetation, constricting channels and floodplains, and regulating flows [altering the natural hydrograph] are primary causes of anadromous fish declines (65 FR 42450 July 10, 2000)"; "Activities affecting this habitat include…wetland and floodplain alteration; (64 FR 50414 Sept. 16, 1999)."

Development proceeding in compliance with NFIP minimum standards ultimately results in impacts to floodplain connectivity, flood storage/inundation, hydrology, and to habitat forming processes. The development consequences of levees, stream bank armoring, stream channel alteration projects, and floodplain fill, combine to prevent streams from functioning properly and result in degraded habitat. Most communities (counties, towns, cities) in Washington and Oregon are NFIP participating communities, applying the NFIP minimum criteria. For this reason, it is important to note that, where it has been analyzed for effects on salmonids, floodplain development that occurs consistent with the NFIP's minimum standards has been found to jeopardize 18 listed species of salmon and steelhead (Chinook salmon, steelhead, chum salmon, coho salmon, sockeye salmon) (National Marine Fisheries Service 2008; National Marine Fisheries Service 2016). The Reasonable and Prudent Alternative provided in NMFS 2016 (Columbia Basin species, Oregon Coast coho salmon, Southern Oregon/Northern California Coast coho salmon) has not yet been implemented.

## State

The Forest Practices Act in Washington as well as the Washington State Forest Practices Rules (Title 222 WAC), establishes rules and guidelines for forest management on non-federal land in Washington State, to be "managed consistent with sound policies of natural resource protection" (RCW 76.09.010 https://apps.leg.wa.gov/RCW/default.aspx?cite=76.09). Washington State Department of Natural Resources states that these rules, "are designed to protect public resources such as water quality and fish habitat while maintaining a viable timber industry" (https://www.dnr.wa.gov/about/boards-and-councils/forest-practices-board/rules-and-guidelines/forest-practices-rules).
The statute (RCW 76.09) and the implementing rules and guidelines (WAC 222) govern forest practices on all private forest lands in Washington as well as all non-DNR state-owned forest lands irrespective of ESA listings. Additionally, these protections are monumented in NMFS's Habitat Conservation Plan (HCP) Biological Opinion (NMFS 2006).

In addition to protections on private and non-DNR state-owned forest lands, DNR's Habitat Conservation Plan (WADNR 2007) addresses compliance with the Federal ESA on state trust lands (NMFS 1997). The HCP covers approximately 1.9 million acres of DNR-owned forest lands within the range of the northern spotted owl (*Strix occidentalis caurina*), which includes all of the Olympic peninsula.  This plan allows for timber harvest and other forest management while complying with the ESA and minimizing and/or mitigating impacts to threatened and endangered species, under section 10 of ESA. This plan may help mitigate impacts to OP steelhead where there is overlap with other Federally listed species and their critical habitat. Furthermore, the Forest Practices Habitat Conservation Plan was established in 2006 and led NMFS and USFWS to issue a 50-year Incidental Take Permit for Washington State for Federally listing species, because of assurances from Washington state that implementation of forest practice and management would comply with ESA (https://www.dnr.wa.gov/programs-and-services/forest-practices/forest-practices-habitat-conservation-plan).

In January 2018, the Washington Legislature passed the Streamflow Restoration law (90.94 RCW) that helps restore stream flows to levels necessary to support robust, healthy, and sustainable salmon populations while providing water for homes in rural Washington. The State law requires that enough water is kept in streams and rivers to protect and preserve instream 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 instream flows, which are flow levels adopted into rule. Instream flows cover nearly half of the state's watersheds and the Columbia River. In Washington – and especially on the east side of the state -- out-of-stream uses, especially irrigation, exacerbate seasonally low flows, leading to passage and temperature problems, and the loss of habitat living space. Other water uses and land use (lack of recharge arising from impervious surfaces) also contribute to low streamflow levels. The Washington State Department of Ecology has a list of critical watersheds where instream flows are thought to be a contributing factor to "critical" or "depressed" fish status, as identified by the Washington Department of Fish and Wildlife.
Washington State has an anti-degradation standard in law (90.48 RCW) which is the basis for its regulations. These regulations include use-based criteria for existing and designated uses to set the Surface Water Quality Standards, (Washington Administrative Code (WAC) 173-201A). These use criteria include aquatic life criteria, and specifically name salmonid life history uses such as spawning, rearing, and migration. The EPA approved the Washington State's updated Water Quality Assessment 305(b) report and 303(d) list in 2012.

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

In 2015, the Washington state legislature created the Fish Passage Barrier Removal Board ((Revised Code of Washington (RCW) 77.95.160) to establish a new statewide strategy for fish barrier removal and administering grant funding available for that purpose. The legislation established several key objectives for the new strategy including:

- Coordination with all relevant state agencies and local governments to maximize state investments in removing fish barriers.
- Realizing economies of scale by bundling projects whenever possible.
- Streamlining the permitting process whenever possible without compromising public safety and accountability.

Chaired by WDFW, the board includes representatives of WSDOT, Washington Department of Natural Resources, Tribes, city and county governments, and the Governor's Salmon Recovery Office. In developing the statewide strategy, the board has been working closely with salmon recovery organizations to approve statewide guidelines. Highlights of the Board's work include:

- Approving two project pathways: 1) Watershed Pathway - Remove multiple barriers within a stream system. 2) Coordinated Project Pathway - Remove additional barriers upstream or downstream of a planned and funded project.
- Approving the initial focus areas for Watershed Pathway.
- Analyzing barriers submitted for Coordinated Project Pathway.

As of June, 2020, the Washington Department of Transportation has corrected more than 73 fish passage barriers in the injunction area and opened more than 329 miles of anadromous fish habitat, including ESA-listed salmon and steelhead habitat (WSDOT 2021). The other responsive state agencies have completed their known barrier corrections and all four agencies continue to monitor their roads to ensure that newly discovered barriers are quickly corrected

Updated in 2021, RCW 77.85 includes information for guiding the monitoring, protection, and recovery of salmonids as well as the Statewide Salmon Recovery Strategy (see summary in Harbison et al. (2022)). This led to the development of Lead entities for specific geographic areas that are tasked with identifying habitat projects, prioritizing projects, and exploring funding for projects. A Salmon Recovery Board approves projects submitted by the Lead entities and local organizations implement the projects. The Co-managers in their 2023 assessment of the petition (Co-Manager Olympic Peninsula Steelhead Working Group 2023) voice that "road maintenance and abandonment plans are now complete and positive progress on addressing culvert blockages is occurring…".

Cumulatively, many laws and regulations are in place to regulate freshwater habitat in Washington; however, it is difficult to assess how effective these are specifically to OP steelhead.

## Harvest regulations and monitoring

This section summarizes harvest regulations, noting that the discussion in listing Factor B describes aspects of harvest regulation, many of which we repeat here for continuity, but additional information can be found above.

The Washington Department of Fish and Wildlife (WDFW) cooperatively manages steelhead with Treaty Native American tribes and other parties and publishes yearly sport fishing regulations for steelhead (National Marine Fisheries Service (1996b)). For background on salmonid fisheries regulations in Washington state and based on the Pacific Salmon Treaty, see the summary in Duda et al. (2018) and/or Harbison et al. (2022). At the time of the 1996 NMFS steelhead review, wild steelhead could be harvested in Washington, but only if the wild run size was projected to have surplus escapement. Per existing court orders and through agreements between the State and Tribes (including U.S. vs. Washington, aka the Boldt decision - https://lib.law.uw.edu/c.php?g=1239321&p=9069754), harvestable surpluses of steelhead (wild and hatchery fish) were allocated approximately equally between treaty and non-treaty fishers. The WDFW defines adult steelhead as sea-run rainbow trout over 20 inches in length and since 1985, has marked all hatchery fish with an adipose clip to facilitate the identification and conservation of wild steelhead while allowing the harvest of hatchery steelhead. Most non-treaty sport fisheries for winter steelhead were directed at hatchery fish early in the season and many seasons were closed prior to the time most natural-origin fish enter the streams. In addition, freshwater recreational regulations (e.g., springtime stream closures and an 8-inch minimum size limit on all rivers statewide) were set to prevent anglers from targeting steelhead smolts. Wild steelhead release regulations (WSR), closed seasons, or area closures were implemented as appropriate to regulate the recreational fishery. As a general strategy in mixed hatchery-wild fisheries, WDFW would institute WSR if wild runs appeared to be under-escaped (or their status was unknown), and invoked area closures.

A summary document on Traditional Ecological Knowledge (TEK) provided by the Makah for this status review provides helpful context on management and biases of certain historic data (Martin 2023). The document from Makah notes that sustainable harvest management is a core principle of traditional resource management and embedded into Tribe societal roles, salmon and steelhead have been managed since time immemorial (including their habitat) and this management included both traditional hatchery and harvest practices. They also highlight that historical documents on harvest from the 1950s-1970s were prepared by non-Tribal entities and contain biases and limitations; not adequately representing historic conditions and biases in reporting of fish. They note that "historical data" may not be reliable. We mainly focus on data since 1996 but note this context for any consideration of more historical data or management information. Makah also highlight Tribal historical documentation that notes previous poor salmon returns due to climate conditions.

Sport harvest on all streams (except the Columbia River) is calculated from returns of permit cards that all persons fishing for steelhead in Washington are required by law to have. In addition, WDFW requests that anglers also keep records of all released steelhead. Information from steelhead permit cards provide WDFW with data valuable for assessing trends in sport catch.

Tribal steelhead harvest is gathered from several sources: state licensed game fish buyers return game fish receipt tickets to WDFW, on-reservation tribal enterprises report purchases of steelhead and steelhead taken for ceremonial/subsistence use, and reports of steelhead caught incidental to salmon fisheries and information gathered through enforcement programs.

The 2008 statewide steelhead management plan (Washington Department of Fish and Wildlife 2008) provided state management guidelines for the steelhead resource in Washington but recognized that individual regional plans were needed to include Tribes. The plan presented a framework to achieve the following goal for steelhead:

> "Restore and maintain the abundance, distribution, diversity, and long-term productivity of Washington's wild steelhead and their habitats to assure healthy stocks. In a manner consistent with this goal, the Department will seek to protect and restore steelhead to achieve cultural, economic, and ecosystem benefits for current and future residents of Washington State." (WDFW 2008 - https://wdfw.wa.gov/publications/00149)

To reach this goal, WDFW outlined implementation of policies related to natural production; habitat protection and restoration; fishery management; artificial production; regulatory compliance; monitoring, evaluation, and adaptive management; research; and outreach and education. More specifically, they noted prioritizing protection of wild steelhead, and protecting and/or restoring the quality, quantity, and productivity of both freshwater and marine habitat. Within fisheries management, they specified protection and restoration of the four criteria for a viable salmonid population, VSP (diversity, spatial structure, abundance, and productivity), while corporately managing resources with Tribes, and also providing diverse recreational opportunities. Within artificial production they noted striving for a net aggregate benefit to the 4 VSPs of wild stocks from artificial programs and enhancing harvest opportunities. More specific strategies and actions for these, including harvest, are detailed in the plan.

Olympic Peninsula rivers support a combination of sport fishing, as well as commercial, ceremonial, and subsistence gill-net fisheries for Pacific Salmon and steelhead. Summer and winter steelhead are collectively managed by WDFW and Treaty Tribes (in the Boldt Case Area) and the National Park Service in the Olympic National Park (ONP). WDFW has jurisdiction over recreational fisheries in Washington state waters and outside of the ONP boundaries. The Treaty Tribes regulate commercial, subsistence, and tribal-guided fisheries. ONP has exclusive federal jurisdiction to manage recreational fisheries within the park boundaries.

Currently, the OP steelhead fisheries are mainly managed for escapement goals for winter-run steelhead based on freshwater productivity (see Gibbons, Hahn and Johnson 1985). Goals are set based on maximum sustainable harvest, which became a priority after U.S. vs. Washington (Boldt decision - Tribes and state will co-manage fisheries and Tribes have the right to half the catch). More specifically, for the term "escapement goal," Harbison et al. (2022) states for WDFW that "In this instance, it refers to the approximate number of fish needed to escape from fisheries to provide enough spawners to perpetuate the run for future generations at maximum sustainable yield (MSY)." Before the Boldt decision, harvest was managed to ensure sufficient returns to the hatcheries for production purposes without regard to returning natural origin fish; WDFW notes that "managers assumed that enough wild fish made it past the fishery to spawn," or in some cases redd counts or abundance counts at dams were used for monitoring and management (see Harbison et al. 2022). Given the lack of data on spawners and recruits for specific watersheds, Gibbons, Hahn and Johnson (1985) developed a Potential Parr Production model to estimate the number of steelhead offspring possible based on habitat, and used this within a modified Beverton-Holt model to determine escapement goals at MSY. Further, while Gibbons et al. is the basis for escapement goals there is some disagreement among co-managers on the escapement goals for some basins (see Table 4 in the Status Review report). Specifically, a separate escapement goal for the Queets River was calculated based on the number of spawners needed for maximum sustainable yield ($S_{msy}$) both in the 1980s and again in the 1990s (based on a Ricker curve) and it the escapement goal used by Quinault (Scott, J.B. OP steelhead follow-up questions. Email to Laura Koehn. 17 July 2024). WDFW has yet to reevaluate these escapement goals and the assumptions from Gibbons et al. upon which they are based. WDFW has stated their intention to recalculate escapement goals based individual population models within a management strategy evaluation framework (Harbison et al. 2022).

With the escapement goals and foundation of Boldt, each year the State and the Tribes agree to yearly management plans that detail harvest of natural-origin and hatchery-origin OP steelhead for the upcoming fishing season. These plans consider forecasted returns and escapement goals to set harvest rates. In certain years and depending on the system, escapement goals are not met (see Factor B above). This may be due to errors in projected returns. The co-managers did state in their 2023 review to the SRT that, "Tribal fisheries are generally shaped by time and area restrictions with in-season management based on monitoring of fishery catches," so there is monitoring of certain catch (Co-Manager Olympic Peninsula Steelhead Working Group 2023), and seasons have been shortened/closed early in recent years in response to monitored catches (see Listing Factor B). Additionally, differing escapement goals (e.g. Queets River) may lead to harvest rates that result in adult returns below the escapement goal, depending on if the State or Tribal escapement goal is considered. Therefore, in certain years and certain systems, projected abundance may be below a certain escapement goal but harvest still occurs, and therefore harvest may not be at MSY and escapement levels may not be at the level to maximize future returns. Note that the info on meeting escapement goals we have is for the major four systems and we do not present information on meeting escapement for rivers along the Strait of Juan de Fuca. For more on harvest that has occurred see Factor B presented above and section *Harvest Rates* above.

Escapement goals and MSY management are not directly related to extinction risk, but not meeting escapement goals suggests that harvest management has inherent impression that can result in effects to populations' overall productivity and represent a potential risk to the DPS. In the face of a declining run size, it is unclear if current management goals and strategies will allow for maintenance or restoration of the runs.

For winter-run steelhead returning to the Olympic Peninsula, in 2016, WDFW changed the recreational fishing regulations to prohibit retention of natural-origin winter-run steelhead in OP steelhead river basins. Sport and Tribal catch of winter-run population has typically occurred from November-April. The number of natural-origin OP steelhead that are captured and released is calculated by WDFW via creel surveys, and it is estimated that catch and release has a 10 percent mortality rate. However, research by Bentley (2017) suggests that angler effort is underestimated (https://wdfw.wa.gov/sites/default/files/publications/01918/wdfw01918.pdf) and this work also suggests that some fish are caught and released more than once. Hooking mortality is assumed to be 10% by WDFW but is not included in estimates of harvest mortality presented in Listing Factor B.

In 2004, Olympic National Park implemented catch-and-release regulations for wild steelhead throughout coastal rivers of the DPS within the park. Steelhead fisheries in Olympic National Park allow for the retention of 2 hatchery-origin fish, but prohibit the retention of natural-origin steelhead (since 2016). A National Park fishing license is required to fish within the Park, rather than a Washington state recreational license, although a Washington state record card is required (see https://www.nps.gov/olym/upload/OLYM_Fish_Brochure_2022-0502-508_all_CHARTs_REMOVED.pdf).

Additional strategies since the 1990s have been employed to support sustainable fishing including harvest restrictions (such as bag limits), shorter seasons, and gear restrictions in the face of declining wild steelhead populations (Harbison et al. 2022), including those listed above. In recent years, WDFW has shortened or closed the recreational fishing season on winter-run OP steelhead, at least in part due to low returns. WDFW also imposed restrictions on recreational angling by banning the use of boats ("no fishing from a floating device") and bait (see links provided in Factor B). In 2022-2023 sport fishing was closed on the Quinault and Queets for December 1st- April 30th because of low returns and because a natural-origin steelhead harvest level was not agreed to across co-managers (see links provided in Factor B). Alternatively, tribal fisheries regulations still allow for the retention of natural-origin steelhead (commercial and ceremonial and subsistence harvest) but the total number of weeks of Tribal fisheries has declined in recent years (see Listing Factor B) specifically on the Queets and Quinault, and as mentioned before, harvest rates have also declined.

In the response to the petition to list OP steelhead, the Co-managers (Co-Manager Olympic Peninsula Steelhead Working Group 2023), explain that they develop abundance forecasts each year and develop fishery plans to meet management objectives under U.S. v. Washington. This includes that Tribal fisheries are shaped by time/area restrictions based on the monitoring of fishery catches within season. Recreational fishery management varies across locations and year but can include bag limits, non-retention of natural-origin steelhead, seasonal closures, gear and access limitations. The Co-Managers provided examples of yearly regulations as illustration of regulation responsive to OP steelhead abundance. This included that in specific past years the release by recreational fishers of unclipped fish (winter-run or summer-run) was required in State waters, including in 1997-1998 for summer-run steelhead due to ongoing concerns regarding status of summer steelhead. In certain other years, retention of unclipped winter-run steelhead was limited (example 1 fish per day, specific months, etc), before the non-retention of unclipped[45] steelhead regulations for recreational fishing was put into effect throughout the DPS in 2016. Currently, recreational fisheries within tribal lands on the Queets and Quinault do not prohibit the retention of natural-origin steelhead.

WDFW has proposed in their recent 2022 Coastal Steelhead Proviso Implementation Plan (Harbison et al. 2022) as part of their Proviso Implementation Strategy, a 3-step process for setting fishery regulations for state fisheries in Pacific coast Washington river systems. Specifically, (1) forecasting wild and hatchery-origin run sizes, (2) pre-season planning of regulations to meet management objectives, and (3) in-season update tools to assess if based on updated information if there can be increased opportunity, additional restrictions, or fishery closure. Additionally, fishery regulations depend on their Adaptive Management Framework which considers if rivers are in a "Maintenance" regime where the full spectrum of recreational fishing possibilities are explored, "Transitional" regime where hatchery-targeted fisheries or fishery limitations or closures are utilized, or "Emergency" regime where recreational steelhead fishing is closed. Finally, WDFW is continuing to pursue actions related to including steelhead impacts from other fisheries (Chinook and coho salmon) in management calculations, evaluating permanent fisheries regulations, and looking into tailoring fishery regulations on other species (i.e. Smallmouth Bass) to reduce predation on steelhead.

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[45] Except in the Queets and Quinault rivers where dorsal fin height is used to segregate hatchery-origin from natural-origin steelhead.

The following information for harvest management from for specific rivers/watersheds within the Olympic Peninsula are summarized from Harbison et al. (2022) and specific regulations are listed in Appendix 12.4 in that plan. The following descriptions are specifically for winter-run steelhead. More specifics on harvest for populations along the Strait of Juan de Fuca are not covered in similar detail as the large rivers in the OP, but as noted in the status review (see section *Population Growth and Harvest in Strait Populations*), especially Table 12, Figure 32) most rivers along the Strait, fishing hasn't occurred in recent years (but see the Hoko River).

**Quinault:** The Quinault River steelhead management is divided into areas either above and below Lake Quinault. Above the lake, but below the ONP boundary ("upper Quinault"), recreational fishing is managed by WDFW, in the ONP it is managed by NPS, and below the lake ("Lower Quinault"), the Quinault Indian Nation manages a tribal gill net fishery and recreational fishing. The entirety of the Quinault system falls within the usual and accustomed fishing grounds of the Quinault Indian Nation. The escapement goal set by WDFW for upper Quinault is 1,600 steelhead. Motorized boats are currently not permitted on Lake Quinault nor the upper Quinault River. Hatchery fish are not adipose fin clipped in the Quinault Basin, making retention targeting hatchery fish difficult. Currently, state regulations allow for retention of steelhead with a dorsal fin of less than 2 1/8 inches, the height of a credit card so named the "credit card rule", because hatchery fish are assumed to have eroded dorsal fins. This system for identification is inexact, and misidentification likely occurs. Other regulations related to prohibiting bait, limits on hooks, size limits etc. are listed in Appendix 12.4 of Harbison et al. (2022). Recreational fisheries on tribal lands do not prohibit the retention of natural-origin steelhead. Monitoring of this system is currently solely spawning ground surveys by the Quinault Indian Nation and ONP.

**Queets/Clearwater:** Most of the Queets River flows through ONP and therefore, sport fisheries are managed by the NPS within the park boundaries. Only the lower four river miles exists out of the park with fisheries in this portion co-managed by WDFW and the Quinault Indian Nation. WDFW manages sport fishing in the Clearwater River. The entirety of the Queets system falls within the usual and accustomed fishing grounds of the Quinault Indian Nation. WDFW's natural origin escapement goal is 4,200 fish. Similar to the Quinault artificial propagation program, hatchery fish are not adipose fin clipped in this river system. So, as in the Quinault River currently, regulations allow for retention of steelhead with a dorsal fin of less than 2 1/8 inches, the height of a credit card so named the "credit card rule", because hatchery fish are assumed to have eroded dorsal fins. But again, there is uncertainty if this rule can achieve management objectives based on variations in dorsal fin lengths. Other regulations related to prohibiting bait, limits on hooks, size limits etc. are listed in Appendix 12.4 of Harbison et al. (2022). Recreational fisheries on tribal lands do not prohibit the retention of natural-origin steelhead. Monitoring consists of solely spawning ground surveys with 90% covered by the Quinault Indian Nation and 10% covered by WDFW.

**Hoh:** 58% of this watershed is contained within the ONP. The whole watershed is within the usual and accustomed fishing groups of the Hoh Tribe. The escapement goal agreed to by co-managers is 2,400 fish. Due to emergency regulations in recent years for recreational fishing, floating devices have not been allowed on the Hoh River, for a portion of the river in 2016 that extended to the whole river in 2020/2021. Other regulations related to prohibiting bait, limits on hooks, size limits etc. are listed in Appendix 12.4 of Harbison et al. (2022). Spawning ground surveys are conducted by the Hoh Tribe, ONP, and WDFW. The Hoh River is also monitored through creel surveys but often suffers from limitations in resources leading to the inability to estimate total steelhead encounters and angling effort. Annual harvest management plans for winter steelhead are prepared by WDFW and the Hoh Tribe including the most recent in 2022-2023 (Hoh Tribe and WDFW 2022-2023 management plan provided by co-managers). For the Tribal fishery, days fished/week are set and have been limited to 15-18 days in recent years (pers. comm Brian Hoffman, Hoh Tribe, Natural Resources, June 21, 2023). Ceremonial-and-Subsistence fisheries can be conducted at other times and included harvest rates of 10 natural-origin and 30 hatchery fish in 2022-2023 (Hoh Tribe and WDFW 2022-2023 management plan). Evidence of depressed run size or exceedance of harvest, can trigger discussions among the co-managers. Commercial catch, harvest returns, and winter steelhead sport catch are monitored and each party (Hoh Tribe, WDFW) enforces its own regulations.

**Quillayute River:** This includes the Quillayute mainstem and four major tributaries: Calawah, Sol Duc, Bogachiel, and Dickey rivers. A portion of the watershed falls within the ONP and fisheries are managed by the NPS there, and the whole system falls in usual and accustomed fishing areas of the Quileute Indian Nation and a portion falls within usual and accustomed fishing areas of the Hoh Tribe. The co-managers agreed that the escapement level for wild steelhead in the Quillayute system is 5,900 fish. Monitoring is conducted by WDFW, the Quileute Tribe, and ONP and includes on-the-ground surveys as well as aerial surveys, where ground to air conversion factors are used and surveyed areas are extrapolated to unsurveyed reaches for escapement totals. In recent years landslides have limited surveys. Also, some creel surveying has occurred but not comprehensively. Scale collection by the Quileute Tribe also helps to estimate run timing and age class. Other regulations related to prohibiting bait, limits on hooks, size limits etc. are listed in Appendix 12.4 of Harbison et al. (2022). WDFW and Quileute Tribe also prepare an annual harvest management agreement for winter-run steelhead in the Quillayute with predicted returns (for example “Annual Agreement for the 2022-23 Harvest Management of Winter Steelhead in the Quillayute River System”). Quileute Tribe fishing is conducted based on a fixed schedule set forth in the annual management plan. These annual plans also outline agreements for enforcement and evaluation of causes of mortality.

**Independent streams:** This includes Tsoo-Yess-Waatch, Ozette River, Goodman Creek, Mosquito Creek, Kalaloch Creek, Moclips River, Copalis River and others. Spawning ground surveys have only been consistently done on Goodman Creek by WDFW. There have also been sporadic spawning ground surveys conducted by WDFW and Tribes on Mosquito Creek, Kalaloch Creek, Cedar Creek, Raft River, and the Moclips River, as well as others.

We note that most estimates of population size, used to inform harvest management, are based on redd counts/surveys that occur after March 15th. Though the majority of natural-origin fish likely spawn after March 15th, a proportion does spawn before this date and therefore are missed in counts and estimates. We discuss the likely effects of the March 15$^{th}$ cut-off date above in Listing Factor B and extensively in the status review (particularly, see *Abundance, Winter-run Steelhead*). Finally, many public comments during the 90-day finding comment period voiced concerns about the inadequacy of state monitoring of OP steelhead especially in relation to being able to make an accurate assessment of the current status of the population. Many of these mentioned seeing redds not counted earlier in the season and voiced support for redd surveys from December-May to count earlier returning natural-origin steelhead. Others noted that they are often not creel checked by management.

### Summer-run steelhead

As mentioned above, Harbison et al. (2022) specifies critical research needs including summer-run steelhead monitoring and data collection, a conclusion voiced earlier by Busby et al. (1996). Similarly, Cram et al. (2018) noted that there was insufficient data for all summer-run populations to assess trends or extinction risk. In 1992, WDFW and ONP implemented catch-and-release-only fishing regulations for summer steelhead; although there is still mortality associated with non-retention fisheries (i.e. hook mortality). Specifically, the onset of the WDFW ruling to protect and release wild summer steelhead occurred in the April 16, 1992-1993 fishing pamphlet (page 22 of 40) and included releasing wild steelhead from June 1-November 30 throughout all rivers in Region 6, which includes the Olympic Peninsula. There are no directed commercial fisheries for summer steelhead in the DPS. The Treaty tribes develop annual regulations for sport fishing on-reservations and those regulations include daily limits for steelhead that are caught during summer months. Time-series of estimates of harvest of summer steelhead are provided in Listing factor B and in the Status review (section *Summer-run steelhead population harvest*). Also, there are no established management goals between Washington State and Treaty Tribes for summer-run steelhead. Therefore, though fisheries management appears responsive to winter-run steelhead, there is no formal management of summer-run (outside of ONP) and no definitive monitoring plans making the adequacy of existing management for summer-run uncertain. And again, this was the case at the time of the last OP steelhead review by NMFS.

### Pacific Fishery Management Council Harvest Management

Salmon fisheries in the exclusive economic zone (three to 200 nautical miles offshore) of Washington, Oregon, and California have been managed under salmon Fishery Management Plans (FMPs) of the Pacific Fishery Management Council (PFMC) since 1977. While all species of salmon fall under the jurisdiction of the current plan (Pacific Fishery Management Council 2022), the FMP currently contains fishery management objectives only for Chinook salmon, coho, pink salmon (odd-numbered years only), and any salmon species listed under the ESA measurably impacted by PFMC fisheries. The FMP contains no fishery management objectives for sockeye salmon (*O. nerka*), even-numbered year pink salmon, chum salmon (*O. keta*), steelhead (*O. mykiss*), sea-run cutthroat (*O. clarki*), or spring run Chinook salmon from mid-Columbia River, states that the Council does not manage fisheries for these species, and also states that incidental catches of these are inconsequential (low hundreds of fish annually) to rare (citing PFMC and NMFS 2011). There is also a prohibition on take or retention of steelhead by any persons other than Indians with judicially-declared rights and licensed recreational fishermen, within the EEZ. (https://www.pcouncil.org/documents/2022/12/pacific-coast-salmon-fmp.pdf/)

## Hatchery regulations

Here we describe overall hatchery regulations and then in listing Factor E, we discussed potential negative impacts of hatchery production on natural-origin OP steelhead. WDFW operations of hatcheries is currently regulated by the Statewide Steelhead Management Plan (SSMP) and the Anadromous Salmon and Steelhead Hatchery Policy C-3624 (Fish and Wildlife Commission 2021), superseding the policy from 2009 (Hatchery and Fishery Reform Policy C-3619). However, the state and Tribal co-managers are currently working to develop Hatchery Management Plans (Harbison et al. 2022). Furthermore, the state Coastal Steelhead Proviso Implementation Plan (Harbison et al. 2022) aligns with the existing policies, and hatcheries on the West Coast are focused on the primary goal of harvest.

Co-Manager Olympic Peninsula Steelhead Working Group (2023) review of the petition noted that after a 2008 assessment of gene flow (Scott and Gill 2008) certain segregated hatchery programs were discontinued as part of the 2008 SSMP. Specifically, Scott and Gill (2008) showed gene flow of early Winter Chambers creek stock into Hoko, Pysht, and Sol Duc, (5.5-14.5%, 12-75%, and 2.5-6% gene flow respectively). Based on this and other information, the SSMP included the action "Where risks are inconsistent with watershed goals, implement one or more of the following actions:…eliminate the segregated hatchery program." (Co-Manager Olympic Peninsula Steelhead Working Group 2023). Therefore, winter steelhead smolt release into Pysht was eliminated in 2009, similarly in Goodman Creek, Clallam River, and Lyre River in 2009, and in 2012 the Sol Duc River was designated by WDFW as a Wild Stock Gene Bank, terminating summer smolt releases in 2011 and winter in 2013 (winter-run was local-origin broodstock steelhead). Local-origin broodstock releases occurred in Calawah and Bogachiel until 2021 when the program was terminated (Co-Manager Olympic Peninsula Steelhead Working Group 2023).

The 2009 Hatchery and Fishery Reform policy was evaluated by Murdoch and Marston (2020) (https://wdfw.wa.gov/publications/02133), but they determined that there was not enough data to evaluate the effectiveness at meeting management goals, including supporting fisheries, of the 159 hatcheries. However, they did look at the effectiveness of policy implementation. The review's conclusions identified several concerns including: a lack of harvest program goals, lack of a comprehensive monitoring and evaluation program, lack of program success definitions, and lack of data analysis for further adaptive management. On the other hand, regulations that were found to be well implemented included hatchery fish external marking (State Hatcheries only), Chinook smolt survival, and compliance of facilities with environmental regulations.

The 2009 plan was superseded by the 2021 Anadromous Salmon and Steelhead Hatchery Policy C-3624 (https://wdfw.wa.gov/about/commission/policies/anadromous-salmon-and-steelhead-hatchery-policy), which outlines guidelines for operations at WDFW-run hatcheries for salmon and steelhead. Specifically, the policy says, " The purpose of the Anadromous Salmon and Steelhead Hatchery Policy (Policy) is to guide hatcheries and their individual rearing programs to advance the conservation and recovery of wild salmon and steelhead by implementing hatchery reform measures; to perpetuate salmon and steelhead in accordance with existing mitigation programs and agreements for permanently lost or impaired habitat; and to provide sustainable economic and stability benefits to recreational, commercial and tribal fisheries in Washington State as appropriate." And "The intent of this Policy is to provide 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 in a manner that optimally achieves the stated purpose of this Policy.” Furthermore, this policy will be superseded when joint policies with Tribal co-managers are completed.

The C-3624 Hatchery Policy lists 10 policy guidelines for managing state hatcheries. Specifically, (1) minimizing genetic risks to wild salmon and steelhead via provisions in Hatchery Management Plans (HMPs), (2) minimizing ecological risks to wild fish through provisions in HMPs, (3) provide benefits, such as boosting recovery of wild populations, maintaining genetic traits, supporting fisheries, supporting at-risk predators, and benefits should be provided based on provisions in HMPs, (4) an HMP will be developed for every hatchery program under this Policy, (5) levels of hatchery production are based on deliberative, transparent, science-based process, (6) hatchery production for Southern Resident killer whale recovery is top priority, (7) all Chinook and coho salmon and steelhead that are hatchery produced will be marked (**with exceptions**), (8) the department shall strive to secure the funding needed for these hatcheries, (9) high protection to wild populations that have had limited negative impacts of hatcheries, and (10) WDFW will plan for and implement technologies for separating wild and hatchery salmonids such as weirs and other emerging technologies.

This policy provides general guidelines and points repeatedly to future HMPs for specifics. We did not find any evidence of completed HMPs at this point. This policy applies to State hatcheries and not to Federal or Tribal facilities. The C-3624 policy is only that, a policy, and not regulation.

Harbison et al. (2022), following the policies listed above, states that WDFW will consider ecological impacts, the ability for angling opportunities, and mitigation agreements when designing hatchery operations. Hatcheries will be designed based on the criteria outlined in the CSPIP depending on the status of the river/system, specifically “maintenance” regime, “transitional”, and “emergency” (i.e. different requirements for hatchery operations or different responses depending on the status of the system). This includes that in an emergency regime, hatchery programs may be discontinued if fisheries frequently need to close due to low natural-run steelhead returns.

Furthermore, within the CSPIP, WDFW notes that they will pursue additional actions as well when designing or updating hatcheries. Specifically: (1) developing adult production goals; (2) minimizing ecological impacts through: (a) techniques for reducing residual juveniles, (b) volitional hatchery releases and transporting non-migratory smolts, (c) field sampling of juvenile dispersion, residual rates, and competition, (d) using predation competition disease risk models, and/or (e) collecting genetic data for specific rivers; (3) relocating surplus smolts or adults; (4) optimization of trapping and hatchery attraction; (5) reducing spatial and temporal overlap between hatchery and natural-origin through release locations; (6) prioritizing hatchery research. WDFW will be conducting modeling to look into how many smolts could be released while staying within the genetic thresholds. Finally, the state will also be looking into potential designations of Wild Stock Gene Banks as the SSMP states that there will be at least one gene bank for each Major Population Group but currently only the Sol Duc River has been identified as a steelhead gene bank (2012). Stock use/placement of the gene bank will follow guidelines/criteria set by the SSMP: each stock used must be sufficiently abundant and self-sustaining, no releases of hatchery steelhead in rivers used by the stocks, and fisheries may occur on stocks if management objectives are met. WDFW also expanded criteria and considerations for gene banks in their CSPIP including: populations must have stable trends and over 300 spawners on average over 6 years, populations may not be where on-station hatchery releases occur but could be where off-station releases occur, consideration of usefulness of populations for research, and considerations of designating other populations in that overlap (Harbison et al. 2022).

Though the CSPIP (Harbison et al. 2022) outlines overall plans for state hatcheries (e.g. only the Bogachiel Hatchery in the OP DPS), co-manager Hatchery Management Plans are still being developed and it is unclear how much of the CSPIP is currently being implemented. We outline current potential impacts of hatcheries below (in Listing Factor E), noting: (1) the use of out-of-DPS origin broodstock, (2) not all hatchery fish are adipose fin clipped, and (3) possible current levels of proportion of hatchery-origin adults spawning (pHOS) with natural origin steelhead that are above desired levels.

# Listing Factor E: Other natural or manmade factors

## 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 2018 were the warmest years on record both on land and in the ocean (2018 was the fourth warmest). Events such as the 2013-2016 marine heatwave (Jacox et al. 2018), have been attributed directly to anthropogenic warming (Herring et al. 2018). 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 5-year status reviews for listed species of salmonids including steelhead 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 National Marine Fisheries Service 2022b).

Crozier et al. (2019), conducted a climate vulnerability assessment that included all anadromous Pacific salmon and steelhead (*Oncorhynchus* spp.) population units listed under the federal ESA. Using an expert-based scoring system, they ranked 20 attributes for the 28 listed units and 5 additional units. Attributes captured biological sensitivity, or the strength of linkages between each listing unit and the present climate; climate exposure, or the magnitude of projected change in local environmental conditions; and adaptive capacity, or the ability of salmon to adjust to cope with new climatic conditions via genetic adaptation or phenotypic plasticity (Crozier et al. 2019; Pachauri et al. 2014). Among species, Chinook salmon had the highest vulnerability rankings overall (mostly very high and high rankings), followed by coho and sockeye. Steelhead and chum DPS scores were generally lower and nearly equally spread across high and moderate vulnerability categories.

Climate change is projected to alter habitat conditions in freshwater, estuarine, and ocean environments. Siegel and Crozier (2020) provide the following observations: as stream temperatures increase, many native salmonids face increased competition with more warm-water tolerant invasive species. Changes in flow regimes may alter the amount of habitat available for spawning. This could lead to a restriction in the distribution of juveniles, further decreasing productivity through density dependence. Along with warming stream temperatures and concerns about sufficient groundwater to recharge streams, another recent study projects nearly complete loss of existing tidal wetlands along the U.S. West Coast, due to sea-level rise (Thorne et al. 2018). Tidal wetlands in California and Oregon are most threatened (expected loss of 100%), while 68 percent of Washington tidal wetlands are 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. Finally, climate change is expected to have profound influences 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).

Increasing stream temperatures can affect salmon and steelhead at multiple life stages depending on species (Hicks 2002), including reproduction (egg viability) (Berman 1990), incubation survival (eggs in the gravel), juvenile rearing (Bear, McMahon and Zale 2007; Fogel et al. 2022), smoltification, adult survival (Keefer, Peery and Caudill 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 degrees 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.

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 due to increasingly 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 predominantely 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 adult summer adult migration patterns (Beechie et al. 2023; Crozier and Zabel 2006; Crozier et al. 2010; Quinn 2007).

Siegel and Crozier (2020) suggest that for some salmon populations, climate change may drive mismatches between juvenile arrival timing and prey availability in the marine environment. 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 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. They recommended that managers maintain and augment such life-history diversity.

## In the Pacific Northwest and Olympic Peninsula

In Washington State, increases in freshwater temperatures for salmon streams are predicted in addition to large shifts in hydrology (Climate Impacts Group 2009). Projected changes in climate for the Olympic Peninsula were summarized in Halofsky et al. (2011); Dalton (2016); the 2020 State of Our Watershed Reports from Northwest Treaty Tribes (Northwest Indian Fisheries Commission 2020) (https://nwifc.org/publications/state-of-our-watersheds/)). Northwest Indian Fisheries Commission (2020) summarizes potential climate change impacts within the Olympic Peninsula stating, “the observed and projected trends include warmer air temperatures; shrinking glaciers and snowpack; lower summer streamflows; higher winter flood flows; shifts in streamflow patterns and timing; higher stream temperatures; larger and more frequent wildfires; warmer ocean temperatures; rising sea levels; and changing ocean chemistry, including ocean acidification and lower levels of dissolved oxygen.” On the OP, warming has already occurred, and is projected to further affect allseasonal temperatures, with the largest increases occurring during summer. Projected decreases in precipitation in summer in combination with increased summer evapotranspiration will further impact stream flows for both juvenile and adult steelhead. Additionally, increases in winter precipitation quantity, combined with an increase in event intensity in the western portion of the DPS, will likely result in redd scouring and habitat degradation (see Halofsky et al. (2011) and references therein). Changes in precipitation and timing of peak streamflow may lead to increased runoff and flood risk, with an increased frequency and magnitude of flooding. Warming is likely to reduce snowpack (less winter snow accumulation) which would in turn decrease the risk of floods in springtime, but also reduce stream cooling and flow augmentation in the late spring and summer from snow melt. The biggest changes in streamflow are projected where rivers flow from the Olympic Mountain Range; where snowpack is likely to decline rapidly, especially for areas that will transition from a mix of rain/snow to rain dominant with warming (Yoder and Raymond 2022). Specifically, model projections show up to 30% decline in average summer flow in reaches of low intrinsic potential (<20% in medium to high intrinsic potential) by 2040 (Reeves et al. 2018), and average winter flows of at least 30% higher (Reeves et al. 2018; Safeeq et al. 2015).

Many of these ongoing changes have already been observed on the OP. On USFS land within the OP, there has been a decrease in wetted bank extent and increases in August temperatures from <14 °C in 2002 to 14-18°C in the late2010s, with data ending in 2018 (Dunham et al. 2023). Additionally WDOE stream temperature data from Sol Duc shows warming water temperatures in April and May in certain recent years[46]. Peak winter flows have already increased while summer low flows have already decreased. An assessment of peak flood flows between 1976 and 2019 found that peak flows have increased for the Hoko, Hoh, Calawah, and Quinault rivers, by 5% to 18% with the Hoh River increasing by 18.4% (Northwest Indian Fisheries Commission 2020). In both the Calawah and Bogachiel rivers, it is becoming common for peak flows to be at or above flood stage. Examination of the peak discharges for the coastal drainages of the OP DPS watersheds found that the two-year flood event has been 10 to 35% greater over the last 40 years, relative to over the entire length of the stream-gage record (East et al. 2017). In the Hoh River basin, the three largest peak flow events recorded have occurred since 2002 (East et al. 2018). The 2-year flood peak calculated for the Hoh River for water years 1978–2013 was 1024 cms, whereas the 2-year flood for the entire period of record at the Hoh River gaging station (12041200) was 924 cms (East et al. 2018). The Hoh, Queets, and Quinault rivers have all widened since 1970 consistent with greater flood activity, and Hoh River is showing greater braiding likely related to increased sediment loads from retreating glaciers (East et al. 2017). The general increase in flood activity throughout the OP after the mid-1970s coincided with the onset of a wet phase of the Pacific Decadal Oscillation (PDO, an index of monthly sea-surface temperature anomalies over the North Pacific) (Mantua et al. 1997). This mid-1970s climatic transition has been identified as a major atmospheric and hydrologic shift that affected a large region of the Pacific in both the northern and southern hemispheres (Castino, Bookhagen and Strecker 2016; East et al. 2018). Summer low flows have decreased further over time in the Calawah River basin, where the average low flow in the in late 1970s through the 1990s was 2.0cms, while in the 2000s average summer low flow has been 1.5cms.

[46] Washington Department of Ecology. 2023. Freshwater DataStream, https://apps.ecology.wa.gov/ContinuousFlowAndWQ/StationDetails?sta=20A070; provided in a public comment on the 90 day finding from The Conservation Angler and Wild Fish Conservancy

Northwest Indian Fisheries Commission (2020) provided information on observed system-specific climate changes. For the Quinault Basin glaciers are receding, including those that supply steady streamflow to Quinault and Queets rivers.  Further, glacier loss has been observed, including the complete loss of the Anderson Glacier (Northwest Indian Fisheries Commission 2020). They note that glacier loss results in "less fish habitat, higher stream temperatures and greater sediment load". Adequate streamflow is needed for fish survival and productivity and with climate change, rain dominated watersheds (for example Chehalis River) will likely have increased frequency of low flows in the summer and Glacier-fed watersheds (e.g. Queets) may become rain dominated and have more extreme low summer flows and more frequent intense winter flows. For the Quileute Tribe and the Quillayute River basin, they note increased spring precipitation and winter streamflows with decreased spring snowpacks and summer flows. Within the Calawah river, there have been increasing peak flows along with decreasing low flows over the last 40 years. Sea level rise, coastal storms, and hydrological events contribute to flooding and erosion that may lead to habitat loss. Within the Hoh watershed, glaciers have already been reduced by 40% from 1981-2015. This impacts streamflows and temperature and water quality within fish spawning and rearing habitat. Increasing trends in peak flows and decreasing trends in low flows have occurred. Makah Tribe also note increasing trends in peak flows and decreasing summer low flows in the Hoko River. On the marine side, ocean warming and associated heatwaves, as well as hypoxia and harmful algal blooms have impacted marine areas of interest for Tribes on the Olympic Peninsula.

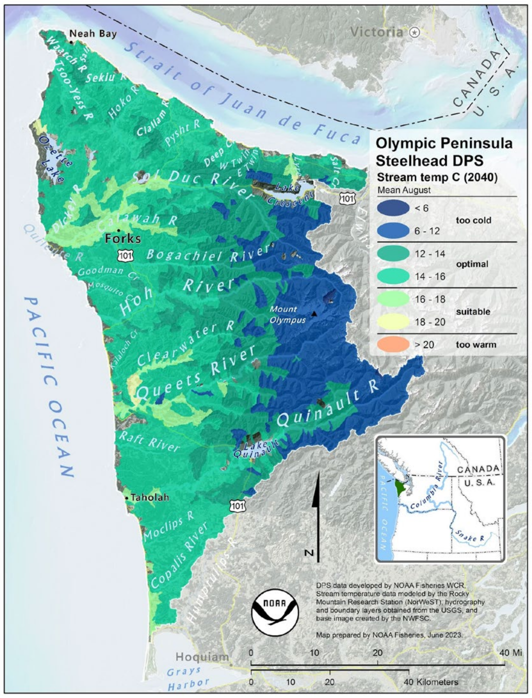
A paper by Riedel et al. (2015) looked at glacial extent in the Olympic Mountains and current contribution of glacial melt to stream flow. This paper showed that all glaciers combined in the Olympics have decreased by 34% over 30 years, resulting in only 4 of the remaining 184 remaining glaciers with an area >1 $km^2$. The greatest losses in area and volume have been on the southern side of the glaciers, at lower elevations, and in northeastern parts of the Olympics. This glacial loss has resulted in a ~20% decline in contribution to summer streamflow of glacial run-off, but still there is significant contribution in the Hoh. For all other major watersheds in the Olympics, glaciers only contribute <5% of summer streamflow.

Additionally, a recent study looked at the changes in number of glaciers and glacial extent over time and predicted future loss of glaciers in the Olympic Mountains due to climate warming (Fountain et al. 2022). Using aerial photograph inventories of the mountains in September 1990, 2009, and 2015, authors determined that the current total ice-covered area is around half of the area in 1900 and that since 1980, glaciers have shrunk -0.59 km^2 per year which has led to the loss of 35 glaciers and 16 perennial snowfields. Models showed that warming winters lead to less snow precipitation accumulating (falling as rain instead of snow), and warming summers result in greater ice melt. Finally, Regional Glaciation Models paired with a "business as usual" carbon emission climate scenario showed that Olympic Mountain glaciers will largely disappear by 2070.

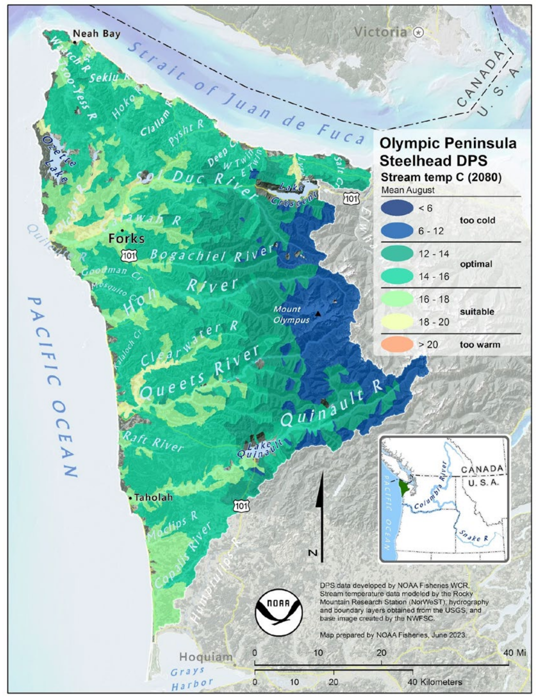
Dalton (2016) describes vulnerability due to climate change within the OP, but notes that certain future conditions that may not be as extreme on the OP as elsewhere, due to the influence of the Pacific Ocean. Further, temperatures and spring precipitation on the Peninsula have increased over the past century, while snowpack and streamflow over the last half century have decreased. Projections from Dalton (2016) suggest a 30% decline in average summer flows that could disrupt migrations, but this may be mitigated by the short migration distances and variation in water temperature throughout the day. Alternatively, there is a predicted 30% increase in winter flows that may impact younger fish through scour but this is likely stream-dependent. Virtual watersheds created in NetMap presented in Dalton (2016) show that temperatures will likely increase in summer in Quinault, Queets, Hoh, and Quillayute rivers but much of the habitat will remain within suitable thermal ranges for salmonids through the 2040s (but growth, predation, and competition could still be impacted). Similarly, a presentation provided to the SRT by Mara Zimmerman on May 15, 2023 and cited by the Co-Manager Olympic Peninsula Steelhead Working Group (2023) noted that throughout the range of the DPS, with climate change, systems are likely to retain either optimal or suitable water temperatures both for juvenile and adult steelhead, based on estimates of future mean water temperatures (Zimmerman presentation to SRT, May 15, 2023). Updated spatial stream network models for the Washington coast region from WDFW show that current August mean, minimum, and maximum temperatures are 14.9°, 1.8, and 25.0 °C and in 2080 are projected to be 15.6, 2.4, and 25.8 °C (Winkowski 2023).

Using stream temperature and flow data from the USDA and USFS Rocky Mountain Research Station (https://www.fs.usda.gov/rm/boise/AWAE/projects/NorWeST/ModeledStreamTemperatureScenarioMaps.shtml, https://www.fs.usda.gov/rm/boise/AWAE/projects/modeled_stream_flow_metrics.shtml), the SRT considered projections of temperature and flow into the future (2040, 2080) for the OP steelhead population range. Average August temperatures projected into the future show minimal areas of unsuitable habitat due to warming but mean weekly maximum temperatures do show larger areas of unsuitable temperatures from high temperatures (Figure B60). For flow, projections into 2040 show extreme change within the Olympic mountains but minimal change in the lowlands, while projections into 2080 show substantial to extreme changes across most of the region (Figure B61).

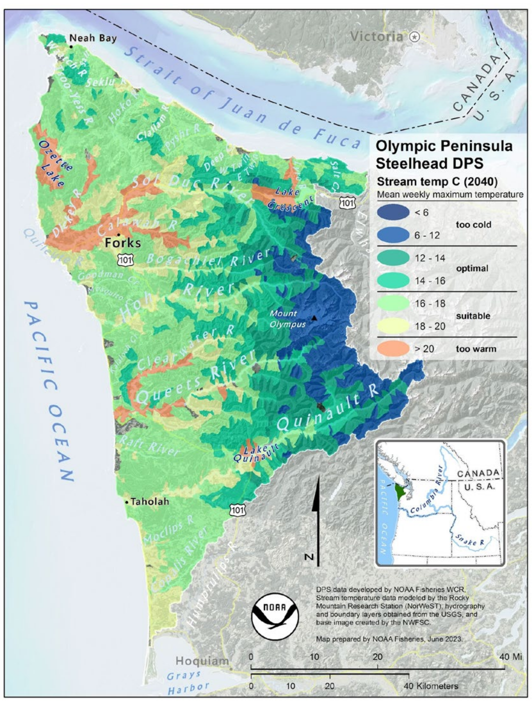
(A)




(B)




(C)




(D)

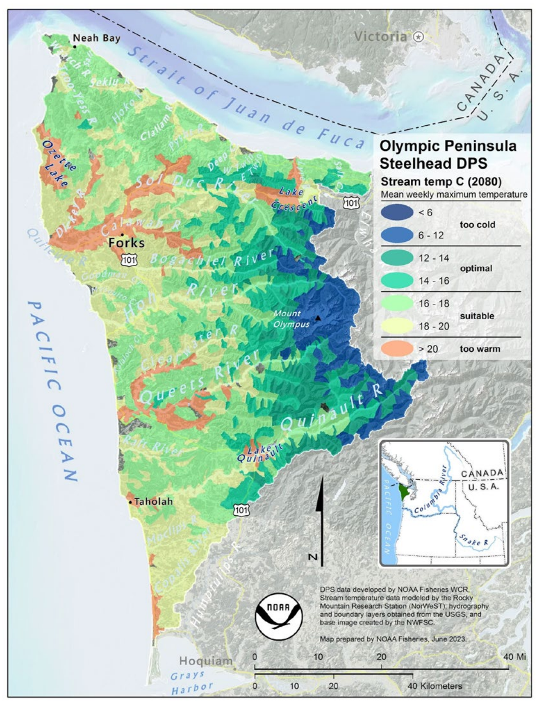



Figure 60 Projected average August stream temperature in 2040 (A) and 2080 (B) and projected maximum August temperature in 2040 (C) and 2080 (D).

(A)

(B)

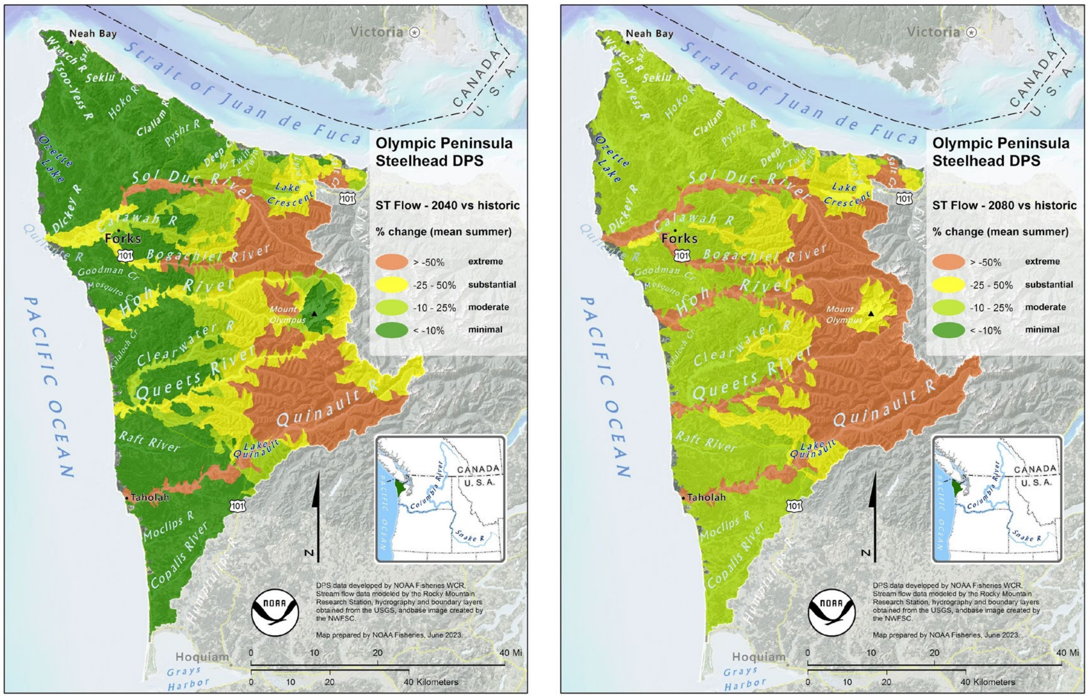



Figure 61. Projected changes in stream flow in 2040 (A) and 2080 (B)

Using stream temperature and flow data from the USDA and USFS Rocky Mountain Research Station (https://www.fs.usda.gov/rm/boise/AWAE/projects/NorWeST/ModeledStreamTemperatureScenarioMaps.shtml, https://www.fs.usda.gov/rm/boise/AWAE/projects/modeled_stream_flow_metrics.shtml), the SRT reviewed projected changes in temperature, flow, and 25 year flood cubic feet per second for individual rivers/streams (Table 31). Changes in summer flow are more likely to affect returning and holding summer-run steelhead, although juvenile and adult winter-run steelhead in the Upper Quinault and Queets rivers and Salt-creek independents tributaries may also be affected. The highest temperatures experienced now and likely into the future are predicted to impact the Lyre winter-run and Clearwater summer-run populations. Summer low flows are predicted to decrease anywhere between 5% and 43% by 2040 in the Quillayute River Basin with the largest changes predicted to occur in the Sol Duc, Upper Bogachiel, and Quillayute River proper. Summer-low flows are already a limiting factor for the range of the DPS, so further declines in low flows due to climate change may restrict the range of the DPS even further.

Table 32 Summary values by population and reaches for future climate change projections including stream flow percent change (now vs. 2040 or 2080), 25 years flood cubic feet per second percent changes (now vs. 2040 or 2080), and mean max temperature now (~2011) vs 2040 and 2080. Summarized for all use types. For flow, green to red gradient is smaller to greater change and for temperature, green to red are lower to higher temperatures.

| | | | **Stream Flow CFS (% change)** | | | | | NorWeST Temp Mean Max Week °C A1B Scenario | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Population | Run | **Use Length m** | Summer season mean NOW vs 2040 | Summer season mean NOW vs 2080 | 25yr Flood NOW vs 2040 | 25yr Flood NOW vs 2080 | | 2011 | 2040 | 2080 |
| Salt Creek-Independents | winter | 31011 | -0.401 | -0.478 | -0.081 | 0.206 | | 14.7 | 15.7 | 16.4 |
| Lyre | winter | 14836 | -0.270 | -0.345 | 0.007 | 0.129 | | 18.8 | 19.8 | 20.5 |
| Pysht-Independents (including the Twins) | winter | 91861 | -0.102 | -0.181 | -0.019 | 0.041 | | 15.6 | 16.6 | 17.3 |
| Clallam | winter | 42838 | -0.056 | -0.141 | -0.057 | -0.087 | | 16.3 | 17.2 | 17.9 |
| Hoko | winter | 117457 | -0.056 | -0.145 | -0.066 | -0.077 | | 16.0 | 17.0 | 17.7 |
| Sekiu | winter | 44658 | -0.050 | -0.143 | -0.051 | -0.042 | | 16.2 | 17.2 | 17.9 |
| Sail | winter | 11449 | -0.053 | -0.157 | -0.006 | 0.006 | | 15.0 | 16.0 | 16.7 |
| Tsoo-Yess-Waatch | winter | 80989 | -0.052 | -0.148 | -0.018 | -0.027 | | 15.9 | 16.9 | 17.6 |
| Ozette | winter | 149053 | -0.051 | -0.142 | -0.024 | 0.000 | | 17.6 | 18.6 | 19.3 |
| Quillayute-Bogachiel | winter | 188336 | -0.286 | -0.398 | -0.001 | 0.058 | | 17.2 | 18.2 | 18.9 |
| Dickey | winter | 185791 | -0.054 | -0.142 | -0.035 | -0.005 | | 17.6 | 18.6 | 19.3 |
| Sol Duc | winter | 250733 | -0.319 | -0.456 | 0.054 | 0.128 | | 16.7 | 17.7 | 18.4 |
| Calawah | winter | 139831 | -0.158 | -0.246 | -0.018 | 0.040 | | 17.9 | 18.8 | 19.5 |
| Hoh | winter | 276356 | -0.277 | -0.495 | 0.200 | 0.376 | | 15.1 | 16.0 | 16.7 |
| Goodman Creek | winter | 44652 | -0.065 | -0.147 | 0.034 | 0.005 | | 16.5 | 17.5 | 18.2 |
| Mosquito Creek | winter | 20269 | -0.065 | -0.147 | 0.047 | 0.027 | | 16.6 | 17.5 | 18.2 |

| Population | Run | **Use Length m** | **Stream Flow CFS (% change)** Summer season mean NOW vs 2040 | Summer season mean NOW vs 2080 | 25yr Flood NOW vs 2040 | 25yr Flood NOW vs 2080 | | NorWeST Temp Mean Max Week °C A1B Scenario 2011 | 2040 | 2080 |
|---|---|---|---|---|---|---|---|---|---|---|
| Kalaloch Creek | winter | 11076 | -0.076 | -0.158 | 0.048 | 0.049 | | 15.4 | 16.4 | 17.1 |
| Queets | winter | 220090 | -0.386 | -0.575 | 0.114 | 0.187 | | 16.1 | 17.1 | 17.8 |
| Clearwater | winter | 156294 | -0.142 | -0.224 | 0.014 | 0.019 | | 18.1 | 19.1 | 19.8 |
| Raft | winter | 39724 | -0.070 | -0.144 | 0.100 | 40.135 | | 16.8 | 17.8 | 18.5 |
| Lower Quinault | winter | 152089 | -0.263 | -0.397 | 0.110 | 0.143 | | 16.1 | 17.1 | 17.8 |
| Upper Quinault | winter | 183483 | -0.488 | -0.698 | 0.111 | 0.197 | | 14.8 | 15.8 | 16.5 |
| Moclips | winter | 17988 | -0.067 | -0.134 | 0.213 | 0.325 | | 17.1 | 18.1 | 18.8 |
| Copalis | winter | 37636 | -0.064 | -0.128 | 0.128 | 0.339 | | 18.1 | 19.1 | 19.8 |
| Quillayute-Bogachiel | summer | 115484 | -0.389 | -0.514 | 0.011 | 0.076 | | 17.4 | 18.4 | 19.1 |
| Sol Duc | summer | 186606 | -0.434 | -0.591 | 0.098 | 0.187 | | 16.1 | 17.1 | 17.8 |
| Calawah | summer | 123122 | -0.147 | -0.235 | -0.019 | 0.040 | | 17.9 | 18.9 | 19.6 |
| Hoh | summer | 123949 | -0.308 | -0.572 | 0.265 | 0.507 | | 15.1 | 16.1 | 16.8 |
| Queets | summer | 106666 | -0.398 | -0.612 | 0.127 | 0.225 | | 16.4 | 17.4 | 18.1 |
| Clearwater | summer | 62959 | -0.164 | -0.247 | 0.013 | 0.020 | | 19.2 | 20.1 | 20.8 |
| Quinault | summer | 127968 | -0.476 | -0.688 | 0.091 | 0.155 | | 16.7 | 17.7 | 18.4 |

A new Climate Adaptation Framework by the Coast Salmon Partnership looked at the resilience to climate change of salmon watershed habitats along the Washington coast (https://www.coastsalmonpartnership.org/current-initiatives/climate-framework/). This work includes a tool to explore the resiliency of various watersheds - https://coast-salmon-partnership.shinyapps.io/CRI_app/. Overall, most of the watersheds on the coast in the OP steelhead DPS range were found to have higher overall resiliency to climate change than watersheds further south. But, certain watersheds in WRIA 20 had lower resiliency, mainly due to metrics around summer low flows. Though this work was made public after the status review teams finalized scoring for the risk assessment, it corroborates that low summer flow is likely going to impact certain streams in the range of the DPS but there also may be some areas where climate change will be less impactful. See the user guide for the tool (Adams and Zimmerman 2024) for more information on the metrics used.

### Effects to OP steelhead

For OP steelhead, increases in summer stream temperatures may especially pose risks to juvenile summer- and winter-run OP steelhead that spend multiple summers in freshwater (Halofsky et al. 2011). Adult summer steelhead require cool water holding pools (Baigún 2003; Baigun 1994; Nakamoto 1994; Nielsen, Lisle and Ozaki 1994) which may be less available with warming temperatures, resulting in higher mortality and/or lower reproductive success (Dalton 2016). Low summer stream flows may affect summer-run steelhead migration by dewatering stream reaches or limiting the accessibility of waterfall or cascades (Halofsky et al. 2011). Increases in flows other times of year may displace juvenile fish and/or reduce the availability of suitable slow-water habitat for young fish. However, winter-run steelhead spawn after peak flow events and may be less susceptible to their redds being scoured (Halofsky et al. 2011). Still, future changes in streamflow could increase stream scouring impacting eggs and embryos, while warmer temperatures may result in early emergence leading to smaller individuals (Dalton 2016). Authors note that salmon fry in low gradient streams may be less vulnerable to displacement from high winter stream flows than fish that emerge later in the year in steeper streams (such as summer steelhead) (Dalton 2016). Changes in flows and temperatures could also impact smolt migration timing (Dalton 2016). Climate Impacts Group (2009) highlighted that salmonids with extended freshwater rearing such as steelhead may experience particularly large increases in temperature and hydrologic stress in summer (from stream temperature increases and lower stream flows), that may result in lower reproductive success. There may be positive impacts from climate change as well, mainly possibly longer growing seasons due to temperature increases, increased productivity within the food-web, and more rapid growth at certain times and life stages (Dalton 2016; Halofsky et al. 2011). Specifically, warmer conditions in summer would likely reduce growth but warmer conditions at other times of year could increase growth rates (Dalton 2016). Warmer temperatures also potentially increase competition with other (especially invasive) species (or predation) and increase susceptibility to disease as well.

For context, Beechie et al. (2023) provided a diagram of overlap of key life stages and effects of climate change based on salmonid populations in the Chehalis basin. We replicate part of Figure 3 from Beechie et al. (2023) to show timing of how climate change would likely impact different stages of winter-run steelhead in the Pacific Northwest and on the OP.

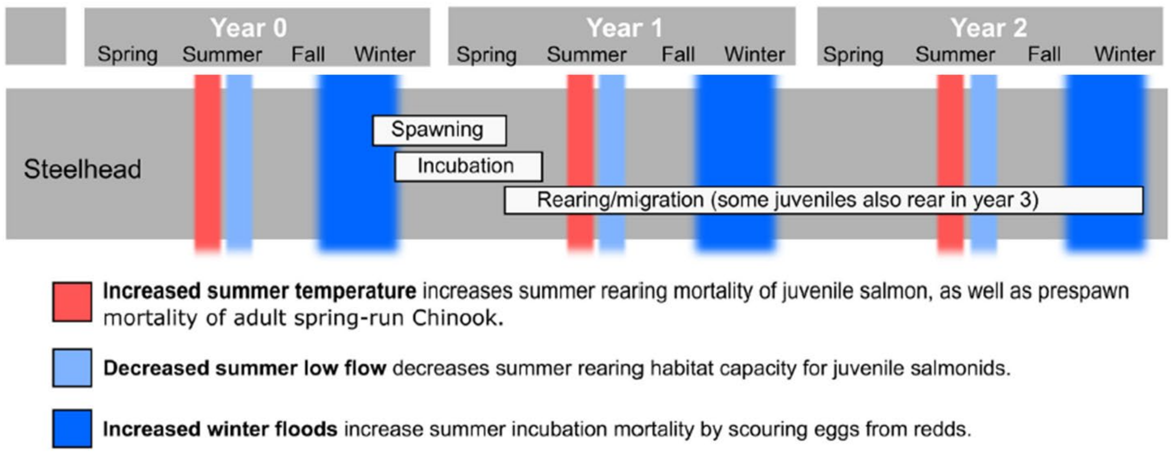



Figure 62. Modified from Beechie et al. (2023), timing of effects of climate change on different life stages of winter-run steelhead.

Within the 2020 State of Watershed Report, the Northwest Treaty Tribes describe that the overall increase in stream temperature leads to salmon being exposed for longer to temperatures outside of their ranges for reproduction and survival (Northwest Indian Fisheries Commission 2020). Also increased temperatures along with changes in streamflow result in lower dissolved oxygen, increased sediment, higher disease susceptibility, competition with other species, and variation in prey for salmonid species. Many of the individual watershed/Tribal reports in the State of Our Watersheds Report note impacts on streamflow and temperature changes to salmon productivity and survival. Within the Quileute report, they note that warmer stream temperatures may lead to accelerated growth and early emergence as well as hydrological impacts on smolting and migration behavior, with overall negative impacts on reproductive success.

At the population level, the ability of organisms to genetically adapt to climate change depends on how selection on multiple traits interact, and whether those traits are linked genetically. Factors that affect genetic diversity can thus limit the ability of a population to adapt to climate change. These include, but are not limited to small population size, domestication in hatchery environments, or introgression by introduced non-native stocks. Though populations may be able to adapt to changes if within the range of what they've experienced historically (Waples, Pess and Beechie 2008), it is unknown if Olympic Peninsula steelhead can adapt quickly enough to the rapid pace of changing climate and habitat conditions. Further, any directional selection effects (i.e. harvest selection on run timing) or general decrease in diversity will decrease the ability of steelhead populations to adapt to these changes. McMillan et al. (2022) note that winter-run steelhead in warmer streams migrate and spawn earlier and that early-run time may be important to the resilience of the population with future climate change, but that there has been a decline in early-returning natural-origin fish.

Dalton (2016) state that climate change driven changes in freshwater ecosystems will be relatively small by the mid-century but that more challenges may present in the marine environment.

## Marine Climate Change impacts and OP steelhead

A 2013 report for the Olympic Coast National Marine Sanctuary (OCNMS) reviewed and summarized literature on projected climate change in the marine environment of the Pacific Northwest (including the area within the OCNMS) (Miller et al. 2013). This includes that ocean water could warm by 1 degree Celsius by 2050 with "corrosive ocean water" within shallower areas (water that has a more acidic pH outside of the contemporary values and reduced carbonate ions). Additionally, sea level could rise by over 1 meter by 2100. Changes in the frequency and intensity of storms and upwelling are more uncertain, but the report notes that it is unlikely that climate change will cause any measurable changes in upwelling favorable winds by 2100. Dissolved oxygen is expected to decrease with warming surface waters and there have been declines in dissolved oxygen in specific locations near the OCNMS. This report also notes likely changes in flows and flooding in the Sol Duc, Hoh, Queets, and Quinault rivers.

In the marine ecosystem, salmon may be affected by warmer water temperatures, increased stratification of the water column, intensity and timing changes of coastal upwelling, loss of coastal habitat due to sea level rise, ocean acidification, and changes in water quality and freshwater inputs (Independent Scientific Advisory Board 2007; Mauger et al. 2015). Salmon marine migration patterns could be affected by climate-induced contraction of thermally suitable habitat. Climate change in the marine environment may also reduce forage fish prey for steelhead and other salmonids. Ocean acidification will likely disrupt the food web (through impacts to calcifying planktonic organisms) and warmer temperatures may constrict salmon habitat, affecting adult returns and reproductive success (if fish are smaller returning) (summarized in Dalton et al. 2016). Bioenergetics models informed by data on steelhead mainly from the Central North Pacific Ocean suggest that growth of steelhead in the ocean environment varies with prey quality, consumption rates, overall total consumption, and temperature, though more consumption can compensate for low quality prey (Atcheson et al. 2012). Models suggest that steelhead growth declines with temperatures that deviate from the optimum and there is a narrow range of temperature that results in optimal growth (Atcheson et al. 2012). Also, a study by Abdul-Aziz, Mantua and Myers (2011) predicted an 8 to 43 percent contraction of steelhead species' marine habitat due to climate change between the 2020s and 2080s (depending on time period). A recent assessment of the vulnerability to climate change for 64 different species in the California Current marine ecosystem ranked steelhead as having both high exposure and high sensitivity to climate change and all salmon species considered ranked either very high or high for vulnerability, likely related to their anadromous life history (McClure et al. 2023). Northward range shifts are a climate response expected in many marine species, including salmon (Cheung et al. 2015). However, salmon populations are strongly differentiated in the northward extent of their ocean migration, and hence would likely respond individualistically to widespread changes in sea surface temperature.

Siegel and Crozier (2020) observe that changes in marine temperature are likely to have a number of physiological consequences on fishes themselves. For example, in a study of small planktivorous fish, Gliwicz et al. (2018) found that higher ambient temperatures increased the distance at which fish reacted to prey. Numerous fish species (including many tunas and sharks) demonstrate regional endothermy, which in many cases augments eyesight by warming the retinas. However, Gliwicz et al. (2018) suggest that ambient temperatures can have a similar effect on fish that do not demonstrate this trait. Climate change is likely to reduce the availability of biologically essential omega-3 fatty acids produced by phytoplankton in marine ecosystems. Loss of these lipids may induce cascading trophic effects, with distinct impacts on different species depending on compensatory mechanisms (Gourtay et al. 2018). The ecological consequences of these effects and their interactions add complexity to predictions of climate change impacts in marine ecosystems.

As stated in Ford (2022) – "Historically, ocean conditions cycled between periods of high and low productivity. However, global climate change is likely to disrupt this pattern, in general, leading to a preponderance of low productivity years, with an unknown temporal distribution (Crozier et al. 2019). Recent (2015–19) ensemble ocean indicator rankings include four of the worst seven years in the past 20, meaning that an entire Chinook salmon generation has been subjected to poor ocean productivity conditions." Additionally, a NOAA presentation by Brian Burke provided in comments to the 90-day finding, noted the increase frequency and magnitude of marine heatwaves in the N.E. Pacific (citing the California Current Ecosystem Status Report - https://www.integratedecosystemassessment.noaa.gov/regions/california-current/california-current-marine-heatwave-tracker-blobtracker)

The assessment by Co-Manager Olympic Peninsula Steelhead Working Group (2023) suggested that interannual variation in recruitment and kelt survival were both partially explained by summer sea surface temperature (SST) (and also pink salmon abundance; as well as North Pacific Gyre Oscillation for recruitment). In other words, this analysis showed a negative correlation between recruitment and summer SST and a negative correlation between kelt survival and summer SST. Work by Kendall, Marston and Klungle (2017) showed variability in smolt survival consistently for Washington coast and Strait populations (but with less magnitude fluctuations for Washington Coast, on average). There is uncertainty in how smolt survival and recruitment and kelt survival will change overtime, but this analysis strongly suggests that ocean survivals are likely to decrease in warm years and the frequency of these warm years will increase with climate change.

## Hatchery impacts

The effects of hatchery fish on the status of an ESU or DPS depends upon which of the four key attributes -- abundance, productivity, spatial structure, and diversity -- are currently limiting the ESU/DPS, and how the hatchery fish within the ESU/DPS affect each of the attributes (70 FR 37204). In general, hatchery programs can provide short-term demographic benefits to salmon and steelhead, such as increases in abundance during periods of low natural abundance. They also can help preserve genetic resources until limiting factors can be addressed. However, the long- term use of artificial propagation may pose risks to natural productivity and diversity. 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 fish while integrated incorporate natural-origin broodstock (Harbison et al. 2022). In order to reduce risks from hatcheries, the WDFW Statewide Steelhead Management Plan (SSMP) and the Hatchery Scientific Review Group (HSRG) (a now disbanded independent scientific panel that reviewed Pacific Northwest hatchery operations), set 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). In the case of OP hatchery programs, most of the hatcheries maintain broodstocks that were founded by non-native stocks and are operated as segregated programs to minimize introgression with natural-origin steelhead.

Extensive hatchery programs have been implemented throughout the range of West Coast steelhead. While these programs may have succeeded in providing harvest opportunities and increasing the total number of naturally spawning fish, the programs have also likely increased risks to natural populations. Hatchery programs and hatchery-produced steelhead can affect naturally produced populations of salmon and steelhead in a variety of ways, including competition (for spawning sites and food) and predation effects, disease effects, genetic effects (e.g., outbreeding depression, hatchery-influenced selection (i.e. domestication)), broodstock collection effects (inadvertent selection for run timing or size, or limited numbers of broodstock), and facility effects (e.g., water withdrawals, effluent discharge) (Hatchery Scientific Review Group 2014; McMillan et al. 2023; Ohlberger et al. 2018; Rand et al. 2012), as well as masking of trends in natural populations through straying of hatchery fish. Additionally, hatchery influence can result in reduced genetic diversity and reproductive fitness through interbreeding between natural and hatchery-origin steelhead, and the masking of trends in natural populations through the straying of hatchery-origin fish onto spawning grounds. State natural resource agencies have adopted or are developing 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. The role of artificial propagation in the conservation and recovery of salmonid populations continues to be the subject of vigorous scientific research.

A recent paper by McMillan et al. (2023) summarized literature on effects of hatchery programs on salmonids. For steelhead, 23/35 papers reviewed found adverse or minimally adverse effects of hatcheries on the corresponding natural steelhead population. Chilcote, Goodson and Falcy (2011) found a negative relationship between recruitment and proportion of hatchery fish spawning naturally for steelhead, Chinook, and coho populations in Oregon, Washington, and Idaho (even after corrections to this publication). One study of steelhead in the Hood River, OR found evidence that hatchery produced steelhead increased numbers on the spawning grounds but reduced the effective population size substantially (especially if >10% of the naturally spawning fish are hatchery-origin) (Christie et al. 2012). On the beneficial effects side, two studies reported on effects of a long-term experiment of a captive breeding program for steelhead (using all wild fish as broodstock), including a paper in 2018 that found that the breeding program led to greater redd abundance, expected heterozygosity, and also allelic richness (though not significant) for a depleted steelhead population (Berejikian et al. 2008; Berejikian and Van Doornik 2018). However, another study reported decreased productivity from a recovery program for steelhead, specifically decreased reproductive fitness of wild-born fish from captive parents (Araki, Cooper and Blouin 2009). A recent paper by Courter et al. (2019) found no negative effect of the hatchery summer steelhead spawner abundance on winter steelhead recruitment.

In its 1996 review, NMFS noted that past hatchery practices and practices at the time of the review were a major threat to the genetic integrity of OP steelhead (Busby et al. 1996). Where hatchery-origin and natural-origin steelhead co-occur on the Olympic Peninsula, there is concern about genetic introgression due to interbreeding, especially because all of the current hatchery broodstocks were founded or have been significantly influenced by out-of-DPS stocks. Estimates of the proportion of naturally spawning steelhead that were of hatchery-origin ranged from 16 to 44 percent, but with the largest runs (Queets and Quillayute) having the lowest proportions of hatchery-origin spawners (Busby et al. 1996).

The recent review of Washington steelhead population include OP steelhead from WDFW (Cram et al. 2018) also named hatchery operations as "a threat to genetic integrity of wild steelhead populations" in the area occupied by OP steelhead. Cram et al. (2018) stated that, as of 2014, there were 11 hatchery programs on the Olympic Peninsula with an average annual release of 1,393,022 smolts from 2000 to 2008 and 1,072,781 from 2009 to 2013. Most hatchery programs (10 of 11) are used for harvest augmentation and most of these were founded from two steelhead populations not native to the Olympic Peninsula – Chambers Creek early winter (Puget Sound) and Skamania early summer (Columbia River: the use of which is being eliminated elsewhere on the West Coast due to impacts on listed steelhead, see Ford (2022)). Of the hatchery programs in the Olympic Peninsula, five are off-site release programs that transfer smolts from their hatchery to another watershed for release. Cram et al. (2018) notes that if adults from these programs are not caught by fisheries, they place natural-origin OP steelhead at risk genetically and ecologically. An integrated hatchery program was initiated in the Bogachiel River in 2013 using hook and line caught natural-origin broodstocks, but has since been discontinued, additionally the program on the Sol Duc River ended and steelhead there are now managed as a "Wild Steelhead Gene Bank" (Cram et al. 2018).

The recent paper by McMillan et al. (2022) discusses potential hatchery impacts on early-returning winter run natural-origin steelhead on the Olympic Peninsula. Specifically, hatchery-origin winter-run steelhead migration overlaps with the historical early-run timing of natural-origin winter-run steelhead so there is high likelihood of interaction between the early-run natural-origin and hatchery-origin steelhead. Additionally, commercial and recreational fisheries targeting hatchery-origin steelhead with early run-timing may be harvesting early-run natural-origin steelhead as well, potentially creating directional selection against early run-timing given that run-timing is a heritable trait. Recent research suggests that hatchery introgression can reduce variation in run timing and even despite reduced fitness of hatchery fish, hatchery alleles can quickly assimilate into natural populations (May et al. 2024).

In the Co-manager's 2023 assessment of the petition (Co-Manager Olympic Peninsula Steelhead Working Group 2023), they state that – Currently, three hatchery stocks are propagated and released in the OP DPS: early winter (Puget Sound origin, sometimes referred to as Chambers Creek origin), early summer (Lower Columbia origin, sometimes referred to as Skamania origin), and Cook Creek early hatchery winters (putatively Olympic Peninsula origin). Currently, there is limited data to understand the genetic relationships of Cook Creek stock hatchery fish and OP native steelhead. All three hatchery stocks are operated as segregated programs, i.e., they use only hatchery origin fish as broodstock. While this management strategy prevents removing natural spawners from the spawning grounds, it does not prevent hatchery-origin adults from spawning naturally, potentially hybridizing with native steelhead. In addition, the progeny of naturally-spawning hatchery fish would be indistinguishable from native fish. Thus, the co-managers state that these hatchery populations should retain their genetic identity, i.e., should not be introgressed with wild OP steelhead populations and, given the origins of the early winter and early summer stocks, we would expect to be able to genetically distinguish them from wild OP *O. mykiss*.

Specific hatchery information for specific watersheds/rivers is summarized in Harbison et al. (2022) and as well as in our Status Review report; specifically see Table 15 in the Status Review with information on all currently operating hatcheries. A few points on certain hatcheries are worth mentioning. Hatchery fish released in Quinault and Queets rivers are not adipose fin clipped, thus preventing any quantification of hatchery influence except through scale sampling and interpretation or genetics. In the Hoh River, there is a tribal program on the Chalaat Creek which, since 2019, uses broodstock from the Bogachiel or natural-origin Hoh River steelhead. Prior to 2019, broodstock came from Quinault National Fish Hatchery. According to the 2022-2023 Management season report from Hoh Tribe and WDFW, the goal is to produce 100,000 smolts (but this has been closer to 52,000 on average in recent years; pers. comm Brian Hoffman, Hoh Tribe, Natural Resources, June 21, 2023), and 100% of smolts are adipose fin clipped.

Our status review in the section *Hatchery Operations in the Olympic Peninsula Steelhead DPS* summarizes extensively the hatchery programs and hatchery outputs. Hatchery releases have stayed consistent since the late 1970s/early 1980s to the present both for winter-run and summer-run hatchery output. Smolt output depending on the run timing, system, and year can range from <10,000 to >700,000. Additionally, see Appendix A: OP DPS Watershed Summaries for specific hatchery output for individual systems.

In the NMFS 1996 review (Busby et al. 1996), NMFS noted the estimated proportion of hatchery stocks on natural spawning grounds ranged from 16 to 44 percent. This proportion was lowest for the two rivers with the largest production of natural-origin steelhead - Queets and Quillayute Rivers. At the time, according to Busby et al. (1996) percent hatchery origin spawners was 43% for the Pysht River, 16% for the Quillayute River, 19% for the Queets River, 44% for the Quinault River, and 37% for the Moclips. As noted in the status review, more recently, the Washington Coast Sustainable Salmon Partnership (WCSSP, 2013) estimated the proportion of hatchery-origin adults that were naturally spawning in Olympic Peninsula DPS basins based on the professional opinion of local biologists. In general, smaller basins with hatchery programs (Tsoo-Yess River, Goodman Creek) and the Quinault River were thought to have higher pHOS levels (26-50%), other basins less so (>25%); although a number of basins were not reported. Most summer-run steelhead pHOS is unknown, however the following website was reported by the petitioners from WDFW (https://fortress.wa.gov/dfw/score/score/hatcheries/hatchery_details.jsp?hatchery=Bogachiel%20Hatchery) which shows that for 2009, pHOS for summer-run steelhead for the hatchery program on the Bogachiel River were 23% and 9% for winter-run.

Scott and Gill (2008) showed gene flow of early Winter steelhead from Chambers creek stock into the Hoko, Pysht, and Sol Duc rivers, (5.5-14.5%, 12-75%, and 2.5-6% gene flow respectively). This led to elimination of winter steelhead smolt release into the Pysht river in 2009, as well as Goodman Creek, Clallam River, and Lyre RiverIn2012River. In

2012RiverIn2012 the Sol Duc River was designated by WDFW as a Wild Stock Gene Bank, terminating summer smolt releases in 2011 and winter in 2013 (winter-run was local-origin broodstock steelhead) (see Hatchery regulations above).

A recent review by Marston and Huff (2022) looked at the compliance of the WDFW operated Bogachiel Hatchery with standards set in the SSMP. This report also summarized existing hatcheries and then looked at compliance of WDFW operated programs. Specific conclusions included that stray rates from the Bogachiel programs are unknown, for early winter steelhead they modeled – 6% of hatchery fish spawning in the overlap period when natural-origin fish are spawning, and for summer steelhead – less than 1% of hatchery fish spawning in the overlap period with natural-origin fish. Marston and Huff (2022) recommended assessing the status, spawn timing, and spatial distribution of summer natural-origin steelhead, and also re-evaluating the March 15th hatchery origin/natural origin spawner cut-off date, amongst other recommendations. Recommendations also included specifics for discontinuing or continuing State-run programs and how to better manage them.
Kassler et al. (2011) provided evidence of hatchery-origin ancestry for collections of natural-origin fish; indicating at some point there was natural-origin spawning with hatchery fish. The absence of up-to-date comprehensive genetic sampling of natural and hatchery populations makes it difficult to draw conclusions on the impact of hatcheries. All genetic samples are at least 6 years old and some nearly 30 years old, and few directed studies have been conducted to specifically evaluate hatchery introgression or genetic impacts on the wild populations. We therefore do not know the current introgression levels of hatchery genes in the natural population. However, hatchery output has continued for multiple decades, while natural-origin populations have declined, the impacts of hatcheries outlined above are likely continuing and may be impacting more as the natural-origin population declines. At the same time, natural-origin survival is higher than hatchery smolt survival and hatchery smolt survival has declined in recent years (Harbison et al. (2022)), so there may be some selection against the maintenance of non-native (hatchery-origin) genes.

We note that many public comments on the 90-day finding voiced support for more broodstock hatchery programs in the OP as a way to continue to have fisheries; we assume because many of these pointed to Oregon programs that use natural-origin stock that this was encouraging the use of natural-origin steelhead as broodstock in hatcheries. Other comments noted that Chinook hatchery releases may be competing with steelhead. Another comment provided reference to epigenetics in steelhead; certain recent studies have found epigenetic differences in natural-origin versus hatchery-origin steelhead but no studies for steelhead have shown if these differences lead to difference in fitness (Gavery et al. 2018; Koch, Nuetzel and Narum 2023).

## Competition and indirect food-web interactions

Ruggerone and Nielsen (2004) summarized literature on competition between pink salmon and other salmonids. Research summarized therein found that pink salmon alter the prey abundance of other species (such as abundance of zooplankton, squid), and that this then can lead to an altered diet, reduced consumption, reduced growth, delayed maturation, and reduced survival depending on the salmon species and location. However, some steelhead specific studies indicate that steelhead abundance increased because spawning pink salmon can provide greater prey (in the form of pink fry or eggs) for steelhead, with pink salmon eggs enhancing steelhead parr growth and survival. Additional papers have looked at possible connections between pink salmon abundance and other salmonid growth and survival (Ruggerone and Irvine 2018; Ruggerone et al. 2023). The assessment by the Co-Manager Olympic Peninsula Steelhead Working Group (2023) that interannual variation in recruitment and kelt survival were both partially explained by Pink salmon abundance (and also SST; as well as North Pacific Gyre Oscillation for recruitment). In other words, this analysis showed a negative correlation between recruitment and Pink salmon abundance and a negative correlation between kelt survival and Pink summer abundance. We note that the co-manager analysis however did not sufficiently consider impacts of pinniped predation on kelt survival or smolt survival because of a lack of data for seal/sea lion (pinniped) abundance (shorter time series compared to other factors) and so there is still uncertainty about impacts of predation on survival for steelhead.

Similarly, potential food web impacts due to loss of in-river chum salmon abundance is discussed in Appendix A. Specifically, the decrease in steelhead abundance in the Lyre River may be related to the decline of chum salmon (Goin 1990). Chum salmon spawning escapement estimates for the Lyre were close to 10,000 fish annually (Goin 1990) but by 1996, abundance levels were reduced to 500 to 1,000 annually (McHenry, Lichatowich and Kowalski-Hagaman 1996). Chum fry would emerge from the gravels in the spring, similar to timing of outmigration for steelhead smolts, and so chum fry were hypothesized as a food resource for steelhead smolts.

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# Appendix C: Status Review Team Scoring

Table 33. Status Review Team scoring of Viable Salmonid Population Criteria for steelhead populations in the Olympic Peninsula steelhead DPS. Scores represent the average of all team members voting on a 1 (low) to 5 (high) risk range. Run: W- winter-run, S – summer-run.

| | | Abundance | Productivity | Spatial Structure | Diversity |
|---|---|---|---|---|---|
| Population | Run | Average | Average | Average | Average |
| Salt Creek | W | 4.0 | 2.9 | 1.8 | 1.9 |
| Lyre River | W | 3.7 | 2.7 | 1.3 | 2.1 |
| Lyre River | S | 4.2 | 3.3 | 1.5 | 3.0 |
| West Twin River | W | 3.7 | 2.5 | 1.3 | 2.0 |
| East Twin River | W | 3.6 | 2.3 | 1.3 | 2.0 |
| Deep Creek | W | 3.6 | 2.5 | 1.3 | 2.0 |
| Pysht River | W | 3.1 | 2.3 | 1.7 | 2.3 |
| Clallam River | W | 3.1 | 2.2 | 1.3 | 2.0 |
| Hoko River | W | 3.4 | 2.3 | 1.3 | 2.1 |
| Sekiu River | W | 4.0 | 3.0 | 1.5 | 2.2 |
| Sail River | W | 4.0 | 3.0 | 1.4 | 2.2 |
| Waatch River | W | | | 1.2 | 2.4 |
| Tsoo-Yess River | W | | | 1.2 | 2.4 |
| Ozette River | W | 3.3 | | 1.2 | 2.6 |
| Quillayute River | W | 2.5 | 2.6 | 1.3 | 2.3 |
| Quillayute River | S | 4.3 | 3.1 | 1.5 | 2.4 |
| >Dickey River | W | 2.8 | 2.4 | 1.3 | 2.0 |
| >Sol Duc River | W | 2.4 | 2.4 | 1.3 | 2.1 |
| >Sol Duc River | S | 3.8 | 3.3 | 1.5 | 2.4 |
| >Calawah River | W | 2.3 | 2.4 | 1.3 | 2.1 |
| >Calawah River | S | 4.3 | 4.0 | 1.5 | 2.4 |
| >Bogachiel River | W | 2.9 | 2.9 | 1.3 | 2.4 |
| >Bogachiel River | S | 4.3 | 3.5 | 1.5 | 2.4 |
| Lonesome Creek | W | 3.5 | | 1.3 | 2.3 |
| Goodman Creek | W | 3.4 | 2.8 | 1.4 | 2.2 |
| Mosquito Creek | W | 3.5 | | 1.3 | 2.3 |
| Hoh River | W | 2.2 | 2.3 | 1.3 | 2.4 |
| Hoh River | S | 4.3 | 3.7 | 1.2 | 2.7 |
| Queets River | W | 2.9 | 2.7 | 1.3 | 2.4 |
| Queets River | S | 3.9 | 3.7 | 1.5 | 2.3 |
| >Clearwater River | W | 2.7 | 2.5 | 1.3 | 2.4 |
| >Clearwater River | S | 4.3 | 4.0 | 1.3 | 2.3 |

| Population | Run | Abundance Average | Productivity Average | Spatial Structure Average | Diversity Average |
|---|---|---|---|---|---|
| Raft River | W | | | 1.3 | 2.0 |
| Quinault River | W | 2.9 | 2.7 | 1.3 | 2.4 |
| Quinault River | S | 4.1 | | 1.3 | 2.4 |
| >Upper Quinault River | W | 3.2 | 2.8 | 1.2 | 2.4 |
| >Upper Quinault River | S | 4.5 | 4.0 | 1.2 | 2.4 |
| Moclips River | W | 2.6 | | 1.3 | 2.0 |
| Copalis River | W | 3.0 | | 1.3 | 2.0 |
| | Mean | 3.4 | 2.9 | 1.4 | 2.3 |
| | Median | 3.5 | 2.7 | 1.3 | 2.3 |
| | Big 4W | 2.6 | 2.5 | 1.3 | 2.4 |
| | Strait W | 3.7 | 2.6 | 1.5 | 2.1 |
| | Summer (5) | 4.2 | 3.6 | 1.4 | 2.5 |

Table 34. Status Review Team scoring of Factors for Decline Threats for steelhead populations in the Olympic Peninsula steelhead DPS. Scores represent the average of all team members voting on a 1 (low) to 5 (high) threat range. Run: W- winter-run, S – summer-run.

| | | Habitat loss or destruction | Over utilization | Inadequate regulation | Disease/. Predation | Hatchery effects | Climate change |
|---|---|---|---|---|---|---|---|
| Population | Run | Average | Average | Average | Average | Average | Average |
| Salt Creek | W | 2.2 | 1.4 | 2.2 | 1.0 | 1.1 | 3.2 |
| Lyre River | W | 2.3 | 2.2 | 2.3 | 1.0 | 2.0 | 3.0 |
| Lyre River | S | 2.4 | 2.6 | 2.8 | 1.0 | 2.0 | 3.3 |
| West Twin River | W | 2.0 | 1.4 | 2.1 | 1.0 | 1.0 | 2.9 |
| East Twin River | W | 2.1 | 1.4 | 2.1 | 1.0 | 1.0 | 2.9 |
| Deep Creek | W | 2.1 | 1.4 | 2.1 | 1.0 | 1.0 | 2.8 |
| Pysht River | W | 2.5 | 1.8 | 2.3 | 1.0 | 2.3 | 2.5 |
| Clallam River | W | 2.5 | 1.8 | 2.1 | 1.0 | 1.7 | 2.6 |
| Hoko River | W | 2.6 | 2.4 | 2.3 | 1.1 | 2.8 | 2.5 |
| Sekiu River | W | 2.7 | 2.5 | 2.3 | 1.0 | 2.2 | 2.6 |
| Sail River | W | 2.3 | 1.8 | 2.5 | 1.0 | 1.7 | 2.7 |
| Waatch River | W | 2.3 | 2.0 | 3.0 | 1.0 | 1.8 | 2.7 |
| Tsoo-Yess River | W | 2.3 | 2.0 | 2.8 | 1.4 | 2.2 | 2.7 |
| Ozette River | W | 2.3 | 2.1 | 3.0 | 1.0 | 1.4 | 2.8 |
| Quillayute River | W | 2.1 | 2.8 | 2.7 | 1.2 | 2.5 | 3.3 |
| Quillayute River | S | 2.1 | 2.7 | 4.0 | 1.0 | 2.4 | 3.9 |
| >Dickey River | W | 2.3 | 2.9 | 2.5 | 1.0 | 2.0 | 2.7 |
| >Sol Duc River | W | 1.8 | 2.6 | 2.5 | 1.2 | 2.1 | 3.4 |
| >Sol Duc River | S | 2.1 | 2.7 | 3.9 | 1.2 | 2.1 | 4.0 |
| >Calawah River | W | 2.1 | 2.9 | 2.7 | 1.2 | 2.3 | 2.9 |
| >Calawah River | S | 2.3 | 2.7 | 3.9 | 1.2 | 2.4 | 3.5 |
| >Bogachiel River | W | 2.0 | 2.9 | 2.8 | 1.6 | 2.5 | 3.3 |
| >Bogachiel River | S | 2.0 | 2.7 | 4.0 | 1.4 | 2.4 | 3.6 |
| Lonesome Creek | W | 1.7 | 3.0 | 2.8 | 1.0 | 2.5 | 2.8 |
| Goodman Creek | W | 1.8 | 3.0 | 2.8 | 1.3 | 2.2 | 2.7 |
| Mosquito Creek | W | 1.7 | 2.3 | 2.8 | 1.0 | 2.3 | 2.7 |
| Hoh River | W | 2.1 | 2.7 | 2.7 | 1.4 | 2.3 | 3.4 |
| Hoh River | S | 2.1 | 2.6 | 3.7 | 1.2 | 2.7 | 3.7 |
| Queets River | W | 2.1 | 3.2 | 3.5 | 1.3 | 2.7 | 3.3 |
| Queets River | S | 2.0 | 2.7 | 3.9 | 1.0 | 1.7 | 3.9 |
| >Clearwater River | W | 2.3 | 3.3 | 3.3 | 1.2 | 2.7 | 2.9 |
| >Clearwater River | S | 2.3 | 2.7 | 3.9 | 1.0 | 2.0 | 3.4 |
| Raft River | W | 1.3 | 3.0 | 2.5 | 1.0 | 1.6 | 3.0 |

| | | Habitat loss or destruction | Over utilization | Inadequate regulation | Disease/. Predation | Hatchery effects | Climate change |
|---|---|---|---|---|---|---|---|
| Population | Run | Average | Average | Average | Average | Average | Average |
| Quinault River | W | 2.1 | 3.3 | 3.4 | 1.5 | 3.0 | 3.4 |
| Quinault River | S | 2.1 | 3.1 | 3.7 | 1.2 | 2.5 | 4.0 |
| >Upper Quinault River | W | 2.0 | 3.1 | 3.4 | 1.3 | 3.0 | 3.6 |
| >Upper Quinault River | S | 1.9 | 2.7 | 3.9 | 1.2 | 2.3 | 4.0 |
| Moclips River | W | 2.0 | 2.7 | 2.9 | 1.0 | 2.0 | 2.8 |
| Copalis River | W | 2.0 | 2.5 | 3.0 | 1.0 | 2.0 | 2.6 |
| | Mean | 2.1 | 2.5 | 2.9 | 1.1 | 2.1 | 3.1 |
| | Median | 2.1 | 2.7 | 2.8 | 1.0 | 2.2 | 3.0 |

| | | | | | | |
|---|---|---|---|---|---|---|
| Big 4W | 2.14 | 3.00 | 3.07 | 1.35 | 2.64 | 3.34 |
| Strait W | 2.31 | 1.82 | 2.25 | 1.02 | 1.66 | 2.77 |
| Summer (5) | 2.14 | 2.70 | 3.77 | 1.13 | 2.19 | 3.75 |

# Appendix D. Hatchery Releases

Table 35. Releases of steelhead juveniles by watershed. Releases of steelhead juveniles less than 2 grams in weight were excluded. Broodyear range is not necessarily continuous. (N) designation by source indicates that natural-origin broodstock were used.

| Release Watershed | Run | Source | Number Released | Broodyears |
|---|---|---|---|---|
| Agency Creek | Winter | Bogachiel H. | 2,027 | 1989 |
| | Winter | Hoko R. H. | 59,663 | 1988-2014 |
| Bogachiel River | Winter | Bogachiel H. | 4,055,699 | 1981-2022 |
| | Winter | Quinault NFH | 50,337 | 1986-1987 |
| | Winer | Hoko R. H. | 80,293 | 2010 |
| Calawah River | Summer | Bogachiel H. | 1,081,556 | 1981-2022 |
| | Summer | Chehalis R. | 30,065 | 1983-1985 |
| | Summer | Skykomish R. | 31,656 | 1990 |
| | Summer | Sol Duc R. | 10,000 | 1981 |
| | Summer | Washougal | 10,802 | 1986 |
| | Winter | Bogachiel H. | 2,491,293 | 1981-2022 |
| | Winter | Calawah R. | 17,346 | 2019-2020 |
| | Winter | Quinault NFH | 24,962 | 1986-1987 |
| Chalaat Creek | Winter | Bogachiel H. | 83,000 | 2022 |
| | Winter | Chalaat Cr. H. | 519,616 | 2011-2019 |
| | Winter | Quinault NFH | 1,008,892 | 1989-2018 |
| | Winter | Quinault R. | 337,359 | 1981-1987 |
| | Winter | Hoh R. | 90,243 | 1981-2021 |
| Clallam River | Winter | Bogachiel H. | 114,986 | 1981-2004 |
| | Winter | Dungeness R. H. | 9,263 | 2005-2006 |
| | Winter | Elwha R. H. | 31,806 | 2005-2008 |
| | Winter | Hoko R. H. | 26,630 | 1990-1996 |
| | Winter | Quinault NFH | 5,208 | 1986 |
| Cook Creek | Winter | Quinault NFH | 9,386,258 | 1972-2022 |
| | Winter | Quinault R. | 1,221,513 | 1973-1988 |
| Dickey River | Winter | Unknown | 35,003 | 1972 |
| Educket Creek | Winter | Hoko R. H. | 14,003 | 2012 |
| | Winter | Quinault R. | 18,000 | 1989 |
| | Winter | Makah NFH | 493,756 | 1986-2011 |
| Goodman Creek | Winter | Bogachiel H. | 479,497 | 1981-2008 |
| | Winter | Quinault NFH | 16,359 | 1986 |
| Hoh River | Winter | Bogachiel H. | 5,428 | 1981 |

| Release Watershed | Run | Source | Number Released | Broodyears |
|---|---|---|---|---|
| | Winter | Quinault NFH | 1,299,800 | 1981-2009 |
| | Winter | Hoh R. | 125,704 | 1977-2012 |
| | Winter | Quinault R. | 512,229 | 1982-1988 |
| | Winter | Unknown | 77,868 | 1972-1981 |
| Hoko River | Winter | Bogachiel H. | 90,647 | 1981-1989 |
| | Winter | Hoko R. H. | 746,083 | 1987-2022 |
| | Winter | Makah NFH | 49,961 | 1986-2009 |
| | Fall | Hoko R. H. | 52,808 | 2009-2016 |
| Lyre River | Summer | Bogachiel H. | 219,973 | 1987-2008 |
| | Summer | Chehalis R. | 20,614 | 1983-1985 |
| | Summer | Sol Duc R. | 4,000 | 1981 |
| | Summer | Skamania H. | 16,945 | 1981-1986 |
| | Winter | Bogachiel H. | 524,619 | 1981-2004 |
| | Winter | Dungeness R. H. | 17,278 | 2005-2006 |
| | Winter | Elwha R. H. | 125,169 | 1990-2008 |
| | Winter | Quinault NFH | 20,677 | 1986 |
| Moclips River | Winter | Unknown | 35,032 | 1972 |
| Pysht River | Winter | Bogachiel H. | 218,283 | 1981-2003 |
| | Winter | Dungeness R. H. | 10,188 | 2005-2006 |
| | Winter | Elwha R. H. | 40,082 | 2005-2008 |
| | Winter | Hoko R. H. | 46,349 | 1990-1992 |
| | Winter | Quinault NFH | 10,302 | 1986 |
| Queets River | *Summer* | *Queets R. (N)* | *2,108* | *2000* |
| | Winter | Quinault NFH | 1,074,507 | 1989-2011 |
| | Winter | Queets R. | 42,945 | 1977-2002 |
| | Winter | Salmon R. FCF | 1,339,299 | 1978-2009 |
| | Winter | Quinault R x+ Queets R. | 184,683 | 1981-1996 |
| | Winter | Quinault R & Lk H. | 3,673,499 | 1978-2022 |
| | Winter | Quinault R & NFH | 189,626 | 1997-2010 |
| Quillayute River | Winter | Quinault R. | 58,810 | 1985-1986 |
| Quinault River | Winter | Quinault R. & NFH | 10,230,470 | 1972-2022 |
| | Winter | Queets R. | 154,914 | 2003 |
| | Winter | unknown | 27,402 | 1972 |
| Raft River | Summer | Quinault R. | 15,513 | 1979 |
| | Winter | Quinault NFH | 480,675 | 1978-1986 |
| | Winter | Quillayute R. | 238,000 | 1975 |
| | Winter | Eagle Creek NFH, OR | 109,314 | 1976 |
| Sail River | Summer | Hoko R. H. | 12,681 | 2009-2016 |

| Release Watershed | Run | Source | Number Released | Broodyears |
|---|---|---|---|---|
| | Winter | Bogachiel H. | 3,317 | 1989 |
| | Winter | Hoko R. H. | 213,282 | 1988-2018 |
| | Winter | Makah NFH | 85,346 | 1986-2009 |
| Sekiu River | Summer | Hoko R. H. | 21,352 | 2009-2016 |
| | Winter | Bogachiel H. | 5,016 | 1989 |
| | Winter | Hoko R. H. | 281,904 | 1988-2020 |
| | Winter | Makah NFH | 12,292 | 2009 |
| Sol Duc River | Summer | Bogachiel H. | 65,000 | 2001-2009 |
| | Summer | Chehalis R. | 74,178 | 1983-1985 |
| | Summer | Quillayute R. | 392,283 | 1987-2010 |
| | Summer | Skykomish R. | 14,300 | 1990 |
| | Summer | Sol Duc R. | 27,725 | 1981 |
| | Summer | Skamania H. | 42,531 | 1981-1986 |
| | Winter | Sol Duc R. | 394,670 | 1975-1993 |
| Sol Duc River | *Winter* | *Sol Duc R. (N)* | *1,035,388* | *1995-2020* |
| Tsoo-Yess | Winter | Quinault NFH | 197,652 | 1984-1989 |
| | Winter | Makah NFH | 4,836,734 | 1982-2022 |
| Village Creek | Winter | Bogachiel H. | 1,897 | 1989 |
| | Winter | Hoko R. H. | 37,267 | 1988-2013 |

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