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

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Contents

Figures

Tables

Plain Language Summary

Background

Steelhead are the ocean-migrating form of the species *Oncorhynchus mykiss*, with rainbow trout being the alternative form that lives only in freshwater. Like Pacific salmon, steelhead lay their eggs in streams. Juveniles rear in freshwater streams (normally for two years) and then migrate to the ocean for two or three more years before returning to spawn. Unlike Pacific salmon, steelhead do not necessarily die after spawning, and may return to the ocean for another year or more before spawning again.

Steelhead are found along the West Coast of the United States, from southern California to Alaska, and range into Asia as well. In the contiguous United States, steelhead are organized into distinct population segments (DPSes) based on similarities in their life-history characteristics, genetics, and the ecology of their rivers. The DPS is the unit that the National Marine Fisheries Service (NMFS) considers for listing as threatened or endangered under the Endangered Species Act. The steelhead of the Olympic Peninsula (OP) may constitute a DPS if they are markedly separate from other steelhead populations. Some OP steelhead begin their return migration to spawn in the summer, and others return in the winter in rivers west of the Elwha River and north of Grays Harbor. Winter-returning steelhead can be found in both small and large rivers in the Olympic Peninsula, while summer-returning steelhead are mostly found in larger rivers. NMFS previously reviewed the status of OP steelhead in the 1990s and determined at that time that the DPS was not at risk of becoming threatened or endangered.

In August of 2022, NMFS was petitioned by the Conservation Angler and the Wild Fish Conservancy to reevaluate the status of OP steelhead, pointing out that a) the numbers of summer-returning steelhead had declined considerably since the last review, b) large numbers of winter-returning steelhead were being caught each year, making the current number of steelhead much smaller than historically was found in the rivers, and c) large numbers of steelhead that originally came from rivers outside of the Olympic Peninsula were being released every year from hatcheries. NMFS decided that these issues justified a reexamination of the status of OP steelhead. NMFS formed a status review team (SRT) to gather and review all available information to answer several guiding questions:

- Is the current steelhead composition of the DPS accurate, and if not, what populations of steelhead need to be added or removed from the DPS?
- How many steelhead are currently present in winter- and summer-returning populations, and how have these numbers changed since the 1990s?
- What are the threats that these steelhead face, and how likely are they to contribute to the risk of extinction?
- Based on the best available information, are OP steelhead at risk of extinction now or in the near future?
- Is the risk of extinction the same for all populations, or are some groups of OP steelhead at a greater risk than the DPS as a whole?

Key Takeaways

The SRT gathered information on the numbers of fish caught and estimates of the numbers that survived to spawn. Much of this information came from federal, state, and tribal biologists. Information on the current environmental condition of the land and rivers in the range of OP steelhead was also considered, including predictions of the effects of climate change on the region. After reviewing and discussing the information gathered, the SRT concluded:

- There was no new information suggesting that changes were needed in the composition of the DPS.
- There had been a strong decline in the number of steelhead returning to the rivers of the Olympic Peninsula.
- Until recently, a large fraction of returning steelhead had been harvested before they could spawn.
- Information on summer-returning steelhead was very limited, but what was available indicated that there are very few remaining in the DPS.
- Hatchery fish, because they did not originally come from the DPS, could put OP steelhead at risk if they spawn with local fish, because genes not adapted for the OP environment could be introduced into OP steelhead. Also, fishers trying to catch hatchery fish may be catching a lot of local native fish.
- Habitat conditions were good in the Olympic National Park. Outside of the Park, however, there has been intensive tree cutting, and much of the forest and river habitat is still slowly recovering. Climate change is likely to result in summertime river temperatures being too warm, with river flows diminished. Winter snowfall will transition to rain in the future, often in the form of major rainfall events. The loss of glaciers in the Olympic Mountains is already proceeding at a rapid rate, and it is likely that they will be gone completely by the end of this century.

The majority of SRT members concluded that the OP Steelhead DPS was at a moderate risk of extinction.

Links used in this section:

- Steelhead: https://www.fisheries.noaa.gov/species/steelhead-trout
- Distinct population segments, Endangered Species Act: https://www.fisheries.noaa.gov/topic/laws-policies/endangered-species-act
- Petitioned: https://www.fisheries.noaa.gov/action/90-day-finding-petition-list-olympic-peninsula-steelhead-threatened-or-endangered-distinct
- Threats that these steelhead face: https://www.fisheries.noaa.gov/species/pacific-salmon-and-steelhead/esa-protected-species
- Climate change: https://www.fisheries.noaa.gov/topic/climate-change/understanding-the-impacts

Executive Summary

In response to a petition to the Secretary of Commerce to list the Olympic Peninsula (OP) Steelhead (*Oncorhynchus mykiss*) Distinct Population Segment (DPS) as a threatened or endangered species under the Endangered Species Act (ESA), the National Marine Fisheries Service (NMFS) convened a biological status 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), and evaluate whether the DPS 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 OP 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 OP 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 DPSes for winter-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 DPSes and ESUs 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.

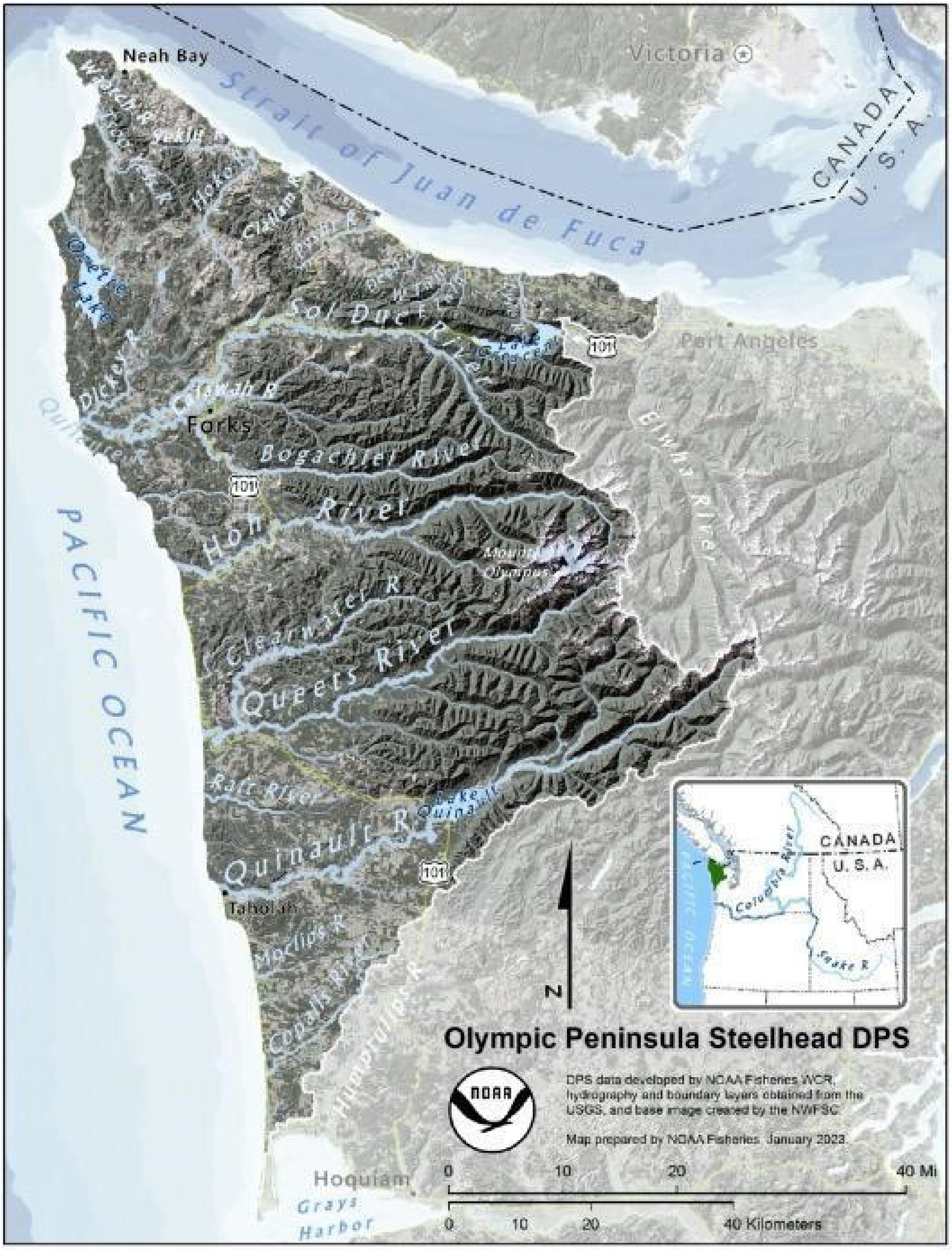

Figure 1. Olympic Peninsula Steelhead Distinct Population Segment, as identified in Busby et al. (1996).

Demographic Risk Analysis

NMFS previously reviewed the coastwide status of steelhead (anadromous *Oncorhynchus mykiss*) in 1996, and at that time identified 15 distinct population segments within the contiguous United States, including the OP Steelhead DPS.[1] It was the conclusion of the Status Review Team (SRT) at the time that the OP Steelhead DPS was not at risk of extinction then or in the foreseeable future (Busby et al. 1996).

Analysis of data relevant to the status of the OP Steelhead DPS was limited by the varying levels of data quantity and quality for each of the 39 steelhead populations (29 winter-, ten summer-run) identified in the Salmon and Steelhead Stock Inventory (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 counts[2]). Even where redd surveys were undertaken, only redds created after 15 March were included in the natural spawner abundance estimates provided by the co-managers. The use of the 15 March 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- 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- and native-origin steelhead spawning naturally. Overall, the SRT concluded that there was evidence for substantial overlap between returning hatchery and native winter-run steelhead, and that ,contrary to management intent, nonselective 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 hatchery and native steelhead have continued to interbreed, although the necessary genetic studies to evaluate this have not been undertaken. Currently, there is no direct harvest of native summer-

[1] 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).

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

run fish in the OP Steelhead DPS, although there is a fishery for hatchery-origin summer-run fish on the Quillayute River. The SRT was unable to establish from the harvest data provided by co-managers whether there was bycatch 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 River 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-year "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, 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 VSP categories of 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 effects of the ESA factors for decline (threats): habitat loss and destruction, overutilization, disease and predation, inadequacy of existing regulatory mechanisms, hatchery effects, and 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. 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 as a whole. In doing this, the team followed advice from 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 Service and NOAA SPOIR policy (USOFR 2014) and subsequent legal rulings.

Population VSP evaluation

In their evaluation of the VSP parameters for all populations within the DPS, the unweighted averages were *moderate* for abundance (2.2), productivity (2.9), and diversity (2.3), and *low* for spatial structure (1.3). The scores for winter-run steelhead populations from the four major coastal tributaries [the "Big Four"][3] 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 populations' VSP categories after concluding that there was insufficient information to make an informed score.

Population threats evaluation

The evaluation of threats (scores out of 5) for all populations identified climate change (3.1), inadequate regulation (2.9), and overutilization (2.5) as the top threats. Habitat loss or destruction (2.1), hatchery effects (2.1), and disease/predation (1.1) were ranked as lesser threats to the DPS. Climate change was unanimously 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 is that three of these threats (inadequate regulation, overutilization, and hatchery effects) to steelhead viability in the OP Steelhead DPS could be directly addressed through management and operational changes.

[3] The Quillayute, Hoh, Queets, and Quinault Rivers.

OP Steelhead 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 to this DPS, the SRT considered not only the current status, but how the status has changed since the last review by Busby et al. (1996), and concluded that the OP Steelhead DPS is at *moderate* risk of extinction because:

1. Escapement has declined in most populations since the previous status review (1996). In 1996, of the 12 populations for which trends could be calculated (1991–95), seven were found to be declining and five increasing (Busby et al. 1996). Currently, in contrast, of the 14 populations for which five-year trends could be calculated (2018–22), no populations were increasing, one was stable, and 13 were declining (ten of which had trends that were significantly different from zero).
2. Run size (escapement + harvest), which is only available for winter-run populations in the Big Four rivers, has declined by 42%, from 32,556 (1991–95, the time of the previous status review) to 18,821 (2018–22).
3. Kelt survival rates went from ~20% to ~12% since 1996 (in the Big Four 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 River 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 (2020–22), harvest rates were lower (~10% on average in the four 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 to 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 had 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). 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 observed in run sizes were in spite of the moderate-to-good conditions the SRT noted in river and riparian habitat, especially those rivers with substantial portions 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; 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 yet 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 points 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 Distinct Population Segment was at a *moderate* risk of extinction.

OP Steelhead DPS SPOIR 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 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- (stream-maturing) or winter-run (ocean-maturing) life histories. In deciding upon the significance of each portion, the majority of the SRT members placed the majority of their likelihood points in the *not significant* category for summer-run steelhead populations, and in the *significant* category 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 drain to the Strait of Juan de Fuca, and 2) steelhead population in rivers that drain to the Pacific Ocean. These two regions were identified as potential portions due to the hydrological and geographic distinctiveness or the rivers supporting Strait and coastal populations. The majority of the SRT members assigned the majority of their likelihood points in *not significant* 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 and coastal steelhead population portions identified as *significant*, the risk of extinction was determined not to be higher than that of the entire DPS.

In summary, the OP Steelhead DPS 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 SPOIRs based on run timing and biogeography, and concluded that none of the significant portions was at a higher risk of extinction than the overall DPS, and therefore, *no change in risk status was prescribed*.

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

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 (OP) Steelhead (*Oncorhynchus mykiss*) Distinct Population Segment (DPS) as a threatened or endangered species under the Endangered Species Act (ESA; TCA and WFC 2022).

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

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

The Petition does not request a reevaluation of the definition of the OP 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 Petitioners assert that there are 30 steelhead populations (26 winter- and four summer-run) in the OP Steelhead DPS. Recent abundance information was presented for approximately half (15) of the populations, all of which were winter-run. Of those populations, only 20% (three) 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 fish co-occurring with "early-returning" hatchery-origin fish.[1] 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 in late autumn and early winter.

[1] 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 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 (USOFR 2023).

In response to the petition, on 6 January 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 the OP 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 WCR.
4. Complete the extinction risk synthesis.
5. Depending on the outcome of (4), conduct a significant portion of its range (SPOIR) analysis and evaluate whether the DPS is at moderate or high risk of extinction in a significant portion of its range.

The Northwest Fisheries Science Center (NWFSC) convened a status review team (SRT) in 2023, with scientists from NWFSC, the Southwest Fisheries Science Center (SWFSC), WCR, and the 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 OP steelhead, past and current harvest and hatchery operations, watershed habitat conditions, past and present fisheries and land-use regulations, and 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 its 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 OP Steelhead DPS or with treaty/management interests within the DPS. In addition, there were presentations by other state and federal agencies and nongovernmental entities. This report presents the information reviewed and analyzed by the SRT, as well as the process by which it made its DPS configuration and risk determinations.

OP Steelhead DPS Configuration

NMFS's DPS Policy

The ESA allows listing of species, subspecies, and distinct population segments (DPSes) 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 *evolutionarily significant units* (ESUs) for defining listable units under the ESA. This concept was adopted by NMFS in applying the ESA to anadromous salmonid species (USOFR 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 (USOFR 2006b). This change was initiated because steelhead are jointly administered with USFWS, and USFWS does not use the ESU policy in its listing decisions (USOFR 2006b). Under the joint USFWS and NMFS DPS policy, a group of organisms is a DPS if it is both "significant" and "discrete" 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" (p. 4). 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 USOFR 1996b 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 DPSes), 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 OP Steelhead DPS

The Olympic Peninsula Steelhead DPS was established in 1996 (USOFR 1996a), 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). The DPS includes Water Resource Inventory Areas (WRIAs) 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 U.S. EPA Ecoregion III Coast Range (#1) to the North Cascades (#77), and their basins include several Level IV Ecoregions (Figure 2). The OP Steelhead DPS was further characterized by habitat, climatic, and zoogeographical characteristics that distinguish it from its neighboring DPSes (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). 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, pp. 59–60) 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

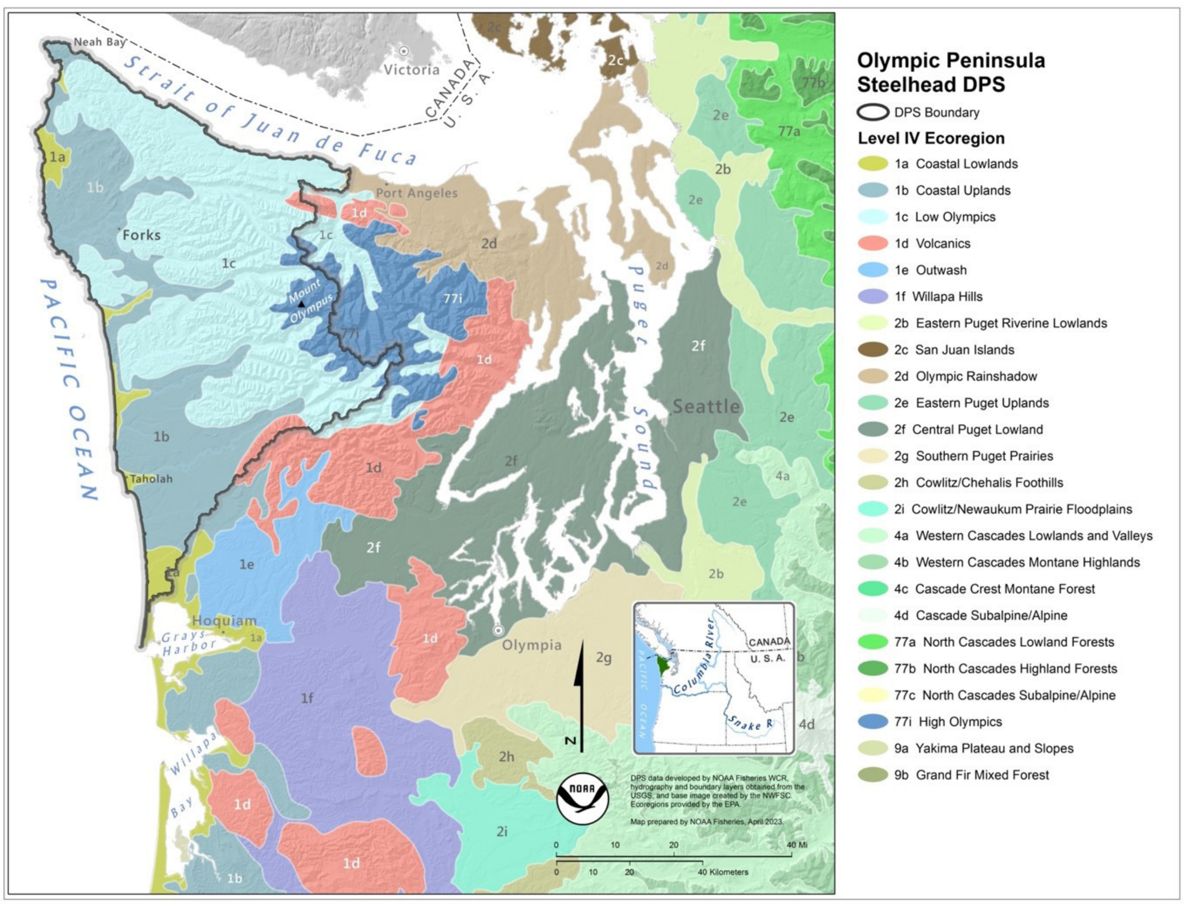

Figure 2. U.S. EPA Level IV Ecoregions. 1 = Coast Range: 1a = Coastal Lowlands, 1b = Coastal Uplands, 1c = Low Olympics, 1d = Volcanics. 77 = North Cascades: 77i = High Olympics (Pater et al. 1998). Ecoregions identify areas with distinct climatic, geologic, and vegetative characteristics.

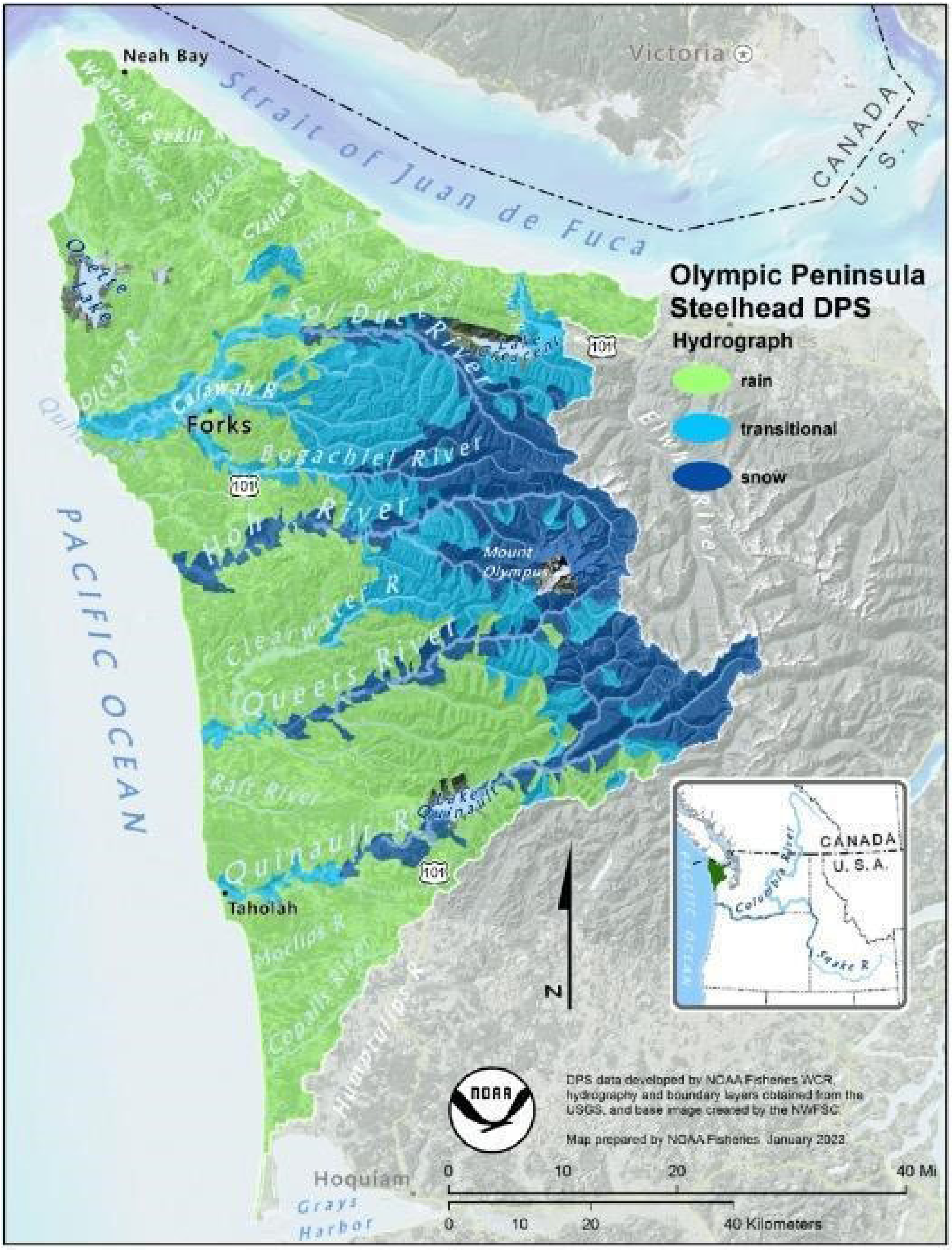

Figure 3. Hydrographic regions within the OP Steelhead DPS.

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) [Figures 3 and 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).

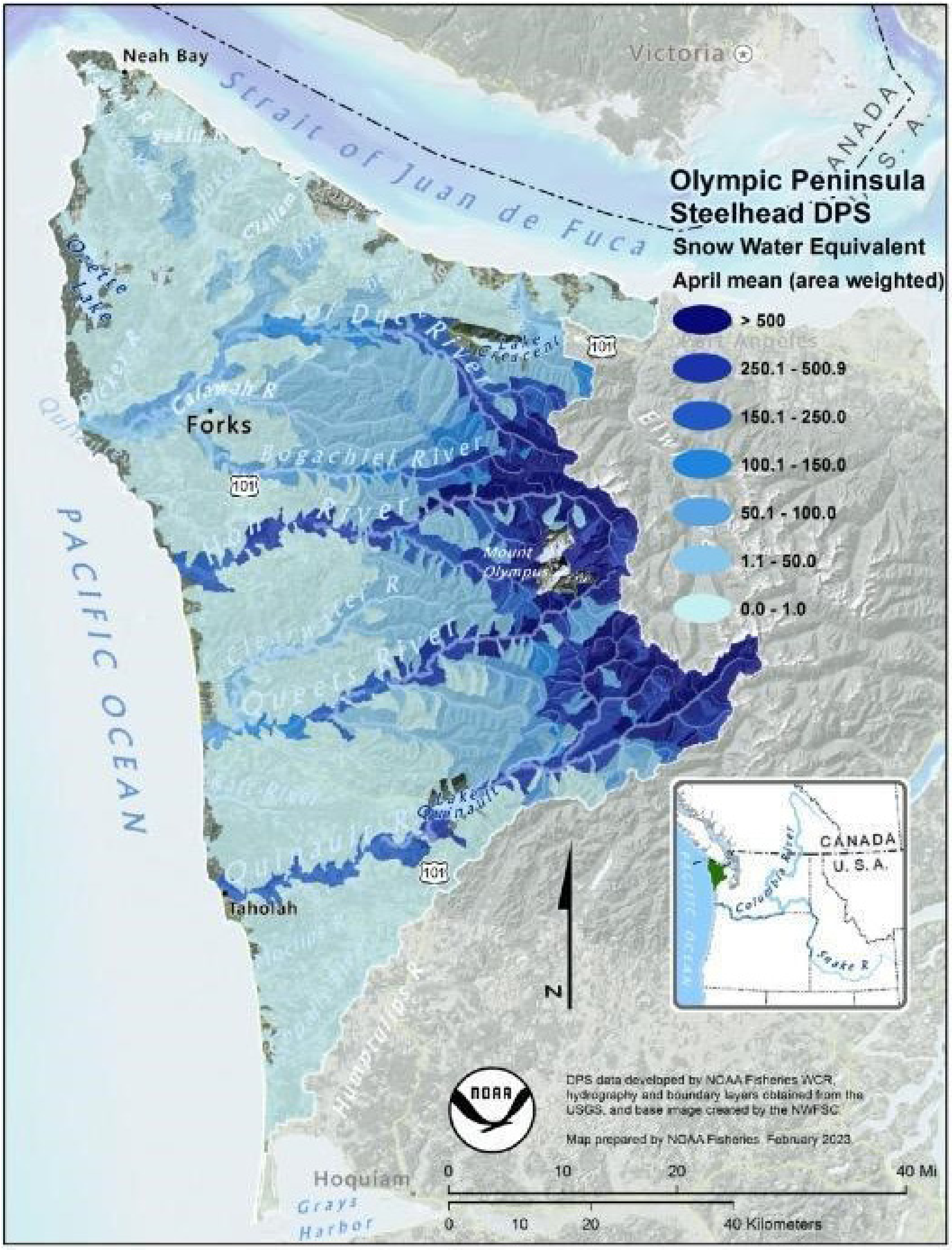

Figure 4. Average April mean snow-water equivalent regions in the OP Steelhead DPS.

In describing the OP 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, 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 (*Oncorhynchus kisutch*, Weitkamp et al. 1995), chum salmon (*O. keta*, Johnson et al. 1997), and Chinook salmon (*O. tshawytscha*, Myers et al. 1998). These other status reviews similarly relied on species-specific genetic and life-history data, as well as ecological conditions.

Biology of steelhead (anadromous *O. 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 enables *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, or 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 1 May to 31 October (WDG 1984).[2] 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 Contor 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 survive when 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 among males) reduces the

[2] Washington Department of Game, 1984 letter to all concerned parties, on summer steelhead harvest management for the Boldt Case area.

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, 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 rivers draining into the Pacific Ocean (Figure 6): Quillayute (Bogachiel, Sol Duc, Sitkum, and Calawah), Hoh (South Fork Hoh), Queets (mainstem, Clearwater), and Quinault (East Fork, North Fork, and mainstem; 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) whose 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 (Busby et al. 1996, Quinn et al. 2016). 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 et al. 2007). The summer-run strategy is observed in anadromous fish and is also known as premature migration (Quinn et al. 2016), so called because the summer-run adults migrate from the ocean to freshwater before sexual maturation, which is fundamentally distinct physiologically from the mature-first-

Table 1. Presumptive run and spawn timing for winter- and summer-run steelhead populations in the OP Steelhead DPS, based on WDF et al. (1993). Shaded areas indicate run timing (green = winter run, orange = summer run). *s* = spawning period, where no *s* is present spawn timing was designated as unknown. Water Resource Inventory Areas (WRIAs) are watershed areas defined by the Washington Department of Ecology.

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

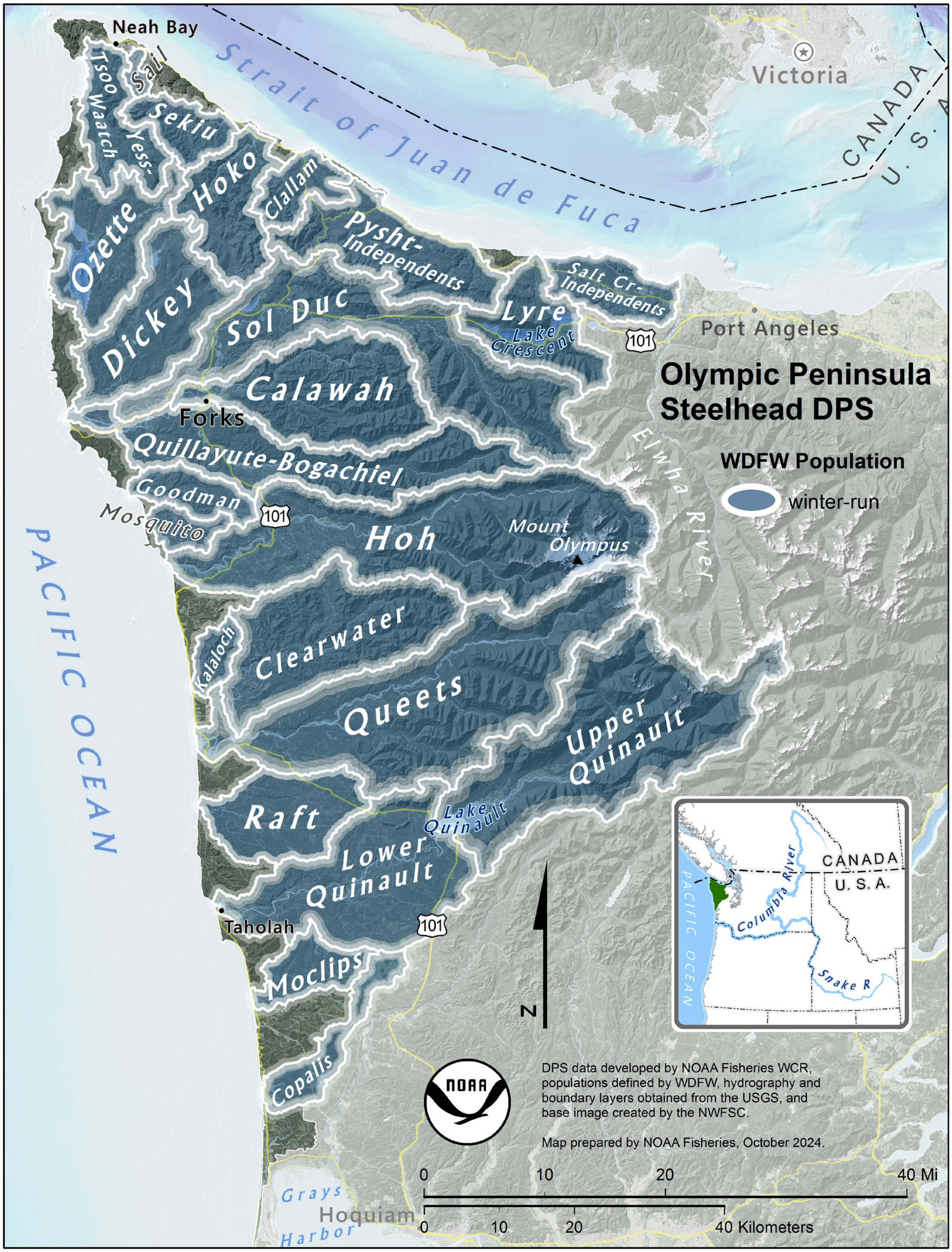

Figure 5. Winter-run steelhead populations in the OP Steelhead DPS. Based on presence and hydrographic basin.

Table 2. Presumptive populations of winter- (blue) and summer-run (red) steelhead in the OP Steelhead DPS, based on WDF et al. (1993), arranged by Water Resource Inventory Area (WRIA): east to west (WRIA 19), and north to south (WRIAs 20 and 21). Winter steelhead also occur in numerous smaller independent tributaries. Similarly, summer-run steelhead have been observed in the Hoko River, but it is unclear if it represents an independent population. Dickey, Sol Duc, Calawah, and Bogachiel Rivers are tributaries to Quillayute River.

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[a]	Winter			
			Pacific Coast	Goodman Creek	Winter			
			Pacific Coast	Mosquito Creek	Winter			
			Pacific Coast	Hoh River	Winter			
			Pacific Coast	Hoh River	Summer			

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

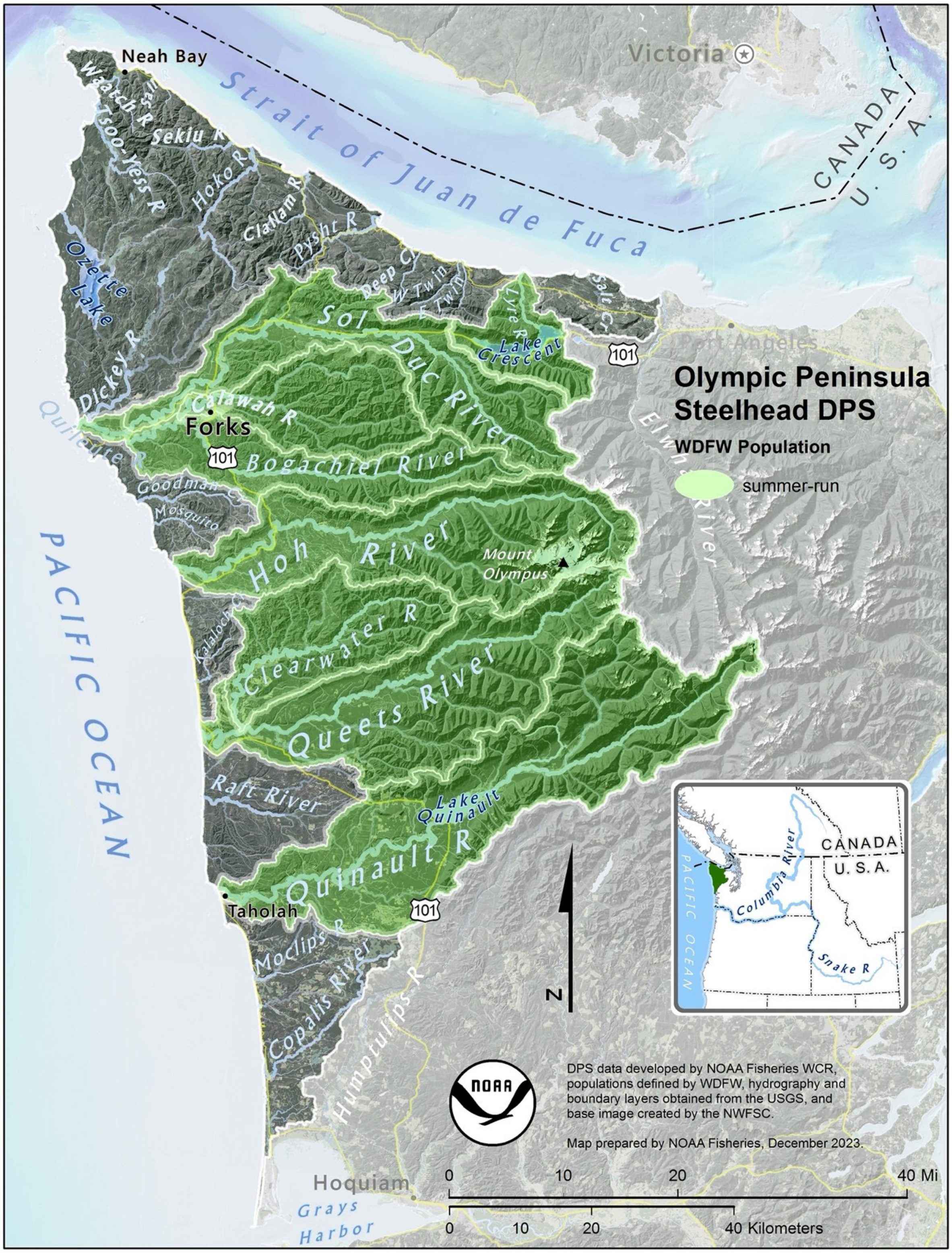

Figure 6. Summer-run steelhead populations in the OP Steelhead DPS. Based on presence and hydrographic basin.

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

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- (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 ten summer-run and 29 winter-run steelhead populations. Based on our assessments, steelhead from individual smaller independent streams may not constitute a demographically independent population (DIP) 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) or geography (the Lower and Upper Quinault Rivers). In some cases, the DIP was defined by coverage of the datasets 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).

Genetics

Genetic studies

There are a limited number of genetic studies that include steelhead samples from Olympic Peninsula watersheds 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 OP Steelhead DPS (Figures 7 and 8; Reisenbichler and Phelps 1989, Phelps et al. 1994). 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

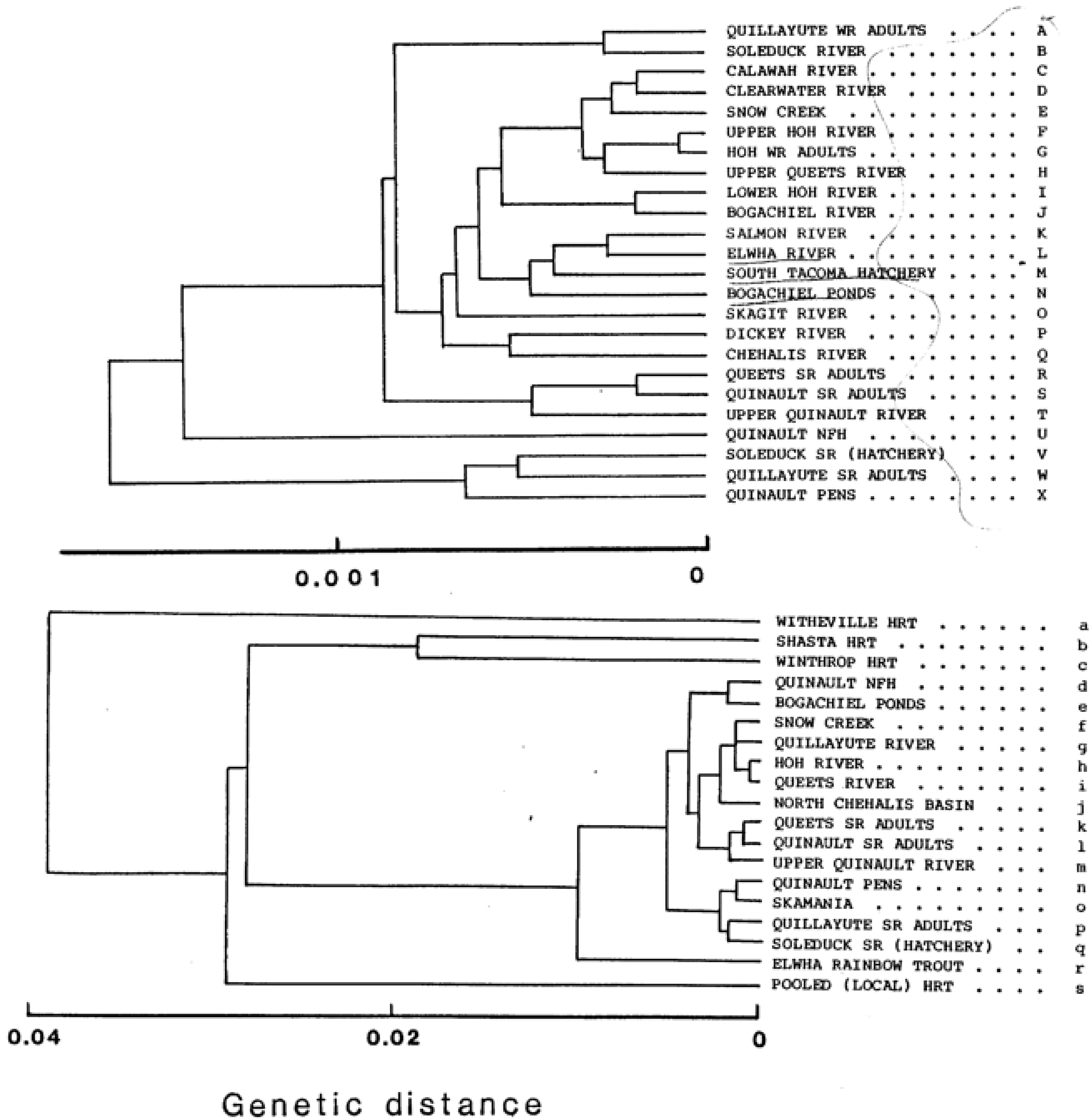

Figure 7. Dendrograms showing results of unweighted pair group method with arithmetic mean (UPGMA) analysis of genetic similarities among samples of steelhead collected (top) and hatchery stocks (bottom). Similarities are based on 19 variable loci. *WR* = winter run, *SR* = summer run, *NFH* = National Fish Hatchery, *HRT* = Hatchery Rainbow Trout. From Reisenbichler and Phelps (1989).

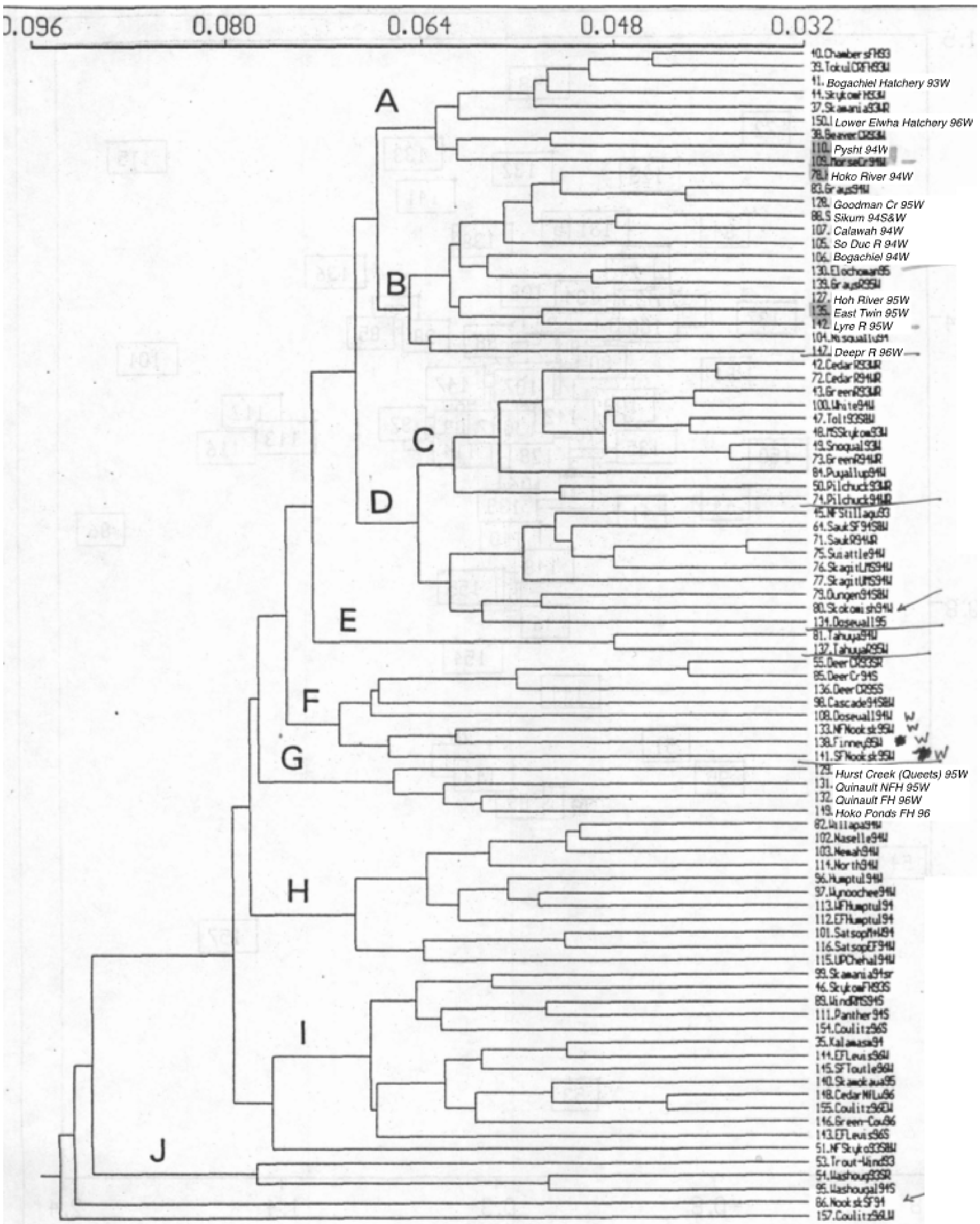

Figure 8. *O. mykiss* populations in Washington State. From Phelps et al. (1997). OP Steelhead DPS samples are in "clear" type.

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 Steelhead DPS and off-station releases appear to have influenced the genetic composition of winter steelhead collected from in-river sampling in many rivers (Figures 9 and 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 Steelhead 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 Steelhead DPS (Figures 7 and 8). Although the early genetic studies provided incomplete coverage of the 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 (Figures 9, 10, and 12). For the limited number of streams investigated, 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, 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 River 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 River basin) and Salmon River Tribal Hatchery (Queets River basin). A small portion (2.1%) of the steelhead sampled above the Highway 101 Bridge were assigned to the Skamania River summer-run hatchery collection.

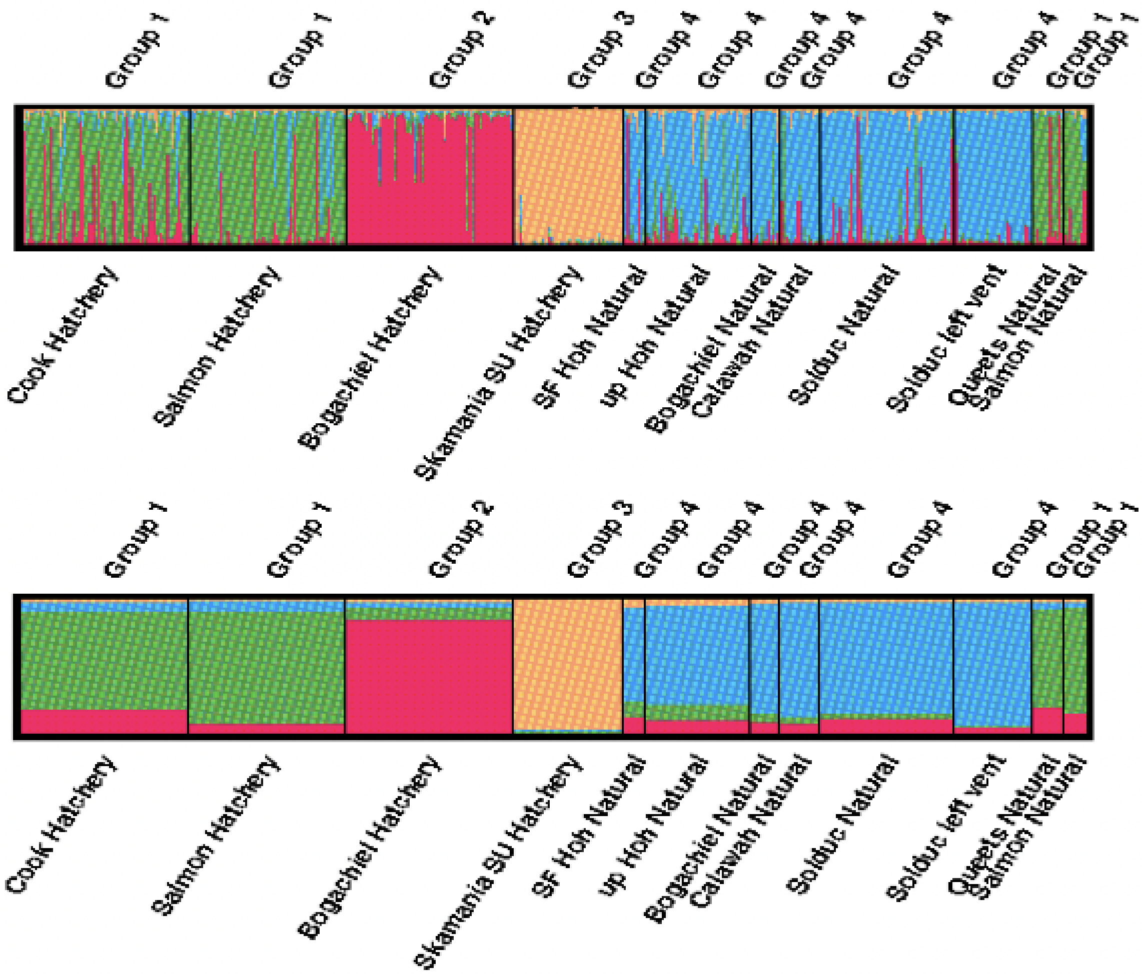

Figure 9. Structure plot showing percent membership of each individual steelhead (top) and the population average (bottom) into the groups that the Structure software found in the dataset. Individuals with more than one color in the bar likely have mixed ancestry. The group number identifies collections 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).

New genetic analysis and DPS configuration

In response to data requests by the SRT, the Northwest Indian Fisheries Commission (NWIFC) embarked on an updated analysis of all genetic data (single nucleotide polymorphism [SNP] markers) that have been collected to date on the OP Steelhead DPS *O. mykiss* (Seamons and Spidle 2023). Samples analyzed by Seamons and Spidle (2023) came 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 Steelhead DPS = 0.008). In particular, the major

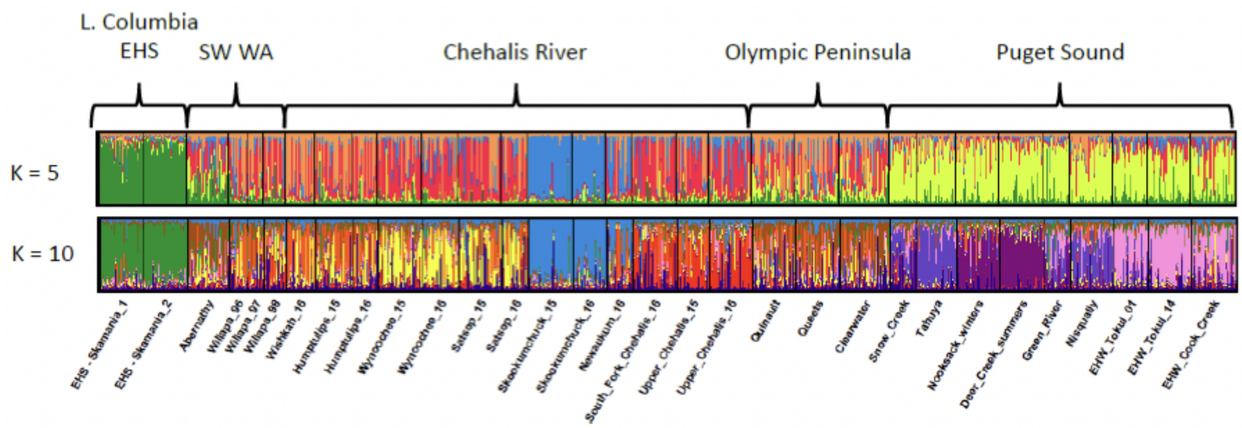

Figure 10. Plots of the results of a Structure analysis of coastal lineage *O. mykiss* collections, including Chehalis River collections at K (number of inferred clusters) = 5 and K = 10. The ΔK method of Evanno et al. (2005) supported K = 5, but the mean ln(K) plot supported K = 10, so both are shown. With K = 5, most of the Chehalis samples cluster with other Washington coast collections, with the Lower Columbia River hatchery early 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 four 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).

coastal streams—which have the best coverage of samples—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 Reisenbichler and Phelps (1985) and Phelps et al. (1997), which used allozymes.

Very few samples from within the OP Steelhead DPS have been analyzed for the small streams draining into the Strait of Juan de Fuca; only the Pysht and Lyre River collections 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 Steelhead DPS (pairwise $F_{ST} = 0.042$). More recent collections are needed to get a definitive understanding of the genetic differentiation among steelhead in the OP Steelhead DPS, and, in particular, the genetic differentiation in the Strait of Juan de Fuca between a) the streams to the west of the Elwha River, and b) the Elwha River and east (in the Puget Sound Steelhead DPS)—though there is clear differentiation between the OP Steelhead DPS and the Puget Sound Steelhead DPS overall ($F_{ST} = 0.026$; Figure 11). 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).

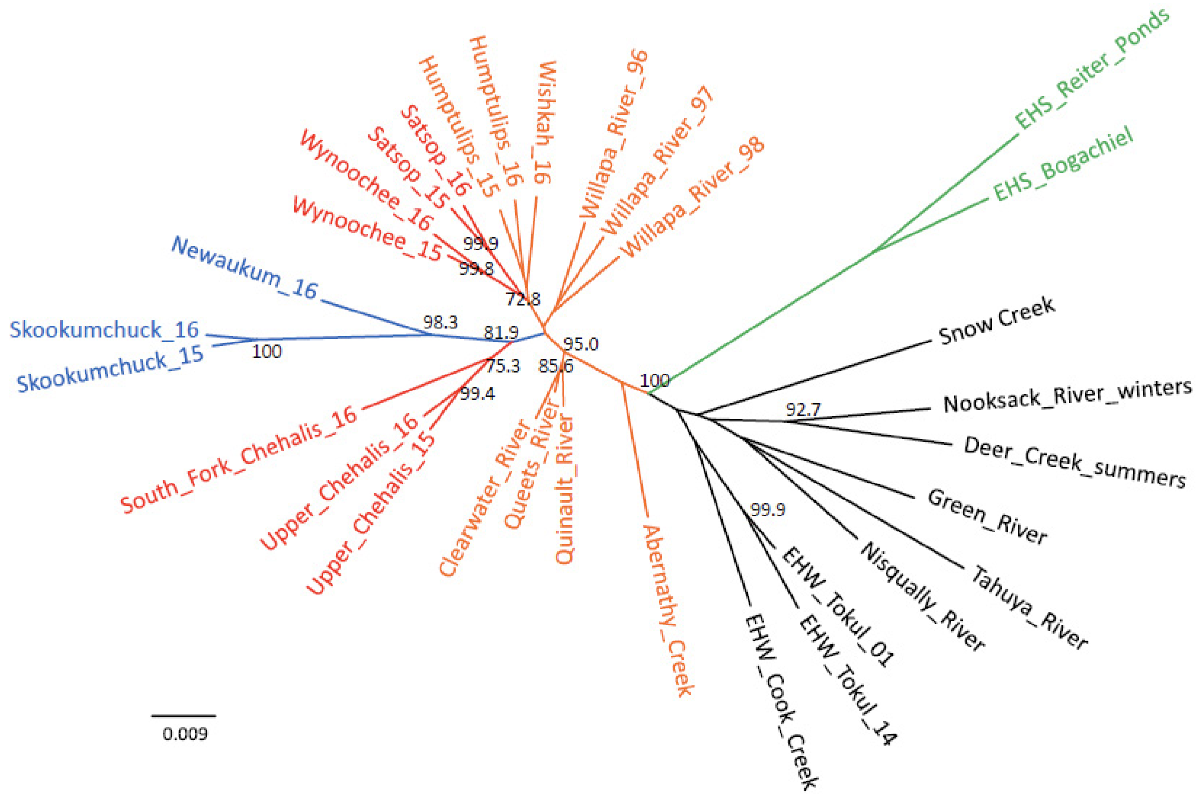

Figure 11. Unrooted neighbor-joining dendrogram constructed from a Cavalli–Sforza genetic distance matrix calculated using the Phylogeny Inference Package (PHYLIP; Felsenstein 1993). The dendrogram is color-coded to roughly match K = 5 of Figure 5: Lower Columbia River in green, Puget Sound in black, 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 and Willapa River collections from all other collections. Moderate to strong bootstrap support existed separating the Willapa from the Chehalis River collections. From Seamons et al. (2017).

Genetic information and life-history diversity

Busby et al. (1996) first highlighted the paucity of information on summer- 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 (Figures 12 and 13), which is not enough to evaluate genetic diversity and similarity to winter-run steelhead, or to evaluate specifically whether summer-run steelhead on the Olympic Peninsula 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, personal communication) 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.

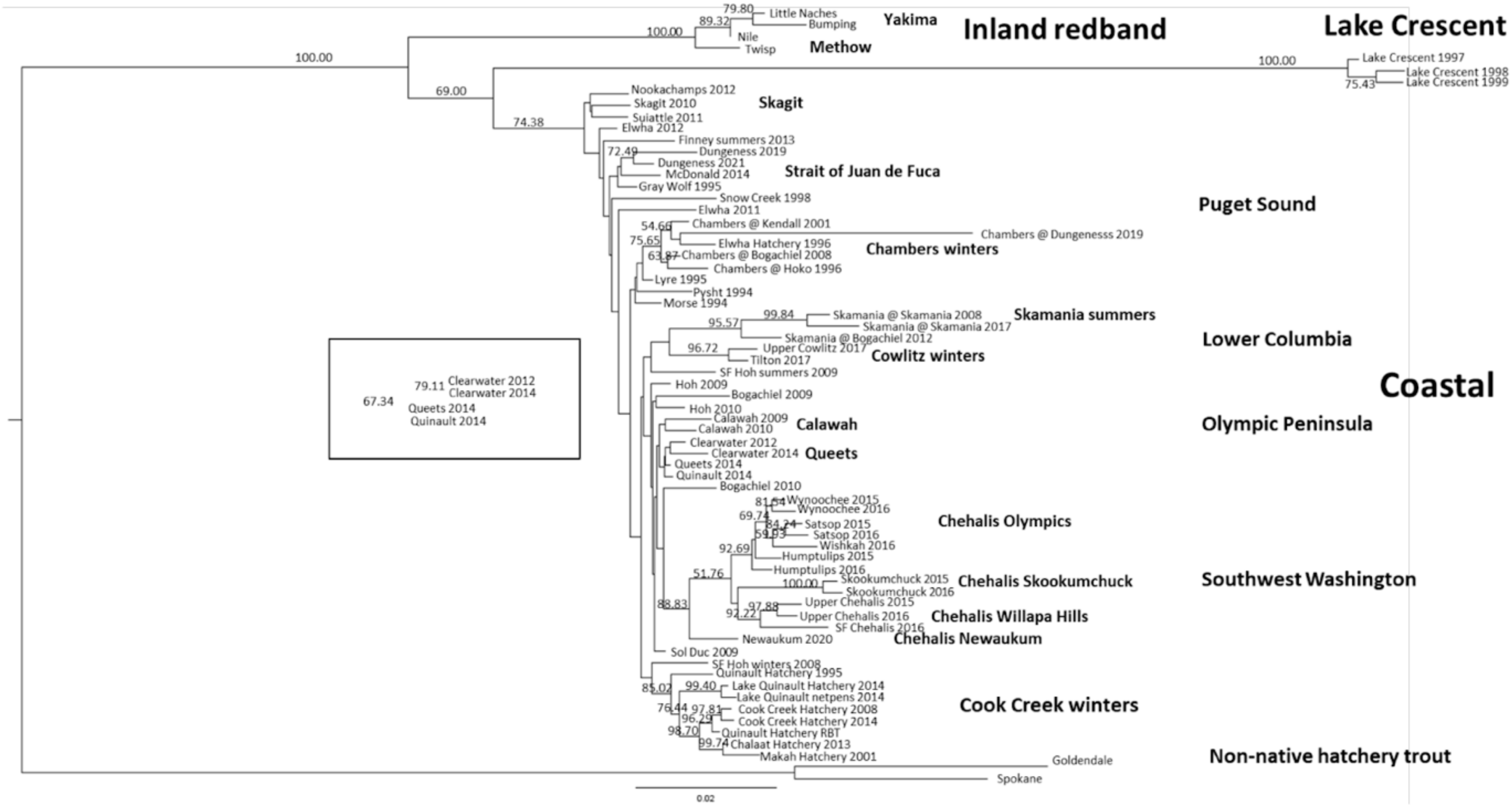

Figure 12. Unrooted neighbor-joining dendrogram based on pairwise Nei's genetic distance of native and non-native Washington *O. mykiss.* From Seamons and Spidle (2023).

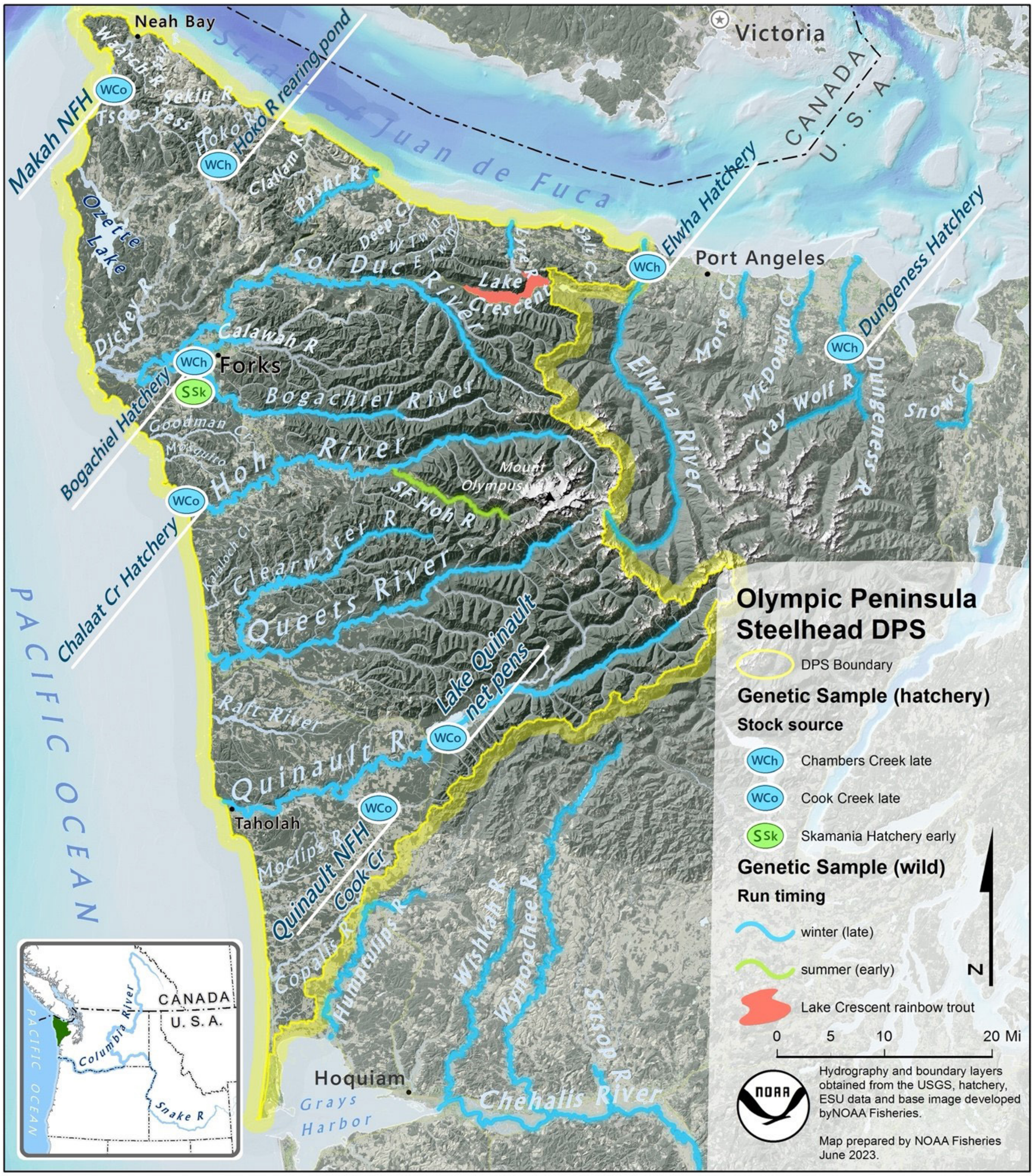

Figure 13. Olympic Peninsula steelhead collections (highlighted in blue or red) used for genetic analysis in Seamons and Spidle (2023). Collections were made from 1994–2021 and were genotyped with SNP markers.

Prior studies in *O. mykiss* in the southern portion of the species range have identified a major genome region (on chromosome 5) associated with migration and residency, but diversity at this region of the genome has not been examined in OP steelhead. 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).

Hybridization with coastal cutthroat trout

Hybridization between *O. mykiss* and *O. clarkii* can occur where the two species co-occur. Seamons and Spidle (2023) specifically evaluated evidence for hybridization with coastal cutthroat trout (*O. clarkii clarkii*) 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 > 3,000 *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 Steelhead 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.

Artificial propagation

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 OP Steelhead 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 Olympic Peninsula. Reisenbichler and Phelps (1989) used protein electrophoresis to evaluate allozyme variation in natural and hatchery-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 natural 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 were more similar to other natural-origin OP steelhead collections (Figure 12). 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-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 OP steelhead (Figure 12).

There is some evidence for hatchery influence on the native steelhead in OP streams in these historical collections. Individuals collected from the Pysht and Lyre Rivers (in 1994 and 1995, 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 hatchery summer steelhead (Kassler et al. 2010, 2011, Seamons and Spidle 2023).

Newer collections would be needed in the OP Steelhead 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 or modification of hatchery programs and releases that occurred relatively recently. Finally, the effective number of breeders calculated by Seamons and Spidle (2023) was in the hundreds to thousands (considering uncertainty in the estimates) for coastal OP naturally produced steelhead collections, but was 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* were not considered by the SRT for this report).

Summary: DPS boundary

The SRT considered new information and analyses relevant to the designation of the OP Steelhead 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 steelhead, with new collections to further evaluate the genetic relationships between streams within the DPS, and with the adjacent Puget Sound Steelhead 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 altering the current configuration was not justified. 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 Steelhead DPS.

OP Steelhead DPS Risk Analysis

Abundance and Productivity

Previous assessments

In an assessment of salmonid stocks (WDF et al. 1993), 31 stocks were identified within the OP Steelhead 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, Hoh, Queets, and Quinault Rivers) were of unknown status. Of the 12 independent steelhead populations (all winter-run) that Busby et al. (1996) reviewed, seven were found to be declining and five 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) of the 31 identified (48%) in the DPS. Of the 15 populations for which there were 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 were slightly declining or stable (Harbison et al. 2022; Figures 14–17). 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 (1991–95) to 18,821 (2018–22), a 42% decline.

Hatchery-origin steelhead survival

A recent study comparing the 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 (Figure 18). 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.

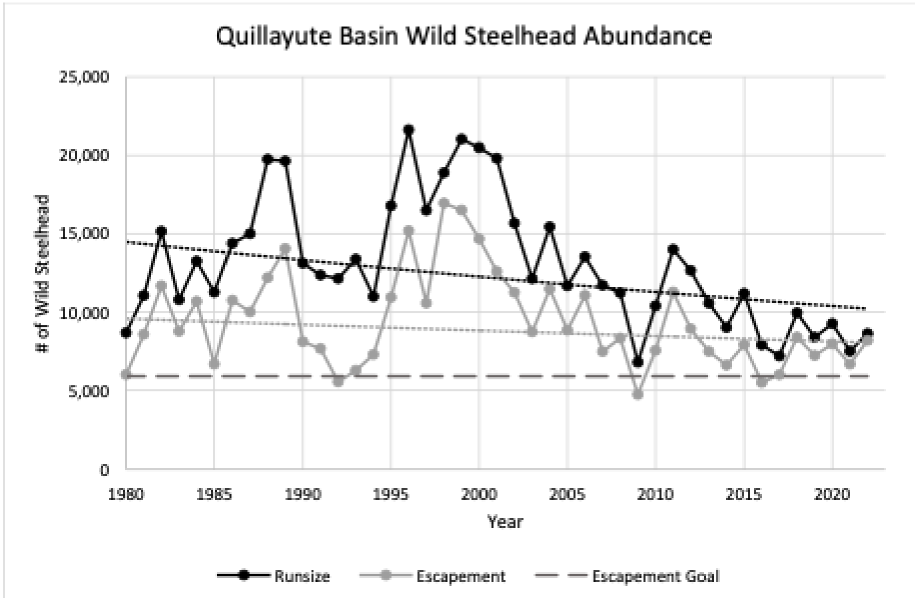

Figure 14. Quillayute River basin naturally produced steelhead run size and escapement from the 1979–80 and 2021–22 recreational steelhead fishery seasons, including the Dickey, Calawah, Bogachiel, and Sol Duc Rivers. The dashed line indicated the 5,900 steelhead escapement goal. The dotted curves show fitted exponential trends. From Harbison et al. (2022).

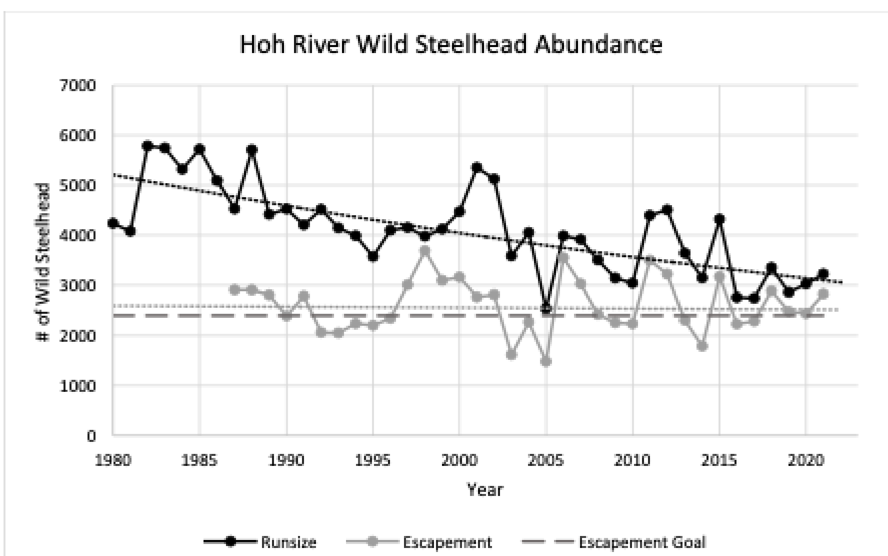

Figure 15. Hoh River naturally produced steelhead run size and escapement from 1979–80 to 2021–22. The dashed line indicates the WDFW 2,400 steelhead escapement goal. The dotted curves show fitted exponential trends. From Harbison et al. (2022).

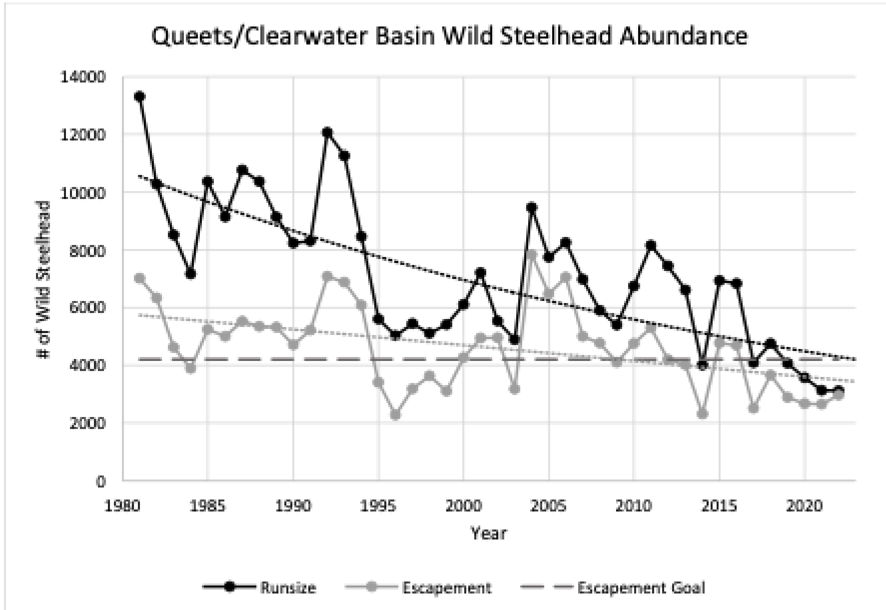

Figure 16. Queets/Clearwater River basin naturally produced steelhead run size and escapement between the 1980–81 and 2021–22 recreational steelhead fishery seasons. The dashed line indicates the 4,200 steelhead WDFW escapement goal; the tribal goal is 2,700. The dotted curves show fitted exponential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

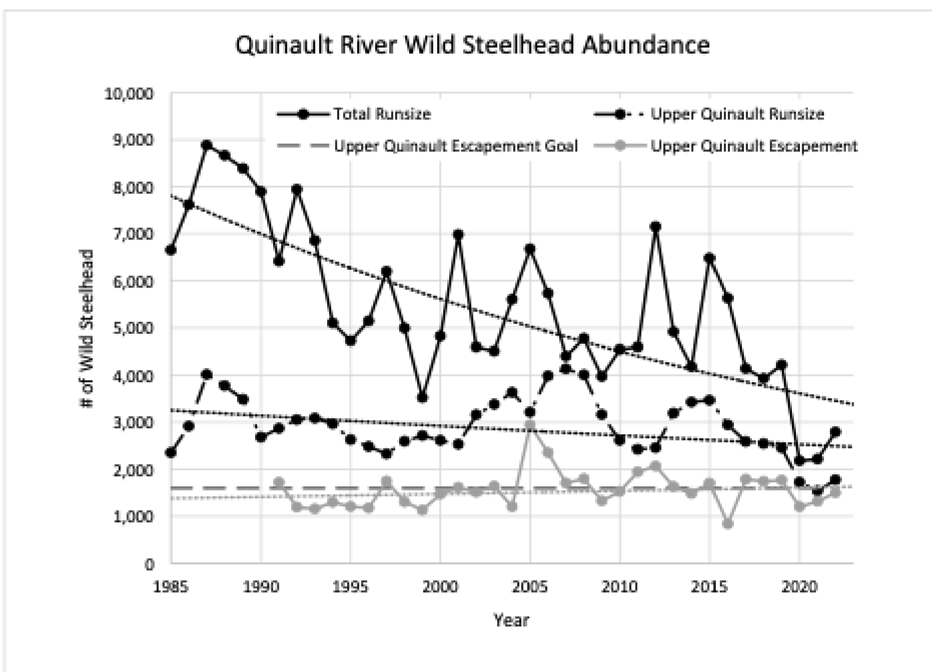

Figure 17. Total run size, Upper Quinault River run size, and Upper Quinault escapement from 1984–85 to 2021–22. The Upper and Lower Quinault River areas are separated because the State of Washington manages a recreational steelhead fishery in the upper river, while the Quinault Tribe manages steelhead 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).

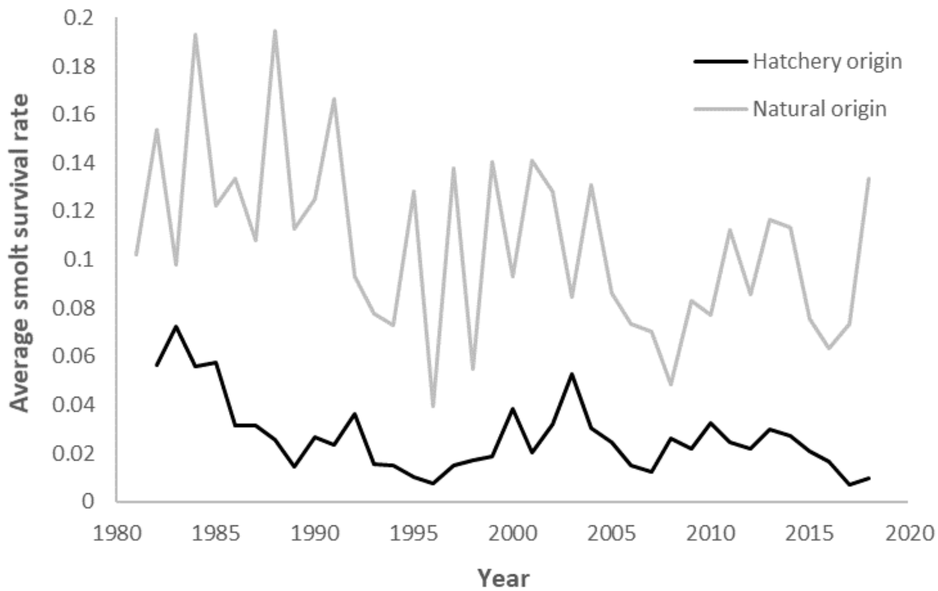

Figure 18. Average coastal Washington steelhead smolt survival rate for 13 hatchery-origin stocks and two natural-origin populations between the early 1980s and 2018. From Harbison et al. (2022).

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 (Table 3). 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 (Quinn 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.

Repeat spawner rate

In contrast to 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 broodyears 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). A 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.

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
19	Clallam River	W	0.7%	1999–2013
19	Pysht River/Independents	W	14.0%	1999–2013
19	Salt Creek/Independents	W	3.9%	1995–2013
20	Quillayute River System	W	29.6%	1978–2013
20	Goodman Creek	W	6.8%	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

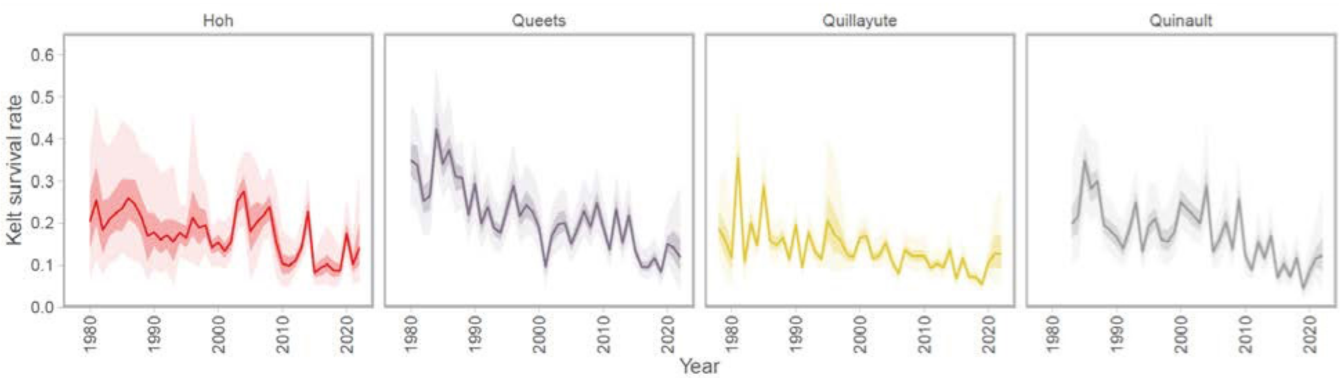

Figure 19. Estimated kelt survival rates by outmigration year for OP steelhead winter populations. Thick lines with dark and light bands are medians with 50% and 90% confidence intervals. From COPSWG (2023), their Figure 13.

Status Review Team Analyses for OP Steelhead

This section provides an overview of demographic data and trends for Olympic Peninsula winter-run steelhead based on the data provided by co-managers (COPSWG 2023). Escapement and catch time-series data are not available for summer-run steelhead, and their status is discussed separately. For some analyses, populations were grouped by geographic area (Figure 20).

Population and data description

Natural-origin and hatchery-origin steelhead contribution to escapement

"Wild" escapement[3] 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 15 March (COPSWG 2023). However, this varies by river (Table 4).

[3] "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).

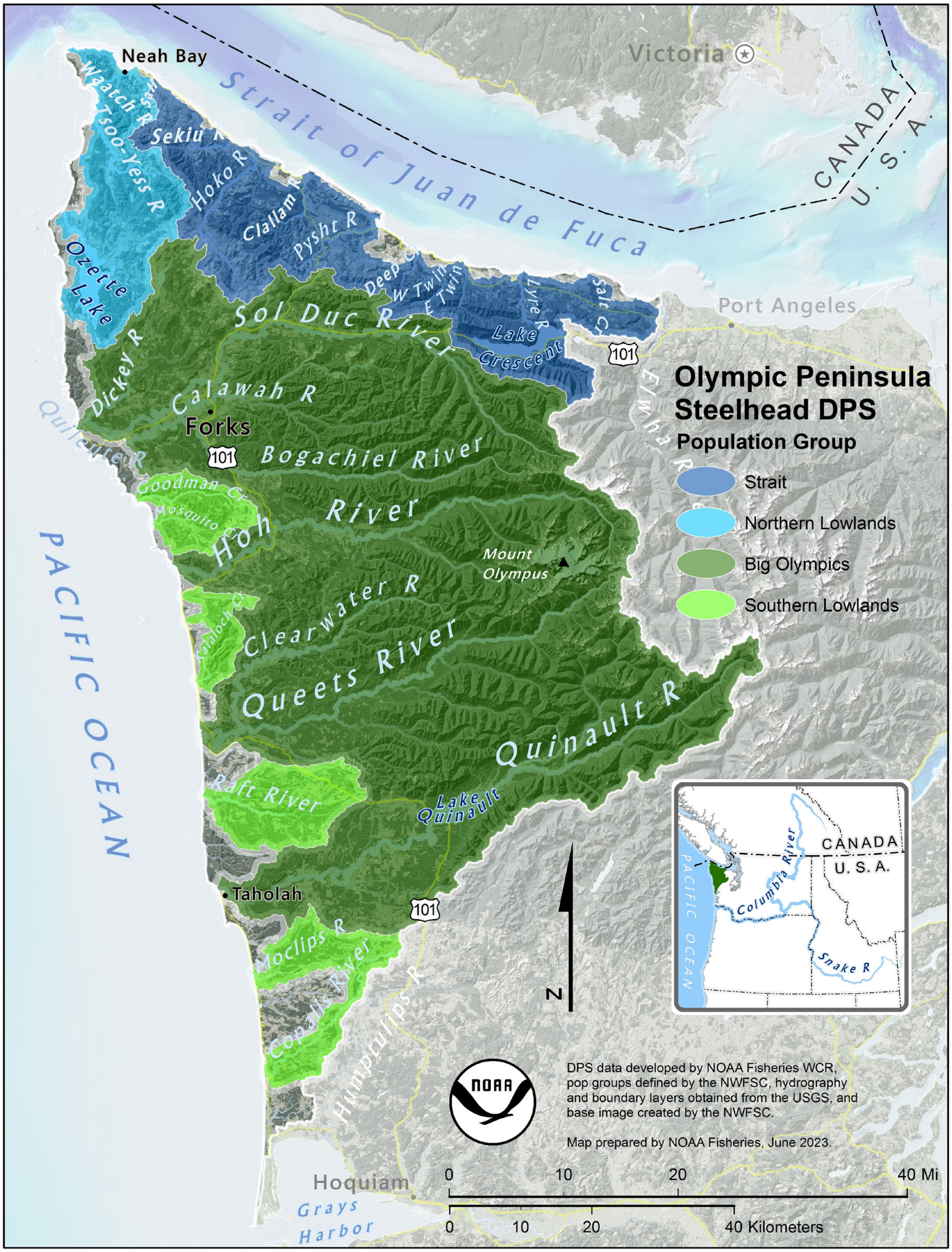

Figure 20. Map of OP steelhead geographic regions used in demographic analyses.

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 provided by co-managers. This is typically 15 March, but varies by region and year. The escapement goals are from the Salmon Conservation and Recovery Engine (SCoRE) public database. *EG* = escapement goal.

Population	**Start**	**End**	**Run**	**Cutoff**	**Goal**
Clallam River	1999	2022	Winter	after 15 March	No established goal
Deep Creek	1995	2022	Winter	after 15 March	
East Twin River	1995	2022	Winter	after 15 March	
Goodman Creek	1995	2022	Winter	after 15 March	Index EG = 206
Hoh River	1976	2022	Winter	after 15 March	EG = 2,400
Hoko/Little Rivers	1985	2022	Winter	after 15 March	
Moclips River	1988	2000	Winter	various dates in March	No established goal
Mosquito Creek	2016	2016	Winter	after 16 March	No established goal
Pysht/SF Pysht rivers	1984	2022	Winter	after 15 March	Index EG = 200, 103, 86[b]
Queets River (incl. Clearwater)	1980	2022	Winter	various dates in March	WDFW goal = 4,200
Quillayute:					
Bogachiel River	1978	2022	Winter[a]	after 15 March	EG = 1,127
Calawah River	2012	2012	Summer	n/a	No established goal
Calawah River	1978	2022	Winter[a]	after 15 March	EG = 1,740
Dickey River	1978	2022	Winter[a]	after 15 March	EG = 123
Sol Duc River	1978	2022	Winter[a]	after 15 March	EG = 2,910
Quinault River	1978	2022	Winter	various dates in March	Upper Quinault EG = 1,200
Salt Creek and tributaries	1995	2022	Winter	after 15 March	Index EG = 137
West Twin River	1995	2022	Winter	after 15 March	

[a] Assumed winter-run, but may include some summer-run as well.
[b] Escapement goal was reduced from 200 to 86 over the 1984–2022 period.

Escapement goals

Harvest and escapement levels of OP steelhead have largely been 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* (a.k.a. the Boldt Decision; 384 F. Supp. 312 [W.D. Wash. 1974]).[4] 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 1972). 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

[4] https://casetext.com/case/united-states-v-state-of-washington-wdwash-1974

rates" (proportion of the total population which may be harvested; Duda et al. 2018). The co-managers have established escapement goals for natural steelhead in several rivers of the OP Steelhead DPS (Table 5).

The river systems throughout the OP Steelhead 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 nonguided 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 5).

Trend analysis

Correlation structure

The correlation plot (Figure 21) shows how the escapement time series correlate across the rivers. Based on clustering, they fall into clusters of smaller systems with tributaries to the Quillayute River, with the other three large watershed systems (Hoh, Queets, and Quinault) being independent of one another.

Table 5. Escapement goals listed in the SCoRE database versus those in Busby et al. (1996) and from WDFW (R. Cooper, WDFW, personal communication) for the Strait of Juan de Fuca group of rivers. *QIN* = Quinault Indian Nation.

Population	WDFW (SCoRE)	WDFW (R. Cooper)	Busby – total	Busby – natural	Current goal
Moclips River			400	250	—
Quinault River			6,300	3,400	1,200[a]
Queets River			7,400	5,900	4,200 (WDFW) 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			

[a] Upper Quinault River.

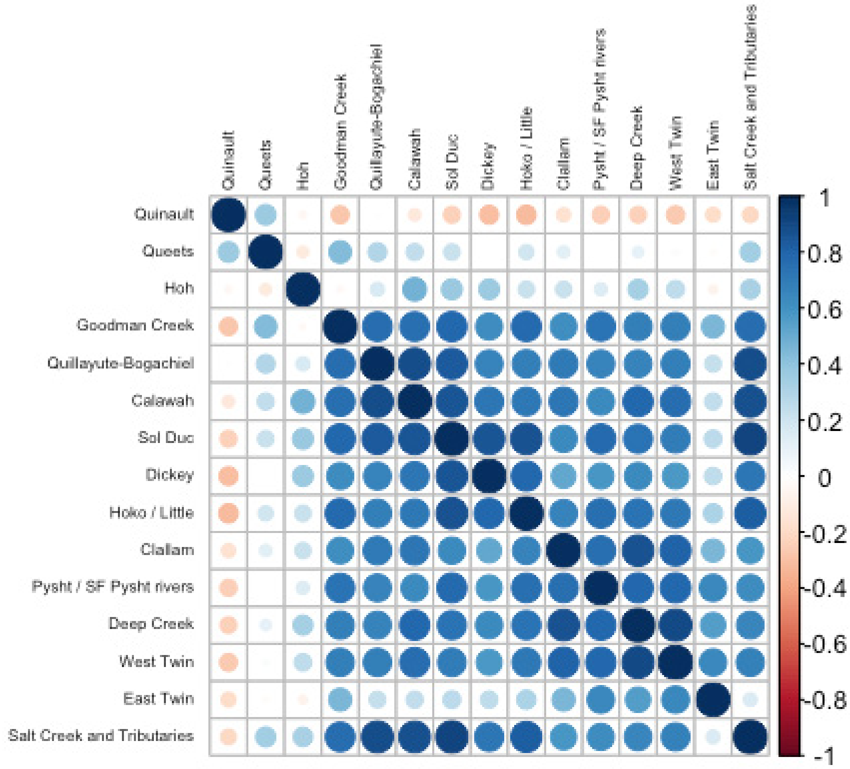

Figure 21. Correlation across the observed naturally produced escapement after the cutoff dates in March. In this plot, a clustering algorithm is applied that finds similar clusters. Populations are arranged south (top/left) to north (bottom/right).

Smoothed escapement estimates

To understand trends in the escapement of OP steelhead, we follow Ford (2022) and use multivariate dynamic linear models (DLMs) to estimate population-specific trends. The DLMs provide an estimate of the smoothed spawner counts after accounting for observation and process errors (see Ford [2022] and citations therein for details). For all component populations, we calculate smoothed time series of spawner abundances, geometric-mean abundances for each five-year window, and population trends over 15-year windows 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 (A. O. Shelton, personal communication) using the statistical software `stan` as implemented in the R computing language (R v.4.2.3, R Core Team 2022; `stan` v.2.26.22, Stan Development Team 2023).

We constructed a DLM using total escapement data for each river and separately for estimated total run size (escapement + 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 multivariate autoregressive state-space [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 13–30 March cut-off date (Table 4) is 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 9). We calculated geometric means for each five-year period for each population using output from the MARSS model (Tables 11 and 12)..

Escapement estimates

By population

These represent the DLM estimates from the data summarized (Table 4) 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).

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, McHenry et al. 1996). In most cases, summer-run steelhead were either not identified or their overall abundance and associated status and trends were categorized as "unknown" (WDF et al. 1993, McHenry et al. 1996). SASSI (WDF et al. 1993) did identify summer steelhead populations as being present in the Sol Duc, Bogachiel, and Calawah River 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 natural 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 Rivers, but reported no actual population estimates. SASSI (WDF et al. 1993) did not recognize summer-run steelhead in any of the watersheds 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

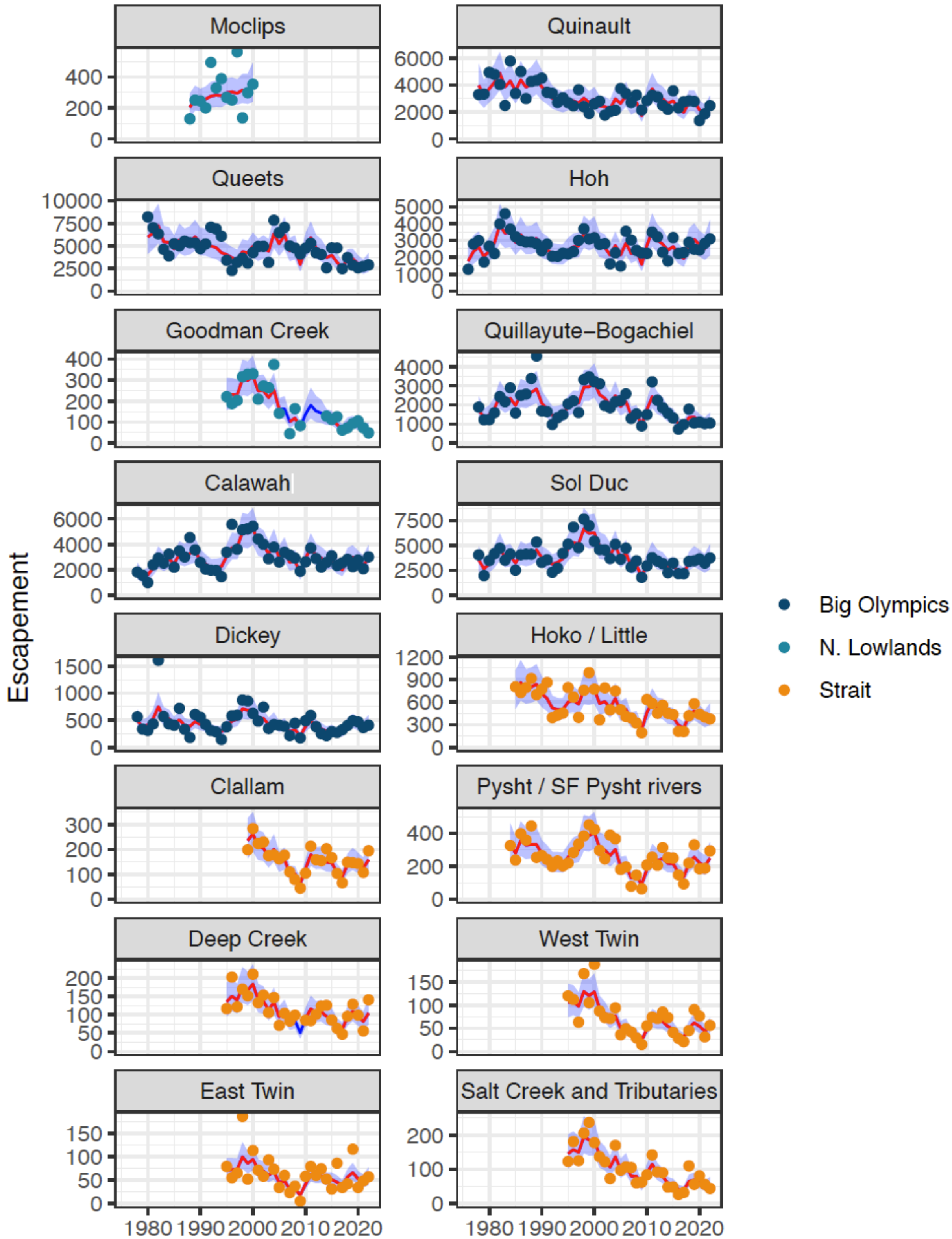

Figure 22. Total escapement after calendar cut-off (assumed primarily natural-origin) for winter-run populations in the OP Steelhead DPS. Points show observations, lines and shaded areas show model predictions of abundance and 95% CI. There was no information to determine hatchery contribution, so plots simply show total escapement after the cutoff dates (Table 4).

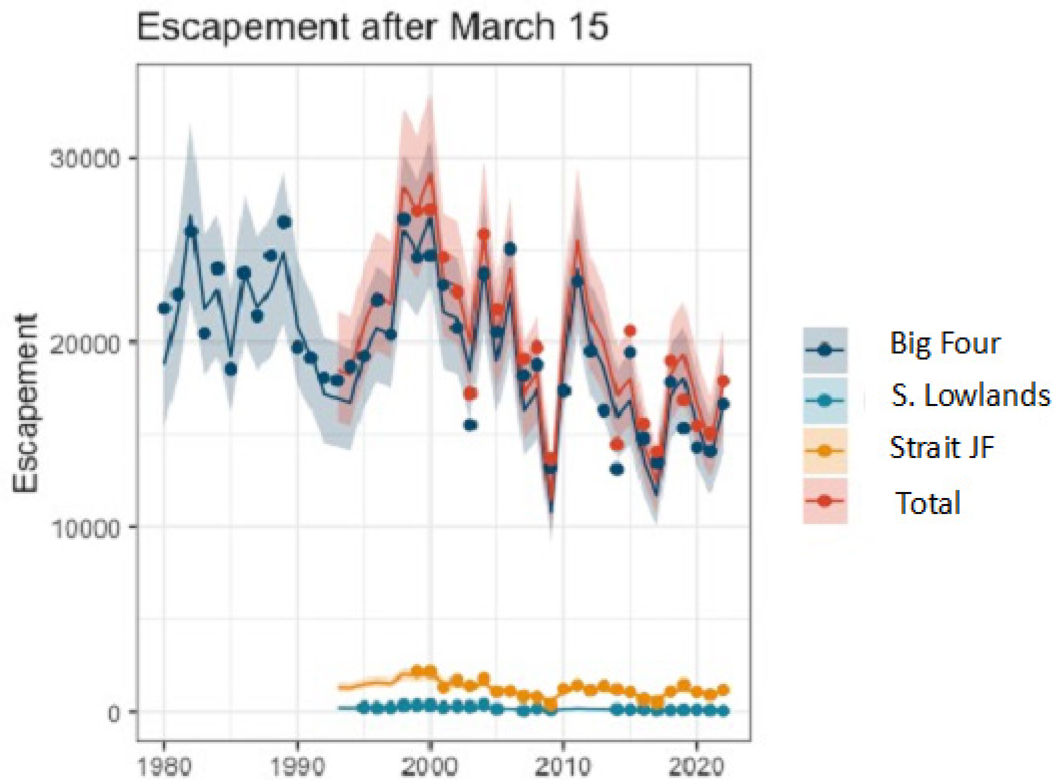

Figure 23. Escapement time series summed across all rivers and creeks for winter-run steelhead, 1993–2021 (excluding Moclips, which has no data after 2000). *Big Four*, *S. Lowlands*, and *Strait JF* escapements represent the aggregate sum of the smoothed estimates or populations within each area. *Total* 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.

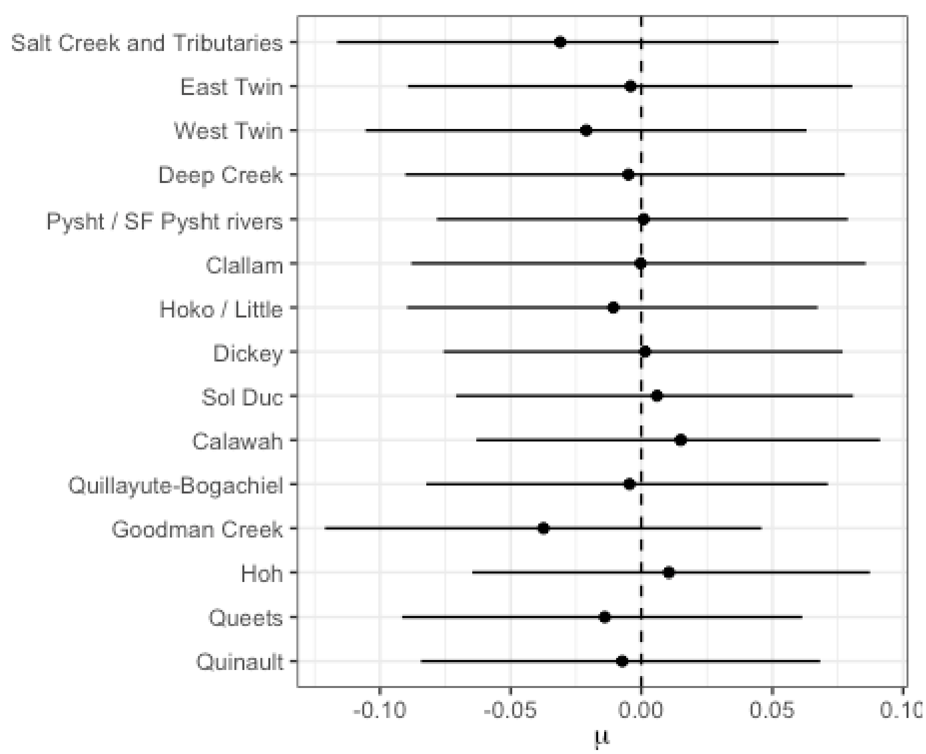

Figure 24. Estimated trend for each population over the time series estimated by the DLM.

the Clearwater, Hoh, Queets, Bogachiel, Quinault, and Sol Duc River 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 have been no spawner surveys 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).

Olympic National Park staff 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 2004–12 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 an inference about total breeding population size. Although not designed specifically to assess trends in summer steelhead, comparisons among years provide information on trends (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.

The counts (Table 6) show no consistent trends up or down, but do show a presence, albeit at consistently low numbers: almost always fewer than three holding adults per kilometer of stream channel, but occasionally as high as 5.6. These reference sites were used mostly by natural-origin summer steelhead, but hatchery-origin fish were observed in all except the Sol Duc River, and outnumbered the natural-origin fish in the South Fork Hoh River (Table 6).

More recently, snorkel surveys were conducted using a more spatially extensive "riverscape" approach. Scientists from Olympic National Park—as well as the U.S. Geological Survey (USGS), U.S. 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 7), while using snorkeling methods similar to the reference site surveys, covered entire

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	Natural	Unknown	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

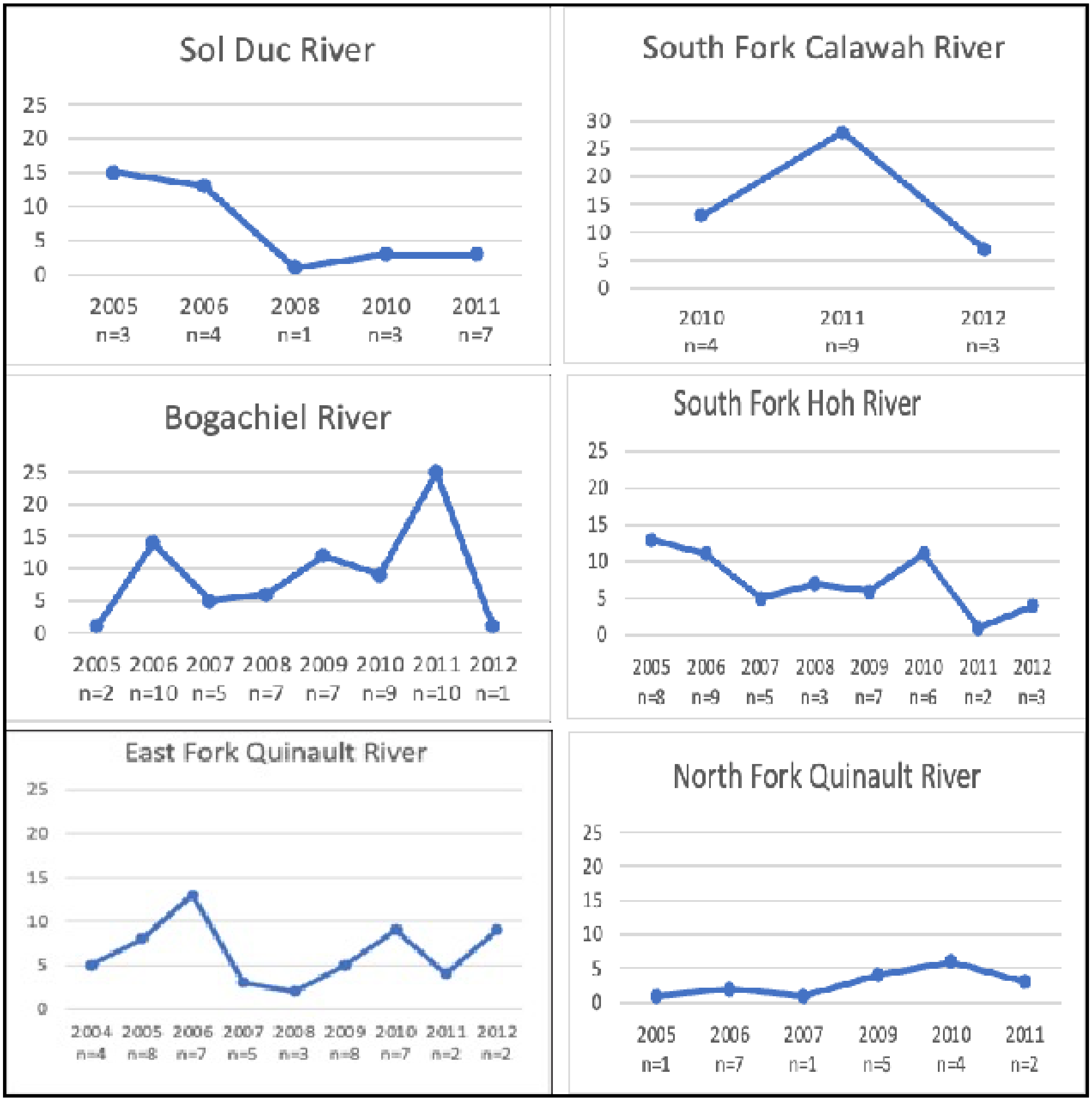

Figure 25. Annual peak counts of adult summer steelhead per 5 km, from snorkel surveys conducted during summer months at reference sites. Reference sites were in each of six rivers draining from Olympic National Park, with the annual number of repeat surveys (*n*) reported in the labels of the *x*-axis. Counts include both natural- and hatchery-origin adults (see Table 6). See Brenkman and Connolly (2008) for details.

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), and totals for surveys conducted by staff of ONP, WDFW, NOAA, USGS, USFWS, treaty tribes, and other project partners (ONP, unpublished data).

Survey coverage				Adult summer steelhead counts			
River	**Survey dates**	**Distance surveyed (rkm)**	**Spatial extent (rkm)**	**Hatchery-origin**	**Natural-origin**	**Unknown-origin**	**Total observed**
Bogachiel River	1–4 Aug 2016	0–55.6	6.2–45.1	4 (15%)	16 (62%)	6 (23%)	26
SF Hoh River	13–15 Sep 2016	0–22.3	1.2–22.3	3 (5%)	19 (33%)	35 (61%)	57
SF Hoh River	23 Sep 2003	0–21.0	n/a	33 (54%)	28 (46%)	0 (0%)	61
SF Hoh River	1 Oct 2002	0–21.0	n/a	21 (27%)	56 (73%)	0 (0%)	77
Quinault River	17–21 Aug 2009	L. Quinault 51.4	0.5–48.5	1 (1%)	108 (95%)	5 (4%)	114
Sol Duc River	18–21 Aug 2014	0–99.3	3.0–88.6	38 (26%)	55 (37%)	54 (37%)	147

river systems and obtained information on the spatial extent, relative abundance among rivers, and relative proportion of hatchery and natural summer steelhead for the larger rivers of the Olympic Peninsula (Brenkman et al. 2012, Duda et al. 2021).

Similar to the reference sites in the riverscape surveys, the rivers were used mostly by natural-origin summer steelhead. However, hatchery-origin fish were observed in all rivers, this time including the Sol Duc, and outnumbered the natural-origin fish in the South Fork Hoh River in one of the three years it was surveyed (Tables 6 and 7).

Assuming that observation probability was high (~1.0) and that unknown-origin fish had the same natural proportion as known-origin fish, breeding population size of the natural component was generally less than 120 summer steelhead per river, and often much less (Table 7). The average across rivers was 66 breeding fish per year, or roughly a breeding population size of ~260 per river assuming a four-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 natural 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 (e.g., 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 natural 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, natural- 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 river kilometer (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 River system in 1983, and there have been no reported hatchery outplantings of summer steelhead into the Quinault River 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 had been straying from unknown release locations to the Hoh, Queets, and Quinault Rivers 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, the risk is not negligible.

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 (Quillayute, Hoh, Queets, and Quinault). 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.

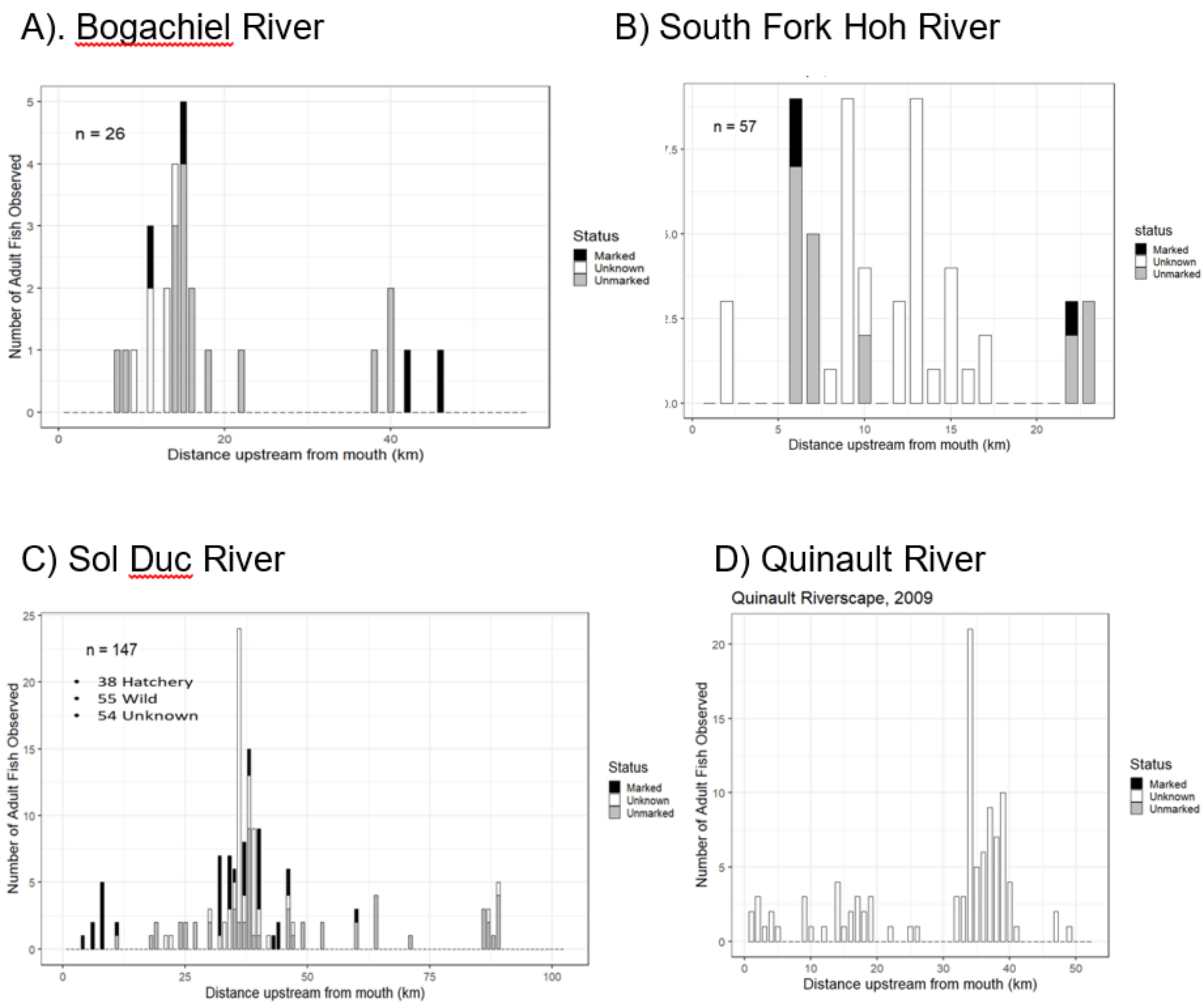

Figure 26. Distribution and relative abundance of adult summer steelhead counted in the continuous snorkel surveys (see Table 7). Longitudinal profiles of adult steelhead were plotted at 1-km spatial scale, indicated as bin lengths.

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, South Fork Calawah, Sitkum, and South Fork Hoh Rivers for Brenkman et al. [2012], Bogachiel, Sol Duc, South Fork Hoh, East Fork Quinault, and North Fork Quinault Rivers for McMillan [2022]). Mean estimates ranged from 3–303 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 3–43%. According to the Petitioners, McMillan and Gayeski (2006) estimated that summer steelhead abundance in the Queets and Clearwater Rivers 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 8). Their analysis utilizes both the index survey data from the fish assemblage data collected by ONP, the riverscape surveys efforts of ONP, and the Petitioners' efforts in the South Fork Calawah River to develop an estimate of both unclipped escapement and terminal run size (COPSWG 2023). The use of expanded index surveys generally introduces considerable uncertainty into population estimates, as reflected in the broad range for estimates (Table 8). 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-managers' estimates of naturally produced summer-run steelhead populations in the OP 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 five years of the 15-year window and two observations in the last five years of the window were required to report a trend estimate. This was to ensure that we did not report trend estimates when there were 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 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.

Pre- and post-Busby trends

In addition to the 15-year trends, the trends for 1977–94 corresponding to the years considered by Busby et al. (1996) were compared to the most recent trends (1995–2021; Figure 28, Table 10). 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 Big Four basins, where the majority of the DPS abundance lies. Differences between the pre- 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.

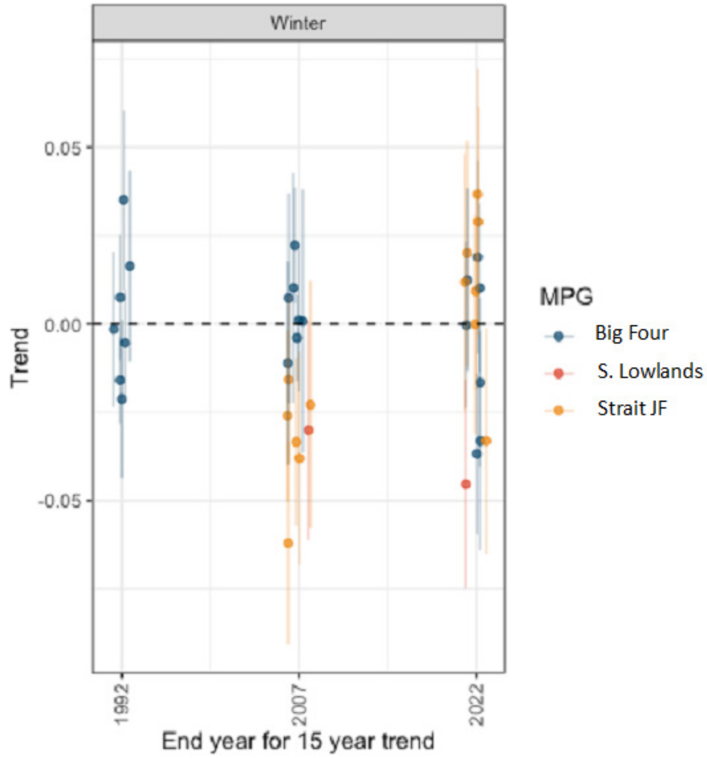

Figure 27. 15-year escapement trends estimated for winter-run stocks (total escapement after cutoff). Points show estimated trend through time and 95% CI for individual stocks. The end of the 15-year window is the year in the *x*-axis. Only 15-year windows where at least two observations (data points)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

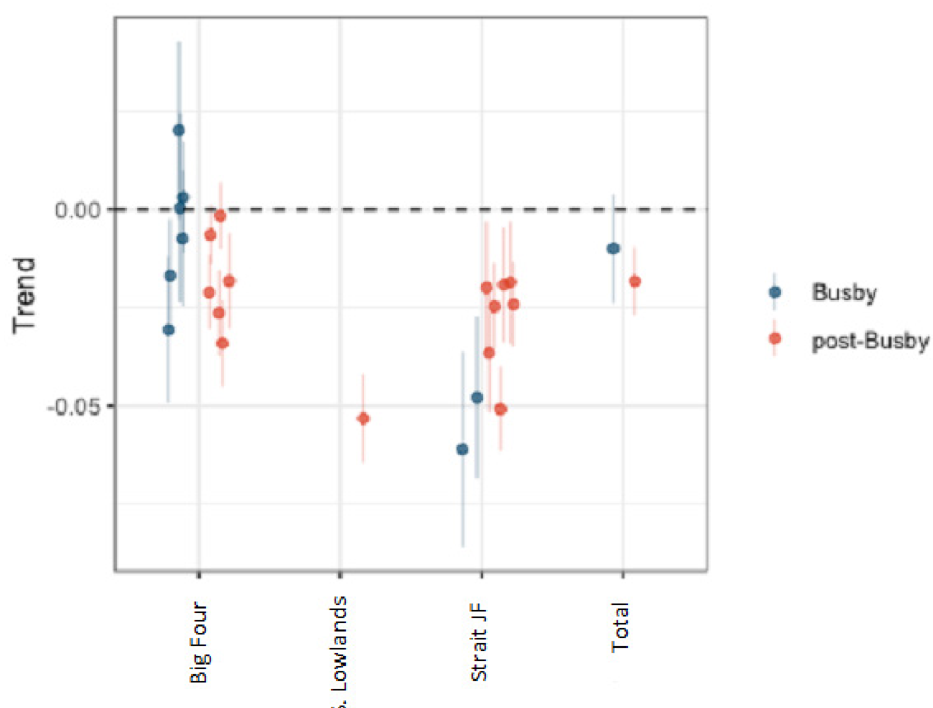

Figure 28. Escapement trends estimated for winter-run stocks (total escapement after cutoff) for the Busby (1977–94) and post-Busby (1995–2022) periods. Points show estimated trend and 95% CI.

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 two data points (observations, not estimates) in the first five years and last five years of the 15-year ranges are shown. Populations are ordered south to north.

Population	Area	1978–92	1993–2007	2008–22
Moclips River	Southern Lowlands			
Quinault River	Big Four	−0.01 (−0.02, 0.01)	0.00 (−0.02, 0.01)	−0.02 (−0.04, 0.01)
Queets River	Big Four	−0.02 (−0.03, 0.00)	0.02 (0.01, 0.04)	−0.04 (−0.06, −0.01)
Hoh River	Big Four	0.00 (−0.02, 0.02)	0.00 (−0.02, 0.02)	0.01 (−0.01, 0.03)
Goodman Creek	Southern Lowlands		−0.03 (−0.06, 0.00)	−0.05 (−0.07, −0.02)
Quillayute–Bogachiel River	Big Four	0.02 (−0.01, 0.04)	0.01 (−0.02, 0.04)	−0.03 (−0.06, 0.00)
Calawah River	Big Four	0.04 (0.01, 0.06)	0.01 (−0.02, 0.04)	0.00 (−0.02, 0.02)
Sol Duc River	Big Four	0.01 (−0.01, 0.03)	−0.01 (−0.04, 0.02)	0.01 (−0.01, 0.04)
Dickey River	Big Four	−0.02 (−0.04, 0.00)	0.00 (−0.04, 0.04)	0.02 (−0.01, 0.05)
Hoko/Little River	Strait		−0.02 (−0.04, 0.01)	0.00 (−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 (0.00, 0.06)
Deep Creek	Strait		−0.03 (−0.05, 0.00)	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 (0.00, 0.07)
Salt Creek and tributaries	Strait		−0.03 (−0.06, −0.01)	−0.03 (−0.07, 0.00)

Table 10. Trends in log total escapement for winter-run stocks in the Busby (1977–94) and post-Busby periods (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	Area	Year range	Busby et al.	Year range post	Post-Busby et al.
Moclips River	Southern Lowlands	1988–94	0.05 (0.02, 0.07)	1995–2000	0.03 (0.01, 0.05)
Quinault River	Big Four	1978–94	–0.02 (–0.03, 0.00)	1995–2022	–0.01 (–0.01, 0.00)
Hoh River	Big Four	1977–94	–0.01 (–0.02, 0.01)	1995–2022	0.00 (–0.01, 0.01)
Goodman Creek	Southern Lowlands	1977–94		1995–2022	–0.05 (–0.06, –0.04)
Quillayute–Bogachiel River	Big Four	1978–94	0.00 (–0.02, 0.02)	1995–2022	–0.03 (–0.04, –0.02)
Calawah River	Big Four	1978–94	0.02 (0.00, 0.04)	1995–2022	–0.02 (–0.03, –0.01)
Sol Duc River	Big Four	1978–94	0.00 (–0.01, 0.02)	1995–2022	–0.03 (–0.04, –0.02)
Dickey River	Big Four	1978–94	–0.03 (–0.05, –0.01)	1995–2022	–0.02 (–0.03, –0.01)
Hoko/Little River	Strait	1985–94	–0.06 (–0.09, –0.04)	1995–2022	–0.02 (–0.03, –0.01)
Clallam River	Strait	1977–94		1999–2022	–0.02 (–0.04, 0.00)
Pysht/SF Pysht Rivers	Strait	1984–94	–0.05 (–0.07, –0.03)	1995–2022	–0.02 (–0.03, 0.00)
Deep Creek	Strait	1977–94		1995–2022	–0.02 (–0.04, –0.01)
West Twin River	Strait	1977–94		1995–2022	–0.04 (–0.05, –0.02)
East Twin River	Strait	1977–94		1995–2022	–0.02 (–0.03, 0.00)
Salt Creek and tributaries	Strait	1977–94		1995–2022	–0.05 (–0.06, –0.04)
Aggregate	**Total**	**1977–94**	**–0.01 (–0.02, 0.00)**	**1995–2022**	**–0.02 (–0.03, –0.01)**

Means and Geomeans of Escapement

15-year mean

The DLM escapement estimates are used to calculate 15-year means. A minimum of two years in the first five years of the 15-year window and two years in the last five years 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.

Pre- and post-Busby abundance means

The mean of the estimated escapement (from DLM) for the pre- (1988–93) and post-Busby (2018–23) 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.

5-year geomeans

5-year geometric means were calculated using the observed escapement after the cutoff and using the estimated escapement from the DLM (Tables 11 and 12).

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 steelhead (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).

Population Growth and Harvest in Strait Populations

The DLM 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 13) and, from the DLM, we get 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.

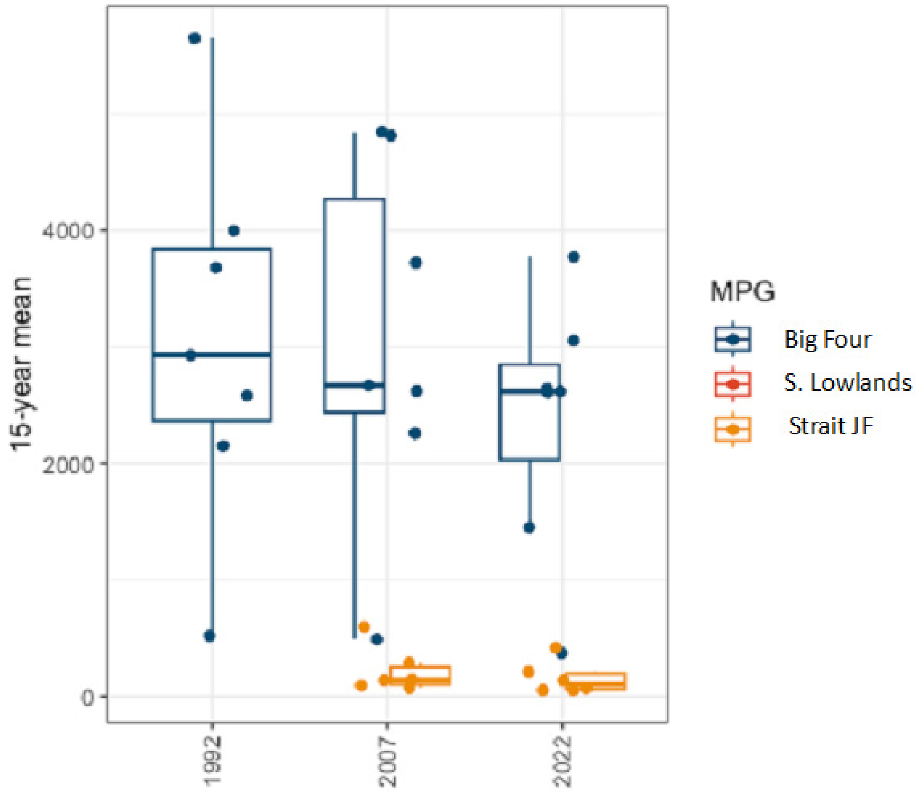

Figure 29. 15-year mean escapement estimated for winter-run stocks (total escapement after cutoff). Points show estimated mean for individual stocks for the 15-year period ending at the year in the *x*-axis. Only 15-year windows where at least two years are in the first five years and two years are in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

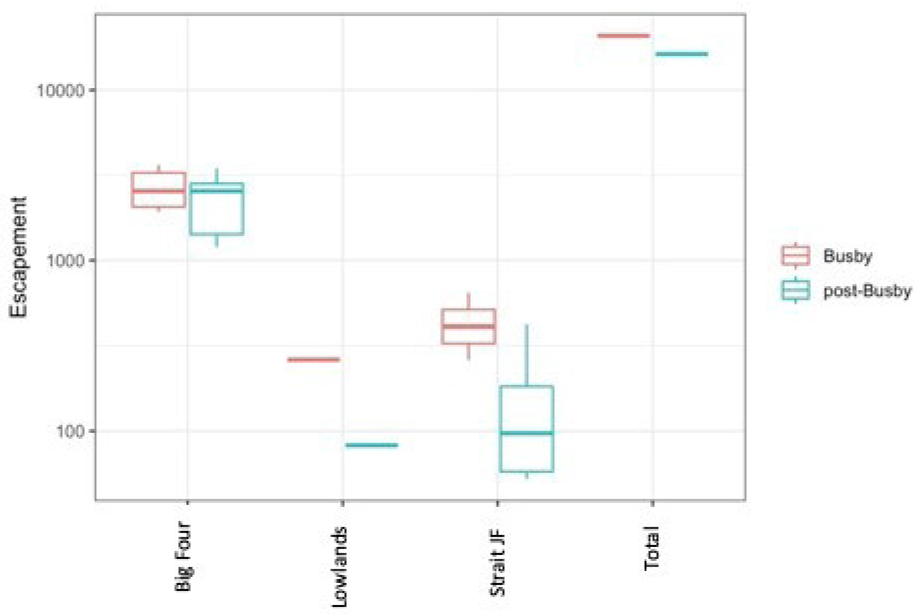

Figure 30. Mean escapement estimated for winter-run stocks (total escapement after 15 March cutoff) for the pre- (1989–93) and post-Busby (2018–23) periods. The *y*-axis is on log10 scale.

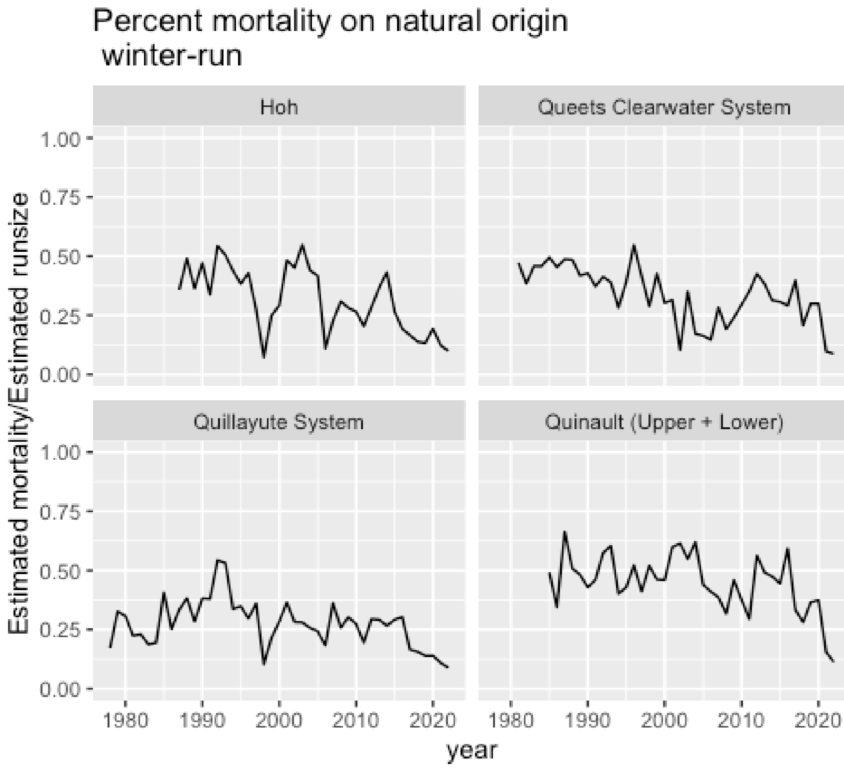

Figure 31. Harvest mortality of natural (escapement after 15 March cutoff) winter-run steelhead reported by co-managers. This is harvest/runsize. Recreational hooking (catch-and-release) mortality is only included in the Hoh River data.

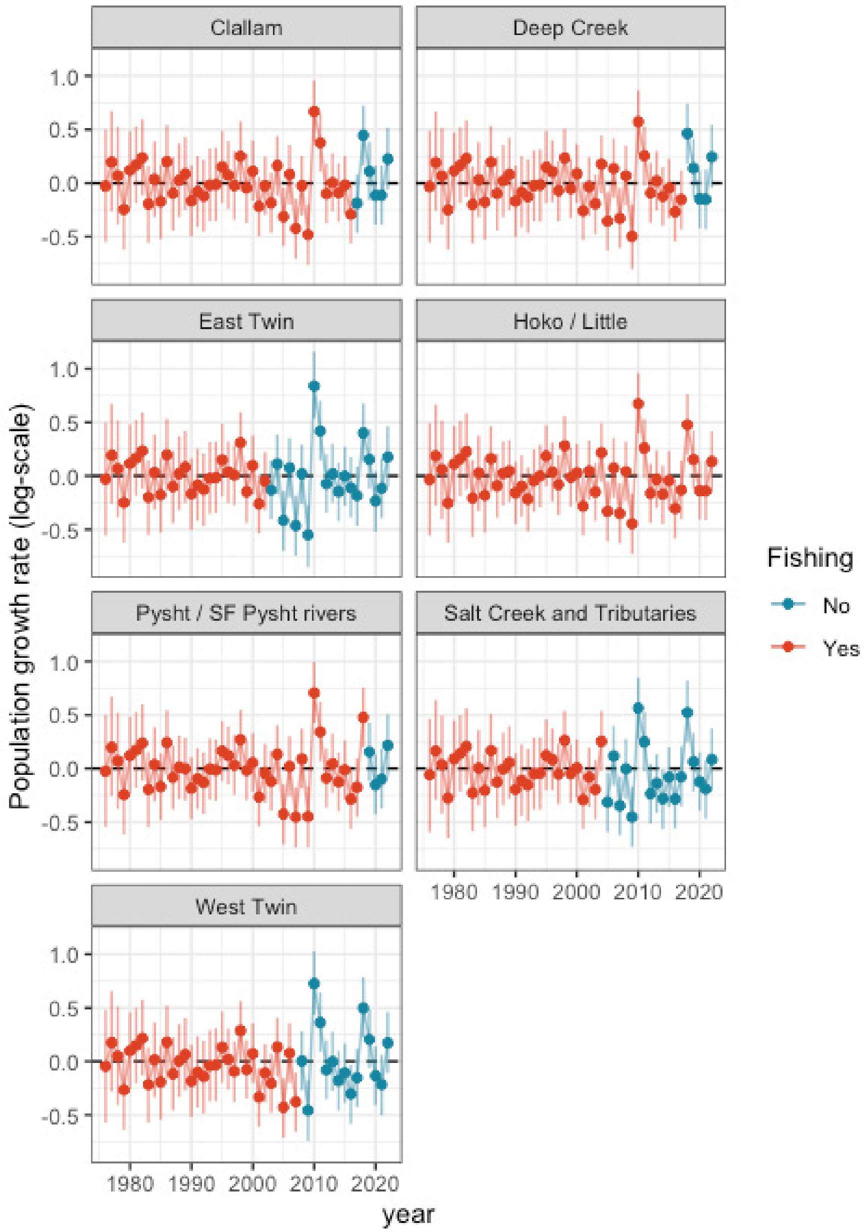

Figure 32. One-year estimates of population growth (log scale) during period with and without harvest on strait populations of OP steelhead. Estimates are from the DLM output. Mean and 95% CI shown in vertical lines.

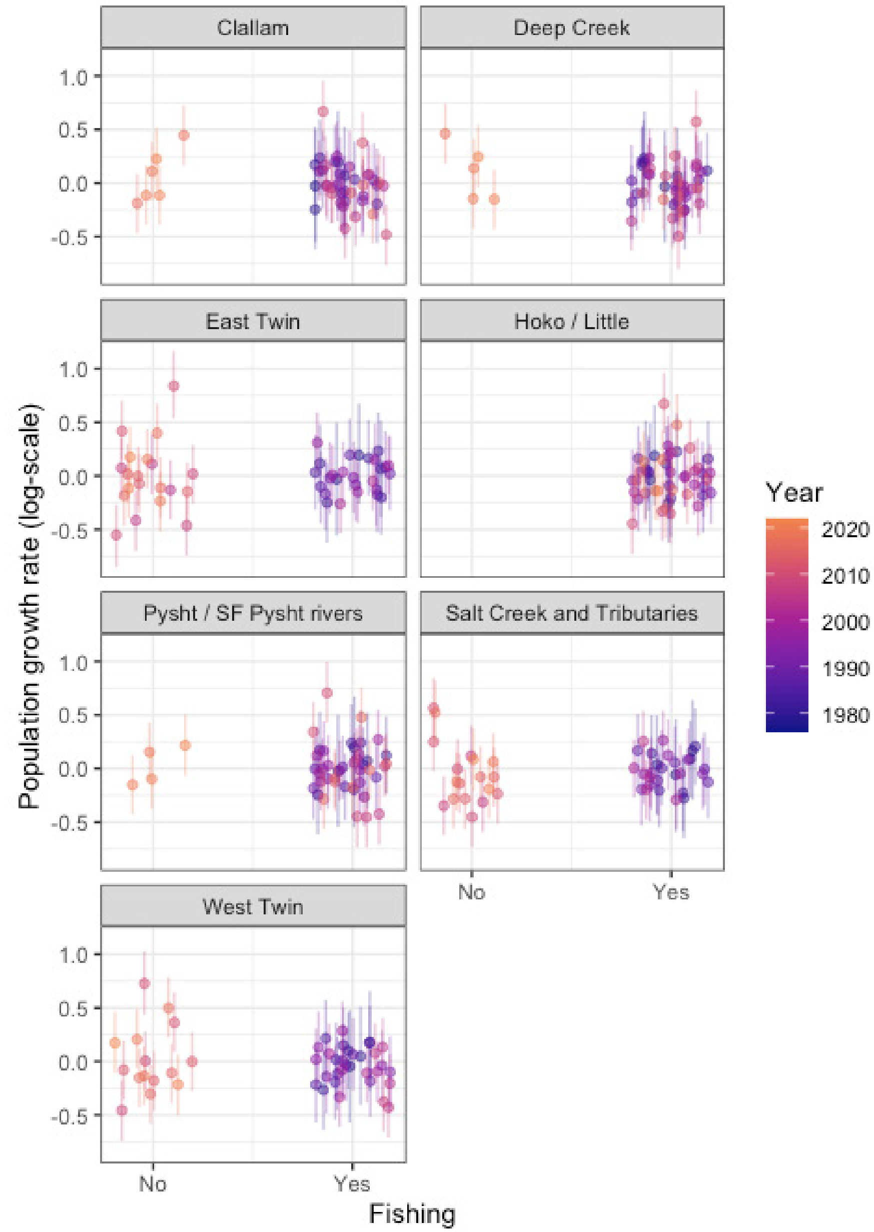

Figure 33. Population growth rates during periods with and without harvest on strait populations of OP steelhead. Estimates are from the DLM output. Mean and 95% CI shown in vertical lines.

Table 11. Five-year geometric mean of winter-run stocks. Observed escapement after cutoff date is 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	1978–82	1983–87	1988–92	1993–97
Moclips River	Southern Lowlands			239 (245)	343 (291)
Quinault River	Big Four	4,018 (3,959)	3,734 (3,977)	3,965 (3,747)	2,887 (2,757)
Queets River	Big Four		4,820 (5,267)	5,480 (5,404)	4,003 (4,029)
Hoh River	Big Four	2,613 (2,694)	3,430 (3,210)	2,569 (2,650)	2,348 (2,376)
Goodman Creek	Southern Lowlands				
Quillayute–Bogachiel River	Big Four	1,613 (1,725)	2,293 (2,281)	2,105 (2,097)	1,703 (1,800)
Calawah River	Big Four	1,810 (1,916)	2,842 (2,803)	2,779 (2,766)	2,862 (2,911)
Sol Duc River	Big Four	3,523 (3,490)	3,597 (3,761)	3,577 (3,761)	4,526 (4,328)
Dickey River	Big Four	527 (497)	473 (481)	379 (393)	350 (385)
Hoko/Little River	Strait			699 (693)	526 (557)
Clallam River	Strait				
Pysht/SF Pysht Rivers	Strait			271 (280)	251 (254)
Deep Creek	Strait				
West Twin River	Strait				
East Twin River	Strait				
Salt Creek and tributaries	Strait				

Table 12. Five-year geometric mean of winter-run stocks. Observed escapement after cutoff date is 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	2003–07	2008–12	2013–17	2018–22
Moclips River	Southern Lowlands					
Quinault River	Big Four	2,259 (2,683)	2,716 (2,673)	2,887 (2,770)	2,625 (2,508)	2,186 (2,356)
Queets River	Big Four	4,111 (4,532)	5,634 (5,285)	4,613 (4,531)	3,583 (3,507)	2,931 (3,140)
Hoh River	Big Four	3,088 (3,032)	2,254 (2,397)	2,677 (2,569)	2,314 (2,366)	2,735 (2,840)
Goodman Creek	Southern Lowlands	287 (283)				76 (81)
Quillayute–Bogachiel River	Big Four	2,957 (2,776)	1,972 (1,980)	1,710 (1,664)	1,221 (1,280)	1,166 (1,191)
Calawah River	Big Four	4,798 (4,590)	3,122 (3,218)	2,732 (2,729)	2,526 (2,488)	2,551 (2,728)
Sol Duc River	Big Four	5,696 (5,678)	3,897 (3,898)	2,980 (3,016)	2,553 (2,676)	3,483 (3,458)
Dickey River	Big Four	699 (628)	344 (391)	384 (344)	268 (297)	423 (418)
Hoko/Little River	Strait	698 (688)	494 (490)	401 (415)	344 (362)	438 (420)
Clallam River	Strait		158 (153)	105 (118)	129 (128)	146 (144)
Pysht/SF Pysht Rivers	Strait	351 (356)	209 (211)	160 (170)	194 (192)	237 (229)
Deep Creek	Strait	162 (160)	99 (104)		83 (83)	99 (97)
West Twin River	Strait	116 (109)	55 (55)	42 (46)	43 (45)	56 (52)
East Twin River	Strait	85 (83)	50 (48)	35 (38)	51 (48)	54 (55)
Salt Creek and tributaries	Strait	171 (165)	106 (105)	84 (82)	44 (53)	66 (60)

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	n/a

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 $\hat{C}_{it}$ be the predicted catch. Then:

$$Z_{it} \sim LogNormal\ (\log Y_{it} - F_{it} - 0.5\sigma^2 R, \sigma^2 R) \quad (1)$$

$$\log \boldsymbol{Y_t} = \log \boldsymbol{X_{t-1}} + \boldsymbol{\mu} + \boldsymbol{\varepsilon_t} \quad (2)$$

$$\boldsymbol{\varepsilon_t} \sim MultivariateNormal\ (0, \boldsymbol{\Sigma_Q}) \quad (3)$$

and for catches,

$$C_{it} \sim Normal\ (\hat{C}_{it}, \phi\hat{C}_{it}) \quad (4)$$

$$\hat{C}_{it} = Y_{it}(1 - \exp(-F_{it})) \quad (5)$$

Following standard notation, bold symbols indicate vectors. Note that, unlike a standard model, this model is nonlinear in log-space and has two likelihood components, one for catch and one for escapement. We estimate the process error covariance $\boldsymbol{\Sigma_Q}$ with a single variance term, σ_Q^2, and correlation among rivers Θ. For a four-population model, $\boldsymbol{\Sigma_Q}$ is σ_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. Prior distributions for parameters include:

$$\bar{F} \sim TruncatedNormal(0{,}2) \quad (6)$$

$$\tau_F \sim Gamma(2{,}10) \quad (7)$$

$$\sigma_Q^2 \sim Gamma(2{,}2) \quad (8)$$

$$\sigma_R^2 \sim Gamma(1{,}1) \quad (9)$$

$$\log X_{i0} \sim Gamma(1{,}1) \quad (10)$$

$\bar{F}$ is truncated at 0 to ensure it is positive, and $\log X_{i0}$ is the prior distribution on the abundance in the first year of each time-series.

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

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, 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 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]).

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 one-year time lags, density dependence, 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.

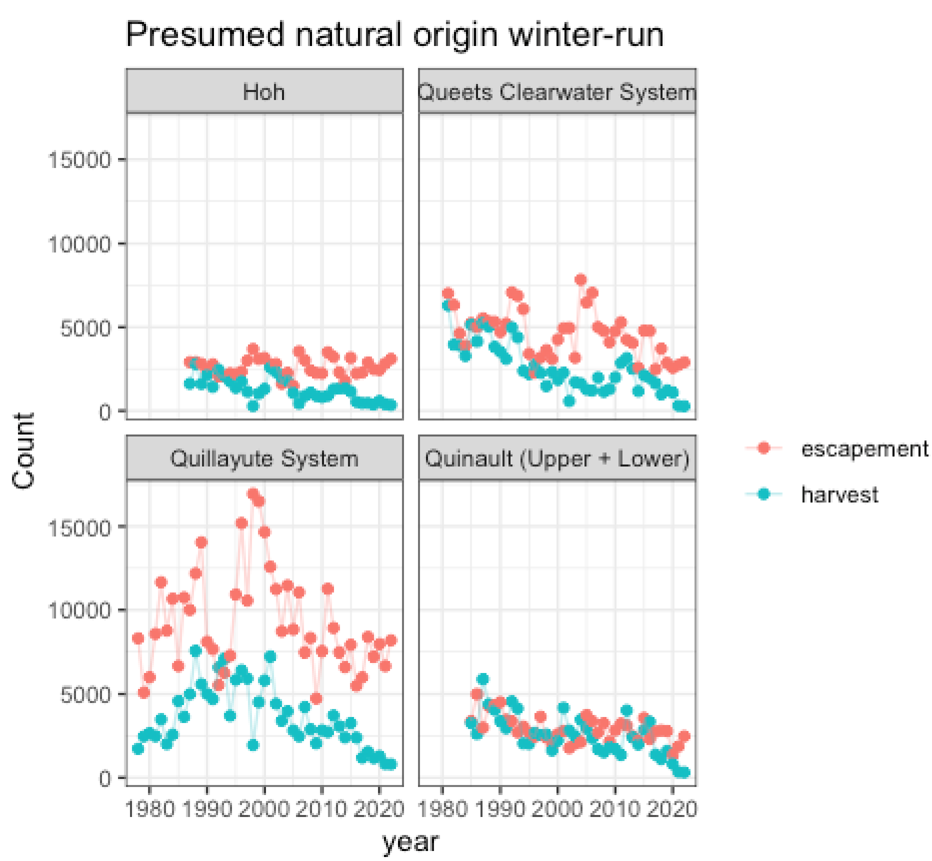

Figure 34. Raw data for escapement and harvest of natural (after March cutoff) winter-run steelhead, as reported by co-managers.

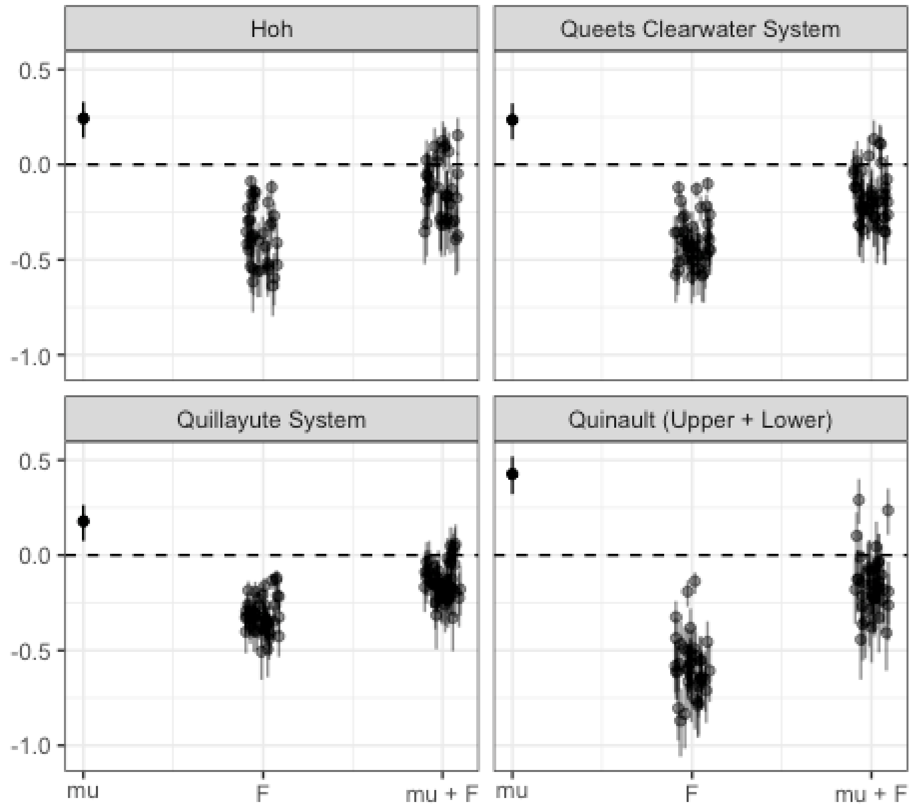

Figure 35. Estimated log-scale population growth rate (*mu*), estimated annual harvest mortality (*F*), and net population growth rate (*mu + F*). For *F* and *mu + F*, each point represents the estimated value in a particular year. For all parameters, means and 95% CI are shown.

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 (*Oncorhynchus nerka*), and Chinook salmon; these bycatch data were not available and are 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 natural and hatchery summer steelhead in the Quillayute, Hoh, Queets, and Quinault Rivers, with generally fewer than 100 fish per year on average. The average harvest of summer-run fish has historically been a few hundred in most basins, except for the Quillayute River basin, where hatchery-origin summer-run steelhead are released and combined hatchery- and natural-origin fish catches are in the thousands of fish (Table 14). 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 natural summer steelhead among OP Steelhead DPS rivers. (Mean annual, range, period of record.)

	Sport harvest		Commercial harvest	
River	**Hatchery**	**Total[a]**	**Total[a]**	**Source**
Quillayute River	611 (27–1,974) 1988–2022	n/a	179 (29–373) 1953–57	WDG, letter
Hoh River	n/a	275 (38–711) 1962–92	291 (23–954) 1975–82	WDG, letter
Queets River	n/a	222 (21–516) 1962–92	104 (43–171) 1975–82	WDG, letter
Quinault River	n/a	132 (0–452) 1962–92	n/a	n/a

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, 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 (2018–22) 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 basins in the OP Steelhead DPS. Several studies (McLachlan 1994, McMillan and Gayeski 2006, 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 OP Steelhead 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 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 stems mostly from a loss of early spawners (November–January), the replacement rate may suffer if conditions for egg and fry survival are higher in midwinter 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. J. Meyer (National Park Service, personal communication) 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-run steelhead in the Quillayute River returned before 1 January. Further, the *q25* of the natural run[5] 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 in early-to-midwinter (November–January), but the fishery harvests natural steelhead as well, and the Petitioners argue that declines in early natural 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 natural steelhead with the hatchery-origin fish, potentially reducing their fitness relative to late-returning natural steelhead; and 2) higher exploitation (harvest rate) of early-returning natural fish relative to late-returning natural fish, which would also reduce their relative fitness. If within-season run timing has a heritable genetic component,

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

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 through 1960 indicates that there was significant harvest of steelhead in November and December in the Queets and Quinault Rivers (Figure 36; M. Moore, Washington Department of Fisheries, personal communication). 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 (Figures 37 and 38).

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, p. 3).

Meyer (personal communication) 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 found this argument plausible and requested data from the co-managers that would allow the team 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 midcentury period (1948–60) 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 natural fish, which ideally would have stalled the ongoing compression of run timing. Genetic data suggest the strategy to segregate hatchery stocks has tended to prevent interbreeding with natural stocks (although these genetic data are 10–20 years out of date and thus represent a weak test). However, the catch of natural-

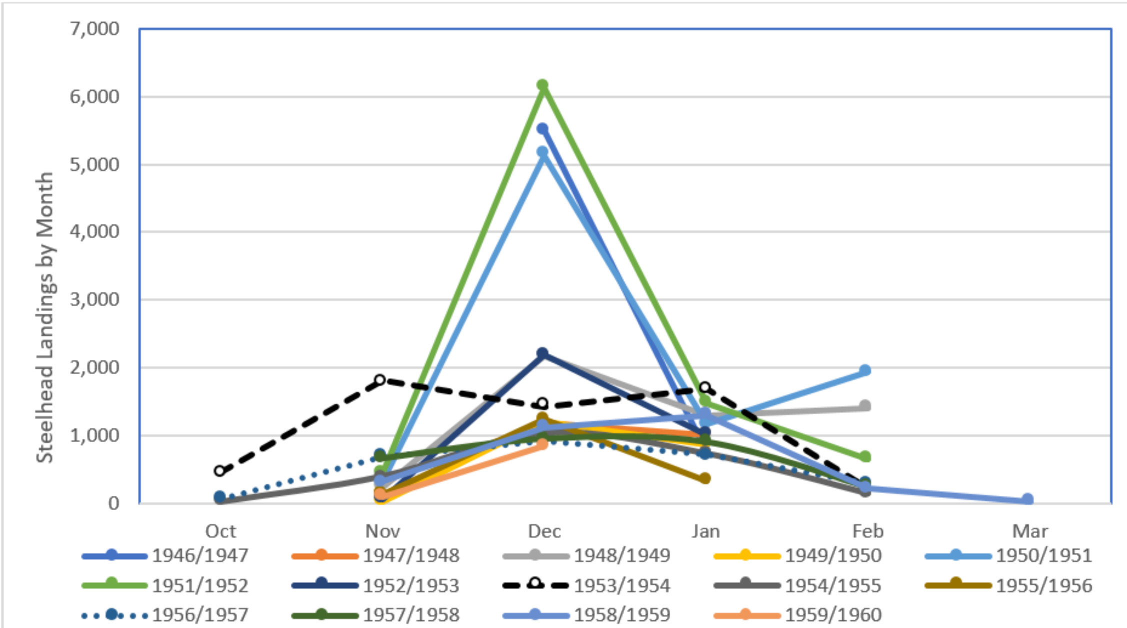

Figure 36. Quinault River steelhead gill and set net harvest, 1946–60. From Moore (personal communication).

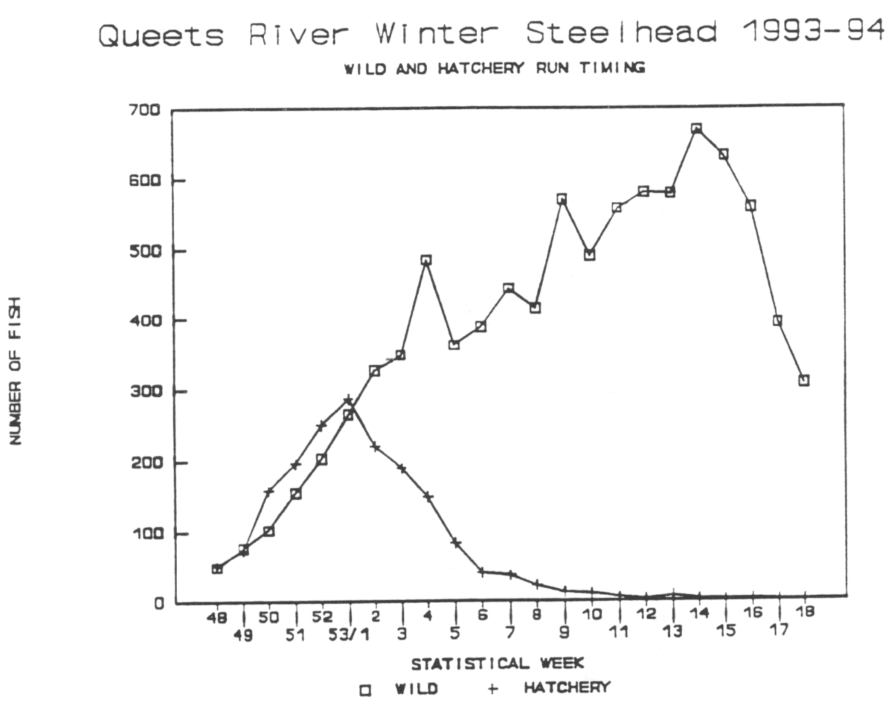

Figure 37. Predicted run timing and magnitude of hatchery and natural winter-run steelhead entering the Queets River during 1993–94. From QFD and WDW (1993), p. 4, their Figure 2.2.

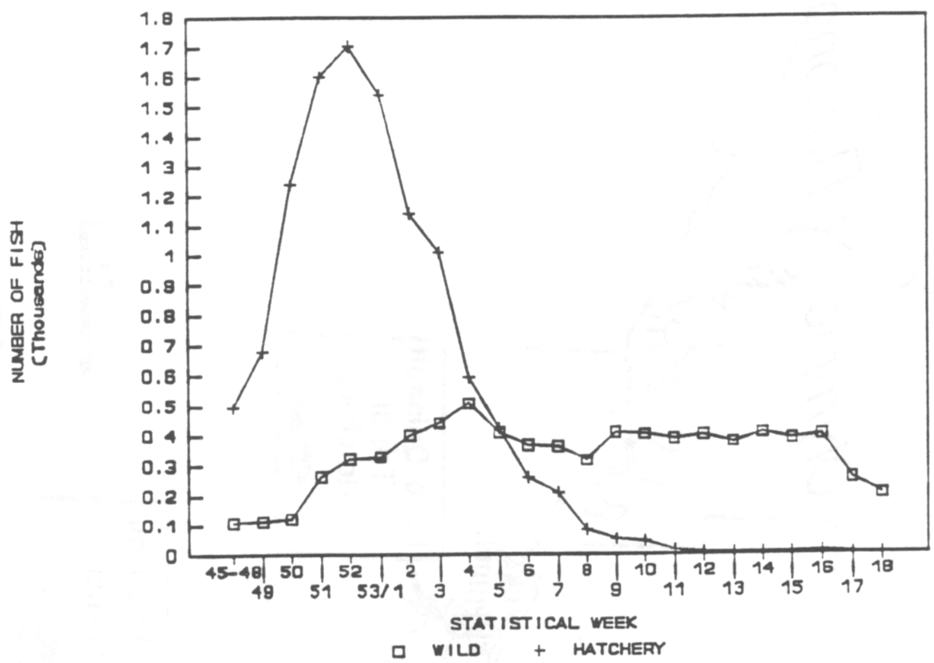

Figure 38. Predicted run timing and magnitude of hatchery and natural winter-run steelhead entering the Quinault River during 1993–94. From QFD and WDW (1993), p. 4, their Figure 2.2.

origin steelhead in the early-winter fishery is appreciable, consistent with the exploitation hypothesis. Meanwhile, the ongoing declines of natural run sizes over the past three decades (see Hatchery operations in the OP Steelhead DPS) 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 data 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 analysis of variance (ANOVA) or linear regression, except the effects are estimated as smooth spline curves rather than categories (ANOVA) or straight lines (linear regression), and so make 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:

$$\text{Weekly spawners} = mean + re(year) + re(week) + s(year) + s(week) + s(year, week),$$

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 natural fish.

Datasets with sufficient granularity to test the above compression hypothesis for natural steelhead were not available.

- The Petitioners provided daily tribal harvest data (see Supporting Files), but the dataset does not distinguish hatchery fish from natural fish.
- The co-managers provided harvest data for natural steelhead (see Supporting Files), but only in a summary form which lumps by year, so monthly or weekly totals are not available.
- The co-managers provided weekly recreational harvest data for natural steelhead (see Supporting Files) broken down by year and month, but only for summer steelhead (June through October); the data include 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[6] maintained by one of the authors, with relevant data for the Quillayute, Hoh, and Queets River systems for the period 1980–2017 (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 reasonably be 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 (Footnote 6) associated with a peer-reviewed paper by McMillan et al. (2022), given that no other weekly or monthly specified data with natural and hatchery separated were 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 a 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,[7] this fishery consistently

[6] https://github.com/MartinLiermann/historicalOPsteelhead
[7] 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, though such effort likely contributed to the loss of early-returning fish in the past.

removes hundreds to thousands of natural-origin fish from the population. Because fishery activities target both 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 natural steelhead in the catch. All analyses were conducted in the statistical R package (R Development Core Team 2021) `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 natural used a beta family with a complementary log–log link.

Natural 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 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 Figures 39–41, 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 patterns with this noise removed.

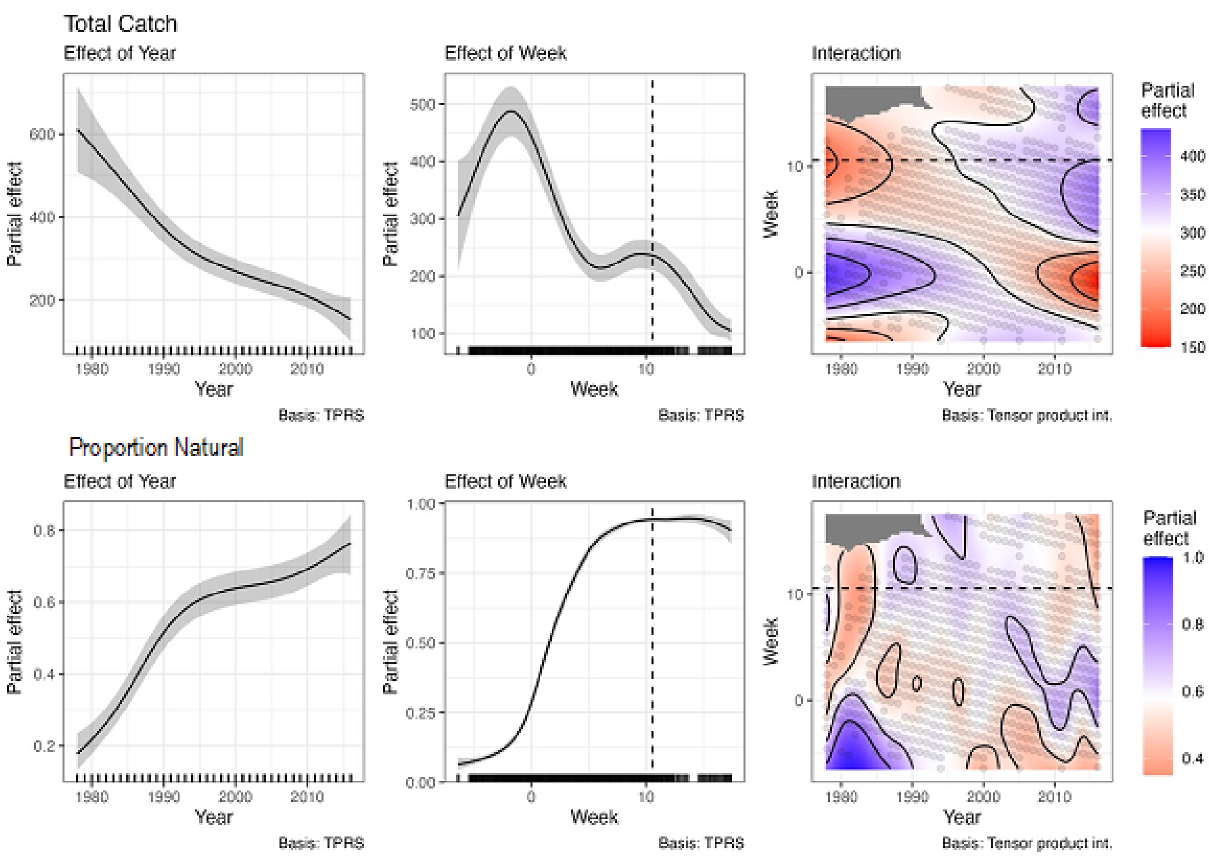

Figure 39. Quillayute River steelhead catch from the gill-net fishery, decomposed into long-term and seasonal trends using generalized additive regression models. Top row shows patterns of total catch, bottom row shows proportion of natural steelhead in total catch. Vertical dashed line marks the week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

In the Quillayute, the total catch has steadily declined since 1980, while the proportion of natural fish in the catch has steadily increased to over 70% of the catch in the latest years of record, and to 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 natural steelhead, but from a total catch initially about twice as high.

Seasonally, on average, the catch peaks in December (top-middle panel; Week 0 = last week in December) but shows a smaller bump in March corresponding to the natural run. However, natural steelhead dominate the catch well before this bump, comprising over half the weekly catch starting in mid-January (bottom-middle panel) and over 80% of catch by early February (Week 10).

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 bottom-right interaction plot shows that even as total catch has disproportionately shifted away from this midwinter period, the proportion of natural fish has increased in it (blue patch for Weeks –3 to 3).

Overall, this suggests that 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 natural steelhead moving earlier and earlier into the season.

Natural 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 River results, but less extreme: total catch is declining, while proportion natural is increasing, though less steadily than in the Quillayute (Figure 40, left panels). As in the Quillayute, natural steelhead catch is larger than hatchery catch by Week 3 (bottom-middle panel) and dominates (>75%) by Week 5 in early February. By the end of the period of record, the proportion of natural steelhead in the catch is disproportionately increasing in Weeks 0–5, beyond what is expected from the main effects (blue patch in bottom-right panel).

Natural steelhead catch in the Queets River

In the Queets River system, both total catch and proportion natural had extremely high statistical significance for all main effects and all interaction effects ($p < 10^{-10}$). The various partial effects for the Queets River (Figure 41) show similar long-term decline of total catch (top-left panel), but a significantly more complicated pattern for the proportion of natural steelhead in the annual catch (bottom-left). Despite these fluctuations in the proportion of natural steelhead in the gill-net 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 natural fish in the catch increased (bottom-left), indicating a diminished role for hatchery fish in the fishery.

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

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

Hatchery operations in the OP 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-

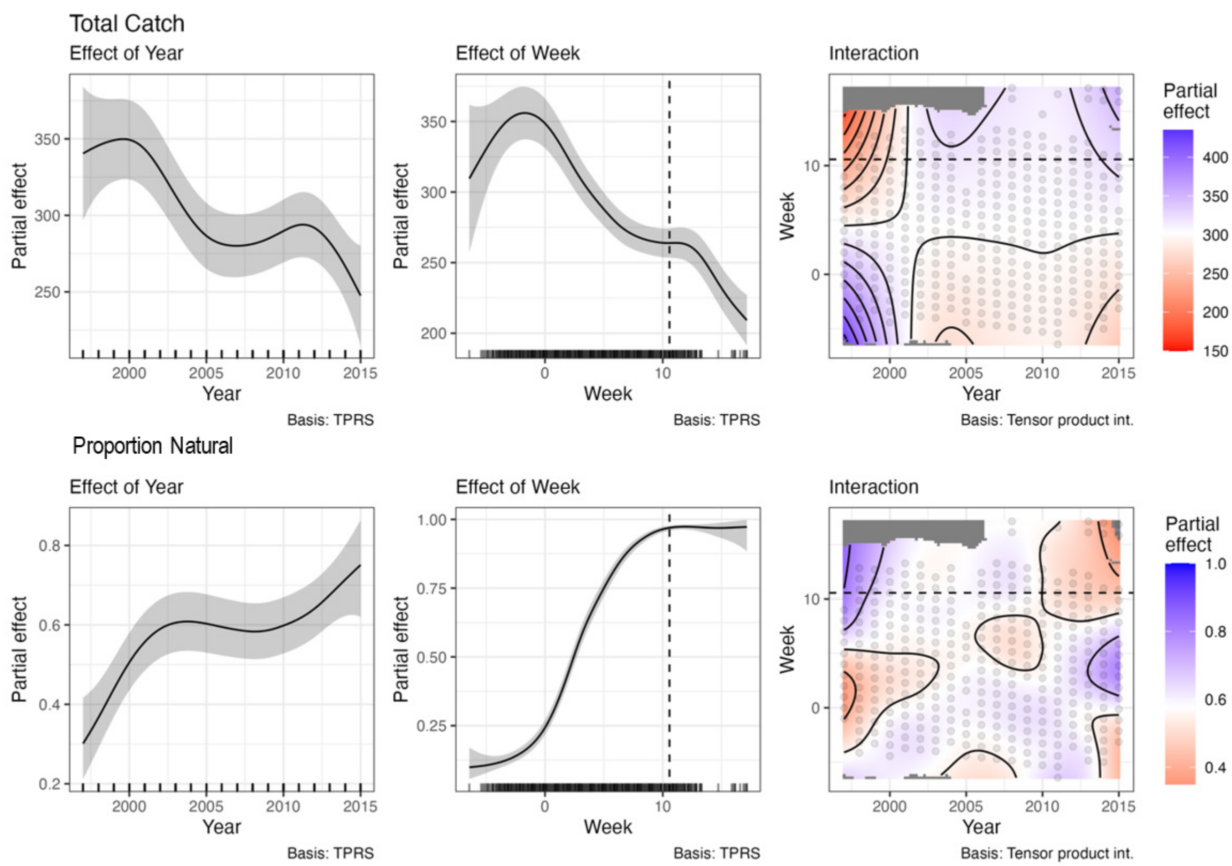

Figure 40. Hoh River steelhead catch from the gill-net fishery, decomposed into long-term and seasonal trends using generalized additive regression models. Top row shows patterns of total catch, bottom row shows proportion of natural steelhead in total catch. Vertical dashed line marks the week of 15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

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 1984, Tatara and Berejikian 2012), as well as the short- 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 the OP Steelhead DPS (Figure 42), the majority (seven) operate segregated hatchery programs that release non-native fish—predominantly of Chambers Creek Hatchery-origin early-winter run or Skamania Hatchery-origin early summer-run (Table 15). The hatchery propagation and release of

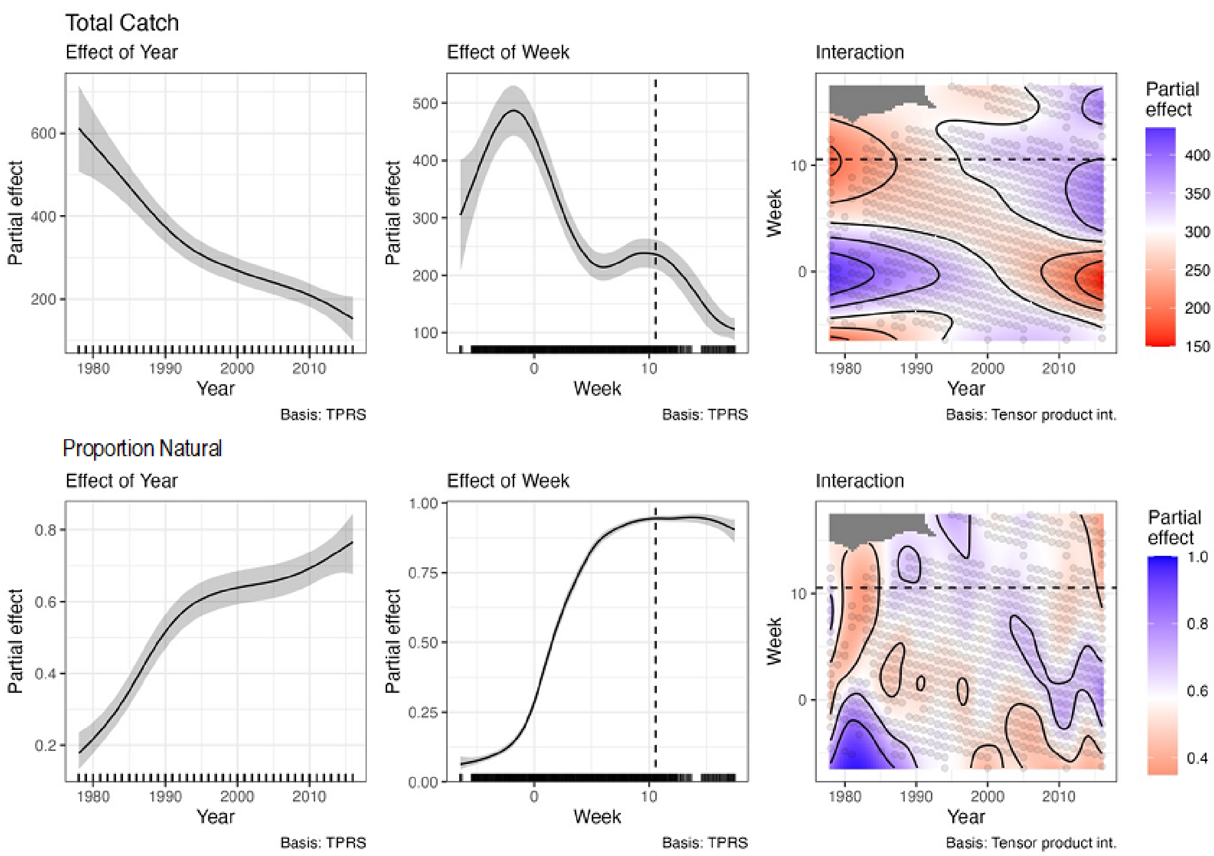

Figure 41. Queets River steelhead catch from the gill-net fishery, decomposed into long-term and seasonal trends using generalized additive regression models. Top row shows patterns of total catch, bottom row shows proportion of natural steelhead in total catch. Vertical dashed line marks the week of 15 March.

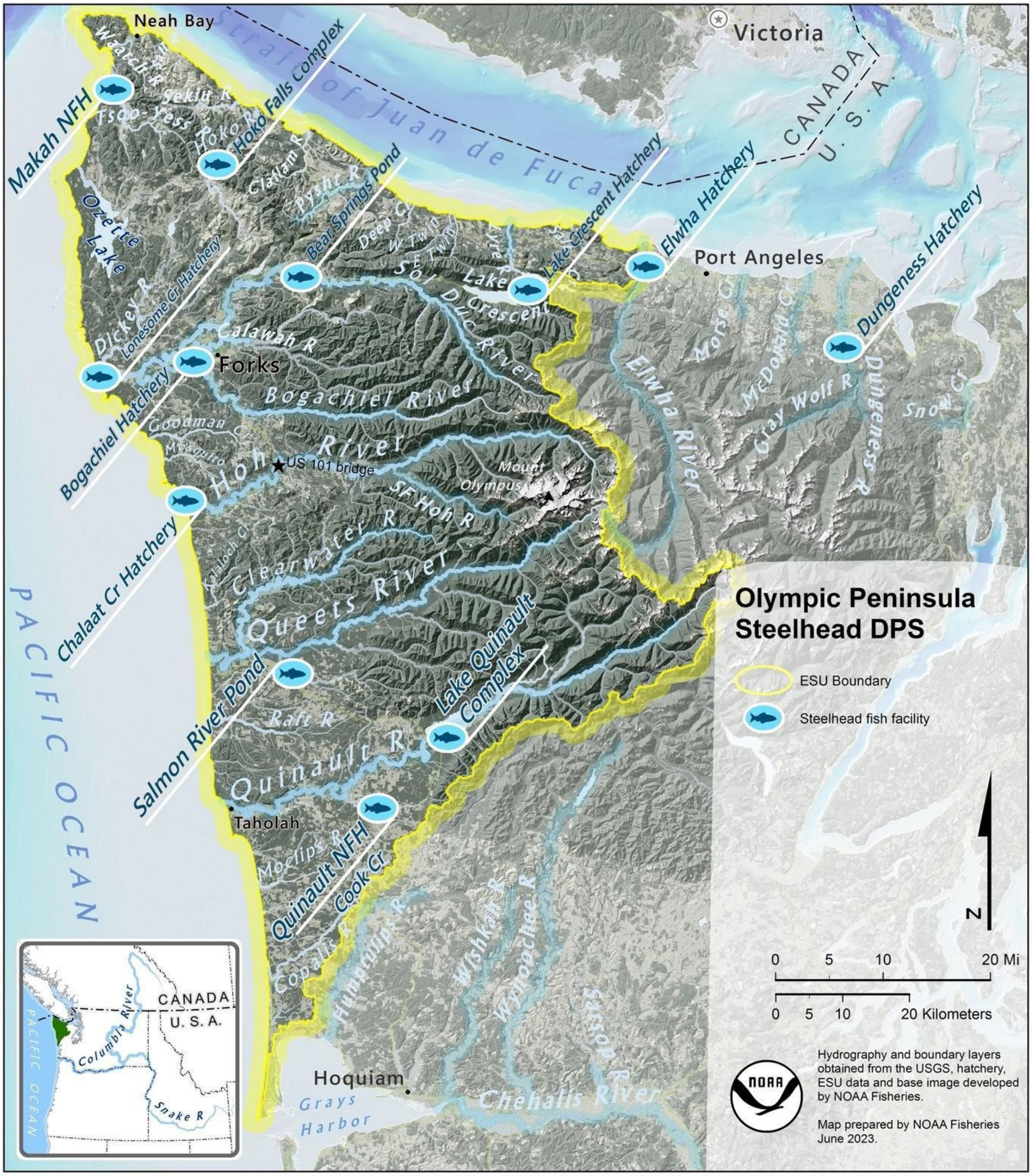

Figure 42. Hatchery facilities in the OP Steelhead DPS that currently release winter- and/or summer-run steelhead. *NFH* = National Fish Hatchery. The Lake Crescent Hatchery released resident *O. mykiss* into the 1970s.

Table 15. Current hatchery programs in the OP Steelhead DPS. *FWS* = U.S. Fish and Wildlife Service, *EWS* = early winter-run steelhead (Chambers Creek Hatchery-origin), *ESS* = early summer-run steelhead (Skamania Hatchery-origin). Program information from 2022 Future Brood Document (draft).[a]

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	W	45,000 EWS	20,354 EWS (Hoko R.) 5,580 EWS (Sekiu R.)	segregated
Educket Creek	20	Tribal	Waatch River	W	22,000 EWS		segregated
Makah NFH	20	FWS/Tribal	Tsoo-Yess River	W	158,000 EWS	128,523 EWS	segregated
Lonesome Creek	20	Tribal	Bogachiel Hatchery	W	80,000 EWS	Transferred to Bogachiel Hatchery	segregated
Chalaat Creek	20	Tribal	Hoh River	W	100,000 EWS	64,354 EWS	segregated
Bear Springs Pond	20	Tribal	Quillayute River	W			segregated
Calawah Pond (North and South)	20	WDFW	Quillayute River	S/W	150,000 EWS 30,000 ESS	56,357 EWS 31,486 ESS	segregated
Bogachiel Hatchery	20	WDFW	Quillayute River	S/W	150,000 EWS 30,000 ESS	108,281 EWS	segregated
Salmon River Fish Culture Facility	21	Tribal	Queets River	W	200,000 WS	171,624 WS	integrated
Quinault Lake Complex	21	Tribal	Quinault River	W	250,000 WS	253,493 WS	integrated
Quinault NFH (Cook Creek)	21	FWS/Tribal	Quinault River	W	190,000 EWS	225,811 EWS	segregated

[a] https://wdfw.wa.gov/sites/default/files/publications/02295/all_alpha_2022_2nd_draft.pdf

winter- and summer-run steelhead in the DPS has remained relatively constant since 1980, with the majority of releases being winter-run steelhead smolts (Figures 43–46). 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 onsite 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).[8] In the Quillayute River 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 this 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 HSRG 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 River basins are still unclipped. In 2014, there were 11 hatchery programs in the OP Steelhead DPS which annually released 1,072,781 smolts (2009–13), of which 61.1% were released off station (Cram et al. 2018). For broodyears 2018–20, 1,105,855 juveniles (> 5 g) were released in the OP Steelhead DPS, predominantly winter-run and primarily on station releases (RMIS Database 2023). Limiting most hatchery production to onsite 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.

While there are multiple hatchery rearing and release sites for steelhead in the OP Steelhead DPS (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, subjected to domestication selection through hatchery rearing practices, or subjected to inbreeding through spawning protocols. 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

[8] 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 Current hatchery broodstocks).

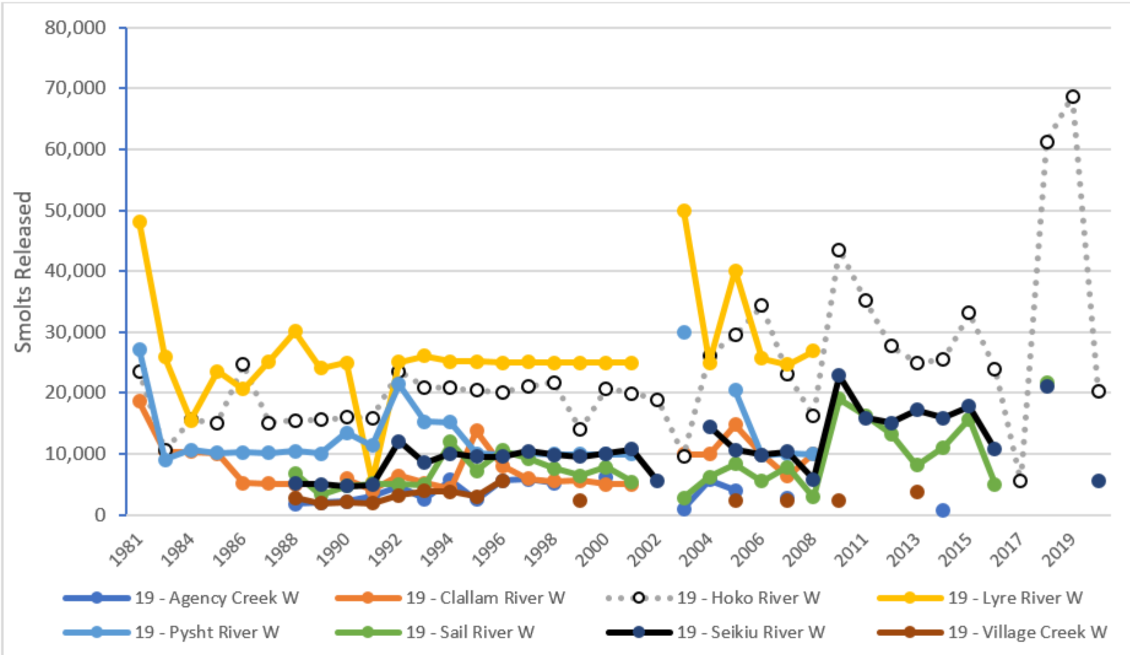

Figure 43. Releases of hatchery-reared winter-run steelhead into Water Resource Inventory Area 19 streams, 1981–2021. Releases of juvenile steelhead weighing less than 5 g are not included. (Data from RMIS database, accessed 23 January 2023.)

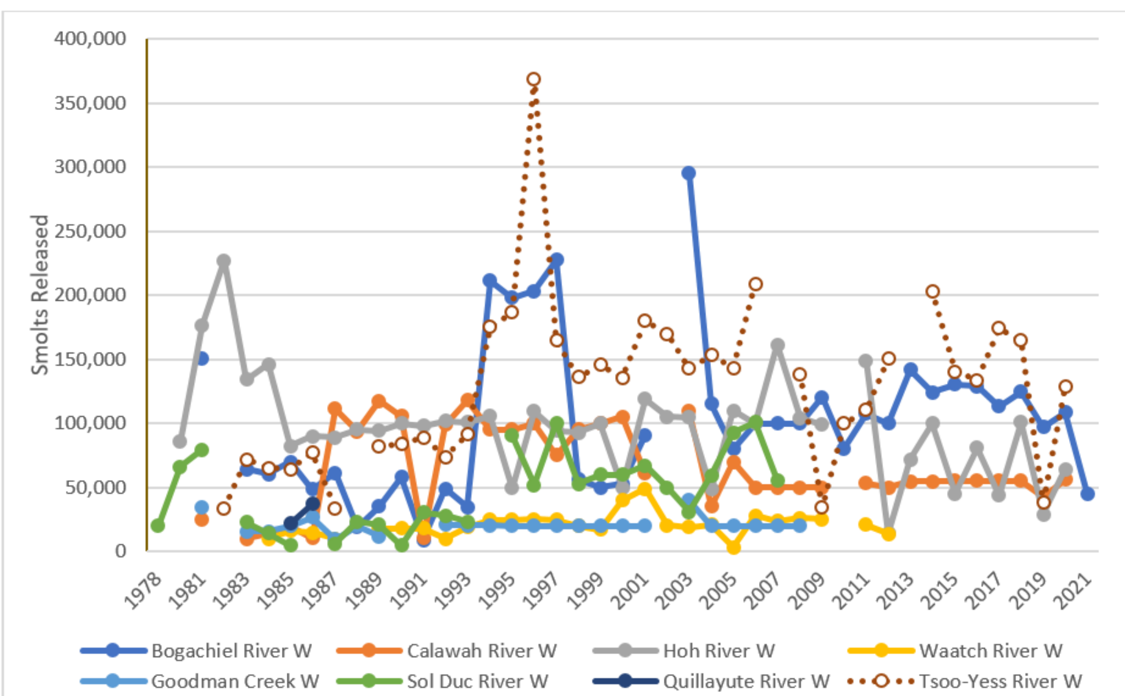

Figure 44. Releases of hatchery-reared winter-run steelhead into Water Resource Inventory Area 20 streams, 1981–2021. Releases of juvenile steelhead weighing less than 5 g are not included. (Data from RMIS database, accessed 23 January 2023.)

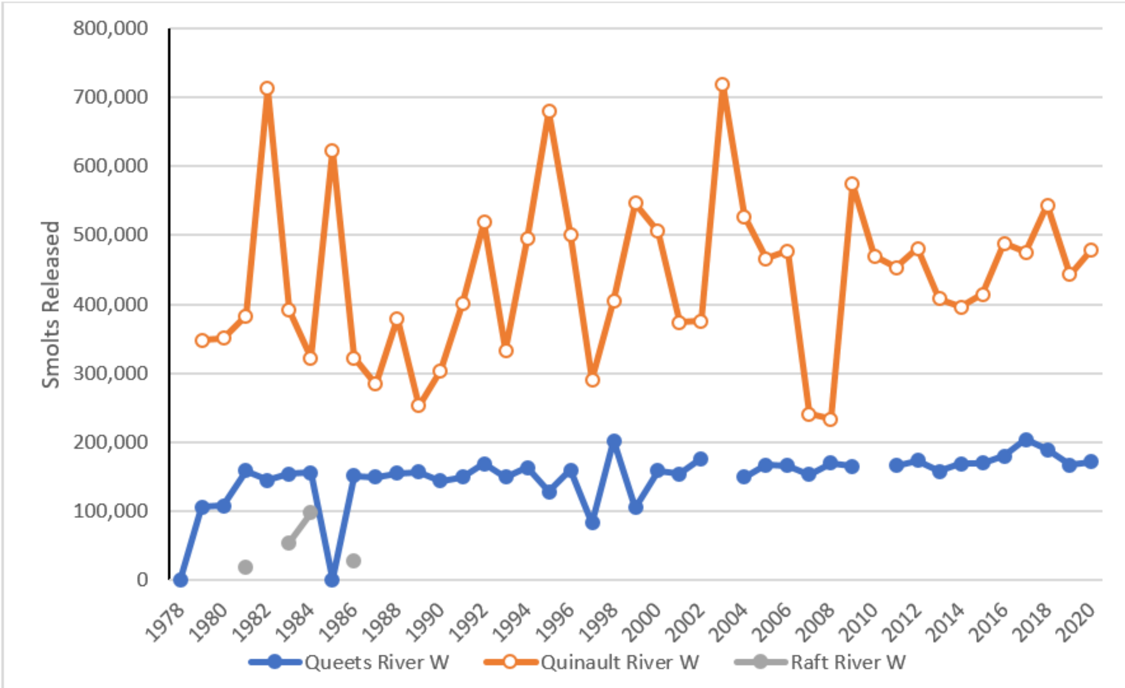

Figure 45. Releases of winter-run hatchery-reared steelhead into Water Resource Inventory Area 21 streams, 1981–2021. Releases of juvenile steelhead weighing less than 5 g are not included. (Data from RMIS database, accessed 23 January 2023.)

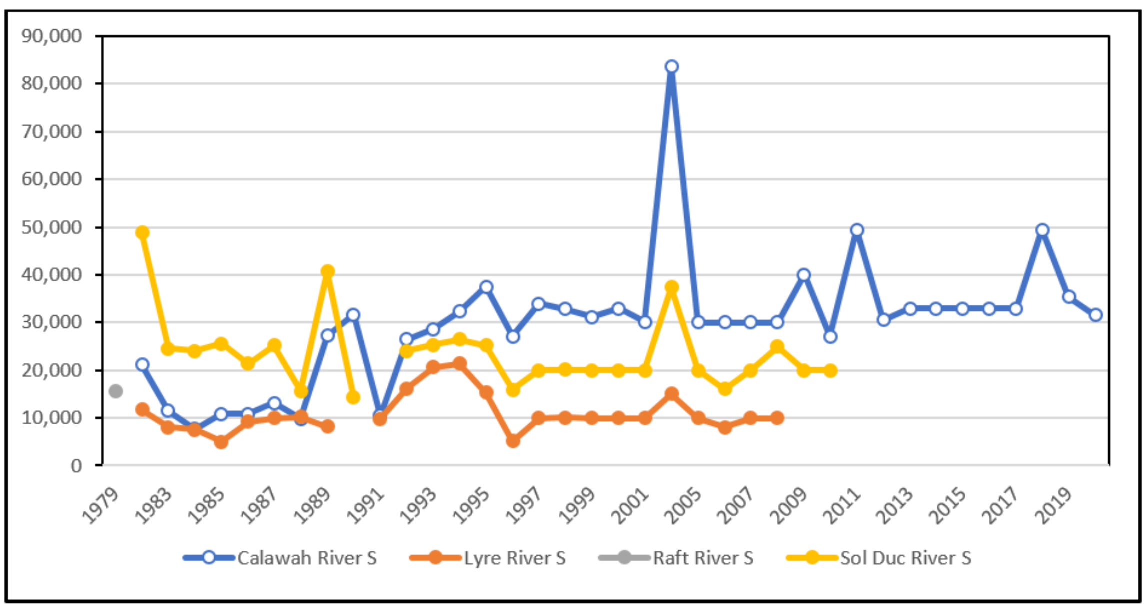

Figure 46. Releases of summer-run hatchery-reared steelhead into OP Steelhead DPS streams, 1981–2021. Releases of juvenile steelhead weighing less than 5 g are not included. (Data from RMIS database, accessed 23 January 2023.)

populations. For example, the reproductive success of early winter-run steelhead from the Bogachiel Hatchery (non-native) spawning in Forks Creek (Willapa River 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 broodyears 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's 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) these out-of-basin transfers have been terminated except in Agency, Sekiu, and Village Creeks. This broodstock is currently being operated as a segregated program, and is considered by the SRT as being non-native from outside of the DPS.[9]

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 U.S. 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 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 is not included in the OP Steelhead DPS.

Quillayute River 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

[9] 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.

from the Skamania Hatchery, Washougal River, Lower Columbia River Steelhead DPS (Scott and Gill 2008). The Skamania stock was established using summer-run steelhead from the Washougal and Klickitat Rivers, 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 River basin. This stock is considered non-native from outside of the OP Steelhead DPS.

Quillayute River 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 River 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 OP Steelhead 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 River hatchery winter steelhead. Neither the Bogachiel Hatchery early-winter steelhead stock nor the Quinault NFH stocks are included in the OP Steelhead DPS; therefore, this hatchery stock is also 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 10). The origins of the Quinault River Hatchery broodstock are 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 steelhead 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 OP Steelhead 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;[10]

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[10] The 2023 co-manager assessment (COPSWG 2023) states that all three hatchery stocks [groups of stocks] released into the OP Steelhead 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

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 et al. (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 resemble Bogachiel Hatchery winter steelhead (Kassler et al. 2010). The Lake Quinault Hatchery broodstock similarly has "mixed" origins (WDFW 2023a). 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. The co-managers expressed some uncertainty about the origins of 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 OP Steelhead DPS.

Hatchery Interactions

The percent hatchery contribution to escapement has been estimated for only a few populations in the OP Steelhead 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%, 19%, and 73% of the sport catch in the Hoh, Sol Duc, and Lyre Rivers at a time of nonselective 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).

On the effect of hatchery releases into rivers with native steelhead, Royal (1973, p. 115) 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.

More recently, the Washington Coast Sustainable Salmon Partnership (WCSSP 2013) estimated the proportion of hatchery-origin adults that were naturally spawning in OP Steelhead 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 (> 25%); although

that the Quinault Lake Hatchery and Salmon River FCF were integrated programs, and does not explain how hatchery- and natural-origin fish can be distinguished in the Queets and Quinault Rivers when the majority of hatchery fish released are unmarked.

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; population estimates varied 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 (personal communication) 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 OP Steelhead DPS lies in a region of the U.S. 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 crossings 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 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 2013), whereas forestry road culverts are covered under the Road Maintenance and Abandonment Plan (RMAP; Table 16). There has been considerable progress in replacing culverts, especially under the RMAP process where over 80% of the culverts are passable (NWIFC 2020).

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 flows and/or high-water temperature barriers (Figure 48) may create temporal passage blockages that may disproportionately affect summer-run steelhead.

Habitat

The quantity and quality of stream, riparian, and upland habitat can directly and indirectly affect the risk of extinction for the OP Steelhead 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

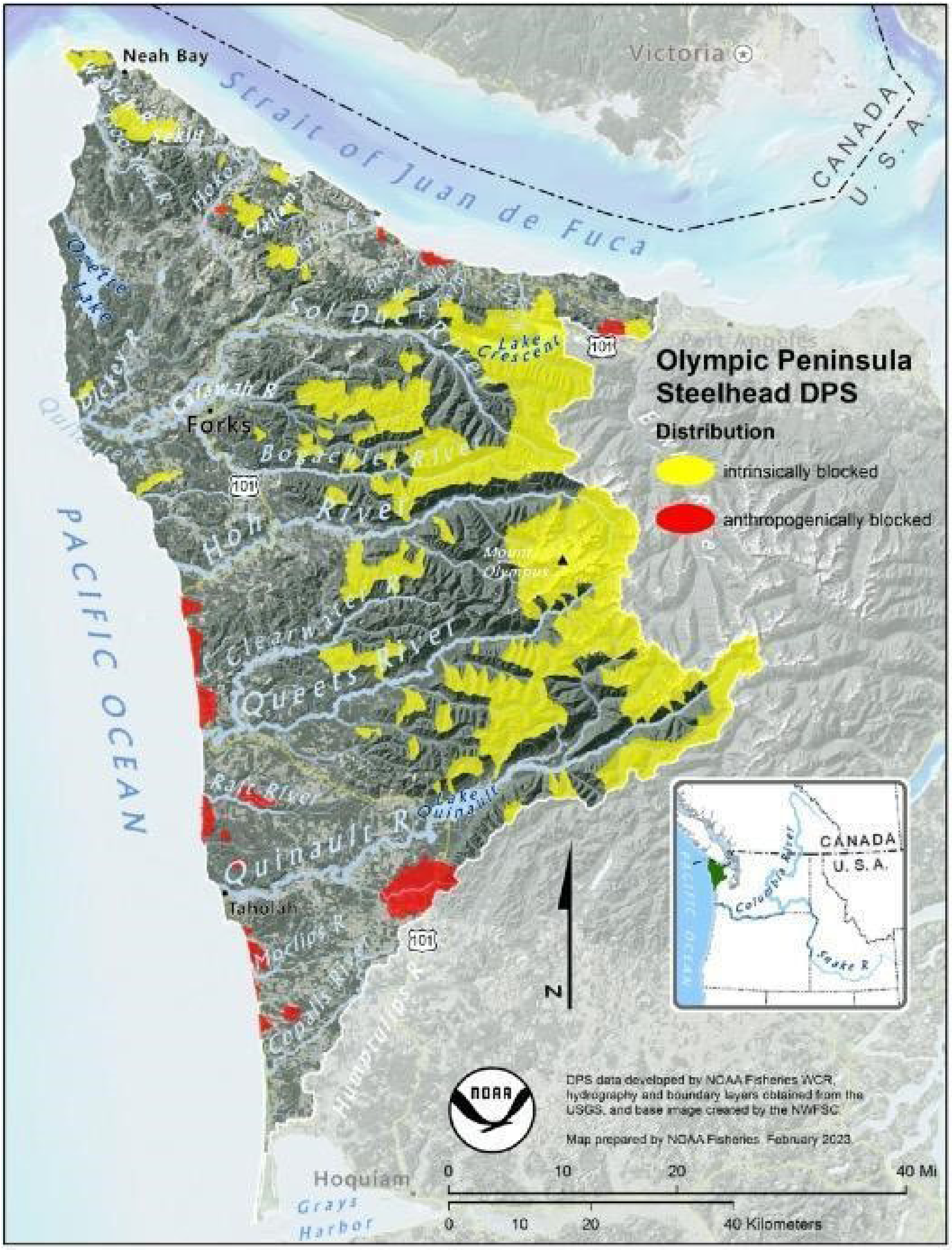

Figure 47. OP Steelhead DPS stream basins blocked by natural (yellow) and anthropogenic (red) barriers.

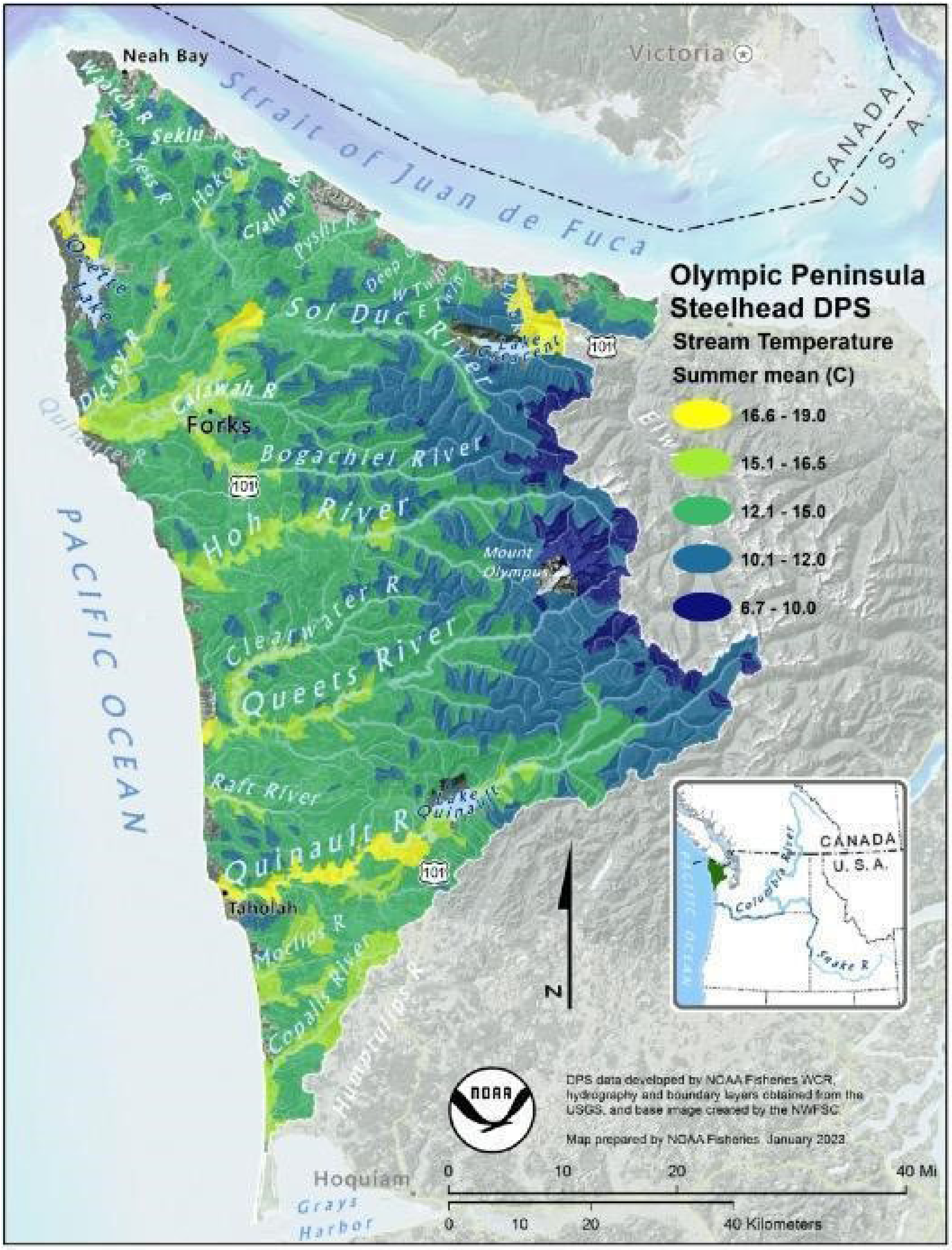

Figure 48. Stream summer mean temperature (°C) for stream reaches in the OP Steelhead DPS.

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

OP areas[a]	Culverts	Total	Fixed	Fixed (%)	Remaining/ impassable[d]	Remaining/ impassable
Makah R.	RMAP[b]	550	448	81%	102	19%
	Non-RMAP[c]	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 R.[e]	RMAP	1,433	1,232	86%	201	14%
	Non-RMAP	3,108	2,380	77%	728	23%

[a] Tribal areas as reported in the State of Our Watershed report (NWIFC 2020).
[b] *RMAP* = Road Maintenance and Abandonment Plan.
[c] *Non-RMAP* = culverts on state, county, and other roads.
[d] Non-RMAP culverts that are 100% impassable.
[e] Quinault River area includes watershed south of the OP Steelhead DPS.

and Caldwell 2001). Most recently, the State of Our Watersheds (SOW) reports reviewed conditions throughout much of Western Washington, including basins in the OP Steelhead DPS (NWIFC 2020). The SRT's assessment of habitat is provided in detail in Appendix A and Appendix B. 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 17). 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.

Analysis of ESA Section 4(a)(1) Listing 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) listing 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) listing factors, otherwise known as threats, specific to OP steelhead in Appendix B. Here, we provide our main findings for each factor, focusing on the time since the last NMFS review of OP steelhead, and present our overall conclusions.

Table 17. Total watershed areas and the proportion of watershed areas inside Olympic National Park (*ONP*) boundaries for the major coastal tributaries in the OP 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	0.00 km^2	0%	223.53 km^2	100%
	Sol Duc R.	603.45 km^2	194.03 km^2	39%	409.42 km^2	61%
Hoh R.	Hoh R.	770.97 km^2	445.63 km^2	58%	325.34 km^2	42%
Queets R.	Queets R.	769.50 km^2	388.92 km^2	51%	380.61 km^2	49%
Quinault R.	Quinault R.	1,123.48 km^2	567.02 km^2	50%	556.42 km^2	50%

Listing Factor 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 continue to be a threat for natural 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 conservation 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 (a.k.a. stream cleaning) due to the fact that both smaller and larger material was removed, resulting 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; 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 Olympic National Park and therefore largely protected from development (see Table 17 for proportion in ONP). However, not all stream/river reach habitat is accessible to steelhead use (see Table 18 for percent of steelhead habitat used within ONP). We note that even if steelhead do not utilize portions of a watershed within 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 percentage 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).

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 geographically 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 19). 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 et al. 2004, NOPLE 2015). The Lyre River watershed includes Olympic National Park (~66%), as well as commercial timberlands (31%) and low-density rural residential lands (~3%; McHenry et al. 1996, NOPLE 2015). The East Twin River basin is

Table 18. Percentage of steelhead habitat used that falls within Olympic National Park (ONP) for various rivers and creeks or basins (e.g., *Hoh R.* contains sub-basins) in coastal Washington that drain directly into saltwater. Note: *Quillayute* presents rivers that comprise the system that had more than 0% in ONP. Basins/rivers not listed have 0% of steelhead habitat used in the park.[a]

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 R.	149,053	14,113	9%	134,940	91%
Mosquito Creek	20,269	1,710	8%	18,558	92%
Upper Quinault R.	183,483	119,663	65%	63,821	35%
Queets R.	220,090	90,816	41%	129,274	59%
Hoh R.	276,356	103,266	37%	173,090	63%
Quillayute R.					
Bogachiel R.	188,336	56,716	30%	131,620	70%
Calawah R.	139,831	24,264	17%	115,567	83%
Sol Duc R.	256,847	44,347	17%	212,500	83%

[a] We attributed the National Hydrography Dataset catchments (Hill et al. 2016) with our proto-populations (usually inheriting the largest river name) and steelhead distribution (WDFW 2023a) by run and use type. These spatial features were then intersected with the land manager polygons from the Protected Areas Database (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 Olympic National Park.

Table 19. Percentage of each landownership type for watershed area, by sub-basin. Modified from NOPLE (2015). *WDNR* = Washington State Department of Natural Resources, *ONP* = Olympic National Park, *USFS* = U.S. Forest Service, *Reserv.* = Reservation, *Ease./ROW* = easements/rights of way. *Total* = WRIA 19.

Sub-basin	Private	WDNR	ONP	USFS	Reserv.	County	Other state land	Other fed land	Ease./ ROW	Other
Salt	50.2%	44.3%	0.0%	0.0%	0.0%	1.1%	0.0%	3.1%	1.3%	0.0%
Lyre	10.4%	17.5%	65.5%	5.7%	0.0%	0.0%	0.0%	0.6%	0.3%	0.0%
East Twin	6.8%	46.1%	< 0.1%	46.2%	0.0%	0.1%	0.0%	0.5%	0.3%	0.0%
West Twin	29.0%	9.9%	0.0%	60.9%	0.0%	0.0%	< 0.1%	0.0%	0.2%	0.0%
Deep	43.2%	4.9%	0.0%	50.4%	0.0%	0.6%	0.0%	0.8%	< 0.1%	0.0%
Pysht	76.7%	5.9%	0.0%	16.6%	0.0%	< 0.1%	0.2%	0.0%	0.5%	0.0%
Clallam	49.6%	47.6%	0.0%	0.1%	0.0%	0.1%	2.1%	< 0.1%	0.6%	< 0.1%
Hoko	72.5%	24.6%	0.0%	0.9%	0.0%	0.2%	1.7%	0.0%	0.1%	< 0.1%
Sekiu	75.7%	17.3%	0.0%	0.0%	7.1%	0.0%	< 0.1%	0.0%	< 0.1%	0.0%
WSI	57.1%	57.1%	0.0%	0.0%	16.8%	0.6%	0.4%	1.2%	1.0%	0.1%
Total	**51.4%**	**22.3%**	**11.6%**	**9.1%**	**3.9%**	**0.3%**	**0.6%**	**0.5%**	**0.4%**	< 0.1%

mostly forestlands; Washington state Department of Natural Resources lands (WA DNR) and U.S. 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; M. McHenry, Lower Elwha Tribe, personal communication). The Sekiu River predominately contains privately owned and state-owned timberlands, but is also partially on the Makah Tribal Reservation (NOPLE 2015).

Pacific Coast

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

NMFS (1996) 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 Appendix B 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 (Swanson et al. 1976, Bisson et al. 1987, Hicks et al. 1991, Sedell and Maser 1994), 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 usable 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, FEMAT 1993, CDFG 1994).

A diverse habitat mosaic is essential for healthy and sustainable salmon and steelhead populations (Hilborn et al. 2003, Brennan et al. 2019). 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, Li et al. 1987, Hicks 1990, Reeves et al. 1993, Munsch et al. 2022). 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 (Appendix A) relative to the impacts of past land-use practices on the Olympic Peninsula. 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, 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). 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 those west of the Lyre River (the East Twin, West Twin, Pysht, Hoko, and Sekiu Rivers). As we discuss extensively in Listing Factor E, increases in winter flow events, decreases in summer flows, and increases in stream temperatures have already been occurring in these watersheds. Finally, the Pysht River estuarine area has been reduced by almost 50% 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 continues to influence stream and riparian habitat quality.

Along the west side of the Olympic 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 in-stream 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, stream flow, 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 the ONP boundary and has had less timber harvest and road building (Jaeger et al. 2023). In general, the reduction in wood loadings and in-stream wood removal has led to the loss of pools and decreases in stabilizing wood jams which led to the loss of channel complexity (particularly in the Queets River; Abbe and Montgomery 2003, Martens et al. 2019). Wood loadings continue to decrease, and the density of large wood in OP forests managed by USFS has decreased by ~50% from 2002–18 (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 substandard and resulted in road failures, increased landslide rates (168 times those of a natural reference area), reduced stream habitat conditions—particularly in some tributaries such as the Clearwater River basin—and 2.5 times the in-stream sediment levels of unclogged OP streams, resulting in reduced salmon egg survival and fry emergence (Cederholm and Salo 1979, Tagart 1984, Cederholm and Reid 1987, McHenry et al. 1998). Additionally, the loss of large trees along riparian zones has 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 et al. 2023). In the Hoh River, increases in sediment supply (from timber harvest and glacial retreat) have led to an increase in channel width and braiding. Due to the high alpine terrain of the Hoh basin, it is 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 Olympic Peninsula. Due to climate change, glacial extent declines have already occurred, with a decline of up to one-third 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; see Listing Factor E).

While, cumulatively, these habitat changes have been large over space and time, the Hoh River basin—as well as the Quillayute, Queets, and Quinault River basins—still exhibits 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 lies within 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 (Sullivan et al. 1987, Kemp 2015). However, restoration actions have occurred and/or are underway to remove culverts and fix fish passage and restore habitat (Table 16). 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 (CSP 2022). This annual report summarizes various restoration efforts for WRIAs within the OP Steelhead DPS boundaries (WRIAs 20 and 21), including many efforts undertaken by the tribes. In WRIA 20, 36 fish passage barriers have been corrected, sediment transport due to the restoration of almost 450 acres of upland area has improved, 1,353 riparian acres have been restored, 11 acres of floodplain have been reconnected, and 30 in-stream miles have been restored. In WRIA 21, corrections to 33 fish passage barriers have occurred, sediment transport has improved due to the nearly 480 acres of upland area restored, 5,939 acres of riparian habitat have been restored, 14 acres of floodplain have been reconnected, and 6 in-stream miles have been restored. For the Pacific Coast region, that includes watersheds south of the Olympic Peninsula, where the State of Washington 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 in-stream 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 from 100 to 225 years (Stout et al. 2018, Martens and Devine 2023).

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

Harvest rates for OP steelhead have declined within the last decade (particularly the last few years) and vary greatly by region (Strait of Juan de Fuca populations vs. the Big Four basins on the coast). We summarize primarily what has occurred since the last NOAA status review (Busby et al. 1996), though we also provide some information for earlier. Most of the information presented here concerns winter-run natural-origin steelhead in the Big Four basins. Data are limited for rivers draining into the Strait of Juan de Fuca (where harvest is mainly terminated) and for summer-run natural-origin steelhead.

OP 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% for natural-origin steelhead averaged across rivers for which there were data through 2013 (Cram et al. 2018). The average harvest rate across the Big Four basins was 36.5% from the 1980s to 2013, including commercial and recreational harvest. Specifically, winter-run natural-origin steelhead in the Quillayute, Hoh, Queets, and Quinault River systems have had harvest rates ranging from 7% to > 40% annually since the 1980s (to 2013). WDFW stated in Cram et al. (2018, p. 86), "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 noted 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 likely not due to harvest alone, but rather 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 Big Four basins were provided by the co-managers along with estimated run size, which can be used to estimate harvest rates (Figure 32, Table 20). Data from recent years (2014–22) not included in Cram et al. (2018) show harvest rates in the Big Four basins ranging from 13.26–59.19% 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 SRT had data (2021 and 2022), there were considerable declines in harvest rates, to 8.66–15.44% across basins, rate declines of approximately 50–70% (Table 20).

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. Martin (2023), from the Makah Tribe, notes that sustainable harvest management is a core principle of traditional resource management and is 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. Further information from Martin (2023) is presented in Listing Factor D).

Recreational and tribal catch of winter-run populations has typically occurred from November to April. In 2004, ONP implemented catch-and-release regulations for natural steelhead throughout coastal rivers of the OP Steelhead 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 systems, the estimates of harvest rates presented here do not include catch-and-release (hooking) mortality. (For further information on where such information is included,

Table 20. Calculated harvest rates (commercial and sport) for natural-origin steelhead in the Quillayute, Hoh, Queets, and Quinault Rivers, 1978–2022, based on total run size and escapement data provided by the co-managers (tribes and WDFW). Harvest = run size – escapement; percent harvest = harvest / run size. It is possible that steelhead harvested post-spawning (kelts) would be counted in both escapement and harvest; however, harvest during the March–May period (when kelts would be encountered) is relatively low.

Year	Quillayute River basin	Hoh River	Queets and Clearwater Rivers basin	Quinault (Upper + Lower) River
1978	17.23%	n/a	n/a	n/a
1979	32.67%	n/a	n/a	n/a
1980	30.73%	0.00%	n/a	n/a
1981	22.40%	0.00%	47.27%	n/a
1982	23.01%	0.00%	38.43%	n/a
1983	18.68%	0.00%	45.78%	n/a
1984	19.45%	0.00%	45.76%	n/a
1985	40.71%	0.00%	49.50%	49.17%
1986	25.28%	0.00%	45.32%	34.38%
1987	33.31%	35.76%	48.71%	66.33%
1988	38.29%	49.07%	48.50%	50.77%
1989	28.45%	36.40%	41.83%	48.24%
1990	38.24%	47.18%	42.84%	42.83%
1991	38.00%	33.83%	37.26%	46.01%
1992	54.38%	54.35%	41.27%	57.40%
1993	53.10%	50.46%	38.97%	60.41%
1994	33.69%	43.86%	28.16%	40.11%
1995	34.89%	38.28%	39.20%	42.85%
1996	29.72%	42.89%	54.80%	52.18%
1997	35.96%	27.55%	41.55%	41.15%
1998	10.30%	7.24%	28.87%	51.93%
1999	21.50%	24.93%	42.77%	46.20%
2000	28.39%	29.23%	30.25%	45.96%
2001	36.48%	48.29%	31.48%	59.85%
2002	28.23%	45.15%	10.40%	61.40%
2003	28.04%	54.90%	35.06%	54.90%
2004	25.74%	44.04%	17.22%	62.01%
2005	24.25%	41.71%	16.37%	43.93%
2006	18.25%	10.97%	14.61%	41.03%
2007	36.14%	22.69%	28.43%	38.63%
2008	25.78%	30.91%	19.22%	31.77%
2009	30.25%	28.18%	23.95%	45.91%
2010	27.32%	26.56%	29.56%	37.54%
2011	19.48%	20.37%	35.07%	29.52%
2012	29.41%	28.50%	42.64%	56.30%
2013	29.16%	36.76%	38.28%	49.12%
2014	26.65%	43.19%	31.31%	47.46%
2015	29.19%	26.58%	30.67%	44.43%
2016	30.34%	19.31%	29.16%	59.19%

Table 20 (continued). Calculated harvest rates (commercial and sport) for natural-origin steelhead in the Quillayute, Hoh, Queets, and Quinault Rivers, 1978–2022.

Year	Quillayute River basin	Hoh River	Queets and Clearwater Rivers basin	Quinault (Upper + Lower) River
2017	16.53%	16.63%	39.78%	33.41%
2018	15.63%	13.79%	20.86%	28.14%
2019	13.90%	13.26%	29.90%	36.51%
2020	13.94%	19.31%	29.91%	37.39%
2021	10.93%	12.29%	9.76%	15.44%
2022	8.93%	9.96%	8.66%	11.31%
2013–22 avg.	**19.52%**	**21.11%**	**26.83%**	**36.24%**

including for the Hoh River, see cross-references below.) Additionally, information from Bentley (2017) led to a sport angler encounter rate calculation of 1.14 for natural 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 that 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).

Notably, outside of the Big Four basins, directed steelhead harvest for most rivers along the Strait of Juan de Fuca was terminated in various years after the late 2000s/2010s (Figure 35; see 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; see Listing Factor D). In recent years, 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 ("fishing from a floating device is prohibited," WDFW 2020) and bait (see WDFW 2020, LaBossiere 2021, WDFW 2022a, Harbison et al. 2022).

In 2022–23, sport fishing was closed on the Queets and Quinault Rivers from 1 December to 30 April because of low returns and "failure to reach agreement on an acceptable level of wild steelhead harvest" (WDFW 2022b, p. 1). The total number of weeks of tribal fisheries has declined in recent years, specifically in the Queets and Quinault Rivers, and, as mentioned before, harvest rates have declined as well. In addition, WDFW added harvest restrictions to protect returns to the Bogachiel Hatchery to ensure broodstock egg take (WDFW 2022b). WDFW implemented similar gear and floating device restrictions for

2023–24 and set a bag limit of two hatchery steelhead (WDFW 2023b). For the 2023–24 season, the National Park Service closed the Queets and Quinault Rivers within the ONP to sports fishing, beginning on 27 November 2023 (NPS 2023).

On 26 January 2024, the co-managers clarified for the SRT in a written response what data are included in estimates of run size and harvest (J. Scott, WDFW, personal communication). For the Hoh River, run size and total catch of natural-origin steelhead included hooking mortality in the sport fishery dating back to the 2003–04 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 natural steelhead does not 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 Quillayute and Hoh 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 is currently included in the data, although it was not until the 2020–21 season that the tribal managed (on-reservation) nontreaty recreational harvest component for the Queets River system was included. Furthermore, there are key differences in estimates of natural-origin steelhead escapement in surveys in Quillayute–Hoh versus Queets River systems. The Quillayute–Hoh estimates are based on number of redds × 0.81 female/redd × 2 fish. In Queets, the estimator is total number of redds × 1 female/redd × 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 15 March, when it is assumed that all the fish present are natural-origin steelhead. Therefore, those natural-origin fish returning and spawning prior to 15 March would not be counted in redd surveys, resulting in potential underestimates of run sizes and an overestimate of harvest (see Life-History Traits for a discussion of 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 1 November) and Week 18 in the following year (approximately April) in directed fisheries, or as bycatch in other fisheries;[11] however, any steelhead caught in other salmonid fisheries outside this time period were not included.

In Listing Factor D, both here and in Appendix B, 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 et al. 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 5). In the Queets River system, the co-managers have differing escapement goals.[12] Each year, specifically for the

[11] Scott, personal communication.
[12] 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 were needed. 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 (Scott, personal communication).

Big Four systems, the co-managers develop management plans that 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 18% of the time since 1970. In recent years (2021–22), 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 River, the state-specific escapement goals were not met in 2020–21 or 2021–22, 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), so the 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

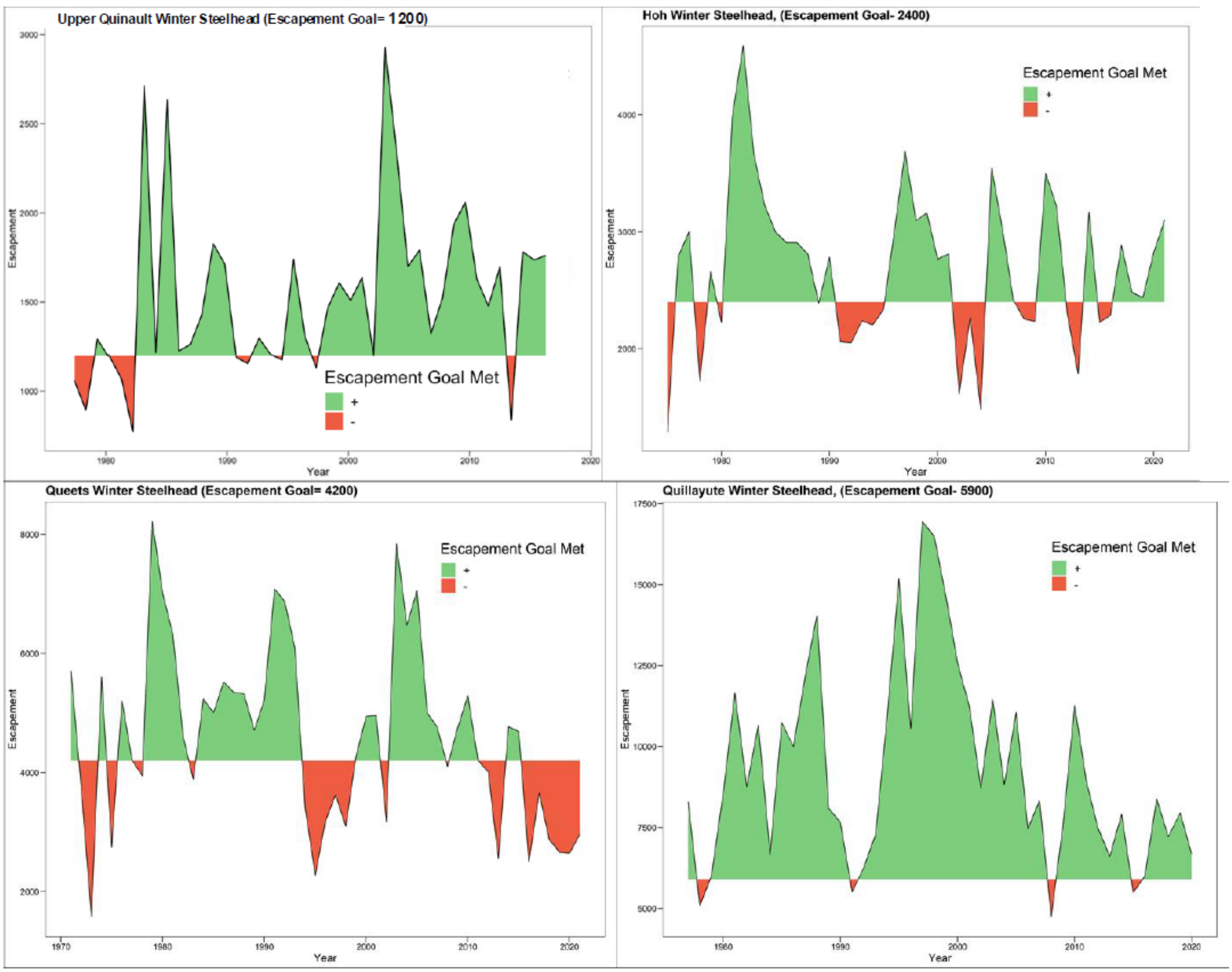

Figure 49. Winter steelhead escapement and escapement goals for the a) Upper Quinault River, b) Queets River, c) Hoh River, and d) Quillayute River. Note: WDFW's escapement goal for the Queets River is 4,200, but the Quinault Tribe's escapement goal is 2,500.

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), and, in general, the escapement goal is more consistently met in the Quillayute (Figure 49). Similarly, for the Hoh River, in 2020 harvest rates were set to 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 systems' harvest rates are still leading to adult returns under the state (but not the tribal) escapement goal in the Queets River.

Forecasting accuracy certainly influences whether harvest rates are set to achieve escapement goals in the OP Steelhead 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" (COPSWG 2023, p. 58). The co-managers provided examples of in-season management and management taken in recent years (Scott, personal communication). Specifically, for the Quillayute River, in-season fishery catch monitoring led to an earlier closure in February 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 were 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, average 104 days in Quinault), but with roughly 50% of harvest levels observed in the 1970s. Between the 2017–18 and 2020–21 seasons, fishing days were again reduced to 78 and 88 days on average in the Queets and Quinault, respectively, and early closures were implemented. Finally, in the most recent seasons (2021–22 and 2023–24), average gill-net 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 (Quillayute, Hoh, Queets, and Quinault Rivers) largely have an intrinsic 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, 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 (θ = 0.83 [0.62, 0.97] mean, [95% CI]; see 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 ONP since 1992 under 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 (Appendix B; see 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 Summer run escapement data), but available information suggests that the harvest of natural-origin summer-run steelhead has declined since the last NMFS review (Appendix B). Further, the SRT 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 to bycatch during their upstream migration to spawn, pre-spawning holding, or as seaward-migrating kelts.[13] Given that summer-run population abundances are inherently smaller, this likely increases the extinction risk for these populations.

Listing Factor C: Disease or predation

Neither disease nor predation effects have been intensively studied for OP 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 group of related viruses (genogroup) of IHNV in the Quillayute, Hoh, Queets, and Quinault River basins (as well as other coastal areas) between 2007 and 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 River watershed at the Salmon River Hatchery (in 1997). Most detections from 2007–11 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 the Columbia River basin; Breyta et al. 2013). The effect of IHNV varied across various streams in Washington, 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 an MD IHNV strain.[14] 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 depend on heritability (see Crozier et al. 2008) and Brieuc et al. (2015) showed that resistance to IHNV is likely heritable. Sockeye salmon are

[13] Bycatch rates depend on the specifics of the gear used, timing, and size/age of steelhead.
[14] M is a phylogenic genogroup of IHNV; sequence types in the M genogroup are designated MA to MF.

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 rivers, like the Quinault, that support a large sockeye run—this could lead to further exposure for steelhead.

Similarly, we obtained data from T. Capps (WDFW, personal communication) on instances of disease, parasites, and viruses in steelhead hatcheries (state, federal, and tribal) on the Olympic 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 River 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 2009–10, with six in December 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 the Bogachiel River in summer 2017. Again, most of the known cases are in hatchery fish populations and not a lot of information exists on impacts to natural-origin steelhead in the OP Steelhead DPS. 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, and 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, whales, cormorants, and/or mergansers on steelhead, including anecdotal accounts of seeing predation by mergansers, otters, and eagles in OP Steelhead DPS 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. Per NWIFC (2020), the following non-native fish species occur in waters of the OP steelhead DPS: 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).

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, as summarized in NMFS (1996). More recently, Caspian terns (*Hydroprogne caspia*) and double-crested cormorants (*Phalacrocorax auritus*) have been documented consuming outmigrating steelhead smolts in the Snake River basin (Hostetter et al. 2015), as have gulls in the Columbia River (Evans 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 OP watersheds, but we are unaware of any unusual or excessive predation events by seabirds or hotspots of seabird predation (T. Good, NWFSC, personal communication).

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*; 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. 2017a,b, Couture et al. 2024), but this work has been mainly focused on predation on Chinook salmon. (Couture et al. 2024 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 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 that OP steelhead along the Strait of Juan de Fuca would migrate through a portion of this area as well, seals are likely impacting steelhead smolt survival to some extent. 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 are 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 U.S. West Coast occurred after marine mammals and many birds were fully protected by law (Cooper and Johnson 1992). Based on this, it would seem unlikely that, in the absence of human 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 and 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. It is possible 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.

Listing Factor 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 toward 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 directed at steelhead. These include both federal and state forest-management plans; 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 2,564 km^2 (990 mi^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. ONP created a General Management Plan in 2008 (USOFR 2008). This plan set desired outcomes for the Park over the course of the next 15–20 years and established management zones within 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 Olympic Peninsula.

A retrospective on 25 years of the NWFP (Spies et al. 2019) reviewed the scientific literature published since the inception of the NWFP and reported several key findings. The NWFP has protected remaining old-growth forests from clear-cutting 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 lands—is not sufficient to achieve recovery (Reeves et al. 2018).

Numerous Washington State regulations also influence steelhead populations in the OP Steelhead DPS. The Forest Practices Act in Washington as well as the Washington State Forest Practices Rules (Washington Administrative Code [WAC] Title 222) establish rules and guidelines for forest management on non-federal lands in Washington State, and that those lands are to be "managed consistent with sound policies of natural resource protection" (Revised Code of Washington [RCW] 76.09.010).[15] 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."[16] The statute (RCW 76.09) and the implementing rules and guidelines (WAC Title 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 (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

[15] https://app.leg.wa.gov/RCW/default.aspx?cite=76.09.010
[16] https://www.dnr.wa.gov/about/boards-and-councils/forest-practices-board/rules-and-guidelines/forest-practices-rules

lands within the range of the northern spotted owl (*Strix occidentalis caurina*), which includes all of the Olympic Peninsula. The Department of Ecology has in-stream flow and water management rules to implement state law requiring that enough water is kept in streams and rivers to protect and preserve in-stream 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 (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 in 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 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 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 only focuses on recreational harvest and state hatchery operations, and does not include the tribal component of harvest nor tribal hatcheries, which currently constitute the majority of landed catch and hatchery production.

A summary document on traditional ecological knowledge (TEK) provided by the Makah Tribe for this status review provides helpful context on fisheries management and biases of certain historic data (Martin 2023). The document notes that sustainable harvest management is a core principle of traditional resource management and embedded into tribe societal roles, that salmon and steelhead have been managed since time immemorial (including their habitat), and that this management included both traditional hatchery and harvest practices. They also highlight that historical documents on harvest from the 1950s to the 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 treaty tribes in the Boldt Case Area and also by ONP. WDFW has jurisdiction over recreational fisheries in Washington State waters and outside of 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. 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 two 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 natural steelhead populations (Harbison et al. 2022). In recent years, recreational fisheries have been closed inside and outside of 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). As noted in Listing 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 Rivers do not prohibit the retention of natural-origin steelhead. Additionally, hatchery steelhead released in the Queets and Quinault Rivers are mostly not marked. State regulations allow for retention of steelhead with a dorsal fin height of less than 2⅛ inches, the height of a credit card—the so-called "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 or 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 et al. 1985). Goals are set based on maximum sustainable harvest, which became a priority after the Boldt Decision (i.e., that tribes and the state will co-manage fisheries and that tribes have the right to half of the catch). More specifically, for the term "escapement goal," Harbison et al. (2022, p. 17) 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 (Harbison et al. 2022, p. 18) notes that "managers assumed that enough wild fish made it past the fishery to spawn (Gibbons et al. 1985)," 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 et al. (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. (1985) is the basis for escapement goals, there is some disagreement among co-managers on the escapement goals for some basins (see Table 4). For the Queets River, the tribal escapement goal differs from that used by the state. The number of spawners needed for maximum sustainable yield (S_{msy}) was calculated separately in the 1980s to be 2,500, but with the caveat that more data were needed (Scott, personal communication). 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. (1985) upon which they are based. WDFW has stated their intention to recalculate escapement goals based on 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 <u>Listing Factor B</u>). This may be due to errors in projected returns. The co-managers did state in their 2023 submission 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 some in-season evaluation of the run relative to forecast (COPSWG 2023, p. 58). Seasons have been shortened/closed early in recent years in response to monitored catches (see <u>Listing Factor B</u>). Additionally, differing escapement goals (e.g., Queets River) may lead to harvest rates that result in adult returns below the escapement goal depending on which escapement goal, state or tribal, is considered. Therefore, in certain years and certain systems, projected abundance may be below a certain escapement goal, harvest may not be at MSY, and escapement levels may not be at the level to maximize future returns. Note that the information we have on meeting escapement goals is for the Big 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 <u>Listing Factor B</u> and <u>Harvest rates</u>.

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 were insufficient data for all summer-run populations to assess trends or extinction risk. In 1992, 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; those regulations include daily limits for steelhead that are caught during summer months. Time-series estimates of harvest for summer steelhead are provided above (see Summer-run steelhead population harvest).

WDFW operation of hatcheries is currently regulated by the Statewide Steelhead Management Plan (SSMP) and 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 U.S. West Coast are primarily operated for harvest augmentation. We outline current potential impacts of hatcheries in Listing Factor E, noting: 1) the use of out-of-DPS-origin broodstock, 2) that not all hatchery fish are adipose fin-clipped, and 3) the possible current proportion of hatchery-origin adults spawning (pHOS) with natural-origin steelhead that are above desired levels.

Listing Factor E: Other natural or man-made factors affecting the species' existence

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 U.S. 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 by masking 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/region.

In Busby et al. (1996), NMFS noted that the estimated proportion of hatchery stocks on natural spawning grounds ranged from 16–44%. This proportion was lowest for the two rivers with the largest production of natural-origin steelhead, the Quillayute and Queets 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 OP Steelhead 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 Petitioners reported a website from WDFW[17] which shows that for 2009, pHOS for summer-run steelhead for the hatchery program on the Bogachiel River was 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 River. In 2012, the Sol Duc River was designated by WDFW as a Wild Steelhead Gene Bank, resulting in the cessation of summer smolt releases in 2011 and winter releases in 2013 (winter-run was local-origin broodstock steelhead; see the Hatchery regulations section in Appendix B).

A recent review by Marston and Huff (2022) looked at compliance by the WDFW-operated Bogachiel Hatchery with standards set in the SSMP. The report also summarized existing hatcheries and then looked at compliance of WDFW-operated programs. It 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, < 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

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

also re-evaluating the 15 March hatchery-origin/natural-origin spawner cut-off date, among 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, p. 85) 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 (2000–08) and 1,072,781 (2009–13). 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 DPS) or Skamania early summer-run steelhead[18] (Lower Columbia River DPS). 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) note 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).

In Hatchery operations in the OP Steelhead DPS, we summarize the hatchery programs and hatchery outputs. Hatchery releases have stayed consistent since the late 1970s and early 1980s to the present, both for winter- and summer-run hatchery output. Depending on the run timing, river, and year, smolt output can range from < 10,000 to > 700,000. Additionally, see Appendix A 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 to hatchery influence (McMillan et al. 2022). Additionally, McMillan et al. (2022) hypothesize 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 transferring stock between watersheds was part of traditional tribal fisheries management. Such movements, most likely between adjacent watersheds, would be akin to returning straying adult steelhead, and would not represent the same level of genetic risk as the cumulative release of millions of steelhead from the Puget Sound or Columbia River hatchery stocks.

[18] The use of Skamania Hatchery and Chambers Creek Hatchery stocks has been eliminated elsewhere on the U.S. West Coast due to negative impacts on listed steelhead. See Ford (2022).

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, p. 11) summarizes potential climate change impacts within the Olympic Peninsula, stating that "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 Olympic Peninsula, 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 (Safeeq et al. 2015, Reeves et al. 2016, 2018). Multiple papers have already documented extensive glacier losses (Riedel et al. 2015, NWIFC 2020, Fountain et al. 2022).

Many of these changes have already been observed on the Olympic Peninsula. 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 show warming water temperatures in April and May in certain recent years.[19] Peak flows (in winter) 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% (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

[19] Washington Department of Ecology 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.

discharges for the OP watersheds draining to the Pacific found that the two-year flood event is 10–35% 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 two-year flood peak calculated for the Hoh River for water years 1978–2013 was 1,024 cubic meters of water per second (cms), whereas the two-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 Olympic Peninsula 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 Ocean in both the northern and southern hemispheres (Castino et al. 2016, East et al. 2018). Summer low flows have decreased over time in the Calawah River basin, where the average low flow in the late 1970s through the 1990s was 2.0 cms, while in the 2000s average summer low flow has been 1.5 cms.

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 west-side OP Steelhead 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,[20] the SRT reviewed projected changes in temperature, flow, and 25-year flood cms for individual rivers/streams between now and 2040 and now and 2080 (Appendix B). 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 independent tributaries may also be affected. The highest temperatures experienced now and likely into the future are predicted to also impact the Lyre River winter-run and Clearwater River summer-run populations.

[20] https://www.fs.usda.gov/rm/boise/AWAE/projects/NorWeST/ModeledStreamTemperatureScenarioMaps.shtml and https://www.fs.usda.gov/rm/boise/AWAE/projects/modeled_stream_flow_metrics.shtml

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 waterfalls 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 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). 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 conditions at other times of year could increase growth rates (Dalton 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.

Within the 2020 State of the Watershed Report, the northwestern 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. The Quileute Tribe report notes 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.[21] This work includes a tool to explore the resiliency of various watersheds.[22] Overall, most of the coastal watersheds in the OP Steelhead DPS range were found to have higher overall

[21] https://www.coastsalmonpartnership.org/current-initiatives/climate-framework/
[22] https://coast-salmon-partnership.shinyapps.io/CRI_app/

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 SRT finalized scoring for the risk assessment, it corroborates that low summer flows will likely impact certain streams within the DPS—though there 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 interacts, 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 factors 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 et al. 2008), it is unknown if OP steelhead can adapt quickly enough to the rapid pace of 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 (2016) states that climate change-driven changes in freshwater ecosystems will be relatively small by mid-century, but that additional changes and challenges may occur in the marine environment. A study by Abdul-Aziz et al. (2011) predicted 8–43% 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, p. 13) 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. 2019)."

The assessment by COPSWG (2023) suggested that interannual variation in recruitment and kelt survival were both partially explained by summer sea surface temperatures (SST)—in addition to pink salmon (*Oncorhynchus gorbuscha*) abundance and North Pacific Gyre Oscillation (NPGO) 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 lower-magnitude fluctuations for Washington coast, on average). There is uncertainty in how smolt survival and recruitment and kelt survival will change over time, but kelt survival has already declined since the 1980s (see Repeat spawner rate and Figure 19). This analysis strongly suggests that ocean survivals are likely to decrease in warm years, and that the frequency of these warm years will increase with climate change.

Competition among salmonid species

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 zooplankton and 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 COPSWG (2023) suggested that interannual variation in recruitment and kelt survival were both partially explained by pink salmon abundance (and also SST and NPGO 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 salmon abundance. We note that the co-manager analysis 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), 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" (p. 165). 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).

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 in recent years (see NWIFC 2020 and CSP 2022) 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 the 2000s and 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, and Anadromous Salmon and Steelhead Hatchery Policy C-3624 (see Listing Factor D and Appendix B). 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.[23] 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 (Stout et al. 2018, Martens et al. 2019), especially as 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 peak flows, elevated water temperatures, and continued loss of glaciers (and melt impacts on 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 Big Four coastal rivers, possibly related to warmer sea surface temperature, pink salmon impacts, and PDO (but there is uncertainty about other potential contributing factors, including predation).

Furthermore, though harvest and hatchery operations have been modified as described above, they continue to have an overall negative influence on steelhead populations within the DPS. Prior to 2021, OP steelhead populations experienced relatively high commercial and recreational fishing pressure (when compared to other DPSes), even while populations declined. There are documented legacy and current impacts associated with harvest. Harvest rates were the highest in the state for the Big Four rivers (13.26–59.19%, depending on year and river, 2014–20) 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 two years (2021, 2022), harvest rates in the Big 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 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-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

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

management changes, there continue 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 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 OP 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 OP Steelhead 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 OP Steelhead DPS [ESU] was 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 [DPS] 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 [DPS], 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 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, p. 166).

Risk assessment

The current SRT has been similarly tasked to assess the status of the OP 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 OP Steelhead DPS used the following categories: "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 DPSes/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, and/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 nor 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 (next section), 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 OP 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, 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) 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 [e.g., 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 was 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 team reviewed the scores and discussed the range of perspectives, 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 DPSes.

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, whether they were at either moderate or high risk of extinction. In doing this, the team followed advice from NMFS WCR and the 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 (USOFR 2014) and subsequent legal rulings.

Based on this advice, this analysis involved identifying and evaluating portions of the DPS that are potentially at moderate or high risk of extinction and are important to the overall long-term viability of the DPS, 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 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 suggests 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—not just those that could be placed into identifiable SPOIRs. Simply put, populations are still important to the overall risk assessment, even if they are 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's 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:

- **Geographic strata:** The OP Steelhead DPS occupies three WDFW WRIAs that occupy the Strait of Juan de Fuca (WRIA 19) and the Washington coast (WRIAs 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 Big Four rivers (the Quillayute, Hoh, Queets, and Quinault Rivers), with higher-elevation headwaters that are glacially fed with rain/snow hydrographs. There are also 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 variations in adult run timing might form the basis for identifying alternative portions. OP 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 the 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 asked each member to 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 its review of the status of the OP 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 15 March 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 15 March, and naturally produced (unmarked) steelhead contribute to this pre-cutoff date production (Marston 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 (i.e, spawning before 15 March) steelhead. 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- and natural-origin spawners prior to 15 March.

Another effect of the post-15 March 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 × native introgression. Although hatchery release practices were 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" is applied to identify hatchery-origin steelhead in state and ONP waters[24]—the majority of the natural-origin harvest is in the commercial fisheries. In

[24] Where hatchery-reared fish are not marked, hatchery origin is determined by the height of the dorsal fin, on the assumption that fin wear in the hatchery during juvenile rearing leaves fish with shorter fins. The height of a credit card—2⅛ in (54 mm)—distinguishes hatchery-origin (shorter fins) from natural-origin (taller fins) steelhead.

tribal waters of the Queets and Quinault Rivers, no distinction is made between hatchery or natural fish in the tribal guide-led recreational fisheries. Although limited information was 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 (November to February) for native winter steelhead. Some members postulated that harvesting the early-returning natural steelhead may affect the spatial distribution of spawners—i.e., 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 has 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 are 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 Big Four basins in this DPS was a risk factor cited by members of the SRT. Combined escapement estimates for the Big Four rivers have decreased by 16%, from 18,597 (1991–95; Busby et al. 1996) to a current level of 15,653 (2018–22); however, total run size has decreased 42%, from 32,556 (1991–95) to 18,821 (2018–22). Additionally, of the 14 populations for which adequate escapement data were available for trend analysis, only one had a stable trend and 13 were negative (ten of them significantly so). Historical harvest levels set for these basins do not appear to be sustainable; although there has been a steady reduction in harvest in the last three to five years, many of the populations have 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- 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 steelhead.

Summer-run steelhead

There was a paucity of data available for summer-run steelhead in the OP Steelhead 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 that 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 within Olympic National Park boundaries, is of high quality. Harvest data are 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 USOFR 2005, 2006a,b). 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 OP Steelhead DPS (USOFR 2006a). While resident fish are known to be present in the watersheds of the Olympic Peninsula, there have been no efforts to quantify their current 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 DPSes, but in the absence of information, the contribution to the OP Steelhead DPS's 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 five of the 15 populations analyzed had negative trends, four of which were significantly different from zero (including the larger Queets and Bogachiel River winter-run populations). Positive trends were observed in eight of the 15 populations, and only two of those were significant—specifically, the smaller Pysht and East Twin River winter populations. Analysis of trends in the total run size, however, suggests declining productivity under varying harvest rates; as stated above, for the Big Four basins, the combined five-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 Big Four basins. It is unclear if this "improvement" in total run size is strictly related to harvest changes or whether improvements in freshwater and ocean conditions has an effect. Overall, the population growth rate (μ) for the Big Four 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; 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 Big Four 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 OP Steelhead DPS lies in a region of the U.S. 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 crossings 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 of 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 2013), 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 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- 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 prior to and following the initiation of hatchery programs in the OP Steelhead DPS.[25] With the beginning of hatchery programs in the DPS utilizing early-returning winter-run steelhead (i.e., Chambers Creek Hatchery stock from southern 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

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

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- and summer-run, was a concern, in that nonadapted genotypes may be integrated into the naturally spawning native population. The co-managers identified three hatchery stocks utilized in the OP Steelhead 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 Chambers 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 indicate 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.

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 River basin, there are few recent specific data on the proportion of hatchery-origin fish on the spawning grounds (pHOS) in OP 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 (the first so designated in Washington State). 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 effects on the native populations.

The effects of hatchery releases relate 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 River basins, where there is some uncertainty regarding the genetic composition of the broodstocks used and whether they are representative of the native population. A few representative genetic samples are available, taken in different years; most are of the hatchery populations rather than the natural populations. Hatchery operations in the Quinault and Queets River 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 fin, 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 Steelhead DPS, the small population abundances presumed for native summer-run steelhead 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, which also helps maintain genetic diversity.

OP Steelhead DPS Risk Assessment

In considering the DPS's overall risk of extinction, the SRT considered a number of factors. First, contemporary census estimates indicate that there are nearly 20,000 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 Big Four basins, escapement has decreased 16% since the last status review. As before, the winter run is 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%[26]) in the Big Four 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 River 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 its 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); however, the current SRT was provided with substantial information to conclude that there was considerable overlap between hatchery- and natural-origin adults.

The management of co-occurring natural and hatchery winter-run populations in the Quillayute, Hoh, Queets, and Quinault River basins has several consequences on the viability estimates for the natural populations. In order to ensure that the hatchery contribution to spawner abundance estimates is minimized, the co-managers use the 15 March threshold for counting natural redds; 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- 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 reduce 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 nonharvested hatchery-

[26] Average of natural harvest/escapement (post-15 March) for Quillayute, Hoh, Queets, and Quinault Rivers, 2016–20; see Figure 31.

origin fish to the hatchery rack. These efforts (segregated broodstocks and eliminating off-station releases) to minimize the interaction between hatchery- 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- 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 River basin) and Quinault Lake Hatchery have more complicated histories, but have been sufficiently influenced by transfers of Chambers Creek stock 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 have 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 have 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 can take decades.[27] Many of the larger basins also had their headwaters in 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 improved habitat and spatial structure, and will continue to do so into the future. In considering future habitat and spatial structure changes, 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 have 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/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 of 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.

[27] 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.

Table 21. Status Review Team risk scores for viable salmonid population criteria. *All populations* represents an unweighted mean and median for all populations (see Appendix C, Table C1 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 into 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

SRT VSP risk scoring

In the unweighted assessment of VSP criteria for steelhead populations in the OP Steelhead DPS (Table 21),[28] 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 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- 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 22), 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 exhibit a rain/snow hydrograph in the DPS. These effects include low summer flows and increased winter flows—especially given 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 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

[28] Populations were weighted equally in the computation of scores for each category, regardless of abundance.

Table 22. Status Review Team risk scores for threats. *All populations* represents an unweighted mean and median for all populations (see Appendix C, Table C2 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 into 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

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

SRT votes	Low	Medium	High
Average	4.00	5.50	0.50
Median	4.00	5.50	0.00
Range	2, 6	4, 7	0, 2

persistence of summer-run steelhead. Hatchery effects and habitat loss and destruction were also identified as threats, but to a lesser extent than harvest 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 cleaning and timber harvest activities continue. Finally, disease and predation was considered a low risk, primarily related to hatchery operations.

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 23). 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—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

Table 24. Scoring for SPOIR using portions based on run-timing life-history strategies: summer- 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**			**Winter-run significant**		**Winter-run risk**		
Yes	**No**	**L**	**M**	**H**	**Yes**	**No**	**L**	**M**	**H**
4.1	5.9	0.6	4.3	5.1	9.3	0.7	5.6	4.3	0.1

Juan de Fuca were stable, but at absolute abundances that are very low (< 100),[29] despite the termination of harvest in most of the basins over a decade ago. While habitat condition was generally good and restoration efforts have 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 nonselective harvest may reduce life-history diversity and limit the ability of these populations to respond to environmental changes.

Significant portion of its 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 24). 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- 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-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

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

Table 25. Scoring for SPOIR using portions based on geography: Strait of Juan de Fuca (*JDF*) and coastal populations. Members scored each portion for significance and risk level, assigning 10 likelihood points to each question for each portion.

Scenario 2: Geographic									
Strait JDF significant		Strait JDF 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

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 greatly increase risks to DPS abundance, productivity, diversity, and spatial structure (most rivers in the DPS contain only winter-run steelhead), and that 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 was: 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 of risk was not higher than that of the overall DPS.

The SRT also discussed and assessed a SPOIR scenario based on geography. In this case, the geographic units included: 1) steelhead populations in rivers that drain to the Strait of Juan de Fuca, and 2) steelhead populations in rivers that drain to the Pacific Ocean. These two regions were identified as potential portions due to the hydrological and geographic distinctiveness of the rivers supporting Strait 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 25). The SRT considered that populations in the Strait 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 likelihood points for risk in the moderate category (6/10), with 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 increased 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 scored 4.7/10, and high risk scored 0.3/10, suggesting 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, and 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.

The OP 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 OP Steelhead DPS. Following completion of the SPOIR process, the SRT reconfirmed the moderate risk of extinction for the OP Steelhead DPS.

The total abundance of steelhead in the DPS was relatively high compared to other DPSes at moderate risk, but the relatively high risk scores estimated for summer-run populations were a factor in the VSP risk analysis, especially for diversity. Further, analyses by the co-managers and the SRT of run sizes for the Big Four winter-run populations suggest that overharvest 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 presents a continued risk to the natural populations, and continued management (harvest, post-15 March redd surveys) under the concept of temporal separation between hatchery and natural stocks is 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 change effects in the next 40–50 years will be increasingly deleterious to steelhead populations in the OP Steelhead DPS.

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Appendix A: Olympic Peninsula Steelhead DPS Watershed Summaries

This appendix has been published as a separate document in the NOAA Processed Reports series. It will be available for download from the NOAA Institutional Repository in mid-December, 2024, at https://doi.org/10.25923/q5p7-0w19.

Appendix B: Review of ESA Listing Factor Threats for the Olympic Peninsula Steelhead DPS, 1996–Present

This appendix has been published as a separate document in the NOAA Processed Reports series. It will be available for download from the NOAA Institutional Repository in mid-December, 2024, at https://doi.org/10.25923/vs8f-gj05.

Appendix C: Status Review Team Scoring

Table C1. Status Review Team scoring (averages) of viable salmonid population criteria for steelhead populations in the OP Steelhead DPS. Scores represent the average of all team members voting on a 1 (low) to 5 (high) risk range. *W* = winter run, *S* = summer run.

Population	Run	Abundance	Productivity	Spatial structure	Diversity
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
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 C2. Status Review Team scoring (averages) of factors for decline threats for steelhead populations in the OP 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.

Population	**Run**	**Habitat loss or destruction**	**Over utilization**	**Inadequate regulation**	**Disease/. Predation**	**Hatchery effects**	**Climate change**
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
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		2.1	3.0	3.1	1.4	2.6	3.3
Strait W		2.3	1.8	2.3	1.0	1.7	2.8
Summer (5)		2.1	2.7	3.8	1.1	2.2	3.8

Appendix D: Hatchery Releases

Table D1. Releases of steelhead juveniles, by watershed. Releases of steelhead juveniles less than 2 g 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	Broodyear(s)
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
	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

Table D1 (continued). Releases of steelhead juveniles by watershed.

Release watershed	Run	Source	Number released	Broodyear(s)
Lyre River	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
	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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