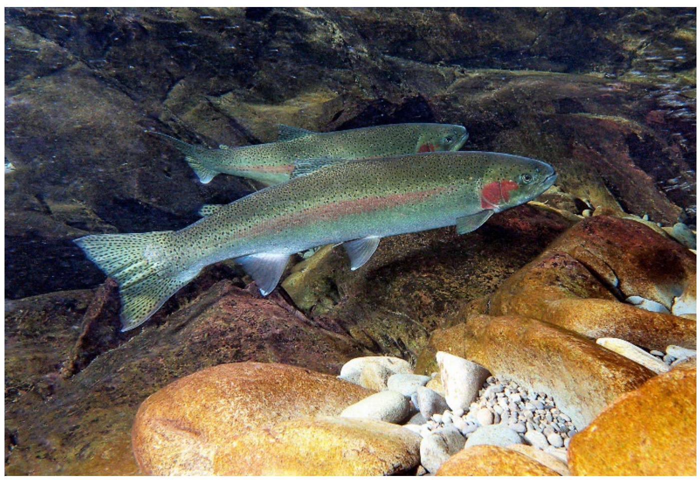
PETITION TO LIST OLYMPIC PENINSULA STEELHEAD (*Oncorhynchus Mykiss*) AS A THREATENED OR ENDANGERED SPECIES



August 1, 2022

<u>Petitioners</u>

The Conservation Angler

Wild Fish Conservancy
NORTHWEST
SCIENCE EDUCATION ADVOCACY

NOTICE OF PETITION

August 1, 2022

The Honorable Gina Raimondo
Secretary of Commerce
U.S. Department of Commerce
1401 Constitution Ave. NW
Washington, D.C. 20230
TheSec@doc.gov
via email

The Honorable Janet Coit
Assistant Administrator for Fisheries
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Dear Secretary Raimondo:

Pursuant to Section 4(b) of the Endangered Species Act (16 U.S.C. § 1533(b)), Section 553(e) of the Administrative Procedures Act (5 U.S.C. § 553(e)), and 50 C.F.R. § 424.14(a), The Conservation Angler and Wild Fish Conservancy (the "Petitioners") hereby petition the Secretary of Commerce, through the National Marine Fisheries Service ("NMFS"), to list the Olympic Peninsula Steelhead Distinct Population Segment (*Oncorhynchus mykiss*) as a threatened or endangered species. Petitioners also request that NMFS designate critical habitat for Olympic Peninsula steelhead concurrent with the distinct population segment being listed as threatened or endangered.

The Conservation Angler (TCA) is a Washington-based, nonprofit, public interest organization that uses scientific expertise and legal advocacy to protect and conserve wild steelhead, salmon, trout, and char throughout their Pacific range. TCA also operates the Kamchatka Steelhead Program, an international research program that has advanced scientific knowledge of wild steelhead for over 25 years.

Wild Fish Conservancy (WFC) is a Washington-based, nonprofit, public interest organization that uses science, education, and advocacy to promote technically and socially responsible habitat, hatchery, and harvest management to better sustain wild fish heritage in Washington State.

NMFS has jurisdiction over this petition. This petition sets in motion a specific process, placing definite response requirements on NMFS. Specifically, NMFS must "make a finding as to whether the petition presents substantial scientific or commercial information indicating that the petitioned action may be warranted." 16 U.S.C. § 1533(b)(3)(A). NMFS must make this finding "[t]o the maximum extent practicable, within 90 days after receiving the petition." *Id.* Petitioners need not demonstrate that the petitioned action *is* warranted. Rather, Petitioners must only present information demonstrating that such action *may* be warranted. While Petitioners believe that the best available science demonstrates that listing Olympic Peninsula steelhead as threatened or endangered is in fact warranted, there can be no reasonable dispute that the available information indicates that listing the species as either threatened or endangered *may* be warranted. As such, NMFS should promptly make a positive initial finding on the petition and commence a status review as required by 16 U.S.C. § 1533(b)(3)(B) and 50 C.F.R. § 424.14(h)(1)-(2).

Thank you.

THE CONSERVATION ANGLER

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EXECUTIVE SUMMARY

Olympic Peninsula steelhead are at risk of becoming an endangered species within the foreseeable future. The summer-run component is nearly extinct, and the winter-run component is declining and losing its life history diversity. The fate of the species now rests on a depressed and contracted mid- to late-spring component of wild fish whose productivity is limited or declining depending on the population. The remnants of these runs that historically numbered in the tens of thousands face declining freshwater and marine habitat conditions, increasing recreational fishing pressure, and ongoing commercial harvest. Because of these and other demographic and ecological threats, Olympic Peninsula steelhead are likely to become endangered within the foreseeable future. Olympic Peninsula steelhead warrant protection under the Endangered Species Act (ESA).

Abundance

Almost every population of winter steelhead on the Olympic Peninsula is in long-term decline over their period of record. For example, mean annual run sizes from 1980 – 2017 declined by 37-46% in the Hoh, Queets, and Quinault Rivers (McMillan et al. 2022), resulting in the Hoh and Queets River populations increasingly failing to meet their escapement goals. The Quillayute River system population has declined sharply since the 1990s (McMillan et al. 2022). All populations have continued to decline since 2017 and run sizes have been so small in recent years that fisheries were closed. Trends appear similar for the smaller populations, with some not meeting their escapement goals for decades, though the data sets are not always as extensive. These patterns underscore why the Olympic Peninsula DPS has the second lowest proportion of populations with increasing trends in Washington State (Cram et al. 2018).

The plight of the winter runs is even more revealing when compared to historical estimates from circa 1940-1960. Based on those historical estimates (McMillan et al. 2022), the Quillayute, Queets, Hoh, and Quinault River wild winter steelhead populations have declined by 61%, 69%, 79%, and 81%, respectively, in relation to their most recent five-year mean run size. Those estimates do not necessarily capture the full extent of decline, however, because they came after decades of harvest and many years of habitat alterations. When compared to cannery data in 1923, for instance, the decline in the Queets River increases to 86%. Hence, it is likely that all the populations are now only a fraction of their former abundance.

The status of the summer-run component is dire. Summer steelhead are not monitored or managed, but snorkel surveys by Brenkman et al. (2012) and McMillan (2022) suggest populations in the Quillayute, Hoh, and Quinault Rivers have declined dramatically and are at critically low levels of abundance, if not already functionally extinct. Unfortunately, almost no information exists for summer steelhead in the Queets River. The long-term, chronic declining trends in winter steelhead coupled with the dire plight of summer steelhead puts the Olympic Peninsula Steelhead DPS at greater risk of extinction than when Busby et al. (1996) conducted the last status review for the DPS (ESU at the time).

Diversity

The diversity of Olympic Peninsula steelhead has been altered in different ways that could impact their productivity, resilience, and capacity to adapt with changing climate effects. First, the early-returning component of winter steelhead populations in the Quillayute, Hoh, and Queets Rivers is severely depleted and consequently, the breadth of run timing is now far more compressed than it was historically (McMillan et al. 2022). This change in run timing was not accounted for in the prior status review (Busby et al. 1996), but it could be critical to their persistence. Steelhead enter and spawn earlier in the winter in warmer, more southerly regions of their native range (Busby et al. 1996), and as climate impacts progress, winter steelhead on the Olympic Peninsula will also need to enter and spawn earlier to keep pace with climate change. Run timing provides one way in which salmonids can adapt to changes in stream flow and water temperature (Manhard et al. 2017), but that could be impossible for winter steelhead in the Olympic Peninsula DPS if their original run timing is not restored.

Second, the data on repeat spawners suggest that rates of iteroparity have declined in most of the four major populations over their period of record. The Queets River population declined from a high of over 50% down to lows of approximately 10% in recent years, while the Quillayute River peaked at over 20% in the late-1970s and is now down to around 10%. The Quinault River population declined from 10-15% to approximately 1-2% in the last couple years. Repeat spawning rates do not show a declining trend in the Hoh River, but that is likely because its period of record does not extend back to the late-1970's to 1980 as it does in the Queets and Quillayute Rivers.

Third, summer runs are not monitored or managed, but they are caught in fisheries. Snorkel surveys by Brenkman et al. (2012) and McMillan (2022) indicate summer steelhead are at critically low levels of abundance in the Quillayute, Hoh, and Quinault Rivers, and they are exposed to high levels of stray hatchery summer steelhead. There is a strong heritable basis for the early maturing summer run life history and such allelic variants do not arise independently via new mutations (Prince et al. 2017; Waples et al. 2022). Given their reduced abundance, it is likely the populations are already genetically compromised, but they are still producing some adults. If the declines continue and summer runs go extinct, the genetic basis for the life history could be lost entirely, leaving behind a massive void in the timing of migration of the overall steelhead populations in the Quillayute, Hoh, Quinault, and Queets Rivers.

Productivity

Information on productivity is very limited for steelhead in the Olympic Peninsula DPS, but what data is available suggests that productivity is declining, and recruitment has increasingly failed to reach replacement levels (Cram et al. 2018). The most recent productivity estimates (Cram et al. (2018) only account for data up to 2010, so they miss many of the worst returns on record that have occurred most recently. Considering the increasingly low levels of productivity in the Quillayute River population, and that peaks and troughs in annual run size in the Hoh, Queets, and Quinault River populations have generally declined each decade over the period of record (lower peaks, deeper troughs) leading to high levels of depletion in recent years, it appears the populations are no longer compensating as effectively as they did 20-40 years ago. If a declining productivity trend continues at the rates estimated by McMillan et al. (2022) and summer runs decline further, which seems conservative given the recent and rapid declines in run size, winter steelhead run sizes will soon be too small to allow fisheries and summer steelhead will be extinct.

Spatial structure

While there are no large dams affecting Olympic Peninsula steelhead, there is a dendritic network of roads and culverts constructed for forest practices, many of which cross salmon bearing streams and create blockages to upstream migration (Smith 2000). Juvenile steelhead are often distributed right up to barriers in many small creeks (McMillan and Starr 2008; McMillan et al. 2013), so the blockages have likely truncated their distribution. It is likely the spatial distribution has been reduced or altered due to the depletion of early-returning winter steelhead and the very low abundance of summer steelhead. Those components of the steelhead population spawn in habitats that are otherwise used less or not at all by other life histories (e.g., Cederholm 1983; McMillan et al. 2007). Hence, changes in run timing of winter steelhead and depletion of summer steelhead, combined with barriers, have all plausibly reduced the populations' spatial distributions.

Harvest impacts

Several winter steelhead populations in the Olympic Peninsula DPS support intensive recreational and commercial fisheries, and they experience the highest harvest levels of any steelhead populations in Washington State (Cram et al. 2018). Harvest has had two apparent effects. First, it has increasingly reduced spawning abundance below management goals. For instance, due to overharvest, winter steelhead in the Hoh and Queets River have failed to meet their minimum escapement goals for abundance in 50% of the years since 2003 and 50% of the years over the past decade, respectively. The Quillayute River system population has generally met its overall escapement goal, but individually, the Bogachiel and Sol Duc Rivers have failed to meet their goals in 60% and 70% of the last ten years, respectively. Unlike the larger populations, many smaller populations no longer experience direct harvest, but nearly all of them have also failed to meet their escapement goals, including some that have not met their goals for 20 years or longer.

Second, the strong focus on maximizing harvest of early-timed hatchery winter steelhead from late-November through early-January has apparently contributed to depletion of early-returning wild winter steelhead that were formerly very abundant in the 1940's – 1950's (McMillan et al. 2022). As a result, the breadth of run timing is now more compressed than it was historically. These new results challenge the claim by Busby et al. (1996) that hatchery and wild winter steelhead were temporally segregated due to differences in run timing. Rather, the hatchery fish have simply replaced the wild fish, underscoring the potential for a Shifting Baseline (Pauley 1995). This should not be a surprise, however, because according to the traditional ecological knowledge of the Quileute Tribe, the month of January meant the "time of steelhead running" and February the "time of steelhead spawning" (Frachtenberg 1916).

Hatchery impacts

Hatcheries can exert a range of genetic and ecological impacts on wild salmon and steelhead (Araki and Schmid 2011), and each year, large numbers of hatchery steelhead are released into several streams in the Olympic Peninsula DPS with the largest number of fish being released into the Quinault River (Duda et al. 2018). Unfortunately, hatchery effects are relatively unstudied, and their impacts are therefore uncertain in the DPS. However, as mentioned earlier, fisheries focused on maximizing harvest of hatchery steelhead and interbreeding between hatchery and wild steelhead have both likely contributed to the depletion of early-timed wild winter steelhead (McMillan et al. 2022). For example, Seamons et al. (2012) found that after three generations of stocking segregated hatchery winter steelhead the proportion of wild ancestry smolts and adults declined by 10-20% and up to 80% of naturally produced winter steelhead were hatchery x wild hybrids. Whatever the relative contributions of each factor are, it is clear that hatchery and wild steelhead once overlapped substantially, and the depletion of wild fish coincides with the onset of the hatchery programs.

There are also concerns about overall genetic impacts due to the number of hatchery steelhead that stray and spawn in nature. Cederholm (1983) noted there was a high degree of within and between river straying of hatchery steelhead, which could change the long-term spawning, timing, growth, and survival of wild fish. Similar concerns were raised in the status review by Busby et al. (1996) because of the "widespread production of hatchery steelhead" in the DPS. Indeed, a review in 2009 indicated that hatchery steelhead greatly exceeded their pHOS goals for winter and summer steelhead in the Bogachiel River (WDFW 2022b). And WDFW suggests that hatchery winter and summer steelhead pose substantial risks to among-population diversity and fitness of wild steelhead due to introgression that has likely occurred in the Pysht, Hoko, and Sol Duc Rivers (WDFW 2008).

The record shows that concerns by Cederholm (1983) and others were warranted, particularly for summer steelhead. As evidenced by snorkel surveys by Brenkman et al. (2012) and McMillan (2022), a substantial number of hatchery summer steelhead stray into the Hoh and Quinault Rivers, both of which do not receive any releases of hatchery summer steelhead. There is also a substantial amount of straying within the Quillayute River system, particularly into the upper Calawah and Bogachiel Rivers (Brenkman et al. 2012). The high levels of pHOS, which frequently reach and even exceed 30-50%, combined with very low abundance of wild summer steelhead, suggest that hatchery impacts could be a significant limiting factor.

Changes to Freshwater and Marine Habitat

The depleted stocks of winter and summer steelhead in the Olympic Peninsula DPS are threatened with a myriad of environmental challenges, including degraded freshwater habitat, climate impacts, and a changing ocean. Although the largest watersheds in the DPS have their headwaters in Olympic National Park, a long history of extensive and unsustainable logging has greatly degraded freshwater habitat outside of the park. Logging and roadbuilding have increased the frequency of mass wasting events, which cause excessive sedimentation of stream channels, diminish supplies of large woody debris, impair water quality, increase the frequency of peak flow events, and reduce habitat connectivity (Smith 2000; East et al. 2017). Climate change is predicted to degrade freshwater habitat even further, contributing to an earlier onset of warmer summer water temperatures, more extreme fluctuations in stream flow, and eventual elimination of all the glaciers that feed the largest, most productive watersheds (Wade et al. 2013; East et al. 2017; Fountain et al. 2022). Climate impacts will also alter the productivity potential of the marine environment where adult steelhead spend a substantial portion of their lives. For example, sea surface temperatures are warming and will continue to do so, while alterations to ocean currents could alter the extent of upwelling and there is the potential for increased acidification and changes in the food web (Klingler et al. 2008; Miller et al. 2013; Dalton et al. 2016). Each of these changes, past and future, singular and cumulative, are expected to negatively affect the survival and productivity of steelhead if they do not have sufficient abundance, diversity, and spatial structure to adapt and keep pace with climate change.

Summary

This petition demonstrates that the Olympic Peninsula Steelhead DPS warrants protection under the ESA. It is divided in two parts. Part One covers the Olympic Peninsula Steelhead DPS's description, taxonomy, life history, distribution, and population status. Part Two describes the current and future threats to the DPS in the context of the ESA's five listing factors. Based on the substantial information provided herein, NMFS should list Olympic Peninsula steelhead as a threatened or endangered species and designate its critical habitat.

LEGAL BACKGROUND

The Endangered Species Act

The ESA defines "species" to mean "any subspecies of fish or wildlife or plants, and any *distinct population segment* of any species of vertebrate fish or wildlife which interbreeds when mature." 16 U.S.C. 1532(16) (Emphasis added).

Olympic Peninsula steelhead are a distinct population segment (DPS) of steelhead (Busby et al. 1996; 61 Fed Reg 41544 (Aug. 9, 1996)). Genetic data from the Washington Department of Fish and Wildlife (WDFW) indicate that the Olympic Peninsula steelhead DPS is "substantially isolated from other regions in western Washington" (Busby et al. 1996). In addition to genetic differences, Olympic Peninsula steelhead are further characterized by habitat, climatic, and zoogeographical differences between it and adjacent DPSs, including the Southwest Washington and Puget Sound DPSs (Busby et al. 1996). NMFS delineates the Olympic Peninsula steelhead DPS to include populations that occur in river basins to the west of the Elwha River and south to, but not including, the rivers that flow into Grays Harbor (Busby et al. 1996; 61 Fed Reg 41544 (Aug. 9, 1996)).

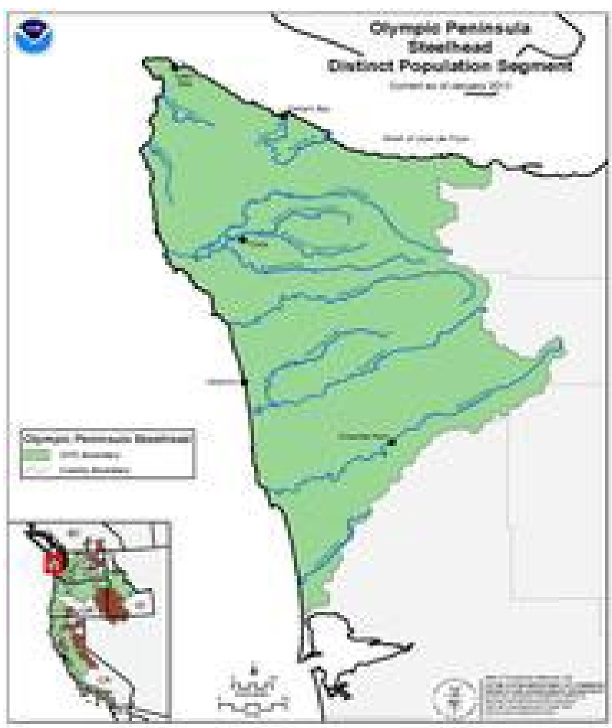


Figure 1. Map of the Olympic Peninsula Steelhead Distinct Population Segment (Source: NMFS).

When making a listing determination, NMFS must analyze the status of a species in conjunction with five statutory listing factors, relying "solely on the best scientific and commercial data available." 16 U.S.C. § 1533(b)(1)(A). The five listing factors include:

1. The present or threatened destruction, modification, or curtailment of its habitat or range;
2. Overutilization for commercial, recreational, scientific, or educational purposes;
3. Disease or predation;
4. The inadequacy of existing regulatory mechanisms; and
5. Other natural or manmade factors that affect its continued existence.

Id. at § 1533(a)(1).

Under the ESA, a species is "endangered" if it "is in danger of extinction throughout all or a significant portion of its range." *Id.* at § 1532(6). A species is "threatened" if it is "likely to become an endangered species within the *foreseeable future* throughout all or a significant portion of its range." *Id.* at § 1532(20) (Emphasis added).

The Foreseeable Future and Climate Change

The foreseeable future extends as far into the future as NMFS can reasonably determine that both future threats and the species' responses to those threats are likely. 50 C.F.R. § 424.11(d). When analyzing the threats of climate change, it is NMFS policy to "project effects over the longest possible period for which credible projections are available in order to ensure the best available science is fully considered" (Tortorici 2016).

NMFS has looked out as far as the end of the 21st Century when making listing determinations (Tortorici 2016). For example, the Ninth Circuit held that it was not arbitrary or capricious for NMFS to list the ringed seal based on climate change models that projected as far out as year 2100. Alaska Oil & Gas Ass'n v. Nat'l Marine Fisheries Serv., 722 F. App'x 666, 669 (9th Cir. 2018) quoting Alaska Oil & Gas Ass'n v. Pritzker, 840 F.3d 671, 680 (2016); *see also* Alaska Oil & Gas Ass'n v. Pritzker, 840 F.3d 671 (9th Cir. 2016), cert denied 138 S. Ct. 924 (2018) (upholding NMFS's decision to list bearded seals as threatened based on climate change models that predicted that the sea ice the seals depend on for birthing and mating would mostly disappear by 2095) and Safari Club International v. Salazar (In re Polar Bear Endangered Species Act Listing and § 4(d) Rule Litigation) 709 F.3d 1 (D.C. Cir. 2013), cert denied 571 U.S. 887 (2013) (upholding the U.S. Fish & Wildlife Service's decision to list the polar bear as threatened based, in part, on projected climate change effects to the species and its habitat 45 years in the future).

As an example of the feasibility of a 100-year time frame, the Intergovernmental Panel on Climate Change ("IPCC") provides climate change projections through 2100 under a range of plausible emissions scenarios (IPCC 2021). NMFS recognizes the IPCC as a credible information source. The Service's guidance on climate change and ESA determinations requires it to "use climate indicator values projected under the [IPCC's] Representative Concentration Pathway 8.5

when data are available" (Tortorici 2016). For these reasons, the use of 100 years as the foreseeable future is consistent with NMFS's climate change policy.

As indicated by NMFS's policy on climate change and listing decisions (Tortorici 2016), the best available science standard does not require NMFS to be certain about climate change and its effects on Olympic Peninsula steelhead.

> "While it requires that decisions not be based on mere generalizations or speculation, the best available science standard does not require that information be free from uncertainty. For example, to support listing a species on the basis of climate change related impacts, we must have information particular to that species to demonstrate that it will be impacted by climate change, such as through a reduction of suitable habitat within its known range. It is not necessary, however, to have projections at a particular geographic scale or to have a complete understanding of the biological reasons for and extent of the species' sensitivity to climate change."

(Tortorici 2016). This petition supports listing Olympic Peninsula steelhead for multiple reasons, including climate change-related impacts. The best available science, including NMFS's own reports, demonstrates that the species is impacted by climate change.

PART ONE

THE STATUS OF OLYMPIC PENINSULA STEELHEAD

Threats to a species' long-term persistence are manifested demographically as risks to its abundance, population growth rate, spatial structure and connectivity, and genetic and ecological diversity. These demographic risks thus provide the most direct indices or proxies of extinction risk. A species at very low levels of abundance and with few populations will be less tolerant to environmental variation, catastrophic events, genetic processes, demographic stochasticity, ecological interactions, and other processes (e.g., Meffe and Carroll 1994, Caughley and Gunn 1996). A population growth rate that is unstable or declining over a long period of time indicates poor resiliency to future environmental change (e.g., Lande 1993, Middleton and Nisbet 1997, Foley 1997). A species that is not widely distributed across a variety of well-connected habitats is at increased risk of extinction due to environmental perturbations, including catastrophic events (Schlosser and Angermeier 1995, Hanski and Gilpin 1997, Tilman et al. 1997, Cooper and Mangel 1999). A species that has lost locally adapted genetic and ecological diversity may lack the raw resources necessary to exploit a wide array of environments and endure short- and long-term environmental changes (e.g., Groot and Margolis 1991, Wood 1995). Assessing extinction risk of a species involves evaluating whether risks to its abundance, population growth rate, spatial structure and or diversity are such that it is at or near an extinction threshold, or likely to become so in the foreseeable future. As demonstrated by this petition, Olympic Peninsula steelhead are likely to become an endangered species within the foreseeable future and, therefore, should be listed as a "threatened" species under the ESA.

I. DESCRIPTION

Olympic Peninsula steelhead are a distinct population segment of steelhead (Busby et al. 1996; 61 Fed Reg 41544 (Aug. 9, 1996)). Steelhead are the anadromous form of *Oncorhynchus mykiss* and may display the most diverse life histories of any salmonid (Kendall et al. 2015). As with other salmonids, they begin their life cycle in freshwater where they spend 1-4 years growing as parr, which are relatively drab and natural in coloration to match their surrounding stream environments (Busby et al. 1996). After reaching a growth threshold in size, typically between 150 – 200 mm in length, they undergo a complex series of physiological changes, such as increased levels of NA+/K+-ATPase in the gills, that assist with osmoregulation and life in saltwater (Busby et al. 1996). As their internal physiology changes, so does their outward appearance. Their body becomes more fusiform, their scales become more deciduous, and they take on a silvery appearance (Busby et al. 1996).

Steelhead are known to display three general categories of ocean migrations. There are typically anadromous fish that spend 1-4 years in the ocean before returning to spawn in freshwater, attaining sizes of 45 – 1125 mm in length and 0.9 – 20.5 kg in weight (Kendall et al. 2015). There are also half-pounders that only undertake a short, near-shore ocean migration for a few months attaining a size of 25 – 40 mm in length and a weight of 0.1 – 0.7 kg, but generally return in an immature state and will undertake a full ocean migration the next year (Hodge et al. 2014; Kendall et al. 2015). Last, there are individuals that undertake short migrations and attain sizes intermediate to the fully anadromous and half-pounder life histories but are mature upon return to freshwater and are typically male (Kendall et al. 2015). The latter are referred to as "estuarine" or "jack" life histories, potentially depending on what part of the ocean they migrate to (Kendall et al. 2015). While the fully anadromous form is common throughout the native range of steelhead, far less information exists on the distribution of the half-pounder life history except for a few rivers where they are abundant in California and southern Oregon, and almost no information exists on estuarine or jack life histories outside of the Kamchatka Peninsula, Russia (Kendall et al. 2015).

Adults that return to freshwater to spawn are further delineated into two run types, including ocean-maturing "winter run" and river-maturing "summer run" life histories (Busby et al. 1996). Olympic Peninsula steelhead include winter and summer run life histories. Winter runs enter freshwater sexually mature or close to sexual maturity from late-October through early-June, while summer runs enter sexually immature from May through October and sexually mature as they stage in freshwater. The physiological processes responsible for osmoregulation will reverse to allow the fish to persist in freshwater and they begin to display signs of secondary sexual characteristics as they approach the spawning season from late-winter through spring to early-summer. Males become much darker shades of green and red, while females become less silvery and display various shades of red and light green. Both sexes may develop fungal infections due to stress, fights, and compromised immune systems.

Individuals that survive the rigors of mating and spawning will once again – if in sufficient condition and health – restart the smoltification process in preparation for saltwater entry (Buelow and Moffit 2015). During emigration to the ocean their outward appearance will also change to become more silvery as they did when they were smolts for the first time.

II. TAXONOMY

Kingdom:	Animalia
Phylum:	Chordata
Class:	Osteichthyes
Order:	Salmoniformes
Family:	Salmonidae
Genus:	*Oncorhynchus*
Species:	*Oncorhynchus mykiss*

III. HABITAT AND RANGE

The Olympic Peninsula is a large arm of land located in western Washington. It is bordered by Hood Canal to the east, the Strait of Juan de Fuca to the north, and the Pacific Ocean to the west. The Olympic Mountains are its natural centerpiece. These coastal mountains reach 1,200 to 2,400 meters above sea level and currently hold 184 glaciers (Busby et al. 1996; Reidel et al. 2017). The area receives copious rainfall (McHenry et al. 1996). The west side of the peninsula receives the most precipitation, ranging from 70 to 100 inches in the lower coastal plains (Dalton et al. 2016) and up to 240 inches in the mountains (McHenry et al. 1996). Olympic National Park protects the only temperate rainforest in the contiguous United States (Dalton et al. 2016). The park's foothills are dominated by undisturbed old western hemlock (*Tsuga heterophylla*), douglas fir (*Pseudotsuga menziesii*), and sitka spruce (*Picea sitchensis*) (Dalton et al. 2016). Outside of the park, the forest has been extensively logged (McHenry et al. 1996). Running through this dichotomous landscape of pristine and degraded habitat are some of the last large, undammed rivers in the Pacific Northwest.

Olympic Peninsula steelhead occur in three distinct water resource inventory areas (WRIA): WRIAs 19, 20, and 21 (Table 1).

A. Water Resource Inventory Area 19

WRIA 19 begins immediately west of the Elwha River and extends eastward to Cape Flattery, the northwesternmost point in the lower forty-eight. The area experiences a cool maritime climate with annual precipitation ranging from 80" to 130" (HSRG 2004). Precipitation is higher on the west side of WRIA 19 (NOPLE 2015). The Sekiu River basin on the western edge of WRIA 19 receives 95-120 inches of precipitation per year (NOPLE 2015). The Salt Creek basin on the eastern edge receives 35-55 inches of precipitation annually (NOPLE 2015).

The forestland has a mix of tree species that vary east to west. In the eastern portion of WRIA 19, Douglas fir (*Psuedotsuga mensiezii*) is the dominant species, with red alder (*Alnus rubra),* vine maple (*Acer cirnicatum*), and bigleaf maple (*Acer macrophylla*) also occurring in the area (NOPLE 2015). In the western portion of WRIA 19, the forests are dominated by western hemlock (*Tsuga heterophylla*) and Sitka spruce (*Picea sitchensis*) (NOPLE 2015).

Commercial forestry accounts for 76% of land use in WRIA 19 (NOPLE 2015). Private interests own 56% of the commercial forestland, the Washington Department of Natural Resources (WDNR) owns 28%, the U.S. Forest Service (USFS) owns 12%, and the remaining 4% is owned by the county and small landowners (NOPLE 2015). The non-commercial forestland occurs inside Olympic National Park (11.6%) or is classified as rural, urban, industrial, tribal reservation, or miscellaneous (12.4%) (NOPLE 2015).

Steelhead are one of the more widely distributed salmonids in WRIA 19 (NOPLE 2015; Table 1). Winter steelhead occur in several small to medium-sized watersheds, including the following subbasins listed in order of size: Hoko River (71 sq mi), Lyre River (67.9 sq mi), Pysht River (46.3 sq mi), Sekiu River (33.2 sq mi), Clallam River (31 sq mi), Salt Creek (19.1 sq mi), East Twin River (13.6 sq mi), and West Twin River (12.6 sq miles) (NOPLE 2015). The Western Strait Independents collectively drain 73.3 square miles (NOPLE 2015).

Summer steelhead may occur in WRIA 19 (Table 1). Summer steelhead are thought to occur in the Lyre River (McHenry et al. 1996; Lyre-Hoko Watershed (WRIA 19) Planning Unit 2008) and may occur in the East and West Twin Rivers, Murdock Creek, Fielding Creek, and Colville Creek (WRIA 19 Watershed Plan 2008).

The Lyre River is the only river in WRIA 19 that originates in an alpine area (approx. 5,500 feet) (NOPLE 2015). All other rivers and streams in WRIA 19 drain from low elevation foothills ranging 2,000 to 3,500 feet in elevation (NOPLE 2015).

B. Water Resource Inventory Area 20

WRIA 20 begins at Cape Flattery and extends south to, but not including, Kalaloch Creek (HSRG 2004). The area receives 80” to 240” of rain per year (McHenry et al. 1996). Wind and heavy rainstorms are common (HSRG 2004). Inside Olympic National Park rests an undisturbed temperate rainforest, which includes old-growth Sitka spruce, western hemlock, and silver fir (*Abies alba*) (HSRG 2004). Outside of the park there is significant habitat disturbance, due in large part to extensive clearcutting and roadbuilding that occurred during the 1960s-1980s (HSRG 2004). Commercial forestry is the predominant land use in the lower reaches of WRIA 20 rivers (NPCLE 2020).

Olympic Peninsula steelhead occur in several large and small systems within WRIA 20 (Table 1). The winter and summer-run components of the Olympic Peninsula DPS are found in the Hoh, Quillayute/Bogachiel, Calawah, and Sol Duc Rivers (Cram et al. 2018). Several other rivers and streams support the winter run life history only, including the Dickey River, Goodman Creek, Mosquito Creek, Ozette River, Tsoo-Yess (Sooes) River, and Wa’atch River (Cram et al. 2018).

1. Quillayute River System (Quillayute, Sol Duc, Bogachiel, Calawah, and Dickey Rivers)

The Quillayute River system includes the Quillayute, Sol Duc, Calawah, Bogachiel, and Dickey Rivers, which collectively drain 628 square miles of land (Klinger et al. 2008). Nearly one-third of the Quillayute River basin lies within Olympic National Park (32% of the Sol Duc River basin, 29% of the Bogachiel River basin, and 20% of the Calawah River basin occur inside the park (Houston and Contor 1984)). The Quillayute River proper, formed by the confluence of the Bogachiel and Sol Duc Rivers, is only 5.6 miles long (Smith 2000).

a. Sol Duc River

The Sol Duc River originates in the northern Olympic Mountains and generally flows west and northwest before leaving the boundary of Olympic National Park. After leaving the park, the river flows south and west through 11.1 miles of the Olympic National Forest, where more than half of the land is in late successional reserve (Smith 2000). Outside the Olympic National Forest, the Sol Duc River winds through private and state-owned land. The watershed receives 90-120 inches of precipitation per year and its upper reaches are influenced by the rain-on-snow zone.

Table 1. Summary of watersheds and steelhead populations, including Watershed Resource Inventory Area (WRIA), whether the watershed supports(ed) winter and/or summer run steelhead, period of record for monitoring data (usually redd counts) for winter runs, mean annual run size and range, mean annual escapement and range, trend in annual run size or abundance, mean harvest rate, percent of last 10-years the population achieved its escapement goal, average number of hatchery smolts released (winter and summer run), and the population risk score generated by WDFW's Steelhead at Risk report (Cram et al. 2018). N/A = not available or applicable. N/A is used to describe "run size" in larger tributaries to Quillayute River and the Clearwater River (tributary of Queets River) because the tribal fishery occurs almost solely below the tributary mouths and hence, the ultimate destination of the fish is unknown. Consequently, only escapement can be estimated for those tributaries.

Watershed (Major tributary)	WRIA	Winter (W) and/or summer run (S)	Monitoring period of record for winter runs	Mean (Range) annual run size winter runs	Mean (Range) annual escapement winter runs	Trend in abundance[2,3,4]	Mean harvest rate (range)	Percent last 10-years met escapement goal	Average hatchery smolts release 2009-2013 (Winter, Summer)	Population risk score as of 2013 (Cram et al. 2018)
Hoko River	19	W	1985-2020	N/A	566 (193 – 990)	40% decline[2]	N/A	80%	24,000 (W)	Low
Pysht River	19	W	1995-2020	N/A	499 (195 – 936)	21% decline[2]	14%[9]	70%	0	Mod.
Clallam River	19	W	1999-2020	N/A	158 (45 – 284)	27% decline[2]	0.7%[8]	No goal	2,000 (W)	Mod.
Deep Creek	19	W	2010-2020	N/A	95 (47 – 129)	Declining[4]	N/A	No goal	0	Insf.
West Twin River	19	W	2010-2020	N/A	60 (21 – 90)	Declining[4]	N/A	0%	0	Insf.

East Twin River	19	W	2010-2020	N/A	57 (31 – 116)	Declining[4]	N/A	10%	0	Insf.
Salt Creek	19	W	1995-2020	N/A	116 (32 – 237)	43% decline[2]	3.9%[9]	0%	0	Mod.
Lyre River	19	W & S[?]	N/A	N/A	N/A	N/A	N/A	No goal	5,000 (W), 2,000 (S)	Insf.
Sekiu River	19	W	N/A	N/A	N/A	N/A	N/A	No goal	12,000 (W)	Insf.
Sail River	19	W	N/A	N/A	N/A	N/A	N/A	No goal	10,000 (W)	Insf.
Tsoo-Yess River	20	W	N/A	N/A	N/A	N/A	N/A	No goal	96,000 (W)	Insf.
Ozette River	20	W	N/A	N/A	N/A	N/A	N/A	No goal	0	Insf.
Quillayute River	20	W & S	1978-2020[1]	13,064 (6,456 – 21,615)	9,340 (5,500 – 16,919)	No trend[2,3]	28% (10% - 55%)	90%	243,000 (W), 47,000 (S)	N/A
Dickey R. (trib)	20	W	1978-2020[1]	N/A	460 (143 – 1,607)	N/A	See Quillayute	100%	0	Low
Sol Duc R. (trib)	20	W & S	1978-2020[1]	N/A	3,864 (1,791 – 7,634)	N/A	See Quillayute	70%	*23,000 (W), 12,000 (S)*[NR]	Low
Calawah R. (trib)	20	W & S	1978-2020[1]	N/A	2,980 (989 – 5,985)	N/A	See Quillayute	100%	40,000 (W), 35,000 (S)	Moderate
Bogachiel R. (trib)	20	W & S	1978-2020[1]	N/A	1,975 (730 – 4,553)	N/A	See Quillayute	60%	180,000 (W)	Low
Goodman Creek	20	W	1995-2020	N/A	184 (45 – 374)	54% decline[2]	6.8%[7]	0%	*4,000*[NR]	High
Mosquito Creek	20	W	N/A	N/A	N/A	N/A	N/A	N/A	0	Insf.

Hoh River	20	W & S	1980-2020[1]	4,117 (2,541 – 5,783)	2,726 (1,616 – 4,593)	37% decline[3]	33% (7% - 55%)	60%	68,000 (W)	Mod.
Kalaloch Creek	21	W	N/A	N/A	N/A	N/A	N/A	No goal	0	Insf.
Queets River	21	W & S	1980-2018[1]	7,648 (4,200 – 13,309)	4,845 (2,271 – 7,841)	45% decline[3]	35% (10% - 55%)	30%	157,000 (W)	Mod.
Clearwater R. (trib)	21	W & S[?]	1980-2018[1]	N/A	1,744 (847 – 2,966)	N/A	See Queets	50%	0	Low
Quinault River	21	W & S	1978-2020[1]	5,883 (2,179 – 9,726)	3,107 (1,366 – 5,774)	44% decline[3]	46% (15% - 65%)	No goal	486,000 (W)	Low – Mod
Upper Quinault	21	W & S	1978-2020[1]	N/A	1,511 (772 – 2,877)	N/A	See Quinalt	1,200	0	Low
Moclips River	21	W	1988-2000	N/A	299 (130 – 560)	27% increase[2]	N/A	No goal	0	Low
Raft River	21	W	N/A	N/A	N/A	N/A	N/A	No goal	0	Insf.
Copalis River	21	W	N/A	N/A	N/A	N/A	N/A	No goal	0	Insf.

? = There are reports of small numbers of unclipped summer steelhead, 1) in the Lyre River, but it has never been determined whether those were wild fish from the Lyre, strays from another river, or naturalized offspring of hatchery summer steelhead scatter plants in the Lyre River; and 2) in the Clearwater River, which could be strays from the Queets River or naturalized offspring of hatchery summer steelhead scatter plants in the Clearwater River.

NR = Hatchery releases have ended on these rivers.

1 = Data range is presumed to be only for winter runs. Data on run size and escapement of summer runs is almost entirely lacking, but redd counts for winter steelhead may also include some summer steelhead since there is overlap in their spawning distribution.

2 = Source is Cram et al. (2018) Steelhead at Risk Report, with period of record ending in 2013.

3 = Source is McMillan et al. (2021).

4 = Source is North Olympic Peninsula Lead Entity (NOPLE) report (2015)

5 = Quillayute River population is not in statistically significant decline over its full period of record, but it has declined sharply from 1996-2017 at a rate of 5,533 adults/decade (McMillan et al. 2022).

6 = Harvest rate is included in whole population estimate.

7 = Period of record 1995-2009, per Cram et al. 2018

8 = Period of record 1999-2013, per Cram et al. 2018

9 = Period of record 1995-2013, per Cram et al. 2018

b. Bogachiel River

The Bogachiel River is formed by the North and South Forks of the Bogachiel River, which originate in the Olympic Mountains (Smith 2000). Its upper reaches are in Olympic National Park, while its middle and lower reaches flow through timber and other agricultural land (Smith 2000). The river's most significant tributary is the Calawah River, an important salmonid tributary itself (Smith 2000). The Bogachiel River's other important salmonid tributaries include Murphy, Maxfield, Weeden, Mill, Grader, and Dry Creeks (Smith 2000).

c. Calawah River

The Calawah River is formed by the North and South Forks of the Calawah River, which originate in the Olympic Mountains (Smith 2000). The South Fork and its largest tributary, the Sitkum River, provides spawning and rearing habitat for winter and summer steelhead and they flow through late-successional reserve land inside the Olympic National Forest (Smith 2000). The Nork Fork Calawah also supports winter and summer steelhead (Smith 2000).

d. Dickey River

The Dickey River is formed by the West, Middle, and East Forks of the Dickey River (Smith 2000). The Dickey River is a low gradient system, and it has numerous wetlands and sloughs (Smith 2000). Unlike the Sol Duc, Bogachiel, and Calawah Rivers, the Dickey River does not originate inside Olympic National Park (Smith 2000). Rather, the river and its tributaries occupy a heavily logged drainage (Smith 2000).

2. Hoh River

The Hoh River originates deep inside Olympic National Park, on the glacial slopes of Mt. Olympus, and flows westward for 56 miles before emptying in the Pacific Ocean (Duda et al. 2018). The Hoh River basin drains nearly 299 square miles of land and has an extensive floodplain (WCSSP 2013). The watershed receives 93-240 inches of rain per year (Klinger et al. 2008). The Hoh River has a strong glacial influence (McHenry et al. 1996).

Fifty-seven percent of the watershed is located within Olympic National Park, with the remaining 43% flowing through state (24.4%), private, and tribal lands (Klinger et al. 2008). The South Fork of the Hoh River, a major steelhead tributary, joins the mainstem of the Hoh River at RM 30 (Smith 2000). Other significant salmonid tributaries to the Hoh River include Slide, Falls, Mt Tom, Jackson, Taft, Snider, East Twin, Canyon, Spruce, Dismal, Pole, Tower, Lindner, Clear, Willoughby, Elk, Alder, Winfield, Hell Roaring, Lost, Pins, Anderson, Nolan, Braden, and Fossil Creeks (Smith 2000).

3. Ozette River

The Ozette River originates at the northern end of Ozette Lake and flows 5.3 miles west to the Pacific Ocean (NPCLE 2020). Coal Creek is the largest tributary to the Ozette River (NPCLE 2020). Multiple tributaries drain into Lake Ozette, including the Big River and Umbrella, Crooked, Siwash, South, Palmquist, Quinn, Elk, and Lost Net Creeks (NPCLE 2020).

4. Tsoo-Yess (Soose) and Wa'atch Rivers

The Tsoo-Yess and Wa'atch Rivers are short, rain-fed systems that originate in coastal foothills located in the northeastern corner of WRIA 20 (NPCLE 2020). Both rivers flow through the Makah Reservation and a small coastal strip of Olympic National Park (NPCLE 2020).

5. Goodman and Mosquito Creeks

Goodman and Mosquito Creeks are located to the north of the Hoh River and south of the Quillayute River system. Both creeks originate in coastal foothills and flow through state and private timberland lands and a coastal section of Olympic National Park before emptying into the Pacific Ocean (NPCLE 2020).

C. Water Resource Inventory Area 21

WRIA 21 begins at Kalaloch Creek in the north and ends at Connor Creek in the south (QINLE 2011). The area receives heavy rainfall, measuring 120" to 200" per year (HSRG 2004). Prior to European American settlement, the area was covered by old growth western red cedar (*Thuja plicata*), Sitka spruce, Douglas fir, and western hemlock (QINLE 2011). Today, the only undisturbed forestland in WRIA 21 is in Olympic National Park. WRIA 21 has an extensive coastal plain and 65 miles of marine shoreline (QINLE 2011).

Within WRIA 21, Olympic Peninsula steelhead occur in the Queets, Clearwater, Quinault, Raft, Copalis, and Moclips Rivers and Kalaloch Creek (HSRG 2004). The Queets and Quinault Rivers are large glacially-influenced systems that originate inside Olympic National Park

(McHenry et al. 1996, QINLE 2011, WCSSP 2013). The Clearwater, Raft, Copalis, and Moclips Rivers and Kalaloch Creek are rain-dominant systems (WCSSP 2013).

1. Queets River

The Queets River originates on Humes Glacier on the southeast side of Mt. Olympus. The river measures 51 miles long, drains 450 square miles of land, and flows at an average rate of 8,000 cubic feet per second (cfs) in the winter and 1,015 cfs in the summer (McMillan 2006). Nearly the entire course of the Queets River flows through Olympic National Park, with only the lower eight miles running through the Quinault Indian Reservation before reaching the Pacific Ocean (Smith and Caldwell 2001). Its major tributaries include the Clearwater, Sams, and Salmon Rivers and Matheny and Tshletshy Creeks (Smith and Caldwell 2001). Steelhead spawn in these tributaries, as well as several smaller ones (e.g., Miller Creek), and in the mainstem of the Queets River (Smith and Caldwell 2001).

Olympic National Park owns 34 miles of tributary streams, including all of Tsheltchy Creek and the lower five miles of the Sams River (Smith and Caldwell 2001). The USFS owns 84% of Matheny Creek watershed, 73% of the Sams River watershed, and 30% of the Salmon River watershed (Smith and Caldwell 2001). The USFS manages these lands as riparian reserves, late successional reserves, or adaptive management areas (Smith and Caldwell 2001). The Quinault Tribe owns the lower eight miles of the Queets River and 54% of the Salmon River drainage (McMillan 2006).

2. Clearwater River

The Clearwater River originates in the foothills of the Olympic Mountains (WCSSP 2013) and flows 39 miles before emptying into the Queets River several miles upstream of the Pacific Ocean. WDNR owns 79% of the Clearwater River watershed and roughly 20% is privately owned (McMillan 2006). Its major tributaries include the Sollecks and Snahapish Rivers and Christmas and Stequaleho Creeks (Smith and Caldwell 2001). The upper watershed receives 120" to 160" of rain per year (McHenry et. al 1996).

3. Quinault River

The headwaters of the Quinault River begin in the Mt. Lawson and the Enchanted Valley watersheds (Smith and Caldwell 2001). The river measures 69 miles long, drains 434 square miles of land, and flows at an average rate of 6,300 cfs in the winter and 1,080 cfs in the summer (McMillan 2006). The river feeds into Lake Quinault, which spans 3,729 acres. Below the lake, the river runs southwesterly through the Quinault Indian Reservation for 33 miles until it meets the Pacific Ocean (Smith and Caldwell 2001).

Nearly half (47%) of the Quinault River basin occurs within Olympic National Park (Houston and Contor 1984). The entire North Fork and most of the East Fork occur inside in the park. Roughly one-third (32%) of the basin occurs on the Quinault Indian Reservation, with the remaining 13% located on USFS land and 4% on private land (Smith and Caldwell 2001).

Winter and summer steelhead occur in the Quinault River (Smith and Caldwell 2001). Winter steelhead spawn in the mainstem below Lake Quinault, the North Fork of the Quinault River, and the following creeks: Cook, Elk, Willaby, Falls, Gatton, Zeigler, Kestner, Inner, Slough, Alder, Big, Fox, Fletcher, Boulder, Ten O'clock, Canoe, Irely, and Bunch Creeks (Smith and Caldwell 2001). Summer steelhead spawn in the East Fork and North Fork of the Quinault River (Brenkman et al. 2012).

4. Raft River

The Raft River is located between the Queets and Quinault Rivers (WCSSP 2013). The basin is 71,824 acres in size and includes the Raft River, North Fork Raft River, Red Creek and several independent tributaries (Smith and Caldwell 2001). The river originates in the foothills of the Olympic Mountains and flows through a coastal plain before emptying into the Pacific Ocean. The Raft River is 11.5 miles long and occurs almost entirely within the Quinault Indian Reservation (81%). The remaining 19% of the basin is private land (Smith and Caldwell 2001).

5. Moclips River

The Moclips River and its main tributary, the North Fork Moclips River, originate in the foothills to the south of the Quinault River. Together, the Moclips River and its north fork measure 17 miles long and flow west to the Pacific Ocean. The Moclips River basin is 53,528 acres in size, most of which is private land (54%) or owned by the Quinault Indian Nation (39%) (Smith and Caldwell 2001). The state owns the remaining 7% of the basin (Smith and Caldwell 2001). Winter steelhead spawn in the Moclips River mainstem, the North Fork of the Moclips River, and Wreck Creek (Smith and Caldwell 2001).

6. Copalis River

The Copalis River is a low gradient system located to the south of the Moclips River. The river originates in the foothills and flows through a coastal plain for a total of 24 miles before reaching the Pacific Ocean (WSCCP 2013). The Copalis River basin is 36,818 acres in size, nearly all of which is privately owned (95%), with the remaining 5% owned by the state (Smith and Caldwell 2001). Winter steelhead spawn in the river's mainstem (Smith and Caldwell 2001).

7. Kalaloch Creek

The Kalaloch Creek Basin spans 13,649 acres and includes Kalaloch Creek and four unnamed tributaries. Most of the land is owned by WDNR (41%) or private interests (40%). Olympic National Park owns 18% of the basin and the Quinault Indian Nation owns 1% (Smith and Caldwell 2001). Kalaloch Creek supports winter steelhead, which spawn in the lower mainstem and the West Fork of Kalaloch Creek (Smith and Caldwell 2001).

IV. LIFE HISTORY

Steelhead emerge from the gravel from spring through early summer and spend one to four years in freshwater before migrating to the ocean, where they spend one to four years before returning to freshwater to spawn (Busby et al. 1996). Steelhead smolts on the Olympic Peninsula typically migrate to the ocean from April through June, with peak emigration occurring in mid- to late-May (Busby et al. 1996).

Juvenile steelhead are present off the Washington coast in May and June, with only a few lingering into July and August (Pearcy et al. 1990). They migrate north and spend their first summer in the Gulf of Alaska and North Pacific Ocean (Atcheson et al. 2012). Afterwards they distribute throughout their north Pacific Ocean range (Atcheson et al. 2012). Some will go as far west as the Kuril Islands (Atcheson et al. 2012).

Steelhead distribution in the Pacific Ocean appears to be driven by temperature and salinity (Okazaki 1983, Sutherland 1973, Light et al. 1989). The south-north distribution of steelhead corresponds with a 3° C to 16° C temperature range, with nearly 96% of steelhead occurring in waters measuring at or below 12° C (Sutherland 1973, as cited in Light et al. 1989). The northern boundary, which is slightly above the Aleutian Islands, may correspond with factors other than sea surface temperatures, such as salinity (Light et al. 1989). Steelhead are generally found within 10 meters of the ocean's surface (Godfrey et al. 1975, as cited in Light et al. 1989).

Except for half-pound steelhead stocks, "[i]nformation from tagging studies shows little or no differences in ocean distribution among stocks, groups, or races." (Light et al. 1989). Therefore, Olympic Peninsula steelhead likely follow the same ocean distribution pattern as all other steelhead, not including half-pounders.

The Olympic Peninsula steelhead DPS includes seven summer-run and twenty-four winter-run populations (Cram et al. 2018). The summer-run populations return from May through October (McHenry et al. 1996) and spawn from January through April; however, the populations remain largely unstudied and, consequently, their status and trends are unknown (WDFW 1992; Busby et al. 1996; McMillan 2006; Brenkman et al. 2012; Cram et al. 2018). Historically, a substantial proportion of the winter-run populations returned between December and January (McMillan et al. 2022). Currently, the winter-run component returns and spawns between January and May, with a peak in March through April depending on the population (Busby et al. 1996; McMillan et al. 2007; McMillan et al. 2022).

Steelhead are iteroparous (Busby et al. 1996). Steelhead on their first spawning run are called "maiden" fish (Light et al. 1989). Maiden fish that survive spawning and return to the ocean are known as "kelts" (Behnke 2002). Steelhead may return to spawn up to five times within their lifetimes; however, most steelhead will only spawn once (Busby et al. 1996; Kendall et al. 2015).

V. HABITAT REQUIREMENTS

Steelhead are poikilotherms and evolved to survive within a range of stream and ocean temperatures. In general, steelhead prefer water temperatures between 10-15°C, but this varies by life stage and population (McCullough et al. 1999; Hicks 2000). For instance, Fuss (1998) suggests 5-11°C is optimal for steelhead egg survival and eggs and Rombough (1988) found alevins that experience water temperatures over 15°C are smaller than individuals that experience cooler water temperatures. This suggests optimal incubation temperatures are below 11-12°C (Hicks 2000). The most favorable temperature range for juvenile growth is between 5-17°C, with the optimal temperatures varying by season (Hicks 2000). Temperatures over 19°C can limit the occurrence and growth of juvenile steelhead if food is not sufficient, with lethal temperatures beginning at around 27°C (Hicks 2000). Temperatures of 11.3-12.3°C are consistently cited as the uppermost constant temperature exposures that will not interfere with smoltification (Zaugg et al. 1972; Zaugg and Wagner 1973; Zaugg 1981), with detrimental effects occurring once water temperatures begin to exceed 12.7°C. Adult migration in winter typically occurs when temperatures are relatively cool, but there is evidence that upstream migration by adults stops when temperatures reach 20-21°C, which is a concern for summer steelhead (Hicks 2000). Adult steelhead can begin to die when water temperatures reach 21.6-23.8°C and generally, steelhead avoid water temperatures of 24°C and above (unless they are genetically adapted to warmer water temperatures) (Hicks 2000).

Steelhead eggs and juveniles require clean gravel. If fine-grained sediment exceeds 12-20% of the surface, young steelhead are buried and suffocate (Phillips et al 1975; Reiser and White 1988; Jensen et al. 2009). Sedimentation in streams can also negatively impact growth and survival of juvenile salmonids in different ways (Suttle et al. 2004). For example, high levels of fine sediment can decrease prey, leading to starvation (Murphy et al. 1981).

Large woody debris (LWD) is also an important component of steelhead habitat (Cederholm et al. 1997; Thompson et al. 2012). LWD stabilizes stream channels, creates pools, reduces erosion, and provides cover, resting, and feeding areas for juveniles (Bilby and Bisson 1998). Complex floodplain habitat is also important to steelhead survival, as side channels and other natural features provide feeding habitat, refuge from high water events, and other benefits (Beechie et al. 1994; Montgomery et al. 1999; Bellmore et al. 2013).

Steelhead adapted to natural hydrographs, and juvenile and adult migrations are timed to coincide with historic flow regimes that maximize survival (Bjornn and Reiser 1991). Changes in flow regimes due to climate change and other anthropogenic causes (e.g., water withdrawals) interfere with these major life history stages, increasing stress and susceptibility to disease, and, occasionally, leading to direct mortality (e.g., stranding due to low streamflows) (Bjornn and Reiser 1991; Wade et al. 2013).

VI. DIET

A. Freshwater

Steelhead fry consume aquatic invertebrates, including chironomids, mayflies, and terrestrial macroinvertebrates (Davis 2015). One-year-old juveniles mostly consume insect larvae and pupae, adult insects, and crustaceans (Davis 2015). By age two, juvenile steelhead are larger in size and can consume larger and more diverse prey items, such as fish larvae and small fish (Davis 2015).

B. Marine

Steelhead smolts appear to spend a relatively short period of time feeding in nearshore waters before dispersing across the vast North Pacific Ocean. In the 1980s, researchers purse seined the coasts of Washington and Oregon to collect juvenile steelhead and study their stomach contents (Pearcy et al. 1990). The researchers intercepted juvenile steelhead in May through August; however, the peak catch occurred in May and June. Their stomach contents contained a variety of fish (juvenile rockfish, sandlance, brown Irish lord, and greenlings), euphasids, barnacle larvae, copepods, and other crustaceans (Pearcy et al. 1990). The researchers estimated that steelhead grow by about 1.1 mm per day between May and early July.

Steelhead in the open ocean consume a wide variety of prey. Sampling of steelhead diets in the Gulf of Alaska and Central North Pacific found that the dominant prey were fish and squid, with the primary species within those categories being Atka mackerel (*Pleurogrammus monopterygius)*, three-spined stickleback (*Gasterosteus aculeatus)*, lantern fish (*Myctophidea spp*.), and the minimal armhook squid (*Berryteuthis anonychus*) (Atcheson 2010). The study also found that steelhead consume euphausiids, copepods, amphipods, polychaetes, crustacean larvae, and, unfortunately, some styrofoam and plastic (Atcheson 2010). Plastic and styrofoam are of particular concern because it could result in mortality, toxicity, or delayed effects, such as heritable alterations in gene expression (Myers et al. 2013).

VII. NATURAL MORTALITY

A. Freshwater

Although a tremendous amount of research has been conducted on the survival of adult steelhead, far less information is available for specific mortality rates for juveniles in freshwater across a gradient of environments. Mortality at the egg stage can be very high, with survival from egg to emergence ranging from 18-95% depending on factors such as temperature, intra-gravel flow, and fine sediment (Shapovalov 1937; Sheppard 1972; Phillips et al. 1975; Bjornn 1978; Biley and Moring 1988). However, Shapovalov and Taft (1954) experimentally concluded that under ideal conditions, survival to emergence averages about 80-90%. Bjornn (1978) and

Phillips et al. (1975) also conducted experiments and found a similar upper limit as Shapovalov and Taft (1954) when fine sediment was absent from the redd, but survival was only 18% when 70% of the redd consisted of fine sediment.

After emergence, evidence suggests - as with most salmonids - that the mortality rate is high during the initial weeks to months of life when juvenile fish are small, weak, and fairly immobile. For example, Burns (1971) estimated June-to-October survival of young-of-the-year salmonids in a California stream was only 27% (range = 20-29%), while age-1 and older juveniles survived at a rate of 56% (range = 34-94%). Bjornn (1978) estimated mortality rate of 80-90% during the first summer of life, with an overall mortality of 94-99% upon completing the first year of life. A review of these papers and others suggests that mortality rates almost always exceed 50-60% early in life and thereafter, conditions being equal, survival tends to increase as fish size and condition increases (Biley and Moring 1988).

Mortality also occurs as smolts migrate to and enter the ocean. For example, Melnychuk et al. (2007) found that 65-73% mortality of juvenile steelhead occurred in the first month of smolt migration. Romer et al. (2012) reported that 63-89% of the steelhead smolts they tagged survived a short freshwater migration to the estuary, but that 50-60% of the fish died in the estuary despite most fish spending less than one day in the estuary.

B. Marine

Quinn (2005) estimated that approximately half of steelhead life cycle mortality occurs in the ocean. Typically, smolt to adult survival rates in the ocean is between 2% and 10% (Quinn 2005). Steelhead ocean survival is largely dependent on sea surface temperature during their first summer in the ocean (Atcheson 2010).

VIII. POPULATION STRUCTURE

Olympic Peninsula steelhead are distinct from other distinct population segments, including the Southwest Washington and Puget Sound DPSs (Busby et al. 1996). That distinction still holds true. WDFW acknowledges it is "not yet able to fully evaluate genetic population structure to aid the process of verifying the 1992 population definitions for Olympia [*sic*] Peninsula and SW Washington DPSs" (Cram et al. 2018).

The Olympic Peninsula Steelhead DPS includes summer-run and winter-run populations (Table 1; Busby et al. 1996). WDFW considers summer steelhead populations in the Sol Duc, Bogachiel, Calawah, Hoh, Queets, Clearwater, and Quinault Rivers to be distinct from each other based on geographic isolation (WDF et al. 1993). WDFW considers the following winter steelhead population to be distinct stocks: Salt Creek/Independents (Salt, Whiskey, Colville, and Field Creeks), Lyre River, Pysht/Independents (Pysht River, Deep Creek, East Twin River, and West Twin River), Clallam River, Hoko River, Sekiu River, Sail River, Sooes/Waatch (Sooes and Waatch Rivers), Ozette River, Sol Duc River, Quillayute/Bogachiel (Quillayute and Bogachiel Rivers), Calawah River, Dickey River, Goodman Creek, Mosquito Creek, Hoh River, Kalaloch

Creek, Queets River, Clearwater River, Quinault/Lake Quinault (Lower Quinault and broodstock net pet steelhead in Lake Quinault), Quinault River (above Lake Quinault), Raft River, Moclips River, and the Copalis River (WDF et al. 1993).

IX. PREVIOUS STOCK ASSESSMENTS

There have been several stock assessments for the largest wild winter steelhead populations in the Olympic Peninsula DPS (Table 2 and Table 3) but assessments for smaller populations with less consistent data are more limited. Below we list and summarize the assessments that have been conducted to date. No assessments are available for wild summer run steelhead because data on those populations is almost entirely lacking (except for data collected by the National Park Service (Brenkman et al. 2012) and John McMillan (Science Director, The Conservation Angler: McMillan 2022)).

Table 2. Reports that reviewed the status and trends of wild winter steelhead populations in the Quillayute, Hoh, Queets, and Quinault Rivers. Only the HSRG 2004 included an evaluation of summer steelhead.

Reference	Status
Nehlsen et al. 1991	Healthy, stable to increasing trend
WDF et al. 1993	Healthy
Busby et al. 1996	Stable to increasing; status review determined ESA-listing was not warranted
McHenry et al. 1996	Healthy, stable to increasing trend
WDFW 2002	Healthy, stable to increasing trend
HSRG 2004	At risk (summer runs listed as critical, except for Quinault population which is “at risk”)
Kendall et al. 2017	Declining trend in Hoh and Queets River populations
Cram et al. 2018	Declining trend in Hoh, Queets, and Quinault River populations
McMillan et al. 2022	Declining trend in Hoh, Queets, and Quinault River populations

A. American Fisheries Society (1991)

In 1991, the American Fisheries Society published a list of 214 native, naturally spawning salmonid stocks that were either at high or medium risk of extinction or were a species of concern (Nehlsen et al. 1991). This list did not include Olympic Peninsula steelhead (Nehlsen et al. 1991).

B. Washington Department of Fish and Wildlife (1992)

In 1992, the Washington Department of Fisheries ("WDF"), Washington Department of Wildlife, and the Western Washington Treaty Indian Tribes assessed the status of Olympic Peninsula winter and summer steelhead stocks (WDF et al. 1993). For WRIA 19 populations, WDF et al. (1993) identified the Hoko River and Pysht/Independent winter stocks as "healthy." WDF et al. (1993) defined "healthy" to mean the stock was "experiencing stable escapement, survival, and production trends and not displaying a pattern of chronically low abundance." The Salt Creek/Independents and Sekiu, Sail, Lyre, and Clallam River populations were listed as "unknown" (WDF et al. 1993).

For WRIA 20 populations, WDF et al. (1993) identified five winter steelhead stocks as "healthy" and five as "unknown." The five healthy stocks included winter steelhead in the Quillayute/Bogachiel, Dickey, Sol Duc, Calawah, and Hoh Rivers (WDF et al. 1993). WDF et al (2013) could not determine the status of winter steelhead populations in the Sooes, Wa'atch, or Ozette Rivers or Goodman, Mosquito, or Kalaloch Creeks (WDF et al. 1993).

For WRIA 21 populations, WDF et al. (1993) determined that four out of five winter steelhead stocks were healthy, including the Quinault, Moclips, Queets, and Clearwater River populations. The Departments and the tribes could not determine the status of the Copalis River population (WDF et al. 1993).

Except for the Queets River summer run population, which WDF et al. (1993) designated as "healthy," the departments and tribes could not determine the status of summer steelhead in the Sol Duc, Bogachiel, Calawah, Hoh, or Clearwater Rivers (WDF et al. 1993).

C. National Marine Fisheries Service (1996)

In 1996, NMFS reviewed the status of Olympic Peninsula steelhead (Busby et al. 1996). At the time, seven of the twelve winter steelhead stocks that had population trend data were declining or had relatively flat trends, while the other five were increasing (Busby et al. 1996). The declining and stable stocks included the following winter steelhead populations: Pysht River (-5.8%), Hoko River (-7.6%) Quillayute/Bogachiel River (-0.2%), Dickey River (-4.4%), Sol Duc River (-0.1%), Clearwater River (-0.5%); and Quinault River/Lake Quinault (-2.6%) (Busby et al. 1996). The increasing stocks included the following winter steelhead populations: Calawah River (1.1%), Hoh River (0.2%), Queets River (0.9%), Upper Quinault River (1.8%), and the Moclips River (13.6%) (Busby et al. 1996). No population trend data were available for summer steelhead (Busby et al. 1996).

At the time, most Olympic Peninsula steelhead populations appeared to be self-sustaining because there were no strong abundance trends (Busby et al. 1996). Additionally, WDFW provided information indicating that there was "substantial temporal separation between hatchery and native winter steelhead" (Busby et al. 1996). However, NMFS noted there were “isolated problems” with the population sustainability, including declining trends in the Pysht/Independents and Quinault River populations (Busby et al. 1996). NMFS also noted there was a “substantial contribution” of hatchery spawners in the Quinault River population (Busby et al. 1996). Based on those facts, NMFS determined that Olympic Peninsula steelhead did not warrant listing at the time (61 Fed Reg. 41541, 41550).

However, the biological review team expressed several concerns regarding Olympic Peninsula steelhead (61 Fed Reg., at 41550).

> “*** [T]he BRT has several concerns about the overall health of this ESU and about the status of certain stocks within it. The majority of recent abundance trends are upward (including three of the four largest stocks), although trends in several stocks are downward. These downward trends may be largely due to recent climate conditions. There is widespread production of hatchery steelhead within this ESU, largely derived from a few parent stocks, and this could increase genetic homogenization of the resource despite management efforts to minimize introgression of the hatchery gene pool into natural populations. Estimates of the proportion of hatchery fish on natural spawning grounds range from 16% to 44%, with the two stocks with the largest abundance of natural spawners (Queets and Quillayute) having the lowest hatchery proportions.”

(Busby et al. 1996).

D. Washington Department of Fish and Wildlife (2002)

In 2002, WDFW updated its status assessment of Olympic Peninsula steelhead populations (WDFW 2002). The 2002 assessment downgraded the Quinault River “mixed” origin winter steelhead stock from “healthy” to “depressed” (WDFW 2002). It changed the statuses of Queets River summer steelhead and Moclips River winter steelhead from “healthy” to “unknown” (WDFW 2002). Finally, it changed the Goodman Creek, Pysht/Independents, and Salt Creek/Independents winter steelhead population statuses from “unknown” to “healthy” (WDFW 2002).

E. North Olympic Peninsula Lead Entity (2004)

In 2004, a technical review team for the North Olympic Peninsula Lead Entity for Salmon (“NOPLE”) assessed winter steelhead populations in WRIA 19. The team determined that the following winter steelhead populations were “depressed”: Sekiu River, Clallam River, West Twin River, East Twin River, and the Western Strait Independents populations (NOPLE 2015). The team identified the Hoko River, Pysht River, and the aggregated Salt Creek populations as “healthy” and the Lyre River population as “unknown” (NOPLE 2015).

F. Hatchery Scientific Review Group (2004)

In 2004, the Hatchery Scientific Review Group (HSRG) issued a report on hatchery operations on the north coast of Washington (HSRG 2004). The report included population viability ratings from managers of Olympic Peninsula winter and summer steelhead populations (HSRG 2004). To assess population viability, they considered multiple factors such as age class structure, spawner escapement, and proportion of hatchery-origin fish in natural spawning populations (HSRG 2004). Each stock's viability was rated as "critical," "at risk," or "healthy" (HSRG 2004).

The report did not list any wild Olympic Peninsula steelhead populations as "healthy" (HSRG 2004). The report identified the following populations' viability as "critical": Copalis River winter steelhead, Goodman Creek winter steelhead, Hoh River summer steelhead, Moclips River winter steelhead, Mosquito Creek winter steelhead, Kalaloch Creek winter steelhead, Ozette River winter steelhead, Quillayute River system summer steelhead, and Sooes River winter steelhead (HSRG 2004). The report rated the following populations as "at risk": Hoh River winter steelhead, Hoko River winter steelhead, Queets River winter steelhead, Quillayute system winter steelhead, Quinault River summer steelhead, and Quinault River winter steelhead (HSRG 2004). HSRG could not assess the viability of Queets River summer steelhead (HSRG 2004).

G. Steelhead at Risk Report (2018)

In 2018, WDFW released the Steelhead at Risk Report, which assessed steelhead populations statewide based on the available, albeit limited, data at the time (Cram et al. 2018). Importantly, the last year of data the report covered was from 2013 and, therefore, it did not include data on the significant declines in abundance that occurred from 2014 through 2018. The authors noted that the "the lack of abundance, productivity, and diversity data was the most common impediment to conducting wild steelhead status assessments statewide" (Cram et al. 2018). The authors also explained they had limited productivity data. Cram et al. (2018) acknowledged "very little is known about temporal and spatial patterns of freshwater population productivity (smolts per spawner) and smolt to adult return rates (SAR) for Olympic Peninsula DPS wild steelhead populations, and this is a substantial data gap" (Cram et al. 2018). However, the authors observed that smolt-to-adult return ratios (SARs) for Olympic Peninsula steelhead have declined significantly over time (Cram et al. 2018). Additionally, the authors highlighted that between brood years 2005 and 2010, the population productivity of Olympic Peninsula steelhead was lower than the productivity of the lower Columbia River, middle Columbia River, Upper Columbia River, and Snake River steelhead DPSs (Cram et al. 2018).

When possible, Cram et al. (2018) generated a risk score based on short- and long-term trends in run size, extinction risk, and proportion of time the population met its escapement goal. Again, the report did not include escapement data for years 2014-2018 and, therefore, it did not consider significant failures to meet escapement goals during those years. They found that 11 (73%) of 15 winter steelhead populations displayed a decreasing trend in abundance

(Table 3), but the Lower Quinault River was the only population that met the high-risk standard for long-term abundance. They also determined Goodman Creek and the Calawah River were considered as high risk based on declines in short-term abundance, while Goodman Creek, Salt Creek, and Clallam River populations were identified as having a high risk of extinction (Table 3). Of the 13 populations with defined escapement goals, Cram et al. (2018) identified six populations at high risk for failure to meet escapement goals, but also noted that 54% of populations had met escapement goals in at least 70% of last 10 years (2004 – 2013).

Table 3 (from Cram et al. 2018). Risk assessment results and ratings for Olympic Peninsula Steelhead DPS populations. Red text indicates values that exceeded specific criterion for each metric. Status relative to abundance goal represents the percentage of years the population achieved its escapement goal. Abbreviations: win. = winter steelhead; sum. = summer steelhead; insuf. = insufficient.

Population	Long Term Abundance Trend	Short Term Decline	Extinction Risk	Status relative to abundance goal	SARR risk score	Population Risk Rating
Goodman Creek win.	-54%	Yes	91%	60%	3.0	High
Salt Creek/Independents win.	-43%	No	68%	20%	2.0	Moderate
Clallam win.	-27%	No	60%	40%	2.0	Moderate
Lower Quinault win.	-69%	No	0%	no goal	1.0	Moderate
Queets win.	-29%	No	0%	50%	1.0	Moderate
Pysht/Independents win.	-21%	No	1%	30%	1.0	Moderate
Hoh win.	-16%	No	0%	50%	1.0	Moderate
Calawah win.	50%	Yes	0%	100%	1.0	Moderate
Hoko win.	-40%	No	0%	80%	0.0	Low
Dickey win.	-22%	No	7%	100%	0.0	Low
Clearwater win.	-12%	No	0%	100%	0.0	Low
Sol Duc win.	-9%	No	0%	80%	0.0	Low
Quillayute/Bogachiel win.	-6%	No	0%	90%	0.0	Low
Upper Quinault win.	24%	No	0%	100%	0.0	Low
Moclips win.	27%	insuf. data	insuf. data	no goal	insuf. data	Low

In addition to lacking summer steelhead data, the data was insufficient to determine extinction risks for the following winter steelhead populations: Copalis River, Kalaloch Creek, Lyre River, Mosquito Creek, Ozette River, Raft River, Sekiu River, Sail River, Sooes River, and Wa'atch River populations (Cram et al. 2018).

According to the results for all four risk metrics, Cram et al. (2018) determined that one population was at a high total risk, seven at moderate total risk, seven at low total risk (Table 3) and 16 at undetermined risk.

X. ABUNDANCE AND POPULATION TRENDS

Based on Cram et al. (2018), McMillan et al. (2022), NOPLE (2015), run size and escapement estimates by WDFW and tribes (co-managers), and an online WDFW data repository (https://fortress.wa.gov/dfw/score/score/species/steelhead.jsp?species=Steelhead, Accessed online 3/20/2022), we compiled a list of all the major and small watersheds known to support wild steelhead by their respective Watershed Resource Inventory Area (WRIA) and other factors, such as mean annual abundance, in cases where such data was available (Table 1). The populations are predominantly or solely the winter run life history, and the most abundant populations are found in the Quillayute, Queets, Quinault, and Hoh Rivers, while the remaining rivers and creeks support much smaller populations (Table 1). Data is almost totally lacking for summer steelhead, and they are only thought to be present in the four largest watersheds (Quillayute, Hoh, Queets, and Quinault Rivers). However, snorkel survey data indicates that summer run populations are at critically low levels and close to extirpation (Table 1). Below we summarize information on the historic and current trends in abundance and escapement of wild steelhead where such information is available.

A. Historical Abundance Winter and Summer Steelhead

Information on the historic abundance of wild winter steelhead is only available for the four largest populations of wild winter steelhead in the Olympic Peninsula DPS, which include the Quillayute, Hoh, Queets, and Quinault River populations (McMillan et al. 2022). There is also some historical catch information for summer steelhead in each watershed, which we reference later. We could not find historical data for the other, smaller populations of wild winter steelhead. McMillan et al. (2022) analyzed old cannery, sport, and tribal catch records to estimate historic run size and run timing for the Quillayute, Hoh, Queets, and Quinault River populations. Information on historic releases of hatchery winter and summer steelhead are available from Duda et al. (2018), which provides a thorough summary of all releases of hatchery steelhead for watersheds draining from Olympic National Park.

Although there is no formal analysis of historical catch of summer steelhead in the four largest watersheds known to support them (Quillayute, Hoh, Queets, and Quinault Rivers), we include recreational and tribal catch data for a few years during the 1950's when data was occasionally available. We also include catch data from 1962-1977, which mostly consists of data from sport record catch cards collected annually by WDFW (Summer steelhead catch data is in excel file "Summer steelhead catch data for Quillayute_Hoh_Queets_Quinault River populations" and based on data from .pdf file "Historic winter and summer steelhead tribal catch by month Hoh, Quinault, Queets, Quillayute 40s-70s"). We ended the "historical" period in 1977 because that was the first year that hatchery summer steelhead were released into two tributaries of the Quillayute River system (Calawah and Sol Duc Rivers). It was not possible to determine the relative proportion of hatchery and wild fish in the catch after 1977 because hatchery summer steelhead were not outwardly marked with a fin clip until 1985. Regardless, this data was available to Busby et al. (1996), but it was not evaluated in their status review.

WDFW and the treaty tribes (collectively, the “co-managers”) considered steelhead to be “summer runs” if they were caught from May 1 through October 31, which is slightly problematic because that suggests they were also potentially catching spawned out winter run kelts in May and June that were emigrating back to the ocean (McMillan 2006). Kelts can be distinguished from unspawned summer steelhead upon capture, but it is not known whether such distinctions were made. Regardless, the overall catch of summer steelhead is quite small in most years and inconsistent across months, which generally suggests that large numbers of kelts were not reported in the catch.

We summarized cumulative summer steelhead catch by year and watershed and then generated a very rough estimate of run size by assuming a harvest rate of 25% and calculating lower and upper bounds at 15% and 35% harvest rates. We do not assume the estimates of summer run are robust owing to the simple calculation. Nonetheless, we provide them because they offer some insight into how catch could translate to abundance.

Below, we summarize what is known about the historical winter steelhead and summer steelhead catch and run sizes. Overall, the estimates suggest the largest watersheds historically contained abundant runs of winter and summer steelhead, with fresh steelhead likely entering the rivers each month of the year (McMillan 2006).

Table 4 (modified from McMillan et al. 2022). Comparison of historical mean annual wild winter steelhead abundance estimates based on cannery record data, expansion of historical commercial and recreational catch data, historical commercial fishery catch per unit effort (*CPUE*), and accessible stream kilometers (SKM) of habitat, an ensemble historical (circa 1948 - 1960) mean estimate, the contemporary (circa 1980 - 2017) mean estimate, and the percent decline of each population in relative to the ensemble historical estimate. Percent decline is the difference between the ensemble historical abundance estimate and the contemporary mean abundance. The percent decline reported for the most recent five-year period in McMillan et al. (2021) refers to 2013-2017 (except for Quinault which referred to 2009-2013). Since the estimate in McMillan et al. (2021) does not capture the most recent run sizes, we updated the estimate so that the comparison between the ensemble estimate and the most recent five-year period for each population refers to 2016-2020.

Population	Cannery records	Historical catch	Historical *CPUE*	Accessible habitat (SKM)	Ensemble historical abundance estimate[1]	Contemporary mean abundance	Percent decline	Most recent 5-yr mean abundance (~ 2016-2020)	Percent decline for most recent 5-yr
Quillayute		22,567	23,391	19,571	21,843	13,595	38 %	8,528	61 %
Hoh		15,923	14,160	10,431	13,505	4,206	69 %	2,880	79 %
Queets	32,659	19,875	13,553	12,144	15,191	7,648	50 %	4,658	69 %
Quinault		13,743	22,226	14,723	16,897	6,181	63 %	3,370	80 %

[1] Ensemble historical estimate does not include cannery record estimate for Queets in 1923

1. WRIA 20 Populations

a. Hoh River

Using historical catch data, McMillan et al. (2022) estimated the average historical run size from 1948 through 1960 to be 15,923 winter steelhead (lower and upper bounds for historic mean: 9,023, 24,901) (Table 4). With that approach, the lowest and highest single-year estimate for the Hoh River population was 7,118 and 24,684 winter steelhead in 1948 and 1956, respectively. Using historical catch per unit effort (CPUE) and total accessible habitat produced average historical run size estimates of 14,160 and 10,431 steelhead, respectively (Table 4). The average ensemble estimate for annual run size for all three approaches was 13,505 steelhead (Table 4).

Estimates of historic run size prior to the modern monitoring period are not available for summer run steelhead. Because they were not a target species less data is available on historic catch. We could only find two years of harvest data prior to 1962 (i.e., tribal catch in 1954 and 1957: data is in excel file "Summer steelhead catch data for Quillayute_Hoh_Queets_Quinault River populations" and based on data from .pdf file "Historic winter and summer steelhead tribal catch by month Hoh, Quinault, Queets, Quillayute 40s-70s"), after which sport catch record card data is available in addition to a few years of tribal catch up to 1977. We consider this timeframe to be the "historical" period because it is prior to the onset of releases of hatchery summer steelhead that began in 1977. Mean catch of summer steelhead for the period beginning in 1954 and ending in 1977 was 118 fish and ranged from a low of 38 fish to a high of 274 fish (Table 5).

Assuming a 25% harvest rate and using bounds of 15% and 35% harvest rates, we estimated a mean run size of 472 summer steelhead (Lower bound = 337, Upper bound = 1,179) with a low run size of 152 fish and a high of 1,096 (Table 6). The mean of the top-five run sizes is 884 summer steelhead (Lower bound = 631, Upper bound = 1,473).

Although there is uncertainty associated with our estimations, wild summer steelhead were formerly abundant in the Hoh River. Further, it is likely that the run sizes were larger if the harvest rates are lower than we assume and not all the catch was reported, both of which are plausible. Last, the earliest data point we found was from 1954, which is decades after the onset of commercial and recreational fisheries and industrial logging practices. Therefore, it is almost certain that wild summer steelhead were more abundant than we report here.

b. Quillayute System

Using historical catch, McMillan et al. (2022) estimated that the average historical abundance for winter steelhead in the Quillayute River from 1948-1960 was 22,567 steelhead (lower and upper bounds for the historical mean: 16,733, 31,591) (Table 4). Annual variability within the historical period ranged from the lowest and highest single year estimates in the Quillayute River of 6,702 and 34,757 winter steelhead in 1948 and 1951, respectively. Using historical CPUE and total accessible habitat produced average historical run size estimates of 23,391 and 19,571 steelhead, respectively (Table 4). The average ensemble estimate for annual run size for all three approaches was 21,843 steelhead.

Estimates of historic run size are not available for summer run steelhead. Because they were not a target species less data is available on historic catch. However, there is evidence that summer steelhead were present in each of the three largest tributaries, including the Bogachiel, Sol Duc, and Calawah Rivers (McMillan 2006). We summarized all the catch data for the tributaries as part of the Quillayute River system (Data is in excel file “Summer steelhead catch data for Quillayute_Hoh_Queets_Quinault River populations” and based on data from .pdf file “Historic winter and summer steelhead tribal catch by month Hoh, Quinault, Queets, Quillayute 40s-70s”). Mean catch of summer steelhead in 1946 and from 1962-1977 was only 67 fish with annual catch ranging from 12 to 309 fish (Table 5), suggesting not many fishers focused on catching summer steelhead in the Quillayute system.

Assuming a 25% harvest rate and using bounds of 15% and 35% harvest rates, we estimated a mean run size of 268 summer steelhead (Lower bound = 191, Upper bound = 670) with a low run size of 48 fish and a high of 848 (Table 6). The mean of the top-five run sizes is 478 summer steelhead (Lower bound = 342, Upper bound = 797). If the population was harvested with the range of rates we proposed, then wild summer steelhead were once quite abundant in the Quillayute River system.

2. WRIA 21 Populations

a. Quinault River

Using historical catch, McMillan et al. (2022) estimated the average historical abundance for winter steelhead in the Quinault River from 1948-1960 was 13,743 steelhead (lower and upper bounds for the historical mean: 9,345, 20,258) (Table 4). Annual variability within the historical period ranged from the lowest and highest single year estimates in the Quinault River of 7,475 and 30,332 winter steelhead in 1956 and 1952, respectively. Using historical CPUE and total accessible habitat produced average historical run size estimates of 22,226 and 14,723 steelhead, respectively (Table 4). The average ensemble estimate for annual run size for all three approaches was 16,897 steelhead.

We summarized all the catch data on wild summer steelhead for the Quinault River system for 1954-1955, 1957, and 1962-1977 (Table 5, Raw data is in excel file “Summer steelhead catch data for Quillayute_Hoh_Queets_Quinault River populations” and based on data from .pdf file “Historic winter and summer steelhead tribal catch by month Hoh, Quinault, Queets, Quillayute 40s-70s”). Mean catch of summer steelhead in 1954-55, 1957, and from 1962-1977 was 162 fish with annual catch ranging from 12 to 447 fish, and peak catch being reported in 1954 (Table 5).

Assuming a 25% harvest rate and using bounds of 15% and 35% harvest rates, we estimated a mean run size of 649 summer steelhead (Lower bound = 464, Upper bound = 1,624) with a low run size of 48 fish and a high of 1,788 (Table 6). The mean of the top-five run sizes is 1,130 summer steelhead (Lower bound = 807, Upper bound = 1,884). This suggests summer steelhead were very abundant in the Quinault River dating back to 1954. Further, as with other populations, catch records were very low in several years, suggesting some of our estimates are likely much lower than the true total run size.

Table 5. Catch of wild sumer steelhead in Quillayute, Hoh, Queets, and Quinault Rivers prior to onset of hatchery summer steelhead releases in Quillayute system in 1977, with 1946-1957 representing tribal gillnet catch and 1962-1977 representing almost solely recreational catch. Tribal catch data was only reported and included here for 1972 and 1975-1978 in Quillayute River system, 1975-1976 in Hoh River, 1974-1977 in Queets River, and 1974-1976 in Quinault River. Bolded values represent peak catch for the period of record for each population. Data is in excel file "Summer steelhead catch data for Quillayute_Hoh_Queets_Quinault River populations" and based on data from .pdf file "Historic winter and summer steelhead tribal catch by month Hoh, Quinault, queets, quillayute 40s-70s"),

Population	1946	1954	1955	1957	1962	1963	1964	1965	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975	1976	1977
Quillayute River	15	N/A	N/A	N/A	12	36	39	30	37	78	105	81	84	116	**309**	66	175	143	136	51
Hoh River	N/A	133	N/A	39	59	106	126	112	160	137	106	105	125	142	**274**	169	188	66	92	72
Queets River	N/A	**373**	29	136	111	143	230	220	266	225	299	239	217	337	345	267	315	385	171	124
Quinault River	N/A	**447**	15	49	165	157	208	119	206	151	238	279	221	180	241	289	236	140	357	12

Table 6. Mean estimated run size of wild summer steelhead based on recreational and tribal catch data in Table 7, where we estimated run size with an assumed 25% harvest rate and we estimated upper and lower bounds using 15% and 35% harvest rates, respectively, for the top five run sizes for each population. Data from 1946-1957 is from tribal gillnet catch records. Data from 1962-1977 almost solely consists of recreational catch, with tribal catch only being reported for 1972 and 1975-1978 in Quillayute River system, 1975-1976 in Hoh River, 1974-1977 in Queets River, and 1974-1976 in Quinault River. Bolded values represent peak catch for the period of record for each population. Calculations for each year's run size, if not clear to the reader in this petition, can be accessed in the excel file "Summer steelhead catch data for Quillayute_Hoh_Queets_Quinault River populations"

Population	1946	1954	1955	1957	1962	1963	1964	1965	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975	1976	1977
Quillayute River	60	N/A	N/A	N/A	48	144	156	120	148	312	420 (300-700)	324 (231-540)	336 (240-560)	464 (331-773)	**848** (606–1,413)	264	308	104	296	204
Hoh River	N/A	532	N/A	156	236	424	504	448	640 (457-1,067)	548 (391-913)	424	420	500	568 (406-947)	**1,096** (783-1,827)	676 (483-1,127)	520	152	360	288
Queets River	N/A	**1,492** (1,066-2,487)	116	544	444	572	920	880	1,064 (760-1,773)	900	1,196 (854-1,993)	956	868	1,348 (963-2,247)	1,380 (986-2,300)	788	676	212	380	496
Quinault River	N/A	**1,788** (1,277-2,980)	60	196	660	628	832	476	824	604	952 (680-1,587)	1,116 (797-1,860)	884 (631-1,473)	720	964 (689-1,607)	484	596	296	212	48

b. Queets River

Unlike the other populations, there are two historic data periods for Queets winter steelhead: one from a cannery record in 1923 and another from tribal catch records from 1948-1960. In 1923, 1,500 cases of wild winter steelhead were packed at the Queets River cannery, with each case containing 48 cans at 1 lb per can, or 48 lb per case (McMillan et al. 2022). This is equivalent to a total weight of canned winter steelhead of 72,000 lb. Based on a wastage rate of 0.40 and a mean weight of 9.8 lb per Queets River winter steelhead, the number of wild winter steelhead processed in 1923 is equivalent to 12,245 fish. Assuming a cannery exploitation rate equivalent to the median contemporary exploitation rate (0.38), the total 1923 Queets return of wild winter steelhead is estimated to be 32,223 fish**,** with upper and lower bounds of 43,732 and 27,829 fish, based on 75% and 25% quartiles of contemporary exploitation rates (0.44, 0.28), respectively (McMillan et al. 2022)**.**

Using historical catch, McMillan et al. (2022) estimated the average historical abundance for winter steelhead in the Queets River from 1948-1960 was 19,875 steelhead (lower and upper bounds for the historical mean: 13,025, 32,878) (Table 4). Annual variability within the historical period ranged from the lowest and highest single year estimates in the Queets River of 6,191 and 52,200 winter steelhead in 1960 and 1954, respectively. Using historical CPUE and total accessible habitat produced average historical run size estimates of 13,553 and 12,144 steelhead, respectively (Table 4). During the peak estimate year in the Queets River, which was the largest estimated run size amongst all years and populations, over 14,000 winter steelhead were harvested, including over 5,200 winter steelhead harvested in the month of December alone. The average ensemble estimate for annual run size for all three approaches was 15,191 steelhead.

We summarized catch data on wild summer steelhead for the Queets River system for 1954-1955, 1957, and 1962-1977 (Table 5: Raw data is in excel file "Summer steelhead catch data for Quillayute_Hoh_Queets_Quinault River populations" and based on data from .pdf file "Historic winter and summer steelhead tribal catch by month Hoh, Quinault, Queets, Quillayute 40s-70s"). Mean catch of summer steelhead in 1954-55, 1957, and from 1962-1977 was 200 fish with annual catch ranging from 29 to 373 fish, with peak catch being reported in 1954 (Table 5).

Assuming a 25% harvest rate and using bounds of 15% and 35% harvest rates, we estimated a mean run size of 802 summer steelhead (Lower bound = 573, Upper bound = 2,004) with a low run size of 116 fish and a high of 1,492 (Table 6). The mean of the top-five run sizes is 1,296 summer steelhead (Lower bound = 926, Upper bound = 2,160). Based on the catch data wild summer steelhead were formerly abundant in the Queets River.

3. WRIA 19 Populations

As mentioned previously, we could not find quantifiable information on historic abundance for populations in WRIA 19. However, there is likely some historic catch information from WDFW punch-cards, and anecdotal references suggest that "the largest steelhead trout populations were found in the Lyre, Pysht, and Hoko rivers. The Clallam and Sekiu rivers, as well as Deep Creek also supported significant steelhead populations" (NOPLE 2015). Additionally, each stream was fished for winter steelhead by Native American tribes that lived in the area, including the Lower Elwha Klallam Tribe and the Makah Tribe.

B. Current Abundance and Population Trends (circa 1980-2020)

Data on run size and escapement is not available for all populations in the Olympic Peninsula DPS (Table 1). The Steelhead at Risk Report for Washington State by Cram et al. (2018) identified 31 populations of wild steelhead (24 winter run populations and 7 summer run populations) in the DPS, which included delineating larger sub-basins within each major watershed (e.g., counted Sol Duc, Bogachiel, Calawah, and Dickey River (tributaries in Quillayute River watershed) as individual populations). We did not delineate the Upper and Lower Quinault River into sub-populations per Cram et al. (2018). We otherwise delineated populations the same as Cram et al. (2018). We identified 22 distinct watersheds, with the Quillayute River system consisting of four sub-basins and the Queets River containing the Clearwater River, resulting in 26 populations of winter steelhead (Table 1). The Hoh, Calawah, Queets, and Quinault Rivers are known to support wild populations of summer steelhead, though data is very limited because they are not monitored. There is also evidence of summer steelhead in the Sol Duc, Bogachiel, and Clearwater Rivers (Table 1). This brings the total number of populations to 30 for the entire DPS.

Cram et al. (2018) evaluated current abundance and population trends for winter steelhead where data was available. Cram et al. (2018) reported that sufficient escapement abundance data were available for 48% (15 of 31) of populations (Figure 2). We use Cram et al. (2018), a report by NOPLE (2015), and a publication by McMillan et al. (2022) to summarize existing information on the run size, escapement, and trends in run size (or escapement if only escapement data is available). We also rely on McMillan et al. (2022) to compare contemporary abundance (circa 1980 – 2018) to historical abundance (circa 1948-1960: Table 4) to illustrate that contemporary populations, most of which are in decline (Cram et al. 2018, McMillan et al. 2022), had already experienced a high level of depletion before the onset of the modern monitoring programs that began in the late-1970's through the early-1980s (Table 4).

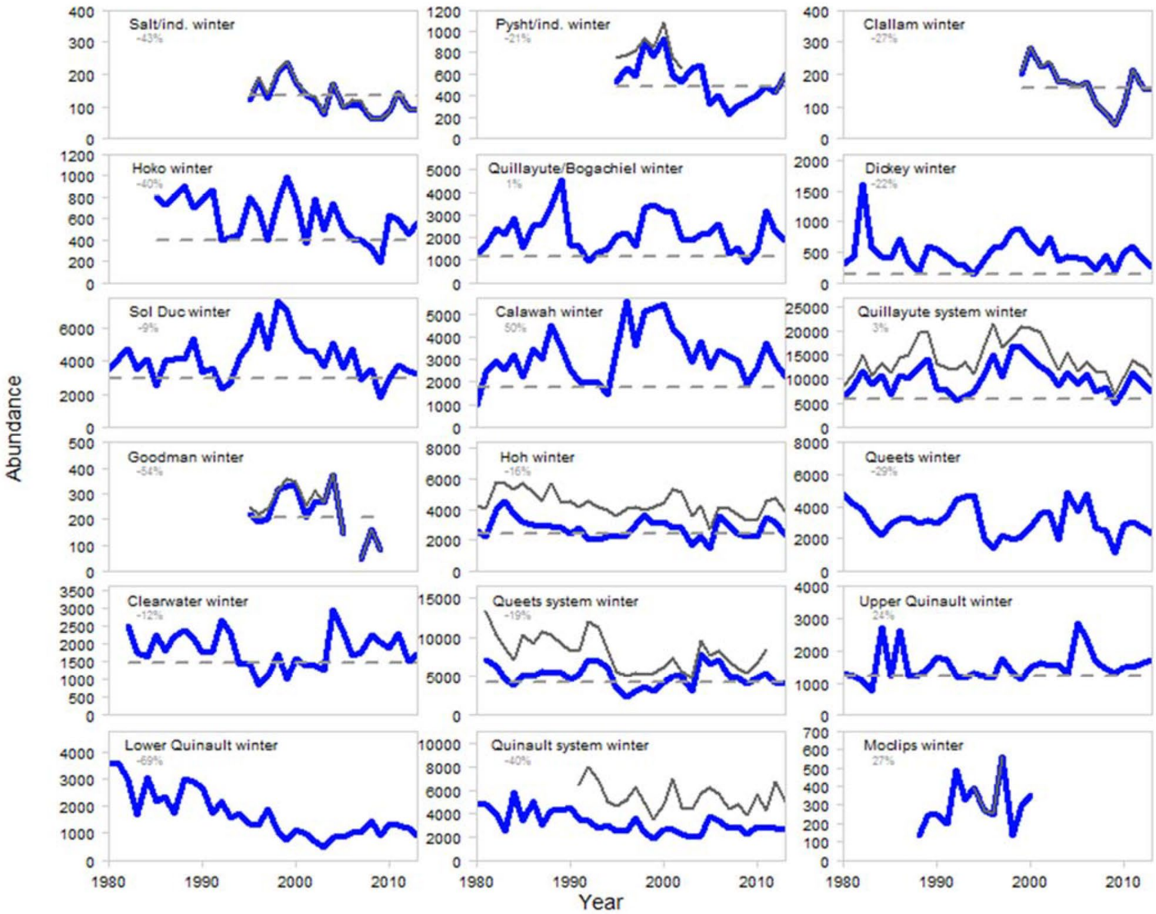


Figure 2 (from Cram et al. 2018). Abundance data for the Olympic Peninsula Steelhead DPS populations available from years 1980 to 2013. The thick blue line represents the total escapement (for the population or a portion of it, based on data availability). The thin grey line represents total run size (escapement plus reported sport and tribal harvest plus, for Hoh River population only, estimated non-tribal and tribal bycatch mortality). The dashed grey line is the escapement goal. The light grey number is the percent change in abundance (increase or decrease) over the time period with data.

In terms of numbers of populations, the Olympic Peninsula DPS is predominantly composed of small populations of wild steelhead in small coastal creeks and streams (Table 1). Data sets for most of those populations are less consistent compared to the larger, major populations, which contribute the greatest proportion of fish to the cumulative total number of wild steelhead in the DPS. We provide a synthesis of the location of each watershed (by WRIA), the type of steelhead life history that is present, the period of record of monitoring, mean and range for run size and escapement, and the status of the population trend – as either declining or increasing, and the rate of decline (based on Cram et al. 2018, NOPLE 2015, and McMillan et al. 2022). As we outline below, the data indicates that most populations of wild winter steelhead are in decline, and comparatively, Cram et al. (2018) estimated the Olympic

Peninsula DPS had the second lowest proportion of populations with increasing trends (20%) among all the steelhead population segments in Washington State (Figure 3).

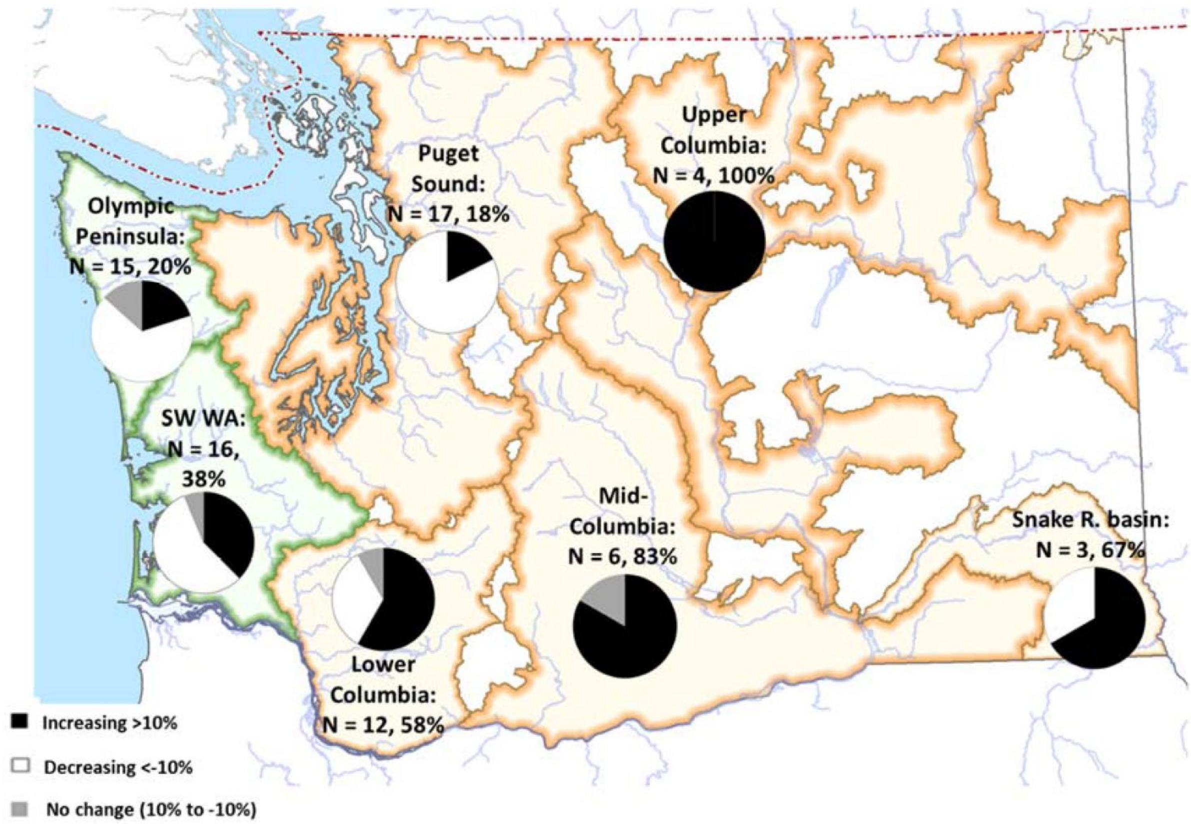


Figure 3 (From Cram et al. 2018). Washington steelhead population escapement abundance trends in each DPS in time periods from 1980-2013. N represents in the number of populations in each DPS that had suitable abundance data for trend analysis and the percentage indicates the proportion of populations that have increased in abundance since 1980.

Run size declines in wild winter steelhead have generally continued since the report by Cram et al. (2018), with most populations reaching their lowest level of abundance in the past several years (Figures 4-8). To account for the most recent data, we updated the most recent five-year estimates by McMillan et al. (2022) to 2016-2020 instead of 2013-2017. Using that adjustment, we estimate the Quillayute, Queets, Hoh, and Quinault River wild winter steelhead populations have declined by 61%, 69%, 79%, and 81%, respectively, when comparing the most recent five-year average run size to the historic estimates from the 1950s (Table 4). The estimated decline for the Queets winter steelhead population increases to 86% when compared to the 1923 cannery estimate in McMillan et al. (2022). Given the older cannery record from the Queets River, declines are likely even greater than estimated in McMillan et al. (2022) because their estimates only dated back to the 1950's, which is after the initial boom in logging and the operation of commercial salmon canneries at the mouths of the Quillayute, Hoh, Queets, and Quinault Rivers in the early-1900's.

The long-term, chronic declining trends coupled with a recent sharp downturn not documented in Cram et al. (2018) puts the Olympic Peninsula Steelhead DPS at greater risk of extinction than when Busby et al. (1996) conducted the last federal status review for the DPS (ESU at the time). Further, as we outline below, the populations of wild summer steelhead are almost all at critically low levels of abundance, something that was also not reported by Busby et al. (1996), and could be facing extirpation in the near term if some are not already functionally extinct.

1. Winter Steelhead

The four largest winter steelhead populations on the Olympic Peninsula have been on a forty-year-long decline.

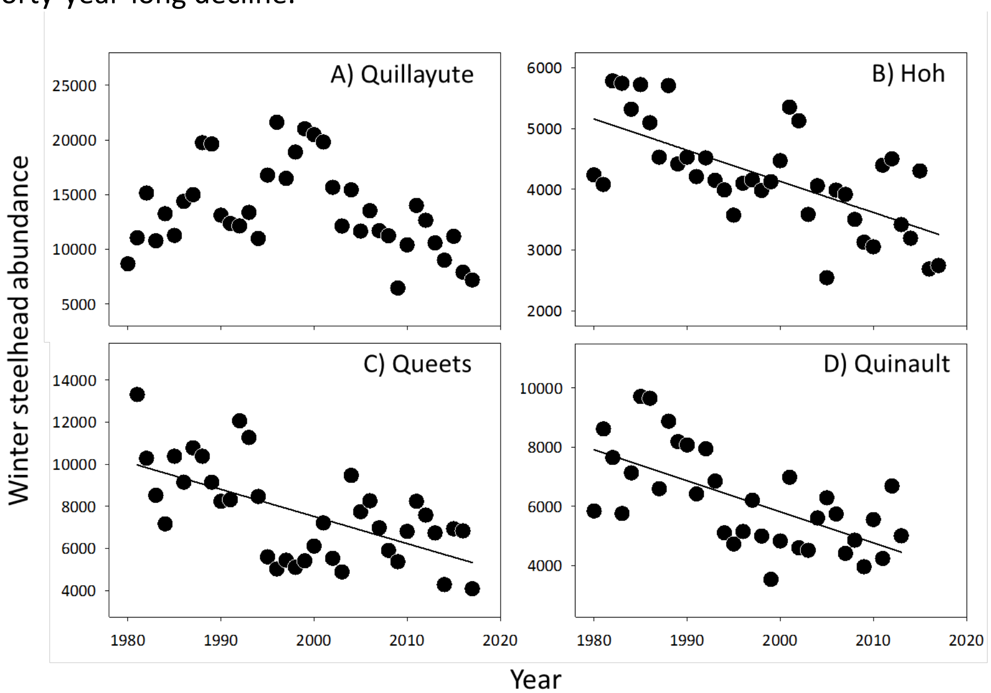


Figure 4 (From McMillan et al. 2022). Contemporary trends (circa 1980 – 2017) in wild winter Steelhead abundance in the a) Quillayute, b) Hoh, c) Queets, and d) Quinault Rivers. Black lines represent the best fit linear regression models.

a. WRIA 20 Populations

i. Hoh River

The Hoh River has one of the largest populations of wild winter steelhead in the Olympic Peninsula DPS (Table 1). From 1980-2020 the mean annual winter steelhead run size of Hoh River winter steelhead was 4,117 fish with annual run sizes ranging from 2,541 – 5,783 fish (Table 1). The population was in a significant declining trend from 1980-2013 (Figure 2) and from 1980-2017 (Figure 4), and it has experienced most of its lowest returns on record in the past decade (Figure 5). Cram et al. (2018) estimated a 16% decline from 1978-2013, while McMillan et al. (2022) estimated a decline of 37% from 1980-2017 (Table 1), which is equivalent to a loss of 513 (95% CI: 320, 710) adults per decade (linear regression on log-transformed abundance, $F_{1, 36} = 29.01$, $P < 0.001$: McMillan et al. 2022). The decline has not reversed since the analysis by McMillan et al. (2022) (Figure 5).

The Hoh River population of winter steelhead has failed to meet its escapement goal in 5 of the past 11 years (2010-2020), and it just barely made its escapement goal in the past two years (Figure 5).

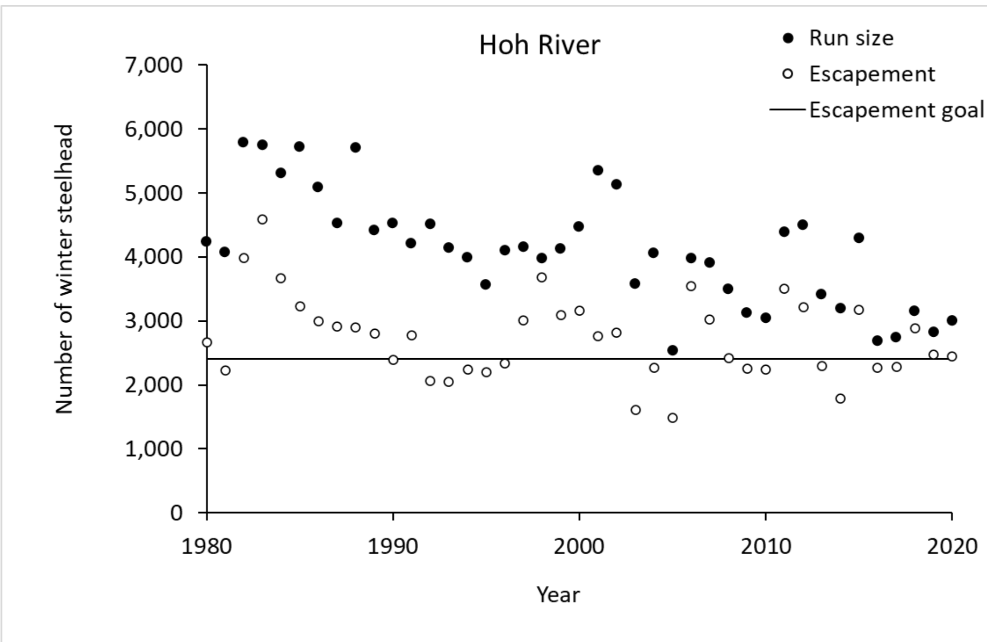


Figure 5. Annual run size and escapement of wild winter steelhead in the Hoh River from 1980-2021 in relation to the escapement goal of 2,400 fish. Years when escapement falls below the escapement goal indicate that the fishery did not meet its spawner target goal.

Compared to the historic period (circa 1948-1960), mean annual abundance in the contemporary period (1980-2017) has declined 69% from its mean annual abundance during the historical period (circa 1948-1960) (Table 4). The decline jumps to 79% when comparing the mean historical abundance to the most recent five-year average run size (Table 4).

We believe this should be considered a minimum level of depletion because the approach assumes all caught fish were sold and reported in the catch. This is unlikely, as Wilcox (1898) indicated for the Quinault River in 1895: "…quite a large number of salmon are taken by Indians for their winter supply of food, and a small amount … was sold to buyers…" It could therefore be anticipated that a considerable number of steelhead caught on the Queets River in

1923 were used by Native Americans for their own winter food supply rather than sold to the cannery (McMillan 2006). In addition, the historical estimates in McMillan et al. (2022) were based on fish populations that had already experienced decades of habitat changes and the onset of canneries and commercial fisheries.

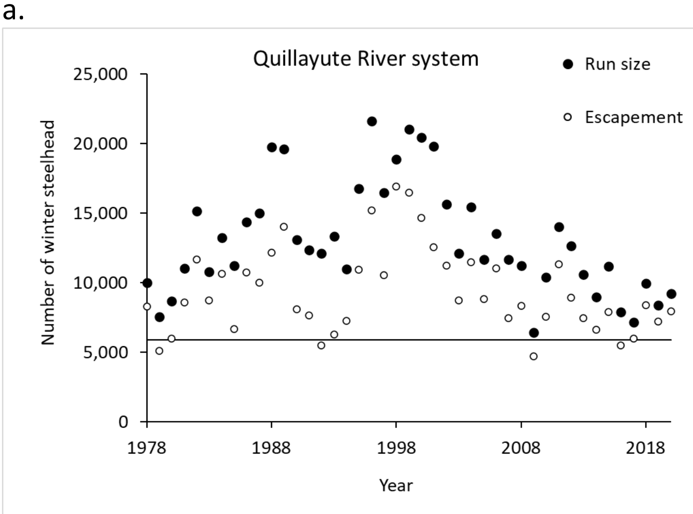
If this downward trajectory continues, run sizes will begin to consistently come in below the escapement goals, eliminating the potential for fisheries and potentially leading to a level where they can no longer effectively compensate (e.g., Ward et al. 2000; Atlas et al. 2015).

ii. Quillayute River System

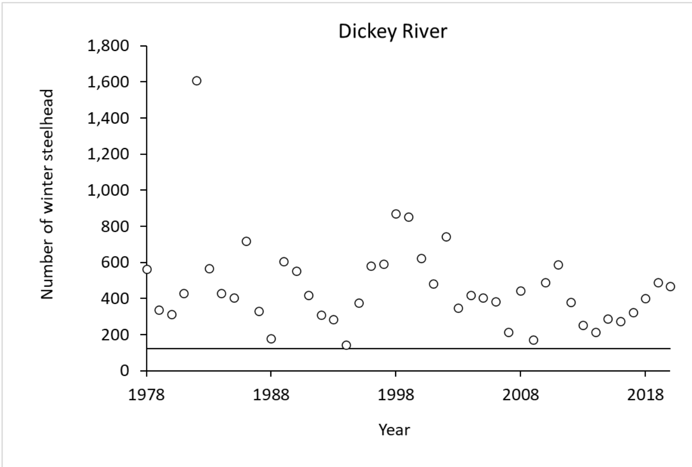
The Quillayute River supports the largest population of wild winter steelhead in the Olympic Peninsula DPS. Its mean annual run size from 1978-2020 was 13,064 steelhead (range = 6,456 – 21,615) (Table 1). The Quillayute River system is largely comprised of three large tributaries, including the Sol Duc, Calawah, and Bogachiel Rivers, and one smaller tributary, the Dickey River (Table 1). We summarize all of these populations as part of the "Quillayute River" system for purposes of simplicity and because each of these populations is showing relatively similar patterns in annual abundance (Figure 2).

Unlike the Hoh, Queets, and Quinault Rivers, there was not a simple linear trend for wild winter steelhead run size in the Quillayute River (Figure 4). This is because winter steelhead returns increased from 8,761 fish in 1980 to a peak of 21,615 fish in 1996. Since the peak in 1996, annual returns of winter steelhead have declined at a rate of 5,533 fish per decade (Figure 4). Run sizes for more recent years, which were not included in McMillan et al. (2022), suggest the recent declining trend has continued (Figure 6a). As a result, winter steelhead run sizes in recent years have been the lowest on record. Although the Quillayute River population has fared slightly better than the others, the co-managers implemented restrictive regulations in the 2020 and 2021 seasons due to poor returns, raising concerns about the long-term sustainability of the population and the fisheries it supports.

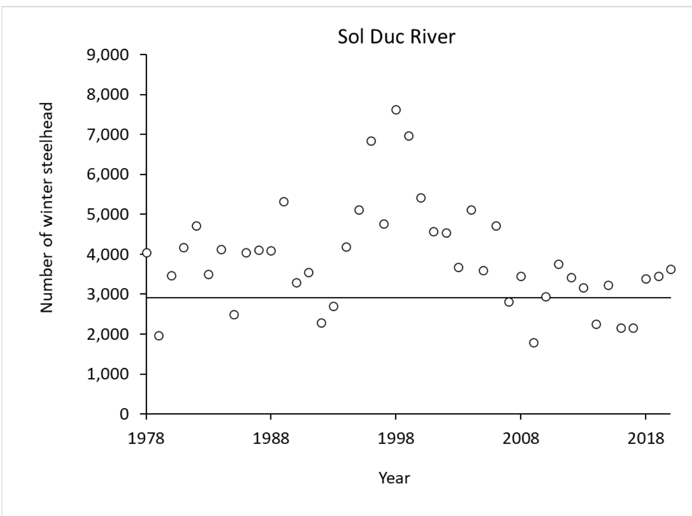
a.



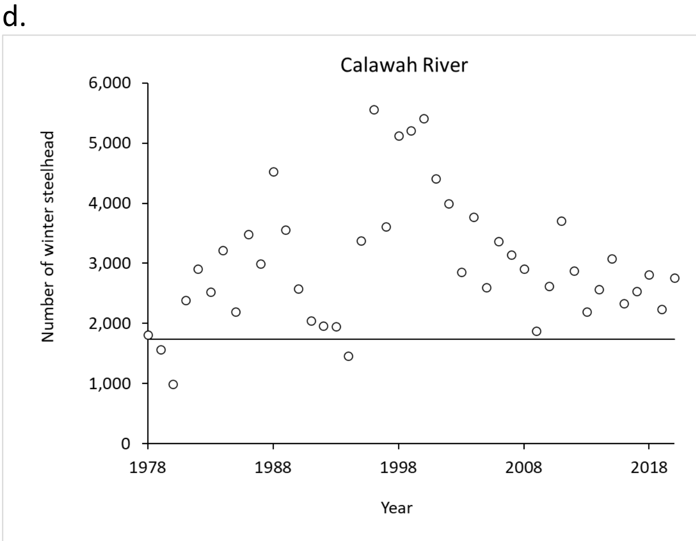
b.


Dickey River
Number of winter steelhead
1,800
1,600
1,400
1,200
1,000
800
600
400
200
0
1978
1988
1998
2008
2018
Year

c.


Sol Duc River
Number of winter steelhead
9,000
8,000
7,000
6,000
5,000
4,000
3,000
2,000
1,000
0
1978
1988
1998
2008
2018
Year

d.


Calawah River
Number of winter steelhead
6,000
5,000
4,000
3,000
2,000
1,000
0
1978
1988
1998
2008
2018
Year

e.

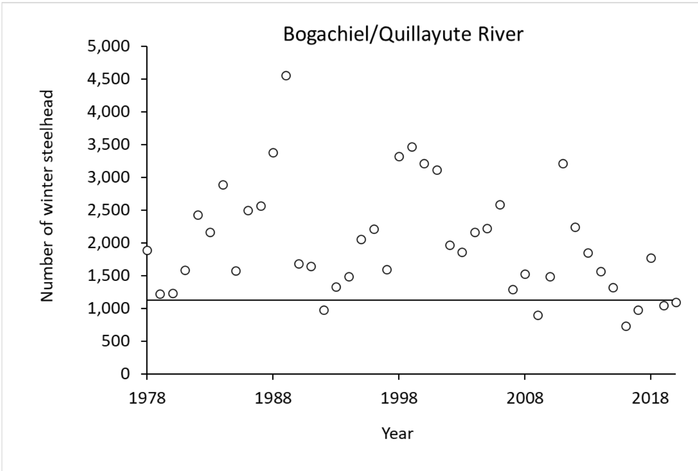


Figure 6. Annual run size and escapement of wild winter steelhead in a) entire Quillayute River watershed system, and annual escapement of wild winter steelhead in the four major individual tributaries, which include the b) Dickey River, c) Sol Duc River, d) Calawah River, and e) Bogachiel and Quillayute River proper. Black circles represent run size while open circles represent escapement. The escapement goals for individual rivers within the Quillayute River system are as follows: Calawah River (1,740 fish), Sol Duc River (2,910 fish), Bogachiel and Quillayute River proper (1,127 fish), and Dickey River (123 fish), resulting in a system-wide goal of 5,900 fish (Duda et al. 2018). Years when escapement falls below the escapement goal indicate that the fishery did not meet its spawner target goal.

Mean annual abundance of wild winter steelhead in the Quillayute River from 1978-2017 has declined by 38% compared to historic mean annual abundance from 1948-1960 (Table 4). The decline increases to 61% when compared to the mean annual abundance for the most recent five-year period (Table 4). If the population continues to decline at its current rate (1996-2020), run sizes in the coming decade to decades could – on average – come in lower than the escapement goal, effectively eliminating the potential for future fisheries while simultaneously increasing the risk of extinction.

iii. Other Populations

There are four other small populations of wild winter steelhead in WRIA 20, including the Tsoo-Yess, Wa'atch, and Ozette Rivers, which are located north of the Quillayute River, and Goodman Creek and Mosquito Creeks, which are located between the Quillayute and Hoh Rivers.

Goodman Creek is the only stream for which we could find monitoring data on escapement (Table 1). From 1995-2020 the mean annual escapement was 184 wild steelhead (range of 45 – 374) and consequently, it has not met its escapement goal of 206 fish once in the past decade ending in 2020 (Table 1). In fact, Goodman Creek has met its escapement goal only once since 2004. Cram et al. (2018) estimated the average annual escapement has declined by

54% over its period of record, and the decline has continued through 2020. Data on run size and escapement was otherwise not available.

b. WRIA 21 Populations

i. Quinault River

From 1978 to 2020 the mean annual winter steelhead run size was 5,883 fish (range = 2,179 – 9,726) in the Quinault River (Table 1). Unlike the Hoh and Quillayute Rivers, the Quinault River is managed in two sections, including: (1) the lower part below the lake, which is owned and managed by the Quinault Indian Nation, and (2) the upper section above the lake, which is owned mostly by the Olympic National Park and its fish managed by WDFW (Figure 2, from Cram et al. 2018). There is no escapement goal for the lower Quinault River, but there is an escapement goal of 1,200 fish for the upper section (Cram et al. 2018).

There is a significant declining trend in Quinault River wild winter steelhead (Table 1). Cram et al. (2018) estimated a decline of 69% for escapement of wild steelhead in the Lower Quinault River and an increase of 24% in escapement for the Upper Quinault River (Figure 2). McMillan et al. (2022) grouped the upper and lower river because run size cannot be estimated for each component of the population separately, and they estimated a 44% decline (Table 1). The McMillan et al. (2022) estimate equates to a loss of 1,052 (590 - 1,510) adults per decade (Figure 4; $F_{1, 32}$ = 21.20, P < 0.001: McMillan et al. 2022). We have access to the full data set from 1978-2020 (included in our data package) and the declining trend has continued, with the smallest run sizes on record occurring in the most recent five-year period (Figure 7). The mean escapement has been 3,107 fish with a range of 1,366 to 5,774 fish (Table 1). While it is unclear whether the population would be meeting a biologically defensible escapement goal for the whole watershed, the declining trend is concerning and more restrictive fishery practices, including early closures, were implemented in 2020 and 2021 to limit impacts on the small run sizes.

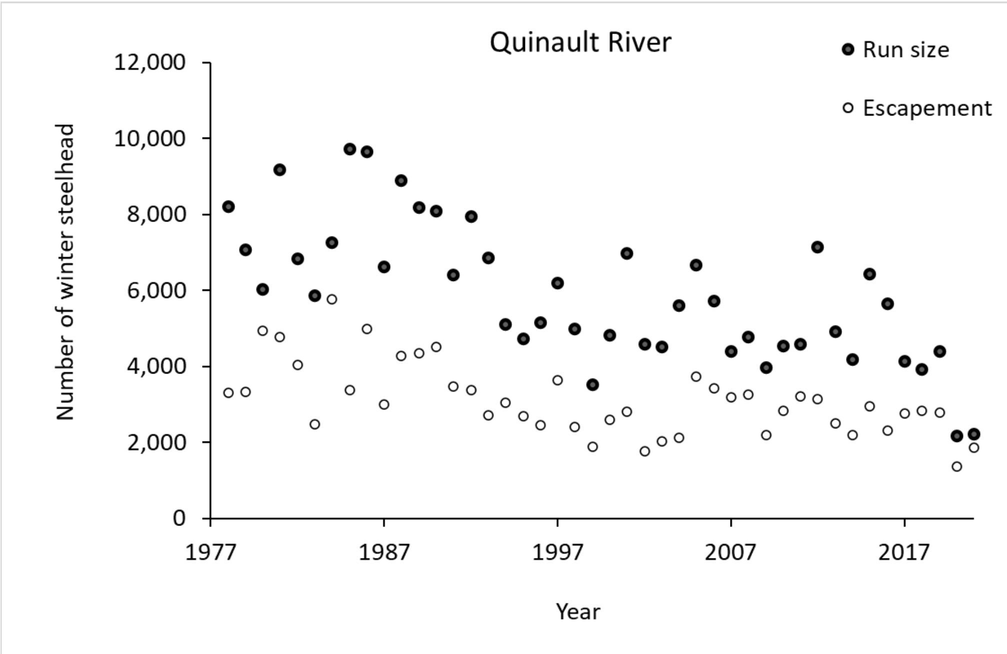


Figure 7. Annual run size and escapement of wild winter steelhead in the Quinault River from 1978-2021. There is no escapement goal for the entire Quinault population, as it is delineated into Upper (above the lake) and Lower (below the lake) populations. Figure 2 illustrates the trend by Cram et al. (2018) for the Upper and Lower Quinault populations.

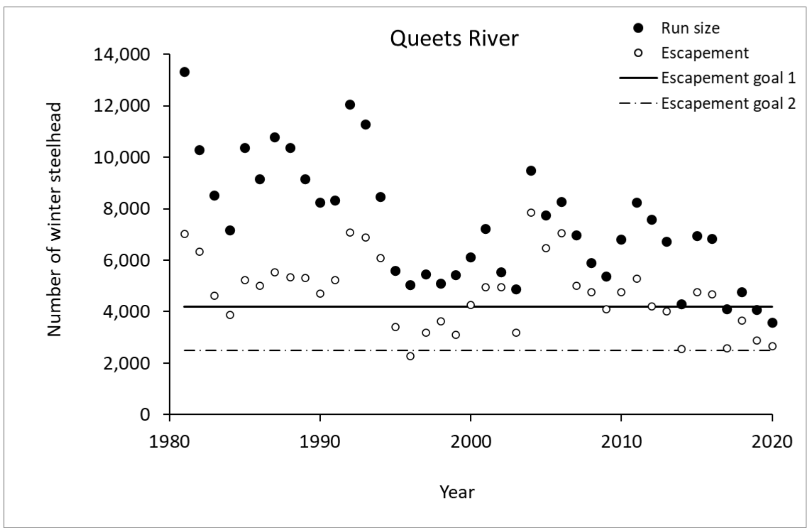
Overall, McMillan et al. (2022) estimated that the contemporary mean annual abundance of wild winter steelhead in the Quinault River from 1980-2013 declined by 63% when compared to historic mean annual abundance from 1948-1960 (Table 4). The decline increases to 80% when compared to the mean annual abundance for the most recent five-year period (Table 4). If the decline rate of 1,052 adults per decade continues, Quinault River winter steelhead will decline from their recent five-year average of 3,370 fish down to 2,318 fish during the following decade. This concern is particularly heightened now because the dramatic declines in 2019 and 2020 highlight how quickly the population could further deplete (Figure 7).

ii. Queets River

From 1980-2018 the mean annual winter steelhead run size was 7,648 fish (range = 4,200 – 13,309) in the Queets River (Table 1). Like the Hoh and Quinault Rivers, there is a significant declining trend for winter steelhead in the Queets River from 1980-2017 (Figure 4), which includes a 12% decline in the escapement trend for its major tributary, the Clearwater River, from 1980-2013 (Figure 2). The extent of decline is greater in McMillan et al. (2022) (45% decline) because Cram et al. (2018) (29% decline) only used data up until 2013, while McMillan et al. (2022) included data up until 2018. The run size continued to decline after 2013 (Figure 4). The decline reported by McMillan et al. (2022) equates to a loss of 1,220 (640, 1,820) adults per decade ($F_{1,36}$ = 19.56, P < 0.001).

Unlike the Hoh River population, the co-managers have not reached an agreement on an escapement goal for the Queets River population, which has affected management for several decades. The Queets Tribe uses a goal of 2,600 fish, while WDFW uses a goal of 4,200 fish (Cram et al. 2018). We show escapement in relation to the latter (Figure 8a) because it is the only goal for which we can find scientific support (Gibbons et al. 1985). The Clearwater River is a major tributary of the Queets watershed that has a separate escapement goal of 1,450 fish (Cram et al. 2018). Clearwater steelhead run size and escapement is included in the overall data for the Queets River population, but Figure 8b depicts the escapement of wild winter steelhead for just the Clearwater River. As a result of the sharp declines in run size in recent years (Figure 8a), the Queets River fishery (as with the Quinault River) was closed to protect small run sizes.

a.



b.

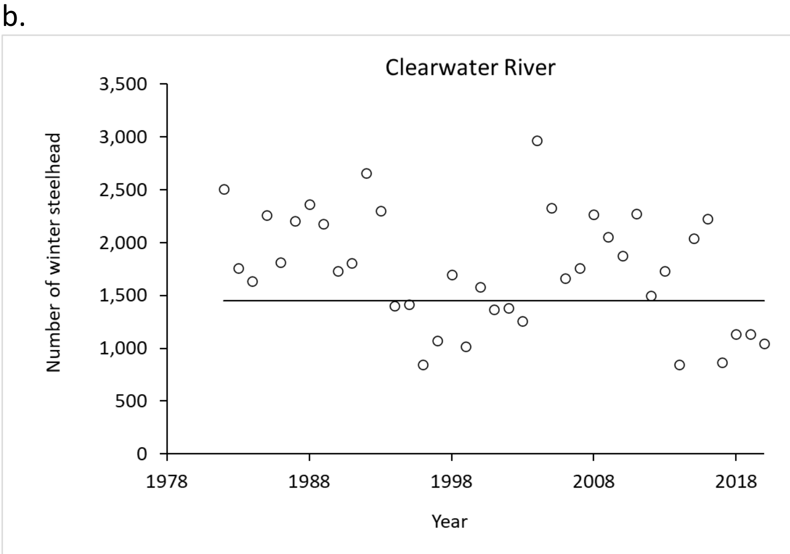


Figure 8. Annual run size and escapement of wild winter steelhead in the a), Queets River from 1980-2020 and b) escapement of wild winter steelhead in the Clearwater River, a major

tributary. Black circles represent run size, open circles represent escapement. In the Queets River panel, the bold line depicts the 4,200 fish escapement goal (Escapement goal 1) that is used by WDFW and is recognized by the National Park Service (a substantial portion of the Queets River watershed lies within the Olympic National Park) and the 2,500 fish escapement goal used by the tribe (Escapement goal 2). The bold line in the Clearwater River panel refers to WDFW's escapement goal of 1,450 fish. Years when escapement falls below the escapement goal indicate that the fishery did not meet its spawner target goal.

The mean annual abundance of wild Queets River winter steelhead from 1980-2017 has declined by 50% when compared to historic mean annual abundance from 1948-1960 (Table 4). The decline increases to 69% when compared to the mean annual abundance for the most recent five-year period (Table 4). However, as we mentioned previously, the declines could be more substantial. For example, McMillan et al. (2022) analyzed one year of cannery data in 1923, when more than nominal cans of steelhead were reported. The run size estimate is 32,659 wild steelhead (Table 4), which would represent a decline of 86% compared to the most recent five-year average run size. If the decline rate of 1,220 adults per decade continues, wild winter steelhead abundance in the Queets River will decline from its recent five-year average of 4,658 fish down to an average of 3,438 fish during the following decade. Like the Hoh River, this suggests that in just one decade run sizes would be coming in under the 4,200 fish escapement goal set by WDFW and would be encroaching on the 2,400 fish goal set by the tribe.

Unfortunately, run sizes in recent years have declined rapidly, resulting in fishery closures and suggesting that the population is on a chronic path that could lead to extinction unless the 40-year trend is reversed.

iii. Other populations

There are four smaller populations of wild winter steelhead in WRIA 21, including Kalaloch Creek, Moclips River, Raft River, and the Copalis River populations (Table 1). Among those populations, we could find escapement data only for the Moclips River, which spans a relatively short period that is not up to date. From 1988-2000, the mean escapement in the Moclips River was 299 steelhead (range = 130 – 560), and based on that data set, Cram et al. (2018) reported a 27% increasing trend in abundance (Table 1 and Figure 2). Given the limited period of record and overall missing data, it is difficult to draw conclusions about the status of these populations.

c. WRIA 19 Populations

There are ten small streams in WRIA 19 that support populations of wild winter steelhead (Table 1). The Hoko River has the largest population with a mean annual escapement of 566 fish (range = 193 – 990). The Hoko River has met an escapement goal of 400 fish in 80% of the last ten years (Table 1). Based on the period of record, however, the Hoko River population declined by 40% from 1985-2013 (Table 1) and that decline has not subsided in recent years (WDFW Score website, Accessed 3/20/2022: https://fortress.wa.gov/dfw/score/score/species/population_details.jsp?stockId=6357).

The Pysht River (and tributaries) and the Clallam River support the second and third largest populations, respectively, and both populations are in long-term decline (Cram et al. 2018). Cram et al. (2018) estimate that winter steelhead abundance declined by 21% in the Pysht River from 1995-2013 and 27% in the Clallam River from 1999-2013 (Table 1). More recent data indicates these declines have continued through 2020 in the Pysht River (WDFW Score website, Accessed 3/20/2022: https://fortress.wa.gov/dfw/score/score/species/population_details.jsp?stockId=6343) and the Clallam River populations (WDFW Score website, Accessed 3/20/2022: https://fortress.wa.gov/dfw/score/score/species/population_details.jsp?stockId=6350). Consequently, the Pysht River population has only made its escapement goal in 70% of the past ten years (Table 1).

Salt Creek supports the fourth largest population in WRIA 19. The population has only met its escapement goal once in the past decade (Table 1) and once overall since 2004. Cram et al. (2018) estimated wild steelhead abundance declined by 43% from 1995-2013, and more recent data suggests the decline has not reversed but may have stabilized (WDFW Score website, Accessed 3/20/2022: https://fortress.wa.gov/dfw/score/score/species/population_details.jsp?stockId=6329).

Escapement estimates are also available for populations in Deep Creek and the East and West Twin Rivers from circa 1995 through 2020 (WDFW Score website, Accessed 3/20/2022: https://fortress.wa.gov/dfw/score/score/species/population_details.jsp?stockId=6343). Using this data set, a regional recovery document suggests that all three populations of wild steelhead were in decline and there is a high level of concern about the depressed status of the stocks (NOPLE 2015). Of particular concern, wild winter steelhead have only met the escapement goal of 86 steelhead once in the past decade (Table 1) and in two of the past twenty years. Even worse, the population in West Twin River has missed its escapement goal of 103 fish every year in the past decade (Table 1) and every year dating back to 2001. We could not find any estimates of escapement for the Sekiu, Sail, and Lyre Rivers (Table 1).

2. Summer Steelhead

The co-managers do not monitor any populations of wild summer steelhead on the Olympic Peninsula, nor do they have any established escapement goals for these populations. The only contemporary data on abundance and distribution of adult summer steelhead comes from snorkel surveys conducted by Olympic National Park (e.g., Brenkman et al. 2012), which manages summer steelhead habitat and fisheries in the park, and McMillan (2022) (Table 7). Those snorkel surveys were conducted in the Sol Duc River, Calawah River, SF Calawah River, NF Calawah River, Sitkum River, Bogachiel River, SF Hoh River, and the EF and NF Quinault Rivers. Snorkel surveys were generally conducted annually from September through October and, occasionally early November, to ensure summer steelhead had time to migrate through the watershed and reach the upper most habitats where they stage for several months before spawning from winter through spring.

Table 7. Counts of summer steelhead via snorkel surveys in the Calawah, Sitkum, SF Calawah, NF Calawah, SF Hoh, Sol Duc, EF Quinault, and NF Quinault Rivers, including year(s) of surveys, mean abundance, and mean proportion of hatchery steelhead (Figures 9 – 11). Data from McMillan (2022) is based on snorkel surveys conducted in early September through early November, while data from Sam Brenkman et al. is based on snorkel surveys conducted in late summer (Brenkman et al. 2012). Standard Deviation and range is provided for McMillan (2022), while only range is provided for Brenkman et al. (2012).

Study and population	Year (s)	Mean abundance (SD and range)	Mean proportion hatchery steelhead (SD range)
McMillan 2022			
Calawah River system[1]	2002	89 (wild), 214 (hatchery)	N/A
Sitkum River	2002 – 2006, 2009 – 2021	53 (25.2, 19-105)	3% (3%, 0% – 10%)
SF Calawah River	2003 – 2006	18 (5.6, 10 – 23)	7% (6%, 0% – 13%)
NF Calawah River	2000 – 2006, 2009 – 2021	4 (4.1, 0 – 14)	33% (35%, 0% – 100%)
SF Hoh River	2000 – 2006, 2009 – 2014, 2016 – 2019	7 (3.9, 1 – 16)	41% (12%, 20% – 67%)
Brenkman et al. 2012			
Bogachiel River	2005 – 2010	15 (8 – 26)	13% (0% – 23%)
Sol Duc River	2005 – 2010	6 (0 – 15)	N/A
SF Hoh River	2005 – 2010	8 (4 – 12)	40% (14% – 76%)
EF Quinault River	2005 – 2010	17 (4 – 35)	16% (0% – 38%)
NF Quinault River	2005 – 2010	3 (0 – 8)	43% (0% – 100%)

[1]Calawah River survey data from 2002 is a census of the entire watershed accessible to adult summer steelhead in September and October.

Below we summarize the results of the surveys and when possible, compare them to our historic estimates. The findings generally suggest that wild summer steelhead now only exist at exceptionally low abundance (Brenkman et al. 2012; McMillan 2022) and are headed towards extirpation if their status and trends are not rapidly improved.

a. WRIA 20 Populations

i. Hoh River

Snorkel surveys on the Hoh River have been conducted by the National Park Service (Brenkman et al. 2012) and McMillan (2022). These surveys focused on several long sections where summer steelhead were observed staging during previous years. The results are not a census, but rather, an estimate of peak abundance in those important habitats.

The snorkel surveys in the SF Hoh River by Brenkman et al. (2012) sometimes covered a longer section of stream than McMillan (2022), though not always, and in most years their surveys overlapped spatially. Unfortunately, both sets of surveys indicate a very low abundance of wild summer steelhead. Brenkman et al. (2012) data cover the period from 2005 – 2010 while surveys by McMillan (2022) ranged from 1999-2006, 2009-2014, and 2016-2019. Brenkman et al (2012) reported a mean abundance of eight wild summer steelhead with peak annual peak counts of 12 and 16 fish (Table 7). The longer time series by McMillan (2022) suggests summer steelhead abundance has remained low, with a mean abundance of seven wild summer steelhead (Figure 9). The lowest abundance estimates have almost all occurred from 2016 to 2020 (McMillan 2022).

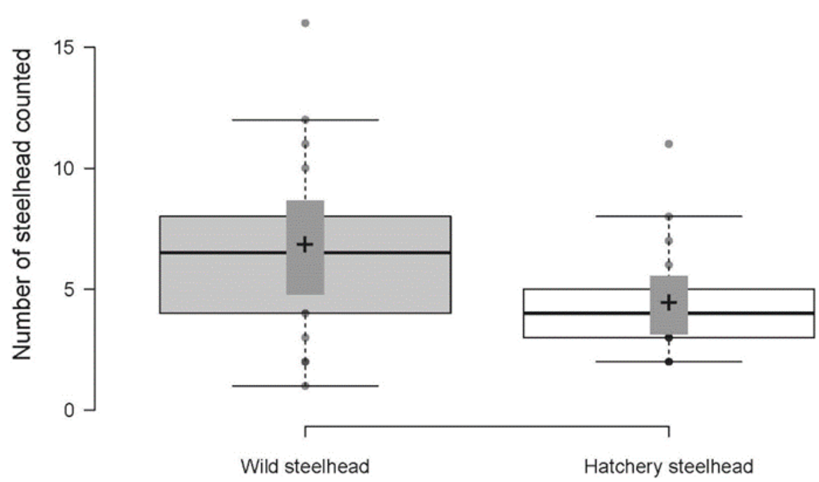


Figure 9 (From McMillan 2022). Box-whisker plot displaying mean (**+**) annual abundance of wild and hatchery summer steelhead counted during snorkel surveys from late-summer through early-fall in the SF Hoh River (1999-2006, 2009-2014, 2016-2019). Internal rectangle indicates 95% confidence interval for the mean. Tukey whiskers extend to data points that are less than 1.5 interquartile range away from $1^{st}/3^{rd}$ quartile.

Both surveyors also reported high levels of hatchery summer steelhead, including more hatchery than wild adults in some years (Table 7). For example, the mean abundance of four hatchery steelhead was not significantly lower than the mean estimate of seven wild steelhead (Figure 9). Hatchery summer steelhead are not released into the Hoh River, but hatchery summer steelhead could have strayed from their release locations in the Bogachiel or Wynoochee Rivers, eventually ending up in the headwaters of the SF Hoh River.

Unfortunately, there is almost no information on summer run abundance in the mainstem Hoh River, which likely supports most of the population. Nonetheless, there is also little evidence to suggest that mainstem component is faring any better, even if more abundant. Overall, the data indicate that SF Hoh River wild summer steelhead exist at low levels of abundance. Considering our rough historic estimates of population size based on recreational catch – a mean run size of 472 steelhead with a high of 1,096 fish (Table 6) – it is likely that the SF Hoh River sub-component is critically depleted, and that the entire population could be threatened with extinction.

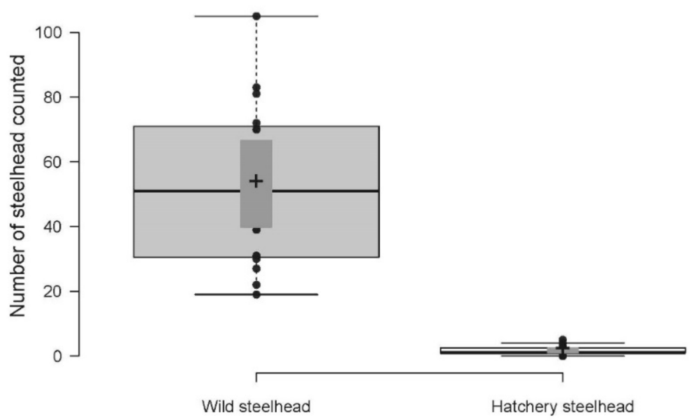
ii. Quillayute System

Data on abundance of wild summer steelhead is limited to snorkel surveys conducted in the Sol Duc, Bogachiel, Calawah, NF Calawah, SF Calawah, and Sitkum Rivers (Table 7). Surveys by Brenkman et al. (2012) found a mean abundance of 15 wild summer steelhead in the upper Bogachiel River from 2005 – 2010 and only six wild summer steelhead in the upper Sol Duc River (Table 4). In some years no wild summer steelhead were observed in the Sol Duc River surveys conducted by Brenkman et al. (2012).

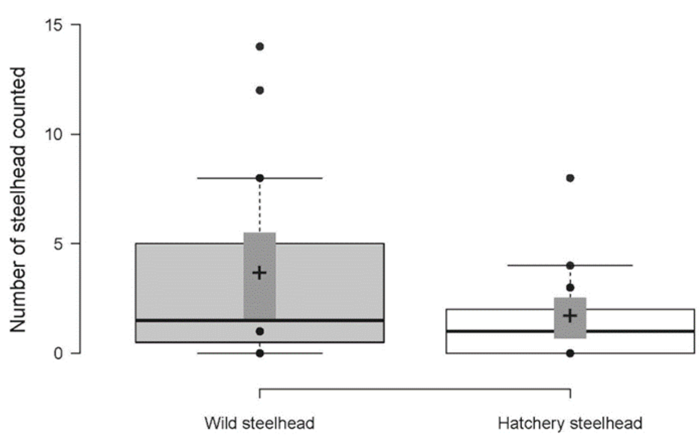
McMillan (2022) conducted snorkel surveys in the NF Calawah River, Sitkum River, and SF Calawah River (only 2003 – 2006) from circa 2000 – 2006 and from 2009 – 2021. These locatoins are thought to be the predominant staging and spawning areas for summer steelhead (McMillan 2006). Data is also available from a snorkel census in 2002 that covered 85% of all habitats available to adult steelhead during the summer (e.g., McMillan et al. 2013). The diver counted adult steelhead during the survey but did not publish the data in McMillan et al. 2013.

During the 2002 census, a total of 89 wild summer steelhead and 214 hatchery summer runs were counted (Table 7). Snorkel surveys in the Sitkum River and NF Calawah River produced a mean annual abundance of 53 and five wild summer steelhead, respectively (Figure 10a and 10b). Peak abundance in the Sitkum River snorkel counts was 105 wild summer runs in 2005, but its lowest estimates have dropped to 19 fish (Table 7) and counts were less than 30 adults in 2020 and 2021 (McMillan 2022). Surveys of sections of the upper SF Calawah from

2003-2006 suggest it also has summer steelhead staging and spawning habitat, though overall abundance appears lower than in the Sitkum (McMillan 2022). Counts of wild summer steelhead in the NF Calawah River have generally ranged between 0 – 10 fish (Figure 10b), with the two peaks occurring in 2001 and 2002 (McMillan 2022). A drought in 2002 dewatered all of the lower NF Calawah except for the lower-most 1km of stream, which killed thousands of salmonids including all the adult steelhead, and since then, abundance of summer runs has been at very low levels (McMillan 2022). In fact, zero adult steelhead were counted in the NF Calawah from 2014-2021 except for 2015, when a single adult was observed (McMillan 2022). The snorkel survey data has not been adjusted to account for potential sources of error in the diver counts, and there is not a whole basin estimate for summer steelhead abundance. With those caveats, the current counts still suggest the Quillayute River system population is severely depleted in comparison to the historical catch estimates.



a.



b.

Figures 10 a-b (From McMillan 2022). Box-whisker plot displaying mean (**+**) annual abundance of wild and hatchery summer steelhead counted during snorkel surveys from late-summer through early-fall in the, a) Sitkum River (2003-2006, 2009-2019), and b) NF Calawah River (2000-2006, 2009-2021). Internal rectangle indicates 95% confidence interval for the mean. Tukey whiskers extend to data points that are less than 1.5 interquartile range away from 1st/3rd quartile.

b. WRIA 21 Populations

i. Quinault River

The only data we could find on the abundance of summer steelhead in the Quinault River is from Brenkman et al. (2012). They conducted snorkel surveys in the EF and NF Quinault Rivers from 2005 – 2010. The mean annual abundance of wild summer steelhead during those surveys was 17 and three wild steelhead in the EF and NF Quinault Rivers, respectively (Table 7). The surveys suggest the Quinault River population of wild summer steelhead has dramatically declined from the historic estimates we calculated based on recreational catch from the 1950s through 1976, which show a mean run size of 649 summer steelhead with a peak run size of 1,788 fish (Table 6). Although the current run size is unknown, it appears the population of summer steelhead in the Quinault River is less than 1% of its historic abundance and is close to extirpation.

ii. Queets River

The are no snorkel survey counts to determine the population trend of summer steelhead in the Queets or Clearwater Rivers. McMillan (2006) estimated that, based on available catch data, the run size of wild summer steelhead returning to the Queets and Clearwater Rivers combined was no more than 100 fish. If the population is close to 100 fish, that would suggest the population is now only a fraction of the historic estimate we calculated based on catch, where the mean population size was 802 summer steelhead and the peak was 1,492 fish (Table 6). This uncertainty creates a great deal of concern for the status and trajectory of the population moving forward.

XI. PRODUCTIVITY

In comparison to estimates of abundance, productivity data is very limited for the Olympic Peninsula Steelhead DPS. Cram et al. (2018) reported on nine productivity data sets in the Olympic Peninsula DPS from circa 1980-2013, eight of which focused on spawner-per-spawner production (Figure 11) and one that focused on smolt-to-adult return survival (Figure 11). Hall et al. (2016) reported on the productivity of adults to juveniles in the Twin Rivers. We reviewed the co-managers' annual data sets and could only find productivity information for three major populations, including 1) recruits-per-spawner data for the Quillayute River (1978-2015), 2) Hoh River (2000-2016) populations of wild winter steelhead, and 3) a raw estimate of number of adult returns per wild/natural winter steelhead smolt in the Queets River (1982-

2013). Cram et al. (2018) did not list which populations were included in their productivity estimates, but it would seem likely that our data sets overlap.

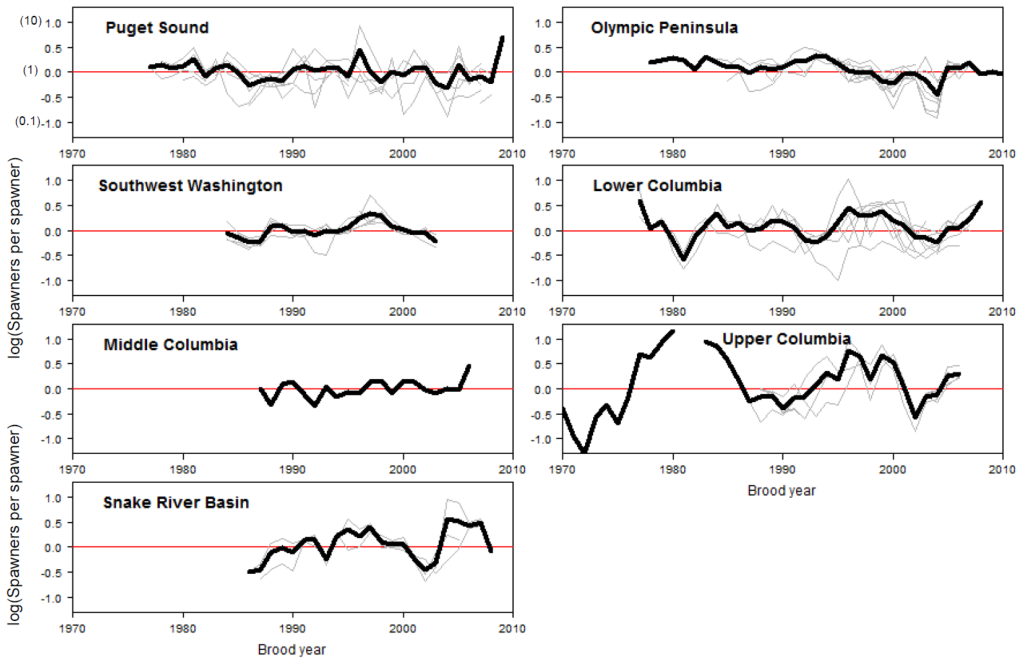


Figure 11 (From Cram et al. 2018). For each DPS, population growth rates represented as the log of the numbers of natural-origin adult spawners produced per spawner for all populations with suitable data for brood years 1970 to 2010. The thick black line represents the average value for each DPS, while the thin lines represent data from individual populations within each DPS. The y-axis numbers in parentheses for the Puget Sound figure are the non-transformed values for reference. A log productivity of 0, shown by the red lines, corresponds to an untransformed productivity of 1 spawner per spawner (replacement). Data are available for multiple populations in each DPS that occur in Washington, including: 7 populations in the Puget Sound DPS; 8 in the Olympic Peninsula DPS; 5 in the Southwest Washington DPS, 8 in the Lower Columbia River (LCR) DPS, 1 in the Mid-Colubmia River (MCR) DPS, 4 in the Upper Columbia River (UCR) DPS, and 3 in the Snake River Basin (SRB) DPS.

There appears to be a slight decline in spawner-to-spawner recruitment from 1978-2010 for wild winter steelhead in the Olympic Peninsula DPS, and after the mid-1990’s, the populations increasingly fail to replace themselves (Figure 11, Cram et al. 2018). Smolt-to-adult return declined significantly over time for the single wild winter steelhead population evaluated by Cram et al. (2018) (Figure 12).

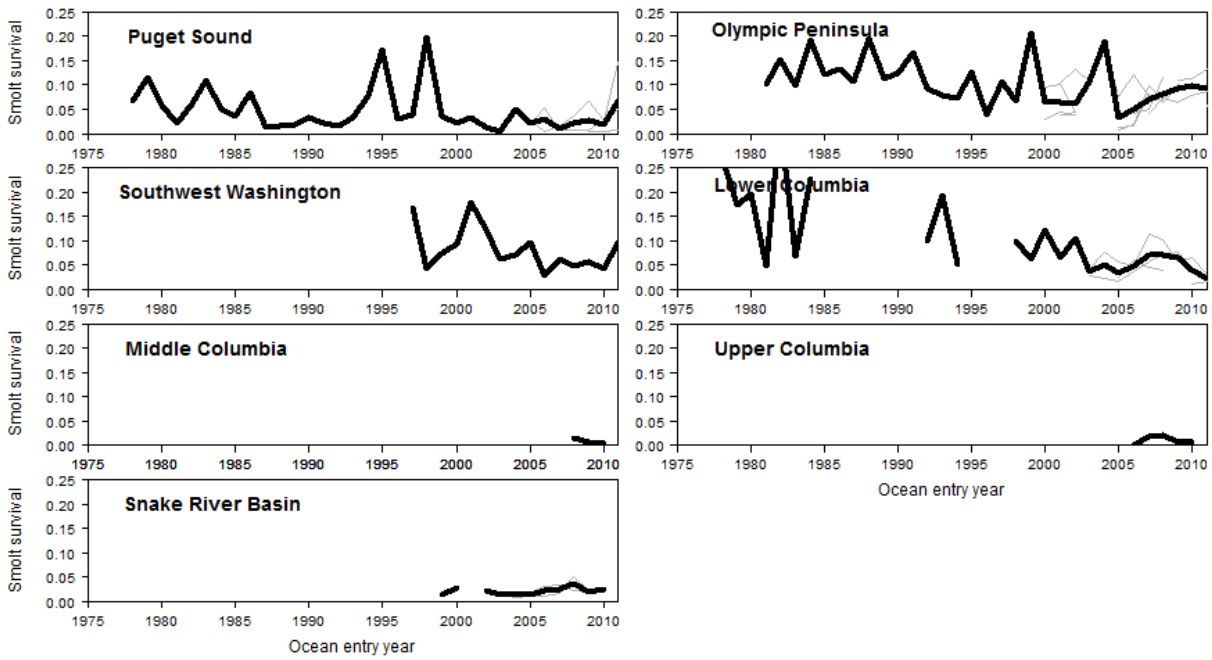
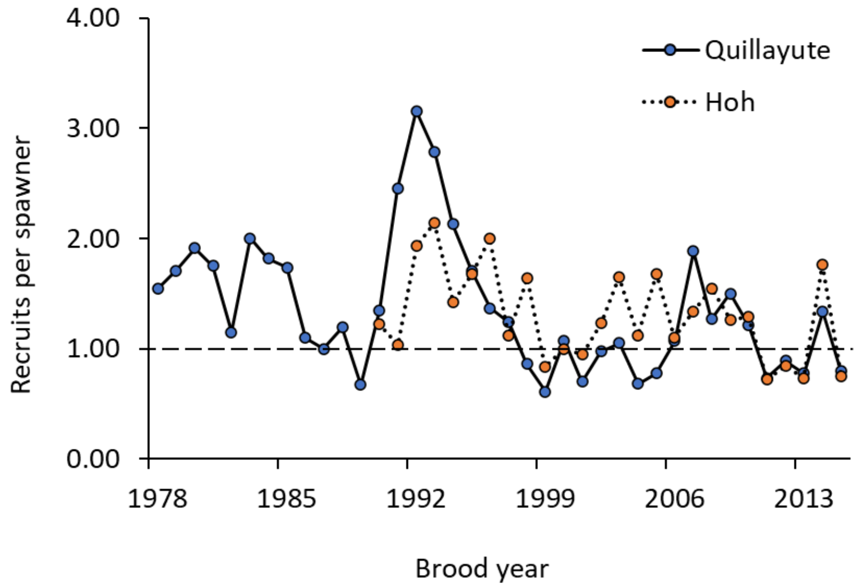


Figure 12 (from Cram et al. 2018). For each DPS, smolt survival, estimated as smolt-to-adult return (SAR) rates of natural-origin steelhead spawners from smolt outmigration year 1975 to 2011 (16 populations represented: 2 populations and 1 sub-population in Puget Sound; 4 sub-populations and 1 population in Olympic Peninsula; 1 population each in Southwest Washington, MCR, and UCR DPSs; 3 populations in the LCR DPS; and 2 populations in the SRB DPS). The thick black line represents the average SAR value for each DPS, and the thin grey lines represent data from individual populations therein.

Cram et al. (2018) indicated recruits-per-spawner productivity was not available for any populations in the DPS. However, the co-manager data sets we analyzed referred to “recruits per spawner” estimates for the Quillayute and Hoh Rivers. Perhaps this is just a minor difference in terminology. We include data on recruits per spawner and extend the productivity period of record for the Quillayute River population to 1978-2015 and the Hoh River population to 1990-2016, both of which suggest a continued declining trend (Figure 13a).

a.



b.

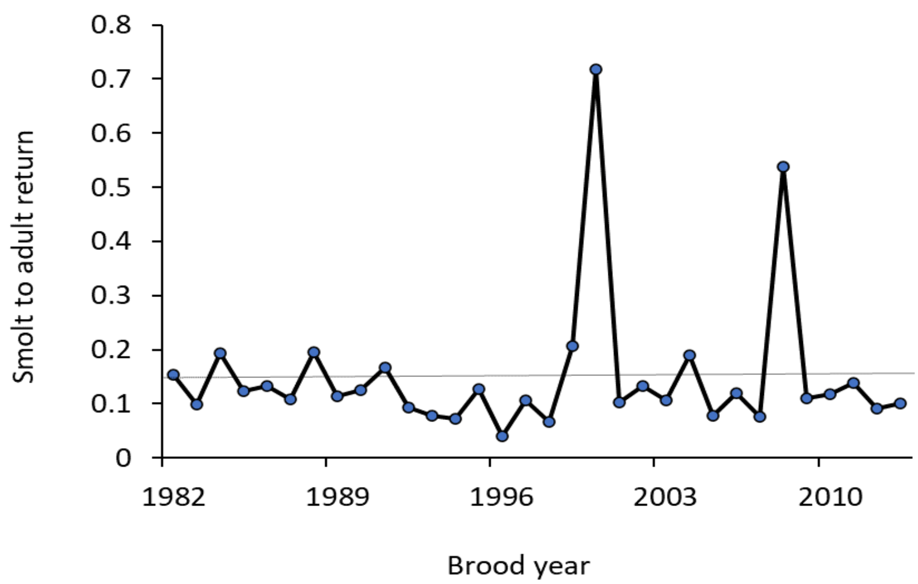


Figure 13. Estimates of wild steelhead productivity, including a.) recruits per spawner for wild winter steelhead in Quillayute (1978-2015 brood years) and Hoh Rivers (1990-2016 brood years), and b.) smolt to adult return data for wild winter steelhead in Queets River from 1982-2013. Dashed line in Panel a.) represents replacement of 1 recruit/spawner, while dashed in Panel b.) represents mean smolt to adult return rate for Queets wild steelhead during period of record, which is 0.15.

Of particular concern, productivity has been below replacement (recruits per spawner = 1.0) in four of the past ten years in both the Quillayute and the Hoh River populations, and the Hoh has been below 1.0 in ten (50%) of the past twenty years (Figure 13a). There is no clear trend in the Queets River population smolt-to-adult return data (Figure 13b), largely because of two peaks in survival that seem abnormally high. Perhaps the co-managers adjust their estimates or there were errors in the data sets that we had access to.

Cram et al. (2018) also reported that variation in freshwater per-capita productivity among populations in the Olympic Peninsula DPS declined as abundance increased, as did Hall et al. (2016) for the Twin Rivers. Indeed, the estimates of recruits per spawner for the Quillayute (Figure 14a) and Hoh River populations (Figure 14b) reveal a similar pattern. As Cram et al. (2018) described, the relationships suggest there are potential density dependent effects on productivity (Walters et al. 2013; ISAB 2015). This could suggest habitat is limiting (Cram et al. 2018). However, density-dependence effects on growth can be highest at low densities (Lobon-Cervia 2007), and there are other factors to consider before we can conclusively determine that productivity is being solely limited by habitat potential and that the populations have the compensatory capacity to rebuild from their current low levels of abundance.

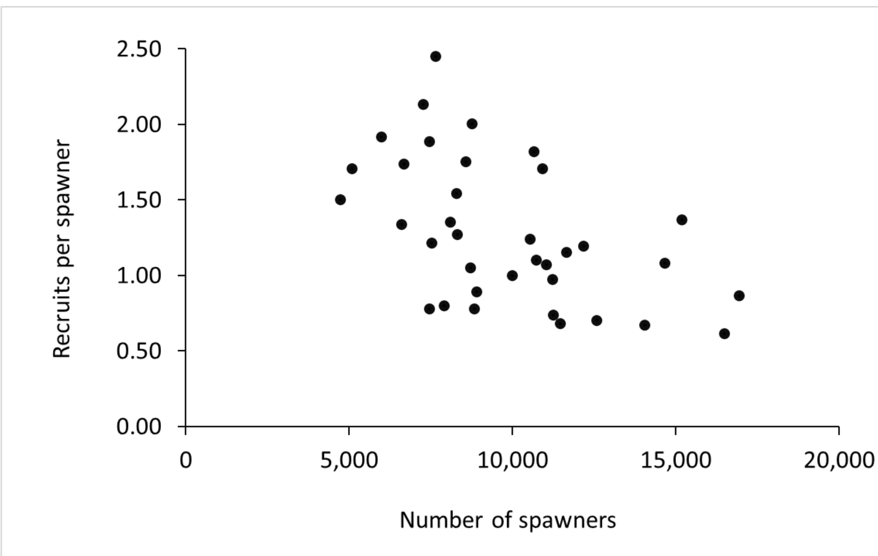


Figure 14a. Estimates of recruits per spawner for wild steelhead populations in the Quillayute River.

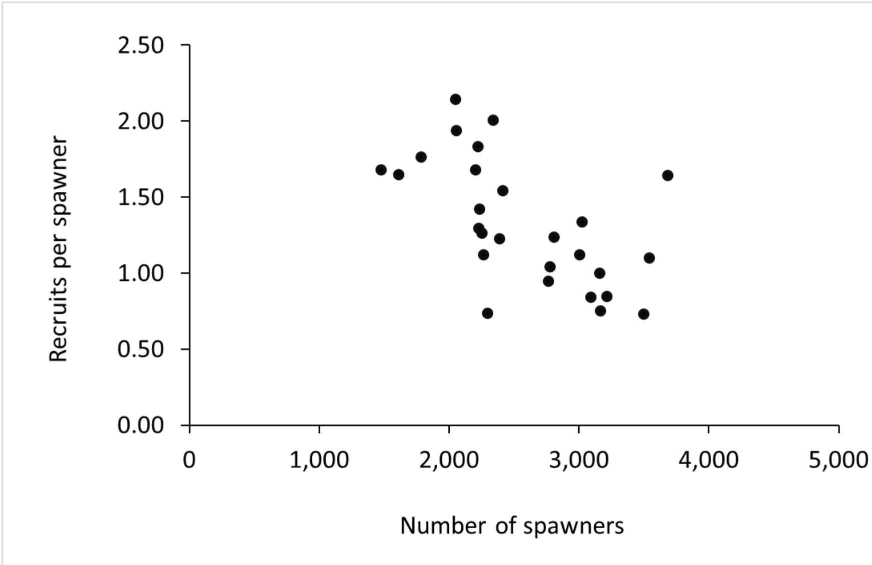


Figure 14b. Estimates of recruits per spawner for wild steelhead populations in the Hoh River (1990-2016).

The first factor to consider is the lack of data pointing to specific causes of productivity declines. While there is suggestive evidence of density dependence in the Quillayute and Hoh River populations based on the adult-to-adult recruits per spawner estimates, the factors limiting these populations are unknown due to a lack of data on juveniles and smolts. There has not been a systematic, long-term decline in marine survival on the Washington Coast (Kendall et al. 2017; Welch et al. 2018), so it is possible that fishery, hatchery, and habitat effects have combined to reduce the productivity of the populations. For example, the historic run timing of wild winter steelhead has changed dramatically in the Quillayute, Hoh, and Queets River populations with early-entering wild fish being replaced by hatchery fish (Figure 15; McMillan et al. 2022), resulting in compressed migration timing that could increase density-dependent effects by reducing spatial and temporal distribution and diversity of adults and juveniles.

Several studies on Atlantic salmon (Einum et al. 2006, 2008; Teichert et al. 2011; Finstad et al. 2013) indicate that contraction in spatial distribution can lead to heightened density dependence even without changes in habitat conditions. In this vein, declines in abundance could have also resulted in reduced spatial distribution of spawners (Isaak and Thurow 2006), which in turn could contribute to depensation (Atlas et al. 2015). There is also evidence that diversity in run timing can help stagger the use of habitat by newly emerged offspring, essentially increasing the productivity of the existing habitat (Gharett et al. 2013).

Additionally, the frequency of repeat spawners – which can be the most productive individuals (Christie et al. 2018) – has apparently declined in the Queets and the Hoh River populations, and they currently occur at low levels in the Quillayute River and are almost non-existent in the Quinault River (Figure 16). Because life history diversity is critical to the productivity and resilience of steelhead (Moore et al. 2014), it is plausible that productivity will continue to decline because the original fabric of diversity has been greatly reduced. Although this is postulation, the dearth of information outside the adult-to-adult data does not provide the65 necessary insight to determine how and when population productivity is being regulated.

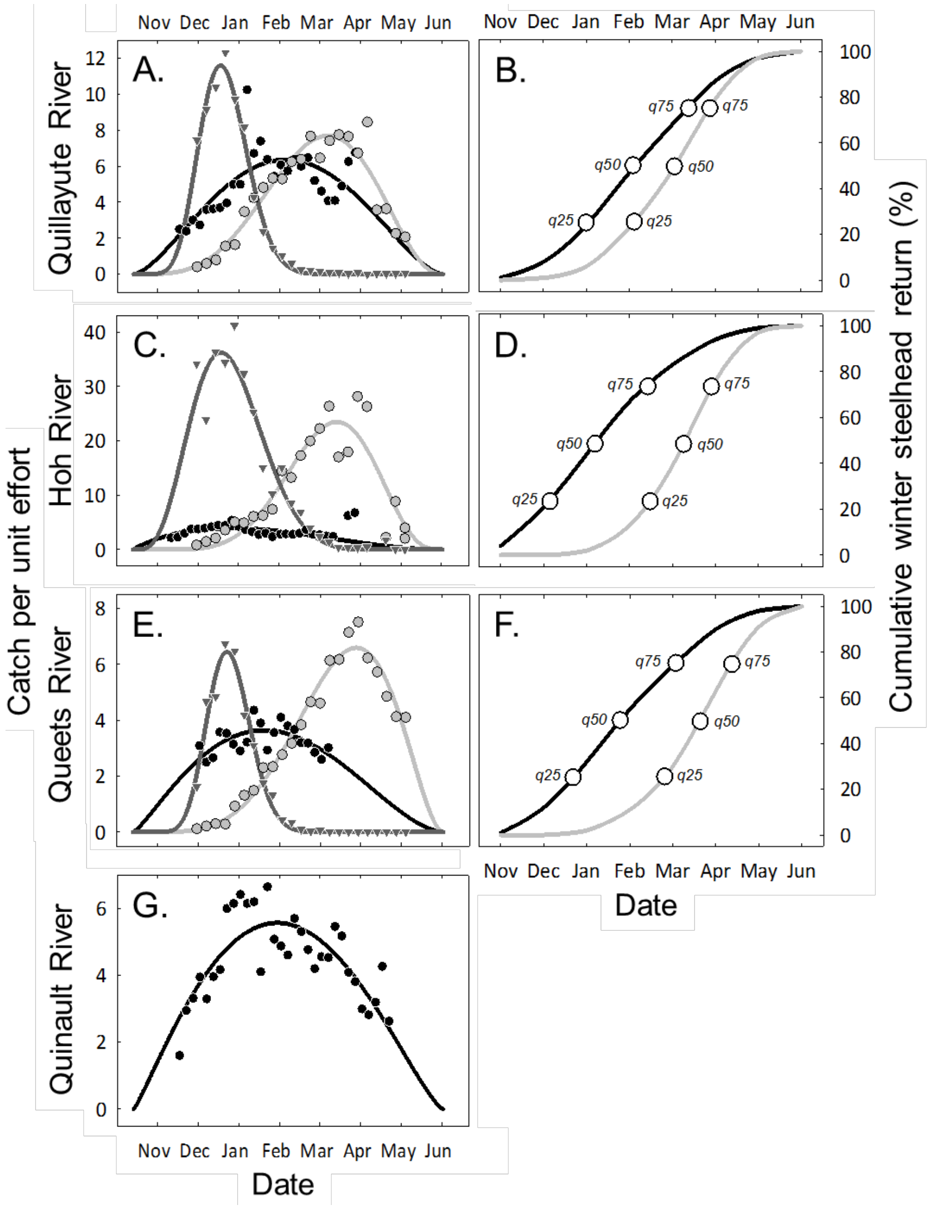


Figure 15 (from McMillan et al. 2022). Historical (circa 1955–1963) and contemporary (2000 - 2017) migration timing estimates based on CPUE of wild and hatchery winter steelhead (left panels) in the (A–B) Quillayute, (C–D) Hoh, and (E–F) Queets rivers, and comparison of cumulative run timing with estimates of dates at which 25% (*q*25), 50% (*q*50), and 75% (*q*75) of the run had passed for historical and contemporary wild winter steelhead (right panels). Dark gray lines and triangles represent contemporary hatchery returns; black lines and black circles represent historical wild returns; and light gray lines and light gray circles represent contemporary wild returns. Run timing estimation in the (G) Quinault River was limited to the historical period because contemporary CPUE data were not available.

The second factor to consider is that productivity estimates for the major populations of wild winter steelhead in Cram et al. (2018) ended in 2010 and the co-manager data sets we accessed (ending in 2015) do not account for the recent declines in run size in the Quillayute, Hoh, and Queets River populations. We also could not find estimates of productivity for the Quinault River population (though we assume the co-managers do have estimates of productivity). It will take several years for the productivity estimates to be updated so that we can better understand the compensatory dynamics that have occurred over the past decade.

Petitioners are concerned that the populations are not compensating as effectively now as they did 20-40 years ago for several reasons. For example, there are increasingly low levels of productivity in the Quillayute River population (Figure 14a). Additionally, the peaks and troughs in annual run size for the Hoh, Queets, and Quinault River populations have generally declined each decade over the period of record (i.e., lower peaks, deeper troughs). These factors have led to high levels of depletion in recent years (Figures 5-8),

The third factor to consider is that there is no data on the productivity of summer steelhead stocks, nearly all of which are completely unstudied. This is a very large data gap. Considering the declines in wild winter steelhead abundance and productivity, there is a high level of risk that summer steelhead are facing an even more dire situation.

The fourth factor to consider is that there is tremendous uncertainty about productivity for numerous smaller populations of wild winter steelhead that inhabit independent tributaries. While these populations are indeed smaller, they collectively represent an important component of diversity that spatially covers a much larger extent of the DPS than the four largest, major populations. An HSRG (2004) report claims productivity is limiting for 15 ½ out of 16 wild steelhead populations (the Quinault River winter steelhead received two ratings), and they associated the reduced productivity with degraded and "inadequate" habitat conditions. And a report by NOPLE (2015) indicates the number of smolts produced in the following WRIA 19 populations had declined: Salt Creek (1995-2006), East Twin River (2001-2005); West Twin River (2001-2005); Deep Creek (1998-2005). NOPLE (2015) did not provide smolt production numbers for the Pysht, Clallam, or Hoko River populations.

In sum, the general assumption, based on fish data and habitat evaluations, is that wild steelhead productivity is in long-term decline for most populations in the Olympic Peninsula DPS. That trend, combined with depleted abundance and diversity, puts them at greater risk of extinction in the coming decades.

XII. DIVERSITY

A robust population should maintain both genotypic and phenotypic diversity and have distributions that are spatially and temporally diverse. The array of life history diversity of a population is analogous to a “Portfolio” in the financial realm, where having different life histories is like having a more diverse financial portfolio (Schindler et al. 2010). The diversity helps buffer individuals and populations from environmental effects, the cumulative effects of which improve the performance and resilience of populations with stronger and more diverse portfolios (Schindler et al. 2010). Moore et al. (2014) applied this to steelhead populations in British Columbia and found that life history diversity helped dampen fluctuations in population abundance and increased their resilience to environmental factors.

Overall, there is very little data on the genetic diversity of wild steelhead on the Olympic Peninsula or on the genetic relationships between the several apparently demographically independent populations. However, there is evidence that life history diversity has been reduced in several ways that could compromise their ability to remain productive and resilient in the face of a changing climate. Important elements of steelhead life-history include: the frequency and age-specific proportions of repeat spawners; proportions of smolt ages of each sex; proportions of spawner ages of each sex; size-specific fecundity; and timing of migrations to spawning habitat.

Repeat spawning is particularly important to the resilience of steelhead populations (Hard et al. 2015; Gayeski et al. 2016). For example, available historical (Withler 1966) and contemporary data from Kamchatka (Pavlov et al. 2001) indicate that the proportion of repeat spawners in the annual returns of the area’s steelhead population historically exceeded 20-30% of total spawners. This is also true for the Queets River, which had relatively high levels of repeat spawners in the 1980s (Figure 16). Such relatively high levels of repeat spawning suggests that this is a life-history characteristic to compensate for high and/or variable juvenile marine survival rates. When this is the case, the proportion of repeat spawners can determine whether a population can sustain levels of recruits per spawner above 1.0 (see Gayeski et al. 2016, pp. 92-93, and Table 1).

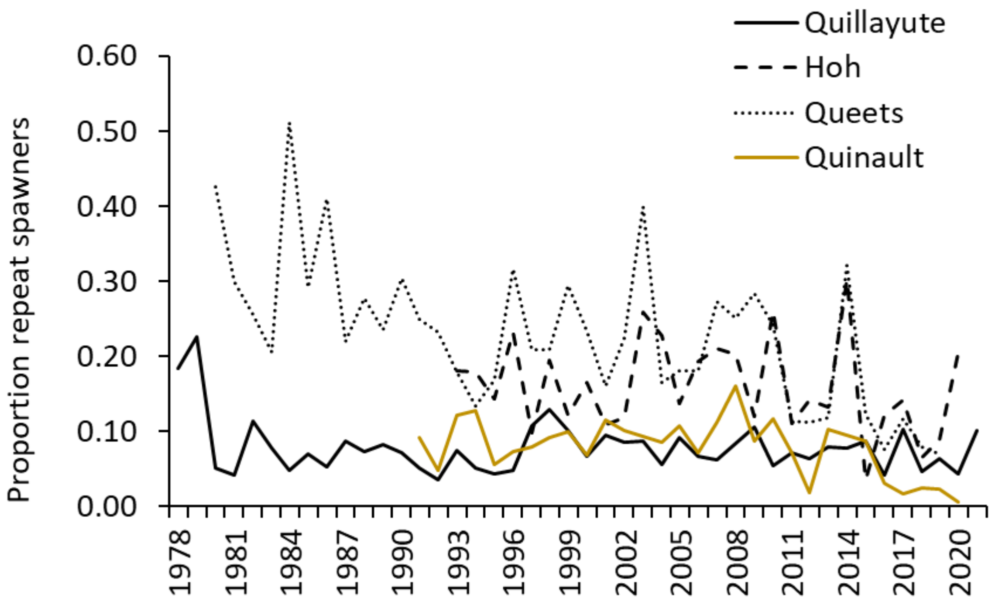


Figure 16. Proportion of repeat spawning individuals (repeats/total run size) by year for wild winter steelhead populations in the Quillayute (1978-2021), Hoh (1993-2020), Queets (1980-2019), and Quinault Rivers (1991-2020).

A. Summer Steelhead

The summer run life history represents an important component of the overall diversity of the Olympic Peninsula Steelhead DPS. There is a heritable basis to the summer run life history in steelhead (Prince et al. 2017; Fraike et al. 2021) that allows them to enter streams in a highly immature state and stage for several months to up to a year in freshwater before spawning. Entering in summer when temperatures are warmer also helps allow them ascend waterfalls that winter runs cannot, providing them with access to habitats to headwater habitats that would otherwise go unused. Thus, losing the summer life history component could reduce, and likely already has, the spatial distribution of Olympic Peninsula steelhead. The extinction of summer steelhead would also create large gaps in migration timing, hindering each population's ability to adapt to and keep pace with changing selective pressures associated with climate change.

As noted by McMillan (2006), the situation for summer steelhead on the Olympic Peninsula is dire:

> "It is apparent that the Olympic Peninsula summer steelhead populations examined are at the edge of extirpation. The Quinault population may be the most dire, with estimated returns of less than 50 fish for the entire watershed whose spawning destinations are further reduced in their split between the

North and East forks – potentially less than 25 fish destined for each. The Clearwater population of the Queets system, and the Sol Duc and Bogachiel populations of the Quileute system may be similarly low, with only 2-3 dozen fish returning to each. In fact, the Quinault, Clearwater, Sol Duc, and Bogachiel populations may already be functionally extinct."

Snorkel data from Brenkman et al. (2012) and McMillan (2022) suggests the situation is even worse now compared to McMillan (2006).

Considering the allelic variants for early migration have not arisen independently via new mutations (Prince et al. 2017) and the life history is at critically low levels of abundance, it is likely the genetic integrity of the populations is impaired. Consequently, there is great concern wild summer steelhead populations are close to extinction and their demise could lead to irreparable loss of the genetic basis for premature migration in the Quillayute, Hoh, Queets, and Quinault River populations.

B. Early Returning Winter Steelhead

There is strong evidence that the migration timing of wild winter steelhead has changed significantly from its historic norm (McMillan et al. 2022). Migration and run timing are strongly heritable in salmonids such as steelhead (Carlson and Seamons 2008; Abadıa-Cardoso et al. 2013) and differences in migration and spawn timing represent adaptive responses that maximize their reproductive fitness to local environmental conditions (Brannon et al. 2004). However, there is concern that indirect and direct hatchery effects, combined with habitat degradation, have selected against the early entering component (November – January) of the wild winter steelhead (Bahls 2001; McMillan 2006). Fisheries can induce directional selection in the timing of migration in salmon (Quinn et al. 2007), and in turn, shorten breeding seasons, reduce phenotypic diversity, and lower population productivity (Tillotson and Quinn 2018).

Recently, McMillan et al. (2022) published a study that evaluated potential changes in run timing of wild winter steelhead in the Quillayute, Hoh, and Queets Rivers. The author's comparison of tribal CPUE data for wild winter steelhead between circa 1948-1960 (historical period) and 1980-2017 (contemporary period) revealed large changes in run timing (Figure 15, McMillan et al. 2022). Overall, wild winter steelhead now enter later, and as a result, their overall period of entry is compressed compared to the historical period (Figure 15, McMillan et al. 2022).

Reconstruction of run timing provided evidence that wild winter steelhead runs began significantly earlier in all three populations during the historical period (Figure 15). For example, the percentage of wild winter steelhead migrating before January 1st (*pJan1*) during the contemporary period was between 18% (14%-23%) and 43% (35%-50%) less than the historical period in the Quillayute and Hoh Rivers, respectively (McMillan et al. 2022). The reduction of earlier returning wild winter steelhead also corresponds with a significant shift in the date at which half of the population had passed (*q50*) in the Quillayute and Hoh Rivers (Figure 15b and

16d). Estimated *q50* occurred 25 (16-33 days) and 61 (47-71 days) days later during the contemporary period in the Quillayute and Hoh Rivers, respectively. The magnitude of the shift in *q50* in the Queets River was similar to other populations, occurring 54 (-23-70 days) days later during the contemporary period. Importantly, the timing of contemporary hatchery steelhead returns directly overlaps with the historical early returning wild fish, which are now greatly depleted (Figure 15).

The later entry timing during the contemporary period is causing run compression (Figure 15). As a result, the number of days elapsed between when 25% and 75% of the run had passed historically is now 16 (6-29 days), 26 (11-53 days), and 22 days (-4-83 days) shorter in the Quillayute, Hoh, and Queets Rivers, respectively (McMillan et al. 2022).

McMillan et al. (2022) could not explicitly determine cause and effect, but they did offer two hypotheses. First, spawning and rearing habitat used by early entering individuals may have been sufficiently degraded to reduce the productivity of those fish (discussed in McMillan et al. 2022). Second, declines in earlier migrating wild winter steelhead are also associated with the introduction of hatchery winter steelhead (Cederholm and TU 1984; Bahls 2001).

Most hatchery programs on the western side of the Olympic Peninsula selectively bred adults to return from November through early January, under the assumption that wild steelhead were not abundant during that period (Crawford 1979; Cram et al. 2018). The assumption was erroneous, however, because the return of the hatchery steelhead directly overlaps with wild run timing in the historical period (Figure 15). This would have created opportunities for interbreeding for wild steelhead that also spawned early. For example, Seamons et al. (2012) found that a segregated winter steelhead hatchery program using the early returning Chambers Creek stock failed to prevent interbreeding with wild winter steelhead in Forks Creek, Washington. After only three generations of hatchery stocking, the proportion of wild ancestry smolts and adults declined by 10-20% and up to 80% of naturally produced winter steelhead were hatchery x wild hybrids (Seamons et al. 2012).

Further, the production and timing of hatchery winter steelhead set the stage for mixed-stock in-river fisheries in which recreational and commercial fisheries targeting hatchery winter steelhead also subjected co-mingling earlier-returning wild winter steelhead to high and potentially unsustainable exploitation rates (Cederholm and TU 1984; Naish et al. 2007; Cram et al. 2018). Contemporary fishing effort is highest from December through mid-January, corresponding with the timing of hatchery winter steelhead returns and thereby providing the potential for fisheries-induced directional selection against earlier migrating wild winter steelhead (Quinn et al. 2007; Tillotson and Quinn 2018). Given the heritable component underlying timing of migration, its plausible that reduced survival due to interbreeding with hatchery fish combined with a variable fishing intensity that peaks from late-November through early-January are the two most important factors responsible for the declines in early returning wild steelhead. Regardless of the exact causes, unless action is taken to restore and protect diversity, the fate of wild Olympic Peninsula steelhead rests on late-returning winter steelhead that may not be able to keep pace with changes in spring and summer streamflow and temperatures.

XIII. SPATIAL STRUCTURE

Ensuring that populations are well represented across diverse habitats helps to maintain and enhance genetic and life history variability and population resilience (McElhany et al. 2000). Additionally, ensuring wide geographic distribution across diverse climate and geographic regions helps to minimize risk from catastrophes (e.g., droughts, floods, etc.; McElhany et al. 2000).

The Steelhead at Risk Report authors evaluated spatial structure of steelhead in the Olympic Peninsula DPS using only previously documented or historical classifications (Cram et al. 2018). The region is fortunate in that no habitat has been lost to large dams or barriers (Cram et al. 2018). However, much of the landscape is covered with a dendritic network of roads that were constructed for forest practices, many of which cross salmon-bearing streams and have culverts (or lack culverts) that prevent upstream access (Smith 2000). We could not determine how much habitat has been blocked by culverts and roads, but it is likely extensive (Smith 2000). Considering that juvenile steelhead are distributed up to barriers in many small creeks (McMillan and Starr 2008; McMillan et al. 2013), it is likely that numerous roads and impassable culverts have truncated their spatial distribution.

It is also possible that distribution has been altered due to changes in run timing (McMillan et al. 2022) due to spatial structure of spawning adults (McMillan et al. 2007). Early returning steelhead often spawn in smaller streams or stream sections within a watershed (Cederholm 1983; McMillan et al. 2007), perhaps because they are only accessible during higher flows of early winter and such places may offer more secure spawning habitat during the high flows of winter. Reduced abundance of those fish could alter the overall distribution of steelhead. Similarly, the depletion of summer steelhead has potentially changed how and when different life histories use parts of the watershed. While any changes in spatial distribution due to the aforementioned effects are speculative, the lack of data and the importance of spatial structure underscore the uncertainty associated with steelhead in this DPS.

Last, spatial structure of a DPS is also related to the patterns and frequency of natural demographic exchange between apparently demographically independent populations. This affects the extent to which the DPS and its component populations represent or exhibit characteristics of metapopulations and/or exhibit source-sink dynamics and over what temporal periods. This requires, among other things, knowledge of the genetic relationships between populations, which appears to be largely lacking on Olympic Peninsula steelhead.

PART TWO

ENDANGERED SPECIES ACT LISTING FACTORS

I. PRESENT OR THREATENED DESTRUCTION, MODIFICATION, OR CURTAILMENT OF ITS HABITAT OR RANGE

Olympic Peninsula steelhead habitat is threatened both inside and outside of Olympic National Park (Klinger et al. 2008). Outside the park, steelhead habitat is degraded from decades of destructive logging operations (McHenry et al. 1996). Timber harvesting has altered hydrographs, increased water temperatures, and accelerated erosion. Extensive road building is causing erosion and delivering fine sediments to streams. A widespread distribution of culverts block access to spawning and rearing habitat and impede the downstream delivery of wood and sediment (Smith 2000). These impacts continue to limit steelhead productivity despite improved forest practices that were adopted in the 1990s.

> "*** [T]he majority of streams on the Olympic Peninsula will not recover for well over a century or possibly longer. The condition and land-use history on the Olympic Peninsula are representative of the Pacific Northwest. The implications for salmonid habitat, population, and resource value are not encouraging. Degraded habitats will continue to produce at levels below their full capacity, while other habitats will decline in quality."

(McHenry et al. 1996). Climate change will only worsen these impacts on steelhead and their habitat (Wade et al. 2013; Frissell et al. 2014).

Inside Olympic National Park, climate change is more than a threat – it is reality. Glaciers that provide cold summer flows are shrinking and some have disappeared. By 2070, they may all be gone (Fountain et al. 2022) and late spring snowpack is expected to decrease by 40-60% (see Figure 17). Summer streamflows are decreasing, and peak winter streamflows are increasing. Most models project drier summers, which will decrease summer and early fall streamflows, especially in rain-dominant basins.

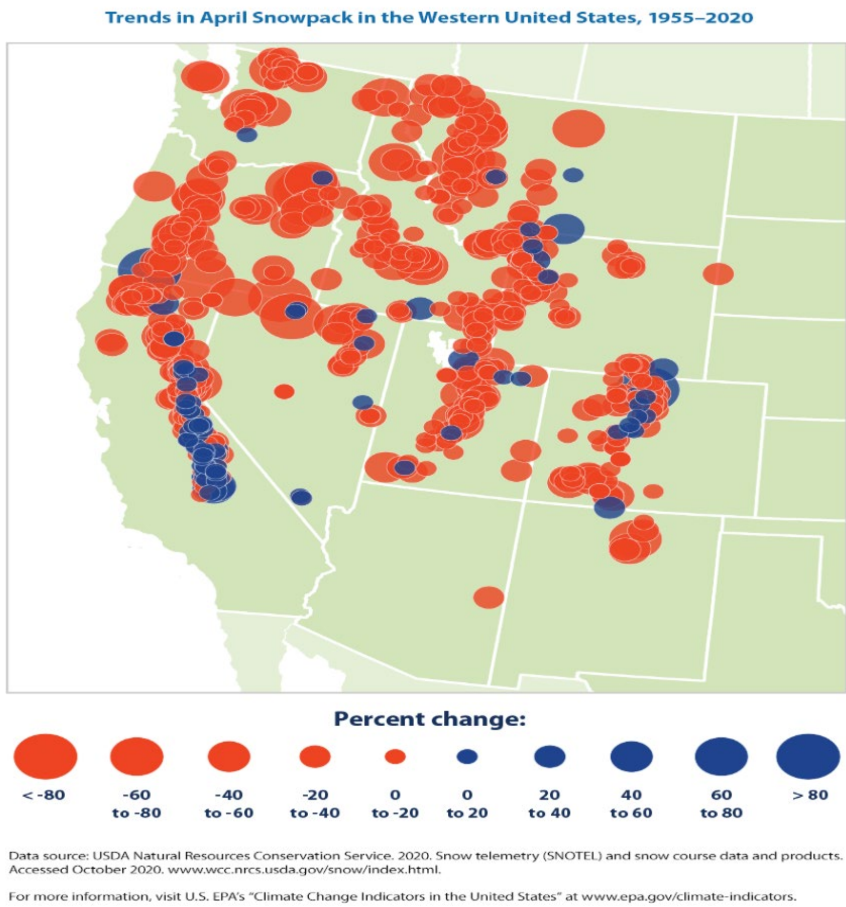


Figure 17. Trends in April Snowpack in the Western United States, 1955–2020, from: https://www.epa.gov/climate-indicators/climate-change-indicators-snowpack.

Summer steelhead, which are nearly extinct, are particularly vulnerable to these habitat changes (e.g., low summer flows during adult migration). Winter steelhead, which are declining, are also vulnerable to these changes. These impacts will increase throughout the century.

A. Overview of Habitat Conditions on the Olympic Peninsula

1. WRIA 20 Habitat

Outside of Olympic National Park, steelhead habitat in WRIA 20 suffers from extensive habitat degradation (Smith 2005, Klinger et al. 2008). A 2005 study conducted for the Washington State Conservation Commission rated salmonid habitat in WRIA 20 as "poor-fair" (Smith 2005). The study rated the following habitat attributes as poor: side-channel floodplain, sediment quantity, sediment quality, bank/streambed/channel stability, instream large woody debris ("LWD"), riparian, and water temperature (Smith 2005). The following attributes were rated as fair: road density and hydro maturity high flows (Smith 2005). The only attribute rated as "good" was pool habitat (Smith 2005).

All rivers in WRIA 20 where Olympic Peninsula steelhead occur have degraded habitat, including but not limited to the following examples:

- The **Sol Duc River** has excessive sedimentation from landslides and high road densities, poor LWD recruitment and riparian conditions, loss of wetland and off-channel habitat, low and warm summer streamflows, loss of fog drip, blockages in tributaries, and loss of cover and winter refuge habitat provided by debris jams (Smith 2000).

- The **Bogachiel River** has poor riparian and LWD conditions, an aggraded mainstem that worsens as the river moves downstream, collapsing banks in the lower mainstem, fines from exposed clay layers, and warm summer water temperatures (Smith 2000).

- The **Calawah River** experiences extensive landslides, high road densities, historic fire and subsequent salvage logging impacts, excessive sedimentation, poor levels of LWD, incised floodplains in the North Fork Calawah and South Fork Calawah Rivers (as well as several tributaries), and low and warm summer streamflows (Smith 2000).

- The **Hoh River** suffers from significant habitat degradation outside of Olympic National Park. Debris flows in the basin have been “common and devastating, resulting in scoured, incised channels with few spawning gravels and LWD” (Smith 2000). There are numerous areas with poor LWD and riparian conditions, passage blockages (in tributaries), degraded water quality, few floodplain complexes, and fog drip loss due to large conifer removal (Smith 2000). The Hoh River is also experiencing higher magnitude flood events and lower summer flows (Piety et al. 2004; East et al. 2017; NIFC 2020)

- The **Dickey River** experiences sedimentation from roads and logging operations. The Dickey River has poor riparian habitat, extensive substrate embeddedness, low and warm summer flow, increased distribution of predators (e.g., northern pikeminnow), passage blockages, reduced fog drip due to the removal of old conifers, altered wetlands, decreasing LWD, and degraded channel and floodplain conditions in several tributaries (Smith 2000). Low streamflows and high water temperatures are believed to limit steelhead production in the river (Klinger et al. 2008).

- The **Ozette River** Basin has low levels of LWD, poor riparian conditions, warm water temperatures, poor hydrologic maturity, invasive plant species, and a lack of marine-derived nutrients (Smith 2000). The Ozette River basin has particularly high road densities, which contribute to high sedimentation rates, a major limiting factor for the area (Smith 2000; Klinger et al. 2008).

- The **Wa’atch** and **Tsoo-yes Rivers** suffer from numerous blockages throughout their basins, lack LWD, and experience substantial sedimentation, mass wasting, high water temperatures, and lack marine-derived nutrients (Smith 2000; Klinger et al. 2008).

- **Goodman** and **Mosquito Creeks** suffer from sedimentation, poor riparian conditions, and, in the middle reach of Goodman Creek, low levels of LWD (Smith 2000).

2. WRIA 21 Habitat

Outside of Olympic National Park, steelhead habitat in WRIA 21 is degraded (QINLE 2011). Throughout most of the 20th century, the forest outside of the park was widely harvested for timber, causing significant effects on the quantity and quality of salmon habitat (Cederholm and Salo 1979; Cederholm et al. 1978; Smith and Caldwell 2001; McMillan 2006). There is excessive sedimentation in the more intensely logged areas, especially in the Clearwater subbasin (Cederholm et al. 1980; Klinger et al. 2008). Generally, water temperatures and side-channel floodplain habitat are in poor condition (Smith 2005; Klinger et al. 2008). Instream LWD, pool habitat, and riparian habitat are in fair condition (Smith 2005). Off-channel habitat is limited in the Clearwater, Sams, and Salmon Rivers and Matheny Creek (Klinger et al. 2008). A glacier that fed the Quinault River melted away in 2011, contributing to lower summer flows (Ahearn 2015). On the Queets River, peak and low flows are intensifying (QINLE 2011).

3. WRIA 19 Habitat

Steelhead habitat in WRIA 19 is degraded from decades of poor logging practices (McHenry et al. 1996). Approximately 95% of the watershed's old growth forest has been converted into tree farms, and landslides triggered by logging operations have been common (McHenry et al. 1996). For example, between 1950 and 1995, at least 330 logging-related landslides occurred in the Hoko River basin alone (McHenry et al. 1996). Mass wasting events have also occurred in the Clallam, Sekiu, and Pysht River basins (McHenry et al. 1996).

The basins in WRIA 19 suffer from floodplain development and alterations, loss of LWD, estuary and nearshore alterations at stream mouths, degraded water quality, warm summer stream temperatures, passage barriers, and conversion of riparian forests to non-forest uses (McHenry et al. 1996; NOPLE 2015). Smith (2005) rated the following habitat attributes as being in "poor" condition: side-channel floodplain habitat, sediment quantity, sediment quality, bank/streambed/channel stability, instream LWD, riparian habitat, and hydro maturity high flows. Water temperature varies from "poor" to "good" (McHenry et al. 1996, Smith 2005).

B. FOREST PRACTICES

Forestry has been, and continues to be, the primary land use outside of the park (McHenry et al. 1998; Smith and Cadwell 2001; Copass 2016). Despite improvements to logging practices on federal, state, and private land, the habitat of Olympic Peninsula steelhead continues to suffer the consequences of past and ongoing logging operations throughout WRIA 19, 20, and 21 (McHenry et al. 1998; Smith 2000; NIFC 2020).

1. Timber Harvest on Federal and State Lands

a. Olympic National Forest

The Olympic National Forest spans 628,115 acres and surrounds most of Olympic National Park. Logging in Olympic National Forest began in the 1920s and grew more intense in the latter half of the 20th century (Halofsky et al. 2011). Extensive road building occurred during the 1950s through the 1980s (Halofksy et al. 2011). Today, there are approximately 3,500 km of roads in the Olympic National Forest, which is roughly the driving distance between Forks, Washington and Milwaukee, Wisconsin (Halosfky et al. 2011). Between the 1960s and 1990s, approximately half of the forest's suitable land base was harvested (Halofksy et al. 2011). Prior to the adoption of the Northwest Forest Plan in 1994, timber operations involved harvesting riparian trees, clearcutting, and broadcast burning (Halofksy et al. 2011). Stream clearing and splash damming removed large wood from numerous stream channels (Halofksy et al. 2011).

The adoption of the 1994 Northwest Forest Plan ushered in a new era of management, which incorporates ecosystem management principles and specific protections for aquatic ecosystems (Halofsky et al. 2011). However, despite improved forest practices, the effects of logging continue to impair steelhead habitat in the Olympic National Forest. Logging roads and associated channel crossings are still major issues for fish habitat quality (Halofsky et al. 2011). The USFWS describes the sediment input from roads in the Olympic National Forest as "chronic" (USFWS 2020). Although conifers have regenerated along some waterways, "many riparian corridors have few conifers to provide large wood to streams" (Halofksy et. al 2011).

b. State Lands

The State of Washington began to overhaul its forest practice rules in the 1990s to benefit salmonids and other species such as the marbled murrelet (*Brachyramphus marmoratus*) and northern spotted owl (*Strix occidentalis caurina*). Prior to 1998, Washington's forestry rules lacked certain measures to protect salmonid habitat such as requiring timber harvesters to leave trees in riparian zones for LWD recruitment (McHenry et al. 1998). However, even with these improvements, the benefits to salmonid habitat are not clear.

> "The effectiveness of currently implemented forest practices for minimizing impacts remains uncertain. For example, incorrectly applied or inadequately designed riparian management zones and incorrect stream typing classifications are known problems that impair habitat protection strategies (Hansen 2001). These practices result in loss of large woody material, fish passage impacts, altered hydrology, water quality impacts, mass wasting (landslides), and elevated stream temperatures (Naiman et al. 1998)."

(Cram et al. 2018).

The Olympic Experimental State Forest

The Olympic Experimental State Forest (OESF) was designated in 1992 and covers approximately 270,000 acres of land (WDNR 2016). The OESF includes parcels of state-owned trust land interspersed with private, federal, and tribal lands (Minkova et al. 2021). WDNR's goal is to manage the state trust lands to generate sustainable revenue for counties, universities, and other trust beneficiaries, while also maintaining ecological values (WNDR 2016). Nearly half of the forest is young, with trees ranging between 20 and 39 years of age (WDNR 2016).

Several major rivers occur in the OESF, including the Hoh, Queets, Quillayute, Clearwater, Hoko, and Pysht Rivers (Minkova et al. 2021). WDNR does not manage streams in the OESF to meet desired future conditions for salmonids (WDNR 2018). Instead, it seeks to maintain or aid the restoration of riparian functions, water quantity, and water quality (WDNR 2018).

It is unclear whether the forest practices in OESF are significantly improving habitat for steelhead. For example, Pollock et al. (2004) found that "within the OESF, a majority of streams do not meet WDOE water quality standards for temperature, and that stream temperatures in harvested basins are often (but not always) higher and more variable than stream temperatures in unharvested basins" (Pollock et al. 2004). Additional study by Pollock et al. (2009) found that the impact of harvest activities could not be fully mitigated by riparian buffers alone. However, two studies by WDNR did not find similar water temperature exceedances within OESF (Martens et al. 2019; Devine et al. 2022).

2. Logging Effects on Steelhead Habitat

Logging operations harm steelhead habitat by increasing water temperatures and sedimentation in streams, removing large woody debris, reducing streamflows, and decreasing habitat connectivity.

a. Temperature

Logging operations increase water temperatures in several ways: by removing upland and riparian vegetation, which increases thermal radiation; decreasing groundwater inflow; increasing sedimentation rates, which widen and shallowing river channels; and reducing large woody debris recruitment (Cederholm and Salo 1979; Lynch et al. 1984; Frissell et al. 2014). High water temperatures harm salmonids by causing direct mortality and metabolic distress, altering migration and breeding behavior, increasing susceptibility to disease, and creating thermal barriers that block access to habitat (Hicks 1999).

Several studies indicate that logging operations on the Olympic Peninsula increase stream temperatures. Hatten and Conrad (1995) studied managed (i.e., logged) and unmanaged stream reaches on tributaries to the Hoh, Queets, and Bogachiel Rivers and

Kalaloch Creek. They found that unharvested sites exceeded the state's 16° C temperature standard at an average of 1.8 days over a 39-day monitoring period (Hatten and Conrad 1995). In contrast, the managed sites exceeded 16° C an average of 18.3 days during the same period (Hatten and Conrad 1995). Because the harvested sites were representative of low elevation harvesting sites, Hatten and Conrad (1995) assumed that most managed stream segments in the western Olympic Peninsula did not meet water quality standards for temperature. Those standards are intended to protect salmonids.

Although riparian buffer standards have been implemented on federal and state forestland, research conducted by Hatten and Conrad (1995) and Pollack et al. (2009) indicate that buffers alone may not be sufficient to improve stream temperatures. For example, Hatten and Conrad (1995) found that the most important variable affecting stream temperature is the total amount of land logged in a basin. However, Devine et al. (2022) did not find similar water temperature exceedances in OESF as Pollock et al (2009), which may indicate that increased shade in riparian zones is benefiting stream temperature.

Logging has had significant impacts in WRIA 19 basins, which have mostly been converted into tree farms (McHenry et al. 1996). For example, water temperatures as high as 69° F and 73.4° F have been recorded in the Hoko River (McHenry et al. 1996).

b. Hydrologic Flows

Timber harvest affects streamflow by increasing permeable surface areas, transporting water directly to streams, removing vegetation that stores water, and altering snow retention and melt rates (Davis and Schroeder 2009, as cited in WCSSP 2013). These impacts change the timing and magnitude of peak and low streamflows (Davis and Schroeder 2009, as cited in WCSSP 2013). Peak flows can scour eggs and flush juveniles out of rearing habitat (Davis and Schroeder 2009, as cited in WCSSP 2013). Low flows can alter flow-related migration cues, disconnect habit, and increase crowding, competition, and vulnerability to predation (Davis and Schroeder 2009, as cited in WCSSP 2013).

A study on the Hoh River showed that logging significantly impacts peak and mean daily streamflows at watershed, subbasin, and basin level in descending order (Achet 1997).

> "If about 27% of the basin was harvested (extracting 46. 9 million m^3 of timber) the impact on peak would increase by 44 to 51% at basin, 34 to 41% at sub-basin and 55 to 141% at watershed level in medium and high hydrologic conditions. By reducing the timber harvest by 26%, the corresponding reductions in peak at the basin level, sub-basin level and watershed level were 9 to 10%, 7 to 9% and 16 to 18% respectively. Regarding mean daily flow, harvesting 27% of the basin would result in increase in mean daily flow by 30 to 47% at the basin level, 22 to 36 % at the at the sub-basin level and 77 to 137% at the watershed level in medium and high hydrologic conditions. When the timber harvest was 34.6 million m3 the

corresponding reduction in impact on mean daily flow at basin, sub-basin and watershed level would be 7 to 10%, 5 to 8% and 14 to 20%" (Achet 1997).

c. Sediment

Logging operations increase sedimentation through debris flows and surface runoff from logging roads, which expand channel widths and reduce pool depth (Furniss et al. 1991; Jones et al. 2000). Increased sedimentation affects salmonids throughout their freshwater life stages by suffocating incubating eggs and fry, reducing macroinvertebrate composition/prey, causing respiratory failure, reducing or blocking access to habitat, and warming stream temperatures by lowering water depth (McHenry et al. 1996, USFWS 2020).

Sedimentation from timber harvest significantly impairs aquatic habitat on the Olympic Peninsula. Logging operations have caused landslides throughout the area. The most severe damage has occurred on the Quinault River, Queets River, Clearwater River, Hoh River, Calawah River, Sol Duc River, Hoko River, Seiku River, Sooes River, and Deep Creek basins (McHenry et al. 1996; Cederholm et al. 1980; WSCC 2001; USFWS 2020).

Landslides have been particularly destructive in the Hoh River basin (Logan et al. 1991). For example, in the winter of 1988-1989, 105 landslides occurred on Huelsdonk Ridge above the Hoh River, mobilizing an estimated 83,000 cubic yards of sediment that deposited in the South Fork and mainstem Hoh River (Logan et al. 1991). Only two of the 105 landslides occurred in entirely old growth stands (Logan et al. 1991).

Sedimentation from logging roads impacts all watersheds where Olympic Peninsula steelhead occur. The Hoh River basin suffers from excessive sedimentation caused by logging roads (USFWS 2020). The sub-basins with the largest percentage of sediment over background levels include the South Fork of the Hoh River (156%), Owl Creek (286%) and Winfield Creek (210%) (Marsh 2012).

d. Large Woody Debris

Logging removes wood from riparian, upland, and headwater riparian areas that could be recruited to streams (Klinger et al. 2008). The loss of wood increases the magnitude of peak flow events, gravel scour, and bank erosion. It also decreases channel sinuosity and stability as well as the quantity of pools and off-channel features (East et al. 2017). As a result, steelhead in streams with low levels of LWD have less cover, food, off-channel habitat, and pools for rearing and refuge. Depletion of LWD has long-term consequences for stream productivity.

Outside of Olympic National Park, nearly all rivers and streams suffer from the loss of LWD (Smith 2000; Piety et al. 2004). Based on GIS imagery, East et al. (2017) found that the Quinault River has less LWD than the Hoh and Queets Rivers. The Quinault River had 1.9% LWD cover by area compared to 2.5% on the Queets River and 2.7% on the Hoh River (East et al. 2017).

e. Habitat Connectivity

Logging roads and culverts decrease access to on-channel and off-channel habitat (Davis and Schroeder 2009, as cited in WCSSP 2013). As a result, there is diminished spawning, rearing, and foraging habitat; increased competition caused by crowding; and less access to refuges during high flow events (Davis and Schroeder 2009, as cited in WCSSP 2013). Undersized culverts also compromise habitat quality for varying downstream distances due to their disruption of the downstream transport of water, wood, and sediment.

f. Roads

Outside of Olympic National Park, most areas in WRIAs 19-21 are not properly functioning for salmonid habitat due to road density (NIFC 2020). According to NMFS, watersheds with road densities greater than three miles per square mile of watershed are not properly functioning for salmonid habitat (NMFS 1996). Based on this criterion, nearly all forestland outside of the park is not properly functioning for salmon habitat (NIFC 2020). The West Fork Dickey River watershed has the highest road density (5.5 miles per square mile) (NIFC 2020). Other high-density watersheds include Elk Creek-Calawah, Crooked Creek, the East Fork Dickey, the Lower Bogachiel River and Big River watersheds (NIFC 2020).

Roads threaten several of the larger watersheds as well. For example, road density remains high in the Queets River watershed on both federal and private land (USFWS 2020). Road density in the Clearwater drainage is 3.2 miles per square mile in the upper section and 3.7 miles per square mile in the lower section (USFWS 2020). There are approximately 2.5 miles of road per square mile in the Queets River floodplain outside of the Olympic National Forest (USFWS 2020).

Roads cause excessive sedimentation on the lower Hoh River (USFWS 2020). Private landowners and the Washington Department of Natural Resources have extensively logged the Hoh River watershed and logging roads cause excessive sedimentation (USFW 2020).

Despite lower road densities, the Quinault River is also impacted by roads (USFWS 2020). Logging roads built during the early 1900s spread throughout the basin and streambank hardening is degrading habitat conditions (USFWS 2020).

Roads have also triggered mass wasting events. For example, the USFS (2012) found that 49% of mass wasting events in the Sitkum and South Fork Calawah watershed were caused by roads.

C. CLIMATE CHANGE

The evidence of climate change is irrefutable. According to the IPCC, "[i]t is unequivocal that human influence has warmed the atmosphere, ocean and land" and that "[w]idespread and rapid changes in the atmosphere, ocean, cryosphere and biosphere have occurred" (IPCC

2021). Each of the last four decades have been successively warmer than the last (IPCC 2021). Under status quo emissions levels, average global temperatures are expected to rise 1.6° C above 1850-1900 levels between 2021 and 2041. Under the very high GHG emissions scenario (SSP5-8.5), global temperature will likely increase by 2°C between 2041 and 2060 (IPCC 2021).

Climate change will affect the climate and hydrology of the Pacific Northwest. The region is projected to lose snowpack and glacier mass and incur frequent and extreme hydrological conditions (Mantua et al. 2010; Halofsky et al. 2011). Most models indicate that winters will be wetter and summers will be warmer and drier (Mote and Salathé 2010; Elsner et al. 2010; Kunkel et al. 2013). Summer droughts will be more frequent and severe (CIG 2009; Mantua et al. 2010; Halofsky et. al 2011).

The western Olympic Peninsula is anticipated to warm, although it may warm slightly less than other areas of the Pacific Northwest due to the moderating effect of the Pacific Ocean (Halofsky et al. 2011; Dalton et al. 2016). Annual precipitation is projected to increase during the winter and spring and decrease during the summer (Dalton et al 2016). Snowpack is projected to decline and streamflows are projected to increase in the winter and decrease in the summer (Dalton et al. 2016). The Olympic Peninsula is also expected to experience sea level rise, stronger and more frequent storms, increased erosion, warmer water temperatures, more low and high flow events, and increased glacial melt (Miller et al. 2013).

Climate change will also significantly alter the marine environment. Under the RCP8.5 scenario, models indicate that more multi-year warming events will occur in the northeast Pacific Ocean (marine heatwaves, MHW), which may cause profound effects on salmonid habitat (Joh and Di Lorenzo 2017; Oliver et al. 2019; Cheung and Frolicher 2020).

> "Widespread and extreme negative impacts on marine life and fisheries associated with the 2014-2015 marine heat wave are well documented. If the projected increases in the area, magnitude and frequency of extreme warm events are realized, and they are superimposed upon a systematic anthropogenic warming trend; this combination would likely cause profound negative impacts on marine life and fisheries all along the west coast of North America, particularly those in the Gulf of Alaska in the second half the 21st century."

(Joh and Di Lorenzo 2017).

These changes will impact Olympic Peninsula steelhead, especially the summer-run component, which is nearly extinct. Climate change is expected to cause widespread declines in the quantity and quality of habitat for Olympic Peninsula steelhead (Halofsky et al. 2011). It is unknown whether these fish will adapt quickly enough to these changes.

> "It remains an open question whether present-day salmonid fish populations on the Olympic Peninsula can adapt (either through phenological, phenotypic, or

evolutionary responses) at rates required to deal with the combination of anthropogenic climate change and other habitat and ecosystem changes that will come in the next century (Crozier et al. 2008)." (Halfosky et al. 2011).

1. Projected Climate and Hydrological Changes

a. Air Temperature

The Pacific Northwest has not experienced the same magnitude of warming since the period between glacial and interglacial periods (Dalton et al. 2016). Between 1900 and 2014, the average air temperature in Washington rose by approximately 1.5°F (USFWS 2020; Mote and Salathe 2010). This warming trend will continue.

> "Regardless of the scenario, warming is projected to continue throughout the 21st century in the Pacific Northwest. For the 2050s (2041 to 2070) relative to 1950-1999, temperature is projected to rise +5.8°F (range: +3.1 to +8.5°F) for a high greenhouse gas scenario (RCP8.5). Much greater warming is possible after mid-century under the more aggressive scenarios (RCP 6.0 and 8.5) ***."[1]

Models using less aggressive emission scenarios also project increased warming over the course of the 21st Century. For example, Elsner et al. (2010) used data from the IPCC Fourth Assessment Report to project that air temperatures will increase 0.3 C per decade in the Pacific Northwest. However, even this lower rate of warming could produce "profound changes in the hydrology and environment of the Northwest" (Mote and Salathe 2010).

Modeling shows that summer air temperatures in the Pacific Northwest will rise significantly. For example, multi-model averages of the A1B scenario indicate that average June through August air temperatures will increase at the following rates over the century: 1.7° C (0.43° C to 3.4° C) by the 2020s; 2.7° C (1.3° C to 5.1° C) by the 2040s; and 4.7° C (2.7° C to 8.1° C) by the 2080s (Mantua et al. 2010). Multi-model averages of the B1 scenario project June through August temperature to increase as follows: 1.2° C (0.18° C to 2.4° C) by the 2020s; 1.8° C (0.2° C to 3.7° C) by the 2040s; and 2.9° C (1.3° C to 5.1° C) by the 2080s (Mantua et al. 2010).

Air temperatures on the Olympic Peninsula are projected to increase (USFWS 2020). Under the business-as-usual emissions scenario, air temperatures in the Olympic National Forest are projected to increase by 2° F to 5° F (approximately 1° C to 2.5° C) between 2016 and 2045 relative to late 20th Century temperatures (USFWS 2020). Air temperatures in Olympic National Park are projected to increase by 2.93° F between 2016 and 2045 and by 5.85° F between 2046 to 2075 (LCD 2022).

[1] https://cig.uw.edu/learn/climate-change/

b. Glaciers & Snowpack

Climate change is melting the Olympic Peninsula's glaciers (USFWS 2020) and reducing snowpack (Figure 17). Between 1980 and 2015, glaciers on the Olympic Peninsula decreased by 34% (Riedel et al. 2015). During that period, thirty-five glaciers and 16 perennial snowfields disappeared (Fountain et al. 2022), including the Anderson Glacier, which contributed to streamflow to the Quinault River (Dalton et al. 2016; NIFC 2020). Other glaciers that provide water to the Quinault, Queets, Hoh, and Bogachiel Rivers (McHenry et al. 1996) are receding at rates higher than previously recorded (NIFC 2020). Between 1981 and 2015, the glaciers that feed into the Hoh River decreased by 40% (NIFC 2020). Under the RCP 8.5 scenario, Fountain et al. (2022) estimates that glaciers will largely disappear from the Olympic Peninsula by 2070.

Climate change is expected to reduce late spring snowpack on the Olympic Peninsula (Halofsky et al. 2011). Parts of the Olympic National Forest historically maintained snowpack until April 1st (USFWS 2020). By mid-century, most spring snowpack will only exist at high elevations in Olympic National Park, which will also experience "dramatic reductions" in spring snowpack (Elsner et al. 2010; USFWS 2020).

c. Precipitation

The majority of climate models project that spring and summer rain will decrease on the Olympic Peninsula, with up to a 24% decrease in summer rainfall (USFWS 2020). At least one model, however, shows slightly wetter summers (LCD 2022).

The majority of climate models project that fall and winter rain will increase on the Olympic Peninsula (USFWS 2020). Precipitation during the months of December through February is likely to increase by 4.5% to 5% on average and depending on location (Halofksy et al. 2011). Across the Pacific Northwest, spring precipitation has already increased by about 2%-5% per decade over the past century, a trend that will likely continue (Abatzoglou et al. 2014). A multi-mean model using a high emissions scenario projects a 6.5% increase in spring precipitation by 2041-2070 (Dalton et al. 2016).

The Pacific Northwest will also experience wetter storm events (Miller et al. 2013). Some models indicate that storms will be wetter and stronger on the Olympic Peninsula (Halofsky et al. 2011).

d. Streamflow

Climate change is altering the hydrology of watersheds on the Olympic Peninsula. Winter peak flows are getting higher, and summer low flows are getting lower (NIFC 2020). These conditions are predicted to become worse for the major steelhead producing watersheds in the coming decades. As illustrated in Table 8, these changes are occurring throughout the DPS and are likely to have significant effects on winter and summer steelhead. For example, as we discussed for steelhead, the lower 9km of the NF Calawah River (tributary of Calawah River in Quillayute River system) went completely dry in 2002 killing thousands of salmonids, including all adult and juvenile steelhead (McMillan et al. 2013; McMillan 2002). The Calawah River is a lower-elevation, rain-fed tributary and is susceptible to drought conditions (Smith 2000). Considering low summer flows are likely to become worse in the future (Wade et al. 2013), streams like the NF Calawah may become uninhabitable during the summer months. This also helps explain why the Calawah River has experienced a large change in summer low flows relative to other watersheds (Table 8).

Table 8: Changes in low and high streamflow in Calawah, Quinault, Hoh, and Hoko Rivers from 1976-2019. (Source: NIFC 2020)

Watershed	Low flow decrease	High flow increase
Calawah River	-48%	+5%
Quinault River	-20%	+7%
Hoh River	-15%	+18.4%
Hoko River	-13%	+17.8%

Summer streamflows will continue to decrease on the Olympic Peninsula (Mantua et al 2010; Wade et al. 2013; Beechie et al. 2013; Dalton et al. 2016). Using A1B and B1 warming scenarios, Mantua et al. (2010) found that annual summer low flows on the Olympic Peninsula will be approximately 5% to 45% lower by mid-century. According to Dalton et al. (2016), average summer flows are projected to decline by 30% across the Quillayute, Hoh, Queets, and Quinault River basins. Summer flows in headwater areas will likely become more ephemeral or stop during the summer months, and the duration of low flow periods will increase significantly in all but the most rain-dominant basins (i.e., Hoh and Queets River basins) (Halofsky et al. 2011). Increased winter flooding may exacerbate summer low flow conditions by increasing porous sediment deposits (Halofsky et al. 2011).

Winter flooding is increasing and will continue to do so throughout the 21st century (NIFC 2020, Mantua et al. 2010). Peak flows are commonly at or above flood stage on the Bogachiel and Calawah Rivers (NIFC 2020). Average winter flows are projected to increase by at least 30% over the majority of stream reaches in the Quillayute, Hoh, Queets, and Quinault River basins (Dalton et al. 2016). Winter flow in stream reaches on the Quinault and Quillayute Rivers are projected to increase by 40% (Dalton et al. 2016). Late fall and early winter flooding is also projected to increase (Halofsky et al. 2011).

It is important to note that these summer and winter streamflow projections are based on moderate (A1B) and low (B1) emission scenarios established by the IPCC. These scenarios project similar amounts of emissions through the mid-21st century as the A2 scenario, which is a high emissions scenario (Halofsky et al. 2011). The A2 scenario, however, projects higher emissions during the latter half of the century (Halofsky et al. 2011).

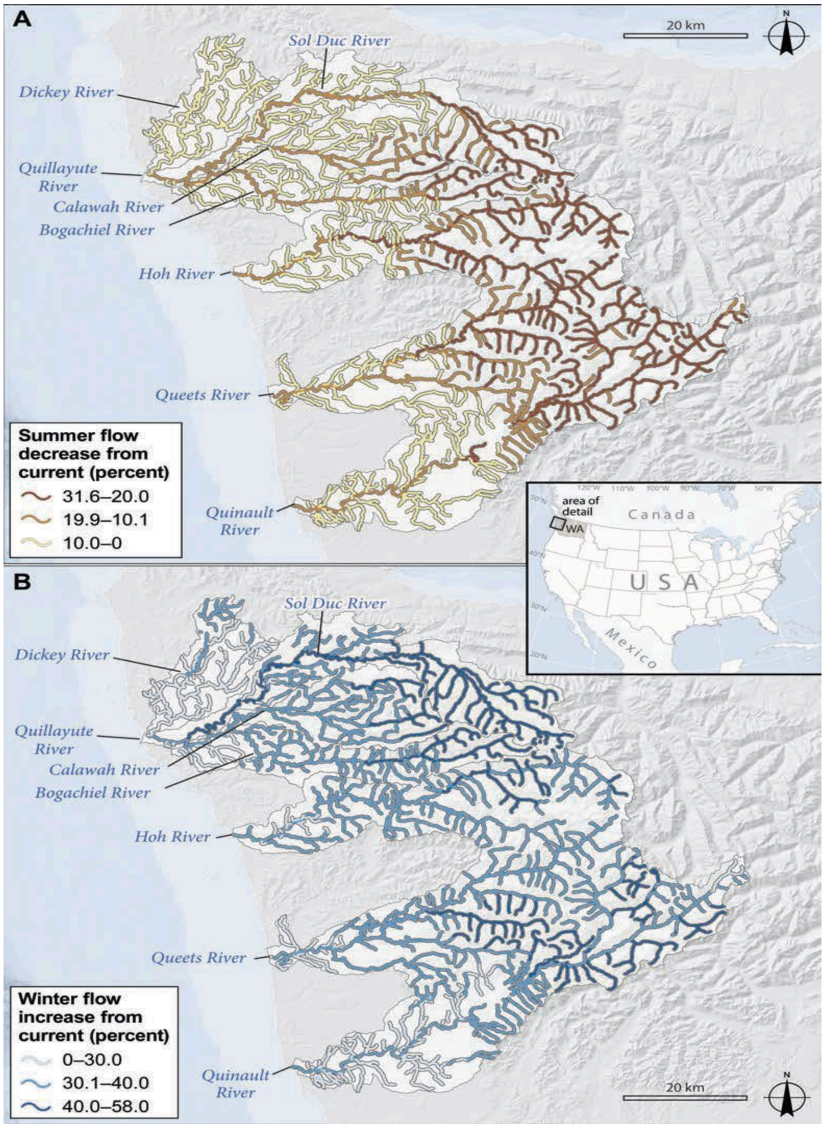


Figure 18. Current and projected (year 2040) reductions in summer streamflow and increases in winter streamflow in the Sol Duc, Dickey, Quillayute, Calawah, Bogachiel, Hoh, Queets, and Quinault Rivers (Reeves et al. 2018).

e. Water Temperature

Water temperatures on the Olympic Peninsula are projected warm. Increasing stream temperatures pose higher risks to the quality and quantity of steelhead habitat because steelhead typically however, it is unlikely that they will exceed the thermal tolerances of steelhead (USFS 2020; Miller et al. 2013; Mantua et al. 2010). Using A1B and B1 emission scenarios, Mantua et al. (2010) projected that water temperatures will warm by 1° to 2° C (

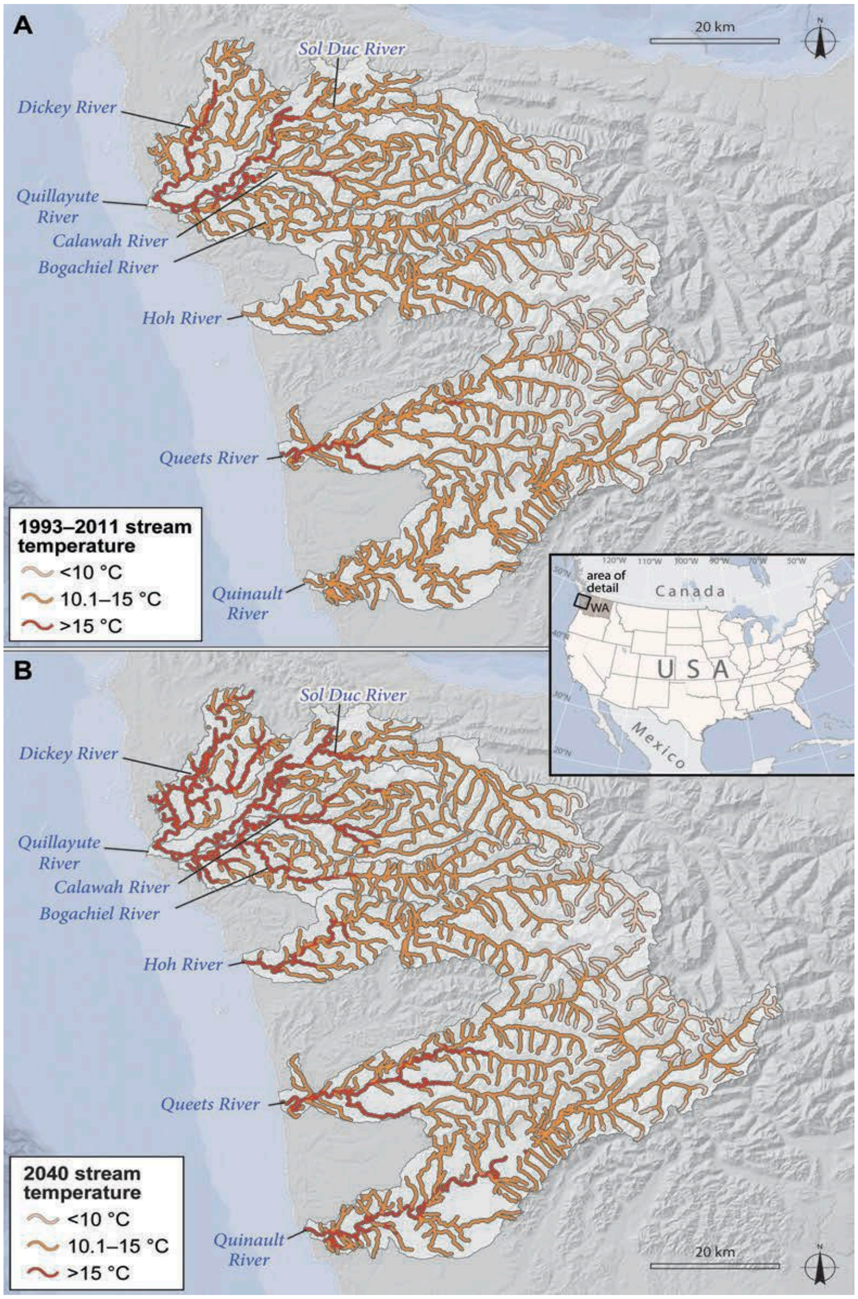


Fig. 19. Current and projected 2040 summer water temperatures in the Sol Duc, Dickey, Quillayute, Calawah, Bogachiel, Hoh, Queets, and Quinault Rivers (Reeves et al. 2018).

f. Other Water Quality Issues

Increased winter precipitation will likely increase runoff and landslides from logging operations, which would increase sediment pollution in Olympic Peninsula rivers and streams (Klinger et al. 2008; Halofsky et al. 2011).

2. Effect of Freshwater Habitat Changes on Olympic Peninsula Steelhead

Climate change will adversely modify freshwater salmonid habitat on the Olympic Peninsula. Because of their longer freshwater residency, steelhead are more sensitive to these freshwater habitat changes than other salmonids (Halofsky et al. 2011).

a. Low Summer and Early Fall Streamflow

Olympic Peninsula steelhead have high exposure to extreme low flows (Wade et al. 2013). Lower flows will decrease habitat availability, elevate water temperatures, and induce thermal stress on salmonids (Crozier and Zabel 2006; Wade et al. 2013; Dalton et al. 2016; Ohlberger et al. 2018). Reduced summer and fall streamflows in rain-dominated basins may adversely affect summer steelhead migration (Holafsky et. al. 2011). For example, there will be less cold water and fewer holding pools for migrating steelhead, which may be particularly stressful for summer steelhead and lower their reproductive success (Dalton et al. 2016). Low streamflows are already a limiting factor on the Calawah River (Phinney et al. 1975), particularly the NF Calawah (McMillan 2002), and summer streamflows are showing a decreasing trend on many Olympic Peninsula Rivers, such as the Hoh River (NIFC 2020). Climate change will exacerbate these effects (NIFC 2020).

b. High Late Fall and Winter Streamflow

Olympic Peninsula steelhead have high exposure to increased winter streamflows (Wade et al. 2013). Average winter flows (January – April) are expected to increase by at least 30% by 2040 (Dalton et al. 2016). Average winter flows are projected to increase by 31-40% in the Hoh, Queets, and Quinault Rivers (Dalton et al. 2016). Increased winter flows could reduce the survival of developing eggs, embryos, and juveniles (Dalton et al. 2016) This is a higher risk in confined streams, which are more susceptible to scour (Halofsky et al. 2011). Summer steelhead juveniles that emerge in winter in high gradient streams are at greater risk of displacement (Dalton et al. 2016). Peak flows could also reduce the availability of slow-water habitat for juveniles (Mantua et al. 2010; Halofsky et al. 2011), which may reduce parr-smolt survival rates (Halofsky et al. 2011). Increased runoff will also likely increase sedimentation in steelhead habitat (East et al. 2017).

c. Water Temperature

Based on models that use A1B and B1 scenarios, water temperatures are unlikely to exceed lethal limits for steelhead, but they would increase to levels that reduce growth and increase the risk of predation (Dalton et al. 2016). For example, temperatures at or above 15° C can inhibit the smolt transformation process (Miller et al. 2013). As explained in the water quality section, many stream reaches on the Olympic Peninsula already exceed 16° C and warmer spring stream temperatures could impede smolting and force the populations to adapt to smolt during an earlier time.

It is also possible that warmer temperatures could expand growing seasons and increase food web productivity, which would benefit steelhead (Halofsky et al. 2011). However, this benefit could be offset by increased flooding, lower summer flows, increase predation risks (Dalton et al. 2016). Additional food availability may be also offset by increased competition (Dalton et al. 2016).

3. Projected Changes in Marine Environment

Climate change is altering nearshore and offshore habitat of Olympic Peninsula steelhead (Klinger et al. 2008; Miller et al. 2013). These changes include warming sea surface temperatures and potential alterations in upwelling, hypoxia, and acidification (USFWS 2020). The scope and intensity of these impacts are uncertain (Miller et al. 2013). For example, it is possible that climate change could decrease upwelling, which would lower productivity (Klinger et al. 2008). It could also increase upwelling, although warmer ocean temperatures could limit productivity benefits (Miller et al. 2013). Increased sea surface temperatures will drive the southern boundary of steelhead northward, potentially contracting the range of the species (Abdul-Aziz et al. 2011). These changes will negatively affect salmon and steelhead survival rates (e.g., Kilduff et al. 2015).

a. Sea Surface Temperature

Sea surface temperatures are projected to increase in the Pacific Northwest (Mote and Salathe 2010; Miller et al. 2013; USFWS 2020). Mote and Salathe (2010) projected that ocean surface temperatures will increase by approximately 1.2° C by 2050 relative to the 1970-1999 average temperature. Other models project that surface temperatures may increase by 1.2° C to 3° C by mid to late century (USFWS 2020). These projected increases are substantially outside 20th century variability (Mote and Salathe 2010). Marine heatwaves such as the 2014-2015 “Blob” are also likely to reoccur more frequently (Oliver et al. 2019; USFWS 2020) as the ocean heat content increases (Cheung and Frölicher 2020; von Schuckmann et al. 2020).

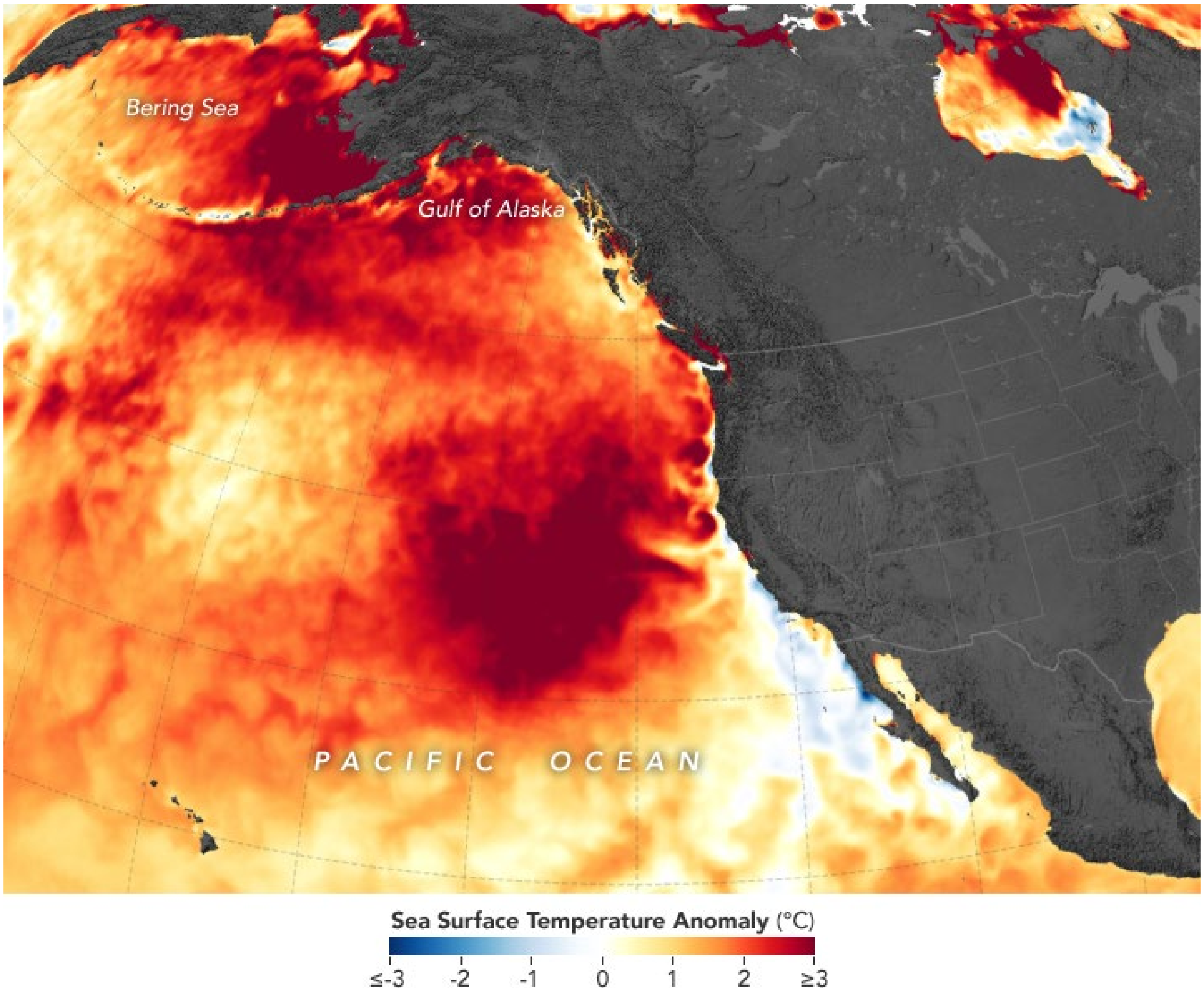


Fig. 20. Sea Surface Temperature Anomaly from August 1 - 31, 2019 (Image source: https://earthobservatory.nasa.gov/images/145602/marine-heat-wave-returns-to-the-northeast-pacific)

b. Upwelling

Climate change may affect upwelling, although there is much uncertainty about this possibility. For example, models have produced mixed results on whether upwelling favorable winds will change (Miller et al. 2013). There is some evidence indicating that upwelling patterns may become shorter and more intense (USFWS 2020). Increasing surface temperatures may influence the timing and magnitude of upwelling as well (Miller et al. 2013).

c. Acidification

The Pacific Northwest is vulnerable to ocean acidification (Miller et al. 2013). Increased acidification threatens key species in the food web, including zooplankton, pteropods, crabs, and krill (Busch et al. 2013; Mathis et al. 2015; Dalton et al. 2016).

d. Anoxic and Hypoxic Events

Anoxic and hypoxic events have occurred off the Washington coast (Klinger et al. 2018; USFWS 2020;). As sea surface temperatures warm, dissolved oxygen levels are expected to decrease (Miller et al. 2013).

4. Effect of Changed Marine Conditions on Olympic Peninsula Steelhead

NMFS anticipates that climate change will continue to limit ocean productivity for salmonids (Ford 2022).

> "Historically, ocean conditions cycled between periods of high and low productivity. However, global climate change is likely to disrupt this pattern, in general, leading to a preponderance of low productivity years, with an unknown temporal distribution (Crozier et al. 2019). Recent (2015–19) ensemble ocean indicator rankings include four of the worst seven years in the past 20, meaning that an entire salmon or steelhead generation could have been subjected to poor ocean productivity conditions." (Ford 2022).

Warming will also cause steelhead ocean habitat to progressively decrease throughout the century (Abdul-Aziz et al. 2011). Summer marine habitat is anticipated to contract 8-10% by the 2020s, 15%-19% by the 2040s, and 24%-43 by the 2080s (Abdul-Aziz et al. 2011).

Although some forage fish may benefit from climate change, forage fish abundance is likely to decrease (USFWS 2020), resulting in less prey for steelhead.

C. WATER QUALITY

Limited water quality monitoring makes it difficult to identify the status and trends of water quality on the Olympic Peninsula (Klinger et al. 2008). However, the limited data that does exist shows that there are many river and stream segments where Olympic Peninsula steelhead occur that do not meet water quality standards (Klinger et al. 2008; WDOE 2016). These water quality standards are intended to protect salmonids.

In WRIA 19, thirteen rivers and streams have segments that do not meet state temperature standards and need Total Maximum Daily Loads (TMDLs). These waterbodies include: Deep Creek, Hoko River, Little Hoko River, Clallam River, Sekiu River, Sekiu River (North Fork), Sekiu River (South Fork), Pysht River, Pysht River (South Fork), Salt Creek, Lyre River, and the West Twin River (WDOE 2016). Deep, Salt, and Bear Creeks are also on the 303(d) list for dissolved oxygen (WDOE 2016).

Dozens of rivers and stream segments in WRIA 20 do not meet water temperature standards (WDOE 2016). The following rivers and streams have at least one river segment that does not meet temperature standards and are in need of a TMDL: Hoh River, Coal Creek, Crooked Creek, Ozette River, Umbrella Creek, Willoughby Creek, Owl Creek, Split Creek, Maple Creek, Anderson Creek, Line Creek, Alder Creek, Winfield Creek, Nolan Creek, Bogachiel River, Coal Creek, Dickey River (mainstem, East Fork, Middle Fork, and West Fork) Elk Creek, Sol Duc River, SF Calawah River, Sitkum River, Sooes River, Trout Creek, Big River, Fisher Creek (McQuarry Creek), Ozette River, and Lake Creek (WDOE 2016). Monitoring by the Hoh Tribe between 2006 and 2015 revealed that all but one major salmonid tributary to the Hoh River had summer water temperatures that exceeded state water quality standards (NIFC 2020).

In addition to temperature exceedances, several rivers and tributaries in WRIA 20 have at least one stream segment that does not meet pH or dissolved oxygen standards and need TMDLs for these parameters (WDOE 2016). The following rivers and stream are on the 303(d) list for pH: Coal Creek, Crooked Creek, Ozette River, Sol Duc River, Big River, and Palmquist Creek (WDOE 2016). The following rivers and streams are on the 303(d) list for dissolved oxygen: Big River, Coal Creek, Ozette River, Siwash Creek, South Creek, Umbrella Creek, Lake Creek, and Bear Creek (WDOE 2016).

There is less water quality data for WRIA 21. Kalaloch Creek, Matheny Creek, and the Sams River are on the 303(d) list for temperature (WDOE 2016).

D. ANTHROPOGENIC MIGRATION BARRIERS

When summarizing habitat conditions for Olympic Peninsula steelhead, Busby et al. (1996) assumed that “minor blockages (such as impassable culverts) are likely throughout the region." Today, we know that at least 1,214 human-made barriers exist in WRIA 19-21, many of which are completely impassable (WDFW 2020). According to WDFW, there are 272, 470, and 473 barriers in WRIAs 19, 20, and 21, respectively (WDFW 2020). Although progress has been made on removing and repairing culverts and other human-caused barriers, there are still passage issues on many streams throughout the range of Olympic Peninsula steelhead.

The Quinault Indian Nation, Quileute Indian Nation, Hoh Tribe, Makah Nation, and other western Washington tribes won a major legal victory for salmonids that requires the State of Washington to repair or replace state-owned culverts throughout the Olympic Peninsula and elsewhere. *United States v. Washington*, 853 F.3d 946 (9th Cir. 2017) *aff* 138 S.Ct. 1832 (2018). However, the Washington State Department of Transportation has less than 12% of the funding it needs to complete these actions by the court-imposed deadline (NIFC 2020). If the state continues at its current rate, it will not complete its work until 2034 (NIFC 2020).

State and private landowners have made progress repairing culverts on state and large private forest roads under the Road Maintenance and Abandonment Plan (NIFC 2020). Although the plan was supposed to be completed in 2021, approximately 85% of the work is done in the coastal region (NIFC 2020). However, there are hundreds of non-RMAP culverts on private, county, and federal land throughout WRIAs 19, 20, and 21 (NIFC 2020). Roughly half of the non-RMAP culverts are not passable (NIFC 2020).

II. OVERUTILIZATION FOR COMMERCIAL AND RECREATIONAL PURPOSES

Wild winter steelhead populations in the Olympic Peninsula DPS have a long history of supporting intensive recreational and commercial fisheries until the last few years when fisheries were altered and eventually closed due to conservation concern over unprecedented low run sizes (WDFW blog post: https://wdfw.medium.com/changes-to-the-coastal-steelhead-season-67131dd05ba7). The chronic declines in run size combined with altered run-timing (McMillan et al. 2022), low levels of repeat spawning, and now, dramatic freshwater and marine effects linked to climate change, suggest that the four major populations of wild winter steelhead are being overutilized in fisheries. Cram et al. (2018) was able to estimate harvest rates for 33% of the populations in the Olympic Peninsula DPS up until 2013 and found a mean annual harvest rate of 25.6%, which was the highest harvest rate among all steelhead DPSs in Washington State. We updated the harvest estimates for the Quillayute, Hoh, Queets, and Quinault Rivers through 2020 (Table 1) and discuss those results for each population specifically in the Harvest Impacts section.

These harvest fisheries put Olympic Peninsula steelhead at risk. WDFW's Steelhead at Risk report did not evaluate "population-specific impacts" within the DPS because the "harvest rates are difficult to interpret without accompanying demographic analyses, since risks posed by harvest depend on a population's productivity" (Cram et al. 20198). WDFW acknowledges there are substantial data gaps regarding the productivity of Olympic Peninsula steelhead (Cram et al. 2018). Nonetheless, WDFW and the Treaty tribes manage Olympic Peninsula steelhead under a maximum sustainable harvest regime that does not include the level of detail necessary to responsibly manage harvest or maintain the persistence of the species (Gibbons et al. 1985; Burge et al. 2006). And summer steelhead are not managed or monitored but are subject to indirect harvest in fisheries.

A. Escapement Goals

The framework for the modern winter steelhead fishery was established following the 1974 "Boldt Decision" (*U.S. v. Washington*, 384 F.Supp. 312 (1974)), which reaffirmed the tribe's Treaty rights to fish (Clark 1985). Shortly after the Boldt Decision was issued, WDFW conducted a study based on a series of snorkel surveys to estimate steelhead parr production in populations across western Washington, including the Quillayute River and its tributaries, which they then used to generate models and estimate a range of escapement goals under varying assumptions (Gibbons et al. 1985). While the effort was laudable at the time, there are several limitations, some of which include sample size and representation (e.g., the juvenile snorkel surveys only occurred in a few rivers), survey counts that were not adjusted for diver error (likely resulting in underestimates), and failure to account for the depletion of early-timed steelhead (McMillan et al. 2022) or the effects of spatial distribution (Finstad et al. 2013; Atlas et al. 2015). Further, the escapement goals do not account for potential mining of certain life histories in species that display a high level of life history diversity, which in turn can reduce productivity and resilience (Ricker 1963). Last, there has not been any effort to evaluate or develop escapement goals or management targets for populations of summer steelhead.

Despite these limitations, the range of escapement goals estimated for each population of winter steelhead were not unreasonable for the time. In fact, the highest estimates could prove to be sufficiently conservative if the major questions are addressed through a sensitivity analysis and aspects of diversity, such as run timing and iteroparity, were accounted for. Unfortunately, WDFW and the co-managers disagreed over the escapement goal science and ultimately, the parr production models were rejected in favor of Maximum Sustained Harvest (MSH) and as a result, the agreed upon escapement goals were either among the lowest or were lower than the lowest estimates in Gibbons et al. (1985). The disagreement continues to this day, and as such, there is no agreed upon escapement goal for the Queets River or the Lower Quinault River (Cram et al. 2018). This is a great concern given the declines of wild winter steelhead specifically in those populations, but also overall because wild steelhead in the Olympic Peninsula DPS experience the highest harvest rates of any steelhead populations in Washington (Cram et al. 2018) and likely the world. The high harvest rates and declining population trends raise serious questions about whether the management targets are sufficient to sustain the winter steelhead populations and the fisheries they provide into an uncertain future.

A previous assessment of winter steelhead management in western Washington argued that a strong reliance on an MSH approach jeopardizes the health and resilience of a diverse species such as steelhead (Burge et al. 2006):

> "It has become abundantly clear that MSH theory and harvest models have not provided adequate protection for wild steelhead in the 20th and 21st centuries because they are too simplistic and allow high harvest rates that are unsustainable. These models do not annually or temporally account or plan for environmental variation, management error, the role of genetic and life history diversity in stock resilience and productivity, or the rebuilding of depleted tributary or seasonal runs. Rather, the models are rigid numbers-based equations and provide management tools that were developed to provide maximum harvests from a population without adequate considerations for long term stock health or the sustainability of annual fisheries."

Limited information and poor data quality on escapement levels can contribute to population decline (Knudsen 2000). A 2018 report from the National Park Service noted that some escapement estimates of salmonids on the Olympic Peninsula "are currently derived from surveys that occur in limited index reaches or are based on one or a few surveys during the peak of spawning" (Duda et al. 2018). The authors also noted there is a need "to fully understand the extent of bycatch and incidental mortality of Pacific salmonids in recreational and commercial fisheries in Olympic Peninsula Rivers" (Duda et al. 2018).

Given what we have outlined above, current escapement goals for the four major Olympic Peninsula populations are likely far too low and should be revised based on consideration of all available data. To ensure adequate escapement, the escapement goals should be revised to incorporate precautionary estimates of parr and smolt densities required to secure the resilience of each population given current and expected future variability in early marine survival. Estimates should be based on spawner age-composition and repeat spawner frequency targets with the primary objective of attaining minimum egg deposition sufficient to attain target mean parr or smolt densities in rearing habitats (see Gayeski et al. 2016 for some likely target parr density values). And the goals should begin to account for life history diversity that underpins the evolutionary strengths of steelhead.

In addition, any harvest – whether tribal commercial, ceremonial, subsistence, or recreational – should be reduced far below current comanager estimates of maximum sustainable yield/harvest (MSY). Recent declines in productivity indicate that MSY harvest rates are likely incompatible with population persistence or recovery. Petitioners obtained population data sets from WDFW for the Hoh River and Quillayute River winter steelhead populations (WDFW 2022d) These data sets contained run reconstructions that provided time series of spawners and maiden recruits spanning more than 25 years. Gayeski conducted a preliminary Bayesian estimation of the Ricker spawner-recruit model with time-varying productivity parameter (alpha) and constant capacity parameter (beta) on both data sets. The analyses were conducted in the program Stan. A copy of this estimation is included with this petition (*See* Appx. A).

Gayeski's estimates are shown in Figures 21 and 22. Both populations display a strong declining trend in alpha and the unfished equilibrium abundance (EQ). The Hoh River population displays a steady decline since 2003; the Quileute River population shows a steady decline beginning in 2007. The average value of alpha for the Hoh River population for the recent period of decline (2003 to 2013) is 2.51; the average value of EQ is 2,297. The average value of alpha for the Quillayute River population for the recent period of decline (2007 to 2015) is 2.16; the average value of EQ is 9,058. For comparison, the peak values of alpha and EQ for the Hoh River population during the period spanned by the data (1987 to 2013) occurred at the beginning of the time series, where alpha averaged between 3.6 and 3.7, and EQ between 3,200 and 3,300. Peak values for the Quillayute River population during the period spanned by the data (1978 to 2015) occurred in the early 1990s, where alpha averaged between 4.2 and 4.5, and EQ averaged between 17,000 and 18,000. These are biologically very significant and worrying declines. Further, values of both parameters in the past three years of record for each population are significantly lower than the recent average values.

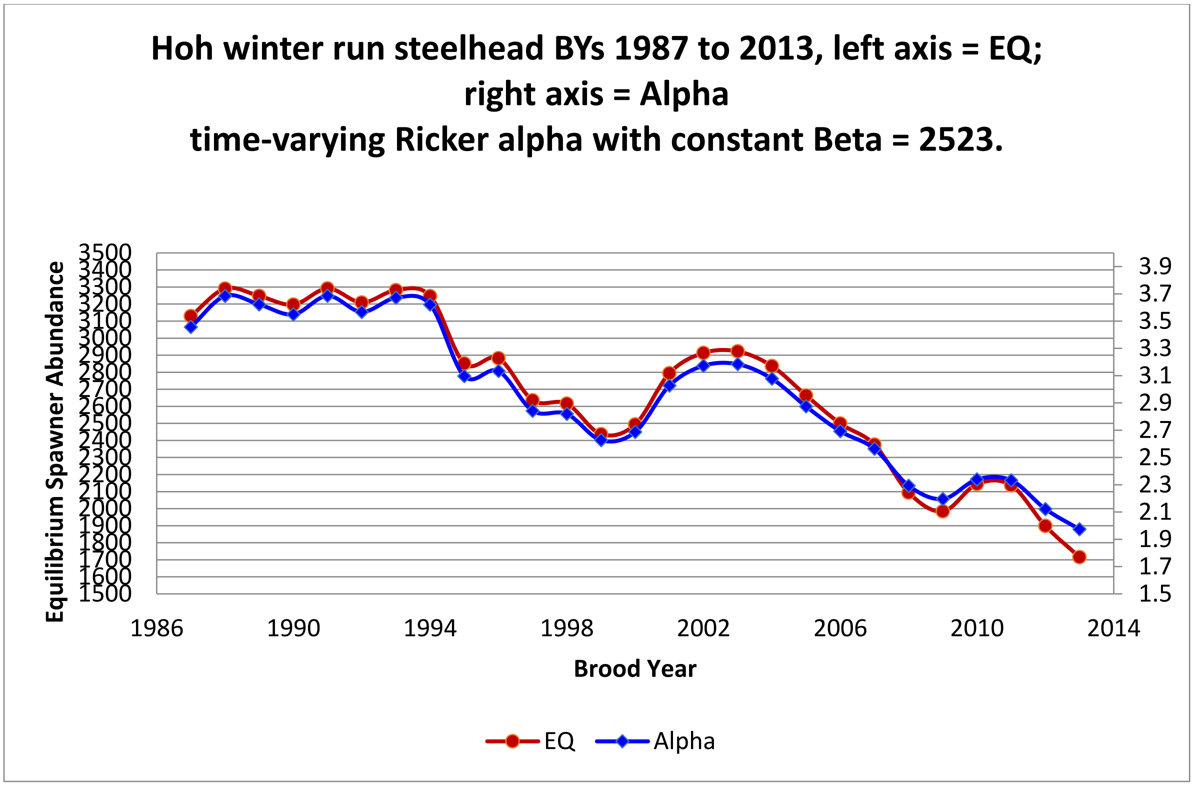


Figure 21. Time series of time-varying Ricker productivity parameter alpha and unfished equilibrium abundance for fixed beta parameter from a Bayesian stock recruit analysis of Hoh winter-run steelhead.

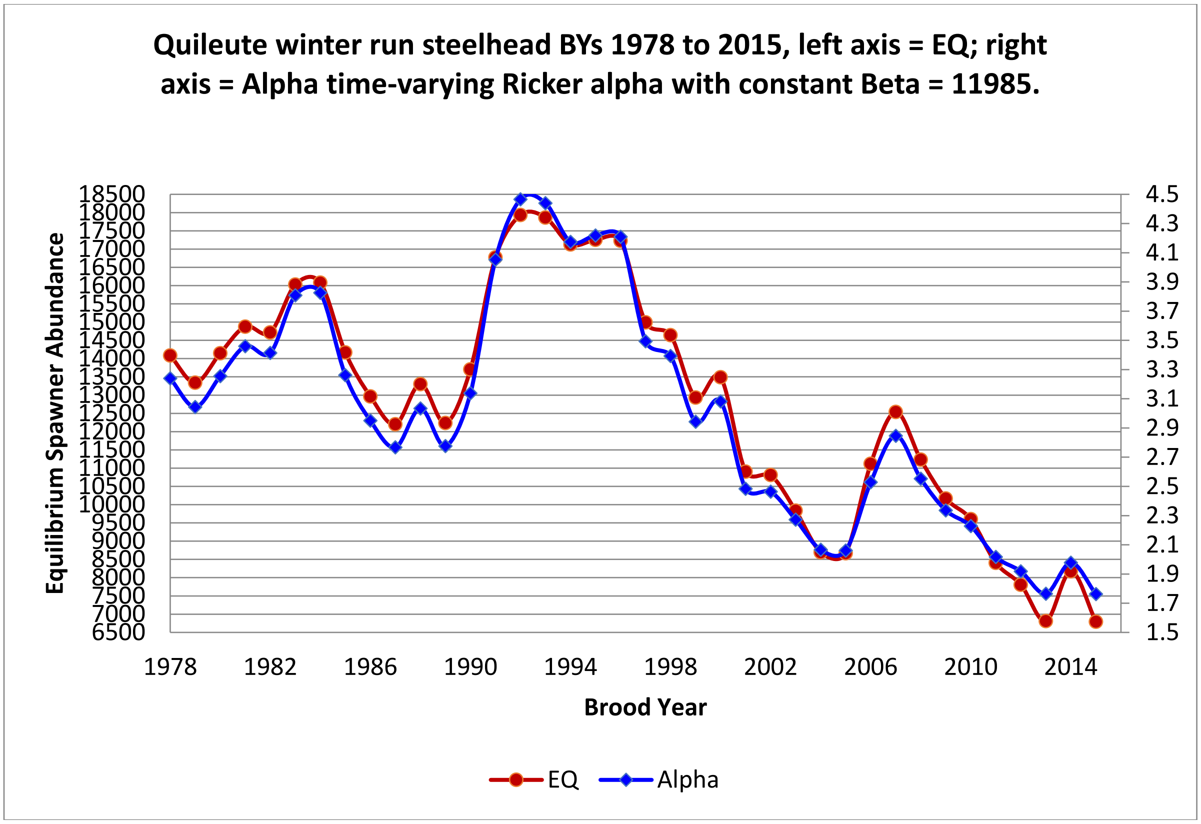


Figure 22. Time series of time-varying Ricker productivity parameter alpha and unfished equilibrium abundance for fixed beta parameter from a Bayesian stock recruit analysis of Quileute winter steelhead.

It is very probable that the other two large winter steelhead populations (i.e., Queets and Quinault River populations) and numerous smaller populations have experienced similar declines. These declines are cause for alarm.

B. Harvest Impacts

Wild winter steelhead populations in the Olympic Peninsula DPS are fished for and harvested in recreational and tribal fisheries at the highest levels of any steelhead populations in Washington State (Cram et al. 2018). Harvest can affect abundance, diversity, productivity, and spatial structure of salmon and steelhead populations, which in turn interact to shape the short- and long-term viability of the exploited populations (McElhaney et al. 2000). There are two main sources of harvest, including impacts from tribal gillnets and from recreational catch and release fishing.

Tribal commercial harvest of wild steelhead annually occurs in the Quinault, Queets, Hoh, and Quillayute River systems, and up until 2016, recreational anglers were permitted to kill wild steelhead on eight rivers on the Olympic Peninsula (Cram et al. 2018). Tribal harvest includes all fish that were reported, including commercial and subsistence catch.

The recreational fishery is now catch and release only for wild steelhead, and although catch and release generally has low mortality rates (2-10%: Hooton 1986; Barnhardt and Taylor 1996; Nelson et al. 2005), recreational fishing pressure and success has increased to the point where anglers are now catching and releasing the entire escapement, on average, more than one time (Bentley 2017; WDFW 2022c). Prior to 2016, WDFW determined recreational harvest using punch cards. Now that the fishery has shifted to catch and release, WDFW relies on creel surveys to estimate angler encounter rates with wild winter steelhead and estimate a 10% mortality to fish that are caught and released (Bentley 2017; Cram et al. 2018).

It is likely that harvest rates in the Quillayute, Hoh, Queets, and Quinault Rivers are underestimates. For example, we could not find estimates of gillnet dropout rates for wild winter steelhead in the tribal commercial fisheries. Injuries sustained by fish that escape gillnets can dramatically reduce their reproductive success if they survive to spawn (Baker et al. 2013). It is unknown how many redds are counted each year that were constructed by fish with lower fitness due to such injuries.

In addition, on average, every wild winter steelhead that escaped to spawn in the Hoh River in 2014 was caught and released, on average, 1.4 times by anglers (Bentley 2017). WDFW has several years of raw creel survey data, but the Hoh River in 2014 was the only stream and year for which they expanded those counts to estimate total encounter rates. The raw creel data suggests high encounter rates in other streams, with possibly even higher encounter rates in the Bogachiel and Sol Duc Rivers (WDFW 2022c, see Fishing Regulations section below). Although the 10% mortality rate that WDFW applies to recreational encounters is relatively high and conservative, those estimates are based on a fish being caught one time (Hooton 1987; Hooton 2001; Nelson et al. 2005; Twardek et al. 2018). The mortality rate for fish caught multiple times is unknown, but fish that enter early and remain in the system longer have a higher frequency of being caught more than once (Hooton 1987; Hooton 2001). Further, emerging research on Atlantic salmon indicates there can be significant sub-lethal impacts that can alter the migration and reduce the reduce fitness of adults (up to ~ 20-25%) that are caught and released (Richard et al. 2013; Bouchard et al. 2021; Papatheodoulou et al. 2021).

Potential effects from net dropouts and high recreational encounter rates, combined with the high harvest levels, raise concerns whether current estimates of harvest fully account for all impacts.

Below we review what is currently known about harvest rates for the four major wild winter steelhead in the Olympic Peninsula DPS and discuss harvest in relation escapement goals, how it has most likely contributed to changes in run timing, and how it is potentially affecting size and age and iteroparity.

We also summarize harvest records for summer steelhead and, when possible, we distinguish between wild and hatchery fish, but that is a challenge with the catch data. *See* (WDFW 2022e). From 1978-1985 returning adults were not marked with an adipose fin clip, and thus, only total catch of summer steelhead is reported for tribal and recreational fisheries. Thereafter, recreational catch consistently delineates hatchery and wild summer steelhead from 1985-2016, while the tribal catch records do not – except for the Quillayute River system from 2001-2020. Retention of wild summer steelhead by recreational anglers was banned by WDFW and the National Park Service in 1992, and consequently, from 1993 onward all recreational fisheries for wild summer runs were catch and release. Even though contemporary catch of wild summer steelhead is not high by tribes and anglers must employ catch and release, impacts could be more deleterious than they seem given the very depleted status of the stocks and the high levels of hatchery fish (see Hatchery Impacts section below).

1. Quillayute River System

Annual harvest rates of wild winter steelhead from 1978-2020 ranged from 10% to 55% in the Quillayute River system (Figure 23a), with a mean annual harvest rate of 28%, which is the lowest harvest rate of the four major populations (Table 1). Compared to the mean annual run size of 13,064 wild winter steelhead, the mean harvest rate equates to 3,724 fish per year. Total wild winter steelhead harvested per year ranged from 1,166 to 7,561 fish. Harvest rates have generally declined the past few years (Figure 23a) in response to smaller run sizes and shortened fishing seasons. The proportion of hatchery to wild winter steelhead in the commercial fishery was 2.1:1, and the proportion of hatchery to wild winter steelhead in the recreational fishery was 3.3:1 (Duda et al. 2018). The mean annual harvest rate of hatchery winter steelhead was 64% (range 32% – 86% of total run size; SD 12%) (Duda et al. 2018).

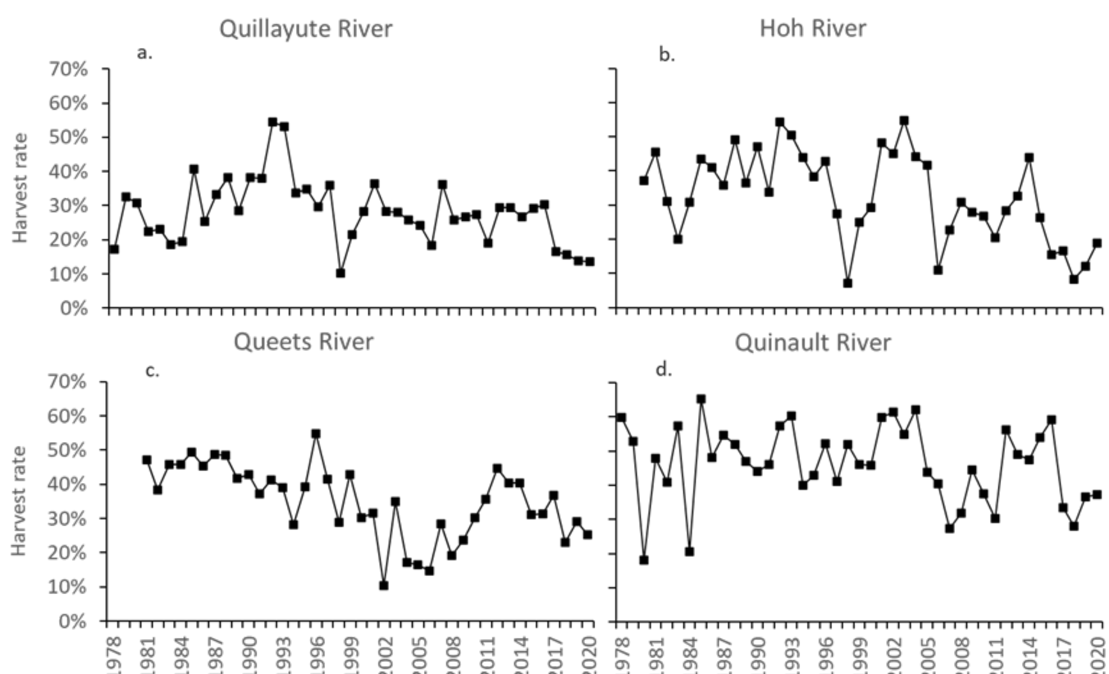


Figure 23. Annual harvest rates of wild winter steelhead populations in the a) Quillayute, b) Hoh, c) Queets, and d) Quinault Rivers. Based on co-manager data sets covering period circa 1980-2020.

Harvest rates of wild summer steelhead from 1978-2020 were not available because co-managers do not monitor summer runs. We do, however, have catch data for summer steelhead, though wild and hatchery fish could not be distinguished prior to 1986, the year the first adults returned with adipose fin-clips. Hatchery summer runs were first planted in the Calawah and Sol Duc Rivers in 1977 and were stopped in the Sol Duc in 2012 (Figure 24), and according to the hatchery records we reviewed, the Quillayute River watershed is the only system to have received direct plants of hatchery summer steelhead (Duda et al. 2018).

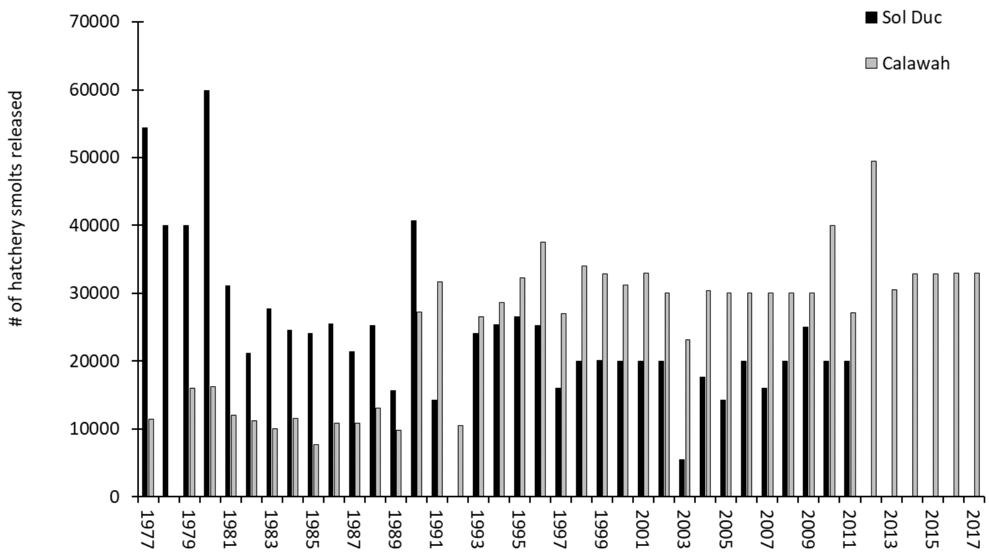


Figure 24. Number of hatchery summer steelhead smolts released into the Sol Duc and Calawah Rivers, which are tributaries to the Quillayute River watershed.

We separated the Quillayute River catch data into recreational (1978-2016) and tribal Treaty catch (1978-2020) because the recreational catch distinguishes between hatchery and wild fish in all years possible while we could only find that delineation for tribal catch from 2001-2020. As mentioned previously, historical catch of wild summer steelhead was very low and inconsistent in the Quillayute prior to hatchery releases in 1977, but from 1978 to 1985 there was a spike in harvest of adult summer steelhead, many of which were likely hatchery (Figure 25). After fin-clipped hatchery adults started to return in 1986, the catch of wild fish declined and remained low while the harvest of hatchery fish increased (Figure 25). Retention of wild summer steelhead by recreational anglers was banned by WDFW and the National Park Service in 1992, and consequently, from 1993 onward all recreational fisheries for wild summer runs were catch and release. However, WDFW records show some wild summer steelhead harvest after 1992, and we are not sure why, though it is possible the data represents fish that were illegally harvested (Figure 25). Regardless, based on the punch card data, 2006 was the last year a wild summer steelhead was reported as harvested by recreational anglers in the Quillayute River system (Figure 25). Mean annual recreational harvest from 1986-2006 was 54 wild summer steelhead (range = 0 – 239) and from 1986-2016 mean annual recreational harvest was 673 hatchery summer steelhead (range = 119 – 1,974).

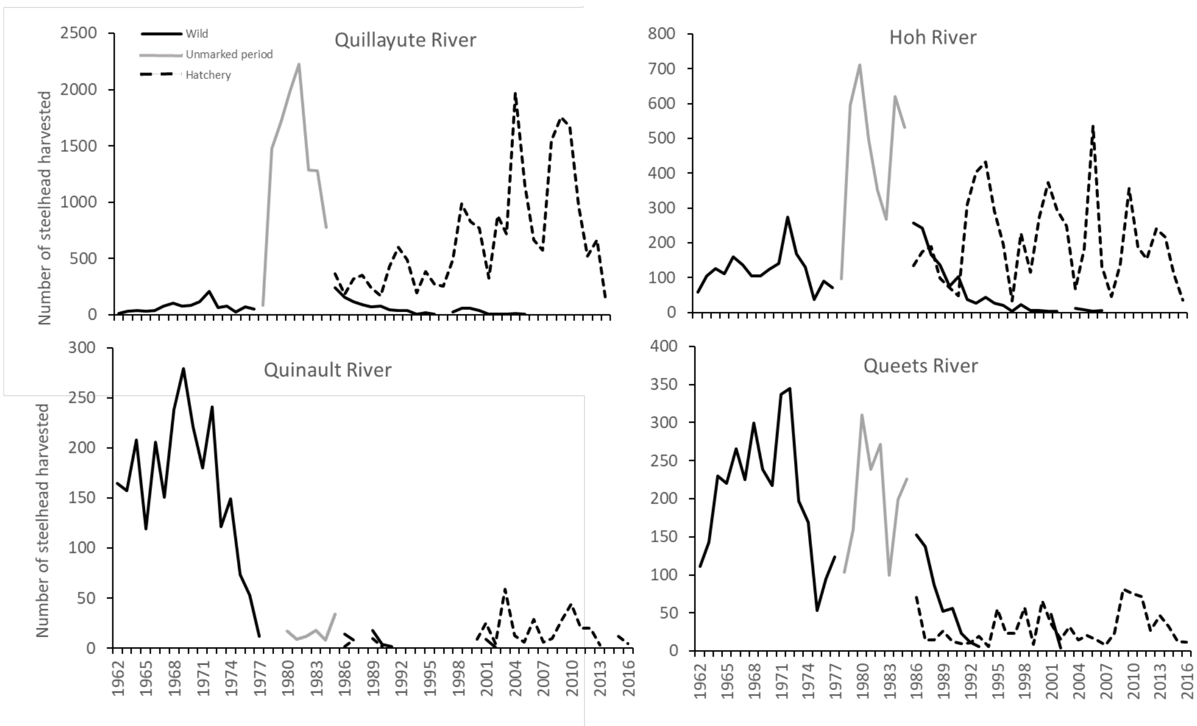


Figure 25. Number of summer steelhead harvested by recreational anglers from 1962-2016 in the four largest populations in the Olympic Peninsula DPS. The black line denotes wild steelhead with an adipose fin. The grey line denotes unknown steelhead because harvest occurred during a period when hatchery fish were introduced, but not outwardly marked, so wild and hatchery could not be distinguished. The dashed black line denotes hatchery steelhead. Retention of wild summer steelhead by recreational anglers was banned from 1993 onward by WDFW and the National Park Service.

We found two sets of tribal catch data with slightly different annual catch numbers. The first ranges from 1978-2016 and does not distinguish between hatchery and wild summer steelhead. The second data set ranges from 2001-2020 and, although it does not contain catch records for all years, it does delineate hatchery and wild steelhead in years when catch records were available (Figure 26 and Figure 27). The data sets are compared in Figure 27, which shows the data from the 1978-2016 and 2001-2020.

Overall, the tribal harvest of summer steelhead in the Quillayute River is generally higher than other populations, except for the Quinault River (Figure 26). Mean annual tribal catch from 1978-2016 was 341 summer steelhead of unknown origin (range = 43 – 1,120), while mean annual tribal catch from 2001-2020 was 107 wild summer steelhead (range = 24 - 244) and 308 hatchery summer steelhead (range = 28 – 839). Given the lack of monitoring and escapement goals, there should be concern because the reported harvest of wild steelhead is high (Figure 24) compared to the snorkel counts of staging adults (Table 7) from circa 2000-2010.

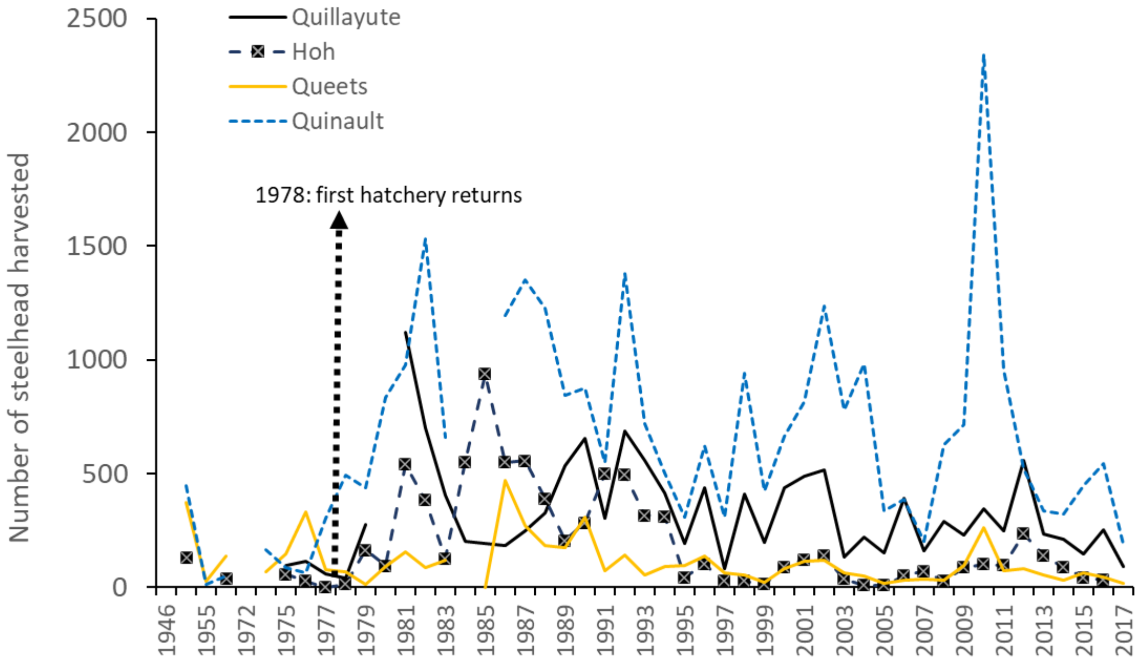


Figure 26. Number of summer steelhead harvested by tribal fishers in Quillayute, Hoh, Queets, and Quinault Rivers, with occasional data available for some populations from 1946-1975 and generally consistent annual catch data available from 1976-2017. Data were provided by WDFW and did not include information on which fish were hatchery or which were wild. Based on releases of hatchery smolts in Quillayute River system in 1977, 1978 would be the first year of potential adult returns. So, data thereafter is conflated by presence of hatchery fish, which could represent a substantial component of the catch – even in watersheds outside the Quillayute River as evidenced by sport catch data in Hoh and Queets Rivers.

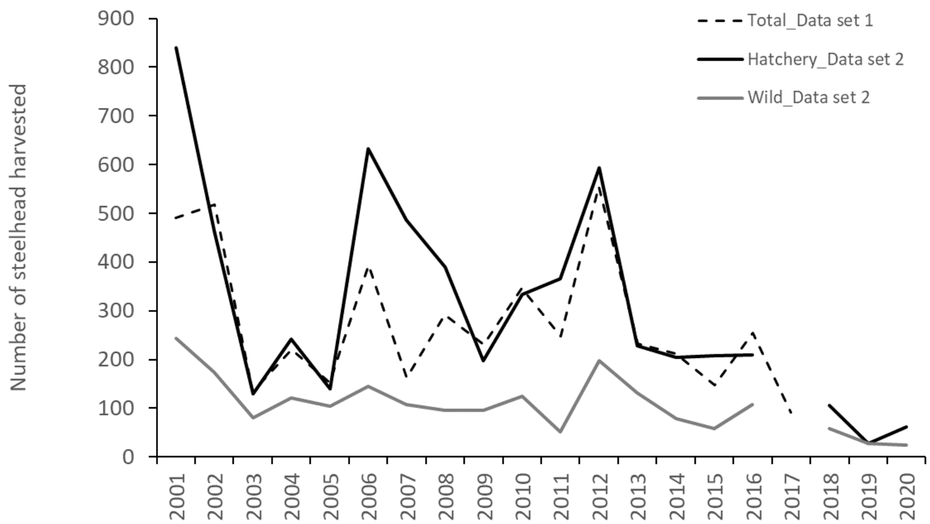


Figure 27. Number of summer steelhead harvested in Quillayute River by tribal fishers, which includes two disparate data sets: data set 1, which only provides total numbers of summer steelhead harvested and does not delineate between hatchery and wild; and data set 2, which delineates total harvest by hatchery and wild. We provide both data sets because they contained different harvest numbers.

2. Hoh River

The mean annual harvest rate from 1980-2020 was 35% for Hoh River wild winter steelhead (Table 1), with a range of 7% to 55% per year (Figure 23b). This equates to a harvest of 1,361 wild winter steelhead per year, with a range of 261 – 2,800, compared to a mean annual run size of 4,118 steelhead. Harvest rates have generally declined the past few years (Figure 23b) in conjunction with declining run sizes and shortened fishing seasons. The ratio of hatchery to wild winter steelhead in the commercial fishery was 2:1, and it was 3.4:1 in the recreational fishery. (Duda et al. 2018).

As with the other populations, we delineate the recreational harvest of summer steelhead in the Hoh River population into three periods, including: (1) the historical period discussed in a previous section dating from circa-1950s through 1977; (2) the post hatchery release period when wild and hatchery steelhead could not be distinguished (1978-1985); and (3) the period from 1985 to the present when wild and hatchery fish could be distinguished (Figure 25). Although the Hoh River did not receive hatchery releases, the harvest of summer steelhead sharply peaked from 1978-1985 like the Quillayute River did. Since 1985, the harvest of wild summer steelhead declined to zero in 2007, while the harvest of hatchery summer runs has remained relatively high (Figure 25). Mean annual recreational catch from 1978-1985 was 459 steelhead (range = 96 - 711). The last year a wild summer steelhead was reported as harvested by recreational anglers was 2006, the same as the Quillayute River system, though retention of wild summer runs was banned by managers in 1993. Mean recreational harvest from 1986-2006 was 52 wild summer steelhead (range = 0 - 257) and, interestingly, 204 hatchery summer steelhead (range = 33 - 433). Given the consistent hatchery catch by recreational anglers over the period of record in the Hoh River, the spike in catch from 1978-1985 was potentially associated with hatchery steelhead straying from the Quillayute River.

Tribal harvest of summer steelhead has been variable since 1978 and was not delineated by hatchery and wild fish (Figure 26). The mean annual tribal catch from 1978-2016 was 207 summer steelhead of unknown origin, with a peak of 937 fish and a low of 10. Overall, the reported catch levels, as with the Quillayute River system, seem high compared to the very low abundance of wild summer steelhead reported in snorkel surveys in the SF Hoh River (Table 4). This is even more concerning due to the lack of monitoring and escapement and fishery goals.

3. Queets River

The mean annual harvest rate from 1978-2020 was 35% for Queets River wild winter steelhead (Table 1), with a range of 10% to 55% per year (Figure 23c). This equates to a harvest of 2,672 wild winter steelhead per year, with a range of 575 – 6,291, compared to a mean annual run size of 7,384 steelhead. There could be a slight declining trend in harvest rate over the period of record (Figure 23c). The ratio of hatchery to wild winter steelhead in the commercial fishery was 0.7:1 and 4.7:1 in the recreational fishery. (Duda et al. 2018). Since 1976, the mean annual commercial harvest of hatchery steelhead has been 1,472 fish (range 298 – 3,308; SD 766) (Duda et al. 2018).

Recreational catch of wild summer steelhead in the Queets River was highest during the historical period, and then showed a similar peak in catch as the Quillayute and Hoh Rivers from 1978-1985, followed by a sharp decline in catch thereafter until recreational harvest of wild summer runs was eliminated in 1993 though some summer steelhead were reported as harvested from 2001-2004 (Figure 25). Like the Hoh River, the Queets River does not receive hatchery summer steelhead smolts, and yet, they have been caught each year by anglers since hatchery fish were first marked (Figure 25). Mean annual recreational catch from 1978-1985 was 201 steelhead (range = 100 - 310), compared to 53 wild summer steelhead (range = 0 - 153) from 1986-1992 and 2001-2004 and 30 hatchery summer steelhead (range = 8 - 81) from 1986-2016.

Tribal catch is not delineated by hatchery and wild. Zero catch was reported in 1984 and only two fish were reported in 1985. Except for these years, relatively low levels of catch have been reported (Figure 26). Mean annual tribal catch from 1978-2016 was 102 summer steelhead (range = 2 – 469). Unfortunately, there is almost no data on the abundance of wild summer steelhead in the Queets River, so it is very difficult to draw inferences about potential harvest impacts. Nonetheless, wild summer steelhead were formerly quite abundant as we described earlier in the Historic Abundance section, and it is likely they are now highly depleted. Consequently, we have grave concern about this population because it is not monitored, there are no escapement goals, and there has been no effort to evaluate status and trends to inform management and conservation.

4. Quinault River

Annual harvest rates of wild winter steelhead from 1978-2020 ranged from 15% to 65% in the Quinault River (Figure 23d), with a mean of 46% (Table 1). That is the highest mean harvest rate among the four largest populations in the DPS (Table 1). It equates to 2,832 fish per year, with a range of 814 – 6,356, compared to a mean annual run size of 5,968 wild winter steelhead. Since 1977, there has been a mean annual commercial harvest of 7,089 hatchery winter steelhead (SD = 3,956) with a range of 1,068 – 15,979 fish, and a mean annual recreational harvest of 692 hatchery winter steelhead (SD = 418) and a range of 61 – 1,656 fish (Duda et al. 2018). The ratio of hatchery to wild winter steelhead in the commercial fishery was 2.9:1 and in the recreational fishery it was 3.0:1 (Duda et al. 2018).

The pattern and extent of annual recreational catch of summer steelhead in the Quinault River differs from other watersheds in that it did not experience the peak in catch during the period when hatchery fish were unmarked and overall, catch remained very low until some hatchery steelhead started to show up in the catch in the late-1990's (Figure 25). Overall, mean annual recreational catch from 1978-1985 was 15 steelhead (range = 8 - 34), compared to 8 wild summer steelhead (range = 0 - 18) from 1986-1992 and 2001-2002 and 15 hatchery summer steelhead (range = 0 - 44) from 1986-2016.

Tribal catch of summer steelhead from 1976-2017 is higher than all other populations (Figure 26). The mean annual catch was 753 steelhead, with a range of 191 to 2,345 fish. However, given the very low snorkel counts of wild adult summer steelhead in the upper EF and NF Quinault Rivers (see Abundance section), it is unclear whether true wild summer runs are being harvested at very high rates or whether some of the fish are winter run kelts. It is also unclear whether hatchery summer runs are contributing. A combination of all three factors could possibly explain the very high catch levels reported by the tribe.

We delineated catch by month. The highest mean catch was in May, which thereafter declined through September until catch slightly increased in October (Figure 28). May is an important month of spawning for Olympic Peninsula steelhead (Busby et al. 1996), so some (or many) of those fish could be kelts. Kelts are likely being captured in June, considering spawn timing in other nearby populations (e.g., McMillan et al. 2007). Perhaps some of the fish are also hatchery summer steelhead from releases in streams further south in Grays Harbor, such as the Wynoochee River. Whatever the case, it seems imperative to determine the life histories and source of the fish, because snorkel surveys in the upper watershed suggest the population is greatly depleted and potentially close to extinction.

C. Recent failures to Meet Escapement Goals

A report by Cram et al. (2018) evaluated the number of years the harvested wild winter steelhead populations met their escapement goals from 2004-2013 (Table 3). We updated those estimates to cover the most recent ten-year period (2011-2020) for all populations and tributary populations (Table 1). There is an overall problem with many populations failing to meet their escapement goals, particularly in recent years (Table 1). For example, among the wild steelhead populations that experience harvest, the Hoh River has only met its escapement goal of 2,400 fish in six of the past ten years (Table 1) and only 50% of the years dating back to 2003 (Figure 5).

The Queets River population has only met its WDFW and National Park Service escapement goal of 4,200 fish in 30% of the last ten years (Table 1), and run sizes are now coming in below the escapement goal (Figure 8a).

The Clearwater River, the major tributary of the Queets, has met its escapement goal of 1,450 fish only 50% of the years in the past decade (Table 1), with recent escapements being the lowest during the period of record (Figure 8b).

There is no escapement goal for the entire Quinault River population, but its run sizes have collapsed to the point where it certainly would be below any reasonable estimate for spawner goals (Figure 7).

The Quillayute River system overall has remained the strongest among the populations, meeting its escapement goal of 5,900 fish for the entire watershed in nine out of the past ten years (Table 1) and only missing the goal in two of the past twenty years (Figure 6a). Among its major tributaries, the Dickey (Figure 6b) and Calawah Rivers (Figure 6d) have met their escapement goal each of the past ten years (Table 1), and they are the only two tributaries that do not have a long history of receiving releases of hatchery winter steelhead. In contrast, the Bogachiel/Quillayute (Figure 6e) and Sol Duc Rivers (Figure 6c) have only met their escapement goal in 60% and 70% of the last ten years, respectively (Table 1).

As mentioned in the abundance and trends section, nearly all the smaller, independent populations of wild winter steelhead have failed to meet their escapement goals during much of – if not all – the past decade (Table 1). Most concerning, some of the populations have not achieved their escapement goals for extensive periods of time dating back 20 years or more.

D. Demographic, Genetic, and Ecological Risks Caused by Harvest

Harvest poses demographic and genetic risk to wild salmon and steelhead (ISAB 2005; Hard et al. 2008) and can alter attributes including, but not limited to, size and age at maturity, migration timing, spatial distribution, and life history diversity, which, in turn, interact to shape the productivity and resilience of wild salmon and steelhead populations (McElhaney et al. 2000; ISAB 2005; Hard et al. 2008). Given the high exploitation rates on the four largest populations of Olympic Peninsula wild winter steelhead, it is possible that chronic, high levels of harvest have imposed a variety of impacts. Unfortunately, research on such topics is almost entirely lacking for the DPS, except for evidence of change in run timing of wild winter steelhead (McMillan et al. 2022).

Cram et al. (2018) noted the risk of fishery selection on run timing since the number of fishing days per week for treaty fisheries is highest early in the season, when the fishery targets greater harvest of hatchery adults. Historical recreational fisheries were structured similarly, with a focus on harvesting early returning hatchery adults that were intermixed with wild adults. The variable historical and contemporary fishing rates indeed appear to have contributed to the substantial depletion of early returning wild steelhead (Figure 15). As a result, run timing is more compressed than it was historically.

The results challenge the findings in Busby et al. (1996), which described hatchery and wild winter steelhead as being temporally segregated because of differences in run timing. It also underscores the importance of the Shifting Baseline Syndrome, where managers and scientists come to accept the current level of abundance and diversity as the norm (Pauly 1995). For example, in the Quileute Tribal language, the month of January is defined as the "time of steelhead running" and February is defined as the "time of steelhead spawning" (Frachtenberg 1916). Thus, based on quantitative data and traditional ecological knowledge, it is clear there has been a large change in run timing that has not been accounted for or addressed in previous reviews of the Olympic Peninsula DPS (e.g., Busby et al. 1996).

The change in run timing could impact the population in several ways. Per McMillan et al. (2022):

> "Changes in run timing can shorten breeding seasons, reduce phenotypic diversity, and lower population productivity (Tillotson and Quinn 2018). More protracted anadromous fish migrations, like those that we estimated to occur historically for wild Winter Steelhead, can allow fish to temporally stagger the use of spawning habitat (Gharrett et al. 2013), thereby reducing density-dependent effects on juvenile survival and increasing local habitat capacity (Chandler and Bjornn 1988).
>
> Migration timing is also often associated with the spatial structure of breeding locations in anadromous fish populations (e.g., Everest 1973; Stewart et al. 2002; Beacham et al. 2012). On the [Olympic Peninsula], McMillan et al. (2007) observed spatial correlations with spawn timing in the Quillayute River, where spawning higher in the stream network in smaller stream channels occurred about a month earlier than in the lowermost mainstem spawning reaches. Previous observations by Cederholm (1984) also found that earlier returning Queets River wild Winter Steelhead were more likely to spawn in smaller tributaries, while mainstem spawning tended to occur several weeks later in the season. Importantly, Cederholm (1984) observed this occurrence in low

elevation tributary streams, so the pattern of earlier spawn timing in tributaries is not necessarily limited to higher elevation headwaters as might be inferred just from the results of McMillan et al. (2007), but rather it appears to be a pattern associated with stream size."

In addition, migration timing underpins a population's adaptive capacity to keep pace with shifting climatic conditions, such as changing streamflow and temperature regimes (Manhard et al. 2017; Reed et al. 2011; Austin et al. 2020). For example, winter steelhead tend to migrate and spawn earlier in the season in warmer streams in more southerly habitats (Busby et al. 1996), presumably because the higher streamflows provide better access and earlier spawning is needed to ensure offspring emerge before the onset of summer baseflows. Climate change models predict water temperature regimes on the Olympic Peninsula (see Climate Change section) will become more similar to those in more southerly climates (Wade et al. 2013). This suggests that early migrating life histories will become increasingly important to population resilience. However, rebuilding earlier-timed wild winter steelhead migrations will be challenging without addressing the current fishery structure and hatchery practices (see Hatchery section below).

Harvest could also be impacting diversity though selection on size and age and selection against iteroparity. Harvest can alter size and age in salmonids (Hard et al. 2008), and Cram et al. (2018) reported that scale samples indicated gillnetting was selectively capturing older adult wild steelhead, while recreational fisheries were capturing more younger fish (Cram et al. 2018). It is unknown how these effects have manifested over the decades, but it is possible that both fisheries have truncated the overall distribution of size and age in the populations experiencing high levels of harvest.

Steelhead are iteroparous, and individuals that repeat spawn are significantly more productive on their second attempt than their first and forgo early reproduction to devote additional energy to continued survival (Christie et al. 2018). Repeat spawning fish may also come back larger and with more eggs (Christie et al. 2018; Gayeski, personal observation on recaptured tagged steelhead in the Sopichnaya River, Kamchatka 2005). In this context, kelts migrating back to the ocean after spawning their first time are only likely to contribute to reproduction if they are allowed to spawn a second time. Harvesting those individuals is analogous to removing the most productive life history from the next generation of spawners. Kelts can appear in the system beginning as early as late-January, but they are most common after the peak of spawning from April through May. During that time kelts are harvested in gillnet fisheries and captured by recreational anglers in catch and release fisheries.

Although the steelhead fishery is closed in May, there have traditionally been recreational and tribal Spring/Summer Chinook Salmon fisheries in each of the rivers that begins in April and runs through May and into June. Those fisheries could have significant impacts on kelts. McMillan (2006) hypothesized a portion of the relatively high numbers of wild steelhead harvested by Treaty fisheries in May are likely kelts – up to 1,800 fish in May in some years – despite being considered "summer runs" by co-managers. This seems possible, especially in the Quinault River where the catch numbers of steelhead in May are particularly high (Figure 28). There is also an intensive sport fishery in the Quillayute and Sol Duc Rivers for hatchery spring/summer Chinook, although it is unclear how many kelts are caught, it could be substantial because most anglers are using bait. And while sport fisheries for spring Chinook salmon are now closed on the Hoh and Queets Rivers, historically they were also open to recreational angling at different periods. Because kelts start to feed after spawning, they are also prone to being caught by recreational anglers and, considering the low lipid levels and poor condition of kelts (Penney and Moffitt 2014), any further expenditure of energy – such as being hooked and caught – could result in delayed mortality.

In addition, as noted by Gayeski et al. 2016, steelhead kelts (if unstressed by post-spawning capture in river fisheries) have higher survival rates to repeat spawning in the following one or two years than smolts that must survive several years in the marine environment in order to become maiden spawners. Thus, repeat spawners provide a considerable boost to the overall productivity and juvenile capacity of the population than smolts that survive to spawn for the first time. This is particularly critical to the recovery of depressed populations.

Ultimately, the sharp declines in iteroparity in the Queets River population since 1980 and low levels of iteroparity in other wild winter steelhead populations (Figure 16) have likely contributed to the chronic declines over the past twenty years or more. The depleted early returning wild steelhead combined with potential changes in size and age and reductions in iteroparity raise serious questions about the sustainability of the fishery model and highlights the need to incorporate run timing and repeat spawning into steelhead management and fisheries.

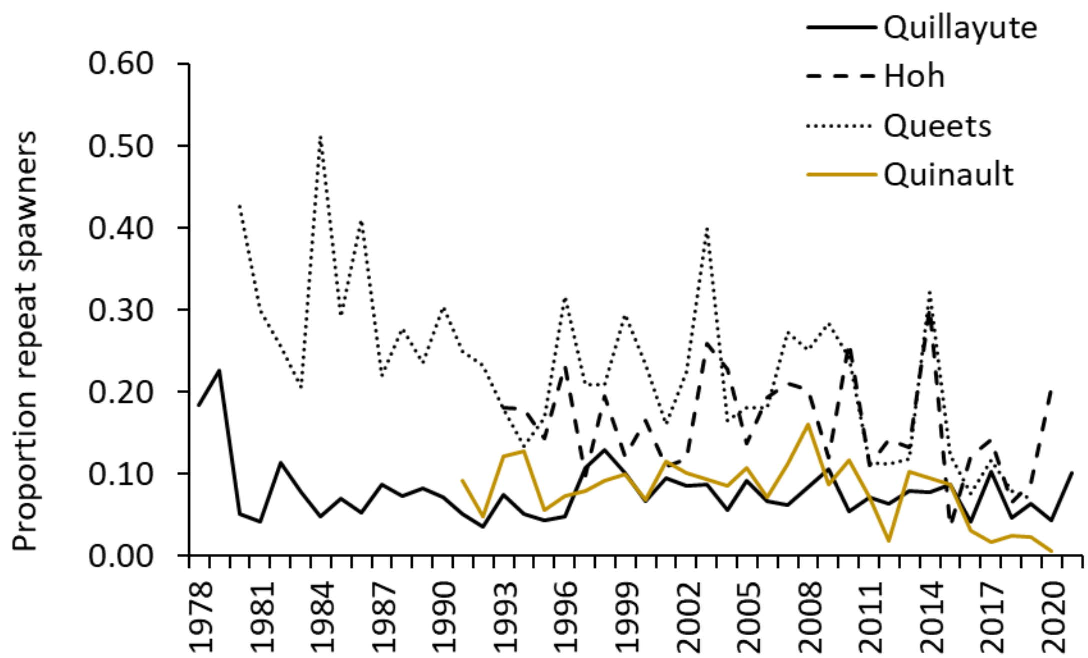


Figure 28. Proportion of repeat spawning individuals (repeats/total run size) by year for wild winter steelhead populations in the Quillayute (1978-2021), Hoh (1993-2020), Queets (1980-2019), and Quinault Rivers (1991-2020).

III. DISEASE AND PREDATION

A. Disease

The M genogroup of the infectious hematopoietic necrosis virus (IHNV), which is harmful to steelhead, has been previously detected in seven hatchery locations where wild Olympic Peninsula steelhead occur (Breyta et al. 2013). Between 2007 and 2011, there were two distinct waves of the virus that occurred in multiple watersheds, including the Hoh, Queets, Quinault, and Quillayute Rivers (Breyta et al. 2013). Most fish detected with the virus were hatchery fish, however, wild fish are less commonly sampled (Breyta et al. 2013). The virus was detected in wild fish in the Hoh and Quinault Rivers (Breyta et al. 2013). According to Breyta et al. (2013), “it is not clear what the future burden of IHNV in coastal steelhead trout might be.”

B. Predation

There is an increased distribution of predators in the Dickey River, likely because of warming water temperatures (Smith 2000). It is likely that predation risks will increase in Olympic Peninsula rivers as summer streamflows decrease and water temperatures increase (Dalton et al. 2016).

IV. INADEQUACY OF EXISTING REGULATORY MECHANISMS

A. FEDERAL

1. National Forest Management Act & Northwest Forest Plan

The National Forest Management Act (NFMA) requires the U.S. Forest Service (USFS) to manage fish and wildlife habitat in the Olympic National Forest to "maintain viable populations of existing native and desired non-native vertebrate species." 36 C.F.R. § 219.19. In 1990, the USFS adopted a Land and Resource Management Plan ("LRMP") for the Olympic National Forest (USFS 1990). The USFS claimed that implementation of the plan would increase fish production potential by more than 10% by the end of first 10 years of implementation (USFS 1990). It also claimed that it would correspond with 1,200,000 additional anadromous smolts and that anadromous fish production would increase by 25% above then current levels due to decreased sedimentation and habitat enhancement projects (USFS 1990).

In 1998, the LRMP was amended to include management changes consistent with the Northwest Forest Plan (1994) (Hoffman 1998). The NFP includes an Aquatic Conservation Strategy (ACS), which establishes measures intended to restore and maintain ecological processes of aquatic and riparian habitat (Thomas et al. 1993; Reeves et al. 2006). Among other things, the ACS requires the USFS to "maintain and restore the sediment regime under which aquatic ecosystems evolved" (USDA 1994). To date, the ACS has not maintained or restored the sediment regime under which Olympic Peninsula steelhead evolved, and Petitioners did not locate evidence to suggest that the LRMP and the improvements made by the ACS have increased anadromous fish production by 25% or more over 1990 levels.

The ACS also established riparian reserves, which place primary emphasis on protecting fish habitat. The size of the buffers depends on whether the streams are designated as fish- or non-fish-bearing. Riparian buffers on fish-bearing streams are two site-potential tree heights (≥ 200 years old) or 300 ft, whichever is greater. Riparian buffers for non-fish bearing streams are one site-potential tree height or 100 ft, whichever is greater (Wilhere and Quinn 2018). The buffers are intended to help ensure that species meet certain viability standards (Wilhere and Quinn 2018). When developing the ACS, the USFS assumed that if a species has at least an 80% likelihood of a stable, well-distributed population over 100 years, it is viable (Wilhere and Quinn 2018).

The USFS is also guided by an Olympic National Forest Strategic Plan, which "integrates aquatics, wildlife, silviculture, and fire, helping to identify priority areas for management activities such as habitat restoration, road decommissioning, forest thinning, and fuel reduction treatments" (Halofsky et al. 2011). It also has a Road Management Strategy, which it developed, in part, to meet ACS standards (Halofsky et al. 2011). The road strategy sets priorities for road maintenance, upgrading, and decommissioning based on several factors, including aquatic risk and high-value watershed goals (Halofsky et al. 2011).

The NFP established the Aquatic and Riparian Ecosystem Monitoring Program (AREMP) to monitor whether implementation of the ACS is improving watershed conditions (Gaines et al. 2022). Unfortunately, the program has been hindered by insufficient funding and changes to monitoring protocols, which make it challenging to measure the impacts of the ACS on aquatic resources (Gaines et al. 2022).

Roads

The ACS has not been effective in reducing road density or improving other road-related factors that affect fish (Frissell et al. 2014). According to Frissell et al. "[t]he magnitude of existing road impacts on watersheds and streams in the [NFP] may equal or exceed the effect of all other activities combined" (Frissell 2014).

Roads present major risks to fish and aquatic resources in the Olympic National Forest. In 2020, the USFWS stated that "forest roads in the Olympic National Forest have been a chronic source of sediments for decades" (USFWS 2020). Although the USFS has decommissioned 435 miles of roads in the Olympic National Forest since 1990 (Halofsky et al. 2011), hundreds of road miles still present significant risks to aquatic resources (USFS 2015). Fifty-one percent (1,032 miles) of all roads in the Olympic National Forest present high aquatic risks (USFS 2015). Thirty-three percent (651 miles) of the roads in the national forest are rated as presenting medium aquatic risks (USFS 2015). As a result, only 17% (338 miles) of roads in Olympic National Forest present low aquatic risk (USFS 2015). Nearly one-third of the Olympic National Forest's roads are proposed for decommissioning (Halofsky et al 2011).

In addition to falling behind on road decommissioning, the Olympic National Forest has not received the funding necessary to bring all of the roads up to current standards (USFWS 2020). Additionally, some roads that have been previously maintained need additional work (USFWS 2020).

Riparian Revegetation

Although there has been some revegetation in riparian corridors in the Olympic National Forest, many corridors continue to have few conifers because of historic logging practices (Halofsky et al. 2011). Reestablishing conifers in these corridors would help restore sources of large woody debris (Halofsky et al. 2011). However, these projects require costly, long-term commitments and, therefore, they have not been a high priority for forest managers (Halofsky et al. 2011).

Additionally, treatments in riparian areas can disturb soils and decrease effectiveness in retaining sediment and nutrients (Frissell et al. 2014). Thinning in riparian areas can diminish summer flows because of increased water demand by regrowth of vegetation (Frissell et al. 2014).

Climate Change

It is unclear if the USFS has reviewed the ACS to determine whether any science-based changes to the strategy are necessary in response to climate change (Frissell 2014). Current forest plans are not flexible enough to adapt to climate change challenges (Gaines et al. 2022). To strengthen watershed resiliency to climate change, the ACS could be revised to include additional steps to reduce non-climatic stressors (Gaines et al. 2022). For example, reducing the impact of roads would foster greater ecosystem resiliency to climate change (Gaines et al. 2022). To date, the ACS has not been updated to specify steps to reduce non-climate-related stressors on watersheds as a method to increase resiliency to climate change.

2. Endangered Species Act

Several populations of Olympic Peninsula steelhead occur in areas where ESA-related conservation and recovery efforts are underway for other species, including bull trout (USFWS 2020). As a result, Olympic Peninsula steelhead may benefit from the habitat protections afforded to bull trout (USFWS 2020). The status of Olympic Peninsula steelhead, however, demonstrates that these benefits are not sufficient to halt or reverse their decline. Olympic Peninsula steelhead need their own protection under the ESA.

a. Critical Habitat for Bull Trout

In 2005, the USFWS designated a series of waterbodies as critical habitat for coastal populations of bull trout. 70 Fed. Reg. 56212, 56304-56306 (Sept. 26, 2005). In 2010, the USFWS updated the designations for 32 critical habitat units (CHUs), including the Olympic Peninsula Unit (75 Fed. Reg. 63898 (Oct. 18, 2010)). The USFWS designated 121 waterbodies as critical habitat in the Olympic Peninsula Unit, excluding certain geographic areas covered by the Washington State Forest Practices Plan (HCP) or covered by tribal plans. *Id.*, at 63968-69369.

Critical habitat for bull trout overlaps with multiple areas where Olympic Peninsula steelhead occur (75 Fed. Reg. 63875-63978) Specifically, Olympic Peninsula steelhead occur in the following critical habitat for bull trout: Clearwater River, Copalis River, Hoh River, Kalaloch Creek, Moclips River, Mosquito Creek, Quinault River, Queets River, Raft River, Salmon River, South Fork Hoh River, and Tshletshy Creek. As a result, Olympic Peninsula steelhead in these rivers and streams may benefit from critical habitat protection afforded to bull trout.

b. Biological Opinion on Forest Management Activities on the Olympic National Forest

On June 24, 2020, the USFWS issued a biological opinion addressing Forest Management Activities on the Olympic National Forest (USFWS 2020). The biological opinion evaluates the potential effects of management actions on ESA-listed bull trout, Northern Spotted Owl, and Marbled Murrelet. It requires certain conservation measures for projects that occur within bull trout core area watersheds or in designated critical habitat for bull trout. Several populations of

Olympic Peninsula steelhead occur in bull trout core area watersheds and critical habitat and therefore, may indirectly benefit from these conservation measures. These measures include the following: steps to prevent erosion and enable large wood recruitment; limits on commercial thinning and road maintenance activities; prohibitions on new culvert installations or culvert replacements; no-cut buffers restrictions; and road standards (USFWS 2020).

However, despite these conservation measures, the USFWS anticipates that adverse impacts will continue to occur in bull trout habitat (USFWS 2020). Olympic Peninsula steelhead that occur in the same areas as bull trout will likely incur some of these impacts as well. For example, new and temporary road construction, existing road reconstruction, road repairs, log hauling, road grading/blading, and drainage maintenance is anticipated to cause indirect adverse effects to bull trout. These effects include increased fine sedimentation, altered watershed hydrology, reduced water quality, and increased substrate embeddedness (USFWS 2020). The USFWS expects that the Queets and Quinault Core Areas, which overlap with the habitat of Olympic Peninsula steelhead, will continue to be depressed by poor water quality (e.g., up to 408 tons of sediment per year in the Queets River) increased substrate embeddedness, and altered flow regimes caused by logging operations (USFWS 2020).

c. Habitat Conservation Plans

Olympic Peninsula steelhead may benefit from the Washington Department of Natural Resources Trust Lands Habitat Conservation Plan (DNR HCP) and the Forest Practices Habitat Conservation Plan (FPHCP). However, these plans have not been fully implemented and there are conflicting data on whether forest practices are improving aquatic resources. These plans are discussed in the state forest management section.

3. Clean Water Act

a. Overview

The Clean Water Act (CWA), 33 U.S.C. §§ 1251-1387, is the principal federal law regulating water quality in U.S. surface waters. The CWA seeks "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters." 33 U.S.C. § 1251(a). The Act establishes a goal to eliminate all discharges of pollutants into navigable waters by 1985. 33 U.S.C. §§ 1251(a)(1). It also sets an interim goal of achieving, wherever attainable, "water quality which provides for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water" by July 1, 1983. *Id.*, at 1251(a)(2).

The U.S. Environmental Protection Agency (EPA) is the primary federal agency responsible for administering the CWA. EPA implements water pollution control programs, establishes pollution control technology standards for industries, and reviews water quality standards set by states. The U.S. Army Corps of Engineers (Corps) regulates the discharge of certain materials (e.g., fill) in the waters of the United States.

The CWA requires states to establish water quality standards to protect public health and welfare, enhance water quality, and serve the purposes of the Act. States follow a two-step process when setting water quality standards. First, they establish "designated uses" for individual waterbodies, such as the protection and propagation of fish. Next, they set allowable levels of pollutants to protect those uses. 40 C.F.R. §§ 131.10, 131.11. Washington has EPA-approved water quality standards; however, it is not meeting them in many rivers and streams where Olympic Peninsula steelhead occur (WDOE 2016; NIFWC 2020).

The CWA requires states to list "impaired waters," which include segments of waterbodies that do not meet water quality standards. The states must develop Total Maximum Daily Loads (TMDLs) that set the maximum amounts of pollutants that may enter impaired waters without violating water quality standards. The state may distribute those amounts or "loads" to various sources of point and nonpoint sources of pollution. In effect, a TMDL operates as a pollution budget for an impaired waterbody. Nonpoint source pollution drives water quality exceedances in the rivers and streams where Olympic Peninsula steelhead occur.

Washington has not developed, and EPA has not approved, any TMDLs for any waterbodies where Olympic Peninsula steelhead occur.

b. Nonpoint Source Pollution

Nonpoint source pollution is the "leading cause of water pollution across the nation and in Washington." Wash. Dep't of Ecology, Publ'n No. 040-010-009, Enforcement Report on Policy and Trends, 49 (2004). Agriculture causes most nonpoint source pollution; however, that industry is exempt from certain CWA regulations.

A limited number of logging-related activities are subject to the CWA. 40 C.F.R. § 1227.27. The EPA's silvicultural rule defines silvicultural point sources to include "any discernable, confined and discrete conveyance related to *** log sorting *** or log storage facilities which are operated in connection with silvicultural activities and from which pollutants are discharged in waters of the United States." *Id.* § 1227.27(b)(1). The rule defines log sorting and log storage facilities to mean "facilities whose discharge results from the holding of unprocessed wood, for example, logs or roundwood with bark or after removal of bark held in self-contained bodies of water (mill ponds or log ponds) or stored on land where water is applied intentionally on the logs (wet decking). *Id.* § 1227.27(b)(3).

The silviculture rule does not apply to major sources of water pollution caused by logging operations. Among other activities, the rule exempts harvesting operations, surface drainage, and road construction and maintenance from which there is natural runoff. 40 C.F.R. at § 1227(b)(1). In 2013, the U.S. Supreme Court upheld the EPA's decision not to regulate stormwater runoff from logging roads. *Nw. Envtl. Def. Ctr. v. Decker*, 133 S. Ct 1326, 1338 (2013). The following year, Congress amended the CWA to effectively prohibit the EPA from requiring NPDES permits for discharges resulting from several silviculture-related activities, including surface drainage, road construction, and road maintenance. 33 U.S.C. § 1342(l). Therefore, the Clean Water Act fails to adequately protect Olympic Peninsula steelhead.

Many river and stream miles occupied by Olympic Peninsula steelhead will experience water quality impacts caused by logging operations for the foreseeable future (McHenry et al. 1996). The synergistic effect of timber harvesting and heavier rainfall caused by climate change will impair water quality in rivers and streams where Olympic Peninsula steelhead occur.

4. National Environmental Policy Act

The National Environmental Policy Act (NEPA), 42 U.S.C. §§4321 et seq., requires federal agencies to identify and evaluate the impacts of "major Federal actions significantly affecting the quality of the human environment." Thus, the U.S. Forest Service, U.S. National Park Service, Bureau of Indian Affairs, and other federal agencies whose major actions may impact Olympic Peninsula steelhead are subject to NEPA. Major federal actions include those actions "subject to federal control and responsibility." 40 C.F.R. §1508.1(q). To determine whether the impacts of a proposed action are "significant" federal agencies must assess the "potentially affected environment and degree of the effects." *Id.* at § 1501.3(b). Federal agencies must consider the short-term and long-term effects, beneficial and adverse effects, effects on public health and safety, and effects that would violate laws that protect the environment. *Id.*

Federal agencies must prepare environmental impact statements (EISs) for proposed actions with significant impacts. 42 U.S.C. § 4332. An EIS must assess the following topics: (1) the environmental impacts of the proposal; (2) unavoidable adverse effects; (3) alternatives to the proposed action; (4) the relationship between the short-term uses of the environment and maintenance of long-term productivity; and (5) any irretrievable commitment of resources involved if the proposed action is implemented. *Id.* An EIS must also provide a detailed statement regarding the scope of its assessment, which must cover the following subjects: (1) connected or similar actions; (2) a reasonable number of alternatives to the proposed action, including no action alternatives and other reasonable alternatives, as well as mitigation measures; and (3) effects. 40 C.F.R. §§ 1501.9, 1502.14. Federal agencies must release draft EISs for comments from other agencies and the public. 42 U.S.C. § 4332(2)(C).

After undertaking all these analyses and procedures, NEPA does not require federal agencies to select the alternatives with the least impacts on the environment; it simply requires federal agencies to take a "hard look" at them. *Robertson v. Methow Valley Citizens Council*, 490 U.S. 332, 350 (1989). Therefore, the USFS and other federal agencies whose actions may impact Olympic Peninsula steelhead are not required to choose the alternatives that best protects the species. If they follow the procedural requirements of NEPA, agencies may select any alternatives, including those that harm Olympic Peninsula steelhead. Therefore, NEPA is not sufficient to protect Olympic Peninsula steelhead. The DPS requires protection under the ESA.

5. Governmental Failure to Adequately Address Climate Change

Climate change impacts in freshwater and marine habitats pose significant threats to the survival of Pacific salmonids. A critical feature of the contemporary and future threats of climate change is the failure of the US government to address these threats through the adoption of "green" energy policies and actions to drastically reduce reliance on fossil fuel energy generation. These actions are necessary to begin a rapid reduction in the amounts of greenhouse gases that are annually released into the atmosphere.

It is widely recognized in the peer-reviewed scientific literature that the concentration of atmospheric carbon dioxide (CO2) is the primary driver of climate warming. The concentration of CO2 in the atmosphere is currently 412.5 ppm [NOAA Climate.gov https://www.climate.gov/news-features/understanding-climate/climate-change-atmospheric-carbon-dioxide (accessed June 1, 2022)]. This level is the major contributor to the current global energy imbalance. In 2020, von Schuckmann et al. estimated the current global energy imbalance to be 0.87 +/0.12 watts-per-square meter (W m^{-2}) of the earth's surface. "The amount of CO2 in the atmosphere would need to be reduced from 410 to 353 ppm to increase heat radiation to space by 0.87Wm^{-2}, bringing Earth back towards energy balance" (von Schuckmann et al, 2020, p. 2014).

Considering both current policy and the fragmented political climate in the US today, there is a high probability that such a timely reduction in atmospheric CO2 will not occur in time to avoid significant and continued (and perhaps even acceleration of) global warming within the next few decades. This further emphasizes the realism of making climate change predictions based on the IPCC's "business-as-usual", RCP8.5 scenario, which Cheung and Frölicher (2020) note leads to global atmospheric surface warming … of 3.2 C by 2081 – 2100 relative to preindustrial" (Cheung and Frölicher (2020). Further, as explained by von Schuckmann et al. (2020), the current large global energy imbalance coupled to the imbalance that has existed for the past several decades means that considerable warming of the atmosphere as a result of the legacy of heat already stored in marine waters is already locked in for several decades. This would still be true even if the current imbalance were to disappear tomorrow. Therefore, the current failure to adopt green energy policies that would rapidly reduce the global energy imbalance should be understood as a fundamental component of the failure of existing regulatory mechanisms to protect salmon and steelhead and the many components of salmonid ecosystems on which they depend.

6. National Park Service Organic Act and Other Regulatory Mechanisms

The habitat in Olympic National Park has been protected for over 100 years, and it is in relatively pristine condition (Halofsky et al. 2011). In 1988, Congress designated 95 percent of the park as the Olympic Wilderness. The Olympic Wilderness, which was renamed the Daniel J. Evans Wilderness, covers 1,370 square-miles, including 48 miles on the Washington coastline.

a. National Park Service Organic Act

Portions of several Olympic Peninsula steelhead populations spawn and rear inside the Olympic National Park, which provides relatively pristine aquatic habitat conditions compared to areas outside of the park. The park's enabling legislation and the National Park Service Organic Act of 1916 (Organic Act) protect these resources. Specifically, the Organic Act requires the NPS "to conserve the scenery and the natural and historic objects and wildlife therein and to provide for the enjoyment of the same in such manner and by such means as will leave them unimpaired for the enjoyment of future generations." 16 U.S.C. § 1.

b. Fishing Regulations

Olympic Peninsula National Park has its own fisheries management program. The program has three goals: (1) "manage aquatic resources as an important part of the park ecosystem;" (2) "preserve and restore native fishes and their habitats;" and (3) "provide recreational fishing opportunities for the enjoyment of park visitors, consistent with the first two objectives." Except for saltwater areas within park, a Washington State fishing license is not required to fish in Olympic National Park. 36 C.F.R. § 2.3(b). However, a Washington State catch record card is required to fish for steelhead.

All wild steelhead must be released in Olympic National Park. Except during limited periods in select areas, bait fishing is generally prohibited in the park in order to reduce catch and release mortality on steelhead. Studies show that catch-and-release fishing mortality is less than 2% when fish are caught on flies and less than 5-7% overall with most gear types, except for bait, which can have much higher rates of mortality – for both juvenile and adult steelhead – and when fish are caught in elevated water temperatures (Warner and Johnson 1978; Hooton 1987; Pauley and Thomas 1993; Schisler and Bergersen 1996; Taylor and Barnhart 1997; Hooton 2001; Nelson et al. 2005; Twardek et al. 2018). However, as mentioned earlier, the regulations do not account for potential sublethal impacts that can reduce fitness of individuals that survive catch and release (up to ~ 20-25% reduction in fitness: Richard et al. 2013; Bouchard et al. 2021; Papatheodoulou et al. 2021).

The National Park Service has taken emergency actions to protect vulnerable steelhead. For example, in February 2021, the NPS closed recreational fishing for steelhead on the Queets River inside Olympic National Park in response to record low returns (LaBossiere 2021).

c. Roads & Structures

Historically, maintenance and repair of Olympic National Park roads that are adjacent to the Sol Duc, Hoh, Queets, and Quinault Rivers have caused major impacts on fish and aquatic life (Halofsky et al. 2011). Today, the National Park Service takes steps to reduce these impacts. For example, when feasible, the National Park Service relocates roads and other facilities from floodplains to other areas (Halofsky et al. 2011). It also limits construction of new facilities within floodplains to protect fish habitat (Halofsky et al. 2011).

B. STATE

1. Washington Department of Fish and Wildlife

WDFW management of Olympic Peninsula steelhead is guided by the Statewide Steelhead Management Plan, Hatchery and Fishery Reform Policy (C-3624), and harvest management plans with the Quinault, Quileute, Hoh, and Makah Tribes (WDFW 2008, WDFW 2021, Hoh Tribe and WDFW 2020; Quileute Tribe 2020; Quinault Dept. of Fisheries 2021). The continued decline of Olympic Peninsula steelhead with these plans in place shows they are insufficient to adequately manage these populations. Further, the Washington Fish and Wildlife Commission recently replaced its previous hatchery policy, C-3619, with its new hatchery policy, C-3624, which further weakens the standards governing the allowable proportion of hatchery-origin spawners (pHOS) on spawning grounds. The adoption of C-3624 represents a step backwards in addressing the harms posed by hatcheries.

a. Statewide Steelhead Management Plan (2008)

The 2008 Statewide Steelhead Management Plan describes strategies intended to help restore and maintain the abundance, distribution, diversity and long-term productivity of the state's wild steelhead and their habitats to assure healthy stocks (WDFW 2008). To achieve this goal, the plan establishes a series of policies to guide natural production, habitat protection and restoration, fishery management, artificial production, regulatory compliance, monitoring and evaluation, adaptive management, research, and education (WDFW 2008).

The plan provides that "[s]teelhead management shall place the highest priority on the protection of wild steelhead stocks to maintain and restore stocks to healthy levels." (WDFW 2008). To achieve this goal, in part, the policy calls for providing sufficient wild steelhead spawners, which requires setting escapement objectives (WDFW 2008). The plan acknowledges that setting informed escapement objectives requires an understanding of stock population dynamics, habitat conditions, and stock status (WDFW 2008). WDFW maintains a list, which was last updated in 2002, that describes the status of each stock as "unknown," "depressed," "critical," or "healthy." If a stock's status is unknown, WDFW is supposed to "apply a precautionary strategy by implementing low risk fishery and hatchery management regimes." (WDFW 2008). If the status is "depressed," "critical," or ESA-listed, WDFW should promote a trend of increasing wild fish numbers. (WDFW 2008). If the status is "healthy," WDFW should maintain wild steelhead escapement objectives at or above MSH levels (WDFW 2008).

Regardless of whether the nearly two-decades-old "healthy" status designations still apply to certain Olympic Peninsula steelhead stocks - which they do not (e.g., McMillan et al. 2022) - recent (2010-2021) escapement numbers indicate that WDFW is not maintaining escapement levels at or above MSH levels. For example, WDFW did not maintain Queets River wild winter steelhead escapement at or above MSH levels during the last eight out of nine years and have often failed to do so in the Hoh River (WDFW 2022a). Further, they do not monitor or manage wild summer steelhead.

Additionally, the Statewide Steelhead Management plan requires WDFW fish and wildlife managers to establish a network of wild stock gene banks across the state. These gene banks are intended to be located where steelhead were “largely protected from the effects of hatchery programs”. This policy came about from a 2004 review of Washington hatchery facilities by the Hatchery Scientific Research Group, recognizing the need to protect populations from the threat of hatchery effects (HSRG, 2004).

The plan calls for establishing one wild stock gene bank for each major population group (“MPG”) within a larger conservation unit, such as a DPS (WDFW 2008). The plan defines “MPG” to mean:

> “[a] group of populations within a larger conservation unit such as a DPS or ESU that share genetic, life-history, or ecological characteristics that are sufficiently distinct from those of other groups of populations to make conservation or recovery of the group essential for the conservation or recovery of the larger conservation unit.”

(WDFW 2008). This definition implies that more than one MPG should occur within a conservation unit, as an MPG is a subcomponent of a DPS or similar conservation unit.

Fourteen years after the plan was adopted, there is still only one wild steelhead gene bank on the Olympic Peninsula: the Sol Duc Wild Steelhead Gene Bank, which WDFW established in 2013. As a result, since the Snider Creek hatchery was closed in accordance with the Sol Duc gene bank designation. Although this has benefitted Sol Duc steelhead, WDFW should be establishing additional gene banks on the Olympic Peninsula.

Considering the differences between the WRIA 19-21 populations, WDFW should have established more gene banks. There are characteristics that make these groups of populations sufficiently distinct from the group of populations represented by the Sol Duc gene bank designation. For example, WRIA 19 populations ecologically differ from WRIA 20 populations, which includes the Sol Duc population. WRIA 19 populations occur in rain-dominant basins that discharge into the Strait of Juan de Fuca. WRIA 20 populations occur rain-dominant basins – several of which are glacially influenced – that discharge into the Pacific Ocean. The WRIA 21 populations also have characteristics that may make them sufficiently distinct from WRIA 20 populations. Despite these differences, WDFW has not announced plans to designate additional gene banks for any other major population groups within the Olympic Peninsula DPS. Although Cram et al. (2018) recommended establishing additional gene banks in other DPSs, it did not recommend establishing more gene banks within the Olympic Peninsula DPS. Based on Cram et al. (2018), it does not appear that WDFW intends to establish additional gene banks for the Olympic Peninsula steelhead DPS.

b. Hatchery and Harvest Reform Policy

In the years following NMFS's 1996 status assessment of Olympic Peninsula steelhead (Busby et al. 1996), WDFW developed two hatchery reform policies. The original Hatchery and Fishery Reform Policy (C-3619) included several guidelines that, had they been implemented, could have reduced hatchery threats to Olympic Peninsula steelhead. Recently, WDFW adopted a new policy that rolls back these guidelines. As a result, WDFW's hatchery policies have failed to protect Olympic Peninsula steelhead.

On November 6, 2009, the Washington Fish & Wildlife Commission (WFWC) adopted a Hatchery and Fishery Reform Policy that focused on the "scientific and systematic redesign of hatchery programs to recover wild salmon and steelhead and support sustainable fisheries." (Policy C-3619). The policy directed WDFW to follow eleven policy guidelines (Policy C-3619). The first three guidelines were the following instructions:

1. Use the principles, standards, and recommendations of the HSRG to guide WDFW's hatcheries, and to enable adaptive management based on a structured monitoring, evaluation, and research program.

2. Prioritize improved broodstock management to reduce the genetic and ecological impacts of hatchery fish.

3. Develop watershed-specific action plans that implement hatchery reform as part of a comprehensive strategy for meeting conservation and harvest goals. (C-3619).

The 2009 policy was reviewed through the State Environmental Policy Act and within that review committed WDFW to follow a "phased approach" where subsequent reform efforts would be developed and implemented for each hatchery facility, known as Hatchery Action Implementation Plans (WDFW, 2009). To the best of our knowledge these plans were never developed, submitted for state environmental review, or implemented.

In 2018, the Commission suspended these three guidelines for salmon species and directed WDFW to review the hatchery and fishery reform policy and the department's performance of its strategies. Subsequently, WDFW produced two reports: (1) A Review of Hatchery Reform Science in Washington State (Anderson et al. 2020), including independent review by the Washington Academy of Natural Sciences; and, (2) WDFW Hatchery and Fishery Reform Policy Implementation Assessment: Final Report, 2009-2019 (Murdoch and Marston 2020).

i. Review of Hatchery Reform Science in Washington State (Anderson et al. 2020)

As directed by the FWC through the C-3619 review process, WDFW and the Washington State Academy of Sciences conducted a Review of Hatchery Reform Science in Washington

State to identify advances in hatchery reform science that have occurred since C-3619 was adopted in 2009. This thorough WDFW-produced and independently reviewed report provides the following key conclusions:

1. "The HSRG principles of reducing pHOS and increasing pNOB to achieve fitness gains in wild populations are well-founded and should be fundamental goals in any hatchery reform management action.
2. Excessive hatchery program size requires more careful scrutiny and scientific justification because it affects virtually every aspect of hatchery risks.
3. Hatcheries have potential for large magnitude ecological impacts on natural populations that are not well understood, not typically evaluated and not measured.
4. Hatchery risks include fishery risks, ecological risks and genetic risks. Fisheries targeting abundant hatchery runs can unintentionally increase mortality of co-mingled natural populations.
5. Research on ecological [HxW] interactions lags far behind the attention devoted to genetic risks of hatcheries. Importantly, research suggests the potential for ecological interactions in marine environments shared between multiple hatchery and natural populations, yet very little is known about the likelihood or magnitude of population-scale ecological impacts of hatcheries."
6. Studies comparing the number of offspring produced by hatchery-origin fish and natural-origin fish when both groups spawn in the wild (relative reproductive success, RRS) have demonstrated a general pattern of lower reproductive success of hatchery-origin fish.
7. In WDFW's hatchery system, a focus on efficiency and maximizing abundance prevents widespread implementation of risk reduction measures.
8. We recommend a more rigorous, consistent and intentional evaluation of cumulative hatchery effects across multiple hatchery programs operating within a geographic region."
9. WDFW invests considerable effort into population monitoring, yet this information does not often achieve its potential as a hatchery evaluation tool because analysis, reporting, and synthesis are typically underfunded. Furthermore, for many hatchery programs, the absence of a clear framework for application of monitoring data in decision making precludes clearly articulated risk tolerance thresholds."

(Anderson et al. 2020).

ii. WDFW Hatchery and Fishery Reform Policy Implementation Assessment (Murdoch and Marston 2020)

As directed by the WFWC through the C-3619 review process, WDFW also conducted a Hatchery and Fishery Reform Policy Implementation Assessment (2020) to evaluate the policy's effectiveness at protecting wild salmon and steelhead populations (Murdoch and Marston

2020). However, for reasons described in the report beginning on page 3, WDFW found that the monitoring data necessary to answer that question were unavailable or inadequate (Murdoch and Marston 2020). Consequently, WDFW re-focused the assessment to evaluate whether and to what extent the agency had implemented the fishery and hatchery reform actions mandated in the 2009 policy C-3619. Among the many findings in this report, the following WDFW conclusions are particularly relevant to this petition: A lack of funding was a common reason that prevented implementation of some guidelines; a lack of comprehensive statewide monitoring and evaluation program are areas of special concern; and defining program success and collecting and analyzing data to adaptively manage our programs are critical missing components.

WDFW's review also found that the department had made little to no progress implementing seven HSRG recommendations for the Bogachiel Hatchery summer and winter steelhead programs, as required under the first guideline of the 2009 Hatchery and Fishery Reform Policy (Murdoch and Marston 2020). Those recommendations included steps intended to set clear goals for conservation, ensure hatcheries are meeting management goals, and minimize risks to natural populations (Murdoch and Marston 2020). As discussed in the hatchery section of this petition, the winter and summer steelhead programs at the Bogachiel Hatchery did not meet pHOS goals in 2009 (WDFW 2022b). Data for other years are not available on WDFW's online database (WDFW 2022b).

On April 9, 2021, the Commission adopted a revised Hatchery Policy (C-3624), superseding the previous policy (C-3619) (WFWC 2021). The revised policy abandons commitments to follow HSRG guidelines, and it sets forth new direction for WDFW to follow when managing salmon and steelhead hatcheries. Among other things, the policy directs WDFW to prepare Hatchery Management Plans for every WDFW salmon and steelhead hatchery program (WFWC 2021). These HMPs must be based on the best available science regarding the risks of hatchery production on wild salmon and steelhead. However, the HMP provisions must "*** reflect a balance between minimizing genetic and ecological risks to coincident wild populations and providing for the ecological and societal benefits of hatchery propagated salmon and steelhead ***." (WFWC 2021).

The Washington Fish and Wildlife Commission replaced C-3619 with C-3624 without State Environmental Policy Act review, which is the subject of ongoing litigation. *Wild Fish Conservancy et al. v. Washington Dep't. of Fish & Wildlife et al.*, King County Superior Court Docket No. 21-2-13546-0 SEA.

Even after adoption, the new C-3624 hatchery policy is behind schedule on creating the evaluation protocols for the yet-to-be-developed hatchery management plans. Despite the known, ongoing harms that the steelhead hatchery programs are causing to the Olympic Peninsula Steelhead DPS, no changes to relevant hatchery management plans have been proposed. Additionally, previous phased state environmental reviews are being abandoned, starting the decade-long process all over again. It is unclear if, or when, WDFW will release a HMP for its Bogachiel steelhead hatchery operations.

c. Harvest Management Plans

The Quinault, Hoh, Quileute, and Makah Tribes have treaty rights to receive shares of salmon and steelhead as determined in *U.S. v. Washington*, 384 F. Supp. 312, aff'd 520 F.2d 676 (9th Cir. 1975), cert. denied 423 U.S. 1086. Each year, WDFW and the tribes prepare harvest management plans that set guidelines for tribal fisheries, seek to promote wild steelhead conservation, establish monitoring protocols (e.g., sample catches for size, sex, and age), and commit the parties to working together to evaluate fishery mortalities that occur in tribal and non-tribal fisheries (e.g., catch and release mortality, net drop out, and marine mammal predation) (Hoh Tribe and WDFW 2020; Quileute Tribe 2020; Quinault Dept. of Fisheries 2021).

Despite the monitoring protocols in harvest management plans, it is unclear how many steelhead are harvested each year in tribal fisheries, as indicated in Cram et al. (2018).

> "Methods used to estimate treaty harvest and non-retention mortality are not well-documented and do not currently contain estimates of uncertainty. This is a problem particularly in areas where harvest rates are relatively high (e.g., Olympic Peninsula, Grays Harbor, upstream of Bonneville Dam on the Columbia River). Work with tribal co-managers is needed to document methods used to estimate treaty harvest, to test assumptions related to estimations, and to report estimates of uncertainty."

(Cram et al. 2018). Cram et al. (2018) did not explain why previous work with tribal co-managers has failed to fill this data gap.

d. Monitoring

WDFW does not sufficiently monitor Olympic Peninsula winter steelhead, and it does not monitor Olympic Peninsula summer steelhead at all. The monitoring recommendations in Cram et al. (2018) acknowledge that there are significant data gaps. For example, Cram et al. (2018) recommends that WDFW "initiate" monitoring of wild summer-run steelhead in the Olympic Peninsula DPS. Cram et al. (2018) also recommends initiating robust population-scale monitoring in one of more Olympic Peninsula steelhead populations and expanding life cycle monitoring of the DPS.

Likewise, WDFW Hatchery and Fishery Reform Policy Implementation Assessment (Murdoch and Marston 2020) found that a lack of comprehensive statewide monitoring and evaluation program are areas of special concern. Defining program success and collecting and analyzing data to adaptively manage WDFW hatchery programs are critical missing components despite previous WDFW plans that required this data to be collected.

2. Washington Forest Practices Act

The Washington Forest Practices Act (FPA) regulates timber harvest on state and private lands. Wash. Rev. Code §§ 76.09.010-.935. Among other goals, the FPA seeks to "recognize both the public and private interest in the profitable growing and harvesting of timber," and "provide for regulation of forest practices so as to avoid unnecessary duplication in such rules," and "achieve compliance with all applicable requirements of federal and state law with respect to nonpoint sources of water pollution from forest practices." Wash. Rev. Code § 76.09.010(2)(c), (e), (g). The Act established a Forest Practices Board charged with adopting forest practice regulations, and it established a permit system operated by DNR that covers certain forest practices. *Id.* §§ 76.090.030, 76.09.050, 76.09.020(7).

a. The State Trust Lands Habitat Conservation Plan

WDNR manages state trust forestlands under the State Trust Lands Habitat Conservation Plan (the "DNR HCP") (WDNR 1997). The OESF is one of nine planning units under the DNR HCP (WDNR 1997). The DNR HCP includes a Riparian Forest Restoration Strategy that aims to restore salmonid habitat (WDNR 1997). Under that strategy, riparian buffers differ in width based on particular stream needs and disturbance history (WDNR 2016). Generally, the average riparian buffers, which are measured horizontally from the outer edge of the 100-year floodplain, are 150 feet on type 1 and 2 streams and 100 feet on type 3 and 4 streams (WDRN 2016). These buffers are intended to minimize disturbance to unstable banks and adjacent hillslopes and maintain key biological and physical functions. This approach is not designed to achieve a desired future condition for salmonid habitat, but rather to "maintain or aid restoration of riparian functions important to salmonid habitat" (WNDR 2016). In addition to riparian buffers, the HCP includes other habitat protections, including road building requirements and wetland protections (WDNR 2016).

Improved management standards have not increased large woody debris in streams in the OESF. Martens et al. (2019) found that large woody debris was either stable at reduced levels or declining, and that using passive restoration alone is unlikely to increase salmonid productivity.

> "Our results add to the current scientific literature that has found passive restoration of salmonid habitat in the Pacific Northwest is a slow process, which could take an additional 12–70 years for riparian forests and over 50 years for instream wood to accumulate (McHenry et al. 1998; Connolly and Hall 1999; Kaylor et al. 2017)."

(Marten et al. 2019).

The most recent monitoring report for the OESF also found that the majority of LWD is in decay, and that historic logging practices continue to interrupt the supply of new LWD to streams (Devine et al. 2022).

Some research indicates that riparian buffers alone will not be sufficient to meet water quality standards for temperature in the OESF (Pollack et al. 2009). A 2009 study collected water temperature data from 42 subbasins on DNR lands, including 22 readings on tributaries to the mainstem of the Hoh River, 10 on the South Fork of the Hoh River, nine on the Clearwater River, and one small tributary to the Bogachiel River (Pollock et al. 2009). The study found that 17 out of 40 streams had at least one seven-day Average Daily Maximum (7DADM) temperature that exceeded 16.0°C, and thus did not meet water quality standards for temperature in core salmon rearing and spawning habitat (Pollock et al. 2009). One of six subbasins with 25-50% harvest, nine of eighteen subbasins with 50-75% harvest, and seven of nine subbasins with >75% harvest had 7DADM exceedances (Pollock et al. 2009). None of the temperature exceedances occurred in unharvested basins (Pollock et al. 2009).

Pollock et al. (2009) explained that several mechanisms related to logging could be responsible for increased stream temperatures, including widening and shallowing of stream channels, widening above-channel canopy openings, decreasing large woody debris and alluvium, and increasing debris flow frequency (Pollock et al. 2009). If these mechanisms are causing these changes, the authors concluded that "*** reestablishment of riparian forests alone will not be sufficient to return stream temperature regimes to natural conditions" (Pollock et al. 2009).

Other researchers have found different temperature results. For example, Martens et al. (2019) found that 7-day average water temperatures for DNR-managed watersheds were below the 16°C temperature standard. Martens et al. (2019) found that summer stream temperatures have decreased and riparian canopy cover has increased (Martens et al. 2019). Devine et al. (2022) reported that the average 7-day maximum water temperatures on DNR managed lands was 14.4°C, with only 16 exceedances of 16°C out of 329 observations of the 7-day maximum. Devine et al (2022) explained that the cool maritime climate and increased shading influenced water temperatures.

b. Washington State Forest Practices Habitat Conservation Plan

The Washington State Forest Practices HCP ("FPHCP") applies to private lands on the Olympic Peninsula. Although the FPHCP includes habitat protections that benefit aquatic species, NMFS and USFWS noted that forestry activities could still potentially adversely affect aquatic habitat by increasing temperature pollution and sedimentation and decreasing large wood recruitment (NMFS and FWS 2006). Additionally, legal questions have been raised regarding whether the HCP's Clean Water Act assurances violate Washington's antidegradation policy and undermine the TMDL program (Steifel 2013). Federal agencies, including NMFS, have also raised concerns about water temperature, riparian function, and Clean Water Act assurances.

There are ongoing issues with water typing classifications as well. Water typing is fundamental to effectively protect riparian buffers from forest practices, and it can be used in other management contexts such as fish population assessments and harvest management.

Within the context of forest practices, the FPHCP riparian buffer width rules are based on a water type classification system intended to identify Type F (fish-habitat) stream reaches and Type N (non fish-habitat) reaches. The state, however, has been operating under Interim Water Type Rules (WAC 222-16-031) since 1999, which inappropriately allows streams to be classified as Type N based on the results of a single point-in-time electrofishing survey.

NMFS and USFWS have warned WDNR that its water typing practices are not consistent with the FPHCP. In 2015, NMFS and USFWS sent a letter to Washington's Commission of Public Lands, describing WDNR's failure to follow the FPHCP.

> "Since the HCP was signed, too many water type determinations that result in permanent modifications continue not to use physical stream characteristics or models specified by the FPHCP. Water typing decisions based on a demonstration of "fish absence" are not consistent with the permanent approaches agreed to in the FPHC."

(Letter from Kim Kratz, Assistant Regional Administrator, NMFS, and Eric V. Rickerson, State Supervisor, USFWS, to Peter Goldmark, Commissioner of Public Lands, DNR (July 2, 2015) (copy provided). As a result of WDNR's water typing practices, NMFS and USFWS do not believe fish habitat has been correctly identified in a "substantial number of instances" *Id*. Therefore, the Services requested that WDNR consider its water type modifications as temporary rather than permanent until "water typing is better aligned with the original HCP requirements." *Id.*

In addition to water typing concerns, NMFS and USFWS expressed concerns about WDNR's compliance reporting. *Id.* Specifically, the Services stated that "sample sizes are decreasing and reported patterns are disconcerting." *Id.* It is unclear if these compliance monitoring issues have been resolved to the Services' satisfaction.

WDNR is still behind schedule on fulfilling its commitment to "rapidly and iteratively improv[e] the water typing map and model" (NMFS and USFWS 2015). Although WDNR has committed to develop and adopt permanent water type rules that more accurately identify fish habitat as defined in WAC 222-16-010, including recoverable fish habitat, this still has not occurred despite being 16 years into the 50-year habitat conservation plan (WDNR, 2019).

2. Water Pollution Control Act

Washington has established water quality standards for surface waters throughout the state. The purposes of these standards are to protect public health and public enjoyment of Washington's surface waters and to "protect[] *** fish, shellfish, and wildlife ***" pursuant to the Washington Pollution Control Act. WAC 173-201A-010(1).

The WPCA prohibits the Washington Department of Ecology (WDOE) from establishing a permit system for nonpoint source pollution caused by forest practices. RCW § 90.48.420(3). It also prohibits WDOE from fining or penalizing nonpoint sources of pollution so long as the polluters follow forest practice rules. *Id.* WDOE is allowed to review forest practice rules, and WDOE and DNR are required to agree that the rules will allow water quality standards to be met. RCW § 90.48.420(3).

At least one author has suggested that the incidental take permit (ITP) associated with the FPHCP may violate Washington's antidegradation policy, which seeks to "restore and maintain the highest possible quality of the surface waters of Washington." (Stiefel 2013). The policy provides that "[n]o degradation may be allowed that would interfere with, or become injurious to, existing or designated uses ***." Wash. Admin. Code 173-201A-200. Because the ITP authorizes activities that impair fish and aquatic life uses, the ITP expressly violates Washington's antidegradation policy (Stiefel 2013).

3. State Environmental Policy Act

The State Environmental Policy Act (SEPA) requires state and county agencies to identify environmental impacts of proposed projects before committing to a particular course of action. RCW Ch. 43.21C. Under SEPA, state and county agencies must evaluate these impacts, consider alternatives and mitigation measures, and involve the public in decision making. Like NEPA, SEPA is a procedural law; it does not require agencies to select the alternative that causes the least harm to the environment.

4. Fishing regulations

The Olympic Peninsula DPS supports the most popular and intensive recreational fisheries for wild winter steelhead in Washington State (Bentley 2017; Cram et al. 2018), largely because most rivers in Puget Sound are now closed to steelheading in winter (Burge et al. 2006). With little other opportunity, many fishing guides have relocated to the Olympic Peninsula and as such, each year thousands of anglers travel to fish famed rivers like the Sol Duc, Bogachiel, Hoh, Queets, and Quinault Rivers from November through April, which has resulted in very high encounter rates with wild steelhead (e.g., Bentley 2017). Some other smaller populations, including but not limited to the Hoko River, Clallam River, and Sekiu River, also support fisheries, though not to nearly the same extent as the largest populations. Further, nearly all the rivers and streams – both large and small – in the DPS are open to recreational fishing during summer (June – October) when anglers may encounter summer steelhead. Below we discuss how fishery regulations and assumptions by WDFW have potentially helped and hindered the productivity and resilience of wild steelhead.

WDFW has taken steps with fishery regulations to reduce impacts on wild steelhead, including eliminating retention of wild summer steelhead (we could not determine the date when retention was eliminated by WDFW and National Park) and eliminating retention of wild winter steelhead in 2015. Further, in response to the sharp declines in run size the past few years WDFW has shortened or closed fishing seasons for wild winter steelhead and reduced angler effectiveness by eliminating fishing from boats and banning the use of bait (https://wdfw.medium.com/frequently-asked-questions-march-2022-coastal-steelheadclosure-364cfa62826f). Each of these actions may have slowed the decline of steelhead, but these actions, as a whole, have not seemed to reverse the plight of wild steelhead.

However, a lack of prior due diligence and regulatory action by WDFW has also likely contributed to the recent declines and closures. The largest watersheds on the Olympic Peninsula experience high levels of angling pressure that has resulted in very high encounter rates with wild winter steelhead (Bentley 2017). Bentley (2017) evaluated the creel data collected by WDFW to estimate total angler effort and total number of wild steelhead caught and released in the Hoh River in 2014-2015. That analysis indicated WDFW's prior method (referred to hereafter as the "old expansion method") for expanding creel surveys underestimated the total angler effort and catch by a large amount (Bentley 2017). For example, using the old expansion method WDFW estimated 2,580 wild steelhead were caught and released by recreational anglers in 2014-2015, but that number jumped to 4,580 steelhead using the improved model by Bentley (2017). That equates to a 1.77x increase. Total escapement that year was 3,171 steelhead, meaning that every steelhead that escaped to spawn was caught, on average, 1.4 times by anglers (Table 9).

Table 9. Estimated number of wild winter steelhead caught and released (CnR) by recreational anglers in 2014-2015 in relation to escapement goal using the old expansion method by WDFW (old method) and using the new model developed by WDFW (Bentley et al. 2017) in the Hoh River and Quillayute River system excluding the Dickey River, which does not get creel surveys. Bentley et al. (2017) only applied their model to the Hoh River in 2014-2015 and based on that analysis, the old method underestimated catch by 1.77 times. We applied that multiplier to estimates of catch using the old method in the Bogachiel, Calawah, and Sol Duc Rivers (summed as the "Quillayute system") for the 2014-2015 season to generate a rough estimate of potential recreational catch using assumptions in Bentley et al. (2017).

Population	Escapement	Old Method Total wild steelhead CnR	Bentley et al. (2017) 1.77 multiplier	Percent of escapement CnR	
				Old method	1.77 multiplier
Hoh	3171	2580	4580	0.8	1.4
Quillayute system	*7914*	*5277*	*9340*	*0.7*	*1.2*
Calawah	3081	576	1019	0.2	0.3
Bogachiel	1315	1893	3350	1.4	2.5

Sol Duc	3229	2808	4970	0.9	1.5

Bentley (2017) did not evaluate any other years or streams but did suggest re-evaluating the prior year's assumptions using the new method. Unfortunately, WDFW has not done that so far, and therefore, they do not have a rigorous estimate of total encounter rates outside of the single year analyzed by Bentley (2017). To help better understand the level of encounters in the neighboring Quillayute River system, we applied the 1.77x correction to the old expansion totals generated by WDFW for the Bogachiel, Calawah, and Sol Duc Rivers, and then summed those for the entire system (Table 9). To be clear, Bentley (2017) did not come up with a "correction factor" that can be applied to other populations. Rather, we did this only as a means of roughly gaging how much WDFW has underestimated total numbers of fish caught and released. The encounter rates rose sharply when we applied an expansion, but even the old method suggested very high encounter rates in the Bogachiel and Sol Duc Rivers, which are more easily accessed than the Calawah River (Table 9). If those estimates are in the general ballpark, then encounter rates in the Bogachiel and Sol Duc River are, like the Hoh River, very high.

It is difficult to estimate what effects such encounter rates could have. WDFW assumes a fairly conservative 10% mortality rate with each encounter, which is higher than the typical 3-5% rate found in many studies (Hooton 1987; Taylor and Barnhart 1997; Hooton 2001; Nelson et al. 2005; Twardek et al. 2018). Nonetheless, they don't account for fish being caught multiple times, and they do not account for potential sublethal impacts associated with catch and release. As discussed earlier, emerging research on Atlantic salmon indicates there can be significant sub-lethal impacts that can alter the migration and reduce the reduce fitness of adult females (up to ~ 20-25%) that are caught and released (Richard et al. 2013; Bouchard et al. 2021; Papatheodoulou et al. 2021). If sublethal impacts are similar in steelhead, then those impacts, combined with the high encounter rates could be far more detrimental to productivity than a simple assumption of a 10% mortality.

There are two other areas of notable concern. First, the fishery for summer steelhead is not monitored and while WDFW has enacted emergency regulations for other stocks, including closures, to protect depleted run sizes of wild winter steelhead (WDFW emergency regulations, accessed online: https://wdfw.wa.gov/news/state-announces-full-closure-coastal-steelhead-fishing-support-conservation-following) and summer coho salmon (WDFW 2021 emergency regulations, accessed online: https://wdfw.wa.gov/sites/default/files/about/regulations/filings/2021/2021_nof_statewide_rec_ces_final_7_1_2021.pdf), they have done nothing to help conserve wild summer steelhead. Although WDFW does not monitor summer steelhead, they have been aware of the snorkel counts in Brenkman et al. (2012) for well over a decade. Even low encounter rates by recreational anglers could have impacts on the struggling populations, particularly because mortality increases when water temperatures are warmer and sublethal impacts are not accounted for. For instance, Barnhardt and Taylor (1996) found mortality greatly increased when water temperatures reached and exceeded 21°C. While it is rare for the Hoh and Queets

Rivers to reach 21°C during summer, such temperatures are fairly common in sections of the Calawah (John McMillan, The Conservation Angler, unpublished instantaneous water temperature data 2000-2006; 2009-2021) and Sol Duc Rivers (Sol Duc stream and temperature gage: https://apps.ecology.wa.gov/ContinuousFlowAndWQ/StationDetails?sta=20A070).

Second, it is unclear how recreational angling is impacting juvenile steelhead. Juvenile steelhead are found throughout a variety of main-stem and tributary habitats (McHenry et al. 1996; Smith 2000; McMillan and Starr 2008; McMillan et al. 2013), most of which are open to angling during the summer months. It is possible that many juvenile steelhead are captured each season by anglers, though at this time we are nor sure to what extent and how such encounters could impact survival. Perhaps more worrisome is the potential for high encounter rates on older, larger steelhead parr and steelhead smolts that are found in the main-stem rivers during the winter steelhead season – from January – April. If juveniles are killed during summer there is potential for compensation by other individuals, but if smolts are killed during their journey downstream, there is almost no chance for compensation. Personal fishing experiences and observations (John McMillan, The Conservation Angler) indicate many juvenile steelhead are captured on steelhead gear in winter, and smolts are commonly captured during the spring when they migrate to the ocean. Because they are non-target fish and the gear is for larger fish, lethal hookings appear more common and smolts are often handled without care. Given the estimated smolt to adult survival of Olympic Peninsula steelhead, a river like the Hoh may only be producing 30,000-40,000 smolts per year. Anglers reported catching and releasing over 4,500 adults in the Hoh River in 2014-2015 (Bentley 2017). Anglers likely encounter more smolts than adults, perhaps substantially so because they are feeding. Understanding these impacts, although challenging, seems important because of the sheer number of anglers and their potential for catching and releasing large numbers of juvenile steelhead. At this point, given the declines in the populations, all smolts are important to productivity and resilience.

V. OTHER NATURAL OR ANTHROPOGENIC FACTORS

A. Hatcheries

In 1996, NMFS acknowledged the genetic risk that hatcheries pose to wild steelhead on the Olympic Peninsula (Busby et al. 1996). At the time, around 40,000 summer steelhead, primarily out-of-DPS Skamania stock, were annually released in the Quillayute River Basin (Crawford 1979; WDF et al. 1993); 840,000 winter steelhead, primarily from Bogachiel/Chambers/Cook Creek stock, were annually planted; (WDF et al. 1993; WDFW 1994a); and winter steelhead of mixed origin were being released into the Quinault River (Busby et al. 1996). The proportion of natural spawning winter steelhead composed of hatchery-origin recruits (pHOS) ranged from 16% in Quillayute River to 44% in Quinault River (Busby et al. 1996).

In the twenty-five years following the 1996 Status Review, additional studies have discussed the genetic and ecological harm hatchery steelhead programs cause to wild steelhead populations (e.g., Goodman 2005, Araki et al. 2008; Araki et al. 2009; Araki and Schmid 2010;

Chilcote 2003; Naish et al. 2007; Chilcote et al. 2011; Kostow et al. 2003; Kostow and Zhou 2006; Kostow 2009). Considering these risks, state, and federal agencies, as well as the Hatchery Scientific Review Group, have raised concerns about the effects of steelhead hatchery production on the Olympic Peninsula (HSRG 2004, 2015; USFWS 2009; Duda et al. 2018; Cram et al. 2018). And research by McMillan et al. (2022) indicates hatchery fish, both directly and indirectly, have likely contributed to the depletion of early returning wild winter steelhead. Below we summarize what is known about the extent of hatchery operations in the DPS, straying of hatchery steelhead, and potential ecological and genetic impacts posed by hatchery fish.

1. Hatchery Operations on the Olympic Peninsula

Hatchery steelhead have been released into Olympic Peninsula rivers for over a century (Duda et al. 2018). Between 1933 and 2014, a total of 9.2 million winter steelhead and 1.6 million summer steelhead were released into the Quillayute River system (Duda et al. 2018). Between 1958 and 2014, a total of 4.1 million winter steelhead and 0.9 million summer steelhead were released into the Hoh River (Duda at al. 2018). And, between 1962 and 2010, a total of 3.01 million juvenile winter-run steelhead were planted in WRIA 19 streams (NOPLE 2015).

The largest releases have occurred in WRIA 21, primarily on the Quinault and Queets Rivers (Table 1). From 1915 to 2014, 26.2 million winter steelhead were released into the Quinault River, and from 1974 to 2014, 5.2 million winter steelhead were released into the Queets (Duda et al. 2018). The Quinault River program continues to release close to half-a-million steelhead every year: 395,606 (2015), 452,482 (2016), 487,190 (2017), and 475,488 (2018) (QDF and WDFW 2021).

As of 2013, eighteen hatchery release operations (on-site and off-site) were ongoing on the following rivers: Calawah River (summer: Figure 24, and winter steelhead), Clallam River, Goodman Creek, Hoh River, Hoko River, Lower Quinault River, Lyre River, Pysht/Independents, Queets River, Quillayute/Bogachiel River, Sail River, Sekiu River, Sooes/Waatch Rivers, Sol Duc Rivers (terminated all hatchery steelhead releases after 2013), and Upper Quinault River (Table 1; Cram et al. 2018). Between 2000 and 2008, these programs released an average of 1,383,024 hatchery steelhead per year (Cram et al. 2018.) As a result of some off-site release program reductions that occurred between 2009 and 2013, the average annual release has reduced to 1,072,781 (Cram et al. 2018). Population-specific data from Cram et al. (2018) are provided in Table 1.

These programs are operated for harvest augmentation and mostly use early returning Chambers Creek winter stock and early returning Skamania summer stock, neither of which are native to the Olympic Peninsula DPS (Cram et al. 2018). It is unclear if the Quinault programs are segregated (Cram et al. 2018). In 2013, an integrated hatchery program using early returning wild broodstock was initiated on the Bogachiel River, and summer and winter steelhead hatchery releases were terminated on the Sol Duc River (Table 1 and Figure 24; Cram et al. 2018)

a. Straying

Hatchery steelhead released within, and outside of, the Olympic Peninsula DPS boundaries are known to stray into the DPS's river and streams (Cederholm 1993; Houston and Contor 1984; Phelps et al. 1997; McMillan 2006; USFWS 2009; Brenkman et al. 2012; Kassler et al. 2011; Cram et al. 2018). WDFW acknowledges this threat to the genetic integrity of Olympic Peninsula steelhead (Cram et al. 2018). As a result of past and current hatchery practices, the Sitkum River summer steelhead population may be the only population of Olympic Peninsula steelhead that has escaped large levels of hatchery straying (Table 7). Almost no information is available for straying of hatchery winter steelhead, if only because it is difficult to discern which populations the fish came from, and because in places like the Queets River, the hatchery steelhead are not marked. The lack of marking for hatchery winter steelhead in the Queets, and one of the hatchery stocks in the Quinault River, is of great concern because it is possible those fish and their feral offspring are masking (Quinones et al. 2013; Willmes et al. 2018) potentially greater declines in wild winter steelhead than is currently thought. Not marking hatchery fish also undermines state selective fishing regulations that are intended to protect wild fish from harvest while reducing pHOS by encouraging harvest of hatchery fish.

Snorkel survey data suggests there is a substantial amount of straying by hatchery summer steelhead into watersheds that do not have any releases of hatchery summer steelhead (Table 7). For example, surveys by Brenkman et al. (2012) from 2005-2010 found a mean percent origin hatchery adult of 40% in the SF Hoh River, 16% in the EF Quinault River, and 43% in the NF Quinault River (Table 7). In some years, there were so few wild summer steelhead and so many hatchery summer runs that the annual proportion of hatchery adults ranged up to 76% and 100% in the SF Hoh and NF Quinault Rivers (Table 7). McMillan (2022) reported similar results as Brenkman et al. (2012) in the SF Hoh River, although over a longer period (Figure 9), with a mean percent hatchery steelhead of 41% and an annual peak of 67% (Table 7). Although it is possible some hatchery steelhead moved downstream in the Quillayute River system and may not have spawned in the upper watershed, the surveys were purposely conducted late in the season to minimize that behavior (McMillan 2022). Regardless, if we translate the proportion of hatchery adults in late summer and early fall into pHOS, the estimates easily exceed the HSRSG's recommended pHOS limits of 5%-10% for segregated programs (WCSSP 2013; HSRG 2015.

Snorkel survey data also suggests hatchery steelhead are straying within the Quillayute River system outside of the release location in the lower Calawah River. Brenkman et al. (2012) reported a mean proportion of hatchery steelhead of 13% in the upper Bogachiel River (Table 7), while McMillan (2022) reported a mean of 33% in the NF Calawah River with an annual range of 0% up to 100% (Table 7). In contrast, relatively few hatchery steelhead have been documented in the Sitkum River and upper SF Calawah River, where most of the spawning is believed to occur (Table 7). The most likely explanation is the series of numerous waterfalls that must be ascended to reach the upper Sitkum and SF Calawah Rivers (McMillan 2006; McMillan 2022). Most steelhead ascend the falls from late summer through early fall after the first freshets in October (McMillan 2022). It is likely that many summer steelhead are not capable of ascending the falls and others are not able to capably time their passage with the appropriate weather window.

As of 2013, weirs and other adult traps intended to remove hatchery steelhead were absent on multiple Olympic Peninsula rivers (Cram et al. 2018). Specifically, as of 2013, there were no weirs or traps on the Calawah River, Goodman Creek, Hoh River, Lyre River, Pysht/Independents, Sail River, or Sekiu River (Cram et al. 2018). It is unclear if weirs or traps were in place on the lower or upper sections of the Quinault River in 2013 (Cram et al. 2018); it is unclear if they are there now.

Even where weirs and adult traps are in place, they may not sufficiently protect wild populations (Seamons et al. 2012). They may not be complete barriers and they do not prevent hatchery fish from spawning elsewhere in a watershed (Quinn 1993; Dittman et al. 2010).

Overall, straying hatchery summer steelhead combined with low abundance of wild summer steelhead would appear to represent a great risk to the genetic integrity and diversity of the wild populations, particularly in the Hoh, Quinault, and NF Calawah Rivers. Further, there is almost no information on how many of those hatchery summer steelhead remain in the system and spawn with wild winter runs. The uncertainty and potential for strong negative hatchery impacts underscores the need for an ESA listing for the Olympic Peninsula steelhead DPS.

b. Genetic and Ecological Risks

Concerns regarding interbreeding between hatchery and wild steelhead on the Olympic Peninsula have been raised for nearly thirty years, if not longer. In the early 1980's, Cederholm (1983) raised concerns about the large number of hatchery smolts released in the Quinault, Queets, Hoh, and Quillayute Rivers. Cederholm (1983) noted there was a high degree of within and between river straying and that there may be interbreeding with wild stocks, which could change the "long-term, spawning, timing, growth and survival of wild fish." (Cederholm 1983). The record shows Cederholm was correct, and in fact, subsequent snorkel surveys (mentioned in previous section) indicate this is also a significant problem for wild summer steelhead.

NMFS expressed similar concerns in 1996 (Busby et al. 1996). In its stock assessment, the BRT mentioned there was "widespread production of hatchery steelhead" within the DPS, which presented genetic risks despite management efforts to minimize genetic introgression (Busby et al. 1996).

In 2004, the Hatchery Scientific Review Group (HSRG) raised several concerns regarding interbreeding on the Olympic Peninsula (HSRG 2004). Specifically, the HSRG identified interbreeding between wild and hatchery steelhead as a concern for Quillayute River winter and summer steelhead populations and the Hoh River winter steelhead population, and it recommended the termination of multiple off-site release programs (HSRG 2004). Several of those off-site release programs have been terminated to avoid genetic and ecological risks. For example, the Quinault Tribe discontinued transfers Quinault NFH winter steelhead to the Hoh River and Cook Creek. (https://www.fws.gov/Quinaultnfh/Hatchery.cfm).

The Washington Coast Sustainable Salmon Plan (2013) indicates that previous proportions of hatchery fish spawning with natural origin spawners likely exceeds the HSRG's recommended pHOS limits of 5%-10% for segregated programs (WCSSP 2013; HSRG 2017). The report includes the following estimated percentages of natural winter steelhead spawners by river system: Sooes/Waatch: 50-74%; Ozette: 75-94%; Quil/Bogie: 75-94%; Dickey: 95-100%; Sol Duc: 75-94%; Calawah: 75-94%; Goodman Creek: 50-74%; Mosquito Creek: 75-94%; Hoh River: 75-94%; Klalaloch Creek 95-100%; Queets River: 75-94%; Clearwater River 95-100%; Raft River: 75-94%; Quinault/Lake Quinault (75-94%); Quinault River (50-74%); Moclips River (95-100%); and Copalis: (95-100%) (WCSSP 2013). Based on these estimates, genetic risk would be increased on all systems except Kalaloch Creek and the Dickey, Moclips, and Copalis Rivers.

In 2008, WDFW reported that introgression resulting from the release of Chambers Creek winter steelhead may have occurred in the Pysht/Independent, Hoko River, and Sol Duc River populations (WDFW 2008). The report describes Chambers Creek winter and Skamania River summer steelhead as "pos[ing] substantial risk to both the among-population diversity and the fitness of natural steelhead populations" (WDFW 2008). Because of introgression, the report estimated that the Pysht/Independents, Hoko, and Sol Duc winter steelhead populations had "high" reductions in diversity (WDFW 2008).

> "A limited amount of information was available to evaluate changes in spatial structure and diversity in the Olympic Peninsula region. Most notably, we compared the 1993 genetic characteristics of the Pysht Winter and Hoko Winter steelhead populations with samples collected in 1973. The analysis indicated a 5.5-14.5% gene flow (modal value of 9.5%) from hatchery-origin, Chambers type stock to the Hoko natural population; a 12-75% gene flow (modal value of 26.5%) to the Pysht natural population; and a 2.5-6% (modal value of 4%) to the Sol Duc winter natural population (see Chapter 4, Artificial Production)."

(WDFW 2008). WDFW did not review gene flow samples from other wild steelhead populations for the 2008 report. Fortunately, in 2012, WDFW stopped releasing hatchery winter and summer steelhead in the Sol Duc River (Cram et al. 2018), but to date, we are not aware of data on potential hatchery effects before and after cessation of hatchery releases.

In 2009, WDFW reported pHOS measurements related to the operation of its hatchery program on the Bogachiel River (*See* https://fortress.wa.gov/dfw/score/score/hatcheries/hatchery_details.jsp?hatchery=Bogachiel%20Hatchery). WDFW did not meet its 5% and 10% pHOS goals for winter and summer steelhead, respectively. WDFW reported a 9% pHOS rate for winter steelhead and a 23% pHOS rate for summer steelhead (WDFW Cons. 2021).

Ecological impacts of hatchery programs may occur through a variety of pathways, and the effects – ranging from predation to competition for food and space – have been reviewed extensively (Einum and Fleming 2001; Weber and Fausch 2003; Kostow 2009; Tatara and Berejikian 2012; Rand et al. 2012). Given the high pHOS levels in many of the largest populations, and the ability of those fish to produce some feral offspring and interbreed with wild fish, there is heightened potential for those feral and hybrid juveniles to compete with wild steelhead for limited food and habitat, and negatively influence declining and depleted populations of wild steelhead.

Outside of freshwater, there is a growing body of evidence that large-scale releases of hatchery pink and chum salmon are negatively impacting growth and survival of other salmonids in the North Pacific (Levin et al. 2001; Ruggerone et al. 2010; Ruggerone and Connors 2015; Ruggerone and Irvine 2018). A recent presentation on the subject found hatchery pink salmon negatively impacted the growth and survival of steelhead in the Thompson River and Chinook salmon in Puget Sound and the Columbia River (Ruggerone et al. 2021). It is therefore possible that wild steelhead in the Olympic Peninsula DPS, particularly populations with older fish that migrate further into the North Pacific, are also being impacted by competition with pink and chum salmon.

c. Hatchery Fishery Structure and Erosion of Adaptive Capacity

As we described earlier, Cram et al. (2018) noted the risk of fishery selection on run timing in wild winter steelhead since the number of fishing days per week for treaty fisheries is highest early in the season to target greater harvest on hatchery adults. However, it should be noted that the recreational fisheries operated in the same manner for decades before hatchery steelhead were marked, and as such, both wild and hatchery steelhead were harvested at high rates (McMillan 2006). Further, while it is now catch and release only for wild winter steelhead, individuals that enter earlier in the season and remain in the system for longer periods are more likely to be caught multiple times, and such effects are unknown (Hooton and Lirette 1986). Owing to this combination of factors, and others, it appears the hatchery fishery structure has contributed to the depletion of early returning wild steelhead and as a result, run timing is more compressed than it was historically (McMillan et al. 2022).

This is a concern because run timing is highly heritable in salmonids and it represents a mechanism through which fish can adapt to changes in stream flow and water temperature (Manhard et al. 2017; Tillotson and Quinn 2018). As outlined in this document previously, stream flows and water temperatures are predicted to become flashier and warmer, respectively (Wade et al. 2013). Winter steelhead enter and spawn earlier in the winter and spring in more southerly areas of their native range (Busby et al. 1996), which suggests wild steelhead on the Olympic Peninsula will need to rebuild the front end of their run timing to keep pace with climate change.

Environmental conditions on the Olympic Peninsula have changed in the past forty years and they will continue to shift in relation to climate effects. For example, early winter peak flows are expected to become more intense and summer streamflows and temperatures will occur earlier and for longer durations (Wade et al. 2013). To take advantage of higher streamflows earlier in the winter and to ensure they get their juveniles out of the gravel before the onset of summer base flows and peak temperatures, steelhead will need to increasingly enter and spawn earlier. This will not be possible if the current hatchery management and fishery framework remains in place, because those impacts are essentially blocking the potential for adaptations in migration timing. Hence, the population could become increasingly less productive if the fishery and hatchery practices continue to select for later entering and spawning adults.

Of course, run timing is only one trait that is likely to be under selection as climate effects unfold. Genetics are a complex field and numerous other traits will also likely need to change.

> "Many traits appear to have responded to recent climate change, apparently without genetic adaptation. However, to keep pace with climate change, genetic adaptation may be necessary in the long-run; thus maintaining genetic diversity within DPSs and species as a whole is a high priority for salmon conservation."

Crozier et al. (2019). *See also* (Pitman et al. 2020).

Although steelhead are highly adaptable, they have their limits (Wade et al. 2013) and it is uncertain whether Olympic Peninsula steelhead can adapt quickly enough to climate change (Halofesky et al. 2011). This concern is particularly heightened for early returning wild winter steelhead and wild summer runs, the latter of which may no longer have the necessary genetic and phenotypic diversity to sustain fitness and productivity in the coming years and decades.

B. Ocean Conditions

Correlations exist between recurring, decadal-scale variability (including the Pacific Decadal Oscillation and the El Nino Southern Oscillation) and salmonid abundance in the Pacific Northwest (Stout et al. 2012). When ocean conditions are unfavorable for salmonids, their abundance declines. Historically, salmonids have persisted through shifts in ocean productivity. However, it is uncertain whether salmonids can withstand these shifts in marine conditions on top of climate change, habitat loss, and other increasing threats.

C. Loss of Salmon Nutrients

The loss of marine-derived nutrients from other salmonids in Olympic Peninsula rivers and streams is likely limiting steelhead productivity (Minkova et al. 2021; Bilby et al. 1996, 1998; NOPLE 2015; McMillan 2006; Halofsky et al. 2011). For example, McMillan (2006) estimated that the combined loss of pink, chum, and coho in the Quillayute River system has resulted in 87,411 - 201,761 fewer salmon as compared to early 20th century numbers, a significant reduction in marine nutrients. As a result of losing these nutrients, streams are less able to produce steelhead smolts that spend an average of two years instream before outmigrating (McMillan 2006). The same is likely true for all other Olympic Peninsula rivers that historically supported significant numbers of salmonids.

EFFECTIVENESS OF CONSERVATION ACTIVITIES BY STATES AND OTHER PARTIES

State, local, and tribal governments, as well as nonprofit organizations, have invested significant resources in recovering steelhead habitat on the Olympic Peninsula. For example, the Wild Salmon Center, Western Rivers Conservancy, The Nature Conservancy, and WDNR have purchased thousands of acres of habitat along the Hoh River to protect migration, rearing, and spawning habitat (McMillan 2006; McMillan and Starr 2008). This is in addition to many other projects to restore habitat, which can be reviewed on the Washington Governor's Salmon Recovery Office website: https://srp.rco.wa.gov/projectmap?mlayer=projects.

WDNR administers a Family Forest Fish Passage Program that assists private landowners with fish passage improvements. Over seventeen years, the program has completed 67 projects on the Olympic Peninsula (WDNR 2020). WDNR also administers a River and Open Space Program that purchases habitat for conservation purposes (WDNR 2020). Since the program's inception, it has funded 16 conservation easements that protect 1,043 acres, although it is unclear if any of those easements occur within Olympic Peninsula steelhead habitat.

Since 1999, the Pacific Coast Salmon Recovery Fund has distributed $12.6 million in funding to habitat projects throughout Washington's coast. This federal spending leveraged an additional $33 million in federal and state funding for these projects.

Although habitat has likely improved in the limited areas where habitat restoration has occurred, the results – as measured by increased abundance, productivity, spatial distribution, and diversity – are unclear. For example, Bilby et al. (2022) reported that habitat restoration in Deep Creek watershed produced mixed results in response to habitat restoration. Additionally, Bilby et al. (2022) indicates that land use practices continue to interfere with restoration efforts.

> "Historical and ongoing land-use practices (road construction, logging) continue to influence the effectiveness of the restoration activities. Mass-wasting events and avulsions can dramatically change or negate the effects of restoration. Equilibration of the systems will take decades or longer to occur. Riparian recovery in particular will take centuries."

(Bilby et al 2022).

To make restoration projects effective, there need to be enough fish to utilize restored habitat. As McMillan (2006) explained, “habitat purchases made to recreate functioning salmon and steelhead ecosystems are rendered ineffective without sufficient numbers of the key species that drive them.”

REQUEST FOR CRITICAL HABITAT DESIGNATION

The Petitioners request the designation of critical habitat for Olympic Peninsula steelhead concurrent with listing. Critical habitat should encompass all known and potential freshwater spawning and rearing areas, migratory routes, estuarine habitats, riparian habitats and buffers, and essential near-shore ocean habitats.

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APPENDIX A

Summary of Bayesian analysis of Hoh and Quillayute wild winter-run steelhead spawner-recruit analyses from which Figures 21 and 22 were created

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June 17, 2022

Spawner and recruit data for each river were provided in the two attached Excel files received by Wild Fish Conservancy (WFC) from WDFW in May 2022. *See* Gayeski Bayesian Analysis data folder provided in submitted materials.

Sheet "Recruits Per Spawner 1" of the file "Quilayutte_Winter_Steelhead_data.xlsx, columns C and G (rows 6 to 43) contained spawner and recruit data for brood years 1978 to 2015 calculated by WDFW staff and was taken at face value. The corresponding data file used in the Stan analysis is named "QuilBY78-15.txt".

The data set for Hoh wild winter run steelhead received by WFC ("Hoh Stlhd broodyear return by age.xlsx") did not include recruits calculated from the cohort analysis (worksheet "Cohort Analysis") so I calculated maiden recruits from the cohort analysis data (Gayeski 2022). This is contained in workheet "R and S" of the file "Hoh Stlhd broodyear return by age_NG_June_2022.xlsx" columns BI and BJ (rows 6 to 32). The corresponding data file used in the Stan analysis is named "HohBY87-13.txt".

The text files "Printfit_Hoh steelhead_RickerRW May 9 2022.txt" and "Printfit_Quil steelhead_RickerRW May 9 2022.txt" contain the summary output of the Bayes analyses conducted in program Stan (Gayeski 2022a)

The Ricker random walk in alpha model treats the natural logarithm of the Ricker alpha as undergoing a random walk with the capacity parameter beta treated as a constant (estimated from the data as part of the fitting of the model). In fitting each of the two data sets, broad uniform prior distributions were placed on beta (between 1000 and 20000), sigmaw, the standard error of the annual random walk (between 0 and 2.0), and sigma, the population-and-time (brood year)-specific process error of the R/S relationship. A normal prior in natural log space was placed on log alpha (mean = 1.5078, standard deviation = 0.3486), which confines alpha to the interval [1.8, 11].

Four chains of 10000 replicates each were run, the first 5000 samples of which were treated as "warmup" and the last 5000 retained as samples from the posterior distribution of the parameters, for a total of 20,000 posterior samples of each parameter. Both models converged rapidly. Trace plots of log alpha, beta, sigmaw, and sigma showed coherent sampling of the parameter posteriors, and Rhat values were all equal to 1.00+ indicating convergence.