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Olympic Peninsula Steelhead DPS Watershed Summaries: Appendix A to NOAA Technical Memorandum NMFS-NWFSC-198

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National Oceanic and Atmospheric Administration
National Marine Fisheries Service
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Olympic Peninsula Steelhead DPS Watershed Summaries

Strait of Juan de Fuca watersheds

Salt Creek

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

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

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

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

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

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

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

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

Lyre River

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

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

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

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

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

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

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

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

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

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

East Twin River

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

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

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

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

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

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

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

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

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

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

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

West Twin River

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

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

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

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

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

Deep Creek

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

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

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

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

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

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

Pysht River

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

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

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

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

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

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

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

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

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

Clallam River

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

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

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

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

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

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

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

Hoko River

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

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

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

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

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

Sekiu River

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

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

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

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

Westside watersheds

Quillayute River Basin

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

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

The climate on the western portion of the Olympic Mountains is temperate, with an average annual precipitation of 350 cm between 1980 and 2010 (Jaeger et al. 2023), most of which occurs between November through March as rain or snow events (Jaeger et al. 2023). Peak flows, for example, in the Calawah River are greatest during late fall/early winter months (November, December, and January), when ~80% of all peak flow events have occurred between 1975 and 2021[5]. Peak flows in the Calawah River during those years average approximately 661 cms [6]. Monthly low-flow in the Calawah, as well as the other watersheds within the Quileute, occur during August or September. Average low flow in August and September in the Calawah River is approximately 5 cms[7]. Spring snow melt occurs during the months of April through June (Jaeger et al. 2023). Summer low flows has been considered a general limiting factor to salmon and steelhead production in the Quileute river basin (Williams et al. 1975). Summer low flows have decreased over time in the Calawah River basin, where the average low flow in the in late 1970s through the 1990s was 2.0cms, while in the 2000s average summer low flow has been 1.5cms[8]. Summer low flows are predicted to decrease anywhere between 5% and 43% by 2040 in the Quillayute River Basin (Wenger et al. 2010, USBOR 2014, USFS-OSC 2022). The largest changes are predicted to occur in the Sol Duc, Upper Bogachiel, and Quillayute River proper (Wenger et al. 2010, USBOR 2014, USFS-OSC 2022). Peak flows have slightly increased from 1975 to 2010 from a decadal average of 585cms in the 1980s to 721cms average during the first decade of the 2000s[9] (Jaeger et al. 2023). In both the Calawah and Bogachiel rivers, it is becoming common for peak flows to be at or above flood stage. These trends could threaten salmon habitat and other aquatic ecosystem functions (NWIFC 2020).

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

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

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[5] https://nwis.waterdata.usgs.gov/nwis/peak?site_no=12043000&agency_cd=USGS&format=html
[6] https://nwis.waterdata.usgs.gov/nwis/peak?site_no=12043000&agency_cd=USGS&format=html
[7] https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&search_site_no=12043000&format=sites_selection_links
[8] https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&search_site_no=12043000&format=sites_selection_links
[9] https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&search_site_no=12043000&format=sites_selection_links

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

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

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

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

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

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

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

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

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

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

Hoh River

The Hoh River originates at the Hoh Glacier on Mt. Olympus and flows approximately 90 kilometers to the Pacific Ocean (Williams et al. 1975). The 770 km^2 watershed includes portions of the Olympic National Park, the Olympic National Forest, state and private timberlands, and the Hoh Tribal reservation. It includes over 350 km of anadromous stream habitat, approximately 89 kilometers of mainstem and the rest in tributaries (Williams et al. 1975). The amount of floodplain habitat is also large but the precise amount has not been quantified. Besides the Hoh Glacier, the drainage includes several other glaciers including the Blue, White, Hubert, and Ice River Glaciers (Williams et al. 1975), all of which serve to sustain summer streamflows. Some notable tributaries include the South Fork Hoh River, Winfield Creek, Elk Creek, Owl Creek, Anderson Creek, Braden Creek, Willoughby Creek, Alder Creek, Pins Creek, and Hell Roaring Creek (Williams et al. 1975). Channel wetted widths along the mainstem Hoh River range between 13 and 27 meters during the summer and 14 to 36 meters in the winter (Williams et al. 1975).

The Hoh watershed has the highest precipitation levels in Washington State (U.S. Weather Bureau 1965, NWIFC 2020). Average annual precipitation ranges from about 225 cm near the Pacific Coast to 600 cm in the Olympic Mountains (U.S. Weather Bureau 1965). Peak flows in the Hoh River are greatest during winter months (e.g., November to February), and average approximately 985 cms[10]. Monthly low-flow typically occurs in August or September, averaging approximately 33 cms[11].

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

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

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

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[10]https://nwis.waterdata.usgs.gov/nwis/peak?site_no=12041200&agency_cd=USGS&format=html
[11]https://nwis.waterdata.usgs.gov/nwis/peak?site_no=12041200&agency_cd=USGS&format=html

particularly vulnerable to increased sediment supply associated with high-altitude warming (East et al. 2017). This includes new sediment resulting from glacial retreat, shrinking perennial snow fields, melting of permafrost, and mass wasting of recently deglaciated valley walls, all drivers of changes to downstream channel characteristics (East et al. 2017).

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

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

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

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

Stream flow, in the form of average annual stream discharge, summer low-flows, and peak flood-flows are typically indicators of stream habitat quality as well as a determinant of the amount of habitat quantity (refs). For the Olympic Peninsula as a whole there has been a decline in average annual discharge, more so than other parts of the western USA where United States National Forest Service (USFS) lands occur (Dunham et al. 2023). Streamflow assessed since

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

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

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

One of the largest predicted changes, with respect to changing climatic conditions, is the decline in glacial extent (Riedel et al. 2015), particularly for the larger west side watersheds. Over the past several decades, glacier decline in the Olympics was greater than in the Cascades and southern Coast Mountains, and is more comparable with Vancouver Island (Riedel et al. 2015). Riedel et al. (2015) estimate that the glacial contribution to summer streamflow has declined ~20% in the past 30 years, but still remains significant for the Hoh River. In the other Westside OP DPS watersheds, glaciers contribute less than 5% to summer streamflow (Riedel et al. 2015). The loss in glaciers over the past 30 years appears to be a result of mean air temperature increases, and illustrates how sensitive these relatively small, thin, and low-elevation glaciers are to climate change (Riedel et al. 2015, East et al. 2018). Continued loss of glaciers will directly impact aquatic ecosystems through higher stream temperatures and lower summer base flows. Summer low flows are predicted to decrease anywhere between 25% and 50% by 2040 for the majority of the Hoh River Basin (Wegner et al. 2010, USBOR 2014, USGS-OSC 2022). The largest changes are predicted to occur in the Upper Hoh basin (Wegner et al. 2010, USBOR 2014, USFS-OSC 2022). Peak flows have slightly increased from 1975 to 2010 from a decadal average of 585cms in the 1980s to 721cms average during the first decade of the 2000s.[12] (Jaeger et al. 2023). Large-scale flood events (i.e. greater than 25-year recurrence interval) are predicted to increase between 10% and 25% by 2040 (Wegner et al. 2010, USBOR 2014, USFS-OSC 2022).

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

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

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

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

Queets River

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

[12] https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&search_site_no=12043000&format=sites_selection_links

Peak flows in the Queets River are greatest during October through March with the majority of peak flows occurring November through January since the 1930s[13]. Average peak flow is 1,992 cms, with a maximum discharge of 3,766cms and a minimum peak flow of 932 cms[14]. Monthly low-flow typically occurs in August or September, averaging approximately 27 and 38 cms, respectively[15].

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

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

Similar to other watersheds in the OP DPS, there has been a large reduction in wood loadings due to riparian harvest, instream wood removal, and logging of floodplain forests. These actions led to changes in stream habitat conditions in tributaries and the mainstem areas, particularly in private and state timberlands (Bilby 1984, McHenry et al. 1998, Abbe and Montgomery 2003, Martens et al. 2014, 2019, 2020). Impacts include the loss of pools in smaller streams and a

[13] https://nwis.waterdata.usgs.gov/usa/nwis/peak/?site_no=12040500
[14] https://nwis.waterdata.usgs.gov/usa/nwis/peak/?site_no=12040500
[15] https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&search_site_no=12040500&format=sites_selection_links

decrease in stabilizing wood jams in the mainstem, which led to loss of stream channel complexity in larger streams (Abbe and Montgomery 2003, Martens et al. 2019). Even with ~25 years of more protective timber harvest regulations related to riparian zones important salmonid habitat components such as instream wood and pools have not recovered through natural recruitment of wood (Martens and Devine 2023). The estimated timeline for recovery of these remaining wood loading degradations could range between 100 and 225 years (Stout et al. 2018, Martens and Devine 2023).

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

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

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

The 5-year geometric observed escapement mean of winter steelhead in the Queets River has changed as follows – 4,111 from 1998 to 2002, 5,634 from 2003 to 2007, 4,613 from 2008 to 2012, 3,583 from 2013 to 2017, and 2,931 from 2018 to 2022 (Table A1).
Hatchery supplementation and operations for the Queets River has been ongoing since 1981 (Table A2). Between 1981 and 1989 the average winter steelhead smolt release was 133,600 (Table A2). Between 1990 and 1999 the average winter steelhead smolt release was 145,527, with a minimum of 83,483 and a maximum of 202,638 (Table A2). From 2000 to 2010 the average winter steelhead smolt release was 162,418, with a minimum of 149,874 and a maximum of 176,580 (Table A2). From 2011 to 2021 the average winter steelhead smolt release was 174,743, with a minimum of 158,204 and a maximum of 204,002 (Table A2).

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

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

Quinault River

The 490-km^2 Quinault River originates in Olympic National Park and flows approximately 111 kilometers to the Pacific Ocean (Williams et al. 1975). It includes over 460 km of anadromous stream habitat and approximately 111 kilometers of mainstem (Williams et al. 1975). Some notable features and tributaries include Lake Quinault, the North Fork Quinault River, Graves Greek, and Cook Creek (Williams et al. 1975). Land ownership varies as a function of the area below and above Lake Quinault. Below Lake Quinault ownership is predominantly the Quinault Tribal reservation (~80%), followed by Olympic National Forest (~14%), and private timberlands (~7%). Above Lake Quinault ownership is dominated by Federal lands (~95%), followed by Quinault Tribal reservation (~4.5%), and private lands (<0.5%),
Peak flows in the Quinault River above the lake are greatest during November through March with the majority of peak flows occurring November through January since the 1930s (https://nwis.waterdata.usgs.gov/usa/nwis/peak/?site_no=12039500). Average peak flow is 697 cms, with a maximum discharge of 1,489 cms and a minimum peak flow of 189 cms[16]. Monthly low-flow typically occurs in August or September, averaging approximately 26 and 28 cms,

[16] https://nwis.waterdata.usgs.gov/usa/nwis/peak/?site_no=12040500

respectively[17]. Like the other large watersheds draining the west side of the OP, there have been changes to peak flows (East et al. 2017). On the Quinault River the Q2 value for 1978–2013 was 808cms, whereas the Q2 value for the entire Quinault River record, dating back to 1909, is substantially lower at 595cms – 26% increase (East et al. 2017).

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

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

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

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

Between 1985 and 1989 winter steelhead catch average was 4,045 with a minimum of 2,618 and a maximum of 5,892 (Table A3). Average winter steelhead catch in the 1990s was 2,855 with a minimum of 1,628 and a maximum of 4,560 (Table A3). Between 2000 and 2010 average winter steelhead catch was 2,472 with a minimum of 1,516 and a maximum of 4,177 (Table A3). From

[17] https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&search_site_no=12039500&format=sites_selection_links

2011 to 2022 average winter steelhead has been 1,796, with a minimum of 316 and a maximum of 4,025 (Table A33). Summer steelhead sport catch between 1962 and 1971 was 321 with a minimum of 197 and a maximum of 463 (Table A3). Summer steelhead sport catch during between 1972 and 2021 is less than 5 annually (Table A3).

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Tables

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

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

Table A2 - Hatchery Supplementation. Annual juvenile releases

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

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

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

Table B3 - Harvest –1976-2006 Abundance. Date oly_pen_esu.xlsx. Steelhead Historical Database WDFW Fish Mgmt HQ, 600 Capitol Way N, Olympia 98501, (360) 902-2820/2817 https://wdfw.wa.gov/sites/default/files/publications/00150/oly_pen_esu.pdf. CRC spreadsheet Jim Scott May 2023 - Personal communication Jim Scott WDFW Subject: RE: extension for providing materials regarding OP steelhead petition Date: Friday, July 14, 2023 2:59 PM. WDFW and Tribal SJD data June 2023 - NOAA_5_15_23 OP Steelhead and SJD Aggregate Harvest Data. Data sent by WDFW May/June of 2023. Data sent by WDFW May/June of 2023 Personal Communication From: Harbison, Toby (DFW) Sent: Friday, June 16, 2023 4:13 PM To: Laura Koehn - NOAA Federal <laura.koehn@noaa.gov> for SJD data

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

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

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

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

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

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