Puntledge River Summer Chinook Habitat Status Report

Mel Sheng , Esther Guimond, Natasha Nahirnick, Jennifer Sutherst, Arthur Bass, Kristi Miller-Saunders and Tedd Sweeten

South Coast Area, Pacific Region, Fisheries and Oceans Canada
65 Front Street, Nanaimo, BC, V9R 5H7

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By

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EXECUTIVE SUMMARY

The Puntledge River watershed covers a 600 km^2 area west of the city of Courtenay on the east coast of Vancouver Island and is home to at least 17 fish species, including Percids, Gasterosteids, Centrarchids, Salmonids, and Cottids. Similar to several other watersheds in BC, the Puntledge River watershed has been greatly affected by several anthropogenic activities including hydroelectric generation and flow management, mining, forestry, fishing, community development and agriculture. These activities have affected several of the fish species, but particularly summer Chinook Salmon. Returns of this run historically averaged around 3,000 (until 1954) before declining drastically to ~400 individuals in the 1960s and 70s. Following captive breeding and stocking programs, in the 1990s and early 2000s, the returns rebounded before crashing again to current return levels of ~500 annual returns.

Puntledge River Summer Chinook are part of Designated Unit (DU) 20 and were assessed as Endangered in 2020 by the COSEWIC (COSEWIC 2020). In response, DFO is completing a Recovery Potential Assessment (RPA) as part of the *Species at Risk Act* listing process for this population (DFO 2023). A key task within this assessment was to bring experts together to identify potential threats to summer Chinook in the Puntledge River watershed and rank them according to the importance for population recovery and sustainability. On March 13-14, 2023, local experts from government (provincial, DFO), K'ómoks First Nation, industry (BC Hydro, Mosaic) and the sport fishing sector (Comox Fish & Game, SFAB) identified over 70 potential threats, of which eight were classified as Very High Risk and eight as High Risk (DFO 2023). Several of these were related to a loss of habitat complexity and availability while others were associated with pinniped predation, fish predation/competition, unfavourable water temperatures, hatchery fish maladaptation to the wild environment, and flow management issues (DFO 2023). Additionally, several potential threats were also identified as a data gap that affected the group's ability to classify them (DFO 2023). These included the prevalence of certain disease/pathogens (e.g., BKD) within the population, impacts of deleterious substances on returning adults and rearing juveniles in the estuary, access to appropriate food for early rearing, and competition with hatchery fish and water temperatures in the estuary.

This report provided a re-evaluation of the potential threats based on a detailed review of available information (i.e., literature review, internal DFO data, information from local experts) to provide updated threat ranking and data gaps. The overall goal was to examine all aspects that affect the productivity of the summer Chinook population and determine which are the most important to the survival so that the population can be correctly classified (e.g., *Species at Risk Act* listing) and recovery and management plans can be developed and implemented to provide positive effects on population recovery. These plans can be assembled by habitat type for ease of knowledge transfer to those who can initiate restoration work (e.g., First Nations, Community Groups, local Governments, and Industry).

The evaluation of the potential threats generally remained similar to DFO's report (DFO 2023), except that one threat was downgraded from "Very High" to "High" (i.e., predation by Coho Salmon) and nine were upgraded to "High" or Very High". Based on this updated evaluation, 22 threats are now considered "High (n=11) or "Very High" (n=11). These key threats include stress due to anthropogenic activities such as hydroelectric facilities (e.g., Eicher screens), migration issues, pre-spawn mortality, increase in heritability of BKD, beach habitat loss and unfavourable water temperatures. Finally, 11 threats could not be classified due to data gaps, which included inter- and intra-specific competition and lack of access to appropriate food. Some data gaps have also been identified for the key threats (i.e., "Very High", "High"). These data gaps would also benefit from additional studies to confirm the findings that were used to classify these key threats to summer Chinook Salmon. Overall, it is clear that a comprehensive stock status analysis of Puntledge summer Chinook is required to evaluate the response of the population to past recovery actions, including habitat and hatchery-related activities, as well as guide future decisions regarding the recovery of this stock.

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1. INTRODUCTION

The Puntledge River watershed covers a 600 km^2 area west of the city of Courtenay on the east coast of Vancouver Island (Map 1) and is home to at least 17 fish species, including Percids, Gasterosteids, Centrarchids, Salmonids, and Cottids. Historically, this watershed supported diverse and abundant stocks of salmon and trout and contributed significantly to an economically viable commercial and sport fishery in the area, as well as sustaining First Nations in the watershed long before the first non-natives arrived (Rimmer *et al.* 1994). However, similar to several other watersheds in BC, the Puntledge River watershed has been greatly affected by several anthropogenic activities including hydroelectric generation, timber harvesting, mining, agriculture, urban growth, and industrial uses of the estuary, as well as impacts from over-harvesting. Additionally, predation from birds and/or seals has also contributed to the declines in some salmon stocks, most notably Chinook salmon and steelhead (Rimmer *et al.* 1994; Hourston 1962).

In particular, the Puntledge River summer-run Chinook salmon run has been impacted by the construction of a hydroelectric facility built in 1955 consisting of the Comox Dam (storage), as well as a diversion dam on the Puntledge River. These impacts led to escapement estimates of summer-run Puntledge River Chinook Salmon declining from an average of about 3,000 to below 600 in 1975 (Guimond 2008). In the last 60 years, this population has been the focus of a significant rebuilding effort (e.g., habitat restoration such as spawning channels and installation of a fishway, a brood program and salmon stocking, and seal predation management, e.g., Yurk and Trites 2000); however, sustainable recovery to pre-hydro expansion escapement levels has not yet been achieved. Additionally, several management measures taken to address the concerns surrounding this stock, such as the captive brood stock program and seal cull efforts, have been expensive and contentious. For example, it has been concluded that hatchery releases represent a serious threat to the wild fish in the watershed due to competition and genetic introgression.

In November 2020, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) declared Designated Unit 20 (DU-20) East Vancouver Island, Ocean, Summer population, which includes summer-run Chinook in the Puntledge and Nanaimo rivers, as endangered, a wildlife species facing imminent extirpation or extinction (COSEWIC 2020). In response, Fisheries and Oceans Canada (DFO) started developing a Recovery Potential Assessment (RPA) as part of the *Species at Risk Act* listing process for these populations (DFO 2023). A key task within this assessment was to bring experts together to identify potential threats to summer Chinook in the Puntledge River watershed and rank them according to importance for population recovery and sustainability. On March 13-14, 2023, local experts from government (provincial, DFO), K'ómoks First Nation, industry (BC Hydro, Mosaic) and the sport fishing sector (Comox Fish & Game, SFAB) identified 70 potential threats, of which eight were classified as Very High Risk and eight as High Risk (DFO 2023). These threats were found to be mostly related to predation by seal and predation/competition with other fish, a loss of habitat and access in the river and estuary, disturbances due to anthropogenic activities, flow issues related to hydroelectric activities, barriers to fish migration, and maladaptation of hatchery juveniles to the natural environment (DFO 2023). Eleven limiting factors were also identified as presenting data gaps that affected the group's ability to classify them (DFO 2023). A summary of these threats along with the DFO RPA ranking are provided in Appendix A, although some of the threats and names have been combined and updated, respectively, to better describe threats for this report.

This report provides a description of the Puntledge River watershed and summer Chinook Salmon population, as well as an evaluation of the limiting threats initially identified by DFO as part of the RPA process. Specifically, it provides a description of limiting factors impacting each life history, which includes adult upstream migration into the watershed, spawning habitat quantity and quality, incubation conditions, freshwater rearing behavior/distribution and habitat, juvenile migration timing to the estuary and estuary rearing habitat and residence. This work was completed by conducting an extensive literature review, obtaining internal DFO data/analysis, as well as reaching out to several local experts (local and provincial government, companies, DFO scientists, hatchery staff and active and retired biologists). The ranking and data gaps that were identified by DFO (DFO 2023) were then re-evaluated based on the additional information that was identified for each limiting threat. It is expected that this report will be used as a reference and guide for DFO and local experts to complete the RPA process for this population to assess *Species at Risk Act* listing, as well as habitat and population restoration efforts.

2. CHARACTERIZING THE WATERSHED

2.1. Watershed Description

The Comox Lake watershed is located in the traditional territory of the K'ómoks First Nation and is the source of drinking water for over 49,000 residents through the Comox Valley Water System and the Cumberland Water System. Over the past 140 years the Comox Lake watershed has been a base for mining, logging and recreation activities. While coal mining operations ended in the 1930's, a large portion of the watershed is still currently privately owned and managed for timber supply. Comox Lake itself is a reservoir controlled by BC Hydro for power generation and is also the main water supply for domestic water use in the Comox Valley Regional District (CVRD). Swimming, boating, and camping are also activities popular in the designated public lake access areas.

The Puntledge River watershed covers a 600 km^2 area west of the city of Courtenay on the east coast of Vancouver Island. Approximately 75% of this watershed area is encompassed by Comox Lake and the influent drainages to the lake. Watershed elevations range from sea level to the Comox Glacier in Strathcona Provincial Park at 2,134 m. Downstream of Comox Lake, the Puntledge River flows in a north-easterly direction for 14.3 km where it joins with the Tsolum River. These two rivers combined become the Courtenay River, which flows for another 2.7 km into the Strait of Georgia. For the purposes of this document, the Puntledge River refers only to the mainstem river downstream of Comox Lake, while the Upper Puntledge River refers to the inlet tributary at the southwest end of Comox Lake (Map 1, Table 1). Comox Lake reservoir lies at 135 m above sea level and has a surface area of 2,118 ha, an average depth of 61 m, and a maximum depth of 109 m (BC Hydro 2003). The two largest influent drainages are the Cruickshank and the Upper Puntledge rivers. Both of these tributaries have historically provided suitable habitat for summer Chinook spawning and rearing.

Map 1. Puntledge River watershed boundary (inset), and major features in the lower river below Comox Lake.

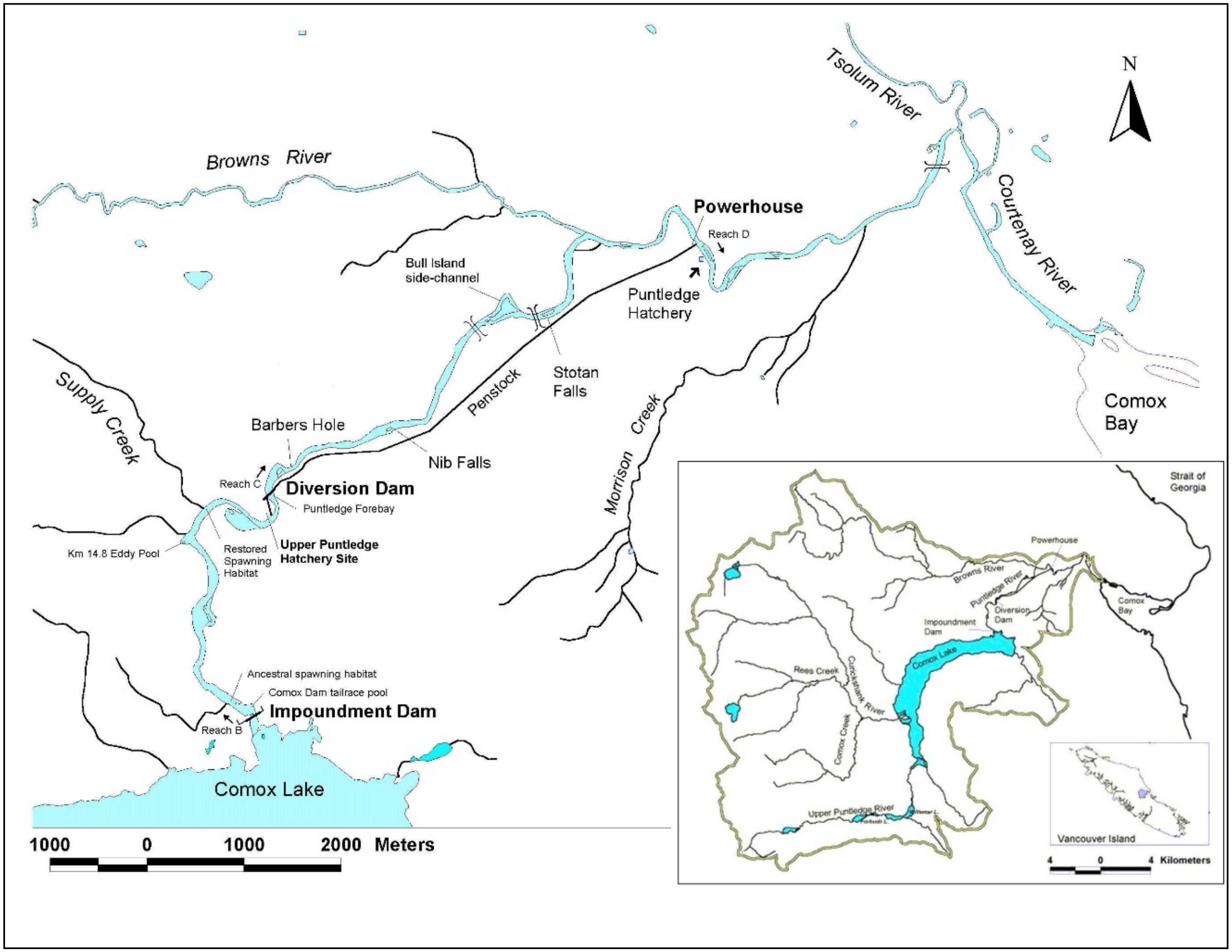


Table 1. Puntledge River watershed general description and reach characteristics (adapted from Fergus *et al.* 2005, Griffiths 1995).

Reach Number	Boundary	General Description	Length (km)	Average Gradient (%)
A	Upper Puntledge River	Second largest tributary to Comox Lake; mainstem arises out of Puntledge Lake at elevation 550 m asl, but receives inflows from tributaries originating in the Comox glacier	20	-
	Comox Lake Reservoir	2118 ha reservoir with a 4.5 m drawdown zone and max depth of 109 m.	15	-
B	Comox dam to Diversion dam (Headpond)	Deep, slow moving reach. Substrate ranges from mud to large gravel and cobble, much of it infilled with sand/silt.	3.7	0.01
C	Diversion dam to BC Hydro Powerhouse	Bedrock dominated channel that carries diminished flows due to diversion. Includes Nib Falls (8%) and Stotan Falls (15 %) and the Browns river tributary.	6.3	1.50
D	Powerhouse to Tsolum River confluence	Low gradient reach rejoined by diverted flows at the powerhouse. Morrison Creek tributary and several side channels in this reach.	5.7	0.05
E	Courtenay River and Estuary	Low gradient, mostly channelized, tidally inundated to Tsolum / Puntledge River confluence	2.7	-

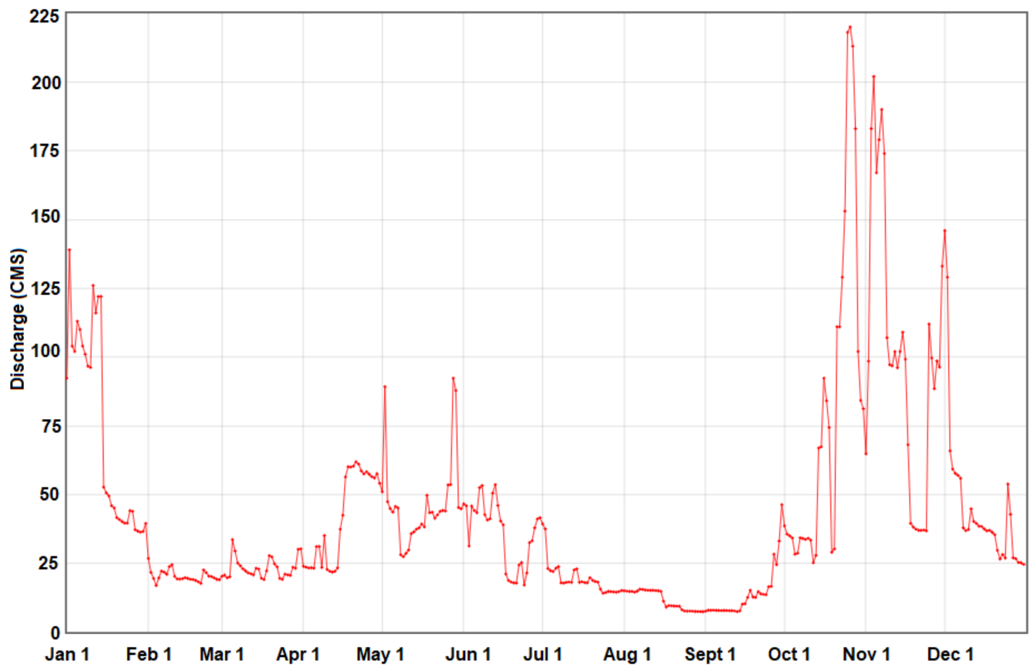
2.1.1. Biogeoclimatic Zones

The geography of the Puntledge River watershed is characterized by a vast richness and diversity of habitats, from its headwaters in the alpine region of the Comox glacier to the estuary in Comox Bay. The majority of the watershed lies within the Coastal Western Hemlock (CWH) and Mountain Hemlock (MH) biogeoclimatic zones with small portions of Alpine Tundra (AT) in the high elevation headwaters. The CWH zone occupies elevations from sea level to 900 m. This biogeoclimatic zone has been classified as the wettest in British Columbia. The Mountain Hemlock zone is located above the CWH. The climate of this coastal subalpine zone is characterized by short, cool summers, and long, cool, wet winters, with heavy snow cover for several months (MoF 1991). The Alpine Tundra, at elevations in the southwest above 1,650 m, is described as harsh, cold, windy, and snowy with average temperatures below 0°C for much of the year (MoF 1991). The watershed is dominated by a large lake ecosystem and numerous smaller lakes, along with several sensitive ecosystem types including wetlands, riparian, and older forest ecosystems as identified by the Sensitive Ecosystem Inventory (SEI) for East Vancouver Island and Gulf Islands (McPhee *et al.* 2000).

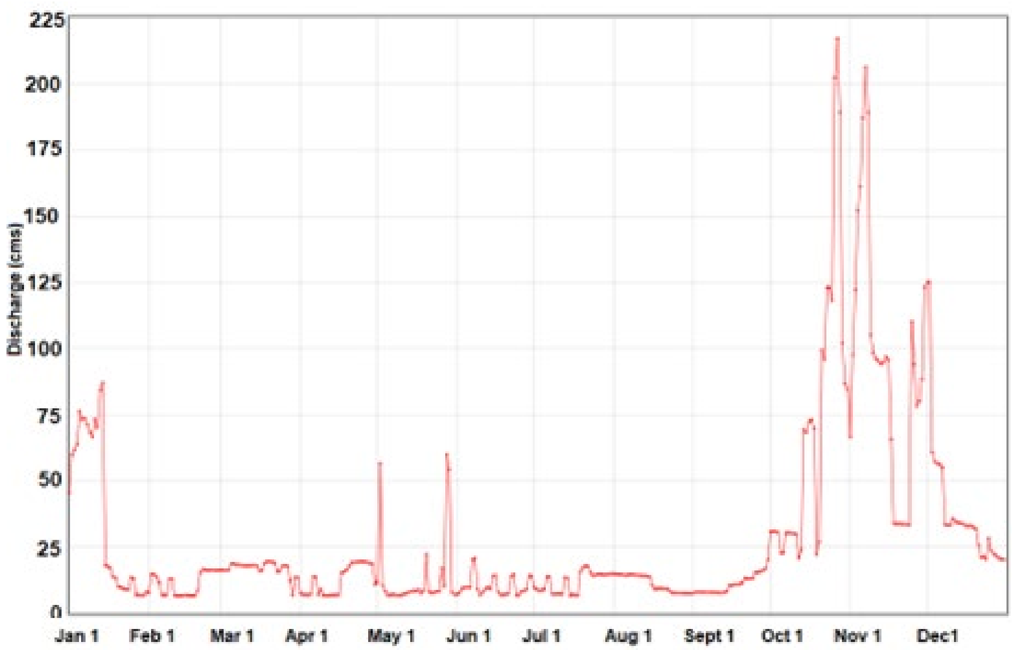
2.1.2. Hydrology

Much of the Puntledge River watershed is located at elevations above 200 m, with the western boundary of the watershed dominated by mountain ranges. The watershed is characterized as a combination of a rainfall driven and snowmelt driven system (Figure 1). The months with the heaviest precipitation are from October to March, and during the winter months, much of this precipitation falls as snow. Water Survey of Canada operates a hydrometric station on the Puntledge River above the confluence with the Tsolum River (08HB006) at Courtenay. This hydrometric station has over fifty years of discharge records from 1914 to 1920, 1955 to 1957, and 1964 to 2010. The drainage area of the Puntledge River at Courtenay (08HB006) hydrometric station is 593 km^2. Discharge recorded at this hydrometric station is influenced by controls for hydro power generation. To estimate the natural or non-regulated flow of the Puntledge River, the average of the discharge runoff per km^2 for the Cruickshank River and Browns River was used (Table 2). BC Hydro calculates inflows into the Comox Lake reservoir using a computer program called FLOCAL, which incorporates a variety of information, including gate openings, reservoir and tailwater elevations, generation, spill, turbine flows, and inflows (BC Hydro 2004).

Figure 1. **Discharge (m^3/s) in the Puntledge River between January 1 and December 31, 2021 at a) the Courtenay Water Survey Canada Station #08HB006, and b) below diversion at Station #08HB084 (GoC 2022).**



a)



b)

Table 2. **Puntledge River mean monthly and mean annual discharge (m^3/sec) for WSC hydrometric station 08HB006 (regulated), and estimated natural discharge, from Riddell and Bryden (1996).**

Discharge (m³/s)[1]	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	MAD[1]
Regulated	47.3	43.8	40	40.3	50.4	48.8	31	22.4	23.5	35.6	53.5	54.9	41
% MAD	115	107	98	98	123	119	76	55	57	87	130	134	100
Natural	48.4	50.1	42.1	47.7	69.8	60.5	25.4	15.9	9	43.4	58.4	47.7	41.4
% MAD	117	121	102	115	169	146	61	38	22	105	141	115	100

[1]MAD is the mean annual discharge

2.1.3. Geology

The underlying geology of Comox Lake is known as the Vancouver Group – Karmutsen Formation (BCWRA 2009). Bedrock, consisting of basaltic volcanic rocks from the middle to upper Triassic period, surround the lake on the northern, southern, and western boundaries. The eastern shore of Comox Lake and eastward is comprised of coal-bearing sedimentary rock of the Nanaimo Group, including mixed sandstone and shale, which are well exposed along and within the Puntledge River downstream of the lake outlet (Benjamin 2009).

Gravel sources and gravel recruitment to the lower Puntledge River reflect geologic processes that took place in south-western British Columbia since the last (Fraser) glaciation, 15,000 years ago. During the climax of this glaciation, the ice extended up to 1,200 to 1,500 m above sea level (4,000 or 5,000 ft asl) on Vancouver Island (Mathews *et al.* 1970). The ice withdrew quickly, leaving the Courtenay area about 13,000 years ago. The land at Courtenay was depressed at least 150 m (500 ft) by the weight of the ice, resulting in a maximum marine limit of 150 m (500 ft) above present-day sea level immediately following ice retreat. The land rebounded quickly following ice retreat and sea level dropped from the 150 m to the 60 m (500 ft to 200 ft) contour in only 300 years. This process left behind a succession of emergent deltas upstream of many creeks on eastern Vancouver Island (Fyles 1963; Clague 1980; Mathews *et al.* 1970). Delta terraces are found along and adjacent to most of the stream valleys of the coastal lowlands along the east coast of central Vancouver Island between 120 m and 150 m (400 ft and 500 ft) asl (Fyles 1963). Marine shells dating back to 12,360 B.P. (before present) have been discovered along the Puntledge River about 46 m (150 ft) asl indicative of marine invasion (Clague 1980). Glaciofluvial deposits and gravel pits have been mapped up to 150 m (500 ft) asl around Comox Lake and the Puntledge River (Fyles 1960). Although not interpreted to be deltaic deposits by the author, the locations of these sediments, mainly around the 150 m (500 ft) contour, correlates with the deltas that have been identified since the map was produced.

2.1.4. Fish Habitat

2.1.4.1. Puntledge River

There are two sets of natural falls, Stotan and Nib, that limited migration of anadromous and resident fish. Improvements were made between 1923 and 1977 by blasting step pools. Chinook, Coho, and Steelhead likely accessed areas above the falls and above Comox Lake prior to passage improvement. The lower Puntledge was an important spawning area for fall Chinook, Pink and Chum Salmon, but these stocks have declined and are being rebuilt or enhanced by the Puntledge River Hatchery.

Summer Chinook historically migrated into Comox Lake and held at depth in cooler water until spawning season (i.e., early October), and then migrated back downstream to spawn in Lower Puntledge, which is now between the Comox Impoundment and Diversion Dams. This section of river is still the main spawning area for summer Chinook. A portion of Chinook also spawn in the upper watershed and have been observed in the Upper Puntledge River and Cruikshank River.

Below Comox Lake, the Puntledge River can be divided into 3 major reaches (Bengeyfield and McLaren 1994; Map 1): Reach B, Reach C, and Reach D. Reach B, the headpond reach, is located between the Comox impoundment dam at the outlet of Comox Lake and the Puntledge diversion dam approximately 3.7 km downstream. This reach is low gradient (<0.01%) with deep, slow-moving water as a result of back flooding from the diversion dam. The average channel width is about 60 m and ranges between 35 m and 105 m (Bengeyfield and McLaren 1994). Substrates range from mud to large gravel and cobble with a small percentage of boulder. Much of the gravel is infilled with sand and silt, and in slower velocity areas the riverbed is covered with algal mats.

Reach C, the diversion reach, extends downstream of the diversion dam for 6.3 km to the BC Hydro Puntledge Generating Station or "Powerhouse". The Browns River, a large tributary of the Puntledge River, enters the mainstem approximately 4.7 km downstream of the diversion dam. This tributary is a moderate gradient system fed by run-off from rainfall and snowmelt and is used to separate Reach C into upper and lower sub-reaches. Upstream of the Browns River confluence, the Puntledge River is dominated by smooth bedrock and has an average gradient of 2%. This upper reach is punctuated by two major waterfalls: Nib (Nymph) Falls and Stotan Falls. The river downstream of the Browns River confluence has a lower gradient (0.5%) and more complex bed materials, thus, providing greater rearing potential for juvenile fish.

Reach D encompasses the remaining 4 km of the Puntledge River from the Powerhouse to the Tsolum River confluence. Flows in this reach are greater than the two upstream reaches due to flow releases from the penstock. This reach is low gradient (0.05%), contains several side-channel and off-channel areas, and Morrison Creek – a low-gradient tributary with an extensive wetland complex in its headwaters.

2.1.4.2. Comox Lake

Phytoplankton

Phytoplankton samples were collected for all three basins in Comox Lake (Map 2) and the dominant species for each site are listed in Table 3, Table 4 and Table 5. Phytoplankton was sampled 13 times between March 2005 and March 2008. A total of 58 species were identified. Overall, the inlet basin tended to have lower plankton concentrations (average of 255 cells/mL) and higher species richness (the number of different species occurring) (average of 48 species). Whereas the main and outlet basin had similar average species richness (38 species). The outlet basin tended to have higher overall concentrations of plankton (average of 312 cells/mL). In general, algal concentrations were quite low in all three basins, which is typical of oligotrophic lakes. Concentrations of chlorophyll a measured in the three basins ranged from < 0.5 μg/L to a maximum of 1.2 μg/L in the main basin also indicating low productivity.

The phytoplankton community in Comox Lake was dominated most years by diatoms from the Order Centrales, with *Cyclotella glomerata* and *Rhizosolenia eriensis/longiseta* comprising the majority of the plankton community in most samples. Pennate diatoms were also common in both the inlet and outlet basins, especially *Achnanthes minutissima*. During the winter months, a number of species from three other orders (Chlorococcales, Cryptomonadales, and Dinokontae) were present, but they were not seen in significant numbers during the summer months (in the June or August samples). In October 2005 and 2006, two species of blue-green algae (*Anacystis cf elachista var. conferta* and *Anacystis limneticus*, from the Order Chroococcales) were present in significant numbers in all three of the basins. Overall, the phytoplankton community found in Comox Lake is consistent with the oligotrophic conditions as indicated by the water chemistry results.

Table 3. **Summary of dominant (i.e., >10% of sample) phytoplankton species for the inlet basin of Comox Lake 2005 – 2008 (number of cells/mL and % of total sample). Source: Epps and Phippen, 2011**

	9-Mar-05	1-Jun-05	11-Aug-05	13-Oct-05	14-Mar-06	30-May-06	16-Aug-06	17-Oct-06	20-Mar-07	21-Jun-07	9-Aug-07	23-Oct-07	19-Mar-08
Order: Centrales													
Cyclotella glomerata		150 40%					70 44%		14 31%	67 14%		31 39%	32 37%
Melosira italica	17 31%												
Rhizosolenia eriensis/longiseta		52 14%	398 76%		17 18%	49 38%		53 25%			529 72%		
Rhizosolenia sp.										333 68%			
Order: Chlorococcales													
Scenedesmus cf dentriculatus				22 12%									
Sphaerocystis schroeteri												8 11%	
Order: Chroococcales													
Anacystis cf elachista var. conferta							22.0 14%	56 26%					
Anacystis limneticus				39 22%									
Order: Cryptomonadales													
Chroomonas acuta				20 11%					8 19%			15 20%	24 27%
Order: Dinokontae													
Peridinium cf inconspicuum				20 11%									
Order : Ochromonadales													
Dinobryon bavaricum													
Dinobryon spp.						13.0 10%							
Order: Pennales													
Asterionella formosa						18% 14%							
Achnanthes minutissima	10 18%				14.0 15%	18% 14%			6 13%				10 11%
Fragilaria crotonensis									10 22%				
Tabellaria flocculosa		50 14%											

Table 4. **Summary of dominant (i.e., >10% of sample) phytoplankton species for the main basin of Comox Lake 2005 – 2008 (number of cells/mL and % of total sample).**

	9-Mar-05	1-Jun-05	11-Aug-05	13-Oct-05	14-Mar-06	30-May-06	16-Aug-06	17-Oct-06	20-Mar-07	21-Jun-07	9-Aug-07	23-Oct-07	19-Mar-08
Order: Centrales													
Cyclotella glomerata	7 23%	224 56%		22 20%	4 14%	39 20%	67 53%	113 38%	17 32%	101 15%	76 10%	35 78%	
Rhizosolenia eriensis/longiseta		86 22%	367 84%			57 30%		62 20%		465 69%	616 82%		11 16%
Order: Chlorococcales													
Crucigenia quadrata				11 10%					6 11%				
Order: Chroococcales													
Anacystis cf elachista var. conferta				28 25%				42 14%					
Order: Cryptomonadales													
Chroomonas acuta	15 50%			20 18%	22.0 73%				11 22%				34 48%
Order : Ochromonadales													
Dinobryon bavaricum						23.0 11%							

Table 5. Summary of dominant (*i.e.* >10% of sample) phytoplankton species for the outlet basin of Comox Lake, 2005 – 2008 (number of cells/mL and % of total sample).

	9-Mar-05	1-Jun-05	11-Aug-05	13-Oct-05	14-Mar-06	30-May-06	16-Aug-06	17-Oct-06	20-Mar-07	21-Jun-07	9-Aug-07	23-Oct-07	19-Mar-08
Order: Centrales													
Cyclotella cf bodanica	4 13%												
Cyclotella glomerata	11 35%	203 61%		35 42%		81 32%	57 56%		7 36%	90 11%		38 66%	10 21%
Rhizosolenia eriensis/longiseta		59 18%	210 81%		17 20%	69 27%		106 25%		666 79%	955 89%		7 15%
Rhizosolenia sp.													
Order: Chroococcales													
Anacystis cf elachista var. conferta				17 20%				238 55%					
Anacystis limneticus				20 23%			24.0 23%						
Order: Cryptomonadales													
Chroomonas acuta					38.0 60%				7 36%				15 32%
Cryptomonas ovata/erosa												13 22%	
Order: Dinokontae													
Peridinium cf inconspicuum													
Order : Ochromonadales													
Dinobryon spp.						70.0 28%							
Order: Pennales													
Achnanthes minutissima	6 17%	32 10%							3 14%				8 18%
Fragilaria crotonensis	7 22%								3 14%				

Zooplankton

The zooplankton community of Comox Lake was composed mainly of four groups: rotifers, cladocerans, calanoid copepods and cyclopoid copepods (Table 6 to Table 8). In all three basins, the zooplankton community was dominated by three rotifer genera: *Keratella cochlearis, Polyarthra*, and S*ynchaeta*. *Keratella* and *Polyartha*. These species are cold water rotifers that normally reach maximal population densities in midwinter to early spring (Wetzel 2001). *Callotheca*, was also observed in the summer and fall of 2006 in all three basins and was the dominant species at the outlet basin in June 2005 and August 2007.

The dominant calanoid copedod during the study period was *Diaptomus oregonensis*, which was typically only observed in March. During the spring, copepod nauplii (newly hatched copepods) were prevalent, dominating the zooplankton population at all three basins. By late summer/early autumn, cladoceran, *Bosmina longirostris*, dominant in response to the breakdown of thermal stratification and increased nutrient regeneration from the deeper waters (Wetzel 2001). Overall, the zooplankton communities observed in Comox Lake are consistent with oligotrophic conditions.

Table 6. **Summary of dominant (i.e., >10% of sample) zooplankton species for the main basin of Comox Lake 2005 – 2008 (number of cells/mL and % of total sample).**

	9-Mar-05	1-Jun-05	11-Aug-05	13-Oct-05	14-Mar-06	30-May-06	16-Aug-06	17-Oct-06	20-Mar-07	21-Jun-07	9-Aug-07	23-Oct-07	19-Mar-08
Subclass : Copepoda										no data			
Copepod nauplii	440 60%				507 40%	413 10%			1320 85%				2532 86%
Order : Cyclopoida													
Diacyclops thomasi											1133 35%		
Order: Calanoida													
Diaptomus oregonensis adult					209 17%								295 10%
Diaptomus oregonensis copepodid		1467 11%											
Order: Cladocera													
Bosmina longirostris			1325 13%				1453 12%				533 16%	3525 38%	
Daphnia ambigua							1893 15%						
Holopedium gibberum													
Phylum: Rotifera													
Callotheca sp.?							1860 15%	860 16%					
Gastropus sp.													
Keratella cochlearis	107 15%		4775 47%	1020 11%	267 21%		3200 25%	1560 30%			533 16%	3025 32%	
Polyarthra sp.		4467 33%		4980 54%		1307 32%	1307 10%	1360 26%					
Synchaeta sp.						1240 30%							

Table 7. **Summary of dominant (i.e., >10% of sample) zooplankton species for the inlet basin of Comox Lake, 2005 – 2008 (number of cells/mL and % of total sample).**

	9-Mar-05	1-Jun-05	11-Aug-05	13-Oct-05	14-Mar-06	30-May-06	16-Aug-06	17-Oct-06	20-Mar-07	21-Jun-07	9-Aug-07	23-Oct-07	19-Mar-08
Subclass : Copepoda										no data	no data		
Copepod nauplii	267 14%				147 11%	1200 14%			427 53%				780 61%
Order : Cyclopoida													
Diacyclops thomasi													159 12%
Order: Calanoida													
Diaptomus oregonensis adult													
Diaptomus oregonensis copepodid		1925 16%				1133 13%							
Order: Cladocera													
Bosmina longirostris	240 13%		1280 13%	1653 13%	167 12%	973 11%	1020 10%					5160 35%	
Daphnia ambigua			1800 18%									3660 25%	
Holopedium gibberum			1227 12%										
Phylum: Rotifera													
Callotheca sp.?							1770 16%	1107 15%					
Gastropus sp.			1240 12%										
Keratella cochlearis	213 11%		1587 16%	1480 12%	440 32%	973 11%	1320 12%	2800 39%	120 15%			2670 18%	
Polyarthra sp.		2225 18%		6440 51%	200 14%	1040 12%	3810 35%	933 13%					
Synchaeta sp.	773 40%	2800 23%			213 15%	1387 16%							

Table 8. Summary of dominant (i.e. >10% of sample) zooplankton species for the outlet basin of Comox Lake, 2005 – 2008 (number of cells/mL and % of total sample).

	9-Mar-05	1-Jun-05	11-Aug-05	13-Oct-05	14-Mar-06	30-May-06	16-Aug-06	17-Oct-06	20-Mar-07	21-Jun-07	9-Aug-07	23-Oct-07	19-Mar-08
Subclass : Copepoda										no data			
Copepod nauplii	360 17%				307 23%				453 41%				1000 60%
Order : Cyclopoida													
Diacyclops thomasi	203 10%												
Order: Calanoida													
Diaptomus oregonensis adult									151 14%				158 10%
Diaptomus oregonensis copepodid													
Order: Cladocera													
Bosmina longirostris			813 13%	2320 34%					112 10%			8970 43%	
Daphnia ambigua													
Holopedium gibberum													
Phylum: Rotifera													
Callotheca sp.?		820 11%						587 12%			3600 27%		
Gastropus sp.											1200 10%		
Keratella cochlearis	870 41%		2973 47%	940 14%	427 33%		9280 61%	2173 46%	253 23%		1120 9%	3960 19%	
Polyarthra sp.	240 11%	2860 37%		2060 30%	267 20%	1147 19%		640 13%			2400 19%	2370 11%	
Synchaeta sp.		1820 23%				3027 49%							

2.1.4.3. Courtenay River Estuary

The Courtenay River starts downstream of the confluence of the Puntledge and Tsolum Rivers and flows for 2.7 km through the city of Courtenay before discharging into Comox Harbour. The Courtenay River estuary's eastern boundary at the mouth of Comox Harbour, and specifically identified by the division of a line extending in a north easterly direction from a point ~1.6 km south of the Trent River to Goose Spit (Adams and Asp 2000). This area encompasses portions of the City of Courtenay, the Town of Comox, the Comox Valley Regional District (formerly the Comox-Strathcona Regional District), and the K'ómoks First Nation.

The Courtenay River estuary is relatively shallow, consisting of mudflats, sand, gravel, and eelgrass beds (Bravender *et al.* 2002). At tide levels of 3 m above datum the mudflats begin to dry and are entirely exposed at tides of 1 m above datum. Tides in this area are typically semi-diurnal with a maximum height of 5 m above datum (Olesiuk 1995). The Courtenay River is tidally influenced and confined to a channel on the northeast side of the estuary, which had been continuously dredged over the last century to provide navigable passage through the estuary and up the river during high tides.

The Courtenay River estuary lies within the Comox Valley Important Bird Area, which is recognized as a globally significant overwintering area for Trumpeter Swans and nationally significant area for waterfowl (IBA Canada). In a collaborative project by Ducks Unlimited and Environment Canada (Canadian Wildlife Service), the Courtenay River estuary was ranked in the top 10 Class 1 estuaries based on criteria used to rank over 440 estuaries in British Columbia (PBHJV 2024).

2.2. Water Quality

The main source of water for the Puntledge River is Comox Lake, which is also the Comox Valley Regional District drinking water supply for the township of Courtenay. Water quality monitoring is conducted within Comox Lake to ensure water quality objectives (i.e., safe limits for the protection of aquatic life and designated water uses in the waterbody or watershed) are met. Such objectives ensure that inputs from recreation, timber harvesting, and residential activities do not impair water quality. Objectives were developed using data collected from the inlet basin, which reflects the natural or background conditions in the watershed. Objectives have been established for the following parameters: Secchi depth, temperature, dissolved oxygen, total phosphorus and chlorophyll (Table 9). Additionally, objectives have also been established for turbidity and microbiological for the protection of drinking water (Epps and Phippen 2011).

Table 9. Summary of proposed water quality objectives for drinking water, recreation, irrigation and aquatic and wildlife in the Comox Lake Community Watershed.

Variable	Objective Value
Secchi Depth	Annual average ≥ 8 m
E. coli	≤10 CFU/100 mL (90th percentile) with a minimum 5 weekly samples collected over a 30-day period
Turbidity	≤2 NTU maximum
Total Phosphorus	≤6 µg/L average during spring overturn
Chlorophyll *a*	≤1.5 µg/L
Water Temperature	≤15°C summer maximum hypolimnetic temperature (>10 m depth)
Dissolved Oxygen	≥5 mg/L at any depth throughout the year

Water quality monitoring as well as education and forestry remediation programs in and around Comox Lake are conducted by Comox Valley Regional District (the CVRD) and Mosaic Forest Management. A partnership between these parties and the ENV is being established to increase protection and understanding of the lake ecology and coordinating monitoring programs. A summary of the current water quality in the Lower Puntledge Watershed is provided from a review of water quality reports including: Water Quality Assessment and Objectives Report for Comox Lake 2011, Comox Valley Regional District annual reports from 2017 to 2019 on Comox Lake Watershed Source Water Quality (of note, annual reports for 2020 and 2021 were not available), BC Provincial Data sheets from a lake survey in 1975, and water quality results from two sets of samples taken by Ecofish Research Ltd. from the upper and lower reach of the Puntledge River Head pond (i.e., Reach B located between the Comox Lake impoundment Dam and the Diversion Dam) in 2020 and 2021.

The main activities in the Comox Lake watershed are forestry, mining and recreational activities. Active forestry management (61%) and park services (33%) make up the majority land use in the watershed (Table 10). Forestry is governed by the Private Managed Forest Land Act as activities mostly take place on privately owned lands. Water quality objectives are primarily set to protect human drinking water. Although forestry managers are also required to retain sufficient streamside mature trees and understory vegetation to protect fish habitat (including water temperatures, channel stability, and stream bank stability).

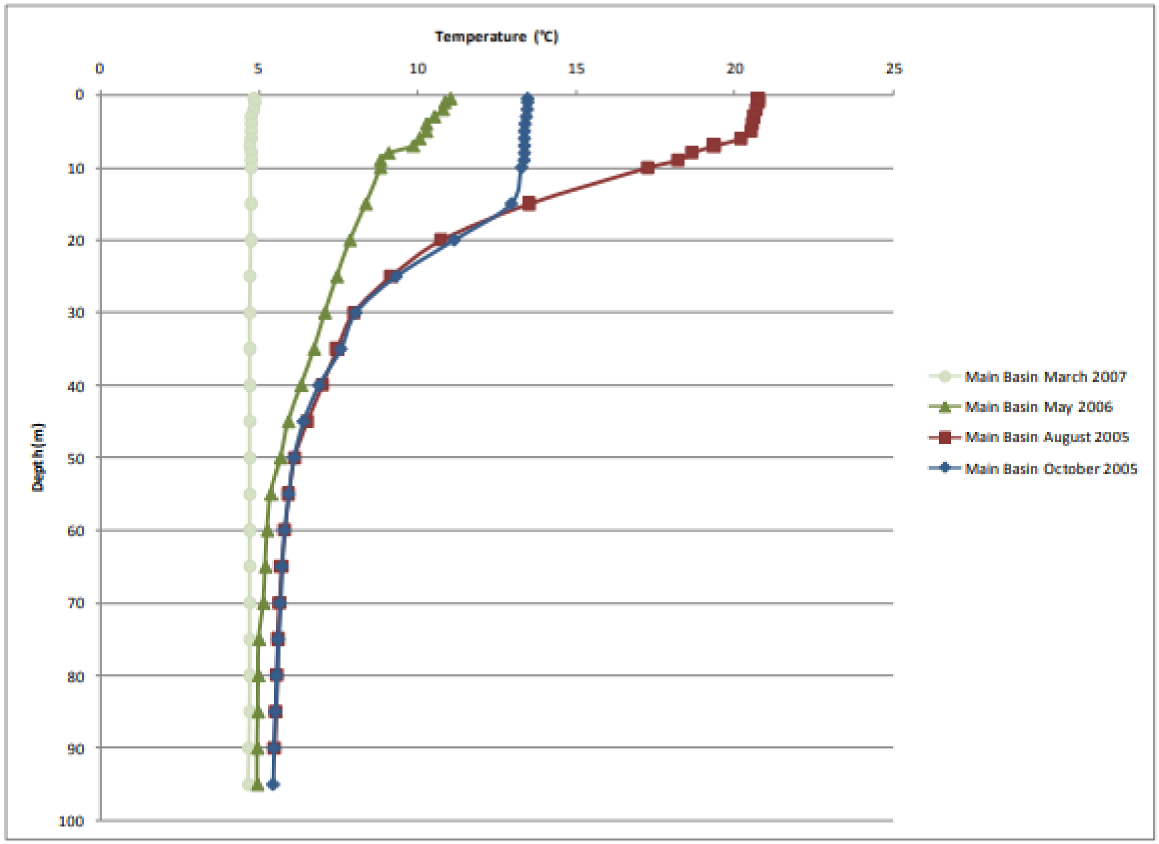
Table 10. Summary of land use within the Comox Lake Watershed (from Benjamin and Varashelvi 2006).

Land Use	Area (ha)	Area (km^2)	% of Total Watershed Area
Forestry	28,075	280.75	60.84
Park	15,141	151.41	32.81
Water	2,250	22.5	4.88
Crown Land	412	4.12	0.89
Private Land	91	0.91	0.20
Bc Hydro	11	0.11	0.02
Municipal Land	158	1.58	0.34
Road Right-of-Way	8	0.08	0.02
Total	**46,146**	**461**	**100**

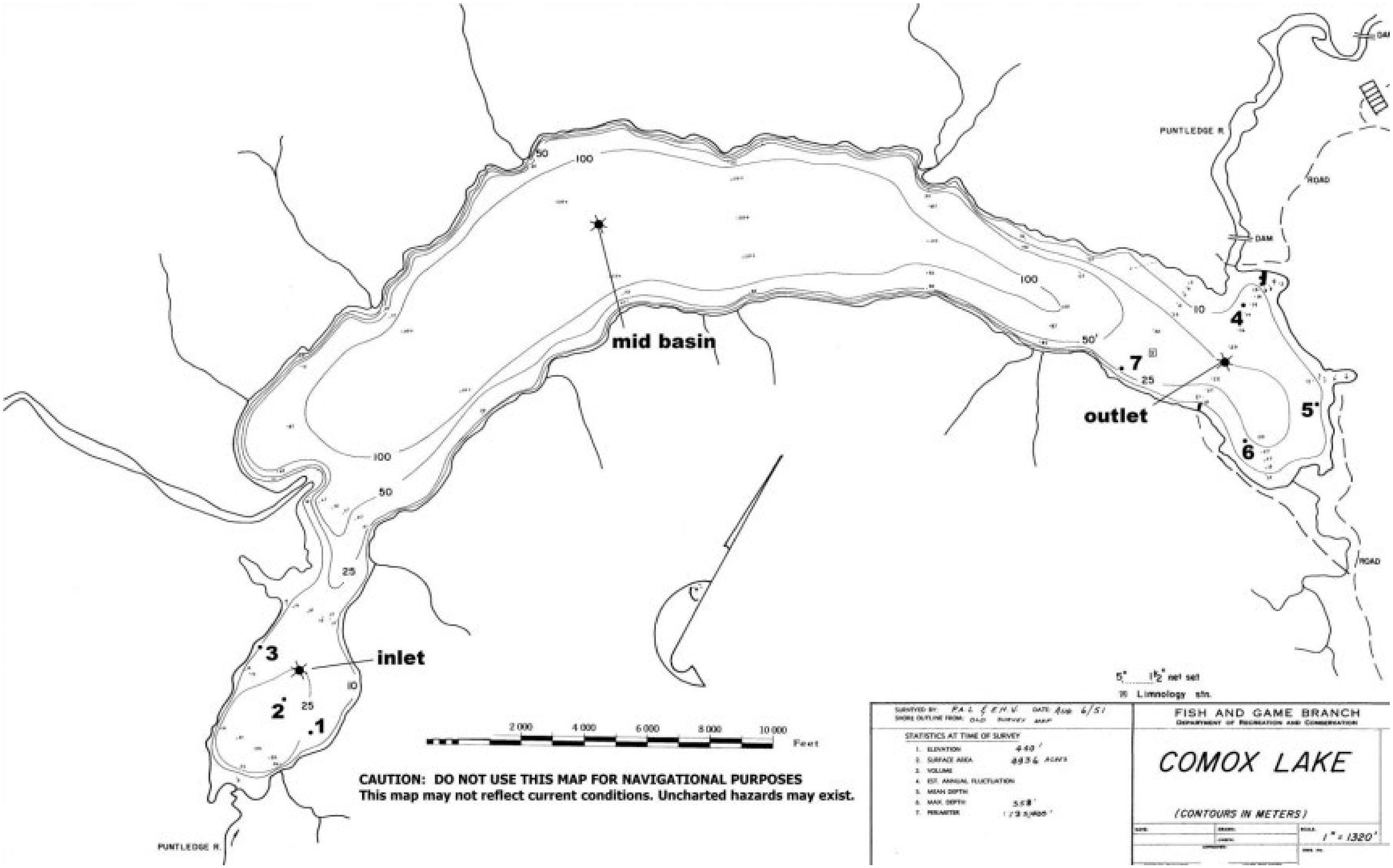
2.2.1. Lake Temperature

As typically occurs in coastal BC lakes, the water column in Comox Lake is unstratified during the winter months by late November, with stratification re-establishing sometime between March and May each year (CVRD 2019). By August each year, the water column is strongly stratified, with the thermocline occurring between 10 m and 30 m depth. Hypolimnetic temperatures remained between 5°C and 6°C throughout the year, while epilimnetic temperatures reached as high as 20.9°C over the course of the summer. All three basins are similar. Therefore, the main basin data is used here to represent the seasonal variability (Figure 2).

Figure 2. **Seasonal water temperatures measured at one to five metre intervals in Comox Lake in the main basin (Figure sourced from CVRD 2019, Section 6.1.1).**



Map 2. **Bathymetric map of Comox Lake, showing sampling locations (Map sourced from Fish wizard 2022).**



2.2.2. Water Temperature

Water temperature in the Puntledge River is highest during the summer and coincidental with low summer flows (i.e., June to September). During this period, average daily temperatures in the lower Puntledge River exceed 20ºC and daily maximum temperatures can often exceed 24ºC (Griffith 2000). River temperature data collected by DFO from the Upper Puntledge Hatchery since 1965 and the lower hatchery since 1977 are illustrated in Figure 3(a) and Figure 3(b), respectively. The Upper Hatchery temperature data reflect temperatures in the Puntledge River headpond while the lower hatchery data reflect temperature of the penstock tailrace discharge.

Suitable water temperatures are critical for fish and can have a significant influence on the behavioural ecology and most physiological processes of Pacific salmon (Hasler *et al* 2011b). Water temperatures above 24ºC can be lethal (Crozier *et al.* 2008). Pacific salmon prefer temperatures between 4ºC and 18ºC (Brett 1971), and exposure to temperatures outside of this range can have negative consequences on fish energetics, health condition, and survival, as well as on gamete viability (Jensen *et al.* 2006). This is especially the case during spawning migration because, as adult salmon cease feeding prior to migrating, they have a finite amount of energy available not only to swim upstream but also fuel reproductive maturation.

The availability of thermal refuge along a river migration route can provide adult salmon with a means of conserving energy through thermoregulation, by seeking cooler water in lakes, tributaries, or groundwater seepage (Berman 1990). In the Puntledge River, no evidence of cool-water refuge exists in the lower mainstem (i.e., Reach B, C and D), until fish reach Comox Lake (Hasler *et al* 2011). Since there is no opportunity for summer Chinook to behaviourally thermoregulate during their upriver migration through Reach C, the long-term survival of this stock is dependent on their ability to access Comox Lake before temperatures increase.

Specific temperature tolerances for summer Chinook have been summarized by Carter (2005). Temperatures above 21.1ºC have been described as a thermal migration barrier. Optimal migration temperatures are between 3.3ºC and 13.3ºC and should not exceed 17ºC. Optimal spawning temperatures range between 5.6ºC and 13.9ºC, and gamete mortality occurs above 15ºC. For incubation and emergence, the optimal temperature range is between 6ºC and 10ºC, and embryo survival is poor below 2ºC and above 15ºC. For juvenile rearing, the optimal growth temperature range is between 10ºC and 15.6ºC, and growth is impaired between 17ºC and 24ºC.

There has been some argument that hydro development has caused an increase in the temperature of the river downstream of the impoundment dam. Flow regulation can have an impact on the thermal regimes upstream and downstream of their footprint, but there is no historic temperature data on the Puntledge to verify these effects. Temperatures in the lower Puntledge River are, for the most part, regulated by the thermal surface mass of Comox Lake. This waterbody warms slowly in the spring and cools slowly in the fall (Sweeten 2005). The Comox Lake reservoir has a large volume of water and thus could act as heat sink. Water temperatures are cooler in the early summer than in nearby streams (without glacial inputs) and warmer in the late fall and early winter. Any large lake or reservoir has the potential to cause this, but in early summer, after the freshet, this appears pronounced – Comox Lake is known to be particularly cold (Lewis pers. comm. 2022). In a study that predicted the thermal impact of 216 current and future dams around the world, thermal modelling predicted that dams' cool temperatures during the summer months and generate warmer water during the winter months (Ahmad *et al.* 2021). The most significant temperature impact likely occurs in the spring and early summer when there is a drop in temperature of approximately 2ºC between the lake surface and a depth of 5 m, which is the depth of the under-sluice gate (i.e., BC Hydro flow control gate at the outlet of Comox Lake) (Figure 4). The relatively low height of the storage dam is suspected of only having a slight impact on temperature (Griffith 2000). The potential for Comox Lake reservoir to act as a heat sink should be assessed (Lewis pers. comm. 2022). However, because of the likely turbulence in front of the under- sluice gates, thermal stratification is likely broken-down causing mixing in the top 5 metres (Chilibeck pers. comm. 2022)

River temperatures tend to increase with increasing distance downstream in Reach C. The intensity of this warming trend depends largely on the snowpack and air temperature. For example, in 1998 – a year with below normal snowpack and warm air temperatures – significant warming (up to 4.5°C) occurred between the impoundment dam at Comox Lake and the Browns River, which is ~ 8 km downstream (Griffith 2000). In 1999, there appeared to be up to a 1ºC difference in the mean temperature between 150 m downstream of Comox Lake Dam, 50 m upstream of the Diversion Dam and upstream of Browns River (Figure 5).

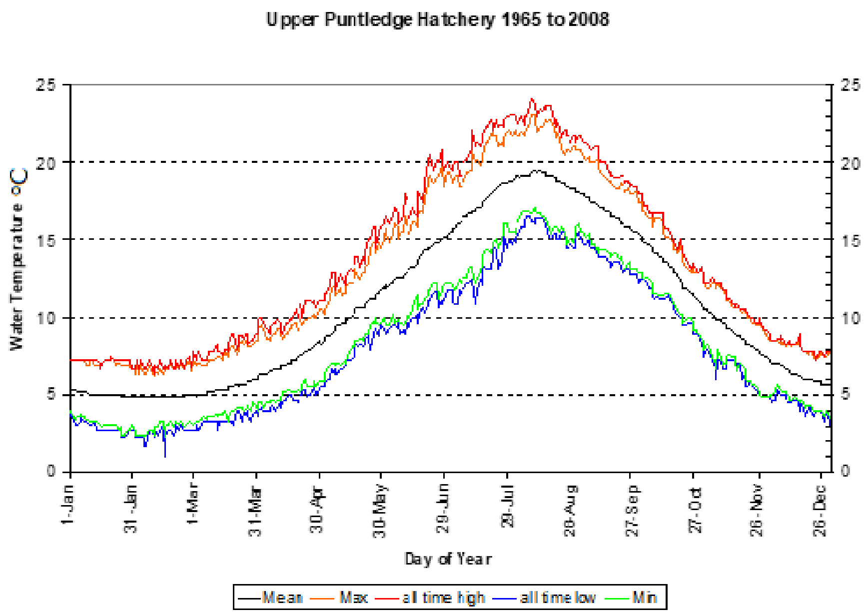
During the Puntledge Water Use Planning process, a modelling study was initiated to: (1) gain a better understanding of how the river is heated during the summer, and (2) determine if increased flow releases from the dam could significantly off-set high river temperatures during critical periods in the summer to reduce thermal stress on juvenile and adult salmonid (Sweeten 2005). The study used continuous temperature data from various sites in the Puntledge River, as well as flow data and weather data (air temperature and rainfall) to model the daily average water temperature in relation to water flow and air temperature. The study showed a significant correlation between water flow and temperature in Reach C, but air temperature had the largest influence on water temperature in the river. Results from the modelling indicated that very large volumes of flow would be required to off-set elevated river temperatures. For example, to alter water temperature by 1ºC, a flow increase of 16 m^3/s would be required. Due to the limited storage capacity of the lake, these volumes would not be available in the reservoir, which must be maintained to ensure delivery of the minimum fishery flows downstream all year.

Results from migration studies clearly demonstrated that summer Chinook that successfully migrate up to Comox Lake, where they can hold in cooler temperatures, have a spawning survival of over 95%. In contrast, Chinook that hold in the lower river all summer only survive at a rate of 50% to

70% (Guimond and Taylor 2010). Recovered temperature data from thermal loggers inserted into radio tagged Chinook indicated a general preference for fish to hold in the lake within a temperature range of between 10°C and 15°C (Guimond and Taylor 2010).

Figure 3. Daily mean temperature for (a) the Upper Puntledge River hatchery, and (b) the Lower Puntledge River hatchery. Data were collected between 1965 and 2008 using a Taylor thermograph (prior 1994) and Tidbit logger (Onset Corp.).

a)



b)

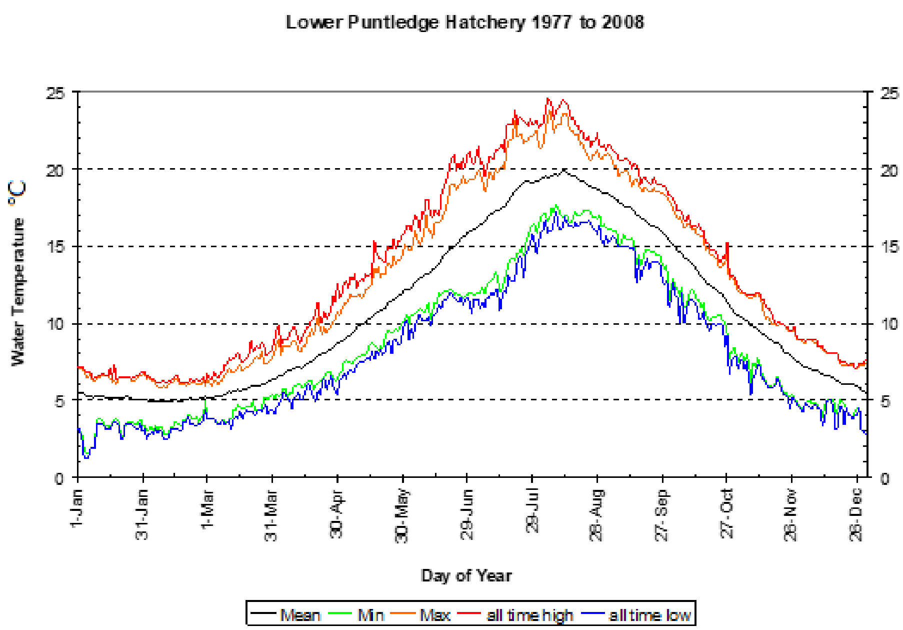


Figure 4. **Outlet Basin Temperature Profile 2015 (Saso, P. 2020. Comox Lake: Water Quality Objectives Attainment (2015 – 2016). Environmental Quality Series. Prov. BC, Victoria BC).**

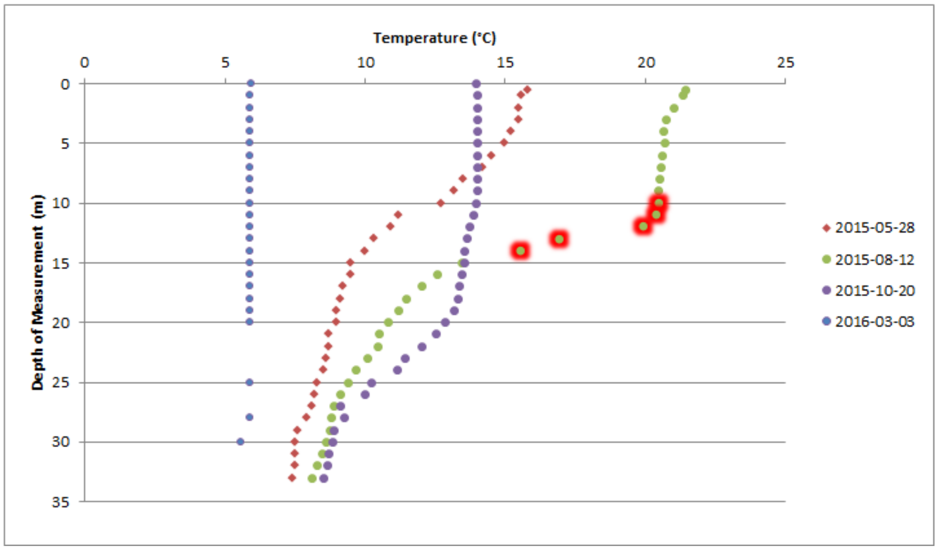
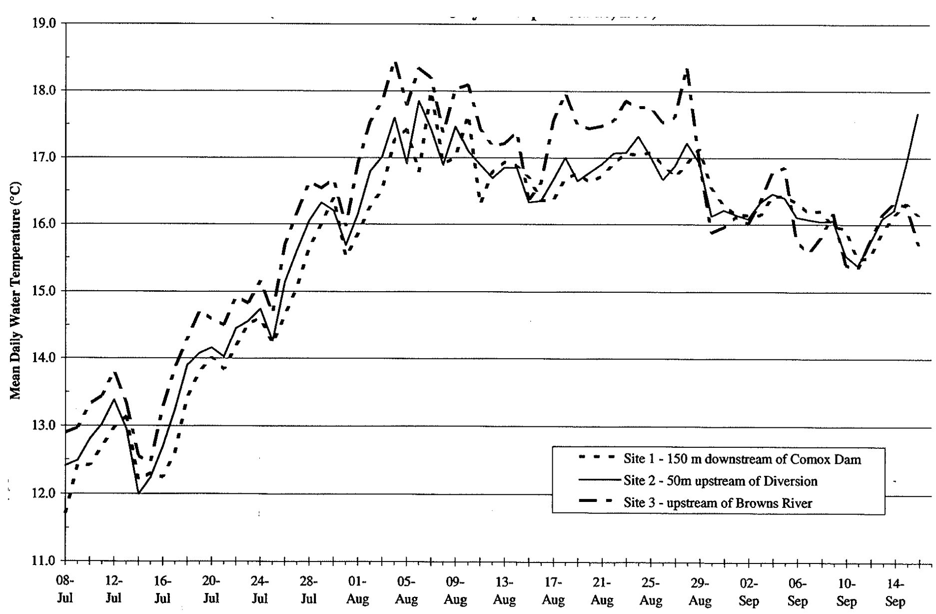


Figure 5. **Puntledge River Mainstem Mean Daily Water Temperatures (from 3 sites between July 8 to September 16, 1999).**



The effects of exposure to elevated water temperatures on various life stages of juvenile and adult salmonids have been studied in the past (Berman 1990; Servizi and Jensen 1977). In adult salmon, exposure to elevated river temperatures increases stress and susceptibility to disease, and results in higher mortality rates (McCullough 1999). However, limited information exists on the effects of exposure to high temperatures on the latter stages of egg maturation, gamete quality, fertilization success and egg development. High water temperatures in Puntledge River in the summer and early fall likely affect the productivity of migrating summer Chinook and pink salmon. An in-depth study was completed at the Puntledge hatchery in 2002 looking at temperature effects on migrating and spawning pink salmon. Specifically, the study captured returning adult pink salmon and exposed them to different temperature regimes for the latter part of maturation and spawning to determine the potential impact on maturation rate, adult mortality, and subsequent gamete viability.

Adult pink salmon were exposed to three declining water temperature regimes prior to spawning. The mean (range) test temperatures for the chilled, ambient, and heated regimes were 14.3°C (18.9°C-11.6°C), 17.8°C (20.3°C-15.0°C), and 20.9°C (23.6°C-17.6°C), respectively, from August 28 to September 17, 2002. During that period, the adult mortality was 2%, 10%, and 82%, respectively. Maturation rates were also affected as 53%, 7%, and 0% of females were ripe by October 1, 2002, respectively. (Pink spawning in Puntledge R. is usually completed by October.) Thirdly, mean egg mortality was 14%, 41%, and 60%, respectively. Hence, the adverse influence of high water temperature during the latter phase of maturation was demonstrated to significantly ($p < 0.05$) increase adult mortality, delay maturation rate, and reduce gamete viability (Jensen *et al.* 2004).

The study was repeated using summer Chinook adults in 2003 and 2005. Despite several unanticipated and unexplainable technical difficulties encountered in both years, results from these studies also demonstrated a delay in maturation (2003) and a higher pre-spawn mortality (2003 and 2005) in the adults exposed to warmer temperatures (17-22°C) compared to the "chilled" (8-9°C) group (Jensen *et al.* 2005; Jensen *et al.* 2006). In addition, exposure of adults to the higher temperature regime also resulted in increased egg mortality and reduced spermatocrits or sperm density (Table 11, Jensen *et al.* 2006).

Spawning surveys conducted between 2014 to 2016 in Reach B at the Supply Creek spawning platform and spawning area below Comox Dam observed spawners between late September and the third week of October (Table 12). Spawning mostly occurs between October 1st and 15th when water temperatures drop below 15°C, which usually occurs by October 7th. In the last 10 years there has been three years (i.e., 2014, 2015 and 2020) when the river temperature did not drop below 15°C until October 10th to 13th (Figure 6). Overall, water temperature data from 1977 to 2021 suggests that there is no advancement in fall warming occurring in early October (Figure 6). However, there appears to be a warming trend between 1981 and 1991 and a cooling trend between 1994 and 2013.

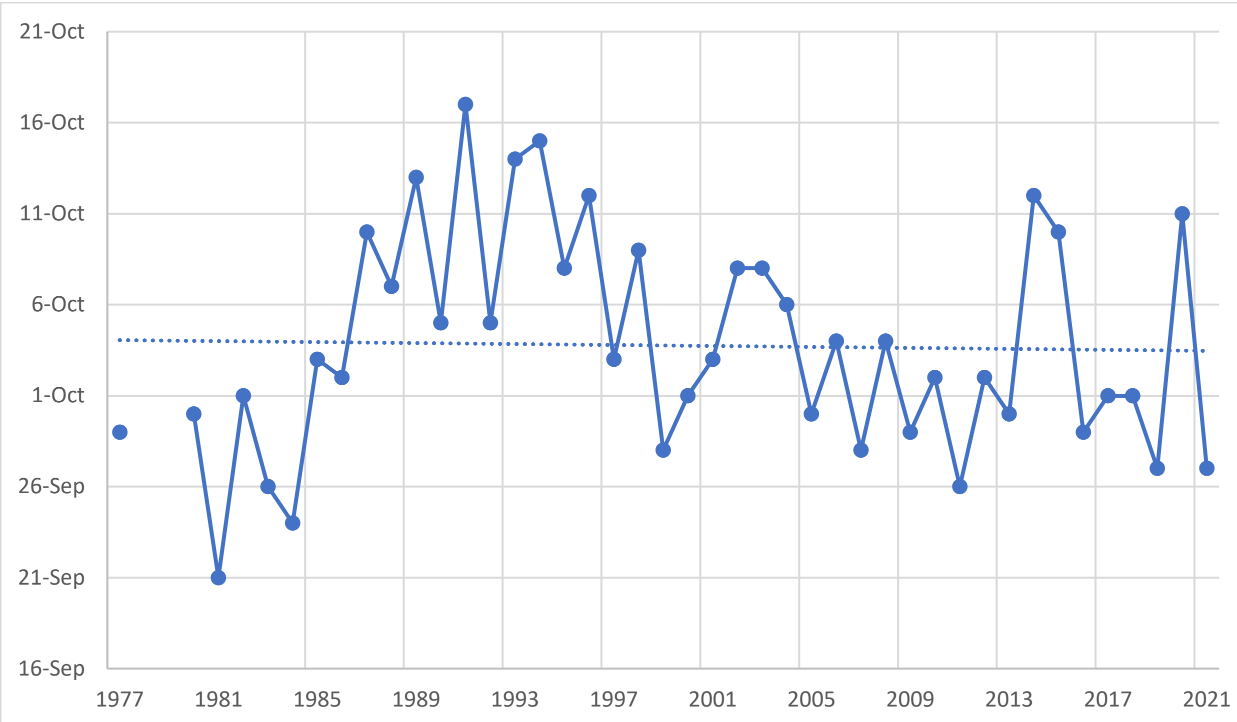
Table 11. **Descriptive statistics of Chinook mean egg mortalities from fish held at Rosewall Creek, Puntledge Upper Site and Puntledge Lower Site (Source: Jensen *et al.* 2006).**

Descriptive Statistics	Mean Mortality (%)		
	Rosewall Creek	Puntledge Upper Site	Puntledge Lower Site
Mean (sample size)	3.1 (30)	13.4 (17)	11.8 (29)
Standard Deviation	4.9	11.1	7.5
Minimum	0	5.9	1.9
Maximum	26.6	52.4	34.5
Confidence Interval (95%)	1.8	5.7	2.8

Table 12. **Observed number of spawners in Reach B of the Puntledge River from 2014-2016 (Guimond unpublished data).**

Year	Date	# Observed Alive
2014	Sept 28	1,212
	Oct 3	3,737
	Oct 8	6,666
	Oct 20	1,221
2015	Sept 30	22
	Oct 3	1,616
	Oct 6	2,020
	Oct 13	2,525
	Oct 22	1,818
2016	Sept 28	1,212
	Oct 8	15
	Oct 22	9

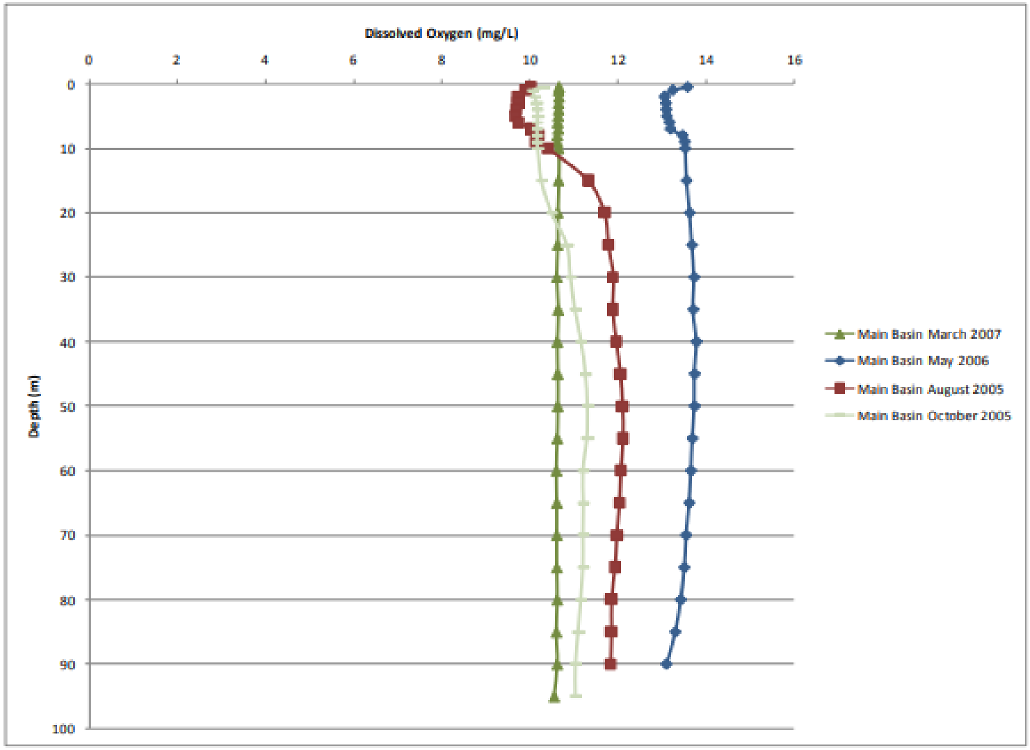
Figure 6. **First day in Fall when the mean temperature dropped below 15°C in the Puntledge River from 1977-2021.**



2.2.3. Dissolved Oxygen

Dissolved oxygen (DO) concentrations recorded between 2005 and 2008 by the Province of BC were at or near saturation in each of the three basins sampled. As DO levels were similar for all three basins, only the main basin data has been provided to illustrate seasonal differences (Figure 7). In general, when the lake was thermally stratified, DO concentrations increased with depth, as water temperatures decreased (resulting in increased oxygen solubility). On occasion, especially during the fall months (when algae would be senescing), DO concentrations near the bottom of the inlet basin decreased slightly, with a minimum recorded value of 6.1 mg/L. This is likely due to decomposition of organic material that grew over the course of the summer. However, at shallower depths (more than 5 m above the substrate), concentrations consistently exceeded 8 mg/L, while all values measured in the main and outlet basins were above 8.5 mg/L. Even when the lake was strongly stratified, oxygen concentrations in the deeper portion of the lake remained high, suggesting that there is low biological productivity and therefore low oxygen demand. As such, it does not appear that DO concentrations are a concern in Comox Lake and Puntledge River at this time. DO levels are near saturation and well within the safety levels for salmonids.

Figure 7. Seasonal dissolved oxygen concentrations (mg/L) measured at one to five metre intervals in Comox Lake in the main basin (Source: Epps and Phippen, 2011).



2.2.4. Water Clarity

Mean Secchi depths between 2005 and 2007 were similar between basins, ranging from 8.5 m in the main and outlet basins (2007) to 11.1 m in the outlet basin (2005). All values met the recreational guideline of 1.2 m. In future, the Secchi depth water quality objective will be set at >8 m and measured at least four times per year (i.e., once during each season) at all monitoring locations.

2.2.5. Turbidity

Turbidity values were consistently low in Comox Lake, ranging from 0.2 NTU to 0.7 NTU in the inlet basin, from 0.1 NTU to 0.8 NTU in the main basin, and from 0.2 to 0.9 NTU in the outlet basin (Table 13). The retention time of water in Comox Lake, from the upper Puntledge River to the BC Hydro spillway, ranges from 14 months in the winter (when the water column is mixed and therefore inputs relative to the entire volume of the lake are small) to 12 weeks in the summer (when the lake is stratified, isolating the water below the thermocline and greatly decreasing the volume of water moving through the lake) (Benjamin and Vasarhelyi 2006). Overall, Comox Lake provides considerable settling time for suspended sediments entering the system from the upper Puntledge River during the winter when sediment levels are highest. However, sediment inputs lower in the watershed (around the perimeter of Comox Lake outlet) have a decreased residence time, such that inputs into the outlet basin could take from 1.5 to 24 hours to reach the BC Hydro spillway (Benjamin and Vasarhelyi 2006). For example, Perseverance Creek, which drains closer to the outlet of the Lake, was only less than 5 NTUs 72.7% of the time and <1 NTU 37.7% of the time in 2018 (Barraclough 2019). Water quality objectives are recommended to ensure that the exceptional water clarity of Comox Lake is maintained. The objective is that the maximum turbidity measured in any sample collected at the three monitoring locations should not exceed 2 NTU. These values are based on an allowable increase of 1 NTU above existing maximum background values, measured at each of these sites. In the latest available CVWS Annual Water Quality Reports for 2016-2019, this level was never exceeded.

Table 13. Summary of turbidity values measured at each of the three monitoring locations on Comox Lake between 2005 and 2008.

Location	NTU				# of
	Min.	Max.	Mean	SD	Samples
Inlet Basin	0.2	0.7	0.4	0.1	30
Main Basin	0.1	0.8	0.3	0.2	30
Outlet Basir	0.2	0.9	0.4	0.2	30

In 2019, UV treatment was added to the new CVRD treatment plant and disinfection process and was fully functional by 2021, which allowed Island Health to increase the allowable turbidity limit for boil water notices from 1.0 up to 3.0 NTU. Over 80% of boil water notices in recent years have been within this range. The new water treatment plant will add filtration as an additional barrier to meet provincial drinking water guidelines and eliminate turbidity related boil water notices completely (CVRD Watershed Protection Plan updated 2022). In 2019, turbidity did not exceed 3.0 NTU and one boil water notice was issued. The boil notice lasted one day, as BC Hydro completed maintenance on their system, shutting down the penstock and requiring the CVRD to rely on the back up pumping station, where water quality was poorer. There have also been instances of sediment inputs into Comox Lake from tributaries such as Beech Creek. The completion of the 5-kilometer-long pipeline from Comox Lake to the treatment facility in 2021 has eliminated the need to extract water from the pumping station.

Overall, turbidity levels between 1 NTUs and 3 NTUs in Comox Lake and subsequently Puntledge River is highly infrequent (i.e., < 3 days per year) and poses little or no risk to summer Chinook Salmon.

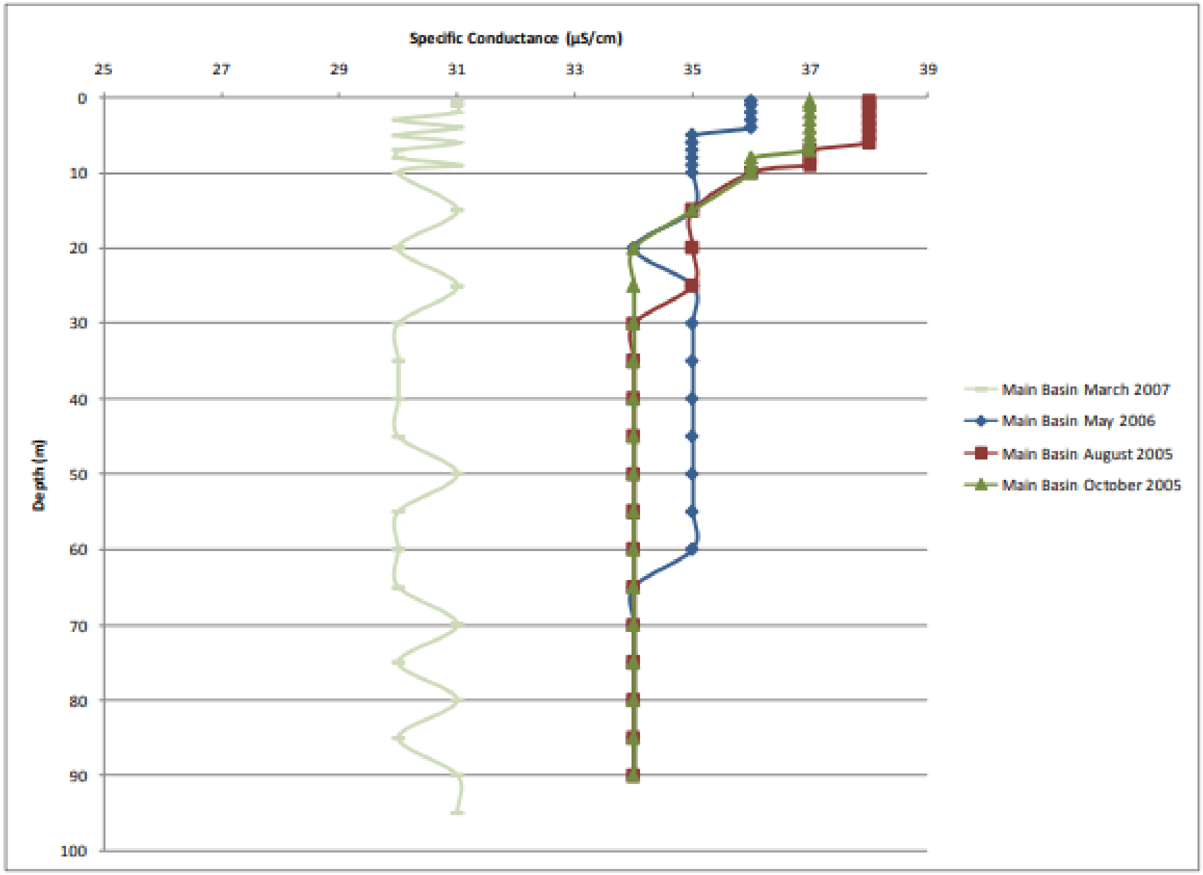
2.2.6. Organic Carbon

Colour is closely correlated with organic carbon concentrations, as humic acids (high in organic carbon) are often major contributors to colour in water. Elevated total organic carbon (TOC) levels (above 4 mg/L) can result in higher levels of disinfection by-products in finished drinking water if chlorination is used to disinfect the water (Moore and Caux 1997). CVRD uses chlorine to disinfect their drinking water, which is affected by TOC concentrations in Comox Lake. TOC concentrations were measured 15 times at each of the three monitoring sites, with values ranging from 0.5 mg/L to a maximum of 1.9 mg/L at the outlet site. All values were well below the drinking water guideline of 4 mg/L (Epps and Phippen 2011).

2.2.7. Conductivity

Specific conductivity values measured in Comox Lake were consistently low, ranging from 21 μS/cm to 44 μS/cm in the inlet basin, from 30 μS/cm to 38 μS/cm in the main basin, and from 30 μS/cm to 41 μS/cm in the outlet basin. Values were correlated with flows, with the highest conductivity occurring during low flows (when dilution was lowest) and dropping during the winter (when dilution from rainfall was highest). Figure 8 illustrates this seasonal variability as represented by the main basin data. As there is no BC Water Quality Guideline for specific conductivity and the specific conductivity results observed were typical of coastal systems, no objective is proposed for specific conductivity in Comox Lake.

Figure 8. Specific conductivity measured at one and five metre intervals in Comox Lake in the inlet, main and outlet basins during summer and main basin only during spring (Source: Epps and Phippen, 2011).



Specific Conductance (µS/cm)
25
27
29
31
33
35
37
39
0
10
20
30
40
50
60
70
80
90
100
Depth (m)
Main Basin March 2007
Main Basin May 2006
Main Basin August 2005
Main Basin October 2005

2.2.8. Nitrogen

Nitrogen concentrations were measured as dissolved nitrite (NO2) + dissolved nitrate (NO3). Nitrogen concentrations in Comox Lake at the inlet site averaged 0.034 mg/L with a range from 0.007 mg/L to 0.067 mg/L. Similarly, nitrogen concentrations in the main and outlet basin of Comox Lake averaged 0.037 mg/L (ranging from < 0.002 mg/L to 0.072 mg/L) and 0.033 mg/L (ranging from 0.006 mg/L to 0.060 mg/L), respectively. Concentrations of nitrate and nitrite were well below the existing aquatic life guidelines.

2.2.9. Phosphorus

Concentrations of phosphorus are generally low in Comox Lake. Total phosphorus concentrations were below the detection limit (2 µg/L) in almost half (44 of 90) of all samples from the three sites (inlet, main basin, and outlet basin). The highest concentrations measured ranged from 6 µg/L at both the main and outlet sites to 10 µg/L at the inlet site.

2.2.10. Metals

Total metals and dissolved metals have been measured at the three sites on Comox Lake. All concentrations of metals were below guidelines for drinking water and aquatic life (Comox Lake: WQ Objectives Attainment (2015-16), Dec 2020) with the exception of cadmium, which detection limit (0.1 µg/L) was greater than the aquatic life guideline of 0.01 µg/L (Nagpal *et al.* 2006). All samples had total cadmium concentrations at or below the detection limit; however, this likely reflects ambient conditions as there are no anthropogenic sources of cadmium within the watershed.

2.2.11. Coliform

The BC water quality guidelines for microbiological indicators developed in 1988 (Warrington 1988) include *E. coli*, *enterococci*, *Psudomonas aeruginosa*, and fecal coliforms. The BC MoE monitoring programs have traditionally measured total coliforms, fecal coliforms, *E. coli* and *enterococci*. As small pieces of fecal matter in a sample can skew the overall results for a particular site, the 90th percentiles (for drinking water) and geometric means (for recreation) are generally used to determine if the water quality guideline is exceeded. The 90th percentile of at least five weekly samples collected in a 30-day period should not exceed 10 CFU/100 mL for either fecal coliforms or *E. coli* (Warrington 2001).

Based on the 90th percentile data for both fecal coliforms (Table 14) and *E. coli* concentrations (Table 15) the inlet basin sites (1-3) are substantially lower than the outlet basin sites (4-7). The *E. coli* 90th percentiles for the inlet basin range from less than detection limits to a maximum of 16.4 CFU/100 mL, which is below the BC MoE drinking water guideline, with only one exceedance occurring in the summer of 2007.

Table 14. **Summary of 90th percentiles of fecal coliform concentrations (CFU/100 mL) for groups of five samples collected within a 30-day period (Source: CVRD 2016).**

Location Basin	Site #	Aug 11 - Sept 9, 2005	Oct 14 - Nov 10, 2005	Aug 30 - Sep 13, 2006	Oct 17 - Nov 22, 2006	Aug 9 - Sept 13, 2007	Oct 23 - Nov 22, 2007
Inlet	1	< 1.0	< 1.0	< 1.0	3.6	5.8	14.6
Inlet	2	< 1.0	9.8	< 1.0	4.8	16.4	3.6
Inlet	3	< 1.0	3.0	< 1.0	2.8	4.0	4.0
Outlet	4	< 1.0	18.6	3.2	42.2	49.2	190.4
Outlet	5	17.0	3.6	23.6	20.6	26.4	10.2
Outlet	6	1.0	4.8	6.0	5.6	192.4	172.0
Outlet	7	4.0	62.2	52.8	3.6	17.0	29.8

Table 15. **Summary of 90th percentiles of E. coli concentrations (CFU/100 mL) for groups of five samples collected within a 30-day period (Source: CVRD 2016).**

Location Basin	Site #	Aug 11 - Sept 9, 2005	Oct 14 - Nov 10, 2005	Aug 30 - Sep 13, 2006	Oct 17 - Nov 22, 2006	Aug 9 - Sept 13, 2007	Oct 23 - Nov 22, 2007
Inlet	1	< 1.0	1.6	< 1.0	3.6	5.8	4.2
Inlet	2	< 1.0	8.2	< 1.0	3.8	16.4	2.2
Inlet	3	< 1.0	2.2	< 1.0	2.8	4.0	2.2
Outlet	4	< 1.0	6.6	2.0	23.2	41.6	88.4
Outlet	5	3.8	4.0	10.4	12.6	23.6	5.0
Outlet	6	< 1.0	3.6	6.2	3.8	142.4	20.0
Outlet	7	1.0	38.6	44.2	2.6	17.0	20.6

For the outlet basin, the *E. coli* 90th percentiles for the sample periods range from below detection limits to a maximum of 142.4 CFU/100 ml. There were 12 exceedances of the BC MoE drinking water guideline for *E. coli* out of the 24 sample period results, including each of the 8 results in 2007. Occasional elevated values of both fecal coliforms and *E. coli* were observed (Table 16; Table 17). Possible contamination from hubs of activity including campgrounds and boat launches, as well as cabins, pets, waterfowl and wildlife could all be contributing to the bacteria levels at these locations.

Overall, the bacteriological results for the inlet basin are relatively low and are reflective of natural conditions. However, it appears that bacteria are a potential concern in the outlet basin of Comox Lake.

Table 16. **Summary of geometric means of fecal coliform concentrations (CFU/100 mL) for groups of five samples collected within a 30-day period (Source: CVRD 2016).**

Site #	Aug 11 - Sept 9, 2005	Oct 14 - Nov 10, 2005	Aug 30 - Sep 13, 2006	Oct 17 - Nov 22, 2006	Aug 9 - Sept 13, 2007	Oct 23 - Nov 22, 2007
1	< 1.0	1.3	< 1.0	1.6	1.6	3.6
2	< 1.0	2.5	1.0	1.8	2.2	1.6
3	< 1.0	1.7	1.0	1.3	1.4	1.4
4	1.0	2.7	1.5	7.9	13.8	20.9
5	4.2	1.4	5.5	6.2	7.5	7.0
6	1.0	1.9	2.9	3.0	64.7	8.9
7	1.4	11.7	7.4	1.6	2.6	3.9

Table 17. **Summary of geometric means of E. coli concentrations (CFU/100 mL) for groups of five samples collected within a 30-day period (Source: CVRD 2016).**

Site #	Aug 11 - Sept 9, 2005	Oct 14 - Nov 10, 2005	Aug 30 - Sep 13, 2006	Oct 17 - Nov 22, 2006	Aug 9 - Sept 13, 2007	Oct 23 - Nov 22, 2007
1	< 1.0	1.5	< 1.0	1.6	1.6	2.3
2	< 1.0	2.1	< 1.0	1.6	2.2	1.2
3	< 1.0	1.4	1.0	1.3	1.4	1.2
4	< 1.0	1.5	1.3	6.1	9.5	12.0
5	1.6	1.3	2.7	4.5	4.0	4.1
6	< 1.0	1.6	2.5	2.1	53.9	3.8
7	1.0	6.2	4.3	1.4	2.2	3.1

2.3. Fish Community

The Puntledge River contains at least 17 fish species, including Percids, Gasterosteids, Centrarchids, Salmonids, and Cottids (BC MoE 2024a). The following sub-sections provide a summary description of the anadromous salmonids, resident salmonids, and other fish species.

2.3.1. Anadromous Salmonids

All five Pacific salmon species are present, including both a summer and fall run of Chinook (*Onchorhynchus tshawytscha*), as well as Coho (*O. kisutch*), Chum (*O. keta*), Pink (*O. gorbuscha*) and Sockeye Salmon (*O. nerka*). It also supports both a summer and winter run of steelhead (*O. mykiss*) and an anadromous population of Cutthroat Trout (*O. clarkii*). Of these species, only Cutthroat Trout is listed provincially (i.e., blue-listed) and none of the species/populations are listed federally (BC MoE 2024b). However, both the summer and winter-run steelhead are considered to be at a high risk of extinction (McCulloch pers. comm. 2022), despite several years of juvenile stocking in the upper watershed in the early 2000s, through the provincial government's "*Living Gene Bank*[1]" program (BC Hydro Bridge CFWRP 2005).

The Puntledge River Hatchery is run by DFO and located in Courtenay, BC. They produce and release three of five salmon species in the Puntledge River watershed to enhance these populations. Pink salmon were enhanced until 2017. Sockeye salmon are not enhanced. However, an intensive transplant of up to 1 million eyed Sockeye salmon eggs into the upper Puntledge River watershed occurred annually between 1923 and 1930, but these efforts were unsuccessful in establishing a viable population.

2.3.2. Resident Salmonids

Resident salmonids that use the Puntledge River watershed, including Comox Lake, consist of Rainbow Trout, Cutthroat Trout, Dolly Varden (*Salvelinus malma*) and kokanee. These species/populations are not listed provincially or federally (BC MoE 2024b).

Kokanee may or may not have been present before the transplants while it is expected that the other three species are native, based on the following evidence:

- *Kokanee* – eggs were transplanted in Comox Lake in the 1920s but it is unclear if the species was present before this time; there are no records.
- *Cutthroat Trout* – eggs were planted in the upper watershed in the 1920s and 1930s, but the species was already present and part of a sport fishery by 1907 (Burridge 1954).
- *Rainbow Trout* – eggs were planted in the upper watershed in the 1920s and 1930s. This species was present in the river below the diversion dam before it was constructed (McCulloch pers. comm. 2022).
- *Dolly Varden* – spawning likely occurred in Reach B (headpond) before the dam was constructed (Bengeyfield and McLaren 1994).

2.3.3. Other Fish Species

Other fish species present in the Puntledge River watershed include: Coastrange Sculpin (*Cottus aleuticus*), Prickly Sculpin (*C. asper*), Threespine Stickleback (*Gasterosteus aculeatus*), Pacific Lamprey (*Lampetra tridentate*), and Western Brook Lamprey (*Lampetra richardsoni var. marifuga)*. Additionally, two Pumpkinseeds (*Lepomis gibbosus*), one Starry Flounder (*Platichthys stellatusand*), and one perch sp. (*Perca sp.*) have also been reported in the Puntledge River (BC MoE 2024a).

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[1]Rearing of captured steelhead smolts in the hatchery to spawning and then rearing and release of their progeny to rebuild stocks (BC Hydro Bridge CFWRP 2005).

Western Brook Lamprey are present in the Puntledge River watershed in Morrison Creek, which is a tributary of the Puntledge River. It is a rare non-anadromous form endemic to Morrison Creek (COSEWIC 2010). This species is unique in that it produces two different life history types from a single population - a non-parasitic “typical” form (no pronounced teeth), and a parasitic form (with teeth). Due to its extremely limited distribution and potential impacts resulting from ongoing development in the Morrison Creek watershed, the species has been designated as "endangered" by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2010). The other species/populations are not listed provincially or federally (BC MoE 2024b).

2.4. Land and Water Uses

2.4.1. Historical

2.4.1.1. Puntledge River

The east coast of Vancouver Island has been inhabited by Indigenous people for thousands of years. The Puntledge River watershed lies within the traditional territory of the K’ómoks Nation, which is currently an amalgamation of the K’ómoks, Pentlatch and Laich-Kwil-Tach cultural groups (BC Hydro 2003). As is often noted, the First Nations translation of the word K’ómoks (Comox) is “plentiful”, and the area is referred to as “the Land of Plenty” because of the abundant resources provided by the ocean and the surrounding land. A recent investigation and mapping of wooden stake fish traps in Comox Harbour points to the historical significance of this area to First Nations people (Greene *et al.* 2015). Comox Lake also served as an important trade route between the K’ómoks Nation and the Nuu-chah-nulth people on the west coast, while also providing sources of food, medicine, clothing, and other cultural and ceremonial materials.

The first European settlers arrived in the Comox Valley in 1862 when the Hudson Bay Company sent a small contingent of migrants to farm the area. Within the next year, the discovery of coal began to attract a greater number of settlers to the area. Over the next century, coal mining, forestry, fishing, and agriculture became the mainstay of the local economy. Today, the surrounding local communities have focused their economic growth on servicing a large retiree community and the military (Cairns 2017).

2.4.1.2. Courtenay River

The Courtenay River and its estuary have undergone significant changes since the arrival of the first European settlers in the 1860s. A variety of land uses including agricultural and urban development, forestry and log storage, recreational boating, commercial fisheries and their associated infrastructure have all had a negative impact on the ecological integrity of the estuary (Envirowest Consultants 2000). Dyking, channelization, dredging, stormwater, sewage and other contaminant discharges have resulted in a deterioration of water quality and loss of foreshore and saltwater marsh habitat. Comox (Dyke) Road, linking the city of Courtenay with the town of Comox, eliminated a significant portion of salt marsh habitat as land behind the road was no longer inundated during tides and was converted into agricultural fields. Between 1950 and 1988, an additional 14 hectares of intertidal and subtidal habitat were lost to agriculture, a sewage treatment lagoon, and urban and industrial development (Bravender *et al.* 2002).

2.4.2. Current

2.4.2.1. Water Uses

The Puntledge River is an important watershed for fish and wildlife but also for human uses within the whole Comox Valley. A query of current water license holders within the Puntledge River Watershed is summarized in Table 18. Of note, it excludes non-relevant water licenses that are: (1) not in the watershed, (2) related to groundwater/springs, (3) in Perseverance Creek, which is not used by Chinook Salmon, and (4) associated with water withdrawals of <10 m^3/day, which is unlikely to have an impact on the water supply. Of the four main water uses (i.e., conservation, irrigation, power and municipal), BC Hydro is the largest user of the resource, representing 79.8% of all water consumption while the remaining uses represent the rest (i.e., conservation: 17.9%, irrigation: 0.06%, and municipal 2.2%). Current land uses are shown in Map 3.

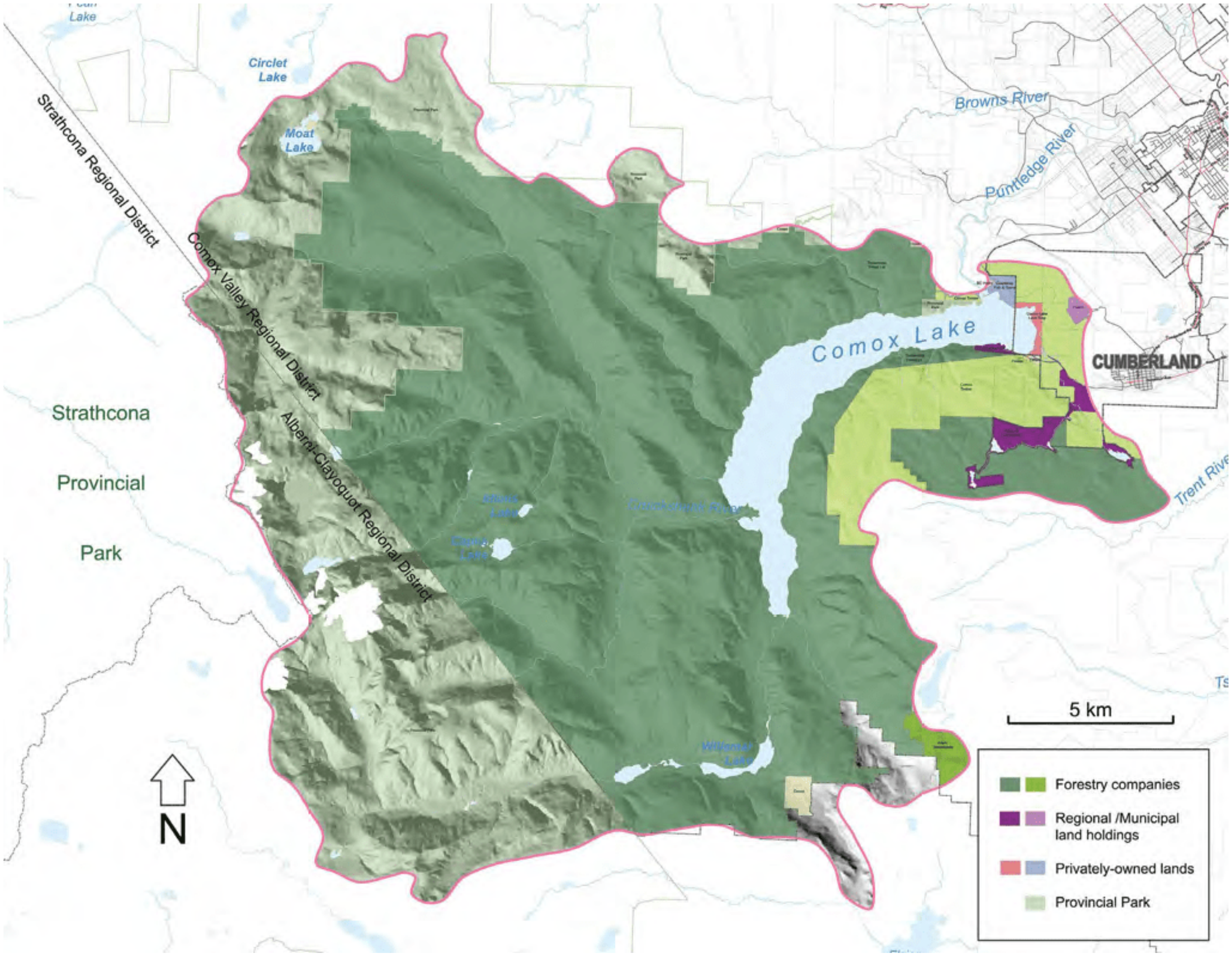
Table 18. Summary of relevant current water licenses for the Puntledge River watershed (Ref: BC Water Rights Database).

Purpose	Licensee[1]	Licensed Quantity (m^3/s)[2]	Source Name
Conservation	Fisheries and Oceans, Canada	1.42	Browns River
		0.58	Puntledge River
		0.58	
		0.85	
		0.42	
		0.51	
	Courtenay and District and Game Protective Association	0.00425	Supply Spring
	BC Hydro and Power Authority	2.83	Puntledge River
	Total	**7.19425**	
Irrigation	Viewfield Farms Ltd.	0.0005	Puntledge River
		0.0027	Happy Creek
	Eloisa H. Tobacca	0.0002	Puntledge River
	George Erler and Marla Simone Limousin	0.0001	Courtenay River
	Ducks Unlimited	0.007	
		0.007	
		0.007	
	Dan Anad	0.0005	Puntledge River
	Total	**0.025**	
Power	BC Hydro and Power Authority	28	Puntledge River
		3.73	
	Total	31.73	
Municipal	Comox Valley Regional District	0.29	
		0.29	
		0.16	
		0.16	
		0.0036	Courtenay River
		0.0018	
	Total	**0.9054**	

[1]Excludes non relevant licensees (i.e., not in the watershed, groundwater/springs, Perseverance Creek because Chinook don't use the creek and licensees that have little or no impact (i.e., < $10m^3$/day).

[2]Provided in a standardized unit for comparative purposes.

Map 3. **Land uses in the Comox Lake Watershed (Source: CVRD Watershed Protection Plan).**



2.4.2.2. Hydroelectric

In 1912, the Wellington Colliery Company Ltd. (Dunsmuir) constructed a small hydro-electric generation facility on the Puntledge River to supply power to local coal mines. This project consisted of an impoundment dam at the outlet of Comox Lake, a diversion dam and intake 3.7 km downstream and the overland penstock, which diverted water to a powerhouse located 5.5 km downstream (Map 1). The impoundment of Comox Lake raised the original lake elevation by about 8 m (BC Hydro 2000). The water license granted to the company for this project allowed 29.7 m^3/s (1,050 ft^3/s) to be diverted from the river. However, the original facilities redirected only 8.5 m^3/s (300 ft^3/s) of water from the river to generate 7 MW of power. A fishway was included in the original diversion dam but not the impoundment dam. A timber fishway was constructed at the impoundment dam in 1927 (Bengeyfield and McLaren 1994) and this was later replaced with a permanent concrete fishway in 1946.

In 1953, the installation was purchased by the British Columbia Power Commission, a predecessor of BC Hydro, and underwent an expansion between 1955 and 1958. In 1955, the power plant was refurbished with a new 24 MW generating unit, requiring the full water license grant to be utilized. The impoundment dam was reconstructed in 1957, and a new diversion dam and intake structure was completed in 1958. The new diversion dam was 0.45 m higher than the original dam, which caused further backflooding in the headpond reach (Bengeyfield and McLaren 1994).

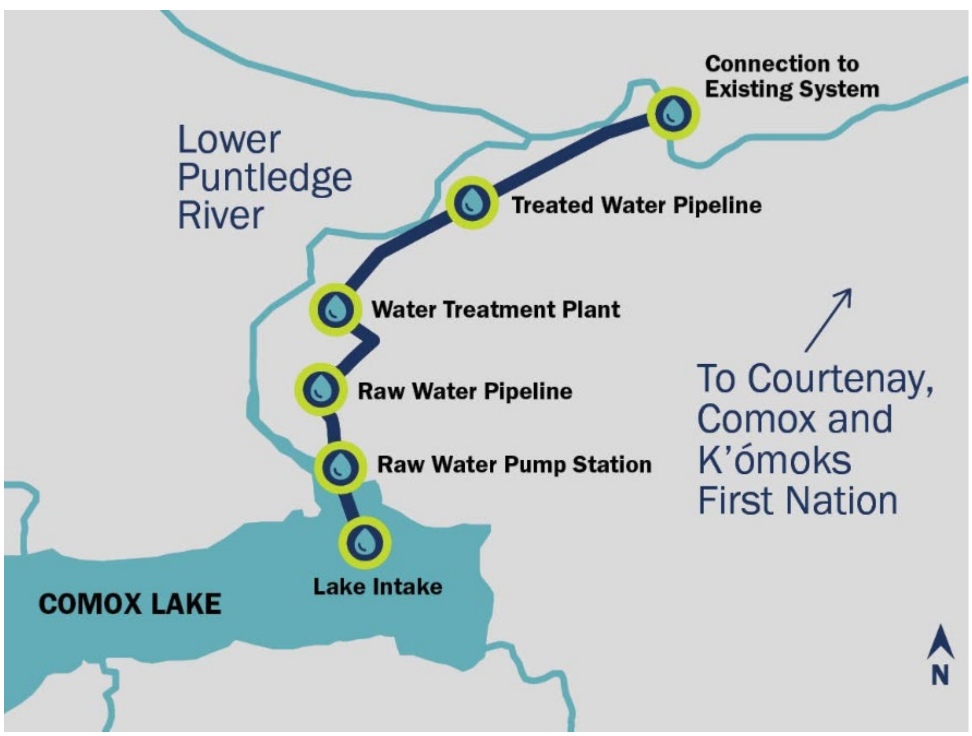
The Puntledge Generating Station provides Vancouver Island with approximately 156 GWh of electricity annually, equivalent to supply approximately 16,000 homes (BC Hydro 2003). Annual demand, calculated as the maximum licensed diversion/withdrawal, multiplied by 365 is 937 million m^3/year. In addition to the amount diverted for power generation, BC Hydro also provides between 1.4 m^3/s and 2.8 m^3/s for conservation purposes, supplying water to the Upper Puntledge hatchery spawning and rearing channels and the fishways, located adjacent to the Puntledge Diversion Dam.

Anadromous fish stocks in the Puntledge River, particularly those associated with the upper watershed were likely affected by construction of the first hydroelectric facilities in 1912. However, the magnitude of these impacts is difficult to quantify due to the lack of early salmon escapement records for comparison. The complete obstruction to migration into Comox Lake would have had significant impacts on anadromous stocks that utilized the lake, until a timber fishway was installed in the dam in 1927. Redevelopment of the hydroelectric facilities in the 1950s had serious detrimental impacts on salmonid production in the watershed, and these were most significant on summer-run populations of Chinook and steelhead (Anon. 1958; Angus 1962; Hourston 1962; Rimmer *et al.* 1994). The increase in the elevation of the diversion dam flooded the most important spawning reach in the lower Puntledge River for these species, and this habitat was further impaired by a sedimentation event during redevelopment of the impoundment dam. The increase in water diversion into the penstock entrained a greater proportion of seaward migrating juveniles and smolts into the turbine, while the lower flows down the river impeded adult upstream migration, and juvenile rearing habitat for other species.

2.4.2.3. Municipal Water Supply

The Puntledge River watershed is the source of drinking water for over 49,000 residents in the City of Courtenay, Town of Comox and the CVRD. The Comox Valley Water System originates in Comox Lake where water is withdrawn from the BC Hydro penstock near the Puntledge Generating Station about 15 km downstream (Figure 9). A standby pump station located beside the Puntledge Generating Station is used only when the BC Hydro penstocks are undergoing repairs or maintenance, a few weeks a year. On average the system withdraws 0.29 m^3/s from the penstock, with a maximum daily allowance of 70,463 m^3. This amount represents less than one percent of the total water use outlined in Table 18. The combined water withdrawal of all other smaller license holders along the Puntledge River is negligible, amounting to less than one half of one percent. Construction of a lake intake to allow domestic water withdrawal directly from the lake and filter it at a new filtration plant is complete. Raw water is chlorinated before entering the distribution system. In 2018, UV disinfection was added to achieve dual disinfection.

Figure 9. CVRD Domestic pipeline water supply and treatment plants (Source: CVRD 2023).



2.4.2.4. Agriculture

The agricultural potential offered by the fertile soils and moderate climate of the Comox Valley was recognized early when the first European settlers began arriving to the region in the 1860s. Today the Comox Valley contains 20,000 hectares of land protected within the Agricultural Land Reserve (ALR), of which approximately half is currently used for agriculture production. The 2001 Census of Agriculture indicates the valley had 445 farms in operation, encompassing a diverse mix of farm activities. The majority of the prime agricultural land lies outside of the Puntledge River watershed. Within the watershed however, most of the production is located below the Puntledge diversion dam. Some of the most dramatic modifications to the habitat surrounding the estuary were a result of agricultural activities. Dyking along the estuary resulted in the conversion of over 660 acres of salt marsh habitat into agricultural land that is still farmed today (Burns 1976 cited in Hamilton *et al.* 2008).

2.4.2.5. Forestry

The potential hazards associated with forestry activities can cause significant impacts on adjacent streams and water bodies. The removal of trees can decrease water retention times within the watershed and result in a more rapid response to precipitation events and earlier and higher spring freshets. Road construction can change drainage patterns, destabilize slopes, and introduce high concentrations of sediment to streams. Water quality can also be impaired by the input of hydrocarbon or other chemicals from the operation of logging machinery or due to accidents, and from increased nutrient loads from forest fertilization.

Forestry development in the Puntledge River watershed has been extensive, but now is the dominant land use activity in the upper watershed. Currently, most of the uplands surrounding Comox Lake are privately owned lands managed for forestry, while in the lower portions of the watershed, a number of smaller privately owned properties are also managed for forestry amongst other land uses. Mosaic Forest Management (MFM) and Hancock Timber Resource Group are the two largest forest companies in the watershed. MFM manages most of the Comox Lake watershed outside of Strathcona Provincial Park, including the bed and the easterly shore of Comox Lake. Hancock owns most the easterly part of the Comox Lake Watershed and part of the Lower Puntledge River Watershed. The majority of forestry activity takes place on privately owned lands and is governed by the Private Managed Forest Land Act. Approximately 61% of the Comox Lake watershed is currently under active forestry management (Table 19), including lands managed by Mosaic (formerly Timber West and Island Timberlands) and Hancock.

Table 19. Summary of land use within the Comox Lake Watershed (Source: Benjamin and Varashelyi 2006).

Land Use	Area (ha)	% of Total Watershed Area
Forestry	28,075	60.84
Park	15,141	32.81
Water	2,250	4.88
Crown Land	412	0.89
Private Land	91	0.20
BC Hydro	11	0.02
Municipal Land	158	0.34
Road Right-of-Way	8	0.02
Total	**46,146**	**100**

The forest harvest history of Puntledge River was reconstructed based on the Provincial Vegetation Resources Inventory (VRI) public dataset (https://www2.gov.bc.ca/gov/content/industry/forestry/managing-our-forest-resources/forest-inventory/data-management-and-access) and the 2022 data release by scientists at Ecofish Research. The VRI provides forest age estimates for forest stands in the forest management land base (FMLB) in the Puntledge River watershed, including private lands (Map 4). Based on the forest age data attribute in the VRI the history of disturbance can be reconstructed back to 1880, as well as equivalent clearcut area (ECA) that accounts for forest regrowth and recovery of hydrologic function (Hudson and Horel 2007) (Figure 10, Figure 11). Cumulative disturbance is a measure of the proportion of the FMLB that has been disturbed (or harvested) over time without accounting for any recovery through re-growth of vegetation. The age datasets do not account for second or third harvest cycles, and therefore historical disturbance may be underestimated. Almost 75% of the FMLB has been disturbed in the Puntledge River watershed with only 25% old growth left (according to this analysis, and excluding high elevation stands). Significant forest harvesting appears to have begun in the 1930s in the watershed, with periods of higher forest harvest in the early 1970's, early 1990's, and 2010s (Figure 10). The percent permanent disturbance in the watershed (red line in Figure 10), which includes all impacts (i.e., urban, mining, agriculture) is underestimated in the VRI and needs to be reviewed and likely re-adjusted.

Equivalent clearcut area trends (Figure 11) are similar to cumulative disturbance, although ECA accounts for hydrologic recovery through forest regrowth. ECA was estimated using the stand age hydrologic recovery curve presented in BC's aquatic cumulative effects protocol (MoECCS and FLNRORD 2020). The Province is using ECA as a harvest intensity criterion in BC; for example, guiding forest harvest targets for important salmon streams in the Great Bear Rainforest (Coast Information Team 2004). Under the Great Bear Rainforest Order, which sets the guidelines for forestry in the Central and North Coast regions, the first objective for maintaining ecological integrity and human well-being of Important Fisheries Watersheds states: "(1) Maintain hydrological and fluvial processes in watersheds within the range of natural variation by maintaining an Equivalent Clearcut Area of less than 20% in each of the Important Fisheries Watersheds shown in Schedule E." The ECA in the Puntledge River has been above or near this 20% ECA threshold for over 50 years since 1971 and was between 10% to 15% ECA from 1945 to 1970 (Figure 11). This analysis can be partitioned out into different sub-regions of the Puntledge watershed (e.g., Upper watershed (Mosaic) and Lower Puntledge R). These two areas likely have different rates and levels of impact (Hocking, pers. comm. 2024).

Ecofish Research analysis of Chum and Pink Salmon has found impacts to Chum from forest harvesting as measured by ECA and cumulative disturbance, with ECAs >20% causing at least 25% decline in Chum recruits per spawner produced per year (Hocking *et al.* 2017; Hocking *et al. in prep*). The impact on summer Chinook in the upper watershed is probably at least the same as found with Chum salmon. Changes in the structure and functioning of watersheds from forestry can affect the survival and growth of salmon during their life cycle in freshwaters through various mechanisms. Examples from the literature include increases in accumulation of fine material and substrate embeddedness that reduces egg-to-fry survival (Bjornn *et al.* 1977; Scrivener and Brownlee 1989; Hartman *et al.* 1998; Bjornn and Reiser 1991), increases in peak flows that can cause stream channel scour and either salmon egg dislodgement or entombment (Scrivener and Tripp 1998), alterations to the stream channels themselves such as channel over widening and loss of large woody debris, pool habitat, and structural complexity that can reduce salmon rearing habitat capacity (Tschaplinski and Pike 2017), and shifts in stream temperatures and low flows (Holtby 1998; Reid *et al.* 2020).

Map 4. **Forest management land base (FMLB; shown in green) for the Puntledge watershed. The black areas are excluded from FMLB because they are high elevation habitat, lakes, agriculture, or permanent disturbances.**

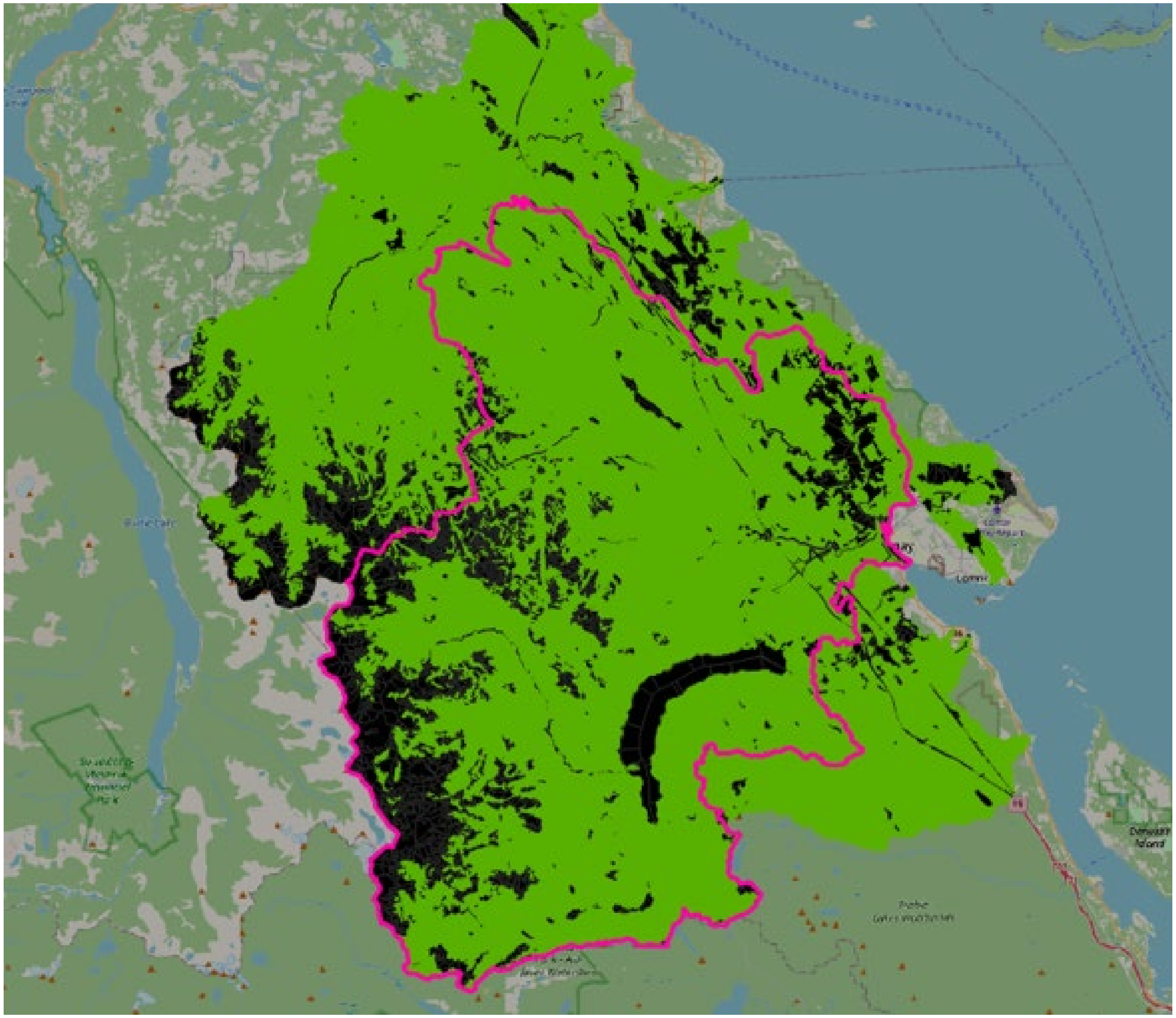


Figure 10. The history of disturbance in the Forest Management Land Base in the Puntledge River watershed: 1880 to 2022.

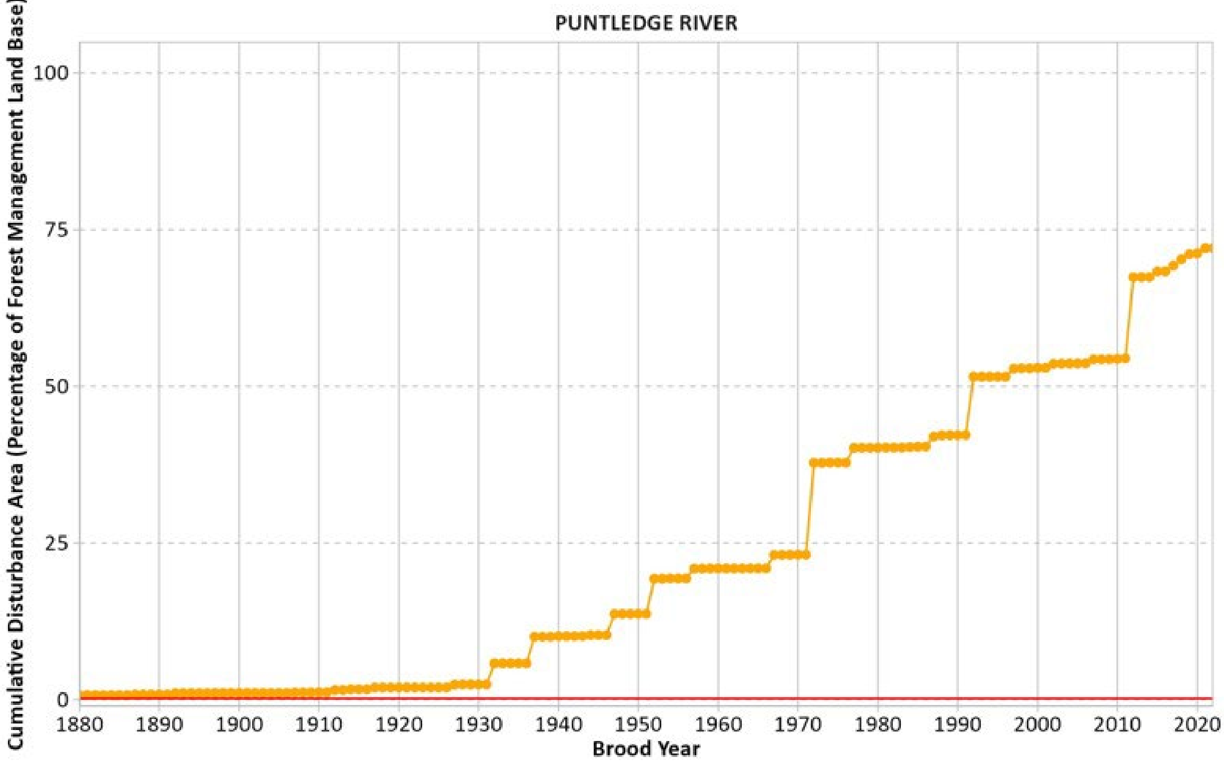
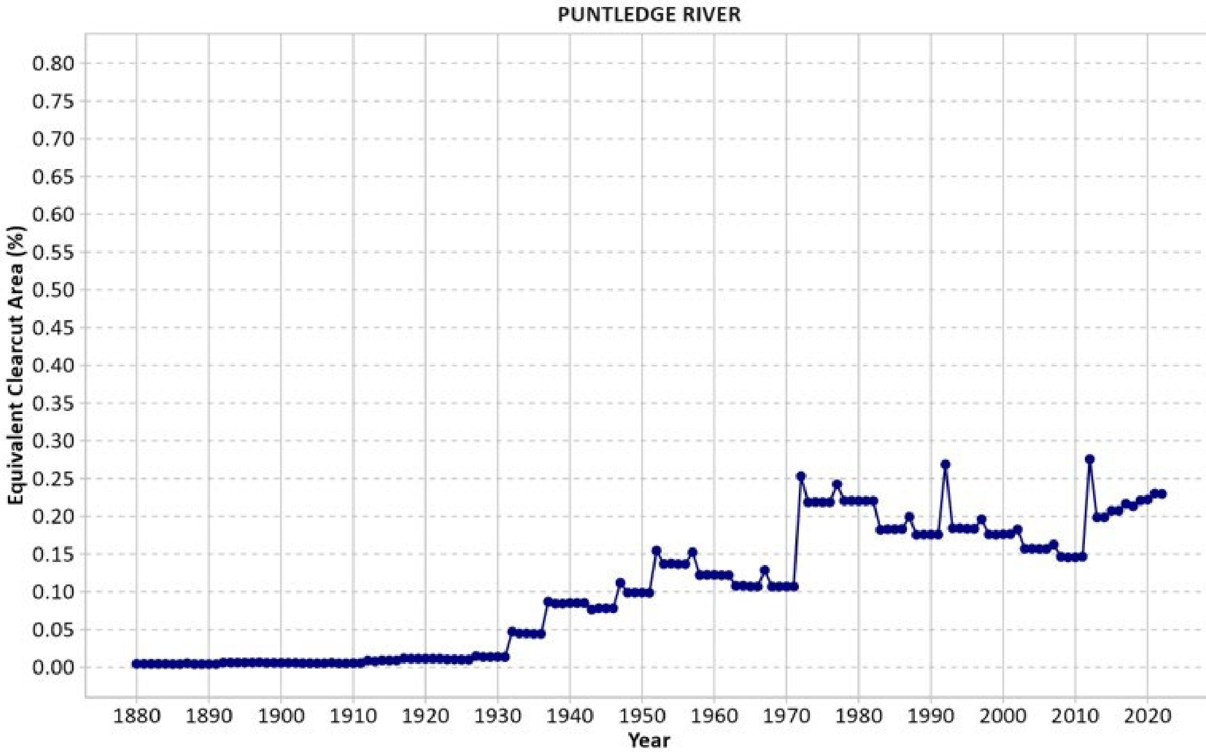


Figure 11. The history of Equivalent Clearcut Area in the Puntledge River.



2.4.2.6. Mining

Historically, coal mining has occurred in the Browns River watershed, below the Comox Lake watershed (Epp and Phippen 2011). There are claims in the upper Cruikshank River watershed that contain a number of minerals, including gold, silver, copper, molybdenum, lead, and zinc. However, these have not been developed, and development would need to undergo environmental impact assessments to ensure that watershed resources (including water quality) were not significantly impacted.

2.4.2.7. Recreational Use on Comox Lake

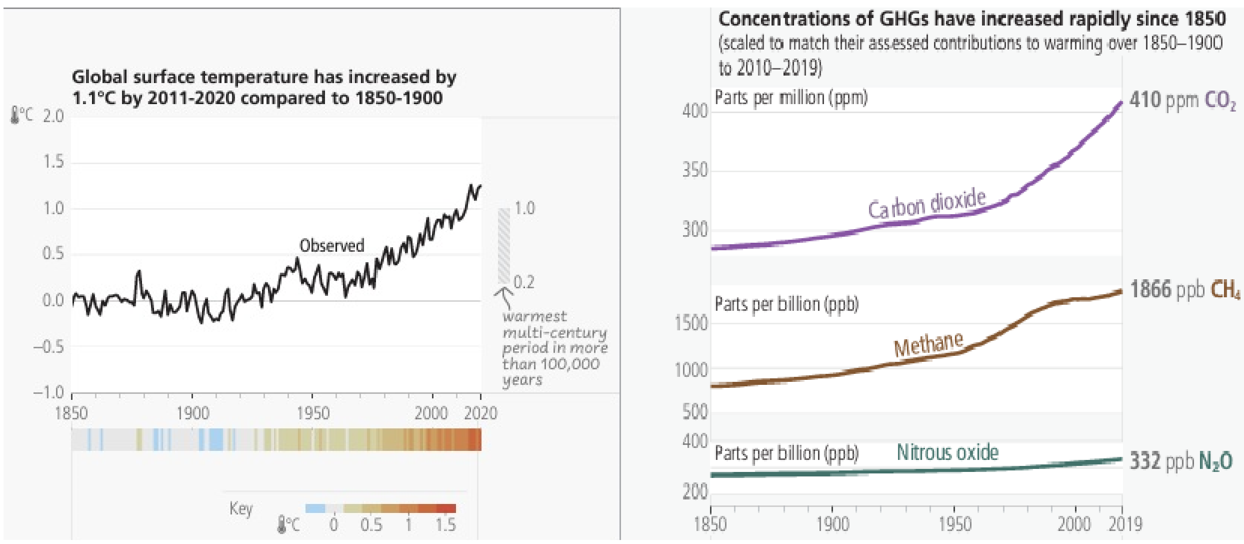
Comox Lake is a popular recreational area. There are 77 cabins on the lake (70 of which are used seasonally, and the remaining are used year-round). There are two designated campgrounds: the Cumberland campground on the south shore of the outlet basin and the Courtenay and District Fish and Game Protective Association campground on the north shore of the outlet basin. There are day-use beaches for swimming, walking, fishing, and boating (power boats as well as canoes and kayaks have access to the lake). Boat launches are located at both campgrounds.

2.5. Climate Change Impacts to the Puntledge River

Changes in climate patterns occur over short- and long-term periods. Ocean circulation patterns, such as the El Niño Southern Oscillation (ENSO) and Pacific Decadal Oscillation (PDO), are the primary short-term events that fluctuate on yearly and multi-year timescales and are driven by the transfer of heat between the oceans and the atmosphere. On the longer term, progressive changes in Earth's orbit around the sun trigger ice ages approximately every 100,000 years. Other cycles occur over millions of years (Jost and Weber 2012).

Human activities are accelerating climate change. Global surface temperature has increase 1.1°C between the periods of 2011-2020 and 1850-1900 and greenhouse gases increased rapidly over the same period (Figure 12, IPCC 2023). Although precipitation records are less reliable, climatologists estimate that precipitation over North America has increased by about 10% during the 20th century (Jost and Weber 2012).

Figure 12. Changes in Global Surface Temperature and Increased Concentrations of GHGs in the Atmosphere (IPCC 2023).



2.5.1. Historical Trends in Temperature and Precipitation

Over the last century, all regions of British Columbia warmed by an average of about 1.2°C (air temperature) (Jost and Weber 2012). Most of the increase was a result of a rise in minimum temperatures (Figure 13). The mean annual temperature is projected to increase by 1.4 to 3.7°C over the next century, which is greater than the range of historical variability for this period of time. All carbon emission scenarios project increasing temperatures in all seasons in all regions of British Columbia (Figure 14). In the Comox watershed the warming will be more evenly distributed throughout the year compared to the rest of the province.

In Courtney, BC (on the mouth of the Puntledge River), mean annual temperatures have increased since 1979 (Figure 15). Notably, nine of the last ten years have been warmer than average over this period of monitoring. Monitoring of temperatures over summer months in this region indicate that warm temperature anomalies are becoming increasingly prevalent (Figure 16). July has been warmer than normal since 2011 while August has been warmer than normal 11 times out of 12 years since 2009.

Over the past century in BC, annual precipitation has increased by about 20 percent (Figure 17). Most of the precipitation increase occurred in fall, winter, and spring, with the highest increases in the northern interior and no change in the southwest (Figure 17). It is projected that much of BC will get modestly wetter (i.e., from 0 to 18%, Table 20). Contrary to temperature projections, however, the projected increase in precipitation is within the range of historical variability. Precipitation increases are projected to be greatest in fall, winter, and spring (Figure 17). In summer, the southern portion of the province, and particularly the southwest, will likely become drier. Indeed, records from 1979-2022 for July and August in the Courtenay Region suggest this to be the case (Figure 16; Meteoblue 2022).

Historic records show a reduction in peak winter snow accumulation over the past 50 years. On average, the peak snow water equivalent (SWE) of 73 long term recorded snow courses dropped by about 18 percent. Vancouver Island dropped by 17 percent. One-half to two-thirds of the reduction in peak SWE over the past 50 years correlates to natural ENSO and PDO cycles. The PDO shift from a cold to a warm phase in 1976 had the most significant effect (Jost and Weber 2012). After factoring out these effects of this natural climate variability, the province-wide SWE trends and snowpack decline just four percent (Table 20). In some regions, adjusting for ENSO and PDO reverses the trend. However, it is important to note that most SWE analysis relies on data taken at mid elevations. Models suggest that colder, higher elevation areas, which are less sensitive to warming, are predicted to have an increase in peak SWE due to increases in precipitation (Jost and Weber 2012).

Glaciers across the province lost about 11% of their area between 1985 and 2005. Coastal glaciers lost less area than interior glaciers, but absolute volume loss was larger in the Coast Mountains than in the Columbia region or the Rocky. The glacier in the Puntledge Watershed only represents a very small percent of the total watershed area and the impact of glacier melt on annual flow volumes is relatively minor.

These changes in temperature and precipitation are having associated impacts on freshwater systems as is evident by a modest historical increase in annual inflows into BC Hydro's reservoirs (though trends are small and statistically not significant). Fall and winter inflows have increased in almost all regions, and there is some evidence for a slight decline in late-summer flows for basins primarily dependent on glacial melt and/or seasonal snowpack. The year-to-year variation in annual reservoir inflow has not changed (Jost and Weber 2012).

On the South Coast (Vancouver Island and Lower Mainland watersheds), more of the precipitation will fall as rain and snow will become less important. Fall and winter flows will increase, and spring and summer flows will decrease. BC Hydro will likely see a modest increase in annual water supply for hydroelectric generation. The Campbell River area and likely most Coastal watersheds will see negligible changes to annual water supply into the 2050s (Jost and Weber 2012).

Figure 13. **Annual trends in (a) minimum, (b) mean, (c) maximum temperature and (d) precipitation for British Columbia. Results are based on 1900 to 2004 data and calculated as degree Celsius change per century. Black solid circles indicate statistically significant results (95% confidence level). Open circles show the location of Adjusted Historical Canadian Climate Station sites (AHCCD) (Source: CANGRID (50 km) data, adapted from Zhang *et al.* 2000).**

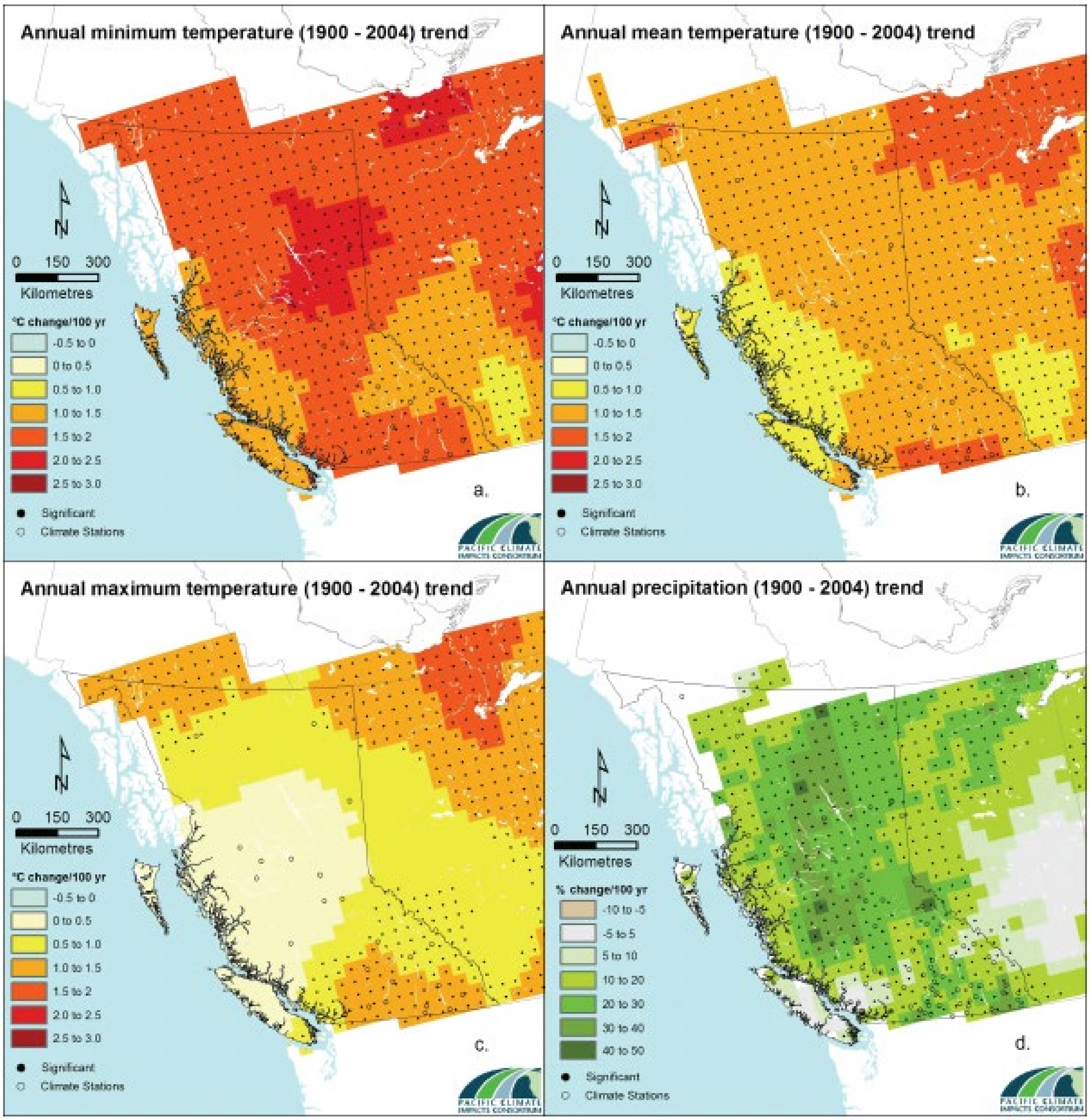


Figure 14. **Seasonal trends in minimum temperature (a) winter, (b) spring, (c) summer and (d) fall for British Columbia. Results are based on 1900 to 2004 data and calculated as degree Celsius change per century. Black solid circles indicate statistically significant results (95% confidence level). Open circles show the location of Adjusted Historical Canadian Climate Station sites (AHCCD) (Source: CANGRID (50 km) data; adapted from Zhang *et al.* 2000).**

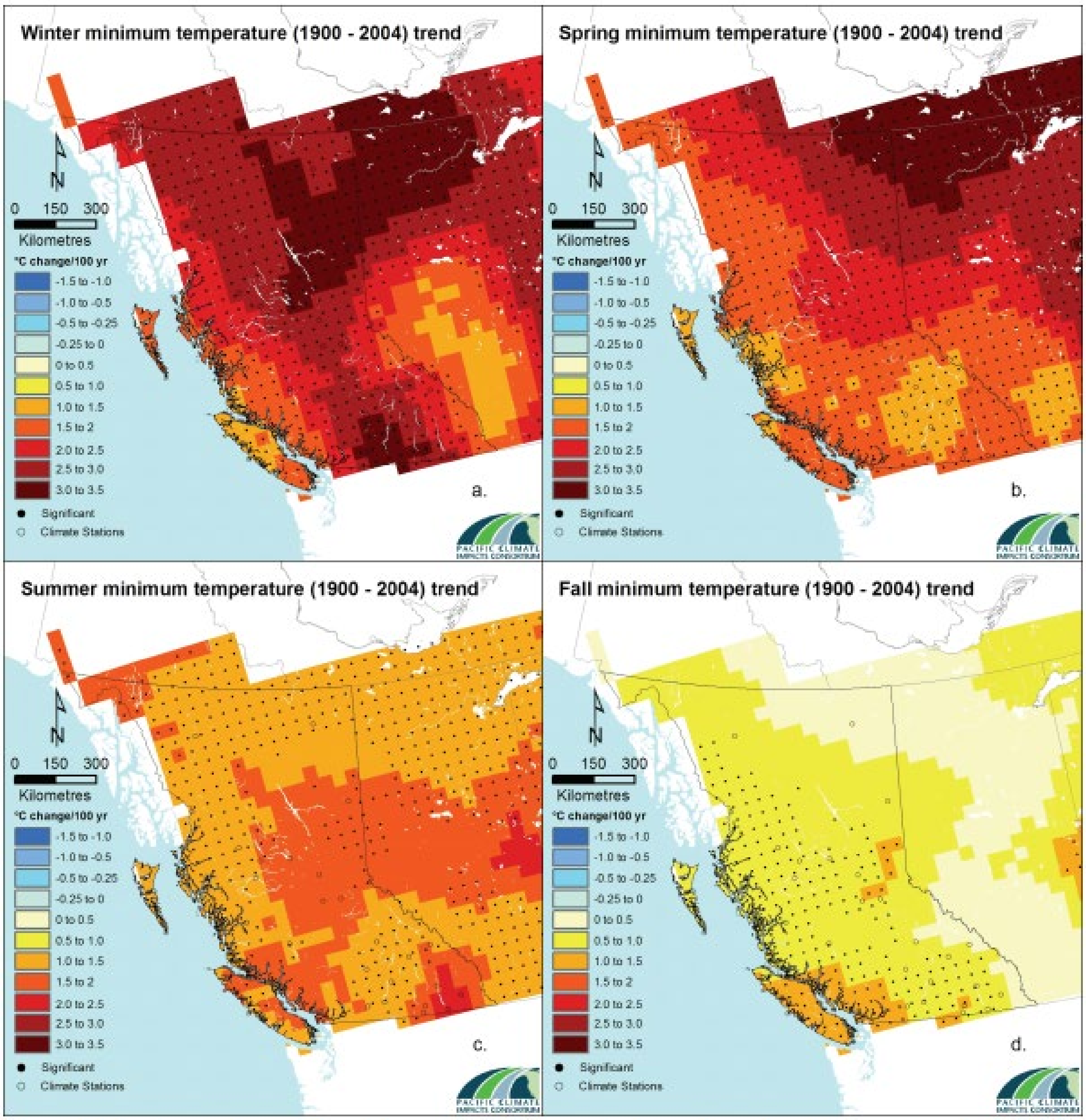


Figure 15. **Mean annual temperature for the larger region of Courtenay, BC. The dashed blue line is the linear climate change trend (Source: Meteoblue 2022).**

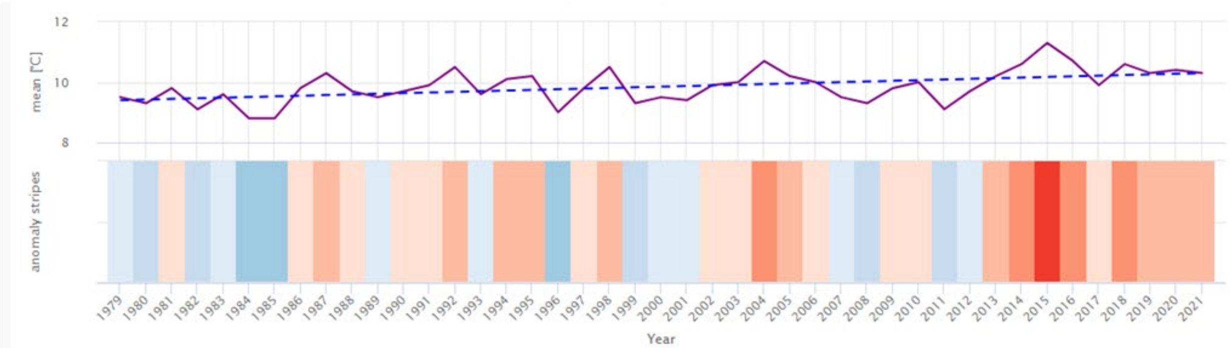
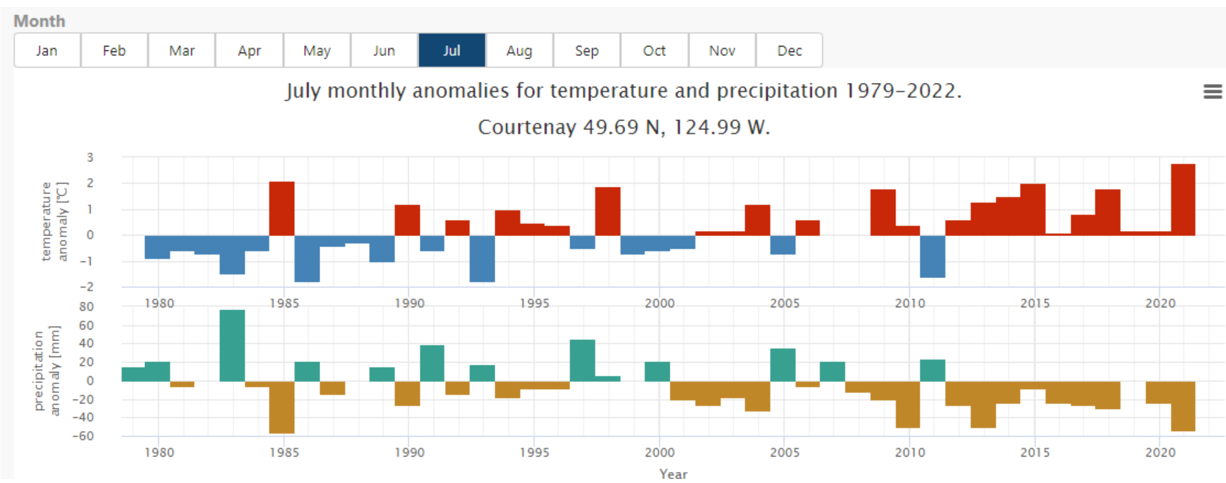


Figure 16. **Temperature and precipitation anomalies for the month of (a) July and (b) August from 1979 to 2021 in Courtenay, BC (Source: Meteoblue 2022).**

a)



b)

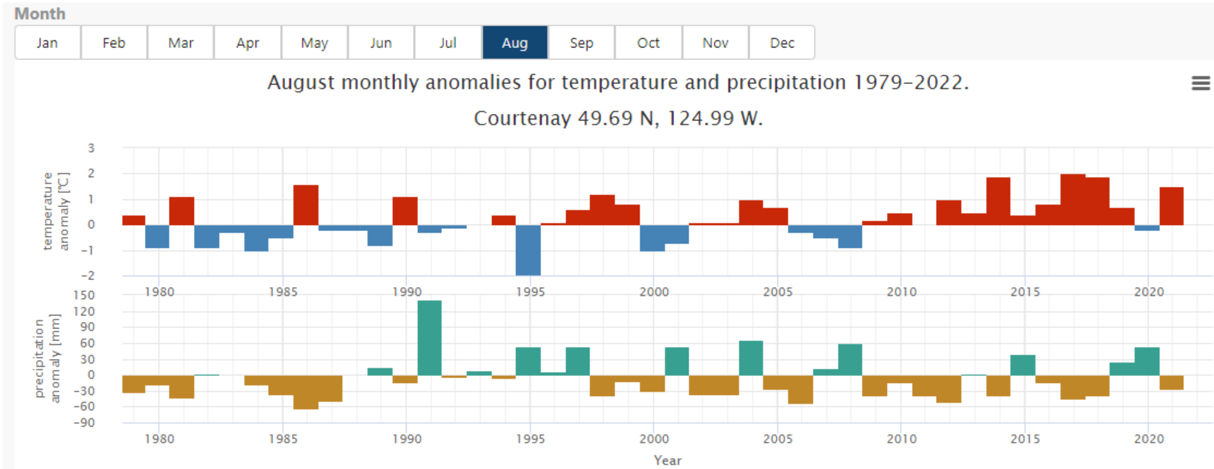

Month
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
August monthly anomalies for temperature and precipitation 1979-2022.
Courtenay 49.69 N, 124.99 W.
temperature anomaly [°C]
precipitation anomaly [mm]
1980
1985
1990
1995
2000
2005
2010
2015
2020
Year

Figure 17. **Seasonal mean precipitation changes in the 2050s (2041-2070) relative to the 1961-1990 baseline period (Source: Schnorbus *et al.* 2011).**

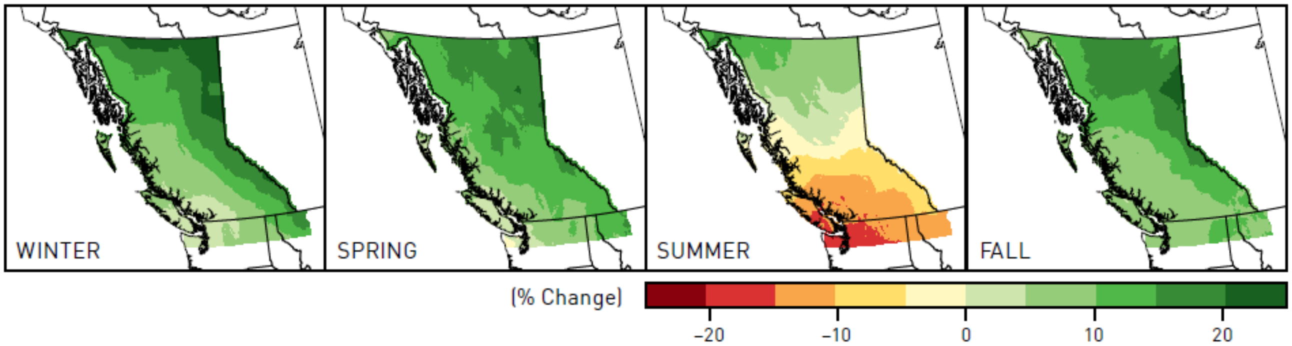


Table 20. **Snow water equivalent (SWE) trends by basin/region before and after adjustment for natural climate variability (Source: Jost and Weber 2012).**

	UNADJUSTED DATA			ADJUSTED DATA*	
BASIN / REGION	Mean [mm]	Change [mm]	Change (%)	Change [mm]	Change (%)
Peace	399	-8	-4	33	7
Columbia	646	-87	-20	7	-5
Kootenay	365	-91	-23	-21	-6
Middle Fraser	213	-82	-47	-31	-27
South Coast/ Vancouver Island	1202	-261	-17	-65	-4
British Columbia (overall)	474	-71	-18	0	-4

% Change Key: little or no change: -5% and 5% increase: > 5% decrease: < -5%

2.5.1.1. Stream Flow Projections

Stephanie Smith, BC Hydro Manager of Hydrology & Technical Services, works with the Pacific Climate Impacts Consortium for all climate and hydrologic future scenarios and has completed scenarios for the Upper Campbell River watershed. She suggested that the following trends could be applied to the Puntledge River (Smith pers. comm. 2022). In addition, BC Hydro has been working on climate change modelling for the Comox watershed, but it is not yet completed.

By 2050, the Campbell River at Strathcona watershed is expected to change from a hybrid snow-rainfall to a rainfall dominated regime (Table 21). Snowfall will decrease, and flows from October to April will increase, with a substantially reduced spring freshet. Global climate models (GCMs) consistently predict the highest flow increases in January and the largest decreases in June. No significant changes to annual inflow volumes are projected (Table 21; Pacific Climate Impacts Consortium 2011).

The greatest changes to seasonal flow regimes can be expected for coastal watersheds. There, rainfall-runoff processes will very likely become dominant over snowmelt. Hybrid rainfall- and snowmelt-dominated watersheds will turn into rainfall-dominated watersheds. With only marginal precipitation increases, the region will see a decline of basin-wide snowpack and consequently a reduction in spring runoff (Figure 18).

In Comox Lake (headwaters of the Puntledge River), winter and spring inflows are ~30-50 m^3/s (Figure 19). Comox Lake natural inflows appear to be driven by precipitation in the fall and winter and snowmelt in the spring. Between mid-June and October, inflows decline to ~10 m^3/s or less. Climate change predictions forecast higher winters and lower summer discharges for 2050 and 2080 (Figure 20; Healey *et al.* 2018). The storage of water in Comox Lake will compensate for climate change-related flow impacts and allow for the mitigation of low natural flows during the summer in the Puntledge River to meet flow requirements in the WUP during most years. (Healey *et al.* 2018).

Table 21. Seasonal and annual inflow anomalies for select BC Hydro watersheds for the 2050s relative to 1961-1990 average. Percentiles refer to the range in projections under different emission scenarios and GCMs. (Source: Pacific Climate Impacts Consortium 2011).

REGION	WATERSHED		WINTER	SPRING	SUMMER	FALL	YEAR
SOUTH COAST	Strathcona	5 percentile	45%	-10%	-68%	8%	-10%
		50 percentile	52%	6%	-64%	10%	1%
		95 percentile	42%	13%	-43%	12%	8%

Figure 18. **Hydrographs showing median monthly discharge rates for the Campbell River at Strathcona Dam. The black line shows the modelled historical pattern of streamflow while the blue line represents the median discharge for all models and emissions scenarios used in the study. The blue shaded areas illustrate the range of results from the various global climate models and emissions scenarios used. The bottom graph shows the same results as the top graph but presents them as a change in discharge rate compared to the baseline period (Zwiers *et al.* 2011).**

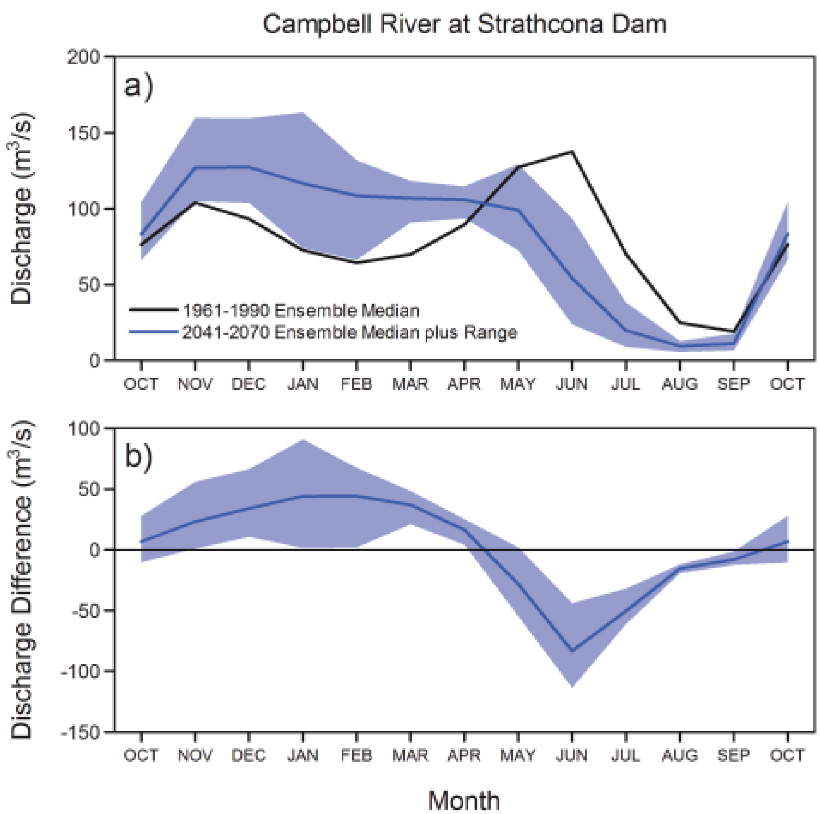


Figure 19. **Natural inflows into Comox Lake (1963-2016) (Healey *et al.* 2018).**

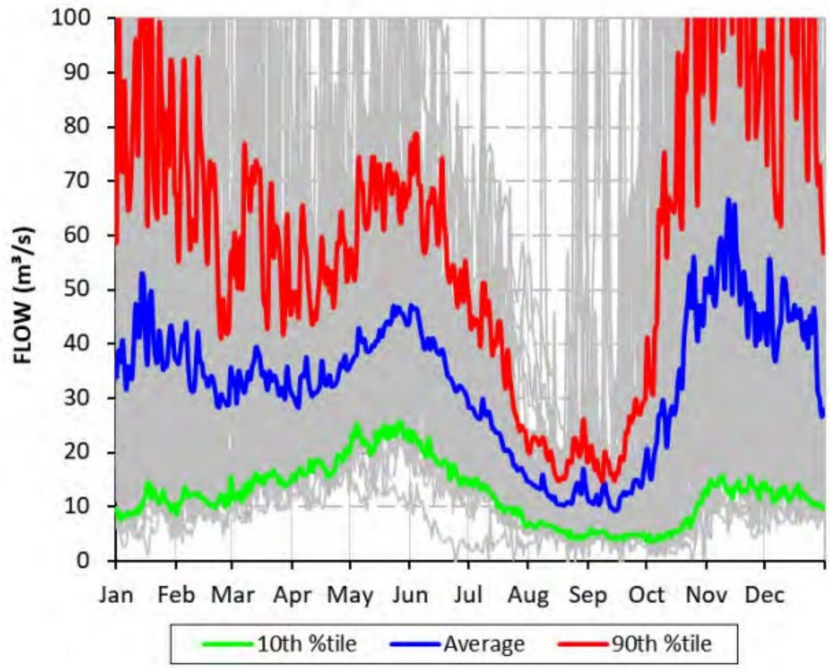
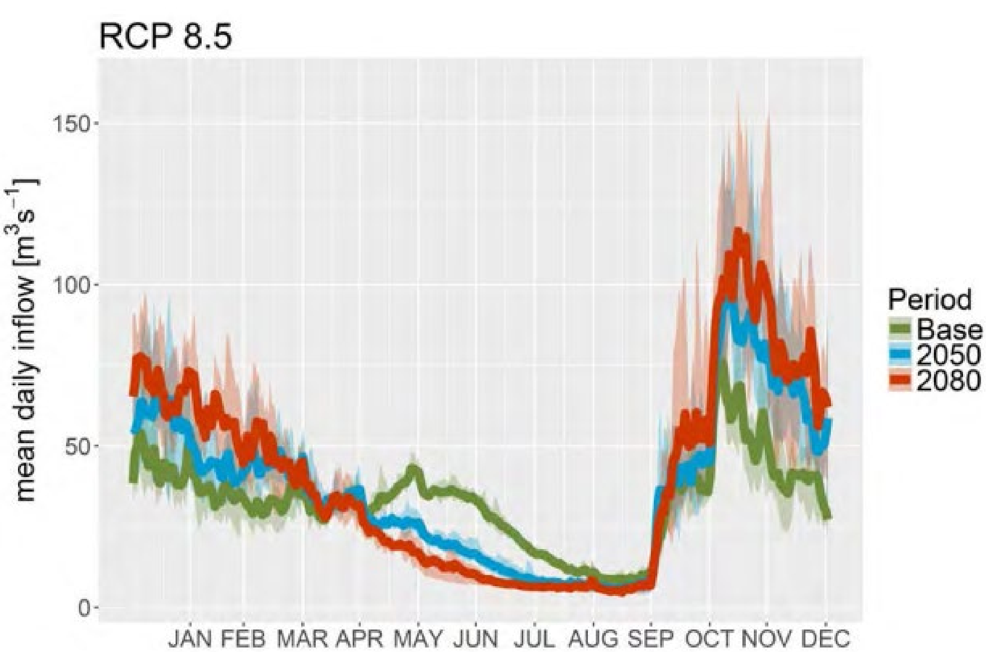


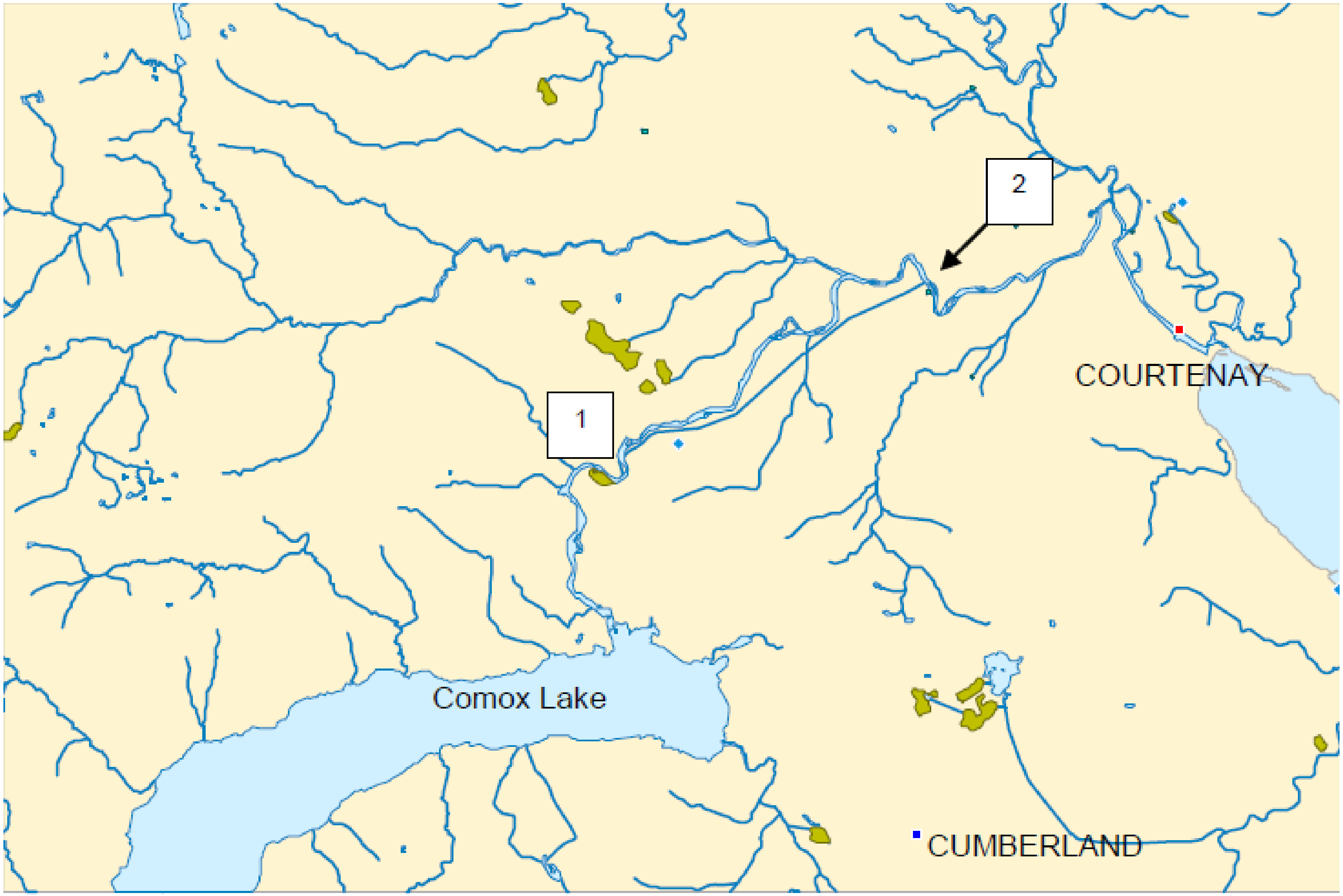
Figure 20. **Mean daily inflow to Comox Lake under historic (base) and forecasted future climate conditions (Healey *et al.* 2011).**



2.5.1.2. Water Temperature Trends

DFO has been operating continuous temperature loggers on the lower Puntledge River at the upper and lower hatchery sites since 1978 (Map 5). The Upper Site was built to aid in the rebuilding of Chinook stocks impacted by hydro power generation. In 1978, the Lower Site was built below the outlet of the penstock. Daily water temperatures at these locations have been summarized in Sweeten (2005). Further, some temperature data associated with the operation of the upper spawning channel is available for the years following 1965. From 1977 until 1998, temperatures were recorded on a Taylor thermograph, while Onset temperature recorders were used afterwards (Sweeten 2005).

Map 5. **Study Area and location of temperature loggers in the Puntledge Hatchery Upper (box 1) and Lower sites (box 2) (Source: Sweeten 2005).**



Puntledge Hatchery Upper Site temperature trends (1965 to 2005)

The warmest time of year in Comox Lake is August when the water has been drawn down off the surface of the thermocline. Mean temperature is above 18ºC and maximum temperatures can exceed 22ºC (Figure 21). Over a span of 39 years the minimum August water temperature for the Upper Site has increased 1.50 ˚C (i.e., 16.30 ˚C in 1965 to 17.80 ˚C in 2004;), a relationship that is significant over time (Figure 22). Similarly, over the same period the maximum August water temperature for the Upper Site has increased 1.09 ˚C (i.e., 20.87 ˚C in 1965 to 21.96 ˚C in 2004).

Figure 21. The average daily minimum, maximum and mean temperatures for June to September from 1965 to 2004 at the Puntledge Hatchery Upper Site. Instantaneous minimum and maximum daily temperatures are also plotted (Source: Sweeten 2005).

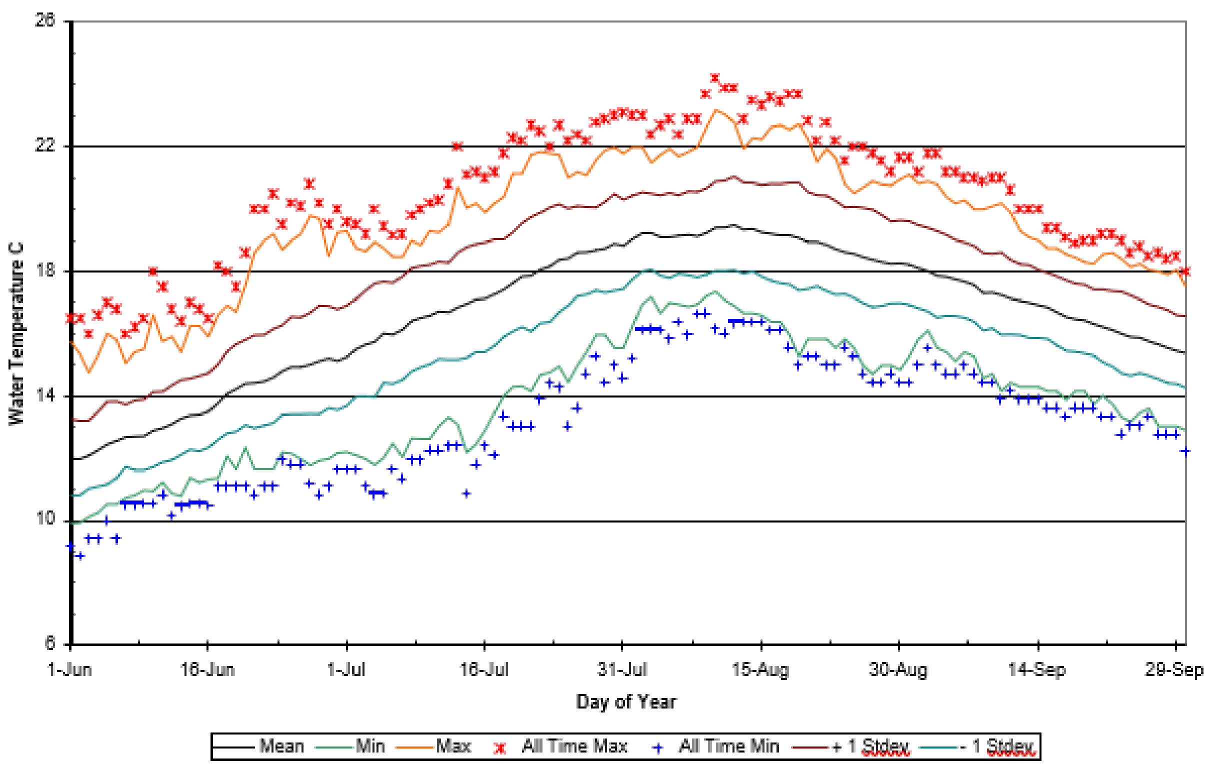
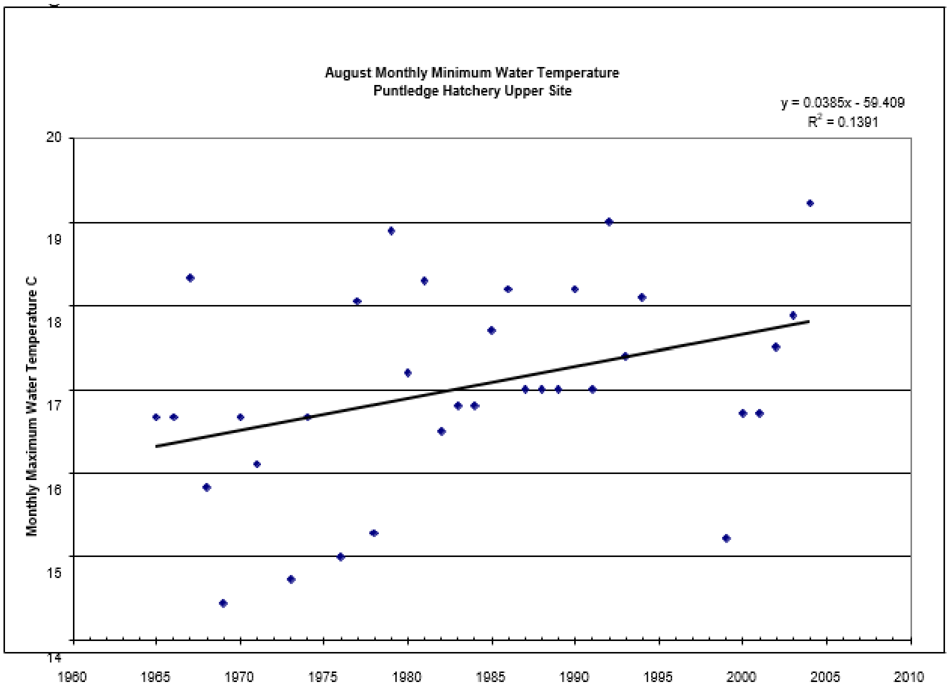
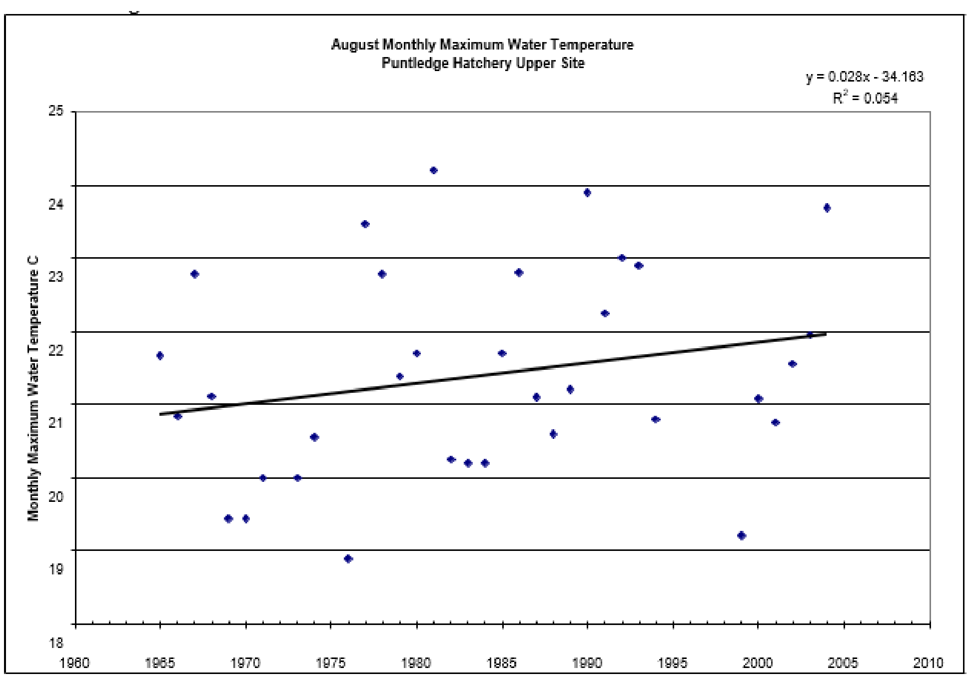


Figure 22. Average minimum (a) and maximum (b) August water temperature trends in the Puntledge Hatchery Upper Site from 1965-2004. (Source: Sweeten 2005). Of note, the y axis in figure a should state "Monthly Minimum Water Temperature °C".

a)

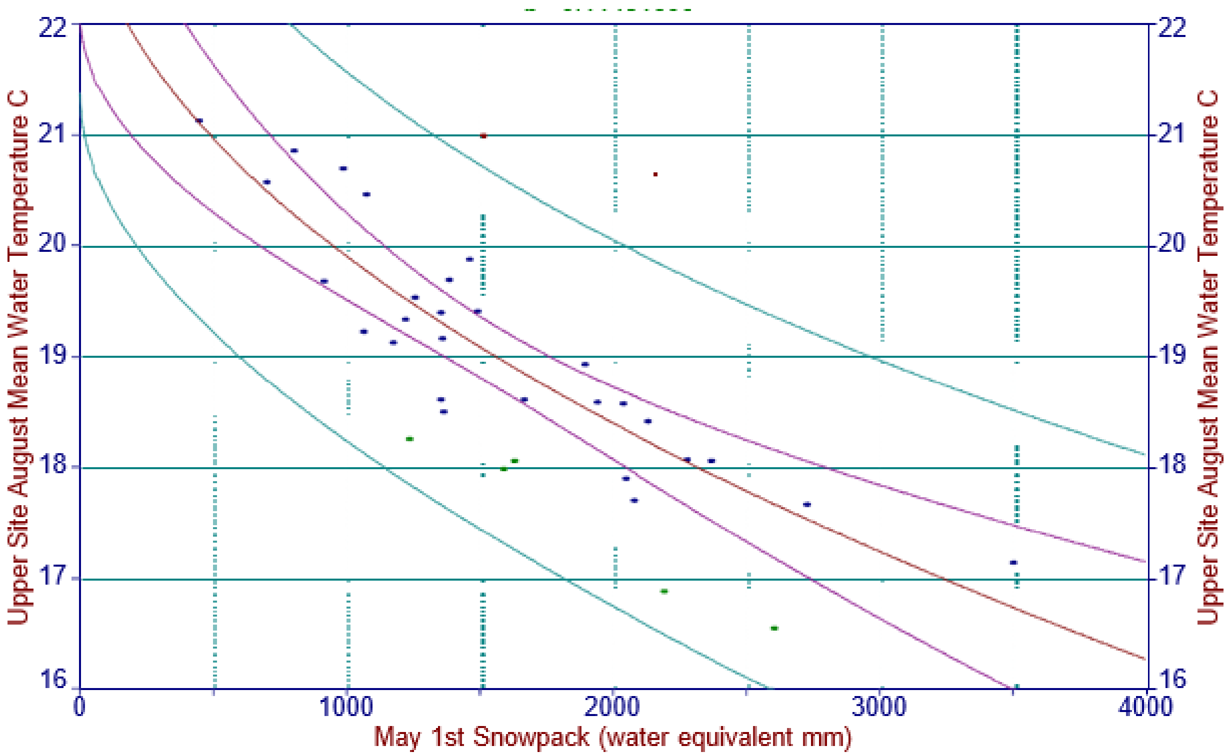


b)



Since 1957, snow-pack depth and water equivalent (mm) have been measured on or around May 1st at the Forbidden Plateau snow depth station near the Puntledge Hatchery Upper Site (located at N 49° 39' W 125° 13' at an elevation of 1130 m; Figure 23). Water equivalent is the amount of water present in the snow column should it be melted. Using this relationship and snow-pack data from May 1st, one can try predicting the mean August water temperature. This information may then shape planning for temperature effects by moving fish or changing fish culture practices. In 2005, the May 1st snowpack was 600 mm of water equivalent, which estimates an August mean water temperature of 20.72 ˚C. This corresponds to the fourth warmest August on record, with only 1980 1990, and 2004 being warmer. It should be noted that this trend line is significant (R^2=0.57, N=35, F=43.97; critical F = 4.14 at 0.05).

Figure 23. The effect of snow-pack level on August mean water temperature at the Puntledge Hatchery Upper Site.

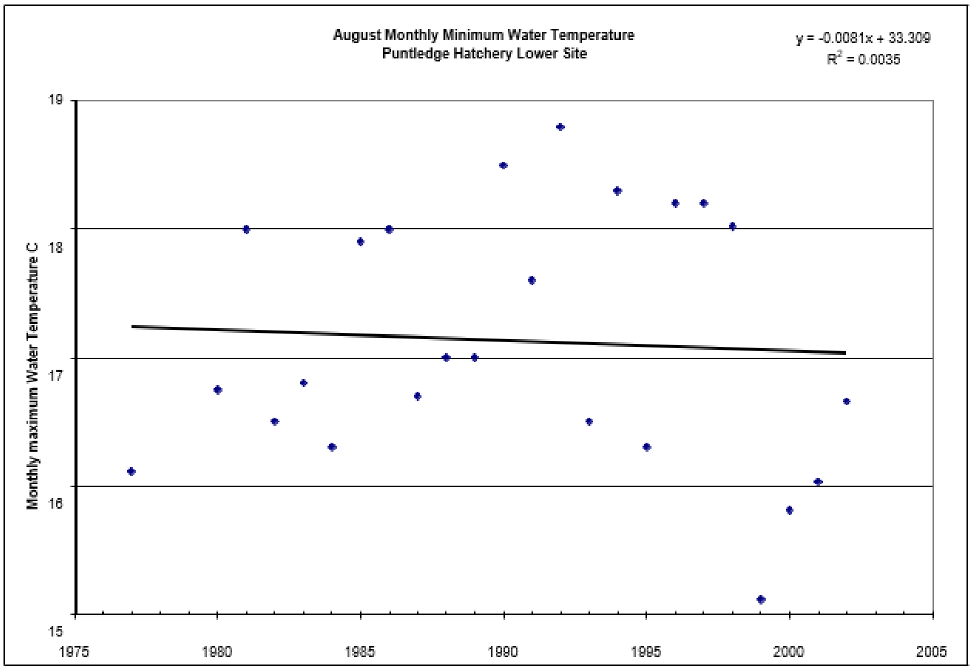


Lower Site Trends

Figure 24 and Figure 25 show the average daily minimum, maximum and mean summer water temperatures (June to September) between 1977 and 2004. A linear regression fit to the data shows a negative relationship (although not significant) between water temperature and time (R^2=0.0035, P>0.05; R^2=0.0051, P>0.05). Thus, based on the obtained trend lines, the minimum water temperature has generally dropped 0.20˚C (from 17.24˚C in 1977 to 7.04˚C in 2004) or 0.0074˚C per year in August while the mean water temperature has dropped 0.27 ˚C (19.46 ˚C in 1977 to 19.19 ˚C in 2004) or 0.01 ˚C per year.

Figure 24. (a) Monthly minimum and (b) maximum average daily summer water temperatures at the Puntledge lower hatchery between 1977 and 2004. (Source: Sweeten 2005).

a)



b)

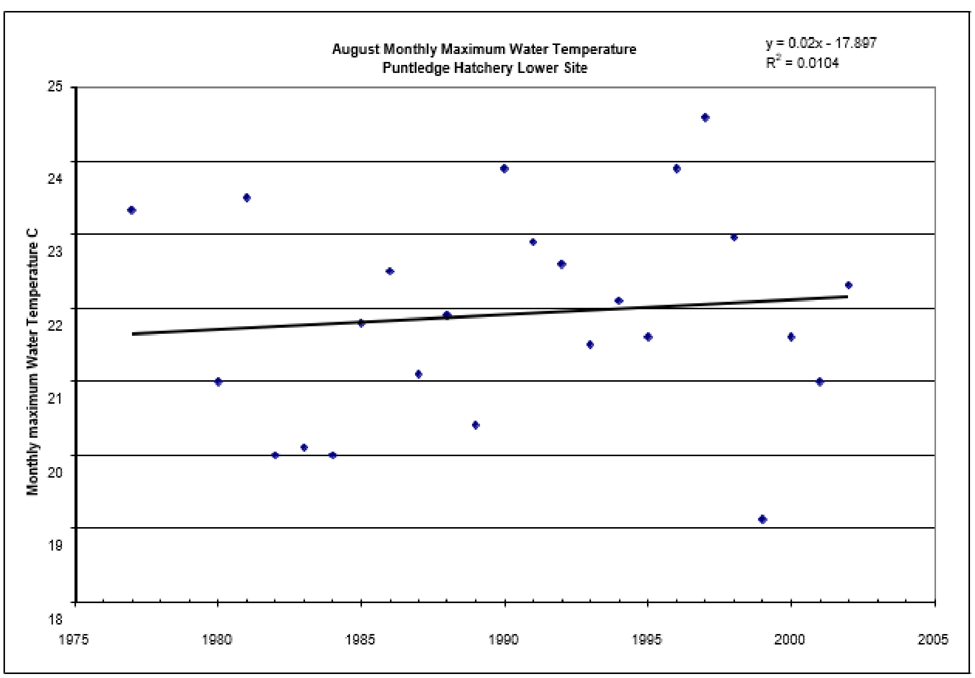
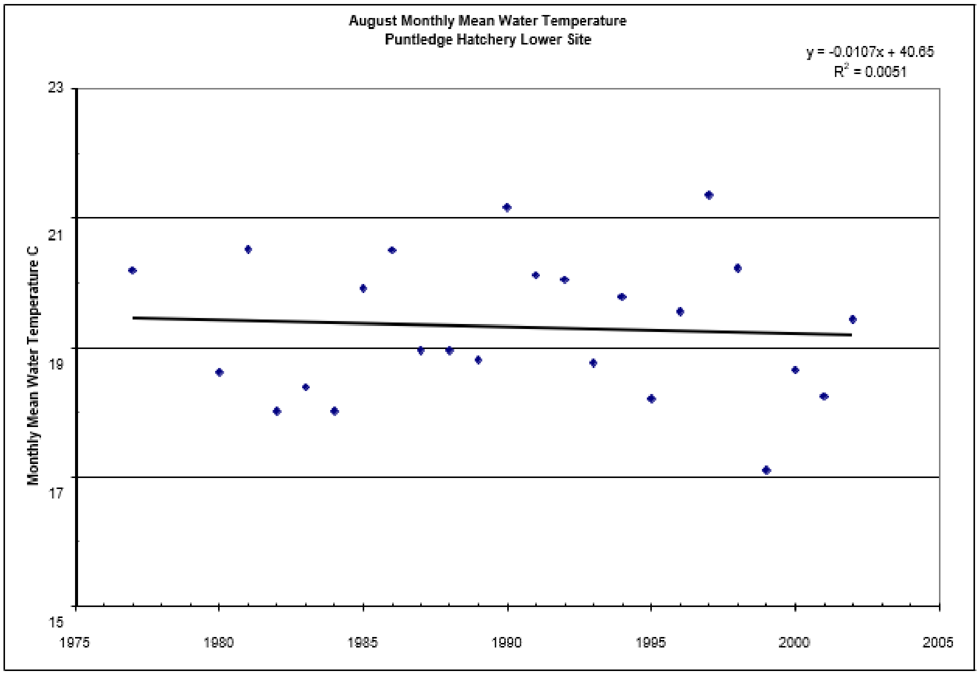


Figure 25. Monthly mean (average daily) summer water temperatures at the Puntledge lower hatchery between 1977 and 2004. (Source: Sweeten 2005).



Upper vs Lower Site Temperature Relationship

Daily temperature records from the upper and lower hatchery sites were compared for the period of July 1st to September 30th over five years (1999 to 2004). A statistically significant relationship was identified between the two sites. On average, the daily minimum water temperature of the lower hatchery site was cooler than the upper hatchery site. However, the lower site was warmer than the upper site when:

- The daily upper site maximum temperature was >15.68˚C, or
- The daily mean temperature was >18.16˚C.

Thus, when the upper site daily mean temperature was 23˚C, the lower site was on average 0.22ºC warmer.

Lower Hatchery Temperature Data (1977-2021)

Water temperature data in the Puntledge River collected at the lower hatchery site between 1977 and 2021 was provided by DFO (Sweeten pers. comm. 2022). Data were summarized by plotting the available mean and mean maximum daily temperatures over time, as well as the frequency of days

>20°C in the summer months (i.e., June, July and August[2]). Data trends were examined using a linear regression (Figure 26 to Figure 28). A small long-term increase in monthly mean stream temperature (for June to August) from 1977 to 2022 is evident in the scatterplots. Furthermore, year appears to be a predictor of warmer water temperatures and the number of days over the threshold of 20°C increases by 0.50°C every year (Figure 28). However, the linear models show very low R^2 values (0.057 to 0.127), likely due to the high variability in observed stream temperature and the monotonic temperature-year relationship assumed by the model. To better compare trends in mean monthly stream temperature, a more detailed analysis could be completed by standardizing the monthly mean stream temperature to the same scale and applying generalized additive models (GAMs) to quantify differences in predicted stream temperature over time. Alternatively, a similar approach could be applied to the mean weekly average temperature (MWAT) per month in each year, as MWAT would reduce the variability in observed temperatures. This dataset combined with water temperature from the upper hatchery and snowpack levels will be analyzed in depth by DFO this fall (Sweeten pers. comm. 2022).

2

Figure 26. Mean water temperature in June, July and August between 1977 and 2021 in the Puntledge River at the lower Puntledge Hatchery (Source: Sweeten pers. comm. 2022).

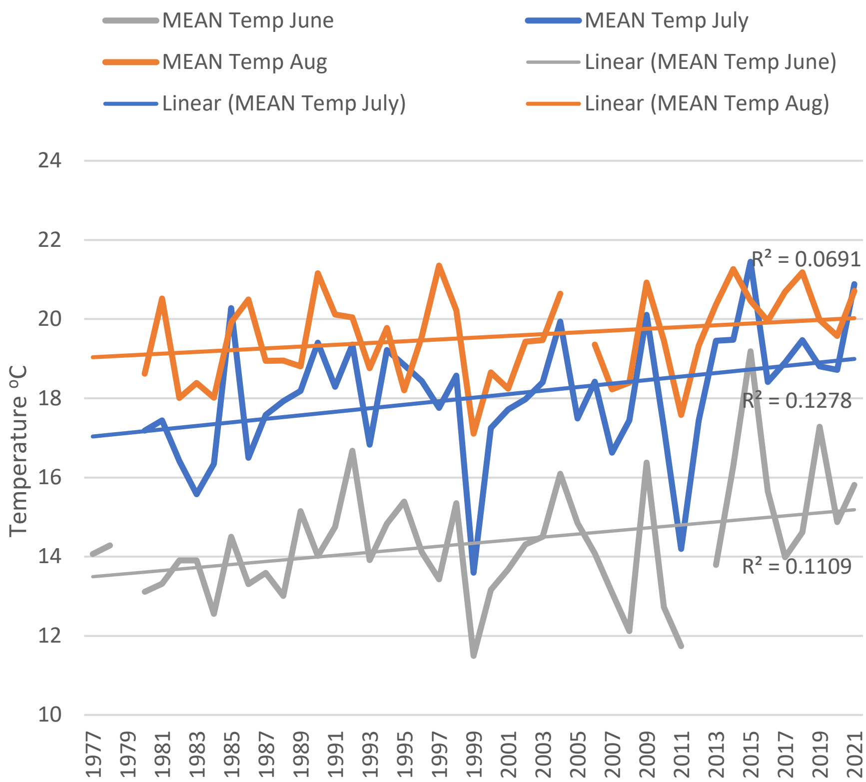


Figure 27. **Mean maximum water temperature in June, July and August between 1977 and 2021 in the Puntledge River at the lower Puntledge Hatchery (Source: Sweeten pers. comm. 2022).**

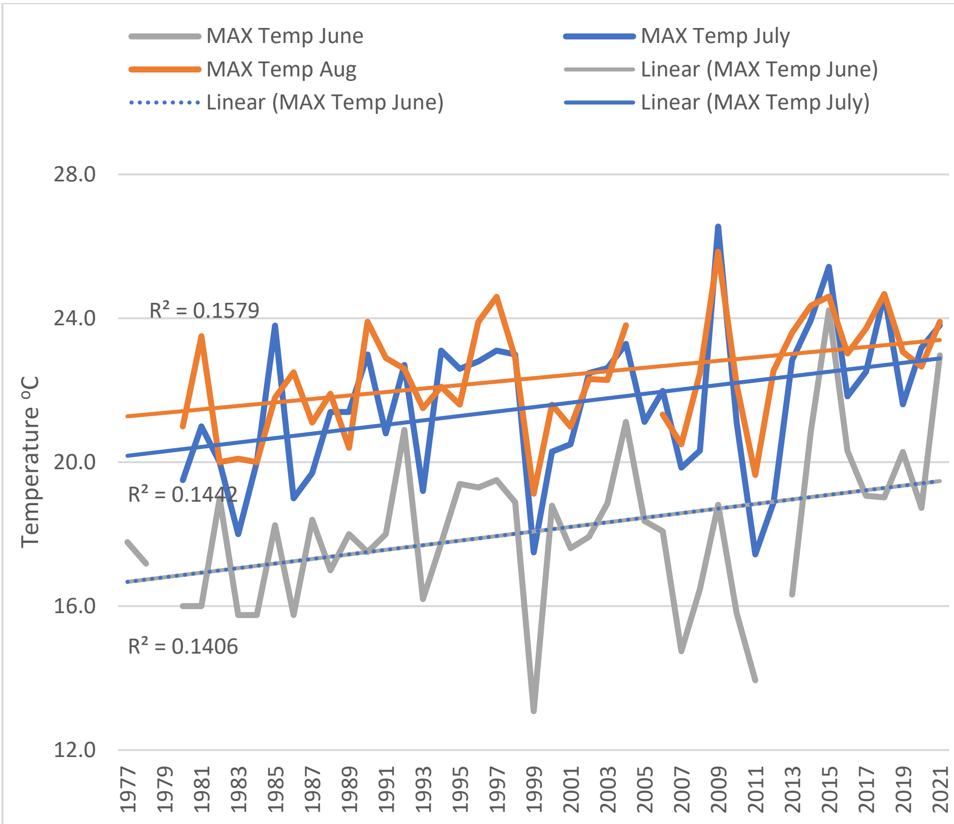
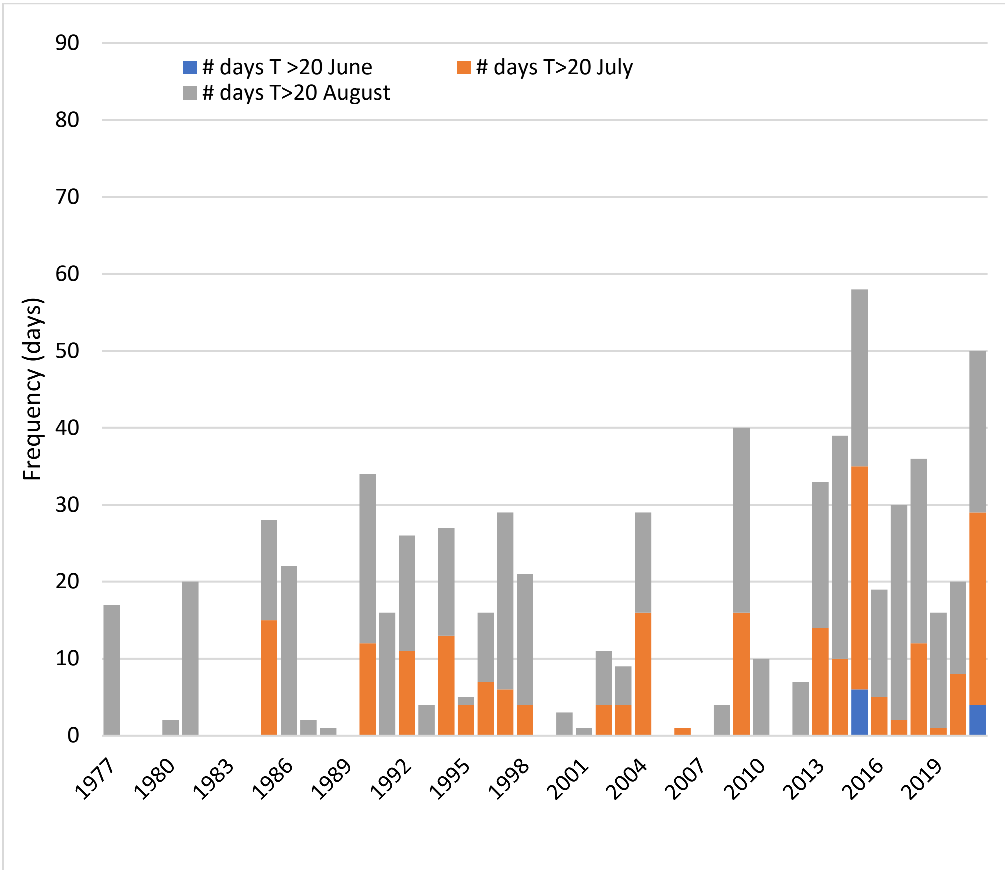


Figure 28. Frequency of # days with water temperature >20°C in June, July and August between 1977 and 2021 in the Puntledge River at the lower Puntledge Hatchery (Source: Sweeten pers. comm. 2022).



2.5.2. Future

The CVRD has forecasted demands on the water supply to double by the year 2020 and more than triple by 2058. A Regional Water Supply Strategy recently commissioned by the CVRD (Gower 2011) estimates that, with the implementation of water conservation measures and universal water metering, the annual average day demand will increase only two-fold by 2058. This translates into a 43% reduction in demand as compared to the current consumption rate.

The Comox Valley in one of the fastest growing regions in the province of BC The trend over the past five years indicates an increase of ~9% over this period (CVRD 2014). The present population of the Comox Valley is estimated at 64,642 (BC Stats 2010). According to the recent Comox Valley Regional Growth Strategy policy document adopted by the CVRD on March 29, 2011, this number is projected to increase by almost 50% over the next 20 years. This rapid development comes with associated threats, including runoff from sewage and suburban storm sewers, infilling of wetlands, urban sprawl and associated transportation networks, and disturbance from increased recreational activities. From 1991 to 2002, at least 42% (~2,700 ha) of the rare and threatened sensitive ecosystem lands in the Comox Valley and 97% of valuable human-modified ecosystems such as older second growth forests and seasonally flooded agricultural fields were either lost, fragmented, or reduced (Fyfe 2008).

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2.6. <u>Sustainable Watershed Use Planning</u>

2.6.1. Water Use Planning

Water Use Planning (WUP) was developed in 1998 by the BC government in response to increasing demands on the province's water resources, with a goal of finding a better balance between competing uses of water such as domestic water supply, fish and wildlife, recreation, heritage, and electrical power needs that are environmentally, socially and economically acceptable to British Columbians. Water use plans were completed for most of BC Hydro's hydroelectric facilities through a consultative planning process that involved government agencies, First Nations, local citizens, and other interest groups. Longer term climate change effects were not a focal point during the initial process but will be included in future scheduled review.

The Puntledge River Water Use Plan (PUN WUP) consultative process began in June 2001 and was completed in June 2003 with a consensus agreement on an operating alternative for the hydroelectric facilities on the Puntledge River (BC Hydro 2003). During that period, several studies were conducted to provide the necessary information for an informed decision-making process by the Consultative Committee (CC). In addition to recommending a preferred operating alternative for the facility, the CC recommended several monitoring studies and physical works that would address uncertainties and answer specific questions associated with the operating alternative that may change future decisions on operations. The PUN WUP was reviewed by the provincial Comptroller of Water Rights under the provisions of British Columbia's Water Act, which involved DFO, other provincial agencies, First Nations, and holders of water licenses who might be affected by the plan. BC Hydro was ordered to implement the conditions proposed in the Puntledge River WUP on January 19, 2005.

Five monitoring programs and one physical work project was recommended in the PUN WUP:

1. Assessment of Adult Fish Passage During Pulse Flow Releases;
2. Assessment of Egg Incubation Success in Puntledge River Reach C;
3. Steelhead Production;
4. Evaluating Effects of Ramping Rates on Fish Stranding in Puntledge River Reach C;
5. Kayak Pulse Flow Cost – Benefit Assessment; and
6. Gravel Placement in the Puntledge River.

The PUN WUP monitoring program commenced in 2006. The CC also recommended a review of the PUN WUP 10 years after implementation, unless results from the monitoring program indicate that an earlier review is necessary. Five years after the implementation of the WUP, BC Hydro, with input from appropriate agencies, will review results from the monitoring programs and assess the need to review the Puntledge River WUP. A review could be triggered sooner if significant risks are identified that could result in a recommendation to change operations.

During the PUN WUP consultative process, it was identified that higher minimum flows in Reach C of the Puntledge River, from the current 5.7 m^3/s to 8 m^3/s, would significantly increase Chinook and steelhead spawning habitat performance measures based on weighted usable width (WUA). However, increasing minimum flows would cost BC Hydro approximately $1.5 million annually and it was agreed that placing gravel in Reach C or B would be a less expensive means of increasing spawning habitat, even when factoring in the cost of future gravel replacement to maintain the habitat. The PUN WUP CC concluded that 2,000 m^2 of new spawning habitat would be created, in lieu of increased flows (BC Hydro 2003).

2.6.2. Land Use Planning

The CVRD has set various sustainability strategies for land use in the Puntledge River watershed, which designates high-level targets for the year 2050 in the CVRD area (Source: CVRD Community Climate Action Plan 2023a, Table 22, Table 23):

- Energy:
    - 50% use per capita.
    - 50% of energy supplied by clean, renewable energy for new building energy demand.
- Water use (non-agricultural):
    - 50% per capita.
- Wastewater:
    - 100% treated to tertiary or reuse standards.
- Ecosystems:
    - 100% sensitive ecosystems and riparian areas conserved.
    - 70% degraded ecosystems are restored.
- Waste Sustainability Strategy targets for the year 2050:
    - 90% diversion rate of compostables and recyclables.
    - All new landfills are designed to maximize methane capture and reuse.
    - All existing landfills are reviewed for viability of landfill gas capture and reuse by 2012.
    - 100% wastewater treated to tertiary or reuse standards.
- Regional Growth Strategy targets for the year 2030:
    - 75% solid waste diversion rate.
    - No net loss of zoned farmland in the ALR, equal to or greater than 23,059 hectares.
    - No net loss of aquaculture farm tenure, 470 hectares.
    - Improve farm access to irrigation water by 25%.
    - Increase farming activity to $55,000,000 in farm receipts and to 9,071,847 kg shellfish production.
    - Raise awareness of the regional importance of the local food system.
- Agriculture Sustainability Strategy targets for the year 2050:
    - 60% of fruits and vegetables consumed are produced on Vancouver Island.
    - 100% of dairy consumed is produced locally.

- 45% of protein products consumed are produced locally.

A simple energy and emissions projection was also performed for rural areas. The open-source land-use energy and emissions model, GHG Proof, was used to estimate a business as usual (BAU) Scenario and a Scenario in which energy saving and emissions reduction actions were taken (S2). Consistent with the Sustainability Strategy, a target year of 2050 was set for Scenario 2. The targets in the Sustainability Strategy (SS) and Regional Growth Strategy (RGS) were used to guide the assumptions for both scenarios.

Table 22. Targets for water consumption, solid waste diversion rate and GHG emissions for 2015, 2020 and 2050 in the CVRD area (Source: CVRD Community Climate Action Plan 2023a).

Water Conservation

MEASURES	Baseline (2008)	TARGETS			Data sources
		Short-term (2015)	Medium- term (2020)	Long-term (2030)	
Reduce daily total water consumption per capita	500-600 litres	20% reduction	30% reduction	40% reduction	CVRD water services

Waste

MEASURES	Baseline (2010)	TARGETS			Data sources
		Short-term (2015)	Medium- term (2020)	Long-term (2030)	
Increase solid waste diversion rate	48%	55%	65%	75%	CVRD
Reduce solid waste GHG emissions	61,605 CO2e(t)	20% Reduction	33% Reduction	50% Reduction	CEEI

Table 23. Rural areas simple energy and emissions projection targets under the business as usual (BAU) and energy savings and emissions reductions (S2) scenarios (Source: CVRD 2023).

Agriculture and forest	BAU	S2	In S2, by 2050...
Area of local farms	23,342	23,342	Farm area (ALR) stays the same.
Intensity of production (hectares/capita)	0.58	0.80	Production intensity increases by 0.22ha/capita. (SS and RGS local food targets)
Percent of production locally consumed	5%	60%	Locally produced goods that are locally consumed increases 55%. (SS and RGS local food targets)
Area of forest	15,015	15,015	Forest area remains unchanged.

2.6.3. Climate Change Planning

Climate change (as described in Section2.5) is a key issue for governments around the world, including in the Comox Valley area. Climate change requires both strategies and mitigation to reduce greenhouse gas (GHG) emissions and adapt to changes that are already taking place (CVRD 2023b). The CVRD has set goals within the RGS, which include minimizing regional GHG emissions, which is being implemented within the following climate action initiatives (CVRD 2023b):

- Coastal Flood Mapping;
- Airshed Roundtable Project;
- Comox Valley Sustainability Strategy;
- Comox Valley Rural Areas Community Climate Action Plan;
- Corporate Energy Plan;
- Transition 2050 Residential Retrofit Acceleration Strategy; and
- Green Shores Local Government Working Group.

As well, the CVRD has joined the Partners for Climate Protection Program, signed the BC Climate Action Charter in 2007, adopted a Climate Change Toolkit in 2008 and installed several solar energy panels in the area to reduce its environmental footprint (CVRD 2023b).

2.7. Habitat Restoration Efforts in the Watershed

2.7.1. Overview

In response to growing concerns over the health of the Courtenay River watershed (Puntledge and Tsolum Rivers) and its estuary, DFO in partnership with local government and first Nations, initiated an estuary management planning process in 1997. The goal of the Courtenay River Estuary Management Plan (CREMP) was to set a direction for the sustainable management of the estuary's resources (Envirowest 2000). The CREMP was completed in 1999 as a "Working Draft" with the understanding that this "living" document would require additional effort on the part of agencies and stakeholders to update the plan to meet future needs and address changing social, environmental, and economic conditions (Adams and Asp 2000). It was designed as a policy-based document and does not have regulatory force at any level of government. In addition to the integrated management plan, fourteen Habitat Classification and Development maps were created that cover the entire estuary planning area and classify habitat and development into three categories based on habitat sensitivity: highly sensitive (red lines), moderately sensitive (yellow lines) and limited habitat values (green lines). The classification includes riparian areas, intertidal zone, and below sea level to 10 m.

Unfortunately, the CREMP was never adopted by any of the three local governments, and the document was not implemented following its completion. However, a recent surge in interest for the protection and rehabilitation of the Courtenay River estuary has renewed collaboration among local municipal governments, and other stakeholders, to review and update the CREMP to reflect current conditions.

In 1992, BC Hydro commissioned a study to document the locations of historic spawning habitat for salmonids in the lower Puntledge River and identify potential sites where spawning habitat could be enhanced or created (Bengeyfield and McLaren 1994). One of the recommendations in the report was to defer spawning gravel placement projects because of a lack of surplus spawners. As summer Chinook returns began to increase in 2001, and the issue of turbine mortality was finally believed to be resolved, the feasibility of spawning habitat restoration became more achievable.

2.7.2. Puntledge River Spawning Channel (Upper Puntledge Hatchery)

Following the expansion of the Hydroelectric facilities in the 1950s, a rapid decline in summer-run Chinook and other salmon populations prompted the Province of British Columbia to order a public inquiry into matters relating to fish conservation and the operation of the Power Plant on the Puntledge River. The inquiry was led by Commissioner Henry F. Angus and was completed in 1962. One of the recommendations was for BC Hydro to install and operate a barrier dam and fish collection facilities downstream of the powerhouse, so that adult salmon and steelhead could be transported to the upper river, as well as a louvre-type screening mechanism at the penstock intake to reduce juvenile mortality (Angus 1962). A final settlement between DFO and BC Hydro was reached in April 1965 with BC Hydro agreeing to construct a spawning channel adjacent to the diversion dam to compensate for the loss of spawning habitat upstream of the diversion dam and mortality from entrainment.

The spawning channel was designed to provide 1,905 m^2 of spawning habitat and was considered to be capable of supporting a population of 1,000 (400 female) summer-run Chinook (Lister 1968). Once operations at the spawning channel commenced in 1965, adult summer Chinook and summer steelhead were no longer allowed access to their native habitat in the headpond, and instead were directed into the spawning channel via a 23.2 m long fishway, the entrance of which was located at the base of the diversion dam. Fish barrier racks were installed along the diversion dam to prevent adults from jumping over the dam, and to redirect them into the fishway. A "steelhead bypass" structure also provided access to the river above the dam for steelhead that were not required for hatchery production. Emerging fry from the channel were released into the river below the dam through the fishway and, therefore, could avoid the problems associated with entrainment during downstream migration. Adults that did not enter the spawning channel were allowed to spawn naturally below the diversion dam. This number ranged from 7 to 90 during the period 1965-1976. The spawning channel was intended to be operated until the channel was no longer effective for its intended purpose (i.e., the summer Chinook return was successfully restored to pre-expansion levels) or the channel was approaching its spawning capacity. However, production from the spawning channel between 1965 and 1971 was very poor with average egg-to-fry survival during this period estimated at 29.3%. Pre-spawning mortality from stress, fatigue, and head and body injury sustained during migration also accounted for a significant loss in production (MacKinnon *et al.* 1979).

While efforts to alleviate the problems with pre-spawn mortality were somewhat successful, sedimentation issues and low egg-to-fry survival continued to be problematic. This prompted supplementary hatchery production of summer Chinook in 1972. Eventually the spawning channel was converted into an adult holding and juvenile rearing channel and a full-scale hatchery was constructed downstream of the BC Hydro Powerhouse in 1979. Once full-scale hatchery operations commenced, salmon enhancement practices involved taking 90-95% of the returning summer Chinook adults for hatchery broodstock (Munro pers. comm. 2023). Although the spawning channel has been credited with maintaining the summer Chinook population during its years of operation, it was not successful in rebuilding the species to its former population level (MacKinnon *et al.* 1979).

2.7.2.1. Reach B - Headpond

In the fall of 2001, surplus summer-run Chinook broodstock from the Puntledge River Hatchery were allowed above the diversion dam for the first time since the diversion dam fishway was closed to adult fish passage in 1965. These fish were closely monitored to determine their spawning behaviour in the headpond and assess egg incubation success is the areas that were selected for spawning (Lightly and Guimond 2001, DFO Unpublished data). The observations and results from this cursory assessment led to further evaluations to determine the feasibility of restoring spawning habitat in this once significant spawning reach (Wright and Guimond 2003). Development of a spawning habitat restoration plan for Reach B involved River2D modelling to calculate the most suitable location, dimensions and elevation for a spawning gravel platform that would be stable under the maximum range of flood flows (1 in 25-year flood event; Chilibeck 2003).

Construction of the spawning platform spanned two years and in-stream activities were carefully monitored to ensure turbidity levels were maintained within manageable levels because of the domestic water supply withdrawal 1 km downstream of the work site. The completed project resulted in a spawning platform approximately 4,756 m^2 in area, sufficient for over 400 Chinook pairs. Since October 2005, summer Chinook adults have been observed spawning at this site every year. Incubation survival was assessed in 2005/2006 and 2006/2007 (using eyed Chinook eggs buried in Jordan-Scotty incubation cassettes) and was excellent (>95% survival) (Guimond 2006c).

In 2021, Northwest Hydraulics Ltd. (NHC) constructed a spawning gravel pad providing 1,873 m^2 of highly suitable spawning habitat for Chinook salmon over the range of flows expected during their spawning period. The site is located immediately downstream of the pool tailout below Comox Impoundment dam.

2.7.2.2. Reach C - Bull Island Side-channel

The Bull Island side-channel is a natural secondary channel located upstream of Stotan Falls between the Island Highway (HWY 4) and the Comox Logging Road. This 600 m long channel was identified in a 1993 gravel placement feasibility study (Bengeyfield and McLaren 1994) and became the focus of a two-year habitat restoration project that saw the addition of 2,165 m^2 of spawning habitat to this side-channel in 2002 and 2003. An accumulation of logs and wood debris in a narrow bend in the channel, approximately 300 m downstream from the inlet, provides deep pool and overhead cover for adult holding. The major components of the project include three Newbury-style rock weirs to ensure optimum hydraulic conditions for spawning and incubation in the side channel, washed spawning gravel additions upstream of the rock weirs, and two rock deflectors at the upstream entrance of the side-channel to reduce high flows into the channel during floods and increased flow diversion into the channel during mainstem base flow conditions (Guimond and Norgan 2003).

One of the unique qualities of this restoration project was the creation of an abundance of high-quality rearing habitat in close proximity to the spawning area in the lower 200 m of channel. In this area, the channel width was reduced from 17 m to 13 m by creating 'false' channel banks within the existing channel using oversized rocks, boulders and LWD. This concentrated the majority of the discharge between the false channel banks and over the spawning gravel platform in order to maintain water depths and velocities suitable for Chinook spawning and incubation. A small percentage of the flow still passes through the rock/LWD banks to wet the area between the spawning channel and original channel banks, creating good access and prime rearing habitat for juveniles.

As many as 2,000 adults were estimated using this habitat following completion. The habitat is used primarily by fall Chinook and Coho; however, summer Chinook remaining in the lower river (below Nib Falls) at the onset of the spawning period likely use the habitat as well. Incubation survival was assessed in 2002/2003 and 2005/2006 and again in 2007/2008 using eyed Chinook eggs buried in Jordan-Scotty incubation cassettes, and results were >90% overall (Guimond 2006c, 2008). The high incubation success observed at the Bull Island spawning platforms demonstrates the value of introducing high quality spawning substrate as a means of enhancing spawning populations of summer run Chinook.

2.7.2.3. Reach C Mainstem

In 2005 and 2006, approximately 1,807 m^2 of spawning habitat for Chinook and steelhead was restored in the Puntledge River mainstem below the diversion dam (Reach C) through the addition of washed spawning gravel (Silvestri 2007). The work was completed by the BC Conservation Foundation (BCCF) with FWCP funding. The gravel additions were completed specifically to target threatened summer steelhead and summer Chinook stocks but would also be utilized by non-target species present including Coho and fall Chinook. Location (river km) of the gravel additions is summarized in Table 24. Using a spawning habitat biostandard of 7.6 m^2 per pair of steelhead trout and 10 m^2 per pair of Chinook salmon (Burt 2004). The total number of pairs of steelhead and Chinook that may be accommodated by the new spawning platforms is 237 and 180, respectively.

Table 24. Spawning gravel placement sites in Reach C of the Puntledge River in 2005 and 2006.

Site Number	Year Completed	Location	Km downstream of diversion dam	Area (m^2)
1	2005	Adjacent to the BCH Diversion Dam	13.2	454
2	2005	Mainstem Bull Island side-channel outlet	9.8	538
3	2006	Right bank of Barber's Pool	12.9	303
4	2006	Left bank immediately upstream of Bull Island Side-channel intake	3.2	170
5	2006	Right bank, opposite Bull Island Side-channel outlet	3.5	342
			Total Area	**1,807**

3. PUNTLEDGE SUMMER CHINOOK

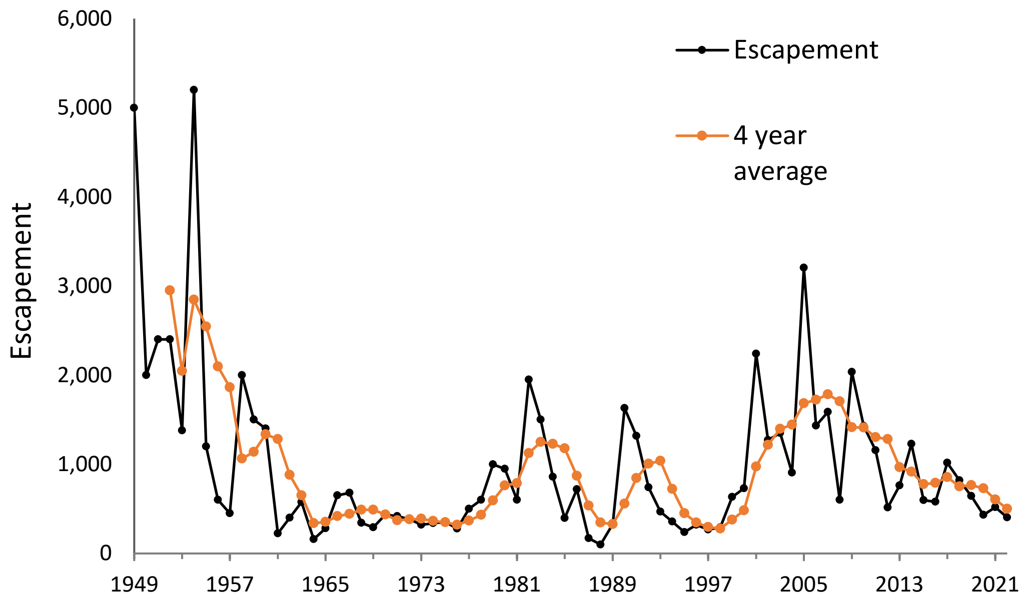
3.1. Introduction

The Puntledge River supports both an early (summer) run and a late (fall) run of Chinook salmon. The two runs have discrete migration timings and spawning distribution in the river. However, both stocks spawn at the same time (October to early November). The Puntledge River summer and fall-run Chinook populations are genetically distinct from each other. In comparison, the Puntledge fall Chinook are more closely related to other southeastern Vancouver Island fall-run populations such as Big Qualicum while the Puntledge summer-run fish are most closely related to the Nanaimo summer-run population. For further details please see Section 1.1.

3.2. Abundance and Stock Status

Annual escapement and four-year moving average for summer-run Chinook are illustrated in Figure 29. For the six-year period between 1949 and 1954, prior to expansion of the hydroelectric facilities on the Puntledge River, the summer-run population averaged about 3,000. Returns declined sharply to an average of 400 through the 1960s and early 1970s. Several management efforts were undertaken at this time such as the construction of a spawning channel, the installation of fishways, and fishing closures and restrictions, which allowed the population to recover to ~1,200 by the mid-1980s but numbers had declined again by the mid 90s (BC Hydro 2011b). A captive breeding program was subsequently initiated in 1997 and the population showed signs of recovery through the early 2000s, with a peak return of 3,200 in 2005. Unfortunately, since then, escapement has been declining steadily, to a four-year average of 500. Currently there is an estimated 0.33% smolt-to-adult survival, which is below that necessary for replacement.

Figure 29. Puntledge River summer Chinook escapement. Records from 1949-1971 are from MacKinnon *et al.* 1979, and 1972-2021 are from NuSEDs. Returns for year 2022 are from Puntledge Hatchery observations.



3.3. Genetic Composition

It is surmised that Puntledge River summer- and fall-run Chinook likely originated from the same population, but the summer-run are now genetically distinct from the fall-run population and from other Chinook populations in the Georgia Basin. Summer Chinook likely evolved from early migrants of the fall-run population that were able to ascend Stotan and Nib Falls as flows increased or decreased before and after peak spring freshet between April and July. These waterfalls physically segregated the summer and fall Chinook populations and were a barrier to other salmon species in the watershed except steelhead and possibly coho.

The two runs have discrete migration timings and spawning distribution in the river, but both stocks spawn at the same time, from early October to early November. Summer-run Chinook enter the river from May to August, peaking in mid-July. Historically, these adults would occupy large deep pools between Stotan Falls and the Comox Lake impoundment dam, and also enter Comox Lake to hold for two to three months before moving onto the spawning grounds (Anon.1958). The main spawning ground of the summer-run was a 4 km section of river immediately below the outlet of Comox Lake, now delimited by BC Hydro's diversion dam and the impoundment dam. They also spawned to a lesser extent in the lower mainstem reaches of the Cruickshank River, a tributary to Comox Lake, and below the diversion dam to Stotan Falls (Anon. 1958).

Fall-run Chinook enter the river from September to October. Historically, their main distribution was in the lower river between the confluence of the Tsolum River and Stotan Falls, but spawning was mainly below the powerhouse and between Morrison Creek and Condensory Bridge. Historical spawning records note that there was minimal temporal separation between the early fish of the fall-run and the late fish of the summer-run, and the fall-run adults were considerably larger in size than the summer-run population (Anon. 1958).

Prior to 2010, summer Chinook broodstock collection practices at Puntledge River Hatchery generally used an August 1st cut-off date (i.e. fish arriving prior to August 1 are called "True" summers and used as broodstock), while fall Chinook brood collection began September 1st. Chinook arriving through the month of August were usually not spawned with earlier migrants and were not permitted upstream of the diversion dam.

The genetic composition of Chinook salmon arriving at the lower Puntledge Hatchery between June and October was analyzed from 2006-2009 to verify the migration timing of summer Chinook (Figure 30). The goal of the study was to determine the extent that Chinook arriving throughout the month of August could be used to increase the effective spawning population of the summer-run, both at the hatchery and in the river (through release above the diversion dam) and accelerate the rebuilding of this stock to historical production levels. Chinook samples were examined from 5 discrete groups based on their time of return, or arrival at the hatchery: before August 1st, August 1–15, August 16–23, August 24–31, and September 1–30. As expected, the results indicated that Chinook returning before August 1st are predominantly summer origin. The tissue samples were screened at 12 microsatellite loci that were surveyed in baseline samples of summer and fall run Puntledge Chinook sampled in earlier years (Table 25; Guimond and Withler 2007).

Figure 30. Proportion of Summer Chinook (SCN), fall Chinook (FCN) and Chinook with a <0.85 probability of assigning to either SCN or FCN (Mixed), determined by microsatellite analysis, for the different return timing groups in years 2006-2009.

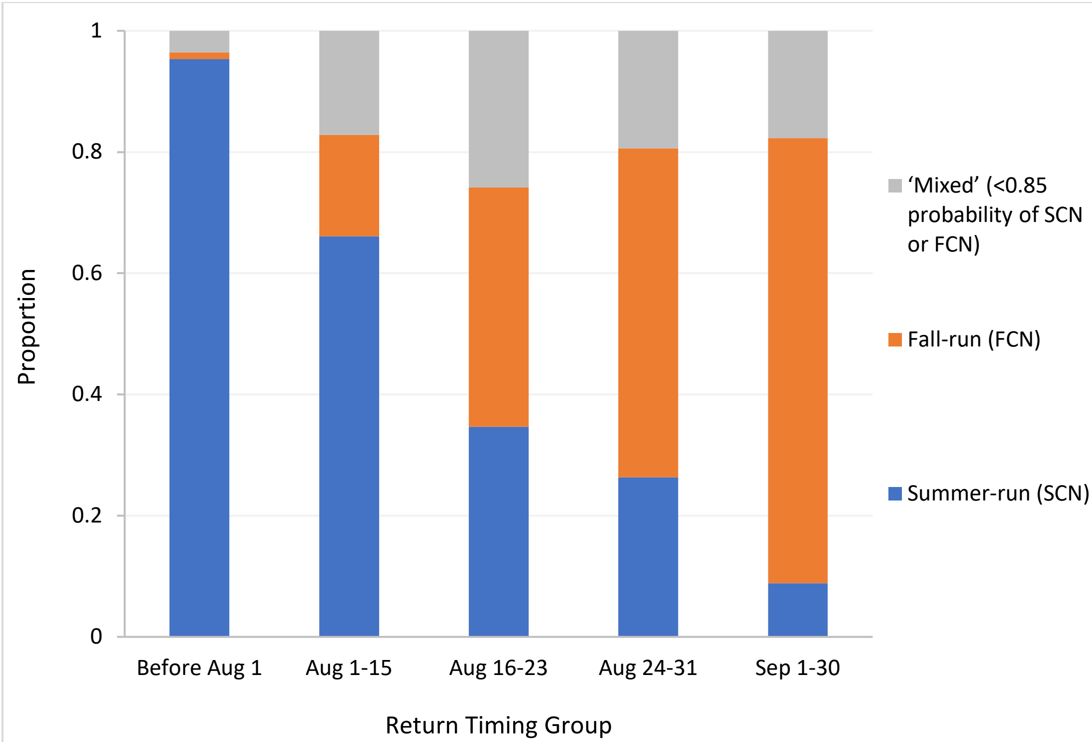
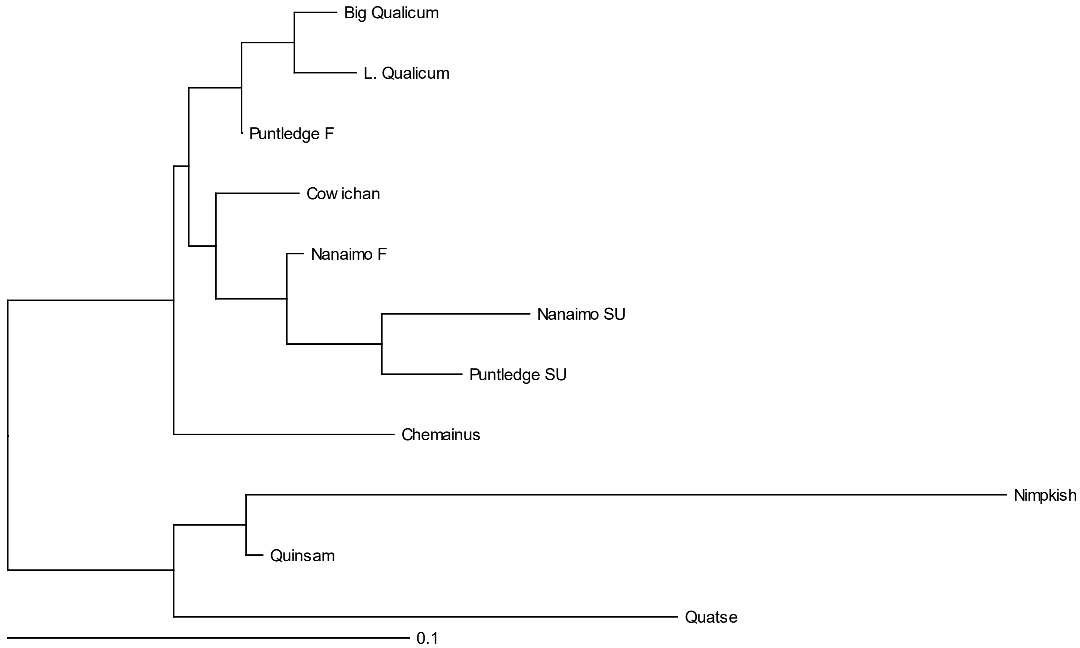


Table 25. Chinook salmon baseline samples for Puntledge River Summer and Fall run Chinook salmon.

Population	Years	Sample sizes	Total sample
Fall	1996 1997 2000 2001	60, 127, 194, 195	576
Summer	1988 1996 1997 1998 2000	131, 196, 209, 164, 201	901

The Puntledge River summer and fall run Chinook populations are genetically distinct from each other at the twelve microsatellite loci with an F_{ST} value of 0.0170 (Figure 31). In comparison, the Puntledge Fall Chinook are more closely related to other southeastern Vancouver Island Fall run populations such as Big Qualicum (F_{ST} = 0.002), while the Puntledge Summer run fish are most closely related to the Nanaimo Summer run population (FST = 0.0136). It should be noted that the fall Chinook samples from 1996-2001 used to develop the dendrogram would have included genetics from Big Qualicum, Little Qualicum, and Quinsam River hatcheries, which were used to rebuild the fall Chinook population following three years of low (<100) returns in 1985-1987.

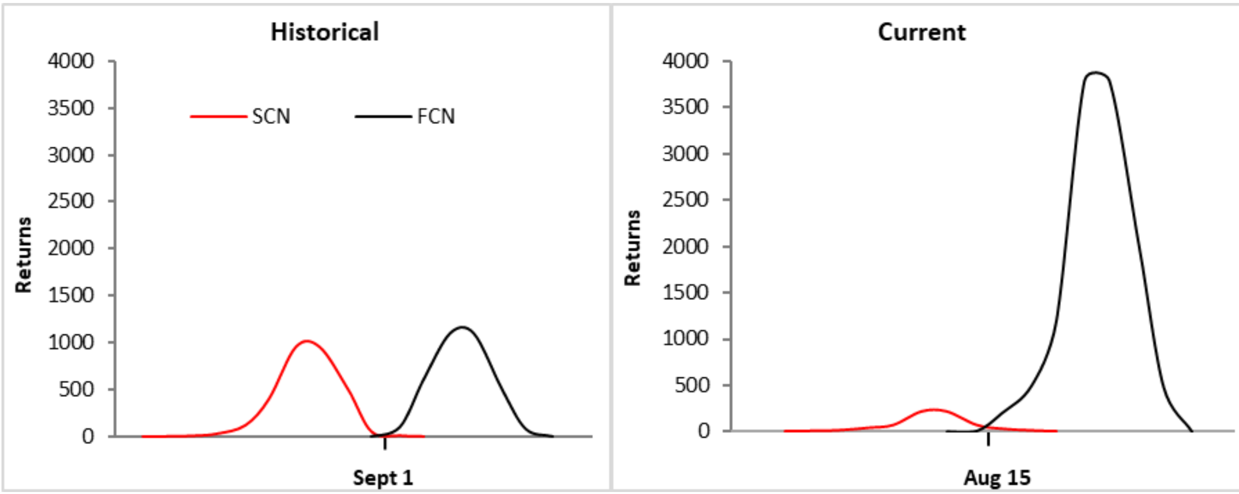
Figure 31. Neighbour-joining dendrogram showing genetic relationships among Chinook salmon populations on the east coast of Vancouver Island based on pairwise measurement of Nei's 1972 genetic distance.



A simplified depiction of the historic and current summer and fall Chinook run timing and escapement is illustrated in Figure 32. Historic information is based on run-time observations and escapements prior to 1955 when the hydroelectric infrastructure was upgraded (Anon.. 1958). Current information is supported by results from the 2006-2009 genetic study, and the current five-year escapement average. The current abundance of summer Chinook is about 6% of the fall Chinook population, while the historic (pre-hydro expansion) escapement average of the two runs were similar.

In addition to confirming that summer Chinook are genetically distinct from fall Chinook, the 2006-2009 analysis highlighted the success of hatchery protocols in maintaining the genetic integrity of the summer-run population using an August 1st cut-off for broodstock collection. Since that study, additional information on the migration success of natural spawners, decommissioning of the Upper Hatchery facility, and advancement in genetic analysis techniques at the Molecular Genetics lab (MGL) at the Pacific Biological Station, resulted in a re-evaluation of using later summer Chinook migrants for rebuilding the summer Chinook population (see Section 4.1.8).

Figure 32. Schematic depiction of the a) historic, and b) current Puntledge River summer and fall Chinook migration run-timing and estimated abundances. Historic abundance from pre-1955 escapement estimates; current 5-year average escapement (2017-2021).



Another observation from the study was an increasing trend in the proportion of fall-run Chinook in later timing groups, in addition to Chinook with lower probabilities of classifying as either Summer-run or Fall-run Chinook. These lower scoring 'Mixed' fish could indicate summer-fall hybrids or could also indicate strays or fish carrying rare alleles. It is reasonable to suspect that a portion of the 'Mixed' fish are the result of historical hybridization between the summer and fall populations, either through natural pairing on the spawning grounds or unintentional pairing at the hatchery. Since fall Chinook are not pre-screened prior to spawning, collection of fall Chinook for broodstock should continue to be deferred to after September 5th to reduce artificial introgression of summer Chinook genes in the fall Chinook population at the hatchery.

3.4. <u>Distribution</u>

Summer-run Chinook salmon enter the Puntledge River from late April to August while fall Chinook enter the river from September to October. Although the two populations are genetically distinct, it is suspected that summer-run Chinook evolved from early migrants of the fall-run population that were able to negotiate Stotan and Nib Falls as flows decreased after peak spring freshet between May and August (Marshall 1972). Fish that entered later in the summer and during lower flows would not have been able to ascend the falls and likely would have perished in the lower river from injury, exhaustion, increasing water temperatures and predation. These waterfalls have therefore been important in maintaining the spatial segregation of the two stocks.

Summer-run adults originally utilized spawning habitat above Stotan Falls and more predominantly in the section of river immediately below the outlet of Comox Lake, which is now bounded by the

Comox Lake impoundment dam and the Puntledge diversion dam. Historically, summer Chinook adults entering the system would have migrated upstream quickly, and likely held in the cooler depths of Comox Lake during the summer to escape elevated river temperatures until they were ready to spawn in the fall. Puntledge Chinook would drop back downstream and spawn below the lake outlet while some would have spawned in the main lake tributaries, notably the Cruickshank and Upper Puntledge Rivers. Spawning downstream of large lake outlets provides some advantages: the lake can buffer large storm events and can serve as a sediment trap, thus preserving gravel quantity and quality in this reach. It has also been speculated that those fish that spawned in the Cruickshank River may have in fact been a unique race of summer Chinook, adapted to the cooler temperatures of a snow-fed inlet stream as opposed to those fish that spawned in the headpond reach (Lister pers. comm. 2023). This life history behaviour is seen in several BC interior rivers that have both an early spring-run Chinook salmon stock and a later summer-run stock. The spring-run Chinook spawn in the cooler inlet streams while the later summer-run stock spawn in the warmer outlet stream (Bailey pers. comm. 2023). If this hypothesis is correct, genetic mixing of the two Puntledge River summer Chinook stocks would have likely occurred, particularly between 1912 and 1922 when passage into the lake was obstructed by the dam. Summer Chinook were occasionally observed spawning in the Cruickshank River following the fishway installation at the impoundment dam (Bengeyfield and McLaren 1994).

Fall-run Chinook salmon are larger than the summer-run Chinook and historically spawned downstream of the Browns River confluence (Trites *et al.* 1996). However, after hydro expansion altered the hydrology through Reach C, modifications to Stotan and Nib Falls in the 1960s and 1970s to improve fish passage for summer Chinook have now facilitated the access of fall Chinook and other salmon species into the upper reaches of the river (Bengeyfield and McLaren 1994)[3]. Presently, the headpond reach is maintained primarily for summer-run Chinook salmon through regulation of a fishway around the Puntledge diversion dam. Fall Chinook are not permitted into the headpond.

3.5. Life History Characteristics

There is very limited pre-hydro development life history information available on Puntledge summer Chinook. The majority of information has been obtained following the period hydro expansion on the river and through subsequent efforts made between the 1960s and 1980s to reverse the decline in returns of both the summer and fall Chinook populations.

3.5.1. Terminal Migration

3.5.1.1. Size and Age at Maturity

Mean age at maturity for summer Chinook females is 23% three-year-olds, 73% four-year-olds, and 4% five-year-olds. For males, 46% return at age two, 44% at age three, and 10% at age four; less than

[3]Earlier work at Stotan falls in 1923 was likely part of a larger sockeye transplant initiative, to improve fish passage for returning sockeye adults.

1% are five-year-olds. Mean age at return was calculated from coded wire tags (CWTs) for return years 2001-2019; fish removed for broodstock (used and unused); and for First Nations food, social, and ceremonial (FSC) purposes; and may not be totally representative of the true population. Mean body size (post-orbital-hypural body length measured from the eye to the end of the spine) for males and females for this period is summarized in Table 26. Results are similar to those described in Trites *et al.* (1996) for the period from 1977 to 1990.

Table 26. Mean post-orbital-hypural length (POH) in mm, and standard deviation (SD) of male and female summer Chinook by age for return years 2001-2019. Ages were determined from coded wire tags (CWTs) in broodstock and other removals (DFO SEP).

Age	**Males**		**Females**	
	Mean	**(SD)**	**Mean**	**(SD)**
2	364.8	(24.9)	-	-
3	539.2	(29.4)	606.3	(22.8)
4	671.4	(26.5)	690.0	(22.0)
5	-	-	726.2	(37.5)

3.5.1.2. Sex Ratios

Since sex ratios were recorded beginning in 1965, the ratio of male to female adults has been quite variable averaging at about two males for every female and has ranged as high as 9:1 (Trites *et al.* 1996). For further discussion on sex ratios please refer to Section 4.1.1.1.

3.5.1.3. Run Timing

Summer-run Chinook salmon typically enter the freshwater environment to begin their upstream migration around late April/early May and peak migration occurs in late June/July. Historically, these fish would have been able to utilize the more variable flows available during the natural spring freshet period between April and June/July to negotiate two large water falls on the river (Stotan and Nib Falls) and continue their migration into Comox Lake where they could hold in the cooler depths over the summer until the onset of spawning in early to mid October.

3.5.1.4. Straying/Homing

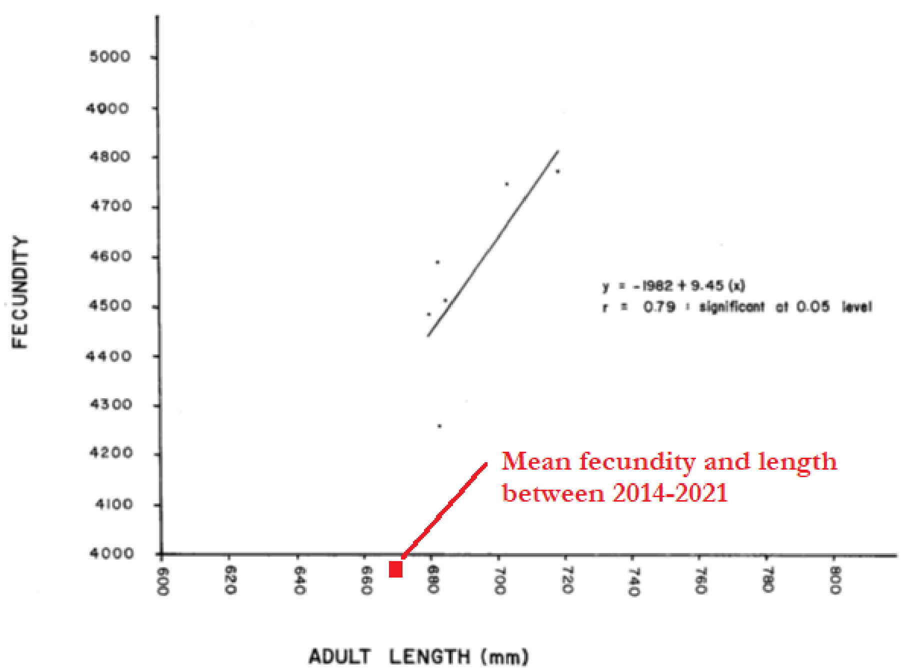
Straying does not appear to be an issue for Puntledge River summer Chinook Salmon. Based on recoveries of CWTs Hatchery Puntledge summer Chinook releases between 1980 and 2021, the stray rate is less than 1%. During this period of hatchery releases only 6, 1, and 9 recoveries were found in Campbell River, Cheakamus River, and Nanaimo River, respectively.

3.5.2. Spawning (Fecundity and Egg Size)

In general, fecundity is highly correlated with fish length and egg size (Beacham and Murray 1993). An analysis of Puntledge summer Chinook fecundity and female length data collected between 1965 to 1976 produced a significant correlation (Figure 33, $R^2 = 0.79$). During the early years of summer Chinook enhancement (1966-1976), average fecundity was calculated at 4,564 eggs per female, which corresponded to an average length (post-orbital hypural - POH) of 691 mm (Figure 33). The average number of eggs produced per female spawner with a mean POH of 669 mm for the period 2014-2021 is estimated at 3,958, which was calculated from live and dead eyed egg count, highlights a decrease in length and fecundity since 1965 to 1976.

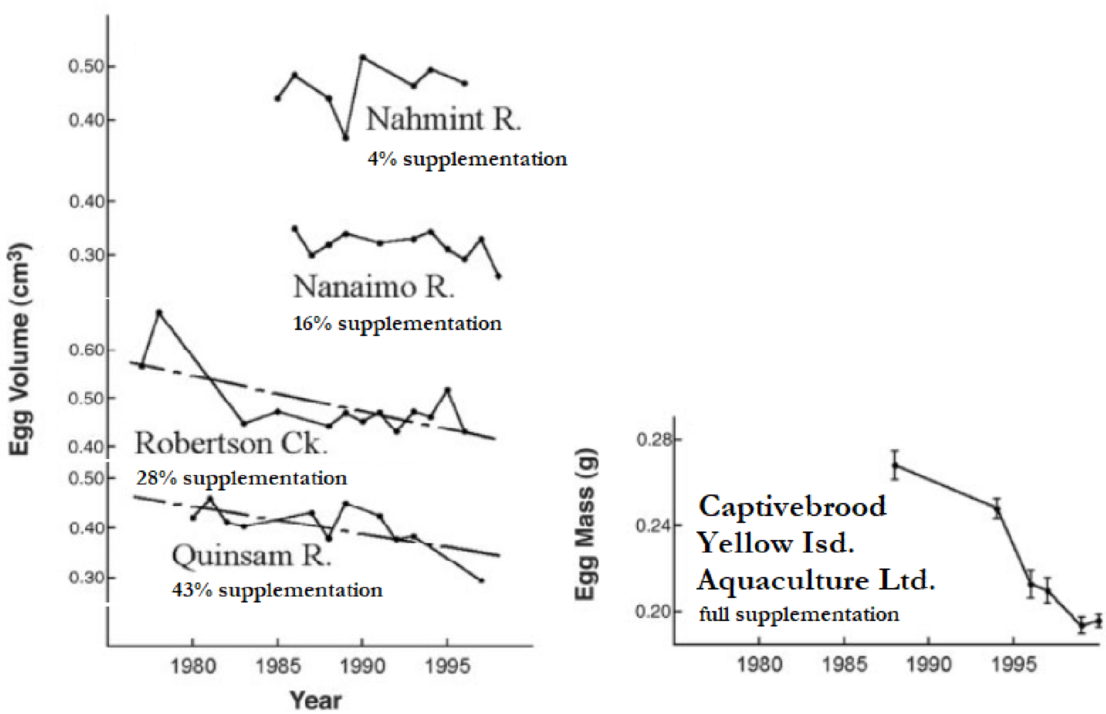
Green egg sample weight data (i.e., unfertilized eggs) indicate that summer Chinook egg size has declined from an average of 0.33 grams in 1987 and 0.31g in 2004 to a dramatic drop of 0.18 g in 2022 and 0.22 g in 2023 (DFO unpublished hatchery data). In contrast, mean weight for fall Chinook green eggs have remained relatively stable and the same for the same period at 0.34 g.

Figure 33. Mean fecundity of summer Chinook (1965-1976) versus post-orbital hypural length (mm) Source: MacKinnnon et al. 1979).



Captive breeding programs have resulted in higher fecundity and smaller eggs (Heath *et al.*2003). Smaller females with smaller eggs and higher fecundity survive as well as larger females with larger summer eggs in a captive brood setting allowing this trait to grow and dominate genetically over time. Overall, the female size over time does not change (Heath *et al.* 2003). In a supplementation hatchery setting, smaller progeny are subject to natural selection once released into the wild and this trait is penalized or selected against. Heath *et al.* (2003) analysed the egg size of four BC coastal Chinook hatcheries with varying degrees of hatchery supplementation (i.e., the number of females spawned in the hatchery divided by the total number of adult females returning to the system, averaged over years that egg size data was collected). Nahmint River and Nanaimo Rivers, which had supplementation rates of 4% and 16%, respectively, showed no decline in egg size. However, Robertson River and Quinsam hatcheries that had supplementation rates of 28% and 43%, respectively, experienced a decline in eggs (Figure 34). There was no significant decline in mean post-orbital hypural female length (mm) for any of the populations with data (p> 0.20, Heath *et al.* 2003).

Figure 34. Egg size versus supplementation rate for captive breeding programs (Source: Heath *et al.* 2003).



In a similar study at the University of Washington Hatchery, where supplementation was considered near 100%, Chinook fecundity, egg size, and freshwater and ocean growth rates were analysed (Quinn *et al.* 2004). Results showed an overall decrease in adult size but the trend was driven by age (i.e., more 3-year-olds returning as adults than 4-year-olds). The 3-year-old females had smaller and fewer eggs than four-year-olds. In contrast, the female age composition of Puntledge summer Chinook has not changed over time (i.e., approximately 75% 4-year-old and 25% 3-year-old females). However, as mentioned earlier female size has decreased from 691 mm between 1966-1976 to 669 mm between 2014-2021. Furthermore, Trite *et al.* (1996) stated that mean female size increased to over 700 mm in 1991, so there was an even more dramatic decrease in size between 1991 and 2014-2021. Part of the decline may be linked to an increase in sportfishing pressure between the 1960s and 1970s compared to the 1990s (MacKinnon *et al.* 1979).

Quinn *et al.* (2004) suggested that egg size and fecundity is developed later in ocean growth and can alter depending on energetic deficiencies. Pink salmon fecundity monitored at sea in late winter maintained stable fecundities but then fecundity dropped sharply under poorer ocean conditions suggesting that eggs could have been reabsorbed (Grachev 1971). Better conditions later in ocean life may results in eggs increasing in size but fecundity does not return to original numbers if eggs were previously completely resorbed. Egg size is the result of the energy attained and then lost in the late marine phase including subsequent energetic losses at maturity. Climate change is causing widespread declines in many species including terrestrial and marine (Oke *et al.* 2020) . In addition, long freshwater migration distances and likely prolong holding could result in resorption and a decrease in eggs size. Overall, it appears that the decreasing eggs size is attributed to a decrease in 4-year-old female length and likely prolonged holding in freshwater under stressful conditions (i.e., high water temperatures), which may have caused eggs to resorb. Another possibility is that larger females have been experiencing higher pre-spawning mortality due to a high vulnerability to water temperature, more difficultly migrating upstream or higher pre-spawning mortality due to predation (see Section 4.1.1.1). The factors mentioned above have not been verified but should be investigated.

Egg size is a major determinate of fry size at emergence (Beacham and Murray 1990). For all five species, incubation temperature was the more important factor in determining alevin length, and egg size was the more important factor in determining alevin weight. Rates of development to hatching and emergence, and alevin and fry size, differed by species in response to changes in temperature. Coho salmon alevins and fry were proportionately larger at 4°C than at 8°C or 12°C, but alevins and fry of Pink salmon and Chum salmon *O. keta* were largest at 8°C. Variation in development characters of Pacific salmon reflected adaptations to each species life history pattern. Puntledge summer Chinooks are currently incubated at Puntledge Hatchery using Puntledge River surface water that is identical to the temperature that wild Chinook spawners and progeny experience throughout the whole incubation period. However, early spawning groups of summer Chinook may be temporarily incubated on chilled water until the ambient river temperature is below 12°C. The hydro reservoir (i.e., Comox Lake storage) likely does not act as a heat sink during the fall (Chilibeck pers. comm.

2022) and therefore water temperatures during the incubation period are not adversely affected (i.e., delayed higher fall water temperatures).

The other consideration related to decreasing egg size amongst salmonids in the Pacific Northwest is changes in the average size and age of return (Ricker 1981). The size of the fish caught, of all species, has decreased because of a size decrease in the population. This decrease did not correlate with ocean temperature or salinity. Chinook salmon have decreased greatly in both size and age since the 1920s. There is evidence that commercial fishing causes a decrease in body size, migration timing and age of maturation, which have implications for fisheries, conservation, and hatchery augmentation (Hard *et. al.* 2008). Trite *et al.* (1996) stated that Puntledge summer Chinook size from the 1970s and 1980s increased in size in the early 1990s after a reduction in fisheries exploitation. However, following a targeted reduction on Cowichan Chinook in the late 1990s and 2000s, exploitation and the potential for size selection on summer Puntledge Chinook remained high. In recent years, sportfishing in the Strait of Georgia (SoG) has been delayed to July 15th and a slot size of a minimum of 62 cm and maximum of 80 cm is now enforced. When combined, these regulations should reduce exploitation and increase the return of larger females, starting in 2021.

Females primarily select male partners equal or larger in size; however, in the hatchery, spawning of broodstock focuses on random pairing to maximize genetic diversity. Models predict that this leads to a decrease in age and size of returns (Chen 2007; Hankin *et. al.* 2009). There is currently a DFO CSAS process underway to review current hatchery practices and develop new preliminary spawning guidelines that emulate natural mating behavior that, amongst other procedures, proposes pairing males with females of equal or greater length to produce progeny that return at an older age and size (Hankin *et. al.* 2009; Hard *et. al.* 2008). A clear outcome is an increase in egg size that is positively related to survivorship (Heath *et al.* 1999). Any subsequent increase in summer Chinook emergent fry size in the Puntledge River should result in higher migration success and survival through the BC Hydro Eicher screens. Overall, selective mating could be adopted as a counter measure strategy to increase hatchery survival and offset the impacts of size selective fisheries on the spawning population age and size structure.

3.5.3. Incubation (and Emergence)

The development of salmonid eggs follows a predetermined course that begins once the egg is fertilized. The rate at which embryonic development proceeds is influenced primarily by temperature. As temperature increases, so does the rate of development, within specific limits. Salmon populations have adapted to the long-term average thermal regime of their environment such that reproduction is timed to optimize growth and survival success of the emergent fry (Quinn 2005). The predicted embryonic development period for Chinook is based on accumulated thermal units (ATUs) and summarized in Table 27. Mean emergence is estimated to range between late January and early March, which is based on mean daily Puntledge River temperature data analysed between 2008 and 2018, and an assumed emergence at 1,000 ATUs.

Table 27. Predicted embryonic development period (in days and ATUs) for Chinook salmon using various models and from IncubWin, a computer program for predicting embryonic stages in Pacific salmon and steelhead trout (Source: Billard and Jensen 1996).

Temperature	Yolk Plug Closure		Eyed Stage		50% Hatch		Emergence	
(°C)	Days	ATUs (°C-days)	Days	ATUs (°C-days)	Days	ATUs (°C-days)	Days	ATUs (°C-days)
5	26.7	133.5	51.5	257.5	102.4	511.8	199.2	996.1
7.5	17.9	134.5	34.2	256.6	70.3	527.5	136.2	1021.8
10	13.4	133.5	24.9	249.2	52.6	526.4	93.2	931.7
12.5	10.6	132.1	19.2	240.5	42.1	525.7	63.7	796.4

3.5.4. Early Rearing

Puntledge Chinook are predominately "ocean-type" fish. After emerging from the gravel in March, fry begin to move downstream with peak migration occurring in April (Lister 1968). The timing of the wild summer Chinook smolt migration is unknown. However, recent monitoring of juvenile summer-run Chinook migration at the Puntledge diversion dam evaluation facility noted that emergent Chinook fry began moving downstream in February at a size of 35-40 mm in length, and continue until April, with another peak in June (Guimond and Taylor, unpublished data; Tryon 2008). For example, between 1966 and 1968, Chinook fry from all the fish that spawned naturally in the Upper hatchery spawning channel were observed either migrating downstream in March/April at the early feeding "fry" stage, or until June/July reaching lengths of 70-80 mm before migrating downstream (Lister 1968). Yet, it is unknown when they migrate to K'omoks Estuary.

3.5.5. Rearing in the Estuary

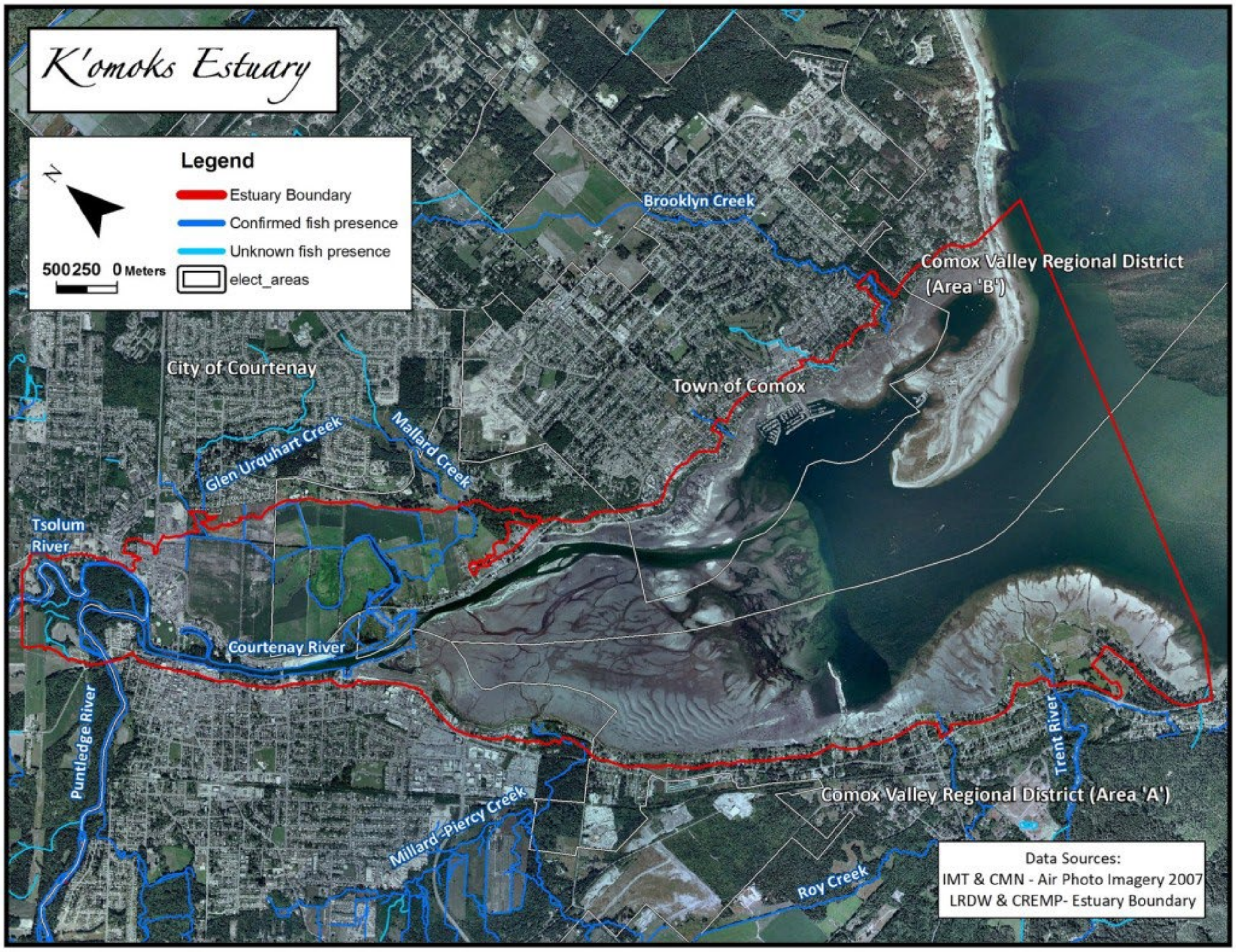
Estuaries are partially enclosed coastal water bodies where freshwater mixes with saltwater. The K'ómoks (Courtney River) estuary (Map 6) is one of the most important estuaries on Vancouver Island and one of eight that are ranked as Class 1 estuaries in BC (WWF 2013). It is a special and unique feature of the Comox Valley and supports 145 bird species (recognized as an internationally important bird area), 218 plant species, 29 fish species including Chinook (summer and fall runs), Coho, Chum, and Pink salmon and a plethora of intertidal life (Hamilton *et al.* 2008). The Puntledge and Tsolum Rivers merge to form the Courtenay River, which is the freshwater body that feeds the estuary.

Estuaries function as important rearing areas for juvenile salmonids. They act as a transition zone for juvenile salmonids out-migrating from freshwater to marine waters providing an opportunity for the physiological changes necessary to adapt to the saltwater phase of their life cycle. In addition, healthy estuaries provide productive foraging areas and refuge from predation. Studies have shown that high estuarine productivity supports the rapid growth of juvenile salmon, which in turn can increase early marine survival (Beamish *et al.* 2004; Duffy and Beauchamp 2011). Therefore, estuaries are productive nearshore habitats that are critical for juvenile salmon survival (Levings 2016; Kennedy 2018).

Nearshore habitat is defined as the area between shoreline bluffs to where the water becomes too deep for light to penetrate and support plant growth. It also includes marine and estuarine habitat but stops at the point where saltwater no longer mixes with freshwater, and can include rocky and sandy beaches, mudflats, kelp and eelgrass beds and lagoons.

Chinook salmon (*Oncorhynchus tshawytscha*) have been shown to be more dependent on estuarine habitat for early rearing than other salmon species. In one study, juvenile Chinook were shown to have a longer window of estuary entry timing (from February to May) and a longer estuary residency compared to pink and chum salmon species (Chalifour *et al.* 2020). Research on the K'ómoks estuary demonstrated that Chinook fry stages were more dependent on the estuary than smolts (the smolts moved through the estuary quickly) and were observed in the estuary from late March (when sampling started) to July, with overall density peaking in June (Tyron 2011). This indicates that estuarine habitat is important for early marine growth and survival of juvenile Chinook salmon. Thus, degradation of this critical nearshore rearing habitat can lead to juvenile Chinook mortality or fitness reduction for a variety of interacting reasons including lack of access to food, increased predation, stress due to anthropogenic activity, inter- and intra-species competition, increases in disease, parasites or pathogens, pollution, and poor water quality, among others.

Map 6. The K'ómoks Estuary, boundary is shown in red (Source: Project Watershed).



3.5.6. Off-shore/Marine Phase

The two stocks of Chinook salmon in the Puntledge River are caught in different fisheries. Summer-run Chinook are mainly caught in the Georgia Strait sport fishery with a small proportion intercepted in the Georgia Strait troll and Johnstone Strait net fisheries, while fall-run Chinook exploitation is concentrated in the northern and central BC and southeast Alaskan fisheries, similar to the Quinsam/Campbell fall Chinook salmon (Trites *et al.* 1996; Nagtegaal *et al.* 2000). The low escapements of Puntledge River summer and fall Chinook prompted the closure of the river and estuary to sport fisheries in 1965, which remain closed, and altered the boundaries of the Georgia Strait troll fishery in 1970 for conservation purposes. Regardless of these measures, the sport fishery caught 70% of the summer Chinook run in 1975, and between 1975 and 1981, the combined sport and commercial fisheries removed 84% of the summer Chinook run (Trites *et al.* 1996). Following the 1985 Pacific Salmon Treaty and the more recent conservation measures imposed for southern BC coho salmon and Georgia Strait Chinook salmon, total exploitation rates on Campbell/Quinsam fall Chinook decreased to between 30% and 40% (Nagtegaal *et al.* 2000). Exploitation on Puntledge River fall Chinook is probably comparable (Nagtegaal pers. comm. 2023) while exploitation rates on summer Chinook have dropped to about 50% between 1990 and 1995 with the majority of exploitation occurring in the Strait of Georgia sport fishery (Trites *et al.* 1996).

One of the significant characteristics of the summer Chinook stock, typical of all Lower Strait of Georgia (LGS) Chinook stocks, is their residence within the Strait of Georgia (DFO 1999). For the most part, LGS Chinook remain in the Strait of Georgia for their entire marine growth stage, while a small portion will migrate further north to areas along central and northern BC and Alaska. This life history trait makes this stock an important component of local fisheries within the Strait, as well as it being particularly vulnerable to exploitation.

3.6. Timeline of Activities Affecting Summer Chinook Population

There are several other noteworthy activities that may have affected summer Chinook escapements between 1921 and 2022, which are listed chronologically below.

- **1912** – First hydro facility constructed with the fishway at the diversion dam, but not at the impoundment dam at the outlet of Comox Lake.
- **1923** – Remedial work done at Stotan and Nibs Falls to improve upstream migration.
- **1927** – Fishway built at the impoundment dam.
- **1953-57** – Hydro facility upgraded. During construction, a coffer dam failed and released silt and sediment impacting the spawning grounds. The diversion dam height was increased, which backwatered the spawning grounds. In subsequent years, concerns grew over the mortality of juveniles entrained through the Francis turbines, which was thought to range between 30-60%.

- **1955** – During the first year of operation of the expanded hydro facility, adult summer-run Chinook salmon were delayed at the tailrace pool of the powerhouse, a phenomenon not previously recorded during the four decades of operation of the facility by Canadian Collieries (Hourston 1962).

- **1954-1967** – Flows greater than 28 m^3/s (1,000 ft^3/s) were found to delay migration and cause serious difficulty in adult passage through Stotan and Nib falls, while flows in the range of 14-23 m^3/s (500-800 ft^3/s) provided easier passage (Lister 1967). Optimum flow for passage was estimated at 200 cfs (Holden 1958).

- **1965** – A 1,900 m^2 spawning channel was built just above the diversion dam and operated until 1971. Fishways at both dams were closed to adult passage. All upstream summer Chinook migrants were diverted into the channel. The site experienced high pre-spawning mortality due to sedimentation, high temperature and gas supersaturation (caused by daily increases and evening decreases in water temperature). Escapements remained below 500.

- **1972** – Hatchery production began at the diversion dam site and became known as the Upper Hatchery Site. All summer Chinook adult captures were used for broodstock. All smolt production was released to the river at the upper hatchery (i.e., just below the diversion dam).

- **1968-1977** – More remediation work in the river to specifically improve summer Chinook migration above the falls.

- **1979** – Hatchery production and fence operation began at the lower hatchery site.

- **1988** – There was a peak high seal count in Comox Estuary in 1988, which corresponds to a very low summer Chinook salmon escapement.

- **1991** – The Comox dam fishway operated as a pool/weir fishway until 1991 but was not operated properly due to the lack of summer Chinook above the dam (i.e., all adults were intercepted and used in the hatchery program). The province modified the fishway to a submerged orifice design; however, attraction flow was not adequate when the lake level dropped in the summer (most if not all summer adults were still being captured at the diversion dam fishway for hatchery broodstock).

- **1993** – Eicher screens were installed to bypass fry that were entrained in the penstock at the Diversion Dam back to the river below the dam. MorInitial tests estimated a 95% efficiency in bypass.

- **Early to mid 1990s** – The resident seal population grew; a fence and electric field was piloted to deter seals but failed.

- **1997-1998** – Seal numbers peaked in the river (over 60 seals/year); 32 were culled in-river in the first year and 21 in second year.

- **1998-1999** – The adult fence at the lower hatchery was upgraded to a fixed steel structure.

- **1997-2001 brood years** – A captive brood hatchery program was in operation with a final juvenile release in 2005.

- **2001** – Puntledge Hatchery also began allowing adults to bypass the diversion dam and migrate into Comox Lake.

- **2004** - The Comox dam fishway was modified again by DFO SEP and the Province, installing wider submerged orifices that maintained higher fishway flow and attraction during low summer lake levels. A study in 2005 measured a success rate of +75% adult passage.

- **2004** – The upper hatchery experienced a near complete loss of summer Chinook broodstock due to high temperatures and gas supersaturation.

- **2005** – The hatchery began collecting summer Chinook at the lower hatchery fence and transported 160 adults to Rosewall Hatchery. Also, BC Hydro implemented the Water Use Plan (WUP) and started providing pulse flows for migrating adults.

- **2005-2006** – A 4,750 m^2 gravel platform was constructed in the headpond at Supply Creek providing spawning habitat for 950 Chinook spawners.

- **2010-2019** – The efficiency of the Eicher Screens were reassessed. Screen efficiency was found to be much lower for emergent fry. At a generation rate of 24 MW the average efficiency was 43% for 38-39 mm fry.

- **2010** – The hatchery began collecting up to 50% of summer Chinook returns for broodstock.

- **2012** – The upper hatchery site was decommissioned, and all summer Chinook production has since been carried out at the lower Puntledge River hatchery while all summer Chinook juvenile production has been released at this location since 2013 (brood year 2012).

- **2013** – Start of DNA sampling (parentage-based tagging or PBT) of all summer Chinook broodstock.

- **2015** – The grating and camera tunnel in the Comox Dam fishway was plugged with debris. Hatchery staff investigated the passage conditions with another underwater camera and suspected that fish passage was severely impaired due to the debris and excessive velocity through the tunnel. The debris and tunnel were not removed until the summer of 2022.

- **2020** – BC Hydro began implementing their Puntledge Spill Planning Application, which governs the diversion dam spill and allowable power generation required to achieve a 90-95% fry diversion rate at the dam during the critical emergent fry migration period (February to April). It is assumed that the number of fry that are diverted over the dam is proportionate to the flow through the penstock (i.e., the relative density (fish/m^3) of fish in spill versus penstock water does not change with water volumes or different river discharges).

- **2021** – A 1,874 m^3 spawning platform was installed in upper Reach B for summer Chinook.

- **~2020-2023** – The hatchery has observed that the summer Chinook eggs and emergent fry are much smaller than in the past.

3.7. Puntledge River Chinook Salmon Restoration Efforts

3.7.1. Chinook Enhancement

3.7.1.1. Historical

Artificial enhancement of Puntledge summer-run Chinook commenced in 1972 following several years of poor survival rates in the spawning channel. This consisted of temporary incubation facilities installed at the old Canadian Collieries powerhouse, and in two 22.9 m (75-feet) concrete Burrows Ponds constructed at the upper hatchery (spawning channel) site (Marshall 1973). During the period 1972 to 1976, the number of adults that spawned in the channel decreased from 255 to 16, while the numbers that were held in the ponds for egg takes increased. Construction of a full-scale hatchery was completed in 1979 downstream of the powerhouse. The main goal of the hatchery was to save the summer and fall-run Chinook stocks from extinction, and to boost other salmonid species in the watershed to healthy levels.

The Upper hatchery spawning channels (also discussed in Section 2.7.2) continued to function as a collection area for summer Chinook broodstock. Adults voluntarily entered the spawning channel through the fishway and were seined and transferred to the Burrows Ponds where they could be sorted, treated for fungal infection, and held until spawning. Adults that escaped being seined were allowed to spawn naturally in the channel with resulting fry passing through the fishway into the river below the dam. Typically, between 30% and 50% of the adults that arrived at the diversion dam did not swim into the upper hatchery channel. Hatchery staff would therefore seine the pool below the dam, as well as other pools in the river downstream (Barbers pool, and Cedar pool above the logging bridge). This was usually conducted during late summer and into early September during the BC Hydro maintenance shutdown (Munro pers. comm. 2023). Since 2001, the use of malachite green was no longer used, and hatchery staff refrained from handling adults when water temperatures increase above 16°C to reduce stress and fungal growth.

During this period, operation of the lower barrier fence also played a critical role in the collection of Chinook broodstock. All summer Chinook broodstock arriving at the fence were allowed to continue their migration upstream until reaching the upper hatchery. The barrier fence was typically closed to summer Chinook migration around August 1st to prevent early fall-run Chinook migrants from accessing habitat upstream and allow pink salmon to be collected at the lower hatchery. This measure likely helped to maintain the genetic separation of the two stocks.

Overall, the spawning channel was successful in maintaining the summer Chinook stock but unable to rebuild the run to historical levels (MacKinnon *et. al.* 1979). The ultimate success of the channel was dependent on survival rate of fry after release into the Diversion Reach of the river (Marshall 1972). Prior to hydro development, fry would have emerged into the wide low gradient stable and more productive reach downstream of Comox Lake (now the headpond), rather than the higher gradient, bedrock-controlled diversion reach. Fry survival was now dependent on flow control through the diversion reach and fry were susceptible to rapid and extreme variations in discharge events. To overcome potential impacts of extreme discharge events, various mitigation options were considered. For example, hydro operations in winter/early spring often required spilling surplus winter storage that coincided with fry emigration from the channel. Another option at this time was to transplant fry into the headpond reach but the installation of an effective diversion around powerhouse intake works would be required. A third proposed option was to rear a portion of fry to smolt stage at hatchery ponds at the upper hatchery site (Marshall 1972). Additional details on restoration efforts such as gravel placement and improvements to fish passage obstructions are provided in Section 2.7.

3.7.1.2. Contemporary

Puntledge River summer Chinook are enhanced at the Puntledge River Hatchery as a 'Conservation and Assessment' objective, as described in the DFO SEP Biological Assessment Framework (DFO 2019). Specifically, the objectives are to increase the abundance of spawning Puntledge River summer Chinook to a level that minimizes risk of functional extirpation while maintaining genetic and adaptive integrity of the population and minimizing potential for failure of any given brood year. Puntledge River summer Chinook are a Pacific Salmon Treaty (PST) indicator stock for estimating exploitation and marine survival of east coast Vancouver Island (ECVI) summer-run Chinook populations. These fish have been tagged with coded-wire tags (CWTs) since 1971. The current enhanced production target is 500,000 sub-yearling smolts, which was last realized in 2014.

3.7.2. Other Initiatives

3.7.2.1. Captive Brood

The Puntledge River "Captive Broodstock Program" was initiated in 1997 to assist the rebuilding of Puntledge summer-run Chinook by raising selected brood stock from fry through to maturity and harvesting eggs to supplement the Puntledge Hatchery summer Chinook enhancement program. The program was prompted by seven consecutive years of escapement levels below 500 and was modeled after the Hurd Creek Captive Brood Stock Program in Washington state, USA, which has been successfully enhancing Chinook since 1992. It was recognized that the success of this program would be dependent on the implementation of a more extensive strategy that would tackle other issues affecting the survival and recovery of the summer Chinook stock (i.e., habitat issues, fishing pressure, and predation).

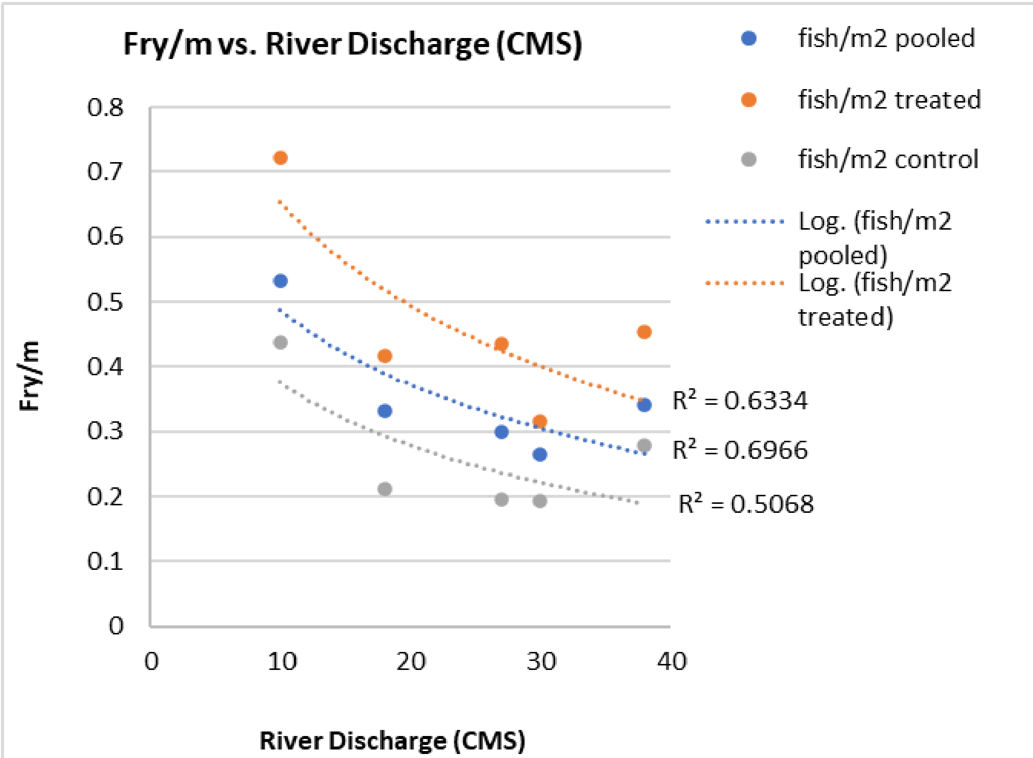
3.7.2.2. Emergent Fry Habitat Assessment Study SWD Installation

In 2016, recycled Christmas trees were installed on the left bank of the Puntledge River headpond as small woody debris (SWD) habitat complexing bundles (Guimond and Sheng 2016). This pilot project was a combined effort of the DFO, the Fish and Wildlife Compensation Program (FWCP), and the Courtenay and District Fish and Game Protective Association (CFGA). The objective was to investigate whether habitat enhancement/restoration activities upstream of the Puntledge diversion dam could mitigate summer Chinook fry mortality and entrainment at the hydro facility. The purpose was to encourage juvenile salmon to rear and migrate at enhanced habitat locations (Map 7) rather than near the hydroelectric intake (on river right) where the risk of entrainment and mortality is higher. Thus, the fish would be encouraged to remain longer in the headpond, growing larger, before bypassing the hydroelectric facility intake (FWCP 2022). Increasing juvenile survival during their seaward migration is an important step in maintaining the genetic integrity of the summer run, and necessary for their long-term adaptation and conservation. Results showed a marginal significant difference in mean Chinook fry density at the enhanced sites compared to the untreated sites ($F = 4.68$, $p = 0.047$, Guimond and Sheng 2016; Figure 35. The difference in density also suggested that cover was important to rearing fry while water depth was not.

Field observations during the pilot project indicated that cover needed to stay rigid in the water and dense enough to reduce water velocity and maintain a stable low velocity environment for refuge and rearing. Many of the installed tree clusters were constantly moving or swirling when discharge and water velocities increased. Areas where the water was deeper along the edge of the riverbank, which was often the case on the left-side of the river, were more exposed to higher velocities and this swirling issue. In contrast, the banks on the right side of the river were generally shallower and the small existing woody debris along the banks was able to provide good rearing and refuge habitat at higher river discharges. It is surmised that the Christmas tree clusters would have performed better on the right side of the river; however, the original objective was to encourage Chinook fry to rear and migrate along the left bank to better avoid the BC Hydro diversion dam intakes and have a better chance of migrating down the fishway or swimming over the dam. In hindsight, improving the right bank and overall refuge and rearing habitat in Reach B would likely increase the survival of summer Chinook fry by encouraging the fry to rear for a longer time, reach a larger size, and improve survival through the Eicher Screens.

Map 7. **Location of small woody debris (SWD) and untreated (UnTr) sites in the Puntledge River headpond upstream of the diversion dam (from Guimond and Sheng 2016).**

Figure 35. **Chinook salmon density at small woody debris (SWD) and untreated (UnTr) sites compared to Puntledge River discharge (Source: Guimond and Sheng 2016).**



4. THREATS AND LIMITING FACTORS

Pacific salmon have a complex life cycle that includes freshwater, estuarine, coastal, and the ocean environments. Each of these habitats provides crucial elements for the salmon's survival as they move through their incubation, freshwater rearing, estuary transition, ocean residence, migration and spawning phases. In each of these phases, human activities can have adverse impacts on their survival. These activities are most significant during their freshwater stages (Waldichuk 1993). The known habitat and hatchery-based features and factors that affect successful Chinook production and overall watershed health in the Puntledge River are outlined in the following sections. Commercial and sport fisheries as well as hatchery and enhancement initiatives can also have a significant affect on Chinook production and are also discussed in this chapter.

This section provides an evaluation of 58 potential threats to Puntledge River summer Chinook production, which are summarized in a table in Appendix A. These threats include the 70 threats identified by DFO in 2023 (DFO 2023) but considers the following changes: (1) some of the 70 threats were combined under one threat, and (2) some threats were added. The potential threats were then considered within the salmon life cycle framework that includes all life stages, life history strategies as well as the various habitats upon which salmon depend in those various life stages. Two broad based categories were also used to segregate these threats: (1) habitat and ecosystem related factors, (2) hatchery and enhancement related factors. It is expected that this list of threats will continue to evolve over time; it started with the recovery/management plan developed for Cowichan River Fall Chinook in 2011 and continued with the development of the DFO workshop report (DFO 2023) and this report. The following sub-sections provide the 58 threats identified in Appendix A for terminal migration and spawning, incubation, early rearing and rearing in the estuary.

4.1. Terminal Migration and Spawning

4.1.1. Elevated Predation

Predation can have an important influence on the dynamics of salmon populations (Fresh 1997). The factors influencing predator-prey relationships are complex, though efforts have been made to understand these general patterns. Mather (1998) proposed that salmonid predation increases when (a) prey are a size that predators can easily consume, (b) predator and prey overlap in time and space, (c) predator numbers are large relative to prey numbers, (d) predators and prey are aggregated in the same place at the same time, (e) no alternative prey is present, preference for the target prey is strong. The following sections summarize predation on salmonids during terminal migration and spawning by two main predators; however, predation can occur from a variety of predators that can consume salmon at different life stages. Further details area provided in Section 4.1.1, but also Sections 4.2.1, 4.3.1, and 4.4.1.

4.1.1.1. Seal Predation

Prior to 1970, Pacific harbour seals (*Phoca Vitulina Richardsi*) were hunted without protection and abundance was depleted to historical low levels of roughly 5,000 individuals in B.C (Figure 36). Following the ban on seal hunting, the population rebounded and is now estimated at roughly 105,000 individuals (DFO 2009). In the SoG, harbour seal abundance increased at a rate of about 12% per year and then stabilized in the 1990s at a population size of around 39,400, with an estimated density of 13 seals/km of shoreline (DFO 2009, Ford 2014). The SoG population has remained stable at this population level in the last decades, although, transient killer whales have impacted seal distribution, causing them to disperse more widely to avoid predation (Trites pers. comm. 2022; Ashley *et al.* 2020).

In the Comox Harbour area, there are no historical records of seal use of the area for feeding or breeding. Between 1974 and 1983, following a period of intensive seal hunting during the first half of the 20th century, fewer than ten seals were counted at one time in Comox Harbour (Figure 37). However, after seal hunting was banned, the seal population had increased exponentially to ~400 individuals by 1990 and had reached 750 by 2006 (Olesiuk 2006).

Figure 36. Census population size of harbour seals in the Strait of Georgia (modified from Trite *et al.* 1996).

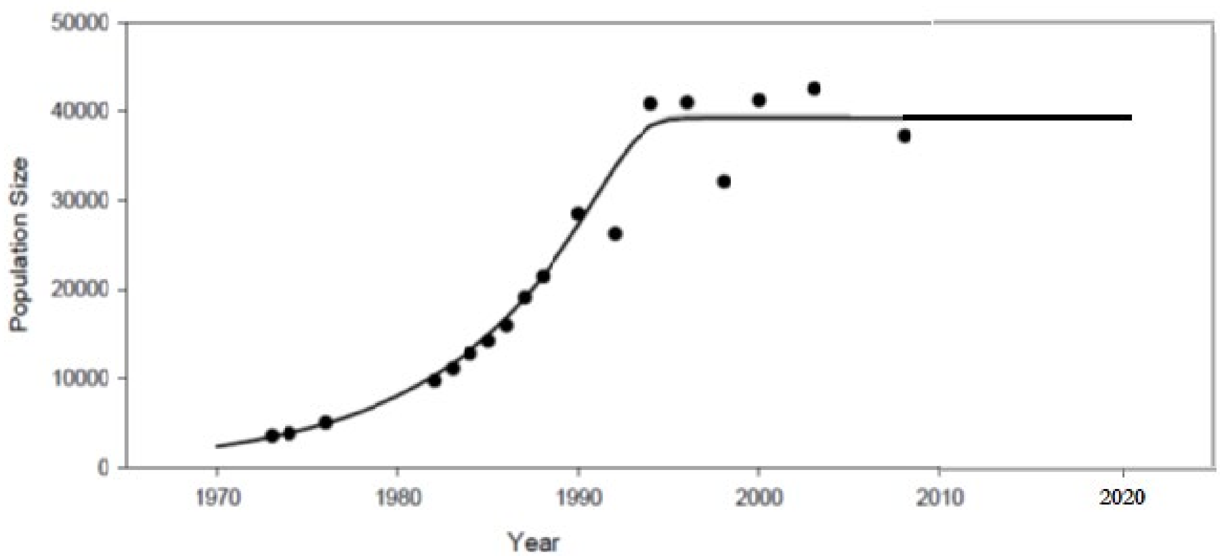
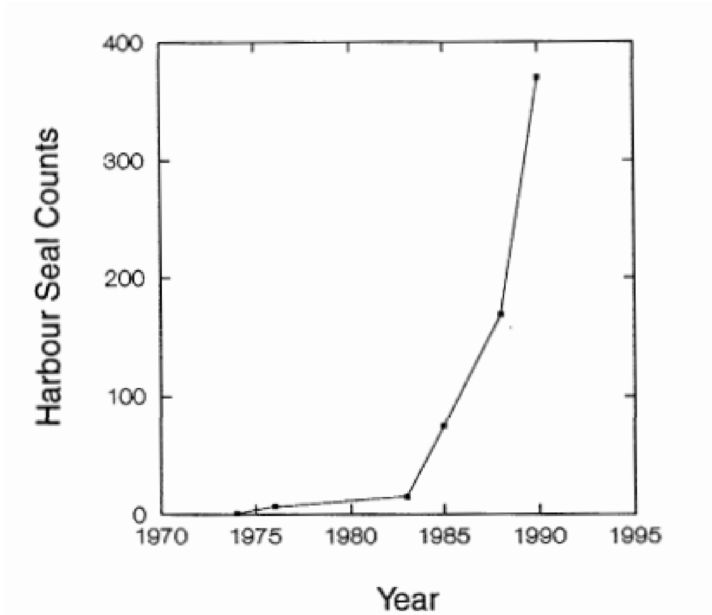


Figure 37. **Number of Harbour Seals counted in Comox Harbour during the month of August (Trite *et al.* 1996).**



Seals have long been suspected of negatively affecting BC's salmon stocks (Austen *et al.* 2016). Seal predation on salmon has been documented in the Puntledge River, but there have been no comprehensive studies on the movement or foraging patterns of harbour seals in the Strait of Georgia in relation to juvenile salmonids. A study conducted in 1995 within the K'omoks estuary area observed harbour seal predation attempts on Chinook salmon over the summer and fall of 1990 (Olesiuk *et al.* 1995). They estimated that harbour seals killed 869 Chinook salmon (based on the average number of successful pursuits observed per hour), of which 362 (42%) Chinook were caught by seals in the K'omoks estuary and the remining 507 (58%) were taken in the Puntledge River (Olesiuk *et al.* 1995). Based on the total escapement that year, they estimated that 35% of the fish that arrived in the K'omoks estuary were captured by harbour seals before they could spawn (i.e., 869 individuals out of an escapement of 2,498).

A study on harbour seal diet based on scat samples (i.e., prey bone/cartilage presence in scat) during the 1980s indicated that most of the annual diet was composed of herring and hake (75%), while salmon species represented only an average of 4% (Olesiuk *et al.* 1990). The salmonids consumed consisted mainly of adult salmon of unknown species that were taken as they returned to rivers to spawn, especially while holding in estuaries. As well, a recent study of seal diet in the Salish Sea using scat DNA samples showed that seal diet was composed of 15% salmonids and 46% hake and herring (Figure 38, Thomas *et al.* 2022). Chinook salmon juveniles were consumed primarily between April and September while Chinook adults were consumed from July to November (Figure 39), which supports previous findings by Olesiuk *et al.* (1996 a,b).

Figure 38. **Harbor Seal DNA diet percent averages (RRA) for all of the Salish Sea samples combined. Considerable diets variability existed between haulout sites and regions in the Salish Sea (Thomas *et al.* 2022).**

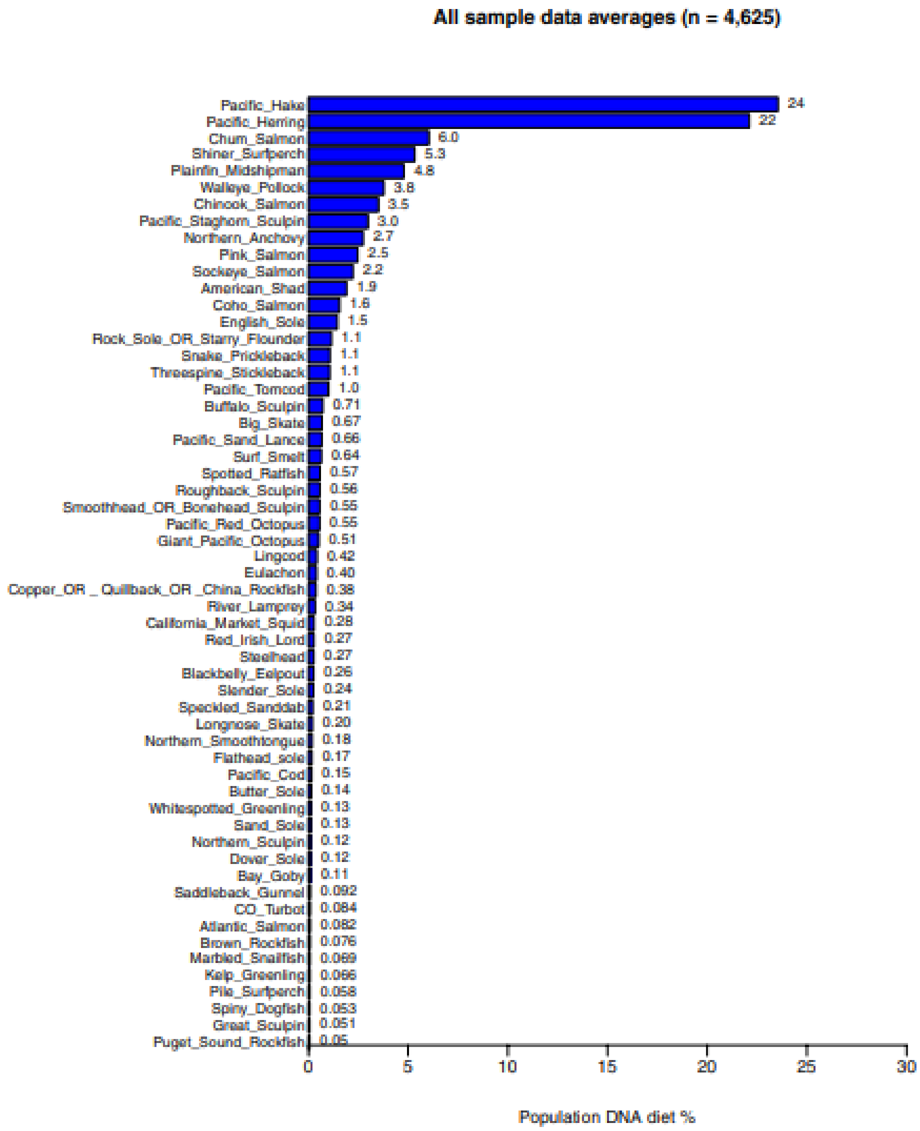
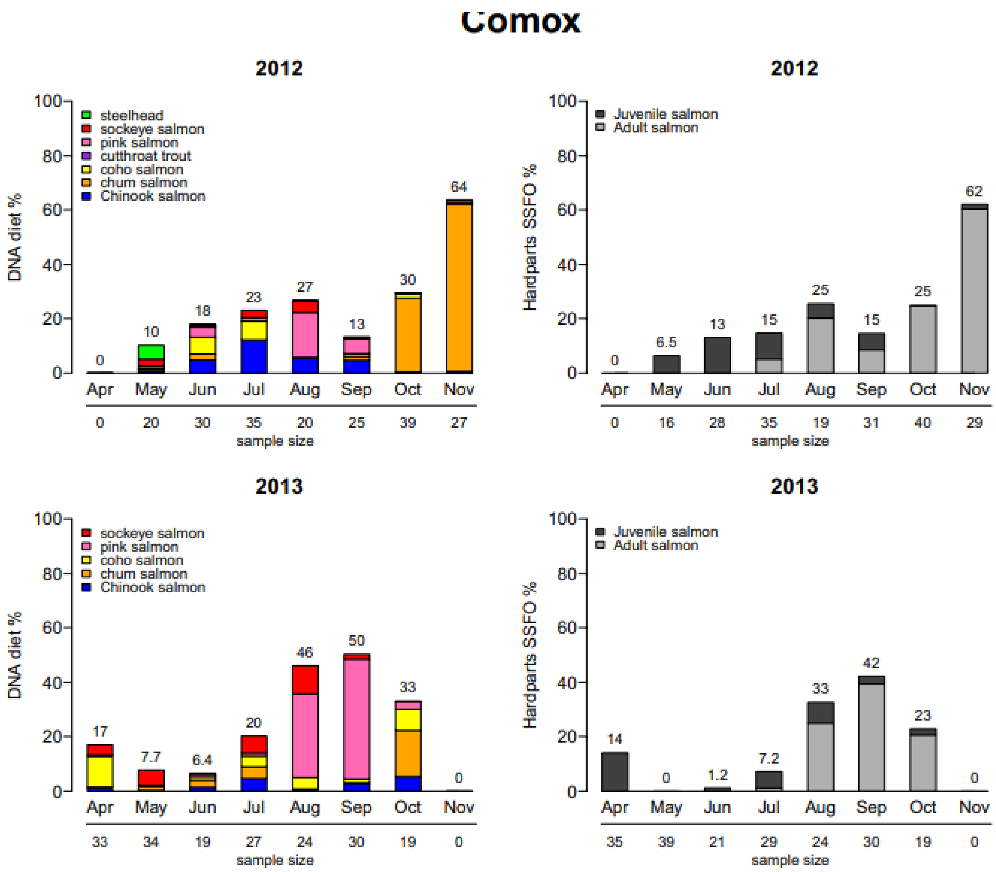


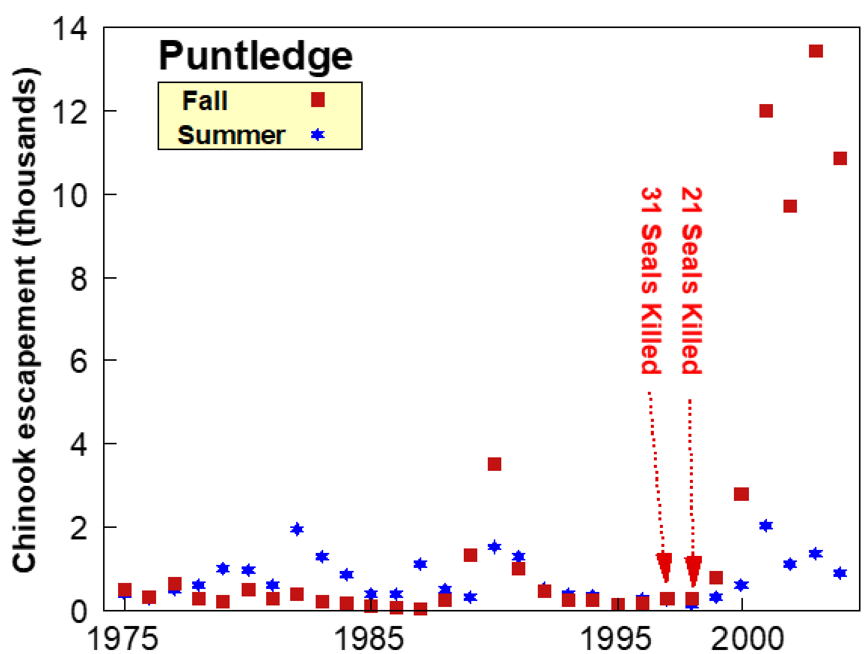
Figure 39. **General agreement between scat DNA RRA (left) and scat prey bone/cartilage remains analyses using wPOO (Hard parts SSFO%) (Right) from the same set of scat samples. Data shown are the salmon component of harbour seal diet from the Comox collection site over two years: 2012 (top) 2013 (bottom) (Source: Thomas *et al.* 2022).**



Various factors that could affect predation success by seals have been identified. These include the density of fish species, the availability of artificial light to aid seals in locating prey, easier accessibility of seals into the lower river during high tide, salmon smolt migration in relation to flows, differences between male and female salmon behaviour and use of the habitat, and the straightening and simplification of the riverbanks due to industrial and urban development, which reduce the amount of cover for salmon. Over the decades, various methods have been used to reduce harbour seal predation in the Puntledge and Courtenay Rivers. The following paragraphs provides details on some of these key factors that affect seal predation success, as well as management methods to reduce the impact of seal predation on salmon in the Puntledge River and K'omoks Estuary.

A barrier fence was operated at the mouth of the Courtenay River between June and September 1998 to prevent seals from moving upstream into the river to prey on spawning Chinook salmon. While the fence did limit upstream movement of seals, its effectiveness in reducing salmon predation was limited because the fence also delayed salmon migration, allowing seals to capture them as they held below the fence. An acoustic deterrent device was used at the seal fence but was found to be ineffective. Additionally, a series of triads (i.e., cement interlocking columns) were placed along one side of the river downstream of the seal fence to provide refuge for adult salmon; however, these also proved ineffective and were poorly used by adult salmon. Finally, due to the poor success of the deterrent methods, 52 seals, likely representing 75% of the resident seal population, had to be culled in 1997-1998 (Figure 40; Brown *et al.* 2003).

Figure 40. Seal culls in the Puntledge River compared to summer and fall Chinook escapements (Bonnell 2004).



Anthropogenic changes to the environment can create 'hot spots' where seals are readily able to prey upon large numbers of fish (Pacific Salmon Foundation 2021). In particular, the former Field Sawmill site – now referred to as Kus-kus-sum - is an area that acts as a "pinch point" and seals have been observed using the steel sheet piling wall to their advantage to trap and feed upon migrating salmon (Figure 41 to Figure 43) as well as passing on this learned behaviour to seal pups (Miller pers. comm. 2022). The site was originally a tidally influenced riparian area and, based on interpretation of air photos from 1931, it was forested and the eastern portion was marsh-like with tidal creeks flowing through it. As the site was developed the original vegetation on the property was cleared, the salt marsh wetland was filled in and paved over, and the foreshore was eventually armoured with a steel-clad retaining. This wall was built into the Courtenay River and backfilled to accrete additional land for the sawmill operations. The sawmill was shut down in 2006 and put up for sale in 2008. The Comox Valley Project Watershed Society (Project Watershed) saw an opportunity to acquire the site and to restore it to its natural functioning condition. Project Watershed spear headed the initiative, along with their two partners the City of Courtenay and the K'ómoks First Nation, to purchase the site from Interfor Corporation for the purposes of restoration and long-term conservation. Project Watershed successfully raised the money to purchase the property and restoration of the site started in 2021 and is slated to be completed in 2023, at which time the steel-sheet piling will be removed and the site will be reconnected back to the river. The reclamation and restoration of the Kus-kus-sum site provide the following benefits:

- Support for both a critical ecologic component/habitat (Class-1 Estuary) and a threatened aquatic species: summer-run Chinook salmon;
- Increased habitat complexity, including high value critical saltmarsh habitat;
- Help to mitigate and/or reduce seal predation on both adult and smolt life-stages; a known and significant source of recruitment mortality; and
- Linkage with and provide further enhancement of existing community-based restoration initiatives (ecological connectivity).

Figure 41. Steel-clad concrete retaining wall along the site (Source: Project Watershed).

Figure 42. Seal catching salmon alongside the Kus-kus-sum site (Photo credit: Miller, pers. Comm. 2022).

Figure 43. Harbour seal with salmon near the Kus-kus-sum (Photo credit: Terry Thormin).

Summer Chinook Escapement Trend Following a Reduction in the Resident Seal Population

A reduction in the seal populations foraging on salmon within the Puntledge River may be associated with an increase in Chinook production. Recent internal data from DFO shows the observed recruitments obtained from previous releases of hatchery Chinook (Figure 44). While this data ignores natural production, it shows a spike in adult returns between 1999 and 2002, which is likely a compounded benefit of: (1) reduced predation on out migrating juveniles by seals in the contributing brood years, and (2) reduced predation by seals on the returning adults. During this period, Chinook returns to the Puntledge River increased substantially more than returns to neighbouring rivers (Figure 45). DFO speculated that seals were attracted to the large numbers of returning pink salmon and remained in the lower river rather than moving upstream to feed on Chinook. Furthermore, seal culls likely reduced predation on Chinook even if the contribution of this factor is unclear.

Hatchery release numbers remained low to moderate, yet there was a strong increase in total escapement starting in 1999 up to and including 2004. In 2005, there was a very large escapement, but that was associated with an unprecedented large hatchery release four years prior (Pellet and Thom 2022). Of note, recruitment starts to drop off in 2003, which, although there were no official counts, may be due to recolonization of pinnipeds to the area. Additional escapement fluctuations may be due to varying annual seal abundance and/or other marine survival influences.

Available data also suggests that natural Chinook production may also have benefited from changes in seal population. Prior to seal culls in 1997 to 1998, the survival of summer Chinook cohorts released from seapens was on average twice (2.4 times) that of the hatchery releases into the river (Figure 46). However, after seal culls occurred, survival of Chinook salmon river releases increased and seapen released survival decreased (1.6 times).

Figure 44. Puntledge River summer Chinook smolt hatchery releases compared to adult returns (escapements) (Source: DFO 2022; Pellet pers. comm. 2022).

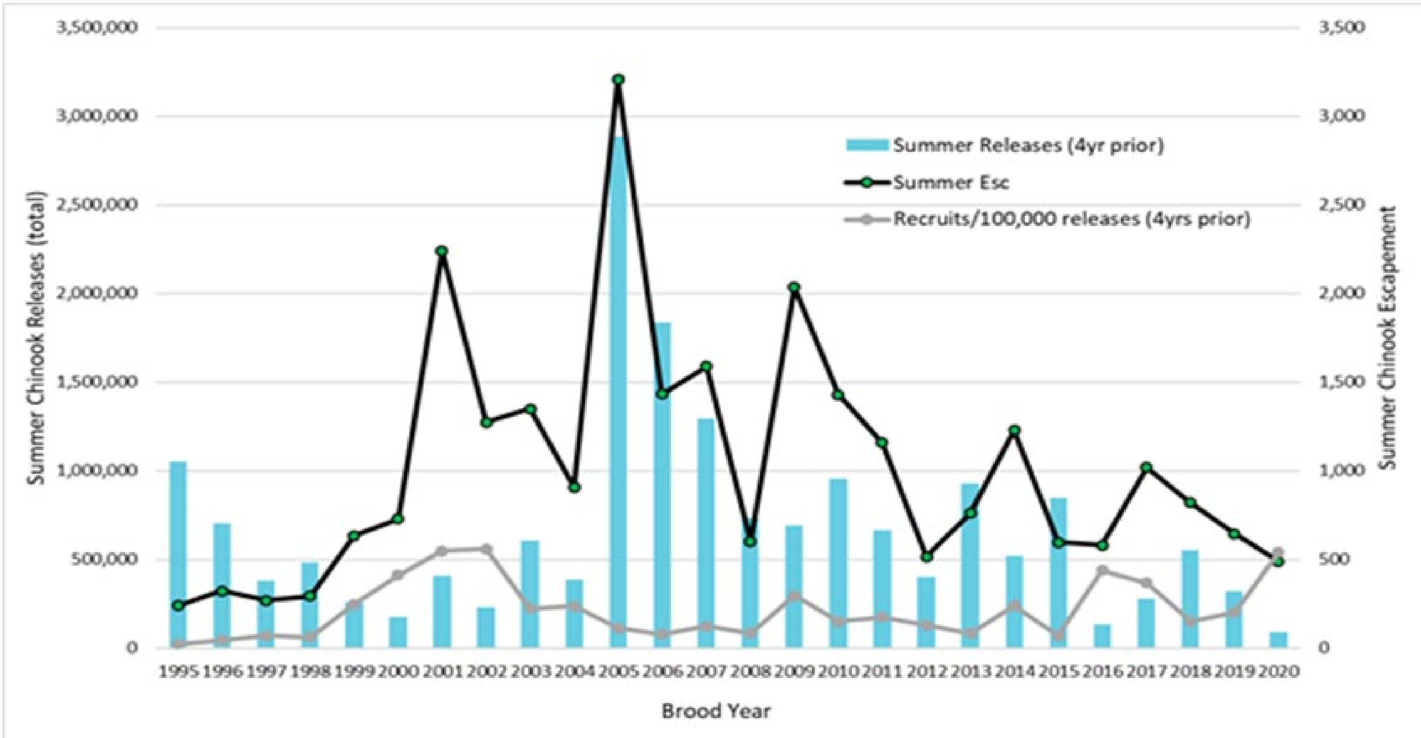


Figure 45. **Ratio of Puntledge to Big Qualicum and Quinsum rivers Chinook salmon escapements over time (Source: Bonnell 2004).**

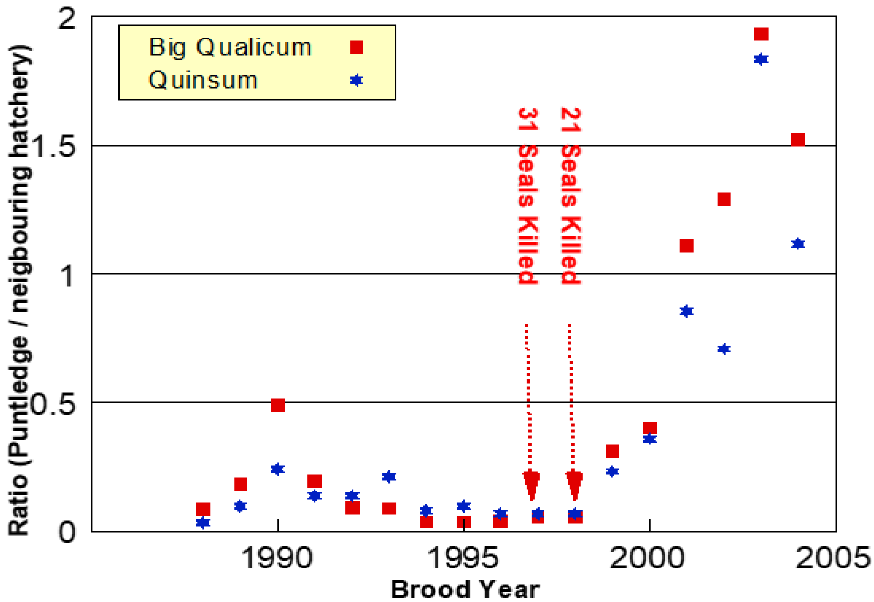
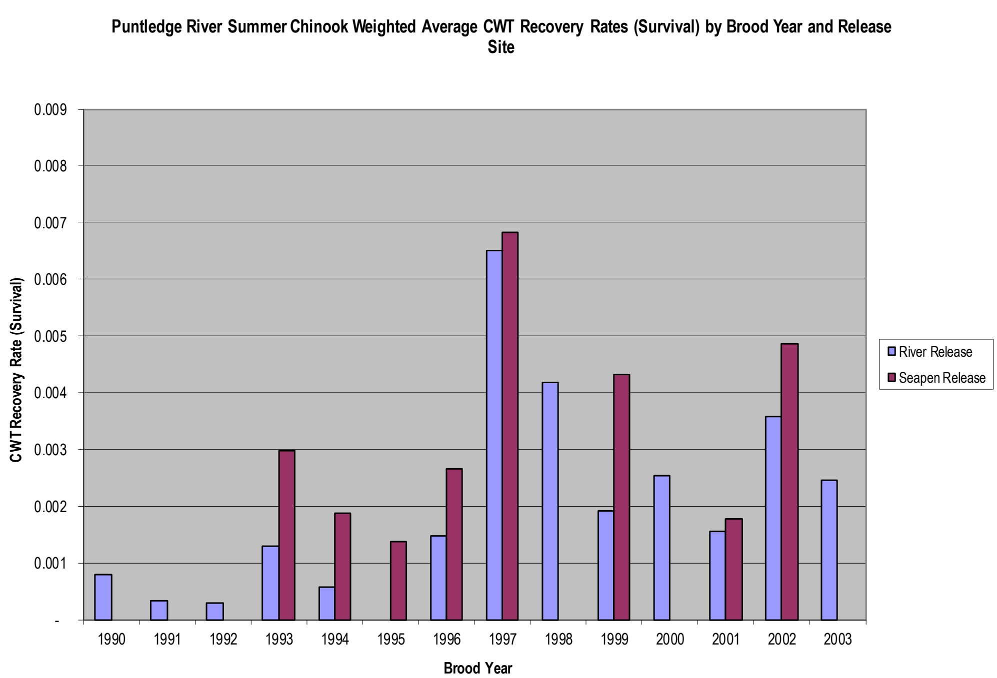


Figure 46. **Summer Chinook survival between river and seapen releases (Source: Bonnell 2004).**



The removal of habituated seals predating on salmonids in the Puntledge River only occurred two consecutive years and it was unclear how long it would take for new individuals to replace the culled ones. The culls involved use of firearms within the vicinity of the city limits and was reassessed and discontinued after two years. The seal population in the river peaked at over 60 seals prior to the culls. Following the cull, the population was significantly reduced for a time, but steadily increased up to 12 individuals per count in 2004 and 2005 (Figure 47) and reaching a count of 20 by 2006 (DFO briefing note 2007). In-river seal counts were not conducted between 2006 and 2015. However, in 2008, the Puntledge Hatchery reported euthanizing up to five seals at the fence location indicating that in-river seal numbers were increasing (Sheng 2007). Although in-river counts were not conducted, the K'omoks Estuary area haulout counts (i.e., Comox Harbour, Sandy Island, Royston, East Cape Lazo Reef), conducted periodically by DFO (Map 8) until 2014, likely provide an indication of the in-river Puntledge River seal population (Trite pers. comm. 2022). There was a peak seal count in the Comox Estuary in 1988, which corresponds to a very low summer Chinook salmon escapement (Figure 48). According to the combined counts in the area, the population numbers also peaked between 1996 and 1998, which corresponds to the official high counts in the river (i.e., over 60, as indicated above).

K'omoks First Nation began in-river seal counts in 2015. Most counts were conducted when summer Chinook adults are absent (i.e., usually in the fall-winter months). Generally, the highest counts recorded (i.e., ~30 to 90 seals) were when pink salmon are returning to spawn in August through September (K'omoks First Nation, unpublished data 2022). Seal counts were also recorded in March and April during downstream migration of salmonid juveniles. K'omoks First Nation staff believe that the same seals remain in the area to target summer Chinook adults. Staff have observed groups of two to three seals chasing down, cornering and killing summer Chinook adults, immediately above the Tsolum River confluence. Harbour seals have now established a haulout area across from the bird viewing platform that is used by more than 40 seals in June and July (Frank pers. comm. 2022). Another 20 seals were also observed in the estuary, around the same period in 2020, and one seal was observed catching a summer Chinook near the Air Park kayak launch site (McCulloch pers. comm. 2022).

Since 2017, K'omoks First Nation staff continue to observe large number of seals (an average of 40 and up to 83) in the estuary and lower river, from Goose Spit to Condensory Bridge and just above the Tsolum and Puntledge River confluence (Frank pers. comm. 2022). Staff believe that the number of seals in the river are the same or more than in 1997 to 1998 when over 60 seals were counted, and 52 seals were culled (Frank pers. comm. 2022). When the adult Chinook first arrived in the Puntledge River in 2023, twenty seals were observed in the river catching Chinooks for two weeks (Frank pers. comm. 2022). However, according to the K'omoks First Nation, seals are first sighted predating on juvenile salmonids in the estuary in the spring, coinciding with wild downstream migration and hatchery juvenile releases (see Section 3.5.4 and 3.5.5). Seal counts of approximately 25 and 35 individuals were observed in March and June, respectively, which may indicate that the seals were feeding on Chinook juveniles (Figure 49).

There is only one in-river seal count between 2014 and 2020 during the months when summer Chinook adults are migrating up the river (i.e., June, July, August; Figure 49). Multiple counts during this period would be required to determine if there is a relationship between seal presence and Chinook salmon predation because counts in-river and at haulout areas vary dramatically throughout the day. The K'omok First Nation are planning to install cameras at key haulout areas to obtain more data. Furthermore, DFO Stock Assessment group will be investigating the use of the Garmin Livecope™System for in-river monitoring of seals.

Figure 47. Seal counts in the Puntledge River between 2000 and 2005 (Brown 2006).

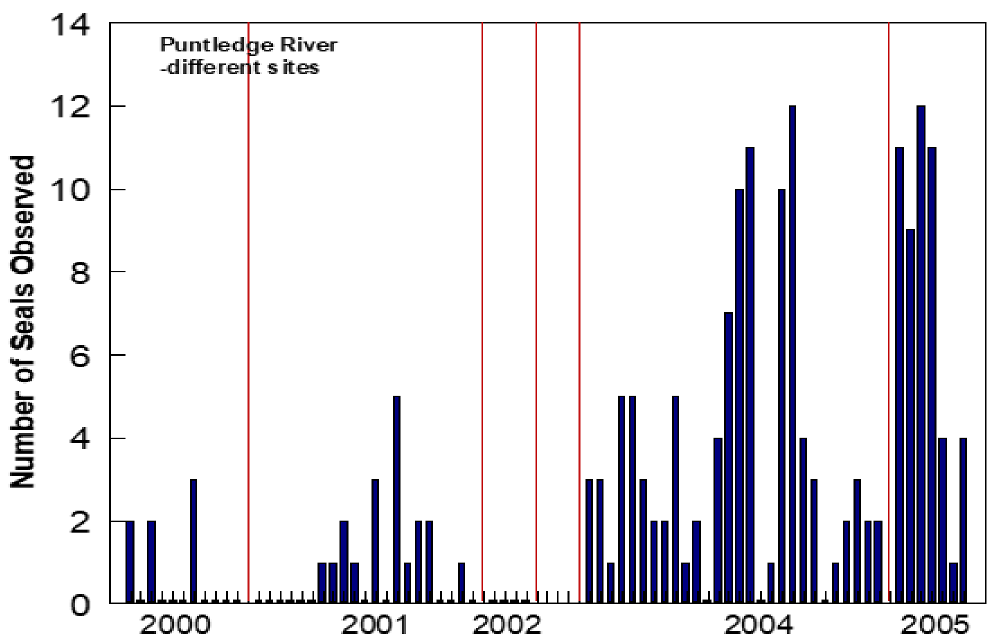


Figure 48. Seal counts in the K'omoks Estuary area (DFO 2018).

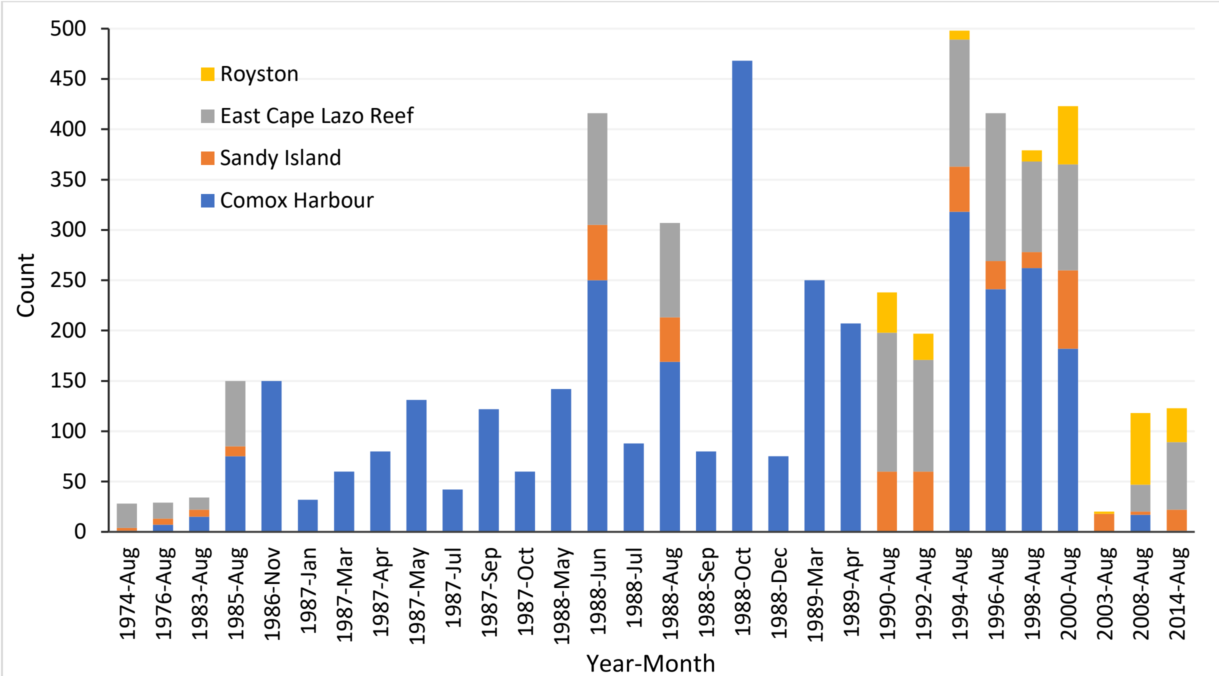
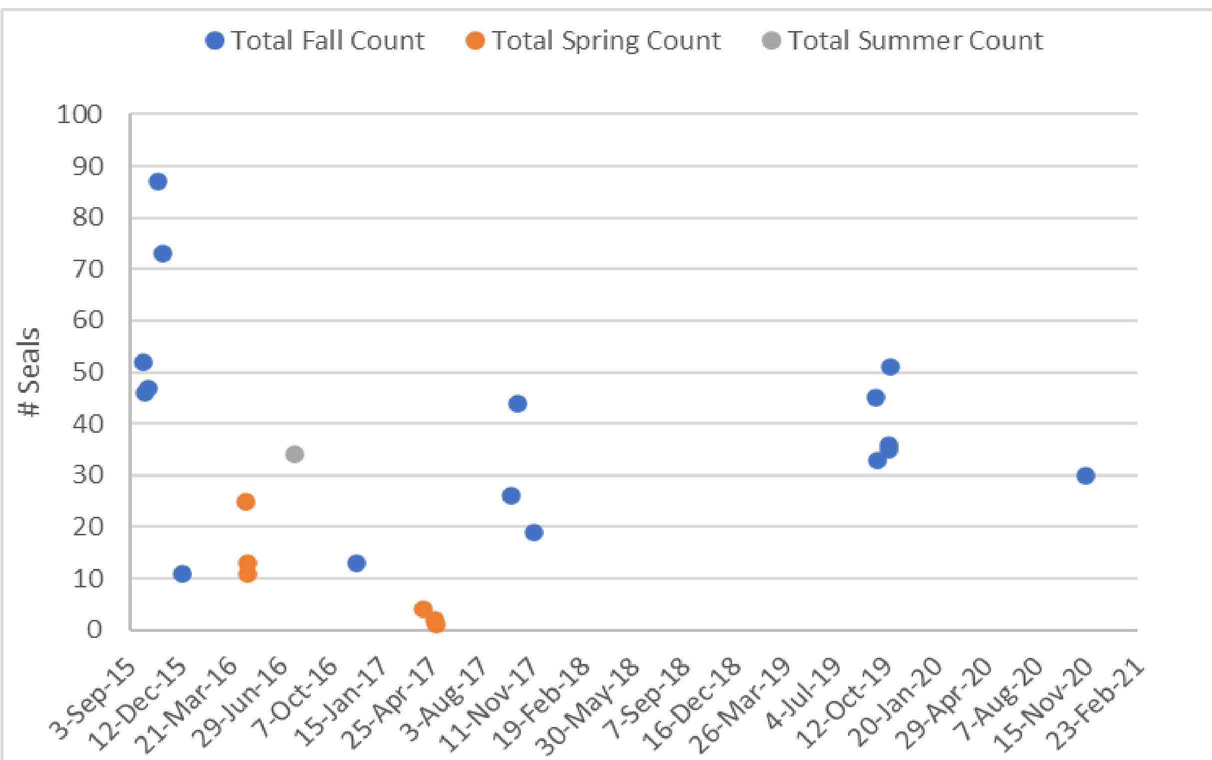


Figure 49. K'omoks First Nation in-river seal counts (Frank pers. comm. 2022).



Map 8. Pacific harbour seal haulout locations within the Comox Harbour area (Source: DFO 2018).

A study conducted in 1998 by Brown *et al.* (2003) continuously observed seal and salmon behaviour (24 hours/day) from mid-June to mid-September close to the seal fence on the Puntledge River, which was installed along the downstream side of the 17th Street bridge. They found that seal numbers were significantly greater ($p < 0.01$) during flood tides and positively correlated to tide height ($p < 0.0001$). The number of seals also varied diurnally with two to three times more seals counted at night than during the day. Similarly, salmon predation rate was also two to three times higher at night compared to daytime. A significantly higher ratio of misses to kills occurred during the day ($p < 0.01$), which suggests that nighttime predation is more successful than daytime pursuits. Overall, seal activity at the fence during the day was lower. Seals may be wary of approaching the fence during the day possibly due to higher human activity.

The distribution of salmon predation further downstream of the seal fence varied with time of day; more kills were recorded further downstream during the day than at night ($p < 0.001$) and primarily on the right side of the river ($p < 0.01$). These differences in diurnal distribution of salmon kills near the fence suggest that salmon may have greater difficulty evading foraging seals at night. However, other areas further downstream of the fence had higher predation rates during the day (e.g., the Old House side of the river between the triad and the seal fence), which appear to contain less cover for salmon to avoid seals (Brown *et al.* 2003).

Chinook salmon counts within the estuary, below the seal fence immediately downstream of the 17th Street Bridge, were greater around mid-day than during morning or evening ($p < 0.01$), negatively correlated with tide height ($p < 0.05$), and positively correlated with flooding tides ($p < 0.001$). Thus, the number of Chinook/hr in the observation area was negatively correlated with the number of seals/hr during the daylight hours ($p < 0.05$). However, it seems feasible that the proportion of salmon counted below the fence reflects the proportion of seals present. Assuming half of the reported probable events resulted in predation, it is estimated that seals killed 144 (38%) of the summer Chinook, 700 (6.5%) pink salmon, and 154 (33%) autumn Chinook (Brown *et al.* 2003).

In recent decades, the sex ratio of summer Chinook salmon has also likely been skewed in favour of males. Figure 50 demonstrates the Chinook salmon sex ratio between 1965 and 1996, which has favored males compared to females at a rate of two males for every female (Trite *et al.* 1996). When the seal numbers were high in the Puntledge River between 1996 and 1998, it was estimated that the sex ratio of male to females was 6:1 (14% female; Beggs pers. comm. 2022) indicating that seals were potentially selecting for females and reducing their numbers. Additional data covering the period between 1992 and 2020 (DFO hatchery and dead pitch data) suggest that the percentage of females in the summer population dropped significantly after 1995 and has remained below 30% (Figure 51). However, this data is generally unreliable because it does not include adults removed for broodstock (which represents >75% of the total return in some years), and the difficulty in identifying sex of salmon when they are 'silvers[4]'. Since 2014, all captured summer Chinook are sampled for DNA so the sex of the adults released above the fish fence or into Comox Lake can be identified. The sex ratio continues to be variable, with the percentage of females ranging from 11% to 42%, and a 2:1 ratio on average between 2014 and 2021 (Figure 52).

Observations in 1996 and 1998 indicate different behaviours between Chinook salmon sexes near the fish fence, which would likely affect their predation rate by seals. Females, which were larger than the males, were often sighted in higher numbers below the fence likely because the smaller males would swim up the fishway into the hatchery. It was suggested that the larger females may have been targeted by seals at a higher rate because they avoided the fast-moving waters of the fishway and congregated at the fish fence where they were easier to for seals to capture (Beggs pers. comm. 2022). Seals likely

[4]Silvers are salmon that have not yet undergone morphological maturation changes.

targeted the larger females, which is a common response by predators (Trites pers. comm. 2022). Female Chinook were also likely dying at a higher rate once they were transferred above the fence. Even though the abundance of females above the fence was lower than males, the hatchery staff often found an equal number of dead males and females during river surveys (Beggs pers. comm. 2022). Beggs suspected that it was possible that the mortality of females could have been caused by delayed impacts from seals (e.g., injury or energy loss due to repeated chases/attacks).

Analysis of scat (scatology) was investigated as an indirectly method to determine sex-biased predation by harbor seals. Results were compromised by extreme differences in DNA density between males and females Chinook consumed and therefore whether the proportions of male and female salmon consumed can be estimated based on genetic analyses. However, despite the bias, it appeared that harbour seal consume a disproportionately higher percent of females. (Balbag 2016)

Figure 50. Ratio of males to female Chinook salmon spawners (the dash line represents a 1:1 ratio) in the Puntledge River (Source: Trites *et al.* 1996).

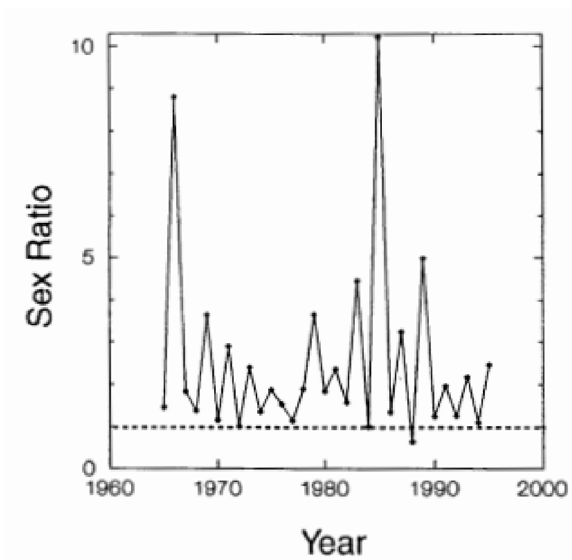


Figure 51. Sex ratio of summer Chinook salmon between 1992 and 2020, from releases and removals not used for broodstock (Thom pers. comm. 2022).

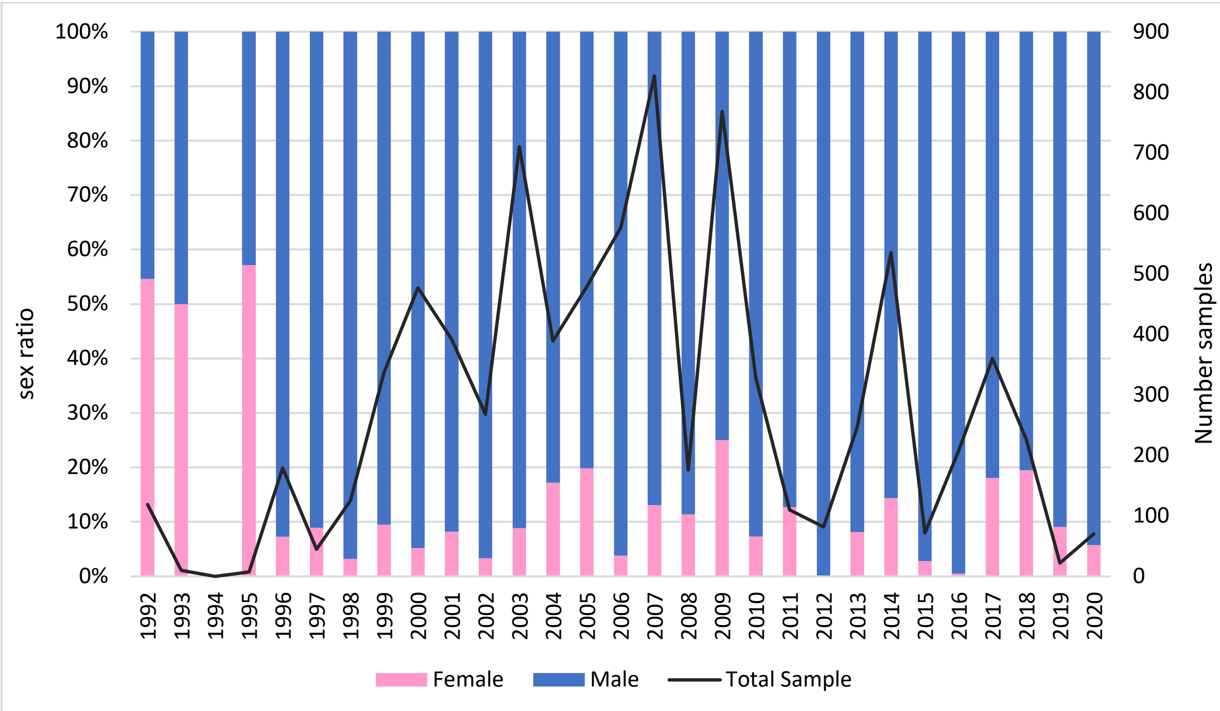
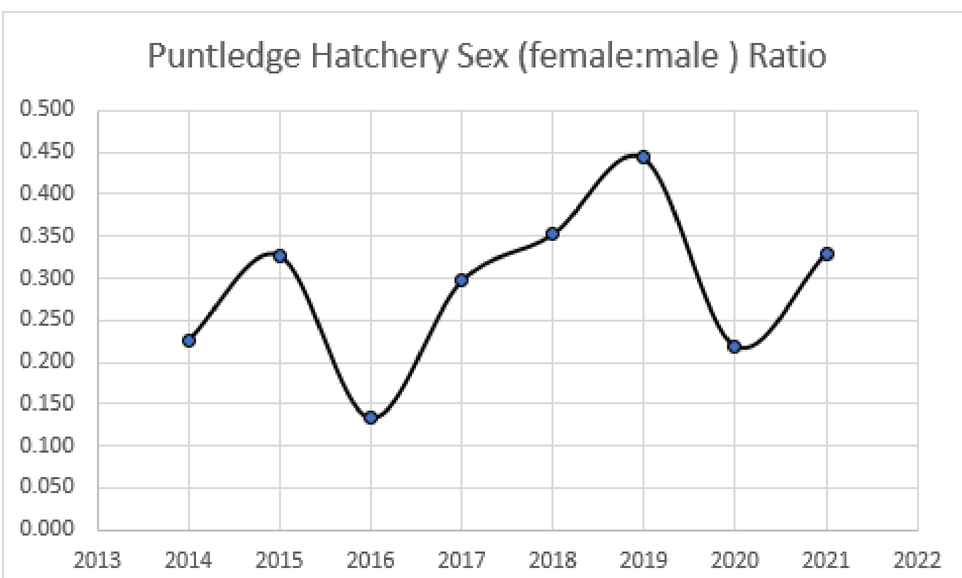


Figure 52. Percent female summer Chinook sex ratio from genetic sampling of broodstock removals at Puntledge Hatchery and natural spawners. (Guimond pers. comm. 2022).



4.1.1.2. Otter Predation

River otters (*Lontra canadensis*) are semi-aquatic, freshwater mammals that use sounds, posturing, and scents to communicate with each other. Males live independently, while females and pups live together in groups. River otters can thrive in a wide range of climates and inhabit a variety of aquatic and coastal environments, including rivers, marshes, lakes, and swamps. Otters construct underwater dens in any location that supplies adequate food resources (NWF 2022).

River otters breed between March and April and give birth between November and May. Females have one to six pups, and average two litters per season. Female otters raise the pups without the assistance of the male. At two months of age, pups can swim. Pups reach sexual maturity in two to three years and will live to an age of eight to nine years (NWF 2022). Adult otters will travel between 16 km and 29 km in search of food with a home range between 4.8 km^2 to 24 km^2 (BD 2020). River otters generally weigh from 5.5 kg to 13.5 kg and can grow three to four feet (0.9 m to 1.2 m) in length. Male otters are usually larger than female otters.

Otters consume a variety of food. Clams, mussels, snails, frogs, and fish make up the bulk their diet, but otters will also feed on birds and vegetation. River otters feed on inter- and sub-tidal fish and consume schooling pelagic fishes when available in the nearshore environment. Foraging success is more efficient and faster swimming fish are more readily captured when otters fish in groups. Otters usually consume food sources under 76 mm in length, especially while in the water, but will consume larger items on shore. When preying on salmonids otter prefer juveniles approximately 80 mm to 150 mm in length, which is smaller than most Chinook smolts. Otters are most efficient preying on salmonids located in off-channel ponds. Several studies of otter diets have confirmed that fish are an important source of otter prey. A 10 kg otter consumes about 1 kg of fish daily (10% of body mass). A study of otter diets in Western Oregon found that fish were the main staple of the diet in the winter, occurring in 80% of all samples. Major fish families represented were Cottidae (31%), Salmonidae (24%), and Cyprinidae (24%). Crustaceans, amphibians, and birds were other important otter food items consumed occurring in 33%, 12%, and 8%, respectively, of all digestive tracts examined (Towell 1974).

Cosby and Gunther (2021) reported that "many fish that otters consumed, including sculpins, gunnels, flounders, toadfish, eelpouts, and sucker fish, tend to be slow moving fish that reside near the bottom of the water column, supporting the finding that otters tend to take slower fish. Other studies have noted the importance of sculpins (Cottidae) to North American river otter diet. Though most sculpin species are marine-oriented, sculpins (*Cottus* sp.) also live in freshwater. The steady level of sculpin consumption over the course of the year indicates a constant, reliable, and accessible, food source for otters". Fish, mostly from the families Gasterosteidae, Cottidae, and Pholidae, were the primary otter prey component, with crustaceans, birds, amphibians, and insects, also important components of river otter diet. Salmonids constituted less than 5% of overall otter diet in the study and consumption of salmon was concentrated during spawning season for otters located near salmon spawning grounds (Cosby and Gunther 2021).

Coastal river otters follow the spawning migration of salmon upriver. North American River Otter (*Lontra canadensis*) predation on salmon is of concern in the Lake Ozette watershed due to potential impacts on ESA listed Lake Ozette Sockeye Salmon (*Oncorhynchus nerka*). Researchers found evidence that otter prey in the area differs by habitat type with significantly greater occurrence of fish and amphibians recovered from scat collected in lake habitat, while a significantly higher occurrence of invertebrate prey was identified in scat from the river habitat (Scordino *et al.* 2016). The study also noted a significantly greater frequency of adult salmon prey remains in scat collected in the river habitat than in the lake habitat. It was hypothesized that a fish counting weir located in the river increased adult salmon vulnerability to river otter predation by slowing fish migration at the location of the weir (Scordino *et al.* 2016). Cutthroat trout are the main prey for river otters in Yellowstone Lake where otters follow the movement of spawning cutthroat trout to tributary streams. The otter population declined in response to a declining cutthroat population in the area (Crait and Merav 2006).

Staff at Puntledge River hatchery annually observe otters within the hatchery and diversion dam fishways and have witness otters predating on Chinook adults enclosed in the hatchery raceways where broodstock are collected. Otters have been observed on video predating on Coho in the Millstone Fishway on Vancouver Island. In the 1990s fisheries officers reported high predation on Sakinaw sockeye within the fishway providing access to Sakinaw Lake. Otters take advantage of locations where salmonid adult migration is slowed, and fish congregate making them easy to capture. Key locations on the Puntledge River are the hatchery fence and fishway, the diversion and Comox dam fishway, the partial obstructions at Stotan and Nib Falls, and the stranding location at the potholes in the Stotan Area. It is unclear how many adult Chinook are taken annually by otters; however, if the otters are responding to the seasonal migration of summer Chinook, these key locations offer good opportunities for otter to capture fish. The main Chinook spawning sites are located in the headpond where large deep runs and pools provide ample area to escape predation and likely are difficult areas for otter to capture summer Chinook.

4.1.2. Non-sanction Fishing Mortality

Interviews with hatchery staff and the last officer posted at the Comox Fisheries Office (F/O) commented that non-sanctioned fishing occurs where summer Chinook congregate at isolated locations in the river, primarily at the BC Hydro tailrace and the Diversion Dam (Gillard pers. comm. 2022). The most active area for summer Chinook poaching was at the Diversion Dam where the fish are easily seen and could be snagged. Most fishing occurred after 5 pm and on average a half dozen people were involved each year. The F/O concluded that based on the difficulty catching fish and the amount of remains found from fish cleaning on the shoreline, yearly catch by non-sanctioned fishing was thought to be low (i.e., less than 20 fish/yr). However, in recent years, hatchery staff observe increased fishing at the hatchery, tailrace and Diversion Dam pools. This may now be a significant factor given the small returns in recent years. A more rigorous program needs to be implemented potentially utilizing security cameras at known non-sanctioned fishing locations to get a more accurate understanding of the level of fishing mortality.

4.1.3. Stress due to Anthropogenic Activity (Non-fishing)

Human presence in the Puntledge River can affect migrating summer Chinook. The Puntledge River attracts thousands of swimmers and tubers every summer. Stotan and Nib falls, and Barbers Hole are some of the most popular swimming areas in the Comox Valley (Figure 53) and tubing the lower river from Puntledge Hatchery to Lewis Park, downstream of the Tsolum River confluence, is a 'rite of passage' for many locals and tourists alike. The Stotan Falls area is characterized by a wide bedrock shelf containing many naturally carved potholes. At low flows, only a thin veneer of water flows over the bedrock, and exposed, dry areas are distributed throughout. Fish ladders were blasted into the bedrock shelf to facilitate fish passage and large pools are located at the base of these ladders providing refuge and resting areas for migrating adults. During the summer when river flows are low, the area is inundated with people sunbathing on the bedrock and swimming in the pools. A few hundred people have been observed in the Stotan Falls area on a hot summer day and Nib Falls also attracts many swimmers. Recent change in ownership of land adjacent to Stotan Falls and access restrictions to this specific area has shifted the concentration of swimmers/sunbathers to Nib Falls, while a ten percent growth in population of the Comox Valley since 2016, and social media exposure, has resulted in a significant increase in recreational use of the Puntledge River overall. One component of the radio telemetry studies conducted in 2003 and 2004 was to determine whether human activity, particularly that associated with swimming, had any influence on the movement of radio-tagged Chinook (Taylor and Guimond 2004, 2005). There was a positive correlation between the frequency of downstream movement and sites with high human activity. However, the manner and degree to which disturbance influences the upstream movement of Chinook is complex and undoubtedly modified by the physical characteristics of the site. Premature mortality (pre-spawn mortality) in salmon has been shown to be up to 8 times higher in females than males during upstream migration, and is highest during challenging conditions such as high water temperatures, turbulent flows, and handling, that increase stress and deplete valuable energy reserves (Hinch *et al.* 2021). Exposure to these stressors can also influence offspring through hormonally-mediated maternal effects. Exposure of eggs to cortisol early in development had persistent effects on juvenile aerobic performance after hatch in Chinook, pink and sockeye salmon (Banet *et al.* 2019). Similarly, offspring from stressed females that were predator-exposed (chased) showed a decrease in burst swimming performance and learning impairment compared to non-stressed mothers (Sopinka *et al.* 2014; Roche *et al.* 2012).

The effect of river tubing/rafting activity in Reach D on summer Chinook migration through the lower river has not been assessed. Higher flows in this reach compared to Reach C may mitigate potential impacts. However, Reach D is dominated by swift laminar runs, riffles, and occasional rapids with only 5% pool area (Griffith 2000). Similar to upstream recreation sites, most pools in this reach become congested with swimmers in the summer. Although the main holding pool for migrating salmon below the barrier fence at the Puntledge Hatchery typically does not attract swimmers, it is the main launching point for tubers. More recent drought conditions however are resulting in earlier reductions in flow in Reach D by BC Hydro to conserve water through the summer and early fall. A study that examined the olfactory perception in migrating salmon found that the chemical l-Serine, a secretory product associated with human sweat, provoked an alarm or avoidance reaction in fish when exposed to a highly diluted concentration (Idler *et al.* 1956). Therefore, it could be reasoned that high concentrations of recreational users at certain key migratory locations could have a significant repellant activity, which may linger after swimmers have left the area, depending on dilution rates. Consequently, the influence of 500+ swimmers at Stotan and Nib Falls during a summer day may affect the ability of Chinook to progress upstream more than from a physical disturbance standpoint.

Not only can the physical presence of humans in the river during migration impact Chinook salmon migration but the detrimental affects of suntan lotion to fish is another anthropogenic cause for concern (see Section 4.3.12.3). These compounds would largely impact summer Chinook adults that can be present in areas where they may interact with swimmers during migration in June, July, and August. Aversion to these compounds could likely delay upstream migration. Regarding the juvenile phase, it is likely that most summer Chinook juveniles have migrated to the ocean by late June before high recreational use. Physical barriers or obstacles built or installed by humans also cause stress to migration adult Chinook. The BC Hydro powerhouse tailrace and fishways at the Diversion and Comox Impoundments Dams impose delays to upstream adult migration. This increases the exposure to in-river seal predation and delays upstream migration to Comox Lake, a critical cold water holding area for the summer months, during the early summer period when river temperatures increase rapidly making upstream migration more stressful and difficult (see Section 4.1.5 and 4.1.6). Likewise, the Hatchery fence also delays migration and can have a similar impact as the BC Hydro tailrace (see Section 4.1.5).

Figure 53. Recreational users at Stotan Falls in 2008 (Source: Hassler *et al.* 2010).

4.1.4. Disease, Parasites or Pathogens

The impacts of diseases, parasites and pathogens on Chinook salmon are detailed in Section 4.3.2 because most of the data are collected from hatchery and early marine rearing fish. Results to date have shown that freshwater pathogens present in Puntledge River fish could indicate that they are also present in the estuary and thus could be present when adult salmon are returning to spawning habitat. Some information on adults is provided in Section 4.3.2 and this aspect requires further studies to assess for adult Puntledge River summer-run Chinook in particular.

4.1.5. Limited or Delayed Access due to Migration Barriers and Lack of Safe Migration Routes

4.1.5.1. Physical Migration Obstacles or Barriers

Various migration obstacles can delay access to spawning habitat for migrating Puntledge summer-run Chinook salmon. The sustainability of the Puntledge summer-run Chinook population is dependent on the ability of these fish to access thermal refuge (i.e., Comox Lake) after migrating upriver in the summer and holding in the lake until they are ready to spawn in the fall. Past migration assessments have indicated that upstream migration in the lower Puntledge River can be delayed at five locations: (1) the powerhouse, (2) Stotan Falls, (3) Nib Falls, (4) the fishway at the Diversion Dam, and (5) the fishway at the Comox Lake Dam (Figure 54). There are also several areas in the lower Puntledge River with difficult passage where shallow water flows over bedrock. Passage at these locations require high amounts of energy and can potentially exhaust fish, delay migration and result in pre-spawning mortality (Brown and Geist 2002). Fish that get delayed are exposed to increasing river temperatures and other stressors that can also lead to physiological stress resulting in further delays and pre-spawn mortality. Climate change forecasts predict higher summer temperatures (IPCC 2023), which will exacerbate physiological stress, increased upstream migration delays and lower success reaching Comox Lake.

Results from a three-year radio telemetry study on the Puntledge River, between 6.4 to 17 km from the estuary, was completed to assess the upstream migration of Puntledge summer Chinook (Table 28; Hasler *et al.* 2011). Further details on the results of this work are provided in Appendix B. This study, which was published in several reports, found that adults (only males were used in the studied) moved at a much slower rate than Chinook in other systems, possibly suggesting that upstream migration is being delayed due to physical or behavioral factors (Hasler *et al.* 2011). For instance, fish can also be stranded and perish in potholes at Stotan and Nibs falls if the pools become isolated. These fish are also highly vulnerable to predators. Hydropower facilities can provide physical migration obstacles as well as flow velocity effects to upstream spawning migration (See Section 4.1.5.2). The effects of hydropower tailrace discharges on the ability of Chinook to detect flow cues in the river has been documented in other watersheds (Hinch and Rand 1998; Brown and Geist 2002; Scruton *et al.* 2007). The potential for the Puntledge powerhouse pool to attract upstream migrating adults and delay migration is discussed in Section 4.1.5.4. Similarly, Stotan and Nib Falls are known to pose a substantial challenge to Chinook migration despite modifications completed in the 1970s to improve passage. A delay or cessation in Chinook migration has also been observed at two other man-made structures. Video footage of Chinook migration through the diversion dam fishway suggests that there may have been Chinook passage difficulties through the fishway when the headpond was closed and the bypass was opened for broodstock collection, compared to when the fishway was opened to allow access directly into the headpond (Figure 54). In 2008, Chinook adults were found to move upstream and downstream through the fishway repeatedly, making several unsuccessful attempts. This resulted in large numbers of Chinook holding or "stacking up" in the pool below the diversion dam. This was a common observation during past broodstock collection operations at the upper hatchery, where as many as 30% to 50% of the Chinook adults that arrived at the diversion dam remained in the Diversion Dam pool (Miller and Munro pers. comm. 2023). When this has occurred, hatchery staff seine adults from the pool to attain broodstock targets.

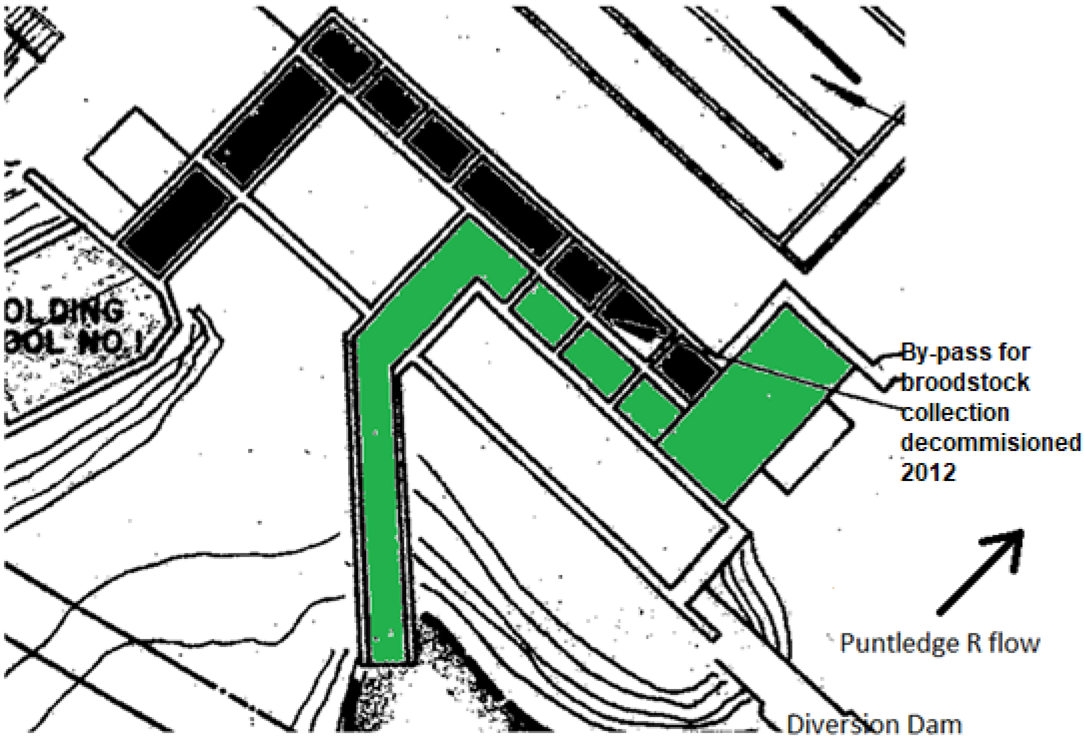
Table 28. Description and location of major holding pools along Reach C and B of the Puntledge River.

Site Name	Distance from Estuary in Km	Description
Lower Hatchery fence	6.4	Large pool below barrier fence - boulders limit fish jumping above fence and force fish to migrate around through the fishway
Powerhouse Pool	6.8	Tailrace pool.
Lower (Powerhouse) Side-channel	7.5	This is a small pool up to 2m deep.
Island Pocket	7.95	Small pool up to 3m deep with little cover. Here fish tend to hold for a short time.
Brown's River confluence	8.4	There are a series of small pools in this area and one slightly larger pool. The water entering the Puntledge from the Browns is very cool by comparison, but low flows in the summer.
Upper (Powerline) Side-channel	8.6	This is a plunge pool that runs perpendicular to the flow of the water. Is has very little cover and a lot of white water.
Lower Stotan	9.3	This pool is 2-3m deep, immediately below the lowest of 3 rock fish ladders. There is not a lot of pool area and much fast water. Swimmers visit this in low numbers.
Mid Stotan	9.35	This is the middle of three fish ladders that make up Stotan falls. Teenagers jump from the rocks into the pool on a regular basis. Average number of swimmers in the water would be 15 at peak. also
Upper Stotan	9.5	A deep pool with overhanging rock cover. Swimmers do use this pool but not as frequently as mid Stotan. Many sunbathers on the rocks around this pool.
HWY / Cut line	10.4	A series of small pools and one deep pool.
Lower to upper Nib Falls	11.5-8	Nib Falls area. 11.5 is a cavernous pool that snorkelers often use. Not heavy with swimmers but steady (moderately easy access).11.6 is a pool were many swimmers and sun bathers accumulate (10-15 people) up to 5 swimmers in the water at a time is not uncommon. 11.7 is accessible to swimmers but not as inviting. it is shallow and offers no cover. 11.8 is simply a series of pools and deeper glides where swimmers visit only sporadically.
Water gauge	12.2 - 12.7	This is a deep slow moving stretch where swimmers visit infrequently. There is a fair amount of cover but little white water.
Barber's Hole	12.9 - 13.0	This deep protected pool is located downstream of a small island, and is used heavily by swimmers. There is some relief closer to the mainstem where no jumping occurs and swimming activity is more relaxed.
Diversion dam	13.3	Pool below diversion dam and entrance to fish way. A very deep pool with a good amount of cover. Very little recreation activity but poaching from this pool is a concern.
Puntledge Forebay	13.2 - 13.3	Deep pool adjacent dam and upstream of penstock intake gates.
Upper Hatchery	13.4 - 13.6	Deep slow moving section between diversion dam and Upper hatchery intake and main pumping station
Eddy pool	14.8	A large deep (30 ft) pool approximately 400 m upstream of a restpored spawning platform. Chinook hold in this pool most frequently.
Comox Dam tailrace	17	Tailrace pool below Comox Dam – turbulence; entrance to fishway is adjacent pool.

In contrast, video monitoring at the diversion dam fishway in 2005-2007 when adults were allowed directly into the headpond (no broodstock collection at the upper hatchery) indicated that Chinook had little difficulty passing into the headpond (i.e., they were not seen dropping down or making repeated attempts). During these three years, over 200 summer Chinook adults accessed the headpond each year with most of these fish migrating before early August. In 2012, the Upper Hatchery and Diversion Dam bypass fishway were decommissioned and just the fishway providing direct access to the headpond remains in operation (Figure 54). A 0.3 m drop at the last fishway cell into the diversion pool is maintained to create hydraulic noise and attract adults to the fishway entrance. When required, broodstock are captured in the first cell. Every year, some adults remain in the diversion pool and fail to proceed further upstream, even when the fishway remains open all summer until the beginning of September. Passage through the fishway into the headpond remains variable. This delay in upstream migration could be attributed to a number of physical, environmental, and behavioral factors including attraction flow, velocities, and depth within the fishway, or the bypass channel into the hatchery raceways, timing of arrival at the diversion pool, water temperature, condition of the fish, and location of juvenile Chinook releases.

Fishways are often constructed at hydroelectric facilities or at other migration barriers to facilitate upstream migration for spawning salmon. The initial impoundment dam at Comox Lake was not equipped with a fishway. At the time of construction in 1912 it was assumed that passage into Comox Lake would be provided through the gates, since they would be continuously wide open (Hunter 1912). However, in 1922 a timber fishway was constructed upon the direction of Fisheries Engineers, who observed that fish passage through the gates was problematic (Ferguson *et al.* 2005). In 1946, the timber fishway was replaced with a concrete structure and replaced again in 1958 when the impoundment dam was rebuilt. However, anecdotal information suggests that the original timber fishway and the succeeding concrete fish passage structure only allowed intermittent access to the lake due to poor design and operation (Anon. 1958; Rimmer *et al.* 1994). Consequently, there was no fish migration between 1912 and 1922 and only sporadic access thereafter. This likely had a significant impact on anadromous fish stocks that utilized the lake and upper tributaries, particularly summer runs of Chinook salmon and steelhead that migrate during low summer flow. In 1991, a new submerged orifice fishway was installed at the impoundment dam by the Ministry of Environment, replacing the pool/weir style fishway. The improvements were likely only successful at providing passage under higher lake levels and flows. When the level of the Comox Lake reservoir dropped below the orifice opening at the upstream inlet of the fishway, flow through the fishway shutdown (KWE and Bixby 2003). Finally in 2002/2003, the Comox Dam fishway was modified by DFO and MOE into a submerged horizontal slot fishway. This design required no operational adjustments as the lake level dropped through the summer and functioned well during summer Chinook adult migration into the lake. Radiotelemetry and PIT tracking indicated that 70-90% or more of the fish that entered the headpond succeeded in migrating through the Comox Dam fishway and into Comox Lake (Guimond 2007a; Guimond and Taylor 2013).

Figure 54. Diversion Dam Fishway Plan (Green – active fishway; Black -decommissioned fishway) (Source: DFO SEP drawings).



4.1.5.2. Flow Related Migration Obstacles or Barriers

Adequate water flows are essential during the upstream migration phase for salmon. Stream discharge is one of several environmental cues that stimulate the upstream migration of Chinook salmon (Keefer *et al.* 2004) and facilitate passage past natural and artificial barriers as well as shallow or hydraulically complex reaches. Downstream of the BC Hydro diversion dam are two major waterfalls, Stotan and Nib Falls, which summer Chinook must ascend to reach their historical spawning and holding areas in the headpond and Comox Lake. It has been suggested that these falls were responsible for the evolution of the summer Chinook stock when early migrants of a fall-run stock were able to negotiate these falls during spring freshets. Prior to hydro development, as the natural spring freshet discharge slowly rose and dropped during the early summer period, the 'summer-run' Chinook salmon were able to negotiate these falls by migrating at a particular flow.

Hydro development on the Puntledge River changed river discharges in Reach C (below the point of diversion) from a more natural flow regime (Table 29, Figure 55) to a constant regulated flow throughout most of the year (BC Hydro 2003). A decrease in both the average flow and in the variability of flow below the diversion dam, as well as an increase in the rate of flow changes during the summer period affected the ability of summer Chinook to migrate though Reach C, and more specifically, to ascend Stotan and Nib falls. The initial diversion of 8.5 m^3/s for power generation was found to cause fish passage difficulties through the falls (Ferguson *et al.* 2005). Access above the falls became even more difficult after hydro expansion in the 1950s.

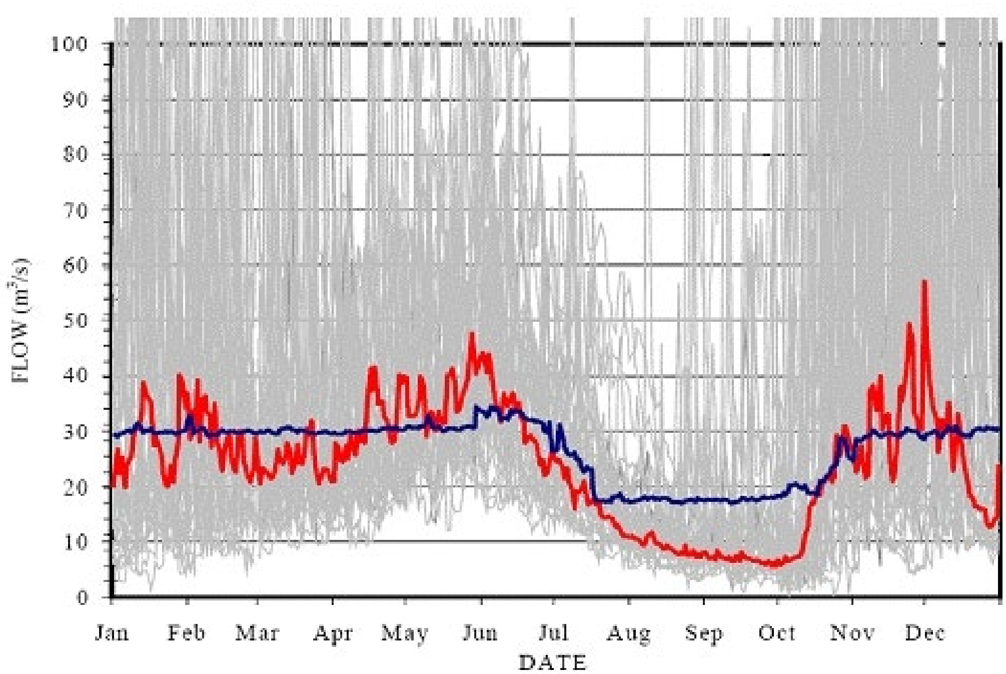
Efforts to improve fish passage at the falls commenced in 1921 when step pools were blasted at Stotan Falls. The fishway works allowed salmonids to ascend the falls in smaller incremental rises of 0.30 m to 0.45 m by swimming and/or jumping up from pool-to-pool. Remedial work on these falls continued sporadically until 1977 (Bengeyfield and McLaren 1994). It has been suggested that this early work may have been to improve access to upstream spawning for Sockeye since it coincided with an intensive transplant program to establish a commercial run of Sockeye salmon in the watershed. The program transplanted eyed Sockeye eggs from Rivers Inlet and Anderson (Henderson) Lake hatchery to the upper watershed, annually from 1923 to 1929, but was never successful (Bengeyfield and McLaren 1994). However, later improvements in the falls between 1968 and 1977 were specifically targeting summer Chinook. It was recognized that passage through these falls was possible only during a limited range of flows. Access was impossible if flows were too low, and similarly if flow were too high. Observations of migrating summer Chinook through this reach between 1954 and 1967 indicated that flows greater than 28 m^3/s (1,000 ft^3/s) could delay migration, result in significantly more injury, and cause serious difficulty in adult passage through the falls, while flows in the range of 14-23 m^3/s (500-800 ft^3/s) provided much easier passage (Lister 1967).

Table 29. **Mean, minimum, and maximum Comox Reservoir daily inflows by month from 1963-1999 (Source: Puntledge River Project Water Use Plan 2004).**

Comox Reservoir daily inflows (1963-1999)

	Mean (cms)	Maximum (cms)	Minimum (cms)
October	33	448	0
November	50	468	4
December	43	483	6
January	38	383	1
February	37	358	4
March	34	356	7
April	33	174	9
May	43	194	12
June	40	135	11
July	24	124	5
August	14	221	2
September	13	230	1

Figure 55. **Daily natural historical inflows into Comox Lake (grey lines), median inflow on each calendar day over the period of record (red line), and median discharge from Comox Dam (blue line) from 1965 - 2001 (Source: BC Hydro 2003).**



Changes in the hydrology of the Puntledge River caused migration problems for summer Chinook migration at other locations as well. In 1955, during the first year of operation of the expanded hydro facility, adult summer-run Chinook salmon were delayed at the tailrace pool of the powerhouse, a phenomenon not previously recorded during the four decades of operation of the smaller Canadian Collieries facility (Hourston 1962). Higher flows through the penstock (28.3 m^3/s versus 8.5 m^3/s prior to expansion) combined with slightly cooler temperatures from the powerhouse compared to the lower flows in the mainstem "diversion" reach (Reach C) inadvertently attracted adult salmon to the tailrace pool and tailrace outlet. This resulted in serious injury and exhaustion, to the extent that many fish died of injuries or became more susceptible to poaching and predation. In 1955, the BC Power Commission attempted to lure fish away from the tailrace pool by releasing a series of artificial freshets down the river (Hourston 1962). The measures were only marginally successful, and from 1963 to 1968, a total shutdown of the power plant during June and July and partially in August was ordered by DFO to reduce injuries at the tailrace.

Since the 1960s, BC Hydro has negotiated fisheries flow agreements with DFO to provide optimum benefits for power generation without negatively impacting fisheries (Hirst 1991). In 1969, a discharge of 8.6-14.3 m^3/s (300-500 ft^3/s) in the river between the diversion dam and the powerhouse was provided to prevent the delay and injury of adults in the tailrace pool and facilitate passage at Stotan and Nib falls. Results from monitoring the effectiveness of these flows led to the formal adoption of minimum flows in the 8.6 m^3/s to 14.3 m^3/s range through Reach C during the adult migration period to facilitate fish passage (Marshall 1973). By 1998, a minimum provisional fisheries flow below the diversion dam was set at 5.7 m^3/s (200 ft^3/s) from June 10th to September 30th to facilitate passage through the falls for summer-run Chinook salmon, and 2.8 m^3/s (100 ft^3/s) from October 1st to June 9th for in-river habitat maintenance. However, since that year, minimum flow requirements have been maintained at 5.7 m^3/s (200 ft^3/s) below the diversion dam and 15.6 m^3/s (550 ft^3/s) below the powerhouse year-round under an Interim Water Order through the Comptroller of Water Rights (Wightman *et al.* 1998).

On 19 January 2005, BC Hydro began implementing the conditions of the Puntledge River Water Use Plan (PUN WUP) (BC Hydro 2003). As per the PUN WUP operating alternative, minimum discharge for Reach C is established at 5.7 m^3/s (daily average) for rearing and spawning. A total of 17 pulse flows are provided yearly to facilitate fish migration and two additional pulse flows between May 15th and June 15th to provide opportunity for a planned kayaking event. One of the key recommendations in the PUN WUP was the release of five pulse flows in Reach C during the months of July and August to improve summer Chinook and steelhead migration. The recommendations were based on results from a radio telemetry study conducted during the PUN WUP process in 2002 to address the limited information on the flows needed for summer steelhead and summer Chinook to stimulate migration past the BC Hydro Powerhouse Tailrace and ascend the barriers in the Puntledge River during their upstream migration. Results from this initial study showed a positive migratory response of radio-tagged summer Chinook to experimental pulse flows in Reach C (KWE and Bigsby 2003). The five pulse flows typically occur weekly: four in July and one during August. Each pulse flow last 48 hours, inclusive of ramping. Each pulse includes a ramp up period, a period at 12 m^3/s, and a ramp down period, and had to be greater than the powerhouse discharge. Minimum discharge in Reach D below the powerhouse is established at 15.6 m^3/s year-round, with additional conditions in place between September 21 and December 31. The preferred operating conditions for the Puntledge River hydroelectric facility relating to instream flows are summarized in Table 30.

Table 30. Recommended operating conditions for the Puntledge River Hydroelectric Facility (from the PUN WUP BC Hydro 2003).

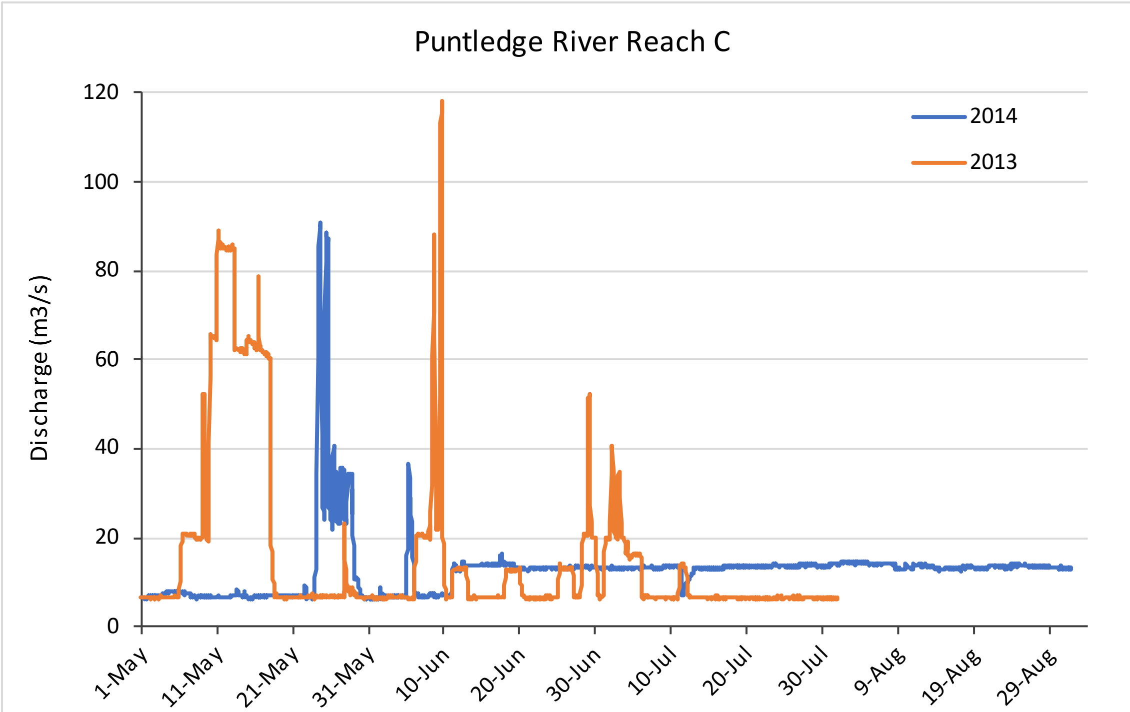
River Reach	Condition	Time of Year	Purpose
Puntledge River Reach C	Minimum 5.7 m^3/s	Year-round	Provide opportunity for fish migration, spawning and rearing.
	17 - 12 m^3/s pulse flows	January – October	Provide opportunity for a kayaking event
	2 – 85 m^3/s pulse flows	15 May - 15 June	
Puntledge River Reach D	15.6 m^3/s	Year-round	Fish Habitat
	Increases in discretionary flows at Gauge 8 up to 20.7 m^3/s must be maintained for the remainder of the period. When flows at Gauge 8 are greater than 20.7 m^3/s, discretionary flow increases need not be maintained.	21 Sept – 31 Dec	
Puntledge Diversion Dam Maximum Ramp Rates	When Gauge 6 flow is 5.7 m^3/s to 19.8 m^3/s, the maximum rate of increase or decrease of discharge is 2.8 m^3/s per hour. When Gauge 6 flow is 19.8 m^3/s and above, there is no maximum rate of increase or decrease of discharge	Year-round	Prevent fish stranding

With respect to river discharge and stage height, the rate of migration through the Comox Dam fishway was similar in both years. The average was 36 minutes and 37 seconds in 2013 and 33 minutes and 25 seconds in 2014. The flow rate through this fishway is a function of the lake level and the water level at the fishway outlet, which was likely similar in both years. In contrast, the migration rate through the Diversion Dam was 16 minutes and 26 seconds in 2013 versus 44 minutes and 52 seconds in 2014. The Diversion Dam Fishway has only five pool weir cells, which are designed to operate with 0.3 m (1 ft) of head. The higher discharge in 2014 may have increased the head drop between the cells and increased the difficulty of upstream migration; however, this requires further investigation.

Despite the difference in river discharge between the two study years (Figure 56), the migration times in 2013 and 2014 were similar. In 2014, the average time from release at the lower Puntledge hatchery to detection at the Diversion Dam array was 18 days, plus an additional 8 days, on average, for fish to reach the upper antenna at Comox Dam. The travel time from the same release location to the diversion dam in 2013 was 21 days, 3 days less than 2014 but not significantly different ($t = 1.81$ $p = 0.359$). Although the range of 2013 data (13- 35 days, variance 61.6) was less than that in 2014 (5-41 days, variance 57.5), the variance was similar. These results are similar to previous studies. In 2003, the mean time was 15.7 days, ranging between 6 days and 25 days to migrate from the lower hatchery to the Diversion Dam (i.e., a distance of 6.3 km); and a mean time of 16.7 days (estimated from graphs) ranging between 9 days and 25 days in 2005 (Taylor and Guimond 2003, 2006). Overall, it took 26 days to migrate from the Lower Hatchery to Comox Dam in both 2013 and 2014, which included a period of residence in the headpond ($t = 1.78$ $p = 0.807$) (i.e., a distance of 10.4 km).

Discharge did not appear to change the rate at which adults migrated through the Diversion and Impoundment dam fishways. Analyses of migration time versus fish size have not been completed. The influence of river temperature, discharge, stranding, predators, delayed migration due to the BC Hydro tailrace, human interaction of recreational swimmers at Stotan and Nib falls and the initial condition of the adults when first entering the river are all suspected to impact migration success rate. Size-related factors likely include river discharge as well as potential size selection attributes of Stotan and Nib Falls, though this is less likely at the fishways. A repeat of a study like the homing study with the addition of a PIT tag array up and downstream of the BC Hydro tailrace, Stotan, Nib, and the fishways would be informative.

Figure 56. **Mean hourly discharge for Gauge 6 below the diversion dam (WSC Gauge No. 08HB084) period in 2013 and 2014.**



The powerhouse tailrace pool is one of only a few large, deep holding pools available for Chinook refuge during their migration to Comox Lake. Fish holding in this pool are typically exposed to the minimum flows through Reach C (5.7 m^3/s) and the discharge from the penstock (28.3 m^3/s during maximum power generation), throughout most of the summer Chinook migration period . Historically, Chinook have tended to hold in this pool for prolonged periods, perhaps as a function of the warmer temperatures and low flows in Reach C (diversion reach), in comparison with that in the penstock. However, the temperature is not statistically different (Sweeten, 2005). In the past, Puntledge Hatchery personnel observed adult Chinook gathering at the powerhouse pool in large numbers where they were at risk of predation and poaching. This phenomenon appears to be variable and less common in recent times, possibly due to the increase in minimum flows in the diversion reach from 2.8 m^3/s to 5.7 m^3/s to facilitate migration, and changes to the barrier fence at the hatchery in the lower river that restricts seal access upstream. However, any delay during the upstream migration phase of summer Chinook increases their risk of being exposed to increasing river temperatures that can occur rapidly in some years. Temperatures can reach 20°C by the end of June in some years.

Several factors are believed to stimulate salmonid migration including stream discharge, water temperature, turbidity, light intensity, barometric pressure, and olfactory cues. It is surmised that a combination of these environmental factors may interact with one another, producing optimal conditions to elicit a migratory response. The role of flow conditions in this combination remains unclear. The summer pulse flows as part of the PUN WUP were implemented for two reasons - to stimulate summer Chinook holding in the powerhouse pool to move upstream, and to facilitate their migration through Reach C, particularly past Stotan and Nib Falls. During the pulse flow release, fish holding in the powerhouse pool would be exposed to a greater flow from Reach C (flow increase from 5.7 m^3/s to 12 m^3/s) compared to the discharge from the tailrace (decrease from 28.3 m^3/s to 10 m^3/s) over the 48-hour period.

Delays can also occur at the BC Hydro Powerhouse tailrace, located 435 m upstream of the hatchery fence. The tailrace discharges are normally two to three times higher than the river discharge in the diversion reach during adult migration. Chinook have been observed congregating in the tailrace pool and may be delayed for a month at this location (Beggs pers. comm. 2022; Miller pers. comm. 2022). Delayed migration at this location, or other choke points along the migration corridor, can increase the exposure of adults to increasing river temperatures. During the period summer Chinook first arrive at the tailrace, and 2-4 weeks later, water temperature commonly increases from 12°C to 18°C. Temperatures above 18°C are known to increase stress for Chinook, whereas reduced migration success and cessation of migration may occur when temperatures exceed 21°C (Cresswell 2004). High temperatures also increase pre-spawning mortality and decrease gamete viability (Jensen *et. al.* 2006). Weekly pulse flows between mid-June to the end of July are provided by BC Hydro to stimulate summer Chinook to migrate upstream and past the BC Hydro tailrace. Minimum flows in Reach C of the river are increased to 12 m^3/s for 48 hours (including ramping up and down), and minimum flow must be greater than the powerhouse flow during this period (BC Hydro 2004).

4.1.5.3. Migration Success

Radio telemetry studies were conducted to assess summer Chinook migration in Reach C and Reach B in the Puntledge River (between 2003, 2005, 2007, and 2009). The latter studies (2007 and 2009) were part of the PUN WUP Monitoring Program, which was implemented following approval of the PUN WUP Consultative Committee report by the Comptroller of Water Rights. The monitoring programs were designed to assess how well the preferred operating alternative achieved the desired fundamental objectives, in this case, whether the pulse flows during July and August stimulated and facilitated summer Chinook and summer steelhead migration in Reach C. Due to the low escapements of summer run steelhead, the migratory response observed for Chinook salmon was used as a proxy for steelhead, based on the assumption that flow conditions that benefit Chinook migration would also benefit steelhead migration.

The degree to which the powerhouse tailrace pool delays summer Chinook migration in Reach C remains unclear. However, the hatchery crew still consistently observes Chinook holding at this location. Based on results from all radio telemetry studies conducted between 2002 and 2009 to monitor the effectiveness of pulse flows, radio tagged Chinook were found to be delayed at the powerhouse pool at varying rates, or not at all. Snorkel surveys conducted in Reach C during July and August of 2007-2009 counted a maximum of 23 Chinook holding in this pool prior to a pulse flow, and no Chinook were counted three days later. Very few (<6) Chinook were counted in the pool during other swims. The efficacy of pulse flows to attract fish from the powerhouse pool during all study years was inconsistent based on the radio telemetry studies.

A series of radio telemetry studies on summer Chinook salmon migration in Reach C between 2003 and 2009 have demonstrated that, despite minimum flows of 5.7 m^3/s, and summer pulse flow releases implemented through the WUP, key points in the river continue to pose a substantial challenge to Chinook migration. Radiotelemetry and on-site observations demonstrated that adults still delay at the Powerhouse, Stotan and Nib falls. The continuous 2009 radio tracking of the gastrically- and electromyogram (EMG) -tagged fish (excluding fish that fell back from the release site) revealed similar migratory behaviours as previous reports (Taylor and Guimond 2006). The first key delay occurs at the Powerhouse Pool. Thirty-three of the 85 fish tracked delayed or failed to ascend any further (Hasler and Cooke 2010). Failure to pass Powerhouse Pool has ranged between 7.9% to 22.2% in previous radio-telemetry studies. Most fish holding at Stotan and Nib falls for considerable time periods (up to 47 days). Fish tended to ascend Nib Falls much faster than Stotan Falls. Mean time to ascend Stotan Falls was 8 days to 10 days while mean time to ascend Nib Falls was 3 days. With the exception of EMG-tagged fish in 2009, EMG migratory behaviours did not noticeably standout when compared to gastrically-tagged fish. In addition, fish tagged in 2009 took longer to migrate from the Powerhouse Pool to Stotan Falls but did not have significant differences for any other measured migratory behaviours.

At Stotan, some causes for delayed migration included difficulties finding the fishway entrance (Fleener pers. comm. 2005, 2006) and straying off the main fishway route and into other arteries of river flow leading to dead end shallows or "potholes", where adults became stranded and vulnerable to predation. Rock work was completed in 2007 to rectify some of these problems (Guimond 2007b). In the five-year period previous to the improvements, the migration success rate at Stotan Falls ranged between 64-88%. In the subsequent two years after the improvements, the success rate was 94% in 2008 and 74% in 2009 (Guimond and Taylor 2010).

Based on the results of six years of telemetry studies, on average 19% of Chinook that reached Stotan Falls and 18% at Nib Falls failed to migrate further (Taylor and Guimond 2010). In some years, this number was as high as 30% of tagged fish failing to pass these barriers (Table 31). Radio telemetry (2003-2009) and PIT tagging (2013-2015) studies on summer Chinook migration from the hatchery to Stotan and Nib Falls, the Diversion Dam and Comox Dam found that the migration success rate into Comox Lake, the traditional holding area for summer Chinook, ranged between 50% and 70% (Guimond and Taylor 2008, 2009, 2010; Guimond *et al.* 2016). The migration success rate above these locations is variable, and dependent on river discharge, water temperature, human disturbance levels, and physiological condition of the fish.

Table 31. Attrition rates at Stotan and Nib Falls including losses expressed as a proportion of total tag releases.

Year	Viable Tags	Stotan Falls			Nib Falls			Failure as a % of Total Releases	
		# Reached	# Passed	% Failure	# Reached	# Passed	% Failure	Stotan	Nib
2003	31	28	24	14.3	24	20	16.7	12.9	12.9
2004 (1)	17	17	15	11.8	14	10	28.6	11.8	23.5
2004 (2)	17	17	14	17.6	14	13	7.1	17.6	5.9
2004 combined	34	34	29	14.7	28	23	17.9	14.7	14.7
2005 (1)	23	21	17	19.0	17	15	11.8	17.4	8.7
2005 (2)	23	22	14	36.4	14	11	21.4	34.8	13.0
2005 combined	46	43	31	27.9	31	26	16.1	26.1	10.9
2007	19	16	14	12.5	14	11	21.4	10.5	15.8
2008	26	17	16	5.9	15	11	26.7	3.8	15.4
2009	29	27	20	25.9	20	17	15.0	24.1	10.3
Average losses				**19.2**			**18.3**	**17.4**	**13.1**

After several years of intensive data collection on Chinook migration in Reach C, the reasons that the Stotan and Nib falls areas affect migrants differently is inconclusive. Migration success is highly variable at both sites, and each can pose the greater obstacle in a given year. Evidence from past studies indicates that neither hydrological conditions, nor physiological variables, are the primary factor dictating migration success (Guimond and Taylor 2010). The observable effects of the parameters that drive the variability in migration past the two falls are likely obscured by subtle interactions among several contributing variables that influence behaviour in individual Chinook. Overall, the data suggests that 25% or less of all tagged fish each year ascended Stotan Falls and Nib Falls during pulse flows. Based on an analysis of flow and fish passage at Stotan and Nib Falls, passage rates at these restrictive points were higher during peak pulse and transitory flows, relative to base flows (Hasler and Cooke 2010).

Interestingly, early migrating Chinook (i.e., adults that arrive in the river between early May and late June) have a greater success of migrating through Reach C and negotiating the Stotan and Nib falls areas compared to later migrants (i.e., adults arriving in the month of July). These early arriving fish can bypass the barrier fence in the river at the lower hatchery during base flows (i.e., tailrace plus minimum Reach C flow (~35 m^3/s) and high river flows (flows likely exceeding 75-80 m3/s at Gauge 8). Snorkel survey counts and video surveillance monitoring at the lower and upper hatchery fishways indicate that over 90% of summer Chinook that arrive in the lower river prior to the end of the freshet flow period, or the kayak pulse flow release scheduled around the end of May, successfully reached the diversion dam pool or had passed through the diversion dam fishway by the end of July compared to only a 50% success rate for later arriving radio tagged Chinook. A combination of the higher flows in May/June, lower water temperatures, and absence of swimmers in the river likely contributed to the success of these early arriving Chinook in reaching the upper river.

The purchase of the Mosaic property adjacent to Stotan Falls to two subsequent developers (i.e., 3M and Valiant Financial) resulted in a reduction in access to Stotan Falls area for recreational use. There was restricted access between 2016 and 2017 when the public had to pay to access the falls; open access in 2018 and now complete restricted access since 2019 (Nestor pers. comm. 2023). This likely has improved the ability of summer Chinook to migrate through the falls more expediently. A follow-up telemetry study would be necessary to determine if this has occurred.

4.1.5.4. Adult Upstream Migration Success Rate versus Size

Data to determine if adult Chinook fish size are correlated to successful upstream migration in the Puntledge River was inconclusive. In 2013-2014, a homing study was conducted that focused on determining if hatchery smolt releases from the lower hatchery had a lower propensity to migrate to Comox Lake compared to smolt releases in the Lake (Guimond and Taylor 2013, 2014). Adult return data from this study was re-examined to determine if adult size impacted the migration success rate into the lake. A total of 133 adults ranging in fork lengths between 350 mm to 990 mm were captured at the lower hatchery fence, PIT tagged, and tracked through the Diversion and Comox Dam fishways. The pooled results of adults from both return years indicate that although the mean size of the adults that failed to swim through Comox dam was higher, there was no statistical difference (t-test: $p = 0.121$; Figure 57). Overall, of the 133 adults in the study, only 49% successfully migrated into the Lake (Figure 58).

If the sizes of the adults in the study are partitioned into three groups (i.e., small, medium, and large), the success rate migrating through Comox Dam in 2013 was approximately 40% for the small and medium size fish and 75% for the largest size adults. However, the sample size was small, and no differences existed between size groups (Table 29). In 2014, a larger number of adults were tagged and tracked (Table 33). The smallest size group attained a 72% success rate, the medium size group achieved a 50% success rate and the largest size group only a 38% success rate, opposite to 2013 (Figure 59).

Figure 57. The mean size of PIT Tagged summer Chinook adults that failed or successful migrated from the lower hatchery and through Comox Dam.

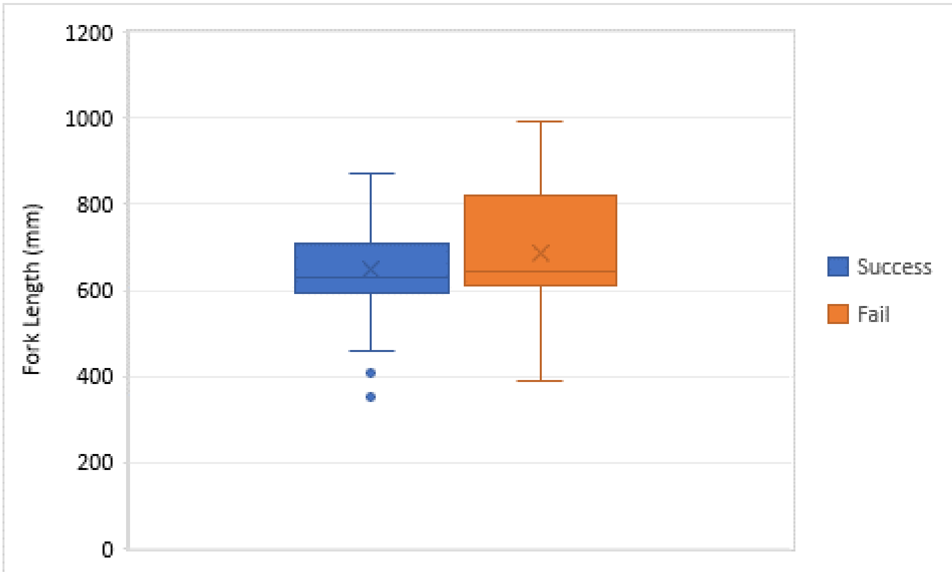


Figure 58. The migration success of 113 individual summer Chinook adults ranging in fork length between 350 mm and 990 mm and PIT tagged and released in 2013 and 2014 from the lower Hatchery.

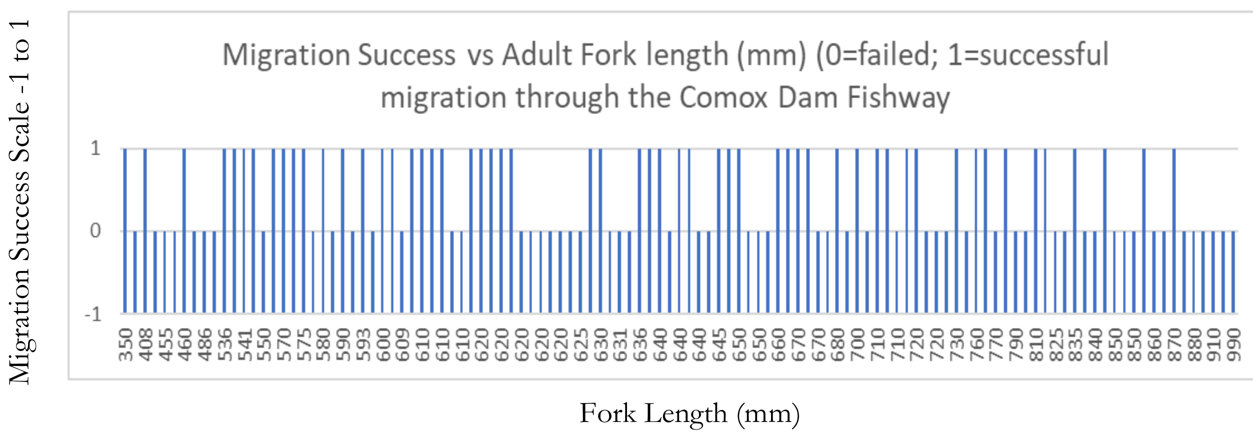


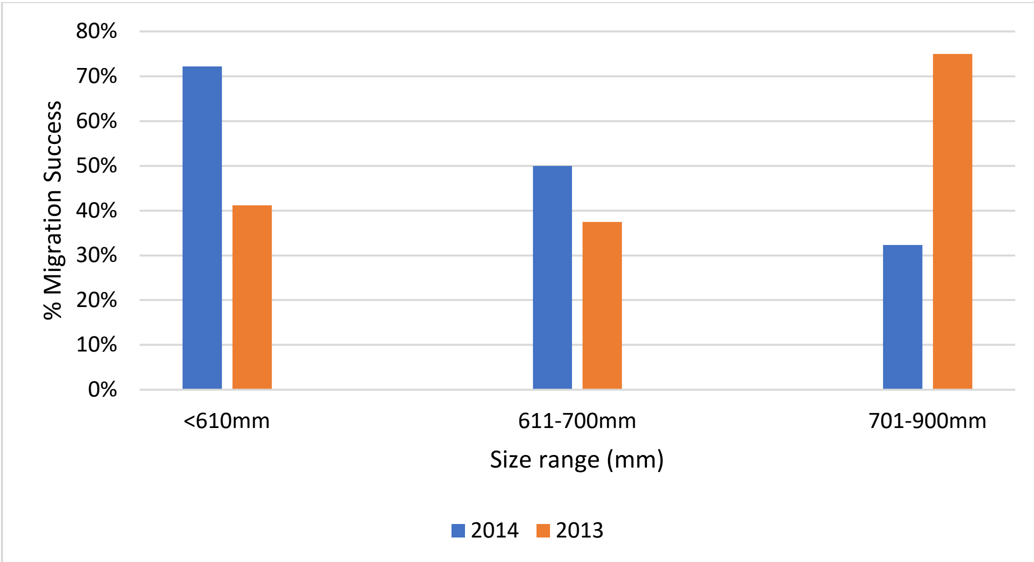
Table 32. **The migration success rate of PIT tagged adults in three size categories migrating from the lower hatchery to Comox Lake in 2013 (N=29).**

Fish Sizes	**Fork Length Range (mm)**	**Sample Size (N)**		**Migration Success Rate**		**Total**
		Failed	**Successful**	**Failed**	**Successful**	
Small	350 - 610	10	7	0.59	0.41	17
Medium	611 - 700	5	3	0.62	0.38	8
Large	701 - 990	1	3	0.25	0.75	4
	Total	**16**	**13**	**0.55**	**0.45**	**29**

Table 33. **The migration success rate of PIT tagged adults in three size categories migrating from the lower hatchery to Comox Lake in 2014 (N=84).**

Fish Sizes	**Fork Length Range (mm)**	**Sample Size (N)**		**Migration Success Rate**		**Total**
		Failed	**Successful**	**Failed**	**Successful**	
Small	400 - 610	5	13	0.28	0.72	18
Medium	610 - 700	16	16	0.50	0.50	32
Large	700 - 990	23	11	0.68	0.32	34
	Total	**44**	**40**	**0.52**	**0.48**	**84**

Figure 59. Fork length and migration success from the lower hatchery through the Comox Fishway to Comox Lake in 2013 (5.7 m^3/s plus WUP pulses) and 2014 (13.5 m^3/s).



In 2013, the river discharge remained at the prescribed summer flow level of 5.7 m^3/s in Reach C and as per the Puntledge Water Use Plan, BC Hydro conducted five weekly 48 hour pulse flow releases for summer Chinook migration between June 11th and July 11, 2013 (Figure 56). However, the discharge in Reach C remained higher than normal for the entire summer Chinook migration period in 2014 (i.e., 13.5 m^3/s versus 5.7 m^3/s) and was slightly above the prescribed summer migration pulse flow (12 m^3/s) as per the Puntledge Water Use Plan. Based on the differences between 2013 and 2014, higher discharges appear to impact the migration success of larger fish.

If migration success is compared between male and female adults in both years, females which were larger on average compared the males in both brood years (Figure 60), had a higher success rate than males in 2013 and poorer success than males in 2014 (Figure 61). Again, this suggests that larger fish had poorer success when higher flows were released through the summer migration period. The WUP flows appear to provide better conditions for adult migration. The sample size in this study are too small to run statistical analyses (i.e., 69 males and 14 females in 2013; and 21 males and 8 females in 2014). Overall, this study strongly suggests that the WUP flow with the five summer pulses provide better upstream migration access for larger fish.

Figure 60. Comparison of male and female fork length (2013 and 2014).

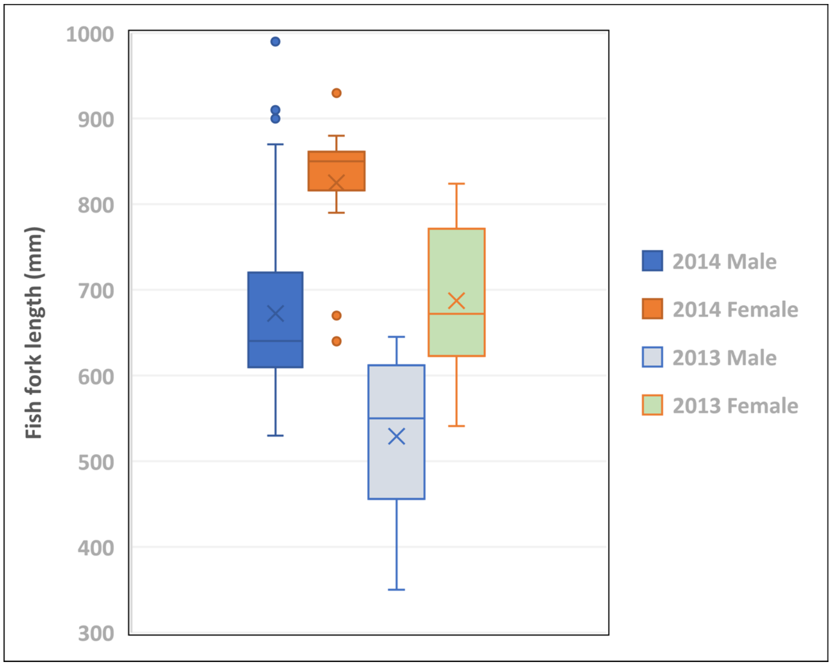


Figure 61. Comparison of male and female upstream migration success rate in 2013 and 2014.

	Brood Year Return	
	2013	**2014**
Female	63%	29%
Male Success	38%	58%

4.1.6. Pre-spawn Mortality

There are a variety of ways that pre-spawn mortality may occur for Puntledge summer-run Chinook salmon as a result of other threats to the population. These causes of pre-spawn mortality are detailed in other sections. For instance, delayed migration or limitation to spawning ground access can result in pre-sawn mortality as indicated in the sections above. Water quality issues can also result in adult fish mortality prior to spawning. For example, increases in summer water temperature can result in the death of fish (see Section 4.1.16). Additionally, deleterious substances found in the Puntledge River during spawning migration timing may result in pre-spawn mortality (see Section 4.1.17.4).

4.1.7. Escapement – Defensible Consistent Enumeration Technique

Although not specifically a threat to summer Chinook in the Puntledge River, an adequate escapement count is critical to monitoring the population. Summer Chinook migration begins in May or earlier, but the majority arrive between June and July and attempts to enumerate returning spawners has evolved over time but remains imperfect. The Puntledge Hatchery fence located 6 km upstream from the Puntledge River mouth operates year-round. The associated fishway allows Chinook to bypass the fence and continue migration upriver or returns can be diverted into the hatchery raceways to sort and transport Chinook to Comox Lake for release and/or collection for broodstock (Map 9). Chinook that are allowed to continue migration upstream can be recorded and enumerated by a video camera in the fence bypass tunnel. However, annual summer Chinook enumerations at Puntledge are not absolute counts due to issues with fish jumping over the fence as detailed below. However, Hatchery staff still use the fence for collecting broodstock throughout the migration period.

BC Hydro regularly spills water through the Comox Impoundment dam during the spring freshet period (i.e., May-June), including a two-day Kayak flow release. Forty-eight-hour pulse flows are also scheduled weekly throughout the summer, from mid-June to end of July, to aid summer Chinook migration above the BC Hydro tailrace, and Stotan and Nib falls. In addition, freshet flow spills are modified to provide additional adult migration pulses. These events often overwhelm the fence hydraulic capacity resulting in Chinook to swimming over the structure.

The fence plugs with gravel and debris during the fall-winter months. When this occurs, flow is constricted through the fence panels and water flows over-top the fence, allowing Chinook to jump over at a greater range of discharges. To improve summer Chinook capture and enumeration at the fence, hatchery staff have begun cleaning the fence starting in 2019 in late-May/early June before summer Chinook arrive (Figure 62). This work must be performed when river discharge is at or below ~30 m^3/s to 35 m^3/s, so that staff can access and clean the fence safely. However, even if the fence is cleaned, and the discharge has remained low to moderate without overtopping the fence, fish have been observed jumping over when the downstream water level is within 0.3 m of the top of the fence. Therefore, a total fence count is rarely possible during the spring-summer period. In addition, at lower flows, water flow through the bypass fishway is blocked by a gravel bar that forms at the entrance. Coroplast sheets are strategically placed along the fence to block and divert water to maintain flow through the bypass fishway. However, even with this modification, gravel bar formation during the winter months often remains at the fishway entrance and disrupts or prevents fishway passage during the summer.

In the last decade, boulders were placed downstream of the fish fence creating several issues. Boulder placement was not a feature of the original design and installation of the fence. The boulders backwater the fence leading to overtopping at relatively low flows allowing fish to swim over the fence. Sediment deposits restrict upstream migration for fish that are allowed to pass through the bypass structure on the right bank and plug the entrance to the fishway entrance. The reason why boulders were placed below the fence is unclear. It’s assumed that the boulders were placed to prevent fish swimming over the fence and prevent recycling of water under the fence and entrap people. A review on this modification is recommended to address both fence operation and safety concerns regarding recreational swimmers, tubers, and kayakers. Different modifications to the streambed immediately downstream of the fence and changes to the fence panel are now being considered to improve fence performance at higher discharges (Frisson pers. comm. 2022). In March 2024, Northwest Hydraulics Ltd. received funding approval from the Pacific Salmon Commission to conduct a hydraulic study and develop new design concepts to improve fence operation.

Once summer Chinook begin arriving at the hatchery, or are observed immediately below the hatchery fence, visual counts (snorkel swims) in the river are initiated. Locations in the river that are monitored include major pools between the hatchery fence upstream to the diversion dam, and in some years, the river section from the fence downstream to Puntledge Park (i.e., 1.5 km downstream) to determine the number of adults that may be expected to arrive at the fence, and the number that may have bypassed the fence unaccounted. Swims have been restricted to inspecting just a few select locations due to difficult access and risky swimming conditions. The limited locations where swims can be conducted safely and effectively reduces the reliability of the escapement count. Swims are typically conducted once a week, and in the future should continue from early June to July.

Counts into the hatchery and below the fence are used in combination with counts at the Diversion dam to develop an escapement count. If Chinook have bypassed the fence, staff may close the diversion fishway for a period and will conduct snorkel surveys at the diversion pool, and occasionally Stotan and Nib Falls (river discharges must be within a range for staff to safely conduct swim counts). The total count upstream of the fence at the diversion dam is doubled and recorded as the number in the river. This expansion is based on a series of radiotelemetry and PIT tagging studies that indicated on average only half of the Chinook that swim past the hatchery successfully migrate all the way into Comox Lake. Chinook are allowed access through the Diversion Dam fishway if the lower hatchery fence has been operating effectively. Under these circumstances, staff are able to accurately conduct migration counts at the fence and can capture the target number of broodstock at the hatchery fence.

Since 2009, the target broodstock capture rate has been 50% of the returns to aid in the rebuilding of this endangered population (Figure 63). This rate of broodstock capture is typically focused on the fish returning from the start of the migration to the end of June; the rate of capture is reduced to 30% for later returns in July. The total number of Chinook collected for broodstock is based on the total number of Chinook counted through the fence fishway plus an indirect estimate on the number of Chinook that swim over the fence which is based on counts upstream of the fence and at the Diversion Dam. This is tracked on a weekly basis so broodstock can be collected and transported to Rosewall hatchery throughout the migration period at the 50% to 30% rate through the summer. For example, if 100 Chinook were counted in the Diversion pool, then the estimated number that had migrated above the fence would have been 200 (assuming a 50% mortality rate). Therefore, if 200 adults were collected at the lower hatchery, all could be retained for broodstock, which equals 50% of the total returns. If 400 fish were collected at the hatchery and 100 were estimated in the Diversion pool, 300 would be kept for broodstock and the remaining 100 would be transported and released in Comox

Lake. This is an attempt to comply with the maximum 50% broodstock target. Transporting adults to Comox Lake maximizes natural survival and productivity by avoiding the 30-50% loss that normally occurs when summer Chinook migrate from the fence to Comox Lake. However, this expansion method ignores the fact that a high proportion of the fish that passed the fence fail to reach the diversion dam and will not contribute to the effective spawning population in the river. Adults that hold in the river (below the dam) all summer experience a high pre-spawning mortality rate. Thus, this estimated 50% retention of captures for broodstock is likely being underestimated.

Map 9. **Puntledge River showing various locations along the migration path of summer Chinook including the Puntledge Hatchery Fence, Stotan and Nib falls, the Diversion Dam Fishway, and Comox Impoundment Dam Fishway.**

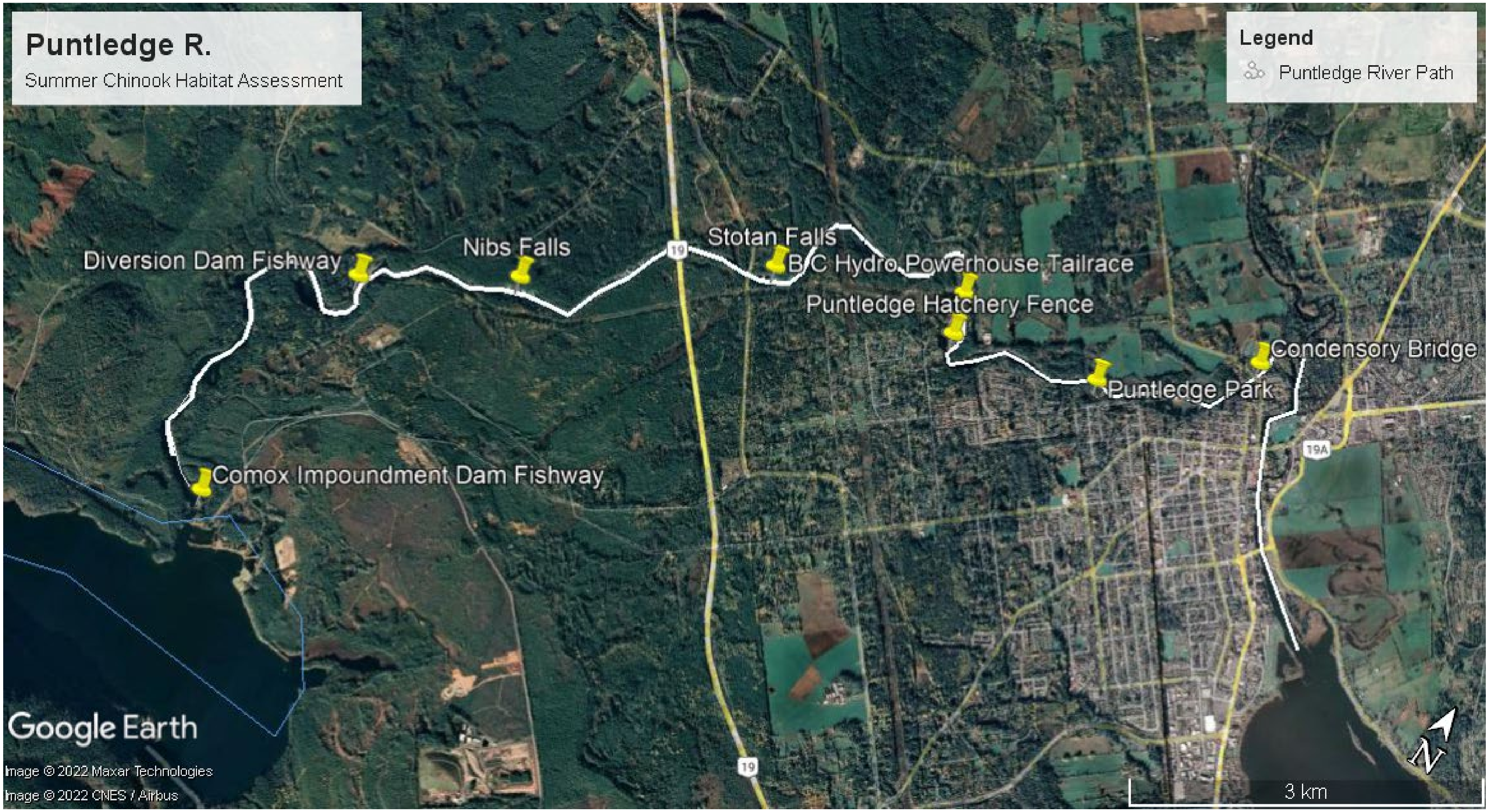
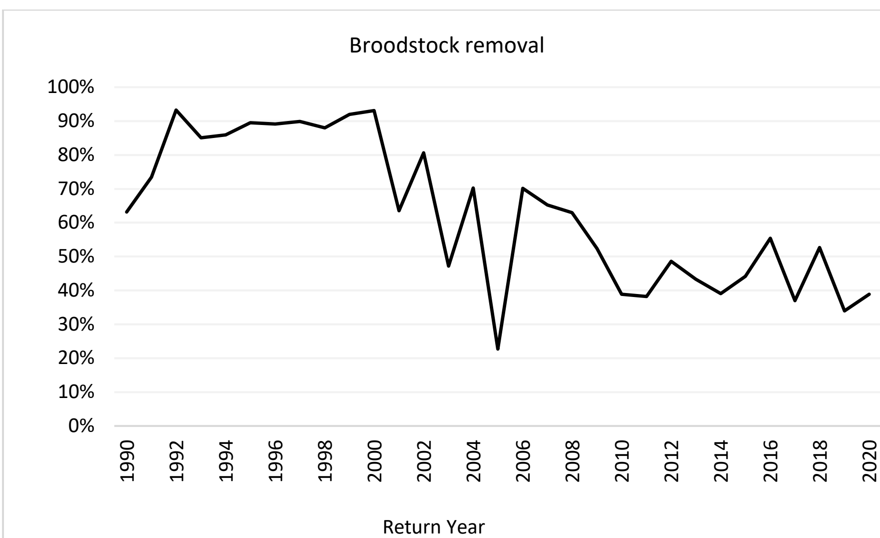


Figure 62. **Puntledge hatchery fence after cleaning gravel/debris. Flow passes through rather than over fence (Photo DFO hatchery staff).**

Figure 63. **Percent of the total summer Chinook adult return to the Puntledge River removed for broodstock (Source: DFO NuSEDS).**



4.1.7.1. An Alternate Standardized Summer Chinook Escapement Procedure

The current escapement methodology does not quantitatively take into consideration the effects that a variable summer discharge regime, water temperature, and recreational swimmer use have on summer Chinook migration success rate into Comox Lake each year. This is essential for estimating escapement, overall survival, and spawning success. The current escapement estimates are also confounded by differential seal predation on females, an unknown number of adults that bypass the fence, in-river mortality, and gamete viability.

To gain insight into the natural and anthropogenic factors that can influence escapement estimates above the hatchery fence, and limit error in escapement estimates, a multi-year monitoring study has been proposed to assess current migration success rates for summer Chinook in Reach C. During the migration-capture period between June to July, a small number (minimum of 25 to maximum of 50) of adults will be PIT tagged and returned to the river to continue their migration. PIT tagging will be conducted such that representative numbers will be released above the fence throughout the migration period. Depending on the sex ratio and overall strength of the return, it is suggested that 20% to 25% of these Chinook should also include females throughout the tagging period to track potential migration differences between sexes and avoid behavioral migration effects if no females are present. It is assumed that both males and females are able to swim over the fence when river discharges over top of the fence.

It is expected that migration success will decrease the later adults begin migration upriver because of higher water temperatures that may result in pre-spawn mortality. PIT tagging may be suspended if river temperatures become elevated since tagging-induced stress on top of temperature related stress can influence migration success and bias results. A PIT antennae receiver will be placed at the diversion dam fishway to record migration success rate of PIT tagged Chinook from the hatchery fence upstream to this receiver location. Simultaneously, a video camera will be operated to get a total count of untagged Chinook (i.e., Chinook that bypassed the fence uncounted). The untagged number would be expanded based on the success rate of the PIT tagged Chinook to estimate total escapement. If the hatchery needs additional broodstock, the diversion dam fishway could be setup to capture untagged broodstock, record tagged and untagged captures, and release fish into the headpond. A second receiver and camera at the Comox Dam fishway would provide absolute numbers of fish migrating into Comox Lake.

Snorkel swims could be conducted at Nib and Stotan Falls and the diversion pool to record Chinook that do not reach the diversion dam fishway. However, radiotelemetry results indicate that many of the Chinook that fail to migrate past the diversion dam fishway perish or experience lower gamete viability (i.e., 50% adult mortality/11.8-13.4% egg mortality; Jensen *et al.* 2006; Guimond and Taylor 2008, 2009, 2010). Even broodstock held at Rosewall Hatchery under ideal water temperature (i.e., 9 °C) recorded similar mortalities (i.e., 13%). Hence, we suspect that egg mortality in the river is likely higher. Furthermore, based on spawning studies conducted at a controlled spawning channel at

Puntledge hatchery, there is a high probability that summer Chinook will indiscriminately spawn with fall Chinook and hybridize (Withler *et al.* 2012). It is likely that larger, more robust fall Chinook males will outcompete smaller, weaker summer-run males that have held in warm water all summer and be more successful mating with summer females. The risk of hybridization is even more likely with the significantly higher abundance of fall Chinook spawners in the river below the diversion dam compared to the summer Chinook population. Since 2014, the fall Chinook escapement above the hatchery fence has averaged 6,600, which is twice the total historic and current escapement target. The summer Chinook escapement is current less than 200.

If the effective spawning population is represented by those adults that successfully reach Comox Lake, then enumeration of the PIT tagged and untagged adults at the Diversion Dam fishway will provide a sufficient estimate with the understanding that ~15% of the Chinook that access habitat upstream of the diversion dam may remain below the impoundment dam fishway (Guimond *et al.* 2016). Ideally, monitoring at both the diversion and Comox dam fishways would provide a more accurate assessment of migration success to the lake and identify potential fish access issues at the impoundment dam fishway. However, restrictions by BC Hydro and associated challenges with accessing and operating equipment in Comox dam fishway currently limits this activity.

Another important factor in Chinook escapement enumeration is the brood year sex ratio, which can be skewed in the summer Chinook returns. The hatchery has recorded ratios as high as 5:1 male to females. Reports of up to 60 seals were observed in the river during this occurrence. It is suspected that resident harbour seals may favour preying on female Chinook over males because of the larger size and nutritional value of the eggs. This will impact potential egg deposition and the size of the effective spawning population. This should be accounted for in the end of season escapement report and could possibly be estimated using the hatchery broodstock sex ratio as an indicator.

In addition to escapement counts, it is important to maintain detailed records on the various environmental parameters (e.g., river discharge, temperature) and operational activities (e.g., dates of fence closures and opening, fence cleaning, fence modifications) that can both positively and negatively impact or influence summer Chinook migration and estimated escapements. Some of these variables include:

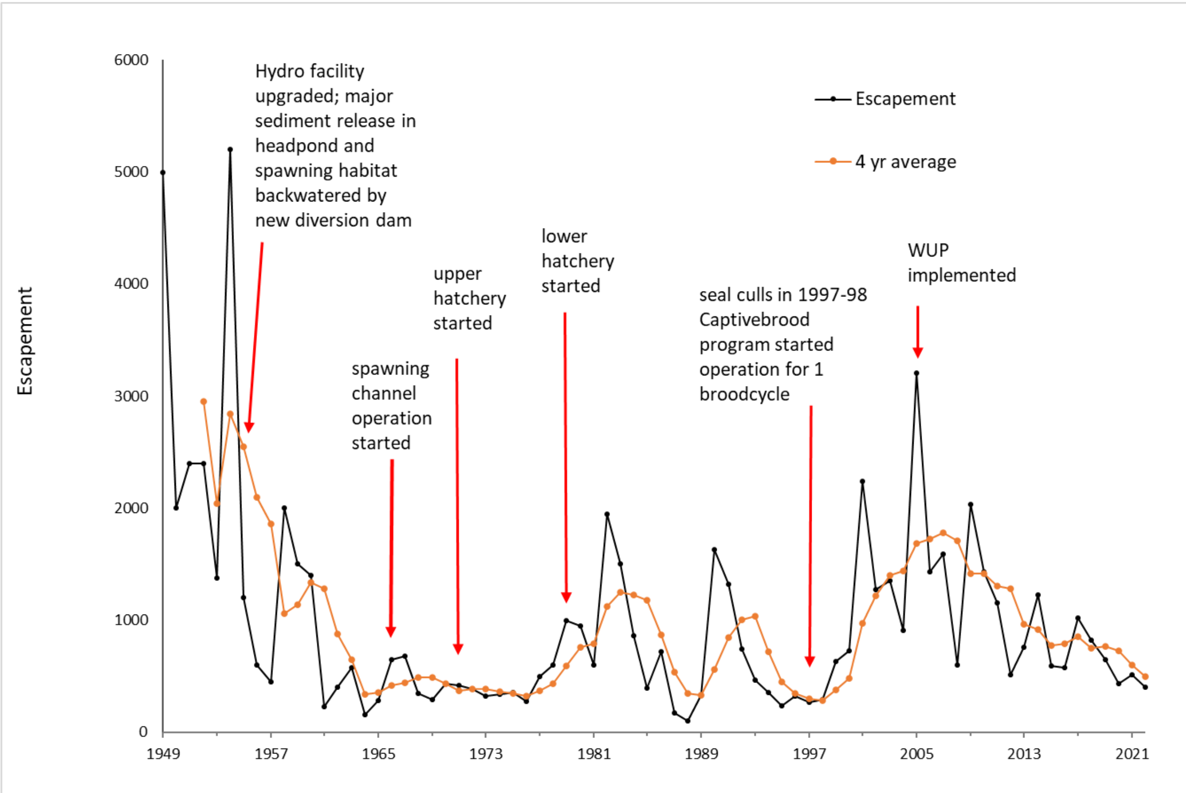
- Timing of cleaning/maintenance activities at the lower barrier fence and their effectiveness. More than one cleaning event may be necessary to maintain optimum performance of the fence. This should be a high priority for hatchery crew.
- Timing, duration, and magnitude of river discharges before and after cleaning, including the kayak pulse flow:
    - Magnitude of discharge may transport gravel to infill fence after cleaning activities, reducing fence effectiveness in blocking adult passage.

- Magnitude of discharge may be sufficient to overwhelm fence and increase passage opportunity.
- Short-term early season (early May) elevated discharges may have minimal consequences as summer Chinook adults are not in the river in high numbers. However, high flows around the end of May/early June, like those prescribed for the kayak event, may stimulate migration into the lower river and upstream, increasing the risk of passage at the fence.

4.1.7.2. Summer Chinook Escapement, Natural Spawning, Sex ratio and Proportion of Natural Influence (PNI) Escapement

A number of key activities affecting summer Chinook productivity have occurred both before and after the onset of escapement monitoring in 1949. The original hydro facility was built in 1912 but did not provide fishway access into Comox Lake until 1927. Upgrades were undertaken at the hydro facility in the 1950s, which impacted the main spawning habitat for summer Chinook located between the Comox impoundment dam and the Diversion dam. These impacts are detailed in a chronologically bullet list below and included (1) back flooding of the spawning habitat; (2) failure of a coffer dam at Comox Lake that deposited thousands of cubic meters of sediment downstream, further impairing the spawning gravel, and (3) a significant increase in water diversion for power generation and a greater juvenile entrainment mortality. Escapement subsequently dropped from a range of 2,500 to 5,000 adults between 1949 and 1954 to below 500 adults between 1968 and 1977. From 1965 to 1971, a spawning channel was constructed and operated just above the diversion dam. During this period, rebuilding of the summer Chinook population failed, and the escapement remained below 500 into the future. Hatchery operation began in 1972 to attempt to rebuild the Puntledge summer Chinook stock with variable results. Escapement increased to approximate 2,000 summer Chinook by 1982, declined to a low of approximate 100 summer Chinook between 1987 to 1989, increased to approximately 1,500 in 1990, then declined to approximately 200 in the mid and late 1990s. This coincided with an increase in resident harbor seals, which were observed preying on downstream migrating juveniles and migrating adults in the river. Seal culls in 1997 to 1998 and implementation of a captive brood program from 1997 to 2000 may have resulted in escapements increasing to 3,000 in 2005. Since this peak, escapement has steadily declined to 405 adults in 2022 (DFO NuSEDS 2022) (Figure 64).

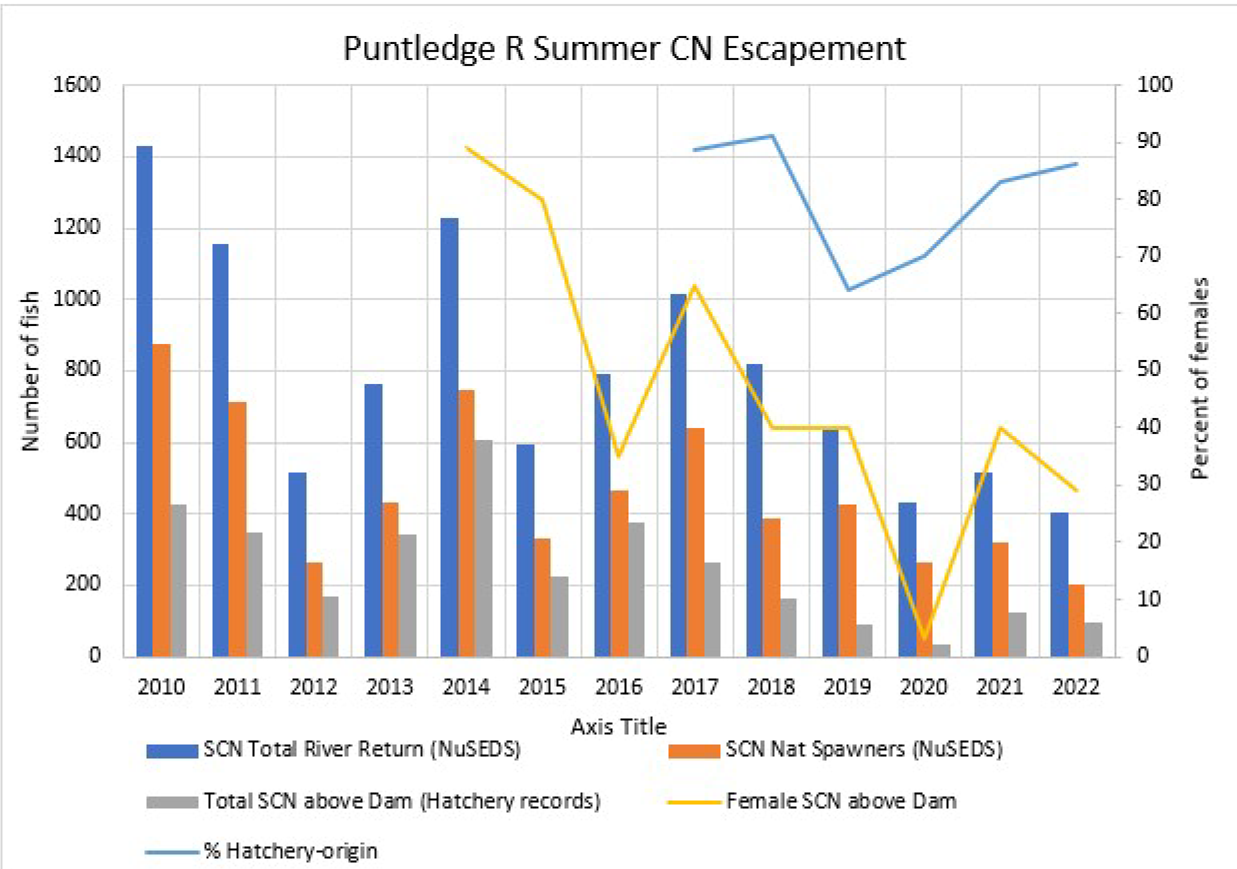
Figure 64. Summer Chinook Escapement 1949 to 2022 (Source: DFO NuSEDS), and timing of key activities.



Since 2010, there has been a decline in the total summer Chinook returns, the number of natural spawners, and the total number that either migrate or are transported from the hatchery to above Comox dam (Figure 65). Since 2013, DNA has been used to verify sex. Although the total returns have been between 1,200 and 400 in the last ten years, the number of females that were recorded above the Comox dam has dropped from 89 in 2014 to 29 in 2022 (only three females were recorded in 2020). Parental based tagging (PBT) samples of the returning population indicate the percent hatchery origin in the returning population has ranged between 64.3% to 91.3% between 2017 and 2022. There is debate on the minimum escapement required to ensure sustainability of small salmon populations. Estimates between 500 to 1,000 have been proposed by DFO. However, the percentage of females in the returns between 2014 and 2021 has been below 35% five times. This low proportion of females has major impacts on the effective population size.

From a genetic conservation perspective, a proportion of natural influence (PNI) of > 0.72 is targeted as an acceptable level for a hatchery-wild integrated hatchery program. In the last six years, PNI ranged approximately between 0.1 to 0.4. Currently, the Puntledge Summer Chinook escapement and PNI status falls far below these sustainability and conservation limits. However, at this low population level, hatchery intervention continues to be critical, to offset freshwater impacts and to maintain the population. In summary, although PNI is currently low, the hatchery program is critical for rebuilding this stock which declined to 29 females in 2022. The hatchery procedures maintain the existing genetic diversity and new rearing strategies are being implemented to increase survival. Fry to adult survival of natural spawner is below replacement rates.

Figure 65. Total summer Chinook (SCN) escapement, SCN above the diversion dam, and SCN natural spawners (Primary Axis) as well as the percentage of female SCN above the dam and hatchery origin fish (Secondary Axis) between 2010 and 2022.



4.1.8. Risk of Hybridization

Historically, Stotan and Nib Falls were strong selective fish barriers to fall-run Chinook but were passable during the spring-early summer period for summer-run Chinook. During the 1960s and 1970s, these waterfalls were modified to improved passage for summer-run Chinook during their upstream migration (Bengeyfield and McLaren 1994). Unfortunately, these activities inadvertently benefited other species that previously were not capable of ascending the falls, including fall Chinook, Pink and Chum salmon, as well as winter steelhead (Rimmer *et al.* 1994). The elimination of this natural physical separation of the two Chinook populations has increased the risk of summer and fall Chinook interactions below the diversion dam and possibly the likelihood of these groups spawning together.

A spawning behaviour study between summer and fall Chinook at Puntledge River Hatchery indicated no preference for either stock in choosing a mate of the same ecotype (summer or fall; Withler *et al.* 2012). The experimental results strongly suggest that it is unfounded to assume that the Puntledge River Chinook salmon ecotypes will avoid 'crossbreeding' in the wild in situations where individuals of the other ecotype constitute a large proportion of available mates. Based on the current flow regimes and channel characteristics of the river, the likelihood of hybridization between the Puntledge summer and fall Chinook populations in the natural environment below the diversion dam is high, particularly given the current escapement trends. Since the rebuilding of the Puntledge River fall Chinook population to levels exceeding the historic average escapement by nearly three-fold in some years, large numbers of fall Chinook access habitat upstream of the hatchery fence. Although they are restricted from habitat upstream of the diversion dam by management of the fishway, the ratio of fall to summer Chinook in the reach downstream of the dam may differ by as much as 40 to 1. The likelihood of summer Chinook that remain below the diversion dam to successfully spawn and produce viable offspring with high summer ancestry in this section of the river is extremely low.

In summary, the risk of hybridization with fall Chinook is high due to the overlap in migration timing exacerbated by delays in summer Chinook upstream migration and now easier access for fall Chinook-above Stotan and Nibs Falls. The hatchery currently maintains access through the Diversion Dam fishway until approximately early August to allow summer Chinook access into the headpond, the summer Chinook Salmon main spawning area and then closes it to prevent fall spawners from getting above. Attempts have been made to intensify the migration timing peaks between summer and fall Chinook back to historic levels. This has been a challenge and has become more difficult because of the declining returns. Early arriving broodstock which have the GREBL1 gene, a genetic marker for early migration, are only spawned with other early arriving brood. Rebuilding the historically earlier timing migration group is also seen as a critical strategy in addressing global warming, which forecasts higher summer temperature increases in July. The hatchery also focuses on not collecting fall Chinook broodstock until after September 5 to enhance the timing separation between the two populations.

4.1.9. Changes in Heritability of Migration Run Timing

Studies on summer Chinook migration in the Puntledge River have indicated that adults arriving in the lower Puntledge River prior to July have greater success migrating to the upper river (at or above the diversion dam) compared to those that arrive later in the summer (Guimond and Taylor 2010). The success of early arriving fish is attributed to cooler river temperatures during migration, low recreational use, and higher availability of spring freshet spills to aid upstream migration into Comox Lake. In contrast, later arrivals contend with warmer river temperatures, lower flows, and a high level of disturbance from people swimming, particularly at Stotan and Nib falls, two areas that present some of the greatest challenges for migration. Adults that hold in the cooler depths of Comox Lake through the summer have a spawning success rate of 95%, while less than 50% of adults that remain below the diversion dam may survive to spawn (Guimond and Taylor 2010). The most productive strategy for summer Chinook adults is therefore early migration into Comox Lake (i.e., before July), summer holding in the lake, and spawning above the diversion dam at the lake outlet (headpond) or in the two main Comox Lake tributaries (Upper Puntledge and Cruickshank rivers).

Migration time has been shown to be genetically controlled (and therefore heritable) in Chinook salmon (Healey 1991). Moreover, families that produce early- or late-migrating progeny tend to do so consistently over adult age classes. Therefore, we expect the early (prior to July) and late (July onwards) adults spawned in the hatchery, and those that spawn in the wild, to produce offspring with similar migration timing. It was hypothesized that selection for early migration time in the summer Chinook would have the added benefit of facilitating genetic separation of the two Chinook salmon populations for hatchery and natural spawning within the Puntledge drainage; maintaining this genetic distinction is necessary for adaptation and long-term conservation of the summer run.

A multi-year study that focused on Puntledge summer Chinook run-time and bacterial kidney disease (BKD) heritability was conducted from 2013 to 2019 to determine the level of genetic and environmental influence on migration time in Puntledge summer Chinook, and the degree to which selection for early migration times may be effective in improving their survival and abundance. The overall goal of the study was to provide guidance for the development of appropriate hatchery protocols that would maintain the genetic distinction of the summer and fall Chinook populations, properly manage BKD in the summer Chinook population (discussed in Section 4.1.10) and optimize their survival.

The study employed genetic analysis methods, known as parentage-based tagging, to identify individual summer-run Chinook salmon back to parental crosses (both those that were performed in the hatchery and those that occurred in the wild). The genotyping of parents and offspring was conducted with a set of fifteen microsatellite loci (genetic markers) that were analyzed in the Molecular Genetics lab (MGL) at the Pacific Biological Station (Beacham *et al.* 2012). Over two full cycles (parental brood years 2014 and 2015, and adult progeny returns from 2016-2019), hatchery broodstock caught at the Lower Puntledge hatchery or at the diversion dam were separated into 'Early' migrants (those arriving before July 1st) and 'Late' migrants (those caught between July 1st and August 1st) for holding, tissue sampling for DNA analysis, and spawning (i.e., all spawning within their own timing group). Similarly, adults that were transported and released directly into Comox Lake as natural spawners were also tissue sampled, distinguishing 'Early' from 'Late' migrants by transport date.

For the 2015 BY, progeny return time was significantly affected by the parental run time group, with progeny from the 'Mid' parental spawning group that arrived from July 8-31, 2015, returning later than those from the Early (May 21 – July 2, 2015) parental group (p=0.0007;). Results from the 2014 parental BY showed no effect. However, the difference between years may have been due to a greater accuracy of establishing run time as the study progressed, and the arrival timing of progeny returns in 2017 and 2018 potentially being confounded by capture locations at both the lower hatchery and the diversion dam. Furthermore, Early returning females produced potentially more returning progeny than do Late returning females (Figure 66, Table 34). Therefore, in summary, the emphasis on enabling early returning fish to spawn successfully both in the hatchery and the natural environment, should be continued to assist in maintenance or enhancement of early spring return to maximize adult migration success. This includes segregating and spawning Early and Late migrants within their run-time group. Continued genetic monitoring of the early- and late-migrating summer Chinook Salmon will be important if spawning will continue within groups for the foreseeable future to ensure artificial subpopulations of the summer Chinook Salmon are not created.

Figure 66. Mean progeny return time (Julian days) among 2017-2019 adult hatchery returns (with 95% confidence limits) given by the run time group of the 2015 BY parents. Parental groups 1, 2 and 3 were the Early, Mid and Late returning fish, respectively.

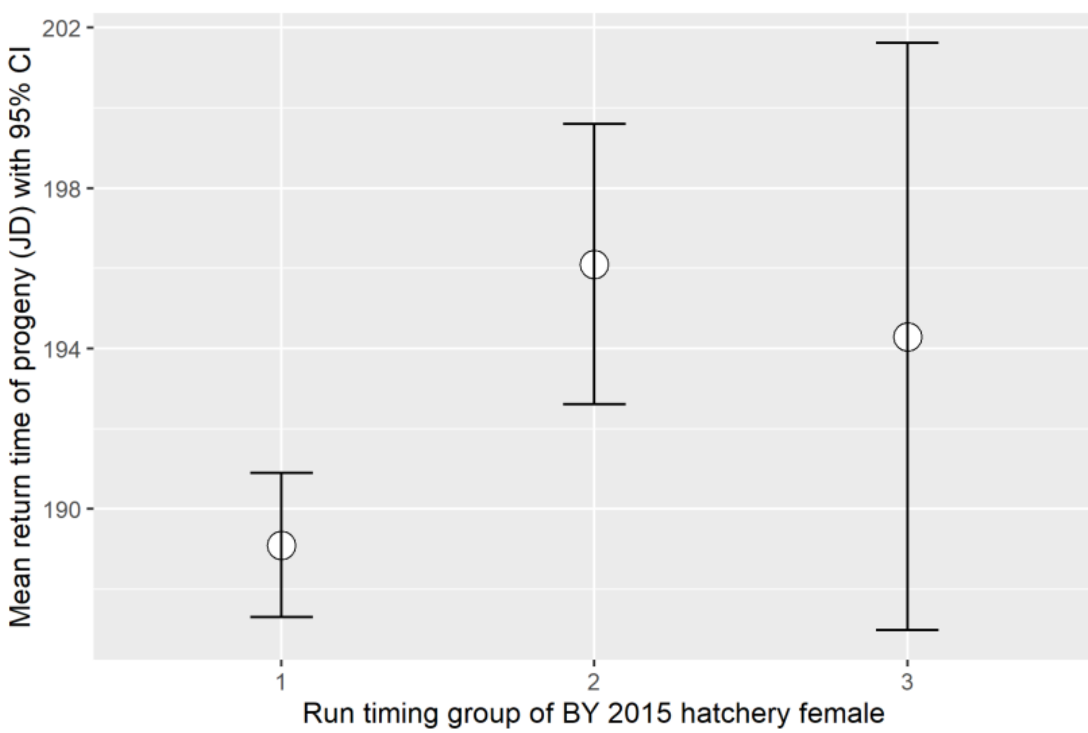


Table 34. Number of adult progeny (n) returning in 2017-2019 from hatchery BY 2015 parental run time groups, with mean return date (Julian days) and 95% CI given. Number of female spawns (dams) and progeny per dam are also given. Parental groups 1, 2 and 3 were Early, Mid and Late returning fish, respectively.

Parental Group	Run Time	Dams (n)	Progeny per dam (n)	Progeny (n)	Mean progeny return time (JD)	95% CI
1	May 21-July 2	54	6.3	342	189.1	187.3 - 191.0
2	July 8-31	19	4.7	89	196.1	192.6 - 199.6
3	August 1-7	6	2.5	15	194.3	187.0 - 201.6

4.1.10. Increases in Heritability of BKD Load

Renibacterium salmoninarum, the causative agent of bacterial kidney disease (BKD), is an endemic pathogen in the Pacific Northwest. BKD is a slowly progressing, lifelong infection of salmonids. The bacterium may be horizontally transmitted between fish and vertically transmitted to the next generation. Fish infected with *R. salmoninarum* will not normally exhibit clinical signs until the fish are a year old. As such, BKD is a serious disease in salmon culture. From a husbandry perspective, good hatchery practice is to eliminate or minimize presence of the pathogen in the hatchery and subsequently the natural environment by culling progeny from BKD-positive female parents. However, there may be a genetic disadvantage to this practice if, in fact, the positive females that are being selected against carry genes that enable tolerance of the pathogen and the ability to survive and reproduce even in the presence of bacterial infection. Despite concerns of genetic loss arising from the practice of culling eggs from females that screen positive for the pathogen, the judicious use of males will ensure that genetic diversity is retained in most situations (Hard *et al.* 2006).

The BKD specific pathogen control plan for DFO fish culture facilities has been devised to prevent clinical BKD epizootics during hatchery rearing and to reduce the risk of disease amplification through hatchery practices. The plan recommends that all Chinook and Coho stocks that have a higher-than-average historical prevalence of BKD, be annually screened and that egg culling and progeny segregation be practiced based on female parental Enzyme Linked Immunosorbent Assay (ELISA) optical density (O.D.) readings of *R. salmoninarum* antigen levels. Other stocks are subjected to periodic prevalence assessment of 60 fish, to confirm BKD risk status. The Puntledge summer Chinook stock was identified as a high risk BKD stock during routine screening of 2009 and 2011 broodstock. As a result of the revised stock BKD risk designation, the production strategy was altered to improve biosecurity and to participate in annual BKD broodstock screening, egg segregation, and culling based disease risk management. Specific biosecurity measures employed include pre-spawning antibiotic administration to females prior to egg collection, iodophor egg disinfection during water hardening, incubation in individual Heath trays until broodstock ELISA results are available, and culling based on levels of soluble *R. salmoninarum*-antigen detected using ELISA.

For the Puntledge summer Chinook Salmon, a stock of conservation concern, the following hatchery rearing protocols were implemented for eggs/progeny from females with various levels of infection following direction from the DFO veterinarian:

- *Negative (N)* **–** fertilized eggs/progeny from females that have a lower optical density (OD) value than those of the kidneys of the negative control fish. No restrictions on progeny rearing.
- *Low Level of Detection (LLD)* **–** OD values <0.1 but greater than the mean negative control. LLD eggs present a low enough risk of BKD to be treated as negative. No restrictions on progeny rearing.
- *Low Positive (LP)* **–** OD value ≥ 0.1 but < 0.25. No restrictions on progeny rearing, but fry are not marked (CWT & adipose clip). For the 2014 and 2015 BYs, the LP female egg lots were further divided into a *Low Low Positive (LLP*; OD ≥ 0.1 <0.14) and *High Low Positive (HLP*; OD ≥ 0.14 <0.25**)** groups. It is generally recommended that these fry be reared separately from progeny from negative and LLD screened female brood.
- *Moderately Positive (MP)* **–** OD value ≥ 0.25 but < 0.6. Progeny outplanted as eyed eggs.
- *High Positive (HP)* **–** OD ≥ 0.6. Eggs are destroyed.

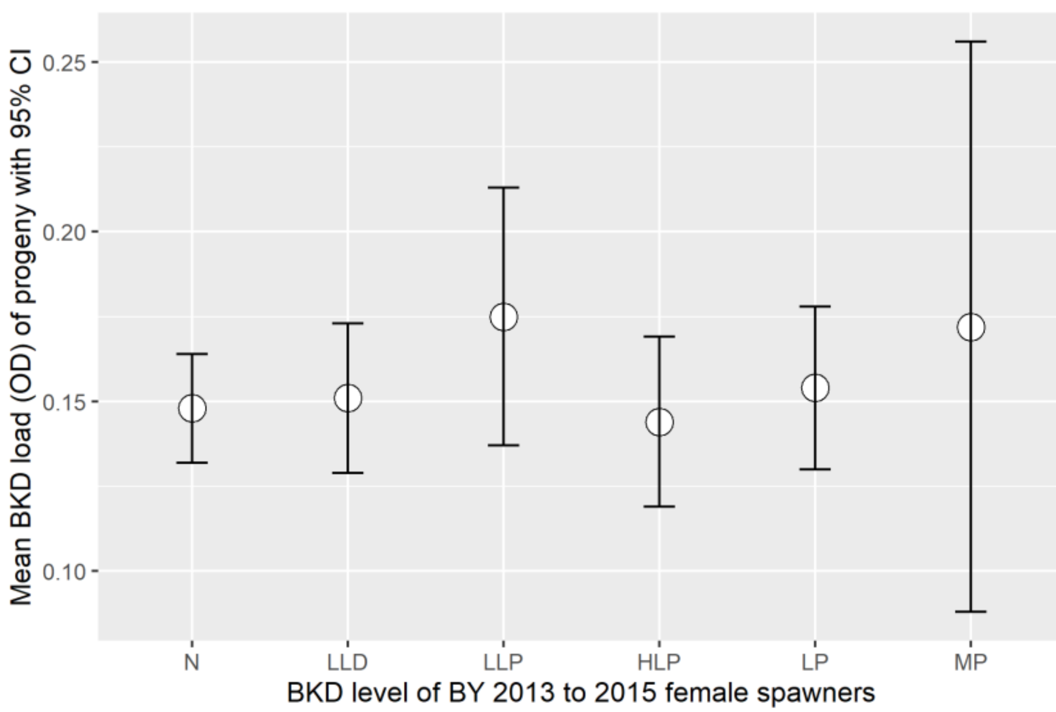
The fate of hatchery individuals that are disease-free (i.e., display no clinical signs of the disease), but carry the *R. salmoninarum* bacterium due to vertical transmission, and their impact, or lack of, on their naturally spawned counterparts following release is neither well understood nor simple to predict due to variable effects of environmental stressors on transmissibility and pathogenicity. However, there is strong evidence that culling and segregation of eggs from higher titre females can reduce prevalence of the disease in subsequent generations (Elliott *et al.* 1995, Munson *et al.* 2010). Good husbandry and a precautionary approach to wild interactions have been the driving factors to date in developing appropriate protocols for responding to infection in the hatchery environment. However, there is value in examining the impacts of exclusion of progeny from *R. salmoninarum* positive females on genetic diversity in populations of conservation concern. Where populations are exceptionally small, the consequences of the loss of genetic diversity must be adequately balanced against the risk of inclusion of females carrying a higher pathogen load. Regardless, the ability to follow the survival and reproductive success of offspring from individual BKD positive and negative females in the Puntledge summer Chinook Salmon population will assist both in its management and in the refinement of general husbandry protocols for BKD affected hatchery populations, as such a study was conducted to evaluate this relationship (Withler and Guimond 2015).

For each brood year, female broodstock were classified for BKD load as negative (N), low level of detection (LLD), and low positive (LP, LLP and HLP). Eggs from moderate positive (MP) females were outplanted to the natural environment at the eyed stage, while those from high positive (HP) females were culled. For all female parents in BYs 2013 to 2015, progeny BKD load among fish incorporated into future broodstocks was compared to maternal BKD load to determine if there was a relationship between maternal and surviving progeny infection level (Figure 67). Route of transmission of *R. salmoninarum* was not examined in this study.

For all brood years, progeny from BKD-positive females (i.e., those classified as 'low positive') that were maintained in hatchery production survived as well as progeny from BKD-free females. Even eyed egg outplants from moderate positive females produced some returns. In addition, there was no relationship between maternal BKD load on the BKD load of her female offspring that returned to the river (Figure 67).

The retention of eggs from BKD low-positive females for in-hatchery rearing will increase abundance and the maintenance of genetic diversity in stocks of conservation concern such as the Puntledge summer Chinook Salmon population. This practice should be continued as long as the hatchery-specific biosecurity protocols implemented to minimize risk of BKD outbreaks can be maintained.

Figure 67. Mean progeny BKD load among 2015-2019 returning adults by maternal (BY 2013-2015) BKD. Classified as negative (N), low level of detection (LLD), low low positive (LLP), low positive (LP), high low positive (HLP) and moderate positive (MP).



4.1.11. Risks to Genetic Diversity

The 'genetic effective size' of a population (N_e - the effective size per generation), represents the size of a 'perfect' population that would contain the same amount of genetic diversity as the actual population or sample being analyzed. A perfect population is one in which the sex ratio is 1:1 and every parent (male and female) contributes the same number of progeny to the spawners in the next generation. Thus, the N_e for a real population is usually smaller than the census size of the real population. The genetic effective size of a population can be estimated from the microsatellite genotypes. Factors that reduce the genetic effective size of a population from one generation to the next include:

- A high proportion of unsuccessful potential parents.
- An unequal sex ratio (both sexes contribute half the genetic information to the next generation; if successful spawners of one sex are scarce, then half the genetic diversity of the next generation comes from only a few males or females.
- Highly unequal contributions of successful parents to the next generation, again limiting the numbers of fish that contribute diversity to succeeding generations.

As a measure of genetic diversity over a reproductive cycle (N_b – the effective number of breeders in one reproductive cycle), the annual effective population size of adult hatchery brood fish from BY 2013 to 2019 was estimated using the linkage disequilibrium method in NeEstimator 2.01 under the assumption of the random mating model and excluding allele frequencies less than 0.02 (Do *et al.* 2014). Effective population size estimates of natural origin and hatchery origin adults in the 2017, 2018, and 2019 returns were also calculated.

The N_b estimates for Puntledge summer Chinook Salmon brood fish were higher in 2013 and 2014 than in subsequent years (Table 35). By 2019, the N_b value was roughly two-thirds of that observed in 2013. Annual estimates of N_b in hatchery brood from 2016 to 2019 were close to the harmonic mean estimate of the annual N_b estimates (n=229). The apparent reduction in effective population size in the summer Chinook Salmon population over the course of the study, coupled with the ongoing low abundance in the overall population, is of concern and supports the continued use of genetic screening to avoid inbreeding and maximize diversity within the population while excluding FCN from summer Chinook broodstock collections (Wetklo *et al.* 2020).

Table 35. Sample size and annual effective population size estimates (Nb) for Puntledge summer Chinook hatchery broodstocks from 2013 to 2019. 95% confidence intervals (CI) are shown for annual N_b values and the harmonic mean (HM) of N_b estimates is given.

BY	n	N_b	95% CI
2013	182	410	327-540
2014	271	321	282-371
2015	186	164	147-184
2016	121	238	195-301
2017	219	211	188-239
2018	291	207	188-229
2019	172	197	173-226
HM	1441	229	

The Ryman-Laikre (R-L) effect is an increase in inbreeding and a reduction in total effective population size (N_{eT}) in a combined captive–wild system, which arises when a few captive parents produce large numbers of offspring. Decreased contributions from natural Puntledge summer Chinook spawners since 2014, or increased contributions from fall spawners from 2013-2014 could account for this decrease in diversity (Wetklo pers. comm. 2023). Since 2015, the number of effective summer breeders has averaged 203 (i.e., 2015-2019; Table 35). If there was a R-L effect occurring we would likely see decreasing effective breeder size, but this is not present in the subsequent years. The stability of genetic diversity for Puntledge summers likely reflects good hatchery practices. However, results from 2020-2022 have not been examined.

A fairly large and similar number of unrelated females contributes to the hatchery broodstock in most years. From 2019 to 2022, the number of unrelated females contributing to the summer broodstock has averaged 124 (Table 36), with 2020 being the only year in which the number of these females was significantly below the average. An additional measure of diversity could also be determined from existing data (e.g., expected heterozygosity).

Table 36. The number of unrelated females contributing to the Puntledge summer hatchery brood.

Collection year	Hat
2019	142
2020	86
2021	137
2022	132

Genetic Diversity and Juvenile Out-migration Timing

DNA sampling of the early emergent juvenile Chinook migrants in 2016 that were progeny from BY2015 natural spawners above the diversion dam indicated a loss of genetic diversity from parents to off-spring. The effective size of the natural fry sampled in 2015 and 2016 was less than a third to one half of that represented in their pool of potential parents. Neither the early nor later emergent juveniles captured all the juvenile genetic diversity. Furthermore, the genetic diversity of juveniles was not randomly distributed over the entire out-migration period with differences in family out-migration timing accounting for the non-random distribution of genetic diversity in the juvenile samples. Only ~20% of the maternal families analyzed in 2016 were exclusively from early emergent migrants (Wetklo *et al.* 2017). However, it is noted that a much lower sampling rate on emergent fry during a period when the BC Hydro facility was shut down for maintenance may have influenced the interpretation of genetic diversity.

This becomes significant when evaluating genetic diversity in the context of our current knowledge of juvenile summer Chinook migration and survival in the upper Puntledge River. The small size and corresponding weaker swimming ability of the emergent juveniles (i.e., 35-50 mm fork length) during the early migration time period (February-May) puts them at greater risk of entrainment and mortality at the BC Hydro facility. Hydro-related mortality of juveniles with fork lengths less than 50 mm was estimated to result in a 19.4% loss of genetic diversity in BY 2015 alone. The cumulative effect of selective mortality to the early emergent juveniles is therefore a concern for the preservation of existing genetic diversity in the summer run Chinook (Wetklo *et al.* 2017).

Contribution of Natural and Hatchery Origin Returns

Both hatchery- and natural-origin fish contributed to genetic diversity in the population. Parentage analysis of the Puntledge summer Chinook population returning from 2015 to 2021 indicates that 86.6% were attributed to hatchery production and 13.4% to natural spawners (Table 37). The 2014 hatchery BY was particularly successful, likely due, in part, to a greater hatchery smolt production in that BY (>500,000 smolts). In contrast, BY2016 had the lowest hatchery contribution, corresponding to a year of low hatchery production (<100,000 smolts). Environmental factors such as river temperatures, discharge, predation, and ocean conditions, would have also influenced survival differentially for these brood years.

Table 37. Number and percentage of adult Puntledge summer Chinook sampled between 2015 and 2021 assigned to hatchery- (BY 2013-2019) and natural- (BY 2014-2019) origin parents. Unassigned adults in 2017 to 2021 were considered natural-origin and attributed to the Unknown BY class.

Origin	Year								Unknown	Total (%)
	Total (n)	2013	2014	2015	2016	2017	2018	2019		
Hatchery (n)	1993	384	826	446	53	149	121	14	-	
(%)		16.7	35.9	19.4	2.3	6.5	5.3	0.6	-	86.6
Natural (n)	309	-	13	69	25	50	10	4	138	
(%)		-	0.6	3	1.1	2.2	0.4	0.2	6	13.4
Total (n)	2302	384	839	515	78	199	131	18	136	

The 2015 natural-origin brood year was also relatively successful, possibly owing to a greater number of naturally spawning females upstream of the diversion dam, and a 40-day shut-down of the BC Hydro Generating Station during peak early emergent fry migration that may have reduced entrainment mortality on this cohort. In 2017 and 2018, fish of natural origin constituted less than 12% of the escapement, whereas in 2019 and 2020, they comprised approximately a third of returns (Table 38).

The contribution of spawners in the natural environment is critical to maintaining adaptation to that environment. Whereas hatchery-produced fish survive well in captivity, they often survive poorly after release from the juvenile (Beamish *et al.* 2012) to the adult reproductive (Fleming and Gross 1993; Ford *et al.* 2016; Christie *et al.* 2014) stage relative to naturally produced fish. Thus, maintaining conditions in the natural environment that support juvenile survival and adult reproduction is of fundamental value to restoration of the Puntledge summer Chinook population. To date, PBT of the Puntledge summer Chinook population has facilitated the evaluation of adult survival and the differential contribution of natural and hatchery spawners to the escapement. In future years, it will also be possible to evaluate how hatchery or natural origin affects spawning success in both the natural and hatchery environments because the origin of all returning fish, both those released to the wild and those collected as hatchery brood, have been determined using PBT.

Table 38. Percentage of hatchery-origin and natural-origin summer Chinook adults determined by parentage-based tagging, returning to the Puntledge River in years 2017 to 2021.

Return Year	% Hatchery-origin	% Natural-origin
2017	88.6	11.4
2018	91.3	8.7
2019	64.5	35.5
2020	70.1	29.9
2021	83.2	16.8

GREB1L Genomic Region in Puntledge Summer CN

A large genomic region called GREB1L has recently been discovered to be associated with seasonal run time in Chinook salmon (Prince *et al.* 2017; Thompson *et al.* 2019). Although GREB1L function is not fully understood, it is found to be associated with foraging and fat storage and expressed in renal and reproductive tissues in other organisms (Willis *et al.* 2021). Numerous variable nucleotide locations within GREB1L (termed Single Nucleotide Polymorphisms, or SNPs) are associated with two major haplotypes – one responsible for early (spring-summer) migration (E) and one with later (fall) migration (L). Puntledge summer-run Chinook were surveyed for GREB1L at two SNPs to determine SNP genotypes and infer GREB1L haplotypes (the combined SNP alleles by chromosome) from them. The two common haplotypes are EE (chromosomes that carry the nucleotide associated with early migration at both positions), and LL (chromosomes that carry the nucleotide associated with late migration at both positions). These haplotypes, when present in the homozygous state in fish, are associated with spring and fall migration times, respectively. Heterozygous fish, carrying one copy of each haplotype, have intermediate migration times and are relatively unsuccessful in the natural environment (Prince *et al.* 2017). Two less common haplotypes, EL and LE, also occur and their effect on run time is unknown.

The near fixation of the EE haplotype in the GREBL1 genomic region in the Puntledge summer Chinook population (and the predominance of the LL haplotype in the Puntledge fall Chinook population) supports the assumption that GREBL1 is responsible for the seasonal run time differences between the two Puntledge populations (Wetklo *et al.* 2020). EARLY and LATE homozygotes migrate in different seasons to different locations, whereas EARLY/LATE heterozygotes demonstrate intermediate run times and tend to be selected against in natural populations (Prince *et al.* 2017; Thompson *et al.* 2019).

Maintenance of the EE haplotype in the summer Chinook population has occurred despite likely hybridization between summer and fall Chinook in both the natural and hatchery environment and underscores the importance of the genotype to fitness of the summer Chinook population.

Therefore, genetic screening to avoid the use of broodstock carrying the LL haplotype should be continued. Without a genetic screening program, avoiding the collection of Puntledge Summer Chinook broodstock in August and possibly in mid to late July, depending on the overall abundance and run-time distribution of the returning adults, should reduce the likelihood of including early returning FCN salmon in the summer Chinook brood collection.

4.1.12. Quality of Spawning Habitat

The reach between the diversion dam and the impoundment dam 3.7 km upstream (headpond, Reach B), was historically the most important spawning area for summer-run Chinook salmon and steelhead (Anon.1958; Hourston 1962; Rimmer *et al.* 1994). However, the designation of this reach as such applies only to the period following initial impoundment in 1912. It is unclear whether this reach was a preferred habitat historically (prior to 1912) or whether it became the preferred alternate spawning location after access to the lake and tributaries was obstructed (Bengeyfield 1992; McLaren. 1994). Spawning habitat below the diversion dam to Stotan Falls has been critical for natural summer Chinook production as access to upstream became limited.

The spawning area upstream of the diversion dam once supported a run of 3,000 adult summer Chinook prior to hydro expansion based on the average return from 1949 to 1954. However, estimates of the potential to support spawners in this reach was reportedly higher. For instance, the gravel in this reach was reported to be capable of supporting 8,000 summer Chinook (Holden 1958). DFO estimated that the accessible spawning habitat within the Upper Puntledge and Cruickshank rivers systems could accommodate up to 7,000 adults (Bengeyfield 1992). However, the Cruickshank River is much cooler than the Puntledge River, and spawning habitats are exposed to more extreme scouring flows, and other logging related impacts due to the prevalence of forestry activities in this watershed. A map of the Puntledge River (Lister 1968) identified some of the former natural spawning areas in the headpond. These were located below the impoundment dam, in a ~400 m stretch of river about 2.4 km further downstream, and another ~400 m stretch of river about 400 m upstream of the diversion dam (upstream of the Upper Hatchery).

A large proportion of post-glacial deltaic gravel deposits and sand sediments in the Puntledge River became isolated after the construction of the diversion dam at an elevation of 130 m (430 ft) asl downstream of Comox Lake (Guimond 2004). Since 1912, gravel contributions from sources above the diversion dam have been isolated by the dam and inputs are now basically limited to downstream deltaic and fluvial terrace sediments. Supply Creek, a small tributary located in Reach B between the two dams, is the only location where some gravel has deposited and has formed a bar on the far left-side of the Puntledge mainstem confluence (Bengeyfield 1992; McLaren 1994). Spawning habitat surveys conducted in Reach B in 1992 and 1993 estimated that between 1,500 m^2 and 4500 m^2 of suitable spawning gravel for summer Chinook spawners remained, depending on discharges from the Comox impoundment dam (Bengeyfield 1992; Bengeyfield and McLaren 1994). Using the minimum estimate of suitable spawning area remaining, and a biostandard of 20 m^2 of spawning area per pair (Burt 2004; Burner 1951), approximately 75 pairs of summer Chinook could spawn in Reach B.

The original diversion dam in the Puntledge River was constructed with a 23 m long fishway; however, it was noted that it did not operate adequately (Holden 1958). It was likely not an issue since adults were still able to gain access to their main spawning area upstream by jumping over the low head dam. The crest elevation of the original dam was 128.5 m (421.5 ft; Holden 1958). Following the reconstruction of the diversion dam in 1958 the elevation of the dam was increased by 1.2 m to a new elevation of 129.7 m. In 1965, the fishway was closed to adult passage into the headpond, due to the loss of spawning habitat and turbine mortality following expansion of the hydro facilities. The Upper Hatchery spawning channel was constructed and utilized part of the fishway to attract adult Chinook holding in the pool below the diversion dam into the earthen spawning channels. Fish barrier racks were installed on top of the diversion dam to prevent adults from jumping over the dam to access their historical spawning habitat, and to direct them into the fishway. However, in some years, Chinook were observed attempting to jump over the dam and were often injured as they contacted the barrier racks. This accounted for a high incidence of pre-spawning mortality at the spawning channel in 1966-1968. In 1970, a wood sill was installed along a section of the crest of the diversion dam to divert flow away from the area where adults were observed trying to ascend the dam. This remedial work was successful at reducing jumping activity and injury, although it likely caused additional backwatering of habitat upstream of the diversion dam by the 0.7 m increase in elevation from the additional wood sill. Access into the headpond remained closed to salmon migration until 2001 when surplus summer Chinook broodstock were allowed to utilize habitat upstream of the diversion dam once again.

Following the expansion of the hydro facilities in the 1950s, the summer-run Chinook stock began to decline to critically low levels. One of the most significant impacts from this expansion was the impairment of summer Chinook spawning habitat upstream of the diversion dam. The increase in the height of the diversion dam and consequent back flooding significantly altered the hydrology in this section of river and resulted in reduced velocities and greater water depth over the spawning beds. Spawning gravels in the headpond reach were further impaired during reconstruction of the impoundment dam at Comox Lake in 1957. An earthen coffer dam failed during a December flood depositing thousands of m^3 of sediments on the spawning grounds downstream. This siltation event was responsible for the near complete loss of fry production during the winter of 1957-1958 (Hourston 1962). Impairment of this brood year was evidenced by the lower-than-average four-year cycle returns following this incident (Marshall 1972). Further back flooding of the headpond was caused by the installation of wood flashboards on the diversion dam in 1971 to attempt to reduce attraction flows over the dam and discourage jumping attempts by adult Chinook (Bengeyfield 1992; McLaren 1994). Due to the presence of the diversion dam, the reduced velocities through this reach and the inability of this section to be subjected to seasonal flushing, the area has been slow to recover, and fine sediments continue to persist throughout much of the spawning grounds today (Rimmer *et al.* 1994). Additional information on the migration barriers is provided in Section 4.1.5.

A 2003 survey of spawning gravel in Reach C (diversion dam to the powerhouse) was completed as part of an "information gap" study during the Puntledge WUP process. It was found that 90% of the functioning gravel in this reach was located in three discrete areas along the river: Barbers Hole and Bull Island side-channel located upstream of Stotan Falls, and in a small area downstream of the Browns River confluence (MJL 2003). The remaining 10% was found in scattered patches along the wetted edge of the channel for a total of approximately 1,955 m^2 of functional gravel available for spawning Chinook and steelhead. It should be noted that this amount includes 857 m^2 of recently placed spawning gravel in Bull Island side channel that was a two-year project restoring over 2,165 m^2 of spawning habitat in the side channel (see Section 2.7.2.2). A summary of spawning habitat in Reach B and Reach C is provided in Table 39.

Table 39. Status of Puntledge River summer Chinook and steelhead spawning habitat and its potential capacity.

Spawning Habitat Status	Reach B (m^2)	Reach C (m^2)	Reach B&C (m^2)	Spawning pairs	Non-functioning spawning habitat (m^2)	Sources / Comments
Target			30,000	1,500		
Spawning Habitat Status (1992 to 2003)	1,500	1,098	2,598	130 [1]	4,705	Bengeyfield and McLaren 1994; MJL Environmental Consultants 2003
Spawning habitat additions since 2002	6,600	3,972	10,572	1,057 [2]		Guimond and Norgan 2003; Silvestri 2007; BC Hydro 2020
Spawning Habitat Status (2008)	8,100 [3]	5070	13,170	1,187		Placement of 1874 m^2 of gravel in 2021, as per the PUNWUP Implementation
Required to meet target			3,130	313		Escapement target for summer Chinook under review by DFO

[1] Used a spawning biostandard of 20 m^2 per spawning pair for natural spawning sites (Burt 2004).

[2] Used a spawning biostandard of 10 m^2 per spawning pair for man-made spawning sites (Burt 2004).

[3] Some spawning gravel additions overlapped with natural spawning sites, mostly in Reach B. An approximate amount has been subtracted from the total.

Restoration efforts have made improvements to spawning habitat in the Puntledge River. Summer Chinook have been observed spawning in a ~4,700 m^2 area of restored gravel located approximately 1 km upstream of the diversion dam, and in a 200 m stretch of river used historically, immediately downstream of the impoundment dam. In 2021, additional gravel was placed in this section of river immediately downstream of the pool tailout below the Comox Impoundment dam increasing the spawning area to 1,873 m^2. This Reach is now capable of supporting over 300 spawning pair at a spawning biostandard of 20 m^2 per pair restoring the potential spawning in this habitat to pre-hydro expansion numbers. However, man-made spawning habitat tends to support higher densities of spawners due to the high quality of their gravels (loosely compacted, optimal range of particle sizes and adequate substrate depth). Therefore, a spawning biostandard of 10 m^2 per pair can be employed (Burt 2004) for introduced spawning habitat doubling that estimated amount of spawning pairs. These areas are the only spawning locations available in Reach B that total 6,573 m^2. Overall, spawning habitat is not limiting. Reach B can support 657 females or 1,315 adults. This is well above the current escapement of less than 200 Chinook plus there is likely several thousand m^2 of spawning habitat in the Upper Puntledge and Cruikshank Rivers.

4.1.13. Fishing Pressure

There is no legal retention of summer Chinook in the Puntledge River or in Comox Lake. Recreational fishers have been reported landing summer Chinook in Comox Lake while legally fishing for trout or kokanee. Bryce Gillard, the last fishery officer stationed at the Comox DFO office, relayed that fishers were able to target summer Chinook off the mouth of the Cruikshank River (Figure 68) by using trolling gear fished at the thermocline (i.e., 10-15ºC). This temperature range is noted to be preferrable for summer Chinook as it is the same temperature range that was recorded from internal temperature probes recovered from Chinook carcasses that when alive held in Comox Lake all summer in 2010 (Guimond and Taylor 2010).

On July 1, 2022, salmon fishing in Comox Lake was closed by DFO following consultation with the province (Variation Order: 2022-RCT-268). Fishers were advised of a new salmon fishing closure in Region 1 to address conservation concerns for at-risk Puntledge River summer Chinook stocks. "This closure is year-round salmon fishing closure in Comox Lake. Puntledge Summer Chinook are a stock of concern that migrate through the Puntledge River and into Comox Lake from March until August. The Chinook stage in Comox Lake until it is time to spawn in the fall". Although fishers can still fish for trout and kokanee, the hope is that fishers will no longer target summer Chinook and are restricted from fishing at the mouth of the Cruikshank River.

Figure 68. Photo of summer Chinook captured in Comox Lake near the mouth of Cruikshank River on September 8, 2021 (Photos provided by Nick Strussi).

4.1.14. Competition with Hatchery Adults or Aquaculture

There are currently no issues involving competition with escaped aquaculture fish for Puntledge River Chinook. However, returns of hatchery fall Chinook adults has been identified as a potential threat to wild and hatchery summer Chinook adults. The large number of hatchery adult returns can potentially compete with summer Chinook spawners in Reach C and D increasing the risk of hybridization (See Section 4.1.8).

4.1.15. Increase in Didymo Abundance

The spread of diatom species known as Didymo (*Didymosphenia geminate)* on Vancouver Island, British Columbia occurred between 1988-1998 (Figure 69). There was no obvious change in water chemistry, hydrological regime, or other environmental metrics associated with the onset of the blooms. Literature searches determine that didymo is native to Canada and BC (Bothwell *et al.* 2009). Initially it was suspected that it was invasive and spread through movement of boats and sportfisherman travelling between watersheds. Didymo (rock snot) is a microscopic algae (diatom) that attaches to solid surfaces (i.e., rocks) and can grow into polysaccharide stalks that form thick mats on stream beds. When growth conditions are favorable, colony expansion allows stalks to coalesce, forming thick, gelatinous masses that can smother pristine, rocky-bottomed rivers.

Blooms of Didymo have recently been observed in the Lower Puntledge River (Guimond pers. comm. 2008) as well as heavier blooms in the Upper Puntledge River indicating that phosphorous is low (Lough pers. comm. 2014; Figure 70).

Didymo studies in New Zealand (2004) found that the prevalence of didymo was related to dissolved phosphorus concentration (Bothwell *et al.* 2014). The identification of very low soluble reactive phosphorus (SRP; below ~2 ppb) was the proximate cause of bloom formation. This has led to the more likely explanation that *D. geminata* blooms are the result of large-scale human intervention in climatic, atmospheric, and edaphic processes that favour this ultra-oligotrophic species. In this new view, blooms of *D. geminata* are not simply due to the introduction of cells into new areas. Rather, bloom formation occurs when the SRP concentration is low, or is reduced to low levels by the process of oligotrophication. Mechanisms that potentially cause oligotrophication on global and regional scales are identified in Bothwell *et al.* (2014).

The potential decrease in phosphorus (P) on Vancouver Island appears to be associated with logging. At a local scale, clear cutting can have a similar effect by accelerating the rapid growth of the understory which also increases nutrient and in particular P uptake. P uptake can be exacerbated by applying urea fertilizer on forest lands to increase the rate of timber regeneration. Nitrogen added to soils results in tighter binding of P in landscapes. Mechanisms vary but this might include mycorrhizae stimulated uptake of P (Bothwell 2008). These factors and treatments have the net effect of increasing growth and thereby P uptake by terrestrial plants which may reduce P in neighbouring rivers (i.e., < 2ppb). Phosphorus leaching from N-fertilized landscapes results in oligotrophication and subsequent *D. geminata* blooms.

Figure 69. **Map of the occurrence of Didymo 1988-1999 (Source: Bothwell *et al.* 2009).**

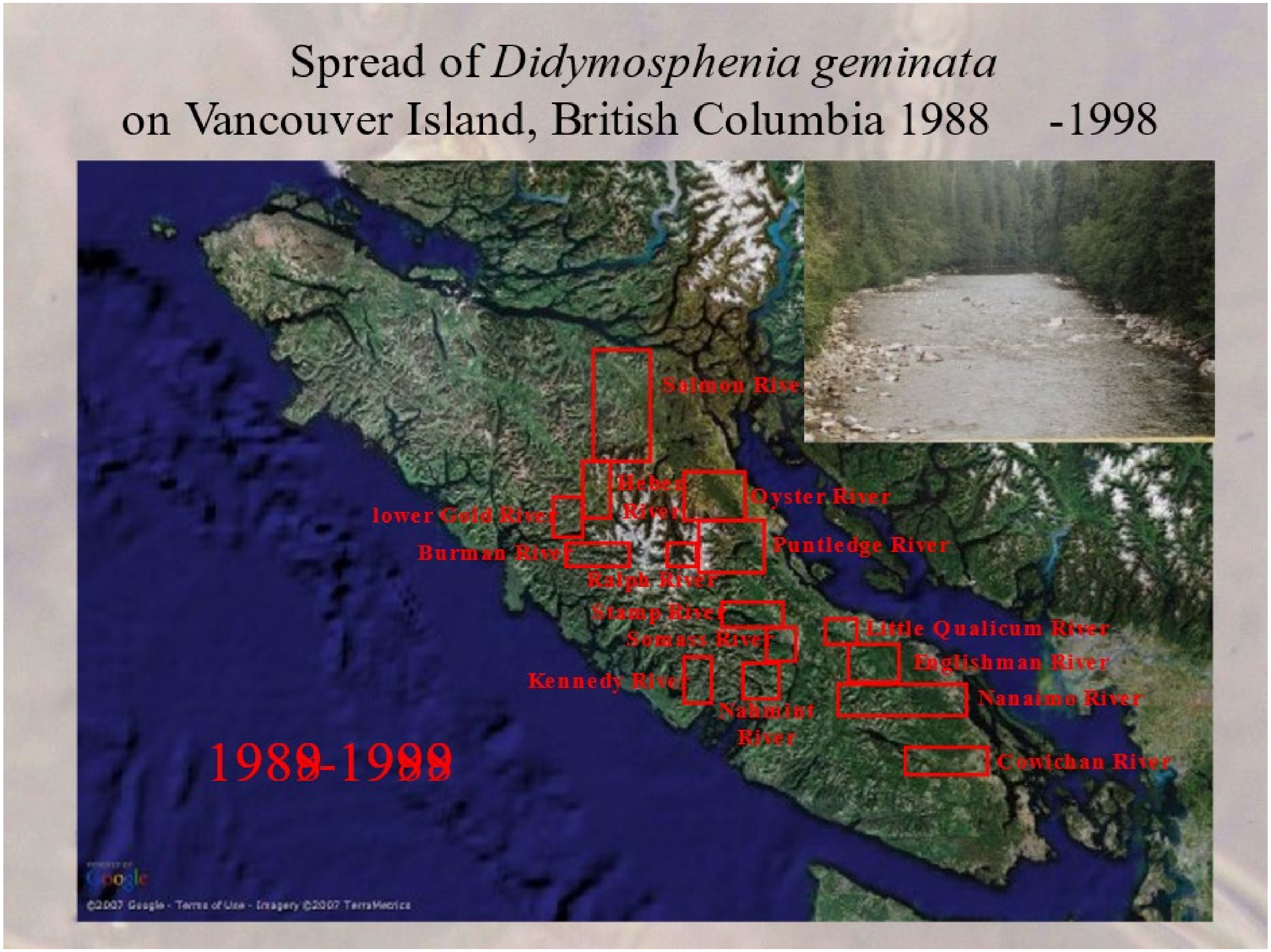


Figure 70. Didymo in the (a) Upper Puntledge River (Source: Lough 2014) and (b) the Lower Puntledge River in Reach C, Barbers Hole during February 2008 (Source: Guimond and Burt 2008).

a)

b)

On a global scale, increases in greenhouse gases and global warming can lead to an earlier growing season, accelerate terrestrial plant growth and an increased uptake of Carbon, Nitrogen, Calcium and Phosphorus. Changes in pollution and climate may be driving the Didymo explosion. The release of nitrogen from fossil fuel combustion eventually gets incorporated into the ground fertilizing and promoting the growth of plants and consumption of other nutrients as well, including phosphorus. Terrestrial plant growth depletes phosphorus that would have otherwise entered rivers and streams. Less phosphorus in the water may now give rise to more Didymo algal blooms (Taylor 2014). This likely has a negative effect on zooplankton and kokanee productivity in Comox Lake, which is already oligotrophic, and productivity in the lower Puntledge River. Furthermore, lakes like Comox, with a corresponding high annual flushing rate (i.e., one exchange per year) typically have low P (Guimond *et al.* 2014).

New research now links climatic warming and the recession of glaciers to the appearance of *D. geminata* blooms in British Columbia (Brahney *et al.* 2021). The input of glacial meltwater to streams maintains unique habitats and supports a diversity of stream flora and fauna. In western Canada, glaciers are anticipated to retreat by 60–80% by the end of the century, and this retreat will invoke widespread changes in mountain ecosystems.

Using a set of streams and rivers in the Upper Columbia River Basin (UCRB) in southeastern British Columbia, a space-for-time substitution approach was used to determine the impact of receding glaciers and reduced glacial meltwater input on *D. geminata* abundance and bloom formation in streams. Rivers in the UCRB were categorized according to the areal extent of glacier coverage in their watershed: (1) heavily glaciated watersheds with >5% areal coverage, (2) transitional watersheds with <5% glacier coverage, and (3) non-glaciated alpine watersheds with streams fed solely by snowmelt. In surveys conducted during the late summer *D. geminata* was completely absent from streams in heavily glaciated watersheds while cells and occasional colonies were observed in transitional watersheds. In stark contrast, *D. geminata* cells and blooms were common in alpine watershed streams without glacier input.

The addition of glacial meltwater to streams does three things: (1) it lowers water temperature, (2) it increases instream shading by adding glacial flour, and (3) it increases the SRP concentration. Since these tend to limit *D. geminata* stalk formation, declining amounts of glacial meltwater in streams make conditions more favourable for bloom formation. Furthermore, with climatic warming, glacial meltwaters enter streams earlier in the year and prior to summer solstice. Therefore, the window of maximum solar radiation (intensity and duration) is now coinciding with lower SRP and higher temperatures. The phenological shift of light, temperature, and SRP in streams caused by reduced and earlier inputs of glacial melt water is hypothesized as a possible mechanism for *D. geminata* bloom formation now being widely observed in British Columbia rivers (Brahney *et al.* 2021). A study by the University of Northern BC found that between 1985 and 2005, the glacier surface on Vancouver Island dropped from 18.2 square kilometres to 14.5 square kilometres, a loss of 20 percent. The losses since then are estimated to be much higher. Brian Menounos, a professor of Earth sciences at the

University of Northern BC who has extensively studied glaciers on BC's coast estimates all of the Island's ice packs will be gone by mid-century, including the iconic Comox Glacier (Kloster 2021).

D. geminata has been observed immediately downstream of the two hydropower dams on the Puntledge River. The key predictor variables of *D. geminata* abundance included dam presence, water clarity, and total phosphorus concentration. The role of dams in the abundance and blooming of *D. geminata*, was investigated at two major headwaters of the South Saskatchewan River Basin (SSRB), Alberta, Canada; including sites just below dams compared to unregulated upstream reference sites in six dammed rivers of the SSRB (Kirwood *et al.* 2006). There was a high degree of seasonal variability in *D. geminata* abundance among sites, but statistical analyses showed a significant propensity for the diatom to have higher cell densities and an increased frequency of blooms at dam sites. This may be the result of a more stable flow regime typical of dam operation resulting in more favorable conditions for Didymo growth and mat formation. Due to the ecological and aesthetic concerns regarding the global spread and blooming of *D. geminata*, it was recommended that dams be viewed as key candidates for mitigating blooms in flow regulated systems and that the threshold discharge required to control Didymo be investigated.

In New Zealand and worldwide, there was limited understanding how *D. geminata* biomass influenced higher trophic levels (e.g., invertebrates and fish). The effect of *D. geminata* biomass on benthic invertebrates, invertebrate drift, and fish communities in 20 rivers in New Zealand was examined with variable hydrology, physical habitat and water chemistry (Jellyman and Harding 2016). Analyses of biotic responses showed that high *D. geminata* biomass did not affect either invertebrate or fish diversity but altered the structure of benthic communities and changed the composition of drifting invertebrate communities (Jellyman and Harding 2016).

The percentage of [Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)] taxa (i.e., EPT) in the drift have been shown to rapidly decline with increasing *D. geminata* biomass while the percentage of Chironomidae increased (Hayes *et al.* 2000). The negative relationship between drift propensity and *D. geminata* biomass indicated that at high biomass sites, fewer invertebrates (relative to benthic density) were present in the drift, although the total biomass of drifting prey available to fish was unchanged across the *D. geminata* biomass gradient. The quantity of drifting food for fish may have remained relatively constant; however, the mean size of individual prey items decreased with increasing *D. geminata* biomass. Declining prey size can have significant implications for fish bioenergetics, particularly of drift-feeding larger salmonids (Hayes *et al.* 2000).

The suspension of Didymo particles, which have been described as sharp silica fragments, has been observed during high flow events in Puntledge River. It is speculated that when benthic algae like Didymo (*Didymosphenia geminate)* blankets the river bottom, particularly during the summer months when growth can be high, increases in discharge can dislodge and suspend Didymo fragments. These sharp siliceous fragments are reported to irritate the eyes of swimmers and the gills of fish. Jensen

(2006) reported high summer Chinook adult pre-spawning mortality in circular tubs that were using pumped river water at Lower Puntledge Hatchery. Didymo fragments are a concern during Chinook migration where adults are already experiencing stressful high temperature conditions. The added stress caused by suspended material could potentially lead to mortality. It is recommended that this concern be assessed and that penstock shutdowns and high increases in Reach C discharge be avoided during summer Chinook migration. DFO staff have not visually observed impacts of suspended Didymo from summer pulse flows in Reach C; however, this should be investigated.

4.1.16. Unfavorable Water Temperatures

Migration

Summer Chinook have a life history that makes them particularly vulnerable to pre-spawn mortality as they arrive in the river in June-July when stream temperatures are increasing. High water temperatures between June and August can delay summer Chinook migration. Stressful conditions for anadromous salmonids begin at temperatures greater than 15.6°C with lethal effects occurring at 21°C (Sauter *et al.* 2001). Puntledge summer Chinook arriving in June, can experience temperature increases over 18ºC by the end of the month and temperatures over 20ºC in July and August (Figure 24 and Figure 25). Prolonged exposure to elevated temperatures during migration can increase metabolic rate and deplete energy reserves before fish reach the spawning grounds, which may result in resorption and reduced egg size and increased incubation mortality. Higher stress ultimately results in pre-spawning mortality. The migration success rate from Puntledge Hatchery into Comox Lake ranges between 50 to 70% and is reflective of the rate of pre-spawning mortality. Research from Washington State indicates that increases in summer stream temperatures will potentially result in a 95% decline in spring-run Chinook, without intervention via habitat restoration to offset warming temperatures (Fogel *et al.* 2022). Unfortunately, there are no sources of cold water draining into the watershed that can be enhanced. High daily summer temperatures over 20ºC in Puntledge River continue to increase in frequency and will have a greater impact on summer Chinook migration in the future (Figure 26).

The longer-term strategy for sustainable survival of summer Chinook is focused on efforts to encouraging upstream migration into the lake early in the summer during a period when water temperatures are more suitable and higher spring freshet flows remain available. Adults that migrate in late June to mid-July have a migration success rate of 70-90% to Comox Lake. Thus, a key strategy for the hatchery program is to focus on selecting, enhancing, and rebuilding the early proportion of migrating adults, replicating historic timing.

Spawning

In the Puntledge River, average temperatures are around 15.2ºC at the beginning of summer Chinook spawning (early-to-mid October). The optimal temperature for spawning is 10°C. Gamete viability starts decreasing at temperatures above 15ºC. Based on a limited number of observations, it does not appear that Chinook delay spawning for 1-1.5 weeks to avoid spawning above 15ºC (Guimond pers.

comm. 2022). However, in 2022, river temperature was above 18ºC in early October when summer Chinook in Reach B were observed spawning. Based on analysed temperature data from the Lower Puntledge River between 1977 and 2021, on average, the lower river reaches 15ºC in the first week of October (Figure 6). The latest date on record to be greater than 15ºC was October 17th in 1991. However, given that the September-October temperatures of this data set were taken downstream of the summer Chinook spawning area, which is subject to fall atmospheric cooling this time of year, temperatures would likely be higher at the spawning grounds.

Overall, the maximum and minimum temperature range that summer Chinook spawners experience are usually within the recommended range. However, there are occasions during early October when the temperature is still 18ºC but drops to 15ºC by mid-October. It is recommended that temperature is monitored closely during the onset of spawning and more observations are conducted on the timing of the start, peak and end of spawning.

4.1.17. Water Quality Threats

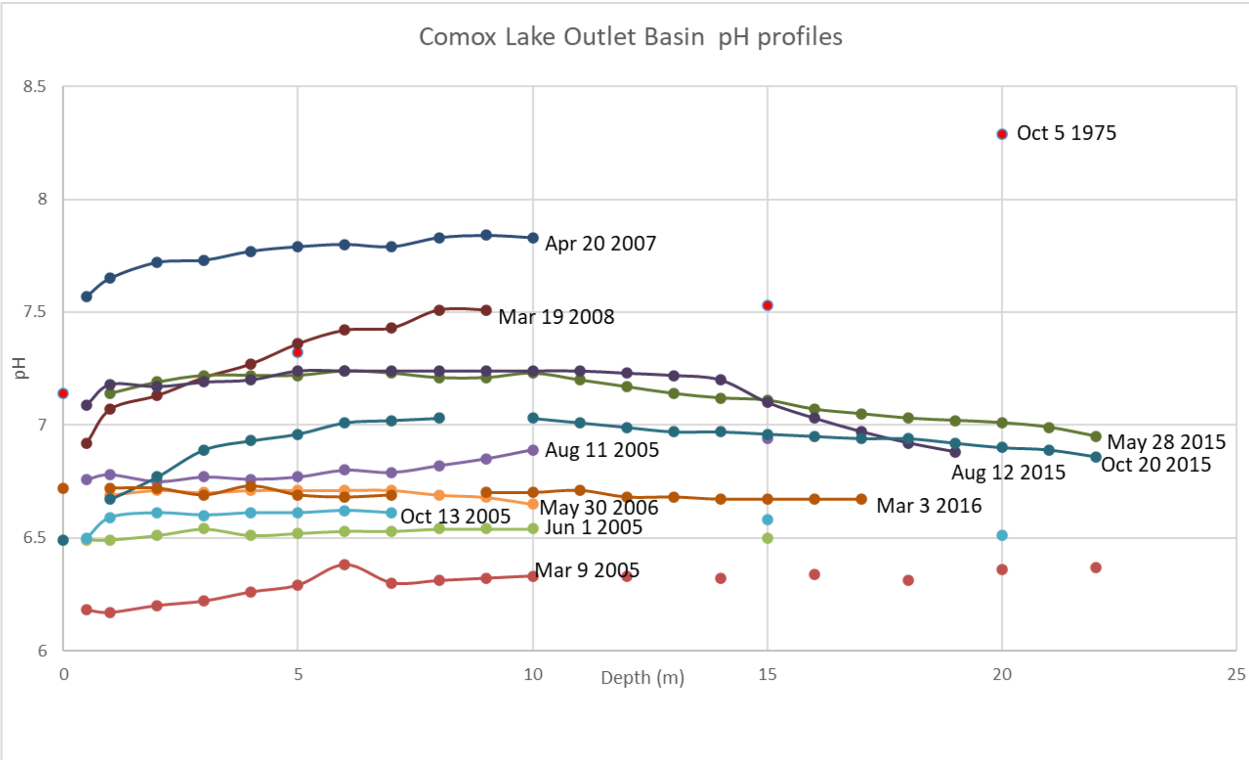
4.1.17.1. Low dissolved oxygen

Low dissolved oxygen is not expected to be an issue for spawners. Based on the temperature data collected historically at the Upper Puntledge Hatchery and currently at the Lower Puntledge Hatchery, oxygen levels in the river are at saturation during adult migration.

4.1.17.2. Poor PH levels

There does not appear to be an issue with pH that would affect summer Chinook in the Puntledge River or Comox Lake. The mean pH at all sampled sites in 2011 was 7.3, except for one low value measured in the outlet basin (6.5 pH units, measured near the surface on October 13 2005), all values were within aquatic life and drinking water guidelines (which allow a minimum pH of 6.5). Deeper water samples measured on October 13, 2005, in the outlet basin of Comox Lake had pH values of 7.2 and 7.4, and it is likely that the low surface water value was a result of heavy rainfall the previous day (rain has a naturally low pH). Higher pH at depth was also recorded in 1975 and has since varied 1.5 units between 2005 and 2016 (Figure 71). There is seasonal variation caused by photosynthesis in in the spring, associated with the mass growth of phytoplankton, increasing pH. Wind mixing during the summer months alters the pH and decomposition of phytoplankton in the fall decreases pH (Krokhin 1962). The pH between 2016 and 2019 averaged 7.17, 7.39, 7.11 and 6.90, respectively. In 2018, alkalinity in Comox Lake averaged 26 mg/l (Barraclough 2019). Overall, natural and anthropogenic activities occurring within the watershed are not likely to have a significant impact on pH and current seasonal levels are not a concern.

Figure 71. **pH depth profile for Comox Lake collected between October 1975 to March 2016 (Source: BC EMS database 2023).**



4.1.17.3. Total Gas Pressure

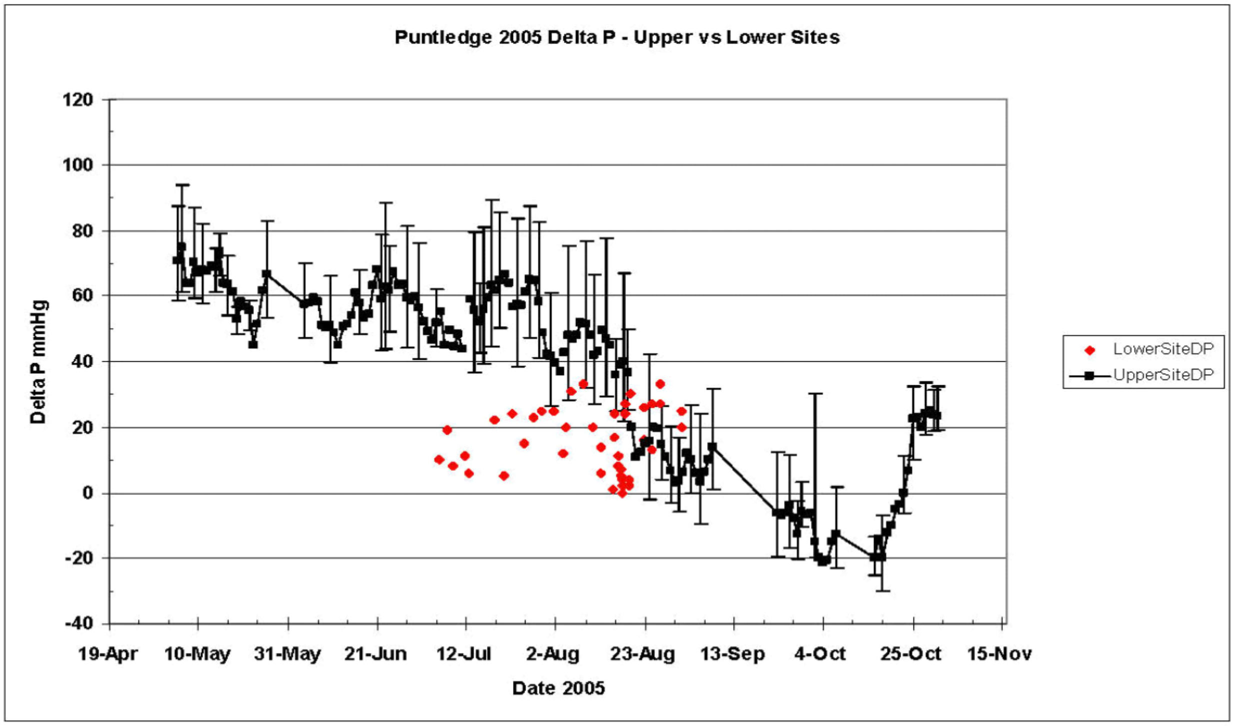
High Total Gas Pressure (TGP) (or gas-supersaturation) can result in gas bubble disease (GBD) and mortality to fish that are exposed for prolonged periods of time. Elevated TGP is caused by the daily solar heating of the lake surface in the spring and summer, with maximum values recorded in the late afternoon or early evening (Jensen *et al.* 2006). Since gas supersaturation is a function of the rate of heating, it is to be expected that warmer years will result in higher TGP and water delivered immediately downstream of the lake or reservoir will present an increased risk to fish.

There is a difference in TGP between the upper and lower Puntledge hatchery areas during the summer months. There is very little potential for degassing through the headpond before the water is diverted into the upper Puntledge hatchery channels due to the low gradient and low velocity of this reach, compared to Reach C where turbulence and a greater surface area of the air/water interface facilitates degassing and decreases TGP. Unlike the lower hatchery, the upper facility is not equipped with an aeration tower to reduce gas supersaturation from the water supply. Thus, fish holding at the upper hatchery site would be exposed to elevated TGP levels during the late spring and summer with daily variations. TGP measured in July through August 2005 at the upper hatchery site yielded an average and peak daily TGP value of 107% and 110% (at BP = 755 mmHg), respectively, compared to an average TGP value of 102% at the Lower Hatchery (post aeration). The difference in TGP between the two hatchery sites is shown in Figure 72.

In 2004, Puntledge River summer-run Chinook holding in raceways at the Upper Puntledge hatchery suffered substantial mortality (>90%); the highest pre-spawn mortality recorded by the hatchery (Munro and Beggs pers. comm. 2022) when only six females out of ~900 spawned. In contrast, a small group of summer Chinook holding at the lower hatchery experienced relatively low pre-spawn mortality even though both groups were exposed to similar high temperatures during the summer (daily average temperatures exceeding 22ºC for a week in August). More typical pre-spawn mortality rates of summer Chinook at the hatchery are between 25% and 30%, and up to 50% in warm years. It was speculated that high temperatures combined with elevated TGP at the upper hatchery was largely responsible for the high mortality.

A reduction in the daily TGP fluctuations was noted when the lower Hatchery switched from the penstock water supply to the river water supply from a nearby pumping station in September 2006. Little degassing occurs in the enclosed penstock and so the maximum daily TGP levels at the inflow to the aeration tower were higher (Jensen *et al.* 2006). Effects of TGP are not a concern for fish in the majority of the Puntledge River due to the aeration tower and because gas supersaturation decreases with increasing hydrostatic pressure (Sigma 1983) and supersaturation dissipates as water moves downstream. For every 1 m increase in water depth, TGP decreases by 10%, therefore, greater water depths in the headpond reach provide fish the ability to avoid exposure to excessive gas supersaturation.

Figure 72. Total gas pressure (ΔP mmHg) at the Upper and Lower sites over the summer of 2005. Daily average, minimum and maximum values (from continuous readings) are shown for the Upper site. Spot checks were made at the Lower site (Source: Jensen *et al.* 2006).



4.1.17.4. Deleterious substances

Sunscreen

Ultraviolet absorbing organic chemicals (UV filters), that are increasingly used in sunscreens and personal care products, can enter the aquatic environment during recreational activities. Many people sunbathe in the lower Puntledge River daily throughout the summer months (see Section 4.1.3). Concentrations of sunscreen chemicals in rivers and lakes can range from a few mg/L to hundreds of mg/L in high use recreational areas. It is known that lipophilic UV filters accumulate in aquatic biota, but little is known about their environmental fate.

A large number of UV filters elicit hormonal effects in fish both in-vitro and in-vivo. For instance, benzophenone-2 (BP2) and 3-benzylidene camphor (3BC) cause feminization in secondary sex characteristics of male fish, alteration of gonads in male and female fish, and decrease in fertility and reproduction. However, in-vitro, and in-vivo studies may not reflect exposure levels found in the field as field data are sparse and highly variable. Most organic UV filters, octocrylene (OCR) and butyl-methoxydibenzoylmethane (BMDBM), have a low water solubility leading to partitioning to suspended solids and deposition into the sediment. While OCR is relatively photostable and considered poorly biodegradable, BMDBM is susceptible to photodegradation and thus biodegrades when exposed to sunlight (Duis *et al.* 2022).

There is evidence of UV filter bioaccumulation in Rainbow and Brown trout and researchers have found some UV filters to inhibit algae growth (Garo-Ferrero *et al.* 2012). The hazard and risks to aquatic ecosystems cannot be ruled out for the UV filter 3BC, where histological and reproductive effects have been observed in fish at low concentrations. However, for other types of UV filter compounds like BP1, BP2 and ethyl-4-amino-benzoate (Et-PABA), the environmental risk is rather low based on current knowledge (Fent *et al.* 2008).

Considering the large number of people using sunscreen, and other endocrine-disrupting compounds, as well as hormonally active UV filters that may act additively, further studies on their potential effects on aquatic species biology and physiology are needed. Water sampling during high recreational use in the summer would be informative and characterise the kinds and concentrations of UV filters present. These compounds would mainly impact summer Chinook adults may be present in these areas during June, July, and August during spawning migration. It is likely that most summer Chinook juveniles have migrated to the ocean by late June.

4.2. Incubation

One of the most critical of the life history phases for salmonids is the period that occurs in the stream gravel, from egg fertilization to fry emergence. Incubation survival is affected by a number of density-dependent and independent factors including location of the redd, water quality, gravel quality, and redd disturbance.

4.2.1. Elevated Predation of Eggs and Alevins

Predation on eggs and alevins in Chinook salmon redds is likely not a significant mortality factor compared to other physical and chemical factors discussed below. Cutthroat trout are known predators on the eggs of other spawning salmon, but once the eggs have been deposited and buried, they are less susceptible to predation from fish and birds. However, freshwater sculpins, family Cottidae, were found to feed extensively on sockeye salmon eggs in redds along beach spawning grounds in Iliamna Lake, Alaska (Foote and Brown 1998). The spawning substrate at Iliamna Lake is characterized by large gravel, with large interstitial spaces and few fines, which facilitates the movement of sculpins within the redd. Large sculpins were able to consume up to 50 eggs/hr, 130 eggs over a 7-day period, and it was estimated that sculpins consumed about 16% of the total eggs laid (Foote and Brown 1998). Following the construction of side-channel spawning habitat in Campbell River, a reduction in Chinook egg-to-fry survival from 39.6% in the first year after construction to between 2.5% and 13.9% survival in the subsequent four years was noted and was suspected to be due to sculpin predation on eggs (Anderson and Sheng 2009). A study comparing sculpin predation impacts on Chinook survival in screened (i.e., 15.2 to2 cm dia.) and native gravel (i.e., 15.2 cm dia. to 3mm) found a significantly lower Chinook survival rate ($p < 0.001$) in the screened gravel (i.e., 47.75%) vs native gravel (i.e., 77.25%) when sculpins were present. The screened gravel used in the study was similar in size to the coarse screened gravel substrate in the constructed side-channel habitat at Campbell R. and is suspected to allow sculpin to easily manoeuvre through the gravel to consume eggs and alevins (Anderson and Sheng 2009). The impact of sculpin predation on summer Chinook production in the Puntledge River is not known. Over 80% of the current spawning habitat in Reach B and C has been placed and is composed of screened and washed gravel. However, the gravel composition resembles a more natural mix and is smaller in diameter than Campbell R. Incubation survival estimates at the Supply Creek spawning platform based on hydraulic sampling and recovery of installed test incubators were >95% (Guimond 2006c) There were no signs of in-gravel intrusion by sculpins or other organism. In contrast, sculpins were found in the gravel at side-channels in Campbell R. following similar assessment methods.

Based on the high survival rate found in 2006, predation on egg and alevins are not suspected to be an issue.

4.2.2. Predation by Invasive Species

Predation by invasive species has not been observed and is not known to be an issue.

4.2.3. Redd Disturbance by Humans

Redd disturbance by humans is possibly an issue because of siltation at the Supply Creek site. Run off from road ditches from an adjacent upslope residential development increased following construction of the Supply Creek spawning platform. Turbidity level >100 NTUs were measured during rain events. Fortunately, this appeared to only extend out and impact the first 5-10 metres off the left-bank. Anthropogenic activities causing siltation include upslope urban development, flow regulation BC Hydro, logging the upper watershed (Mosaic), and recreational activity along the headpond pond trail network during early alevin emergence. However, in general water quality is good and turbidity is low during incubation and human impacts are considered low during this phase of life history

4.2.4. Redd Over-spawn

The available spawning area for summer Chinook in the lower Puntledge River is in Reach B, the historic spawning areas for summer Chinook Salmon. The first platform was built in 2005-06 at the Supply Creek confluence with a spawning area of 4,750 m^2 and has an estimated capacity of 950 spawning pairs; the second platform was built in 2021, 135 metres downstream of the Comox Impoundment Dam with a spawning area of 1,874 m^2 and an estimated capacity of 375 spawning pairs. A smaller proportion of summer Chinook (i.e., approximately 10-20% of the escapement) migrate to Upper Puntledge and Cruikshank Rivers, have an abundance of available spawning area relative to the current escapements. Over-spawning by the summer Chinook population is not an issue due to the low escapement (i.e., <400).

4.2.5. Dewatered Redds at Low Flows and Overall Egg-to-fry Survival

Stream discharge plays a significant role in the survival of eggs during the incubation and alevin-rearing stage. One mechanism is that stream flow directly influences hydraulic gradient across a given redd, which affects intra-gravel velocity, in turn affecting oxygen supply to the redd, and removal of metabolic wastes (Wu 2000). Incubation studies on the Cowichan River found that intra-gravel oxygen levels at monitoring sites were positively correlated with flow (Burt *et al.* 2005). Thus, surface flow has the potential to impact incubation survival if it causes oxygen levels within redds to drop below critical levels (the BC guideline for intra-gravel oxygen includes an instantaneous minimum of 6 mg/L and a 30-day mean of 8 mg/L; RIC 1998). In Reach B, BC Hydro regulates the river discharge between 15.6 m^3/s and 30 m^3/s during the spawning incubation period (i.e., early October to March).

In the period of operation since the Supply Creek spawning platform was installed in 2005, which was designed to withstand 1:100 discharge occurrences, there have been no incidences of dewatering at this site. There have been some brief flow reduction events in Reach C due to equipment failure. The minimum flow during the spawning-incubation period is 15.6 m^3/s. The platform is designed to remain fully submerged at this discharge. Incubation survival was assessed in 2005/2006 and 2006/2007 using eyed Chinook eggs buried in Jordan-Scotty incubation cassettes and was excellent (>95% survival) (Guimond 2006c).

Discharge in Reach B is approximately 33 m^3/s when the generation facility is operating at full capacity and minimum flows are released below the diversion dam. However, flows through this reach can be substantially lower when the reservoir is at its lowest point (early to mid-October) and generation is reduced, to ensure that minimum flows in Reach C and Reach D are met, particularly during a dry inflow year. This corresponds to the beginning of the spawning/incubation period, but the duration of these events is typically brief, since the reservoir can fill quickly once fall rains increase inflows (BC Hydro 2003). The spawning platform constructed in Reach B in 2005 was based on a modelling study that used flow releases from Comox Dam over the past 36-year time series and was designed to provide suitable habitat for spawning even during minimum flows (Chilibeck 2004). However, since construction in 2005, the weighted usable spawning area at the Supply Creek spawning platform was below the target of 4,750 m^2, at a discharge of 30-60 m^3/s, 4 times for more than a week and 3 times for more than 3 weeks following October 7th – the start of summer Chinook spawning (i.e., 2002, 2006, 2009, 2012, 2017 and 2022). Discharges ranged between 15.4 m^3/s and 7.1 m^3/s, which resulted in a reduction in usable area of approximately 30%. It is suspected that this reduction currently has little impact on Chinook incubation survival and productivity due to the low escapement; however, these low flow occurrences will increase due to climate change. In 2022, peak discharged remained below 15.3 m^3/s and averaged 10.2 m^3/s until late December.

In 1997, minimum flows in Reach C of the Puntledge River were increased from 2.8 m^3/s (from October 1st to June 9th) to 5.7 m^3/s year-round. This higher minimum flow was retained as a recommendation by the Puntledge Water Use Plan Consultative Committee (PUN WUP CC) to improve fish habitat in the lower river. A two-year study was implemented in 2006 to assess incubation success of salmonid eggs in response to the 5.7 m^3/s minimum flow recommendation (Guimond and Burt 2008). The field study involved monitoring survival of fall Chinook eggs buried in Jordan-Scotty incubators to the fry stage at three sites in Reach C of the Puntledge River – two sites containing native spawning gravel, and a third site where spawning gravel was recently introduced. Environmental parameters monitored during the study included water level, water depth, velocity, water column and intra-gravel water temperature, dissolved oxygen (DO), conductivity, pH, and turbidity. Substrate composition was assessed throughout the study using both visual estimation and Wolman pebble counts. Fall Chinook survival estimates completed in Reach C are a conservative proxy to survival rates in Reach B. Discharge-velocity conditions and gravel quality are not as stable or good as the spawning habitat in Reach B, which is the primary spawning area for summer Chinook.

The study found that incubation survival (i.e., the eyed egg-to-fry stage) of fall Chinook in Reach C ranged from 42.7% to 90.7%. The highest survival occurred at the site with introduced spawning gravel (minimal fines and highly porous gravel) while the lowest survival occurred at the site with the highest quantity of fines (10–15% fines; Guimond and Burt 2008). Inter-gravel DO levels were higher than would normally cause impairment to survival. The average egg-to-fry survival rate for Chinook populations from the literature is calculated at 38% (Quinn 2005), which is much higher than for other Pacific salmon species due to the tendency for Chinook to spawn in large rivers (i.e., large gravel) that are often associated with water flow from a large lake that inherently have more moderated, stable flows (Bradford 1995; Bradford pers. comm. 2023). These results indicate that the current WUP flow regime and minimum flow releases in Reach C provide suitable conditions for the incubation of fall Chinook salmon eggs. However, keep in mind that there is always remanent summer Chinook that fail to migrate above the Diversion Dam and may survive to spawn in Reach C. Summer Chinook Salmon should be affected by low flow similar to fall chinook.

Flow regulation or unexpected flow reductions due to operational problems may often result in dewatering of redds during incubation, during which time egg or alevin mortalities can occur. Under such circumstances, the extent of mortalities is influenced by a number of physical factors including increased or decreased temperatures, drying (desiccation), reduced dissolved oxygen, increased concentration of biotic wastes, and settling of the gravel (Neitzel and Becker 1985). The magnitude and duration of the flow reduction would also be expected to influence egg and alevin survival. In Reach C, unplanned changes in the rate of withdrawal of water from the penstock may result in flows that drop below the mandated minimum flow of 5.7 m^3/s. Target releases for this reach have therefore been increased to 6.2 m^3/s to compensate for such incidents (BC Hydro 2003). Short-duration flow reductions down to 5.1 m^3/s were not considered to have a significant impact on fish values, and this flow was established as the absolute instantaneous minimum flow for this reach.

4.2.6. Frequent and Higher Peak Flows causing Redd Scour

High flows can scour spawning beds, causing mortality of embryos and alevins from displacement from the stream bed, or from mechanical shock if it occurs during sensitive developmental periods. In the Puntledge River, spawning habitat in the lowermost reach (Reach D) was cited as being impacted from accentuated fall-winter freshets following the expansion of the hydro facilities in 1955, combined with a lack of gravel inputs, and was a leading cause to declines in the fall-run Chinook stock (Marshall 1971). In Reach B, where the summer Chinook spawn, the spawning platforms at both the Supply Creek and the new spawning platform just below Comox Dam are designed to be stable at 260 m^3/s. The highest flow recorded since 2005 was ~254 m^3/s (Chilibeck pers. comm. 2022). High flows in Reach B are moderated by the back flooding effect from the diversion dam. The storage capacity of Comox Lake is limited relative to the total run-off of the upper watershed, and the maximum range in normal reservoir operating levels of the lake is 4.83 m (Griffith 2000). BC Hydro manages the storage capacity of the reservoir and release capability of the impoundment dam to offset high inflow events against tributary inflows downstream of the diversion dam (Browns and Tsolum Rivers) and daily tidal variations to reduce flooding in susceptible areas downstream. This regulation has greatly reduced the frequency and magnitude of high flow events in the river. Studies at the Supply Creek platform from 2005 to 2007, where test incubators were installed, attained incubation survival >95% (Guimond 2006c); however, several of the incubators experienced either an accumulation or scouring of gravel that exposed one of the incubators in the first year after construction. In the second year, discharges during incubation were higher and all 21 test incubators were retrieved at a depth similar to the installation depth. It appears that there was more gravel re-distribution at the bottom of the platform in the first year after construction and minimal movement in the second year indicating that the platform stabilized after the first year of operation.

It has been 16 years since the Supply Creek spawning platform (installed in 2005-06) has been assessed for incubation survival and gravel stability. The latest spawning platform installed in 2021 has never assessed. It is recommended that both sites are evaluated.

4.2.7. Variable Lake Water Levels

This limiting factor is not expected to be an issue for Puntledge River summer Chinook salmon that spawn below Comox Dam. Based on radio telemetry studies summer Chinook appear to mainly spawn in the Upper Puntledge and Cruikshank River.

4.2.8. Lack of Groundwater Upwelling on Lakeshore

Summer Chinook have not been observed spawning along the shoreline of any of the lake in the watershed. This limiting factor is not expected to be an issue for Puntledge River summer Chinook salmon.

4.2.9. Lower Quality Spawning Gravel

Summer Chinook salmon mainly spawn in Reach B where spawning gravel quality was assessed in the mid- to late to 2000s. At this time, the quality of spawning gravel was not an issue, but it needs to be reassessed to ensure this is still the case. Summer Chinook are also known to spawn in the upper Puntledge River and the Cruikshank River; however, spawning and incubation quality has never been assessed. Based on the current escapement levels and the knowledge of gravel quality in the system, this is not expected to be a limiting factor. However, it has been 16 years since incubation survival has been assessed at the Supply Creek spawning platform and the platform installed below Comox dam in 2020 has never been assessed.

4.2.10. Increase in Didymo Abundance

Didymo grows in thick mats in several river systems on the South Island of New Zealand, often smothering entire riverbeds. Salmonid eggs deposited in redds, depend on constant water exchange across the riverbed to provide oxygen-rich water for development. Thick didymo mats might restrict the flow of oxygen-rich water into spawning gravels, resulting in increased egg mortality and reduced trout recruitment. Studies indicate that semen, when activated with uncontaminated river water, had an average motility time of 60±21s while in rivers contaminated with *D. geminata* semen achieved a time of 30±12s (James *et al.* 2015). The present study measured hyporheic hydraulic conditions in trout redds with varying didymo cover in the Clutha River catchment, South Island, New Zealand1. Didymo cover had no significant effects on three hydraulic variables (flow into the substrate, hydraulic conductivity and hyporheic oxygen concentration). However, there was a significant difference in the potential surface water–groundwater exchange between sites, suggesting some effect of didymo on hydraulic conditions. Considering the limited number of replicates, the impact of didymo on trout redds in the Clutha River cannot be excluded. The present study highlights the need for further research on the possible effects of didymo on important surface water–groundwater exchange processes (Bickel and Closs 2008).

The presence of high levels of Didymo will certainly impact the performance of the Eicher screens and the likelihood of impingement of Chinook fry on the screens. BC Hydro staff have reported that Didymo is extremely difficult to clean off the screens. Increasing cleaning cycles, if initiated immediately after the screens have been initially cleaned in time for Chinook fry emergence, may help maintain the screens but the longer Didymo has been allowed to accumulate on the screen, the less likely it is to be swept off during a cleaning cycle. Flushing of the riverbed prior to Chinook fry emergence, which naturally occurs during the winter months, could help remove Didymo matts that formed the previous summer and reduce Eicher screen fouling (Gillis *et al.* 2018). Avoiding high discharges during fry emergence and migration would minimize screen fouling and the risk of fry impingement. Monitoring the pressure gauges up and downstream of the screens and inspection of the screens through the viewing ports could inform staff when the risk of impingement is high due to Didymo buildup and that flow through the penstocks needs to be reduced to maintain screen efficiency. It was also reported that growth of Didymo occurs during the winter months and that thick mats were present during an incubation study on Puntledge River in 2008 (Guimond and Burt 2008). There were concerns with emergent fry getting exposed to Didymo and potential for suspended fragments impacting fry during emergence, migration, and rearing.

4.2.10.1. Increased Spawning Superimposition by Coho Salmon

Coho salmon enter the Puntledge River from September until mid-January with peak spawning and migration occurring around November (Griffith 2000). The principal spawning area is Morrison Creek and the adjacent Tsolum River watershed, with some spawning in the Puntledge River mainstem. It is also suspected that Coho may have had a historical presence in the upper watershed prior to 1912, being able to ascend Stotan and Nib Falls in some years, though there is little evidence to support this (Bengeyfield and McLaren 1994). The later migration timing of Coho combined with the improved passage at the falls and limited spawning habitat available in Reach C and B may increase the risk of over-spawning by Coho and may result in lower egg-to-fry survival rates on Chinook. The superimposition of Chinook redds by Coho spawners may be influenced by Coho broodstock collection procedures and the operation of the lower hatchery barrier fence, as well as the hydrological conditions encountered on the spawning grounds. However, hydraulic sampling of salmon redds on the Supply Creek gravel platform did not find any signs of Coho spawning (Guimond 2007c). This may have been a result of a low Coho escapement year.

The Puntledge Hatchery currently enhances Coho and collects adults at the lower facility. A portion of the adults arriving at the barrier fence (considered surplus) are allowed to continue their migration through Reach C. During past assessment programs and video monitoring at the dams, Coho have been observed migrating into Comox Lake. However, large numbers have also been observed spawning in the headpond and below the diversion dam in areas previously utilized by Fall Chinook. In summary, based on spawning surveys in the Headpond reach and the occurrence of coho redds during incubation field assessments at the Supply Creek spawning platform superimposition by coho does not appear to be a problem.

4.2.11. Unfavorable Water Temperatures

The rate of egg development is primarily affected by temperature, and secondly, by dissolved oxygen. Higher water temperatures result in earlier hatching and emergence from the gravel redds. Aside from altering the rate of development to a certain degree, exposure of developing embryos to very low or very high temperatures can be lethal. However, suboptimal temperatures are not likely a significant cause of egg mortality in most cases when compared to other variables that can influence incubation success. In-situ studies on Chinook embryo survival found that survival was high at 5ºC, 8ºC, and 11ºC, moderate at 14ºC, and poor at 2ºC (Murray and McPhail 1988). It was concluded that 4.5ºC to 14ºC is the acceptable range of temperatures for normal embryo development.

In the Puntledge River, average temperatures are around 15.2ºC at the beginning of summer Chinook spawning (early-to-mid October) (Figure 6). In 2022, river temperature was above 18ºC in early

October when summer Chinook in Reach B were observed spawning. However, given that the September-October temperatures of this dataset were taken downstream of the summer Chinook spawning area, which is subject to fall atmospheric cooling this time of year, temperatures would likely be higher at the spawning grounds, which would potentially impact gamete survival. In future, priority should be given to maintaining temperature logger and updating the dataset from the historic upper hatchery site.

In a study to assess the effects of elevated stream temperature on egg/alevin mortality, Jensen (2006) incubated fall-run Chinook salmon eggs under a declining temperature regime (simulating decreasing fall temperatures). Results showed an increase in mortality of 4% when eggs were exposed to the maximum recorded water temperatures compared to eggs exposed to average recorded temperatures. Since both fall and summer Chinook spawn at the same time, the influence of elevated temperatures on egg incubation would likely be similar (Section 3.1).

Low water temperatures data in the lower Puntledge River was provided (Sweeten pers. comm. 2022), which cover the period of 1977 to 2021. During this period, temperature in the lower river dropped below 2ºC in 1978 from January 4-9th and in 2017 on January 18th. The lower limit of 3ºC was exceeded in 2014, 2017, and 2019 for a single day; in 1977 and 1989 for 2 days; in 2014 for 3 days; and in 1978 for 17 days. Water temperatures in Reach B, where the summer Chinook spawn, are higher than the lower river during the winter period due to atmospheric cooling from the lake outlet to the lower river. The low temperature threshold is currently not an issue and not likely to be one in the future due to climate change trends.

Overall, the maximum and minimum temperatures that summer Chinook experience is usually within the recommended range. However, there are occasions during early October when the temperature is 18ºC, though it drops to 15ºC by mid-October. Jensen (2006) estimated a reduction in incubation survival of 25% when eggs are exposed to water temperatures above 20ºC.

4.2.12. Water Quality Threats

4.2.12.1. Low Dissolved Oxygen

Dissolved oxygen (DO) is an essential parameter for embryo survival and development. The DO requirements of embryos vary with the stage of development, increasing steadily from fertilization and reaching a peak just prior to hatching (Alderdice *et al.* 1958). After hatch, alevins can move more freely through the gravel and pump water with their gills. Exposure to low DO levels can result in numerous adverse consequences for the embryo, such as reduced growth and rate of development that can impact the timing of hatching and emergence (Alderdice *et al.* 1958; Shumway *et al.* 1964). Low DO exposure can also result in lower survival to emergence, as well as reduced growth and survival of emergent fry (Mason 1969). Although DO may be the critical parameter, it is the function of the hyporheic environment to deliver the oxygen to the embryo and remove metabolic waste products (Coble 1961). Spawning gravel containing high levels of fines has also been demonstrated to adversely affect the survival of salmonid eggs and alevins (Chapman 1988). As fine sediments infill the interstitial spaces within the redd, permeability decreases thereby reducing the delivery of oxygenated water to the embryos and removal of wastes and causing entombment of alevins. Several studies suggest that substrates should not contain more than 12-14% of fine sediments smaller than 0.85 mm in diameter for successful incubation (Kondolf 2000). For emergence, the upper threshold of the fine sediment sizes affecting emergence is more variable, and particle sizes of 3 mm, 6.35 mm and 9.52 mm are commonly reported in the literature (CCME 1999). Generally, less than 28-30% of gravels should be smaller than 6.35 mm in diameter (MOE 1998, CCME 1999; Table 40).

Table 40. Incubation requirements of Chinook salmon from various sources.

Parameter	Requirements	Source
Temperature	5.0–14.4 °C	Bell (1986)
		J. Jensen's studies
Intergravel Oxygen	5.0 mg/L	Leitritz and Lewis (1980)
	6.0 mg/L (instantaneous min.)	BC guidelines (RIC 1998)
	8.0 mg/L (30-day mean)	
Substrate Fines:	Maximum Acceptable Levels :	BC Approved Water Quality Guidelines (MoE 1998)
Particles < 2 mm	10%	Canadian Water Quality Guidelines (CCME 1999)
Particles < 3 mm	19%	
Particles < 6.35 mm	25%	

From the fall of 2007 to the spring of 2008, a field study was conducted monitoring three sites in Reach C (i.e., pipeline crossing (Site 1) located 500 m downstream of the Browns River confluence, Bull Island Side-channel (Site 2) located 300 m upstream of Stotan Falls, and Barbers Hole (Site 3) located 250 m downstream of the Puntledge Diversion Dam). These sites are downstream of the summer Chinook spawning area and subject to higher shear force and sediment inputs. Intra-gravel DO was measured using in-situ OxyGuard probes buried at each microsite. Average DO at the three sites ranged from 11.2 mg/l to 11.7 mg/l for egg to hatch and 11.8 mg/l to 12.4 mg/l for hatch to fry stage. Incubation survival from eyed egg to fry for Sites 1, 2, 3, and the control were 82.3%, 90.7%, 42.7%, and 98.7 %, respectively, indicating that intra-gravel oxygen was near saturation and survivals were generally high.

Intra-gravel oxygen has not been monitored at the Supply Creek spawning platform for summer Chinook. However, overall egg-to-fry survival in Jordan cassettes at the Puntledge Headpond Supply Creek was 96.4% in 2005 and 95.6% in 2006, suggesting that intra-gravel levels are high. The latest spawning platform installed below Comox Dam has never been assessed. It is assumed that intra-gravel oxygen and incubation survival is high at both platforms; however, it is recommended that these variables are reassessed. It has been 17 years since the Supply Creek platform has been assessed.

Overall, based on oxygen measurements in Reach C and the high incubation survivals in both Reach C and B, the main summer Chinook spawning area, dissolved oxygen levels appear high and are not an issue in the Lower Puntledge River. Oxygen levels in the Cruikshank and Upper Puntledge River have not been assessed and is a data gap but is not likely an issue.

4.2.12.2. Poor pH Levels

A discussion of pH in the Puntledge River and Comox Lake is provided in Section 4.1.16.2. Based on the available information, pH is not expected to be an issue for incubating summer Chinook eggs.

4.2.12.3. Deleterious Substances

A discussion of deleterious substances is provided in Section 4.1.16.4. Based on the available information, this limiting factor is not expected to be an issue for incubating summer Chinook eggs.

4.3. Early Rearing

4.3.1. Elevated Predation

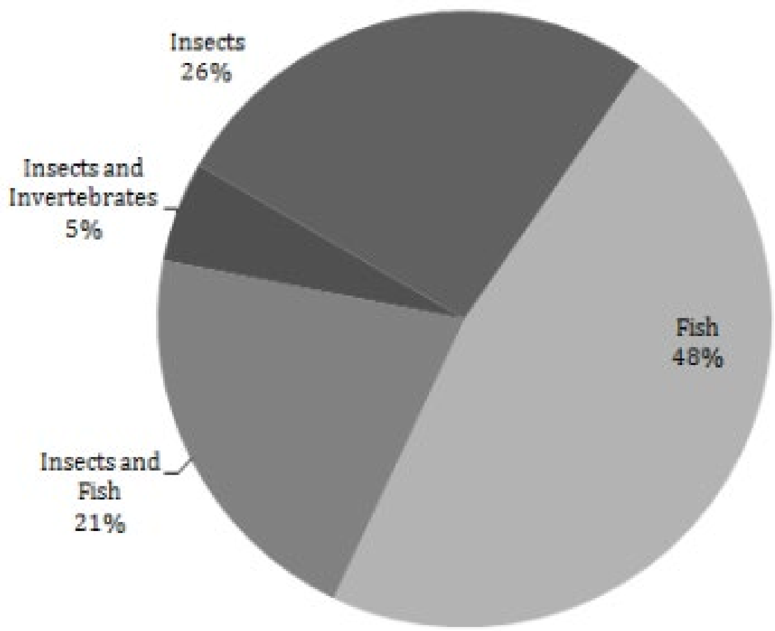
Predation of juvenile salmon appears to be most intense when refuge habitat is lacking, and when river discharge and turbidity are low (Mather 1998), making it easier for predators to find and access prey. Vulnerability to predation is also largely dependent on abiotic factors such as temperature (Hartman 1965), light intensity (Patten 1971; Ginetz and Larkin 1976), tide height (Mace 1983), and cover (Holtby and Hartman 1982). Many animals are known to feed on juvenile salmon, including mammals, fish, and birds (Fresh 1997). A few predators of rearing salmon within the Puntledge River as discussed below but it should be kept in mind that predators may consume salmonids during various life stages (e.g., smolts may be consumed as they move downstream to rear in estuaries, eggs may be consumed by trout or sculpin). The most important predators are detailed below but this is not an exhaustive list of potential predators. For instance, invasive fish species such as pumpkinseed and perch may also prey on juvenile salmon (see Section 4.3.9).

4.3.1.1. Resident Trout Predators

Resident trout are not suspected to be a primary predator on summer Chinook salmonids (Ptolemy pers. comm. 2022). The Freshwater Fisheries Society of BC does not release rainbow or cutthroat trout in the Puntledge Water. The Rainbow and Cutthroat trout population in Comox Lake is estimated to be in the low thousands and estimated in the low hundreds in the lower Puntledge River. (McCulloch and Wightman pers .comm. 2022). A search of the gofishbc.com website fish stock database indicates no hatchery releases of trout in the system.

Trout in Comox Lake are known to primarily feed pelagically on stickleback and nerkid juveniles as well as Coho in May (McCulloch pers. comm. 2022). The Provincial Fish and Wildlife staff analyzed the stomach contents of 19 Cutthroat trout sampled in August 2009 and 2010 and found that almost half of the stomachs sampled contained only fish (stickleback and sculpins) while an additional 21% contained both insects and fish (Figure 73). Just over a quarter of the stomachs contained only insects, and 5% contained both insects and invertebrates (e.g., worms) (Michalski 2011). The stomach content study was conducted in August and predation on summer Chinook juveniles would not be expected this time of year. Based on scale analyses, the summer-run Chinook exhibit an ocean-type life history, migrating out of the watershed by mid summer so would not be expected in the lake in August.

Figure 73. **Percent of fish (stickleback and sculpins), insects, insects/invertebrates, and insects/fish, found in the stomachs of cutthroat trout captured in gillnets in Comox Lake in August 2009 and 2010 (n=19). Adapted from Michalski 2011.**



No other relevant studies were found in a literature search on monitoring and stomach analyses of rainbow and cutthroat trout throughout the year in specific coastal watersheds in the Salish Sea (Wightman and Ptolemy pers. comm. 2022). However, predation of juvenile Chinook by resident trout was extensively monitored in the Lower Cedar River, Washington between 2006 and 2010 (Tabor *et al.* 2014). Analyses found that 85% of the prey fish consumed by resident trout were sculpins, while the summer diet consisted mainly of insects. Results from one year of the study indicated that resident Cutthroat trout can be important predators of Chinook salmon. For three ages of Cutthroat trout (i.e., <150mm, 150-250 mm, >250mm) during that year, juvenile Chinook salmon represented from 5% to 30% of the combined diet from January to April, while Chinook salmon never represented more than 2% of the January-April diet of any size class of rainbow trout. Similar results were found in 2010, where an estimated 66,000 Chinook salmon were consumed mainly by cutthroat trout, resulting in a rough estimate of 30% predation assuming other sources of mortality were minimal.

Rainbow trout consumed fewer Chinook salmon than cutthroat trout and sizes of Chinook salmon consumed by rainbow trout were generally smaller than those consumed by Cutthroat trout. Earlier sampling in the Cedar River (1995 to 2000) demonstrated a similar trend (Tabor *et al.* 2004). These results potential show differences in habitat preference between the two trout predators. Rainbow trout are often found in riffles and the thalweg of large pools, whereas Cutthroat trout are more common in low velocity habitats (Bisson *et al.* 1988). Juvenile Chinook salmon typically inhabit shallow, low velocity areas such as secondary pools along the edge of rivers edge. Few Chinook juveniles are present in large, deep pools and thus juvenile Chinook may overlap more due to habitat use similarities with Cutthroat trout.

The habitat choices of Chinook salmon fry immigrating into Lake Washington from the Cedar River remain unclear, though this is likely to influence their encounter rates with predators. Chinook migrating through high velocity habitats may increase their susceptibility to rainbow trout encounters and associated predation. During sampling in the Cedar River from 1995 to 2000, predation of Chinook salmon by rainbow trout was observed primarily in those collected in large, deep pools; whereas predation by cutthroat trout was observed primarily in those collected in secondary pools (Tabor *et al.* 2004). Although, it is not known exactly what habitat type the predation occurred, this does provide some preliminary evidence that habitat preferences of trout may influence their predation of juvenile Chinook salmon. The headpond reach of the Puntledge River, which is backwatered by the diversion dam, is deep and slow moving with steep banks and limited instream cover. Emergent Chinook fry would have limited access to shallow, low velocity refuge habitat and could potentially be vulnerable to Cutthroat predators, particularly as river discharge increases and displace the fry. Cutthroat trout present in the Puntledge River headpond reach could have originated from Comox Lake, but the population numbers in this reach are unknown. Cutthroat trout are likely piscivorous and are often observed in the Comox Dam fishway (video surveillance).

Although resident Cedar River trout appear to be an important predator of Chinook salmon fry in the months of March though April, predation of emigrating juvenile Chinook salmon into Lake Washington in May and June appears to be extremely rare. Tabor *et al.* (2014) sampled 292 resident trout in May and June and there were no Chinook salmon found in trout diets during that period. Predation may have been low due to high streamflow conditions in late May to mid June in both 2008 and 2010. However, in collections of resident trout in the lower two kilometers of the Cedar River in May through June 1995 to 2000 during low streamflow conditions, only one Chinook salmon was found in 326 trout samples (Tabor *et al.* 2001). During this period, the author pointed out that juvenile Chinook salmon are probably large enough to effectively avoid predation by resident trout. Additionally, the availability of some types of alternative trout prey (e.g., aquatic insects, sculpin, crayfish, and largescale sucker and peamouth chub eggs) is much higher during this period than earlier in the year.

The effect of streamflow on predation of juvenile salmonids by resident trout and other fish is not well known but the information available does suggest that during fry migration higher discharges decrease the exposure time to predators thereby decreasing predation. In 1998 and 1999 in the Cedar River, predation of sockeye salmon fry by trout and sculpin was much lower in riffles (high-velocity habitats) than in pools (low-velocity habitats). Similarly, predation on sockeye salmon fry by prickly sculpin appeared to be reduced during periods of high streamflow (Tabor *et al.* 2014).

It is speculated that Puntledge summer Chinook fry and juveniles likely experience similar periods of trout predation between February and June and if the trout population is low, predation may also be low (as suggested by Provincial staff). However, there is concern over the availability of submerged LWD or rearing and refuge habitat, which is limited throughout the entire Puntledge River. The local municipality aggressively removes LWD accumulations downstream of the Puntledge River diversion dam due to concerns with potential entrapment of recreational users in the river. This practice could potentially increase juvenile Chinook vulnerability to predation by reducing cover for fish.

4.3.1.2. Coho Smolts

Enhancement that increases Coho smolt numbers in the watershed could potentially lead to predation of emergent or small summer Chinook fry; however, predation on summer Chinook fry by Coho smolts has not been assessed on the Puntledge River. On the Cedar River, Washington, yearling Coho salmon abundance appeared to be an important predator of Chinook salmon fry abundance (Tabor *et al.* 2014). In this study, total predation of Chinook salmon ranged from 42,776 to 99,674 in 2008. Coho salmon may have consumed from 5.1% to 11.1% of the Chinook salmon fry in the river. No predation of Chinook salmon by juvenile Coho salmon was observed in 2010, likely due to small sample sizes and a smaller number of Chinook salmon fry in 2010 compared to 2008 (fry migration to Lake Washington in 2008 was 691,200 while in 2010 115,500 fry were estimated to migrate to the lake).

Feeding trials investigating Coho salmon predation on wild fall Chinook salmon using fish from Cedar and Yakima River, Washington, have been conducted in fiberglass troughs to determine the maximum size of fall Chinook salmon that juvenile coho salmon could, or would attempt to consume (Pearsons and Fritz 1999). Both large and small Coho smolts (i.e., 135–171 mm and 129–149 mm) consumed fall Chinook salmon that were between 40% to 46% of their length, but generally consumed Chinook fry that were 40% of their lengths.

Coho smolts enumerated at the BC Hydro diversion dam Eicher Screen Facility, located downstream of the main summer Chinook spawning grounds, recorded a range in Coho smolt size between 80 mm to 145 mm for 1+ smolts (Guimond and Taylor 2015, Figure 74) and an average length of approximately 100 mm for brood years 2010 to 2015 (Guimond and Taylor 2015). In a field study conducted in the 1990s, Coho smolts ranging in size between 75 mm and 100 mm were recorded consuming between one and three Chum fry per day in a groundwater-fed side channel in the Squamish area (Sheng *et al.* 1990; Figure 75). The modal length of the chum fry measured at the same location in an earlier study was 38 mm (Lister *et al.* 1980), which is similar in size to the Puntledge summer Chinook fry enumerated at the sampling facility. Based on this information, the average number of smolts enumerated between 2010 and 2017 with a fork length of 95 mm to 100 mm (in a total population of 35,000), could consume 105,000 Chinook fry per day (the total Coho smolt population was approximately twice this estimate). However, the mean size of Chinook fry is only below 40 mm from February until the end of March.

Figure 74. Length-frequency distributions for coho smolts (1+ and 2+) captured at the Eicher assessment facility over three periods in 2015 (Guimond and Taylor 2015).

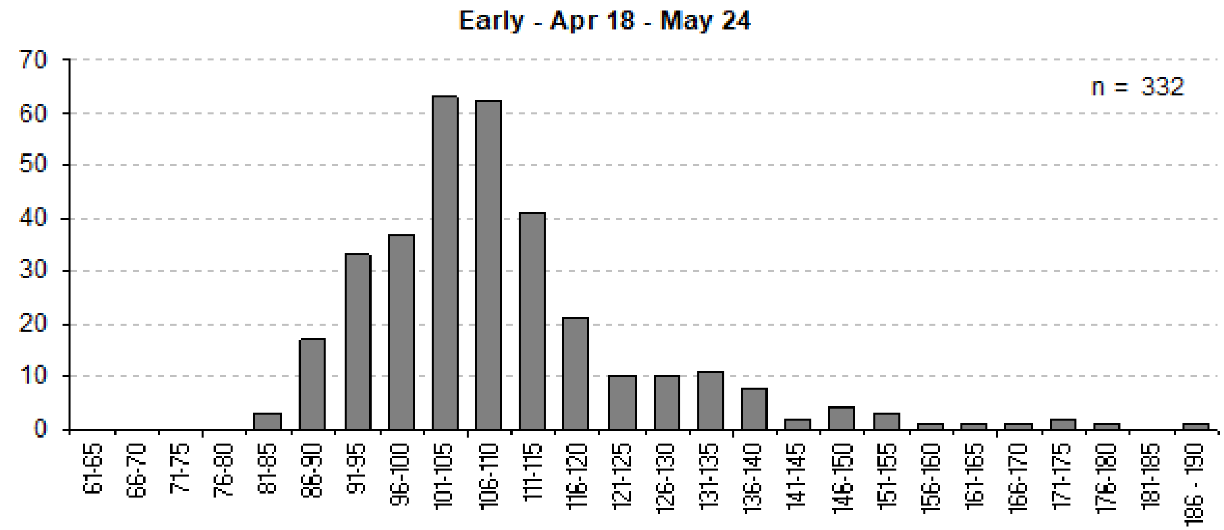
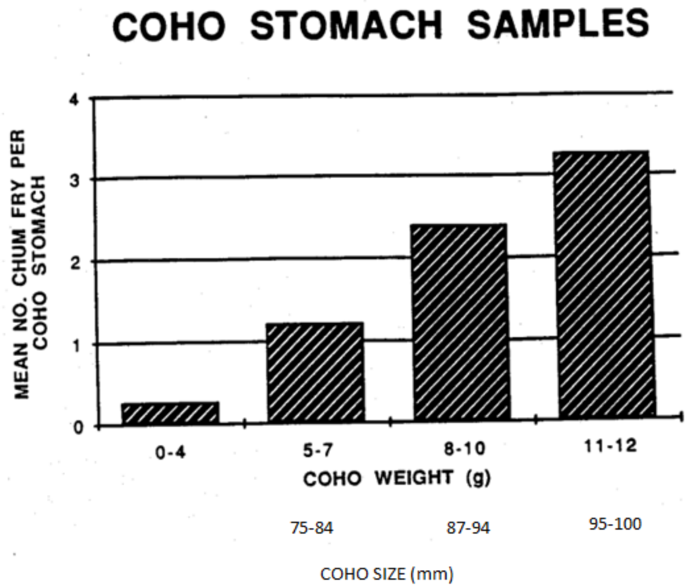


Figure 75. Relationship between juvenile Coho weights and the average number of Chum fry in the stomach, Upper Paradise Channel, May 1985 (Sheng *et al.* 1990).



Coho from three side-channels in March on Vancouver Island had mean fork lengths ranging between 80.0 mm to 82.5 mm (Table 41). Length-weight data collected from Coho juveniles in the Eicher Screen Sampling Facility between February and April 2014 indicated that Coho pre-smolts have the capability to consume Chinook fry (Table 42). Staff observed Chinook fry in the mouths of Coho, which also appeared satiated. If Coho pre-smolts measure between 75 mm to 84 mm in February through March, each juvenile could consume one Chinook fry per day for a total of 35,000 Chinook fry per day in the population. It is presumed that Coho pre-smolts are present and inhabit the Puntledge River headpond where summer Chinook spawn. The Coho pre-smolts are likely encountering the emergent Chinook fry as they migrate downstream seeking out low velocity shoreline habitat, which is also the preferred habitat of over-wintering Coho smolts.

Table 41. **Mean size of Coho juveniles in three side-channels on Vancouver Island in March ([1]Clough 2000, [2]Nitinat Hatchery 2000, [3]Guimond 2000).**

Region	Location	Sample Date	Mean length (mm)	Mean wt (gr)
East Vancouver Island	Nile Creek side channel[1]	Mar 2 2000	82.1	7.1
West Vancouver Island	Caycuse Side-channel[2]	Mar 31 2000	82.5	5.75
Puntledge River	Forbidden side-channel[3]	March 8-10 2017	80.0	-

Table 42. **Mean fork length and weight of Coho juveniles (yearling smolts) sampled at the BC Hydro Eicher Screen sampling centre between February and April 2014.**

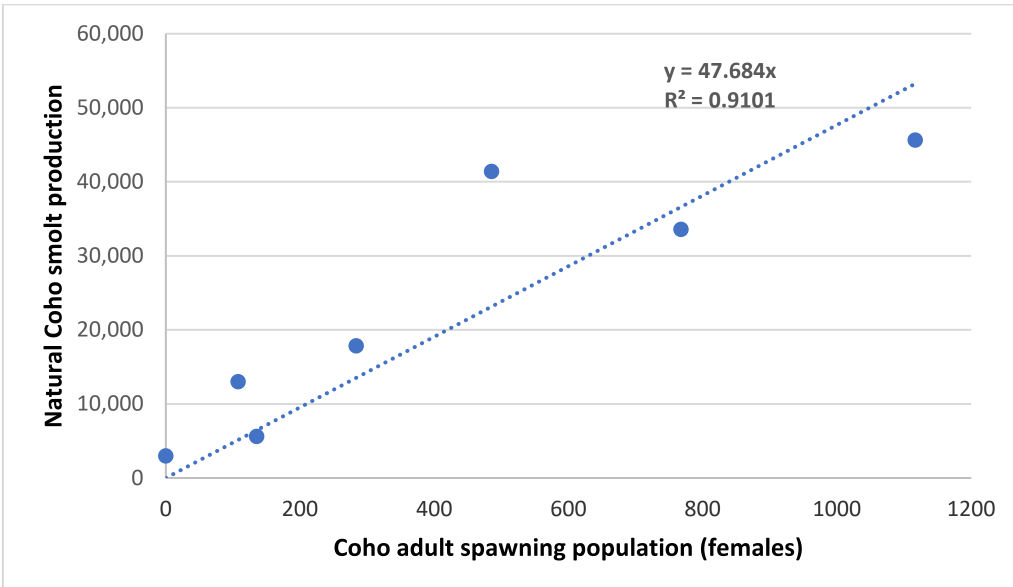
Month	Mean FL (mm)	Mean WT (gr)
Feb	77.9	5.1
Mar	78.2	5
April	97.8	10

The Puntledge River Hatchery has operated a Coho fed fry out-planting program in Comox Lake since 2003 and has been releasing between 46,532 and 1.8 million fry. Releases stabilized between brood years 2011 and 2020 at an average release of 800,000 fry. Based on a mean hatchery fry-to-smolt survival rate of 7.2% estimated for the period 2011-2015, an estimated 50,000 smolts are produced annually from the fry release program.

Adult Coho also migrate into Comox Lake naturally, and the hatchery transports adult Coho from the hatchery to Comox Lake. Transport numbers have been very sporadic over the last 19 years. However, analyses of multiple years of smolt enumeration at the diversion dam sampling center has allowed staff to develop a relationship between the number of Coho adults in Comox Lake (i.e., the number transported by DFO staff and the number that migrated into the lake via the Comox Dam fishway) and the number of Coho smolts produced (Figure 76). Two thousand Coho adults (assuming a 50:50 sex ratio) or 1,000 females produces 47,684 smolts.

In 2021, no Coho fed fry were released into Comox Lake and staff anticipate that no fry will be released into the lake for the foreseeable future. However, hatchery staff still plan to release adults into the lake. It is believed that most of the adults will likely spawn in upper tributary streams such as the Cruikshank and the Upper Puntledge River. This will likely result in more juveniles rearing in these streams and the lake shores at the upper end of the lake, resulting in a lower number migrating into the headpond. However, it would be of interest to conduct minnow trapping in the headpond during Chinook fry emergence (i.e., February-March) to determine if Coho smolts are present in high numbers and if predation is occurring on Chinook fry.

Figure 76. Relationship between the Coho adult spawning population (females) from observations/transports and natural Coho smolt production above the Puntledge diversion dam for brood years 2009 to 2015.

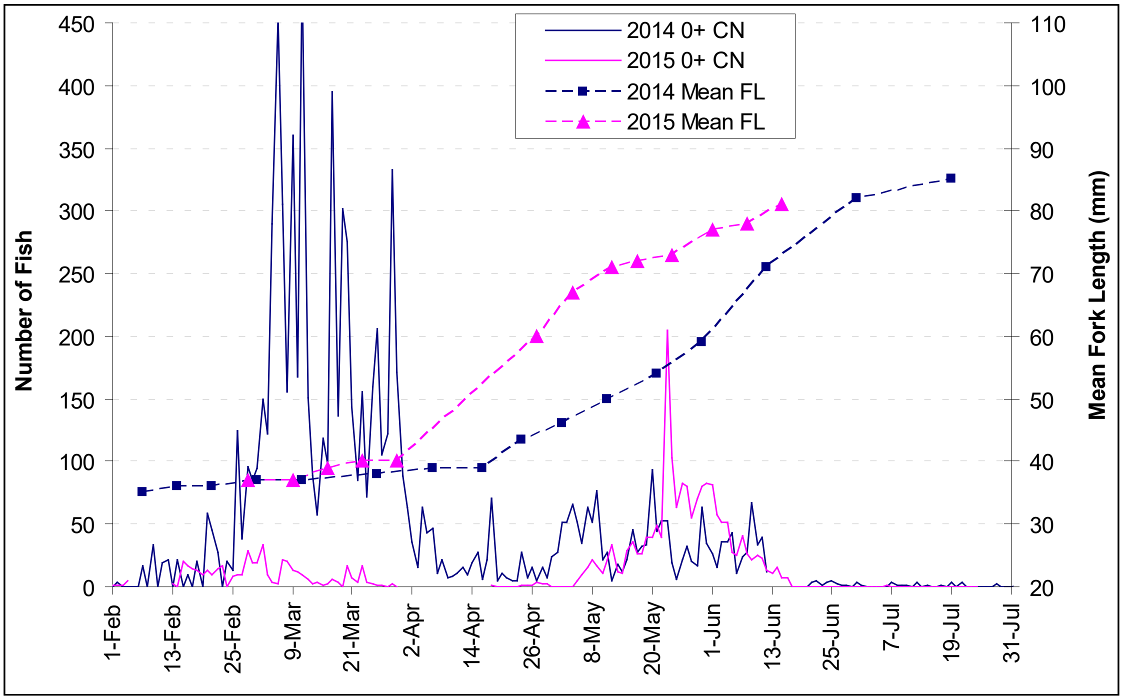


In 2016, after a hiatus of approximately eight years, the Puntledge Hatchery has established a production plan for producing and releasing 100,000 Coho yearling smolts annually. All releases have occurred below the enumeration-broodstock collect fence. The release size has ranged between 15.79 g and 19.25 g or an estimated length of 109.5 mm to 117.1 mm assuming a condition factor of 1.2 (Table 43). The release date is typically the third week in May, which is similar to the natural smolt migration timing in this region (Figure 77). Based on the assumption that Coho predators can consume fish 40% to 46% of their body length, Chinook fry between a maximum size of 43.8 mm to 53.6 mm may be at risk. Based on a 2014 to 2015 study at the Eicher screen sampling center, Chinook fry at the time of smolt release are approximately on average 52 mm or longer. These sizes will vary depending on emergence times and rearing temperatures each year. Furthermore, there is also a risk that there is a proportion of larger size hatchery smolts releases that can consume fry larger than 50 mm (Figure 77).

Table 43. Coho smolt production at Puntledge Hatchery for brood years from 2016 to 2020.

Brood Year	Release Date Start	Release Date End	Total Released	Mean wt at release (g)	Estimated length (mm)	40% of fork length	46% of fork length
2016	25-Apr-18	10-May-18	45,669	15.79	109.5	43.8	50.4
2017	13-May-19	13-May-19	105,465	17.57	113.5	45.4	52.2
2018	25-May-20	25-May-20	89,199	19.78	118.2	47.3	54.4
2019	23-May-21	23-May-21	71,129	18.77	116.2	46.5	53.5
2020	19-May-22	19-May-22	91,279	19.25	117.1	46.8	53.9

Figure 77. Comparison of 0+ Chinook migration timing and mean fork length from the upper Puntledge River in 2014 and 2015.



4.3.1.3. Racoon Predation

Little information could be found on the population of raccoons in the Puntledge River Watershed or on the occurrence of racoon predation of Chinook juveniles. The Pacific Salmon Commission reported on Chinook movement in relation to raccoon presence in 2016 (Pellett 2017). As cited in the Pacific Salmon Commission report: "A total of 114 tags were detected moving upstream in the channel of which 73 originated from the five mainstem tagging locations. The number of tags from each location that were available at the confluence of the Major Jimmy Side Channel and Cowichan River were estimated using the slope of the survival line from Allenby Road array. It was estimated that raccoons interacted with 3% of wild fish and 7% of hatchery fish in the lower river between May 16 and June 24. The Road Pool hatchery release group appeared to be targeted the hardest with a loss of 13.7%. Given that 12 tags from this group were detected and only 1 in 365 fish carried a HDX PIT tag, it was estimated that 4,380 fish from the late hatchery release could have been consumed by raccoons.". The study occurred during an unusually dry year with high numbers of stranded fish, which would provide easy access to fish for racoon predation and thus may overrepresent predation in a typical year (Table 44). Further, only fish crossing arrays were detected, making it hard to quantify predation impacts to the Chinook population without a raccoon population estimate.

Table 44. The impact of low flows during spring 2016 combined with above average predation pressure/stranding is summarized below (Pellet 2022, unpublished data).

Brood Year	Natural Adults	Smolt Year	Age 2	Age 3	Agre 4	Age 5	Total	Adults
2011	2,786	2012	2,313	4,014	4,797	473	11,597	9,284
2012	2,668	2013	668	1,853	3,739	92	6,352	5,684
2013	4,406	2014	887	4,335	4,623	141	9,986	9,099
2014	4,185	2015	2,179	8,439	5,420	115	16,153	13,974
2015	5,984	2016	13,282	10,476	11,415	173	35,173	22,064
2016	7,671	2017	6,445	6,569	6,051	136	19,201	12,756
2017	12,572	2018	3,335	3,905	3,437	125	10,802	7,467
2018	13,975	2019	14,597	11,196	12,200		37,993	23,396
2019	15,103	2020	8,975	7,500				
2020	8,849	2021	6,500					

4.3.1.4. Blue Heron Predation

Great Blue Heron have been placed on BC's Blue List of vulnerable species due to declining populations and sensitivity to human activity. The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) has designated the Pacific Great Blue Heron as Vulnerable (MoE 1998). Two subspecies occur in British Columbia: Ardea herodias fannini breeds along the Coast, whereas A. h. herodias breeds in the Interior (COSEWIC 2008). Both sub-species of Great Blue Heron are Species at Risk under the Forest and Range Practices Act and are Blue-listed in British Columbia. The coastal subspecies, A. herodias fannini, is designated as a species of Special Concern by COSEWIC. The most recent published estimate of population size for the Pacific Great Blue Heron in British Columbia is about 3,600 nesting adults, of which 3,300 were thought to occur in the Strait of Georgia (COSEWIC 2008). Great Blue Heron are also common throughout the Comox Valley (Map 10).

From 2008 to 2018, juvenile salmon predation by heron was studied at nesting sites within 35 km of three British Columbia rivers (i.e., Cowichan River, Big Qualicum River, and Capilano River) by recovering passive integrated transponder (PIT) tags from over 100,000 tagged juvenile salmon (Sherker 2016). Heron fecal surveys recovered 1,205 tags, representing a minimum annual predation rate of 0.3% to 1.3% of all juvenile salmon. Most of this predation (99%) was attributed to 420 adult Pacific Great Blue Herons from three rookeries. Correcting for tags not found because herons defecated outside of the rookeries raised the predation rates to 0.7% to 3.2%, and as high as 6% during a year of low river flow. Predation mainly occurs during chick-rearing in late spring and juvenile salmon account for 4.1% to 8.4% of the Pacific Great Blue Heron chick diet. Smaller salmon smolts were significantly more susceptible to Pacific Great Blue Heron predation than larger conspecifics. The proximity of rookeries relative to salmon-bearing rivers is likely a good predictor of Pacific Great

Blue Heron predation on local salmon runs and can be monitored to assess coast-wide effects of Pacific Great Blue Herons on salmon recovery (Sherker *et al.* 2021).

Chatwin *et al.* (2017) monitored Vancouver Island heron populations between 2013 and 2015. The number of active nests in the two colonies in the Courtenay area (i.e., Point Holmes and Minto Road) decreased from 59 and 11 to 35 and 5, respectively over the three years. Bald Eagle predation is identified as the main causative factor in decreasing heron nest numbers. In general, heron are mainly identified feeding in the lower Puntledge River and in the estuary in shallow slow-moving water (Sherker *et al.* 2021). Hatchery staff and the K'omok First Nation Guardians observe only small numbers of heron at these locations. Thus, it is presumed, based on the low population size, low population estimates of local Heron and study results at the Cowichan, Big Qualicum and Capilano River PIT tag study which indicated a 0.3-3% predation rate, predation by Heron on Chinook is likely low in the Puntledge River and the estuary.

Map 10. Great Blue Heron colonies in the Comox Valley area from 2000-2015 as per the British Columbia Great Blue Herons Atlas (Source: CMNMaps 2022).

4.3.1.5. Merganser Predation

Mergansers rank among the largest (in terms of appetite) and most efficient predators of juvenile salmon; furthermore, they are relatively common and congregate wherever salmon density is high (Wood 1984). Mergansers are common throughout British Columbia, including within the Puntledge Watershed (Frisson pers. comm. 2022; Frank pers. comm. 2022). Hundreds of mergansers have been observed in the Puntledge River headwaters and local smaller lakes during the months of October and November (Bolton pers. comm. 2022). These birds are known to over-winter in lakes and forage in the lower river and at other watershed during the spring (Wood 1984). The vulnerability of summer Chinook in the Puntledge River is inherently dependent on the amount of time juveniles spend in freshwater. While this freshwater rearing duration remains uncertain for the Puntledge River Chinook, observations of juveniles in the nearby Cowichan River and Big Qualicum River suggest juveniles spend two to three months in freshwater during fry rearing and migration (Lister 1978).

Recent observations of mergansers in the Puntledge River from the late winter to early spring period appear to correspond with the timing of Chinook fry emergence (Frisson pers. comm. 2022). Mergansers have been reported in the lower river by First Nation Guardian Crews in recent years ranging between two to three flocks, each consisting of between 20 to 40 birds during the onset of Chum fry migration in March through April and during the hatchery release of Coho and Chinook smolts in May through June (Frank pers. comm. 2022). This observational timing likely covers the peak period when wild summer Chinook smolts are migrating to the ocean. Juvenile salmon may be particularly vulnerable to predation during this time given that they aggregate along a fixed, predictable route (Mather 1998). Mergansers are often observed in the lower river within the tidal zone around the bird viewing platform and where Portuguese Joe fish market use to operate (Frank pers. comm. 2022). This area in the lower river has been heavily altered and simplified by industrial development, likely making migrating juveniles more vulnerable to predation. The riverbanks upstream (for 1.5 km) are also highly modified due to urbanization and likely offer limited habitat complexity and escape refuge for Chinook juveniles.

Foraging efforts of breeding mergansers (i.e., summer) are most likely to occur within proximity of their nests (Covich 1976), given that breeding females must continually return to their nests throughout egg-laying, incubation, and the period when hatchlings are unable to fly (Wood 1984). Wood (1987) reviewed studies investigating the gut contents of mergansers feeding in a wide range of habitats and found that mergansers eat salmonids wherever they are prominent relative to other available prey species. The proportion of salmonids in the diet of mergansers is likely to change spatiotemporally based on the movements of juvenile salmon within the river. For example, White (1938) observed that the proportion of salmonids comprising the diet of red-breasted mergansers in the Margaree River, Nova Scotia, declined from 100% to 73% to 12% for birds sampled from the headwaters, lower reaches, and upper estuary, respectively. These data were consistent with observations of diet among common mergansers on Vancouver Island streams (Wood 1987). Outside of the breeding season, Salyer and Lagler (1940) reported that mergansers wintering on lakes and streams in Michigan tend to congregate on the lower reaches of rivers, as observed on Vancouver Island during the spring and summer when juvenile salmon were out-migrating (Wood and Hand 1985).

Mergansers usually move continuously while foraging and may do so alone or in groups. Although mergansers were commonly observed foraging in close proximity to each other and occasionally along parallel search paths, their feeding rate was no higher under these conditions than when foraging alone (Wood 1985). Coordinated foraging behaviour was most often observed among groups of birds (particularly juveniles) searching along the shoreline of estuaries, though this is rarely seen within rivers (Wood 1985). Coordinated foraging has been observed by K'omoks First Nations fisheries staff who observed fish being pushed to the shore and boiling on the surface prior to being eaten by mergansers (Frank pers. comm. 2022). However, with the exception of these observations, no published evidence was found to support claims that mergansers forage cooperatively (White 1957; Huntingdon and Roberts 1959; Miller 1973; Des Lauriers and Brattstrom 1965). Wood (1984) found that merganser search paths seldom remain coordinated once fish schools have been encountered.

Merganser predation on juvenile salmon will inherently vary across locations due to a range of environmental conditions. In an enclosure study conducted on the Big Qualicum River in 1980 and 1981, fish density, available cover, prior prey exposure to mergansers, and merganser hunger level were all found to influence salmon predation (Wood and Hand 1985). Although merganser foraging behaviour changed in enclosures with increased cover, the capture efficiency of salmon remained similar highlighting the plasticity of merganser foraging. However, the authors noted that undercut banks tended to provide more protection from merganser predation than sunken logs, branches, and debris, which perhaps suggests a potential restoration design to consider to reduce potential impacts of merganser predation where it is prevalent.

Studies on captive mergansers have also provided important insights into the rate of prey consumption, including on juvenile salmonids. White (1957) reported that a tame (but not captive) male merganser consumed 440 g of prey/day or 38% of its body weight, while four other captive birds in his study averaged 380 g of prey/day or 30% of body weight. Daily consumption was estimated to be 225g to 450 g of prey or 20-40% of body weight (depending on sex), based on ingestion rates of wild birds feeding on Coho smolt of known size in captivity (Wood and Hand 1985). Upon reviewing several studies, Wood and Hand (1985) proposed that mergansers could clearly satisfy daily appetites of 400 g at relatively low fish densities (Wood and Hand 1985). In general, a merganser's daily energy gain appears to be constrained by the time required for digestion, rather than hunting performance (Wood and Hand 1985), suggesting there may be a limit to the number of fish a single merganser can consume over a given period. Indeed, Wood and Hand (1985) found that one hour was required to digest a Coho smolt (43 grams), such that daily consumption was unlikely to exceed 500 g over a 12-hour period.

The study by Wood and Hand (1985) also revealed that Coho smolt were eaten more frequently than Coho fry when stocked together in enclosures with common merganser (Wood and Hand 1985).

Based on the contents in stomach samples taken from mergansers, it appears these birds prefer large salmonids over smaller individuals (Salyer and Lagler 1940; Elson 1962; Alexander 1979). From an energetic perspective, this preference for larger prey is understandable given the time required to pursue and swallow prey of each size class, and the probability of successfully capturing an individual that is larger. However, it is also possible that preference for larger prey (such as smolt) relates to their behaviour (i.e., downstream migration), which may make them more susceptible to predation (Wood and Hand 1985).

As merganser predation appears to be depensatory for the population, from a hatchery perspective, losses can simply be mitigated by increasing hatchery production. The counter argument is that increasing hatchery production may increase the recruitment of mergansers, which may in turn have a larger impact on the wild Chinook. In Coastal streams, merganser brood densities tend to be greater in streams with enhancement than in streams with only natural production (Wood 1984). In the summer months, hatchery fish are protected in rearing channels until their release the following spring, such that all merganser predation pressure will be concentrated on wild salmonids (White 1957; Wood 1984). Therefore, wild fish populations in these streams may be subjected to intense predation for many weeks as an indirect consequence of salmonid enhancement (Wood 1984). This is potentially relevant to the Puntledge River as the lower section has a Chinook, Coho, Chum and Pink hatchery facility operated by DFO.

4.3.1.6. Seal Predation

Harbour seals inhabiting the Courtenay River have developed a unique and highly specialized nocturnal foraging behaviour that was first observed in 1993. Seals were found to congregate at the 5th Street and 17th Street bridges, which span the Courtenay River roughly 2 km and 1 km upstream of the estuary respectively and use the light cast from the bridges to silhouette and capture out-migrating salmon fry and smolts (Olesiuk *et al.* 1995). A detailed assessment conducted from March 22 to June 20, 1995 estimated that seals consumed 3.1 million chum fry and 138,000 coho smolts, each representing about 15% of the projected total chum fry and coho smolt production in 1995 (Olesiuk *et al.* 1995). Seals continued to forage on out-migrating Chinook smolts following the end of the coho smolt out-migration, but observations on feeding rates during this period were too limited to estimate predation levels on Chinook smolts. Predation on Chinook smolts was not assessed but was estimated to be ~33% of the annual smolt production. Since that time, lights on the bridges have been replaced and shielded, reducing the amount of light cast, onto the river. This effort reduced seal hunting at night under bridges, although seal predation continues in the river and estuary.

4.3.2. Stress due to Anthropogenic Activities

Human presence could cause stress to juveniles due to noise or light pollution, ATV or boat recreational use, swimming with salmon, camping, horse riding, log handling and salvages, and in-stream construction. However, these anthropogenic stresses, which are assessed separately from hydroelectric impacts (see below) are not expected to be a potential threat for summer Chinook salmon at this time.

4.3.3. Impact of Hydroelectric Development on Downstream Chinook Juvenile Migration

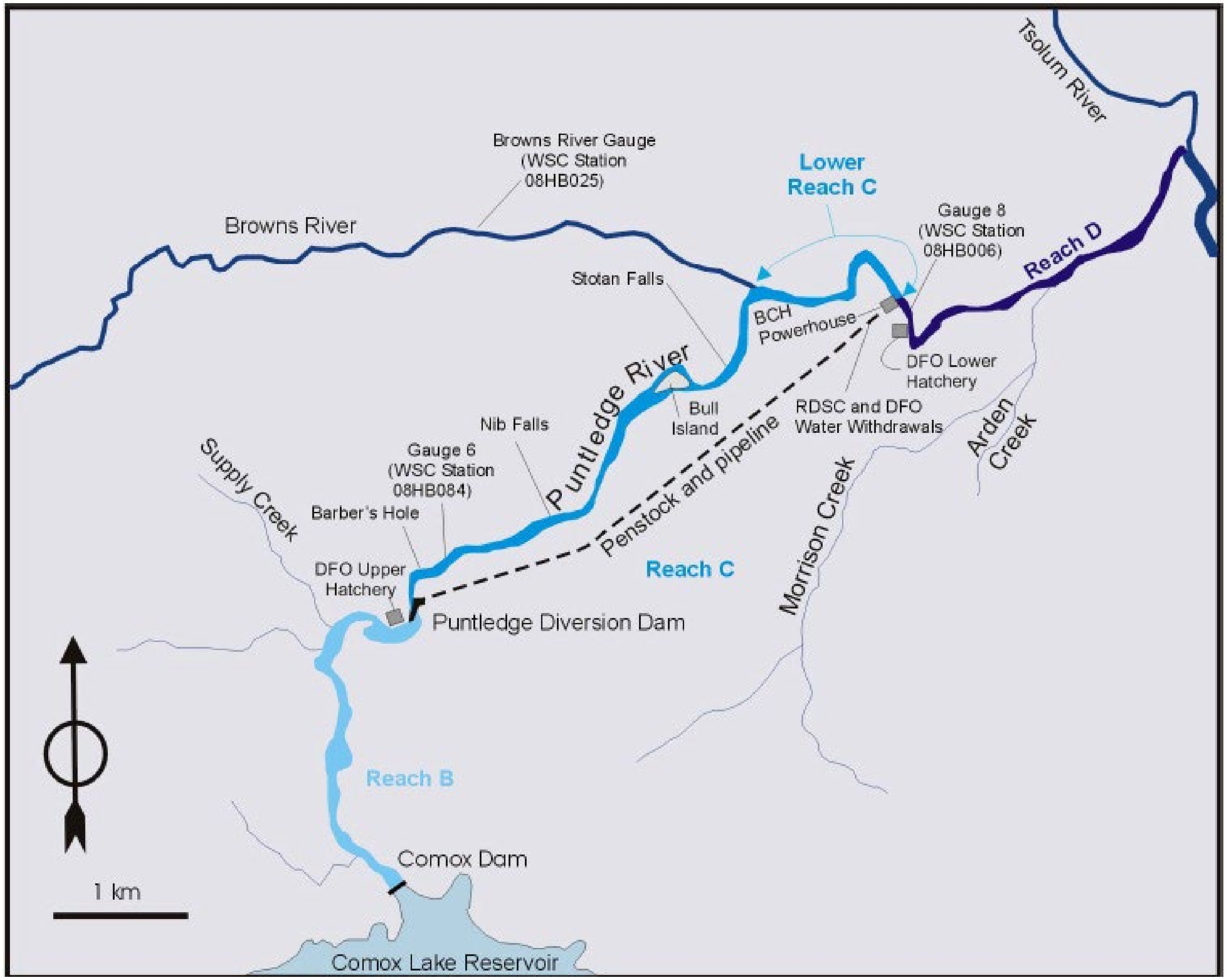
Two key obstacles for summer Chinook juvenile migration are the BC Hydro Diversion Dam and the two power generation intakes, which interfere with downstream migration and entrain a large proportion of the migrants. The Puntledge River flows approximately 6.3 km between the Diversion Dam and the powerhouse (BC Hydro 2003). The current BC Hydro infrastructure includes the Comox impoundment dam, a 3.7 km headpond, the Puntledge Diversion Dam and penstock intakes, a 5 km long woodstave and steel penstock, and a powerhouse with a single 24 MW generating unit that passes about 27 m^3/s at maximum generation (BC Hydro 2003; Figure 78).

Injury and mortality (direct and latent) on fry/juveniles entrained into the penstock and through the Francis turbines have never been properly assessed in the Puntledge River. Francis turbines are responsible for a high fish mortality rate due to severe collisions/blade strike, shear forces, cavitation and pressure decreases (Zhiqun *et al.* 2016; Pracheil *et al.* 2016). Based on assessment at other facilities with similar turbines, the mortality rates from Francis turbines average approximately 30% (Wilson 1962).

Studies were conducted to determine if entrainment at the Puntledge intakes could be minimized. Early abatement measures included using lights and air bubble curtains (1953), fish salvages using fyke nets (1956-1958) (Rimmer *et al.* 1994), and louver type deflectors (1957 and 1959) (Marshall 1972). Unfortunately, none of the early investigations and trials were effective in reducing entrainment.

In spring 1989, BC Hydro tested a combination of mitigation measures including a steel chain curtain, strobe light, and underwater hammer to deter Coho smolts from entering the Puntledge River intakes (Bengeyfield and Smith 1989). The test indicated behavioural devices did not result in a significant difference in proportion of Coho smolts using the intake bypass route. An electrified array (Smith-Root Inc. "Graduated Field Fish Guidance" (GFFG) system) was installed in spring (May 13 to June 22) 1990 to determine if a pulsed electric field could be used to guide Coho salmon smolts to the fish bypass (Bengeyfield 1990). The tests indicated the electrical field did not have a high success rate guiding fish away from the intake (Birch 1992) and this method would not meet today's safety and animal care standards.

Figure 78. Puntledge Generating Station Overview (Source: Puntledge River Project Water Use Plan 2004).



In 1991 and 1992, temporary Eicher screens were tested and estimated to prevent approximately 99% of entrainment of fish into the penstock (Bengeyfield 1992, 1993). As a result, two permanent Eicher fish screens were installed in the penstock at the Puntledge Diversion dam in 1993 (BC Hydro 2003). The screens provide a physical barrier that divert juvenile fish that enter the intakes and returns them to the Puntledge River downstream of the diversion dam. Initially, the screens were intended to operate only during the spring out-migration period, although additional assessment after their installation found juveniles migrating other times of the year.

Function of the BC Hydro Diversion Dam Eicher Fish Screens

Twin intakes at the diversion dam entrain a large proportion of migrating fish from the upper Puntledge River into two smaller penstocks. Each intake is equipped with an elliptical wedge wire Eicher screen oriented at 16.5 degrees to the flow in the penstock with the top of the screen oriented at a shallower angle to aid fish diversion into the bypass pipe (Bengeyfield 1994). As fish approach the screen, they are diverted into a bypass pipe located at the top of each penstock pipe and returned to the river downstream of the dam (referred to as "fish diversion position") (Figure 79). Fish may also pass over the diversion dam during spill events, or through a small spillway adjacent to the intakes (Figure 80; Figure 81).

The Eicher screens operate year-round except during short periods of time for self-cleaning (Figure 82), when they trip open due to pressure increase, and for regular maintenance. The screens are susceptible to fouling from floating debris and periphyton growth (Figure 82 and Figure 83), which can interfere with the hydraulics and efficiency of the screen. In more recent years the build-up of Didymo on the screens has been observed. The screens can automatically rotate into a horizontal non-fish diversion position allowing water in the pentstock to sweep across the screens to aid in the removal of debris. BC Hydro can regulate the frequency of these cycles (usually once a day) so that the screens will automatically rotate from the fish diversion position into the cleaning position every few hours for a duration of approximately 180 seconds per cycle. The screens can also be triggered to cycle out of the fish diversion position by a pressure sensing system, whereby debris build-up on the screens causes a change in pressure beyond specified limits (Guimond and Taylor 2012). When this threshold pressure protection limit is reached, the screens move into a cleaning position. These limits are established to protect the aging wood stave penstock and prevent equipment damage from over pressurization (Tryon 2008). Entrainment of fish into the intakes can occur when the screens are moved horizontally into the cleaning position. As part of the facility maintenance schedule, the penstock is shutdown 2-3 times a year so the screens can be pressure washed. These are usually scheduled early February prior to summer Chinook fry emergence, early spring, and in September.

The spill over the dam is set at a minimum of 5.7 m^3/s to maintain flows in Reach C and typically flow of 12 m^3/s, 20 m^3/s, or 26 m^3/s is drawn through the two intakes to generate 10 MW, 18 MW, or 24 MW of power, respectively, although a wider range of discharges can be diverted for power generation. Approximately 0.5 m^3/s of water is used by each Eicher screen to bypass fish to the river. The operation of the Puntledge Eicher Screens is now a requirement of BC Hydro's *Fisheries Act*

Authorization for the Puntledge Generating Station (PATH#05-HPAC-PA3-00314), as well as a commitment to assess and improve screening efficiency under the Puntledge River Project Fish Entrainment Strategy Action Plan. An annual review of Eicher screen function found that the screens were in the fish diversion position >97% of the year in recent years (Easton 2022).

Figure 79. Schematic of Eicher Screen- Top and Cross Section View (modified from Guimond 2012).

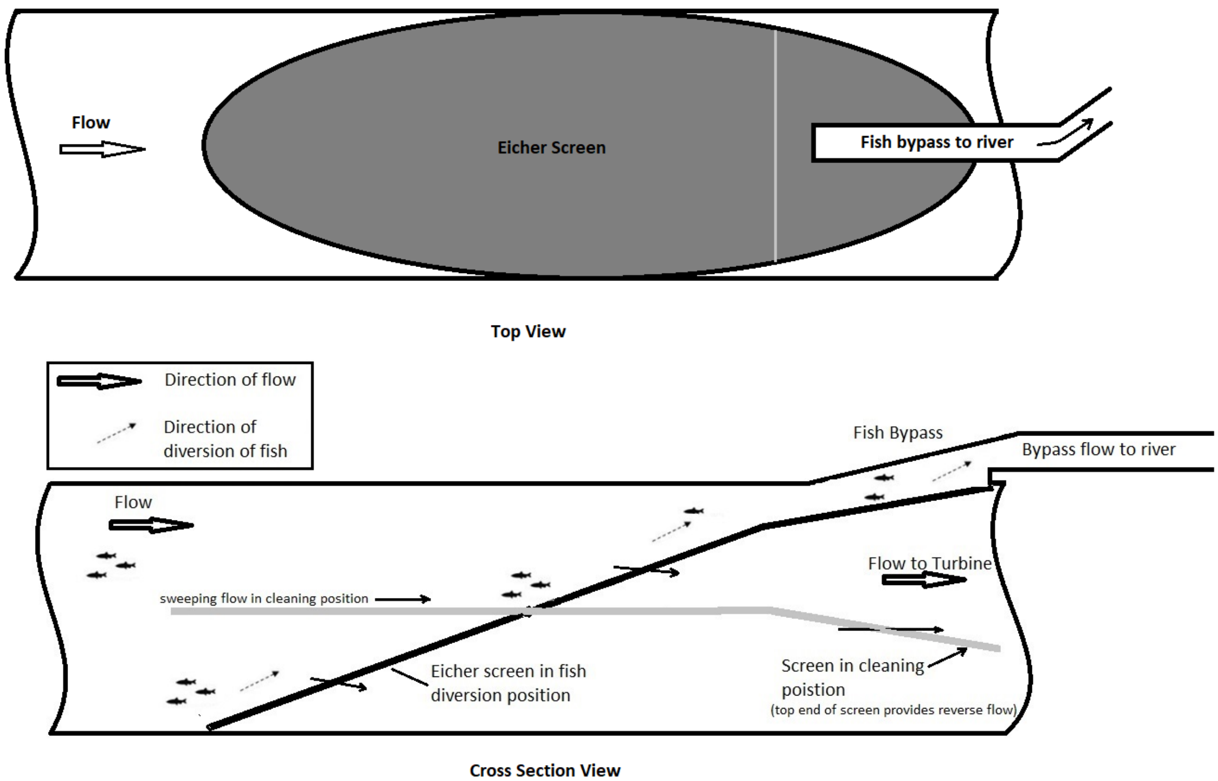


Figure 80. Schematic diagram of the diversion dam, the two intakes, Eicher screens, bypasses, diversion dam overflow and the bypass weir (Source: Bengeyfield 1995).

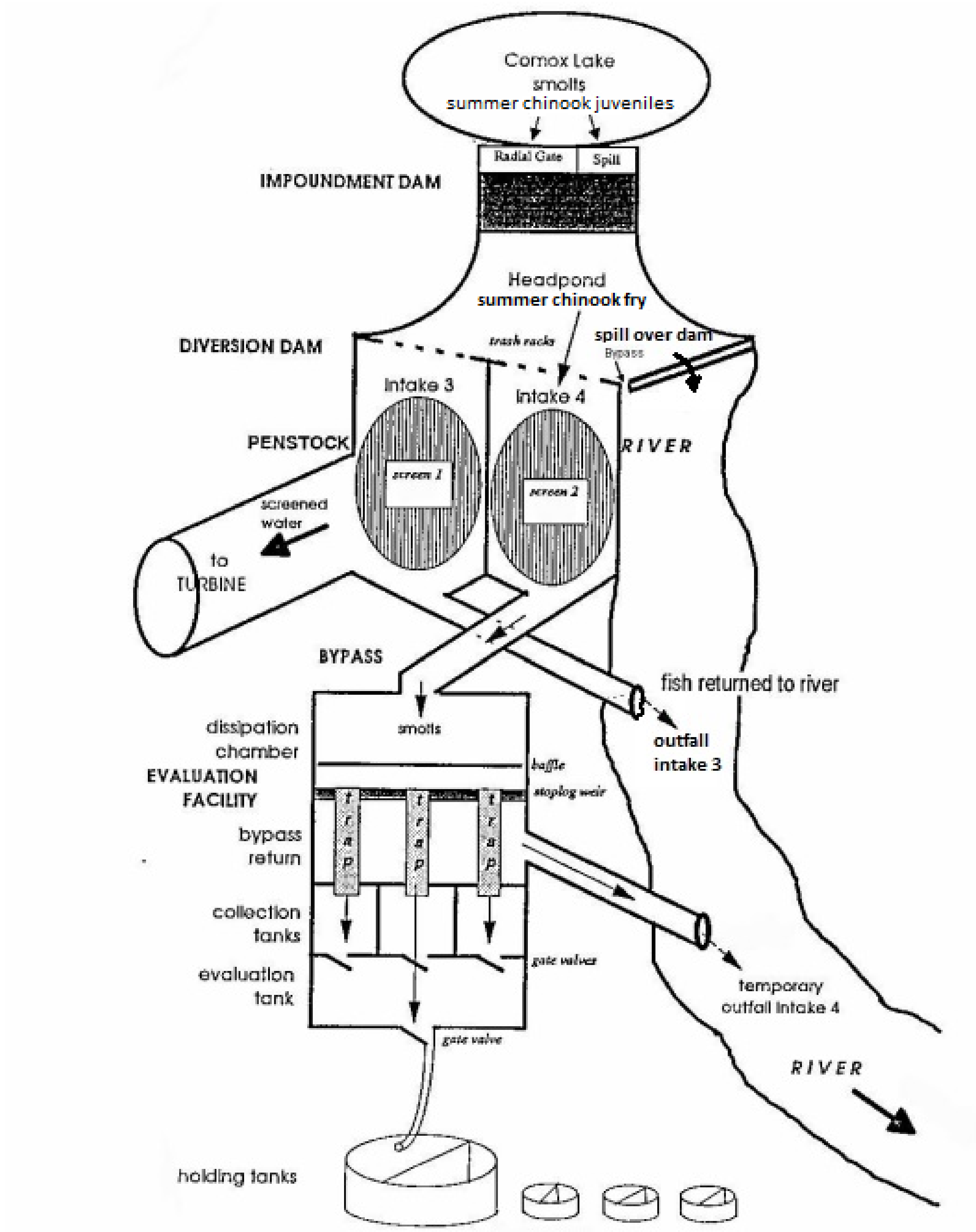


Figure 81. An aerial of the diversion dam site showing the intakes, Eicher screens, bypass flow, diversion dam spill and bypass weir (Source: BC Hydro).

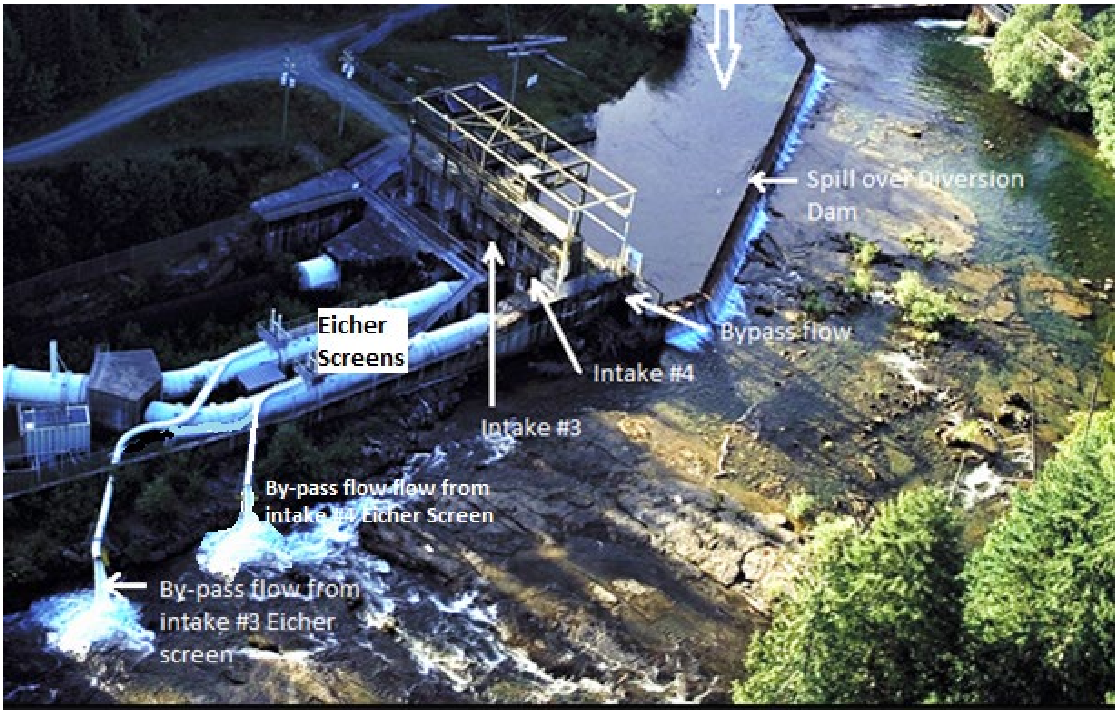


Figure 82. Eicher Screen in cleaning position in preparation for pressure washing in September 2010. This photo shows the screen clogged with organic material (Photo: Sheng 2010).

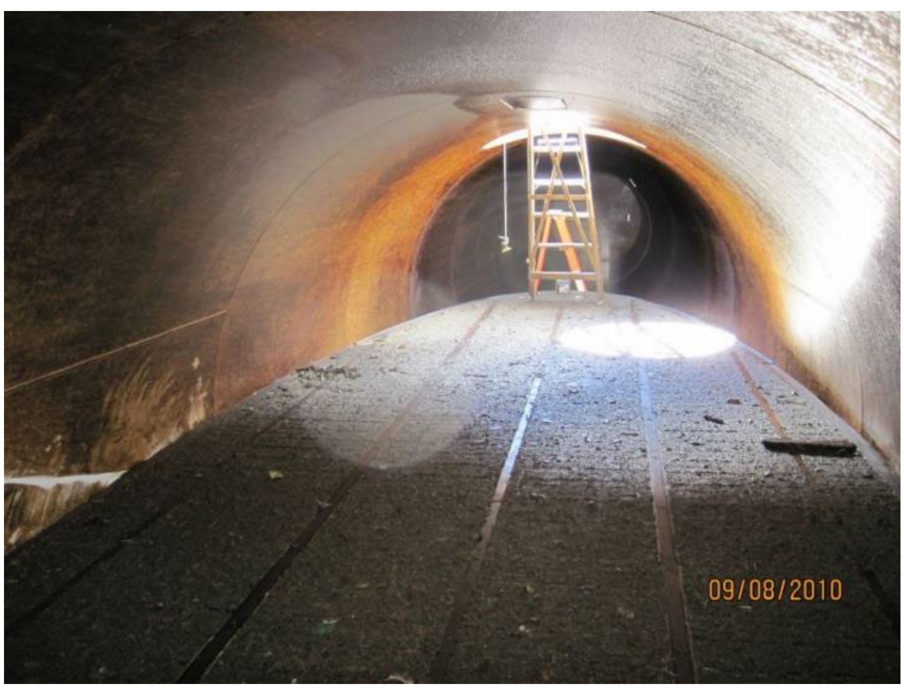


Figure 83. **Top of the upstream end of the Eicher Screen. This section of screen is typically cleaner than the rest of the screen (Photo: Sheng 2010).**

Eicher Screen Efficiency Modelling and Monitoring

The efficiency of the screens was assessed between 1993 and 2005 and again from 2010 to 2019. Efficiency (on diverting Coho) for the Puntledge Eicher screens immediately following their installation was 99.8% (BC Hydro 1995). Studies on Eicher screen efficiency at the Elwha dam in Washington State in 1990-1991 indicated a diversion rate greater than 98% for Coho, Chinook, and steelhead juveniles (EPRI 1992). By 2010, information regarding the performance of the screens at the Puntledge diversion dam was still limited to the initial evaluations conducted in 1993-1994 (Bengeyfield 1994, 1995), leading to a growing concern among DFO staff and other stakeholders regarding the actual diversion efficiencies and operations of the Eicher screens and their influence on the survival of downstream migrating Coho and Chinook juveniles. Bengeyfield (1995) conducted an efficiency trial using chum fry between 41 mm and 54 mm. Results indicated an overall mortality rate of 3.5%, though this value is likely underestimated. Observations made through the viewing ports in the penstock found that some of the fry disappeared completely through the screen, some became impinged on the face of the screen, or were observed sliding up the screen, often colliding with debris that was lodged in the screen (Bengeyfield 1995). The screens were originally designed to pass 37 mm Chinook salmon migrants, but they have never adequately been evaluated for Chinook fry at the Puntledge diversion dam due to the absence of adult Chinook spawning above the dam over the 10-year period after their installation. Recent emergent summer fry at Puntledge hatchery ranged between

30 mm and 33.6 mm in brood years 2020 and 2022. Additional studies to address these uncertainties were proposed in 2010 (Guimond 2018).

From 2010 to 2017, summer-run Chinook and Coho salmon production from the upper Puntledge River watershed was evaluated at the Puntledge diversion dam with funding from BC Hydro's Fish and Wildlife Compensation program (FWCP). This assessment program was implemented as part of an agreement between DFO and BC Hydro to decommission the Upper Puntledge Hatchery facility and supported DFOs new strategy for rebuilding sustainable populations of summer Chinook and Coho salmon in the upper Puntledge River watershed. Fundamental to this new strategy is access to and utilization of habitat above BC Hydro's diversion dam, and successful juvenile migration past the diversion dam for both species (Guimond and Taylor 2014).

Recent evaluations of summer Chinook and Coho migration at the Puntledge diversion dam (Guimond and Taylor 2011; Guimond *et al.* 2013) clearly demonstrated that Chinook fry begin emigrating from spawning habitat upstream of the dam soon after emergence (early February) and that this component may account for over 80% of the total juvenile migrating population. Furthermore, in 2013, assessment of Eicher screen efficiency (rate of diversion of fry/smolts to bypass the turbines) was found to be significantly compromised when debris accumulates on the screens, resulting in higher mortality for small emergent fry and concern from BC Hydro.

To address the potential issues more adequately, the Puntledge Fish Entrainment Strategy (FES) Technical Committee was formed in 2013 to investigate renewed concerns that the Eicher screens might not be effectively mitigating the entrainment during the early spring Chinook fry out-migration. This committee was tasked with evaluating entrainment risks and updating the existing action plan to mitigate summer Chinook entrainment-related mortality. Over the following four years, the committee applied a demographic population model to help quantify impacts of several scenarios while working to balance other important social and economic values brought forward by group (Connors and Parkinson 2015).

In 2015, the Puntledge FES Technical Committee produced several recommendations to help mitigate entrainment impacts on summer Chinook. These recommendations included operational recommendations like optimizing routine shutdowns at the facility to avoid peak out-migration; increasing the number of routine scheduled cleanings of the Eicher screens to maximize efficiency and supporting ongoing operations to reduce pre-spawn mortality of adult summer Chinook. Additionally, the group highlighted several uncertainties that should be further investigated to inform future operations (Connors and Parkinson 2015).

Several of the uncertainties in the Chinook population model were re-visited in 2019 with newly collected and historical data (Siegle and Parkinson 2020). Refinements were made to the model parameters as follows:

- Screen efficiency based on penstock flow, fry length and volume of flow through the screens since cleaning.

- Fry length as a function of the time of the year.
- Fry migration timing (mean, standard deviation, and distribution).
- Spill ratio (fish/m^3) in water moving over diversion dam relative to penstock).
- The required threshold fish length to achieve 100% screen efficiency.

Screen Efficiency based on Penstock Flow, Fry Length, and Volume of Flow through the Eicher Screens since Cleaning

Trials were conducted between 2010-2016 using fall Chinook emergent fry as a surrogate for summer Chinook. Fry at a range of sizes were released in the penstock of intake #4 at various power generation rates to determine screen efficiency. The culmination of yearly trials resulted in the following observations:

- Eicher screen performance (screen efficiency) is compromised when debris accumulates on the screens, resulting in significant injury and mortality of fish, particularly on smaller (i.e., emergent sized) fry.
- Operation of the evaluation facility (currently associated with Intake #4) may impact the optimum design flow in the bypass pipe and velocities at the downstream end of the Eicher screen to some extent due to flow restrictions from stop logs on the dissipation weir, compared to the unmonitored Intake #3. Although a minor improvement on diversion efficiency was observed with the manipulation of stop logs, the effect was not statistically significant. However, since 2014, the facility has been operated with four fewer stop logs on the weir to increase the bypass flow compared to the initial assessments in 2012 and 2013.
- Under low generating flows (5 MW, or 7 m^3/s) Chinook fry responded to a light stimulus at the top of the penstock resulting in a slightly higher diversion rate compared to unlighted conditions. However, there was no light induced response at higher generating flows.

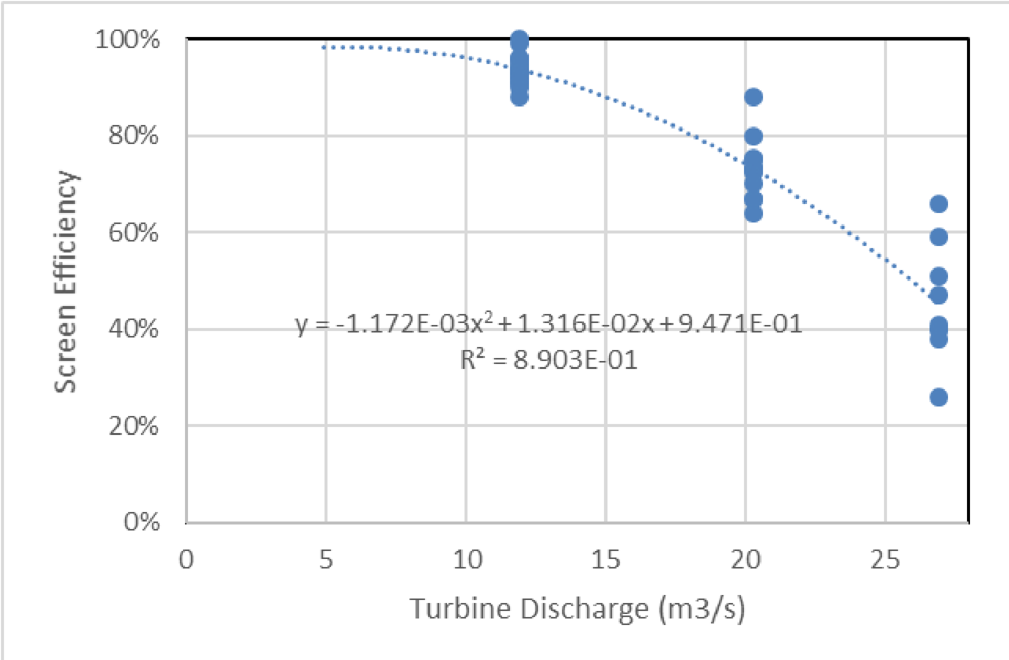
Turbine flow (penstock discharge) had the greatest influence on fry survival, particularly on fry <40 mm fork length. At the highest turbine flows tested (max power generation), the smallest fry only achieved an average of 43% survival in passage across the screen while at 18 MW, survival almost doubled. When generation was reduced to10 MW and 5 MW, survival closely approximated the design specification efficiencies for the Eicher screens (95%; Table 45).

BC Hydro typically operates the generation plant at one of three powers levels (10 MW, 18 MW, and 24 MW) or at discharges of 11.9 m^3/s, 22.45 m^3/s, and 26.9 m^3/s. When storage is available, the plant mainly operates at 24 MW or 18 MW during the spring, fall, and winter months. The most comprehensive set of data on screen efficiency using fry ranging in size between 38-39 mm was collected in 2014. Results clearly showed a decrease in screen efficiency at higher turbine generation rates (Figure 84).

Table 45. Summary of screen efficiency trials conducted between 2014 and 2016 at various generating flows and sizes of fall Chinook hatchery fry (Source: DFO 2018, unpublished report).

Fry Size (FL)	**37-40 mm (Avg 38mm)**				**40-45mm (Avg 44mm)**		**46-54mm (Avg 50mm)**		**55-62mm (Avg 57mm)**	
Generating	**5 MW**	**10 MW**	**18 MW**	**24 MW**	**18 MW**	**24 MW**	**18 MW**	**24 MW**	**18 MW**	**24 MW**
Conditions	Lighted & Unlighted penstock; up to 4 stop logs removed				Lighted penstock, 4 stop logs removed					
# Trials	11	18	21	27	7	2	1	1	1	1
Efficiency (%)	93.1	93.7	78.4	42.7	90.3	83.5	98.0	92.0	98.0	93.0

Figure 84. Screen efficiencies measured in 2014 using 38-39 mm Chinook fry released directly into the intake (Source: PUN 2018).



In response to the poor screen performance results, BC Hydro initiated operational measures to maximize Eicher screen efficiency for summer Chinook fry. Automated screen cleaning cycles, which have been shown to influence screen performance, were set to a daily daytime auto-cleaning schedule. The penstock is shutdown in early February to clean the Eicher Screens prior to emergent fry migration. The plant is also shutdown for two weeks in March for infrastructure maintenance, and the screens are again power washed. However, BC Hydro commits to this scheduled two-week spring maintenance shutdown once a year in advance to accommodate crew scheduling at several facilities, with very little opportunity to adjust dates if the shutdown does not coincide with peak fry migration. Currently, it is assumed that migration timing is similar every year; however, based on ATU calculations during incubation and hatchery emergence time, which is similar to wild summer Chinook, emergence timing and presumably fry migration, can vary up to one month depending on whether the incubation season was cold or warm (Figure 85).

Based on the growth rate and size of Chinook fry in Puntledge River between February 1st and late April (Figure 86), BC Hydro operates the generation plant between February and March to potentially maintain 90% weekly survival past the Diversion Dam. This operation includes increasing spill over the dam, with the seasonal target of 95% bypass efficiency.

Figure 85. **Hatchery summer Chinook start and end ponding dates versus ATU river temperature calculations between 2008 and 2018 (Source: Sweeten 2005).**

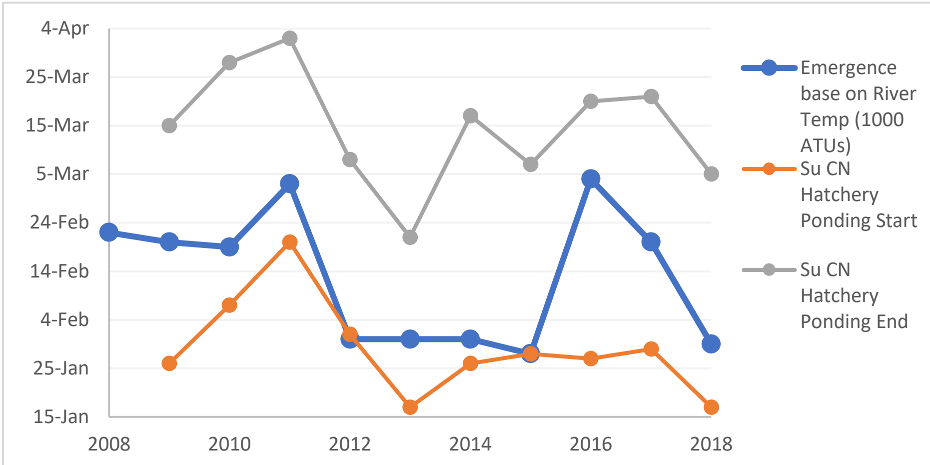
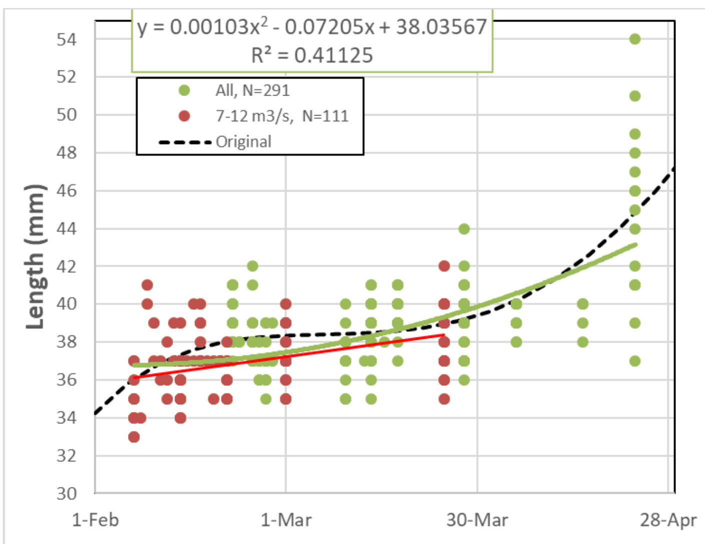


Figure 86. **Chinook fry lengths obtained from Wolf Trap catches in the Puntledge River from the 2011, 2013 and 2014 migration seasons.**



Note: Many of the 291 points are superimposed because lengths are collected on selected days and measurements are to the closest mm. The black line represents a 5th order polynomial fitted to data collected between February 8 to July 26, 2013. Red dots and line represent data from low discharge days when length bias due to screen inefficiency is less likely to be a factor. For modelling, a 3rd degree polynomial was adopted. The two functions are similar but the extrapolation to February 1 in the new relationship does not include a sudden decline in size.

Summer Chinook are mainly composed of hatchery returns and in recent years escapement has declined below 400 adults. Hatchery production has also declined, but more precipitously due to increases in BKD and subsequent culls of fertilized eggs from infected female parents. Staff have also observed differences in emergent fry size, growth rate, and mortality rates during rearing compared to fall Chinook fry which have larger eggs, and thus are larger at emergence, have a higher growth rate, are more robust, and experience less disease issues during the rearing phase.

In contrast, the emergence size of summer Chinook fry is very small and more similar to the size of emergent sockeye salmon. Based on analyses of 2022 brood summer Chinook, mean weight at ponding averaged 0.264 g and fork length was 30 mm (Figure 87). In comparison, in Brood year 2020, fall Chinook ponding size averaged 0.44 g or an estimated length of 39 mm (Figure 88), which have been the proxy used to represent the summer Chinook in all Eicher screen efficiency tests. The low weight of the summer 2022 brood Chinook fry is alarming when compared to the emergent fry weight between 1973-1976. Weight ranged between 0.406 g to 0.4734 g (MacKinnon *et al.* 1979).

Figure 87. Puntledge River Summer Chinook ponding size (weight and length) and growth for brood year 2022 (Source: DFO SEP unpublished data).

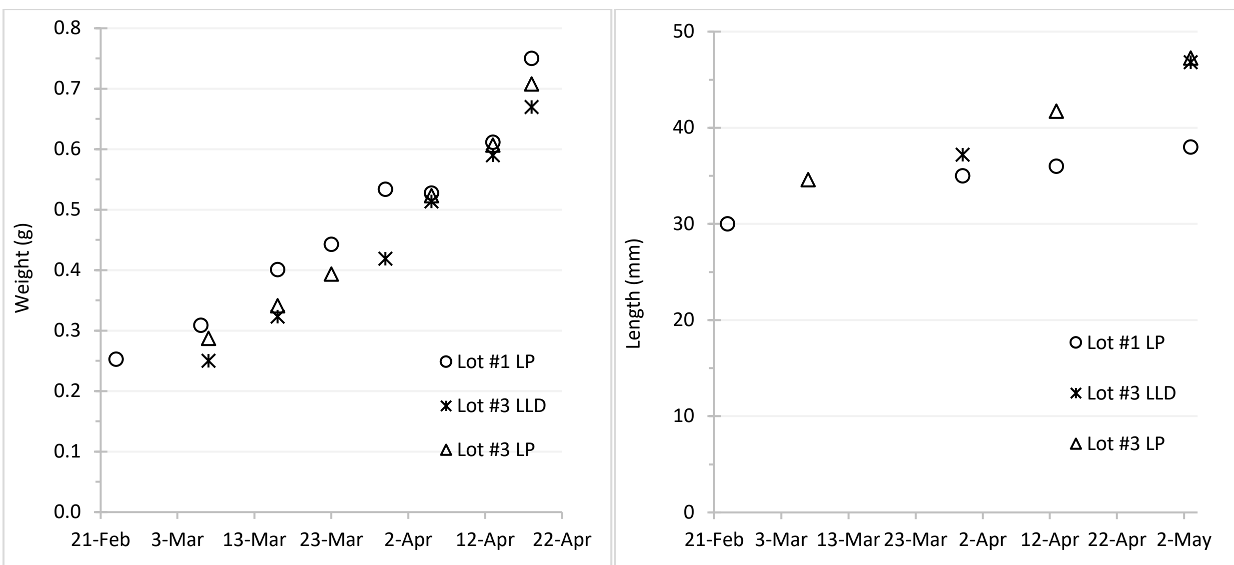
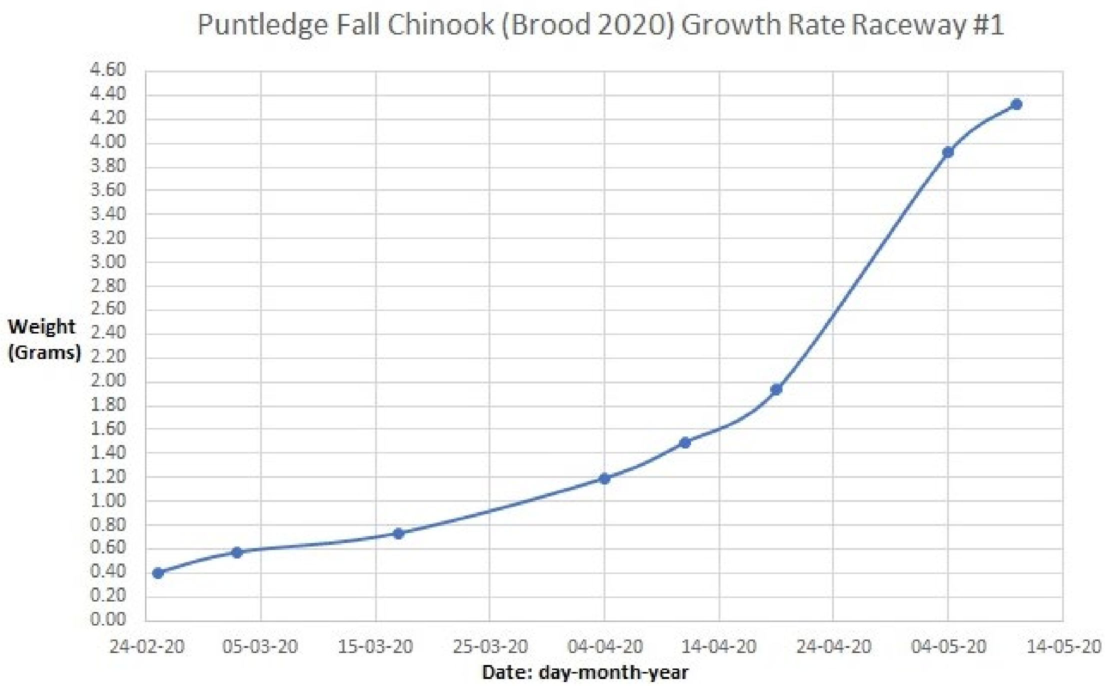
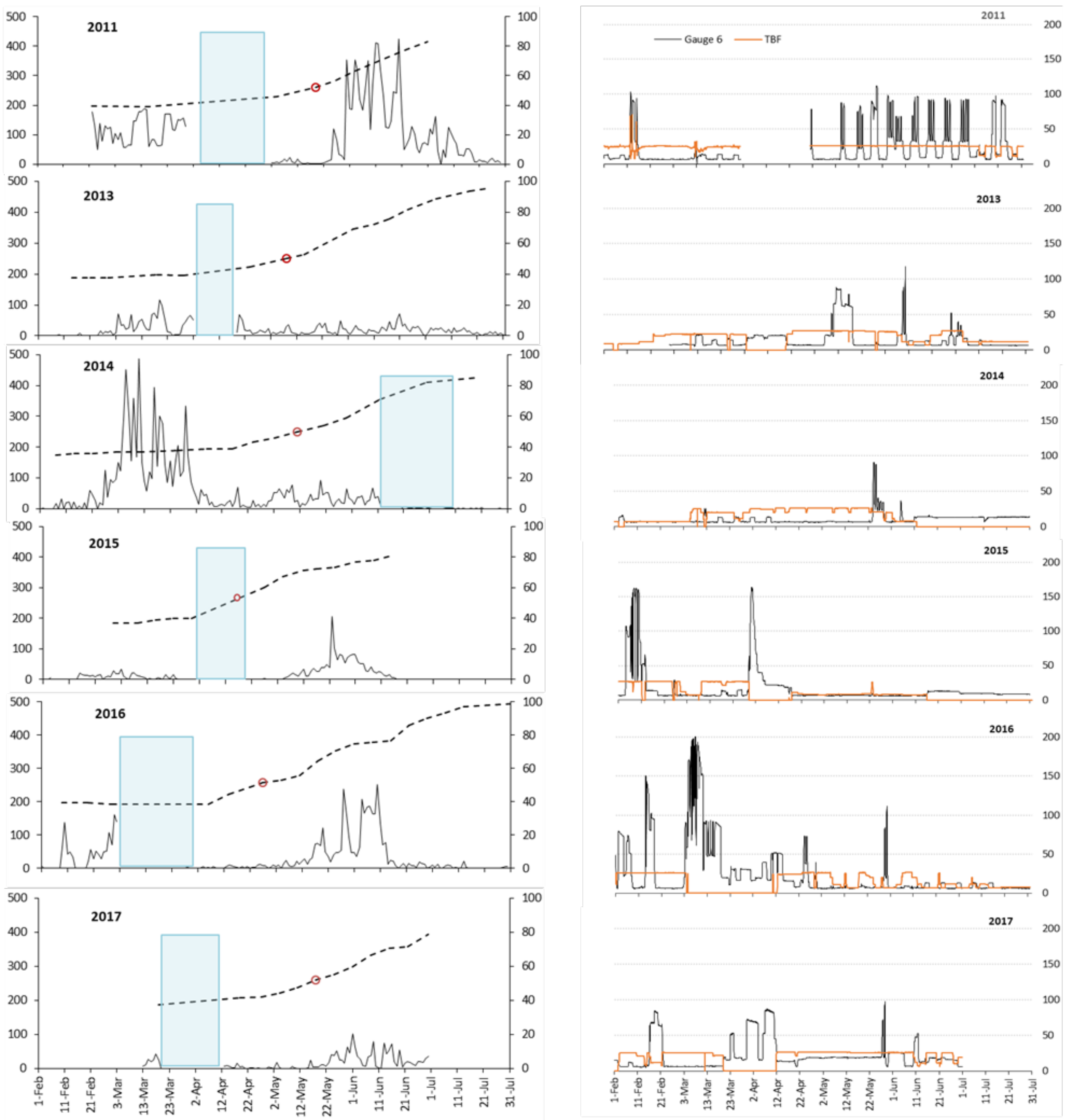


Figure 88. Puntledge fall Chinook ponding weight and growth rate.



In the context of entrainment mortality versus fry size, trials on Eicher screen efficiency using larger fall Chinook fry as a surrogate may be overestimating actual survival rates for natural-origin summer Chinook. Smaller fry are only prevalent in the Wolf trap when turbine discharge is low, while at higher discharges, smaller fry are impinged on the screens and therefore are not present in the Wolf traps (Table 45). Power generation at 18 and 24 MW has severe impacts on the smaller emergent summer Chinook, which would not be detected in the Wolf Traps. Based on fry size data from the hatchery, the emergence of small fry occurs throughout the ponding period. In the 2022 brood year, small fry emerged between mid-February and early-April (Figure 87). Small fry would not be present in the Wolf traps at the sampling center, later during the migration period when BC Hydro typically increases and maintains generation at 18-24 MWs and the smaller fry are likely impinged on Eicher screens. Fry migration timing and the BC Hydro generation rate between 2011-17 during fry migration is shown in Figure 89.

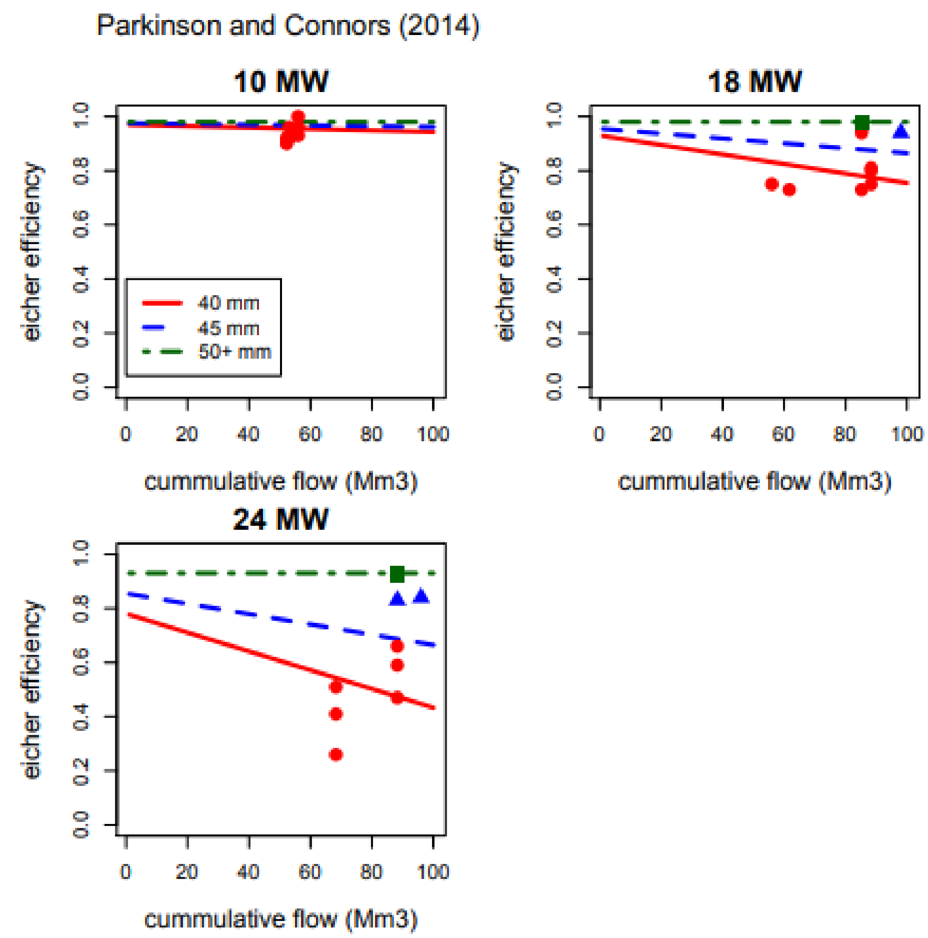
Figure 89. **Temporal pattern and size distribution of wild summer-run Chinook fry and under yearling smolt migration from the Puntledge River upstream of the diversion dam (left), and corresponding river and turbine discharge (right) for sampling years 2011-2017. BC HYDRO shutdown periods (shaded blue) and 50 mm fork length thresholds (red symbol) are denoted. Under yearling hatchery smolt migration (from CWT only releases) is included in 2011 catches (Source: Guimond 2018).**



Eicher Screen Fouling

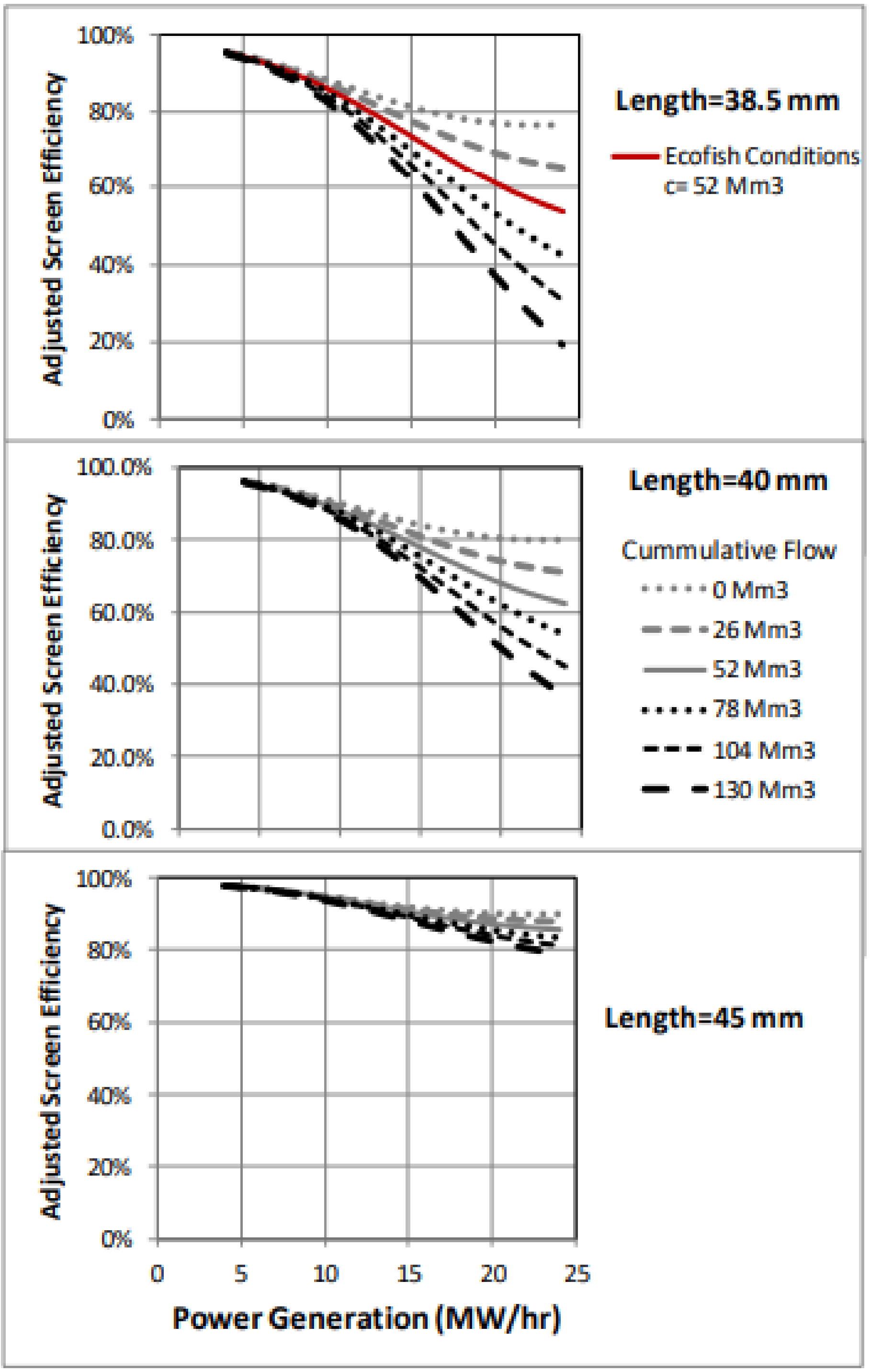
As an indicator of Eicher screen fouling over time, the accumulative amount of flow passed through the screens was calculated and linear relationships were developed illustrating the relationship between Eicher efficiency as a function of Chinook fry size, cumulative flow, and power generation (Figure 91). Another family of equations and curves were created as a function of screen efficiency in successfully passing fish as a function of the time of the year and fry migration timing (mean, standard deviation, and distribution), which is presented by fish size for modelling purposes from the data collected in 2019 (Figure 91). This conforms with the composite of data collected over all the years the screen was assessed. In summary, screen efficiency decreases as the accumulative flow through the Eicher screens increases and as power generation (i.e., MW) increases. As fry size increases, screen efficiency increases (Figure 90 and Figure 91).

Figure 90. Eicher efficiency as a function of Chinook fry size, cumulative flow, and power generation.



Note: Lines are the predicted efficiency based on a function developed by Parkinson and Conner (2014) and points are empirical estimates of efficiency from trails in 2014 (red, blue, green circles are 40mm fry; triangles are 45 mm fry and squares are +50 mm fry, respectively). There is an absence of data at the y-intercept. Each line is chosen to produce a negative slope for all lines and a larger spread at higher flow generations (Source: Parkinson and Connors 2014).

Figure 91. **Adjusted screen efficiency in successfully passing fish as a function of the time of the year and fry migration timing (mean, standard deviation, and distribution), which is presented by fish size. Family of curves generated representing the experimental data collected (Source: Hocking *et al.* 2019).**



Data Collected on Chinook Fry Size and Migration Timing Between 2011 and 2017

Information on the survival of summer-run Chinook from natural spawning areas above the dam, downstream migration patterns, and size distributions of emigrants was collected between 2011 and 2017 (Figure 89). However, the data were confounded because of variation in the study commencement and termination dates, interruptions to the annual monitoring schedules due to Generation Station maintenance shutdowns and regulated high flow spill events over the 8-year monitoring program.

Fry Emergence Time

Juvenile migration typically begins in early February soon after fry emergence, although the onset of emergence and peak migration timing is strongly influenced by temperature, river discharge, and intraspecific interactions (Healey 1991). Emergence timing at the main spawning grounds in Reach B was calculated by multiplying the mean daily lower river temperature data (Sweeten 2005) by the number of days of incubation (i.e., accumulated temperature units (ATU)) that totalled 1,000 ATUs. The ATU emergence time was later than the emergence time recorded at Puntledge hatchery. However, fry emergence varied up to one month for both ATU estimates, and the recorded hatchery emergence times (Figure 85). The hatchery water supply comes from the BC Hydro penstock, which diverts surface water from the Diversion dam, approximately 1.35 km downstream from the main spawning grounds and therefore should closely represent river emergence time in Reach B. The mean river temperature was recorded at the lower river and therefore is subjected to more atmospheric cooling as water flows from Reach B to the lower river, resulting in a more delayed emergence estimate than the emergence recorded at the hatchery. There was only four years of water temperature data that was available between 2015 and 2019 that could be compared to the lower river monitoring station. It was shown that the cooling effect is more pronounced if the river water is warmer (DFO, unpublished data).

The timing of the Diversion Dam shutdown and Eicher Screen maintenance and cleaning should be scheduled during the estimated period of peak fry emergence. However, this is never taken into consideration. Shutdowns are strictly based on BC Hydro crew availability, which is linked to the scheduling of a suit of facilities. Between 2011 and 2017, the two-week shutdown completely missed the peak period of emergence in at least one of the brood years (Figure 89). Ideally, based on emergence times estimated between 2008 and 2018 and daily counts at the Eicher Screen Sampling Centre, shutdowns should be scheduled between January 15th and the end of March. However, if the shutdown period is limited to two weeks, this should be scheduled from late February/early March to mid-March to protect the emergence of the smaller more vulnerable emergent fry.

The out-migration of Chinook juveniles follows a bimodal pattern consistent with observations of summer Chinook migration from the Puntledge spawning channel between 1966 and 1968 (Lister 1968). Based on the fish samples collected at the Diversion Dam Sampling Centre, the majority of juvenile Chinook migrate as newly emerged fry between February and April (average FL ~40 mm), followed by a second smaller migration of larger under yearling smolts between May and July (average FL ~75 mm). However, the size range of the recovered smaller fry at the Centre may only represent a sub-set of the larger earliest migrating fry due to the inherent size-selective limitations of the Eicher Screens. The screens were originally designed for diversion of fry approximately 38 mm or larger. Smaller fry are highly prone to screen impingement and therefore would not get diverted to the Sampling Centre. The dead impinged fry would either disintegrate on the screen, decompose and/or get washed off and carried down the penstock and completely bypass the Sampling Centre during screen cleaning cycling events.

Over the course of the monitoring period, the emergent fry and under yearling smolt migration period was differentiated using a fry size of 50 mm fork length, which is similar to historical observations (Lister 1968). However, in the context of fry survival at Puntledge diversion dam Eicher screens, this size threshold determines the point at which entrainment mortality and injury of emergent fry due to impingement on the screens is significantly reduced. Emergent-sized fry (38 mm) have half the survival rate (~43%) as larger juveniles (>50 mm) (>90% survival) (Table 45).

The 2014 (brood year 2013) juvenile monitoring program was the most comprehensive assessment of emergent Chinook out-migration timing and abundance and yielded the largest catches and the largest population estimate over the 8-year time series (Figure 89). The success of data collection in this year is attributed to uninterrupted data collection, as well as lower generating flows and absence of freshet flows during the out-migration period (apart from the annual 2-day kayak pulse flows in May).

In 2013 and 2014, 80-85% of the total juvenile production migrated during the emergent period (<50 mm), similar to historical migration patterns (Lister 1968), while this proportion was much lower in 2015 and 2016 (25-35%). Warmer spring temperatures and higher snow melt discharges may have affected these latter years of data collection. Water temperature influences fry growth rate and determines when fry reach the critical size threshold of 50 mm fork length for out-migration. Over the 8-year time series, the date that fry reach this size threshold varied by as much as four weeks. In 2015, an unseasonably warm winter/early spring produced 50 mm fry (average) by late April, while in cooler years, fry did not reach this size until late May. The reduced size of fry at migration time can significantly increase the potential for major yearly losses of Chinook fry due to entrainment at the Puntledge River penstock intake at high generation levels.

DNA analysis of early versus later migrating Chinook fry at the diversion dam found that the different timing groups were genetically unique, suggesting that differentially higher mortality of the early smaller migrating fry could be impacting genetic diversity in the population. There was a loss of genetic diversity when comparing parents and their offspring. The effective size of the natural fry sampled in 2015 and 2016 was less than a third to one half of that represented in the pool of potential parents sampled the previous fall. Neither the early emergent nor later juveniles represented all the juvenile genetic diversity. Furthermore, the genetic diversity of juveniles was not randomly distributed over the entire out-migration timing. Over 20% of the maternal families analysed in 2016 were exclusively from early emergent migrants (Wetklo *et al.* 2017).

Spill Ratio (fish/m^3) in Water Moving over Diversion Dam Relative to Penstock

The proportion of fry that migrate directly over the diversion dam spillway strongly influences the impact the diversion dam operation has on the population because these fry do not encounter the screen and do not experience entrainment mortality. The relative concentration of Chinook fry in spill versus penstock water (Spill Ratio – SR in text, r in equations) affects the overall migration success rate past the dam. SR is used to model the effects of changes in spill and penstock discharge on overall entrainment mortality. The key assumption is that the relative density (fish/m^3) of fish in spill versus penstock water does not change with varying discharges and water volumes. At a turbine rate of 10 MW, 90% screen efficiency is initially expected. To achieve the 95% target, a spill rate of 12 m^3/s is required to increase to number of fry diverted directly over the dam spillway. When taking into consideration the countervailing effects of fry growth overtime and screen fouling, the model is able to adjust and calculate how much spill is required.

Using the updated values, summer Chinook population models were revisited, and Siegle and Parkinson (2020) developed an application to estimate fry entrainment using stream discharge and power generation. The generally accepted entrainment objective that was agreed upon through the process was >95% of the total out-migrating Chinook fry population should avoid entrainment, to migrate downstream naturally between February 1st and April 30th. While it was acknowledged that the level of fry diversion may vary over the course of the out-migration period, the weekly fry diversion efficiency should remain above 90% to maintain the diversity of the population.

There is no data on the downstream migration behavior or the 3-D distribution of summer Chinook fry in the PUN headpond and no indication that Puntledge summer fry distribute evenly bank-to-bank or throughout the water column. Field data collected in 2014 and 2016 showed variable results suggesting that distribution may be different for different size fish and that there could be a positive rheotactic behavioural response to flow over the dam. Trials using a Didson and side scan sonar technology (Svein Vagle IOS) in 2015 failed to identify the location and distribution of Chinook fry in the headpond. In field studies, Marshall (1973) observed that summer Chinook emergent and small fry in Puntledge River were closely associated with shallow and low velocity nearshore habitat. Craig (2015) observed the same distribution and observed that the juveniles migrated downstream along the banks and utilized side-channel habitat for rearing in the Cowichan River. Similarly, emergent Chinook fry preferred very shallow water habitat and low velocities and only moved into higher depth/velocity as they grew larger in Big Qualicum River (Lister and Genoe 1970). When fry approach the weir crests there is a positive rheotactic response (i.e., turn to face into an oncoming current) and avoid exiting over the division dam spillway. Velocity increases above 1 cm/s per cm distance, which elicits this type of behavioural response (Enders *et al.* 2012). Fry were often observed on the upstream side of the diversion dam spill just below the crest where velocity is low. Behavior studies suggest that there is a preference for the low depth, low velocity, low acceleration habitat near the dam and that fry distribution is non-random.

Although increased spills may help achieve the overall +95% efficiency of Chinook juveniles successfully migrating past the diversion dam, there may be an off-setting impact on early emergent fry that are displaced by the higher spill flows and result in fry being diverted over the dam or through the Eicher screens at smaller size than when the river discharge remains low. A preliminary analysis of the cross-sectional velocities in Reach B, where most of the summer Chinook emerge, demonstrated that velocities and depths at a discharge of 29.4 m^3/s are only suitable for emergent fry rearing up to 2 m from the riverbank and potentially less at higher discharges. Furthermore, summer Chinook fry that are displaced into Reach C are exposed to a rearing environment that has a high amount of bedrock, high water velocity, and marginal fry habitat. Marshall (1973) reported that high river discharge during early fry rearing had an impact on overall adult survival. He noted that survival was higher when river discharge was low during the early fry rearing period. Summer Chinook fry now also must compete with a large population of fall Chinook fry below the diversion dam. Fall fry are larger in size and potentially have established rearing territories before the summer Chinook arrive. In the last 10 years, fall Chinook escapement has averaged 9,000. Based on bio-standards, fry density capacity is likely exceeded.

Latest Operational Plan

In spring 2020, following the recommendations of the FES Technical Committee, BC Hydro piloted constrained generation under the guidance of the 'Puntledge Spill Planning Application' (Siegle and Parkinson 2020). This involved increasing the total amount of volume that is spilled past the intakes and by curtailing generation during key periods of migration. Following recommendations made in Connors and Parkinson (2015) and endorsed by the technical committee, an additional Eicher screen cleaning occurs in late January/early February and an annual shut down maintenance of the facility has been scheduled to coincide with the peak summer Chinook out-migration (late February to mid March). During the maintenance shutdowns 100% of the river's flow bypasses the penstock intakes, ensuring that no out-migrating summer Chinook become entrained. During 2021 and 2022, the scheduled maintenance outages were 38 and 19 days, respectively. The facility has now operated three years under this constrained application (i.e., 2020-2022).

Modelled estimates, derived from undated Puntledge Chinook population model (Siegle and Parkinson 2020), calculated that operations on the Puntledge River operational plan have thus-far been successful at achieving the accepted entrainment criteria. During the out-migration periods in 2021 and 2022 (February to April), model estimates indicate that that 99.3% and 95.8% of out-migrating smolt, respectively, have been diverted around the facility, while estimated weekly average diversion rates have remained >90% of individuals.

4.3.4. Disease, Parasites or Pathogens

4.3.4.1. Pathogens that could Pose Emerging Population Threats to Juvenile Puntledge Chinook

Broadscale pathogen monitoring across 60 viral, bacterial, fungal, and protist agents, most known or suspected to be salmon pathogens, but also including agents recently discovered in salmon around the world where linkages with disease is less well understood was conducted as part of the Strategic Salmon Health Initiative (SSHI). The SSHI dataset, spanning three Pacific salmon species (Chinook, Coho, Sockeye), and two aquaculture species (Atlantic and Chinook salmon), included tens of thousands of fish surveyed in freshwater and marine environments, largely centred on smolt out-migration (for Pacific salmon), but also including some returning adult migrant data. This section is included in the early rearing heading because the testing is mostly on juvenile fish, but it should be noted that the diseases and pathogens could be present in adult fish as well. Smolt outmigrant data, collected over a decade, to develop models assessing associations between pathogen prevalence and survival, and individual-based models associating pathogen abundance/load and body condition was used to identify agents that are associated with negative impacts on wild salmon (Bass *et al.* 2022). While models do not go as far as establishing cause and effect relationships, the consistency of associations across species to rank the pathogens of greatest concern can be used. Those pathogens that are also positively associated with climate change and/or with disease outbreaks in cultured salmon (hatcheries or open net pen salmon farms), are classified as being emerging threats to wild salmon population productivity, and most likely to influence the decreasing trends in survival observed over the past 20 or more years.

There were too few Puntledge Summer-run Chinook fish sampled to establish a baseline of infective agents for this stock alone. Hence, agents detected in all Puntledge River runs were combined to provide the most robust signals of agents that may be impacting fish from the Puntledge system. Below, the freshwater detections are presented separately from saltwater and show prevalence levels by year. In all, the data presented are based on 340 fish sampled over eight years (note that additional collections of 435 fish sampled in both freshwater (hatchery) and marine environments from 2008-2014 have not been run). The results of models and findings on salmon worldwide are put into context to identify pathogens found in Puntledge River Chinook that may pose the strongest threats to summer-run Puntledge Chinook into the future.

4.3.4.2. Agents Detected in Puntledge River Chinook in Freshwater

All freshwater pathogen collections presented in the SSHI were from the Puntledge hatchery. Eight of 57 of the fish collected at the hatchery and analyzed for pathogens are juveniles from the summer-run population. Most detections within the hatchery suggested infection at low levels (>25 Ct). The combination of weak detections and low prevalence indicates that for the most part, this small group of fish collected from the Puntledge Hatchery was relatively “clean” in terms of pathogens that we tested for, and not likely to be experiencing disease at the time of sampling.

The other agents detected at Puntledge hatchery that have been associated with disease and/or survival in Chinook salmon include *Ichthyophthirius multifiliis*, *Flavobacterium psychrophilum*, piscine orthoreovirus (PRV), and Candidatus *Branchyomonas cysticola*. Of these, two are known to be positively associated with climate change, the ciliate *I. multifiliis* and bacterium Ca. *B. cysticola*, while the other two are more associated with cool over winter temperatures. *I. multifiliis* is a protozoan that causes white spot disease (aka "Ich") that is highly problematic for freshwater aquaculture worldwide, and known to cause large-scale die-offs in Fraser River sockeye salmon spawning channels (Traxler *et al.* 1998). *I. multifiliis* detection in juvenile salmon was negatively associated with population-level survival among Chinook and Sockeye salmon studied in the SSHI (Bass *et al.* 2022; Teffer *et al.*, unpublished data). Given consistent negative impacts across species and associations with climate change, *I. multifiliis* can be classified as an emerging threat to sustainability of wild salmon, and given associations with hatchery fish, the impacts of this parasite on wild fish could be amplified by anthropogenic activities, if not carefully controlled. *I. multifiliis* could cause serious problems for both juvenile and adult Chinook salmon migrating through the Puntledge at elevated temperatures. Ca. *B. cysticola* is more associated with impact in saltwater, although it can be transmitted in both environments.

F. psychrophilum is a freshwater-transmitted bacterial pathogen and causative agent of bacterial coldwater disease, which affects a wide array of freshwater teleost species (Starliper 2011). As this bacterium proliferates in cooler water, it is not positively associated with climate change. However, as it is an issue within hatcheries, culture environments could increase risks to wild salmon if it is not carefully controlled. The levels observed in the two hatcheries *F. psychrophilum* was detected in are not suggestive of a disease event. While worldwide, PRV has been detected in fish from both freshwater and marine environments, and there is mounting evidence that it can cause disease in both environments, it has typically been associated with disease in the marine environment. PRV that is present in the Pacific Northwest today was introduced some 30-35 years ago from Norway (Mordecai *et al.* 2020), but while highly prevalent on farms and in the Columbia River salmon, it is less prevalent in wild/hatchery BC Pacific salmon (although it is prevalent on Chinook salmon farms). Detection of PRV in Puntledge hatchery Chinook in two years was unusual (although restricted to two individuals), as large-scale surveillance studies have rarely detected PRV in freshwater SEP hatcheries; however, it has been detected in Atlantic salmon in freshwater hatcheries (Bateman *et al.* 2021). Aquaculture hatcheries have considerably reduced PRV infection in freshwater in recent years through triple disinfection of eggs.

PRV is highly associated with the disease jaundice/anemia, which generally occurs on BC Chinook salmon farms the first winter at sea and is highly similar in presentation to PRV-caused diseases described in Pacific salmon worldwide (Di Cicco *et al.* 2018). We also find wild migrating Chinook with the early signs of this disease over fall and winter periods (Wang 2018). Population-level survivorship models identify a strong negative association with survival and body condition of Chinook carrying this virus in the fall and winter period (Bass *et al.* 2022). Given that the virus is relatively newly introduced and amplified on farms (Mordecai *et al.* 2021), this virus is not considered to be of concern to productivity of Chinook salmon populations. Note that *Renibacterium salmoninarum,* the etiological agent of Bacterial Kidney Disease (BKD), was not detected in juvenile Chinook collected at the Puntledge hatchery. This pathogen was reported to be observed at high prevalence in returning Puntledge adults. It is likely that pathogen profiles of Puntledge fish outside the hatchery are dissimilar from those of the hatchery fish, while in the river (Thakur *et al.* 2018).

4.3.4.3. Agents Detected in Puntledge River Chinook in Saltwater, First Year at Sea

In the marine environment, collections from the coastal environment in the first year following marine entry have been examined. The sample sizes between 2008 and 2015 ranged from zero fish in 2013 to 280 fish in 2010 and that most fish ranged between 100 mm to 250 mm in size.

Calculations of agent prevalence included both clipped and unclipped Puntledge Chinook (very few reported as summer Puntledge from GSI analysis). Pathogens detected in Puntledge Chinook sampled in marine waters that are traditionally expected to infect fish in freshwater include *C. shasta*, *F. psychrophilum*, *I. multifiliis*, *M. arcticus*, *N. salmonicola*, *P. minibicornis*, and *T. bryosalmonae.* Most other agents primarily infect fish in seawater but for many agents, freshwater versus saltwater infection is not a hard and fast rule. The freshwater pathogens identified here could indicate that they are present in the Puntledge River or estuary and thus could be present when adult Chinook are migrating upstream.

Of those freshwater pathogens listed, *T. bryosalmonae* might be of greatest concern due to its ability to cause large-scale mortality at elevated water temperatures, especially in juveniles, as has been observed for salmonids in Europe and the US (Sudhagar *et al.* 2019). While this pathogen has been observed in multiple Fraser River adult salmon, associated mortality events has not been observed in our adults. However, *T. bryosalmonae* was weakly negatively associated with population-level survival of juvenile Chinook salmon (Bass *et al.* 2022), with even stronger evidence in sockeye salmon (Teffer *et al.* unpublished data). Low to moderate prevalence of this parasite was found in marine collected Puntledge Chinook (marked and unmarked) in all collection years. Given that *T. bryosalmonae* productivity is positively associated with climate change and is consistently associated with population-level impacts across salmon and trout species worldwide, this parasite is potentially an emerging population threat to BC wild salmon, and one whose impact should be carefully monitored as rivers and oceans warm.

Of the marine species listed, *Tenacibaculum maritimum* is of particularly high interest due to its high ranking in recent studies of infectious agents and population-level survival in Chinook, Coho, and Sockeye salmon (Bass *et al.* 2022; Teffer *et al.* unpublished data), linkages with climate change, and with risks posed by salmon farms. *T. maritimum* was detected in marine migrating Puntledge Chinook in four of the seven years examined. This bacterium is highly concentrated around active salmon farms (Shea *et al.* 2020), and Fraser sockeye show an elevated risk of infection when they pass by open net farms in the Discovery Islands (Bateman *et al.* 2022). Two other closely related species of this bacterium have recently been found in B.C., one of which, *Tenacibaculum dicentrarchi*, was linked in the summer of 2023 to disease and mortality of adult Chinook caught in a recreational fishery while nearing the completion of their marine residence and preparing to enter freshwater on the West Coast of Vancouver Island. Due to its associations with climate change, farm activities, and population-level survival, the marine bacterium *T. maritimum* is considered to be an emerging threat to population productivity of BC wild salmon.

Ca. *B. cysticola* is a highly prevalent bacterium originally discovered in Norway in association with proliferative gill inflammation disease on farms (Toenshoff *et al.* 2012). Assessment of archived sockeye salmon tissues showed that this bacterium is not a recent introduction to BC, but rather has been detected in BC salmon since at least the early 1980s (Thakur *et al.* 2019). Ca *B. cysticola* is considered opportunistic, but recent evidence in farmed Norwegian salmon suggests its role in gill disease (epitheliosistis) is greater than originally assumed (Gjessing *et al.* 2021). Despite its high prevalence, this bacterium has been negatively associated with survival and condition in Chinook, Coho and Sockeye salmon (Bass *et al.* 2022; Teffer *et al.* unpublished data). In BC Chinook and Sockeye salmon, field infections with Ca. *B. cysticola* are associated with disruptions in osmoregulation and/or upregulated inflammatory markers (Wang 2018 in Review; Stevenson *et al.* 2020), consistent with its role in epitheliosistis in gills of farmed salmon. A recent longitudinal study in farmed Atlantic salmon identified an association between Ca. *B. cysticola* infection and levels of dissolved oxygen (Ferriera 2021), an environmental factor that is positively influenced by climate change (Marcogliese 2016). Taken together, the consistency in the negative relationship between *Ca. B. cysticola* infection with condition and survival of Pacific salmon, evidence that this bacterium is associated with gill disease on farms, and associations between bacterial productivity and climate-driven environmental variation would rank this bacterium as an emerging population threat to wild BC salmon. Laboratory challenge studies are still required to establish cause and effect relationships of Ca. *B. cysticola* with disease in Pacific salmon.

Among the other pathogens detected in marine caught fish, viral hemmoragic septicemia virus (VHSV) can be a virulent pathogen, and is, in fact, reportable to the Canadian Food Inspection Agency when found on farmed or wild salmon, although it is a known BC endemic agent, and uses herring as a host reservoir. Being an acute agent of disease, where mortality and disease ensues soon after transmission, it has not been observed commonly enough in the SSHI monitoring program to include it in models. The bacterium Candidatus *Syngnamydia salmonis* and microsporidian parasite *Paranucleospora theridion* are agents discovered on Norwegian farms whose association with proliferative gill disease in Atlantic salmon is still tenuous; both show a negative association with survival of Chinook salmon when detected in the warm summer period, but not Coho or Sockeye (Bass *et al.* 2022; Teffer *et al.*, unpublished data).

Rickettsia-like organisms, the causative agent of red mark disease, has a negative impact on the aquaculture product, resulting in downgrading of the quality of filets. It is not considered an established pathogen. This bacterium is also associated with reduced survival of Chinook and freshwater sampled sockeye salmon (Bass *et al.* 2022; Teffer *et al.* unpublished data). In all, Puntledge

Chinook carry a similar array of pathogens as observed in other areas of the west coast of BC, some of which are associated with patterns consistent with negative impacts at individual and population levels. Those of most concern are pathogens that respond positively to climate change, first and foremost, and to cultured environments, where some measure of control is possible. SSHI models in Chinook identified stronger negative associations of select pathogens, and overall pathogen richness, impacting survival than those based on temperature, a well-known factor influencing survival.

From these findings, it is possible that infectious disease is a contributing factor to declining trends in wild salmon, one which is likely considerably exacerbated under climate change. More research is needed to understand what levers humans can control to reduce impacts of pathogen infection in wild salmonids, and which populations may be at greatest risk. Note that these data, and the data used in SSHI models, are based on agent detections, which should not be confused with detection of disease. In nature, all organisms carry a mixture of potential pathogens, many of which may be tolerated until the host organism becomes stressed. SSHI is now undertaking research with a new technology, called salmon Fit-Chips, that will allow a better understanding of the interlinkages between environmental stress, infection, and disease, as well as identifying where and when salmon are most compromised. These studies, combined with knowledge of population-specific migration patterns and habitat usage, will help define areas of the BC coast that require urgent attention for habitat restoration or remediation of human-derived threats, and populations that will be most impacted by such remediation activities. If stressors and diseases act synergistically, it is possible that mitigation of one or two critical stressors could result in substantive improvements to population sustainability. Of note, pathogen data from adult salmon returning to the Puntledge River are lacking where it is estimated that 30-50% of adult Chinook experience pre-spawn mortality. All hatchery broodstock have only been routinely sampled for BKD. As an indication of the level of BKD in the adult returns, between 2013 and 2015, approximately 15% of the broodstock were classified in each category as negative, low level of detection, high-low positive and low positive; approximately 18% in the two other categories were low positive and moderate positive (see Section 4.1.9 for the definitions). If other pathogens play a role in this poor freshwater survival, having these data are necessary for understanding if and which agents might be driving such mortality. Currently, there are no major or emerging threats identified during the fry-juvenile rearing phase.

4.3.5. Lack of Access to Appropriate Food

Reduced phosphorus (P) levels in rivers and lakes in the Comox watershed in conjunction with heavy blooms of *Didymosphenia geminata* (Didymo) recently observed may have a negative effect on zooplankton and hence kokanee productivity in Comox Lake (Guimond *et al.* 2014). The Puntledge River, Comox Lake and its tributaries are categorized as nutrient limited, with total phosphorus concentrations below the detection limit (2 μg/L) in Comox Lake (CVRD 2019). This limits biological productivity in Comox Lake (Guimond *et al.* 2014). Furthermore, heavy blooms of Didymo have been observed in the Upper Puntledge River as well in the Lower Puntledge River, but to a lesser degree, and is associated with low phosphorus (Bothwell 2009, 2014, 2021). The low phosphorus levels may also be linked with changes in logging activity in the watershed. Phosphorus is often the critical nutrient that limits productivity of BC coastal lakes and clear cutting can result in an increase in nutrient uptake by plants and a consequential reduction of P in the groundwater table and in the aquatic environment.

Algal growth can be predicted by the ratio of dissolved inorganic nitrogen to phosphorus, the N:P ratio (Miller *et al.* 1978). Many algae have a cellular N:P ratio of about 12:1; a similar ratio in the environment is considered optimal for growth. Ratios much higher than 12:1 indicates phosphorus limitation of the algae, whereas ratios much less than 12:1 suggest nitrogen limitation. In field studies between 1978 and 1979, Puntledge had a high N:P ratio (i.e., 14:1 to 23:1) suggesting that Puntledge R. is phosphate limited. Very low total phosphate concentrations, chlorophyll a and periphyton accumulations were measured at all sites sampled. Benthic was also low (i.e. 6,000 organisms per sq.m.) (Munro et al, 1985) The counts in March were higher than June 1979.

Total alkalinity may also be limiting biological productivity in the Comox watershed. Trout/char fry densities in tributaries in the Upper Comox Lake watershed were well below the predicted maximum densities. Based on a productivity-based model using total alkalinity as a predictor of maximum biomass (i.e., Chinook: $g/100m^2=36*(Alk)^{0.62}$), estimates are low due to low alkalinity (i.e., 230.6 $g/100\ m^2$ per Age Group or Size Class; Ptolemy 1993). This is consistent with findings in previous surveys where Coho juvenile densities were assessed (Russell 1990; Griffiths 1995).

Data on seasonal food availability (i.e., abundance of aquatic and terrestrial invertebrates), consumption, and stomach fullness through the spring-summer months in freshwater for Puntledge summer Chinook could not be located; however, benthic invertebrate abundance in the Puntledge were in the lower range (i.e., $9.1x\ 10^3$ organisms/m^2 compared to other rivers studied in Southcoast of B.C. (i.e., range between 2.3 to 24.9 organisms/m^2 in 1978-79 (Munro et.al. 1985). However, based on length-weight data collected at the Eicher Screen Sampling Centre between 2010 and 2017 and mean monthly water temperature, growth is within the normal range and similar to growth rate in the Cowichan River, a system ranked as highly productive (Ptolemy pers. comm. 2023) (Figure 92 and Figure 93).

Figure 92. Mean weight and length of summer Chinook Captured at the Eicher Screen Sampling Centre between 2010 and 2017 (Source: Guimond 2018).

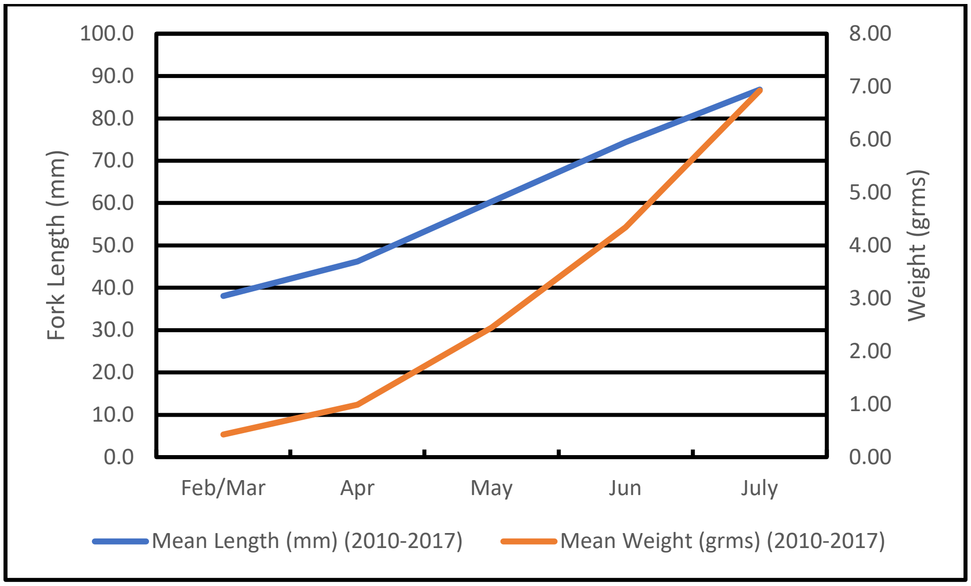
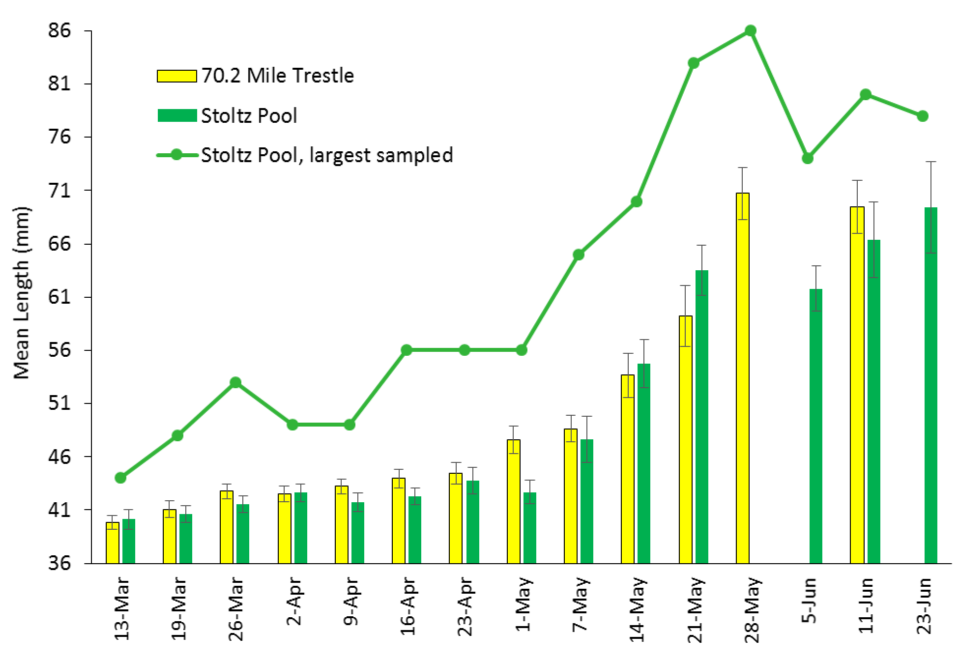


Figure 93. Mean length of Cowichan Chinook juveniles at 70.2 Mile Trestle, Stoltz Pool and the largest fish sampled at Stoltz Pool in 2014 (Source: Craig 2015).



The condition factor (i.e., $K = 10^5 \times weight(g)/(length(mm)^3)$) K is positively correlated with lipid storage (Spangenberg 2023). Summer Puntledge Chinook K increased from 0.78 at fry emergence, when water temperature is low and food availability is likely limited, to between 1 and 1.06 during the spring-summer months when food resources were more available. A K of 1 is considered normal for Northwest Pacific coastal Chinooks and it is common for K to increase from later winter to summer (Ptolemy pers. comm. 2023) (Figure 94 and Figure 95). The condition factor of Cowichan Chinook in 2015 is similar to Puntledge Summer Chinook that reared in Reach B and follow the same trend in K as the growth year progressed from late winter to summer (Figure 96).

Figure 94. Linear Regression of K versus percent lipid for individual Yakima Spring Chinook (Source: Spangenberg *et al.* 2023).

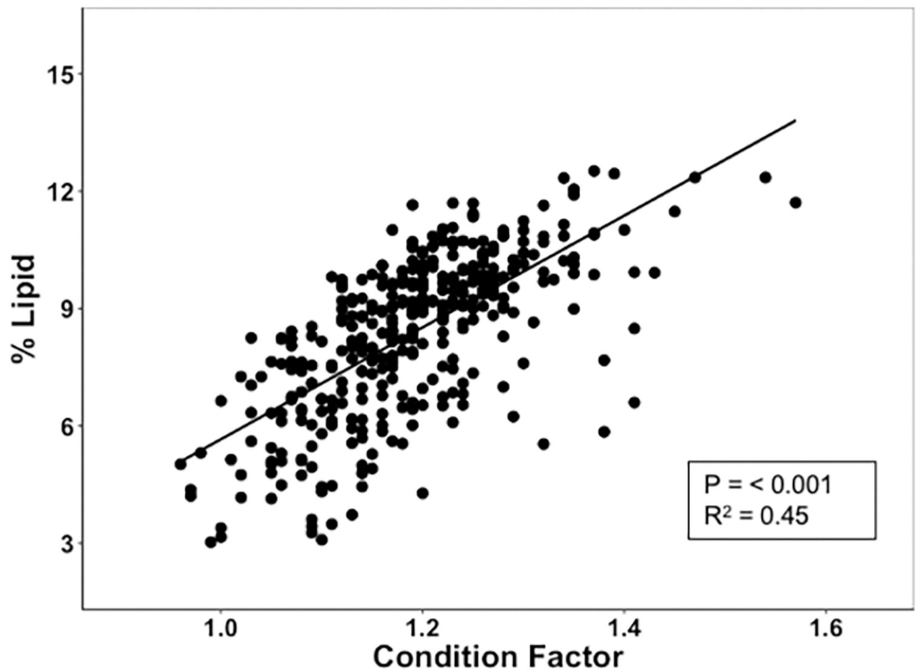


Figure 95. **Condition factor of Puntledge summer Chinook sampled at the Eicher Screen Sampling Centre between 2010 and 2017 (Source: revised data from Guimond 2018).**

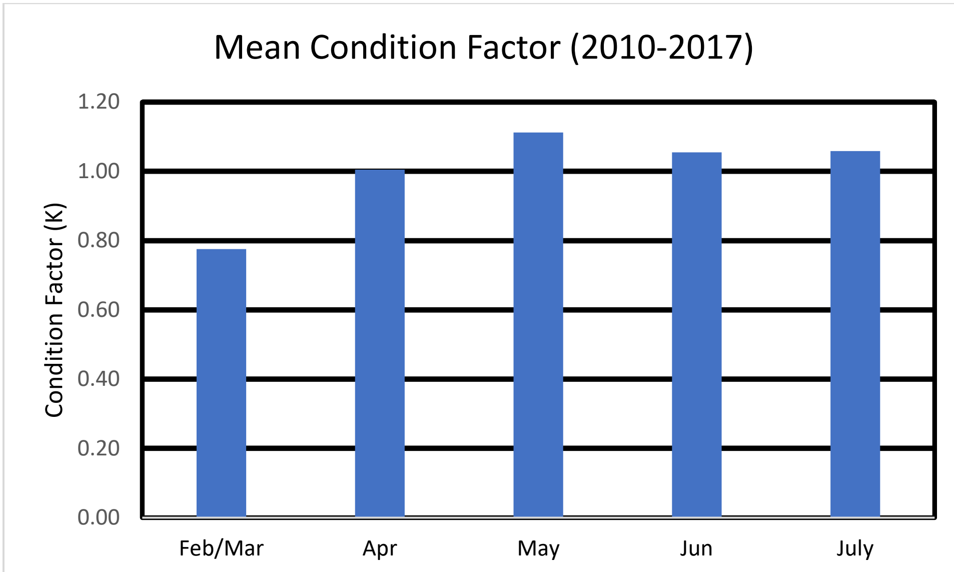
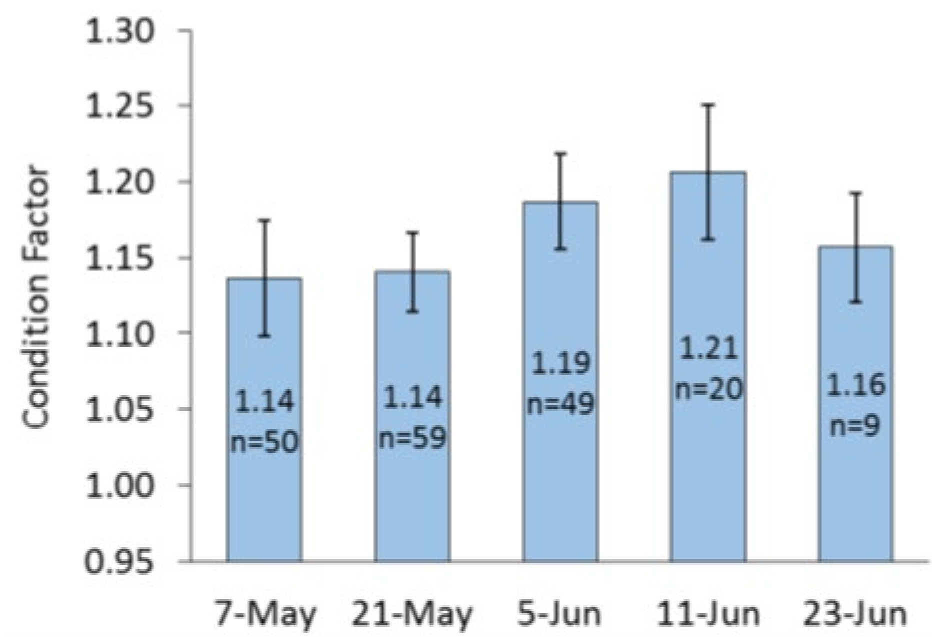


Figure 96. **Mean fork lengths (mm) of wild Chinook fry sampled from 70.2 Mile Trestle and Stoltz Pool index sites in 2014. Error bars represent 95% confidence intervals for each sample mean.**



Overall, growth of summer chinook appears normal compared to the Cowichan River. This is interesting given the Cowichan River is ranked as a productive stream with a chinook stock that is primarily wild and returns have been increasing in the last 15+ years whereas the Puntledge River is considered an oligotrophic stream and ranks low-to-moderate in stream productivity compared to other Northwest Pacific coastal streams (Ptolemy pers. comm. 2024). Furthermore, the Puntledge River has low Nitrogen:Phosphorus (N:P) and alkalinity, which are indicators of food productivity.

In recent years, summer Chinook escapement has been low (i.e., <100 female) undoubtedly resulting in low fry abundance suggesting that food availability is unlikely a limiting factor for juveniles in Reach B. However, this may not be the case for summer Chinook juveniles that migrate and rear downstream in Reach C and D where competition is expected to be substantially higher with fall Chinook juveniles that were produced from an average escapement of 5,000 spawners.

4.3.5.1. Reach B

Habitat for Chinook fry is generally limited to the stream margin and would be ranked as 'high' suitability in areas with gradual sloping banks and instream cover, and as 'low' suitability in areas with steep banks and no instream cover. In summary, a total of 8.4 km of stream margin was visually assessed and 2.86 km ranked low, 2.96 km ranked moderate, and 2.58 km ranked as high habitat suitability for Chinook fry. Cover in the form of LWD (1,733 pieces identified) and boulders (5,701 identified) is present in moderate abundance, although these cover components are often located at depth, and in habitats that appear to be in higher velocity and would not be heavily utilized by Chinook fry (Map 7). Furthermore, snorkel observations (Guimond and Sheng 2016; Harwood *et al.* 2018) found that newly emerged Chinook fry (approximately 40 mm fork length) occupy shallow habitat (<1.25 m deep) with aquatic and emergent vegetation, as well as SWD (Christmas tree clusters). Since construction of the diversion dam, although there is now more wetted habitat, the stream banks are predominantly steep, water depth is >1 m and water velocities often exceed the sustained swimming speed of fry (i.e., 0.12 m/s to 0.16 m/s), especially on the left-side of the river.

4.3.5.2. Reach C

Aerial imagery results obtained from the RPAS survey indicated that aquatic habitat within Reach C is complex in comparison to Reach B, with smaller mesohabitat units (Faulkner *et al.* 2021). Reach C is comprised of 25.8% riffle, 28.7% cascade, 16.5% glide, 21.7% run, 4.0% falls, and 0.4% unknown (this area was located in a 'no fly zone'; Faulkner *et al.* 2021). Gravel enhancement activities are visually evident in the first habitat unit below the diversion dam and within and adjacent to the Bull Island Side Channel. Spawning gravel additions at Barber's Pool were less obvious from the aerial imagery. A total of 15.43 km of stream margin was visually assessed for depth, cover and velocity to establish habitat suitability rankings for Chinook fry rearing. In summary, 3.74 was ranked as low suitability, 5.74 was moderate suitability and 4.68 kms high suitability for Chinook fry habitat. A total of 1.28 km of margin habitat was in 'no fly zones' and was therefore unclassified. An example of habitat rankings (red/orange = low; green = moderate; blue = high) are displayed in Figure 97. The moderate suitability habitat (green line) has a fairly steep sloping bank with less cover than higher suitability areas identified with blue lines. The cascade habitat (orange line) is ranked as low suitability due to high velocity and turbulent flow.

Reach C rearing habitat quality has likely increase post dam operation due to the regulation of flow through this reach which is maintained at minimum discharge of 5.7 m^3/s. Higher discharges up to approximately 20 m^3/s from Reach B are diverted into the Hydro penstock at the diversion dam for power generation and returns to the river near the top of Reach D.

Cover in the form of LWD (772 pieces identified) is present in low abundance, except for Bull Island Side Channel, where there is large jam. Boulders are present in moderate to high abundance (39,961 identified). The distribution of boulder cover is even, except for the ~ 1 km long cascade section upstream of the Highway bridge crossing, which is predominantly bedrock cascade habitat with less boulder cover.

Habitat enhancement with LWD and boulder complexing could offer some benefit to Chinook fry by provided additional cover in areas that may otherwise be lacking. Anchoring of structures would be feasible with abundant bedrock substrate. This type of enhancement would need to consider the Puntledge River's high recreational use (e.g., swimming, kayaking) and potential hazard to navigation imposed by instream structures, which is beyond the scope of this assessment.

Figure 97. Chinook fry habitat suitability rankings applied to RPAS imagery collected in Reach C midway between the diversion dam and Nib Falls.

Note: Habitat rankings are indicated as red/orange line = low; green line = moderate; and blue line = high.

4.3.5.3. Reach B Riparian Habitat

The total area encompassed by the 100 m Riparian Management Area (RMA) of the Puntledge River in Reach B is 108.5 ha, and the total area within the 30 m Riparian Reserve Zone is 52.7 ha (Table 46). The area of land cover types are provided in Table 46. A review of the Ballin *et al.* (2017) land cover dataset indicated that some features such as wetlands are not mapped accurately at the site scale. Therefore, these results should be considered indicative of the relative proportion of each habitat type, but not used for site-specific planning. For example, bank classification in Reach B has been conducted for 8.04 km of bank (both sides of the river) downstream of the Comox Dam and the amounts and proportions of habitat by class are:

- 4.27 km forested (53.1%);
- 2.91 km marsh (36.2%); and
- 0.86 km unvegetated/modified/cleared/potential erosion (10.7%).

Table 46. Land cover within the RRZ (30 m) and RMA (100 m) of Puntledge River Reach B according to Ballin *et al.* (2017).

Land Cover[1]	Riparian Reserve Zone (30 m)	Riparian Management Area (100 m)
Riparian	24.17 ha	65.01 ha
Wetland	12.81 ha	29.16 ha
Development/Disturbed	1.21 ha	5.36 ha
Unclassified[2]	21.38 ha	29.51 ha

[1]Riparian, Wetland, and Development/Disturbed classes may overlap, as such the area values do not add to equal the total area within the RRZ or RMA.

[2]Unclassified area is generally comprised of river area not considered riparian and second growth forest that is not within the modelled riparian extent.

4.3.6. Decreased Quantity and Quality of Rearing Habitat

Hydraulic transect data taken for the Puntledge River Watershed WUP spawning habitat analyses for summer Chinook in 2002 (Burt 2002; Figure 98) were re-examined by Ptolemy using standardized habitat suitability curves (Figure 99; Ptolemy pers. comm. 2022). Wetted usable area (WUA) was determined for emergent summer Chinook fry using select transects from Reaches B, C, and D (Figure 98) to develop WUA curves based on depth and velocity at specific river discharges (Figure 100 to Figure 102).

A summary of the emergent summer Chinook analyses conducted by Ptolemy (pers. comm. 2022), is provided as follows:

- Transect cells nearest the stream margin or mid-channel bars provided the highest usable width for fry.
- Cell resolution across the stream width was high with upwards of 27 verticals per transect (a vertical is data from a tape station with depth and mean velocity recorded).
- All three reaches (B, C, and D) show a trend of declining usable width with increasing flow with usable width for emergent Chinook fry being generally less in Reach D (Reach B example, see Figure 100). Natural Long-Term (LT) MAD is higher in Reach D and includes the return flows from diversion.
- Width into mean depth ratios (>300) for broad riffles gave high usable widths regardless of changing flows per transects.
- Transect PT08 had the greatest usable width at 20 m over the flow range 2 m^3/s to 4 m^3/s. The lowest usable width of the transects examined came from Transect PT05 at 5.2 m. Both transects are from Reach C.
- In Reach D, Transect PT17 had the greatest usable width at 12.3 m at a flow of 15.6 m^3/s. The lowest usable width of the transects at lowest flow examined came from Transect PT20 at 3.1 m. All transects displayed very low usable widths (1.5 m to 6.0 m) at the highest flow of 25 m^3/s.
- Both increasing depth and velocity with increased flow limited the potential distribution of emergent Chinook salmon fry.
- Since preferred habitat conditions are not static and change with increasing fish size, the application of the results should be time-limited to the early part of the growth season after emergence.
- Wetted width changes with flow were minor; should you be looking at wetted perimeter methods.

Figure 98. Location of hydraulic transects on the Lower Puntledge River used to determine summer Chinook spawning capacity in 2002 (Burt 2002). Red lines across the river are transects used in analysis.

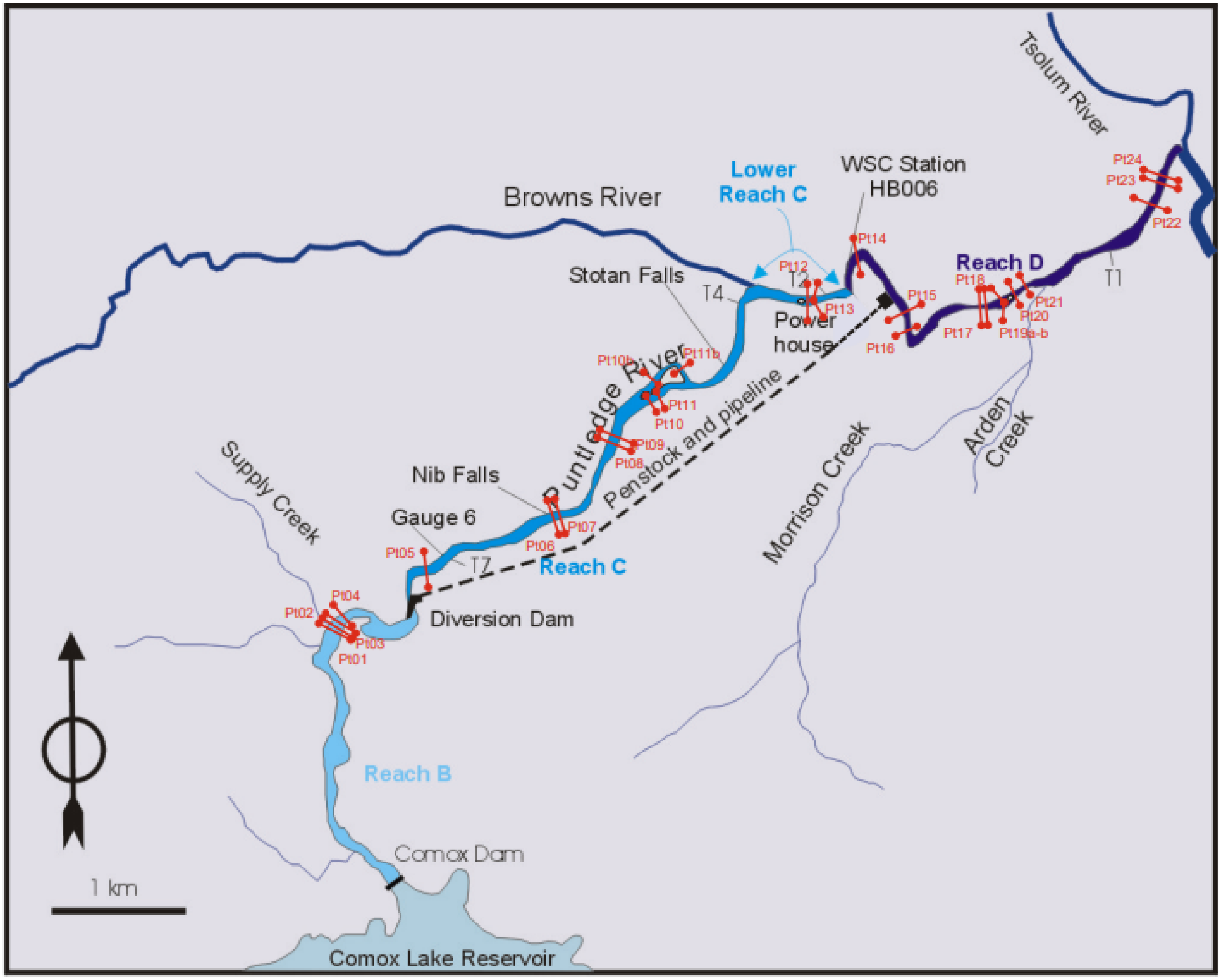


Figure 99. Depth-velocity curves used for emergent Chinook habitat suitability analyses (Ptolemy pers, comm. 2022).

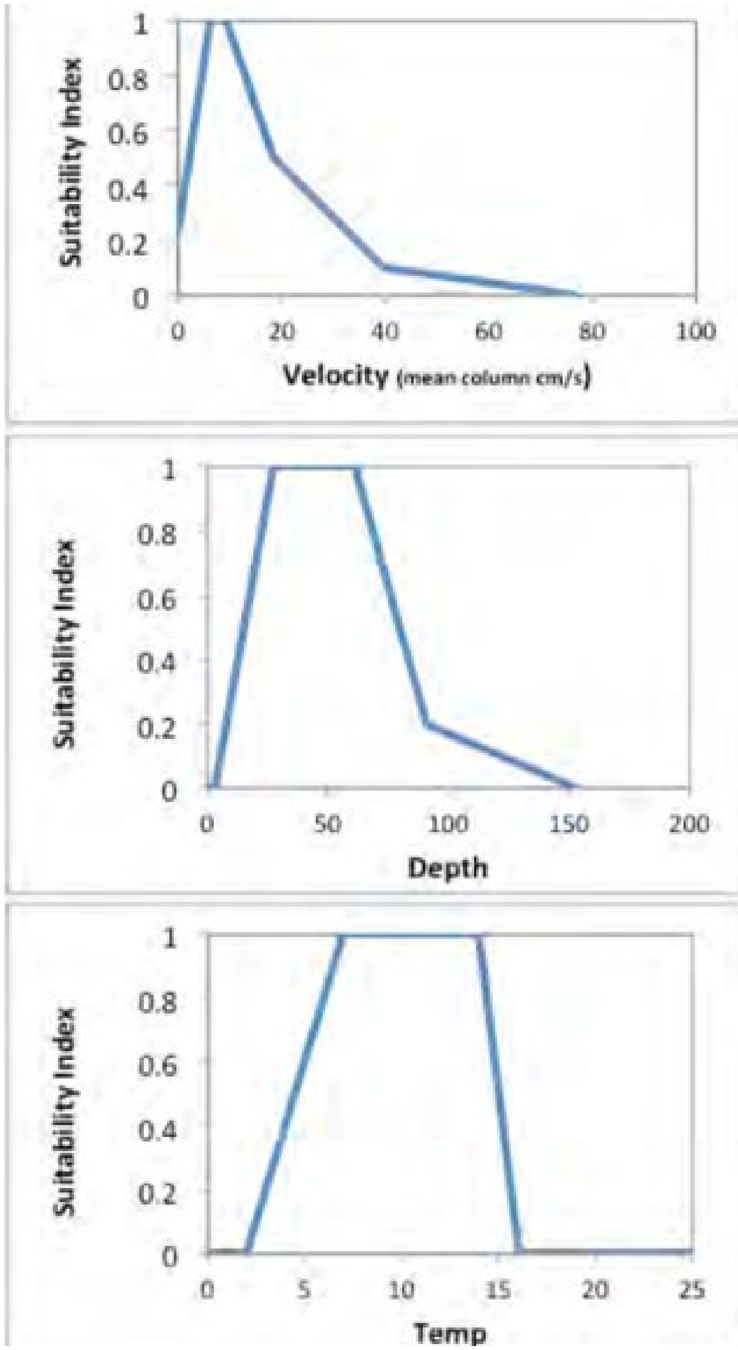


Figure 100. Discharge, usable width, and flow at transects Pt01-04 of the Puntledge River in Reach B (Ptolemy pers. comm. 2022).

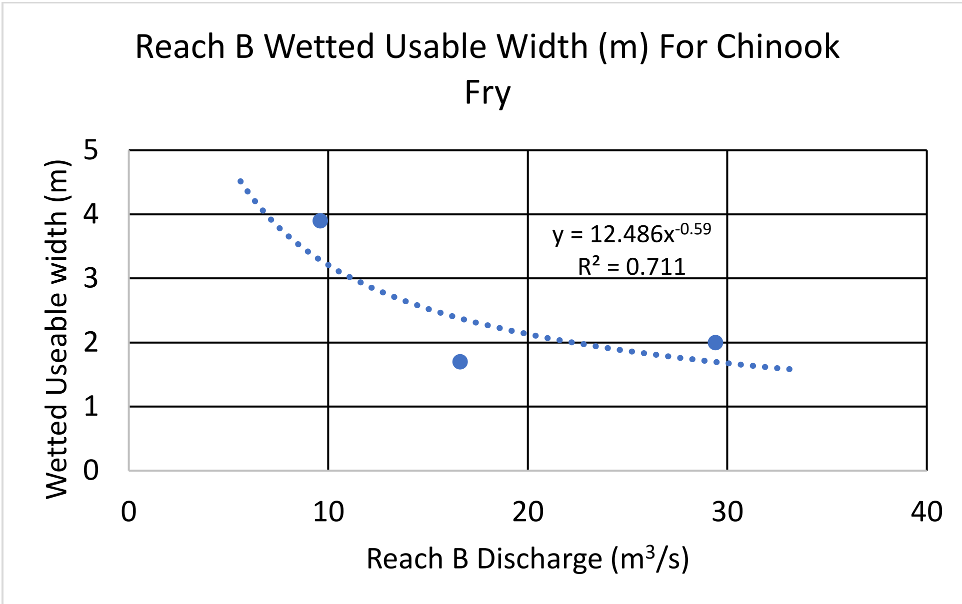


Figure 101. Usable width for young Chinook fry in Reach C of the Puntledge River at four different transects (Source: Ptolemy pers. comm. 2022).

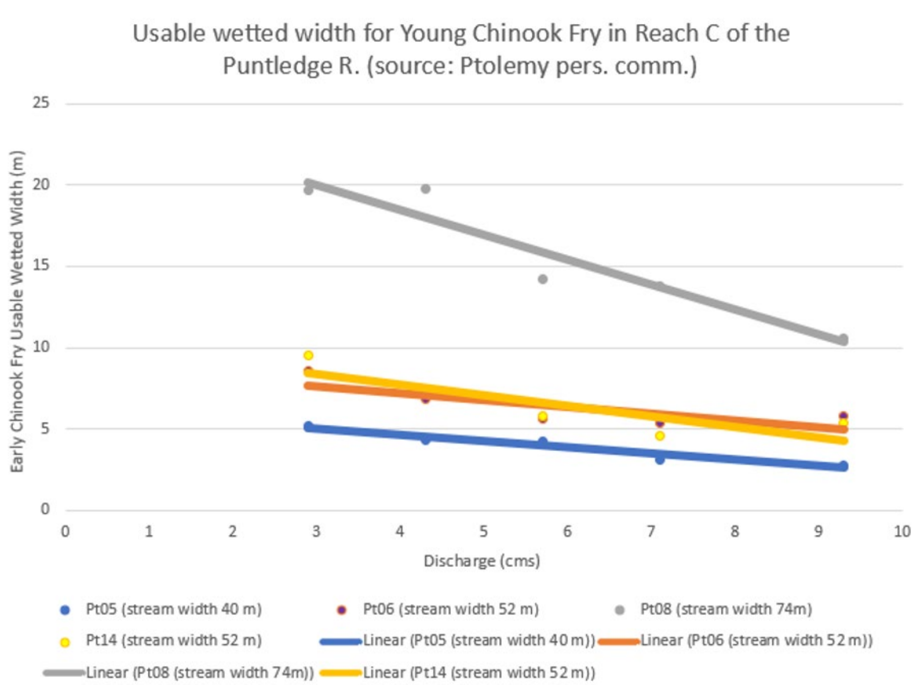
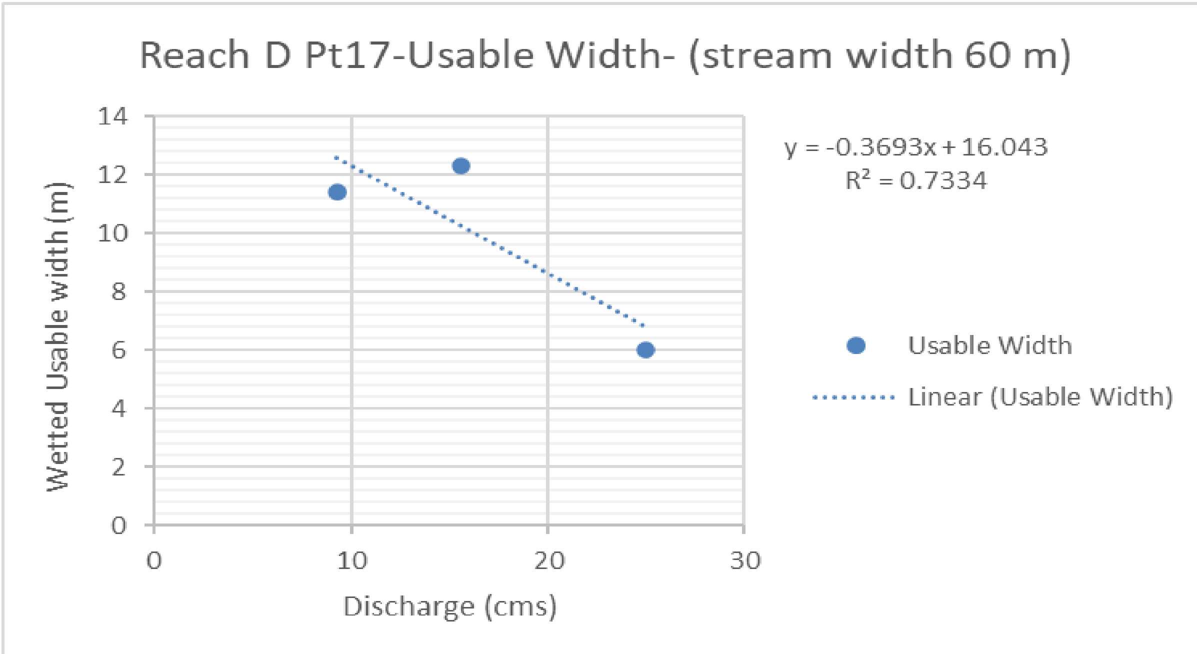


Figure 102. **Usable width for young Chinook fry in Reach D of the Puntledge River at transect PT17 (stream width of 60 m) (Source: Ptolemy pers. comm. 2022).**



Ptolemy (pers. comm. 2022) suggested that an ideal flow for emergent Chinook fry tied to flooded in-channel riparian habitat is near 20% of LT MAD (Long Term, Mean Annual Discharge) depending on channel confinement. The natural LT MAD for the Puntledge River in Reach C is 32.7 m^3/s. LT MAD of 20% is equivalent to 6.54 m^3/s. A lesser flow is required if the stream width is significantly smaller than 34 m. In general, fry habitat is best at 10% LT MAD or 3.3 m^3/s. Suitable hydraulic conditions for emergent fry are nearer 3.45-4.4 m^3/s (Burt 2002). A closer examination of Griffith and Burt's instream flow study (Burt 2002) at finer depth-velocity interval could be conducted to verify suitable flow conditions.

The empirical biomass data (Allen Plot) for the Puntledge River suggests that the upper biomass limit for Chinook or steelhead is near 143 g/100m^2 per age or size class (Ptolemy pers. comm. 2022). Data collected in 2016 in the Puntledge River measured maximum densities of 0.6 fry/m^2, which is much lower than Ptolemy's empirical estimate (Guimond and Sheng 2016). In comparison with other coastal streams in BC, Puntledge River biomass productivity ranks in the lower most quartile (Ptolemy pers. comm. 2023).

Based on Ptolemy's generic maximum biomass equation, for a total alkalinity of 20 mg/L in the Puntledge River, the equation $36*(Alk)^{0.62}$ yields an estimate of 230.6 g/100 m^2 (Ptolemy 1993). Therefore, for emergent Chinook fry with a 38 mm FL weighing on average 0.5 g, the watershed can achieve a maximum density in suitable habitat of 230.6 g/0.5 m^2 or 5.3 individuals per m^2. If this maximum is applied to the suitable habitat calculated from 2002 Puntledge River, habitat transects in Reach B, C, and D at base flows of 21.3 m^3/s, 6 m^3/s (regulated flow), and 21.3 m^3/s, respectively, a theoretical capacity estimate can be calculated for the base discharges in each of the reaches. Emergent fry capacity estimates are 42,400 for Reach B, 262,880 for Reach C, and 125,875 for Reach D. Reach B is the only reach that is restricted for summer Chinook use (Table 47). Most of the summer Chinook spawners utilize the man-made spawning platform located in the lower third of the reach so it is likely that only a portion of the available Reach B habitat has been utilized by the emergent fry. Since 2021, a second spawning platform was constructed at the top of Reach B 300 m below Comox Impoundment Dam, which now potentially allows emergent fry to utilize all of Reach B.

Table 47. Calculated emergent fry rearing capacity in Reaches B, C, and D.

Reach	Reach length (m)	WUA Bank width (m)	Potential Emergent Fry Capacity (5.3 fry/m^2)
B	4,000	2	42,400
C narrow	1,600	6	50,880
C wide	4,000	10	212,000
D	4,750	5	125,875
Total	**-**	**-**	**431,155**

Incubation assessments conducted on the first spawning platform in Reach B after construction in 2005, which was the main spawning habitat available in the reach attained an overall Jordon incubation box survival rate of 92% (Guimond 2005). If a conservative overall survival rate of 30% for wild spawners is applied to the platform, 100 summer Chinook females could produce 120,000 emergent fry (Table 48). However, a reassessment of the incubation survival rate should be conducted to confirm the platform is maintaining good incubation conditions.

Observations conducted by boat and snorkel surveys during an assessment of conifer tree bundles installed along the riverbanks in prime locations in Reach B in 2016 indicated that velocities higher than BC Hydro's base discharge (i.e., 21.3 m^3/s or 65% of MAD) resulted in water velocities and depth that reduced habitat use for Chinook (Figure 103). Based on snorkel surveys conducted in 2016, velocities appeared too high for emergent fry to find refuge, indicating that estimates calculated using habitat suitability curves and hydrologic transects likely overestimate emergent fry capacity (Guimond and Sheng 2016).

At a spawning escapement of 200 adults (i.e., 50% female) emergent fry habitat in Reach B is limited if the discharge is sustained at 100% of MAD. In 2016, during an emergent fry habitat study, the river, exceeded 200% MAD for extended periods of time. Under this circumstance emergent fry would be displaced downstream, be at risk of impingement in the Eicher Screen, and/or be forced to seek refuge in Reach C, which is a higher gradient section largely composed of bedrock with limited shallow low velocity stream margins.

The summer Chinook fry population migrating downstream of the diversion dam was estimated during Eicher Screen efficiency assessments between 2010 and 2017. In all study years, high discharge and/or BC Hydro shutdowns disrupted sampling for several days each year, lowering confidence in the population estimates. However, despite this problem, when examining the mean daily discharge each year for the period between the 47th to the 65th Julian Day (i.e., the estimated mean period for most of the fry emergence) discharge ranged between 16.55 m^3/s and 63.6 m^3/s or 50.6% MAD to 195.4% MAD (Figure 104). The correlation between the population estimates and discharge or MAD was R^2= 0.853 indicating that higher discharge had a negative impact on the population estimate. It is unknown if this is a result of a decrease in sampling efficiency and/or an increase in the number of fish perishing (e.g., Eicher screen mortalities or predation on displaced fry.

Based on the chart equation, using Reach B fry population estimates between 2010 and 2017, as discharge increases to between 100% MAD and 200% MAD (i.e., 32.7 m^3/s to 65.4 m^3/s), the fry population estimate decreases from 30,000 to 11,345, which is even lower than the WUA estimated ranges. Overall, the graphed field data estimates have a similar negative linear relationship as the Reach B WUA for discharges over 100% MAD (Figure 100). The field data also suggests that the fry population decreases slowly as discharges over 100% MAD increases and that the population increases exponentially as discharges decrease below 100% MAD. This suggests that at high discharges, fry are restricted to rearing close to the riverbanks but are then able to utilize habitat exponentially further away from the banks as discharges decreases.

The emergent summer fry displaced into Reach C would be in competition with fall Chinook cohorts (present in Reach C and D), which are larger at emergence and more numerous. Fall Chinook are enhanced in the Puntledge River, and the escapement routinely reaches a spawning capacity of 5,000. This could potentially result in the production of 2 million fall fry, which is potentially over 3 times the capacity of the lower watershed (Table 49).

Table 48. **Estimated emergent fry production for the spawning platform in the Puntledge River based on estimated fecundity, incubation survival, fry produced/female, and total female spawners.**

Parameter	Value
Fecundity	4,000
Incubation survival	0.3
fry produced/female	1,200
Total female spawners	100
Total emergent fry production	120,000

Figure 103. **Puntledge River mean hourly discharge for Reach B (CMC sluice gate discharge), penstock discharge (turbine flow), and river temperature from March 1st to April 30th 2016, showing snorkel survey dates (Source: Guimond 2018).**

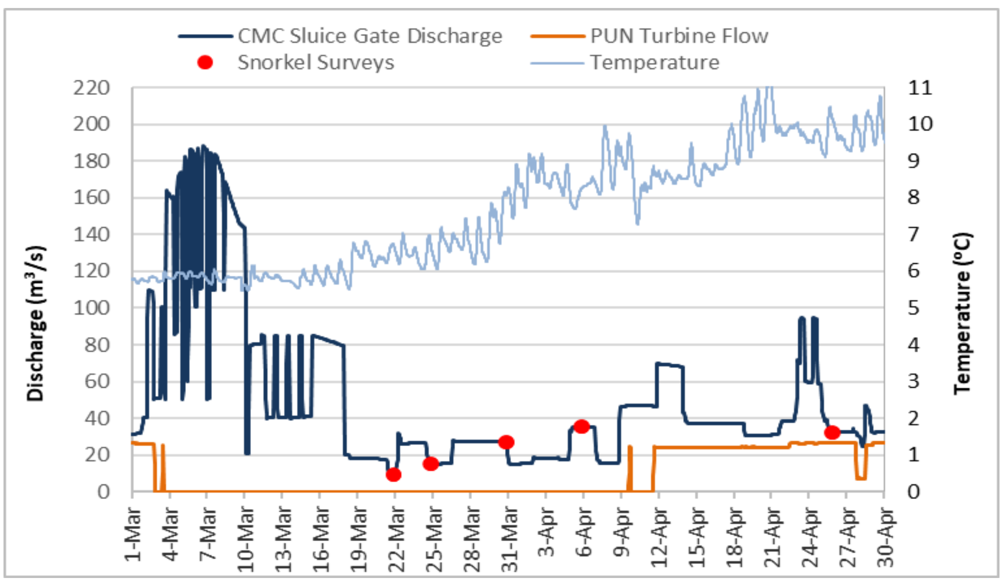


Figure 104. **Summer Chinook fry population estimate and discharge in the Puntledge River by Julian date.**

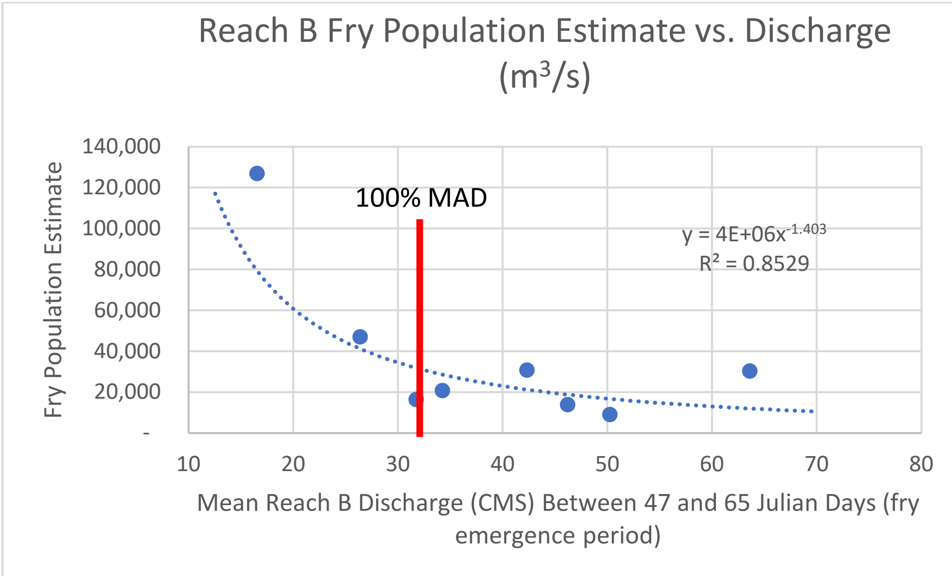


Table 49. **Estimated emergent Fall Chinook fry production in the Puntledge River based on escapement, number of females, fecundity, and egg to fry survival.**

Parameter	Value
Fall CN escapement	5,000
Females	2,500
Fecundity	4,000
Egg-to-fry survival	0.2
# Emergent fry	2,000,000

Overall, this review of the emergent summer fry habitat capacity using the 2002 WUP transects demonstrates that Reach B is unlikely to accommodate many emerging fry under BC Hydro's base flow conditions. Fry that move downstream (either voluntarily or displaced downstream due to increases in river discharge above 20% MAD) become increasingly at risk when reaching the Eicher screen, especially the smaller fry. Reach B has reached discharges of 100% MAD or higher during 57.4% of the emergence fry period (i.e., Jan 27th to Mar 31st based on the approximate date that emergent fry reach 1,000 ATUs plus 27 days buffer) and 60% MAD 84.3% of the time in last 30 years.

Furthermore, fry displaced into Reach C and D are in high competition with enhanced emergent fall fry, which are more numerous and larger in size at emergence. Between 1993 and 2022, Reach D reached discharges of 100% MAD or higher 68% of the time during of the emergence fry period (i.e., Jan 27th to Mar 31st based on the approximate date that incubating fry reach 1,000 ATUs plus 27 days buffer) and 60% MAD 91% of the time. MAD discharges are further exceeded in the lower parts of Reach D due to additional flow inputs from Browns and Tsolum Rivers.

Reach C, which operates at a minimum (three-day rolling average) discharge of $5.7m^3/s$ (i.e., 17.5% MAD), is likely the most important area for summer Chinook early fry rearing (Marshall 1975). A MAD of 20% or less is considered an appropriate higher range rearing flow for small fry. In Reach C, between 1993 and 2022, 20% MAD was maintained 36.3% of the time during the emergent fry period, 86.8% of the time at 60% MAD and 9.6% of the time at 100% MAD (Table 50).

Table 50. Occurrence of %MAD in Reach B, C, and D during the fry emergence period (i.e., Jan 27th to Mar 31st based on fry development reaching 1,000 ATUs) between 1993 and 2022.

	Reach B	Reach C	Reach D
%MAD	% Occurrence	% Occurrence	% Occurrence
20=/<	0	36.3	0
60=/>	84.3	13.2	91
100=/>	57.4	9.6	66

The current BC Hydro Eicher Screen diversion dam spill strategy focuses on increasing river discharge when Eicher screen efficiency is low. This is so that overall bypass efficiency over the diversion dam can theoretically be increased to over 95% by increasing the number of fry that get diverted directly over the diversion dam instead of being diverted through the Eicher Screens. This approach potentially increases the displacement of smaller fry in Reach B and C, exacerbates impingement issues at the Eicher screens for small fry, decreases the amount of available emergent fry habitat in Reach C, and increases competition with summer and fall emergent fry in Reach C and D. Since the implementation of the strategy in 2020, the occurrence of flows suitable for emergent and small fry (i.e., 20% MAD) has decreased from 36% between 1993 and 2019 to 1% since 2020. For flows 30.5% MAD and 44% MAD or less, these occurrences have decreased by over two times (Table 51).

Table 51. Comparison of the Occurrence of % MAD in Reach C between 1993 and 2019, and 2020 and 2023.

Condition (m^3/s)	% MAD	1993-2019	2020-2023
		% Occurrence	% Occurrence
<=6.54	20	0.36	0.01
<=10	31	0.78	0.29
<=14.5	46	0.87	0.28
>=19.7	60	0.11	0.03
>=32.7	100	0.07	0.04

For fry entrained into the penstock and unsuccessfully diverted by the Eicher screens, the injuries and mortalities (both direct and latent) will occur when passing through the BC Hydro power plant. In 1955, tests conducted at the Puntledge powerhouse found mortalities of juveniles passing through Francis turbines ranging between 30% and 40% (Marshall 1973). Based on assessment at other facilities with similar turbines the mortality rates average approximately 30% (Pracheil *et al.* 2016, see Section 4.3.1.3).

The likely accumulative negative impacts on Chinook fry caused by the latest Eicher Screen flow strategy may negate the overall benefit to summer Chinook survival compared to the previous operational strategy. It is possible that the latest strategy has a negative effect on overall fry survival and possibly creates genetic and adaptive consequences. The displacement of fry early in life history likely results in a higher percentage of fry forced to rear in the estuary and a smaller proportion rearing in fresh water. Studies at the Cowichan River have demonstrated that the majority of juveniles rear for an extended period in freshwater to a size of approximately 70 mm and contribute to adults returns at a much higher rate than fry that migrate earlier and rear in the estuary.

4.3.7. Freshwater and Estuary Habitat Utilization due to River Flow

In the Puntledge River, based on fry-juvenile enumeration data collected between 2011 and 2017 at the BC Hydro Eicher Screen Assessment Centre, natural-origin summer-run Chinook juveniles were found to follow a bimodal pattern in outmigration with the largest proportion (i.e., <50 mm fork length) entering the estuary between the end of February to the beginning of May, and a smaller proportion (i.e., >60 mm) entering the estuary around June (Lister 1968; Guimond *et al.* 2014). Emergent fry <50 mm, in some years, exceed over half of the total population migrating downstream, despite a significantly high entrainment mortality and injuries at the BC Hydro diversion dam (Guimond and Taylor 2014). Hatchery origin Chinook, by contrast, are all released in late May at >65 mm from the lower river and are therefore not impacted by the diversion dam. Although there is much smaller contribution of natural-origin adults compared to hatchery-origin adults in the total escapement, the natural-origin returns are critical to providing additional genetic diversity and maintaining adaptive potential in the overall small returning population (Wetklo *et al.* 2020).

There is limited information on the use of the estuarine environment and how it may differ between the two predominant migratory phenotypes within the summer Chinook population (emergent fry and subyearling smolts). Ocean type Chinook rely on the estuarine and nearshore marine environments to accelerate their growth (Healey 1971; Healey 1991; Chalifour *et al.* 2021). In the Nanaimo River, Chinook were found to grow over 1 mm a day in the estuarine environment, underscoring the importance of this habitat for smolt rearing (Healey 1971). Differences in estuary use amongst returning adults may explain some of the trends in smolt to adult survival between natural origin and hatchery origin Puntledge River summer Chinook.

In 2021, a study was conducted to compare the estuary entry and residence time for Puntledge Summer Chinook by analyzing otolith microchemistry of both natural and hatchery origin summer Chinook adult returns, to identify key life-history differences between these two groups, and the percent contribution of the different life history strategies (Quindazzi 2023a). Technological advances in otolith microchemistry analysis provide a powerful method for determining fish migratory pathways at the individual level. Due to its metabolically inert structure, otoliths (fish ear stones) incorporate elemental ions from the surrounding environment onto their growing surface, providing a chronological record of the environment to which the fish has been exposed to during its lifetime (Campana 1999). In particular, the element strontium (Sr), which is generally found in low concentrations in freshwater, is an intrinsic part of the salt concentrations within seawater. As overall strontium concentrations increase with salinity, strontium is substituted in the calcium carbonate lattice of the otolith, and the strontium/calcium (Sr/Ca) ratio profile of the otolith can show transitions from freshwater to saltwater habitat (Kalish 1990).

A total of 96 otoliths collected from 2021 brood Puntledge summer Chinook adults were analyzed by Laser Ablation Inductively Coupled Plasma Mass Spectrometry (LA-ICP-MS). Of these, 32 were natural origin and 64 were hatchery origin returns. Natural origin Chinook were found to enter the estuary at a smaller size on average (53.4 mm versus 59.4 mm for hatchery origin). Approximately 46% of the <50 mm natural fry that entered the estuary utilized the estuary for an extended period. Extended estuary use was defined as individuals with a predicted growth of above 10 mm in the estuarine environment. Natural origin fry 50-60 mm utilized the estuary to a far lesser extent (i.e., 14.3%) (Table 52). Natural origin Chinook entered the estuary between 30.9-82.0mm, which was a greater range than the hatchery origin Chinook (47.3-73.9 mm). None of the hatchery origin Chinook used the estuary for an extended period. In contrast, extended estuary use by <50 mm natural origin Chinook resulted in faster growth and attained a size similar to hatchery origin fish once both finally migrated from the estuary into a fully marine environment, (i.e., 60.1 mm for natural origin versus 62.5 mm for hatchery origin).

Overall, 41% of the natural origin Puntledge summer Chinook entered the estuary at <50 mm. In contrasts, between 2014 and 2016, on average, 19% of the fingerlings in Cowichan River entered the estuary at <55 mm, 63% entered at a size range between 55-75 mm, and 18% entered as yearling smolts (>75 mm) (Atkinson 2023) (Table 53). Similar to Puntledge, otolith analyses of Sarita Chinook adult returns pooled from between 2015 and 2021 clustered into two main estuary entry groups. The first group entered the estuary at around 45 mm and the second entered at around 60 mm (Quindazzi 2023b). Based on an "eye-ball area estimate of the violin plot for the two-size group in the 2023 Quindazzi report, approximately 45% entered at 45 mm, 45% at 60 mm, and 10% entered at >60 mm.

Three main characteristics of the Puntledge River watershed potentially affect the time and size of natural origin juvenile saltwater entry. As noted earlier, a proportion of downstream migrating emergent fry are entrained and perish at the diversion dam intake. This potentially affects the time and size of when the remaining summer Chinook fry migrate to the estuary. If the mortality at the diversion dam could be eliminated or reduced, this would increase the number of remaining summer fry and would likely increase the number migrating to the estuary. In Section 4.3.5, it was shown that fry rearing capacity declines rapidly as discharge increases from 50% to 100% MAD. BC Hydro flow regulation through Reaches B, C and D could therefore impact the quantity and quality of available rearing habitat by increasing discharge. The recent implementation of the BC Hydro spill ratio rule (See Section 4.3.2 – Spill ratio (fish/m^3) in water moving over diversion dam relative to penstock) to reduce entrainment of emergent fry can result in increased flows during this critical period of outmigration and may force more fry to displace downstream in the lower river and estuary earlier than under lower hydraulic conditions. The third factor that would also likely alter the time and size of summer fry migrating to the estuary is competition with fall fry, originating from the large return of hatchery spawners (i.e., >5000). Fall fry emerge larger, are likely more territorial and would outcompete summer fry.

It should be noted that fork length back-calculations through otolith microchemistry can be difficult and prone to error (Thibault *et al.* 2010). The back-calculated fork length sizes for Puntledge summer Chinook at ocean emergence were based on a fork length/otolith radius baseline developed from Chinook in the Nass River and several lower Fraser River systems. Therefore, caution should be taken when assessing the exact size at outmigration using this non-Puntledge baseline. Fork length reconstructions could likely be +/- 5 mm due to the fork length baseline that was used, as well as the delay time it takes for the barium signal to diminish and the strontium signal to get incorporated in the otolith (Atkinson pers. comm. 2024). In other words, the calculated lengths for each of the estuary entry periods are relative to each other; however, the absolute lengths are not relative to each other. A comparison of back calculated fork lengths for Sarita River Chinook using three different relationships found that lengths based on the wild Sarita juvenile measurements were 19% smaller than those based on Puget Sound (LaForge and Quindazzi 2023). Efforts to obtain a stock specific otolith base size to fork length regression to reduce potential error are in progress (Quindazzi pers. comm. 2023). Summer Chinook juveniles will need to be captured in Puntledge River.

Table 52. **Number of natural origin and hatchery origin Puntledge summer Chinook salmon that displayed extended estuary use, for the three life history categories based on predicted estuary entry size, <50 mm, 50-60 mm, and >60 mm. These sizes were selected from previous studies that describe the two different size classes in the out-migrating fry.**

Life History Category	Fish Size (mm)	No Estuary Use	Extended Estuary Use	Total	% of Total in the Life History Category Using the Estuary for an Extended Period	% of the Total of Fish[1]
Natural Origin Adult Otolith Analyses						
	<50	7	6	**13**	46.2	40.6
	50-60	6	1	**7**	14.3	21.9
	>60	12	0	**12**	0	37.5
Total		**25**	**7**	**32**	**21.9**	
Hatchery Origin Adult Otolith Analyses						
	<50	3	0	**3**	0	4.7
	50-60	31	0	**31**	0	48.4
	>60	30	0	**30**	0	46.9
Total		**64**	**0**	**64**	**0**	

[1]Total is 32 for Natural Origin Adult Otolith Analyses and 64 for Hatchery Origin Adult Otolith Analyses

Table 53. **Saltwater Entry Life history phenotype percent contributions to adult return age classes for Cowichan River Chinook for outmigration years 2014-2016 (Source: Atkinson 2023).**

Year	% Fry <55 mm	% Fingerling 55-75 mm	% Yearling >75 mm
2014	15	68	16
2015	17	58	25
2016	24	64	13
Average	**18.7**	**63.3**	**18**

While there is a strong relationship between otolith Sr/Ca ratios and variable saltwater salinities, it is difficult to reconstruct the spatial and temporal behavior of fish in estuarine environments without a greater understanding of the various salinity gradients within the lower Puntledge River. Estuary Sr/Ca is relatively constant in fully marine environment but can change significantly transitioning from freshwater to full saltwater (Kraus and Secor 2003). The dynamics of the freshwater lens and salt wedge intrusion in the Courtenay River is influenced seasonally by tidal cycles and hydrologic regimes. Reconstructing the rearing history of juveniles through these different environments based on otolith chemistry is contingent on knowing the spatial and temporal extent of chemical variation in ambient water and otolith chemistry. Additional water sample and otolith analysis could provide more insight. Water samples could be collected from mainstem and tributary sites upstream of, and within the estuary, to characterize the chemical gradients in the watershed and correlate biomineralization rates in otoliths to background water chemistries and gain a better understanding where summer Chinook juveniles rear both in freshwater and brackish areas. (Atkinson and Anderson 2020)

BCCF is currently operating PIT tag arrays in Puntledge River. They first propose running lab trials on a range of Chinook juvenile sizes with PIT smaller tags to verify this can be successfully done. If successful, tagging in-river fall and summer Chinook juveniles and then releasing and tracking the movement of the juveniles with the PIT arrays would provide additional information on movement into the estuary and the effects of varying river discharges (Atkinson pers. comm. 2024).

Regardless of the extent of this type of habitat use, the current results are similar to other systems in the Fraser River (Chalifour *et al.* 2021). Emergent Chinook fry that emigrate early in the season and rear in the estuary grow to a size that is comparable to hatchery juveniles that rear in freshwater for months and then migrate to full saltwater. Early estuary rearing juveniles are able to access more productive food resources and grow quickly. PIT tagging studies in the Cowichan River and Bay found that Chinook freshwater growth rates for juvenile Chinook salmon were on average, 0.6 mm/day, compared to 1.0 mm/day in the estuary and bay (Atkinson 2023). However, it is currently unknown if there is a trade-off resulting in lower overall fry-to-adult survival due to higher mortality during the initial period of early rearing in the estuary. This will be investigated in the coming years at Cowichan (Atkinson pers. comm. 2024).

Overall, the findings of this research indicate that natural origin Puntledge Chinook fry produce a phenotypic life-history absent in hatchery origin releases. Early entry into the estuary may be an important life stage that adds to the diversity, adaptability, and overall survival of this endangered population. Additional brood year otolith analyses should be conducted to verify the consistency and prevalence of these early marine life stages and the associated adult age of return. In conjunction with this, field distribution observations on summer Chinook fry-juvenile habitat use and DNA, length/weight sampling in the Reaches of lower Puntledge R. and intertidal areas are needed. This is currently a major data gap. In addition, based on the sensitivity of early emergent Chinook fry to river discharge (i.e., the decrease in quality and quantity of availability of suitable freshwater habitat in Puntledge River as MAD increases from 50% to 100%), brood year otolith analyses should target emergence years where fry were predominantly subjected to these ranges of discharges. BC Hydro is often able to regulate river discharge during this period and therefore could impact fry and juvenile estuary entry timing and size and thus adult survival and age of return (see Section 4.3.5). When comparing the mean discharge during the period of fry-juvenile freshwater rearing in Cowichan River, the number of adult returns from progeny that were subjected to low discharges resulted in significantly higher returning escapements than fry-juvenile seasons that were subjected to medium or high freshwater rearing discharges (unpublished DFO data; Pellet pers. comm. 2024).

The hatchery adult escapements of fall Chinook has averaged 5,000 and could potentially produce two million fry (see Section 4.3.4). This is estimated to be three times the rearing capacity of the lower Puntledge River. Future otolith sampling of both summer and fall Chinook is recommended to determine freshwater and estuary use and potential juvenile interaction between the two populations.

There is evidence that Chinook fry utilizing the estuary for an extended period and consequently growing rapidly and larger tend to return as jacks or 3-year-old adults (Pellet pers. comm. 2024). This phenomenon is similar to Coho jacking, which is often the result of juveniles growing too fast and getting too large early in the freshwater or ocean phase. This should be reviewed as more data is collected. If this is occurring, high river discharge during early freshwater rearing could potentially be driving this outcome.

4.3.8. Decreased Access or Quality of Floodplain Habitat

In the presence of large expanses of accessible floodplain, flooding during the period of fry emergence and in the following weeks create ideal conditions in floodplain for primary and secondary production and ultimately provide an abundant food source for juvenile Chinook (Ahearn *et al.* 2006; Grosholz and Gallo 2006). This is more common in watersheds that experience spring freshets.

The associated increase in temperature is one of the factors that distinguished floodplain habitat from the river habitat. The optimum temperature for growth of juvenile salmon is often reached leading to higher food productivity. Ephemeral floodplain habitat is also important for increased growth of juvenile salmon by maintain ideal flow conditions when the mainstem river is experiencing a variety of high flow conditions. Temperatures from 14°C to 19°C provide optimal growing conditions for juvenile Chinook salmon fed at 60% to 80% of satiation (Marine and Cech 2004; Richter and Kolmes 2005). Zooplankton biomass can be 10–100 times greater in floodplain sites than in river sites (Grosholz and Gallo 2006).

Although growth rates can be very high on the floodplain, fish are at a higher risk of stranding and can be exposed to poorly circulated bodies of water, causing lethal conditions to juvenile salmon. However, juveniles are very mobile, can quickly detect water quality conditions, and move to more favorable habitats (Ahearn *et al.* 2006; Carson *et al.* 2008).

Overall, when juvenile Chinook salmon leave fresh water at a larger size, as seen in fish reared on floodplains, overall survivorship to adulthood is increased (Unwin 1997; Galat and Zweimuller 2001).

The headpond reach which is approximately 3.75 km long, the main spawning area for summer Chinook, is permanently flooded by the Hydro diversion dam. The banks are steep and water depths along the banks are primarily over 1 m. During late winter and early spring flood events, slow moving floodplain habitat is limited in this reach.

A large portion of the lower Puntledge River in Reach C is incised and down cut providing limited opportunities for access to floodplain habitat. Reach C is also regulated by BC Hydro and experiences less flooding than Reach B and D. The SEP Resource Restoration Division (RRD) has assessed opportunities in the entire lower Puntledge River and has restored off-channel habitat in the last 30 years to provide stable spawning habitat for Coho, Chum and Pink salmon, as well as rearing habitat for coho and chinook juveniles. There are 4 main side-channels that can potentially provide off-channel rearing for Chinook and refuge during floods. Starting from the furthest upstream location at the Island Highway is Forbidden Plateau/Island Highway channel on river-right, adjacent on river-left is Bull Island channel, next downstream is Powerline channel on river-right and then approximately 1.3 km downstream of the Hatchery is Jack-Haines channel on river-right. Utilization of these side-channel by juvenile Chinook has been poorly assessed and should be regarded as a data gap. Minnow trap sampling in Forbidden Channel in May 2008, June 2008, March 2017, April 2022 and May 2022, when Chinook juveniles are present in the river, there were no recorded captures of Chinook juveniles in the channel. However, fry access into the channel would be difficult due to high water velocities and beaver dams. Access into the other channels is easier. Monitoring would be required between March and June and DNA analyses would be required to determine if the juveniles are fall or summer Chinook.

The inter-tidal zone in Reach D is the historic floodplain area that has been impacted by training dikes on the right side of the river in the city of Courtenay and Comox Road on the left side, (e.g., Fields Sawmill), which cut off a major intertidal zone. This area covers the lower reaches of Glen Urquhart and Millard Creeks, which is now diked and used for agriculture. Diking has constrained the historic floodplain and has had a permanent impact on available ephemeral floodplain habitat.

Overall, off-channel and floodplain habitat is limited in lower Puntledge River and floodplain flooding is of short duration. The lack of access to floodplain and off-channel habitat potentially leads to a higher percentage of the fry being displaced into the estuary.

4.3.9. Stranding in Rearing Habitats

Operational changes to the BC Hydro Puntledge River facilities, as outlined in the PUN WUP (BC Hydro 2003) included seventeen pulse flows between January and October, to facilitate salmon and steelhead migration. The increased flows down Reach C and D from these scheduled flow releases, as well as any unplanned releases, would also likely increase the wetted area along the river margins, and limited access to off-channel habitat. This could potentially attract rearing juvenile salmonids into ephemeral areas not available during normal base flows, only to become stranded as flows suddenly decline when the pulse flow is terminated. To prevent the risk of stranding fish, BC Hydro adopted ramping rates. For releases from the diversion dam, the maximum rate of change in flow (increasing or decreasing) is 2.8 m^3 per second per hour when river discharge is below 19.8 m^3/s, and no maximum rate of change when discharge is above 19.8 m^3/s (BC Hydro 2003). During previous fish stranding assessments in Reach C and D of the Puntledge River, 18 individual sites were identified as having a high risk of stranding fish. These sites were assessed in July and October during a PUN WUP monitoring program and only one site in Reach D was found to have a high incidence of fish stranding (Hay and Lough 2008).

It is recommended that further investigations be conducted to determine stranding impacts to salmon outmigrants during the spring period. For example, in the Skagit River during the spring, Chinook salmon were most susceptible to pothole entrapment, which is also present in the Puntledge River. Salmonid fry that have recently emerged from the gravel (in the spring) are the most vulnerable due to limited swimming capabilities and they seek refuge along stream margins (Hunter 1992). Once salmon fry grow to 50-60 mm, stranding vulnerability is reduced substantially. As juveniles grow to a more advance stage of development, they transition toward deeper, faster flowing waters gradually over time (i.e., a process known as ontogenetic niche shift).

Altered downstream migration or displacement can also affect overall survival. In the WUP timetable, four of the seventeen 48-hr pulse events of 12 m^3/s are scheduled to occur between mid-January and mid-February, and another four between mid-March and mid-April. Given the smaller size of emergent summer Chinook compared to juveniles observed in July and October, these fish would be at greater risk of downstream displacement or stranding due to fluctuating water levels.

Overall, based on the morphology of the river and diking in the lower river, this risk is considered moderate to low.

4.3.10. Frequent and Higher Peak Flows causing Flushing

Frequent and higher peak flows are expected to particularly affect early rearing habitats for Chinook salmon, which are often located along the margins of rivers (Marshall 1972). This negative effect, which can be caused by hydroelectric facilities, can lead to a reduction in recruitment and a reduction in fish abundance (see Section 4.3.4). Intense spring pulse flows can also potentially reduce Chinook rearing survival indirectly by affecting their food supply. Benthic macroinvertebrates comprise the principal food source of both migratory and resident Chinook populations in streams. However, their density, biomass and species richness are negatively affected by high flow fluctuations (Young *et al.* 2011).

Climate change predictions forecast an increase in winter flows (i.e., until mid-March) and a decrease in spring freshet (PCIC 2011). Overall, this may be positive for early summer Chinook fry and juveniles that require shallow, slow moving wetted habitat for initial rearing. BC Hydro also has some capabilities to moderate flood flows during this time of year (Healey *et al.* 2018).

4.3.10.1. Puntledge River (Reach B – Upstream of the Diversion Dam)

Before the dam was built, when adults spawned in the headpond, juveniles reared in what was identified as ideal habitat for Chinook (Marshall 1973). The low gradient reach was wide, deep, and offered a variety of protected lagoons and bays for fry rearing (Marshall 1973). Natural fluctuations in flow would not have had a significant impact on the rearing conditions of fry in this habitat within this reach. However, after construction of the diversion dam in 1965, when this reach was no longer accessible to adults, the rearing habitat conditions changed (i.e., backwatered to steeper banks) and there were less Chinook fry rearing in this reach.

As indicated in Section 3.6.2.2, SWD was installed in the headpond to attract summer Chinook fry during migration and rearing to increase their survival. However, there was concern that the velocities at the enhanced sites may have been too high for rearing fry. For emergent fry (0.3-0.4 cm in length), the sustained swimming speed is between 0.12 m/s to 0.16 m/s (Ptolemy pers. comm. 2022). However, at higher river discharges (e.g., 40 m^3/s), this velocity would likely be exceeded at many of these sites in Reach B. Velocities at the SWD treatment sites in 2016 varied between 0.07 m/s and 0.35 m/s, with four out of 12 enhanced sites having velocities ≥0.18 m/s (Table 54, Map 7). Overall, the headpond reach is radically different from the rest of the Lower Puntledge River and from what it provided historically for rearing and spawning. For example, mean depth (primarily) and bank velocities exceed conditions for emergent and early developing fry (Grifith 2000; Ptolemy and Lewis 2002).

Table 54. Location and description of small woody debris (SWD) and untreated (UnTr) sites in the Puntledge River headpond (Source: Guimond and Sheng 2016).

Site Name	X	Y	Site Description	Velocity (m/s)[1]	Length (m)	Avg. depth (m)[1]
SWD 01	-125.10263	49.66348	adjacent spawning platform, shallow, marshy area	0.07	15	0.5
UnTr 01	-125.10231	49.66363	Contiguous and downstream of SWD 01	0.3	18	0.5
SWD 02	-125.1017	49.66383	circular arrangement of SWD	0.08	9	0.5
SWD 03	-125.101	49.66381	Some overhanging cover	0.18	11	1
SWD 04	-125.10031	49.66381	extensive overhanging cedar	0.35	12	1.5
SWD 05	-125.09978	49.66358	overhanging cedars	0.35	7.5	1.5
UnTr 02	-125.09947	49.6636	Contiguous and downstream of SWD 05	0.12	15	1.5
UnTr 03 (SWD 06)	-125.09903	49.66343	No SWD installed here. Used as an untreated site.	0.15	15	0.8
SWD 07	-125.09853	49.66313	SWD along log; deep water; trees on bottom	0.13	11	2.5
UnTr 04	-125.09797	49.66296	contiguous and upstream of SWD 8	n/a	15	n/a
SWD 08	-125.09787	49.66283	Overhanging conifers	0.18	13	1
SWD 09	-125.09696	49.66238	No overhanging cover	0.07	13.4	1.8
UnTr 05	-125.09623	49.66194	contiguous and upstream of SWD 10	0.02	15	0.2
SWD10	-125.09601	49.66182	Upper hatchery intake log boom	0.03	13	1.8
SWD 11	-125.09846	49.66196	lagoon at small island	0	12	0.8
UnTr 06	-125.09826	49.66172	contiguous and downstream of SWD 10	0	16	0.8
SWD 12	-125.09407	49.66378	fishway inlet	n/a	n/a	n/a
UnTr 07	-125.0982	49.66267	Right bank near lagoon; shallow	0.09	15	1

[1]Average velocity and depth measured at river discharge of ~33 m^3/s. Both control and treatment sites showed a negative correlation between River Discharge and fry/m (i.e., $R^2 = 0.5068$ and 0.6966, respectively demonstrating that velocity is a key factor. Fry have a sustained swim speed of approximately 4 times the fork length (pers. comm. R. Ptolemy).

4.3.10.2. Puntledge River (Reach C – Diversion Reach)

After construction of the diversion dam in 1965, Reach C became the main juvenile rearing habitat for Chinook salmon in the Puntledge River due to restricted access issues upstream of the diversion dam (i.e., Reach B). Reach C is dominated by bedrock substrate and has a higher gradient and limited habitat for fry rearing. High flow releases by BC Hydro during fry emergence and early rearing further decreases habitat availability. For example, the new Eicher screen proportional flow strategy increases fry bypass efficiency past the diversion dam by increasing river discharge. Consequently, this impacts available rearing habitat by increasing flows and water depth along the margins of the river where emergent and small fry take refuge early in freshwater life history. During these higher flow events, emergent fry are restricted to the immediate river margin and are at risk of being flushed downstream to the lower river or estuary where they are most susceptible to predation and mortality.

Spill events appear to be most critical during March and April when newly emerged fry are present in the river (Marshall 1973). In the past, Lister (1968) noted that the majority of Chinook fry emigrated from the spawning channel during March and April at an average size of 38 mm. A smaller second migration of migrants occurred between late-May until mid-July at a length exceeding 55 mm and averaging 70-80 mm. The spawning channel was protected from the mainstem river and had regulated flows during emergence allowing fry to rear in the channel in the absence of high river discharges. Similarly, more recent observations at the diversion dam intake (i.e., Eicher screen facility) found emergent Chinook fry moving downstream at the commencement of the monitoring period on February 22, 2011 and peaked in the month of March (Guimond and Taylor 2011). This first migration was followed by a second larger migration (70-80 mm average length) between late May and mid-July. These emigration patterns are like those observed in fall Chinook stocks on Vancouver Island (i.e., Big Qualicum and Cowichan River; Lister 1968).

Marshall (1973) found that emergent fry survival in Reach C was higher during low flows periods in late winter-early spring. The estimated rate of adult returns from the 1966 brood fry that were exposed to low flows during emergence in the spring months was 0.54%, which was considerably higher than the returns that were estimated for the 1965, 1967, and 1968 brood fry years, which experienced lower survival rates of 0.23%, 0.10% and 0.29% respectively, when emergent fry flows were higher.

Similarly, an assessment of Chinook fry migration in the Cowichan River during a low (mean 7 m^3/s; 12% MAD) and high flow (mean 35 m^3/s; 61.4% MAD) year in 2016 and 2017, respectively, found that adult return survival in 2016 was twice as high. This may be evidence that higher discharge following emergence displaces fry into the ocean at a smaller size and results in lower adult survival (Pellet pers. comm. 2022).

Observations made in 1971 and 1972 found that Puntledge emergent Chinook fry are only present in quiet, shallow pools along the stream margins. Typically, these pools were less than 10 cm in depth with gravel, rubble, or rock substrate that provided shelter. Fry were notably absent in flowing water; in pools deeper than 0.3 m; in pools without cover; or low velocity areas that dropped off sharply into deeper, flowing water (Marshall 1973). The optimum discharge for early emergent fry is estimated and observed to be between 2.83 m^3/s and 5.66 m^3/s or 10-20% MAD. Discharges exceeding 60% MAD (19.62 m^3/s) during the first few weeks of emergence potentially decrease overall adult survival by 50% because of displacement and predation (Marshall 1973; Craig 2015).

4.3.10.3. Other Rivers

In a more recent study on the early life-history and critical rearing habitat requirements of Cowichan River Chinook salmon (Craig 2015), similar habitat preferences and ontogenetic niche shift were observed compared to the Puntledge River. Although Cowichan Chinook fry used velocities and depths that were generally higher early in the seasons and shallower late in the seasons (Craig 2015) compared to general habitat suitability preferences (i.e., Dephi curves[5]), the later season fish used higher velocities and depths than early season fish (Figure 105). A similar ontogenetic niche shift is visible on Figure 106 where early Chinook fry use slower velocities and stay closer to the river shoreline than late Chinook fry.

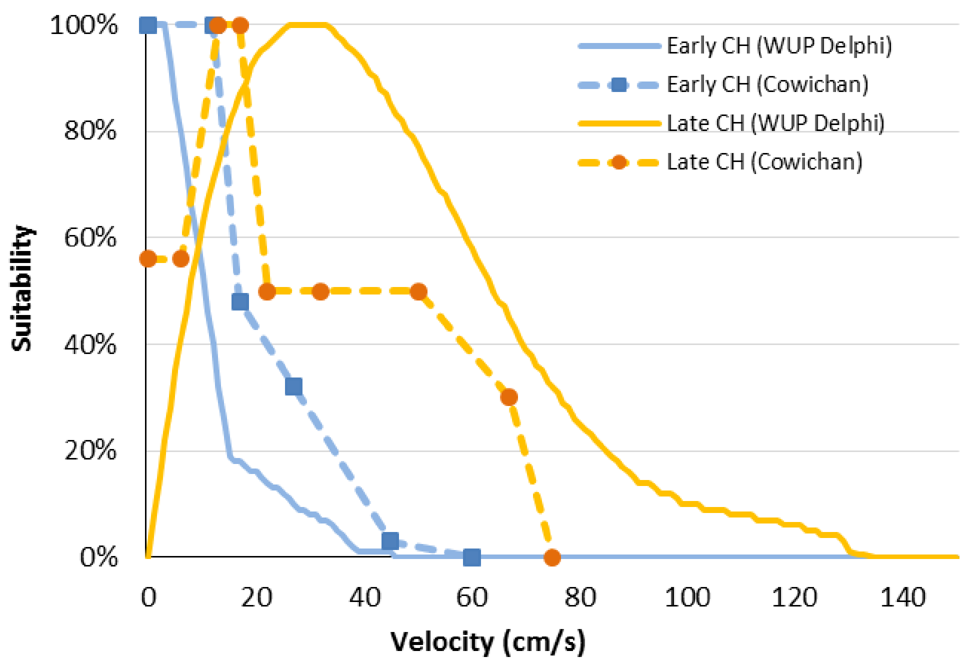
With spring discharge ranging from 100-200% MAD (similar to the Puntledge River), tight schools of fry were typically found occupying shallow (<10 cm deep) stream edges choked with vegetation (i.e., riparian shrub habitats on the wetted perimeter that are regularly inundated during spring freshet). Craig (2015) thus concluded that Cowichan Chinook fry preferred to stay close to the streambanks in zero to very low velocities, in shallow areas and in association with vegetative cover (e.g., willows, Red Osier Dogwood, Nootka Rose, grasses, etc.), particularly early in the season after emergence. The presence of all three of these features provided the highest densities of fry. As fry grew larger, the distribution of fish spread to faster flowing and deeper waters with use of LWD, but still in close association with the streambanks. This behaviour can be explained by the capabilities of a larger fish size, which includes higher swimming speeds to use larger areas and avoid predators.

It is expected that loss of shallow margin habitats with low velocities and vegetative cover would adversely affect Chinook abundance, particularly in the spring. Thus, the amount of intact riparian habitat that acts as wetland habitat or instream cover along the rivers and large side channels, may be a limiting factor for Chinook production in Puntledge River (particularly in years of high fry production when habitat can be limiting). Loss of such habitats due to flood protection, farming, and range tenures, as well as residential/industrial development is therefore a concern for the productive capacity of Chinook salmon in rivers. Overall, this threat is considered a high risk.

[5]The differences between the Delphi and Cowichan curves could be due to the limited data points for the Cowichan curves.

Figure 105. Habitat suitability curves of preferred a) water velocities and b) water depths, for early "spring" and late "summer" rearing juvenile Chinook salmon based on WUP Delphi and 2014 Cowichan River data (Source: Craig 2015).

a)



b)

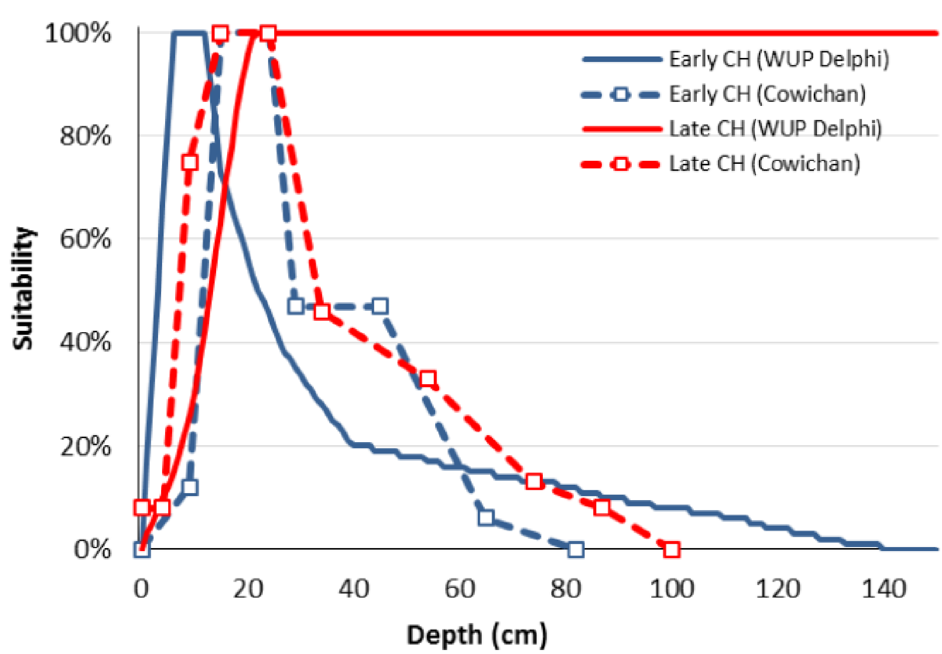
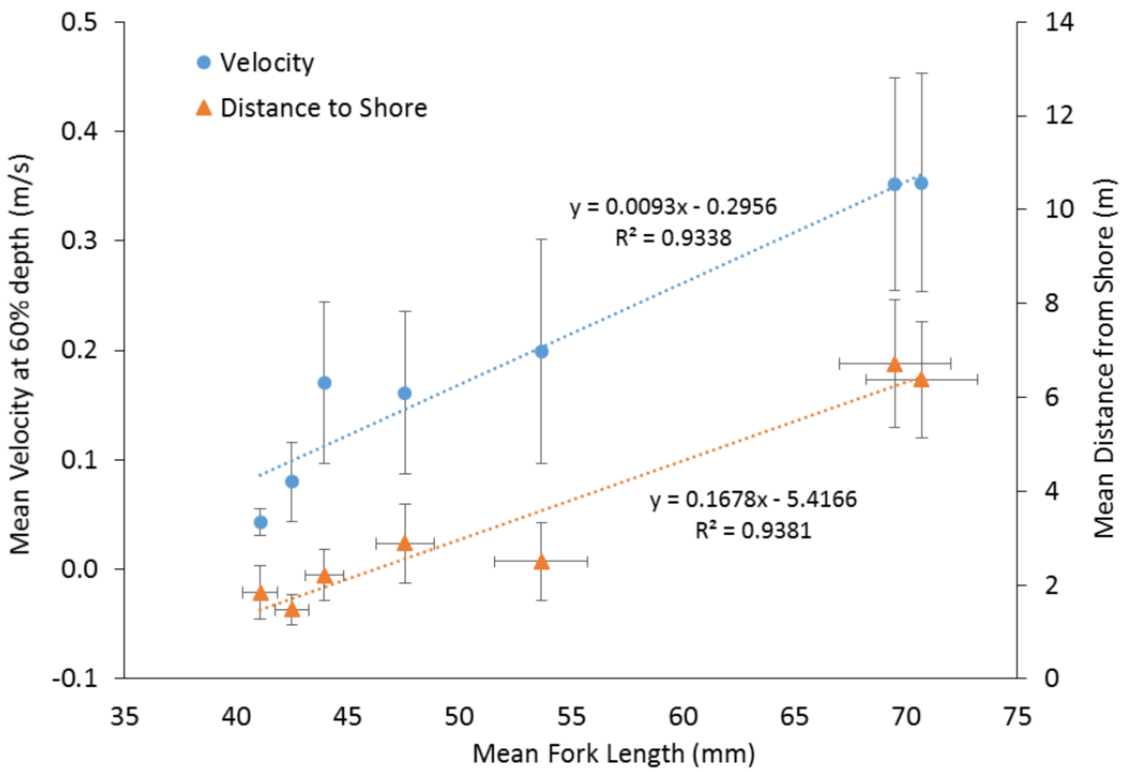


Figure 106. Relationships of mean velocity (at 60% depth) and mean distance from shore to Chinook fry fork length at the 70.2 Mile Trestle index site from March 19-June 11, 2014 (Cowichan River). Error bars represent 95% confidence intervals around means. Horizontal error bars for distance to shore are also applicable to velocity (Source: Craig 2015).



4.3.11. Competition with Invasive Species

Invasive fish species come from other parts of the country or world and are usually transported or allowed to migrate due to anthropogenic factors (e.g., use of boats, aquaculture facilities, etc.) to another location where they can establish and cause potential harm to native fish species (BC MoE 2022). The province of BC has classified invasive fish species under five categories based on their recommended management action (BC MoE 2023):

- *Prevent* – Prevent the introduction of high-risk species that are not yet established.
- *Early detection and rapid response* - Eradicate high risk species that are new to the province.
- *Provincial Containment* - Prevent expansion of invasive species outside of the province.
- *Regional containment/control* - Prevent further expansion of invasive species into new areas of high risk and well established or medium risk with high potential for spread.

- *Management* - Reduce the invasive species impacts on native species once it is widespread in the province.

The fish observations in the Puntledge River watershed identify four potential species that are non-native in BC:

- *Pumpkinseed sunfish (Lepomis gibbosus)* – Two individuals have been observed in the Puntledge River, one in 2010 and one in 2014. This species was also found in high numbers (>80) in the Smit Creek area in 2022, which joins with the Tsolum and Puntledge Rivers, and likely came from backyard ponds (BC MoE 2024a, Chek News 2022).

- *Perch (Perca sp.)* – A perch was observed in the Puntledge River in 2001 (BC MoE 2024) and was likely a Yellow Perch (*Perca Flavescens*) as this species has been intentionally released in BC as stock fish, accidentally spread by boats and bait buckets, as well as introduced voluntarily from aquariums and private ponds (ISCBC 2024).

- *Brook Trout (Salvelinus fontinalis)* – One individual was apparently observed in Comox Lake in 1928. As well, an angler also indicated that he observed a Brook Trout in Comox Lake in 2017 (AA 2022). However, this species was voluntarily introduced into several lakes through provincial stocking programs as a game fish in the province (BC MoE 2024a).

- *Atlantic Salmon (Salmo salar)* – One individual was apparently observed in Comox Lake in 1921. Atlantic salmon were introduced voluntarily between 1905 and 1935 in BC waters, including Comox Lake (Clifford Carl and Guiguet 1958); however, no viable population has been established. To this day, individuals from aquaculture pen facilities sometime escape into BC waters (Shore 2017).

Of these species, only two are considered invasive: pumpkinseed and yellow perch. The former is considered a species that is established and that should be managed to reduce their impact on local fish species while the yellow perch is under regional containment/control and should be refrained from spread in BC waters. These species may outcompete salmonid species for habitat and prey, as well as prey on salmonids. For example, the yellow perch reproduces quickly and competes with salmonids for prey and habitat (ISCBC 2024), and adult pumpkinseeds can prey on salmonid eggs and fry (Jordan *et al.* 2009). Introduced pumpkinseed may have caused a decline in salmonids from a Vancouver Island Lake (Jordan *et al.* 2009). Based on the observations to date, the greatest concern for Chinook salmon in the Puntledge watershed is likely the high numbers of pumpkinseeds that were observed in Smit Creek and that could have access to the Puntledge River over time.

4.3.12. Competition with Hatchery Fry

Puntledge River Hatchery begins releasing their Chinook juveniles, both fall- and summer-run, around the same date, usually between the last week of May to the first week of June. This timing is often associated with the prescribed kayak pulse flow (BC Hydro 2004), or an alternate pulse flow upon request by DFO to BC Hydro. This timing is based on an assumption that higher flows will reduce the presence of seals in the lower river, thereby providing a window of lower predation during smolt migration to the estuary (see Section 4.1.1.1). This timing also aligns with a second peak in the migration of natural summer Chinook parr (>60 mm), as observed during assessment activities at the diversion dam between 2011-2017 (Figure 89). Emergence and migration of juvenile fall Chinook from natural spawning in the river has not been monitored, but likely follows a similar pattern to summer Chinook, and other ECVI fall populations. Based on the number of fall Chinook returns to the Puntledge River over the past two decades, the average natural production of fall Chinook emergent fry could be as high as ~2 million annually. In addition, up to 1.8 million enhanced fall Chinook sub-yearling smolts are released from the hatchery each year. Fall Chinook fry are larger at emergence compared to summer Chinook (0.44 g versus 0.26 g or approximately 39 mm verses 33 mm; DFO unpublished data 2010-2020), due to their larger egg size (in Fleming and Peterson 2001).

Since 2010, releases of fall Chinook smolts from the hatchery tend to have been 50% larger than summer Chinook releases. Thus, the potential for competitive interactions between large hatchery releases of fall Chinook, and both hatchery- and natural-origin summer Chinook, as well as between natural-origin fall and natural-origin summer Chinook may be high (Table 55).

Table 55. Ranking of the potential for interactions between different groups of hatchery (H) and wild (W) summer (S) and fall (F) Chinook in the Puntledge River and estuary.

Group	Potential Interaction
HF x WS	High
WF x WS	High
HF x HS	High
WF x HS	Medium
HS x WS	Low

It is generally believed that the majority of hatchery releases of large (>5 g) smolts disperse downstream fairly rapidly and spend little time rearing in freshwater prior to entering the estuary (Levings *et al.* 1986; Korman *et al.* 1997). In the Campbell River estuary, Levings *et al.* (1986) captured higher numbers of hatchery Chinook in the transition zone, an area on the seaward side of the estuary, compared to wild Chinook, and hatchery released Chinook were found to use the estuary for half of the duration as the wild Chinook. The transition zone was identified as an area with the greatest potential for competitive interaction between hatchery and wild Chinook due to the abundance and timing of the hatchery releases (Levings *et al.* 1986). In the Cowichan River, both wild Chinook fry and smaller hatchery releases tended to have an extended rearing period in freshwater and estuarine habitats (Pellett 2017).

The downstream dispersal and migration time to the estuary of Puntledge River Chinook after release from the hatchery has not been closely examined. Habitat use by wild Chinook salmonids in the lower Puntledge River and upper estuary is limited. Studies in the late 1990s found low numbers of salmonids rearing in the lower river and estuary, compared to other East Coast Vancouver Island systems (i.e., Campbell and Nanaimo rivers; MacDougall *et al.* 1999; Bravender *et al.* 2002; Jenkins *et al.* 2006). However, the timing of these studies coincided with a period of extremely low natural production for both summer and fall Chinook, and releases of up to 1.9 million 6-8 g sub-yearling Chinook smolts from Puntledge Hatchery. The researchers also observed temperatures within the lethal range (21ºC to 25ºC; Walters and Nener 1997) at many sampling sites, making many areas within the estuary unsuitable for juvenile rearing due to poor water quality.

Competition between hatchery and wild salmonids has frequently been cited as an important negative ecological interaction but has seldom been tested rigorously. An extensive literature review of studies testing different hypotheses about competition between hatchery-reared and wild salmonids in streams concluded that two types of controlled experiments provided the strongest evidence for competition (Weber and Fausch 2001). Additive experiments quantify the effects of stocking hatchery fish on wild fish, whereas substitutive designs measure the relative competitive ability of wild versus hatchery fish, recognizing fish density, fish size and stream carrying capacity as key variables.

Various studies have confirmed that competition from hatchery fish can reduce fitness of wild fish, particularly when densities were increased to high levels by the supplemented hatchery fish (e.g., Petrosky and Bjornn 1988; Fenderson and Carpenter 1971). Peery and Bjornn (2004) found that the aggressive and habitat-use behaviour in natural Chinook was altered after the addition of hatchery fish to the stream section and was dependent on localized densities, and relative sizes of the natural and hatchery fish. Natural Chinook were dominated by the larger more aggressive hatchery fish in spring/summer, whereas in fall natural Chinook exhibited greater aggression when paired with hatchery fish of the same size (Peery and Bjornn 2004). The advantage of 'prior residence' of wild fish may be inadequate to the larger size and rearing experience of hatchery-produced salmon, which can outcompete smaller wild fish (Nickelson *et al.* 1986; Rhodes and Quinn 1998). These interactions between hatchery and natural fish could result in a lower survival potential of natural fish and affect the productivity of the population. From the standpoint of summer Chinook conservation and rebuilding objectives in the Puntledge watershed, large releases of fall Chinook for harvest, combined with high fall Chinook escapement to the river, may be incompatible due to the high potential for competitive interactions.

4.3.13. Increase in Didymo Abundance

The higher presence of Didymo at Dams might be related to a pH shift that interferes with the balance between pH-sensitive taxa in the periphyton community and Didymo gemination. Seven pH lake profiles taken in the outlet basin of Comox Lake between 1975 and 2015 shows an increase in pH from the lake surface to a depth of 5 meters or more, suggesting that the Comox Dam control outlet, which draws water at a depth of 4-5 m, may result in an increase of pH during the chinook juvenile rearing period, although this has not been verified. Alkaline conditions are preferrable for Didymo growth (i.e., pH 7 to 9) (Kirwood, *et al.* 2009; Figure 71). As well, as discussed in Section 4.1.15, although the quantity of drifting food for fish may remain relatively constant, the mean size of individual prey items may decrease with increasing *D. geminata* biomass. Declining prey size can have significant implications for fish bioenergetics, particularly of drift-feeding larger salmonids (Hayes *et al.* 2000). In an Atlantic salmon juvenile foraging study in the Restigouche River system in eastern Canada, increasing Didymo biomass led to a significant positive increase in the proportions of benthic forays versus drift forays (Gillis and Bergeron 2018). Weight gain was significantly lower in Didymo affected sites than Didymo free sites. However, it is unknown what effect this would have on smaller salmonid juveniles like summer Chinook that migrate to the estuary by June.

4.3.14. Unfavorable Water Temperatures

Summer Chinook juveniles migrate out of the system by June when water temperatures area generally below 15°C; thus, summer freshwater temperatures may not be an issue. Current analysis of adult otoliths is underway to determine the exact timing of habitat uses over time and results should be available in 2023. These results will provide clarification on the importance of this limiting factor, if any. Given the temperatures up to mid-June are within the optimal range for rearing (i.e., 16°C or below; Oliver and Fidler 2001). it is not anticipated that increases in water temperature due to climate change will be directly stressful for Chinook in the early rearing phase.

4.3.15. Water Quality Threats

4.3.15.1. Low Dissolved Oxygen

Low dissolved oxygen is not expected to cause mortality or reduce fitness of juvenile summer Chinook. Oxygen levels monitored at the lower hatchery have indicated that oxygen is normally saturated and not limiting (Puntledge Hatchery Staff pers. comm. 2023).

4.3.15.2. Poor pH Levels

pH level is not expected to cause mortality or reduce fitness of juvenile summer Chinook. For a discussion of pH in the Puntledge River watershed please refer to Section 4.1.15.2.

4.3.15.3. Deleterious Substances

Deleterious substances such as Didymo could cause mortality or fitness reductions in summer Chinook rearing in the river. For a discussion of this threat, please see Section 4.1.1.

4.3.16. Ingestion of Microplastics in Lake Environments

Microplastics are a significant problem in the oceans (Collicut *et al.* 2018) where they get ingested by fish and may affect their health and survival, including salmon. However, their presence is likely less dense in freshwater environments (where runoff from industry and communities are less common) and unlikely to have an impact on juvenile life stages of salmon, including summer Chinook. However, the identification of microplastics in the Puntledge watershed is very limited and would need to be further assessed to confirm this assessment.

4.4. Rearing in the Estuary

4.4.1. Elevated Predation

The channelization of the Courtenay River, the alternation of the natural shoreline, changes in the hydrology of the estuary delta due to disconnection and habitat fragmentation and the loss of saltmarsh and eelgrass habitats have resulted in an overall decrease in the amount of refugia available for out-migrating Chinook juveniles and returning spawners. In particular, the channelization of the Courtenay River has made it an area where salmonids are easily trapped and preyed upon by pinnipeds (seals and sea lions), as well as bald eagles, mergansers and other fish-eating birds. In particular, harbor seal populations have rebounded due to protections for marine mammals introduced in the 1970s and one study indicated that seals consumed an estimated 36% of endangered Chinook runs in the K'ómoks estuary (Map 11; DFO 1998). The following sections discuss predation on juvenile fish rearing in the Puntledge River estuary (Map 11). While this includes the main predators on estuary rearing juvenile Chinook, it is not exhaustive and may include other predators that are not as impactful on population dynamics.

4.4.1.1. Seal Predation

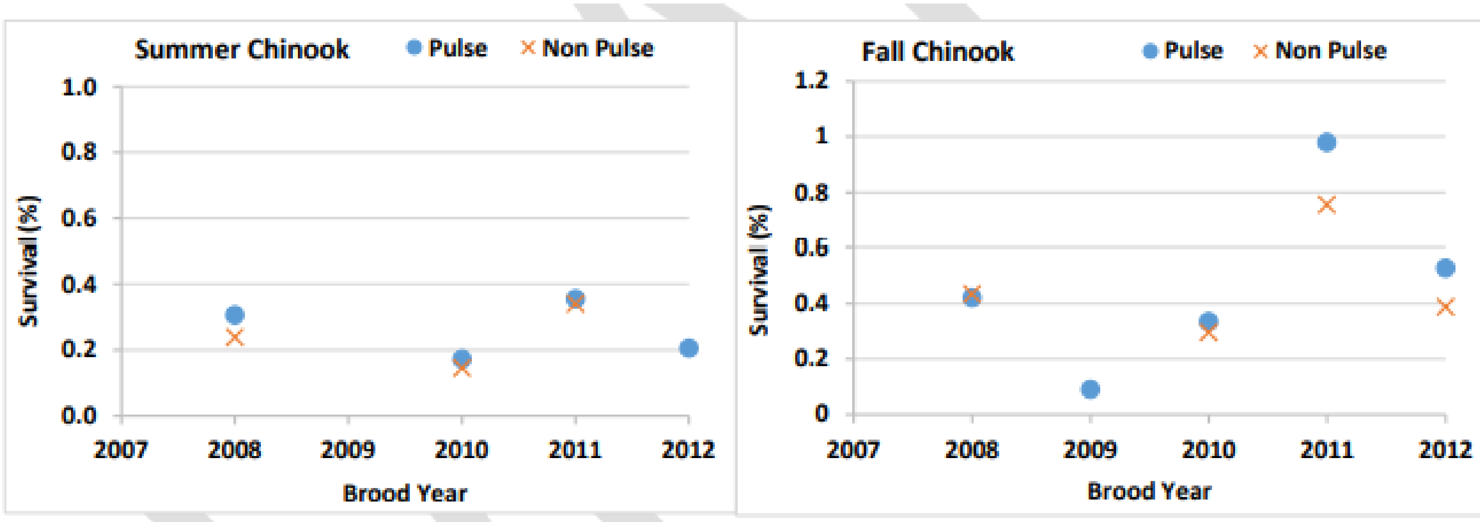
Historically, wild summer Chinook smolts migrating to the estuary in late spring/early summer would have coincided with the natural spring freshet in May through June. This timing may have reduced predation by seals because high flows could reduce the presence of seals, and smolts would be harder to capture during higher flows. However, spring freshet has been muted by BC Hydro dam operations, particularly in low snowpack years. Thus, between 2009 and 2013, trials were conducted to assess whether the release of Chinook smolts from the Puntledge Hatchery during higher river flows would positively influence smolt-to-adult survival.

Between 2009 and 2013, trials were conducted to assess whether the release of Chinook smolts from Puntledge Hatchery during higher river flows positively influenced adult survival. It was anticipated that the higher flows would reduce the presence of seals in the lower river, thereby providing a window of lower predation during smolt migration to the estuary. During the trials, unique tag codes of summer and fall Chinook (BY2008-BY2012) were released each year (except 2009) during different river discharges, which typically included base conditions and a period of elevated discharges provided by BC Hydro. Despite the variations in flow provisions in some years, there seemed to be a minor improvement in survival for releases during pulse flows (Figure 107).

Reduced availability of forage fish, lack of complexity in nearshore habitats, and the consolidated timing of Chinook hatchery releases may affect the number of salmon consumed by seals in a given year or location (PSF 2021). Chinook fry would benefit from more habitat enhancement projects in the estuary that naturalize hardened shorelines, such as the Kus-kus-sum initiative (Section 4.1.1). As mentioned previously, saltmarsh and riparian habitat degradation can result in a less robust insect community within the estuary and a decline in this food source for coastal cutthroat trout can potentially increasing their predation on Chinook fry (Tyron 2011).

Map 11. **Overview map of the estuary.**

Figure 107. Smolt-to-adult survival of summer and fall Chinook salmon released during pulse and non-pulse flow periods from the lower and upper Puntledge River hatcheries between 2009 and 2013 (Guimond pers. comm. 2022).



4.4.1.2. Bird Predation

A predatory species must be very abundant to inflict appreciable mortality on large populations (like that of the Puntledge River), which are at risk for so short a time. Although no quantitative estimates of merganser predation exist for the Puntledge River, information from previous reviews provide some insight on the potential for mergansers to influence productivity in this system. A review by Salyer and Lagler (1940), found that mergansers posed the greatest threat to salmon populations when they are found in high densities on productive salmon-rearing waters (e.g., production from a large local hatchery). Wood (1984) found that up to 8% of downstream migrating juveniles succumb to merganser predation on coastal streams of Vancouver Island. Despite this significant level of mortality, the overall mortality rate due to mergansers appears to be depensatory, such that the overall impact to the population is lower at higher salmon abundance. Although there are likely some contexts whereby mergansers are limiting salmon production, the challenge (Elson 1962; Miegs and Wieck 1967; Anderson *et al.* 1985) and ecological uncertainty (Anderson *et al.* 1985) associated with managing merganser populations make this management approach impractical.

It is unlikely that other fish-eating birds impose more serious mortality on juvenile salmon during their seaward migration in Vancouver Island streams than mergansers. Blue Heron may be the next most impactful bird predator, consuming up to 6% of the Chinook juveniles in Cowichan River; however, there is not a large population of these birds in the Puntledge watershed (Sherker 2020). Eagles and kingfishers are also efficient predators with large appetites (Alexander 1979), but all are less common than mergansers on eastern Vancouver Island streams. These species may cause significant mortality among stream-resident salmonids that are vulnerable for long durations, but their depredations are unlikely to be serious during the brief period of seaward migration in these coastal streams.

4.4.2. Predation by Invasive Species

There is limited information on the role of predation by invasive species on juvenile Chinook rearing in the estuary. However, is not expected to cause substantial mortality or reduce fitness of this life stage.

4.4.3. Inter- and Intra-specific Competition

There is limited information on inter- and intra- specific competition with juvenile Chinook rearing in the estuary; however, progeny from returning fall hatchery adults could potentially compete with juvenile summer Chinook rearing in the estuary (see Section 4.3.10). The Puntledge and Tsolum hatcheries also produces and releases Chinook/Chum and Pink Salmon juveniles, respectively, which may compete with summer chinook juveniles in the estuary.

4.4.4. Stress due to Anthropogenic Activity

Causes of coastal habitat degradation fall into two categories – those that are natural in origin and those that stem from anthropogenic, or human made, disturbances. In general, ecosystems have been shown to recover more quickly from natural disturbances than they do from human induced ones (Jones and Schmitz 2009). Examples of natural disturbances that can stress habitat in the coastal environment include storms and storm surges, flooding by tsunamis, earthquakes, or landslides. Human disturbances in the coastal environment can result in either an outright loss of marine vegetative habitat, fragmented habitat and/or impaired functioning condition through the alteration of physical, chemical or biological processes. These disturbances can lead to an overall reduction of marine life biodiversity and productivity, and a general reduction of ecosystem services. However, the impacts from these stressors are complex and some are interrelated, compounding each other leading to rapid declines in coastal ecosystem health.

Much of the historic industrial activity that affected the K'ómoks estuary, including industrial sawmills, log booming and dredging of the Courtenay River, have all disappeared and the community largely supports the restoration of the estuary. However, there are still many anthropogenic stressors, related to historic and ongoing activities and their associated impacts, at play in the estuary. Some key anthropogenic stressors include an increase in human population density along the coastline, channelization of the upper estuary, and invasive plant species, which are described in the following subsections. These stressors can negatively affect summer-run juvenile Chinook rearing in the estuary (e.g., by reducing availability of saltmarsh and eelgrass habitats, which are key rearing habitats for this species and other salmon; Kennedy *et al.* 2018, Bottom *et al.* 2005) and reduce their robustness and/or increase mortality.

4.4.4.1. Increase in Population Density

The majority of coastal water quality problems are the result of human activities due to concentrated populations along the coastline and from land-use practices throughout coastal watersheds. The K'ómoks estuary (Map 6) has limited exchange with offshore Pacific waters, meaning that pollution from overland flow can accumulate and have considerable impact on water quality.

Pollution from a complex network of storm drains and sewer outfalls carries many contaminants that can impact the nearshore environment. For example, excess nutrients (i.e., garden fertilizers, sewage, detergents, etc.) can cause algae blooms that block the amount of light available in the water column. Excessive algal epiphyte growth on eelgrass blades can lead to plant die-offs, as they block the light needed for photosynthesis. Herbicides, which are used on adjacent coastal land, can runoff into nearshore habitat killing or damaging coastal vegetation, including saltmarsh and eelgrass, which are key rearing habitats for juvenile salmon, including Chinook. Oil and other petrochemicals that enter the marine environment can either directly or indirectly kill aquatic plants, and direct contact with oil has been shown to cause eelgrass to lose its leaves (ToG 2017).

Recent research in the Salish Sea also demonstrated an inverse correlation between eelgrass meadow coverage and residential housing density, along with its associated shoreline activities (Nahirnick 2018). Over time, as urbanization increased, eelgrass coverage and complexity decreased, pointing to a general decline in coastal health due to a marked increase in the use of the coastal zone. The loss of eelgrass habitat results in secondary effects such as re-suspension of sediments, increased turbidity and reduced light penetration, all of which compounds the loss (Eamus 2005). This leads to a reduction of ecosystem services (Thayer 1994), the outcome of which can be an autocatalytic decline (Larkum 1982).

In the early 2000s, the Comox Valley Project Watershed Society (Project Watershed) undertook water quality sampling at over 200 outfall locations around the estuary and Baynes Sound. Areas that had high fecal coliform counts were noted and any cross contamination between storm water and sanitary sewer lines was rectified to improve the overall water quality. As well, two bio-retention wetland ponds were installed in areas draining properties with septic fields to help address water quality. However, no additional water quality data has been collected since the original project was conducted and, given the amount of development that has occurred in the area in the last 20 years, it would make sense to re-assess water quality within the estuary. One initiative that Project Watershed has recently engaged in with other partners is conduct research to determine the amount of micro-plastic pollution within Baynes Sound. Other water quality parameters could potentially be measured within the estuary as part of this work and provide a more complete picture of the current status of water quality within the estuary.

Clearing of the marine riparian vegetation within the nearshore supralittoral and backshore zones, due to coastal development and associated landscaping, removes the vegetative cover that provides valuable habitat, insect production, and shading and cooling of the water. This vegetation can also be impacted by infrastructure that facilitates public access into sensitive shoreline habitats such as stairs, docks, and boat ramps. In general, the estuary suffers from major shoreline alterations that have occurred over time due to industrial and residential development along the coastline.

4.4.4.2. Channelization of the Upper Estuary

Hard armoring such as bulkheads, seawalls, dikes, tide-gates, groynes, and rip rap are common in coastal communities and have been the conventional approach to protect shorelines and hard infrastructure such as roads from coastal flooding and erosion. However, armoured shorelines disrupt natural shoreline sediment transport processes, can increase erosion in adjacent unprotected areas and degrade the natural shoreline habitat. Such hard armouring of shorelines does not support abundant fish habitat and results in ‘coastal squeeze’ – the prevention of the landward migration of intertidal habitat that would otherwise naturally occur with sea level rise. The installation of hard armouring has also been shown to result in cumulative ecosystem impacts. Specifically, reductions in riparian vegetation, beach width, number of accumulated logs and amounts and types of beach wrack and associated invertebrates have resulted from hard armouring (Dethier 2016).

In the K’ómoks estuary, notable areas of armouring exist starting around Lewis Park on the Courtenay River (Map 6) where a cement wall has been installed on the left bank of the river along the entire edge of the park. Riverside Park, across from Lewis Park, has also been treated with a concrete wall. Furthermore, the downstream properties between Riverside Park and the 5th Street bridge have been protected with wooden cribbing and extensive rip rap, resulting in significant channelization of this section of the river. Starting in 1937, the whole section of river from the confluence of the Puntledge and Tsolum Rivers to where the Courtenay Slough meets the river adjacent to present day Simms Park, has been armoured to protect the community from flooding (CCHAC 2015). The wooden cribbing that was installed in the 1930s was eventually replaced with a concrete bin wall in the 1970s. This channelization straightened the river, taking out some of the natural meanders, which greatly altered the shoreline and potentially resulted in increased flow velocity. As a result, shoreline complexity and overhanging vegetation was reduced, making this area much less appropriate habitat for salmonid rearing and refuge (i.e., eliminate shallower areas, cover, aquatic vegetation and habitat complexity that are key habitats for rearing salmon; PSF 2024). Moving further downstream, armouring of the river is associated with past development of transportation corridors (i.e., Dyke Road) and industrial activities that use the lower river and estuary (i.e., log sorting and cement production). A good example of a shoreline in the estuary that has an extensive amount of hard armouring is the Lafarge site (Figure 108, which previously housed an old cement silo.

Various actions could be taken to rehabilitate the affected shoreline to a more natural shoreline that supports saltmarsh. For example, the Lafarge site (Figure 108) property was officially donated to the regional district for park land in 2016 and could be rehabilitated. Furthermore, next to the park is a property on the ocean side with two dwellings. This property is a prime candidate for coastal retreats, as it is on a vulnerable section of coastline and the buildings on it are non-conforming with current building codes, including coastal setbacks.

Figure 108. **Hardening of shoreline at Lafarge/Dyke Road Park site. Note the non-conforming property adjacent to park on right side of image (Source: ShoreZone (https://www.shorezone.org/interactive-shorezone-maps/), photo taken August 29, 2023).**

4.4.4.3. Invasive Plant Species

Anthropogenic activities can lead to the introduction of foreign species, which can become invasive and outcompete native coastal species for resources and habitat. According to the International Union for the Conservation of Nature, the threat of invasion by non-native species is second only to habitat loss in terms of the impact on native ecosystems (Netherlands Committee for the IUCN 2001). Coastal zone species known to be in the K'ómoks estuary include saltmarsh cordgrass species (*Spartina spp.*), Japanese wireweed (*Sargassum muticum*) (Figure 109) and introduced resident Canada geese (*Branta canadensis*).

Spartina spp. are invasive perennial salt-tolerant grasses that threaten shorelines by out-competing native saltmarsh plants, forming a monoculture, and reducing liveable habitat for wildlife such as fish, crabs, shellfish, shorebirds and waterfowl (Williams 2009). There are four invasive species of cordgrass – *Spartina anglica*, *Spartina densiflora*, *Spartina patens* and *Spartina alterniflora*. The dominant growth of these species can cause the elevation of intertidal areas to rise, changing the structure and function of these areas (Dethier and Hacker 2004).

In 2018, Project Watershed undertook a coastal mapping of eelgrass, saltmarsh and kelp habitats along 120 kms of coastline, including the K'ómoks estuary, and encountered a surprising amount of *S. muticum*. There are many reports of *S. muticum* competing with and successfully displacing native species of seaweed in BC as well as native eelgrass (*Zostera marina*) (Druehl 1973). *S. muticum* has a variety of life-history strategies and adaptations that make it a very successful invader including a high growth rate, quick maturation, ability to grow in a wide range of temperatures, an enduring holdfast, and multiple dispersal techniques (Pawluk 2016). In Baynes Sound, *S. muticum*'s rapid growth allows it to shade out eelgrass, growing in areas with suitable substrate (i.e., small stones) where it can attach itself. This may lead to destabilization and eventual loss of eelgrass-supporting finer sediments. Further invasion may be facilitated with only larger sediments remaining, in a possible example of an alternative stable state induced by an invasive species. The most likely vector of entry for *S. muticum* into BC was via oyster seed (*Crassostrea gigas*) taken from beaches in Japan and then dumped into Baynes Sound in the early 1900's. Several nonprofit organizations such as Ducks Unlimited Canada work to restore BC coastal estuaries (e.g., saltmarsh presence), which are key habitats for salmon (i.e., for refuge and rearing as they transition from freshwater to saltwater) by finding and eradicating these invasive plant species (e.g., *Spartina spp.*; DU 2023).

Figure 109. Japanese wireweed (*Sargassum muticum*) (Source: iNaturalist).

4.4.5. Disease, Parasites, or Pathogens

The impacts of saltwater diseases, parasites and pathogens on Chinook salmon are detailed in Section 4.3.2.3 and are mostly discussed for juvenile hatchery fish and those that have been rearing in the marine environment for one year. Results to data have shown that freshwater pathogens present in Puntledge River fish could indicate that they are also present in the estuary and thus could be present when juveniles are rearing in the estuary. This aspect requires further studies to assess.

4.4.6. Lack of Access to Appropriate Food

Feeding opportunities for Chinook rearing in the estuary are closely associated with the habitat that supports their invertebrate diets, including healthy marsh and riparian ecosystems. In particular, sedges (*Carex lyngyei*), rushes (*Scripus spp.*, *Typha spp.*) and riparian shrubs and trees in the middle and upper intertidal zones have been shown to provide detritus and habitat for Chinook food organisms (Maier and Simenstad 2009), making them essential components of the estuarine food web that supports Chinook rearing. In the Comox Estuary, stomach content analysis of Chinook fry and smolts showed that they are highly dependent on insects, both terrestrial that drop from over hanging vegetation and aquatic insects from detrital habitat, particularly from May through July (Tyron 2011). Consequently, ecosystem alteration that affects these habitats may decrease the availability and quality of food webs in the estuary and reduce the ability of the estuary to support Chinook salmon via access to the appropriate food sources (Maier and Simenstad 2009). Healthy insect communities in the estuary may also reduce trout predation on Chinook fry (Tryon 2011), by providing the trout with alternative prey. Impacts to salt and brackish habitats within the estuary include all those outlined in the previous section. Therefore, floodplain restoration has been generally recommended as a priority for improving salmonid habitat (Sommer *et al.* 2011; Beechie *et al.* 2013) and it has been specifically suggested that restoring saltmarsh in the upper K'ómoks estuary floodplains would improve the productivity of the area for salmonids (Buffet 2008).

4.4.7. Freshwater and Estuary Habitat Utilization due to River Flow

Ptolemy examined the hydraulic data collected for the Puntledge WUP (Burt 2002, Ptolemy pers. comm. 2022) and showed that BC Hydro's base flow conditions are generally too high during fry emergence (>20% MAD) and tend to displace juveniles from Reach B downstream towards the estuary where they are at risk of mortality due to Eicher screens and competition/predation with large salmonids in Reach C and D. This negative impact is expected to affect rearing in the estuary by increasing the number of fry that are displaced to the estuary at a smaller size. Further details on the findings of this work are provided in Section 4.3.5 (i.e., early rearing).

4.4.8. Increased Frequency and Magnitude of Algal Blooms

There is limited information on the frequency and magnitude of algal blooms in the Puntledge River estuary, as well as the potential for algal blooms to have an impact on rearing summer Chinook within the estuary.

4.4.9. Vegetation/Beach Habitat Loss

Salmonid habitat in the Courtenay River Estuary has been significantly reduced over the past 150 years from human activities such as agriculture, urban development, and road building. Agricultural fields now cover about 75% of the original estuary. The only functioning estuarine salt marsh habitat remaining is Hollyhock Flats, which is adjacent to the old Field sawmills site. Small pockets of salt marsh habitat still exist throughout the estuary but are largely isolated from each other. Urban and industrial encroachment along the banks of the lower river and estuary has alienated off-channel, riparian and wetland areas. Shoreline habitat has also been simplified, increasing fish susceptibility to seal predation. Hydroelectric development in the watershed has altered natural estuarine processes due to the reduction of woody debris inputs and the loss of large flood events to flush accumulated sediments from the estuary.

Coastal mapping analysis of the extent of saltmarsh habitat in the K'ómoks estuary shows a loss of 16.41 hectares (ha) or a 25.79% decline in the amount of habitat between 1980 and 2016 (Figure 110). In particular, there has been substantial erosion and recession of the saltmarsh fringe between the Courtenay Airpark and Millard Creek estuaries (Figure 111). The marsh fringe in this area has receded by around 60 m resulting in an overall loss of six hectares of habitat. This loss is likely due to the impact from south-easterly storms and their associated wave action leading to erosion of the saltmarsh platform. In addition, grazing and grubbing by resident Canada geese likely contribute to weakening of the plant communities, which facilitates erosion. Another area that has experienced a significant decline in the amount of marsh habitat is the outlet of the Courtenay River. This area has been greatly impacted by historic human activity. The entire Courtenay River has been artificially channelized and narrowed as the community of Courtenay developed protection to prevent flooding in low-lying areas.

Additional impacts to the estuary include historical dredging of the river and concomitant dredge spoil deposition, the in-filling of brackish saltmarsh habitat to create the Courtenay Airpark airstrip, the development of the now defunct Courtenay sewage lagoon, diking, and the construction of the Courtenay Marina, all of which have altered the habitat and the distribution of freshwater across the estuary delta. Moreover, the impacts of resident Canada geese continue to hamper the growth of native plant communities in these areas.

Increased frequency and magnitude of algal blooms surrounding residential areas (Comox and Courtenay). The Courtenay Sewage Lagoon, located in the K'omoks Estuary, operated from 1962 to 1983 with marginal improvements to the effluent entering the environment. High levels of total coliform continued to be a problem, further aggravated by other sources, such as leaking shoreline septic tanks and agricultural runoff that continued into the late 1990s (Asp and Adams 2000).

In the early 1990s, the park around the lagoon was built as habitat compensation for the expansion of the Comox marina. As part of the park development the dike around the lagoon was breached at the south end to reconnect it to the estuary. In 2015, the Project Watershed undertook a restoration project to breach the dike at the north end of the lagoon via the installation of a culvert to further restore habitat connectivity in the estuary. The culvert reconnected the river through the lagoon, and changed the hydrology of the area, to improve its ecological functioning condition. This area has experienced a 1.14 ha gain in saltmarsh habitat (see the blue triangular area to the left of the image in Figure 110. However, even these successful restoration efforts are threatened by year-round herbivory by resident Canada geese that use this sheltered area during winter storms.

Many studies have demonstrated that eelgrass, *Zostera marina*, provides significant foraging opportunities for juvenile Pacific salmon and that the protection and restoration of nearshore eelgrass habitat may be critical for the health of salmon runs (Beamish *et al.* 2004; Kennedy 2016; Kennedy *et al.* 2018). Eelgrass habitat within the estuary declined by 94.71 ha or 37.94% between 1979 and 2016 (Figure 112). This rate of loss of eelgrass is consistent with other trends in eelgrass habitats within the Salish Sea (Nahirnick *et al.* 2020). Most of this loss (79.20 ha) has occurred in the high intertidal sand flats of the estuary and are likely due to an increase of sediment inputs, which alter the structure and function of the ecosystem, as a result of human activities in coastal areas (Thrush *et al.* 2004). Another area of loss was due to the previously mentioned expansion of the Comox marina that resulted in the loss of 4.21 ha of eelgrass habitat. The Airpark habitat compensation project, which was intended to offset the marina development, resulted in an increase of 1.14 ha of saltmarsh habitat, but this does not compensate for the loss of eelgrass habitat. Another significant area of eelgrass habitat loss is around the Trent River estuary, where 11.11 ha have disappeared on the east side of the estuary. The community of Cumberland, which has been experiencing rapid population growth, discharges its partially treated sewage into Maple Creek, which flows into the Trent River estuary and then out to Baynes Sound. The Village of Cumberland currently has an antiquated sewage treatment process, whereby raw sewage is directed into a lagoon to settle out before being released to the environment. In fact, the province of BC has issued several notifications to the Village for being in non-compliance of the province's wastewater treatment regulations over the past several years. Nutrient loading of the marine environment from sewage and run-off from agriculture and industry activities have been shown to be a major cause of seagrass death (Waycott 2009). The excessive nutrients trigger the growth of algae that grow above or on seagrass and stop it from getting the sunlight required for photosynthesis. Thus, it is surmised that the loss of eelgrass in this area could be due in part to the substantial nutrient loading from sewage effluent. Furthermore, large blooms of invasive *Sargassum muticum* have been linked to warmer ocean temperatures and nutrient loading from sewage. As mentioned previously, mapping of this area showed that there is a large amount *of S. muticum* growing, which appears to be outcompeting the eelgrass in this area. Eight to ten years ago *S. muticum* was absent in the Trent River estuary. The Village of Cumberland is currently in the process of upgrading its sewage treatment facilities to comply with provincial standards.

There are some areas where eelgrass has expanded within the estuary including 6.39 ha at the Royston Wrecks site due to the end of log handling operations. From 1911 to the early 1950s, steam locomotives hauled approximately 6-8 billion board feet of logs from logging camps throughout the Comox Valley to the Royston log dump, which was located at this site. A mile-long wharf extended from the end of the road into the estuary. Logs were tipped off the wharf and sorted into booms and towed to more protected waters for transport to sawmills. Starting in 1937, large ships and tugboats were sunk off the site to form a breakwater to protect the log booms. Logs were stored north of the breakwater until 2006, thereafter, Interfor abandoned the log booming tenure. The past industrial activities at the site caused major damage to the coastal habitats. In addition to impacts to eelgrass, the breakwater has disrupted the normal pattern of sediment delivery along the shoreline, starving the site of sediment and slowing saltmarsh recovery in the area. Another 0.65 ha of eelgrass has been gained at Goose Spit where long handling activities were also discontinued in the intertidal area. The outlet of the Courtenay River has migrated over time resulting in a loss of around 3.92 ha on the left side but gained 8.67 ha on the right side of the river outlet. All of these changes in the nearshore habitat likely alter the feeding and migration patterns of summer Chinook salmon within the estuary.

Figure 110. Change in saltmarsh habitat extent in the K'ómoks Estuary between 1980 and 2016 (Source: Ecofish Research Ltd.).

Figure 111. Change in saltmarsh habitat between Courtenay Airpark and Millard Creek (Source: Ecofish Research Ltd.).

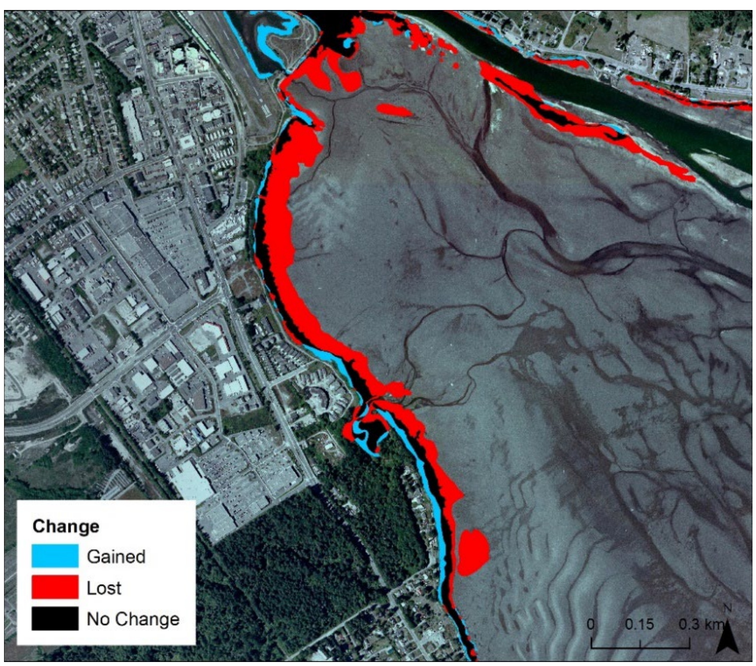
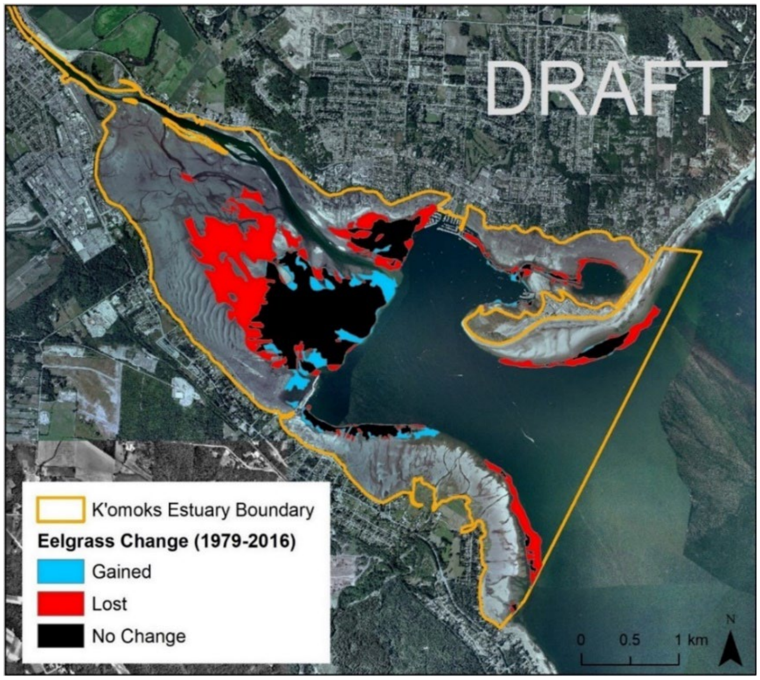


Figure 112. Change in eelgrass habitat within the K'omoks Estuary between 1979 and 2016 (Source: Ecofish Research Ltd.).



Results from three years of salmonid surveys in the Courtenay River estuary identified a deficiency in low tide refuges for Chinook salmon, and those present were, for the most part, located in areas that had been dredged (Hamilton *et al.* 2008). Off-channel estuarine tidal channels are also important nursery feeding areas for Chinook. Young Chinook will move from low tide refuge habitat and into tidal channels during flooding tides, and out of the channels on the ebbing tide (Healey 1991). Chinook in the Salmon River exhibited positive growth while rearing in these habitats (Hering *et al.* 2006).

Several hectares of intertidal estuary habitat has been cut off resulting in channelization of the upper estuary, which has affected summer Chinook juvenile distribution and likely growth (see Section 4.4.4.1). Beach habitat has also deteriorated over the last decades due to invasion by non-native shoreline species (see Section 4.4.4.2). The impacts of this change in beach habitat quantity may have an impact on mortality or fitness of summer Chinook rearing in the estuary, but additional studies would be required to determine that effect, if any.

4.4.9.1. Resident Canada Geese (*Branta canadensis*)

Resident Canada geese (*Branta canadensis*) numbers are increasing along the east coast of Vancouver Island, causing habitat impacts due to herbivory and degradation of intertidal marsh and eelgrass areas (Figure 113 and Figure 114). An intensive juvenile bird introduction program in the 1950s and 1960s led to a well-established resident population, before which these birds were only migratory and winter visitors (Dawe and Stewart 2010). As of 2010, the population was around 10,000-12,000 birds residing on eastern Vancouver Island; current estimates are similar or somewhat reduced (Clermont pers comm. 2022). These geese feed on the intertidal marsh vegetation and the calorie-dense rhizomes of the plants that hold the sediments in place. Once the rhizomes are removed the sediment becomes loose and unconsolidated and is easily washed away by wave action, leaving no material at the correct elevation for remaining plants to re-colonize. The result is degraded mudflat habitat that cannot support the highly productive species requiring a higher elevation. In particular, *Carex lyngbyei* (Lyngbyei's sedge) dominates brackish tidal marsh channel edges and catches and retains sediments and organic matter, which builds up over time forming thick mats – the saltmarsh platform. Growing at a precise elevation depending on prevailing salinity and sediment texture, the *Carex* meadow plant community has been documented as the most bio-productive in the marsh (Dawe and Stewart 2010). Overgrazing and grubbing of the *Carex* rhizomes by resident geese leads to erosion and loss of this platform and its associated biological productivity and provision of habitat to rearing juvenile Chinook salmon.

The K'ómoks First Nation in partnership with the Guardians of the Mid-Island Estuaries Society (GOMIES), have been working in the K'ómoks estuary to control the impacts of goose overgrazing using a three-pronged approach. The primary intervention is an annual harvest of adult geese that occurs in partnership with local First Nations from Sooke to Campbell River and across the Georgia Strait to Powell River. The harvest methods are permitted by Canadian Wildlife Service and comply with provincial animal welfare standards. By combining goose population reduction with cultural harvest of meat for food, the community buy-in to this activity is increased while mitigating resistance from members of the public concerned with animal welfare.

GOMIES also partners with First Nations and local governments and the province to execute an annual egg addling program with the goal of achieving good coverage of the most used nesting sites. The third prong of GOMIES strategy to restore intertidal marsh is building "eco-cultural" fencing, using alder stakes interwoven with willow, to create enclosure fencing around areas to prevent the goose herbivory and allow the native vegetation to regrow. The principle behind this is that the geese need an open area for take off and landing and will therefore will not land into the fenced off polygons of the marsh. These areas are often replanted with donor *Carex* from nearby healthy areas. Although there has been some success with these enclosure areas preventing goose herbivory in fenced areas, there are also problems with this management tool. First, the geese tend to concentrate their feeding in the areas that are not fenced off, which then suffer increased impacts. In addition, if not designed and maintained properly the geese can breach the fencing, and they have been noted inside the fenced areas. Finally, this method does not replace the eroded saltmarsh platform and in some areas the platform might need to be enriched with sediment in order to rebuild the historic elevations of the area to pre-browsing conditions, so that the *Carex* is being transplanted into a suitable soil matrix and at an elevation that will facilitate its bio-productivity and hence value to the ecosystem, including its role in providing habitat to juvenile Chinook salmon.

The success of the GOMIES strategy depends on deployment at the regional scale because the resident geese use multiple estuaries in any given season. Actions taken in only one estuary will be less effective if populations continue to grow in surrounding areas. Partnerships with First Nations, and local, provincial, and federal government departments are critical to the success of this strategy.

Figure 113. Resident Canada geese congregating in the K'ómoks Estuary.

Figure 114. **View from the water side looking landward of the progression of marsh platform degradation owing to resident Canada goose herbivory in the K'ómoks estuary near the Courtenay Airpark. Areas closer to the water are grubbed first while areas closer to shore are more intact.**

4.4.10. Competition with Hatchery Fish

Competition of fall hatchery Chinook juveniles and progeny from returning fall Chinook could compete with naturally emerging summer chinook fry, as discussed in Section 4.4.3.

4.4.11. Unfavourable Water Temperatures

The Salish Sea annual mean water temperature is predicted to increase by 1.51°C by 2095 (i.e., RCP8.5 scenario). Higher temperatures in estuaries and inner bays are predicted to cause increased thermal stress for the ecosystem. The warmest temperatures have been found in the lower estuaries, due to the shallower depths and exposure of mudflats at low tides that can warm incoming tidal waters. Model simulations predict algal species shifts in the Salish Sea. Dinoflagellates will increase by 196% and diatoms will decrease by 14%. The overall annual mean algal biomass is predicted to increase by 23% in the RCP8.5 scenario simulation relative to historical levels because of higher temperatures and higher nutrient loads (Khangaonkar *et al.* 2019). Excessive algal blooms can have a negative impact on Chinook habitat suitability (e.g., DO) in the estuary.

Temperatures recorded at many sites throughout the entire estuary often exceed 21°C; the lower range of thermal tolerance (LT50) for juvenile salmonids (Walters and Nener 1997). Although juvenile Chinook were found in small numbers in all regions sampled in the estuary, the greatest catches occurred in the remaining natural slough area as well as in man-made slough and boat moorage areas in the upper estuary, underscoring the importance of this type of habitat for juvenile rearing. However, these sloughs can also be of concern for increasing water temperatures. Therefore, mortality or fitness reduction due to unfavourable water temperatures may be a limiting factor for summer Chinook in the estuary and is likely to become more of an issue with anthropogenic climate change. However, more research needs to be done to understand the specific spatial and temporal thermal regimes that are present in the estuary, and how to enhance or augment these via floodplain reconnection as part of the summer Chinook recovery plan.

4.4.11.1. Climate Change Impacts in the Estuary

Degradation of the K'ómoks estuary habitat is expected due to climate change and associated impacts from increased temperatures, sea level rise and increasing frequency and severity of storm surges. Coastal salt and brackish marsh species are adapted to live within a very specific set of elevations. Sea level rise will cause coastal shorelines to be inundated more frequently and for longer resulting in some salt and brackish marshes to be permanently inundated if they cannot migrate inland. The estuarine-freshwater transition zone will also migrate up the Courtenay River with sea level rise. Likewise, subtidal and intertidal varieties of eelgrass have adapted to live within a certain range of tidal heights, and modelling of an increase in sea level showed that eelgrass mortality will increase due to reduction in the amount of light available for photosynthesis (Scalpone *et al.* 2020). Research has also demonstrated that eelgrass (*Zostera marina*) in our area is very sensitive to increases in temperature change and rising ocean temperatures may predispose it to secondary stressors such as eelgrass wasting disease (Kaldy 2011). Increases in the severity and frequency of storm events will also impact estuarine rearing habitats due to a concomitant increase coastal erosion of these habitats. Finally, as CO_2 levels in the atmosphere rise oceans are absorbing more CO_2 and becoming more acidic, this changing chemistry of the ocean in combination with other anthropogenic stressors is predicted to have significant impacts on coastal ecosystems (Fabry *et al.* 2008).

4.4.11.2. Water Quality and Contaminant Loading in the Estuary

There is limited information on water quality and contaminant loading in the Puntledge River estuary, as well as its potential to affect summer Chinook rearing. Currently this is not expected to be a threat.

4.4.12. Water Quality Threats

4.4.12.1. Low Dissolved Oxygen

Seabed hypoxia (DO <2 mg/L) is forecasted to cover as much as 16% of the Salish Sea by 2095 and annual mean DO throughout the Salish Sea is predicted to decrease by 0.77 mg/L (i.e., RCP8.5 scenario). Annual exposure to hypoxic waters in Salish Sea is predicted to be 57 times higher (Khangaonkar *et al.* 2019). Changes in dissolved oxygen specifically in the Comox estuary over time are unclear and will need to be assessed further to identify potential impacts of these changes on summer Chinook juvenile rearing.

4.4.12.2. Poor pH Levels

Fossil evidence indicates that ocean pH has fluctuated over the past 20 million years, but only within the range of 8.1 to 8.3. Ocean pH has been declining since the 1960s due to CO_2 absorption from greenhouse gas emissions (Pfister *et al.* 2011). Pfister has now measured pH values as low as 7.7 and 7.8 in the Strait of Juan de Fuca. In the central Salish Sea, Island Scallops, a local land-based scallop farm had to layoff staff in 2014, due to high mortality, mainly caused by a decrease in pH to 7.3 (Parksville Qualicum Beach News 2014). The Salish Sea is high in carbon and low in pH relative to the surrounding ocean (Lanson *et al.* 2016). Surface pH is now expected to range between 7.8 to 8.0 throughout the year. Mussel shells have become 30% thinner and are increasingly made of fossil fuel-sourced carbon (C_{12}). This is due to higher concentration of free hydrogen ions, which lowers pH and reduces available carbonate, reducing the ability for organisms to form their calcium carbonate shells impacting the populations of shell-forming organisms, which include many species that salmon commonly rely on for food (e.g., crabs, krill, and shrimps). Acidification has also been shown to decrease microbial community diversity in the Salish Sea that could in turn impact the microbial food web and the availability of energy to higher trophic levels (Crummett 2020). Overall, these changes could lessen the availability of salmon's food sources, which could impact their diet, growth rate, and survival.

The olfactory cues in salmonids are also sensitive to disruptions in pH, particularly when caused by increases in CO_2 (Williams *et al.* 2018). A previous study by Williams *et al.* (2018) determined that juvenile Coho salmon exposed to elevated CO_2 experienced significantly disrupted neural pathways on a physiological level. This has potential implications for a salmon's ability to home to its natal stream for spawning following the oceanic developmental phase of the life cycle (Dittman and Quinn 1996; Gerlach *et al.* 2007).

Damage to the olfactory bulb of juvenile Coho salmon caused by excessive CO_2 exposure has been shown to decrease their ability to respond to alarm cues (Williams *et al.* 2018; Michelson 2015). The failure to elicit an avoidance response to an alarm cue may make juveniles more susceptible to predators as well as increasing challenges experienced in locating and capturing prey (Gallagher *et al.* 2018; Williams *et al.* 2019).

Changes in pH levels in the estuary over time in conjunction with higher temperatures and potentially higher nutrient loading is still unclear and will need to be assessed further to identify potential impacts of these changes on summer Chinook juvenile rearing. Predicting how chinook will respond to the changing water quality environment is complex. It is imperative that the mechanisms and interactions in response to lower pH are understood to effectively maintain conservation and sustainability.

4.4.12.3. Increases in Salinity

Juvenile Chinook begin entering the estuary in March, soon after emergence, and may reside there all summer until they reach smolt size. As they grow, they move progressively downstream from the upper estuary where there is greater freshwater influence and lower salinities, to the lower estuary and offshore areas of higher salinity (Jenkins *et al.* 2006). In 2001, sixty-eight sampling sites in the Courtenay River estuary were assigned to one of nine salinity zones, which ranged from 0 ppt in the upper river to 29.3 ppt in Baynes Sound (outer estuary). Additional information on salinity trends in the estuary is available but limited, and its impact on summer Chinook juvenile rearing is unclear and should be assessed further. However, evidence to date does not suggest that change in salinity, if any, has had an impact on juvenile rearing.

4.4.12.4. Deleterious Substances

Pollution from storm and sewer outfalls carries several contaminants that can impact water quality in estuaries. For example, excess nutrients from garden fertilizers, sewage, and detergents can cause algae blooms that can negatively impact water quality for fish. The impact of the increase in deleterious substances within the estuary can affect fish rearing in this area, including summer Chinook juveniles (see Section 4.4.4).

4.4.13. Ingestion of Microplastics

Microplastics are small particles less than 5 mm in size that are now abundant in the world's oceans and that take hundreds of years to decompose. About 10% of this type of waste is discharged into the oceans and accounts for ~60% to 80% of marine water wastes (You Li *et al.* 2021). Inputs for microplastics are from both land and water flow (e.g., plastic wastes are washed by wind and rain; You Li *et al.* 2021) and can accumulate quickly in certain areas due to current and tidal effects. Unfortunately, these particles can then be ingested by living organisms such as fish where they can cause toxic effects, as well as reduced foraging, delayed growth, increased susceptibility to disease, and abnormal behavior (You Li *et al.* 2021).

Collicutt *et al.* (2018) investigated the incidence of microplastics in juvenile Chinook on the east coast of Vancouver Island, BC, by completing fish and fish habitat sampling in preferred Chinook rearing habitats (e.g., beach seines, plankton tows, and sediment cores). Microplastics were found in all samples at varying concentrations. Overall, Juvenile Chinook microplastic concentrations (i.e., 1.2 ±1.4 SD per fish) were low compared to other studies conducted elsewhere, suggesting that microplastics in juveniles do not represent an immediate threat in the area. However, microplastics less than 100 μm where not included in this study and may be a greater issue due to their ability to pass through tissues, and this potential threat should be assessed.

5. CONCLUSION

The 2023 report provided by DFO as part of the RPA process for *Species at Risk* listing (Section 1) identified around 70 potential threats to the summer Chinook Salmon population in the Puntledge River (DFO 2023). A ranking of these threats identified 16 that were classified as "Very High" Risk (n=8) or "High" Risk (n=8; Appendix A). Several of these were related to a loss of habitat complexity and availability while others were associated with pinniped predation, fish predation/competition, unfavourable water temperatures, hatchery fish maladaptation to the wild environment, and flow management issues (DFO 2023). Additionally, several potential threats were also identified as a data gap that affected the group's ability to classify them (DFO 2023). These included the prevalence of certain disease/pathogens (e.g., BKD) within the population, impacts of deleterious substances on returning adults and rearing juveniles in the estuary, access to appropriate food for early rearing, and competition with hatchery fish and water temperatures in the estuary.

This report provides a re-evaluation of the potential threats identified by DFO in 2023 (DFO 2023) based on a detailed review of available information (i.e., literature review, internal DFO data, information from local experts) to provide updated threat ranking and data gaps. It is expected that this report will contribute to efforts to (1) correctly classify the population status for Puntledge River summer Chinook (e.g., *Species at Risk Act* listing) and (2) develop and implement recovery and management plans to focus restoration efforts for this population.

The evaluation of the potential threats identified by DFO (DFO 2023), which is provided in Section 4, suggests that the primary threats (those that are high or very high) generally remain the similar, except for some proposed changes, which are highlighted below and Appendix A, which have increased the classification of threats to 11 "Very High" and 11 that are likely "High". These changes are in the sub-section below while a summary of the key threats are provided in a table in the following sub-section.

5.1. Proposed Changes to Key Threats

One potential threat was downgraded from a "Very High" to a "High" while 22 threats were upgraded after further evaluation to High or Very High. Finally, two potential threats that were initially classified as key threats (High, Very High) were determined to have too much of a data gaps to accurately classify at this time.

- Recommended Threat Downgrade:
  - ***Elevated Predation as a Result of Enhancement of Predatory Fish Species (Coho)*** – Predation within freshwater habitats for early rearing was initially assessed as "Very High" but could potentially be downgraded to "High" because the impact of the high numbers of hatchery produced juveniles that are released (e.g., annual releases of 100,000 Coho smolts), is unknown. Additional studies would be needed to confirm the predatory impact.

- Data Gap and Cannot Classify:
  - ***Inter- and Intra- Specific Competition*** - Competition was initially assessed as "Very High" but is more of a data gap than previously anticipated. There is limited information on inter- and intra- specific competition with juvenile Chinook rearing in the estuary; however, progeny from returning fall hatchery adults could potentially compete with juvenile summer Chinook rearing in the estuary. Puntledge hatchery also produces and releases fall Chinook and Chum Salmon, which may compete with summer Chinook juveniles in the estuary. Additional studies would be needed to better assess this potential threat.
  - ***Lack of Access to Appropriate Food*** - lack of access to appropriate food in the estuary was initially assessed as "High" but is more of a data gap than previously anticipated. Preliminary otolith microchemistry analyses of a small number of natural origin summer Chinook indicates that these juveniles grow to a size that is comparable to hatchery juveniles that are reared in freshwater for months. It appears that estuary rearing juveniles are able to access productive food resources and grow quickly; however, this needs to be confirmed with additional sampling.

- Recommended Threat Upgrade:
  - ***Stress due to Anthropogenic Activity (Non-Fishing)*** –
    - This threat to terminal migration and spawning was originally assessed as "High" but is actually expected to be "Very High". Swimmers have in the past congregated at the base of Stotan falls but now more so at Nibs Falls and all

areas of the river. (Stotan falls now has restricted access though some minor activity still persists). Swimmers impacts migration through the fishways during the day. Repetitive and chronic stress can reduce adult survival and affect the fitness of offspring through hormonally-mediated maternal effects. For example, exposure of eggs to cortisol early in development can have persistent effects on juvenile aerobic performance after hatch in Chinook, pink and sockeye salmon (Banet *et al.* 2019). However, there are also data gaps in this assessment that should be evaluated (e.g., re-evaluation of monitoring of migration success).

- ***Stress due to Anthropogenic Activity (Estuary)***
  - This threat to juvenile rearing in the estuary was initially assessed as "Very Low" but is expected to be "High" after review of the available information because there are still many anthropogenic stressors related to historic and ongoing activities and their associated impacts, at play in the estuary. Some key anthropogenic stressors include an increase in human population density along the coastline, channelization of the upper estuary, and invasive plant species.
- ***Limited or Delayed Access due to Migration Barriers and Lack of Safe Migration Routes*** - This threat was initially assessed as "High" but may actually be "Very High" because it looks like there may only be ~30-50% success rate at migrating to the lake due to partial barriers, fishway, recreational interference and high-water temperatures.
- ***Pre-Spawn Mortality*** – Terminal migration leading to pre-spawn mortality was initially assessed as "Very Low" but is expected to be "Very High" after a review of available information. It appears that most of the adult summer Chinook are unsuccessful when migrating to Comox Lake and perish from factors such as high river temperatures.
- ***Increases in heritability of BKD*** – This threat for spawners was not rated initially but is expected to be "Moderate" or "High"; It is unclear if BKD is exacerbated in hatchery broodstock and to what degree it naturally occurs in wild salmon. More studies are needed on the spread and heritability of BKD, particularly for wild fish.
- ***Change in Biological Characteristics*** – This threat was not rated initially but is expected to be "High" because reductions in size of spawners, fecundity and egg size, as well as age structure changes are evident in the population and likely linked to selective fishing pressure, seal predation, hatchery selection/practices and climate change.

- *Impact of Hydroelectric Development on Downstream Chinook Juvenile Migration* – This threat to juveniles in the river was initially not assessed or rated but is expected to be "Very High" due to high fry mortality at Eicher screens.

- *Beach Habitat Loss* – Beach habitat loss was initially rated as "Very Low" but should really be combined with the Vegetation Habitat Loss" threat since they both relate to the loss of saltmarsh and eelgrass habitats, which were rated as "Very High" threat in DFO (2023). Thus, beach habitat loss should also be rated as "Very High".

- *Unfavourable Water Temperatures (Estuary)*– Unfavourable water temperatures in the estuary was initially assessed as a data gap but a detailed evaluation of the available information has provide enough information to classify this threat as "High". Specifically, higher water temperatures in the estuary are likely to become more of an issue with climate change. The Salish Sea annual mean water temperature is predicted to increase by 1.5°C by 2095 (i.e., RCP8.5 scenario; Khangaonkar *et al.* 2019). Higher temperatures in estuaries and inner bays are predicted to cause increased thermal stress for the ecosystem. The warmest temperatures have been found in the lower estuaries, due to the shallower depths and exposure of mudflats at low tides that can warm incoming tidal waters. However, more research needs to be done to understand the specific spatial and temporal thermal regimes that are present in the estuary.

5.2. Summary of Key Threats

Table 56 provides a summary of the 22 key threats that have been identified as part of this assessment of threats for summer Chinook in the Puntledge River watershed.

Table 56. Summary of key threats (rated as "High" or "Very High") for summer Chinook Salmon in the Puntledge River.

Limiting Factor #	Life Stage	Potential Threat	Recommended Threat Classification	Description
1	Terminal Migration and Spawning	Elevated predation (pinnipeds, aquatic species)	Very High	In the 1990s it was estimated that harbour seal consumed 35% of adult and 33% of downstream migration summer Chinook. Following a cull in the late 1990s, seal numbers are rebounding to early 1990s levels and predation on summer Chinook is expected to be an important threat to this population. Seal also use illumination from bridge lights to improve their predation rate on salmon. While some mitigation has been implemented to help address this issue (e.g., light shielding), additional efforts are likely needed and the last assessment of lighting conditions along the river is long overdue. Fences along the river (e.g., hatchery fish fence) may also contribute to the issue by causing accumulations of fish that are easy prey for pinnipeds.
3		Stress due to anthropogenic activity (non fishing)	Very High	Repetitive and chronic stress due to recreational activity, B.C. Hydro flow manipulation (e.g., attraction to the tailrace), fishway passage and operation can reduce adult survival and affect the fitness of offspring through hormonally-mediated maternal effects. For example, exposure of eggs to cortisol early in development can have persistent effects on juvenile aerobic performance after hatch in Chinook, pink and sockeye salmon (Banet *et al.* 2019)
5		Limited or delayed access due to migration barriers and lack of safe migration routes	Very High	Upstream migration in the lower Puntledge River can be delayed at five main locations: 1) the powerhouse, 2) Stotan Falls, 3) Nib Falls, 4) the fishway at the Diversion Dam, and 5) the fishway at the Comox Lake Dam. There are also several areas in the Lower Puntledge River with difficult passage where shallow water flows over bedrock. Delays result in increased physiological stress due to increasing water temperatures during the summer months which can impact the migration success rate into Comox Lake. Climate change is expected to exacerbate this issue (e.g., impact the migration success rate into Comox Lake, which is an essential cold-water refuge for summer Chinook). In summary, decades of either no or sporadic fish access into Comox Lake from 1912 until 2002-03 likely had harmful and possible irreparable impact on the Summer Chinook population, genetic composition and long-term adaptability.
6		Pre-spawn mortality	High	
7		Escapement – Defensible Consistent Enumeration Technique	High	The current adult fence and operational counting procedures do not provide an accurate or consistent estimate of salmon escapement or sex ratio. At low discharges, adults stall at the fence and are often unable to reach the by-bass fishway entrance because of a shallow riffle at the entrance from gravel deposition during the fall-winter. Gravel also deposits on the same side of the river upstream of the fence which reduces attraction flow through the fishway that provides migration further upriver. Finally, some adults can jump over the fence at higher flows. However, defensible escapement estimates are required to assess the status of this endangered population and develop/implement appropriate management decisions to restore/protect this population.
10		Increases in heritability of BKD load	Moderate to High	*Renibacterium salmoninarum* is the causative agent of bacterial kidney disease (BKD), which is an endemic pathogen in the Pacific Northwest. BKD is a lifelong infection of salmonids that can be transmitted among fish and to the next generation. The Puntledge summer Chinook stock was identified as a high risk BKD stock during routine screening of 2009 and 2011 broodstock. In response, hatcheries eliminate or minimize presence of the pathogen (and subsequently the natural environment) by removing progeny from BKD-positive parents; however, this practice could lead to loss of genetic diversity, which can have negative impacts on small populations like the Puntledge summer Chinook run.
11		Changes in biological characteristics	High	There has been a reductions in size of spawners, fecundity and egg size, as well as age structure changes are evident in the population and likely linked to selective fishing pressure, seal predation, hatchery selection/practices and climate change.
16		Unfavorable water temperatures	Very High	The maximum and minimum temperature range that summer Chinook spawners experience is usually within the recommended range. (i.e., 10-15°C). However, water temperature during summer Chinook Salmon migration from late June to August can exceed 20°C, leading to delayed migration and pre-spawning mortality. A migration success rate of only 50-70% is reached when migrating from the Hatchery and into Comox Lake. In the last decade, the frequency of temperatures over 20°C has increased, and is expected to increase further due to climate change, which will likely lead to increased pre-spawning mortality.

Table 56. Continued.

Limiting Factor #	Life Stage	Potential Threat	Recommended Threat Classification	Description
30	Early Rearing	Elevated predation as a result of enhancement of predatory fish species (Coho)	Potentially high but unknown	Little is known about Coho smolt predation on summer Chinook Salmon fry but there is a potential for the coho smolt population in the Puntledge Watershed to consume 35,000 fry /day. This could potentially have a devasting impact on summer Chinook Salmon.
32		Impact of Hydroelectric Development on Downstream Chinook Juvenile Migration	Very High	To achieve 95% fry migration survival past the diversion dam, BC Hydro increases discharge over the dam during higher levels of power generation to presumably increase the proportion of downstream migrating fry that are diverted directly over the dam spillway and thereby avoid diversion into the penstock. However, there is a concern that this displaces and diverts early emergent fry through the Eicher screens, which decreases fry survival. Fry survival is estimated to be >90% only when fish attain a minimum size of 50 mm, which is larger than many summer Chinook fry.
35		Decreased quantity of rearing habitats	High	Reach B, where the majority of emergent fry originate and Reach D, the lowest gradient reach, are subjected to discharges of 100% MAD 57-66% of the time, respectively, during summer Chinook Salmon rearing, which is well over the optimum MAD of 20% . The current BC Hydro Eicher Screen Diversion Dam spill strategy focuses on increasing river discharge when Eicher screen efficiency is low. This is so that overall bypass efficiency over the Diversion Dam can theoretically be increased to over 95% by increasing the number of fry that get diverted directly over the Diversion Dam instead of being diverted through the Eicher Screens. This approach potentially increases the displacement of smaller fry in Reach B and C, exacerbates impingement issues at the Eicher screens for small fry, and decreases the amount of available emergent fry habitat in Reach C.
		Decreased quality of rearing habitats	High	There is information lacking on the amount of habitat that may have been lost historically. However, a summer chinook habitat assessment completed on the Puntledge R. in Reach B and C in 2022 indicated that the percent of bank habitat classified as either moderate or high suitability was 66 and 73.6%, respectively. Additionally, while the wetted area has increased in Reach B due to the construction of the Diversion Dam, the stream banks are steep, water depths are generally over 1 m, and velocities are high for summer Chinook Salmon, reducing the quality of the habitat for rearing.
37		Decreased access to or quality of floodplain habitat	Very High	The headpond, (Reach B), which is approximately 3.75 km long, is the main spawning area for summer Chinook Salmon and is permanently flooded by the BC Hydro Diversion Dam. The banks are steep and water depths along the banks are primarily over 1 m. During late winter and early spring flood events, slow moving floodplain habitat is limited in this reach for rearing salmon. Additionally, a large portion of Reach C is incised and down cut, providing limited opportunities for access to low velocity floodplain habitat. As well, the inter-tidal zone in Reach D is a historic floodplain area that has been impacted by training dikes on the river right in the city of Courtenay and Comox Road on river left (e.g., Fields Sawmill), which has cut off intertidal areas.
39		Frequent and higher peak flows causing flushing	High	It is expected that loss of shallow margin habitats with low velocities and vegetative cover would adversely affect summer Chinook abundance, particularly in the spring. Thus, the amount of intact riparian habitat that acts as wetland habitat or instream cover along the mainstem and side channels may be a limiting factor for Chinook production in Puntledge River (particularly in years of high fry production when habitat can be limiting). Loss of such habitats due to flood protection, farming, and range tenures, as well as residential/industrial development, is therefore a concern for the productive capacity of Chinook salmon. BC Hydro discharge increases during fry emergence and early rearing have been observed to have impacts on rearing.
41		Competition with hatchery fry	Very High	Fall Chinook and hatchery fry are larger in size, which favours them in a competitive setting with the smaller summer Chinook. As presence of fall and hatchery Chinook fry is quite high (e.g., average natural production of fall Chinook emergent fry could be as high as ~2 million annually while up to 1.8 million enhanced fall Chinook sub-yearling smolts are released from the hatchery each year), summer Chinook fry are at a disadvantage for resource and this could result in a lower survival rate and potential fish production.

Table 56. Continued.

Limiting Factor #	Life Stage	Potential Threat	Recommended Threat Classification	Description
46	Rearing in the estuary	Elevated predation	Very High	Lack of complexity and channelization in the intertidal areas has likely contributed to an increase in seal predation in the estuary. Overall, studies have shown that predation on downstream migrating chinook juveniles and returning summer chinook adults can reach 30% mortality.
49		Stress due to anthropogenic activity	High	Past industrial activities (e.g., industrial sawmills, log booming and dredging) has caused damage to the coastal habitats. In addition, there have been impacts to saltmarsh and eelgrass habitats and the breakwater has disrupted the normal pattern of sediment delivery along the shoreline, starving the site of sediment and slowing aquatic vegetation recovery in the area. Additional impacts to the estuary include historical dredging of the river and concomitant dredge spoil deposition, the in-filling of brackish saltmarsh habitat to create the Courtenay Airpark airstrip, the development of the now defunct Courtenay sewage lagoon, diking, and the construction of the Courtenay Marina, all of which have altered the habitat and the distribution of freshwater across the estuary delta.
		Estuary channelization	High	Hard armoring such as bulkheads, seawalls, dikes, tide-gates, groynes, and rip rap are common in coastal communities and have been the conventional approach to protect shorelines and hard infrastructure such as roads from coastal flooding and erosion. For example, the entire Courtenay River has been artificially channelized and narrowed as the community of Courtenay developed protection to prevent flooding in low-lying areas. However, these changes disrupt natural shoreline sediment transport processes, increases erosion in adjacent unprotected areas and degrades the quality of natural shoreline habitats.
52		Freshwater and estuary habitat utilization	Very High	The inter-tidal zone habitat in Reach D has been reduced due to training dikes on the right side of the river in the city of Courtenay and Comox Road on the left side.
54		Vegetation habitat loss	Very High	Several anthropogenic have led to the loss of aquatic vegetation within the estuary. Agricultural fields now cover about 75% of the original estuary and the remaining salt marsh habitats are isolated from each other. Another area that has experienced a significant decline in the amount of salt marsh habitat is the outlet of the Courtenay River. K'ómoks estuary salt marsh habitat also shows a loss of 16.41 hectares (ha) or a 25.79% decline in the amount of habitat between 1980 and 2016. Additionally, eelgrass habitat within the estuary declined by 94.71 ha or 37.94% between 1979 and 2016. The loss of eelgrass in this area could be due in part to the substantial nutrient loading from sewage effluent, which can promote the growth of an algae that grows above or on seagrass and prevents it from getting the required sunlight for photosynthesis Furthermore, large blooms of invasive *Sargassum muticum* have been linked to warmer ocean temperatures and nutrient loading from sewage, which now appears to be outcompeting the eelgrass.
		Beach habitat loss	Very High	Beach habitat has also deteriorated over the last decades due to invasion by non-native shoreline species. The impacts of this change in beach habitat quantity is likely having an impact on mortality or fitness of summer Chinook rearing in the estuary, but additional studies would be required to determine the scale of that effect.
56		Unfavourable water temperatures	High	Climate Change impacts include increased temperature, sea level rise, higher increase in the number of hypoxia events, and increased CO_2, all of which impact rearing water quality, habitats and food availability for juvenile Chinook.

6. DATA GAPS

A total of 11 threats could not be properly assessed due to data gaps, which are summarized below:

- ***Non-Sanction Fishing Mortality*** - A monitoring program is recommended to properly assess this risk to spawners.
- **Increase in Didymo Abundance**
  - *Incubation* - Effect of didymo on intragravel environment, hyporheic flow and potential gill irritation for alevins/emerging fry is a general data gap that also applies to the Puntledge River wateshed.
  - *Early Rearing* - Effect of didymo on fry is also relatively unknown.
- **Lack of Access to Appropriate Food** - Data gap regarding where and when juveniles feed in the river or what they feed on. Mortality or fitness impacts as a result of lack of food.
- **Freshwater and Estuary Habitat Utilization due to River Flow** - Preliminary results of otolith analysis indicate that summer Chinook juveniles use the freshwater and estuarine habitat equally. However, sample size is small and needs to be increased to confirm results.
- **Inter- and Intra-Specific Competition** - There is high potential for competition; however, the number of wild fall chinook fry derived from +5000 spawners has not been assessed. The interaction and competition with wild and hatchery chum fry is unknown. The proportion of summer Chinook juveniles that rear for an extended period in the river versus the estuary has just recently been investigated and the current sample size is small and only from one brood year.
- **Disease, Parasites, or Pathogen**s - Disease sampling and analyses of juveniles rearing in the estuary is lacking.
- **Lack of Access to Appropriate Food** - There is limited growth data in the estuary that indicates access to food is a limiting factor. Preliminary otolith microchemistry analyses of a small number of natural origin summer Chinook, although only available for one brood year, indicates that summer Chinook juveniles grow quickly suggesting that access to food may not be limiting but additional studies are needed.
- **Increased Frequency and Magnitude of Algal Blooms** - More research is needed to understand the pH conditions in the estuary and the impact of algal blooms on summer Chinook Salmon.
- **Competition with Hatchery Fish** - Known level of competition, but not the degree of impact on summer Chinook Salmon in the estuary.

- **Water Quality Threats (pH, salinity**) - Current issue may be domestic runoff. No monitoring at current time so severity of this issue cannot be determined for the estuary.

Some data gaps have also been identified for the key threats (i.e., "Very High", "High") which are summarized in Appendix A. These data gaps would also benefit from additional studies to confirm the findings that were used to classify these key threats to summer Chinook Salmon. Overall, it is clear that a comprehensive stock status analysis of Puntledge summer Chinook is required to evaluate the response of the population to past recovery actions, including habitat and hatchery related activities, as well as guide future decisions regarding the recovery of this stock.

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