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### Recovery Potential Assessment for 11 Designatable Units of Chinook Salmon, *Oncorhynchus tshawytscha*, Part 2: Elements 12 to 22

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## Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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## LIST OF ACRONYMS

- AIS** – Aquatic Invasive Species
- BC** – British Columbia
- CNR** – Chinook Non-Retention
- COSEWIC** - Committee on the Status of Endangered Wildlife in Canada
- CTC** – Chinook Technical Committee of the Pacific Salmon Commission
- CU** – Conservation Unit
- CWT** – Coded Wire Tag
- CYER** – Calendar Year Exploitation Rate
- DFO** – Fisheries and Oceans Canada
- DU** – Designatable Unit
- eDNA** – Environmental DNA
- ERG** – Exploitation Rate Goal
- FFHPP** – Fish and Fish Habitat Protection Program
- FLNRORD** - Ministry of Forests, Lands, Natural Resource Operations and Rural Development
- FRC** – Fraser River Chinook
- IFC** – Interior Fraser Coho
- IPCC** – International Panel of Climate Change
- LET** – Lower Escapement Target
- MLE** – Maximum Likelihood Estimate
- MoE** – Ministry of Environment and Climate Change
- MU** – Management Unit
- PST** – Pacific Salmon Treaty
- PVA** – Population Viability Analysis
- QET** – Quasi-Extinction Threshold
- RER** – Rebuilding Exploitation Rate
- RPA** – Recovery Potential Assessment
- SARA** – Species at Risk Act
- S<sub>gen</sub>** – The spawner level required to achieve the number of spawners at maximum sustainable yield (**S<sub>msy</sub>**) within one generation
- S<sub>max</sub>** – Spawner level above which density-dependence results in reduced recruitment with increasing spawner abundances
- S<sub>msy</sub>** – Spawners at maximum sustainable yield
- S-R** – Stock-Recruitment
- S<sub>rep</sub>** – Spawners required to produce replacement
- UET** – Upper Escapement Target
- WSP** – Wild Salmon Policy

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## ABSTRACT

Eleven Designatable Units (DUs) of Chinook Salmon (*Oncorhynchus tshawytscha*), within the Fraser River were assessed as Threatened or Endangered by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) in 2018, and are currently under consideration for addition to Schedule 1 of the *Species at Risk Act* (SARA). The first part of the Recovery Potential Assessment (RPA) (Elements 1-11) provided DU descriptions, status updates and an assessment of the threats and factors limiting recovery. This second half provides potential recovery targets, a discussion of mitigation measures, population projections and a recommendation on allowable harm. Survival and recovery targets for each DU were suggested based on Wild Salmon Policy (WSP) benchmarks, with additional requirements about observed percent change in spawners. A projection model was used to assess likely future trajectories and the chances of meeting these targets, however, results are only available for DU2 (LFR-Harrison). Despite efforts to produce the required input parameters for the stream-type DUs, significant uncertainties and a considerable lack of data prevented quantitative assessment, and hence these DUs were assessed qualitatively. Results for DU2 (LFR-Harrison) indicate that reaching the survival target under recent conditions is about as likely as not likely (48% chance), while meeting the recovery goal is unlikely (16% chance). Risks imposed by climate change and continued anthropogenic development add additional uncertainty that was only described qualitatively. Based on the quantitative assessment for DU2 (LFR-Harrison) and the qualitative assessment for the remaining DUs, it is recommended that human-induced mortality and other sources of harm identified in the threats assessment should be significantly reduced and in some cases prevented to provide the best chance for these populations to recover.

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## 1. INTRODUCTION

Subsequent to the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) assessing an aquatic species as Threatened, Endangered or Extirpated, Fisheries and Oceans Canada (DFO) undertakes a number of actions required to support implementation of the *Species at Risk Act* (SARA). Many of these actions require scientific information on the current status of the wildlife species, threats to its survival and recovery, and the feasibility of recovery. Formulation of this scientific advice is typically developed through a Recovery Potential Assessment (RPA) within a designated timeframe following the COSEWIC assessment. This timing allows for consideration of peer-reviewed scientific analyses into SARA processes. The RPA provides the Species at Risk Program the information needed to understand the current state and likely future state of the species, which allows for the evaluation of potential management options and provides the base for recovery planning if listed. The information provided in the RPA is based on 22 core Elements. Part 1 of this RPA, Elements 1 to 11, discussing species biology and threats to the populations, was reviewed in December 2019 (DFO 2020a). Part 2 of the RPA (elements 12 to 22) covers recovery targets, forward projections, discussion of mitigation measures and an assessment of allowable harm, are addressed in this report. Many of the elements in this report rely on information provided in Part 1 of the RPA. It is highly recommended that the reader review Part 1 prior to Part 2.

### 1.1. SPECIES INFORMATION

Chinook Salmon are the largest of five semelparous and anadromous Pacific salmon species native to North America, ranging from central California to the Mackenzie River (Northwest Territories, Canada) along the North American coast (Netboy 1958; McPhail and Lindsey 1970; McLeod and O'Neil 1983; Healey 1991). Chinook Salmon have the most diverse life history patterns of all the semelparous Pacific salmon (Brannon et al. 2004), with considerable variation in size, age at maturation, habitat requirements, and duration of freshwater and saltwater rearing stages. In Canada, Chinook Salmon are an important food source for other fish, mammals, birds, amphibians and reptiles, as well as a key target species for recreational and commercial fisheries, and are highly significant to Indigenous peoples in British Columbia (BC) as a cultural symbol and connection to a way of life for subsistence (COSEWIC 2018).

Chinook Salmon populations in Southern BC are subdivided into 28 Designatable Units (DUs) by COSEWIC based on geographic distribution, life-history variation, and genetic data (COSEWIC 2018). Delineation of COSEWIC DUs follow the same fundamental approach for maintaining genetic variability at the wildlife species level as Wild Salmon Policy (WSP) or Conservation Units (CUs) with some similarities and differences between these delineation types (COSEWIC 2018). In some instances, multiple CUs can make up a DU. For Fraser River Chinook Salmon (FRC), 25 of the 28 DUs are exactly the same as the CUs, while 3 of the DUs have different population boundaries. Each of the DUs discussed in this RPA represent a single FRC CU. Detailed descriptions of COSEWIC DUs and WSP CUs for FRC can be found in COSEWIC (2015, 2018) and Brown et al. (2019) respectively.

For the context of this RPA, FRC DUs are genetically distinct populations that do not readily interbreed, and spawn within different geographical reaches of the Fraser River drainage (see (COSEWIC (2018) for detailed description of FRC genetics and geographic distribution). The DUs assessed in this RPA, and their corresponding WSP CUs and fisheries Management Units (MUs), are summarized in Table 1.

*Table 1 - Southern British Columbia Chinook (CK) Salmon Designatable Units (DU) in the Fraser River assessed as Endangered or Threatened, reasons for designation from COSEWIC (2018), and their relation to Wild Salmon Policy Conservation Units (CU) and fisheries Management Units (MU). The MU numerical notation refers to the dominant life history type for each DU: 4<sub>2</sub> and 5<sub>2</sub> are stream-type Chinook salmon where juveniles migrate to sea as yearlings and return at a total (freshwater + marine) age of 4 or 5 years; 4<sub>1</sub> is an ocean-type life history where juvenile migrate to sea as underyearlings and primarily return at 4 years total age.*

<b>Management Unit (MU)</b>	<b>Conservation Unit (CU)</b>	<b>Designatable Unit (DU)</b>	<b>COSEWIC Status</b>	<b>Reasoning for Status</b>
Spring 5 <sub>2</sub>	CK-08 FR Canyon-Nahatlatch	DU7 - Middle Fraser River Stream Spring (Nahatlach)	Endangered	This population of spring run Chinook spawning in the Nahatlatch River watershed has declined to very low levels. Declines in freshwater and marine habitat quality, and harvest, are threats facing this population.
	CK-10 MFR Spring	DU9 - Middle Fraser River Stream Spring	Threatened	This spring run of Chinook spawning in multiple Middle Fraser River tributaries has declined in abundance. Declines in marine and freshwater habitat quality, and harvest, and pollution from mining activities are threats to this population.
	CK-12 UFR Spring	DU11 - Upper Fraser River Stream Spring	Endangered	This spring run of Chinook spawning in the Upper Fraser River watershed has declined in abundance. Declines in marine and freshwater habitat quality, and harvest, are threats facing this population. Anticipated changes to North Pacific weather systems that affect ground water availability, will impact spawning sites and overwinter survival.
	CK-18 NTh Spring	DU16 - North Thompson Stream Spring	Endangered	This spring run of Chinook spawning in the North Thompson River has steeply declined in abundance to a low level. Declines in marine and freshwater habitat quality, and harvest, are threats facing this population. Anticipated changes in North Pacific weather systems that affect groundwater availability will impact spawning sites and overwinter survival.
	CK-05 LFR Upper Pitt	DU4 - Lower Fraser River Stream Summer (Upper Pitt)	Endangered	This summer run of Chinook spawning in the Pitt River in the Lower Fraser River watershed has declined, and is now at its lowest recorded abundance. Declines in freshwater and marine habitat quality, and harvest, are continuing threats to this population.
Summer 5 <sub>2</sub>	CK-06 LFR Summer	DU5 - Lower Fraser River Stream Summer	Threatened	This summer run of Chinook spawning in the Lillooet and Harrison Rivers in the Lower Fraser watershed has declined to low levels. Declines in freshwater and marine habitat quality, and harvest, are threats facing this population.
	CK-09 MFR Portage	DU8 - Middle Fraser River Stream Fall (Portage)	Endangered	This population of fall run Chinook spawning in the Seton and Anderson River watersheds along the Middle Fraser River has declined to very low levels, and decline is anticipated to

Management Unit (MU)	Conservation Unit (CU)	Designatable Unit (DU)	COSEWIC Status	Reasoning for Status
				continue. Declines in freshwater and marine habitat quality, and harvest, are threats facing this population.
	CK-11 MFR Summer	DU10 - Middle Fraser River Stream Summer	Threatened	This summer run of Chinook spawning in multiple Middle Fraser River tributaries has declined in abundance. Declines in marine and freshwater habitat quality are threats facing this population.
	CK-19 NTh Summer	DU17 - North Thompson Stream Summer	Endangered	This summer run of Chinook spawning in the North Thompson River has steeply declined in abundance. Declines in marine and freshwater habitat quality, and harvest, are threats facing this population.
Spring 4 <sub>2</sub>	CK-16 SThBessette Creek	DU14 - South Thompson Stream Summer (Bessette)	Endangered	This summer run of Chinook spawning in the South Thompson River has steeply declined in abundance to a very low level. Declines in marine and freshwater habitat quality, and harvest, are threats facing this population.
Fall 4 <sub>1</sub>	CK-03 LFR Fall	DU2 - Lower Fraser River Ocean Fall	Threatened	While the calculation of rates of decline are complicated by hatchery releases from 1981 to 2004, this fall run of Chinook spawning in the Lower Fraser River has steadily declined in abundance. The abundance data over all available years was thought to best represent natural spawner abundance. Declines in marine and freshwater habitat quality, harvest and ecosystem modification in the Lower Fraser River estuary, are threats facing this population.

## 1.2. OVERVIEW OF PART 1 ELEMENTS 1 TO 11

Part 1 of the RPA provided updated trend assessments for the populations, an overview of the biology and habitat requirements, and an assessment of the threats and factors limiting recovery. The trends assessment included three additional years of data (2016-18) and the results suggested that all DUs have continued to decline since the COSEWIC assessment in 2018 (Figure 1 and Figure 2). Through a workshop with local experts, the major threats impacting the DUs were determined to be climate change, natural system modifications, fishing and pollution (Table 2). Threats to individual DUs include: recent landslides posing serious risks to DUs 8 (MFR-Portage), 9 (MFR-Spring), 10 (MFR-Summer) and 11 (UFR-Spring); competition with hatchery fish for DU2 (LFR-Harrison); and particularly high impacts due to natural systems modifications for DUs 9 (MFR-Spring) and 14 (STh-Bessette). All eleven DUs are considered to be at a High-Extreme or Extreme threat risk, due to the severity and number of threats these DUs are facing. It was concluded that alleviating the multiple and complex threats to these DUs will be difficult, especially as many of the threats are exacerbated by climate change. It will be critical to ensure that efforts are appropriately coordinated through effective governance to successfully mitigate the cumulative impacts of these diverse threats. Further information can be found in DFO (2020a).

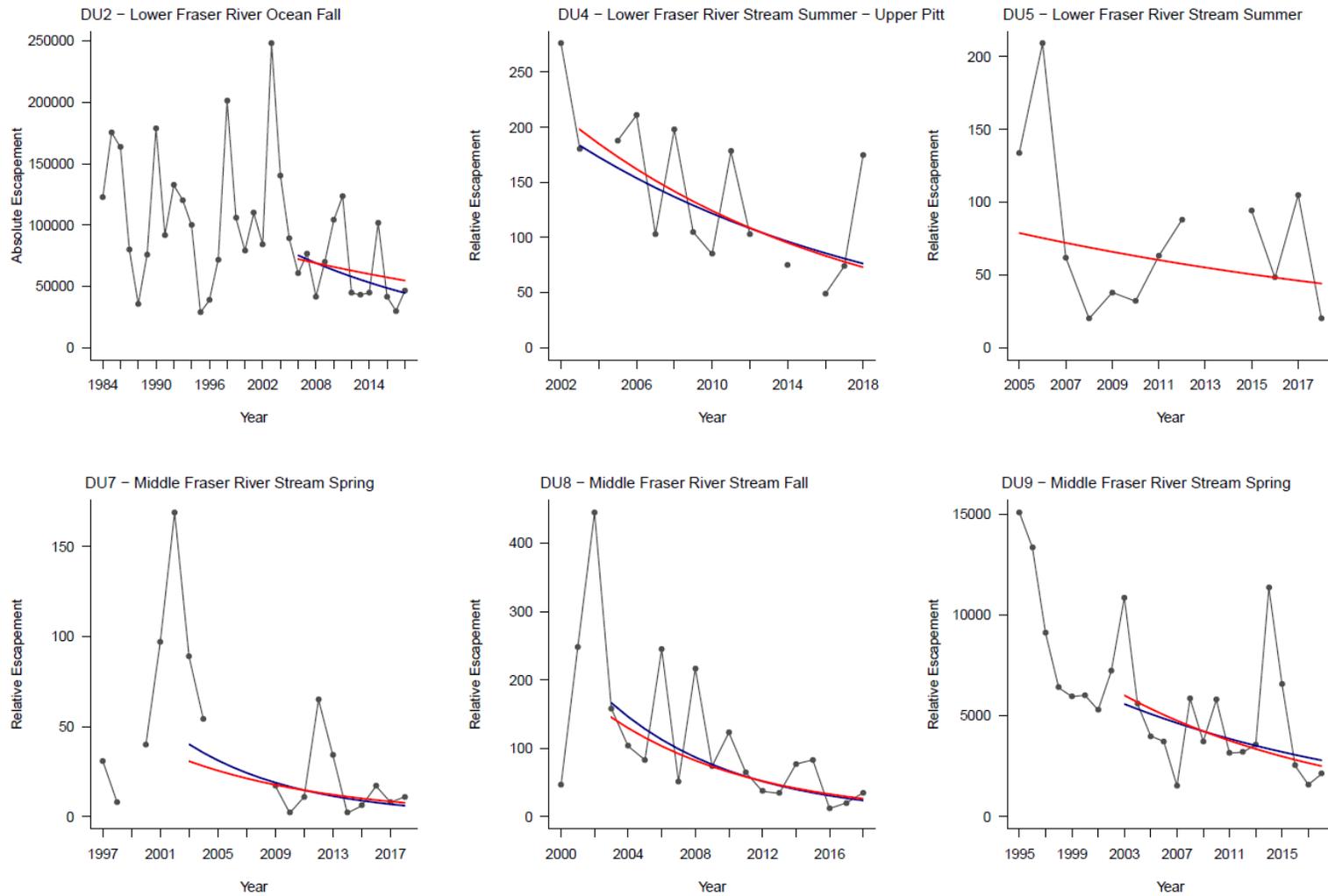


Figure 1 - Time series in absolute (DU2 only) and relative escapement with two estimates of the rate of change in log-escapement through time: (blue) rate of change over the last three generations based only on the last three generations of data, and (red) rate of change over the last three generations based on all available data. Note that DU5, does not have three generations of data, and hence only has a red line.

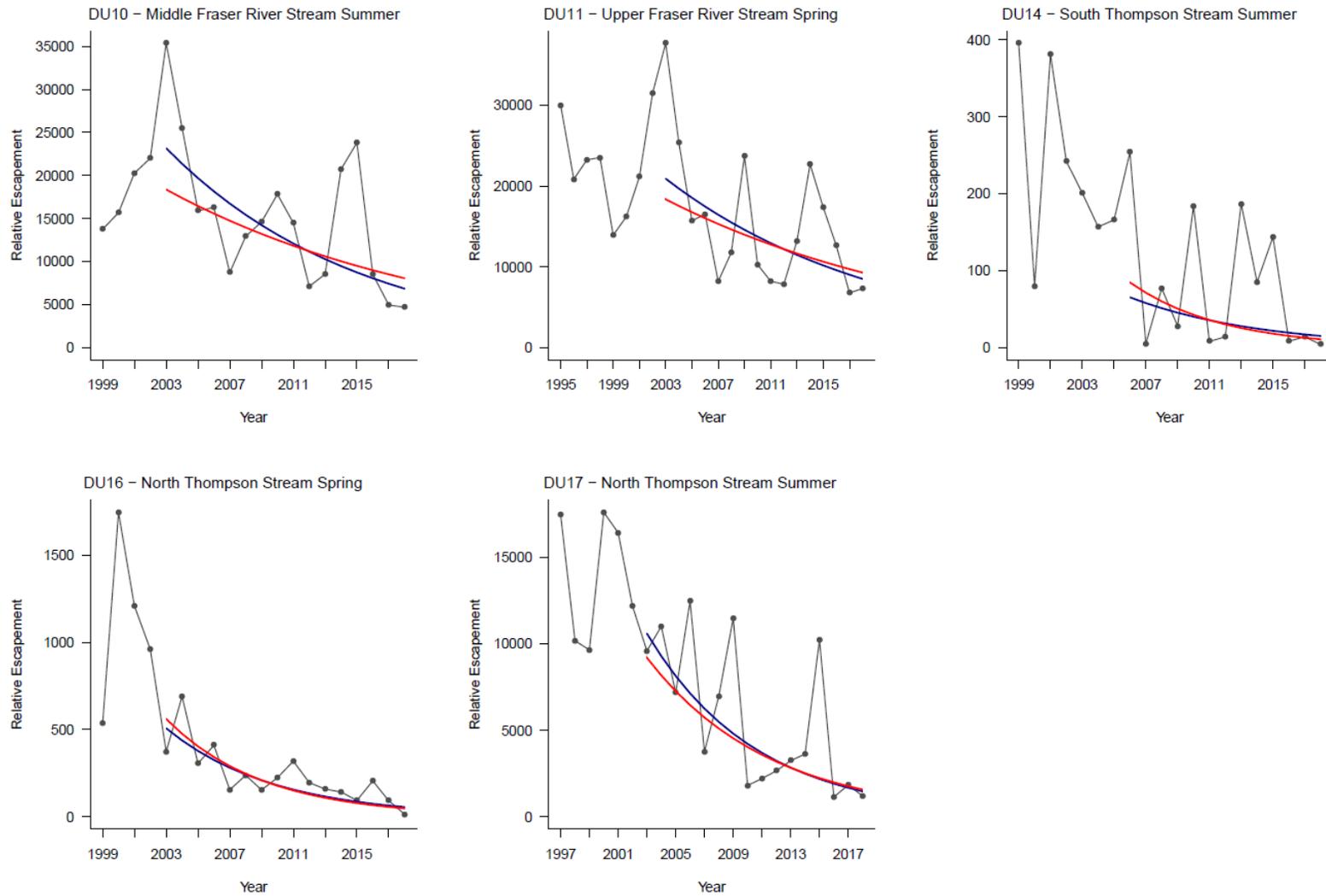


Figure 2 - Time series of relative escapement with two estimates of the rate of change in log-escapement through time: (blue) rate of change over the last three generations based only on the last three generations of data, and (red) rate of change over the last three generations based on all available data.

Table 2 - Summary threats table from Part 1 of the RPA, with the rolled up COSEWIC threat categories and overall threat ranking for FRC DUs assessed in the RPA. Additional details about the threats assessment can be found in Part 1 of the RPA.

COSEWIC Major Threat Category	DU2	DU4	DU5	DU7	DU8	DU9	DU10	DU11	DU14	DU16	DU17
Residential and commercial development	Low	Low	Low	Negligible	Negligible	Negligible	Negligible	Negligible	Negligible	Negligible	Negligible
Agriculture & aquaculture (Hatchery competition)	High-Medium	Low	Low	Low	Low	Low	Low	Low	Medium-Low	Low	Low
Energy production & mining	Medium-Low	N/A	Low	Low	Low	Medium	Low	Low	Low	Low	Low
Transportation & service corridors	Unknown	Unknown	Unknown	Unknown	Unknown	Negligible	Negligible	Negligible	Unknown	Low	Low
Biological resource use (Fishing)	High-Low	High-Low	High-Low	High-Low	High-Low	High-Low	High-Low	High-Low	High-Low	High-Low	High-Low
Human intrusions & disturbance	Negligible	Medium-Low	Negligible	Negligible	Negligible	Low	Negligible	Low	Low	Low	Low
Natural systems modifications (Water management, ecosystems modifications)	Medium-Low	Medium-Low	Medium-Low	Medium-Low	Medium	High-Medium	Medium-Low	Medium-Low	Extreme - High	Medium-Low	Medium-Low
Invasive & other problematic species & genes	Medium-Low	Medium-Low	Medium-Low	Low	Low	Low	Low	Low	Medium-Low	Low	Low
Pollution (From all sources and threats)	Medium	Medium-Low	Medium-Low	Medium-Low	Medium-Low	Medium	Medium-Low	Medium-Low	Medium	Medium	Medium
Geological events (Landslides)	Unknown	Unknown	Unknown	Unknown	High	Extreme	Extreme	Extreme	Unknown	Medium-Low	Medium-Low
Climate change & severe weather (Shifting habitats)	High-Low	High-Medium	High-Medium	High-Medium	High-Medium	High-Medium	High-Medium	High-Medium	High-Medium	High-Medium	High-Medium
<b>OVERALL THREAT RANKING</b>	Extreme-High	Extreme-High	Extreme-High	Extreme-High	Extreme	Extreme	Extreme	Extreme	Extreme	Extreme-High	Extreme-High

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## 2. ELEMENT 12: RECOVERY TARGETS

*Element description: Propose candidate abundance and distribution target(s) for recovery.*

For all FRC DUs considered in this report, both a survival and recovery target were proposed and used in this RPA (Table 3). The survival target is aimed at reaching a COSEWIC status of Special Concern, whereas the recovery target represents a benchmark of recovery or a status of Not at Risk. This approach is consistent with DFO advice on setting SARA recovery targets (DFO 2011). The survival target may represent a limit reference point that triggers rebuilding and recovery plans when spawner abundances drop below the target; whereas the recovery target may indicate an ideal management objective. In other words, the survival target represents the minimum population level required for long-term persistence, and could be viewed as a short-term goal on the way to recovery. The definition of the survival target in this report does not match the definition of survival under SARA guidance, as the survival target defined here is based on the COSEWIC approach and can include a declining trend if abundance is sufficiently high. Biological recovery benchmarks for these FRC DUs were selected based on both the COSEWIC criteria for status designation and the WSP benchmarks. While the targets presented here attempt to be consistent with the COSEWIC and WSP assessments, the suggested targets are highly simplified targets compared to the more nuanced criteria used in the expert driven processes involved in the COSEWIC and WSP assessments, which include a broad range of criteria. Accordingly, achieving either the survival or recovery target does not necessarily mean that there will be a corresponding change in the COSEWIC or WSP status of a DU.

The majority of these DUs were designated Threatened or Endangered by COSEWIC due to observed declining trends in total number of mature individuals. However, DU7 (MFR-Nahatlatch) was designated Endangered solely due to its small population size, as there was insufficient data to evaluate trends at the time of the COSEWIC assessment. DU8 (MFR-Portage) was also assessed due to a small population size, in addition to the decline in total number of individuals. Many DUs examined in this RPA have abundance indices below 1,000 spawners, however, there is uncertainty in assessing total population size as population estimates are generated from relative abundance. Additionally, for some DUs the relative abundance estimates are based on only one or two spawning sites while multiple sites exist but are not surveyed.

The recovery targets proposed here contain abundance and population trajectory benchmarks. There are other variables that could be considered as part of the recovery, such as maintenance or expansion of distribution, productivity levels or inferences of productivity (e.g. trends in fecundity, size at age or maturation rates), genetic diversity, or threat mitigation which all could provide indications of the state of the population and its resiliency. A discussion of trends in many of these life history parameters were discussed in Part 1, but many of these variables are unknown for these populations and hence no specific targets were set. These variables, to the extent that data existed were considered when assessing the ability of these stocks to achieve the survival and recovery targets, and further discussion is in Section 9. All of these aspects are recommended for consideration in the definition of recovery and setting targets if these DUs are listed by SARA, however, for the purpose of the RPA, targets focused on those attributes which can be objectively assessed against.

Based on the COSEWIC criteria and the reasons for designation for these DUs, most DUs only require positive population growth to potentially achieve Special Concern status. However, minimum abundance targets were also included since many of these DUs currently have historically low estimates of relative abundance and the percent change requirements may not adequately ensure persistence or recovery. The spawner level required to achieve the number

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of spawners at maximum sustainable yield ( $S_{msy}$ ) within one generation ( $S_{gen}$ ) was selected for the survival abundance target as this metric has performed well in evaluations under scenarios with varying productivity (Holt 2009; Holt and Bradford 2011) and is consistent with the WSP abundance lower benchmark. The recovery abundance target was set to 85% of  $S_{msy}$ , to correspond with the abundance component of WSP green status for Chinook Salmon. These abundance benchmarks are evaluated as a generational average abundance. For DUs with an  $S_{gen}$  or  $S_{msy}$  less than 1,000 spawners, the abundance target was set to a minimum of 1,000 to ensure that COSEWIC Criterion D is exceeded.

An alternative approach to set abundance targets, as was done in the Interior Fraser Coho (IFC) RPA (Arbeider et al. 2020), was considered for DUs with spawning sites in multiple watersheds. Unfortunately, there were no conditions where 1,000 spawners were present in all the enumerated sites for any of the DUs with multiple spawning locations. Ultimately it was determined that habitat-based benchmarks would provide the best abundance target despite the uncertainties. However, the intention is to maintain all spawning locations and escapement should be monitored across the spatial extent of the DU area to confirm persistence or expansion of current distributions.

The population trajectory component of the recovery targets is measured through the percent change over three generations. Percent change requirements associated with the two abundance targets described above, are based roughly on COSEWIC criterion A and C. When the abundance target is above 10,000 spawners, a less than 30% decline is required (Criterion A), and when the target is below 10,000, positive population growth is required (Criterion C). Both criterion A and C, have additional nuanced requirements that are not used here.

DU estimates of  $S_{gen}$  and  $S_{msy}$  were generated using a habitat-based method (Parken et al. 2006; referred to as “the habitat model”), with the exception of DU2 (LFR-Harrison) where stock recruit (S-R) data were available. The habitat-based estimates presented in this report are updates to the benchmarks presented in the 2014 WSP Assessment, and use the most recent version of the habitat model (Table 4). An overview of the process to calculate the benchmarks is provided below. An excerpt from the forthcoming WSP Assessment Research Document with the detailed description of the methods used to calculate the benchmarks is provided in Appendix A.

The habitat model is a predictive regression model based on a meta-analysis of stock-recruitment reference points (i.e.  $S_{msy}$  and  $S_{rep}$ ) and the accessible watershed area. The updated equation from Parken et al (2006) used for the benchmarks in this report is provided in Table 4 for these stream-type DUs. Watershed areas were previously calculated for the 2014 WSP Assessment with ArcGIS, using the BC Watershed Atlas, the Fisheries Information Summary System (FISS), and were peer reviewed by field program staff who conduct spawning ground surveys. This information was applied to determine the Chinook accessible watershed area for each DU. DUs with spawning in a single watershed have only one estimate of  $S_{msy}$  and  $S_{rep}$ , while other DUs with spawning across multiple watersheds, have several estimates of watershed areas with individual estimates of  $S_{msy}$  and  $S_{rep}$  that align with the stock units and the population dynamics. To arrive at a DU-level habitat estimate of  $S_{msy}$  and  $S_{rep}$  for DUs with multiple watershed areas, joint distributions of  $S_{msy}$  and  $S_{rep}$  from the individual estimates for all watersheds contributing to the DU were calculated, from which a DU level estimate of  $S_{gen}$  and  $S_{msy}$  could be calculated. The watershed areas used for the stream-type DUs in this report are presented in Table 5. The estimates of  $S_{gen}$  and  $S_{msy}$  estimated from the habitat model output could vary from estimates derived using stock-recruit (S-R) analyses with DU specific data, based on the leave-one-out analysis conducted in Parken et al. (2006). At this time it is not possible to verify these model estimates against DU specific data. If S-R data for these DUs becomes available, they can be used to generate more representative recovery targets for the

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DU, and be included in the model to provide more accurate predictions and better represent the productive capacity of each DU.

Habitat-based benchmarks are abundance targets for all DUs, with the exception of DU5 (LFR-Summer). The recovery target for DU5 (LFR-Summer) does not apply to the abundance for the DU as a whole, but rather segments of the DU where estimates of spawner abundance are available as the Lillooet River system is not included in this estimate for  $S_{gen}$  or  $S_{msy}$ . Since the habitat-based benchmarks are true abundance benchmarks, it is difficult to compare for stream-type DUs where only relative abundance data is available. The escapement estimates available for these DUs may underestimate the population size and there will be a discrepancy when comparing to the absolute abundance benchmarks. Until unbiased abundance estimates are available for these DUs, either through a significant expansion of stock assessment activities or the development of scalars to relate relative abundance to true abundance, it is recommended that evaluating whether the abundance target is met or not be done using the relative abundance estimates available. The lack of absolute abundance data is a gap that needs attention, but using relative abundance for now will provide a precautionary assessment of DU status.

DU2 (LFR-Harrison) estimates of  $S_{gen}$  and  $S_{msy}$  were not updated from the 2012 WSP assessment. There was discussion within the author group about updating these values, as a recent S-R analysis with time-varying productivity provided evidence of a recent decline in productivity for DU2 (LFR-Harrison) to a historic low (see Section 5.1). If the S-R values were updated, using the recent low productivity, it would produce a higher  $S_{gen}$  value (33,988 vs 15,318), but a significantly lowered 85%  $S_{msy}$  value (40,146 from 63,808). These values were not updated for several reasons. First, previous guidance recommended that biological reference points only be updated using a time-varying productivity model when a decline in productivity is well documented, quantifiable, and likely to be a persistent state compared to the management regime (Duplisea and Cadigan 2012; Holt and Michielsens 2020). To meet these criteria, additional research and data points in the time series, along with a broader group discussion, would be required to determine if this is likely a persistent state of low productivity. Second, the WSP  $S_{gen}$  estimate for DU2 (LFR-Harrison) is above the 10,000 spawners required to exceed the COSEWIC C criterion to achieve a status of Not Endangered or Threatened, and hence is already more precautionary than the COSEWIC guidelines. Lastly, the  $S_{msy}$  value of 75,068 used to calculate the WSP benchmark has been committed to in the bi-laterally agreed-upon Pacific Salmon Treaty (PST) as an escapement goal for DU2 (LFR-Harrison) (Pacific Salmon Treaty, 1985). If in the future the WSP benchmarks or the PST escapement goal are adjusted, or if a new persistent productivity regime is quantified and documented it may be appropriate to adjust the recovery targets.

As noted above, there are many variables and factors that could change the selection and estimation of survival and recovery targets. Many of these variables (e.g. distribution, fecundity, size at age, productivity) are data gaps for most of the DUs assessed here. For the most data-rich DU, DU2 (LFR-Harrison), an alternative approach used to determine its abundance benchmark using a shifting baseline is described in Section 6. As mandated by SARA, models and targets should be reviewed as more data become available.

Table 3 - Survival and recovery targets for each DU assessed. The survival target aims to achieve COSEWIC Special Concern status. The recovery target is set to achieve Recovered or Not at Risk status. To meet the target each population must achieve both the abundance and % change requirement. Abundance is based on S<sub>gen</sub> or 85% S<sub>msy</sub> for the survival or recovery targets respectively, unless otherwise indicated, and is measured against a generational average.

DU	DU Short Name	Survival Targets		Recovery Targets	
		Abund.	% Change Requirement	Abund.	% Change Requirement
DU2	LFR-Harrison	15,318	< 30% decline	63,808	< 30% decline
DU4	LFR-Upper Pitt	1,000 <sup>2</sup>	Positive population growth	1,000 <sup>2</sup>	Positive population growth
DU5 <sup>1</sup>	LFR-Summer	1,000 <sup>2</sup>	Positive population growth	1,285	Positive population growth
DU7	MFR-Nahatlach	1,000 <sup>2</sup>	Positive population growth	1,000 <sup>2</sup>	Positive population growth
DU8	MFR-Portage	1,000 <sup>2</sup>	Positive population growth	1,358	Positive population growth
DU9	MFR-Spring	5,331	Positive population growth	22,216	< 30% decline
DU10	MFR-Summer	5,878	Positive population growth	25,260	< 30% decline
DU11	UFR-Spring	5,273	Positive population growth	24,883	< 30% decline
DU14	STh-Bessett	1,000 <sup>2</sup>	Positive population growth	1,000 <sup>2</sup>	Positive population growth
DU16	NTh-Spring	1,000 <sup>2</sup>	Positive population growth	3,865	Positive population growth
DU17	NTh-Summer	1,824	Positive population growth	7,773	Positive population growth

<sup>1</sup> For DU5, the recovery target only represents a target for the sampled systems, not the DU as a whole, as the Lillooet River system is not included in this estimate.

<sup>2</sup> For DUs with an S<sub>gen</sub> or S<sub>msy</sub> abundance target of < 1,000, the abundance target was set to a minimum of 1,000 to ensure that COSEWIC Criterion D is exceeded.

Table 4 - The equation and parameter values to estimate S-R benchmarks based on watershed area developed in Parken et al 2006. These parameters estimates are updated values compared to the initial report and represent the most up-to-date values for the habitat model.

Equation		
$\ln(\hat{y}) = \ln(\hat{a}) + (\hat{b} * \ln(x)) + (\hat{\sigma}^2/2)$		
Parameters		
	Stream-type S <sub>msy</sub>	Stream-type S <sub>rep</sub>
$\hat{y}$	S <sub>msy</sub>	S <sub>rep</sub>
$\ln(\hat{a})$	3.06	3.99
$\hat{b}$	0.686	0.691
$x$	Accessible watershed area	Accessible watershed area
$\hat{\sigma}^2$	0.260	0.208

Table 5 - Accessible watershed areas, listed as major tributary names, for the stream-type DUs in the RPA.

DU	Watershed(s)	Area (km <sup>2</sup> )
DU4	Upper Pitt	342
DU5	Big Silver	495
DU7	Nahatlatch	293
DU8	Portage	538
DU9	Bridge (below Terzaghi Dam)	416

DU	Watershed(s)	Area (km <sup>2</sup> )
DU9	Endako	814
DU9	Chilako	2,233
DU9	Horsefly	576
DU9	Westroad, Baker, Naver, Narcosli, Cottonwood	10,028
DU9	Chilcotin (to confluence with Chilko)	2,331
DU10	Chilko (to confluence with Taseko)	2,458
DU10	Quesnel (to confluence with Fraser)	4,648
DU10	Nechako (Stuart, Stellako, Nechako below Kenney Dam)	18,016
DU11	Upper Fraser (to confluence with and including Salmon River)	14,451
DU14	Bessette	276
DU16	Upper North Thompson (all tributaries mainstem upstream of Avola)	5,167
DU17	Lower North Thompson (to confluence with South Thompson)	6,849

### 3. DESCRIPTION OF MODEL USED FOR FORWARD PROJECTIONS

To address many of the remaining elements of the RPA, forward projections were used to predict the trajectory of the populations under recent and future conditions, and to inform the allowable harm assessment. The Chinook Projection Model was used to accomplish these tasks. The model is briefly described below with supplemental information available in Appendix B.

#### 3.1. CHINOOK PROJECTION MODEL

To assess the expected trajectory of these DUs under recent and changing future conditions, the Chinook Projection Model recently developed by co-authors Kendra Holt and Brooke Davis was used (Appendix B).

The Chinook Projection Model simulates trajectories of future abundance for individual stocks under specified scenarios about future exploitation rates and biological processes. Stochastic projections can be parameterized to represent the effects of uncertainty in survival, maturity, productivity and exploitation rates. The model includes scalars that can be applied to both fishing rates and productivity, easily allowing for the examination of population trajectories under varying future conditions as required by Element 15 (Section 9).

The model is based on Chinook population dynamics and set up so that catch is taken by one or more fisheries, operating as either pre-terminal or terminal fisheries. An overview of annual fishery and maturation timing in relation to abundance and catch for an ocean-type stock is shown in Figure 3. Stream-type stocks are modelled in a similar manner, but with an offset parameter added to account for the additional year spent in freshwater as juveniles (Appendix B). Pre-terminal and terminal fisheries differ in their timing relative to maturation. Pre-terminal fisheries occur prior to annual maturation while terminal fisheries occur after maturation and only intercept fish that are on-route to spawning grounds. As a result, pre-terminal fisheries are implemented using exploitation rates (ERs) applied to total at-sea abundance at the start of a year while terminal fisheries are implemented using harvest rates (HRs) applied to the mature abundance returning to spawn in each year. Escapement after terminal fisheries is then used to predict the ocean age-1 recruitment to the next year based on a Ricker S-R model that is adjusted using a spawner equivalence factor. The Ricker model predicts total adult recruitment from age  $\geq 3$  escapement. The spawner equivalence factor, which is the product of survival and maturation rate schedules, is then used to back-calculate the ocean age-1 abundance that would be required to produce the predicted adult recruitment. This approach to modelling

ocean-age-1 abundance is the same as that used by the Chinook Technical Committee’s (CTC) Chinook Model (Pacific Salmon Commission 2019a, 2019b). The abundance of fish remaining in the ocean after maturation (i.e., those that were not captured in pre-terminal fisheries and did not return to spawn), have an age-specific over-winter survival rate applied before advancing into the next year and age class (Figure 3).

Appendix B provides a detailed explanation and equations of the Chinook Projection Model.

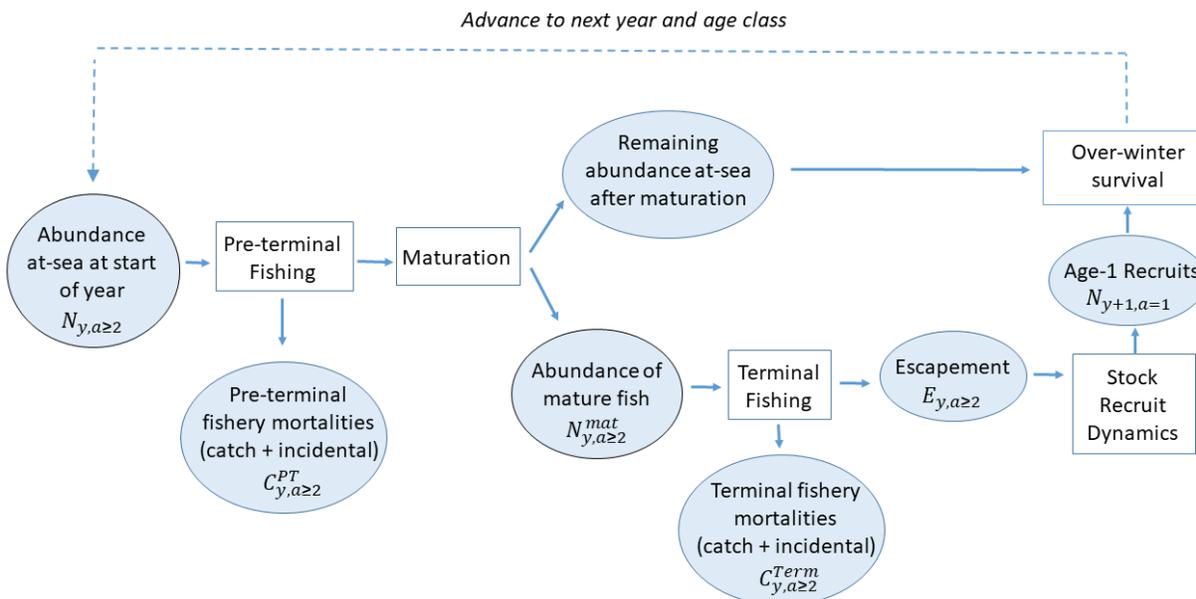


Figure 3 - An overview of annual fishery and maturation timing in relation to abundance and catch in the model for an ocean-type population such as DU2 (LFR-Harrison). Note that while fish age  $\geq 2$  are included in escapement, only escaped fish age  $\geq 3$  contribute to escapement in the stock-recruit function.

#### 4. DECISIONS ON FORWARD PROJECTIONS

For 10 of the 11 DUs in this RPA (all DUs except DU2 (LFR-Harrison)), it was determined that reliable quantitative forward projections could not be completed due to extensive data gaps. Escapement time series of relative abundance are the only recent information available from each of these DUs for projections. For DUs 4 (LFR-Upper Pitt), 5 (LFR-Summer), 7 (MFR-Nahatlatch), 14 (STh-Bessette), and 16 (NTh-Spring) the time series of escapement is particularly unreliable, and the possibility of forward projecting these DUs was rejected due to data limitations (Appendix C).

Significant efforts were made to produce representative inputs from proxy sources of information for the remaining data-limited DUs (8, 9, 10, 11, 17). Fishing rates and cohort sizes were estimated from the outputs of the CTC Chinook Model, maturation rates were estimated from 1986–2002 CWT data from Dome Creek (a system within DU11 (UFR-Spring)), and the habitat model (Parken et al. 2006) was used to estimate S-R parameters and abundance benchmarks. However, there are significant uncertainties associated with these estimates, in particular the harvest rates and S-R parameters. In addition to these uncertainties, there were also concerns over the representativeness of the cumulative assumptions associated with using multiple proxy sources.

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One of the major assumptions that prevented attempts of forward projections for the data-limited DUs (8, 9, 10, 11, 17) was that the harvest rates estimated from the CTC Chinook Model needed to be adjusted due to unrealistic estimates of harvest rates-at-age (Appendix D - Section 1.5). This included redistributing the escapement at-age for the DUs in the Summer 52 MUs and using the relationship between harvest rates-at-age from DU2 (LFR-Harrison) to adjust age-6 and age-3 harvest rates for the Spring and Summer 52 stocks. It also became evident that the CTC Chinook Model significantly underestimates harvest rates and poorly represents the inter-annual variation in harvest rates when compared to the Dome Creek CWT harvest rates from 1986–2002. Hence, there is variable and unknown uncertainty in the accuracy of the estimated harvest rates-at-age for the Spring and Summer 52 stocks. This situation arises from the poor data quality described above, and the use of proxy information.

The habitat model was initially used to produce estimates of S-R parameters for DUs 8 (MFR-Portage), 9 (MFR-Spring), 10 (MFR-Summer), 11 (UFR-Spring), and 17 (NTh-Summer). However, these estimates are unlikely to represent current productivity for these DUs given the significant declines in abundance observed, even after reductions in fisheries. The habitat model uses S-R and watershed data mainly from the mid-1970s to the mid-1990s (Parken et al. 2006). For Chinook populations that are facing multiple threats causing freshwater habitat degradation and reductions in marine survival, such as these DUs, the relationship between productive capacity and watershed area may have shifted over time. Additionally, the habitat model provides an estimate of average long-term productivity, which is unlikely to represent current productivity for these DUs, as they are suspected to be experiencing a declining or reduced productivity. Attempts were made to reduce the estimates based on the trend in productivity observed at DU2 (LFR-Harrison), however, this may underestimate the decline as the stream-type DUs have experienced steeper declines in abundance compared to DU2 (LFR-Harrison). Ultimately the S-R relationships for these DUs remain unknown.

Due to the considerable uncertainty in the accuracy of the input values for the projection model, and the inability to identify plausible S-R parameters, allowable harm and future trends were assessed qualitatively. The qualitative assessment was based on the threats assessment, results of the forward projections for DU2 (LFR-Harrison), and projected future climatic conditions. Details of the steps taken to estimate model parameters, associated uncertainty and example projections with varying productivities are provided in Appendix D.

For the reasons stated above and in Appendices 3 and 4, forward projections were only completed for DU2 (LFR-Harrison).

## **5. DATA SOURCES AND PARAMETER ESTIMATION**

This section only describes data sources and parameter estimation for DU2 (LFR-Harrison) that are used in the subsequent sections of the RPA. The main data sources for DU2 are the Harrison Mark-Recapture data from Fraser Stock Assessment, CWT data, CTC Chinook Model Outputs and a recent S-R analysis (Vélez-Espino et al. in press). Tables with the complete input values and sources for the Chinook Projection Model are available in Appendix E. Discussion of the S-R parameters and the fishing rates used are in the sections below.

### **5.1. STOCK RECRUIT ANALYSIS**

There are concerns about historic changes in productivity for DU2 (LFR-Harrison) that need to be considered when parameterizing projections (Appendix F). These concerns led to a model selection analysis of alternative S-R model formulations (Appendix G). Three models were considered: a traditional Ricker model, a Ricker model with temporal autocorrelation in recruitment residuals and a state-space model with time-varying productivity. The outcome of

the analysis suggested that the time-varying productivity model was the most appropriate for DU2 (LFR-Harrison) based on preferred residual patterns (Appendix G). This model was additionally supported by evidence of productivity changes among several biological indicators for this DU (Appendix F). Unlike the traditional Ricker model, the time-varying productivity model allows the productivity parameter to vary among years, which allows us to consider changes in productivity over time and isolate recent productivity.

The DU2 (LFR-Harrison) S-R analyses used to parameterize projections were based on analyses originally conducted by Vélez-Espino et al. (in press.), which were re-run to include a log-normal bias correction factor. The bias correction factor was added to be consistent with the Chinook Projection Model which uses a correction factor during projections. The analysis was conducted using S-R data of DU2 (LFR-Harrison) Chinook Salmon from brood years 1984 to 2013. A Bayesian state-space model with time-varying productivity (implemented using a Recursive Bayes estimation procedure) was fit to the data using R (R Core Team 2019) and TMB software (Kristensen et al. 2016) via the `tmbstan` package for R.

The formulation of the Bayesian state-space model for time-varying productivity is as follows:

$$R_t = S_t * \alpha_t * e^{(-b \cdot S_t + \epsilon_t - \sigma^2/2)} \quad \text{Eq. 1}$$

$$\log \frac{R_t}{S_t} = a_t - b * S_t + \epsilon_t - \sigma^2/2 \quad \text{Eq. 2}$$

$$\alpha = e^{a_t} \quad \text{Eq. 3}$$

$$S_{max} = \frac{1}{b} \quad \text{Eq. 4}$$

$$\epsilon_t \sim N(0, \sigma) \quad \text{Eq. 5}$$

Where,  $R_t$  is the abundance of adult recruits in year  $t$ ,  $S_t$  is the number of spawners that generated those recruits, and  $a_t$  is the annual productivity of recruits in year  $t$ , the  $b$  parameter represents the strength of density-dependence per unit spawning biomass, and  $\epsilon_t$  represents normally distributed residuals around the spawner-recruit curve with a standard deviation of  $\sigma$ . The model assumes that  $a_t$  changes over time following a simple random walk, with standard deviation  $\sigma_a$ , given by a recursive Bayes function as follows:

$$\begin{cases} a_t = a_0 + \gamma_0 & t = 0 \\ a_t = a_{t-1} + \gamma_t & t > 0 \end{cases} \quad \text{Eq. 6}$$

$$\gamma_t \sim N(0, \sigma_a) \quad \text{Eq. 7}$$

The model's observation ( $\sigma$ ) and process ( $\sigma_a$ ) standard errors were partitioned as follows:

$$\sigma = \sqrt{\rho} * \sigma_{total} \quad \text{Eq. 8}$$

$$\sigma_a = \sqrt{1 - \rho} * \sigma_{total} \quad \text{Eq. 9}$$

where  $\rho$  is the proportion of total variance associated with observation error, and  $\sigma_{total}$  is the total standard deviation. Priors for estimated parameters or derived quantities were not explicitly specified except for the  $\rho$  parameter. For the other parameters, bounds were placed, which is comparable to using uniform priors. All Bayesian posteriors were based on 100,000 iterations and three MCMC chains. A burn-in period of 50,000 iterations was used and convergence was

evaluated with visual inspection of standard diagnostic plots available for the R package `tmbstan`.

In order to evaluate the impact of the prior on the parameter estimates, the model was fit with three different priors on  $\rho$ . The value of  $\rho$  was centered around 0.5 (base case), 0.3, and 0.7 using a beta distribution with different shape parameters. The base case of 0.5, (Beta (3,3)) was selected due to a lack of information available about the distribution of error and poor convergence with an uninformative prior. The base case model fit and the trajectory of  $\alpha$  parameter are available in Figure 4 and Figure 5, respectively.

The time-varying productivity model fit indicates a declining trend in productivity over time, but the estimated magnitude of the changes in the  $\alpha$  parameter was sensitive to the specified prior distribution on  $\rho$ , the error allocation parameter. If the prior allowed greater allocation to observation error, the changes in productivity were smoother. However, the overall declining trend was consistent regardless of choice of prior and the base case ( $\rho$  centered around 0.5) was used for the forward projections.

Given the declining trend in productivity, the average productivity from the most recent generation available was used to represent the state of current productivity for forward projections. This average is denoted  $\bar{\alpha}_t$ , where  $t =$  brood years 2010 – 2013. For each simulated projection with the Chinook Projection Model, stock recruitment parameters  $\bar{\alpha}_{t=2010-2013}$ ,  $b$ ,  $\sigma_{total}$  and  $\rho$  were sampled from the estimated joint posterior from the Bayesian model fit. The median, 5% and 95% quantiles, for the marginal posterior distributions for each of these parameters are provided in Table 6.

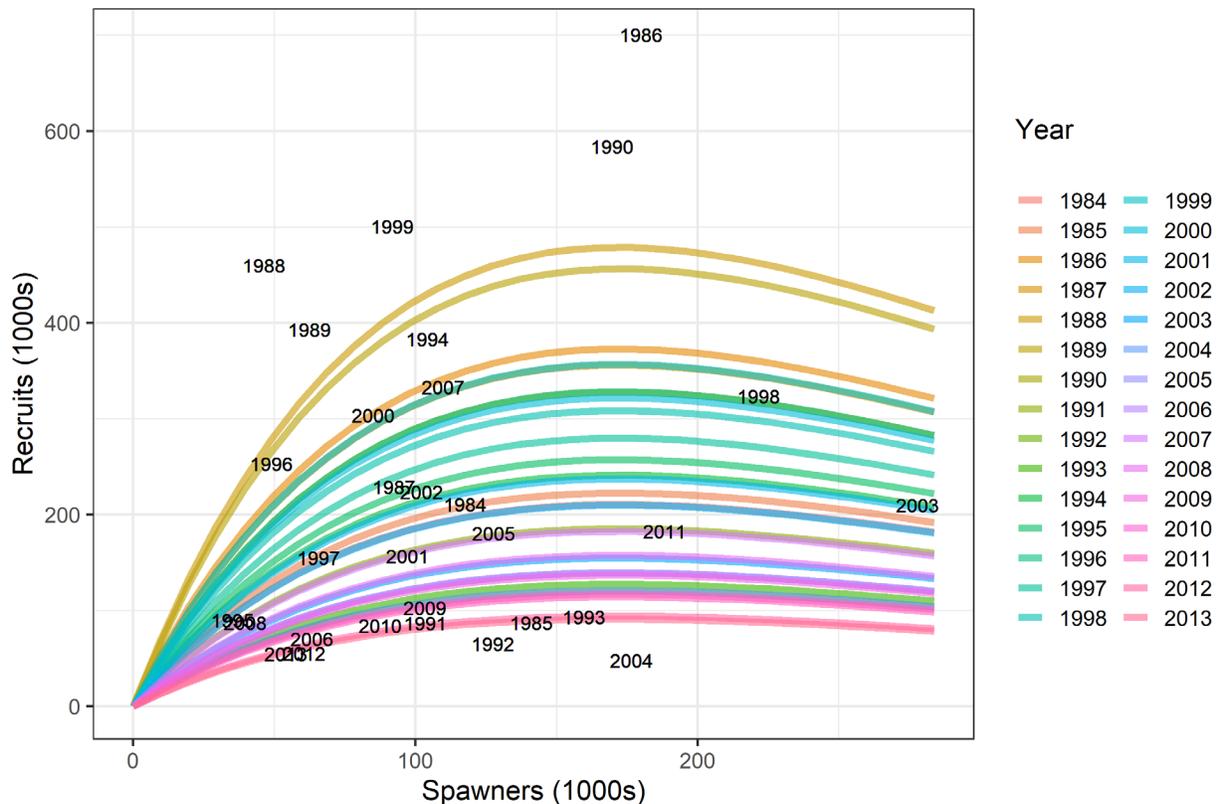


Figure 4 - The Bayesian state-space model with time-varying productivity for DU2 (LFR-Harrison). Colours indicate predicted values using year-specific  $\alpha$  parameters. Individual observations are represented by their associated years in text on the graph.

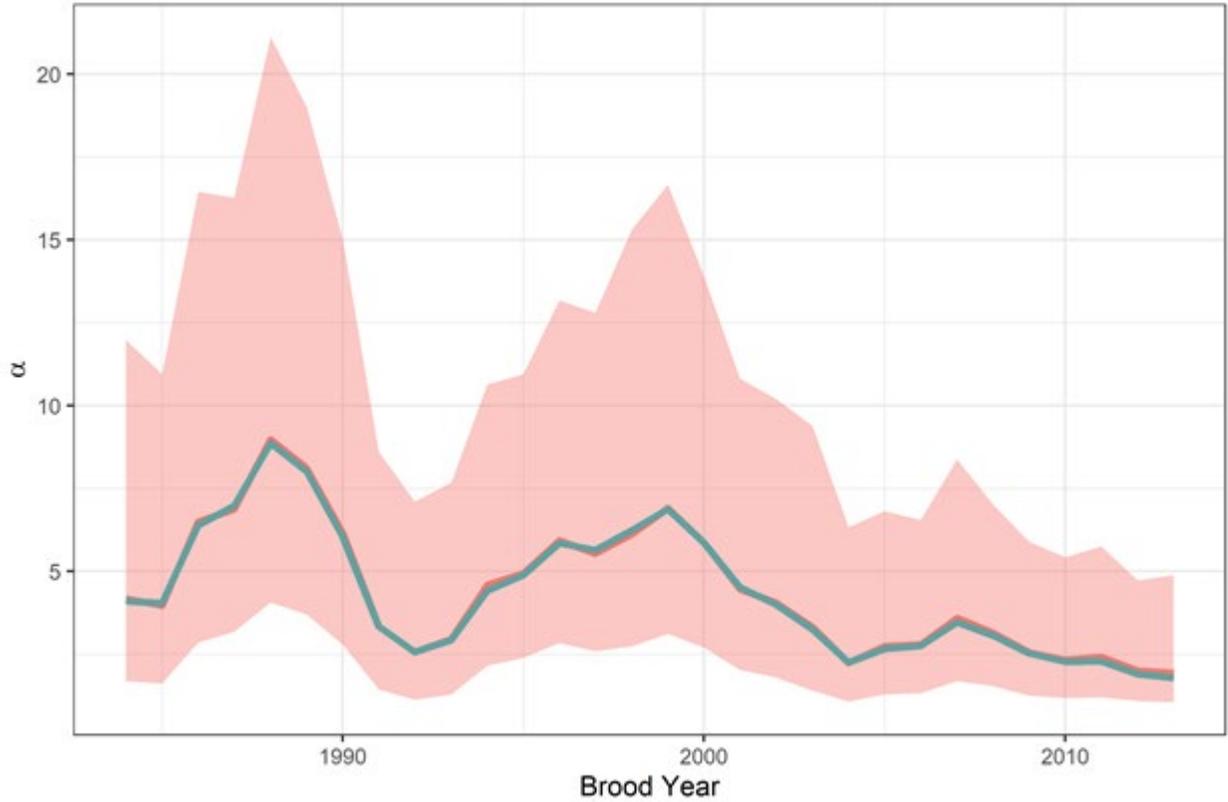


Figure 5 - Time series of  $\alpha$  estimated by the time-varying Ricker model, with the corresponding 95% credible interval. Lines represent the posterior median (red) and MLE estimates (blue). Bayesian 95% credible intervals are represented by the shaded area in red.

Table 6 - Posterior medians, as well as the 95<sup>th</sup> and 5<sup>th</sup> posterior quantiles, for the time-varying productivity base case scenario for the Chinook Project Model.

Quantile	$\alpha_{t=2010-2013}$	$b$	$S_{max}$	$\rho$	$\sigma_{total}$
Upper 95 Quantile	4.44	1.05 E-05	423287	0.91	1.00
Median	2.17	5.83 E-06	171451	0.66	0.75
Lower 95 Quantile	1.35	2.36 E-06	95553	0.33	0.59

## 5.2. FISHING RELATED MORTALITY

Harvest rates were calculated using coded wired tag (CWT) data from the CTC Exploitation Rate Analysis provided by Gayle Brown (CTC Co-Chair, July 2019). These data are produced in the total mortality output files, and are reported annually by the CTC (Pacific Salmon Commission 2019a, 2019b). Pre-terminal and terminal harvest rates-at-age are required for the Chinook Projection Model. Pre-terminal fisheries occur in the ocean, catching both immature and mature fish, whereas terminal fisheries harvest only mature individuals returning to the spawning grounds, and can include ocean fisheries near river mouths during the months when mature individuals return. These were calculated for each year ( $y$ ) and age ( $a$ ) using the following equations:

$$Pre\text{-Terminal Harvest Rate}_{y,a} = Catch_{y,a}/Cohort\ Size_{y,a}$$

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$$\text{Terminal Harvest Rate}_{y,a} = \text{Terminal Catch}_{y,a} / \text{Terminal Run}_{y,a}$$

where cohort size is the estimated ocean abundance of a cohort-at-age after natural mortality, which is applied when individuals in a fishery transition from one age to the next (e.g. October 1 for ocean troll fisheries). The terminal run is the number of mature individuals at each age returning to spawn. When calculating the rates for the Chinook Projection Model, the year ( $y$ ) refers to the return (run) year. It is important to note that harvest rates are not the same as the Calendar Year Exploitation Rates (CYER) and Mortality Distributions because the denominator is much different, and the CYER values are adjusted to adult equivalents whereas the cohort-based harvest rates are unadjusted (i.e. in units of nominal fish).

The period selected to represent harvest rates for DU2 (LFR-Harrison) is 2009-2015, as it represents relatively recent harvest rates, is a period with high quality CWT data (Pacific Salmon Commission 2015), and because the most recent PST Agreement identifies that Chinook Salmon Individual Stock Based Management Fisheries should be managed in reference to the 2009–2015 period.

Harvest rates for the Chinook Projection Model were calculated separately for American and Canadian fisheries (Table 7), so that when varying conditions for the forward projections (Section 9), Canadian harvest rates could be decreased or increased, while holding American harvest rates constant. This was done because Canada does not control American harvest rates, although there are harvest rate limit obligations for certain Canadian stocks in the Individual Stock Based Management fisheries specified in the 2019 PST Agreement, and is consistent with the approach used to assess the threat of fishing in the threats process in Part 1 of the RPA (DFO 2020a).

It is important to note that the pre-terminal and terminal harvest rates-at-age presented in the table below, include total fishing related mortality (i.e. both landed and incidental mortality; Pacific Salmon Commission 2019a). Landed catch is estimated through CWT recoveries, while incidental mortality (IM) is estimated from sampling data and/or internal algorithms (e.g., size-at-age vulnerability algorithms and gear-specific mortality rates) (Pacific Salmon Commission 2019b). IM rates are estimated for the four following categories (as defined in Pacific Salmon Commission 2019b):

1. Shakers: Chinook Salmon below the legal size limit that are encountered, brought to the boat, and released during a Chinook salmon retention fishery.
2. Sublegal Chinook Non-Retention (CNR): Chinook Salmon below the legal size limit that are encountered, brought to the boat, and released during a Chinook Salmon non-retention fishery. The mortality rate per encounter applied to sublegal CNR is the same as applied to shakers.
3. Legal CNR: Chinook Salmon above the legal size limit that are encountered, brought to the boat, and released during a Chinook Salmon non-retention fishery and some retention fisheries that have empirical estimates of legal releases.
4. Drop-off: Chinook Salmon above or below the legal size limit that encounter the fishing gear, but are lost from the gear (for any reason, including predation mortality for fish removed or lethally maimed by predators (Pacific Salmon Commission 1997)) before they reach the boat during either retention or non-retention fisheries. Drop-off mortality is assumed the same for legal and sublegal fish, but varies by gear type and fishery location.

The only IM rate, of the suggested categories in Patterson et al. (2017), that is not quantified is avoidance mortality. No attempts were made to estimate this mortality rate due to an absence of estimates for Chinook Salmon and time constraints. The methods used to calculate IM in the

CTC Exploitation Rate Analysis are currently under review by the CTC to identify possible improvements, including a better representation of gear-specific rates based on recent research.

For additional context, the brood year exploitation rate time series for Canadian Pre-Terminal, Canadian Fraser River and American fisheries are provided in Table 8. These exploitation rates are calculated for each year ( $y$ ) and fishery ( $f$ ) as follows:

$$Exploitation\ Rate_f = Catch_{f,y} / (Catch_y + Escapement_{y,ages\ 3:5})$$

Where  $Catch_{f,y}$  is calculated by CWT adult equivalent catch at ages 2 to 5 for each of the three fisheries, scaled up by the ratio of total escapement to CWT escapement for that age and brood year. These catch values represent total fishing related mortality, as described above, and they are in units of adult equivalence. Escapement is only calculated across ages 3 to 5, consistent with the methods used to calculate exploitation rates for S-R analyses. Canadian Pre-Terminal exploitation rate includes all Canadian ocean harvest for DU2 (LFR-Harrison), while Canadian Fraser River is the in-river harvest, and the American exploitation rate includes all pre-terminal and terminal American harvest. The exploitation rates in Table 8 were not used in this report, but have been included to provide context of recent harvest conditions in a format more familiar to most readers. For a discussion on the trends in exploitation rates and the threat risk associated with fishing please refer to Part 1 (DFO 2020a).

Table 7 - The average pre-terminal and terminal harvest rates-at-age for Canadian (CA) and American (US) fisheries from 2009–2015 used in the Chinook Projection Model.

Period	Country	Pre-Terminal				Terminal			
		Age 2	Age 3	Age 4	Age 5	Age 2	Age 3	Age 4	Age 5
2009–2015	US	1.1%	5.6%	6.3%	1.0%	2.9%	6.5%	3.0%	0.8%
2009–2015	CA	1.7%	7.4%	13.0%	5.1%	0.0%	0.4%	0.0%	0.6%

Table 8 - Brood year total exploitation rates excluding age 2 spawner escapement that aligns with data used the Stock Recruit Analysis. In addition to total exploitation rate, the individual rates for Canadian pre-terminal, Canadian Fraser River and American (US) harvest have been provided.

Brood Year	Total	Canadian Pre-Terminal	Canadian Fraser River	All US
1981	84%	69%	5%	10%
1982	91%	73%	6%	13%
1983	76%	64%	4%	8%
1984	78%	61%	4%	13%
1985	64%	49%	4%	10%
1986	71%	54%	2%	15%
1987	77%	47%	2%	28%
1988	73%	50%	2%	21%
1989	72%	50%	2%	21%
1990	75%	50%	1%	24%
1991	77%	73%	2%	2%
1992	47%	32%	4%	10%
1993	46%	26%	2%	18%

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<b>Brood Year</b>	<b>Total</b>	<b>Canadian Pre-Terminal</b>	<b>Canadian Fraser River</b>	<b>All US</b>
1994	36%	21%	1%	14%
1995	17%	3%	2%	12%
1996	51%	30%	0%	20%
1997	46%	31%	0%	15%
1998	60%	23%	1%	37%
1999	53%	36%	2%	15%
2000	47%	22%	4%	21%
2001	50%	32%	4%	15%
2002	70%	19%	3%	48%
2003	56%	40%	0%	17%
2004 <sup>1</sup>	-	-	-	-
2005	32%	24%	3%	6%
2006	56%	41%	4%	10%
2007	26%	15%	2%	8%
2008	39%	23%	3%	13%
2009	41%	19%	1%	20%
2010	34%	18%	5%	12%
2011	27%	16%	3%	9%
2012	34%	17%	4%	12%
2013	28%	19%	2%	6%
2014	49%	35%	4%	10%
2015	36%	17%	4%	15%

<sup>1</sup> For brood year 2004, no CWTs were applied to released Harrison hatchery smolts, hence no exploitation rates were measured for that brood year.

## **6. CAVEATS AND CONDITIONS**

### **6.1. ASSUMPTIONS AROUND THE APPROACH TO ABUNDANCE BENCHMARKS AND PROJECTIONS**

In assessing the probability of recovery (Element 13 and Element 15) for DU2 (LFR-Harrison), the choice was made to use parameters from a time-varying productivity model to project forward, while comparing future trajectories to already existing, fixed, benchmarks as recovery targets. This means that the recovery targets are not directly linked to either the assessment or projection model and were estimated independently and treated as fixed points without uncertainty. This choice is to allow consistency between the recovery target and the previously established Pacific Salmon Treaty escapement goal. It is also consistent with other Pacific salmon RPAs carried out in recent years, where uncertainty around recovery targets was not incorporated into forward simulations, and recovery targets were determined independently from the forward projection model, either from WSP benchmarks, or other applicable targets used for a given DU (e.g., DFO 2019c, 2020b; 2020c, Arbeider et al. 2020). This results in recovery targets that remain constant as underlying population dynamics change and may not include the full time-series of data. For example, under the scenario of declining productivity, this avoids the downward trend in  $S_{msy}$  targets that would accompany a decline in productivity.

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Possible shortcomings to using different models for determining targets and doing projections are that the recovery targets are not directly estimated from the model used in the projection and so do not incorporate uncertainty derived from those dynamics, the recovery targets may be different from ones estimated from the projection model fit to the same data, or may be based on an earlier time series of data then used to fit the projection model, and they may not reflect the changing underlying conditions when parameters are time-varying. To avoid the underestimation of uncertainty associated with using fixed benchmark estimates, an alternative approach is to define targets as estimated quantities that are themselves uncertain and functions of estimated model parameters fit to the full time series of data, including forward projections. For example, where  $0.8 S_{msy}$  is set as the rebuilding benchmark, the value can be treated as an uncertain random variable in the estimation and in the population projections. The probability of exceeding,  $0.8 S_{msy}$  can be computed without  $0.8 S_{msy}$  being treated as a fixed value and the uncertainty in the estimate of  $S_{msy}$  can be directly accounted for in the projections. This approach has been considered in the literature (e.g., Holt and Michielsens 2020) and applied in other RPAs and evaluations of management options for both short-lived semelparous and long-lived Canadian fish stocks (McAllister and Duplisea 2011; Schweigert et al. 2012; Stanley et al. 2012; Yamanaka et al. 2012). Although the approach used in this RPA to determine abundance targets may not reflect the true nature of the certainty in forward projections, it reflects the reality of how these benchmarks are used as fixed points to infer conservation status.

An alternative approach to address the second possible shortcoming, that the benchmarks do not reflect changing underlying conditions, is to re-estimate the benchmarks in each subsequent step of the forward projection. In scenarios where productivity is non-stationary, and there is uncertainty over future trends in productivity, this alternative approach projects associated changes in the benchmarks for each alternative scenario considered for future changes in productivity (e.g., Licandeo et al. 2020). Definitions of the benchmarks applied are thus kept internally consistent within future simulations and the probabilities and risks associated with the benchmarks can be computed in an internally consistent manner. While this alternative approach has been applied in evaluations of management options for non-salmonid Canadian fish stocks (e.g., Licandeo et al. 2020), and results obtained could be considerably different from the current approach of applying fixed long-term average values for benchmarks. Recent literature (e.g., Berger 2019; Holt and Michielsens 2020; O'Leary et al. 2020) has recommended further simulation-based research into the efficacy of these alternative approaches when applied in different contexts. In particular, the performance of time-varying benchmarks when stocks are depleted and productivity is low, may result in management actions that are not precautionary (Berger 2019). For Pacific salmon, instead of changing benchmarks in response to annual declines in productivity (e.g., reducing  $S_{MSY}$ ), poor productivity can in some cases be mitigated by improvements to freshwater habitat or other freshwater management levers that are not be available to marine fish species. In the absence of such simulation evaluation on the impacts of time-varying recovery and survival targets for Pacific salmon species at risk, the analyses presented here have relied on the historically derived (long-term) survival and recovery targets (Duplisea and Cadigan 2012).

## **6.2. TIME-VARYING MODEL ASSUMPTIONS**

A critical assumption of this analysis is that the Ricker recruitment model and the data used to generate the S-R relationship accurately represents the underlying population dynamics of DU2 (LFR-Harrison), and will continue to do so over the projection period. Another consideration is that S-R data are estimates derived from spawner abundance, exploitation rates, and age information that each have associated uncertainty. This uncertainty is excluded from the S-R

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model analysis which results in an inherent underestimation of the uncertainty in the model output.

The time-varying productivity model results for the S-R parameters for DU2 (LFR-Harrison) have an additional source of uncertainty. The parameter estimates are sensitive to the prior used to inform the proportion of total variance associated with observation error ( $\rho$  parameter) vs. variance associated with changes in productivity over time. This sensitivity highlights that prior choice is important as it may influence the magnitude of the changes in the productivity parameter ( $\alpha$ ) over time. Sensitivity analyses (unpublished analyses C. Wor) show that alternative prior assumptions lead to more or less error being allocated to the process error ( $\sigma_a$ ), and consequently alter the magnitude of the changes in  $\alpha$  over time. However, the general trend of declining productivity does not appear to change under a variety of priors used for the  $\rho$  parameter. The use of more informative priors, and estimates of observation errors for spawners and recruitment might improve the estimates and confidence bounds around  $\alpha$  and the uncertainty in the population dynamics, but may also introduce bias. Any future assessments should include an updated S-R analysis, with any newly available data, which may help the model parse out variance between observation error, process error, and changes in productivity. A properly structured integrated assessment that accounts for known error in both escapement and recruitment estimates (rather than assuming they are the true values, as was done in this analysis) may improve convergence, rather than relying on informative priors based on untested assumptions.

A key assumption of the time-varying analysis described in Section 5.1 is that  $S_{\max}$  ( $1/b$ ; the spawner level above which density-dependence results in reduced recruitment with increasing spawner abundances) remains constant over time (Peterman et al. 2003). This assumption was carried forward for the projections under future productivity scenarios (decreasing or increasing alpha ( $\alpha$ ) while  $S_{\max}$  remains constant) as well as for the example projections in Appendix D. The validity of this assumption and alternative assumptions were discussed within the author group for the RPA. A notable consequence of assuming a constant  $S_{\max}$  is that it requires  $S_{\text{rep}}$  (spawners required to produce replacement) to vary over time. In the linearized Ricker curve, the alpha and beta parameters co-vary. As a result, if carrying capacity is stable and constant, a reduction in productivity, such as from a decrease in survival for one or more life stages (e.g. smolt-age 2 survival) or a decrease in fecundity, will lead to a reduction in beta and an increase in  $S_{\max}$  (Table 9; S-R curve shifts to the right and the spawner abundances at which overcompensation starts to occur increases). In comparison, if  $S_{\max}$  and beta are held constant and alpha is reduced, then  $S_{\text{rep}}$  will decrease in direct proportion to the decrease in alpha (beta remains constant because  $S_{\max}$  is constant, S-R shifts downwards and spawner abundances at which overcompensation occurs remains the same).

To examine whether holding  $S_{\max}$  constant is a valid procedure, DU2 (LFR-Harrison) S-R data were used to fit stationary (i.e., no time-varying productivity) Ricker curves in 20 year rolling windows (Figure 6), and a comparison between two time periods was completed: 1984-1999 and 1984-2013. The year 1999 was selected as the cut-off point for the period comparison, as productivity has declined steeply since that year, with only a slight recovery in the mid-2000s (Figure 7). In the rolling window analyses, there was some indication of both  $S_{\text{rep}}$  (point where curve meets the diagonal line) and  $S_{\max}$  changing (Figure 6; curves shifting down, and left to right). Comparing the early time-period to the entire time period, there was some evidence that  $S_{\max}$  had increased over time, while  $S_{\text{rep}}$  appears to have decreased (Table 10). In contrast,  $S_{\text{msy}}$  was virtually unchanged between the two time periods used. These estimates were not examined for any statistical differences.

Ultimately the assumption of holding  $S_{\max}$  constant was determined to be preferable over the alternative of holding  $S_{\text{rep}}$  constant for three reasons. First, given current productivity regimes

and the stability of DU2 (LFR-Harrison) spawning grounds, few biological mechanisms could be identified that would explain an increase in  $S_{max}$ . Second, holding a constant  $S_{rep}$  created scenarios that challenged biological explanation at low productivities, such as significant increase in recruits per spawner at spawner abundances above  $S_{rep}$ . Third, the assumption of constant  $S_{max}$  as productivity varies is consistent with previous salmon literature (Peterman et al. 2003; Dorner et al. 2008; Peterman and Dorner 2012; Malick et al. 2017). Although Table 10 shows some evidence for changes in both  $S_{max}$  and  $S_{rep}$  between the time periods, these could be a consequence of changes in underlying productivity and exploitation rates. Holt and Michielsens (2020) found that S-R parameters from standard Ricker models tend to be biased when there are concurrent trends in productivity and exploitation. The estimated strength of density-dependence in S-R curves may be an artifact of exploitation and productivity histories driving the distributions of underlying S-R data. Future analysis exploring the changes in both alpha ( $\alpha$ ) and beta ( $\beta$ ) over time would be beneficial for exploring this issue further, as there is evidence that estimated  $S_{max}$  values have shifted over time.

*Table 9 - The effects of changes in alpha on  $S_{max}$ , beta, and  $S_{rep}$  for scenarios where  $S_{rep}$  is constant and where  $S_{max}$  is constant.*

<b>Scenario</b>	<b><math>S_{max}</math></b>	<b>Alpha Prime</b>	<b>Alpha Reduction</b>	<b>Beta</b>	<b><math>S_{rep}</math></b>
<b><math>S_{rep}</math> constant</b>	155,948	1.51	0.0	6.41E-06	235,334
	159,131	1.48	0.02	6.28E-06	235,334
	162,446	1.45	0.04	6.16E-06	235,334
	165,902	1.42	0.06	6.03E-06	235,334
	169,509	1.39	0.08	5.90E-06	235,334
	173,276	1.36	0.10	5.77E-06	235,334
<b><math>S_{max}</math> constant</b>	155,948	1.51	0.0	6.41E-06	235,334
	155,948	1.48	0.02	6.41E-06	230,627
	155,948	1.45	0.04	6.41E-06	225,920
	155,948	1.42	0.06	6.41E-06	221,214
	155,948	1.39	0.08	6.41E-06	216,507
	155,948	1.36	0.10	6.41E-06	211,800

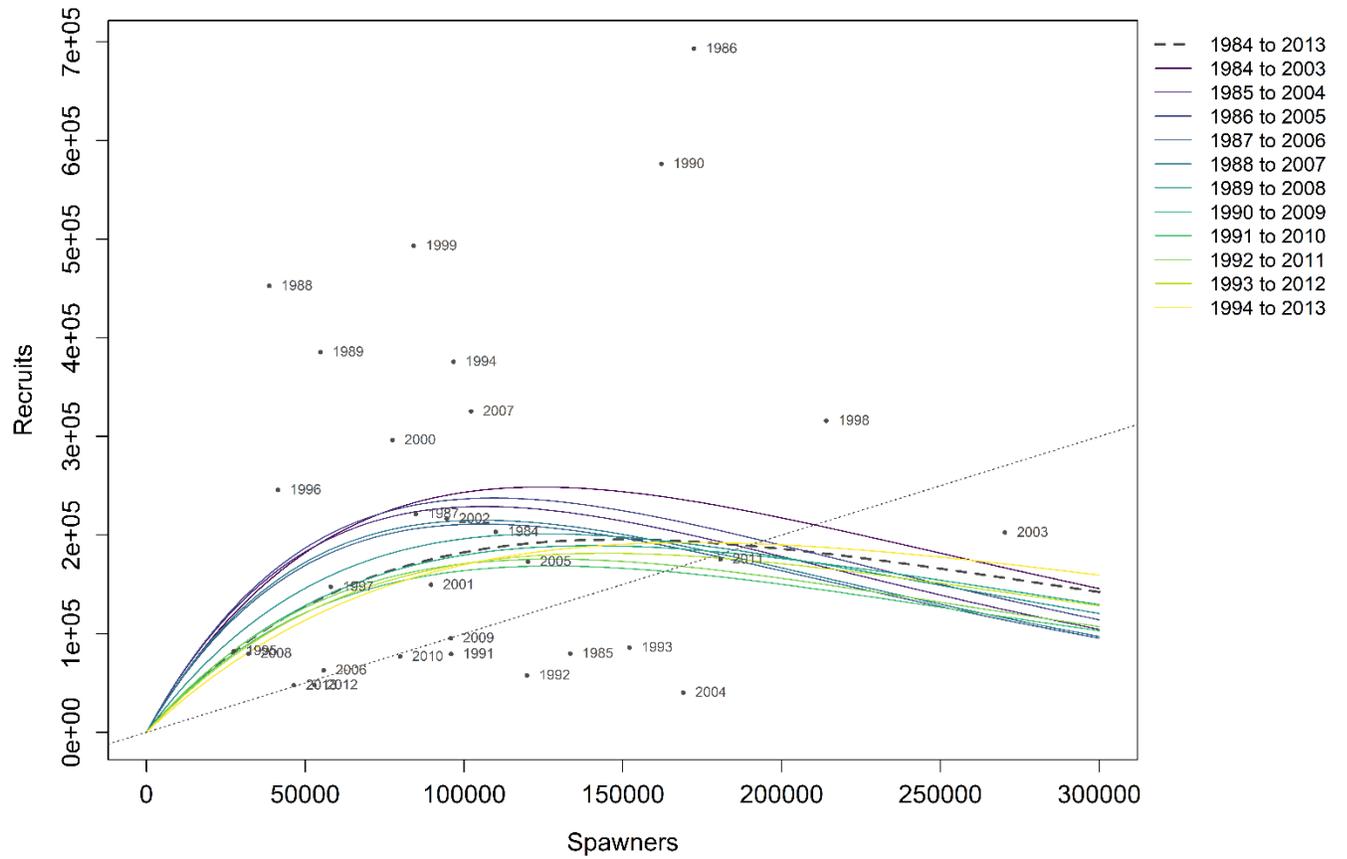


Figure 6 - Stationary Ricker S-R curves fit to rolling 20 year windows for DU2 (LFR-Harrison), with the fit to the full time series represented by the black dashed line and the 1:1 line in light grey.

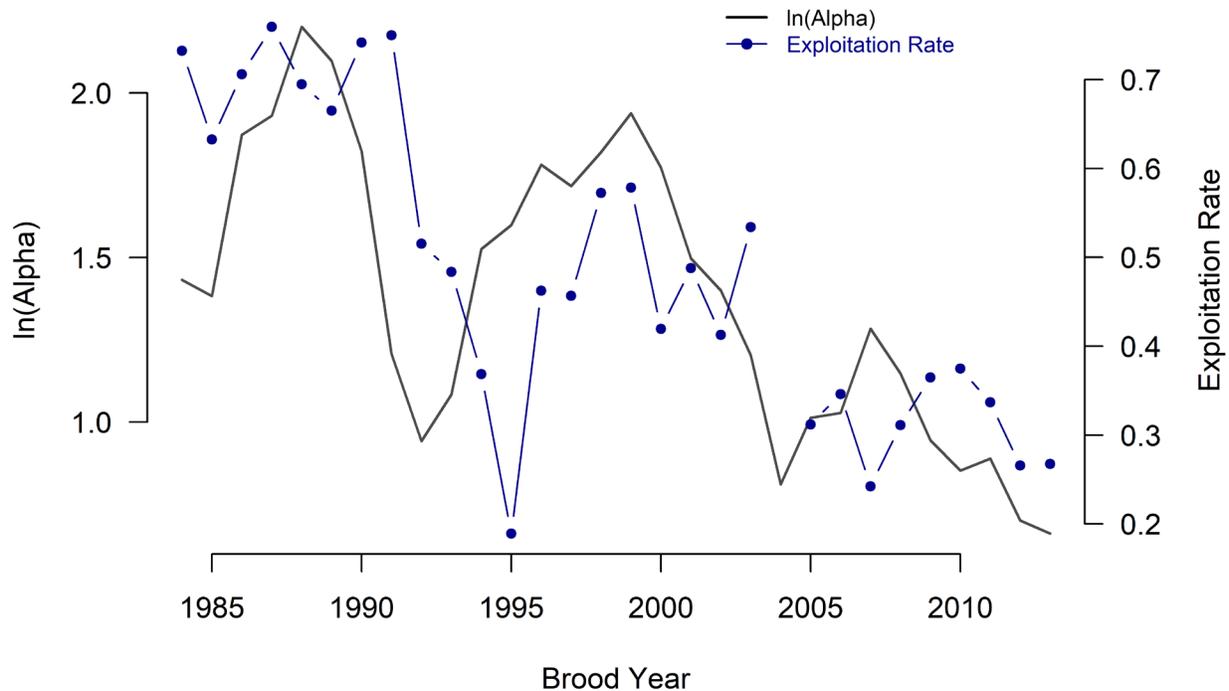


Figure 7 - The trend in the natural log of alpha (black) and exploitation rates for DU2 (LFR-Harrison) (blue). The alphas presented are the median values from the time-varying productivity analysis. Exploitation rates are based on adult equivalency adjusted total mortality in fisheries for ages 2 to 5 and spawner escapements for ages 3 to 5.

Table 10 - Summary of stationary Ricker S-R model attributes estimated for 1984-1999 and 1984-2013 for the DU2 (LFR-Harrison).

Stock Recruitment Attribute	Time Period	
	1984-1999	1984-2013
Ricker Alpha	4.7	3.8
Gamma	0.683	0.592
Median Smolt-Age2 survival	-4.64	-4.67
Alpha Prime <sup>B</sup>	1.5	1.1
Beta	6.41E-06	4.92E-06
S <sub>max</sub>	155,948	203,379
S <sub>rep</sub>	235,334	216,860
S <sub>msy</sub>	92,808	92,291

<sup>A</sup>  $\ln\left(\frac{R}{S}\right) = \alpha - \beta S + \gamma \ln(M)$

<sup>B</sup> Based on the median smolt-age2 survival for the time period and Ricker alpha

### 6.3. OTHER POPULATION MODELLING ASSUMPTIONS

Changes in natural mortality or marine survival were not explicitly modelled. Instead for this process we have opted to vary productivity as it captures aspects of both freshwater and marine conditions, which is important when modelling Pacific Salmon populations that occupy both environments. Future modelling exercises for DU2 (LFR-Harrison) could consider looking at systematic changes in marine survival. However, it is important to understand that future productivity, survival, and carrying-capacity may differ from historic and more recent time periods when extrapolating the results presented in the subsequent sections. Frequent changes

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in population dynamics and parameter estimates are being driven by climate change, continued human population growth, development, mitigation, and restoration, and the variable ways that ecosystems react to those changes. Analyses that could incorporate predictions about how climatic and anthropogenic stressors may affect survival and carrying capacity at a fine scale would be beneficial if there was sufficient data to support them. The analysis in Element 15 (Section 9) presents possible future scenarios based on different levels of productivity and adult mortality (examined through changes in harvest rates), but this assessment's uncertainty may be underestimated due to unknown future conditions and data quality issues. Accordingly, the results presented in Element 13 and 15 should be viewed as hypotheses of how DU2 (LFR-Harrison) will respond to a given productivity and harvest regime rather than a prediction of future escapements.

## **7. ELEMENT 13: POPULATION TRAJECTORIES AT RECENT PRODUCTIVITY AND MORTALITY**

*Element description: Project expected population trajectories over a scientifically reasonable time frame, and assess the ability of the trajectories over time to reach the potential recovery target(s), given current population dynamics parameters.*

### **7.1. DU2 (LFR-HARRISON)**

To assess the likely trajectory of DU2 (LFR-Harrison) under recent conditions (defined here as the average productivity of the last generation available), forward projections for the next 12 years (three generations) were completed using the previously described Chinook Projection Model. To represent recent conditions, it was assumed in the model that the average recent productivity (brood years 2010-2013) and recent (2009-2015) levels and patterns of Canadian and US fishery removals would continue unchanged. The model parametrization under recent conditions (defined above) is subsequently referred to as the *base case* in subsequent sections of this RPA.

The 2019 and 2020 measures to reduce Chinook Salmon harvest may have decreased Canadian harvest rates below the 2009-2015 period for those years, but estimates of the effects of these measures on harvest rates are not yet available. The effects of reduced harvest rates relative to the base case, such as those anticipated from the 2019 and 2020 reductions, can be explored through the different harvest rate scenarios considered in Element 15 (Section 9).

Details on model parameterization of the S-R relationship and base period harvest rates are described in Section 5, while all input parameters are documented in Appendix E. The forward projection model and S-R model included a log normal bias correction factor which is consistent with recent literature (Cox et al. 2011, 2019a, 2019b; Grandin and Forrest 2017; Ohlberger et al. 2019; Olmos et al. 2019; Forrest et al. 2020). The results of the projections are shown in Figure 8, relative to the observed escapement time series. The forward projections of abundance are log-normally distributed, with 50% of the simulated outcomes centered around the median (darker blue shaded region centered on the dark blue line; Figure 8).

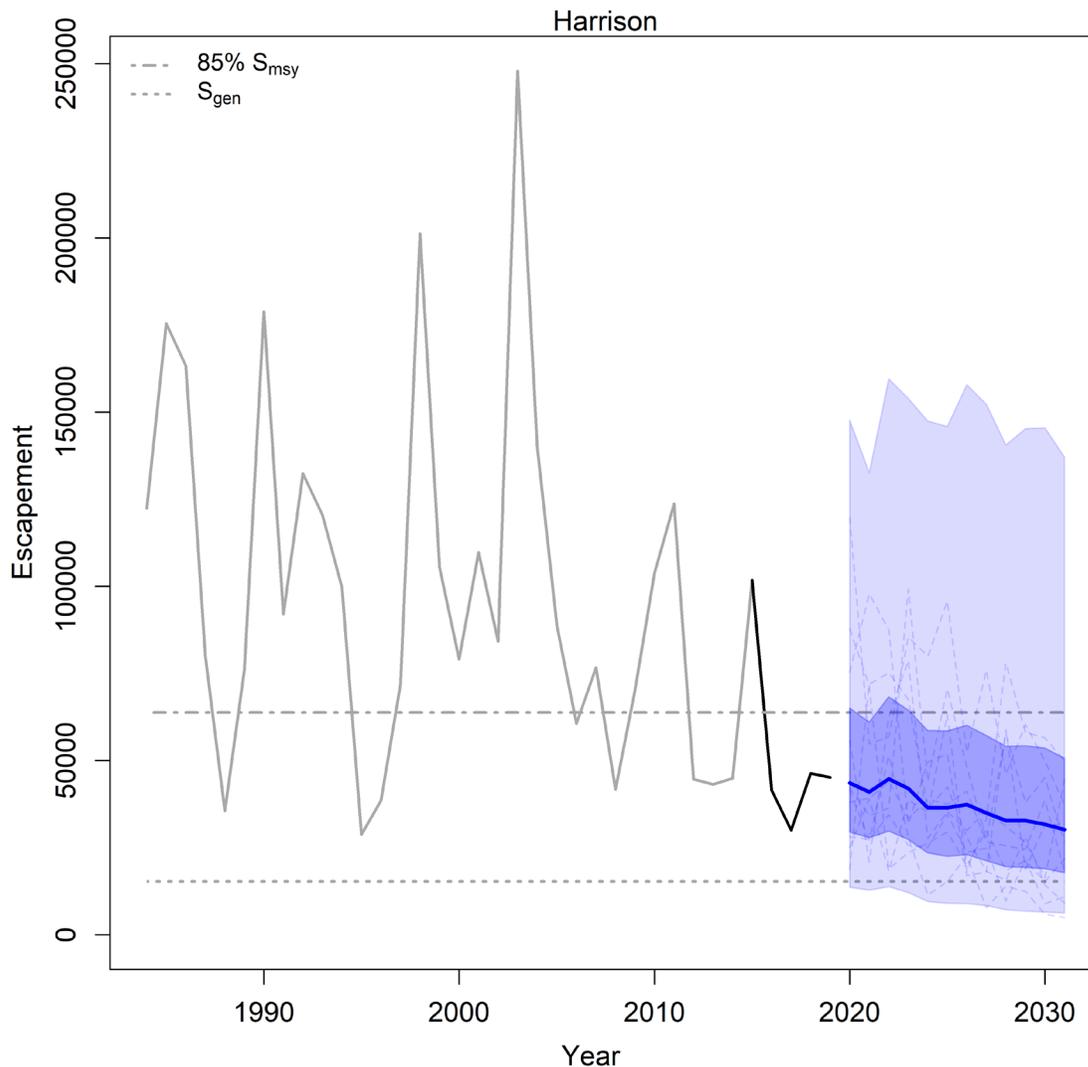


Figure 8 - Forward projection for DU2 (LFR-Harrison) from 2020 to 2031 assuming recent productivity estimates (2010-2013) and base period exploitation rates (2009-2015). The median value of abundance in each projected year is represented by the dark blue line. The area shaded light blue represents the outcome of 95% of the simulations (0.025 and 0.975 quantiles), while the dark blue polygon represents the outcome of 50% of the simulations (0.25 and 0.75 quantiles). The escapement time series from 1984 to 2019 is shown in light grey, with the black line representing the initialization period for the Chinook Projection Model. Dashed lines show outcomes of ten randomly selected individual simulations. Horizontal lines indicate abundance benchmarks for survival ( $S_{gen}$ ) and recovery targets (85%  $S_{msy}$ ).

The ability to achieve targets under recent conditions was assessed by evaluating the number of the 5,000 simulations that met the survival and recovery targets. Percent change in abundance was calculated using a natural log linear regression, consistent with Part 1 (DFO 2020a) and the COSEWIC assessment (COSEWIC 2018), over the three generations projected and compared to the goal of a less than 30% decline for both the survival and recovery targets. Abundance was averaged over the last generation (4 years) and was compared to the abundance targets for both the survival (15,313) and recovery (63,808) targets. Both the

percent change target and the abundance targets have to be met for the survival and recovery targets to be met. The International Panel of Climate Change (IPCC) risk/certainty categories are used to describe the likelihood of achieving targets (Table 11; Mastrandrea et al. 2010).

Table 11 - International Panel of Climate Change (IPCC) Risk Categories used to describe the likelihood of recovery in the RPA.

International Panel of Climate Change	
Risk Categories	
Extremely Unlikely	0%–1%
Very Unlikely	1%–10%
Unlikely	10%–33%
As Likely As Not	33%–66%
Likely	66%–90%
Very Likely	90%–99%
Certain	>99%

The results indicate that if recent productivity conditions (2010-2013) and base period (2009-2015) harvest rates persist over the next 12 years, DU2 (LFR-Harrison) is unlikely (16% of simulations) to recover to the recovery target, and is as likely as not (48% of simulations) to meet the survival target (Table 12). The ability to achieve the survival target is limited most by the percent change requirement, as 90% of the simulations exceeded the survival abundance target of 15,313. However, despite meeting the minimum abundance target 90% of the time, as described in Section 2, the survival target should be treated as a limit reference point, and both the percent change goal and survival abundance target should be exceeded most of the time to ensure population persistence. Reaching the recovery target was limited most by the recovery abundance goal, with only 17% of simulations exceeding a generational average abundance over 63,808 at the end of the projection period.

Table 12 - The percent of simulations that meet the survival and recovery targets, broken out by the separate generational average abundance and percent change requirements. Note that both the survival and recovery target are only considered met for a given simulation replicate if **both** the percent change target and the applicable abundance target are met in that replicate.

Target	Simulations Meeting Target
Percent Change Target (< 30% decline)	49%
Survival Abundance Target ( $\geq$ 15,313)	90%
<b>Survival Target (both metrics)</b>	<b>48%</b>
Recovery Abundance Target ( $\geq$ 63,808)	17%
<b>Recovery Target (both metrics)</b>	<b>16%</b>

## 7.2. QUALITATIVE ASSESSMENT FOR DATA-LIMITED STREAM-TYPE DUS

Due to data constraints described in Section 4 and Appendices 3 and 4, the remaining DUs are assessed qualitatively.

The recent trends in escapement and the number of threats documented in Part 1 provide no indication that trends observed in these DUs (Figure 1) are likely to improve in the short term (DFO 2020a). Declines in abundances have continued in recent years despite efforts to reduce

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harvest rates. It remains uncertain whether harvest reduction measures have effectively reduced harvest pressure on populations. A recent review determined that the measures put in place in 2012 aimed at reducing harvest rates on the Spring and Summer 5<sub>2</sub> MUs were likely successful; however, several uncertainties were noted that prevented a definitive conclusion (DFO 2019b; Dobson et al. 2020). Assuming harvest rates have in fact declined, the concurrent decrease in abundance indicates that these DUs are likely experiencing declining productivity. This aligns with the quantitative evidence that DU2 (LFR-Harrison) is experiencing declining productivity (based on time-varying productivity models). Furthermore, the stream-type DUs have seen more severe declines in abundance than DU2 (LFR-Harrison) (Figure 1; Table 13), which raises the possibility that the decline in their productivity may be steeper. Given the variability in escapement estimates over the time series there is uncertainty in the change in abundance over time (Table 13). For a detailed discussion of trends refer to Part 1. The data necessary to assess trends in productivity in the stream-type DUs is lacking, but the qualitative evidence supports the hypothesis of a decline. This is supported by other studies which have documented widespread declines in Chinook salmon productivity (Dorner et al 2018) and survival (Welch et al 2021).

The impacts of threats facing these DUs were discussed extensively in Part 1 (DFO 2020a), and provided significant evidence that these threats will likely continue to impact the survival and recovery of FRC DUs covered in this report. For two of the main threats to these DUs, removal of forest cover and climate change, there is limited control over the impacts and no short-term mitigation options available. Massive losses of forest cover in the Fraser River watersheds in the Interior of BC have led to unstable freshwater habitat and hydrological conditions. Mitigation measures (e.g. reforestation) to improve forest cover are unlikely to improve these conditions for several generations (Perry and Jones 2017; Tschaplinski and Pike 2017). The loss of forest cover in the Fraser River basin will become increasingly evident in the future, as climate change effects are anticipated to exacerbate shifting hydrological conditions (earlier onset of freshet, changes in snowpack, drought, flooding; Brown 2002; Shrestha et al. 2012; Kang et al. 2014, 2016; Islam et al. 2019). This can profoundly affect the quantity and quality of freshwater rearing habitats, particularly for stream-type Chinook Salmon which use these freshwater rearing habitats for longer (Brown et al. 2019). These changes in hydrological conditions may also result in timing mismatches regarding the windows of habitat availability in the lower Fraser and estuary (R. Bailey, pers comm). While in the North Pacific Ocean, climate driven changes (increasing temperatures/heat waves, ocean acidification, shifts in prey distribution) are expected to continue to lower ocean productivity, and ocean conditions are not expected to improve in the near future (Walsh et al. 2018; Young and Galbraith 2018; Galbraith and Young 2019). Lower ocean productivity is likely to negatively impact the productivity of these DUs.

Currently, DUs 9, 10 and 11 are additionally impacted by a landslide in the Fraser River. The current and future impacts from the slide are unknown on the populations at this time, but it is anticipated that the work done to improve the slide site has resulted in improved migration survival over the slide. Harvest restrictions for 2019 and 2020 were put in place to reduce harvest impacts on these populations; however, without CWT indicator programs the harvest rates cannot be determined, nor if it was sufficient to negate the mortality at the slide. Additional monitoring will be required to determine the likely ongoing impact of the slide. While the full impact of the slide is unknown, it is anticipated that in the long-term it will have a neutral or negative impact and will not aid in the recovery of these DUs.

Given suspected declines in productivity, and the number and severity of threats impacting these FRC DUs, it is anticipated that these DUs will either continue to decline or level off at the recently observed low population abundances. It is unlikely they will recover without effective measures to mitigate threats.

Table 13 - The percent change values presented in Part 1 of the RPA (DFO 2020a), representing the median percent change observed over the last three generations, data permitting.

DU	DU Short Name	Time Series Length	Years	Median % Change	95% CI
DU2	LFR-Harrison	3 Gens	2007-2018	-40	-73, 33
DU4	LFR-Upper Pitt	3 Gens	2003-2018	-57	-80, -2
DU5 <sup>1</sup>	LFR-Summer	All Years	2005-2018	-43	-87, 139
DU7	MFR-Nahatlach	3 Gens	2003-2018	-83	-98, 74
DU8	MFR-Portage	3 Gens	2003-2018	-84	-94, -53
DU9	MFR-Spring	3 Gens	2003-2018	-49	-81, 45
DU10	MFR-Summer	3 Gens	2003-2018	-69	-86, -32
DU11	UFR-Spring	3 Gens	2003-2018	-58	-80, -12
DU14	STh-Bessette	3 Gens	2007-2018	-75	-98, 310
DU16	NTh-Spring	3 Gens	2003-2018	-87	-95, -64
DU17	NTh-Summer	3 Gens	2003-2018	-84	-95, - 55

<sup>1</sup> Three generations of data are not currently available for DU5 (LFR-Summer)

## 8. ELEMENT 14: SUITABLE HABITAT SUPPLY

*Element description: Provide advice on the degree to which supply of suitable habitat meets the demands of the species both at present and when the species reaches the potential recovery target(s) identified in element 12.*

RPA's aim to provide advice on the status of habitat *supply* and *demand*, and to inform discussion about whether habitat availability is currently limiting population growth, both at present, and when the species reaches its recovery target(s) (DFO 2014). *Supply* in this context refers to the amount of different habitat types known to exist, and how much each habitat type can be expected to support, should the population of the species saturate the habitat. *Demand* refers to habitat usage by the species, and is estimated from the population size and densities that can be reached in different types of habitat.

Freshwater habitat has been generally described for FRC (see Part 1 of RPA), yet it is difficult to assess this habitat in the context of requirements and supply and demand. This is particularly true for stream-type FRC that rear in freshwater for one or more years (10 of 11 DUs covered in RPA), some of which cover large geographic areas within the watershed (e.g. DU9 (MFR-Spring; 4490 km<sup>2</sup>), DU10 (MFR-Summer; 2616 km<sup>2</sup>), and DU11 (UFR-Spring; 4065 km<sup>2</sup>); COSEWIC 2018b). Stream-type FRC have been observed to exhibit three main strategies during the freshwater rearing stage: 1) juveniles rear in their natal stream from emergence until smolting; 2) juveniles rear in their natal stream from emergence to late summer and then migrate into a larger mainstem river such as the Thompson or Fraser where they overwinter before smolting the following spring; or 3) juveniles immediately leave their natal stream after emergence and disperse (actively and passively) downstream to overwinter in the mainstem, side channels, and small tributaries of the lower Fraser River and the estuary. Collectively this

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rearing habitat in the Fraser watershed makes up thousands of kilometers of streams of variable width and depth, and suitability within this habitat may change annually or seasonally due to environmental conditions (i.e. flow conditions, temperatures, turbidity, etc.). Further to rearing habitat, quantifying the supply and demand of spawning habitat for stream-type DUs also poses challenges as many DUs have multiple spawning sites, and not all are surveyed or surveyed consistently through time due to a variety of constraints (remote access, water turbidity, financial constraints, etc.). The availability and quality of spawning habitat and substrates can also change annually or seasonally due to environmental conditions or extreme weather events (high or low flows and temperatures, sediment inputs, anchor/frazil ice formation, etc.), posing further challenges in estimating habitat supply. However, we note it is unlikely that all imperilled stream-type DUs within the Fraser River watershed are simultaneously being limited from reaching their recovery targets from insufficient habitat supply within the freshwater environment.

DU2 is the only ocean-type population covered in this RPA, and spawns in a single and well defined area directly downstream of Harrison Lake in the Harrison River. This spawning habitat has historically supported much higher numbers of Chinook and there is currently no indication that a lack of spawning habitat is limiting this DU from reaching its recovery targets. There is, however, evidence to suggest that limited habitat in the Fraser River estuary may be contributing to declines in productivity for DU2 (Chalifour et al. 2020). Levy and Northcote (1982) reported Chinook Salmon had the highest density in brackish marsh channels in the Fraser estuary (maximum of 0.18 fish·m<sup>2</sup>), which is approaching densities in which substantially shorter residency times and decreased growth rates were observed in juvenile Chinook in the Nisqually River Delta (0.20-0.25 fish·m<sup>2</sup>; (Davis et al. 2018)). It is highly likely that the estuarine carrying capacity for Harrison River Chinook has been diminished through a variety of historical activities, and there continues to be an increase in hatchery production that potentially exacerbates this loss by increasing density-dependent effects in the remaining habitat (David et al. 2016; Chalifour et al. 2020). Estuarine habitat supply may therefore be limiting DU2 from reaching the recovery targets identified in Element 12, yet our limited understanding of habitat use in the estuary, paired with limited surveying and monitoring of estuarine habitat suitability, restricts our ability to provide advice on habitat supply and demand in the context of the RPA.

Marine habitat supply and demand is not well understood for FRC due to inherent challenges in surveying and monitoring vast unconstrained areas. Ocean carrying capacity is a highly dynamic ecosystem principle that fluctuates often, is strongly influenced by a plethora of ecological variables, and is in general poorly understood (Heard 1998). It has been suggested that the carrying capacity of the ocean may have been reached in recent decades, supported by relatively stable biomass estimates of both adult and immature wild and hatchery salmon in the north Pacific Ocean (Ruggerone and Irvine 2018). The size-structure and age-structure of Chinook salmon has also changed considerably across the Northeast Pacific Ocean since the late 1970s, with lower proportions of older age classes throughout most regions, and simultaneous declines in length-at-age of older fish and increased length-at-age of younger fish (Ohlberger et al. 2018). It remains unclear whether these demographic changes are a result of high levels of competition from hatchery production, changing environmental conditions impacting habitat and resource availability, changes in predator and prey interactions, or a multitude of other concurrent marine ecosystem processes. Despite this uncertainty, this may be an indication that habitat supply and demand in the marine environment is an important factor limiting the recovery of FRC DUs considered in this RPA.

This Element represents a notable gap in knowledge in the context of FRC, and has been highlighted as a major research need (Appendix H). For this element to be properly addressed, research on DU-level fry dispersal, behaviour, densities, and survival is required in combination with an assessment of the state of knowledge on habitat throughout the Fraser River watershed,

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the estuary and the North Pacific Ocean. Future assessment of the supply of suitable habitat would benefit from collaboration between DFO Science, DFO Fish and Fish Habitat Protection Program (FFHPP), the Ministry of Forests, Lands, Natural Resources Operations and Rural Development (FLNRORD) and the Ministry of Environment and Climate Change (MoE), as well as many individuals who have compiled information in various mapping databases who are associated with other organizations. Future assessments may also benefit from attempting to assess changes that have likely impacted the carrying capacity in the marine environment.

## **9. ELEMENT 15: ABILITY TO ACHIEVE RECOVERY TARGETS UNDER CHANGING CONDITIONS**

*Element description: Assess the probability that the potential recovery target(s) can be achieved under current rates of population dynamics parameters, and how that probability would vary with different mortality (especially lower) and productivity (especially higher) parameters.*

### **9.1. DU2 (LFR-HARRISON) RESULTS FROM VARYING PRODUCTIVITY AND HARVEST RATES**

To examine the probability of achieving recovery targets under different conditions, additional simulations with the Chinook Projection Model were conducted, varying both productivity and fishing mortality away from the base case. The bounds for future productivity scenarios are based on the changes observed in the time-varying analysis of alpha for DU2 (LFR-Harrison). The greatest changes in alpha observed for DU2 (LFR-Harrison), based on the time-varying analysis, are the decrease from a peak alpha in 1988 to a low in 1992 representing a 71% decline, followed by the greatest increase from 1992 to the second peak in 1999, a 168% increase. Accordingly, productivity change scenarios are simulated from a -70% to 170% change from the base case alpha, to capture the largest changes that have been observed previously. These changes in productivity were applied in equal increments over each projection period, so that in the last year the percent change from the baseline level is achieved. These scenarios do not represent the greatest instantaneous change in productivity as these historical changes in productivity occurred over shorter time scales, but they do exceed the greatest twelve year decreasing and increasing trends in productivity. It is important to note that these historical changes may not represent future changes in productivity, particularly given that the greatest changes observed were earlier in the time series and the more recent time period has seen changes of lower magnitude. Rather, these scenarios represent a way to explore the likely impacts of changing productivity on the probability of meeting targets. Scenarios of increased productivity can be viewed as cases where productivity either increased naturally, or through effective mitigation measures, whereas scenarios of decreasing productivity may occur if threats continue unabated or if mitigation measures take many years to be effective, causing productivity to remain below average or decline further in the future.

Fishing mortality was also decreased or increased by 10% increments relative to the estimated average Canadian harvest rates from the 2009–2015 base period. Changes were expressed as a percentage of age-specific harvest rates because the Chinook Projection Model does not include a vulnerability-at-age function that would allow a total annual harvest rate to be allocated among age classes. While a vulnerability at age function could be added to the model in the future, it was assumed for the current analysis that vulnerability-at-age was constant for the projection years. Harvest rates cannot be readily converted into a single annual exploitation rate because both immature and mature fish are captured in fisheries. To facilitate interpretation of the fishing mortality scenarios, the average catch year exploitation rates were calculated as  $catch/(catch+escapement)$  from the model outputs for each associated percent reduction in at-age harvest rates (Table 14). The changes in harvest rates were applied to the first year of the

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projection and then held constant, assuming that the implementation of management changes would be immediate and without error. It is important to note that the changes only represent reductions relative to Canadian harvest rates, while American harvest rates are held constant. This was determined to be the preferable option, as Canada does not control the American harvest rates, and hence cannot change those harvest rates as part of different management strategies.

When extrapolating from the results, it is important to consider that the harvest rates represent total mortality, and hence a reduction in harvest rates represents a reduction across all types of fishing related mortality and not just landed catch. Additionally, any potential increases in mortality not related to fishing (such as increased natural or invasive species predation, migration mortality or any human induced mortality) need to be considered in determining the ability to meet these mortality reductions. When assessing allowable harm and overall recovery potential, all additional factors of mortality should be considered, including all sources identified in the threats assessment in Part 1 (DFO 2020a).

As in Element 13 (Section 7), percent change was calculated over the three generations projected and the abundance was averaged over the last generation to determine if both requirements exceeded or met the survival and recovery targets. The likelihood of recovery was assessed as the percent of projections that reached the recovery targets under different combinations of changes to productivity and harvest. The heat maps displaying the likelihood of meeting the survival (Figure 9) and recovery (Figure 10) targets are shown below. As with Element 13, the IPCC risk categories are used to describe likelihoods (Table 11; Mastrandrea et al. 2010).

Results from the projections indicate that at recent productivity, DU2 (LFR-Harrison) would require harvest rates to be decreased by 80% or greater (~4% Canadian catch year exploitation rate) in order to be likely (66–90% certain) to reach the survival target in the next three generations (Figure 9). Alternatively, if harvest rates remain at current levels (2009–2015 average) (~18% Canadian catch year exploitation rate), a 40% increase in productivity would be required to be likely (66–90% certain) to reach the survival target. Several combinations of both reduced harvest rates and increased productivity were also likely to meet the survival target. If the recent productivity trend continues (-50% over the last three generations) it is unlikely (10–33%) that the survival target will be achieved unless harvest rates are reduced by > 70% (~6% Canadian catch year exploitation rate).

Under recent productivity conditions, none of the harvest rate scenarios resulted in an outcome where it was likely (66-90%) that the recovery target would be met (Figure 10). Holding recent harvest rates constant, it would require at least a 150% increase in productivity to achieve an outcome that was likely (66-90%) to meet the recovery target. Depending on the harvest levels, a productivity increase of >50 to 100% is required for DU2 to be likely (66-90%) to meet the recovery target. To move into a scenario of as likely as not (33-66%) to reach the recovery target requires either a > 70% decrease in harvest rates, a > 50% increase in productivity or a combination of increasing productivity and decreasing harvest rates.

Overall, the results from the Chinook Projection Model suggest that significant reductions in mortality are required to reach the recovery target, and that even if Canadian fishing mortality is eliminated it may not be possible to reach the recovery target at recent productivity levels.

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Table 14 - Catch year exploitation rate (ER) calculated from the model output, as catch/(catch+escapement), averaged over all simulations for each harvest rate (HR) scenario. The average American ER is 12%, which is why the Average Total ER is not 0% after a -100% reduction.

HR Scenario	Average Catch Year ER
+10%	31%
Base	30%
-10%	28%
-20%	27%
-30%	25%
-40%	23%
-50%	21%
-60%	19%
-70%	18%
-80%	16%
-90%	14%
-100%	12%

## Probability of Meeting Survival Target

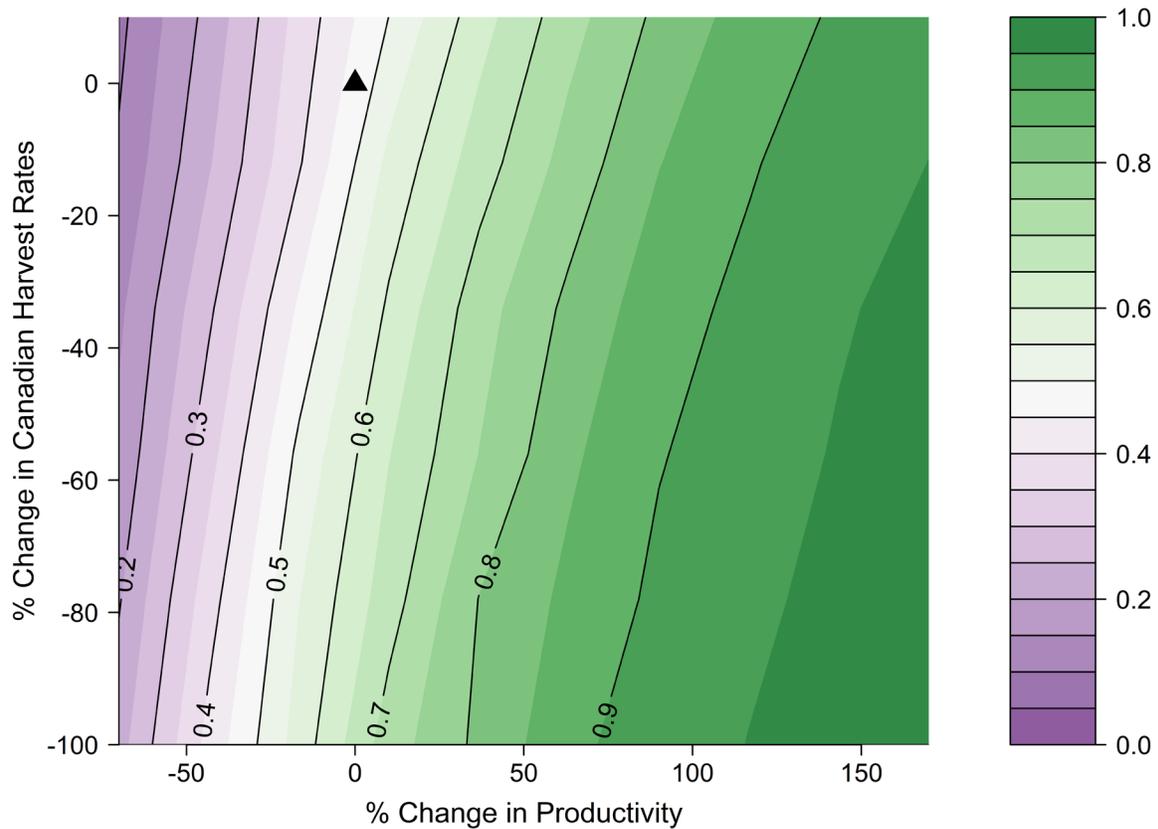


Figure 9 - Heat map showing the probability of reaching the Survival Target under changing productivity and percent reductions in Canadian harvest rates for DU2 (LFR-Harrison). Meeting the Survival Target requires an average abundance over the last generation greater than 15,313 and a percent decline over the last three generations of less than 30%. Triangle indicates base case conditions. Productivity is assumed to change linearly over the 12 year simulation from the base case value to the indicated percent change from the base case value. Percent reductions in Canadian harvest rates are based on the base case harvest rate (2009-2015) and are assumed to occur instantaneously in the first year and remain constant afterwards.

## Probability of Meeting Recovery Target

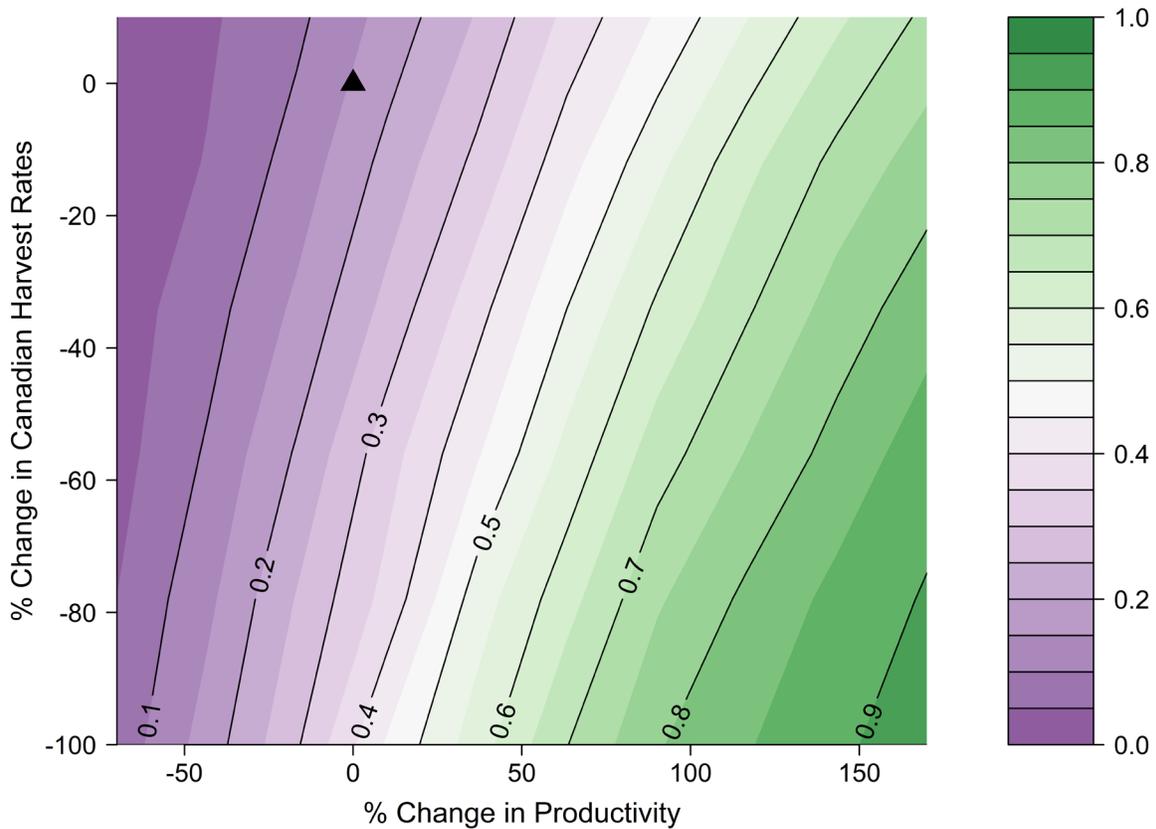


Figure 10 - Heat map showing the probability of reaching the Recovery Target under changing productivity and percent reductions in Canadian harvest rates for DU2 (LFR-Harrison). Meeting the Survival Target requires an average abundance over the last generation greater than 63,808 and a percent decline over the last three generations of less than 30%. Triangle indicates base case conditions. Productivity is assumed to change linearly over the 12 year simulation from the base case value to the indicated percent change from the base case value. Percent reductions in Canadian harvest rates are based on the base case harvest rate (2009-2015) and are assumed to occur instantaneously in the first year and remain constant afterwards.

### 9.2. QUALITATIVE DISCUSSION OF DATA-LIMITED STREAM-TYPE DUS

As mentioned in previous sections, data limitations prevented reliable modelling for the remaining 10 stream-type DUs covered in this RPA. The qualitative assessment conducted for these DUs in Element 13 (Section 7.2) indicated that under current conditions it is expected that these populations will continue to decline. Any increase in the number or severity of the threats discussed in Part 1, delays in the effects of mitigation measures, or even the continued unabated impacts of current threats is likely to result in the continued or even an increased rate of decline for these DUs. Efforts to improve the productivity and survival of these populations, through mitigating of both current and past threats, and preventing or mitigating future impacts, will increase the chances of recovery for these populations. While current harvest rate levels on these DUs is unknown, reducing the impacts of fishing will likely increase the chance of recovery for these DUs. As these populations appear to be in steeper declines than DU2 (LFR-

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Harrison) (Table 1), it is possible that they will require higher increases in productivity and greater reductions in harvest rates to reach recovery targets.

## **10. ELEMENTS 16 TO 20: EVALUATION OF POTENTIAL MITIGATION OPTIONS**

### **10.1. ELEMENT 16 INVENTORY OF MITIGATION MEASURES AND ALTERNATIVE ACTIVITIES**

*Element description: Develop an inventory of feasible mitigation measures and reasonable alternatives to the activities that are threats to the species and its habitat (as identified in elements 8 and 10).*

FRC use an extensive and diverse range of habitats throughout their life cycle, with considerable variability in habitat use and migration timing between populations (e.g. ocean-type vs. stream-type; see Part 1 for detailed descriptions of FRC life-history). This variability causes some DUs to be at greater risk than others, particularly for stream-type variants that rear in freshwater for one or more years (10 of 11 DUs considered in this RPA). There is also considerable inter-annual variability within freshwater and marine environments, in addition to a suite of threats and limiting factors of variable severity that can influence FRC survival or spawning success year-to-year (see Table 2 for summary of threats to FRC). Further to this, many of the threats identified in Part 1 of the RPA are extremely difficult to mitigate due to the many interrelated physical, biological, and chemical processes involved in large ecosystems such as the Fraser River watershed. The combination of these factors poses many challenges for mitigation planning and creates uncertainty associated with quantifying the effectiveness of mitigation measures once they are employed. There is also currently insufficient information to quantify DU-level benefits from individual mitigation activities for stream-type DUs, which greatly limits our ability to prioritize mitigation activities by both their importance to FRC recovery and the efficient use of recovery resources. For these reasons, this section does not attempt to prioritize mitigation options, rather provides a discussion of both broad and specific mitigation actions that could address threats identified in Part 1 of the RPA.

#### **10.1.1. Development**

Mitigation of threats associated with new developments can be addressed through project-specific measures to reduce, eliminate, or buffer the harmful effects associated with them. Coker et al. (2010) developed a broad guidance document to accompany Central and Arctic Region RPAs but it is relevant to all fish-bearing systems. Coker et al. (2010) comprehensively detailed linkages between works and activities and their “pathways of effects”, as well as mitigation strategies to break those pathways. These are specific mitigation measures that can be undertaken by those working in and around water. When development activities do not directly occur in fish habitat, the potential larger-scale implications on fish productivity are often not considered. Planning for development within all sectors needs to consider the cumulative hydrological effects within watersheds and the existing state of a watershed’s hydrological health, as which is inextricably linked to salmon survival and productivity (Hartman and Brown 1988; Tschaplinski and Pike 2017).

There are a number of legislated Acts and their associated guidance policies and documents that detail the regulations and best practices for works or activities which impact fish. These products include but are not exclusive to: the *Provincial Riparian Area Regulations* under the *Riparian Areas Protection Act*, the *Forest and Range Practices Act*, the *Mines Act*, the *Water Sustainability Act*, the *Federal Fisheries Act* and the *Fisheries Protection Policy Statement*. These Acts recognize the link between activities and habitat threats and provide the regulatory

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framework for reducing those threats; however, cooperation within multijurisdictional regulation frameworks, policy interpretation, planning, monitoring and enforcement are all areas which require support and funding.

The Acts listed above, policies, and guidance documents are only as useful as they are enforceable. In many cases mitigation is associated with extra costs. Significant gaps have been identified in models which use professional reliance or self-declared development plans with habitat impacts to ensure compliance with regulations (Ombudsperson of BC 2014; Haddock 2018). These planning and monitoring methods create a conflict of interest between profit and fish protection, which has detrimental effects on mitigation enforcement (Haddock 2018). Adequate resourcing to assist with third party planning, monitoring and enforcement of regulations is required. In addition to enforcement and third party planning, mandatory financial safety-nets for unforeseen problems (e.g. spills or breaches) would be beneficial. A legal and policy framework that is consistently applied at the municipal, regional district, provincial, federal, and First Nations levels would help to ensure the protection of salmon.

### 10.1.2. Agriculture and Aquaculture

Several threats to FRC associated with agriculture (loss/degradation of habitat, livestock entering streams) and aquaculture (various competitive interactions with hatchery fish) were identified in Part 1 of the RPA. Other threats related to agriculture and aquaculture such as water extraction and pollution were also identified, but are discussed in separate sections within this document (sections 10.1.7 Dams and Water Management, and 10.1.8 Pollution respectively).

Agricultural activities occur throughout the majority of the Fraser Basin, yet the threat to FRC exists mostly within the lower Fraser River where land use is highest and habitat is most limited. Intensification or conversion of existing agricultural land in the lower Fraser River was thought to be the most likely threat to most FRC in the future with increasing human populations and subsequent increasing demands for food resources. As within other sectors, mitigating the impacts of new agricultural development needs to consider both the direct physical impacts from those activities such as loss or degradation of habitat, and the larger scale implications such as impacts on stream hydrologic function, runoff dynamics, and pollution, among others. In addition to the acts listed above in section 10.1.1 Development, there are additional pieces of legislation that aim to reduce the impacts from agriculture, and include: the *Environmental Management Act*, *Public Health Act*, and *Integrated Pest Management Act*. Further to this, better planning of on-site agricultural activities would likely contribute to FRC recovery. Programs such as the [Environmental Farm Plan](#) aim to support agricultural operations in order to minimize environmental risks, and provide on-site assessments and guidance for factors such as riparian integrity, irrigation and drainage, water quality, air quality and emissions control, and on-farm materials storage. Programs such as these should be utilized when possible to ensure the protection of FRC habitat.

Fish aquaculture is pervasive in the Fraser River basin and nearshore rearing habitats, and it is probable that all FRC will encounter aquaculture as open net pens or hatchery fish at some point in their life cycle. There are likely negligible impacts on FRC resulting from the footprint of open net pens, yet there are concerns surrounding transmission of disease, introduction of genetic material, and fish escaping into the wild, among other. Transitioning to closed containment or land-based aquaculture will likely eliminate these interactions; however, there are also concerns surrounding competitive interaction between FRC and hatchery-origin fish, which compete for resources at all life stages and in all associated habitats and can negatively affect wild populations when resources are limited (Tatara and Berejikian 2012; see Part 1 of

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RPA for discussion of competitive interactions between wild and hatchery Chinook). Interactions between hatchery and wild FRC are discussed further in section 10.1.6 Hatchery Enhancement.

### **10.1.3. Fishing Impacts**

The nature of fisheries impacting FRC has changed significantly over the past 40 years. Reduced marine survival in the 1980s and subsequent management actions throughout the 1990s to conserve at-risk populations resulted in coast-wide reductions in fishing effort and landed catches observed over time (Brown et al. 2019). In 1997 and 1998, Canadian ocean fisheries were dramatically reduced to lessen impacts on Interior Fraser River Coho Salmon, further altering marine catch distributions and lowering ocean catches of FRC (Brown et al. 2019).

There are, however, a number of factors confounding the true effects of the reduction in fisheries. The harvest of co-migrating stronger and weaker salmon populations, whether wild or enhanced, is an inherent challenge in estimating the impacts of fisheries (Brown et al. 2019). In mixed-stock fisheries, there are risks of overfishing reproductively weaker or less abundant salmon populations that are mixed with stronger or more abundant wild or enhanced populations (DFO Salmonid Enhancement Program 2013). There is currently an inadequate understanding of the full impact of non-retention fisheries due to the potential for under-reporting of bycatch and uncertainties in the mortality rates of released fish. There are also unaccounted impacts from illegal fishing activity that further confound the response of populations to changes in fisheries. The impacts of non-retention fisheries and illegal fishing activity have been identified as future research needs, and are noted in Appendix H.

Impacts from net fisheries during co-migration of FRC could potentially be reduced by stipulating time and area closures, shorter gill-net set times, shorter nets, larger gill-net mesh size or tangle tooth gear and active fishing of set nets as opposed to passive fishing methods. Making use of brailing methods on seine boats facilitates recovery of released fish, as do recovery tanks when they are properly used. Recreational fisheries mitigation may include but is not limited to: use of gear which decreases impacts to released fish such as barbless hooks, mandatory fish handling and fish identification courses/exams (similar to a Conservation and Outdoor Recreation Education exam for hunting), and diminished fishing opportunities when compliance with regulations fail to reach target levels. Research and stock assessment activities must use the least invasive methods when possible.

Reduced harvest represents one of the few immediate mitigation measures available to reduce impacts on FRC, but even in the absence of fishing many DUs may not recover in the short-term. This is particularly true for spring-run FRC that spawn above the Big bar landslide (DU9 MFR-Spring, DU11 UFR-Spring). Recent joint conservation efforts between First Nations, conservation stakeholders, and DFO will aim to reduce fishing opportunities for these stocks until passage improvements at the slide evident.

### **10.1.4. Forestry and Wildfire Management**

Numerous activities related to forestry and wildfire management, both historical and current, were identified as threats to FRC in Part 1 of this RPA (see sections 4.1.4 Shipping Lanes, 4.1.5 Biological Resource Use, 4.1.7 Natural Systems Modifications). In summary, historical clear-cut logging and riparian vegetation removal have had significant negative impacts on stream channel stability, stream temperatures, runoff dynamics, seasonal hydrographs, and overall forest health throughout areas of the Fraser Basin. Current forestry practices aim to reduce these impacts by employing more sustainable and selective cutting rates, requiring buffer zones in riparian habitat, and considering information such as forest health/diversity, wildfire and fuel

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management, fish and wildlife status, climate change, and cumulative effects into timber management goals (FLNRORD 2017); however, wildfires, pest infestations and disease are becoming more recurrent threats within BC, and subsequent salvage logging operations following these events were identified in Part 1 of the RPA as a likely threat to FRC in the future. Salvage logging typically covers larger areas than conventional cutblocks and can occur within riparian habitat due to exemptions for salvaging timber damaged by fire, insects, or disease, suggesting that unless forest regulations and practices change, impacts from future salvage logging on FRC is probable.

Future planning for salvage logging and timber harvesting needs to consider and align with the recovery goals of FRC, including both the physical impacts from these activities and more importantly, the larger implications on hydrological function through modified catchment surfaces. There are several pieces of provincial legislation in place to guide sustainable forestry practices both on public and private land, including the *Forest Act*, *Forest and Range Practices Act*, and *Private Managed Forest Land Act*, yet as with other sectors, these acts need to be updated regularly and require support for monitoring and enforcement. Changing legislation to eliminate or reduce aggressive salvage logging operations following forest disturbances, as was seen following the outbreak of Mountain Pine Beetle in BC, is also critical for the long-term recovery of FRC.

Log storage in the lower Fraser was also identified in Part 1 of the RPA as a threat to FRC transiting and rearing in the lower Fraser River (sections 4.1.4.3 Shipping Lanes). The lower Fraser is a highly active channel for log boom shipping and contains a high concentration of log booms and barges, which can lead to a variety of adverse physical, chemical, and biological effects to the surrounding environment (Power and Northcote 1991; Nelitz et al. 2012). Log booms can also provide cover and attract inbound migrating Chinook Salmon seeking refuge; however, they can also attract predators such as Harbour Seals, which use log booms as haul-out sites and for pupping (Baird 2001; Brown et al. 2019). This area is also known to support millions of outmigrating salmon which occupy marine foreshore areas after smoltification, and prior to migrating out to sea (Nelitz et al. 2012). Removals or reductions of current log storage areas in the lower Fraser River and estuary will likely improve the quantity and availability of nearshore habitat for FRC (and other Pacific salmon species) rearing in or travelling through the lower Fraser River, and should be considered as a mitigation activity to improve FRC habitat.

### **10.1.5. Invasive and Problematic Species**

The introduction of aquatic invasive species (AIS) is extremely difficult to mitigate as it takes only a few individuals, sometimes introduced unintentionally, to irrevocably alter a watershed. There has been a long history of failures to manage aquatic invasive species before irreversible damage has been done to ecosystems, both on the federal and provincial/state level in the Pacific Northwest (i.e. Columbia and tributaries), therefore early action is paramount in managing AIS. Once AIS become established, they can be extremely difficult to manage without impacting native biological communities using conventional suppression techniques such as physical removal (netting, electrofishing) and chemical intervention (i.e. Rotenone). Where AIS are detected, all efforts to eradicate those species should be undertaken as quickly as possible and monitoring programs should be implemented and sustained to ensure eradication is complete. This is particularly true for species that have short maturation times, high fecundity, and great dispersal mechanisms such as Dressedid mussels and European Green Crab (see Part 1 for detailed descriptions of threats to FRC from AIS (DFO 2020a)), which have been identified as potential major threats to ecosystem function in the Fraser River drainage. Detection of biological invasions in their early stages is, however, challenging when population densities are at a minimum, and conventional surveying techniques require considerable

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resources to conduct and have the potential to negatively impact non-target species, in addition to having questionable effectiveness when target species abundance is low (Olsen et al. 2015). The use of environmental DNA (eDNA) sampling has gained considerable interest since its inception (Ficetola et al. 2008) as a non-invasive technique to detect and monitor invasive or rare freshwater species, requiring minimal effort in the field and eliminating potential negative impacts on non-target species. The implementation of routine eDNA monitoring programs in likely areas of introduction may be an option to track the colonization and/or spread of AIS.

Mitigation of AIS should involve a multipronged approach of public education, monitoring of areas likely to be points of introduction, and enforcement through strong disincentives. Preventing or slowing the secondary spread of already established invasive populations is also an important consideration in long-term management of AIS (Vander Zanden and Olden 2008).

Predation by pinnipeds (Harbour Seals, Stellar Seal Lions, California Sea Lions) was identified as a potentially major source of mortality for FRC in Part 1 of the RPA, particularly for DUs with significantly depressed abundances (see section 4.1.8.2 Problematic Native Species). While there has been considerable work investigating the effects of predatory interactions between FRC and pinnipeds, there are a vast number of other ecological processes at play within the Salish Sea confounding our understanding of these interactions and their impacts on FRC.

There are few direct mitigation strategies available to reduce impacts of predation, with the exception of lethal removal (culling) or non-lethal removal (capture or relocation). A recent technical workshop hosted by the Institute for the Oceans and Fisheries (University of British Columbia), which included a broad group of scientists and managers from both Canada and the US with technical expertise on pinnipeds and salmonids, convened to evaluate the current state of knowledge and uncertainties surrounding the diets and population dynamics of pinnipeds, as well as the impacts that pinnipeds may be having on Pacific Salmon in the Salish Sea (Trites and Rosen 2019). The proceedings from this workshop go into considerable detail surrounding pinnipeds and their interactions with Pacific Salmon (see Trites and Rosen (2019)); however, the general consensus from this workshop was that data are insufficient at this time to justify mitigation in the form of culling pinnipeds in the Salish Sea, due to high levels of uncertainty in the both our current state of information and the indirect effects of conducting a cull. Non-lethal alternatives such as capturing or harassing pinnipeds during critical times were also discussed, yet considerable thought would have to be given to implement such actions as to avoid habituation over time. As mentioned in section 10.1.5 Forestry and Wildlife Management, log booms were identified to attract FRC and other salmon seeking refuge, but also attract other predators and serve as haul-out sites for Harbour Seals. Removal of log booms in key areas, particularly in estuaries, may be beneficial in reducing the number of pinnipeds that prey on FRC seeking refuge.

Further research is needed to better understand the indirect effects of culling predators and other factors that influence ecosystem function such as food web relationships, shifting prey/predator distributions, and hatchery practices. Further to this, with our limited understanding of both Pacific Salmon and pinniped population dynamics, we have little capability in determining whether removals are producing the intended effect. Further investigation of pinniped predation has been identified as a future research need for FRC mitigation planning, and is noted in Appendix H.

#### **10.1.6. Hatchery Enhancement**

Hatchery enhancement has been used both as a conservation tool and for maintaining Pacific salmon fisheries in Canada following recognition of rapidly declining catches in the 1970s. Hatcheries have been used successfully to meet certain conservation goals, yet they have also

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raised a number of ecological concerns and have become a controversial issue in conservation biology (National Research Council 1996; Myers et al. 2004; Lackey 2013). In Part 1 of the RPA potential issues stemming from high levels of hatchery production are discussed in detail (DFO 2020a). In summary, enhancement and hatchery programs can reduce genetic diversity and fitness, increase intraspecific competition in highly degraded habitat, and can lead to higher fishing mortality rates for wild salmon. In the U.S. Pacific Northwest, Chinook Salmon conservation activities have been occurring for many decades, dating back to the period when dams were being constructed on the mainstem of the Columbia River, which can provide helpful information for programs aimed at rebuilding depleted FRC populations. The negative effect of hatcheries on wild Chinook Salmon survival has been reported, although the presence and magnitude of effects varies depending on the nature and magnitude of the hatchery-wild interaction. Relatively little information is available about ecological interactions between hatchery and wild salmon in the marine environment in and around the Strait of Georgia.

Mitigating interactions between hatchery and wild fish across their entire shared environment, and over their entire life cycle, is particularly challenging due to the migratory behaviour of salmon where multiple stocks often mix. Genetic effects from interbreeding with hatchery fish may also remain for several generations after hatchery reductions are made. However, even moderate decreases in the level of hatchery production may decrease hatchery-wild fish interactions and allow wild fish to locally adapt to their environment (Kostow 2009). Therefore, genetic impacts may only be mitigated by reductions of interbreeding in a natal river but other ecological processes may be mitigated by managing regional hatchery production.

The landslide near Big Bar on the mainstem Fraser River poses a serious threat to all DUs that spawn above the slide, and there is potential for some populations to be extirpated without some form of hatchery supplementation. Numerous streams above the landslide have been proposed for broodstock collection within DU9 (Chilako River, Endako River, Horsefly River, Upper Cariboo River, Upper Chilcotin River, West Road (Blackwater) River), and DU11 (Bowron River, McGregor River, Salmon River, Slim Creek, Swift Creek, Tete Jaune (Fraser River), Torpy River, Willow River). There is, however, considerable uncertainty surrounding the feasibility or effectiveness of these activities, and ongoing assessment will inform and determine future enhancement decisions for these DUs.

### **10.1.7. Dams and Water Management**

The threat to FRC through water management and utilization (for a variety of sectors) in the Fraser River basin is pervasive for all DUs discussed in this RPA. This includes threats from structures related to flood control (i.e. dikes, flood boxes, tide gates), dams and hydroelectric development, and water extraction.

There are no hydroelectric dams on the mainstem Fraser River; however, major facilities on the Seton (Seton Dam), Bridge (Terzaghi Dam), and Nechako (Kenney Dam) rivers have ongoing impacts on DUs 8 (MFR-Portage), 9 (MFR-Spring) and 10 (MFR-Summer), respectively. The Seton Dam is the only passable structure of these three facilities; all returning DU8 (MFR-Portage) spawners must migrate over the fishway to reach their spawning grounds in Portage Creek, and all out-migrating smolts must migrate downstream and may be entrained at the facility. It is therefore imperative to maintain and maximize passage of DU8 (MFR-Portage) spawners through the fishway and reduce mortalities of out-migrating juveniles. The footprint of the Terzaghi and Kenney dams has significantly altered the surrounding ecosystems and ongoing impacts with water release strategies have been identified and discussed in detail in Part 1 of the RPA (DFO 2020a). Water release strategies must adhere to methods informed by system-specific ecological flow requirements, which may be important for both adults and juveniles. Ecological flow requirements must include spring freshets to incorporate

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allochthonous material, clear sediments from spawning gravel, and introduce woody debris and inundate off channel habitat (Biggs et al. 2005). Water release must also be mindful of summer temperature and flow management requirements for FRC and other salmon species.

Temperature may be better controlled by designing dams that can release from lower stages of the water column as well as spilling from the surface of impoundments.

In addition to large dams, there are many smaller water impoundment structures on lake headed systems in FRC watersheds and in the lower Fraser River. Water management with regard to extraction of overland flows and aquifers may be in direct conflict with the water needs of FRC and other stream-dwelling animals. These structures are mostly in place for irrigation and flood mitigation purposes, most of which are not currently managed in a manner that addresses passage or flow requirements for fish. Flood mitigation structures impede the dispersal of juvenile Chinook Salmon into favoured off channel areas during spring freshets. Recognition and protection of off-channel habitat for FRC rearing is critical to maintaining productivity into the future.

Mitigation of smaller water impoundment structures is difficult because mitigation often involves maintaining or restoring the flood function of streams, which is frequently in direct conflict with human settlement (see Estuary Restoration section above). The current water extraction network is difficult to govern, monitoring of surface extraction is inadequate, and monitoring of groundwater removal is almost non-existent. As well, in times of drought, the enforcement response is frequently slow and until conditions are extreme, mitigation is strictly voluntary. Though modern water licences are granted with metering requirements and within associated allocations, many water licences still exist that are unmetered. Water extraction in some river systems is now recognized to be over-allocated, but there are few options to retract licenses (Brown et al. 2019). There is growing recognition in BC's regulatory framework of the importance of aquifer sources to environmental needs. Section 55(4) of *The Water Sustainability Act* now clarifies that government has the discretion to consider environmental flow needs when adjudicating both new and pre-existing groundwater use. Though *The Water Sustainability Act's* move to licence ground water is a step forward, there is still work required to incorporate current ground water wells into the regulatory framework, meter all extraction activities, and create water allocation regimes that include planning for fish-habitat requirements in order to sustain salmon habitat.

### **10.1.8. Pollution**

Numerous sources of pollution, both historical and present, were identified in Part 1 of the RPA as posing a significant threat to FRC, and include: Household Sewage & Urban Waste Water; Industrial & Military Effluents; Agriculture & Forestry Effluents; Garbage & Solid Waste; and Air-Borne Pollutants. Many of these contaminants are persistent in the environment, may travel long distances, and have a tendency to accumulate in sediments and food chains from multiple sources. Further to this, contaminants generated from multiple sources accumulate as mixtures in the environment therefore the effects from individual pollutants are extremely difficult to ascertain from one another, and thus prioritize mitigation activities to reduce their harm.

The principal pieces of legislation in place for environmental pollution issues in British Columbia include the provincial [Environmental Management Act](#) and [Waste Discharge Regulation](#), and the federal *Canadian Environmental Protection Act*, *Fisheries Act*, and *Canada Water Act*. Legislation and operational changes over the last several decades have been effective in reducing pollution from a variety of sectors, and while current legislation/regulation aims to reduce environmental contamination, the effects of historical activities still pose a noteworthy threat to FRC at all life stages. This is particularly true within the lower Fraser River and estuary, which has historically been the epicenter of anthropogenic activities within the province that

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generate pollution, in addition to serving as a bottleneck for pollutants accumulated throughout the Fraser Basin (refer to section 4.1.9, Pollution & Contaminates in Part 1 of RPA for discussion of pollution in the Fraser Basin). All FRC must transit the lower Fraser and estuary during outmigration to the ocean and during their return spawning migration, and are thus exposed to environmental pollutants twice within these areas.

One of the few current options we have available for mitigating future pollution is the adoption and enforcement of more strict regulations on activities that generate and release contaminants into the environment. There are, however, inherent challenges in monitoring the release of pollution due to the vast number of sources within the Fraser Basin and surrounding coastal areas. This is particularly true when self-reliance of reporting and potential loss in revenue is involved (see section 10.1.1 Development). Monitoring programs like [PollutionTracker](#) are currently working to document the levels and trends of a variety of contaminants within coastal BC. Expansion of monitoring programs such as this, particularly within the interior Fraser Basin, would be beneficial for identifying and reducing the release of pollution that may impact FRC.

Remediation of polluted sites that are either within salmon habitat, or that influence salmon habitat through the release of contaminants (effluents, runoff, groundwater inputs, etc.), is another important component for the recovery of FRC. Remediation of contaminated sediments commonly employs activities such as dredging (mechanical or hydraulic removal of contaminated sediment), dry excavation (de-watering and physical removal of contaminated sediment), capping (covering contaminated sediments with clean material or geotextiles), the use of sorptive agents (mixing of sediments with reactive sorbants to isolate contaminants), and in-situ amendments (addition of chemicals/compounds to promote destruction or immobilization of contaminants) (Perelo 2010; Bullard et al. 2015). An alternative non-invasive mitigation strategy for contaminated sediments is monitored natural recovery (MNR), which relies on the metabolic potential of microorganisms, paired with naturally occurring physical and chemical processes to degrade contaminants over time (Perelo 2010; Bullard et al. 2015). Each of these mitigation strategies have number of associated considerations in terms of their usefulness, feasibility, and sustainability, and should be thoroughly investigated on a project-specific basis.

Considerable work is needed in order to inventory and prioritize remediation of environmental pollution for FRC, particularly at the DU level, and has been identified as a major knowledge gap that needs to be addressed for future recovery planning.

### **10.1.9. Climate Change**

Climate change encompasses a large suite of complex and inter-related issues that threaten FRC, and is likely to exacerbate many of the threats discussed in Part 1 of the RPA (DFO 2020a). These cumulative impacts may impede progress on many of the previously recommended mitigation measures. For example, more extreme precipitation events caused by climate change will compound with the increased run-off rates that result from logging and forest fires. Impediments to mitigation activities for those threats may occur through creation of new impoundment structures, increased failures of tailings ponds and water treatment facilities that introduced effluent, as well as higher rates of scouring and the increase in the likelihood of bank failure and of avulsion events. In addition, failures of infrastructure due to extreme events may lead to a greater number of in-stream work that may in turn contribute to threats as discussed under the Development threats section in Part 1 (DFO 2020a).

The current regulatory framework and best practices with regard to emergency works, water and tailings dam planning and management, forestry cut rates and block planning, bridge engineering, storm-water management and occupation of flood plains through urban encroachment may all need to be reconsidered to mitigate for the more regular arrival of higher

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flood flows, and altered snowpack melt regimes. The current practices of unregulated groundwater extraction, unmonitorable and hence unmonitored surface water extraction activity, slow reaction times to drought conditions, and lack of planning around watershed-level hydrological function will all need to improve and be more responsive to climate change.

Combating the effects of climate change is a global issue, and there are no simple measures available to mitigate the impacts in the short term. The negative effects from climate change are not anticipated to diminish or reverse in the foreseeable future, therefore, considerable preparation and planning is needed to restore and conserve the remaining habitat available to FRC and other imperiled salmonids. The recent [Paris Agreement](#) and the [United Nations Intergovernmental Panel on Climate Change](#) provide guidelines to aid in the global effort of combatting and adapting to climate change, and FRC populations and their habitats should be managed according to these guidelines so that they are resilient and can adapt to future environmental changes.

#### **10.1.10. Estuarine, Intertidal, and Riparian Habitat Restoration**

There has been significant degradation of historical rearing habitat in the lower Fraser River and estuary from various developments and flood control structures (e.g. dikes, flood boxes, tide gates, etc.). These developments have led to major losses of the Fraser River estuary (70-90 %; Levings 2004), and restricted access to floodplain and off-channel habitat that provide critical foraging and growth opportunities for juvenile FRC. There can be substantial early natural mortality in the marine environment resulting mostly from predation when juvenile Chinook do not grow large enough to reach a critical minimum size by July (Duffy and Beauchamp 2011) or the end of their first marine summer (Beamish et al. 2011). Fostering the restoration of freshwater, brackish, and saline marshes is one possibility for increasing the functional capacity of the estuarine habitat, and represents a crucial mitigation option to prevent habitat loss from rising sea levels (Temmerman et al. 2013). Habitat restoration in estuaries is, however, often confounded by the complexity of the salmon life cycle and variation in habitat needs at multiple spatiotemporal scales (Simenstad et al. 2000). Additionally, there frequently appears to be no effect, or even detrimental effects related to biological interventions undertaken to promote the recovery of biodiversity and functionality in estuaries (Moreno-Mateos et al. 2015), and there are demonstrated risks to over-engineering a restoring ecosystem or encouraging homogeneity among habitats (Elliot et al. 2016). Careful consideration must therefore be put into restoration planning to overcome these challenges.

While not FRC specific, recent habitat restoration efforts in the Nisqually River Delta, Washington, provide evidence that re-establishing tidal influences to a heavily modified estuarine ecosystem can increase prey resources and forage opportunities for juvenile salmon. Post-restoration monitoring data indicates substantial increases in invertebrate biomass following re-establishment of tidal inundation, greatly enhancing foraging capacity of salmon (Woo et al. 2018). Similar enhancements of habitat within the Fraser River estuary may be a viable mitigation measure to provide valuable prey resources for juvenile salmon and other fishes, and to increase the recovery and survival of FRC; however, as seen with the aforementioned example in the Nisqually River Delta, significant modifications and losses of existing habitat are required to complete similar enhancements. This could be accomplished through the removal of engineered barriers to tidal exchange (i.e. tide gates, flood boxes) encouraging the formation of tidal channel networks, increasing overhanging riparian vegetation, and improving environmental conditions for invertebrate productivity loss (Davis et al. 2019). The development of complex tidal channel networks with overhanging vegetation can lead to shaded waterways with more stable water temperatures (Beck et al. 2001; Bertness and Ewanchuk 2002; Whitcraft and Levin 2007), while also providing habitat and structure for

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terrestrial prey (Kneib 1984; Allan et al. 2003; Woo et al. 2018). There may be more beneficial implications for wild Chinook Salmon populations, which appear to have longer delta residence times and are more likely to use estuarine tidal wetlands during their out-migration to the sea when compared to larger hatchery-origin fish (Chittenden et al. 2018; Davis et al. 2018). The broader trophic niche and longer delta residence times of wild juvenile Chinook Salmon may allow them to exploit resources better than hatchery Chinook and thus to have higher bioenergetic growth potentials (Davis et al. 2018).

There are currently efforts underway through a variety of organizations to restore marsh and tidal channel habitat in the lower Fraser River, to enhance connectivity within the Fraser River delta, and improve habitat within the interior Fraser. Examples include: the Fraser River Estuary Connectivity Project (Raincoast Conservation Foundation); Connected Waters (Watershed Watch Salmon Society); Resilient Waters (MakeWay Foundation); and the Tsawwassen Eelgrass Project (Vancouver Fraser Port Authority Habitat Enhancement Program). There would be a major benefit from improved coordination and planning of restoration activities within the Fraser River estuary, as mitigating historical damages to this highly degraded habitat will require both considerable planning and the use of large-scale operations to make meaningful improvements to ecosystem function.

### **10.1.11. Conclusion**

The above sections have identified a broad range of mitigation activities/strategies and their relation to threats identified in Part 1 of the RPA, yet alleviating many of these threats will be extremely challenging, especially since many are interrelated and exacerbated by climate change. Table 15 provides a summary of the various threats facing Chinook Salmon and the possible mitigation measures to address each of the threat categories.

A rapid change in practices for many activities and their regulation is needed to reduce further impacts on FRC and the other imperiled Pacific Salmon species in the Fraser (Interior Fraser Coho, Interior Fraser Steelhead, Fraser Sockeye) in the future, and more recognition of the cumulative effects these activities have is needed within management. Further to alleviating future threats, there is also a great need to restore historical damages from development and resource extraction activities that continue to impact hydrologic function within the Fraser Basin. Re-stabilization of more natural hydrological regimes and restoration of highly degraded habitat, particularly in the lower Fraser River and estuary, would facilitate work to address many of the aforementioned issues negatively impacting freshwater and estuarine productivity. These are, however, multi-generation endeavors, and is only possible if future management/planning from all sectors is in line with the recovery goals of FRC.

A common theme within the mitigation categories discussed above is that a more coordinated and informed approach to managing anthropogenic activities is needed. Undertaking a more coordinated approach would promote more efficient use of limited human resources, and facilitate access to the broad range of specialists required to develop such a strategy and manage its implementation over time. There is also a need to incorporate adaptive management strategies when planning mitigation activities, including current research on land use changes, intra- and interspecific competition, changing ocean and estuarine habitat conditions, and climate change, in addition to being regularly updated based on new information (Maas-Hebner et al. 2016).

Appendix H provides a summary of research needs for FRC recovery planning, and considerable work is needed in these areas before prioritizing mitigation actions for the DUs assessed in this RPA; however, by promoting the recovery actions as beneficial across multiple

species, there may be greater acceptance of measures and financial cost required to achieve recovery.

Table 15 - Possible mitigation strategies to address threats to FRC identified in Part 1 of the RPA

<b>COSEWIC Major Threat Category</b>	<b>Threat Category Description</b>	<b>Possible Pathway(s)</b>	<b>Possible Mitigation Options</b>	<b>Notes</b>
<i>Residential and commercial development</i>	<ul style="list-style-type: none"> <li>• Footprints of residential, commercial, and recreational development</li> </ul>	<ul style="list-style-type: none"> <li>• Loss or degradation of habitat</li> </ul>	<ul style="list-style-type: none"> <li>• Manage ongoing and future development in the context of salmon habitat requirements, mandate and monitor compensatory works for loss of habitat</li> </ul>	-
<i>Agriculture &amp; aquaculture</i>	<ul style="list-style-type: none"> <li>• Footprints of agriculture, horticulture, and aquaculture</li> <li>• Competitive interactions with hatchery fish</li> </ul>	<ul style="list-style-type: none"> <li>• Loss or degradation of habitat</li> <li>• Competition</li> </ul>	<ul style="list-style-type: none"> <li>• Manage ongoing and future activities/development in the context of salmon habitat requirements, mandate and monitor compensatory works for loss of habitat</li> <li>• Transition to closed containment aquaculture</li> <li>• Reduce hatchery production, employ adaptive and alternative hatchery production strategies (e.g. time and size of release)</li> </ul>	Note that there is a large amount of surplus hatchery production outside of the Fraser River; the Chilliwack River Hatchery is a notable exception
<i>Energy production &amp; mining</i>	<ul style="list-style-type: none"> <li>• Footprints and extraction activities from mining (e.g. gravel extraction, placer mining, etc.).</li> </ul>	<ul style="list-style-type: none"> <li>• Loss or degradation of habitat</li> </ul>	<ul style="list-style-type: none"> <li>• Manage ongoing and future activities/development in the context of salmon habitat requirements, mandate and monitor compensatory works for loss of habitat</li> </ul>	-
<i>Transportation &amp; service corridors</i>	<ul style="list-style-type: none"> <li>• Footprints from roads, railroads, utility and service lines, and shipping lanes</li> </ul>	<ul style="list-style-type: none"> <li>• Loss or degradation of habitat</li> </ul>	<ul style="list-style-type: none"> <li>• Manage ongoing and future activities/development in the context of salmon habitat requirements, mandate and monitor compensatory works for loss of habitat</li> <li>• Use salmon friendly stream crossings (e.g. free span bridges, baffles, etc.), upgrade old passages (e.g. hanging culverts)</li> </ul>	-
<i>Biological resource use</i>	<ul style="list-style-type: none"> <li>• Logging and wood harvest in riparian areas, transport of logs via rivers</li> <li>• Fishing</li> </ul>	<ul style="list-style-type: none"> <li>• Loss or degradation of habitat</li> <li>• Direct and indirect mortality</li> </ul>	<ul style="list-style-type: none"> <li>• Update/improve forestry policy in the context of protecting and restoring salmon habitat and riparian areas, managing the time and abundance of log booms in river, monitor and enforce water quality requirements for salmon health</li> <li>• Manage the time and abundance of log booms in river, monitor and enforce water quality and effluent targets around booms</li> <li>• Adaptive fisheries management, increased monitoring and enforcement, minimize fisheries related mortality (direct and incidental), education on identification of salmonids and conservation concerns</li> </ul>	Fishing effects are transboundary and are associated with mixed stocks and mixed species

<b>COSEWIC Major Threat Category</b>	<b>Threat Category Description</b>	<b>Possible Pathway(s)</b>	<b>Possible Mitigation Options</b>	<b>Notes</b>
<i>Human intrusions &amp; disturbance</i>	<ul style="list-style-type: none"> <li>• Recreational activities (e.g. ATVs in streams, jet boats, etc.)</li> </ul>	<ul style="list-style-type: none"> <li>• Loss or degradation of habitat</li> <li>• Direct and indirect mortality</li> <li>• Alteration of behaviour</li> </ul>	<ul style="list-style-type: none"> <li>• Manage access (e.g. infrastructure) to water and allowable activities (e.g. regulations) over time and space, increased monitoring and enforcement</li> <li>• Increased education on interacting with streams and salmon</li> </ul>	-
<i>Natural systems modifications</i>	<ul style="list-style-type: none"> <li>• Fire and fire suppression</li> <li>• Dams and water Management</li> <li>• Modifications to catchment surfaces, forestry, and linear development</li> </ul>	<ul style="list-style-type: none"> <li>• Loss or degradation of habitat</li> <li>• Direct and indirect mortality</li> <li>• Alteration of behaviour</li> </ul>	<ul style="list-style-type: none"> <li>• Update/improve forestry policy in the context of conserving watershed functions that support salmon; mandate, monitor, and manage reforestation and restoration activities (including managing for mature forest characteristics)</li> <li>• Use strategic burning to prevent large fires</li> <li>• Manage ongoing and future development of water resources, increase monitoring and enforcement of surface and ground water, specifically with salmon biological requirements as targets</li> <li>• Decommission or remove dams, increase, monitor, and maintain fish passage infrastructure for adults and juveniles (fishways, fish ladders, etc.)</li> <li>• Adaptively manage water in the face of climate change and increased variability</li> <li>• Manage ongoing and future linear developments by imitating more natural waterways, reconnecting off-channel habitat, removing or restoring old developments, and set and monitor water quality and sediment targets</li> <li>• Consider the impacts of cumulative effects in decision making</li> </ul>	-

<b>COSEWIC Major Threat Category</b>	<b>Threat Category Description</b>	<b>Possible Pathway(s)</b>	<b>Possible Mitigation Options</b>	<b>Notes</b>
<i>Invasive &amp; other problematic species &amp; genes</i>	<ul style="list-style-type: none"> <li>• Aquatic invasive species (AIS), introduced pathogens and viruses, problematic native species (e.g. pinnipeds, parasites, and disease), interbreeding with hatchery-origin fish</li> </ul>	<ul style="list-style-type: none"> <li>• Loss or degradation of habitat</li> <li>• Alteration of behaviour</li> <li>• Predation and competition</li> <li>• Increased prevalence of infection</li> <li>• Reduced genetic diversity and natural selection forces</li> </ul>	<ul style="list-style-type: none"> <li>• Removals of AIS, prevention of introduction through increased monitoring for new and of existing AIS populations, increased enforcement and education surrounding introductions of AIS</li> <li>• Monitoring and treatment of pathogens in aquaculture, transition to land-based aquaculture and increased treatment of aquaculture effluent, implement and monitor predator control measures</li> <li>• Reductions in log booms in lower Fraser and estuary that serve as haul-out sites for pinnipeds</li> <li>• Monitor hatchery and wild genetics and implement adaptive production planning, mass mark hatchery fish to identify and remove from natural breeding population, minimize hatchery production</li> </ul>	Pinniped populations have increased due to protection of marine mammals; research is required on the efficacy and direct applicability of predator controls
<i>Pollution</i>	<ul style="list-style-type: none"> <li>• Introduction of exotic and/or excess materials or energy from point and nonpoint sources, including nutrients, toxic chemicals, and/or sediments from urban, commercial, agricultural, and forestry activities</li> </ul>	<ul style="list-style-type: none"> <li>• Altered behaviour and physical condition due to hormone and developmental que mimics, gene regulation, and other toxicities, potentially reducing survival and resilience</li> </ul>	<ul style="list-style-type: none"> <li>• Manage ongoing and future activities/developments that contribute to pollution, improve waste water management and monitoring, increase enforcement of best practices for water quality</li> <li>• Removal or remediation of contaminated sediments</li> </ul>	-
<i>Geological events</i>	<ul style="list-style-type: none"> <li>• Avalanches and landslides</li> </ul>	<ul style="list-style-type: none"> <li>• Stop or reduce passage</li> <li>• Increased mortality associated with passage</li> </ul>	<ul style="list-style-type: none"> <li>• Increase, monitor, and maintain fish passage infrastructure for adults and juveniles (e.g. fishways, fish ladders, etc.)</li> <li>• Proactively identify areas that are at risk of landslides that could result in passage impediments, and implement regular monitoring to decrease mitigation response times to initiate mitigation activities</li> </ul>	-
<i>Climate change &amp; severe weather</i>	<ul style="list-style-type: none"> <li>• Freshwater and marine habitats shifting, and increasing frequency of severe weather events (e.g. droughts, floods, temperature extremes, etc.)</li> </ul>	<ul style="list-style-type: none"> <li>• Loss or degradation of habitat</li> <li>• Direct and indirect mortality</li> <li>• Exacerbate impacts from other threats</li> </ul>	<ul style="list-style-type: none"> <li>• Follow guidelines from the recent Paris Accord and International Panel on Climate Change reports</li> <li>• Proactively manage habitats and populations so that they are resilient and may adapt to future changes</li> </ul>	Adaptive management is required for all mitigation activities in the context of climate change and the increased frequency of severe weather events

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## **10.2. ELEMENT 17 INVENTORY OF ACTIVITIES THAT COULD INCREASE PRODUCTIVITY OR SURVIVAL**

*Element description: Develop an inventory of activities that could increase the productivity or survivorship parameters*

In Element 16 an inventory of activities was provided that could mitigate threats and limiting factors identified in Part 1 of the RPA, most of which could potentially increase productivity or survival of FRC. To avoid redundancy they are not listed again here (see Table 15 for list of threats, pathways of effect, and possible mitigation actions); however, as noted in the previous section, due to limited information on distribution and habitat use, a dynamic suite of threats and limiting factors, and increasing inter-annual variation in environmental conditions throughout the Chinook Salmon life cycle, we are currently unable to evaluate the effect of any mitigation activity on productivity or survival.

## **10.3. ELEMENT 18 ADVICE ON THE FEASIBILITY OF RESTORING LIMITING HABITAT**

*Element description: If current habitat supply may be insufficient to achieve recovery targets (see element 14), provide advice on the feasibility of restoring the habitat to higher values. Advice must be provided in the context of all available options for achieving abundance and distribution targets.*

As discussed in Section 8, there is currently insufficient information to evaluate whether the supply of suitable FRC habitat is currently limiting these 11 DUs from reaching their recovery targets (see section 8). However, many of the mitigation activities outlined in Element 16 (see Table 15) are likely to restore habitat properties to higher qualities. Further to this, Coker et al. (2010) has previously identified a suite of activities to mitigate threats in aquatic environments that would result in increased habitat quality. The vast number of confounding ecological processes that may change habitat supply and demand through time greatly limits our ability to provide advice on the feasibility or effectiveness of habitat restoration, and considerable research is needed to begin prioritizing habitat restoration for FRC. This has been identified as a major research need and is noted in Appendix H.

## **10.4. ELEMENT 19 REDUCTIONS IN MORTALITY RATE EXPECTED BY MITIGATION MEASURES AND INCREASE IN PRODUCTIVITY OR SURVIVAL ASSOCIATED WITH MEASURES IN ELEMENT 17**

*Element description: Estimate the reduction in mortality rate expected by each of the mitigation measures or alternatives in Element 16 and the increase in productivity or survivorship associated with each measure in Element 17.*

With the current state of information surrounding FRC we are unable to quantify reductions in mortality from the mitigation options discussed in Element 16 (section 10), nor their increase in productivity or survival (section 10.2). The interaction between changes in habitat quality and quantity to changes in life-history parameters represents a large knowledge gap for FRC, and has been identified as a future research need (Appendix H). These interactions are likely system specific and will require substantial resources and time to assess. Additionally, the success of mitigation activities would likely vary substantially for different types of projects and between individual projects of a similar nature. As more research is conducted on the effectiveness of mitigation measures it may be possible in the future to estimate reductions in mortality and ranges of productivity changes for certain projects.

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## 10.5. ELEMENT 20 PROJECTED EXPECTED POPULATION TRAJECTORY GIVEN MORTALITY RATES AND PRODUCTIVITIES ASSOCIATED WITH THE SPECIFIC MEASURES IDENTIFIED FOR EXPLORATION IN ELEMENT 19

*Element description: Project expected population trajectory (and uncertainties) over a scientifically reasonable time frame and to the time of reaching recovery targets, given mortality rates and productivities associated with the specific measures identified for exploration in element 19. Include those that provide as high a probability of survivorship and recovery as possible for biologically realistic parameter values.*

Neither mortality rates nor productivities were identified in Element 19 (Section 10.4), as it is not currently possible to identify mitigation specific productivity or mortality parameters. However, the results from varying productivity in Element 15 (Section 9) can be used to evaluate the likelihood of recovery under improved productivity, which would represent scenarios where there were successful mitigation measures.

## 11. ELEMENT 21: RECOMMENDED PARAMETER VALUES FOR FUTURE ASSESSMENTS

*Element description: Recommend parameter values for population productivity and starting mortality rates and, where necessary, specialized features of population models that would be required to allow exploration of additional scenarios as part of the assessment of economic, social, and cultural impacts in support of the listing process.*

The parameter estimates used in the modelling for DU2 (LFR-Harrison) are available in Appendix E. Many of these values are updated periodically as recent data becomes available and hence could become out-of-date. Additionally careful consideration of the model structure must be conducted before applying the harvest rates at age to another model as they could be applied erroneously. It is highly recommended to contact the lead author of this document before any exploratory analysis is done for economic, social, and cultural impacts based off of the modelling and parameter estimation described in Sections 3 and 5. There are considerable caveats and conditions associated with the modelling and parameter estimation done for this report which are briefly described in Section 6.

## 12. ELEMENT 22: ALLOWABLE HARM ASSESSMENT

*Element description: Evaluate maximum human-induced mortality and habitat destruction that the species can sustain without jeopardizing its survival or recovery.*

**Allowable harm** is: “Harm to the wildlife species that will not jeopardize its recovery or survival” (DFO 2014). It is important to note that **survival** represents a stable or increasing state where a species is not facing imminent extirpation, and **recovery** is a return to a state in which the population and distribution are within the normal range of variability (DFO 2014). Therefore, recovery is higher on the spectrum of population persistence than survival, and is more likely represented by the recovery target. The recommendations for allowable harm will first be discussed for DU2 (LFR-Harrison) and then separately for the remaining stream-type DUs.

### 12.1. DU2 (LFR-HARRISON)

Part 1 of the RPA identified numerous significant threats facing DU2 (LFR-Harrison), along with a continued downward trend in observed abundances. Many of the threats facing this DU are pervasive across all life stages and are challenging or costly to mitigate. These threats may have an ongoing effect on productivity and there is no indication that they are dissipating.

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Overall, Part 1 concluded that the risk of population decline for DU2 (LFR-Harrison) from all threats was High to Extreme (DFO 2020a).

There is uncertainty in current productivity as there is always a lag in the available data to complete the recruitment of a given brood year. Results from the time-varying Ricker model also indicated that there can be large changes in productivity within twelve years (3 generations). Element 15 provides results that may be used to interpolate what may happen to the probability of recovery at varying levels of harvest if productivity increases or decreases. Since the weight of evidence from the threats assessment indicated that DU2 (LFR\_Harrison) is at risk of declining from various threats, and that there is no current data to indicate that the productivity is increasing, the base case model scenario from Element 13 (Section 7.1) is presented below as a precautionary assessment of allowable harm.

The results from the forward projections from the model base case presented in this report align with the threats assessment from Part 1. The outcomes from the Chinook Projection Model indicate that at 2009-2015 Canadian catch year harvest rate levels, DU2 (LFR-Harrison) is likely to continue to decline in abundance and may not reach the recovery target in three generations if the productivity persists at the most recent available generation's average (2010-2013 brood years; base case scenario). Sixteen percent of simulations reached the recovery target in the base case scenario (see Element 13) and 48% of simulations met the survival target. Assuming that productivity remains at base case values, if Canadian salmon harvest rates are reduced to the 0 to 5% range (-100% to -80% change from the base case), the percent of simulations that reached the recovery target in 12 years was 34% (5% harvest) to 41% (0% harvest); however, the likelihood of reaching the survival target increased to either 64% (5% harvest) or 69% (0% harvest).

Considering the impact from all activities in the allowable harm assessment is vital because any additional impacts from the various threats not directly modelled will further hinder recovery.

**The results from both the modelling and the threats assessment suggest that under model base case productivity, human-induced mortality and other sources of harm identified in the threats assessment should be significantly reduced from base case mortality so as to not jeopardize recovery.** There is greater uncertainty in our understanding of allowable harm on habitat and the effects of harm to habitat on recovery outcomes could not be quantified. The impact of any activities on survival and recovery outcomes should be evaluated on a case-by-case basis, and considered in the broader context of cumulative impacts on recovery. Activities that are in support of the survival or recovery of the species that may result in mortalities but will have a net positive effect on the population should be allowed. As the productivity of this population has exhibited large fluctuations in the recent past, abundances and productivity should be continually monitored to determine if progress towards recovery is sufficient to warrant a re-assessment of allowable harm.

## 12.2. STREAM-TYPE DUs

Quantitative forward projections are not reliable nor robust for the remaining ten DUs due to the uncertainty that stems from the quality of the relative escapement data and lack of reliable exploitation estimates (See Section 7.2). Therefore, the allowable harm assessment is based on the threats assessment from Part 1, recent trends in relative abundance, the possible future trajectory of these populations based on qualitative assessments, and the results for DU2 (LFR-Harrison). The results of the threats workshop from Part 1, indicated that all DUs were considered to be at High or Extreme risk, due to the severity and number of threats that each of the DUs are facing (DFO 2020a). Alleviating many of these threats will be difficult given the widespread nature of these threats (summarized in Table 2), especially as many are

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exacerbated by climate change, posing a risk of extinction for these DUs within the next three generations.

There is considerable uncertainty about the future trajectory of these populations, but based on the threats assessment and the qualitative description in Section 7.2, these populations are at greater risk, and the potential for recovery is less likely than was the case for DU2. It is likely that many of the assessed threats pose a more serious risk to these stream-type DUs than compared to DU2 (LFR-Harrison), as stream-type populations rely on freshwater habitat for more of their life cycle than ocean-type stocks. Stream-type DUs have experienced more severe declines in relative abundance compared to DU2 (LFR-Harrison) and many are currently extremely small (Figure 1 and Figure 2). Based on this information, and the allowable harm assessment for DU2 (LFR-Harrison), a precautionary approach is suggested unless sufficient increases in generational average abundances and trends in abundances are confirmed due to mitigation measures or changes in natural conditions. **Harm is likely to continue to jeopardize recovery. Therefore, to promote the survival and recovery of these DUs, it is advised that all future and ongoing human-induced harm should be prevented so as not to jeopardize recovery.** As with DU2, it is important to note that some activities in support of survival or recovery may result in mortalities but will have a net positive effect on the population and should be allowed.

**For DUs 7, 8 and 14, there is additional concern due to the limited area of the spawning habitat and small population sizes.**

**For DUs 9, 10, and 11, additional concern due to the increased threat risk from the Big Bar landslide will remain until the impacts from the slide are alleviated.**

### 13. RESEARCH NEEDS

There are significant knowledge gaps surrounding FRC, particularly for the Spring and Summer 5<sub>2</sub> populations. The following is a brief summary of the main sources of uncertainty specific to the second half of the RPA process, with a focus on recovery targets and projections:

- The largest knowledge gap for this process was a lack of productivity and S-R data for these DUs, with the exception of DU2 (LFR-Harrison). The absence of S-R data made the evaluation of potential changes in productivity and the identification of S-R parameters extremely problematic and ultimately prevented the ability to provide meaningful forward projections for ten DUs. The lack of S-R data, typically provided through CWT indicator stock programs, also prevented direct estimates for the recovery targets, which instead relied on habitat-based methods (Parken et al. 2006).
- The reliance on relative abundance data for 10 of 11 DUs contributes significant uncertainty. It will likely not be possible to provide absolute abundance estimates for all systems, but additional resources should be added to improve estimates that are particularly uncertain. DUs 4 (LFR-Upper Pitt), 5 (LFR-Summer), and 16 (NTh-Spring) should be priorities, as there is only moderate to high quality relative abundance data from one or two tributaries in each DU, which may not capture DU level trends.
- The impacts of fisheries (both targeted and non-targeted at Chinook Salmon) are currently limited or unknown for the majority of DUs. DU2 (LFR-Harrison) is the only population with a long-standing time series of CWT data. Harvest rate data is required for both the Summer and Spring 5<sub>2</sub> MUs, to be able to accurately assess the fishing pressure facing these DUs and to properly assess if reduction targets are being achieved (DFO 2019b; Dobson et al. 2020).

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- Management or implementation error is an important feature in evaluating the risk associated with setting a specific target and the ability to achieve those targets. Clear management targets, with a framework for assessing success and error, would be highly beneficial to provide better information for future assessments.

Without additional information about S-R data, harvest rates estimates, and more accurate abundance estimates it will be difficult to ever accurately assess stock status or set meaningful recovery targets for the data-limited DUs. A detailed list of research needs from both Part 1 and 2 is provided in Appendix H. The list provided is extensive and would require substantial time and resources to accomplish. Additional work with a broader group of contributors is required to carry out the cost/benefit analysis required to rank research priorities.

## 14. CONCLUDING REMARKS

The results of this assessment are similar to the outcomes of recent RPAs on other salmonid species (DFO 2018, 2019c, 2019a, 2020c, 2020b). The threats and mitigation measures required to recover these populations are not unique to these Fraser Chinook DUs, but are common among the various declining salmonid populations. It will be critical to ensure that efforts are appropriately coordinated through effective governance to successfully mitigate the cumulative impacts of these diverse threats. Additional research will be essential for improved prediction of outcomes, and to develop approaches to mitigate the impacts of the threats and limiting factors, under a more variable and uncertain climate.

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## APPENDIX A. EXCERPT FROM WILD SALMON POLICY ASSESSMENT: BENCHMARK CALCULATION METHODS

Provided below is an excerpt from the Research Document that is pending from the 2014 Assessment of Southern British Columbia Chinook Salmon Conservation Units, Benchmarks and Status. The excerpt provided from the report describes the abundance status metric (2.1.5. **WSP status metrics**) and the benchmark calculations (2.1.6. **Benchmarks for the Relative Abundance metric**). These exact methods were used to calculate the abundance benchmarks using the updated habitat model provided in the RPA research document.

The anticipated citation for the forthcoming document is as follows:

Brown, G., Thiess, M.E., Pestal, G., Holt, C.A and Patten, B. 20xx. Integrated Biological Status Assessments under the Wild Salmon Policy Using Standardized Metrics and Expert Judgement: Southern British Columbia Chinook Salmon (*Oncorhynchus tshawytscha*) Conservation Units. DFO Can. Sci. Advis. Sec. Res. Doc. 20xx/nnn. vi + xx p.

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### 2.1.5. WSP status metrics

#### ABUNDANCES

The (geometric) average spawner abundance in the most recent generation was compared against the lower benchmark,  $S_{gen}$ , and an upper benchmark, 85% of  $S_{MSY}$ , where  $S_{gen}$  is defined as the spawner abundance that will result in recovery to spawner abundances at maximum sustainable yield ( $S_{MSY}$ ) within one generation under equilibrium conditions (Holt et al. 2009). The upper benchmark (i.e., 85% of  $S_{MSY}$ ) is a slight deviation from that proposed by Holt et al. 2009 (i.e., 80% of  $S_{MSY}$ ), and was adopted to be consistent with an agreed benchmark for Chinook salmon assessment specified in the Pacific Salmon Treaty (PST 2008). Benchmarks (and 90% confidence intervals) were obtained through published stock-recruit parameters where available (CK-01: Okanagan; CK-03: Harrison; CK-22: Cowichan), or otherwise estimated from habitat models of freshwater capacity for rivers where Chinook salmon spawn (Parken et al. 2006). See Section 2.1.6 for further details on this calculation. In short,  $S_{gen}$  is estimated by solving the following equation iteratively:

$$(3) S_{MSY} = \alpha S_{gen} e^{-\beta S_{gen}}$$

### 2.1.6. Benchmarks for the Relative Abundance metric

For the majority of southern BC Chinook salmon CUs, it is not possible to calculate traditional stock-recruit parameters, due to insufficient data. For these cases, a habitat-based approach has been developed to provide comparable estimates of productivity and capacity (Parken et al. 2006), and these can then be used to provide upper and lower abundance benchmarks (as outlined in the previous section).

The habitat model predicts  $S_{MSY}$  and  $S_{REP}$ , spawner abundances at maximum sustainable yield or replacement, and the associated confidence levels from watershed characteristics (Parken et al. 2006; updated by C. Parken, DFO, unpublished data). Benchmarks were then estimated from  $S_{MSY}$  and  $S_{REP}$  using the Ricker model:

$$(4) R = \alpha S e^{-\beta \cdot S + \omega}, \omega \sim (N, \sigma_{\omega}^2)$$

where  $\alpha$  is the productivity parameter,  $\beta$  is the capacity parameter,  $\omega$  is a stochastic term, and  $\sigma_{\omega}^2$  is the variance of the recruitment anomalies. Using first principles (Ricker 1975) and an approximation for  $S_{MSY}$  (Hilborn and Walters 1992), Ricker  $\alpha$  and  $\beta$  parameters could then be estimated as:

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$$(5) \log_e(\alpha) = \frac{0.5 - \frac{S_{REP}}{S_{MSY}}}{0.07}, \text{ and } \beta = \frac{\log_e(\alpha)}{S_{REP}}$$

Finally,  $S_{gen}$  was estimated by solving Equation (3) iteratively as outlined in the previous section.

For CUs with spawning sites across multiple watersheds, an extra step was required to arrive at CU-level habitat estimates of  $S_{MSY}$  and  $S_{REP}$ . Prior to estimating  $S_{gen}$  as outlined above, joint distributions of  $S_{MSY}$  and  $S_{REP}$  for the CU were calculated from the individual estimates for all watersheds contributing to the reported escapement time series (i.e. this meant including habitat-based estimates for all persistent, aggregated or extirpated census sites, but not from data deficient or deleted census sites). For each CU, the following non-parametric procedure was used:

1. Generate 10,000 samples of  $S_{MSY}$  for each of the  $n$  contributing census sites in the CU, where  $S_{MSY,i,j} \sim \text{lognormal}(\text{median}(S_{MSY,i}), \text{std error}(S_{MSY,i}))$ ,  $i = 1, \dots, n; j = 1, \dots, 10\,000$ . The median and standard errors of  $S_{MSY}$  for each contributing census site were provided by C. Parken (unpublished data).
2. Estimate the  $S_{MSY}$  for the CU ( $S_{MSY,CU}$ ) by summing across the  $n$  contributing census sites'  $S_{MSY}$  estimates for each of the 10,000 random samples (thus generating 10,000 samples of  $S_{MSY,CU}$ ) and calculating the mean and standard deviation of the resulting distribution.

$$S_{MSY,CU_j} = \sum_{i=1}^n S_{MSY,i} \text{ where } i = 1, \dots, n; j = 1, \dots, 10\,000$$

$$S_{MSY,CU} \sim \text{lognormal}(\text{median}(S_{MSY,CU_j}), \text{std error}(S_{MSY,CU_j}))$$

3.  $S_{REP}$  was identified as a proportion of  $S_{MSY}$ , so in order to maintain this relationship, the point estimate of  $S_{REP}$  was determined dependent on the ratio of median  $S_{REP}$  to median  $S_{MSY}$  for each contributing census site, multiplied by the random sample of  $S_{MSY}$ 's:

$$S_{REP.ratio_i} = \frac{\text{med}(S_{REP,i})}{\text{median}(S_{MSY,i})} \text{ where } i = 1, \dots, n$$

$$S_{REP,i,j} = S_{REP.ratio_i} * S_{MSY,i,j} \text{ where } i = 1, \dots, n \text{ and } j = 1, \dots, 10\,000$$

In a manner similar to Step 3,  $S_{REP}$  for the CU ( $S_{REP,CU}$ ) was approximated by

$$S_{REP,CU} \sim \text{lognormal}(S_{REP,CU_j}, \text{std error}(S_{REP,CU_j})), j = 1, \dots, 10\,000$$

$$\text{where } S_{REP,CU_j} = \sum_{i=1}^n S_{REP,i}, i = 1, \dots, n$$

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## APPENDIX B. CHINOOK PROJECTION MODEL EQUATIONS

Kendra Holt & Brooke Davis

February 5, 2021

With additions from Lauren Weir

### B.1. OVERVIEW

The notation used to describe the Chinook Projection Model is presented in Table B1, while model equations are presented in Table B2 and described below.

The Chinook Projection Model simulates trajectories of future abundance for individual stocks under specified scenarios about future exploitation rates and biological processes while incorporating stochasticity. Stochastic projections can be parameterized to represent uncertainty in recruitment, maturity, productivity and base period exploitation rates<sup>1</sup>. Catch is taken by one or more fisheries ( $f$ ), with fisheries operating as either pre-terminal or terminal fisheries. A flow chart of annual fishery and maturation timing in relation to abundance and catch is shown in Figure B1. Each year in the simulation routine starts at the end of the “over-winter” period.

Within our model notation, superscripts are used to differentiate names of related variables while subscripts are used to show indices. For example  $N_{y,a}$  and  $N_{y,a}^{mat}$  are two different variables that both denote fish abundance.  $N_{y,a}$  is the abundance of fish at the beginning of the year with indices  $y$  (year) and age ( $a$ ), and  $N_{y,a}^{mat}$  is the abundance of mature fish after pre-terminal fisheries that will be exposed to terminal fisheries with indices year and age (Figure B1).

### B.2. POPULATION DYNAMICS

A Ricker stock recruitment model was used to represent recruitment dynamics (Eq. 1). Process error in the recruitment function,  $v$ , was added as multiplicative, lognormal error. A lognormal bias-correction factor of  $-\sigma_v^2/2$  is applied to projected recruitment in Equation 1 based on the assumption that stock-recruitment parameters used to parameterize the model have been corrected to estimate the expected (mean) recruitment. In this case, the bias correction is necessary because the expected value of  $e_v^\sigma$  is  $e^{\sigma_v^2/2}$  rather than zero when  $v$  is normally distributed. For each Monte Carlo simulation trial used in projections, stock recruitment parameters  $\alpha$ ,  $\beta$ , and  $\sigma_v$ , were sampled from an estimated joint posterior from a Bayesian model fitting procedure (Appendix F). For each posterior sample, the productivity parameter ( $\alpha$ ) was based on the average of the last generation (4 years) of year-specific  $\alpha$  estimates from the time-varying productivity model (which are denoted  $\alpha_t$  in Appendix F, where  $t$  = brood year 2010 - 2013). An annual scalar parameter was then applied to sampled  $\alpha$  values to introduce trends in productivity when required for projection scenarios. The resulting annual productivity parameter used in Equation 1 is denoted  $\alpha_y$  accordingly.

The expected number of adult recruits from fish that spawned in year  $y$ ,  $R_y$ , was calculated as a function of the sum of age 3 to age 5 fish returning to the spawning ground in year  $y$  ( $S_y$ , Eq. 2). While a small proportion of age 2 fish do return to spawn in some ocean-type populations (i.e.,

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<sup>1</sup> Base period exploitation rates are the exploitation rates representing a particular period as specified by the user

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jacks that have spent one year or less in the ocean), these fish are assumed to not effectively contribute to spawning abundance.

The spawner equivalence factor  $Q_y$  described in Equation 3 is used to adjust the expected recruits calculated from Equation 1 to ocean age-1 abundance. Back-calculating from recruitment to ocean age-1 abundance in this way is necessary in order to represent pre-terminal Chinook fisheries that capture immature fish.  $Q_y$  is the product of annual survival and maturation rate schedules,  $surv$  and  $m$ , respectively (Equation 3). While the annual survival rate schedule is constant among years, maturation rate schedules vary among years as a function of the random distribution described in Equation 7. Maturation rates in Equation 3 are applied from “future year”  $y+i$  (where  $i$  is age-of-maturity) to align the maturation rates of fish from brood year  $y$  with their eventual maturation in return year  $y+i$ . The parameter  $k$  in Equation 3 is used to adjust for ocean-type versus stream-type life histories;  $k = 0$  for ocean-type stocks that migrate to the ocean as smolts within a few months of emerging from eggs and  $k = 1$  for stream-type stocks that spend an entire year in freshwater before migrating as smolts. For ocean-type stocks, the actual (total) age  $a$  is equal to ocean age, while for stream-type stocks, the total age will be 1 year greater than ocean age due to their first year spent in freshwater. The index  $a$  in all model equations refers to total age.

The at-sea abundance of age-1 fish at the start of the year,  $N_{y,a=1}$ , is calculated using Equation 4. For ocean-type stocks,  $N_{y,a=1}$  is calculated by dividing the expected adult recruits produced by the spawner-recruit function by the ocean age-1 to adult survival rate for brood year  $y$ ,  $Q_y$ . For stream-type stocks,  $N_{y,a=1}$  is set to zero since these fish will still be in freshwater habitats.

The at-sea abundance of age-2 fish at the start of the year,  $N_{y,a=2}$ , is calculated using Equation 5. Age-1 fish become Age-2 fish at the beginning of the next year, which is set to occur in the spring for the current parameterization of the model. As age-1 fish are generally too small to be captured in fisheries,  $N_{y,a=2}$  for ocean-type stocks is simply a function of the cohort abundance from the previous year and over-winter survival. For stream-type stocks just entering the ocean,  $N_{y,a=2}$  is calculated by dividing the expected adult recruits produced by the spawner-recruit function by the ocean age-1 to adult survival rate,  $Q$ .

For ages 3 and higher, at-sea abundance at the start of the year,  $N_{y,a}$ , is calculated by applying over-winter survival,  $surv_i$  (where,  $i = a-k$  is the ocean age after winter) to the remainder of the at-sea abundance from age class  $a-1$  in year  $y-1$  that did not get removed by pre-terminal fisheries or undergo maturation in the previous year (Equation 6). Note that in Equation 6,  $C^{PT}$  represents pre-terminal fishing mortalities in number of fish while  $N^{mat}$  represents the number of fish maturing.

Annual maturation is applied starting at age 2 (Equation 7). The number of fish maturing at age  $a$ ,  $N_{y,a}^{mat}$  is calculated as a function of age-specific maturation rate,  $m_{y,a}$ , and the at-sea abundance remaining after total pre-terminal mortalities have been removed from the population. Stochasticity in  $m_{y,a}$  is incorporated using a beta distribution, which is naturally bound between 0 and 1 (Equation 7). For the oldest age class, where the maturation rate is always 1, no stochasticity was incorporated.

Annual spawning escapement,  $E_{y,a}$ , is calculated by subtracting total terminal fishery mortalities,  $C^{Term}$ , from the number of fish maturing in each year (Equation 8).

### B.3. FISHERY DYNAMICS

Two types of fishery-management regimes are applied:

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1. Pre-terminal exploitation rate-based fisheries
  2. Terminal harvest rate-based fisheries

These fisheries are described below. When parameterizing the model for our current case studies, total fishery mortality (including release and drop-off mortality) is represented for all fisheries. Mortality from Chinook non-retention fisheries (i.e., bycatch) is not explicitly represented at this time.

Note that there is no feedback control between annual abundance and harvest decisions in the current version of this model. Both pre-terminal and terminal fisheries apply exploitation rates (or harvest rates) that are independent of annual stock status.

### **B.3.1. Pre-Terminal Exploitation Rate-Based Fisheries**

For pre-terminal fisheries, total fishery mortalities in units of numbers of fish ( $C_{y,f,a}^{PT}$ ) are calculated by applying year-specific fishery scalars,  $\delta_{y,f}^{PT}$ , to a base period exploitation rate. The base period exploitation rates, denoted by  $ER_{y,f,a}$ , are drawn each year from beta distributions, with the shape parameters determined by the  $\mu_{f,a}^{ER}$  and  $\sigma_{f,a}^{ER}$  of the base period exploitation rates specified for each fishery and age (Equation 9). The exploitation rates are selected from distributions to represent the inter annual variation in exploitation rates observed during the base period. When selecting a base period for the exploitation rates, it is important to consider that this approach is only appropriate for base periods where there was no major change in the management regime.

For pre-terminal fisheries, annual age-specific exploitation rates are modelled as a function of age-specific base period exploitation rates,  $ER_{y,f,a}$ , and annual fishery-specific scalars,  $\delta_{y,f}^{PT}$ . This parameterization allows scenarios about relative increases or decreases in exploitation rates to be easily specified (e.g., a 50% reduction from the base period level). Total pre-terminal fishery mortalities in units of numbers of fish ( $C_{y,f,a}^{PT}$ ; which includes both catch and incidental mortalities) is then calculated by applying  $ER_{y,f,a} * \delta_{y,f}^{PT}$  to  $N_{y,a}$ .

### **B.3.2. Terminal Fisheries**

Terminal fisheries are represented in a similar way; total fishery mortalities in numbers of fish ( $C_{y,f,a}^{Term}$ ) are calculated by applying year-specific fishery scalars,  $\delta_{y,f}^{Term}$ , to base period harvest rates (Equation 10). The main difference between pre-terminal and terminal fisheries is that terminal fisheries are implemented using harvest rates that are applied to the maturing population, not the entire non-mature cohort.

## **B.4. MODEL INITIALIZATION**

The model has two modes of initialization that use: (i) initial at-sea-abundance levels ( $N_{y=1,a}$ ) as model inputs, or (ii) a time series of escapement values. The model is currently set up to be initialized using the second approach. As many years as ages being modelled are required to fully initialize the model (in the case of DU2 (LFR-Harrison) 5 years; 2015-2019. Model results are not used for inference within this initialization period. In the first year (2015) the  $N$  array is initialized as an array of zeros, and escapement is input in the maturation period. For each subsequent initialization year escapement are the input values (rather than calculated based on maturation rates and abundance). Catches from these years are not counted towards model performance as not all age classes will have abundance in all years during initialization.

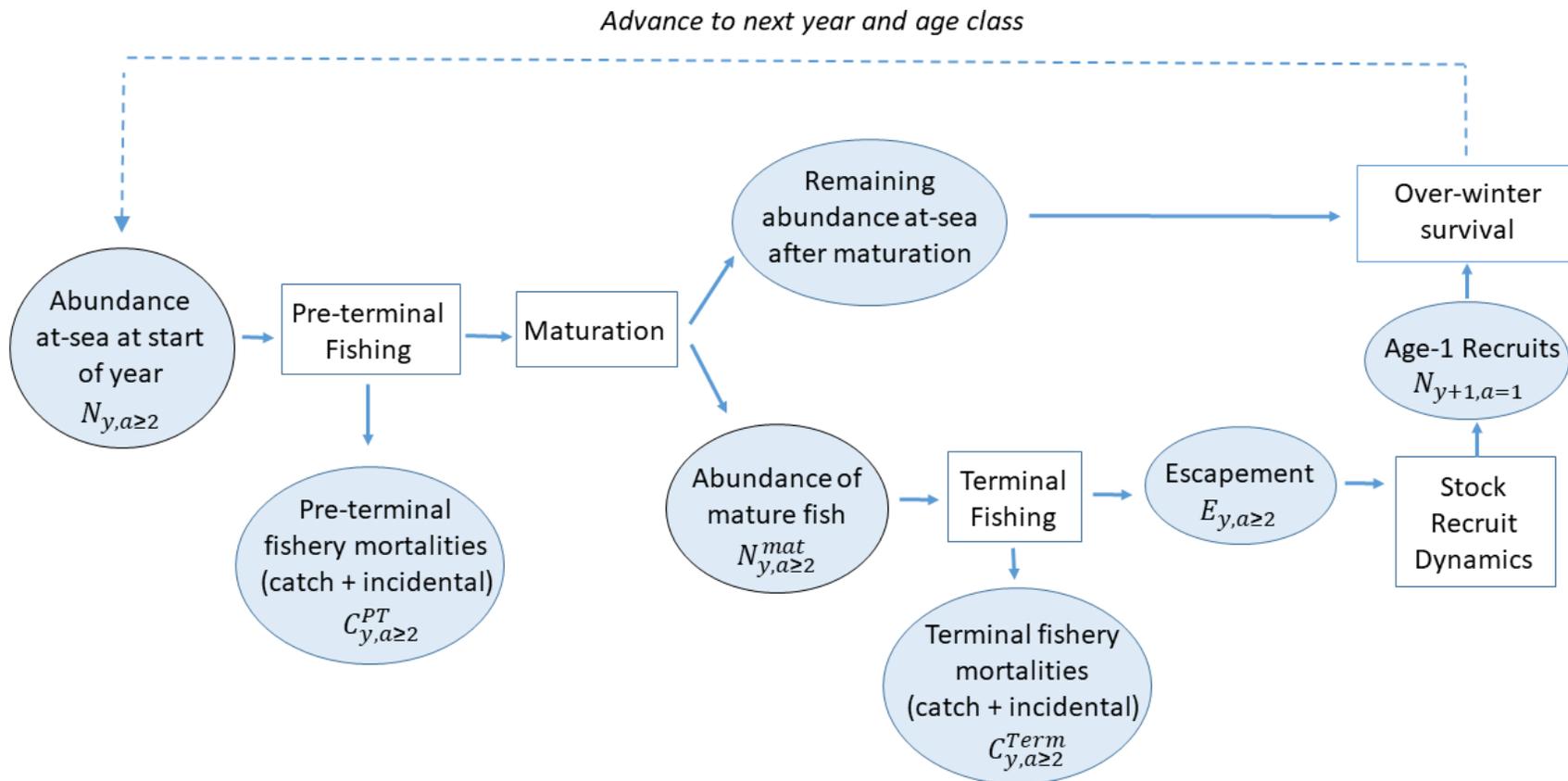


Figure B1 - An overview of annual fishery and maturation timing in relation to abundance and catch in the model for an ocean-type population such as DU2 (LFR-Harrison). Note that while fish age  $\geq 2$  are included in escapement, only escaped fish age  $\geq 3$  contribute to spawner abundance in the stock recruit relationship.

Table B1 - Definition of model notation.

Symbol	Description, with fixed values where appropriate
<b>Indices (all subscripts)</b>	
$y$	Year
$f$	Fishery
$a$	Age
<b>Index ranges</b>	
$NY$	Number of years
$NAges$	Number of ages
<b>Parameters</b>	
$F^{PT}$	List of pre-terminal fisheries
$F^{Term}$	List of terminal fisheries
$k$	Life history type indicator ( $k = 0$ if ocean-type and $k = 1$ if stream-type)
$\alpha_y$	Ricker $\alpha$ coefficient for projection year $y$ . For each Monte Carlo simulation replicate, the base $\alpha$ value against which annual changes in $\alpha_y$ are scaled is randomly sampled from a joint posterior distribution with $\beta$ and $\sigma$ .
$\beta$	Ricker $\beta$ coefficient. For each Monte Carlo simulation replicate, $\beta$ is randomly sampled from a joint posterior distribution with $\alpha$ and $\sigma$ .
$\sigma_v$	Standard deviation of recruitment error. For the current parameterization of the Chinook Projection Model, in which stock recruitment parameters were estimated using a model with time-varying productivity, process error associated with annual deviations in $\alpha$ were removed from the estimated total standard deviation according to $\sigma_v = \sqrt{\rho} * \sigma_{Total}$ . See Appendix F for details of the time-varying productivity model. For each Monte Carlo simulation replicate, $\sigma_{Total}$ and $\rho$ are randomly sampled from a joint posterior distribution with $\alpha$ and $\beta$ .
$\mu_a^{mat}$	Average proportion of the stock that matures at age $a$
$\sigma_a^{mat}$	Standard deviation of the proportion of the stock that matures at age $a$
$\mu_{f,a}^{ER}$	Average total pre-terminal exploitation rate at age $a$ and fishery $f$ during the base period
$\sigma_{f,a}^{ER}$	Standard deviation of the total pre-terminal exploitation rate at age $a$ and fishery $f$ during the base period
$\mu_{f,a}^{HR}$	Average terminal harvest rate at age $a$ and fishery $f$ during the base period
$\sigma_{f,a}^{HR}$	Standard deviation of the terminal harvest rate at age $a$ and fishery $f$ during the base period
$surv_i$	Proportion of fish that survive from ocean age $i-1$ to ocean age $i$ (i.e., overwinter survival); $surv_{2:5} = 0.6, 0.7, 0.8, 0.9$ . This means that for ocean-type stocks $surv_2$ will represent survival from total age 1 to 2,

Symbol	Description, with fixed values where appropriate
	whereas for stream-type stocks $surv_2$ represents survival from total age 2 to 3.
$Q_y$	Proportion of offspring from brood year $y$ that survive from ocean age 1 to maturation
<b>State Variables</b>	
$N_{y,a}$	Abundance of fish of age $a$ that are present at the beginning of year $y$
$m_{y,a}$	Proportion of the stock that matures at age $a$ , in year $y$
$N_{y,a}^{mat}$	Number of fish maturing in year $y$ that are of age $a$
$E_{y,a}$	Escapement abundance for age $a$ in year $y$
$S_y$	Spawner abundance in year $y$ (escapement age-3 and older)
$R_y$	Adult recruits in year $y$
$C_{y,f,a}^{PT}$	Total pre-terminal fishery mortalities (includes catch and incidental mortalities) of age $a$ fish in year $y$ from fishery $f$
$C_{y,f,a}^{Term}$	Total terminal fishery mortalities (includes catch and incidental mortalities) of age $a$ fish in year $y$ from fishery $f$
<b>Fishery Controls</b>	
$ER_{y,f,a}$	Total exploitation rate (includes landed catch and incidental mortality) in year $y$ of age $a$ fish for fishery $f$ in the base time period, where $ER_{y,f,a} \sim Beta(\alpha_a^{ER}, \beta_a^{ER})$ . Annual exploitation rates are scaled relative to this base period.
$HR_{y,f,a}$	Total terminal fishery harvest rate (includes landed catch and incidental mortality) in year $y$ of age $a$ fish for fishery $f$ in the base time period, where $HR_{y,f,a} \sim Beta(\alpha_a^{HR}, \beta_a^{HR})$ . Annual terminal harvest rates are scaled relative to this base period.
$\delta_{y,f}^{PT}$	Exploitation rate scalar for pre-terminal fisheries representing the ratio of the current years exploitation rate to base exploitation rate for fishery $f$ in year $y$
$\delta_{y,f}^{Term}$	Harvest rate scalar for terminal fisheries representing the ratio of current years terminal harvest rate to base harvest rate for fishery $f$ in year $y$

Table B2 - Chinook Projection model equations.

Eq. #	Equation
<b>Population Dynamics</b>	
Recruitment:	
Eq. 1	$R_y = \alpha_y S_y e^{-\beta S_y} e^{v - \sigma_v^2/2}$ where $v \sim Normal(0, \sigma^v)$
Eq. 2	$S_y = \sum_{a=3}^{NAges} E_{y,a}$

Eq. #	Equation
Eq. 3	$Q_y = \sum_{i=(2+k)}^{NAges} q_{y,i} \times m_{y+i,i}$ <p>where,</p> $q_{y,(2+k)} = surv_2$ $q_{y,i} = q_{y,i-1}(1 - m_{y+i,i-1})surv_{i-k} \text{ for } i \in (3+k):NAges$ <p>and,</p> $k = \begin{cases} 0 & \text{if stock is ocean type} \\ 1 & \text{if stock is stream type} \end{cases}$
<b>Annual survival:</b>	
Eq. 4	<p>For <math>a = 1</math>:</p> $N_{y,a=1} = \begin{cases} 0 & \text{if stock is stream – type} \\ \frac{R_{y-1}}{Q} & \text{if stock is ocean – type} \end{cases}$
Eq. 5	<p>For <math>a = 2</math>:</p> $N_{y,a=2} = \begin{cases} \frac{R_{y-2}}{Q} & \text{if stock is stream – type} \\ N_{y-1,a=1}surv_2 & \text{if stock is ocean – type} \end{cases}$
Eq. 6	<p>For <math>a \geq 3</math>:</p> $N_{y,a} = surv_{a-k} \left( N_{y-1,a} - \sum_{f \in F^{PT}} C_{y-1,f,a}^{PT} - N_{y-1,a}^{mat} \right)$
<b>Maturation and Escapement:</b>	
Eq. 7	$N_{y,a}^{mat} = m_{y,a} \left( N_{y,a} - \sum_f^{F^{PT}} C_{y,f,a}^{PT} \right)$ <p>where <math>m_{y,a} \sim \text{Beta}(\alpha_a^{mat}, \beta_a^{mat})</math> and</p> $\alpha_a^{mat} = \mu_a^{mat2} \left( \frac{1 - \mu_a^{mat}}{\sigma_a^{mat2}} - \frac{1}{\mu_a^{mat}} \right)$ $\beta_a^{mat} = \alpha_a^{mat} \left( \frac{1}{\mu_a^{mat}} - 1 \right)$
Eq. 8	$E_{y,a} = N_{y,a}^{mat} - \sum_{f \in F^{Term}} C_{y,f,a}^{Term}$
<b>Fishery Dynamics</b>	
<b>Pre-Terminal Exploitation Rate-based Fisheries</b>	
Eq. 9	$C_{y,f,a}^{PT} = N_{y,a} \times ER_{y,f,a} \times \delta_{y,f}^{PT}$ <p>where <math>ER_{y,f,a} \sim \text{Beta}(\alpha_a^{ER}, \beta_a^{ER})</math></p>

Eq. #	Equation
	<p style="text-align: center;">and</p> $\alpha_a^{ER} = \mu_{f,a}^{ER2} \left( \frac{1 - \mu_{f,a}^{ER}}{\sigma_{f,a}^{ER2}} - \frac{1}{\mu_{f,a}^{ER}} \right)$ $\beta_a^{ER} = \alpha_a^{ER} \left( \frac{1}{\mu_{f,a}^{ER}} - 1 \right)$
<i>Terminal Fisheries</i>	
Eq. 10	$C_{y,f,a}^{Term} = N_{y,a}^{mat} \times HR_{y,f,a} \times \delta_{y,f}^{Term}$ <p style="text-align: center;">where <math>HR_{y,f,a} \sim \text{Beta}(\alpha_a^{HR}, \beta_a^{HR})</math></p> <p style="text-align: center;">and</p> $\alpha_a^{HR} = \mu_{f,a}^{HR2} \left( \frac{1 - \mu_{f,a}^{HR}}{\sigma_{f,a}^{HR2}} - \frac{1}{\mu_{f,a}^{HR}} \right)$ $\beta_a^{HR} = \alpha_a^{HR} \left( \frac{1}{\mu_{f,a}^{HR}} - 1 \right)$

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## APPENDIX C. DATA-LIMITED ISSUES PART 1

As stated in the main document, with the exception of DU2 (LFR-Harrison), escapement time series of relative abundance are the only recent abundance information available directly for each DU. For DUs 4 (LFR-Upper Pitt), 5 (LFR-Summer), 7 (MFR-Nahatlatch), 14 (STh-Bessette), and 16 (NTh-Spring), the escapement time series are subject to large uncertainties, and are considered unsuitable for initializing the Chinook Projection Model. Additionally, in some cases, there were concerns about using the habitat-based (Parken et al 2006) S-R parameters in combination with the available relative abundance spawner estimates, since there may be a mismatch in scale (relative spawner abundance doesn't represent the entire population). The issues for each DU are summarized below.

### C.1. DU4 (LFR-UPPER PITT)

For this DU, a major concern is that there are only relative abundance data for one tributary stream from the much larger overall DU area. There is no evidence to support whether the one system regularly surveyed, Blue Creek, is a good indicator for overall abundance at the DU level. This system was selected for spawner surveys because it is relatively clear compared to the other tributaries in the Upper Pitt watershed, so visual counts are possible. Additionally, Chinook counts are opportunistically done by DFO staff while conducting sockeye surveys in the area, so they are likely inconsistent with regard to timing and length of stream surveyed annually, and hence may be an unreliable estimate of trends. Surveys were not conducted in every year, and data from one of the missing years would be required for the initialization of the Chinook Projection Model. Given the lack of available information, it was not possible to initialize the Chinook Projection Model, thus forward projections were not possible for this DU.

### C.2. DU5 (LFR-SUMMER)

As with DU4 (LFR-Upper Pitt), there are only sampling data available from a single system in the DU and the time series is incomplete. Big Silver Creek is the only system in the DU that had data of sufficient quality to be used in the trend assessment. Even though the visual spawner counts are considered to be of moderate quality, there remains some concern about the timing of the surveys in some years, as these counts were also conducted opportunistically by DFO stock assessment staff who were surveying sockeye escapements. The Lillooet River spawning population is included in this DU, but there is very limited information about Chinook spawning in this river (it is highly glacially turbid) and there are no escapement estimates. It is unknown if the trend observed at Big Silver Creek represents the DU level trend in escapement.

Due to the incomplete escapement time series, it was not possible to initiate the Chinook Projection Model. Estimates of cohort sizes are possible to attain from the CTC Chinook Model, however, those projections wouldn't represent the DU as a whole, only the Big Silver Creek spawning population. For these reasons, it was determined that providing a qualitative description of recent and likely future trends based on available information would be more informative.

### C.3. DU7 (MFR-NAHATLATCH)

There are concerns with the quality of the available relative abundance estimates for DU7 (MFR-Nahatlatch). The counts at Nahatlatch may significantly underestimate total abundance, and may represent an inconsistent fraction of spawners across years. The yearly estimates are a combination of counts from two different visual methods, from a boat floating down the river and an over-flight, which perhaps should not be combined. Due to the likelihood of the counts

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being under estimates, it was deemed inappropriate to use the relative abundance data with the habitat-based<sup>1</sup> estimates of the S-R parameters.

#### **C.4. DU14 (STH-BESSETTE)**

There are several issues with data quality and availability for DU14 (STh-Bessette) which prevent generating meaningful forward projections. The available escapement data are likely overestimates for some years, as a variable proportion of Summer 4.1 spawners will move into Bessette creek to spawn, and may erroneously be counted as part of DU14 escapement at the end of the survey period. The habitat in this DU is severely degraded, and there were concerns that the habitat model<sup>1</sup> S-R parameter estimates would not appropriately represent the true recruitment dynamics of the population. Lastly, the CWT indicator used to represent Bessette is the Nicola River, which may not be a good indicator for this DU, as there is no evidence that these two populations have similar ocean distributions. With no good alternate indicator, and concern about the applicability of habitat-based estimates<sup>1</sup>, it was decided not to provide forward projections for this DU.

#### **C.5. DU16 (NTH-SPRING)**

Only two systems in DU16 (NTh-Spring) are surveyed annually for escapement, Finn Creek and Blue River. Many of the tributary systems within the DU are glacially turbid, and are unsuitable for visual survey techniques. Counts at Finn Creek may reflect how attractive conditions are within Finn Creek for spawners as opposed to the North Thompson mainstem in a particular year, rather than the abundance of DU16 (NTh-Spring) in total. There are expansive spawning dune habitats in the vicinity of the confluence of Finn Creek and the North Thompson River, but turbidity during the spawning period prevents visual counts. The amount of suitable spawning habitat available within Finn Creek is a small fraction of that available in the dune habitat in the mainstem, and fish spawning in the dunes would be less vulnerable to predation. The annual counts on Blue River have been undertaken by raft surveys in most years, with the recent addition of helicopter overflights. The raft surveys only cover the lower reach of the river, while the helicopter counts are believed to survey all accessible habitat. There is ongoing work to develop a calibration factor between the floats and the helicopter counts, but the ratio of fish in the lower reach versus above has been inconsistent to date, thus we are uncertain of the value of the previous counts in the lower Blue River as an index of escapement for the entire reach, as well as to DU16. As with DUs 4 and 5, these two systems represent only a small fraction of the entire DU area, and many other systems are not surveyed due to the glacially turbid nature. There is insufficient information to produce a scalar to expand the relative abundance counts to the whole DU, so the habitat model<sup>1</sup> productivity estimates are inappropriate to use with this time series, preventing forward projections.

#### **C.6. CONCLUSION**

For the DU specific reasons mentioned above, it was determined that forward projections should not be completed. While there are still issues with the time series of relative abundance and the application of the Habitat Model S-R parameters for the remaining 5 DUs with data limitations, those escapement time series provided more accurate estimates of abundance that are likely to be closer to representing absolute abundance. Efforts were continued to produce representative parameter estimates to allow forward projections for DUs 8, 9, 10, 11 and 17.

#### **C.7. REFERENCES**

Parken, C.K., McNicol, R.E., and Irvine, J.R. 2006. [Habitat-based methods to estimate escapement goals for data limited Chinook salmon stocks in British Columbia, 2004](#). DFO Can. Sci. Advis. Sec. Res. Doc. 2006/083. vii + 67 p.

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## APPENDIX D. DATA-LIMITED ISSUES PART 2

As noted in section 4 of the body of the document, the Chinook Projection model was used for forward projections. This model requires specific inputs and parameters to initialize. This appendix provides brief descriptions of the efforts made to produce the required inputs and parameter estimates for DUs 8 (MFR-Portage), 9 (MFR-Spring), 10 (MFR-Summer), 11 (UFR-Spring), and 17 (NTh-Summer). Ultimately the projections were not included in the main research document due to the uncertainty surrounding the input parameters. Example projections from the Chinook Projection Model under different potential productivity values are provided in Section 2. The projections should not be treated as quantitative analysis of the likely trajectory of these populations, but rather a qualitative demonstration of potential trajectories depending on a range of suspected productivities for these DUs where the true current productivity is unknown.

### D.1. PARAMETER ESTIMATION

DU2 (LFR-Harrison) is the only DU where there is enough data to calculate DU specific inputs. For the remaining five DUs for which forward projections were attempted, proxy information from model outputs and/or data from other stocks were used to generate the required input values. The sections below describe those efforts and the associated uncertainty for DUs 8, 9, 10, 11, and 17.

#### D.1.1. Initial Escapement Estimates

To initialize the Chinook Projection Model, six years of escapement estimates are required. Although the available time series for DUs 8, 9, 10, 11, and 17 are still relative and not absolute abundance estimates, they were expected to be a closer representation of absolute abundance than DUs 4, 5, 7, 14 and 16, that were already excluded from the analysis. Therefore, the first assumption required for projection is that these relative escapements are accurate proxies of absolute abundance. However, for DUs 10 (MFR-Summer) and 17 (NTh-Summer) adjustments to the escapement time series were required, because the systems included in the escapement time series did not align with the systems included in the estimate of watershed area used in the habitat model to estimate proxy S-R parameters. Specifically, the Stuart and North Thompson escapement time series are not included in the escapement estimates of DUs 10 and 17, respectively, but the watersheds were used as part of the watershed area estimates for the habitat model. In order to pair the escapement data with the habitat model S-R estimates, an expansion factor is required. The expansion factors used were calculated based on the average proportion that the spawning population in these two systems contributed to the overall estimates of DU population size in the Run Reconstruction model. An expansion factor of 1.21 was used for DU10 and 1.58 for DU17. It is unknown whether these scalars accurately represent the systems' contributions to the DUs; therefore, DUs 10 and 17 have an additional assumption required for their projection.

#### D.1.2. Maturation Rates

For all five DUs, maturation rates from Dome Creek data were used as a proxy and averaged over the 10 years they are available. The data available from Dome Creek is outdated, as the most recent year maturation rates are available from is 2002. Additionally, Dome Creek is a Spring  $5_2$  population and may not accurately represent the Summer  $5_2$  populations.

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### D.1.3. Fishing Related Mortality

Harvest rates at age are difficult to estimate accurately for the Spring and Summer 5<sub>2</sub> MUs, because of the lack of CWT data. There are CWT harvest rate data from Dome Creek, but those rates were considered too outdated to represent recent conditions, so the CTC Chinook Model estimates of catch were used. The CTC Chinook Model produces estimates of cohort size, catch, and escapement by age at the MU level. Catch is provided for various fisheries, which can be identified as either terminal or pre-terminal. From this information it is possible to estimate harvest rates-at-age at the MU level. The MU-level harvest rates were applied at the DU level, assuming that harvest would be evenly distributed across the DUs in each MU.

Both pre-terminal and terminal harvest rates were calculated for each year ( $y$ ) and age ( $a$ ) using the following equations:

$$\text{Pre-Terminal Harvest Rate}_{y,a} = \text{Catch}_{y,a} / \text{Cohort Size}_{y,a}$$

$$\text{Terminal Harvest Rate}_{y,a} = \text{Terminal Catch}_{y,a} / \text{Terminal Run}_{y,a}$$

The Spring and Summer 5<sub>2</sub> MU estimates of pre-terminal harvest rates mostly appeared reasonable across each age of return. However, the estimate of age 6 harvest rates for the Spring 5<sub>2</sub> MU was close to zero and for the Summer 5<sub>2</sub> MU it was closer to 50%. Unfortunately, there is limited information about age-6 harvest rates for the Spring and Summer 5<sub>2</sub> MUs because the sample size of CWT tagged age 6 fish that return is very small. Given the uncertainty in the age-6 harvest rates, the relationship between age-4 and age-5 harvest rates at DU2 (LFR-Harrison) was used to recalculate the age-6 harvest rates for the Spring and Summer 5<sub>2</sub> MUs. The Spring and Summer age-5 harvest rates in each year were multiplied by the ratio of age-5 to age-4 harvest rates at DU2 (LFR-Harrison) to produce the new age-6 harvest rates.

The terminal harvest rate estimates for the Spring 5<sub>2</sub> MU appeared plausible, but the Summer 5<sub>2</sub> terminal harvest rates did not reflect expected patterns in harvest rates with regard to the distribution across ages. The older (and larger) age classes had lower harvest rates than the younger (smaller) age classes, which had harvest rates close to 100%. This may be due to the very limited age composition information in the Summer 5<sub>2</sub> escapement data for the Summer 5<sub>2</sub> MU for the CTC Chinook Model to fit to. To address this issue, escapement data in each year was redistributed to the age classes using the proportion of fish-at-age in escapement data from the Chilko River mark-recapture program. Chilko River is in DU10 (MFR-Summer) and is in development to be an indicator for the Summer 5<sub>2</sub> MU. While re-distributing the escapement data helped to improve the estimates for most age classes, the age-3 harvest rates remained high. To provide a more reasonable estimate of age-3 harvest rates, the relationship between age-2 and age-3 harvest rates at DU2 (LFR-Harrison) was used to recalculate the age-3 harvest rates of the Summer 5<sub>2</sub> MU based on the age-4 harvest rates in that year.

When comparing the 1986-2002 Dome Creek CWT data to the CTC Chinook Model estimates of harvest rates over those years, the CTC Chinook Model significantly underestimated the total exploitation rate and did a poor job of capturing the inter-annual variability. The CTC Chinook Model estimates of harvest rates have remained relatively constant over time, but it is expected that harvest rates have actually decreased due to changes in fisheries management. If harvest rates have decreased, the CTC Chinook Model estimates may better approximate current harvest levels compared to the period harvest rates are available from Dome Creek.

Unfortunately, this will remain difficult to determine until there is a new indicator for the Spring and Summer 5<sub>2</sub> MUs. An additional source of uncertainty is whether adjusting the Spring and Summer 5<sub>2</sub> harvest rates based on Harrison harvest rate relationships at-age is accurate, given the large differences in distance to terminal areas, ocean migration, and residence time

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between the MUs. Overall, these values represent an estimate of harvest rates-at-age, but it is unknown if they are accurate.

#### **D.1.4. Stock Recruit Data**

The habitat model estimates stock recruitment (S-R) parameters for Chinook Salmon based on watershed area, and can be used to provide S-R estimates for Chinook populations lacking S-R data sets (Parken et al. 2006). The habitat model was used to produce estimates of S-R parameters for these five DUs (Table D1). However, these values are unlikely to represent current productivity for these DUs, given observed declines. The habitat model uses mainly S-R and watershed data from the mid-1970's to the mid-1990's from non-Fraser populations (Parken et al. 2006). For Chinook populations, including these DUs, that are facing multiple threats causing freshwater habitat degradation and changes in marine survival, the relationship of production based on watershed area may have shifted over time. Additionally, the habitat model provides an estimate of average long-term productivity, which is unlikely to represent current productivity for populations experiencing a declining trend in productivities, as is suspected for these DUs.

In order to produce a more plausible estimate for current productivity, the habitat model estimate was decreased by 50%, based on the decline seen at DU2 (LFR-Harrison) over the past three generations. However, using the adjusted S-R variables produced forward projections that implied the populations would rebound quickly to abundances over those seen in the early part of the time series. Given the observed declining trends and knowledge of the threats facing these DUs, this outcome seems unlikely. As the trend in declining abundance is less steep at DU2 (LFR-Harrison) compared to the stream-type DUs, using the trend in productivity at DU2 (LFR-Harrison) might be under estimating the decline in productivity in these stream-type DUs.

A retrospective analysis of the productivity required to see the declining trends observed is difficult to conduct when the harvest rates are likely underestimated and generally inaccurate across ages. Using the harvest rates estimated described in Section 1.5 could lead to selecting a lower productivity value when higher harvest rates earlier in the time series could have played a role in the decline. The trade-off between the uncertainty in the decline in productivity and the uncertainty in harvest rates cannot be determined with any confidence. Initializing the Chinook Projection Model at the beginning of the time series for DU11 and projecting forward to 2008 using the yearly CWT harvest rates from Dome Creek, required a 50% reduction in alpha from the habitat model estimates to produce a trend over that period similar to that observed in the early 2000's. If productivity has continued to decline since the early 2000's this would imply that the reduction in productivity needs to be greater than 50% to reproduce the trend from the early 2000s. Despite efforts to produce reasonable S-R parameters, current productivity in these DUs is unknown, precluding the provision of forward projections with any certainty. The forward projections in Section 2 provide examples of trajectories with reductions in the habitat model-estimated alpha from 50-85%.

*Table D1 - The alpha and beta values for each of the DUs estimated from the habitat model.*

<b>DU</b>	<b>Alpha</b>	<b>Beta</b>
DU8 (MFR-Portage)	5.33	4.01E-04
DU9 (MFR-Spring)	5.60	2.50E-05
DU10 (MFR-Summer)	5.77	2.22E-05
DU11 (UFR-Spring)	6.28	2.33E-05
DU17 (NTh-Summer)	5.72	7.21E-05

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### **D.1.5. Big Bar Scenarios**

Big Bar impacts are preliminary; but given the work completed at the slide in early 2020, survival is expected to improve in 2020. The estimated mortality rates from the slide in 2019 are 89% for the Spring 5<sub>2</sub> MU and 51% for the Summer 5<sub>2</sub> MU. The decision to not provide projections for these DUs occurred before Big Bar scenarios were finalized by the working group. For the purposes of the example projections in Section 2, the estimated mortalities for 2019 were used, then subsequently in 2020 the mortality was reduced to 10% for both Spring and Summer DUs. For the subsequent years no additional mortality was assumed. While this might be an optimistic scenario, it is difficult to predict future mortality without additional data that will be collected in 2020.

### **D.1.6. Summary**

Almost all the model inputs are based on other models' outputs and there is very limited DU-specific data available for use in the projections. Due to the considerable (and usually unknown) uncertainty in the accuracy of the input values and the inability to identify S-R parameters, allowable harm was assessed based on the threats assessment, results of the forward projections for DU2 (LFR-Harrison), and projected future conditions with climate change. To provide more accurate forward projections, DU-level S-R and harvest rate at age data will be required.

## **D.2. EXAMPLE PROJECTIONS**

The following sections provide example projections with varying reductions in alpha from the habitat model estimate, the input parameters used in the projections, and the number of trajectories within each alpha scenario that achieved the lower and upper recovery targets. As noted above, these projections should only be seen as a qualitative demonstration of possible trajectories given alternative alpha values for these populations. These are not intended to inform the allowable harm assessment and were not used in the determination made in the main research document.

## DU8 (MFR-Portage)

### Portage

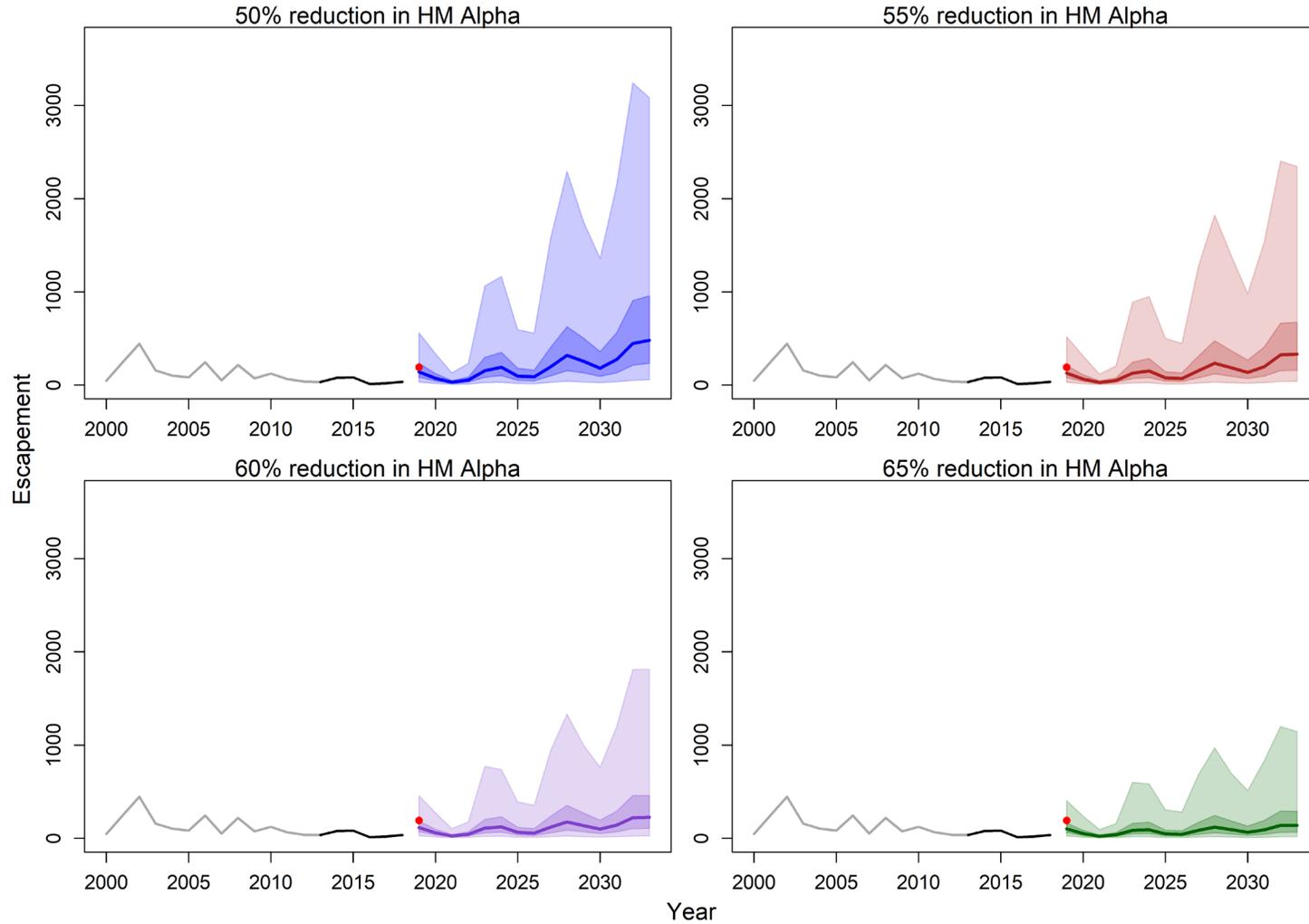


Figure D1 - Forward projections from 2019 to 2033 for DU8 (MFR-Portage) under recent harvest rate conditions with varying alpha values, calculated as a 50 to 65 percent reduction from the habitat model (HM) output. Initial 2019 estimates are displayed by the red circle.

**Portage**

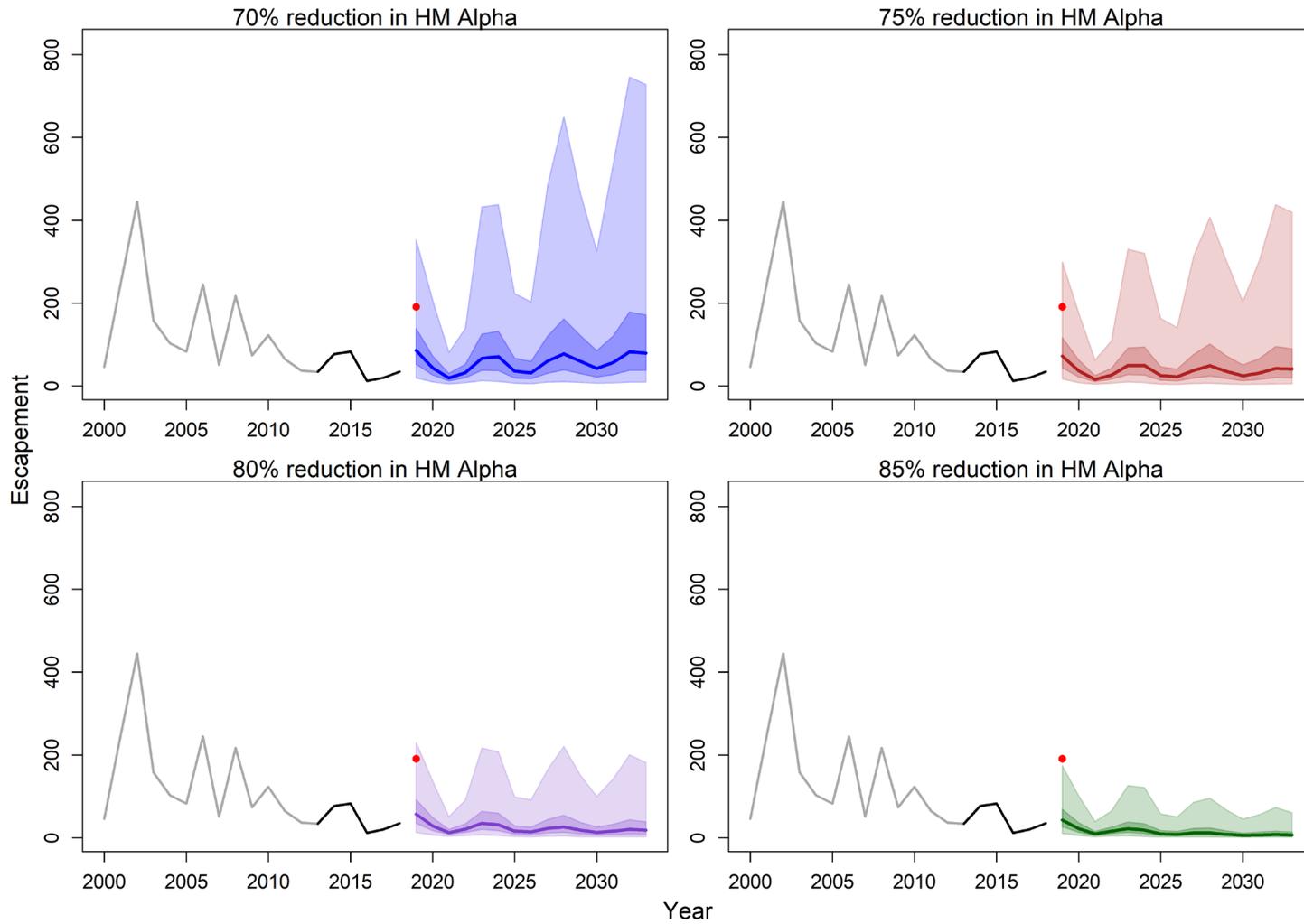


Figure D2 - Forward projections from 2019 to 2033 for DU8 (MFR-Portage) under recent harvest rate conditions with varying alpha values, calculated as a 70 to 85 percent reduction from the habitat model (HM) output. Initial 2019 estimates are displayed by the red circle. Note the change in y axis from the previous figure.

Table D2 - Input parameters for the Chinook Projection Model for DU8 (MFR-Portage)

DU8 (MFR-Portage)			
Data type	Information needed	Values	Source
S-R function	S-R parameters, alphas reduced by: 50%, 55%, 60%, 65%, 70%, 75%, 80%, 85%	Alphas: 2.7, 2.4 , 2.1 , 1.9, 1.6, 1.3, 1.1, 0.8 Constant Beta: 4.01E-04	Adjusted Habitat Model Estimates
	Tau R	1.51	Tau R used from Harrison as a proxy
Recovery Targets	85% Smsy	1362 and positive population growth	WSP Benchmarks and COSEWIC Criteria
	Sgen or 1000	1000 and positive population growth	
Max Age	Maximum Age	6	-
	2013	34	
	2014	77	
Escapement Lead In	2015	83	NuSEDS high quality estimates
	2016	12	
	2017	20	
	2018	35	
Natural Survival	Age 1	0.5	CTC Cohort Analysis assumption
	Age 2	0.6	
	Age 3	0.7	
	Age 4	0.8	
	Age 5	0.9	
	Age 6	0.9	
Maturation Rates (Average and SD of BY 1986-2002)	Age 2	0.0000, SD 0.000	2019 CTC Exploitation Rate Analysis Out files Dome used as proxy
	Age 3	0.0040, SD 0.008	
	Age 4	0.4532, SD 0.210	
	Age 5	0.9913, SD 0.016	
	Age 6	1.0000, SD 0.000	
Pre-terminal ERs (Average of RY 2009-2015)	Age 3	CA 1.2%   US 0.4%   Total 1.7%	CTC Chinook Model, 2019
	Age 4	CA 2.2%   US 1.5%   Total 3.8%	
	Age 5	CA 5.9%   US 1.9%   Total 7.8%	
	Age 6	CA 1.9%   US 0.0%   Total 1.9%	
Terminal ERs (Average of RY 2009-2015)	Age 3	CA 10.0%   US 0.0%   Total 10.0%	CTC Chinook Model, 2019
	Age 4	CA 23.3%   US 0.0%   Total 23.30%	
	Age 5	CA 34.9%   US 0.0%   Total 34.9%	
	Age 6	CA 38.4%   US 0.0%   Total 38.4%	

Table D3 - The percentage of simulations meeting the Lower and Upper Recovery Targets for DU8 (MFR-Portage).

<b>DU8 (MFR-Portage)</b>			
<b>Percent Reduction in Alpha</b>	<b>Alpha</b>	<b>Lower Recovery Target Met</b>	<b>Upper Recovery Target Met</b>
50	2.7	12%	5%
55	2.4	5%	2%
60	2.1	2%	1%
65	1.9	1%	0%
70	1.6	0%	0%
75	1.3	0%	0%
80	1.1	0%	0%
85	0.8	0%	0%

## DU9 (MFR-Spring)

### MFR-Springs

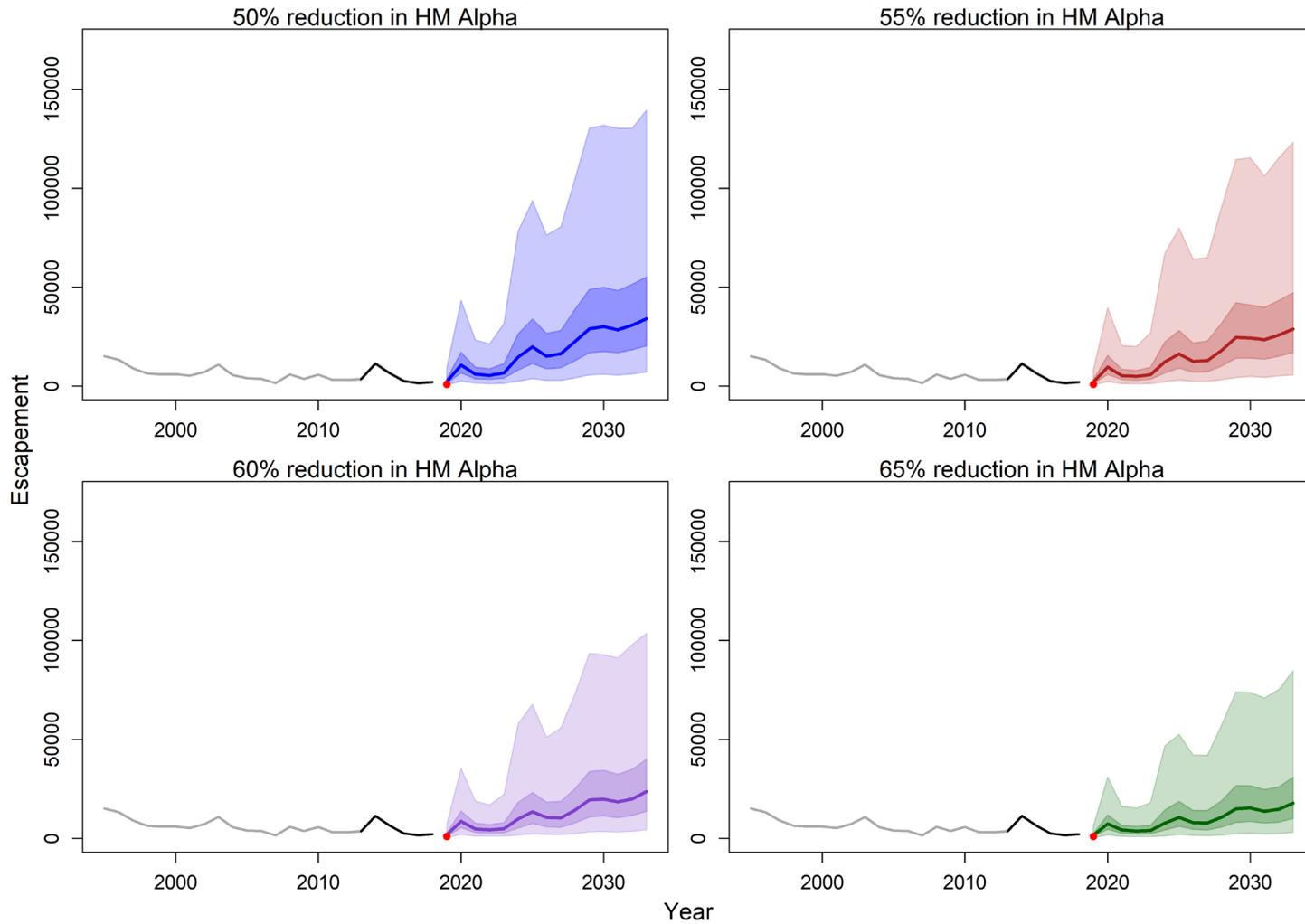


Figure D3 - Forward projections from 2019 to 2033 for DU9 (MFR-Spring) under recent harvest rate conditions with varying alpha values, calculated as a 50 to 65 percent reduction from the habitat model (HM) output. Initial 2019 estimates are displayed by the red circle.

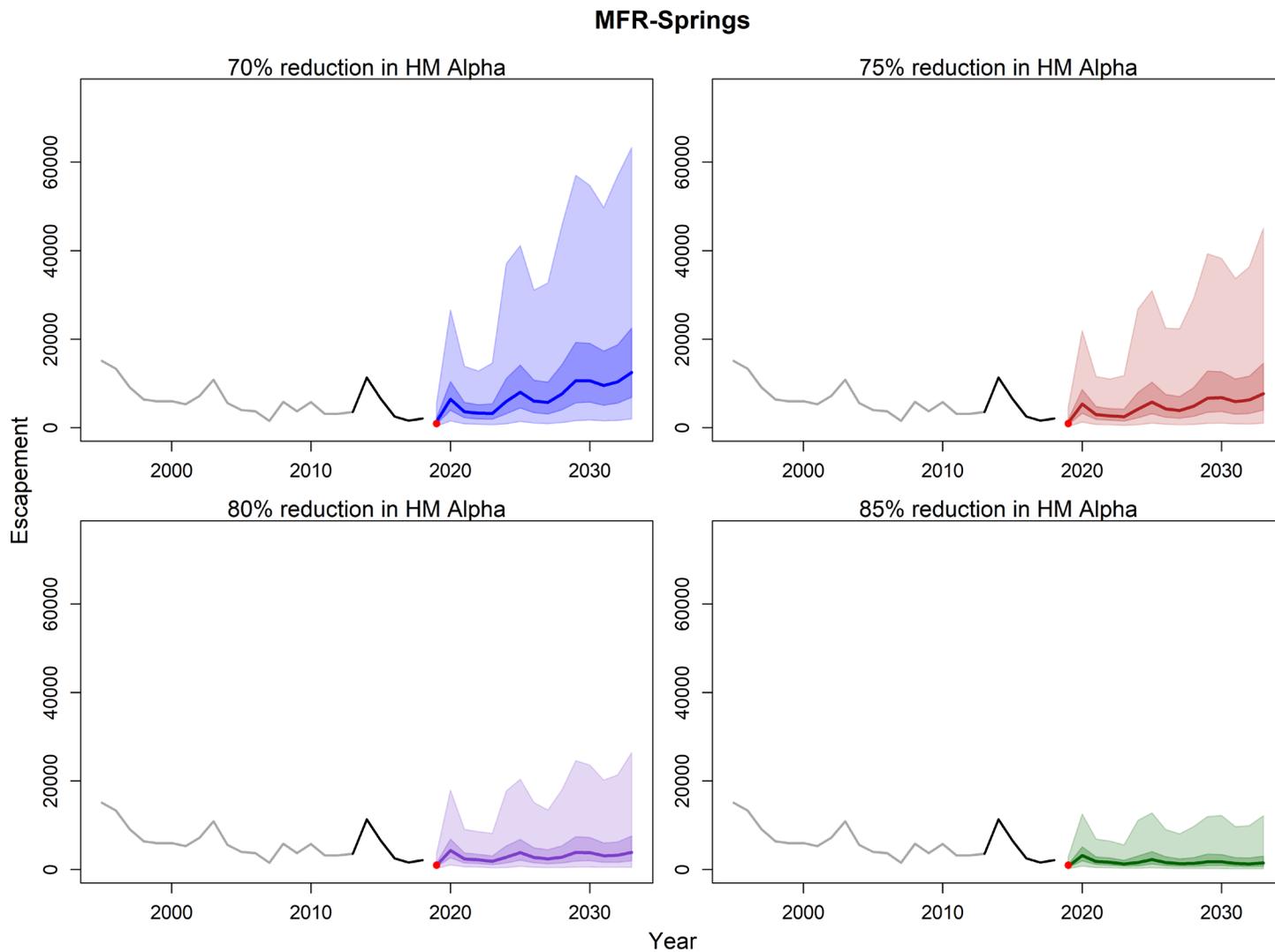


Figure D4 - Forward projections from 2019 to 2033 for DU9 (MFR-Spring) under recent harvest rate conditions with varying alpha values, calculated as a 70 to 85 percent reduction from the habitat model (HM) output. Initial 2019 estimates are displayed by the red circle. Note the change in y axis from the previous figure.

Table D4 - Input parameters for the Chinook Projection Model for DU9 (MFR-Spring)

DU9 (MFR-Spring)			
Data type	Information needed	Values	Source
S-R function	S-R parameters, alphas reduced by: 50%, 55%, 60%, 65%, 70%, 75%, 80%, 85%	Alphas: 2.8, 2.5, 2.2, 2.0, 1.7, 1.4, 1.1, 0.8 Constant Beta: 2.51E-05	Adjusted Habitat Model Estimates
	Tau R	1.51	Tau R used from Harrison as a proxy
Recovery Targets	85% Smsy	22,152 and $\leq 30\%$ decline	WSP Benchmarks and COSEWIC Criteria
	Sgen or 1000	4,927 and positive population growth	
Max Age	Maximum Age	6	-
	2013	3567	
Escapement Lead In	2014	11336	
	2015	6553	NuSEDS high quality estimates with infilling
	2016	2518	
	2017	1584	
	2018	2111	
Age 1	0.5		
Natural Survival	Age 2	0.6	
	Age 3	0.7	CTC Cohort Analysis assumption
	Age 4	0.8	
	Age 5	0.9	
	Age 6	0.9	
Maturation Rates (Average and SD of BY 1986-2002)	Age 2	0.0000, SD 0.000	
	Age 3	0.0040, SD 0.008	2019 CTC Exploitation Rate Analysis Out files Dome used as proxy
	Age 4	0.4532, SD 0.210	
	Age 5	0.9913, SD 0.016	
	Age 6	1.0000, SD 0.000	
Pre-terminal ERs (Average of RY 2009-2015)	Age 3	CA 0.3%   US 0.1%   Total 0.4%	
	Age 4	CA 1.0%   US 1.7%   Total 2.7%	CTC Chinook Model, 2019
	Age 5	CA 5.5%   US 2.0%   Total 7.5%	
	Age 6	CA 1.7%   US 0.3%   Total 2.0%	
Terminal ERs (Average of RY 2009-2015)	Age 3	CA 0.8%   US 0.0%   Total 0.8%	
	Age 4	CA 3.7%   US 0.0%   Total 3.7%	CTC Chinook Model, 2019
	Age 5	CA 29.6%   US 0.0%   Total 29.6%	
	Age 6	CA 40.5%   US 0.0%   Total 40.5%	

Table D5 - The percentage of simulations meeting the Lower and Upper Recovery Targets for DU9 (MFR-Spring).

<b>DU9 (MFR-Spring)</b>			
<b>Percent Reduction in Alpha</b>	<b>Alpha</b>	<b>Lower Recovery Target Met</b>	<b>Upper Recovery Target Met</b>
50	2.8	100%	87%
55	2.5	100%	75%
60	2.2	100%	59%
65	2.0	98%	38%
70	1.7	93%	18%
75	1.4	77%	6%
80	1.1	43%	1%
85	0.8	10%	0%

**DU10 (MFR-Summer)**

**MFR-Summers**

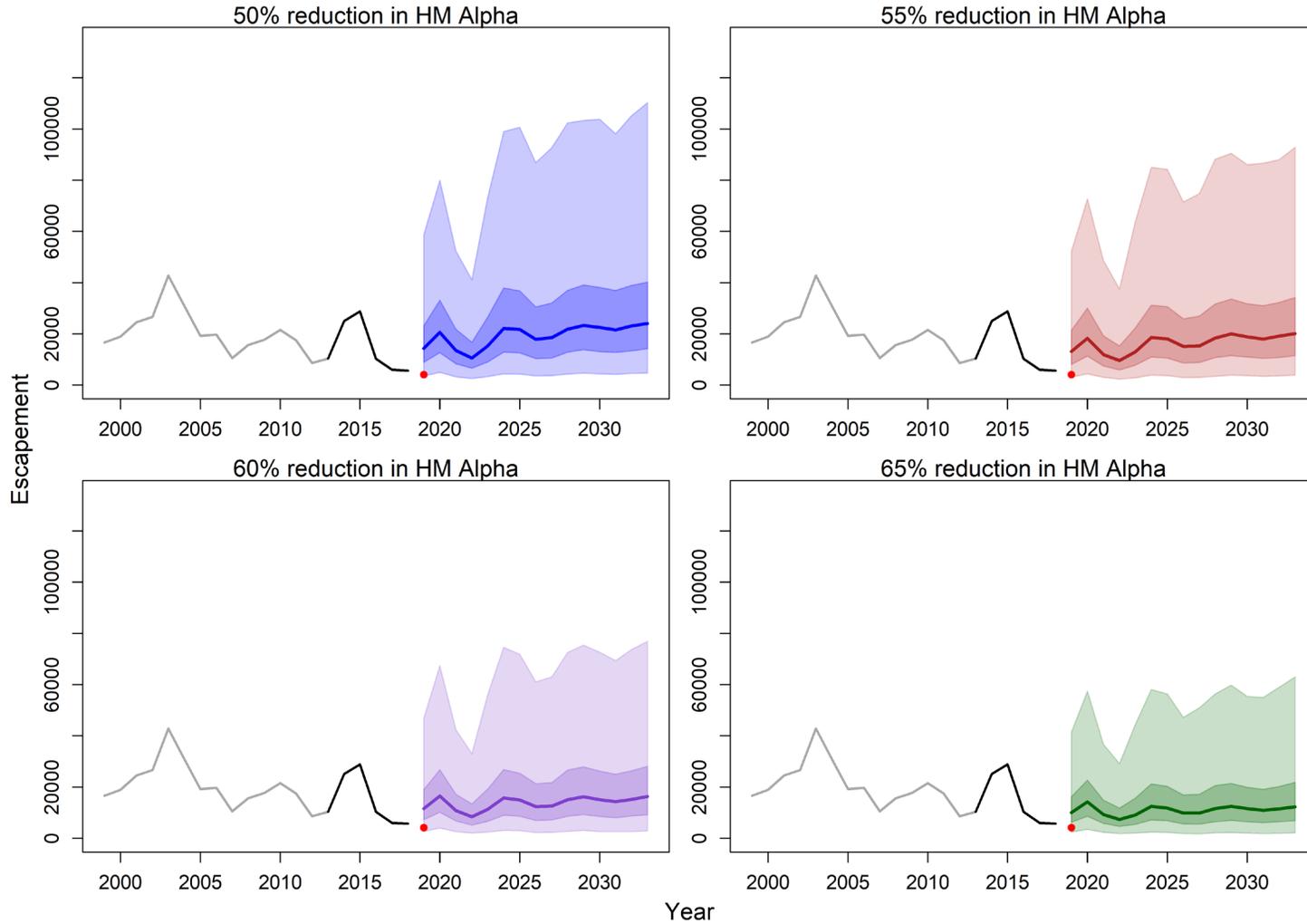


Figure D5 - Forward projections from 2019 to 2033 for DU10 (MFR-Summer) under recent harvest rate conditions with varying alpha values, calculated as a 50 to 65 percent reduction from the habitat model (HM) output. Initial 2019 estimates are displayed by the red circle.

**MFR-Summers**

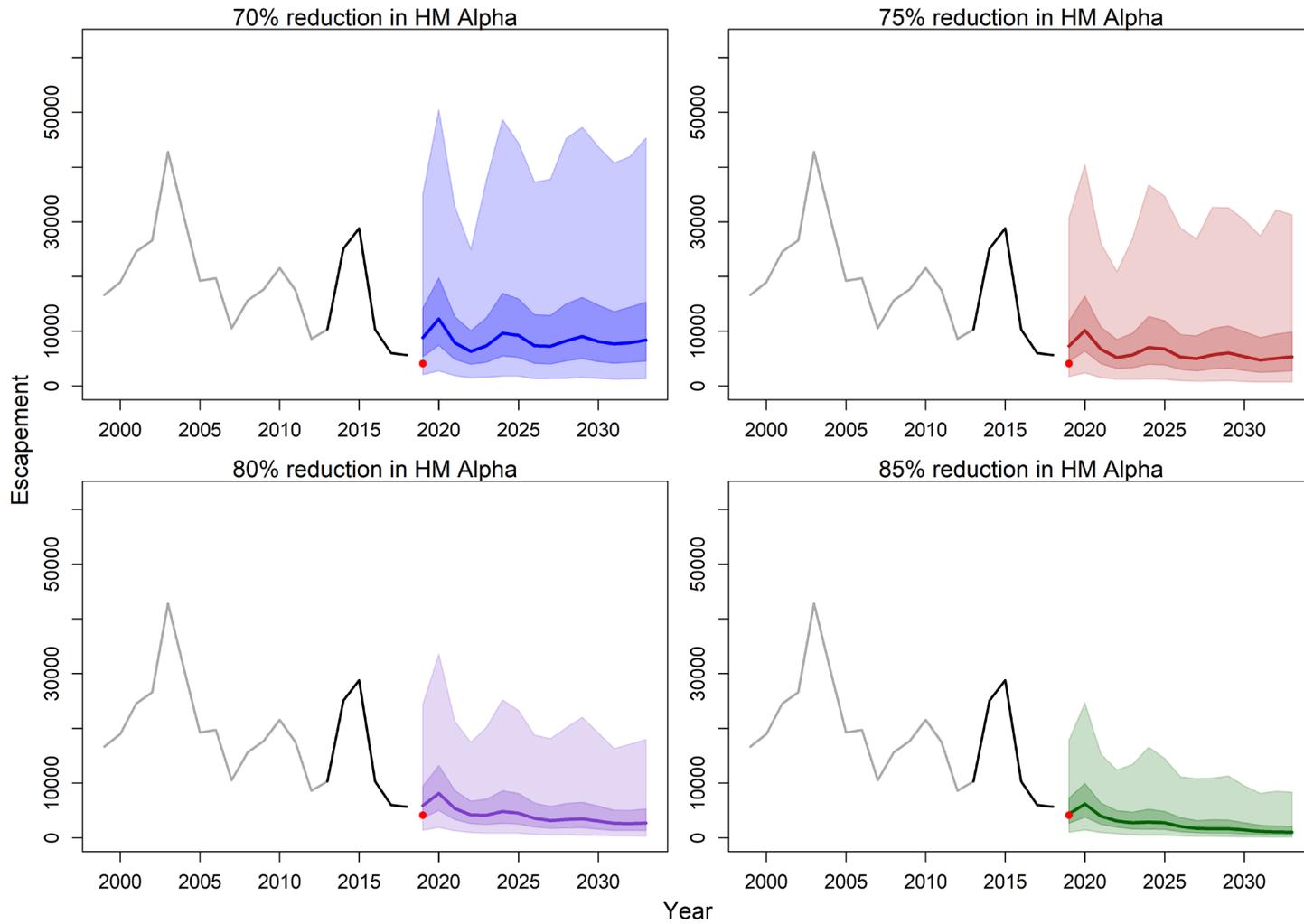


Figure D6 - Forward projections from 2019 to 2033 for DU10 (MFR-Summer) under recent harvest rate conditions with varying alpha values, calculated as a 70 to 85 percent reduction from the habitat model (HM) output. Initial 2019 estimates are displayed by the red circle. Note the change in y axis from the previous figure.

Table D6 - Input parameters for the Chinook Projection Model for DU10 (MFR-Summer)

DU10 (MFR-Summer)			
Data type	Information needed	Values	Source
S-R function	S-R parameters, alphas reduced by: 50%, 55%, 60%, 65%, 70%, 75%, 80%, 85%	Alphas: 2.9, 2.6, 2.3, 2.0, 1.7, 1.4, 1.2, 0.9 Constant Beta: 2.22E-05	Adjusted Habitat Model Estimates
	Tau R	1.51	Tau R used from Harrison as a proxy
Recovery Targets	85% Smsy	25,312 and $\leq 30\%$ decline	WSP Benchmarks and COSEWIC Criteria
	Sgen or 1000	5,371 and positive population growth	
Max Age	Maximum Age	6	-
	2013	10323	
Escapement Lead In	2014	25069	NuSEDS high quality estimates with infilling and 1.21 Scalar for the Stuart River
	2015	28803	
	2016	10310	
	2017	5993	
	2018	5668	
Natural Survival	Age 1	0.5	CTC Cohort Analysis assumption
	Age 2	0.6	
	Age 3	0.7	
	Age 4	0.8	
	Age 5	0.9	
	Age 6	0.9	
Maturation Rates (Average and SD of BY 1986-2002)	Age 2	0.0000, SD 0.000	2019 CTC Exploitation Rate Analysis Out files Dome used as proxy
	Age 3	0.0040, SD 0.008	
	Age 4	0.4532, SD 0.210	
	Age 5	0.9913, SD 0.016	
	Age 6	1.0000, SD 0.000	
Pre-terminal ERs (Average of RY 2009-2015)	Age 3	CA 1.2%   US 0.4%   Total 1.7%	CTC Chinook Model, 2019
	Age 4	CA 2.2%   US 1.5%   Total 3.8%	
	Age 5	CA 5.9%   US 1.9%   Total 7.8%	
	Age 6	CA 1.9%   US 0.0%   Total 1.9%	
Terminal ERs (Average of RY 2009-2015)	Age 3	CA 10.0%   US 0.0%   Total 10.0%	CTC Chinook Model, 2019
	Age 4	CA 23.3%   US 0.0%   Total 23.30%	
	Age 5	CA 34.9%   US 0.0%   Total 34.9%	
	Age 6	CA 38.4%   US 0.0%   Total 38.4%	

Table D7 - The percentage of simulations meeting the Lower and Upper Recovery Targets for DU10 (MFR-Summer).

DU10 (MFR-Summer)			
Percent Reduction in Alpha	Alpha	Lower Recovery Target Met	Upper Recovery Target Met
50	2.9	76%	53%
55	2.6	72%	40%
60	2.3	66%	26%
65	2.0	58%	13%
70	1.7	44%	5%
75	1.4	27%	1%
80	1.2	10%	0%
85	0.9	1%	0%

## DU11 (UFR-Spring)

### UFR-Springs

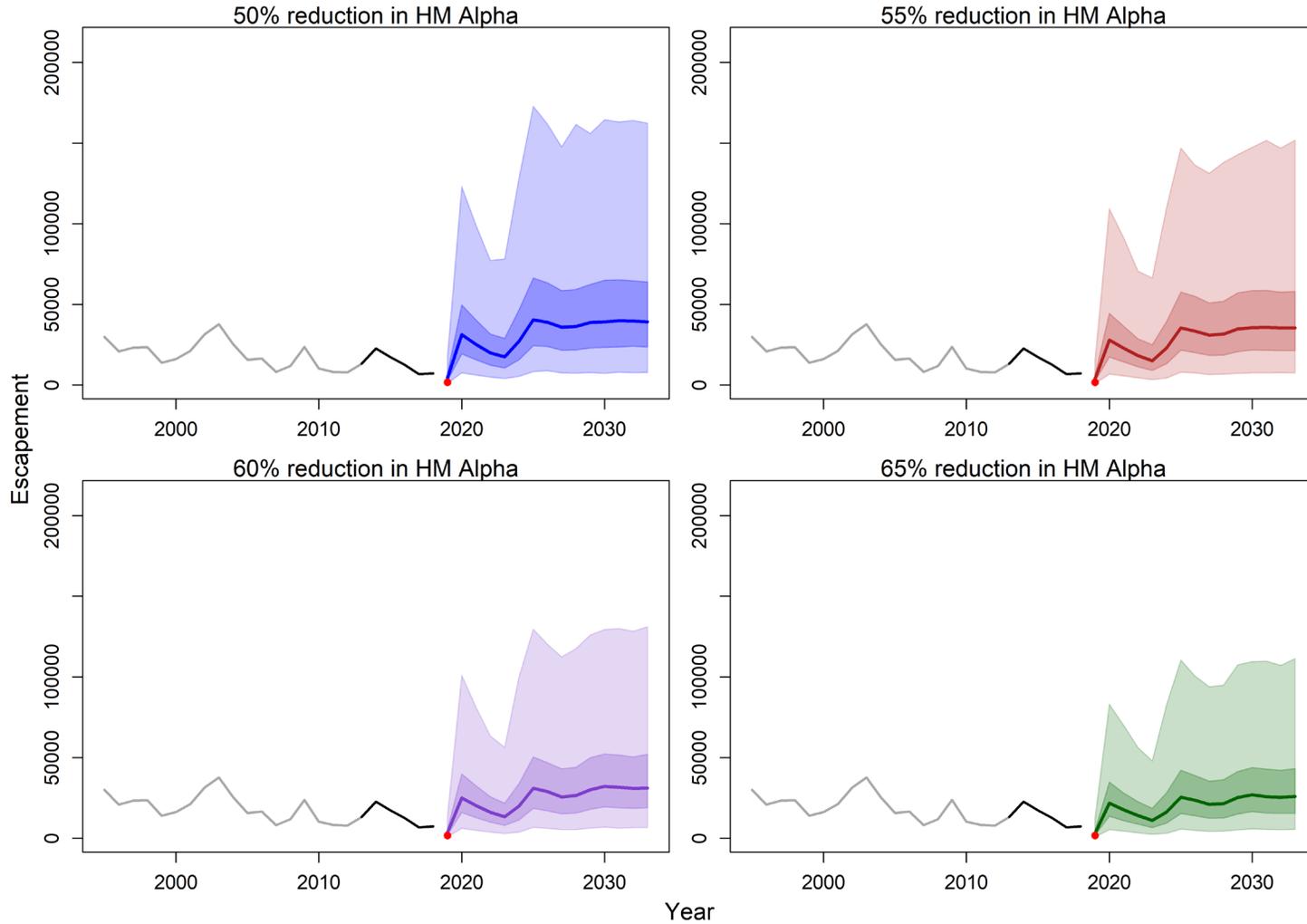


Figure D7 - Forward projections from 2019 to 2033 for DU11 (MFR-Spring) under recent harvest rate conditions with varying alpha values, calculated as a 50 to 65 percent reduction from the habitat model (HM) output. Initial 2019 estimates are displayed by the red circle.

### UFR-Springs

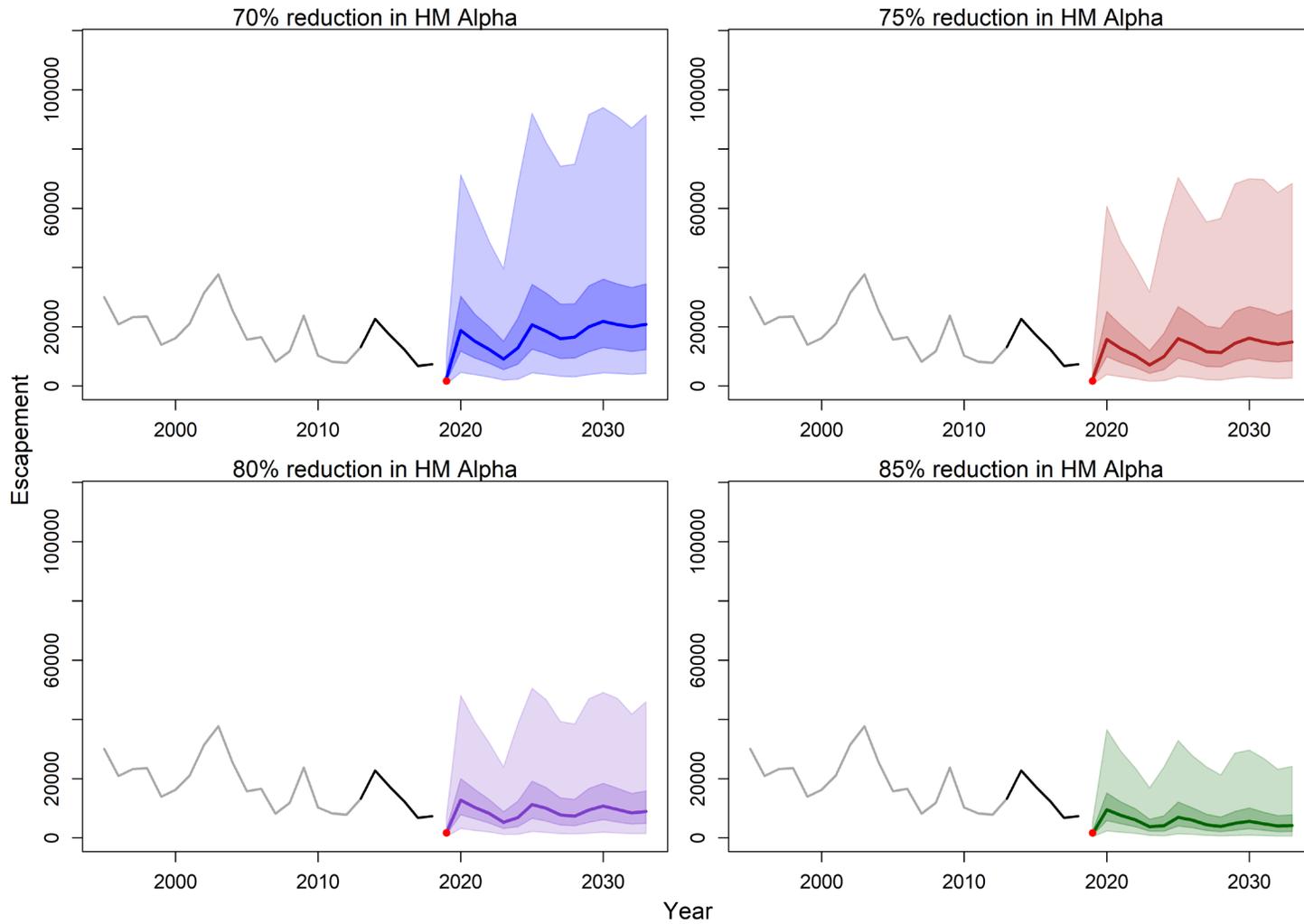


Figure D8 - Forward projections from 2019 to 2033 for DU11 (MFR-Spring) under recent harvest rate conditions with varying alpha values, calculated as a 70 to 85 percent reduction from the habitat model (HM) output. Initial 2019 estimates are displayed by the red circle. Note the change in y axis from the previous figure.

Table D8 - Input parameters for the Chinook Projection Model for DU11 (UFR-Spring)

DU11 (UFR-Spring)			
Data type	Information needed	Values	Source
S-R function	S-R parameters, alphas reduced by: 50%, 55%, 60%, 65%, 70%, 75%, 80%, 85%	Alphas: 3.1, 2.8, 2.5, 2.2, 1.9, 1.6, 1.3, 0.9 Constant Beta: 2.50E-05	Adjusted Habitat Model Estimates
	Tau R	1.51	Tau R used from Harrison as a proxy
Recovery Targets	85% Smsy	25,664 and $\leq 30\%$ decline	WSP Benchmarks and COSEWIC Criteria
	Sgen or 1000	5,671 and positive population growth	
Max Age	Maximum Age	6	-
	2013	13206	
Escapement Lead In	2014	22696	NuSEDS high quality estimates with infilling
	2015	17362	
	2016	12596	
	2017	6763	
	2018	7322	
Natural Survival	Age 1	0.5	CTC Cohort Analysis assumption
	Age 2	0.6	
	Age 3	0.7	
	Age 4	0.8	
	Age 5	0.9	
Maturation Rates (Average and SD of BY 1986-2002)	Age 6	0.9	2019 CTC Exploitation Rate Analysis Out files Dome used
	Age 2	0.0000, SD 0.000	
	Age 3	0.0040, SD 0.008	
	Age 4	0.4532, SD 0.210	
	Age 5	0.9913, SD 0.016	
Pre-terminal ERs (Average of RY 2009-2015)	Age 6	1.0000, SD 0.000	CTC Chinook Model, 2019
	Age 3	CA 0.3%   US 0.1%   Total 0.4%	
	Age 4	CA 1.0%   US 1.7%   Total 2.7%	
	Age 5	CA 5.5%   US 2.0%   Total 7.5%	
Terminal ERs (Average of RY 2009-2015)	Age 6	CA 1.7%   US 0.3%   Total 2.0%	CTC Chinook Model, 2019
	Age 3	CA 0.8%   US 0.0%   Total 0.8%	
	Age 4	CA 3.7%   US 0.0%   Total 3.7%	
	Age 5	CA 29.6%   US 0.0%   Total 29.6%	
	Age 6	CA 40.5%   US 0.0%   Total 40.5%	

Table D9 - The percentage of simulations meeting the Lower and Upper Recovery Targets for DU11 (UFR-Spring).

<b>DU11 (UFR-Spring)</b>			
<b>Percent Reduction in Alpha</b>	<b>Alpha</b>	<b>Lower Recovery Target Met</b>	<b>Upper Recovery Target Met</b>
50	3.1	94%	90%
55	2.8	95%	86%
60	2.5	95%	79%
65	2.2	94%	66%
70	1.9	92%	46%
75	1.6	87%	23%
80	1.3	74%	6%
85	0.9	39%	1%

## DU17 (NTh-Summer)

### NTh-Summers

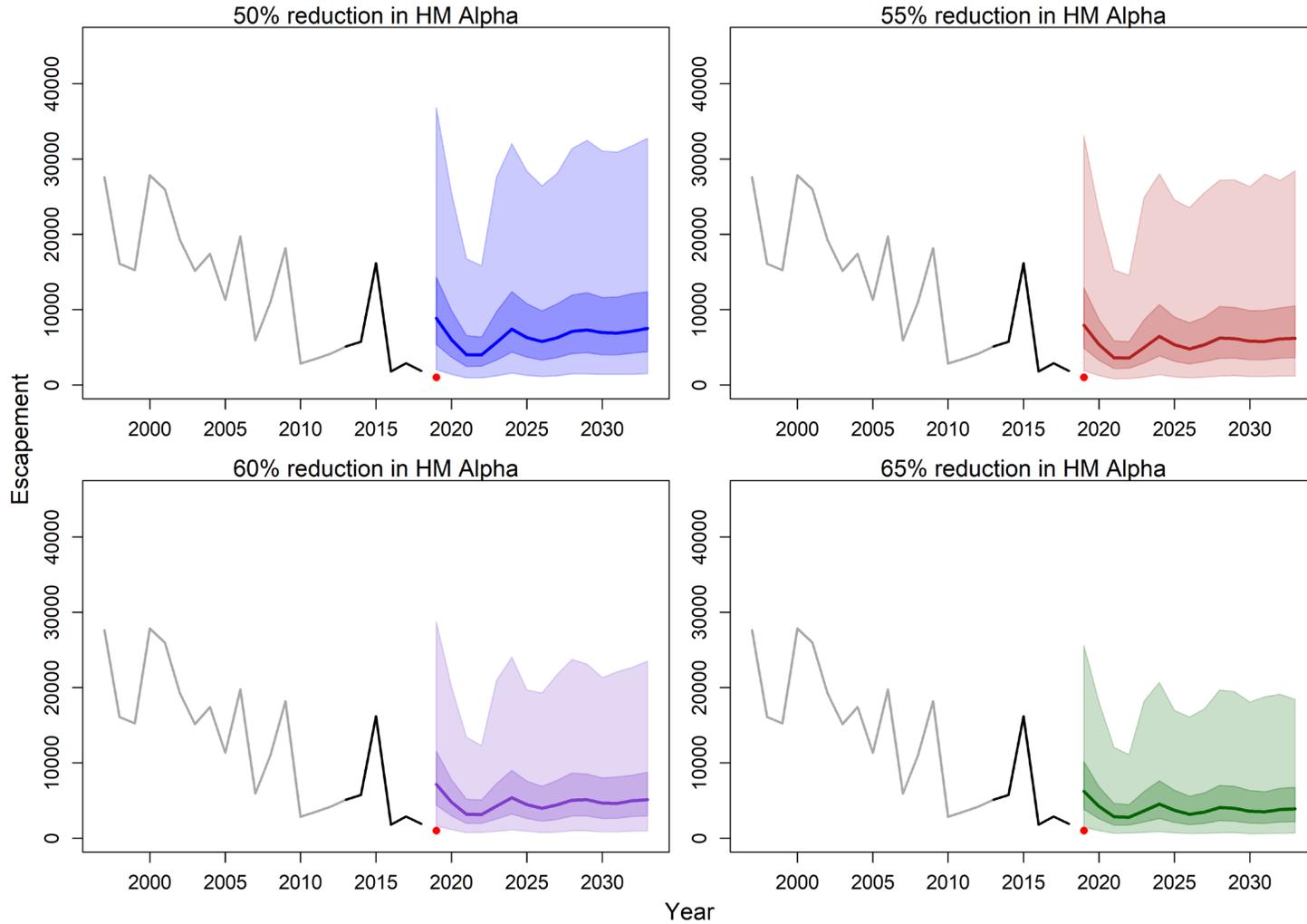


Figure D9 - Forward projections from 2019 to 2033 for DU17 (NTh-Summer) under recent harvest rate conditions with varying alpha values, calculated as a 50 to 65 percent reduction from the habitat model (HM) output. Initial 2019 estimates are displayed by the red circle.

### NTh-Summers

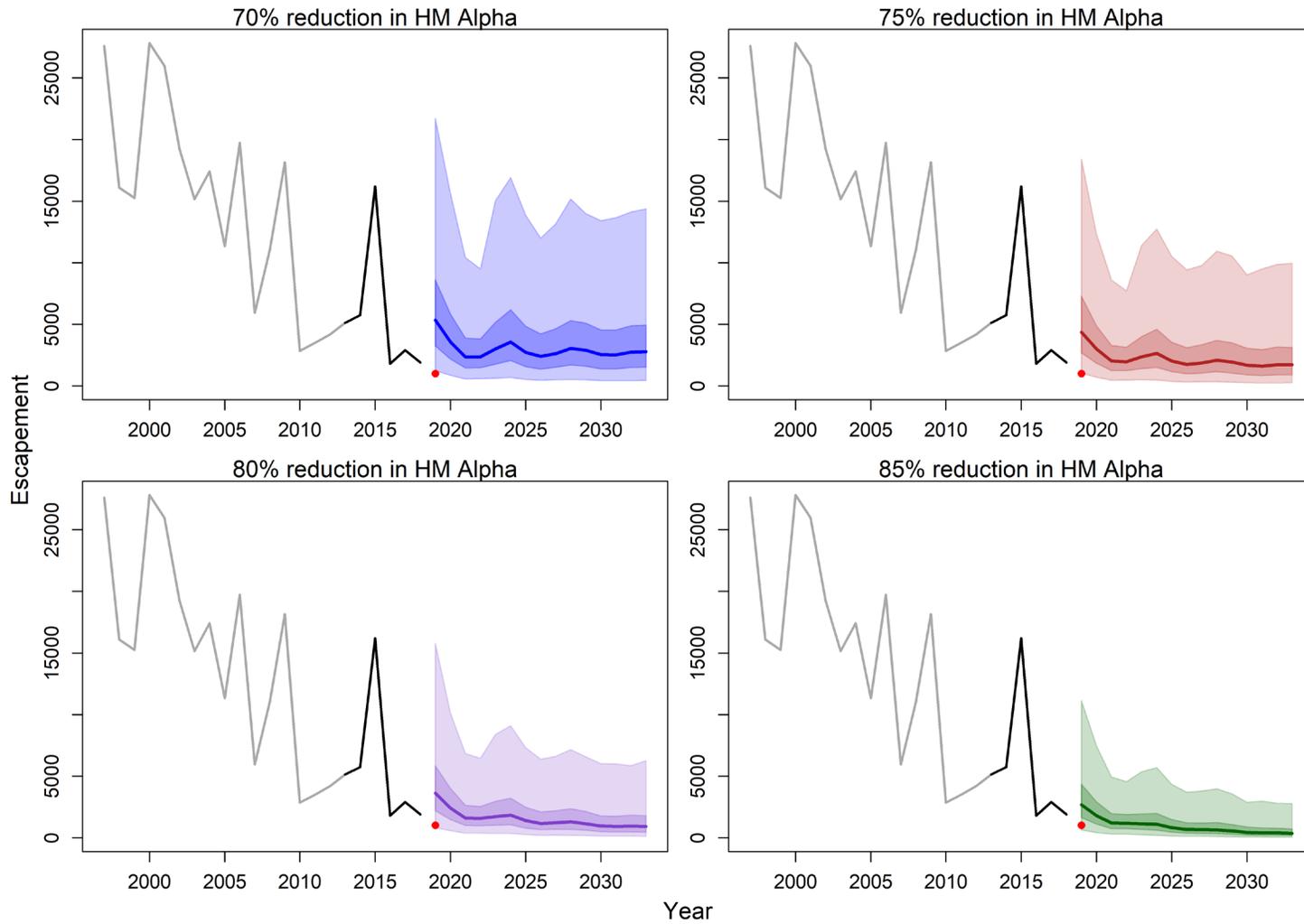


Figure D10 - Forward projections from 2019 to 2033 for DU17 (NTh-Summer) under recent harvest rate conditions with varying alpha values, calculated as a 70 to 85 percent reduction from the habitat model (HM) output. Initial 2019 estimates are displayed by the red circle. Note the change in y axis from the previous figure.

Table D10 - Input parameters for the Chinook Projection Model for DU17 (NTh-Summer)

DU17 (NTh-Summer)			
Data type	Information needed	Values	Source
S-R function	S-R parameters, alphas reduced by: 50%, 55%, 60%, 65%, 70%, 75%, 80%, 85%	Alphas: 2.9, 2.6, 2.3, 2.0, 1.7, 1.4, 1.1, 0.9 Constant Beta: 7.21E-05	Adjusted Habitat Model Estimates
	Tau R	1.51	Tau R used from Harrison as a proxy
Recovery Targets	85% Smsy	7,822 and positive population growth	WSP Benchmarks and COSEWIC Criteria
	Sgen or 1000	1,686 and positive population growth	
Max Age	Maximum Age	6	-
	2013	5133	
Escapement Lead In	2014	5739	NuSEDS high quality estimates with infilling and 1.58 scalar for North Thompson River
	2015	16189	
	2016	1808	
	2017	2901	
	2018	1899	
Natural Survival	Age 1	0.5	CTC Cohort Analysis assumption
	Age 2	0.6	
	Age 3	0.7	
	Age 4	0.8	
	Age 5	0.9	
	Age 6	0.9	
Maturation Rates (Average and SD of BY 1986-2002)	Age 2	0.0000, SD 0.000	2019 CTC Exploitation Rate Analysis Out files Dome used as proxy
	Age 3	0.0040, SD 0.008	
	Age 4	0.4532, SD 0.210	
	Age 5	0.9913, SD 0.016	
	Age 6	1.0000, SD 0.000	
Pre-terminal ERs (Average of RY 2009-2015)	Age 3	CA 1.2%   US 0.4%   Total 1.7%	CTC Chinook Model, 2019
	Age 4	CA 2.2%   US 1.5%   Total 3.8%	
	Age 5	CA 5.9%   US 1.9%   Total 7.8%	
	Age 6	CA 1.9%   US 0.0%   Total 1.9%	
Terminal ERs (Average of RY 2009-2015)	Age 3	CA 10.0%   US 0.0%   Total 10.0%	CTC Chinook Model, 2019
	Age 4	CA 23.3%   US 0.0%   Total 23.30%	
	Age 5	CA 34.9%   US 0.0%   Total 34.9%	
	Age 6	CA 38.4%   US 0.0%   Total 38.4%	

Table D11 - The percentage of simulations meeting the Lower and Upper Recovery Targets for DU17 (NTh-Summer).

DU17 (NTh-Summer)			
Percent Reduction in Alpha	Alpha	Lower Recovery Target Met	Upper Recovery Target Met
50	2.9	64%	48%
55	2.6	60%	37%
60	2.3	54%	24%
65	2.0	44%	13%
70	1.7	33%	5%
75	1.4	19%	1%
80	1.1	7%	0%
85	0.9	1%	0%

### D.3. REFERENCES

- Dobson, D., Holt, K., and Davis, B. 2020. [A Technical Review of the Management Approach for Stream-Type Fraser River Chinook](#). DFO Can. Sci. Advis. Sec. Res. Doc 2020/027. x + 280 p.
- Parken, C.K., McNicol, R.E., and Irvine, J.R. 2006. [Habitat-based methods to estimate escapement goals for data limited Chinook salmon stocks in British Columbia, 2004](#). DFO Can. Sci. Advis. Sec. Res. Doc. 2006/083. vii + 67 p.

## APPENDIX E. INPUT PARAMETERS FOR DU2 (LFR-HARRISON)

Table E1 below has the final input parameters used for the Chinook Projection Model.

*Table E1 - Input parameters for the Chinook Projection Model for DU2 (LFR-Harrison)*

<b>DU2 (LFR-Harrison)</b>			
<b>Data type</b>	<b>Information needed</b>	<b>Values</b>	<b>Source</b>
S-R function	S-R parameters (a, b)	2.17, 5.83 E-06	Harrison Updated SR analysis
	Error ( $\sigma_{total}, \rho$ )	0.75, 0.66	
Recovery Targets	85% Smsy	63,808 and < -30% decline	WSP Benchmarks and COSEWIC Criteria
	Sgen or 1000	15,313 and < -30% decline	
Max Age	Maximum Age	5	-
	2015	44907	
Escapement Lead In	2016	101759	NuSEDS high quality estimates
	2017	41526	
	2018	30049	
	2019	46336	
Natural Survival	Age 1	0.5	CTC Cohort Analysis assumption
	Age 2	0.6	
	Age 3	0.7	
	Age 4	0.8	
	Age 5	0.9	
Maturation Rates (Average and SD of BY 2005-2013)	Age 2	0.1088, SD 0.079	2019 CTC Exploitation Rate Analysis Out files
	Age 3	0.2321, SD 0.142	
	Age 4	0.9470, SD 0.050	
	Age 5	1.0000, SD 0.000	
Pre-terminal ERs (Average of RY 2009-2015)	Age 3	CA 1.7%   US 1.1%   Total 2.8%	HRJ file, 2019
	Age 4	CA 7.4%   US 5.6%   Total 12.9%	
	Age 5	CA 13.0%   US 6.3%   Total 19.4%	
	Age 6	CA 5.1%   US 1.0%   Total 6.0%	
Terminal ERs (Average of RY 2009-2015)	Age 3	CA 2.9%   US 0.0%   Total 2.9%	HRJ file, 2019
	Age 4	CA 6.5%   US 0.4%   Total 6.9%	
	Age 5	CA 3.0%   US 0.0%   Total 3.1%	
	Age 6	CA 0.8%   US 0.6%   Total 1.5%	

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## **APPENDIX F. TEMPORAL VARIATION IN PRODUCTIVITY FOR DU2 (LFR-HARRISON)**

For Chinook Salmon, population productivity results from a combination of the average fecundity per female spawner and survival rates among many life stages from egg deposition to adult spawner. Survival is influenced by both density-independent and density-dependent factors, whereas average fecundity for Chinook Salmon is influenced by population-specific fecundity, size and age composition of spawners, and other factors (Healey and Heard 1984). Recently, there has been considerable literature published about the trends for some of the factors influencing productivity for many North American populations of Chinook Salmon (Ohlberger et al. 2018, 2020; Xu et al. 2020; DFO 2018; Sharma et al. 2013; Kendall and Quinn 2011; Lewis et al. 2015; Dorner et al. 2018).

There are several lines of evidence that DU2 (LFR-Harrison) has time varying productivity based on information from (1) a simple Ricker stock-recruitment analysis, (2) egg to age-2 survivals, (3) smolt to age-2 CWT survivals, (4) female lengths, (5) population level maturation rates, (6) population level average fecundity, (7) brood year recruitment patterns, and (8) oviposition.

### **F.1. SIMPLE RICKER STOCK RECRUITMENT ANALYSIS**

Ricker (1973, 1975) described a simple approach to estimate the relationship between the spawning stock and recruitment for the average environmental and population conditions that occurred during the period when the data were collected. The approach produces a stock-recruitment relationship and a set of model residuals that represent spawner density-independent annual variations in productivity, relative to the average during the time period of the data (Figure F1). For DU2 (LFR-Harrison), the simple Ricker model identifies periods of above average productivity (brood years 1986—1990 and 1994—2003) and below average productivity (1991—1993 and 2004—2013). There are temporal patterns among the residuals, as identified by groups of positive or negative values, however there was no statistically significant autocorrelation detected at lag-1 ( $P = 0.22$ ). Dorner et al. (2018) identified time trends in productivity and reported that the productivity for many of the North American Chinook salmon stocks varied temporally with variation in the North Pacific Gyre Oscillation, and to a lesser extent with the location of the bifurcation in the North Pacific Current as it reaches the west coast of North America.

### **F.2. EGG TO AGE-2 COHORT SURVIVAL**

The survival from egg to age-2 life stage was estimated for brood years 1984—2013 to identify temporal patterns that may contribute to time varying productivity for DU2 (LFR-Harrison). The egg-age-2 survival is a combination of density-dependent and density-independent mortality sources. Egg production was estimated from the mean fecundity by age and the abundance of female Chinook spawners by age (excluding oviposition), whereas the abundance of natural-origin age-2 fish before natural mortality was estimated from the number of natural-origin spawners by age, CWT exploitation rates by age, and backward cohort analysis. This approach does not produce the same values as the recruitment time series used in the stock-recruitment analysis, but it can be thought of as a different measure of recruitment (i.e. abundance of fish at the age-2 life stage prior to any fishing or natural mortality at ages 2 and older). The survival rate was the abundance of natural-origin age-2 fish divided by egg production by cohort. For DU2 (LFR-Harrison), there are no abundance estimates for specific life stages of natural-origin fish until age-2. No survival estimate was generated for 2004 because no Harrison CWT data were available for this cohort.

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The egg-age-2 survival has a temporal pattern (Figure F2) that resembles the temporal variation of productivity, as indicated by the residuals from the simple Ricker stock-recruitment model (Figure F1). Survival was above average for brood years 1986—1990, below average from 1991—1993, generally above average from 1994—2000, and then generally below average from 2001—2013. Most (83%) of the unexplained variation in productivity from the Ricker stock-recruitment relationship is represented by egg-age-2 survival (Figure F3), thus a relatively minor amount of the variation in productivity arises from survival variation for older life stages and measurement errors (<17% combined). The egg-age-2 survival had significant autocorrelation at lag-1 ( $P = 0.034$ ).

### **F.3. SMOLT TO AGE-2 CWT SURVIVAL**

The survival from the smolt to age-2 life stages was indexed using CWT-marked hatchery fish released into the Harrison River, and these survival rates are used as a surrogate for the survival of natural-origin fish, since there are no abundance estimates of natural origin smolts emigrating from the Harrison River. Survival from the smolt life stage to age-2, the youngest age of maturity and the age when DU2 (LFR-Harrison) Chinook recruit to fisheries, can represent much of the temporal variation in productivity, as indicated by the simple Ricker model residuals for Chinook Salmon (PSC 1999; Brown et al. 2001; Parken et al. 2006).

For DU2 (LFR-Harrison), the smolt-age-2 CWT survival varies substantially and had a temporal pattern of above average (1986—1990), below average (1991—2004) and random above and below average survivals (2005—2013; Figure F4). The temporal pattern of the smolt-age-2 survival represented 38% of the variability in productivity, as represented by the Ricker residuals (Figure F5). There was no significant autocorrelation among the smolt-age-2 CWT survival data at lag-1 ( $P = 0.879$ ).

There can be temporal patterns in smolt-age-2 survival because of ecological factors affecting the abundance of prey and the degree of predation on juvenile Chinook Salmon. In a meta-analysis of Pacific Northwest Chinook stocks, Sharma et al. (2013) reported that local ocean conditions following the outmigration path of Chinook smolts had an effect on smolt-age-2 survival for the majority of stocks, and they also reported linkages with time lags for other environmental indices from local to distant scales. These environmental indices can influence the abundance of prey, whereas the abundance of predators (Beamish and Neville 1995) could also contribute to temporal variation in smolt survival.

### **F.4. FEMALE LENGTH**

Fecundity is related to the length of reproductive females in fishes (Barneche et al. 2018), including Chinook Salmon (Healey and Heard 1984). Female length has been declining for Chinook Salmon in many parts of North America, including Alaska (Lewis et al. 2015, Ohlberger et al. 2018, 2020), Washington and Oregon (Ohlberger et al. 2018), British Columbia (DFO 2018) and the Fraser River (Xu et al. 2020). Average length-at-age has varied temporally for female Chinook in DU2 (LFR-Harrison) for ages 3 to 5, with a period of above average length (1996—2010) and more recently below average length (2011—2019; Figure F6). Significant autocorrelation was detected for age-3, -4 and -5 females at lag-1 (and others;  $P < 0.018$ ). For DU2 (LFR-Harrison), the growth rate of age-3 fish was associated with a local environmental index, the spring salinity at Entrance Island, B.C. near the mouth of the Fraser River, whereas the growth rates for age-4 and age-5 Chinook were associated with broad scale environmental indices, the Aleutian Low Pressure Index and the North Pacific Gyre Oscillation (Xu et al. 2020). As these different environmental indices each have a specific temporal pattern, Xu et al. (2020) developed models to predict age-specific growth rates for different Fraser Chinook management units. Since fecundity is related to female length for Chinook salmon, the temporal variation in

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female length-at-age could influence the temporal pattern in productivity for DU2 (LFR-Harrison).

## F.5. MATURATION RATE

Maturation rates describe the proportion of fish in a salmon population that mature at a specific age. The oldest mature age has a maturation rate of 100%, thus these rates do not vary for DU2 (LFR-Harrison) at age-5. Above average maturation rates imply that a higher than average proportion of fish mature in the age class for that specific cohort. Thus, the maturation rates can influence the age composition in the escapement, and for example, when maturation rates increase for all ages, the average age of the population decreases. Maturation rate patterns differ between males and females, based on differences between the age composition of males and females from the same cohort; however, maturation rates are only measured for both sexes combined in a cohort because fishery exploitation is calculated at the population level since fishery harvests are not measured by sex. Although the maturation rates are not measured specifically for female cohorts, temporal variation in the population level maturation rates can indicate that shifts are occurring in the age composition of female spawners. The age composition of the female spawners can affect the productivity of specific cohorts since younger fish are on average smaller than older fish, and for Chinook Salmon fecundity increases with age and size (Healey and Heard 1984).

For DU2, the maturation rates vary temporally for ages 2 to 4 for natural-origin Chinook (Figure F7), but no significant autocorrelation was detected at lag-1 ( $P > 0.079$ ). For DU2 (LFR-Harrison), females mature at ages 3, 4 and 5, whereas males mature at ages 2 to 5. For age-3 Chinook, maturation rates were generally below average from 1984—1988, near average from 1989—1995, below average from 1996—2003, and above average from 2006—2013. For age-4, maturation rates were below average from 1984—1987, above average from 1988—1991, below average from 1992—2000, and above average from 2001—2013. For age-3 and -4, maturation rates were above average during 2006—2013, which indicates DU2 (LFR-Harrison) Chinook are maturing earlier and returning to spawn at younger ages recently. The temporal patterns evident in the maturation rates could influence the temporal variation in productivity.

## F.6. FECUNDITY

Many species of fish have a disproportionate increase in fecundity with increasing size (Barneche et al. 2018). Relative to other fish species, fecundity increases at a slower rate per unit length for Chinook Salmon with length often representing less than 50% of the fecundity variation and other factors, such as stock, age at maturity, and year contributing to fecundity variation (Healey and Heard 1984).

To calculate average fecundity per female for DU2 (LFR-Harrison), first a stock-specific relationship between length and fecundity was developed from Harrison River fish measured during 2018 and 2019. Then the relationship was applied to the mean length of female Chinook at age by brood year, described previously. Next, the age-specific fecundity values were weighted by the age composition of the female escapement to create a time series of population level fecundity for DU2 (LFR-Harrison).

For DU2 (LFR-Harrison), there is a strong temporal pattern, with significant lag-1 autocorrelation ( $P < 0.0005$ ; Figure F8). Fecundity appeared to vary randomly during 1984—1997, but then the pattern changed to above average fecundity during 1997—2009, and then below average fecundity from 2010—2019. Although the average fecundity is influenced by the mean length of females at ages 3 to 5 and the maturation rates and age composition of female spawners from

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ages 3 to 5, there is a much more distinctive temporal pattern for fecundity than either female length or maturation rates.

### **F.7. BROOD YEAR RECRUITMENT**

The brood year recruitment, which contributes to catch and escapement in subsequent years, had a temporal pattern for DU2 (LFR-Harrison), but no significant autocorrelation at lag-1 ( $P = 0.262$ ; Figure F9). For brood years 1984—1990, there were several large recruitments that experienced high simple exploitation rates (i.e. Total Mortality/(Total Mortality + Escapement)), averaging 70%, including both US and Canadian fisheries. For brood years 1991—1997, recruitments were generally below average (Figure F10) and exploitation rates averaged 42%, until recruitments increased above average for brood years 1998—2000 when exploitation rates averaged 53%. Since then, brood year recruitment has been below average and exploitation rates averaged 38%. Recruitment did not appear to vary randomly through time.

### **F.8. OVIPOSITION**

Oviposition is the expulsion of eggs from the female spawner's body into the environment where the redd is constructed. On the spawning grounds, the body cavities of female Chinook salmon carcasses have been examined for egg retention, recorded as 100%, 50% or 0%, where the 50% category represents a volume of eggs that would fill a person's hands when cupped together. These categorical data were converted to the percentage oviposition for each brood year of spawners. Oviposition had a temporal pattern for DU2 (LFR-Harrison), but no significant autocorrelation at lag-1 ( $P = 0.391$ ; Figure F11). There was below average oviposition during 1986-1995, and since then most years have had above average oviposition. Oviposition does not appear to vary randomly through time.

### **F.9. SUMMARY**

Temporal patterns were identified among seven types of factors affecting population dynamics that provide evidence that productivity is more likely to be time varying than randomly varying for DU2 (LFR-Harrison) over brood years 1984 to 2013. There were brief stanzas of random variation apparent with some of the factors (i.e. smolt-age-2 CWT survival, fecundity) and significant autocorrelation at lag-1 was only detected in the egg-age-2 survival, female lengths for ages 3, 4, and population level fecundity. Although we did not detect any significant autocorrelation at lag-1 for the simple Ricker model residuals, there were temporal patterns that aligned most strongly with the temporal pattern for egg-age-2 survival and smolt-age-2 CWT survival, with these survival rates representing 83% and 38% of the Ricker residual variation, respectively. Similar ongoing and widespread declining trends in body size, age at maturity and fecundity have been documented in Alaskan Pacific Salmon populations, with the largest changes observed in Chinook Salmon (Oke et al. 2020). These trends were associated with climate change and marine competition from salmon (Oke et al. 2020), providing further evidence that these observed trends are not the result of random variation.

Overall, the weight of evidence for DU2 (LFR-Harrison) suggests that time varying productivity with a recent period of depressed productivity is the most likely scenario. This is supported by recent research that documented synchronous temporal patterns in Chinook Salmon productivity across the Northeastern Pacific, with recent declines in productivity across most of the stocks (Dorner et al. 2018). This biological information is important to consider along with statistical diagnostics during the model selection process.

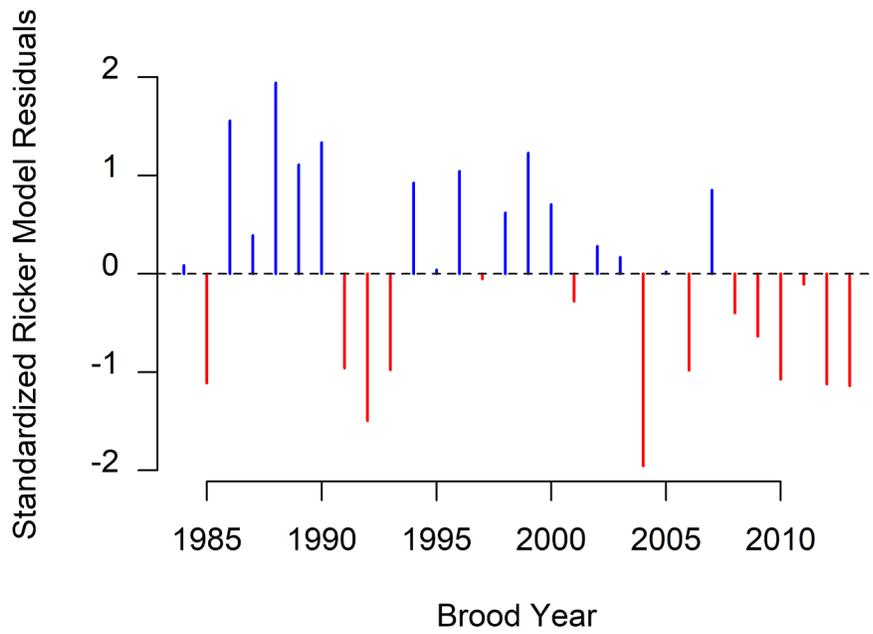


Figure F1 - Standardized (Z-scores;  $[(obs-mean)/SD]$ ) residuals from the simple Ricker model for DU2 (LFR-Harrison).

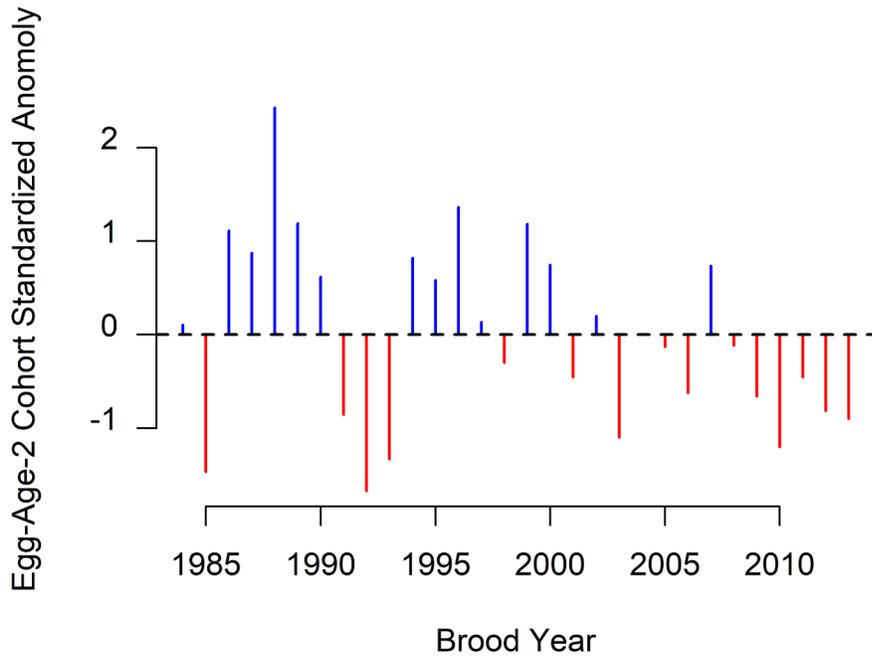


Figure F2 - Standardized survival anomalies (Z-scores;  $[(obs-mean)/SD]$ ) for survival from the total number of eggs to natural-origin age 2 cohort abundance for DU2 (LFR-Harrison).

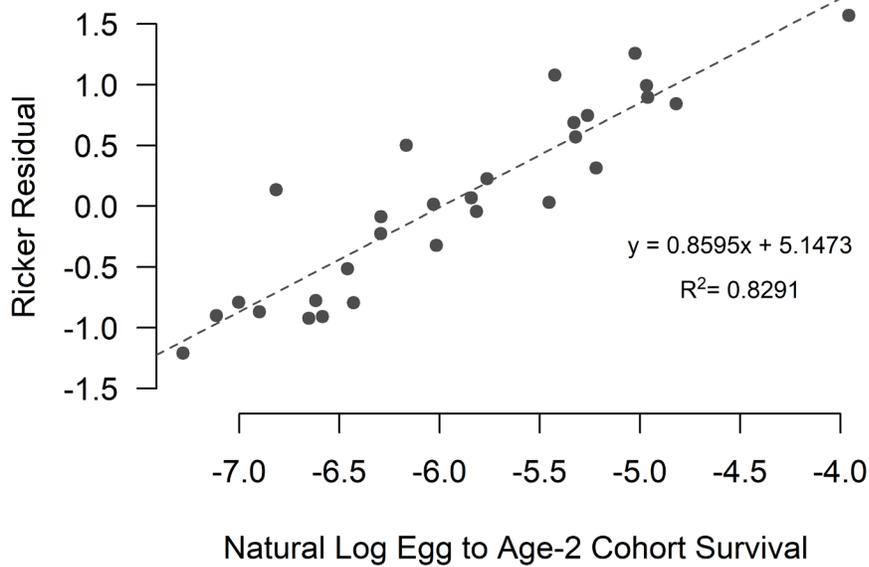


Figure F3 - Scatterplot of natural log transformed egg to natural-origin age-2 cohort abundance and residuals from Ricker stock-recruitment relationship for DU2 (LFR-Harrison).

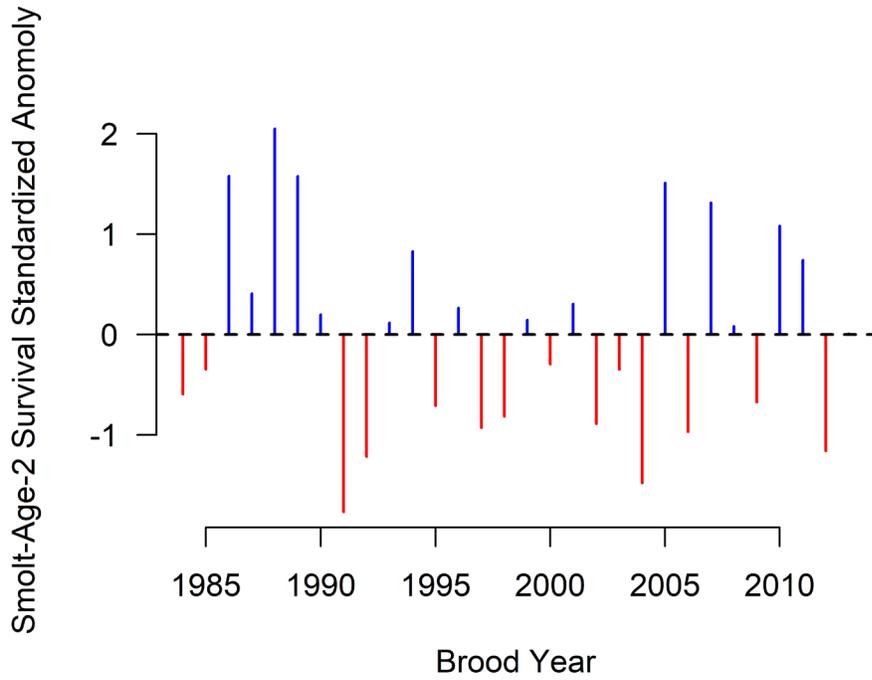


Figure F4 - Standardized survival anomalies (Z-scores;  $[(obs-mean)/SD]$ ) for natural log transformed survival estimated from the number of CWT-marked hatchery fish released into the Harrison River and the hatchery-origin age 2 cohort abundance for DU2 (LFR-Harrison).

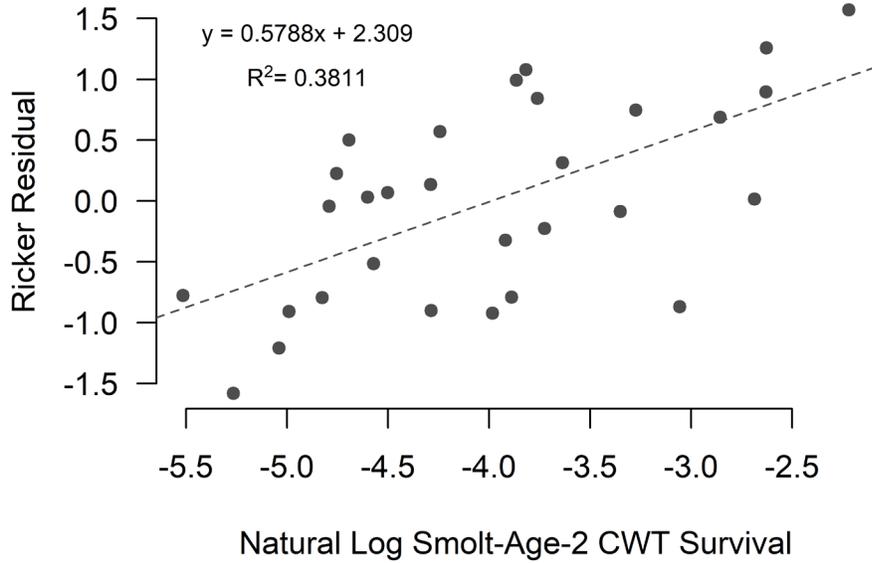


Figure F5 - Scatter plot of natural log transformed smolt-age-2 survival, for CWT-marked hatchery-origin fish released into the Harrison River, and the residuals from the simple Ricker model.

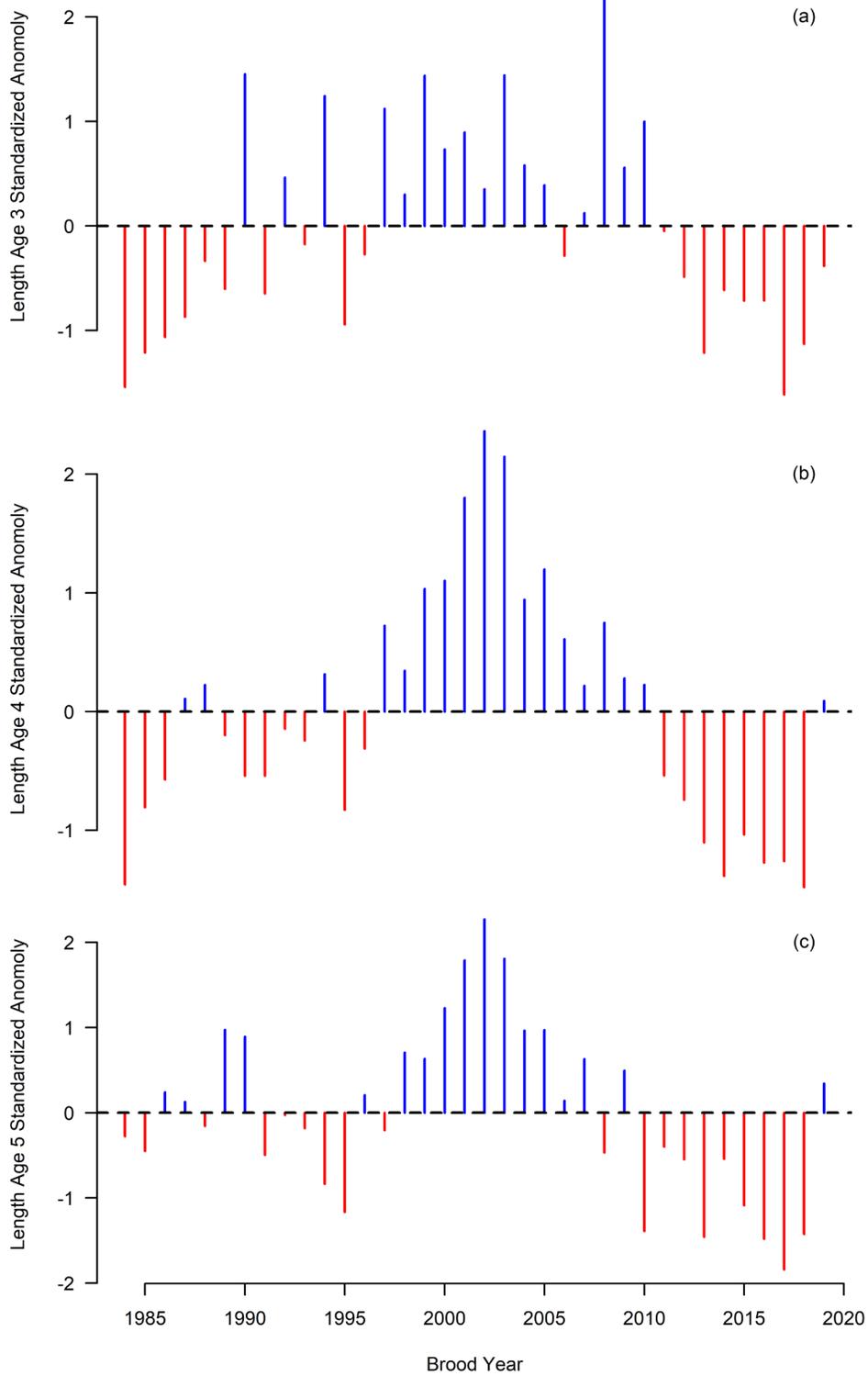


Figure F6 - Mean female length standardized anomalies (Z-scores;  $[(obs-mean)/SD]$  for ages 3 (a), 4 (b) and 5 (c) Chinook Salmon for DU2 (LFR-Harrison).

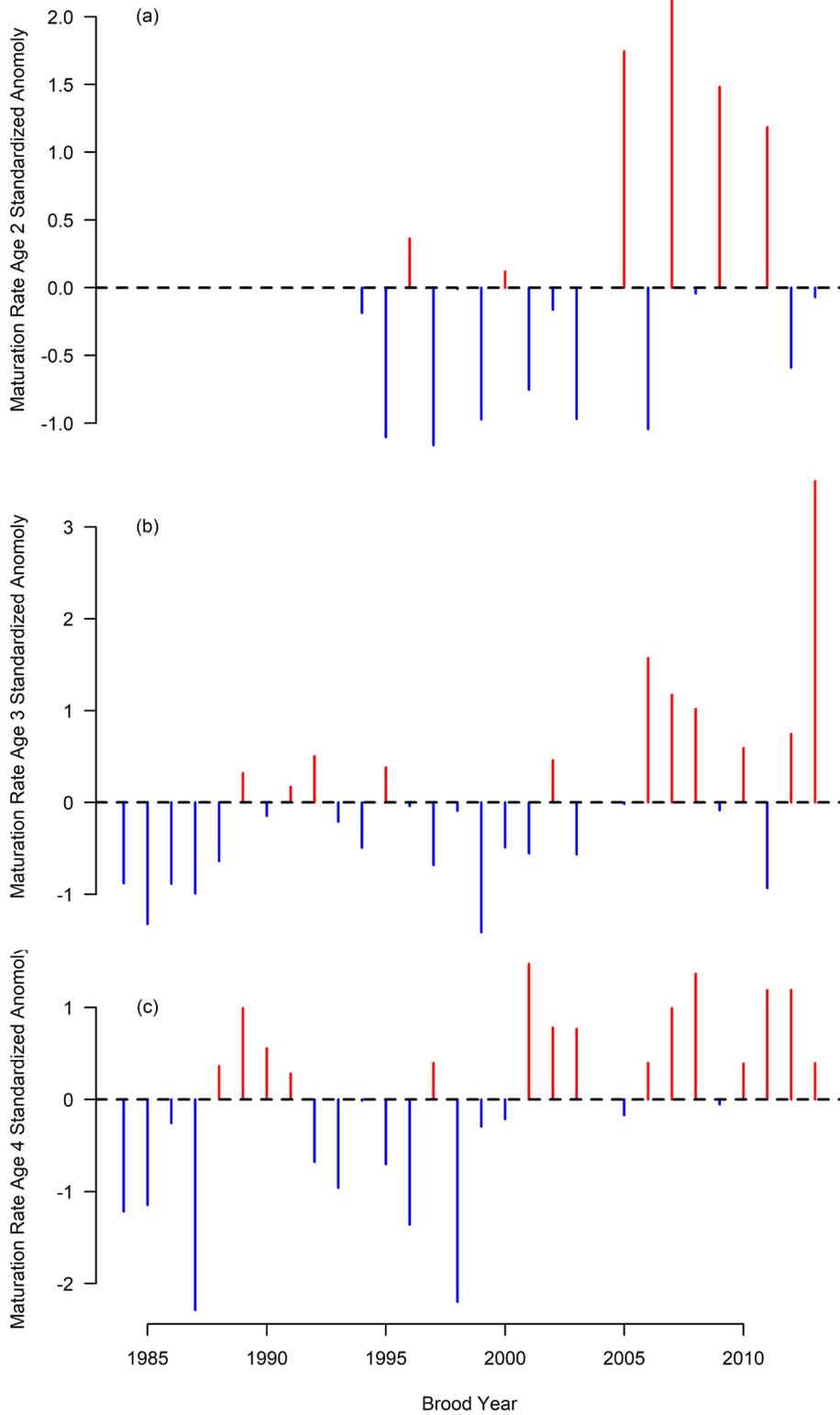


Figure F7 - Maturation rate standardized anomalies (Z-scores;  $[(obs-mean)/SD]$  for ages 2 (a), 3 (b), and 4 (c) Chinook Salmon for DU2 (LFR-Harrison).

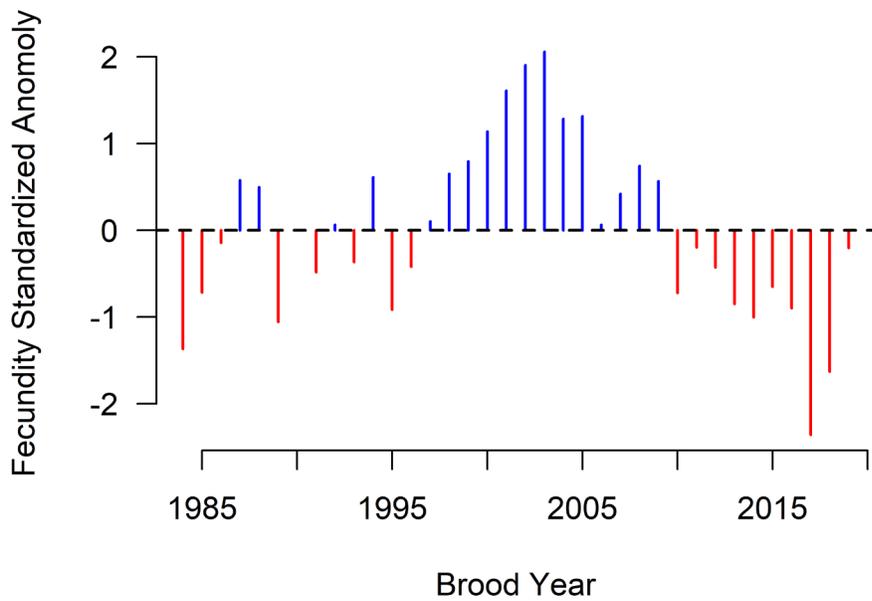


Figure F8 - Standardized anomalies for the population-level fecundity (Z-scores;  $[(obs-mean)/SD]$  for Chinook Salmon for DU2 (LFR-Harrison), where one standard deviation is 2.3% of the mean.

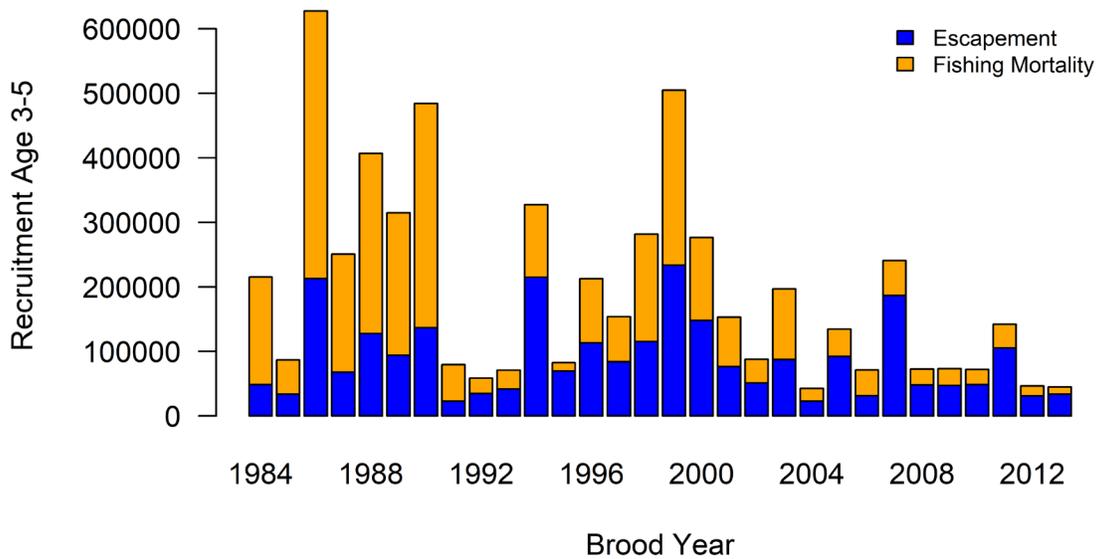


Figure F9 - The amount of the brood year recruitment estimated as fishing mortality (i.e. catch and incidental mortality) or as escapement for DU2 (LFR-Harrison).

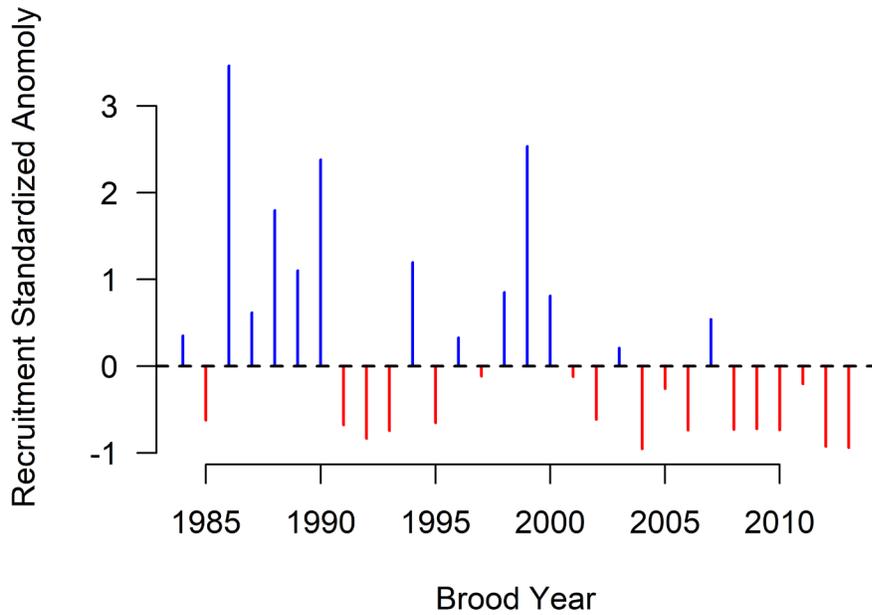


Figure F10 - Standardized anomalies for recruitment (Z-scores;  $[(obs-mean)/SD]$  of Chinook Salmon for DU2 (LFR-Harrison).

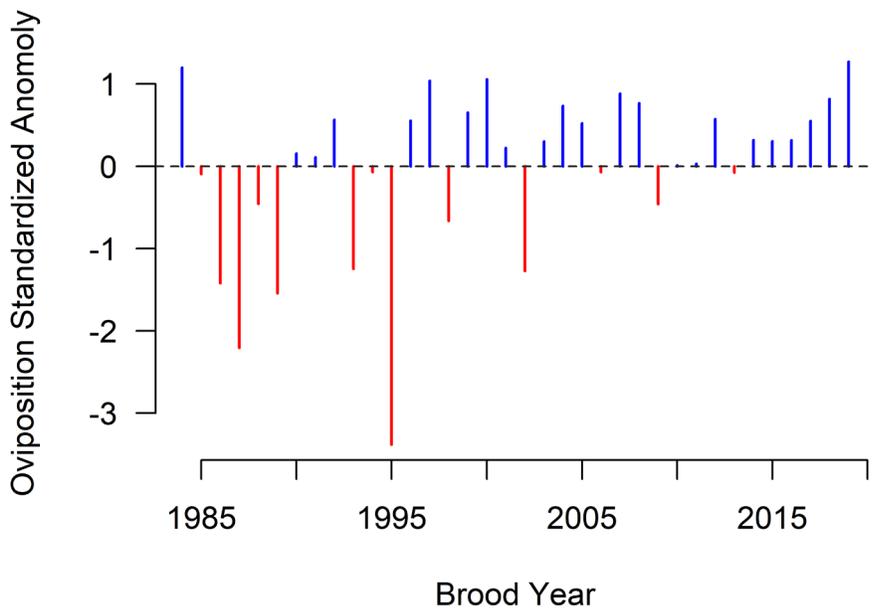


Figure F11 - Standardized anomalies for oviposition (Z-scores;  $[(obs-mean)/SD]$ ) for DU2 (LFR-Harrison) Chinook Salmon

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## APPENDIX G. RICKER MODELS FOR HARRISON DU2 (LFR-HARRISON)

To provide parameter estimates for the forward projections for DU2 (LFR-Harrison) in the RPA three different Ricker models were considered, including an auto-correlated model that was suggested during the CSAS process. Each of the models are briefly described below, followed by a short comparison between them.

### G.1. MODEL VERSIONS

In each of the subsections below, the form of the model and the parameter estimates are provided.

#### G.1.1. Ricker Simple

Simple Ricker model of the form:

$$R_t = \alpha S_t e^{-bS_t} e^{\epsilon_t - (\sigma^2/2)}$$

$$\epsilon_t \sim \text{Normal}(0, \sigma)$$

Which is modelled in its linearized form as:

$$\log\left(\frac{R_t}{S_t}\right) = a - bS_t + \epsilon_t - \sigma^2/2$$

Where  $a = \log(\alpha)$

$R_t$  is the abundance of adult recruits in year  $t$ ,  $S_t$  is the number of spawners that generated those recruits, the parameter  $a$  represents the log recruits-per-spawner as spawner abundance approaches zero (i.e., productivity), the  $b$  parameter represents the strength of density dependence per unit spawning biomass, and  $\epsilon_t$  represents normally distributed residuals around the spawner-recruit curve with a standard deviation of  $\sigma$ . This model provides the base for the following two models.

Table G1 - Parameter estimates for the simple Ricker estimated through a Bayesian framework.

Parameters	MLE	Lower 95 CI	Upper 95 CI	Median	Lower 95 CI	Upper 95 CI
$a$	1.59	0.98	2.21	1.50	0.95	2.22
$\alpha$	4.92	1.88	7.96	4.49	2.58	9.24
$b$	6.74E-06	1.70E-06	1.18E-05	5.45E-06	1.21E-06	1.11E-05
$S_{max}$	148444	37351	259537	183609	89992	828056
$\sigma$	0.80	0.60	1.00	0.84	0.66	1.12

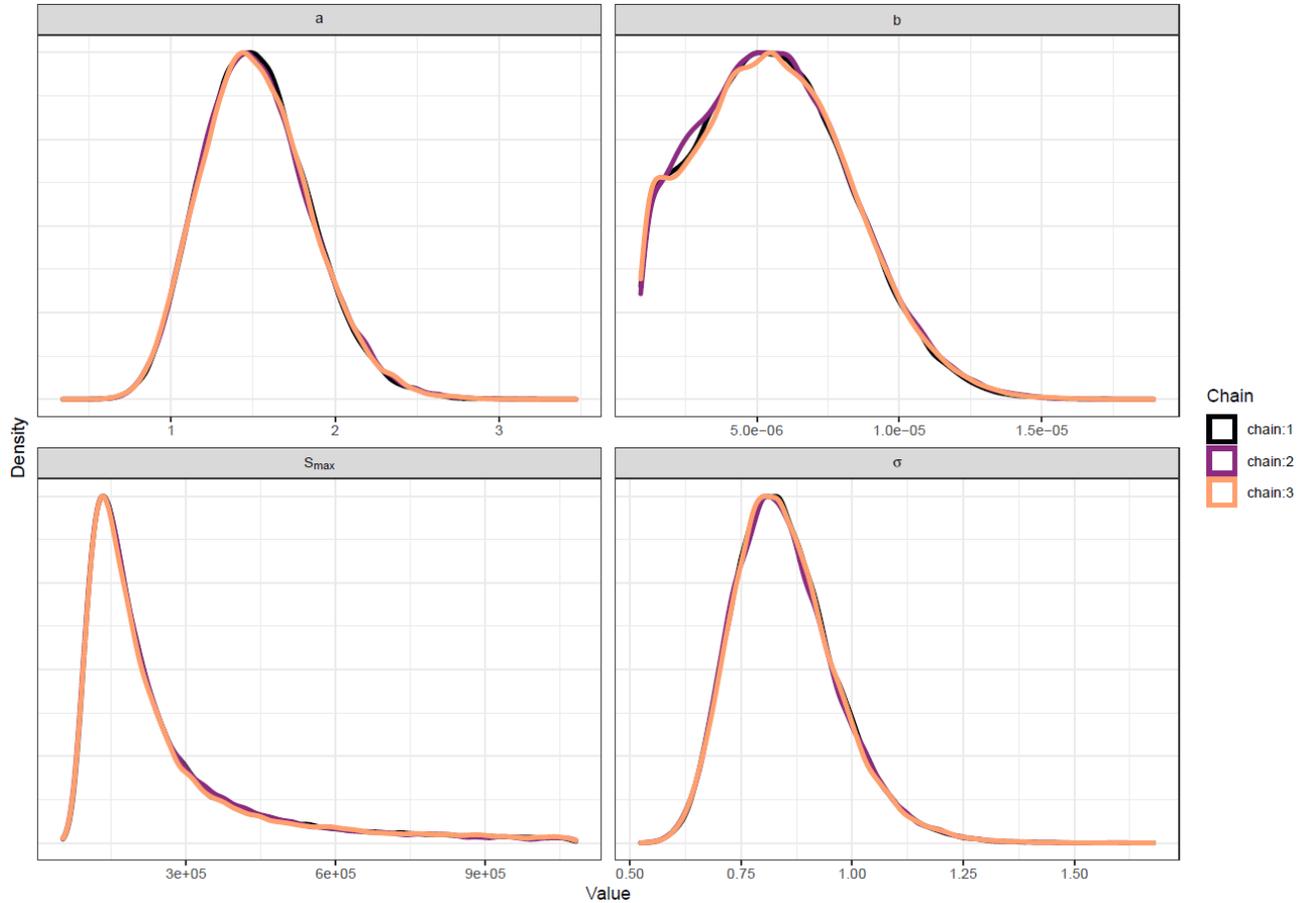


Figure G1 - Posterior probability distributions for the simple Ricker model corresponding to the median estimates in Table 1. Posteriors are based on 100,00 iterations from 3 MCMC chains with a burn-in period of 50,000 iterations.

### G.1.2. Ricker with Autocorrelation

Ricker model with autocorrelation in residuals of the form:

$$R_t = \alpha S_t e^{-b S_t} e^{\epsilon_t - (\sigma_{AR}^2/2)}$$

Where:  $\epsilon_t = \epsilon_{t-1} * \rho + \delta_t$

$$\delta_t \sim Normal(0, \sigma_{AR})$$

$$\sigma_{AR} = \sigma_{total} \sqrt{1 - \rho^2}$$

Which is modelled in its linearized form as:

$$\log\left(\frac{R_t}{S_t}\right) = a + \beta S_t + \epsilon_{t-1} * \rho + \delta_t - \sigma_{AR}^2/2$$

Where  $a = \log(\alpha)$

Where  $R_t$ ,  $S_t$ ,  $\alpha$ ,  $a$ , and  $b$  are as defined in the “Ricker Simple” model above and  $\rho$  is a coefficient representing first-order autocorrelation among annual recruitment residuals from year

t and year t-1. The standard deviation  $\sigma_{AR}$  represents the portion of the total standard deviation  $\sigma_{total}$  that is not accounted for by the autocorrelation process.

Table G2 - Parameter estimates for the Ricker with autocorrelation estimated through a Bayesian framework.

Parameters	MLE	Lower 95 CI	Upper 95 CI	Median	Lower 95 CI	Upper 95 CI
$a$	1.36	0.67	2.06	1.36	0.86	7.95
$\alpha$	3.91	1.20	6.62	3.91	2.36	7.95
$b$	4.83E-06	-8.74E-07	1.05E-05	4.12E-06	1.94E-06	9.67E-06
$S_{max}$	207046	-37466	451559	242551	103413	515685
$\rho$	0.24	-0.08	0.56	0.26	-0.07	0.53
$\sigma_{AR}$	0.80	0.58	1.01	0.82	0.64	1.09

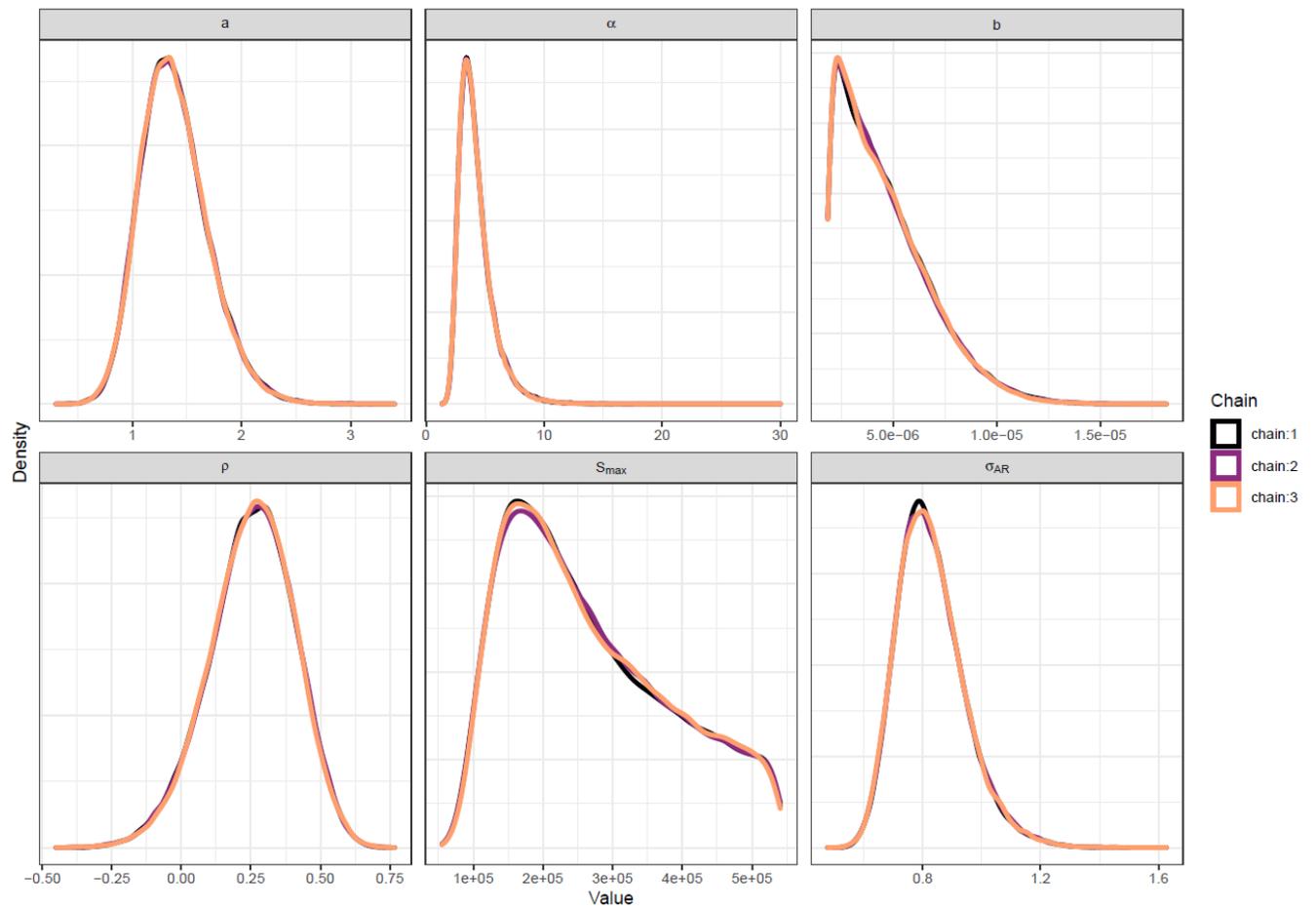


Figure G2 - Posterior probability distributions for Ricker model with autocorrelation corresponding to the median estimates in Table 2. Posteriors are based on 100,00 iterations from 3 MCMC chains with a burn-in period of 50,000 iterations.

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### G.1.3. Ricker With Time-Varying Productivity

The Ricker model with time-varying productivity allows the productivity parameter,  $a$ , from the “Ricker Simple” model to vary among years. This model is most easily interpreted in its linearized form:

$$\log\left(\frac{R}{S}\right) = a_t + \beta S + \epsilon_t - \sigma^2/2$$

$$a_t = a_{t-1} + \gamma_t$$

$$\gamma_t \sim \text{Normal}(0, \sigma_a)$$

$$\epsilon_t \sim \text{Normal}(0, \sigma)$$

where,  $R_t$ ,  $S_t$ , and  $b$  are as defined in the “Ricker Simple” model above, and  $a_t$  is the annual productivity of recruits in year  $t$ . The model assumes that  $a_t$  changes over time following a simple random walk, with standard deviation  $\sigma_a$ . In order to project forward, total standard deviation in the spawner-recruit relationship is calculated as follows:

$$\sigma_{total} = \text{sqr}t(\sigma^2 + \sigma_a^2)$$

Due to issues with model convergence, the total variance is split between process variance,  $\sigma$ , and variance in  $a$ ,  $\sigma_a$ , using a variable,  $\rho$ , which has a beta prior put on it (see section 5.1 of the Research Document for further details).

$$\sigma = \sqrt{\rho} * \sigma_{total}$$

$$\sigma_a = \sqrt{1 - \rho} * \sigma_{total}$$

Unlike the models above, which provide a single long term average estimate of productivity ( $a$ ), this model formulation produces an estimate of productivity for each year of the time series.

Table G3 - Parameter estimates for model with time-varying productivity estimated through a Bayesian framework. Each year a different  $a_t$  value is estimated, only the values for the last generation are shown in the table below, as those were the values used to calculate  $a_{avg}$ . Figure 6 shows the change in  $a_t$  over the whole time series.

Parameters	MLE	Lower 95 CI	Upper 95 CI	Median	Lower 95 CI	Upper 95 CI
$a_{2010}$	0.82	-0.03	1.68	0.85	0.17	1.70
$a_{2011}$	0.83	-0.02	1.69	0.89	0.19	1.76
$a_{2012}$	0.64	-0.23	1.52	0.70	0.09	1.56
$a_{2013}$	0.59	-0.37	1.54	0.66	0.06	1.59
$a_{avg}$	0.72	-0.01	1.46	0.77	0.30	1.49
$\alpha_{avg}$	2.06	0.55	3.57	2.17	1.35	4.44
b	5.98E-06	1.17E-06	1.08E-05	5.83E-06	2.36E-06	1.05E-05
$S_{max}$	167156	32556	301755	171451	95553	423287
$\rho$	0.67	0.24	1.09	0.66	0.33	0.91
$\sigma_{total}$	0.72	0.53	0.90	0.75	0.59	1.00

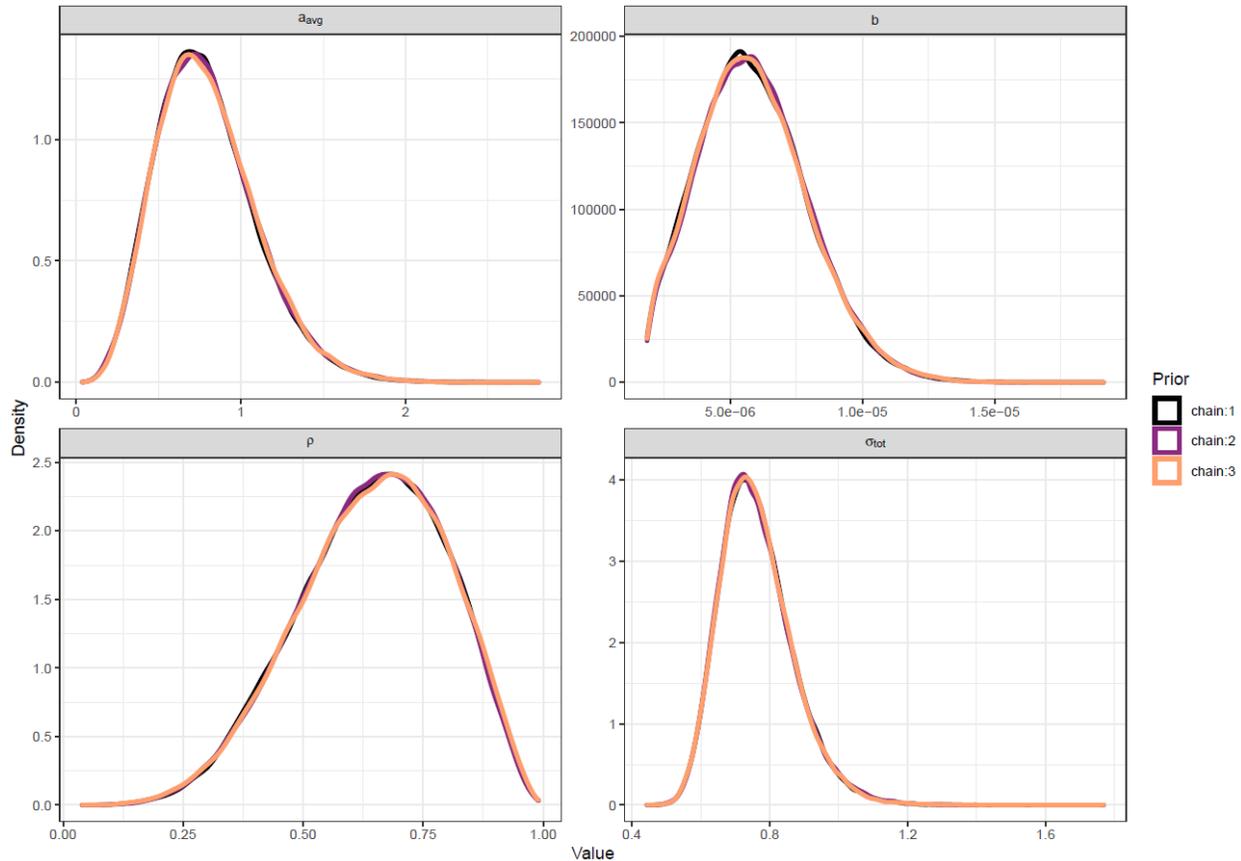


Figure G3 - Posterior probability distributions for the Ricker model with time-varying productivity corresponding to the median estimates in Table 1. Posteriors are based on 100,00 iterations from 3 MCMC chains with a burn-in period of 50,000 iterations.

## G.2. COMPARISON

Model comparison was based on visual inspection of temporal residual patterns as well as calculated Akaike Information Criterion (AIC) values for the maximum likelihood estimates. Residuals are the difference between the value predicted by the model and that of the observed data. The residuals from all three models were examined to see if there appeared to be trends in the residuals, if the model was tracking the changes seen in the data (i.e. if the points were distributed evenly above and below the zero line), the relative size of the residuals and if there were any outliers ( $<3$  SDs in standardized residuals). The second model selection criteria used, AIC, helps to identify the model that optimizes the trade-off between variance and bias (Burnham and Anderson 2002). When comparing models of similar quality the AIC criteria will identify the model that is the most parsimonious. In other words, it helps to identify the model that best explains the data using the fewest parameters. Time-series biases are common in standard Ricker models, and may be at least partially alleviated in time-varying models that account for trends in productivity. Holt and Michielsens (2020) found that time-varying models provided parameter estimates that were less biased than standard Ricker models under a variety of historic patterns in productivity and exploitation rates, despite AIC model selection criteria favoring the standard Ricker model. They suggested that AIC may not be appropriate for selecting between standard and time varying Ricker models.

Of the three models considered here, the state-space model with time-varying productivity is the one with the lowest residual values and absence of a temporal pattern (Figure G4). This

indicates that the time-varying model is doing the best job of tracking the changes we have observed in recruitment events over time. In particular, it more closely estimates the most recent generation which is important when we are trying to forward project what will occur if recent conditions persist. The AIC values, however, indicate that the simple Ricker is the most parsimonious model (the lowest AIC value; Table G4), despite more marked temporal residuals resulting from that fit (Figure G4).

Given the differing results of the diagnostics, there is no definitive statistical basis to select either of these models over the other. However, as mentioned above, there are concerns with using AIC with time-varying models, which could bias the results of that diagnostic. Overall there is more support for the time-varying model empirically because it had the smallest absolute residual values with no temporal pattern in residuals, and more broadly because of significant biological evidence of changing productivity for DU2 (LFR-Harrison) consistent with coast-wide patterns in Chinook populations (see Appendix F). Future stock recruit modelling should explore the possibility of using survival to age-2 as a co-variate in the Ricker model for Harrison, as preliminary analyses indicate that it may be promising for future stock recruit modelling.

These statistical diagnostics are informative of how the model fits the data, but provide little insight on the potential performance of these models for management advice (Walters and Martell 2004). More recently in the fisheries science literature, there has been a push to base model selection for management advice on closed-loop simulation and management strategy evaluation analyses (Butterworth 2007; Punt et al. 2016). Additionally, future analysis could consider different methods of model selection, such as cross validation approaches. These analyses are beyond the scope of the current RPA given time constraints.

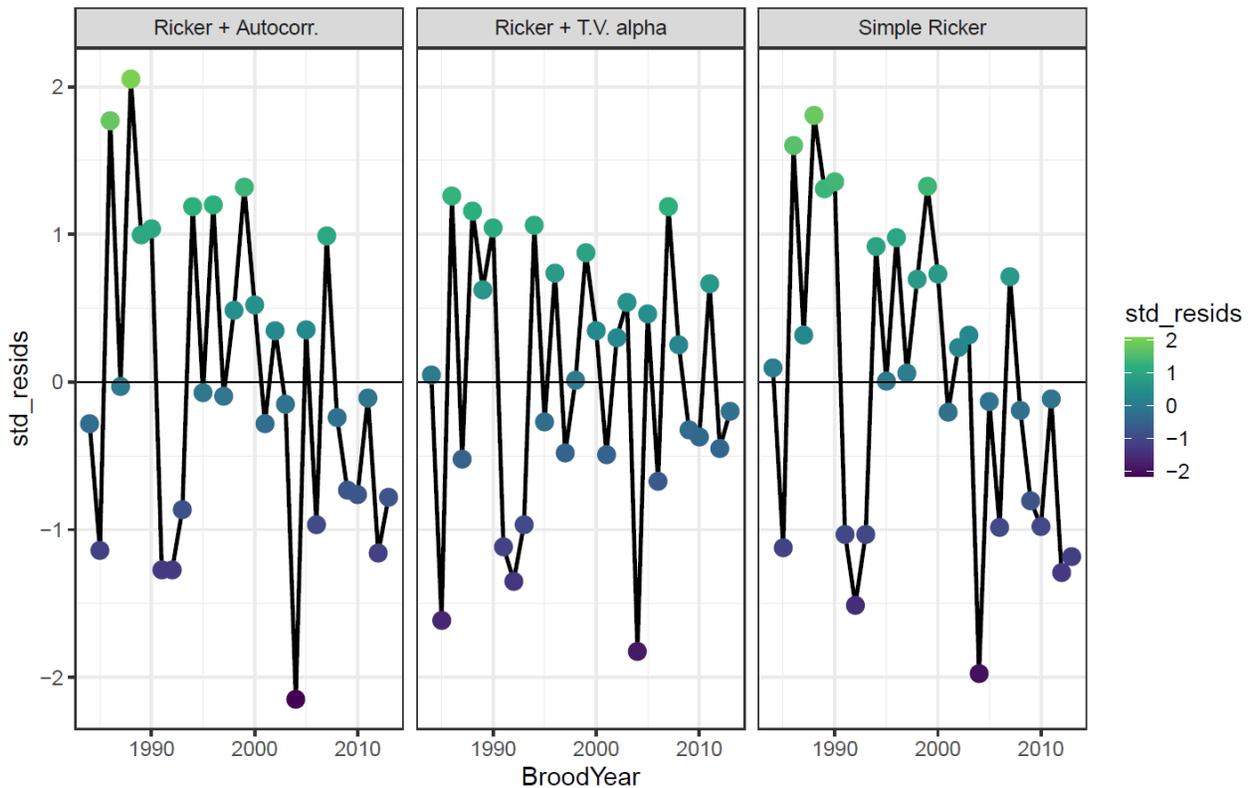


Figure G4 – Standardized residuals vs. Brood year comparison over years with all models.

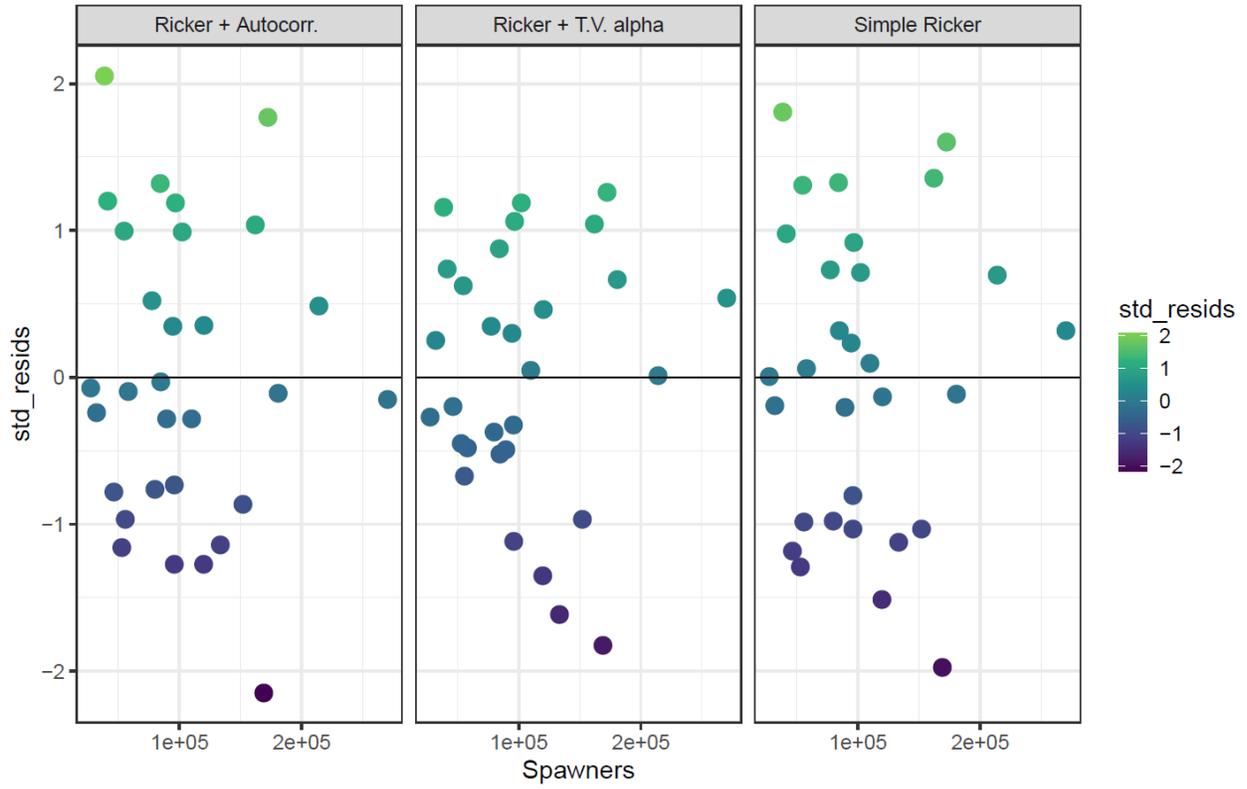


Figure G5 – Standardized residuals vs Spawners comparison over years with all models.

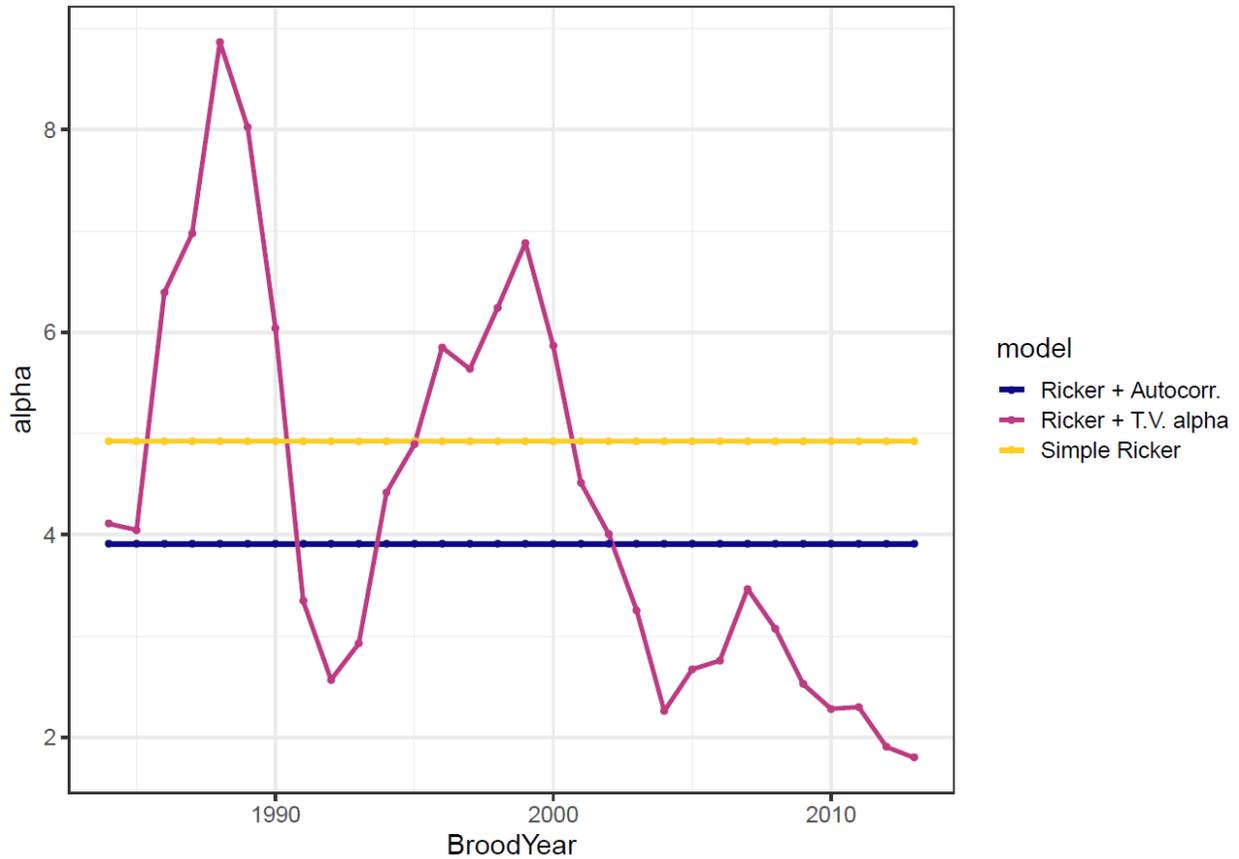


Figure G6 - Productivity comparison across models.

Table G4 - AIC comparison between the three Ricker models.

Model	AIC	deltaAIC
Simple Ricker	77.57	0
Ricker with Autocorrelation in Recruitment	77.63	0.06
Ricker with Time-Varying Productivity	80.54	2.97

### G.3. REFERENCES

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Punt, André E, Doug S Butterworth, Carryn L de Moor, José A A De Oliveira, and Malcolm Haddon. 2016. "Management Strategy Evaluation: Best Practices." *Fish and Fisheries*, 303–34. DOI: 10.1111/faf.12104

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The repository for this work is available from [Github](#)

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## APPENDIX H. FRASER RIVER CHINOOK - RESEARCH NEEDS

This section provides a summary list of research needs identified during this RPA process, many of which were discussed in detail in the threats assessment in Part 1. In some cases, there is developing and ongoing research in these areas through various organizations, academia, and different levels of federal and provincial governments. For a more in-depth summary of the literature surrounding these topics, please refer to Part 1 of the RPA. The [Science Advisory Report \(2020/023\)](#) and [Research Document \(2021/063\)](#) are available online.

### H.1. FRESHWATER HABITAT

- There is a need to expand our knowledge of Chinook habitat use in the mainstem of the Fraser River. Surveys in the mainstem of the lower Fraser River (e.g. near Agassiz) have identified this as important rearing habitat for many Fraser Chinook DUs. There is some, albeit limited knowledge of habitat use in the mainstem Fraser, but there is more opportunity to gain a better understanding of DU-specific life history and the temporal and spatial aspects of habitat use.
- Previous studies have reported physiological limitations of FRC for turbidity that are lower than levels observed in some systems known to contain Chinook Salmon. This has likely led to an under-estimation of freshwater habitat use within the Fraser drainage, and future research should aim to investigate both the physiological limits of Fraser Chinook for turbidity, and habitat use in turbid systems thought to contain juvenile FRC.
- There is a growing body of information indicating that climate change will lead to an earlier spring freshet, which can impact migration and affect the quantity, availability and quality of freshwater rearing habitats. Considerable research could help understand the implications of changes in timing and duration of the spring freshet.
- There have been massive losses in forest cover in the Fraser River drainage through logging, wildfires, and pest infestations. Studies are required to investigate alternate reforestation strategies to address optimizing watershed rehabilitation and restoration, while taking into account climate change, fire and pest resilience and future fibre supplies.
- Research is needed to better characterize Fraser Chinook freshwater distribution and suitable habitat supply at the DU level. Element 14 of the RPA aims to provide advice on the status of habitat *supply* and *demand*, and to inform discussion about whether habitat availability is currently limiting population growth. This element was not addressed in the RPA (see section 8), and will require considerable study of fry dispersal, behaviour, densities, and survival. This information can then be used to coordinate habitat preservation and/or restoration efforts for FRC.
- Historical development in the lower Fraser River has led to losses in off-channel and stream habitat, and reductions in floodplain connectivity has likely reduced the freshwater carrying capacity for Fraser River Chinook. Research is therefore needed to understand the potential mitigation effects of reconnecting off channel habitat, particularly in the lower Fraser River.
- Research is needed to gain a better understanding of spawning levels and spawner distribution at the DU level.

### H.2. MARINE HABITAT USE

- There is limited available data on the marine distribution and habitat use of Fraser Chinook due to the vast areas that they inhabit in the Pacific Ocean. Much of the available data relate to recoveries in fisheries and little is available in terms of pre-fishery distributions. There are some CWT recovery data available for areas along the Pacific Coast outside of Pacific

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Salmon Treaty waters, and up into the Bering Sea, however these data are limited and inconsistent over time. While large-scale tagging studies are difficult to approach for a variety of logistical reasons, future research should aim to increase our knowledge of Fraser Chinook marine distribution to better manage fishing activities and marine protected areas.

- It would be beneficial to determine if there are “carrying capacity” bottlenecks in the nearshore and distant marine habitats, and what (if anything) could be done to alleviate those constraints on production.

### **H.3. ABUNDANCE AND LIFE HISTORY PARAMETERS**

- Due to a lack of indicator stocks for many Fraser Chinook DUs, current productivity, survival, and biological data are limited or non-existent. In addition, abundance estimates for many DUs rely heavily on indices of relative abundance, and in some cases, may not be representative of the DU as a whole (i.e. DU4 LFR-Upper Pitt, DU5 LFR-Summer, and DU16 NTh-Spring). As a result, our current understanding DU-level population trends are highly uncertain for these DUs. Obtaining this information will be difficult due to the logistic challenges associated with developing CWT programs. If possible, through CWT (or other) programs, future research should aim to investigate the following at the DU level:
  - Absolute abundance estimates
  - Biological sampling of spawners
  - Stock recruit time series data
  - Freshwater and marine survival
  - Length at age
  - Changes in fecundity
  - Maturation rates
  - Trends in age proportions of returns

### **H.4. POLLUTION**

- The effects of pollution at all life stages was identified as a major knowledge gap for Fraser Chinook. There are many sources of contaminants in the Fraser River drainage and along the Pacific coast (both current and historic) that impact Fraser Chinook, many of which have been shown to have negative effects on various Pacific salmon populations in both Canada and the US. These contaminants were broken into the following categories in Part 1 of the RPA (see Part 1 for further breakdown of contaminants within these categories):
  - Household Sewage & Urban Waste Water
  - Industrial & Military Effluents
  - Agriculture & Forestry Effluents
  - Garbage & Solid Waste
  - Air-Borne Pollutants

It is critical to understand the numerous and dynamic sources and effects of these contaminants for future FRC mitigation and recovery planning. Considerable research is needed in order to inventory and prioritize pollution risk and subsequent mitigations, and should be considered at the individual DU-level.

### **H.5. ENHANCEMENT**

- All enhancement activities occurring within FRC DUs need to be reviewed to ensure that objectives and protocols are aligned with the conservation strategies and recovery of these DUs.

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- Competition between hatchery-origin and wild fish can occur at all life stages and in all habitats, the latter of which may be limiting in the lower Fraser River and estuary due to reductions in habitat caused by extensive historical development. High levels of hatchery production may therefore lead to increased competition for finite and limited resources, particularly for Fraser Chinook DUs that have similar life histories to those that receive high levels of enhancement (i.e. ocean-type Fraser Chinook). While there are some studies available that attempt to characterize these interactions, further research is needed to determine the risk of hatchery competition in the Fraser River drainage, and identify the carrying capacity of estuarine habitats.
  - There is a need to investigate the extent of genetic introduction into DUs from outside of those populations. Genes can be introduced by the straying of hatchery fish from other enhanced populations, which has been observed for DU2 (LFR-Harrison) with the detection of hatchery-origin fish on spawning grounds from the Cowichan River and Robertson Creek hatcheries. Deliberate introductions have also occurred, such as the re-introduction of genes that were previously selected out over a period of time under different conditions. The impacts from introduction of genes from hatchery-origin fish, in addition to stock transfers, and the use of stored genetic materials should be thoroughly investigated.

## **H.6. LIVESTOCK RANCHING**

- There is a need to investigate and monitor the extent of stream bank/bed impacts from livestock ranching in the Interior Fraser. Cattle were frequently identified as a threat to smaller stream systems within some DUs (e.g. DU9 MFR-Spring, DU14 STh-Bessette), and cattle are routinely observed in and around streams within spawning areas during aerial and ground spawner surveys. Despite regulations surrounding the use of fences to prevent cattle from entering streams, enforcement is difficult and often lacking within the middle and upper Fraser River DUs (DUs 9 MFR-Spring, 10 MFR-Summer, and 11 UFR-Spring) where cattle are often observed in streams. The presence of cattle in streams poses a risk of increased streambank erosion, sedimentation, and the potential for trampling redds. There are few available studies looking at the direct effects on Chinook Salmon.

## **H.7. SHIFTS IN PREDATOR/PREY SPECIES INTERACTIONS**

- With rapidly changing climatic conditions, there will likely be a continued shift in predator/prey species composition within both freshwater and marine environments. There is a need to better characterize the changes and understand the implications to future Chinook Salmon production. Examples of this are the changing distribution of zooplankton prey species with warming ocean temperatures and the recent increase of coastal jellyfish populations, both of which could change prey availability.
- The distribution of marine predators of Chinook may be shifting due to warming ocean temperatures. An example of this is the presence of salmon sharks in the Bering sea, where the onset of colder ambient water temperatures were generally thought to drive some predators out of these cold habitats as winter sets in. There is a need to better understand the abundance and distribution of large predators such as salmon sharks, in addition to the magnitude of late ocean mortality of Chinook Salmon from predation by large predators.
- There are significant knowledge gaps in the abundance and population trends of a variety of co-occurring freshwater predators, such as pikeminnow, seals, and river otters that may also be contributing to declining trends in abundance. Future research is needed to better our understanding of these predatory interactions for juvenile and adult Fraser Chinook, and the magnitude of these effects.

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- Research is needed to investigate the impacts of pinniped (in particular Harbour Seals, Stellar Sea Lions, and California Sea Lions) predation on FRC, particularly for low-abundance DUs in which predation effects could be significant. There has been increasing pressure in recent years to reduce pinniped numbers by conducting a cull, however, further research is needed to understand the indirect effects of conducting a cull in addition to other factors that influence ecosystem function such as food web relationships, shifting prey/predator distributions, and hatchery practices. Further to this, with our limited understanding of both Pacific Salmon and pinniped population dynamics, we have little capability in determining whether removals would produce the intended effect.

## **H.8. INVASIVE SPECIES**

- The timing of invasion and establishment of invasive species was identified as a significant knowledge gap for all Fraser DUs, and should be considered in mitigation planning. There are a number of invasive fish species that may have detrimental impacts on juvenile Fraser Chinook abundance, including Largemouth and Smallmouth Bass, Yellow Perch, Pumpkinseed, Black Crappie, Bullhead, and Northern Pike, in addition to a variety of non-fish species such as European Green Crab and dressenid mussels (i.e. Zebra/Quagga mussels). European Crab in particular was identified as a major potential threat due to their capacity to alter habitats with abundant aquatic vegetation such as eelgrass meadows, which are critical components of juvenile Chinook rearing habitat. While some research is ongoing through provincial and academic organizations in BC, there is a need to clarify a process and platform to better quantify the current distributions and population status of these invasive species, and to determine the levels of risk they pose to Fraser Chinook through predation and competition.

## **H.9. DISEASE**

- Disease prevalence and intensity is difficult to study in wild salmon populations due to the extensive geographic range they inhabit, and because fish mortality is generally not observed and carcass recovery can be difficult. However, there are opportunities to investigate disease in migrating adult salmon returning to spawning grounds, and to improve upon monitoring and detection protocols for disease. Future research should aim to better characterize the linkages between disease transmission and frequency in Fraser Chinook populations with the many stressors these stocks are facing, such as climate change and increasing frequency of drought, high temperatures, and periods of low flows.

## **H.10. FISHING**

- Chinook Salmon from some Fraser DUs are caught in fisheries outside the Pacific Salmon Treaty waters; however there is little in the way of specific accounting for these impacts as some fisheries are without formal CWT monitoring programs or effective alternatives.
- There is a need to collect more and better encounter data from non salmon-targeted fisheries and distant fisheries such as the Gulf of Alaska Pollock fishery and mid-water trawl fisheries for Hake off US/Southern BC coast.
- Recently, concerns surrounding the potential impacts of mass-marking programs and the implementation of mark-selective fisheries have been raised, as injured and/or stressed wild salmon can be subject to substantial mortality following release. The impacts from mark-selective fisheries should be investigated for FRC DUs, and compared to the benefits of the information provided and possible alternatives.

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- Considerable research is needed to better characterize harvest rates for Fraser Chinook both at the MU and DU level. The current paucity of CWT-indicator programs for the Spring and Summer 5.2s has resulted in a lack of information on age- and fishery- specific harvest rates. Developing DU-specific encounter rate information, for both retention and non-retention fisheries would be very valuable.
  - There is considerable uncertainty surrounding illegal fishing activity in both the freshwater and marine environments, in addition to fisheries that intercept FRC as bycatch. Research is needed to investigate the impact these activities have on FRC, particularly at the DU-level, and to provide information for potential mitigations.

### **H.11. BIG BAR LANDSLIDE**

- Ongoing research needs for the 2018 landslide in the mainstem Fraser River near Big Bar will be determined by the Big Bar working group. We do, however, put forward the following research recommendations for consideration:
- An assessment of straying of fish from DUs that would ordinarily spawn upstream of Big Bar and have spawned in other river systems as a result of the slide. The assessment should include how many fish spawned in other locations and whether the spawning events were successful. Identifying successful spawning is important, as it may result in some degree of genetic introgression into the recipient populations.
- Continued monitoring of hydrological conditions at the Big Bar slide as attempts to mitigate the channel constriction continue, and determining how certain flows impact or halt migration.

### **H.12. MITIGATION MEASURES**

- Considerable research is needed to investigate the feasibility and potential effectiveness of mitigation measures that may benefit FRC. In Element 16 of the RPA a broad inventory of mitigation measures that may benefit FRC was discussed, using examples from both within the Fraser River watershed and distant regions, yet there is a great amount of uncertainty with regards their applicability or practicality. Due to our limited knowledge of FRC habitat use and supply (particularly for stream-type FRC with no stock-recruit data, or 10 of 11 DUs assessed), variable inter-annual environmental conditions, and a large and often inter-related suite of threats and limiting factors that lead to FRC mortality, there is insufficient information to accurately quantify the benefits of individual mitigation measures at the DU or even MU level. As more research is conducted on the effectiveness of mitigation measures it may be possible in the future to estimate ranges of productivity changes for certain projects.
- There is an enormous amount of variation in habitat type, hydrology, and environmental conditions between streams within the FRC DUs considered in this RPA, and often major differences exist between watersheds within a single DU. This is particularly challenging for mitigation planning for multiple DUs in which there are a large number of watersheds (i.e. DU9 MFR-Spring, DU10 MFR-Summer, DU11 UFR-Spring). Future research on FRC mitigation should explore DUs on an individual basis to better represent these aggregate populations.