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Stock status and biological and fishery consequences of alternative harvest and rebuilding actions for Yukon River Chinook salmon (*Oncorhynchus tshawytscha*)

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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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ABSTRACT

The Yukon River is one of the largest and most remote salmon producing rivers in the world. Chinook salmon (*Oncorhynchus tshawytscha*) from the river have supported subsistence and First Nations fisheries for millennia, and commercial and recreational fisheries for over a century. Within the Canadian portion of the Yukon River basin Chinook salmon make up two Stock Management Units (SMUs; Yukon Chinook and Porcupine), comprising 12 Conservation Units (CUs) and over a hundred spawning locations. We used genetic information and run reconstruction models to estimate spawner abundances for the nine Yukon River CUs in the Yukon Chinook SMU from 1985 to 2024. These estimates were combined with age composition and harvest data to characterize population dynamics, estimate biological benchmarks, assess CU status, and quantify expected biological and fishery consequences of current and alternative harvest management measures via closed loop simulations. Returns to the Yukon Chinook SMU declined ~87% over the past four decades culminating in five of the lowest returns on record in 2020–2024. Five of the nine CUs in the Yukon Chinook SMU, and two of three CUs in the Porcupine SMU, were assessed as in the Red status zone under the Wild Salmon Policy and so both SMUs are considered to be below their CU-status based Limit Reference Points. Aggregate Yukon Chinook SMU spawner abundances over the most recent generation (2019–2024) averaged ~25,000 which is well below a proposed Upper Stock Reference point and rebuilding target of 86,000 spawners. Declines in female Chinook salmon age at maturity, and to a lesser extent the proportion of females in the spawning population, have led to declines in reproductive output per spawner (~25% from the late 1980s to the early 2020s). Intrinsic productivity has also declined by ~ 60% over this same time period in the Yukon Chinook SMU. Leading hypotheses for factors driving the declines in both SMUs include heat stress and *Ichthyophonus* infection during spawning migrations, and climate induced changes to predator, prey, and competitor abundances in the Bering Sea ecosystem in which Yukon Chinook salmon spend much of their marine life. Simulations for the Yukon Chinook SMU suggest that under the current agreement between ADF&G and DFO that limits directed harvest unless border passage is expected to exceed 71,000 fish, spawner abundances are projected to exceed CU-specific lower biological benchmarks for six CUs, exceed upper biological benchmarks for four of those six, and that there could be modest, but infrequent, harvest opportunities. Alternative harvest management measures (e.g., lower escapement goals, caps on maximum harvest rates) were projected to influence rebuilding potential and harvest opportunities to varying degrees, but alternative assumptions about future productivity were far more consequential. Data limitations precluded reconstructing the dynamics of Chinook salmon from CUs in the Porcupine SMU. We conclude by describing key knowledge gaps and areas of potential future work for both SMUs.

1. INTRODUCTION

The Yukon River is one of the largest and most isolated river systems in North America. It drains over 850,000 square kilometers from its headwaters in northern British Columbia through the Yukon Territory in Canada and Alaska in United States, before emptying into the Bering Sea (Brabets et al. 2000). It is also one of the world's largest salmon producing river basins with millions of salmon entering the river in most years (JTC 2025). Among the Pacific salmon species that return to the Yukon River are Chinook salmon (*Oncorhynchus tshawytscha*) which spawn throughout the basin with nearly half returning to one of hundreds of Canadian streams to spawn each year (Brown et al. 2017; JTC 2025).

Yukon River Chinook salmon have supported subsistence and First Nations fisheries and cultural practices for millennia. In more recent years they have also supported commercial and recreational fisheries. In 2001, Canada and the United States finalized the Yukon River Salmon Agreement, a chapter of the Pacific Salmon Treaty (1985), where a spawning escapement goal for mainstem Canadian-origin Yukon River Chinook salmon was set at 33,000 to 43,000 and joint assessment projects were initiated. Prior to 2002, annual harvests of Chinook salmon in the Yukon River averaged 150,000 fish, but dropped to near 50,000 in the early 2000s as a result of low return sizes and fishery closures. An updated "interim management" escapement goal (IMEG) of 42,500 to 55,000 Chinook salmon was established in 2010. Aggregate returns of Canadian-origin Yukon River Chinook salmon have declined in the decades since culminating in five of the lowest returns on record in 2020–2024 (JTC 2025). These declines have placed immense hardship on subsistence users and First Nations who depend on Yukon River Chinook salmon for food, social and cultural uses, and there have been simultaneous declines in average female reproductive potential (due to declines in size and age, Ohlberger et al. 2020) and changing freshwater and marine conditions experienced throughout the Chinook salmon life cycle (Howard and von Biela 2023; Murdoch et al. 2024; Feddern et al. 2024). In 2024, an agreement was reached between the Alaska Department of Fish and Game (ADF&G) and Fisheries and Oceans Canada (DFO) which included fishery closures throughout the drainage and established a rebuilding target of 71,000 Chinook salmon to pass the U.S.–Canada international border.

In addition to the provisions of the Yukon River Salmon Agreement, Yukon River Chinook salmon and Porcupine Chinook salmon have been proposed as a major fish stock under the Fish Stocks provisions of Canada's recently amended *Fisheries Act*. The major fish stocks identification comes with a requirement to develop three reference points: a Limit Reference Point (LRP), a Upper Stock Reference (USR), and a Removal Reference (RR; Section 3). These candidate reference points should be relevant to the scale at which assessment and management occurs, the types of information that are collected, how assessments are used to support decision making, and should be consistent with existing policies (e.g., Canada's Wild Salmon Policy, Holt et al. 2023). The Fish Stocks provisions require that the Government of Canada use these reference points to "maintain major fish stocks at or above the level necessary to promote the sustainability of the stock, taking into account the biology of the fish and the environmental conditions affecting the stock" (DFO 1985). A rebuilding plan is required per section 6.2 of the Fish Stocks provisions if the stock is assessed to be below its LRP. In support of these legislative requirements, the Fisheries Management sector of DFO requested science advice on stock status and the expected impacts of harvest and rebuilding actions on Canadian-origin Yukon River Chinook salmon. Throughout this document, salmon originating from the Canadian portion of the Yukon River basin will be referred to as Yukon River Chinook salmon for brevity.

1.1. OBJECTIVES

The specific objectives of this research document are to:

1. Describe current understanding of (a) major stock structure and distribution, (b) major stock, and component Conservation Units, status and trends, and (c) anthropogenic, ecosystem, and climate factors affecting Canadian-origin Yukon River Chinook salmon;
2. Describe potential rebuilding and enhancement activities that could contribute to Canadian-origin Yukon River Chinook salmon rebuilding;
3. Provide estimates of a Limit Reference Point and other candidate reference points (e.g., Upper Stock Reference and Removal Reference) for the Yukon Chinook Stock Management Unit (SMU);
4. Provide quantitative estimates of the expected biological and fishery consequences of current and alternative harvest management measures (e.g., no fishing, interim aggregate escapement goal, rebuilding goal, exploitation rate caps based on Conservation Unit overfishing risk, etc.) that could contribute to Yukon Chinook SMU rebuilding;
5. Propose exceptional circumstances or assessment triggers for Yukon Chinook SMU (e.g., circumstances or conditions that represent a substantial departure from those under which the advice in this assessment was developed); and
6. Identify key knowledge gaps and areas requiring future work.

2. STOCK STRUCTURE, BIOLOGY, AND DISTRIBUTION

Under Canada's Wild Salmon Policy (DFO 2005), Yukon River Chinook salmon belong to 12 conservation Units (CUs) which are genetically and ecologically unique population units unlikely to naturally recolonize if they were to be extirpated (Table 1, Holtby and Ciruna 2007; Holtby 2011). Conservation Units were established in 2007 and 2009 based on genetics, life history characteristics (such as stream-type or ocean-type), and biogeographic zones of the near-shore and freshwater environments (Holtby and Ciruna 2007; Holtby 2011; Wade et al. 2019). Conservation Unit delineations can be updated and revised when new information warrants it following a standardized framework (Wade et al. 2019). The 12 Canadian-origin Yukon River Chinook salmon CUs represent two stock management units (SMU): Yukon River Chinook and Porcupine River Chinook (DFO 2024a). Nine CUs are from watersheds draining into the mainstem of the Yukon River and three CUs from the Porcupine River. The Porcupine River SMU joins the Yukon River at Fort Yukon, Alaska, downstream of the U.S.–Canada border, and downstream of the assessment site near Eagle, Alaska which estimates Yukon Chinook SMU border passage. DFO manages these two SMUs separately due to differences in assessment projects, data availability, harvest decision rules, and fishery management objectives (DFO 2024a). However, under the Fish Stocks provisions, both SMUs are combined into one "major stock" under legislation. Much of the analysis herein applies to the Yukon Chinook SMU, and not the Porcupine SMU due to limited data availability.

Chinook salmon that spawn in the Canadian portion of the river basin are genetically unique, and genetic sampling in the lower river near the river mouth provides in-season estimates of the Canadian component of the return migration that enter the system. On average, the annual proportion of the total return of Chinook salmon to the Yukon River that are Canadian origin is 42% and has historically varied from 31%–54% (JTC 2025). These Chinook salmon spawn in over 100 spawning locations throughout the Canadian portion of the watershed (Figure 1, Brown et al. 2017).

The Porcupine River basin, located in northern Yukon is comprised of three Chinook salmon CUs: Old Crow, Salmon Fork, and Porcupine (Figure 1). Little data exists from Chinook salmon from the Salmon Fork CU, and spawning abundance is believed to be very low. The Porcupine CU is the most abundant of the SMU (DFO 2024a). Chinook salmon passage in the Porcupine River has been estimated using sonar since 2014 near the community of Old Crow, and estimates the combined abundance of adult Chinook salmon returning to the Porcupine and Old Crow CUs (Figure 3).

Further south, along the mainstem Yukon River within Canada, nine CUs form a single SMU which contains the majority of Canadian-origin Yukon River Chinook salmon (Figure 1).These nine CUs include: Northern Yukon and tributaries (hereafter “Northern Yukon”), White and tributaries (“White”), Stewart, Middle Yukon River and tributaries (“Middle Yukon”), Pelly, Nordenskiold, Big Salmon, Upper Yukon River (“Upper Yukon”), and Yukon River-Teslin headwaters (“Teslin”). Spawner and return abundances among the CUs varies considerably and is primarily assessed via a monitoring and sampling program located near the U.S.–Canada border. In addition, there are assessment sites throughout the Canadian portion of the watershed, which in recent years have been operated by Yukon First Nations, Yukon Energy Corporation, or consultants (Figure 3, JTC 2025). Sampling for genetics on the spawning grounds has helped support the development of a genetics baseline which has been used to determine population and CU delineations (Figure 2). While, historical and current tributary assessments provide insights into the general distribution of spawners within and among CUs they are unable to shed much light on on changes in distributions over time.

Yukon River Chinook salmon have a stream-type life history where fry remain in streams, rivers or lakes prior to out-migrating to the Bering Sea as 1- or 2-year-old smolts (Bradford et al. 2009), though occasionally age-0 smolts have been observed (Bradford et al. 2008). During their first summer, fry will migrate into non-natal streams, sometimes up to 75 kilometers upstream or hundreds of kilometers downstream in search of optimal foraging and rearing habitats (Bradford et al. 2008; Daum and Flannery 2011). Smolt out-migration is believed to be directional and occurs between April and June but understanding of smolt behaviour and out-migration timing is limited due to logistical constraints during the sampling period as smolts can migrate under ice (Bradford et al. 2008; Wilson and Peacock 2025).

Once smolts reach the Yukon River estuary, they exit the area by July or August (Myers et al. 2010; Howard et al. 2017). Juveniles move offshore into the northeastern Bering Sea and are found in depths greater than 30 meters by September – October. They overwinter in the southeastern Bering Sea at the edge of the shelf break (Myers et al. 2010; Murphy et al. 2023). Juvenile and immature Chinook salmon spend summers in the central Bering Sea and winters in the southeastern Bering Sea (Myers et al. 2010). After 2–6 years at sea, mature adults return to the shallow northeastern Bering Sea in the early summer to begin the return spawning migration (Myers et al. 2010; Wilson and Peacock 2025).

Adult Yukon River Chinook salmon begin their upstream freshwater migration in May to June each year (Brown et al. 2017; Wilson and Peacock 2025). Spawning populations located further upstream tend to return later on average, when compared with populations lower in the Canadian portion of the drainage (Connors et al. 2022b). Performing one of the longest freshwater migrations known for Pacific salmon, they traverse approximately 2,000 km in 30 days to reach the U.S.–Canada border in late June – August (Eiler et al. 2015; Wilson and Peacock 2025). From here, the distance to spawning habitat varies by population, ranging from less than 100 km to over 1,000 km (Brown et al. 2017), and spawning is completed by approximately mid-September (JTC 2025; Wilson and Peacock 2025). Adaptation to this long migration has resulted in Canadian-origin Yukon River Chinook salmon being, on average, older and larger, with a higher fat content when compared to other populations in the Yukon River basin (Eiler et al. 2015).

3. STOCK STATUS AND TRENDS

3.1. OVERVIEW OF ASSESSMENTS AND SOURCES OF DATA

A detailed review of assessment projects for Yukon River Chinook salmon is provided in Pestal et al. (2022). This data review spanned assessment projects for Chinook salmon across the Yukon River basin including mainstem passage (e.g., sonar, fishwheels, and mark-recapture projects), tributary spawning escapement (e.g., weir, tower, aerial, and sonar projects), harvest estimation, and associated biological sampling (e.g., genetics and age, sex, and length information for stock apportionment).

Briefly, the number of adult Chinook salmon returning to the Yukon River has been estimated near the mouth of the river at Pilot Station since the mid 1980s and scale pattern analysis and more recently genetics have been used to apportion returns to populations spawning in the lower, middle, and upper (Canadian) portion of the river basin. Chinook salmon migrating into the Canadian portion of the mainstem Yukon River has been estimated using a variety of methods over the past four decades, largely focused on the Yukon Chinook SMU. Fishwheels and a mark-recapture program operated in several locations near the border (White and Sheep rocks, Figure 3) from the early 1980s to the mid 2000s to estimate abundance and collect biological samples. In 2005, a sonar site and drift gillnet test fishery were established near Eagle, Alaska, to also estimate border passage and collect biological samples, and this assessment project has continued to present. Within the Canadian portion of the Yukon River, spawning escapement has been enumerated at various tributaries using foot, aerial, and sonar based assessment methods (Figure 3; Table 2). Many of these assessment projects have changed methods over time and/or have been relatively short in duration (Figure 4). The number of adult Chinook salmon migrating into the Canadian portion of the Porcupine River has been estimated using sonar near the community of Old Crow since 2014.

Surface trawl surveys targeting juvenile Chinook salmon in the northern Bering Sea have been used most years since 2003 to derive an estimate of the abundance of Yukon River salmon stocks (Canadian, middle Yukon [Alaska], and lower Yukon [Alaska]) and their diet, energy, and health during their first summer at sea (Murphy et al. 2023). These data have been used to forecast adult returns (JTC 2025), understand juvenile Chinook salmon marine ecology, and shed light on life stage specific drivers of survival (Cunningham et al. 2018; Howard and von Biela 2023).

Harvest of Chinook salmon in Yukon has occurred in First Nations fisheries, commercial, domestic, and public angling fisheries, while in Alaska fisheries have included subsistence, personal use, commercial, and sport. In Alaska, harvest of Chinook salmon was historically apportioned to lower, middle, and upper/Canadian stocks, by age, using scale pattern analysis. Since 2003, harvest of Yukon River Chinook salmon in Alaska by age and stock of origin has been estimated using genetic methods. In Canada, harvests have been estimated and/or are inferred from commercial landings, recreational catch cards, and from communal harvest information provided by individual First Nation Lands and Resources staff to DFO.

Yukon River Chinook salmon are caught as bycatch in Bering Sea Aleutian Islands (BSAI) groundfish fisheries along with other salmon stocks from Alaska, the west coasts of Canada and the United States, eastern Asia, and Russia. Canadian-origin Yukon River Chinook salmon have typically made up a relatively small percentage (2–5%) of the total catch in recent years (2011–2016), and it is estimated that on average ~1% of the total annual return is caught in these fisheries (Ianelli and Stram 2018), though at times it has been greater and impacts on individual CUs are not known. Because harvest in BSAI fisheries is a relatively minor portion of total returns, and because estimates of harvest are not available for all years, we did not explicitly account for BSAI fisheries interceptions of Canadian-origin Yukon River Chinook salmon in our analyses.

Biological information on the Yukon Chinook SMU has been collected annually from the fishwheels at White and Sheep rocks (most years from 1985–2008) and the gillnet test fishery at Eagle (2005–present) to determine fish age, sex, and length. Samples have typically been taken over the annual upstream adult migration period, with the number of samples taken each day roughly proportional to daily passage (Connors et al. 2022b). Genetic material has been recovered from 150–300 archived scale samples for most years between 1985–2005, and since then 200–1000 tissue samples have been taken per year for genetic stock assignment. These samples have been used to assign individual fish to one of nine CUs via microsatellite markers (1985–2016) or Single Nucleotide Polymorphisms (SNPs, 2017–present, Appendix A).

A state-space run reconstruction and spawner-recruitment model for the Yukon Chinook SMU was recently developed (Connors et al. 2022a). The model can be fit to data from various assessment projects that estimate mainstem passage, harvests, tributary escapements, stock proportions, and age composition and has been used to derive annual estimates of total harvest, escapement and returns, by age, for mainstem Yukon River Chinook salmon (i.e., not including Porcupine Chinook salmon) (JTC 2025).

3.2. RUN RECONSTRUCTION AND SPAWNER-RECRUITMENT MODELS

We reconstructed spawner abundance and population dynamics for the nine CUs in the Yukon Chinook SMU following the approach described in Connors et al. (2022b). This approach consisted of three overarching steps: (1) estimate daily CU composition (i.e., proportion of daily total abundance assigned to each of nine CUs) in test fisheries at the U.S.–Canada border via molecular analyses of archived scale and tissue samples collected from returning spawners; (2) estimate annual CU-specific border passage and abundance via a state-space run-reconstruction model fitted to daily estimates of total border passage and CU composition; and (3) characterize CU scale population dynamics and estimate biological benchmarks by fitting age-structured, state-space spawner (or egg mass) recruitment models to estimated spawner abundances, harvest, and age composition. Details on the molecular analyses, run-reconstruction model, and spawner-recruitment analyses and estimation of benchmarks are detailed in Appendix A, B, and C, respectively. Due to data limitations, CU-level scale analysis for the Porcupine Chinook SMU was not able to be undertaken.

3.3. BIOLOGICAL BENCHMARKS AND REFERENCE POINTS

3.3.1. CONSERVATION UNIT BIOLOGICAL BENCHMARKS

Biological benchmarks are used to assess the biological status of CUs under Canada's Wild Salmon Policy (WSP, Holt 2009, Table 3). One commonly used lower biological benchmark is S_{GEN}, the spawning abundance that is expected to lead to recovery to S_{MSY} (the spawner abundance at maximum sustained yield) in one salmon generation in the absence of fishing under equilibrium conditions (Holt 2009). This benchmark is meant to identify when there is high risk of irreversible harm to the CU and is aligned with a Committee on the Status of Endangered Wildlife in Canada (COSEWIC) assessment of "endangered". A commonly used upper biological benchmark is 80%S_{MSY} which is 80% of the spawner abundance associated with maximum sustainable yield under equilibrium conditions. These two benchmarks (S_{GEN} and 80%S_{MSY}) are typically used to assess WSP status when there is sufficient data to estimate spawner-recruitment relationships at the CU scale, or information on freshwater spawning and rearing habitat capacity are available. Other alternatives have been applied in some contexts (e.g., 20% and 40% of S_{MSR} [spawner abundance associated with maximum recruitment]) as a proxy for the lower and upper WSP benchmarks, respectively.

While the use of MSY-based benchmarks under the WSP and DFO's Precautionary Approach framework (DFO 2009) is consistent with international standards, they have been criticized for being explicitly tied to yield based objectives (Frid et al. 2023), difficult to estimate, and sensitive to temporal variation and trends in intrinsic productivity (Holt and Michielsens 2020). For these reasons we used benchmarks based on estimates of the spawner abundance associated with maximum returns (S_{MSR}) to assess the biological status of CUs in the Yukon SMU instead (Table 3). Specifically we used 20% and 40% of S_{MSR} as lower and upper biological benchmarks, respectively (Holt 2009). We argue these alternatives are more biologically based (because they are not tied to yield based objectives), easier to estimate, and less sensitive to temporal variation in intrinsic productivity than MSY based benchmarks. While S_{MSR} based benchmarks can be slightly more biologically conservative than S_{MSY} based ones in some systems they are still generally consistent with the intent of upper and lower benchmarks under the WSP and in their application to status assessments across salmon CUs in the Pacific Region. In addition, the use of S_{MSR} based benchmarks is aligned with the proposed use of S_{MSR} based reference points at the SMU scale (see next section).

We also used 100% of S_{MSR} as an illustrative CU rebuilding target to quantify rebuilding potential and the biological consequences of alternative fisheries management measures (Section 7). We argue this target is better aligned with Indigenous and ecosystem based management principles and the concept of "take only what you need and leave lots for the ecosystem" (Reid et al. 2022b) than S_{MSY} based targets. Because they are not tied to concept of maximum sustainable yield and are associated with lower fishery removals, they allow for greater resilience to climate change and other uncertainties, and better preserve ecosystem functions, both principles often found in Indigenous Knowledge (Frid et al. 2023). These CU rebuilding targets were based on analyses that explicitly attempted to account for changing reproductive potential due to shifts in age structure and size at age in female Chinook salmon (described in detail in section on "Demographic Change" in Appendix C).

Consistent with emerging recommendations for Pacific salmon (Holt et al. 2025), our estimates of biological benchmarks (and reference points; see next section) were derived from spawner-recruitment, or egg mass-recruitment, models that assumed long-term stationary dynamics as opposed to models that allowed intrinsic productivity to vary through time (Appendix C).

We recognize that there is an inherently subjective dimension to the selection of biological benchmarks and so to be transparent about our use of S_{MSR} based benchmarks and rebuilding targets at the CU scale we compared their values to other commonly used ones with, and without, also attempting to explicitly account for recent demographic characteristics of spawners (Figure C.1).

3.3.2. STOCK MANAGEMENT UNIT REFERENCE POINTS

A Limit Reference Point, Upper Stock Reference Point, and maximum Removal Reference need to be defined for the Yukon Chinook salmon Stock Management Unit (SMU) to meet the Fish Stocks provisions of the *Fisheries Act* (House of Commons 2019) (Table 4). Data limitation did not allow for development of these same reference points for the Porcupine Chinook SMU (only ~10 observations of spawner abundance, Figure 4).

The Limit Reference Point (LRP) is intended to be set at a point above which serious harm to the stock and resultant impacts to the ecosystem, associated species, and/or a long-term loss of fishing opportunities may occur (Barrett et al. 2024). Regional and national DFO guidance is that LRPs for salmon SMUs should be 'CU status-based' to meet the Fish Stocks provisions (DFO 2023). This means that a SMU is below its LRP if any component CUs are in the WSP 'Red' status zone. The WSP rapid status framework (Pestal et al. 2023) is the default approach to formally assess the WSP status of each CU in an SMU in order to meet the Fish Stocks provisions. Wild Salmon Policy statuses under this framework use multiple abundance and trends in abundance metrics as well as additional information when available (e.g., distribution, genetics, demographics) to assess status, and the framework emphasizes iterative local expert review.

The Upper Stock Reference (USR) point is the aggregate SMU spawner abundance at which socio-ecological objectives for the system are expected to be met, and above which maximum allowable harvest rates (up to the Removal Reference point) can be sustained. The USR should also be set at an appropriate distance above the LRP to provide (a) sufficient opportunity for the management system to recognize a declining stock status and (b) time for management actions (e.g., reduced harvest) to have effect. The Removal Reference is the maximum acceptable harvest rate the SMU can be subject to and is typically set to be less than or equal to the removal rate associated with maximum sustainable yield (U_{MSY}).

Other aggregate abundance based reference points can be defined for an SMU. A 'Fishery Reference Point - Lower' is the aggregate SMU spawner abundance below which all fishery removals should be limited to the maximum extent possible and below which there is a high risk of serious and irreversible harm to the stock. This is referred to as a lower fishery reference point to avoid confusion with the CU status-based LRP noted above. This type of reference point is helpful for informing management since assessment and fishery decisions are often abundance based and so can be operationalized.

We used a CU-status based LRP to formally assess SMU status relative to the LRP per national guidance where CU statuses are based on applying the rapid status assessment framework for Pacific salmon under Canada's WSP (Pestal et al. 2023, Appendix **??**).

We propose that an USR equal to the median estimate of S_{MSR} for the Yukon Chinook SMU, when changes in demographics are taken into account (86,000 spawners based on Connors et al. 2022a), be used for the SMU (Table 4). This could be considered the aggregate spawner abundance at which socio-ecological objectives for the system are potentially maximized, and above which maximum allowable fishery removals could occur. We also propose that this be considered as the rebuilding target for the Yukon Chinook SMU.

We propose that the USR be based on analyses of the Yukon Chinook Salmon at the SMU scale and not analyses of component CUs because we consider these to be the most scientifically defensible estimates of stock aggregate dynamics, and hence reference points. This is because they are based on an integrated run-reconstruction and spawner-recruitment models fit to data from various assessment projects that estimate mainstem passage, harvest, tributary escapements, stock proportions, and age composition. In addition, this model is used bilaterally under the Yukon River Salmon Agreement to derive annual estimates of total harvest, escapement and returns, by age, for Canadian-origin Yukon River Chinook salmon. While this model does not explicitly take CU scale dynamics into account, the closed loop simulations described in Section 7 allowed us to evaluate CU scale consequences of operationalizing the SMU reference points we propose and other harvest control rules that have been used or considered for this system. Based on the simulations described in Section 7, and under assumption of long-term average productivity, the proposed USR has a very low probability of any CUs falling below their lower biological benchmarks.

We propose a maximum Removal Reference point for the Yukon Chinook SMU that is equal to the long-term average U_{MSY} for the least productive CUs (36%; Table 4). Adopting this reference point would help to ensure that maximum allowable fishery removals, in mixed-CU mainstem fisheries in Alaska and Yukon, are unlikely to overfish the least productive CUs in the system. We argue this is consistent with Federal and international fishery policy which typically set maximum fishery removals to be less than or equal to those associated with U_{MSY}, and propose this value be from the least productive CU to align with Wild Salmon Policy objective to take biodiversity at the scale of CUs into account. While the use of U_{MSY} is somewhat inconsistent with our proposal to adopt a USR based on maximizing returns instead of harvest, we note that the proposed removal reference is somewhat similar to estimates of U_{MSR} that have been previously derived for the stock aggregate (~45%, Connors et al. 2022a).

Lastly, we propose a Fishery Reference Point - Lower equal to 31,000 spawners for the Yukon Chinook SMU (Table 4). This value is the average SMU spawner abundance associated with no greater than 50% chance of at least one CU falling in the Wild Salmon Policy Red status zone based on the forward simulations described in Section 7, under an assumption of long-term average productivities. To be consistent with the intent of the *Fisheries Act*, at aggregate returns below this reference point we propose that fishery removals be limited to the extent possible, while above the reference point removals can progressively be increased to the Removal Reference point (the maximum removal rate when the USR point is met).

All of the above reference points are proposed for the Yukon Chinook SMU only. There is very limited data for the Porcupine SMU which precluded us from characterizing its dynamics and/or estimating reference points for it.

3.4. CONSERVATION UNIT STATUS AND TRENDS

3.4.1. RUN-TIMING AND RECONSTRUCTED SPAWNER ABUNDANCES

Run-timing, as inferred by the timing of passage from Alaska into Yukon, varied among CUs (Figure 5). Conservation Units with Chinook salmon that spawn in the lower (more northern) portion of the Canadian Yukon River basin tended to return, on average, earlier than those in the upper (more southern) portions of the basin. Chinook salmon from the Middle Yukon and Upper Yukon CUs tended to return later than Teslin fish, the longest migrating Chinook salmon in the Yukon River. Chinook salmon from the Nordenskiold CU tended to have the latest, and most variable, run-timing. This run-timing variation results in an average return migration duration that is approximately twice as long (~70 days) as the average individual CU migration (~35 days).

All CUs exhibited considerable inter-annual variation in reconstructed spawner abundance over time (Figure 6). The Middle Yukon CU had on average the largest spawning abundances

(~13,000) followed by the Pelly (~10,000) and Northern Yukon (~7,500) CUs. The White (~3,800) and Nordenskiold (~1,900) CUs had the smallest average spawner abundances. Across CUs spawner abundances in the most recent generation (2018–2024) were on average 46% lower than preceding ones (1985–2017). The Northern (68% lower), Pelly (62% lower), and White (60% lower) CUs have experienced the largest magnitude declines.

There was not strong evidence for changes in the proportional contribution of different CUs to total spawning abundance over time (Figure 7). This is in contrast to the apparent increase in the contribution of the Middle Yukon CU over time based on the raw GSI based estimates of CU contributions to border passage over time (e.g., Appendix B24 in JTC 2025). The likely reason for these differences is that the raw estimates of contributions are confounded by the sometimes unrepresentative genetic sampling of total returns. For example, in recent years (e.g., 2020–2024) genetic samples tended to be biased toward the second half of the run to varying degrees (Figure A.1) and because the Middle Yukon CU tends to have a run-timing that is slightly later than peak of aggregate return (Figure 5) it may be over represented in the raw genetic proportions.

Reconstructed CU spawner abundances were not fitted to data from tributary scale assessments, but tended to correlate positively with them (Figure 8). Tributary assessments that indexed a larger fraction of the total CU and/or were higher precision methods (e.g., Takhini and Klondike sonars, Tatchun weir) tended to have the strongest positive correlations with reconstructed CU scale spawner abundances. One exception was the Whitehorse Rapids Fishladder that had little relationship with reconstructed spawner abundance for the Upper Yukon CU, possibly because fish enumeration at the fishway is confounded by relatively low rates of passage for returning adults.

3.4.2. SPAWNER-RECRUITMENT RELATIONSHIPS

There was clear evidence of heterogeneity in intrinsic productivity (α) and spawner abundance associated with maximum recruitment (S_{MSR}) across CUs (Figure 9; Tables 5 and 6). Mean estimates of intrinsic productivity ranged from 2.25 to 7.62 recruits-per-spawner and S_{MSR} ranged from ~ 1,600 to 20,000 spawners. These intrinsic productivities translate into harvest rates predicted to maximize long-term harvest (i.e., U_{MSY}) that range from ~36% to 73%. Most CUs showed some visual evidence of overcompensation with modestly declining recruitment at relatively high spawner abundances, and for many of the CUs the largest recruitments were observed in brood years with relatively low to moderate spawning abundances in the early part of the time series (1985–1995 brood years, Figure 10). Interannual variation in recruitment was positively correlated among CUs (mean pairwise correlation of ~0.5) and was strongest for CUs that were geographically closest to each other. There was strong evidence of time-trends in recruitment and productivity over time (Figure 11) with productivity estimated to have declined by an average of 60% across CUs between the late 1980s and the early 2020s, though Nordenskiold brood years in the mid 2000s appeared to experience a large spike in productivity.

3.4.3. DEMOGRAPHIC CHANGE

As has been well documented previously (Ohlberger et al. 2018, 2020; Connors et al. 2022a), age, size, and sex ratios of Yukon River Chinook salmon, as measured at the U.S.–Canada border, have changed to varying degrees over time (Figure 12). The proportion of returning fish that were sexed as female declined from an average of approximately 53% in the 1980s to 46% in the 2020s, with considerable interannual variation (as high as 59% in 1989 and low as

31% in 2023). In contrast, there is little evidence of declines in female size at age (Figure 12b). Female Chinook salmon age composition has changed considerably over time. In the 1980s, approximately 20% of females were seven year olds, 70% were six year olds, and fewer than 10% of females returned on average as five year olds. By the 2020s an average of 5% of females returned as seven year olds, 70% as six year olds and 25% as five year olds (Figure 12c).

When the relationship between female size and fecundity (Ohlberger et al. 2020) is taken into account, these observed demographic changes translate into a marked decline in average reproductive output per spawner (Figure 12d). Specifically, the average total eggs per spawner declined by ~25% from the late 1980s to the early 2020s.

These demographic changes result in modest differences in the inferred shape of the relationship with recruitment when spawners are the measure of reproductive output versus when total egg mass is used instead (Figure 13 vs. Figure 10). As a result of the observed declines in reproductive output, the spawner abundance expected to maximize recruitment (S_{MSR}), given recent (2019–2024) demographic characteristics, was estimated to be on average 40% greater across all CUs than spawner abundance estimated given historical (1985–1995) demographic characteristics (Table 7). In contrast, S_{MSR} given recent (2019–2024) demographic characteristics was estimated to be on average 8% greater across all CUs than when demographic change was not taken into account (Table 7) but this was highly variable (53% to -17%) and for several CUs estimates of S_{MSR} corresponded to smaller spawner abundances when demographic change was explicitly accounted for (i.e., Big Salmon, Middle Yukon, Nordenskiold, Pelly, White CUs). The reason why estimates were lower for some CUs may be due to subtly different shapes between the two alternative reproductive output (i.e., spawners vs. total egg mass) and recruitment relationships; CUs like Big Salmon and the Middle Yukon, for example, appeared to have more evidence for overcompensation when considering egg mass and this could in turn result in estimates of the corresponding spawner abundance expected to maximize recruitment being lower than when the reference point is based on a total spawner to recruitment relationship.

3.5. WILD SALMON POLICY RAPID STATUS ASSESSMENTS

An assessment of CU biological statuses was carried out following the methods detailed in the Wild Salmon Policy (WSP) rapid status algorithm (Pestal et al. 2023). The key inputs into these assessments were biological benchmarks (Table 5) and time series of reconstructed spawner abundances (i.e., from the the multi-CU run reconstruction model) in most cases, but tributary assessments were used instead in some cases. The data inputs and resulting statuses were determined through a series of meetings with subject matter experts. The rapid status algorithm is a decision tree where branches in the tree compare metric values (e.g., absolute abundance, trends in abundance, relative abundance) to thresholds to ultimately arrive at end points (or nodes) that define CU status and associated confidence ratings. The outcome of these assessments are briefly summarized here; more details are provided in Supplement A.

Within the Yukon Chinook SMU, four CUs (Nordenskiold, Upper Yukon, Stewart, and White) were assessed as in the Red status zone due to spawner abundances over the most recent generation being below an absolute abundance lower threshold of 1,500 spawners. These statuses were based on run-reconstruction model estimates of spawner abundance with the exception of the Upper Yukon CU which instead was based on tributary assessment projects that are believed to assess the majority of the CU. The Teslin and headwaters CU was assessed as in the Red status zone based on trends in spawner abundance from two long-running aerial surveys. These assessment projects were considered to be more consistent with local observations and expert feedback than run-reconstruction based WSP status. The WSP rapid statuses for other CUs in this SMU were assessed as in the Amber status zone (Figure 14).

Status assessment for the Porcupine Chinook SMU is limited to the Porcupine and Old Crow CUs, as the Salmon Fork CU is data deficient. The Porcupine and Old Crow CUs have been jointly assessed by sonar beginning in 2014. Since then the two CUs averaged ~4,000 during the first six years, and have fallen to ~400 over the most recent four years (2021–2024). The most recent generation (2019–2024) geometric mean for the CUs combined is 695 which is well below the absolute abundance lower threshold of 1,500. Both CUs were therefore assessed as in the Red WSP status zone.

3.6. SMU STATUS AND TRENDS

Aggregate returns, harvest and spawner abundances for the Yukon Chinook SMU were derived from updated fits (through 2024) of the model described in Connors et al. (2022a).

The Yukon Chinook SMU experienced relatively large run-sizes from 1981–1995, followed by a sharp decline in the late 1990s, a modest recovery in the mid-2000s, and then a general decline since with the lowest observed run-sizes on record in 2021–2024 (Figure 15). Overall the SMU declined by ~87% from 1981–2024. In contrast to run size, spawning escapements have varied from year to year but have been relatively consistent over time (grey bands in Figure 15a) with the exception of the very low spawning abundances in recent years.

Aggregate SMU harvest has also varied considerably over time, and realized harvest rates have ranged from an average of 68% in the late 1980s and early 1990s to 18% over the most recent decade (Figure 15b). In the last four return years harvest was dramatically curtailed and averaged 3%–8%.

The Yukon Chinook SMU is assessed as being below its CU-status based LRP because there are five CUs that were assessed as being in the Red status zone. Aggregate spawner abundances over the most recent generation (6 years) averaged 25,852 which is below the proposed Fishery Reference Point–Lower of 31,000 and well below the proposed Upper Stock Reference Point of 86,000. The harvest rate over this same time period averaged 16%.

The Porcupine SMU is also below its CU-status based LRP because there two CUs are in the Red status zone, but data limitations preclude abundance and harvest trend metrics for this SMU.

4. ANTHROPOGENIC, ECOSYSTEM, AND CLIMATE FACTORS AFFECTING THE STOCK

The past decade has seen a considerable amount of research into anthropogenic, ecosystem, and climate factors that may have contributed to the recent and long-term declines in body size and productivity of Yukon River Chinook salmon. Anthropogenic climate change is causing clear, directional changes in the ecosystems used by Yukon River Chinook salmon, but is also likely to increase the variability of conditions experienced year-to-year. The arctic is warming up to four times faster than the globe (Rantanen et al. 2022), meaning that Northern populations of salmon are likely to experience more rapid change than more Southern ones. The northernmost populations in the Yukon River Chinook stock (i.e., Porcupine SMU) should be considered more vulnerable overall to future climate-caused changes in freshwater environments. While the Yukon River watershed is relatively minimally modified by human activity, some populations are subject to large localized anthropogenic impacts (e.g., from mining, hydroelectric facilities, etc.) in addition to overarching environmental change.

Taken together, these factors impact every life stage and ecosystem used by Yukon River Chinook salmon. However, some are more likely to be drivers of declining abundance and size. Strong correlations between juvenile salmon abundances and total returns to the Yukon River suggest that the driving factors of recent declines occur between the migrating adult life stage and their offspring's first summer at sea (Finster et al. 2025). Recent returns to the U.S. –Canada border have been lower than predicted from run estimation at the mouth of the Yukon River, suggesting that migrating adults bound for the Canadian portion of the Yukon basin face high en route mortality rates.

The sections below, and Table 8, summarize what is currently known about factors affecting the stock by life stage, but is not an exhaustive review. Additional factors affecting the stock pertaining to habitat are found Section 6, and impacts from fisheries were described in Section 3.

4.1. MIGRATING AND SPAWNING ADULTS

The adult freshwater return migration of Yukon River Chinook salmon is one of the longest of any Pacific salmon populations, up to 3,200km (Brown et al. 2017), and this is believed to be the life stage where both environmental and intrinsic factors (e.g., age and/or size) are most limiting the productivity of Yukon River Chinook salmon and present the highest risk to the future of the stock (Howard and von Biela 2023; Finster et al. 2025). Over their long freshwater migrations, Yukon River Chinook salmon can be exposed to a wide range of temperature and flow conditions, especially those that spawn in smaller tributaries. High maximum daily temperatures and maximum weekly temperatures are negatively associated with the productivity of some CUs (White, Stewart, and Pelly) (Murdoch et al. 2024; Feddern et al. 2024). This is hypothesized to occur via reduced survival and spawning success with effects that carry over to juvenile production in the following generation (von Biela et al. 2020; Murdoch et al. 2024). Heat stress begins at 18°C and intensifies at 21°C for Yukon River Chinook salmon (U.S. EPA 2003; Bowen et al. 2020). Adult Chinook salmon that experienced warmer water temperatures and lower discharge in the mainstem Yukon River during their return migrations tended to produce fewer juveniles per spawning adult (Howard and von Biela 2023). The timing of upstream migrations differs among CUs (Figure 5), likely in part due to the thermal regimes in which eggs incubate (Connors et al. 2022b), and they may be differently adapted to temperatures during their spawning migration. This adaptation along with cumulative heat exposure for longer-migrating fish may explain variation among CUs in their relationships with migration temperatures, for example, the productivity of mainstem CUs tend to be minimally associated with migration temperature, while the Pelly, Stewart, and White CUs tend to be the most strongly related (Feddern et al. 2024).

Pathogens and disease are also a concern in spawning adults, and their impact is likely mediated by environmental conditions (e.g., temperature during return migrations). *Ichthyophonus hoferi* is a pathogen that can infect immature or maturing Chinook salmon in the marine environment (Murphy et al. 2023; Murphy 2024), but infection is most likely to impact survival and reproductive success during spawning migrations. It is believed to infect female Chinook salmon more than males (Kocan et al. 2004). Infections appear to be most severe near the midpoint of migration for Yukon River Chinook salmon (between river entry and U.S.–Canada border) and it is believed that *Ichthyophonus* results in significant en route mortality of Yukon River Chinook downstream of the U.S.–Canada border, though it has not been quantified (Liller and Ferguson 2023; Finster et al. 2025).

Another issue affecting spawners is thiamine deficiency, a condition that can arise in part from a poor diet in the marine environment (Larson and Howard 2019). Low thiamine levels in spawning adults and subsequently low thiamine stores in eggs may be contributing to lower run sizes (Larson and Howard 2019; Howard and von Biela 2023), but research on its impacts and causes in Yukon River Chinook salmon is limited. Although CU-specific data are not available, thiamine deficiency likely disproportionately affects Yukon River Chinook salmon with the longest migrations (e.g., Upper Yukon, Teslin, and Pelly CUs), as evidenced by low thiamine levels observed on the spawning grounds of far-migrating Chinook salmon (Larson and Howard 2019).

In addition to these biotic and abiotic stressors, hydroelectric facilities at Mayo and Whitehorse Rapids pose localized challenges to adult migration. The Mayo facility, built in the 1950s, obstructs adult fish passage and locally extirpated Chinook Salmon from the upper reaches of the Mayo River and Mayo Lake, in the Stewart CU. Gravel starvation and channel works potentially contribute to reduced quality of spawning habitat below the dam as well (Finster et al. 2025). The Whitehorse Rapids hydroelectric facility (Upper Yukon CU) and storage dam have fish ladders for adult passage, but the efficacy of the ladder is low (31%), particularly for females (13%, Twardek et al. 2023a). Many female salmon in the Yukon River downstream of the Whitehorse Rapids facility were found to have retained (unspawned) eggs, with 16% of total fecundity retained there compared to 2% in the Teslin River (Jessup and Graff 2024). It is hypothesized that females who fail to pass the Whitehorse Rapids fishway either do not find suitable spawning locations below the dam or lack energy to complete spawning (Jessup and Graff 2024).

Harvest rates of Yukon River Chinook salmon were historically very high in some years for which estimates exist (e.g., greater than 70%; Figure 15), and may have been as high or higher in earlier years. This almost certainly led to the extirpation of some of the least productive spawning populations over time (Connors et al. 2022b), though the extent to which this has occurred is unknown.

4.2. EGGS, FRY, AND SMOLTS

The survival of salmon during their incubation (egg) and freshwater rearing life stages is influenced by hydrology and broader climate conditions. There is some evidence that above average streamflows during egg incubation in the fall are negatively associated with Yukon River Chinook salmon productivity (Feddern et al. 2024). Warmer winter temperatures in freshwater during incubation and rearing are associated with above average Yukon River Chinook salmon productivity (Murdoch et al. 2024). However, it is considered unlikely that changes in egg survival are principally responsible for the declining productivity observed in Yukon River Chinook salmon populations (Lamborn et al. 2025).

Climate and hydrological conditions during fry rearing are also associated with Yukon River Chinook salmon productivity. High levels of precipitation during the fry stage are associated with lower productivity (Murdoch et al. 2024). The date of Yukon River ice out (break-up of river ice in spring) is associated with smolt outmigration timing, and earlier ice out years have been linked with increased survival and productivity for Chinook salmon (Cunningham et al. 2018; Murdoch et al. 2024). In smaller tributaries, the formation of stream ice (aufeis) can reduce or even stop stream flow, which can be fatal to rearing salmon (Bradford et al. 2001). Climate change is expected to result in earlier ice out, but also increase winter temperature variability, potentially increasing the prevalence of irregular ice formations like aufeis. Observed impacts of climate change on rearing Yukon River Chinook salmon appear to have complex causes, and future responses are uncertain. The arctic is warming more rapidly than more Southern latitudes (Rantanen et al. 2022), meaning that the freshwater habitats used by more Northern CUs (Porcupine, Old Crow, and Salmon Fork) are likely to experience the most severe cumulative impacts of climate change, resulting permafrost thaw, and glacial retreat, which are discussed in section 6.

The Whitehorse Rapids hydroelectric facility requires smolts from the Upper Yukon CU to pass downstream over the spillway, through the power generation turbines, or through the fish ladder. Recent studies indicate that downstream mortality rates could be as high as 30%, and could stun juveniles as they pass through the facility, increasing their vulnerability to predators downstream (Twardek et al. 2023b). It is possible that flow regimes and gravel starvation from the Mayo hydroelectric facility are impacting incubation and rearing below the dam, but the extent of impact is unclear (Finster et al. 2025).

Placer mining, nearly all (96%) of which occurs in the central Yukon (van Loon 2025), can have a large impact on stream beds, banks, and surrounding terrain. Sluicing, road development, and tailings ponds can increase fine sediment suspension, deposit sediment downstream, and cause bank destabilization. Sedimentation can impact the incubation of salmon eggs by impeding oxygen absorption, thereby reducing the egg-to-fry survival rate (Jensen et al. 2009). Elevated levels of sedimentation and turbidity can also can impact fry survival by reducing primary and secondary productivity of aquatic systems, and may also change the behaviour of fry (DFO 2000). Mining activities can also release heavy metals and change the pH of streams, which may have deleterious effects on eggs and fry. Current regulations for placer mining do not consider cumulative impacts of multiple mining locations throughout a watershed, and mining takes place in multiple Yukon River Chinook salmon CUs. Most of the gold produced by placer mining (47%) takes place near Dawson in the Northern Yukon and White CUs, with smaller operations near the lower Stewart River, Mayo, the South McQuesten River, Kluane, Whitehorse, Livingstone, and Dawson Range (van Loon 2025). Cumulatively, placer mining for gold takes place in six of the twelve Yukon River Chinook salmon CUs (Northern, White, Stewart, Middle Yukon, Upper Yukon, and Teslin). Additional water quality changes can come from permafrost melt through the release of stored heavy metals or toxins (Finster et al. 2025).

4.3. JUVENILES

The climatic and ecosystem conditions that salmon encounter in their first year in the marine environment have a major impact on growth and survival. Similar to river ice, sea ice in the Bering Sea is related to the outmigration of Yukon River Chinook salmon, and productivity is higher when sea ice melts early (Bradford et al. 2008; Howard et al. 2017; Feddern et al. 2024). Less ice cover in the estuary may also be associated with increased food resources available to juvenile salmon (Murphy et al. 2021). Earlier outmigration and abundant food resources allow higher growth and survival in the first months of marine life, which are considered a critical life stage with relatively high mortality (Cunningham et al. 2018). Warmer winter temperatures in the first year at sea are also associated with higher productivity for Yukon River Chinook salmon, but there is less evidence of a relationship with summer temperatures (Murdoch et al. 2024; Feddern et al. 2024). Future climate change will likely continue to reduce sea ice and increase winter temperatures, but it should not be assumed that this will result in positive outcomes for Yukon River Chinook salmon. The rate of ocean warming in the Bering Sea is several times higher than elsewhere in the North Pacific, and historically positive relationships between salmon and warm ocean conditions can erode or even reverse under new conditions (Litzow et al. 2018, 2020). Warm conditions during early marine life may already have detrimental effects, for example, increasing temperatures during this life stage have been correlated with an increase in the number of jacks (salmon that return early), thereby contributing to overall declines in the average size of spawners (Murphy et al. 2021). Furthermore, warm sea surface temperatures favour a community of smaller, less energy-dense copepods, as opposed to the larger copepods that are high-quality prey for salmon and forage fish (Siddon 2023). As a result, juvenile energy density of Yukon River Chinook salmon measured in the Eastern Bering Sea was low in 2019, during a marine heatwave Siddon (2023). Simultaneously, large numbers of rearing pink salmon in the eastern Bering Sea (primarily originating from Russia) can similarly shift the zooplankton community toward smaller copepods (Siddon 2023). The availability and quality of prey is being dually affected by both bottom-up (climate) and top-down (pink salmon) processes during a critical growth stage, potentially impacting the survival, size, and condition of salmon that return or are caught in fisheries.

Subadult and adult Chinook salmon are vulnerable to predation by apex predators including salmon sharks and killer whales, including in the Eastern Bering Sea (Seitz et al. 2019). There is some evidence that killer whale abundance is increasing in the North Pacific generally, possibly contributing to declines in both survival and size via size-selective predation on large Chinook salmon in coastal waters (Ohlberger et al. 2019). However, there is no research indicating the role apex predators may play in the declining size and abundance of Yukon River Chinook salmon specifically.

4.4. ADULTS

Data pertaining to the adult marine life stage are relatively scarce because their habitat during this time includes offshore areas in the central and Western Bering Sea (Myers et al. 2010). Similarly to juvenile Chinook salmon, adult Chinook may be affected by abundant pink salmon via competitive interactions (Ruggerone 2016; Feddern et al. 2024; Lamborn et al. 2025). While pink and Chinook salmon typically occupy a different trophic niche as adults (Johnson and Schindler 2009), pink salmon are capable of causing trophic cascades that alter the availability of prey for other salmon species (Ruggerone et al. 2023). The abundance of pink salmon has been increasing since the mid-1980s due in part to releases from large-scale hatcheries in Russia and Alaska. Prey quality and availability is also a concern for adult salmon, as climate change-induced shifts in the marine food web similar to those documented in the Eastern Bering Sea are likely occurring across the Bering Sea. Like in the juvenile stage, marine diets are the root of some conditions that present during upriver migration to spawning grounds, including thiamine deficiency (Larson and Howard 2019) and *Ichthyophonus* infection (White et al. 2014).

Suboptimal prey and increased competition for Chinook salmon combined with a projected shrinking of suitable ocean habitat (Abdul-Aziz et al. 2011) are likely already contributing to poorer marine habitat conditions for adult Yukon River Chinook salmon in their marine life stage. However, the high-seas habitat that Chinook salmon occupy as adults is governed by large-scale climate processes that are known to exhibit regime shift-like changes with major implications for growth and survival of adult salmon (Litzow et al. 2018). The trends observed may not continue into the future, as the climate of the North Pacific continues to change and ecosystems shift in response.

5. ENHANCEMENT

5.1. HISTORY OF ENHANCEMENT OF YUKON RIVER CHINOOK SALMON

Over the last 40 years several enhancement activities have been directed towards Yukon River Chinook salmon, though they have been fairly modest in nature. These activities have included hatcheries, incubation facilities, and in-stream incubation projects (Figure 16). The largest and longest running enhancement project has been the Whitehorse Rapids Fish Hatchery which began operations in 1984 as mitigation for turbine induced mortalities of migrating Chinook salmon juveniles associated with the Whitehorse Rapids Generating Facility (W.R. Ricks Consulting and DNA Enterprises 1996). The hatchery was designed for a maximum capacity of 400,000 Chinook smolt 0+ reared to a weight of 2 grams. There is currently an annual release target of 150,000 smolt 0+ Chinook, with the majority (80,000) destined for Michie Creek for rebuilding purposes. Releases have occurred in several other unique tributaries (i.e. Wolf Creek, McClintok River, Fox Creek, Byng Creek, and Judas Creek). Average overall releases from 2019–2024 was approximately 101,000 smolt 0+. To varying degrees juveniles have been adipose fin clipped and coded-wire tagged (CWT) since 1984. More recently, parentage-based tagging (PBT) was implemented beginning with the 2019 brood year. Fin clipping and tagging enables in-season hatchery genetic management by allowing hatchery fish to be identified and reducing total hatchery origin broodstock as necessary. In theory it also provides harvest and assessment information though recoveries of clipped and tagged fish but harvest sampling programs are likely not robust enough to reliably estimate encounters in fisheries (Connors et al. 2022a). In addition to its primary mitigation objective, the Whitehorse Rapids Fish Hatchery supports stewardship and education activities by annually providing up to 2,000 eggs for local Stream to Sea classroom incubators.

The Whitehorse Rapids Fish Hatchery is owned and operated by Yukon Energy. Hatchery staff, in collaboration with local First Nations, establish operating plans and production decisions annually. DFO's Salmonid Enhancement Program publishes and engages on the hatchery production targets for the facility more broadly through an Integrated Production Planning process, which considers DFO's priorities and mandate, the Wild Salmon Policy, fish health, and hatchery-wild interactions. These plans include details on target releases by stock, release strategy and location, and fin clipping and tagging[1]. Whitehorse Rapids Fish Hatchery became the only active enhancement facility in the watershed after the McIntyre Creek Fish Incubation Facility was destroyed by a fire in 2018. Enhancement activities at Whitehorse Rapids Fish Hatchery appears to have increased total adult returns to the Upper Yukon River CU, though no dedicated work has been published on the efficacy of the program to mitigate mortalities associated with the hydroelectric dams.

Genetic management is a priority for in-hatchery operational planning and salmon management that has evolved over time. Recent work by Withler et al. (2018) and updated enhanced contribution guidelines (DFO 2024b) provide guidance on the genetic risk of prolonged hatchery enhancement activities on wild stocks, identifying Proportionate Natural Influence (PNI) as a key metric to measure hatchery genetic influence. PNI is a widely applied metric to evaluate, classify, and monitor levels of hatchery influence and genetic risk to the natural adaptive state of 'integrated' populations (Hatchery Scientific Review Group 2009, 2015). Integrated populations in this context

[1] DFO. In Press. Integrated Fisheries Management Plan Salmon Enhancement Program (SEP) data tables, Pacific Region https://www.pac.dfo-mpo.gc.ca/sep-pmvs/data-donnees/index-eng.html

are those in which natural and hatchery origin salmon spawn in both the hatchery and natural environments, with gene flow between the two. PNI is calculated by dividing the proportion of natural origin broodstock by the combination of natural origin broodstock and hatchery origin spawners. The PNI metric is associated with specific biological designations which progress from "wild" to "integrated-hatchery", as defined in Withler et al. (2018). With an average PNI of 0.72 (2019–2024), the Yukon River Chinook salmon populations above the fishway (part of the Upper Yukon River CU; CK-69) are considered 'integrated-transition', where natural origin salmon are in the majority, and gene flow is dominated by the natural environment (Figure 17). To minimize long-term genetic risks it is typically recommended that rebuilding programs work toward an integrated-wild designation (PNI>0.8), where natural productivity is sufficient.

In-stream incubators are used as a low-cost means to improve fertilization rates and in-stream egg to emergence survival. Although they have been used at multiple sites in the Yukon watershed (e.g., Deadman Creek (Environmental Dynamics Inc. 2021a), Ibex Creek (Environmental Dynamics Inc. 2021b), Nisultin River (Environmental Dynamics Inc. 2023a), Fox Creek (Ta'an Kwäch'än Council 2024)), there is currently no centralized database of these activities. Success of in-stream incubators can be highly variable depending on site conditions, weather, and methodology, among other factors (Environmental Dynamics Inc. 2021a).

5.2. POTENTIAL ENHANCEMENT PLANNING ACTIVITIES THAT COULD CONTRIBUTE TO STOCK REBUILDING

Enhancement activities for rebuilding/conservation purposes should address identified bottlenecks to salmon survival and should be part of a comprehensive recovery strategy that considers habitat and harvest measures. However, enhancement for rebuilding/conservation often incurs heightened risks to wild populations and must be characterized by well-defined objectives, a clear plan to transition through recovery stages, a robust monitoring program, and strong safeguards that acknowledge the potential genetic, ecological and disease risk associated with enhancement (DFO 2013). This can be best achieved with enhancement plans that summarize enhancement objectives, goals and intended outcomes, developed in consultation with First Nations and stakeholders. Key initial activities to consider for Yukon River Chinook salmon, drawn from guidance on conservation hatcheries, include (Bradford et al. 2025):

1. Identify threats and stressors that are contributing to low returns and determine if hatchery enhancement could contribute to rebuilding/conservation efforts. For example, hatchery management measures may help mitigate or avoid freshwater bottlenecks or promote returns to preferable spawning habitat through various release strategies.
2. Assess the likelihood of recovering a self-sustaining wild population, which will inform the enhancement goals, approach, and possible outcomes. For example, where recovery is possible, short-term supplementation can accelerate recovery or stabilize a population while threats/stressors are addressed. Where recovery is unlikely, a hatchery program may be used to satisfy other values, including human and ecosystem use. However, exceptional clarity on the objective of the enhancement would be needed prior to developing these programs, and for Yukon Chinook, should be limited to conservation or small scale low-risk stewardship or assessment objectives.

«««< HEAD 3. Design an enhancement plan that is based on spawner recovery goals and corresponding recovery phases and triggers. Rebuilding focused enhancement plans should include defined spawning escapement targets, fisheries reference points, benchmarks that are explicitly tied to wild escapement, and timelines; where hatchery, habitat, and harvest measures are coordinated to meet clearly articulated population, Conservation Unit or stock management goals.

======= 3. Design an enhancement plan that is based on spawner recovery goals and corresponding recovery phases and triggers. Rebuilding/conservation focused enhancement plans should include defined escapement targets, fisheries reference points, benchmarks that are explicitly tied to wild escapement, and timelines; where hatchery, habitat, and harvest measures are coordinated to meet clearly articulated population, Conservation Unit or stock management goals. »»»> 48bbb9cc833f0a7b0fff86075bbfcc115d460476

6. HABITAT

6.1. FRESHWATER HABITAT CONDITIONS

The terrain of Yukon includes mountains, plateaus, and valleys that support a diverse mosaic of forest, tundra, grassland, shrubland, and wetland ecosystems. Terrestrial and aquatic ecosystems are considered to be relatively intact across the Yukon watershed (Brabets et al. 2000) and access to most of the watershed is limited to boat or aircraft. The Yukon River has considerable seasonable variability in environmental conditions, with summer flows substantially higher than winter flows (Brabets et al. 2000). Streamflows as low as 1,200 m^3s^{-1} occur in the winter, and streamflow discharge pulses of more than 19,800 m^3s^{-1} occur in the spring (USGS 2022). The river freezes throughout the entire watershed in the winter and ice breakup occurs in late spring, typically by mid-May (Eiler et al. 2023). The mainstem river is relatively deep, greater than 20m in the lower river, but becomes shallower, 8–10m, in the Canadian portion of the drainage (Eiler et al. 2023).

The freshwater habitats that Yukon River Chinook salmon utilize are undergoing significant environmental transformations caused by climate change, which are thought to play a role in the decline of their populations (Smith et al. 2004; Murdoch et al. 2024). Chinook salmon migrating through the Yukon River basin regularly encounter water temperatures that induce stress responses (above 18°C, U.S. EPA (2003)), particularly in the Porcupine drainage, leading to reduced reproductive success (von Biela et al. 2020; Murdoch et al. 2024). Over the past several decades, earlier spring melt and river ice breakup have altered migration patterns and habitat availability for aquatic species (Smith et al. 2004; Bush and Lemmon 2019). Thawing permafrost and increased summer precipitation are contributing to higher streamflows and greater sedimentation, and more frequent extreme weather events impact water clarity and quality. These changes are also contributing to unstable riverbanks, further degrading critical spawning and rearing habitats (Murdoch et al. 2024). There is continuous permafrost throughout the Porcupine SMU and part of the Northern Yukon CU (Heginbottom et al. 1995), and these populations are therefore most likely to experience changes in water quality due to permafrost thawing.

These environmental impacts are expected to intensify under future climate change. More frequent and intense floods due to increased precipitation during the summer months could displace juvenile salmon from their rearing habitats, increasing mortality risks (Murdoch et al. 2024). As glaciers retreat, summer water flows in glacial systems may become less predictable, with potential reductions in water availability during critical salmon migration periods (Smith et al. 2004). Effects of glacial retreat on stream flow and timing have already been seen, notably in Kusawa Lake in southern Yukon (Upper Yukon CU) (Zdanowicz et al. 2018) and in Kluane Lake and Slims River (White CU) with the retreat of the Kaskawulsh Glacier (Shugar et al. 2017).

6.2. HABITAT RESTORATION AND MONITORING ACTIVITIES THAT COULD CONTRIBUTE TO REBUILDING

In response to the challenges faced by Chinook salmon across the Canadian portion of the Yukon and Porcupine watersheds, a range of stewardship and restoration activities have been implemented or are being considered. These efforts include watershed and habitat protection, targeted monitoring programs, and cold-water refuge conservation, all aimed at promoting the long-term health and recovery of Yukon River Chinook salmon.

Protecting and restoring creeks and rivers, particularly those vulnerable to increased precipitation and sedimentation, is thought to be important for juvenile salmon survival (Murdoch et al. 2024). Improved environmental monitoring can help to track stream temperatures, flow regimes, and sediment levels to detect early changes in freshwater conditions (Murdoch et al. 2024). Yukon First Nations, Yukon First Nations Salmon Stewardship Alliance, Government of Yukon, and Fisheries and Oceans Canada are working to standardize temperature monitoring throughout the watershed. Sedimentation and high water temperatures frequently occur in areas where the riparian zone is degraded, negatively affecting spawning and rearing habitats (Pusey and Arthington 2003). Restoring riparian zones involves revegetating stream banks and adding large organic debris to stabilize banks and improve habitat quality, which provides shade, regulates water temperature (Kirkland and Wondzell 2020; Varner et al. 2023), and contributes essential nutrients for salmon. Restoration and reclamation should also include improving floodplain connectivity and lateral stability (Chevreaux and Clarkson 2015; Varner et al. 2023).

Improving fish passage may also support habitat restoration efforts in some portions of the river basin. Barriers like dams and culverts can obstruct salmon migration routes, preventing access to spawning and rearing habitats. Only two significant dams impact the Yukon River Chinook salmon –the Mayo Generating Station completely blocks upstream passage of Chinook salmon of the Stewart CU and the Whitehorse Rapids Generating Station has been shown to have low passage success (Twardek et al. 2023a) for the Upper Yukon CU. Water relicensing processes for the Mayo Generating Station and the Whitehorse Rapids Generating Station are underway, and public comment during both processes have called for installing fish passage (Yukon Energy 2024) and improving fish passage, respectively. Such projects should identify and prioritize barriers for remediation, design and implement fish passage solutions such as fish ladders or culvert replacements, and evaluate their success in facilitating salmon movement. Yukon River Drainage Fisheries Association (YRDFA) identified multiple culverts of potential concern for salmon passage, highlighting the need for such assessments (YRDFA 2025) and research has emphasized the need for connectivity between spawning habitats and unobstructed access to them (Isaak et al. 2007; Brown et al. 2017). More work with landowners and biologists to idenitify problem culverts in the Canadian portion of the watershed is required. Additionally, there has been increasing scrutiny on beaver dams, which can create passage barriers and reduce connectivity, both for juveniles accessing non-natal streams and for returning spawners (Connors et al. 2016; Malison et al. 2016). Conversely, in some systems there is evidence that ponds created by beaver activity can benefit rearing salmon (Pollock et al. 2004), and the effect of beaver presence may be highly case-dependent.

Identification and preservation of cold-water refugia are also being considered to protect areas critical to salmon migration and spawning, especially as in-river temperatures rise. Migrating salmon rely on cooler, non-mainstem rivers to reduce stress during these warmer conditions (von Biela et al. 2020; Eiler et al. 2023). More work is required to locate these refugia. Additionally, identification of groundwater sites used for rearing is critical, as groundwater supplies possibly impact nutrient loads when surface flows are frozen in the winter (Bradford et al. 2001). Identification of juvenile habitat through eDNA or juvenile sampling can help to identify the groundwater locations and habitat to target for regulatory protection. Projects like these, which have already begun primarily in the upper watershed, help identify the habitat use of Yukon River Chinook salmon throughout their life cycle. Increasing the coverage to identify thermal refugia and groundwater throughout the watershed will give a more fulsome picture of productive habitats and contribute to hydrological databases.

Furthermore, water quality is a component of habitat restoration efforts. Pollution, nutrient runoff, and downstream impacts from mining can degrade water quality, affecting salmon health and survival (Gormley et al. 2005; Sergeant et al. 2022). Identifying sources and levels of pollutants, and understanding their impact on salmon populations, are necessary to inform mitigation strategies. While much of the Yukon River watershed is minimally impacted from land use practices and wastewater management, gold and quartz mining are sources of water contaminants, and legacy mining can have persistent effects for years after remediation (Sergeant et al. 2022). In June 2024, a catastrophic failure of the Eagle Gold Mine heap leach pad resulted in a landslide releasing approximately 300 million liters of cyanide laden solution into the surrounding environment. Remediation efforts are ongoing, but elevated levels of cyanide, cobalt, copper, and nitrite have persisted to varying degrees in nearby water bodies, including a tributary of the McQuesten River, a known spawning system in the Northern Yukon CU (Government of Yukon 2025). The water quality is being closely monitored, but the impacts of the incident on Yukon River Chinook salmon are not yet known. A mining-related spill of 100 million liters of cyanide and heavy metal-laden solution into a tributary of the Tisza River in Romania and Hungary in 2000 caused mass die-offs of many fish species. Following the acute toxicity of the spill, an ecotoxicology study conducted over two years found that while the aquatic community recovered, plants and algae bioaccumulated heavy metals, possibly putting higher tropic levels at risk (Lakatos et al. 2003). Continued water quality monitoring, particularly in the area of the McQuesten River and in the Klondike River watershed, is critical for determining impacts to salmon from mining.

There is increasing recognition that a key aspect of any forward-looking restoration is to incorporate Indigenous Knowledge into restoration efforts, as Indigenous communities possess valuable traditional ecological knowledge that can lead or contribute to grounded and impactful conservation strategies. For example, Indigenous Knowledge can provide insights into ecological baselines, such as indicators of salmon abundance (e.g., number of carcasses, smell along riverbanks) or harvest (e.g., reduction in kilograms consumed) which are not captured through conventional monitoring methods (Reid et al. 2022a; Connoy et al. 2024). Complementing Indigenous Knowledge with western science confirmed the role of salmon as ecological and cultural keystone species and found that riparian tree growth and nitrogen enrichment was positively related to salmon escapement (Connoy et al. 2024).

Future efforts may also require changes to regulatory processes to protect salmon habitats. Key recommendations from other works include updating the Fish Habitat Management System for Yukon placer mining to reflect current science, addressing outdated habitat models, and fostering collaboration with Yukon First Nations and industry (YSSC 2025). Additionally, assessing cumulative impacts under key legislation like the Waters Act and modifying the Yukon Environmental and Socio-economic Assessment Act process would allow for stronger salmon conservation measures (YSSC 2025). Removing the free-entry mining system and linking staking rights to development would enable more deliberate decision-making, incorporating environmental reviews and community input (YSSC 2025). These changes aim to strengthen governance and safeguard Yukon's salmon-bearing rivers.

Targeting restoration efforts to those habitats that are hypothesized to be bottlenecks to salmon survival is likely to be the most impactful use of resources. However, some challenges are unlikely to be fully addressed (e.g., those resulting from climate change) except through international or global scale policy. There is growing appreciation for the dynamic nature of salmon habitat and the importance of protecting and preserving the heterogenous and complex habitat landscapes that salmon use and the processes that generate them (Moore and Schindler 2022).

6.2.1. CURRENT HABITAT RESTORATION AND MONITORING PROJECTS

Numerous projects funded through the Yukon River Panel's Restoration and Enhancement Fund are advancing stewardship and habitat restoration efforts across the Yukon River watershed. Additional funding mechanisms and projects are conducted throughout the watershed, though not touched on here. The following paragraphs are just some of the examples of restoration and monitoring occurring. Habitat restoration initiatives such as the Fox Creek stream channel restoration and McIntyre Creek streambank stabilization aim to improve salmon habitat by addressing impacts of beaver dams and streambank erosion (Crawford 2023). In-stream incubation of eggs collected from local broodstock has been seen as an alternative to hatchery operations to restore Yukon River Chinook. Multiple year-long in-stream egg incubation projects have been conducted on the traditional territories of Kwanlin Dün First Nation, Teslin Tlingit Council, Tr'ondëk Hwëch'in, Vuntut Gwitchin First Nation, and Ta'an Kwäch'än Council (M. Milligan, pers. comm.) with varying success (Environmental Dynamics Inc. 2021a, 2021b).

Monitoring projects underway or recently completed include the Michie Creek Salmon Monitoring Project, which tackles migration barriers for outmigrating Chinook (Jessup and Graff 2024), and the Takhini River Chinook Restoration and Sonar Initiative, which employs drone and sonar technology to monitor Chinook populations (Hoogland and Therriault 2025). The Beaver River watershed Chinook survey (Mantyka-Pringle et al. 2020) focused on understanding Chinook population dynamics to inform conservation strategies while in the Porcupine River the Chinook Salmon Telemetry Project tracked migratory behavior using telemetry (DFO 2017; Environmental Dynamics Inc. 2017). Tr'ondëk Hwëch'in is monitoring Chinook salmon passage through sonar, and supporting community engagement, youth education, and stewardship through their First Fish initiative (Environmental Dynamics Inc. 2023b; Vittrekwa 2023).

These restoration projects also provided promising future directions for ecosystem recovery and stock rebuilding. For instance, drone surveys in the Takhini River will be expanded to identify critical habitats and improve species monitoring. A new sonar site on the Stewart River will assess hydrology, turbidity, and fish passage to support management. In the Klondike watershed, a revised restoration plan could pave the way for small community enhancement efforts. Temperature monitoring will be broadened in key migration corridors such as the Porcupine, Upper Pelly, and White rivers, as well as work to identify cold-water refuges like the Big Salmon and White River tributaries. Planned sonar station upgrades and expanded telemetry studies will boost data reliability, aiding both population tracking and habitat restoration planning efforts. Additionally, targeted habitat enhancement in the face of a changing climate could help to alleviate some of the pressures faced in the freshwater environment.

7. IMPACT OF FISHERY MANAGEMENT MEASURES

7.1. CLOSED LOOP SIMULATION MODEL

We developed a simple closed loop forward simulation, conditioned on our estimates of historical spawner abundance, harvest, age composition, and time-varying productivity (Figure 18). This model was used to shed light on expected biological and fishery consequences of current and alternative harvest management measures (i.e., Harvest Control Rules [HCRs]) for the Yukon Chinook SMU. The simulation started in the last year for which we had empirical observations (2024) and projected the dynamics of individual CUs forward in time. In each year of the simulation, the model (1) generated a forecast of total returns (with empirically based forecast error), (2) applied a given HCR (see fishery sub-model description below) with outcome uncertainty because management control is not perfect, and (3) allowed remaining fish to spawn and then return in subsequent years across a range of ages at maturity. Details on model components and calculation of performance are provided in the sections below.

7.1.1. OPERATING MODEL

The operating model was used to simulate future population trajectories for the nine Yukon River Chinook salmon CUs (those in the Yukon Chinook SMU) to which we were able to fit spawner-recruitment models. The operating model projected their population dynamics forward over 26 years (i.e., to 2050, approximately four Chinook salmon generations), thereby generating a distribution of future states conditioned on the historical data. By simulating Yukon River Chinook salmon dynamics in this manner, we ensured that predicted future spawner abundance and age structure were conditioned on the incomplete cohorts at the end of the data series (i.e., those cohorts from which one or more older age classes have not yet returned to spawn) and that uncertainties in the spawner-recruit relationships were propagated through time (i.e., by drawing from the joint posterior distributions of each estimated parameter and abundance state in each iteration of the simulation).

Population dynamics were assumed to be driven by Ricker (1954) type spawner (S) and recruitment (R) relationships:

$$\ln(R_{i,y}) = \ln(S_{i,y}) + \ln(\alpha_i) - \beta S_{i,y} + v_{i,y} \tag{1}$$

where α is productivity (intrinsic rate of growth), β is the magnitude of within brood year (y) density-dependent effects for CU i, and $v_{i,y}$ is lag-1 autocorrelated recruitment residuals (i.e., $v_{i,y} = \phi v_{i,y-1} + \varepsilon_{i,y}$). Correlation in recruitment residuals (ϕ) was fixed at the average estimate across CUs (0.75) and the portion of recruitment anomalies that is temporally independent, $\varepsilon_{i,y}$, was modeled as a multivariate normal vector:

$$\varepsilon_{i,y} \sim \mathcal{MVN}\left(\mathbf{0}, \varepsilon_R\right) \tag{2}$$

where the root diagonal of the covariance matrix ε_R is the CU specific inter-generational variation in survival and where correlation in the recruitment residual time series vectors $\boldsymbol{\varepsilon}_i$ and $\boldsymbol{\varepsilon}_k$ between CU i and k ($\rho_{i,k}$) is captured in the off-diagonal covariance elements of ε_R. This variance-covariance matrix was fixed at the median estimates of the variance-covariance matrix of recruitment residuals by CU and year as estimated in the spawner-recruitment model described in Appendix C.

Returns in calendar year t, $N_{t,i}$, were then modeled as a function of the proportion of individuals that mature and return at each age:

$$N_{t,i} = \sum_{a=4}^{7} R_{(t-a)} p_{(a-3)} \tag{3}$$

where p is the maturity schedule composed of four age classes (ages four through seven). In each year of the forward simulation, the number of spawners in Equ. 1 was calculated as the total return (Equ. 3) minus aggregate harvest which was determined according to the harvest control rules described in the following section.

In each iteration of the forward simulation (n = 1000) the operating model was parameterized by taking a draw from the joint posterior distributions of the relevant parameters from the spawner-recruitment models with time-varying intrinsic productivity fit to each CU (Equ. C.1-C.10). A summary of the parameters is provided in Tables 10 and 11.In our reference (base) scenario, we projected system dynamics forward using the average α estimates of the six most recent completely observed brood years (2013–2018), which corresponds with the 6-year generation length for Yukon River Chinook salmon (Figure 19). As a robustness (alternative or "worst case") scenario we also projected the system forward in time assuming intrinsic productivity was instead equal to the last empirical estimate of time varying productivity (i.e., 2017 brood year) which serves as a test of the consequences of alternative harvest management measures if productivity were to remain severely depressed for the next several generations (~35% on average lower than most recent generation average; Figure 19). We also considered another robustness ("best case") scenario where the system was projected forward using the median α estimates from all fully observed brood years (1985–2017), which illustrates the outcomes if the stock were to return to long term average productivity (Figure 19, Table 11). It is possible that productivities in future years may also become more variable (e.g., due to climate induced extreme events) but for simplicity such scenarios were not considered.

For simplicity we chose to not explicitly model spawner demographics and the consequences of changing demographic characteristics. We chose to do this because we consider our demographic spawner recruitment models to be exploratory/illustrative (described in more detail in Appendix C) and because by conditioning our operating model on recent productivity we are in effect implicitly accounting for the cumulative consequences of both demographic and environmental/ecosystem change on the intrinsic productivity of each CU.

7.1.2. FISHERY SUB-MODEL

In each year of the simulation, total returns were assumed to be forecasted with error. This error was assumed to be lognormally distributed with a mean equal to the true run size and a standard deviation of 0.79, based on a retrospective assessment of the mid-point of preseason forecasts and true run size 2000-present (JTC 2025). Forecasted returns were then used as an input into a given HCR that specified a target exploitation rate given the expected run-size. Outcome uncertainty (i.e., deviations from targeted catch due to imperfect management control) was then applied to calculate realized catch, where outcome uncertainty was assumed to be equal to a CV of 10% which is of a magnitude similar to that assumed in other large Chinook salmon fisheries in the region (Connors et al. 2022b). We do not simulate in-season management or forecast adjustments, therefore our simulations likely over-estimate the true magnitude of combined forecast and management error.

The harvest control rules (HCRs) we considered are shown in Figure 20 and described in Table 9. These alternative fishery management measures ranged from a no fishing scenario to illustrate expected recovery potential, to fixed exploitation rate rules intended to simply illustrate the expected impact of a range of harvest rates, to escapement goal type rules (with and without caps on maximum exploitation) that were intended to approximate a range of general harvest management strategies. For the latter general harvest management strategies it was assumed that the current 7-year agreement ("moratorium") rule was in effect until 2030, after which the given HCR being simulation tested took effect.

With the exception of the fixed exploitation rates, harvest control rules were defined by a a lower management reference point and in some cases an upper reference point and/or maximum removal reference, all defined at the SMU scale. When there was both a lower and upper reference point, harvest rates linearly increased from zero at spawner abundances below the lower reference point to the maximum removal reference at run sizes equal to the spawner abundances associated with the upper reference point. The maximum removal rate was never allowed to exceed 80% under any HCRs because this is the maximum estimated aggregate harvest rate to have been experienced by the stock (in 1987) and for those HCRs with a cap on the maximum allowable exploitation rate it was set to 36% which was the the median estimate of U_{MSY} for the least productive CU (Section 3.3.2). Conservation Units were assumed to be equally vulnerable to harvest, and low levels of incidental harvest (~1000 fish) were assumed to occur even in years where the HCR being applied would call for a fishery closure to account for mortality due to assessment projects and unregulated harvest.

7.1.3. PERFORMANCE MEASURES

We quantified the expected performance of current and alternative harvest management measures against biological and fishery objectives and associated performance measures (Table 12). We considered the primary biological objectives to be minimizing the number of CUs below their lower biological benchmarks, and maximizing the number of CUs above their upper biological benchmarks. We derived these benchmarks from the spawner-recruitment models that assumed time-invariant productivity (Appendix C) and note that simply reporting the number of CUs that fall above their lower biological benchmarks may underestimate extinction risk which is more explicitly accounted for in the formal Wild Salmon Policy rapid status assessments. We also calculated the number of CUs above their rebuilding target, which was (100%) S_{MSR} based on egg mass recruitment models with time-invariant productivity where S_{MSR} is based on demographic characteristics of spawners over the most recent generation of data (2019-2024). In addition to these CU scale status performance measures, we also calculated average annual aggregate spawner abundance.

Fishery related performance measures included realized total aggregate harvest, realized aggregate harvest rate, the proportion of years when the fishery is closed, and average annual total harvest occurring in Canada. Under the harvest sharing provisions of the Yukon River Salmon Agreement (Panel 2001), 20%–26% of total allowable catch is allocated to Canada when projected Total Allowable Catch is less than 110,000, which we simplified to a target of 23% in order to calculate the total Canadian harvest.

7.2. CONSEQUENCES OF CURRENT AND ALTERNATIVE HARVEST MANAGEMENT MEASURES

Under the reference productivity scenario and no fishing, median spawner abundances were projected to be at or below lower biological benchmarks from 2030–2050 for three CUs: Northern Yukon, White, and Pelly. Two CUs (Middle Yukon and Big Salmon) were projected to be between lower and upper benchmarks, and two (Stewart and Upper Yukon) were above their upper benchmarks and lastly, the Nordenskiold and Teslin CUs were projected to approach or exceed their CU-specific rebuilding targets (Figure 21).

Simulations across a range of fixed harvest rates highlighted a clear tradeoff between mixed-stock fishery harvests and risks to component CUs. Aggregate harvests were projected to be maximized at mixed-stock harvest rates of ~60%, and dominated by harvest from the Middle Yukon and Nordenskiold CUs, but at this harvest rate at least four CUs were projected to be extinct and another three below their lower biological benchmark and hence at an elevated risk of extinction (Figure 22).

The current moratorium on fishing at run-sizes below 71,000 allowed for populations to be stable and modest opportunities for harvest in the forward projections under the reference productivity scenario (Figure 21). Specifically, annual exploitation rates averaged 10%, and total harvest was projected to be ~3,000 fish on average per year, though this was variable from year to year. Notably, this harvest control rule resulted in the fishery being closed to harvest in ~80% of years between 2030–2050. Harvest in Canada averaged ~600 fish per year, which is well below Yukon First Nations Basic Needs Allocation for Chinook salmon which, while not finalized, is the interim allocation of Chinook salmon (10,000 fish) reserved for Yukon First Nations within the current Canadian fishery management strategy (DFO 2024a). Under this harvest control rule median spawner abundances were projected to exceed CU specific lower biological benchmarks for six CUs, and exceed upper biological benchmarks for four of those six. Placing a maximum harvest rate cap equal to the removal reference (RR) on this harvest control rule very slightly reduced average annual harvest rates and harvest but otherwise had little impact on the biological performance measures.

The interim management escapement goal (IMEG) harvest control rule of 42,500 resulted in the highest average harvest rates (~16%) and harvest (~5,000), and lowest aggregate spawner abundance (~25,000), of the alternative fishery management measures considered. These higher harvests resulted in the highest number of CUs falling below their lower biological benchmarks. Under this scenario ~70% of years were expected to have complete fishery closures. Placing a maximum harvest rate cap on the IMEG harvest control rule reduced the average harvest rate to ~14% and reduced the chances of any CUs falling below their lower biological benchmarks. Lastly, the Precautionary Approach alternative, which allowed harvest at lower abundances than the other harvest control rules, had similar biological performance to the moratorium scenarios, but reduced the frequency of fishery closures from four in five years (80%) to approximately one in four (25%, Figure 23, Table 13).

Under the robustness scenario where intrinsic productivity was assumed to remain severely depressed for the next several generations, all harvest control rules resulted in very limited opportunities for harvest and resulted in over half of CUs being extinct or at risk of extinction (Table 14). While relative fishery performance varied slightly among HCRs similarly to under the reference scenario, absolute fishery performance was low in all cases under the depressed productivity scenario.

Under the second robustness scenario where productivity was assumed to return immediately to its long-term average, all harvest control rules resulted in more positive fishery and biological outcomes. Under the IMEG rule, one CU was at risk of falling below its lower benchmark, but placing a cap on the maximum allowable harvest rate removed this risk (Table 15). Notably, this was the only harvest control rule and productivity scenario under which a Basic Needs Allocation for Yukon First Nations was consistently met, assuming that Yukon First Nations continue to get priority access to Canadian harvest.

8. DISCUSSION

8.1. SUMMARY OF KEY FINDINGS

In this report we describe Yukon River Chinook salmon stock structure and distribution, stock assessment history, and analyses that combine genetic stock identification of archived scale and tissue samples with run-reconstruction models to estimate migration timing and annual return abundance for nine Conservation Units (CUs) of the Yukon Chinook SMU. We then characterize CU scale population dynamics for the Yukon Chinook SMU and estimated biological benchmarks by fitting age-structured, state-space spawner (or egg mass) recruitment models to observations of spawner abundances, harvest, and age composition. Anthropogenic, ecosystem, and climate factors affecting both the Porcupine and Yukon Chinook SMUs are described along with general rebuilding and enhancement activities that could contribute to growth of the SMUs. Lastly, quantitative estimates of the expected biological and fishery consequences of current and alternative harvest management measures are generated from a closed loop simulation model for the Yukon Chinook SMU.

Aggregate returns of Yukon River Chinook salmon have declined in recent decades culminating in five of the lowest returns on record in 2020–2024. These declines have placed immense hardship on subsistence users and Yukon First Nations who depend on Chinook salmon for food, social and cultural uses, and prompted a seven-year moratorium on directed harvest in Alaska and Canada beginning in 2024. Declines in female Chinook salmon age at maturity, and to a lesser extent the proportion of females in the spawning population, have resulted in a marked decline in average reproductive output per spawner (~25% from the late 1980s to the early 2020s). Concurrent with these changes intrinsic productivity has declined by an average of 60% across CUs.

Reconstructed spawner abundances at the CU scale for the Yukon Chinook SMU were on average 42% lower in the most recent generation (2019–2024) than preceding generations (1985–2017). Four CUs (Nordenskiold, Upper Yukon, Stewart, and White) were assessed as in the Wild Salmon Policy Red status zone due to their spawner abundances being below an absolute abundance lower threshold of 1,500 spawners. The Teslin and headwaters CU was assessed as in the Red status zone based on trends in spawner abundance from two long-running aerial surveys. Two additional CUs, the Porcupine and Old Crow, were assessed as in the Red status zone due to their combined average abundance over the most recent generation being 695. The remaining CUs were assessed as in the Amber status zone.

Because at least one CU was assessed as falling in the Red status zone in each of the SMUs, both SMUs were assessed as below their CU status based Limit Reference Point. At Yukon Chinook SMU scale run-sizes declined by ~87% from 1981-2024 and aggregate spawner abundances over the most recent generation (2019–2024) averaged 25,852 which is below a proposed Fisheries Reference Point–Lower of 31,000 and well below the proposed Upper Stock Reference point of 86,000 (Section 3.3.2). Realized harvest rates over this same time period averaged 16%. The Porcupine Chinook salmon SMU is also below its CU status based LRP because two CUs were assessed as being in the Wild Salmon Policy Red zone.

The past decade has seen a considerable amount of research into anthropogenic, ecosystem, and climate factors that may have contributed to the recent and long-term declines in body size and productivity of Yukon River Chinook salmon. Leading hypotheses for drivers of these changes include heat stress and *Ichthyophonus* infection during return migrations, and climate induced changes to predator, prey and competitor abundances in the Bering Sea ecosystem that Yukon River Chinook salmon spend much of their marine life in. Several enhancement activities have been directed towards Yukon River Chinook salmon over the past 40 years including hatcheries, incubation facilities, and in-stream incubation projects. These enhancement projects have been modest in nature and while there is general agreement that enhancement may support rebuilding efforts, more research and work is needed to determine where enhancement could be effective and how to minimize the potential for adverse impacts. A range of stewardship and restoration activities have been implemented or considered for Yukon River Chinook salmon. These efforts include watershed and habitat protection through specific changes in regulations, targeted monitoring programs, and cold-water refuge conservation, all aimed at promoting the long-term health and recovery of Chinook salmon returning to the Canadian portion of the Yukon River basin.

Simulations of the Yukon Chinook SMU suggest that the current moratorium on fishing at aggregate run sizes less than 71,000 allows for some stock growth and hence future harvest opportunities over the next 26 years. Spawner abundances are projected to exceed CU-specific lower biological benchmarks for six CUs, and exceed upper biological benchmarks for four of those six provided that productivity remains at its most recent generation average. Under this harvest control rule (HCR), harvest is projected to be ~3,000 fish on average per year (2030–2050) though this is highly variable from year to year and only 15% of years are expected to result in any harvest opportunities. Evaluation of the interim management escapement goal (IMEG) in place prior to the 2024 moratorium suggests that it can achieve higher average harvests but at the cost of one addition CU projected to fall below its lower biological benchmark and only two CUs projected to be above their upper biological benchmarks. Setting a maximum harvest rate cap equal to U_{MSY} for the least productive CU (36%) had little impact on the performance of the moratorium HCR and reduced the chances of any CUs falling below their lower biological benchmarks under the IMEG HCR. Lastly, a Precautionary Approach compliant HCR, which allowed harvest at lower abundances than the other HCRs, had similar biological performance but reduced the frequency of fishery closures from four in every five years to one in every two years. These simulations assumed the intrinsic productivity of the CUs that make up the Yukon Chinook SMU is depressed but remains the same as it has been over the last generation. Under an alternative “worst-case” scenario where productivity is assumed to remain severely depressed (~35% below most recent generation average) all alternative harvest management measures considered resulted in very limited opportunities for harvest, and up to six CUs were projected to fall below their lower biological benchmarks by 2050, and another was projected to reach extinction. Under an alternative “best-case” scenario where productivity is assumed to revert to its long-term average, harvest opportunities predictably increased, all CUs were projected to be above their lower biological benchmarks and eight CUs were projected to exceed their rebuilding targets by 2050 in the absence of fishing.

8.2. EXCEPTIONAL CIRCUMSTANCES OR ASSESSMENT TRIGGERS FOR THE STOCK

Exceptional circumstances are assessment triggers intended to proactively identify conditions and/or circumstances that may represent a substantial departure from those under which the advice in this assessment was developed.

In addition to routine re-assessment every generation to ensure stock status is updated and captures contemporary fishery and biological processes, we recommend a re-assessment be triggered if any of the following occur:

- Stock productivity declines drastically, where the median estimate of time-varying productivity (annual CU-specific Ricker α) falls outside the 50th percentiles of the productivity used to condition our forward simulation under the robustness test (most recent α);
- Management measures, and performance statistics, other than those evaluated in this research document are considered by fisheries management;
- New information becomes available that results in changes to our understanding of stock-structure (e.g., the current Conservation Unit delineations) and/or major drivers of stock dynamics; or
- New information becomes available that results in changes to the historical time-series of spawner abundance and/or catches used in this research document (e.g., historical estimates of migration mortality are derived and incorporated into the run-reconstructions).

8.3. KEY KNOWLEDGE GAPS AND AREAS OF POTENTIAL FUTURE WORK

As is inevitable with any analyses of large and complex ecological and fishery datasets, we had to make a number of simplifying assumptions. These include:

1. **Assuming all Conservation Units have historically been equally vulnerable to fisheries**. Our reconstructions of CU scale harvests, and subsequent population dynamics modelling and estimation of biological benchmarks, relied on reconstructed estimates of Yukon River Chinook salmon harvest rates (Connors et al. 2022a) which were then used to calculate CU-specific harvest based on reconstructed spawner abundance. This approach implicitly assumes that all CUs have historically experienced the same harvest rates in both Yukon Territory and Alaska. We consider this a reasonable assumption because, in a general sense, Canadian-origin Chinook salmon make up the first “pulse” of Chinook salmon to enter the Yukon River each year and subsistence and commercial fisheries (e.g., openings, closures, and gear restrictions) have historically been managed by pulse. And for those few years where genetic sampling of harvest in Alaska allowed for a comparison of CU composition in both border passage and harvest there was no evidence of consistent over/under-representation of individual CUs in Alaskan harvest (Appendix D, Figure D.3). Nonetheless, it is possible that differences in run timing, and the timing of fisheries during the first pulse in the lower river in Alaska, have led to earlier returning fish being more vulnerable to harvest than later returning fish. This has the potential to underestimate harvest rates for populations that return relatively early and overestimate harvest rates for those that return relatively late (e.g., Northern Yukon and White CUs) which in turn may under- and over-estimate productivity, respectively. However, the actual timing of lower river harvests has varied over time and is difficult to separate from the harvest of co-migrating Chinook salmon that spawn in the middle and lower portions of the Yukon River Basin. Thus, it may not be possible to accurately reconstruct the timing and magnitude of the gauntlet of commercial and subsistence

fisheries that historically occurred and that Canadian-origin Yukon River Chinook salmon would have experienced along the length of the Yukon River.

2. **Assuming there has been no en-route adult migration mortality from the U.S.–Canada border to spawning grounds**. Our reconstructions of CU scale spawner abundances assumed that the only source of mortality after passing the U.S.–Canada border was harvest. However, elevated water temperatures and *Ichthyophonus hoferi* disease have been identified as two relatively common stressors that may result in premature mortality of Yukon River Chinook salmon throughout the U.S. or Canadian portions of the drainage during their adult spawning migrations (von Biela et al. 2020; Connors et al. 2022a; von Biela et al. 2022; Howard and von Biela 2023). Historical tagging studies of return adult Chinook salmon have found that years when freshwater migration conditions were relatively cool and relatively warm had comparable movements and survival rates (Eiler et al. 2023). However, more recent tagging studies, undertaken as a collaboration between ADFG, DFO, and Yukon First Nations, and during a period of higher migration temperatures than in previous studies, suggest that not all tagged fish make it to their spawning grounds. However, this work is ongoing and at present the magnitude of en-route adult migration mortality exhibited by Yukon River Chinook salmon on an annual basis is not known, but likely varies substantially in space and over time. Failing to account for en-route adult migration mortality in our CU run-reconstructions would result in overestimates of spawner abundance in years in which it occurs, and potential for biologically optimistic estimates of CU status. For this reason, assessment of biological status for headwater CUs were deemed to be of lower confidence if they relied on border passage reconstructions instead of assessments on or closer to spawning grounds.

3. **Assuming Conservation Unit age composition can be approximated by Canadian-origin stock aggregate age composition**. Our CU scale spawner recruitment analyses assumed all CUs shared a common age-at-return that varied through time. While there is undoubtedly some among-CU variation in age-at-return, this assumption was a necessary simplification due to the availability of age composition information at the CU level. Age composition data from fish sampled at the U.S.–Canada border can be assigned to individual CUs from ~2005 to present, but our ability to link genetic and age information for samples prior to 2005 was limited, and tributary scale age composition is very patchy. For those years where we can compare CU-specific and aggregate age composition, there is not strong evidence of differences in estimated age compositions (Appendix D; Figures D.1 and D.2), but it is possible that relatively subtle among-CU differences in age composition could bias our estimates of recruitment (the sum of all fish that returned for a given spawning cohort). This could in turn influence our inference about the shape of the spawner-recruitment relationship and resulting estimates of population productivity and abundance (Zabel and Levin 2002) though these impacts may be relatively small (Peacock et al. 2020).

4. **Assuming future productivity will remain similar to that observed over the most recent generation**. Our closed-loop simulation evaluation of alternative fishery management measures required us to make assumptions about what CU-scale intrinsic productivity was likely to be over the course of the simulations. There is clear evidence of interannual variation and declines in intrinsic productivity over time, so we assumed that estimates of most recent generation of CU-specific productivity were a reasonable proxy for the coming decades. Given uncertainty in what future productivities might be we also considered two contrasting alternative scenarios where productivity is either assumed to remain severely depressed (~35% below most recent generation average), or reverts to it's long-term average,

moving forward. Assumptions about intrinsic productivity will impact projection of recovery potential and fishery outcomes, and while both productivity scenarios we considered were based on empirical observations, it is possible that productivity continues to decline further.

There are several areas of assessment and research that warrant attention moving forward. Doing so can help to resolve key uncertainties and improve the scientific basis upon which management decisions are made. These include, but are not limited to:

- **Increased/continued tributary scale assessment effort**. While the analytical work detailed in this report represents our best efforts to shed light on CU scale dynamics and status based on the patchwork of available information, quantitative models that attempt to reconstruct dynamics from limited data are no substitute for long-term investment in spawning ground assessments at the tributary and/or CU scale. All Yukon Chinook SMU CUs currently have tributary scale assessments projects in operation with the exception of the Stewart and Nordenskiold CUs (Table 2), and estimates of spawner abundances from these tributary projects tend to correlate fairly well with the CU scale run reconstructions (Figure 8). Continued investment to ensure these assessment projects remain in operation, and consideration of opportunities to develop assessment(s) in the Stewart and Nordenskiold CUs, will help ensure that empirical estimates of spawner abundance are available to compliment the run-reconstructions and assess CU status moving forward. In addition, further development of the CU scale run-reconstruction models could consider attempting to fit to the tributary indices of abundance where possible.
- **Improved assessments, run reconstructions, and benchmarks and reference points for the Porcupine Stock Management Unit**. The Porcupine Chinook SMU is largely data limited. Sonar assessments for two of three CUs (Old Crow and Porcupine) began in 2014, but there is no assessment of the Salmon Fork CU. Improved assessment and continued monitoring of Chinook salmon in the mainstem Porcupine River would help to produce run reconstructions and reference points for two of the CUs. The Salmon Fork River breaks off from the mainstem Porcupine River near Old Rampart, Alaska, and due to the remoteness of the location, there are no assessment projects for this portion of the watershed. Beginning assessments for the Salmon Fork River, or developing a radio-tagging project to identify the spawning locations of the Chinook salmon in the Salmon Fork River, would help to address knowledge gaps in the Porcupine SMU.
- **Improved understanding of en-route migration mortality and its implications for CU assessment and evaluations of fishery management measures**. Given that the magnitude of en-route adult migration mortality exhibited by Yukon River Chinook salmon on an annual basis is not known, but likely varies substantially in space and over time, continued investment in telemetry assessment projects, and research into correlates of potential migration mortality back through time, will enable future evaluation of the consequences of accounting for this mortality on CU assessment and forward looking evaluations of fishery management measures. This work could include exploration of the use of tributary indices of spawner abundance as an additional source of data the multi-CU reconstruction model is fit to and potential to directly estimate migration mortality (e.g., as informed by the difference between border passage and spawning ground reconstructed abundance).
- **Improved understanding of contribution of individual CUs to harvest in marine and in-river fisheries**. Given that the vulnerability of individual CUs to harvest in both BSAI groundfish fisheries, and fisheries in the Yukon River, is largely unknown, consideration should be give to routine estimation of CU scale composition of harvest in fisheries that may

intercept Yukon River Chinook salmon. This should include marine and in-river Chinook harvest in the United States and Canada and would allow for a potentially more rigorous assessment of degree to which there is differential CU vulnerability to harvest, and development of CU scale fishery management measures if/as necessary.

- **Climate and ecosystem informed life cycle modelling**. Assessments of juvenile Yukon River Chinook salmon abundances in the Bering Sea have occurred for the past several decades. Fish sampled in these surveys have been assigned to Canadian, middle Yukon, and lower Yukon stock groups based on genetics, but consideration should be given to whether they could be assigned (historically and/or moving forward) to CUs in a manner similar to what we have detailed for adults sampled at the U.S.–Canada border. At either scale, these juvenile data can be used to inform life cycle models that could explicitly incorporate and update those ecosystem and environmental conditions that have previously been shown to correlate with Yukon River Chinook salmon productivity (Cunningham et al. 2018; Murdoch et al. 2024; Feddern et al. 2024). When coupled with future projections of climate and ecosystem change, such analyses may help shed light on future recovery potential and impacts of alternative management measures.
- **Revisiting Conservation Unit delineations and measures of genetic diversity**. The current CU delineations are based on a genetics baseline and analyses that are over a decade old (Holtby and Ciruna 2007; Holtby 2011). These delineations should be revisited based on the updated genetics baseline collection, and updated evaluations of genetic evidence for population structure along with traditional and local ecological knowledge.

9. ACKNOWLEDGEMENTS

We are deeply appreciative of the countless individuals and organizations that have collected data on Yukon Chinook salmon for over 60 years in both the U.S. and Canada. Without their efforts the information and analyses we describe in this report would not have been possible.

We thank Dan Greenberg for analytical advice, Luc Glover and Marina Milligan for their input on assessment projects, Joe Tadey for providing additional hatchery context, Jaclyn Kendall and Steve Smith for helpful background context, Cheyenne Bradley and Priyadarshini Dutta for feedback on an earlier version of this document, and Carmen Gemmell for creating the maps. Eric Rondeau and Janine Supernault carried out the genetic analyses we relied upon for our work and Fred West provided historical forecasts and postseason run-size estimates that we based our forecast error estimates on. Sue Grant and Gottfried Pestal helped lead the Wild Salmon Policy rapid status assessments. Lastly, Carrie Holt, Zachary Liller, and Allison Dennert provided detailed and constructive reviews which, along with feedback from participants at the Regional Peer Review meeting, greatly improved the Research Document.

10. FIGURES

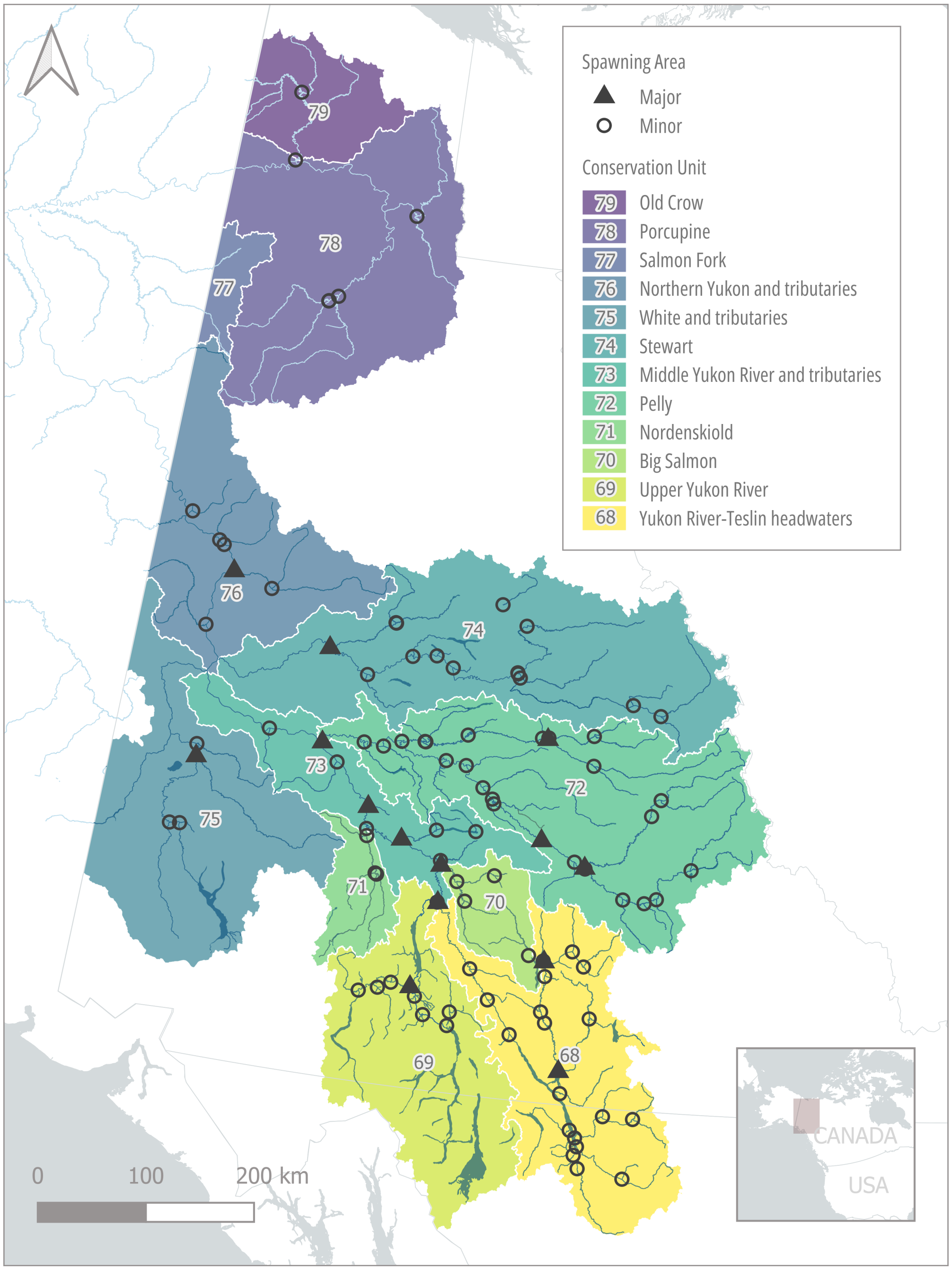

Figure 1. Yukon River Chinook salmon Conservation Units and "major" and "minor" spawning locations as cataloged in Brown et al. (2017).

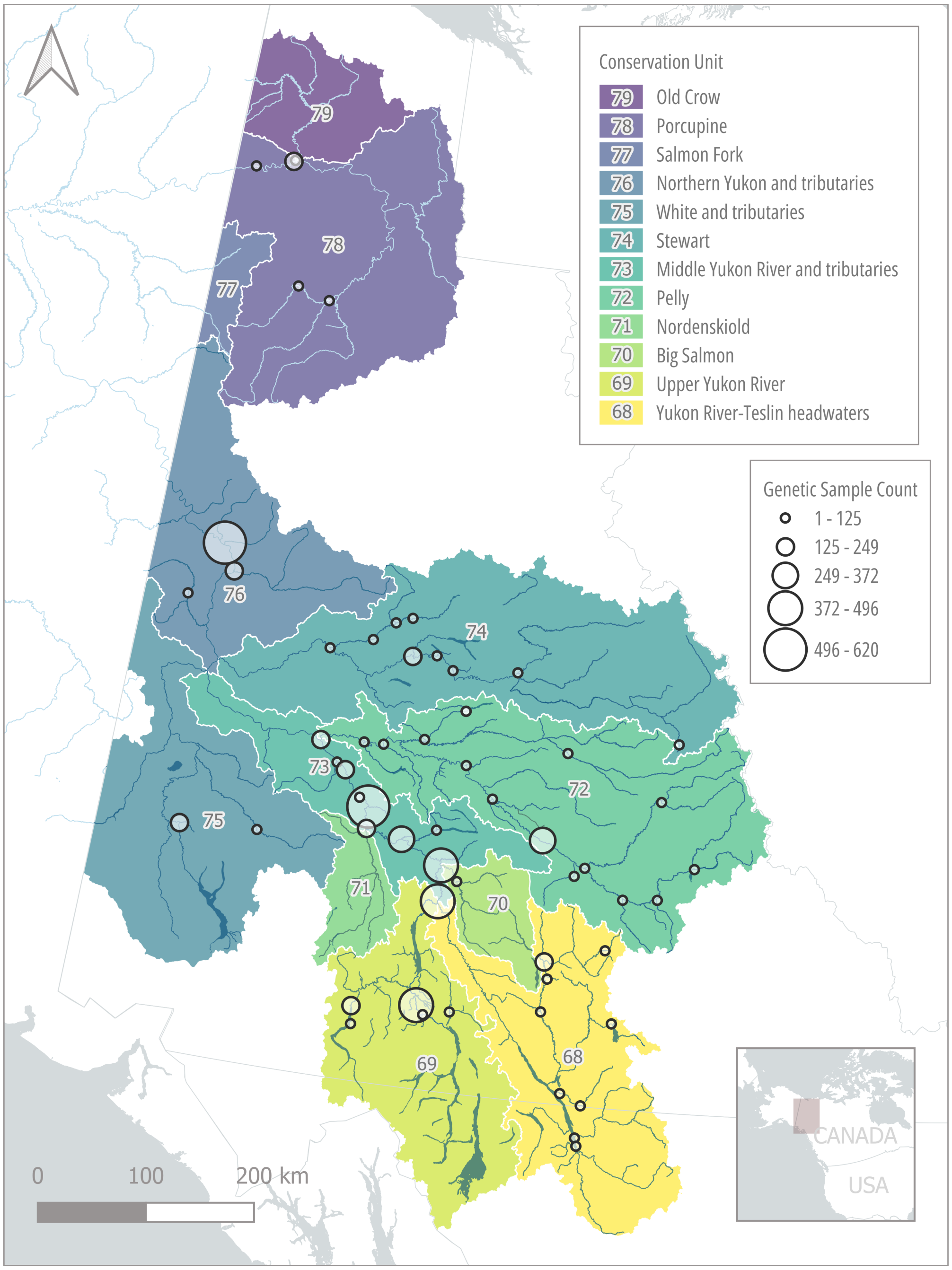

Figure 2. Locations of Chinook salmon genetic baseline samples in the Canadian portion of the Yukon River watershed. Sites are scaled to the number of samples collected to date.

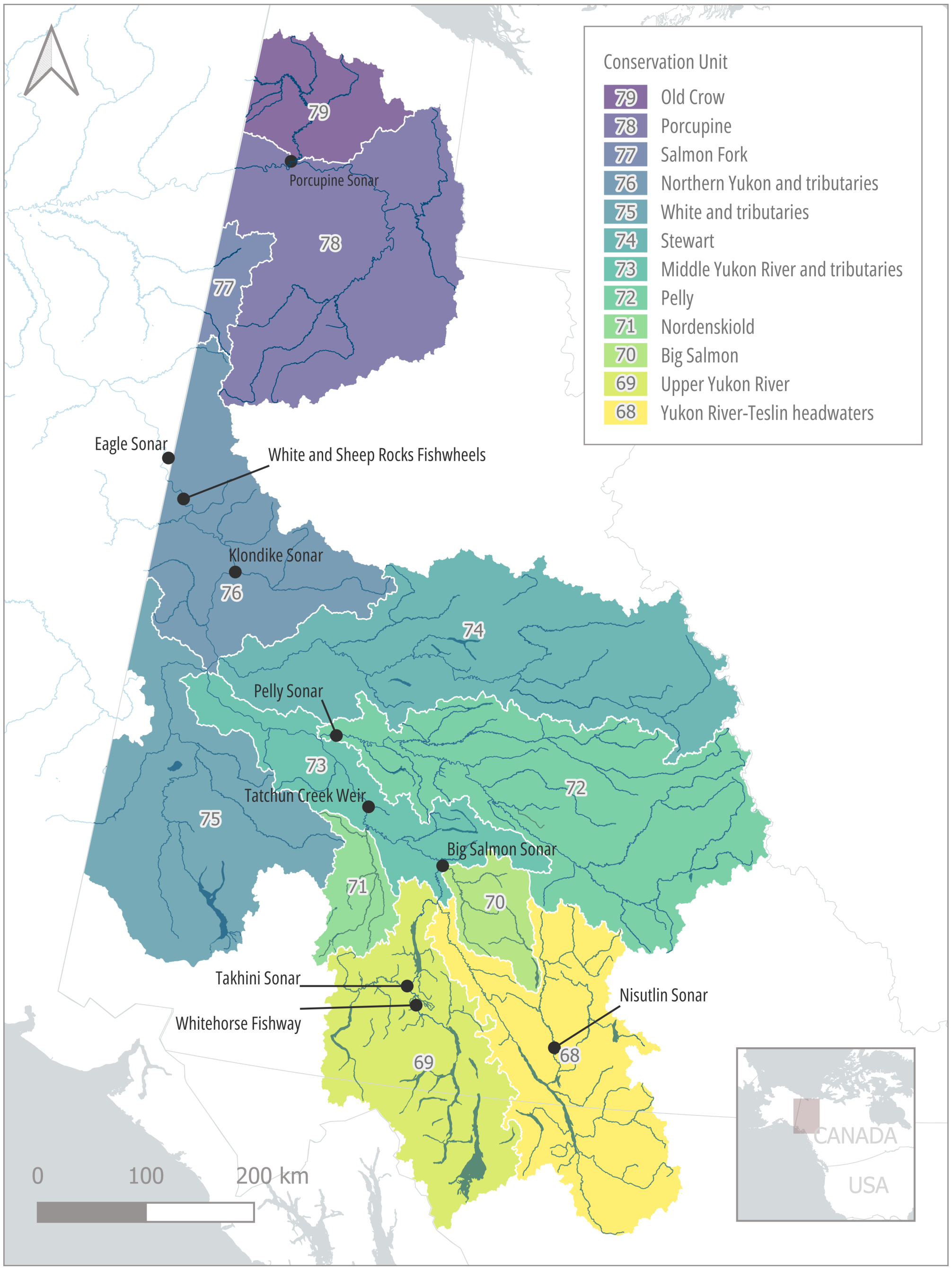

Figure 3. Major assessment projects for adult Chinook salmon in the Canadian portion of the Yukon River watershed. A full list of assessment projects is provided in Table 2.

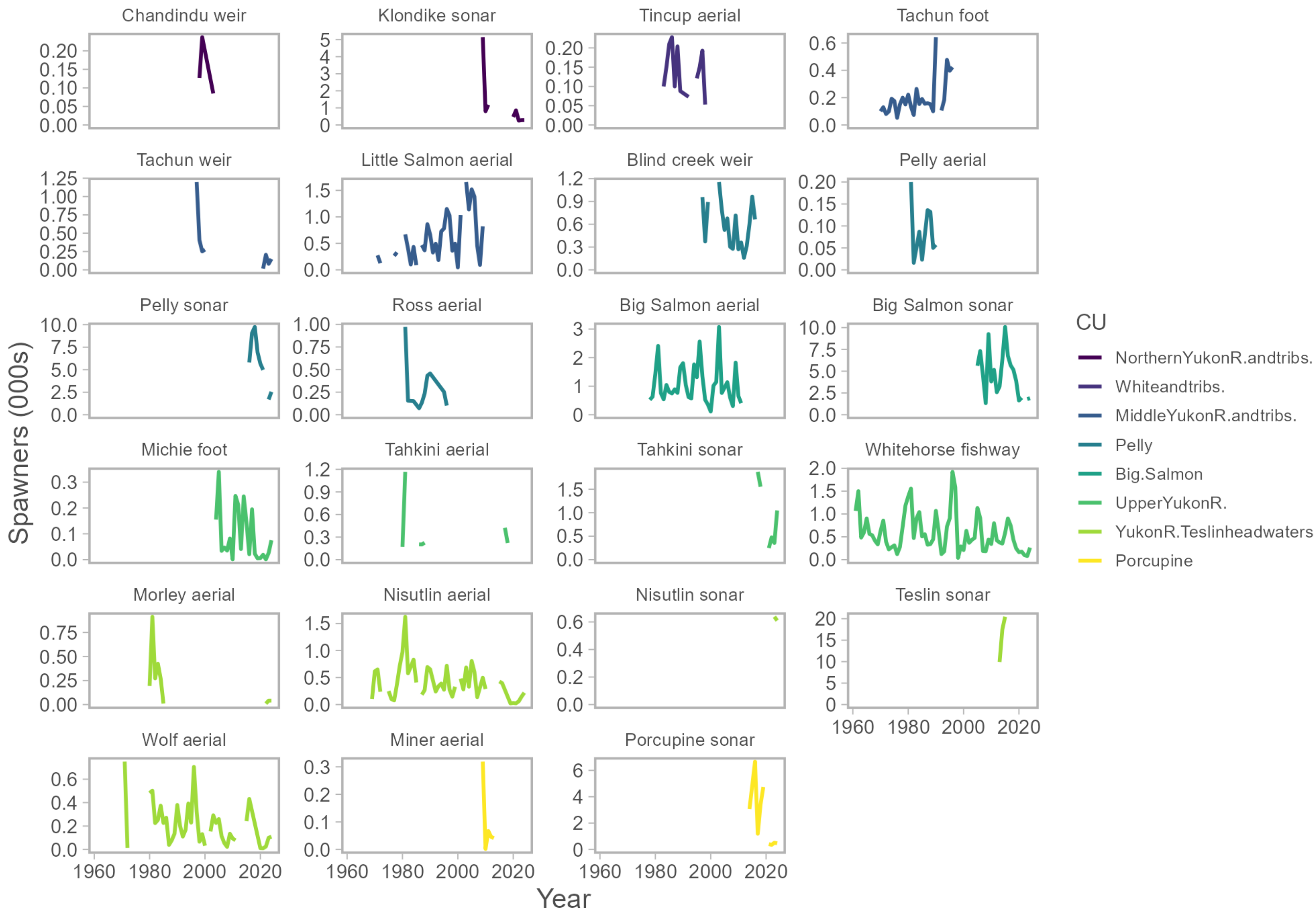

Figure 4. Estimates of spawner abundances over time by tributary assessment project. For more details on each assessment project see Table 2.

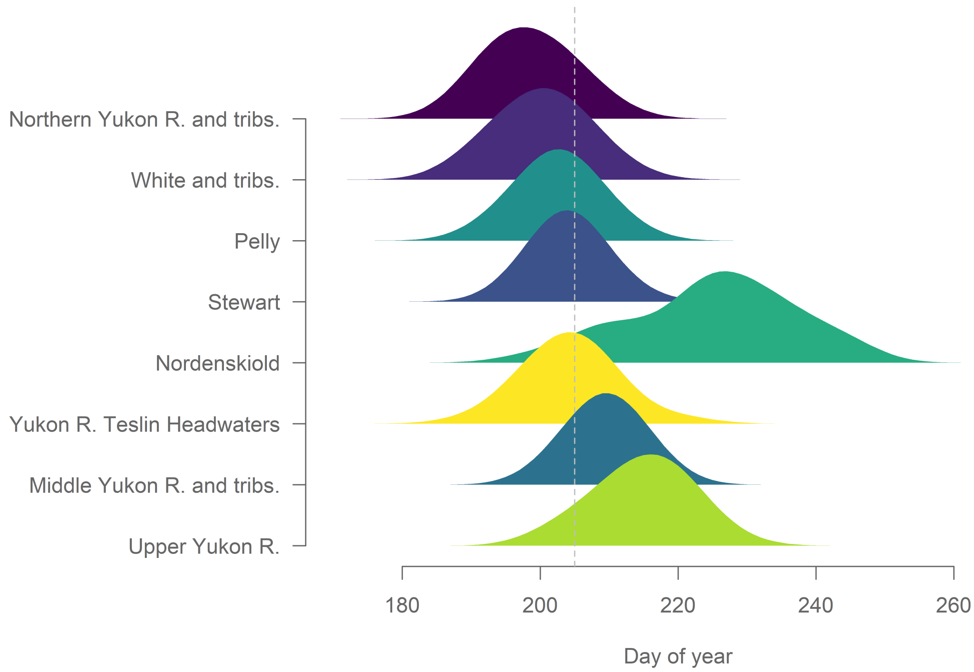

Figure 5. Average annual daily passage (scaled to maximum) from the U.S. into Canada on the mainstem Yukon River for eight Conservation Units. The Big Salmon Conservation Unit is not shown because it cannot be seperated out from the Middle Mainstem Conservation Unit based on genetics. For reference, in a non-leap year the 180th, 210th and 240th day of the year correspond to 29 June, 29 July, and 28 August, respectively. The dashed line represents the approximate average peak run-timing for the stock aggregate.

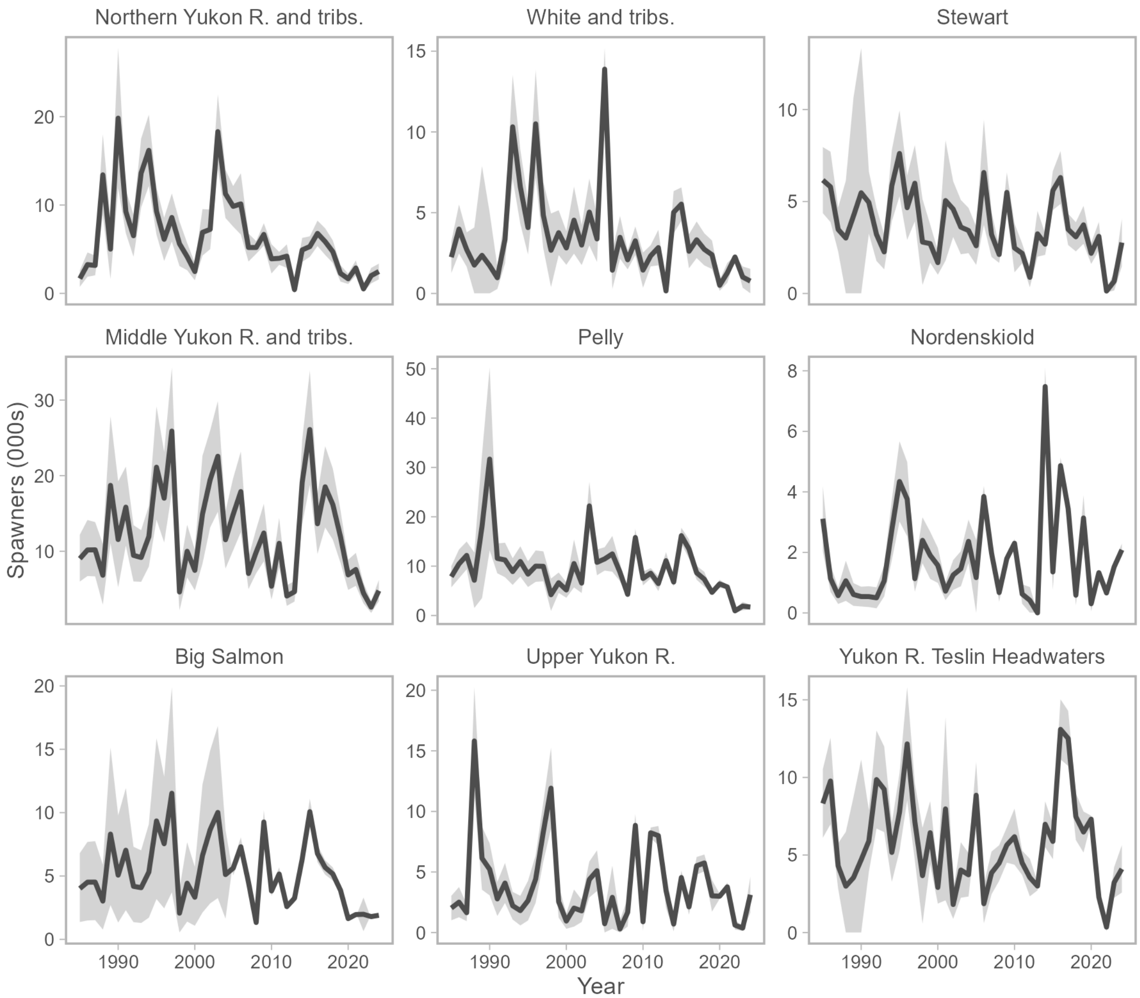

Figure 6. Reconstructed spawner abundance over time by Conservation Unit (median and 95% confidence intervals).

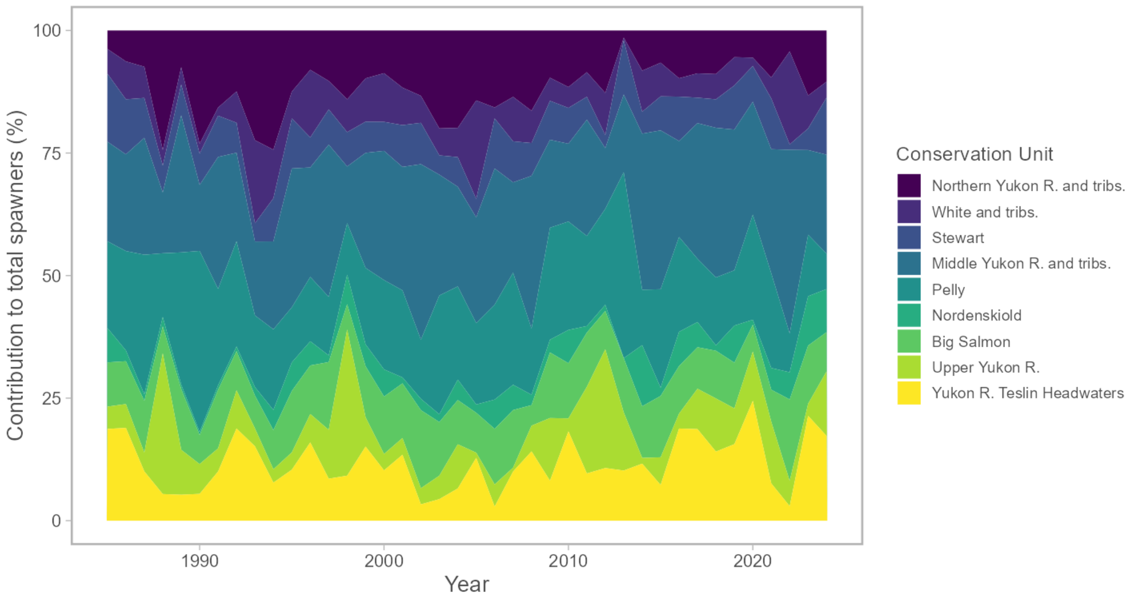

Figure 7. Percentage contribution of each Conservation Unit (CU) to reconstructed total return abundance of the Yukon Chinook Stock Management Unit over time. Estimates are based on the multi-CU run-reconstruction model that explicitly accounts for missing years and the sometimes unrepresentative genetic sampling of total returns.

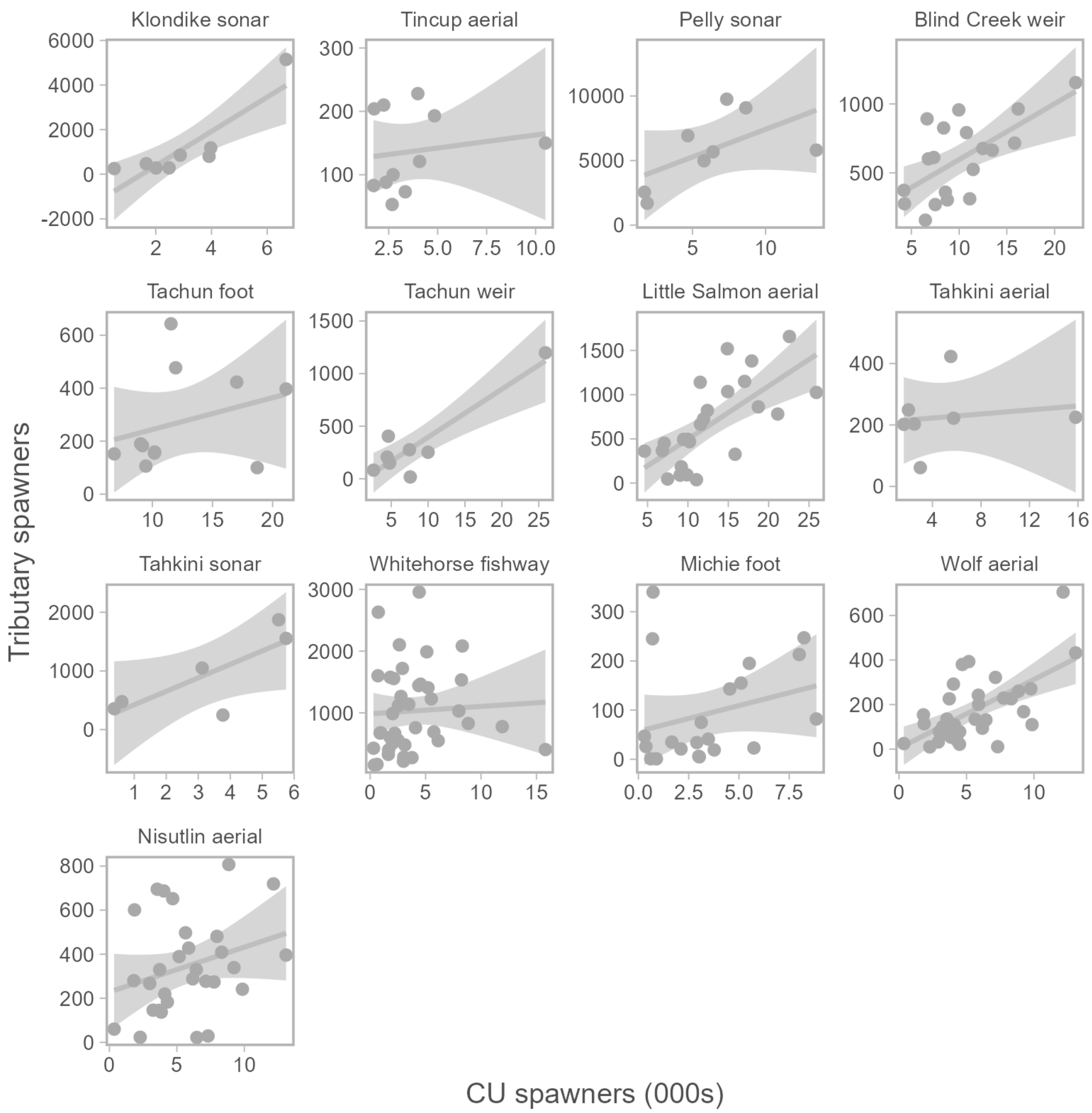

Figure 8. Relationship between estimates of spawner abundance from select tributary scale assessment projects and run-reconstruction based estimates at the Conservation Unit scale. Linear model fit shown in grey. For more details on each assessment project and corresponding Conservation Units see Table 2.

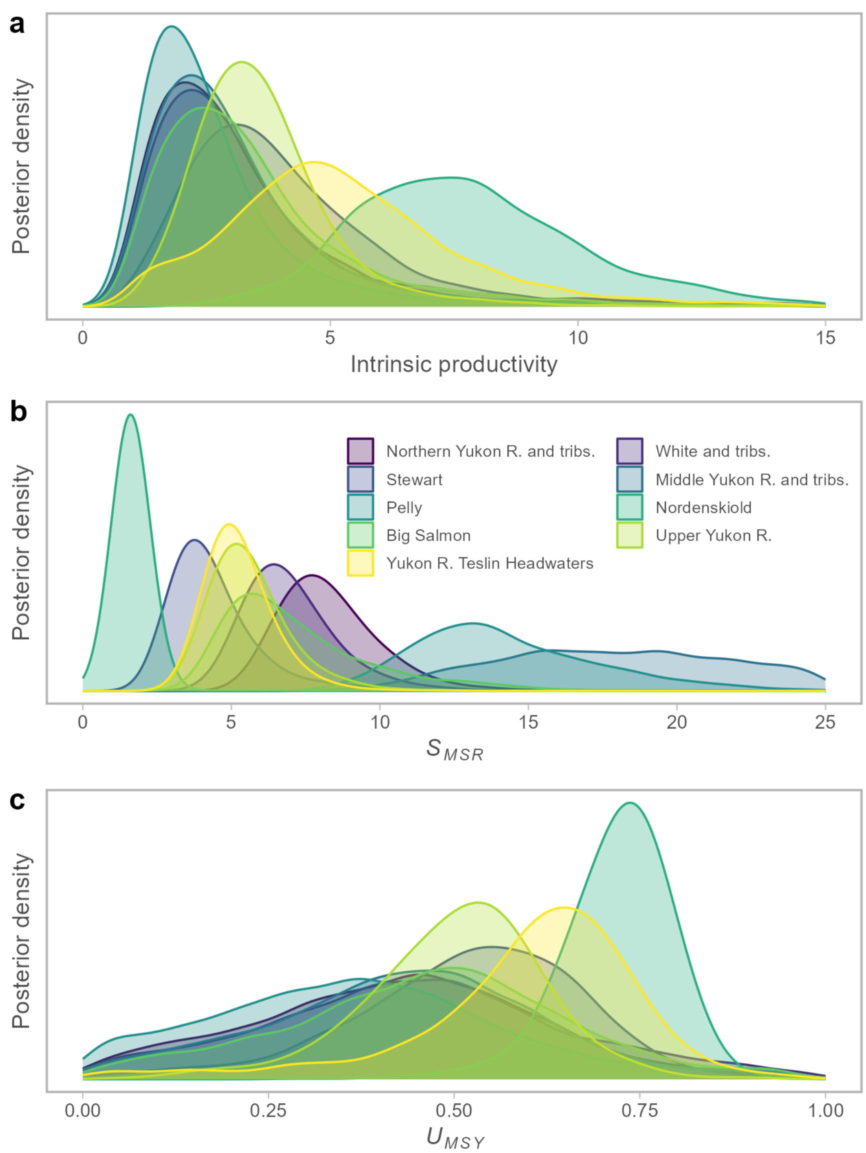

Figure 9. Conservation Unit specific posterior distributions of estimates of intrinsic productivity (a;α in units of recruits-per-spawner), spawner abundance associated with maximum recruitment (b; S_{MSR}), and exploitation rate associated with maximum sustainable yield (c; U_{MSY}) .

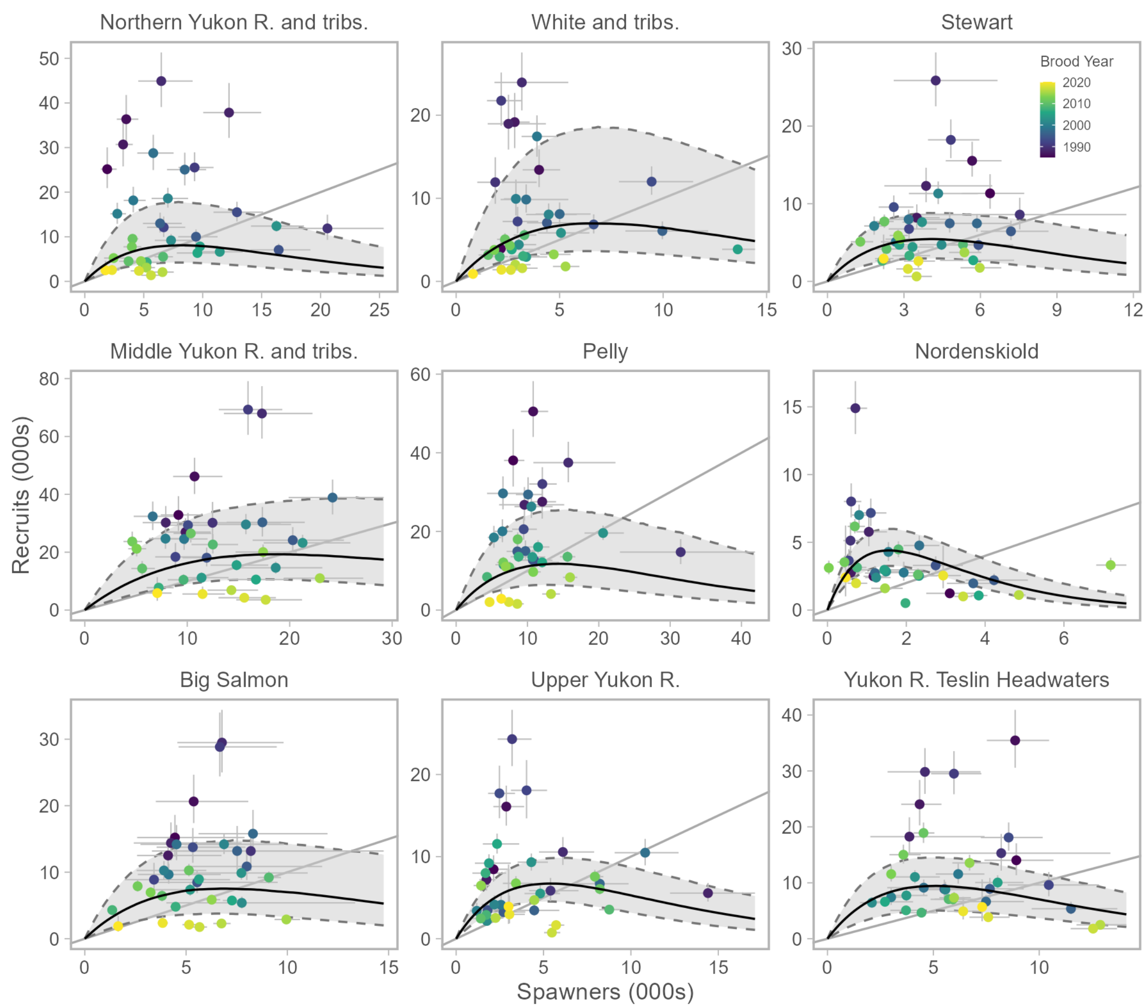

Figure 10. Relationship between spawner abundance and recruitment by Conservation Units. Individual spawner-recruitment pairs are color coded according to brood year with lines representing 80% credible intervals. The thick black line is the predicted median relationship between spawner abundance and recruitment along with 80% credible intervals in the shaded region. The replacement line, where spawners equal recruits, is shown in grey. To aid in visualizing uncertainty we use 80% credible intervals here instead of 95%.

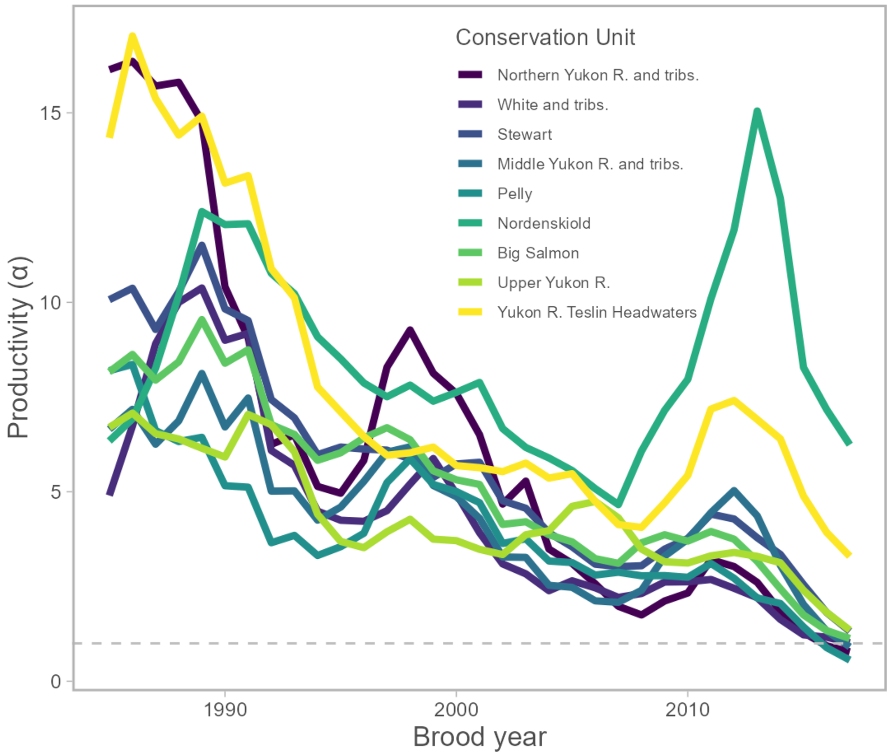

Figure 11. Estimated average intrinsic productivity (Ricker α parameter; maximum recruits per spawner) over time by Conservation Unit in the Yukon Chinook Stock Management Unit. The dashed grey line is replacement (recruits equal spawners).

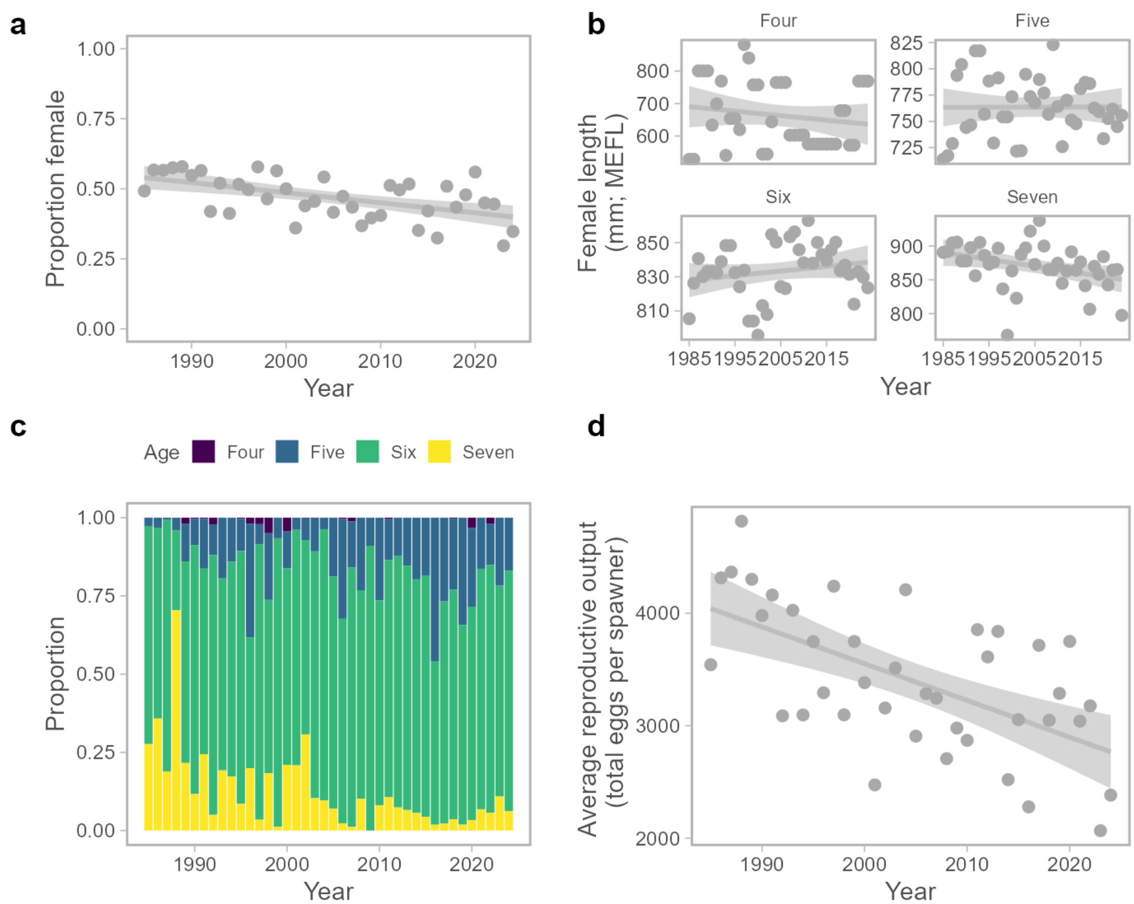

Figure 12. Age, sex, length, and reproductive output trends in Yukon River Chinook salmon over time. a) sex ratio, represented as proportion of spawners that are female, b) body length of female spawners by age (mid eye to tail fork length), c) proportion of returning spawners in each age class over time, d) average reproductive output of female spawners (eggs per female spawner). Data taken from fish sampled at Eagle and White and Sheep rocks.

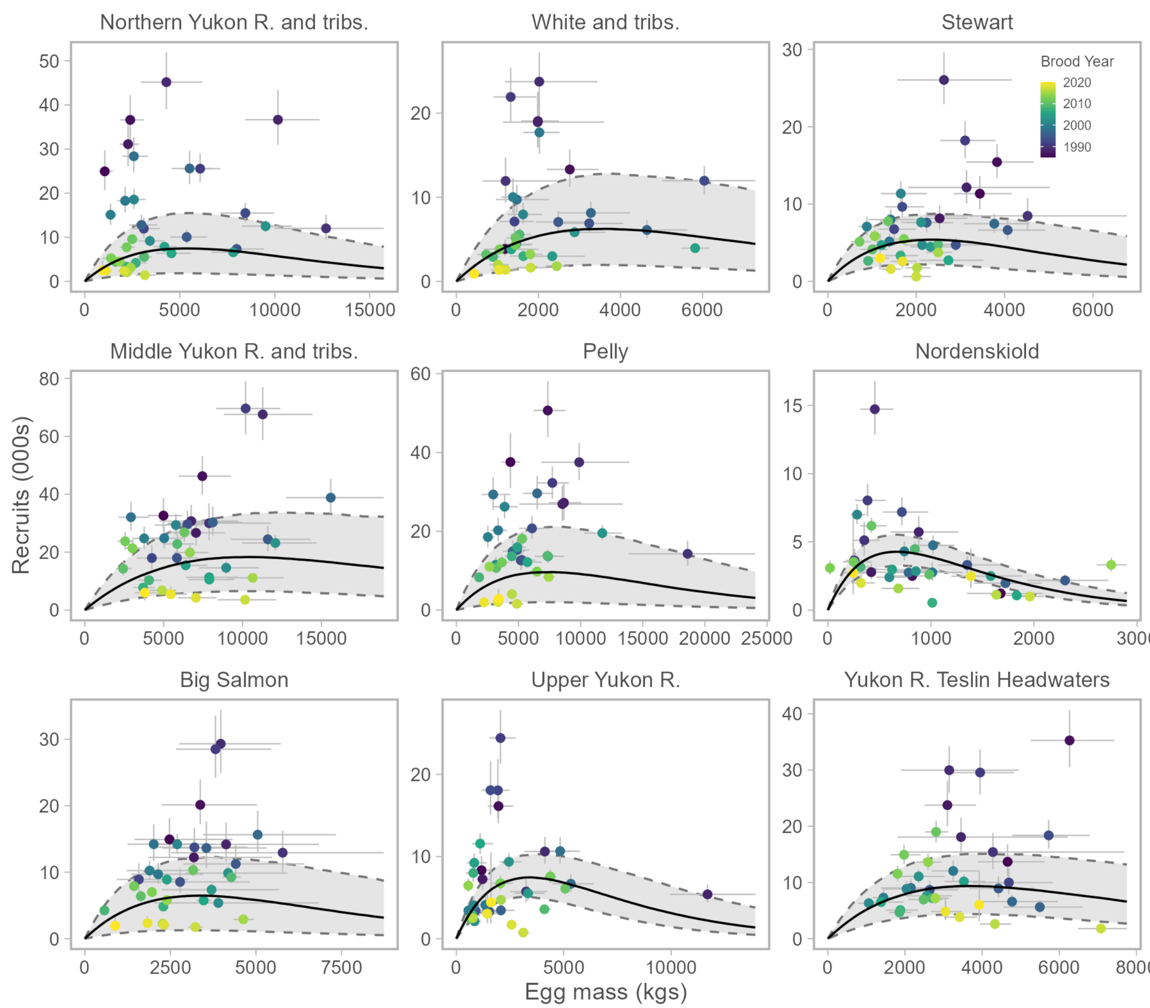

Figure 13. Relationship between total egg mass of spawning females and recruitment by Conservation Units. Individual egg mass-recruitment pairs are color coded according to brood year with 80% credible intervals. The thick black line is the predicted median relationship between egg mass and recruitment along with 80% credible intervals in the shaded region. To aid in visualizing uncertainty we use 80% credible intervals here instead of 95%.

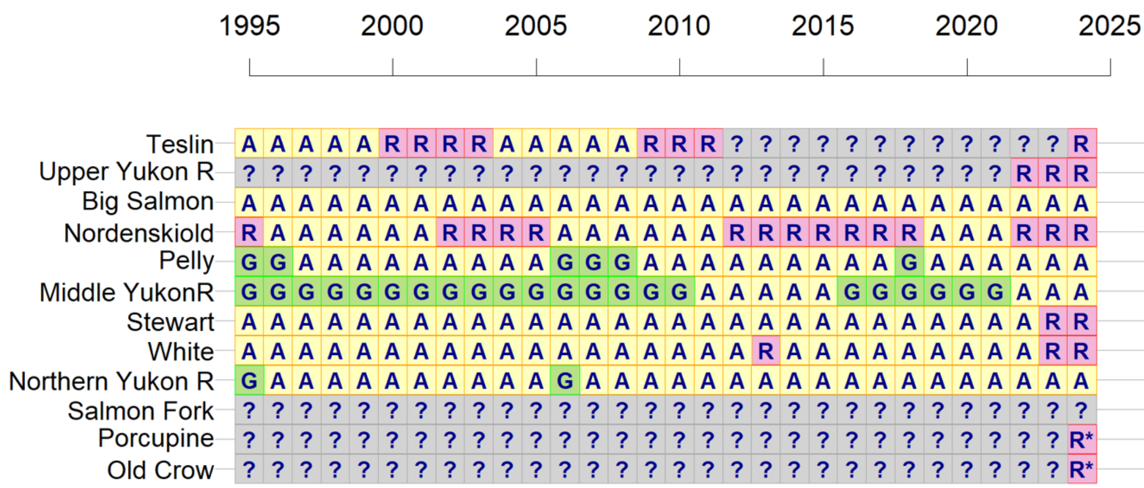

*Figure 14. Yukon River Chinook salmon Wild Salmon Policy statuses for years with applicable data. Each row summarizes the statuses available for each Conservation Unit in the Yukon River and Porcupine Stock Management Units.The * for the Porcupine and Old Crow statuses indicates they are based on a combined assessment.*

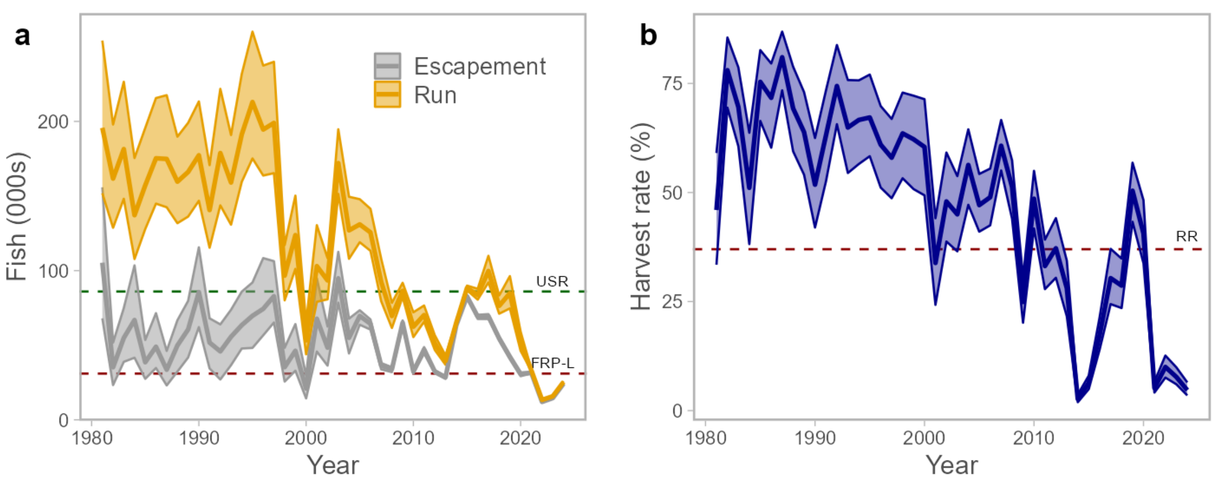

Figure 15. Reconstructed (a) total run size (orange) and spawning escapement (grey), with proposed Upper Stock Reference point (USR; green dashed line) and Lower Fisheries Reference Point (FRP-L; red dashed line), and (b) harvest rates for the Yukon Chinook Stock Management Unit (blue) and Removal Reference point (RR; red dashed line). Thick lines are medians and shaded areas indicate 95% credible intervals. Note that harvest rate in some years (e.g., 2017 and 2019) may be overestimated due to spawning migration mortality that was unaccounted for in the model that was used to estimate them (Connors et al. 2022a).

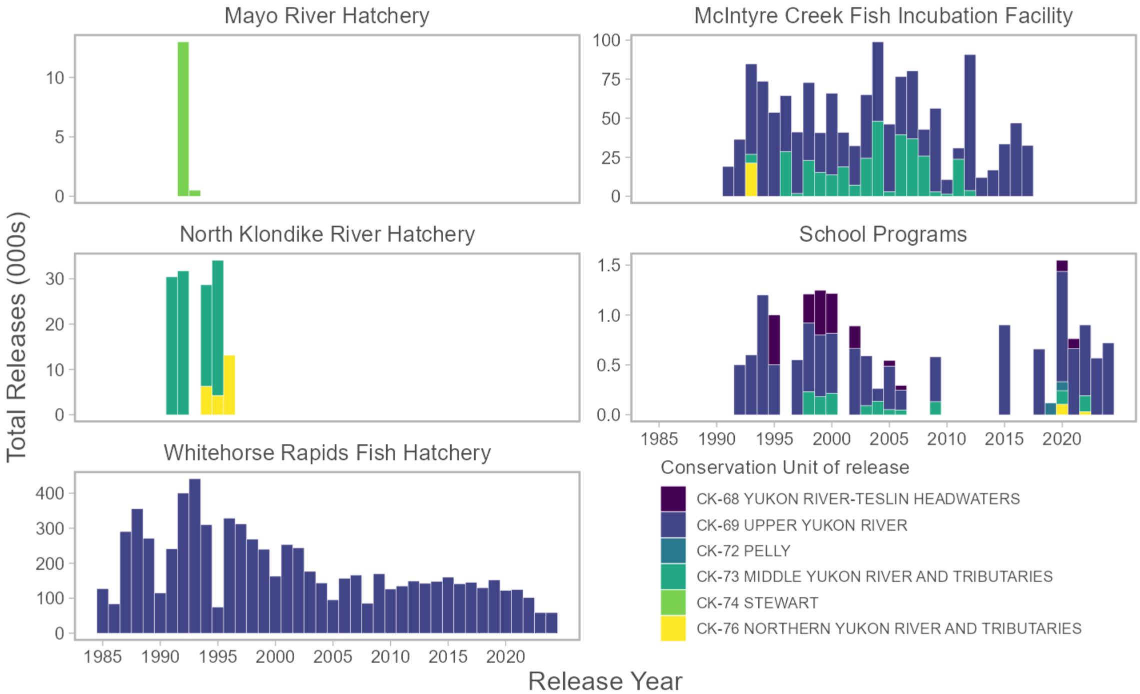

Figure 16. Total Yukon River Chinook salmon releases by facility/type and Conservation Unit. Note: Location of brood collection and release location, by CU, are the same in all cases except for ~250,000 fry from the Upper Yukon that were released in the Tatchun River (Middle Yukon CU) between 2006–2011.

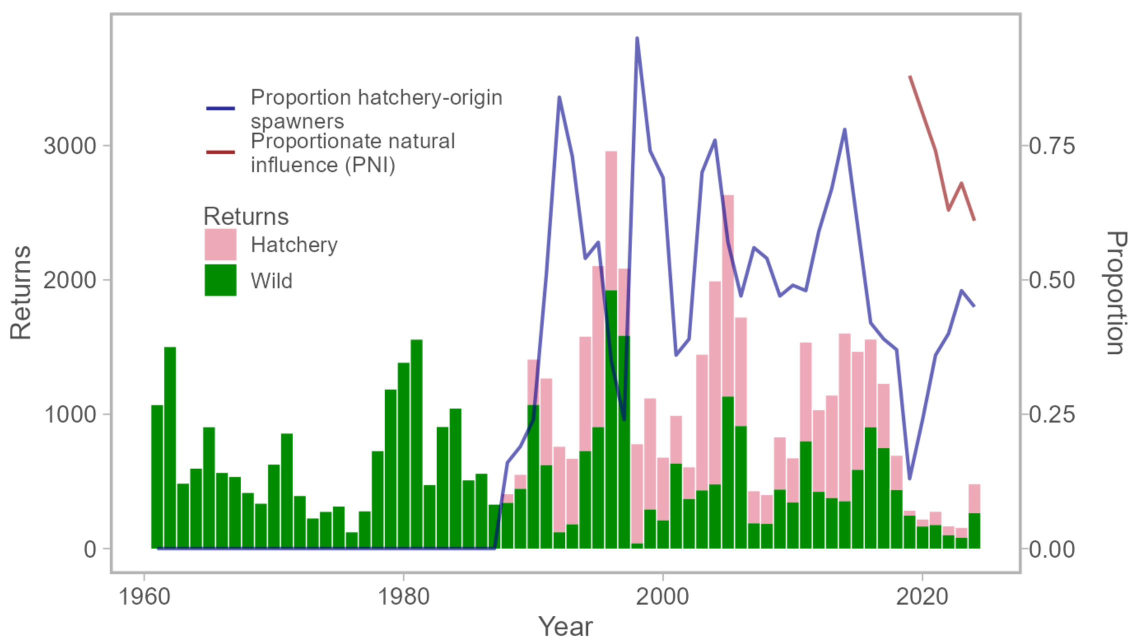

Figure 17. Chinook returns to Whitehorse Rapids fishway (Upper Yukon CU) and proportion hatchery origin spawners (pHOS), based on observed adipose fin clip rates. Proportionate Natural Influence (PNI) for the last six years, calculated from adipose fin clip status and Parentage Based Tagging (PBT) results of broodstock, when available.

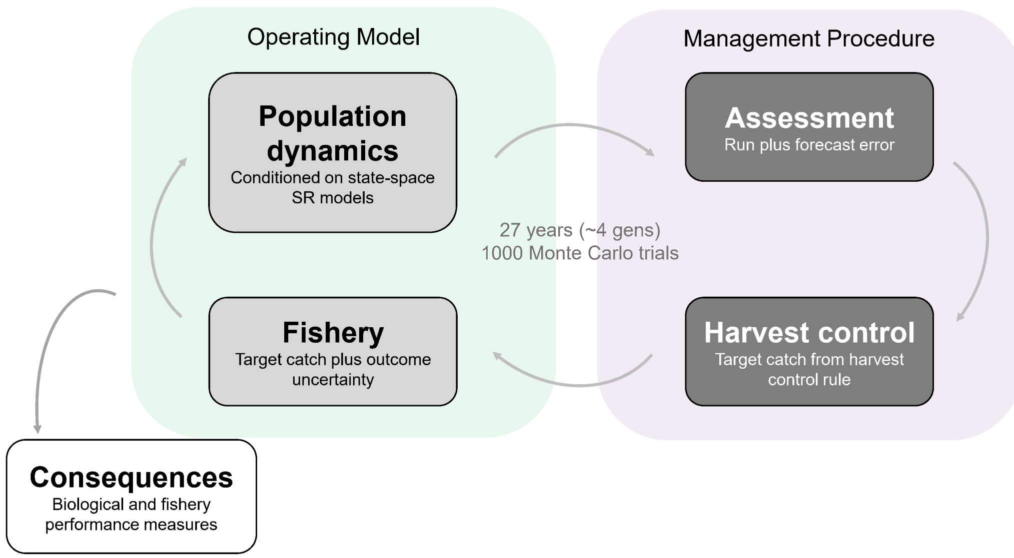

Figure 18. Illustration of steps in the closed loop simulations that were used to quantify future biological and fishery consequences of current and alternative harvest management measures. Run size forecast in forward simulation refers to run size at the U.S.–Canada border.

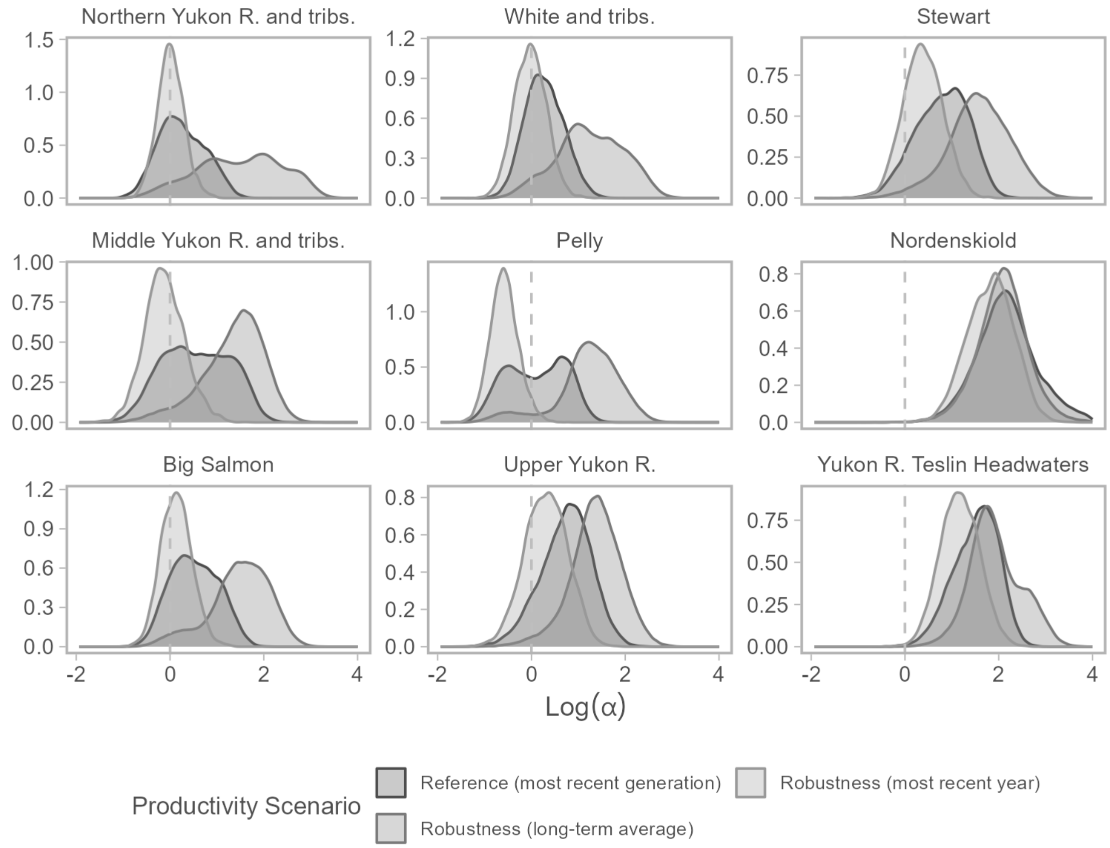

Figure 19. Alternative operating model productivity scenarios considered in the closed-loop simulations including the reference (base case) scenario, the average α of the most recent fully observed generation (2013–2018 brood years), and robustness scenario, the most recent α (2017 brood year).

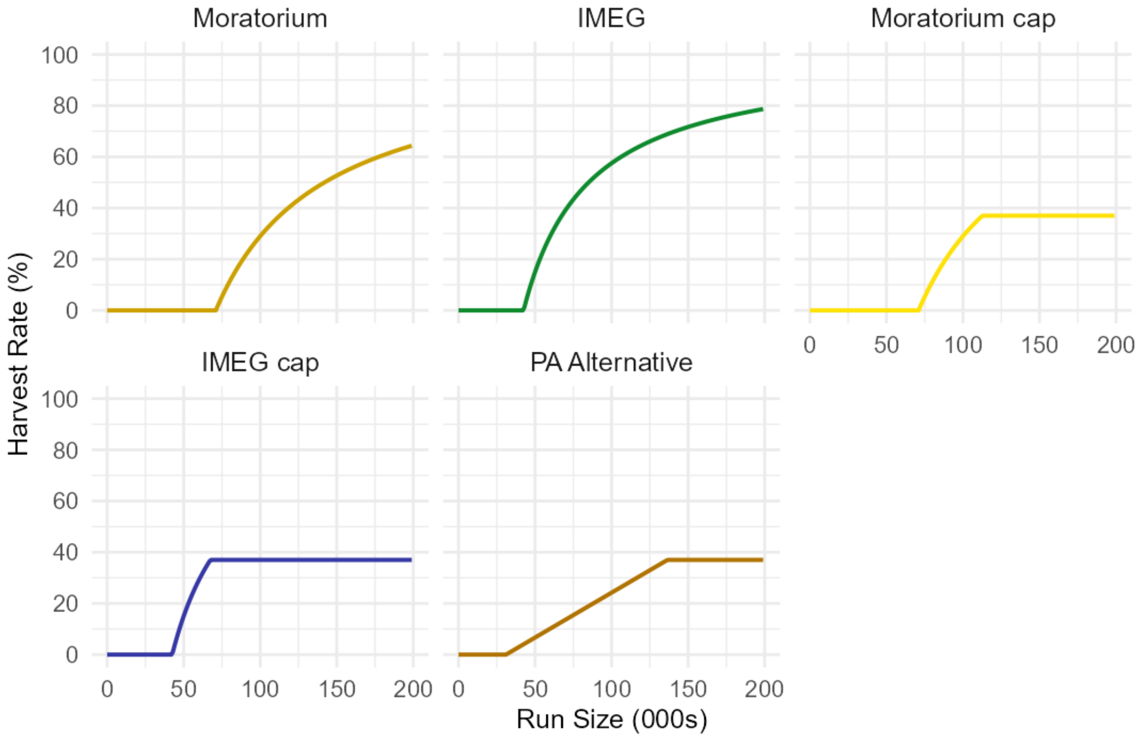

Figure 20. Illustration of the suite of harvest control rules considered in the closed-loop simulations. See Table 9 for description of each harvest control rule. The x-axis range in the plot corresponds approximately to the historical range of Yukon River Chinook salmon run sizes.

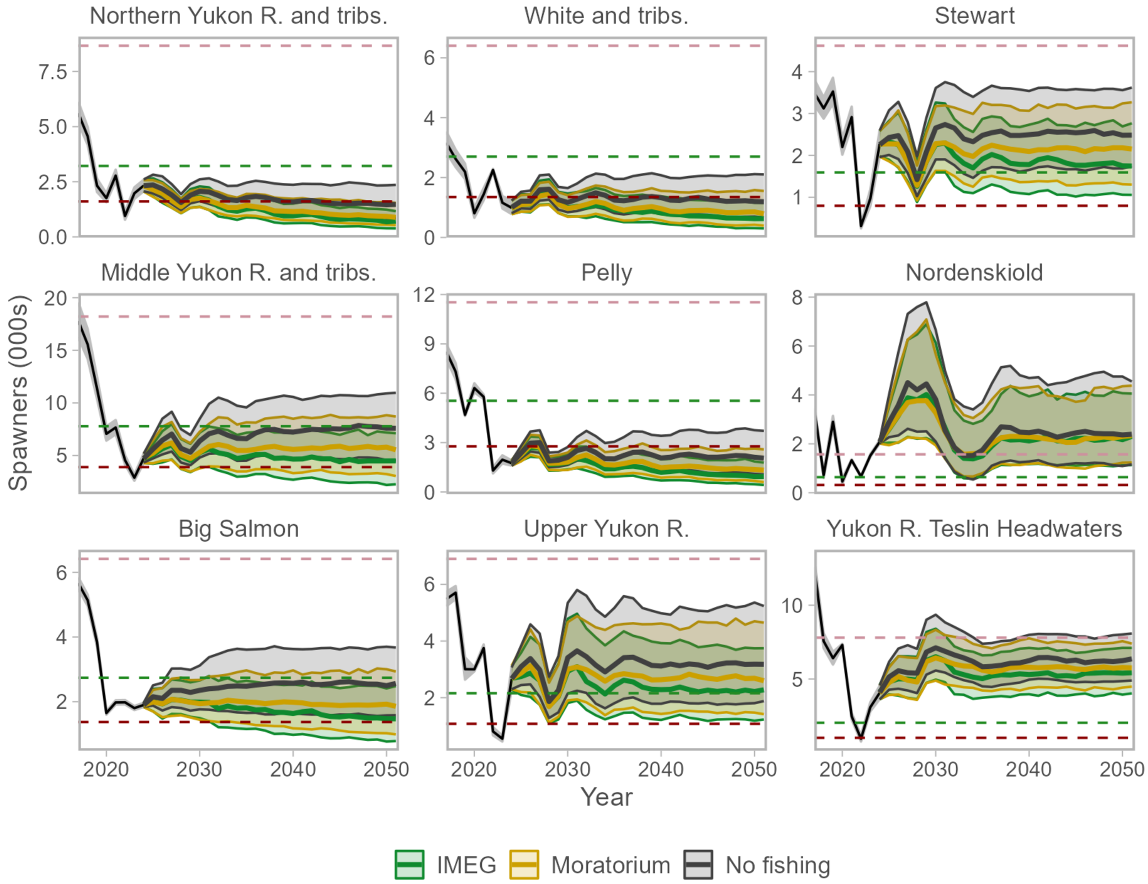

Figure 21. Projected spawner abundance over time (median and 50th percentiles) by Conservation Unit (CU) under three alternative fishery management measures, and the reference (base case) operating model productivity scenario. The last 6 years of observed spawners are shown prior to projections. Red and green dashed lines correspond to proposed lower and upper biological benchmarks for each CU (20%S_{MSR} and 40%S_{MSR}, respectively) from spawner-recruitment models with stationary productivity. Salmon colored dashed lines are CU rebuilding targets based on analyses that explicitly took demographic characteristics into account (S_{MSR} based on demographic characteristics from 2019–2024).

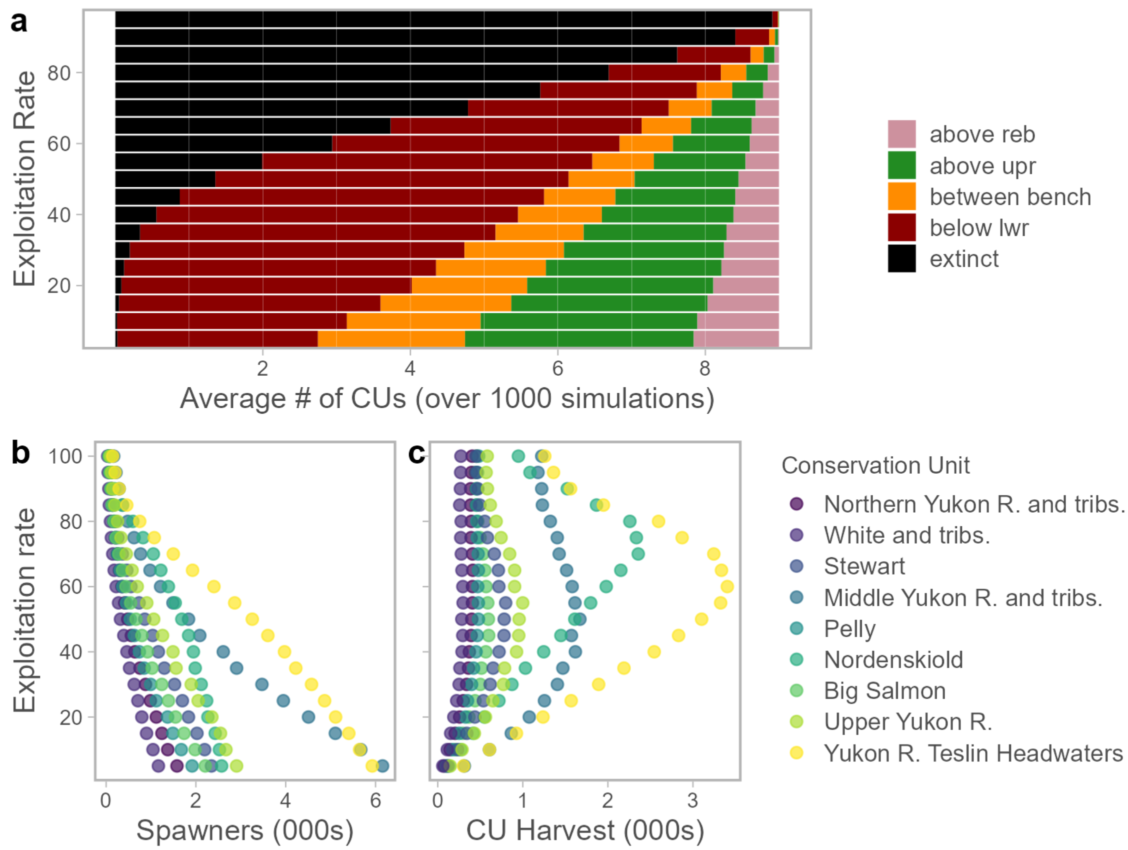

Figure 22. Projected consequences of a range of fixed exploitation rates from the closed loop simulations under the 'reference' (base case) operating model scenario. (a) Mean number of Conservation Units that fall above, below, and between their lower and upper biological benchmarks (20% S_{MSR} and 40%S_{MSR} based on analyses that assume stationary productivity) and illustrative rebuilding target (S_{MSR} based on demographic characteristics from 2019–2024) in the final simulated year (2050), over 1000 replicate simulations. (b) Mean spawner abundance and (c) mean harvest, by Conservation Units, over all simulation years (2025–2050).

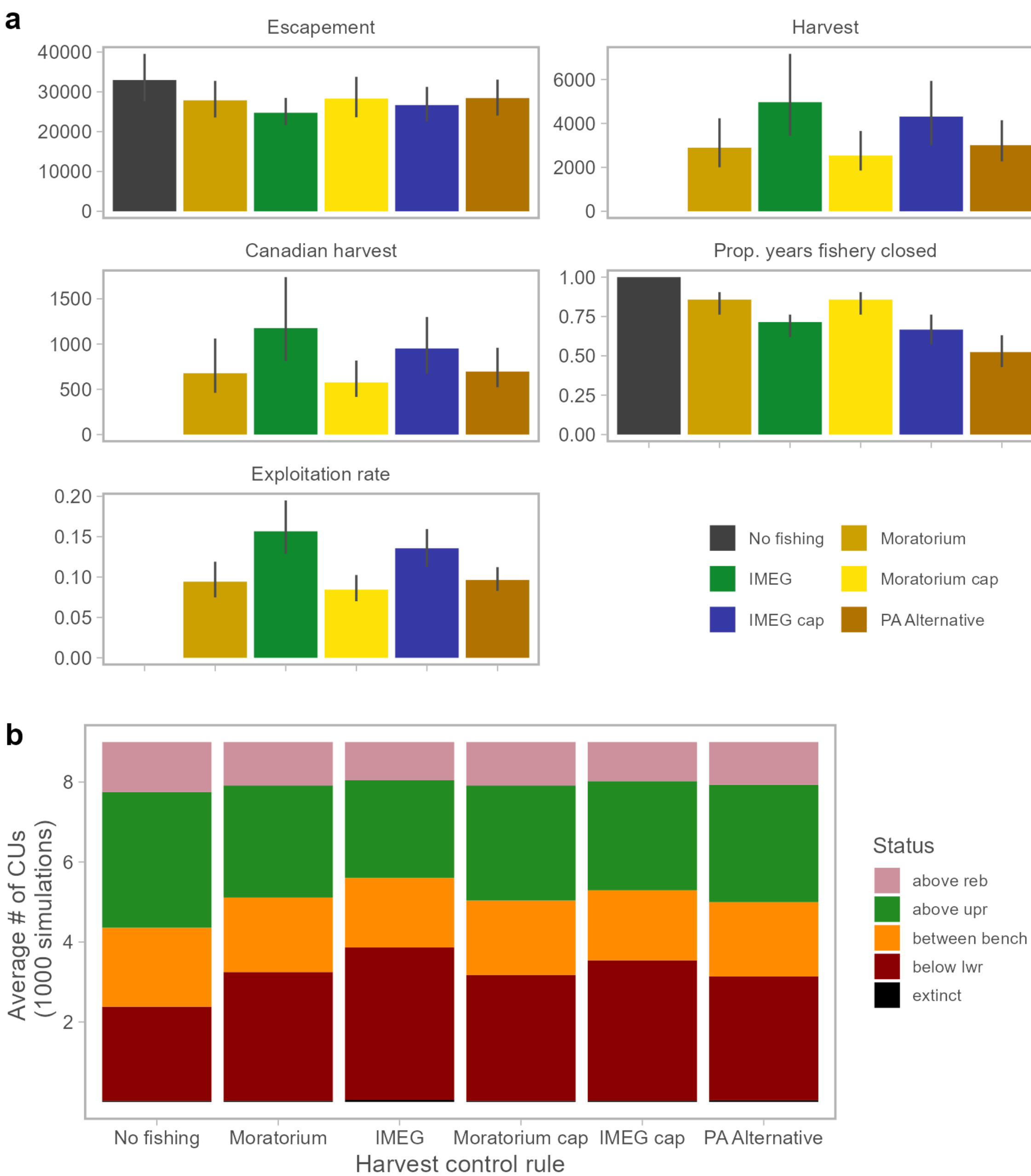

Figure 23. Performance of alternative harvest control rules the reference (base case) operating model productivity scenario. (a) Median (coloured bars) and 50% quantiles (vertical lines) of performance measures calculated over the years 2030–2050 and 1000 replicate simulations. (b) Mean number of Conservation Units that fall above, below, and between their lower and upper biological benchmarks (20% S_{MSR} and 40%S_{MSR} based on analyses that assume stationary productivity) and illustrative rebuilding target (S_{MSR} based on demographic characteristics from 2019–2024) in the final simulated year (2050), over 1000 replicate simulations.

11. TABLES

Table 1. Stock Management and Conservation Units (CUs) that make up the Yukon Chinook major fish stock prescribed under the Fish Stock Provisions of Canada's Fisheries Act*. The average annual percentage contribution of each CU to the Yukon Chinook Stock Management Unit based on reconstructed border passage is also shown. Contributions of individual CUs to the Porcupine SMU are unknown at present.*

Stock Management Unit	Conservation Unit ID	Conservation Unit	Average Contribution (1984-2024)
Yukon Chinook Salmon	CK-68	Yukon River-Teslin headwaters	11.7%
	CK-69	Upper Yukon River	7.89%
	CK-70	Big Salmon	10%
	CK-71	Nordenskiold	3.78%
	CK-72	Pelly	18%
	CK-73	Middle Yukon River and tributaries	22.9%
	CK-74	Stewart	7.16%
	CK-75	White and tributaries	6.72%
	CK-76	North Yukon River and tributaries	11.8%
Porcupine Chinook Salmon	CK-77	Salmon Fork	N/A
	CK-78	Porcupine	N/A
	CK-79	Old Crow	N/A

Table 2. Adult Chinook salmon assessment projects in the Canadian portion of the Yukon River basin.

Project	Assessment method	Proponent	Conservation Unit	Status	Years of operation
Big Salmon*	Sonar	Metla Environmental Inc.	Big Salmon	Active	2005-2021, 2023-24
Big Salmon	Aerial	Fisheries and Oceans Canada and Alaska Department of Fish and Game	Big Salmon	Historical	1978-2011
Tatchun Creek*	Weir/video box	Little Salmon Carmacks First Nation	Middle Yukon River and Tribs	Active	2021-24
Tatchun Surveys	Foot/aerial/weir	Fisheries and Oceans Canada, Alaska Department of Fish and Game, Independent consultants, Little Salmon Carmack First Nation	Middle Yukon River and Tribs	Historical	1970-90, 92-2000
Little Salmon	Aerial	Fisheries and Oceans Canada and Alaska Department of Fish and Game	Middle Yukon River and Tribs	Historical	1969-72, 1977-78, 1981-82, 1984-85, 1987-2011
Klondike*	Sonar	Tr′ondëk Hwëch′in	Northern Yukon	Active	2009-11, 2020-24
Chandindu	Weir	Fisheries and Oceans Canada	Northern Yukon	Historical	1998-99, 2003
Pelly*	Sonar	Selkirk First Nation	Pelly	Active	2016-21, 2023-24
Ross River	Aerial	Fisheries and Oceans Canada and Alaska Department of Fish and Game	Pelly	Historical	1981-82, 1984, 1986-90, 1995-96
Porcupine*	Sonar	Fisheries and Oceans Canada and Vuntut Gwitchin First Nation	Porcupine	Active	2014-19, 2021-24

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Project	Assessment method	Proponent	Conservation Unit	Status	Years of operation
Miner	Aerial	Fisheries and Oceans Canada	Porcupine	Historical	2009-13
Takhini*	Sonar	Kwanlin Dün First Nation	Upper Yukon River	Active	2017-18, 2021-24
Whitehorse Fishway*	Fishladder/video	Yukon Energy Corporation	Upper Yukon River	Active	1961-2024
Michie	Foot	Kwanlin Dün First Nation	Upper Yukon River	Active	2004-24
Takhini	Aerial	Fisheries and Oceans Canada and Alaska Department of Fish and Game	Upper Yukon River	Historical	1980-81, 1986-88, 2001, 2017-18, 2020
Takhini	Aerial	Fisheries and Oceans Canada	Upper Yukon River	Historical	1980-83, 1986-89, 2001-02, 2017-20
Tincup	Aerial	Fisheries and Oceans Canada and Alaska Department of Fish and Game	White and tribs	Historical	1983-1990, 1992, 1995-98
Eagle*	Sonar	Alaska Department of Fish and Game and Fisheries and Oceans Canada	Yukon Chinook SMU	Active	2005-24
Border mark/recapture	Fishwheel	Fisheries and Oceans Canada	Yukon Chinook SMU	Historical	1982-2004
Radio mark/recapture Canada	Fishwheel	Alaska Department of Fish and Game and Fisheries and Oceans Canada	Yukon Chinook SMU	Historical	2003-05

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Project	Assessment method	Proponent	Conservation Unit	Status	Years of operation
Nisutlin*	Sonar	Teslin Tlingit Council	Yukon River-Teslin headwaters	Active	2023-24
Nisutlin	Aerial	Teslin Tlingit Council, Fisheries and Oceans Canada, Alaska Department of Fish and Game	Yukon River-Teslin headwaters	Active	1969-72, 1975-85, 1987-99, 2001-10, 2015-16, 2019-24
Wolf	Aerial	Teslin Tlingit Council, Fisheries and Oceans Canada, Alaska Department of Fish and Game	Yukon River-Teslin headwaters	Active	1971-72, 1980-00, 2002-11, 2015-16, 2020-24
Morley	Aerial	Teslin Tlingit Council, Fisheries and Oceans Canada, Alaska Department of Fish and Game	Yukon River-Teslin headwaters	Active	1980-85, 2022-24

Note:
*Project provides in-season information on returning spawners.

Table 3. Definitions, values, and rationale for proposed biological benchmarks and rebuilding targets at the Conservation Unit (CU) scale.

Metric	Definition	Value	Rationale
Lower biological benchmark	Spawner abundances below which CU is expected to have high risk of irreversible harm and which aligns with a Committee on the Status of Endangered Wildlife in Canada (COSEWIC) assessment of “endangered”	20% of S_{MSR}	More biologically based than MSY based biological benchmarks that are implicitly tied to yield based objectives (Frid et al. 2023). Less sensitive to temporal variation in intrinsic productivity than MSY based benchmarks (Holt and Michielsens 2020). Aligned with proposed use of S_{MSR} based reference points at SMU scale
Upper biological benchmark	Spawner abundances associated with low risk of irreversible harm, delineatinating amber and green Wild Salmon Policy biological status zones	40% of S_{MSR}	Same as above
Rebuilding target	Spawner abundances associated with a “healthy” state that meets socio-ecological objectives and is expected to be resilient in the face of rapid and unpredictable environmental change	S_{MSR} based on demographic characteristics of spawners in most recent generation (2019-2024)	Targets linked to stock size greater than those associated with MSY, and that take demographics into account, have been argued to be more consistent with indigenous and ecosystem based principles and to afford greater resilience to environmental and ecosystem change (Reid et al. 2022; Frid et al 2023)

Table 4. *Definitions, values, and rationale for proposed reference points for the Yukon Stock Management Unit (SMU).*

Metric	Definition	Value	Rationale
Limit Reference Point (LRP)	Point below which serious harm may occur to the stock and where there may also be impacts to the ecosystem, associated species, and/or a long-term loss of fishing opportunities	One or more component CUs for the MSU in Wild Salmon Policy Red status zone	Required under Fish Stock Provision of *Fisheries Act*. Reflects requirement for salmon LRPs to be aligned with Wild Salmon Policy objective of preserving biodiversity of salmon at the scale of CUs (Holt et al. 2023)
Upper Stock Reference Point (USR)	Stock size with a low probability of the stock dropping below it's LRP, and a target for meeting biological and/or socio-economic objectives	86,000 spawners; median estimate of S_{MSR} for the SMU based on demographic characteristics of spawners (and impacts on total egg mass) in most recent decade (2009-2019) at time of analysis (Connors et al. 2022)	Required under Fish Stock Provision of *Fisheries Act*. Reference points linked to stock size greater than those associated with MSY, and that take demographics into account, have been argued to be more aligned with indigenous and ecosystem based principles (Reid et al. 2022; Frid et al 2023). Consistent with use of S_{MSR} as border passage target under current Yukon River 7-year agreement

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Metric	Definition	Value	Rationale
Maximum Removal Reference (RR)	The maximum acceptable harvest rate the SMU can be subject to	36%; Harvest rate associated with MSY (U_{MSY}) for the least productive CU in the SMU	Required under Fish Stock Provision of *Fisheries Act*. Federal and international fishery policy typically set maximum fishery removals to be less than or equal to those associated with U_{MSY}. Propose this value be from least productive CU to align with Wild Salmon Policy objective to take biodiversity at the scale of CUs into account
Fishery Reference Point - Lower (FRP-L)	Aggregate SMU spawner abundance below which fishery removals should be limited to the maximum extent possible	31,000 spawners; average SMU spawner abundance associated with no greater than 50% chance of at least one CU falling in the Wild Salmon Policy Red status zone	Not required under Fish Stock Provision of *Fisheries Act* but helpful for informing management since assessment and fishery decisions are often abundance based and so can be operationalized (compared to the CU-status based LRP)

Table 5. *Estimates of biological benchmarks by Conservation Unit, based on parameters from spawner-recruitment models assuming stationary intrinsic productivity. Values are in thousands of fish (except U_{MSY}).*

Conservation Unit	Reference Point	Median (80th percentile)
Big Salmon	S_{GEN}	1.11 (0.56,2)
	S_{MSR}	6.8 (4.83,11.52)
	20%S_{MSR}	1.36 (0.97,2.3)
	40%S_{MSR}	2.72 (1.93,4.61)
	S_{MSY}	3.15 (1.42,5.19)
	U_{MSY}	0.46 (0.19,0.69)
Middle Yukon R. and tribs.	S_{GEN}	3.29 (1.66,5.88)
	S_{MSR}	19.96 (13.5,34.88)
	20%S_{MSR}	3.99 (2.7,6.98)
	40%S_{MSR}	7.98 (5.4,13.95)
	S_{MSY}	8.28 (3.9,14.97)
	U_{MSY}	0.43 (0.17,0.66)
Nordenskiold	S_{GEN}	0.17 (0.11,0.26)
	S_{MSR}	1.59 (1.31,1.96)
	20%S_{MSR}	0.32 (0.26,0.39)
	40%S_{MSR}	0.64 (0.53,0.78)
	S_{MSY}	1.14 (0.98,1.37)
	U_{MSY}	0.73 (0.63,0.8)
Northern Yukon R. and tribs.	S_{GEN}	1.35 (0.68,1.87)
	S_{MSR}	8.07 (6.49,10.5)
	20%S_{MSR}	1.61 (1.3,2.1)
	40%S_{MSR}	3.23 (2.6,4.2)
	S_{MSY}	3.48 (1.34,5.57)
	U_{MSY}	0.43 (0.16,0.68)
Pelly	S_{GEN}	2.31 (1.06,3.34)
	S_{MSR}	14.04 (10.74,19.26)
	20%S_{MSR}	2.81 (2.15,3.85)
	40%S_{MSR}	5.62 (4.3,7.71)
	S_{MSY}	5.15 (1.75,9.01)
	U_{MSY}	0.36 (0.12,0.63)
Stewart	S_{GEN}	0.65 (0.36,1.11)
	S_{MSR}	3.97 (2.93,6.24)
	20%S_{MSR}	0.79 (0.59,1.25)
	40%S_{MSR}	1.59 (1.17,2.5)
	S_{MSY}	2.1 (1.31,3.03)
	U_{MSY}	0.53 (0.3,0.69)

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Conservation Unit	Reference Point	Median (80th percentile)
Upper Yukon R.	S_{GEN}	0.94 (0.61,1.36)
	S_{MSR}	5.39 (4.2,7.34)
	20%S_{MSR}	1.08 (0.84,1.47)
	40%S_{MSR}	2.16 (1.68,2.94)
	S_{MSY}	2.72 (1.99,3.58)
	U_{MSY}	0.51 (0.35,0.64)
White and tribs.	S_{GEN}	1.11 (0.57,1.62)
	S_{MSR}	6.76 (5.27,9.32)
	20%S_{MSR}	1.35 (1.05,1.86)
	40%S_{MSR}	2.7 (2.11,3.73)
	S_{MSY}	3.03 (1.31,5.09)
	U_{MSY}	0.45 (0.19,0.71)
Yukon R. Teslin Headwaters	S_{GEN}	0.71 (0.4,1.11)
	S_{MSR}	5.04 (4.1,6.55)
	20%S_{MSR}	1.01 (0.82,1.31)
	40%S_{MSR}	2.02 (1.64,2.62)
	S_{MSY}	3.12 (2.16,3.97)
	U_{MSY}	0.63 (0.41,0.75)

Table 6. Estimates of key spawner-recruitment relationship parameters by Conservation Unit along with two measures of parameter estimability and model convergence. Estimates of β are on the scale of 10,000 fish for ease of summarizing parameters in the table.

Conservation Unit	Parameter	Median (80th percentile)	Effective Sample Size (N_{eff})	Chain Mixing ($\widehat{R}$)
Big Salmon	α	2.95 (1.5,6.49)	3387	1.0006
	β	1.5 (0.9,2.1)	1570	1.0012
	ϕ	0.85 (0.68,0.96)	4389	1.0003
	σ	0.94 (0.84,1.05)	4973	0.9998
Middle Yukon R. and tribs.	α	2.71 (1.44,5.73)	4343	0.9999
	β	0.5 (0.3,0.7)	5895	1.0004
	ϕ	0.81 (0.63,0.95)	5257	1.0008
	σ	1 (0.91,1.11)	8907	1.0005
Nordenskiold	α	7.64 (5.14,11.15)	7104	0.9998
	β	6.3 (5.1,7.6)	9282	0.9996
	ϕ	0.35 (0.08,0.61)	4290	0.9999
	σ	1.29 (1.19,1.41)	8671	1.0002
Northern Yukon R. and tribs.	α	2.68 (1.41,6.18)	4172	1.0005
	β	1.2 (1,1.5)	7982	1.0001
	ϕ	0.87 (0.75,0.96)	6184	1.0001
	σ	0.96 (0.87,1.07)	6767	0.9999
Pelly	α	2.25 (1.28,5.03)	2811	1.0012
	β	0.7 (0.5,0.9)	5772	1.0004
	ϕ	0.86 (0.72,0.97)	4339	1.0006
	σ	0.96 (0.87,1.06)	5228	1.0005
Stewart	α	3.67 (1.92,6.56)	5612	0.9999
	β	2.5 (1.6,3.4)	4867	1.0003
	ϕ	0.73 (0.52,0.9)	5147	1.0000
	σ	1.05 (0.96,1.16)	8010	1.0002
Upper Yukon R.	α	3.39 (2.17,5.18)	4234	1.0003
	β	1.9 (1.4,2.4)	6220	1.0002
	ϕ	0.53 (0.27,0.77)	3765	1.0001
	σ	1.17 (1.07,1.29)	6476	1.0002
White and tribs.	α	2.88 (1.49,6.95)	4451	1.0000
	β	1.5 (1.1,1.9)	7091	1.0000
	ϕ	0.85 (0.69,0.96)	5568	0.9997
	σ	1 (0.9,1.12)	5271	1.0006
Yukon R. Teslin Headwaters	α	5.09 (2.55,8.66)	4591	0.9999
	β	2 (1.5,2.4)	6222	0.9999
	ϕ	0.76 (0.54,0.93)	3638	1.0010
	σ	0.96 (0.87,1.06)	6714	1.0001

Table 7. Biological benchmarks based on estimated egg mass-recruitment relationships by Conservation Unit. Values are in thousands of fish, and shown are biological benchmarks assuming different age, size and sex composition observed in the first six years of time series ("Early"), last six years of time series ("Recent"), or averaged over all years .

Conservation Unit	Reference Point	Period	Median (80th percentile)
Big Salmon	S_{MSR}	Average (all years)	5.7 (0.92,11.31)
	S_{MSY}	Average (all years)	3.04 (0.45,6.09)
	S_{MSR}	Early (1985-1990)	4.6 (0.73,9.61)
	S_{MSY}	Early (1985-1990)	2.7 (0.36,5.23)
	S_{MSR}	Recent (2019-2024)	6.42 (0.84,12.04)
	S_{MSY}	Recent (2019-2024)	3.22 (0.41,6.69)
Middle Yukon R. and tribs.	S_{MSR}	Average (all years)	16.92 (2.82,35.88)
	S_{MSY}	Average (all years)	8.39 (1.38,20.24)
	S_{MSR}	Early (1985-1990)	14.38 (2.7,32.12)
	S_{MSY}	Early (1985-1990)	7.76 (1.32,17.89)
	S_{MSR}	Recent (2019-2024)	18.21 (2.19,38.2)
	S_{MSY}	Recent (2019-2024)	8.63 (1.08,22.37)
Nordenskiold	S_{MSR}	Average (all years)	1.31 (1,1.83)
	S_{MSY}	Average (all years)	0.99 (0.8,1.28)
	S_{MSR}	Early (1985-1990)	1.01 (0.77,1.41)
	S_{MSY}	Early (1985-1990)	0.81 (0.65,1.05)
	S_{MSR}	Recent (2019-2024)	1.57 (1.2,2.2)
	S_{MSY}	Recent (2019-2024)	1.12 (0.91,1.45)
Northern Yukon R. and tribs.	S_{MSR}	Average (all years)	8.2 (0.81,13.49)
	S_{MSY}	Average (all years)	3.86 (0.4,8.38)
	S_{MSR}	Early (1985-1990)	7.01 (0.87,10.96)
	S_{MSY}	Early (1985-1990)	3.6 (0.43,6.96)
	S_{MSR}	Recent (2019-2024)	8.67 (0.75,15.35)
	S_{MSY}	Recent (2019-2024)	3.95 (0.37,9.6)
Pelly	S_{MSR}	Average (all years)	11.13 (0.82,19.68)
	S_{MSY}	Average (all years)	5.22 (0.41,12.46)
	S_{MSR}	Early (1985-1990)	9.58 (1.17,16.67)
	S_{MSY}	Early (1985-1990)	4.91 (0.58,10.33)
	S_{MSR}	Recent (2019-2024)	11.52 (0.98,22.52)
	S_{MSY}	Recent (2019-2024)	5.29 (0.49,14.21)
Stewart	S_{MSR}	Average (all years)	4.05 (1.07,8.62)
	S_{MSY}	Average (all years)	2.24 (0.52,4.29)
	S_{MSR}	Early (1985-1990)	3.24 (1.07,7.92)
	S_{MSY}	Early (1985-1990)	1.97 (0.51,3.97)
	S_{MSR}	Recent (2019-2024)	4.62 (0.94,9.07)
	S_{MSY}	Recent (2019-2024)	2.38 (0.46,4.54)

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Conservation Unit	Reference Point	Period	Median (80th percentile)
Upper Yukon R.	S_{MSR}	Average (all years)	6.22 (3.02,9.98)
	S_{MSY}	Average (all years)	3.12 (1.42,4.98)
	S_{MSR}	Early (1985-1990)	4.99 (3.1,8.41)
	S_{MSY}	Early (1985-1990)	2.82 (1.54,4.48)
	S_{MSR}	Recent (2019-2024)	6.9 (2.62,10.7)
	S_{MSY}	Recent (2019-2024)	3.24 (1.26,5.28)
White and tribs.	S_{MSR}	Average (all years)	5.89 (0.89,10.89)
	S_{MSY}	Average (all years)	2.92 (0.44,6.25)
	S_{MSR}	Early (1985-1990)	4.9 (0.91,9.19)
	S_{MSY}	Early (1985-1990)	2.68 (0.44,5.33)
	S_{MSR}	Recent (2019-2024)	6.4 (0.68,12.11)
	S_{MSY}	Recent (2019-2024)	3.03 (0.34,6.94)
Yukon R. Teslin Headwaters	S_{MSR}	Average (all years)	6.87 (2.17,14.18)
	S_{MSY}	Average (all years)	3.76 (1.02,7.63)
	S_{MSR}	Early (1985-1990)	5.46 (2.29,11.89)
	S_{MSY}	Early (1985-1990)	3.32 (1.07,6.79)
	S_{MSR}	Recent (2019-2024)	7.8 (2,15.46)
	S_{MSY}	Recent (2019-2024)	4.02 (0.97,8.33)

Table 8. *Summary of anthropogenic, ecosystem, and climate factors affecting hypothesized to influence Yukon River Chinook salmon.*

Life stage	Anthropogenic, ecosystem, or climate factor	Description or hypothesis	References
Spawner	Water temperature	Temperatures >18C cause heat stress and impact spawning success	Feddern et al. 2024; Murdoch et al. 2024
	Discharge/flow levels	Low flow reduce spawning success via higher temperature, habitat fragmentation	Howard and von Biela 2023; Neuswanger et al. 2015
	Ichthypohonus hoferi (pathogen)	Primarily infects maturing adults in the marine environment, potentially high mortality during spawning migration	Carroll and Liller 2023; Murphy et al. 2023; 2024; Kocan et al. 2004
	Thiamine deficiency complex (TDC)	Low thiamine in adults has transgenerational effects	Larson and Howard 2019; Howard and von Biela 2023
Egg	Stream flow	High flows scour streambeds, reducing egg survival	Feddern et al. 2024
	Winter stream temperature	Cold and/or long winters reduce incubation survival	Murdoch et al. 2020; 2024
Fry / smolt	Aufeis (icing)	Reduced or halted streamflow from ice formations decreases fry survival	von Finster et al. 2025; in press
	Stream flow / precipitation	High streamflow (high precipitation) may reduce foraging efficiency	Murdoch et al. 2024; Feddern et al. 2024
	Ice out date	Early ice-out allows earlier outmigration	Murdoch et al. 2024; Cunningham et al. 2018; Feddern et al. 2024
	Hydroelectric dam	Fry/smolts pass through dam, decreasing downstream survival	Twardek et al. 2023b
	Placer mining	Sediment deposition and suspension can decrease egg and fry survival	Jensen et al. 2009; von Finster et al. in press
Juvenile	Sea surface temperature - winter	Colder first winters result in less productivity	Feddern et al. 2024
	Sea ice cover	Productivity is lower in years when ice persists in the Bering Sea later	Feddern et al. 2024; Murdoch et al. 2024
	Predation	Increase in apex predator abundance	Ohlberger et al. 2019

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Life stage	Anthropogenic, ecosystem, or climate factor	Description or hypothesis	References
Adult	Pink salmon competition	Abundant pink salmon cause trophic cascades in the Bering Sea	Ruggerone et al. 2016; Cunningham et al. 2018
	Prey quality	Declining prey quality in the Eastern Bering Sea hypothesized to be the cause of thiamine deficiency in spawning adults	Howard and von Biela 2023

Table 9. Descriptions of the alternative fishery management measures (aka Harvest Control Rules) considered in the closed loop simulations.

Harvest Control Rule	Description
Moratorium	Current agreement between ADF&G and DFO (aka 'moratorium') that limits directed harvest unless border passage is expected to exceed 71,000 fish, which for simplicity is operationalized as a forecasted run-size that exceeds 71,000
IMEG	Interim Management Escapement goal (IMEG) that was in place prior to the current moratorium. At forecasts below 42,500 fish passing border (lower bound of escapement goal) no directed harvest and target harvest equal to 'surplus' if forecast exceeds escapement goal
Moratorium w/ harvest cap	Current moratorium unless border passage is forecast to exceed 71,000 fish in which case there is a harvest rate cap at large run-sizes that is equal to U_{MSY} of the least productive Conservation Unit (36%)
IMEG w/ harvest cap	Interim Management Escapement goal with a harvest rate cap at large run-sizes that is equal to U_{MSY} of the least productive Conservation Unit
Precationary Approach (PA) alternative	Harvest Control Rule that is consistent with DFO's Precautionary Approach Framework (DFO 2009): no harvest below lower fishery reference point (31,000) with progressive increase in allowable harvest rate up to maximum equal to U_{MSY} of the least productive Conservation Unit when run-sizes reach spawner abundances associated with the Upper Stock Reference Point (86,000)

Table 10. Parameters upon which closed-loop simulation model to assess impact of possible future fishery management measures was conditioned. α values are those under the reference scenario (most recent generation productivity; 2013-2018), and α values for alternative (robustness) productivity scenarios are found in Table 11. Mean age-at-maturity probabilities (ps) correspond to ages 4-7.

Conservation Unit	Parameter	Median (50th percentiles) or value
Big Salmon	α	2.1 (1.73,2.56)
	β	2 (1.72,2.27)
Middle Yukon R. and tribs.	α	1.18 (1.03,1.37)
	β	1.41 (1.24,1.58)
Nordenskiold	α	2.18 (1.72,2.76)
	β	2.78 (2.31,3.24)
Northern Yukon R. and tribs.	α	1.56 (1.26,1.97)
	β	1.72 (1.42,2)
Pelly	α	1.12 (0.95,1.36)
	β	0.74 (0.63,0.86)
Stewart	α	8.06 (6.12,10.56)
	β	6.74 (6.07,7.47)
Upper Yukon R.	α	1.24 (1.06,1.47)
	β	1.8 (1.56,2.06)
White and tribs.	α	1.64 (1.3,2.1)
	β	0.63 (0.51,0.75)
Yukon R. Teslin Headwaters	α	4.43 (3.48,5.58)
	β	2.23 (1.99,2.47)
	p_1	0.09 (0.08,0.09)
	p_2	0.34 (0.34,0.35)
	p_3	0.49 (0.48,0.49)
	p_4	0.08 (0.08,0.08)
	ϕ	0.75
	Forecast Error	0.79
	Outcome Error	0.1

Table 11. Alternative productivity (α) scenarios tested in forward simulations. Values are the posterior medians of (α) estimated by time-varying productivity models across three time frames: the most recent Chinook salmon generation (brood years 2012–2017), the long term average (1985–2017), and the most recent fully observed single brood year (2017).

Conservation Unit	Productivity Scenario	Median (95th percentile)
Big Salmon	Reference (most recent generation)	0.5 (-0.39,1.48)
	Robustness (long-term average)	1.55 (-0.05,2.54)
	Robustness (most recent year)	0.13 (-0.48,0.81)
Middle Yukon R. and tribs.	Reference (most recent generation)	0.56 (-0.73,1.82)
	Robustness (long-term average)	1.44 (-0.25,2.4)
	Robustness (most recent year)	-0.13 (-0.9,0.78)
Nordenskiold	Reference (most recent generation)	2.15 (0.94,3.62)
	Robustness (long-term average)	2.09 (0.97,3.17)
	Robustness (most recent year)	1.85 (0.85,2.73)
Northern Yukon R. and tribs.	Reference (most recent generation)	0.21 (-0.63,1.23)
	Robustness (long-term average)	1.53 (-0.29,3.03)
	Robustness (most recent year)	0.01 (-0.52,0.6)
Pelly	Reference (most recent generation)	0.13 (-1.01,1.15)
	Robustness (long-term average)	1.27 (-0.65,2.31)
	Robustness (most recent year)	-0.59 (-1.13,0.05)
Stewart	Reference (most recent generation)	0.83 (-0.36,1.77)
	Robustness (long-term average)	1.59 (0.14,2.81)
	Robustness (most recent year)	0.38 (-0.41,1.21)
Upper Yukon R.	Reference (most recent generation)	0.78 (-0.42,1.71)
	Robustness (long-term average)	1.39 (0.13,2.37)
	Robustness (most recent year)	0.32 (-0.59,1.16)
White and tribs.	Reference (most recent generation)	0.26 (-0.48,1.11)
	Robustness (long-term average)	1.24 (-0.17,2.54)
	Robustness (most recent year)	-0.03 (-0.67,0.64)
Yukon R. Teslin Headwaters	Reference (most recent generation)	1.56 (0.5,2.32)
	Robustness (long-term average)	1.87 (0.9,3.07)
	Robustness (most recent year)	1.17 (0.37,1.98)

Table 12. Biological and fishery performance measures considered in the closed-loop simulations to assess the performance of alternative fishery management measures. Where applicable, the years over which metrics are calculated were t_1=2030 – t_2=2050.

Metric	Description	Equation
Escapement	Average spawner abundance across all CUs across replicate simulations and years, where n rep is the number of replicate simulations and t_1 and t_2 are the first and last years over which the metric is calculated	$\frac{\sum_i \sum_{n_{rep}=1} \sum_{t_1}^{t_2} S_{i,t}}{t_2 - t_1 + 1}$
Harvest	Average annual harvest (H) across all CUs across replicate simulations and years	$\frac{\sum_i \sum_{n_{rep}=1} \sum_{t_1}^{t_2} H_{i,t}}{t_2 - t_1 + 1}$
Canadian Harvest	Average annual harvest (H) targeted to take place in Canada after accounting for harvest sharing provisions with Alaska	$\frac{\sum_i \sum_{n_{rep}=1} \sum_{t_1}^{t_2} H_{i,t}}{t_2 - t_1 + 1} * 0.23$
Exploitation Rate	Average annual harvest rate (U) for stock aggregate across replicate simulations and years	$\frac{\sum_{t_1}^{t_2} U_t}{t_2 - t_1 + 1}$
Prop. years fishery closed	Proportion of years across replicate simulations and years in which the simulated harvest control rule would cause the fishery to close (total allowable catch [TAC] = 0)	$\frac{\sum_i \sum_{n_{rep}=1} \sum_{t_1}^{t_2} TAC_{i,t}=0}{t_2 - t_1 + 1}$
No. CUs below lower benchmark	Number of Conservation Units (CUs; i) with spawner abundance (S) at end of simulations that falls below CU specific lower biological benchmark, defined as 20% of the spawner abundance associated with maximum returns (20%S_{MSR}) as estimated from spawner recruitment models with stationary productivity	$\sum_i S_{i,t=2050} < 0.2 S_{MSR,i}$

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Metric	Description	Equation
No. CUs above upper benchmark	Number of Conservation Units (CUs) with spawner abundance at end of simulations that is above CU specific upper biological benchmark, defined as 40% of the spawner abundance associated with maximum returns (40%S_{MSR}) as estimated from spawner recruitment models with stationary productivity	$\sum_i S_{i,t=2050} > S_{MSR,i}$

Table 13. Consequences of alternative Harvest Control Rules based on range of biological and fishery performance measures under the simulated reference (base case) scenario where productivity remains at the most recent generational average. CU status performance metrics (i.e. No. CUs below lower benchmark, between benchmarks, above upper benchmark, above CU rebuilding target, and extinct) reflect status in the final year of simulations (2050). Harvest and escapement metrics are in units of thousands of fish and are the average value across 2030-2050. All mean and median values are summarized across 1000 simulations.

Harvest Control Rule	Performance measure	Mean	Median (50^{th} percentiles)
No fishing	Escapement	33.84	32.96 (27.61,39.53)
	Harvest	0	0 (0,0)
	Canadian Harvest	0	0 (0,0)
	Exploitation Rate	0	0 (0,0)
	Prop. years fishery closed	1	1 (1,1)
	No. CUs below lower benchmark	2.36	2 (1,3)
	No. CUs between benchmarks	1.98	2 (1,3)
	No. CUs above upper benchmark	3.39	3 (2,4)
	No. CUs above CU rebuilding target	1.25	1 (0,2)
	No. CUs extinct	0.03	0 (0,0)
Moratorium	Escapement	28.32	27.87 (23.56,32.75)
	Harvest	3.43	2.89 (2,4.23)
	Canadian Harvest	0.85	0.68 (0.46,1.06)
	Exploitation Rate	0.1	0.09 (0.07,0.12)
	Prop. years fishery closed	0.84	0.86 (0.76,0.9)
	No. CUs below lower benchmark	3.22	3 (2,4)
	No. CUs between benchmarks	1.86	2 (1,3)
	No. CUs above upper benchmark	2.8	3 (2,4)
	No. CUs above CU rebuilding target	1.09	1 (0,2)
	No. CUs extinct	0.03	0 (0,0)
IMEG	Escapement	25.05	24.74 (21.64,28.47)
	Harvest	5.72	4.97 (3.44,7.17)
	Canadian Harvest	1.4	1.18 (0.81,1.74)
	Exploitation Rate	0.17	0.16 (0.13,0.19)
	Prop. years fishery closed	0.68	0.71 (0.62,0.76)
	No. CUs below lower benchmark	3.81	4 (3,5)
	No. CUs between benchmarks	1.74	2 (1,2)
	No. CUs above upper benchmark	2.44	2 (2,3)
	No. CUs above CU rebuilding target	0.95	1 (0,1)
	No. CUs extinct	0.05	0 (0,0)

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Harvest Control Rule	Performance measure	Mean	Median (50th percentiles)
Moratorium cap	Escapement	28.78	28.32 (23.6,33.75)
	Harvest	2.95	2.54 (1.86,3.66)
	Canadian Harvest	0.66	0.58 (0.42,0.82)
	Exploitation Rate	0.09	0.08 (0.07,0.1)
	Prop. years fishery closed	0.84	0.86 (0.76,0.9)
	No. CUs below lower benchmark	3.15	3 (2,4)
	No. CUs between benchmarks	1.86	2 (1,3)
	No. CUs above upper benchmark	2.87	3 (2,4)
	No. CUs above CU rebuilding target	1.09	1 (0,2)
	No. CUs extinct	0.03	0 (0,0)
IMEG cap	Escapement	27.01	26.66 (22.58,31.24)
	Harvest	4.74	4.31 (3,5.94)
	Canadian Harvest	1.04	0.95 (0.67,1.3)
	Exploitation Rate	0.14	0.14 (0.11,0.16)
	Prop. years fishery closed	0.67	0.67 (0.57,0.76)
	No. CUs below lower benchmark	3.51	3 (3,4)
	No. CUs between benchmarks	1.75	2 (1,3)
	No. CUs above upper benchmark	2.73	3 (2,4)
	No. CUs above CU rebuilding target	0.98	1 (0,1)
	No. CUs extinct	0.03	0 (0,0)
PA Alternative	Escapement	28.64	28.43 (24.03,33.07)
	Harvest	3.38	3.01 (2.27,4.15)
	Canadian Harvest	0.78	0.7 (0.52,0.96)
	Exploitation Rate	0.1	0.1 (0.08,0.11)
	Prop. years fishery closed	0.54	0.52 (0.43,0.63)
	No. CUs below lower benchmark	3.1	3 (2,4)
	No. CUs between benchmarks	1.86	2 (1,3)
	No. CUs above upper benchmark	2.94	3 (2,4)
	No. CUs above CU rebuilding target	1.07	1 (0,2)
	No. CUs extinct	0.04	0 (0,0)

Table 14. Consequences of alternative Harvest Control Rules based on range of biological and fishery performance measures under the robustness scenario where productivity remains severely depressed (at 2017 levels). CU status performance metrics (i.e. No. CUs below lower benchmark, between benchmarks, above upper benchmark, above CU rebuilding target, and extinct) reflect status in the final year of simulations (2050). Harvest and escapement metrics are in units of thousands of fish and are the average value across 2030-2050. All mean and median values are summarized across 1000 simulations.

Harvest Control Rule	Performance measure	Mean	Median (50th percentiles)
No fishing	Escapement	19.11	18.36 (14.94,22.5)
	Harvest	0	0 (0,0)
	Canadian Harvest	0	0 (0,0)
	Exploitation Rate	0	0 (0,0)
	Prop. years fishery closed	1	1 (1,1)
	No. CUs below lower benchmark	4.26	4 (3,5)
	No. CUs between benchmarks	1.28	1 (1,2)
	No. CUs above upper benchmark	1.97	2 (1,3)
	No. CUs above CU rebuilding target	0.79	1 (0,1)
	No. CUs extinct	0.71	1 (0,1)
Moratorium	Escapement	16.43	15.99 (12.54,19.61)
	Harvest	1.79	1.6 (1.32,2.02)
	Canadian Harvest	0.42	0.37 (0.3,0.47)
	Exploitation Rate	0.11	0.1 (0.08,0.12)
	Prop. years fishery closed	0.95	0.95 (0.9,1)
	No. CUs below lower benchmark	4.37	4 (3,5)
	No. CUs between benchmarks	1.03	1 (0,2)
	No. CUs above upper benchmark	1.71	2 (1,2)
	No. CUs above CU rebuilding target	0.75	1 (0,1)
	No. CUs extinct	1.14	1 (0,2)
IMEG	Escapement	15.5	15.05 (11.99,18.61)
	Harvest	2.54	2.11 (1.58,3)
	Canadian Harvest	0.6	0.49 (0.36,0.7)
	Exploitation Rate	0.14	0.13 (0.11,0.16)
	Prop. years fishery closed	0.86	0.86 (0.81,0.95)
	No. CUs below lower benchmark	4.41	4 (4,5)
	No. CUs between benchmarks	1	1 (0,2)
	No. CUs above upper benchmark	1.63	2 (1,2)
	No. CUs above CU rebuilding target	0.69	1 (0,1)
	No. CUs extinct	1.26	1 (0,2)

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Harvest Control Rule	Performance measure	Mean	Median (50th percentiles)
Moratorium cap	Escapement	16.49	15.98 (12.86,20.04)
	Harvest	1.66	1.53 (1.28,1.9)
	Canadian Harvest	0.38	0.35 (0.29,0.43)
	Exploitation Rate	0.1	0.09 (0.08,0.11)
	Prop. years fishery closed	0.95	0.95 (0.9,1)
	No. CUs below lower benchmark	4.34	4 (4,5)
	No. CUs between benchmarks	1.08	1 (0,2)
	No. CUs above upper benchmark	1.73	2 (1,2)
	No. CUs above CU rebuilding target	0.73	1 (0,1)
	No. CUs extinct	1.12	1 (0,2)
IMEG cap	Escapement	15.81	15.32 (12.34,18.91)
	Harvest	2.19	1.9 (1.51,2.56)
	Canadian Harvest	0.49	0.43 (0.34,0.57)
	Exploitation Rate	0.12	0.12 (0.1,0.14)
	Prop. years fishery closed	0.86	0.86 (0.81,0.95)
	No. CUs below lower benchmark	4.41	4 (4,5)
	No. CUs between benchmarks	1.04	1 (0,2)
	No. CUs above upper benchmark	1.69	2 (1,2)
	No. CUs above CU rebuilding target	0.7	1 (0,1)
	No. CUs extinct	1.16	1 (0,2)
PA Alternative	Escapement	16.14	15.71 (12.25,19.68)
	Harvest	1.81	1.66 (1.38,2.06)
	Canadian Harvest	0.42	0.38 (0.32,0.48)
	Exploitation Rate	0.11	0.1 (0.09,0.12)
	Prop. years fishery closed	0.77	0.76 (0.67,0.86)
	No. CUs below lower benchmark	4.37	4 (3,5)
	No. CUs between benchmarks	1.04	1 (0,2)
	No. CUs above upper benchmark	1.66	2 (1,2)
	No. CUs above CU rebuilding target	0.72	1 (0,1)
	No. CUs extinct	1.21	1 (0,2)

Table 15. Consequences of alternative Harvest Control Rules based on range of biological and fishery performance measures under the robustness scenario where productivity returns to its long-term average. CU status performance metrics (i.e. No. CUs below lower benchmark, between benchmarks, above upper benchmark, above CU rebuilding target, and extinct) reflect status in the final year of simulations (2050). Harvest and escapement metrics are in units of thousands of fish and are the average value across 2030-2050. All mean and median values are summarized across 1000 simulations.

Harvest Control Rule	Performance measure	Mean	Median (50^{th} percentiles)
No fishing	Escapement	94.02	93.17 (85.46,101.41)
	Harvest	0	0 (0,0)
	Canadian Harvest	0	0 (0,0)
	Exploitation Rate	0	0 (0,0)
	Prop. years fishery closed	1	1 (1,1)
	No. CUs below lower benchmark	0.08	0 (0,0)
	No. CUs between benchmarks	0.13	0 (0,0)
	No. CUs above upper benchmark	2.63	2 (1,4)
	No. CUs above CU rebuilding target	6.14	7 (5,8)
	No. CUs extinct	0.02	0 (0,0)
Moratorium	Escapement	63.04	62.64 (57.66,68.04)
	Harvest	28.51	26.7 (20.97,35.29)
	Canadian Harvest	8.36	7.61 (5.77,10.41)
	Exploitation Rate	0.29	0.28 (0.24,0.33)
	Prop. years fishery closed	0.4	0.38 (0.33,0.48)
	No. CUs below lower benchmark	0.33	0 (0,0)
	No. CUs between benchmarks	1.05	0 (0,1)
	No. CUs above upper benchmark	4.45	5 (2,7)
	No. CUs above CU rebuilding target	3.14	2 (0,6)
	No. CUs extinct	0.03	0 (0,0)
IMEG	Escapement	47.04	46.86 (42.47,51.11)
	Harvest	40.42	38.99 (31.65,48.21)
	Canadian Harvest	11.35	10.9 (8.62,13.43)
	Exploitation Rate	0.43	0.43 (0.38,0.48)
	Prop. years fishery closed	0.22	0.19 (0.14,0.29)
	No. CUs below lower benchmark	0.89	0 (0,1)
	No. CUs between benchmarks	1.98	1 (0,3)
	No. CUs above upper benchmark	4.26	4 (2,6)
	No. CUs above CU rebuilding target	1.87	1 (0,3)
	No. CUs extinct	0.01	0 (0,0)

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Harvest Control Rule	Performance measure	Mean	Median (50th percentiles)
Moratorium cap	Escapement	74.4	73.98 (67.75,80.42)
	Harvest	20.83	20.01 (16.02,24.8)
	Canadian Harvest	4.67	4.42 (3.5,5.6)
	Exploitation Rate	0.21	0.21 (0.17,0.24)
	Prop. years fishery closed	0.38	0.38 (0.29,0.48)
	No. CUs below lower benchmark	0.05	0 (0,0)
	No. CUs between benchmarks	0.26	0 (0,0)
	No. CUs above upper benchmark	4.29	4 (2,7)
	No. CUs above CU rebuilding target	4.39	4.5 (2,7)
	No. CUs extinct	0	0 (0,0)
IMEG cap	Escapement	67.92	67.3 (61.28,73.84)
	Harvest	28.12	27.74 (23.35,32.53)
	Canadian Harvest	5.86	5.73 (4.69,6.71)
	Exploitation Rate	0.28	0.28 (0.26,0.31)
	Prop. years fishery closed	0.19	0.19 (0.1,0.24)
	No. CUs below lower benchmark	0.07	0 (0,0)
	No. CUs between benchmarks	0.31	0 (0,1)
	No. CUs above upper benchmark	5.08	5 (3,7)
	No. CUs above CU rebuilding target	3.52	3 (1,6)
	No. CUs extinct	0.01	0 (0,0)
PA Alternative	Escapement	73.9	73.37 (67.43,79.77)
	Harvest	22.14	21.6 (17.72,26.06)
	Canadian Harvest	5.43	5.27 (4.24,6.38)
	Exploitation Rate	0.22	0.22 (0.19,0.25)
	Prop. years fishery closed	0.11	0.1 (0.05,0.14)
	No. CUs below lower benchmark	0.06	0 (0,0)
	No. CUs between benchmarks	0.22	0 (0,0)
	No. CUs above upper benchmark	4.26	4 (2,6)
	No. CUs above CU rebuilding target	4.45	5 (2,7)
	No. CUs extinct	0.01	0 (0,0)

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APPENDIX A. GENETIC STOCK INFORMATION

Biological information on Yukon River Chinook salmon has been collected annually from the fish wheels at White and Sheep rocks (scales; most years from 1985–2008) and the gillnet test fishery at Eagle (tissue samples; 2005–present; 3). Samples have typically been taken over the annual upstream adult migration period, with the number of samples taken each day roughly proportional to daily passage (Figure A.1) (Connors et al. 2020). Genetic material has been recovered from 150–300 archived scale samples for most years 1985–2005, and since then 500–1500 tissue samples have been taken per year for genetic stock assignment. These samples have been used to genotype individual fish by microsatellite panel [1985–2016; Beacham et al. (2006)) or single nucleotide polymorphism (SNP) panel (2017–2019, Beacham et al. (2018) and then assigned to one of eight Conservation Units (CUs). These CU groupings differ slightly from the grouping reported in annual Yukon River Panel Joint Technical Committee reports (JTC 2025). This benchmarked baseline consists of 4,713 Chinook salmon sampled at 29 unique spawning locations (or tributaries) throughout the Canadian portion of the Yukon River Basin (Table A.1).

To evaluate how accurate genetic assignments were we conducted a simulation analysis on the baseline samples (i.e., samples of known origin) in rubias (Moran and Anderson 2019). Simulations indicated that the average simulation accuracy within each CU was greater than 75%, and typically greater than 95%, for all CUs using the microsatellite baseline and similar for the SNP baseline with the exception of the Chandindu, Earn, and Nisling rivers which had poorer assignment accuracies (Table A.1). Given the variable uncertainty in genetic stock assignments uncovered by these simulations we ensured the run-reconstruction model (Appendix B) fit to these data to derive annual estimates of CU border passage explicitly took uncertainty in genetic stock assignments into account. Genetic samples from the Big Salmon and Teslin (below Teslin lake) tributaries could not reliably differentiated from baseline sampling locations in the Middle Yukon CU; as a result they were grouped with the Middle Yukon CU for the purposes of our genetic analyses. Similarly, genetic samples from the McQuesten River more closely grouped with the Northern CU than with the Stewart CU that they originated from (Table A.1).These genetic misassignments were subsequently accounted for in the CU scale run-reconstructions (Appendix B).

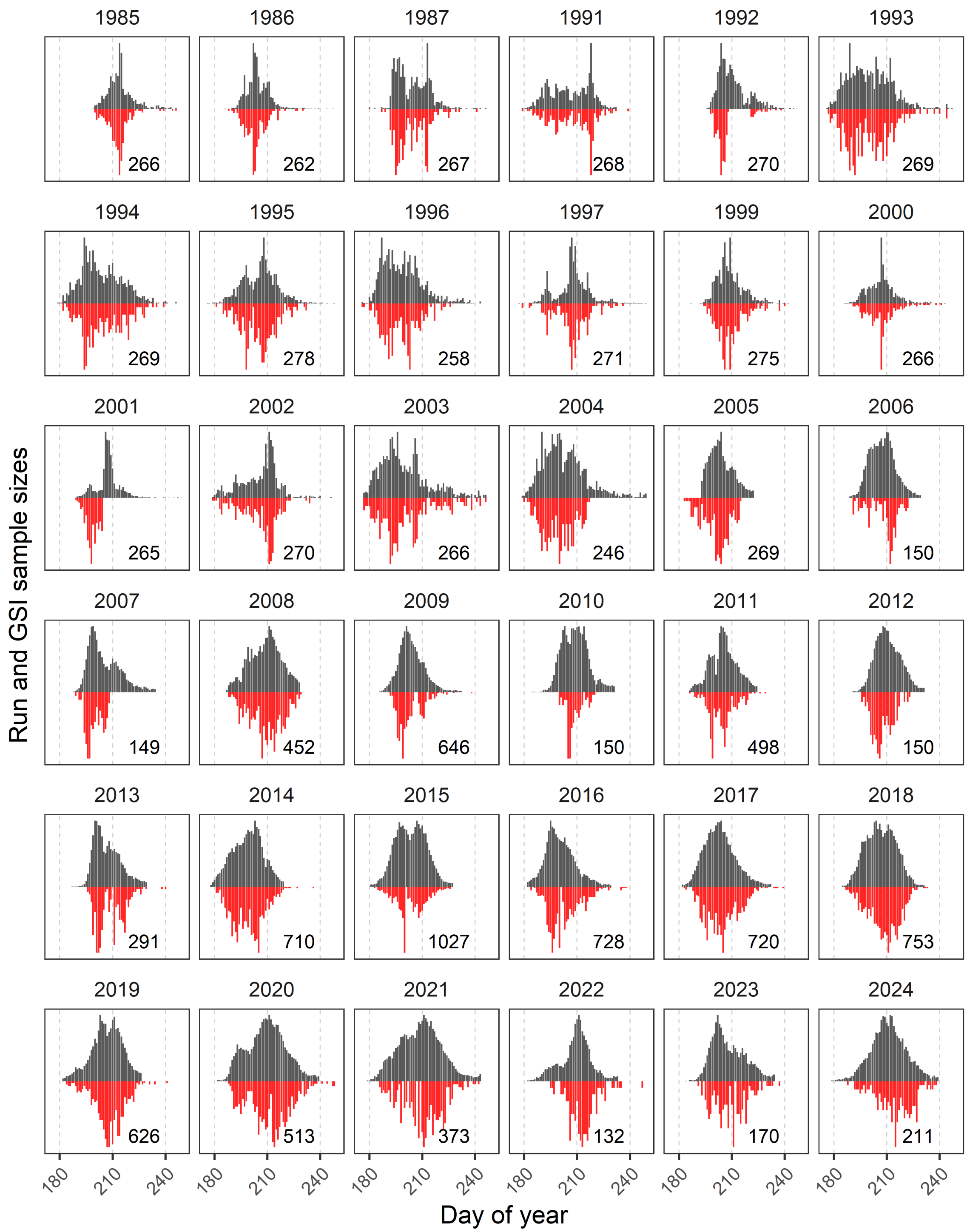

Figure A.1. Daily estimates of Chinook salmon border passage (grey bars) and scales or tissues sampled (red bars) from the run to determine genetic stock identification (GSI). Each bar is relative to the maximum for a given year with GSI samples inverted to allow for visualization of samples relative to border passage. The number of GSI samples in each year is in the bottom right of each panel. Note that scale samples were unavailable for 1988–1990, and 1998. For reference, in a non-leap year 29 June, 29 July and 28 August are equivalent to the 180th, 210th and 240th day of the year, respectively (dashed grey bars).

Table A.1. Yukon River Chinook salmon populations considered the primary spawning tributaries or location where samples were collected for the genetic baseline (tributary name) total number of baseline genetic samples (Msat(n), SNP(n)), corresponding Conservation Units (CU), CU codes, and probability of correctly assigning a genetic sample for a tributary (collection) back to its known tributary or CU based on simulation analysis.

Tributary name	Msat(n)	SNP(n)	CU	Conservation Unit code[2]	Probability to tributary (Msat)	Probability to CU (SNP)
Big Salmon River[1]	394	195	MidYR	CK 70	0.99	0.89
Little Salmon River	285	175	MidYR	CK 73	0.97	0.95
Tatchun River	327	95	MidYR	CK 73	0.95	0.95
Yukon River At Yukon Crossing	197	170	MidYR	CK 73	0.98	0.92
Teslin River[1]	328	200	MidYR	CK 73	0.95	0.94
Nordenskiold River	126	96	Norden	CK 71	0.91	0.91
Chandindu River	526	102	NYR	CK 76	0.95	0.49
Klondike River	247	163	NYR	CK 76	0.99	0.9
Mcquesten River	144	135	NYR	CK 76	0.93	0.9
Blind Creek	251	134	Pelly	CK 72	0.98	0.94
Earn River	34	42	Pelly	CK 72	0.9	0.67
Glenlyon River	61	58	Pelly	CK 72	0.96	0.94
Hoole River	22	106	Pelly	CK 72	0.92	0.92
Kalzas River	8	73	Pelly	CK 72	0.99	0.87
Pelly River	178	59	Pelly	CK 72	0.83	0.88
Ross River[1]	103	99	Pelly	CK 72	0.98	0.91
Mayo River	142	56	Stew	CK 74	0.84	0.83
Stewart River	231	140	Stew	CK 74	0.85	0.78
Morley River	59	120	Teslin	CK 68	0.93	0.95
Nisutlin River	117	168	Teslin	CK 68	0.97	0.96
Wolf River	40	90	Teslin	CK 68	0.85	0.96
Takhini River	198	169	UpperYR	CK 69	0.98	0.96
Yukon River At Whitehorse	200	200	UpperYR	CK 69	0.93	0.95
Nisling River	40	38	White	CK 75	0.75	0.64

[1] Samples from the Big Salmon river and Teslin river (below Teslin lake) are both assigned to the Middle Yukon CU for the purposes of this analysis because they cannot be reliably differentiated from other baseline sampling locations in the Middle Yukon CU. [2] Holtby and Ciruna (2007)

APPENDIX B. RUN RECONSTRUCTION MODEL

We estimated the daily passage of Chinook into Canadian (CDN) waters using a multi-CU run reconstruction (RR) model, which simultaneously accounts for uncertainty in observations and underlying population processes and allows for the inclusion of incomplete datasets. This model was originally described in Connors et al. (2022b), and was adapted to allow for time-varying catchability and fit to updated data (including genetics and an externally derived border passage index) through 2024. We separately modelled the dynamics of the nine component Conservation Units (CUs), indexed by i, in the Yukon Chinook Stock Management Unit (SMU). Annual, CU-specific border passage was the main parameter of interest estimated by the model, though we also estimated parameters for run timing and catchability. We separately modelled time-varying catchability to two fishing "gears" (index by g): fish wheels and sonar, which have been used to derive annual aggregate border passage estimates (fishwheel mark-recapture from 1985–2008; sonar from 2005–2024). Model notation and equations are listed in Table B.1 and B.2, respectively. While we refer to this model as a run reconstruction, it could also (perhaps more accurately) be referred to as a border passage reconstruction.

Table B.1. Model notation for multi-CU run-reconstruction model.

Symbol	Description
Indices	
i	Conservation Unit, i=1, . . . ,9
y	Year, y=1985, . . . ,2016
d	Julian day, d=160, . . . ,285
g	Gear type, 1=Sonar, 2=Fish Wheel
Data and inputs	
x_{igyd}	Observed CU composition by gear/year/day
E_{gyd}	Border passage counts by gear/year/day
I_y	Border passage index from stock aggregate run-reconstruction (Connors et al., 2022b)
CV_y	Coefficient of variation around border passage index
Parameters	
$\widehat{R}_{iy}$	Annual border passage by CU
$\widehat{\mu}_{iy}$	Mean Julian date of arrival by CU in first year
σ_i	Standard deviation around mean Julian date of arrival by CU
ε_{iy}	Process error in arrival timing by CU/year
Σ	Process error covariance in arrival timing
$q_{gy}^{(E)}$	Daily counts catchability by gear and year
Latent variables	
N_{iyd}	Daily numbers arriving by CU /year
ρ_{iyd}	Daily arrival proportions by CU /year
μ_{iyd}	Mean Julian date of arrival by CU /year
$\widehat{E}_{igyd}$	Predicted daily counts by CU /gear/year
$\widehat{I}_{iy}$	Predicted border passage index
p_{igyd}	CU composition by gear/year/day
τ_y	Standard deviation for log border passage index

Table B.2. Model equations for multi-CU run-reconstruction model. "N.B" stands for negative binomial and "NLL" stands for negative log likelihood.

Equation	Formula
Population Dynamics	
(TB.2.1) Estimated Parameters	$\Theta = \left\{ \hat{R}_{iy},\ \hat{\mu}_i,\ \sigma_i,\ \varepsilon_{iy},\ \sum,\ q_{gy}^{(E)},\ \tau \right\}$
(TB.2.2) Arrival timing, y=1	$\mu_{i,1} = \widehat{\mu}_i$
(TB.2.3) Arrival timing, y>1	$\mu_{i,y} = \mu_{i,y-1} \exp(\varepsilon_{i,y-1})$
(TB.2.4) Daily arrival proportions	$\rho_{iyd} = \frac{exp(-0.5(d-\mu_{iy})^2\sigma_i^{-2})}{\sum_k exp(-0.5(k-\mu_{iy})^2\sigma_i^{-2}}$
(TB.2.5) Daily arrivals (numbers)	$N_{iyd} = \widehat{R}_{iy}\rho_{iyd}$
Model predictions	
(TB.2.6) Predicted daily border counts	$\widehat{E}_{igyd} = q_g y^{(E)} N_{iyd}$
(TB.2.7) Predicted border passage index	$\widehat{I}_y = \sum_i \widehat{R}_{iy}$
(TB.2.8) Predicted CU composition	$p_{iyd} = N_{iyd} / \sum_j N_{jyd}$
Objective function	
(TB.2.9) Mean for N.B. count likelihood	$\eta_{gyd} = \sum_i \widehat{E}_{igyd}$
(TB.2.10) Variance for N.B. count likelihood	$\tau_{gyd}^2 = \eta_{gyd} + \eta_{gyd}^2 \phi_{gy}$
(TB.2.11) Re-parameterized mean predicted stock composition by gear type for N.B. count likelihood	$p_{gyd} = \eta_{gyd} / \tau_{gyd}^2$
(TB.2.12) Re-parameterized variance for predicted stock composition by gear type for N.B. count likelihood	$r_{gyd} = \eta_{gyd} p_{gyd} (1 - p_{gyd})$
(TB.2.13) N.B. NLL for daily counts	$L_E = \sum_g \sum_y \sum_d [-\ln \Gamma(E_{gyd} + \eta_{gyd}) + \ln \Gamma(\eta_{gyd}) + \ln \Gamma(E_{gyd} + 1) - \eta_{gyd} \ln(1 - p_{gyd})]$
(TB.2.14) Multinomial NLL for stock composition	$L_x = \sum_g \sum_y \sum_d [-\ln \Gamma(\sum_i x_{igyd} + 1) + \sum_i \ln \Gamma(x_{igyd} + 1) - \sum_i x_{igyd} \ln(p_{iyd})]$
(TB.2.15) Border passage index std. dev.	$\tau_y = \sqrt{\ln(CV_t^2 + 1)}$
(TB.2.16) Lognormal NLL for run size index	$L_I = \sum_{y=1985}^{2008} \left[\ln \tau_y + \frac{\ln 2\pi}{2} + \frac{(\ln I_y - \ln \widehat{I}_y)^2}{2\tau_y^2} \right]$
(TB.2.17) Multivariate-normal likelihood penalty on process errors	$L_\epsilon = 0.5 \left[\ln \lvert \boldsymbol{\sigma} \rvert - \boldsymbol{\epsilon}^T \boldsymbol{\sigma}^{-1} \boldsymbol{\epsilon} \right]$

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Equation	Formula
(TB.2.18) Border passage likelihood penalty in fish wheel years	$L_R = \sum_{y=1985}^{2005} \left[\ln 1 + \frac{\ln 2\pi}{2} + \frac{\left(\ln \widehat{R}_{iy} - \ln \widehat{R}_{i,y-1}\right)^2}{2} \right]$
(TB.2.19) Objective function	$L = L_E + L_x + L_I + L_\varepsilon + L_R$

B.1. POPULATION DYNAMICS

For each population, the daily proportion of salmon entering the model (i.e., crossing the U.S.–Canada border into CDN waters) in a given year is calculated based on a normal probability density function, with mean μ_{iy} with variance σ_i^2, which is divided by its sum to ensure that the individual values sum to 1 (Table B.2.4). Numbers of salmon arriving daily for each CU/year combination is subsequently calculated as the product of the daily arrival proportions and border passage (Table B.2.5). To model interannual variability in mean run timing, we assume that the CU-specific mean dates of arrival in a given year ($\{\mu_{i,y}\}_{i=1}^{9}$ or μ_y) are a function of run timing in the previous year, i.e.:

$$\mu_y = (\mu_{y-1})^{\epsilon_y} \tag{B.1}$$

where ϵ_y is a vector of nine normally distributed process errors with mean 0 and covariance $\sum$. The covariance matrix is constructed as $\sum = DCD$, where D is a diagonal matrix with the variance of μ_i as the iith element, and C is a symmetric correlation matrix, i.e.:

$$C_{i,j} = \begin{cases} 1 & i = j \\ c_{i,j} & i \neq j \end{cases} \tag{B.2}$$

where $c_{i,j}$ is the correlation between arrival timing deviations for CU i and j. We constrained all off-diagonal elements of C to have the same value, which assumes all stocks have equal correlation in arrival timing deviations.

B.2. OBSERVATION MODEL AND OBJECTIVE FUNCTION

Daily border passage counts were predicted by scaling the daily model-predicted passage by a gear-specific catchability factor (Table B.2.6). The fishwheel samples an unknown portion of the run, so fishwheel catchability was estimated, and allowed to vary from year to year to account for the potential influence of environmental conditions (e.g., discharge) on fishwheel catchability. While catchability may vary across CUs for a variety of reasons (i.e., bank orientation, run-timing, etc.), the data were not informative enough to estimate CU-specific catchability. In contrast, all passing fish were assumed to be observable by sonar, and so we fixed sonar catchability at 1. Similarly, the predicted border passage index was assumed to represent the total run at he border, so catchability to the border passage index was also fixed at 1 (Table B.2.7), and this index was taken from an updated fit of the model described in Connors et al. (2022a). Conservation Unit composition was predicted as the relative proportion of a CU present on each day (Table B.2.8).

Daily salmon counts at the border were assumed to arise from negative-binomial (NB) distributions (Table B.2.13). The (NB) distribution, describing the number of successes in a series of Bernoulli trials before r failures occur (with probability of success p), is broadly applicable as a model for overdispersed count data. We parameterized the NB distribution in terms of the mean count η and the dispersion ϕ (Table B.2.10-11). The variance of this distribution is $\tau^2 = \eta + \eta^2\phi$, hence the NB distribution is equivalent to the Poisson when $\phi = 0$. We set $\phi = 0$ for sonar counts, as these counts are believed to be relatively accurate. We set $\phi = 0.1$ for the fishwheel counts.

Conservation Unit composition data (x_{igyd}), which were based on summing the individual genetic assignment probabilities and rescaling to the total sample size for that CU and day thereby accounting for uncertainty in genetic assignments, were fitted using a multinomial likelihood (Table B.2.14). For each year/day/gear combination, the observed number of Chinook salmon by CU, $n_1, \ldots, n_9$, was assumed to arise from a multinomial distribution with sample size $n = \sum_i n_i$

and probabilities equal to the relative proportions of passage by CU for that year/day. Sample sizes in multinomial distributions for fisheries composition data are typically down-weighted to an "effective" sample size to account for correlations among fish within a given sample. However, we did not down-weight sample sizes as they were already relatively small. We note, however, that by applying a relatively large and fixed value for the NB dispersion parameter when modelling daily counts, as well as to the total run size index likelihood (see below), we are effectively down weighting the stock composition data.

Total border passage indices were assumed to arise from a lognormal distribution with standard deviation τ_y, which was derived from the annual estimates of the CV in the border passage index (Table B.2.13). Process errors in arrival timing are assumed to arise from a zero-mean multivariate normal distribution with covariance $\sum$ (Table B.2.15).

In some years prior to the implementation of sonar monitoring, the fishwheel data was not informative enough to estimate border passage for some CUs. To stabilize estimates in these cases, we applied a likelihood penalty on the interannual deviations of log-scale border passage for each CU (Table B.2.16). The penalty was set to 1, which is wide enough to have negligible effects in years for which data were sufficiently informative to estimate border passage and was omitted from 2005 onward, after the introduction of sonar assessment of border passage.

B.3. MODEL FITTING AND RECONSTRUCTIONS OF SPAWNER ABUNDANCE

We fit the run-reconstruction model using Template Model Builder (Kristensen et al. 2016). Model parameters were estimated via maximum likelihood and standard errors of parameters and quantities of interest were calculated using a delta method routine within TMB and were treated as equivalent to standard deviations.

Fishwheel catchability was estimated to average around 0.02 but varied considerably from year to year (Figure B.1), presumably as a result of different environmental conditions (e.g., flow) and/or fishwheel operation. Interannual deviations in arrival timing were estimated to be weakly correlated among CUs (pairwise correlation, which was assumed to be the same for all CUs, estimated to be equal to 0.15).

The model tended to fit the sonar count data well (Figure B.2) and in contrast, model fits to the fishwheel counts for 1985–2007 were more variable, which was expected due to the relatively imprecise nature of the fishwheel data (Figure B.3). In years when sonar and fishwheel counts overlapped (2005–2007), the tight correspondence between model estimates and sonar counts resulted in relatively poorer fits to the fishwheel data.

In the pre-sonar years the model attempted to simultaneously fit to the fishwheel relative abundance indices and the externally derived absolute border passage index (based on mark-recapture projects) so the model tended to generate estimates of total border passage that were in close alignment with the border passage index (Figure B.3). As more reliable abundance data (i.e., sonar based estimates beginning in 2005) became available, the border passage index had essentially no effect on model estimates, and the model more closely fit to the sonar based estimates of total border passage 2005–2024.

Because the reconstructions of Northern CU border passage included fish from the McQuesten River, we post-hoc estimated the annual contribution of the McQuesten River based on genetic analyses of samples collected at Eagle from 2013-2023 (mean:0.008; SD:0.006). Specifically, for each year we took a random draw from the average McQuesten River contribution to border

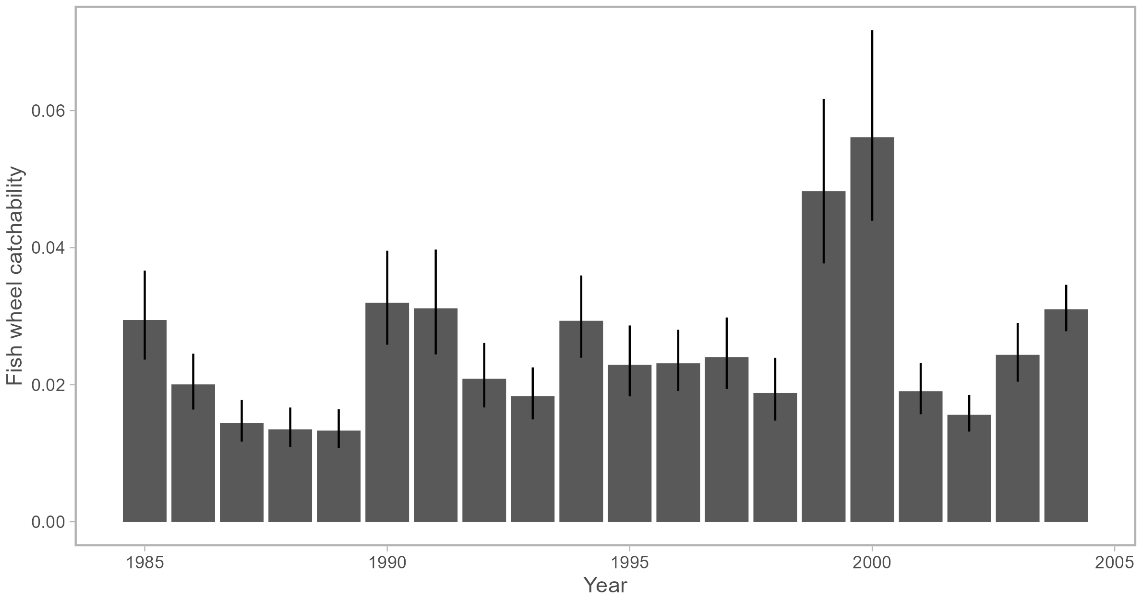

Figure B.1. Estimated fish wheel catchability, by year. Bars are median estimates +/- 95% confidence intervals.

passage, and a random draw from the border passage estimate for that year, and repeated this 1000 times. We then removed these estimates of McQuesten River border passage from the Northern CU and added them to the reconstructions of Stewart CU border passage.

The run-reconstruction model estimated border passage and so estimates of spawner abundance were derived by subtracting the inferred harvest of each CU by fisheries that occurred upstream (in Canada) in the mainstem Yukon River from estimated border passage. To do this we assumed all CUs were historically equally vulnerable to Canadian fisheries and calculated spawner abundance as $S_{i,y} = \widehat{R}_{i,y}(1 - U_{CDN,y})$ where $\widehat{R}_{i,y}$ is border passage for CU i in year y and $U_{CDN,y}$ is the aggregate harvest rate from Canadian fisheries as estimated by Connors et al. (2022a).

For the Big Salmon CU, which cannot be reliably differentiated with genetics from the Middle Mainstem CU, but which has had a sonar assessment project in operation since 2005, we inferred spawner abundance as a proportion of the reconstructed spawner abundance from the Middle Mainstem CU. This proportion was based on those years where there was data from the sonar assessment project (i.e, 2005 to present with exception of 2022) and the reconstructed spawner abundances explicitly accounted for variability in the proportional Big Salmon CU contribution, and uncertainty in Middle Mainstem spawner abundance, by simulating 1000 estimates of spawner abundance for both CUs by taking random draws from the average proportional contribution of the Big Salmon CU and estimates of Middle Mainstem CU spawner abundance in each year that did not have sonar based Big Salmon assessment. The resulting time series of Big Salmon spawner abundance was then removed for the entire time series from the Middle Mainstem CU.

For the Upper Yukon CU we removed the estimated contribution of hatchery fish based on the count of hatchery fish at the Whitehorse Fishway.

Resulting estimates of spawner abundance by CU, summed across CU, and as previously estimated for the stock aggregate (Connors et al. 2022a) are provide in Table B.3. Annual aggregate estimates of spawner abundance from our multi-CU reconstructions and previous stock aggregate reconstructions (Connors et al. 2022a) were very similar and, on average, within 6% of each other (~9% in years prior to Eagle sonar, and ~3% in years thereafter). This provides some confidence that the CU-scale and aggregate analyses yield very similar reconstructions of the system. While the CU scale reconstructions were necessary to support our assessments of CU status and evaluation of consequences of alternative domestic fishery management measures, we caution against rolling them up to derive a stock aggregate brood table which we argue is instead more appropriately done using the integrated run-reconstruction and spawner-recruitment model that is used bilaterally under the Yukon River Salmon Agreement to derive annual estimates of total harvest, escapement and returns, by age, for Canadian-origin Yukon River Chinook salmon (JTC 2025).

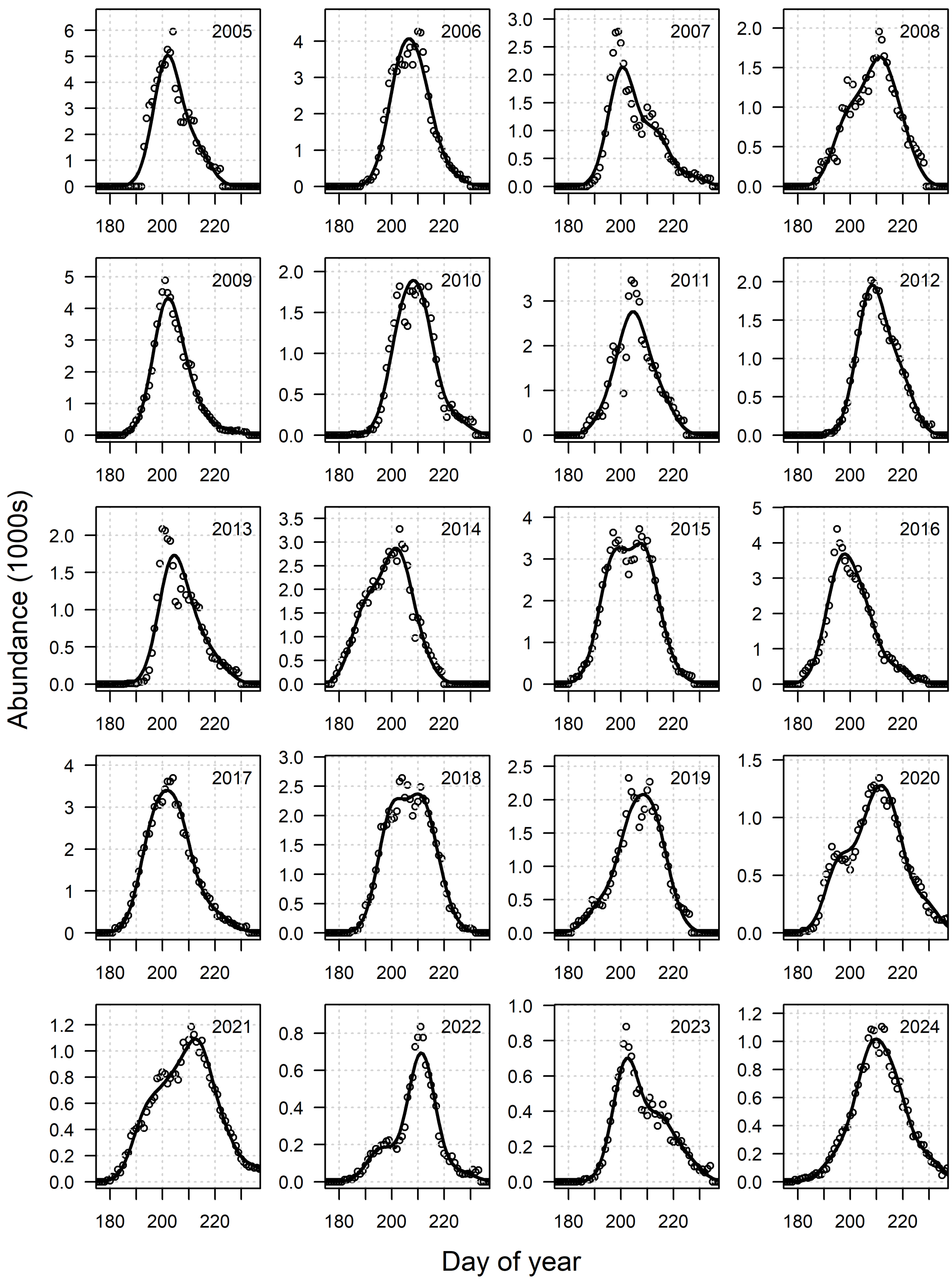

Figure B.2. Fits of the run-reconstruction model (lines) to daily sonar counts (circles) at Eagle, near the U.S. –Canada border.

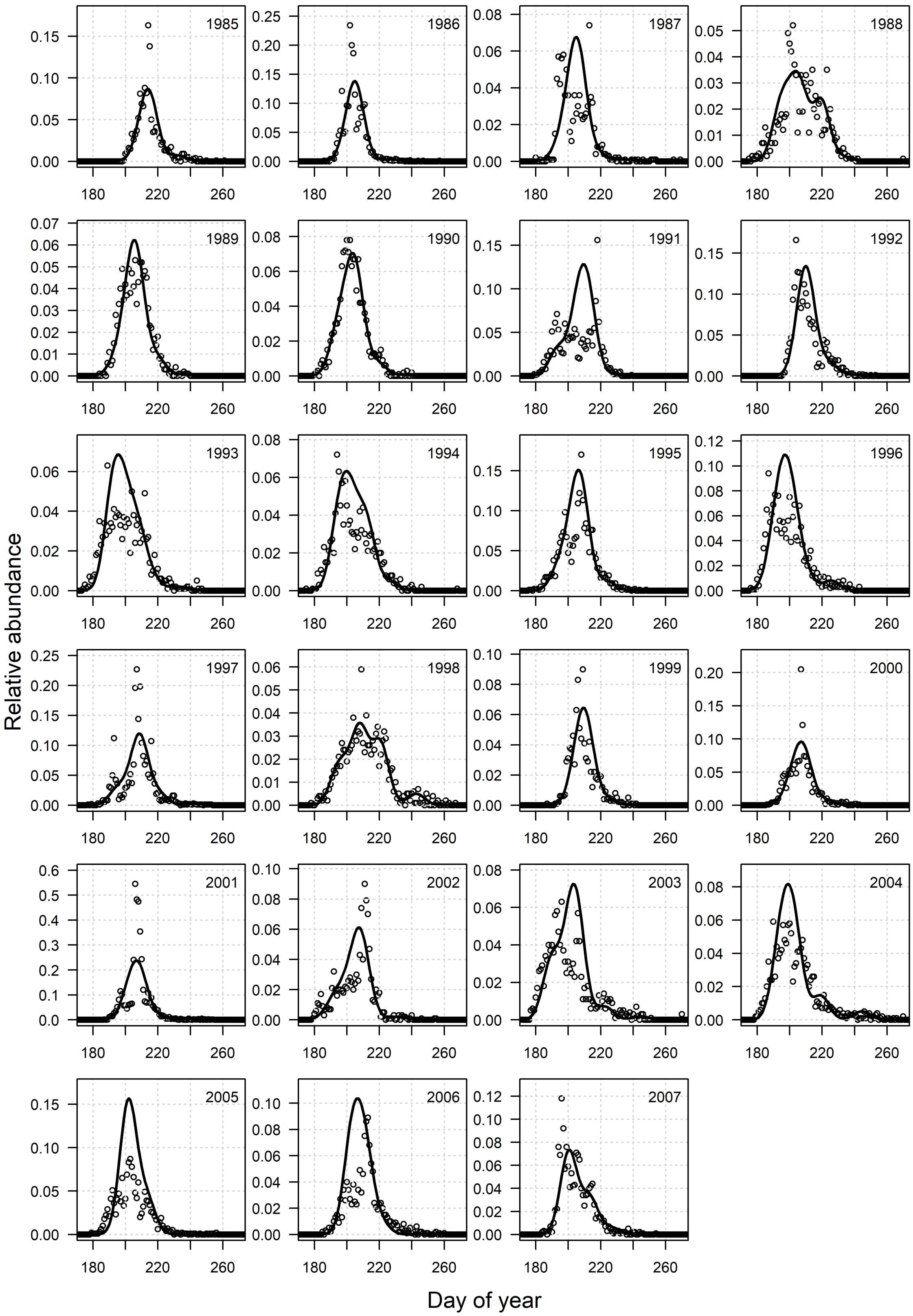

Figure B.3. Fits of the run-reconstruction model (lines) to daily fishwheel counts (circles) at White and Sheep Rocks, near the U.S. –Canada border.

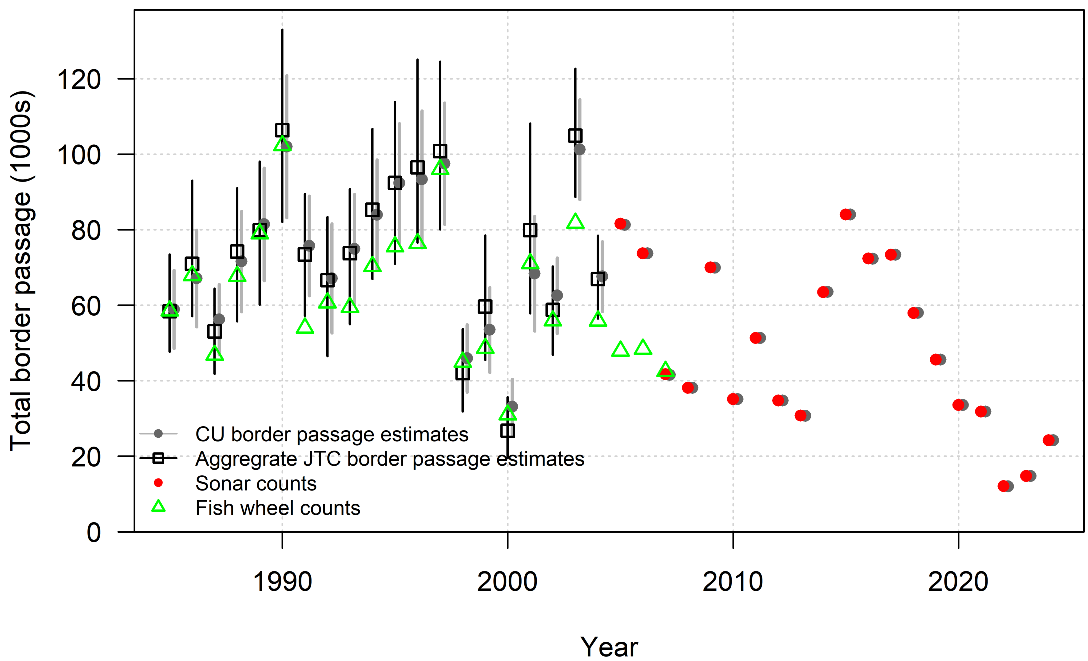

Figure B.4. Total border passage estimates based on the run-reconstruction model (grey circles with bars indicating the central 95% confidence interval), mark-recapture based estimates of run size from fish sampled from fish wheels after accounting for selectivity (open square), annual sum of daily fishwheel counts (scaled by estimated catchability, open green triangle) and the Eagle sonar (red circle).

Table B.3. Median estimates of spawner abundance by CU, summed across CU ("Aggregate"), and as previously estimated for the stock aggregate (Connors et al. 2022) but with data updated through 2024.

Conservation Unit or stock aggregate	Year	Spawners
NorthernYukonR.andtribs.	1985	1676
NorthernYukonR.andtribs.	1986	3260
NorthernYukonR.andtribs.	1987	3165
NorthernYukonR.andtribs.	1988	13420
NorthernYukonR.andtribs.	1989	5010
NorthernYukonR.andtribs.	1990	19833
NorthernYukonR.andtribs.	1991	9260
NorthernYukonR.andtribs.	1992	6526
NorthernYukonR.andtribs.	1993	13629
NorthernYukonR.andtribs.	1994	16192
NorthernYukonR.andtribs.	1995	9305
NorthernYukonR.andtribs.	1996	6123
NorthernYukonR.andtribs.	1997	8593
NorthernYukonR.andtribs.	1998	5615
NorthernYukonR.andtribs.	1999	4161
NorthernYukonR.andtribs.	2000	2475
NorthernYukonR.andtribs.	2001	6908
NorthernYukonR.andtribs.	2002	7248
NorthernYukonR.andtribs.	2003	18310
NorthernYukonR.andtribs.	2004	11265
NorthernYukonR.andtribs.	2005	9853
NorthernYukonR.andtribs.	2006	10127
NorthernYukonR.andtribs.	2007	5194
NorthernYukonR.andtribs.	2008	5197
NorthernYukonR.andtribs.	2009	6676
NorthernYukonR.andtribs.	2010	3913
NorthernYukonR.andtribs.	2011	3972
NorthernYukonR.andtribs.	2012	4214
NorthernYukonR.andtribs.	2013	429
NorthernYukonR.andtribs.	2014	4919
NorthernYukonR.andtribs.	2015	5303

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Conservation Unit or stock aggregate	Year	Spawners
NorthernYukonR.andtribs.	2016	6792
NorthernYukonR.andtribs.	2017	5860
NorthernYukonR.andtribs.	2018	4716
NorthernYukonR.andtribs.	2019	2263
NorthernYukonR.andtribs.	2020	1658
NorthernYukonR.andtribs.	2021	2877
NorthernYukonR.andtribs.	2022	508
NorthernYukonR.andtribs.	2023	2008
NorthernYukonR.andtribs.	2024	2478
Whiteandtribs.	1985	2246
Whiteandtribs.	1986	3987
Whiteandtribs.	1987	2722
Whiteandtribs.	1988	1757
Whiteandtribs.	1989	2361
Whiteandtribs.	1990	1725
Whiteandtribs.	1991	974
Whiteandtribs.	1992	3345
Whiteandtribs.	1993	10316
Whiteandtribs.	1994	6596
Whiteandtribs.	1995	4078
Whiteandtribs.	1996	10503
Whiteandtribs.	1997	4838
Whiteandtribs.	1998	2675
Whiteandtribs.	1999	3764
Whiteandtribs.	2000	2817
Whiteandtribs.	2001	4537
Whiteandtribs.	2002	2996
Whiteandtribs.	2003	5038
Whiteandtribs.	2004	3376
Whiteandtribs.	2005	13885

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Conservation Unit or stock aggregate	Year	Spawners
Whiteandtribs.	2006	1441
Whiteandtribs.	2007	3477
Whiteandtribs.	2008	2085
Whiteandtribs.	2009	3259
Whiteandtribs.	2010	1445
Whiteandtribs.	2011	2332
Whiteandtribs.	2012	2838
Whiteandtribs.	2013	157
Whiteandtribs.	2014	5028
Whiteandtribs.	2015	5530
Whiteandtribs.	2016	2612
Whiteandtribs.	2017	3323
Whiteandtribs.	2018	2730
Whiteandtribs.	2019	2396
Whiteandtribs.	2020	505
Whiteandtribs.	2021	1281
Whiteandtribs.	2022	2259
Whiteandtribs.	2023	1017
Whiteandtribs.	2024	767
Pelly	1985	7868
Pelly	1986	10456
Pelly	1987	12177
Pelly	1988	7144
Pelly	1989	18036
Pelly	1990	31697
Pelly	1991	11576
Pelly	1992	11260
Pelly	1993	8897
Pelly	1994	10950
Pelly	1995	8380

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Conservation Unit or stock aggregate	Year	Spawners
Pelly	1996	10004
Pelly	1997	9965
Pelly	1998	4207
Pelly	1999	6644
Pelly	2000	5172
Pelly	2001	10533
Pelly	2002	6562
Pelly	2003	22212
Pelly	2004	10771
Pelly	2005	11471
Pelly	2006	12487
Pelly	2007	8767
Pelly	2008	4277
Pelly	2009	15809
Pelly	2010	7499
Pelly	2011	8564
Pelly	2012	6465
Pelly	2013	11114
Pelly	2014	6780
Pelly	2015	16183
Pelly	2016	13475
Pelly	2017	8650
Pelly	2018	7344
Pelly	2019	4682
Pelly	2020	6407
Pelly	2021	5791
Pelly	2022	941
Pelly	2023	1898
Pelly	2024	1715
Stewart	1985	6164

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Conservation Unit or stock aggregate	Year	Spawners
Stewart	1986	5797
Stewart	1987	3481
Stewart	1988	3015
Stewart	1989	4218
Stewart	1990	5476
Stewart	1991	4952
Stewart	1992	3210
Stewart	1993	2262
Stewart	1994	5858
Stewart	1995	7614
Stewart	1996	4666
Stewart	1997	5994
Stewart	1998	2798
Stewart	1999	2711
Stewart	2000	1679
Stewart	2001	5052
Stewart	2002	4586
Stewart	2003	3609
Stewart	2004	3425
Stewart	2005	2595
Stewart	2006	6577
Stewart	2007	3210
Stewart	2008	2119
Stewart	2009	5498
Stewart	2010	2477
Stewart	2011	2180
Stewart	2012	877
Stewart	2013	3244
Stewart	2014	2682
Stewart	2015	5594

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Conservation Unit or stock aggregate	Year	Spawners
Stewart	2016	6304
Stewart	2017	3484
Stewart	2018	3081
Stewart	2019	3732
Stewart	2020	2183
Stewart	2021	3105
Stewart	2022	132
Stewart	2023	667
Stewart	2024	2769
Nordenskiold	1985	3112
Nordenskiold	1986	1138
Nordenskiold	1987	572
Nordenskiold	1988	1061
Nordenskiold	1989	608
Nordenskiold	1990	540
Nordenskiold	1991	539
Nordenskiold	1992	500
Nordenskiold	1993	1058
Nordenskiold	1994	2747
Nordenskiold	1995	4344
Nordenskiold	1996	3761
Nordenskiold	1997	1131
Nordenskiold	1998	2399
Nordenskiold	1999	1903
Nordenskiold	2000	1571
Nordenskiold	2001	718
Nordenskiold	2002	1254
Nordenskiold	2003	1459
Nordenskiold	2004	2366
Nordenskiold	2005	1167

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Conservation Unit or stock aggregate	Year	Spawners
Nordenskiold	2006	3848
Nordenskiold	2007	1978
Nordenskiold	2008	672
Nordenskiold	2009	1799
Nordenskiold	2010	2302
Nordenskiold	2011	623
Nordenskiold	2012	417
Nordenskiold	2013	1
Nordenskiold	2014	7481
Nordenskiold	2015	1354
Nordenskiold	2016	4867
Nordenskiold	2017	3443
Nordenskiold	2018	585
Nordenskiold	2019	3138
Nordenskiold	2020	303
Nordenskiold	2021	1330
Nordenskiold	2022	660
Nordenskiold	2023	1524
Nordenskiold	2024	2085
YukonR.Teslinheadwaters	1985	8320
YukonR.Teslinheadwaters	1986	9767
YukonR.Teslinheadwaters	1987	4298
YukonR.Teslinheadwaters	1988	2991
YukonR.Teslinheadwaters	1989	3544
YukonR.Teslinheadwaters	1990	4692
YukonR.Teslinheadwaters	1991	5902
YukonR.Teslinheadwaters	1992	9860
YukonR.Teslinheadwaters	1993	9235
YukonR.Teslinheadwaters	1994	5167
YukonR.Teslinheadwaters	1995	7756

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Conservation Unit or stock aggregate	Year	Spawners
YukonR.Teslinheadwaters	1996	12162
YukonR.Teslinheadwaters	1997	7146
YukonR.Teslinheadwaters	1998	3670
YukonR.Teslinheadwaters	1999	6440
YukonR.Teslinheadwaters	2000	2911
YukonR.Teslinheadwaters	2001	7971
YukonR.Teslinheadwaters	2002	1809
YukonR.Teslinheadwaters	2003	4031
YukonR.Teslinheadwaters	2004	3725
YukonR.Teslinheadwaters	2005	8850
YukonR.Teslinheadwaters	2006	1861
YukonR.Teslinheadwaters	2007	3844
YukonR.Teslinheadwaters	2008	4483
YukonR.Teslinheadwaters	2009	5642
YukonR.Teslinheadwaters	2010	6178
YukonR.Teslinheadwaters	2011	4493
YukonR.Teslinheadwaters	2012	3548
YukonR.Teslinheadwaters	2013	2999
YukonR.Teslinheadwaters	2014	6980
YukonR.Teslinheadwaters	2015	5876
YukonR.Teslinheadwaters	2016	13096
YukonR.Teslinheadwaters	2017	12513
YukonR.Teslinheadwaters	2018	7472
YukonR.Teslinheadwaters	2019	6482
YukonR.Teslinheadwaters	2020	7310
YukonR.Teslinheadwaters	2021	2282
YukonR.Teslinheadwaters	2022	353
YukonR.Teslinheadwaters	2023	3236
YukonR.Teslinheadwaters	2024	4104
MiddleYukonR.andtribs.	1985	9030

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Conservation Unit or stock aggregate	Year	Spawners
MiddleYukonR.andtribs.	1986	10178
MiddleYukonR.andtribs.	1987	10197
MiddleYukonR.andtribs.	1988	6829
MiddleYukonR.andtribs.	1989	18718
MiddleYukonR.andtribs.	1990	11553
MiddleYukonR.andtribs.	1991	15839
MiddleYukonR.andtribs.	1992	9462
MiddleYukonR.andtribs.	1993	9182
MiddleYukonR.andtribs.	1994	11941
MiddleYukonR.andtribs.	1995	21121
MiddleYukonR.andtribs.	1996	17020
MiddleYukonR.andtribs.	1997	25906
MiddleYukonR.andtribs.	1998	4617
MiddleYukonR.andtribs.	1999	9998
MiddleYukonR.andtribs.	2000	7480
MiddleYukonR.andtribs.	2001	14925
MiddleYukonR.andtribs.	2002	19464
MiddleYukonR.andtribs.	2003	22572
MiddleYukonR.andtribs.	2004	11525
MiddleYukonR.andtribs.	2005	14880
MiddleYukonR.andtribs.	2006	17898
MiddleYukonR.andtribs.	2007	7064
MiddleYukonR.andtribs.	2008	9885
MiddleYukonR.andtribs.	2009	12416
MiddleYukonR.andtribs.	2010	5380
MiddleYukonR.andtribs.	2011	11057
MiddleYukonR.andtribs.	2012	4087
MiddleYukonR.andtribs.	2013	4649
MiddleYukonR.andtribs.	2014	19104
MiddleYukonR.andtribs.	2015	26108

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Conservation Unit or stock aggregate	Year	Spawners
MiddleYukonR.andtribs.	2016	13637
MiddleYukonR.andtribs.	2017	18527
MiddleYukonR.andtribs.	2018	16179
MiddleYukonR.andtribs.	2019	11924
MiddleYukonR.andtribs.	2020	6882
MiddleYukonR.andtribs.	2021	7591
MiddleYukonR.andtribs.	2022	4460
MiddleYukonR.andtribs.	2023	2613
MiddleYukonR.andtribs.	2024	4790
UpperYukonR.	1985	2023
UpperYukonR.	1986	2509
UpperYukonR.	1987	1640
UpperYukonR.	1988	15815
UpperYukonR.	1989	6135
UpperYukonR.	1990	5217
UpperYukonR.	1991	2778
UpperYukonR.	1992	4089
UpperYukonR.	1993	2215
UpperYukonR.	1994	1813
UpperYukonR.	1995	2632
UpperYukonR.	1996	4421
UpperYukonR.	1997	8312
UpperYukonR.	1998	11919
UpperYukonR.	1999	2543
UpperYukonR.	2000	952
UpperYukonR.	2001	2022
UpperYukonR.	2002	1792
UpperYukonR.	2003	4382
UpperYukonR.	2004	5106
UpperYukonR.	2005	729

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Conservation Unit or stock aggregate	Year	Spawners
UpperYukonR.	2006	2906
UpperYukonR.	2007	301
UpperYukonR.	2008	1663
UpperYukonR.	2009	8850
UpperYukonR.	2010	895
UpperYukonR.	2011	8235
UpperYukonR.	2012	7999
UpperYukonR.	2013	3477
UpperYukonR.	2014	703
UpperYukonR.	2015	4543
UpperYukonR.	2016	2120
UpperYukonR.	2017	5514
UpperYukonR.	2018	5746
UpperYukonR.	2019	3030
UpperYukonR.	2020	3003
UpperYukonR.	2021	3772
UpperYukonR.	2022	614
UpperYukonR.	2023	374
UpperYukonR.	2024	3124
Big.Salmon	1985	4010
Big.Salmon	1986	4511
Big.Salmon	1987	4517
Big.Salmon	1988	3021
Big.Salmon	1989	8304
Big.Salmon	1990	5067
Big.Salmon	1991	7033
Big.Salmon	1992	4184
Big.Salmon	1993	4079
Big.Salmon	1994	5308
Big.Salmon	1995	9347

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Conservation Unit or stock aggregate	Year	Spawners
Big.Salmon	1996	7546
Big.Salmon	1997	11531
Big.Salmon	1998	2065
Big.Salmon	1999	4432
Big.Salmon	2000	3322
Big.Salmon	2001	6614
Big.Salmon	2002	8657
Big.Salmon	2003	10017
Big.Salmon	2004	5112
Big.Salmon	2005	5618
Big.Salmon	2006	7308
Big.Salmon	2007	4506
Big.Salmon	2008	1329
Big.Salmon	2009	9261
Big.Salmon	2010	3817
Big.Salmon	2011	5156
Big.Salmon	2012	2584
Big.Salmon	2013	3242
Big.Salmon	2014	6321
Big.Salmon	2015	10078
Big.Salmon	2016	6761
Big.Salmon	2017	5672
Big.Salmon	2018	5159
Big.Salmon	2019	3874
Big.Salmon	2020	1635
Big.Salmon	2021	1958
Big.Salmon	2022	1978
Big.Salmon	2023	1795
Big.Salmon	2024	1899
Aggregate	1985	44448

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Conservation Unit or stock aggregate	Year	Spawners
Aggregate	1986	51603
Aggregate	1987	42768
Aggregate	1988	55054
Aggregate	1989	66934
Aggregate	1990	85800
Aggregate	1991	58851
Aggregate	1992	52436
Aggregate	1993	60871
Aggregate	1994	66571
Aggregate	1995	74577
Aggregate	1996	76206
Aggregate	1997	83416
Aggregate	1998	39966
Aggregate	1999	42597
Aggregate	2000	28377
Aggregate	2001	59279
Aggregate	2002	54369
Aggregate	2003	91631
Aggregate	2004	56670
Aggregate	2005	69048
Aggregate	2006	64454
Aggregate	2007	38342
Aggregate	2008	31711
Aggregate	2009	69210
Aggregate	2010	33906
Aggregate	2011	46612
Aggregate	2012	33029
Aggregate	2013	29313
Aggregate	2014	59998
Aggregate	2015	80568

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Conservation Unit or stock aggregate	Year	Spawners
Aggregate	2016	69665
Aggregate	2017	66986
Aggregate	2018	53012
Aggregate	2019	41521
Aggregate	2020	29885
Aggregate	2021	29987
Aggregate	2022	11904
Aggregate	2023	15132
Aggregate	2024	23732
Aggregate (Connors et al. 2022)	1985	39082
Aggregate (Connors et al. 2022)	1986	49804
Aggregate (Connors et al. 2022)	1987	33795
Aggregate (Connors et al. 2022)	1988	49442
Aggregate (Connors et al. 2022)	1989	60545
Aggregate (Connors et al. 2022)	1990	85985
Aggregate (Connors et al. 2022)	1991	51662
Aggregate (Connors et al. 2022)	1992	46024
Aggregate (Connors et al. 2022)	1993	55346
Aggregate (Connors et al. 2022)	1994	64047
Aggregate (Connors et al. 2022)	1995	69485
Aggregate (Connors et al. 2022)	1996	76425
Aggregate (Connors et al. 2022)	1997	83707
Aggregate (Connors et al. 2022)	1998	35520
Aggregate (Connors et al. 2022)	1999	47002
Aggregate (Connors et al. 2022)	2000	21345
Aggregate (Connors et al. 2022)	2001	68219
Aggregate (Connors et al. 2022)	2002	48542
Aggregate (Connors et al. 2022)	2003	94909
Aggregate (Connors et al. 2022)	2004	55433
Aggregate (Connors et al. 2022)	2005	69335

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Conservation Unit or stock aggregate	Year	Spawners
Aggregate (Connors et al. 2022)	2006	64261
Aggregate (Connors et al. 2022)	2007	36229
Aggregate (Connors et al. 2022)	2008	33885
Aggregate (Connors et al. 2022)	2009	64971
Aggregate (Connors et al. 2022)	2010	32250
Aggregate (Connors et al. 2022)	2011	47008
Aggregate (Connors et al. 2022)	2012	32328
Aggregate (Connors et al. 2022)	2013	28488
Aggregate (Connors et al. 2022)	2014	63370
Aggregate (Connors et al. 2022)	2015	82695
Aggregate (Connors et al. 2022)	2016	69164
Aggregate (Connors et al. 2022)	2017	69387
Aggregate (Connors et al. 2022)	2018	54595
Aggregate (Connors et al. 2022)	2019	42193
Aggregate (Connors et al. 2022)	2020	30646
Aggregate (Connors et al. 2022)	2021	31509
Aggregate (Connors et al. 2022)	2022	12009
Aggregate (Connors et al. 2022)	2023	14576
Aggregate (Connors et al. 2022)	2024	24184

APPENDIX C. SPAWNER RECRUITMENT MODELS

To characterize Yukon River Chinook salmon CU spawner-recruitment relationships we fit individual age-structured state-space spawner-recruitment models (Fleischman et al. 2013) to all available data for each CU (i.e., time series of estimated catch, spawner abundance and age composition).

These sources of information, along with measures of observation variance, were externally derived from the multi-CU run-reconstructions described in Appendix A for spawner abundance and published estimates for catch and age composition (Connors et al. 2022a), updated through 2024.

An alternative to the sequential modelling approach we took would have been to integrate the run-reconstructions and spawner-recruitment analyses into a single model. However, we note that previous research has found that as long as the spawner-recruitment model carries forward uncertainty in input estimates from the run-reconstruction models, sequential analyses like those that we conducted (which still do not account for parameter covariance among models) produce very similar estimates of abundance, population dynamics parameters, and management reference points, both in terms of point estimates and uncertainty (Staton et al. 2017).

These models could have also been fit in a multi-population framework (Staton et al. 2020; Connors et al. 2020). However, these integrated models tend to yield very high estimates of recruitment variability and negative estimates of recruitment autocorrelation which appear to be spuriously too high and low, respectively, and which could lead to biases in other model parameters in manners that are difficult to predict (Staton et al. 2020).

Because the same model was fitted to each CU, all variables in the presentation of the state-space model actually have a j subscript appended to them to represent individual CUs. We omit this for the presentation here, both to simplify it and to emphasize that this approach fits to data from each CU separately.

C.1. PROCESS MODEL

The process model is intended to represent the true population dynamics (i.e., free of measurement error), and this component of our state-space spawner-recruitment model specifies productivity, density-dependence, and age-at-return by cohort (i.e. brood year, y).

Recruitment abundances of adult Chinook salmon (i.e., the total number of adults from a given brood year that will ever return, R_y) were treated as unobserved states and modeled as a function of spawner abundance in year y (S_y) assuming a Ricker (1954) spawner-recruitment relationship with serially auto-correlated log-normal process variation:

$$\ln(R_y) = \ln(S_y) + \ln(\alpha) - \beta S_y + v_y \tag{C.1}$$

where α is productivity (intrinsic rate of growth), β is the magnitude of within brood year density-dependent effects, and v_y reflects inter-annual variation in survival from egg to adulthood, which we term “recruitment anomalies”.

This variation was assumed to follow a lag-1 autoregressive process (correlation coefficient denoted by ϕ) over time:

$$\begin{aligned} v_y &= \phi v_{y-1} + \varepsilon_y \\ \varepsilon_y &\sim \mathcal{N}(0, \sigma_R) \end{aligned} \tag{C.2}$$

where ε_y reflects the portion of the recruitment anomaly v_y that is temporally independent (i.e., white noise).

The first seven years of recruitment states were not linked to observations of spawner abundance in the spawner-recruitment relationship (Equ. C.1) and were modeled as random draws from a log-normal distribution with mean $\ln(R_0)$ and variance σ_R^2.

Rather than estimating $\ln(R_0)$ as a free parameter as in Fleischman et al. (2013), we choose to follow Staton et al. (2020) and inform its value using the expected recruitment under equilibrium unfished conditions $\ln(\alpha)/\beta$.

The number of Chinook salmon that returned to spawn in year y at age a ($a \in$ 4:7) was the product of the total recruitment produced by spawning that occurred in brood year $y - a$ and the proportion from brood year $y - a$ that returned at age a:

$$N_{y,a} = R_{y-a} p_{y-a,a} \tag{C.3}$$

where $p_{y,a}$ is the probability that a fish spawned in brood year y will return at age a. Because we lacked empirical data on life stage-specific abundances (e.g., smolts, fish abundance of different ages/brood years in the ocean) from which to directly estimate maturation rates and life-stage specific mortality, the sole purpose of this probability of return-at-age model (Equ. C.3-C.4) is to apportion the abundance of total adult recruitment to the correct age and year of return.

We modeled brood year variation in age at return as Dirichlet random vectors drawn from a common hyperdistribution characterized by a mean age-at-maturity probability vector (π) and an inverse dispersion parameter ($1/D^2$):

$$p_{y,a} \overset{\text{iid}}{\sim} \mathcal{D}(\pi(\frac{1}{D^2})) \tag{C.4}$$

The total run in year y was made up of recruitments from multiple brood years as shown in Equ. C.3 and was calculated as the sum of the age-specific run abundances:

$$N_y = \sum_{a=4}^{7} N_{y,a} \tag{C.5}$$

Harvest in a given year (H_y) was modeled as the product of total run size and the harvest rate (U_y) experienced that year:

$$H_y = N_y U_y \tag{C.6}$$

and spawner abundance (S_y) was modeled as the portion of N_y remaining after harvest H_y:

$$S_y = N_y(1 - U_y) \tag{C.7}$$

It is this spawner abundance S_y (which is itself made up of fish of multiple ages from multiple brood years) that is used in Equ. C.1 to produce the recruitment abundance from spawning in brood year y.

C.2. OBSERVATION MODEL

We assumed a coefficient of variation (CV) for spawner observations that was based on the annual CV in estimated spawner abundances derived from the run reconstruction model (Appendix A).

Observed spawner abundance ($S_{obs,y}$) was therefore assumed to be log-normally distributed with the CVs converted to log-normal variance following Forbes et al. (2011):

$$\sigma_{o,y}^2 = \ln\left(\text{CV}_y^2 + 1\right) \tag{C.8}$$

Direct estimates of harvest by CU do not exist because most harvest occurs in mixed-stock fisheries in the mainstem of the Yukon River, and this harvest has never been apportioned back to CUs. Harvest observations ($H_{obs,y}$) were therefore approximated based on the stock aggregate harvest rate ($U_{obs,y}$, Connors et al. 2022a) and CU specific spawner abundance:

$$H_{obs,y} = S_{obs,y}(1 - U_{obs,y})^{-1}U_{obs,y} \tag{C.9}$$

and were assumed to have a CV equal to that of the joint variability in spawner abundance and aggregate harvest rate estimates which was log-normally distributed and converted to log-normal variance as per Equ. C.8.

Age composition by return year at the CU scale was available from fish passing from the U.S. into Canada most years from 2008 to present, but samples sizes are small because not all fish that were aged have corresponding genetic stock identification information, and vice-versa. In earlier years (i.e., pre 2008) it was often difficult to match individual scale samples used for ageing with scales that had genetic material extracted making any inference on CU age composition unreliable. In addition, while these sources of data provide insights into age composition of fish prior to spawning they may not be representative of total return age composition because harvest can be size selective and these samples are taken after the bulk of harvest occurs. We therefore made the simplifying assumption that estimates of age composition by return year, taken from an integrated run-reconstruction and spawner-recruitment model that explicitly takes age composition in harvest and spawners into account (Connors et al. 2022a), were a reasonable proxy for CU specific age compositions. We felt this was a reasonable assumption because CU specific border age composition from 2008 to present tended to be very similar among CUs (Figure D.1).

These age composition by return year estimates were assumed to be observed with multinomial sampling error, where uncertainty in age proportions in a given year was generated by specifying an “effective sample size” (ESS) of 50 pre-2007 and 100 thereafter. These ESSs are lower than the likely observed sample sizes to account for the non-independence of observation within a given sampling event (Maunder 2011) and were chosen to reflect the reduced confidence in age composition data prior to the Eagle test fishery sampling program which was standardized beginning in 2007.

C.3. TIME-VARYING PRODUCTIVITY

Given widespread evidence for changes in salmon productivity over time (Peterman and Dorner 2012; Malick and Cox 2016; Dorner et al. 2018), and interest in understanding the evidence for changing productivity in Yukon River Chinook salmon, we also fit a version of the model described in Equ. C.1-C.9 with time-varying intrinsic productivity (Table C.1). Specifically, we allowed the α parameter to evolve through time as a random walk, yielding time-varying estimates of productivity:

$$\begin{aligned} \alpha_y &= \alpha_{y-1} + \varepsilon_y \\ \varepsilon_y &\sim \mathcal{N}(0, \sigma_\alpha) \end{aligned} \tag{C.10}$$

where σ_α is the magnitude of process variation in productivity and where recruitment anomalies were no longer modeled as being auto-correlated but all other parameters in equation C.1 otherwise remained the same.

Table C.1. Estimates of key parameters from time-varying $alpha$ spawner-recruitment models by Conservation Unit along with two measures of parameter estimability and model convergence. Estimates of β are on the scale of 10,000 fish for ease of summarizing parameters in the table. For α and σ_α, which are time-varying parameters, values are averages of the posterior median estimates for each year along with the standard deviation among years.

Conservation Unit	Parameter	Mean (SD) or Posterior Median	Effective Sample Size (N_{eff})	Chain Mixing ($\widehat{R}$)
	α	1.3 (0.69)	3990	1.001
Big Salmon	β	1.72	2874	1.002
	σ_α	-0.2 (0.47)	9153	1.000
	α	1.14 (0.71)	3411	1.001
Middle Yukon R. and tribs.	β	0.63	1958	1.002
	σ_α	-0.15 (0.57)	7257	1.000
	α	2.06 (0.28)	4511	1.001
Nordenskiold	β	6.75	4653	1.000
	σ_α	0 (0.41)	8474	1.000
	α	1.31 (0.9)	5878	1.000
Northern Yukon R. and tribs.	β	1.41	4139	1.001
	σ_α	-0.19 (0.59)	9306	1.000
	α	0.94 (0.86)	6300	1.000
Pelly	β	0.74	4070	1.000
	σ_α	-0.23 (0.54)	12058	1.000
	α	1.42 (0.63)	5897	1.000
Stewart	β	2.76	4625	1.001
	σ_α	-0.18 (0.46)	13622	1.000
	α	1.23 (0.5)	5992	1.000
Upper Yukon R.	β	1.98	6659	1.000
	σ_α	-0.14 (0.4)	10320	1.000
	α	1.07 (0.74)	5099	1.001
White and tribs.	β	1.80	3271	1.001
	σ_α	-0.14 (0.51)	8349	1.000
	α	1.84 (0.5)	5112	1.000
Yukon R. Teslin Headwaters	β	2.25	3994	1.000
	σ_α	-0.16 (0.46)	10456	1.000

C.4. DEMOGRAPHIC CHANGE

The spawner-recruitment models described in the previous section assumed that demographic attributes of spawning escapement have not changed over time such that reproductive output is homogeneous among individuals and has been static over the past 40 years. However, changes in age, sex, and length-at-age of returning Chinook salmon, including in the Yukon (Lewis et al. 2015; Ohlberger et al. 2018, 2020), have been widely observed, suggesting concurrent declines in per capita reproductive output as smaller females carry disproportionately fewer eggs of a smaller mass compared to larger females (i.e., fecundity and total egg mass scale exponentially with female size) (Ohlberger et al. 2020). While changes in age, sex, and length-at-age (aka “escapement quality”) are not typically considered when deriving spawner-recruitment based reference points and setting escapement goals, previous research on Chinook salmon in the Yukon and Kuskokwim rivers of western Alaska suggests that failing to account for declines in escapement quality may result in underestimating the escapement needed to maximize long-term sustainable yield or recruitment (Staton et al. 2021; Connors et al. 2022a). In addition, simulation analyses suggest that accounting for the consequences of declining “escapement quality” in fishery management (via reference points) can reduce conservation risks relative to assessment frameworks that do not take demographic change into account (Ohlberger et al. 2025).

To evaluate the potential consequences of changes in escapement quality over time (which we operationalized as total egg mass) we translated time trends in sex ratios, age composition, and size-at-age to annual estimates of total egg mass production. We chose to focus on total egg mass instead of eggs because it it tightly correlated with egg energy content (Ohlberger et al. 2020) and may be a better proxy for female reproductive output than egg number. Large females produce not only more but also larger eggs compared to small females (Ohlberger et al. 2020).

Estimates of annual age and sex composition, and mean length were derived from records of individual fish sampled by fish wheels (pre-2007), and more recently by multi-mesh size gill net test fishery, near the U.S.–Canada border (ADF&G AYK Database Management System). These data were corrected for the known size selectivity of fish wheels using the length selectivity method described in Hamazaki (2018) .The allometric relationship between female Chinook salmon size and total egg mass that we assumed was based on 140 samples of female fish with paired measurements of length (mid-eye to tail fork; METF), egg count and egg (ovary) mass, based on collections in the gill net test fishery between 2008 and 2010 (Ohlberger et al. 2020).

With this alternative measure of reproductive potential we then refit our our spawner-recruitment models, replacing the spawner abundance term (S_y) in Equ. C.1 with:

$$Z_{m,y} = \sum_a S_y \cdot q_{y,a} \cdot w_{em,y,a} \tag{C.11}$$

where $Z_{m,y}$ is the year-specific total egg mass em, S_y is the year specific total spawner abundance, $q_{y,a}$is the year- and age-specific proportion females, and $w_{em,y,a}$ is the average total egg mass (em) per individual in each year (y) and age (a) (Table C.2).

This model makes several key assumptions including that total reproductive output is limited by females and that male abundance is always sufficient to fertilize all eggs (Staton et al. 2021). We believe these were reasonable assumptions across the range of spawner abundances that have been observed, and at the scale at which we are modelling spawner-recruitment dynamics, though at very low abundances and in small populations, male abundance will also presumably limit reproductive output. The model also assumes that egg mass suitably indexes within-stock density effects whereas total spawners (males and females) contribute to density effects in the base model formulation. For these reasons, plus necessary assumption that border age, size, and sex composition is a reasonable proxy for CU scale demographic characteristics, we consider these demographic models exploratory and encourage greater consideration of consequences of potentially violating these assumptions in future analyses.

Table C.2. *Estimates of key egg mass-recruitment relationship parameters by Conservation Unit along with two measures of parameter estimability and model convergence. Estimates of β are on the scale of 10,000 fish for ease of summarizing parameters in the table.*

Conservation Unit	Parameter	Median 95th percentiles)	Effective Sample Size (N_{eff})	Chain Mixing ($\widehat{R}$)
Big Salmon	α	0.005 (0.00027 , 0.02)	888	1.001
	β	0.003 (0.0014 , 0.0046)	887	1.004
	ϕ	0.87 (0.56 , 0.99)	1434	1.001
Middle Yukon R. and tribs.	α	0.0046 (0.00047 , 0.016)	1774	1.000
	β	0.00093 (0.00038 , 0.0016)	3012	1.000
	ϕ	0.8 (0.49 , 0.99)	2029	1.001
Nordenskiold	α	0.017 (0.0093 , 0.03)	765	1.004
	β	0.015 (0.011 , 0.019)	4782	1.000
	ϕ	0.26 (-0.068 , 0.72)	1371	1.000
Northern Yukon R. and tribs.	α	0.0038 (0.00029 , 0.017)	1397	1.000
	β	0.0019 (0.0012 , 0.0026)	5012	1.001
	ϕ	0.87 (0.66 , 0.99)	2191	1.000
Pelly	α	0.0035 (0.00021 , 0.016)	1085	1.003
	β	0.0014 (0.00079 , 0.002)	2883	1.000
	ϕ	0.89 (0.63 , 0.99)	2025	1.001
Stewart	α	0.0061 (0.0008 , 0.018)	953	1.000
	β	0.0044 (0.0017 , 0.0071)	2109	1.000
	ϕ	0.73 (0.37 , 0.97)	1729	1.003
Upper Yukon R.	α	0.0059 (0.0024 , 0.011)	1105	1.003
	β	0.0029 (0.0017 , 0.0042)	3994	1.000
	ϕ	0.46 (0.036 , 0.88)	1067	1.005

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Conservation Unit	Parameter	Median 95th percentiles)	Effective Sample Size (N_{eff})	Chain Mixing ($\widehat{R}$)
White and tribs.	α	0.0047 (0.0004 , 0.022)	1103	1.000
	β	0.0028 (0.0015 , 0.0042)	2925	1.001
	ϕ	0.85 (0.54 , 0.99)	2349	1.000
Yukon R. Teslin Headwaters	α	0.0067 (0.0011 , 0.019)	1211	1.002
	β	0.0026 (0.0011 , 0.0042)	2358	1.001
	ϕ	0.74 (0.39 , 0.98)	1978	1.001

C.5. MODEL FITTING AND DIAGNOSTICS

We fit the spawner-recruitment models in a Bayesian estimation framework with Stan (Carpenter et al. 2017; Stan Development Team 2023), which implements the No-U-Turn Hamiltonian Markov chain Monte Carlo (MCMC) algorithm (Hoffman and Gelman 2014) for Bayesian statistical inference to generate a joint posterior probability distribution of all unknowns in the model. Priors were generally uninformative or weakly informative and are summarized in Table C.3.

We sampled from 4 chains with 3000 iterations each and discarded the first 1000 as warm-up. We assessed chain convergence visually via trace plots and by ensuring that $\hat{R}$ (potential scale reduction factor, Vehtari et al. 2021) was generally less than 1.1 and that the effective sample size was greater than 400, or 10% of the iterations. Posterior predictive checks were also used to make sure that the model returned data similar to the data used to fit parameters. A summary of model diagnostics is available online here. Most models had leading parameters that were well estimated but the stationary spawner-recruitment model fit to data from the Teslin CU had relatively low effective sample size suggesting posterior samples were not well mixed with high autocorrelation. This was likely a result of the strong temporal pattern in recruitment residuals as evidenced by high effective samples sizes for the model that allowed for variation in intrinsic productivity over time.

Table C.3. Prior probability distributions for leading spawner-recruitment model parameters. Note that the egg mass and recruitment model shared the same priors except for $ln(\alpha)$ which was given a prior of $\sim N(-6, 2)$ with bounds $[-10, 5]$ because the parameter was on the scale of grams and not individual fish.

Parameter	Prior	Bounds	Description
$ln(\alpha)$	$\sim N(1, 2)$	$[0, \inf]$	Natural log of intrinsic rate of growth.
β	$\sim N(0, 1)$	$[0, \inf]$	Magnitude of within brood-year density-dependence
ϕ	$\sim U(-1, 1)$	$[-1, 1]$	Lag-one correlation in interannual variation in survival.
σ_R	$\sim N(0, 2)$	$[0, \inf]$	White noise process standard deviation in survival.
$ln(R_0)$	$\sim N(0, 20)$	$[0, \inf]$	Natural log of unobserved recruitment in the first year of process model.
σ_{R_0}	$\sim Inv - Gamma(2, 1)$	$[0, \inf]$	Standard deviation in unobserved recruitment in the first year of process model.
π	$\sim Dir(0.25, 0.25, 0.25, 0.25)$		Mean maturation-at-age probability for ages 4:7.
D	$\sim Beta(1, 1)$		Dispersion parameter that governs variability in maturation-at-age probabilities across cohorts.

C.6. BIOLOGICAL BENCHMARKS

We calculated biological benchmarks points for each MCMC sample and then summarized them across MCMC samples. The spawning abundance expected to maximize sustainable yield over the long-term under equilibrium conditions, S_{MSY} was derived as:

$$S_{MSY} = 1 - W(e^{1-ln(\alpha)})/\beta \tag{C.12}$$

where W is the Lambert function (Scheuerell 2016), and α and β are intrinsic productivity and the magnitude of within stock density dependence, respectively. We chose to apply this exact solution for S_{MSY} instead of the commonly applied Hilborn (1985) approximation because the approximation only holds for $0 < ln(\alpha) \leq 3$ such that infrequent, but large, posterior samples of α can result in biased estimates of the posterior distribution of S_{MSY}.

The spawner abundance expected to result in the stock rebuilding to S_{MSY} in one generation in the absence of fishing (S_{gen}, Holt 2009), often used as a lower biological benchmarks for CUs under the Wild Salmon Policy, was solved numerically according to:

$$S_{MSY} = S_{gen} \alpha e^{-\beta S_{gen}} \tag{C.13}$$

The spawner abundance expected to maximize recruitment over the long-term under equilibrium conditions (S_{MSR}, also commonly referred to as S_{MAX}) was estimated as:

$$S_{MSR} = \frac{1}{\beta} \tag{C.14}$$

Equilibrium spawner abundance (S_{eq}), where recruitment exactly replaces spawners, was estimated as:

$$S_{eq} = ln(\alpha)/\beta \tag{C.15}$$

The harvest rate expected to lead to maximum sustainable yield, U_{MSY}, was derived according to the solution proposed by Scheuerell (2016) as:

$$U_{MSY} = 1 - W(e^{1-ln(\alpha)}) \tag{C.16}$$

To derive biological benchmarks from the egg mass-recruitment models we used the yield-per-recruit algorithm described in Staton et al. (2021). Specifically, we calculated reproductive output-per-recruit (z_{PR}) under fishing mortality F as a function of the age and sex class (i) - specific reproductive output ($z_{i,j}$) in a given time block (j) weighted by the probability of returning by age and sex in that time block ($\omega_{i,j}$) and the probability of escaping harvest ($1 - U_{i,j}$):

$$z_{PR,j} = \sum_i (1 - U_{i,j}) z_{i,j} \omega_{i,j} \tag{C.17}$$

where $U_{i,j} = 1 - exp(-F)$ assuming all ages and sexes were equally vulnerbale to harvest, and time block j was either the first generation (1985-1990), all years (1985-2024), or the last generation (2019-2024) of observed spawners. We then could calculate equilibrium recruitment as:

$$R_{eq,j} = \frac{log(\alpha z_{PR,j})}{\beta z_{PR,j}} \tag{C.18}$$

Once the scale of the equilibrium population fished at F was obtained from Eqn. C.18, we calculated the age- or sex-structured total return, harvest, and spawning escapement:

$$\begin{aligned} N_{eq,i,j} &= R_{eq,j} \omega_{i,j} \\ H_{eq,i,j} &= N_{eq,j} U_{i,j} \\ S_{eq,i,j} &= N_{eq,j}(1 - U_{i,j}) \end{aligned} \tag{C.19}$$

and calculated totals for harvest and spawning escapement by summing over ages and sexes.

We then treated $H_{eq,i,j}$ or $R_{eq,i,j}$ as the objective value to maximize via iterative numerical search on the quantity F to estimate demographically based biological benchmarks $S_{MSY,j}$ or $S_{MSR,j}$ for a given time period, respectively.

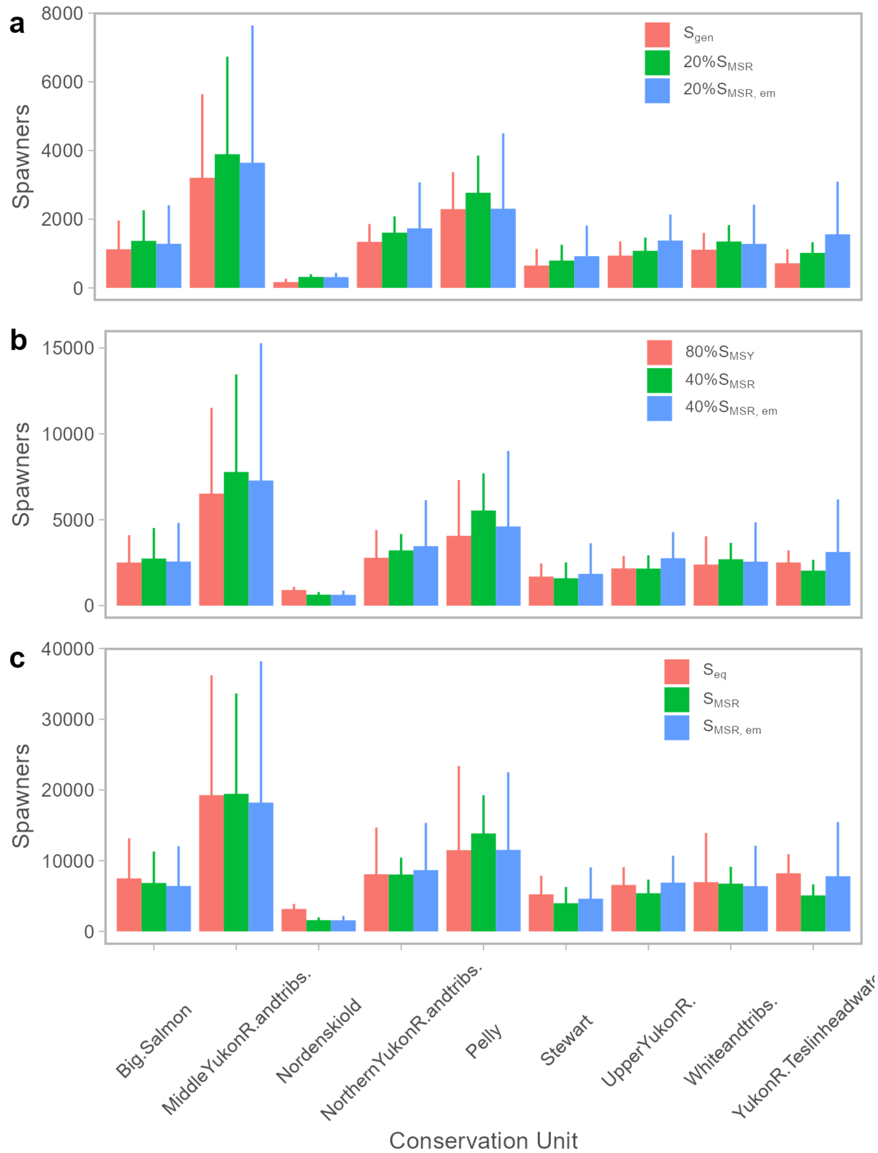

Figure C.1. Comparison of alternative (a) lower and (b) upper biological benchmarks as well as (c) estimates of maximum returns (S_{MSR}) or equilibrium population size (S_{EQ}) across CUs. The numeric value in legends refers to a percentage of the benchmark and those labelled "egg-mass" are derived from analyses that account for demographic change and are based on demographic characteristics observed over the most recent generation (2019-2024). Bars are medians and whiskers are upper 95th percentiles. Note than in instances where the intrinsic productivity of a CU is less than ~2.7 recruits-per-spawner equilibrium population size (S_{EQ}) occurs at lower spawner abundances than those associated with maximum recruits (S_{MSR})

APPENDIX D. SENSITIVITY ANALYSES

D.1. AGE COMPOSITION

Our analysis of CU spawner recruitment dynamics made the simplifying assumption that estimates of age composition by return year, taken from an stock aggregate run-reconstruction and spawner-recruitment model that explicitly takes age composition in harvest and spawners into account (Connors et al. 2022a), were a reasonable proxy for CU specific age compositions. We made this assumption because age composition information at the CU scale was unavailable because individual samples taken from fish for genetics at the border, and samples taken for ageing, could not be reliably linked in many years.

However, those years where age and genetic samples could be linked allow for a qualitative examination of the degree to which this simplifying assumption might be violated. For those years where there were 1000 or more linked genetic and age samples we compared age proportions by CU (Figure D.1). These comparisons illustrate that while age composition differed among some CUs in some years, overall it was broadly similar across CUs within years and there was no evidence of consistent over/under representation of age classes unique to a given CU. The exception to this was the Nordenskiold CU which is some years exhibited very different age composition, though we caution against overinterpreting this pattern because it is the smallest CU and often had 10 or less samples in a given year.

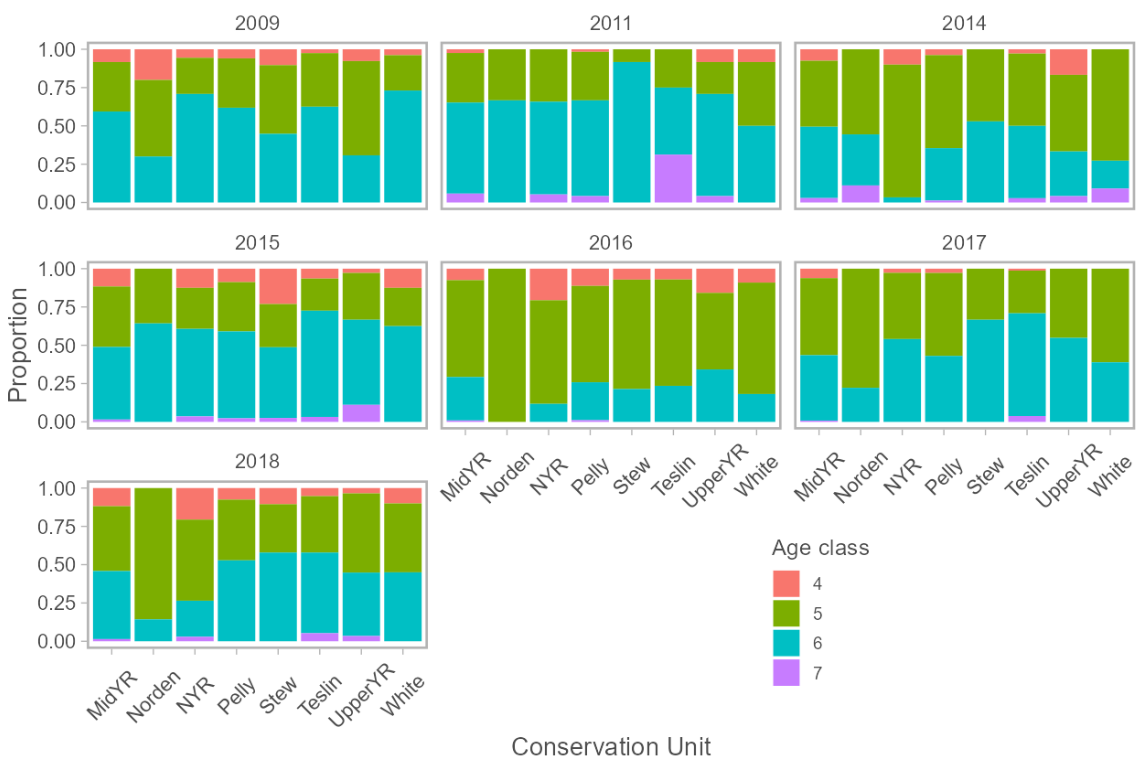

Figure D.1. Age composition in Chinook salmon sampled at the U.S.–Canada border, by Conservation Unit.

In addition to the comparison of stock aggregate and CU scale border age composition we also compared stock aggregate age composition to Blind Creek (Pelly CU) age composition based on samples taken from fish on the spawning grounds. This comparison allowed us to qualitatively evaluate the degree to which aggregate age composition estimates at the border were a reasonably proxy for age composition on the spawning grounds. This comparison highlights that for those years where a comparison could be made there was reasonable agreement between the two sources of data on age composition.

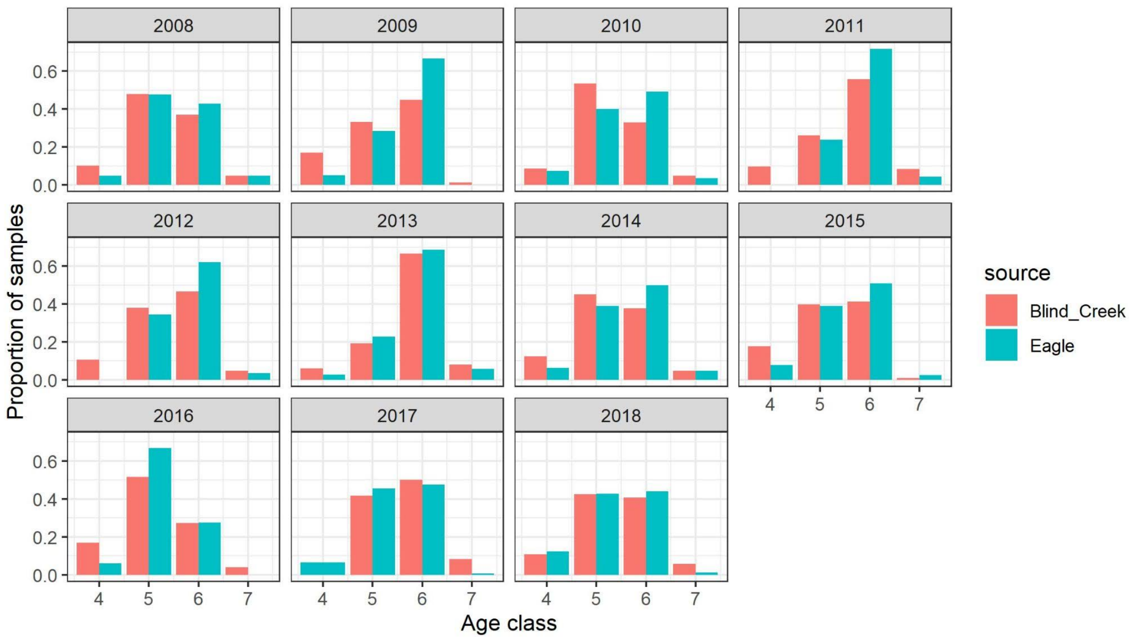

Figure D.2. Age composition of Chinook salmon sampled at the the Blind Creek weir (in Pelly CU) and those sampled at the U.S.–Canada border.

D.2. VULNERABILITY TO HARVEST

Our reconstructions of CU scale harvests, and subsequent population dynamics modelling and estimation of biological benchmarks, relied on the assumption that all CUs have historically experienced the same harvest rates. While this was a necessary assumption due to a lack of routine genetic assignment of fish sampled in fisheries to CUs, a multi-year study of genetic composition of fish harvested in the Alaskan portion of the Yukon in the mid to late 2000s allowed us to qualitatively examine this assumption. From 2005 to 2010 Chinook salmon were sampled in commercial and subsistence fisheries in the Alaskan portion of the Yukon River and estimates of genetic stock composition were derived for stock groupings that consisted of either individual or groups of CUs (e.g., DeCovich and Howard 2011). For each year, and river district, we calculated harvest by CU group as the product of total Canadian-origin Yukon River Chinook harvest and CU group composition (C.9). We then summed these estimates across districts to calculate the total CU group composition for Canadian-origin Yukon River Chinook harvested in Alaska for a given year, and compared it to estimates of CU group composition at the U.S.–Canada border based on the border passage reconstructions described in this document (Appendix B).

For these few years where genetic sampling of harvest in Alaska allowed for a comparison of CU composition in both border passage and harvest there was no evidence of consistent over/under-representation of individual CUs in Alaskan harvest, though in some years for some CU groups there were qualitative differences in representation (Figure D.3). This suggests that overall, across years with data, CUs were generally equally vulnerable to harvest in Alaskan fisheries.

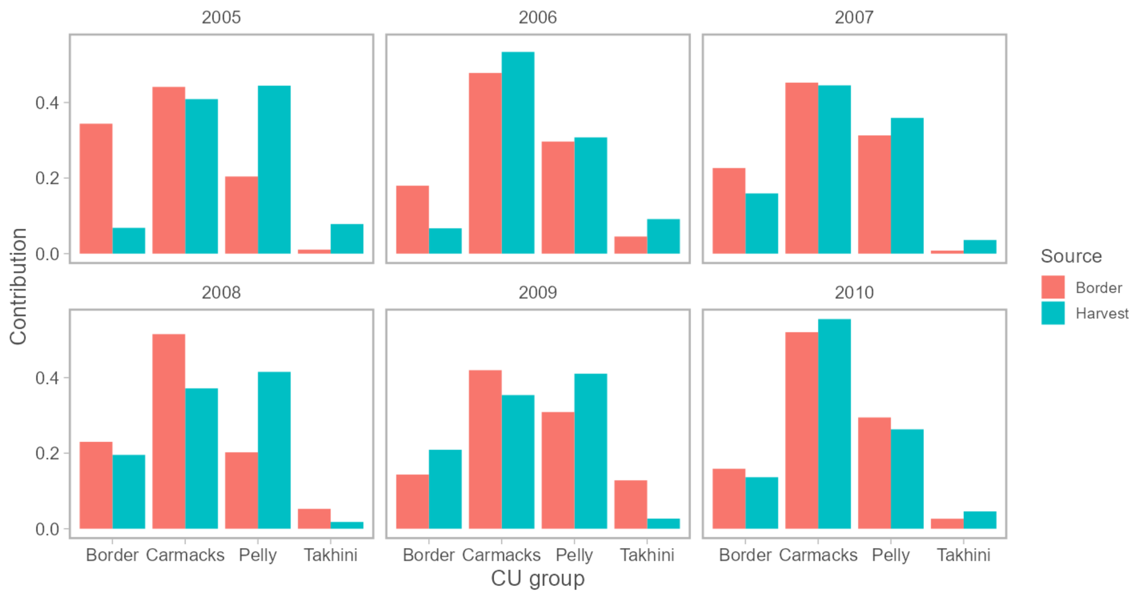

Figure D.3. Conservation Unit group composition of Yukon River Chinook salmon sampled in commercial and subsistence fisheries in Alaska (“Harvest”) and fish sampled in test fisheries at the U.S.–Canada border (“Border”). The “Border” CU grouping includes the Northern Yukon and White and tributaries CUs, the “Pelly” grouping includes the Pelly and Stewart CUs, the “Carmacks” grouping includes the Nordenskiold, Middle Yukon, Big Salmon and Teslin CUs, and the “Takhini” grouping includes the Upper Yukon CU.

APPENDIX E. COMPUTING ENVIRONMENT

This document aims to be transparent and reproducible. All data and code to reproduce the analysis in the report, and generate it, is available in the GitHub repository.

A document describing model diagnostics and some additional figures can be found in Supplement B.

R packages, versions, and dependencies necessary to recreate our analysis and documentation are tracked by the 'renv' package (Ushey and Wickham 2025) in the 'renv.lock' file. This ensures full reproducibility by allowing others to recreate the R environment where the analyses herein were run.