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Estimates of a Biologically-Based Spawning Goal and Management Benchmarks for the Canadian-Origin Taku River Sockeye Salmon Stock Aggregate

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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 purpose of this paper is to identify a range of spawning escapements that would likely result in maximum sustained yields and identify the appropriate biological benchmarks (management reference points) for management of the Canadian-origin Taku River Sockeye salmon *Oncorhynchus nerka* stock aggregate. A Bayesian state-space Ricker model that included age-structure and a one year-lag autoregressive component was fit to 1980–2018 data for Taku River Sockeye salmon greater than 349 mm mid eye to fork length. Data for the state-space model included:

1. estimates of harvest of naturally-spawned and enhanced (hatchery-produced) Sockeye salmon above and below the U.S./Canada border in the lower Taku River;
2. pooled Petersen capture-recapture estimates of above-border abundance; and
3. weighted age composition estimates of Taku River Sockeye salmon harvested in the U.S. District 111 traditional commercial drift gillnet fishery and Sockeye salmon captured in the Canyon Island fish wheels in the lower Taku River.

Coefficients of variation were also associated with these data sources. Historical annual terminal run abundance and inriver run abundance, spawning abundance, stock-recruitment parameters, and biological benchmarks were estimated from this model. The median estimate of spawner abundance that maximizes sustained yield, S_{MSY}, was 43,857 fish. A sensitivity analysis on the beta prior of the Ricker model concluded that a uniform distribution produced similar median estimates of key model outputs and biological reference points as the normal distribution prior, although the computation time was greatly increased with the uniform prior. Likewise, a normal prior on beta that was not constrained to be greater than 1.00 x 10^{-6} greatly reduced the precision on the reference points, but produced similar median estimates of key model outputs. A sensitivity analysis on the early years (1980–1983) concluded that uncertainty in the early years of model data may bias the estimate of S_{MSY} low. Based on the analyses from the state-space model, consideration for the uncertainty in the stock-recruit curve, and the minimal contrast within the time series, the recommendation from the Taku River Sockeye Salmon Working Group is a biological escapement goal range of 40,000–75,000 naturally-spawned fish. This range has a greater than 50% probability of achieving at least 70% of maximum sustained yield at the lower and upper bounds, and minimizes the risk of overfishing (less than a 10% probability of overfishing the stock at lower bound if optimal yield based on 80% or more of maximum sustained yield).

1 INTRODUCTION

1.1 SCOPE OF THIS PAPER

1.1.1 Purpose

The purpose of this paper is to provide science advice with respect to development of a biologically-based spawning escapement goal for the Canadian-origin Taku River Sockeye salmon *Oncorhynchus nerka* stock aggregate. The escapement goal is to be based on maximum sustained yield (MSY) and to have biological benchmarks that are consistent with Fisheries and Oceans Canada's (DFO's) Precautionary Approach and Wild Salmon Policy, and the Alaska Sustainable Salmon Fisheries Policy (Alaska Board of Fisheries' regulations, the *Policy for the Management of Sustainable Salmon Fisheries*: 5 AAC 39.222 and the *Policy for Statewide Salmon Escapement Goals*: 5 AAC 39.223). Terminal harvests and above-border inriver run estimates spanning nearly four decades form the basis of this work. Advice regarding biological benchmarks will contribute to a future assessment of status to meet Canada's Precautionary Approach and Wild Salmon Policy (see DFO 2005; Holt 2009; Grant *et al.* 2011) commitments and the Alaska Sustainable Salmon Fisheries Policy (Munro 2019) commitments.

The specific objectives of this work are to:

Objective 1: Identify the range of spawning escapements that would produce MSY for the Taku River Sockeye salmon stock aggregate; and

Objective 2: Identify the appropriate biological benchmarks for management of the Taku River Sockeye salmon stock aggregate.

These tasks primarily emerged from an obligation in the most recent provisions of Chapter 1, Annex IV of the Pacific Salmon Treaty that calls for the development of a bilaterally-agreed MSY escapement goal prior to the 2020 fishing season. Paragraph 3(b)(i) states:

> (B) "The Parties shall develop a joint technical report and submit it through the Parties' respective review mechanisms with the aim of establishing a bilaterally-approved maximum sustainable yield (MSY) goal for Taku River Sockeye salmon prior to the 2020 fishing season."

The Transboundary Panel requires escapement goal recommendations at the aggregate level to support the management and stock assessment regime that has been developed by the Parties through the joint Transboundary Technical Committee (TTC). The historical spawning escapement objective, established in 1985, was 71,000 to 80,000 fish with a point goal of 75,000 fish. In February 2019, a revised "interim" objective, based on the historical objective, but adjusted downward by 22% to account for historical dropout rates observed through radiotelemetry studies conducted in 1984, 2015, 2017, and 2018 (TTC 2019a), was agreed to by the Parties for the 2019 fishing season. The interim goal specified was 55,000 to 62,000 fish with a management target of 59,000 fish.

These two objectives and a concurrent review of the Taku River Sockeye salmon stock assessment program were conducted by the Taku River Sockeye Salmon Working Group (Taku Working Group), which included staff from DFO, the Alaska Department of Fish and Game (ADF&G), the Taku River Tlingit First Nation, and mark-recapture specialists from both Canada and the U.S. Under direction from the Transboundary Panel, the Taku Working Group conducted a review of the Taku River Sockeye salmon stock assessment program to address an obligation identified in the Pacific Salmon Treaty Chapter 1 Annex IV Paragraph 3(b)(i).

(C) “The Taku River Sockeye salmon assessment program will be reviewed by two experts (one selected by each Party) in mark-recovery estimation techniques. The Parties shall instruct these experts to make a joint recommendation to the Parties concerning improvements to the existing program including how to address inherent mark-recovery assumptions with an aim to minimize potential bias prior to the 2020 fishing season.”

The process took two years and was supported by funding from the Northern Endowment Fund of the Pacific Salmon Commission.

The Taku River Sockeye salmon stocks are grouped into conservation units for Canadian domestic status assessments under DFO’s *Wild Salmon Policy*, but Taku River Sockeye salmon conservation units are currently aggregated into a single Taku River Sockeye salmon stock aggregate for management purposes (e.g., inseason run size, spawning escapement objective; *see section 1.2.4 Canada’s Wild Salmon Policy*). The analyses in this report are primarily at this aggregate stock level. All references to “border” in the document pertain to the U.S./Canada border on the lower Taku River. The capture-recapture abundance estimates of the aggregate stock exclude fish smaller than 350 mm as measured from mid-eye to fork-of-tail (MEF). In keeping with this, the escapement goal and associated benchmarks refer to naturally-spawned fish greater than 349 mm in MEF length.

1.1.2 Definitions

- Return: the aggregation of salmon over several years that represent the surviving adult offspring from a single brood year
- Run: the total number of mature salmon that migrate from ocean-rearing areas to spawn in freshwater areas in a given calendar year and includes fish from multiple brood years
- Catch: all fish caught, whether retained or released
- Harvest: all fish caught and retained
- Terminal run: abundance of Sockeye salmon entering the Taku River including marine harvest in the U.S. District 111 traditional commercial drift gillnet and the Amalga Harbor special harvest area purse seine fishery
- Inriver abundance: abundance of Sockeye salmon passing the Canyon Island assessment site into Canada
- Dropout: any fish tagged at the Canyon Island fish wheels that did not cross the border; this includes mortality of marked fish due to predation, fish spawning below the border, or due to capture, handling, and tagging at the Canyon Island fish wheels
- Stock: group of Sockeye salmon in a spawning area

1.1.3 Analytical Approach

Taku River Sockeye salmon spawner-recruit data were analyzed using a Bayesian state-space Ricker spawner-recruit model (Ricker 1954) that included age-structure and a one year-lag autoregressive component. State-space models are time series models that feature both observed variables and unobserved states. Use of a Bayesian age-structured state-space model allows for consideration of process variation (natural fluctuations) in stock productivity, recruitment, and age-at-maturation independently from observation error (uncertainty in measurements of observed data) in run size, harvest, and age composition and allows for missing data. Such models have been used with increasing frequency in place of traditional methods in spawner-recruit analysis of Pacific salmon (e.g., Bernard and Jones III 2010; Schmidt and Evans 2010; Eggers and Bernard 2011; Fleischman *et al.* 2013; Fleischman and Reimer 2017). Biological benchmarks (e.g., 80% of S_{MSY}, S_{MSY}, S_{MAX}, S_{GEN}, S_{EQ}) were estimated based on samples of posterior distributions. Sensitivity analyses of the prior on beta and of the early years of data (1980–1983) were also explored.

1.1.4 Brief Overview of the Taku River Watershed

The Taku River is a transboundary river system originating in the Stikine Plateau of northwestern British Columbia. The merging of two principal tributaries, the Inklin and Nakina rivers, approximately 50 km upstream from the border, forms the mainstem of the Taku River. The river flows southwest from this point through the Coast Mountain Range eventually draining into Taku Inlet in Southeast Alaska, about 30 km east of Juneau (Subdistrict 111-32) (Figure 1). A majority of the 17,000 km^2 Taku River watershed lies within Canada (Neal *et al.* 2010). The river produces one of the largest runs of Sockeye salmon in northern British Columbia and Southeast Alaska and Sockeye salmon spawn throughout the drainage in both river and lake habitats.

The Taku River is turbid. Water discharge in the winter (November–March) ranges from approximately 49 to 196 m^3/s at the U.S. Geological Survey (USGS) water gauging station located on the lower Taku River near Canyon Island (USGS 2019a; 1988–2018). Discharge increases in April and May and reaches a maximum average flow of 890-1,000 m^3/s during June. Flow usually remains high in July but drops to approximately 500 m^3/s in late August. Sudden increases in discharge in the lower river result from a Jökulhlaup; release of the glacially impounded waters along the Tulsequah Glacier (Kerr 1948; Marcus 1960). These floods usually occur once or twice a year between June and September. During the floods, water levels fluctuate dramatically and the river carries a tremendous load of debris. Between 1987 and 2003, a majority of the annual peak floods from the Jökulhlaup occurred in August (53%) and since 2004 to 2018 only annual peak floods from the Jökulhlaup occurred in August with majority of the peaks occurring in July (53%) (USGS 2019b). During water years 1987 to 2018 the instantaneous peak flow due to a Jökulhlaup event was as high as 3,200 m^3/s (22 July, 2007; USGS 2019b).

1.1.5 Population Structure of Taku River Sockeye Salmon

The Canadian Taku River Sockeye salmon stock aggregate is currently described as five Sockeye salmon stocks. There are currently four main lake-type Sockeye salmon stocks within the Taku River drainage. Moving upstream from the mouth of Taku River, these stocks are King Salmon Lake, Kuthai Lake, Little Trapper Lake, and Tatsamenie Lake (Aaron Foos, DFO, Whitehorse, personal communication). Each of these stocks has individual monitoring and assessment (TTC 2019b). The remaining stocks are currently grouped together as river-type Sockeye salmon. These five stocks have been identified over the years through observations of life history, through scale pattern analysis (Heinl *et al.* 2014), and through genetic stock identification (Beacham *et al.* 2004; Rogers Olive *et al.* 2018).

1.1.6 Fisheries Harvesting Taku River Sockeye Salmon

Sockeye salmon returning to the Taku River drainage are primarily harvested in the U.S. District 111 traditional commercial drift gillnet fishery (hereafter referred to as D111 gillnet fishery) in Taku Inlet (Subdistrict 111-32) and in the inriver Canadian commercial fishery. Other harvests occur in the inriver U.S. personal use fishery, a test/assessment fishery and Canadian

Aboriginal food, social, and ceremonial fishery (hereafter referred to as Aboriginal fishery). The D111 gillnet fishery is managed primarily to target wild (naturally-spawned) Sockeye salmon, and Coho salmon, *O. kisutch,* and more recently, enhanced Chum salmon, *O. keta*, from local Alaska hatchery enhancement programs. In years of surplus production, Chinook salmon, *O. tshawytscha,* are also targeted. Pink salmon, *O. gorbuscha,* are also harvested and sold while incidentally caught Steelhead, *O. mykiss,* may not be sold. Non-terminal harvests of Taku River Sockeye salmon also occur in U.S. drift gillnet and purse seine fisheries.

For harvest stock assessment purposes, King Salmon Lake stock, Kuthai Lake stock, Little Trapper Lake stock and the Tatsatua stock are combined into one reporting group referred to as Taku Lakes, while Tatsamenie Lake is reported individually. All the remaining river-type stocks are grouped into one reporting group referred to as 'Mainstem.' This reporting group is not enumerated directly, but based on the difference between inriver abundance and the sum of the lake surveys

1.2 BENCHMARKS FOR PACIFIC SALMON

Two institutional frames of reference have emerged for Pacific Salmon in Canada and Alaska. Both relate back to a common biological frame of reference, but differ in how the biological information is used. Section 1.2.3 explains the Alaska Sustainable Salmon Fisheries Policy and Section 1.2.4 explains Canada's Wild Salmon Policy, and outline how biological information is used by the two agencies to determine management goals and evaluate status. There are conceptual differences, which are characterized as *frames of reference*. Section 1.2.5 compares some recent ADF&G and DFO reports which differ in terms of conceptual approach, definitions, and computational methods depending on the available data and policy setting.

1.2.1 Frames of Reference

The Taku River Sockeye salmon stock aggregate is jointly managed by DFO, ADF&G, and the Taku River Tlingit First Nation. The Pacific Salmon Commission, via the Pacific Salmon Treaty of 1985 (Treaty), commits Canada and the U.S. to conservation and allocation obligations for salmon originating in the waters of the Canadian portion of the Taku River. The Treaty mandates cooperative international management and has established conservation (via a spawning escapement goal) and harvest sharing (percentage sharing of the allowable catch) obligations for Taku River Sockeye salmon. Taku River Sockeye salmon are managed as an aggregate under provisions of Chapter 1, Annex IV of the Treaty. The historical spawning objective of 71,000 to 80,000 fish, with a point goal of 75,000 fish, was established in 1985 and was considered an "interim" objective since it was based on harvest and escapement data that were very limited at the time. For the 2019 fishing season, a revised "interim" objective of 55,000 to 62,000 fish and a management target of 59,000 fish were established by the Transboundary Panel of the Pacific Salmon Commission (TTC 2019a). The "interim" objective incorporates a 22% reduction to account for historical tag dropout rates observed through radiotelemetry studies completed in 1984, 2015, 2017 and 2018 which biased capture-recapture estimates high. The Parties are obligated under the Treaty to establish a bilaterally-approved MSY-based goal for Taku River Sockeye salmon prior to the 2020 fishing season (TTC 2019a).

DFO and ADF&G operate under similar policy frameworks and recent work related to biological benchmarks is conceptually consistent. However, there are important differences in both methodological details and subsequent use of the results.

1.2.2 Biological Parameters

The fundamental step is to fit a biological production model and estimate population parameters relating to productivity and capacity based on the shape of the fitted model. Biological benchmarks can then be directly calculated from samples of posterior distributions of parameters of the fitted state-space model. Methods and criteria were previously developed around Canada's Precautionary Approach and Wild Salmon Policy (e.g., Holt 2009, Holt *et al.* 2009, Grant *et al*. 2011) and the Alaska Sustainable Salmon Fisheries Policy (Munro 2019).

Biological Benchmarks include the following, all of which pertain to long-term trends:

- S_{MAX}: spawner abundance that maximizes the number of recruits;
- S_{MSY}: spawner abundance that maximizes sustained yield;
- S_{EQ}: spawner abundance that results in long-term equilibrium, whereby spawners equal the number of recruits;
- U_{MSY}: harvest rate (the proportion of the total run that is harvested) at maximum sustained yield;
- S_{GEN}: spawner abundance with a high probability of rebuilding to S_{MSY} in one generation in the absence of harvest; and
- 80%S_{MSY}: spawner level that is 80% of the spawner level that produces maximum sustained yield.

These benchmarks have widely accepted technical definitions and can be calculated independently of any management considerations. For example, S_{MSY} is always defined as the spawner abundance that maximizes sustained yield over the long-term (i.e., spawning abundance that produces the largest difference between spawner abundance and subsequent recruits), independently of how the stock is managed.

1.2.3 Alaska Sustainable Salmon Fisheries Policy

Two policies adopted into regulation by the Alaska Board of Fisheries (board): the *Policy for the Management of Sustainable Salmon Fisheries* (5 AAC 39.222) and the *Policy for Statewide Salmon Escapement Goals* (5 AAC 39.223) require a review of salmon escapement goals every three years that coincides with the regulatory cycle for each management area and provide process and criteria to be followed. The Alaska Sustainable Salmon Fisheries Policy defines three types of escapement goals that are set by the department (summarized from the *Policy for the Management of Sustainable Salmon Fisheries*: 5 AAC 39.222 under section (f)):

> *(3) "biological escapement goal" or "(BEG)" means the escapement that provides the greatest potential for maximum sustained yield; BEG will be the primary management objective for the escapement unless an optimal escapement or inriver run goal has been adopted; BEG will be developed from the best available biological information, and should be scientifically defensible on the basis of available biological information; BEG will be determined by the department and will be expressed as a range based on factors such as salmon stock productivity and data uncertainty; the department will seek to maintain evenly distributed salmon escapements within the bounds of a BEG;*
>
> *(25) "optimal escapement goal" or "(OEG)" means a specific management objective for salmon escapement that considers biological and allocative factors and may differ from the SEG or BEG; an OEG will be sustainable and may be expressed as a range with the lower bound above the level of SET, and will be adopted as a regulation by the board;*

the department will seek to maintain evenly distributed escapements within the bounds of the OEG;

(36) "sustainable escapement goal" or "(SEG)" means a level of escapement, indicated by an index or an escapement estimate, that is known to provide for sustained yield over a 5 to 10 year period, used in situations where a BEG cannot be estimated or managed for; the SEG is the primary management objective for the escapement, unless an optimal escapement or inriver run goal has been adopted by the board; the SEG will be developed from the best available biological information; and should be scientifically defensible on the basis of that information; the SEG will be determined by the department and will take into account data uncertainty and be stated as either a "SEG range" or "lower bound SEG"; the department will seek to maintain escapements within the bounds of the SEG range or above the level of a lower bound SEG; and

(39) "sustained escapement threshold" or "(SET)" means a threshold level of escapement, below which the ability of the salmon stock to sustain itself is jeopardized; in practice, SET can be estimated based on lower ranges of historical escapement levels, for which the salmon stock has consistently demonstrated the ability to sustain itself; the SET is lower than the lower bound of the BEG and lower than the lower bound of the SEG; the SET is established by the department in consultation with the board, as needed, for salmon stocks of management or conservation concern.

According to the Alaska Sustainable Salmon Fisheries Policy (summarized from the *Policy for the Management of Sustainable Salmon Fisheries*: 5 AAC 39.222 under section (c) (2)):

(B) "salmon escapement goals, whether sustainable escapement goals, biological escapement goals, optimal escapement goals, or inriver run goals, should be established in a manner consistent with sustained yield; unless otherwise directed, the department will manage Alaska's salmon fisheries, to the extent possible, for maximum sustained yield."

According to the Alaska Sustainable Salmon Fisheries Policy (summarized from the *Policy for the Management of Sustainable Salmon Fisheries*: 5 AAC 39.222 under section (f)):

(22) "maximum sustained yield" or "(MSY)" means the greatest average annual yield from a salmon stock; in practice, MSY is achieved when a level of escapement is maintained within a specific range on an annual basis, regardless of annual run strength; the achievement of MSY requires a high degree of management precision and scientific information regarding the relationship between salmon escapement and subsequent return; the concept of MSY should be interpreted in a broad ecosystem context to take into account species interactions, environmental changes, an array of ecosystem goods and services, and scientific uncertainty.

During its regulatory process, the board will review BEGs, SEGs, and SETs, and with the assistance of the department, determine the appropriateness of establishing an OEG. Although SETs are part of Alaska Sustainable Salmon Fisheries Policy, there are no SETs currently defined in Alaska (Munro 2019).

The Alaska Sustainable Salmon Fisheries Policy also defines three levels of concern that can be attributed to salmon stocks with escapement goals (summarized from the *Policy for the Management of Sustainable Salmon Fisheries*: 5 AAC 39.222 under section (f)):

(6) "conservation concern" means concern arising from a chronic inability, despite the use of specific management measures, to maintain escapements for a stock above a

sustained escapement threshold (SET); a conservation concern is more severe than a management concern;

(21) "management concern" means a concern arising from a chronic inability, despite use of specific management measures, to maintain escapements for a salmon stock within the bounds of the SEG, BEG, OEG, or other specified management objectives for the fishery; a management concern is not as severe as a conservation concern; and

(42) "yield concern" means a concern arising from a chronic inability, despite the use of specific management measures, to maintain expected yields, or harvestable surpluses, above a stock's escapement needs; a yield concern is less severe than a management concern, which is less severe than a conservation concern.

Numerous reviews of escapement goals have been completed since the Alaska Sustainable Salmon Fisheries Policy was formally adopted (e.g., Brannian *et al.* 2006; Clark *et al.* 2014; Heinl *et al.* 2017; Schaberg *et al.* 2019; Munro 2019).

1.2.4 Canada's Wild Salmon Policy

After a long development and consultation process (Irvine 2009), DFO released *Canada's Policy for Conservation of Wild Salmon* (Wild Salmon Policy) in 2005 (DFO 2005). The policy identifies six strategies and outlines action steps for implementing each strategy. The strategies can be summarized as:

1. Determine population status;
2. Determine habitat status;
3. Incorporate ecosystem considerations;
4. Establish collaborative strategic planning processes;
5. Include outcomes from 1–4 in annual implementation; and
6. Regularly review performance.

Strategy one, which describes the requirements for standardized monitoring of naturally-spawned salmon status is the most relevant to the work presented here on biological benchmarks for Taku River Sockeye salmon. It has three action steps:

1. Identify conservation units;
2. Develop criteria to assess conservation units and identify benchmarks to represent biological status; and
3. Monitor and assess the status of conservation units.

Substantial work has been completed on all three steps.

Holtby and Ciruna (2007) developed a framework for identifying conservation units and presented an initial list of 420 conservation units covering all five salmon species in BC. DFO (2009) summarized the framework. The conservation unit delineations have since been revised as data are compiled and verified for the status assessments, but no updated master list of coastwide conservation units has been formally published at this time. Taku River Sockeye salmon are grouped into two major life history types (river-type and lake-type) and five conservation units. The river-type life history is one conservation unit (Table 49 of Holtby and Ciruna 2007) and the lake-type life history has four conservation units (Kuthai, Little Trapper/Trapper, Tatsamenie, and Tatsatua) (Table 50 of Holtby and Ciruna 2007). The Sockeye salmon genetic stock identification management reporting groups, recommended and agreed upon by the TTC in April 2013 (TTC 2019a), are Tatsamenie Lake, Taku Lakes Other, and Taku River-type (Waugh *et al.* 2015[1]).

Holt *et al.* (2009) developed a framework for status assessment based on a suite of metrics (e.g., relative abundance, short-term and long-term trends in abundance), as well as upper and lower benchmarks for each metric to identify three status zones (red/amber/green). For the relative abundance metric, the upper benchmark is set at 80% of S_{MSY} (80% S_{MSY}), and the lower benchmark at S_{GEN}, the spawner abundance that allows rebuilding to S_{MSY} in one generation in the absence of fishing (i.e., high probability that total adult recruits meet or exceed S_{MSY}). Holt (2009) documented the analyses behind the choice of benchmarks and Holt and Bradford (2011) further explored the properties of alternative biological benchmarks. Holt and Ogden (2013) introduced a software package for calculating the benchmarks and resulting values of the different status metrics.

Grant *et al.* (2011) and Brown *et al.* (2014a[2]) compiled available data for a large number of conservation units and presented Wild Salmon Policy status metrics, which were evaluated in subsequent expert workshops (Grant and Pestal 2012; Brown *et al.* 2014b[3]).

Key concepts emerging from this body of work were:

- Assessments under the Wild Salmon Policy focus on biological status. Yield considerations are explicitly excluded from this step but are part of integrated planning under strategy four.
- No single metric can reflect the many different aspects of status that are considered by experts, so status integration has been done in a large workshop format (Grant and Pestal 2012; Brown *et al.* 2014b[3]).
- Status benchmarks are fundamentally different from management reference points (e.g., Chaput *et al*. 2012, Holt and Irvine 2013), but management can choose to set them at the same value (e.g., fixed escapement policy with goal set to S_{MSY}).

1.2.5 Comparing the Two Frames of Reference

This section briefly summarizes how the Alaska Sustainable Salmon Fisheries Policy and Canada's Wild Salmon Policy align conceptually and in terms of recent implementation. To formally reconcile the two frames of reference for the management of transboundary stocks, a document summarizing guidelines for the development of biological benchmarks and management reference points for Canada/U.S. transboundary river stocks has been suggested in prior reports (Pestal and Johnston 2015; Section 5.4).

The terminology roughly matches up as follows:

[1] Waugh, B., P. Etherton, I. Boyce, and S. Stark, 2015. Stock Composition of Stikine Chinook and Sockeye (2010 & 2013) and Taku Sockeye (2009 & 2013) In-river Fisheries - Genetic Stock Identification Sample Analysis. Final Report. Unpublished report.

[2] Brown, G.S., S.J. Baillie, R.E. Bailey, J.R. Candy, C.A. Holt, C.K. Parken, G.P. Pestal, M.E. Thiess, and D.M. Willis. 2014a. Pre-COSEWIC review of southern British Columbia Chinook salmon (*Oncorhynchus tshawytscha*) conservation units, Part II: Data, analysis and synthesis. Unpublished DFO Canadian Science Advisory Secretariat working paper.

[3] Brown, G., M.E. Thiess, G. Pestal, C.A. Holt, and B. Patten. 2014b. Integrated biological status assessments under the Wild Salmon Policy using standardized metrics and expert judgement: southern British Columbia Chinook Salmon (*Oncorhynchus tshawytscha*) Conservation Units. Unpublished DFO Canadian Science Advisory Secretariat working paper.

- Alaska's OEGs are equivalent to Canada's Management Reference Points, because both incorporate socio-economic considerations and recognize practical constraints on implementation. The difference is that Management Reference Points could also be specified in terms of exploitation rate or run size. The development process differs between the two jurisdictions; OEGs are established by the Alaska Board of Fisheries and Management Reference Points are set by DFO as part of the annual *Integrated Fisheries Management Plan* after public consultation.
- Alaska's SEGs are equivalent to Canada's Interim Management Escapement Goals, because both are often based on percentiles of observed spawner abundance.
- Alaska's SET has a similar intention as Canada's S_{GEN}, because both are designed to flag serious conservation concerns. The difference is that S_{GEN} has a formal quantitative definition and has been tested for robustness (e.g., Holt 2009; Holt and Bradford 2011), whereas the specific choice and justification of an SET is left open on a case by case basis.

Both agencies have delineated spawner abundance ranges based on biological benchmarks, and in both cases S_{MSY} serves as the anchor point. However, interpretation differs substantially (Table 1). Under the Alaska Sustainable Salmon Fisheries Policy, S_{MSY} is the basis for setting a BEG and the starting point for choosing an OEG once a BEG has been developed. Under Canada's Wild Salmon Policy, S_{MSY} is used to delineate three status zones (green > 80% S_{MSY}, red < S_{GEN}, amber in between) for one of several status indicators used in an integrated assessment. Canada's Wild Salmon Policy does not specify how management goals should be set in relation to S_{MSY} or the three status zones, because status assessments look back at observed patterns, and harvest rules are designed around plausible future scenarios. Due to these differences in policy frameworks, recent implementation by the two agencies has diverged (Table 2).

The key differences in policy frameworks are:

- Recent ADF&G analyses include yield profiles as a standard part of the results (e.g., Hamazaki and Conitz 2015; Fleischman and Reimer 2017), but yield-related information was not part of the Wild Salmon Policy case studies for Fraser Sockeye salmon and Southern BC Chinook salmon, which focused on metrics of biological status, including lower and upper benchmarks for *Relative Abundance* (Grant *et al.* 2011; Grant and Pestal 2012; Brown *et al*. 2014a[2]; Brown *et al.* 2014b[3]);
- ADF&G analyses commonly use a modified form of the Ricker model that accounts for autocorrelation in the residuals (e.g., Fleischman *et al.* 2011; Hamazaki and Conitz 2015; Fleischman and Reimer 2017), but this is not used in DFO reports, except for the theoretical explorations in Holt (2009);
- Several reports from both agencies use simple linear regressions with bootstrap. In some cases this is the only estimation method (e.g., Bernard *et al.* 2000; Tompkins *et al*. 2005; Fair *et al.* 2011), but others use them side-by-side with Markov Chain Monte Carlo (MCMC) to explore the robustness of results (e.g., McPherson *et al.* 2010; Holt and Ogden 2013).

In each case, the chosen approach depended on available data and the institutional frame of reference. The analysis for this paper was shaped by three considerations:

1. Where methods diverged (e.g., bias correction), the approach followed other transboundary stocks as the base case (e.g., Bernard *et al*. 2000 for Stikine Chinook salmon; Bernard and Jones III 2010 for Alsek Chinook salmon; McPherson *et al*. 2010 for Taku Chinook salmon; Eggers and Bernard 2011 for Alsek Sockeye salmon; and Pestal and Johnston 2015 for Taku Coho salmon).

2. Given the transboundary management system for Taku River Sockeye salmon, information required by both agencies was presented for their individual and joint planning processes.

3. As Bayesian models have been used with increasing frequency in place of more traditional methods in spawner-recruit analysis of Pacific salmon, Taku River Sockeye salmon spawner-recruit data were analyzed using a Bayesian age-structured state-space model. MCMC methods were employed within the R environment (R Core Team 2019; version 3.6.0) using the packages *rjags* (Plummer 2019) and *R2jags* (Su and Yajima 2015). The packages *rjags* and *R2jags* provide an interface between Program R and the software JAGS (Just Another Gibbs Sampler for Bayesian data analysis).

2 METHODS

Data for the state-space spawner-recruit model included:

1. estimates of harvest of naturally-spawned and enhanced (hatchery-produced) Sockeye salmon above and below the border, and associated coefficients of variation (CVs);
2. pooled Petersen capture-recapture estimates of above-border abundance, and associated CVs; and
3. weighted age composition estimates of D111 gillnet fishery harvests and lower Taku River (Canyon Island) fish wheel catches (Appendices A1, A2, A3).

Sources of these data components are described in the following sections. More detailed explanation of the data compilation and verification of the data sources is found in Pestal *et al.* (2020).

2.1 DATA

2.1.1 Data Sources

Harvest Data

Directed fisheries on Taku River Sockeye salmon occur in the Taku River drainage in Canada (above-border harvest), and in the U.S. inriver personal use fishery (below-border harvest), as well as in the marine water of the D111 gillnet fishery (below-border harvest). Harvest included in this analysis is only from directed fisheries within terminal marine areas (U.S. District 111), which includes Taku River Sockeye salmon harvested incidentally in the hatchery purse seine fishery at Amalga Harbor in D111. The reporting requirements for harvest in both countries have been rigorous. U.S. salmon landings from individual commercial fishermen are recorded on fish tickets as catch in units of total weight. Total weight is then converted by processors into units of fish numbers, based on the individual processor’s method of determining the average weight of a fish. U.S. harvest must be reported to ADF&G within seven days of landing in the form of a fish ticket. The fish tickets are processed and edited promptly inseason and entered into an internal ADF&G Fish Ticket Database System. Likewise, Canadian harvest is reported daily to DFO on fish tickets which are verified and entered into an internal DFO database each day; although Canadian harvest is reported as individual fish on the fish tickets.

Below-border harvest

The D111 gillnet fishery harvests an average of 122,900 Sockeye salmon fish (1983–2017; TTC 2019b) with a minimum harvest of 25,800 fish in 1998 and a maximum harvest of 203,000 fish in 2001; first quartile 49,000 fish and third quartile 107,500 fish (1983–2017; TTC 2019b). Of the Sockeye salmon harvest in the D111 gillnet fishery, the average harvest of naturally-spawned

Taku River Sockeye salmon fish is 80,885 fish (TTC 2019b). Every season, the small U.S. inriver personal use fishery occurs mainly near Canyon Island and average harvest is approximately 1,100 Taku River Sockeye salmon.

U.S. domestic enhanced Sockeye salmon originating from Snettisham Hatchery in Port Snettisham (1994 to present) and several domestic naturally-spawned stocks (including Speel and Crescent lakes) contribute to the D111 gillnet fishery. An average of 29,000 U.S. domestic enhanced fish are harvested in the D111 fishery with a minimum harvest of 2,600 enhanced fish in 1994 and with a maximum harvest of 92,800 enhanced fish in 2001 (1994–2017; TTC 2019b). Transboundary enhanced Sockeye salmon of Tatsamenie, Trapper, and King Salmon lakes contribute to harvests in both U.S. and Canadian fisheries.

Downstream of the border in marine waters, the D111 gillnet fishery is sampled weekly to collect matched age, sex, and length (ASL), otolith, and genetic tissue samples (Table 3). On average 4,400 ASL samples are taken each year in the D111 gillnet fishery (1982–2018). From 1986 to 2011, scale pattern analysis, incidence of brain parasites, and otolith thermal mark data were used to determine stock composition of the D111 gillnet fishery using mixed-stock analysis (TTC 2009; Heinl *et. al* 2014). Since 2012, stock composition of harvest has been estimated through a mark- and age-enhanced genetic mixed-stock analysis (MAGMA) model, which is an extension of the Pella-Masuda genetic stock identification model (Pella and Masuda 2001) that incorporates paired otolith mark and age data. The MAGMA model outputs posterior distributions of stock composition, which incorporate uncertainty in the stock composition due to sampling error and fish misallocation. Contribution of hatchery-enhanced Taku River Sockeye salmon to personal use harvests are estimated using Canadian commercial fishery otolith samples.

Above-border harvest

Above the border, extending from the border to approximately 18 km upstream, Taku River Sockeye salmon are harvested in the inriver commercial, Aboriginal, and test/assessment fisheries; the majority of harvest occurs within 5 km of the border. Commercial fishing periods have ranged from zero to seven days per week, and are chosen weekly and inseason by fishery managers based on available stock assessment data. From 1984–2017, the Canadian commercial fishery harvested an average of 24,700 naturally-spawned and enhanced Taku River Sockeye salmon each year. The Canadian Aboriginal fishery harvests are monitored and reported to DFO by the Taku River Tlingit First Nation and have averaged less than 200 Taku River Sockeye salmon per year.

From 1986–2011, stock composition and contribution of enhanced Taku River Sockeye salmon to these fisheries was based on otolith thermal marks, scale pattern analysis, and brain parasites; since 2012, stock composition was estimated from genetic stock identification based on tissue samples collected from the inriver commercial gillnet fishery (TTC 2019b; Table 4). The commercial and test/assessment fisheries are sampled weekly. On average DFO samples about 2,200 Sockeye salmon per season from the Canadian commercial fishery for ASL samples, otoliths (non-matched), and genetic tissue (matched since 2018).

Capture-Recapture Abundance Estimates

Above-border Canadian-origin inriver Sockeye salmon abundance was estimated through capture–recapture studies conducted annually starting in 1984 (TTC 2019b). This data was compiled, cross-verified, and subjected to a battery of tests to identify sources of potential bias and compare alternative estimation approaches (Pestal *et al.* 2020*).* The primary objective of the capture–recapture project is to estimate the inriver abundance of Sockeye salmon above the border (Figure 1); therefore, run abundance is germane to the Canadian-origin Taku River

Sockeye salmon. Each year inriver run estimates are generated weekly over the run to inform salmon harvest management, and a final inriver run estimate is generated post-season. These data, along with harvest data, are used to reconstruct and estimate the annual terminal run of Taku River Sockeye salmon (TTC 2019b). Detailed summaries of the annual assessment results have been documented in periodic reports, initially in the ADF&G Regional Report Series and starting in 1998 in the Pacific Salmon Commission Technical Report series. The most recent published report is for the 2013 season (Boyce and Andel 2014).

First Event

Sockeye salmon are captured at fish wheels located at Canyon Island, in the lower Taku River, and fish greater than 349 mm MEF are tagged with spaghetti tags, and marked with secondary marks (e.g., fin clips) as the first of two sampling events (Bednarski *et al*. 2019). The two fish wheels are positioned in the vicinity of Canyon Island on opposite riverbanks, approximately 200 m apart. The Taku River channel at this location is ideal for fish wheel operation since the river is fully channelized through a relatively narrow canyon that has very steep walls. In 2016 and 2017 a third fish wheel was operated downriver from Canyon Island across from Yehring River.

From 1984 to 2017 fish wheels were generally sampled twice a day around 8:00 and 16:00, which included a holding time of over 12 hours. Several studies have documented adverse effects on fish captured and handled in fish wheels with extended holding times (Bromaghin and Underwood 2003; Cleary 2003; Underwood *et al.* 2004; Bromaghin *et al*. 2007; Liller *et al.* 2011). In order to reduce stress associated with fish wheel capture and tagging, in 2018, fish wheel methods were changed to hourly fish wheel sampling. General hours of operation were from 4:00 to 12:00 and from 16:00 to 22:00 (Andel *et al*. 2018).

Second Event

The second event (recapture) occurs upriver, above the border, in the inriver Canadian commercial and test/assessment (scientific) fisheries and consists of inspecting Sockeye salmon catches for tags and secondary marks (Bednarski *et al.* 2019). Spaghetti tag return is a condition of licence for all Canadian Commercial Fishing Licences, and field staff gather these from fishers on a daily basis and record individual tag numbers. Compliance is verified through inspection of harvested fish for tag scars. Tagged-to-untagged ratios of salmon caught in these Canadian inriver gillnet fisheries are used to develop estimates of the inriver abundance of Sockeye salmon.

Size Stratification and Dropout Rate Adjustments

Canadian-origin inriver Sockeye salmon abundance was estimated annually from 1984 to 2018 using data generated in capture–recapture studies and implemented within the R environment (R Core Team 2019; version 3.6.0) using the Bayesian Time Stratified Population Analysis System (*BTSPAS)* package with custom extensions (Schwarz 2006; Schwarz *et al.* 2009; Bonner and Schwarz 2020); although these estimates were Pooled Petersen estimates and were not implemented in a Bayesian framework.

Year-specific size-stratified Petersen capture-recapture estimates were only available for 2003–2018, as tag records could not be matched to size data for the earlier years (Appendix A3, Table 5). Size-stratified estimates simply apply the Pooled Petersen estimate twice, once for smaller fish and once for larger fish, then add the individual estimates. Data for these years were split into ‘small’ and ‘large’ fish based on the 30th percentile of the size distribution of the Canadian inriver commercial harvest in each year (Pestal *et al.* 2020). Size-stratified estimates compared to the simple Pooled Petersen estimates from 2003–2018 showed a consistent bias, with the size-stratified estimates about 6.4% smaller on average. Therefore, a downward adjustment of 6.4% was applied to non-stratified pooled Petersen capture-recapture abundance estimates in years when matched size data was not available (1984–2002; excluding 1986). (Appendix A3). Release and recapture data were unavailable for 1986, therefore an abundance estimate was not available.

All estimates (1984–2018, excluding 1986) were adjusted to account for dropout of tagged fish (Appendix A4). For years 2016 and prior, a weighted-average dropout rate of 25.5% was applied to the pooled Petersen capture-recapture abundance estimates. The dropout rate was based on a weighted average of results from radiotelemetry studies conducted in 1984, 2015, 2017, and 2018 (Appendix A4). Year-specific dropout rates were applied to 2017 (32.1%) and 2018 (14.6%) based on radiotelemetry studies conducted in those years. Capture-recapture abundance estimates used in this analysis do not include expansions based on fish wheel operations (i.e., catch per unit effort (CPUE)) (TTC 2019b; Table 5).

Expansion of Capture-Recapture Abundance Estimates

Some historical, published capture-recapture abundance estimates were expanded by the average cumulative proportion of fish wheel CPUE based on late installation of the fish wheel, early removal of the fish wheel, or low tag recovery and effort prior to or after the inriver commercial fishery. For example, in 1989, the fish wheels began operating on 5 May and the first sockeye was tagged on 31 May, but the Canadian test/assessment fishery did not start until 19 June and the Canadian commercial fishery operated one to four days per week beginning in late June (McGregor *et al.* 1991). Based on the capture-recapture study, it was estimated that 99,467 Sockeye salmon passed Canyon Island between 18 June and 25 September. This estimate was expanded to 114,068 using the 1989 cumulative proportion of fish wheel CPUE through 17 June (12.8%) to estimate the number of Sockeye salmon that had passed prior to the start of the Canadian inriver fishery and tag recapture efforts (McGregor *et al.* 1991). A similar occurrence arose in 1995 when the estimate was expanded by 0.8% to account for the number of Sockeye salmon that had passed before the period of the capture-recapture estimate (Kelley *et al.* 1997; TTC 1997). In 2010, fish wheel CPUE was used to expand the inriver capture-recapture abundance estimate for periods of low tag recovery and effort in statistical weeks 22-23 and 35-37. This increased the estimate from 103,257 fish to 109,028 fish (Andel and Boyce 2014). Yearly expansions ranged from no expansions in years 1998–2006, 2011, 2014, and 2016 to a maximum expansion of 12.8% in 1989 (Table 5; Appendix D15 in TTC 2019b). Historical estimates of escapement (the above border expanded capture-recapture abundance estimate minus the Canadian harvest above the border) and terminal run abundance (the above border expanded capture recapture abundance estimate plus U.S. harvest below the border) were based on the expanded capture-recapture abundance estimates (Appendix D15 in TTC 2019b). Although the yearly expansion factors are available and well documented, overall the reasons for the expansions along with the raw data associated with the expansions are not well documented for every year from 1984–2018. Therefore, these expansions were not applied to the updated capture-recapture abundance estimates.

The non-expanded, updated, size-stratified and adjusted for dropout Petersen capture-recapture abundance estimates were used in model as observed inriver run abundance (see equation 8). The difference between inriver run abundance and the total harvest of Taku River Sockeye salmon above the border is the spawning escapement (see equation 9).

Age Composition Data

Age compositions were estimated from a weighted combination of age data from the D111 gillnet fishery harvest of Sockeye salmon in Taku Inlet and age data from the Canyon Island fish wheels (see Section 2.2.2.1 Age Composition) (Table 3 and Table 6). Scale pattern analysis to identify the stock composition of the D111 gillnet fishery started in 1986. Therefore, only age data from the Canyon Island fish wheels was used to estimates age composition for years

1983–1985. The age composition from the fish wheels was weighted by the pooled Petersen capture-recapture abundance estimates, therefore 1986 was only based on the stock-specific harvest age composition (since there was no capture-recapture abundance estimate for 1986). Age composition data for 1980–1982 was considered missing in the model.

2.2 STATE-SPACE MODEL

Taku River Sockeye salmon spawner-recruit data were analyzed using a Bayesian state-space Ricker spawner-recruit model (Ricker 1954) that included age-structure and a one year-lag autoregressive component to assess the uncertainty introduced into the estimate of spawning size that produces MSY (Fleischman *et al.* 2013).

2.2.1 Process Model

Returns *R* of Taku River Sockeye salmon originating from spawners in brood years *y* = 1980–2014 were modeled as a function of spawning escapements *S* using a linearized Ricker (1954) spawner-recruit function with an autoregressive lognormal process error with a lag of one year (AR[1]) (Noakes *et al.* 1987),

(1) $$ln\,(R_y) = ln\,(\,S_y) + ln\,(\alpha) - \beta S_y + \phi\omega_{y-1} + \varepsilon_y.$$

In Equation 1, α is the productivity parameter, β is the inverse capacity parameter (density-dependence), ϕ is the lag-1 AR coefficient, and ω_y are the model residuals,

(2) $$\omega_y = ln\,(R_y) - ln\,(S_y) - ln\,(\alpha) + \beta S_y,$$

and the ε_y are independently and normally distributed process errors with standard deviation σ_R. The six initial returns (R_{1974}–R_{1979}), not linked to previous monitored escapements by the spawner-recruit relationship, were modeled as draws from a common lognormal distribution with parameters ln(R_0) and σ_{R0}.

Age-at-maturity vectors ($p_{y,a}: a = 4:6$) from brood year *y* and returning at ages 4–6 (ages 2–4 were combined and ages 6–8 were combined) were drawn from a common Dirichlet distribution that was implemented by generating independent random variables ($g_{y,a}: a = 4:6$) from the gamma distribution,

(3) $$g_{y,a} \sim gamma\,(shape = \gamma_{a,} inverse\ scale\ =\ 0.01),$$

and dividing each by their sum (Evans *et al.* 1993),

(4) $$p_{y,a} = \frac{g_{y,a}}{\sum_a g_{y,a}}.$$

The expected proportions returning at age, π_a, (Gelman *et al.* 2004) were calculated as

(5) $$\pi_a = \frac{\gamma_a}{\sum_a \gamma_a} = \frac{\gamma_a}{D},$$

and implemented as a series of nested beta distributions that reflect age-at-maturity central tendencies that sum to one. The sum of the γ_a can be interpreted as the inverse dispersion (*D*) of the Dirichlet distribution. While a low value of *D* is reflective of a large amount of variability of age-at-maturity proportions (*p)* among brood years, a high value of *D* is indicative of more consistency in *p* over time.

The terminal abundance *N* of age-*a* Taku River Sockeye salmon returning in calendar year *t* (1980–2018) is the product of the age proportion scalar *p* and the terminal return (recruitment) *R* from brood year *y* = *t*-*a* and age *a,*

(6) $$N_{t,a} = R_{t-a}p_{t-a,a}.$$

Terminal run abundance during calendar year *t* is the sum of abundance-at-age across ages,

(7) $$N_t = \sum_{a=4}^{6} N_{t,a}.$$

Inriver run abundance, *IR,* at the border was modeled as

(8) $$IR_t = N_t - Hbelow_t^{NS},$$

which is the terminal run abundance minus naturally-spawned harvest below the border (excluding naturally-spawned personal use harvest). U.S. personal use harvest in the Taku River is excluded from the naturally-spawned harvest below the border because it is already accounted for in the capture-recapture dropouts (Appendix A2).

Finally, the spawning abundance, *S*, or escapement of age-*a* salmon in calendar year *t* is the difference between inriver abundance and the total harvest of Taku River Sockeye salmon above the border (including enhanced and naturally-spawned harvest but excluding naturally-spawned and enhanced broodstock take),

(9) $$S_{t,a} = IR_{t,a} - Habove_{t,a}.$$

Annual terminal harvest below the border (harvest of naturally-spawned and enhanced fish in the U.S. D111 gillnet, the incidental harvest of naturally-spawned and enhanced fish in the purse seine fishery at Amalga Harbor in D111, and the harvest of naturally-spawned and enhanced fish in the U.S. personal use fishery) was modeled as the product of the terminal run abundance and annual harvest rate,

(10) $$Hbelow_t = N_t \mu.below_t,$$

where the annual harvest rate, $\mu.below_t$, is drawn from a beta distribution (Appendix A1). Similarly, annual naturally-spawned harvest below the border $Hbelow_t^{NS}$ was modeled as the product of the terminal run abundance and annual naturally-spawned harvest rate,

(11) $$Hbelow_t^{NS} = N_t \mu.below_t^{NS},$$

where the annual naturally-spawned harvest rate, $\mu.below_t^{NS}$, is drawn from a beta distribution (Appendix A1).

Annual harvest above the border (i.e., commercial gillnet, test/assessment, Aboriginal) was modeled as the product of inriver abundance and annual harvest rate,

(12) $$Habove_t = IR_t \mu.above_t,$$

where annual harvest rate, $\mu.above_t$, is drawn from a beta distribution (Appendix A1). Similarly, annual naturally-spawned harvest above the border $Habove_t^{NS}$ was modeled as the product of inriver abundance and annual naturally-spawned harvest rate,

(13) $$Habove_t^{NS} = IR_t \mu.above_t^{NS},$$

where annual naturally-spawned harvest rate, $\mu.below_t^{NS}$, is drawn from a beta distribution (Appendix A1).

2.2.2 Observation Model

Observed data (Appendix A2; Appendix A3) included pooled Petersen capture-recapture abundance estimates of inriver run abundance (see Section 2.1.1.2 Capture-Recapture Abundance Estimates), annual naturally-spawned harvest below the border $Hbelow_t^{NS}$, total annual harvest (naturally-spawned and enhanced) below the border $Hbelow_t$, annual naturally-spawned harvest above the border $Habove_t^{NS}$, total annual harvest (naturally-spawned and enhanced) above the border $Habove_t$, coefficients of variation (CV; harvest above the border, harvest below the border, capture–recapture), and inriver age compositions. It was assumed that there was no other harvest of Taku River Sockeye salmon.

Estimated inriver run abundance of Taku River Sockeye salmon from the pooled Petersen capture-recapture abundance estimates were modeled as,

(14) $$\widehat{IR}_t = IR_t e^{\varepsilon_{IR_t}} \quad ,$$

where the ε_{IR_t} were normal (0, $\sigma^2_{IR_t}$) and

(15) $$\sigma^2_{IR_t} = ln\,(CV^2_{IR_t} + 1).$$

Estimated below-border harvest (naturally-spawned and enhanced) was

(16) $$H\widehat{below}_t = Hbelow_t e^{\varepsilon_{Hbelow_t}} \quad ,$$

where the ε_{Hbelow_t} were normal (0, $\sigma^2_{Hbelow_t}$) and variances followed equation 15.

Estimated above-border harvest (naturally-spawned and enhanced) was

(17) $$H\widehat{above}_t = Habove_t e^{\varepsilon_{Habove_t}} \quad ,$$

where the ε_{Habove_t} were normal (0, $\sigma^2_{Habove_t}$) and variances followed equation 15.

Estimated naturally-spawned harvest above the border and naturally-spawned harvest below the border were also estimated as in equations 16 and 17 and variances followed equation 15.

The CVs of below and above-border harvest were uniformly set at 0.05 (Appendix A2; cv.hb, cv.ha) to represent low uncertainty given the rigour of the below and above-border harvest reporting, except for the beginning years (1980–1983) when estimates of below-border harvest of Taku River Sockeye salmon stock were not available; these were set at 0.90 (similar to Fleischman *et al.* 2013). The CVs for capture–recapture estimates of inriver abundance were the standard error divided by the estimate (Appendix A3; ir.cv; output from the *BTSPAS* package within the R environment (R Core Team 2019; version 3.6.0) that was used to analyze the capture–recapture data), except for years 1980–1983 and 1986 when the CV was set to 0.90 to represent high uncertainty.

Age Composition

Inriver abundance age compositions were estimated from annual ASL samples taken at the fish wheels (see Section 2.1.1.3 Age Composition Data; Table 6) and weighted by statistical week. D111 gillnet fishery harvest age compositions of Sockeye salmon in Taku Inlet were from annual ASL data from the harvest (see Section 2.1.1.3 Age Composition Data; Table 3) weighted by statistical week. First, proportions-by-age by return year were converted to numbers-at-age, based on annual harvest and pooled Petersen capture-recapture abundance estimates, respectively (e.g., the proportion of freshwater age-1 and saltwater age-2 (age 1.2 fish) in 2016 was multiplied by the pooled Petersen capture-recapture abundance estimate in 2016 to calculate the number of age 1.2 fish in 2016 captured at the fish wheels). Next, the numbers by age for each method (harvest or fish wheels) were combined for each age group (i.e., age-4 fish; freshwater age-0 and saltwater age-3 (an age 0.3 fish) from the fish wheel was combined with freshwater age-1 and saltwater age-2 (an age 1.2 fish) from the fish wheel). Then, these numbers-at-age by method (harvest or fish wheels) were combined (i.e., age-4 fish numbers from the 2016 fish wheel samples were added to the age-4 fish numbers from the 2016 D111 gillnet harvest). This essentially weighted the numbers-at-age by the inriver abundance (based on the pooled Petersen capture-recapture abundance estimates) and harvest numbers (i.e., if harvest was larger, the numbers-at-age in the harvest received more weight). Next, the combined numbers-at-age were converted to annual proportions by age, $q_{(ob)t,a}$. Then, the annual proportions of age-2 were combined with age-3 and age-4 (i.e., ages 2, 3, and 4 became one age group; ages 2-4). Likewise, the annual proportions of age 6 were combined with age-7 and age-8 to create one age group; ages 6-8. Finally, the weighted annual proportions by age were multiplied by 100,

(18) $x_{t,a} = q_{(ob)t,a} n_{Et}$ where $\sum x_{t,a} = n_{Et} = 100$ across all ages for each year,

to calculate the surrogate terminal run age counts, $x_{t,a}$ (Appendix A2). The age counts ($x_{t,a}$) were modeled as multinomial distributions with order parameter n_{Et} and proportion parameters,

(19) $$q_{t,a} = \frac{N_{t,a}}{N_t},$$

where $\sum q_{t,a} = 1$ across all ages for each calendar year *t*. Age composition in years 1983 to 1985 were based only on fish wheel age composition, and age composition in year 1986 was based only on D111 gillnet harvest age composition (see Section 2.1.1.3 Age Composition Data).

Key model results from state-space analyses of Pacific salmon are typically not sensitive to the choice of n_{Et} (Fleischman and McKinley 2013), therefore, an arbitrary annual effective sample size of n_{Et} = 100 was used and surrogate terminal run age counts, $x_{t,a}$, were obtained that summed to n_{Et} (Appendix A2). Other analyses using a Bayesian state-space modeling method have set n_{Et} = 100 as an arbitrary effective sample size (e.g., Hamazaki *et al.* 2012, Fleischman and Reimer 2017).

2.2.3 Model Fitting

Model fitting involves finding the values of population parameters that can plausibly result in the observed data. Using the package *rjags* (Plummer 2019) and the package *R2jags* (Su and Yajima 2015) within the R environment (R Core Team 2019; version 3.6.0), MCMC methods were employed to provide a more realistic assessment of uncertainty than is possible with traditional spawner-recruit methods. The packages *rjags* and *R2jags* provide an interface between the R environment and the software JAGS (Just Another Gibbs Sampler for Bayesian data analysis).

Prior Distributions

For all unknowns in the model, Bayesian analysis requires that prior probabilities be specified. Most prior distributions in this model were vague with a few exceptions (Table 7). For the parameter beta (Millar 2002) and log alpha, normal priors with mean 0, extremely large variances, and constrained to be greater than 1.00×10^{-6} were used. The parameter phi was constrained to be between -1.0 and 1.0. Log transformed initial recruitments R_{1974}–R_{1979} (those with no linked spawner abundance) were modeled as drawn from a common normal distribution with mean ln(R_0) and variance σ^2_{R0}. Fleischman *et al.* (2013) found that an informative prior on σ_{R0} may have a large effect on the posterior distribution of σ_{R0} and the initial values of R_y, but negligible effects on key model quantities. The initial model residual ω_0 was given a normal prior with mean zero and variance $\frac{\sigma^2_R}{(1-\phi^2)}$. Annual harvest rates ($\mu.below_t, \mu.above_t, \mu.below^{NS}_t, \mu.above^{NS}_t$) were given a beta(1,1) prior distribution. The vector of age proportion hyperparameters $\{\pi_a = 4{:}6\}$ was given a Dirichlet prior distribution, implemented as a series of nested beta distributions. Diffuse conjugate inverse gamma priors were used for $\sigma^2_{R\,0}$ and σ^2_R.

Diagnostic Tools for Assessing Mixing and Convergence

MCMC methods were used to generate the joint posterior probabilities of the unknown quantities using the package *rjags* (Plummer 2019) and the package *R2jags* (Su and Yajima 2015) within the R environment (R Core Team 2019; version 3.6.0). Three MCMC chains were initiated. After a 10,000 sample burn-in period was discarded, 3,000 samples (1,000,000 iterations, thinned by 1000; 1000 samples per chain) were retained for analysis to estimate posterior medians, standard deviations, and percentiles. A variety of diagnostic tools from the package *rjags* (Plummer 2019) were used to assess mixing and convergence including: time series plots (plot of the sampled value against its number on the chain to check for adequate mixing in visual assessment of overlaid chains), autocorrelation plots (the correlation between the samples *i* iterations apart in the chain to determine adequate mixing and assessing whether a chain needs further thinning), and density plots. The density plots of the posterior distribution should be smooth and should not reach the bounds of the priors. In addition, the Gelman-Rubin-Brooks convergence statistic R_c (Gelman and Rubin 1992; Brooks and Gelman 1998), Monte Carlo standard errors (i.e., an estimate of the difference between the mean of the sampled values (the posterior mean) and the true posterior mean; MC error should be less than 5% of the sample standard deviation; Toft *et al.* 2007), and the Geweke statistic were assessed. The Gelman-Rubin-Brooks diagnostics monitor the convergence of iterative simulations by comparing between and within variances of multiple chains. Brooks and Gelman (1998) have suggested that if $R_c < 1.2$ for all model parameters, one can be fairly confident that convergence has been reached. The Geweke statistic and plots (Geweke 1992) should fall within the range [-2, 2]. The Geweke statistic is a z-score and values in this range indicate that early and late sample means fall within 2 standard deviations. If not, then the earlier part of the MCMC chain differs from the later part, and a longer burn-in period is warranted.

Biological Benchmark Estimation for Management

Biological benchmarks were calculated for each individual MCMC sample. Spawning abundance at MSY, S_{MSY} was calculated based on the Lambert W function (Scheuerell 2016)

(20) $$S_{MSY} = \frac{1-W(\mathrm{e}^{1-ln\,(\alpha')})}{\beta},$$

where $ln(\alpha') = ln(\alpha) + \frac{\sigma_R^2}{2(1-\phi^2)}$, to correct for the difference between the median and the mean of a lognormal error distribution from an AR(1) process (Pacific Salmon Commission 1999). Sustained yield at a specified level of S was obtained by subtracting spawning escapement from recruitment,

(21) $$Y_S = R - S = S\mathrm{e}^{(ln\,(\alpha')-\beta S)} - S.$$

Spawning escapement at maximum sustained return, S_{MAX}, was calculated as $1/\beta$ and equilibrium spawning abundance in the absence of fishing (recruitment that exactly replaces spawners) as

(22) $$S_{EQ} = \frac{ln\,(\alpha')}{\beta}.$$

Harvest rate leading to MSY, U_{MSY}, was approximated by Scheuerell (2016) as

(23) $$U_{MSY} = \beta S_{MSY}.$$

The spawner abundance that allows rebuilding to S_{MSY} in one generation in the absence of fishing, S_{GEN}, was calculated using a solver function from the Wild Salmon Policy Metrics Package (Holt and Pestal 2019), which implements the calculations developed by Holt and Ogden (2013). For each alpha and beta in the posterior sample, the solver uses Hilborn's approximation (Hilborn 1985) to calculate S_{MSY}, and then finds the smallest spawner abundance that produces expected recruits larger than S_{MSY} (i.e., build up to S_{MSY} in one generation in the absence of fishing). The biological reference points S_{MSY}, U_{MSY}, and 80%S_{MSY} based on Scheuerell (2016) were calculated based on posterior samples of $ln(\alpha')$ and beta and the package *gsl* (Hankin 2006) within the R environment (R Core Team 2019; version 3.6.0).

2.3 OVERVIEW OF SENSITIVITY ANALYSIS

2.3.1 Sensitivity Analysis to the Prior Distribution for the Beta Parameter

Sensitivity analyses were performed on the prior distribution for the beta parameter from the spawner-recruit function as beta, the level of density dependence, is stock specific (Hilborn and Liermann 1998; Mueter *et. al.* 2002) and can vary greatly among stocks. The base prior distribution for the beta parameter was a normal distribution with mean 0, precision (inverse of the variance) 0.000001, and constrained to be greater than 1.00×10^{-6}. An alternative prior distribution for the beta parameter, beta$_1$ (β_1), was a uniform distribution from 0.000001 to 1.0. A second alternative prior distribution for the beta parameter, beta$_2$ (β_2), was a normal distribution with mean 0 and precision 0.000001 without constraints.

The JAGS input for the dispersion parameter of a normal prior is the precision, which is the inverse of the variance.

2.3.2 Sensitivity Analysis on the Early Years of Missing Capture-Recapture Abundance Estimates (1980-1983)

Sensitivity analyses were also performed on the early years of missing capture-recapture abundance estimates (Table 8). For all scenarios, 1984–1985 and 1987–2018 pooled Petersen capture-recapture abundance estimates were the input for estimates of above-border abundance with the associated coefficients of variation from the base model (Table 8). There was no inriver abundance estimate for 1986. Five scenarios were explored for the sensitivity analysis. For scenarios 1a, 1b, and 1c, the inriver abundance estimates for years 1980–1983, and 1986, were the median capture-recapture abundance estimates output from the posterior distribution of the base model. The coefficients of variation were adjusted by the scenario (0.90, 0.50, and 0.10, respectively) to represent varying degrees of confidence in these estimates. For scenario two, median capture-recapture abundance estimates from the posterior distribution of the base model for years 1980–1983, and 1986 were multiplied by 0.75 to represent potentially lower abundance than the base model for these years. The coefficients of variation were set to 0.10 for these years. For scenario three, median capture-recapture abundance estimates from the posterior distribution of the base model for years 1980–1983, and 1986 were multiplied by 1.33 to represent potentially higher abundance than the base model for these years. The coefficients of variation were set to 0.10 for these years.

2.4 PRESENTATION OF RESULTS

2.4.1 Optimal Yield Profiles

Optimal yield probabilities are the probabilities that a given level of spawning escapement (*S*) will produce average yields exceeding X% of MSY: P(Y_S > X% of MSY). These probabilities were calculated as

(28) $$P(Y_S > X\%MSY) = \frac{\text{number of } Ys > X\%MSY}{\text{number of MCMC samples}}.$$

Optimal yield profiles are plots of *P* versus *S* (Fleischman *et al.* 2013).

2.4.2 Overfishing Profiles

Overfishing probability was calculated as 1 – P(Y_S > *X*% of MSY) at $S < S_{MSY}$, and 0 at $S > S_{MSY}$. These profiles show the probability that sustained yield is reduced to less than a percentage (70%, 80%, 90%) of MSY given a fixed level of escapement (Bernard and Jones III 2010).

2.4.3 Maximum Recruitment Profiles

Maximum recruitment probability is the probability that a given spawning escapement *S* would produce average recruit exceeding X% of maximum recruit. These probabilities were calculated as

(29) $$P(Y_S > X\%MAX) = \frac{\text{number of } Y_S > X\%MAX}{\text{number of MCMC samples}}.$$

Maximum recruitment probability profiles are then a plot of *P* versus *S* (Hamazaki *et al.* 2012).

2.4.4 Sustained Yield

Expected sustained yield, or the numbers of fish over and above those necessary to replace spawners averaged over the brood years 1974–2014, is maximized near S_{MSY}.

3 RESULTS

3.1 DIAGNOSTIC TOOLS FOR ASSESSING MIXING AND CONVERGENCE

Given the data and model structure, no major problems were encountered based on the diagnostics. The time series plots for key model parameters were centered around a mean and the three chains were indistinguishable, suggesting that the chains had each converged and had converged to similar values. To improve mixing of the chain and prevent autocorrelation, samples were thinned (every 1000th iteration). The Gelman-Rubin convergence diagnostic was < 1.2 for all model parameters (Brooks and Gelman 1997), suggesting that all chains converged to the same posterior distribution. The MC error was calculated and compared to the sample standard deviation to determine if a sufficient number of samples had been obtained after the burn-in period; all ratios of MC error divided by the sample standard deviation were < 0.05 suggesting that the number of samples remained adequate after burn-in. Ten thousand iterations were discarded at the beginning of the chain (burn-in period) to ensure that the first and last part of the chain had the same distribution. As the diagnostic tools suggested that there was adequate mixing and convergence of the MCMC chains, interval estimates were constructed from the percentiles of samples of the posterior distribution.

3.2 ABUNDANCE, TIME-VARYING PRODUCTIVITY, HARVEST RATES, AGE COMPOSITIONS, AND AGE-AT-MATURITY

Reconstructed terminal run abundance (*N*) estimates for the autoregressive Ricker model had CVs that ranged from 4% to 46% (Figure 2C; Table 9). Years with higher uncertainty corresponded to years with missing capture-recapture abundance estimates (1980–1983, 1986; Appendix A3), and missing age composition data (1980–1982; Appendix A2). Excluding the initial returns, reconstructed brood year recruitments had CVs that ranged from 6% to 18% (Table 9). The Ricker recruitment or productivity residuals (Figure 3B) are deviations in recruitment from those predicted by the Ricker spawner-recruit relationship. After controlling for density-dependent effects, these residuals reflect the time-varying changes in productivity. The lack of trend in these residuals, which were spread around 0, indicated a good model fit. Terminal run abundance was relatively stable except for small spikes in years 1996, 2001, and 2003 (Table 9 and Table 10; Figure 2C).

Taku River Sockeye salmon matured primarily at age 5 (mean range: 42–66%; principally age 1.3 and a few age 2.2) and ages 2–4 (mean range: 28–54%; principally age 1.2 and 0.3 and fewer 0.2 and 1.1), followed by much smaller proportions at ages 6–8 (mean range: 2–10%; mainly ages 1.4 and 2.3) (Figure 4A). Median below-border harvest rates of naturally-spawned Sockeye salmon varied throughout the time series without a consistent period of highs or lows (Figure 5A; maximum 65% and mean 45%). Excluding the first few years of the time series (1980–1983), median above-border harvest rates of naturally-spawned Sockeye salmon were very stable with minimal variability throughout the time series (Figure 5B; mean 15% and maximum 19%).

3.3 STOCK PRODUCTIVITY, CAPACITY, AND YIELD

Results of the Ricker spawner-recruit relationships take into account measurement error in both *S* and *R* when derived from the age-structured state-space model fitted to capture-recapture abundance estimates, harvest data, and age composition data; these results are depicted as the error bars in Figure 6, which weight the individual data pairs based on how precisely they were estimated. Plausible relationships (grey shaded regions) generated from the observed data were diverse, often deviating substantially from the posterior medians of $\ln(\alpha')$ and β (Figure 6; dark dashed line). In particular, beta, the strength of the density dependence or how quickly the curve comes down, varied greatly as the data was constricted below the replacement line (Figure 6; solid black line). The median estimate of $\ln(\alpha')$ was 2.11, corresponding to $\alpha' = 8.17$ (high productivity stock; $\alpha \geq 4$; Su and Peterman 2012) and the median estimate of the density dependent parameter β was 1.69×10^{-5} (Table 11). Uncertainty about β is reflected in variability in the values of *S* leading to maximum recruitment $S_{MAX} = 1/\beta$, and uncertainty about equilibrium abundance, S_{EQ} is reflected by variability in the values of *S* where the curves intersect the replacement line. The contrast in the spawner data used in the spawner-recruit analysis (1980–2014) is low (3.8; Clark *et al.* 2014); statistical stock-recruit analysis with ranges less than four are likely to produce poor estimates of S_{MSY} (CTC 1999). Stock productivity, $\ln(R/S)$, (Figure 7) was strongly cyclic ranging from 0.28 to 1.79 (R/S ranged from 1.3 to 6.0). The estimated AR(1) parameter ϕ was 0.24, suggesting weak positive lag-1 serial correlation in residuals.

Estimates of escapement obtained by fitting a state-space model to Taku River Sockeye salmon data ranged from 24,075 fish in 1982 to 102,456 fish in 2016 (Figure 8). To incorporate uncertainty about the plausible spawner-recruit relationships (Figure 6), the success or failure of a given number of spawners to achieve biological reference points across plausible spawner-recruit relationships were tallied to create overfishing profiles (Figure 9; top panel), maximum recruitment profiles (Figure 9; middle panel), and optimal yield profiles (Figure 9, bottom panel). The maximum recruitment profiles, which are highest near S_{MAX} = 59,145 fish (Table 11), display the probability of achieving at least 70%, 80%, or 90% of maximum recruitment for specified levels of escapement. Optimal yield profiles show the probability of a given number of spawners achieving at least 70%, 80%, or 90% of MSY. These probabilities, which are highest near S_{MSY} = 43,857 fish (Table 11) can be used to quantify the yield performance of prospective escapement goals taking into consideration uncertainty about the true abundance and productivity of the stock (Figure 9 and Figure 10). Overfishing profiles show the probability that sustained yield would be reduced to less than 70%, 80%, or 90% of MSY by not allowing enough fish to spawn.

3.4 SENSITIVITY ANALYSES ON THE PRIOR DISTRIBUTION ON THE BETA PARAMETER

The results of a sensitivity analysis on the prior distribution for the beta parameter of the Ricker model shows that a uniform distribution (alternative prior: β_1) produced similar median estimates of key model outputs (e.g., ln(α'), β, and reference points) to those produced with the normal prior distribution (Table 12), although the computation time was greatly increased with the uniform prior distribution. For example, the median estimate of S_{MSY} was 43,857 fish for the base model and 44,032 fish for the alternative prior on beta based on a uniform distribution. Likewise, a normal prior distribution on beta that was not constrained to be greater than 1.00 x 10^{-6} (alternative prior β_2) greatly increased the uncertainty of the estimated reference points, but produced similar median estimates of key model outputs (Table 12). For example, the median estimate of S_{MSY} was 43,857 fish for the base model and 43,692 fish for the alternative normal prior distribution on beta that was not constrained to be greater 1.00 x 10^{-6}, but the CVs were drastically different (0.67 for the base model and 4.05 for the alternative prior distribution on beta). This result held for other reference points such as S_{EQ} and S_{MAX}. Therefore, the base prior distribution was used for median estimates of key model outputs and biological reference points.

3.5 SENSITIVITY ANALYSES ON THE EARLY YEARS OF MISSING CAPTURE-RECAPTURE ABUNDANCE ESTIMATES (1980–1983)

A sensitivity analysis on the early years (scenarios 1a, 1b, and 1c; median capture-recapture abundance estimates output from the posterior distribution of the base model used as input data for years 1980–1983, and 1986 and coefficients of variation adjusted by the scenario (0.90, 0.50, and 0.10, respectively)) indicated that arbitrarily increasing certainty to abundance data from these years increased the posterior estimate of S_{MSY}. The increase in S_{MSY} varied from approximately 2,900 fish to 15,100 fish (S_{MSY} varied from 46,720 fish to 58,987 fish) as the coefficients of variations of the capture-recapture abundance estimates decreased from 0.90 to 0.10 (Table 13); although the coefficients of variations of S_{MSY} were relatively stable (range from 0.62 to 0.72 for the three scenarios and 0.67 for the base model). If the coefficients of variation of the 1980–1983 and 1986 capture-recapture abundance estimates were set at 0.10 (i.e., similar to the other inputs from years 1984, 1985, 1987–2018), but the inputs for years 1980–1983 and 1986 were multiplied by 75% (scenario two), the estimate of S_{MSY} remained relatively unchanged from the base model although the precision on the reference point increased (CV decreased to 0.43; Table 13). If the coefficients of variation of the 1980–1983 and 1986 capture-recapture abundance estimates were set at 0.10 (i.e., similar to the other inputs from years 1984, 1985, 1987–2018), but the inputs for years 1980–1983 and 1986 were multiplied by 1.33 (scenario three), the estimate of S_{MSY} increased by about 20,600 fish, but the precision on the reference points remained similar to the base model (0.74; Table 13).

3.6 BIOLOGICAL BENCHMARKS FOR MANAGEMENT

Using base case assumptions and data from the 1980–2014 brood years, the estimated biological benchmarks (5th and 95th percentiles of the posterior distribution, capturing the central 90% of parameter samples) were:

- Spawner level that produces maximum sustained yield (S_{MSY}) estimated at 43,857 spawners (30,422 to 99,699 spawners);
- Spawner level that is 80% of that needed to produce maximum sustained yield (80%S_{MSY}) estimated at 35,086 spawners (24,337 to 79,760 spawners);
- Spawner level that produces the maximum adult recruits (S_{MAX}) estimated at 59,145 spawners (35,843 to 164,901 spawners);
- Equilibrium spawner level in the absence of fishing (S_{EQ}) estimated at 124,106 spawners (97,418 to 252,655 spawners);
- Spawner level with a high probability of rebuilding to S_{MSY} in one generation in the absence of harvest (S_{GEN}) estimated at 5,873 spawners (1,967 to 25,146 spawners).

4 DISCUSSION

4.1 BIOLOGICAL BENCHMARKS BASED ON SPAWNER-RECRUIT ANALYSIS

The beta parameter was difficult to estimate due to the absence of escapements that did not replace themselves. None of the median point estimates of recruits versus spawners crossed the replacement line suggesting little overall contrast within this time series (Figure 6) and only two of the 95% credible intervals of the estimated number of spawners cross the replacement line (1983 and 2013). Constraining the beta parameter to be greater than 1.00×10^{-6} in the base model greatly increased the precision on the reference points, while maintaining similar median estimates (Table 12). Therefore, additional information at high spawner abundance is needed to reduce uncertainty in beta.

4.2 CONSIDERATIONS FOR CHOOSING A SPAWNING ESCAPEMENT GOAL

Fishery policy guides the frame of reference, which determines how we present and interpret the results of biological analyses. As summarized in Section 1.2, ADF&G and DFO operate under separate policies and the results of our analyses may be applied in different ways.

Although the median estimate of S_{MSY} was 43,857 fish, a variety of escapement goal ranges can be considered that encompass this value. Given the uncertainty in outputs of the spawner-recruit model, the minimal contrast within the time series, and lack of high spawner abundances, the Taku Working Group recommended a conservative approach be applied when considering the tradeoff between achieving MSY and guarding against overfishing the stock. For example, the largest spawning escapement used in the spawner-recruit analysis (1980–2014) was 91,294 fish (2003) and the overall contrast between this and the lowest escapement (24,075 fish in 1983) was only 3.8. In comparison, salmon stocks considered to have greater information about the spawn-recruit relationship have a contrast of eight or more (Clark *et al.* 2014). With few escapements beyond 75,000 and no evidence of density dependence, there is substantial uncertainty in estimates of recruits at the higher end of the range (Figure 6). Table 14 compares the probability of biological escapements achieving at least 70%, 80%, or 90% of median MSY for each parameter set sampled from the posterior distribution (i.e., the yield profiles underlying the ADF&G optimal yield reference ranges) along with the probability of overfishing the stock such that sustained yield is reduced to less than a percentage (70%, 80%, 90%) of MSY by fishing too hard and supplying too few spawners (Figure 9). For example, Figure 11 shows a hypothetical escapement goal range of 40,000–75,000 fish. Based on the lower bound of this example, the overfishing profile shows the probability of overfishing the stock to a point where sustained yield is reduced to less than 70%, 80%, or 90% of MSY is 5%, 8%, and 17%, respectively. This probability should be considered when choosing an appropriate range as the consequences of error are more severe for the overfishing objective than the yield objective, so consideration could be given to raising the lower bound slightly.

Besides the uncertainty in the stock-recruit curve, the minimal contrast in estimated escapement within the time series, and the risk of overfishing, another consideration in choosing an escapement goal range is the Treaty negotiations. Both Parties share in the responsibility of achieving escapements within the escapement goal range, and harvest shares and management accountability are based on the agreed escapement objective. Chapter 1, Annex IV, paragraph 3(b)(i) of the Pacific Salmon Treaty (January 2019) states:

> (A) "Annual abundance of wild Taku River Sockeye salmon shall be estimated by adding the catch of wild Taku River Sockeye salmon in U.S. District 111 to the estimated above-border abundance of wild Sockeye salmon. The annual TAC of wild Taku River Sockeye salmon shall be estimated by subtracting the agreed escapement objective as defined in the annual management plan from the annual terminal run abundance estimate."

Chapter 1, Annex IV, paragraph 4(a) of the Pacific Salmon Treaty (January 2019) states:

> "(Trigger 1)…the Parties shall review the overall management regime and recommend adjustments commencing the following year to better address conservation requirements if the lower end of agreed escapement goal ranges in three consecutive years is not achieved."

Therefore, managers in both the U.S. and Canada are held accountable to the agreed escapement objective and the lower end of the goal; thus, the lower bound should be appropriately conservative to avoid overfishing while also striving to maximize yield. In addition, the escapement goal range should be wide enough for managers to be able to reasonably manage a mixed stock fishery within it (i.e., multiple lake and river-type Sockeye salmon stocks, enhanced Sockeye salmon returns, Chinook salmon management, etc.) given year to year fluctuations in stock abundance and marine survival, the uncertainty of inseason run size estimates, and bilateral allocation guidelines. The 2019 revised "interim" objective, based on the historical objective adjusted by 22%, of 55,000 to 62,000 fish with a management target of 59,000 fish (TTC 2019a), is a relatively narrow range. Escapement goal ranges of 22 Alaska Sockeye salmon stocks, based on estimates of escapements that provide maximum sustained yield (S_{MSY}), have a spread, on average, of 2.1 (i.e., lower bound multiplied by 2.1 equals the upper bound), with a range of 1.3 to 2.5 (Miller and Heinl 2018). Eggers (1993) suggests that a goal range of 0.8 to 1.6 times S_{MSY} is appropriate for sufficient management flexibility and to maintain catch levels to within 90% of S_{MSY}, although this was based on a simulation and should be applied strictly to single-stock fisheries.

To identify an MSY-based escapement goal range, we recommend using the following risk criterion (developed from the performance profiles):

- The recommended escapement goal should provide a greater than 90% probability that sustained yield would be at least 80% of MSY; a probability of "overfishing" of less than 10%;

- Prior escapement goal recommendations have been informed by a low probability of overfishing (Chinook salmon in the Copper River (Savereide *et al.* 2018); Chinook salmon in the Alsek River (Bernard and Jones III 2010));

- The recommended escapement goal should provide a greater than 50% probability of achieving at least 70% of MSY over the long-term if the stock is managed to the proposed escapement goal range.

Table 14 can be used to directly identify candidate bounds for an escapement goal range, once a probability level has been identified. Shaded areas delineate the spawner abundances that meet the two proposed criterion (above). Any spawner abundance above 38,000 has a less than 10% probability of achieving less than 80% of MSY (6th column of Table 14). A lower bound for the escapement goal range could be selected at any point above this value and be consistent with the first risk criterion above. Any spawner abundance between (and including) 28,000 and 79,000 has a probability of 50% or higher of achieving 70% of MSY (4th column of Table 14). An upper bound for the escapement goal range could be selected at any point in this range and be consistent with the second risk criterion. Additional qualitative considerations could be used to further refine the candidate values for lower and upper bounds.

4.3 CAPTURE-RECAPTURE ABUNDANCE ESTIMATES IN A NEW FRAMEWORK

The historical spawning escapement objective, established in 1985, was from 71,000 to 80,000 fish with a point goal of 75,000 fish. Dropout rates of tagged fish observed through radiotelemetry studies completed in 1984, 2015, 2017 and 2018 biased capture-recapture estimates high. Therefore, pooled Petersen capture-recapture abundance estimates in this study were adjusted downward by 25.5% to account for dropouts and adjusted downward by an additional 6.4%, which represented the average bias observed between size-stratified and pooled Petersen estimates (see 2.1.1.2 Capture-Recapture Abundance Estimates). Non-expanded previously published estimates from the TTC (TTC 2019b; Appendix D15) report (Table 5; column 6), adjusted by the year-specific dropout rate and by the size selectivity bias (6.4% downward) (Table 5; column 7), are very similar to the new capture-recapture abundance estimates that were used as input in the Bayesian state-space model (Figure 12).

4.4 SOURCES OF UNCERTAINTY

- *Process and observation error:* Use of a Bayesian age-structured state-space model allowed for consideration of process variation (natural fluctuations) in stock productivity, recruitment, and age-at-maturation independently from observation error (uncertainty in measurements of observed data) in run size, harvest, and age composition.

- *Alternative estimation approaches*: The derived estimates of the reference points S_{MSY}, 80% S_{MSY}, and U_{MSY} based on the calculations of Lambert W (Scheuerell 2016), Peterman *et al.* (2000), and Hilborn (1985) approximations all produced similar results. Therefore, only reference points using the more explicit Lambert W function are shown for simplicity.

- *Alternative model assumptions*: Sensitivity analysis on the choice of prior distribution for the beta parameter were explored and biological benchmarks were estimated for each model for comparison. The base case for beta was a prior distribution which is a normal distribution, mean 0, precision 0.000001, and constrained to be > 1.00 x 10^{-6}. An alternative prior distribution, $beta_1$, was a uniform distribution from 0.000001 to 1.0. A second alternative prior distribution, $beta_2$, was a normal distribution, mean 0, and precision 0.000001. Median estimates of key model outputs such as ln(α'), β , and reference points were similar across the base case and the two alternative prior distributions on beta, although the precision on the reference points was much lower on the model that was implemented with the $beta_2$ prior (Table 12).

- *Uncertainties in environmental variability:* Temporal changes in ocean conditions (e.g., sea surface temperature, acidification, freshwater discharge) can potentially affect salmon survival rates and cause large year-to-year variations in Northeast Pacific salmon productivity (Adkison *et al.* 1996; Mueter *et al.* 2002). Time-varying management policies (target spawner abundance changing in response to changes in the Ricker productivity parameter), may have the potential to result in higher escapement and more harvest while reducing the risk across a range of harvest rates (Collie *et al.* 2012).
- *Expansions of Capture-Recapture Abundance Estimates:* In future analyses of an MSY-based escapement goal and estimation of biological benchmarks based on a state-space modeling framework, the capture-recapture abundance estimates could be expanded by fish wheel CPUE in years with low tag recovery and effort or early removal of the fish wheel during early or late statistical weeks; or the Bayesian time-stratified estimate (*BTSPAS*; Schwarz 2006; Schwarz *et al.* 2009; Bonner and Schwarz 2020) could be used. The *BTSPAS* estimate is a hierarchical model that will extrapolate the run curve before the commercial catch occurred and after the fishery ended, or if the fish wheel is removed early due to low water or other unforeseen reason.
- *Uncertainties in data*: Unreported harvest and incidental fishing mortality such as escape mortality (mortality of fish that actively escape after contact with fishing gear such as a hook or gillnet prior to landing), depredation (fish that die as a result of predators directly removing fish from fishing gear during the capture process; not including predation of released fish), and fishery drop-out (fish that die and drop out of fishing gear such as gillnets prior to landing) were not accounted for in this analysis (Patterson *et al*. 2017). This unaccounted harvest can bias available harvest estimates. Second, total fish numbers from the U.S. commercial fisheries are based on total weight converted to numbers of fish. Third, an unknown number of Taku River Sockeye salmon are harvested in non-directed interception fisheries in Southeast Alaska outside of the terminal area (defined as District 111). Pursuant to the Treaty, this analysis only included directed harvest of Taku River Sockeye salmon in terminal areas. Chapter 1, Annex IV, paragraph 3(b)(i) of the Pacific Salmon Treaty (January 2019) states:

  > "… the following provisions apply to the U.S. District 111 drift gillnet fishery and to Canadian in-river fisheries. Directed fisheries on Taku River Sockeye salmon will occur only in the Taku River drainage in Canada and in District 111 in the U.S." (A) "Annual abundance of wild Taku River Sockeye salmon shall be estimated by adding the catch of wild Taku River Sockeye salmon in U.S. District 111 to the estimated above-border abundance of wild Sockeye salmon. The annual TAC of wild Taku River Sockeye salmon shall be estimated by subtracting the agreed escapement objective as defined in the annual management plan from the annual terminal run abundance estimate."

  Therefore, run abundance and return only encompass Taku River Sockeye salmon inside the Taku River drainage and in the terminal areas adjacent to the outlet. Future analyses could include harvest in non-terminal areas to better represent the total run abundance and total return of Taku River Sockeye salmon.

  Years with higher uncertainty corresponded to years with missing capture-recapture abundance estimates (1980–1983, 1986; Appendix A3), missing age composition data (1980–1982; Appendix A2), and missing harvest below the border (1980–1982; stock composition of naturally-spawned Taku River Sockeye salmon was unavailable for these years). Modeled escapements from these years were some of the lowest in the time series

(24,075 to 36,106). Subsequent productivity and yields from these escapements are highly uncertain. Future analyses of an MSY-based escapement goal and estimation of biological benchmarks based on a state-space modeling framework could consider providing additional information for these early years with the potential for added bias with the accompanying additional assumptions.

- Excluding years 1980–1983 from the analysis; or

- Adding additional harvest, capture-recapture abundance estimates, and age composition data to years 1980–1983 and 1986 with more assumptions about the data (see Section 3.5: Sensitivity Analyses on the Early Years of Missing Capture-Recapture Abundance Estimates).
  - Include the non-expanded post season capture recapture abundance estimate of 1986 (104,162 fish; TTC 2019b) adjusted downward by 6.4% (representing the average bias observed between size-stratified and pooled Petersen estimates from 2003 to 2018) and adjusted to account for dropout (adjusted to 72,634 fish);
  - Include D111 gillnet harvest below the border from 1980–1982 adjusted by the Taku stock proportion from a 10-year average (0.81 from 1983–1992) as the Taku stock was partitioned out in these years (1983 on) and enhancement had not begun;
  - Age compositions from 1980–1982 are missing as a model input (Appendix A2); total run age composition from 1982 could be represented by D111 gillnet harvest (not separated out by the naturally-spawned Taku stock) (see Section 2.2.2.1: Age Compositions) as fish wheel age data is unavailable for 1980–1982;
  - Age compositions from 1983–1985 are only weighted by fish wheel data, but the age composition could be weighted by the fish wheel data and the D111 gillnet harvest (not separated out by the naturally-spawned Taku stock) (see Section 2.2.2.1: Age Compositions);
  - The age composition from 1986 is only based on D111 gillnet harvest age composition; it could be weighted by the fish wheel and D111 gillnet harvest age composition if the capture-recapture abundance estimate from the TTC (2019b) report is adjusted by size selectivity and dropout (i.e., the 1986 non-expanded adjusted capture-recapture abundance estimate from the TTC (2019b) report would serve as a weighting for the fish wheel age composition data; 104,162 fish adjusted downward by 6.4% (representing the average bias observed between size-stratified and pooled Petersen estimates from 2003 to 2018) and adjusted to account for dropout (adjusted to 72,634 fish)).

Finally, the low contrast in escapements results in uncertainty in parameter estimates and derived estimates of biological benchmarks.

4.5 SUMMARY OF PROJECT OBJECTIVES

Section 1.1.1 identified the two specific objectives of this work.

Objective 1: Identify the spawning escapements that would produce MSY for the Taku River Sockeye salmon stock aggregate.

The median estimate of S_{MSY} was 43,857 fish.

Objective 2: Identify the appropriate biological benchmarks for the management of the Taku River Sockeye salmon stock aggregate.

Biological benchmarks (5th and 95th percentiles of the posterior distribution, capturing the central 90% of parameter samples) based on data for the 1974–2014 brood years are:

- Spawner level that produces maximum sustained yield (S_{MSY}) estimated at 43,857 spawners (30,422 to 99,699 spawners);
- Spawner level that produces 80% of maximum sustained yield (80%S_{MSY}) estimated at 35,086 spawners (24,337 to 79,760 spawners);
- Spawner level that produces the maximum adult recruits (S_{MAX}) estimated at 59,145 spawners (35,843 to 164,901 spawners);
- Equilibrium spawner level in the absence of fishing (S_{EQ}) estimated at 124,106 spawners (97,418 to 252,655 spawners);
- The spawner abundance with a high probability of rebuilding to S_{MSY} in one generation in the absence of harvest, S_{GEN}, estimated at 5,873 spawners (1,967 to 25,146 spawners).

A variety of escapement levels are presented in Table 14.

5 CONCLUSIONS

5.1 RECOMMENDED SPAWNING ESCAPEMENT GOAL

Based on the analyses from the state-space model and given consideration for the low contrast and high uncertainty of data, parameter, and biological benchmarks, the recommendation from the Taku Working Group is a biological escapement goal range of 40,000 to 75,000 naturally-spawned fish. Based on the lower bound of this range, there is a 92% probability of achieving at least 80% of MSY, and the probability of overfishing, where sustained yield is reduced to less than 80% of MSY, is 8%. The upper bound of this range has a minimum 59% probability of achieving at least 70% of MSY and a minimum 43% probability of achieving at least 80% of MSY.

5.2 FUTURE DATA REQUIREMENTS

To continue to monitor the status of the Taku River Sockeye salmon stock and to facilitate future updates to the escapement goal, it is recommended that collection of the following data continues:

- Harvest and effort data along with age and stock composition data (matched genetic tissue, otolith, and age-size samples) from the D111 gillnet commercial fishery;
- Harvest and effort data from the U.S. Taku River personal use fishery;
- Harvest and effort data from the Canadian inriver Taku commercial fishery along with the proportion of enhanced Sockeye salmon, stock composition data, and age-size data (based on otolith, genetic tissue, and age-size samples);
- Harvest and effort data from the Aboriginal fishery on the Taku River;
- Effort, age-size, and stock composition data (including genetic tissue) from the Canyon Island fish wheels;
- Capture-recapture abundance estimates derived from spaghetti tagging at the fish wheels and recovery in the Canadian commercial fishery;
- Escapement enumeration at King Salmon, Tatsamenie, Little Trapper, and Kuthai lakes;

- Age-size data, tag recovery, tag retention rates, and contribution of enhanced fish (otolith samples) at enumeration weirs;
- Genetic stock composition of marine non-terminal harvest that may intercept Taku River Sockeye salmon; and
- Continue radiotelemetry studies to assess annual dropout rates and to reassess the long-term average dropout based on historical handing methods at the fish wheels (Bednarski *et al*. 2019).

5.3 GUIDANCE FOR FUTURE WORK

As noted by Pestal and Johnston (2015), a document summarizing guidelines for the development of biological benchmarks and management reference points for U.S/Canada. transboundary river stocks would reconcile the DFO and ADF&G frames of reference into a single transboundary policy statement.

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Julie Bednarski – ADF&G – Fisheries Biologist - Co-chair
Aaron Foos – DFO – Sr. Aquatic Science Biologist - Co-chair
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8 TABLES

Table 1: Matching three Frames of Reference for spawner abundance of Pacific salmon. Evaluations are based on a chronic inability to meet the target (ADF&G) or average generation time (DFO). In practice, both definitions result in a 4–5 year time window for most salmon stocks. Note that DFO Status Zones for relative abundance only reflect one of a suite of status metrics that need to be evaluated together. Sources and definitions listed in Section 1.2.

ADF&G Level of Concern	DFO Status Zones
No management concern: Escapements within or above the escapement goal range	Green: No concern indicated by Relative Abundance metric, but integrated status not automatically green. (> 80% S_{MSY})
Management concern: Escapements fall below the lower bound of the escapement goal range	Amber: Escapements >S_{GEN} but <80% S_{MSY}
Conservation concern	Red: Relative abundance metric falls below S_{GEN}

Table 2: Comparison of ADF&G and DFO salmon policy implementation. References for recent implementation examples are listed in Section 1.2.

Aspect	ADF&G	DFO
Focus of recent work	Spawning goals	Biological status (status benchmarks explicitly defined as NOT being goals)
Stock level	Stocks delineated based on management and assessment	Conservation Units delineated based on biological characteristics (genetics, life history, migration timing, freshwater and marine adaptive zones)
Assessment Process	Peer-reviewed status assessments (ADF&G Fishery Manuscript Series)	Peer-review of available data and status metrics, followed by expert workshop to develop integrated status designations (CSAS Research Documents)
Harvest Planning Process	Status assessments reviewed by the Alaska Board of Fisheries	IFMP, IHPC consider status, plan accordingly

Table 3: Estimated harvest of Taku River Sockeye salmon and harvest proportions by age in the U.S. District 111 traditional commercial drift gillnet fishery, 1983–2018.

		Harvest Proportion by Age						
Year	Harvest	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8
1983	23,460	---	---	---	---	---	---	---
1984	57,619	---	---	---	---	---	---	---
1985	73,367	---	---	---	---	---	---	---
1986	60,644	0.000	0.043	0.272	0.585	0.099	0.000	0.000
1987	54,963	0.003	0.033	0.345	0.564	0.055	0.000	0.000
1988	25,785	0.001	0.057	0.412	0.500	0.030	0.000	0.000
1989	62,804	0.000	0.000	0.229	0.698	0.071	0.000	0.000
1990	108,492	0.000	0.010	0.197	0.708	0.085	0.000	0.000
1991	104,471	0.000	0.017	0.237	0.662	0.084	0.000	0.000
1992	119,959	0.000	0.007	0.232	0.678	0.083	0.000	0.000
1993	140,888	0.000	0.008	0.239	0.690	0.062	0.001	0.000
1994	96,952	0.000	0.005	0.163	0.776	0.056	0.000	0.000
1995	86,929	0.000	0.068	0.240	0.639	0.054	0.000	0.000
1996	181,776	0.000	0.008	0.452	0.507	0.033	0.000	0.000
1997	76,043	0.000	0.010	0.213	0.638	0.139	0.000	0.000
1998	47,824	0.000	0.011	0.156	0.729	0.103	0.001	0.000
1999	61,205	0.000	0.014	0.372	0.577	0.036	0.000	0.000
2000	128,567	0.000	0.021	0.372	0.540	0.066	0.000	0.000
2001	194,091	0.000	0.002	0.249	0.723	0.026	0.000	0.000
2002	114,460	0.000	0.015	0.262	0.681	0.042	0.000	0.000
2003	134,957	0.000	0.013	0.157	0.807	0.024	0.000	0.000
2004	75,186	0.000	0.024	0.410	0.533	0.033	0.000	0.000
2005	44,360	0.000	0.023	0.297	0.657	0.023	0.000	0.000
2006	62,814	0.000	0.019	0.170	0.766	0.045	0.000	0.000
2007	60,879	0.000	0.029	0.292	0.613	0.065	0.000	0.000
2008	63,002	0.000	0.012	0.306	0.649	0.033	0.000	0.000
2009	35,121	0.000	0.007	0.229	0.688	0.076	0.000	0.000
2010	44,837	0.000	0.008	0.443	0.520	0.029	0.000	0.000
2011	65,090	0.000	0.010	0.137	0.799	0.054	0.000	0.000
2012	45,410	0.000	0.005	0.388	0.561	0.046	0.000	0.000
2013	84,567	0.000	0.004	0.203	0.701	0.091	0.000	0.000
2014	30,672	0.000	0.085	0.410	0.461	0.045	0.000	0.000
2015	40,904	0.000	0.015	0.529	0.431	0.025	0.000	0.000
2016	66,980	0.000	0.017	0.492	0.472	0.019	0.000	0.000
2017	67,706	0.001	0.010	0.179	0.762	0.049	0.000	0.000
2018	24,472	0.000	0.141	0.340	0.434	0.085	0.000	0.000

Table 4: Estimated harvest of Taku River Sockeye salmon and harvest proportions by age in the inriver Canadian commercial fishery, 1980–2018.

		Harvest Proportion by Age						
Year	Harvest	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8
1980	22,602	0.000	0.011	0.209	0.692	0.089	0.000	0.000
1981	10,922	0.000	0.003	0.142	0.788	0.067	0.000	0.000
1982	3,144	0.000	0.016	0.126	0.770	0.087	0.000	0.000
1983	17,056	0.000	0.013	0.245	0.668	0.074	0.000	0.000
1984	27,242	0.000	0.008	0.227	0.721	0.044	0.000	0.000
1985	14,244	0.000	0.036	0.186	0.749	0.029	0.000	0.000
1986	14,739	0.000	0.023	0.254	0.625	0.099	0.000	0.000
1987	13,554	0.000	0.014	0.277	0.663	0.046	0.000	0.000
1988	12,014	0.033	0.031	0.333	0.559	0.043	0.001	0.000
1989	18,545	0.000	0.023	0.246	0.688	0.044	0.000	0.000
1990	21,100	0.000	0.021	0.221	0.698	0.059	0.000	0.000
1991	25,067	0.000	0.041	0.322	0.565	0.071	0.000	0.000
1992	29,472	0.000	0.014	0.311	0.625	0.050	0.000	0.000
1993	33,217	0.000	0.011	0.227	0.696	0.067	0.000	0.000
1994	28,762	0.000	0.009	0.182	0.754	0.055	0.000	0.000
1995	32,640	0.000	0.096	0.234	0.605	0.064	0.000	0.001
1996	41,665	0.000	0.022	0.396	0.556	0.026	0.000	0.000
1997	24,003	0.000	0.012	0.277	0.622	0.087	0.002	0.000
1998	19,038	0.000	0.019	0.147	0.758	0.076	0.000	0.000
1999	20,681	0.000	0.039	0.431	0.500	0.030	0.001	0.000
2000	28,009	0.000	0.066	0.449	0.450	0.035	0.000	0.000
2001	47,660	0.000	0.006	0.234	0.718	0.042	0.000	0.000
2002	31,053	0.000	0.057	0.398	0.511	0.034	0.000	0.000
2003	32,730	0.000	0.027	0.141	0.802	0.030	0.000	0.000
2004	20,148	0.000	0.036	0.400	0.558	0.005	0.000	0.000
2005	21,697	0.000	0.011	0.208	0.762	0.020	0.000	0.000
2006	21,099	0.002	0.040	0.205	0.711	0.042	0.000	0.000
2007	16,714	0.002	0.079	0.411	0.454	0.052	0.000	0.000
2008	19,284	0.000	0.024	0.299	0.654	0.023	0.000	0.000
2009	10,980	0.000	0.041	0.354	0.551	0.054	0.000	0.000
2010	20,180	0.000	0.030	0.497	0.461	0.012	0.000	0.000
2011	23,898	0.000	0.033	0.202	0.743	0.023	0.000	0.000
2012	29,938	0.000	0.026	0.569	0.382	0.023	0.000	0.000
2013	25,074	0.000	0.009	0.218	0.727	0.043	0.000	0.002
2014	17,568	0.000	0.195	0.441	0.345	0.018	0.000	0.000
2015	19,715	0.000	0.019	0.559	0.411	0.010	0.000	0.000
2016	37,120	0.000	0.036	0.642	0.309	0.012	0.000	0.000
2017	30,150	0.000	0.022	0.188	0.750	0.040	0.000	0.000
2018	17,948	0.000	0.270	0.342	0.346	0.040	0.002	0.000

Table 5: Summary of the updated and previous published (historic) capture-recapture abundance estimates for 1984–2018 (excluding 1986). Updated capture-recapture abundance estimates from 1984 through 2002 (excluding 1986) were based on pooled Petersen estimates adjusted downward by 6.4% which represented the average bias observed between size-stratified and non-size-stratified pooled Petersen estimates from 2003 to 2018. Updated capture-recapture abundance estimates from 2003 onward were based on annually size-stratified Petersen estimates. Release and recapture data were unavailable for 1986, therefore an updated capture-recapture abundance estimate was not available. For years 2016 and prior, a weighted-average dropout rate of 25.5% was applied to the updated capture-recapture abundance estimates, which decreased the estimates. Year-specific dropout rates were applied to updated capture-recapture abundance estimates in 2017 (32.1%) and 2018 (14.6%). The previously published estimates are from the TTC (2019b) report (columns six and eight). The 'previously published estimate (adjusted)' is the published estimate adjusted by the year-specific dropout rate (column four) and by the size selectivity bias (column five). In some years, an expansion factor was first applied to the published estimate (column 10). These expansion factors were applied to the previously published estimates (column 8) before the adjustments for dropout rate and size selectivity (column 9). For example, 133,414 fish (value in 1984) expanded by ~5.6% (expansion factor) is 141,254 fish. If 141,254 is reduced by 25.5% and then by 6.4%, the result is 98,494 fish.

							Expanded		
Year	Updated pooled Petersen capture-recapture abundance estimate	Standard deviation	Dropout rate adjustment	Size selectivity adjustment	Previously published estimate	Previously published estimate (adjusted)	Previously published estimate	Previously published estimate (adjusted)	Expansion factor
1984	88,272	8,689	-0.255	-0.064	133,414	93,027	141,254	98,494	0.056
1985	84,479	8,573	-0.255	-0.064	118,160	82,391	123,974	86,445	0.047
1986	---	----	----	----	104,162	72,634	115,045	80,223	0.095
1987	56,362	5,386	-0.255	-0.064	87,554	61,050	96,023	66,955	0.088
1988	55,580	5,466	-0.255	-0.064	86,629	60,405	92,641	64,597	0.065
1989	80,997	7,605	-0.255	-0.064	99,467	69,356	114,068	79,537	0.128
1990	75,801	6,981	-0.255	-0.064	117,385	81,850	117,573	81,981	0.002
1991	104,895	9,899	-0.255	-0.064	153,773	107,223	154,873	107,990	0.007
1992	99,643	9,121	-0.255	-0.064	162,003	112,961	167,376	116,708	0.032
1993	92,933	8,351	-0.255	-0.064	138,523	96,589	142,148	99,117	0.026
1994	90,128	8,231	-0.255	-0.064	129,119	90,032	131,580	91,748	0.019
1995	104,242	9,531	-0.255	-0.064	145,264	101,290	146,450	102,117	0.008
1996	97,477	8,788	-0.255	-0.064	132,322	92,265	134,651	93,889	0.017
1997	73,255	6,697	-0.255	-0.064	93,816	65,416	95,438	66,547	0.017
1998	64,755	6,069	-0.255	-0.064	89,992	62,750	89,992	62,750	----
1999	83,588	7,886	-0.255	-0.064	113,706	79,285	113,706	79,285	----

Year	Updated pooled Petersen capture-recapture abundance estimate	Standard deviation	Dropout rate adjustment	Size selectivity adjustment	Previously published estimate	Previously published estimate (adjusted)	Expanded: Previously published estimate	Expanded: Previously published estimate (adjusted)	Expansion factor
2000	83,190	7,583	-0.255	-0.064	115,693	80,670	115,693	80,670	----
2001	132,502	12,049	-0.255	-0.064	192,245	134,049	192,245	134,049	----
2002	94,605	8,637	-0.255	-0.064	135,233	94,295	135,233	94,295	----
2003	133,593	12,338	-0.255	-0.064	193,390	134,847	193,390	134,847	----
2004	85,257	7,828	-0.255	-0.064	127,047	88,587	127,047	88,587	----
2005	87,496	8,521	-0.255	-0.064	142,155	99,122	142,155	99,122	----
2006	106,545	10,175	-0.255	-0.064	167,597	116,862	167,597	116,862	----
2007	60,320	5,352	-0.255	-0.064	104,815	73,085	105,012	73,223	0.002
2008	78,031	7,647	-0.255	-0.064	84,073	58,622	87,568	61,059	0.040
2009	59,817	6,237	-0.255	-0.064	83,028	57,894	83,097	57,942	0.001
2010	80,747	8,034	-0.255	-0.064	103,257	71,999	109,028	76,023	0.053
2011	82,116	7,741	-0.255	-0.064	139,926	97,568	139,926	97,568	----
2012	102,670	9,534	-0.255	-0.064	155,590	108,490	156,877	109,387	0.008
2013	88,535	8,506	-0.255	-0.064	96,928	67,586	106,350	74,156	0.089
2014	68,532	6,357	-0.255	-0.064	109,984	76,690	109,984	76,690	----
2015	102,506	10,262	-0.255	-0.064	150,483	104,929	152,372	106,246	0.012
2016	146,294	13,284	-0.255	-0.064	213,851	149,114	216,536	149,114	----
2017	91,164	5,030	-0.321	-0.064	138,518	87,979	138,796	88,155	0.002
2018	84,806	5,206	-0.146	-0.064	135,351	108,135	136,995	109,448	0.012

Table 6: Taku River Sockeye salmon inriver pooled Petersen capture-recapture abundance estimates and estimated age compositions of Taku River Sockeye salmon captured at the Canyon Island fish wheel, 1983–2018. The age composition at the fish wheel was weighted by the inriver capture-recapture abundance estimates. Note that these estimates differ from previously published estimates in the TTC (2019b) report.

		Proportion by Age						
Year	Pooled Petersen Capture-Recapture Abundance Estimates	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8
1983	---	0.000	0.018	0.437	0.484	0.058	0.004	---
1984	88,273	0.002	0.039	0.268	0.662	0.029	0.000	---
1985	84,479	0.003	0.100	0.214	0.628	0.055	0.000	---
1986	---	0.000	0.032	0.359	0.526	0.083	0.000	---
1987	56,362	0.008	0.060	0.319	0.565	0.048	0.000	---
1988	55,580	0.003	0.127	0.380	0.442	0.048	0.000	---
1989	80,998	0.003	0.072	0.268	0.615	0.042	0.000	---
1990	75,801	0.004	0.084	0.312	0.549	0.050	0.000	---
1991	104,896	0.001	0.111	0.416	0.426	0.046	0.000	---
1992	99,643	0.002	0.100	0.396	0.449	0.053	0.000	---
1993	92,933	0.002	0.074	0.272	0.603	0.049	0.000	---
1994	90,129	0.009	0.102	0.267	0.591	0.030	0.000	---
1995	104,242	0.003	0.176	0.369	0.422	0.030	0.000	---
1996	97,478	0.000	0.051	0.355	0.571	0.023	0.000	---
1997	73,255	0.001	0.034	0.376	0.513	0.076	0.000	---
1998	64,756	0.001	0.075	0.261	0.617	0.046	0.000	---
1999	83,588	0.009	0.082	0.543	0.346	0.020	0.000	---
2000	83,190	0.000	0.075	0.433	0.470	0.022	0.000	---
2001	132,503	0.005	0.106	0.315	0.559	0.016	0.000	---
2002	94,606	0.003	0.117	0.398	0.461	0.020	0.000	---
2003	133,594	0.004	0.102	0.326	0.555	0.014	0.000	---
2004	85,258	0.003	0.082	0.470	0.425	0.020	0.000	---
2005	87,496	0.006	0.068	0.389	0.516	0.020	0.000	---
2006	106,545	0.002	0.096	0.327	0.548	0.027	0.000	---
2007	60,321	0.003	0.098	0.467	0.391	0.041	0.000	---
2008	78,031	0.005	0.092	0.312	0.568	0.023	0.000	---
2009	59,818	0.004	0.179	0.318	0.453	0.046	0.000	---
2010	80,747	0.006	0.097	0.529	0.362	0.005	0.000	---
2011	82,117	0.005	0.108	0.220	0.609	0.058	0.000	---
2012	102,671	0.003	0.057	0.586	0.335	0.018	0.001	---
2013	88,536	0.013	0.085	0.289	0.554	0.060	0.000	---
2014	68,533	0.007	0.342	0.390	0.241	0.019	0.001	---
2015	102,506	0.004	0.101	0.509	0.375	0.011	0.000	---
2016	146,294	0.002	0.061	0.616	0.305	0.017	0.000	---
2017	91,164	0.002	0.042	0.250	0.663	0.042	0.000	---
2018	84,807	0.000	0.336	0.431	0.195	0.039	0.000	---

Table 7: Prior distributions for model parameters. The parameter ϕ was constrained to be between -1.0 and 1.0. For the base prior, the ln(α) and β parameters were constrained with a lower bound of 1.00 x 10^{-6}. The JAGS input for the dispersion parameter of a normal prior is the precision, which is the inverse of the variance.

Parameter	rjags Package Coding	Prior	Alternative priors
$\ln(\alpha)$	lnalpha	$\ln(\alpha)$ ~ Normal (0, 1.00 x 10^{-6})	-
β	beta	β_{Base} ~ Normal (0, 1.00 x 10^{-6})	β_1~ Uniform (1.00 x 10^{-6},1.0) β_2 ~ Normal (0, 1.00 x 10^{-6})
σ_R	sigma.R	$1/\sigma_R^2$ ~ gamma (0.001, 0.001)	-
ϕ	phi	ϕ~ Normal (0, 1.00 x 10^{-6})	-
ω_0	log.resid.0	$\omega_o \sim \text{Normal}\ (0, \sigma_R^2/(1-\phi^2))$	-
D	D.sum	$1/\sqrt{D}$~Uniform (0, 1)	-
$\ln(R_0)$	mean.log.R0	$\ln(R_0)$ ~ Normal (0, 1.00 x 10^{-6})	-
σ_{R0}	sigma.R0	$1/\sigma_{R0}^2$~gamma(0.001,0.001)	-
R_1: R_6	R[1:6]	lognormal($\ln(R_0)$,σ_{R0}^2)	-

Table 8: Data used in the sensitivity analysis. In the table, 1984–1985 and 1987–2018 are 'ir' are the pooled Petersen capture-recapture estimates of above-border abundance with the associated coefficients of variation (ir.cv). There was no estimate for 1986. These pooled Petersen capture-recapture estimates are from Pestal et al. (2020) and differ from the published estimates in the TTC (2019b) report. The 'ir' for years 1980–1983, and 1986 in scenarios 1a, 1b, and 1c, are the abundance estimates output from the base model. The coefficients of variation vary by the scenario (0.90, 0.50, and 0.10, respectively). The 'ir' for years 1980–1983, and 1986 in scenario 2 are the abundance estimates output from the base model multiplied by 0.75. The coefficients of variation are set to 0.10 for these years. The 'ir' for years 1980–1983, and 1986 in scenario 3 are the abundance estimates output from the base model multiplied by 1.33. The coefficients of variation are set to 0.10 for these years.

		scenario 1a	scenario 1b	scenario 1c	scenario 2		scenario 3	
year	ir	cv.ir	cv.ir	cv.ir	ir	cv.ir	ir	cv.ir
1980	55,179	0.90	0.50	0.10	41,384	0.10	73,388	0.10
1981	38,444	0.90	0.50	0.10	28,833	0.10	51,130	0.10
1982	27,149	0.90	0.50	0.10	20,362	0.10	36,108	0.10
1983	51,747	0.90	0.50	0.10	38,810	0.10	68,823	0.10
1984	88,273	0.10	0.10	0.10	88,273	0.10	88,273	0.10
1985	84,479	0.10	0.10	0.10	84,479	0.10	84,479	0.10
1986	50,926	0.90	0.50	0.10	38,195	0.10	67,732	0.10
1987	56,362	0.10	0.10	0.10	56,362	0.10	56,362	0.10
1988	55,580	0.10	0.10	0.10	55,580	0.10	55,580	0.10
1989	80,998	0.09	0.09	0.09	80,998	0.09	80,998	0.09
1990	75,801	0.09	0.09	0.09	75,801	0.09	75,801	0.09
1991	104,896	0.09	0.09	0.09	104,896	0.09	104,896	0.09
1992	99,643	0.09	0.09	0.09	99,643	0.09	99,643	0.09
1993	92,933	0.09	0.09	0.09	92,933	0.09	92,933	0.09
1994	90,129	0.09	0.09	0.09	90,129	0.09	90,129	0.09
1995	104,242	0.09	0.09	0.09	104,242	0.09	104,242	0.09
1996	97,478	0.09	0.09	0.09	97,478	0.09	97,478	0.09
1997	73,255	0.09	0.09	0.09	73,255	0.09	73,255	0.09
1998	64,756	0.09	0.09	0.09	64,756	0.09	64,756	0.09
1999	83,588	0.09	0.09	0.09	83,588	0.09	83,588	0.09
2000	83,190	0.09	0.09	0.09	83,190	0.09	83,190	0.09
2001	132,503	0.09	0.09	0.09	132,503	0.09	132,503	0.09
2002	94,606	0.09	0.09	0.09	94,606	0.09	94,606	0.09
2003	133,594	0.09	0.09	0.09	133,594	0.09	133,594	0.09
2004	85,258	0.09	0.09	0.09	85,258	0.09	85,258	0.09
2005	87,496	0.10	0.10	0.10	87,496	0.10	87,496	0.10
2006	106,545	0.10	0.10	0.10	106,545	0.10	106,545	0.10
2007	60,321	0.09	0.09	0.09	60,321	0.09	60,321	0.09
2008	78,031	0.10	0.10	0.10	78,031	0.10	78,031	0.10
2009	59,818	0.10	0.10	0.10	59,818	0.10	59,818	0.10
2010	80,747	0.10	0.10	0.10	80,747	0.10	80,747	0.10
2011	82,117	0.09	0.09	0.09	82,117	0.09	82,117	0.09
2012	102,671	0.09	0.09	0.09	102,671	0.09	102,671	0.09
2013	88,536	0.10	0.10	0.10	88,536	0.10	88,536	0.10

		scenario 1a	scenario 1b	scenario 1c	scenario 2		scenario 3	
year	ir	cv.ir	cv.ir	cv.ir	ir	cv.ir	ir	cv.ir
2014	68,533	0.09	0.09	0.09	68,533	0.09	68,533	0.09
2015	102,506	0.10	0.10	0.10	102,506	0.10	102,506	0.10
2016	146,294	0.09	0.09	0.09	146,294	0.09	146,294	0.09
2017	91,164	0.06	0.06	0.06	91,164	0.06	91,164	0.06
2018	84,807	0.06	0.06	0.06	84,807	0.06	84,807	0.06

Table 9: Annual abundance estimates for Taku River Sockeye salmon obtained by fitting a state-space model to data for calendar years 1980–2018. Point estimates are posterior medians and coefficients of variation are the posterior standard deviations divided by the posterior means. Recruitment values are listed by brood year. Years with higher uncertainty corresponded to years with missing capture-recapture abundance estimates (1980–1983, 1986), and missing age composition data (1980–1982). Log transformed initial recruitments R_{1974}–R_{1979} were not linked to spawner abundance. The years with missing data are shaded.

				Coefficients of Variation		
Year	Terminal Run *N*	Escapement *S*	Recruitment *R*	Terminal Run *N*	Escapement S	Recruitment *R*
1974	---	---	92,107	---	---	2.12
1975	---	---	89,052	---	---	0.60
1976	---	---	76,712	---	---	0.54
1977	---	---	75,484	---	---	0.51
1978	---	---	66,889	---	---	0.42
1979	---	---	132,073	---	---	0.14
1980	85,531	32,407	156,254	0.43	0.62	0.08
1981	76,265	27,408	121,560	0.45	0.71	0.12
1982	72,812	24,075	102,843	0.46	0.77	0.10
1983	75,550	34,562	86,662	0.37	0.72	0.09
1984	141,854	56,713	140,563	0.06	0.15	0.07
1985	153,351	66,095	172,768	0.06	0.12	0.07
1986	111,571	36,106	180,950	0.15	0.44	0.08
1987	109,141	40,471	217,425	0.05	0.12	0.08
1988	84,958	45,860	235,771	0.07	0.12	0.07
1989	142,426	60,762	203,578	0.05	0.12	0.08
1990	181,937	52,450	159,288	0.05	0.13	0.09
1991	203,708	74,518	250,773	0.05	0.12	0.08
1992	216,690	67,339	208,973	0.05	0.13	0.08
1993	230,899	57,220	124,394	0.05	0.15	0.08
1994	185,980	60,144	104,647	0.05	0.14	0.09
1995	188,863	69,229	188,877	0.05	0.13	0.08
1996	271,568	51,908	310,398	0.04	0.16	0.07
1997	150,725	49,692	230,150	0.05	0.13	0.08
1998	113,989	46,538	262,705	0.06	0.13	0.07
1999	142,912	60,424	160,773	0.05	0.12	0.09
2000	210,923	53,832	154,616	0.04	0.14	0.08
2001	316,357	76,982	163,022	0.05	0.14	0.07
2002	210,083	63,405	122,845	0.05	0.14	0.08
2003	257,332	91,294	141,320	0.05	0.12	0.07
2004	159,665	64,029	104,664	0.05	0.12	0.09
2005	134,367	67,425	99,608	0.06	0.13	0.09
2006	163,016	79,063	171,313	0.06	0.11	0.07
2007	121,172	42,850	112,096	0.05	0.12	0.10
2008	137,958	55,759	192,775	0.06	0.13	0.07
2009	95,900	49,164	87,993	0.07	0.13	0.11
2010	124,962	59,311	125,242	0.06	0.13	0.08
2011	145,489	55,687	167,007	0.05	0.13	0.09
2012	145,562	69,695	239,049	0.06	0.12	0.06
2013	167,680	57,774	76,426	0.06	0.16	0.13

				Coefficients of Variation		
Year	Terminal Run *N*	Escapement *S*	Recruitment *R*	Terminal Run *N*	Escapement S	Recruitment *R*
2014	101,138	51,846	168,832	0.06	0.12	0.18
2015	141,543	80,770	---	0.07	0.12	---
2016	206,949	102,456	---	0.06	0.12	---
2017	158,205	60,125	---	0.04	0.09	---
2018	110,162	67,597	---	0.05	0.08	---
Average	155,107	58,025	153,134	0.09	0.19	0.18
Minimum	72,812	24,075	66,889	0.04	0.08	0.06
Maximum	316,357	102,456	310,398	0.46	0.77	2.12

Table 10: Terminal run abundance by age ($N_{y,a}$) obtained by fitting a state-space model to data from Taku River Sockeye salmon for calendar years 1980–2018. Point estimates are posterior medians and coefficients of variation are the posterior standard deviations divided by the posterior means. Note: The $N_y = \sum_{a=4}^{6} N_{y,a}$ for the mean values of $N_{y,a}$.

				Coefficients of Variation		
Year	Ages 2-4	Age 5	Ages 6-8	Ages 2-4	Age 5	Ages 6-8
1980	30,201	48,180	3,889	0.57	0.60	1.40
1981	29,262	41,578	3,746	0.55	0.56	0.94
1982	26,750	41,051	3,241	0.53	0.54	0.88
1983	35,936	35,519	4,069	0.35	0.42	0.55
1984	50,129	87,099	3,876	0.14	0.10	0.47
1985	50,011	95,746	7,189	0.14	0.09	0.37
1986	36,671	65,001	9,867	0.19	0.17	0.31
1987	40,869	62,200	5,770	0.13	0.10	0.36
1988	41,708	39,502	3,438	0.11	0.11	0.40
1989	48,327	87,847	5,885	0.14	0.09	0.37
1990	59,239	111,259	10,513	0.14	0.08	0.33
1991	82,414	108,211	12,290	0.12	0.10	0.33
1992	81,721	121,796	12,759	0.12	0.09	0.32
1993	72,177	145,692	12,616	0.13	0.08	0.34
1994	55,080	122,272	8,200	0.14	0.08	0.40
1995	85,273	94,853	8,035	0.11	0.10	0.38
1996	111,807	150,475	8,537	0.12	0.10	0.44
1997	46,910	88,704	14,540	0.13	0.09	0.27
1998	33,664	72,109	7,916	0.14	0.09	0.32
1999	74,804	62,691	4,898	0.10	0.12	0.43
2000	96,960	105,422	7,909	0.11	0.10	0.38
2001	102,815	205,437	7,721	0.14	0.08	0.48
2002	82,894	118,287	7,834	0.12	0.09	0.43
2003	76,284	172,571	7,431	0.14	0.08	0.48
2004	72,836	80,473	6,186	0.11	0.10	0.43
2005	54,894	75,148	3,848	0.12	0.10	0.48
2006	55,222	101,353	5,824	0.13	0.09	0.43
2007	52,377	62,327	6,177	0.11	0.10	0.37
2008	49,485	83,092	4,918	0.13	0.09	0.42
2009	37,821	52,014	5,557	0.12	0.11	0.34
2010	67,996	53,893	2,526	0.10	0.12	0.53
2011	40,321	97,399	7,214	0.15	0.08	0.34
2012	79,136	60,716	5,183	0.10	0.12	0.43
2013	46,899	109,489	10,820	0.16	0.09	0.32
2014	59,511	37,280	3,958	0.10	0.13	0.44
2015	77,716	60,008	3,506	0.11	0.12	0.47
2016	118,956	82,333	4,935	0.10	0.12	0.49
2017	37,351	113,714	6,632	0.16	0.07	0.39
2018	69,871	33,906	6,023	0.09	0.14	0.37

Table 11: Parameter estimates from the autoregressive state-space Ricker model for Taku River Sockeye salmon in calendar years 1980–2018. Posterior medians are point estimates; the 2.5th and 97.5th percentiles define 95% credible intervals for the parameters and the 5th and 95th percentiles define the 90% credible intervals (parameter definitions are in the Methods section). Coefficients of variation (CVs) are the posterior standard deviations divided by the posterior means.

Parameter	2.5th	5th	Median	95th	97.5th	Posterior CV
α	3.49	3.95	7.63	14.60	16.66	0.42
α'	3.91	4.34	8.17	15.61	17.52	0.45
ln(α)	1.25	1.37	2.03	2.68	2.81	0.20
ln(α')	1.35	1.47	2.11	2.76	2.89	0.19
β_{Base}	4.18 x 10^{-6}	6.06 x 10^{-6}	1.69 x 10^{-5}	2.79 x 10^{-5}	3.01 x 10^{-5}	0.39
ϕ	-0.27	-0.20	0.24	0.63	0.72	1.09
σ_R	0.27	0.28	0.36	0.46	0.48	0.15
S_{EQ}	94,406	97,418	124,106	252,655	327,102	0.64
S_{MAX}	33,210	35,843	59,145	164,901	239,065	0.85
S_{MSY}	28,830	30,422	43,857	99,699	130,640	0.67
$S_{MSY\ 80\%}$	23,064	24,337	35,086	79,760	104,512	0.67
U_{MSY}	0.55	0.59	0.75	0.85	0.87	0.11
S_{GEN}	1,688	1,967	5,873	25,146	38,871	1.26
D	29.08	30.39	46.67	69.82	77.15	0.26
π_4	0.36	0.37	0.39	0.42	0.42	0.04
π_5	0.52	0.53	0.55	0.58	0.58	0.03
π_6	0.04	0.04	0.05	0.06	0.07	0.13

Table 12: Posterior medians and coefficients of variation for key model quantities, with base and alternative versions of prior distributions on the parameter beta.

Parameter	Median	CV	Alternative priors: Medians		Alternative priors: CVs	
			β_1	β_2	β_1	β_2
α	7.63	0.42	7.63	7.56	0.41	0.44
$\ln(\alpha)$	2.03	0.20	2.03	2.02	0.20	0.20
$\ln(\alpha')$	2.11	0.19	2.11	2.10	0.19	0.20
β	1.69×10^{-5}	0.39	1.68×10^{-5}	1.68×10^{-5}	0.39	0.40
S_{EQ}	124,106	0.64	124,370	123,505	0.49	3.67
S_{MAX}	59,145	0.85	59,509	59,197	0.83	5.37
S_{MSY}	43,857	0.67	44,032	43,692	0.59	4.05
$S_{MSY\ 80\%}$	35,086	0.67	35,226	34,953	0.59	4.05
U_{MSY}	0.75	0.11	0.74	0.74	0.11	0.12

Note:

β_{Base}~ Normal (0, 1.00 x 10^{-6}) and constrained to be greater than 1.00 x 10^{-6}

β_1~ Uniform (1.00 x 10^{-6}, 1.0)

β_2 ~ Normal (0, 1.00 x 10^{-6})

Table 13: Posterior medians and coefficients of variation for key model quantities, with base and alternative versions of assumptions on the early years of capture-recapture abundance estimates. The coefficients of variation for the S_{MSY} reference point for the different scenarios are in parentheses. Sensitivity analyses were performed on the early years of missing capture-recapture abundance estimates. In all scenarios, 1984–1985 and 1987–2018 were the pooled Petersen capture-recapture estimates of above-border abundance with the associated coefficients of variation (ir.cv) from Pestal et al. (2020). There was no estimate for 1986. Five scenarios were explored. In scenarios 1a, 1b, and 1c, years 1980–1983, and 1986, were the capture-recapture abundance estimates output from the base model. The coefficients of variation varied by the scenario (0.90, 0.50, and 0.10, respectively). The capture-recapture abundance estimates (ir) for years 1980–1983, and 1986 in scenario two were the abundance estimates output from the base model multiplied by 0.75. The coefficients of variation were set to 0.10 for these years. The capture-recapture abundance estimates (ir) for years 1980–1983, and 1986 in scenario three were the abundance estimates output from the base model multiplied by 1.33. The coefficients of variation were set to 0.10 for these years.

Parameter	Base Model: Median	Base Model: CV	Scenario1a	Scenario1b	Scenario1c	Scenario2	Scenario3
α	7.63	0.42	7.06	6.59	5.55	7.66	4.95
$\ln(\alpha)$	2.03	0.20	1.95	1.88	1.71	2.04	1.60
$\ln(\alpha')$	2.11	0.19	2.03	1.96	1.79	2.11	1.68
β	1.69×10^{-5}	0.39	1.56×10^{-5}	1.45×10^{-5}	1.14×10^{-5}	1.66×10^{-5}	9.96×10^{-6}
S_{MSY}	43,857	0.67	46,720 (0.62)	48,991 (0.65)	58,987 (0.72)	44,811 (0.43)	64,412 (0.74)

Table 14: Sensitivity of yield-based reference escapements to alternative definitions. The optimal yield probabilities (probability of achieving at least x% of MSY) are the probabilities that a given level of spawning escapement will produce average yields exceeding 70%, 80%, or 90% of MSY. Overfishing probabilities (probability of overfishing below x% of MSY) are the probabilities that sustained yield is reduced to less than a percentage (70%, 80%, 90%) of MSY. The % probability for the overfishing profiles refers to the lower bound of the escapement goal range. Values that meet the criterion suggested by the authors are shaded. The values in bold are the biological escapement goal range of 40,000–75,000 fish recommended by the Taku River Sockeye Salmon Working Group.

Escapement	**Probability of achieving at least x% of MSY**			**Overfishing probability yields below x% of MSY**		
	90%	**80%**	**70%**	**90%**	**80%**	**70%**
28,000	51%	76%	88%	49%	25%	12%
29,000	56%	78%	89%	44%	22%	11%
30,000	60%	80%	90%	40%	20%	10%
31,000	63%	82%	91%	37%	18%	9%
32,000	67%	84%	92%	33%	16%	8%
33,000	69%	85%	92%	31%	15%	8%
34,000	72%	87%	93%	28%	13%	7%
35,000	74%	88%	93%	26%	12%	7%
36,000	76%	89%	93%	24%	11%	7%
37,000	78%	90%	94%	22%	10%	6%
38,000	80%	90%	94%	20%	10%	6%
39,000	81%	91%	95%	18%	9%	5%
40,000	**82%**	**92%**	**95%**	**17%**	**8%**	**5%**
41,000	83%	92%	95%	16%	8%	5%
42,000	83%	92%	96%	15%	7%	4%
43,000	83%	93%	96%	14%	7%	4%
44,000	83%	93%	96%	13%	7%	4%
45,000	82%	93%	96%	12%	6%	4%
46,000	81%	93%	96%	11%	6%	4%
47,000	81%	93%	96%	10%	6%	4%
48,000	79%	93%	96%	10%	5%	3%
49,000	77%	93%	96%	9%	5%	3%
50,000	75%	92%	96%	9%	5%	3%
51,000	73%	91%	97%	8%	5%	3%
52,000	71%	90%	96%	8%	4%	3%
53,000	68%	89%	96%	8%	4%	3%
54,000	66%	87%	96%	7%	4%	2%
55,000	63%	86%	95%	7%	4%	2%
56,000	61%	84%	94%	7%	4%	2%
57,000	59%	82%	93%	6%	4%	2%
58,000	57%	79%	92%	6%	4%	2%
59,000	54%	77%	91%	6%	3%	2%
60,000	52%	75%	90%	6%	3%	2%
61,000	50%	72%	89%	5%	3%	2%

Escapement	Probability of achieving at least x% of MSY			Overfishing probability yields below x% of MSY		
	90%	**80%**	**70%**	**90%**	**80%**	**70%**
62,000	47%	70%	87%	5%	3%	2%
63,000	45%	68%	85%	5%	3%	2%
64,000	43%	65%	83%	5%	3%	2%
65,000	41%	63%	81%	5%	3%	2%
66,000	39%	61%	79%	4%	3%	2%
67,000	38%	60%	76%	4%	2%	2%
68,000	36%	57%	74%	4%	2%	1%
69,000	34%	55%	72%	4%	2%	1%
70,000	33%	52%	69%	4%	2%	1%
71,000	31%	51%	67%	4%	2%	1%
72,000	29%	48%	65%	4%	2%	1%
73,000	28%	47%	63%	4%	2%	1%
74,000	27%	45%	61%	4%	2%	1%
75,000	**26%**	**43%**	**59%**	**3%**	**2%**	**1%**
76,000	25%	41%	57%	3%	2%	1%
77,000	24%	39%	55%	3%	2%	1%
78,000	23%	38%	52%	3%	2%	1%
79,000	22%	36%	51%	3%	2%	1%
80,000	21%	35%	49%	3%	2%	1%
81,000	20%	33%	47%	3%	2%	1%
82,000	19%	32%	45%	3%	2%	1%
83,000	19%	30%	44%	3%	2%	1%
84,000	18%	29%	42%	3%	1%	1%

9 FIGURES

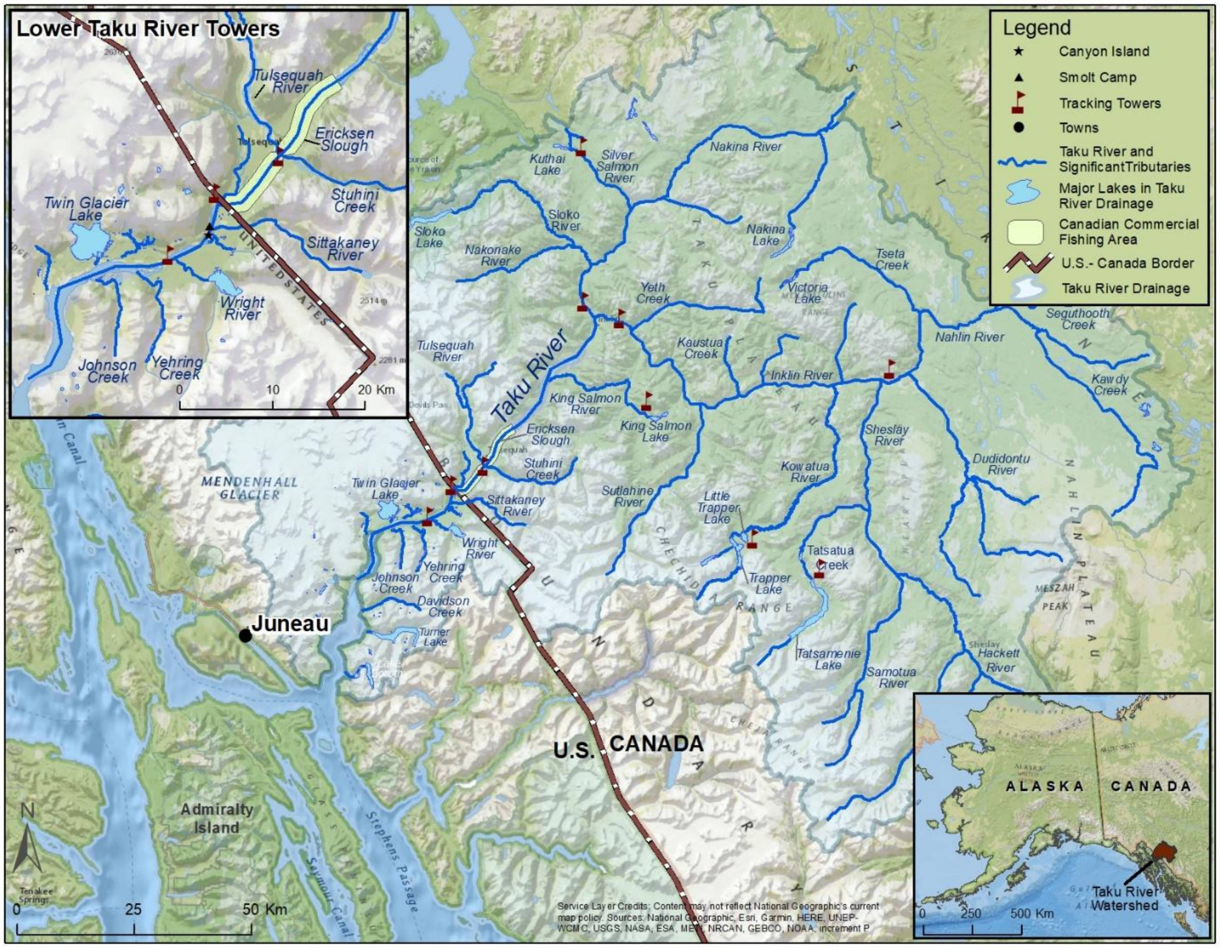

Figure 1: Taku River drainage in Southeast Alaska and British Columbia identifying key landmarks, including the tagging (Canyon Island) and recovery (Canadian fishery) locations of the capture–recapture experiment.

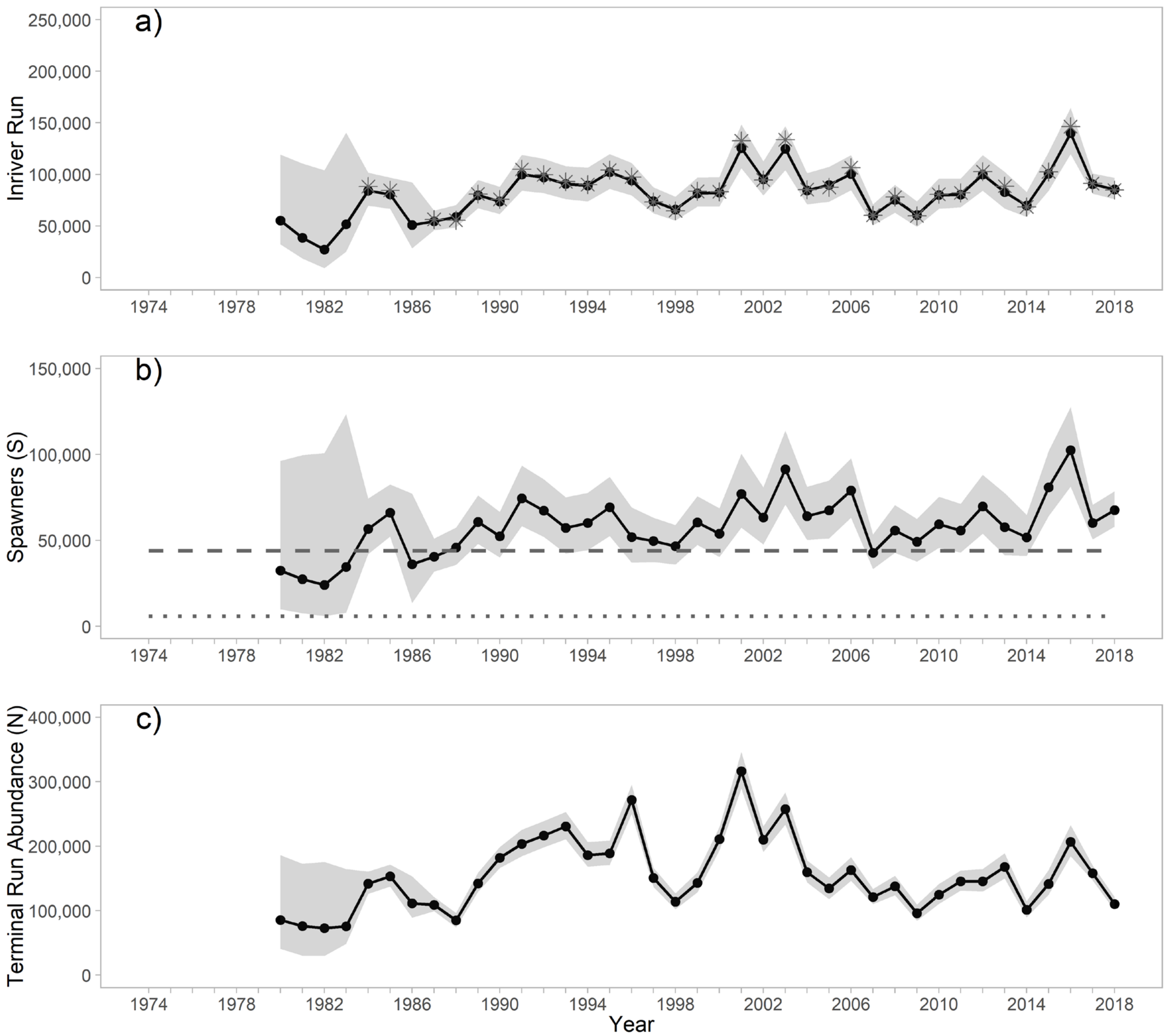

Figure 2: Point estimates (posterior medians; circles with solid lines) and 95% credible intervals (shaded areas) of (A) annual inriver abundance, (B) annual escapement, and (C) terminal run abundance by calendar year t from a Bayesian state-space model of Taku River Sockeye salmon. The posterior median of S_{MSY} is plotted as a dashed horizontal reference line in Figure B and the posterior median of S_{GEN} (based on the Hilborn's approximation) is plotted as a dotted horizontal line in Figure B. The stars in figure A are the observed pooled Petersen capture-recapture abundance estimates.

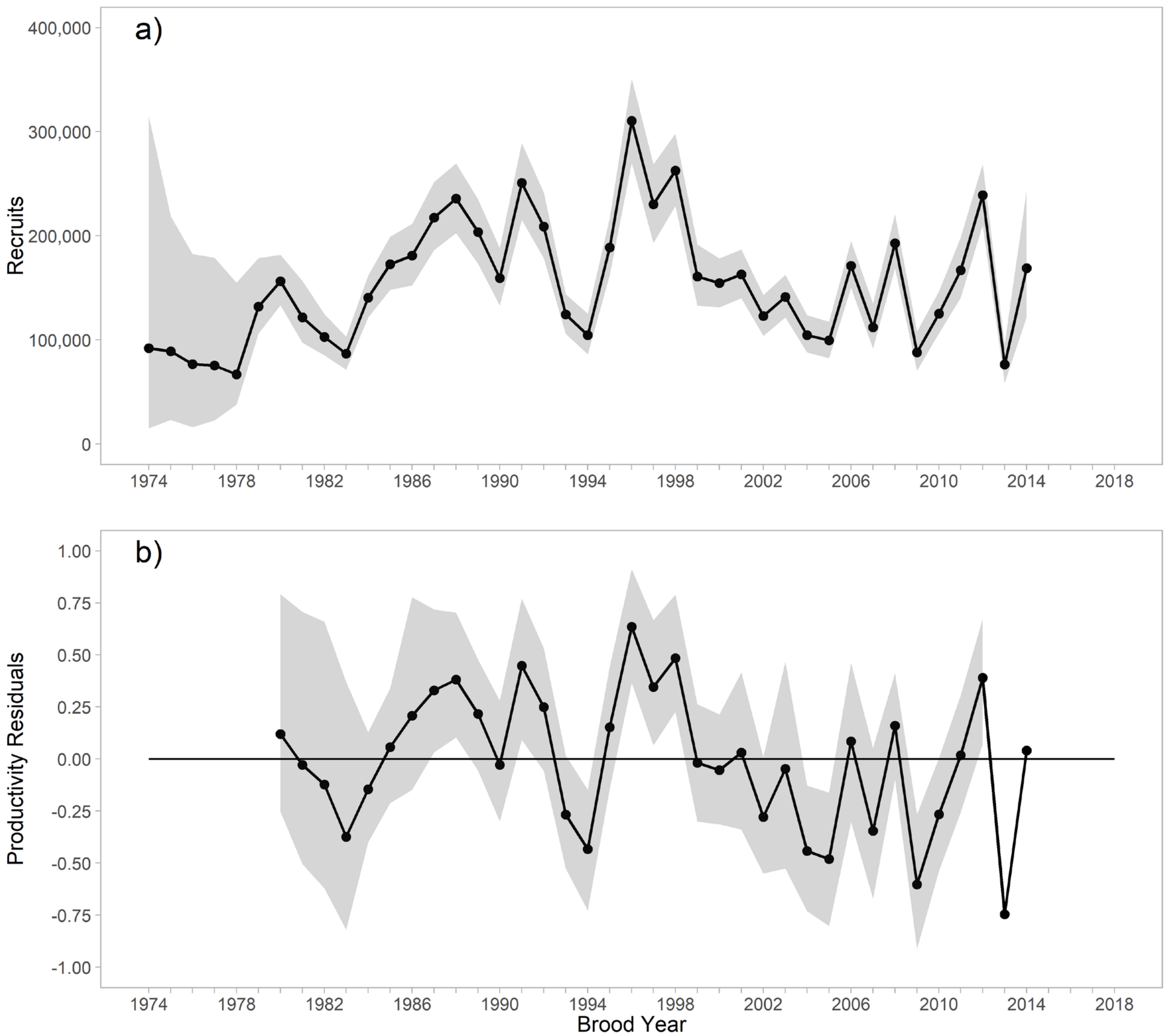

Figure 3: Point estimates (posterior medians; circles with solid lines) and 95% credible intervals (shaded areas) of (A) recruitment and (B) productivity residuals by brood year (y) from a Bayesian state-space model of Taku River Sockeye salmon.

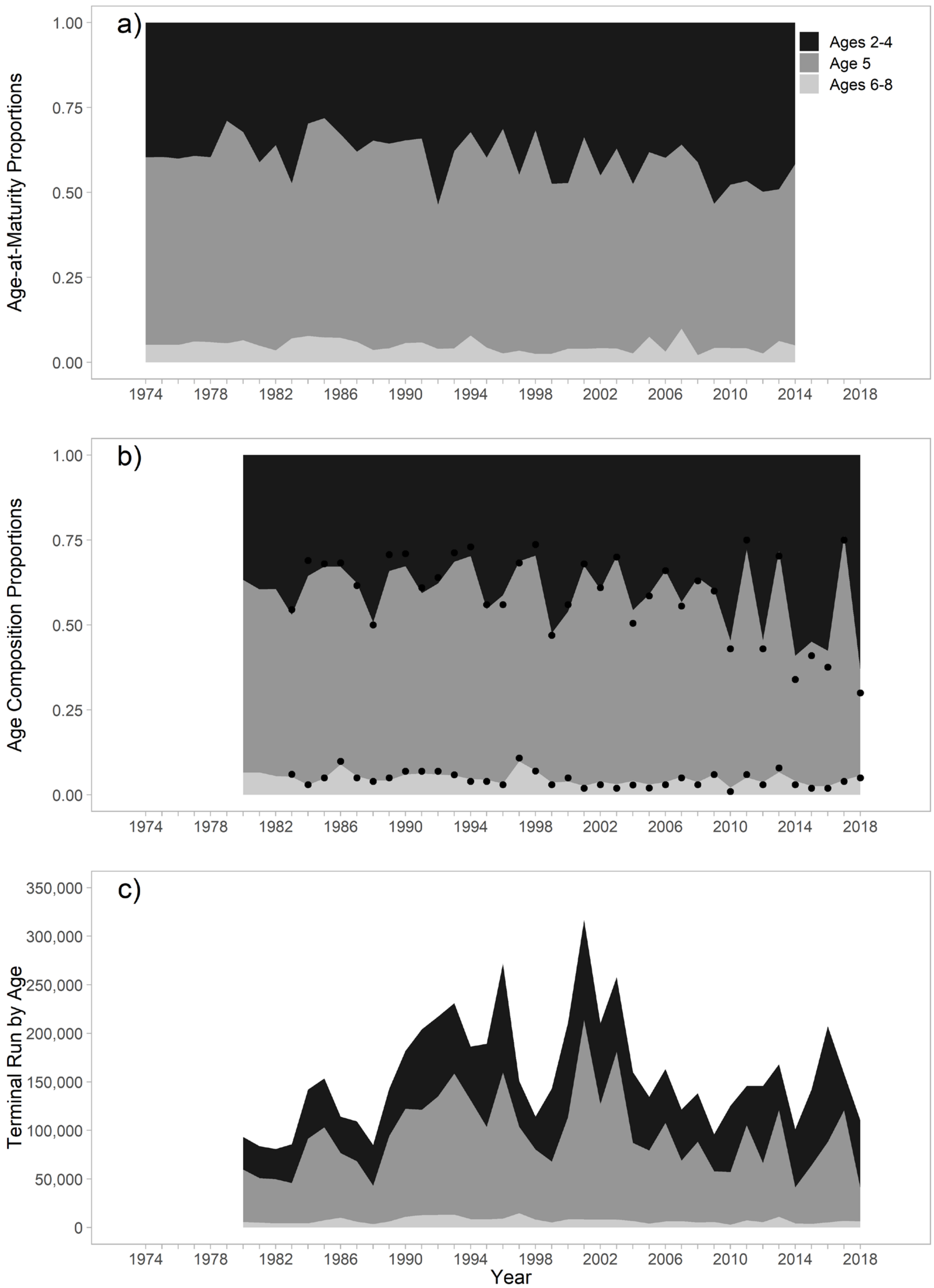

Figure 4: Estimated mean age-at-maturity proportions (p) by brood year (1974–2014; A), mean age composition proportions (q) of annual run by calendar year (1980–2018; B), and mean terminal run by age by calendar year (1980–2018; C), from a Bayesian state-space model fitted to data from Taku River Sockeye salmon. Top and middle figures are area graphs in which the distance between lines represents the age proportions. Dots in the middle plot are data-based estimates of age composition from Appendix A2.

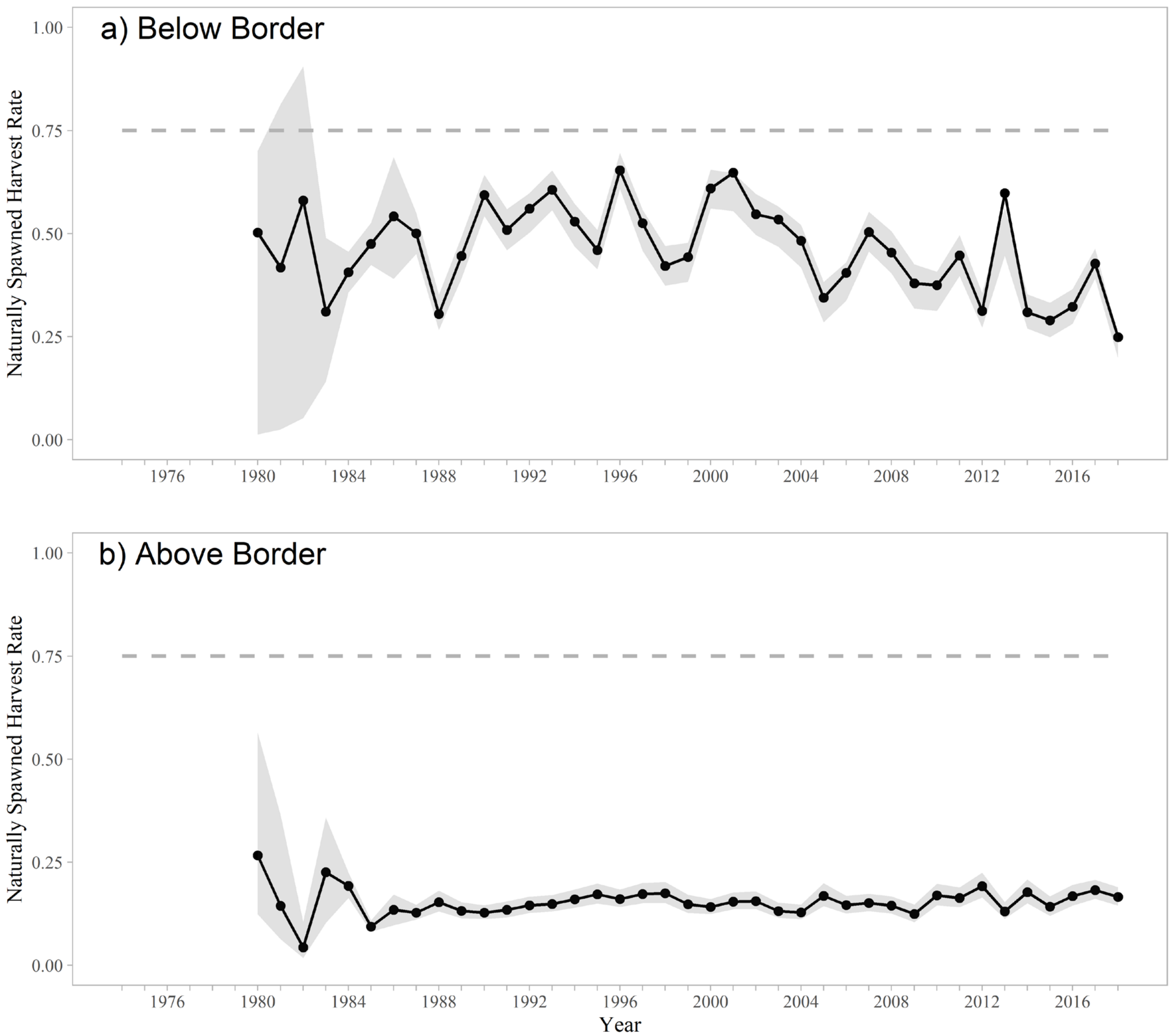

Figure 5: Point estimates (posterior medians; circles with solid lines) and 95% credible intervals (shaded areas) of naturally-spawned harvest rates (A) above and (B) below the border by calendar year from the Bayesian state-space model of Taku River Sockeye salmon, 1980–2018. Posterior median of U_{MSY} is plotted as a dashed horizontal reference line in Figures A and B.

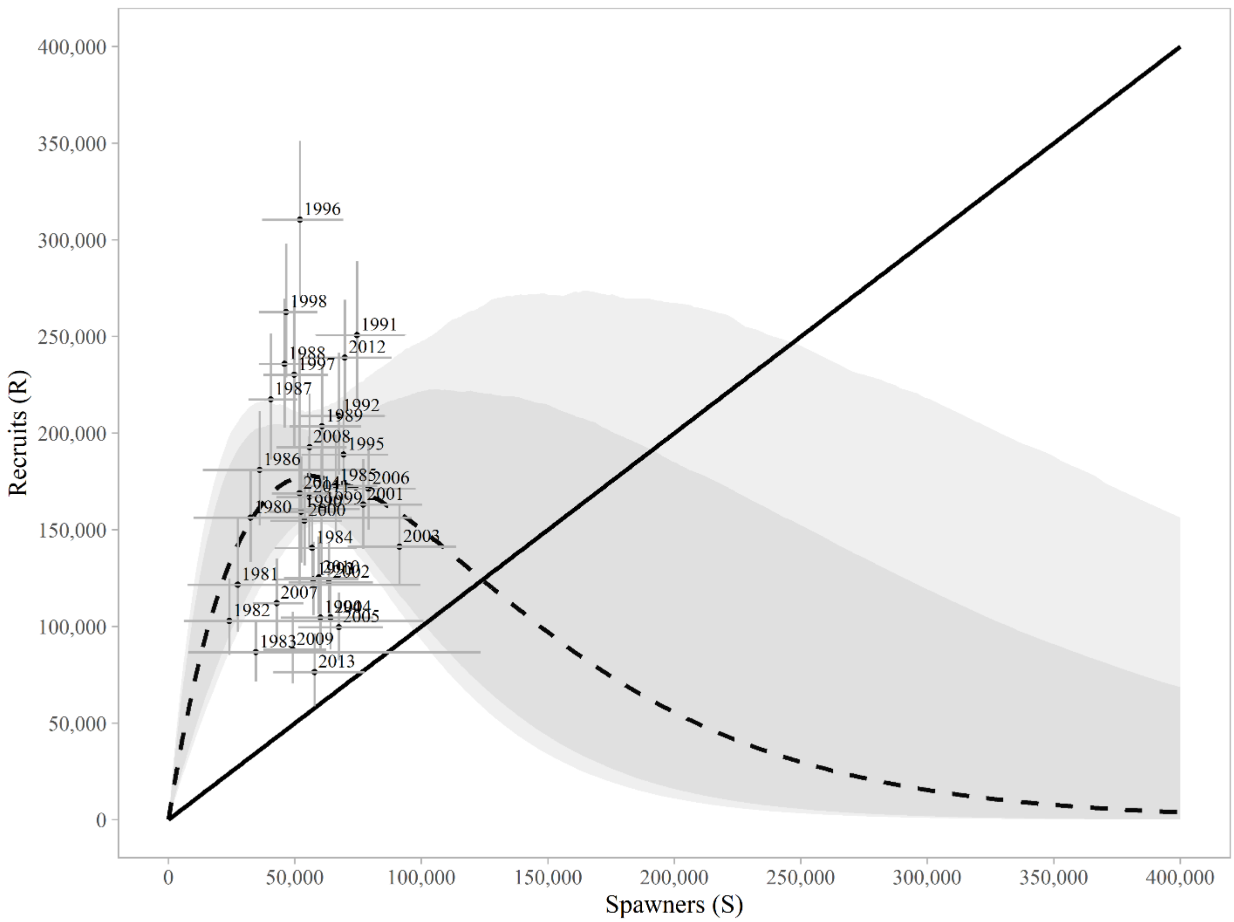

Figure 6: Plausible spawner-recruit relationships (shaded regions around the dashed line) for Taku River Sockeye salmon as derived from a Bayesian state-space model fitted to abundance, harvest, and age data for calendar years 1980–2018. Posterior medians of recruits and spawners are plotted as brood year labels with 95% credible intervals (grey lines). The heavy dashed line is the Ricker relationship constructed from ln(α') and β posterior medians with 90% and 95% credible intervals (shaded areas). Recruits replace spawners on the solid diagonal line.

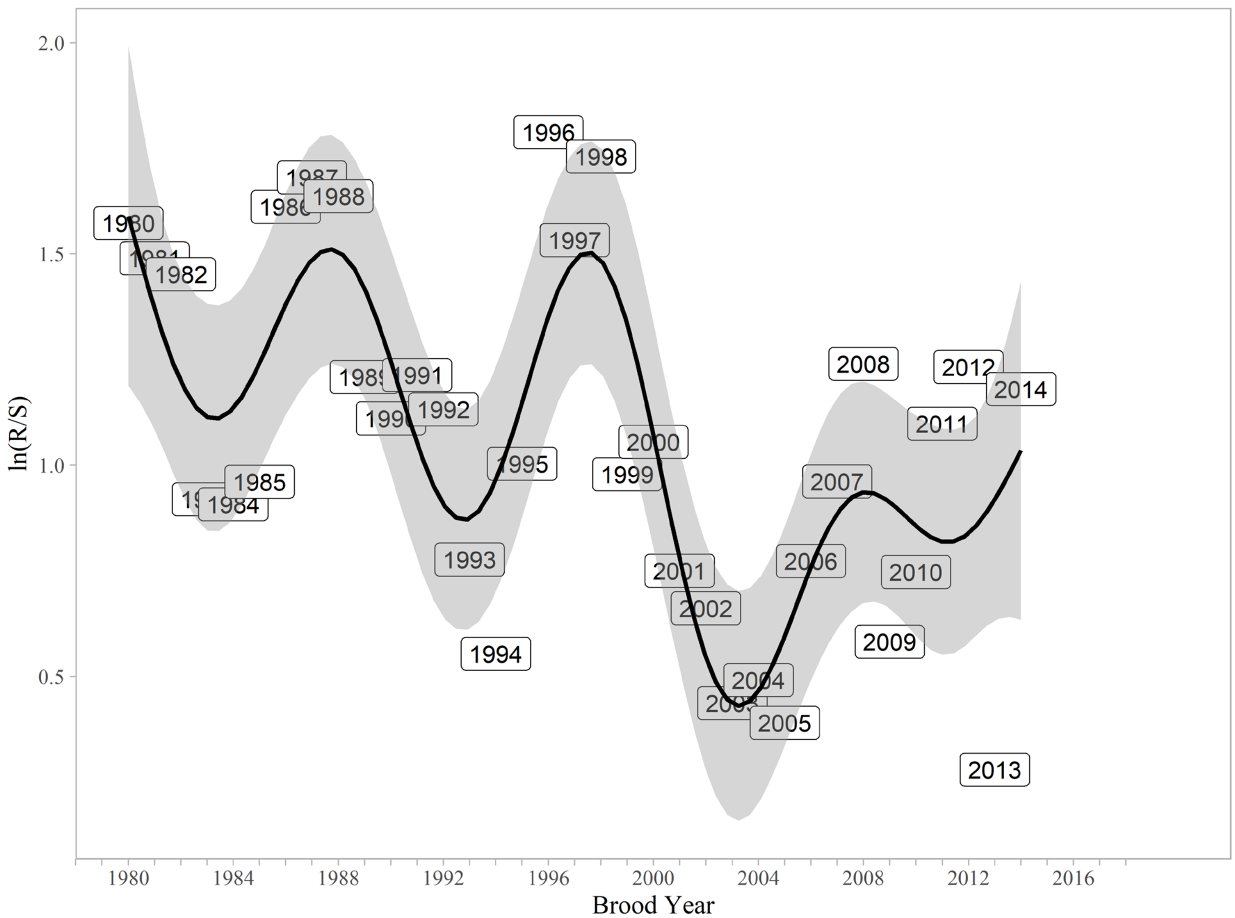

Figure 7: Natural logarithm of recruits per spawner for Taku River Sockeye salmon by brood year (1980–2014).

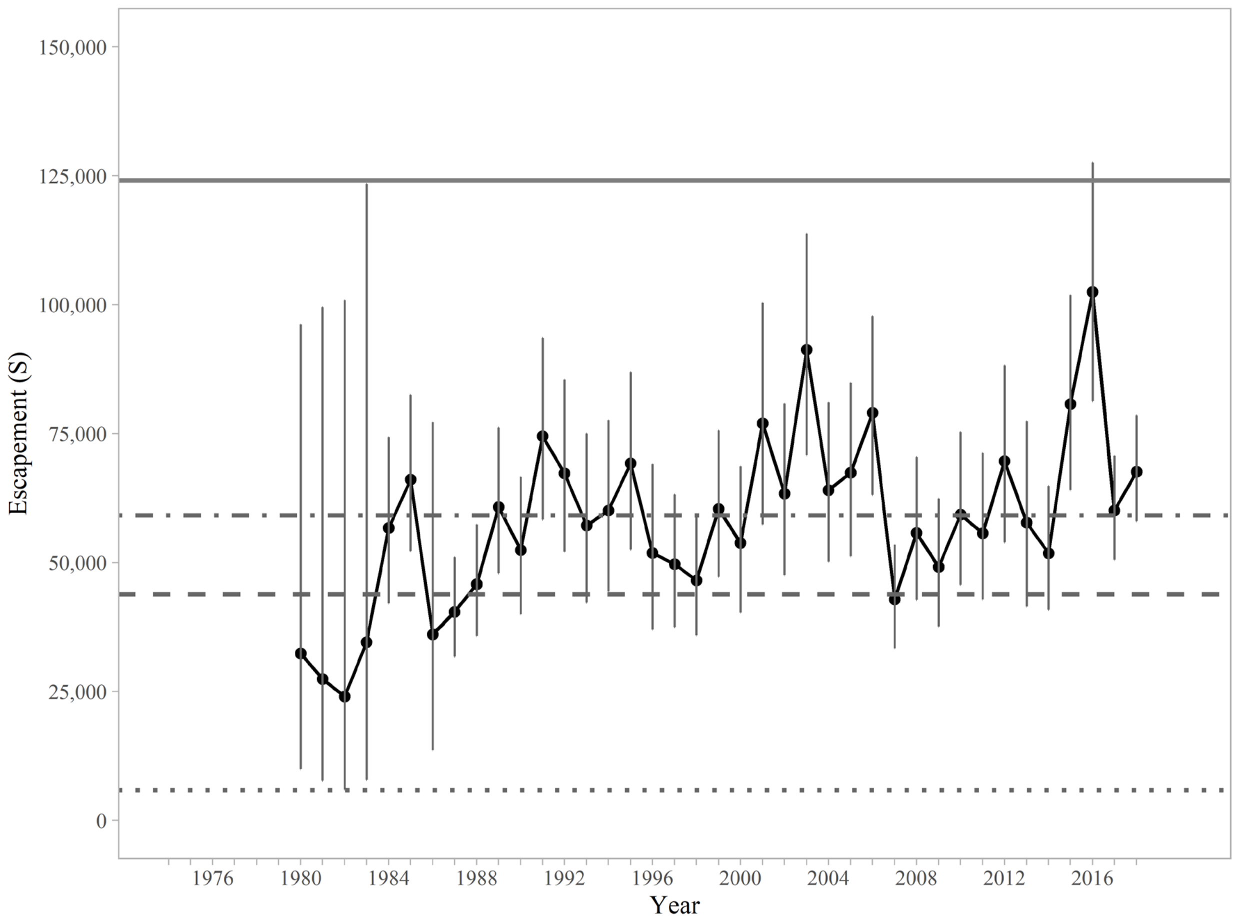

Figure 8: Posterior medians of escapement estimates (spawners (S)) and 95% credible intervals (vertical lines) for Sockeye salmon obtained by fitting a Bayesian state-space model to Taku River Sockeye salmon data, 1980–2018. Posterior medians of S_{MAX} (horizontal dashed line with dots), S_{MSY} (horizontal dashed line), S_{GEN} (horizontal dotted line), and S_{EQ} (horizontal solid line) are plotted as reference lines. Years with higher uncertainty corresponded to years with missing capture-recapture abundance estimates (1980–1983, 1986).

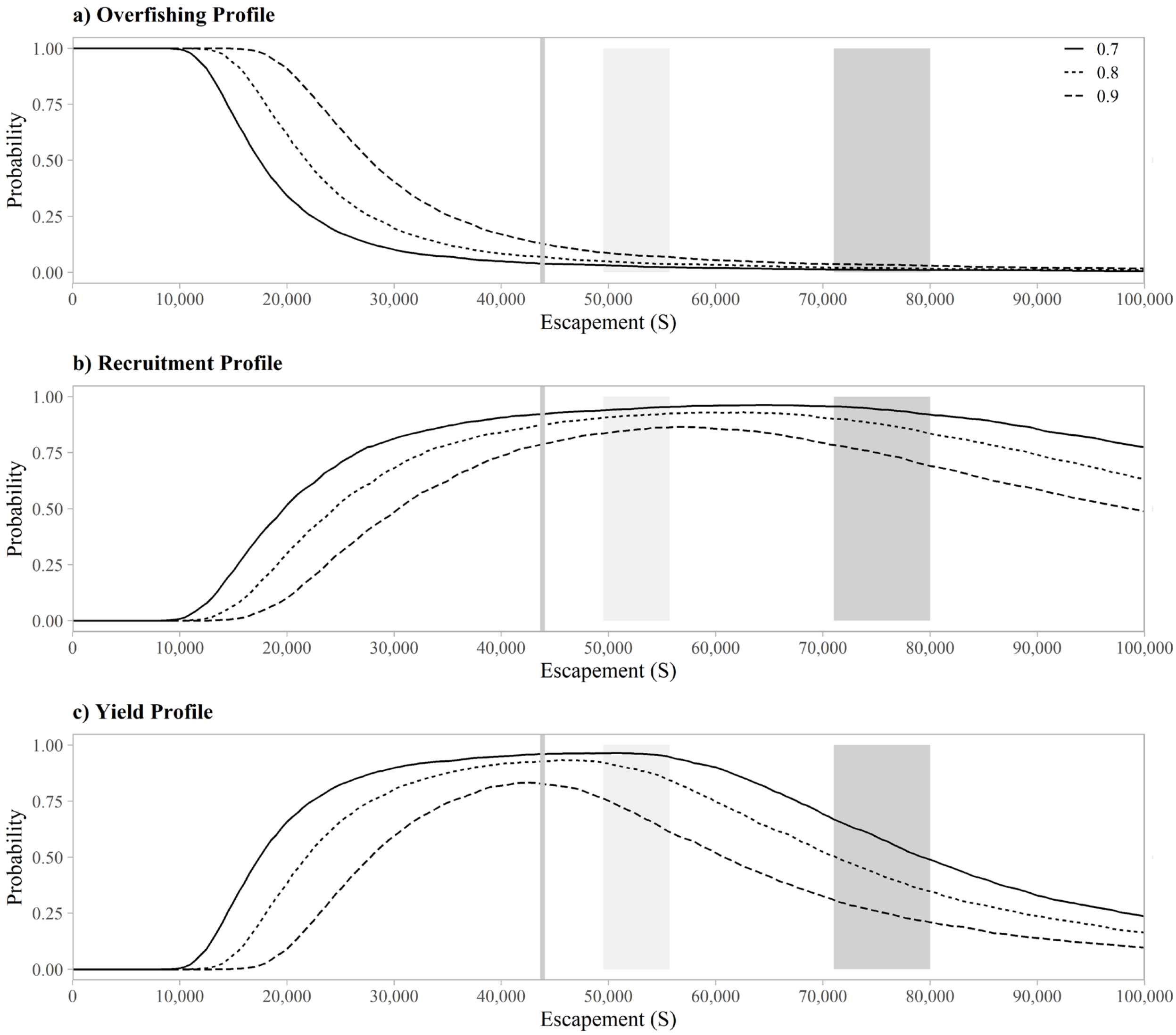

Figure 9: Overfishing profiles (OFPs), maximum recruitment probability profiles (MRPs), and optimal yield profiles (OYPs) for Taku River Sockeye salmon. The OYPs and MRPs show the probability that an escapement will result in specified fractions (0.70, 0.80, and 0.90 line) of maximum sustained yield or maximum recruitment. The OFPs show the probability that reducing escapement to a specified level will result in less than specified fractions of maximum sustained yield. The shaded darker area brackets the system-wide 1985 historical escapement goal and the lighter shaded area is the equivalent system-wide 1985 historical escapement goal after adjusting for dropout rate (25.5% reduction) and size selectivity (6.4% reduction) (49,500 to 55,700 with a point estimate of 52,200). The solid vertical line is the posterior median of spawning abundance at maximum sustained yield obtained from the state-space model (S_{MSY} = 43,857). For the 2019 fishing season, a revised "interim" objective of 55,000 to 62,000 fish and a management target of 59,000 fish was established by the Transboundary Panel of the Pacific Salmon Commission (TTC 2019a). The "interim" objective incorporates a 22% reduction to account for historical tag dropout rates.

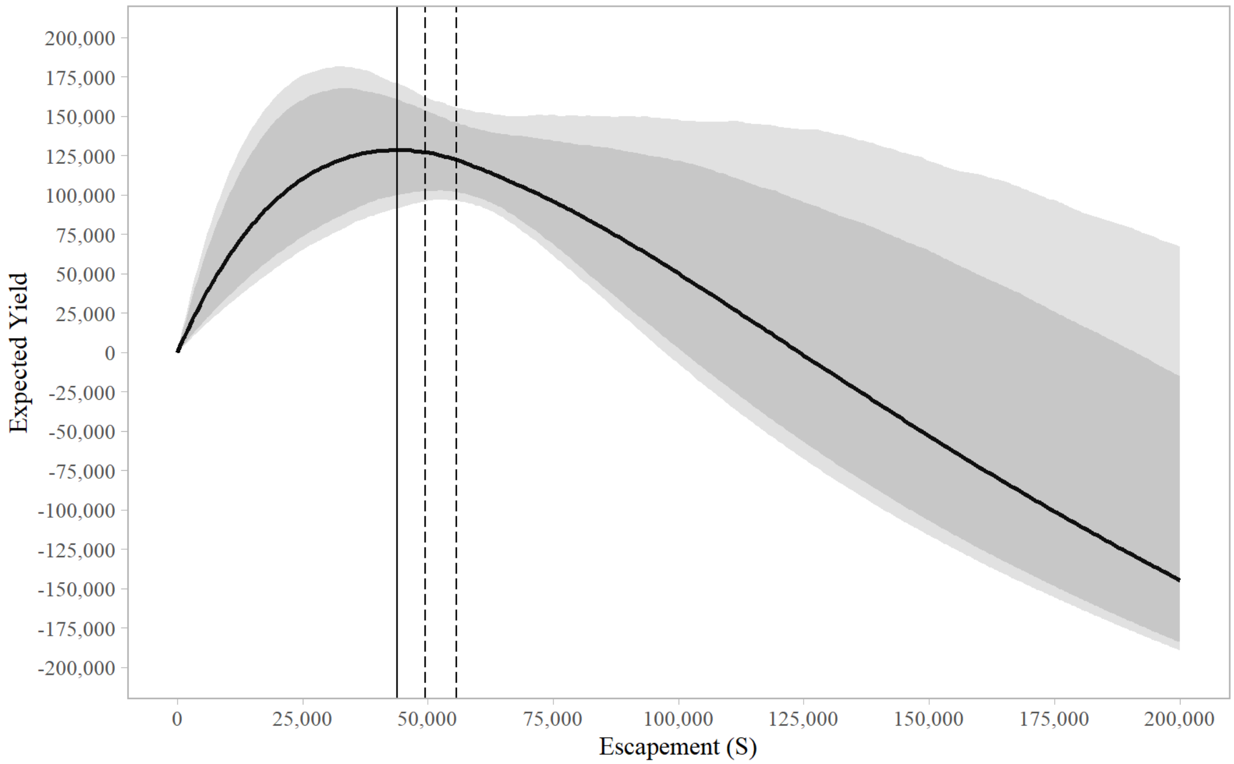

Figure 10: Expected sustained yield (solid black line) and 90% and 95% credible intervals (shaded areas) versus spawning escapement for Taku River Sockeye salmon. The dotted vertical lines bracket the system-wide 1985 escapement goal adjusted for dropout rate (25.5% reduction) and size selectivity (6.4% reduction) (49,500 to 55,700 fish with a point estimate of 52,200 fish). The solid vertical line is the posterior median of spawning abundance at maximum sustained yield obtained from the state-space model (S_{MSY} = 43,857 fish).

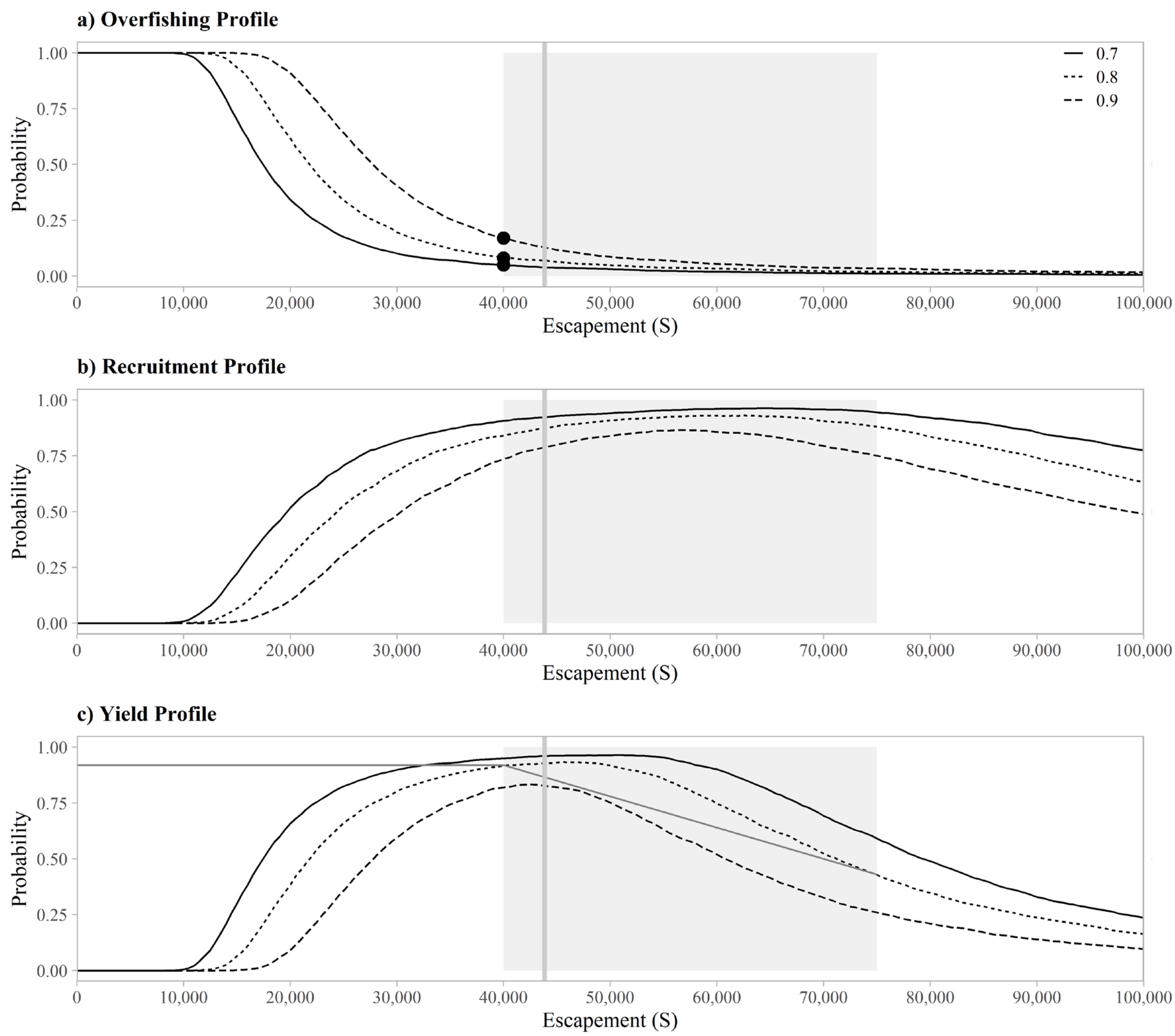

Figure 11: Example of a hypothetical escapement goal range (40,000–75,000 fish; grey box) where the probability of average optimal yields exceeding 0.80 MSY is 0.92 (lower bound; horizontal line in the yield profile) and 0.43 (43%) (upper bound; diagonal line in the yield profile). Overfishing profiles show the probability that sustained yield is reduced to less than a fraction (0.70, 0.80, 0.90) of MSY given a fixed level of escapement. This probability refers to the lower bound of the hypothetical escapement goal range (40,000 fish or lower bound of grey region). The overfishing probability of the stock is 0.05, 0.08, and 0.17 based on 0.70, 0.80, and 0.90 MSY, respectively (solid black points in the overfishing profile). The light grey vertical line in all three figures is the S_{MSY} value of 43,857 spawners. The historical escapement goal range conversion is the escapement goal range that does not include a dropout rate adjustment (25.5% reduction) or size selectivity adjustment (6.4% reduction) (e.g., the historical escapement goal range of 57,400–107,700 fish reduced by 25.5% and then reduced by 6.4% would equate to ~40,000–75,000 fish in the current framework).

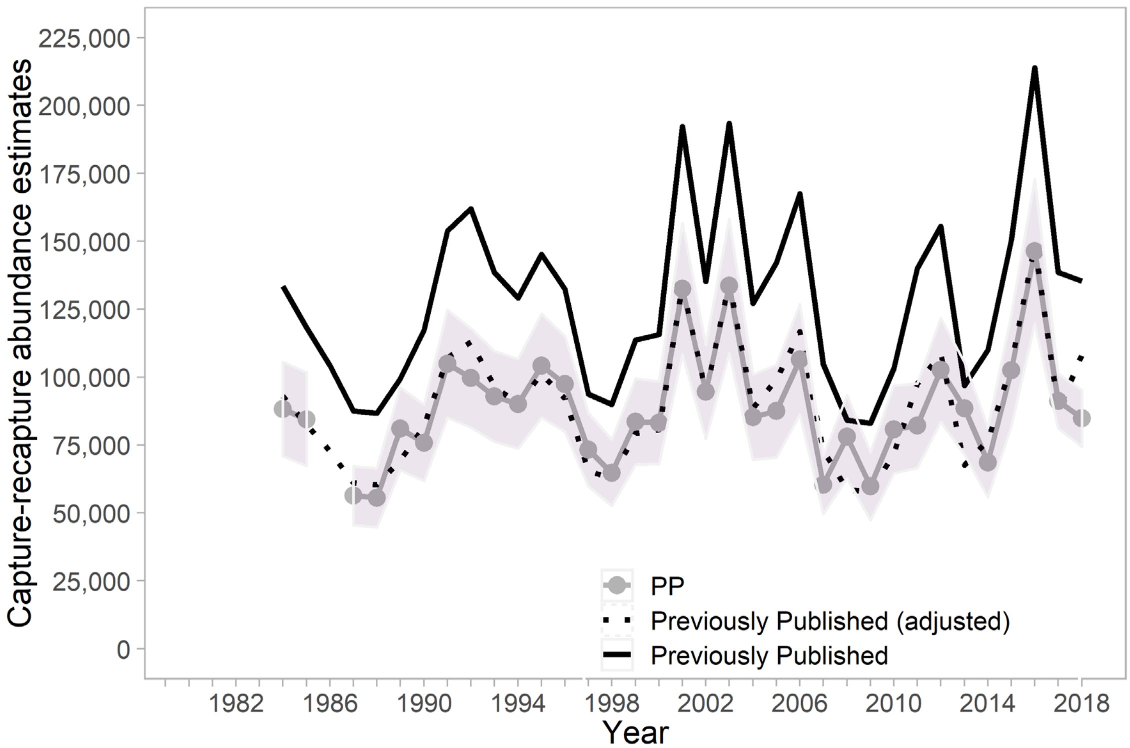

Figure 12: Comparison of the updated and non-expanded previous published (historic) capture-recapture abundance estimates ('PP') for 1984–2018 (excluding 1986). Updated capture-recapture abundance estimates from 1984 through 2002 (excluding 1986) were based on pooled Petersen estimates adjusted downward by 6.4% which represented the average bias observed between size-stratified Petersen and non-size-stratified Petersen estimates from 2003 to 2018. Updated capture-recapture abundance estimates from 2003 onward were based on year-specific size-stratified Petersen estimates. Release and recapture data were unavailable for 1986, therefore an updated capture-recapture abundance estimate was not available. For years 2016 and prior, a weighted-average dropout rate of 25.5% was applied to the updated Petersen capture-recapture abundance estimates, which decreased estimates. Year-specific weighted dropout rates were applied to updated size-stratified Petersen capture-recapture abundance estimates in 2017 and 2018 (32.1% in 2017 and 14.6% in 2018). The grey confidence interval band around the ('PP') estimates are +/- 2 standard deviations. The non-expanded previously published estimates ('Previously Published') are from the TTC (2019b) report. The 'Previously Published (adjusted)' are the non-expanded published estimates adjusted by the year-specific dropout rate and by the size selectivity bias (6.4% downward).

APPENDIX A: RJAGS CODE, DATA OBJECTS, AND MULTINOMIAL AGE COUNTS

APPENDIX A1:

The rjags model code for the Bayesian Markov Chain Monte Carlo statistical analysis of the Taku River Sockeye salmon data run reconstruction model, 1980–2018. The spawner abundance that allows rebuilding to S_{MSY} in one generation in the absence of fishing, S_{GEN}, was calculated using a solver function from the Wild Salmon Policy Metrics Package (Holt and Pestal 2019), which implements the calculations developed by Holt and Ogden (2013). The biological reference points S_{MSY}, U_{MSY}, and 80%S_{MSY} based on Scheuerell (2016) were calculated based on posterior samples of $ln(\alpha')$ and beta and the package gsl (Hankin 2006). The code is not presented. This code was adapted from Fleishman et al. 2013.

```
mod=function(){
# spawner-recruit function with an autoregressive lognormal process error with a lag of one year
   for (y in (A + a.min):(Y + A - 1)) {
      log.R[y] ~ dnorm(log.R.mean2[y], tau.R)
      R[y] <- exp(log.R[y])
      log.R.mean1[y] <- log(S[y - a.max]) + lnalpha - beta * S[y - a.max]
      log.resid[y] <- log(R[y]) - log.R.mean1[y]
   }
   log.R.mean2[A + a.min] <- log.R.mean1[A + a.min] + phi * log.resid.0
   for (y in (A + a.min + 1):(Y + A - 1)) {
      log.R.mean2[y] <- log.R.mean1[y] + phi * log.resid[y - 1]
   }

# prior distribution for model parameters
   lnalpha ~ dnorm(0.00000E+00, 1.00000E-06)  T(1.00000E-06, )
   beta ~ dnorm(0.00000E+00, 1.00000E-06)  T(1.00000E-06, )
   phi ~ dnorm(0.00000E+00, 1.00000E-06)  T(-1, 1)
   mean.log.R0 ~ dnorm(0.00000E+00, 1.00000E-06)
   tau.R0 ~ dgamma(0.001, 0.001)
   log.resid.0 ~ dnorm(0.00000E+00, tau.red)
   tau.R ~ dgamma(0.001, 0.001)
   sigma.R <- 1/sqrt(tau.R)
   alpha <- exp(lnalpha)
   sigma.R0 <- 1/sqrt(tau.R0)
   tau.red <- tau.R * (1 - phi * phi)
   lnalpha.c <- lnalpha + (sigma.R * sigma.R/2/(1 - phi * phi))

# the first several cohorts originate from unmonitored spawning; these are drawn from a
     common lognormal distribution
   R.0 <- exp(mean.log.R0)
   for (y in 1:a.max) {
      log.R[y] ~ dnorm(mean.log.R0, tau.R0)
      R[y] <- exp(log.R[y])
   }

 # biological reference points
   S.max <- 1/beta
```

```
   alpha.c <- min(exp(lnalpha.c), 10000)
   U.msy.c <- lnalpha.c * (0.5-0.07*lnalpha.c)
   S.eq.c <- lnalpha.c * S.max  # equilibrium spawning abundance
   S.msy.c <- S.eq.c * (0.5 - 0.07 * lnalpha.c) # Hilborn (1985) approximation of spawner
abundance that maximizes sustained yield

   positive.lna.c <- step(lnalpha.c)
   lnalpha.c.nonneg <- lnalpha.c * positive.lna.c
   S.eq.c2 <- lnalpha.c.nonneg * S.max
   peterman.approx.c <- (0.5 - 0.65 * pow(lnalpha.c.nonneg, 1.27)/(8.7 + pow(lnalpha.c.nonneg,
1.27)))

   U.msy.c2 <- lnalpha.c.nonneg * peterman.approx.c # Peterman et al. (2000) approximation of
harvest rate leading to MSY

   S.msy.c2 <- U.msy.c2/beta # Peterman et al. (2000) approximation of spawner abundance
that maximizes sustained yield
   U.max.c2 <- 1 - 1 / exp(lnalpha.c.nonneg)
   S.msy.c.80 <- S.msy.c *0.80
   S.msy.c2.80 <- S.msy.c2 *0.80

  # maturity schedule based on the Dirichlet distribution
   D.scale ~ dunif(0, 1)
   D.sum <- 1/(D.scale * D.scale)
   pi.2p ~ dbeta(1, 1)
   pi.1 ~ dbeta(1, 1)
   pi[1] <- pi.1
   pi[2] <- pi.2p * (1 - pi[1])
   pi[3] <- 1 - pi[1] - pi[2]
   for (a in 1:A) {
      gamma[a] <- D.sum * pi[a]
      for (y in 1:(Y + A - 1)) {
         g[y, a] ~ dgamma(gamma[a], 0.01)
         p[y, a] <- g[y, a]/sum(g[y, ])
      }
   }
# terminal abundance
   for (a in 1:A) {
      for (y in a:(Y + (a - 1))) {
         N.ya[y - (a - 1), (A + 1 - a)] <- p[y, (A + 1 - a)] * R[y]
      }
   }
   # multinomial age counts
   for (y in 1:Y) {
      N[y] <- sum(N.ya[y, 1:A])
      for (a in 1:A) {
         q[y, a] <- N.ya[y, a]/N[y]
      }
   }
   for (t in 1:Y) {
      x[t, 1:A] ~ dmulti(q[t, ], n.a[t])
```

```
  }

 # harvest below and above the border
  for (y in 1:Y) {
    mu.hbelow_ns [y] ~ dbeta(1, 1)
    h.below_ns[y] <- mu.hbelow_ns[y] * N[y]
    log.hb_ns[y] <- log(h.below_ns[y])
    tau.log.hb.ns[y] <- 1/log(cv.hb[y] * cv.hb[y] + 1)
    hbelow_ns[y] ~ dlnorm(log.hb_ns[y], tau.log.hb.ns[y])

    mu.habove_ns[y] ~ dbeta(1, 1)
    h.above_ns[y] <- mu.habove_ns[y] * N[y]
    log.ha_ns[y] <- log(h.above_ns[y])
    tau.log.ha.ns[y] <- 1/log(cv.ha[y] * cv.ha[y] + 1)
    habove_ns[y] ~ dlnorm(log.ha_ns[y], tau.log.ha.NS[y])
    inriver.run[y] <- max(N[y] - h.below_ns[y], 1)

    log.ir[y] <- log(inriver.run[y])
    tau.log.ir[y] <- 1/log(cv.ir[y] * cv.ir[y] + 1)
    ir[y] ~ dlnorm(log.ir[y], tau.log.ir[y])

    mu.habove[y] ~ dbeta(1, 1)
    h.above[y] <- mu.habove[y] * inriver.run[y]
    log.ha[y] <- log(h.above[y])
    tau.log.ha[y] <- 1/log(cv.ha[y] * cv.ha[y] + 1)
    habove[y] ~ dlnorm(log.ha[y], tau.log.ha[y])

    mu.hbelow[y] ~ dbeta(1, 1)
    h.below[y] <- mu.hbelow[y] * N[y]
    log.hb[y] <- log(h.below[y])
    tau.log.hb[y] <- 1/log(cv.hb[y] * cv.hb[y] + 1)
    hbelow[y] ~ dlnorm(log.hb[y], tau.log.hb[y])

    mu[y] <- (h.below[y] + h.above[y])/N[y]
    S[y] <- max(inriver.run[y] - h.above[y], 1) # spawning abundance or escapement
    log.S[y] <- log(S[y])
  }
}
```

APPENDIX A2:

Table A2:The rjags data objects for the Bayesian Markov Chain Monte Carlo statistical analysis of the Taku River Sockeye salmon data run reconstruction model, 1980–2018. The terminal run age count x4 represents ages 2–4, the terminal run age count x5 represents age 5, and the terminal run age count x6 represents ages 6–8. In the table, 'hbelow_ns' is naturally-spawned harvest below the border (excluding naturally-spawned personal use harvest) with the associated coefficients of variation (cv.hb), and 'habove_ns' is naturally-spawned harvest above the U.S./Canada border (including the naturally-spawned broodstock) with the associated coefficients of variation (cv.ha). Annual harvest below the border 'hbelow' is naturally-spawned and enhanced harvest in the by U.S. District 111 traditional commercial drift gillnet and the Amalga Harbor special harvest area purse seine fishery, and the U.S. personal use harvest in the Taku River. Annual harvest above the border 'habove' is inriver commercial gillnet harvest, inriver test/assessment fishery harvest, and the Aboriginal harvest.

year	hbelow_ns	habove_ns	habove	hbelow	cv.hb	cv.ha	x4	x5	x6
1980	NA	22,752	22,752	NA	0.90	0.05	0	0	0
1981	NA	10,922	10,922	NA	0.90	0.05	0	0	0
1982	NA	3,144	3,144	NA	0.90	0.05	0	0	0
1983	23,460	17,056	17,056	23,460	0.05	0.05	45	48	6
1984	57,619	27,292	27,292	57,619	0.05	0.05	31	66	3
1985	73,367	14,411	14,411	74,287	0.05	0.05	32	63	5
1986	60,644	14,939	14,939	60,644	0.05	0.05	32	59	10
1987	54,963	13,887	13,887	54,963	0.05	0.05	38	56	5
1988	25,785	12,967	12,967	25,785	0.05	0.05	50	46	4
1989	62,804	18,805	18,805	63,366	0.05	0.05	29	65	5
1990	108,492	23,140	21,474	109,285	0.05	0.05	29	64	7
1991	104,471	27,321	25,380	105,271	0.05	0.05	39	54	7
1992	119,959	31,502	29,862	121,176	0.05	0.05	36	57	7
1993	140,888	34,270	33,523	142,089	0.05	0.05	29	66	6
1994	96,952	29,748	29,001	98,063	0.05	0.05	27	69	4
1995	86,929	32,467	32,711	91,984	0.05	0.05	44	52	4
1996	181,776	43,541	42,025	187,727	0.05	0.05	44	53	3
1997	76,043	26,002	24,352	79,127	0.05	0.05	32	58	11
1998	47,824	19,914	19,277	49,832	0.05	0.05	26	66	7
1999	61,205	21,059	21,151	63,058	0.05	0.05	53	44	3
2000	128,567	29,765	28,468	131,262	0.05	0.05	44	51	5
2001	194,091	48,729	48,117	204,433	0.05	0.05	32	66	2
2002	114,460	32,658	31,726	116,400	0.05	0.05	39	58	3
2003	134,957	33,845	33,024	136,942	0.05	0.05	30	68	2
2004	75,186	20,468	20,359	77,012	0.05	0.05	50	48	3
2005	44,360	22,583	22,102	46,089	0.05	0.05	41	56	2
2006	62,814	23,696	21,446	65,828	0.05	0.05	34	63	3
2007	60,879	18,285	17,249	65,129	0.05	0.05	44	50	5
2008	63,002	19,982	19,509	75,692	0.05	0.05	37	60	3
2009	35,121	11,873	11,260	36,232	0.05	0.05	40	54	6
2010	44,837	21,138	20,661	46,767	0.05	0.05	57	42	1
2011	65,090	23,697	24,543	71,805	0.05	0.05	25	69	6
2012	45,410	27,917	30,113	50,736	0.05	0.05	57	40	3

year	hbelow_ns	habove_ns	habove	hbelow	cv.hb	cv.ha	x4	x5	x6
2013	85,621	21,894	25,173	100,144	0.05	0.05	30	63	8
2014	31,208	17,914	17,795	33,226	0.05	0.05	66	31	3
2015	40,904	20,063	19,849	42,054	0.05	0.05	59	39	2
2016	66,980	34,616	37,434	74,874	0.05	0.05	63	36	2
2017	67,706	28,798	30,379	74,604	0.05	0.05	25	71	4
2018	24,472	18,246	17,962	27,402	0.05	0.05	70	25	5

APPENDIX A3:

Table A3: The rjags data objects for the Bayesian Markov Chain Monte Carlo statistical analysis of the Taku River Sockeye salmon data run reconstruction model, 1980–2018. In the table, 'ir' are the pooled Petersen capture-recapture estimates of above-border abundance with the associated coefficients of variation (ir.cv). There was no estimate for 1986. These pooled Petersen capture-recapture estimates are from the Pestal et al. 2020 and differ from the published estimates in the TTC (2019b) report.

year	ir	cv.ir
1984	88,273	0.10
1985	84,479	0.10
1986	---	0.90
1987	56,362	0.10
1988	55,580	0.10
1989	80,998	0.09
1990	75,801	0.09
1991	104,896	0.09
1992	99,643	0.09
1993	92,933	0.09
1994	90,129	0.09
1995	104,242	0.09
1996	97,478	0.09
1997	73,255	0.09
1998	64,756	0.09
1999	83,588	0.09
2000	83,190	0.09
2001	132,503	0.09
2002	94,606	0.09
2003	133,594	0.09
2004	85,258	0.09
2005	87,496	0.10
2006	106,545	0.10
2007	60,321	0.09
2008	78,031	0.10
2009	59,818	0.10
2010	80,747	0.10
2011	82,117	0.09
2012	102,671	0.09
2013	88,536	0.10
2014	68,533	0.09
2015	102,506	0.10
2016	146,294	0.09
2017	91,164	0.06
2018	84,807	0.06

APPENDIX A4: DROPOUT RATE ADJUSTMENT

Estimates for dropout rate, adjusted Petersen estimate, and associated standard errors are computed as follows:

$$
\begin{aligned}
dr &= 1-(x/n)\\
se_{dr} &= \sqrt{dr \times (1-dr)/n}\\
N_{adj} &= N \times (1-dr)\\
se_{N_{adj}} &= \sqrt{se_N{}^2 \times se_{dr}{}^2 + se_N{}^2 \times (1-dr)^2 + N^2 \times se_{dr}{}^2)}\\
U_{adj} &= U \times (1-dr)\\
se_{U_{adj}} &= \sqrt{se_U{}^2 \times se_{dr}{}^2 + se_U{}^2 \times (1-dr)^2 + U^2 \times se_{dr}{}^2)}
\end{aligned}
$$

where

n = number fish tracked for dropout
x = number that did NOT fall back
N = estimated population size without adjustment for dropout of tagged fish
U = estimated untagged population size without adjustment for dropout of tagged fish
N_{adj} = estimated population size with adjustment for dropout of tagged fish
U_{adj} = estimated untagged population size with adjustment for dropout of tagged fish

The observed dropout proportion varies among years; however, there is no year-specific dropout estimates for most of the capture-recapture estimates. Therefore, an imputed dropout proportion for years without radiotelemetry studies must account for the uncertainty in the dropout proportion caused by a small number of fish tagged with radio tags in a particular study and the year-to-year variation in the dropout probability. The long-term average dropout adjustment was modelled using synthetic values of *n* and *x* that incorporated the weighted average of the results from 1984, 2015, 2017, and the 2018 side project radiotelemetry studies. The side project data for 2018 was used because fish wheel operation was simulated to previous years operations (Bednarski *et al.* 2019). The synthetic values address the effect of the implied radio tag sample size on the variance of the adjusted capture-recapture estimate. A "synthetic" set of telemetry data was created to represent both sources of uncertainty as follows (Figure A4):

1. Fit a generalized linear mixed model to the four years of telemetry data with a common mean and a random effect for years. The overall estimated mean dropout probability was approximately 0.25 (SE 0.028). The estimated year-to-year standard deviation in the dropout probability was 0.45.
2. The total uncertainty that accounts for both year-to-year variation and uncertainty in estimating the mean dropout probability is found as $\sqrt{.045^2+.028^2}=0.054$.
3. Synthetic values of *n* and *x* were created such that a telemetry study with these synthetic values matched the mean dropout probability (i.e., *x*/*n* = 0.25) and matched the combined uncertainty (i.e., $\sqrt{\frac{\frac{x}{n}(1-\frac{x}{n})}{n}}=0.054$), but rounded to integer values. This gives *n* = 51 and *x* = 13.
4. The scaled-down, weighted average dropout rate of 13/51 = 25.5% was applied to capture-recapture abundance estimates in 1984–2016.

Year-specific weighted dropout rates were applied to 2017 and 2018 (32.1% in 2017 and 14.6% in 2018) based on radiotelemetry studies conducted in those years.

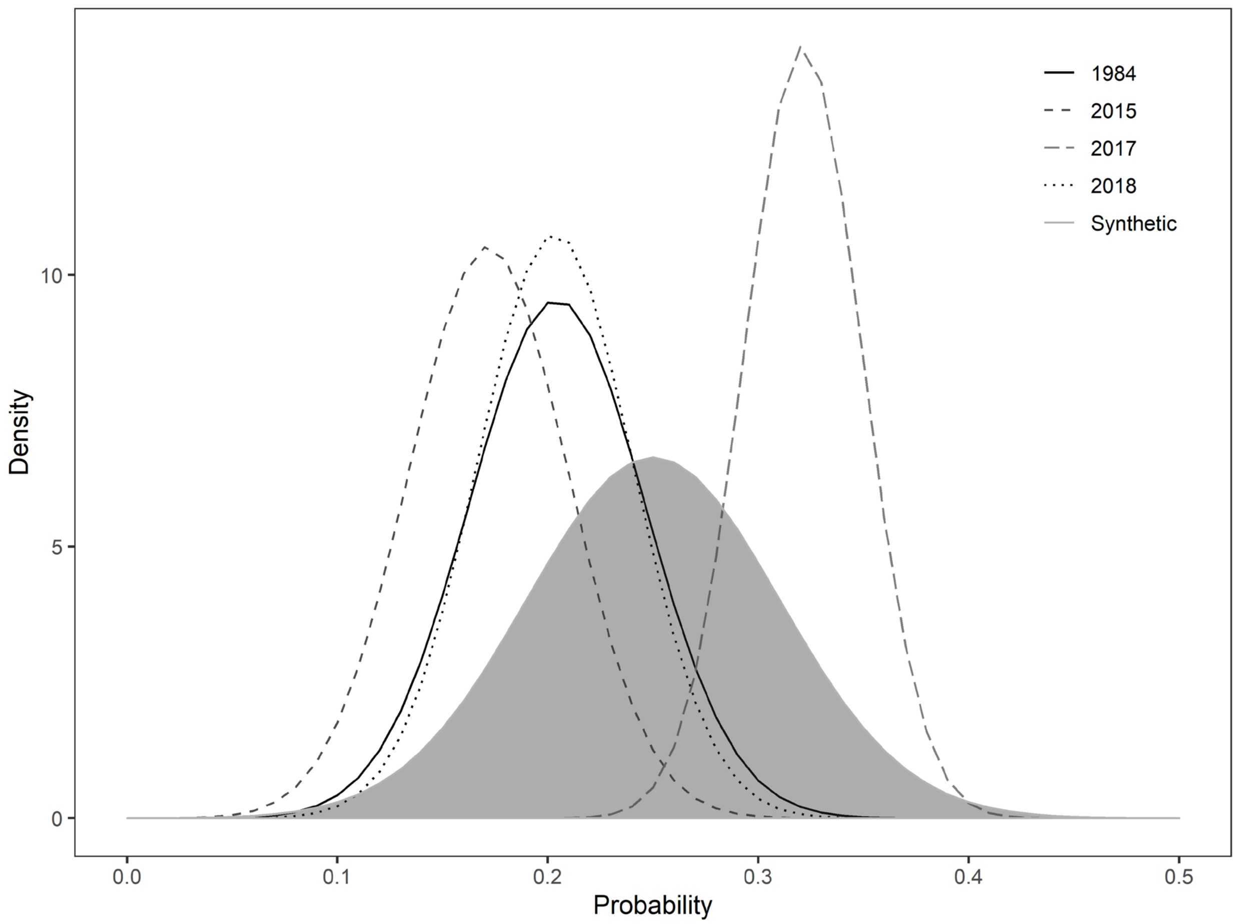

Figure A4: The four years of radiotelemetry data (1984, 2015, 2017, 2018) are shown as solid and dotted density lines. The synthetic dropout rate (13/51), based on the four studies, is shown as the filled, grey density profile.

APPENDIX B: POSTERIOR DISTRIBUTION OF MODEL PARAMETERS

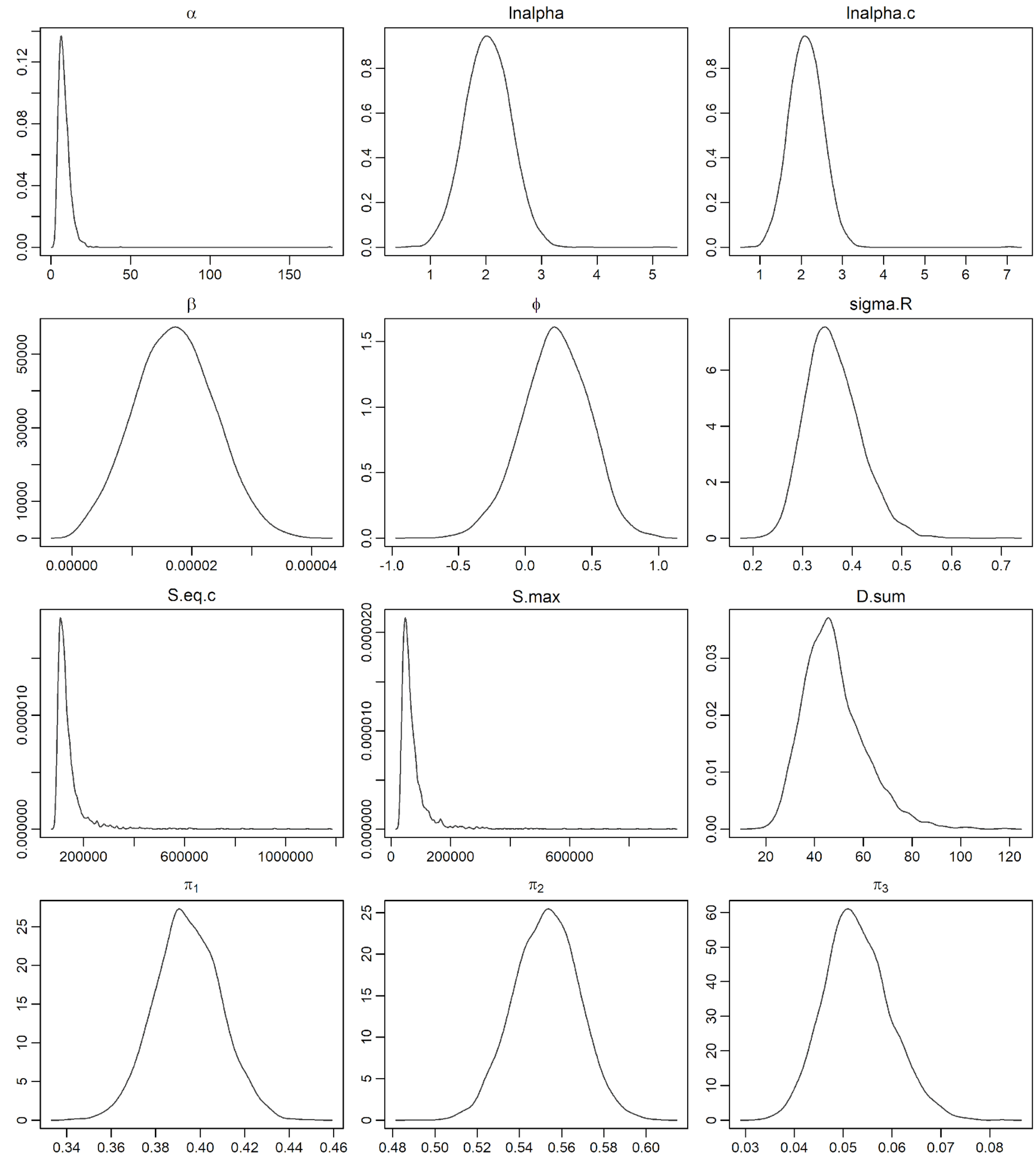

Figure B1: Density plots of key model parameters (alpha (α), lnalpha ln(α), lnalpha.c ln(α′), beta β, ϕ, sigma.R σ_R, S.eq.c S_{EQ}, S.max S_{MAX}, D.sum , pi π).

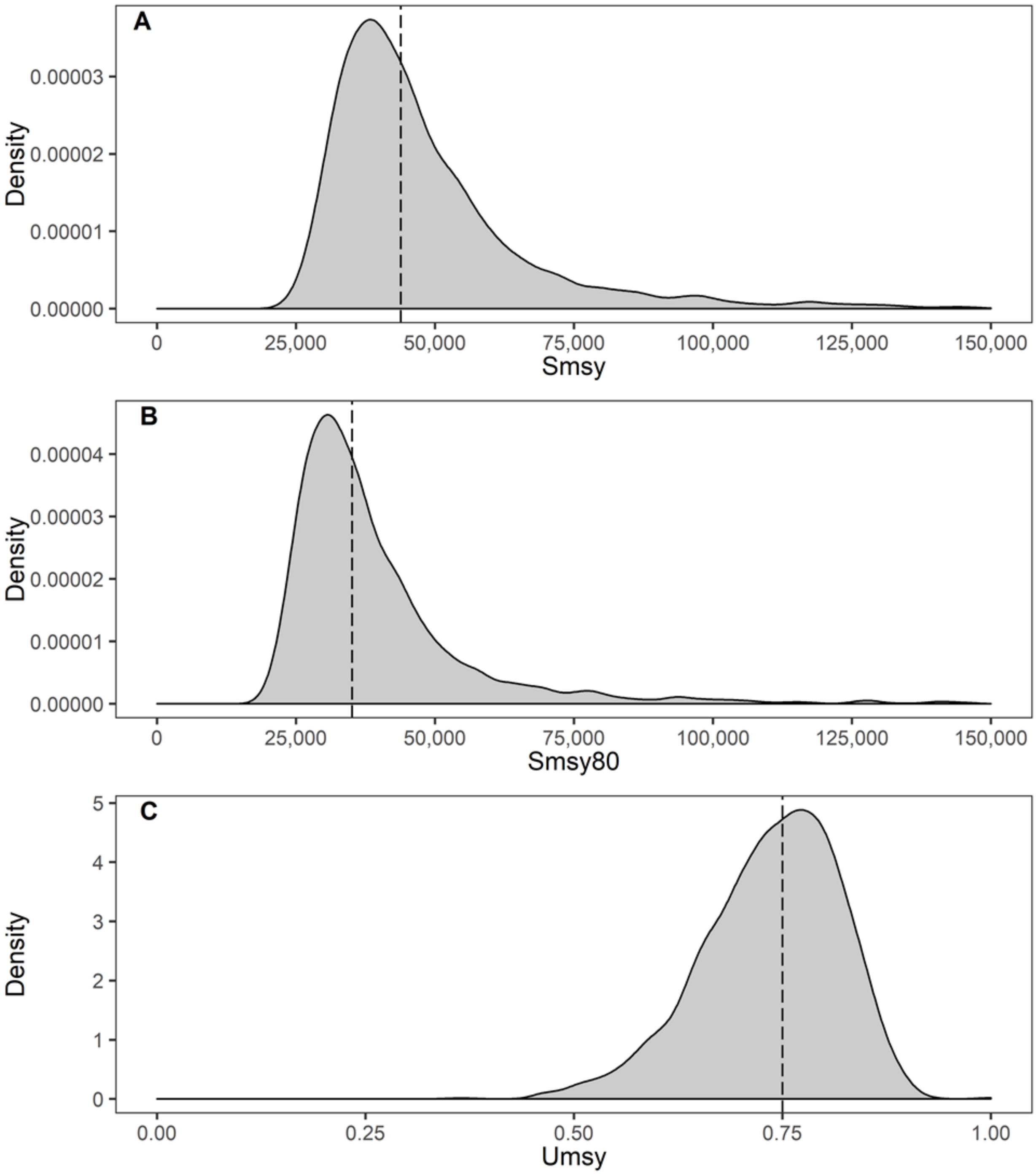

Figure B2: Density plots of biological reference points (S_{MSY}, 80%S_{MSY}, U_{MSY}).