Three Decades of ESA-Listing: Are Snake River Spring/summer Chinook Salmon and Steelhead on the Path to Recovery?

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1 Background

Freshwater and marine fish species are declining globally due to a variety of stressors, including habitat loss, overfishing, climate change, and pollution. Freshwater species have experienced an alarming 83% decline in monitored populations since 1970, with significant losses observed in the Pacific region (WWF, 2022). A recent study conducted for the International Union for the Conservation of Nature (IUCN)'s Red List of Threatened Species found 25% of all freshwater species are at high risk of extinction (Sayer et al., 2025). Similarly, marine species such as Atlantic Cod *Gadus morhua*)* and Bluefin Tuna *Thunnus thynnus* have suffered steep population declines from over exploitation and environmental changes (FAO, 2020). The downward patterns and root causes are shared with most North American diadromous fish as well (Waldman and Quinn, 2022), and mirrored by populations of anadromous salmonids along the western coast of the United States, where fish abundance has been negatively impacted since commercial harvest began with European settlement (Chapman, 1986) and continues to be threatened from altered river flows, warming waters, and habitat fragmentation caused by dams and urban development.

Nowhere are these declines more evident than in the Columbia River basin, a critical habitat for pacific salmonid species, including Chinook Salmon *O. tshawytscha* and steelhead *O. mykiss*. Historically Chinook Salmon and steelhead returned to the Columbia River basin in the millions, estimated returns range between 5 and 16 million fish annually (Chapman, 1986; ISAB, 2015; NPCC, 1986), with spring and summer (sp/su) Chinook Salmon making up the largest component (2.5-3.0 million) and steelhead returns consisting of approximately 450-550 thousand individuals (Chapman, 1986). Chapman and Chandler (2003) later decomposed total Columbia River returns into Snake River specific stocks, a major tributary of the Columbia River, using available habitat area as developed by (**need source: PFMC 1979**). Chapman and Chandler (2003) found the Snake River contained the majority of available Columbia River spawning habitat, suggesting historic returns of sp/su Chinook Salmon to the Snake River were approximately 1.4-2.2 million with an additional 250-400 thousand steelhead.

The abundant Snake River returns started declining soon after European settlement beginning in the 1850's due to overharvest, hydropower development, natural resource extraction and urban growth along the Columbia and Snake rivers (Nehlsen, Williams, and Lichatowich, 1991). With the largest loss of returns being observed during the construction of hydroelectric dams along the Columbia and Lower Snake rivers beginning with Bonneville Dam in 1933 and ending with Lower Granite Dam in 1978. Between 1980 and 1990 sp/su Chinook Salmon escapement into the Snake River basin ranged between 3,343 to 21,870 adults (57 FR 14653)(*Federal Management Regulation; Internet GOV Domain* 1992).

Based on their rapid decline and low numbers relative to their large geographical area of occupancy the Snake River sp/su Chinook Salmon evolutionary significant unit (ESU) was listed as threatened under the Endangered Species Act (ESA) in 1992 (57 FR 14653)(*Federal Management Regulation; Internet GOV Domain* 1992). During the same period, the returning Snake River summer steelhead distinct population segment (DPS) experienced similar rates of decline, with an average annual return of only 9,400 when they were listed as threatened under ESA in 1997 (62 FR 43937) (*Federal Management Regulation; Internet GOV Domain* 1997).

The Snake River sp/su Chinook Salmon ESU and steelhead DPS historically supported 68 sp/su Chinook Salmon and 40 steelhead populations located across central Idaho, southeast Washington, and northeast Oregon [CBP (2020); @**??**fig:esu_map)]. The historical populations contained **X** river kilometers of spawning habitat that represented more than 88% of the total spawning habitat in the Columbia River Basin (Chapman and Chandler, 2003). However, habitat accessibility was permanently reduced to 41 sp/su Chinook Salmon and 24 steelhead populations after the completion of Dworshak Dam on the North Fork Clearwater River in 1969 and the three Hell's Canyon project dams on the upper Snake River River from 1959-1967 which lacked fish passage facilities. The resulting habitat loss led to the immediate extinction of 60% of the historical populations in the Snake River basin. With the remaining populations at historically low abundances and facing continued threats of habitat degradation, hydrosystem impacts, and a rapidly changing climate (NOAA, 2022; Crozier et al., 2021; Nakamura, 2023).

After ESA listing of Snake River anadromous fish in the 1990's regulatory agencies established methods to asses population viability. Population viability and extinction is commonly thought of as a stochastic event with risk conveyed probabilistically, requiring thresholds for population persistence, and determined through "rules of thumb", analytical, or simulation studies (Thompson, 1991). Often, population viability occurs at a point when abundance or other metrics indicate the probability of persistence is greater than 0.95. Similarly, population viability evaluations can examine the opposite and determine at which point is the probability of extinction greater than 0.95, thus, indicating the point at which the population is at high risk of extinction and unlikely to recover (Thompson, 1991). The viability of Snake River sp/su Chinook Salmon and steelhead is assessed using a combination of minimum abundance thresholds (MAT), productivity, genetic diversity and spatial distribution metrics (McElhany et al., 2000; ICTRT, 2007). Minimum abundance thresholds for each population were developed independently using available habitat and intrinsic potential models (Cooney and Holzer, 2006), with MAT for individual populations ranging from 500-2,000 sp/su Chinook Salmon and 500-1,500 steelhead (ICTRT, 2007). At these established MAT levels populations were determined to be viable and have a low risk of extinction (NOAA, 2017). Conversely, (ICTRT, 2007) set a quasi-extinction threshold (QET) for each population at 50 or fewer spawners for four consecutive years to determine the point when extinction was likely. And recently, in 2020, Columbia River and Snake River fisheries co-managers recognized population viability thresholds were far below desired levels in order to meet ecological and state and tribal harvest needs. In collaboration with the regulatory agencies the co-managers formed the Columbia Basin Partnership (CBP) and developed common abundance goals focused on healthy, harvestable, and sustainable levels capable of supporting ecosystem processes (CBP, 2020). These established "healthy and harvestable" levels, combined with population viability thresholds, form an agreed upon and complete set of abundance evaluation points to determine a population's recovery status.

Three decades have passed since the ESA listing of Snake River sp/su Chinook Salmon and steelhead, and during this time, one of the largest and most expensive species recovery efforts has unfolded. Approximately 17 billion dollars has been spent in the Columbia River basin to protect and restore habitat, improve survival through mainstem hydrosystems, provide fisheries with hatchery production, and to rebuild natural populations supplementation (**citation**), yet, annual fish runs continue to be only 1% of historical levels (Thurow, Copeland, and Oldemeyer, 2019; Storch et al., 2022; Ford, 2022). Our aim for this research is to provide an updated current status and future predicted status of Snake River sp/su Chinook Salmon and steelhead populations relative to healthy and harvestable, delisting, and quasi-extinction abundance thresholds (CBP, 2020). To establish the current status and trends of Snake River populations we first fit multivariate dynamic linear models (DLM) to empirical population abundance estimates, similar to Ford (2022), to identify common patterns across populations. Second, we use the best fitting DLM to estimate population trends and annual growth rates for the most recent 10 year period to evaluation recovery status. Third, we summarize the percent of populations currently, or predicted, above or below abundance thresholds, and finally, we discuss our findings in

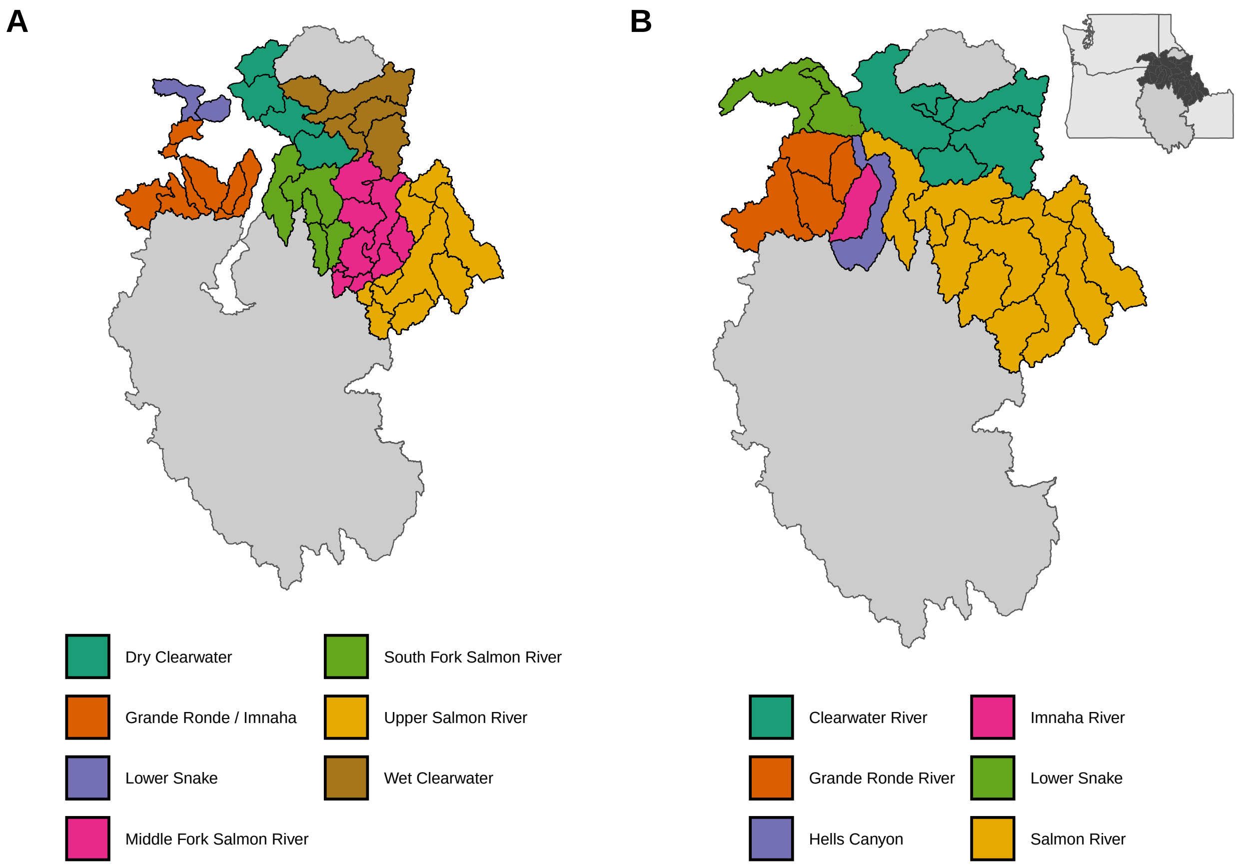

Figure 1: *Currently accessible and extant spring-summer Chinook Salmon (A) and steelhead (B) populations within the Snake River Basin evolutionary significant unit (ESU) or distinct population segment (DPS) are grouped in similar colors while the grey shape indicates extict populations.*

relation to other studies regarding Snake River anadromous fish population status.

2 Methods

2.1 Data

This study seeks to use the best available abundance information to understand the current status of Snake River populations. We begin by modeling spawner abundance using two independent sources of information, when available, for each population with a state-space model accounting for observation and process error. The model seeks to describe the unknown state process for each population while allowing imperfect observations. Next, we fit a linear regression (i.e., trend line) to each state process for the last 10-year period to describe contemporary trends and the last two generations of returns. Last, we extend the trend lines five years into the future to identify populations at highest risk of reaching quasi-extinction.

Time-series abundance estimates of natural-origin fish returning to accessible populations within the Snake River Basin were provided from two independent observation sources [**??**; **??**]. Both time-series estimates were obtained from StreamNet's Coordinated Assessments natural-origin spawner abundance (NOSA) dataset. We first subset the dataset to include only those records from Snake River Basin ESU/DPS populations. The first population time-series were primarily derived from observations of redds and weir collections, and intended to represent NOSA including jacks (NOSAij). The NOSAij estimates were generated and submitted to Coordinated Assessments by Idaho Department of Fish Game, Washington Department of Fish and Game, Oregon Department of Fish and Game, and Nez Perce Tribe biologists. Specifc details regarding the estimation of NOSAij for each population can be found in agency reports [**need**

citations] or by contacting the contributing source listed in Coordinated Assessments metadata. The second population time-series represents escapement of natural-origin fish (including jack) to the population. Escapement estimates were derived from PIT-tag observations and a Bayesian hierarchical patch occupancy model described in Waterhouse et al. (2020) slightly modified by estimating time-varying transition probabilities which can account for 1) differential adult run timings among populations in the basin and 2) potentially inconsistent trapping and tagging rates at Lower Granite Dam. Further details regarding the escapement estimates can be found in **See2020** and **See2019**.

Abundance trends for individual sp/su Chinook Salmon and steelhead populations were analyzed using data from StreamNet's Coordinated Assessments database and passive integrated transponder (PIT) tag model results. For most combinations of return years and sp/su Chinook Salmon populations, data originated from the Coordinated Assessments database and were produced by the state or tribal co-managers responsible for monitoring those populations. The dataset was retrieved from Coordinated Assessments on January X, 2025, and filtered to only include Snake River sp/su Chinook Salmon and steelhead populations with natural-origin spawner abundance estimates, including jacks (NOSAij). For many Snake River populations, multiple NOSAij estimates were available for the same year due to variations in estimation methods (e.g., spawning ground surveys, mark-recapture) and/or data generators. In these cases, we selected annual estimates labeled as the method used by the National Oceanic and Atmospheric Administration (NOAA) technical recovery team. Steelhead abundance data were filtered from Coordinated Assessments by selecting estimates generated through the STate-space Adult Dam Escapement Model (STADEM) (See, Kinzer, and Ackerman, 2021) and the Dam Adult Branch Occupancy Model (DABOM) (See et al., 2016; Kinzer et al., 2020), as indicated in the "Comments" field. Population data for sp/su Chinook Salmon returning to Big Sheep Creek, Lookingglass Creek, Little Salmon River, and Panther Creek were unavailable in the Coordinated Assessments database. For these populations, we included estimates derived from the STADEM and DABOM models but obtained from **kinzerSnakeRiverBasin2020**.

[Figure of stochasiticity across time, and the synchronization after 1980.]

2.2 Dynamic Linear Model

The reported population abundance data generated by fisheries comanagers are not the exact truth, and in many cases, they are reported without levels of certainty (i.e., precision estimates) or measurement error. Reported estimates instead are generally the result of incomplete observations (i.e., spatial and temporal coverage is incomplete) and the product of combining information from multiple data collection activities and fish metrics together (i.e., redd count surveys, mark/recapture events, carcass collections), with each item having the possibility of being incorrect or biasing the abundance estimate from the unknown truth. To better understand the true unknown abundance and underlying trends contained within the data we removed measurement error using a multivariate dynamic linear model (DLM) (Zuur et al., 2003). A DLM is a time series model constructed as a type of state-space model that separates observation and state processes, and assumes the processes change dynamically following linear models (Royle and Dorazio, 2008). Dynamic linear models are capable of capturing the serial correlation in the data and can share covariance across multiple time-series, thus, effectively allowing information to be shared from data rich populations with long time-series, to data poor populations with short datasets or missing years.

Our DLM was fit to natural-log transformed abundance for each fish population simultaneously using shared parameters across all populations. We followed the methods described in Ford (2022), where the model was fit with a single variance and covariance across all populations for the process error, and a single variance term for observational error using the statistical software R (**R-base**) and the MARSS package (Holmes, Ward, and Wills, 2012; Holmes et al., 2024). We chose to include only those Snake River populations in the DLM with greater than 5 years of abundances to reduce risks of parameter bias and overfitting. More in depth model details and fitting procedures are described in the Ford (2022).

The unobserved state process model follows the linear equation,

$$x_t = x_{t-1} + u + w_t; w_t \sim MVN(0, \mathbf{Q})$$

where, x_t is an $mX1$ matrix of true unknown states for return year t. Parameter u is $mX1$ and represents drift between

Table 1: *Summary of abundance time-series used in modeling Snake River spring/summer Chinook Salmon populations (n = 48). Time-series length and data sources varied by population, with observations spanning from 1980 to 2024. Abundance estimates were primarily based on spawning ground surveys and weir counts, with PIT-tag detections included for select populations. The 10th, 50th (median), and 90th percentiles of observed abundance values are provided to summarize distributional characteristics across years.*

Grande Ronde / Imnaha

MPG	Population	Method	Years	# Years	10%	50%	90%
Dry Clearwater	Upper South Fork Clearwater	PIT-tag	2012-2024	12	144	273	757
	Big Sheep Creek	PIT-tag	2011-2024	13	20	49	120
	Catherine Creek	SGS and Weir	1980-2024	45	36	129	527
		PIT-tag	2015-2024	9	90	132	361
	Grande Ronde River Upper Mainstem	SGS and Weir	1980-2024	45	14	56	187
		PIT-tag	2018-2024	6	16	33	71
	Imnaha River Mainstem	SGS and Weir	1980-2024	45	189	417	1028
		PIT-tag	2011-2024	13	222	483	1062
	Lookingglass Creek	PIT-tag	2010-2024	14	38	88	228
	Minam River	SGS and Weir	1980-2024	45	141	327	684
	Wallowa/Lostine Rivers	SGS and Weir	1980-2023	44	72	300	1018
	Wenaha River	SGS and Weir	1980-2024	45	61	279	652
Middle Fork Salmon River Lower Snake		PIT-tag	2019-2024	5	119	133	404
	Asotin Creek	SGS and Weir	1984-2016	33	0	4	18
		PIT-tag	2010-2024	14	0	19	123
	Tucannon River	SGS and Weir	1980-2024	45	26	213	611
	Bear Valley Creek	SGS and Weir	1980-2024	45	85	244	880
		PIT-tag	2015-2024	9	10	232	648
	Big Creek	SGS and Weir	1980-2024	45	38	131	417
		PIT-tag	2012-2024	12	198	585	1121
	Camas Creek	SGS and Weir	1980-2024	44	9	43	104
	Chamberlain Creek	SGS and Weir	1985-2024	36	30	187	526
	Loon Creek	SGS and Weir	1980-2024	44	10	53	107
	Marsh Creek	SGS and Weir	1980-2024	45	31	167	578
	Middle Fork Salmon River Lower Mainstem	SGS and Weir	1987-2024	37	0	3	23
	Middle Fork Salmon River Upper Mainstem	SGS and Weir	1995-2024	30	21	57	154
South Fork Salmon River	Sulphur Creek	SGS and Weir	1980-2024	45	6	41	182
	East Fork South Fork Salmon River	SGS and Weir	1987-2024	38	62	212	503
		PIT-tag	2010-2024	14	243	629	1079
Upper Salmon River	Secesh River	SGS and Weir	1996-2024	29	161	434	1045
		PIT-tag	2010-2024	14	270	666	1191
	South Fork Salmon River	SGS and Weir	1980-2024	45	162	551	1235
		PIT-tag	2010-2024	14	213	675	2534
	East Fork Salmon River	SGS and Weir	1980-2024	45	18	219	630
	Lemhi River	SGS and Weir	1980-2024	45	60	142	363
		PIT-tag	2010-2024	14	129	235	665
	North Fork Salmon River	SGS and Weir	1991-2024	34	6	52	184
		PIT-tag	2016-2024	7	40	60	211
	Pahsimeroi River	SGS and Weir	1980-2024	37	24	122	354
	Panther Creek	PIT-tag	2018-2024	6	93	166	303
	Salmon River Lower Mainstem	SGS and Weir	1980-2024	45	19	98	230
	Salmon River Upper Mainstem	SGS and Weir	1980-2024	45	100	326	679
	Valley Creek	SGS and Weir	1980-2024	45	13	77	229
		PIT-tag	2010-2024	14	87	242	484
	Yankee Fork	SGS and Weir	1980-2024	44	3	23	148
		PIT-tag	2012-2024	12	35	62	279
Wet Clearwater	Lochsa River	PIT-tag	2017-2024	7	160	245	461
	Lolo Creek	PIT-tag	2012-2024	12	41	78	257

Table 2: *Summary of abundance time-series used in modeling Snake River summer steelhead populations (n = 25). Time-series length and data sources varied by population, with observations spanning from 2010 to 2024. Abundance estimates were primarily based on PIT-tag detections, with spawning ground surveys and weir counts included for select populations. The 10th, 50th (median), and 90th percentiles of observed abundance values are provided to summarize distributional characteristics across years.*

Clearwater River MPG	Population	Method	Years	# Years	10%	50%	90%
	Clearwater River Lower Mainstem	SGS and Weir	2010-2024	15	147	254	785
	Lochsa River	SGS and Weir	2017-2024	8	289	439	941
	Lolo Creek	SGS and Weir	2012-2024	13	104	187	589
Grande Ronde River	Selway River	SGS and Weir	2017-2024	8	232	408	802
	South Fork Clearwater River	SGS and Weir	2012-2024	13	131	353	1040
	Grande Ronde River Lower Mainstem	SGS and Weir	2019-2023	5	281	418	467
	Grande Ronde River Upper Mainstem	SGS and Weir	2013-2024	12	356	539	1301
		PIT-tag	2010-2018	9	1300	2556	3589
	Joseph Creek	SGS and Weir	2011-2024	14	372	747	2045
		PIT-tag	2010-2017	8	1487	1990	3735
Salmon River	Wallowa River	SGS and Weir	2014-2024	11	359	508	956
Imnaha River Lower Snake	Imnaha River	SGS and Weir	2011-2024	14	683	1241	2881
	Asotin Creek	SGS and Weir	2010-2024	15	224	366	1312
	Tucannon River	SGS and Weir	2010-2024	15	232	452	841
		PIT-tag	2010-2023	14	347	611	1196
	East Fork Salmon River	SGS and Weir	2011-2019	8	0	14	40
	Lemhi River	SGS and Weir	2010-2024	15	49	161	416
	Middle Fork Salmon River Lower Mainstem	SGS and Weir	2011-2024	14	84	244	586
	Middle Fork Salmon River Upper Mainstem	SGS and Weir	2020-2024	5	19	29	53
	North Fork Salmon River	SGS and Weir	2017-2024	7	17	57	378
	Pahsimeroi River	SGS and Weir	2011-2024	14	9	33	140
	Panther Creek	SGS and Weir	2018-2024	7	101	137	196
	Salmon River Upper Mainstem	SGS and Weir	2010-2024	15	36	106	297
	Secesh River	SGS and Weir	2010-2024	15	27	57	252
	South Fork Salmon River	SGS and Weir	2010-2024	15	154	445	1511

subsequent return years for each state m. Process error (w_t) is assumed multivariate normal with mean zero and the mXm variance-covariance matrix $\mathbf{Q}$.

The observation model is represented as,

$$y_t = \mathbf{Z}x_t + a + v_t; v_t \sim MVN(0, \mathbf{R})$$

where, y_t is an $nX1$ matrix of observed abundance at return year t for n independent time-series. Return year abundances for each n and t observation is mapped to the unknown state (x_t) with the nXm matrix $\mathbf{Z}$. Parameter a is $nX1$ and provides a y-intercept scaling (i.e., bias correction) for the m^{th} state and each n^{th} time-series. Observational error (v_t) is modeled multivariate normal with mean zero and the nXn variance-covariance matrix $\mathbf{R}$.

We modeled four different scenarios to represent true unknown states for Snake River Chinook Salmon and steelhead populations. The scenarios included an individual state for each of the n time-series, states for each of the populations, states for each of the major population groups (MPGs), and a single state for the entire Snake River Basin. Parameters u and a were set to unequal across the modeled states and time-series mapped to each state.

We modeled three different variance-covariance matrices for process error and two variance-covariance matrices of observation error for each of the four state scenarios. The first process error matrix was assumed to have equal variance and covariance terms for each state, the second and third options assumed no covariance and either equal or unequal variances on the diagonal. The observation error matrix was assumed to have either equal or unequal variance on the diagonal, and off diagonal covariance terms equal to zero.

The different combinations resulted in 24 model runs fit with the MARSS package (Holmes, Ward, and Wills, 2012) and the statistical programming language R (**R-base**). The best model was selected as having the lowest bootstrapped AIC (AICb) value of all successfull (i.e., converged) model runs (Holmes, Ward, and Wills, 2012; Holmes et al., 2024).

Residuals from selected models were examined visually for assumption violations of heterogeneity and normality.

2.3 Annual Growth Rate

Ford (2022) examined sp/su Chinook Salmon and steelhead 15-year (2004-2019) population trends using the fitted slope parameter from a linear regression model with year as the independent variable. From our initial examination of abundance data at the aggregate (i.e., Lower Granite Dam) and population levels, it appeared that abundance was relatively low in the beginning and end of the 15-year time period used in Ford (2022). We hypothesized that linear models fit to the last 15 years of population data would effectively be anchored at similar beginning and end points, resulting in less steep trends and potential insignificant slope parameters. We decided to alter the dBSA approach, and instead capture the magnitude of the more recent downward trend that we are currently experiencing. To do so, we choose to use only the last 10-years of abundance returns beginning in 2011. We based our rational on a visual change point (i.e., trend/slope switches direction from positive to negative) in sp/su Chinook Salmon and steelhead returns starting around 2011, and the desire to include two generations of returns to capture contemporary patterns in population trends.

We estimated the population trend using a linear regression model with year and population as predictor variables, and the natural-log transformed abundance (i.e., state process) from the DLM model as the response. First, the full linear model was fit including an interaction term for year and population to allow for different population y-intercept and slope parameters. We then fit a reduced model, excluding the interaction term, which constrained populations to a common slope parameter while allowing for different population y-intercept parameters. The best fitting model was determined with an F-test (Casella and Berger, 2021)

2.4 Above Abundance Thresholds

The proportion of populations currently meeting the QET was calculated as the number of populations below the threshold divided by the total populations included in the DLM analysis. We determined populations were currently below the QET threshold if the DLM abundance estimates fell below 50 spawners for each of the last four years (**2017-2020**). The number of future populations meeting the QET threshold was determined with predictions from the trend analysis (i.e., objective 2) and the best fitting linear model. A population was classified as below the QET when returns within the next 5-years of predictions fell below 50 in a single year, because all subsequent predictions would also be below 50 due to the linear model.

3 Results

3.1 Dynamic Linear Model

The best fitting model for sp/su Chinook Salmon included separate state processes for each major population group [**??**], whereas summer steelhead was best represented by a single basin-wide process [**??**]. For both species, the difference in AICc values between the top two models exceeded 10.00, a threshold commonly interpreted to indicate the second ranked model has little to no support as compared to the first ranked model (Burnham and Anderson, 2004). For sp/su Chinook Salmon, the process error matrix included two parameters: one for an equal variance and the other for an equal covariance across the seven MPG processes. In contrast, the best fitting model for summer steelhead included only a single basin-wide process, as such, process error consisted of a single variance parameter. Observational error for both species was best modeled with unique variance parameters for each time-series.

We evaluated model assumptions by visually examining residual plots for patterns indicating autocorrelation, non-normality, or heteroscedasticity. Overall, these diagnostic checks supported the adequacy of the model structure and parameter estimates [8; 9]. However, in a few populations residuals did show evidence of serial correlation and any future models may benefit from additional terms. model convergence was confirmed with MARSS package output indicating the optimization algorithm reached a stable peak in the likelihood (**holmes_2021**).

Table 3: *Candidate dynamic linear models fitted to 48 spring/summer Chinook Salmon abundance time-series to evaluate population dynamics and identify models best explaining Snake River trends. The number of state processes estimated in each model is indicated in column U. Unique state process error parameters are denoted in column Q, and observation error parameters in column R. Model fit is summarized using log-likelihood (logLik), corrected Akaike Information Criterion (AICc), and relative model support (ΔAIC).*

Model Id	U	Q	R	logLik	AICc	ΔAIC
4	7	2	48	-1409.39	3048.45	0.00
2	1	1	48	-1466.43	3146.20	97.75
10	34	2	48	-1434.48	3163.32	114.87
8	7	7	48	-1483.20	3207.85	159.39
6	7	1	48	-1497.10	3221.52	173.07
3	7	2	1	-1592.07	3307.45	259.00
1	1	1	1	-1644.39	3396.90	348.45
9	34	2	1	-1619.06	3421.60	373.15
7	7	7	1	-1658.74	3451.76	403.30
5	7	1	1	-1675.38	3471.89	423.44
14	34	34	48	-1652.97	3680.66	632.20
12	34	1	48	-1710.60	3713.10	664.65
13	34	34	1	-1787.79	3833.66	785.20
11	34	1	1	-1894.17	3969.55	921.09

Table 4: *Candidate dynamic linear models fitted to 25 summer steelhead abundance time-series to evaluate population dynamics and identify models best explaining Snake River trends. The number of state processes estimated in each model is indicated in column U. Unique state process error parameters are denoted in column Q, and observation error parameters in column R. Model fit is summarized using log-likelihood (logLik), corrected Akaike Information Criterion (AICc), and relative model support (ΔAIC).*

Model Id	u	Q	R	logLik	AICc	ΔAIC
2	1	1	25	-173.6063	477.5700	0.00000
4	5	2	25	-172.4959	490.7482	13.17829
10	22	2	25	-152.4930	508.5072	30.93727
6	5	1	25	-197.1921	537.0077	59.43771
8	5	5	25	-195.7174	546.7564	69.18648
1	1	1	1	-290.3841	645.4864	167.91640
12	22	1	25	-227.4331	654.7353	177.16530
3	5	2	1	-290.2449	657.8599	180.28994
9	22	2	1	-279.5091	683.3980	205.82803
7	5	5	1	-304.4895	694.1822	216.61226
5	5	1	1	-313.8774	702.5548	224.98481
13	22	22	1	-258.2618	705.6388	228.06887
11	22	1	1	-300.5271	722.4827	244.91276
14	22	22	25	-221.9866	728.4811	250.91111

Over the full time-series State processes

3.2 Annual Growth Rate

3.3 Population Status

In recent years, sp/su Chinook Salmon and steelhead returns to the Snake River basin have decreased quickly and are expected to continue declining into the future. In the last 10 years, abundances at LGD ranged from 4,108 to 26,351 from 2011-2020 for sp/su Chinook Salmon, and steelhead ranged from 8,287 to 45,789. A moving average suggests potential declines in sp/su Chinook Salmon abundance began as early as 2010 (Figure), and around 2015 for steelhead

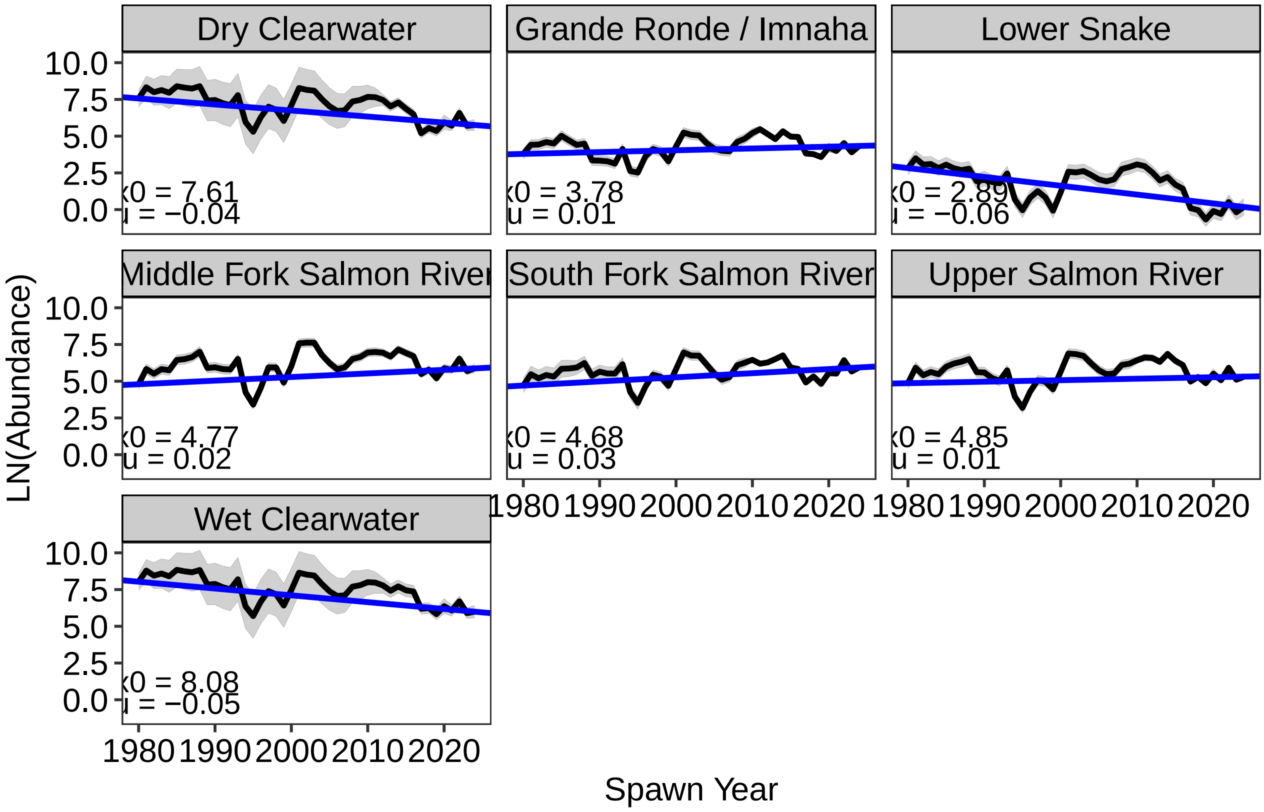

Figure 2: *Estimated abundance (natural-log) trends for the seven state processes (xtT) estimated from the best fitting Snake River spring/summer Chinook Salmon model (grey shading represents 95% CI's). The abundance trend for each process across the full time-series is shown by the dashed line and generated from the estimated drift (u) and initial abundance (x0) parameters.*

(Figure). Mantua et al. (1997) and Crozier et al. (2021) have shown returning abundance is highly correlated with ocean metrics (e.g., pacific decadal oscillation and sea surface temperature), and as such, the recent decreases in abundance are expected given the additional stress of poor ocean conditions.

returning fish continuously fail in reaching abundance thresholds and goals. Sp/su Chinook Salmon returning to the Snake River have only exceeded the aggregated MAT (i.e., sum of population MAT) in three years post ESA listing, and in more recent years returns have been closer to the aggregated QET–similar to the levels observed in the mid-1990's (Figure). Steelhead returns to the basin follow a similar annual abundance pattern, but have exceeded the aggregated MAT in more years and remain above the QET (Figure). Neither species, however, has had abundances near aggregated healthy and harvestable goals set by the CBP at any point since the construction of LGD in 1975.

4 Conclusions

The best fitting models for sp/su Chinook Salmon and steelhead describe biological processes that are logical and match our expectations and understanding of the system. Populations within an MPG are geographically close, often have similar habitat and rearing conditions, exposed to relatively similar migration distances, and managed with similar goals and objectives (Copeland et al., 2024). It stands to reason populations with these similar underlying traits would have a similar trend. Beginning in the late 1970's, after the last mainstem dam was constructed on the Lower Snake River, populations became more synchronous and showed less annual run-size diversity (@ref(fig.synchronicity)). Thus, a common process variance and co-variance between MPG processes match our expectation that populations

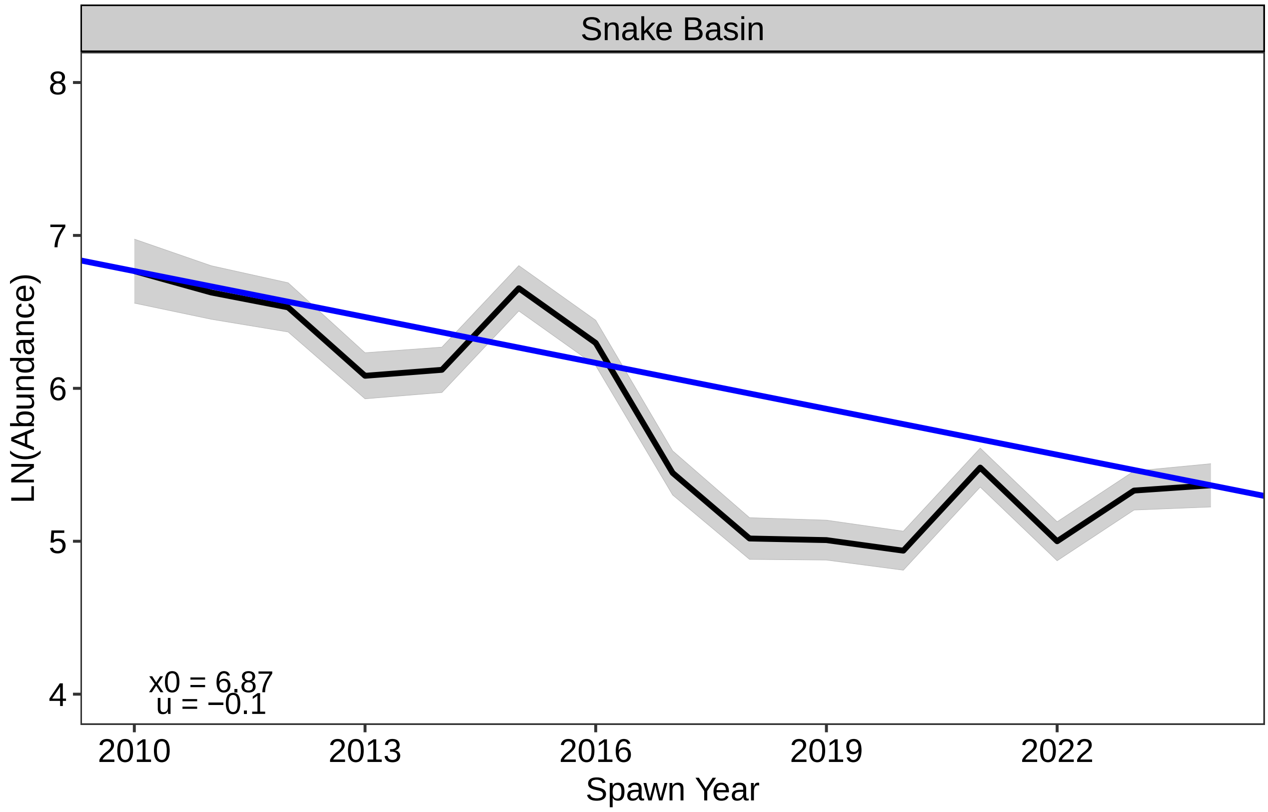

Figure 3: *Estimated abundance (natural-log) trend for a single state process (xtT) as estimated from the best fitting Snake River summer steelhed model (grey shading represents 95% CI's). The abundance trend for each process across the full time-series is shown by the dashed line and generated from the estimated drift (u) and initial abundance (x0) parameters.*

respond similarly to out-of-basin factors. Last, observations in each time-series are often generated by data collected at different spatial and temporal scales within the population, and analyzed by different people and agencies using different methods. Unequal variances across time-series also make sense.

The Lower Snake River and Clearwater River populations showed the fastest rate of decline over the last 10 years. These areas are lowest in elevation and contain some of the most degraded habitat in the basin.

The models indicated population time-series have been largely stationary since 1980 when the analysis began, with short periods of increases and decreases. Currently, the Snake River Basin is experiencing a declining trend that has, or will, push many populations to below the quasi-extinction threshold if continued. Species recovery after reaching this depressed state will be difficult and will require a major improvement in their environment to increase productivity. Productivity increases may come from a cooler climate and more hospitable ocean conditions, restored habitat and migration corridors, or the combination of many items. The model is clear, however, despite the fish management actions taken over the last 45 years, populations have seen little to no improvement, and recently started declining at a faster rate than previously observed.

Our analysis was intended to offer us a greater understanding of NOAA's dBSA, but instead yielded dire results that call for action. We learned that all sp/su Chinook Salmon populations experienced the same rate of decline (19%) from 2011-2020, and the decline was remarkably similar to the common steelhead population decline (18%). We also learned that 13 sp/su Chinook Salmon populations (42%) are currently at levels considered "quasi-extinct", and 24 (77%) will be at the extinction risk threshold if current trends continue. Fewer steelhead populations (n = 3, 19%) are experiencing an immediate risk of extinction, but their rate of decline suggest seven (44%) populations will reach QET within 5-years.

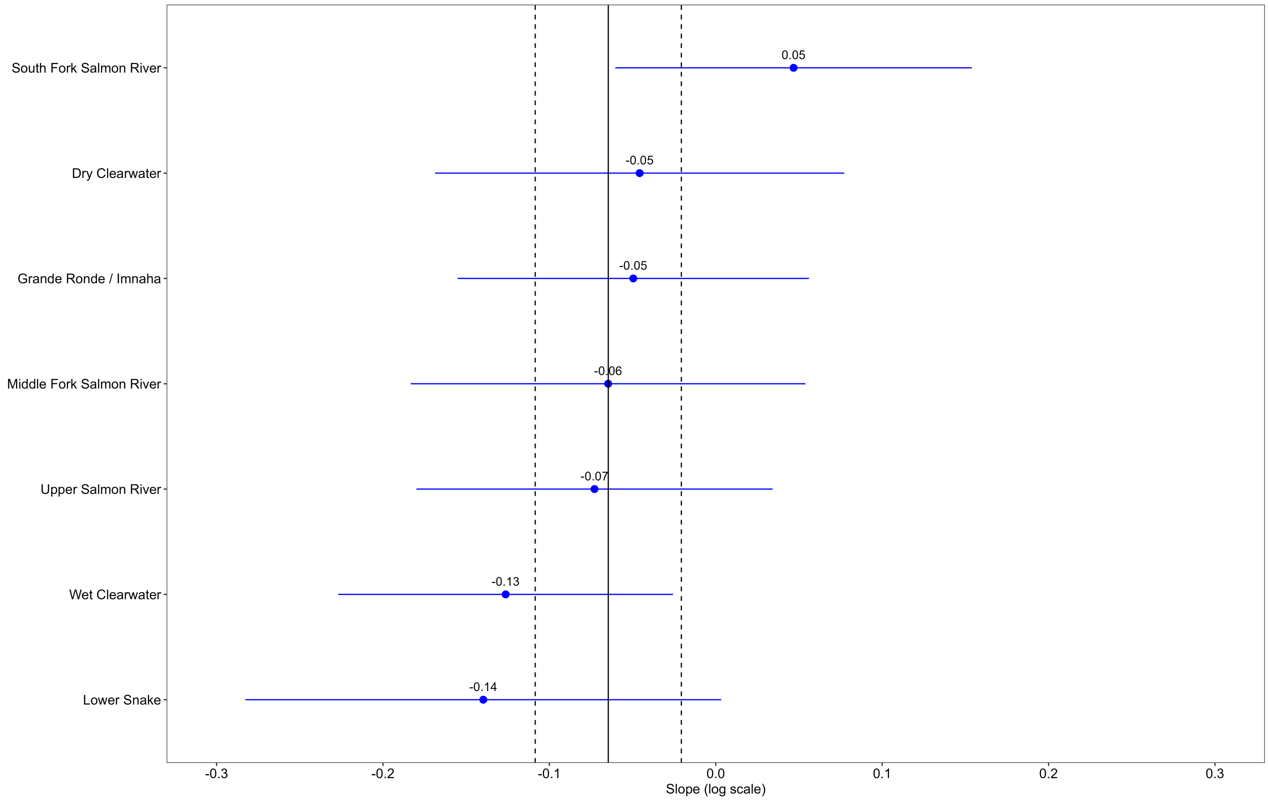

Figure 4: *Estimated slope parameters for natural-origin Snake River spring/summer Chinook Salmon abundance trends indicate an average annual decline of 6% for the last 10-years.*

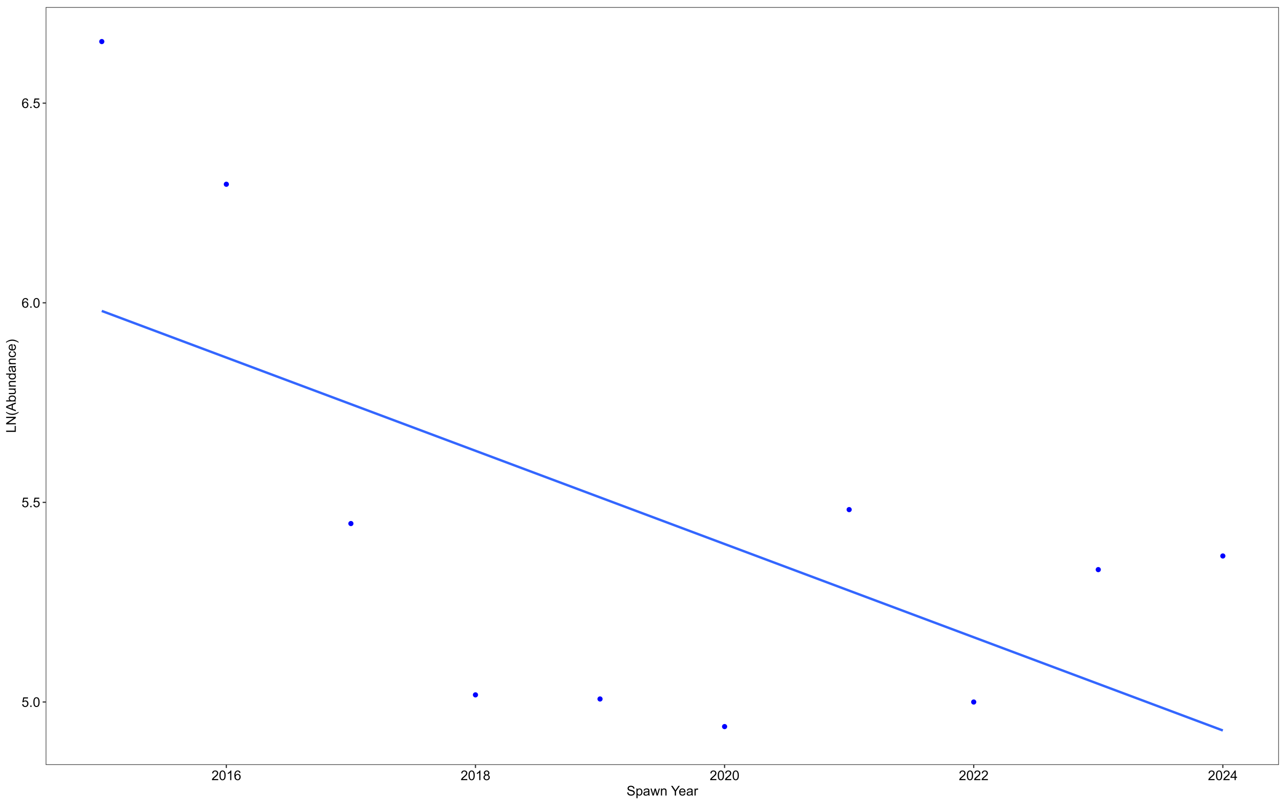

Figure 5: *Modeled abundance trends of natural-origin Snake River summer steelhead indicate an annual 11% decline for the last 10-years.*

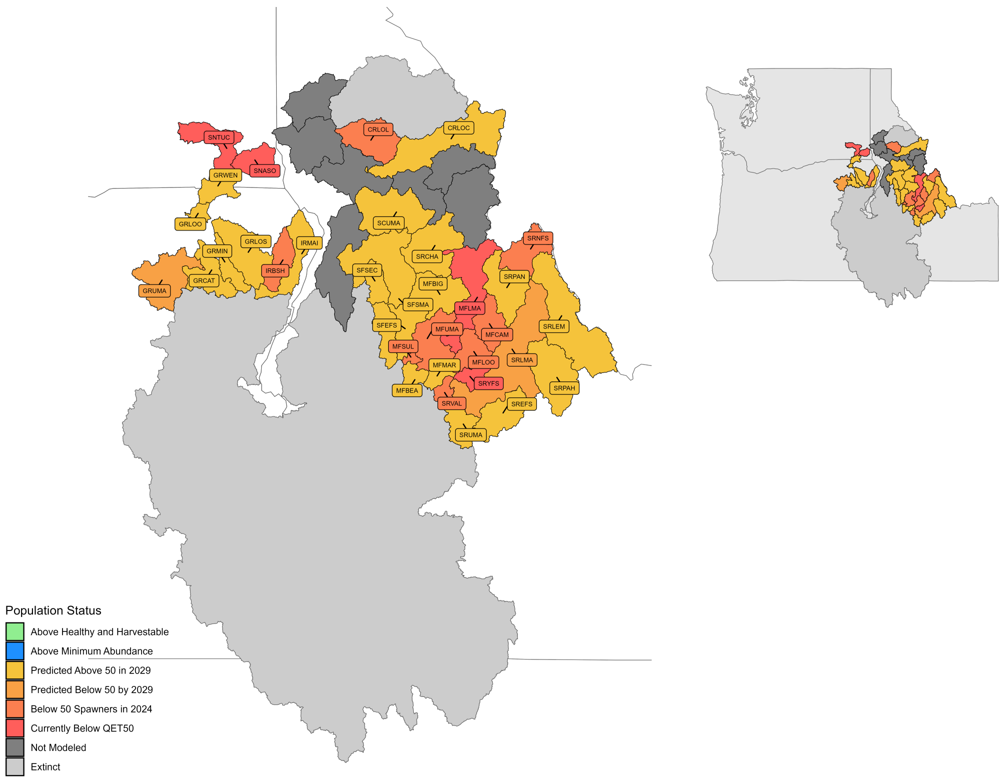

Figure 6: *Current status of natural-origin Snake River spring/summer Chinook Salmon relative to the quasi-extinction threshold (QET) and Columbia Basin Partnership goals.*

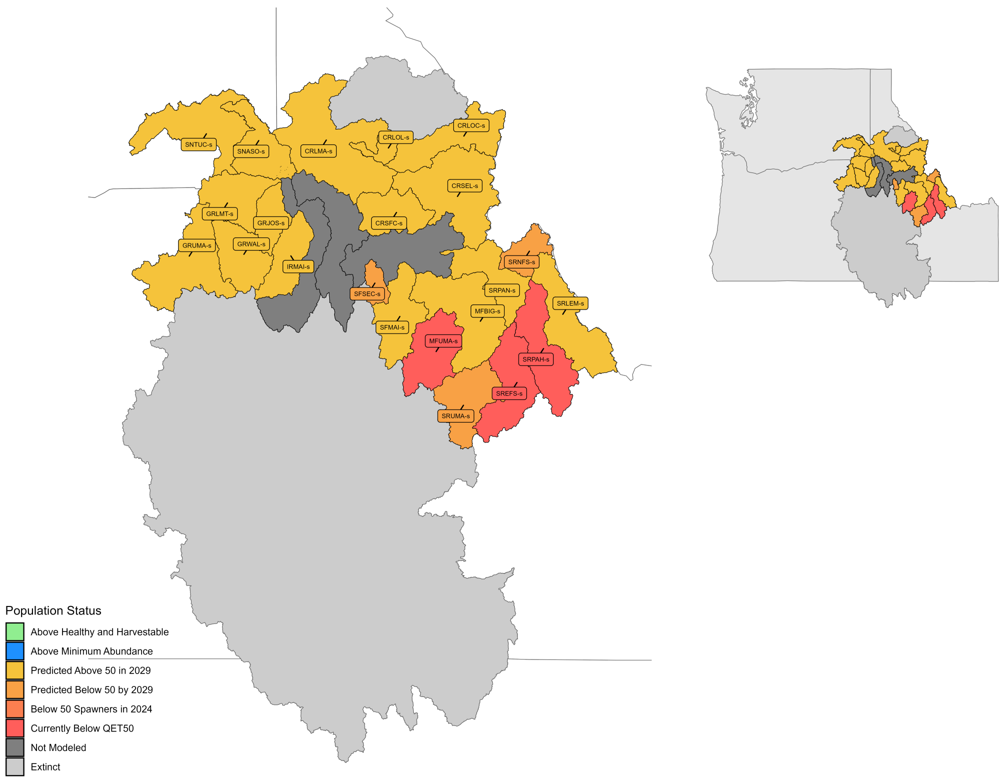

Figure 7: *Current status of natural-origin Snake River summer steelhead relative to the quasi-extinction threshold (QET) and Columbia Basin Partnership goals.*

Our linear predictions assume the current 10-year trend will continue, ignores serial correlation, and disregards important environmental drivers that may be cyclical (e.g., pacific decadal oscillation). Spawn year 2021 returns of sp/su Chinook Salmon and steelhead have returned in higher numbers than we predicted and provide a good example to discuss. The higher realized returns in 2021, resulting from recently improved ocean conditions, were not predicted by our model, but they were also not unexpected. Given our model and assumptions, our predictions should be viewed more as a long-term average trend. We did not attempt to predict actual annual returns by capturing year-to-year variability, management changes, and fluctuating environmental conditions. Instead, we were interested in understanding the average return across years and populations; thus, actual returns are expected to be higher and lower than our results. And although the higher returns of 2021 are welcomed, they still fall short of delisting criteria thresholds, and healthy and harvestable goals.

In the last 30 years since ESA listing of salmon and steelhead, billions of dollars were spent to recover and increase salmon and steelhead returns to the the Snake River. The actions we have taken included small and large scale habitat restoration projects, major hydrosystem over halls and passage improvements, and we have released millions of hatchery fish to jump start recovery. Although each of these actions have provided small localized benefits, our analysis shows Snake River anadromous populations are acting similarly, which suggests common drivers of survival for all populations (i.e., hydrosystem and ocean). Climate change scenarios predict worsen ocean conditions for salmon and steelhead, of which, our actions can not immediately change or improve. We can, however, change and improve migration and rearing conditions for all salmon and steelhead in the Lower Snake River and Columbia River.

5 Code and Data Availability

The code and processed data to reproduce the analyses, summaries, tables, and figures are available on the GitHub project repository https://github.com/ryankinzer/SRAFS.git. Raw time-series observation data are available from Stream Net's Coordinated Assessments Indicators of Fish Population Health web application; https://cax.streamnet.org.

6 Acknowledgements

Providing the current status of Snake River fish stocks would not be possible without the gigantic efforts of hundreds of biologists, technicians, and field crews who have collected monitoring data across the basin over the past three decades. We are especially grateful to the many tribal, state, and federal agency staff whose long-term dedication to data collection, stewardship, and recovery efforts have provided the foundation for this analysis. We also thank the regional biologists and data managers who ensured the consistency and integrity of datasets spanning multiple generations of monitoring programs. Evan Brown, Rebecca Waskovich and Luciano Chiramonte from Idaho Department of Fish and Game, Joseph Feldhaus and Kasey Bliesner from Oregon Department of Fish and Wildlife, and Ethan Crawford and Michael Gallinat from Washington Department of Fish and Wildlife deserve a special thanks for meeting our data requests and helping us understand the methods and nuances. We thank our, now retired, program manager, David Johnson, for encouraging this work and using it to guide future recovery decisions and inform policy makers. Manuscript review and guidance pre-journal submittal were generously provided by **placeholder**, **placeholder**, **placeholder**, and **placeholder**. We also extend our gratitude to the anonymous journal reviewers whose thoughtful feedback greatly improved the quality and clarity of this manuscript.

7 References

8 Supplemental Material

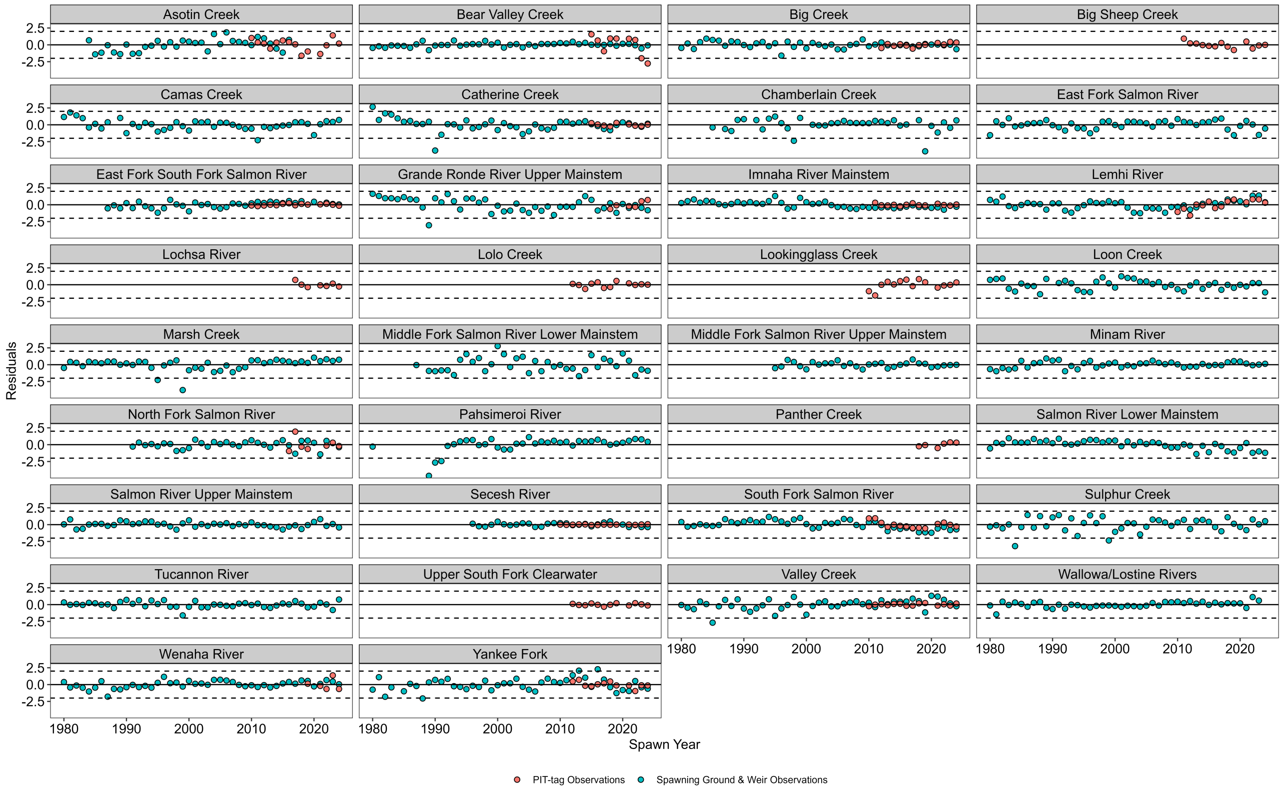

Figure 8: *Residuals from the best fitting spring-summer Chinook Salmon MARSS model.*

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