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Hanford Reach Fall Chinook Salmon Productivity Analysis, Brood Years 1975–2016

December 2022

RA Harnish
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Prepared for the U.S. Department of Energy
under Contract DE-AC05-76RL01830

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RA Harnish¹
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Prepared for
the Pacific Salmon Commission
under DOE Contract No. DE-AC05-76RL01830

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Executive Summary

The Hanford Reach fall Chinook salmon productivity analysis was updated to include seven additional brood years (BY) of data (BY 2010–2016) to update management parameter estimates, identify the mechanism driving density dependence in the Hanford Reach, conduct a more thorough evaluation of past and present Priest Rapids Dam operations, and assess factors outside of the Hanford Reach that may be affecting adult production. Stock–recruit analyses were used to determine the effect of Priest Rapids Dam operations on the freshwater productivity of the Hanford Reach fall Chinook salmon population for BY 1975–2016. Freshwater productivity was defined as the number of presmolts (recruits) produced from a BY divided by the egg escapement (stock) present to produce that brood. This definition of productivity ensured that only the life stages expected to be directly affected by Priest Rapids Dam operations in the Hanford Reach were considered. Of the models tested, the Beverton-Holt model provided the best fit to the data, and management parameters were calculated. Both the Beverton-Holt and Ricker models were fit to the egg-to-presmolt survival estimates from each Priest Rapids Dam operational period to determine whether the productivity (α) parameter differed among operation periods. The spawning abundance required to achieve maximum sustainable yield (S_{MSY}) was estimated from fitting the Ricker AR1 model to adult/spawner data from BY 1975–2016. Factors outside of the Hanford Reach that may be limiting adult production were evaluated for BY 1997–2016 by regressing annual presmolt-to-age-2 survival estimates against inriver and ocean covariates using simple and multivariable regression. Inriver covariates included measures of Columbia River discharge and temperature and an index of spill at Columbia River dams located downstream of the Hanford Reach. Ocean covariates included local and basin-scale ocean condition indicators thought to be biologically important for their influence on the source waters that feed the northern California Current, which affect zooplankton and forage fish communities.

Freshwater productivity of the Hanford Reach fall Chinook salmon population appeared to have increased with each flow fluctuation constraint enacted on the operations of Priest Rapids Dam. From the Beverton-Holt model fit to each period separately, the productivity parameter (α) was estimated to be 0.39 presmolts/egg for the pre-Vernita Bar Settlement Agreement (VBSA) period (BY 1975–1988), 0.58 presmolts/egg during the VBSA period (BY 1989–2003), and 0.81 presmolts/egg during the Hanford Reach Fall Chinook Protection Program period (HRFCPP; BY 2004–2016). Using the Ricker model and analysis of covariance to test for a period effect, the productivity parameter was estimated to be 0.31 presmolts/egg for the pre-VBSA period, 0.36 presmolts/egg for the VBSA period, and 0.43 presmolts/egg for the HRFCPP period.

Although productivity appeared to increase as a result of the HRFCPP operations, the estimated egg-to-presmolt survival was below the 42-year average for each new BY added to the data set. Below-average egg-to-presmolt survival was to be expected due to the high escapements and the density dependent relationship observed in past productivity analyses. However, the mechanism driving density dependence remained unknown after past productivity analyses. With the inclusion of BY 2010–2016, the Beverton-Holt model provided the best fit to the egg-to-presmolt data, indicating juvenile competition for limited resources is driving density dependence in the Hanford Reach.

Since implementation of the interim VBSA in 1984, maximum recruitment (R_{max}) was estimated to be 87.8 million presmolts, with a spawning stock size of about 59,800 adult spawners necessary to produce $\frac{1}{2}$ of R_{max} . These management parameter estimates agree well with those obtained from fitting the Ricker AR1 model to the adult/spawner data. Excluding BY 2015, which was identified as a clear outlier with a high degree of leverage on model parameters, the Ricker productivity parameter (α) was estimated to be 13.84 adults/spawner, the spawning stock size required to produce maximum sustainable yield (S_{MSY}) was

estimated to be 48,925 adult spawners, maximum adult recruitment (R_{max}) was estimated to be 299,936 adults, and the escapement necessary to produce R_{max} was 58,912 adult spawners.

Analyses conducted to identify limiting factors outside of the Hanford Reach revealed the importance of ocean conditions on adult production. The May–September Pacific Decadal Oscillation (PDO) index explained 72% of the variability in presmolt-to-age-2 survival in a simple linear regression model. Adults per spawner estimates were fit as a function of egg-to-presmolt survival and the PDO index to evaluate the relative contribution of freshwater and ocean effects on adult production. These two variables explained 88% of the variability in spawner-to-adult survival with the PDO index explaining 3 times more variability than Hanford Reach (i.e., egg-to-presmolt) survival.

Updating the Hanford Reach fall Chinook salmon productivity analysis to include seven additional years of data (BY 2010–2016), including three years of record escapements (>150,000 adults), reaffirmed the highly productive nature of this population and the positive effect of flow fluctuation constraints as implemented at Priest Rapids Dam by the VBSA and HRF CPP. The addition of several years of high escapements also revealed that competition among juveniles (and not spawning habitat limitations) is likely the mechanism driving density dependence in the Hanford Reach. The addition of the high escapement brood years also provided evidence for the high recruitment capacity of the population. As such, the Washington Department of Fish and Wildlife (WDFW) may consider increasing the maximum escapement goal from 42,000 adults to 49,000 adults, which would be more in-line with S_{MSY} as estimated from this update. Given the in-river protections that are in place as part of the HRF CPP, ocean conditions appear to be the primary factor controlling adult production. Therefore, the success of any additional management actions taken in freshwater are likely to be dependent on survival in the ocean. As such, the most effective management actions may be those that limit juvenile competition and density dependent growth effects in the Hanford Reach. Limiting competition would be expected to increase growth, resulting in larger smolts that would be less likely to suffer size-dependent mortality in the marine environment, even during years of poor ocean conditions. Increasing terminal harvest rates when high escapements are anticipated (from downstream dam counts) is one management option that could help to curtail juvenile competition.

Acronyms and Abbreviations

AABM	Aggregate Abundance-Based Management
AEQ	adult equivalents
AICc	small sample size corrected Akaike's Information Criterion
ANCOVA	analysis of covariance
BY	brood year(s)
CPD	coefficient of partial determination
CWT	coded wire tag
DART	Data Access in Real Time
DOY	day of year
FERC	Federal Energy Regulatory Commission
Grant PUD	Public Utility District No. 2 of Grant County
HR	Hanford Reach
HRFCPP	Hanford Reach Fall Chinook Protection Program Agreement
IHR	Ice Harbor Dam
m	meter(s)
MCN	McNary Dam
MSE	mean squared error(s)
NDBC	National Data Buoy Center
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
ONI	Oceanic Niño Index
PDO	Pacific Decadal Oscillation
PIT	passive integrated transponder
PRD	Priest Rapids Dam
PRH	Priest Rapids Hatchery
PSC CTC	Pacific Salmon Commission Chinook Technical Committee
PUD	Public Utility District
r	Pearson product-moment correlation coefficient
R^2	coefficient of determination
$\bar{R}^2$	adjusted coefficient of determination
rkm	river kilometer(s)
R_{max}	maximum recruitment
SE	standard error

SEAK	southeast Alaska
S_{max}	the stock size expected to generate maximum recruitment
S_{MSY}	the stock size expected to generate maximum sustainable yield
SSE	sum of squared error(s)
SST	sea surface temperature
TAC	Oregon Technical Advisory Committee
URB	upriver bright
VBSA	Vernita Bar Settlement Agreement
WDFW	Washington Department of Fish and Wildlife

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1.0 Introduction

1.1 Background

Priest Rapids Dam is located in southeastern Washington State on the Columbia River at river kilometer (rkm) 639 and marks the upstream boundary of the Hanford Reach (Figure 1.1). The Hanford Reach, which extends downstream to the head of McNary Pool (rkm 557), is the most productive mainstem spawning area for fall Chinook salmon (*Oncorhynchus tshawytscha*) in the entire Columbia River basin and supports one of the largest spawning populations of fall Chinook salmon in the Pacific Northwest (Huntington et al. 1996; Dauble and Watson 1997; Harnish et al. 2012, 2014a). This productivity is particularly noteworthy because many naturally spawning anadromous fish populations of the Columbia River basin have experienced declines in the recent past (NMFS 2022a, 2022b, 2022c, 2022d, 2022e).

Chinook salmon that spawn in the Hanford Reach comprise the majority of the Columbia River upriver bright (URB) stock, which is a driver stock for several important fisheries. The URB stock is an important contributor to Aggregate Abundance-Based Management (AABM) fisheries under the Pacific Salmon Treaty and a primary contributor to Columbia River fisheries. For example, in 2014 and 2015, the URB stock comprised more than 40% of southeast Alaska (SEAK) and northern British Columbia troll catches.

Priest Rapids Dam is part of the seven-dam hydroelectric complex on the mid-Columbia River that includes Wanapum, Rock Island, Rocky Reach, Wells, Chief Joseph, and Grand Coulee dams. This complex of dams is typically operated under a load-following mode to meet electrical demand in the Pacific Northwest and California. Therefore, stream flow in the Hanford Reach is governed largely by hydropower generation, which can result in substantial hourly and daily fluctuations in discharge (Langshaw and Pearsons 2010). Without some operational constraints, these fluctuations would likely lead to widespread dewatering of fall Chinook salmon redds and stranding or entrapment of juvenile fall Chinook salmon in the Hanford Reach.

In 1988, following nine years of intensive studies and experimental protection measures, the Vernita Bar Settlement Agreement (VBSA) was signed by the Public Utility District No. 2 of Grant County (Grant PUD), Chelan PUD, Douglas PUD, Bonneville Power Administration, National Oceanic and Atmospheric Administration (NOAA) Fisheries, Washington Department of Fish and Wildlife (WDFW), Oregon Department of Fish and Wildlife, and the Yakama, Umatilla, and Colville tribes. The agreement included management constraints on flows to minimize fall Chinook salmon spawning above elevations that are frequently dewatered. Flow is manipulated during the fall Chinook salmon spawning period using reverse load factoring at Priest Rapids Dam. The intent of reverse load factoring is to limit Chinook salmon redd site selection (which was thought to occur mainly during daylight hours) to lower elevations on Vernita Bar by reversing the normal load-following pattern, providing low flows during the day and higher flows at night. Spawning activities are monitored to establish the protection level flow, which is the minimum flow required to keep redds fully watered through the incubation period. The protection level minimum flow ceases after emergence is estimated to be completed (from accumulated thermal units). Although the VBSA was an important step in protecting fall Chinook salmon redds on Vernita Bar from dewatering, adaptive management led to development of the Hanford Reach Fall Chinook Protection Program Agreement (HRFCPP) in 2004 to provide additional protections during other life stages (Langshaw and Pearsons 2010).

Prior to the Interim HRFCPP (1999–2003) and the signed HRFCPP (2004 to present), large fluctuations (up to 2.1 m/h and 4 m/d) would occur in the Priest Rapids Dam tailrace during the period of fall Chinook salmon emergence and rearing. These fluctuations caused stranding, entrapment, and ultimately mortality

of juvenile fall Chinook salmon (Nugent et al. 2002a, 2002b, 2002c, 2002d; McMichael et al. 2003). The HRF CPP provided constraints on flow minimums and/or fluctuations from Priest Rapids Dam during fall Chinook salmon spawning, incubation, emergence, and early rearing periods to prevent dewatering of redds during incubation and to prevent stranding and entrapment of juveniles during emergence and early rearing.

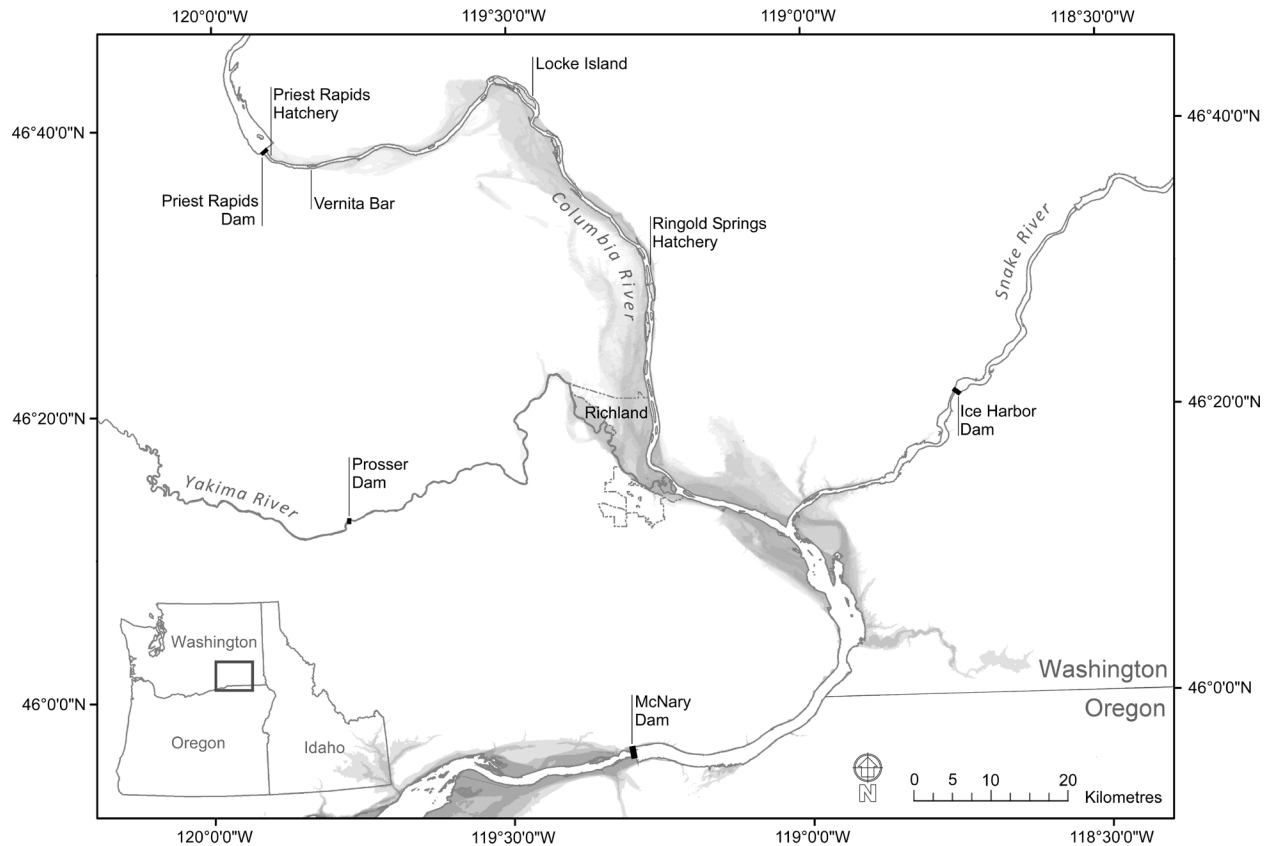


Figure 1.1. The Hanford Reach of the Columbia River Extends from Priest Rapids Dam Downstream to the Head of McNary Pool Located Near Richland, Washington.

A new operating license for Priest Rapids Dam was issued by the Federal Energy Regulatory Commission (FERC) on April 17, 2008. Conditions of the 401 Water Quality Certification, issued as part of FERC's relicensing of Priest Rapids Project, required that Grant PUD develop a study plan with the objective of better understanding whether hydroelectric operations of Priest Rapids Dam under the HRF CPP are causing substantial harm to fall Chinook salmon in the Hanford Reach. Previous stock–recruit analyses indicated the Hanford Reach fall Chinook salmon population was highly productive prior to the implementation of the VBSA and HRF CPP (Bernard and Clark 1999; Peters et al. 1999; Langness and Reidinger 2003). Therefore, an analysis was needed to evaluate how operations of Priest Rapids Dam under these flow constraint agreements have affected productivity of fall Chinook salmon in the Hanford Reach.

The original productivity analysis utilized stock–recruit modeling to determine the effect of Priest Rapids Dam operations on the presmolt/egg productivity (Ricker α parameter) of the Hanford Reach fall Chinook salmon population for brood years (BY) 1975–2004 (Harnish et al. 2012, 2014a). The results from the original productivity analysis indicated that the constraints on flow fluctuations from Priest Rapids Dam

during the periods of spawning, incubation, and rearing increased presmolt/egg productivity of the Hanford Reach fall Chinook salmon population. However, limitations of the data prevented thorough examinations of productivity, potential productivity bottlenecks, density dependence, and carrying capacity under the VBSA and HRF CPP due to the narrow escapement range and small number of BY over which these operational regimes could be evaluated.

The productivity analysis was updated in 2017 to include five additional BY, including three years (2007, 2008, and 2009) of low adult escapements that fell below WDFW's escapement goal (Harnish 2017). As such, the updated analysis provided a wider range of escapements over which to evaluate the effect of Priest Rapids Dam operations on productivity of the Hanford Reach URB fall Chinook salmon population. Results from this analysis reaffirmed the highly productive nature of the population and the positive effect of flow fluctuation constraints as implemented by the HRF CPP. However, the mechanism driving the density-dependent relationship and the factor(s) controlling carrying capacity remained unclear due to a lack of high escapement years in the data set. Results from the 2017 update also indicated that factors outside of the Hanford Reach appear to be influencing adult production. Record adult escapements of URB fall Chinook salmon, which greatly exceeded escapement goals, occurred in 2013, 2014, and 2015. The return of all adults from these broods, which has now been completed, provided an opportunity to conduct a more comprehensive evaluation of the mechanism driving density dependence and the factor(s) controlling carrying capacity in the Hanford Reach.

1.2 Study Objectives

The objectives of this study were to (1) update the Hanford Reach fall Chinook salmon productivity analysis to include BY 2010–2016, (2) estimate egg-to-presmolt, presmolt-to-adult, and spawner-to-adult survival for each BY added to the data set, (3) identify the stock-recruit model that best fits the egg-to-presmolt data and estimate α and β parameters, maximum recruitment (R_{max}), and the stock size required to achieve maximum recruitment (S_{max}) and maximum sustainable yield (S_{MSY}) for the entire 42-year dataset, and (4) evaluate these estimates to provide a better understanding of the factors controlling density dependence and carrying capacity, conduct a more thorough evaluation of past and present Priest Rapids Dam operations, and evaluate factors outside of the Hanford Reach that influence adult production.

2.0 Methods

2.1 Juvenile Production

Stock–recruit analyses were used to achieve most of the project objectives. For the evaluation of juvenile production, productivity was defined as the number of presmolts (recruits) produced from a BY divided by the number of eggs (stock) that were present in the Hanford Reach to produce that brood. This definition of productivity ensured that only the life stages expected to be directly affected by Priest Rapids Dam operations in the Hanford Reach were considered. Egg escapement was used as the measure of stock because it provides a better measure of reproductive potential than adult or female escapement.

2.1.1 Stock Estimation

The number of eggs in female fall Chinook salmon in the Hanford Reach was estimated for each BY 1975–2016 by multiplying age-specific female escapement estimates by age-specific fecundity values. (Table 2.1). Escapement of adult (age-3+) Hanford Reach fall Chinook salmon spawners is based on dam ladder counts, sport and tribal harvest, and tributary spawning populations. Ladder counts at Priest Rapids (adjusted for fallback), Ice Harbor, and Prosser dams; Hanford Reach and Yakima River sport and tribal harvest; returns to Priest Rapids and Ringold Springs hatcheries; and spawning escapement in Moran Creek and the lower Yakima River (downstream of Prosser Dam) are subtracted from the ladder count at McNary Dam to estimate adult and jack fall Chinook salmon escapement to the Hanford Reach.

The number of adults of each age was estimated for each BY 1975–2016 by multiplying adult escapement estimates by the proportion of each age (Table 2.1). The proportion of each age was estimated from scales collected during carcass surveys. Carcass surveys have been conducted by WDFW throughout the Hanford Reach annually since 1979. During carcass surveys, biological data (e.g., length, gender, and age [scales]) are recorded through a random systematic design, whereby every k^{th} fish is sampled, with a goal of sampling 633 carcasses annually (Hoffarth 2010). For years in which age data were not available (BY 1975–1980, 1995), age structure data for the entire run of URBs was used (Harlan et al. 1998).

Next, the number of females of each age was estimated for each BY 1986–2016 by multiplying the age-specific escapement estimates by the proportion of females from each age group. Because there is typically a positive bias toward females associated with carcass recovery (Hoffarth 2010; Hankin et al. 2011), the proportion of females from each age group was obtained from data collected during WDFW Hanford Reach sport fishery creel surveys, whereby length, gender, and age (scales) were recorded for all coded-wire tag (CWT) salmon and from every fourth fish encountered during the surveys (Table 2.2). For BY 1975–1985, prior to WDFW creel surveys, average female proportions from BY 1986–2021 were used. Finally, the number of eggs in females was estimated (Table 2.2) by multiplying the number of females of each age by the age-specific mean fecundity of fall Chinook salmon adult females that returned to Priest Rapids Hatchery in 2011. Of the 441 hatchery females sampled, 291 were age-3 with a mean fecundity of 3,698 (SE = 54) eggs, 59 were age-4 with a mean fecundity of 4,379 (SE = 118) eggs, and 91 were age-5 with a mean fecundity of 4,652 (SE = 110) eggs. The total escapement of eggs in females was estimated for each brood year by adding the age-specific estimates.

Table 2.1. Adult Escapement (from WDFW), Proportions of Each Age (Prop.), and Estimated Number of Adults of Each Age (*N*) for Hanford Reach Fall Chinook Salmon, BY 1975–2021. Age structure data obtained from WDFW carcass surveys for BY 1981–1994, 1996–2021, and from Harlan et al. (1998) for BY 1975–1980, 1995.

Brood Year	Adult Escapement	Age-3		Age-4		Age-5+	
		Prop.	<i>N</i>	Prop.	<i>N</i>	Prop.	<i>N</i>
1975	22,242	0.352	7,829	0.476	10,577	0.172	3,836
1976	21,140	0.520	11,002	0.336	7,108	0.143	3,030
1977	31,528	0.413	13,029	0.505	15,913	0.082	2,586
1978	20,577	0.210	4,318	0.674	13,871	0.116	2,388
1979	23,559	0.105	2,483	0.723	17,035	0.172	4,041
1980	21,861	0.155	3,387	0.380	8,312	0.465	10,162
1981	15,116	0.091	1,369	0.715	10,814	0.194	2,933
1982	20,543	0.310	6,366	0.466	9,575	0.224	4,602
1983	36,022	0.262	9,445	0.663	23,891	0.075	2,686
1984	41,982	0.212	8,917	0.613	25,726	0.175	7,339
1985	65,796	0.270	17,777	0.441	29,016	0.289	19,003
1986	72,559	0.258	18,738	0.560	40,622	0.182	13,199
1987	88,761	0.291	25,866	0.470	41,725	0.239	21,170
1988	74,034	0.122	9,005	0.541	40,085	0.337	24,944
1989	65,913	0.084	5,559	0.221	14,583	0.694	45,771
1990	40,117	0.039	1,581	0.424	16,995	0.537	21,541
1991	31,971	0.015	469	0.279	8,907	0.707	22,595
1992	29,448	0.110	3,236	0.458	13,499	0.432	12,713
1993	30,650	0.064	1,957	0.642	19,691	0.294	9,002
1994	48,857	0.025	1,245	0.447	21,828	0.528	25,784
1995	38,381	0.216	8,289	0.182	6,991	0.602	23,101
1996	37,547	0.240	9,001	0.566	21,249	0.194	7,297
1997	34,007	0.028	944	0.682	23,180	0.291	9,883
1998	29,410	0.189	5,560	0.133	3,919	0.678	19,931
1999	27,013	0.127	3,434	0.679	18,348	0.194	5,231
2000	36,028	0.077	2,765	0.372	13,416	0.551	19,847
2001	51,394	0.223	11,462	0.540	27,762	0.237	12,169
2002	69,343	0.164	11,367	0.612	42,439	0.224	15,537
2003	89,312	0.052	4,677	0.556	49,618	0.392	35,017
2004	79,464	0.138	10,968	0.211	16,739	0.651	51,757
2005	64,355	0.110	7,095	0.557	35,849	0.333	21,411
2006	47,095	0.079	3,725	0.365	17,206	0.556	26,164
2007	13,888	0.098	1,364	0.546	7,587	0.355	4,937
2008	23,361	0.194	4,533	0.371	8,662	0.435	10,166
2009	26,345	0.103	2,714	0.645	16,989	0.252	6,642
2010	80,408	0.273	21,958	0.409	32,902	0.318	25,549
2011	65,724	0.123	8,052	0.712	46,785	0.166	10,888
2012	51,818	0.274	14,182	0.411	21,274	0.316	16,362
2013	157,484	0.507	79,871	0.456	71,816	0.037	5,797
2014	152,052	0.081	12,271	0.866	131,648	0.053	8,133
2015	233,927	0.198	46,352	0.466	108,908	0.336	78,667
2016	109,860	0.145	15,939	0.561	61,648	0.294	32,273
2017	71,303	0.102	7,254	0.493	35,118	0.406	28,931

Brood Year	Adult Escapement	Age-3		Age-4		Age-5+	
		Prop.	N	Prop.	N	Prop.	N
2018	42,277	0.305	12,898	0.457	19,313	0.238	10,066
2019	58,856	0.161	9,492	0.790	46,472	0.049	2,892
2020	65,923	0.170	11,221	0.677	44,633	0.153	10,069
2021	49,569	0.135	6,671	0.743	36,824	0.123	6,074

Table 2.2. Estimated Proportion (Prop.) and Number of Females (N) in Each Age Group and the Estimated Number of Eggs in those Females for Hanford Reach Fall Chinook Salmon, BY 1975–2021. Proportions of females were obtained from WDFW creel surveys conducted in the Hanford Reach during BY 1986–2021; averages from these BYs were used to estimate female proportions by age for BY 1975–1985. Total eggs represent the summation of eggs within that brood year across all ages.

Brood Year	Age-3 Females			Age-4 Females			Age-5+ Females			Total Eggs (1,000s)
	Prop.	N	Eggs (1,000s)	Prop.	N	Eggs (1,000s)	Prop.	N	Eggs (1,000s)	
1975	0.222	1,736	6,420	0.606	6,409	28,065	0.596	2,285	10,630	45,115
1976	0.222	2,440	9,022	0.606	4,307	18,860	0.596	1,805	8,397	36,279
1977	0.222	2,889	10,685	0.606	9,642	42,223	0.596	1,540	7,166	60,074
1978	0.222	958	3,541	0.606	8,405	36,805	0.596	1,423	6,618	46,963
1979	0.222	551	2,036	0.606	10,322	45,200	0.596	2,407	11,198	58,435
1980	0.222	751	2,778	0.606	5,036	22,055	0.596	6,053	28,161	52,993
1981	0.222	304	1,123	0.606	6,553	28,693	0.596	1,747	8,128	37,944
1982	0.222	1,412	5,221	0.606	5,802	25,406	0.596	2,741	12,753	43,379
1983	0.222	2,095	7,745	0.606	14,476	63,931	0.596	1,600	7,443	78,580
1984	0.222	1,977	7,312	0.606	15,588	68,260	0.596	4,372	20,338	95,910
1985	0.222	3,942	14,578	0.606	17,582	76,990	0.596	11,320	52,661	144,229
1986	0.094	1,752	6,479	0.671	27,261	119,378	0.684	9,031	42,011	167,868
1987	0.091	2,351	8,695	0.657	27,417	120,061	0.690	14,601	67,924	196,680
1988	0.125	1,126	4,163	0.692	27,751	121,521	0.571	14,253	66,305	191,988
1989	0.214	1,191	4,405	0.610	8,893	38,941	0.646	29,568	137,551	180,897
1990	0.286	452	1,670	0.667	11,331	49,617	0.741	15,970	74,295	125,582
1991	0.235	110	408	0.692	6,160	26,975	0.689	15,572	72,443	99,826
1992	0.125	405	1,496	0.564	7,613	33,339	0.644	8,191	38,104	72,940
1993	0.100	196	724	0.528	10,401	45,545	0.564	5,077	23,619	69,888
1994	0.000	0	0	0.674	14,721	64,462	0.625	16,115	74,967	139,429
1995	0.231	1,913	7,075	0.778	5,438	23,811	0.650	15,016	69,853	100,739
1996	0.199	1,788	6,614	0.610	12,962	56,760	0.542	3,958	18,412	81,786
1997	0.211	199	735	0.615	14,265	62,466	0.629	6,215	28,914	92,115
1998	0.138	765	2,831	0.668	2,618	11,465	0.635	12,658	58,886	73,183
1999	0.301	1,033	3,820	0.564	10,339	45,275	0.412	2,154	10,021	59,116
2000	0.165	455	1,683	0.663	8,894	38,946	0.545	10,826	50,361	90,989
2001	0.207	2,370	8,764	0.570	15,826	69,302	0.549	6,676	31,058	109,124
2002	0.172	1,955	7,228	0.614	26,063	114,128	0.460	7,149	33,257	154,612
2003	0.095	442	1,636	0.569	28,255	123,727	0.583	20,427	95,024	220,388
2004	0.188	2,066	7,642	0.531	8,884	38,902	0.614	31,788	147,877	194,421
2005	0.228	1,616	5,975	0.633	22,682	99,326	0.513	10,991	51,130	156,430
2006	0.000	0	0	0.538	9,248	40,498	0.548	14,328	66,653	107,151
2007	0.208	284	1,051	0.570	4,327	18,947	0.453	2,236	10,400	30,398

Brood Year	Age-3 Females			Age-4 Females			Age-5+ Females			Total Eggs (1,000s)
	Prop.	N	Eggs (1,000s)	Prop.	N	Eggs (1,000s)	Prop.	N	Eggs (1,000s)	
2008	0.237	1,076	3,978	0.527	4,570	20,013	0.400	4,066	18,916	42,907
2009	0.254	689	2,547	0.517	8,787	38,480	0.431	2,863	13,318	54,345
2010	0.253	5,546	20,510	0.432	14,208	62,216	0.449	11,464	53,331	136,057
2011	0.230	1,852	6,848	0.603	28,225	123,599	0.548	5,971	27,776	158,223
2012	0.256	3,631	13,426	0.679	14,443	63,245	0.680	11,126	51,758	128,429
2013	0.310	24,758	91,557	0.583	41,893	183,449	1.000	5,797	26,965	301,971
2014	0.331	4,065	15,031	0.626	82,464	361,109	0.590	4,796	22,313	398,453
2015	0.422	19,555	72,314	0.640	69,731	305,352	0.714	56,138	261,152	638,818
2016	0.345	5,500	20,339	0.573	35,345	154,775	0.543	17,531	81,554	256,669
2017	0.440	3,194	11,813	0.579	20,347	89,099	0.722	20,894	97,201	198,114
2018	0.256	3,307	12,230	0.526	10,165	44,511	0.510	5,132	23,872	80,614
2019	0.376	3,571	13,205	0.601	27,930	122,305	0.714	2,066	9,609	145,118
2020	0.265	2,969	10,979	0.662	29,554	129,419	0.593	5,967	27,757	168,155
2021	0.397	2,651	9,804	0.585	21,528	94,271	0.563	3,417	15,894	119,970

2.1.2 Recruit Estimation

The results from cohort reconstructions of natural-origin CWT releases were used to estimate the number of presmolts and age-3 adult equivalents produced from each BY. The general approach of the cohort reconstruction method began with escapement estimates, then worked backward in time to account for interdam loss, in-river and ocean harvest, and natural ocean mortality, to estimate the number of age-2 ocean recruits produced. This estimate was then divided by a CWT-based estimate of survival from tagging (as presmolts) to age-2. Adult escapement estimates were obtained from WDFW for BY 1975–2021 (Table 2.1). Jack escapements, estimated annually since 1990, were also obtained from WDFW for BY 1990–2021 (Table 2.3). For BY 1975–1989, jack escapement was estimated using the same “subtraction method” used by WDFW to estimate escapement, whereby counts of jacks at Ice Harbor Dam, Priest Rapids Dam, Priest Rapids Hatchery, Hanford Reach sport fishery harvest, Ringold Springs Hatchery, Yakima River escapement and sport harvest, and Wanapum tribal fisheries were subtracted from the count of jacks at McNary Dam. However, data of Hanford Reach sport fishery jack harvest, Ringold Springs Hatchery jack returns, Yakima River jack escapement and sport harvest, and Wanapum tribal jack harvest were either not estimated or not available for many of these years (Table 2.4). Most years, Ringold jack returns, Yakima River jack escapement and harvest, and Wanapum jack harvest made up less than 3% of the McNary Dam jack count. Therefore, these values were included in the jack escapement estimation when available but otherwise ignored. Harvest of jacks in the Hanford Reach sport fishery was typically about 5% of the McNary Dam jack count and was linearly correlated with the count at McNary Dam for years in which these data were available (Figure 2.1). Hanford Reach sport harvest of jacks was estimated using this linear relationship for years in which no data were available (1975–1985; Table 2.4).

Table 2.3. Estimated Hanford Reach Fall Chinook Salmon Jack Escapement, 1975–2015

Year	Jacks	Year	Jacks	Year	Jacks
1975	20,723	1991	20,225	2007	8,385
1976	44,473	1992	12,503	2008	5,697
1977	35,660	1993	6,697	2009	10,374
1978	8,673	1994	14,246	2010	6,608

Year	Jacks	Year	Jacks	Year	Jacks
1979	9,523	1995	16,827	2011	9,532
1980	3,033	1996	5,701	2012	5,897
1981	6,152	1997	9,486	2013	17,356
1982	13,675	1998	5,983	2014	31,162
1983	17,634	1999	2,800	2015	32,401
1984	33,713	2000	11,993	2016	6,427
1985	66,266	2001	15,708	2017	2,456
1986	89,150	2002	15,167	2018	4,344
1987	34,400	2003	11,196	2019	7,135
1988	41,931	2004	8,231	2020	8,909
1989	14,868	2005	7,612	2021	12,224
1990	17,034	2006	4,606		

Table 2.4. Estimated Numbers of Fall Chinook Salmon Jacks that Passed McNary (MCN), Ice Harbor (IHR), and Priest Rapids (PRD) Dams, Escaped To or were Harvested in the Yakima River, Returned to Ringold Springs or Priest Rapids (PRH) Hatcheries, or were Harvested in the Hanford Reach (HR) Sport Fishery or Wanapum Tribal Fishery, 1975–1989. Data unavailability is indicated by n/a.

Year	MCN	IHR	PRD	Yakima (below Prosser)	Yakima (above Prosser)	Yakima Sport	Ringold	PRH	HR Sport	Wanapum
1975	39,119	658	9,045	n/a	n/a	n/a	n/a	6,951	n/a	n/a
1976	59,191	374	5,301	n/a	n/a	n/a	n/a	6,240	n/a	n/a
1977	46,770	556	2,796	n/a	n/a	n/a	n/a	5,650	n/a	n/a
1978	15,469	493	1,736	n/a	n/a	n/a	383	3,518	n/a	n/a
1979	17,879	855	2,869	n/a	n/a	n/a	0	3,849	n/a	n/a
1980	8,911	579	2,410	n/a	n/a	n/a	n/a	2,564	n/a	n/a
1981	11,869	1,332	1,593	n/a	n/a	n/a	n/a	2,309	n/a	n/a
1982	26,196	1,892	4,330	n/a	n/a	n/a	14	5,119	n/a	n/a
1983	25,233	964	2,364	n/a	116	n/a	50	2,982	n/a	n/a
1984	48,878	795	4,672	n/a	637	n/a	n/a	6,846	n/a	n/a
1985	109,881	7,118	8,944	n/a	92	n/a	n/a	18,800	n/a	n/a
1986	127,277	2,679	10,387	n/a	238	n/a	710	11,588	12,525	n/a
1987	46,745	1,624	4,787	n/a	64	n/a	2,131	1,939	1,800	n/a
1988	54,068	2,035	4,570	n/a	42	2	n/a	1,685	3,803	n/a
1989	19,251	1,352	2,253	n/a	0	n/a	n/a	381	397	n/a

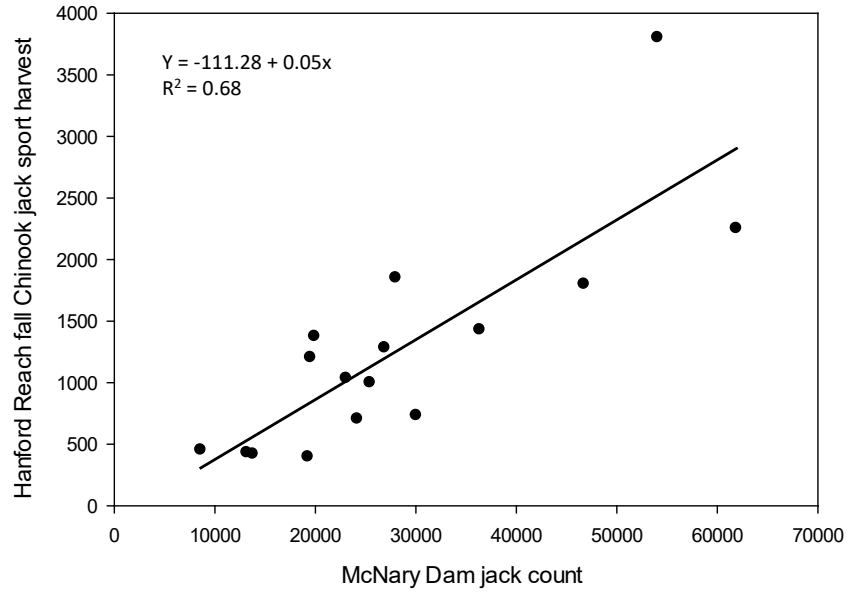


Figure 2.1. Relationship between Harvest of Fall Chinook Salmon Jacks in the Hanford Reach Sport Fishery and the Count of Fall Chinook Salmon Jacks at McNary Dam. Relationship was developed using data from 1987–1993 and 2001–2009.

The proportion of adult and jack hatchery spawners was estimated from recoveries of CWT hatchery fish during spawning ground surveys for return years 1979–2021 (Table 2.5) using the equation

$$CarcHatchProp_{BY} = \frac{\sum \left(\frac{CarcHatchRec_{CWT,BY}}{PropCWT_{CWT,BY}} \right)}{Carc_{BY}}$$

where $CarcHatchRec_{CWT,BY}$ = the number of hatchery carcasses with a specific CWT code recovered during Hanford Reach fall Chinook salmon carcass surveys during a run year,
 $PropCWT_{CWT,BY}$ = the proportion of the hatchery release group associated with a specific CWT code that was implanted with CWTs (obtained from the Fish Passage Center and the Regional Mark Information System database), and
 $Carc_{BY}$ = the number of fall Chinook salmon carcasses sampled in the Hanford Reach during a run year.

Table 2.5. Estimated Proportion (Prop.) and Numbers (N) of Adult and Jack Fall Chinook Salmon that Spawned in the Hanford Reach that were of Hatchery Origin, 1975–2021.

Year	Prop. Jacks	N Jacks	Prop. Adults	N Adults
1975	0.053	1,098	0.110	2,441
1976	0.053	2,357	0.110	2,320
1977	0.053	1,890	0.110	3,460
1978	0.053	460	0.110	2,258
1979	0.000	0	0.071	1,664
1980	0.000	0	0.124	2,713

Year	Prop. Jacks	N Jacks	Prop. Adults	N Adults
1981	0.000	0	0.052	779
1982	0.143	1,958	0.111	2,287
1983	0.000	0	0.051	1,839
1984	0.001	41	0.056	2,335
1985	0.081	5,338	0.015	988
1986	0.036	3,215	0.068	4,932
1987	0.000	0	0.106	9,408
1988	0.086	3,616	0.113	8,338
1989	0.000	0	0.044	2,902
1990	0.000	0	0.044	1,755
1991	0.000	0	0.044	1,417
1992	0.000	0	0.002	61
1993	0.000	0	0.006	196
1994	0.000	0	0.043	2,101
1995	0.416	6,995	0.074	2,838
1996	0.000	0	0.122	4,587
1997	0.000	0	0.128	4,357
1998	0.000	0	0.052	1,525
1999	0.002	7	0.228	6,172
2000	0.001	9	0.280	10,088
2001	0.047	742	0.072	3,702
2002	0.064	967	0.138	9,586
2003	0.000	0	0.132	11,771
2004	0.027	226	0.106	8,450
2005	0.000	0	0.175	11,250
2006	0.000	0	0.136	6,414
2007	0.070	585	0.062	860
2008	0.000	0	0.108	2,534
2009	0.016	164	0.236	6,211
2010	0.000	0	0.082	6,606
2011	0.073	694	0.162	10,647
2012	0.360	2,124	0.091	4,712
2013	0.143	2,490	0.302	47,609
2014	0.095	2,975	0.064	9,756
2015	0.027	880	0.081	18,963
2016	0.071	454	0.099	10,826
2017	0.000	0	0.108	7,704
2018	0.340	1,476	0.061	2,596
2019	0.000	0	0.067	3,947
2020	0.154	1,371	0.245	16,158
2021	0.026	312	0.356	17,657

Average hatchery proportions from 1979–2021 were used for 1975–1978 when no carcass surveys were conducted in the Hanford Reach. After hatchery spawners were subtracted, age-specific escapement

estimates were adjusted by adult- and jack-specific Bonneville Dam to McNary Dam conversion rates (Table 2.6) to account for interdam loss, which is any mortality not accounted for during upstream migration. Next, the age-specific run to the mouth of the Columbia River was constructed by dividing the interdam loss-adjusted estimates by the in-river harvest rates obtained from CWT data (Pacific Salmon Commission Chinook Technical Committee [PSC CTC]; Table 2.7). Priest Rapids Hatchery CWT groups were used to estimate in-river harvest rates for years when no wild Hanford Reach CWT groups existed or when recoveries of wild Hanford Reach CWT fish were low.

Table 2.6. Bonneville Dam to McNary Dam Conversion Rates for Adult (age-3+) and Jack (age-2) Fall Chinook Salmon, 1975–2021 (Cindy LeFleur, Stuart Ellis, U.S. vs. Oregon Technical Advisory Committee (TAC), Personal Communication, February 2012, November 2016, May 2022). Jack conversion rates were unavailable from the TAC for years 2012–2021; therefore, conversion rates of PIT-tagged URB jacks were used for these years.

Return Year	Age-2	Age-3+	Year	Age-2	Age-3+	Year	Age-2	Age-3+
1975	0.838	0.772	1991	0.744	0.864	2007	0.768	0.827
1976	0.725	0.460	1992	0.728	0.915	2008	0.673	0.798
1977	0.689	0.803	1993	0.751	0.864	2009	0.728	0.894
1978	0.333	0.495	1994	0.831	0.857	2010	0.621	0.882
1979	0.482	0.560	1995	0.712	0.889	2011	0.707	0.851
1980	0.516	0.505	1996	0.578	0.809	2012	0.957	0.992
1981	0.393	0.329	1997	0.707	0.824	2013	0.810	0.900
1982	0.463	0.519	1998	0.878	0.757	2014	0.789	0.972
1983	0.547	0.746	1999	0.540	0.770	2015	0.882	0.960
1984	0.546	0.794	2000	0.714	0.697	2016	0.903	0.990
1985	0.767	0.935	2001	0.782	0.689	2017	0.750	0.960
1986	0.708	0.995	2002	0.830	0.805	2018	0.811	1.000
1987	0.698	0.924	2003	0.939	0.705	2019	0.905	0.970
1988	0.786	0.986	2004	0.856	0.650	2020	0.797	0.920
1989	0.631	0.949	2005	0.978	0.733	2021	0.833	0.960
1990	0.793	0.898	2006	0.838	0.653			

Table 2.7. In-River Harvest Rates of Hanford Reach Fall Chinook Salmon, brood years 1975–2018. Rates in bold indicate Priest Rapids Hatchery CWT groups were used to estimate harvest. Wild Hanford Reach CWT groups were used to estimate all other harvest rates. n/a = data not available.

Brood Year	Age-2	Age-3	Age-4	Age-5+
1975	7.4%	24.9%	53.8%	12.8%
1976	0.1%	56.5%	24.0%	7.1%
1977	8.9%	10.2%	14.3%	3.5%
1978	8.1%	1.9%	5.4%	0.0%
1979	<0.1%	3.3%	20.6%	17.8%
1980	1.6%	12.0%	40.6%	53.0%
1981	3.6%	47.6%	69.9%	55.2%
1982	5.3%	59.6%	53.8%	66.3%
1983	16.2%	62.1%	63.7%	51.4%
1984	20.1%	68.6%	85.8%	71.8%
1985	0.2%	64.5%	81.0%	49.5%
1986	56.6%	63.4%	48.3%	32.7%

Brood Year	Age-2	Age-3	Age-4	Age-5+
1987	0.1%	73.1%	28.9%	32.3%
1988	0.1%	75.1%	28.4%	24.6%
1989	0.1%	39.6%	33.8%	12.0%
1990	47.6%	14.4%	27.6%	11.5%
1991	0.1%	4.5%	35.8%	28.7%
1992	0.1%	9.7%	36.0%	19.9%
1993	14.8%	45.5%	25.1%	30.1%
1994	0.2%	10.8%	14.5%	15.0%
1995	0.1%	12.9%	18.0%	39.5%
1996	0.1%	<0.1%	47.9%	11.8%
1997	0.1%	48.2%	40.6%	25.2%
1998	0.1%	6.8%	16.5%	28.2%
1999	18.8%	24.3%	22.2%	22.3%
2000	0.1%	24.2%	48.2%	23.8%
2001	23.3%	62.0%	39.0%	25.6%
2002	0.1%	15.7%	20.5%	10.2%
2003	0.1%	8.2%	17.8%	57.7%
2004	0.2%	<0.1%	39.4%	0.0%
2005	0.2%	<0.1%	82.7%	28.1%
2006	7.9%	41.3%	22.6%	65.8%
2007	0.1%	9.9%	32.1%	60.6%
2008	0.1%	20.0%	47.7%	59.4%
2009	54.8%	19.5%	41.2%	52.0%
2010	9.4%	20.1%	27.9%	36.9%
2011	0.1%	18.9%	26.4%	39.9%
2012	13.0%	15.0%	36.8%	30.9%
2013	0.1%	12.5%	46.0%	18.6%
2014	0.1%	<0.1%	21.5%	10.2%
2015	<0.1%	3.8%	8.8%	30.5%
2016	5.9%	2.0%	14.1%	n/a
2017	0.1%	41.2%	n/a	n/a
2018	5.9%	n/a	n/a	n/a

Next, ocean recruits were calculated for each age using the cohort analysis algorithm of the PSC CTC, ocean harvest rates obtained from CWT data (PSC CTC), and conditional ocean survival rates currently used by the PSC CTC ($S_2 = 0.60$, $S_3 = 0.70$, $S_4 = 0.80$, and $S_5 = 0.90$). Priest Rapids Hatchery CWT groups were used to estimate ocean exploitation rates for years when no wild Hanford Reach CWT groups existed (Table 2.8). For an individual CWT group, the approach began with the oldest age and worked back through successive ages until ocean age-2 was reached. The following equations outline this approach:

$$\hat{A}_{5+}(t+5, t+6) = \frac{(((E_5(t+5) + E_6(t+6))/I_5(t+5))/1 - r_5(t+5))/1 - u_5(t+5))}{S_5}$$

$$\hat{A}_4(t+4) = \frac{(\hat{A}_{5+}(t+5, t+6)) + (((E_4(t+4)/I_4(t+4))/1 - r_4(t+4))/1 - u_4(t+4))}{S_4}$$

$$\hat{A}_3(t+3) = \frac{(\hat{A}_4(t+4)) + (((E_3(t+3)/I_3(t+3))/1 - r_3(t+3))/1 - u_3(t+3))}{S_3}$$

$$\hat{A}_2(t+2) = \frac{(\hat{A}_3(t+3)) + (((E_2(t+2)/I_2(t+2))/1 - r_2(t+2))/1 - u_2(t+2))}{S_2}$$

where $E_i(t)$ = age-specific Hanford Reach natural-origin spawning escapement in year, $i = 2, 3, 4, 5, 6$;
 $I_i(t)$ = interdam conversion rate in year t , calculated separately for jacks and adults; number of Hanford Reach destined fall Chinook salmon counted at McNary Dam as compared to Bonneville Dam;
 S_i = conditional ocean natural survival rate from age i to $i+1$, given not captured in ocean fisheries at age i and does not mature at age i ;
 $r_i(t)$ = in-river fishery exploitation rate at age i in year t ;
 $u_i(t)$ = ocean fishery exploitation rate at age i in year t ; and
 $A_i(t)$ = numbers of fish alive at ocean age i in year t , prior to fishing or maturation.

Table 2.8. Ocean Exploitation Rates of Hanford Reach Fall Chinook Salmon, brood years 1975–2018. Rates in bold indicate Priest Rapids Hatchery CWT groups were used to estimate harvest. Wild Hanford Reach CWT groups were used to estimate all other harvest rates. n/a = data not available.

Brood Year	Age-2	Age-3	Age-4	Age-5+
1975	4.0%	13.1%	35.4%	39.1%
1976	4.0%	13.9%	31.3%	40.5%
1977	5.6%	18.4%	35.1%	23.3%
1978	2.3%	3.9%	15.9%	29.3%
1979	3.4%	11.4%	36.1%	58.8%
1980	4.4%	9.2%	41.6%	46.8%
1981	4.4%	10.2%	32.4%	30.4%
1982	3.1%	10.7%	26.5%	33.3%
1983	4.3%	13.0%	29.8%	36.6%
1984	3.5%	14.2%	25.5%	47.3%
1985	3.2%	4.7%	17.1%	33.6%
1986	2.4%	4.6%	14.4%	30.4%
1987	2.5%	4.4%	10.0%	59.1%
1988	3.6%	4.3%	11.2%	43.6%
1989	2.4%	4.3%	14.0%	33.2%
1990	6.4%	3.6%	9.2%	25.6%
1991	3.6%	11.8%	<0.1%	14.4%
1992	2.4%	1.7%	6.9%	31.1%
1993	2.0%	4.2%	9.5%	27.0%
1994	3.1%	2.0%	15.6%	29.8%
1995	2.1%	1.4%	15.7%	18.6%
1996	0.9%	2.2%	11.2%	6.0%
1997	1.8%	1.3%	4.9%	44.6%

Brood Year	Age-2	Age-3	Age-4	Age-5+
1998	1.4%	1.4%	10.0%	24.9%
1999	1.4%	2.0%	7.5%	33.9%
2000	2.3%	2.9%	19.8%	34.8%
2001	1.8%	2.2%	12.7%	34.6%
2002	1.9%	3.3%	10.1%	50.5%
2003	2.4%	3.8%	24.2%	39.9%
2004	2.9%	3.8%	33.2%	21.8%
2005	3.5%	5.5%	15.6%	45.1%
2006	2.4%	5.0%	22.5%	37.6%
2007	2.0%	2.3%	21.1%	58.2%
2008	2.3%	4.2%	23.7%	36.1%
2009	1.7%	4.0%	17.9%	34.3%
2010	1.7%	1.8%	16.7%	34.9%
2011	1.6%	1.6%	12.2%	32.1%
2012	1.8%	3.1%	19.3%	27.2%
2013	1.3%	2.7%	12.3%	29.3%
2014	1.4%	3.3%	14.8%	<0.1%
2015	2.8%	1.8%	23.4%	34.4%
2016	0.9%	2.0%	13.6%	n/a
2017	0.2%	1.9%	n/a	n/a
2018	<0.1%	n/a	n/a	n/a

Finally, the number of presmolts associated with BY t , $P(t)$ was estimated by dividing the number of fish estimated to have reached ocean age-2 by a CWT-based estimate of survival from presmolt to ocean age-2 (Table 2.9 and Table 2.10):

$$\hat{P}(t) = \frac{\hat{A}_2(t+2)}{\hat{S}_0(t)}$$

where $\hat{S}_0(t)$ is the survival from release to ocean age-2, prior to fishing or maturation, for ad+CWT fish released from brood year t .

Table 2.9. CWT-Based Survival from Tagging (as Presmolts) to Ocean Age-2 for Hanford Reach Fall Chinook Salmon, BY 1975–2016. Survival rates in bold indicate survival rates for Priest Rapids Hatchery CWT groups were used. Wild Hanford Reach CWT groups were used to estimate all other survival rates.

Brood Year	Age-2 Survival Rate	Brood Year	Age-2 Survival Rate	Brood Year	Age-2 Survival Rate
1975	7.8%	1989	1.3%	2003	0.3%
1976	3.3%	1990	1.7%	2004	0.3%
1977	2.5%	1991	0.4%	2005	0.7%
1978	4.0%	1992	1.5%	2006	0.6%
1979	2.4%	1993	1.9%	2007	2.7%
1980	2.0%	1994	0.2%	2008	0.9%
1981	1.9%	1995	0.9%	2009	2.6%

Brood Year	Age-2 Survival Rate	Brood Year	Age-2 Survival Rate	Brood Year	Age-2 Survival Rate
1982	2.6%	1996	0.5%	2010	5.8%
1983	6.1%	1997	0.6%	2011	2.1%
1984	5.9%	1998	2.3%	2012	3.7%
1985	1.7%	1999	4.4%	2013	0.8%
1986	1.6%	2000	0.7%	2014	0.1%
1987	0.4%	2001	1.6%	2015	0.2%
1988	0.5%	2002	0.7%	2016	0.4%

Because of the relatively strong correlation in presmolt-to-age-2 survival of CWT Hanford Reach presmolts and Priest Rapids Hatchery smolts (Figure 2.2), presmolt-to-age-2 survival rates of Priest Rapids Hatchery CWT groups were used for BY in which no wild Hanford Reach presmolts were implanted with CWTs (BY 1975–1985).

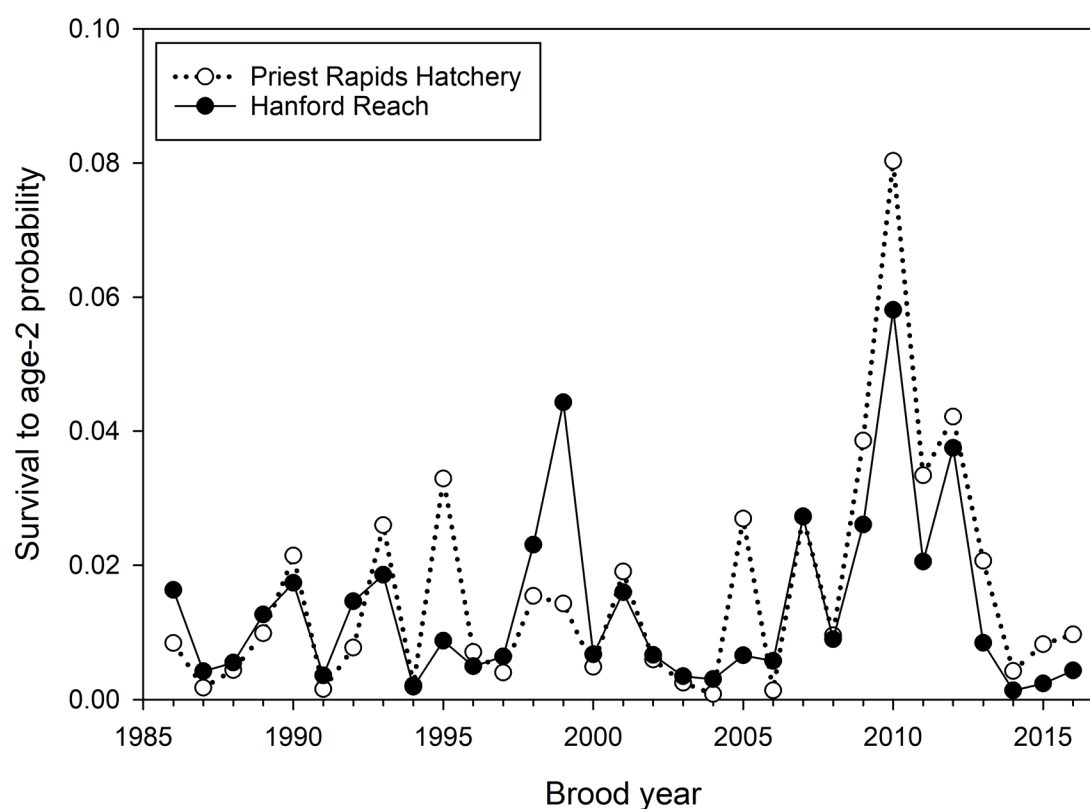


Figure 2.2. Survival to Age-2 of Coded-Wire Tagged (CWT) Wild Hanford Reach Presmolts and Priest Rapids Hatchery Smolts

Table 2.10. Numbers of Ocean Age-2 and Presmolt Recruits Produced each BY, 1975–2016, from the Hanford Reach Fall Chinook Salmon Population as Estimated from Cohort Reconstruction

Brood Year	Recruits to Ocean Age-2	Presmolt Production	Brood Year	Recruits to Ocean Age-2	Presmolt Production
1975	511,951	6,560,964	1996	164,180	33,027,537
1976	201,762	6,142,613	1997	368,163	57,318,482
1977	259,252	10,354,674	1998	515,655	22,352,708
1978	98,266	2,445,590	1999	790,652	17,832,512
1979	320,663	13,569,919	2000	410,425	61,064,450
1980	608,114	30,410,593	2001	572,870	35,820,903
1981	690,449	36,091,616	2002	155,640	23,443,569
1982	870,058	33,515,344	2003	233,556	67,201,994
1983	1,003,893	16,570,630	2004	124,613	40,917,641
1984	2,523,295	42,853,433	2005	503,466	76,375,263
1985	645,216	37,227,422	2006	409,040	71,085,528
1986	527,989	32,337,475	2007	659,528	24,214,315
1987	267,415	63,393,067	2008	264,668	29,439,633
1988	191,686	34,939,465	2009	543,940	20,854,168
1989	341,281	26,926,702	2010	1,433,773	24,673,107
1990	296,134	17,016,600	2011	788,409	38,321,028
1991	96,030	26,713,537	2012	711,445	18,972,922
1992	238,908	16,309,389	2013	347,907	41,291,718
1993	342,067	18,413,450	2014	118,900	87,437,388
1994	77,058	40,401,102	2015	307,681	128,282,174
1995	261,159	29,790,212	2016	197,599	45,534,235

2.1.3 Stock–Recruit Modeling

Assuming lognormal errors (Hilborn and Walters 1992), the log-transformed nonlinear forms of multiple models (Ricker, Beverton-Holt, Smooth Hockey Stick) were fit to the data, and the sum of squared errors (SSE), and mean squared errors (MSE) were used to determine the model that best fit the data.

$$\text{Ricker: } \ln R = \ln \alpha + \ln S - \beta S$$

$$\text{Beverton-Holt: } \ln R = \ln \alpha + \ln S - \ln (1 + \beta S)$$

$$\text{Smooth Hockey Stick: } \ln R = A + \ln \left(1 - \exp \left(-1 \left(\frac{\alpha}{\exp A} \right) S \right) \right)$$

where R = number of presmolts,
 S = egg escapement, and
 A = $\ln R_{\infty}$ = maximum recruitment.

The Ricker, Beverton-Holt, and Smooth Hockey Stick models provided similar fits to the data (Table 2.11). However, the Beverton-Holt model was chosen as the model with which to proceed because it had the lowest SSE and MSE and because the data conformed closely to the assumption of the Beverton-Holt that recruitment approaches an asymptote at high spawning stock abundance (Figure 2.3). Residuals of the fitted model were normally distributed (Shapiro Wilk $W = 0.959$; $P = 0.133$).

Table 2.11. Comparison of Sum of Squared Errors (SSE) and Mean Squared Errors (MSE) from Nonlinear Ricker, Beverton-Holt, and Smooth Hockey Stick Models Fit to Hanford Reach Fall Chinook Salmon Presmolts/Egg Stock–Recruit Data, BY 1975–2016

Model	SSE	MSE
Beverton-Holt	13.6418	0.3410
Smooth Hockey Stick	13.7745	0.3444
Ricker	13.9018	0.3475

The productivity parameter α and maximum presmolt recruitment (R_{max}) were estimated from model parameter estimates (Table 2.12). Estimating the productivity parameter α from the model required the conversion of $\hat{\alpha}$ to α (Hilborn and Walters 1992), which was accomplished by

$$\alpha = \hat{\alpha}e^{\sigma^2/2},$$

with the MSE from the model fit used as an estimate of σ^2 . Maximum recruitment (R_{max}) was estimated as:

$$R_{max} = \frac{\alpha}{\beta}$$

and $1/\beta$ represents the spawning stock size required to produce recruitment equal to half the maximum.

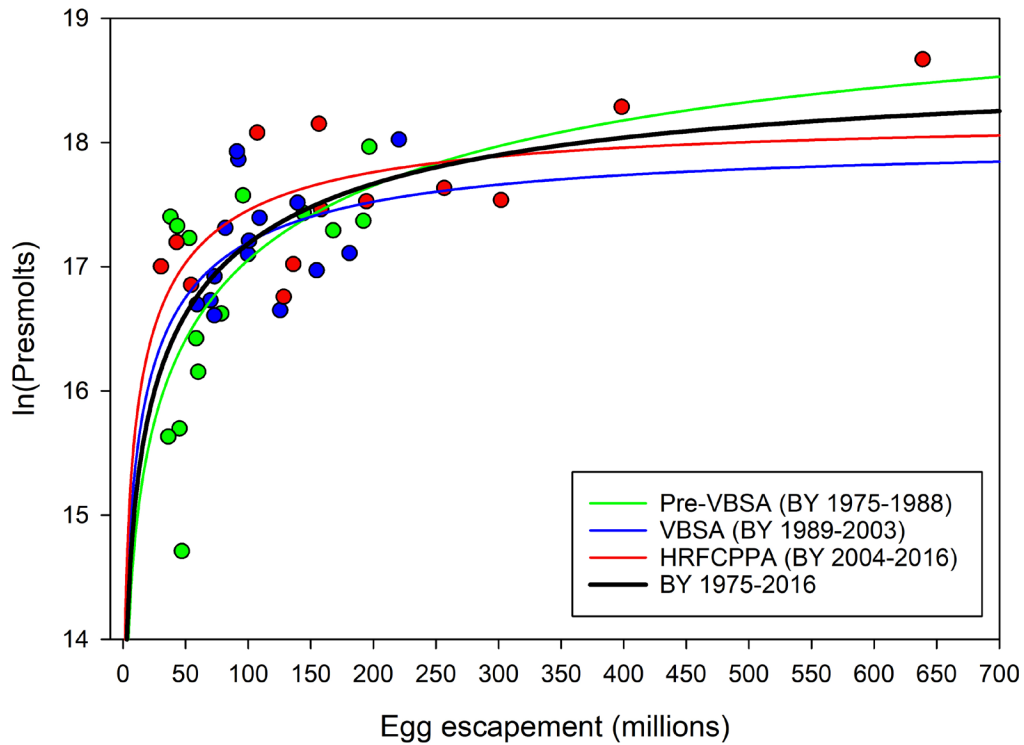


Figure 2.3. Beverton-Holt Model Fit to Hanford Reach Fall Chinook Salmon Presmolts/Egg Stock–Recruit Data for All BY 1975–2016 and for Each Priest Rapids Project Operational Period.

Table 2.12. Results of the Beverton-Holt Model Fit to the Hanford Reach Fall Chinook Salmon Presmolts/Egg Stock–Recruit Data, BY 1975–2016.

Parameter	Coefficient	Standard Error
$\hat{\alpha}$	0.380	0.078
β	3.061×10^{-9}	2.180×10^{-9}

2.1.4 Effect of Dam Operations on Productivity

Two approaches were used to evaluate whether freshwater productivity of the Hanford Reach fall Chinook salmon population differed among the pre-VBSA (BY 1975–1988), VBSA (BY 1989–2003), and HRFCPP (BY 2004–2016) periods. Because the Beverton-Holt model provided the best fit to the egg-to-presmolt survival data, it was fit to each period separately and productivity (α) parameter estimates were cursorily compared between periods. The second approach, which took advantage of the linear form of the Ricker model as in past analyses of Hanford Reach fall Chinook salmon productivity (Harnish et al. 2012, 2014a, 2017), tested for nonstationary behavior in the recruitment function. A temporal change in density-independent mortality, such as that imposed by dewatering of redds or stranding and entrapment of juveniles, would be expected to cause nonstationary behavior in the α parameter of the Ricker regression model (Schaller et al. 1999; Schaller and Petrosky 2007). Nonstationarity in the α parameter was accounted for using analysis of covariance (ANCOVA; JMP Version 15.2.1; SAS Institute, Cary, North Carolina) to examine differences in the intercepts of the relationship of $\ln(R/S)$ versus S in the linear form of the Ricker equation

$$\ln\left(\frac{R_{ij}}{S_{ij}}\right) = \ln(\tau_i + \alpha) - \beta(S_{ij} - \bar{S}_{..}) + \varepsilon_{ij}$$

where

- R_{ij} = the number of presmolts produced from each brood year,
- τ_i = the period effect,
- α = the overall intercept,
- β = the overall slope,
- S_{ij} = the egg escapement for each brood year,
- $\bar{S}$ = the average egg escapement for all brood years during both periods,
- ε_{ij} = the normally distributed residual,
- i = the period, and
- j = the brood year.

First, the homogeneity of slopes was tested for significant interaction between the treatment (period) and the covariate (eggs) using a Student's t -test. An ANCOVA was then run to estimate the period effect on $\ln(R/S)$, taking into account spawning level. The measure of productivity by period was estimated as $\tau_i + \alpha$ from the ANCOVA results, which is equivalent to the Ricker α parameter by period (assuming a common slope for all periods; Schaller and Petrosky 2007). Using the Ricker model to test for a difference in α was deemed appropriate due to the similar model fit of the Ricker and Beverton-Holt models to the presmolt/egg data (Table 2.11).

2.2 Adult Production

2.2.1 Stock Estimation

The number of adult (age-3+) spawners, as estimated by WDFW (Hoffarth 2010; Table 2.1) was used as the measure of stock in the stock–recruit analysis conducted to estimate the spawning abundance required to achieve maximum sustainable yield (S_{MSY}) for fall Chinook salmon in the Hanford Reach.

2.2.2 Recruit Estimation

The number of age-3 adult equivalents (hypothetical adult returns to the spawning grounds in the absence of fisheries) produced from each BY 1975–2016 was used as the measure of recruitment for the analysis to estimate S_{MSY} for the Hanford Reach fall Chinook salmon population. Age-3 adult equivalents were estimated using the cohort reconstruction methods of the PSC CTC (CTC 1988, 2009). The general approach of the cohort reconstruction method begins with escapement estimates (minus hatchery spawners), working backward in time to account for interdam loss and in-river and ocean harvest to estimate the number of age-3 ocean recruits produced. Age-specific natural-origin adult escapement estimates for BY 1975–2021 were divided by adult-specific dam conversion rates (Table 2.6) to account for interdam loss. Next, the age-specific run to the mouth of the Columbia River was constructed by dividing the interdam loss-adjusted estimates by the inriver harvest rates obtained from CWT data (Table 2.7). Priest Rapids Hatchery CWT groups were used to estimate inriver harvest rates for years when no wild Hanford Reach CWT groups existed or when recoveries of wild Hanford Reach CWT fish were low.

Next, ocean recruits were calculated for each age using the cohort analysis algorithm of the PSC CTC and ocean harvest rates obtained from CWT data (Table 2.8). Priest Rapids Hatchery CWT groups were used to estimate ocean exploitation rates for years when no wild Hanford Reach CWT groups existed. For an individual CWT group, the approach began with the oldest age and worked back through successive ages until ocean age-3 was reached. The number of age-3 adult equivalents was calculated by adding the age-3, age-4, and age-5+ estimates (Table 2.13). The following equations outline this approach:

$$\hat{A}_{5+}(t+5, t+6) = \frac{(((E_5(t+5) + E_6(t+6))/I_5(t+5))/1 - r_5(t+5))}{(1 - u_5(t+5))}$$

$$\hat{A}_4(t+4) = \frac{(((E_4(t+4))/I_4(t+4))/1 - r_4(t+4))}{(1 - u_4(t+4))}$$

$$\hat{A}_3(t+3) = \frac{(((E_3(t+3))/I_3(t+3))/1 - r_3(t+3))}{(1 - u_3(t+3))}$$

$$AEQ = \hat{A}_3 + \hat{A}_4 + \hat{A}_{5+}$$

where

- $E_i(t)$ = age-specific natural-origin Hanford Reach spawning escapement in year t , $i = 3, 4, 5, 6$;
- $I_i(t)$ = interdam conversion rate in year t , calculated separately for jacks and adults; number of Hanford Reach destined fall Chinook salmon counted at McNary Dam as compared to Bonneville Dam;
- $u_i(t)$ = ocean fishery exploitation rate at age i in year t ;
- $r_i(t)$ = in-river fishery exploitation rate at age i in year t ;

$A_i(t)$ = numbers of fish alive at ocean age i in year t , prior to fishery exploitation or maturation; and
 AEQ = numbers of age-3 adult equivalents (hypothetical adult returns to the spawning grounds in the absence of fisheries).

Table 2.13. Numbers of Hanford Reach Fall Chinook Salmon Ocean Age-3 Adult Equivalents (AEQ) and the Number of AEQ Recruits per Spawner (R/S) Produced from each BY 1975–2016 as Estimated from Cohort Reconstruction

Brood Year	AEQ	R/S	Brood Year	AEQ	R/S	Brood Year	AEQ	R/S
1975	139,822	6.29	1989	94,838	1.44	2004	37,983	0.48
1976	53,903	2.55	1990	76,396	1.90	2005	156,145	2.43
1977	74,666	2.37	1991	25,972	0.81	2006	126,501	2.69
1978	29,654	1.44	1992	69,985	2.38	2007	206,978	14.90
1979	98,345	4.17	1993	103,785	3.39	2008	81,621	3.49
1980	183,354	8.39	1994	19,105	0.39	2009	167,248	6.35
1981	212,510	14.06	1995	76,797	2.00	2010	471,844	5.87
1982	254,960	12.41	1996	49,050	1.31	2011	248,902	3.79
1983	284,365	7.89	1997	117,141	3.44	2012	219,541	4.24
1984	743,761	17.72	1998	157,746	5.36	2013	97,630	0.62
1985	185,419	2.82	1999	239,678	8.87	2014	37,152	0.24
1986	110,256	1.52	2000	123,616	3.43	2015	101,401	0.43
1987	72,042	0.81	2001	181,137	3.52	2016	65,241	0.59
1988	49,653	0.67	2002	47,505	0.69			
			2003	69,724	0.78			
Pre-VBSA Mean	178,051	5.94	VBSA Mean	96,832	2.65	HRFCPP Mean	155,245	3.55

2.2.3 Stock–Recruit Modeling

The Ricker model was chosen as the model with which to examine the relationship between adult spawners and age-3 adult equivalent recruits because of (1) its flexibility in fitting a wide variety of datasets, (2) its ready calculation of management or policy parameters from fitted Ricker model parameters (Ricker 1975; Langness and Reidinger 2003), and (3) it being used previously for stock–recruit analyses of the URB fall Chinook salmon population (Bernard and Clark 1999; Peters et al. 1999; Langness and Reidinger 2003). The linearized Ricker model was fit to the data (Figure 2.4).

Previous stock–recruit analyses (Bernard and Clark 1999; Langness and Reidinger 2003) have shown that some Columbia River fall Chinook salmon populations exhibit significant levels of autocorrelation between residuals when fit to the linearized Ricker model, indicating a first-order autoregressive process is at work (Langness and Reidinger 2003). This implies non-independence in the residuals and possibly biased estimates (Langness and Reidinger 2003). Residuals from the linearized Ricker model were tested for significant levels of autocorrelation. Autocorrelation coefficients were calculated as

$$r(k) = \left[\frac{T-1}{T-1-k} \right] \frac{\sum_{t=1}^{T-k} \hat{w}_t \hat{w}_{t+k}}{\sum_{t=1}^T \hat{w}_t^2}$$

where T = 42 (the number of brood years),
 k = the time lag,
 t = year, and

$\hat{W}$ = residuals.

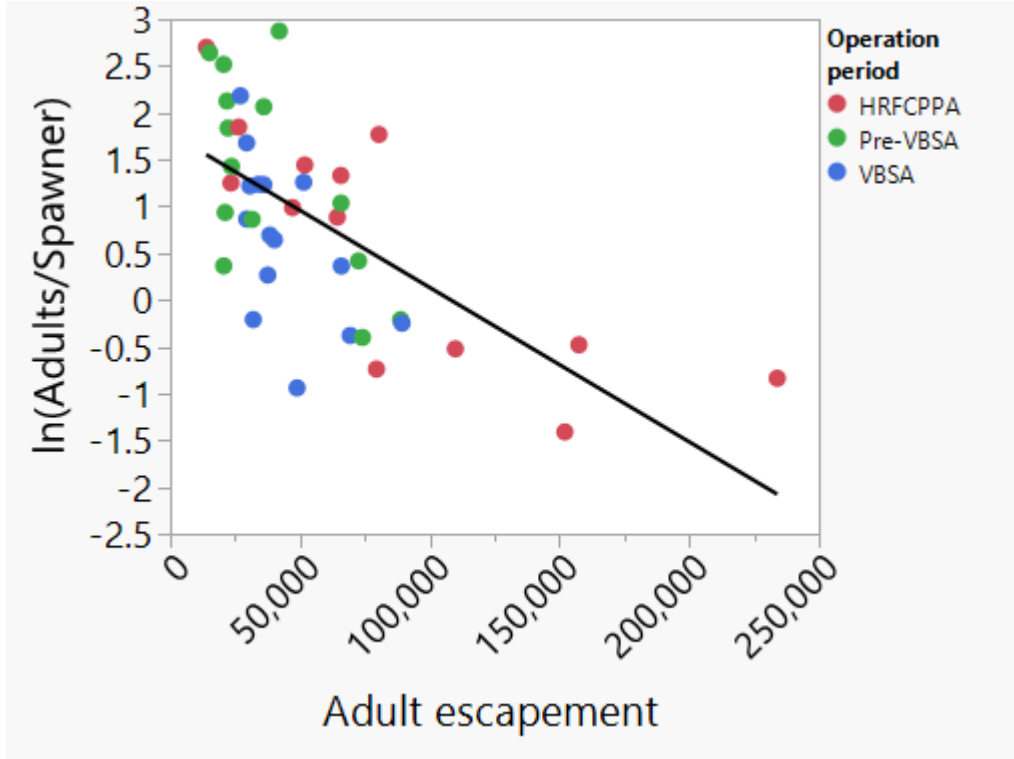


Figure 2.4. Linearized Ricker Model Fit to Hanford Reach Fall Chinook Salmon Adults/Spawner Stock–Recruit Data, BY 1975–2016. Estimated numbers of adult spawners were used as the measure of stock, and estimated numbers of age-3 adult equivalents were used as the measure of recruits in this analysis. Pre-Vernita Bar Settlement Agreement (VBSA) brood years (1975–1988) are shown in green; brood years of the VBSA (1989–2003) are shown in blue, and brood years of the Hanford Reach Fall Chinook Protection Program Agreement (HRFCPPA; 2004–2016) are shown in red.

Significant autocorrelation was identified as correlation coefficients that were less than or greater than the 95% confidence bounds, calculated as $\pm 1.96/\sqrt{T}$ (Figure 2.5).

To ameliorate the autocorrelation problem, an additional parameter, ϕ , was introduced to the model. The linear form of the Ricker model with the additional parameter follows the AR1 model of Noakes et al. (1987):

$$\ln\left(\frac{R_{BY}}{S_{BY}}\right) = (1 - \phi)\ln \alpha + \phi \ln\left(\frac{R_{BY-1}}{S_{BY-1}}\right) - \left(\frac{1}{b}\right)S_{BY} + \left(\frac{\phi}{b}\right)S_{BY-1} + \gamma_{BY}$$

where R_{BY} = the estimated number of age-3 adult equivalents from a brood year,
 S_{BY} = the estimated number of adult spawners from a brood year,
 R_{BY-1} = the estimated number of age-3 adult equivalents from the previous brood year, and
 S_{BY-1} = the estimated number of adult spawners from the previous brood year.

In this equation, the parameter ϕ represents the effect of the serial correlation at a lag of one brood year such that $-1 < \phi < 1$. This equation is of a linear form such that $Y = c_0 + c_1X_1 + c_2X_2 + c_3X_3$ in which $c_0 = (1 - \phi) \ln \alpha$, $c_1 = \phi$, $c_2 = 1/b$, $c_3 = \phi/b$, $X_1 = R_{BY-1}/S_{BY-1}$, $X_2 = S_{BY}$, and $X_3 = S_{BY-1}$.

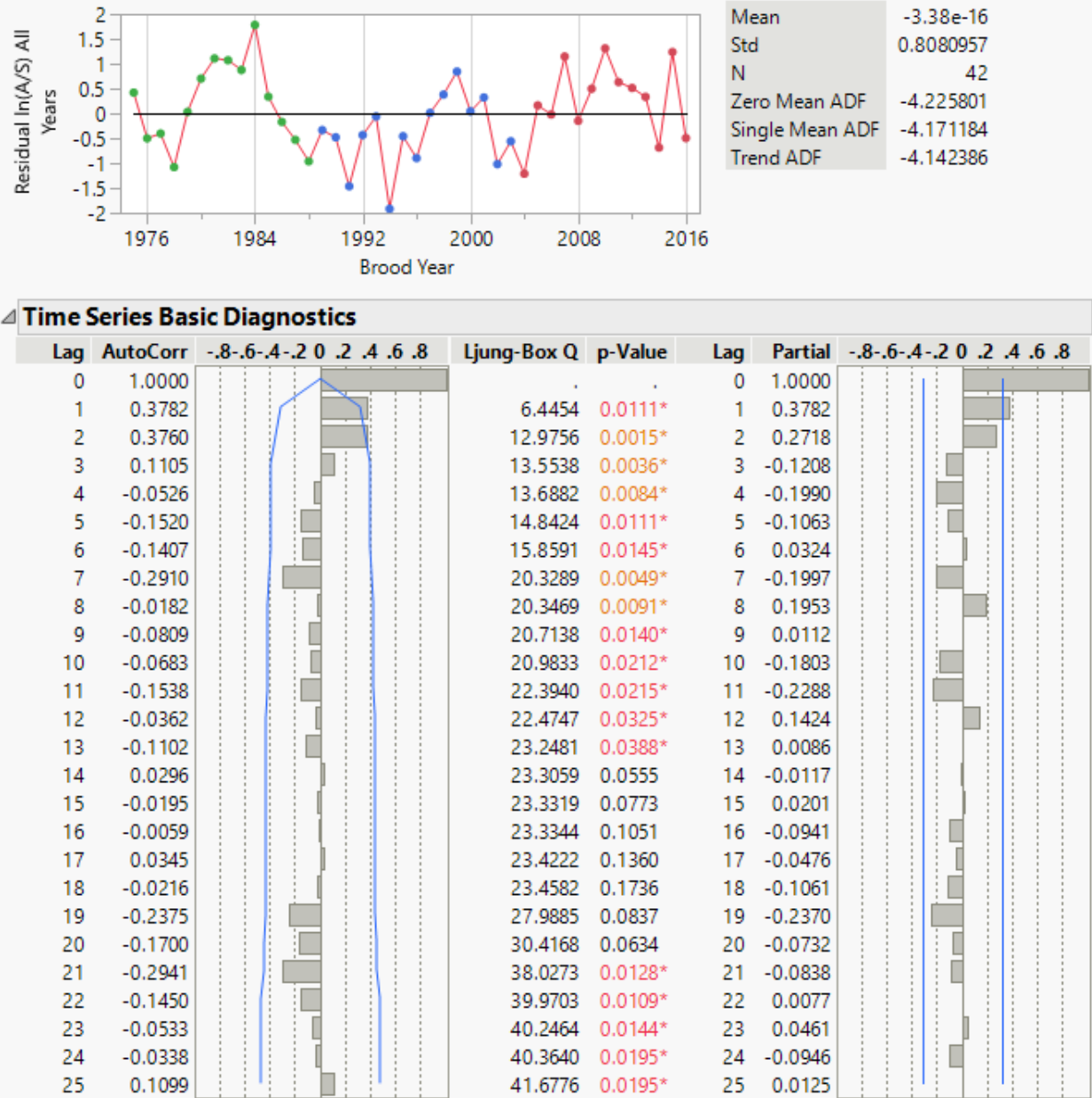


Figure 2.5. Time Series Diagnostics Displaying the Autoregressive Process of Lag One Brood Year Present when the Linearized Ricker Model was Fit to the Hanford Reach Fall Chinook Salmon Age-3 Adult Equivalents/Adult Spawners Data

Estimating the productivity parameter α from the reduced model (in the presence of an autoregressive lag of 1 year) required the conversion of $\hat{\ln} \alpha$ to $\ln \hat{\alpha}$ (Hilborn and Walters 1992), which was accomplished by

$$\ln \hat{\alpha} = \hat{\ln} \alpha + \frac{\hat{\sigma}_Y^2}{2(1-\hat{\phi}^2)},$$

with the MSE from the fit of the reduced model used as an estimate of $\hat{\sigma}_Y^2$. The adult spawning escapement required to achieve maximum sustainable yield ($\hat{S}_{MSY}$) was estimated using Hilborn's approximation (Hilborn and Walters 1992):

$$\hat{S}_{MSY} = (\ln \hat{\alpha})\beta (0.5 - 0.07 \ln \hat{\alpha}).$$

Maximum recruitment (R_{max}) and S_{max} were estimated as

$$R_{max} = \frac{0.3679\hat{\alpha}}{\beta}$$

$$S_{max} = 1/\beta$$

2.3 Factors Affecting Presmolt-to-Age-2 Survival

Factors outside of the Hanford Reach that may be limiting adult production were evaluated by regressing annual log-transformed CWT-based presmolt-to-age-2 survival estimates against inriver and ocean covariates using simple and multivariable regression. Inriver covariates included measures of Columbia River discharge and temperature and an index of spill at Columbia River dams located downstream of the Hanford Reach (i.e., McNary, John Day, The Dalles, and Bonneville dams). These inriver variables were selected due to their influence on smolt migration rate through the hydrosystem, the metabolic rate of piscivorous fishes, and the probability of turbine passage.

Measures of inriver discharge, temperature, and spill were calculated by first determining average daily discharge, temperature, and spill proportions (from hourly means) at each dam during the period of subyearling Chinook salmon smolt migration (June 1–August 15). Daily averages were then weighted by the daily proportion of subyearling Chinook salmon smolts sampled in the juvenile fish facilities of the dams to obtain weighted means for each dam. In the case of The Dalles Dam, which is not equipped with a juvenile fish facility, daily smolt passage proportions from John Day Dam were used. Weighted mean discharge and temperature values were averaged across the four dams and weighted mean spill proportions were summed across the four dams to produce one annual value for discharge, temperature, and spill. Hourly temperature, discharge, and spill data were obtained from the U.S. Army Corps of Engineers data query 2.0 website (<https://www.nwd-wc.usace.army.mil/dd/common/dataquery/www/>) and daily smolt passage data were obtained from the Columbia River Data Access in Real Time (DART) smolt index (https://www.cbr.washington.edu/dart/query/smolt_daily).

Ocean covariates included local and basin-scale ocean condition indicators thought to be biologically important for their influence on the source waters that feed the northern California Current, which affect zooplankton and forage fish communities. Ocean condition indicator data associated with the year of ocean entry were obtained from <https://www.fisheries.noaa.gov/west-coast/science-data/ocean-ecosystem-indicators-pacific-salmon-marine-survival-northern> and included large-scale ocean and atmospheric indicators (i.e., Pacific Decadal Oscillation [PDO] and Oceanic Niño Index [ONI]), local and regional physical indicators (e.g., sea surface temperature [SST], coastal upwelling), and biological indicators (e.g., copepod and ichthyoplankton biodiversity and community structure). Ocean covariate data were only available dating back to smolt migration year 1998; therefore, evaluations of inriver and ocean conditions and density dependent effects on presmolt-to-age-2 survival were limited to BY 1997–2016.

Correlations between each variable and estimates of presmolt-to-age-2 survival were first assessed using simple linear regression. Prior to conducting multivariable analyses, a correlation matrix was calculated to examine multicollinearity between predictor variables. Simple linear regression and correlation matrix results were used to define the final list of candidate variables for use in the multivariable model building procedure. The candidate variables that showed the greatest correlation with presmolt-to-age-2 survival estimates in simple linear regression models were evaluated for multicollinearity with other predictor variables. Predictor variables with a Pearson product-moment coefficient $|r| > 0.7$ were identified as

displaying a very large degree of linear association with one another (i.e., multicollinearity). In such cases, the predictor that explained less of the variability in presmolt-to-age-2 survival was removed from the list of candidate variables in the multivariable model selection procedure, which was performed using forward stepwise regression. All main effects were considered as candidate terms for inclusion in the multivariable model. Interaction terms were not considered as candidates due to the relatively small number of BY (20) for which ocean condition data were available. The best-fitting model was identified as the model that produced the smallest sample size-corrected Akaike's Information Criterion (AICc) value. Coefficients of partial determination were calculated for each variable included in the best-fitting model to express the relative importance of individual contributions of covariates to the model.

3.0 Results

3.1 Juvenile Production

The mean estimated egg-to-presmolt survival was generally similar between Priest Rapids Dam operational periods. Estimated egg-to-presmolt survival averaged 0.334 during the pre-VBSA period, 0.314 during the VBSA, and 0.349 during the HRFCPP (Table 3.1). However, the mean deviation from the Beverton-Holt model fit to the entire 42-year period increased with each successive constraint placed on flow fluctuations in the Hanford Reach. The mean of the pre-VBSA residuals was -0.152 , compared to -0.002 for the VBSA, and 0.165 for the HRFCPP. Additionally, 9 of 14 (64%) pre-VBSA BY had below-average egg-to-presmolt survival as indicated by negative residuals from the Beverton-Holt model fit, whereas 7 of 15 (47%) VBSA BY and 7 of 13 (54%) HRFCPP BY had above-average survival (Table 3.1).

Table 3.1. Presmolt/Egg Estimates and Residuals ε_y from the Non-Linear Beverton-Holt Model Fit to the Hanford Reach Fall Chinook Salmon Presmolts/Egg Stock–Recruit Data, BY 1975–2016

Brood Year	Presmolts/Egg	ε_y	Brood Year	Presmolts/Egg	ε_y	Brood Year	Presmolts/Egg	ε_y
1975	0.145	-0.831	1989	0.149	-0.497	2004	0.210	-0.124
1976	0.169	-0.703	1990	0.136	-0.706	2005	0.488	0.642
1977	0.172	-0.622	1991	0.268	-0.084	2006	0.663	0.841
1978	0.052	-1.853	1992	0.224	-0.329	2007	0.797	0.829
1979	0.232	-0.328	1993	0.263	-0.173	2008	0.686	0.714
1980	0.574	0.562	1994	0.290	0.084	2009	0.384	0.163
1981	0.951	1.027	1995	0.296	0.018	2010	0.181	-0.392
1982	0.773	0.834	1996	0.404	0.284	2011	0.242	-0.056
1983	0.211	-0.374	1997	0.622	0.741	2012	0.148	-0.613
1984	0.447	0.419	1998	0.305	-0.017	2013	0.137	-0.368
1985	0.258	-0.021	1999	0.302	-0.065	2014	0.219	0.248
1986	0.193	-0.265	2000	0.671	0.814	2015	0.201	0.446
1987	0.322	0.306	2001	0.328	0.142	2016	0.177	-0.182
1988	0.182	-0.274	2002	0.152	-0.532			
			2003	0.305	0.295			
Pre-VBSA Mean	0.334	-0.152	VBSA Mean	0.314	-0.002	HRFCPP Mean	0.349	0.165

Fitting the Beverton-Holt model to the egg-to-presmolt data from each Priest Rapids Dam operational period indicated productivity increased with each subsequent flow fluctuation constraint. The productivity parameter α from the Beverton-Holt model fit was estimated to be 0.39 presmolts/egg for the pre-VBSA period, 0.58 presmolts/egg during the VBSA period, and 0.81 presmolts/egg during the HRFCPP period.

Fitting the linearized Ricker model with a period effect to the egg-to-presmolt data also indicated productivity increased with successive flow fluctuation constraints (Figure 3.1). The resulting productivity (Ricker α) for the pre-VBSA period $e^{\ln(\tau_1 + \alpha)} = 0.310$ presmolts/egg compared to $e^{\ln(\tau_2 + \alpha)} = 0.362$ presmolts/egg for the VBSA period, and $e^{\ln(\tau_3 + \alpha)} = 0.433$ for the HRFCPPA. However, the effect of operational period was not statistically significant ($P = 0.414$; Table 3.2) using this approach.

Table 3.2. Summary of Presmolt/Egg ANCOVA “Period Effect” Test Results

Intercept $\ln(\tau_1 + \alpha)$ Pre-VBSA	Intercept $\ln(\tau_2 + \alpha)$ VBSA	Intercept $\ln(\tau_3 + \alpha)$ HRFCPP	Intercept H_0 : $\tau_1 = \tau_2 = \tau_3, p$	Slope β	Slope H_0 : $\beta \leq 0, p$	$\bar{R}^2$	Homogeneity of Slope, p
-1.173	-1.015	-0.838	0.414	-2.009×10^{-9}	0.035	0.049	0.820

Due to the lack of density dependence in the pre-VBSA presmolt/egg data, which was characterized by high variability in egg-to-presmolt survival at low escapements, the estimate of maximum presmolt recruitment was extremely high ($R_{max} = 147.3$ million presmolts) for the entire 42-year period (BY 1975–2016). The spawning stock size required to produce $\frac{1}{2}$ of R_{max} was estimated to be 326.7 million eggs, which is equivalent to about 130,000 adult spawners. Fitting the Beverton-Holt model to a large series of sets of years wherein each successive set left out the earliest year of its predecessor revealed instability in R_{max} when the earliest years of the data set were included (Figure 3.2). Maximum presmolt recruitment stabilized around BY 1983, which corresponds well with enactment of the interim VBSA in 1984 (Figure 3.2). Since enactment of the interim VBSA, R_{max} was estimated to be 87.8 million presmolts and the spawning stock size required to produce $\frac{1}{2}$ of R_{max} was estimated to be 143.6 million eggs, which is equivalent to about 59,800 adult spawners. The productivity parameter (α) of the Beverton-Holt model fit to the BY 1984–2016 dataset was estimated to be 0.611 presmolts/egg, which is higher than the α parameter estimate obtained for the entire BY 1975–2016 dataset, which was 0.451 presmolts/egg.

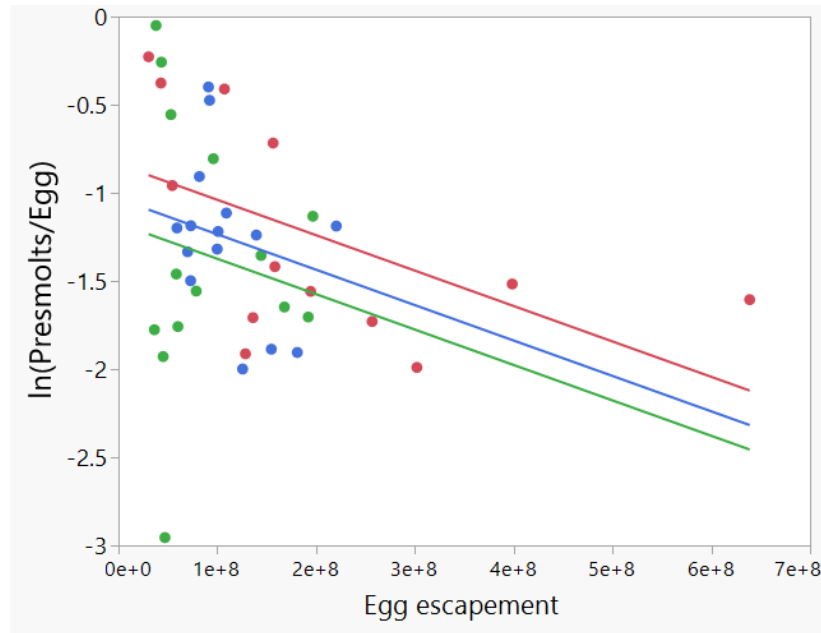


Figure 3.1. Linearized Ricker Model Fit to the Hanford Reach Fall Chinook Salmon Presmolts/Egg Stock–Recruit Data for the Pre-VBSA (BY 1975–1988; shown in green), VBSA (BY 1989–2003; shown in blue), and HRFCPP (BY 2004–2016; shown in red) Periods

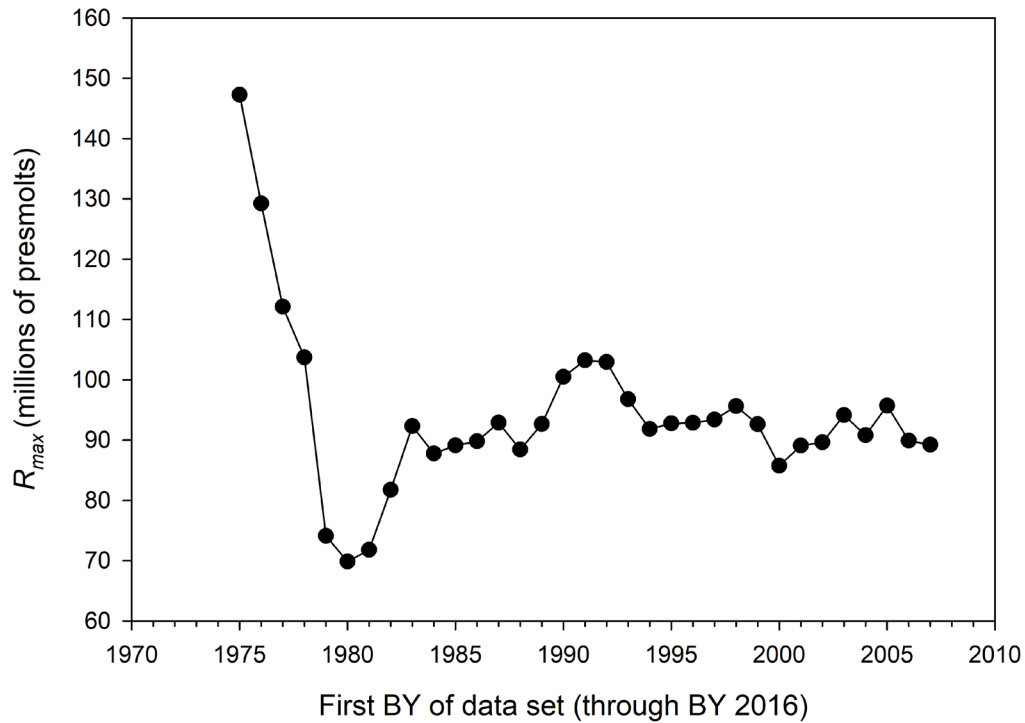


Figure 3.2. Maximum Hanford Reach Fall Chinook Salmon Presmolt Recruitment (R_{max}) Values Estimated for a Large Series of Sets of Years wherein Each Successive Set Leaves Out the Earliest Year of its Predecessor

3.2 Adult Production

When fit to the adult/spawner data, the linearized Ricker AR1 model had an R^2 of 0.55, an adjusted coefficient of determination ($\bar{R}^2$) of 0.51, and an MSE of 0.58. The Durbin–Watson statistic for this fit indicated no remaining serial correlation ($P = 0.49$). However, the value for c3 (S_{BY-1}) was not significantly different from zero ($P = 0.92$; Table 3.3). Therefore, c3 was dropped from the model, and the data were refit with a simpler model that incorporated only the intercept, c1 (R_{BY-1}/S_{BY-1}), and c2 (S_{BY}):

$$\ln\left(\frac{R_{BY}}{S_{BY}}\right) = (1 - \phi)\ln \alpha + \phi \ln\left(\frac{R_{BY-1}}{S_{BY-1}}\right) - \left(\frac{1}{b}\right)S_{BY} + \gamma_{BY}$$

Table 3.3. Results of the Linearized Ricker AR1 Full Model Fit to the Hanford Reach Fall Chinook Salmon Age-3 Adult Equivalents/Spawner Stock–Recruit Data, BY 1975–2016

Term	Parameter	Variable	Coefficient	Standard Error	t	P -value
c0	$(1 - \phi) \ln \alpha$	intercept	1.166	0.331	3.52	0.001
c1	ϕ	$\ln(R_{BY-1}/S_{BY-1})$	0.387	0.147	2.64	0.012
c2	$1/b$	S_{BY}	-1.15×10^{-5}	3.98×10^{-6}	-2.88	0.007
c3	ϕ/b	S_{BY-1}	-4.75×10^{-7}	4.66×10^{-6}	-0.100	0.919

The reduced model had an R^2 of 0.55, an adjusted coefficient of determination ($\bar{R}^2$) of 0.52, and an MSE of 0.57. The intercept, c1, and c2 of the reduced model were significantly different from zero ($P \leq 0.003$; Table 3.4). Applying Hilborn’s approximation (Hilborn and Walters 1992) to the linearized Ricker AR1 reduced model fit to BY 1975–2016 produced an $\hat{S}_{MSY}$ estimate of 65,531 adult spawners and $\hat{\alpha} = 9.32$ for the Hanford Reach URB fall Chinook salmon population. Maximum adult recruitment (R_{max}) and S_{max} were estimated to be 292,918 age-3 adult equivalents and 85,397 adult spawners, respectively. However, BY 2015 was identified as a clear outlier (Cook’s $D = 1.415$) with high leverage on model parameters. Therefore, the reduced AR1 model was rerun with BY 2015 omitted from the data set. The resulting model had an improved fit with an R^2 of 0.59, an $\bar{R}^2$ of 0.56, and an MSE of 0.50. Again, the intercept, c1, and c2 parameters of the model were significantly different from zero ($P \leq 0.001$; Table 3.5). Applying Hilborn’s approximation (Hilborn and Walters 1992) to the linearized Ricker AR1 reduced model fit to BY 1975–2016 with 2015 omitted produced an $\hat{S}_{MSY}$ estimate of 48,925 adult spawners and $\hat{\alpha} = 13.84$ for the Hanford Reach URB fall Chinook salmon population. Maximum adult recruitment (R_{max}) and S_{max} were estimated to be 299,936 age-3 adult equivalents and 58,912 adult spawners, respectively.

Table 3.4. Results of the Linearized Ricker AR1 Reduced Model Fit to the Hanford Reach Fall Chinook Salmon Adults/Spawner Stock–Recruit Data, BY 1975–2016

	Parameter	Variable	Coefficient	Standard error	<i>t</i>	<i>P</i> -value
c0	$(1 - \phi) \hat{\ln} \alpha$	intercept	1.147	0.274	4.18	<0.001
c1	ϕ	$\ln(R_{BY-I}/S_{BY-I})$	0.395	0.125	3.15	0.003
c2	$1/b$	S_{BY}	-1.171×10^{-5}	3.091×10^{-6}	-3.79	<0.001

Table 3.5. Results of the Linearized Ricker AR1 Reduced Model Fit to the Hanford Reach Fall Chinook Salmon Adults/Spawner Stock–Recruit Data, BY 1975–2014, 2016 (BY 2015 omitted)

	Parameter	Variable	Coefficient	Standard error	<i>t</i>	<i>P</i> -value
c0	$(1 - \phi) \hat{\ln} \alpha$	intercept	1.371	0.273	5.02	<0.001
c1	ϕ	$\ln(R_{BY-I}/S_{BY-I})$	0.411	0.118	3.49	0.001
c2	$1/b$	S_{BY}	-1.697×10^{-5}	3.594×10^{-6}	-4.72	<0.001

3.3 Factors Affecting Presmolt-to-Age-2 Survival

The estimated number of adult equivalents (AEQs) produced per adult spawner was significantly positively correlated with estimates of egg-to-presmolt survival ($P < 0.001$; Figure 3.3). However, egg-to-presmolt survival explained only 35% of the variability in AEQ per spawner production. Therefore, it is apparent that other factors, outside of the Hanford Reach, are affecting adult production. Log-transformed presmolt-to-age-2 survival estimates from BY 1997–2016 were significantly correlated with many ocean condition indicators as well as Columbia River temperature and discharge (Table 3.6). The greatest amount of variability in simple linear regression models of presmolt-to-age-2 survival was explained by the May–September PDO index ($R^2 = 0.718$) (Figure 3.4).

The correlation matrix revealed a high degree of multicollinearity between the May–September PDO index and many of the other ocean condition indicators as well as Columbia River temperature and discharge as evidenced by Pearson product-moment correlations ($|r| > 0.5$) (Table 3.6). Variables that

displayed a very high degree of multicollinearity ($|r| > 0.7$) with the May–September PDO index were removed from the list of candidate variables in the multivariable model selection procedure. Therefore, the final list of candidate variables included the main effects of the May–September PDO index, the January–March nearshore/offshore ichthyoplankton community index, the May–September northern and southern copepod biomass anomalies, the biological ocean transition day of year, the November–March and May–September upper 20 m ocean temperature (measured 9 km off shore from Newport, Oregon), sea surface temperature (measured by National Data Buoy Center [NDBC] buoys), the hydrographic survey-based physical spring transition day of year, May–September deep water (50 m) ocean temperature and salinity (measured 9 km off shore from Newport, Oregon), April–May ocean upwelling anomaly, the upwelling index-based length of the ocean upwelling season, upwelling index-based physical spring transition day of year, mean weighted Columbia River discharge and temperature, and the sum of weighted Columbia River dam spill proportions.

Performing forward stepwise regression with this list of candidate variables resulted in a best-fitting model that included the May–September PDO index, May–September northern copepod biomass anomaly, and mean weighted Columbia River temperature (Table 3.7). The model indicated presmolt-to-age-2 survival was negatively correlated with the PDO index and Columbia River temperature and positively correlated with the northern copepod biomass anomaly. The model explained a high degree of variability in presmolt-to-age-2 survival, having an R^2 of 0.84 and an adjusted $\bar{R}^2$ of 0.81. Coefficients of partial determination ranged from 0.267 to 0.319, indicating nearly equal contributions of the three covariates to the overall model.

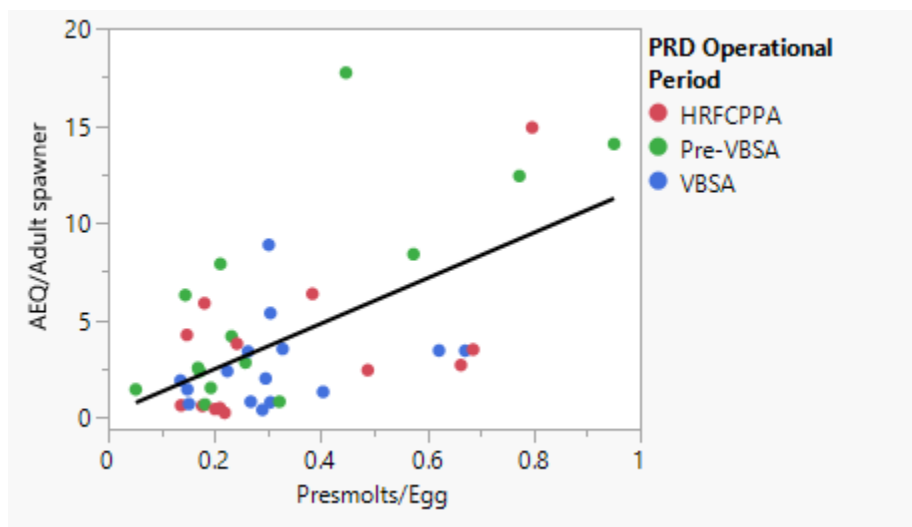


Figure 3.3. Simple Linear Regression Displaying the Relationship between Egg-to-Presmolt Survival (Presmolts/Egg) and the Estimated Number of Age-3 Adult Equivalents (AEQ) Produced per Adult Spawner (AEQ/Adult Spawner) for the Hanford Reach Upriver Bright Fall Chinook Salmon Population, Brood Years 1975–2016. Data points are labeled by Priest Rapids Dam (PRD) operational period. VBSA = Vernita Bar Settlement Agreement, HRF CPPA = Hanford Reach Fall Chinook Protection Program Agreement.

Table 3.6. Intercept (β_0), Slope (β_1), P -value, and Coefficient of Determination (R^2) Results from Simple Linear Regression Models of Log-Transformed Presmolt-to-Age-2 Survival Estimates Fit Versus Ocean Condition Indicator and Inriver Variables. Significant relationships ($P < 0.05$) are indicated by an asterisk (*). Linear associations between PDO (May-Sep) and all other variables are displayed as Pearson product-moment correlation coefficients (r). Pearson's correlation values > 0.7 indicate variables have a very large degree of linear association with PDO (May-Sep) and are highlighted in bold.

Variable	β_0	β_1	$\text{Pr} > t $	R^2	r
PDO (May-Sep)	-4.728	-0.190	<0.001*	0.718	1.000
Copepod richness anomaly (May-Sep)	-4.373	-0.251	<0.001*	0.626	0.712
PDO (Dec-Mar)	-4.470	-0.189	<0.001*	0.579	0.775
Nearshore/offshore ichthyoplankton community index (Jan-Mar)	-4.750	-0.033	<0.001*	0.575	0.686
Copepod community index (May-Sep)	-5.108	-1.292	<0.001*	0.557	0.727
Mean weighted Columbia River temperature	9.267	-0.758	<0.001*	0.525	0.656
S. copepod biomass anomaly (May-Sep)	-4.499	-2.548	<0.001*	0.505	0.687
ONI (Jan-Jun)	-4.676	-1.101	<0.001*	0.484	0.705
N. copepod biomass anomaly (May-Sep)	-4.559	1.727	<0.001*	0.478	-0.555
Biological ocean transition (DOY)	-3.391	-0.008	<0.001*	0.473	0.666
Nearshore ichthyoplankton biomass (Jan-Mar)	-5.888	1.311	<0.001*	0.464	-0.752
Upper 20 m ocean temperature (Nov-Mar)	3.339	-0.752	0.002*	0.431	0.615
Mean weighted Columbia River discharge	-7.151	0.011	0.002*	0.415	-0.551
SST NDBC buoys (May-Sep)	9.592	-1.058	0.019*	0.268	0.668
Physical spring transition hydrographic based (DOY)	-2.151	-0.017	0.038*	0.218	0.479
Upper 20 m ocean temperature (May-Sep)	3.201	-0.771	0.082	0.159	0.509
Deep ocean temperature (May-Sep)	5.857	-1.336	0.108	0.137	0.466
Upwelling anomaly (Apr-May)	-4.825	-0.013	0.141	0.116	0.140
Length of upwelling season upwelling index based	-3.838	-0.005	0.670	0.010	-0.031
Deep ocean salinity (May-Sep)	-26.661	0.653	0.778	0.005	-0.240
Sum of weighted Columbia River dam spill proportions	-4.877	0.163	0.819	0.003	0.145
Physical spring transition upwelling index based (DOY)	-4.360	-0.003	0.839	0.002	0.163

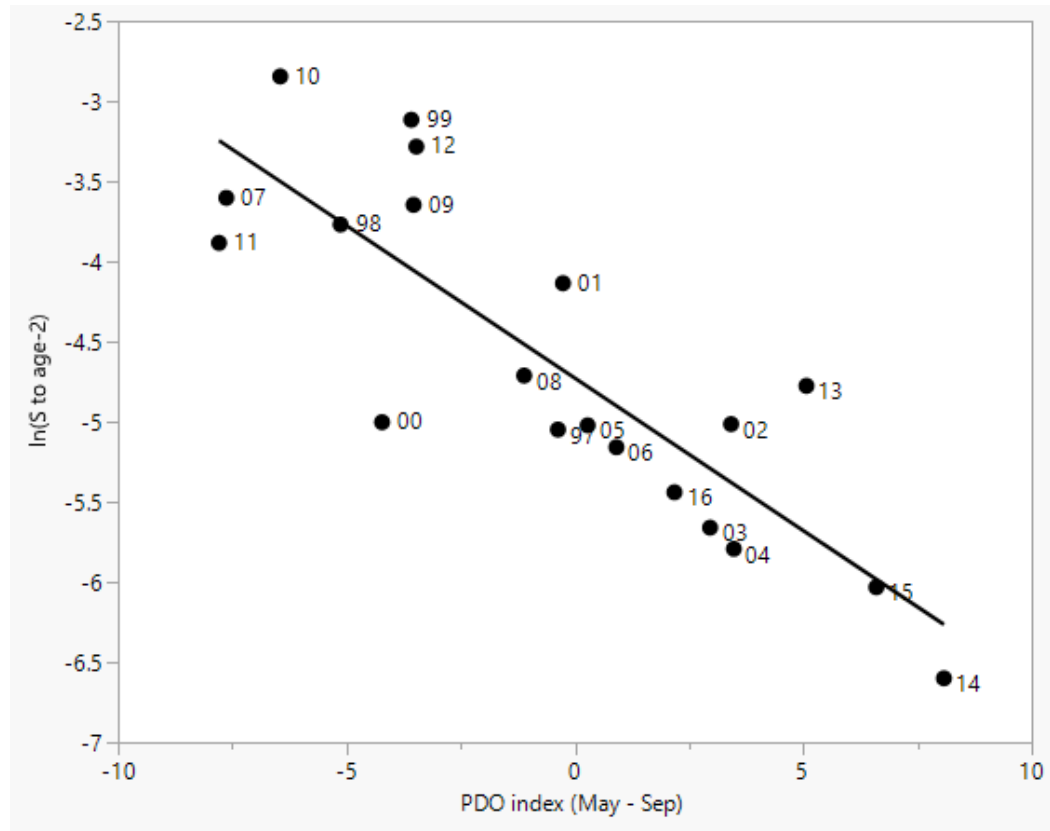


Figure 3.4. Simple Linear Regression Model Displaying the Relationship between the May–September Pacific Decadal Oscillation (PDO) Index and the Log-Transformed Presmolt-to-Age-2 Survival of Hanford Reach Upriver Bright Fall Chinook Salmon ($\ln(S \text{ to age-2})$), Brood Years 1997–2016. Numbers next to points indicate brood year.

Table 3.7. Best-Fitting Multivariable Regression Model Identified for Predicting Log-Transformed Presmolt-to-Age-2 Survival of Hanford Reach Upriver Bright Fall Chinook salmon, BY 1997–2016. CPD = coefficient of partial determination.

Term	Estimate	SE	<i>t</i> Ratio	Pr > <i>t</i>	CPD
Intercept	2.429	2.599	0.93	0.364	
May–Sep PDO index	−0.088	0.037	−2.42	0.028	0.267
May–Sep N. copepod biomass anomaly	0.869	0.320	2.71	0.015	0.315
Mean weighted Columbia R. temp.	−0.386	0.141	−2.74	0.015	0.319

The May–September PDO index, May–September northern copepod biomass anomaly, and mean weighted Columbia River temperature covariates were added to the linearized Ricker AR1 reduced model and fit to the log-transformed adult/spawner estimates of BY 1997–2016. The model had an R^2 of 0.87 and an adjusted coefficient of determination ($\bar{R}^2$) of 0.83. However, the coefficient estimates for R_{BY-1}/S_{BY-1} , Columbia River temperature, and northern copepod biomass were not significantly different from zero ($P > 0.44$). Therefore, these terms were dropped from the model, and the data were refit with a simpler model that incorporated only the intercept, adult spawners (S_{BY}), and the May–September PDO index. The simpler model had an R^2 of 0.87 and an adjusted coefficient of determination ($\bar{R}^2$) of 0.85

(Table 3.8). Coefficients of partial determination (CPD) were calculated to express the relative importance of individual contributions of escapement and ocean conditions to the model. According to the CPDs, the May–September PDO index explained 64% of the variability in adults/spawner after adult escapement had already been added to the model and adult escapement explained 20% of the variability after the PDO index had already been added.

Table 3.8. Results of the May–September PDO Index Added to the Linearized Ricker Model and Fit to the Hanford Reach Fall Chinook Salmon Adults/Spawner Stock-Recruit Data, BY 1997–2016. CPD = coefficient of partial determination.

Variable	Coefficient	Standard Error	<i>t</i>	<i>P</i> -value	CPD
intercept	1.073	0.232	4.63	<0.001	
<i>S_{BY}</i>	-5.656×10^{-6}	2.274×10^{-6}	-2.08	0.053	0.202
May–Sep PDO	-0.179	0.032	-5.54	<0.001	0.644

To evaluate the relative contribution of freshwater and ocean life stages on adult production, the log-transformed adult/spawner estimates were fit as a function of Hanford Reach (i.e., egg-to-presmolt) survival and the May–September PDO index. These two variables explained 88% of the variability in adult/spawner production. According to the CPDs, egg-to-presmolt survival explained 26% of the variability in spawner-to-adult survival when the May–September PDO index was already included in the model and the PDO index explained 83% of the variability when egg-to-presmolt survival was already included (Table 3.9).

Table 3.9. Results of a Multivariable Model in which Log-Transformed Spawner-to-Adult Survival was Fit as a Function of Egg-to-Presmolt Survival (P/E) and the May–September PDO Index. CPD = coefficient of partial determination.

Variable	Coefficient	Standard Error	<i>t</i>	<i>P</i> -value	CPD
intercept	0.215	0.200	1.07	0.298	
P/E	1.206	0.496	2.43	0.027	0.258
May–Sep PDO	-0.208	0.023	-9.14	<0.001	0.831

4.0 Discussion

Past productivity analyses, which included BY 1975–2009, were conducted to evaluate changes to freshwater (i.e., egg-to-presmolt) productivity of the Hanford Reach fall Chinook salmon population that corresponded with operational constraints enacted at Priest Rapids Dam to reduce the effects of discharge fluctuations on intra-gravel and post-emergent life stages (Harnish et al. 2012, 2014a, 2017). Results from those analyses indicated freshwater productivity increased as a result of operational constraints by limiting dewatering of redds and stranding and entrapment of juveniles. However, examinations of productivity, density dependence, carrying capacity, and limiting factors were somewhat limited due to the relatively narrow escapement range over which they could be evaluated. In the current study, seven additional years (BY 2010–2016), including three years of record escapements (>150,000 adults), were added to the data set to improve our understanding of this important population.

Updating the Hanford Reach fall Chinook salmon productivity analysis with several years of high escapements reaffirmed the highly productive nature of this population and the positive effect of flow fluctuation constraints as implemented at Priest Rapids Dam by the VBSA and HRF CPP. Productivity appeared to have increased with each additional operational constraint. Fitting the Beverton-Holt model to each Priest Rapids Dam operational period independently indicated the productivity parameter increased from 0.39 presmolts/egg during the pre-VBSA period, to 0.58 presmolts/egg during the VBSA, to 0.81 presmolts/egg during the HRF CPP.

Although productivity appears to have increased because of the HRF CPP operations, estimated egg-to-presmolt survival probability was below-average for each year added to the data set. Egg-to-presmolt survival averaged 0.187 (range: 0.137–0.242) for BY 2010–2016, compared to a mean of 0.332 for the entire 42-year data set. Below-average egg-to-presmolt survival was to be expected from these brood years due to the high escapements and the density dependent relationship observed in past productivity analyses (Harnish et al. 2012, 2014a, 2017). Because the mechanism driving the density dependent relationship remained unknown after past productivity analyses due to a lack of data at high escapements, the level of density dependent mortality caused by the high escapements was difficult to predict. With the inclusion of BY 2010–2016, the Beverton-Holt model provided the best fit to the egg-to-presmolt data, indicating juvenile competition for limited resources is driving density dependence in the Hanford Reach.

The addition of multiple years of high escapements provided little evidence that spawning habitat is limiting production of Hanford Reach fall Chinook salmon. Geist et al. (2008) predicted there to be about 337 to 423 ha of fall Chinook salmon spawning habitat in the Hanford Reach. Assuming a mean redd size of 20 m² (Burner 1951; Chapman et al. 1986; Visser et al. 2002) and 7.5 adults per redd (Hoffarth, unpublished data), the Hanford Reach has capacity for 1.26 to 1.58 million adult spawners. Using a more conservative estimate of two adults per redd results in a capacity of 337,000 to 423,000 adult spawners. These estimates of spawner capacity greatly exceed all escapement estimates dating back to 1975, supporting the theory that limited spawning habitat is not the mechanism driving density dependence in the Hanford Reach. The theory is further supported by the findings of McMichael and James (2017) who examined egg loss in relation to spawning density in the Hanford Reach during the high escapement year of 2014 and did not detect large numbers of dislodged eggs in areas of high redd density.

Since enactment of flow fluctuation constraints at Priest Rapids Dam, which began with the interim VBSA in 1984, maximum recruitment has stabilized around 90 million presmolts with an escapement of about 110 to 130 million eggs (~47,000 to 55,000 adult spawners) necessary to produce half of maximum recruitment. This range of escapements also comports well with the estimated adult escapement required to achieve maximum sustainable yield ($S_{MSY} = 48,925$ adult spawners), which is slightly greater than the

WDFW's current maximum escapement goal (42,000 adults spawners) for the Hanford Reach fall Chinook salmon population that was based on results from past productivity analyses.

Estimated adult per spawner production was below-average for 5 of the 7 years added to the data set. The number of adults produced per spawner averaged 2.26 (range = 0.24–5.87) for BY 2010–2016, compared to a mean of 4.02 for the entire 42-year data set. An evaluation of the factors affecting presmolt-to-age-2 survival and adult per spawner production indicated the importance of ocean conditions on survival outside of the Hanford Reach. For BY 1997–2016, the May–September PDO index explained 72% of the variability in presmolt-to-age-2 survival and appeared to explain substantially more variability in adult per spawner production than spawner abundance or Hanford Reach (i.e., egg-to-presmolt) survival. The average estimated number of age-3 adult equivalents produced per spawner was 4.63 (range = 3.79–5.87) for BY 2010–2012, which were associated with negative (i.e., cold phase) PDO index values (mean = –5.90; range = –3.47 to –7.79). Conversely, BY 2013–2016 were associated with strong positive temperature anomalies (i.e., “the blob”) that developed in the northeast Pacific Ocean during the winter of 2013–2014 and persisted into 2016 (Bond et al. 2015; Peterson et al. 2016). These positive (i.e., warm phase) PDO index values (mean = 5.48; range = 2.18 to 8.08) resulted in considerably smaller numbers of adults produced per spawner (mean = 0.47; range = 0.24 to 0.62). These findings are supported by evidence from throughout the world that suggests fish abundances, including Pacific salmon, are highly correlated with climate indices such as the PDO (Beamish and Boullion 1993; Mantua et al. 1997; Hannah 2011). As evidenced by the strong correlations between the May–September PDO index and the biological (i.e., copepod and ichthyoplankton richness, biomass, and community) and physical (e.g., sea surface temperature) ocean condition indicators, the PDO index provides a good overall indicator of the factors important to Hanford Reach fall Chinook salmon survival during their early ocean residence.

In addition to ocean conditions, factors within the hydrosystem appear to affect survival of smolts once they emigrate from the Hanford Reach. We observed significant correlations between presmolt-to-age-2 survival and Columbia River temperature and discharge for Hanford Reach fall Chinook salmon from BY 1997–2016. In addition to reaching potentially stressful limits during the summer, water temperatures in the Columbia River also affect the metabolic demands of predators (Kelso 1972; Rice et al. 1983). Some evidence suggests mortality of Hanford Reach fall Chinook salmon smolts emigrating through the hydrosystem, particularly in McNary and John Day pools, is quite high (McMichael et al. 2006; Fryer 2010; Harnish et al. 2014b) with much of the mortality attributed to piscivorous fishes (Tabor et al. 1993; Harnish et al. 2014b; McMichael and James 2017). Higher flows would be expected to move smolts through the hydrosystem more quickly, thereby reducing their exposure to predators. River temperature and discharge showed strong correlations with the May–September PDO index ($|r| > 0.55$), which is not surprising since the climate patterns that affect conditions in the northeast Pacific Ocean also affect weather patterns of the Columbia River basin. These associations make it difficult to tease apart the effect of conditions in the river from those in the ocean. However, the strong correlation of the PDO index with presmolt-to-age-2 survival and adult per spawner estimates suggest a greater influence of ocean conditions. In an evaluation of the factors affecting natural-origin Snake River spring/summer Chinook salmon survival across multiple life stages (smolt migration, ocean, adult migration), Gosselin et al. (2021) found that the greatest variability in survival occurred in the ocean with sea surface temperature having the largest effect.

Some research has suggested that Columbia basin salmon experience delayed or latent mortality in the marine environment as a result of their hydrosystem passage experience (e.g., Petrosky and Schaller 2010; Haeseker et al. 2012; Schaller et al. 2014; McCann et al. 2019). Specifically, these studies hypothesized that higher numbers of powerhouse passages are related to reductions in marine survival caused by stress or injuries incurred during powerhouse passage. They postulated that higher spill levels at Columbia River dams can improve salmon survival in the marine environment by guiding smolts away from the powerhouse to the spillway and by increasing smolt travel rates through the hydrosystem (e.g.,

McCann et al. 2016). However, we observed no effect of spill at lower Columbia River dams on presmolt-to-age-2 survival for BY 1997–2016. The lowest flow year experienced by Hanford Reach fall Chinook salmon smolts from BY 1997–2016 occurred in 2001. Due to low flows, little to no water was spilled at the lower Columbia River dams in 2001, resulting in a sum of weighted mean spill proportions of 0.30 for BY 2000, which was substantially lower than the sum of weighted mean spill proportions from all other years (range = 1.16–1.88). Because there was little to no spill, nearly all smolts emigrating through the hydrosystem passed through the powerhouses of the dams. Most powerhouses of the lower Columbia River are equipped with turbine intake screens that divert a portion of the smolts that enter the intakes away from the turbines and into a juvenile bypass system. In 2001, nearly all smolts that were diverted into the bypass system at McNary Dam were loaded onto barges and transported downstream where they were released below Bonneville Dam to reduce exposure to the poor river conditions (i.e., low flow, high temperature). Although the number of smolts passing through the powerhouse of McNary Dam would have been quite high, many of those fish (i.e., those that were transported) would not have had an opportunity to pass through powerhouses at downstream dams. Therefore, BY 2000 was something of an outlier in the evaluation of spill effects on presmolt-to-age-2 survival. However, even when BY 2000 was excluded, presmolt-to-age-2 survival remained uncorrelated with the sum of weighted mean spill proportions ($R^2 < 0.01$; $P = 0.91$).

Our results agree with those of Harnish et al. (2021a) who evaluated the delayed or latent effects of powerhouse passage for over 6.6 million PIT-tagged inriver migrant hatchery Snake River spring/summer Chinook salmon and steelhead over a 15-year period. Results from that evaluation found ocean condition covariates (i.e., May–September PDO, SST) were significantly correlated with marine survival rates, explaining 45 to 59% of the variability in marine survival, whereas indices of powerhouse passage and spill displayed no correlation with marine survival. Similarly, Gosselin et al. (2021) found only a small and insignificant negative association between powerhouse passage and ocean survival, which was best explained by sea surface temperature, for wild Snake River spring/summer Chinook salmon.

Smolt size has been found to be a significant factor affecting marine survival of Pacific salmon (Holtby et al. 1990; Koenings et al. 1993; Miyakoshi et al. 2001; Farley et al. 2007; Cross et al. 2009; Drenner et al. 2012; Faulkner et al. 2019; Gosselin et al. 2021). The two mechanisms of size-selective mortality most plausible for Pacific salmon include differential vulnerability to predation and differential susceptibility to starvation or exhaustion (Sogard 1997; Zabel and Williams 2002). Potential causes of differential vulnerability to predation by size include limitations of predator mouth gape size, behavioral selection, or size-dependent escape ability of salmon (Faulkner et al. 2019). Most marine mortality incurred by salmonids occurs soon after ocean entry (Percy 1992) when size-selective predation has been demonstrated (Parker 1971; Healy 1982; Hargreaves and LeBrasseur 1986; Holtby et al. 1990). As predators themselves, larger salmon smolts would also have a wider selection of prey due to their larger gape size and faster swimming speed, allowing faster growth and lower risk of starvation (Faulkner et al. 2019).

Differential susceptibility to starvation or exhaustion may be explained by the critical size hypothesis, which suggests that juvenile salmonids must reach some minimum critical size to survive their first winter at sea and may be related to having enough energy reserves available at the onset of winter (Beamish and Mahnken 2001; Farley et al. 2007). Zabel and Williams (2002) postulated that the starvation/exhaustion mechanism is supported by strong relationships between marine survival or overall salmon productivity and ocean conditions, such as those observed by Mantua et al. (1997), Hare et al. (1999), and Beamish et al. (2000). Multiple studies have found that larger smolts displayed a survival advantage over smaller individuals when marine survival was relatively poor but not in years of good ocean conditions (Holtby et al. 1990; Saloniemi et al. 2004). When ocean conditions are good, more smolts would be expected to reach the critical size to survive their first ocean winter. During poor ocean conditions, food availability and growth would be reduced; therefore, only the largest smolts at the time of ocean entry would be

expected to reach the critical size in time. Because smaller smolts grow more slowly (Ruggerone et al. 2009), they would be less likely to reach the critical size in time, particularly if ocean conditions are poor (Holtby et al. 1990; Saloniemi et al. 2004).

In addition to improved survival in the marine environment, larger smolts are also likely to experience higher survival in freshwater. In an acoustic telemetry study conducted by Harnish et al. (2014b), natural-origin Hanford Reach fall Chinook salmon and Priest Rapids Hatchery fall Chinook salmon juveniles showed a strong positive correlation between fish length and survival to McNary Dam. In an evaluation of the factors affecting survival of acoustic-tagged steelhead, spring/summer Chinook salmon, and fall Chinook salmon smolts through multiple reaches of the Columbia River hydrosystem, Harnish et al. (2021b) found that survival was often positively correlated with fish length. As in the ocean, survival in freshwater may depend on size-related vulnerability to predation and susceptibility to starvation or exhaustion. The mainstem Columbia River is a predator-rich environment where much of the mortality incurred by fall Chinook salmon juveniles can likely be attributed to predation from resident piscivorous fish and size may determine the risk of predator consumption (Stephens and Krebs 1986; Harnish et al. 2014b, 2021b). In addition to an increased risk of size-related predation, smaller salmon smolts have been shown to be more susceptible to passage through turbines at Columbia River dams (Zabel et al. 2005; Hostetter 2015; Faulkner et al. 2019; Harnish et al. 2021b). Of all passage routes available to smolts at the dams, the turbines have historically been associated with the highest mortality rates (Skalski et al. 2021).

Our results indicate escapements in excess of about 50,000 adults may cause competition among juveniles in the Hanford Reach, which could result in density-dependent mortality or reduced growth. McMichael et al. (2015) detected a significant negative relationship between Hanford Reach fall Chinook salmon escapement and subsequent smolt size, indicating density dependent effects on juvenile growth. If spawning habitat is not limited and environmental conditions are favorable, as they generally have been in the Hanford Reach since implementation of the VBSA and HRFCPP, then high egg-to-fry survival and fry abundance would be expected from high escapements. During the high escapement year of 2014 (~152,00 adult spawners), McMichael et al. (2015) found that emergence timing was shifted later in areas of very high spawning density and postulated that a combination of later emergence and factors affecting post-emergent fry growth could be important density dependence mechanisms influencing the Hanford Reach fall Chinook salmon population. Density-dependent effects on growth are supported by many examples in the literature. For example, in a review of 199 published data sets (21 species) to assess the relative frequency of various density-dependent processes in salmonids, Grossman and Simon (2020) found that 71% of the studies showed density-dependent growth effects.

Management actions aimed at reducing competition among juveniles in the Hanford Reach may be the most beneficial for improving survival to adulthood. Such actions may include increasing harvest when high escapements are anticipated to reduce the number of juveniles that are competing for limited resources. Limiting competition would be expected to result in larger juveniles that would be less likely to suffer size-dependent mortality and improve their chances of surviving their first ocean year even during years of poor ocean conditions.

5.0 Conclusions and Recommendations

Updating the productivity analysis to include three years of record escapements reiterated the effectiveness of flow fluctuation constraints implemented at Priest Rapids Dam in the form of the VBSA and HRFCPP. Freshwater (egg-to-presmolt) productivity appears to have increased with each additional constraint. Competition among juveniles for limited resources seems to be the mechanism driving density dependence within the Hanford Reach. Based on results from the initial productivity analysis, the WDFW Region 3 altered the escapement goals for the Hanford Reach fall Chinook salmon population for the purpose of managing the terminal sport fishery. These alterations consisted of increasing the minimum escapement goal from 28,800 adults to 31,100 adults and setting a maximum escapement goal of 42,000 adults. Results from the updated productivity analysis, which estimated S_{MSY} to be 48,925 adult spawners, suggest the WDFW may consider increasing the maximum escapement goal to 49,000 adult spawners. However, under current HRFCPP operations, ocean conditions may be the primary factor controlling adult production. Therefore, the success of any additional management actions taken in freshwater may be dependent on survival in the ocean. The most effective management actions may be those that reduce competition among juveniles in the Hanford Reach. Limiting competition would be expected to increase growth, resulting in larger smolts that would be less likely to suffer size-dependent mortality and improve their chances of surviving their first year in the marine environment even during periods of poor ocean conditions. As such, it may be more important to avoid years of high escapements, which could reduce smolt size and ultimately survival to adulthood. Increasing terminal harvest rates when high escapements are anticipated (i.e., from robust downstream dam counts) is a low risk and direct management option for addressing this circumstance.

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