Conservation Objective for Southern Oregon coastal Chinook

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Introduction

The current conservation objective for Southern Oregon coastal Chinook (SOCC) in the Pacific Coast Salmon Fishery Management Plan was based on information and reports that are now outdated. SOCC stocks are included as an unspecified portion of an aggregate Oregon coast natural adult spawner goal of 60-90 spawners per mile. As a result of the Amendment 16 process SOCC were classified as a component stock of the Southern Oregon Northern California complex regulated by Annual Catch Limits that use Klamath River fall Chinook as the indicator stock. Oregon Department of Fish and Wildlife (ODFW) recently completed the *Conservation Plan for Fall Chinook Salmon in the Rogue Species Management Unit* (Rogue Plan) and it was adopted by the Oregon Fish and Wildlife Commission in January, 2013 (ODFW 2013). The Rogue Plan covers the geographic area and fall Chinook stocks that are defined as SOCC.

Analyses used in developing the Rogue Plan are described in detail in the plan itself and its appendices, which may be found at:

http://www.dfw.state.or.us/fish/CRP/rogue_fall_chinook_conservation_plan.asp

and

http://dfw.state.or.us/fish/CRP/docs/rogue_fall_chinook/Rogue_fall
Chinook_Plan_Final_Appendixes_1-11-13.pdf

Development of new conservation objectives may be implemented without plan amendment upon approval by the Council. The Rogue Plan provides new information and data analyses for use in updating the current conservation objective for SOCC.

Methods and Results

Spawner and Freshwater Escapement Estimates For Rogue River Basin Fall Chinook

There are five populations of fall Chinook present in the Rogue River Basin. The populations are generally defined as Upper Rogue, Middle Rogue, Lower Rogue, Applegate, and Illinois. Availability of adult fall Chinook abundance data within these populations varies markedly. An overview of the sources of available data follows. Only those data sets which covered at least five consecutive years are described. Also described is the relevance of the data to the development of the Rogue Plan.

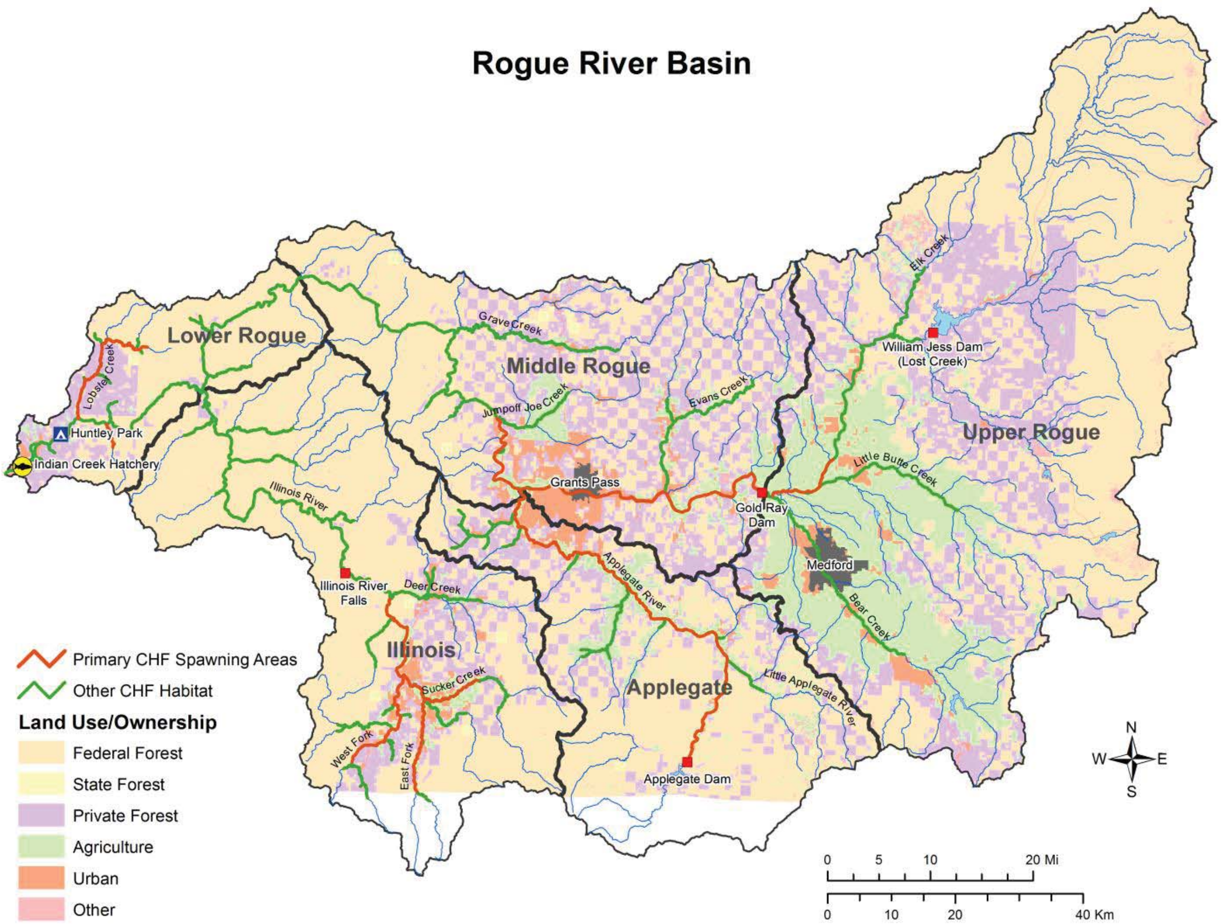

Figure 1. Spatial distribution of independent populations of fall Chinook salmon in the Rogue stratum of the Species Management Unit. The boundaries of the different population areas are shown as black lines.

Lower Rogue population:
Spawners (live and dead fall Chinook) were counted in portions of various tributary streams during 1986-2010. This database was directly relevant to the purpose of the Rogue Plan because spawner abundance could be estimated for the entire population, criteria could be developed in

relation to desired status and conservation status, and monitoring is ongoing and expected to continue for the foreseeable future.

Middle Rogue population:
Carcasses were counted within two survey areas located on the mainstem of the Rogue River during 1974-2004. The areas surveyed included Valley of the Rogue State Park - the city of Rogue River (RM 113-111) and Lathrop's Landing - Robertson Bridge (RM 97-87). This database was judged to not be directly relevant to the purpose of the Rogue Plan because the data could not be used to estimate spawner abundance for the entire population, and monitoring has been terminated and is not expected to be implemented again within the foreseeable future. As a result it was judged there was minimal value in trying to craft criteria for desired status or conservation status for this population.

Upper Rogue population:
Fish passage at Gold Ray Dam (RM 125) was estimated at a counting station during 1942-2009. In August 2010, the fish counting station became inoperable with the removal of Gold Ray Dam. Substitution of spawning surveys to estimate fall Chinook spawner abundance is not possible because of the spatial and temporal overlap in spring and fall Chinook spawning in this area (ODFW 1991). With the loss of the fish counting station, it was judged there was minimal value in trying to craft criteria for desired status or conservation status for this population.

Illinois population:
Spawners (live and dead fall Chinook) were counted within portions of three tributary streams (Mendenhall Creek, Elk Creek, and Sucker Creek) during 1996-2004. This database was judged to not be directly relevant to the purpose of the Rogue Plan because the data could not be used to estimate spawner abundance for the entire population, and monitoring has been terminated and is not expected to be implemented again within the foreseeable future. In addition, ODFW staff judged that fall Chinook spawn primarily in the mainstem, and in the East and West Forks. It is currently unknown whether spawner counts in the smaller tributaries may be representative of spawning escapement throughout the entire basin. As a result of these factors, it was judged there was minimal value in trying to craft criteria for desired status or conservation status for this population.

Applegate population:
Carcasses were counted within three survey areas located on the mainstem of the Applegate River, and in Slate Creek, during 1974-2004. The areas surveyed in the Applegate River included the town of Applegate - Williams Creek (RM 25-20), the town of Murphy - Hog Ranch (RM 13-11), and Highway 199 - the mouth (RM 4-0). The Slate Creek survey covered the lowest 5.0 miles of Slate Creek. This database was judged to not be directly relevant to the purpose of the Rogue Plan because the data could not be used to estimate spawner abundance for the entire population. Monitoring has been terminated and is not expected to be implemented again within the foreseeable future. As a result, it was judged there was minimal value in trying to craft criteria for desired status or conservation status for this population.

Aggregate populations

Migrating adults were captured during 1976-2010 with a 300' beach seine fished at Huntley Park (RM 8) three days weekly during July 15-October 28 (ODFW 1992). Each day, sampling began early in the morning and continued until the seine had been fished 15 times. This sampling effort was standardized in 1978 and tagging studies indicated that all of the fall Chinook populations in the Rogue River Basin are susceptible to capture (ODFW 1992).

The Oregon Game Commission (OGC, now ODFW) began beach seining near the mouth of the Rogue River in 1974 in order to capture adult salmonids that entered the Rogue River. Initially, the sampling was designed to collect fish in order to obtain life history information and to estimate freshwater escapement through the use of mark-recapture methods. Mark-recapture efforts were terminated after 1976 when it became apparent that mortality rates of tagged Chinook salmon resulted in biased estimates of freshwater escapement (Cramer 1979). Instead, catch per unit of seining effort was used as an index of abundance. This approach continued through the 1980s, although it became apparent that unusually high flows in 1983 and 1984 affected the efficacy of sampling with the beach seine at the Huntley Park site.

Establishment of a run of hatchery coho salmon in the early 1980s afforded an opportunity to generate annual estimates of seining efficiency. Available data indicated that few coho salmon died during upstream migration, few hatchery fish strayed to spawn naturally, and at the time, there was no directed freshwater fishery for coho salmon. Seining efficiency on coho salmon of hatchery origin was estimated, compared to flow at time of seine capture, and a catch efficiency model was developed (ODFW 1989). This flow-based model was subsequently used to estimate freshwater escapement for other runs of anadromous salmonids, including fall Chinook (ODFW 1989; ODFW 1992; ODFW 1994).

In 1992, ODFW determined that the flow-based model significantly underestimated the number of fall Chinook that returned to the Rogue River. During some years, known numbers of fall Chinook exceeded the estimate produced by the flow-based model. Known numbers of fall Chinook included: (1) those that passed the fish counting station at Gold Ray Dam, (2) those recovered as carcasses during spawning surveys, and (3) those estimated to be harvested by anglers based on returns of salmon-steelhead cards. In light of these results, ODFW subsequently termed estimates derived from the flow-based model as the "Huntley Park Index" of fall Chinook freshwater escapement, to differentiate it from a formal abundance estimate.

Estimating fall Chinook Passage at Huntley Park

ODFW has developed two methods to estimate the number of fall Chinook that passed Huntley Park. Both methods entail expansion of the Huntley Park Index. The first method uses the results of mark-recapture efforts during 2000-2002 to calibrate the passage index. The second method uses historic fall Chinook passage estimates at Gold Ray Dam to calibrate the passage index. Both methods resulted in similar passage estimates at Huntley Park and a description of each method follows.

Calibration of Huntley Park Index with mark-recapture estimates:
ODFW has tried twice (once during 1975-76 and once during 2000-2002) to estimate fall Chinook escapement in the Rogue River using mark-recapture methods. Freshwater returns were estimated to be about 63,000 fish in 1975 and about 93,000 fish in 1976 (Table 1). However, these estimates were judged to be inflated by disproportionately high mortality of tagged fish (Smith et al. 1978).

Instances of significant prespawning mortality decreased markedly after the mid-1980s as water release strategies at Lost Creek Dam were modified to increase flow during critical periods of fall Chinook migration (ODFW 1992). The only instance of significant prespawning mortality occurred during the drought year of 2001 (Satterthwaite 2002). The decrease in fall Chinook prespawning mortality led ODFW to attempt another series of mark-recapture efforts with fall Chinook during 2000-2002. Resultant mark-recapture estimates of the number of fall Chinook that passed Huntley Park during these years ranged between 126,000 and 405,000 fish (Table 1).

Table 1. Petersen mark-recapture estimates of the number of fall Chinook salmon that entered the Rogue River, 1975-2002 and associated data relevant to the calibration of the Huntley Park Index of freshwater escapement. River physical factors are reported as mean daily maximum values during August at Agness (RM 30).

Year	Mark-recapture estimate (95% CI)		Huntley Index	River physical factors	
				Flow (cfs)	Temperature (F)
1975	63,235	(47,160-87,655)	37,175	1,716	--
1976	92,977	(61,807-147,043)	23,469	2,149	71
2000	126,085	(88,540-208,919)	40,047	2,317	73
2001	404,660	(192,880-616,440)	42,577	1,762	74
2002	203,267	(150,057-290,622)	80,545	2,027	72

All of the mark-recapture estimates listed in Table 1 are believed to be inflated to some degree because of delayed mortality among tagged fish. Bias related to tagging mortality was judged to be differentially high in 2001 because of low flow and extensive prespawning mortality (Satterthwaite 2002). Consequently, the 2001 estimate was excluded from further consideration. Tagging related mortality was assumed to be 10% in the other years, resulting in adjusted mark-recapture estimates of 113,476 in 2000 and 182,940 in 2002. The escapement estimates exceeded the Huntley Passage Indexes by 2.70-fold in 2000 and 2.27- fold in 2002. The average (2.485) of these values was used to expand the Huntley Passage Index for the period of record. Expanded values were then used as estimates of the total number of fall Chinook that passed Huntley Park. Resultant estimates are shown as "Method 1" in Table 2.

Calibration of Huntley Park Index with Gold Ray Dam counts:
ODFW has estimated fish passage at Gold Ray Dam since 1942. All passing fish were counted during 1942-1947. During 1948-92, fish were counted eight hours daily, five days weekly. Partial count sampling designs were intended to estimate biweekly passage with an average error of less than 10% (Li 1948). Since 1993, passage has been estimated with 24-hour, seven day per

week video recordings; a procedure which is assumed to have minimal uncertainty. Chinook salmon that pass the counting station by August 15 are classified as spring Chinook, while later migrants are classified as fall Chinook (ODFW 2000).

ODFW tagged numerous fall Chinook in the lower Rogue during 1974-78 and looked for tags on carcasses found during spawning surveys. Most tags were recovered in the Middle Rogue, Applegate, and Lower Rogue population areas. However, five tags were recovered upstream of Gold Ray Dam (Upper Rogue population area). All of these fish had been captured and tagged at Huntley Park between July 15 and August 4 and were considered early-run fall Chinook. Remaining tag recoveries indicated that later migrating fall Chinook eventually spawned in population areas farther downstream in the Rogue River Basin (ODFW 1992). Early-run fish are therefore defined as those fall Chinook that pass Huntley Park before August 4.

Application of radio-tags to a few fall Chinook caught at Huntley Park in 2008 (ODFW 2009) afforded the opportunity to examine the assumption that early-run fall Chinook migrate upstream to spawning areas above Gold Ray Dam. There were three early-run fall Chinook tagged at Huntley Park that passed Gold Ray Dam. One passed on August 11 and was therefore classified as a spring Chinook. The other two passed after August 5 and were thus classified as fall Chinook. Another three fall Chinook were tagged at Huntley Park after August 4 and were subsequently detected on spawning grounds downstream of Gold Ray Dam.

The OGC tagged large numbers of fall Chinook near the mouth of the Rogue River in 1970 and in 1971 during a summer steelhead research project (Everest 1973). Efforts to recover tagged fall Chinook were limited to a few spawning surveys, but 36 tagged fall Chinook were trapped at Gold Ray Dam. The mean date of tagging at the river's mouth was August 11 (95% CI = ±4 days). These results, coupled with the 1974-78 tag recoveries, confirmed that early-run fall Chinook primarily spawn upstream of Gold Ray Dam. Assuming that all early-run fish pass the Gold Ray Dam counting station, an appropriate expansion factor could be developed for the Huntley Park Index.

The early-run component of the Huntley Passage Index accounted for an average of 40% (95% CI for arc-sine transformed data = 17-64%) of the early-run fall Chinook that subsequently reached Gold Ray Dam during 1992-2008. Data from years prior to 1992 were not included because of concern that the population of fall Chinook in the upper Rogue was still increasing during that time relative to fall Chinook in the remainder of the basin (ODFW 2000). Because 17 years (1992-2008) of data are available, it was judged that effects due to variations in fall Chinook migration timing would likely be mostly cancelled provided that annual variations in fall Chinook migration timing were random in nature. Assuming that 40% of fish passing Huntley Park are reflected in the Huntley Park Index, an expansion factor of 2.5 (1/40%) was used to produce the Huntley Park estimates referred to as "Method 2" in Table 2.

Table 2. Four abundance metrics for abundance of fall Chinook salmon in the Rogue River Basin, 1974-2010.

Year	Total known[a]	Huntley Index[b]	Passage at Huntley Park: Method 1[c]	Passage at Huntley Park: Method 2[d]
1974	--	42,656	106,021	106,660
1975	--	37,175	92,383	92,940
1976	--	23,469	58,329	58,680
1977	12,697	32,038	79,615	80,095
1978	18,501	74,575	185,321	186,438
1979	13,239	69,730	173,281	174,325
1980	6,497	33,478	83,194	83,695
1981	13,552	41,420	102,942	103,563
1982	10,568	55,735	138,506	139,340
1983	9,314	21,464	53,336	53,658
1984	8,336	18,212	45,257	45,530
1985	20,282	36,109	89,722	90,263
1986	39,760	98,314	244,291	245,763
1987	51,204	65,133	161,857	162,833
1988	61,078	33,930	84,319	85,423
1989	24,787	38,767	96,337	96,918
1990	9,472	10,187	25,315	25,468
1991	10,749	7,544	18,747	18,860
1992	13,403	31,288	77,751	78,220
1993	22,515	19,002	47,220	47,505
1994	30,740	33,114	82,290	82,786
1995	28,580	35,444	88,079	88,610
1996	20,283	27,004	67,105	67,509
1997	10,056	24,625	61,193	61,562
1998	12,435	19,967	49,618	49,917
1999	9,500	23,710	58,920	59,275
2000	21,624	42,047	104,489	105,118
2001	29,095	42,577	105,805	106,442
2002	42,491	80,545	200,157	201,363
2003	57,760	94,231	234,167	235,577
2004	--	63,561	157,950	158,902
2005	--	25,821	64,167	64,553
2006	--	17,972	44,660	44,929
2007	--	20,366	50,740	50,914
2008	--	17,336	43,080	43,340
2009	--	30,453	75,676	76,132
2010	--	26,633	66,184	66,582

[a] *Carcasses were not surveyed after 2003 and only three areas were surveyed during 1974-76.*
[b] *Index values for 1974-77 were adjusted for non-standardized sampling (ODFW 1992).*
[c] *Huntley Park Index calibrated with mark-recapture estimates from 2000 and 2002.*
[d] *Huntley Park Index calibrated with 1992-2009 passage estimates at Gold Ray Dam.*

In summary, the Gold Ray Dam counts provide an accurate abundance estimate of the Upper Rogue population and tag recoveries indicate that this population passes Huntley Park by August 5. A comparison of the Huntley Park Index, calculated only for the period July 15-August 5, with the Gold Ray dam counts resulted in the conclusion that fall Chinook passage at Huntley Park is best estimated with the application of a 2.5X expansion factor (0.40/Huntley Park Index).

Estimation of Life History Parameters

Aggregate populations:
Scale interpretations were used to estimate the origin and age composition of fall Chinook that entered the Rogue River during 1974-1988 (ODFW 1992). Scales were also used to estimate fall Chinook age composition for the 2007-2011 returns. The age composition of the 1989-2006 returns was estimated based on length-at-age criteria developed from scale samples obtained from the 2007-2011 returns. Annual proportions of hatchery fish within the 1989-2011 returns were estimated by expanding the number of fin-clipped fish caught at Huntley Park by the mark rates among cohorts released from hatcheries in the Rogue River Basin. Fin-clipped fish were assigned to specific brood years based on their length. Final estimates of fall Chinook passage at Huntley Park can be found in Table 3.

All spawners were assumed to be naturally produced fish. During 1991-2004, ODFW recovered about 80,000 fall Chinook carcasses during spawner surveys conducted throughout the Rogue River Basin. Only 54 of those fish were marked with adipose fin clips and expansions for the proportions of smolts indicated that hatchery fish composed about 0.2% of the spawners, which was judged to be insignificant.

Table 3. Estimated number of adult fall Chinook salmon that passed Huntley Park and the estimated age composition of naturally produced fish, 1974-2011.

Return	Passage Estimate			Proportion by Age				
Year	Natural	Hatchery	Total[1]	Age 2	Age 3	Age 4	Age 5	Age 6
1974	106,021	0	106,021	0.189	0.271	0.461	0.070	0.009
1975	92,383	0	92,383	0.118	0.195	0.567	0.113	0.008
1976	58,329	0	58,329	0.414	0.174	0.343	0.069	0.000
1977	79,615	0	79,615	0.676	0.167	0.129	0.028	0.000
1978	185,321	0	185,321	0.162	0.377	0.405	0.056	0.000
1979	173,281	0	173,281	0.054	0.101	0.802	0.044	0.000
1980	83,010	184	83,194	0.343	0.110	0.284	0.262	0.000
1981	101,429	1,513	102,942	0.258	0.486	0.175	0.073	0.010
1982	134,684	3,822	138,506	0.274	0.266	0.432	0.027	0.001
1983	45,441	7,895	53,336	0.148	0.487	0.336	0.030	0.000
1984	42,255	3,002	45,257	0.231	0.374	0.360	0.029	0.005
1985	84,141	5,582	89,722	0.581	0.110	0.261	0.048	0.000
1986	229,858	14,433	244,291	0.373	0.497	0.113	0.016	0.001
1987	147,944	13,914	161,857	0.210	0.398	0.364	0.028	0.000
1988	79,078	5,241	84,319	0.144	0.198	0.606	0.052	0.000
1989	89,144	7,193	96,337	0.170	0.323	0.421	0.070	0.016
1990	23,915	1,400	25,315	0.183	0.370	0.395	0.051	0.000
1991	18,364	383	18,747	0.184	0.476	0.309	0.031	0.000
1992	76,456	1,295	77,751	0.415	0.232	0.277	0.069	0.008
1993	46,668	552	47,220	0.228	0.598	0.128	0.040	0.006
1994	80,707	1,584	82,290	0.164	0.435	0.357	0.043	0.001
1995	82,745	5,334	88,079	0.224	0.510	0.215	0.046	0.005
1996	64,445	2,660	67,105	0.243	0.380	0.338	0.036	0.003
1997	58,860	2,333	61,193	0.302	0.386	0.241	0.061	0.010
1998	47,732	1,886	49,618	0.142	0.577	0.257	0.024	0.000
1999	56,350	2,570	58,920	0.333	0.264	0.287	0.093	0.023
2000	100,701	3,787	104,489	0.128	0.581	0.216	0.070	0.004
2001	103,026	2,778	105,805	0.259	0.274	0.314	0.134	0.020
2002	196,948	3,209	200,157	0.217	0.318	0.313	0.119	0.033
2003	224,139	10,027	234,167	0.086	0.287	0.425	0.173	0.029
2004	152,081	5,869	157,950	0.130	0.197	0.446	0.188	0.040
2005	61,323	2,843	64,167	0.079	0.281	0.455	0.158	0.026
2006	41,845	2,815	44,660	0.162	0.254	0.428	0.134	0.023
2007	46,778	4,264	51,041	0.070	0.326	0.343	0.256	0.005
2008	39,495	3,751	43,246	0.384	0.181	0.336	0.099	0.000
2009	73,883	2,369	76,252	0.185	0.419	0.342	0.055	0.000
2010	63,849	2,335	66,184	0.223	0.348	0.390	0.038	0.002
2011	97,875	3,044	109,919	0.308	0.242	0.397	0.052	0.001

[1] Total is from Method 1, Table 2.

Estimation of Freshwater Harvest

Aggregate populations: Freshwater harvest (includes the estuary) was estimated from salmon-steelhead cards (punchcards) returned to ODFW by anglers. ODFW (1992) reported estimates of total harvest for 1956-1984. Estimates for later years were obtained from ODFW records. Harvest estimates from salmon-steelhead cards do not include jacks; which are almost all age 2 fish (ODFW 1992). Harvest of jacks was estimated based on their proportion among fall Chinook captured at Huntley Park (i.e., it was assumed that the freshwater fishery did not selectively harvest fall Chinook of different ages).

Estimates of fall Chinook harvest were segregated into areas upstream and downstream of Huntley Park. Angler harvest downstream of Huntley Park was assumed to equal the salmon-steelhead card estimates applicable to the Rogue River downstream of Elephant Rock (RM 3). Estimates for this area were only available for 1993 and later years. During this period, the area downstream of Elephant Rock accounted for an average of 53% (95% CI = 48-58% as estimated from arcsine transformed data). The mean estimate of harvest distribution was applied to years prior to 1993 in order to estimate angler harvest downstream of Elephant Rock.

Table 4. Estimates of population and harvest metrics for aggregated populations of naturally produced fall Chinook salmon in the Rogue River Basin, 1972-2006 brood years.

			River Harvest[a]		Total	Brood		
Brood year	Ocean harvest	River[a] return	Below Huntley	Above Huntley	river harvest	harvest rate	Recruits	Parent Spawners[b]
1972	64,832	41,380	1,125	989	2,115	0.668	100,232	--
1973	98,268	32,036	1,246	1,095	2,341	0.882	114,121	--
1974	212,244	97,548	1,712	1,505	3,218	0.744	289,621	82,518
1975	479,073	233,165	1,572	1,382	2,954	0.732	658,090	78,840
1976	105,752	48,979	444	390	834	0.756	141,072	32,474
1977	60,128	30,961	433	380	813	0.722	84,397	23,486
1978	176,491	110,628	1,642	1,444	3,086	0.630	285,148	134,691
1979	182,635	53,266	942	828	1,769	0.489	377,237	29,875
1980	73,955	42,684	1,198	1,053	2,252	0.320	238,336	23,206
1981	19,258	43,052	1,572	1,382	2,954	0.365	60,935	65,448
1982	45,547	40,631	1,157	1,017	2,173	0.578	82,591	92,768
1983	120,683	179,001	5,409	4,754	10,163	0.450	290,726	37,696
1984	84,580	122,570	9,563	8,406	17,968	0.509	201,344	31,683
1985	51,980	59,010	4,616	4,058	8,674	0.565	107,284	33,414
1986	27,551	42,414	2,981	2,620	5,601	0.486	68,221	140,969
1987	17,434	22,151	2,071	1,831	3,902	0.576	37,061	109,293
1988	3,721	34,422	2,578	2,337	4,915	0.228	37,919	58,733
1989	3,751	29,509	1,892	1,883	3,775	0.229	32,910	68,177
1990	8,882	67,132	6,456	6,666	13,122	0.292	75,470	17,403
1991	6,890	60,663	4,884	3,841	8,725	0.232	67,192	12,581
1992	9,291	73,109	5,536	3,671	9,207	0.226	82,007	42,112
1993	4,353	43,522	2,429	2,019	4,448	0.185	47,566	29,866
1994	3,294	42,572	1,909	1,418	3,327	0.145	45,622	60,887
1995	4,810	55,370	2,598	1,874	4,472	0.155	59,722	59,464
1996	11,699	59,517	2,623	2,430	5,054	0.241	69,616	44,949
1997	27,674	126,338	5,523	5,139	10,662	0.255	150,542	38,785
1998	50,222	139,168	4,503	5,952	10,455	0.330	183,635	38,864
1999	65,609	194,007	6,007	7,682	13,688	0.311	254,879	35,293
2000	68,144	147,337	4,559	5,743	10,302	0.372	210,998	82,935
2001	22,469	65,713	2,066	2,448	4,514	0.312	86,379	63,555
2002	9,134	49,755	2,629	1,650	4,279	0.230	58,211	144,954
2003	6,902	32,407	1,864	955	2,819	0.250	38,938	191,999
2004	5,004	34,601	1,917	1,101	3,017	0.205	39,173	124,571
2005	124	38,319	3,400	1,383	4,783	0.128	38,442	53,208
2006	1,987	66,201	5,301	2,355	7,656	0.140	68,654	32,873

[a] *Includes estuary.*

[b] *Age 3-6; includes hatchery fish.*

Estimation of Spawning Escapement

Aggregate populations:
Spawning escapement in the Rogue River Basin was estimated as:

Passage estimates at Huntley Park - (prespawning mortality + angler harvest)

Fall Chinook in the Rogue River are susceptible to high rates of prespawning mortality during years of low flow and warm water temperatures. Rates of prespawning mortality were estimated during 1978-1986 (ODFW 1992) and during 2001 (Satterthwaite 2002). Prespawning mortality rates in all other years were assumed to equal 2% because there were no anecdotal reports of significant prespawning mortality in years after 1986 (2001 excepted).

Estimation of Ocean Fishery Impacts

Annual exploitation rates on age 3 and age 4 fish in the ocean fisheries were assumed to equal those estimated for fall Chinook salmon of Klamath River Basin origin, as reported by the Pacific Fishery Management Council (PFMC 2010). Ocean exploitation rates on age 5 and age 6 fish were assumed to equal those on age 4 fish. These assumptions had to be made because there were no consistent releases of CWT marked fall Chinook from hatcheries within the Rogue River Basin that would allow for direct estimation of exploitation rates in the ocean fisheries.

The assumption of equal ocean exploitation rates on age 3 and age 4 fish of Rogue and Klamath origin appeared reasonable because (1) Rogue and Klamath fall Chinook exhibit indistinguishable catch distribution patterns in the ocean fisheries (Table 5) and (2) freshwater returns of fall Chinook in the Rogue and Klamath rivers are positively correlated to each other (*see* Comparisons to Other Populations, page 69 in the Rogue Plan). Weitkamp (2010) also documented very similar ocean landing distributions of Chinook salmon released from hatcheries in the Southern Oregon - Northern California ecoregion.

Table 5. Comparisons of landing distributions of CWT-marked fall Chinook salmon released from hatcheries in the Rogue and Klamath River basins, 1987-2003 brood years. Data incorporated in the analyses include only those CWT groups released after the month of August. Comparisons were made with paired t-tests of arcsine transformed data. Data from Iron Gate and Trinity River hatcheries in the Klamath River Basin were pooled because no difference in landing distributions could be detected (paired t-test $P = 0.52$).

	Mean by Stock		*P* for
	Rogue	Klamath	difference
Proportion landed in California and Oregon	0.99	0.99	0.76
Proportion landed in Oregon only	0.43	0.45	0.38

Estimation of Ocean Abundance

Cohort reconstructions (Ricker 1975) were employed to estimate the number of naturally-produced fall Chinook that resided in the ocean during the spring prior to onset of any fishing mortality. Estimation procedures began with age 6 fish and ended with age 3 fish and were analogous to those employed by Hankin and Healy (1986) and Mohr (2006). Estimates of cohort abundance began with age 6 fish because all naturally-produced fall Chinook of Rogue River Basin origin mature at ages 2-6. The abundance of younger cohorts were estimated as the sum of (1) the number of fish that resided in the ocean during the succeeding year, (2) natural mortality, (3) harvest in the ocean fisheries, and (4) the number of fish that returned to the river.

For each cohort, we used the equation:

$$N_i = \frac{\dfrac{N_{i+1}}{1-A_i} + E_i}{1-u_i}$$

where

N_i = number of age i fish resident in the ocean prior to fishing during year t,
N_{i+1} = number of age i+1 fish resident in the ocean during the next year,
A_i = rate of natural mortality for age i cohorts resident in the ocean between years t and t+1,
u_i = exploitation rate of age i fish in the ocean during year t, and
E_i = freshwater return of age i fish during year t.

Estimation of Recruitment

For each cohort (brood year), recruitment was estimated as the sum of the estimated freshwater returns of age 3-6 fall Chinook under a scenario of no ocean fishing mortality (as termed “adult equivalents” by Mohr (2006)).

Population Models

For each population, we assessed the relationship between the abundance of spawners on a given year and the resulting number of adult progeny (recruits) produced by those spawners. This spawner-recruit analysis yields information about trans-generational population dynamics that is subsequently used to assess extinction risk in a population viability analysis (PVA). This appendix begins by describing spawner-recruit analysis and then goes on to describe how results from this analysis are used in a PVA.

Spawner-Recruit Relationships

Annual estimates of spawner abundance and recruits produced for each population were used to assess the shape and strength of relationship between estimates of spawners and recruits. A simple straight linear relationship between spawners and recruits is biologically unrealistic because, among other reasons, it suggests that there is no upper limit to the number of recruits that can be produced. Thus, a nonlinear relationship between spawners and recruits is more appropriate. We considered the two most common relationships commonly used by fish scientists; the Ricker (1954) and Beverton-Holt (1957) curves.

The Ricker curve (Ricker, 1954):

$$R = \alpha S e^{-\beta_{RK} S} \qquad \textit{Eq. 1}$$

and the Beverton-Holt function (Beverton and Holt, 1957):

$$R = \frac{\alpha S}{1 + \frac{\alpha}{\beta_{BH}} S} \qquad \textit{Eq. 2}$$

Both functions model recruit abundance (*R*) as a two-parameter function of spawner abundance (*S*). In both equations, α represents "intrinsic productivity," which is the number of recruits produced per spawner as spawner abundance approaches zero. This value therefore represent the reproductive output when animals are uninhibited by density-dependent effects, and is an important component of population resiliency. The meaning of the β parameter is different in the two functions, and so we denote this difference by using subscripts RK and BH in equations 1 and 2 above. In a Ricker function, β_{RK} indicates the rate of decline in recruit abundance (*R*) as spawner abundance (*S*) increases. There are different algebraic ways of writing the Beverton-Holt function (*see* Equations 7-9 below), but as it is written in Equation 2, the β_{BH} parameter represents the asymptote of recruit abundances as *S* increases. Example Ricker and Beverton-Holt functions are plotted in Figure 2. The Ricker function (red) assumes that recruitment drops at very high spawner abundances while the Beverton-Holt function (blue) assumes that recruitment asymptotes as spawner abundance increases. The maximum sustained yield occurs at the spawner abundance (S_{MSY}) with the maximum vertical distance (dotted) between the model (colored) and population replacement (black). In the absence of fishing mortality, spawner abundance will reach equilibrium at N_{eq}.

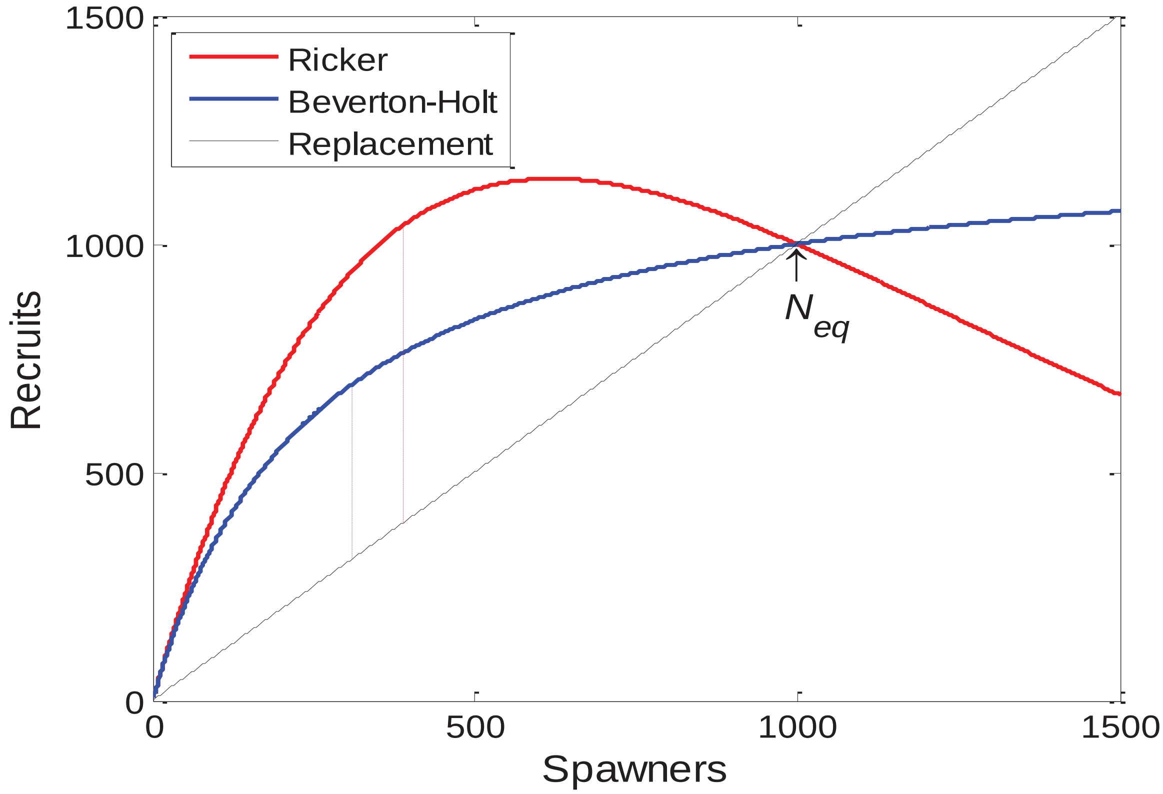

Figure 2. Example Ricker and Beverton-Holt functions.

Both functions in Figure 2 assume α = 5. Values of β were selected for each function so that they cross the replacement line (spawners = recruits) at a point corresponding to 1000 spawners and recruits•. The point where the function crosses the replacement line is denoted N_{eq}. If there was no harvest, then all recruits would become spawners, and abundance of spawners would be in equilibrium at N_{eq}.

Maximum sustained yield is a mathematical concept that can be derived from spawner-recruit relationships. It is the maximum number of animals that can be harvested such that the abundance of animals escaping harvest should produce an equally harvestable surplus of recruits. Graphically, the number of spawners that produces MSY (S_{MSY}) is the point on the x-axis of a spawner-recruit plot where there is maximum vertical distance between recruits and the replacement line (Figure F 2). Attempts to manage fish abundances to attain MSY have been implicated in the overfishing and collapse of many fisheries (Larkin 1977; Finley 2011). Although we do not advocate MSY-based fish management, we nonetheless compute S_{MSY} because we define critical conservation abundance as 50% of the 75th percentile of our estimate of S_{MSY} (*see* next section).

• Solve Equation 1 and Equation 2 for R=S, call the result Neq, then rearrange for β to obtain:

$$\beta_{RK} = \frac{\log(\alpha)}{N_{eq}}, \beta_{BH} = \frac{N_{eq}\alpha}{\alpha - 1}$$

For the Ricker and Beverton-Holt functions, S_{MSY} is, respectively:

$$S_{MSY} = \frac{\ln(\alpha)}{\beta_{RK}}\left(0.5 - 0.07\ln(\alpha)\right) \qquad \textit{Eq. 3}$$

$$S_{MSY} = \beta_{BH}\sqrt{\frac{1}{\alpha}} - \frac{\beta_{BH}}{\alpha}\ , \qquad \textit{Eq. 4}$$

Statistical Fits of Spawner-Recruit Relationships:

We used Bayesian methods to fit spawner-recruit functions for two reasons. First, as noted above, the spawner-recruit relationship is used to drive trans-generational population dynamics in the PVAs. An important aspect of any PVA is incorporation of statistical uncertainty in underlying parameters. Bayesian methods yield probability densities for the parameters of the spawner-recruit functions whereas non-Bayesian (i.e. "frequentist") methods yield point estimates. Thus, Bayes' method provides results that can be directly used to simulate parameter uncertainty in a PVA, which is one reason why Bayesian methods are appealing to conservation biologists (Wade 2002).

The second reason we used Bayesian methods to fit spawner-recruit relationships derives from our desire to characterize uncertainty in our estimate of S_{MSY}. Fifty percent of S_{MSY} has been used as a critically low abundance for triggering conservation action (AHSAC 2011). However, since S_{MSY} is never known perfectly, we define critically low abundance as 50% of the 75^{th} percentile of the estimate of S_{MSY}. This definition of a critically low abundance explicitly acknowledges uncertainty in S_{MSY}, and in response reduces the conservation risk associated with overestimating the true value of S_{MSY}. Stated another way, when ambiguity in the data increases, fish managers should respond more conservatively. Since S_{MSY} is computed from α and β parameters (*see* Equations 3 and 4), our assessment of statistical uncertainty in S_{MSY} depends on uncertainty in both of those parameters as well as their covariance. Quantifying uncertainty in S_{MSY} is therefore a very complex problem. Indeed, an exact analytical solution is not known to science. However, the Bayesian statistical paradigm offers a method for numerically estimating uncertainty in S_{MSY}. S_{MSY} can be computed as Markov Chain Monte Carlo (MCMC) methods sample from parameter posterior distributions (Haddon 2011). This yields a probability density of S_{MSY}, which, unlike results of frequentist methods, can be used to make probability statements about the value of S_{MSY}.

We modeled recruits as a log-normally distributed random variable. Specifically, we let

$$\log(R) \sim \text{Normal}(\mu,\tau) \qquad \textit{Distribution 1}$$

and, for the Ricker function, we get

$$\mu = \log(\alpha)+\log(S) - \beta_{RK}*S. \qquad \textit{Eq. 5}$$

If environmental covariates are included in the Ricker function, then we have:

$\mu = \log(\alpha)+\log(S) - \beta_{RK} * S + \gamma_1 Env_1 + \gamma_2 Env_2$. *Eq. 6*

We had difficulty getting Beverton-Holt models to converge, so we tried several parameterizations of the Beverton-Holt function. Specifically, we explored three:

$\mu = \log(\alpha)+\log(S) - \log(1+ \alpha / \beta_{BH} * S)$ {logarithmic version of Equation 2} *Eq. 7*

$\mu = \log(S) - \log(1/\exp(\alpha) + S/\exp(\beta_{BH}))$ *Eq. 8*

$\mu = \log(a)+\log(S) - \log(b + S)$. *Eq. 9*

As in Equation 6, we included environmental covariates to the Beverton-Holt function by simply including them as additive terms.

We used WinBUGS to carry out MCMC fitting of our spawner-recruit functions. Here, we follow WinBUGS distributional notation and note that τ in Distribution 1 is the precision of the normal distribution, where $\tau = 1/\sigma^2$. We first transform τ to the more familiar standard deviation, σ, and then let

$\sigma \sim \text{Uniform}(0,6)$ *Distribution 2*

We also tried the more common:
$\tau \sim \text{Gamma}(0.005, 0.005)$ *Distribution 3*

and found that the choice of prior parameterizations had little effect of our posterior results. For the intrinsic productivity parameter of both Ricker and Beverton-Holt functions, we specified noninformative priors with:

$\alpha \sim \text{Uniform}(0,10)$. *Distribution 4*

The prior distribution we used for β_{RK} is
$\beta_{RK} \sim \text{Normal}(\mu = 0.0000001, \tau = 0.005)$, *Distribution 5*

but we also explored the effects of assuming
$\beta_{RK} \sim \text{Uniform}(0.00001, 0.005)$. *Distribution 6*

We tried a host of different prior distributions for Beverton-Holt functions given in equations 7-9 because of the difficulty we experienced getting good convergence. Specifically, we tried normal, lognormal, and uniform priors in conjunction with several different non-informative parameterizations of these distributions. We did not obtain satisfactory fits and good evidence to support the use of Beverton-Holt functions for any of the populations we modeled. Thus, all of the spawner-recruit models presented in the Rogue Plan were derived from the Ricker function.

We always ran two Markov chains, and typically allowed them "burn-in" for 5,000 iterations. We obtained a total of 3,500 samples from each chain, after thinning the chains out to every 31st iteration. We plotted the "trace" of the resulting samples and computed Gelman-Rubin statistics to verify that the chains had properly mixed. For many models, including all those assuming a Beverton-Holt function, we did not obtain good evidence of convergence. If we were not able to remedy the convergence problem by adjusting the length of the burn-in and making minor adjustments to the prior distribution values and/or starting values, we concluded that the model was not well suited to the data and abandoned further attempts to fit the model.

We looked at the resulting parameter estimates to ensure that there were not any biologically unrealistic values. For example, if non-informative normal priors are used for the parameters in *Equation 9* (a Beverton-Holt function), then we frequently obtained huge uncertainty intervals that include negative values. Since the parameters of *Equation 9* represent non-negative entities, we did not entertain results with negative estimates. Using such the results of such models in our PVA would have carried absurd assumptions into our estimates of extinction risk. As noted above, we were unable to obtain satisfactory results for any population using the Beverton-Holt function.

We included environmental covariates (Tables 6 and 7) in the spawner-recruit modeling for two reasons. First, it provides an opportunity to possibly better quantify, the effects of primary factors that have been previously shown to limit recruitment within fall Chinook populations of the Rogue SMU. Second, scatterplots of our spawner and recruit data look nothing like the recruitment functions we attempted to fit. Including environmental covariates provides a means of getting better parameter estimates if the covariates can significantly account for some of the apparent randomness in the spawner-recruit data. Covariates were z-transformed so that values approximately come from a standard normal distribution in order to improve convergence performance. Descriptions of the chosen covariates, and the rationale associated with those choices, can be found in the Rogue Plan (*see* Spawner Abundance, page 60 and page 88).

We computed a Deviance Information Criterion (DIC) for each model. DIC is a Bayesian analogue of Akaike Information Criterion (AIC), which represents the tradeoff between model fit and complexity (Spiegelhalter et al. 2002). Much like AIC, a practical rule of thumb is that models receiving DIC scores within 1-2 of the "best" (i.e. smallest DIC) deserve consideration, whereas scores 2-7 greater than the "best" have considerably less support (Table 8). The fitted model parameters for the three best models, determined by DIC, are provided in Table 9.

Table 6. Estimated mean annual survival rates of coded-wire tagged juvenile spring Chinook salmon released at Cole M. Rivers Hatchery during September and October, 1980-2004 brood years.

Brood Year	Ocean Survival Rate[a]	Normalized Survival Rate
1980	0.0824	0.6143
1981	0.0646	0.1623
1982	0.0930	0.8837
1983	0.1449	2.2027
1984	0.0597	0.0369
1985	0.1144	1.4275
1986	0.0283	-0.7619
1987	0.0179	-1.0257
1988	0.0237	-0.8771
1989	0.0272	-0.7885
1990	0.0374	-0.5291
1991	0.1062	1.2174
1992	0.0859	0.7032
1993	0.0623	0.1039
1994	0.0107	-1.2084
1995	0.0544	-0.0986
1996	0.0104	-1.2169
1997	0.0914	0.8434
1998	0.1155	1.455
1999	0.0778	0.4972
2000	0.0735	0.3869
2001	0.0400	-0.4638
2002	0.0095	-1.2391
2003	0.0142	-1.1188
2004	0.0108	-1.2063

[a] *Estimated survival to age 2 in the ocean before the onset of any fishing mortality.*

Table 7. Indicators of freshwater environmental conditions experienced by naturally produced fall Chinook salmon in the Rogue River Basin, 1969-2009.

	July-Aug flow[a]		Peak flow[b]		Oct-Nov[c]	
Year	Mean	Normalized	Mean	Normalized	Mean	Normalized
1969/70	1,404	-1.309	59,200	0.931	107	-1.100
1970/71	1,130	-1.765	87,100	1.994	409	0.392
1971/72	2,191	0.004	82,500	1.818	253	-0.378
1972/73	1,799	-0.650	13,400	-0.814	96	-1.153
1973/74	932	-2.094	96,400	2.348	992	3.276
1974/75	2,045	-0.240	56,000	0.809	80	-1.231
1975/76	2,149	-0.068	26,800	-0.303	273	-0.278
1976/77	1,985	-0.341	1,950	-1.250	53	-1.367
1977/78	916	-2.121	44,600	0.375	363	0.165
1978/79	2,216	0.044	18,600	-0.616	76	-1.253
1979/80	2,130	-0.099	38,400	0.138	283	-0.231
1980/81	2,069	-0.200	16,100	-0.711	75	-1.259
1981/82	1,970	-0.365	78,700	1.674	612	1.396
1982/83	2,621	0.720	73,300	1.468	521	0.947
1983/84	2,966	1.294	32,500	-0.086	767	2.162
1984/85	3,409	2.031	19,000	-0.601	806	2.358
1985/86	2,405	0.359	32,400	-0.090	268	-0.303
1986/87	2,328	0.231	22,600	-0.463	307	-0.112
1987/88	2,282	0.155	16,400	-0.700	249	-0.397
1988/89	1,844	-0.575	25,300	-0.361	393	0.317
1989/90	2,464	0.458	13,700	-0.803	312	-0.084
1990/91	1,983	-0.344	18,300	-0.627	291	-0.189
1991/92	2,166	-0.039	7,590	-1.035	279	-0.247
1992/93	1,534	-1.092	20,800	-0.532	234	-0.472
1993/94	2,895	1.175	4,950	-1.136	306	-0.113
1994/95	1,441	-1.246	16,800	-0.684	237	-0.458
1995/96	2,767	0.963	28,700	-0.231	314	-0.077
1996/97	2,528	0.564	90,800	2.135	485	0.771
1997/98	2,707	0.862	39,000	0.161	369	0.194
1998/99	3,157	1.612	43,400	0.329	638	1.524
1999/00	3,419	2.048	21,200	-0.517	265	-0.320
2000/01	2,376	0.312	3,010	-1.210	272	-0.286
2001/02	1,434	-1.258	13,000	-0.829	94	-1.162
2002/03	1,911	-0.463	34,800	0.001	305	-0.118
2003/04	2,042	-0.245	20,770	-0.533	266	-0.312
2004/05	2,040	-0.248	24,600	-0.387	308	-0.106
2005/06	2,273	0.140	78,200	1.655	333	0.019
2006/07	2,627	0.729	29,400	-0.204	282	-0.236
2007/08	2,029	-0.267	22,400	-0.471	328	-0.006
2008/09	2,988	1.331	18,000	-0.639	274	-0.273

[a] *Mean flow (cfs) at Agness when juveniles reared in freshwater.*

[b] *Greatest mean daily flow (cfs) at Grants Pass when eggs and alevins incubated in the gravel.*

[c] *Mean flow (cfs) at Applegate town when adults migrated and spawned in the Applegate River.*

Table 8. Deviance Information Criterion scores for Ricker spawner-recruit models fitted to the populations of naturally produced Rogue fall Chinook salmon. The model with the lowest DIC is marked with an asterisk, along with models with similar (<2 difference) DICs.

Model Covariate(s)	Deviance Information Criterion score
Survival rate to age 2 for CWT-marked CHS (Table 6)	45.0*
Survival rate and Jul-Aug flow (Tables 6 and 7)	45.6*
Survival rate and peak flow (Tables 6 and 7)	46.4*
Jul-Aug rearing flow (Table 7)	51.7
None	53.9
Oct-Nov spawning flow (Table 7)	55.3
Peak flow during incubation (Table 7)	55.8

Table 9. Parameter values of the best fit Ricker stock-recruitment models built for the aggregated populations of naturally produced fall Chinook salmon in the Rogue River Basin, 1980-2004 brood years.

Model 1: lnRecruits = lnα + lnSpawners - β*Spawners + e1*survival rate		
Parameter	Coefficient	95%CI
Ricker α	4.07	2.11 – 6.28
Ricker β	1.57x10-5	9.76x10-6 – 2.24x10-5
e1[a]	0.37	0.14 – 0.61

Model 2: lnRecruits = lnα + lnSpawners - β*Spawners + e1*survival rate + e2*peak flow		
Parameter	Coefficient	95%CI
Ricker α	3.92	2.03 – 6.16
Ricker β	1.56x10-5	0.88x10-5 – 2.17x10-5
e1[a]	0.38	0.15 – 0.61
e2[b]	-0.10	-0.38 – -0.17

Model 3: lnRecruits = lnα + lnSpawners - β*Spawners + e1*survival rate + e2*summer flow		
Parameter	Coefficient	95%CI
Ricker α	3.93	2.03 – 6.01
Ricker β	1.56x10-5	0.95x10-5 – 2.20x10-5
e1[a]	0.32	0.08 – 0.57
e2[c]	0.16	-0.12 – 0.43

Population Viability Analysis

Population viability analysis (PVA) is a quantitative assessment of a population's risk of extinction (Morris and Doak 2002). Extinction risk can be characterized as either (1) mean time to extinction or (2) probability of extinction over some time horizon, typically 100 years. Here, we adopt the latter meaning of extinction risk. Since we are interested in the probability of extinction over 100 years, we require a principled, empirically-based method of simulating population dynamics through time. The purpose of this section is to describe how the spawner-recruit assessments are used to drive a PVA simulator.

A spawner-recruit curve is a model of trans-generational dynamics, and can therefore simulate population dynamics through time. However, to function as proper PVA, assumptions about (1) statistical uncertainty, (2) harvest, and (3) critically low abundance are needed. These three components of the PVA are addressed below.

Statistical Uncertainty: A deterministic spawner-recruit curve describes the number of recruits *expected* from some number of spawner. Clearly, however, observations of recruitment do not perfectly match this expectation. Rather, there is considerable deviation from this expectation every year. If these deviations are not incorporated into a simulation of a spawner-recruit relationship, then the simulated populations will converge on a stable age distribution and a stable spawner size (N_{eq} in Appendix Figure F-1). It is the principled incorporation of statistical uncertainty that distinguishes a PVA from other forms of population projection. To incorporate stochasticity, we simply compute the variance of the residuals in a spawner-recruit curve and then incorporate those deviations into the simulation. We also compute the lag-1 autocorrelation of the spawner-recruit residuals so that observed trends above or below the spawner-recruit curve are included in our simulations. With estimates of the variance and lag-1 autocorrelation on hand, a 100-year time series of simulated spawner-recruit residuals (or "environmental deviates") was computed using the formula:

$$\varepsilon_t = \rho\varepsilon_{t-1} + \sqrt{\sigma^2}\sqrt{1-\rho^2}\,N(0,1) \qquad \textit{Eq. 10}$$

where ρ is the estimate of the lag-1 autocorrelation of the residuals, σ^2 is the variance of the residuals and $N(0,1)$ is a standard normal deviate. At each time-step of the PVA, the corresponding ε_t was added to the expected number of recruits for a given number of spawners. This adds stochasticity to the otherwise deterministic spawner-recruit function. Note that this procedure assumes a homoscedastic distribution of random deviates.

As noted in the section above on statistically fitting spawner-recruit relationships, it is important for the PVA to incorporate uncertainty in parameter estimates. Indeed, a motivation for using Bayesian methods to fit the spawner-recruit curves is that it permits us to make probability statements about different spawner-recruit parameter values instead of only point estimates. For each population, we randomly drew values of the spawner-recruit curve from their posterior probability distributions 1,000 times. For each of these draws, residual variance and autocorrelation were recomputed, and then the PVA was repeated 50 times. Thus, the PVA was repeated a total of 50,000 times for each population. The frequency of extinction events among these 50,000 replicates is extinction risk as reported in the Rogue Plan. For the Rogue Plan extinction is defined as population abundance falling below a quasi-extinction threshold of 950 spawners for three consecutive years in the PVA simulations (*see* Viability of the Species Management Unit, page 111 in the Rogue Plan).

Harvest:
Chinook return to the spawning grounds at different ages, and these differences must be captured in the PVA. If **a** is a vector of the probabilities of spawning at different ages, then **a***R is a vector containing the number of fish that will return to the spawning grounds at different ages. The values for **a** represent the observed mean age composition of age 2-6 NP fall Chinook

spawners for the period of record (**a** = 0.207, 0.366, 0.324, 0.091, 0.011 for the Rogue populations and **a** = 0.072, 0.183, 0.572, 0.168, 0.004) for the coastal populations). Thus, under a scenario of no harvest for the Rogue populations, if brood year 1 produces 1,000 recruits (*R*), then the model estimates that 366 of these fish will return to spawn three years later. These fish would spawn with 414 two-year olds if brood year 2 produces 2,000 recruits; again under the assumption of no harvest. The total number of spawners in a given simulated year is obtained by summing the products of recruits produced in previous years and the probabilities of spawning at different ages; and then removing harvested fish. Specifically, spawner abundance on a given year (S_t) is:

$$S_t = \sum_{i=1}^{6} R_{t-i} a_i (1 - H) \qquad \textit{Eq. 11}$$

where *H* is the estimated brood harvest rate. Note that *H* and **a** were needed to construct the original spawner-recruit dataset. Harvested fish were included in the number of recruits in the original spawner-recruit dataset. *Equation 11* removes the same number of fish before they spawn, which reflects the real-world harvest process.

Simulations of the Rogue populations incorporated brood harvest rates that were scaled to simulated values of population abundance. This procedure was implemented because brood harvest rates of fall Chinook in the ocean fisheries are dependent on the stock size of Klamath fall Chinook, and the abundance of Klamath fall Chinook and Rogue NP fall Chinook are correlated (*see* Comparisons to Other Populations, page 69 in the Rogue Plan). A function for brood harvest rates was incorporated into the PVA to replicate this process (Figure 3).

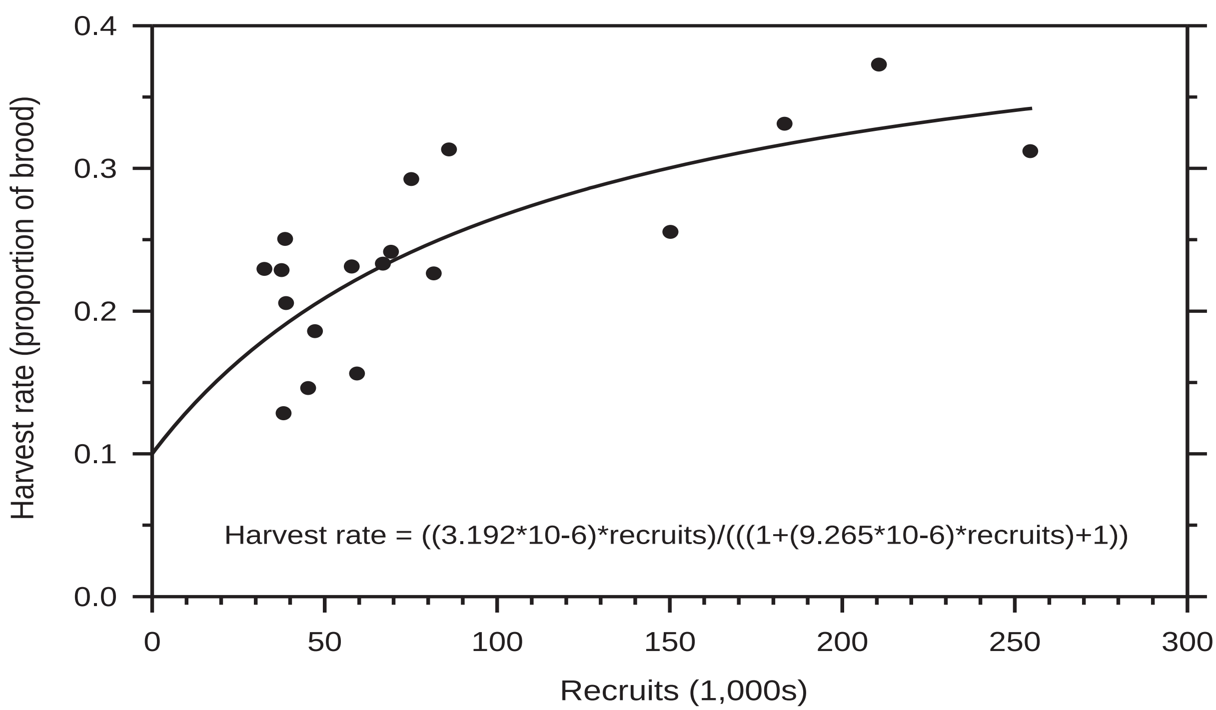

Figure 3. Harvest rate function used in the population viability assessment of naturally produced fall Chinook salmon within aggregated populations of the Rogue River Basin. The function also incorporates a baseline harvest rate of 0.10 for freshwater harvest.

Estimated S_{MSY}:

Population models developed for the Rogue aggregate populations were used to generate estimates of S_{MSY}. To account for model uncertainty, modeled estimates of S_{MSY} were bootstrapped by re-sampling the spawner and recruit data 1,000 times and refitting the best recruitment model. The upper 75th percentiles of these bootstrapped estimates were chosen as the most appropriate metrics for the numerical component of conservation criteria for Rogue fall Chinook spawner escapements.

Model point estimates, and bootstrap estimates of the 75th percentile, for the number of spawners estimated for maximum sustained yield within independent populations of naturally produced Rogue fall Chinook salmon. The table also conveys a rounded value proposed for the MSST conservation criteria (50% of the 75th percentile of S_{MSY}). Values included in the table below reflect estimates generated from population models that included smolt survival rates and summer flow as environmental covariates.

	S_{MSY} estimate		Proposed Conservation
	Point Estimate	75th Percentile	Criteria (MSST)
Rogue Aggregate	34,992	36,880	18,440

Discussion

Subpart D of the federal Magnuson-Stevens Act includes National Standard 1 (§600.310). This standard describes conservation and management measures designed to prevent overfishing while achieving, on a continuing basis, the optimum yield from each fishery managed by the federal government. Status determination criteria to determine overfished stocks are to be based on minimum stock size thresholds and must be expressed in terms of spawning biomass or other measure of reproductive potential, and should equal whichever of the following is greater: one-half (50%) of the spawning stock needed to maintain MSY, or the minimum stock size at which rebuilding to attain MSY would be expected to occur within ten years.

In 2011, the Pacific Fisheries Management Council (PFMC) adopted Amendment 16 to the Pacific Coast Salmon Plan. Included in Amendment 16 are status determination criteria related to minimum stock size thresholds (MSST) and these criteria options (Ad Hoc Salmon Amendment Committee 2011) functionally serve the same purpose as conservation criteria included in the Rogue Plan. ODFW agrees with the Ad Hoc Salmon Amendment Committee (2011) conclusion that a definition of MSST as 0.5* S_{MSY} is appropriate because salmon populations are relatively productive compared to other managed fish species. Consequently, this guidance was used to identify appropriate conservation criteria for fall Chinook spawning escapements in the Rogue SMU.

Table 10. Current conservation objective and reference points governing harvest control rules and status determination criteria for SOCC.

Stocks in Fishery	Conservation Objective	S_{MSY}	MSST	MFMT (F_{MSY})	ACL
Southern Oregon	Unspecified portion of an aggregate 150,000 to 200,000 natural adult spawners for Oregon coast streams measured by 60-90 fish per mile in index streams (Thompson 1977 and McGie 1982). ODFW developing specific conservation objectives for spring and fall stocks that may be implemented without plan amendment upon approval by the Council.	60 fish per mile in index streams	30 fish per mile in index streams	78% Proxy (SAC 2011a)	Component stock of SONC complex; ACL indicator stock is KRFC

Recommendations

ODFW proposes that the current conservation objective and reference points shown in Table 10 be replaced with those shown in Table 11.

Table 11. Proposed conservation objective and reference points governing harvest control rules and status determination criteria for SOCC.

Stocks in Fishery	Conservation Objective	S_{MSY}	MSST	MFMT (F_{MSY})	ACL
Southern Oregon	At least 41,000 naturally produced adults passing Huntley Park in the Rogue River annually to meet S_{MSY}. MSST would be reached at 20,400 measured at Huntley Park. S_{MSY} and MSST values must be inflated by to account for pre-spawn losses between Huntley Park and spawning areas (ODFW 2013).	36,880	18,440	78% Proxy (SAC 2011a)	Component stock of SONC complex; ACL indicator stock is KRFC

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