Analyses to Support a Review of an ESA Jeopardy Consultation on Fisheries Impacting Lower Columbia River Tule Chinook Salmon

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Introduction

The purpose of this report is to summarize information that NOAA's National Marine Fisheries Service (NMFS) used for evaluating alternative harvest impact limits on the ESA-listed Lower Columbia River Chinook salmon (LCR Chinook) Evolutionarily Significant Unit (ESU). LCR Chinook exhibit life-history types based on adult migration timing, including early fall runs ("tules"), late fall run ("brights") and spring-runs (reviewed by Myers 1998). The ESU is subdivided into 32 populations, some of which existed historically but are now extinct (Myers et al. 2006) (Figure 1, Table 1). Of the different life-history types, the tules are subject to the highest level of harvest (Kope 2005), and are the sole focus of this report.

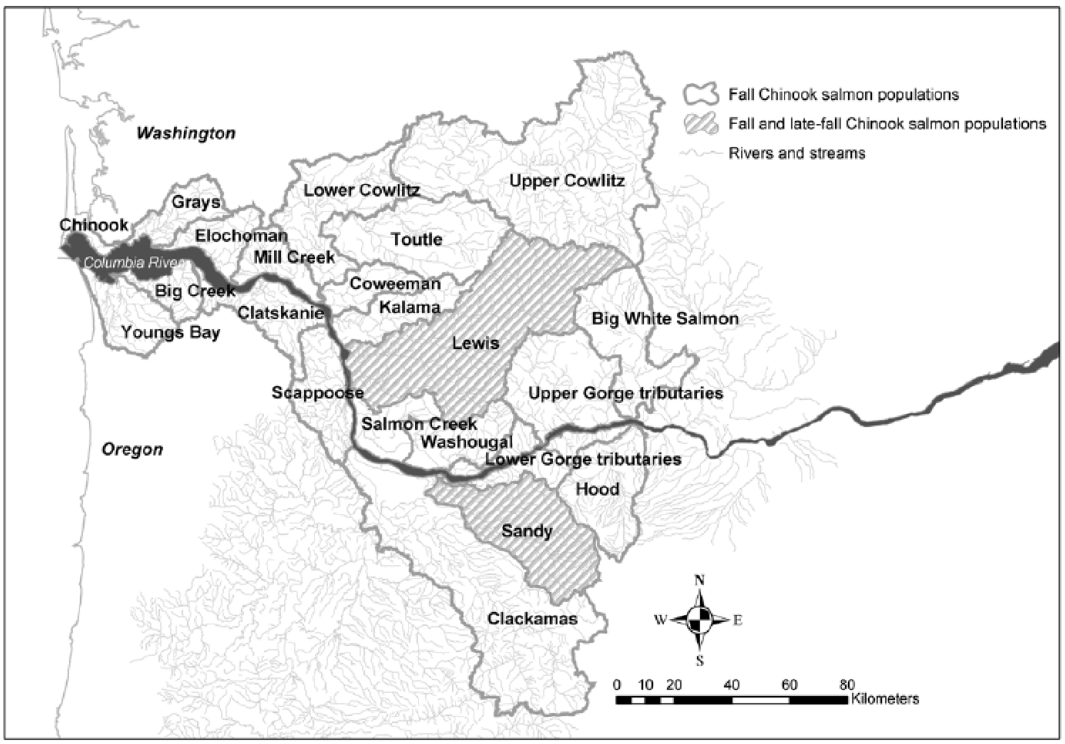

Figure 1 – Fall-run (tule and bright) Chinook salmon populations in the Lower Columbia River identified by the Technical Recovery Team. Reproduced from Myers et al. (2006).

NMFS has used a variety of approaches for evaluating the effects of harvest actions on ESA listed salmon (NMFS 2004). For LCR tules, NMFS has previously used an analytical approach (Viability Risk Assessment Procedure – VRAP; NMFS (2001)) that involves calculating a "rebuilding exploitation rate" (RER). The RER for a specific population is defined as the maximum exploitation rate that will result in a low probability of the population falling below a specified lower abundance threshold, and a high probability that the population will exceed an upper abundance threshold over a specific time period (discussed in more detail below).

In past biological opinions regarding the effects of harvest on LCR tule Chinook, NMFS used the VRAP approach to calculate an RER of 49% for the Coweeman River (Figure 1) tule population (NMFS 2002; NMFS 2005). This RER was used as the jeopardy standard for the tule component of the LCR Chinook ESU from 2002 to 2006. Prior to the start of the 2006 preseason planning process NMFS indicated, in its annual guidance letter to the Pacific Fishery Management Council (Council), its intention to review the 49% standard prior to the 2007 season (Lohn and McInnis 2006). Such a review was called for in the Interim Regional Recovery Plan for the Lower Columbia River (LCFRB 2004), and NMFS concurred that, after five years, a review was warranted. The review provided an opportunity to update the earlier analysis, and consider more recent information developed by the Willamette/Lower Columbia Technical Recovery Team (WLC TRT) (McElhany et al. 2003; McElhany et al. 2004; McElhany et al. 2006), and through the recovery planning process (LCFRB 2004) and other sources.

NMFS initiated the review by forming a Work Group in the summer of 2006 including representatives from the NMFS' Northwest Region and Science Center, and the Washington Department of Fish and Wildlife. The goal of the Work Group was to provide a more comprehensive review of the status of LCR tule populations, and review considerations related to harvest. The Coweeman population was used in the earlier VRAP analysis because it was apparent at the time that it was an important natural-origin population, and because the necessary data were readily available. The Coweeman then served as a harvest indicator stock for natural-origin LCR tule populations. In the years since the previous analysis, state agencies and the WLC TRT have compiled or summarized additional data on other Lower Columbia Chinook populations, and the Interim Recovery Plan also developed a recovery scenario that described which populations should be prioritized for recovery. The Work Group therefore determined that it was important to extend its analyses to additional tule populations in the ESU.

In its initial evaluation, the Work Group found it useful to divide the tule populations in the ESU into three categories (Table 3): 1) medium-to-large natural populations without large scale hatchery programs and relatively few hatchery strays, 2) natural populations with escapements that are dominated by large, in-basin hatchery programs, and 3) small natural populations with limited data. The Work Group decided to initially focus its quantitative analyses on category 1 populations, for the following reasons. First, a lack of data proved to be a significant impediment to the analysis and the category 1 populations tended to have the highest quality data. Information on age structure and the number of hatchery strays in the spawning escapement, both essential to the analyses, was particularly lacking for category 3 populations. Second, the Work Group was concerned that the RER concept is difficult to meaningfully apply to natural populations whose spawning escapements consist largely of stray hatchery fish, a topic that is discussed in the Discussion Section of this report.

The Work Group spent most of their time on the review and analysis of data for the three category 1 tule populations: Coweeman, Grays, and Lewis. These were all identified as high priority populations in the Interim Recovery Plan, and appeared to be subject to relatively little hatchery straying. Information related to escapement, hatchery stray rates, exploitation rates and age composition are all important to the analyses, and WDFW updated all of these estimates for the three category 1 populations. Data for other populations were compiled from existing sources, but not otherwise reviewed in detail.

As stated above, NMFS' intended to use the results from this review to reconsider the 49% jeopardy standard. To be useful for the preseason planning process, NMFS had to provide its guidance by early March, 2007. By February, 2007 the Work Group was not finished with a comprehensive review of the available data, but had made significant progress in updating and analyzing information related to the three category 1 populations, and summarizing information available for other populations. NMFS used this information for developing its guidance to the Council for the 2007 season. The guidance was subsequently considered in the associated biological opinion on Council fisheries (Lohn 2007).

The primary purpose of this report is to summarize information developed by the Work Group through February 2007. The Work Group has not met since February 2007, but some additional work as been done by NMFS since then. The more recent work has focused primarily on exploring alternative methods for estimating exploitation rates, looking at the affect of adding a marine survival covariate to the estimation of a spawner recruit relationship, conducting analyses on category 2 and 3 populations, and writing this report. Because there is interest in understanding the information available in February 2007 when decisions were made, this report distinguishes, as best we can, information available at the time from that developed more recently. Although the Work Group is not currently meeting, NMFS expects to continue its analyses of harvest actions including those affecting LCR tule Chinook.

Table 1 -- The ecological zones and populations for the Lower Columbia River Chinook salmon ESU (Jim Myers, NWFSC, updated from LCFRB 2004).

Fall Run (tule)	
Coastal Tribs	
1	Youngs Bay
2	Grays River (w/ Chinook River)
3	Big Creek
4	Elcohoman River
5	Clatskanie River
6	Mill Creek (w/ Germany and Abernathy Cr)
7	Scappoose Creek
Western Cascade Tributaries	
8	Upper Cowlitz River
9	Lower Cowlitz River
10	Toutle River
11	Coweeman River
12	Kalama River
13	Lewis River
14	Salmon Creek
15	Clackamas River
16	Washougal River
17	Sandy River
Gorge Tributaries	
18	Lower Gorge
19	Upper Gorge
20	Hood River
21	Big White Salmon

Late Fall Run	
Western Cascade Tributaries	
22	Lewis River
23	Sandy River

Spring Run	
Western Cascade Tributaries	
24	Tilton River
25	Upper Cowlitz River
26	Cispus River
27	Toutle River
28	Kalama River
29	North Fork Lewis River
30	Sandy River
Gorge Tributaries	
31	Big White Salmon River
32	Hood River

Methods

Summary of overall approach to analyzing the effects of harvest on population viability

Our overall approach consisted of the following steps: 1) estimate the current intrinsic productivity (theoretical adult recruits per spawner at zero spawners based on spawner/recruit curves) and capacity for (maximum recruits and/or number of spawners that produce the maximum recruits) each category 1 population over the period for which data are available, 2) quantitatively evaluate the maximum rate of harvest that either allows the population to meet its viability criteria, or allows the population to meet some interim (lower) criteria that still allows rebuilding of the population, 3) qualitatively evaluate how the results from the category 1 populations can be extrapolated to category 2 and 3 populations, and 4) evaluate the results in the context of the ESU-level viability criteria.

For the quantitative analyses, we use two complementary modeling approaches: the viability curve approach using the Salmon Population Analyzer (SPAZ) computer program developed by the WLC TRT (available at http://www.nwfsc.noaa.gov/trt/spaz.cfm), and the Viability and Risk Assessment Procedure (VRAP) modeling approach used for previous Biological Opinions (NMFS 2002; NMFS 2005). Both approaches are described in greater detail later in this section.

Summary of ESU and population viability criteria

The WLC TRT has developed a hierarchical approach for determining ESU-level viability criteria (Figure 2). Briefly, an ESU is divided into populations (*sensu* McElhany et al. 2000). The risk of extinction of each population is evaluated, taking into account population-specific measures of abundance, productivity, spatial structure and diversity. Populations are then grouped into ecologically and geographically similar *strata*, which are evaluated on the basis of population status. In order to be considered viable, a stratum generally must have at least half of its historically present populations meeting their population-level viability criteria (this is only an approximation -- see McElhany et al. 2006 for details). Finally, the ESU-level viability criteria require that each of the ESU's strata be viable. The tule fall Chinook populations and strata are listed in Table 2.

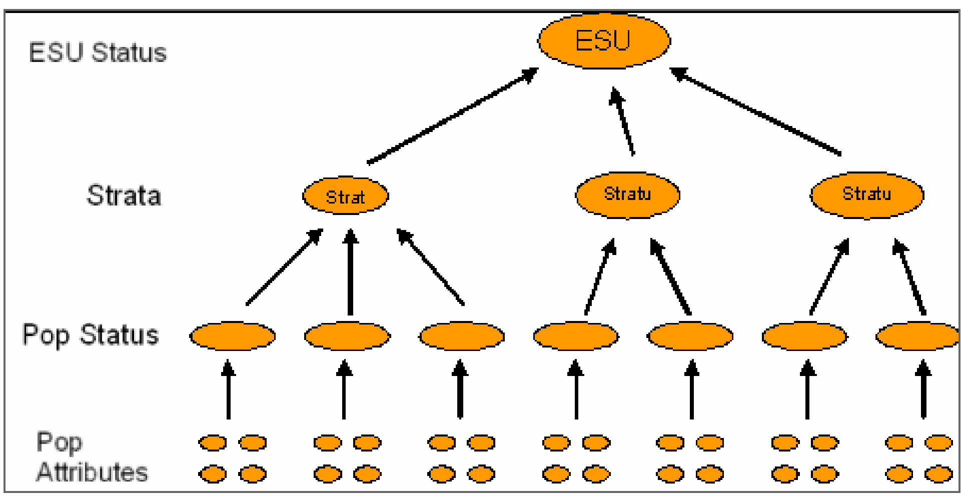

Figure 2 -- Hierarchical approach to ESU viability criteria

The LCFRB has used the TRT viability criteria to define recovery goals (LCFRB 2004). For tule Chinook, the LCFRB has identified six "primary" populations on which to focus recovery efforts (Table 2). NMFS endorsed the LCFRB plan as an Interim Recovery Plan. The LCFRB plan made certain assumptions about Oregon populations, but the State of Oregon has not yet developed formal recovery goals for Oregon populations. Once Oregon completes the recovery planning process for the Oregon side of the ESU, the two states' plans will need to be combined and reconciled. A final comprehensive recovery plan for the Willamette and Lower Columbia River ESUs will then follow.

The LCFRB plan summarizes information related to the status of the tule populations. That information is summarized in Table 3 and Table 4. The table notes provide a brief explanation for information provided, but see the LCFRB plan for more detailed explanations.

Table 2 -- Lower Columbia tule Chinook population and basin information.

Strata	State	Population	LCFRB Goal	size category
Coast Fall	WA	Grays	P	S/M
	WA	Elochomann	P	S
	WA	Mill/Abernathy/Germany	C	S
	OR	Youngs Bay	S	S
	OR	Big Creek	S	M
	OR	Clatskanie	P	S
	OR	Scappoose	S	S
Cascade Fall	WA	Lower Cowlitz	C	L
		Coweeman	P*	S/M
		Toutle	S	M
		Upper Cowlitz	S	M
		Kalama	P	M
		Lewis/Salmon	P	S/M
		Washougal	P	M
	OR	Sandy	S	M
		Clackamas	C	M
Gorge Fall	WA	Lower Gorge	S	S
		Upper Gorge (includes Wind)	C	S
		Big White Salmon	C	S
	OR	Hood	S	S

Notes:
<u>LCFRB Goal</u>: P=primary population/low risk; P* = primary population/very low risk; C = contributing population/moderate risk; S = sustaining population/maintain current status. Based on the TRT criteria, lower risk < 5% risk of extinction in 100 years; very low risk < 1% risk of extinction in 100 years, and moderate risk < 25% in 100 years.
<u>Size category</u> is used to determine the appropriate quasi-extinction threshold for population modeling. Size categories for the Oregon populations are taken directly from the WLC-TRT recommendations (McElhany et al. 2006) and are based on historical km of spawning habitat (<50, 50-150, >150). Size categories for the Washington populations were determined by the work group, based on analogies to the Oregon populations. L = Large = QET of 250/year for four years; M = medium = QET of 150/year for four years; S = small = QET of 50/year for four years.

Table 3 -- Lower Coumbia River tule Chinook population summary of escapement information.

Strata	State	Population	5 year geomean natural origin spawners	5 year % hatchery origin spawners	natural spawner trend	LCFRB abund. goal (natural spawners)	Current EDT equil. Abund.	EDT produc.	category
Coast Fall	WA	Grays	206	16%	1.16	1,400	550	3.5	1
	WA	Elochomann	132	69%	1.01	1,400	2,076	3.1	3
	WA	Mill/Abernathy/Germany	461	77%	0.95	1,100	1,366	3.4	3
	OR	Youngs Bay	no data	no data					3
	OR	Big Creek	no data	no data					3
	OR	Clatskanie	38	15%	1.04				3
	OR	Scappoose	no data	no data					3
Cascade Fall	WA	Lower Cowlitz	2593	41%	1.19	2,300	8,873	5.9	2
		Coweeman	927	7%	1.00	3,600	1,839	4.3	1
		Toutle	0			1,000	4,370	3.2	NA
		Upper Cowlitz	0				3,097	2.5	NA
		Kalama	341	93%	0.85	1,300	1,581	3.3	2
		Lewis	729	6%	1.05	2,900	1,380	3.5	1
		Washougal	1788	61%	1.01	5,800	1,624	3.8	2
	OR	Sandy	183	3%					3
		Clackamas	40	no data					3
Gorge Fall	WA	Lower Gorge	no data	no data		100	124		3
		Upper Gorge (includes Wind)	311	47%		700	954		3
		Big White Salmon	544	81%		900			3
	OR	Hood	no data	no data			1337	1.46	3

Notes:
5 year geometric mean of natural origin spawners is based on the estimated natural origin spawning numbers from 2001-2005, provided in Appendix A.
5 year % hatchery origin spawners is the average percentage of hatchery origin spawners on the natural spawning grounds from 2001-2005, again using the data in Appendix A.
Natural spawner trend was calculated as the exponential of the slope of the regression line of log transformed natural origin spawners from 1990 – 2005. Values >1 indicate an increasing trend.
LCFRB abundance goal is the average natural origin spawning goal from the LCFRB Recovery Plan.
Current EDT equilibrium abundance is the estimated average abundance each population would maintain in the absence of harvest based on current habitat conditions. Values are from the LCFRB Plan.
EDT productivity is the estimated intrinsic productivity for a Beverton-Holt spawner-recruit curve based on current habitat conditions and was obtained from the FCFRB Plan.
Category 1 = natural populations with data verified by the Work Group; Category 2 = relatively large natural population with high fraction hatchery fish; Category 3 = small natural populations with uncertain data.

Table 4 – Lower Columbia River tule Chinook population status summaries (from LCFRB).

Strata	State	Population	Persistance	Spatial Structure	Diversity	Habitat

Coast Fall	WA	Grays	1.5	4	2.5	1.5
	WA	Elochomann	1.5	3	2	2
	WA	Mill/Abernathy/Germany	1.8	4	2	2
	OR	Youngs Bay				
	OR	Big Creek				
	OR	Clatskanie				
	OR	Scappoose				
Cascade Fall	WA	Lower Cowlitz	1.7	4	2.5	1.5
		Coweeman	2.2	4	3	2
		Toutle	1.6	3	2	1.75
		Upper Cowlitz	1.2	2	2	2
		Kalama	1.8	4	2.5	2
		Lewis/Salmon	2.2	4	3	2
		Washougal	1.7	4	2	2
	OR	Sandy				
		Clackamas				
Gorge Fall	WA	Lower Gorge	1.8	3	2.5	2.5
		Upper Gorge	1.8	2	2.5	2
		Big White Salmon	1.7	2	2.5	1.5
	OR	Hood				

Notes:
Summaries are taken directly from the LCFRB Recovery Plan. All are on a 4 point scale, with 4 being lowest risk and 0 being highest risk.

Persistence: 0 = extinct or very high risk of extinction (0-40% probability of persistence in 100 years); 1 = Relatively high risk of extinction (40-75% probability of persistence in 100 years); 2 = Moderate risk of extinction (75-95% probability of persistence in 100 years); 3 = Low (negligible) risk of extinction (95-99% probability of persistence in 100 years); 4 = Very low risk of extinction (>99% probability of persistence in 100 years)
Spatial Structure: 0 = Inadequate to support a population at all (e.g., completely blocked); 1 = Adequate to support a population far below viable size (only small portion of historic range accessible); 2 = Adequate to support a moderate, but less than viable, population (majority of historical range accessible but fish are not using it); 3 = Adequate to support a viable population but subcriteria for dynamics or catastrophic risk are not met; 4 = Adequate to support a viable population (all historical areas accessible and used; key use areas broadly distributed among multiple reaches or tributaries)
Diversity: 0 = functionally extirpated or consist primarily of stray hatchery fish; 1 = large fractions of non-local hatchery stocks; substantial shifts in life-history; 2 = Significant hatchery influence or periods of critically low escapement; 3 = Limited hatchery influence with stable life history patterns. No extended intervals of critically low escapements; rapid rebounds from periodic declines in numbers; 4 = Stable life history patterns, minimal hatchery influence, no extended intervals of critically low escapements, rapid rebounds from periodic declines in numbers.
Habitat: 0 = Quality not suitable for salmon production; 1 = Highly impaired; significant natural production may occur only in favorable years; 2 = Moderately impaired; significant degradation in habitat quality associated with reduced population productivity; 3 = Intact habitat. Some degradation but habitat is sufficient to produce significant numbers of fish; 4 = Favorable habitat. Quality is near or at optimums for salmon.

Data

Estimates of annual spawner escapements, hatchery fraction, and age structure were obtained from several sources, including the Washington Department of Fish and Wildlife, Oregon Department of Fish and Game, and the WLC TRT. For the three category 1 populations (Coweeman, Grays and Lewis), estimates were further reviewed and updated to adjust for changes in survey and sampling methods (Dan Rowling, WDFW, as described in Attachment 1). The Work Group was confident that the data for these three populations were sufficiently accurate to be used in the VRAP and SPAZ modeling frameworks. For the remaining populations, we used the estimates compiled by the WLC TRT (McElhany et al. 2004; McElhany et al. 2003; McElhany et al. 2006), also provided in Appendix A. The Work Group was less confident in the data for the remaining populations. All the escapement estimates included fish age 2 and up, and all analyses assumed an unbiased estimate of age composition. The data are provided in Appendix A and are available electronically at upon request.

Hatchery Composition

Estimates for the hatchery composition of spawning escapements were provided by WDFW or the WLC TRT for most populations. In some cases a constant value was given for all or most of the years, and in other cases the work group elected to use an average value for all years due to low confidence in the annual values. These estimates were generally made from coded-wire-tag recoveries on the spawning grounds from a limited number of recent years. Due to low tagging rates and variable sampling effort, the hatchery fraction estimates are generally quite uncertain.

Age Composition

Estimates of age composition of the natural origin spawners were provided for all populations, although for a few populations, these were just an average applied to all years. Scale sampling was conducted by WDFW on the spawning grounds with age samples available starting in 1988 for the Coweeman, 1977 for the Grays and Lewis, and 1991 for the other populations having data. Estimates for the Grays and Lewis were updated January 5, 2007 with a memo from WDFW. For the Coweeman, age estimates originally provided for years prior to 1988 were based on hatchery estimates and were, therefore, not used. In order to generate cohort run reconstructions for the natural populations, we required information on the age composition of the wild portion of the natural escapement. In cases where age data were missing for some years, we used an estimation method that starts with using the available data to determine an average cohort age distribution and then adjust the proportion of each age within a cohort according to the relative total natural origin escapements for the calendar years in which the cohort returned to the spawning grounds. An iteration procedure was then used to determine the cohort return sizes that resulted in the closest approximation to the annual escapements given the adjusted cohort age distributions. See Appendix B for a more detailed description of this method.

Age Specific Harvest Rates

Harvest rates estimated from indicator hatchery stocks were used to determine fishing mortalities of the natural populations being analyzed. In the context of this report, harvest rate is defined as the proportion of an available stock removed or killed by fishing during a specific time period.

Harvest rates used in these analyses are age- and fishery specific; fisheries are grouped into two categories, pre-terminal and terminal. Pre-terminal fisheries are assumed to harvest a mixture of mature and immature fish and are sometimes referred to as mixed-maturity fisheries, and terminal fisheries are assumed to harvest only mature fish that would have spawned in the current year and are sometimes referred to as mature fisheries.

Age specific harvest rates were estimated by cohort analysis of coded wire tag (CWT) recoveries of indicator hatchery fish using methods employed by the Pacific Salmon Commission's joint Chinook Technical Committee (CTC 2005). This procedure reconstructs the cohort recursively, starting from the oldest age and working backward. For a given age and year, the abundance of CWT recoveries is calculated as:

$$N_{a,t} = MC_{a,t} + TC_{a,t} + S_{a,t} + N_{a+1,t+1}/(1-m_{a+1}) \qquad \textbf{Equation 1}$$

where N is marine abundance at the beginning of the year (after natural mortality at age and before any fisheries), MC is pre-terminal catch including incidental mortality, TC is the terminal catch including incidental mortality, S is spawning escapement, and m is the natural mortality rate, with a denoting age (2,..,5), and t denoting year. Age specific natural mortalities (m_a) are constants, being 0.50, 0.40, 0,30, and 0,20 for ages 2-5, respectively. Terminal harvest rates (th) and pre-terminal harvest rates (mh) are defined in terms of the reconstructed cohort as:

$$th_{a,t} = TC_{a,t} /(TC_{a,t} + S_{a,t}) \qquad \textbf{Equation 2}$$

and

$$mh_{a,t} = MC_{a,t} / N_{a,t} \qquad \textbf{Equation 3}$$

respectively.

For the purpose of natural population run reconstruction, the harvest rates for a natural population are assumed to be the same as its CWT hatchery indicator stock. Four hatchery indicator stocks were available for the LCR tules: the Cowlitz (1977-2001 BYs), Washougal (1973, 1976-1987, 1989-2000 BYs), Grays (1974-1982, 1984-1985, 1988-1996 BYs), and Big Creek (1976-1981, 1986-1988, 1990-1997, 1999-2001 BYs). The Cowlitz was considered the best indicator for the Coweeman, since the Coweeman River is tributary of the Cowlitz River. The other hatchery indicator stocks had gaps in the time series of data, but since the harvest rate estimates for the major age returns (3 and 4) were similar, we decided to use a composite estimate. The three Washington hatchery stocks (Cowlitz, Washougal, and Grays) were combined to represent Washington tule populations and all four hatchery stocks were combined to represent Oregon tule populations. The composite estimates were made by averaging over stocks for each year; this resulted in a full data series of harvest rate estimates from 1973-2001 for both composite stocks. Annual patterns of harvest rates were similar for all stocks as seen in the example graph for the mixed-maturity fishery age 4 harvest rate shown in Figure 3 and for the mature terminal fishery in Figure 4. All ages showed similar patterns, although there is more variability in the age 2 and age 5 harvest rate estimates as those have a much lower level of harvest.

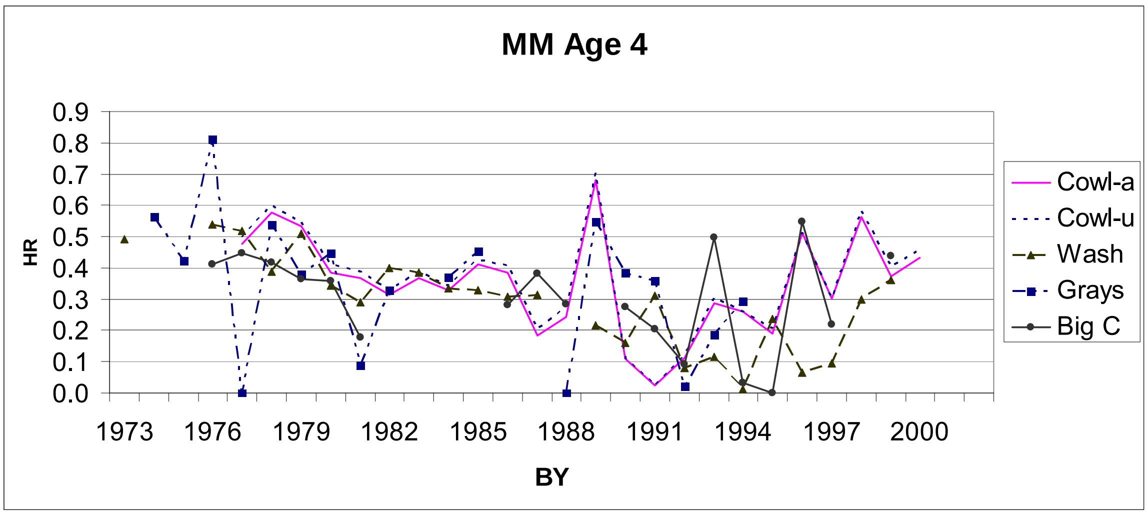

Figure 3. The comparison of harvest rates for the indicator stocks for the mixed maturity fishery age 4. Included are the Cowlitz stock, adjusted and unadjusted, the Washougal, Grays, and Big Creek.

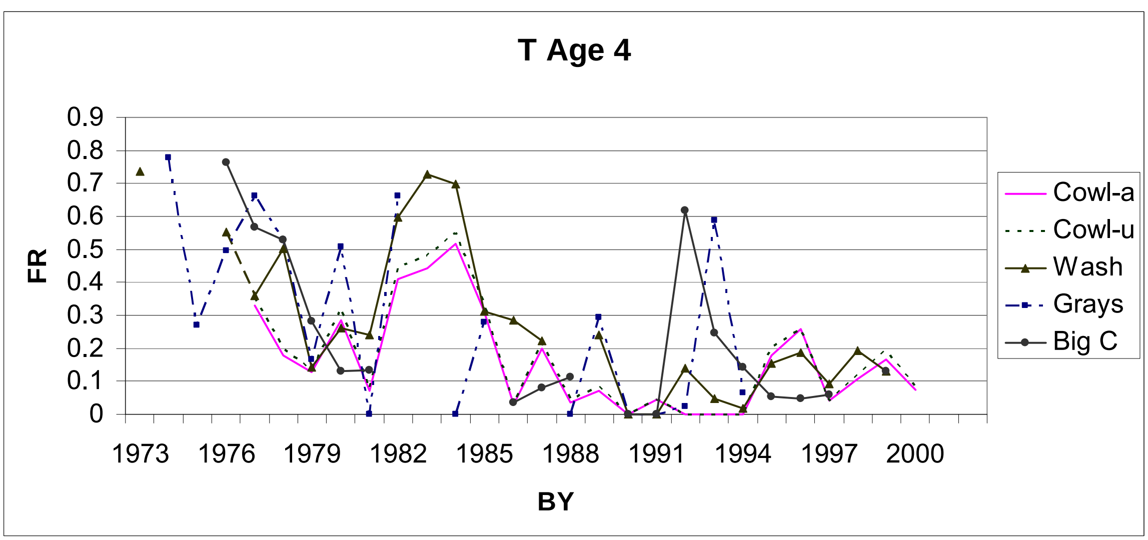

Figure 4. The comparison of harvest rates for the indicator stocks for the mature or terminal fishery age 4. Included are the Cowlitz stock, adjusted and unadjusted, the Washougal, Grays, and Big Creek.

Harvest rates calculated from Cowlitz Hatchery CWT recoveries were used to represent the natural Coweeman tule stock. The Cowlitz has a substantial recreational fishery that is not sampled for CWTs and the Coweeman River is closed to salmon fishing. To correct for the effects of this tributary harvest on hatchery escapement, the CWT recoveries from spawning escapement were divided by (1-tributary harvest rate). The correction did not result in a large change as can be noted in Figure 4 for the Cowlitz adjusted and unadjusted curves.

Harvest rate estimates for the three indicator stocks are provided in Appendix A.

Covariate data

Marine survival indices for the LCR tule populations were obtained from the CWT analyses of CWT hatchery indicator stock recoveries and the total releases of CWT fish by brood year. Marine survival is estimated as the total number of CWT estimated to be in the AEQ returns divided by the total releases of CWT fish. The index values for a specific hatchery are derived by dividing the annual marine survival by the average marine survival; this allows us to average indices over several hatchery stocks to make a composite marine survival index. For these analyses the survival indices from Cowlitz hatchery were used for the Coweeman population and the composite indices from the Cowlitz, Washougal, and Grays were used for the other natural populations. The pattern of survival was noted to be very similar for the three hatchery stocks. The indices for the Cowlitz and the composite stock are shown in Figure 5.

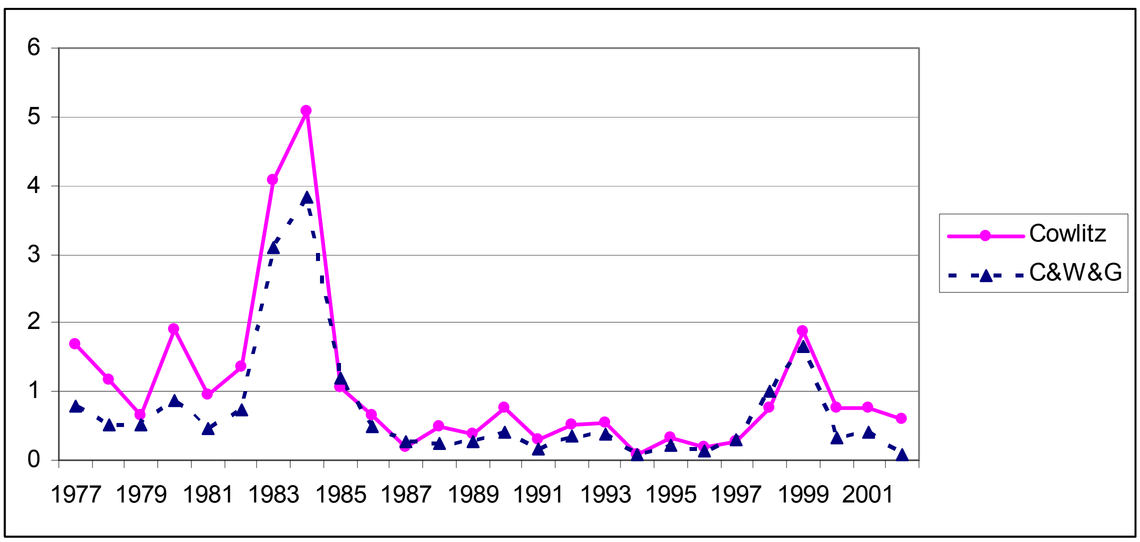

Figure 5. Marine survival indices for the Cowlitz and composite (Cowlitz&Washougal&Grays) hatchery indicator stocks.

Management error data

In the Columbia River, net and sport fisheries are planned preseason based on anticipated abundance of fall stocks (river mouth return), hatchery broodstock and natural escapement needs, and ESA impact constraints. Generally, most often when these objectives are not met, it is due to inaccuracies in the return abundance forecasts.

Therefore, we suggest that the 'Management error' input to the VRAP model be represented by the difference between predicted and actual total river mouth returns of tule stocks. The gamma distribution parameters have been calculated based on errors in return predictions for Columbia River tule stocks. The stocks used were the 'Lower River Hatchery' (LRH) stock, composed of all hatchery and natural origin tules returning to areas below Bonneville dam, and 'Spring Creek Hatchery' tules, composed of all hatchery and natural origin tules returning to areas above Bonneville Dam. Data used to compute the parameters of the Gamma distribution were taken from Preseason Report I, Stock Abundance Analysis for 2006 Ocean Salmon Fisheries (PFMC, 2006) and are presented here in Table 5 and Figure 6.

Table 5. Ratio of predicited to observed return of LCR tules.

	April Pre/PostSeason		
Year	LRH	SPR	
1984	0.87	0.57	
1985	0.78	1.12	
1986	1.12	0.98	
1987	0.87	1.01	
1988	0.80	0.49	
1989	0.74	0.86	
1990	1.09	1.25	
1991	1.17	1.17	
1992	1.94	1.40	
1993	1.49	1.08	
1994	0.87	1.56	
1995	0.91	0.67	
1996	0.64	1.07	
1997	1.20	0.94	
1998	0.50	0.70	
1999	0.96	1.22	
2000	0.98	1.31	
2001	0.32	0.50	
2002	0.85	0.85	
2003	0.75	0.56	
2004	0.73	0.86	
2005	1.00	1.24	
			combined
average	0.935	0.973	0.954
variance	0.111	0.092	0.100
Gamma a	7.853	10.304	**9.137**
Gamma b	0.119	0.094	**0.104**

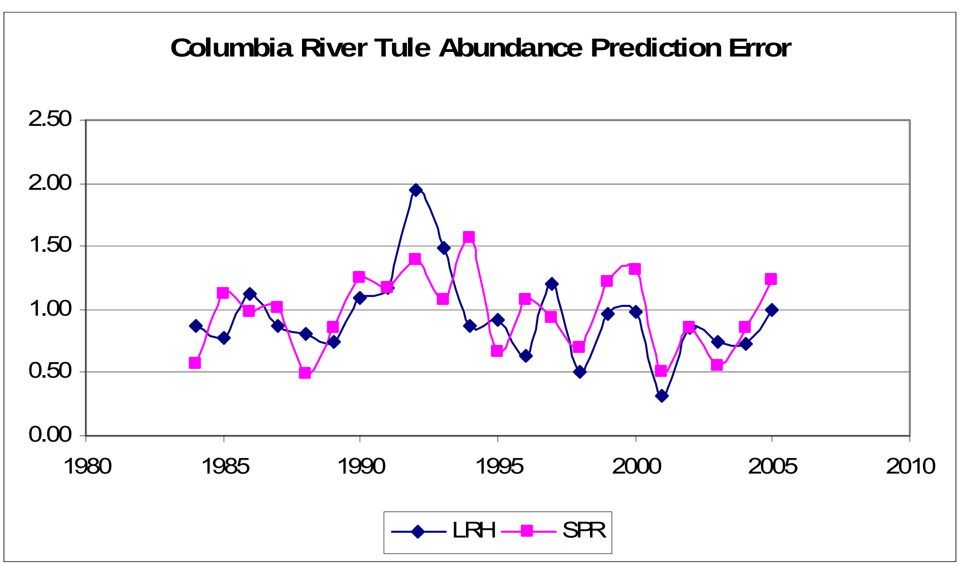

Figure 6. Time series of run size prediction errors for two LCR tule stock groups.

Run reconstruction, recruits, and recruits per spawner

Cohort run reconstruction is used to estimate cohort recruits from each spawning escapement. The Abundance and Productivity (A&P) Excel tables are used to consolidate the input data, reconstruct cohorts, and provide the resulting estimates in the form needed for both the VRAP and SPAZ analyses; they are available in electronic form upon request. The cohort reconstruction procedure, like the CWT analyses that produces harvest rates, reconstructs the cohort recursively, starting from the oldest age and working backward:

$$N_{a,t} = MC_{a,t} + TC_{a,t} + S_{a,t} + N_{a+1,t+1}/(1-m_{a+1}) \qquad \textbf{Equation 4}$$

But in this case, not having catch estimates for the natural populations based on observation, we use the harvest rates from the CWT cohort analysis to calculate catch:

$$TC_{a,t} = S_{a,t}\,\{th_{a,t}/(1-th_{a,t})\} \qquad \textbf{Equation 5}$$

$$MC_{a,t} = \{S_{a,t}/(1-th_{a,t}) + N_{a+1,t+1}/(1-m_{a+1})\}\ \{mh_{a,t}/(1-mh_{a,t})\} \qquad \textbf{Equation 6}$$

Age specific maturation rates (p_a) and adult equivalent (AEQ) factors for the populations, by cohort, may then be calculated from the run reconstruction.

$$p_{a,t} = (TC_{a,t} + S_{a,t})/(N_{a,t} - MC_{a,t}). \qquad \textbf{Equation 7}$$

$$AEQ_{a,b} = \sum_{j=a}^{5}\{p_{j,b}\prod_{i=a}^{a-1} m_{i+1}(1-p_{i,b})\} \qquad \textbf{Equation 8}$$

Where *a* is the age for the brood year (or cohort) *b*.

AEQ recruits (R) can then be estimated as:

$$R_b = \sum_{a=2}^{5}\{(AEQ_{a,b} * MC_{a,b}) + TC_{a,b} + S_{a,b}\} \qquad \textbf{Equation 9}$$

Productivity[1] in terms of the number of recruits per spawner (*R/S*) for a cohort can be calculated as the brood year recruits defined above divided by the parent spawners for the cohort.

1 Productivity may also be expressed as the slope of a spawner/recruit curve at the origin. This is referred to as intrinsic productivity and is a theoretical value related to the current condition of a population and its habitat. See sections on spawner/recruit analysis.

Actual spawners, hatchery contributions, potential spawners, and age distribution for the actual plus potential spawners were used as input for SPAZ, and are available electronically upon request.

AEQ Exploitation Rates

The calculation of the AEQ exploitation rate for the natural population is not necessary for the estimation of RERs but may easily be done from the cohort run reconstruction results. It is useful to estimate the AEQ exploitation rates for both a current check on how recent AEQ exploitation rates compare with the RER and for post-season compliance purposes. Since the exploitation rates are based on cohort run reconstruction, one will not see the results for a cohort until 4 or 5 years after the spawning year; however, the exploitation rate for a given cohort reflects fishing patterns for 2-5 years after the parent spawning year.

The total brood year AEQ exploitation rate for a brood year *t* over all fisheries and ages is the reduction in potential spawning escapement attributable to fishing mortality over the life of a brood:

$$AEQER_t = 1 - \sum_{a=2}^{5} S_{a,t+a} \Bigg/ N_{2,t+2} \sum_{a=2}^{5} \{ p_{a,t+a} \prod_{i=2}^{a-1} m_{i+1} (1 - p_{i,t+i}) \}$$

Equation 10

where *p* is the maturation rate as defined in Equation 7 and *m* is the natural mortality.

SPAZ/Viability Curve Approach

The majority of the methods are described in detail in the most recent WLC-TRT viability report (McElhany et al. 2006), which builds on the basic framework in the NOAA Technical Memoradum on Viable Salmonid Populations (VSP (McElhany et al. 2000)). Only a summary is provided here.

The abundance and productivity evaluation is predicated on two basic observations: 1) all else being equal, a larger population is less likely to go extinct than a small one and 2) all else being equal, a more productive population is less likely to become extinct than a less productive population. Productivity in this context refers to “intrinsic” productivity, and is an indication of a population’s “resilience” or tendency to return to high abundance if perturbed to low abundance. Intrinsic productivity can be broadly defined as the number of offspring per parent when there are few parents.

In the context of the viability curve analyses, “extinction” is defined by a “quasi-extinction threshold” (QET), which for the three category 1 populations was defined by the WLC TRT to be less than either 50 or 150 spawners/year for four consecutive years, depending on whether the populations are believed to "small" or "medium." References to "extinction risk" therefore relate to the probability of being below the QET value for four consecutive years sometime in the 100 year simulation, and should not be interpreted to represent probabilities of literal extinction. The rationale for a QET is that, in most cases, there is very little information on spawner/recruit relationship at spawning densities lower than those corresponding to the QET's, genetic viability decreases at low populations levels, and that risks and uncertainty to population viability increase sharply below the QET level. See McElhany (2006, especially pp. 22-24 and Appendix E of that document) for more discussion of QETs.

Viability Curves

As described in the viability reports (McElhany et al. 2004; McElhany et al. 2006), a viability curve describes a relationship between population abundance, productivity and extinction risk (Figure 7). The viability curve approach was developed as a way of graphically expressing recovery goals and assessing current status. These curves can be generated for various AEQ exploitation rates. Abundance is then defined as the maximum recruits (adult returns) that can be produced prior to harvest and productivity is the intrinsic productivity calculated from a spawner-recruit curve (slope of curve at origin). All abundance and productivity combinations defined by any of the extinction curves in Figure 3 indicate the same level of risk. Populations with productivity and abundance combinations above and to the right of the viability curve have a lower extinction risk than those that fall on the curve, while those below and to the left have a higher risk than those that fall on the curve.

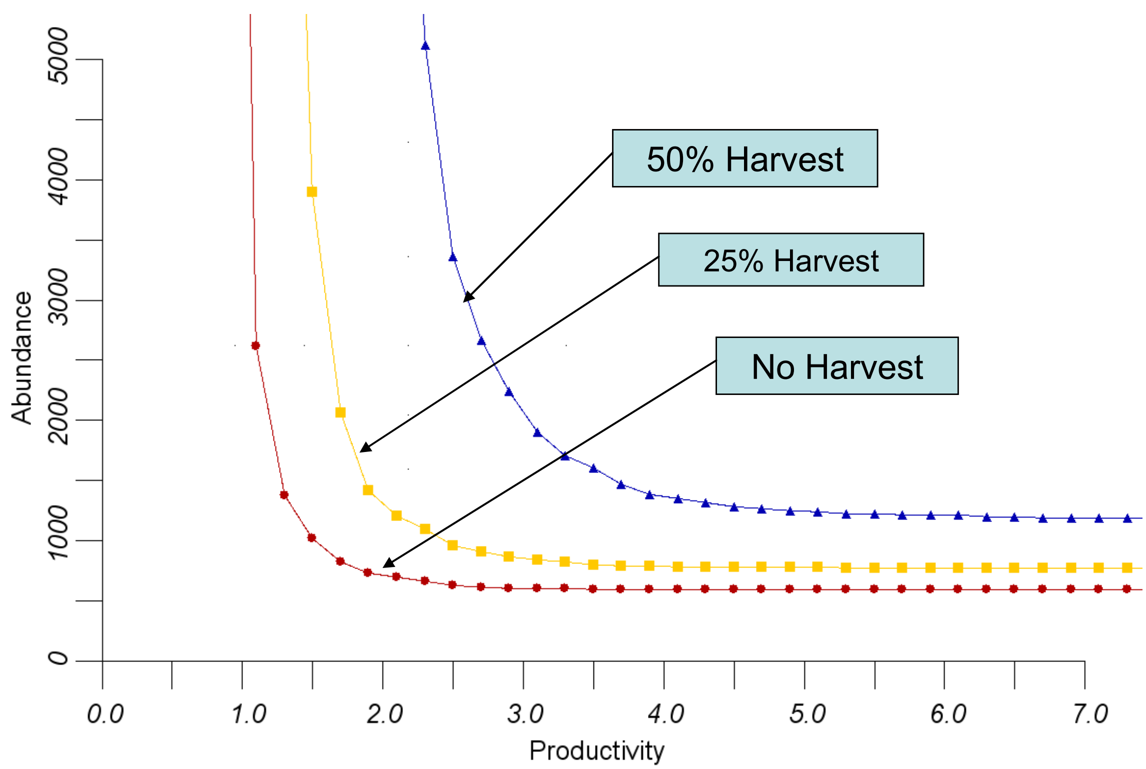

Figure 7 -- Viability curves showing the relationship between harvest levels, productivity, abundance and extinction risk. The curves are extinction risk isoclines with all points on the curve indicating

parameter combinations with the same extinction risk (i.e.., a 5% probability of declining to a QET of 150 fish in 100 years). Each of the curves indicates a different harvest level. As harvest level increases the abundance and/or productivity must also increase to maintain the same extinction risk. This set of isoclines is based on average LCR Chinook annual variability, autocorrelation and age structure and was used to evaluate all three of the category 1 LCR tule populations in this analysis.

The mathematical model that was used to construct the viability curve was the Hockey-stick spawner-recruit function with autocorrelated error (McElhany et al. 2006). To assess the status of a population relative to the curve, the MeanRS Method used in the TRT's viability assessments (McElhany et al. 2004; McElhany et al. 2006). In the MeanRS method, the productivity is estimated as the geometric mean recruits per spawner for broods where the number of spawners is less that than the median number of spawners in the time series (i.e the lowest half of the brood sizes). The MeanRS abundance estimate is the calculated as the geometric mean recruit abundance over the time series. The MeanRS method was selected over simply fitting a hockey-stick model because of concerns the quality of the fit (see McElhany et al. 2006). A key issue in the analysis is how we incorporate uncertainty in the estimation of a population's current abundance and productivity values. We can not precisely estimate abundance and productivity, so we present probability contours for these parameters (Figure 8). See McElhany et al. (2006) for a detailed description of the methods (see especially p. 12-39 for a description of how current population status is assessed relative to the viability curves).

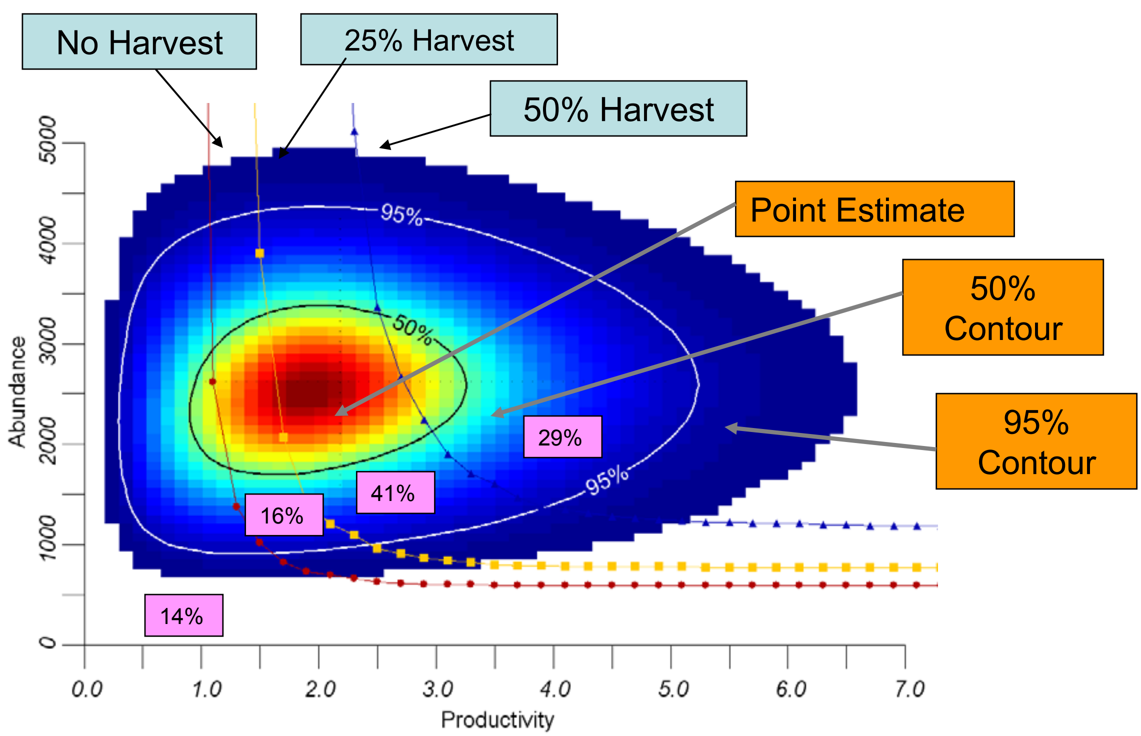

Figure 8 -- Example of current status contours of a theoretical population combined with the viability curves from Figure 3. The contours and color gradations indicate the probability the

population has a particular combination of abundance and productivity given the error in the estimation of these statistics from the population specific data. The values shown in pink are the estimated probability that the population statistics lie between two viability curves (e.g., the probability that the population's actual current abundance and productivity levels lie between the no harvest and 25% harvest curve is 16%). We can also calculate the probability that the population could persist with a harvest rate greater than some threshold. For example, the probability of being viable with a harvest rate of 25% is estimated at 70% (i.e. 41% + 29%). In this figure, the point estimate of the population indicates that a harvest level between 25 and 50% would be viable at the extinction risk defined by this set of viability curves. However, there is some chance (14%) that the population would not be viable even with no harvest or, alternatively, there is some chance (29%) that the population would be viable at harvest rates greater than 50% (i.e. there is a great deal of uncertainty).

If a population has a high intrinsic productivity, the viability curve analysis may indicate that the population is expected to be viable at a very low abundance level. If average abundance is too low, however, the population may be at risk from phenomena that are not incorporated into the SPAZ analyses. For example, very small populations are more likely to suffer from inbreeding depression or may not be able to maintain sufficient genetic variability for long-term survival (reviewed by McElhany et al. 2000). The results of the SPAZ analyses should be therefore interpreted carefully, and in some cases it may be appropriate to specify an abundance floor for viability that is higher than the viability curve alone would indicate. The WLC-TRT has suggested that for a viable Chinook salmon population, the minimum long-term geometric mean naturally origin abundance should be greater than 500 for a small population, 600 for medium population and 1,000 for a large population. These minimum abundance criteria are in addition to viability curve analysis criteria (i.e., both criteria need to be met – not just one or the other).

Rebuilding Exploitation Rates

The goal of the VRAP (NMFS 2001) analysis is to estimate a rebuilding exploitation rate (RER), which is the highest allowable ("ceiling") exploitation rate for a population, under current conditions, that meets certain conservation criteria. The RER is defined as the rate that would result in escapements unlikely to fall below a critical lower escapement threshold and likely to grow or remain above an upper escapement threshold. The VRAP approach uses a stochastic modeling framework to project the population into the future under current productivity and capacity conditions.

The VRAP model takes into account uncertainty in the data and the natural conditions in several ways. In particular, uncertainty can be introduced at three levels: the fit of data to the spawner-recruit model, uncertainty in fishery management process, and variability in environmental conditions. Results are expressed as the percentage of time the population 1) goes below a quasi-extinction threshold, 2) goes below a critical escapement level at any time during the run, and 3) achieves an average abundance above the upper escapement threshold for the last five years of the simulation. The later two conditions are used in developing RERs.

Lower Escapement Threshold

The lower escapement threshold (LET) represents a boundary below which uncertainties about population dynamics increase substantially. In the rare cases where sufficient stock-specific information is available, we can use the population dynamics relationship to define this point. Otherwise, we use alternative population-specific data, or general literature-based guidance. NOAA Fisheries has provided some guidance on the range of critical thresholds in its document, *Viable Salmonid Populations* (McElhany et al. 2000). The VSP guidance suggests that effective population sizes of less than 500 to 5,000 per generation, or 125 to 1,250 per annual escapement, are at increased risk. For the LCR tule analyses, we set both the quasi-extinction threshold and the LET at the QET level recommended by the WLC TRT (50-150 spawners/year for four consecutive years, for the three populations analyzed). Formally, the LET was set to the QET+1, since the VRAP program requires LET to be larger than the QET.

Upper Escapement Threshold

The purpose of the upper escapement threshold (UET) is to ensure that the analyzed action is consistent with population recovery, which the Work Group interpreted to mean consistent with a trend of increasing spawning escapements. According to NMFS policy, RERs are not intended to be the sole means of achieving recovery (NMFS 2004), but are meant to encourage increases in spawning abundance while other recovery actions are put in place to rebuild the productivity and capacity of a population. NMFS intends that as the productivity and/or capacity conditions for the population change, the UET should be changed to reflect the change in conditions (NMFS 2004). The expectation is that as recovery actions are implemented, habitat conditions will improve and population productivity and/or capacity will increase. Thus, the UET serves as a step in the progression to recovery, which will occur as the contributions from all recovery management sectors are realized.

For these analyses, we explored a variety of UETs, including the LCFRB's viability goals, the mean natural spawner escapement, the mean natural origin spawner escapement, and the spawner escapement that would produce maximum sustained yield (MSY) associated with the spawner/recruit function used in the VRAP analysis. None of the options for selecting a UET is ideal, but the modeling requires that some choice be made. After some consideration, the Work Group chose to use as an UET the larger of either the estimated spawning abundance that would produce modeled MSY, or the average natural origin spawning abundance over the time series analyzed. The spawning abundance that is expected to produce MSY are often highly uncertain due to poor model fits and are often lower than estimates of the current abundance. Setting the UET at the mean natural origin spawning abundance implies that the resulting RER will allow for population growth (since the UET must be met 80% of the time).

Parameter estimation

There are two phases to the process of determining an RER for a population. The first, or model fitting phase, involves using data from the target population itself, or a representative indicator population, to fit a spawner-recruit relationship representing the performance of the population over the time period to be analyzed. Population performance is modeled as

$$R = f(S, e) \qquad \textbf{Equation 11}$$

where S is the number of fish spawning in a single return year, R is the number of adult equivalent recruits[2], and **e** is a vector of environmental, density-independent correlates of brood year survival.

Several data sets are necessary for this: a time series of natural spawning escapement, a time series of total recruitment by cohort, and time series for the environmental correlates of survival. In addition, one must assume a functional form for f , the spawner-recruit relationship. Given the data, one can numerically estimate the parameters of the assumed spawner-recruit relationship to complete the model fitting phase.

The data are fitted using three different models for the spawner recruit relationship: the Ricker (Ricker 1975), Beverton-Holt (Ricker 1975), and Hockey stock (Barrowman and Meyers 2000). The simple forms of these models were augmented by the inclusion of environmental variables correlated with brood year survival.

To estimate the parameters of the spawner-recruit function, an excel spreadsheet model, first developed by Jim Scott (WDFW) and adapted by Norma Sands (NMFS), was used. This model is referred to as the Dynamic Model. This model, instead of using the estimated total recruitment as input, uses the harvest rates used to estimate total recruitment and the resulting maturation rates from the run reconstruction such that the model can estimate both total recruits and age specific progeny spawners for each cohort. The age specific progeny spawners can be rearranged to estimate the predicted calendar year escapement. The model then utilizes the solver utility in EXCEL to iteratively solve for parameters that minimize the error between predicted calendar year escapement and the observed calendar year escapement. Minimizing the error between predicted and estimated calendar year escapement is used since the escapements are closer to being observed data than recruits, which, are based on harvest rate estimates times escapement estimates. The error in the estimated recruit values includes the error contained within the escapement estimates as well as the error associated with the harvest rate estimation. This fitting procedure results in larger error estimates in predicted recruits v. estimated recruits than if the spawner-recruit function parameters were estimated by minimizing on recruits; this reflects some of the uncertainty or error that exists in our estimates of recruits, that would have been ignored if minimization of error were based on recruits. Noting the difficulty the EXCEL solver sometimes has in finding the global minimum, a

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[2] Equivalently, this could be termed "potential spawners" because it represents the number of fish that would return to spawn absent harvest-related mortality.

macro has been written to run the model from 66 different starting values for the 2-4 parameters being estimated and then choosing the results with the minimum error.

Input for the Dynamic Model includes natural spawners, natural origin spawners, one or two covariates, harvest rates by age and two fisheries (mixed-maturity and mature), and maturation rates, which are used for the cohort run reconstruction process. In addition, the estimate of AEQ recruits from the A&P run reconstruction is needed in the input to the Dynamic Model, so that the mean squared error of the recruit estimates may be calculated. The cohort run reconstruction done in the Dynamic Model differs from the procedure used in the A&P cohort run reconstruction in that the Dynamic Model starts with estimating the age two cohort size from the spawner recruit parameters and the AEQ factors and uses the harvest and maturation rates to calculate, starting with age 2 fish, the fishing mortality and terminal run size; the resulting escapement values are, thus, calculated values.

The model iteratively picks stock-recruit function parameters, predicts the cohort size at age 2-years (adult equivalent (AEQ) recruits divided by age-2 AEQ), and uses the age specific harvest rates and maturation rates to conduct a cohort run reconstruction. The resulting predicted age specific escapements are rearranged to construct the predicted calendar year escapements and these escapements are tested against the observed escapements for minimizing error. Spawner recruit parameters are iteratively tested for those producing the minimum error. Predicted AEQ recruits are compared with the AEQ recruits estimated in the A&P Tables to determine the process error (mean square error) to supply the VRAP model.

Equations for the three models are as follows:

$$R = (aS\mathrm{e}^{-bS})(M^{c}\mathrm{e}^{dF})$$ **[Ricker]** **Equation 12**

$$R = (S/[bS + a])(M^{c}\mathrm{e}^{dF})$$ **[Beverton-Holt]** **Equation 13**

$$R = (\min[aS, b])(M^{c}\mathrm{e}^{dF})$$ **[hockey stick]** **Equation 14**

In the above, M is the index of marine survival and F is a freshwater environmental co-factor.

Spawner/recruit functions are then estimated numerically from 1) natural spawners, 2) harvest rates (% of standing stock taken by fishery), and 3) maturation rates for the natural origin population using an iterative process of minimizing error between the predicted values and the 'observed' data.

Test statistics for model selection include mean square error (MSE), F-statistic, probability of data fitting the model (p), and the AIC statistic. The F-statistic is calculated as:

$$F = \frac{R^2/(k-1)}{(1-R^2)/(N-k)}$$ **Equation 15**

Where R is the correlation between the predicted and observed values, N is the number of years used in the calculations, and k is the number of parameters being estimated.

The p-value associated with the model is derived from the F-distribution function of EXCEL for the statistic F and the degrees of freedom k-1, and N-k.

The Akaike's information criterion (AIC) is used to help select the best model for the data (Burnham & Anderson 1998). In our case, we are choosing both between the three spawner-recruit functions and between using the marine survival covariate or not. The AIC statistic is calculated as

$$AIC = N\ln(SSE/N) + 2(k+1)$$ **Equation 16**

where the number of parameters being estimated included the error parameter as well as the number of parameters used in fitting the spawner-recruit function. The best fit is indicated by the lowest AIC value. Therefore, one can calculate delta AIC such that the minimum AIC is represented by zero: Δ AIC = AIC - min(AIC). Models with Δ AIC < ~2 are generally considered to explain the data equally well (Burnham & Anderson 1998).

Estimating RERs using VRAP

The second, or projection phase, of the analysis uses the VRAP model and involves using the fitted model in a Monte Carlo simulation to project the probability distribution of the near-term future performance of the population assuming that current conditions of productivity continue. Besides the fitted values of the parameters of the spawner-recruit relationships, one needs estimates of the probability distributions of the variables driving the population dynamics, including the process error (including first order autocorrelation) of the spawner-recruit relationship itself and each of the environmental correlates. Also, since fishing-related mortality is modeled in the projection phase, one can estimate the distribution of the deviation of actual fishing-related mortality from the intended ceiling. This is termed "management error" and its distribution, as well as the others are estimated from available recent data.

For each trial RER the population is repeatedly projected for 3000 runs over 25 years. The 25 year time horizon was chosen to represent a short-to-medium term time horizon. In particular, it provides ~4-5 generations for populations to meet the upper threshold targets, and allows for estimation of the fairly near term probability of going below the lower threshold. In practice, however, the resulting RERs are not very sensitive to small changes in the time horizon (N. Sands, unpublished data). From the simulation results we computed the fraction of years in all runs where the escapement is less than the critical escapement threshold and the fraction of runs for which the mean escapement over the final five years in the simulation is greater than the upper escapement threshold. Trial RERs for which the first fraction is less than 5% and the second fraction is greater than 80% were considered acceptable for use as ceiling exploitation rates for harvest management.

Additional parameters and assumptions in the VRAP model include:

- A hatchery effectiveness of 1 is used (hatchery fish on the spawning grounds have equal reproductive success as wild fish).
- The quasi-extinction level (QET) was set to both 50 and 150 to capture the range associated with either “small” or “medium” sized populations.
- For each population, age data specific to the naturally spawning component of the population was used in the run reconstructions. Because sample sizes for the age data were not available, they were assumed to be reasonably large and that the age information was known without error.
- Age 2 fish are included in the escapement estimates provided and the spawner recruit analyses include age 2 fish in both the spawning escapement and the recruits.

Results

Current Status of Populations

The current status of WLC Chinook populations has recently been evaluated by the WLC TRT and the LCFRB. In particular, for Washington populations, Chapter 1 of Appendix E of the LCFRB Recovery Plan provides a thorough summary of population status, and the TRT has recently evaluated the status of Oregon populations. Table 4 summarizes the current status of each population based on these assessments, and Figure 9 illustrates the recent abundance trend in each population that has time series data available.

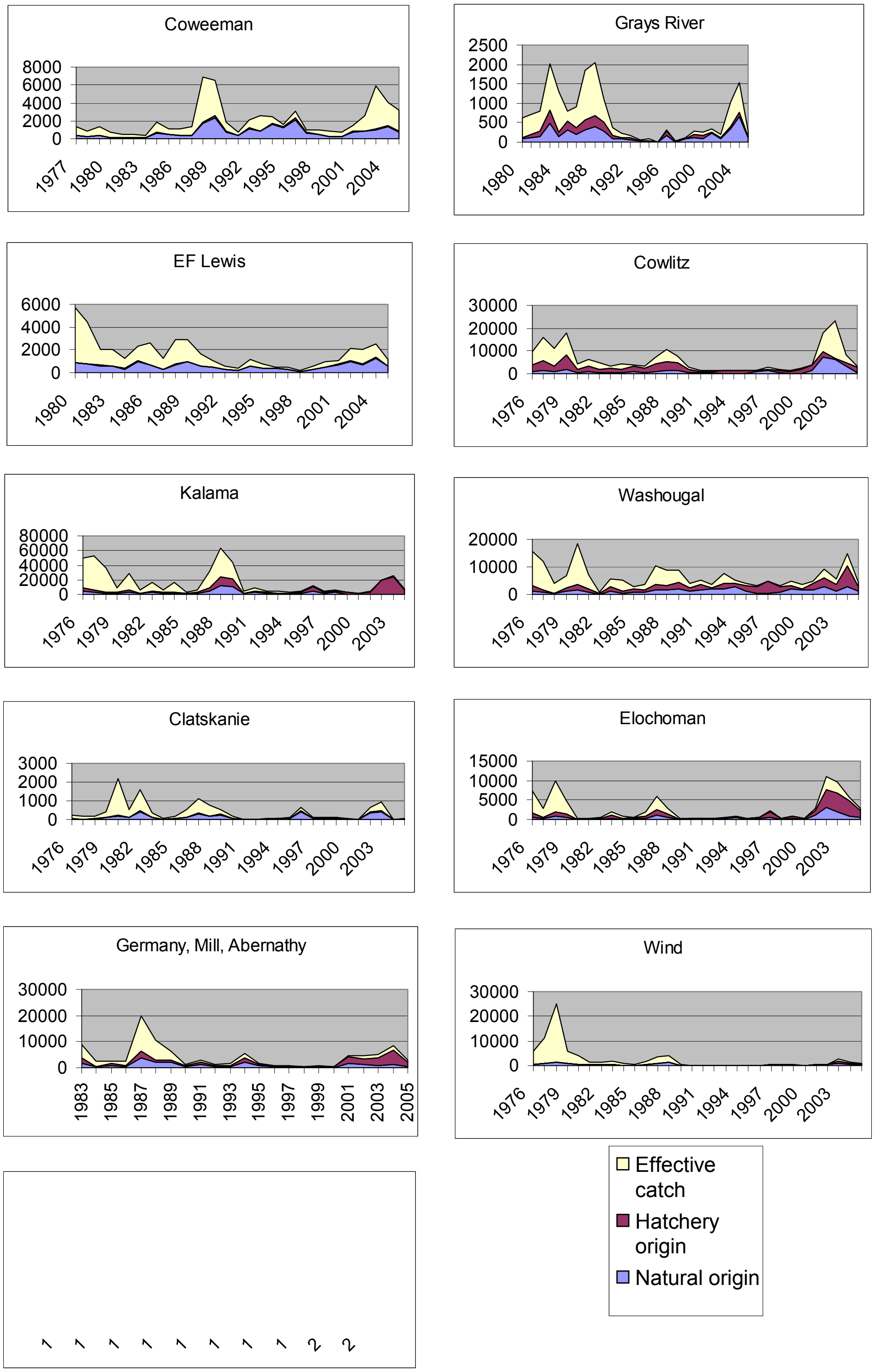


Figure 9 – Trends in natural and hatchery origin escapement to natural spawning areas. Effective catch is the estimated number of additional natural origin fish that would have escaped to spawn in a given year had there been no harvest. Data are in Appendix A.

Run reconstructions and spawner/recruit tables

Estimated Recruits

Recruits per spawner (R/S) estimates for the three category 1 populations are shown in Table 6 and Figure 10. There appears to be a somewhat cyclic pattern to the productivity with a period of relatively high productivity in the early1980s, low productivity in the late 1980s and early 1990s, and then higher productivity in the late 1990s. The high R/S for the Grays in 1995 is due to the low estimate of spawners for that year (9 fish). The apparent low productivity for 2001 and 2002 may be the results of incomplete data for the estimates. Summaries of other estimates from the cohort run reconstruction including progeny spawners, fishing mortalities, recruits and maturation rates are given in the Appendix A for brood years 1977-2002 for Coweeman, Grays, and Lewis and 1973-2002 for the other populations, except Germany/Abernath/Mill that starts with 1980.

Table 6. Spawning escapement, adult equivalent recruits, and recruits per spawner (R/S) for the Coweeman, Grays, and Lewis tule Chinook populations for brood years 1977-2002.

	Coweeman			Grays			Lewis		
BY	Spawners	Recruits	R/S	Spawners	Recruits	R/S	Spawners	Recruits	R/S
1977	337	421	1.2	1,009	907	0.9	1,086	3993	3.7
1978	243	604	2.5	1,806	398	0.2	1,448	2338	1.6
1979	344	56	0.2	344	2627	7.6	1,304	2363	1.8
1980	180	2363	13.1	125	28	0.2	899	1895	2.1
1981	116	651	5.6	208	977	4.7	799	1163	1.5
1982	149	1588	10.7	272	605	2.2	646	2148	3.3
1983	122	620	5.1	825	1997	2.4	598	2925	4.9
1984	683	8891	13.0	252	1124	4.5	340	3099	9.1
1985	491	4814	9.8	532	1323	2.5	1,029	2793	2.7
1986	396	1321	3.3	370	174	0.5	696	1080	1.6
1987	386	995	2.6	555	72	0.1	256	588	2.3
1988	1,890	2360	1.2	680	157	0.2	744	726	1.0
1989	2,549	2011	0.8	516	91	0.2	972	824	0.8
1990	812	2022	2.5	166	24	0.1	563	1051	1.9
1991	340	2028	6.0	127	15	0.1	470	257	0.5
1992	1,247	2581	2.1	109	111	1.0	335	700	2.1
1993	890	1682	1.9	27	111	4.1	164	483	2.9
1994	1,695	804	0.5	30	70	2.3	610	156	0.3
1995	1,368	293	0.2	9	179	19.9	409	467	1.1
1996	2,305	1146	0.5	280	201	0.7	403	865	2.1
1997	689	1571	2.3	15	172	11.5	305	787	2.6
1998	491	5483	11.2	96	606	6.3	127	2821	22.2
1999	299	3454	11.6	195	815	4.2	331	2182	6.6
2000	290	4489	15.5	169	1154	6.8	515	3088	6.0
2001	802	2133	2.7	261	390	1.5	750	978	1.3
2002	877	2332	2.7	107	521	4.9	1,032	889	0.9

Table 7. Population specific adult equivalent exploitation rates and average escapement age for BY 1977-2002 from the A&P run reconstruction for the Coweeman, Grays, and Lewis tule populations. The indicator stock used for the Coweeman was the adjusted Cowlitz and for the Grays and Lewis, the composite Cowlitz/Washougal/Grays.

Population	Coweeman		Grays		Lewis	
Brood Year	ER	Age	ER	Age	ER	Age
1977	0.758	4.0	0.839	3.9	0.799	3.5
1978	0.736	3.9	0.815	4.1	0.698	3.0
1979	0.832	4.2	0.774	3.9	0.702	3.5
1980	0.646	4.1	0.657	3.8	0.654	3.6
1981	0.534	4.0	0.567	3.9	0.497	3.9
1982	0.723	3.9	0.801	3.9	0.743	3.6
1983	0.775	4.3	0.820	4.0	0.771	3.3
1984	0.680	4.4	0.685	3.9	0.688	4.2
1985	0.737	3.9	0.726	3.8	0.731	4.1
1986	0.467	3.5	0.536	3.8	0.537	3.7
1987	0.417	3.8	0.488	3.6	0.464	3.5
1988	0.471	4.0	0.405	3.6	0.415	3.6
1989	0.754	4.1	0.722	4.2	0.681	3.9
1990	0.331	3.9	0.483	3.7	0.476	4.2
1991	0.214	3.9	0.343	4.4	0.274	3.9
1992	0.176	3.9	0.169	4.0	0.173	3.6
1993	0.439	3.8	0.317	3.4	0.349	3.6
1994	0.656	4.1	0.360	3.8	0.555	4.2
1995	0.375	3.9	0.399	3.8	0.441	4.1
1996	0.673	3.9	0.436	3.6	0.448	3.7
1997	0.549	4.1	0.307	4.1	0.294	4.0
1998	0.843	4.1	0.638	3.7	0.610	3.8
1999	0.627	4.1	0.596	4.1	0.604	4.1
2000	0.727	4.1	0.537	4.0	0.363	4.0
2001	0.697	4.0	0.557	3.5	0.525	3.9
2002	0.663	4.0	0.567	3.9	0.475	3.8
average	0.596	4.0	0.559	3.9	0.537	3.8

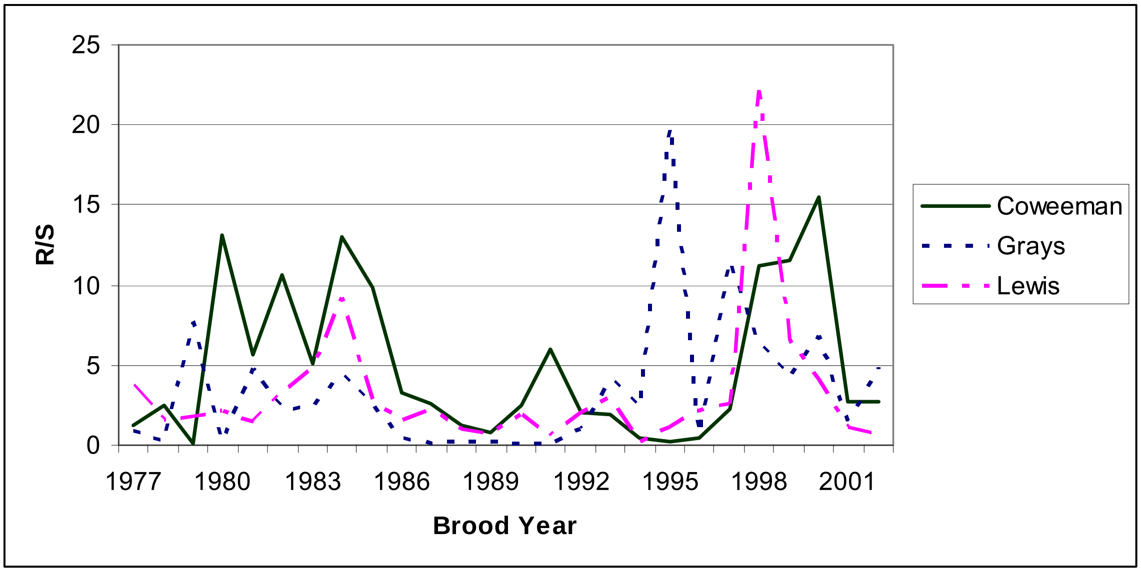


Figure 10 -- Recruits per spawner by brood year for the Coweeman, Grays, and Lewis tule Chinook populations for brood years 1977-2002.

AEQ exploitation rates were calculated from the cohort reconstruction for the natural origin populations and the results for the three category 1 populations are shown in Table 7; the average age of the natural origin spawning escapement is also given. The exploitation rates are generally a little higher for the Coweeman, especially in later years, than the Grays or the Lewis, averaging over all years (1977-2002) 0.60 for the Coweeman and 0.56 and 0.55 for the Grays and Lewis, respectively. Trends in exploitation rate show a decline from 1977 to 1992 and then an increase to the present through 1998 and a leveling off at around 0.60 (Figure 11). Note that the exploitation rate for brood year 2000 reflects harvest taken in 2002-2005. Exploitation rate estimates for brood years 2001 and 2002 are based on expected catches of age 4 and 5 fish and will change as we get new data. The absolute value and trend of the estimated exploitation rates for all the tule populations is expected to be similar (Figure 12) as we use the same few indicator stocks in our estimation of catch (the same composite stock is used for 9 of the populations and the Cowlitz indicator stock is used in some way for all populations. The higher values for the Coweeman population from 1996 to present are due to the higher estimated harvest rates for the Cowlitz indicator stock (Figure 3) than the Washougal indicator stock. For the composite indicator stock in these later years, only the Cowlitz and Washougal contribute, as CWT releases for the Grays seems to have been discontinued by 1996.

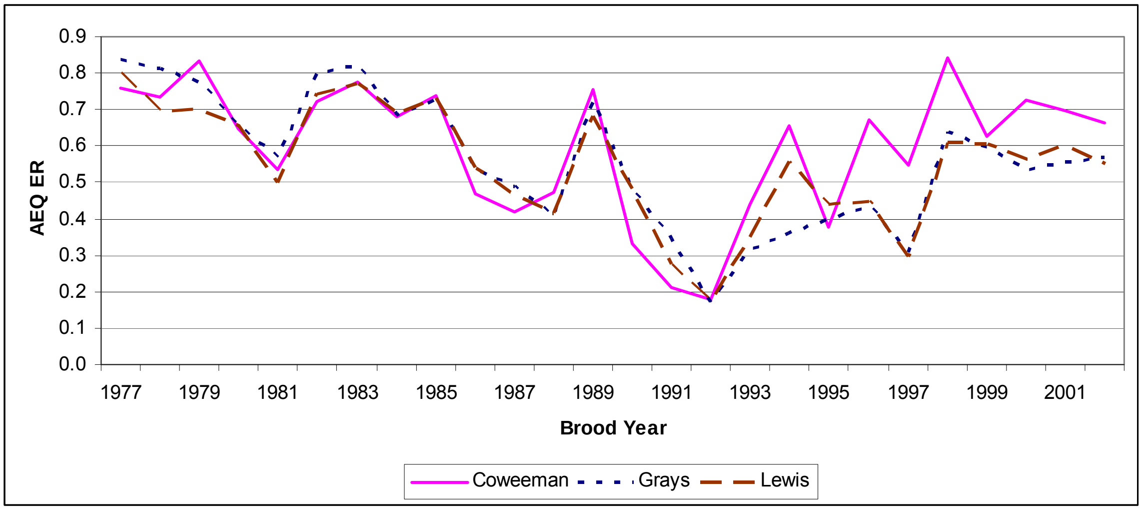


Figure 11. Adult equivalent (AEQ) exploitation rates (ERs) for the Coweeman, Grays, and Lewis tule Chinook populations for brood years 1977-2002.

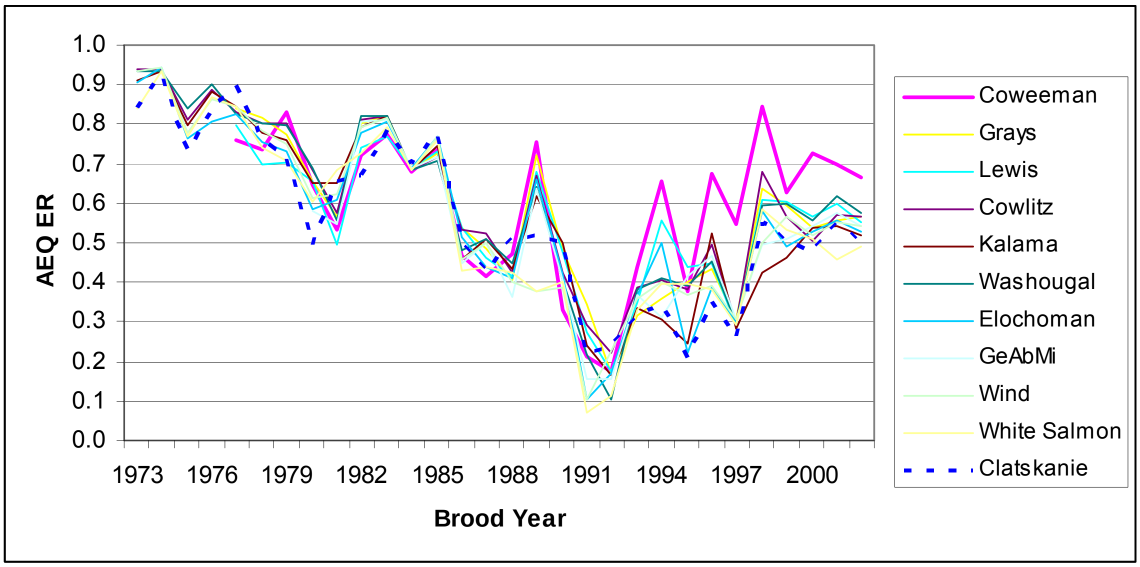


Figure 12. Adult equivalent (AEQ) exploitation rates (ERs) for all eleven tule Chinook populations in our analyses for brood years 1977-2002. The Coweeman (thick line) used the adjusted-Cowlitz indicator stock, the Clatskanie (doted line) used the 4-stock composite indicator stock, and the remaining populations used the 3-stock composite indicator stock.

The results from the Dynamic Model runs to estimate the spawner recruit functions are given in Table 8 and the fitted models are illustrated in Figures 8-10. The run were made for the case of using no covariates and using the one marine survival covariate. Parameters and test statistics were made for the three spawner-recruit functions. The number of spawners for maximum sustainable yield (MSY sp) is used as a potential upper threshold. However, this cannot be estimated if the resulting curve is a straight line (either horizontal, uniform recruits per spawner, or diagonal (no density dependence). The mean squared error of the predicted recruits against estimated (observed) recruits is used as a measure of error in the VRAP model. Parameter and model selection is based on statistics around fitting predicted calendar year escapements to observed escapements;

MSE for escapement is given along with the F statistic with degrees of freedom and the associated p-value for the model (P(esc)) are shown in Table 8 along with the delta AIC statistic. For comparison, the F statistic and probability based on recruits is also provided. If the parameter selection had been based on minimizing the error in predicting recruits the MSE for recruits would have been smaller, but would have been based on the assumption that recruits were measured without error. As is, we are measuring process error using the assumption that escapement is measured without error.

Table 8 -- Parameter estimates and test statistics for three spawner-recruit(S-R) functions (Ricker (Ric), Beverton-Holt (Bev) and hockey stick (Hoc)) for the three category 1 populations. The a parameter is intrinsic productivity or the slope of the curve at the origin. The b parameter is the spawners to achieve maximum recruits for the Ricker function and the maximum recruits for the Beverton-Holt and hockey stick functions. The mean square error (MSE) and p-value for the model are given based on escapement (esc) and on recruits (rec). The statistics used for model selection are based on the error of predicting escapement. The MSE(rec) and the autocorrelation (autocorrelation in MSE(rec)) statistics are inputs to VRAP. Yellow shading indicates best model fits according to the AIC statistic.

Population	Parameter		No covariates				Marine survival covariate		
S-R function			Ric	Bev	Hoc		Ric	Bev	Hoc
Coweeman	a		12.6	182.0	13.3		8.4	9.1	6.3
	b		536	1,760	1,727		1,099	5,277	3,222
	c						0.65	0.76	0.70
	MSY sp		440	120	130		790	1070	510
	MSE (rec)		1.38	1.16	1.16		1.05	1.04	1.00
	Autocorrel		-0.017	-0.028	-0.025		-0.027	-0.052	-0.019
	F statistic	F(1,27)	0.25	1.15	1.29	F(2,26)	2.37	2.54	3.07
	P(rec)		62%	30%	27%		12%	10%	7%
model	MSE (esc)		0.30	0.40	0.40		0.23	0.25	0.21
selection	F statistic	F(1,21)	29.7	16.7	17.0	F(2,20)	24.6	21.7	26.9
	P(esc)		0.0%	0.1%	0.0%		0.0%	0.0%	0.0%
	Δ AIC		7.2	13.7	13.6		1.7	3.2	0.0
Grays	a		3.0	5.8	2.2		8.5	*na*	*71*
	b		465	477	454		266	615	614
	c						0.89	1.17	1.17
	MSY sp		220	120	210		180	*na*	*Na*
	MSE (rec)		2.14	1.92	2.09		2.52	1.48	1.48
	Autocorrel		0.29	0.36	0.23		0.16	0.29	0.29
	F statistic	F(1,27)	1.72	2.22	2.60	F(2,26)	2.20	6.58	6.58
	P(rec)		20%	15%	12%		14%	1%	1%
model	MSE (esc)		1.60	1.48	1.51		1.11	0.75	0.75
selection	F statistic	F(1,21)	3.2	3.5	4.2	F(2,20)	6.8	13.4	13.5
	P(esc)		8.8%	7.4%	5.3%		0.6%	0.0%	0.0%
	Δ AIC		16.6	14.8	15.4		9.1	0.1	0.0
Lewis	a		3.3	4.8	3.9		7.3	17.8	6.5
	b		1,504	2,364	1,236		860	2,456	1,980
	c						0.69	0.69	0.69
	MSY sp		760	590	320		550	440	310
	MSE (rec)		0.76	0.72	0.75		0.27	0.22	0.27
	Autocorrel		-0.11	-0.10	-0.10		-0.22	-0.24	-0.16
	F statistic	F(1,27)	0.62	0.68	0.01	F(2,26)	22.28	31.06	22.40
	P(rec)		44%	42%	91%		0%	0%	0%
model	MSE (esc)		0.43	0.42	0.44		0.10	0.08	0.09
selection	F statistic	F(1,21)	2.4	2.6	2.5	F(2,20)	27.0	36.6	32.2
	P(esc)		13.5%	12.2%	12.9%		0.0%	0.0%	0.0%
	Δ AIC		36.7	36.3	37.7		5.1	0.0	2.1

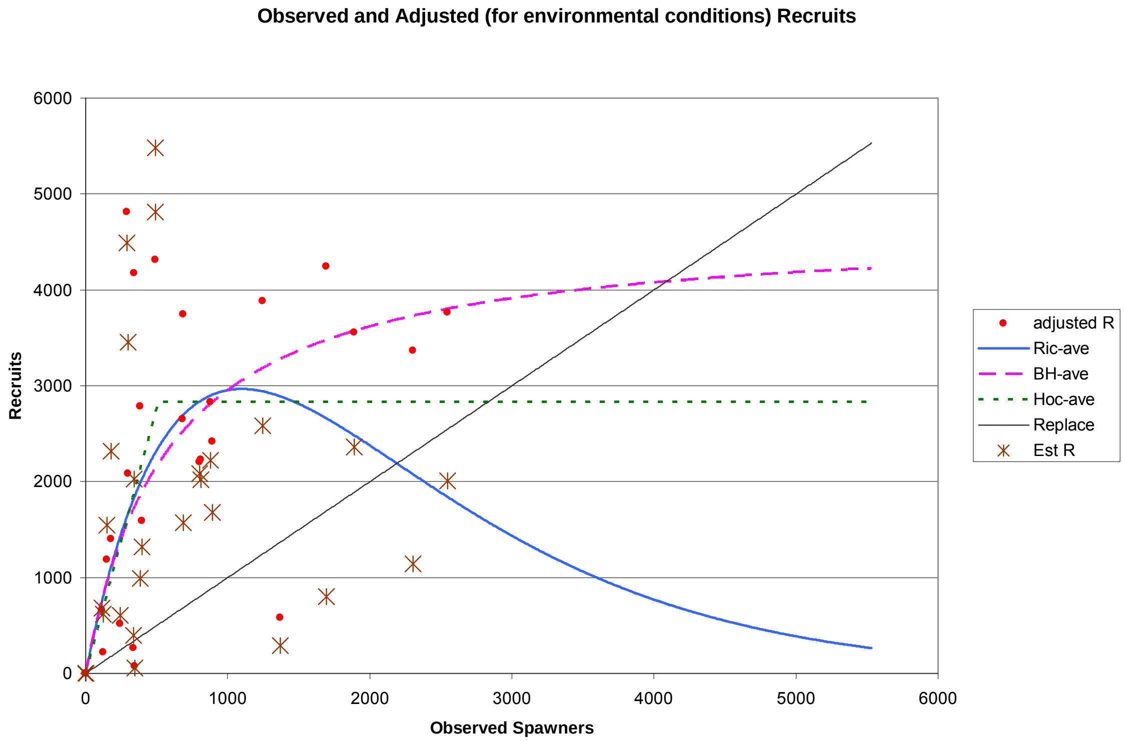


Figure 13 – Graph of spawners and recruits for the Coweeman population with fitted spawner/recruit curves for threemodels: Ricker (Ric), Beverton-Holt (B H), and hockey stick (Hoc). Data are from broodyears 1977-2002. Stars are the estimated recruits, dots are estimated recruits adjusted for variation in marine survival of the Cowlitz Hatchery stock. Curves are fit to the adjusted data.

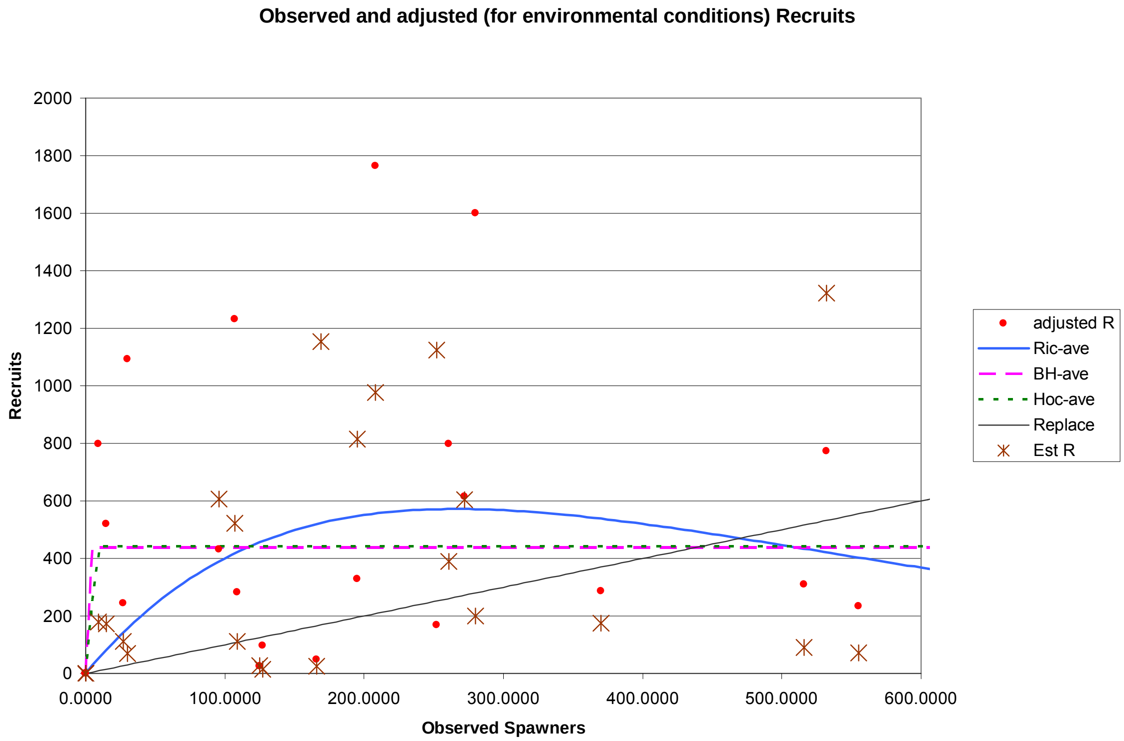


Figure 14 -- Graph of spawners and recruits for the Grays population with fitted spawner/recruit curves for threemodels: Ricker (Ric), Beverton-Holt (B H), and hockey stick (Hoc). Data are from broodyears 1977-2002. Stars are the estimated recruits, dots are estimated recruits adjusted for variation in marine survival of the Cowlitz Hatchery stock. Curves are fit to the adjusted data.

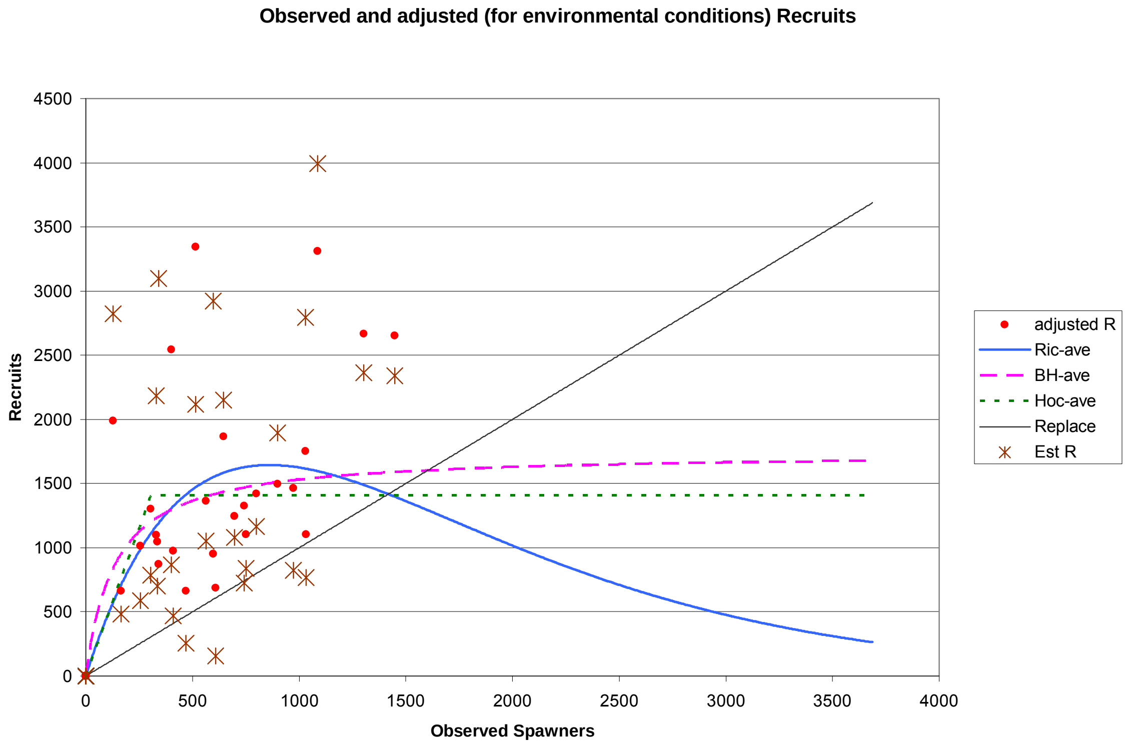


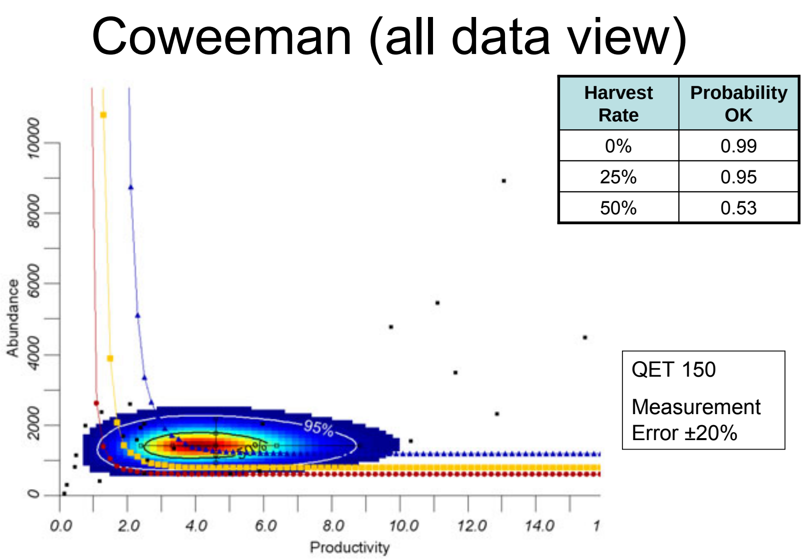
Figure 15 -- Graph of spawners and recruits for the Lewis population with fitted spawner/recruit curves for threemodels: Ricker (Ric), Beverton-Holt (B H), and hockey stick (Hoc). Data are from broodyears 1977-2002. Stars are the estimated recruits, dots are estimated recruits adjusted for variation in marine survival of the Cowlitz Hatchery stock. Curves are fit to the adjusted data.

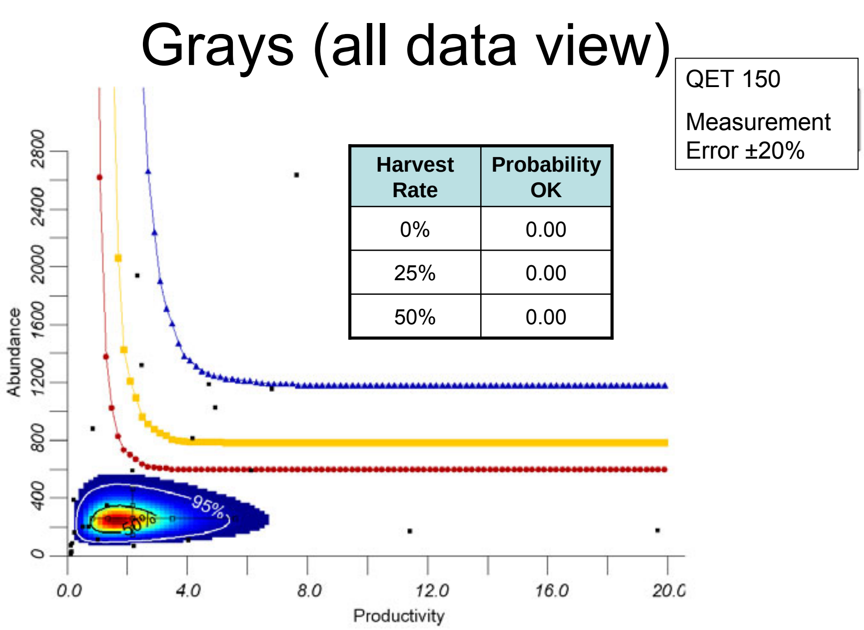
Viability curve results for each population

Current population status relative to viability curves generated under three alternative exploitation rate scenarios were generated for each of the category 1 populations (Table 9, Figure 16). The exploitation rate scenarios examined assumed no harvest, a 25% AEQ exploitation rate, and a 50% AEQ exploitation rate. We also explored two alternative QET assumptions: 50 spawners/year for four years, and 150 spawners/year for your years, corresponding to the recommendations by the WLC TRT for small and medium sized populations, respectively (McElhany et al. 2006).

Table 9 -- Probabilities of meeting viability criteria for abundance and productivity under alternative future exploitation rates for category 1 populations and assuming current habitat and environmental conditions.

Strata	State	Populations	Probability of meeting viability criteria					
			QET = 50			QET = 150		
			0 harvest	25% harvest	50% harvest	0 harvest	25% harvest	50% harvest
Coast Fall	WA	Grays	54%	16%	0%	0%	0%	0%
Cascade Fall	WA	Coweeman	100%	99%	93%	99%	95%	53%
	WA	Lewis	100%	98%	71%	98%	78%	5%





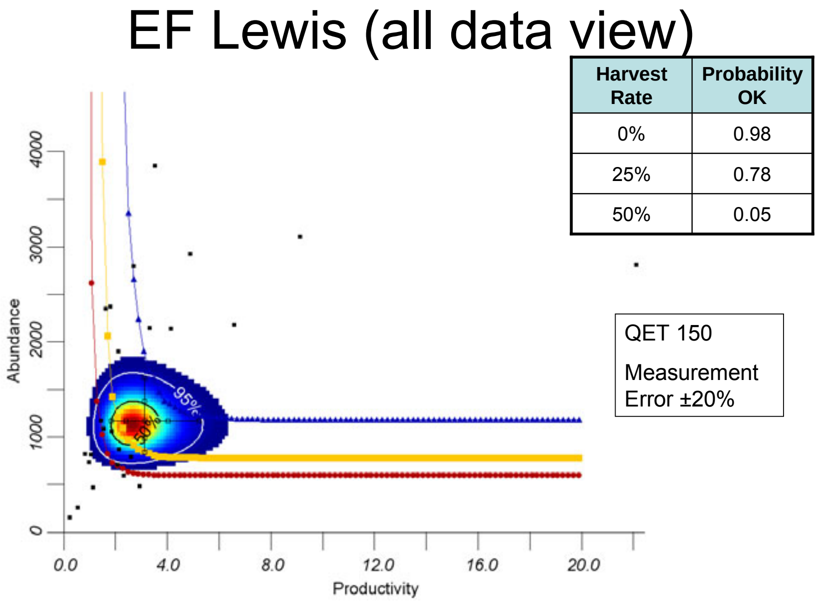


Figure 16 -- Abundance and productivity status of the Coweeman, Grays and Lewis populations relative to three viability curves: 0% AEQ exploitation rate (bottom curve), 25% AEQ exploitation rate (middle curve), and 50% AEQ exploitation rate (top curve). Quasi-extinction level set to 150/year for four years. The risk curves describe a 5% probability of declining to the QET in 100 years. The measurement error is a rough approximation based on principles described in McElhany et al 2006.

VRAP Results for Each Population

Rebuilding Exploitation Rates

RERs for the three category 1 populations are reported in Table 10 under several alternative assumptions regarding spawner/recruit model used and with and without marine survival as a covariate. Note that the marine survival covariate analyses were all conducted subsequent to February 2007. In general, including marine survival improved the fit of the model to the data and had a large effect on the RERs (Table 8). With the exception of the Ricker model for the Coweeman, the RERs were not very sensitive to which spawner/recruit model was used. For the Lewis using the marine survival covariate produced slightly higher RERs than not using the covariate; for the Coweeman including the covariate produced higher RERs; and for the Grays it reduced the RER to zero. Results from using the marine survival covariate in the VRAP procedure is dependent on the assumption of marine survival levels assumed for the runs. We used the full range of marine survival seen in the 23 year data series (1977-2002) for the 25 year runs in VRAP.

Table 10 -- Rebuilding exploitation rates (RERs) for the category 1 populations. Shaded results correspond to the best fitting spawner-recruit models (Table 8). The bold values were reported in the draft Feb 2007 report.

			RER - no covariates					RER - with marine survival as covariate				
Strata	**State**	**Population**	**LEL[2]**	**UEL[3]**	**MOD[4]**	**RER[5]**	**TRIG[6]**	**LEL**	**UEL**	**MOD**	**RER**	**TRIG**
Coast	WA	Grays [1]										
Fall			51	221	BEV	**0.42**	UEL	51	221	BEV	0	UEL
			51	221	HOC	0.46	UEL	51	221	HOC	0	UEL
								151	221	RIC	0	both
			151	221	BEV	0.30	LEL	151	221	BEV	0	LEL
			151	221	HOC	0.32	LEL	151	221	HOC	0	LEL
Cascade	WA	Coweeman	51	750	RIC	0.00	UEL	51	790	RIC	0.48	UEL
Fall												
			51	750	HOC	**0.42**	UEL	51	750	HOC	0.58	UEL
			116	750	RIC	0.00	UEL	116	790	RIC	0.46	UEL
			116	750	HOC	0.44	UEL	116	750	HOC	0.56	UEL
			151	440	RIC	0.52	LEL					
			151	750	RIC	0.00	UEL	151	790	RIC	0.46	UEL
			151	750	BEV	**0.42**	UEL	151	1070	BEV	0.54	UEL
			151	750	HOC	**0.42**	UEL	151	750	HOC	0.58	UEL
		Lewis										
			51	645	BEV	**0.42**	UEL	51	645	BEV	0.46	UEL
			51	645	HOC	0.4	UEL	51	645	HOC	0.44	UEL
			127	645	BEV	0.44	UEL	127	645	BEV	0.44	UEL
			127	645	HOC	0.4	UEL	127	645	HOC	0.44	UEL
			151	645	BEV	0.44	UEL	151	645	BEV	0.46	UEL
			151	645	HOC	0.40	UEL	151	645	HOC	0.44	UEL

Notes

[1] The draft version of this report available in February 2007 reported an RER of 0.42 for the Grays using an LEL of 151 and an UEL of 220 and no marine survival covariate data. However, this RER was in fact calculated using an LEL of 51, not 151 as reported in the February version of the report.

[2] Lower escapement level -- indicates lower threshold used for risk assessment, and is the QET + 1.

[3] Upper escapement level -- indicates upper threshold used for risk assessment, and is generally the average NOR escapement for the time period analyzed.

[4] Spawner/recruit model used.

[5] Recovery exploitation rate.

[6] Threshold that limits RER

Discussion

Key assumptions and uncertainties

The current analyses for the LCR tule populations are predicated on several assumptions and subject to considerable uncertainty. The most critical assumption is that the population time series used to generate spawner-recruit relationships reflect population processes that will continue into the future. In other words, we are assuming that past population performance will reflect future population performance. This assumption is critical to most population forecasts, but nonetheless may well be incorrect. Another key uncertainty associated with these analyses is simply uncertainty about the basic data. In particular, the annual spawner and recruit numbers in the A&P tables are not observed data, but rather are estimates derived from the primary spawner survey counts, harvest rate estimates, age structure estimates, and estimated hatchery fractions. Each of the spawner and recruit values in the A&P tables is, therefore, a point estimate, with associated uncertainty around it.

The SPAZ/viability curve analyses attempt to take into account some of the uncertainty by assuming a certain level of variation around the point estimates, but this will not necessarily capture all of the uncertainty about the data, especially if the point estimates themselves are biased or if the uncertainty is not symmetrical. The VRAP analyses take into account some of the uncertainty in the fit of the data to the particular model(s) employed, but do not take into account measurement uncertainty in the abundance data, age structure, harvest rates, or hatchery fraction. For the three population that are the focus of this report, the Work Group spent considerable time and effort trying to make the data sets as accurate as possible, but even so considerable uncertainty in the basic data remains. Data for those populations that the Work Group did not have time to evaluate in detail are likely to be even more uncertain.

An additional major source of uncertainty is model uncertainty. Both the VRAP and SPAZ/viability curve approaches assume a spawner/recruit model in order to simulate the populations under alternative harvest rates. None of the population data fit any of the modeled spawer/recruit relationships very well (Figure 13, Figure 14, Figure 15), so assuming any particular spawner/recruit model is problematic. However, this is a problem that is common to these sorts of spawner/recruit analyses and is not unique to these data sets.

In an effort to address the uncertainties of the spawner/recruit models, both the VRAP and SPAZ approaches use a quasi-extinction threshold when evaluating the probability that a population will go “extinct”. In other words, the models are not considered sufficiently reliable, particularly at low spawning levels, to accurately describe the dynamics of a population all the way to extinction. Instead, they use a quasi-extinction threshold (QET; either 50 or 150 spawners/year for your consecutive years, for the category 1 populations in this report) that is associated with low spawning density result in uncertainty and increased risk (McElhany et al. 2006). QETs are commonly employed in population viability modeling, but the choice of a particular QET can substantially affect the modeling results (Tables 9 and 10).

Mechanically, the RER analyses and the viability analyses are fairly similar, differing only in details such as how productivity is estimated and how uncertainty is treated. The methods differ more in how they treat population abundance goals. The viability curve approach uses viability goals as the 'yard stick' with which the populations are compared. In particular, the viability curve approach as applied in this report assumed that 'viability' means <5% probability of quasi-extinction over a 100 year time frame. In contrast, the RER approach uses a shorter time horizon (25 years) for evaluating extinction risk, and combines this with an upper abundance goal that is typically lower than the ultimate recovery goal. The upper abundance threshold for the RER is not necessarily intended to reflect the viability level under recovered conditions, but rather what the population can support under current depressed condition and still allow for rebuilding towards the recovery goal as recovery actions related to other threats occur. Determining the appropriate time horizon and risk threshold for evaluating jeopardy decisions under the ESA are largely policy decisions.

Viability Curve Results

The three category 1 populations analyzed differed substantially in their probabilities of meeting viability criteria under alternative assumptions about future harvest scenarios. Based on these analyses, the Coweeman population appeared to be the most robust, with a 53% - 93% probability (depending on the QET used) of meeting the viability criteria assuming a 50% AEQ exploitation rate, and 95%-99% probability of meeting the criteria assuming a 25% AEQ exploitation rate (Table 9). The Grays River population, in contrast, had an estimated 0%-54% probability of meeting the viability criteria assuming no exploitation, 0%-16% assuming a 25% AEQ exploitation rate, and 0% assuming a 50% AEQ exploitation rate. The Lewis River population was intermediate between the other two (Table 9).

The differences in results between the populations appear at least broadly consistent with the habitat information summarized by the LCFRB (Table 4). In particular, the Grays River was judged by the LCRFB to have more degraded habitat than other two populations, and lower estimated productivity and capacity from the EDT method (Table 3). The Grays River also has a long history of hatchery production that ceased relatively recently, and it is possible that existing natural population in the Grays is genetically less fit than populations with less hatchery influence.

Interpreting the viability results in terms of a consultation standard is not entirely straightforward. In particular, the viability criteria (<5% extinction in 100 years) were intended to be related to recovery or delisting criteria, and their relationship to jeopardy is less clear. It is also important to note that the viability criteria used in this report do not exactly correspond to the recovery criteria described by the LCRFB. In particular, the LCRFB criteria were based on an early version of the WLC TRT criteria (population change criteria -- (McElhany et al. 2003)) that was in some cases more conservative than the SPAZ/viability curve approach taken in later assessments (McElhany et al. 2006, this report). For example, in our analyses the Coweeman population appears to meet the 5% extinction risk in 100 years viability criteria at a considerably lower abundance than the 3600 average spawner goal in the LCFRB Plan.

VRAP Results and RERs

In general, the RERs were quite sensitive to whether or not marine survival was used as a covariate in the analysis, and less sensitive to the choice of spawner/recruit model (Table 10). Of the three populations analyzed, the Grays River had the lowest RER (0 – 46%, depending on the model) and the Coweeman River the highest (0-58%, depending on the model), consistent with the viability curve results. The Lewis population had an RER of ~40-44% under a wide range of assumptions. Note that February version of this report did not include any results that incorporated marine survival as a covariate.

Like the viability probabilities, the RERs are also sensitive to the choice of thresholds. For example, using the no-covariate Beverton-Holt model, the Grays River RER is 0.42 when the lower threshold is 51, and is 0.30 when the lower threshold is 151 (Table 10). (Note that the February version of this report erroneously reported an RER of 0.42 for the Grays River associated with a lower threshold of 151; in fact the lower threshold used was 51.) The work group explored a range of alternatives for the lower threshold, but ultimately recommended using 1 + the QET value recommended by the WLC TRT. However, the WLC TRT has formally recommended QET values only for Oregon populations (McElhany et al. 2006). It is unclear from the TRT criteria whether the three category 1 populations analyzed are "small" or "medium" sized populations, so we have included results for both alternatives in this report. We believe this choice for the lower threshold is reasonable and is consistent with the QETs used in viability curve analysis, but other choices may be reasonable as well.

The choice of upper escapement threshold (UET) is also difficult. A UET is a necessary part of the VRAP analysis, and the agency has considered alternative methods for choosing an UET in the past (NMFS 2004). The rationale has been to choose a QET that is consistent with the current status and conditions of the population, and allows the population to progress toward recovery as habitat improvements are made and the capacity and productivity of the population improves. For these analyses, the Work Group ultimately focused, for a UET, on the higher of either estimated spawners that achieves MSY, or the average natural origin escapement over the time series analyzed. For the three category 1 populations analyzed, the average natural origin escapement was the larger value and was therefore used as the UET (Table 10). The results generally indicate how higher UETs lead to estimates of lower RERs.

ESU level considerations

We focused our quantitative analyses on three relatively large natural tule populations in the ESU that also had relatively low proportions of stray hatchery fish in their spawning escapements (Table 3). However, there are an additional 17 tule populations in the ESU (Table 1), and the effects of harvest on these populations need to be considered as well.

Subsequent to the February 2007 recommendation regarding a consultation standard, we used the data compiled by the WLC TRT to conduct viability curve and VRAP analyses on all populations with any time series abundance data (results not shown). There are several reasons why we have little confidence in the results for populations other than the three category 1 populations. First, the Work Group spent considerable time evaluating the data quality for the three category 1 populations and updating the estimates of spawning escapement, age structure, and hatchery fraction. This process resulted in considerable revision of the time series for these populations. We have no reason to believe that the time series for the remaining populations would not also be subject to similar revision, and therefore we have little confidence in the accuracy of the existing data. Second, the remaining populations in the ESU tend to be smaller and/or are dominated by stray hatchery fish. In many cases, the fraction of hatchery fish that make up the spawning escapements is estimated poorly, if at all, leading to very large uncertainties in the productivity estimates, especially for the smaller populations. In particular, we believe that productivity in small populations may be frequently overestimated due underestimates of the hatchery fraction.

Since we had little confidence in the RER or viability probabilities for the category 2 and 3 populations, we instead treated these populations qualitatively by comparing their status as evaluated by the WLC TRT and the LCFRB to that of the three category 1 populations we did analyze.

The status of the populations has been extensively reviewed by the WLC TRT and the LCFRB, and results from these reviews are summarized in Table 2, Table 3, and Table 4. Here we comment on a few patterns that appear relevant to interpreting our results. The biggest difference among the populations in terms of current status appears to be related to the proportion of hatchery fish on the spawning grounds. The Coweeman, Grays, and Lewis population all have relatively low (<20%) recent average hatchery contributions.

In contrast, the Elochoman, Mill/Germany/Abernathy, and Kalama populations all have ~70% or more hatchery fish on the spawning grounds in recent years. The Washougal and Cowlitz populations are intermediate, with ~40-60% hatchery contributions. In contrast to the hatchery situation, the populations all have fairly similar habitat conditions, at least as summarized by the LCFRB. In particular, of the 13 Washington populations evaluated by the LCRFB, 8 received habitat rating of “2”, indicating moderately impaired habitat. Three populations, including the Grays, Lower Cowlitz, and Big White Salmon received ratings of “1.5” (between highly and moderately impaired), and only one population (Lower Gorge) received a rating >2. The EDT evaluations reported by the LCFRB appear generally consistent with the habitat rankings, with most EDT productivity estimates falling in the 3-4 recruits/spawner range (Table 3). Likewise, the populations all received generally similar spatial structure and diversity ratings.

Fewer data or analyses were available for populations on the Oregon side of the river, but we have no reason to believe that these populations are likely to be substantially different from the Washington populations in terms of habitat quality. Oregon has fewer large tule hatchery programs, so the degree of hatchery influence in the Oregon populations is likely to be, on average, less than in the Washington populations.

Evaluating the impacts of harvest on hatchery dominated natural populations, such as the Cowlitz and the Kalama, appears particularly problematic. Both the VRAP and viability curve approaches assume that that population being analyzed is demographically closed, an assumption that is clearly violated in the case of populations with large in-basin hatchery programs. When applied to the populations with high hatchery fractions, the approaches therefore implicitly assume 1) that in the future populations will consist entirely of natural origin fish, and 2) that the estimates of natural productivity made from the time series with high hatchery fractions are indicative of what the population would experience with no hatchery straying. Neither assumption appears particularly well founded, and one of our recommendations below is to better coordinate hatchery and harvest jeopardy analyses. The Hatchery Science Review Group has recently released a report on Lower Columbia River hatcheries (available at http://www.hatcheryreform.us/prod/site/alias__default/hsrg_document_library/306/hsrg_document_library.aspx), and the approach taken by the group may be a good starting point for a combined hatchery/harvest analysis.

For the reasons discussed above, we were uncomfortable attempting to determine RERs for the hatchery dominated populations. However, based on the comparison of the status of the populations discussed above, we see no reason to believe that populations such as the Cowlitz or the Kalama could sustain exploitation rates any higher than the RERs that we did estimate for the three relatively hatchery free populations were they to be managed for natural production. In fact, the natural productivity of the hatchery dominated populations may well be lower than expected based on the habitat quality of the basin if large scale hatchery programs have themselves genetically damaged the populations and lowered their productivity, at least in the short term. Therefore, if they could be estimated, we speculate that the RERs for these populations would probably be lower than those we estimated for the three category 1 populations. The Grays River results may be informative in this regard. In the past, this population has high hatchery contributions that have recently gone down, and it is possible that the relatively low RER and probabilities of viability for this population are, in part, related to that hatchery history. If so, the population may over time readapt to the wild environment, and perhaps its productivity will improve.

Similarly, we see no reason to expect that the remaining small tule populations in the ESU would be able to sustain greater harvest impacts than those we calculated for the category 1 populations. On the contrary, many of these populations are so small that it seems likely that they would require lower exploitation rates than those calculated for the category 1 populations in order to achieve a similar level of viability.

Conclusion: The RERs estimated for the three category 1 populations are likely to be higher than RERs for the remaining populations in the ESU.

As mentioned elsewhere, we would highlight the need for continued efforts to combine conclusions developed through the recovery planning process, particularly with respect to populations priorities, with a more integrated approach that couples harvest and hatchery reform.

Comparison with earlier RER and AEQ ER estimations

Estimating brood year AEQ fishing mortality that is population specific is an essential step in estimating brood year recruits. As described in the methods, the Work Group used a method of using harvest rates estimated from CWT-recoveries of an indicator stock and applying them to the age specific estimates of the wild population escapement to estimate fishing mortality (in numbers of fish) at age. We compare this with the method used previously to generate an RER for the Coweeman population (Simmons 2001) that used the AEQ exploitation rate estimate for the indicator hatchery stock and applied that to the total escapement of the wild population to get to get total (over all ages) AEQ fishing mortality and recruits.

The primary difference between the two methods is the input data used in reconstructing cohorts. The method used in previous analyses (hatchery cohort analysis) used estimated recoveries of hatchery CWTs in fishery catches and spawning escapements at each age over the life of a cohort to calculate age specific harvest rates and maturation rates. The current method (natural cohort analysis) uses estimated escapement at age and reconstruct cohorts using age specific harvest rates from the CWT cohort analyses to estimate catches and maturation rates. Both methods are likely to be biased: The natural-cohort method may be biased due to stream surveys missing the smaller (younger) fish (e.g., Zhou 2002), resulting in an estimate biased toward older ages. On the other hand, the estimates derived from the age structure estimated from hatchery returns may also be biased if the hatchery population has a different maturity schedule from the natural population of interest. For example, it is not uncommon for hatchery stocks to mature at younger ages than closely related wild populations (e.g., Knudsen et al. 2006).

Brood year AEQ exploitation rates calculated from Cowlitz hatchery CWT data, adjusted for the terminal recreational fishery, have previously been used to assess the harvest impact rates on the Coweeman tule population (Kope 2006). This is based on the assumption that the Cowlitz hatchery indicator stock represents the maturation rate schedule and harvest rates on the natural Coweeman population, the same assumption used by Simmons (2001) in estimating the 2001 RER of 0.49. The Cowlitz CWT hatchery stock is also used in the fishery regulation assessment model (FRAM) used by PFMC to represent Washington lower Columbia River tule fall Chinook; so these hatchery AEQ exploitation rates are comparable to the projections for lower Columbia River tule exploitation rates made using FRAM during the PFMC pre-season process, except that the FRAM ERs are calculated for calendar year returns, while brood year ERs are based on cohort returns. In addition, the FRAM process only scales pre-terminal Council area fisheries, and assumes that impacts in terminal fisheries will remain unchanged.

We directly compared the AEQ exploitation rate estimated by cohort analysis of Cowlitz CWT data with the AEQ exploitation rates generated from the A&P tables by natural cohort reconstruction (Table 11, Figure 17). The AEQ exploitation rates we estimated for the Coweeman population are generally higher than those reported by Kope (2006). Compared to the Cowlitz Hatchery stock, the estimated natural escapement in the Coweeman had a lower proportion of 2 and 3 year old fish, resulting in lower maturation rates and, therefore, higher lifetime exploitation rates. Regardless of the estimator used, however, a trend of increasing exploitation rates since the early 1990's is clearly evident.

Table 11 -- Brood year adult-equivalent exploitation rates estimated for the Coweeman tule Chinook population in two ways: a) the hatchery cohort analysis method and b) the natural cohort analysis method.

	Total AEQ ER	
Brood year	Hatchery cohort analysis[1]	Natural cohort analysis
1990	34%	33%
1991	9%	21%
1992	18%	18%
1993	30%	44%
1994	46%	66%
1995	28%	38%
1996	57%	67%
1997	32%	55%
1998	54%	84%
1999	58%	63%
2000	61%	73%
2001*	62%	70%

* incomplete broods
[1] Estimates are from Kope (2006)

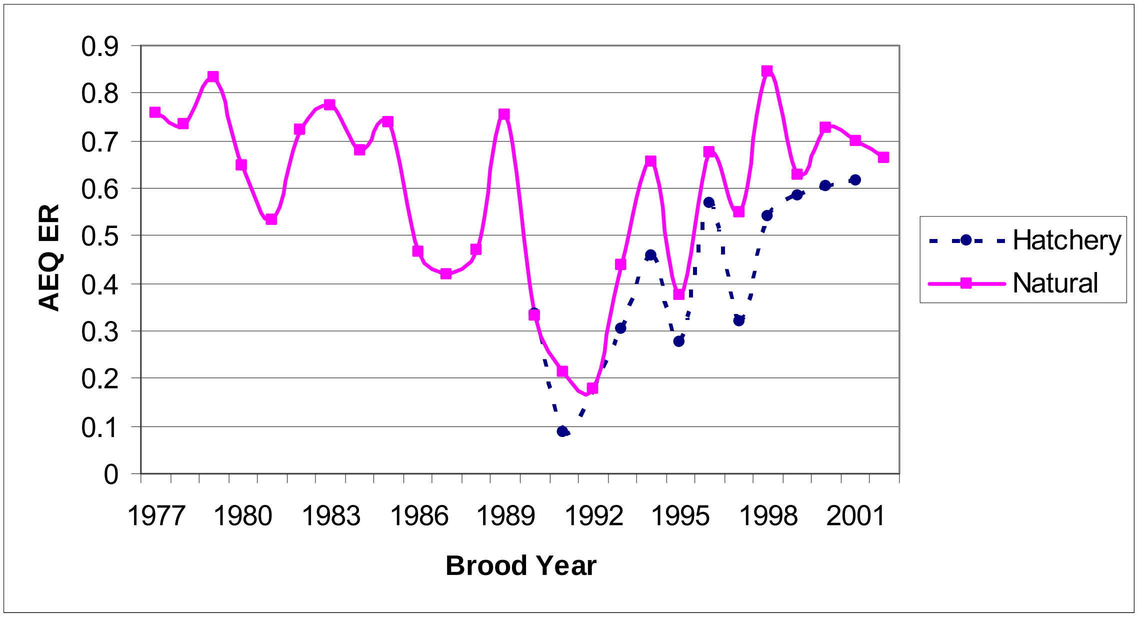


Figure 17 -- Brood year adult equivalent exploitation rates on Coweeman River tule fall Chinook estimated in two ways: a) hatchery cohort analysis method and b) natural cohort analysis method. Data from Tables 6 & 10.

Application of AEQ ER estimates to a consultation standard

The issue of how best to characterize the exploitation rates on natural stocks remains unresolved. Past consultation standards used the Coweeman fall Chinook stock as the sole representative of Lower Columbia tule fall Chinook, and exploitation rates used to derive the RERs were based on cohort analyses of Cowlitz CWTs. The same CWT indicator stock has been used in the fishery regulation assessment model (FRAM) during the Pacific Fishery Management Council's preseason planning process to assess the impacts of management measures on tule fall Chinook, and in past assessments of historic fishery impacts.

The exploitation rates used in the development of the RERs presented in this report were calculated by cohort reconstruction of the escapement of natural origin fish using the age-specific harvest rates from hatchery CWT cohort analyses. As a result of the apparent older age composition of the natural spawners relative to that of the Cowlitz Hatchery escapement, the exploitation rates used to calculate the RERs were generally higher than those used in the past. This disparity produces inconsistency between the RERs presented in this report, and the fishery impacts projected during the Council's preseason process to assess whether or not the management measures are consistent with NMFS' ESA guidance.

In order to assess compliance of fisheries with a consultation standard, it is necessary for both the standard and the fishery metric compared to the standard to be expressed in comparable units. Since the RER is expressed as AEQ ER for a cohort, the achieved exploitation rates should also estimated in this manner. In addition, the same method or assumptions of age structure and maturation rates should be used for both the RER and the achieved ERs.

A couple of discrepancies arise in the application of a consultation standard based on a RER to management decisions evaluated using the FRAM model, however. First, exploitation rates derived using FRAM model, while based on CWT recoveries, are expressed on an annual basis rather than a cohort basis. These two sets of exploitation rates cannot be directly compared, but over time, both should have the same average value. Second, the exploitation rates on which the RERs are based, are derived from escapement estimates for the natural origin populations while the FRAM ERs are derived from hatchery CWT recoveries. Differences in the age composition between CWT and natural escapements lead to differences in the average values of exploitation rates derived from them. This has been previously observed for Puget Sound Chinook populations, and the degree of difference between the two rates varies with the population (N. Sands, unpublished data). One approach may be to compare projected FRAM exploitation rates with reconstructed cohort exploitation rates to estimated a bias correction to bring FRAM rates in line with RERs.

In general, the choice of a method for estimating exploitation rates should not substantially affect the results, as long as the same method is used consistently. However, the current difference in the methods used to calculate RERs and to project fishery impacts appears to be inconsistent.

Recommendations for future work

Coordinate hatchery and harvest analyses – The Work Group struggled a great deal over how to evaluate the effects of harvest on populations dominated by high proportions of stray hatchery fish. There are both technical and policy problems related to this issue. A key policy question that needs to be addressed is whether any of the populations that are currently dominated by hatchery strays will be managed primarily for natural production in the near future. Reducing the exploitation rates on these hatchery dominated populations will not be sufficient to move these populations toward recovery goals unless the hatchery fractions are substantially reduced as well. This highlights the need for a more coordinated and phased approach that couples hatchery and harvest reform, particularly for populations to be managed for natural production to meet overall recovery objectives for the Lower Columbia River Chinook ESU.

Develop alternative methods of evaluating effects of harvest – The VRAP/RER method of determining harvest rates consistent with a ‘no jeopardy’ determination is conceptually attractive. In particular, by combining an avoidance of extinction (lower threshold) and a measure of progress toward recovery (upper threshold), the approach appears to fulfill both aspects of a jeopardy analysis (NMFS 2004). However, in practice the approach can be difficult to apply, for several reasons. First, the approach is fairly data intensive, requiring accurate estimates of escapement, age structure, hatchery fraction, and exploitation rates. These data are typically available only for a subset of populations (e.g, 3/20 for the tule Chinook populations that are the subject of this report), and attempting to apply the method using poor quality data is likely to produce uncertain results. Second, the method is sensitive to choice of thresholds.

As it is applied to harvest rate analysis, the viability curve approach suffers from the same data quality problems as the VRAP/RER method, although it attempts to deal with this issue by explicitly accounting for measurement error. The viability method also explicitly relates harvest to viability goals, which may be a higher standard than the jeopardy standard requires. Like the VRAP/RER approach, the viability curve method is also sensitive to the choice of a QET, with alternative QETs lead to different estimates of extinction risk. In this report we followed the QET recommendations made by the WLC TRT.

It is clear that it would be desirable to develop consistent methods for establishing harvest consultation standards that are capable of being applied to populations with poor quality data. For example, Holmes and Fagan (2002) and Holmes (2004) developed methods of estimating extinction risk that are robust to poor quality data. Perhaps these or similar methods could be adapted to analysis of harvest impacts. Alternatively, perhaps an approach that attempted to use multiple sources of data to generate generic estimates of productivity for Lower Columbia tules would work better than the population-by-population approach taken in this report. Further work on this subject is warranted.

Finally, in order to effectively manage and conserve natural origin Chinook population in the Lower Columbia River, it is essentially that sufficient data be collected on these populations to conduct meaningful analyses.

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Appendix A – Input Data for the Cohort Reconstructions and RER estimations.

Table A1. Annual Spawning Escapement Estimates for Lower Columbia River tule Chinook populations. From LCR TRT August 2006 and updated for the Coweeman, Grays and Lewis, November 2006 by WDFW.

	Cowee -man	Grays	Lewis	Cow-litz	Kala-ma	Wash-ougal	Clat-skanie	Elocho -man	Ge/Ab/ Mi	Wind	White Salmo n
1973				8,390	6,262	203	17	500		487	904
1974				7,566	12,834	2,977	164	245		610	882
1975				4,766	18,123	982	379	220		574	1,899
1976				3,726	8,352	3,037	219	1,682		646	2,063
1977	337	1,009	1,086	5,837	6,549	1,652	4	568		971	231
1978	243	1,806	1,448	3,192	3,711	593	523	1,846		1,527	1,063
1979	344	344	1,304	8,253	2,731	2,388	76	1,478		946	662
1980	180	125	899	1,793	5,850	3,437	4	64	516	401	1,598
1981	116	208	799	3,213	1,917	1,841	25	138	1,367	256	839
1982	149	272	646	2,100	4,595	330	67	340	2,750	365	1,579
1983	122	825	598	2,463	2,722	2,677	48	1,016	3,725	495	280
1984	683	252	340	1,737	3,043	1,217	62	294	614	134	393
1985	491	532	1,029	3,200	1,259	1,983	51	464	1,815	170	153
1986	396	370	696	2,474	2,601	1,589	67	918	980	422	116
1987	386	555	256	4,260	9,651	3,625	177	2,458	6,168	776	161
1988	1,890	680	744	5,327	24,549	3,328	34	1,370	3,133	1,206	382
1989	2,549	516	972	4,917	20,495	4,578	17	122	2,792	112	243
1990	812	166	563	1,833	2,157	2,205	34	174	650	11	145
1991	340	127	470	935	5,152	3,673	143	196	2,017	58	75
1992	1,247	109	335	1,022	3,683	2,399	228	190	839	54	1,078
1993	890	27	164	1,330	1,961	3,924	143	288	885	4	108
1994	1,695	30	610	1,225	2,190	3,888	455	706	3,854	11	288
1995	1,368	9	409	1,370	3,094	3,063	143	156	1,395	4	253
1996	2,305	280	403	1,325	10,676	2,921	17	533	593	166	32
1997	689	15	305	2,007	3,548	4,669	76	1,875	603	282	124
1998	491	96	127	1,665	4,355	2,971	143	228	368	213	242
1999	299	195	331	969	2,655	3,129	337	718	575	126	401
2000	290	169	515	2,165	1,420	2,155	194	196	416	14	167
2001	802	261	750	3,647	3,714	3,901	278	2,354	4,024	444	2,072
2002	877	107	1,032	9,671	18,952	6,050	76	7,581	3,343	375	1,859
2003	1,106	398	738	7,001	24,782	3,444	8	6,820	3,810	1,574	11,898
2004	1,503	766	1,388	4,621	6,680	10,597	8	4,796	6,804	795	8,850
2005	853	147	607	2,968	9,272	2,678	17	2,204	2,083	462	1,504

Table A2. Estimates of hatchery contribution in the natural spawners for Lower Columbia River tule Chinook populations. From LCR TRT August 2006 and updated for the Coweeman and Lewis, November 2006 and for the Grays February 2007 by WDFW. Values for Elochoman, Germany/Abernath/Mills, Wind, and White populations are 5 year averages due to the large annual fluctuations in data.

	Cowee-man	Grays	Lewis	Cowlitz	Kalama	Wash-ougal	Clat-skanie	Elocho-man	Ge/Ab/Mi	Wind	White Salmon
1973				74%	50%	55%	0%	59%		0%	0%
1974				74%	50%	55%	0%	59%		0%	0%
1975				74%	50%	55%	0%	59%		0%	0%
1976				74%	50%	55%	0%	59%		0%	0%
1977	7.2%	43%	3.4%	74%	50%	55%	0%	59%		0%	0%
1978	7.2%	43%	3.4%	74%	50%	55%	0%	59%		0%	0%
1979	7.2%	43%	3.4%	74%	50%	55%	0%	59%		0%	0%
1980	7.2%	43%	3.4%	74%	50%	55%	0%	59%	51%	0%	0%
1981	7.2%	43%	3.4%	74%	50%	55%	0%	59%	50%	0%	0%
1982	7.2%	43%	3.4%	74%	50%	55%	15%	59%	50%	0%	0%
1983	7.2%	43%	3.4%	74%	50%	55%	15%	59%	49%	0%	0%
1984	7.2%	43%	3.4%	74%	50%	55%	15%	59%	49%	0%	0%
1985	7.2%	43%	3.4%	74%	50%	55%	15%	59%	47%	0%	0%
1986	7.2%	43%	3.4%	74%	50%	55%	15%	59%	44%	0%	0%
1987	7.2%	43%	3.4%	74%	50%	55%	15%	59%	40%	0%	0%
1988	7.2%	43%	3.4%	74%	50%	55%	15%	59%	38%	0%	0%
1989	7.2%	43%	3.4%	74%	50%	55%	15%	65%	31%	0%	0%
1990	7.2%	43%	3.4%	74%	50%	55%	15%	53%	33%	0%	0%
1991	7.2%	43%	3.4%	74%	46%	53%	15%	46%	33%	0%	0%
1992	7.2%	43%	3.4%	74%	53%	24%	15%	35%	39%	0%	0%
1993	7.2%	43%	3.4%	94%	11%	48%	15%	33%	41%	0%	3%
1994	7.2%	43%	3.4%	81%	27%	30%	15%	22%	47%	0%	3%
1995	7.2%	43%	3.4%	87%	31%	61%	15%	40%	52%	0%	3%
1996	7.2%	43%	6.3%	42%	56%	83%	15%	50%	54%	0%	3%
1997	7.2%	24%	6.3%	29%	60%	88%	15%	65%	49%	13%	20%
1998	7.2%	24%	6.3%	63%	31%	76%	15%	62%	47%	13%	17%
1999	7.2%	43%	6.3%	84%	97%	32%	15%	59%	50%	27%	34%
2000	7.2%	43%	6.3%	90%	81%	30%	15%	61%	54%	34%	49%
2001	7.2%	16%	6.3%	56%	81%	57%	15%	53%	54%	46%	66%
2002	7.2%	16%	6.3%	24%	99%	53%	15%	58%	68%	39%	62%
2003	7.2%	16%	6.3%	12%	99%	61%	15%	69%	77%	47%	81%
2004	7.2%	16%	6.3%	30%	90%	75%	15%	82%	81%	42%	81%
2005	7.2%	16%	6.3%	83%	97%	59%	15%	76%	76%	45%	83%

Table A3. Age composition for the natural origin spawners for Lower Columbia River tule Chinook populations. From LCR TRT and WDFW.

	Coweeman					Grays				
	2	3	4	5	6	2	3	4	5	6
1973										
1974										
1975										
1976										
1977	4.1%	10.9%	45.3%	39.4%	0.3%	2.3%	32.5%	51.3%	13.8%	0.0%
1978	1.3%	34.0%	40.5%	23.6%	0.6%	0.7%	9.4%	74.0%	15.9%	0.0%
1979	1.1%	6.9%	78.6%	13.1%	0.2%	4.2%	5.8%	43.4%	46.6%	0.0%
1980	3.1%	11.9%	32.5%	52.2%	0.3%	3.3%	37.4%	29.4%	29.9%	0.0%
1981	0.2%	28.9%	50.6%	19.3%	1.0%	14.5%	10.5%	67.7%	7.3%	0.0%
1982	12.4%	1.2%	69.2%	17.0%	0.2%	0.2%	56.5%	23.1%	20.2%	0.0%
1983	3.7%	68.9%	3.0%	24.2%	0.2%	6.0%	0.7%	88.5%	4.9%	0.0%
1984	2.5%	10.4%	86.4%	0.5%	0.1%	2.8%	44.1%	2.8%	50.4%	0.0%
1985	0.7%	19.9%	36.7%	42.7%	0.0%	5.4%	9.4%	84.5%	0.7%	0.0%
1986	10.1%	5.1%	66.8%	17.3%	0.6%	5.9%	29.5%	29.0%	35.6%	0.0%
1987	3.4%	58.7%	13.3%	24.4%	0.2%	5.7%	22.3%	63.5%	8.5%	0.0%
1988	7.3%	15.3%	73.4%	4.0%	0.0%	1.5%	24.0%	53.7%	20.8%	0.0%
1989	3.0%	8.4%	33.0%	55.6%	0.0%	1.0%	7.7%	70.1%	21.3%	0.0%
1990	10.1%	25.7%	37.3%	22.8%	4.1%	5.1%	8.4%	38.6%	47.8%	0.0%
1991	0.0%	31.6%	37.9%	30.5%	0.0%	6.0%	36.0%	36.0%	22.0%	0.0%
1992	2.3%	7.4%	73.4%	15.7%	1.2%	0.0%	0.0%	100.0%	0.0%	0.0%
1993	6.6%	30.9%	35.4%	27.1%	0.0%	7.0%	37.2%	53.5%	2.3%	0.0%
1994	5.6%	31.5%	55.5%	7.4%	0.0%	0.0%	0.0%	25.5%	74.5%	0.0%
1995	2.5%	30.0%	51.9%	15.6%	0.0%	0.0%	0.0%	51.7%	48.3%	0.0%
1996	0.2%	15.4%	66.3%	18.1%	0.0%	3.8%	34.5%	57.8%	3.8%	0.0%
1997	0.0%	0.7%	61.9%	37.4%	0.0%	14.3%	14.3%	71.4%	0.0%	0.0%
1998	1.4%	8.2%	49.3%	41.1%	0.0%	0.0%	39.8%	43.0%	17.2%	0.0%
1999	3.1%	35.4%	45.8%	15.6%	0.0%	0.0%	35.6%	59.4%	5.0%	0.0%
2000	1.6%	17.2%	74.2%	7.0%	0.0%	8.2%	4.1%	76.3%	11.3%	0.0%
2001	2.2%	20.3%	68.1%	9.4%	0.0%	4.0%	49.8%	46.2%	0.0%	0.0%
2002	1.0%	25.7%	55.3%	18.0%	0.0%	4.9%	31.7%	47.6%	15.9%	0.0%
2003	0.7%	8.2%	66.1%	25.0%	0.0%	3.6%	10.6%	68.0%	17.8%	0.0%
2004	2.5%	7.9%	62.0%	27.6%	0.0%	2.6%	13.7%	73.7%	10.1%	0.0%
2005	1.1%	17.0%	47.9%	34.0%	0.0%	18.1%	33.6%	31.5%	16.8%	0.0%

Table A3. Continued.

	Lewis					Cowlitz				
	2	3	4	5	6	2	3	4	5	6
1973						*4.2%*	*20.9%*	*43.7%*	*30.2%*	*0.9%*
1974						*4.3%*	*11.5%*	*65.5%*	*17.8%*	*0.9%*
1975						*9.7%*	*14.0%*	*43.4%*	*32.2%*	*0.6%*
1976						*1.5%*	*29.4%*	*48.5%*	*19.6%*	*1.0%*
1977	6.0%	37.6%	47.4%	9.1%	0.0%	*13.3%*	*3.1%*	*68.4%*	*14.7%*	*0.4%*
1978	29.0%	19.1%	44.7%	7.1%	0.0%	*0.7%*	*48.6%*	*13.0%*	*37.2%*	*0.6%*
1979	12.0%	29.7%	45.0%	13.3%	0.0%	*6.3%*	*1.0%*	*88.9%*	*3.1%*	*0.6%*
1980	40.9%	12.9%	39.4%	6.8%	0.0%	*2.9%*	*29.4%*	*5.6%*	*62.0%*	*0.2%*
1981	9.4%	8.9%	68.7%	13.0%	0.0%	*7.7%*	*6.9%*	*81.7%*	*2.0%*	*1.6%*
1982	30.6%	32.4%	35.5%	1.4%	0.0%	*2.1%*	*27.1%*	*27.8%*	*42.9%*	*0.1%*
1983	8.7%	10.5%	70.4%	10.5%	0.0%	*12.0%*	*4.8%*	*72.4%*	*9.7%*	*1.1%*
1984	7.1%	8.9%	76.8%	7.1%	0.0%	*3.7%*	*40.7%*	*18.6%*	*36.6%*	*0.4%*
1985	17.4%	21.1%	46.2%	15.3%	0.0%	*9.6%*	*6.3%*	*78.7%*	*4.7%*	*0.7%*
1986	12.6%	39.3%	41.1%	7.0%	0.0%	*11.0%*	*29.9%*	*22.3%*	*36.6%*	*0.2%*
1987	13.9%	24.1%	44.3%	17.7%	0.0%	*9.7%*	*20.3%*	*63.1%*	*6.1%*	*0.7%*
1988	10.1%	14.3%	58.2%	17.3%	0.0%	*2.0%*	*22.5%*	*53.6%*	*21.8%*	*0.2%*
1989	4.3%	14.0%	38.4%	43.3%	0.0%	*1.9%*	*5.5%*	*70.2%*	*21.8%*	*0.6%*
1990	4.5%	18.7%	30.3%	24.5%	22.0%	*5.4%*	*9.5%*	*31.3%*	*52.6%*	*1.2%*
1991	7.9%	31.7%	31.7%	23.7%	5.0%	*2.4%*	*24.6%*	*49.2%*	*21.2%*	*2.6%*
1992	5.6%	15.7%	69.4%	9.3%	0.0%	*7.6%*	*5.9%*	*68.2%*	*17.8%*	*0.6%*
1993	7.1%	23.8%	48.8%	20.2%	0.0%	*3.1%*	*29.8%*	*26.4%*	*40.0%*	*0.8%*
1994	25.1%	6.1%	52.1%	16.7%	0.0%	*18.0%*	*6.1%*	*67.3%*	*7.8%*	*0.8%*
1995	9.9%	16.2%	26.1%	47.7%	0.0%	*28.1%*	*36.8%*	*14.3%*	*20.6%*	*0.2%*
1996	1.2%	19.2%	70.1%	9.6%	0.0%	*5.0%*	*36.7%*	*55.2%*	*2.8%*	*0.3%*
1997	0.0%	2.2%	62.0%	35.9%	0.0%	*0.6%*	*8.9%*	*75.6%*	*14.8%*	*0.1%*
1998	5.5%	49.1%	23.6%	21.8%	0.0%	*0.9%*	*2.7%*	*45.4%*	*50.2%*	*0.7%*
1999	2.7%	45.0%	42.3%	9.9%	0.0%	*29.3%*	*5.8%*	*19.5%*	*42.0%*	*3.3%*
2000	5.6%	14.9%	64.6%	14.9%	0.0%	*34.9%*	*48.4%*	*11.1%*	*4.8%*	*0.7%*
2001	0.6%	39.9%	56.6%	2.9%	0.0%	*10.8%*	*33.8%*	*53.8%*	*1.6%*	*0.1%*
2002	5.0%	19.6%	67.3%	8.0%	0.1%	*7.6%*	*17.3%*	*62.2%*	*12.9%*	*0.0%*
2003	1.9%	16.3%	61.2%	20.5%	0.0%	*0.6%*	*20.5%*	*53.4%*	*25.1%*	*0.4%*
2004	3.5%	6.0%	71.5%	18.5%	0.4%	*0.3%*	*1.9%*	*72.4%*	*24.6%*	*0.8%*
2005	1.5%	12.6%	40.9%	43.1%	2.0%	*0.8%*	*2.0%*	*16.0%*	*79.3%*	*1.9%*

Table A3. Continued.

	Kalama					Washougal				
	2	3	4	5	6	2	3	4	5	6
1973	*4.4%*	*49.0%*	*38.9%*	*7.7%*	*0.0%*	*0.2%*	*65.3%*	*2.1%*	*32.4%*	*0.0%*
1974	*2.2%*	*42.7%*	*49.7%*	*5.3%*	*0.0%*	*11.1%*	*0.1%*	*88.5%*	*0.2%*	*0.0%*
1975	*1.7%*	*29.6%*	*59.3%*	*9.3%*	*0.0%*	*6.6%*	*37.4%*	*1.0%*	*55.1%*	*0.0%*
1976	*2.7%*	*30.0%*	*53.0%*	*14.3%*	*0.0%*	*0.6%*	*7.3%*	*91.9%*	*0.2%*	*0.0%*
1977	*0.3%*	*40.9%*	*47.5%*	*11.3%*	*0.0%*	*10.4%*	*1.5%*	*43.0%*	*45.1%*	*0.0%*
1978	*4.1%*	*6.0%*	*77.7%*	*12.1%*	*0.0%*	*22.0%*	*37.3%*	*12.2%*	*28.5%*	*0.0%*
1979	*3.6%*	*67.9%*	*10.4%*	*18.0%*	*0.0%*	*5.7%*	*19.5%*	*72.9%*	*2.0%*	*0.0%*
1980	*1.4%*	*33.0%*	*64.3%*	*1.3%*	*0.0%*	*0.1%*	*9.1%*	*69.2%*	*21.6%*	*0.0%*
1981	*4.5%*	*22.7%*	*57.7%*	*15.1%*	*0.0%*	*12.9%*	*0.2%*	*53.3%*	*33.7%*	*0.0%*
1982	*2.1%*	*57.4%*	*30.2%*	*10.3%*	*0.0%*	*4.2%*	*53.3%*	*1.7%*	*40.7%*	*0.0%*
1983	*1.3%*	*24.1%*	*69.7%*	*4.9%*	*0.0%*	*7.8%*	*3.2%*	*88.8%*	*0.2%*	*0.0%*
1984	*0.4%*	*27.3%*	*52.0%*	*20.2%*	*0.0%*	*5.0%*	*23.7%*	*21.4%*	*49.9%*	*0.0%*
1985	*8.4%*	*10.1%*	*64.9%*	*16.6%*	*0.0%*	*11.1%*	*7.3%*	*75.9%*	*5.7%*	*0.0%*
1986	*5.1%*	*77.0%*	*9.6%*	*8.3%*	*0.0%*	*9.0%*	*24.6%*	*35.5%*	*30.9%*	*0.0%*
1987	*4.4%*	*36.7%*	*58.0%*	*1.0%*	*0.0%*	*9.9%*	*11.6%*	*70.1%*	*8.5%*	*0.0%*
1988	*1.3%*	*47.8%*	*42.0%*	*8.9%*	*0.0%*	*2.4%*	*19.9%*	*51.7%*	*26.0%*	*0.0%*
1989	*2.0%*	*18.7%*	*70.9%*	*8.4%*	*0.0%*	*3.7%*	*4.1%*	*75.7%*	*16.4%*	*0.0%*
1990	*1.7%*	*40.1%*	*38.6%*	*19.6%*	*0.0%*	*17.9%*	*11.3%*	*28.0%*	*42.8%*	*0.0%*
1991	2.4%	22.8%	62.8%	12.0%	0.0%	10.3%	44.5%	45.2%	0.0%	0.0%
1992	5.1%	41.1%	41.8%	12.0%	0.0%	12.9%	5.1%	82.0%	0.0%	0.0%
1993	1.1%	9.7%	76.4%	12.8%	0.0%	4.3%	0.0%	62.6%	33.1%	0.0%
1994	10.6%	69.2%	9.6%	10.6%	0.0%	9.6%	5.9%	84.5%	0.0%	0.0%
1995	2.4%	27.2%	59.1%	11.3%	0.0%	7.8%	54.8%	0.0%	37.4%	0.0%
1996	0.0%	42.0%	49.1%	8.9%	0.0%	20.6%	0.0%	51.4%	28.0%	0.0%
1997	*2.6%*	*19.1%*	*67.8%*	*10.5%*	*0.0%*	25.8%	8.7%	65.5%	0.0%	0.0%
1998	1.2%	54.5%	28.2%	16.1%	0.0%	14.9%	27.7%	38.4%	19.1%	0.0%
1999	*1.1%*	*4.1%*	*86.0%*	*8.8%*	*0.0%*	1.1%	22.8%	71.5%	4.6%	0.0%
2000	*2.5%*	*40.3%*	*14.9%*	*42.3%*	*0.0%*	4.4%	5.9%	89.7%	0.0%	0.0%
2001	2.4%	33.7%	39.9%	24.0%	0.0%	9.7%	24.8%	55.2%	10.3%	0.0%
2002	1.3%	59.2%	39.4%	0.0%	0.0%	5.4%	17.6%	71.3%	5.7%	0.0%
2003	3.0%	19.0%	72.9%	5.1%	0.0%	0.1%	6.3%	79.8%	13.9%	0.0%
2004	1.7%	41.3%	57.0%	0.0%	0.0%	1.8%	7.4%	61.3%	29.5%	0.0%
2005	0.0%	11.1%	82.4%	6.6%	0.0%	0.3%	11.0%	51.2%	37.5%	0.0%

Table A3. Continued.

	Clatskanie					Elochoman				
	2	3	4	5	6	2	3	4	5	6
1973	*0.0%*	*65.3%*	*34.3%*	*0.5%*	*0.0%*	0.1%	*30.1%*	*69.3%*	*0.5%*	*0.0%*
1974	*0.0%*	*79.3%*	*20.1%*	*0.6%*	*0.0%*	9.7%	*3.4%*	*70.3%*	*16.5%*	*0.0%*
1975	*0.0%*	*82.4%*	*17.3%*	*0.2%*	*0.0%*	2.0%	*90.2%*	*2.5%*	*5.3%*	*0.0%*
1976	*0.0%*	*22.9%*	*76.2%*	*0.9%*	*0.0%*	1.4%	*21.7%*	*76.7%*	*0.2%*	*0.0%*
1977	*0.0%*	*79.6%*	*17.2%*	*3.2%*	*0.0%*	4.8%	*36.3%*	*43.2%*	*15.6%*	*0.0%*
1978	*0.0%*	*83.3%*	*16.5%*	*0.2%*	*0.0%*	0.0%	*59.9%*	*35.7%*	*4.3%*	*0.0%*
1979	*0.0%*	*87.7%*	*12.2%*	*0.1%*	*0.0%*	0.4%	*0.9%*	*93.1%*	*5.7%*	*0.0%*
1980	*0.0%*	*73.0%*	*26.7%*	*0.2%*	*0.0%*	0.3%	*30.3%*	*6.1%*	*63.3%*	*0.0%*
1981	*0.0%*	*66.2%*	*33.1%*	*0.7%*	*0.0%*	20.5%	*8.5%*	*69.6%*	*1.4%*	*0.0%*
1982	*0.0%*	*83.0%*	*16.5%*	*0.5%*	*0.0%*	1.4%	*93.0%*	*3.0%*	*2.6%*	*0.0%*
1983	*0.0%*	*53.4%*	*46.1%*	*0.5%*	*0.0%*	1.0%	*15.5%*	*83.3%*	*0.3%*	*0.0%*
1984	*0.0%*	*32.2%*	*64.7%*	*3.1%*	*0.0%*	3.2%	*32.5%*	*41.5%*	*22.8%*	*0.0%*
1985	*0.0%*	*94.2%*	*5.2%*	*0.6%*	*0.0%*	4.0%	*50.1%*	*40.6%*	*5.3%*	*0.0%*
1986	*0.0%*	*75.4%*	*24.6%*	*0.1%*	*0.0%*	4.3%	*46.4%*	*45.5%*	*3.8%*	*0.0%*
1987	*0.0%*	*79.4%*	*20.2%*	*0.4%*	*0.0%*	0.0%	*51.5%*	*44.1%*	*4.4%*	*0.0%*
1988	*0.0%*	*63.8%*	*35.7%*	*0.5%*	*0.0%*	0.4%	*0.9%*	*90.7%*	*7.9%*	*0.0%*
1989	*0.0%*	*78.5%*	*20.8%*	*0.7%*	*0.0%*	2.2%	*31.8%*	*6.1%*	*59.9%*	*0.0%*
1990	*0.0%*	*44.9%*	*54.3%*	*0.8%*	*0.0%*	6.0%	*43.8%*	*49.2%*	*1.0%*	*0.0%*
1991	*0.0%*	*48.3%*	*48.4%*	*3.3%*	*0.0%*	2.8%	*59.1%*	*34.1%*	*3.9%*	*0.0%*
1992	*0.0%*	*71.4%*	*27.0%*	*1.5%*	*0.0%*	0.0%	10.0%	90.0%	0.0%	0.0%
1993	*0.0%*	*80.1%*	*19.5%*	*0.4%*	*0.0%*	6.3%	8.0%	83.9%	1.8%	0.0%
1994	*0.0%*	*88.9%*	*11.0%*	*0.2%*	*0.0%*	2.6%	82.1%	14.6%	0.7%	0.0%
1995	*0.0%*	*73.6%*	*26.2%*	*0.2%*	*0.0%*	7.7%	24.4%	55.1%	12.8%	0.0%
1996	*0.0%*	*86.1%*	*13.6%*	*0.3%*	*0.0%*	7.2%	69.3%	21.2%	2.3%	0.0%
1997	*0.0%*	*25.2%*	*74.2%*	*0.7%*	*0.0%*	0.3%	23.9%	73.7%	2.1%	0.0%
1998	*0.0%*	*91.1%*	*7.6%*	*1.3%*	*0.0%*	8.8%	61.3%	24.6%	5.3%	0.0%
1999	*0.0%*	*67.6%*	*32.3%*	*0.2%*	*0.0%*	6.1%	73.9%	0.0%	20.0%	0.0%
2000	*0.0%*	*34.0%*	*64.2%*	*1.7%*	*0.0%*	0.0%	38.0%	58.7%	3.3%	0.0%
2001	*0.0%*	*59.8%*	*36.4%*	*3.9%*	*0.0%*	14.0%	46.9%	34.9%	4.2%	0.0%
2002	*0.0%*	*98.2%*	*1.8%*	*0.1%*	*0.0%*	0.7%	38.5%	59.9%	0.9%	0.0%
2003	*0.0%*	*65.1%*	*34.8%*	*0.0%*	*0.0%*	0.8%	27.7%	65.9%	5.6%	0.0%
2004	*0.0%*	*22.8%*	*75.0%*	*2.3%*	*0.0%*	0.3%	15.2%	79.2%	5.3%	0.0%
2005	*0.0%*	*80.7%*	*16.3%*	*3.0%*	*0.0%*	1.4%	22.9%	48.5%	27.2%	0.0%

Table A3. Continued.

	Germany/Abernathy/Mills					Wind				
	2	3	4	5	6	2	3	4	5	6
1973						0.0%				
1974						0.0%				
1975						*0.6%*	*34.8%*	*49.8%*	*14.8%*	*0.0%*
1976						*0.7%*	*36.2%*	*53.1%*	*10.1%*	*0.0%*
1977						*0.6%*	*38.5%*	*51.0%*	*9.9%*	*0.0%*
1978						*0.3%*	*34.3%*	*55.6%*	*9.8%*	*0.0%*
1979						*0.3%*	*22.4%*	*63.6%*	*13.7%*	*0.0%*
1980	*15%*	*31%*	*53%*	*1%*	*0%*	*0.3%*	*27.8%*	*52.2%*	*19.7%*	*0.0%*
1981	*0%*	*51%*	*42%*	*7%*	*0%*	*1.1%*	*25.3%*	*59.0%*	*14.7%*	*0.0%*
1982	*7%*	*0%*	*86%*	*7%*	*0%*	*0.3%*	*56.0%*	*33.4%*	*10.3%*	*0.0%*
1983	*0%*	*65%*	*1%*	*34%*	*0%*	*0.0%*	*15.6%*	*78.2%*	*6.2%*	*0.0%*
1984	*13%*	*0%*	*86%*	*0%*	*0%*	*1.6%*	*2.3%*	*57.7%*	*38.4%*	*0.0%*
1985	*6%*	*74%*	*1%*	*19%*	*0%*	*0.5%*	*86.1%*	*3.1%*	*10.4%*	*0.0%*
1986	*7%*	*16%*	*76%*	*0%*	*0%*	*1.3%*	*17.6%*	*80.7%*	*0.4%*	*0.0%*
1987	*2%*	*44%*	*37%*	*18%*	*0%*	*0.1%*	*64.5%*	*22.0%*	*13.4%*	*0.0%*
1988	*2%*	*8%*	*83%*	*7%*	*0%*	*0.0%*	*4.2%*	*91.6%*	*4.1%*	*0.0%*
1989	*12%*	*18%*	*34%*	*36%*	*0%*	*0.1%*	*3.3%*	*24.9%*	*71.7%*	*0.0%*
1990	*8%*	*74%*	*18%*	*1%*	*0%*	*2.7%*	*26.3%*	*35.5%*	*35.5%*	*0.0%*
1991	9%	77%	9%	5%	0%	10.3%	75.9%	13.8%	0.0%	0.0%
1992	7%	29%	62%	1%	0%	0.0%	50.0%	50.0%	0.0%	0.0%
1993	2%	21%	64%	14%	0%	*0.3%*	*17.8%*	*36.0%*	*46.0%*	*0.0%*
1994	3%	44%	30%	23%	0%	0.0%	72.7%	27.3%	0.0%	0.0%
1995	31%	22%	46%	1%	0%	0.0%	75.0%	25.0%	0.0%	0.0%
1996	1%	33%	50%	17%	0%	0.0%	72.9%	27.1%	0.0%	0.0%
1997	6%	9%	66%	20%	0%	0.0%	26.4%	66.8%	6.8%	0.0%
1998	0%	61%	34%	5%	0%	5.2%	18.8%	66.6%	9.4%	0.0%
1999	11%	29%	56%	4%	0%	*0.7%*	*14.8%*	*60.0%*	*24.5%*	*0.0%*
2000	8%	40%	43%	9%	0%	14.3%	28.6%	35.7%	21.4%	0.0%
2001	1%	28%	70%	1%	0%	3.3%	46.7%	45.8%	4.2%	0.0%
2002	0%	21%	74%	5%	0%	2.7%	66.7%	25.6%	5.1%	0.0%
2003	0%	11%	85%	4%	0%	4.8%	30.2%	63.5%	1.6%	0.0%
2004	1%	15%	58%	26%	0%	1.4%	17.3%	63.2%	18.1%	0.0%
2005	0%	0%	0%	0%	0%	2.2%	19.0%	64.1%	14.7%	0.0%

Table A3. Continued.

	White Salmon				
	2	3	4	5	6
1973	*4.0%*	*53.3%*	*30.9%*	*11.7%*	*0.0%*
1974	*9.1%*	*41.8%*	*41.9%*	*7.2%*	*0.0%*
1975	*4.5%*	*65.9%*	*22.8%*	*6.8%*	*0.0%*
1976	*0.1%*	*45.1%*	*49.7%*	*5.1%*	*0.0%*
1977	*13.6%*	*1.6%*	*64.0%*	*20.9%*	*0.0%*
1978	*0.5%*	*89.3%*	*0.8%*	*9.4%*	*0.0%*
1979	*11.8%*	*5.8%*	*82.2%*	*0.2%*	*0.0%*
1980	*1.2%*	*82.8%*	*3.1%*	*12.9%*	*0.0%*
1981	*9.1%*	*14.2%*	*75.8%*	*0.8%*	*0.0%*
1982	*0.1%*	*76.6%*	*9.1%*	*14.3%*	*0.0%*
1983	*5.5%*	*1.0%*	*90.3%*	*3.2%*	*0.0%*
1984	*0.2%*	*72.3%*	*1.0%*	*26.5%*	*0.0%*
1985	*4.2%*	*2.9%*	*92.5%*	*0.4%*	*0.0%*
1986	*6.6%*	*60.7%*	*3.1%*	*29.6%*	*0.0%*
1987	*14.6%*	*49.9%*	*34.9%*	*0.5%*	*0.0%*
1988	*0.6%*	*75.8%*	*19.5%*	*4.0%*	*0.0%*
1989	*3.3%*	*8.8%*	*81.6%*	*6.2%*	*0.0%*
1990	*15.2%*	*48.3%*	*9.8%*	*26.7%*	*0.0%*
1991	10.7%	76.0%	13.3%	0.0%	0.0%
1992	*1.0%*	*61.9%*	*34.7%*	*2.5%*	*0.0%*
1993	2.8%	19.4%	58.4%	19.4%	0.0%
1994	1.7%	72.3%	26.0%	0.0%	0.0%
1995	16.0%	81.7%	0.0%	2.3%	0.0%
1996	0.0%	75.0%	25.0%	0.0%	0.0%
1997	0.0%	37.9%	62.1%	0.0%	0.0%
1998	7.9%	23.1%	53.7%	15.3%	0.0%
1999	*7.7%*	*60.8%*	*24.7%*	*6.8%*	*0.0%*
2000	12.0%	29.3%	35.3%	23.4%	0.0%
2001	5.1%	41.5%	47.4%	6.0%	0.0%
2002	3.9%	71.4%	24.0%	0.6%	0.0%
2003	7.9%	15.0%	72.9%	4.2%	0.0%
2004	1.8%	52.7%	44.6%	0.9%	0.0%
2005	3.7%	37.6%	54.8%	3.9%	0.0%

Table A4. Harvest rate estimates for the three indicator stock groups for harvest. From PSC CTC CWT exploitation rate analysis.

Adjusted Cowlitz

Brood Year	Mixed Maturity Fishery Fishing Rate by Total Age (a)				Mature Fishery Fishing Rate by Total Age (b)			
	2	3	4	5+	2	3	4	5+
1977	0.0516	0.3341	0.4761	0.2869	0.0000	0.5569	0.3293	0.0000
1978	0.0426	0.2103	0.5775	0.5269	0.0894	0.1920	0.1778	0.0000
1979	0.0619	0.2593	0.5334	0.6765	0.7583	0.3265	0.1275	0.0000
1980	0.0435	0.2103	0.3848	0.2097	0.2042	0.1081	0.2840	0.0859
1981	0.0281	0.0728	0.3668	0.2155	0.5644	0.3984	0.0725	0.2764
1982	0.0386	0.1874	0.3133	0.3692	0.4815	0.3734	0.4098	0.5632
1983	0.0399	0.2192	0.3675	0.2954	0.4277	0.6494	0.4413	0.4224
1984	0.0283	0.1307	0.3294	0.2605	0.2677	0.5211	0.5158	0.1358
1985	0.0446	0.1102	0.4120	0.5445	0.3552	0.8064	0.3130	0.0000
1986	0.0381	0.1018	0.3841	0.2264	0.0000	0.2584	0.0285	0.2998
1987	0.0272	0.2058	0.1849	0.2852	0.0000	0.0000	0.1998	0.0000
1988	0.0306	0.1604	0.2435	0.4427	0.0000	0.0879	0.0357	0.0909
1989	0.0461	0.1914	0.6807	0.2422	0.6815	0.0000	0.0712	0.0000
1990	0.0129	0.1005	0.1098	0.3171	0.1108	0.4152	0.0000	0.0738
1991	0.0291	0.0000	0.0241	0.4445	0.0000	0.0000	0.0437	0.0000
1992	0.0092	0.0536	0.1113	0.2547	0.0000	0.0000	0.0000	0.0000
1993	0.0348	0.0432	0.2871	0.5533	0.0000	0.1331	0.0000	0.0000
1994	0.0132	0.0382	0.2602	0.8737	0.0000	0.1625	0.0000	0.0000
1995	0.0172	0.0267	0.1888	0.3310	0.0000	0.0229	0.1771	0.3331
1996	0.0224	0.0990	0.5110	0.4191	0.0000	0.2309	0.2572	0.2453
1997	0.0175	0.0709	0.3006	0.6174	0.0000	0.0000	0.0415	0.1226
1998	0.0315	0.1497	0.5639	0.8157	0.1483	0.0771	0.1077	0.0000
1999	0.0273	0.1962	0.3740	0.3517	0.0311	0.3268	0.1671	0.1185
2000	0.0157	0.0780	0.5639	0.5949	0.0000	0.6901	0.0735	0.0804
2001	0.0103	0.1425	0.5006	0.5874	0.0000	0.3414	0.1161	0.0663
2002	0.0000	0.1389	0.4795	0.5114	0.0000	0.4528	0.1189	0.0884

Table A4. Continued.

Cowlitz, Grays, Washougal Composite

Brood Year	Mixed Maturity Fishery Fishing Rate by Total Age (a)				Mature Fishery Fishing Rate by Total Age (b)			
	2	3	4	5+	2	3	4	5+
1973	0.0868	0.4939	0.4928	0.5262	0.0000	0.0000	0.7362	0.7295
1974	0.0516	0.4812	0.5613	0.2920	0.0000	0.9436	0.7768	0.5813
1975	0.0950	0.5386	0.4231	0.3183	0.0000	0.0000	0.2699	0.3994
1976	0.0767	0.3756	0.6755	0.2190	0.0000	0.3165	0.5250	0.4545
1977	0.1109	0.4336	0.3383	0.5885	0.1829	0.7097	0.4584	0.1488
1978	0.0502	0.2766	0.5098	0.5938	0.3517	0.3985	0.4092	0.0818
1979	0.0548	0.3144	0.4782	0.7362	0.2523	0.4372	0.1470	0.1189
1980	0.0475	0.1824	0.4007	0.4166	0.2688	0.0593	0.3640	0.1898
1981	0.0409	0.1526	0.2555	0.1669	0.1878	0.6877	0.1061	0.3143
1982	0.0347	0.1526	0.3518	0.4914	0.1599	0.4600	0.5681	0.6297
1983	0.0392	0.1995	0.3864	0.3283	0.3215	0.6840	0.6048	0.6021
1984	0.0341	0.1095	0.3493	0.3411	0.2238	0.6561	0.4155	0.2692
1985	0.0506	0.1464	0.4026	0.4966	0.2714	0.7590	0.3091	0.0900
1986	0.0375	0.1052	0.3580	0.3605	0.0000	0.2631	0.1576	0.1665
1987	0.0353	0.1960	0.2586	0.4078	0.0000	0.0512	0.2206	0.0000
1988	0.0363	0.3053	0.1362	0.2389	0.0000	0.0553	0.0217	0.0526
1989	0.0394	0.1538	0.4900	0.4681	0.4476	0.1656	0.2050	0.0000
1990	0.0747	0.1118	0.2181	0.3271	0.2337	0.3824	0.0000	0.0818
1991	0.0604	0.0130	0.2305	0.1482	0.0000	0.0000	0.0148	0.0000
1992	0.0109	0.0405	0.0738	0.1307	0.0000	0.0639	0.0541	0.1628
1993	0.0273	0.0255	0.2026	0.2631	0.0000	0.2149	0.2116	0.0311
1994	0.0067	0.0525	0.1884	0.6111	0.0000	0.3725	0.0282	0.0000
1995	0.0237	0.0894	0.2216	0.2393	0.0000	0.0997	0.1782	0.1665
1996	0.0237	0.0584	0.2881	0.2646	0.0633	0.2329	0.2227	0.1226
1997	0.0132	0.0687	0.1970	0.0499	0.0000	0.1349	0.0664	0.0000
1998	0.0268	0.1111	0.4402	0.6035	0.0741	0.1324	0.1568	0.1049
1999	0.0255	0.1410	0.3830	0.3810	0.0174	0.2683	0.1622	0.1345
2000	0.0106	0.0512	0.4583	0.3448	0.0000	0.4287	0.0827	0.0798
2001	0.0055	0.1528	0.4271	0.4431	0.0000	0.2765	0.1339	0.1064
2002	0.0000	0.1150	0.4228	0.3897	0.0000	0.3245	0.1263	0.1069

Table A4. Continued.

Cowlitz, Grays, Washougal, Big Creek Composite

Brood Year	Mixed Maturity Fishery Fishing Rate by Total Age (a)				Mature Fishery Fishing Rate by Total Age (b)			
	2	3	4	5+	2	3	4	5+
1973	0.0868	0.4939	0.4928	0.5262	0.0000	0.0000	0.7362	0.7295
1974	0.0516	0.4812	0.5613	0.2920	0.0000	0.9436	0.7768	0.5813
1975	0.0950	0.5386	0.4231	0.3183	0.0000	0.0000	0.2699	0.3994
1976	0.1015	0.4726	0.5876	0.1460	0.0000	0.3172	0.6043	0.3030
1977	0.1215	0.5111	0.3653	0.5882	0.1371	0.7786	0.4859	0.3608
1978	0.0640	0.3542	0.4864	0.5987	0.2638	0.4096	0.4387	0.0613
1979	0.0747	0.4015	0.4496	0.5521	0.3886	0.4731	0.1808	0.0892
1980	0.0647	0.2886	0.3898	0.3125	0.2016	0.1346	0.3055	0.1424
1981	0.0477	0.2320	0.2360	0.1251	0.1408	0.6142	0.1134	0.3856
1982	0.0347	0.1526	0.3518	0.4914	0.1599	0.4600	0.5681	0.6297
1983	0.0392	0.1995	0.3864	0.3283	0.3215	0.6840	0.6048	0.6021
1984	0.0341	0.1095	0.3493	0.3411	0.2238	0.6561	0.4155	0.2692
1985	0.0506	0.1464	0.4026	0.4966	0.2714	0.7590	0.3091	0.0900
1986	0.0632	0.2651	0.3325	0.2404	0.2466	0.2423	0.1171	0.1110
1987	0.0561	0.2844	0.2997	0.2719	0.0000	0.0988	0.1737	0.0000
1988	0.0511	0.3493	0.1851	0.1593	0.0000	0.1867	0.0518	0.0351
1989	0.0394	0.1538	0.4900	0.4681	0.4476	0.1656	0.2050	0.0000
1990	0.0826	0.1744	0.2320	0.2453	0.1753	0.3758	0.0000	0.0614
1991	0.0687	0.0814	0.2238	0.1111	0.0000	0.0000	0.0111	0.0000
1992	0.0127	0.0745	0.0786	0.0980	0.0000	0.0553	0.1950	0.1221
1993	0.0335	0.0522	0.2762	0.1973	0.0000	0.2224	0.2204	0.1482
1994	0.0083	0.0886	0.1495	0.4074	0.0546	0.3028	0.0565	0.0000
1995	0.0177	0.0671	0.1478	0.1595	0.0000	0.0748	0.1363	0.1110
1996	0.0233	0.0978	0.3743	0.1764	0.1385	0.2065	0.1639	0.0817
1997	0.0145	0.1282	0.2040	0.0250	0.0000	0.1004	0.0638	0.0000
1998	0.0268	0.1111	0.4402	0.6035	0.0741	0.1324	0.1568	0.1049
1999	0.0292	0.1897	0.4017	0.3810	0.0655	0.2887	0.1521	0.1345
2000	0.0186	0.1667	0.4583	0.3365	0.0793	0.3610	0.0827	0.0798
2001	0.0094	0.1528	0.4334	0.4403	0.0000	0.3822	0.1305	0.1064
2002	0.0000	0.1697	0.4311	0.3860	0.0000	0.3440	0.1218	0.1069

TableA5. Marine survival indices for the three hatchery indicator stock groups.

Brood Year	Adjusted Cowlitz	Cowlitz, Grays, Washougal Composite	Cowlitz, Grays, Washougal, Big Creek Composite
1975		2.21	2.21
1976		1.45	1.45
1977	1.49	0.80	0.80
1978	1.05	0.51	0.51
1979	0.58	0.51	0.51
1980	1.70	0.86	0.86
1981	0.87	0.46	0.46
1982	1.21	0.75	0.75
1983	3.60	3.10	3.10
1984	4.68	3.84	3.84
1985	0.98	1.20	1.20
1986	0.64	0.49	0.49
1987	0.19	0.28	0.28
1988	0.47	0.25	0.25
1989	0.34	0.27	0.27
1990	0.73	0.42	0.42
1991	0.30	0.15	0.15
1992	0.47	0.34	0.34
1993	0.50	0.38	0.38
1994	0.08	0.07	0.07
1995	0.32	0.21	0.21
1996	0.18	0.13	0.13
1997	0.24	0.29	0.29
1998	0.67	1.01	1.01
1999	1.71	1.65	1.65
2000	0.76	0.31	0.31
2001	0.77	0.41	0.41
2002	0.59	0.36	0.36

Appendix B – Age Engine

Age Engine

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Draft 9/28/07

Introduction

Cohort run reconstruction for Pacific salmon populations requires age distribution estimates for the natural spawning escapements. In the case when there are both natural origin salmon and first generation hatchery salmon spawning naturally, we also need to know the hatchery percentage contribution, so we can remove the hatchery fish from our reconstructions of recruits, and the age distribution of the remaining natural origin salmon. Often, in the Pacific Northwest, we have escapement estimates for many more years than we have age composition data, and we need to estimate the age composition for the years without direct sampling data.

Assuming either a constant calendar year age composition or a constant brood year age composition for years without sampling estimates is not a very satisfactory solution due to the variability seen in escapement abundance from year to year and the variability in age composition seen for populations with a number of years of age sampling data.

The "age engine" was developed to make these estimates for Puget Sound Chinook run reconstructions and is a part of the Abundance and Productivity (A&P) Tables, an excel that is used for cohort run reconstruction. It has since been used for Hood Canal summer chum (Sands et al 2007).

Methods

We start by using existing data for annual age compositions of the natural origin escapements for a population. If enough years of continuous data are available, estimates of cohort age composition may be made. Otherwise, the average annual age composition may be used as a starting point for the annual cohort age composition estimation. The age composition of a returning cohort to the spawning ground is influenced both by biological tendencies of the population, influences on the juveniles in freshwater, and by calendar year effects such as fishery pressures, ocean conditions, and annual prey events. This is evident from the large variability in annual returns to escapement that support neither a constant cohort nor annual age distribution. Therefore, for each brood year, a fixed starting age composition is weighted according to the relative abundance of the escapements for years in which the cohort returns. Weights (w) are calculated for each calendar year (t):

$$w_t = s_t / s.$$

where s_t is the escapement for year t and *s.* is the average escapement over all years. Just using the weights results in calendar age distribution that approach being a constant. To get an estimate between a constant cohort age composition and a constant annual age composition we calculate the factor (*f*):

$$f_t = (w_t + 1) / 2$$

Within a cohort, these factors are applied to the fixed starting cohort age distributions for the return years for the cohort; this updated cohort age distribution is then adjusted to sum to 1.

Initial guesses are used for the cohort progeny spawner abundance and this is then multiplied by the adjusted cohort age distribution to get age specific abundances. The age specific abundances are summed across calendar years and compared to the observed escapement. This may be done over all ages (2-6) or just for ages 3-6 depending what is reported in the observed escapement. The error (difference) between the predicted and observed calendar year escapement is calculated. This may then be raised to a chosen power (1 for absolute difference, 2 for squared difference, etc.) and is summed to calculate the over all error to be minimized.

Calendar years with observed data are indicated in the calculation matrix and are not changed. Since, for a cohort with fixed observed age components, changing the input cohort escapement size only influences the age components for years without data. In the minimization process this could lead to large differences in the input cohort size and that obtained by summing over the component years. Therefore, the difference between the two is also calculated and raised to the error power already indicated and added to the total error to be minimized.

The EXCEL solver is used to change the cohort escapement sizes until the error is minimized. It is a good idea to start by using the average calendar escapement size for a starting point; after the age engine has been run and one is updating just by adding a new year, the past estimates are a good starting point and generally do not change much with minor changes/updates to estimated calendar year escapements and hatchery contribution estimates.

Minimizing on the squared error gives a lower total error, but the error (e.g., greater than 1 fish) is distributed over more years than using the absolute error. The solver solution using the squared error is faster to reach than using the absolute error. A recommendation is to first use the squared error to find a solution, and then use the absolute error; this results in few years having a difference in predicted v. observed escapement being greater than 1.

The calendar year age distributions are then calculated using the cohort age sample sizes over years contribution to the calendar year. The calendar year age distribution for years with observed data (so indicated) remains the same and the distributions are estimated for missing years of data.

The above procedure describes the mixed-model method, also described as the half and half method being a type of average between a constant cohort age distribution and a constant calendar year age distribution. The age engine can also be run using a constant cohort age distribution for calculating annual age distributions for years with missing data.

Options

Changeable model input parameters include:

1) Choosing to test on adult or total escapement, depending on which is the provided as “observed” data.
2) Indicating which years, if any, to using as the fixed, observed calendar year age distributions.
3) The starting age distribution for the cohort is generally calculated from observed data, but may be entered separately.
4) Choosing the constant brood year, constant calendar year, or mixed model age method for estimating missing calendar year .
5) Minimizing the absolute error raised to the nth power (usually 1 or 2 should be used).

Discussion

The age composition estimates from existing sampling from the natural spawners needs to be filtered to include samples of NOR fish but not both NOR and hatchery fish or just hatchery fish. In most (all) cases where the data could be compared, the age distribution of hatchery fish on the spawning grounds and NOR fish on the spawning grounds are different for Chinook salmon in Puget Sound and in the Lower Columbia River, with hatchery fish having more younger age fish and NOR salmon having more older fish. It is thought that carcass sampling, often used for spawning escapement estimation, may miss some of the younger fish as they are washed down stream before sampling or removed at a higher rate than older fish by predators. This would result in the different age composition of hatchery fish on the spawning grounds and at the hatchery rack, where presumably, fish of all sizes are sampled at the same rate. However, there is also a difference in the age composition of the NOR fish and the hatchery fish taken in the same carcass samples for some Puget Sound populations.