2011 SALMON METHODOLOGY REVIEW

Each year, the Scientific and Statistical Committee (SSC) and Salmon Technical Team (STT) complete a methodology review to help assure new or significantly modified methodologies employed to estimate impacts of the Council's salmon management use the best available science. The Methodology Review is also used as a forum to review updated stock conservation objective proposals. This review is preparatory to the Council's adoption, at the November meeting, of all anticipated methodology and conservation objective changes to be implemented in the coming season, or in certain limited cases, of providing directions for handling any unresolved methodology problems prior to the formulation of salmon management options in March. Because there is insufficient time to review new or modified methods at the March meeting, the Council may reject their use if they have not been approved the preceding November.

This year the SSC and STT are expected to report on:

- Abundance-based management framework for Lower Columbia River (LCR) tule fall Chinook (Agenda Item C.1.a, Attachment 1).
- Cohort reconstruction and development of a harvest model for Sacramento River Winter Chinook (Agenda Item C.1.a, Attachments 2 and 3).
- Examination of the potential bias in Coho and Chinook Fishery Regulation Assessment Model (FRAM) of fishery-related mortality introduced by mark-selective fisheries (Agenda Item C.1.a, Attachment 4).
- A multi-year review and evaluation of preseason and postseason mark-selective fisheries both north and south of Cape Falcon (Agenda Item C.1.a, Attachment 5).

The National Marine Fisheries Service (NMFS) identified the developing options for an abundance-based approach for 2012 LCR tule Chinook management in its 2010 Endangered Species Act (ESA) guidance letter to the Council. The Tule Chinook Workgroup Report (Attachment 1) addressed this task, which was intended to accelerate the tule recovery process and reduce uncertainties in key elements of the overall recovery strategy. Therefore, in addition to reviewing the models and technical analyses used to develop and analyze the alternatives, the Council should provide guidance and recommendations to NMFS for a preferred alternative for consideration in the 2012 tule Chinook ESA consultation process.

Council Action:

1. **Approve new and modified methodologies as appropriate for implementation in the 2012 salmon season.**
2. **Provide recommendations to NMFS on an abundance-based management approach for Lower Columbia River tule Chinook.**
3. **Provide guidance, as needed, for any unresolved methodology issues.**

Reference Materials:

1. Agenda Item C.1.a, Attachment 1: Exploration of Abundance-Based Management Approaches for Lower Columbia River Tule Chinook.
2. Agenda Item C.1.a, Attachment 2: Sacramento River Winter Chinook Cohort Reconstruction: Analysis of Ocean Fishery Impacts.
3. Agenda Item C.1.a, Attachment 3: The Winter-Run Harvest Model (WRHM).
4. Agenda Item C.1.a, Attachment 4: Application of Bias-corrected Methods for Estimating Mortality in Mark-Selective Fisheries to Coho FRAM.
5. Agenda Item C.1.a, Attachment 5: Causes and Effects of Bias in Anticipated Mark Rates in Mark-Selective Fisheries for Coho Salmon.
6. Agenda Item C.1.b, STT Report.
7. Agenda Item C.1.b, MEW Report.
8. Agenda Item C.1.b, Supplemental SSC Report.

Agenda Order:

a. Agenda Item Overview — Chuck Tracy
b. Reports and Comments of Advisory Bodies and Management Entities
c. Public Comment
d. **Council Action**: Adopt Final Methodology Changes for 2012 and Provide Recommendations to NMFS on Abundance-Based Methodology for Tule Chinook

PFMC
10/14/11

Agenda Item C.1.a
Attachment 1
November 2011

EXPLORATION OF ABUNDANCE-BASED MANAGEMENT APPROACHES FOR LOWER COLUMBIA RIVER TULE CHINOOK

PACIFIC FISHERY MANAGEMENT COUNCIL

AD HOC TULE CHINOOK WORK GROUP (TCW)

Ray Beamesderfer, Cramer Fish Sciences

Tom Cooney, NMFS NWFSC

Peter Dygert, NMFS NWR

Stuart Ellis, CRITFC

Matt Falcy, ODFW

Larrie LaVoy, NMFS NWR

Cindy LeFleur, WDFW

Hap Leon, Makah Tribe

Guy Norman, WDFW

John North, ODFW

Tom Stahl, ODFW

Chuck Tracy, PFMC Staff

OCTOBER 2011

Executive Summary

This report explores the potential utility of abundance-based management alternatives for Lower Columbia River tule Fall Chinook. This stock is currently managed with fixed annual impact rate limits intended to avoid jeopardy of long-term persistence of the natural stock component. Fishery limits have been substantially reduced from historical levels and the need for further reductions has been contemplated in Endangered Species Act (ESA) consultations by the National Marine Fisheries Service (NMFS). These limits are a significant constraint in fisheries administered by the Pacific Fishery Management Council (Council) and by the states of Washington and Oregon in the Columbia River. Abundance-based management is a variable rate alternative to the current fixed-rate strategy that potentially reduces conservation risks in years of low returns and increases fishery flexibility in years of high returns.

This report summarizes investigations of the feasibility and effectiveness of abundance-based management of lower Columbia River tule Fall Chinook by the Ad Hoc tule Chinook Work Group (TCW) convened by the Council at their June 2010 meeting. Four fundamental questions were evaluated:

1. What is abundance-based management and where has it been used?
2. Can Columbia River tule Fall Chinook abundance be predicted with sufficient accuracy and precision to feasibly implement an abundance-based strategy?
3. What are the effects of different fishing rates for Columbia River tule Fall Chinook on Council fisheries and in-river?
4. Can alternatives be implemented with negligible effects on escapement and viability of natural tule Chinook populations?

Abundance-Based Management Alternatives

Abundance-based fishery management strategies are currently employed in a variety of salmon fisheries throughout the Pacific Northwest. These strategies employ a variety of estimators or indicators related to natural fish abundance including abundance forecasts, brood year spawner numbers, marine survival, and ocean conditions related to marine survival. Indicators might be based on wild or hatchery fish at an aggregate or indicator population level. Fishery management strategies also involve different combinations of exploitation rates and thresholds at which different rates might be applied. Different rates and thresholds might be selected depending on the desired balance of conservation risks and fishery objectives.

Most management approaches are based on a preseason abundance forecast where allowable exploitation rates are either a stepped function based on abundance status bins or a continuous function designed to target a specific escapement value (spawners, dam count, etc.). These are essentially a one-dimensional matrix; examples include Puget Sound coho and Sacramento River fall Chinook, respectively. Oregon Coast Natural coho exploitation rates are based on a two-dimensional matrix using parental spawner abundance and estimated marine survival. Parental spawner abundance is further stratified into sub-aggregate populations so harvest in any year is based on status of the weakest population.

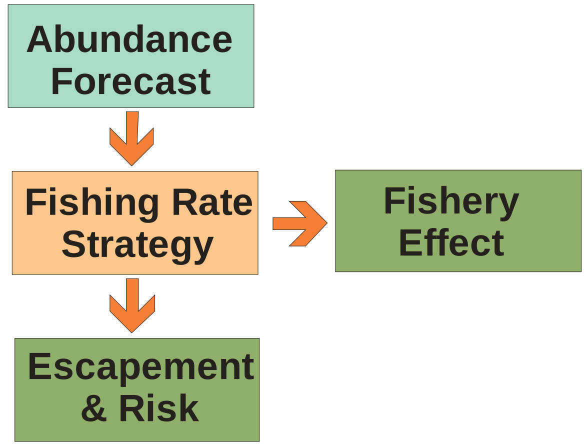

Figure 1. Conceptual depiction of an abundance-based management analysis.

Forecasting Abundance

Tule Fall Chinook from the lower Columbia River are forecast using sibling models from run reconstruction of the Lower River Hatchery (LRH) stock management unit. Although this is considered a hatchery stock unit, the accounting of the LRH run does include a small, but not easily quantifiable, proportion of naturally produced tule Fall Chinook from the lower Columbia River tributaries (LCN tules). LCN tule numbers cannot be accurately predicted at this time in aggregate or by population due to a lack of reliable age composition data for many natural populations.

Forecasts of the aggregate run of LRH Fall Chinook have been relatively accurate but imprecise. Error averaged just -2 percent for predicted number of total adults over the period from 1980 through 2009 but annual predictions have ranged from -66 percent to 85 percent of the actual return with a standard deviation of 37 percent. Errors were highly autocorrelated among years.

While LCN tule numbers cannot be predicted directly, aggregate forecasts based primarily on hatchery returns would appear to be a suitable proxy due to common effects of marine conditions to which both hatchery and wild fish are subject. The aggregate LRH return was at least partially correlated with LCN numbers considered in aggregate and in many or most natural populations throughout the historical data set and in the recent 10-year period. Correlations were similar for a brood year survival rate index, and the total LRH return.

Evaluations of the effects of incorporating indicators of ocean environmental conditions into the forecast models suggested that it might be feasible to improve forecast accuracy of aggregate tule abundance. However, any proposed improvements in forecast techniques will need to be tested with pre- and post- season comparisons. In the interim, considerations of abundance-based fishing strategies for tule Fall Chinook will necessarily rely on current forecast methods.

Fishing Rates, Contributions & Effects

Tule Fall Chinook are harvested in ocean fisheries from Alaska to Oregon and in the Columbia River. They are a major contributor to ocean fisheries north of Cape Falcon, Oregon managed by the Council. Prior to 1990, cumulative total exploitation rates (ER) regularly reached or exceeded 0.65. Following ESA listing, fishery impact ceilings were established by NMFS at 0.49 in 2002-2006, 0.42 in 2007, 0.38 in 2009-2010, and 0.37 in 2011.

Fishery limits on tule Fall Chinook are one of several potential constraints on mixed stock fisheries managed by the Council and Columbia River Compact. In some years, these limits can significantly constrain access to harvest of other fish stocks in these fisheries.

Analyses of 2009-2011 fisheries were conducted using the Fishery Regulation Assessment Model (FRAM) to identify example changes in fishery-specific ERs that would have resulted from a low status estimate for LCN tule Chinook under implementation of an abundance-based management system. Exploitation rates in Council fisheries can approach 0.15 depending on overall Columbia River stock abundances and extent of fishing South of Cape Falcon (primarily Oregon troll) that can add 0.01-0.03. Exploitation rates in the mainstem Columbia net and sport fisheries have averaged about 8 percent. Approximately two thirds of the ER in Council fisheries is in the non-Indian troll and sport fisheries (treaty Indian troll averages about 0.05 in these runs). In these examples, impacts on LCN tules in the southern U.S. would have to be reduced by nearly 50 percent in order to remain under an ER ceiling of 0.28 and by 23 percent to remain under a ceiling of 0.33, assuming current average conditions in northern fisheries.

Natural Population Risk Analysis

Abundance-based fishing strategies were evaluated for their effects on fisheries and on escapement and risk for LCN tule fall Chinook. Wild population risks were estimated with stochastic stock-recruitment modeling in a Population Viability Analysis framework like that employed in salmon ESA status assessments and recovery plans. Similar modeling approaches have previously been utilized by the Council in conservation risk analyses for other stocks including Klamath River fall Chinook.

Based on a review of abundance-based approaches for other fisheries and an assessment of information available for lower Columbia River tule Fall Chinook, the TCW initially identified a series of alternative strategies for further evaluation. Alternatives included a variety of fixed exploitation rate strategies ranging from 0 to 0.53 that were used for comparison with variable rate alternatives. Variable rate strategies used a one-dimensional matrix based on abundance of LRH tules; initial alternatives were three-tiered and used bins representing approximately 20 percent of the lowest preseason forecasts (< 40,000), 20 percent of the highest abundance forecasts (> 100,000) and the remaining 60 percent in the middle bin. A variety of ERs were assigned to these bins and assessed for changes to population risk and overall LRH harvest availability. Additional alternatives with different numbers of tiers and abundance frequencies were developed to achieve more specific objectives of fishery stability and population risk reduction.

Changes in risks and LRH harvest levels were compared relative to a fixed 0.37 ER limit, which represents the 2011 ESA consultation standard. Fishing rate scenarios were categorized based on whether risk and harvest levels were substantively greater or lower than corresponding values at the 0.37 exploitation rate. Substantive differences were based on changes to risk and harvest benefits approximating a one percentage point change in the fixed ER limit. Corresponding values were ±3.5 percent change in 100-year risk, ±0.25 percent change in 20-year risk, and ±3.0 percent change in average 100-year harvest. These numbers were used to classify fishing rate scenarios into one of four categories:

- The Win/Win group involved both a substantive reduction in risk to the natural population and a substantive improvement in fishing opportunity for tule Fall Chinook. This group would represent the ideal abundance-based strategy.
- The Risk Reduction group involves a substantive decrease in wild population risk with little or no fishery benefit.
- The Fishery Opportunity group involves a substantive increase in harvest opportunity.
- The Equivalent group provides similar wild population risk and tule Fall Chinook harvest level as the fixed 37 percent exploitation rate strategy.

No fishery scenarios were contemplated that increase natural population risks and reduce fishery opportunities.

Results indicate that a variety of variable exploitation rate strategies based on forecasts of aggregate tule Fall Chinook abundance can increase fishery management flexibility while also effectively reducing wild population risks. However, increased flexibility associated with higher exploitation rates in years of higher abundance must be compensated by reduced exploitation rates in years of lower abundance. This tradeoff would potentially increase variability in harvest of mixed stocks.

Model sensitivity analyses indicate that this conclusion is relatively robust to uncertainties in model inputs and functions related to the lack of specific data on wild population status and dynamics. Some examples of Win/Win scenarios and the current fixed 37 percent ER limit are presented below:

Table ES-1 Effects of variable rate fishing strategies based on abundance tiers. (Scenarios are sorted by the change in 100 year risk.)

ER Scenario	Abundance Tier Frequency	Population risk 100 yr	Population risk 20 yr	LRH Harvest	Change in risk 100 yr	Change in risk 20 yr	Δ Harvest 100 yr
F37	22/55/23	0.210	0.016	30,130	0%	0.0%	0%
V30/35/38/41	11/11/46/32	0.173		31,910	-3.7%		5.9%
V28/38/50	22/55/23	0.158	0.007	34,470	-5%	-0.9%	14%
V25/40/45	22/55/23	0.136	0.006	33,250	-7%	-1.0%	10%
V28/37/50	22/55/23	0.142	0.007	34,080	-7%	-0.9%	13%
V40-15	22/55/23	0.120	0.006	31,480	-9%	-1.0%	4%
V25/37/50	22/55/23	0.111	0.005	33,860	-10%	-1.1%	12%
V35±10	22/55/23	0.072	0.003	31,320	-14%	-1.3%	4%

Contents

INTRODUCTION

This report explores the potential utility of an abundance-based management alternative for Lower Columbia River (LCR) tule Fall Chinook in fisheries managed by the Pacific Fishery Management Council (Council). The Lower Columbia Chinook evolutionarily significant unit (ESU) was listed as threatened under the U.S. Endangered Species Act (ESA) in 1999. The fall run tule stock component of the ESU is currently managed with fixed annual impact rate limits intended to avoid jeopardy of long-term persistence of the wild stock. Fishery limits have been substantially reduced from historical levels and the need for further reductions has been contemplated in ESA consultations by the National Marine Fisheries Service (NMFS). Tule Chinook are harvested in fisheries from Oregon to Alaska, and while no single fishery harvests a large number of this stock, the combined impact of all fisheries can be significant. Because much of the tule fishery impact currently occurs in Canada and Alaska, outside the Council's direct management authority, reduced impact limits have seriously constrained Oregon and Washington ocean and Columbia River fisheries.

Abundance-based management is a variable rate alternative to the current fixed-rate strategy. An abundance-based strategy might offer two potential benefits. It reduces conservation risks in years of low returns and increases fishery flexibility in years of high returns. When abundance is low, decreased impact rates reduce the risk of low spawning escapements that can damage the long-term viability of weak wild populations. When annual abundance is high, increased impact rates increase access to harvestable surpluses of stronger stocks and hatchery components of the run. Effective use of this strategy could potentially increase average harvest of both tule Chinook and other salmon with no additional long-term risk to wild tule Chinook. The benefit of increased harvest on large runs is effectively bought with the cost of reduced harvest in the small run years.

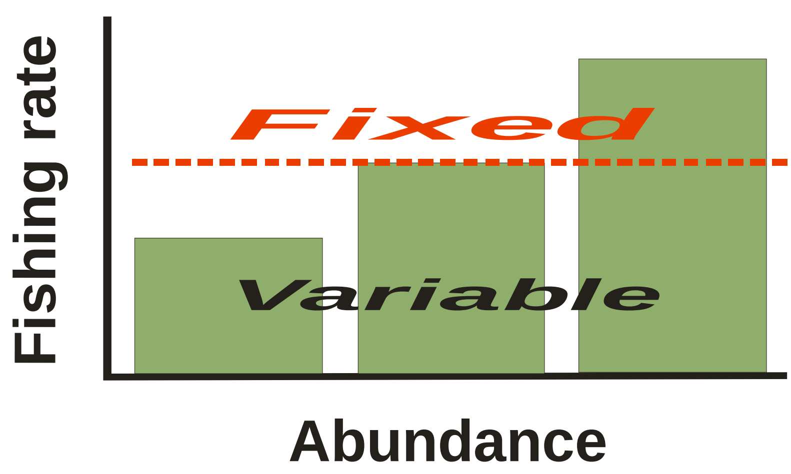

Figure 2. Illustration of a variable exploitation rate fishing strategy based on abundance.

The potential application of abundance-based management of Lower Columbia Natural (LCN) tule Fall Chinook is also complicated by selection and interpretation of appropriate indices by which to measure fishery impacts and effects on wild population risks. Impacts on wild fish were historically indexed based on Cowlitz Hatchery tags to represent the Coweeman wild population.[1] However, Coweeman tules are one of the stronger extant populations[2] and a more effective conservation strategy will involve protection of both weak and strong tule populations.

ESA Recovery Plans adopted in 2010 for lower Columbia River salmon include specific measures calling for the evaluation of abundance-based management for tule Chinook. NMFS also identified the need to develop options for incorporating abundance-driven management principles into LCN tule Chinook management in a 2010 letter to the Council summarizing consultation standards and guidance regarding the potential effects of the 2010 season on listed salmonid species. This guidance letter described a set of tasks designed to accelerate the recovery process by completing actions with immediate benefit to tule populations. NMFS advice indicated that total exploitation rate limits on Lower Columbia River tule Chinook will be contingent on satisfactory progress in completing tasks.

At their June 2010 meeting, the Council convened an Ad Hoc tule Chinook Work Group (TCW) to explore abundance-based approaches for LCN tule Chinook. The work group included members from NMFS, Washington and Oregon Departments of Fish and Wildlife (WDFW and ODFW), Columbia River treaty Tribes, and the Makah tribe, and is facilitated by Council staff. The TCW met on September 30, 2010 to identify a process, tasks, schedule and assignments. A draft work plan and schedule was developed from initial TCW discussions. The draft work plan also described a schedule for integrating this review with the Council's annual salmon methodology review process that produces recommendations in November of each year. Initial assessments by the TCW showed that an abundance-based approach was potentially practical and effective. In April the TCW initiated steps to complete a draft technical analysis for further consideration. This report is the product of that analysis.

This analysis addresses four fundamental questions regarding the feasibility and effectiveness of an abundance-based fishery strategy for LCR tule Fall Chinook:

1. What is abundance-based management and where has it been used?
2. Can Columbia River tule Fall Chinook abundance be predicted with sufficient accuracy and precision to feasibly implement an abundance-based strategy?
3. What are the effects of different fishing rates for Columbia River tule Fall Chinook on Council fisheries?
4. Can alternatives be implemented with negligible effects on escapement and viability of wild tule Chinook populations?

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[1] *Fishery impacts are currently indexed using coded-wire tags (CWT's) from all lower Columbia River hatchery tule Chinook. This provides better representation in ocean fisheries than the previous use of Cowlitz Hatchery tags alone.*

[2] *In the case of LCN tules, strong is a relative term. All populations have been found to be at significant risk of extinction. However, risks for the Coweeman population are more moderate than for most other populations.*

STOCK DESCRIPTION

LCR tule Fall Chinook are part of a lower Columbia River Evolutionarily Significant Unit (ESU) that was initially listed in 1999 as threatened under the U.S. ESA. The listing was reaffirmed in 2005. This ESU includes all naturally spawned populations of spring, fall (tule) and late fall (bright) Chinook salmon from the Columbia River and its tributaries from its mouth at the Pacific Ocean upstream to a transitional point between Washington and Oregon east of the Hood River and the White Salmon River. Celilo Falls, which historically presented a migration barrier to Chinook salmon under certain flow conditions, is the eastern boundary of the ESU (Myers et al. 2006).

Fall Chinook historically spawned in large tributaries of the lower Columbia River from the mouth of the Columbia River to the Klickitat River. Columbia tule Fall Chinook typically enter freshwater from August to September and spawn from late September to November, with peak spawning activity in late September to mid-October. A total of 21 historical populations of tule Fall Chinook were identified by NMFS (Myers et al. 2006). All were estimated to be at high or very high risk of extinction at the time of listing (Figure 3) in recovery plans prepared by Washington and Oregon, and adopted by NMFS. Risks were determined by a combination of qualitative assessments relative to viable salmonid population characteristics (abundance, productivity, spatial structure, and diversity) and quantitative assessments using population viability analysis involving stochastic population models.

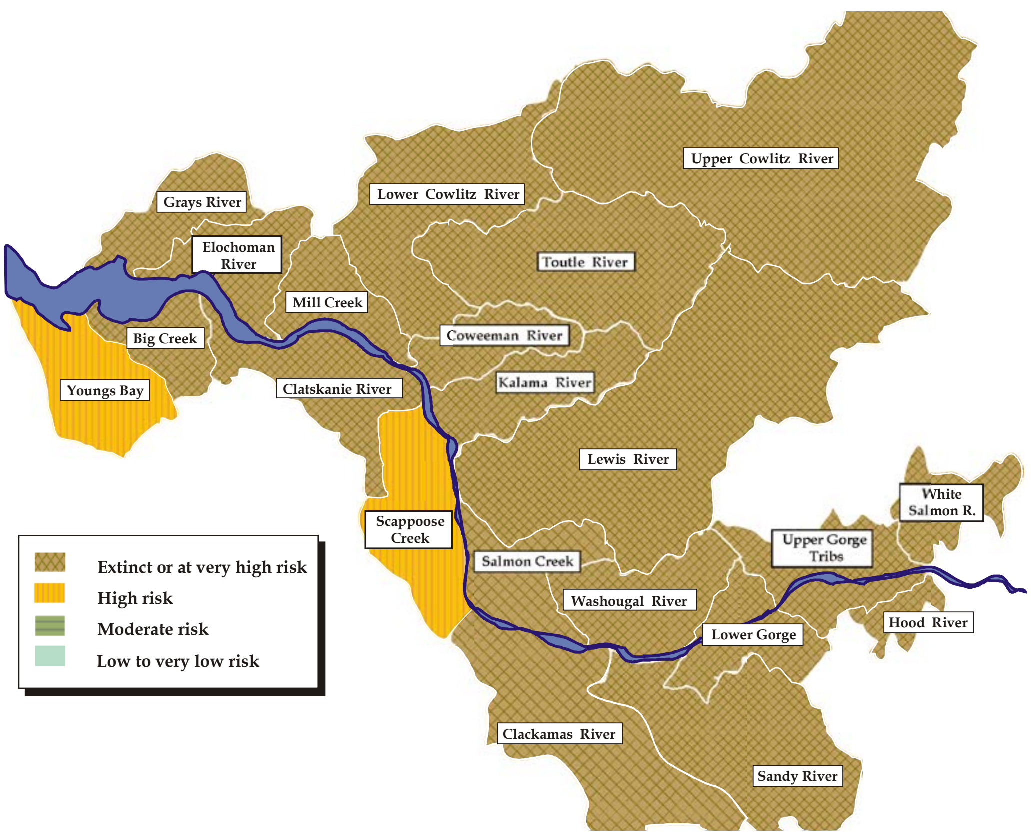

Figure 3. Current status of historical demographically-independent lower Columbia fall (tule) Chinook populations (LCFRB 2010). Risk thresholds are >60% (very high), 26-60% (high), 6-25% (medium), 1-5% (low) and <1% (very low) within 100 years.

Columbia River tule Fall Chinook have been distinguished into two stocks for fishery management purposes. The Lower River “Hatchery” stock (LRH) includes all fish returning to hatchery and natural production areas in the lower 145 miles of the Columbia River downstream from Bonneville Dam. The naturally-produced component (LCN) of the LRH stock includes both wild fish and naturally-produced offspring from hatchery-origin fish spawning in the wild. The Bonneville Pool Hatchery stock (BPH) includes tules returning to hatchery and natural production areas upstream from Bonneville Dam.

Total numbers of wild tule Chinook returning to lower Columbia River streams have declined in the last century from over 100,000 (LCFRB 2010, ODFW 2010) to less than 10,000 per year (Table 1). As wild numbers declined, hatcheries were established to produce fish for fisheries in an effort to mitigate the decline. Hatchery production of LRH tules averaged approximately 60 million juveniles per year from 1978 through 1991 but has since been reduced to an annual average of about 22 million (Figure 5). An additional 15 million BPH tules are produced in Spring Creek Hatchery upstream from Bonneville Dam. Over 90 percent of the current tule run to the lower Columbia River is comprised of hatchery-origin fish. Hatchery-origin fish also appear to contribute 40 percent to 80 percent of the total number of naturally-spawning fish in Washington lower Columbia streams since 1977. Hatchery fractions vary considerably from stream to stream (Table 1).

Returns to the Columbia River of LRH Chinook has averaged 110,000 and varied between 30,000 and 348,000 between 1977 and 2010. Average annual survival (estimated from hatchery releases) declined from about 0.7 percent per year prior to 1977 to just 0.3 percent per year since (Table 1, Figure 5). This change coincided with the shift in ocean environmental regime in the mid 1970s. Average survival improved somewhat since the mid-1990s, in part due to the termination of several of the less-successful hatchery programs in the basin following reductions in federal Mitchell Act funding.

Variable marine survival has resulted in variable and unpredictable tule run size over the last 30 years. Very low numbers corresponded to a severe *El Niño* event in the early 1980s and mid to late 1990s. High returns were produced by apparently favorable survival conditions in years immediately following *El Niños*. From 1977-2010, approximately runs of 40,000 fish or less occurred 6 percent of the time and runs of 140,000 or greater occurred 24 percent of the (Figure 4).

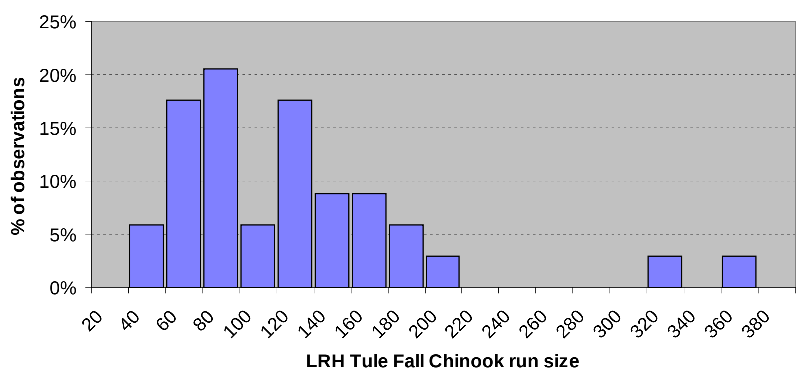

Figure 4. Frequency distribution of LRH tule Fall Chinook run size, 1977-2010.

Table 1. LRH Tule Chinook data including hatchery releases, total run size to the Columbia River and population-specific estimates, naturally-produced fractions , and wild numbers (WDFW unpublished data).

Run year	hatchery releases (millions)	LRH run (1,000's)	LRH wild fraction	LRH run yr survival	Grays			Mill/Abernathy/Germany			Elochoman/Skamokawa			Coweeman			Cowlitz			Kalama			Lewis			Washougal			Natural escapement Total		Wild
					# total	% wild	# wild	# total	% wild	# wild	# total	% wild	# wild	# total	% wild	# wild	# total	% wild	# wild	# total	% wild	# wild	# total	% wild	# wild	# total	% wild	# wild	number	% wild	number
1977	39.3	171.5	4.5%	0.74%	1,009	0.46	464				568	0.42	239	337	1.00	337	5837	0.26	1518	6,549	0.50	3275	1,086	1.00	1,086	1,652	0.46	760	17,038	0.45	7678
1978	66.0	174.9	3.6%	0.83%	1,806	0.46	831				1,846	0.42	775	243	1.00	243	3192	0.26	830	3,711	0.50	1856	1,448	1.00	1,448	593	0.46	273	12,839	0.49	6255
1979	72.2	126.1	5.6%	0.51%	344	0.46	158				1,478	0.42	621	344	1.00	344	8253	0.26	2146	2,731	0.50	1366	1,304	1.00	1,304	2,388	0.46	1,098	16,842	0.42	7037
1980	83.4	111.0	5.8%	0.34%	125	0.46	58	516	0.49	253	64	0.42	27	180	1.00	180	1793	0.26	466	5,850	0.50	2925	899	1.00	899	3,437	0.46	1,581	12,864	0.50	6388
1981	50.4	103.0	4.2%	0.21%	208	0.46	96	1,367	0.48	656	138	0.42	58	116	1.00	116	3213	0.26	835	1,917	0.50	959	799	1.00	799	1,841	0.46	847	9,599	0.45	4366
1982	51.4	149.0	3.6%	0.22%	272	0.46	125	2,750	0.50	1,375	340	0.42	143	149	1.00	149	2100	0.26	546	4,595	0.50	2298	646	1.00	646	330	0.46	152	11,182	0.49	5433
1983	56.9	91.1	7.3%	0.12%	825	0.46	380	3,725	0.51	1,900	1,016	0.42	427	122	1.00	122	2463	0.26	640	2,722	0.50	1361	598	1.00	598	2,677	0.46	1,231	14,148	0.47	6659
1984	34.4	104.6	3.9%	0.17%	252	0.46	116	614	0.52	319	294	0.42	123	683	1.00	683	1737	0.26	452	3,043	0.50	1522	340	1.00	340	1,217	0.46	560	8,180	0.50	4115
1985	52.7	128.0	4.1%	0.24%	532	0.46	245	1,815	0.53	962	464	0.42	195	491	1.00	491	3200	0.26	832	1,259	0.50	630	1,029	1.00	1,029	1,983	0.46	912	10,773	0.49	5295
1986	52.2	184.9	2.6%	0.37%	370	0.46	170	980	0.49	480	918	0.42	386	396	1.00	396	2474	0.26	643	2,601	0.50	1301	696	1.00	696	1,589	0.46	731	10,024	0.48	4803
1987	64.2	348.2	3.8%	0.89%	555	0.46	255	6,168	0.59	3,639	2,458	0.42	1032	386	1.00	386	4260	0.26	1108	9,651	0.50	4826	256	1.00	256	3,625	0.46	1,668	27,359	0.48	13169
1988	61.5	314.2	6.6%	0.87%	680	0.46	313	3,133	0.69	2,162	1,370	0.42	575	1,890	1.00	1,890	5327	0.26	1385	24,549	0.50	12275	744	1.00	744	3,328	0.46	1,531	41,021	0.51	20874
1989	51.1	133.4	14.5%	0.30%	516	0.46	237	2,792	0.69	1,926	122	0.42	51	2,549	1.00	2,549	4917	0.26	1278	20,495	0.50	10248	972	1.00	972	4,578	0.46	2,106	36,941	0.52	19368
1990	53.6	66.6	6.8%	0.12%	166	0.46	76	650	0.63	410	174	0.42	73	812	1.00	812	1833	0.26	477	2,157	0.50	1079	563	1.00	563	2,205	0.46	1,014	8,560	0.53	4503
1991	52.2	71.9	10.2%	0.12%	127	0.47	60	2,017	0.85	1,714	196	0.09	18	340	1.00	340	935	0.26	243	5,152	0.54	2782	470	1.00	470	3,673	0.47	1,726	12,910	0.57	7353
1992	42.7	68.9	8.9%	0.12%	109	0.76	83	839	0.47	394	190	1.00	190	1,247	1.00	1,247	1022	0.26	266	3,683	0.48	1768	335	1.00	335	2,399	0.76	1,823	9,824	0.62	6106
1993	37.3	54.8	10.6%	0.10%	27	0.52	14	885	0.71	628	288	0.78	225	890	1.00	890	1330	0.06	80	1,961	0.89	1745	164	1.00	164	3,924	0.52	2,040	9,469	0.61	5787
1994	38.8	56.3	15.9%	0.11%	30	0.7	21	3,854	0.40	1,542	706	0.98	692	1,695	1.00	1,695	1225	0.19	233	2,014	0.71	1430	610	1.00	610	3,888	0.70	2,722	14,022	0.64	8944
1995	47.9	49.9	12.1%	0.11%	9	0.39	4	1,395	0.51	711	156	0.50	78	1,368	1.00	1,368	1370	0.13	178	3,012	0.69	2074	409	1.00	409	3,063	0.39	1,195	10,782	0.56	6017
1996	33.7	79.5	11.9%	0.20%	280	0.17	48	593	0.54	320	533	0.66	352	2,305	1.00	2,305	1325	0.58	769	10,630	0.44	4728	403	1.00	403	2,921	0.17	497	18,990	0.50	9421
1997	25.9	58.8	8.1%	0.15%	15	0.12	2	603	0.23	139	1,875	0.11	206	689	1.00	689	2007	0.72	1445	3,539	0.40	1402	305	1.00	305	4,669	0.12	560	13,702	0.35	4748
1998	16.8	47.2	11.1%	0.11%	96	0.24	23	368	0.60	221	228	0.25	57	491	1.00	491	1665	0.37	616	4,294	0.69	2973	127	1.00	127	2,971	0.24	713	10,240	0.51	5221
1999	22.2	40.7	9.1%	0.10%	195	0.68	133	575	0.69	397	718	0.25	180	299	1.00	299	969	0.16	155	2,577	0.03	81	331	1.00	331	3,129	0.68	2,128	8,793	0.42	3703
2000	20.9	30.6	10.7%	0.11%	169	0.7	118	416	0.58	241	196	0.62	122	290	1.00	290	2165	0.10	217	1,284	0.21	266	515	1.00	515	2,155	0.70	1,509	7,190	0.46	3277
2001	24.1	103.6	8.4%	0.53%	261	0.43	112	4,024	0.39	1,569	2,354	0.82	1930	802	0.73	585	3647	0.44	1605	3,553	0.18	654	750	0.70	525	3,901	0.43	1,677	19,292	0.45	8659
2002	24.5	159.4	7.6%	0.84%	107	0.47	50	3,343	0.05	167	7,581	0.00	0	877	0.97	851	9671	0.76	7350	18,627	0.01	106	1,032	0.77	795	6,050	0.47	2,844	47,288	0.26	12162
2003	23.3	156.5	10.2%	0.74%	398	0.39	155	3,810	0.56	2,134	6,820	0.65	4433	1,106	0.89	984	7001	0.88	6161	24,684	0.00	74	738	0.98	723	3,444	0.39	1,343	48,001	0.33	16007
2004	23.4	111.4	7.8%	0.51%	766	0.25	192	6,804	0.02	136	4,796	0.01	48	1,503	0.91	1,368	4621	0.70	3235	6,434	0.11	686	1,388	0.29	403	10,597	0.25	2,649	36,909	0.24	8716
2005	17.7	79.5	4.3%	0.34%	147	0.41	60	2,083	0.13	271	2,204	0.05	110	853	0.60	512	2968	0.17	505	9,053	0.03	264	607	1.00	607	2,678	0.41	1,098	20,593	0.17	3427
2006	22.4	61.3	6.6%	0.25%	302	1.00	302	636	0.62	394	317	1.00	317	561	1.00	561	2,051	0.47	964	10,386	0.01	140	1,300	0.82	1,068	1,936	0.14	279	17,489	0.23	4025
2007	18.3	36.7	8.9%	0.16%	63	1.00	63	335	0.48	161	165	1.00	165	234	1.00	234	1,401	0.53	743	3,296	0.06	208	492	0.73	359	1,528	0.87	1,325	7,514	0.43	3257
2008	25	66.9	7.2%	0.34%	40	0.68	27	750	0.49	368	841	0.10	84	404	0.52	210	1,259	0.90	1136	3,734	0.04	149	567	0.87	495	2,491	0.93	2,324	10,086	0.48	4794
2009		85.6	5.4%	0.44%	312	0.43	133	604	0.93	563	2,246	0.18	412	780	0.63	494	2,602	0.45	1165	7,548	0.10	736	299	1.00	299	2,741	0.30	814	17,132	0.27	4616
2010		108.5	7.9%	0.56%	19	1.00	19	3,030	0.57	1,712	913	0.16	150	421	0.44	186	2,489	0.52	1305	5,576	0.29	1625	2,198	0.86	1,883	5,212	0.33	1,704	19,858	0.43	8584
g 1977-20	40.8	109.8	7.6%	0.35%	327	0.51	151	1,983	0.51	899	1,311	0.44	426	760	0.93	715	3,009	0.36	1,221	6,555	0.38	2,053	718	0.94	653	3,083	0.46	1,336	17,572	0.45	7,375
g 2000-20	22.2	90.9	7.7%	0.44%	235	0.61	112	2,349	0.44	701	2,585	0.42	706	712	0.79	570	3,625	0.54	2,217	8,561	0.09	446	899	0.82	697	3,885	0.47	1,597	22,850	0.34	7,048

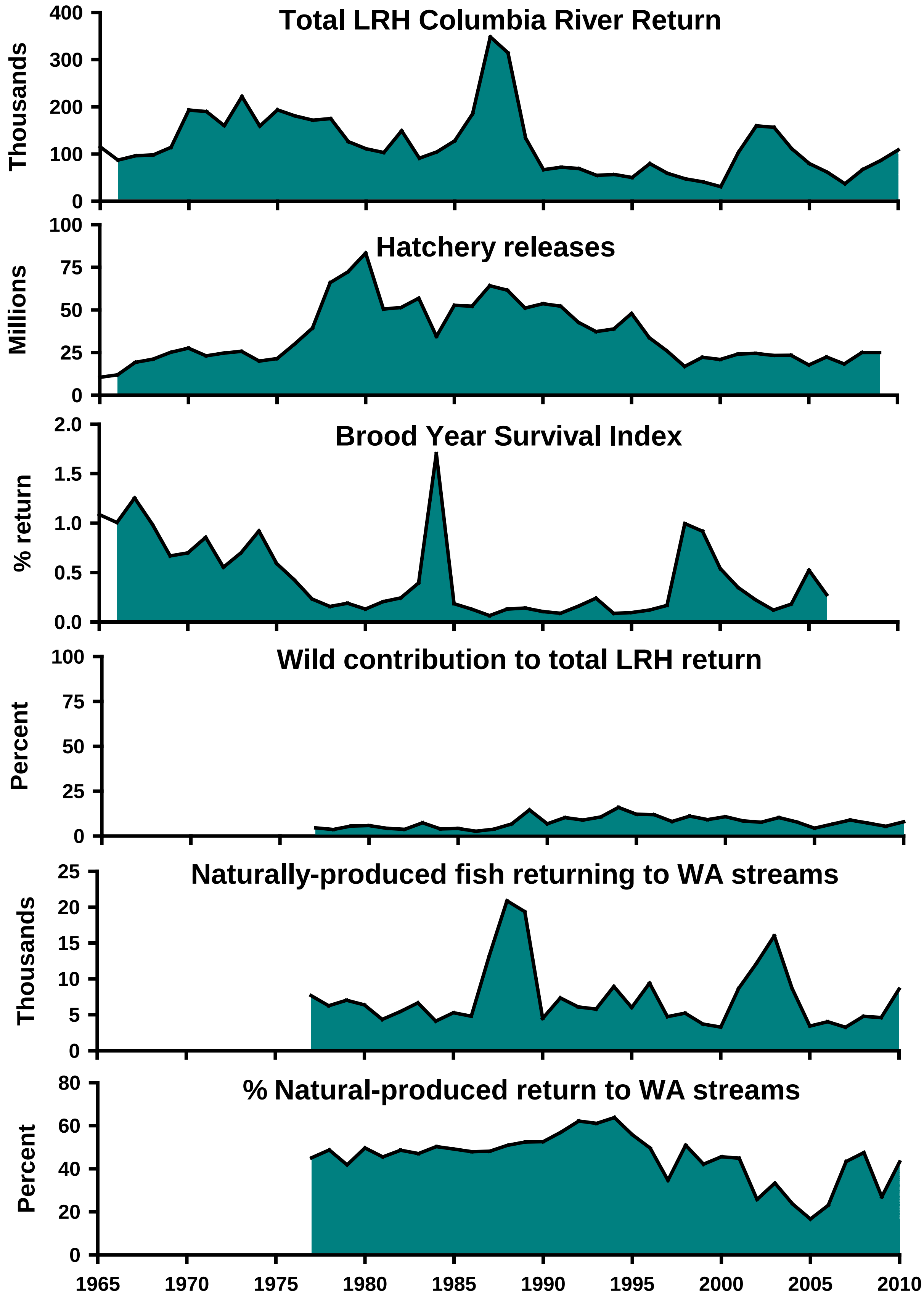

Figure 5. **Historical trends in lower Columbia tule Fall Chinook numbers.**

ABUNDANCE-BASED MANAGEMENT EXAMPLES

Abundance-based fishery management strategies are currently employed in a variety of salmon fisheries throughout the Pacific Northwest. These strategies employ a variety of estimators or indicators related to natural fish abundance including abundance forecasts, brood year spawner numbers, marine survival, and ocean conditions related to marine survival. Indicators might be based on wild or hatchery fish at an aggregate or indicator population level. These examples can help identify a range of indicators that might be considered for application to LCN tule Fall Chinook.

Fishery management strategies also involve different combinations of exploitation rates and thresholds at which different rates might be applied. Related considerations include both conservation and fishery objectives. Many different combinations of rates and thresholds might be contemplated. Single year alternatives might be based on annual run size expectations. Multi-year alternatives might also include extra conditions on adoption of higher or lower rates (for instance, limits if coming off successive low run years). Different rates and thresholds might be selected depending on the desired balance of conservation risks and fishery objectives.

This section reviews examples of abundance-based strategies employed in other fisheries throughout the region.

Pacific Salmon Commission

Aggregate Abundance-based Management (AABM): For Chinook fishery management under the PST, an abundance-based approach is used in three regional fisheries: southeast Alaska troll, net and sport; northern British Columbia troll and sport, and WCVI troll and sport. The abundance measurement used to set allowable landed catches in these fishery groups is an aggregate stock abundance index of stocks that contribute to each of these fisheries. The abundance index (AI) is calculated from the Pacific Salmon Commission (PSC) Chinook Model and is the ratio of the modeled catch in each fishery under 1979-82 base period exploitation rates and current year abundances divided by the catch under base period exploitation rates and base period abundances. There are several different AI tiers per fishery where the fishery harvest rate steps up to a higher level (Table 2). The AIs that contain these incremental harvest rate increases are associated with a total allowable landed catch per fishery in Table 1 of the PST. There are additional provisions in the treaty that reduce AI catch levels when selected stock and stock aggregates are below conservation objectives recognized by the PSC.

Table 2. Pacific Salmon Treaty AABM stepped harvest regime (from Appendix B to Annex IV, Chapter 3, updated January 27, 2009).

Relationships between AIs, Catches and HRIs[15]

Southeast Alaska All Gear	North BC Troll & QCI Sport	WCVI Troll & Outside Sport
Proportionality Constant (PC) = 12.38	Proportionality Constant (PC) = 11.83	Proportionality Constant (PC) = 13.10
Harvest Rate Index (HRI) = EXP(LN(Troll Catch / AI) - PC) Troll Catch = (Total Catch - 17,000) * 0.8 = EXP(PC + LN(HRI * AI))	Harvest Rate Index = EXP(LN(Troll Catch / AI) - PC) Troll Catch = Total Catch * 0.8 = EXP(PC + LN(HRI * AI))	Harvest Rate Index = EXP(LN(Troll Catch / AI) - PC) Troll Catch = Total Catch * 0.8 = EXP(PC + LN(HRI * AI))
Total Catch = 17,000 + Troll Catch / 0.8	Total Catch = Troll Catch / 0.8	Total Catch = Troll Catch / 0.80
Reduction in catch from 1999 Agreement: 15%	Reduction in catch from 1999 Agreement: 0%	Reduction in catch from 1999 Agreement: 30%
For AIs less than 1.005	For AIs less than 1.205	For AIs less than 0.5
Total Catch = 17,000 + 110,500 * AI	Total Catch = 130,000 * AI	Total Catch = 128,347 * AI
Troll Catch = (110,500 * AI) * 0.8	Troll Catch = (130,000 * AI) * 0.8	Troll Catch = (128,347 * AI) * 0.8
HRI = 0.371	HRI = 0.757	HRI = 0.21
For AIs between 1.005 and 1.2	For AIs between 1.205 and 1.5	For AIs between 0.5 and 1.0
Total Catch = -114,750 + 242,250 * AI	Total Catch = -20,000 + 146,667 * AI	Total Catch = 149,739 * AI
Troll Catch = (-131,750 + 242,250 * AI) * 0.8	Troll Catch = (-20,000 + 146,667 * AI) * 0.8	Troll Catch = (149,739 * AI) * 0.8
HRI increasing from 0.371 to 0.445	HRI increasing from 0.757 to 0.777	HRI = 0.245
For AIs between 1.205 and 1.5	For AIs greater than 1.5	For AIs greater than 1.0
Total Catch = 17,000 + 151,721 * AI	Total Catch = 145,892 * AI	Total Catch = 171,130 * AI
Troll Catch = (151,721 * AI) * 0.8	Troll Catch = (145,892 * AI) * 0.8	Troll Catch = (171,130 * AI) * 0.8
HRI = 0.51	HRI = 0.85	HRI = 0.28
For AIs greater than 1.5		
Total Catch = 17,000 + 164,364 * AI		
Troll Catch = (164,364 * AI) * 0.8		
HRI = 0.5525		

[15] If alternative harvest rate metrics are adopted in any of the AABM fisheries this will necessitate a recalculation of the proportionality constants in the affected fisheries and will in turn lead to an adjustment of the associated HRI values in this appendix. However, the formulas to estimate total catch in this appendix and the catches in Table 1 will remain unaffected.

Table 3. Catches specified for AABM fisheries at levels of the Chinook abundance index (January 27, 2009 update).

Values for catch at levels of abundance between those stated may be linearly interpolated between adjacent values.

Abundance index	SEAK	NBC	WCVI
0.25	44,600	32,500	32,100
0.30	50,200	39,000	38,500
0.35	55,700	45,500	44,900
0.40	61,200	52,000	51,300
0.45	66,700	58,500	57,800
0.495	71,700	64,400	63,500
0.50	72,300	65,000	74,900
0.55	77,800	71,500	82,400
0.60	83,300	78,000	89,800
0.65	88,800	84,500	97,300
0.70	94,400	91,000	104,800
0.75	99,900	97,500	112,300
0.80	105,400	104,000	119,800
0.85	110,900	110,500	127,300
0.90	116,500	117,000	134,800
0.95	122,000	123,500	142,300
1.00	127,500	130,000	149,700
1.005	128,700	130,700	172,000
1.05	139,600	136,500	179,700
1.10	151,700	143,000	188,200
1.15	163,800	149,500	196,800
1.20	176,000	156,000	205,400
1.205	199,800	156,700	206,200
1.25	206,700	163,300	213,900
1.30	214,200	170,700	222,500
1.35	221,800	178,000	231,000
1.40	229,400	185,300	239,600
1.45	237,000	192,700	248,100
1.50	244,600	200,000	256,700
1.505	264,400	219,600	257,600
1.55	271,800	226,100	265,300
1.60	280,000	233,400	273,800
1.65	288,200	240,700	282,400
1.70	296,400	248,000	290,900
1.75	304,600	255,300	299,500
1.80	312,900	262,600	308,000
1.85	321,100	269,900	316,600

Puget Sound Coho

Puget Sound coho stocks are managed under the PST using a stepped harvest rate control rule (Figure 6) (Southern Coho Management Plan Chapter 5, Annex IV, Article XV, PST 2009). Under this control rule, exploitation rate ceilings are determined on the basis of abundance, where abundance is divided into three zones defined by two breakpoints defined as:

$A = \frac{MSST}{1-F_{low}}$, breakpoint between critical and low abundance,

$B = \frac{S_{MSY}}{1-MFMT}$, breakpoint between low and normal abundance.

The exploitation rate ceiling has a maximum value of maximum fishing mortality threshold (MFMT; F_{MSY}) when N > B, is reduced to a low exploitation rate (F_{low}) when A < N < B, and further reduced to a critical exploitation rate ($F_{critical}$) to allow for *de minimis* impacts not to exceed 0.20 when N < A. For all Puget Sound coho stocks, the critical/low spawning escapement breakpoint and low exploitation rate are used to define minimum stock size threshold (MSST).

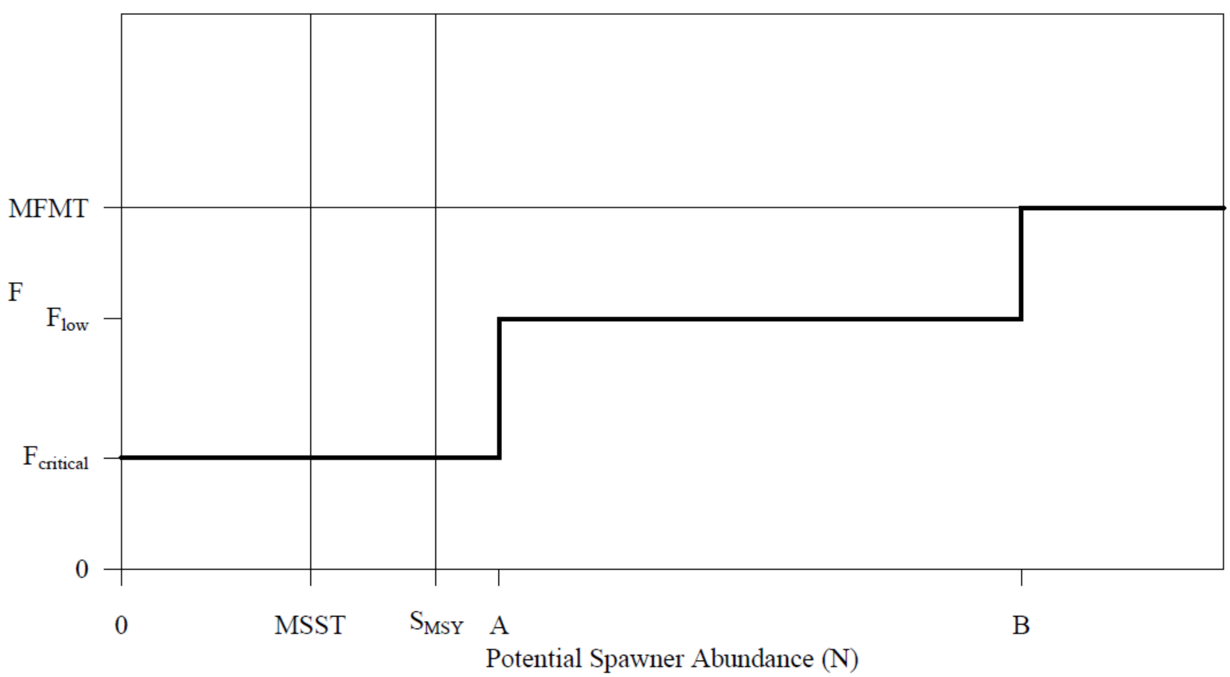

Figure 6. Control rule for Puget Sound coho. Abundance is pre-fishery ocean abundance in spawner equivalent units, and F is the exploitation rate.

Klamath & Sacramento Fall Chinook

Klamath River fall Chinook (KRFC) and Sacramento River fall Chinook (SRFC) have a control rule defined in terms of the reference points F_{ABC} ($0.95*F_{MSY}$ and $0.9*F_{MSY}$ for KRFC and SRFC, respectively), MSST, S_{MSY} (maximum sustainable yield spawning escapement), and two levels of *de minimis* exploitation rates, F = 0.10 and F = 0.25. The allowable exploitation rate, F, in a given year, depends on the pre-fishery ocean abundance in spawner equivalent units, N, as shown in Figure 7, with the abundance breakpoints defined as:

$$\mathrm{A} = \mathrm{MSST} \;/\; 2$$

$$\mathrm{B} = (\mathrm{MSST} + \mathrm{S}_{\mathrm{MSY}}) \;/\; 2$$

$$\mathrm{C} = \mathrm{S}_{\mathrm{MSY}} \;/\; (1 - 0.25)$$

$$\mathrm{D} = \mathrm{S}_{\mathrm{MSY}} \;/\; (1 - \mathrm{F}_{\mathrm{ABC}})\;.$$

For N between 0 and A, F increases linearly from 0 at N = 0, to 0.10 at N = A. For N between A and MSST, F is equal to 0.10. For N between MSST and B, F increases linearly from 0.10 at N = MSST, to 0.25 at N = B. For N between B and C, F is equal to 0.25. For N between C and D, F is the value that results in S_{MSY} spawners. For N greater than D, F is equal to F_{ABC}. The control rule may thus be summarized as follows:

$$\mathrm{F} = \begin{cases} 0.10 \times (\mathrm{N} \;/\; \mathrm{A}), \\ 0.10, \\ 0.10 + (0.15 \times \;((\mathrm{N} - \mathrm{MSST}) \big/ (\mathrm{B} - \mathrm{MSST}))) \,, \\ 0.25, \\ \left(\mathrm{N} - \mathrm{S}_{\mathrm{MSY}}\right) / \;\mathrm{N}\,, \\ \mathrm{F}_{\mathrm{ABC}}, \end{cases}$$

$$\begin{array}{ll} \text{if} & 0 \le \mathrm{N} \le \mathrm{A}; \\ \text{if} & \mathrm{A} < \mathrm{N} \le \mathrm{MSST}; \\ \text{if} & \mathrm{MSST} < \mathrm{N} \le \mathrm{B}; \\ \text{if} & \mathrm{B} < \mathrm{N} \le \mathrm{C}; \\ \text{if} & \mathrm{C} < \mathrm{N} \le \mathrm{D}; \\ \text{if} & \mathrm{D} < \mathrm{N}. \end{array}$$

The control rule describes maximum allowable exploitation rates at any given level of abundance. The Council may recommend lower exploitation rates as needed to address uncertainties or other year-specific circumstances.

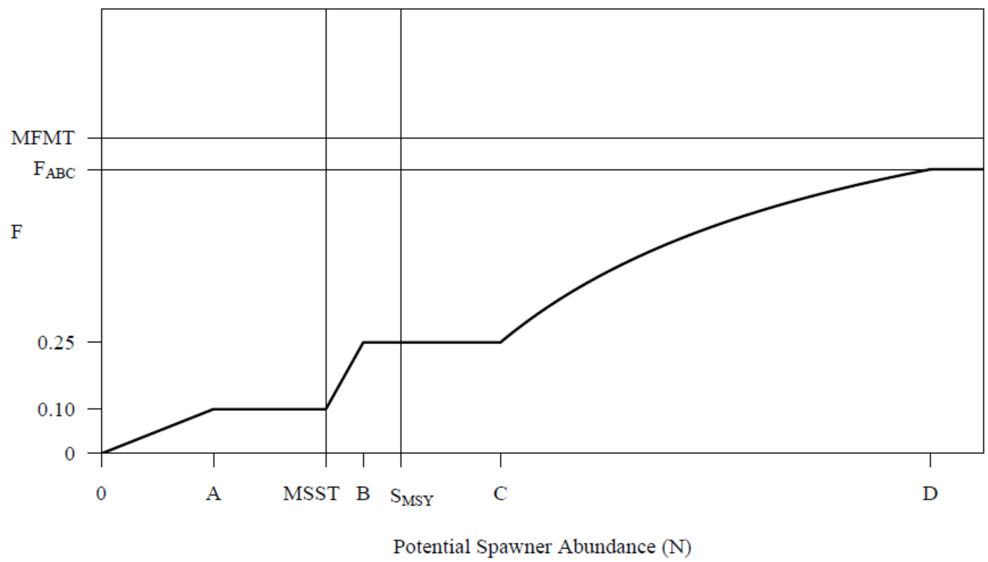

Figure 7. Control rule for SRFC and KRFC. Abundance is pre-fishery ocean abundance in spawner equivalent units, and F is the exploitation rate.

Oregon Coast Natural and Columbia River coho

An abundance-based exploitation strategy was adopted by the Council in 1997 for management of fisheries for Oregon Coast natural (OCN) and Columbia River natural (LCN) coho. The maximum allowable exploitation rates for OCN vary in response to changes in observed brood year-specific parental spawner abundance and marine survival conditions.

Table 4. Harvest management matrix identifying allowable fishery impacts and ranges of resulting recruitment based on parental spawner abundance and marine survival (OCN work group revisions to original Council matrix).

	Marine Survival Index (based on return of jacks per hatchery smolt)			
Parental Spawner Status*	**Extremely Low (<0.0008)**	**Low (0.0008-0.0014)**	**Medium (>0.0014-0.0040)**	**High (>0.0040)**
High (>75% of full seeding)	≤8%	≤15%	≤30%	≤45%
Medium (>50% to ≤75% of full seeding)	≤8%	≤15%	≤20%	≤38%
Low (>19% to ≤50% of full seeding)	≤8%	≤15%	≤15%	≤25%
Very Low (>4 fish/mile to ≤19% of full seeding)	≤8%	≤11%	≤11%	≤11%
Critical (≤4 fish/mile)	0-8%	0-8%	0-8%	0-8%

Sub-aggregate and Basin-specific Spawner Criteria Data

			Critical		Spawner Status Intervals		
Sub-aggregate	**Miles of Available Spawning Habitat**	**100% of Full Seeding**	**4 fish/mile**	**12% of full seeding**	**19% of full seeding**	**50% of full seeding**	**75% of full seeding**
Northern	899	21,700	3,596	NA	4,123	10,850	16,275
North-Central	1,163	55,000	4,652	NA	10,450	27,500	41,250
South-Central	1,685	50,000	6,740	NA	9,500	25,000	37,500
Southern	450	5,400	NA	648	1,026	2,700	4,050
Total	*4,197*	*132,000*		*15,636*	*25,099*	*66,050*	*99,075*

* Parental spawner abundance status for the OCN aggregate assumes the status of the weakest sub-aggregate.

** Critical parental status is defined as ≤4 fish per mi for the Northern, North-Central, and South-Central sub-aggregates; because of high quality spawning habitat in the Rogue River basin, critical status for the Rogue River (Southern sub-aggregate) is defined as 12% of full seeding of high quality habitat.

Columbia River Upriver Bright Fall Chinook

The parties to U.S. v. Oregon are currently operating under the 2008-2017 Management Agreement. This agreement provides specific fishery management constraints for upriver spring, summer, and Fall Chinook, coho, sockeye and steelhead. Fall season fisheries in the Columbia River Basin below the confluence of the Snake River are managed according to the abundance-based harvest rate schedule shown in Table 5. In this table, Upriver Bright (URB) stock Chinook harvest rates are used as a surrogate for Snake River wild Fall Chinook harvest rates. Upriver Bright Fall Chinook escapement goals include 60,000 adult Fall Chinook (natural and hatchery) management goal above McNary Dam. Total harvest rates in combined Treaty Indian and non-Indian Columbia River fisheries increase with increased run size based on forecasted returns to the Columbia River.

Table 5. Columbia River Fall Management Period Chinook Harvest Rate Schedule for upriver bright Fall Chinook included the listed Snake River wild component.[1]

Expected URB River Mouth Run Size	Expected River Mouth Snake River Natural Origin Run Size [1]	Treaty Total Harvest Rate	Non-Treaty Harvest Rate	Total Harvest Rate	Expected Escapement of Snake R. Natural Origin Past Fisheries
<60,000	<1,000	20%	1.50%	21.50%	784
60,000	1,000	23%	4%	27.00%	730
120,000	2,000	23%	8.25%	31.25%	1,375
>200,000	5,000	25%	8.25%	33.25%	3,338
	6,000	27%	11%	38.00%	3,720
	8,000	30%	15%	45.00%	4,400

[1] If the Snake River natural fall Chinook forecast is less than level corresponding to an aggregate URB run size, the allowable mortality rate will be based on the Snake River natural fall Chinook run size.

FORCASTING TULE ABUNDANCE

The feasibility and effectiveness of an abundance-based fishing strategy depends in part on whether abundance can be predicted with reasonable accuracy and precision. Annual tule run size is currently predicted for fishery management purposes using sibling models for the LRH stock aggregate that consists primarily of lower river hatchery fish. Effective conservation-based management objectives would ideally be based on population-specific forecasts of wild fish. However, forecasts of aggregate or population-specific wild run components are not available for LCN tule Chinook at this time. Preliminary examinations by NMFS suggested that, absent better age composition data for the wild populations, it will be difficult to obtain forecasts of wild abundance that are meaningful to managers attempting to set harvest limits based on adult run size (Scheurell 2009) using currently available information. Therefore, this assessment also examined correlations between hatchery and wild population run sizes in order to evaluate whether the aggregate LRH forecast might serve as an effective indicator of wild population run strength.

This section reviews: 1) current methods of forecasting Columbia River LRH tule Fall Chinook abundance from LRH stock accounting, 2) forecast accuracy and precision of LRH, 3) correlations between hatchery and wild run size, and 4) the potential for forecast improvements.

Current Forecast Methods

Current forecast methods were summarized along with aggregate and population-specific information on run size and escapement, wild and hatchery composition, and age-composition. Correlations between age cohorts utilized in the sibling-based forecasts were reported.

Wild Fall Chinook numbers cannot practically be predicted due to a lack of reliable age composition data for many wild populations. Data is available for some of the larger populations (e.g. Coweeman and the Cowlitz). However, sample sizes are quite limited for the smaller, less productive populations due to the simple fact that escapement numbers are currently very low.

Sizes of the LRH Chinook run to the Columbia River mouth are currently predicted each year based on sibling relationships. Thus, the number of age 2 fish predicts the number of age 3 fish in the following year, 3's predict 4's, and 4's predict 5's. Figure 8 illustrates these relationships for the historical dataset back to the 1961 brood year. Forecasts of Age 2 numbers are typically based on a recent year average. Run composition typically averages 6 percent age 2, 37 percent age 3, 48 percent age 4, 8 percent age 5, and <1 percent age 6 (Table 6). Forecasts of LRH include both hatchery and natural fish. However, the hatchery component comprises the large majority of this run.

Relationships between age cohorts are not stable over time. For instance, Figure 8 shows a change in observed ratios coinciding with the ocean regime shift in the mid-1970s. Patterns also appear to be temporally autocorrelated at a smaller scale (although we did not attempt to quantify this effect). To accommodate these effects, annual forecasts are based on yearly decisions regarding which data periods appear to be most appropriate based on ad hoc judgments by a committee of stock assessment experts involved with the Columbia River fishery.

Current practice estimates numbers of fish recruited to ocean fisheries by back-calculating from the Columbia River forecasts based on approximate ocean harvest rates representative of recent return years.

Table 6. Historical run size of Columbia River tule Fall Chinook by age.

run	return						brood	Return						Age composition (by brood year)				
year	Age 2	Age 3	Age 4	Age 5	Age 6	total	year	Age 2	Age 3	Age 4	Age 5	Age 6	total	Age 2	Age 3	Age 4	Age 5	Age 6
1959							1959				0.8							
1960							1960			41.8	3.0							
1961							1961		42.4	85.7	7.7							
1962							1962	1.6	13.6	24.9	4.5		44.6	0.036	0.305	0.558	0.101	0.000
1963							1963	12.7	50.6	68.0	15.0		146.3	0.087	0.346	0.465	0.103	0.000
1964	1.6	42.4	41.8	0.8	0.0	86.6	1964	3.7	18.0	41.8	6.6		70.1	0.053	0.257	0.596	0.094	0.000
1965	12.7	13.6	85.7	3.0	0.0	115.0	1965	5.7	35.0	58.5	13.7		112.9	0.050	0.310	0.518	0.121	0.000
1966	3.7	50.6	24.9	7.7	0.0	86.9	1966	6.3	34.3	72.8	5.7		119.1	0.053	0.288	0.611	0.048	0.000
1967	5.7	18.0	68.0	4.5	0.0	96.2	1967	14.5	90.2	123.7	12.1		240.5	0.060	0.375	0.514	0.050	0.000
1968	6.3	35.0	41.8	15.0	0.0	98.1	1968	16.4	51.6	101.1	38.5		207.6	0.079	0.249	0.487	0.185	0.000
1969	14.5	34.3	58.5	6.6	0.0	113.9	1969	8.7	38.4	106.5	13.8		167.4	0.052	0.229	0.636	0.082	0.000
1970	16.4	90.2	72.8	13.7	0.0	193.1	1970	8.3	70.1	93.6	20.2		192.2	0.043	0.365	0.487	0.105	0.000
1971	8.7	51.6	123.7	5.7	0.0	189.7	1971	6.7	46.8	123.5	20.5		197.5	0.034	0.237	0.625	0.104	0.000
1972	8.3	38.4	101.1	12.1	0.0	159.9	1972	4.6	39.8	74.4	17.6		136.4	0.034	0.292	0.545	0.129	0.000
1973	6.7	70.1	106.5	38.5	0.0	221.8	1973	9.8	76.1	85.9	8.0		179.8	0.055	0.423	0.478	0.044	0.000
1974	4.6	46.8	93.6	13.8	0.0	158.8	1974	9.6	61.6	102.2	10.4	0.1	183.9	0.052	0.335	0.556	0.057	0.001
1975	9.8	39.8	123.5	20.2	0.0	193.3	1975	6.4	56.3	57.4	6.5	0.0	126.6	0.051	0.445	0.453	0.051	0.000
1976	9.6	76.1	74.4	20.5	0.0	180.6	1976	8.4	50.9	63.1	5.1	0.0	127.5	0.066	0.399	0.495	0.040	0.000
1977	6.4	61.6	85.9	17.6	0.0	171.5	1977	7.4	35.9	43.1	4.8	0.1	91.2	0.081	0.394	0.472	0.053	0.001
1978	8.4	56.3	102.2	8.0	0.0	174.9	1978	5.4	46.6	48.4	2.5	0.1	103.1	0.053	0.452	0.470	0.025	0.001
1979	7.4	50.9	57.4	10.4	0.0	126.1	1979	8.2	86.2	40.6	1.6	0.0	136.6	0.060	0.632	0.297	0.012	0.000
1980	5.4	35.9	63.1	6.5	0.1	111.0	1980	9.5	44.9	47.9	6.3	0.2	108.8	0.088	0.413	0.440	0.058	0.001
1981	8.2	46.6	43.1	5.1	0.0	103.0	1981	3.0	49.3	42.7	8.5	0.1	103.6	0.029	0.476	0.412	0.082	0.001
1982	9.5	86.2	48.4	4.8	0.0	149.0	1982	5.7	62.0	49.3	7.9	0.1	125.0	0.045	0.496	0.395	0.064	0.001
1983	3.0	44.9	40.6	2.5	0.1	91.1	1983	17.0	96.8	98.7	11.7	0.1	224.4	0.076	0.432	0.440	0.052	0.000
1984	5.7	49.3	47.9	1.6	0.1	104.6	1984	30.1	237.3	270.8	48.1	1.8	588.2	0.051	0.404	0.460	0.082	0.003
1985	17.0	62.0	42.7	6.3	0.0	128.0	1985	4.1	27.3	57.3	8.6	0.1	97.4	0.042	0.281	0.588	0.088	0.001
1986	30.1	96.8	49.3	8.5	0.2	184.9	1986	4.3	25.5	33.5	3.5	0.0	66.8	0.064	0.381	0.502	0.053	0.001
1987	4.1	237.3	98.7	7.9	0.1	348.2	1987	2.5	16.0	19.7	2.6	0.0	40.8	0.061	0.392	0.482	0.064	0.000
1988	4.3	27.3	270.8	11.7	0.1	314.2	1988	6.6	39.4	30.4	3.8	0.0	80.2	0.083	0.491	0.379	0.047	0.000
1989	2.5	25.5	57.3	48.1	0.1	133.4	1989	9.2	29.6	28.0	4.8	0.0	71.6	0.129	0.413	0.391	0.067	0.001
1990	6.6	16.0	33.5	8.6	1.8	66.6	1990	6.3	20.5	24.3	5.2	0.1	56.4	0.112	0.364	0.432	0.091	0.001
1991	9.2	39.4	19.7	3.5	0.1	71.9	1991	2.4	24.5	17.0	1.9	0.0	45.8	0.053	0.533	0.371	0.042	0.000
1992	6.3	29.6	30.4	2.6	0.0	68.9	1992	2.7	24.1	36.3	4.9	0.1	68.2	0.040	0.354	0.532	0.072	0.002
1993	2.4	20.5	28.0	3.8	0.0	54.8	1993	3.5	37.2	39.6	9.1	0.1	89.5	0.039	0.416	0.442	0.101	0.001
1994	2.7	24.5	24.3	4.8	0.0	56.3	1994	4.0	12.9	14.9	1.4	0.0	33.1	0.120	0.389	0.449	0.041	0.000
1995	3.5	24.1	17.0	5.2	0.0	49.9	1995	1.4	21.2	20.7	2.3	0.0	45.6	0.031	0.464	0.454	0.050	0.001
1996	4.0	37.2	36.3	1.9	0.1	79.5	1996	2.0	17.8	18.3	2.3	0.0	40.4	0.049	0.440	0.453	0.057	0.000
1997	1.4	12.9	39.6	4.9	0.0	58.8	1997	0.8	6.4	31.5	4.6	0.1	43.4	0.018	0.148	0.726	0.106	0.002
1998	2.0	21.2	14.9	9.1	0.1	47.2	1998	3.6	60.5	86.2	16.8	0.6	167.7	0.022	0.361	0.514	0.100	0.004
1999	0.8	17.8	20.7	1.4	0.1	40.7	1999	9.3	65.6	107.0	21.5	0.5	203.9	0.046	0.322	0.525	0.106	0.002
2000	3.6	6.4	18.3	2.3	0.0	30.6	2000	2.9	31.1	63.1	16.0	0.4	113.5	0.026	0.274	0.556	0.141	0.003
2001	9.3	60.5	31.5	2.3	0.0	103.6	2001	1.5	23.8	45.5	13.2	0.2	84.3	0.017	0.283	0.540	0.157	0.003
2002	2.9	65.6	86.2	4.6	0.0	159.4	2002	2.3	16.3	32.1	3.8	0.1	54.7	0.042	0.298	0.587	0.070	0.002
2003	1.5	31.1	107.0	16.8	0.1	156.5	2003	1.2	12.6	12.5	1.7	0.0	28.0	0.044	0.449	0.445	0.062	0.000
2004	2.3	23.8	63.1	21.5	0.6	111.4	2004	2.9	16.2	20.8	2.1	0.0	42.0	0.070	0.384	0.496	0.051	0.000
2005	1.2	16.3	45.5	16.0	0.5	79.5	2005	4.0	38.9	44.9	4.8		92.7	0.043	0.420	0.485	0.052	0.000
2006	2.9	12.6	32.1	13.2	0.4	61.3	2006	5.3	29.7	26.5								
2007	4.0	16.2	12.5	3.8	0.2	36.7	2007	8.9	71.6									
2008	5.3	38.9	20.8	1.7	0.1	66.9	2008	5.5										
2009	8.9	29.7	44.9	2.1	0.0	85.6	2009											
2010	5.5	71.6	26.5	4.8	0.0	108.5	2010											
Mean	6.5	44.8	59.1	9.3	0.1	119.8		6.5	44.8	59.1	9.3	0.2	120.4	0.055	0.373	0.495	0.076	0.001
Min	0.8	6.4	12.5	0.8	0.0	30.6		0.8	6.4	12.5	0.8	0.0	28.0	0.017	0.148	0.297	0.012	0.000
Max	30.1	237.3	270.8	48.1	1.8	348.2		30.1	237.3	270.8	48.1	1.8	588.2	0.129	0.632	0.726	0.185	0.004

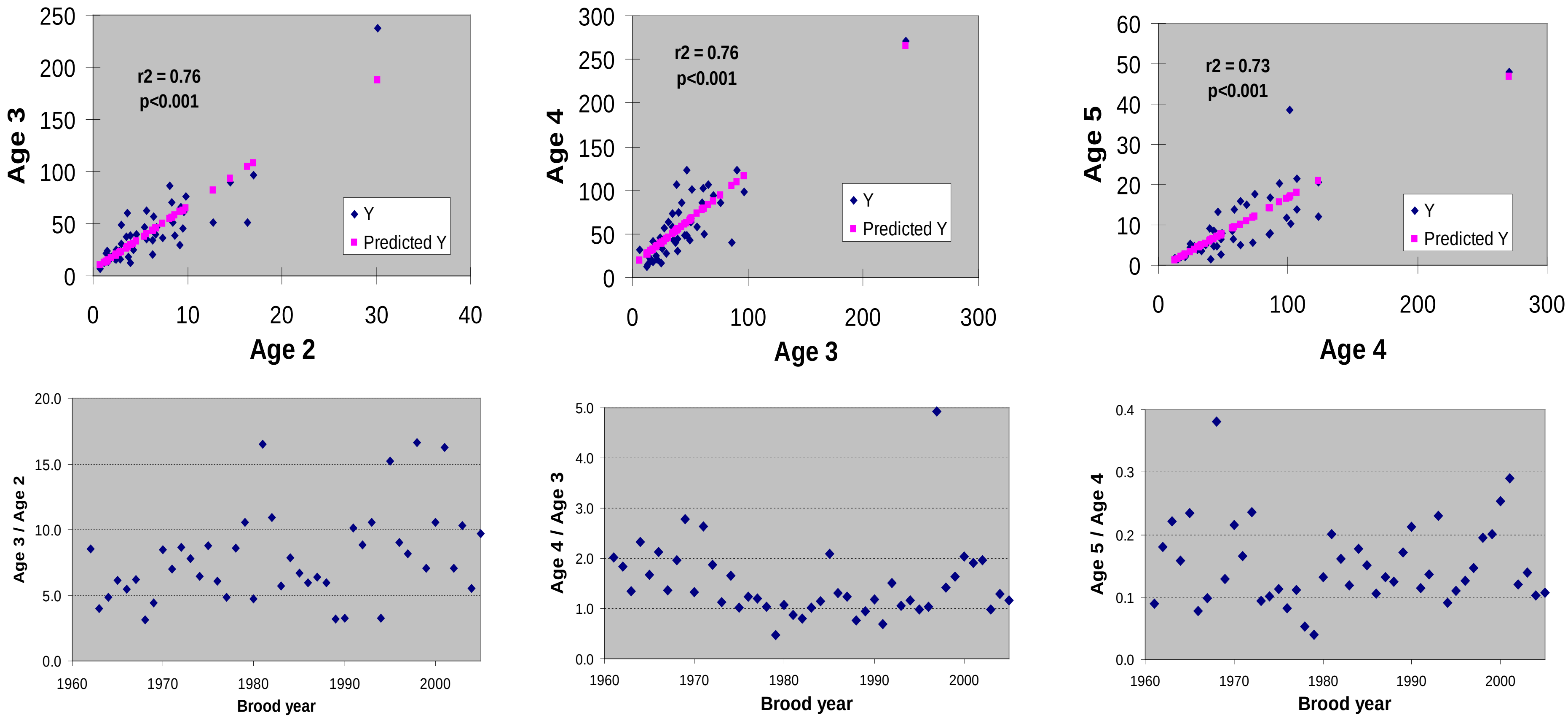

Figure 8. Relationships between age groups by brood year of Columbia River tule Fall Chinook at return (numbers in thousands).

Forecast Accuracy & Precision

Forecast accuracy and precision was estimated by a retrospective comparison of pre and post-season estimates of predicted and actual run size numbers for the aggregate LRH return. Numbers were based on Columbia River mouth returns of adults. Forecast error was estimated as [(predicted-actual)/actual] expressed as a percentage. Thus, negative numbers reflect under-predictions and positive number represent over-predictions. Accuracy was described based on the average of errors. Precision was described as the standard deviation of errors.

On average, forecasts of LRH Chinook have been relatively accurate over the period from 1980 through 2009 (Table 7). Error averaged just -2 percent for predicted number of total adults. However, errors were highly auto-correlated among years with a consistent pattern of under prediction from 1994 through 2006 (Figure 9). Forecasts were relatively accurate on average for age 3 (1 percent average error) and age 4 (-3 percent average error) (Table 7). Age 5 fish were more consistently under-predicted (-16 percent average error) although this age group typically comprises less than 10 percent of the run.

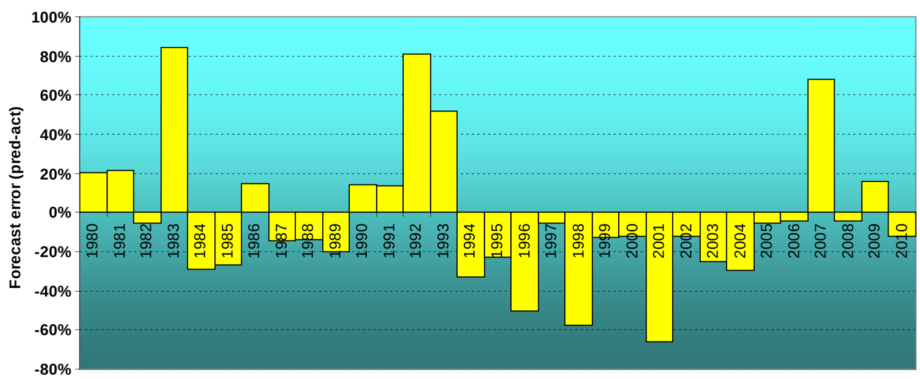

Figure 9. Past errors in forecasts of total annual adult returns of LRH tule Chinook to the Columbia River mouth, 1980-2009.

Annual predictions ranged from -66 percent to 85 percent of the actual return with a standard deviation of 37 percent over the period of record. The distribution of errors is slightly skewed to negative values although three quarters of values are within ±30 percent of the actual number.

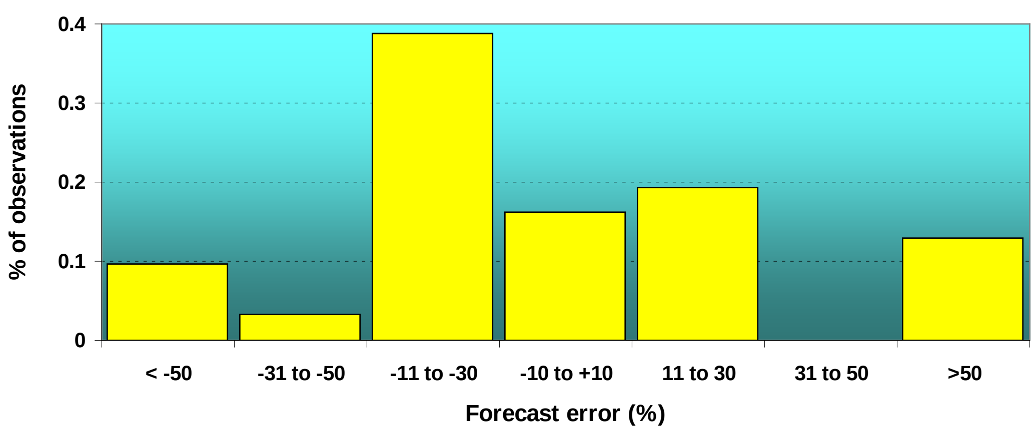

Figure 10. Frequency distribution of forecast errors.

Table 7. Forecast error based predicted and actual returns (thousands) of LRH Chinook to the Columbia River mouth, 1980-2009.

	Total Adults			Age 3			Age 4			Age 5		
Year	Predicted	Actual	Error	Predicted	Actual	Error	Predicted	Actual	Error	Predicted	Actual	Error
1980	127.3	105.6	21%									
1981	115.0	94.9	21%									
1982	132.2	139.5	-5%									
1983	162.5	88.1	85%									
1984	70.4	98.9	-29%	25.0	49.3	-49%	41.7	47.9	-13%	3.7	1.6	132%
1985	81.5	111.0	-27%	37.7	62.0	-39%	38.7	42.7	-9%	5.1	6.3	-19%
1986	177.6	154.8	15%	108.0	96.8	12%	65.4	49.3	33%	4.2	8.5	-51%
1987	294.9	344.1	-14%	189.0	237.3	-20%	100.9	98.7	2%	5.0	7.9	-37%
1988	267.7	309.9	-14%	36.5	27.3	33%	219.1	270.8	-19%	12.1	11.7	3%
1989	104.9	130.9	-20%	32.5	25.5	28%	40.6	57.3	-29%	31.8	48.1	-34%
1990	68.5	60.0	14%	22.4	16.0	40%	39.1	33.5	17%	7.0	8.6	-19%
1991	71.4	62.7	14%	52.1	39.4	32%	15.8	19.7	-20%	3.5	3.5	-1%
1992	113.2	62.6	81%	65.1	29.6	120%	47.2	30.4	55%	0.9	2.6	-65%
1993	79.3	52.3	51%	45.5	20.5	122%	30.7	28.0	10%	3.1	3.8	-18%
1994	36.1	53.6	-33%	14.1	24.5	-42%	19.1	24.3	-22%	2.9	4.8	-39%
1995	35.8	46.4	-23%	16.8	24.1	-30%	17.7	17.0	4%	1.3	5.2	-75%
1996	37.7	75.5	-50%	22.0	37.2	-41%	15.3	36.3	-58%	0.4	1.9	-79%
1997	54.2	57.4	-6%	25.3	12.9	96%	26.2	39.6	-34%	2.7	4.9	-45%
1998	19.2	45.3	-58%	7.6	21.2	-64%	8.0	14.9	-46%	3.6	9.1	-60%
1999	34.8	39.9	-13%	12.3	17.8	-31%	20.8	20.7	0%	1.7	1.4	25%
2000	23.7	27.0	-12%	5.5	6.4	-14%	16.2	18.3	-12%	2.0	2.3	-12%
2001	32.2	94.3	-66%	23.5	60.5	-61%	6.7	31.5	-79%	2.0	2.3	-13%
2002	137.6	156.4	-12%	60.6	65.6	-8%	72.7	86.2	-16%	4.3	4.6	-7%
2003	115.9	155.0	-25%	21.8	31.1	-30%	80.1	107.0	-25%	14.0	16.8	-17%
2004	77.1	109.1	-29%	13.3	23.8	-44%	45.8	63.1	-27%	18.0	21.5	-16%
2005	74.1	78.3	-5%	19.2	16.3	18%	44.6	45.5	-2%	10.3	16.0	-35%
2006	55.8	58.3	-4%	12.4	12.6	-1%	34.8	32.1	8%	8.6	13.2	-35%
2007	54.9	32.7	68%	19.4	16.2	20%	29.2	12.5	134%	6.3	3.8	64%
2008	59.0	61.6	-4%	26.6	38.9	-32%	30.9	20.8	48%	1.5	1.7	-13%
2009	88.8	76.7	16%	36.8	29.7	24%	48.7	44.9	8%	3.3	2.1	55%
2010	90.6	103.0	-12%	43.7	71.6	-39%	38.2	26.5	44%	8.7	4.8	80%
2011	133.5			38.7			90.6			4.3		
Mean	*94.6*	*99.5*	*-3%*	***36.9***	41.3	***0%***	***45.9***	48.9	***-2%***	***6.2***	8.1	***-12%***
SD	*63.8*	*70.9*	*36%*	***36.7***	45.1	***51%***	***42.2***	51.4	***41%***	***6.8***	9.7	***46%***

Hatchery-Wild Correlations

Methods

Correlations were examined among historical data on total LRH returns, hatchery releases, a hatchery survival index, and naturally-produced fish returning to Washington streams. LRH return data to the Columbia River mouth were available for 1964-2010 from WDFW. Hatchery release data for LRH tule fall Chinook released downstream from Bonneville Dam from 1964-2008 were compiled from the Pacific Marine Fisheries Commission (PSMFC) Regional Mark Information System database. An annual survival rate index was estimated as the quotient of total brood year LRH return across all ages and total hatchery release. For comparison with run year returns, a run year survival index was also calculated by averaging brood-year survival rates among ages weighted in proportion to the age composition in each year. Both these indices would of course be inflated by a small amount in proportion to the number of wild fish included the LRH return (since wild juveniles are not included in the denominator).

Results

The aggregate LRH return was at least partially correlated with wild numbers considered in aggregate and in many or most wild populations throughout the historical data set and in the recent 10-year period. Table 8 and Table 9 show the correlation coefficients and significance levels among pair-wise comparisons of LRH and wild run size numbers. The strongest relationships were observed between the LRH and wild total returns (Figure 11). Population-specific correlations to the LRH return varied as did correlations among the individual wild populations.

Correlations were similar for the survival rate index and the total LRH return. Stronger correlations were expected to the survival rate index which controlled for effects of variable hatchery release numbers over time. Forecasts based strictly on hatchery numbers may be confounded by effects of changes in hatchery release levels. Indices based on survival rather than numbers should avoid this effect. However, this was not apparent. The survival index did appear to be related to hatchery release numbers. Survival rates increased concurrent with reduction in hatchery release numbers in the late 1990s in response to reductions in Federal Mitchell Act funding. Effects of ocean conditions, ecological factors, and release numbers were not distinguished, but at least some of this improvement likely resulted from the reduction in the less successful hatchery programs measured in terms of lower juvenile to adult survival success.

These results suggest that further analyses of an abundance-based approach to tule harvest rate management are appropriate. While wild numbers cannot be predicted directly at this time, aggregate forecasts based primarily on hatchery returns appear to be a suitable proxy due to common effects of marine conditions to which both hatchery and wild fish are subject.

Table 8. Correlation matrix (r values) of LRH run size to the Columbia River and wild run size numbers to Washington streams, 1977-2010 run years .

	LRH run	LRH surv.	Wild total	Grays	Mill/Ab/Ger	Elochoman	Coweeman	Cowlitz	Kalama	Lewis	Washougal
LRH run size	1.00	--	--	--	--	--	--	--	--	--	--
LRH survival	0.80***	1.00	--	--	--	--	--	--	--	--	--
Wild total	0.65***	0.59***	1.00	--	--	--	--	--	--	--	--
Grays	0.47***	0.51***	0.18	1.00	--	--	--	--	--	--	--
Mill/Ab/Ger	0.67***	0.49***	0.66***	0.47***	1.00	--	--	--	--	--	--
Elochoman	0.27*	0.42**	0.41**	0.14	0.46***	1.00	--	--	--	--	--
Coweeman	0.08	0.00	0.63***	0.12	0.12	0.07	1.00	--	--	--	--
Cowlitz	0.28**	0.61***	0.47***	0.02	0.08	0.49***	0.12	1.00	--	--	--
Kalama	0.55***	0.27*	0.75***	0.21	0.48***	0.08	0.60***	0.12	1.00	--	--
Lewis	0.24*	0.43**	0.18	0.44***	0.21	0.05	0.15	0.18	0.07	1.00	--
Washougal	0.03	0.10	0.35**	0.31	0.15	0.01	0.27*	0.31**	0.03	0.13	1.00

**** p-value<0.01*
*** p-value<0.10*
** p-value<0.20*

Table 9. Correlation matrix (r values) of LRH run size to the Columbia River and wild run size numbers to Washington streams, 2000-2010 run years .

	LRH run	LRH surv.	Wild total	Grays	Mill/Ab/Ger	Elochoman	Coweeman	Cowlitz	Kalama	Lewis	Washougal
LRH run size	1.00	--	--	--	--	--	--	--	--	--	--
LRH survival	0.99***	1.00	--	--	--	--	--	--	--	--	--
Wild total	0.92***	0.90**	1.00	--	--	--	--	--	--	--	--
Grays	0.06	0.14	0.01	1.00	--	--	--	--	--	--	--
Mill/Ab/Ger	0.50*	0.49*	0.63**	0.05	1.00	--	--	--	--	--	--
Elochoman	0.50*	0.44*	0.70**	0.20	0.79***	1.00	--	--	--	--	--
Coweeman	0.64**	0.57**	0.62**	0.46*	0.03	0.34	1.00	--	--	--	--
Cowlitz	0.88***	0.87***	0.89***	0.00	0.24	0.45*	0.66**	1.00	--	--	--
Kalama	0.11	0.15	0.03	0.24	0.39	0.18	0.16	0.25	1.00	--	--
Lewis	0.25	0.26	0.23	0.10	0.46*	0.03	0.23	0.05	0.58**	1.00	--
Washougal	0.44*	0.48*	0.42*	0.47*	0.14	0.16	0.35	0.52*	0.06	0.07	1.00

**** p-value<0.01*
*** p-value<0.10*
** p-value<0.20*

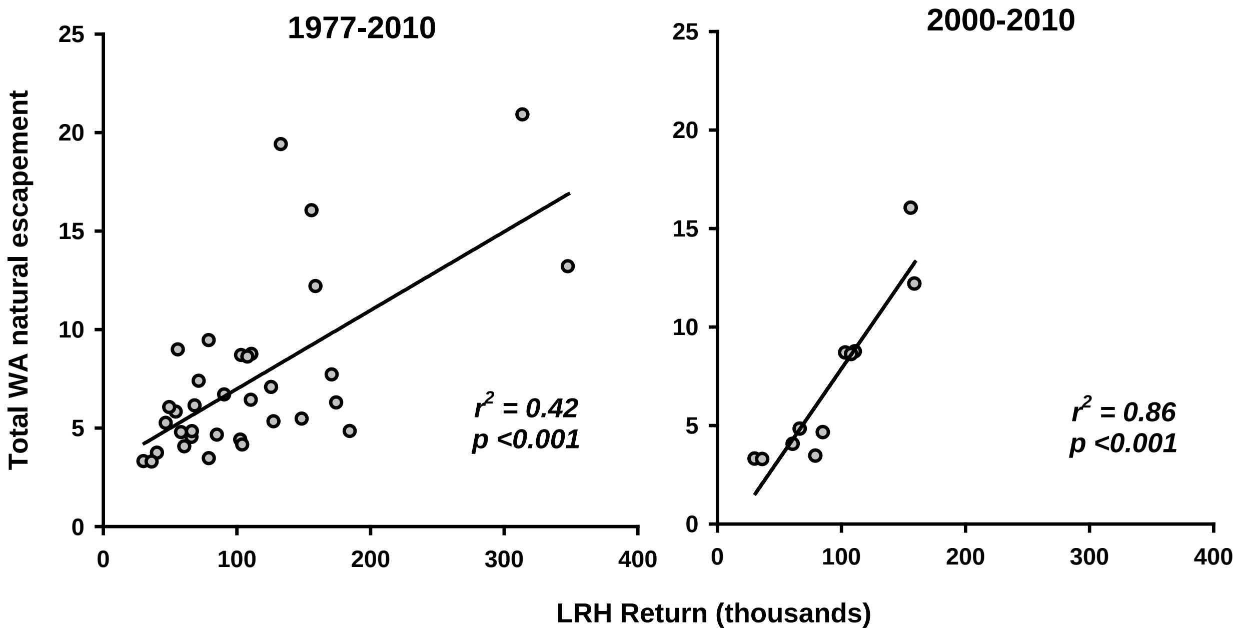

Figure 11. Correlations between the aggregate LRH lower river hatchery return of tule Fall Chinook to the Columbia River and total escapement of natural-origin tules to Washington streams downstream from Bonneville Dam.

Discussion

It should be noted that the reported significance levels assume independence among data points but annual run sizes and survival are not independent of adjacent years. This is not important for the purpose of this analysis, to find a pattern in the data without inferring the underlying biological process. The abundance of a given age class at a given time is used to compute both LRH run and LRH survival. Thus, observational error (not biological variation) of abundance can induce a false correlation between the two variables. P-values of 0.1 and 0.2 are a relatively low standard for “significance,” especially since the data are not independent. For the 55 correlations examined above, we should expect about three spurious correlations if we applied the usual 0.05 significance level. However, results are clearly indicative of a significant partial correlation between hatchery and natural tule Fall Chinook numbers.

Potential for Forecast Improvements

Forecasts can be improved in a number of ways. Recommendations for improvements include:

1) Forecasts based on wild fish returns, rather than hatchery fish returns, should be developed. Ideally these forecasts would be population-specific with mixed stock management decisions based on consideration for an aggregate of the weaker stocks. Development of such forecasts entails implementing, or continuing to implement, monitoring of wild fish.

2) In order to make forecasts more accurate, environmental variables that account for variability in returns should be identified and included in forecast models.

3) Alternative forecasting methods should be explored.

The remainder of this section describes an initial exploratory effort addressing the second and third recommendations above.

Methods

The potential for improvement in forecast accuracy was evaluated by examining the effects of incorporating indicators of ocean environmental conditions into the forecast models. However, the accuracy of traditional forecasting methods such as multiple regression can be compromised by high co-linearity among independent variables – metrics of marine conditions are known to be highly correlated with one another. Therefore an autoregressive neural network approach was explored to test the feasibility for improving tule forecasts by incorporating various metrics related to marine conditions.

A neural network is a machine learning method, with origins in the field of artificial intelligence. Neural networks are widely applied in engineering and economic contexts (e.g. missile guidance systems, stock market prediction) but are seldom used in ecological science. Nonetheless, neural networks have properties that make them inherently and demonstrably superior to more traditional methods such as generalized linear models. In particular, neural networks are well-suited to problems where multiple interacting factors nonlinearly influence some phenomenon of interest. This is precisely the nature of the LRH forecast problem, with the exception that both LRH abundance and the marine conditions used to predict LRH abundance are time-series. For this reason, a neural network was applied with internal structure that accommodates the time-series nature of these data was evaluated. This kind of neural network is known as a NARX network (nonlinear autoregressive network with exogenous inputs), and has the form

$$y_t = f(y_{t-1}, y_{t-2}, \ldots y_{t-4}, x1_{t-1}, x1_{t-2}, \ldots x1_{t-4}, x2_{t-1}, x2_{t-2}, \ldots x3_{t-4}, \ldots xn)$$

where the function f includes complex interactions among the n different predictor variables, x.

Variables compared to the LRH aggregate run size included the number of jacks in the previous two runs, the Pacific Decadal Oscillation (PDO), North Pacific Gyre Oscillation (NPGO), Ocean Nino Index (ONI), and multivariate ENSO Index (MEI). Ocean data were as reported by Scheurell (2009) and Rupp et al. (2010). Analyses were limited to data that are available in the pre-season time frame when forecasts are made. (Some ocean data considered in other analyses were only available post-season.)

Fitting a neural network is unlike traditional methods because the predictions of a complex network are capable of exactly matching observations. Thus, the essence of fitting a neural network is to prevent the network from becoming overfit (i.e. the model not only fits the signal, but also fits the noise. This results in false confidence in the model's prediction of new observations). Overfitting is prevented by withholding data from the model fitting process and using it to evaluate model performance. The original data set includes 40 observations (1962-2001). Since lag-4 autoregressive framework was used, there are 36 observations that can be predicted. These 36 observations were pseudorandomly broken into three groups: i) 26 observations were used to fit the model, ii) five observations were used to determine when the model begins to become overfit, and iii) five observations were used as an independent test of model predictions. The partitioning of the 36 observations into these three groups is pseudorandom because I repeated the process of dividing data and fitting the model was repeated several times. Estimation stopped when the model effectively predicted the data withheld for testing (group iii).

Model performance was assessed based on a statistic called Ordinary Cross-Validation (OCV) that describes the predictive ability of the model (Rupp et al. 2010). The process will: (1) leave out a single point, (2) fit the model, (3) obtain a prediction of the point that was left out, (4) subtract the empirically observed value from the prediction, and (5) square this difference. These steps are repeated until every point has been sequentially left out. Summing all the values obtained on the fifth step yields the numerator in the equation below. The denominator is simply the variance of the entire data set.

$$\text{OCV} = 1 - \frac{\sum_{i=1} (\hat{y}_i - y_i)^2}{\sum_{i=1} (\bar{y} - y_i)^2}$$

The approach applied simultaneously leaves out 12 points rather than sequentially leaving out all the points. A statistic that is similar to OCV was calculated.

$$1 - \frac{\sum_{i=1}^{j=6} (\hat{y}_i - y_i)^2 / 6}{\sum_{k=1}^{l=36} (\bar{y} - y_k)^2 / 36}$$

Where j=5 are the five points used as independent tests (red dots, Figure 12). Using averages in the numerator and denominator rather than the sum, as in OCV, rescales the statistic to a single observation.

Results

Results of the nonlinear autoregressive network model are displayed in Figure 12. The model appeared to reasonably predict aggregate LRH abundance. Model results provided a rescaled goodness of fit value of 0.86. This compares favorably with OCV scores of approximately 0.6 – 0.7 estimated by Rupp et al. (2010) for coho forecast models. However, LRH model results should be interpreted cautiously because the year of extremely high abundance (Time = 18) contributes to the denominator of the equation but does not contribute to the numerator. This would not be true of the OCV value. Furthermore, as with any neural network, concern that this model is over fit is legitimate.

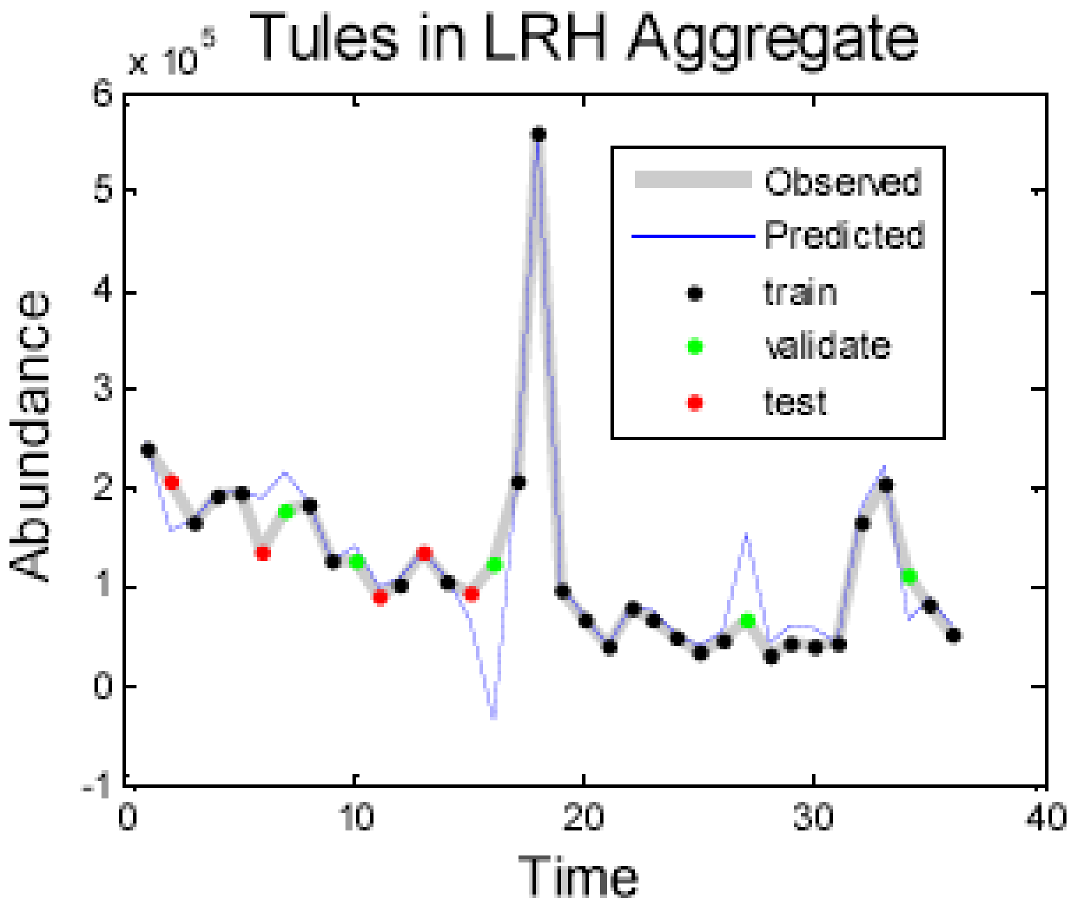

Figure 12. Results of an autoregressive neural network fit to LRH aggregate Tule Chinook abundance. The x-axis is years, beginning with 1966. All points are empirical observations. The black points were pseudo-randomly chosen to train/fit the model. Training/fitting stopped when the difference between predictions and the green points began to increase (i.e. the model showed evidence of overfitting). The red points were never used during model development and can therefore be used as an independent test of model performance.

Discussion

While wild numbers cannot be forecast directly, their partial correlation with LRH abundance makes the aggregate forecasts a suitable index for implementation of an abundance-based management approach.

Analyses of alternative forecast methods suggest that it might be feasible to improve forecast accuracy of LRH abundance by incorporation of some combination of marine indicator variables. However, any proposed improvements in forecast techniques will need to be tested with pre- and post-season comparisons of alternative forecast methods. In the interim, the TCW recommends that considerations of abundance-based fishing strategies for LCN tule Fall Chinook continue to be based on current forecast methods.

Analyses also highlight the need for improvements in data on natural population status and trends, including numbers, age composition, hatchery fractions, and productivity. Both Oregon and Washington have initiated significant efforts to augment existing monitoring of natural populations with additional sampling. It is likely that additional data will at some point improve our ability to more directly forecast natural abundance at a population scale.

FISHING RATES, CONTRIBUTIONS & EFFECTS

While fishery impact limits on LCN tule Fall Chinook obviously affect harvest of this stock, they can also constrain access to harvest of other Chinook stocks in mixed stock fisheries occurring in the ocean and Columbia River. Thus, conservation benefits of lower impact rates limits on natural stocks can come at the cost of significant harvest reductions of other stocks. Impact reductions and costs in foregone harvest do not fall on all fisheries evenly or in proportion to their share of the harvest impact due to the particulars of regulatory authorities and management agreements governing fisheries that impact LCN tule Fall Chinook.

This section: 1) summarizes recent harvest patterns of Columbia River tule Fall Chinook, 2) describes management of LCN tule Fall Chinook including fishery effects, and 3) evaluates potential the impact of different ceiling exploitation rates on ocean and Columbia River fisheries.

Harvest patterns

Recent harvest patterns were summarized based on total and fishery-specific annual impact rates to establish a baseline point of reference. All rates are expressed in terms of adult equivalents, the same metric used for fishery impact assessment for ESA. Observed rates were taken from post-season runs of FRAM using actual landings by fishery and post season estimates of FRAM stock abundances. Observed and target rates were compared for each year to identify fishery implementation uncertainty.

LCN tule Fall Chinook are harvested in ocean fisheries from Alaska, Oregon, and in the Columbia River, and the cumulative exploitation rate in combined fisheries is significant. Prior to 1990, total exploitation rates regularly reached 0.65 and rates exceeding 0.80 were seen in some years (Figure 13, Table 10). Rates were substantially reduced around the mid-1990s with reductions in Council and Canadian ocean fisheries during a period of low runs for many stocks. However, exploitation increased again by 2000 as fisheries recovered. Fishery impact limits were established by NMFS beginning in 2002. Limits were reduced from 0.49 in 2002-2006, to 0.42 in 2007, 0.38 in 2009-2010, and 0.37 in 2011.

The majority of the ocean harvest of LCN tule Fall Chinook currently occurs in fisheries off Alaska and Canada which are governed by the PST (Figure 13). Canadian fisheries, primarily off the west Coast of Vancouver Island (WCVI), accounted for about 39 percent of the total fishery exploitation rate on LCN tules from 2001-2010. In 2010, impact included approximately 0.14 in Canada and SE Alaska ocean, 0.14 in Council fisheries, and 0.06 in Columbia River fisheries.

Tule Fall Chinook typically comprise only a limited portion of the harvest in mixed stock fisheries. LRH tules typically comprise only about 1 percent of the total Chinook harvest in southeast Alaska and northern British Columbia fisheries, increasing to about 10-15 percent in the WCVI fishery (CTC 2011). These fisheries harvest a broad mixture of stocks originating in Alaska, Canada, Oregon and Washington. In the Council fisheries north of Cape Falcon, Oregon, LRH tules typically comprise about 20-30 percent of the harvest (CTC 2009).

In the Columbia River, LRH tule Fall Chinook typically comprise only about 20 percent on average of the annual Chinook harvest in sport and commercial fisheries downstream from

Bonneville Dam. The majority of the Columbia River harvest is of other Fall Chinook stocks, including Upriver Brights destined for natural spawning areas upstream from Bonneville Dam, hatchery-produced bright stocks destined for mid-Columbia facilities, and Bonneville Pool Hatchery tules produced at Spring Creek Hatchery. Tules are more important in some Columbia River fisheries than others. For instance, tules typically comprise a higher percentage of the harvest in the Buoy 10 sport fishery at the mouth of the Columbia than in other sport and commercial fisheries upstream.

Since fishery impact limits were established, ERs have averaged 0.02 less than the established limits although annual rates may have been more or less than the limits due to variability in run sizes relative to forecasts and variability in observed versus expected catches (Table 10, Figure 14). Actual rates have been 0.01 to 0.08 less than the ceiling for the last three years.

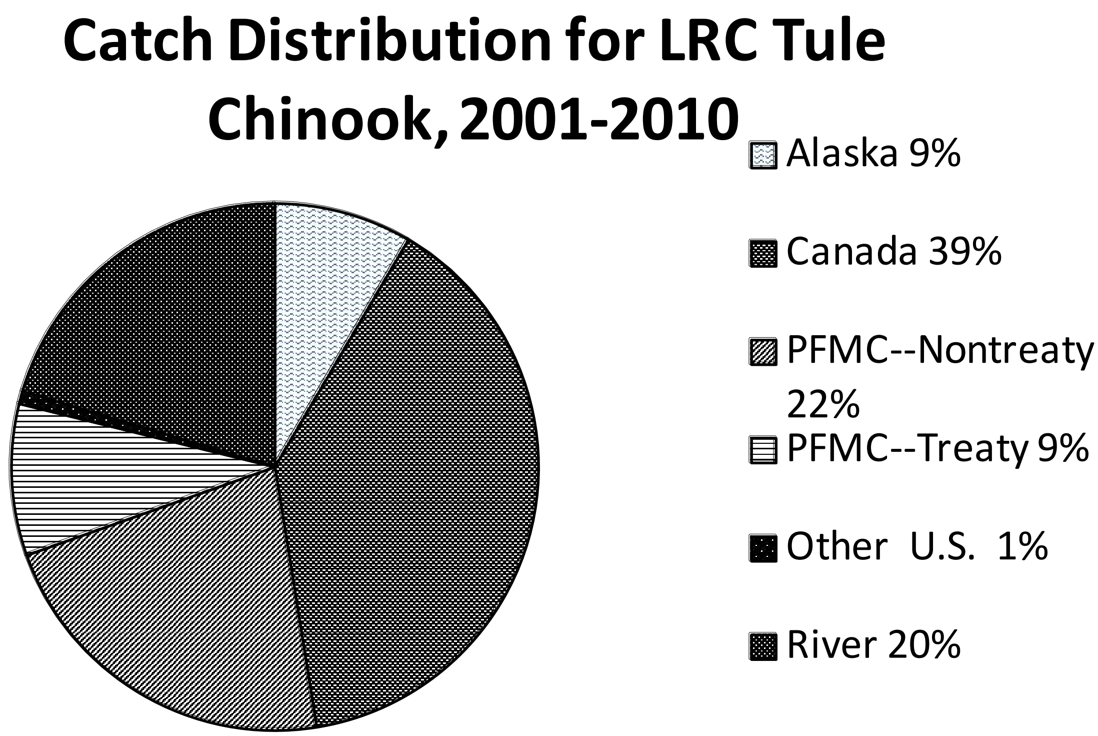

Figure 13. Catch distribution for lower Columbia natural-origin tule Fall Chinook, 2001-2010.

Table 10. Exploitation rates (% in adult equivalents) by fishery for lower Columbia natural-origin tule Fall Chinook from FRAM post-season model runs.

Year	Alaska	Canada	Council--Nontreaty	Council--Treaty	Other So. U.S. Marine	River	Total ER	ESA ER Ceiling
1983	4%	37%	16%	2%	2%	7%	68%	--
1984	4%	40%	4%	1%	3%	17%	68%	--
1985	4%	30%	11%	2%	3%	10%	60%	--
1986	3%	30%	11%	1%	4%	29%	78%	--
1987	4%	28%	12%	2%	3%	29%	78%	--
1988	2%	29%	14%	4%	3%	30%	81%	--
1989	3%	22%	17%	5%	3%	16%	66%	--
1990	3%	31%	18%	6%	3%	5%	65%	--
1991	3%	30%	9%	4%	3%	12%	61%	--
1992	3%	37%	15%	4%	3%	7%	68%	--
1993	3%	32%	12%	5%	3%	9%	64%	--
1994	4%	34%	0%	1%	1%	3%	44%	--
1995	4%	22%	3%	2%	1%	6%	38%	--
1996	4%	5%	3%	3%	1%	9%	24%	--
1997	5%	14%	5%	3%	2%	11%	39%	--
1998	4%	11%	4%	3%	0%	11%	33%	--
1999	4%	11%	6%	5%	0%	15%	41%	--
2000	5%	18%	7%	2%	0%	10%	42%	--
2001	3%	14%	7%	3%	0%	8%	35%	--
2002	4%	17%	12%	3%	0%	7%	42%	49%
2003	3%	20%	16%	3%	0%	4%	47%	49%
2004	4%	21%	10%	5%	0%	6%	46%	49%
2005	4%	17%	9%	6%	0%	12%	49%	49%
2006	4%	17%	9%	6%	0%	16%	53%	49%
2007	5%	19%	9%	6%	0%	9%	48%	42%
2008	3%	14%	5%	3%	0%	7%	33%	41%
2009 a/	3%	15%	5%	2%	1%	11%	37%	38%
2010 a/	3%	11%	11%	3%	0%	6%	35%	38%

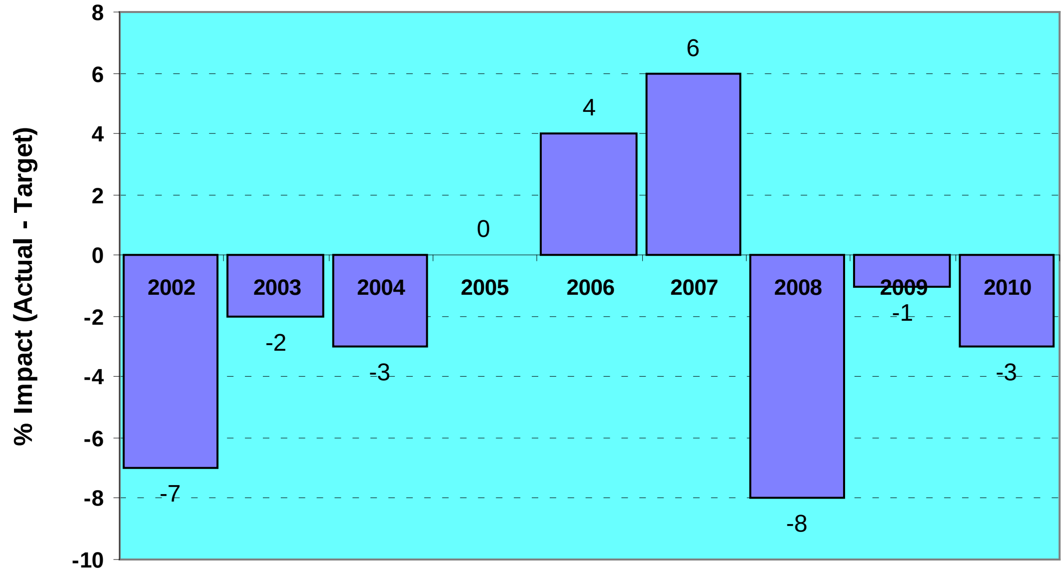

Figure 14. Differences between actual and target fishery impact ceilings for Lower Columbia tule Fall Chinook.

Fishery Management

Management of LCN tule Fall Chinook in freshwater and ocean fisheries was described to provide a context for consideration of future changes associated with implementation of potential abundance-based management strategies. These descriptions establish how different LCN tule Fall Chinook abundance and fishing rates generally affects ocean and Columbia River fisheries.

In Council ocean fisheries from the U.S. Canada Border to Cape Falcon, Chinook harvest is managed to:

1. Comply with ESA consultation standards for LCN tule Fall Chinook, Lower Columbia River wild bright Fall Chinook, and Snake River wild Fall Chinook;
2. Meet treaty Indian sharing obligations and the allocation provisions in the Salmon Fishery Management Plan (FMP);
3. Meet provisions of the PST; and
4. To the extent possible, provide for viable ocean and in-river fisheries while meeting natural stock escapement objectives and hatchery broodstock needs (PFMC 2011).

Exploitation rate limits for ESA-listed coho stocks can also constrain fisheries and limit access to otherwise harvestable Chinook. Which of these constraints limits the fisheries in any given year depends on the mixture of stock-specific abundances and the resulting stock composition available to each fishery. In 2010, the primary constraint for North of Falcon ocean fisheries was the LRN tule Chinook ESA consultation standard of no more than a 0.38 exploitation rate in all combined marine and freshwater fisheries. Exploitation rates are estimated for a composite of Washougal, Kalama, Cowlitz, and Big Creek hatchery tules as a surrogate for natural tules. Other ESA consultation standards include a spawning escapement of 5,700 for Lower Columbia

River wild bright Fall Chinook in the North Fork Lewis River, at least a 30 percent reduction in the total ocean age-3 and age-4 adult equivalent (AEQ) exploitation rate of Snake River Fall Chinook from the 1988-1993 average, and McNary Dam escapement targets.

Fisheries in southeast Alaska and troll and sport fisheries in northern Canadian and WCVI have been managed since 1999 under a PST framework regulating Chinook harvest under aggregate abundance-based management (AABM) regimes. These fishery management regimes establish a catch ceiling derived from estimates of total aggregate abundance of all stocks contributing to the AABM fisheries and indexed to a series of stepped target harvest rates. For fisheries not driven by AABM regimes, management is individual stock based (ISBM) with provisions in the treaty limiting the aggregate impact on any depressed stock across all ISBM fisheries. The 1999 agreement established conservation obligations to reduce harvest rates on depressed Chinook stocks by 36.5 percent for Canadian fisheries and 40 percent for U.S. fisheries, relative to levels observed during 1979-1982. In May, 2008 the PSC recommended to the Governments of Canada and the United States a new bilateral agreement for the conservation and harvest sharing of Pacific salmon. The new fishing regimes are in force from the beginning of 2009 through the end of 2018 and are contained in Chapters 1, 2, 3, 5, and 6 of Annex IV of the Treaty. The 2008 Agreement, contained two key provisions pertaining to the AABM fisheries; a shift towards management for total mortality rather than landed catch and reductions from the levels in the 1999 Agreement of 15 percent in southeast Alaska and 30 percent in WCVI fisheries. Impacts to all stocks in the AABM fisheries are reduced; hence, LCN tules have lower ERs in the northern fisheries, especially in WCVI.

Columbia River treaty Indian and non-Indian fisheries are managed under a 10-year agreement adopted in 2008 between *U.S. versus Oregon* parties. This agreement limits non-Indian fisheries in the lower 145 miles of Columbia River downstream from Bonneville Dam in order to provide adequate numbers of salmon to treaty Indian fishing areas upstream from Bonneville Dam. Combined Columbia River fisheries are typically constrained by consultation standards for LCN tule and Snake River wild Chinook. Fall fisheries below the confluence of the Snake River are managed according to an abundance-based harvest rate schedule that allocates harvest rates for Upriver Bright Fall Chinook between treaty Indian and non-Indian fisheries within the ceiling established by NMFS for total allowable harvest rates. Fisheries in the lower Columbia River downstream from Bonneville Dam are shaped within the constraints of this agreement to optimize harvest and opportunity while also meeting other objectives and constraints including the portion of the LCN tule harvest rate ceiling identified via the Council regulatory process for Columbia River fisheries. Current limits on LCN tule impacts are being met by a combination of fishery reductions and area restrictions. The recent year strategy has been to limit the Buoy 10 sport fishery and to move other fisheries targeting upriver fall Chinook to areas above the Lewis River. In 2010, ERs on LCN tule Chinook in combined sport and commercial fisheries were limited to just 0.08. Further restrictions might require mark-selective regulations in specific fisheries (such as the Columbia River Buoy 10 sport fishery) or shorter retention fisheries.

Impact Ceiling Effects on Fisheries

For perspective on the effect to preseason fisheries shaping of a “low” status under an abundance-based management system for LCN tule Chinook, the 2009-2011 preseason estimates of total ER for LCN tule Chinook were modeled in FRAM. A “low” status abundance forecast was defined as returns to the Columbia River of 40,000 LRH adult age 3-5 Chinook. All fishery catches/inputs and stock abundances (including mark rates) were unchanged from the preseason runs except that the LRH abundances were lowered to achieve a terminal run of about 40,000 adults.

Table 3 contains exploitation rates by fishery group for 2009-2011 preseason model runs and corresponding estimates with a “low” abundance for LRH stock using the FRAM and in-river harvest model system currently employed during Council preseason management. LCN tule Chinook exploitation rates in Alaska and Canada fisheries ranged between 0.15-0.19 and averaged 0.17 in both the preseason and low LRH abundance examples. Exploitation rates in Council fisheries ranged between 0.12-0.15 and averaged 0.14 in the preseason runs and 0.17 in the low abundance runs (modeled with same catch quotas). The exploitation rate in the river fisheries ranged from 0.06-0.08. Exploitation in the treaty Indian troll fishery was slightly lower than the river fishery impacts. Council managed fisheries south of Cape Falcon were severely restricted in 2009 and 2010 but approached a more normal season structure in 2011 when the exploitation rate was estimated to be about 0.02. In general, the fishing seasons in the southern U.S. were constrained by LCN tule Chinook in river and ocean fisheries north of Cape Falcon and by Sacramento-Central Valley Chinook in fisheries south of Cape Falcon. Stocks contributing to northern fisheries were generally abundant, which provided for higher aggregated abundances and fishing levels in the AABM fisheries. In these examples, impacts on LCN tule Chinook in the southern U.S. would have to be reduced by nearly 50 percent in order to remain under an ER ceiling of 0.28 and by 23 percent to remain under a ceiling of 0.33, assuming current conditions in northern fisheries.

Table 11. Projected exploitation rates for LCN tule Chinook from FRAM and in-river harvest models.

	2009		2010		2011		2009-11 Average	
Fishery	**LRH at 88.8K**	**LRH at 40K a/**	**LRH at 90.6K**	**LRH at 40K a/**	**LRH at 133.5K**	**LRH at 40K a/**	**Preseason**	**LRH at 40K a/**
AK-BC	0.164	0.164	0.147	0.148	0.188	0.189	0.166	0.167
Council Total	**0.149**	**0.188**	**0.163**	**0.180**	**0.121**	**0.143**	**0.144**	**0.170**
No. of Falcon	0.149	0.188	0.146	0.163	0.101	0.123	0.132	0.158
(treaty troll only)	0.072	0.088	0.045	0.051	0.039	0.046	0.052	0.062
So. of Falcon	0.000	0.000	0.017	0.017	0.020	0.020	0.012	0.012
Other So. U.S. marine	0.006	0.005	0.005	0.003	0.005	0.005	0.005	0.004
River @ preseason HR	0.078	0.074	0.079	0.079	0.056	0.054	0.071	0.069
So. U.S subtotal	0.233	0.267	0.247	0.262	0.182	0.202	0.221	0.244
LCN Tule Total ER	**0.397**	**0.431**	**0.394**	**0.410**	**0.370**	**0.391**	**0.387**	**0.411**

% Reduction in So. U.S.	**2009**		**2010**		**2011**		**2009-11 Average**	
to achieve total ER of:	**Preseason**	**LRH at 40K**	**Preseason**	**LRH at 40K**	**Preseason**	**LRH at 40K**	**Preseason**	**LRH at 40K**
0.20	85%	87%	79%	80%	93%	95%	85%	86%
0.22	76%	79%	70%	73%	82%	85%	76%	78%
0.27	55%	60%	50%	53%	55%	60%	53%	58%
0.32	33%	42%	30%	34%	27%	35%	30%	37%
0.37	12%	23%	10%	15%	0%	10%	8%	17%

a/ Modeled with same preseason quotas in Council fisheries North of Cape Falcon.

Table 12. Range and average ER for LCN tule Chinook for 2009-11 FRAM preseason runs with preseason abundances and LRH abundance at 40,000.

	2009-11 Preseason			2009-11 Low Abundance		
Fishery	High	Low	Average	High	Low	Average
AK-BC	18.8%	14.7%	16.6%	18.9%	14.8%	16.7%
NoF Nontreaty	10.1%	6.2%	8.0%	11.2%	7.7%	9.6%
NoF Treaty	7.2%	3.9%	5.2%	8.8%	4.6%	6.2%
So. of Falcon	2.0%	0.0%	1.2%	2.0%	0.0%	1.2%
Other U.S	0.6%	0.5%	0.5%	0.5%	0.3%	0.4%
River	7.9%	5.6%	7.1%	7.9%	5.4%	6.9%
So. U.S. subtotal	24.7%	18.2%	22.1%	26.7%	20.2%	24.4%

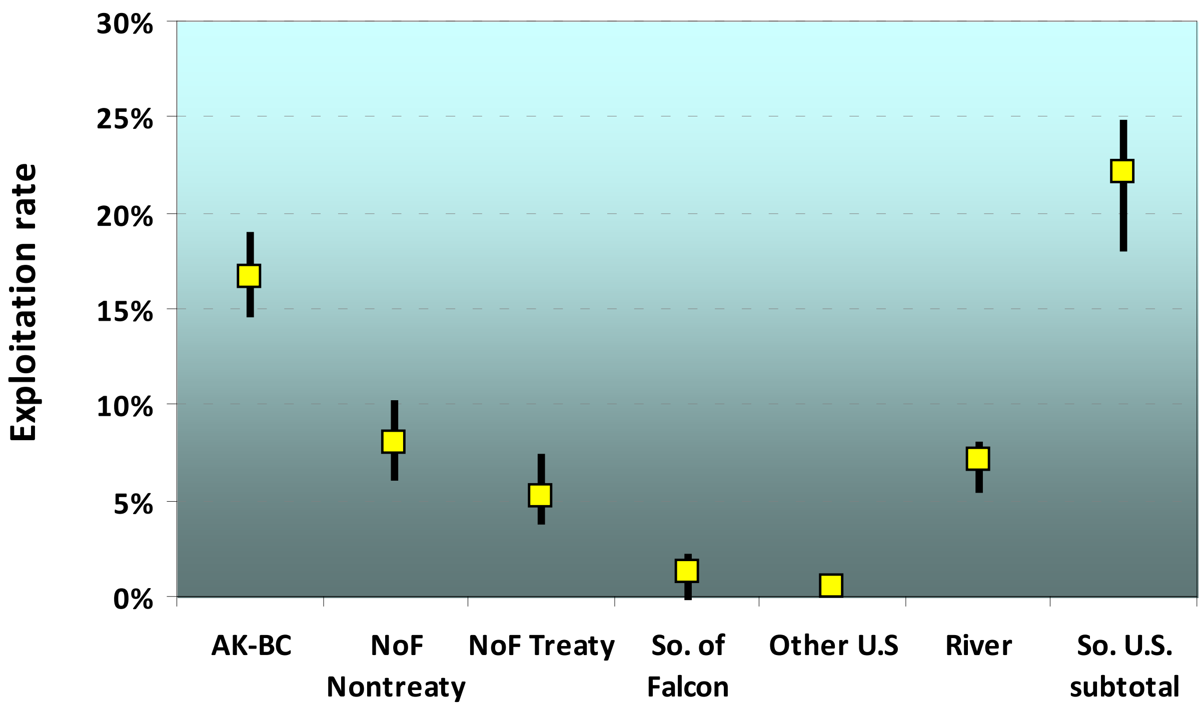

Figure 15. Range and average LCN tule ER for 2009-11 Preseason FRAM.

Effect on 2011 Chinook Harvest Example

Landed catch in Council fisheries north of Cape Falcon and in the Columbia River were estimated from FRAM using the 2011 preseason model run as a base for stock abundances and catch levels in Alaska, Canada and Council waters south of Cape Falcon. LCN tule exploitation rates modeled were 0.42 and 0.47, representing a high status level of +0.05 and +0.10 from the 0.37 ceiling in 2011. The low status exploitation rate modeled was -0.05 and -0.10 from the 0.37 base and the LRH river return was reduced to 40,000 fish to represent the abundance under a low status tier. Landed catch in fisheries north of Cape Falcon and harvest rates on LRH Chinook in the lower Columbia River were uniformly increased or decreased by the same level for modeling simplicity. The in-river harvest rate model was used to estimate the harvest rate on other fall Chinook stocks when river fisheries are shaped to achieve the target LRH harvest rate in the different scenarios. Of course for annual management, fishing levels and the seasons in ocean and river fisheries vary according to the abundances and circumstances that arise each year. The fishing levels in non-treaty and treaty ocean fisheries and between ocean and the river sharing are not a uniform multiplier of the previous year's rates. However, this simplistic approach does provide a way of comparing the effects of different ceiling exploitation rates on total landed catch of all stocks in ocean and river fisheries.

Table 13 contains landed catch estimates under high (0.42 and 0.47) and low (0.32 and 0.27) status exploitation rates for LCN tule Chinook. Under high status, total allowable catch (TAC) in the Council fisheries would increase by 38 percent under an LCN tule ceiling ER of 0.42 and by 75 percent for a ceiling of 0.47. In the river, catch of Chinook for "bright" stocks (Lower River Wild, Select Area Bright and Upriver Bright units) and tule stocks (LRH and BPH) would increase by a lower amount that reflects the varying change in stock specific harvest rates associated with the different river fisheries. Under low status of 40,000 LRH Chinook, the ocean TAC would decrease for the 0.37 ceiling as well as at the 0.32 and 0.27 levels. The TAC would be reduced by 15 percent for a 0.37 ceiling exploitation rate on LCN tule Chinook, 46 percent for 0.32 ceiling, and 76 percent for a 0.27 ceiling. Catch reductions in the river are not nearly as dramatic, although fishing seasons and opportunity are significantly restricted, especially at the 0.27 ceiling.

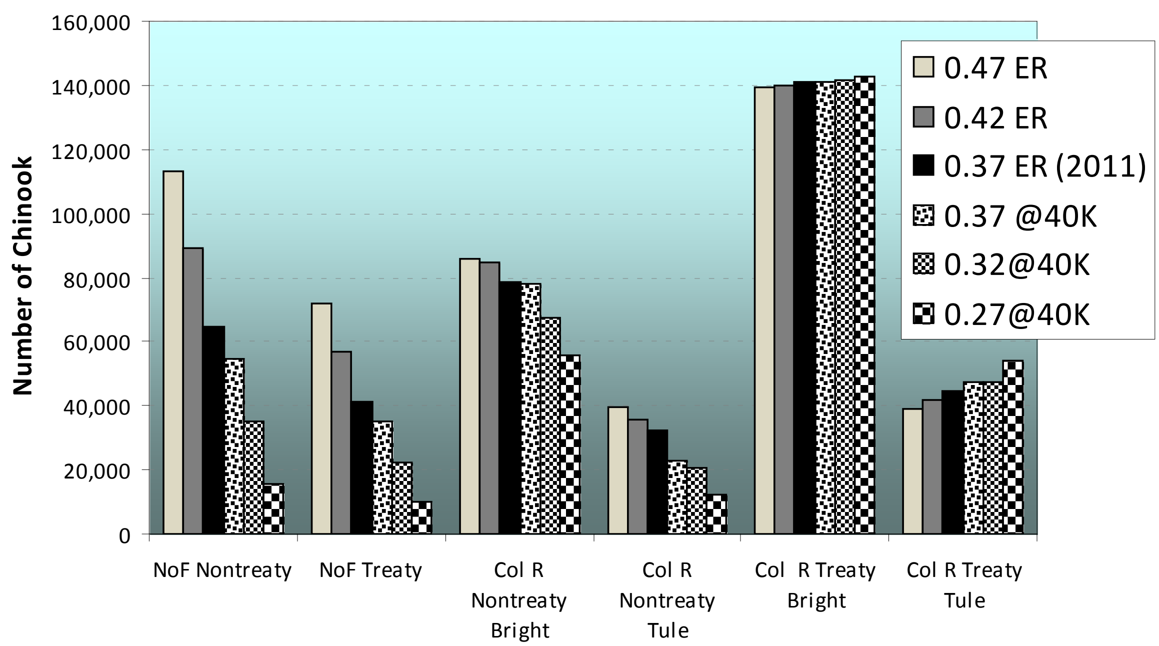

Figure 16. Landed Catch of Chinook in North of Falcon and River Fisheries under variable ERs for LCN Tule Chinook (2011 with LRH abundance at preseason of 133,500 or low abundance of 40,000).

Table 13. Landed catch of Chinook in Council waters North of Cape Falcon and in the Columbia River under different ceiling exploitation rates (ER) for LCR tule Chinook.

LCN Tule Total ER	0.37	LCN Tule Total ER	0.42	LCN Tule Total ER	0.47
2011 Preseason		@1.38X for 42%		@1.75X for 47%	
Council NoF TAC All Stocks		Council NoF TAC All Stocks		Council NoF TAC All Stocks	
Nontreaty	64,600	Nontreaty	89,100	Nontreaty	113,000
Treaty	41,000	Treaty	56,600	Treaty	71,800
Columbia River		Columbia River		Columbia River	
Nontreaty		Nontreaty		Nontreaty	
Bright	78,600	Bright	84,900	Bright	86,000
Tule	32,100	Tule	35,900	Tule	39,700
Treaty		Treaty		Treaty	
Bright	141,000	Bright	140,000	Bright	139,500
Tule	44,400	Tule	41,700	Tule	39,000
LCN ER		LCN ER		LCN ER	
NoF ocean	0.101	NoF ocean	0.137	NoF ocean	0.173
Columbia Ri	0.056	Columbia Ri	0.074	Columbia Ri	0.090

LCN Tule Total ER	0.37	LCN Tule Total ER	0.32	LCN Tule Total ER	0.27
@0.85X for 37% w LRH 40K		@0.54X for 32% w LRH 40K		@0.24X for 27% w LRH40K	
Council NoF TAC All Stocks		Council NoF TAC All Stocks		Council NoF TAC All Stocks	
Nontreaty	54,900	Nontreaty	34,900	Nontreaty	15,500
Treaty	34,900	Treaty	22,100	Treaty	9,800
Columbia River		Columbia River		Columbia River	
Nontreaty		Nontreaty		Nontreaty	
Bright	78,000	Bright	67,500	Bright	55,800
Tule	22,700	Tule	20,500	Tule	12,200
Treaty		Treaty		Treaty	
Bright	140,800	Bright	141,800	Bright	142,700
Tule	44,400	Tule	47,600	Tule	53,900
LCN ER		LCN ER		LCN ER	
NoF ocean	0.108	NoF ocean	0.070	NoF ocean	0.032
Columbia Ri	0.047	Columbia Ri	0.031	Columbia Ri	0.015

Note: @N.NNX represent the all-stocks harvest increase over 2011 preseason expectations.

In these examples, the allowable catch in the Council fisheries fall within the very broad range of annual management during 1991-2010 (Table 14). In most years, actual landed catch was less than 90 percent of the TAC for the non-Indian and treaty Indian fisheries. The magnitude of the TAC did not correlate very well with whether the TAC was achieved or not. A low TAC did not necessarily mean the TAC was achieved nor did a large TAC necessarily mean that there was significant number of fish remaining on the quotas at the end of the season. The 2011 example shows that both high and low status tiers may provide TACs that have been used in the past. The tiers need to be evaluated in terms of the ability of the fisheries to take advantage of the high status ceiling exploitation rates and the restrictions required in the fisheries at the low status in addition to the potential conservation benefits that may accrue to LCN tule Chinook. Also, because the annual abundances are highly correlated between years, consecutive years of low status could occur creating several years of very restrictive fisheries.

Table 14. Landed catch and total allowable catch (TAC) in Council waters North of Cape Falcon, Oregon.

	May-Sep Landed Chinook Catch				Allowable Catch (TAC)		Actual % of TAC	
Year	NT Troll	Sport	Total	Treaty Troll	Nontreaty	Treaty	Nontreaty	Treaty
1991	29,800	13,700	43,400	21,900	80,000	33,000	54%	66%
1992	45,900	18,700	64,700	23,100	80,000	33,000	81%	70%
1993	30,500	13,900	44,300	25,000	60,000	33,000	74%	76%
1994	0	0	0	4,600	0	16,400	--	28%
1995	0	600	600	9,800	0	12,000	--	82%
1996	0	400	200	12,300	0	11,000	--	112%
1997	6,500	4,200	10,600	14,200	23,000	15,000	46%	95%
1998	6,000	2,300	8,200	14,700	10,000	15,000	82%	98%
1999	18,600	10,800	29,400	27,500	50,000	30,000	59%	92%
2000	13,000	9,200	22,200	7,600	25,000	25,000	89%	30%
2001	26,500	25,600	52,000	28,800	60,000	37,000	87%	78%
2002	81,600	60,600	142,100	39,800	142,883	60,000	99%	66%
2003	69,800	36,500	106,200	35,200	124,000	60,000	86%	59%
2004	47,000	27,100	74,100	49,700	89,000	49,000	83%	101%
2005	45,200	40,000	85,100	42,000	86,500	48,000	98%	88%
2006	27,300	11,200	38,300	30,500	65,000	42,200	59%	72%
2007	15,800	9,500	25,200	22,900	32,500	35,000	78%	65%
2008	14,100	15,500	29,500	20,900	40,000	37,500	74%	56%
2009	13,100	13,300	26,300	12,400	41,000	39,000	64%	32%
2010	56,200	38,700	94,900	33,400	102,350	55,000	93%	61%
2011	--	--	--	--	64,600	41,000	--	--

NATURAL POPULATION RISK ANALYSIS

Background

Current fishery limitations for ESA-listed salmon species, including LRN tule Fall Chinook, are intended to avoid jeopardizing the continued existence of the species. NMFS' approach to making determinations regarding the effects of harvest actions involves analysis of effects of a proposed action on abundance, productivity, or distribution of the species (NMFS 2009). Determinations are ultimately based on whether the proposed action, taken together with any cumulative effects and added to the environmental baseline, can be expected, directly or indirectly, to reduce appreciably the likelihood of both survival and recovery of the affected species.

Biological risk assessments for listed salmon species have widely taken the form of a Population Viability Analysis (PVA). PVAs use quantitative methods to predict the likely future status of a population or collection of populations of conservation concern (Morris and Doak 2002; Beissinger and McCullough 2002). Salmon are believed to go extinct when population abundance and resilience are reduced to low levels where numbers "bottom out" under periods of low survival associated with variable environmental conditions. Current salmon recovery plans for listed Columbia River salmon define status in terms of risk which is estimated as the probability that a population will be above some minimum size over a prescribed period of time. Salmon PVA's typically utilize stochastic stock-recruitment models to estimate species survival and recovery likelihoods from population abundance, productivity and spatial structure, and population variability. PVA models have been developed and applied by NMFS, ODFW and WDFW to status assessments and recovery plan analyses for Columbia River tule Fall Chinook (LCFRB 2010; ODFW 2010).

The traditional approach to fishery effects analysis involved simple comparison of escapement numbers relative to goals. Fishery risk analyses consider the combined effects of fishing, fishery uncertainty, and variable production and survival on escapement levels that may threaten the long-term persistence or viability of a population or group of populations. PVA models are particularly well-suited for fishery risk assessments because effects of exploitation rates on demographic risk can be directly quantified. This approach can also effectively evaluate fishing effects on populations of different productivity including weak populations that are most at risk of falling to critical low levels where they are no longer capable of sustaining themselves.

This assessment adapted and applied the PVA framework from the lower Columbia salmon recovery plans to evaluate risks associated with alternative fishing strategies for lower Columbia River tule Fall Chinook. Adaptation of an existing model will ensure that results are consistent with salmon conservation needs driving current salmon management and associated consultations. Similar modeling approaches have previously been utilized by the Council in conservation risk analyses for other stocks including KRFC.

Model Description

Viability risks associated with different implementation strategies were estimated using the PopCycle model. PopCycle is a simple stochastic stock-recruitment model developed for the analysis of population viability of Washington lower Columbia salmon and steelhead populations addressed by the Recovery Plan (LCFRB 2010). This analytical framework is consistent with the approach used in Oregon's CATAS and NOAA's SLAM models. Each of these models are stochastic life cycle models built around the salmon stock-recruitment function and both models can be expected to produce relatively similar results if parameterized with equivalent inputs. Models differ in the detail by which stages of the salmon life cycle are represented. Both the PopCycle model employed by Washington and the SLAM model employed by NMFS have been utilized to evaluate effects of fixed exploitation rates, including evaluating scenarios incorporating potential impacts of habitat and hatchery recovery actions. Additional analysis using multiple models may be contemplated in the future depending on resource availability.

The model used in the analyses described in this report estimates annual run size, harvest and spawner numbers over a prescribed number of years (Figure 17). The model estimates average and frequencies of values over a prescribed number of iterations (typically 1,000). Model populations include the aggregate Lower Columbia River tule stock which consists primarily of hatchery (LRH) fish but also includes a small proportion of naturally-produced fish. The model simultaneously simulates a wild tule population. This wild population can be parameterized to represent a specific population (e.g. Coweeman) or a generic population representative of low, medium, or high viability. However, the simulated wild population number is not included in the aggregate stock numbers. The aggregate number is thus effectively defined in terms of hatchery fish alone (LRH) consistent with current information which indicates that even a significant portion of the natural production is driven by stray hatchery fish spawning in the wild. The total wild/natural aggregate (LCN) consisting of multiple populations was not simulated by the model because individual wild populations behave differently due to difference in size and productivity, and because we lack population-specific information. The model thus simulates representative wild populations subjected to the same and fishery conditions as the aggregate LRH stock.

Number of LRH adults recruiting to ocean fisheries is estimated based on hatchery releases and juvenile to adult survival rates. Number of wild adults in the representative wild population is estimated from recruitment generated by a stock-recruitment function from the brood year number of spawners for that population. This analysis models three representative wild population types describing a productivity and abundance range believed to be representative of the current status of most LRN populations. Recruits of LRH and of the wild population are estimated as an ocean adult cohort. Annual numbers of fish from this cohort are apportioned among years based on an input age schedule. The annual run is subjected to fishing with the surviving wild population spawning to seed the next wild generation and the hatchery adults dead-ending into the hatchery. The model does not simulate straying of hatchery fish into the wild population. Wild population parameters are thus assumed to represent an equilibrium contribution of hatchery fish and any changes in hatchery contributions due to changes in fishery strategy are not captured. While it is computationally simple to simulate hatchery strays, assumptions regarding their effects on population productivity over time would be highly subjective.

Random annual variability is introduced into the model at the juvenile-to-adult survival stage for the LRH population and in the stock-recruitment relationship for the representative wild population. Variances are proportional to survival or productivity, log-normally distributed, annually autocorrelated, and partially correlated between hatchery and wild fish. Log-normal distributions provide for the occasional very high survival or productivity years that we see periodically. Autocorrelation means that poor survival or production years are generally more likely to be followed by poor years, and good years by good years.

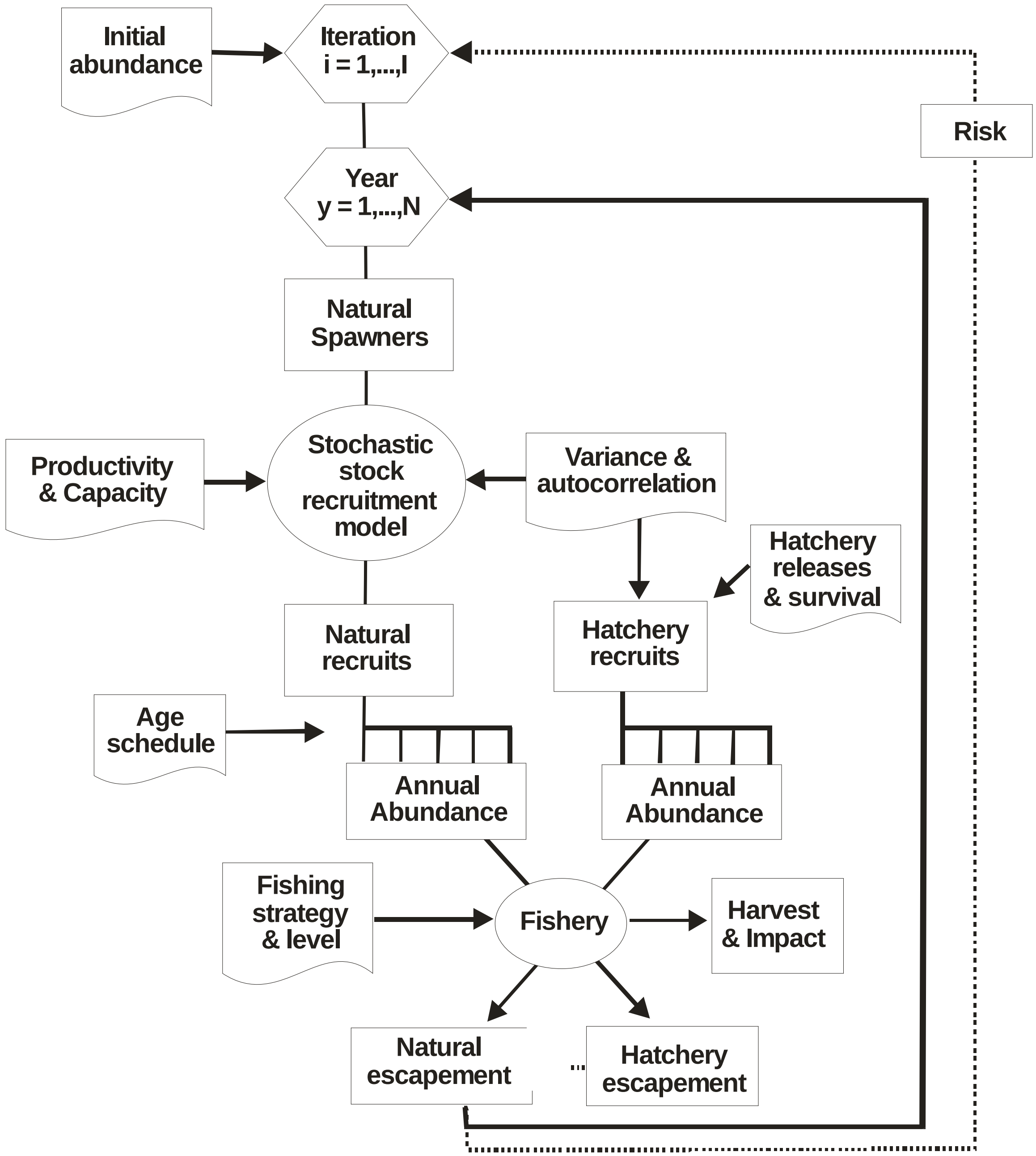

Figure 17. Conceptual depiction of model algorithm.

The model includes optional inputs to apply fishing rates in each year to calculate harvest and fishery effects on population dynamics. Either fixed or abundance-based rates may be utilized. Abundance-based rates are applied according to forecast abundance tiers (e.g. <40,000, 40,000 to 100,000, > 100,000). Input parameters allow for forecast errors which introduce uncertainty and variability into model estimates, notably including errors in predicting which fishing rate tier should be operated in. Inputs also allow for normal differences in target and actual fishing rates which result from a variety of factors mostly related to lack of predictability in stock composition, fishery catch rates, etc.

Viability risk was defined in this analysis as the probability of average abundance of a generation of salmon falling below a critical abundance threshold over the course of a simulation. A quasi-extinction risk threshold (QET) was defined as a population size where functional extinction occurs due to the effects of small population processes (McElhany et al. 2006). The model assumes that extinction occurs if the average annual population size over a moving generational average falls below this threshold at any point in a modeled trajectory. Extinction risk is thus estimated as the proportion of all iterations where the moving generational average spawner number falls below the threshold at any point in each simulation period.

The model is built in Microsoft Excel using Visual Basic. A simple interface page facilitates model use and review of results.

Model Parameters

Table 15. Model input variables and parameters used for fishery risk analysis.

Variable or parameter	Notation	Value
Initial spawner abundance	$S_{y-6},\ldots,S_{y-1}$	Equilibrium abundance @ avg. fishing rate
Stock-recruitment		
Function	Option 2	Beverton-Holt
Productivity	p	Pop A = 3.0; Pop B = 2.0; Pop C = 1.5
Equilibrium abundance	N_{eq}	Pop A = 2,000; Pop B = 1,000; Pop C = 300
Maximum spawner constraint	lim S_y	(10) (N_{eq})
Maximum recruit constraint	lim R_y	(10) (N_{eq})
Production trend	PT	0%
Recruitment failure threshold	RFT	50
Critical risk threshold	CRT	50 (avg. per generation)
Recruitment stochasticity		
Variance	σ^2	0.5
Autocorrelation	Ø	0.5
Age schedule	$m_2,\ldots,m_7$	Age 2 = 0.055; Age 3 = 0.373; Age 4 = 0.495; Age 5 = 0.076
Hatchery fish		
Annual releases	HR	22,000,000
Smolt-to-adult-survival (to ocean)	SAR	0.0031 (0.0020 geomean to Col. River @ a 30% ocean ER)
Wild population correlation	r_w	0.5
Run size forecast error (CV)	E_f	0.75
Fishery implementation error (CV)	E_i	0.10

Wild Populations

Rather than modeling specific wild populations, this assessment identified three general categories of populations and modeled representative abundance and productivity parameters for each category. Categories generally correspond to moderate, low, and very low levels of population viability identified in lower Columbia River salmon recovery plans.

This approach was taken because population-specific data was generally inadequate for confident estimation of model parameters. However, general information was adequate to identify a reasonably representative range of parameters for LCR population based on values reported in Washington and Oregon recovery plans.

Table 16. Representative population parameters.

Category	Abundance	Productivity	Viability response	Examples
A	2,000	3.0	Moderate	Coweeman[a], Washougal[a], Cowlitz,
B	1,000	2.0	Low	EF Lewis[a], Mill/Abernathy/Germany[a], Elochoman/Skamokawa[a], Kalama, Toutle[a]
C	300	1.5	Very low	Clatskanie[ab], Scappoose[ab], Hood[ab], Grays, Sandy, Clackamas

[a] *"Primary" populations targeted for restoration to high or very high levels of viability in lower Columbia River salmon recovery plans.*

[b] *Denotes high degree of uncertainty in current population status.*

Example populations were identified for each population category based on viability estimates and parameters reported in salmon recovery plans and analyses by Washington, Oregon, and NMFS. The binning of populations represented above is based on information from those efforts, not the specific conclusions. The best available data includes a mixture of population-specific parameter estimates based on reconstructions of spawning ground survey information and inference from habitat amounts and conditions.

While the range of population categories are reasonably representative of the status of most LRN tule Fall Chinook, different analyses and assumptions might result in specific populations being classified in different categories (

Table 17). These differences reflect both different approaches and assumptions utilized in the available analyses, and uncertainty in population parameters due to data limitations. NMFS described groups of populations in three categories based on SLAM modeling results and assumptions regarding current production. The NMFS analysis was limited to "primary" populations identified in recovery plans for improvement to high or very high levels of viability. The SLAM model did not explicitly model adult-to-adult stock-recruitment equation parameters – rather, a functional relationship was implicit in life stage-specific parameters and functions used in that mode. The Washington Recovery Plan analysis did report adult to adult stock-recruitment parameters but included only Washington populations. The Oregon Recovery Plan analysis also utilized an adult-to-adult stock recruitment analysis but did not report parameters comparable to the Washington plan. However, all three modeling efforts reported model-derived abundance levels under roughly approximate conditions.

Categorization of example populations reflects current conditions including habitat quantity and quality, and hatchery influences. The category of any given population can change in the future as habitat, hatchery, or hydropower-related actions improve productivity and numbers. Thus, populations that may currently be relatively unresponsive to exploitation rate changes may become more responsive and benefit from harvest limitations at some point in time.

It should also be noted that there are additional populations of extremely low viability that currently appear to be consistently below replacement levels – limitations on these populations are predominately driven by factors other than fisheries and were not modeled. Examples include Big Creek, Youngs Bay tributaries, Salmon Creek, White Salmon River, Lower Gorge, and Upper Gorge populations.

Table 17. Population parameters for lower Columbia River natural tule fall Chinook populations based on analyses and population viability modeling conducted by Washington, Oregon, and NMFS for the purposes of salmon Recovery Plans. Populations are sorted by maximum modeled abundance reported for any of the plans.

		WA plan parameters[a]		Modeled abundance		
Population	**State**	N_{eq}	**R/S**	**WA**[b]	**OR**[c]	**NMFS**[d]
Lower Cowlitz	WA	8,200	3.0	4,260	--	--
Washougal	WA	1,100	1.9	310	--	1,700
Lewis	WA	800	1.7	100	--	1,700
Coweeman	WA	1,700	3.2	920	--	1,400
Mill/Abernathy/Germany	WA	1,000	2.2	360	--	700
Clackamas	OR	--	--	--	558	--
Hood	OR	--	--	--	33	400
Elochoman/Skamakowa	WA	1,300	1.9	390	--	200
Toutle	WA	2,400	1.6	380	--	--
Youngs Bay	OR	--	--	--	379	--
Scappoose	OR	--	--	--	356	100
Kalama	WA	1,000	2.0	280	--	--
Big Creek	OR	--	--	--	216	--
Sandy	OR	--	--	--	144	--
Clatskanie	OR	--	--	--	6	100
Grays/Chinook	WA	300	1.9	<50	--	
L. Gorge	WA/OR	500	--	<50	--	--
U. Gorge	WA/OR	500	--	<50	--	--
White Salmon	WA	--	--	<50	--	--
Upper Cowlitz	WA	--	--	--	--	--
Salmon	WA	--	--	--	--	--

a *Beverton-Holt stock-recruitment parameters from the Washington Recovery Plan (LCFRB 2010) under habitat conditions and hatchery impacts in the listing period baseline (late 1990s). Values reflect pre-harvest equilibrium inferred from habitat conditions with the Ecosystem Diagnosis and Treatment model with productivity reduced by the Hatchery Scientific Review Group hatchery impact.*

b *Modeled population abundance using the Washington recovery plan model at a fixed exploitation rate of 0.37 (unpublished data).*

c *Modeled abundance for Oregon populations from the Oregon Recovery Plan (ODFW 2010) represent the average of 100-year forward projections that assume environmental conditions effecting survival are similar to those from 1974 to 2004; as such is not comparable to observed wild spawner counts in more recent times when the natural survival rates have been lower and fishery impacts generally less than in the early 2000s.*

d *Modeled abundance in NMFS' SLAM model at an ER of approximately 0.37 under current habitat and hatchery conditions, assuming hatchery fish depress natural survival (NMFS Scenario 2).*

Hatchery Populations

LRH abundance was estimated based on annual hatchery releases of lower Columbia River programs which have averaged 22 million juveniles per year from 1998 through 2008. This production level reflects program changes in the mid-1990s to reduce production and selectively eliminate programs with lower success rates. This production level does not reflect any future changes that may be implemented based on conservation and recovery plans for wild populations.

Average annual survival of hatchery fish to the Columbia River mouth was estimated from brood year run reconstructions at 0.0028 for the 1987-2006 brood years. This was less similar to the long-term (1962-2006 run year) average of 0.0045 (Figure 5). Average survival to ocean recruitment was estimated from Columbia River mouth run size estimates expanded for average ocean exploitation rates [0.0028/(1-0.35) =0.0043]. These survival estimates are slight overestimates of actual hatchery fish survival because they are based on total Lower Columbia River tule returns, which include a small percentage of wild fish. However, this approach is consistent with the definition of the LRH population in the model and produces a total LCR tule return similar in number to the actual number. (This is critical for application of the fishery rules where the tiers are based on total adult run size.)

Age Composition

Age composition of both LRH and the wild population was based on 1962-2005 brood year data for adults only (Figure 18). Average percentages were similar based brood year (age 2 = 5.5, age 3 = 37.3 percent, age 4 = 49.5 percent, age 5 = 7.6 percent) and run year (age 2 = 6.0 percent, age 3 = 37.2 percent, age 4 = 48.5 percent, age 5 = 8.2 percent) analyses over the long time frame of this data.

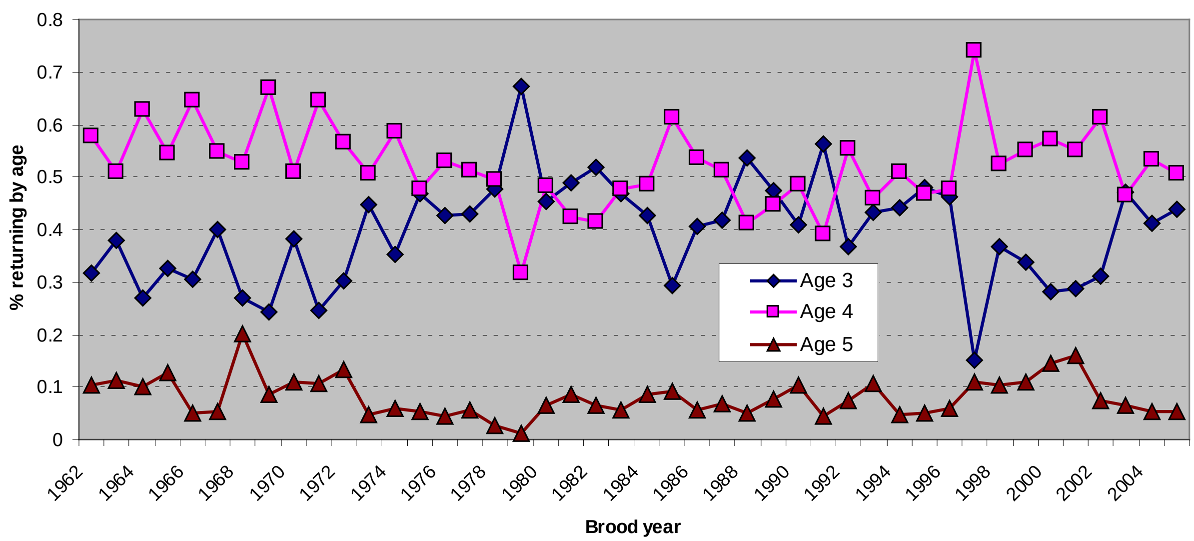

Figure 18. Age composition of adult LRH return to the Columbia River by brood year, 1964-2010.

Variation in Survival & Recruitment

Annual variability in natural production of the wild population is incorporated in the stock-recruitment relationship. The same relative variance was applied to annual survival of hatchery fish under the assumption of common effects of ocean conditions on wild and hatchery fish in the same cohort. The variance in recruits per spawner was parameterized with a variance of 0.5. This parameter produced an average hatchery survival rate to the Columbia River in the model equivalent to the recent 20-year average (0.00276) and CV (0.97). Note that this is less than the 0.9 value used in population viability analyses for all wild LCR fall Chinook population in the Washington Recovery Plan (LCFRB 2010) based on recommendations by NMFS' Technical Recovery Team (TRT) following review of all available population data.

Variance was assumed to be auto-correlated with a coefficient of 0.50. This value was used in the Washington Recovery Plan PVA as recommended by the TRT. The autocorrelation coefficient was independently estimated at 0.50 for the hatchery survival rate index from the long-term dataset.

Finally, wild and hatchery population variability was assumed to be only partially correlated from year to year based on correlation analyses summarized in Table 8, Table 9, and Figure 11. A correlation coefficient of 0.5 was applied to all wild populations.

Forecast & Fishery Errors

Forecast and fishery errors were based on data reported earlier in this report. Forecast error was estimated to have a CV of 0.75. Fishery implementation error was estimated to have a CV of 0.1.

Conservation risks

Wild population risks were based on a QET of 50 estimated as a moving average of years in one generation of the species in question (4 years for Chinook) as per McElhany et al. (2006). Estimates of absolute risk are extremely sensitive to the selection of this parameter, which is why model-derived risks are most useful for relative comparisons among risk factors. While there is an extensive amount of literature on the relationships among extinction risk, persistence time, population abundance, and level of variation in demographic parameters, there are no simple generic abundance levels that can be identified as viable (McElhany et al. 2000). Because empirical data on actual extinction and conservation risk levels is lacking, this QET value was based on theoretical numbers identified in the literature based on genetic risks. Effective population sizes between 50 and 500 have been identified as levels which theoretically minimize risks of inbreeding depression and losses of genetic diversity, respectively (Franklin 1980, Soule 1980, Thompson 1991, Allendorf et al. 1997). Effective population size assumes balanced sex ratios and random mating. Relatively low QET values are supported by recent observations of salmon rebounds from very low numbers (e.g. Oregon lower Columbia River coho: ODFW 2005 and Washington lower Columbia winter steelhead: D. Rawding, WDFW, unpublished) and apparently-sustainable small population sizes of salmon in other regions (e.g. King Salmon River Chinook population in Alaska: McPherson et al. 2003).

Simulations

A series of model simulations were conducted to:

1. Evaluate the effects of exploitation rate on risk for wild populations in each abundance/productivity category.
2. Describe short versus long-term risks associated with exploitation rates.
3. Explore the effect of abundance tier selection on population risks.
4. Identify the risk reduction and fishery opportunity benefits of a variety of fixed and abundance-based fishery scenarios.
5. Evaluate the sensitivity of model results to key input parameters.

Population sensitivity to exploitation rates was evaluated based on simulations of A, B, and C population types to a series of fixed annual ERs ranging from 0.0 to 0.70.

Effects of the simulation duration on risk were used to identify an appropriate time period for analysis of fishery effects. Recovery plan risk assessments involving all threat categories (fisheries, habitat, hydropower, hatcheries, ecological factors, etc.) were typically based on 100-year simulations. However, fishery plans addressed by this analysis are intended primarily for use in an interim period until the longer-term benefits of other recovery measures begin to be realized. Thus, 100-year simulations would overestimate the risk when other improvements are not considered. Conversely, very short-term simulations may not accurately describe risks related to harvest because they do not allow for the compounding effects high harvest rates on unproductive populations over time, particularly under a series of temporally auto-correlated poor ocean survival years. Therefore, evaluations of abundance-based fishery scenarios considered both 100-year and 20-year simulations.

Abundance tiers refer to run size forecast trigger points identifying the appropriate fishing rate for use in any particular year. For instance, the TCW identified a 3-tier variable rate strategy involving the LRH aggregate stock operating approximately 25 percent of time in the high tier, 25 percent of time in the low tier, and 50 percent of the time in the middle tier. This is a "balanced" tier structure with equal frequencies in the lower and higher tiers. The model was used to identify corresponding forecast levels that provide the desired tier frequency. Effects of other tier frequencies were also evaluated for a three-tier scenario. Five-tier examples were also considered.

Based on a review of abundance-based approaches for other fisheries and an assessment of information available for lower Columbia River tule Fall Chinook, the TCW initially identified a series of alternative scenarios for further evaluation. Alternatives included a variety of fixed ER strategies ranging from 0.0 to 0.53. Variable rate strategies were evaluated for different base fishing rates. Alternatives also included a variety of more specific combinations of higher or lower variable fishing rates.

Risks were compared among scenarios based on the Population B category. In a 100-year simulation, the B populations provide the most sensitive index of fishery effect on risk. The B populations also represent populations which are a primary concern of the recovery strategies.

Changes in risks and LRH harvest levels were compared relative to a fixed 0.37 ER, which represents the 2011 ESA consultation standard. Fishing rate scenarios were categorized based on whether risk and harvest levels were substantially greater or lower than corresponding values at the 0.37 ER. For the purposes of this analysis, changes in risks and harvest levels were classified as substantial when they exceeded the difference observed for a ±0.01 change in fixed harvest rate from 0.37 values. Corresponding values were ±3.5 percent change in 100-year risk, ±0.25 percent change in 20-year risk, and ±3.0 percent change in average 100-year harvest.

These numbers were used to classify fishing rate scenarios into one of four categories:

The Win/Win group involved both a substantial reduction in risk to the natural population and an improvement in fishing opportunity for tule Fall Chinook. This group would represent the ideal abundance-based strategy.

The Risk Reduction group involves only a substantial decrease in wild population risk with little or no fishery benefit. This group includes scenarios that greatly reduce fishing opportunity.

The Fishery Opportunity group involves only a substantial increase in harvest opportunity relative to the fixed 0.37 ER standard with either no substantial risk reduction or increased risk. This group includes some scenarios that increase natural population risks.

The Equivalent group provides the same or similar wild population risk and tule Fall Chinook harvest level as the fixed 0.37 ER strategy. Equivalent scenarios include those where some change might occur but the magnitude falls short of the above definition of substantial.

No fishery scenarios were contemplated that increase natural population risks and reduce fishery opportunities.

Results

Population Sensitivity to Exploitation Rates

The sensitivity of long-term risks to fishery impacts varies with population status. Long-term population risks can be substantially reduced by reducing fishery impacts only for populations with significant intrinsic capacity or productivity (e.g. category B populations). Smaller, less productive populations are less affected and cannot be brought to high levels of viability over the long term even at very low fishing rates (e.g. category C populations).

Incremental benefits of fishery reductions progressively decrease at lower and lower fishing rates. Fishing rates below which population viability is largely independent of the effects of fishing are sometimes referred to as *de minimis* fishing rates. Definition of an appropriate *de minimis* rate depends of the specification of an acceptable risk level. Rates may vary among populations in relation to differences in abundance and productivity.

Average abundance of a natural population increases in direct proportion to the decrease in fishing rate over the 100-year period of the simulation. Improvements are greatest in the most productive populations and least in relatively unproductive populations. While risk of falling below a critical small-population threshold may be relatively insensitive to fishing at low impact rates, abundance is consistently sensitive to fishing at all impact levels. Thus, while reductions to very low fishing rates do not substantially affect risk, they do translate into ever larger numbers of spawners.

Of course, harvest of LRH Chinook increases in direct proportion to increasing ER. However, under an assumption of a fixed northern (Alaska and Canada) exploitation rate of 0.18, the southern US share of the harvest depends on the total ER.

Table 18. Modeled effects of different exploitation rates on short term (20-year) and long term (100-year) risks falling below critical wild population abundance thresholds, median wild abundance by population, average total harvest of hatchery and wild tule fall Chinook.

Outcome	Population category	Exploitation rate							
		0	10	20	30	40	50	60	70
Risk (20 yr)	A	0.000	0.000	0.000	0.000	0.000	0.002	0.025	0.231
	B	0.000	0.000	0.000	0.004	0.024	0.134	0.431	0.891
	C	0.020	0.049	0.134	0.291	0.533	0.850	0.982	1.000
Risk (100 yr)	A	0.000	0.000	0.000	0.000	0.001	0.027	0.365	0.957
	B	0.000	0.001	0.007	0.059	0.344	0.832	0.997	1.000
	C	0.123	0.324	0.644	0.901	0.992	1.000	1.000	1.000
Wild number	A	2,200	1,880	1,560	1,240	920	600	240	<50
(100 year)	B	1,120	920	700	500	280	60	<50	<50
	C	340	230	120	<50	<50	<50	<50	<50
Total LRH Harvest	--	0	8,140	16,260	24,430	32,580	40,720	48,870	56,820
US Harvest		0	0	1,598	9,768	17,918	26,058	34,208	42,158

(US harvest south of Canada assumes first exploitation of 18% comes from north)

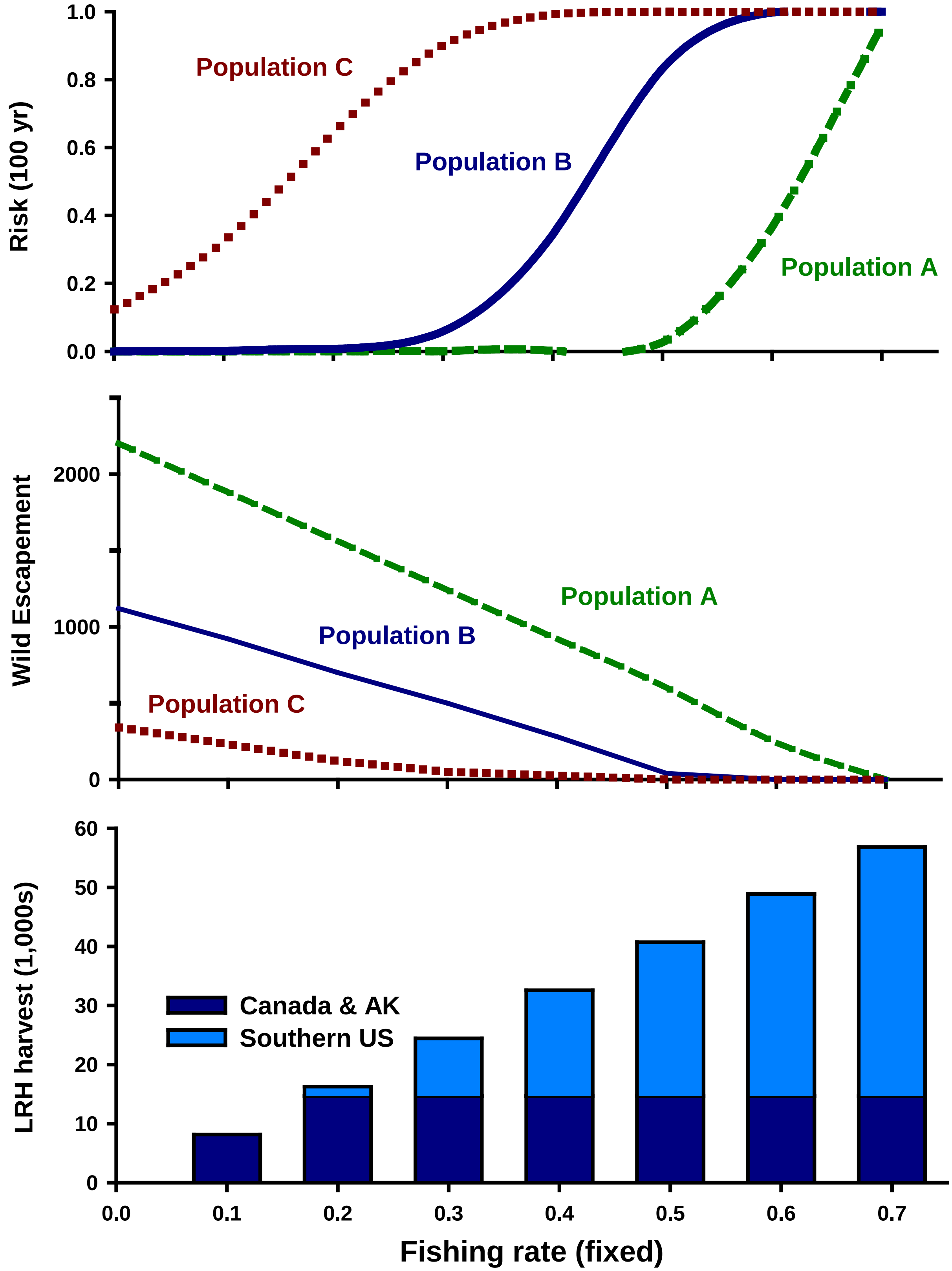

Figure 19. Modeled effects of different exploitation rates on long-term risk of falling below critical wild population abundance thresholds, median wild abundance by population, and average total harvest of hatchery and wild tule fall Chinook.

Short-term vs. long-term risk

Fishery risks to natural populations generally increase with simulation duration as compounding effects of low run sizes and chance occurrences of poor ocean survival years have a chance to accrue. Even relatively high fishing rates are extremely unlikely to drive numbers to low levels within a couple of fish generations starting at recent average numbers.[3] Absolute values of risk estimates vary considerably but relative values of risk are generally similar in short-term versus long-term calculations. Sensitivity varies with population category.

Shorter simulation periods have the effect of shifting population risk profiles to the right. As a result, population sensitivity to variable fishing rates around a mean rate depends greatly on the duration of the simulation. For instance, Population B is most sensitive to fishing rates around 40 percent in a 100-year simulation. In a 20-year simulation, the effects to the B populations are smaller – not as measurable, but still meaningful/relevant. In contrast, Population C is relatively insensitive to ERs around 0.40 in a 100-year simulation but highly sensitive to the same rates in a 20-year simulation.

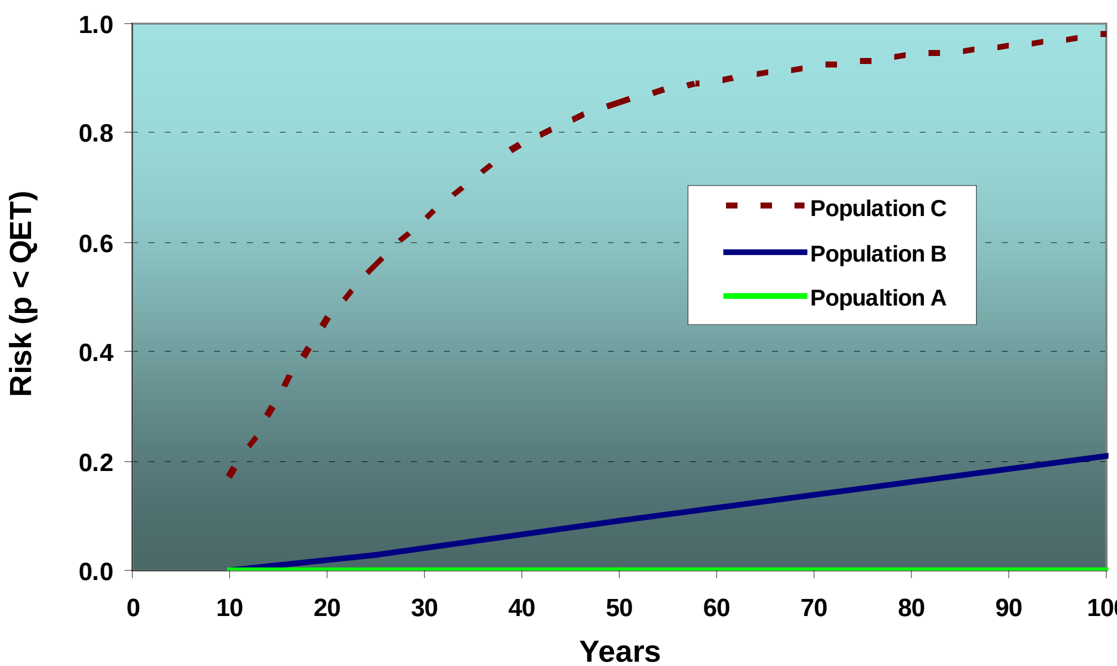

Figure 20. Effect of simulation years on natural population risk under a fixed 37% fishing rate scenario.

[3] Short-term risks are influenced by initial population values. Lower initial population abundances will increase near-term risks.

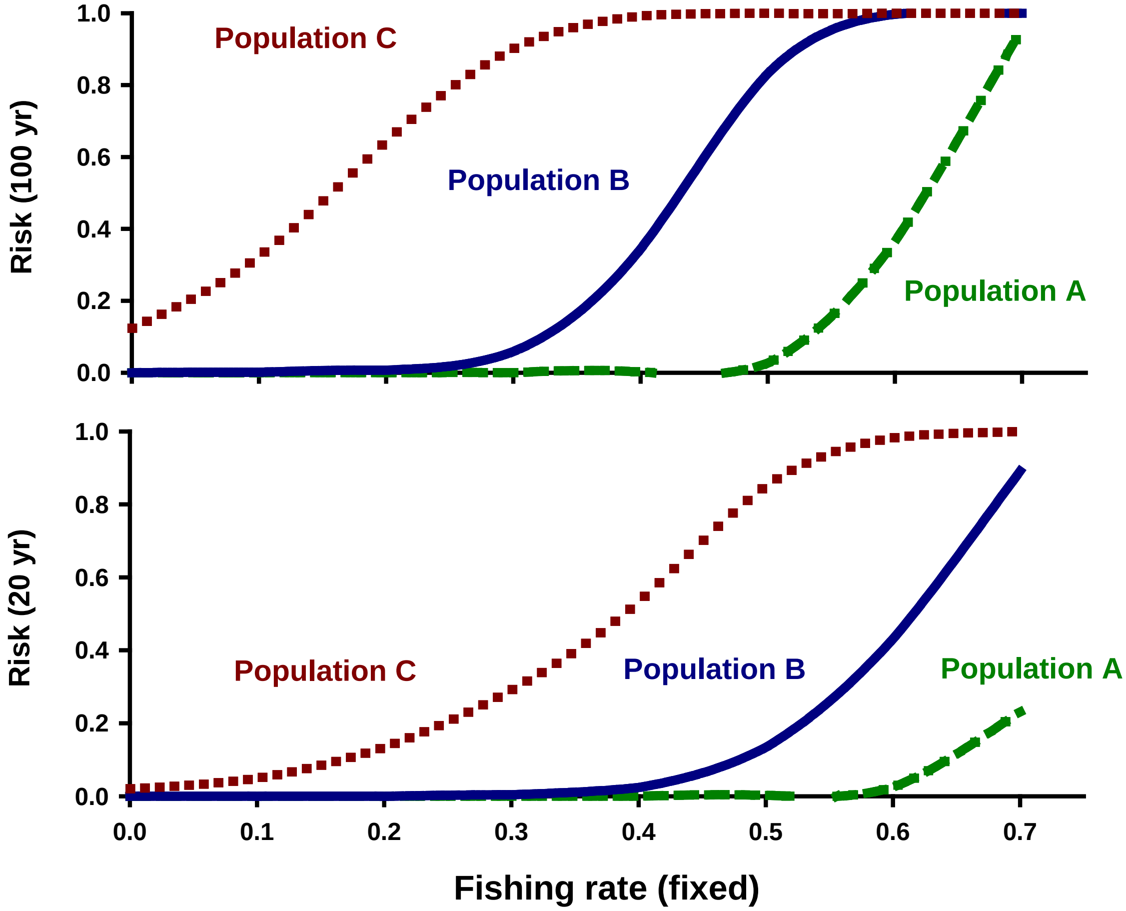

Figure 21. Modeled effects risks of different exploitation rates on long-term (100 year) and short-term (25 year) risk of falling below critical wild population abundance thresholds.

Effect of Fixed Harvest Rates

Average LRH harvest increases in direct proportion to increasing harvest rate (Figure 22). Risk increases in a curvilinear relationship to increasing harvest rate (Figure 23).

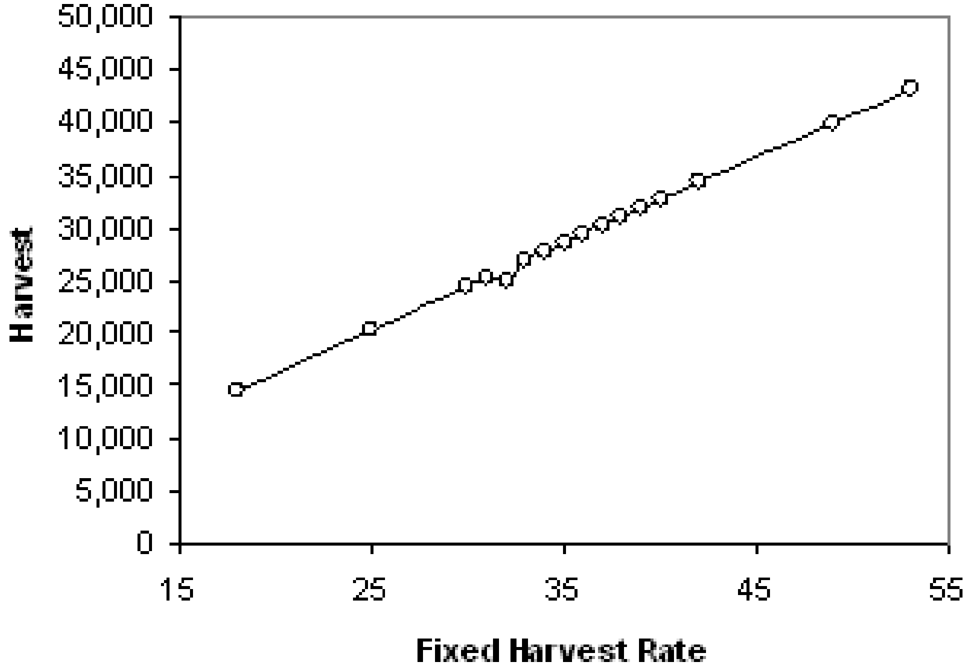

Figure 22. Simulated change in 100-year average harvest in response to fixed exploitation rate scenarios.

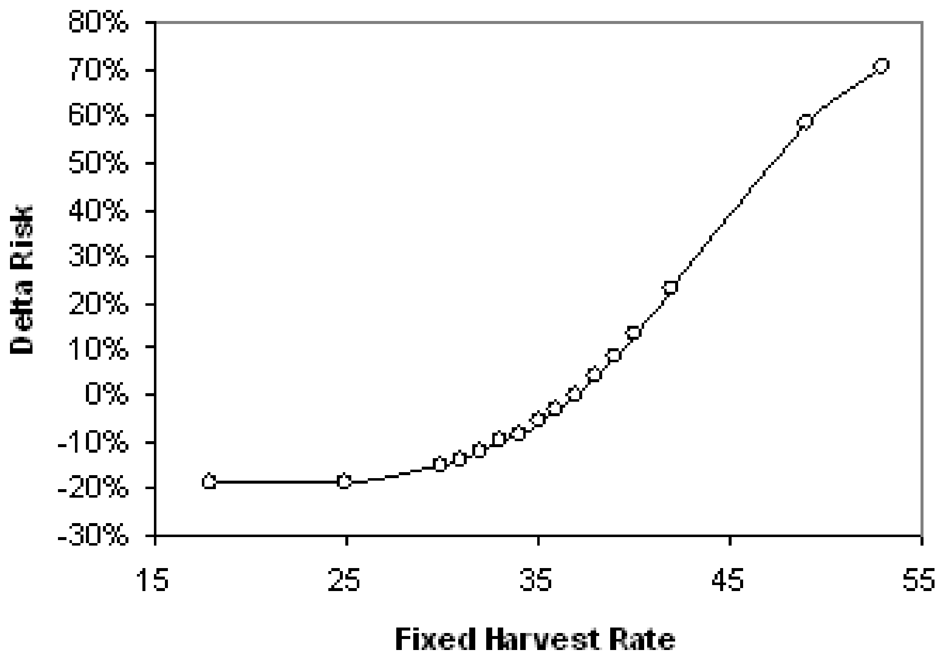

Figure 23. Simulated change in 100-year risk for category B populations in response to fixed exploitation rate scenarios.

Effect of Tier Selection

Frequencies of occurrence of predicted values projected by the model are depicted in Figure 24 and Table 19. Modeled frequencies closely correspond with frequencies observed during the last 20 years. Tier frequencies are of course affected by the ocean exploitation rate because forecasts are to the Columbia River mouth.

Figure 24 is helpful for identifying forecast tier levels consistent with a desired frequency of occurrence in model simulations. For instance, forecasts of less than 40,000 are modeled to occur approximately 22 percent of the time at an ocean exploitation rate of 0.30. Forecasts greater than 100,000 are modeled to occur approximately 23 percent of the time.

Wild population risks can be affected by the tier selection of forecast triggers, which determines the frequency with which a tier-specific rate is applied. Risks do not vary with tier frequencies for fixed-rate scenarios because the same rate is applied no mater what tier we are in. For abundance-based scenarios, risks generally increase as fewer years occur in tiers with lower fishing rates and more years occur with higher fishing rates. In general, the lower the forecast numbers used to define the tiers, the greater the wild population risk. However, tier values and abundance-based rates can be mixed and matched to provide a net benefit relative to any given fixed-rate strategy.

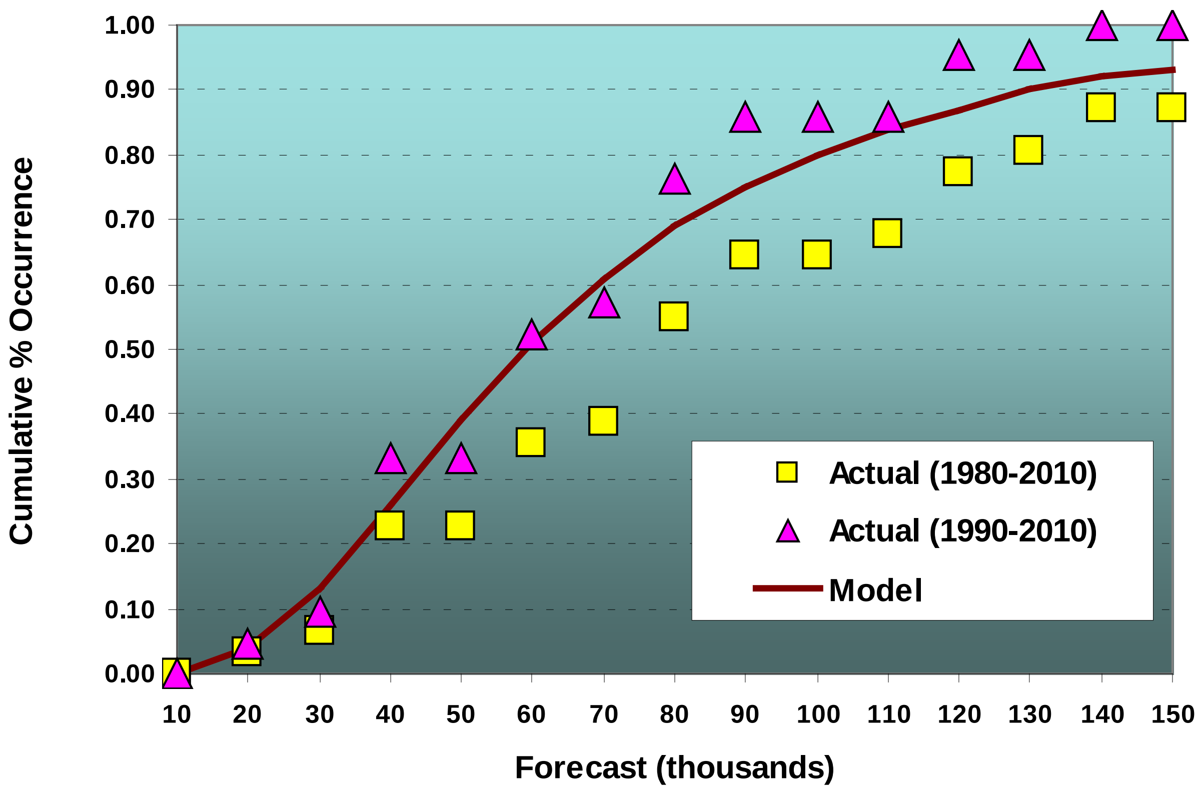

Figure 24. Cumulative frequency of occurrence of LRH run size to the Columbia River mouth based on model simulations.

Table 19. Observed and model frequencies of preseason forecasts and actual run sizes for LRH tule Chinook.

Tier	1980-2010		1990-2010		Model simulation[a]	
	Predicted	Actual	Predicted	Actual	ER = 49%	ER = 37%
<40,000	0.22	0.10	0.33	0.14	0.26	0.22
40,000-100,000	0.42	0.55	0.52	0.67	0.54	0.55
>100,000	0.36	0.35	0.14	0.19	0.20	0.23

[a] *Assuming ocean exploitation rates of 35% when 49% total and 30% when 37% total. The 49% example approximates the recent period. The 37% example is a reference for simulations of scenarios under current conditions.*

Tier frequencies in the future will depend on average and variability in LRH run size which in turn is affected by hatchery production, ocean survival patterns, and ocean exploitation rates. If parameters change significantly in the future, then the tier break points would change as well. The risk assessment thus presumes some base level of hatchery production – e.g., 22 million smolts. Ocean survival patterns are assumed to be similar to those occurring over the last 20 years, which represented an extended period of lower-than-average productivity for LRH Chinook. The relative benefits of different fishing alternatives could be affected by any future changes in conditions which affect tier frequencies.

Table 20. Effects of different tier frequencies on risks and harvest levels associated with fixed and abundance-based fishery scenarios (Population B, 100-year simulation).

Scenario	Frequency	Risk	Harvest	Δ Risk	Δ Harvest	Category
F37	22/55/23	0.210	30,130	0%	0%	Equivalent
V37±5	22/55/23	0.163	31,540	-5%	5%	Win/Win
V37±10	22/55/23	0.117	32,950	-9%	9%	Win/Win
V37-5	22/55/23	0.147	29,770	-6%	-1%	Risk reduction
V37-10	22/55/23	0.100	29,400	-11%	-2%	Risk reduction
F37	46/31/23	0.210	30,130	0%	0%	Equivalent
V37±5	46/31/23	0.135	30,890	-8%	3%	Win/Win
V37±10	46/31/23	0.065	31,650	-15%	5%	Win/Win
V37-5	46/31/23	0.121	29,120	-9%	-3%	Risk reduction
V37-10	46/31/23	0.056	28,100	-15%	-7%	Risk reduction
F37	22/34/44	0.210	30,130	0%	0%	Equivalent
V37±5	22/34/44	0.175	32,479	-4%	8%	Win/Win
V37±10	22/34/44	0.163	34,810	-5%	16%	Win/Win
V37-5	22/34/44	0.147	29,770	-6%	-1%	Risk reduction
V37-10	22/34/44	0.100	29,400	-11%	-2%	Risk reduction

[a] *tiers: <40,000; 40,000-100,000; >100,000*
[b] *tiers: <60,000; 60,000-100,000; >100,000*
[c] *tiers: <40,000; 40,000-70,000; >70,000*

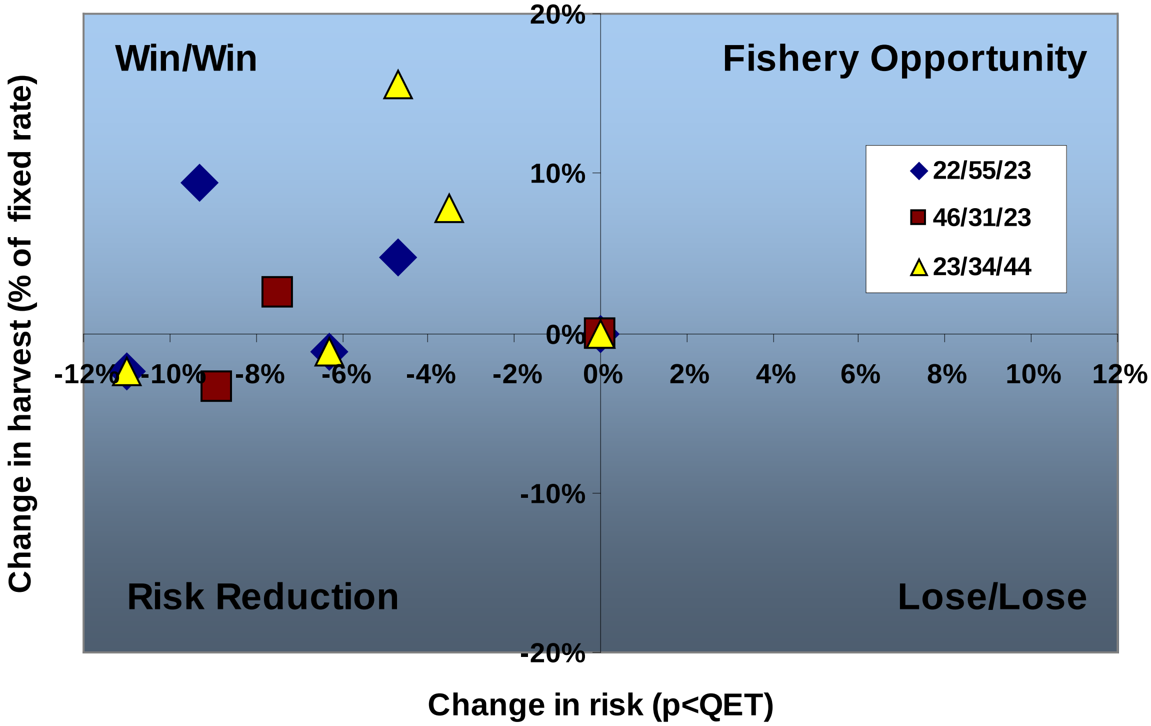

Figure 25. Example plot of the relative changes in natural population risk and LRH tule Chinook harvest level associated with different tier frequencies and fishery scenarios (Population B, 100-year simulations from Table 20).

Scenario Analysis

A variety of abundance-based approaches effectively reduce risks to wild B populations while also increasing average LRH harvest relative to a benchmark fixed-rate of 0.37 used for comparison purposes. Risks are generally most sensitive to reductions in fishing rates in the lower tier and relatively less sensitive to increased fishing rates in higher tiers where the frequency of occurrence is similar in the lower and upper tiers. Risks are reduced by lower ERs in poor ocean survival years which are most likely to result in low wild spawning escapements. Risks are relatively unaffected by higher ERs in years of good survival. Harvest benefits in years of higher abundance exceed harvest reductions in years of lower abundance. It should be noted however that lower ERs may be borne disproportionately by specific fisheries. The fishery implications of lower ERs are substantially more complex than the simple harvest numbers reflect.

Scenario results are summarized in Table 21, Table 22, and Table 23. For the purposes of this analysis, risks and harvest levels within the range observed for a ±1 percent change in fixed harvest rate were classified as equivalent to the fixed 0.37 values (shaded blue). Corresponding values were ±3.5 percent change in 100-year risk, ±0.25 percent change in 20-year risk, and ±3.0 percent change in average 100-year harvest. Lower risks and higher harvests (both desirable conditions) were shaded green. Higher risks and lower harvests (both undesirable conditions) were shaded yellow.

Table 21. Effects of variable rate fishing strategies based on three abundance tiers. Simulations are sorted by the 100-year Population B risk.

		Tier			Tier	Pop B risk		LRH	Change in risk		Δ Harvest	
	Scenario	Lower	Middle	Upper	Frequency	100 yr	20 yr	Harvest	100 yr	20 yr	100 yr	Category
1	F0	0.00	0.00	0.00	9/47/44	0.000	0.000	0	-21%	-1.6%	-100%	Risk reduction
2	F18	0.18	0.18	0.18	15/53/32	0.005	0.000	14,660	-21%	-1.6%	-51%	Risk reduction
3	F25	0.25	0.25	0.25	19/54/27	0.022	0.002	20,360	-19%	-1.4%	-32%	Risk reduction
4	V36-15	0.21	0.36	0.36	22/55/23	0.044	0.002	28,220	-17%	-1.4%	-6%	Risk reduction
5	F30	0.30	0.30	0.30	22/55/23	0.059	0.004	24,430	-15%	-1.2%	-19%	Risk reduction
6	V37-15	0.22	0.37	0.37	22/55/23	0.060	0.003	29,040	-15%	-1.3%	-4%	Risk reduction
7	V36±15	0.21	0.36	0.51	22/55/23	0.066	0.003	33,540	-14%	-1.3%	11%	Win/Win
8	V20/37/53	0.20	0.37	0.53	22/55/23	0.068	0.003	34,560	-14%	-1.3%	15%	Win/Win
9	F31	0.31	0.31	0.31	22/55/23	0.071	0.005	25,250	-14%	-1.1%	-16%	Risk reduction
10	V35±10	0.25	0.35	0.45	22/55/23	0.072	0.003	31,320	-14%	-1.3%	4%	Win/Win
11	V38-15	0.23	0.38	0.38	22/55/23	0.080	0.004	29,850	-13%	-1.2%	-1%	Risk reduction
12	V36-10	0.26	0.36	0.36	22/55/23	0.080	0.005	28,590	-13%	-1.1%	-5%	Risk reduction
13	V20/38/53	0.20	0.38	0.53	22/55/23	0.087	0.003	34,950	-12%	-1.3%	16%	Win/Win
14	F32	0.32	0.32	0.32	22/55/23	0.089	0.007	26,060	-12%	-0.9%	-14%	Risk reduction
15	V37±15	0.22	0.37	0.52	22/55/23	0.090	0.003	34,350	-12%	-1.3%	14%	Win/Win
16	V36±10	0.26	0.36	0.46	22/55/23	0.099	0.005	32,130	-11%	-1.1%	7%	Win/Win
17	V25/36/50	0.25	0.36	0.50	22/55/23	0.100	0.005	33,480	-11%	-1.1%	11%	Win/Win
18	V39-15	0.24	0.39	0.39	22/55/23	0.100	0.005	30,670	-11%	-1.1%	2%	Risk reduction
19	V37-10	0.27	0.37	0.37	22/55/23	0.100	0.005	29,400	-11%	-1.1%	-2%	Risk reduction
20	V25/37/50	0.25	0.37	0.50	22/55/23	0.111	0.005	33,860	-10%	-1.1%	12%	Win/Win
21	F33	0.33	0.33	0.33	22/55/23	0.114	0.008	26880	-10%	-0.8%	-11%	Risk reduction
22	V35±5	0.30	0.35	0.40	22/55/23	0.115	0.007	29,910	-10%	-0.9%	-1%	Risk reduction
23	V25/38/50	0.25	0.38	0.50	22/55/23	0.124	0.005	34,250	-9%	-1.1%	14%	Win/Win
24	V36-5	0.31	0.36	0.36	22/55/23	0.123	0.007	28,950	-9%	-0.9%	-4%	Risk reduction
25	V40-15	0.25	0.40	0.40	22/55/23	0.120	0.006	31,480	-9%	-1.0%	4%	Win/Win
26	V38-10	0.28	0.38	0.38	22/55/23	0.119	0.007	30,220	-9%	-0.9%	0%	Risk reduction
27	V37±10	0.27	0.37	0.47	22/55/23	0.117	0.005	32,950	-9%	-1.1%	9%	Win/Win
28	V38±15	0.23	0.38	0.53	22/55/23	0.116	0.005	35,170	-9%	-1.1%	17%	Win/Win
29	F34	0.34	0.34	0.34	22/55/23	0.129	0.011	27690	-8%	-0.5%	-8%	Risk reduction
30	V36±5	0.31	0.36	0.41	22/55/23	0.132	0.007	30,730	-8%	-0.9%	2%	Risk reduction
31	V25/40/45	0.25	0.40	0.45	22/55/23	0.136	0.006	33,250	-7%	-1.0%	10%	Win/Win
32	V28/37/50	0.28	0.37	0.50	22/55/23	0.142	0.007	34,080	-7%	-0.9%	13%	Win/Win
33	V39±15	0.24	0.39	0.54	22/55/23	0.144	0.007	35,980	-7%	-0.9%	19%	Win/Win
34	V38±10	0.28	0.38	0.48	22/55/23	0.147	0.007	33,760	-6%	-0.9%	12%	Win/Win

35	V39-10	0.29	0.39	0.39	22/55/23	0.147	0.008	31,030	-6%	-0.8%	3%	Risk reduction
36	V37-5	0.32	0.37	37	22/55/23	0.147	0.010	29,770	-6%	-0.6%	-1%	Risk reduction
37	F35	0.35	0.35	0.35	22/55/23	0.154	0.012	28,500	-6%	-0.4%	-5%	Risk reduction
38	V28/37/53	0.28	0.37	0.53	22/55/23	0.156	0.007	35,150	-5%	-0.9%	17%	Win/Win
39	V28/38/50	0.28	0.38	0.50	22/55/23	0.158	0.007	34,470	-5%	-0.9%	14%	Win/Win
40	V37±5	0.32	0.37	0.42	22/55/23	0.163	0.011	31,540	-5%	-0.5%	5%	Win/Win
41	V30/37/50	0.30	0.37	0.50	22/55/23	0.165	0.009	34,230	-5%	-0.7%	14%	Win/Win
42	V28/38/53	0.28	0.38	0.53	22/55/23	0.166	0.008	35,530	-4%	-0.8%	18%	Win/Win
43	V38-5	0.33	0.38	0.38	22/55/23	0.173	0.014	30,580	-4%	-0.2%	1%	Risk reduction
44	V30/37/53	0.30	0.37	0.53	22/55/23	0.174	0.011	35,290	-4%	-0.5%	17%	Win/Win
45	V39±10	0.29	0.39	0.49	22/55/23	0.175	0.011	34,580	-4%	-0.5%	15%	Win/Win
46	V30/38/50	0.30	0.38	0.50	22/55/23	0.176	0.012	34,620	-3%	-0.4%	15%	Fishery opportunity
47	V40-10	0.30	0.40	0.40	22/55/23	0.176	0.012	31,850	-3%	-0.4%	6%	Fishery opportunity
48	F36	0.36	0.36	0.36	22/55/23	0.179	0.014	29,320	-3%	-0.2%	-3%	Risk reduction
49	V40±15	0.25	0.40	0.55	22/55/23	0.180	0.007	36,800	-3%	-0.9%	22%	Fishery opportunity
50	V30/38/53	0.30	0.38	0.53	22/55/23	0.181	0.012	35,680	-3%	-0.4%	18%	Fishery opportunity
51	V38±5	0.33	0.38	0.43	22/55/23	0.183	0.014	32,360	-3%	-0.2%	7%	Fishery opportunity
52	V39-5	0.34	0.39	0.39	22/55/23	0.201	0.014	31,400	-1%	-0.2%	4%	Fishery opportunity
53	V40±10	0.30	0.40	0.50	22/55/23	0.204	0.012	35,390	-1%	-0.4%	17%	Fishery opportunity
54	F37	0.37	0.37	0.37	22/55/23	0.210	0.016	30,130	0%	0.0%	0%	Equivalent
55	V39±5	0.34	0.39	0.44	22/55/23	0.220	0.014	33,170	1%	-0.2%	10%	Fishery opportunity
56	V40-5	0.35	0.40	0.40	22/55/23	0.239	0.016	32,210	3%	0.0%	7%	Fishery opportunity
57	F38	0.38	0.38	0.38	22/55/23	0.248	0.019	30,950	4%	0.3%	3%	Fishery opportunity
58	V40±5	0.35	0.40	0.45	22/55/23	0.266	0.016	33,980	6%	0.0%	13%	Fishery opportunity
59	F39	0.39	0.39	0.39	22/55/23	0.294	0.019	31,760	8%	0.3%	5%	Fishery opportunity
60	F40	0.40	0.40	0.40	22/55/23	0.344	0.024	32,580	13%	0.8%	8%	Fishery opportunity
61	F42	0.42	0.42	0.42	22/55/23	0.438	0.034	34,210	23%	1.8%	14%	Fishery opportunity
62	F49	0.49	0.49	0.49	26/54/20	0.792	0.116	39,910	58%	10.0%	32%	Fishery opportunity
63	F53	0.53	0.53	0.53	28/54/18	0.912	0.200	43,160	70%	18.4%	43%	Fishery opportunity

Table 22. Effects of variable rate fishing strategies based on five abundance tiers. Simulations are sorted by the 100-year Population B risk.

	Scenario	Tier 1	Tier 2	Tier 3	Tier 4	Tier 5	Tier Frequency	Pop B risk 100 yr	Pop B risk 20 yr	LRH Harvest	Change in risk 100 yr	Change in risk 20 yr	Δ Harvest 100 yr	Category
65	V25/30/35/40/45	0.25	0.30	0.35	0.40	0.45	10/25/30/25/10	0.076	0.004	31,090	-13%	-1.2%	3%	Risk reduction
54	F37	0.37	0.37	0.37	0.37	0.37	10/25/30/25/10	0.210	0.016	30,130	0%	0.0%	0%	Equivalent
64	V30/35/40/45/50	0.30	0.35	0.40	0.45	0.50	10/25/30/25/10	0.210	0.013	35,160	0%	-0.3%	17%	Fishery opportunity

Tiers: <29,000; 29,000-50,000; 51,000-80,000; 81,000-137,000; >137,000

Table 23. Effects of variable rate and smoothed rate fishing strategies based on achieving desired abundance thresholds, tier frequencies, risk reduction, and harvest benefits.

	Scenario	Tiers 1	Tiers 2	Tiers 3	Tiers 4	Tiers 5	Exploitation Rate Limits (by 1	2	3	4	5	(hide) Ocean	Tier Frequency	Pop B Risk 100 yr	LRH Harvest	Δ Risk 100 yr	Δ Harvest 100 yr	Category
54	F37	<40	40-100	100+			37%	37%	37%			30%	22/55/23	0.210	30,130	0.0%	0.0%	Equivalent
68c	4-tier	<30	30-40	40-100	>100	--	30%	35%	38%	40%	--		11/11/55/23	0.169	31,300	-4.1%	3.9%	Win/Win
68d[a/]	4-tier	<30	30-40	40-80	>80	--	30%	35%	38%	40%	--		11/11/42/35	0.172	31,530	-3.8%	4.6%	Win/Win
68h1	4-tier	<30	30-40	40-80	>80	--	30%	35%	38%	41%	--		11/11/42/35	0.175	32,010	-3.5%	6.2%	Win/Win
68h2	4-tier	<30	30-40	40-85	>85	--	30%	35%	38%	41%	--		11/11/46/32	0.173	31,910	-3.7%	5.9%	Win/Win
68h3	4-tier	<30	30-40	40-90	>90	--	30%	35%	38%	41%	--		11/11/49/29	0.172	31,820	-3.8%	5.6%	Win/Win
68f	4-tier	<30	30-40	40-80	>80	--	30%	36%	38%	40%	--		11/11/42/35	0.178	31,580	-3.2%	4.8%	Fishery ↑
68g	4-tier	<30	30-40	40-80	>80	--	30%	36%	38%	41%	--		11/11/42/35	0.179	32,050	-3.1%	6.4%	Fishery ↑
68i	4-tier	<30	30-50	50-110	>110	--	30%	36%	39%	41%	--		11/24/42/23	0.175	31,900	-3.5%	5.9%	Win/Win
68j	4-tier	<30	30-50	50-110	>110	--	30%	36%	39%	42%	--		11/24/47/19	0.175	32,110	-3.5%	6.6%	Win/Win
69a	5-tier	<30	30-40	40-80	80-120	>120	30%	36%	38%	39%	40%		11/11/42/20/15	0.175	31,370	-3.5%	4.1%	Win/Win
69c	5-tier	<30	30-40	40-100	100-130	>130	30%	36%	38%	40%	42%		11/11/55/11/12	0.175	31,790	-3.5%	5.5%	Win/Win
70a	S30@30/41@80[b/]	<30		30-80		>80	30%		Linear		41%		11/54/36	0.137	31,490	-7.3%	4.5%	Win/Win
70b	S30@30/41@70	<30		30-70		>70	30%		Linear		41%		11/45/44	0.151	31,360	-5.9%	4.1%	Win/Win
70c	S30@30/41@60	<30		30-60		>60	30%		Linear		41%		11/35/54	0.166	32,220	-4.4%	6.9%	Win/Win
70e	S32@30/41@80	<30		30-80		>80	32%		Linear		41%		11/54/36	0.165	31,830	-4.5%	5.6%	Win/Win
70g	S30@30/45@100	<30		30-100		>100	30%		Linear		45%		11/66/23	0.151	33,060	-5.9%	9.7%	Win/Win
70h	S31@30/45@100	<30		30-100		>100	31%		Linear		45%		11/66/23	0.165	33,300	-4.5%	10.5%	Win/Win

a/ Scenario recommended by the SAS at the September Council meeting.

b/ The interpolation between 30,000 and 80,000 is based on the formula: 0.30 + [(forecast - 30,000)*((0.41 - 0.30) / (80000 - 30000))].

Scenario results may be highlighted with several examples. All of the following examples represent a significant reduction in risk and increase in average harvest relative to the fixed 0.37 ER scenario.

Simple abundance-based centered on current ER

These scenarios decrease ER by a fixed increment (0.05, 0.10, and 0.15) from the base rate in years when Columbia River run forecasts are less than 40,000, which occur about 22 percent of the time over the last 20 years. ER is increased by the same increment in years of forecasts over 100,000, which occur about 23 percent of the time. About 55 percent of the time, the base rate of 0.37 applies. Each of these simple scenarios would substantially reduce risk to the wild populations and increase harvest opportunity for LRH relative to a fixed 0.37 strategy. The greater the steps, the greater the effects.

		Pop B risk		LRH	Change in risk		Δ Harvest	
	Scenario	100 yr	20 yr	Harvest	100 yr	20 yr	100 yr	Category
54	F37	0.210	0.016	30,130	0%	0.0%	0%	Equivalent
40	V37±5	0.163	0.011	31,540	-5%	-0.5%	5%	Win/Win
27	V37±10	0.117	0.005	32,950	-9%	-1.1%	9%	Win/Win
15	V37±15	0.090	0.003	34,350	-12%	-1.3%	14%	Win/Win

Simple abundance-based with reduced ER

Abundance-based strategies may also be centered around different exploitation rates. This example employs a 0.35 ER at forecasts between 40,000 and 100,000, and ERs of 0.25 or 0.45 at lower or higher forecasts. This example also substantially reduces risk and increases harvest relative to the fixed 0.37 ER scenario.

		Pop B risk		LRH	Change in risk		Δ Harvest	
	Scenario	100 yr	20 yr	Harvest	100 yr	20 yr	100 yr	Category
54	F37	0.210	0.016	30,130	0%	0.0%	0%	Equivalent
10	V35±10	0.072	0.003	31,320	-14%	-1.3%	4%	Win/Win

V25/37/50

This is a symmetrical scenario centered on the current ER with bottom and top ends reflecting other fishery constraints. The bottom end (0.25) provides for very limited southern fisheries while recognizing current ERs of northern (Canada & Alaska) fisheries. The top end represents the limit of what is likely to be achievable in light of constraints on other stocks such as Snake River Wild fall Chinook. This example also represents a significant reduction in risk and increase in average harvest relative to the fixed 0.37 ER scenario.

		Pop B risk		LRH	Change in risk		Δ Harvest	
	Scenario	100 yr	20 yr	Harvest	100 yr	20 yr	100 yr	Category
54	F37	0.210	0.016	30,130	0%	0.0%	0%	Equivalent
20	V25/37/50	0.111	0.005	33,860	-10%	-1.1%	12%	Win/Win

V40-15 (V25/40/40)

This is an example of an unbalanced scenario that increases the base ER in the middle and upper tiers from 0.37 to 0.40 with an offsetting reduction in ER in the lower tier from 0.37 to

0.25. The corresponding increase in risk relative to the fixed 0.37 scenario is offset by a large reduction in ER in the lower tier. This example illustrates the required cost in low years to achieve a modest increase in most years. There is no risk of higher ERs in the upper tier if the forecast is off. This example also represents a significant reduction in risk and increase in average harvest relative to the fixed 0.37 ER scenario.

		Pop B risk		LRH	Change in risk		Δ Harvest	
	Scenario	100 yr	20 yr	Harvest	100 yr	20 yr	100 yr	Category
54	F37	0.210	0.016	30,130	0%	0.0%	0%	Equivalent
25	V40-15	0.120	0.006	31,480	-9%	-1.0%	4%	Win/Win

V25/40/45

This scenario is similar to V40-15 above, with slightly more risk/benefit from high forecast years when the ER in the upper tier is increased from 0.40 to 0.45. This scenario produces comparable risks and harvest to V25/37/50 which illustrates that a variety of scenarios can be configured to produce similar effects depending with different frequencies of operating at any given ER level.

		Pop B risk		LRH	Change in risk		Δ Harvest	
	Scenario	100 yr	20 yr	Harvest	100 yr	20 yr	100 yr	Category
54	F37	0.210	0.016	30,130	0%	0.0%	0%	Equivalent
31	V25/40/45	0.136	0.006	33,250	-7%	-1.0%	10%	Win/Win
32	V25/37/50	0.142	0.007	34,080	-7%	-0.9%	13%	Win/Win

V28/38/50

This scenario steps down the ER to 0.28 at forecasts under 40,000, increases the base ER from 0.37 to 0.38 in the middle tier, and goes to a substantially higher ER of 0.50 in the upper tier. This scenario produces similar benefits to V25/37/50 shown above. It illustrates that substantial flexibility can be gained in the middle and upper tiers if substantial reductions are implemented in the lower tier.

		Pop B risk		LRH	Change in risk		Δ Harvest	
	Scenario	100 yr	20 yr	Harvest	100 yr	20 yr	100 yr	Category
54	F37	0.210	0.016	30,130	0%	0.0%	0%	Equivalent
39	V28/38/50	0.158	0.007	34,470	-5%	-0.9%	14%	Win/Win
32	V25/37/50	0.142	0.007	34,080	-7%	-0.9%	13%	Win/Win

V30/35/38/41

These scenarios set the ER to 0.30 at forecasts under 30,000, which was identified as a threshold below which non-Indian fisheries north of Cape Falcon and in-river fisheries may need to be restricted to incidental impacts only to ensure achieving hatchery escapement goals. The 0.30 exploitation rate was also identified as the minimum rate necessary to accommodate expected impact from northern fisheries, consideration of treaty Indian troll fisheries, Puget Sound fisheries, south of Cape Falcon fisheries, and Chinook non-retention fisheries in non-Indian north of Cape Falcon and in-river fisheries. A number of scenarios were developed to try and optimize other parameters (Table 23). For example, in scenario 68h, the 0.38 and 0.41 tiers were intended to provide some harvest benefit over the current 0.37 limit at most LRH forecast levels. The 0.35 tier was intended to provide some risk reduction without completely eliminating north of Cape Falcon and in-river fisheries when LRH forecasts were low. The abundance level for the 0.35 tier was selected to keep the combined 0.30 and 0.35 tier frequencies at 22 percent, as in the three tiered scenarios while providing for some Chinook directed opportunity for non-Indian fisheries north of Cape Falcon and in-river. The 0.41 tier was selected to provide additional harvest benefit during large forecast years while maintaining an overall Win/Win categorization for the alternative, and the three abundance levels for this tier provide perspective on incremental risk reductions and harvest benefits (Figure 26).

S30-41

Scenarios 70a-70h use a sloped function exploitation rate generally based on the V30/35/38/40 scenario (Table 23). The intent was to reduce the effect on allowable exploitation rates of small abundance changes at threshold levels while maintaining the balance between minimum fishery needs, risk reduction, and harvest benefits (Figure 26).

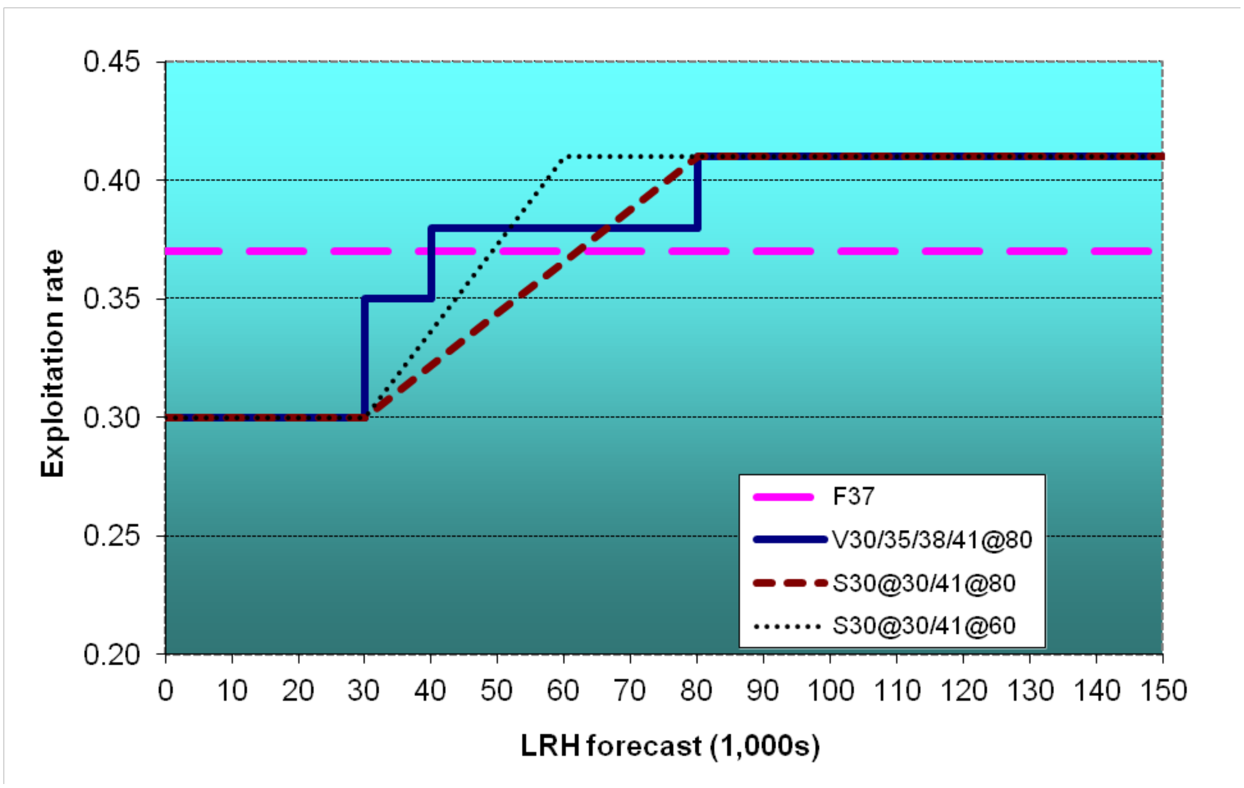

Figure 26. Illustration of variable and sloped scenarios compared to the fixed 0.37 exploitation rate standard.

Sensitivity Analysis

Effects on wild population risk of fishery “errors” in abundance-based management scenarios are illustrated in Figure 277. Fishery “error” in these simulations refers to the difference between target and actual exploitation rates. Over the last 10 years, the fishery error rate has averaged a CV of about 0.10. The sensitivity analysis shows that risks are relatively insensitive to error rates in that range at recent exploitation rates around 0.40. However, higher error rates increase risks under all scenarios as some run years, including low run abundance years that drive the risk calculation, may be subjected to substantially higher exploitation rates.

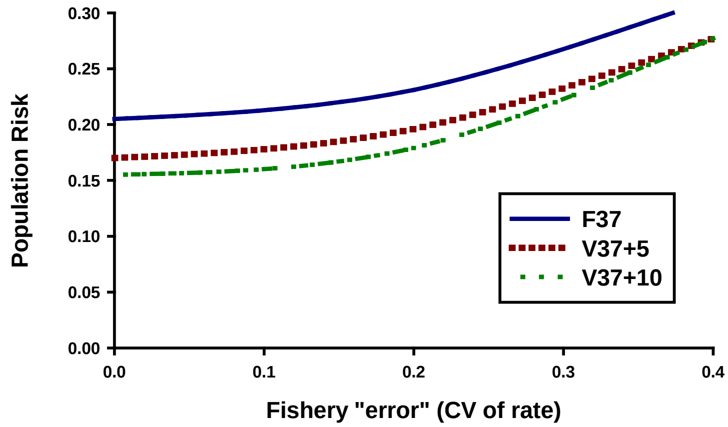

Figure 27. Effect of fishery "error" on 100-year risk for a category B population under several exploitation rate scenarios. Effects do not include the potential impacts of habitat/hatchery actions aimed at increasing natural production.

Effects on wild population risk of forecast errors in abundance-based management scenarios are illustrated in Figure 288. Of course, forecast error has no effect on risk in a fixed-rate scenario where the same exploitation rate occurs in every run size tier. Risk increases with increased forecast error rates for abundance-based strategies as higher exploitation rates are implemented for lower runs than would otherwise occur if run forecasts were more accurate. However, risk is not extremely sensitive to forecast error rates up to the current LRH level (CV=0.75).

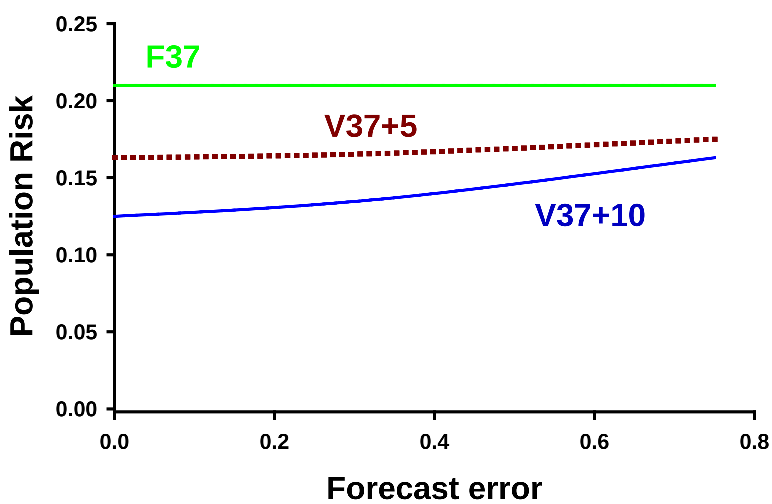

Figure 28. Effect of forecast error on 100-year risk for a category B population under several exploitation rate scenarios.

Discussion

Simulations of the effects of different fishing rate scenario suggest that wild population risks of low escapements can be reduced by an abundance-based approach, which reduces exploitation rates in years of low run forecasts. The general result of the model is that lower ERs provide benefits when run sizes are low, which offsets the cost of higher ERs when runs size is up. At the same time, fishery opportunities may be improved by higher exploitation rates in years of larger returns. A variety of scenarios were identified that provide both risk reduction and fishery opportunity benefits.

Both risk reduction and harvest benefits are relatively modest in the majority of abundance-based scenarios examined. However, the stochastic population model provided a systematic means of identifying scenarios that provide equivalent or lower risks to wild populations in comparison to the simple fixed-rate approach currently employed.

While several abundance-based approaches appear to provide conservation and fishery benefits based on average numbers, this analysis did not attempt to evaluate the implications of different scenarios. Different scenarios consisting of different combinations and frequencies of fishing rates may produce very similar risk and average harvest numbers, but have very different implications to specific fisheries. For instance, a V37±10 scenario reduces risk and increases average harvest relative to a V37±-5 scenario, but depending on how impacts are allocated, certain fisheries might not be fishing at all 22 percent of the time under the V37±10 scenario. Fishery stability will also be a critical consideration in identifying appropriate or acceptable scenarios. A desirable fishing strategy will seek to balance risk reduction, harvest benefits, and the frequency of being able to fish. Different fishing rates identified earlier in this report for specific fisheries provide a basis for application of some logical expectations regarding the implications of different fishing rate alternatives.

Expectations about tier frequency change depending on whether conditions remain similar to the preceding 20-year period or return to conditions more representative of the longer term. The model was parameterized to represent the last 20 years, which represent a prolonged period of low productivity and survival. The model estimates that forecasts under 40,000 LRH will occur approximately 22 percent of the time under similar conditions in the future. These projects also assume continuing production of about 22 million LRH hatchery juveniles per year, which is less than historical levels. However, the expectation changes depending on whether we look at the last 10, 20, or 30 years. The fact is we really have no way to know what the relative frequency of bad, medium and good years will be over the next several years. Because run sizes are highly variable as well as temporally autocorrelated, we could also easily see a sequence of low or high runs simply due to chance within any short-term period.

Scenario comparisons focused primarily on category B populations, which the risk modeling predicts will benefit the most from fishery limitations over the long term. The modeling indicates that previous reductions in ERs from very high levels observed in the late 1980s and early 1990s have substantially reduced the fishing-related risks to the more productive Category A populations. Modeling also showed that the long-term viability of the relative small and unproductive Category C populations will depend on improvements in the full spectrum of factors that are limiting, including habitat and hatchery influences.

It is important to underscore that the risk estimates provided in this analysis should not be used or interpreted as absolute values, but rather as relative changes in risk depending on the population type and fishing strategy. Absolute estimates of risk depend on the combined effect of a suite of model inputs and functions. The details of the model are based on the best available science, but in many cases information is limiting and the model structure is necessarily limited in turn. For instance, population-specific data is lacking for most lower Columbia River Chinook populations. Simulations assume that current conditions persist without consideration of improvements related to other recovery actions or degradation due to climate change. Effects of hatchery fish on spawner abundance or changes in hatchery contributions on population productivity are not modeled. Absolute estimates of risk also depend on definition of a standard for defining risk. In this case, risk is defined as the probability of the average number of wild spawners falling below a critical threshold of 50 fish for one generation (four years) within a prescribed period. However, selection of both the critical threshold and the simulation duration are somewhat subjective.

It should also be emphasized that the model runs included in this analysis do not include any potential contributions from the proposed habitat and hatchery improvements, and that the absolute value of the risk indices used to contrast harvest alternatives would be a function of those improvements. Absolute risk would be a function of assumptions regarding how much and how fast the response will be to implementing habitat actions (and in many cases local hatchery actions) designed to achieve the habitat targets. That, along with uncertainty in current model parameters, is the reason for focusing on relative reductions in risk instead of absolute risk values in the comparisons.

Application of these risk models are relatively robust to comparisons of the relative changes in risk (Morris and Doak 2002; Ralls et al. 2002). Comparisons of the effects of different fishing strategies on example populations of varying status are an example of a relative analysis. In this case, all the assumptions, inputs, and functions driving estimates of the absolute value of risks are common to all strategies. Thus, relative differences in risk estimated by the model reflect only the effects of the strategies being evaluated. Therefore, an abundance-based management approach would have similar effects on A and C populations as those modeled for B populations, relative to changes in fixed-rate management approaches.

CONCLUSIONS & RECOMMENDATIONS

1. Fishery models indicate that an abundance-based exploitation rate schedule can provide benefits to wild populations and the fisheries.
    - An abundance-based approach reduces ERs at low run sizes and increases ERs at high run sizes.
    - Wild populations benefit from reduced risk of low spawning escapements in years following poor ocean survival.
    - Fisheries benefit from increased ERs and greater harvest in years of larger returns.
    - Fishery benefits come at a cost of reduced opportunity in low run years.
2. Several things need to be considered for an abundance-based approach
    - The run sizes that are used to put into the three tiers are important. The run sizes should accurately reflect what the future conditions will be. This needs to incorporate both the abundances in each tier and the frequency in each tier.
    - Other fishery constraints need to be considered in weighing the feasibility and benefits of implementing some abundance-based scenarios. For instance, other stock limitations will constrain opportunities for increased exploitation rates on LRH tule Chinook. Similarly, low run sizes result in other constraints on ERs, such as risk of reaching hatchery brood stock goals.
3. Different fishery sectors are likely to share the benefits and burdens of the variable exploitation rate strategy unevenly.
    - The fishers will have to feel comfortable with the expectation of being in the lowest and highest tiers a certain amount of the time for each, and recognize the likelihood of consecutive years in those tiers.
4. There are numerous scenarios that should be considered when choosing an abundance-based matrix. Scenarios should meet the following criteria:
    - Wild population risks should be less than the risk associated with a fixed 0.37 ER.
    - On average, there should be a benefit to the fisheries – fisheries/harvest should be increased over the fixed 0.37 ER.
    - There are low-end ERs that could essentially eliminate fisheries off of Washington and Oregon and may not be considered reasonable choices.

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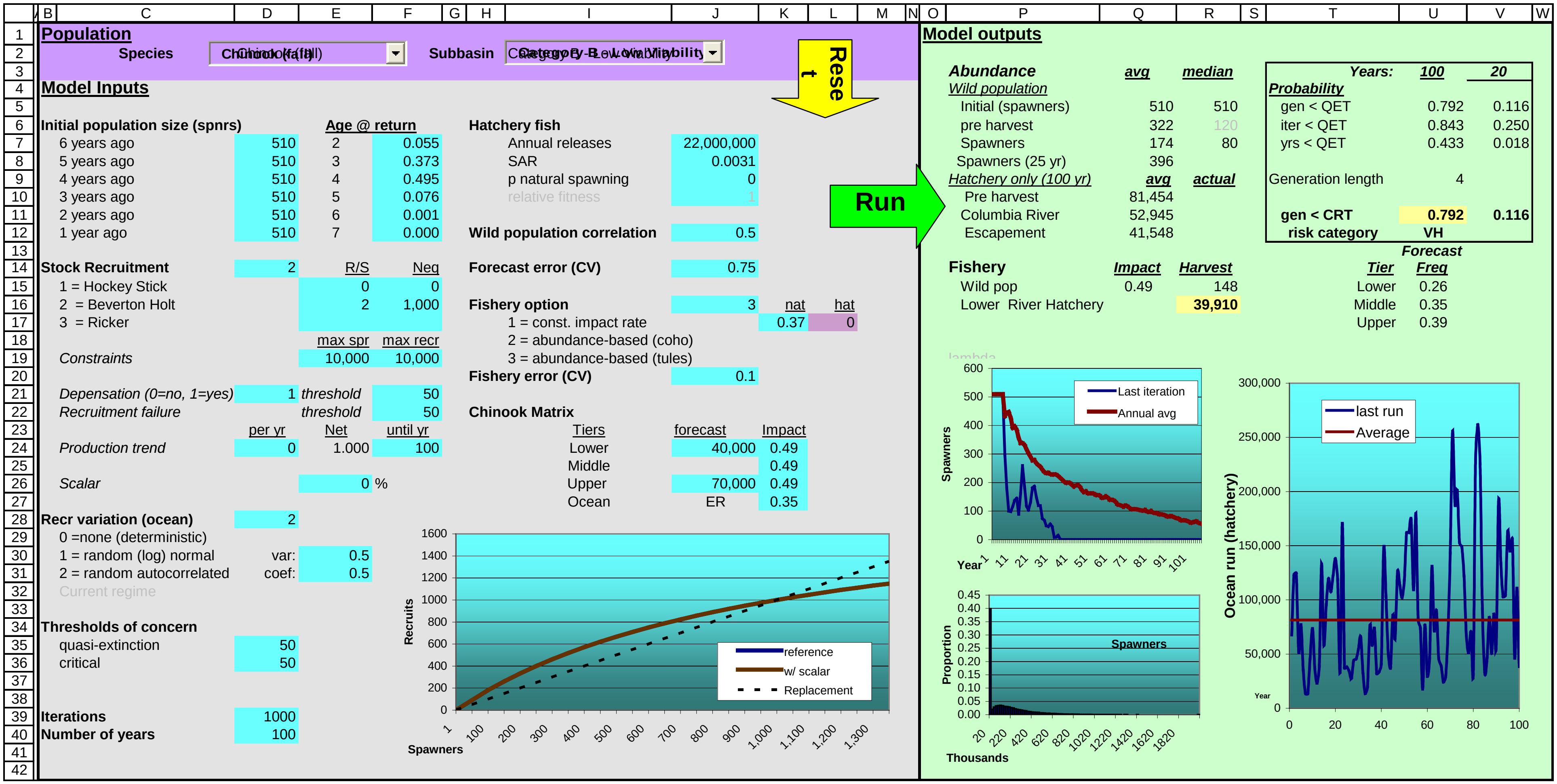
APPENDIX A – POPULATION VIABILITY ANALYSIS DOCUMENTATION

Interface Page



Formulae

Stock-Recruitment Function

The model stock recruitment function was based on the Beverton-Holt functional forms.

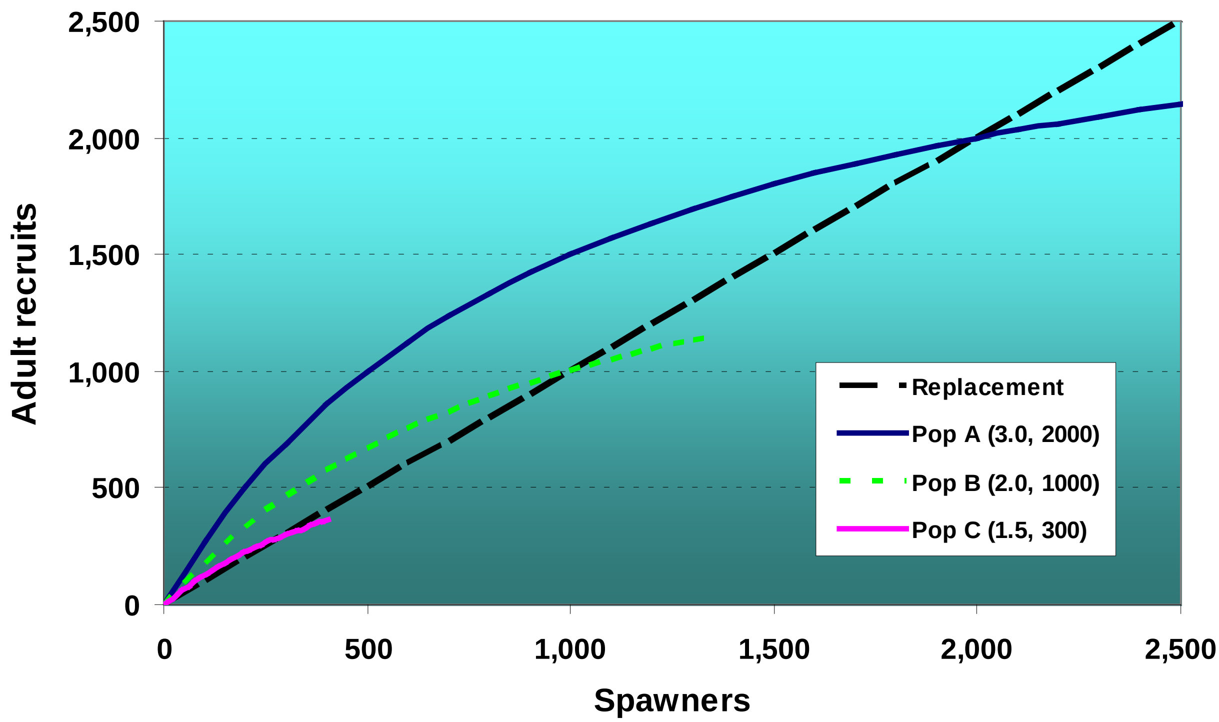


Figure 29. Examples of Beverton-Holt stock-recruitment curves.

The Beverton-Holt form of the relationship is:

$$R_y = \{a\, S_y / [1 + (S_y\,(a - 1)/ N_{eq})]\}\, e^{\varepsilon}$$

where

R_y = recruits,
S_y = spawners,
a = productivity parameter (maximum recruits per spawner at low abundance),
N_{eq} = parameter for equilibrium abundance,
e = exponent, and
ε = normally-distributed error term ~ $N(0, \sigma^2)$.

Stock-Recruitment Variance

The stochastic simulation model incorporated variability about the stock-recruitment function to describe annual variation in fish numbers and productivity due to the effects of variable freshwater and marine survival patterns (as well as measurement error in stock assessments). This variance is modeled as a lognormal distribution (e^{ε}) where ε is normally distributed with a mean of 0 and a variance of σ_z^2 .

The model allows for simulation of autocorrelation in stock-recruitment variance as follows:

$$Z_t = \emptyset\, Z_{t-1} + \varepsilon_t, \qquad \varepsilon_t \sim N(0, \sigma_e^2)$$

where

Z_t = autocorrelation residual,

Ø = lag autoregression coefficient,

ε_t = autocorrelation error, and

σ_e^2 = autocorrelation error variance.

The autocorrelation error variance (σ_e^2) is related to the stock-recruitment error variance (σ_z^2) with the lag autoregression coefficient:

$$\sigma_e^2 = \sigma_z^2 (1 - Ø^2)$$

Model simulations using the autocorrelated residual options were seeded in the first year with a randomly generated value from N(0, σ_z^2).

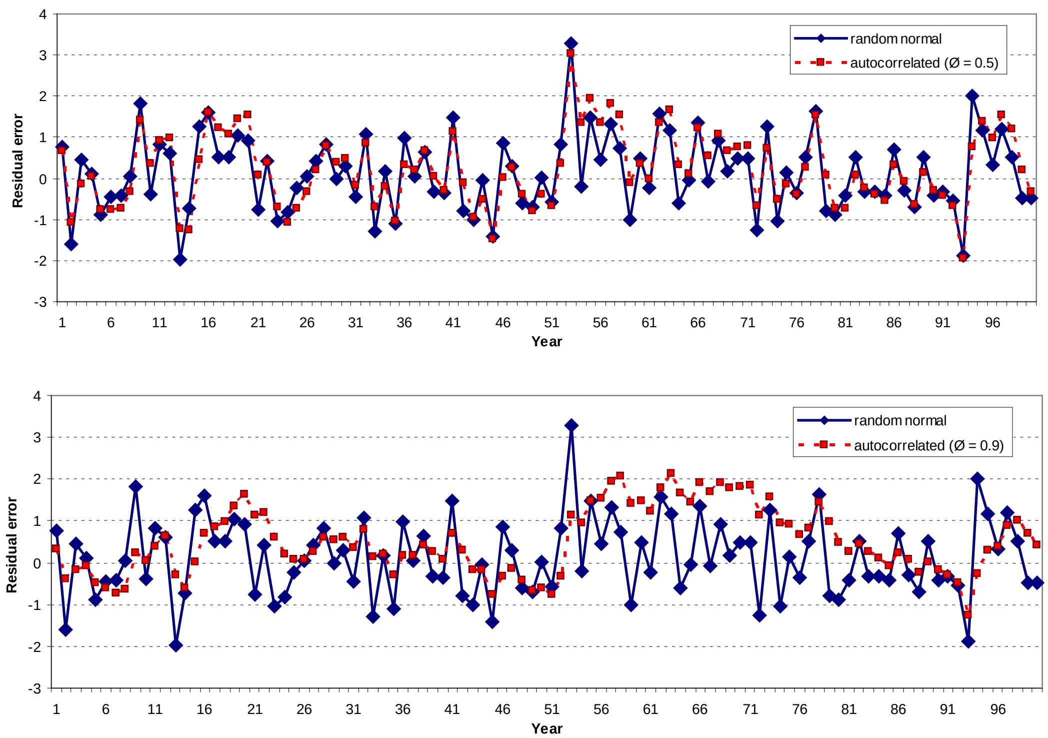


Figure 30. Examples of autocorrelation effect on randomly generated error patterns (σ_z^2 = 1).

Depensation & Recruitment Failure Thresholds

The model provides options to limit recruitment at low spawner numbers consistent with depensatory effects of stock substructure and small population processes. Options include 1) progressively reducing productivity at spawner numbers below a specified recruitment depensation threshold (RDT) and/or 2) setting recruitment to zero at spawner numbers below a specified recruitment failure threshold (RFT):

$$R' = R * (1 - Exp((Log(1 - 0.95) / (RDT - 1)) * S)) \text{ when } S > RFT$$

$$R' = 0 \text{ when } S < RFT$$

where

R' = Number of adult recruits after depensation applied,

R = Number of adult recruits estimated from stock-recruitment function,

S = spawners, and

RDT = Recruitment depensation threshold (spawner number).

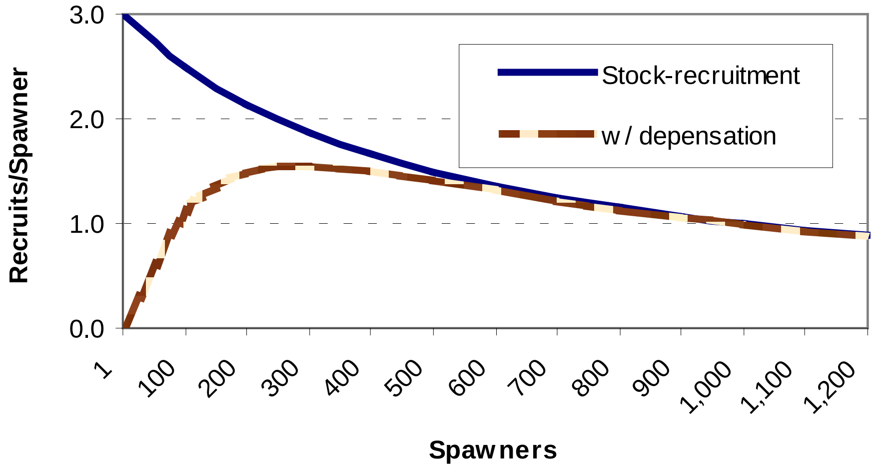


Figure 31. Example of depensation function effect on recruits per spawner at low spawner numbers based on a Beverton-Holt function (a = 3.0, Neq =1,000, γ =500).

Generic sensitivity analyses of production and abundance effects were based on a recruitment failure threshold of 50 (equal to the QET) and a recruitment depensation threshold equal to the CRT. Thus, spawning escapements of fewer than 50 spawners are assumed to produce no recruits and the depensation function reduces productivity of spawning escapements under the CRT value in any one year. Population-specific analyses were similarly based on a RFT of 50 and a recruitment depensation threshold equal to the CRT.

Production Trend

The model includes an optional input to allow average productivity to be annually incremented upward or downward so that effects of trends in habitat conditions might be considered:

$$R'' = R' (1 + t)^{y}$$

where

R' = Number of adult recruits after depensation applied, and

t = proportional annual change in productivity.

McElhany et al. (2006) assumed a median annual decline of ln(y) = 0.995 to future simulations based on a precautionary expectation of declining snow packs, survival indices, and climate change. Generic sensitivity and population-specific analyses included in this analysis did not assume a trend but additional sensitivity analyses were conducted to evaluate the effect of a range of declining trends on projected risks.

Annual Abundance

Numbers of naturally-produced fish ($N_{.y}$) destined to return to freshwater in each year are estimated from a progressive series of recruitment cohorts based on a specified age composition:

$$N_{.y} = \Sigma N_{xy}$$

$$N_{xy} = R^{*}_{y-x} m_{x}$$

where

N_{xy} = Number of mature naturally-produced adults of age x destined to return to freshwater in year y, and

m_x = Proportion of adult cohort produced by brood year spawners that returns to freshwater in year x

Hatchery Fish

The model includes option inputs for modeling co-occurring natural and hatchery populations. Number of hatchery-produced fish ($H_{.y}$) destined to return to freshwater in each year is estimated based on input juvenile release numbers (J), release-to-adult survival rates (SAR), and age composition (m_x):

$$H_{.y} = \Sigma H_{xy}$$

$$H_{xy} = (J)(SAR)(e^{\varepsilon})(m_x)$$

where

H_{xy} = Number of mature hatchery-produced adults of age x destined to return to freshwater in year y

Note that the model incorporates random normal variation in hatchery survival rates among release cohorts using a scalar based on natural productivity derived from the stock-recruitment variance. Thus, hatchery and natural numbers varied in tandem. The corresponding assumption would be that variation in hatchery and wild production was highly correlated due to common effects of freshwater and marine factors.

Run Forecasts

```
Forecast3(y) = NAd2H(i, y - 1) * (m3 / (m2 + 0.0000000001)) * (1 - LRHOcnER)
Forecast4(y) = NAd3H(i, y - 1) * (m4 / (m3 + 0.0000000001)) * (1 - LRHOcnER)
Forecast5(y) = NAd4H(i, y - 1) * (m5 / (m4 + 0.0000000001)) * (1 - LRHOcnER)
Forecast6(y) = NAd5H(i, y - 1) * (m6 / (m5 + 0.0000000001)) * (1 - LRHOcnER)
Forecast7(y) = NAd6H(i, y - 1) * (m7 / (m6 + 0.0000000001)) * (1 - LRHOcnER)

Forecast(y) = Forecast3(y) + Forecast4(y) + Forecast5(y) + Forecast6(y) + Forecast7(y) 'adults only
ForeVar(y) = Sqr(-2 * Log(Rnd())) * Sin(6.283185 * Rnd()) * (ForeErr * Forecast(y))
Forecast(y) = Forecast(y) + ForeVar(y)
```

Where

ForecastX(y) = age-specific forecast to Columbia River

mX = average proportion of brood year return by age X

Fisheries & Harvest

Annual numbers are subject to optional fishing rates. This option is useful for adjusting future projections for changes in fisheries and evaluating the effects of alternative fishing strategies and levels. Fishery impact is defined in the model in terms of the adult equivalent number of fish that die as a result of direct and indirect fishery effects:

$$IN_y = N_{.y} fN_y \text{ and } IH_y = H_{.y} fH_y$$

where

IN_y = fishery impact in number of naturally-produced fish,

fN_y = fishery impact mortality rate on naturally produced fish including harvested catch and catch-release mortality where applicable,

IH_y = Fishery impact in number of hatchery-produced fish, and

fH_y = fishery impact mortality rate including harvested catch and other mortality where applicable.

Spawning Escapement

Estimates of natural spawning escapement (S_y) include naturally-produced fish that survive fisheries plus a proportion of the hatchery escapement that spawns naturally decremented by the relative spawning success of a hatchery fish:

$$S_y = SN_y$$

$$SN_y = (N_{.y} - IN_y)$$

where

SN_y = Naturally-produced spawners in year y.

Model Processing Code

```
Option Explicit
'Dimension variables
   Public i    'iteration count
   Public y    'year count
   'Input variables
   Public Spp  'species
   Public Subbasin
   Public Nyr
   Public iter
   Public NSpn6ago
   Public NSpn5ago
   Public NSpn4ago
   Public NSpn3ago
   Public NSpn2ago
   Public NSpn1ago
   Public SRopt
   Public HSrps
   Public HSneq
   Public BHrps
   Public BHneq
   Public Rrps
   Public Rneq
   Public limitS 'large spawner number where recruitment no longer declines with increasing spawners
   Public limitR 'max recruitment allowed
   Public Depopt
   Public depthres
   Public RFT
   Public FWtrend
   Public FWyrs
   Public FWcond
   Public Ropt
   Public RMSE
   Public Rlag
   Public m3
   Public m4
   Public m5
   Public m2
```

```
Public m6
Public m7

Public Quasi
Public Genetic
Public Recov
Public Rgoal
Public Fopt
Public FrateNin
Public FrateN(110)
Public tgFrateN(110)
Public FrateHin
Public FrateH
Public Fcv
Public Fvar(110)
Public HatRel
Public HatSAR
Public Hatp

Public PopCor
Public ForeErr

Public Gen  'Weighted mean generation time by species for calc of moving avg

Public ISpp(150)
  Public ISubbas(150)
  Public ICRT(150)
  Public ISRtype(150)
  Public INeq(150)
  Public IRpS(150)
  Public IVar(150)
  Public Ilag(150)
  Public Iage2(150)
  Public Iage3(150)
  Public Iage4(150)
  Public Iage5(150)
  Public Iage6(150)
  Public Iage7(150)
  Public IER(150)

'State variables
Public NSpn(1000, 110) 'total spawners
Public NSpn2(1000, 110)
Public NSpn3(1000, 110)
Public NSpn4(1000, 110)
Public NSpn5(1000, 110)
Public NSpn6(1000, 110)
Public NSpn7(1000, 110)

Public NSpnN(1000, 110) 'natural origin spawners
Public NSpn2N(1000, 110)
Public NSpn3N(1000, 110)
Public NSpn4N(1000, 110)
Public NSpn5N(1000, 110)
Public NSpn6N(1000, 110)
```

```
Public NSpn7N(1000, 110)

Public NSpnH(1000, 110) 'hatchery origin spawners
Public NSpn2H(1000, 110)
Public NSpn3H(1000, 110)
Public NSpn4H(1000, 110)
Public NSpn5H(1000, 110)
Public NSpn6H(1000, 110)
Public NSpn7H(1000, 110)

Public Nocn(1000, 110)      'total
Public NocnN(1000, 110)     'natural origin ocean recruits
Public NocnH(1000, 110)     'ocean recruits hatchery

Public NEsc(1000, 110) 'total escaping fishery
Public NEsc2(1000, 110)
Public NEsc3(1000, 110)
Public NEsc4(1000, 110)
Public NEsc5(1000, 110)
Public NEsc6(1000, 110)
Public NEsc7(1000, 110)

Public NEscN(1000, 110) 'natural escaping fishery
Public NEsc2N(1000, 110)
Public NEsc3N(1000, 110)
Public NEsc4N(1000, 110)
Public NEsc5N(1000, 110)
Public NEsc6N(1000, 110)
Public NEsc7N(1000, 110)

Public NEscH(1000, 110) 'hatchery escaping fishery
Public NEsc2H(1000, 110)
Public NEsc3H(1000, 110)
Public NEsc4H(1000, 110)
Public NEsc5H(1000, 110)
Public NEsc6H(1000, 110)
Public NEsc7H(1000, 110)

Public NAd(1000, 110) 'total adults returning to freshwater
Public NAd2(1000, 110)
Public NAd3(1000, 110)
Public NAd4(1000, 110)
Public NAd5(1000, 110)
Public NAd6(1000, 110)
Public NAd7(1000, 110)

Public NAdN(1000, 110) 'natural adults returning to freshwater
Public NAd2N(1000, 110)
Public NAd3N(1000, 110)
Public NAd4N(1000, 110)
Public NAd5N(1000, 110)
Public NAd6N(1000, 110)
Public NAd7N(1000, 110)

Public NAdH(1000, 110) 'hatchery adults returning to freshwater
```

```
Public NAd2H(1000, 110)
Public NAd3H(1000, 110)
Public NAd4H(1000, 110)
Public NAd5H(1000, 110)
Public NAd6H(1000, 110)
Public NAd7H(1000, 110)

Public NHarN(1000, 110) 'natural adults harvested
Public NHar2N(1000, 110)
Public NHar3N(1000, 110)
Public NHar4N(1000, 110)
Public NHar5N(1000, 110)
Public NHar6N(1000, 110)
Public NHar7N(1000, 110)

Public NHarH(1000, 110) 'hatchery adults harvested
Public NHar2H(1000, 110)
Public NHar3H(1000, 110)
Public NHar4H(1000, 110)
Public NHar5H(1000, 110)
Public NHar6H(1000, 110)
Public NHar7H(1000, 110)

Public NHar(1000, 110) 'total adults harvested
Public NHar2(1000, 110)
Public NHar3(1000, 110)
Public NHar4(1000, 110)
Public NHar5(1000, 110)
Public NHar6(1000, 110)
Public NHar7(1000, 110)

'working variables
Public SRvar(1000, 110)
Public SRvarH(1000, 110)
Public HSvRate(1000, 110)
Public Z1(110)
Public Z2(110)
Public eSRvar
Public eSRvarLast
Public alphax        'revised stock-recruit alpha for fw production trend
Public Nsp

Public NRec5ago        'natural recruits 5 years ago
Public NRec4ago
Public NRec3ago
Public NRec2ago
Public NRec1ago

Public HRec5ago        'hatchery recruits 5 years ago
Public HRec4ago
Public HRec3ago
Public HRec2ago
Public HRec1ago
Public r
Public n
```

```
Public x
Public j    'counter for freq distr
Public jj
Public k    'counter
Public CountE(105)
Public CountR(105)
Public CounttgF(105)
Public CountF(105)
Public CntExtinct
   Public CntExtinctST 'short term
Public CntGenetic
   Public CntGeneticST 'gen < CRT short term
Public flagEx(1000)
      Public flagExST(1000)
Public MovGenAvg(1000, 110)
Public CntQETiter
Public CntQETiterST 'short term
   Public flagQET(1000)
   Public flagQETST(1000) ' short term
Public flagGR(1000)
   Public flagGRST(1000)
Public CntQETyr
   Public CntQETyrST
Public ENSpn
Public ENSpn10
Public ENocnN
Public ENocnH
Public ENAdH
Public ENEscH
Public EFrate
Public GNSpn(1000)
Public GNSpnE
Public NspnAvg(110)
'   Public cntFloor1
'   Public minNSpn1
 '  Public maxNSpn1
 '  Public ssNSpn1
'Coho matrix inputs
Public FIR(5, 5)
Public Neq
Public seedN(110)
Public MSIN(110)
Public seed(110)
Public MSI(110)
Public CntErr(30)
Public CntCell(5, 5)

Public LRHLT   'lower theshold forecast level for use in Chinook abundance-based modeling
Public LRHUT   'upper threshold
Public LRHLTER 'impact rate to apply to forecasts below lower threshold
Public LRHMTER 'impact rate to apply to forecasts below lower threshold
Public LRHUTER 'impact rate to apply to forecasts below lower threshold
Public LRHOcnER 'ocean impact used to back CR forecast out to preharvest recruits
Public Forecast(110) 'tule forecast used to drive fishing rate (derived)
Public Forecast2(110)
```

```
    Public Forecast3(110)
    Public Forecast4(110)
    Public Forecast5(110)
    Public Forecast6(110)
    Public Forecast7(110)
    Public ForeVar(110)
    Public CntTierA
    Public CntTierB
    Public CntTierC

Public Sub RunModel()
    Load UserForm1
    UserForm1.Show vbModeless

'Initialize inputs
    'Nyr = 100
    Nyr = Cells(40, 4)
    x = Rnd(-1234567)  ' initializes random number seed so that the same sequence of random numbers are
generated for any simulation
    eSRvarLast = 0

    'read from model sheet
    Spp = Cells(2, 4)
    Subbasin = Cells(3, 4)
    NSpn6ago = Cells(7, 4)
    NSpn5ago = Cells(8, 4)
    NSpn4ago = Cells(9, 4)
    NSpn3ago = Cells(10, 4)
    NSpn2ago = Cells(11, 4)
    NSpn1ago = Cells(12, 4)

    SRopt = Cells(14, 4)
    HSrps = Cells(15, 5)
    HSneq = Cells(15, 6)
    BHrps = Cells(16, 5)
    BHneq = Cells(16, 6)
    Rrps = Cells(17, 5)
    Rneq = Cells(17, 6)
    limitS = Cells(19, 5)
    limitR = Cells(19, 6)

    Depopt = Cells(21, 4)
    depthres = Cells(21, 6)

    RFT = Cells(22, 6)
    FWtrend = Cells(24, 4)
    FWyrs = Cells(24, 6)

    Ropt = Cells(28, 4)
    RMSE = Cells(30, 5)
    Rlag = Cells(31, 5)

    m2 = Cells(7, 6)
    m3 = Cells(8, 6)
    m4 = Cells(9, 6)
```

```
m5 = Cells(10, 6)
m6 = Cells(11, 6)
m7 = Cells(12, 6)

iter = Cells(39, 4)
Quasi = Cells(35, 4)
Genetic = Cells(36, 4)
Recov = Cells(26, 5)
Fopt = Cells(16, 10)
FrateNin = Cells(17, 11)
FrateHin = Cells(17, 12)

Fcv = Cells(20, 10)

HatRel = Cells(7, 10)
HatSAR = Cells(8, 10)
Hatp = Cells(9, 10)

PopCor = Cells(12, 10)
ForeErr = Cells(14, 10)

'MSI(1) = Cells(45, 10)
'MSI(2) = Cells(45, 10)
'MSI(3) = Cells(45, 10)

'coho matrix inputs
FIR(1, 1) = Cells(47, 10)
FIR(1, 2) = Cells(47, 11)
FIR(1, 3) = Cells(47, 12)
FIR(1, 4) = Cells(47, 13)

FIR(2, 1) = Cells(48, 10)
FIR(2, 2) = Cells(48, 11)
FIR(2, 3) = Cells(48, 12)
FIR(2, 4) = Cells(48, 13)

FIR(3, 1) = Cells(49, 10)
FIR(3, 2) = Cells(49, 11)
FIR(3, 3) = Cells(49, 12)
FIR(3, 4) = Cells(49, 13)

FIR(4, 1) = Cells(50, 10)
FIR(4, 2) = Cells(50, 11)
FIR(4, 3) = Cells(50, 12)
FIR(4, 4) = Cells(50, 13)

FIR(5, 1) = Cells(51, 10)
FIR(5, 2) = Cells(51, 11)
FIR(5, 3) = Cells(51, 12)
FIR(5, 4) = Cells(51, 13)

LRHLT = Cells(24, 10)
LRHUT = Cells(26, 10)
LRHLTER = Cells(24, 11)
LRHMTER = Cells(25, 11)
```

```
    LRHUTER = Cells(26, 11)
    LRHOcnER = Cells(27, 11)

    'If Spp = 1 Then  'coho
    '    Gen = 3
    'ElseIf Spp = 2 Then  'steelhead
    '    Gen = 7
    'ElseIf Spp = 3 Then  'spring chinook
    '    Gen = 6
    'ElseIf Spp = 4 Then 'fall chinook
    '    Gen = 5
    'ElseIf Spp = 5 Then 'chum
    '    Gen = 5
    'Else 'default
    '    Gen = 5
    'End If
    Gen = Round(2 * m2 + 3 * m3 + 4 * m4 + 5 * m5 + 6 * m6 + 7 * m7)

'Initialize summary statistics
    CntExtinct = 0
    CntExtinctST = 0
    CntGenetic = 0
    CntGeneticST = 0
    CntQETiter = 0
    CntQETiterST = 0
    CntQETyr = 0
    CntQETyrST = 0
    For j = 1 To 30
        CntErr(j) = 0
    Next j
    CntTierA = 0
    CntTierB = 0
    CntTierC = 0
    ENSpn = 0
    ENAdH = 0
    ENSpn10 = 0
    ENEscH = 0
    ENocnN = 0
    ENocnH = 0
    EFrate = 0
    '    minNSpn1 = 1000000
    '    maxNSpn1 = 0
    '    ssNSpn1 = 0
    For j = 1 To 100
        CountE(j) = 0
        CountR(j) = 0
        CounttgF(j) = 0
        CountF(j) = 0
    Next j
    For j = 1 To 5
        For k = 1 To 4
            CntCell(j, k) = 0
        Next k
    Next j
    If SRopt = 1 Then
```

```
      jj = HSneq * ((1 + FWtrend) ^ (FWyrs)) * (1 + (Recov / 100))
    ElseIf SRopt = 2 Then
      jj = BHneq * ((1 + FWtrend) ^ (FWyrs)) * (1 + (Recov / 100))
    ElseIf SRopt = 3 Then
      jj = Rneq * ((1 + FWtrend) ^ (FWyrs)) * (1 + (Recov / 100))
    End If
    jj = Int(jj * 2 / 100)
    For y = 1 To Nyr + 6
      NspnAvg(y) = 0
      tgFrateN(y) = 0
      FrateN(y) = 0
      seedN(y) = 0
      MSIN(y) = 0
      seed(y) = 0
      MSI(y) = 0
      Fvar(y) = 0
      'MovGenAvg(y) = 0
    Next y
    For i = 1 To iter
      GNSpn(i) = 1
      flagQET(i) = 0
      flagQETST(i) = 0
      flagEx(i) = 0
      flagExST(i) = 0
      flagGR(i) = 0
      flagGRST(i) = 0
    Next i
    GNSpnE = 0

  'Iterations
  For i = 1 To iter
    'reset annual values to 0 from previous iteration
    For y = 1 To Nyr + 6
      MovGenAvg(i, y) = 0
    Next y

    For y = 1 To Nyr + 6
     'Estimate recruits (for bookkeeping purposes recruits assumed to be 1 year old)
      If y > 1 Then Nsp = NSpn(i, y - 1)

      'Call GetSRvar
      'Hatchery recruits - estimate annual variation based on variance input
        Z1(y) = Sqr(-2 * Log(Rnd())) * Sin(6.283185 * Rnd())
        If Ropt = 0 Then
          SRvarH(i, y) = 0
        ElseIf Ropt = 1 Then
          SRvarH(i, y) = Z1(y) * Sqr(RMSE)
        ElseIf Ropt = 2 Then
          If y = 1 Then
            SRvarH(i, y) = Z1(y) * Sqr(RMSE * (1 - (Rlag ^ 2)))
          Else
            SRvarH(i, y) = (Rlag * eSRvarLast) + Z1(y) * Sqr(RMSE * (1 - (Rlag ^ 2)))
          End If
        End If
      'Natural recruits - estimate annual variation based on variance input
```

```
        Z2(y) = Sqr(-2 * Log(Rnd())) * Sin(6.283185 * Rnd())
        Z2(y) = (PopCor * Z1(y)) + (Sqr(1 - (PopCor ^ 2)) * Z2(y))  'Adjustment for partial correlation
        If Ropt = 0 Then
          SRvar(i, y) = 0
        ElseIf Ropt = 1 Then
          SRvar(i, y) = Z2(y) * Sqr(RMSE)
        ElseIf Ropt = 2 Then
        '   SRvar(i, y) = Z2(y) * Sqr(RMSE)
          If y = 1 Then
            SRvar(i, y) = Z2(y) * Sqr(RMSE * (1 - (Rlag ^ 2)))
          Else
            SRvar(i, y) = (Rlag * eSRvarLast) + Z2(y) * Sqr(RMSE * (1 - (Rlag ^ 2)))
          End If
        End If

        eSRvarLast = SRvarH(i, y) ' for autocorrelated reference
        'eSRvarLast = SRvar(i, y) ' for autocorrelated reference

      'Stock-recruitment calculation
      If SRopt = 1 Then 'hockey stick
        If Nsp * HSrps > HSneq Then NocnN(i, y) = HSneq * Exp(SRvar(i, y)) Else NocnN(i, y) = Nsp * HSrps *
Exp(SRvar(i, y))
        'If Nsp * HSrps > HSneq Then NocnN(i, y) = HSneq Else NocnN(i, y) = Nsp * HSrps
        Neq = HSneq
      ElseIf SRopt = 2 Then 'Beverton Holt
        NocnN(i, y) = Nsp * (BHrps / (1 + (((BHrps - 1) / BHneq) * Nsp))) * Exp(SRvar(i, y))
        Neq = BHneq
      ElseIf SRopt = 3 Then 'Ricker
        If Nsp > limitS Then Nsp = limitS 'bound spawner number to avoid weird descending limb effects
        NocnN(i, y) = Nsp * Exp((Rrps * (1 - (Nsp / Rneq))) + SRvar(i, y))
        'NocnN(i, y) = (Nsp * Exp(-(beta * Nsp) + SRvar(i, y)))
        Neq = Rneq
      End If

      If y > 6 Then
        If y < 7 + FWyrs Then FWcond = ((1 + FWtrend) ^ (y - 6)) Else FWcond = ((1 + FWtrend) ^ (FWyrs))
        NocnN(i, y) = NocnN(i, y) * FWcond * (1 + (Recov / 100)) 'adjust for freshwater production trend * recovery
increment
      End If

      If NocnN(i, y) > limitR Then NocnN(i, y) = limitR  'guards against unrealistic recruitment that exceeds observed
range
      If Depopt = 1 Then NocnN(i, y) = NocnN(i, y) * (1 - Exp((Log(1 - 0.95) / (depthres - 1)) * Nsp)) ' apply as
appropriate
      If Nsp < RFT Then NocnN(i, y) = 0  'assume critical depensation when spawners below recruitment failure
threshold

    'Hatchery recruits
      'NocnH(i, y) = HatRel * HatSAR * (1 + SRvarH(i, y - 1))
      NocnH(i, y) = HatRel * HatSAR * Exp(SRvarH(i, y - 1))
      If NocnH(i, y) < 0.2 * HatRel * HatSAR Then NocnH(i, y) = 0.2 * HatRel * HatSAR ' limits for unrealistic
randomly-generated values
      If NocnH(i, y) > 8 * HatRel * HatSAR Then NocnH(i, y) = 6 * HatRel * HatSAR
      HSvRate(i, y) = NocnH(i, y) / (HatRel + 0.0000000001)
```

```
 'total recruits
   Nocn(i, y) = NocnN(i, y) + NocnH(i, y)

 'annual number of adult escapement
   If y > 1 Then
     NAd2N(i, y) = NocnN(i, y - 1) * m2
     NAd2H(i, y) = NocnH(i, y - 1) * m2
   End If
   If y > 2 Then
     NAd3N(i, y) = NocnN(i, y - 2) * m3
     NAd3H(i, y) = NocnH(i, y - 2) * m3
   End If
   If y > 3 Then
     NAd4N(i, y) = NocnN(i, y - 3) * m4
     NAd4H(i, y) = NocnH(i, y - 3) * m4
   End If
   If y > 4 Then
     NAd5N(i, y) = NocnN(i, y - 4) * m5
     NAd5H(i, y) = NocnH(i, y - 4) * m5
   End If
   If y > 5 Then
     NAd6N(i, y) = NocnN(i, y - 5) * m6
     NAd6H(i, y) = NocnH(i, y - 5) * m6
   End If
   If y > 6 Then
     NAd7N(i, y) = NocnN(i, y - 6) * m7
     NAd7H(i, y) = NocnH(i, y - 6) * m7
   End If

   NAd2(i, y) = NAd2N(i, y) + NAd2H(i, y)
   NAd3(i, y) = NAd3N(i, y) + NAd3H(i, y)
   NAd4(i, y) = NAd4N(i, y) + NAd4H(i, y)
   NAd5(i, y) = NAd5N(i, y) + NAd5H(i, y)
   NAd6(i, y) = NAd6N(i, y) + NAd6H(i, y)
   NAd7(i, y) = NAd7N(i, y) + NAd7H(i, y)

   NAdN(i, y) = NAd2N(i, y) + NAd3N(i, y) + NAd4N(i, y) + NAd5N(i, y) + NAd6N(i, y) + NAd7N(i, y)
   'NAdH(i, y) = NAd2H(i, y) + NAd3H(i, y) + NAd4H(i, y) + NAd5H(i, y) + NAd6H(i, y) + NAd7H(i, y)
   NAdH(i, y) = NAd3H(i, y) + NAd4H(i, y) + NAd5H(i, y) + NAd6H(i, y) + NAd7H(i, y) 'jacks not counted in hatchery
adults
   NAd(i, y) = NAd2(i, y) + NAd3(i, y) + NAd4(i, y) + NAd5(i, y) + NAd6(i, y) + NAd7(i, y)

  'Fishing rates (rates don't matter before year 6 because spawners overwritten by historic observed)
   If y > 6 Then
     FrateN(y) = 0
     FrateH = 0
     If Fopt = 1 Then
       tgFrateN(y) = FrateNin
'       FrateH = FrateHin
     End If
     If Fopt = 2 Then
       FrateH = FrateHin
       If NSpn(i, y - 3) > 0.75 * Neq Then
         seedN(y) = 1
       ElseIf NSpn(i, y - 3) > 0.5 * Neq Then
```

```
            seedN(y) = 2
          ElseIf NSpn(i, y - 3) > 0.2 * Neq Then
            seedN(y) = 3
          ElseIf NSpn(i, y - 3) > 0.1 * Neq Then
            seedN(y) = 4
          ElseIf NSpn(i, y - 3) <= 0.1 * Neq Then
            seedN(y) = 5
          Else
            seedN(y) = 0
          End If

          If SRvar(i, y) > 1.3 * RMSE Then
            MSIN(y) = 4
          ElseIf SRvar(i, y) > 0 * RMSE Then
            MSIN(y) = 3
          ElseIf SRvar(i, y) > -0.7 * RMSE Then
            MSIN(y) = 2
          ElseIf SRvar(i, y) <= -0.7 * RMSE Then
            MSIN(y) = 1
          Else
            MSIN(y) = 0
          End If

          tgFrateN(y) = FIR(seedN(y), MSIN(y))
        End If
        If Fopt = 3 Then
          'Forecast includes an an adjustment for avg ocean harvest because matrix is indexed by Col R returns
          'Forecast based on hathery fish only
          'Forecast2(y) = (NAd2H(i, y - 1) + NAd2H(i, y - 2) + NAd2H(i, y - 3) + NAd2H(i, y - 4) + NAd2H(i, y - 5) +
NAd2H(i, y - 6)) / 6
          'Forecast2(y) = (NAd3H(i, y - 1)) * (0.055 / 0.373)
          Forecast3(y) = NAd2H(i, y - 1) * (m3 / (m2 + 0.0000000001)) * (1 - LRHOcnER)
          Forecast4(y) = NAd3H(i, y - 1) * (m4 / (m3 + 0.0000000001)) * (1 - LRHOcnER)
          Forecast5(y) = NAd4H(i, y - 1) * (m5 / (m4 + 0.0000000001)) * (1 - LRHOcnER)
          Forecast6(y) = NAd5H(i, y - 1) * (m6 / (m5 + 0.0000000001)) * (1 - LRHOcnER)
          Forecast7(y) = NAd6H(i, y - 1) * (m7 / (m6 + 0.0000000001)) * (1 - LRHOcnER)

          'Forecast(y) = Forecast2(y) + Forecast3(y) + Forecast4(y) + Forecast5(y) + Forecast6(y) + Forecast7(y)
          Forecast(y) = Forecast3(y) + Forecast4(y) + Forecast5(y) + Forecast6(y) + Forecast7(y) 'adults only
          ForeVar(y) = Sqr(-2 * Log(Rnd())) * Sin(6.283185 * Rnd()) * (ForeErr * Forecast(y))
          If ForeVar(y) > 0.8 * Forecast(y) Then ForeVar(y) = 0.8 * Forecast(y)
          If ForeVar(y) < 0.2 * Forecast(y) Then ForeVar(y) = 0.2 * Forecast(y)
          Forecast(y) = Forecast(y) + ForeVar(y)

          If Forecast(y) < LRHLT Then
            tgFrateN(y) = LRHLTER
            CntTierA = CntTierA + 1
          ElseIf Forecast(y) > LRHUT Then
            tgFrateN(y) = LRHUTER
            CntTierC = CntTierC + 1
          Else
            tgFrateN(y) = LRHMTER
            CntTierB = CntTierB + 1
          End If
        End If
```

```
        Fvar(y) = Sqr(-2 * Log(Rnd())) * Sin(6.283185 * Rnd()) * (Fcv * tgFrateN(y))
        FrateN(y) = tgFrateN(y) + Fvar(y)
        'If FrateN(y) > 3 * tgFrateN(y) Then FrateN(y) = 3 * tgFrateN(y)
        'If FrateN(y) < tgFrateN(y) / 3 Then FrateN(y) = tgFrateN(y) / 3
        If FrateN(y) > 0.8 Then FrateN(y) = 0.8
        If FrateN(y) < 0 Then FrateN(y) = 0
        FrateH = FrateN(y)
    End If

'Number escaping fishery
    NEsc2N(i, y) = NAd2N(i, y) * (1 - FrateN(y))
    'NEsc2N(i, y) = NAd2N(i, y) ' no harvest of jacks
    NEsc3N(i, y) = NAd3N(i, y) * (1 - FrateN(y))
    NEsc4N(i, y) = NAd4N(i, y) * (1 - FrateN(y))
    NEsc5N(i, y) = NAd5N(i, y) * (1 - FrateN(y))
    NEsc6N(i, y) = NAd6N(i, y) * (1 - FrateN(y))
    NEsc7N(i, y) = NAd7N(i, y) * (1 - FrateN(y))
    NEscN(i, y) = NEsc2N(i, y) + NEsc3N(i, y) + NEsc4N(i, y) + NEsc5N(i, y) + NEsc6N(i, y) + NEsc7N(i, y)

    'NEsc2H(i, y) = NAd2H(i, y) * (1 - FrateH)
    NEsc2H(i, y) = 0 ' only counting hatchery adults from this point forward
    NEsc3H(i, y) = NAd3H(i, y) * (1 - FrateH)
    NEsc4H(i, y) = NAd4H(i, y) * (1 - FrateH)
    NEsc5H(i, y) = NAd5H(i, y) * (1 - FrateH)
    NEsc6H(i, y) = NAd6H(i, y) * (1 - FrateH)
    NEsc7H(i, y) = NAd7H(i, y) * (1 - FrateH)
    NEscH(i, y) = NEsc2H(i, y) + NEsc3H(i, y) + NEsc4H(i, y) + NEsc5H(i, y) + NEsc6H(i, y) + NEsc7H(i, y)

    NEsc2(i, y) = NEsc2N(i, y) + NEsc2H(i, y)
    NEsc3(i, y) = NEsc3N(i, y) + NEsc3H(i, y)
    NEsc4(i, y) = NEsc4N(i, y) + NEsc4H(i, y)
    NEsc5(i, y) = NEsc5N(i, y) + NEsc5H(i, y)
    NEsc6(i, y) = NEsc6N(i, y) + NEsc6H(i, y)
    NEsc7(i, y) = NEsc7N(i, y) + NEsc7H(i, y)
    NEsc(i, y) = NEsc2(i, y) + NEsc3(i, y) + NEsc4(i, y) + NEsc5(i, y) + NEsc6(i, y) + NEsc7(i, y)

'Number impacted by the fishery farvest
'NHar2N(i, y) = NAd2N(i, y) * FrateN(y)
    NHar2N(i, y) = 0  'not counting harvest of jacks
    NHar3N(i, y) = NAd3N(i, y) * FrateN(y)
    NHar4N(i, y) = NAd4N(i, y) * FrateN(y)
    NHar5N(i, y) = NAd5N(i, y) * FrateN(y)
    NHar6N(i, y) = NAd6N(i, y) * FrateN(y)
    NHar7N(i, y) = NAd7N(i, y) * FrateN(y)
    NHarN(i, y) = NHar2N(i, y) + NHar3N(i, y) + NHar4N(i, y) + NHar5N(i, y) + NHar6N(i, y) + NHar7N(i, y)

    'NHar2H(i, y) = NAd2H(i, y) * FrateH
    NHar2H(i, y) = 0   'not counting harvest of jacks
    NHar3H(i, y) = NAd3H(i, y) * FrateH
    NHar4H(i, y) = NAd4H(i, y) * FrateH
    NHar5H(i, y) = NAd5H(i, y) * FrateH
    NHar6H(i, y) = NAd6H(i, y) * FrateH
    NHar7H(i, y) = NAd7H(i, y) * FrateH
    NHarH(i, y) = NHar2H(i, y) + NHar3H(i, y) + NHar4H(i, y) + NHar5H(i, y) + NHar6H(i, y) + NHar7H(i, y)
```

```
  NHar2(i, y) = NHar2N(i, y) + NHar2H(i, y)
  NHar3(i, y) = NHar3N(i, y) + NHar3H(i, y)
  NHar4(i, y) = NHar4N(i, y) + NHar4H(i, y)
  NHar5(i, y) = NHar5N(i, y) + NHar5H(i, y)
  NHar6(i, y) = NHar6N(i, y) + NHar6H(i, y)
  NHar7(i, y) = NHar7N(i, y) + NHar7H(i, y)
  NHar(i, y) = NHar2(i, y) + NHar3(i, y) + NHar4(i, y) + NHar5(i, y) + NHar6(i, y) + NHar7(i, y)

'Natural Escapement to spawners
 If y = 1 Then
    NSpn(i, y) = NSpn6ago
  ElseIf y = 2 Then
    NSpn(i, y) = NSpn5ago
  ElseIf y = 3 Then
    NSpn(i, y) = NSpn4ago
  ElseIf y = 4 Then
    NSpn(i, y) = NSpn3ago
  ElseIf y = 5 Then
    NSpn(i, y) = NSpn2ago
  ElseIf y = 6 Then
    NSpn(i, y) = NSpn1ago
  Else
    NSpn2N(i, y) = NEsc2N(i, y)
    NSpn3N(i, y) = NEsc3N(i, y)
    NSpn4N(i, y) = NEsc4N(i, y)
    NSpn5N(i, y) = NEsc5N(i, y)
    NSpn6N(i, y) = NEsc6N(i, y)
    NSpn7N(i, y) = NEsc7N(i, y)
    NSpnN(i, y) = NSpn2N(i, y) + NSpn3N(i, y) + NSpn4N(i, y) + NSpn5N(i, y) + NSpn6N(i, y) + NSpn7N(i, y)

    NSpn2H(i, y) = NEsc2H(i, y) * Hatp
    NSpn3H(i, y) = NEsc3H(i, y) * Hatp
    NSpn4H(i, y) = NEsc4H(i, y) * Hatp
    NSpn5H(i, y) = NEsc5H(i, y) * Hatp
    NSpn6H(i, y) = NEsc6H(i, y) * Hatp
    NSpn7H(i, y) = NEsc7H(i, y) * Hatp
    NSpnH(i, y) = NSpn2H(i, y) + NSpn3H(i, y) + NSpn4H(i, y) + NSpn5H(i, y) + NSpn6H(i, y) + NSpn7H(i, y)

    NSpn2(i, y) = NSpn2N(i, y) + NSpn2H(i, y)
    NSpn3(i, y) = NSpn3N(i, y) + NSpn3H(i, y)
    NSpn4(i, y) = NSpn4N(i, y) + NSpn4H(i, y)
    NSpn5(i, y) = NSpn5N(i, y) + NSpn5H(i, y)
    NSpn6(i, y) = NSpn6N(i, y) + NSpn6H(i, y)
    NSpn7(i, y) = NSpn7N(i, y) + NSpn7H(i, y)
    NSpn(i, y) = NSpn2(i, y) + NSpn3(i, y) + NSpn4(i, y) + NSpn5(i, y) + NSpn6(i, y) + NSpn7(i, y)
    If NSpn(i, y) < 0 Then NSpn(i, y) = 0

  seed(y) = NSpn(i, y) / (Neq + 0.0000000001)

  End If
'update iteration totals
  NspnAvg(y) = NspnAvg(y) + NSpn(i, y)
  'spawner frequencies
  If y > 6 Then
```

```
If NSpn(i, y) <= Quasi Then
   CntQETyr = CntQETyr + 1
   flagQET(i) = 1
   If y < 32 Then
      CntQETyrST = CntQETyrST + 1
      flagQETST(i) = 1 'count interations in 1st 25 yrs where i yr < QET
   End If
End If
ENocnN = ENocnN + NocnN(i, y)
ENocnH = ENocnH + NocnH(i, y)
ENAdH = ENAdH + NAdH(i, y)
ENSpn = ENSpn + NSpn(i, y)
ENEscH = ENEscH + NEscH(i, y)
If y < 17 Then ENSpn10 = ENSpn10 + NSpn(i, y)
EFrate = EFrate + FrateN(y)
'        GNSpn(i) = GNSpn(i) * (NSpn(i, y) + 1)
j = (Int(NSpn(i, y) / jj) + 1)
   If j > 100 Then j = 100
   If j < 1 Then j = 1
   CountE(j) = CountE(j) + 1
j = (Int(NocnN(i, y) / jj) + 1)
   If j > 100 Then j = 100
   If j < 1 Then j = 1
   CountR(j) = CountR(j) + 1

j = (Int(100 * tgFrateN(y) / 5) + 1)
   If j > 20 Then j = 20
   If j < 1 Then j = 1
   CounttgF(j) = CounttgF(j) + 1

j = (Int(100 * FrateN(y) / 5) + 1)
   If j > 20 Then j = 20
   If j < 1 Then j = 1
   CountF(j) = CountF(j) + 1

For k = 0 To Gen - 1
   MovGenAvg(i, y) = MovGenAvg(i, y) + (NSpn(i, y - k) / (Gen + 0.0000000001))
Next k
If MovGenAvg(i, y) < Quasi Then flagEx(i) = 1
If MovGenAvg(i, y) < Genetic Then flagGR(i) = 1
If y < 27 Then
   If MovGenAvg(i, y) < Quasi Then flagExST(i) = 1
   If MovGenAvg(i, y) < Genetic Then flagGRST(i) = 1
End If
If SRvar(i, y) > 1.4 * RMSE Then
   CntErr(1) = CntErr(1) + 1
ElseIf SRvar(i, y) > 1.3 * RMSE Then
   CntErr(2) = CntErr(2) + 1
ElseIf SRvar(i, y) > 1.2 * RMSE Then
   CntErr(3) = CntErr(3) + 1
ElseIf SRvar(i, y) > 1.1 * RMSE Then
   CntErr(4) = CntErr(4) + 1
ElseIf SRvar(i, y) > 1# * RMSE Then
   CntErr(5) = CntErr(5) + 1
ElseIf SRvar(i, y) > 0.9 * RMSE Then
```

```
      CntErr(6) = CntErr(6) + 1
    ElseIf SRvar(i, y) > 0.8 * RMSE Then
      CntErr(7) = CntErr(7) + 1
    ElseIf SRvar(i, y) > 0.7 * RMSE Then
      CntErr(8) = CntErr(8) + 1
    ElseIf SRvar(i, y) > 0.6 * RMSE Then
      CntErr(9) = CntErr(9) + 1
    ElseIf SRvar(i, y) > 0.5 * RMSE Then
      CntErr(10) = CntErr(10) + 1
    ElseIf SRvar(i, y) > 0.4 * RMSE Then
      CntErr(11) = CntErr(11) + 1
    ElseIf SRvar(i, y) > 0.3 * RMSE Then
      CntErr(12) = CntErr(12) + 1
    ElseIf SRvar(i, y) > 0.2 * RMSE Then
      CntErr(13) = CntErr(13) + 1
    ElseIf SRvar(i, y) > 0.1 * RMSE Then
      CntErr(14) = CntErr(14) + 1
    ElseIf SRvar(i, y) > 0 * RMSE Then
      CntErr(15) = CntErr(15) + 1
    ElseIf SRvar(i, y) > -0.1 * RMSE Then
      CntErr(16) = CntErr(16) + 1
    ElseIf SRvar(i, y) > -0.2 * RMSE Then
      CntErr(17) = CntErr(17) + 1
    ElseIf SRvar(i, y) > -0.3 * RMSE Then
      CntErr(18) = CntErr(18) + 1
    ElseIf SRvar(i, y) > -0.4 * RMSE Then
      CntErr(19) = CntErr(19) + 1
    ElseIf SRvar(i, y) > -0.5 * RMSE Then
      CntErr(20) = CntErr(20) + 1
    ElseIf SRvar(i, y) > -0.6 * RMSE Then
      CntErr(21) = CntErr(21) + 1
    ElseIf SRvar(i, y) > -0.7 * RMSE Then
      CntErr(22) = CntErr(22) + 1
    ElseIf SRvar(i, y) > -0.8 * RMSE Then
      CntErr(23) = CntErr(23) + 1
    ElseIf SRvar(i, y) > -0.9 * RMSE Then
      CntErr(24) = CntErr(24) + 1
    ElseIf SRvar(i, y) > -1 * RMSE Then
      CntErr(25) = CntErr(25) + 1
    ElseIf SRvar(i, y) > -1.1 * RMSE Then
      CntErr(26) = CntErr(26) + 1
    Else
      CntErr(27) = CntErr(27) + 1
    End If

    'Sheet9.Cells(y, 14) = seedN(y)
    'Sheet9.Cells(y, 15) = MSIN(y)

    CntCell(seedN(y), MSIN(y)) = CntCell(seedN(y), MSIN(y)) + 1
  End If
Next y
GNSpnE = GNSpnE + (GNSpn(i) ^ (1 / Nyr))
If flagQET(i) = 1 Then CntQETiter = CntQETiter + 1
  If flagQETST(i) = 1 Then CntQETiterST = CntQETiterST + 1
If flagEx(i) = 1 Then CntExtinct = CntExtinct + 1
```

```
      If flagExST(i) = 1 Then CntExtinctST = CntExtinctST + 1
    If flagGR(i) = 1 Then CntGenetic = CntGenetic + 1
      If flagGRST(i) = 1 Then CntGeneticST = CntGeneticST + 1
Next i
Call RunModelOutputs
Unload UserForm1
End Sub

'Public Sub GetSRvar()

'End Sub

Public Sub RunModelOutputs()
'Output summary statistics
    Sheet3.Cells(6, 17) = ENocnN / ((iter * Nyr) + 0.0000000001)
    Sheet3.Cells(7, 17) = ENSpn / ((iter * Nyr) + 0.0000000001)
    Sheet3.Cells(8, 17) = ENSpn10 / ((iter * 10) + 0.0000000001)  'short term (10 yr)spawners
    Sheet3.Cells(10, 17) = ENAdH / ((iter * Nyr) + 0.0000000001)
    Sheet3.Cells(11, 17) = (ENAdH / ((iter * Nyr) + 0.0000000001)) * (1 - LRHOcnER)
    'Sheet3.Cells(10, 17) = ENocnH / ((iter * 100) + 0.0000000001)
    'Sheet3.Cells(11, 17) = (ENocnH / ((iter * 100) + 0.0000000001)) * (1 - LRHOcnER)
    Sheet3.Cells(12, 17) = ENEscH / ((iter * Nyr) + 0.0000000001)
    Sheet3.Cells(15, 17) = EFrate / ((iter * Nyr) + 0.0000000001)
    'Sheet3.Cells(6, 18) = GNSpnE / (iter + 0.0000000001)
    Sheet3.Cells(5, 21) = CntExtinct / (iter + 0.0000000001)
      Sheet3.Cells(5, 22) = CntExtinctST / (iter + 0.0000000001)
    'Sheet3.Cells(5, 21) = CntExtinct
    Sheet3.Cells(6, 21) = CntQETiter / (iter + 0.0000000001) 'prob of gen<QET (100 Yr)
      Sheet3.Cells(6, 22) = CntQETiterST / (iter + 0.0000000001) 'prob of gen<QET (short term)
    Sheet3.Cells(7, 21) = CntQETyr / ((iter * Nyr) + 0.0000000001)
      Sheet3.Cells(7, 22) = CntQETyrST / ((iter * Nyr) + 0.0000000001)
    Sheet3.Cells(9, 21) = Gen
    Sheet3.Cells(11, 21) = CntGenetic / (iter + 0.0000000001)
      Sheet3.Cells(11, 22) = CntGeneticST / (iter + 0.0000000001)
    Sheet3.Cells(15, 21) = CntTierA / ((iter * Nyr) + 0.0000000001) 'tule forecast tiers
    Sheet3.Cells(16, 21) = CntTierB / ((iter * Nyr) + 0.0000000001)
    Sheet3.Cells(17, 21) = CntTierC / ((iter * Nyr) + 0.0000000001)
```

Sacramento River winter Chinook cohort reconstruction: analysis of ocean fishery impacts

DRAFT REPORT

Michael R. O'Farrell

Michael S. Mohr

Allen M. Grover

William H. Satterthwaite

Fisheries Ecology Division

Southwest Fisheries Science Center

National Marine Fisheries Service

110 Shaffer Road, Santa Cruz, CA 95060

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1 Abstract

Endangered Sacramento River winter Chinook (SRWC) are harvested incidentally in ocean salmon fisheries that target more abundant stocks. To evaluate the effect of these fisheries, cohort reconstructions were performed for ten broods (1998–2007) of hatchery-origin SRWC using coded-wire-tag data. Results indicate that the majority of ocean fishery impacts were attributed to recreational fisheries south of Point Arena, California. For complete broods 1998–2005, the number of potential SRWC spawners was reduced by an estimated 11 to 28 percent owing to ocean salmon fisheries. The spawner reduction rate for incomplete broods 2006 and 2007 will likely be zero, or nearly zero, due to the closure of most ocean salmon fisheries for 2008 and 2009 in California and Oregon. SRWC were predominantly caught as age-3, consistent with estimates of high ($>$ 85 percent) age-3 maturation rates that resulted in low ocean abundance of age-4 and older fish. Spawner reduction rates and ocean fishery age-3 impact rates were largely concordant and no temporal trend in these rates was observed over the range of years considered here, with the exception of recent years with widespread fisheries closures. In contrast to the relative consistency in ocean fishery effects on the SRWC population, the composite (hatchery and natural-origin) SRWC stock has experienced recent increases, and subsequent declines, in spawner escapement. These recent trends in spawner escapement cannot readily be explained by the exploitation history estimated during the same time frame.

2 Introduction

Chinook salmon fisheries in the ocean are conducted on a mixture of stocks, and fishing regulations are developed by the Pacific Fishery Management Council (PFMC) to primarily harvest abundant and/or productive target stocks. However, several non-target stocks, which may be listed as threatened or endangered by the Endangered Species Act (ESA), are caught incidentally in ocean fisheries. Sacramento River winter Chinook (SRWC) is one such stock. Measures intended to reduce or maintain the level of ocean fishery impacts on SRWC have been specified in the form of a National Marine Fisheries Service (NMFS) ESA consultation standard since the early 1990s. The primary focus of this Technical Memorandum is to evaluate the impact ocean fisheries have had on the SRWC stock.

Cohort reconstruction is a method commonly used in salmon stock assessment for estimation of exploitation rates. The basic principle of cohort reconstruction is the sequential estimation of a cohort's abundance from the end of the cohorts life span, when abundance is zero, to a specified earlier age (commonly age-2). A full cohort reconstruction can be completed only once the cohort's life span has ended. Age-specific escapement and harvest data are required and, in general, the natural mortality rates are assumed. Incomplete cohorts (i.e., cohorts whose life span has not yet ended) can also be partially reconstructed, but age-specific maturation rates must be assumed for the portion of the cohort yet to be observed. The reconstruction of a cohort's abundance enables the estimation of maturation, ocean harvest, contact, and impact rates, all of which allow for inference about the degree to which ocean fisheries impact a stock.

Cohort analysis methods can be applied to SRWC owing to the availability of age structured ocean harvest, river harvest, and escapement data derived from coded wire tag (CWT) recoveries. The cohort reconstructions described herein apply only to the hatchery-origin portion of the SRWC stock. No attempt was made to perform cohort reconstructions for the natural-origin portion of the stock, hence, the total abundance of SRWC is larger than the estimated abundances reported here. For other cohort reconstructions, such as those performed for Klamath River fall Chinook (Mohr 2006; Goldwasser et al. 2001), the natural area (non-hatchery) origin component of the cohort is reconstructed to obtain estimates of abundance for the composite natural and hatchery-origin stock. To rebuild the abundance of the natural stock, the assumption is made that the hatchery-origin portion of the stock shares the same harvest, impact, and contact rates as the natural-origin portion of the stock. Our goal with this analysis is limited to estimation of fishery impact and maturation rates, and not to obtain estimates of composite stock abundance. Hence, only the hatchery-origin portion of the stock is reconstructed.

SRWC was first listed under the Endangered Species Act (ESA) as threatened in 1989, and, since 1994, has been listed as endangered. The SRWC Evolutionary Significant Unit (ESU) has a high extinction risk, primarily owing to the lack of spatial structure in river spawning areas. Most SRWC historical spawning habitat lies behind impassable Keswick and Shasta Dams and current spawning is nearly all limited to a short stretch of the mainstem Sacramento River below Keswick Dam, an area not historically utilized by SRWC for spawning. Previous analysis of ocean harvest and impacts on SRWC has been confined to periods when marking and tagging of SRWC has occurred. In brood years 1969 and 1970, naturally produced SRWC were marked with a fin clip and estimates of marked SRWC harvest were made by CDFG (1989). The fin clips used to distinguish SRWC were also used for other stocks at this time, which likely confounded estimates of marked SRWC harvest in areas north of Point Arena, California. Nevertheless, these data and estimates indicated that marked SRWC were harvested primarily by the recreational fishery south of Point Arena. In addition, harvest of SRWC was highest in the months of February, March, July, and August. Marking and coded wire tagging of SRWC at Coleman National Fish Hatchery (CNFH) occurred in the early to mid 1990s, and harvest estimates for the tagged portion of the stock exist from broods 1991–1995 (Grover et al. 2004). For these broods, the spatiotemporal pattern of harvest was similar to that reported for the marked broods of 1969 and 1970 with the exception of a notable reduction in February and March harvest. These relative reductions in February and March harvest are clearly due to changes in fishing regulations in 1990 which closed or greatly reduced February and March fisheries south of Point Arena to protect SRWC. Prior to these changes in 1990, the recreational fishery in areas south of Point Arena began in mid-February. In 1998, marking and coded wire tagging of nearly 100% of SRWC hatchery production began at

Livingston Stone National Fish Hatchery (LSNFH), a conservation hatchery that produces SRWC at the upstream terminus of anadromy on the Sacramento River. The marking and CWT program at LSNFH enabled the reconstructions of the 1998–2000 SRWC broods (Grover et al. 2004). An analysis of harvest on these broods again found that the recreational fishery in areas south of Point Arena contributed most heavily to the total harvest. February harvest was nonexistent and March harvest was extremely low owing to restrictions on opening and closing dates for salmon seasons and minimum size limits specified by the ESA consultation standard. In addition, maturation rates for age-3 SRWC were estimated to be very high ($> 90\%$) and age-3 ocean impact rates ranged between 20% and 23%. Since the Grover et al. (2004) report, data from five new complete broods (2001–2005) and two incomplete broods (2006–2007) have become available.

The purpose of this memo is to estimate the degree that ocean salmon fisheries impact the endangered SRWC stock. We describe and present the cohort reconstructions for the hatchery component of this stock, and the subsequent estimation of maturation and ocean fishery impact rates that the reconstructions enable. Section 3 describes in detail the data and methods used for the cohort reconstructions and the estimators for maturation, harvest, contact, and impact rates. Results, including estimates of ocean impact rates and other key metrics are presented in Section 4. Discussion of the results, and comparisons of these results to those found in past assessments are presented in Section 5. We finish with a set of conclusions arising from our analysis. Additional details that pertain to the cohort reconstructions are presented in a set of Appendices at the end of this report.

3 Data and Methods

3.1 Data

Age-specific estimates of natural area escapement, hatchery escapement, river harvest, and ocean harvest of the hatchery-origin SRWC stock component are base requirements for cohort reconstruction. Estimates of these quantities can be derived from expanded CWT recoveries. CWTs recovered from river and ocean sampling programs are expanded for marking/tagging rates of less than 100% as well as non-exhaustive sampling of escapement and fisheries.

Nearly 100% of SRWC hatchery production is marked with a clipped adipose fin and tagged with a CWT. To account for the remaining unmarked/untagged portion of hatchery production, a *production expansion factor* $(1/\phi)$ is applied to each CWT recovered. The quantity ϕ is the proportion of hatchery releases for a particular tag code that received an adipose fin clip and a CWT. This proportion is estimated for each tag code in each brood year by the staff of LSNFH and is reported to the Regional Mark Processing Center (RMPC; http://www.rmpc.org).

All hatchery-origin SRWC caught in fisheries and returning to the river to spawn are not sampled in ocean and river monitoring programs, and all CWTs are not recovered and decoded. To account for the non-exhaustive sampling of harvest and escapement, a *sample expansion factor* $(1/\lambda)$ has been developed and is applied to each CWT recovered. The sampling fraction, λ, represents the fraction of the total escapement or harvest that was effectively sampled for CWTs in a particular stratum. Descriptions of escapement, river harvest, and ocean harvest sampling programs, and the methods used to estimate sample expansion factors, are described in the Escapement, River harvest, and Ocean harvest sections that follow.

Definitions for the notation used in this report are found in Table 1. A list of every CWT used in this analysis, as well as production and sample expansion factors associated with each CWT recovery, is available from the first author upon request.

3.1.1 Escapement

Spawner escapement of hatchery-origin SRWC occurs both to a trap operated by LSNFH at the base of Keswick Dam (the upstream anadromous boundary to the Sacramento River), as well as to natural spawning areas in the mainstem Sacramento River. SRWC returning to the Keswick trap serve as broodstock at LSNFH and are directly enumerated. Heads of SRWC with a clipped adipose fin used as hatchery broodstock are retained for CWT extraction and decoding. Estimation of natural area escapement of the SRWC hatchery-origin stock relies on CWT recoveries as well as sample expansion factors that account for the nonexhaustive sampling of natural spawning areas.

Carcass surveys and mark-recapture estimation methods have been used in the Sacramento

Table 1. Notation used in this analysis.

Symbol	Definition
a	Subscript denoting age, $a \in \{2,3,4,5\}$
C	Ocean fishery contacts
c	Contact rate
com	Term denoting the commercial fishery
D	Number of deaths due to "drop off" mortality
d	Drop off mortality rate
E	Escapement of hatchery-origin SRWC (to natural areas and the hatchery)
f	Fishing effort
H	Harvest
h	Harvest rate
hat	Term denoting hatchery spawner
I	Ocean fishery impacts
i	Impact rate
l	Total length, in inches
l^*	Minimum size limit for ocean fisheries; total length in inches
$1/\lambda$	Sample expansion factor
M	Number of mature, hatchery-origin SRWC returning to the river mouth
M^0	Simulated level of M absent the effects of ocean fisheries
m	Maturation rate
N	Ocean abundance of hatchery-origin SRWC
nat	Term denoting natural spawning areas
o	Subscript denoting ocean
p	Proportion of ocean harvest expected to be $\geq$ the minimum legal size
$1/\phi$	Production expansion factor
R	Number of decoded CWT recoveries
r	Subscript denoting river
rec	Term denoting the recreational fishery
rel	Subscript denoting releases from the hatchery
S	Number of deaths due to release mortality
s	Release mortality rate
t	Subscript denoting month
V	Number of deaths due to natural mortality
v	Monthly natural mortality rate
x	Subscript denoting fishery, $x \in$ {commercial, recreational}
y	Subscript denoting year
z	Subscript denoting area, $z \in$ {NO,CO,KO,KC,FB,SF,MO}

River to estimate natural area escapement of Chinook continuously since 1996. SRWC targeted carcass surveys are conducted jointly by the California Department of Fish and Game (CDFG) and the United States Fish and Wildlife Service (USFWS) from May–August in the mainstem Sacramento River upstream from Red Bluff Diversion Dam (RBDD). While Killam and Kreb (2008) and USFWS (2008) describe the SRWC carcass survey in detail, a general description of the survey and the application of CWT production and sample expansion factors follows.

Carcass surveys are conducted by field crews which examine carcasses of Chinook salmon found both on the bank and the bottom of the river. Carcasses encountered during the survey are considered to be "fresh" if they exhibit characteristics of recent death (e.g., at least one clear eye), or "decayed" if death was obviously not recent. A clipped adipose fin on any fresh or decayed carcass indicates hatchery-origin and the heads of all adipose clipped fish (and those with an unknown disposition of the adipose fin) are removed for CWT recovery and decoding. Fresh carcasses receive a visible, uniquely numbered, external tag and are returned to the river. If an externally tagged carcass is later recovered on a subsequent survey, it is noted and chopped in half to preclude counting at a later date. Decayed carcasses are noted then chopped in half and returned to the river. Data from the carcass surveys are used to estimate total escapement of SRWC to natural areas by applying Jolly-Seber mark-recapture estimation methods.

The number of hatchery-origin SRWC utilizing natural spawning areas is not directly estimated using Jolly-Seber methods. Instead, the escapement estimate for the hatchery-origin portion of the stock is derived from the total number of CWTs recovered and decoded, tag code specific production expansion factors, and spawning year specific carcass survey sample expansion factor. The method used to derive this sample expansion factor, developed by the authors of this report, is described in Appendix B. The derived sample expansion factors for spawning years 2001–2010, along with the data from which they were derived, is provided in Appendix C.

The spawner escapement of age a hatchery-origin SRWC in year y is estimated by the sum

$$E_{ay} = E_{ay}^{hat} + E_{ay}^{nat}. \tag{1}$$

The first term on the right-hand side of equation (1) is the hatchery-origin SRWC escapement to LSNFH, via the Keswick trap. The estimate of the hatchery-origin escapement to LSNFH, per CWT recovered and decoded, is equal to the production expansion factor $1/\phi$ associated with that decoded CWT. E_{ay}^{hat} is estimated by summing the expanded CWTs by year and age. The second term on the right-hand side of equation (1) is the hatchery-origin SRWC escapement to natural spawning areas. The estimate of hatchery-origin escapement to natural areas, per CWT recovered and decoded from the carcass survey, is equal to $(1/\lambda)(1/\phi)$. Both sample and production expansion factors are specific to each decoded CWT. E_{ay}^{nat} is estimated by summing the expanded CWTs by year and age.

3.1.2 River harvest

Recreational Chinook salmon fisheries have occurred annually in the Sacramento River, typically beginning in June or July and ending in December. Due to the timing of the river fishery, and the run timing of SRWC, few SRWC are expected to be harvested in the Sacramento River. The peak migration period of SRWC into the Sacramento River occurs in March (Fisher 1994) when the river fishery is closed to salmon retention.

CDFG conducted angler surveys on the Sacramento River from 2000–2002, resulting in eight winter Chinook CWT recoveries[1]. The sampling program was eliminated in 2003–2005, and most of 2006. In November of 2006, river fishery sampling began again in the upper Sacramento River and continues to the present time. Since the resumption of the sampling program, one winter run CWT was recovered in 2008, and two winter run CWTs were recovered in 2009.

The primary sampling method used for the river fishery has been a roving creel survey conducted by boat. The survey results in estimates of the number of angler hours by time and area. Historical time and area specific estimates of catch-per-angler-hour are then applied to estimate the total catch by time and area. These catch-per-angler-hour estimates were derived from his-

[1]Seven of the eight fish were caught between 28 Dec 2000 and 14 Jan 2001, just prior to the 15 Jan 2001 closure of the fishery. The California Fish and Game Commission responded to this finding by advancing the fishery closure date in all years subsequent to 2001 from Jan 15 to Jan 1 in order to minimize fishery impacts on SRWC.

torical exit surveys of anglers conducted along the Sacramento River. During the survey angler interviews, samplers collect heads from adipose fin clipped fish for CWT recovery and decoding.

The sampling fraction is computed as the ratio of the number of fish sampled to the total catch by time and area. The estimate of the hatchery-origin SRWC river harvest, per CWT recovered and decoded from the angler survey, is equal to $(1/\lambda)(1/\phi)$. Both production and sample expansion factors are specific to each decoded CWT. H_{ray} is then estimated by summing the expanded CWTs by year and age.

3.1.3 Ocean harvest

Commercial and recreational ocean salmon fishery harvest is sampled by CDFG and Oregon Department of Fish and Wildlife (ODFW) in their respective states using similar methods. Both state agencies maintain a CWT extraction and decoding laboratory, and report data associated with CWTs to RMPC. Each state agency also reports catch and fishing effort by month, management area, and fishery each year in the PFMC "Review of Ocean Fisheries" document series (e.g., PFMC 2011). The seven ocean management areas used to spatially stratify ocean harvest estimates of hatchery-origin SRWC for this analysis are described in Table 2. Previous cohort reconstructions considered an eighth area, South of Sur, that resulted from splitting the MO management area into separate northern and southern areas. Fishing effort and landings south of Point Sur, California are generally quite low relative to more northern areas, and typically fisheries management measures are equivalent over the entire region from Pigeon Point to the U.S./Mexico border. Splitting MO into two areas has no effect on reconstructed SRWC abundance, impact rate, or spawner reduction rate estimates. For these reasons, the South of Sur management area was not included in this analysis.

Commercial fishery sampling primarily occurs during fish sales transactions. Salmon are counted, weights are recorded, and heads or snouts are collected from all adipose fin clipped Chinook salmon for CWT extraction and decoding. At this time, fishermen are interviewed to determine the number of days fished and area of catch. The sampling fraction is computed as the ratio of salmon sampled to the total landing estimate, which is based on landing receipts.

Table 2. Ocean management areas used in this analysis. Areas are contiguous, listed from north to south. The southern border of the MO area is the U.S./Mexico border. KMZ denotes Klamath Management Zone.

Area	Abbreviation	Northern border	Major ports
Northern Oregon	NO	Cape Falcon, OR	Newport, Tillamook
Central Oregon	CO	Florence South Jetty, OR	Coos Bay
Oregon KMZ	KO	Humbug Mountain, OR	Brookings
California KMZ	KC	OR/CA border	Eureka, Crescent City
Fort Bragg	FB	Horse Mountain, CA	Fort Bragg
San Francisco	SF	Point Arena, CA	San Francisco
Monterey	MO	Pigeon Point, CA	Monterey

Recreational fishery sampling is performed differently depending if the fishing activity occurs on commercial passenger fishing vessels (CPFV) or privately operated fishing vessels (POFV). For the CPFV recreational fishery, sampling to determine catch and effort is similar to that of the commercial fishery, and landing receipts reported to the respective state agencies are used to make total landings estimates. Heads or snouts are taken from all adipose fin clipped salmon examined by dockside samplers, and the sampling fraction is computed in the same manner as described for commercial fisheries. For the POFV recreational fishery, the sampling is structured differently because landings receipts are not required for private boaters. POFV sampling programs are typically a stratified random creel survey of all available points of landing within a port area. Sampling effort is also stratified by day-type: weekend/holiday versus weekday. Samplers attempt to interview all returning anglers, record the number of Chinook landed per angler, and collect heads or snouts from adipose fin clipped Chinook salmon for CWT extraction and decoding. Estimates of total catch and fishing effort are made based on the sampled catch and the ratio of days and sites sampled to the total number of possible days and sites in the stratum. The catch and effort estimates are then aggregated to an estimate of catch and effort by port and month.

Both CDFG and ODFW attempt to sample at least 20% of the landed catch in the recreational and commercial fishery, for each month and management area. Sample expansion factors reported by CDFG and ODFW to RMPC were derived from sampling fractions and include corrections for heads not collected and for CWTs that were lost or not readable, as was done for the carcass surveys. Estimated hatchery-origin SRWC harvest, per decoded CWT recovery, is equal to $(1/\lambda)(1/\phi)$. Both sample and production expansion factors are specific to each decoded CWT. H_{oatzxy}, the ocean harvest of hatchery-origin SRWC by age, month (t), area (z), fishery (x) and year is then estimated by summing the respective expanded CWTs.

3.2 Methods

3.2.1 Cohort reconstruction

The reconstruction of a cohort with no extant individuals (i.e., a "complete" cohort) proceeds sequentially from the end of that cohort's life span. Given the estimated quantities E_a, H_{ra}, and H_{oatzx} (hereafter ignoring the year y subscripts), we defined the age-specific number of mature, hatchery-origin SRWC leaving the ocean for the Sacramento River

$$M_a = E_a + H_{ra} \tag{2}$$

and the following metrics pertaining to ocean fisheries:

$$C_{oatzx} = H_{oatzx}/p_{oatzx} \tag{3}$$

$$S_{oatzx} = (C_{oatzx} - H_{oatzx}) \times s_{oatzx} \tag{4}$$

$$D_{oatzx} = C_{oatzx} \times d \tag{5}$$

$$I_{oatzx} = H_{oatzx} + S_{oatzx} + D_{oatzx}. \tag{6}$$

M_a is defined as escapement from ocean fisheries, and in the absence of river fishery harvest, is equal to spawner escapement. Natural mortality is assumed to be zero in the river. The quantity p_{oatzx} was estimated based on a length-at-age model for SRWC and the month, area, and fishery specific size limit (Appendix A). The release mortality rate conventions employed are $s = 0.26$ for the commercial fishery and $s = 0.14$ for the recreational fishery, based on a review of hook and release mortality studies by the PFMC Salmon Technical Team (STT 2000). In addition, for recreational fisheries in SF and MO, the estimate of s has dependence on the proportion of anglers "mooching", a style of fishing that results in greater release mortality rate than trolling (Grover et al. 2002). The dropoff mortality rate d was assumed to be 0.05, the value recommended by STT (2000).

Ocean impacts were aggregated over management areas and fisheries to produce an estimate of the total ocean-wide fishery impacts incurred by age and month,

$$I_{oat} = \sum_{zx} I_{oatzx}. \tag{7}$$

Given these quantities, individual cohorts were reconstructed in the following manner:

$$N_{oat} = \begin{cases} 0 & a \geq 6 \\ I_{oat} + V_{oat} + M_a + N_{o(a+1)(t+1)} & a \in \{2,3,4,5\};\ t = Feb \\ I_{oat} + V_{oat} + N_{oa(t+1)} & a \in \{2,3,4,5\};\ t \neq Feb \end{cases} \tag{8}$$

where

$$V_{oat} = \begin{cases} (M_a + N_{o(a+1)(t+1)}) \times [v_a/(1-v_a)] & a \in \{2,3,4,5\};\ t = Feb \\ N_{oa(t+1)} \times [v_a/(1-v_a)] & a \in \{2,3,4,5\};\ t \neq Feb. \end{cases} \tag{9}$$

The cohort reconstruction approximates river entry timing, and exit from ocean fisheries, by specifying that mature fish enter the river on the last day of February. The monthly, age-specific natural mortality rate (v_a) for age-2 is assumed to be 0.0561, which corresponds to a 50% annual rate. The monthly natural mortality rate for ages 3, 4, and 5 is assumed to be 0.0184, corresponding to a 20% annual rate. The use of assumed values for v_a is necessary for estimation of exploitation rates through cohort analysis, and the values used here are consistent with those used for other Pacific salmon (e.g., Goldwasser et al. 2001).

For the most recent cohorts with life spans not yet completed, we used an approximation to perform a partial cohort reconstruction. For the 2006 brood, the age-5 river harvest and escapement have not yet been estimated. Since the data do not extend into the age-5 portion of this cohort, the reconstruction of ocean abundance begins at the month of ocean exit (February) prior to age-4 escapement. Cohort abundance of age-4 individuals on Feb 1 was approximated as

$$N_{o(4)(Feb)} = I_{o(4)(Feb)} + V_{o(4)(Feb)} + \frac{M_4}{avg\{m_4\}}, \tag{10}$$

where $avg\{m_4\}$ is the mean age-4 maturation rate estimated from all complete cohorts and

$$V_{o(4)(Feb)} = \frac{M_4}{avg\{m_4\}} \times \frac{v_4}{1 - v_4}. \tag{11}$$

The ocean abundance of the 2006 cohort was then rebuilt from February 1, age-4, using equation sets (8) and (9).

For the 2007 brood, both age-4 and age-5 river harvest and escapement have not yet been estimated. For this brood, the reconstruction of ocean abundance begins on Feb 1 prior to age-3 escapement. This is accomplished by using equations (10) and (11), modifying the equations such that age-3 is substituted for age-4.

3.2.2 Estimation

Using the quantities defined in equations (2) through (7) and the reconstructed abundances, maturation, harvest, contact, and impact rates were estimated as follows. The maturation rate was estimated as the age-specific fraction of fish alive at the end of February that enter the river:

$$m_a = \frac{M_a}{N_{oa(Feb)} - I_{oa(Feb)} - V_{oa(Feb)}}. \tag{12}$$

Age, month, area, and fishery specific contact, harvest, and impact rates were estimated as:

$$c_{oatzx} = C_{oatzx}/N_{oat} \tag{13}$$

$$h_{oatzx} = H_{oatzx}/N_{oat} \tag{14}$$

$$i_{oatzx} = I_{oatzx}/N_{oat}. \tag{15}$$

Note that the denominator in these equations is the age-specific ocean-wide abundance at the beginning of month t. The annual age-specific impact rate, was estimated as

$$i_{oa} = \frac{\sum_{t=Mar}^{Feb} \sum_{zx} I_{oatzx}}{N_{oa(Mar\ 1)}}, \tag{16}$$

with the denominator in this case being the age-specific ocean-wide abundance at the beginning of the SRWC biological year (i.e., March 1).

The SRR, also referred to as the adult equivalent exploitation rate, is a measure of the effect of ocean fisheries on the adult spawning potential of a brood. It is the reduction in a brood's potential spawning escapement owing to ocean fisheries, relative to its escapement potential in the absence of ocean fishing:

$$SRR = \frac{M^0 - M}{M^0}. \tag{17}$$

M^0 is a brood's projected river return of adult SRWC (age 3–5), absent the effect of ocean fisheries, and M is a brood's observed adult river return. M^0 is derived by projecting the March 1, age-2 abundance forward through age-5 spawners, assuming that maturation rates are the cohort- and age-specific estimates determined by equation (12) and that all mortality is due to natural factors. This formulation isolates the impact of ocean fisheries on the spawning potential, and makes the assumption that no mortality is incurred after river entry. For incomplete cohorts, the SRR was expressed as a range of plausible estimates because maturation and ocean impact rates are unavailable for the final year, or two years, of the cohort's life span and therefore these values must be assumed. Maximum bounds of the SRR for incomplete cohorts were estimated by assuming that the unestimated, age-specific maturation rates were the maximum maturation rates (at age) observed from all complete broods. Impact rates were assumed be 1.0 after the last observed escapement (i.e., all fish died due to fisheries after the last observed escapement and the cohort therefore did not contribute to future escapement). Minimum bounds of the SRR were calculated by assuming that the unestimated maturation rates were equal to the minimum maturation rate at age observed for all complete broods. Impact rates after the last observed spawning escapement estimate were assumed to be zero. Hence, future returns were only limited by natural mortality.

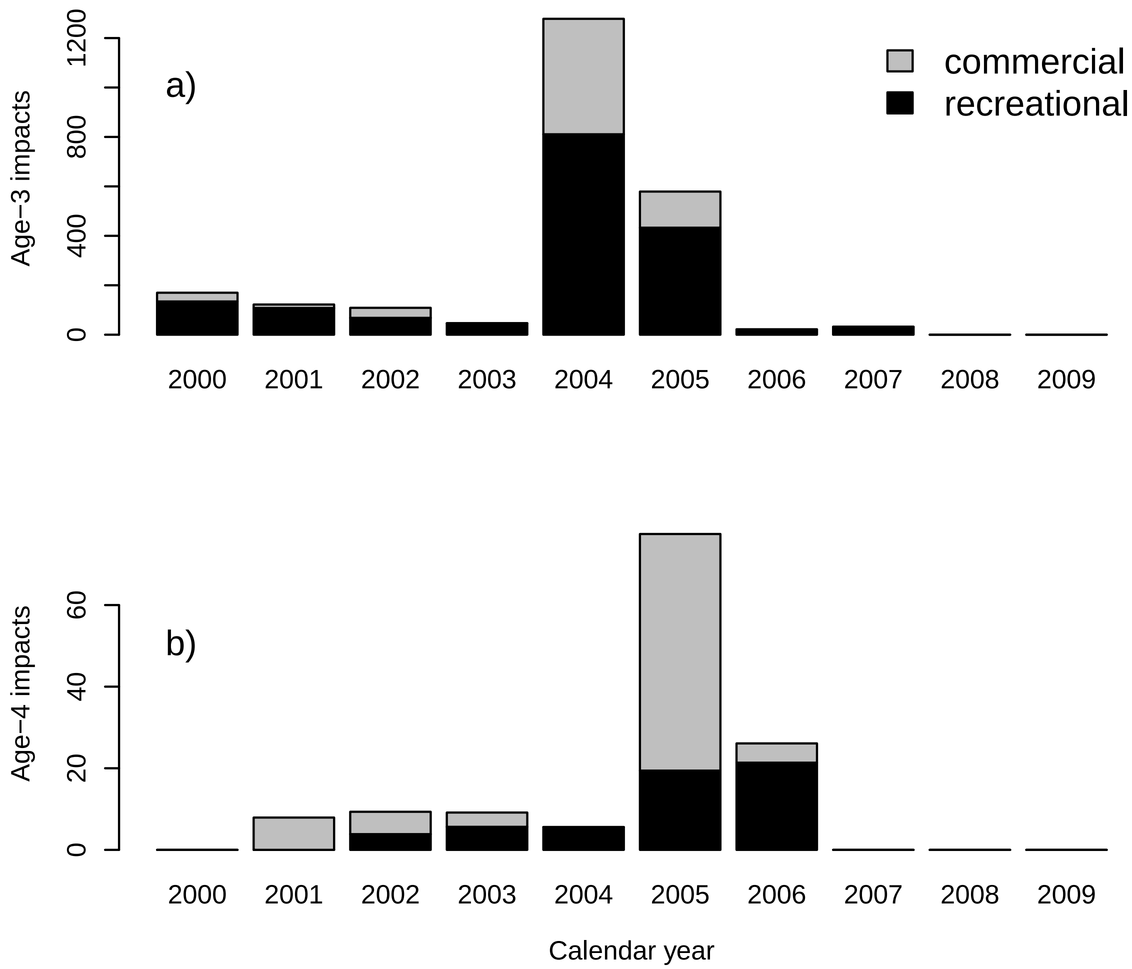

Figure 1. Ocean fishery impacts for hatchery-origin (a) age-3 and (b) age-4 Sacramento River winter Chinook estimated by calendar year. Total impacts by year are the sum of impacts over all ocean fishery management areas.

4 Results

The number of ocean fishery impacts on hatchery-origin SRWC has been quite variable between the years 2000 and 2009 (Figure 1). Age-3 impacts greatly outnumber age-4 impacts (note the scale difference between Figure 1a and 1b), and were primarily the result of recreational fisheries. Recreational fisheries have smaller minimum size limit regulations than commercial fisheries and therefore the relatively small age-3 SRWC are more vulnerable to retention in the recreational fishery. In general, a larger proportion of the age-4 impacts are attributed to the commercial fishery, likely reflecting the increased vulnerability of older and larger fish to retention in that fishery. The

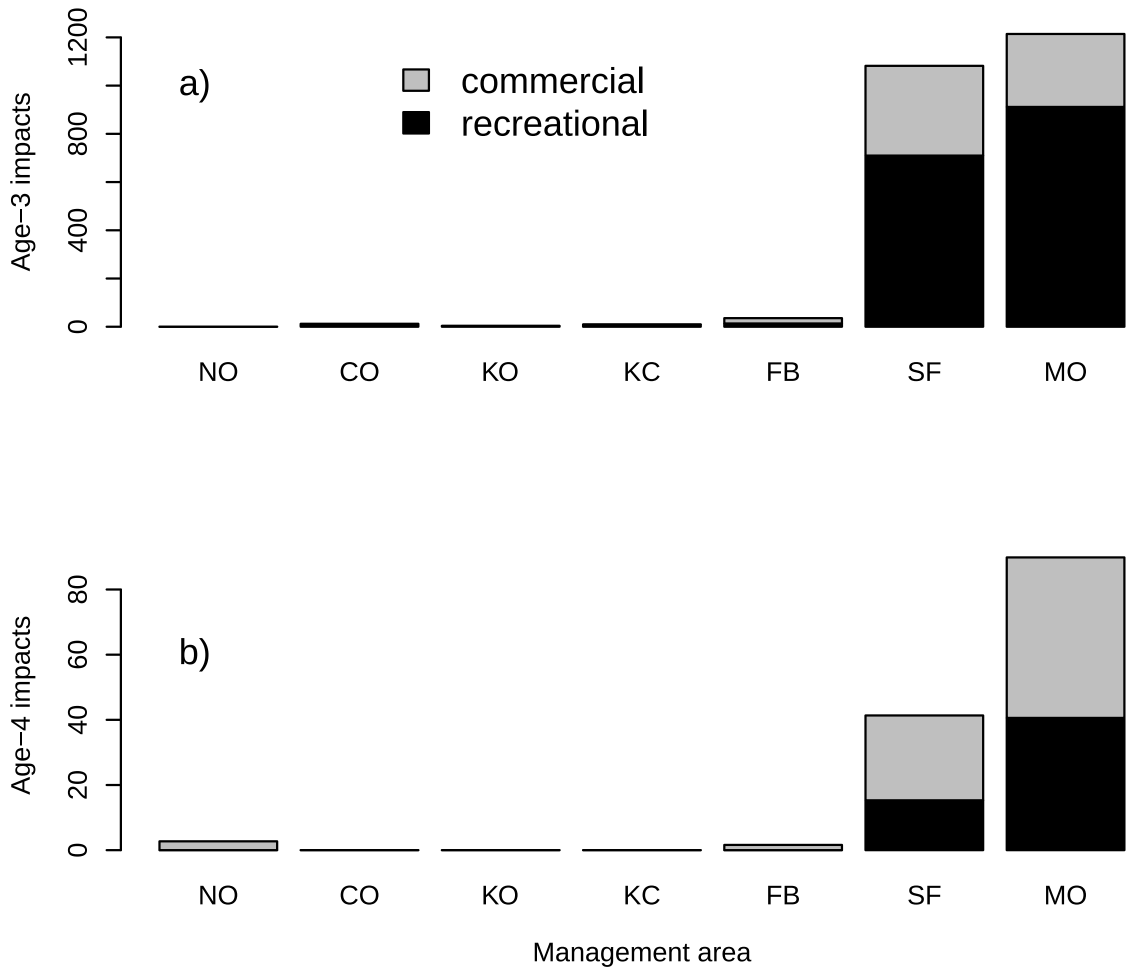

Figure 2. Ocean fishery impacts for hatchery-origin (a) age-3 and (b) age-4 Sacramento River winter Chinook estimated by ocean fishery management area. Total impacts by area are the sum of impacts over calendar years 2000–2009.

highest age-3 impacts occurred in 2004 and 2005, and these anomalies were apparent as age-4 impacts in 2005 and 2006. Nearly all ocean salmon fisheries were closed in 2008 and 2009, for those years impacts are zero. A clear pattern in the spatial distribution of ocean fishery impacts is evident in Figure 2. Impacts in areas north of the SF management area are rare or absent for both age-3 and age-4 SRWC; the SF and MO areas contribute the great majority of ocean fishery impacts.

The reconstruction of cohorts from brood years 1998–2007 enabled the estimation of maturation rates, the SRR, and ocean fishery impact rates. Table 3 displays estimated maturation rates for age-2 through age-4. Of particular relevance is the consistently high age-3 maturation rate. The

Table 3. Estimated age specific maturation rates (m_a), impact rates (i_a), and the spawner reduction rate (SRR). Maturation rate and SRR estimates reported only for complete broods 1998–2005.

Brood year	m_2	m_3	m_4	i_3	i_4	SRR
1998	0.0419	0.8542	0.8274	0.2338	0.1247	0.2641
1999	0.1639	0.9545	1.0000	0.2512	0.7163	0.2278
2000	0.0632	0.9453	1.0000	0.2183	0.5471	0.2322
2001	0.0605	0.9739	1.0000	0.1034	0.6721	0.1131
2002	0.0345	0.9305	1.0000	0.2559	0.3827	0.2759
2003	0.0403	0.9487	0.9467	0.1717	0.2306	0.1803
2004	0.0227	0.9590	1.0000	0.1505	0.0000	0.1538
2005	0.0101	1.0000	1.0000	0.1778	0.0000	0.1861
2006	–	–	–	0.0000	0.0000	–
2007	–	–	–	0.0000	–	–

high age-3 maturation rate results in relatively low age-4 ocean abundance (see Appendix D), since the preponderance of SRWC return to spawn at age-3. This maturation schedule also contributes to the high level of age-3 ocean fishery impacts relative to age-4 impacts.

The SRR, estimated for complete broods 1998–2005, ranged from 11.31% to 27.59% (Figure 3; Table 3). Brood year 2006 is incomplete because it is missing the age-5 river harvest and escapement components and therefore the potential SRR is expressed as a range of possible values. Potential SRR values are very low for this brood because nearly all ocean salmon fisheries were closed in 2008 and 2009, when this brood would be vulnerable as age-3 and age-4, respectively. Furthermore, the range of potential SRR is very small for this brood owing to the the very small contribution of age-5 spawners; reconstructed ocean abundances are either very small or zero after age-4 escapement (Appendix D). The 2007 brood is also incomplete, with no estimates of age-4 and age-5 river harvest and escapement. Since assumptions must be made for age-3 and age-4 maturation rates, as well as the unobserved age-4 and age-5 ocean harvest, the range of possible SRR for this cohort is larger.

Annual impact rates (i_{oa}), estimated using equation (16) for age-3 and age-4 SRWC, are displayed in Table 3 and Figure 4. When ocean fisheries have been open (2000–2007), age-3 impact rates have ranged from 10.34% to 25.59%, with little obvious trend. In contrast, age-4 impact rates have been much more variable and can be quite high. Substantial uncertainty exists for the

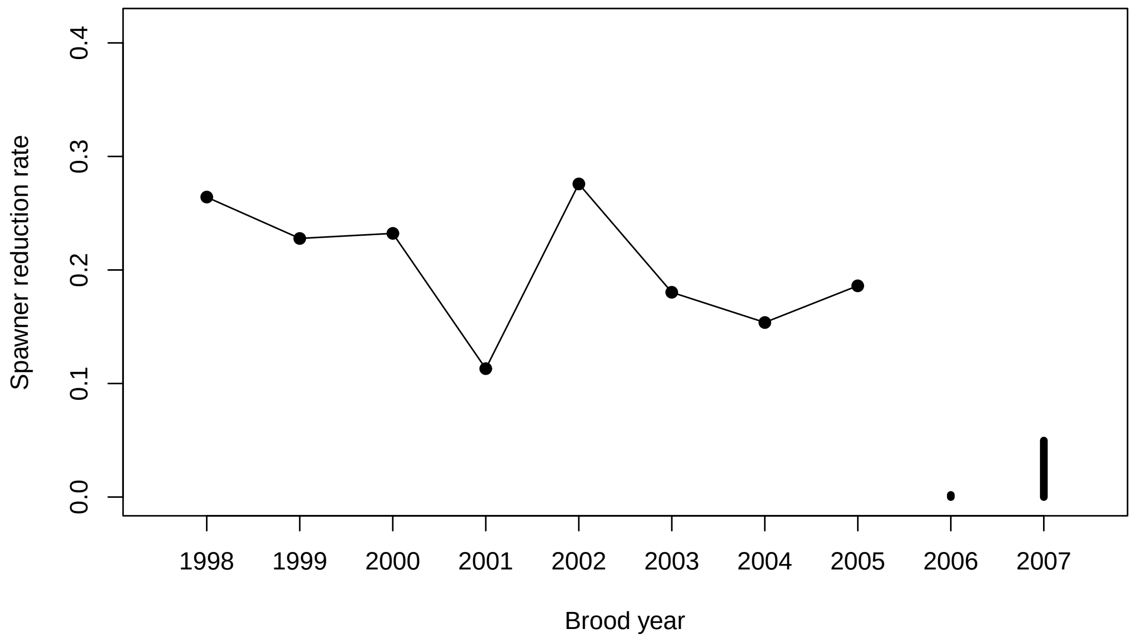

Figure 3. The spawner reduction rate for brood years 1998–2007. Brood years 2006 and 2007 are incomplete and estimates are expressed as a range of potential outcomes.

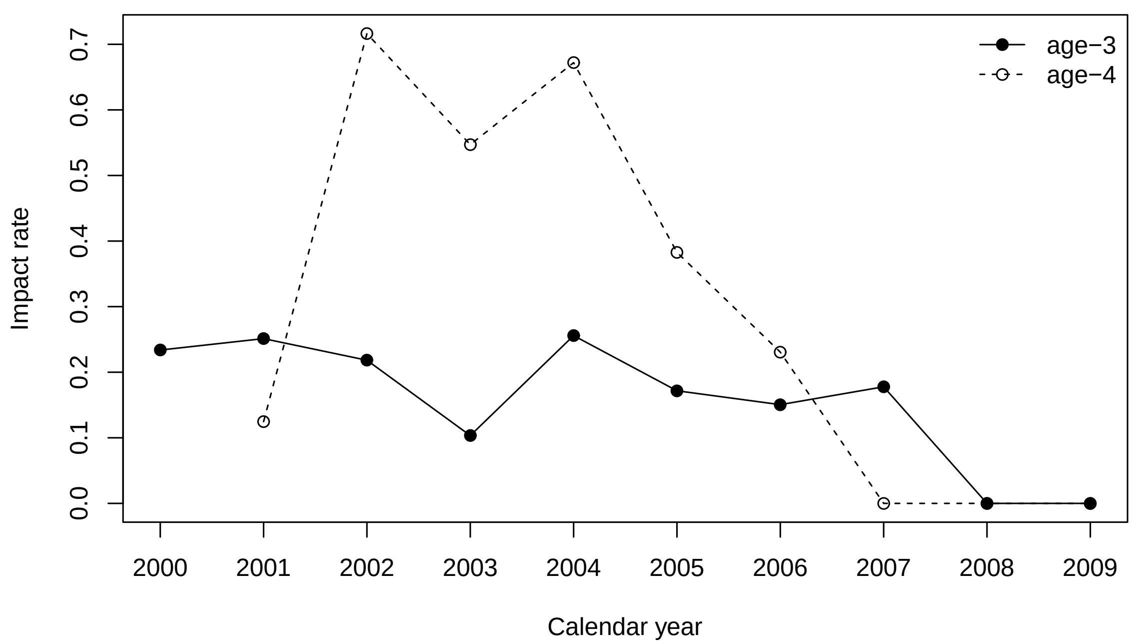

Figure 4. Ocean fishery impact rates for age-3 and age-4 plotted by calendar year.

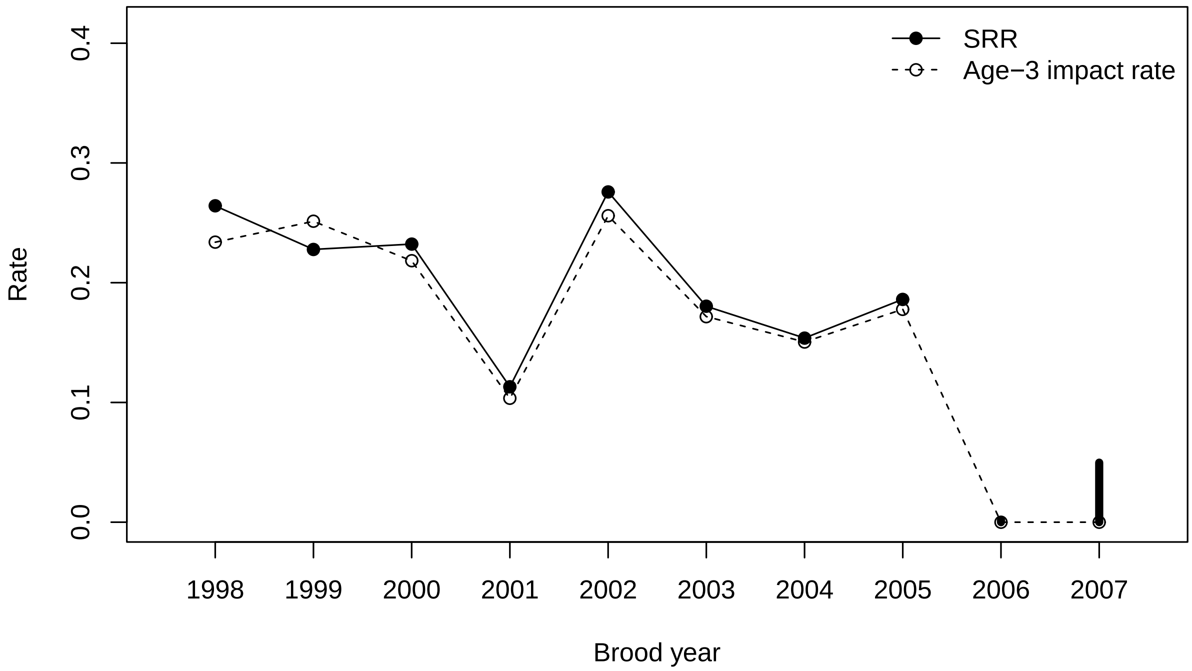

Figure 5. The spawner reduction rate (SRR) and age-3 impact rate plotted by by brood year.

age-4 impact rate estimates owing to the very low numbers of CWT recoveries that contribute to these estimates. Note that such high impact rates do not translate into very large age-4 impacts (Figure 1). This result can be explained by the low age-4 abundance of SRWC, a byproduct of the high age-3 maturation rate. The high age-3 maturation rate also suggests that the age-3 impact rate and the SRR should be concordant. Figure 5 demonstrates this to be the case as the trend and actual values of the SRR and the age-3 impact rate (here plotted by brood year) coincide with each other.

Stratifying the instances of nonzero impact rates by fishery, month, and management area (i_{oatzx}), enables additional inference about how ocean fisheries have affected SRWC. Figure 6 displays these rates (estimated by equation (15)) for the recreational fishery. Very few CWT recoveries from age-3 SRWC exist from management areas north of SF, resulting in few estimates of nonzero impact rates in these areas. Zero CWTs from age-4 SRWC were recovered in areas north of SF. For age-3, the bulk of the CWT recoveries and highest impact rates occur in the SF and MO areas between the months of April and July. Age-4 impacts are also clustered in the SF and MO areas and estimated age-4 impact rates can be much higher than age-3 impact rates (note y-axis scale differences between age-3 and age-4). Note also that very few CWT recoveries contributed to these age-4 estimates, as indicated by color coding of the impact rate estimates. For the commercial fishery (Figure 7), a similar spatiotemporal pattern is observed, yet with fewer nonzero impact rate estimates. Impacts north of SF are rare or absent. Nonzero age-3 impact rates are observed in SF and MO, with the highest rates observed from June–August. This pattern holds for age-4, though estimates are relatively sparse.

The relationship between the age-3 ocean fishery impact rates and fishing effort for recreational and commercial fisheries is displayed in Figures 8 and 9, respectively. A zero-intercept linear model representing the average impact rate per unit of effort was fit to these estimates using the ratio estimator, $\beta_{o(3)tzx} = avg\{i_{o(3)tzx}\}/avg\{f_{otzx}\}$, where f denotes fishing effort and the average is over years. Effort in recreational fisheries is defined as *angler days*, while effort in the commercial fishery is defined as *boat days*. Since the two effort metrics are not equivalent, comparisons of fishing effort between recreational and commercial fisheries are not valid. For the recreational fishery, it is clear that the highest impact rates per unit of fishing effort occur in the SF and MO areas. Recreational fishing effort was comparable between SF and MO through the month of June. After June, the SF region has experienced higher effort relative to MO, though impact rates tend to be low in SF after August. We note that recreational fisheries do not continue to operate in February and March and therefore SRWC are not currently "sampled" by the fishery in these months. The sparsity of data points in March indicate how infrequently fisheries have operated in this month for the cohorts examined here. A similar pattern to the one described above exists for the commercial fishery. It is clear that for this fishery, the highest age-3 impact rates per unit effort are clustered in the SF and MO areas from June–August.

Cohort reconstructions rebuild the abundance of SRWC broods to age-2, March 1. Using the age-2 March abundance ($N_{o(2)(Mar1)}$)and the number of hatchery SRWC released from LSNFH (N_{rel}), it is possible to estimate an early life survival rate ($N_{o(2)(Mar1)}/N_{rel}$). The early life survival rate includes all sources of mortality, both in the river and the ocean from hatchery release to age-2 in the ocean. This survival rate is most likely completely independent of ocean fishery sources of mortality as SRWC prior to age-2 are unlikely to be contacted by ocean fisheries. Estimates of

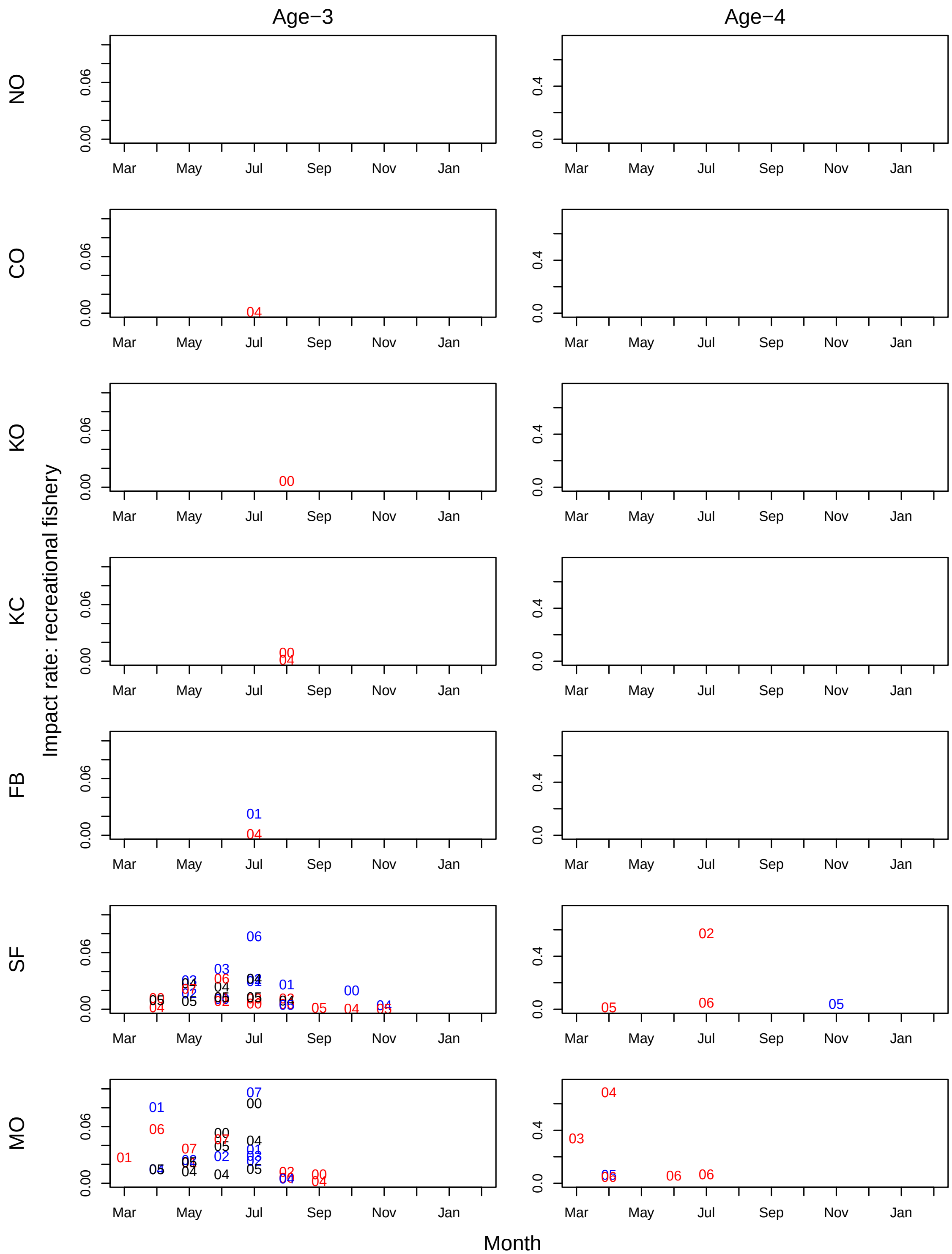

Figure 6. Recreational fishery impact rates for age-3 and age-4 Sacramento River winter Chinook, by month and management area, plotted for instances when rates are nonzero. Two-digit values in plots represent calendar years. Red text indicates that the impact rate estimate is the result of one coded wire tag (CWT) recovery. Blue represents two–five CWT recoveries. Black indicates greater than five CWT recoveries contributed to the estimate.

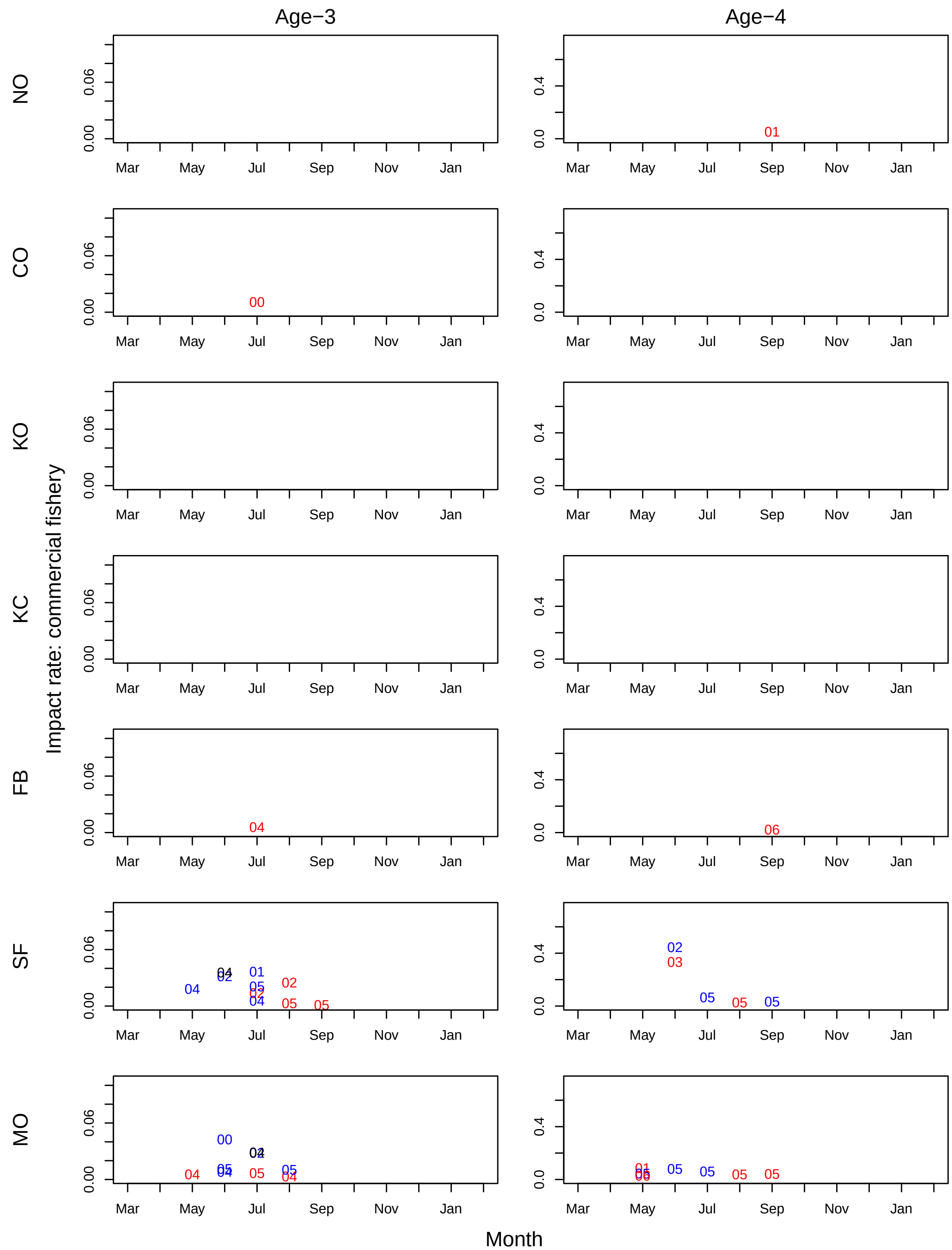

Figure 7. Commercial fishery impact rates for age-3 and age-4 Sacramento River winter Chinook, by month and management area, plotted for instances when rates are nonzero. Two-digit values in plots represent calendar years. Red text indicates that the impact rate estimate is the result of one coded wire tag (CWT) recovery. Blue represents two–five CWT recoveries. Black indicates greater than five CWT recoveries contributed to the estimate.

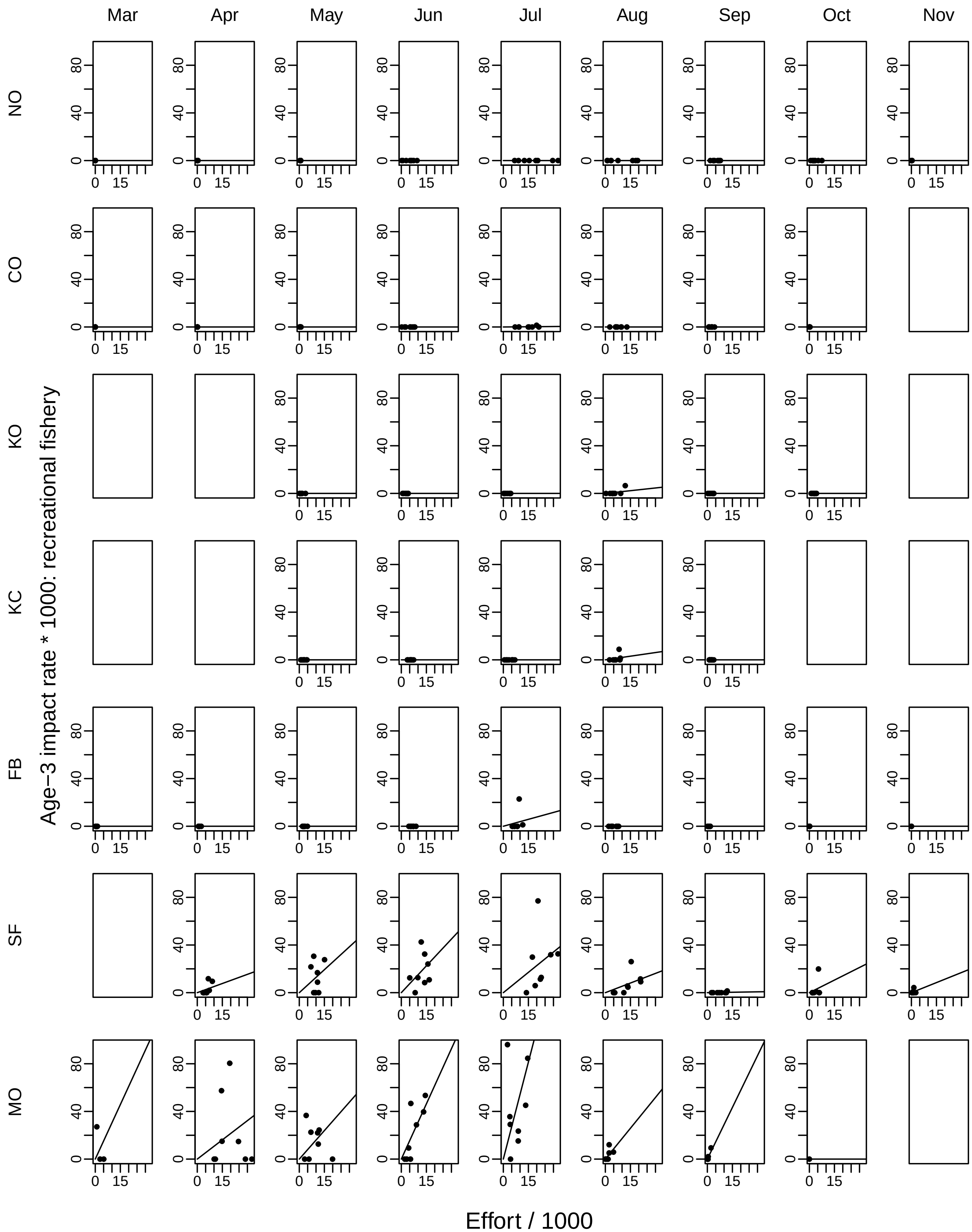

Figure 8. Recreational fishery age-3 impact rates, plotted as a function of fishing effort, by month and management area. Each point represents one year of estimates.

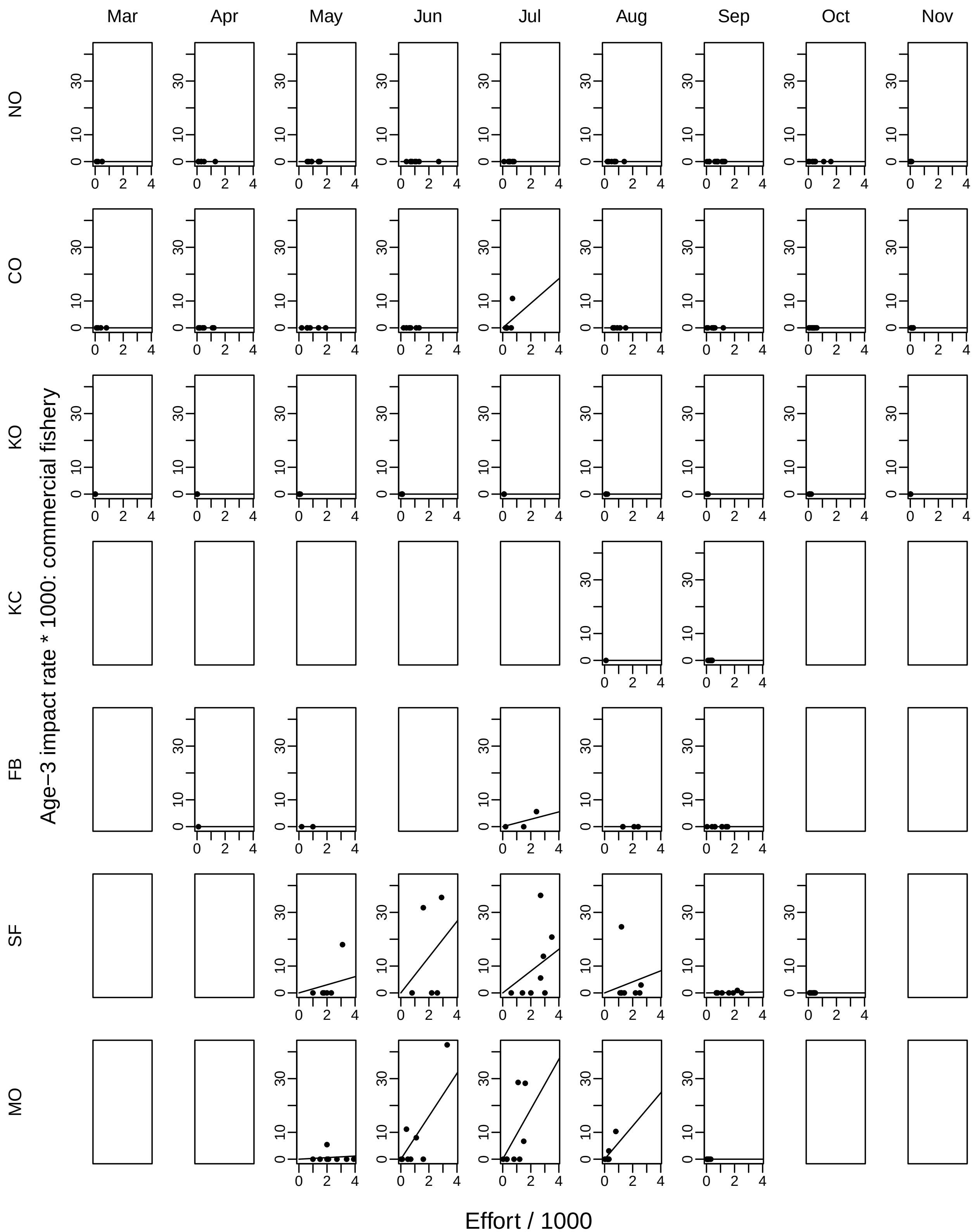

Figure 9. Commercial fishery age-3 impact rates, plotted as a function of fishing effort, by month and management area. Each point represents one year of estimates.

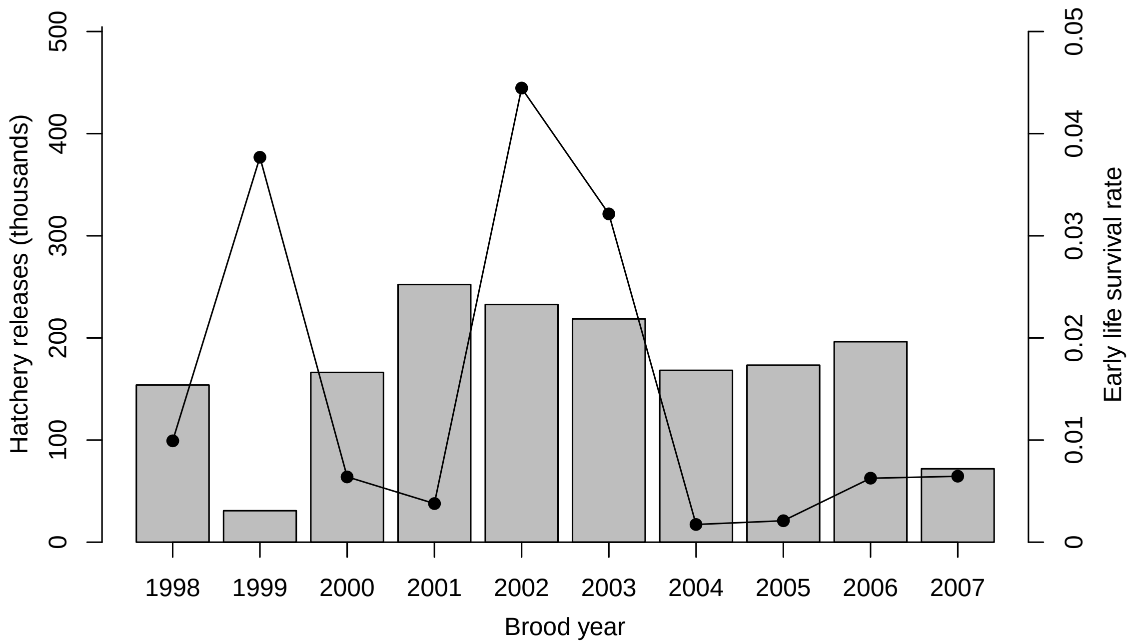

Figure 10. The number of Sacramento River winter Chinook released from the hatchery (bars) and the early life survival rate (line) for brood years 1998–2007.

early life survival, and the number of SRWC released from the hatchery, are presented in Figure 10. With the exception of brood years 1999 and 2007, hatchery release numbers have been fairly consistent. Conversely, early life survival estimates have varied considerably. The highest survival rates occurred for brood years 1999, 2002, and 2003. The relatively high survival rates for the 2002 and 2003 broods coincided with relatively high levels of hatchery releases. These broods in turn incurred the relatively high age-3 ocean impacts observed in 2004 and 2005 (Figure 1). These results suggest that the relatively high age-3 impacts observed in 2004 and 2005 were the result of relatively good early life survival and slightly higher than average hatchery releases, since the age-3 ocean fishery impact rate varied little over the 2000–2007 period (Figure 4).

5 Discussion

The estimation of hatchery-origin SRWC harvest and impacts for brood years 1998–2007 allows for comparison of harvest estimates made at various times since brood year 1969. Furthermore, cohort reconstructions and the estimates of maturation rates, the SRR, and ocean fishery impact rates derived from these reconstructions can be compared to estimates derived from the prior cohort analysis performed in support of the 2004 Biological Opinion (Grover et al. 2004).

With the exception of the 2004 cohort reconstructions completed for brood years 1998–2000, past work using ocean fishery data has focused on the estimation of SRWC ocean harvest by time and area. Figure 11 displays ocean harvest estimates by month for (a) fin-clipped marked SRWC from the pooled 1969–1970 broods (b) tagged SRWC from CNFH in pooled brood years 1991–1995, and (c) the hatchery-origin harvest of pooled brood years 1998–2007 considered in this report. Examination of estimates from the three time periods allows for some inference regarding the effect that ocean fishery regulations have had on the SRWC stock. Recreational ocean fisheries that contacted the 1969–1970 cohorts opened in mid February in areas south of Point Arena. The relatively high marked SRWC ocean harvest estimates from February and March (28% of the total estimated harvest) noted for these broods was largely absent in the harvest estimates for broods 1991–1995 and 1998–2007, owing to regulatory measures requiring salmon fisheries south of Point Arena to be closed for much of February and March for the express purpose of protecting SRWC. Since 2004, the recreational fishery south of Point Arena has been closed the entire month of February and March. As a result, the small or nonexistent recent estimates of harvest in February and March are reflective of regulations that have constrained the fisheries in these months. Given the SRWC run timing, with peak returns of mature adults to the river mouth in March, and the temporal distribution of harvest estimates from the 1969–1970 broods, it would be reasonable to expect that SRWC fishery impacts would be significant in February and March if fisheries were again allowed during that time frame.

For SRWC, the pattern of ocean fishery impacts, the SRR, impact rates, and maturation rates has maintained a consistent pattern. Ocean impacts are dominated by age-3, are taken primarily in the recreational fishery, and are nearly all the result of fisheries in areas south of Point Arena. The SRR and the age-3 annual impact rate have been consistent with each other and have ranged from approximately 10% to 28%. One reason for the consistency between the SRR and the age-3 impact rate is the high ($> 85\%$) and stable age-3 maturation rate. The bulk of the CWT recoveries that contributed to impact rate estimates were recovered in MO and SF, from April–July, and in

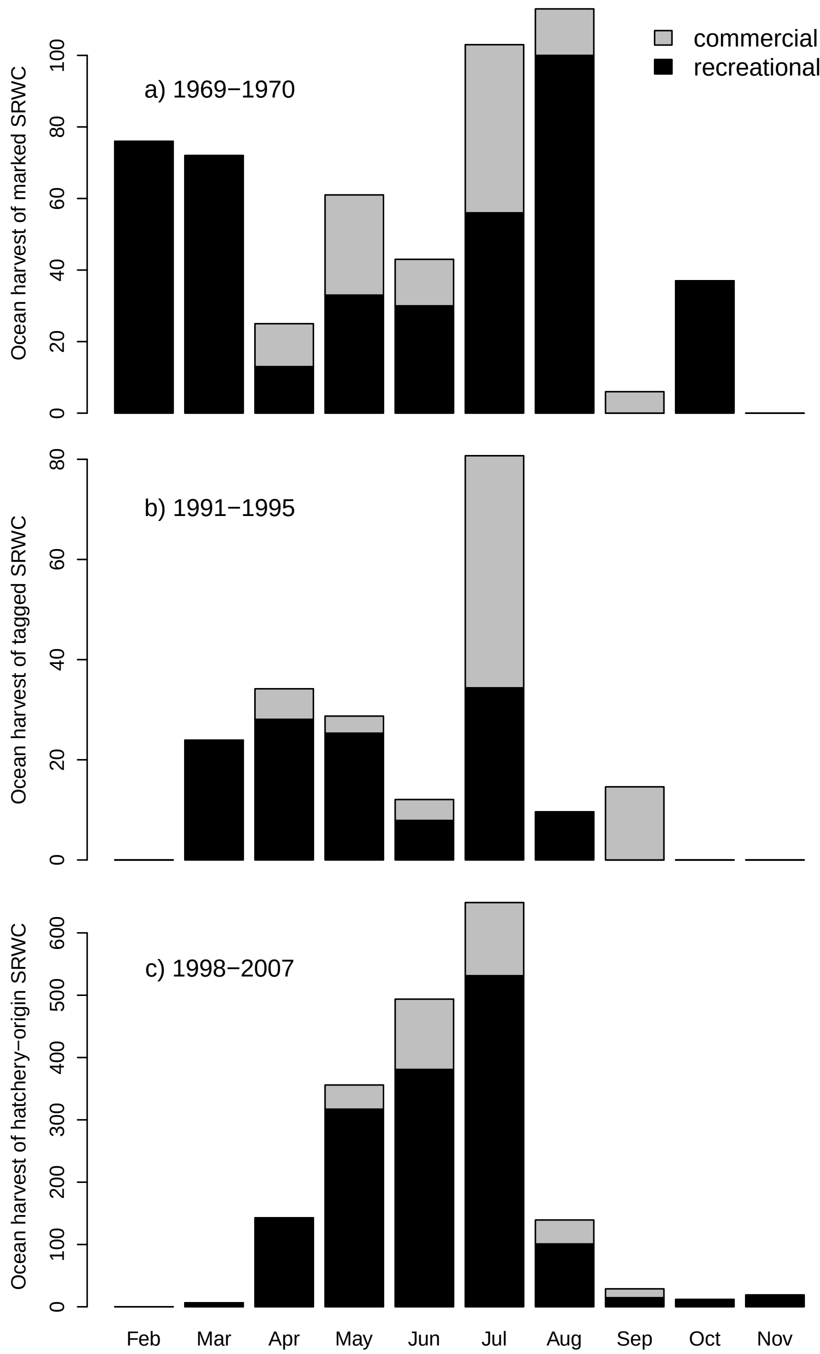

Figure 11. Estimated harvest of a) marked SRWC from pooled brood years 1969–1970, b) coded wire tagged SRWC from pooled brood years 1991–1995, and c) hatchery-origin SRWC from pooled brood years 1998–2007.

the recreational fishery. Fewer CWTs were recovered for age-4 and the commercial fishery. In particular, very few CWTs were recovered in ocean management areas north of Point Arena. All of these results are consistent with those presented in the Grover et al. (2004) report for a subset of the brood years considered here.

One weakness of using fishery-dependent CWT data for cohort analysis is that impacts on tagged fish may be underestimated for subsets of the population that are not retained in fisheries. This is the case for age-2 SRWC, which are too small to be retained in salmon fisheries with minimum size limits, yet may incur release and dropoff mortality. The data available do not allow for quantification of the magnitude of these mortalities. However, we note that errors in fishing impacts on age-2 would only affect the estimate of the SRR and the age-2 maturation rate. Age-3 and 4 impact and maturation rates would not be affected by additional age-2 mortality than what is accounted for herein.

Figure 1 in this report demonstrates that age-3 fishery impacts for hatchery-origin SRWC were much greater in calendar years 2004 and 2005 relative to other years, and these relatively high levels of impacts were observed in 2005 and 2006 for age-4. Coupling this information with the results presented in Figure 10 suggests that pre-fishing recruitment of the 2002 and 2003 hatchery-origin broods was relatively strong, and that this resulted from normal to high levels of hatchery releases and relatively high early life survival rates. The strength of these broods was clearly apparent in the associated estimates of ocean impacts and spawning escapement. Similarly, for the SRWC stock composite (hatchery- and natural-origin), the two highest spawner escapement estimates over the analysis period were in 2005 and 2006 (Figure 12), corresponding primarily to brood years 2002 and 2003, respectively. Together, these results suggest that early life survival (pre-fishery) plays a strong role in determining SRWC realized ocean abundance, ocean fishery impacts, and spawning escapement.

Spawner escapement of SRWC has experienced a precipitous decline, very low abundances, and more recently, a modest increase and subsequent decline (Figure 12). During the period of steep declines (1970 through the early 1980s) it was likely that ocean fishery impact rates were higher than they were after the early 1990s because recreational fishing seasons commenced in

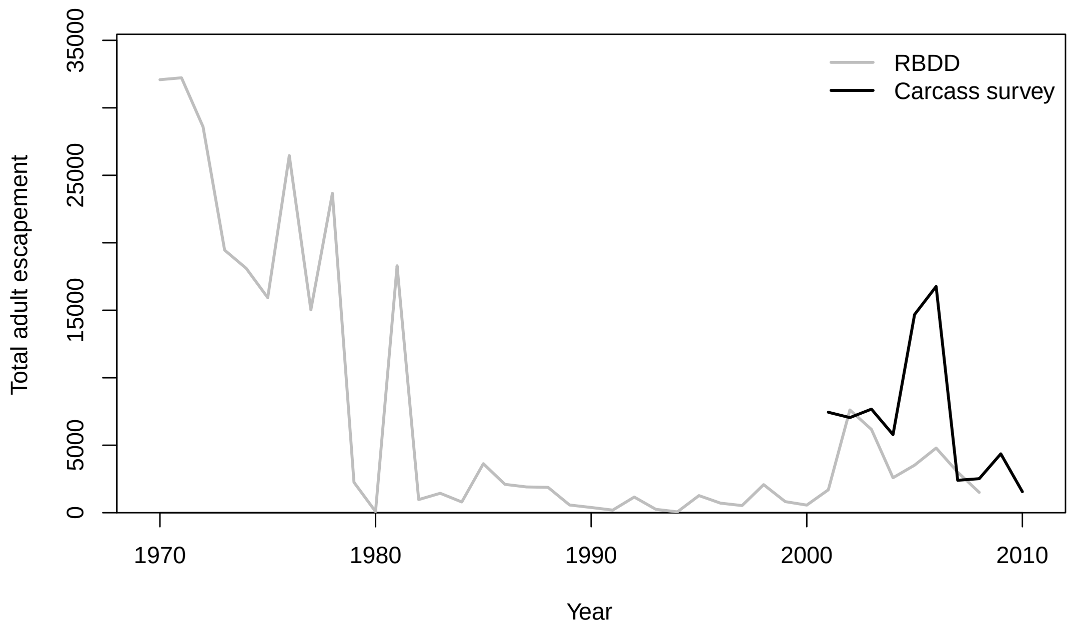

Figure 12. Escapement of combined natural and hatchery-origin adult Sacramento River winter Chinook. The grey line represents estimates based on counts at Red Bluff Diversion Dam (RBDD). The black line represents estimates based on the carcass survey. Carcass survey escapement estimates are considered to be of higher quality than RBDD estimates and are used to determine the "official" SRWC escapement.

February, when impacts would likely be high. During the end of the period with very low escapement (the 1990s) fisheries began to be contracted, with little to no recreational fishing occurring in SF and MO in February or March and restrictions on commercial fisheries owing to conservation concerns for other stocks. In the time since 2000, the period for which this cohort analysis has estimated exploitation rates, escapement has generally increased, with the exception of very recent years. This modest increase has occurred as SRRs have remained relatively stable. Finally, commercial and recreational fishing was closed in the SF and MO areas in 2008 and 2009, hence spawners in 2009 and 2010 were exposed to little or no fishing mortality. Despite fisheries closures, the spawner escapement has decreased in recent years relative to the early to mid 2000s. In sum, recent increases in escapement have occurred under a "typical" modern level of fishing, while the very recent decreases in escapement have occurred in spite of the closure of all salmon fisheries that typically contact SRWC.

Hatchery-origin SRWC make up a very small portion of the ocean salmon harvest off California and Oregon. Were it not for the 100% marking and tagging of LSNFH production, coupled with the ocean fishery and river escapement sampling programs' practice of processing the heads of all observed adipose fin clipped salmon for CWT extraction and decoding, it would not have been possible to conduct the cohort analyses described in this report—the recovery of SRWC CWTs would simply be too rare to support meaningful analysis and inference. Because of these programs, in core month and area strata, SRWC ocean fishery impact rate estimates based on multiple tag recoveries are common (e.g., see Figure 6). Tag recoveries are less frequent when ocean abundance of the hatchery-origin stock is low (such as for age-4) or outside of the core distribution of the stock (i.e., north of Point Arena). The raw CWT recovery pattern observed for SRWC imparts confidence in our core estimates.

6 Conclusions

Based on the results developed here, and those derived from earlier studies, we identify the following conclusions.

1. Cohort analysis results suggest stability in the SRR and ocean fishery impact rate estimates by management area, month, and fishery.

2. Changes in ocean fishing regulations that have limited or eliminated February and March recreational fisheries south of Point Arena, California, have likely been effective in reducing SRWC impacts. The estimates for marked SRWC from the pooled 1969–1970 broods approximate the temporal distribution of harvest that would be expected without these restrictions.

3. Early life survival (pre-fishery) plays a strong role in determining SRWC realized ocean abundance, ocean fishery impacts, and spawning escapement.

4. Recent increases, and subsequent declines, in SRWC adult escapement since 2000 cannot be readily explained by trends in ocean fishery exploitation rates.

5. Current SRWC tagging/marking and monitoring programs should be continued so that future updates of the cohort analysis can be made as these data accumulate. In particular, the collection of heads for CWT recovery from all adipose fin clipped salmon in ocean and river sampling programs should continue to receive high priority. Without such data and analysis, it is impossible to provide a direct estimate of the realized impacts of ocean salmon fisheries on SRWC.

7 Acknowledgements

We wish to acknowledge all those who contributed to the collection of the data used here in both ocean and river monitoring programs. In particular, thanks go to Melodie Palmer-Zwahlen (CDFG) for assistance with ocean CWT data. We sincerely thank Kevin Offill (USFWS) and Doug Killam (CDFG) for answering our questions about the SRWC escapement carcass survey, and for readily providing the data necessary for us to develop estimates of the effective sampling fraction for the survey's CWT recoveries. Our cohort reconstruction work would not have been possible without their cooperation and contribution. Finally, we thank three reviewers from the Center for Independent Experts (Mike Bradford, Marc Labelle, and David Levy) for their thoughtful comments on a previous version of this report.

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Appendix A Proportion legal size

A.1 Introduction

Most ocean salmon fisheries have a minimum size (length) limit provision. Salmon below this minimum size limit must be released while salmon larger than the limit can be retained for harvest. Data on the number of released fish are generally not available, particularly at the individual stock level, yet this information is needed to account for all sources of mortality since some released fish will die. To estimate the proportion of fish that were greater than or equal to the minimum legal size in each year, month, area, and fishery, we utilize a length-at-age model and the minimum size limit in place for that particular year/month/area/fishery.

Previous cohort reconstructions used a length-at-age model developed for this purpose by CDFG (1989). The model is age- and month-specific and was constructed by using adult river recoveries of fin-clipped broods (1969–1970) to estimate the mean length of age-2 and age-3 spawners, which was assumed to be representative of ocean fish as well. Linear interpolation of these ocean mean length-at-age "endpoints" was then used to derive mean length for the in-between months, with a further assumption that 50 percent of the annual increase in length-at-age occurs during the April–June period. Individual lengths-at-age were assumed to be normally distributed with a constant coefficient of variation which was estimated from adult river recoveries of Sacramento River fall Chinook CWT broods (1975–1978).

Since the time of the CDFG (1989) model formulation, a CWT program for Sacramento River winter Chinook has been established by Livingston Stone National Fish Hatchery. With these data we have developed a new length-at-age model for Sacramento River winter Chinook as described below.

A.2 Data

The RMPC database[2] was queried for all available Sacramento River winter Chinook CWT recoveries from recreational and commercial ocean fisheries off the coast of California and Oregon. Recoveries were screened to include only fish with a fork length measurement, and this yielded a dataset of 507 observations, of which 6 were in 1980 and the remainder spanned calendar years 1993–2007, with no recoveries in 1998. Recorded fork length (FL), measured in mm, was converted to total length (TL) in inches using the equation (M. Palmer-Zwahlen[3], personal communication, 2011)

$$TL = 1.04346 + (0.04096 \cdot FL), \tag{A-1}$$

and individual fish were assigned to management area based on the port of landing. The minimum size limit l^* associated with the year, month, area, and fishery in effect at the time of recovery was also determined for each fish. The ageing convention used for ocean recoveries was the same as that used in the cohort reconstruction, with a "birthday" of March 1 (age increments by one year on March 1). Finally, days-at-age of recoveries (number of days between recovery date and previous March 1) was calculated for each fish. No fish were recovered during the December–February period or exceeded four years of age.

Fisheries with minimum size limits provide a truncated sample of the ocean size distribution, and an analysis of size-at-age must take this truncation into account. Of the 507 fish in this dataset, 486 were at or above the legal size limit in effect at their time and place of capture. Our analysis was limited to these fish. However, we found that including fish as much as 0.5 inches below the minimum size limit (to account for possible measurement error) only changed our mean length estimates for March at each age by a maximum of 0.012 inches.

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[2]http://www.rmpc.org/

[3]Melodie Palmer-Zwahlen, CDFG, Ocean Salmon Project, 475 Aviation Blvd, Suite 130, Santa Rosa, CA, 95403.

A.3 Model

Our model assumes that length-at-age (l) on a particular day is normally distributed with mean μ, standard deviation σ, and probability density

$$f(l|\mu,\sigma) = \frac{1}{\sqrt{2\pi}\sigma} e^{-(l-\mu)^2/2\sigma^2}. \tag{A-2}$$

We further assumed a constant daily mean growth rate (g) and coefficient of variation $(CV = \sigma/\mu)$, so that at τ days-at-age the mean length and standard deviation in length are given by

$$\mu_{a,\tau} = \mu_{a,0} + g_a \cdot \tau, \tag{A-3}$$

$$\sigma_{a,\tau} = CV_a \cdot \mu_{a,\tau}, \tag{A-4}$$

where $\mu_{a,0}$ is the mean length of age-a fish on March 1 (day 0)[4]. This model was assumed to apply independently to age-3 and age-4 fish over the March–November period (the period for which CWT recovery data exist). For completeness, the above model was extended to the intervening age-3 December–February period by assuming a constant daily mean growth rate between these two mean "end points". We did not model age-4 length-at-age beyond the month of November (no CWT fish this old or older have been recovered).

Given this model, the proportion of fish-at-age greater than or equal to a particular minimum size limit (l^*) is

$$P\{l \geq l^*|\mu,\sigma\} = 1 - P\{l < l^*|\mu,\sigma\} = 1 - \int_{-\infty}^{l^*} f(l|\mu,\sigma)dl = 1 - \Phi(l^*|\mu,\sigma), \tag{A-5}$$

where $P\{A\}$ denotes the probability of event A, and $\Phi(\cdot)$ is the cumulative probability distribution function for the normal density $f(\cdot)$.

[4] We increment the age on March 1 at 12:00 A.M., but set $\tau = 0$ at 12:00 P.M. (day midpoint) to better reflect the capture (recovery) process.

Table A-1. Maximum likelihood estimates for March–November length-at-age model parameters.

Age (a)	$\mu_{a,0}$	g_a	CV_a
3	20.2372	0.0355	0.0820
4	28.5064	0.0317	0.0868

A.4 Estimation

The parameters of the length-at-age model were estimated using the method of maximum likelihood. Because of the minimum size limit, the sampling density for the length-at-age of a recovery is truncated at l^*, and is therefore given by equation (A-2) normalized over the observable range (Goldwasser et al. 2001),

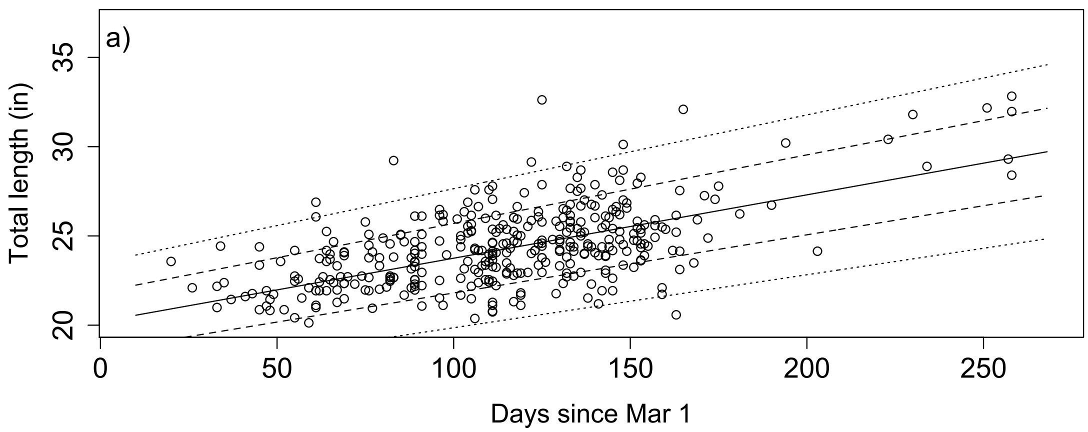
$$f(l|l \geq l^*, \mu, \sigma) = \frac{f(l|\mu, \sigma)}{P\{l \geq l^*|\mu, \sigma\}} = \frac{f(l|\mu, \sigma)}{1 - \Phi(l^*|\mu, \sigma)}. \tag{A-6}$$

The likelihood function for each age ($\mathscr{L}_a$) is the joint density over the recoveries, viewed as a function of the parameters conditional on the data. For an individual recovery i the data are $\{l_i, l_i^*, \tau_i\}$ and the likelihood over the n_a recoveries is thus

$$\mathscr{L}_a(\mu_{a,0}, g_a, CV_a|\{l_i, l_i^*, \tau_i\}) = \prod_{i=1}^{n_a} f\left(l_i|l_i \geq l_i^*, [\mu_{a,0} + g_a \cdot \tau_i], CV_a[\mu_{a,0} + g_a \cdot \tau_i]\right). \tag{A-7}$$

The maximum likelihood estimates for age-a are those values of $\mu_{a,0}$, g_a, and CV_a that together maximize the $\mathscr{L}_a$ function. This was found numerically with the R (R Development Core Team 2011) statistical computing software using the functions "dnorm" to calculate $f(\cdot)$ and "pnorm" to calculate $\Phi(\cdot)$, respectively. Numerical optimization was performed using function "mle2" (Bolker 2010). The resulting parameter estimates are given in Table A-1, and the fitted model is displayed with the observations in Figure A-1.

For the age-3 December–February period, day-specific mean length was modeled by linearly



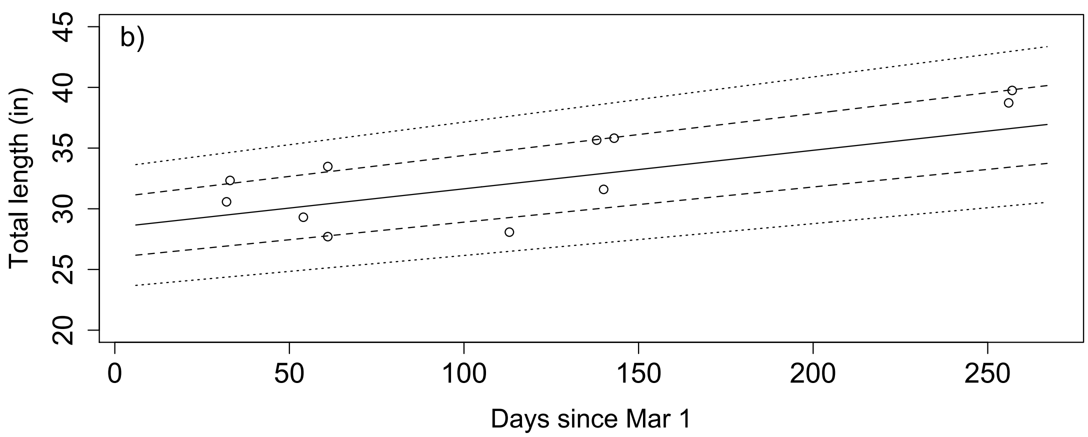


Figure A-1. Fitted March–November length-at-age model for a) age-3 and b) age-4 fish. Solid line is the estimated mean length-at-age; dashed lines represent one and two standard deviations from the mean. Only observations from fisheries with a 20 inch minimum size limit are shown for the clearest interpretation of model fit.

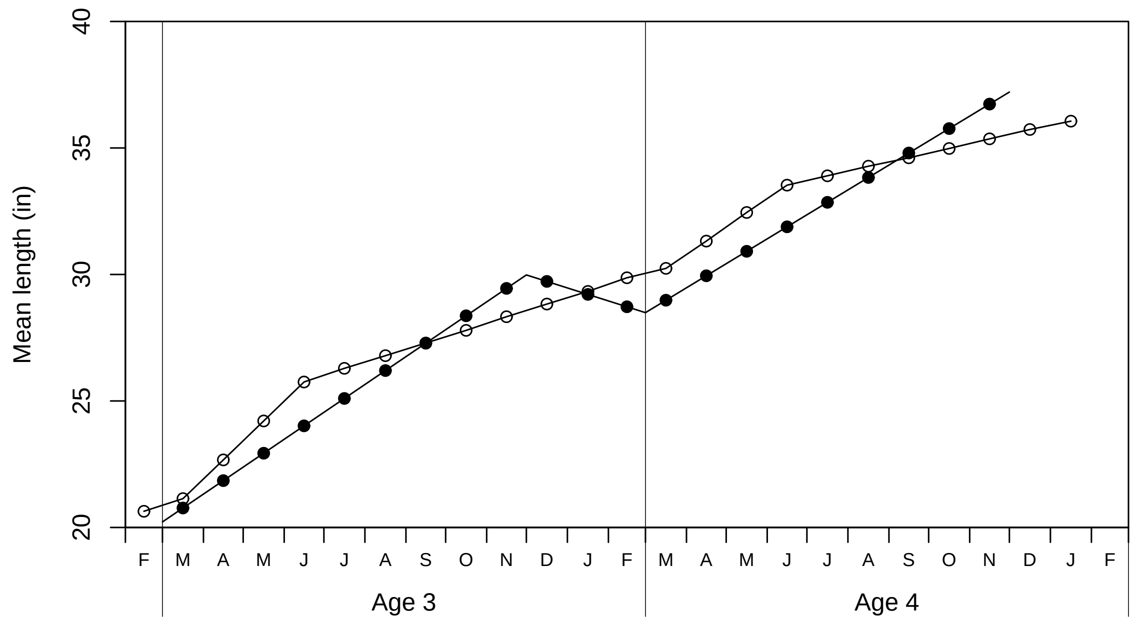


Figure A-2. Estimated mean length-at-age by our model (solid circles) and that specified by CDFG (open circles). For our model, the line traces the daily mean values and the solid circles are the midpoint values reported in Table A-2. For the CDFG model, the line connects their monthly mean values (CDFG 1989, Table 2) (open circles) plotted at the monthly midpoint.

interpolating between the December 1 and March 1 mean lengths[5]:

$$\mu_{3,\tau} = \mu_{3,\tau(\text{Dec 1})} + \left(\mu_{4,\tau(\text{Mar 1})} - \mu_{3,\tau(\text{Dec 1})}\right)\left(\frac{\tau - \tau(\text{Dec 1})}{\tau(\text{Mar 1}) - \tau(\text{Dec 1})}\right), \quad \tau(\text{Dec 1}) \leq \tau \leq \tau(\text{Mar 1}). \tag{A-8}$$

Figure A-2 displays our estimated mean length-at-age relationship for age-3 and age-4 fish, and the monthly midpoint values are listed in Table A-2 along with the corresponding standard deviation and proportion legal size, assuming minimum size limits typical for the recreational (≥ 20 inches, ≥ 24 inches) and commercial (≥ 26 inches) fisheries. For age-4 fish beyond November, and all older age fish, it is assumed that the proportion legal size equals one, noting that an estimated 99.69% of age-4 fish in November exceed 28 inches in total length.

[5](at 12:00 A.M.)

Table A-2. Length-at-age model and proportion legal size at the midpoint of each month (t) over the modeled period. Mean and standard deviation is for total length in inches. Proportion legal size was computed using equation (A-5). The 20 and 24 inch minimum size limits are typical for the recreational fishery, and the 26 inch minimum size limit is typical for the commercial fishery.

Age (a)	Month (t)	μ_{at}	σ_{at}	$P\{l \geq 20 \text{ in}\}$	$P\{l \geq 24 \text{ in}\}$	$P\{l \geq 26 \text{ in}\}$
3	Mar	20.7697	1.7029	0.67	0.03	0.00
3	Apr	21.8524	1.7917	0.85	0.12	0.01
3	May	22.9351	1.8804	0.94	0.29	0.05
3	Jun	24.0178	1.9692	0.98	0.50	0.16
3	Jul	25.1004	2.0580	0.99	0.70	0.33
3	Aug	26.2009	2.1482	1.00	0.85	0.54
3	Sep	27.2836	2.2370	1.00	0.93	0.72
3	Oct	28.3663	2.3257	1.00	0.97	0.85
3	Nov	29.4490	2.4145	1.00	0.99	0.92
3	Dec	29.7247	2.4371	1.00	0.99	0.94
3	Jan	29.2111	2.3950	1.00	0.99	0.91
3	Feb	28.7224	2.3549	1.00	0.98	0.88
4	Mar	28.9820	2.5155	1.00	0.98	0.88
4	Apr	29.9490	2.5994	1.00	0.99	0.94
4	May	30.9161	2.6834	1.00	1.00	0.97
4	Jun	31.8831	2.7673	1.00	1.00	0.98
4	Jul	32.8502	2.8512	1.00	1.00	0.99
4	Aug	33.8331	2.9366	1.00	1.00	1.00
4	Sep	34.8001	3.0205	1.00	1.00	1.00
4	Oct	35.7672	3.1044	1.00	1.00	1.00
4	Nov	36.7343	3.1884	1.00	1.00	1.00

A.5 Discussion

Recoveries were not equally distributed among months, with the majority occurring during the May–July period. While this period is the most important one to model for the purpose of cohort reconstruction (since it is when most harvest occurs), we explored the potential impacts of uneven temporal representation on the model parameter estimates for age-3 fish by fitting the model to bootstrapped replicate datasets consisting of: (a) 35 samples for each month April–August (all of which had at least 35 recoveries); and (b) five samples each month March–November (except for October, which had only three data points). In both cases, the March 1 length estimated by the complete dataset was close to the mode of the bootstrapped estimates. Fitting the April–August data implied slightly slower growth (with an approximate 0.7 inch difference by the end of November) than the full dataset, while fitting the data from March–November implied faster growth by a similar amount. Thus, our fit to the full dataset seems appropriate.

We evaluated alternate models of growth allowing for lognormally distributed individual lengths, exponential growth in length, or von Bertalanffy growth in length. However these alternative formulations did not substantially improve model fit (or decreased it in the case of von Bertalanffy) and yielded very similar predictions of the proportion of the population above minimum size limits for the various fisheries. There was little evidence for seasonal variation in growth rate (Figure A-1) and a comparison of size-at-age curves for other Central Valley Chinook runs (spring, fall, and late-fall) with more data available suggested that seasonal variation in size-at-age was mostly driven by the timing of return to freshwater by spawning adults. Apparent growth in mean size slows, stops, or even reverses during this period (as we found for winter Chinook, Figure A-2), likely owing to the preferential loss of large fish at age to spawning (Healey 1991; Morita et al. 2005). Thus the assumption of linear growth within age classes during the non-spawning period when the fishery is operational, and linear interpolation over the intervening period, is well supported.

Limited sample sizes and uneven temporal and spatial coverage limited our ability to model the effects of year or ocean management area. Had we been able to include such effects, we would likely have estimated slightly different means for each year/location, with a smaller standard deviation around that year/location's estimated mean. Thus in any one year, we might expect the standard deviation in length to be smaller than that implied by our model which excludes year or spatial effects. However, without an ability to predict the mean for a given year or location, our approach provides a simple method of averaging over our uncertainty about year and location effects. Inspection of data for other Chinook stocks did show a smaller mean size-at-age for fall and late-fall Chinook from the Central Valley during 1983, 1993, and 1998, which correspond to El Nino conditions in the ocean. Unfortunately, the SRWC CWT dataset does not include data from these years to allow for such a comparison.

Our fitted model of SRWC size-at-age was not radically different from CDFG (1989), but there were some small yet potentially important differences in estimated mean size, estimated variation in individual sizes, and resultant proportion of the population that can be legally retained. Our estimated mean lengths of age-3 fish in March were 0.37" smaller than CDFG's estimate, with the difference growing to 1.73" by June and then shrinking with our model predicting larger fish by October due to CDFG (1989)'s assumption that 50 percent of growth occurs between April and June. The assumption of accelerated growth in spring does not appear to be well supported for SRWC (see below). In addition, CDFG assumed a larger coefficient of variation in mean length (0.107, based on SRFC data) than we fit for either age-3 (0.082) or age-4 (0.087) SRWC.

These different predictions of size-at-age result in different calculations of the proportion legal and thus change our estimates of fishery impacts when taking non-landed mortalities into account. Fishery impacts on SRWC are most significant for age-3 fish in May, June, and July. Due to the larger mean and standard deviation in fish sizes predicted for this time period by the CDFG (1989) model, it would predict a larger fraction of fish can be retained than our model does (e.g., 46 percent of age-3 fish legal sized with a 26" limit in June compared to 16 percent predicted by our model). Differences are also pronounced for a 26" limit in May (24 percent vs. 5 percent) and July (54 percent vs. 33 percent). The smaller fraction legal predicted by our model suggests more non-landed mortality per sampled fish, and thus increases our estimated total impact of the fishery. Predicted differences in a 20" limit recreational fishery are smaller since most fish are of legal size for either model (e.g. 94 percent legal in May according to our model and 95 percent according to

CDFG, both models predict 99 percent or more legal by July and predictions for March and April also agree within 2 percent). However, when the recreational fishery size limit is 24”, the situation becomes more like the 26” size limit evaluated for the commercial fishery. For age-4 fish, either model predicts a large fraction legal (at least 97 percent for a 26” size limit by May).

Appendix B Carcass survey sample expansion factor: derivation

Formulas for determining the *effective sampling fraction* (λ) for carcass survey decoded recoveries, and its inverse the *sample expansion factor* ($1/\lambda$), are presented in this Appendix. The sampling fraction λ is specific to the natural area SRWC carcass survey (it does not pertain to fish caught in the Keswick fish trap and used for hatchery broodstock). The sampling fraction λ is also year-specific, but it is not age-specific, not CWT code-specific, and not stock-specific[6]. Table B-1 provides a list of the notation used in the development of the CWT expansion formulas presented in this section.

Table B-1. Notation used to derive carcass survey sample expansion factor.

Symbol	Definition
E	natural area escapement (SRWC + strays)
E_{cwt}	number of E with CWT
R	number of E_{cwt} recovered and CWT decoded
λ	effective sampling fraction for CWTs
$p_{\text{ad-clipped}}$	proportion of E that is adipose fin clipped
p_{cwt}	proportion of E with CWT
$p_{\text{cwt}\|\text{ad-clipped}}$	proportion of E ad-clipped fish with CWT
n_{fresh}	number of fresh carcasses sampled in survey (SRWC + strays)
$n_{\text{fresh,ad-clipped}}$	number of n_{fresh} that are adipose fin clipped
$n_{\text{fresh,head-processed}}$	number of $n_{\text{fresh,ad-clipped}}$ heads processed for CWT detection
$n_{\text{fresh,cwt-detected}}$	number of $n_{\text{fresh,head-processed}}$ in which CWT was detected
$n_{\text{fresh,cwt-decoded}}$	number of $n_{\text{fresh,cwt-detected}}$ in which CWT was decoded

By definition, λ is equal to the number of decoded CWT sample recoveries divided by the

[6]Stray fall or late-fall CWT'd Chinook have occasionally (less than or equal to five per year) been recovered in the SRWC survey. These fish and their respective non-CWT counterparts are part of the overall pool of carcasses on which CWT recovery sampling is performed, and they are therefore included in the estimation of λ (as they are in the estimation of SRWC escapement).

number of CWT fish present in the escapement:

$$\lambda = R/E_{\text{cwt}}. \tag{B-1}$$

R is known from the survey, but E_{cwt} must be estimated and we do this by appealing to the product

$$E_{\text{cwt}} = E \times p_{\text{cwt}}, \tag{B-2}$$

where E is the natural area escapement, and p_{cwt} is the proportion of E that are CWT'd. For E we substitute the survey's Jolly-Seber estimate of overall natural area escapement. For p_{cwt}, because the probability of misclassification of ad-clipped status in a non-fresh carcass is appreciable (due to the carcass's deteriorated state), we restrict ourselves to the fresh carcass portion of the survey data and make the following three assumptions:

1. A fresh carcass that is adipose fin clipped (ad-clipped) may not have a CWT, but not vice-versa: a fresh carcass that has a CWT is ad-clipped. That is,

   $$P\{\text{CWT present} \mid \text{fresh, not ad-clipped}\} = 0,$$

   where $P\{A|B\}$ denotes the probability of event A given that event B occurs.

2. There is no misclassification of ad-clipped status for a sampled fresh carcass:

   $$P\{\text{classify carcass as ad-clipped} \mid \text{sampled, fresh, ad-clipped}\} = 1,$$
   $$P\{\text{classify carcass as not ad-clipped} \mid \text{sampled, fresh, not ad-clipped}\} = 1.$$

3. There is no CWT detection failure for a sampled fresh carcass whose head has been pro-

cessed for CWT recovery[7]:

$$P\{\text{CWT detected} \mid \text{sampled, fresh, head processed, CWT present}\} = 1.$$

We then re-express p_{cwt} as

$$p_{\text{cwt}} = p_{\text{ad-clipped}} \times p_{\text{cwt}|\text{ad-clipped}} \tag{B-3}$$

since $p_{\text{cwt}|\text{not ad-clipped}} = 0$ (assumption 1), and use the fresh carcass survey data to estimate these component proportions:

$$p_{\text{ad-clipped}} = \frac{n_{\text{fresh,ad-clipped}}}{n_{\text{fresh}}} \tag{B-4}$$

(assumption 2), and

$$p_{\text{cwt}|\text{ad-clipped}} = \frac{n_{\text{fresh,cwt-detected}}}{n_{\text{fresh,head-processed}}} \tag{B-5}$$

(assumption 3). In summary, equations (B-4) and (B-5) are used in (B-3) to estimate p_{cwt}, and this is multiplied by the Jolly-Seber estimate of natural area escapement E to estimate E_{cwt} (equation (B-2)), and λ is then estimated as R/E_{cwt} (equation (B-1)).

We note that while the non-fresh CWT decoded recoveries contribute to R, and hence the estimate of λ, the assumptions made above are not similarly required for the non-fresh portion of the survey data, in particular, assumptions 2 and 3. Thus, for sampled non-fresh carcasses, misclassification of ad-clipped status and CWT detection failure are not an issue with respect to the estimation of λ. Indeed, because it can be difficult to accurately determine whether a non-fresh carcass is ad-clipped, samplers are encouraged to collect heads for CWT processing from non-fresh carcasses considered to be "potentially" ad-clipped. The effect of this practice is to increase the magnitude of R which increases the effective sampling fraction λ, which in turn increases the precision of all cohort reconstruction derived quantities and estimates. This strategy implicitly assumes that the percent composition of CWT codes among fresh and non-fresh carcasses does not differ for the overall survey, but this is warranted given that all non-fresh carcasses were once

[7]The head of a fresh carcass is collected for CWT processing if and only if the carcass is ad-clipped.

fresh carcasses and that sampling is conducted throughout the SRWC spawning period.

The derivation of λ could be simplified if we limited our analysis entirely to the fresh carcass portion of the survey results, i.e. by excluding the non-fresh carcass CWT recoveries. For a fresh carcass only analysis, the CWT effective sampling fraction is simply the fraction of the escapement examined after adjusting for the fraction of ad-clipped carcass heads not processed and the fraction of detected CWTs not decoded[8]:

$$\frac{n_{\text{fresh}}}{E} \times \frac{n_{\text{fresh,head-processed}}}{n_{\text{fresh,ad-clipped}}} \times \frac{n_{\text{fresh,cwt-decoded}}}{n_{\text{fresh,cwt-detected}}}.$$

While restricting the analysis to the fresh carcass only data would not inherently bias the analysis results, it would substantially reduce the precision of the analysis by reducing the CWT effective sampling fraction (by a factor of $n_{\text{fresh,cwt-decoded}}/R$).

[8]The formulation of λ previously provided reduces to this product for a fresh carcass only data set, with $R = n_{\text{fresh,cwt-decoded}}$.

Appendix C Carcass survey sample expansion factor: data and derived values

The U.S. Fish & Wildlife Service (USFWS) and California Department of Fish and Game (CDFG) have co-operatively performed the SRWC spawning escapement carcass survey since 1996. CDFG has primary responsibility for the collection of information relevant to the estimation of spawning escapement. USFWS has primary responsibility for the collection of information relevant to the estimation of temporal/spatial/gender/age/length/origin-composition of the escapement, which includes the collection and processing of heads from carcasses for CWT recovery. The spawning escapement estimates (E) reported in this Appendix were provided to us by CDFG (D. Killam[9], personal communication, 2011). All other data reported in this Appendix were provided to us by USFWS (K. Offill[10], personal communication, 2011).

Summary data for the 2001–2010 surveys and the estimates resulting from application of our Appendix B formulas are presented in Table C-0 below. While the estimated escapement ranged from approximately 1500 to 17200 fish over the 2001–2010 period, the CWT effective sampling fraction was fairly consistent over the period, ranging from approximately 0.34 to 0.49 (except for 2007 when it reached 0.63). CWT expansion factors range from approximately 1.6 to 3.4 over the period, which is rather remarkable given the scope and complexity of the SRWC carcass survey. We note that, had the analysis been restricted to fresh carcass CWT recoveries only, this would have reduced the CWT effective sampling fraction by a factor ranging from 0.49 to 0.75.

The basic data and calculations that result in the Table C-0 values are presented in sections C.1–C.10 of this Appendix for survey years 2001–2010, respectively. An electronic file[11] of the data and estimates reported in this Appendix is available from the authors of this report.

[9] Doug Killam, CDFG, Red Bluff Field Office, P.O. Box 578, Red Bluff, CA, 96080.

[10] Kevin Offill, USFWS, Red Bluff Fish & Wildlife Office, 10950 Tyler Road, Red Bluff, CA, 96080

[11] SRWC.cwt.expansion.factors.NMFS.29jun2011.xls

Table C-0. Carcass survey CWT expansion summary data and results.

	Year									
Quantity	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010
					Survey					
R	117	141	125	164	1266	767	66	46	115	95
E	8120	7360	8133	7784	15730	17197	2487	2725	4416	1533
n_{fresh}	2235	2021	2423	1621	4177	3083	785	547	802	472
$n_{\text{fresh,ad-clipped}}$	116	108	138	140	840	440	48	34	91	74
$n_{\text{fresh,head-processed}}$	113	106	138	139	832	437	48	34	91	73
$n_{\text{fresh,cwt-detected}}$	92	81	91	97	699	385	33	27	72	59
					Estimates					
$p_{\text{ad-clipped}}$	0.0519	0.0534	0.0570	0.0864	0.2011	0.1427	0.0611	0.0622	0.1135	0.1568
$p_{\text{cwt\|ad-clipped}}$	0.8142	0.7642	0.6594	0.6978	0.8401	0.8810	0.6875	0.7941	0.7912	0.8082
p_{cwt}	0.0423	0.0408	0.0376	0.0603	0.1690	0.1257	0.0420	0.0494	0.0898	0.1267
E_{cwt}	343.1199	300.5484	305.4490	469.1425	2657.6475	2162.2760	104.5490	134.5064	396.4489	194.2500
λ	0.3410	0.4691	0.4092	0.3496	0.4764	0.3547	0.6313	0.3420	0.2901	0.4891
$1/\lambda$	2.9326	2.1315	2.4436	2.8606	2.0992	2.8191	1.5841	2.9241	3.4474	2.0447

C.1 2001 survey: basic data and calculations

Table C-1. Carcass survey CWT expansion basic data, 2001 (source: USFWS).

Gender	Carcass condition	Adipose fin	Head not taken	Head taken, but not processed or lost	Head processed, but CWT not detected	CWT detected, but not extracted or lost	CWT extracted, but unreadable	CWT decoded, SRWC	CWT decoded, not SRWC	Total
Female	Fresh	Hatchery	0	1	15	3	0	43	0	62
Female	Fresh	Unknown	0	0	0	0	0	0	0	0
Female	Non-fresh	Hatchery	0	1	6	0	0	14	0	21
Female	Non-fresh	Unknown	6	0	0	0	0	0	0	6
Female	Unknown	Hatchery	0	0	0	0	0	1	0	1
Female	Unknown	Unknown	0	0	0	0	0	0	0	0
Male	Fresh	Hatchery	0	0	6	1	0	44	0	51
Male	Fresh	Unknown	1	0	0	0	0	0	0	1
Male	Non-fresh	Hatchery	0	0	4	1	0	10	0	15
Male	Non-fresh	Unknown	0	0	0	0	0	0	0	0
Male	Unknown	Hatchery	0	0	0	0	0	1	0	1
Male	Unknown	Unknown	0	0	0	0	0	0	0	0
Unknown	Fresh	Hatchery	0	0	0	0	0	1	0	1
Unknown	Fresh	Unknown	1	0	0	0	0	0	0	1
Unknown	Non-fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Unknown	Hatchery	0	0	0	0	0	3	0	3
Unknown	Unknown	Unknown	0	0	0	0	0	0	0	0
Total			8	2	31	5	0	117	0	163
	Fresh		2	1	21	4	0	88	0	116

Survey

$$
\begin{aligned}
R &= (\textit{Total}\text{: CWT decoded, SRWC}) + (\textit{Total}\text{: CWT decoded, not SRWC}) \\
&= 117 + 0 = 117 \\
E &= 8120 \quad \text{(source: CDFG)} \\
n_{\text{fresh}} &= 2235 \quad \text{(source: USFWS)} \\
n_{\text{fresh,ad-clipped}} &= (\textit{Fresh}\text{: Total}) \\
&= 116 \\
n_{\text{fresh,head-processed}} &= n_{\text{fresh,ad-clipped}} - (\textit{Fresh}\text{: Head not taken}) - (\textit{Fresh}\text{: Head taken, but not processed or lost}) \\
&= 116 - 2 - 1 = 113 \\
n_{\text{fresh,cwt-detected}} &= n_{\text{fresh,head-processed}} - (\textit{Fresh}\text{: Head processed, but CWT not detected}) \\
&= 113 - 21 = 92
\end{aligned}
$$

Estimates

$$
\begin{aligned}
p_{\text{ad-clipped}} &= n_{\text{fresh,ad-clipped}}/n_{\text{fresh}} = 0.0519 \\
p_{\text{cwt}|\text{ad-clipped}} &= n_{\text{fresh,cwt-detected}}/n_{\text{fresh,head-processed}} = 0.8142 \\
p_{\text{cwt}} &= p_{\text{ad-clipped}} \times p_{\text{cwt}|\text{ad-clipped}} = 0.0423 \\
E_{\text{cwt}} &= E \times p_{\text{cwt}} = 343.1199 \\
\lambda &= R/E_{\text{cwt}} = 0.3410 \\
1/\lambda &= 2.9326
\end{aligned}
$$

C.2 2002 survey: basic data and calculations

Table C-2. Carcass survey CWT expansion basic data, 2002 (source: USFWS).

Gender	Carcass condition	Adipose fin	Head not taken	Head taken, but not processed or lost	Head processed, but CWT not detected	CWT detected, but not extracted or lost	CWT extracted, but unreadable	CWT decoded, SRWC	CWT decoded, not SRWC	Total
Female	Fresh	Hatchery	0	0	18	5	0	57	1	81
Female	Fresh	Unknown	0	0	1	0	0	1	0	2
Female	Non-fresh	Hatchery	0	0	32	0	0	60	0	92
Female	Non-fresh	Unknown	0	0	1	0	0	0	0	1
Female	Unknown	Hatchery	0	0	0	0	0	2	0	2
Female	Unknown	Unknown	0	0	0	0	0	0	0	0
Male	Fresh	Hatchery	0	0	5	2	0	15	0	22
Male	Fresh	Unknown	0	0	1	0	0	0	0	1
Male	Non-fresh	Hatchery	0	0	1	0	0	4	0	5
Male	Non-fresh	Unknown	0	0	1	0	0	1	0	2
Male	Unknown	Hatchery	0	0	0	0	0	0	0	0
Male	Unknown	Unknown	0	0	0	0	0	0	0	0
Unknown	Fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Fresh	Unknown	0	2	0	0	0	0	0	2
Unknown	Non-fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Unknown	Hatchery	0	0	0	0	0	0	0	0
Unknown	Unknown	Unknown	0	0	0	0	0	0	0	0
Total			0	2	60	7	0	140	1	210
	Fresh		0	2	25	7	0	73	1	108

Survey

$$
\begin{aligned}
R &= (\textit{Total}\text{: CWT decoded, SRWC}) + (\textit{Total}\text{: CWT decoded, not SRWC}) \\
&= 140 + 1 = 141 \\
E &= 7360 \quad \text{(source: CDFG)} \\
n_{\text{fresh}} &= 2021 \quad \text{(source: USFWS)} \\
n_{\text{fresh,ad-clipped}} &= (\textit{Fresh}\text{: Total}) \\
&= 108 \\
n_{\text{fresh,head-processed}} &= n_{\text{fresh,ad-clipped}} - (\textit{Fresh}\text{: Head not taken}) - (\textit{Fresh}\text{: Head taken, but not processed or lost}) \\
&= 108 - 0 - 2 = 106 \\
n_{\text{fresh,cwt-detected}} &= n_{\text{fresh,head-processed}} - (\textit{Fresh}\text{: Head processed, but CWT not detected}) \\
&= 106 - 25 = 81
\end{aligned}
$$

Estimates

$$
\begin{aligned}
p_{\text{ad-clipped}} &= n_{\text{fresh,ad-clipped}} / n_{\text{fresh}} = 0.0534 \\
p_{\text{cwt}|\text{ad-clipped}} &= n_{\text{fresh,cwt-detected}} / n_{\text{fresh,head-processed}} = 0.7642 \\
p_{\text{cwt}} &= p_{\text{ad-clipped}} \times p_{\text{cwt}|\text{ad-clipped}} = 0.0408 \\
E_{\text{cwt}} &= E \times p_{\text{cwt}} = 300.5484 \\
\lambda &= R / E_{\text{cwt}} = 0.4691 \\
1/\lambda &= 2.1315
\end{aligned}
$$

C.3 2003 survey: basic data and calculations

Table C-3. Carcass survey CWT expansion basic data, 2003 (source: USFWS).

Gender	Carcass condition	Adipose fin	Head not taken	Head taken, but not processed or lost	Head processed, but CWT not detected	CWT detected, but not extracted or lost	CWT extracted, but unreadable	CWT decoded, SRWC	CWT decoded, not SRWC	Total
Female	Fresh	Hatchery	0	0	26	0	6	65	0	97
Female	Fresh	Unknown	0	0	16	0	0	1	0	17
Female	Non-fresh	Hatchery	0	0	17	0	2	32	0	51
Female	Non-fresh	Unknown	0	0	10	0	0	3	0	13
Female	Unknown	Hatchery	0	0	0	0	0	0	0	0
Female	Unknown	Unknown	0	0	0	0	0	0	0	0
Male	Fresh	Hatchery	0	0	4	0	1	18	0	23
Male	Fresh	Unknown	0	0	1	0	0	0	0	1
Male	Non-fresh	Hatchery	0	0	1	0	0	5	0	6
Male	Non-fresh	Unknown	0	0	1	0	0	1	0	2
Male	Unknown	Hatchery	0	0	0	0	0	0	0	0
Male	Unknown	Unknown	0	0	0	0	0	0	0	0
Unknown	Fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Unknown	Hatchery	0	0	0	0	0	0	0	0
Unknown	Unknown	Unknown	0	0	0	0	0	0	0	0
Total			0	0	76	0	9	125	0	210
	Fresh		0	0	47	0	7	84	0	138

Survey

$$\begin{aligned} R &= (\textit{Total}\text{: CWT decoded, SRWC}) + (\textit{Total}\text{: CWT decoded, not SRWC}) \\ &= 125 + 0 = 125 \end{aligned}$$

$$E = 8133 \quad \text{(source: CDFG)}$$

$$n_{\text{fresh}} = 2423 \quad \text{(source: USFWS)}$$

$$\begin{aligned} n_{\text{fresh,ad-clipped}} &= (\textit{Fresh}\text{: Total}) \\ &= 138 \end{aligned}$$

$$\begin{aligned} n_{\text{fresh,head-processed}} &= n_{\text{fresh,ad-clipped}} - (\textit{Fresh}\text{: Head not taken}) - (\textit{Fresh}\text{: Head taken, but not processed or lost}) \\ &= 138 - 0 - 0 = 138 \end{aligned}$$

$$\begin{aligned} n_{\text{fresh,cwt-detected}} &= n_{\text{fresh,head-processed}} - (\textit{Fresh}\text{: Head processed, but CWT not detected}) \\ &= 138 - 47 = 91 \end{aligned}$$

Estimates

$$p_{\text{ad-clipped}} = n_{\text{fresh,ad-clipped}} / n_{\text{fresh}} = 0.0570$$

$$p_{\text{cwt|ad-clipped}} = n_{\text{fresh,cwt-detected}} / n_{\text{fresh,head-processed}} = 0.6594$$

$$p_{\text{cwt}} = p_{\text{ad-clipped}} \times p_{\text{cwt|ad-clipped}} = 0.0376$$

$$E_{\text{cwt}} = E \times p_{\text{cwt}} = 305.4490$$

$$\lambda = R / E_{\text{cwt}} = 0.4092$$

$$1/\lambda = 2.4436$$

C.4 2004 survey: basic data and calculations

Table C-4. Carcass survey CWT expansion basic data, 2004 (source: USFWS).

Gender	Carcass condition	Adipose fin	Head not taken	Head taken, but not processed or lost	Head processed, but CWT not detected	CWT detected, but not extracted or lost	CWT extracted, but unreadable	CWT decoded, SRWC	CWT decoded, not SRWC	Total
Female	Fresh	Hatchery	0	0	22	0	1	51	1	75
Female	Fresh	Unknown	0	0	7	0	0	0	0	7
Female	Non-fresh	Hatchery	0	0	19	0	3	34	0	56
Female	Non-fresh	Unknown	0	0	8	0	0	2	0	10
Female	Unknown	Hatchery	0	0	0	0	0	1	0	1
Female	Unknown	Unknown	0	0	0	0	0	0	0	0
Male	Fresh	Hatchery	1	0	11	0	0	43	0	55
Male	Fresh	Unknown	0	0	2	0	0	1	0	3
Male	Non-fresh	Hatchery	0	0	11	0	0	31	0	42
Male	Non-fresh	Unknown	0	0	1	0	0	0	0	1
Male	Unknown	Hatchery	0	0	0	0	0	0	0	0
Male	Unknown	Unknown	0	0	0	0	0	0	0	0
Unknown	Fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Unknown	Hatchery	0	0	0	0	0	0	0	0
Unknown	Unknown	Unknown	0	0	0	0	0	0	0	0
Total			1	0	81	0	4	163	1	250
	Fresh		1	0	42	0	1	95	1	140

Survey

$$\begin{aligned}
R &= (\textit{Total}\text{: CWT decoded, SRWC}) + (\textit{Total}\text{: CWT decoded, not SRWC}) \\
&= 163 + 1 = 164 \\
E &= 7784 \quad \text{(source: CDFG)} \\
n_{\text{fresh}} &= 1621 \quad \text{(source: USFWS)} \\
n_{\text{fresh,ad-clipped}} &= (\textit{Fresh}\text{: Total}) \\
&= 140 \\
n_{\text{fresh,head-processed}} &= n_{\text{fresh,ad-clipped}} - (\textit{Fresh}\text{: Head not taken}) - (\textit{Fresh}\text{: Head taken, but not processed or lost}) \\
&= 140 - 1 - 0 = 139 \\
n_{\text{fresh,cwt-detected}} &= n_{\text{fresh,head-processed}} - (\textit{Fresh}\text{: Head processed, but CWT not detected}) \\
&= 139 - 42 = 97
\end{aligned}$$

Estimates

$$\begin{aligned}
p_{\text{ad-clipped}} &= n_{\text{fresh,ad-clipped}} / n_{\text{fresh}} = 0.0864 \\
p_{\text{cwt|ad-clipped}} &= n_{\text{fresh,cwt-detected}} / n_{\text{fresh,head-processed}} = 0.6978 \\
p_{\text{cwt}} &= p_{\text{ad-clipped}} \times p_{\text{cwt|ad-clipped}} = 0.0603 \\
E_{\text{cwt}} &= E \times p_{\text{cwt}} = 469.1425 \\
\lambda &= R / E_{\text{cwt}} = 0.3496 \\
1/\lambda &= 2.8606
\end{aligned}$$

C.5 2005 survey: basic data and calculations

Table C-5. Carcass survey CWT expansion basic data, 2005 (source: USFWS).

Gender	Carcass condition	Adipose fin	Head not taken	Head taken, but not processed or lost	Head processed, but CWT not detected	CWT detected, but not extracted or lost	CWT extracted, but unreadable	CWT decoded, SRWC	CWT decoded, not SRWC	Total
Female	Fresh	Hatchery	0	2	86	1	0	508	0	597
Female	Fresh	Unknown	0	0	27	0	0	3	0	30
Female	Non-fresh	Hatchery	6	3	96	0	0	405	0	510
Female	Non-fresh	Unknown	0	0	31	0	0	5	0	36
Female	Unknown	Hatchery	0	0	0	0	0	3	0	3
Female	Unknown	Unknown	0	0	0	0	0	0	0	0
Male	Fresh	Hatchery	2	3	16	0	0	184	1	206
Male	Fresh	Unknown	0	1	4	0	0	0	0	5
Male	Non-fresh	Hatchery	2	0	19	0	0	148	0	169
Male	Non-fresh	Unknown	0	0	3	0	0	2	0	5
Male	Unknown	Hatchery	0	0	0	0	0	1	0	1
Male	Unknown	Unknown	0	0	0	0	0	0	0	0
Unknown	Fresh	Hatchery	0	0	0	0	0	2	0	2
Unknown	Fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Hatchery	0	0	0	0	0	1	0	1
Unknown	Non-fresh	Unknown	0	0	1	0	0	0	0	1
Unknown	Unknown	Hatchery	0	0	0	0	0	3	0	3
Unknown	Unknown	Unknown	0	0	0	0	0	0	0	0
Total			10	9	283	1	0	1265	1	1569
	Fresh		2	6	133	1	0	697	1	840

Survey

$$\begin{aligned}
R &= (\textit{Total}\text{: CWT decoded, SRWC}) + (\textit{Total}\text{: CWT decoded, not SRWC}) \\
&= 1265 + 1 = 1266 \\
E &= 15730 \quad \text{(source: CDFG)} \\
n_{\text{fresh}} &= 4177 \quad \text{(source: USFWS)} \\
n_{\text{fresh,ad-clipped}} &= (\textit{Fresh}\text{: Total}) \\
&= 840 \\
n_{\text{fresh,head-processed}} &= n_{\text{fresh,ad-clipped}} - (\textit{Fresh}\text{: Head not taken}) - (\textit{Fresh}\text{: Head taken, but not processed or lost}) \\
&= 840 - 2 - 6 = 832 \\
n_{\text{fresh,cwt-detected}} &= n_{\text{fresh,head-processed}} - (\textit{Fresh}\text{: Head processed, but CWT not detected}) \\
&= 832 - 133 = 699
\end{aligned}$$

Estimates

$$\begin{aligned}
p_{\text{ad-clipped}} &= n_{\text{fresh,ad-clipped}} / n_{\text{fresh}} = 0.2011 \\
p_{\text{cwt|ad-clipped}} &= n_{\text{fresh,cwt-detected}} / n_{\text{fresh,head-processed}} = 0.8401 \\
p_{\text{cwt}} &= p_{\text{ad-clipped}} \times p_{\text{cwt|ad-clipped}} = 0.1690 \\
E_{\text{cwt}} &= E \times p_{\text{cwt}} = 2657.6475 \\
\lambda &= R / E_{\text{cwt}} = 0.4764 \\
1/\lambda &= 2.0992
\end{aligned}$$

C.6 2006 survey: basic data and calculations

Table C-6. Carcass survey CWT expansion basic data, 2006 (source: USFWS).

Gender	Carcass condition	Adipose fin	Head not taken	Head taken, but not processed or lost	Head processed, but CWT not detected	CWT detected, but not extracted or lost	CWT extracted, but unreadable	CWT decoded, SRWC	CWT decoded, not SRWC	Total
Female	Fresh	Hatchery	0	1	35	4	2	282	0	324
Female	Fresh	Unknown	0	0	13	0	0	0	0	13
Female	Non-fresh	Hatchery	0	3	53	5	0	267	0	328
Female	Non-fresh	Unknown	0	0	16	0	0	7	0	23
Female	Unknown	Hatchery	0	0	0	0	0	0	0	0
Female	Unknown	Unknown	0	0	0	0	0	0	0	0
Male	Fresh	Hatchery	0	2	3	0	1	96	0	102
Male	Fresh	Unknown	0	0	1	0	0	0	0	1
Male	Non-fresh	Hatchery	0	4	10	2	0	106	0	122
Male	Non-fresh	Unknown	0	0	4	0	0	4	0	8
Male	Unknown	Hatchery	0	0	0	0	0	0	0	0
Male	Unknown	Unknown	0	0	0	0	0	0	0	0
Unknown	Fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Unknown	Hatchery	0	0	0	0	0	0	0	0
Unknown	Unknown	Unknown	0	0	0	0	0	5	0	5
Total			0	10	135	11	3	767	0	926
	Fresh		0	3	52	4	3	378	0	440

Survey

$$\begin{aligned} R &= (\textit{Total}\text{: CWT decoded, SRWC}) + (\textit{Total}\text{: CWT decoded, not SRWC}) \\ &= 767 + 0 = 767 \end{aligned}$$

$$E = 17197 \quad \text{(source: CDFG)}$$

$$n_{\text{fresh}} = 3083 \quad \text{(source: USFWS)}$$

$$\begin{aligned} n_{\text{fresh,ad-clipped}} &= (\textit{Fresh}\text{: Total}) \\ &= 440 \end{aligned}$$

$$\begin{aligned} n_{\text{fresh,head-processed}} &= n_{\text{fresh,ad-clipped}} - (\textit{Fresh}\text{: Head not taken}) - (\textit{Fresh}\text{: Head taken, but not processed or lost}) \\ &= 440 - 0 - 3 = 437 \end{aligned}$$

$$\begin{aligned} n_{\text{fresh,cwt-detected}} &= n_{\text{fresh,head-processed}} - (\textit{Fresh}\text{: Head processed, but CWT not detected}) \\ &= 437 - 52 = 385 \end{aligned}$$

Estimates

$$p_{\text{ad-clipped}} = n_{\text{fresh,ad-clipped}} / n_{\text{fresh}} = 0.1427$$

$$p_{\text{cwt|ad-clipped}} = n_{\text{fresh,cwt-detected}} / n_{\text{fresh,head-processed}} = 0.8810$$

$$p_{\text{cwt}} = p_{\text{ad-clipped}} \times p_{\text{cwt|ad-clipped}} = 0.1257$$

$$E_{\text{cwt}} = E \times p_{\text{cwt}} = 2162.2760$$

$$\lambda = R / E_{\text{cwt}} = 0.3547$$

$$1/\lambda = 2.8191$$

C.7 2007 survey: basic data and calculations

Table C-7. Carcass survey CWT expansion basic data, 2007 (source: USFWS).

Gender	Carcass condition	Adipose fin	Head not taken	Head taken, but not processed or lost	Head processed, but CWT not detected	CWT detected, but not extracted or lost	CWT extracted, but unreadable	CWT decoded, SRWC	CWT decoded, not SRWC	Total
Female	Fresh	Hatchery	0	0	9	0	0	27	1	37
Female	Fresh	Unknown	0	0	5	0	0	0	0	5
Female	Non-fresh	Hatchery	0	0	5	0	0	29	0	34
Female	Non-fresh	Unknown	0	0	2	0	0	0	0	2
Female	Unknown	Hatchery	0	0	0	0	0	0	0	0
Female	Unknown	Unknown	0	0	0	0	0	0	0	0
Male	Fresh	Hatchery	0	0	0	0	0	5	0	5
Male	Fresh	Unknown	0	0	1	0	0	0	0	1
Male	Non-fresh	Hatchery	0	0	3	0	0	4	0	7
Male	Non-fresh	Unknown	0	0	0	0	0	0	0	0
Male	Unknown	Hatchery	0	0	0	0	0	0	0	0
Male	Unknown	Unknown	0	0	0	0	0	0	0	0
Unknown	Fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Unknown	Hatchery	0	0	0	0	0	0	0	0
Unknown	Unknown	Unknown	0	0	0	0	0	0	0	0
Total			0	0	25	0	0	65	1	91
	Fresh		0	0	15	0	0	32	1	48

Survey

$$\begin{aligned}
R &= (\textit{Total}\text{: CWT decoded, SRWC}) + (\textit{Total}\text{: CWT decoded, not SRWC}) \\
&= 65 + 1 = 66 \\
E &= 2487 \quad \text{(source: CDFG)} \\
n_{\text{fresh}} &= 785 \quad \text{(source: USFWS)} \\
n_{\text{fresh,ad-clipped}} &= (\textit{Fresh}\text{: Total}) \\
&= 48 \\
n_{\text{fresh,head-processed}} &= n_{\text{fresh,ad-clipped}} - (\textit{Fresh}\text{: Head not taken}) - (\textit{Fresh}\text{: Head taken, but not processed or lost}) \\
&= 48 - 0 - 0 = 48 \\
n_{\text{fresh,cwt-detected}} &= n_{\text{fresh,head-processed}} - (\textit{Fresh}\text{: Head processed, but CWT not detected}) \\
&= 48 - 15 = 33
\end{aligned}$$

Estimates

$$\begin{aligned}
p_{\text{ad-clipped}} &= n_{\text{fresh,ad-clipped}} / n_{\text{fresh}} = 0.0611 \\
p_{\text{cwt}|\text{ad-clipped}} &= n_{\text{fresh,cwt-detected}} / n_{\text{fresh,head-processed}} = 0.6875 \\
p_{\text{cwt}} &= p_{\text{ad-clipped}} \times p_{\text{cwt}|\text{ad-clipped}} = 0.0420 \\
E_{\text{cwt}} &= E \times p_{\text{cwt}} = 104.5490 \\
\lambda &= R/E_{\text{cwt}} = 0.6313 \\
1/\lambda &= 1.5841
\end{aligned}$$

C.8 2008 survey: basic data and calculations

Table C-8. Carcass survey CWT expansion basic data, 2008 (source: USFWS).

Gender	Carcass condition	Adipose fin	Head not taken	Head taken, but not processed or lost	Head processed, but CWT not detected	CWT detected, but not extracted or lost	CWT extracted, but unreadable	CWT decoded, SRWC	CWT decoded, not SRWC	Total
Female	Fresh	Hatchery	0	0	5	0	0	20	1	26
Female	Fresh	Unknown	0	0	0	0	0	0	0	0
Female	Non-fresh	Hatchery	0	0	5	0	0	11	0	16
Female	Non-fresh	Unknown	0	0	5	0	0	0	0	5
Female	Unknown	Hatchery	0	0	0	0	0	0	0	0
Female	Unknown	Unknown	0	0	0	0	0	0	0	0
Male	Fresh	Hatchery	0	0	1	0	0	6	0	7
Male	Fresh	Unknown	0	0	1	0	0	0	0	1
Male	Non-fresh	Hatchery	0	0	3	0	0	8	0	11
Male	Non-fresh	Unknown	0	0	0	0	0	0	0	0
Male	Unknown	Hatchery	0	0	0	0	0	0	0	0
Male	Unknown	Unknown	0	0	0	0	0	0	0	0
Unknown	Fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Unknown	Hatchery	0	0	0	0	0	0	0	0
Unknown	Unknown	Unknown	0	0	0	0	0	0	0	0
Total			0	0	20	0	0	45	1	66
	Fresh		0	0	7	0	0	26	1	34

Survey

$$
\begin{aligned}
R &= (\textit{Total}\text{: CWT decoded, SRWC}) + (\textit{Total}\text{: CWT decoded, not SRWC}) \\
&= 45 + 1 = 46 \\
E &= 2725 \quad \text{(source: CDFG)} \\
n_{\text{fresh}} &= 547 \quad \text{(source: USFWS)} \\
n_{\text{fresh,ad-clipped}} &= (\textit{Fresh}\text{: Total}) \\
&= 34 \\
n_{\text{fresh,head-processed}} &= n_{\text{fresh,ad-clipped}} - (\textit{Fresh}\text{: Head not taken}) - (\textit{Fresh}\text{: Head taken, but not processed or lost}) \\
&= 34 - 0 - 0 = 34 \\
n_{\text{fresh,cwt-detected}} &= n_{\text{fresh,head-processed}} - (\textit{Fresh}\text{: Head processed, but CWT not detected}) \\
&= 34 - 7 = 27
\end{aligned}
$$

Estimates

$$
\begin{aligned}
p_{\text{ad-clipped}} &= n_{\text{fresh,ad-clipped}}/n_{\text{fresh}} = 0.0622 \\
p_{\text{cwt}|\text{ad-clipped}} &= n_{\text{fresh,cwt-detected}}/n_{\text{fresh,head-processed}} = 0.7941 \\
p_{\text{cwt}} &= p_{\text{ad-clipped}} \times p_{\text{cwt}|\text{ad-clipped}} = 0.0494 \\
E_{\text{cwt}} &= E \times p_{\text{cwt}} = 134.5064 \\
\lambda &= R/E_{\text{cwt}} = 0.3420 \\
1/\lambda &= 2.9241
\end{aligned}
$$

C.9 2009 survey: basic data and calculations

Table C-9. Carcass survey CWT expansion basic data, 2009 (source: USFWS).

Gender	Carcass condition	Adipose fin	Head not taken	Head taken, but not processed or lost	Head processed, but CWT not detected	CWT detected, but not extracted or lost	CWT extracted, but unreadable	CWT decoded, SRWC	CWT decoded, not SRWC	Total
Female	Fresh	Hatchery	0	0	9	1	0	50	1	61
Female	Fresh	Unknown	0	0	6	0	0	4	0	10
Female	Non-fresh	Hatchery	0	0	5	0	0	28	0	33
Female	Non-fresh	Unknown	0	0	5	0	0	2	0	7
Female	Unknown	Hatchery	0	0	0	0	0	0	0	0
Female	Unknown	Unknown	0	0	0	0	0	0	0	0
Male	Fresh	Hatchery	0	0	3	0	0	13	0	16
Male	Fresh	Unknown	0	0	1	0	0	3	0	4
Male	Non-fresh	Hatchery	0	0	4	0	0	11	0	15
Male	Non-fresh	Unknown	0	0	2	0	0	3	0	5
Male	Unknown	Hatchery	0	0	0	0	0	0	0	0
Male	Unknown	Unknown	0	0	0	0	0	0	0	0
Unknown	Fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Unknown	Hatchery	0	0	0	0	0	0	0	0
Unknown	Unknown	Unknown	0	0	0	0	0	0	0	0
Total			0	0	35	1	0	114	1	151
	Fresh		0	0	19	1	0	70	1	91

Survey

$$\begin{aligned} R &= (\textit{Total}\text{: CWT decoded, SRWC}) + (\textit{Total}\text{: CWT decoded, not SRWC}) \\ &= 114 + 1 = 115 \end{aligned}$$

$$E = 4416 \quad \text{(source: CDFG)}$$

$$n_{\text{fresh}} = 802 \quad \text{(source: USFWS)}$$

$$\begin{aligned} n_{\text{fresh,ad-clipped}} &= (\textit{Fresh}\text{: Total}) \\ &= 91 \end{aligned}$$

$$\begin{aligned} n_{\text{fresh,head-processed}} &= n_{\text{fresh,ad-clipped}} - (\textit{Fresh}\text{: Head not taken}) - (\textit{Fresh}\text{: Head taken, but not processed or lost}) \\ &= 91 - 0 - 0 = 91 \end{aligned}$$

$$\begin{aligned} n_{\text{fresh,cwt-detected}} &= n_{\text{fresh,head-processed}} - (\textit{Fresh}\text{: Head processed, but CWT not detected}) \\ &= 91 - 19 = 72 \end{aligned}$$

Estimates

$$p_{\text{ad-clipped}} = n_{\text{fresh,ad-clipped}} / n_{\text{fresh}} = 0.1135$$

$$p_{\text{cwt|ad-clipped}} = n_{\text{fresh,cwt-detected}} / n_{\text{fresh,head-processed}} = 0.7912$$

$$p_{\text{cwt}} = p_{\text{ad-clipped}} \times p_{\text{cwt|ad-clipped}} = 0.0898$$

$$E_{\text{cwt}} = E \times p_{\text{cwt}} = 396.4489$$

$$\lambda = R / E_{\text{cwt}} = 0.2901$$

$$1/\lambda = 3.4474$$

C.10 2010 survey: basic data and calculations

Table C-10. Carcass survey CWT expansion basic data, 2010 (source: USFWS).

Gender	Carcass condition	Adipose fin	Head not taken	Head taken, but not processed or lost	Head processed, but CWT not detected	CWT detected, but not extracted or lost	CWT extracted, but unreadable	CWT decoded, SRWC	CWT decoded, not SRWC	Total
Female	Fresh	Hatchery	0	1	5	1	0	33	3	43
Female	Fresh	Unknown	0	0	8	1	0	0	0	9
Female	Non-fresh	Hatchery	0	1	6	1	0	30	0	38
Female	Non-fresh	Unknown	0	0	12	0	0	0	0	12
Female	Unknown	Hatchery	0	0	0	0	0	0	0	0
Female	Unknown	Unknown	0	0	0	0	0	0	0	0
Male	Fresh	Hatchery	0	0	0	2	0	18	1	20
Male	Fresh	Unknown	0	0	1	0	0	0	0	1
Male	Non-fresh	Hatchery	0	0	0	0	0	9	1	9
Male	Non-fresh	Unknown	0	0	3	0	0	0	0	3
Male	Unknown	Hatchery	0	0	0	0	0	0	0	0
Male	Unknown	Unknown	0	0	0	0	0	0	0	0
Unknown	Fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Hatchery	0	0	0	0	0	0	0	0
Unknown	Non-fresh	Unknown	0	0	0	0	0	0	0	0
Unknown	Unknown	Hatchery	0	0	0	0	0	0	0	0
Unknown	Unknown	Unknown	0	0	0	0	0	0	0	0
Total			0	2	35	5	0	90	5	135
	Fresh		0	1	14	4	0	51	4	74

Survey

$$\begin{aligned}
R &= (\textit{Total}\text{: CWT decoded, SRWC}) + (\textit{Total}\text{: CWT decoded, not SRWC}) \\
&= 90+5 = 95 \\
E &= 1533 \quad \text{(source: CDFG)} \\
n_{\text{fresh}} &= 472 \quad \text{(source: USFWS)} \\
n_{\text{fresh,ad-clipped}} &= (\textit{Fresh}\text{: Total}) \\
&= 74 \\
n_{\text{fresh,head-processed}} &= n_{\text{fresh,ad-clipped}} - (\textit{Fresh}\text{: Head not taken}) - (\textit{Fresh}\text{: Head taken, but not processed or lost}) \\
&= 74-0-1 = 73 \\
n_{\text{fresh,cwt-detected}} &= n_{\text{fresh,head-processed}} - (\textit{Fresh}\text{: Head processed, but CWT not detected}) \\
&= 73-14 = 59
\end{aligned}$$

Estimates

$$\begin{aligned}
p_{\text{ad-clipped}} &= n_{\text{fresh,ad-clipped}}/n_{\text{fresh}} = 0.1568 \\
p_{\text{cwt|ad-clipped}} &= n_{\text{fresh,cwt-detected}}/n_{\text{fresh,head-processed}} = 0.8082 \\
p_{\text{cwt}} &= p_{\text{ad-clipped}} \times p_{\text{cwt|ad-clipped}} = 0.1267 \\
E_{\text{cwt}} &= E \times p_{\text{cwt}} = 194.2500 \\
\lambda &= R/E_{\text{cwt}} = 0.4891 \\
1/\lambda &= 2.0447
\end{aligned}$$

Appendix D Reconstructed cohorts: 1998–2007 broods

Tables D-1 through D-10 display the cohort reconstructions of hatchery-origin SRWC, brood years 1998–2007. Notation used for column headings: BY is brood year; CY is calendar year; N is ocean-wide abundance at the beginning of the month; I_{com} is ocean commercial fishery impacts; I_{rec} is ocean recreational fishery impacts; V is natural mortalities; H_r is river harvest; E_{hat} is hatchery escapement; E_{nat} is natural area escapement. For a given Age/Month combination, the sum of the columns to the right of N equals the decrement in abundance for that Age/Month.

Table D-1. Reconstructed cohort: 1998 brood.

				Ocean				River		
BY	CY	Age	Month	N	I_{com}	I_{rec}	V	H_r	E_{hat}	E_{nat}
1998	1999	2	3	1528.38	0.00	0.00	85.78	0.00	0.00	0.00
1998	1999	2	4	1442.60	0.00	0.00	80.97	0.00	0.00	0.00
1998	1999	2	5	1361.63	0.00	0.00	76.42	0.00	0.00	0.00
1998	1999	2	6	1285.21	0.00	0.00	72.13	0.00	0.00	0.00
1998	1999	2	7	1213.08	0.00	0.00	68.08	0.00	0.00	0.00
1998	1999	2	8	1144.99	0.00	8.68	63.78	0.00	0.00	0.00
1998	1999	2	9	1072.53	0.00	0.00	60.20	0.00	0.00	0.00
1998	1999	2	10	1012.34	0.00	0.00	56.82	0.00	0.00	0.00
1998	1999	2	11	955.52	0.00	0.00	53.63	0.00	0.00	0.00
1998	1999	2	12	901.89	0.00	0.00	50.62	0.00	0.00	0.00
1998	2000	2	1	851.27	0.00	0.00	47.78	0.00	0.00	0.00
1998	2000	2	2	803.49	0.00	0.00	45.10	23.48	8.29	0.00
1998	2000	3	3	726.63	0.00	0.00	13.39	0.00	0.00	0.00
1998	2000	3	4	713.24	0.00	8.37	12.99	0.00	0.00	0.00
1998	2000	3	5	691.88	0.00	0.00	12.75	0.00	0.00	0.00
1998	2000	3	6	679.14	28.93	43.65	11.17	0.00	0.00	0.00
1998	2000	3	7	595.38	6.52	53.84	9.86	0.00	0.00	0.00
1998	2000	3	8	525.16	0.00	14.14	9.41	0.00	0.00	0.00
1998	2000	3	9	501.60	0.00	4.73	9.15	0.00	0.00	0.00
1998	2000	3	10	487.72	0.00	9.71	8.81	0.00	0.00	0.00
1998	2000	3	11	469.20	0.00	0.00	8.64	0.00	0.00	0.00
1998	2000	3	12	460.56	0.00	0.00	8.49	0.00	0.00	0.00
1998	2001	3	1	452.07	0.00	0.00	8.33	0.00	0.00	0.00
1998	2001	3	2	443.74	0.00	0.00	8.18	90.83	13.18	268.04
1998	2001	4	3	63.52	0.00	0.00	1.17	0.00	0.00	0.00
1998	2001	4	4	62.35	0.00	0.00	1.15	0.00	0.00	0.00
1998	2001	4	5	61.20	5.21	0.00	1.03	0.00	0.00	0.00
1998	2001	4	6	54.96	0.00	0.00	1.01	0.00	0.00	0.00
1998	2001	4	7	53.95	0.00	0.00	0.99	0.00	0.00	0.00
1998	2001	4	8	52.95	0.00	0.00	0.98	0.00	0.00	0.00
1998	2001	4	9	51.98	2.71	0.00	0.91	0.00	0.00	0.00
1998	2001	4	10	48.36	0.00	0.00	0.89	0.00	0.00	0.00
1998	2001	4	11	47.47	0.00	0.00	0.87	0.00	0.00	0.00
1998	2001	4	12	46.59	0.00	0.00	0.86	0.00	0.00	0.00
1998	2002	4	1	45.74	0.00	0.00	0.84	0.00	0.00	0.00
1998	2002	4	2	44.89	0.00	0.00	0.83	31.54	0.00	4.92
1998	2002	5	3	7.61	0.00	0.00	0.14	0.00	0.00	0.00
1998	2002	5	4	7.47	0.00	0.00	0.14	0.00	0.00	0.00
1998	2002	5	5	7.33	7.33	0.00	0.00	0.00	0.00	0.00
1998	2002	5	6	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1998	2002	5	7	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1998	2002	5	8	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1998	2002	5	9	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1998	2002	5	10	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1998	2002	5	11	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1998	2002	5	12	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1998	2003	5	1	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1998	2003	5	2	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table D-2. Reconstructed cohort: 1999 brood.

				Ocean				River		
BY	CY	Age	Month	N	I_{com}	I_{rec}	V	H_r	E_{hat}	E_{nat}
1999	2000	2	3	1162.47	0.00	0.00	65.24	0.00	0.00	0.00
1999	2000	2	4	1097.23	0.00	0.00	61.58	0.00	0.00	0.00
1999	2000	2	5	1035.65	0.00	0.00	58.13	0.00	0.00	0.00
1999	2000	2	6	977.52	0.00	0.00	54.86	0.00	0.00	0.00
1999	2000	2	7	922.65	0.00	0.00	51.78	0.00	0.00	0.00
1999	2000	2	8	870.87	0.00	0.00	48.88	0.00	0.00	0.00
1999	2000	2	9	821.99	0.00	0.00	46.13	0.00	0.00	0.00
1999	2000	2	10	775.86	0.00	0.00	43.55	0.00	0.00	0.00
1999	2000	2	11	732.31	0.00	0.00	41.10	0.00	0.00	0.00
1999	2000	2	12	691.21	0.00	0.00	38.79	0.00	0.00	0.00
1999	2001	2	1	652.42	0.00	0.00	36.62	0.00	0.00	0.00
1999	2001	2	2	615.80	0.00	0.00	34.56	0.00	0.00	95.27
1999	2001	3	3	485.97	0.00	13.19	8.71	0.00	0.00	0.00
1999	2001	3	4	464.07	0.00	37.31	7.86	0.00	0.00	0.00
1999	2001	3	5	418.89	0.00	9.25	7.55	0.00	0.00	0.00
1999	2001	3	6	402.10	0.00	5.03	7.32	0.00	0.00	0.00
1999	2001	3	7	389.76	14.15	34.44	6.29	0.00	0.00	0.00
1999	2001	3	8	334.89	0.00	8.74	6.01	0.00	0.00	0.00
1999	2001	3	9	320.14	0.00	0.00	5.90	0.00	0.00	0.00
1999	2001	3	10	314.24	0.00	0.00	5.79	0.00	0.00	0.00
1999	2001	3	11	308.45	0.00	0.00	5.68	0.00	0.00	0.00
1999	2001	3	12	302.77	0.00	0.00	5.58	0.00	0.00	0.00
1999	2002	3	1	297.19	0.00	0.00	5.48	0.00	0.00	0.00
1999	2002	3	2	291.72	0.00	0.00	5.37	0.00	5.06	268.24
1999	2002	4	3	13.04	0.00	0.00	0.24	0.00	0.00	0.00
1999	2002	4	4	12.80	0.00	0.00	0.24	0.00	0.00	0.00
1999	2002	4	5	12.56	0.00	0.00	0.23	0.00	0.00	0.00
1999	2002	4	6	12.33	5.49	0.00	0.13	0.00	0.00	0.00
1999	2002	4	7	6.72	0.00	3.85	0.05	0.00	0.00	0.00
1999	2002	4	8	2.81	0.00	0.00	0.05	0.00	0.00	0.00
1999	2002	4	9	2.76	0.00	0.00	0.05	0.00	0.00	0.00
1999	2002	4	10	2.71	0.00	0.00	0.05	0.00	0.00	0.00
1999	2002	4	11	2.66	0.00	0.00	0.05	0.00	0.00	0.00
1999	2002	4	12	2.61	0.00	0.00	0.05	0.00	0.00	0.00
1999	2003	4	1	2.56	0.00	0.00	0.05	0.00	0.00	0.00
1999	2003	4	2	2.52	0.00	0.00	0.05	0.00	0.00	2.47
1999	2003	5	3	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1999	2003	5	4	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1999	2003	5	5	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1999	2003	5	6	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1999	2003	5	7	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1999	2003	5	8	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1999	2003	5	9	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1999	2003	5	10	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1999	2003	5	11	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1999	2003	5	12	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1999	2004	5	1	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1999	2004	5	2	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table D-3. Reconstructed cohort: 2000 brood.

				Ocean				River		
BY	CY	Age	Month	N	I_{com}	I_{rec}	V	H_r	E_{hat}	E_{nat}
2000	2001	2	3	1063.81	0.00	0.00	59.71	0.00	0.00	0.00
2000	2001	2	4	1004.11	0.00	0.00	56.36	0.00	0.00	0.00
2000	2001	2	5	947.75	0.00	0.00	53.19	0.00	0.00	0.00
2000	2001	2	6	894.56	0.00	0.00	50.21	0.00	0.00	0.00
2000	2001	2	7	844.35	0.00	0.00	47.39	0.00	0.00	0.00
2000	2001	2	8	796.96	0.00	0.00	44.73	0.00	0.00	0.00
2000	2001	2	9	752.23	0.00	0.00	42.22	0.00	0.00	0.00
2000	2001	2	10	710.01	0.00	0.00	39.85	0.00	0.00	0.00
2000	2001	2	11	670.16	0.00	0.00	37.61	0.00	0.00	0.00
2000	2001	2	12	632.55	0.00	0.00	35.50	0.00	0.00	0.00
2000	2002	2	1	597.05	0.00	0.00	33.51	0.00	0.00	0.00
2000	2002	2	2	563.54	0.00	0.00	31.63	0.00	3.11	30.50
2000	2002	3	3	498.30	0.00	0.00	9.18	0.00	0.00	0.00
2000	2002	3	4	489.12	0.00	0.00	9.01	0.00	0.00	0.00
2000	2002	3	5	480.11	0.00	19.81	8.48	0.00	0.00	0.00
2000	2002	3	6	451.82	14.33	16.81	7.75	0.00	0.00	0.00
2000	2002	3	7	412.93	17.30	22.86	6.87	0.00	0.00	0.00
2000	2002	3	8	365.91	9.01	8.66	6.42	0.00	0.00	0.00
2000	2002	3	9	341.82	0.00	0.00	6.30	0.00	0.00	0.00
2000	2002	3	10	335.53	0.00	0.00	6.18	0.00	0.00	0.00
2000	2002	3	11	329.35	0.00	0.00	6.07	0.00	0.00	0.00
2000	2002	3	12	323.28	0.00	0.00	5.96	0.00	0.00	0.00
2000	2003	3	1	317.32	0.00	0.00	5.85	0.00	0.00	0.00
2000	2003	3	2	311.48	0.00	0.00	5.74	0.00	6.13	282.88
2000	2003	4	3	16.73	0.00	5.65	0.20	0.00	0.00	0.00
2000	2003	4	4	10.88	0.00	0.00	0.20	0.00	0.00	0.00
2000	2003	4	5	10.68	0.00	0.00	0.20	0.00	0.00	0.00
2000	2003	4	6	10.48	3.50	0.00	0.13	0.00	0.00	0.00
2000	2003	4	7	6.85	0.00	0.00	0.13	0.00	0.00	0.00
2000	2003	4	8	6.72	0.00	0.00	0.12	0.00	0.00	0.00
2000	2003	4	9	6.60	0.00	0.00	0.12	0.00	0.00	0.00
2000	2003	4	10	6.47	0.00	0.00	0.12	0.00	0.00	0.00
2000	2003	4	11	6.36	0.00	0.00	0.12	0.00	0.00	0.00
2000	2003	4	12	6.24	0.00	0.00	0.11	0.00	0.00	0.00
2000	2004	4	1	6.12	0.00	0.00	0.11	0.00	0.00	0.00
2000	2004	4	2	6.01	0.00	0.00	0.11	0.00	0.00	5.90
2000	2004	5	3	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2000	2004	5	4	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2000	2004	5	5	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2000	2004	5	6	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2000	2004	5	7	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2000	2004	5	8	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2000	2004	5	9	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2000	2004	5	10	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2000	2004	5	11	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2000	2004	5	12	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2000	2005	5	1	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2000	2005	5	2	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table D-4. Reconstructed cohort: 2001 brood.

				Ocean				River		
BY	CY	Age	Month	N	I_{com}	I_{rec}	V	H_r	E_{hat}	E_{nat}
2001	2002	2	3	954.19	0.00	0.00	53.55	0.00	0.00	0.00
2001	2002	2	4	900.63	0.00	0.00	50.55	0.00	0.00	0.00
2001	2002	2	5	850.09	0.00	0.00	47.71	0.00	0.00	0.00
2001	2002	2	6	802.37	0.00	0.00	45.03	0.00	0.00	0.00
2001	2002	2	7	757.34	0.00	0.00	42.51	0.00	0.00	0.00
2001	2002	2	8	714.83	0.00	0.00	40.12	0.00	0.00	0.00
2001	2002	2	9	674.71	0.00	0.00	37.87	0.00	0.00	0.00
2001	2002	2	10	636.84	0.00	0.00	35.74	0.00	0.00	0.00
2001	2002	2	11	601.10	0.00	0.00	33.74	0.00	0.00	0.00
2001	2002	2	12	567.36	0.00	0.00	31.84	0.00	0.00	0.00
2001	2003	2	1	535.52	0.00	0.00	30.06	0.00	0.00	0.00
2001	2003	2	2	505.46	0.00	0.00	28.37	0.00	1.09	27.76
2001	2003	3	3	448.24	0.00	0.00	8.26	0.00	0.00	0.00
2001	2003	3	4	439.99	0.00	0.00	8.11	0.00	0.00	0.00
2001	2003	3	5	431.88	0.00	13.19	7.71	0.00	0.00	0.00
2001	2003	3	6	410.98	0.00	17.51	7.25	0.00	0.00	0.00
2001	2003	3	7	386.22	0.00	15.64	6.83	0.00	0.00	0.00
2001	2003	3	8	363.75	0.00	0.00	6.70	0.00	0.00	0.00
2001	2003	3	9	357.05	0.00	0.00	6.58	0.00	0.00	0.00
2001	2003	3	10	350.47	0.00	0.00	6.46	0.00	0.00	0.00
2001	2003	3	11	344.01	0.00	0.00	6.34	0.00	0.00	0.00
2001	2003	3	12	337.68	0.00	0.00	6.22	0.00	0.00	0.00
2001	2004	3	1	331.46	0.00	0.00	6.11	0.00	0.00	0.00
2001	2004	3	2	325.35	0.00	0.00	5.99	0.00	8.21	302.82
2001	2004	4	3	8.32	0.00	0.00	0.15	0.00	0.00	0.00
2001	2004	4	4	8.17	0.00	5.59	0.05	0.00	0.00	0.00
2001	2004	4	5	2.53	0.00	0.00	0.05	0.00	0.00	0.00
2001	2004	4	6	2.48	0.00	0.00	0.05	0.00	0.00	0.00
2001	2004	4	7	2.44	0.00	0.00	0.04	0.00	0.00	0.00
2001	2004	4	8	2.39	0.00	0.00	0.04	0.00	0.00	0.00
2001	2004	4	9	2.35	0.00	0.00	0.04	0.00	0.00	0.00
2001	2004	4	10	2.30	0.00	0.00	0.04	0.00	0.00	0.00
2001	2004	4	11	2.26	0.00	0.00	0.04	0.00	0.00	0.00
2001	2004	4	12	2.22	0.00	0.00	0.04	0.00	0.00	0.00
2001	2005	4	1	2.18	0.00	0.00	0.04	0.00	0.00	0.00
2001	2005	4	2	2.14	0.00	0.00	0.04	0.00	0.00	2.10
2001	2005	5	3	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2001	2005	5	4	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2001	2005	5	5	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2001	2005	5	6	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2001	2005	5	7	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2001	2005	5	8	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2001	2005	5	9	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2001	2005	5	10	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2001	2005	5	11	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2001	2005	5	12	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2001	2006	5	1	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2001	2006	5	2	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table D-5. Reconstructed cohort: 2002 brood.

				Ocean				River		
BY	CY	Age	Month	N	I_{com}	I_{rec}	V	H_r	E_{hat}	E_{nat}
2002	2003	2	3	10345.83	0.00	0.00	580.67	0.00	0.00	0.00
2002	2003	2	4	9765.16	0.00	0.00	548.08	0.00	0.00	0.00
2002	2003	2	5	9217.09	0.00	0.00	517.32	0.00	0.00	0.00
2002	2003	2	6	8699.77	0.00	0.00	488.28	0.00	0.00	0.00
2002	2003	2	7	8211.49	0.00	0.00	460.88	0.00	0.00	0.00
2002	2003	2	8	7750.61	0.00	0.00	435.01	0.00	0.00	0.00
2002	2003	2	9	7315.61	0.00	0.00	410.59	0.00	0.00	0.00
2002	2003	2	10	6905.01	0.00	0.00	387.55	0.00	0.00	0.00
2002	2003	2	11	6517.46	0.00	0.00	365.80	0.00	0.00	0.00
2002	2003	2	12	6151.67	0.00	0.00	345.27	0.00	0.00	0.00
2002	2004	2	1	5806.40	0.00	0.00	325.89	0.00	0.00	0.00
2002	2004	2	2	5480.51	0.00	0.00	307.60	0.00	0.00	178.45
2002	2004	3	3	4994.46	0.00	0.00	92.02	0.00	0.00	0.00
2002	2004	3	4	4902.45	0.00	81.23	88.82	0.00	0.00	0.00
2002	2004	3	5	4732.39	110.61	190.31	81.64	0.00	0.00	0.00
2002	2004	3	6	4349.84	189.42	145.65	73.97	0.00	0.00	0.00
2002	2004	3	7	3940.81	156.66	316.65	63.88	0.00	0.00	0.00
2002	2004	3	8	3403.61	10.42	53.77	61.52	0.00	0.00	0.00
2002	2004	3	9	3277.89	0.00	7.04	60.26	0.00	0.00	0.00
2002	2004	3	10	3210.59	0.00	2.58	59.10	0.00	0.00	0.00
2002	2004	3	11	3148.90	0.00	13.49	57.77	0.00	0.00	0.00
2002	2004	3	12	3077.65	0.00	0.00	56.70	0.00	0.00	0.00
2002	2005	3	1	3020.95	0.00	0.00	55.66	0.00	0.00	0.00
2002	2005	3	2	2965.29	0.00	0.00	54.63	0.00	3.12	2705.25
2002	2005	4	3	202.29	0.00	0.00	3.73	0.00	0.00	0.00
2002	2005	4	4	198.56	0.00	15.06	3.38	0.00	0.00	0.00
2002	2005	4	5	180.12	8.20	0.00	3.17	0.00	0.00	0.00
2002	2005	4	6	168.76	13.28	0.00	2.86	0.00	0.00	0.00
2002	2005	4	7	152.61	19.05	0.00	2.46	0.00	0.00	0.00
2002	2005	4	8	131.10	8.71	0.00	2.25	0.00	0.00	0.00
2002	2005	4	9	120.13	8.75	0.00	2.05	0.00	0.00	0.00
2002	2005	4	10	109.33	0.00	0.00	2.01	0.00	0.00	0.00
2002	2005	4	11	107.32	0.00	4.37	1.90	0.00	0.00	0.00
2002	2005	4	12	101.05	0.00	0.00	1.86	0.00	0.00	0.00
2002	2006	4	1	99.19	0.00	0.00	1.83	0.00	0.00	0.00
2002	2006	4	2	97.36	0.00	0.00	1.79	0.00	1.03	94.54
2002	2006	5	3	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2002	2006	5	4	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2002	2006	5	5	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2002	2006	5	6	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2002	2006	5	7	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2002	2006	5	8	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2002	2006	5	9	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2002	2006	5	10	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2002	2006	5	11	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2002	2006	5	12	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2002	2007	5	1	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2002	2007	5	2	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table D-6. Reconstructed cohort: 2003 brood.

				Ocean				River		
BY	CY	Age	Month	N	I_{com}	I_{rec}	V	H_r	E_{hat}	E_{nat}
2003	2004	2	3	7026.64	0.00	0.00	394.37	0.00	0.00	0.00
2003	2004	2	4	6632.26	0.00	0.00	372.24	0.00	0.00	0.00
2003	2004	2	5	6260.02	0.00	0.00	351.35	0.00	0.00	0.00
2003	2004	2	6	5908.67	0.00	0.00	331.63	0.00	0.00	0.00
2003	2004	2	7	5577.05	0.00	0.00	313.02	0.00	0.00	0.00
2003	2004	2	8	5264.03	0.00	0.00	295.45	0.00	0.00	0.00
2003	2004	2	9	4968.58	0.00	0.00	278.87	0.00	0.00	0.00
2003	2004	2	10	4689.72	0.00	0.00	263.21	0.00	0.00	0.00
2003	2004	2	11	4426.50	0.00	0.00	248.44	0.00	0.00	0.00
2003	2004	2	12	4178.06	0.00	0.00	234.50	0.00	0.00	0.00
2003	2005	2	1	3943.57	0.00	0.00	221.34	0.00	0.00	0.00
2003	2005	2	2	3722.23	0.00	0.00	208.91	0.00	0.00	141.67
2003	2005	3	3	3371.65	0.00	0.00	62.12	0.00	0.00	0.00
2003	2005	3	4	3309.53	0.00	81.20	59.48	0.00	0.00	0.00
2003	2005	3	5	3168.86	0.00	99.43	56.55	0.00	0.00	0.00
2003	2005	3	6	3012.88	33.68	157.09	51.99	0.00	0.00	0.00
2003	2005	3	7	2770.12	76.05	77.48	48.21	0.00	0.00	0.00
2003	2005	3	8	2568.38	34.15	12.01	46.47	0.00	0.00	0.00
2003	2005	3	9	2475.76	2.28	3.59	45.50	0.00	0.00	0.00
2003	2005	3	10	2424.38	0.00	0.00	44.67	0.00	0.00	0.00
2003	2005	3	11	2379.72	0.00	2.01	43.81	0.00	0.00	0.00
2003	2005	3	12	2333.91	0.00	0.00	43.00	0.00	0.00	0.00
2003	2006	3	1	2290.91	0.00	0.00	42.21	0.00	0.00	0.00
2003	2006	3	2	2248.70	0.00	0.00	41.43	0.00	2.02	2092.10
2003	2006	4	3	113.15	0.00	0.00	2.08	0.00	0.00	0.00
2003	2006	4	4	111.07	0.00	5.33	1.95	0.00	0.00	0.00
2003	2006	4	5	103.79	3.11	0.00	1.85	0.00	0.00	0.00
2003	2006	4	6	98.82	0.00	5.52	1.72	0.00	0.00	0.00
2003	2006	4	7	91.57	0.00	10.51	1.49	0.00	0.00	0.00
2003	2006	4	8	79.57	0.00	0.00	1.47	0.00	0.00	0.00
2003	2006	4	9	78.10	1.61	0.00	1.41	0.00	0.00	0.00
2003	2006	4	10	75.09	0.00	0.00	1.38	0.00	0.00	0.00
2003	2006	4	11	73.70	0.00	0.00	1.36	0.00	0.00	0.00
2003	2006	4	12	72.35	0.00	0.00	1.33	0.00	0.00	0.00
2003	2007	4	1	71.01	0.00	0.00	1.31	0.00	0.00	0.00
2003	2007	4	2	69.70	0.00	0.00	1.28	0.00	2.18	62.59
2003	2007	5	3	3.65	0.00	0.00	0.07	0.00	0.00	0.00
2003	2007	5	4	3.58	0.00	0.00	0.07	0.00	0.00	0.00
2003	2007	5	5	3.52	0.00	0.00	0.06	0.00	0.00	0.00
2003	2007	5	6	3.45	0.00	0.00	0.06	0.00	0.00	0.00
2003	2007	5	7	3.39	0.00	0.00	0.06	0.00	0.00	0.00
2003	2007	5	8	3.33	0.00	0.00	0.06	0.00	0.00	0.00
2003	2007	5	9	3.26	0.00	0.00	0.06	0.00	0.00	0.00
2003	2007	5	10	3.20	0.00	0.00	0.06	0.00	0.00	0.00
2003	2007	5	11	3.15	0.00	0.00	0.06	0.00	0.00	0.00
2003	2007	5	12	3.09	0.00	0.00	0.06	0.00	0.00	0.00
2003	2008	5	1	3.03	0.00	0.00	0.06	0.00	0.00	0.00
2003	2008	5	2	2.97	0.00	0.00	0.05	0.00	0.00	2.92

Table D-7. Reconstructed cohort: 2004 brood.

				Ocean				River		
BY	CY	Age	Month	N	I_{com}	I_{rec}	V	H_r	E_{hat}	E_{nat}
2004	2005	2	3	291.71	0.00	0.00	16.37	0.00	0.00	0.00
2004	2005	2	4	275.34	0.00	0.00	15.45	0.00	0.00	0.00
2004	2005	2	5	259.88	0.00	0.00	14.59	0.00	0.00	0.00
2004	2005	2	6	245.30	0.00	0.00	13.77	0.00	0.00	0.00
2004	2005	2	7	231.53	0.00	0.00	12.99	0.00	0.00	0.00
2004	2005	2	8	218.54	0.00	0.00	12.27	0.00	0.00	0.00
2004	2005	2	9	206.27	0.00	0.00	11.58	0.00	0.00	0.00
2004	2005	2	10	194.69	0.00	0.00	10.93	0.00	0.00	0.00
2004	2005	2	11	183.77	0.00	0.00	10.31	0.00	0.00	0.00
2004	2005	2	12	173.45	0.00	0.00	9.74	0.00	0.00	0.00
2004	2006	2	1	163.72	0.00	0.00	9.19	0.00	0.00	0.00
2004	2006	2	2	154.53	0.00	0.00	8.67	0.00	0.00	3.31
2004	2006	3	3	142.55	0.00	0.00	2.63	0.00	0.00	0.00
2004	2006	3	4	139.92	0.00	8.04	2.43	0.00	0.00	0.00
2004	2006	3	5	129.45	0.00	0.00	2.38	0.00	0.00	0.00
2004	2006	3	6	127.06	0.00	4.12	2.27	0.00	0.00	0.00
2004	2006	3	7	120.68	0.00	9.29	2.05	0.00	0.00	0.00
2004	2006	3	8	109.33	0.00	0.00	2.01	0.00	0.00	0.00
2004	2006	3	9	107.32	0.00	0.00	1.98	0.00	0.00	0.00
2004	2006	3	10	105.34	0.00	0.00	1.94	0.00	0.00	0.00
2004	2006	3	11	103.40	0.00	0.00	1.90	0.00	0.00	0.00
2004	2006	3	12	101.49	0.00	0.00	1.87	0.00	0.00	0.00
2004	2007	3	1	99.62	0.00	0.00	1.84	0.00	0.00	0.00
2004	2007	3	2	97.79	0.00	0.00	1.80	0.00	7.62	84.43
2004	2007	4	3	3.94	0.00	0.00	0.07	0.00	0.00	0.00
2004	2007	4	4	3.86	0.00	0.00	0.07	0.00	0.00	0.00
2004	2007	4	5	3.79	0.00	0.00	0.07	0.00	0.00	0.00
2004	2007	4	6	3.72	0.00	0.00	0.07	0.00	0.00	0.00
2004	2007	4	7	3.66	0.00	0.00	0.07	0.00	0.00	0.00
2004	2007	4	8	3.59	0.00	0.00	0.07	0.00	0.00	0.00
2004	2007	4	9	3.52	0.00	0.00	0.06	0.00	0.00	0.00
2004	2007	4	10	3.46	0.00	0.00	0.06	0.00	0.00	0.00
2004	2007	4	11	3.39	0.00	0.00	0.06	0.00	0.00	0.00
2004	2007	4	12	3.33	0.00	0.00	0.06	0.00	0.00	0.00
2004	2008	4	1	3.27	0.00	0.00	0.06	0.00	0.00	0.00
2004	2008	4	2	3.21	0.00	0.00	0.06	0.00	0.00	3.15
2004	2008	5	3	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2004	2008	5	4	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2004	2008	5	5	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2004	2008	5	6	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2004	2008	5	7	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2004	2008	5	8	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2004	2008	5	9	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2004	2008	5	10	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2004	2008	5	11	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2004	2008	5	12	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2004	2009	5	1	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2004	2009	5	2	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table D-8. Reconstructed cohort: 2005 brood.

				Ocean				River		
BY	CY	Age	Month	N	I_{com}	I_{rec}	V	H_r	E_{hat}	E_{nat}
2005	2006	2	3	364.07	0.00	0.00	20.43	0.00	0.00	0.00
2005	2006	2	4	343.64	0.00	0.00	19.29	0.00	0.00	0.00
2005	2006	2	5	324.35	0.00	0.00	18.20	0.00	0.00	0.00
2005	2006	2	6	306.14	0.00	0.00	17.18	0.00	0.00	0.00
2005	2006	2	7	288.96	0.00	0.00	16.22	0.00	0.00	0.00
2005	2006	2	8	272.74	0.00	0.00	15.31	0.00	0.00	0.00
2005	2006	2	9	257.44	0.00	0.00	14.45	0.00	0.00	0.00
2005	2006	2	10	242.99	0.00	0.00	13.64	0.00	0.00	0.00
2005	2006	2	11	229.35	0.00	0.00	12.87	0.00	0.00	0.00
2005	2006	2	12	216.48	0.00	0.00	12.15	0.00	0.00	0.00
2005	2007	2	1	204.33	0.00	0.00	11.47	0.00	0.00	0.00
2005	2007	2	2	192.86	0.00	0.00	10.82	0.00	0.00	1.83
2005	2007	3	3	180.20	0.00	0.00	3.32	0.00	0.00	0.00
2005	2007	3	4	176.88	0.00	0.00	3.26	0.00	0.00	0.00
2005	2007	3	5	173.63	0.00	10.12	3.01	0.00	0.00	0.00
2005	2007	3	6	160.50	0.00	7.50	2.82	0.00	0.00	0.00
2005	2007	3	7	150.18	0.00	14.43	2.50	0.00	0.00	0.00
2005	2007	3	8	133.25	0.00	0.00	2.45	0.00	0.00	0.00
2005	2007	3	9	130.80	0.00	0.00	2.41	0.00	0.00	0.00
2005	2007	3	10	128.39	0.00	0.00	2.37	0.00	0.00	0.00
2005	2007	3	11	126.02	0.00	0.00	2.32	0.00	0.00	0.00
2005	2007	3	12	123.70	0.00	0.00	2.28	0.00	0.00	0.00
2005	2008	3	1	121.42	0.00	0.00	2.24	0.00	0.00	0.00
2005	2008	3	2	119.19	0.00	0.00	2.20	0.00	4.29	112.70
2005	2008	4	3	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2008	4	4	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2008	4	5	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2008	4	6	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2008	4	7	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2008	4	8	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2008	4	9	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2008	4	10	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2008	4	11	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2008	4	12	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2009	4	1	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2009	4	2	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2009	5	3	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2009	5	4	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2009	5	5	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2009	5	6	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2009	5	7	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2009	5	8	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2009	5	9	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2009	5	10	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2009	5	11	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2009	5	12	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2010	5	1	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2005	2010	5	2	0.00	0.00	0.00	0.00	0.00	0.00	0.00

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Table D-9. Reconstructed cohort: 2006 brood.

				Ocean				River		
BY	CY	Age	Month	N	I_{com}	I_{rec}	V	H_r	E_{hat}	E_{nat}
2006	2007	2	3	1228.82	0.00	0.00	68.97	0.00	0.00	0.00
2006	2007	2	4	1159.85	0.00	0.00	65.10	0.00	0.00	0.00
2006	2007	2	5	1094.75	0.00	0.00	61.44	0.00	0.00	0.00
2006	2007	2	6	1033.31	0.00	0.00	58.00	0.00	0.00	0.00
2006	2007	2	7	975.31	0.00	0.00	54.74	0.00	0.00	0.00
2006	2007	2	8	920.57	0.00	0.00	51.67	0.00	0.00	0.00
2006	2007	2	9	868.90	0.00	0.00	48.77	0.00	0.00	0.00
2006	2007	2	10	820.14	0.00	0.00	46.03	0.00	0.00	0.00
2006	2007	2	11	774.11	0.00	0.00	43.45	0.00	0.00	0.00
2006	2007	2	12	730.66	0.00	0.00	41.01	0.00	0.00	0.00
2006	2008	2	1	689.65	0.00	0.00	38.71	0.00	0.00	0.00
2006	2008	2	2	650.94	0.00	0.00	36.53	8.70	3.35	22.35
2006	2008	3	3	580.01	0.00	0.00	10.69	0.00	0.00	0.00
2006	2008	3	4	569.32	0.00	0.00	10.49	0.00	0.00	0.00
2006	2008	3	5	558.83	0.00	0.00	10.30	0.00	0.00	0.00
2006	2008	3	6	548.54	0.00	0.00	10.11	0.00	0.00	0.00
2006	2008	3	7	538.43	0.00	0.00	9.92	0.00	0.00	0.00
2006	2008	3	8	528.51	0.00	0.00	9.74	0.00	0.00	0.00
2006	2008	3	9	518.77	0.00	0.00	9.56	0.00	0.00	0.00
2006	2008	3	10	509.22	0.00	0.00	9.38	0.00	0.00	0.00
2006	2008	3	11	499.84	0.00	0.00	9.21	0.00	0.00	0.00
2006	2008	3	12	490.63	0.00	0.00	9.04	0.00	0.00	0.00
2006	2009	3	1	481.59	0.00	0.00	8.87	0.00	0.00	0.00
2006	2009	3	2	472.72	0.00	0.00	8.71	0.00	6.23	423.12
2006	2009	4	3	34.66	0.00	0.00	0.64	0.00	0.00	0.00
2006	2009	4	4	34.02	0.00	0.00	0.63	0.00	0.00	0.00
2006	2009	4	5	33.39	0.00	0.00	0.62	0.00	0.00	0.00
2006	2009	4	6	32.78	0.00	0.00	0.60	0.00	0.00	0.00
2006	2009	4	7	32.17	0.00	0.00	0.59	0.00	0.00	0.00
2006	2009	4	8	31.58	0.00	0.00	0.58	0.00	0.00	0.00
2006	2009	4	9	31.00	0.00	0.00	0.57	0.00	0.00	0.00
2006	2009	4	10	30.43	0.00	0.00	0.56	0.00	0.00	0.00
2006	2009	4	11	29.87	0.00	0.00	0.55	0.00	0.00	0.00
2006	2009	4	12	29.32	0.00	0.00	0.54	0.00	0.00	0.00
2006	2010	4	1	28.78	0.00	0.00	0.53	0.00	0.00	0.00
2006	2010	4	2	28.25	0.00	0.00	0.52	0.00	0.00	26.83
2006	2010	5	3	NA	NA	NA	NA	NA	NA	NA
2006	2010	5	4	NA	NA	NA	NA	NA	NA	NA
2006	2010	5	5	NA	NA	NA	NA	NA	NA	NA
2006	2010	5	6	NA	NA	NA	NA	NA	NA	NA
2006	2010	5	7	NA	NA	NA	NA	NA	NA	NA
2006	2010	5	8	NA	NA	NA	NA	NA	NA	NA
2006	2010	5	9	NA	NA	NA	NA	NA	NA	NA
2006	2010	5	10	NA	NA	NA	NA	NA	NA	NA
2006	2010	5	11	NA	NA	NA	NA	NA	NA	NA
2006	2010	5	12	NA	NA	NA	NA	NA	NA	NA
2006	2011	5	1	NA	NA	NA	NA	NA	NA	NA
2006	2011	5	2	NA	NA	NA	NA	NA	NA	NA

Table D-10. Reconstructed cohort: 2007 brood.

				Ocean				River		
BY	CY	Age	Month	N	I_{com}	I_{rec}	V	H_r	E_{hat}	E_{nat}
2007	2008	2	3	464.03	0.00	0.00	26.04	0.00	0.00	0.00
2007	2008	2	4	437.99	0.00	0.00	24.58	0.00	0.00	0.00
2007	2008	2	5	413.40	0.00	0.00	23.20	0.00	0.00	0.00
2007	2008	2	6	390.20	0.00	0.00	21.90	0.00	0.00	0.00
2007	2008	2	7	368.30	0.00	0.00	20.67	0.00	0.00	0.00
2007	2008	2	8	347.63	0.00	0.00	19.51	0.00	0.00	0.00
2007	2008	2	9	328.12	0.00	0.00	18.42	0.00	0.00	0.00
2007	2008	2	10	309.70	0.00	0.00	17.38	0.00	0.00	0.00
2007	2008	2	11	292.32	0.00	0.00	16.41	0.00	0.00	0.00
2007	2008	2	12	275.91	0.00	0.00	15.49	0.00	0.00	0.00
2007	2009	2	1	260.43	0.00	0.00	14.62	0.00	0.00	0.00
2007	2009	2	2	245.81	0.00	0.00	13.80	15.72	0.00	0.00
2007	2009	3	3	216.29	0.00	0.00	3.98	0.00	0.00	0.00
2007	2009	3	4	212.31	0.00	0.00	3.91	0.00	0.00	0.00
2007	2009	3	5	208.40	0.00	0.00	3.84	0.00	0.00	0.00
2007	2009	3	6	204.56	0.00	0.00	3.77	0.00	0.00	0.00
2007	2009	3	7	200.79	0.00	0.00	3.70	0.00	0.00	0.00
2007	2009	3	8	197.09	0.00	0.00	3.63	0.00	0.00	0.00
2007	2009	3	9	193.46	0.00	0.00	3.56	0.00	0.00	0.00
2007	2009	3	10	189.90	0.00	0.00	3.50	0.00	0.00	0.00
2007	2009	3	11	186.40	0.00	0.00	3.43	0.00	0.00	0.00
2007	2009	3	12	182.96	0.00	0.00	3.37	0.00	0.00	0.00
2007	2010	3	1	179.59	0.00	0.00	3.31	0.00	0.00	0.00
2007	2010	3	2	176.28	0.00	0.00	3.25	0.00	0.00	163.65
2007	2010	4	3	NA	NA	NA	NA	NA	NA	NA
2007	2010	4	4	NA	NA	NA	NA	NA	NA	NA
2007	2010	4	5	NA	NA	NA	NA	NA	NA	NA
2007	2010	4	6	NA	NA	NA	NA	NA	NA	NA
2007	2010	4	7	NA	NA	NA	NA	NA	NA	NA
2007	2010	4	8	NA	NA	NA	NA	NA	NA	NA
2007	2010	4	9	NA	NA	NA	NA	NA	NA	NA
2007	2010	4	10	NA	NA	NA	NA	NA	NA	NA
2007	2010	4	11	NA	NA	NA	NA	NA	NA	NA
2007	2010	4	12	NA	NA	NA	NA	NA	NA	NA
2007	2011	4	1	NA	NA	NA	NA	NA	NA	NA
2007	2011	4	2	NA	NA	NA	NA	NA	NA	NA
2007	2011	5	3	NA	NA	NA	NA	NA	NA	NA
2007	2011	5	4	NA	NA	NA	NA	NA	NA	NA
2007	2011	5	5	NA	NA	NA	NA	NA	NA	NA
2007	2011	5	6	NA	NA	NA	NA	NA	NA	NA
2007	2011	5	7	NA	NA	NA	NA	NA	NA	NA
2007	2011	5	8	NA	NA	NA	NA	NA	NA	NA
2007	2011	5	9	NA	NA	NA	NA	NA	NA	NA
2007	2011	5	10	NA	NA	NA	NA	NA	NA	NA
2007	2011	5	11	NA	NA	NA	NA	NA	NA	NA
2007	2011	5	12	NA	NA	NA	NA	NA	NA	NA
2007	2012	5	1	NA	NA	NA	NA	NA	NA	NA
2007	2012	5	2	NA	NA	NA	NA	NA	NA	NA

Agenda Item C.1.a
Attachment 3
November 2011

The winter-run harvest model (WRHM)

DRAFT REPORT

Michael R. O'Farrell

Shanae Allen

Michael S. Mohr

Fisheries Ecology Division

Southwest Fisheries Science Center

National Marine Fisheries Service

110 Shaffer Road, Santa Cruz, CA 95060

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1 Abstract

The Winter Run Harvest Model (WRHM) forecasts the annual age-3 ocean impact rate on Sacramento River winter Chinook resulting from fisheries south of Point Arena, CA. This impact rate includes both landed and non-landed mortality attributable to fisheries. The model is a tool developed for use in the Pacific Fishery Management Council (PFMC) arena for managing fisheries to comply with the National Marine Fisheries Service Endangered Species Act consultation standard for Sacramento River winter Chinook beginning in 2012. Analogous to other models used for assessment and management of salmon through the PFMC process, the WRHM is temporally and spatially stratified. Impact rates are forecast for each month, area, and sector (commercial, recreational) to capture variation in exploitation patterns and fishery management measures that occur at that scale. A forecast of the total age-3 impact rate is then made by aggregating impacts over all strata where fishing occurred. The WRHM is capable of accounting for the customary fishery management measures used by the PFMC (e.g., time/area/sector closures, quotas, and minimum size limits). Hence, the WRHM will readily integrate into the PFMC salmon management process.

2 Introduction

Sacramento River winter Chinook (SRWC) is an endangered salmon stock harvested incidentally in ocean fisheries. SRWC were first listed as threatened in 1989, and then downgraded to endangered in 1994. Most recently, in the 2010 Biological Opinion for ocean fisheries (NMFS 2010), the National Marine Fisheries Service (NMFS) found that ocean fisheries are likely to jeopardize the continued existence of SRWC owing to a lack of measures and tools to constrain or reduce fishery impacts when SRWC population status is poor. NMFS offered a reasonable and prudent alternative (RPA) to comply with the ESA, which included (1) establishing thresholds related to the status of SRWC, (2) establishing fishery management objectives, and (3) development of analytical tools and assessment models that can implement the fishery management objectives in the salmon fishery management process. This report documents one portion of component (3): the Winter Run Harvest Model (WRHM).

Development of the new SRWC fishery management objectives is in progress and the final form of the ocean fishery management framework rule is not known as of September 2011. However, some aspects of the framework are known at this time. In particular, a control rule will annually specify a maximum allowable age-3 ocean fishery impact rate, and this age-3 impact rate will apply only to fisheries occurring south of Point Arena, California. The impact rate includes both landed and non-landed mortality attributable to fisheries, and the region covered includes the San Francisco (SF) management area (Point Arena to Pigeon Point), and the Monterey (MO) management area (Pigeon Point to the US/Mexico border).

For SRWC, the age-3 ocean fishery impact rate is an appropriate metric for use in controlling overall fisheries exploitation. The age-3 impact rate closely approximates the cohort's spawner reduction rate, which is the fraction of a cohort's potential spawners that are eliminated by the fishery (see Figure 5 in O'Farrell et al. 2011). The concordance between the age-3 impact rate and the spawner reduction rate is due to the very high ($>$ 85 percent) age-3 maturation rates SRWC exhibit (O'Farrell et al. 2011). In addition, the age-3 impact rate can be forecast in the absence of a SRWC preseason abundance forecast. A preseason abundance forecast cannot be made for SRWC

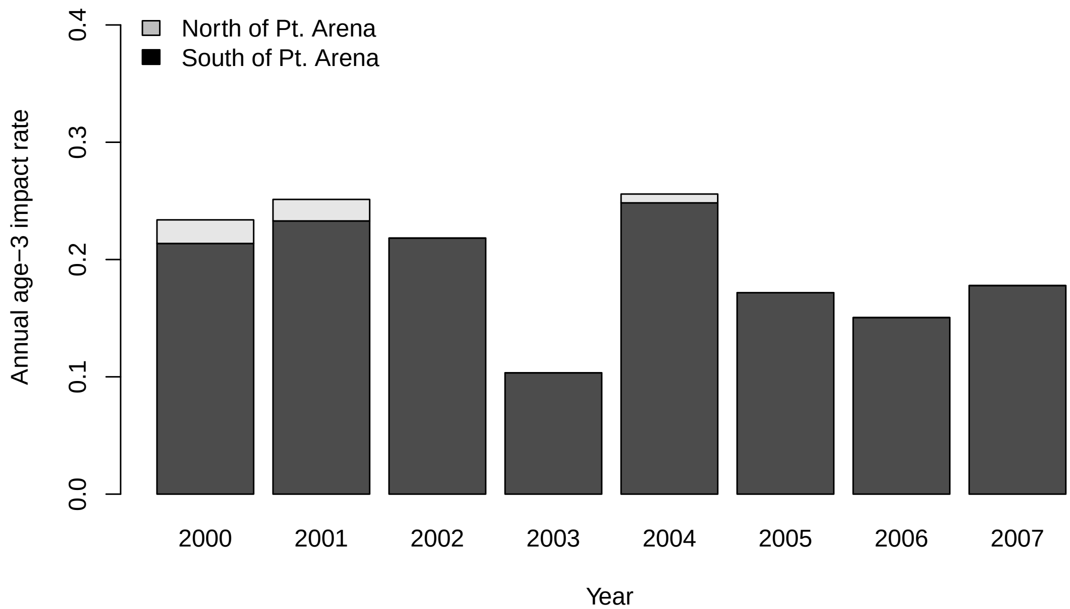

Figure 1. Age-3 ocean fishery impact rate, partitioned by contributions from fisheries north and south of Point Arena, CA.

in time for the PFMC preseason management process due to the timing of SRWC spawning and the timing of annual ocean salmon fisheries. The age-2 (jack) river return data that would be necessary to forecast age-3 abundance prior to spring/summer ocean salmon fisheries are not available until the fall or winter following those fisheries, and therefore are not useful for making a timely age-3 ocean abundance forecast.

Forecasts of the age-3 impact rate will be confined to fisheries occurring in management areas south of Point Arena because the overwhelming majority of SRWC impacts occur in this region. Figure 1 demonstrates that in most years for which the age-3 impact rate has been estimated, zero impacts resulted from fisheries north of Point Arena, and when they did occur, they represented a very small portion of the overall age-3 impact rate. Between years 2000 and 2007, the age-3 impact rate attributed to fisheries north of Point Arena averaged 0.0058.

The WRHM consists of projecting an age-3 cohort abundance through ocean fisheries on a monthly basis between March 1 (year y) and the last day of February ($y+1$). The starting abundance is arbitrary and does not affect the forecast of the annual age-3 impact rate; hereafter we assume that the March 1 (y) ocean abundance is equal to 1. March 1 was chosen as the "birth date" for SRWC, based on the reported peak migration period into the Sacramento River basin from Fisher (1994). Monthly age-3 impacts, forecast by area (SF, MO), and sector (commercial, recreational) under the proposed fishery management measures, are deducted from the monthly abundance. The total, age-3 impact rate is then computed by totaling the month/area/sector impacts and dividing by the assumed March 1 (y) ocean abundance. The WRHM is able to accommodate days-open (fisheries specified as the number of days open to fishing and not as a harvest limit) and quota fishery management measures, with one exception that is explained in more detail in section 4.2. The WRHM is also able to account for variation in minimum size limits. Hence, management measures such as month/area/sector closures and minimum size limits commonly used by the PFMC to constrain the salmon fishery can be directly accounted for in the WRHM-derived forecast of the age-3 impact rate.

Documentation of the WRHM follows in sections 3 and 4. Section 3 defines the main model structure and methods used to project the age-3 cohort through ocean fisheries, and the expression used to forecast the age-3 impact rate. Section 4 describes the submodels and input variables used to parameterize the WRHM. The report ends with a discussion of key components of the model and a comparison to existing PFMC harvest models for Chinook salmon.

3 Main model

The age-3 SRWC cohort abundance is projected through ocean fisheries sequentially from $t =$ March (y) through $t =$ February ($y+1$). The method of forward projection of the age-3 cohort is consistent with the backward reconstruction of cohorts described for SRWC in O'Farrell et al. (2011).

For each month t, the following metrics are computed by area z and sector x:

$$C_{tzx} = c_{tzx} \times N_t \tag{1}$$

$$H_{tzx} = C_{tzx} \times p_{tzx} \tag{2}$$

$$S_{tzx} = (C_{tzx} - H_{tzx}) \times s_{tzx} \tag{3}$$

$$D_{tzx} = C_{tzx} \times d \tag{4}$$

$$I_{tzx} = H_{tzx} + S_{tzx} + D_{tzx}, \tag{5}$$

with cohort abundance (N), contacts (C), harvest (H), release mortality (S), dropoff mortality (D), and impacts (I) dependent on the contact rate (c), the proportion of fish that are greater than or equal to the minimum size limit (p), the release mortality rate (s), and the dropoff mortality rate (d). Because the model confines itself to age-3, we have for simplicity suppressed the use of a subscript denoting age for these quantities.

To project the cohort abundance forward in one month increments, total monthly impacts

$$I_t = \sum_{z,x} I_{tzx} \tag{6}$$

are first deducted from N_t, followed by application of the monthly natural survival rate

$$N_{t+1} = (N_t - I_t) \times (1 - v), \tag{7}$$

where v denotes the monthly natural mortality rate.

Following projection of the cohort abundance across months, the age-3 impact rate (i_3) is forecast as

$$i_3 = \frac{\sum_t I_t}{N_{\text{March}}}. \tag{8}$$

In practice, N_{March} is specified as 1, and i_3 reduces to the numerator in (8).

In the following section, the submodels and input variables used to parameterize the c, p, s, and d rates are described.

4 Submodels and input variables

4.1 Contact rate

Age-3 month/area/sector contact rates have been estimated for years 2000–2009 through cohort reconstruction (O'Farrell et al. 2011). Pairing postseason estimates of c_{tzx} with postseason fishing effort estimates f_{tzx} allows for forecasting the contact rate per unit effort (β_{tzx}), and ultimately, the contact rate expectation in proposed fisheries.

Forecasts of β_{tzx} are determined by the slope of a zero-intercept linear model fitted to historical c_{tzx} and f_{tzx} data. Figures 2 and 3 displays these relationships for the commercial and recreational sectors, respectively. A ratio estimator is used to determine the month/area/sector contact rate per unit effort forecast,

$$\beta_{tzx} = \frac{\bar{c}_{tzx}}{\bar{f}_{tzx}} \tag{9}$$

following the methodology used for the KOHM (Mohr 2006a), where $\bar{c}_{tzx}$ and $\bar{f}_{tzx}$ denote the respective average of these quantities over the historical data.

Expected contact rates are then forecast using β_{tzx} and the effort forecast for that month/area/sector:

$$c_{tzx} = \beta_{tzx} \times f_{tzx}, \tag{10}$$

with the effort forecast determined as described in the following section.

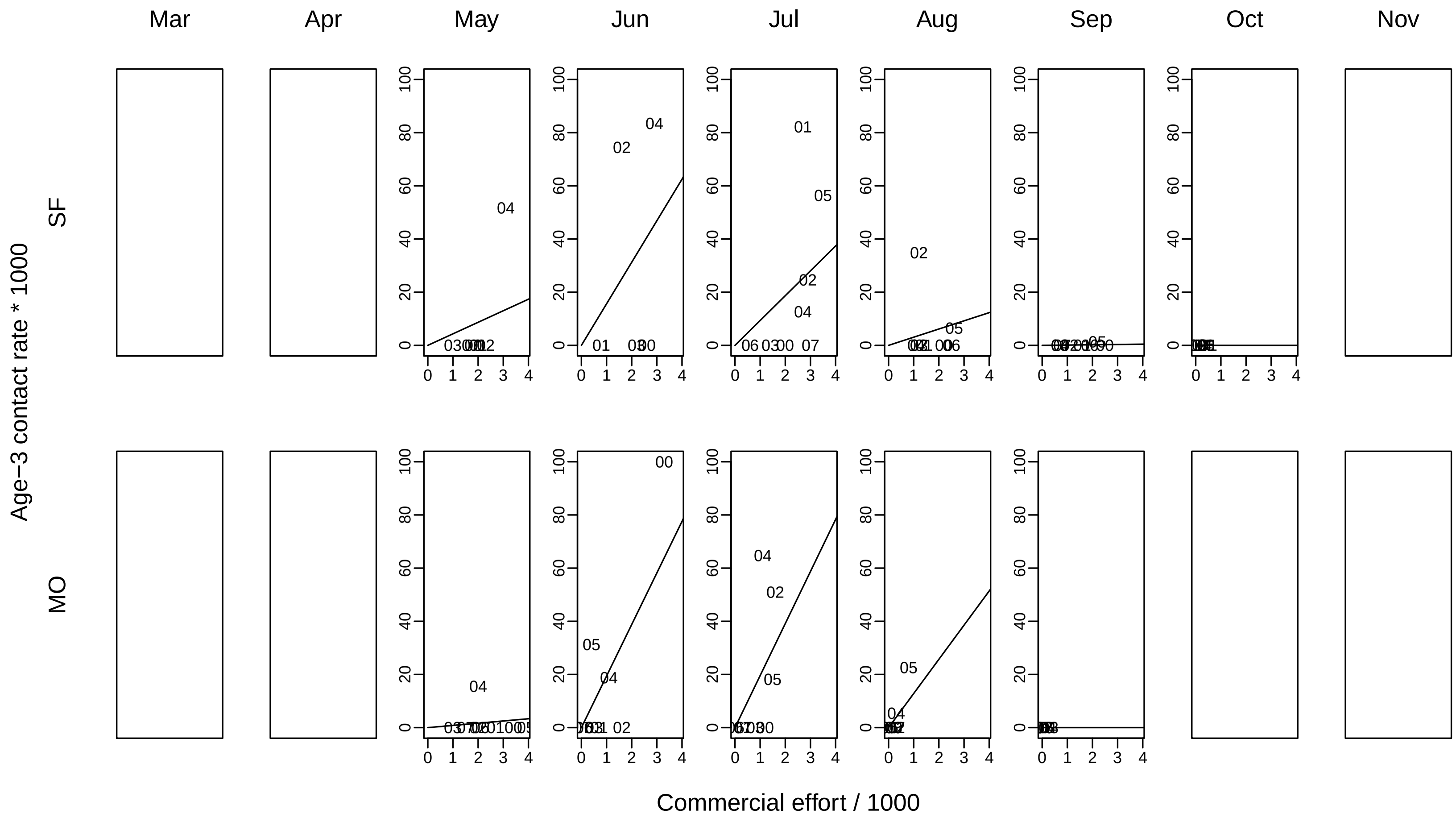

Figure 2. Commercial sector age-3 contact rates, plotted as a function of fishing effort, by month and management area. Numbers in the plot are the last two digits of the calendar year for the contact rate and effort estimates.

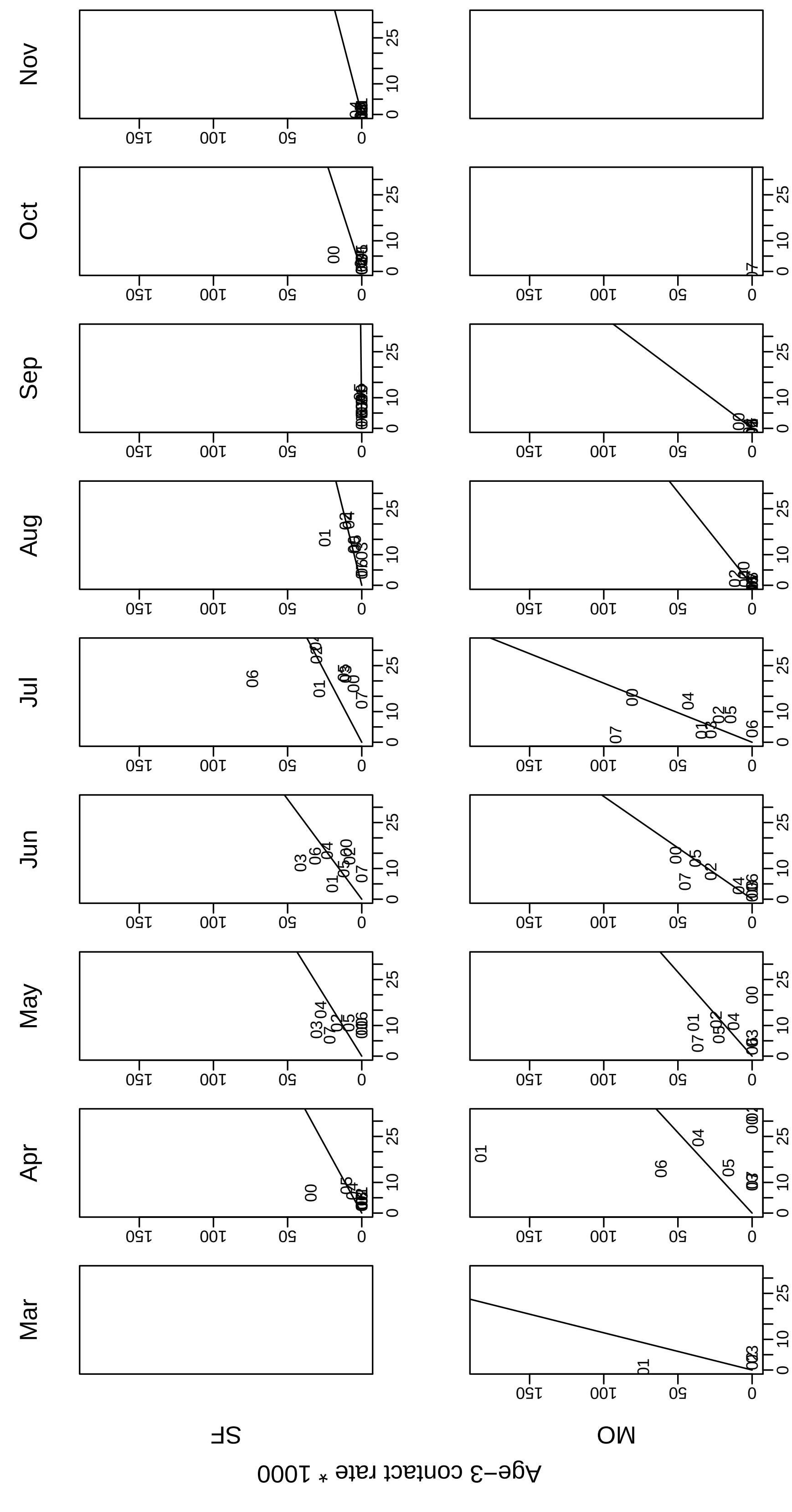

Figure 3. Recreational sector age-3 contact rates, plotted as a function of fishing effort, by month and management area. Numbers in the plot are the last two digits of the calendar year for the contact rate and effort estimates.

4.2 Fishing effort

Fishing effort is forecast for each month/area/sector external to the WRHM. Effort forecasts are necessary inputs for the Klamath Ocean Harvest Model (KOHM) and Sacramento Harvest Model (SHM), as well as the WRHM, hence they are shared across models. Fishing effort forecast methods for both days-open fisheries and quota fisheries are documented in Mohr (2006a,b).

As described in Mohr (2006a), quota fishery effort is forecast in a different manner than days-open fishery effort. Effort expected in a quota fishery is determined by the size of the mixed-stock quota and the stock contribution rate of abundant target stocks (e.g., Klamath and Sacramento River fall Chinook). For quota fisheries occurring between September (y) and February ($y+1$), the stock contribution rate of these target stocks is not known at the time of the PFMC preseason salmon management process because the ocean abundance of these fall run stocks has not yet been forecast (O'Farrell 2009). As a result, it is currently not possible to forecast effort in quota fisheries for these months. Note, however, that quota fisheries in the SF and MO area during this period have been extremely rare.

4.3 Proportion legal size

Determination of p_{tzx} requires a specified minimum size limit (l^*_{tzx}), and the mean (μ_t) and standard deviation (σ_t) of the length distribution of age-3 SRWC in the ocean for month t. Minimum size limits are specified for nearly all ocean fisheries, and are a standard input to the WRHM. The model used to estimate monthly size-at-age is described in O'Farrell et al. (2011, Appendix A). Size-at-age in month t is assumed to be normally distributed so that given l^*_{tzx}, μ_t, and σ_t,

$$p_{tzx} = P\{l \geq l^*_{tzx} | \mu_t, \sigma_t\} = 1 - \Phi(l^*_{tzx} | \mu_t, \sigma_t), \tag{11}$$

where $P\{A\}$ denotes the probability of event A, and $\Phi(\cdot)$ is the cumulative probability distribution function for the normal distribution.

4.4 Release mortality rate

Based on the Salmon Technical Team (STT) review of hook and release mortality rates (STT 2000), we employ the conventional rate values of $s_{tz(\text{com})} = 0.26$ for the commercial sector, and $s_{tz(\text{rec})} = 0.14$ for the recreational sector when the method of fishing is exclusively trolling.

However, for the recreational sector, if the method of fishing known as "mooching" is used in addition to trolling in a particular month/area, then $s_{tz(\text{rec})}$ is formulated as a weighted average of the troll release mortality rate (0.14) and the elevated mooch release mortality rate (0.422) (Grover et al. 2002). Mooching is a fishing technique that consists of drifting whole bait, encourages swallowing of the bait, and results in a high proportion of these fish being gut-hooked (hence the high release mortality rate). Mooching is popular in the SF and MO areas, but its use varies by month/area. Denoting by $\bar{\rho}_{tz}$ the 5-year average of the month- and area-specific proportion of the recreational catch taken by mooching, the $s_{tz(\text{rec})}$ forecast for the SF and MO areas is derived as

$$s_{tz(\text{rec})} = (\bar{\rho}_{tz} \times 0.422) + ((1 - \bar{\rho}_{tz}) \times 0.14). \tag{12}$$

Grover et al. (2002) presents details pertaining to the parameterization of this relationship.

4.5 Dropoff mortality rate

Fish that contact fishing gear yet are not brought to the boat may experience dropoff mortality. This source of mortality could result from a variety of causes, such as predation events or wounds inflicted by the fishing gear. Following STT (2000), we employ the conventional rate value of $d = 0.05$.

4.6 Natural mortality rate

The natural mortality annual rate is assumed to be 20 percent, and this corresponds to a monthly rate value of $v = 0.0184$. This is consistent with values used in the assessment of other Pacific salmon (e.g., Goldwasser et al. 2001; Mohr 2006a).

5 Discussion

We have formulated a harvest model capable of forecasting the annual age-3 impact rate for SRWC, given a proposed set of ocean salmon fishery management measures. This model will be used as a tool to meet, in expectation, maximum allowable age-3 impact rates specified by the SRWC consultation standard. The PFMC will have the customary fishery management controls of time/area/sector closures, spring/summer quotas, and minimum size limits available to meet the SRWC objectives.

Key inputs to the WRHM such as fishing effort and contact rates are based on relationships that utilize new information as it becomes available. Cohort reconstructions will be performed annually, providing new data that will be incorporated into the WRHM each year. This process allows the model to integrate changes in effort or exploitation patterns should they occur.

Contact rate forecasts are a very important component of the WRHM. For the commercial sector, examination of contact rate and effort relationships illustrate the relative rarity of coded-wire tagged age-3 SRWC harvest. In part, this can be explained by the low abundance of SRWC relative to target stocks such as Sacramento River fall Chinook. In addition, for the spring and summer months, a large proportion of age-3 SRWC are smaller than typical commercial minimum size limits (O'Farrell et al. 2011, table A-2) and therefore landed catch is low. These factors contribute to the many instances of zero contact rates, with occasional nonzero estimates for most month and area strata. This pattern is not evident for recreational fisheries, where age-3 SRWC become largely vulnerable to retention in the spring, and nearly all are vulnerable in the summer, given typical recreational sector size limits. As a result, fewer zero contact rate estimates exist, and patterns in contact rates per unit effort are more clearly evident for the recreational sector.

Since 2004, recreational fisheries south of Point Arena have been required to open no earlier than the first Saturday in April, yet these fisheries traditionally opened in mid-February. Sufficient data do not exist to allow for robust contact rate estimation in February and March. Resumption of these early fisheries would result in highly uncertain forecasts of age-3 impact rates for those months because β_{tzx} would need to be assumed rather than directly estimated. Because of SRWC river return timing, it is likely that β_{tzx} is high in February and March (see Figure 11 in O'Farrell et al. 2011). This potential problem also exists for the commercial sector if fisheries prior to May 1 are proposed, and for both sectors if fisheries are proposed for late-fall or winter, when contact rate estimates are sparse or nonexistent.

The WRHM shares many structural similarities to existing PFMC harvest models for Chinook salmon, and to the KOHM in particular. Like the KOHM, the WRHM is an age-structured model, though it only accounts for one age class. It is linked to a cohort reconstruction model with the same structure, which is updated annually. A size-at-age model is incorporated into both the KOHM and WRHM to allow for forecasting of release mortality incurred by sublegal size fish. Contact rates per unit effort are forecast in the same manner. Finally, many of the same conventions for s, d, and v are shared across models. In contrast to the existing harvest models, the WRHM does not account for river fisheries as SRWC are rarely harvested in the Sacramento River. Most importantly, neither preseason ocean abundance forecasts nor spawner escapement forecasts are made by the WRHM. As such, the WRHM can be considered a simplified harvest model in the same family as the KOHM and SHM.

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Application of Bias-corrected Methods for Estimating Mortality in Mark-selective Fisheries to Coho FRAM

Robert H. Conrad
Northwest Indian Fisheries Commission
6730 Martin Way East
Olympia, WA 98516

and

Angelika Hagen-Breaux
Washington Department of Fish and Wildlife
600 Capitol Way North
Olympia, WA 98502

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Abstract

The current Fishery Regulation Assessment Models (FRAM) used in the Pacific Fishery Management Council's pre-season planning process to project mortalities during proposed coho and Chinook salmon fisheries underestimate the number of unmarked mortalities occurring in mark-selective fisheries and concurrent non-selective fisheries. This is a concern because all natural (wild) stocks are unmarked. The bias is caused by unmarked fish encountering fishing gear more than once in a modeled time step due to the required release of all unmarked fish and by the change in the unmarked-to-marked fish ratio resulting from the selective removal of marked salmon by the mark-selective fishery. The bias-correction methods proposed by Conrad and Yuen (2010) were applied to the 2009 and 2010 Coho FRAM final preseason runs to assess the amount of bias introduced by FRAM's computational structure and algorithms. The model was implemented with no catch and release (CNR) mortalities, no drop-off mortalities, and no mark misidentification errors to simplify the interpretation of results.

At current levels of exploitation for coho salmon, the bias in the FRAM estimate of the <u>total</u> exploitation rate of unmarked stocks is minimal. The mean and median differences by which FRAM underestimated the total exploitation rate for an unmarked stock were -0.003 in 2009 and -0.002 in 2010. For the 2009 and 2010 FRAM preseason runs, there was only a single instance where the difference between the FRAM estimate of the total exploitation rate for an unmarked stock and the bias-corrected estimates was ≥ -0.01 (-0.015 for Area 12A wild stock in 2009).

However, bias-correction is important when considering exploitation rate guidelines (limits) for coho stocks of concern. Although in the 2009 and 2010 preseason runs there were no instances where the bias-corrected estimate of total exploitation for a stock of concern exceeded its guideline (when CNR mortalities, drop-off mortalities, and mark misidentification errors were not included), the potential exists. For example, in 2009 the FRAM estimate for the total exploitation rate on the Upper Fraser River Wild stock (Thompson River coho with an exploitation rate guideline of 0.10 in southern US fisheries) of 0.094 was increased to 0.097 after bias-correction. If current FRAM projections of total exploitation rate for a stock of concern are very near a guideline, then there is a very real possibility that the bias-corrected estimate may exceed that guideline.

It is recommended that:

1. The bias-correction methodology is incorporated into Coho FRAM.
2. The implementation of bias-correction into FRAM be evaluated by comparing results from bias-corrected FRAM to bias-corrected results calculated outside the model as was done for this report.
3. The effects of adding in additional sources of mortality not included in the bias-correction evaluation to date (i.e., CNR mortality, drop-off mortality, and mark-recognition errors) be evaluated.
4. A process similar to that used to evaluate bias-correction in Coho FRAM be considered for Chinook FRAM.

Introduction

Mark-selective fisheries for coho salmon (*Oncorhynchus kisutch*) were introduced as a management tool in 1998 off the Washington coast (PFMC 1999a). Regulations for mark-selective fisheries permit the retention of legal-size coho which have had their adipose fin removed (marked) and require the release of all coho salmon with an adipose fin (unmarked) that are brought to the boat. The objective of mark-selective fisheries is to provide meaningful fisheries on abundant (marked) hatchery salmon while reducing the impact on wild salmon.

The Fishery Regulation Assessment Model (FRAM) is used by the Pacific Fishery Management Council (PFMC) during the pre-season planning process to project mortalities during proposed coho and Chinook salmon fisheries. FRAM is a single-pool, deterministic model that has discrete time steps that vary in length from one month to several months (PFMC 2008a). All fisheries during a time step are assumed to operate simultaneously on a single pool of fish. The pool of modeled fish consists of all stocks that have been caught historically in the fishery as estimated from coded-wire tag (CWT) recoveries (Nandor et al. 2010). Historical exploitation rates estimated from CWTs recovered during a base period when salmon abundances were relatively high and fisheries were widely distributed in both time and area are the basis for the FRAM predictions of fishery mortalities by stock (PSC 2005). Details for the methods and algorithms used in FRAM are presented in PFMC (2008b). PFMC (2007a and 2007b) provides a description of the base-period data used for the coho FRAM.

Prior to the implementation of mark-selective fisheries, a key FRAM assumption was that the exploitation rate for specific tagged salmon stocks (sometimes called indicator stocks) was representative of the exploitation rate for unmarked (typically wild) and marked stocks with similar life histories and ocean distributions. With the advent of mark-selective fisheries, the model was restructured so that the exploitation rates for these tagged indicator stocks were used to estimate the encounter rates in mark-selective fisheries for the unmarked stocks that they represent (PFMC 2008b). These encounter rates are used to produce stock-specific estimates of the number of encounters of unmarked fish in a mark-selective fishery which, combined with an estimate of the release-mortality rate, provide estimates of the mortalities due to the catch and release of unmarked salmon. In FRAM, the exploitation rate on the unmarked stock is a linear function of the exploitation rate on the tagged indicator stock used to represent the unmarked stock and the release-mortality rate. Since all marked fish encountered die, the exploitation rate of the tagged indicator stock is synonymous with the exploitation rate of the marked stock. Therefore, the exploitation rate calculation for an unmarked stock in FRAM can also be described as the exploitation rate of the marked stock component (ER_M) multiplied by the release mortality rate (δ).

Lawson and Sampson (1996) demonstrated that in a mark-selective fishery, the actual mortality rate of unmarked fish is an increasing function of the apparent harvest rate on the marked fish. This causes the total number of unmarked mortalities in mark-selective fisheries to be underestimated by models relying on the linear relationship between exploitation rate and release-mortality rate.

Conrad and Yuen (2010) described a bias correction method where the unbiased exploitation rate of the selectively exploited unmarked stock (ER_U) can be computed as an exponential function of the encounter rate of the corresponding marked stock component and the release mortality rate (δ):

$$ER_U = 1 - (1 - ER_M)^{\delta}. \qquad [1]$$

FRAM's computational structure poses some challenges to applying the bias-corrected equation. As mentioned above, all fisheries occurring in a time step operate simultaneously on a single pool of fish.

Therefore, ER_M is computed as the sum of the marked exploitation rates[1] of all fisheries affecting a stock in a given time step. These fisheries can have a range of release mortality rates for the unmarked stock component. The bias-correction procedure used in this analysis is a modification of that proposed by Conrad and Yuen (2010) as suggested by Hagen-Breaux. Specifically, the total exploitation rate in all fisheries (both non-selective and mark-selective) for the marked component of the stock is used in equation 7 of Conrad and Yuen (2010) and a weighted release-mortality rate (equations 8 and 9) is calculated using 1.00 as the release-mortality rate for non-selective fisheries (NSF).

While previous work was focused on developing methods to compute unbiased mark-selective exploitation rates, the purpose of this report is to assess the magnitude of the bias arising from FRAM's computational structure and algorithms. Bias-correction methods for FRAM estimates of stock-specific exploitation rates on unmarked stocks were applied to the final 2009 and 2010 Coho FRAM preseason model runs (runs C0921 and C1016, respectively). The model was implemented with no CNR mortalities, no drop-off mortalities, and no mark misidentification errors. These adjustments were not included so that the bias resulting specifically from exploitation rates and release mortality rates could be assessed. FRAM's PopStat report was used as the source for stock cohort sizes and exploitation rates at each time step and within each time step ("After Natural Mortality" and "After Pre-terminal").

The years 2009 and 2010 were quite different in coho abundance forecasts thus providing an interesting contrast for comparing results. While the total predicted abundance of British Columbia and Puget Sound stocks was similar between these years, some individual stocks varied considerably. However, it is primarily Columbia River stocks that support the Oregon and Washington ocean mark-selective fisheries and for those stocks the predicted abundance of Columbia River and the US coastal stocks was much greater in 2009 than 2010. Total preseason abundance (sum of marked and unmarked fish) for Southern US stocks (no Canadian or Alaskan stocks) was predicted to be 2.81 million in 2009 compared to 1.87 million in 2010. Table 1 summarizes preseason abundance forecasts in each year.

Table 1. Summary of preseason abundance forecasts in the 2009 and 2010 FRAM preseason runs.

Year	Stock Group	Marked	Unmarked	Total
2009	Canadian and Southern US Stocks	1,723,617	2,108,322	3,831,939
	Southern US Only	1,648,857	1,156,796	2,805,653
	Columbia River, WA Coast, and OR Coast	1,275,357	782,904	2,058,261
2010	Canadian and Southern US Stocks	1,020,556	2,617,378	3,637,934
	Southern US Only	919,854	948,834	1,868,688
	Columbia River, WA Coast, and OR Coast	563,956	524,081	1,088,037

[1] For a marked or unmarked stock component, a time-step specific exploitation rate uses all fishery-related mortalities occurring in the time step (harvest plus release mortalities from mark-selective fisheries) for the numerator and the cohort abundance "After Natural Mortality" for the time step as the denominator (see Appendix A).

Methods

For each unmarked stock with a preseason abundance > 0, the time-step specific fishery exploitation rate (ER) for the marked component of the stock is the basis for the bias-corrected calculation of the exploitation rate for the unmarked stock. Since all marked fish encountered are assumed killed in a selective fishery, marked stock exploitation rates are a convenient surrogate for the encounter rates used by FRAM to compute the exploitation rates for the selectively exploited unmarked stock components. However, not all FRAM stocks have a marked stock equivalent. In these cases, the encounter rates can be computed using FRAM's base period exploitation rates and current year fisheries scalars:

$$ER_{M_{S,T}} = \sum_F (BPER_{S,F,T} \ x \ FishScalar_{F,T}) \qquad [2]$$

where,

BPER	=	base period exploitation rate for stock (*S*), fishery (*F*), and time step (*T*), and
FishScalar	=	fishery effort scalar for fishery and time step.

The procedure used to calculate the bias in the FRAM estimate of an exploitation rate for an unmarked stock is described in detail below.

Bias-correction Procedure:

Time Step 1
Step 1: Calculate a weighted release-mortality rate *(δ_W)* for the unmarked stock specific to the time step. As was demonstrated in Conrad and Yuen (2010), this can be calculated as the ratio of the FRAM ER for the unmarked stock ($\widetilde{ER}_U$)[2] to the FRAM ER for its marked stock component ($\widetilde{ER}_M$). For the time-step specific calculation,

$$\delta_W = \frac{\widetilde{ER}_U}{\widetilde{ER}_M} \qquad [3]$$

where both exploitation rates are time-step specific ERs based on cohort sizes in that time step after natural mortality. $\widetilde{ER}_M$ was extracted from a table containing time-step specific estimates of the exploitation rate for the marked component of the stock calculated from FRAM base period data as previously described. $\widetilde{ER}_U$ was calculated by first estimating unmarked fishery mortalities in the time step by subtracting PopStat's "After Preterminal" cohort size from the "After Natural Mortality" cohort size. This difference (the mortalities due to fisheries) was then divided by the "After Natural Mortality" cohort size.

Step 2: Calculate a bias-corrected, time-step specific, exploitation rate for the unmarked stock

$$\widehat{ER}_U = 1 - \left(1 - \sum \widetilde{ER}_M\right)^{\delta_W} . \qquad [4]$$

Step 3: Calculate the bias-corrected "After pre-terminal" fishery cohort abundance for the unmarked stock ($\widehat{N}_U$)

$$\widehat{N}_U = N_U \ x \left(1 - \widehat{ER}_U\right) \qquad [5]$$

where N_U is the cohort size after natural mortality for the time step.

[2] ~ is used to indicate a FRAM calculation and ^ to indicate a bias-corrected calculation.

Step 4: Calculate the bias-corrected number of fishery mortalities for the unmarked stock ($\widehat{D}_U$). This is simply the difference between the N_U and $\widehat{N}_U$,

$$\widehat{D}_U = N_U - \widehat{N}_U. \qquad [6]$$

The relative bias ($\widehat{B}$) of the FRAM exploitation rate for the unmarked stock was calculated as:

$$\widehat{B} = \frac{\widetilde{ER}_U - \widehat{ER}_U}{\widehat{ER}_U}\ x\ 100\%. \qquad [7]$$

A negative bias indicates that FRAM underestimated the true exploitation rate for the unmarked stock. The difference (Δ) between the two calculations of the ER (as defined in the numerator above) was examined, also.

Time Steps 2 through 5

The same four-step procedure is followed in subsequent time steps (2, 3, 4, and 5) with the following modification. In each new time step *i*, the new cohort size for the unmarked stock is the bias-corrected "After pre-terminal" fishery cohort abundance ($\widehat{N}_U$) calculated in the previous step (*i* - 1). This becomes the starting cohort size for the time step from which natural mortality is subsequently removed and then steps 1 through 4 implemented.

The bias-corrected total exploitation rate for an unmarked cohort is calculated as the sum of the bias-corrected fishery mortalities in each time step divided by those summed mortalities plus escapement (with the escapement re-estimated based on the bias-corrected mortalities). Appendix A provides an example of these calculations.

Summary Statistics:

Basic summary statistics for $\widehat{B}$ (relative bias of exploitation rate estimates) and Δ (absolute difference of exploitation rate estimates) were estimated for Washington, Oregon, and California origin stocks in each time step. Each stock was placed in a regional grouping so that differences by regions could be examined. Regions and number of unmarked stocks in each region are summarized in Table 2. Appendix B describes the specific stocks in each regional grouping. Box-and-whiskers plots were used to compare $\widehat{B}$ and Δ across time steps and regional groupings by time step.

Table 2. Definition of regions and number of unmarked stocks in each region.

Region Label	Description	Number of Stocks
NPS	North Puget Sound	13 (14[a])
MPS	Mid Puget Sound	9
SPS	South Puget Sound	8
HC	Hood Canal	7 (8[a])
SJF	Strait of Juan de Fuca	6
WAC	Washington Coast	15
CR	Columbia River	8
ORC	Oregon and California	9

[a] Stocks in 2010 model run.

Coho Stocks of Concern:

There are four coho "stocks of concern" in the PFMC management process which have ER guidelines that often constrain fisheries. The four stocks (or stock groups) are (with their ER guideline):

- Interior Fraser River Natural – 10 percent total ER in southern US fisheries,
- Lower Columbia River Natural (2009) – 20 percent total ER in marine and mainstem Columbia River fisheries), (2010) – 15 percent total ER in marine and mainstem Columbia River fisheries)
- Oregon Coastal Natural – 15 percent total ER, and
- Southern Oregon | Northern California Coast – 13 percent total ER.

The Interior Fraser River (also referred to as Thompson River) Natural ER is measured by a single unmarked stock (Upper Fraser River). Lower Columbia River Natural ER is measured by combining abundance and impacts for three stocks (Columbia River Wild Unmarked - Oregon, Columbia River Early Wild Unmarked – Washington, and Columbia River Late Wild - Washington). Oregon Coastal Natural (OCN) ER is also measured by combining abundance and impacts for three stocks (Oregon North Coastal Wild, Oregon North Mid Coastal Wild, and Oregon South Mid Coastal Wild). The southern Oregon|Northern California Coast ER is measured using the exploitation rate for the unmarked component of two hatchery stocks as surrogates (Oregon South Coast Hatchery and California North Coast Hatchery). The relative biases and differences between the exploitation rate estimates for these stocks, or stock aggregates, were examined separately so they could be compared to their ER guidelines. This was done on a cumulative FRAM ER basis (with a fixed denominator consisting of the sum of the catch over all time steps plus escapement) and on a time-step specific basis.

The Lower Columbia River Natural ER guideline is for marine fisheries combined with mainstem Columbia River fisheries. For this stock aggregate, only the marine impacts (including Buoy 10 sport) are calculated in the FRAM model. The total ER guideline for this stock is calculated outside the model and combines FRAM mortalities with impacts from mainstem Columbia River fisheries. Thus the total ER from FRAM fisheries is always well below the guideline to allow the mainstem Columbia River fisheries to occur. It is important to note that for this stock the ER guideline is always reached, as these river fisheries are structured to fish up to the total ER guideline.

Results

In 2009, there were 75 unmarked stocks for which bias-corrected exploitation rates were calculated[3]. In 2010, there were 77 unmarked stocks for which bias-corrected exploitation rates were calculated[3].

Relative Bias ($\hat{B}$):[4]

In both years, mean and median relative bias increased by time step through time step 3 (Tables 3 and 4)[5]. Mean and median bias in time step 4 was less than in time step 3 for both years. The mean and median biases for time step 5 estimates of exploitation rate were relatively small (less than 0.2%). Mean relative bias for the final FRAM estimate of the exploitation rate for the unmarked stocks was -0.90% in 2009 (median = -0.65%) and -0.68% in 2010 (median = -0.54%). As measured by the coefficient of variation (CV), there was considerable variability in relative bias across stocks within a time step during both years. The greatest observed relative bias for the total ER was -3.67% in 2009 and -2.11% in 2010. It is important to note that relative bias is expressed as a proportion of the bias-corrected ER estimate and is not the absolute difference between the biased and bias-corrected estimates.

Table 3. Summary statistics, by time step, for the relative bias of the FRAM exploitation rate estimates for unmarked coho stocks in the 2009 FRAM pre-season model run (N = 75).

Time Step	Mean	Standard Deviation	Coefficient of Variation	Median	Minimum	Maximum
1	-0.17%	0.177%	104%	-0.125%	-0.55%	0.00%
2	-2.30%	1.275%	55%	-2.022%	-6.32%	-0.78%
3	-3.40%	2.152%	63%	-2.614%	-9.72%	0.00%
4	-2.70%	3.005%	111%	-2.213%	-20.35%	0.16%
5	-0.01%	0.279%	559%	-0.001%	-0.66%	1.13%
Final	-0.90%	0.720%	80%	-0.649%	-3.67%	0.48%

Relative bias is expected to be negative whenever a stock has been subjected to a mark-selective fishery. The small positive relative biases present in some time steps are typically associated with stocks with very small exploitation rates in a time step, and|or very small cohort sizes, and are the result of rounding because the original FRAM estimates were rounded to the nearest whole fish compared to the bias-corrected estimates which did not round cohort numbers (i.e., fractional fish were carried forward in the computations).

[3] Canadian and Alaskan stocks are excluded from these summaries.
[4] Relative bias is expressed as a proportion of the bias-corrected ER estimate and is not the absolute difference between the biased and bias-corrected estimates.
[5] References to increases and decreases (or larger and smaller) are made without regard to the sign of $\hat{B}$ and Δ, e.g., -10% is considered larger than -5%.

Table 4. Summary statistics, by time step, for the relative bias of the FRAM exploitation rate estimates for unmarked coho stocks in the 2010 FRAM pre-season model run (N = 77).

Time Step	Mean	Standard Deviation	Coefficient of Variation	Median	Minimum	Maximum
1	-0.06%	0.104%	187%	-0.008%	-0.36%	0.17%
2	-1.31%	0.838%	64%	-1.214%	-4.49%	0.00%
3	-2.61%	1.612%	62%	-2.059%	-7.51%	0.00%
4	-1.74%	1.084%	62%	-1.571%	-6.06%	0.00%
5	-0.11%	0.336%	305%	-0.003%	-1.30%	0.78%
Final	-0.68%	0.537%	79%	-0.541%	-2.11%	0.74%

Figure 1 (top panel) summarizes the relative bias for the unmarked stocks by time step for each year. In 2009, several stocks had a relative bias exceeding -5% during time steps 2 (3 stocks), 3 (14 stocks), and 4 (4 stocks). There were only four estimates of relative bias exceeding -10%; all occurred during time step 4:

- Columbia River Early Hatchery Unmarked (-11.7%, bias-corrected time step ER = 0.073),
- Youngs Bay Hatchery Unmarked (-20.3%, bias-corrected time step ER = 0.112),
- Columbia River Wild Unmarked - Oregon (-11.7%, bias-corrected time step ER = 0.074), and
- Columbia River Early Wild Unmarked - Washington (-11.7%, bias-corrected time step ER = 0.073).

In 2010, there were only 9 stocks with a relative bias exceeding -5% during time step 3 and one stock with a bias exceeding -5% in time step 4 (Figure 1, bottom panel). In 2010, there were no stocks during any of the time steps with estimates of relative bias exceeding -10%.

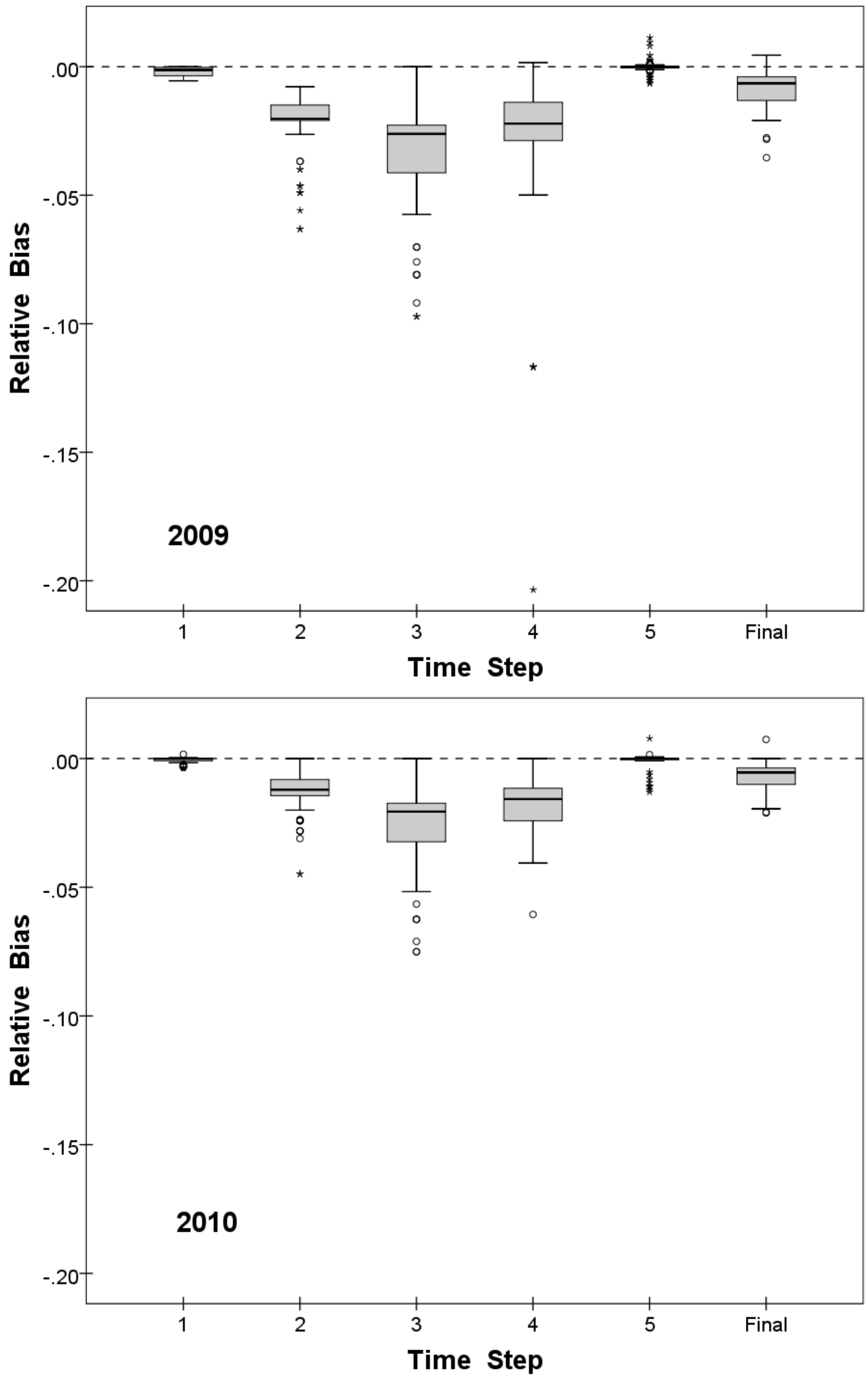

Figure 1. Box-and-whiskers plots summarizing the relative bias of the FRAM exploitation rates for unmarked coho stocks, by time step, for the final 2009 and 2010 preseason model runs with no CNR mortalities, no drop-off mortalities, and no mark misidentification errors.

Figures 2 and 3 compare the relative bias for the unmarked stocks by region in each time step for 2009 and 2010, respectively. Median relative bias was negative for all regions in all time steps in both years with the following exceptions:

- in 2009, median relative bias for the SJF region was 0 in time step 1 and positive in time step 5,
- in 2009, median relative bias for the COR and ORC regions was positive in time step 5,
- in 2010, median relative bias for the SPS, HC, and SJF regions was 0 in time step 1, and
- in 2010, median relative bias for the COR region was positive in time step 5.

As described earlier, these small positive relative biases are typically associated with stocks with very small exploitation rates in a time step, and|or very small cohort sizes, and are the result of rounding in the original FRAM model output. In both years, the COR (Columbia River) and ORC (Oregon-California) regions generally had larger negative relative biases compared to the other regions during time steps 1, 2, and 3.

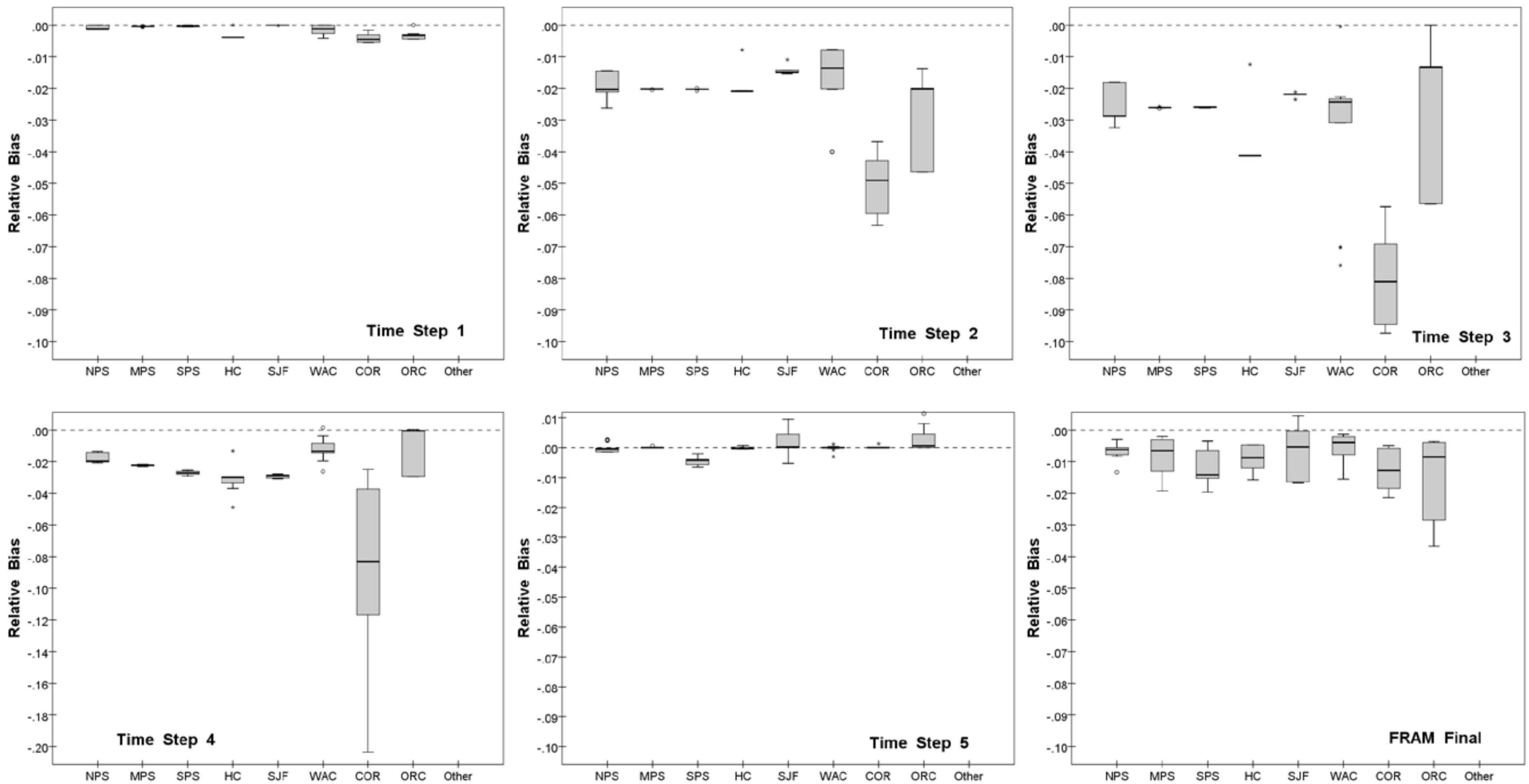

Figure 2. Box-and-whiskers plots comparing the relative bias of the FRAM exploitation rates for unmarked coho stocks, by region and time step, for the final 2009 preseason model run with no CNR mortalities, no drop-off mortalities, and no mark misidentification errors. Note that x-axis scale for time steps 4 and 5 is different from other time steps.

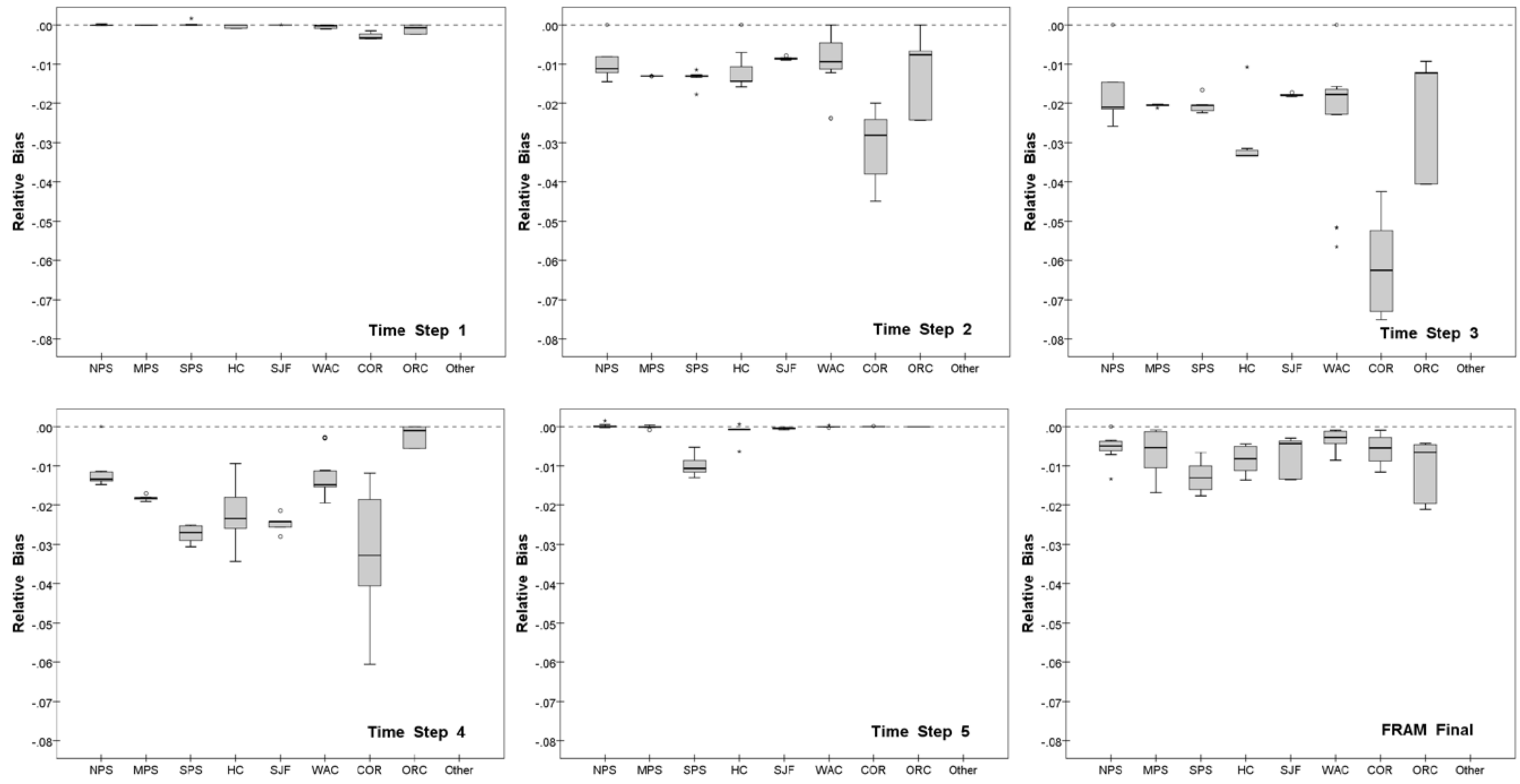

Figure 3. Box-and-whiskers plots comparing the relative bias of the FRAM exploitation rates for unmarked coho stocks, by region and time step, for the final 2010 preseason model run with no CNR mortalities, no drop-off mortalities, and no mark misidentification errors.

Difference (Δ):

In both years, the mean and median difference between the FRAM and bias-corrected estimates increased by time step through time step 4 (Tables 5 and 6). The mean and median differences for time step 5 estimates of exploitation rate were relatively small (less than 0.006). Mean difference for the final FRAM estimates of the exploitation rate for the unmarked stocks was -0.003 in 2009 (median = -0.0029) and -0.002 in 2010 (median = -0.002). As measured by the coefficient of variation (CV), there was considerable variability in the differences between the estimates across stocks within a time step.

Table 5. Summary statistics, by time step, for the difference between the FRAM and the bias-corrected estimates of exploitation rate for unmarked coho stocks in the 2009 FRAM pre-season model run (N = 75).

Time Step	Mean	Standard Deviation	Coefficient of Variation	Median	Minimum	Maximum
1	0.0000	0.0000	135%	0.0000	0.0000	0.0000
2	-0.0005	0.0004	72%	-0.0005	-0.0018	0.0000
3	-0.0012	0.0011	87%	-0.0011	-0.0048	0.0000
4	-0.0039	0.0063	163%	-0.0022	-0.0452	0.0001
5	-0.0002	0.0011	431%	0.0000	-0.0060	0.0022
Final	-0.0031	0.0025	82%	-0.0029	-0.0151	0.0013

Table 6. Summary statistics, by time step, for the difference between the FRAM and the bias-corrected estimates of exploitation rate for unmarked coho stocks in the 2010 FRAM pre-season model run (N = 77).

Time Step	Mean	Standard Deviation	Coefficient of Variation	Median	Minimum	Maximum
1	0.0000	0.0000	357%	0.0000	0.0000	0.0000
2	-0.0002	0.0002	80%	-0.0002	-0.0009	0.0000
3	-0.0008	0.0007	82%	-0.0006	-0.0029	0.0000
4	-0.0028	0.0041	146%	-0.0015	-0.0283	0.0000
5	-0.0005	0.0020	365%	0.0000	-0.0118	0.0021
Final	-0.0024	0.0022	94%	-0.0020	-0.0099	0.0027

Figure 4 summarizes the difference between the FRAM and the bias-corrected estimates of exploitation rate for unmarked stocks by time step for each year. Differences between the ERs exceeding -0.01 were rare. In 2009, differences greater than -0.01 occurred in:

- Quilcene Hatchery Unmarked (-0.022) in time step 4,
- Area 12A Natural Unmarked (-0.045) in time step 4,
- South Puget Sound Net Pens Unmarked (-0.014) in time step 4,
- Youngs Bay Hatchery Unmarked (-0.023) in time step 4, and
- Area 12A Natural Unmarked (-0.015) for the final ER.

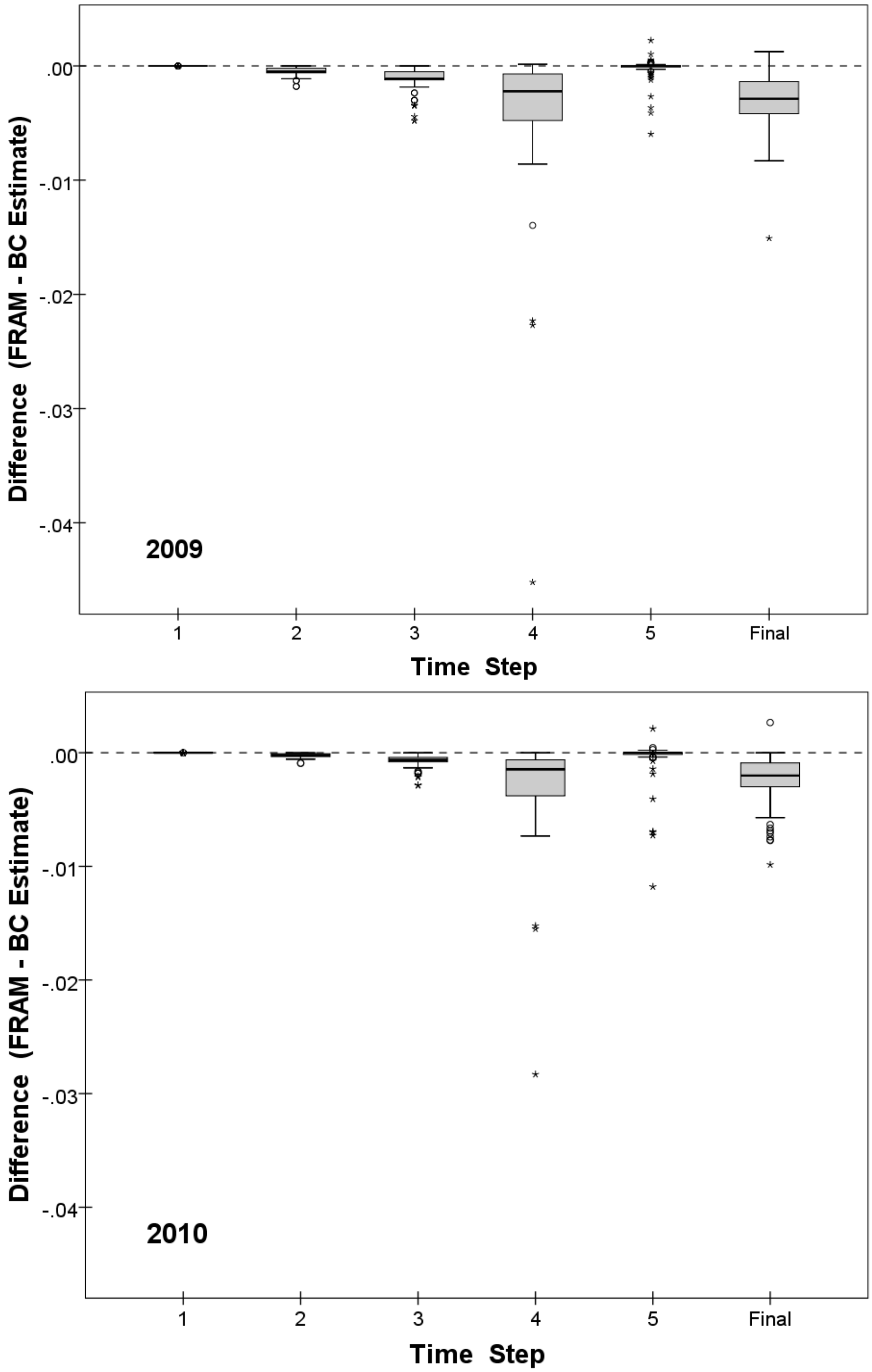

Figure 4. Box-and-whiskers plots summarizing the differences between the FRAM and bias-corrected (BC) exploitation rates of unmarked coho stocks, by time step, for the final 2009 and 2010 preseason model runs with no CNR mortalities, no drop-off mortalities, and no mark misidentification errors.

In 2010, there were differences greater than -0.01 for three of the same stocks:

- Quilcene Hatchery Unmarked (-0.015) in time step 4,
- Area 12A Natural Unmarked (-0.028) in time step 4,
- South Puget Sound Net Pens Unmarked (-0.016) in time step 4, and
- South Puget Sound Net Pens Unmarked (-0.012) in time step 5.

Stock sizes for the Quilcene Hatchery Unmarked, Area 12A Natural Unmarked, and South Puget Sound Net Pens Unmarked stocks are relatively small (less than 2,500 fish) while time-step specific ERs for time steps 4 and 5 are relatively high (generally > 0.50) for these stocks.

Figures 5 and 6 compare the difference between the FRAM and the bias-corrected estimates of exploitation rate for unmarked stocks by region in each time step for 2009 and 2010, respectively. Mean difference was negative for all regions in all time steps in both years with the following exceptions:

- in 2009, mean difference for the SJF region was 0 in time step 1,
- in 2009, mean differences for the MPS, SJF, COR, and ORC regions were positive in time step 5,
- in 2010, mean difference for the SPS and SJF regions was 0 in time step 1, and
- in 2010, mean difference for the NPS, WAC, and COR regions was positive in time step 5.

Similarly to relative bias, these small positive differences are due to rounding effects in the exploitation rate estimates for the original FRAM estimates (where cohort sizes were rounded to the nearest whole fish before calculation) compared to the bias-corrected estimates which did not use rounded cohort numbers. The COR region also had the largest negative differences compared to the other regions during time steps 2 and 3. The HC region had the largest differences in time step 4 and the SPS region had the largest differences in time step 5. The great majority of all differences were less than -0.005. Across all time periods and stocks, the difference between the FRAM ER estimate and bias-corrected estimate was greater than -0.005 in only 7 percent of the comparisons in 2009 and 5 percent in 2010.

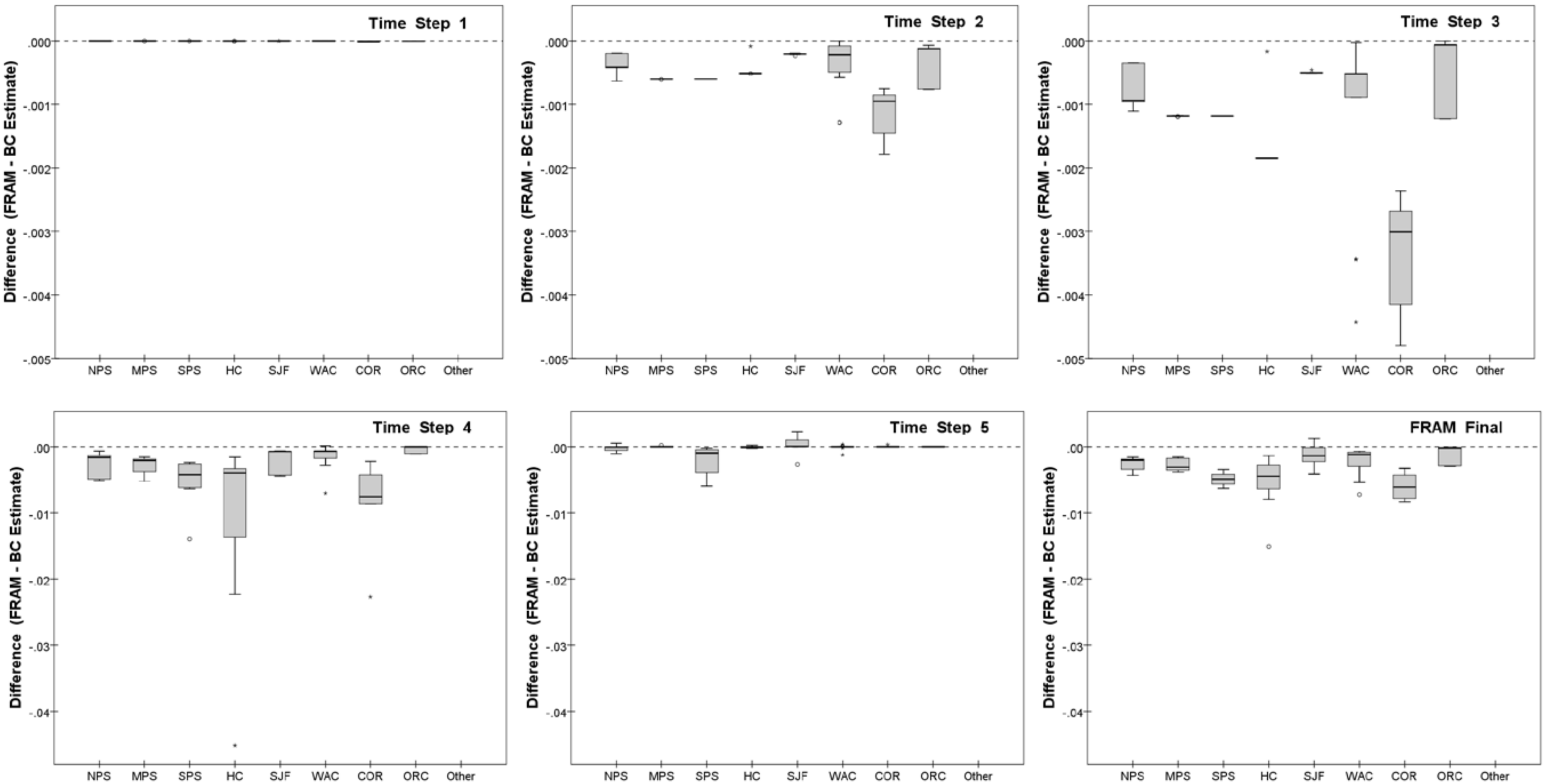

Figure 5. Box-and-whiskers plots comparing the differences between the FRAM and bias-corrected (BC) exploitation rates of unmarked coho stocks, by region and time step, for the final 2009 preseason model run with no CNR mortalities, no drop-off mortalities, and no mark misidentification errors. Note that x-axis scale for time steps 1 through 3 is different from the x-axis scale for time steps 5 though Final.

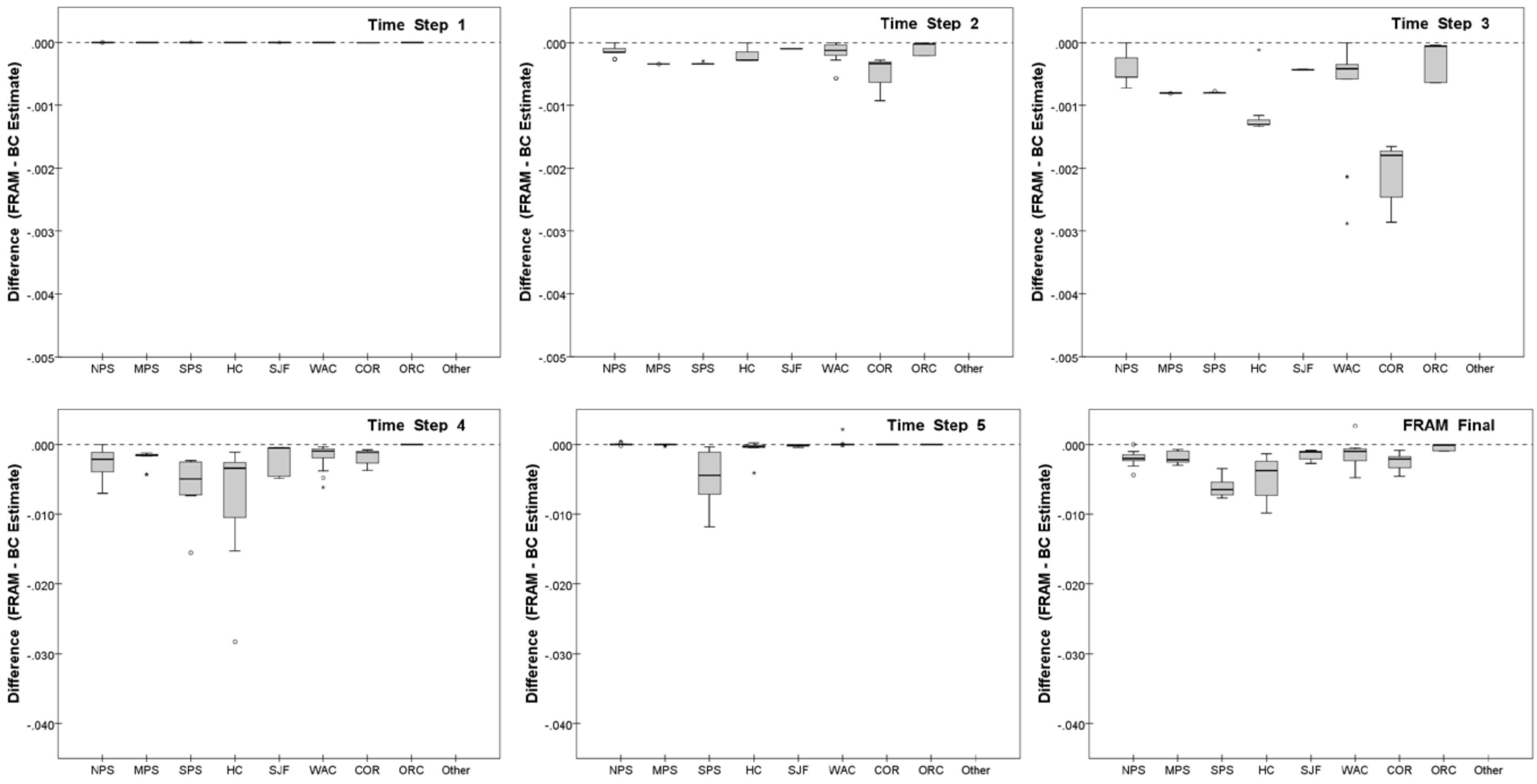

Figure 6. Box-and-whiskers plots comparing the differences between the FRAM and bias-corrected exploitation (BC) rates of unmarked coho stocks, by region and time step, for the final 2010 preseason model run with no CNR mortalities, no drop-off mortalities, and no mark misidentification errors. Note that x-axis scale for time steps 1 through 3 is different from the x-axis scale for time steps 5 though Final.

Coho Stocks of Concern:

The bias-corrected (BC) estimates of exploitation rate are compared to the FRAM estimates for the stocks (or stock aggregates) of concern below. These summaries compare the FRAM and bias-corrected estimates of exploitation rates on a cumulative basis over each time step with the ER calculated using a fixed denominator. It is important to remember that these ER estimates do not include the additional mortalities due to CNR, drop-off, or mark misidentification errors and are, therefore, underestimated. The time-step specific estimates of ER are compared in Appendix C.

Upper Fraser River Wild (Thompson R.) – 0.100 ER guideline: Although FRAM underestimated the final Upper Fraser River Wild exploitation rate by -2.7% (relative bias), the bias-corrected ER was still slightly below the 0.100 guideline in 2009. In 2010, relative bias and Δ were less than in 2009 in each time step.

Time Step	FRAM	BC	$\hat{B}$	Δ
2009				
1	0.000128	0.000128	-0.02%	0.000000
2	0.020981	0.021293	-1.46%	-0.000312
3	0.051145	0.052559	-2.69%	-0.001414
4	0.089089	0.091656	-2.80%	-0.002567
5	0.094190	0.096760	-2.66%	-0.002570
2010				
1	0.000136	0.000136	-0.01%	0.000000
2	0.013135	0.013251	-0.88%	-0.000116
3	0.039224	0.039997	-1.93%	-0.000773
4	0.075311	0.076795	-1.93%	-0.001484
5	0.079220	0.080700	-1.83%	-0.001480

Lower Columbia River Natural (three stock aggregate) – The ER guideline was 0.200 in 2009 and 0.150 in 2010: In 2009, FRAM underestimated the Lower Columbia River Natural exploitation rate through time step 4 by slightly more than 0.01. In 2010, relative bias and Δ were less than in 2009 in each time step. Although relative bias in time steps 3 and 4 was about 5 percent in 2010, the actual difference in exploitation rates (Δ) was small (less than 0.005). Because the fisheries in the mainstem of the Columbia River are structured to impact the stock up to the full ER guideline, any underestimate of the ER in the marine fisheries would result in the ER guideline being exceeded.

Time Step	FRAM	BC	$\hat{B}$	Δ
2009				
1	0.001857	0.001865	-0.45%	-0.000008
2	0.025876	0.027349	-5.38%	-0.001473
3	0.066029	0.071423	-7.55%	-0.005394
4	0.116625	0.126720	-7.97%	-0.010095
5				
2010				
1	0.001448	0.001452	-0.33%	-0.000005
2	0.016515	0.017081	-3.31%	-0.000566
3	0.046980	0.049730	-5.53%	-0.002750
4	0.074023	0.077678	-4.71%	-0.003655
5				

Guideline measured by combining FRAM impacts with mainstem Columbia River fishery impacts (most of which occur in time step 5) outside the model.

Oregon Coastal Natural (three stock aggregate) – 0.150 ER guideline: FRAM underestimated the final Oregon Coastal Natural exploitation rate by -3% (relative bias) but the bias-corrected ER was still well below the 0.150 guideline in 2009. In 2010, relative bias and Δ were less than in 2009 in each time step.

Time Step	FRAM	BC	$\hat{B}$	Δ
2009				
1	0.001659	0.001666	-0.43%	-0.000007
2	0.018230	0.019041	-4.26%	-0.000811
3	0.039174	0.041216	-4.95%	-0.002041
4	0.072247	0.075215	-3.95%	-0.002968
5	0.093427	0.096324	-3.01%	-0.002898
2010				
1	0.001005	0.001007	-0.24%	-0.000002
2	0.009970	0.010194	-2.20%	-0.000224
3	0.025545	0.026424	-3.33%	-0.000879
4	0.032135	0.033045	-2.75%	-0.000910
5	0.042577	0.043477	-2.07%	-0.000900

Southern Oregon | Northern California Coast (two stock aggregate) – 0.130 ER guideline: There were only very small differences between the FRAM and bias-corrected estimates for the Southern Oregon | Northern California Coast stock aggregate in 2009. In 2010, relative bias and Δ were less than in 2009 in each time step.

Time Step	FRAM	BC	$\hat{B}$	Δ
2009				
1	0.001232	0.001236	-0.33%	-0.000004
2	0.007336	0.007466	-1.74%	-0.000130
3	0.011817	0.012006	-1.58%	-0.000189
4	0.017249	0.017440	-1.09%	-0.000191
5	0.022737	0.022913	-0.77%	-0.000175
2010				
1	0.000372	0.000372	-0.09%	0.000000
2	0.002789	0.002808	-0.67%	-0.000019
3	0.006880	0.006950	-1.00%	-0.000070
4	0.007252	0.007322	-0.96%	-0.000070
5	0.012737	0.012807	-0.54%	-0.000070

Discussion

As was demonstrated in Conrad and Yuen (2010), the negative bias of the FRAM estimate of the exploitation rate for an unmarked stock when mark-selective fisheries occur during a time step (1) increases as the exploitation rate for the stock's marked component increases and (2) decreases as the weighted release mortality rate increases. The weighted release mortality rate is a gross indicator of the proportion of total exploitation occurring in mark-selective fisheries, e.g., a δ_W near 1.00 indicates that almost all of the exploitation is occurring in non-selective fisheries while a δ_W less than 0.20 indicates that most of the exploitation is occurring in mark-selective fisheries.

Figures 7 and 8 show the trend across time steps in the distributions of the exploitation rates for the marked stock components (upper panel) in comparison to the corresponding weighted release mortality rates (lower panel) for 2009 and 2010, respectively. Exploitation in time step 1 is minimal and therefore not a major contributor to bias despite the relatively low weighted release mortality rate. The exploitation rates then increase during time steps 2 and 3 which have weighted release mortality rates that are similar (usually between 30 percent to 40 percent). This explains the relatively large increase in bias during these time steps. While the exploitation rate for time step 4 is generally greater than that in time steps 2 and 3, its weighted release mortality rate increases greatly (to about 60 percent to 80 percent) which reduces bias relative to time steps 2 and 3. Although, the marked component exploitation rates generally continue to increase in time step 5, the weighted release mortality rate for the time step is nearly 100 percent for most stocks which results in a minimal contribution to overall bias for the last time step.

Generally, the relative biases for time steps 2, 3, and 4 were larger than the relative bias for the overall exploitation rate (Tables 3 and 4 and Figure 1). This is a result of 50 percent (on average) of the total exploitation rate for most unmarked stocks occurring in time step 5 where the majority of the fisheries are non-selective (Figures 7 and 8). Table 7 presents summary statistics for the proportion of the total FRAM exploitation rate that occurred in the largely non-selective time step 5 for the 2009 and 2010 FRAM runs.

Having a large proportion of the total exploitation on an unmarked stock occurring in non-selective fisheries during any time step reduces the overall (total) relative bias considerably. As a general rule, the overall bias will never exceed the greatest observed time step bias. Interestingly, when fisheries are modeled as rates, time steps are interchangeable. The overall exploitation rate and bias will be almost the same regardless of which time step selective and non-selective fisheries are modeled in; i.e., exploitation rates and release mortality rates for time steps 4 and 5 could be swapped and have very little effect on the overall catch, final exploitation rate, and bias.

Table 7. Summary statistics for the proportion of the total FRAM exploitation rate for the unmarked stock component occurring in time step 5, by year.

Year	Mean	Standard Deviation	Coefficient of Variation	Median	Minimum	Maximum
2009	49.8%	22.3%	45%	51.8%	0.0%	82.9%
2010	50.1%	24.5%	49%	50.1%	0.0%	91.6%

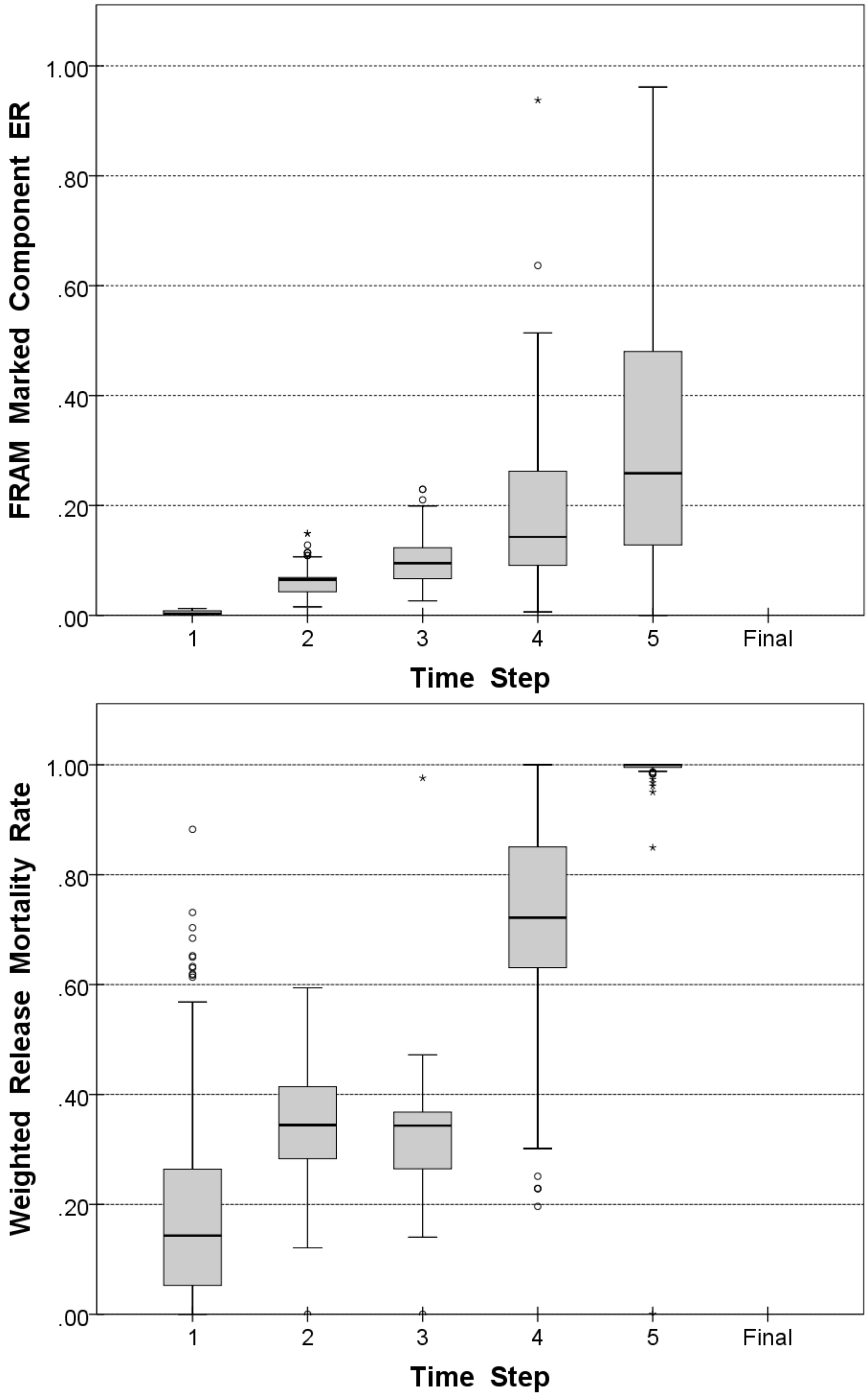

Figure 7. Box-and-whiskers plots summarizing FRAM exploitation rates for marked stock components (upper panel) and weighted release mortality rates (lower panel) by time step in 2009.

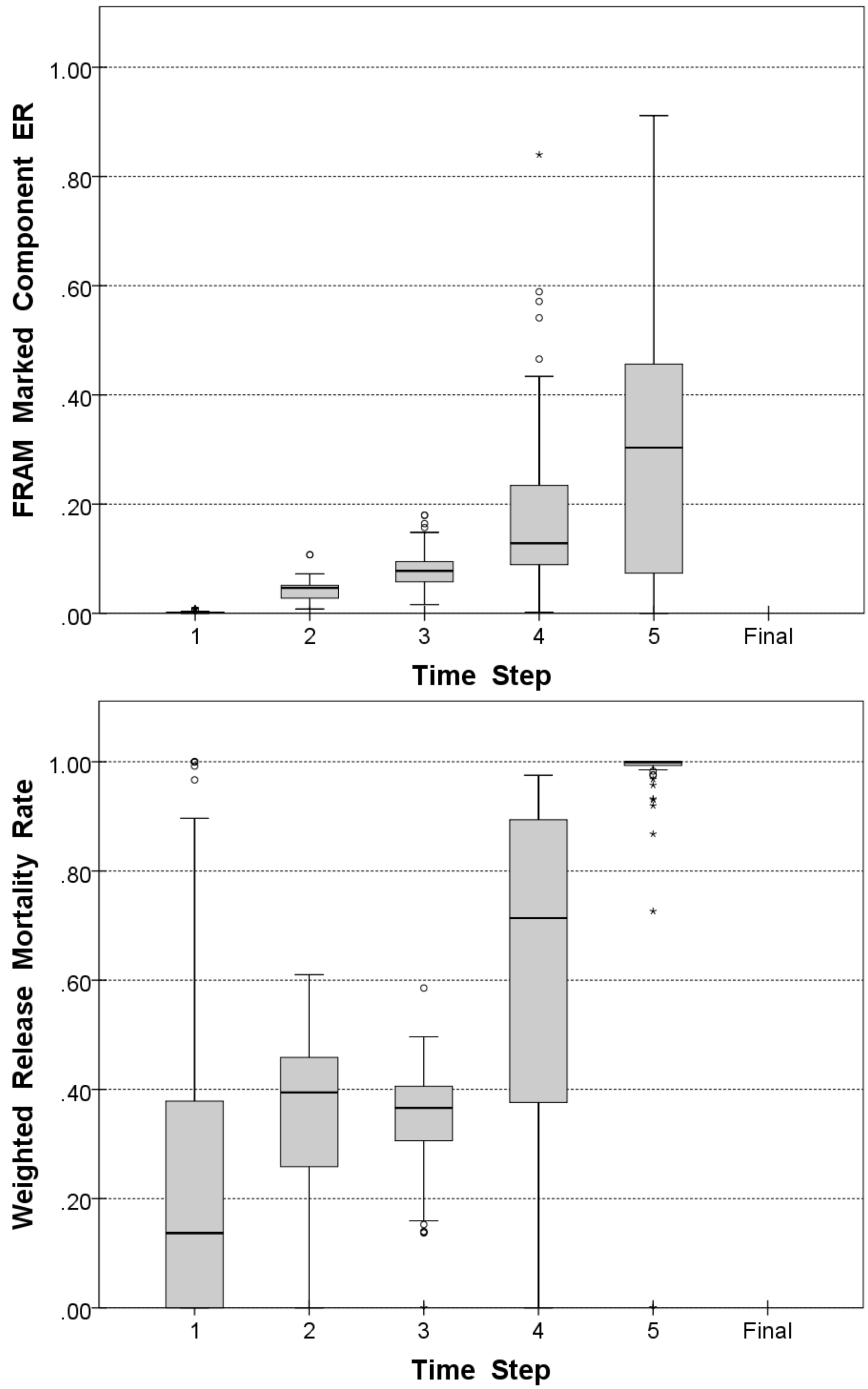

Figure 8. Box-and-whiskers plots summarizing FRAM exploitation rates for marked stock components (upper panel) and weighted release mortality rates (lower panel) by time step in 2010.

While the order of the time steps is not important, it does matter whether a non-selective fishery occurs in the same time step as a mark selective fishery or in a different time step. If the non-selective fishery occurs in the same time step as the mark-selective fishery, unmarked impacts will be underestimated by FRAM, thus adding to the existing bias. If the non-selective fishery occurs in a subsequent time step, FRAM will overestimate unmarked impacts, because it overestimates unmarked abundance, thus compensating for the already existing bias. Since the overall bias is a function of the mix of selective and non-selective fisheries occurring over all time steps, a simple rule as to what constitutes acceptable bias that is based on the size of the mark-selective fisheries alone will not work. The “30-10” rule, which recommended that the FRAM is “suitable for modeling mark-selective fisheries of low intensity, with 'low intensity' provisionally defined as those fisheries with fishery-specific exploitation rates on marked stocks of less than 10 percent and overall selective fishery exploitation rates of less than 30 percent” (PFMC 2011) can produce vastly different bias outcomes depending on the timing and magnitude of the non-selective fisheries.

There is not a similar compensating effect for the absolute differences between the FRAM and bias-corrected estimates of exploitation rates. For the differences between the estimates, the mean and median differences for time step 4 are only slightly greater than those for the FRAM final estimate (Tables 5 and 6 and Figure 4). The differences in exploitation rates in time steps 2 and 3 are generally less than those for the FRAM final estimate.

Therefore, we recommend focusing any evaluation of the impact of bias on the FRAM estimates of exploitation rates on the differences between the FRAM and bias-corrected estimates rather than relative bias. As has been demonstrated, what could be considered a large relative bias for the FRAM exploitation rate for an unmarked stock in one time step (say -10%) will be substantially reduced in another time step if the majority of the exploitation in the time step occurs in non-selective fisheries. Also, current guidelines for stocks of concern are usually expressed in terms of a total allowable exploitation rate.

Escapement Projections:

An important management consideration when using FRAM to develop management options for a fishing season is the projected escapement for certain stocks, especially wild stocks. Tables 8 and 9 summarize the relative percent differences[6] in projected escapement between the FRAM and bias-corrected methods for 2009 and 2010, respectively. Across all unmarked stocks, FRAM overestimated escapement (relative to the bias-corrected estimate) by about 2 percent in 2009 and 1 percent in 2010, on average. The largest relative differences between the escapement projections occurred for the same two stocks in both years. The escapement for the Area 12 Wild (unmarked) stock was 62 percent greater than the bias-corrected projection in 2009 (53 fish compared to 33 fish) and 17 percent greater in 2010. The escapement for the South Puget Sound Net Pen (unmarked) stock was 18 percent greater than the bias-corrected projection in 2009 and 17 percent greater in 2010. Relative percent differences in escapement projections for all other stocks were less than 6 percent.

In terms of number of fish, the largest differences between the escapement projections were 711 fish for the Columbia River Hatchery Unmarked stock in 2009 and 214 fish for the Snohomish Wild Unmarked stock in 2010. In both years, most of the larger differences in numbers of fish for escapement projections occurred in the Columbia River region (Figure 9). Across all unmarked stocks, FRAM overestimated escapement (relative to the bias-corrected estimate) by about 43 fish in 2009 and 17 fish in 2010, on average.

[6] Relative percent difference = (FRAM projected escapement – bias-corrected projected escapement)/bias-corrected projected escapement.

Table 8. Summary statistics, by region, for the relative percent difference between the FRAM and the bias-corrected estimates of escapement for unmarked coho stocks in the 2009 FRAM pre-season model run.

Regional Group	Mean	Standard Deviation	Coefficient of Variation	Median	Minimum	Maximum
NPS	0.6%	0.37%	65%	0.4%	0.2%	1.2%
MPS	0.5%	0.20%	36%	0.5%	0.3%	0.9%
SPS	3.2%	6.11%	189%	1.1%	0.5%	18.3%
HC	10.1%	22.81%	226%	0.7%	0.3%	61.6%
SJF	0.2%	0.31%	170%	0.1%	-0.3%	0.7%
WAC	0.6%	1.10%	174%	0.2%	0.1%	4.4%
COR	1.4%	0.68%	49%	1.3%	0.9%	3.0%
ORC	0.1%	0.21%	246%	0.0%	-0.4%	0.3%
Total	1.8%	7.35%	419%	0.5%	-0.4%	61.6%

Table 9. Summary statistics, by region, for the relative percent difference between the FRAM and the bias-corrected estimates of escapement for unmarked coho stocks in the 2010 FRAM pre-season model run.

Regional Group	Mean	Standard Deviation	Coefficient of Variation	Median	Minimum	Maximum
NPS	-0.6%	3.62%	620%	0.3%	-13.0%	1.5%
MPS	0.4%	0.25%	65%	0.3%	-0.1%	0.7%
SPS	3.3%	5.53%	166%	1.4%	0.2%	16.8%
HC	3.2%	5.54%	175%	0.6%	0.4%	16.5%
SJF	0.1%	0.33%	221%	0.1%	-0.3%	0.6%
WAC	-0.1%	1.91%	1293%	0.2%	-6.9%	1.4%
COR	0.5%	0.17%	35%	0.4%	0.3%	0.7%
ORC	0.0%	0.17%	533%	0.0%	-0.3%	0.3%
Total	0.7%	3.24%	498%	0.3%	-13.0%	16.8%

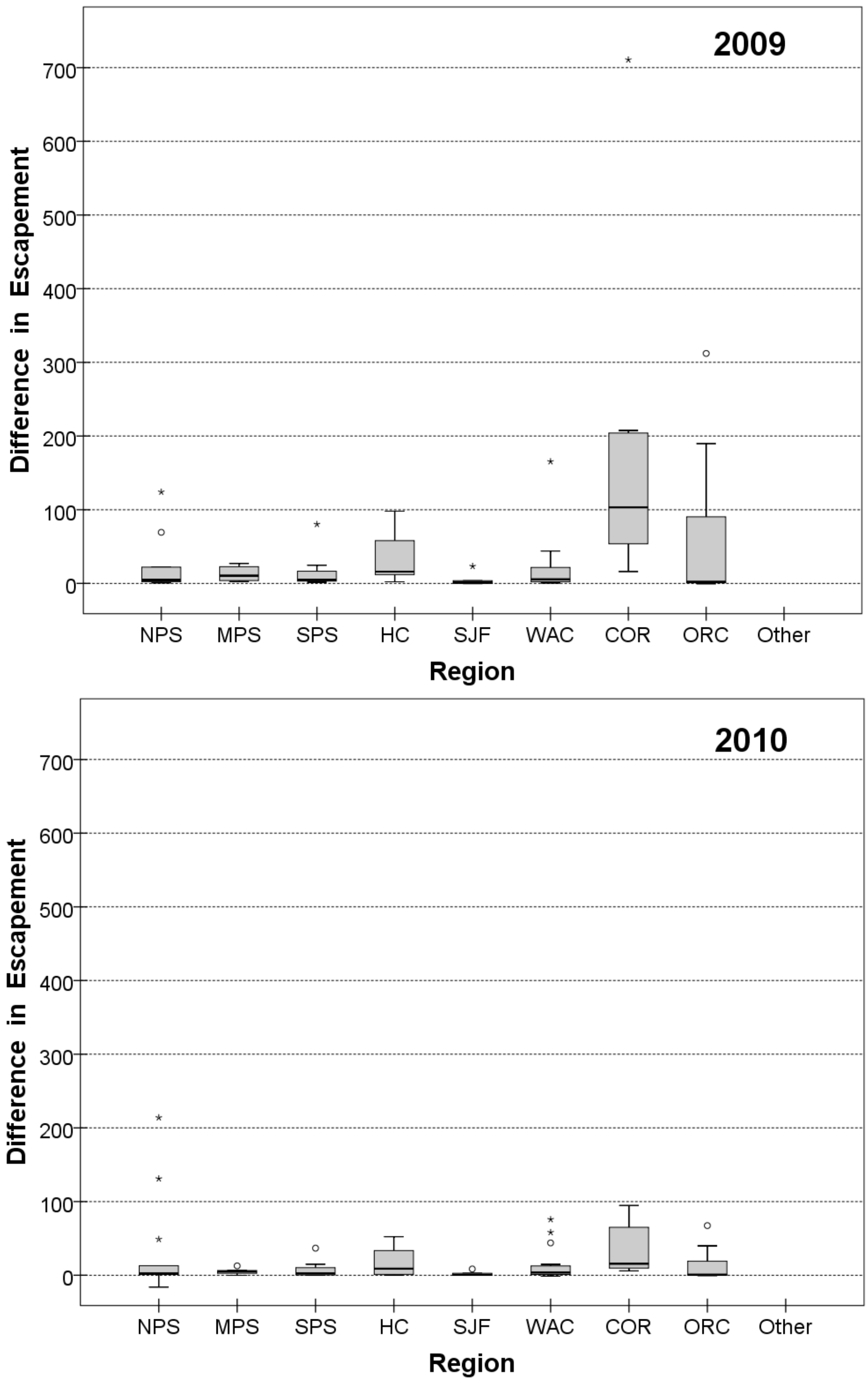

Figure 9. Box-and-whiskers plots summarizing the differences in escapement estimates between the FRAM and bias-corrected methods for unmarked coho stocks, by regional group, for the final 2009 and 2010 preseason model runs with no CNR mortalities, no drop-off mortalities, and no mark misidentification errors.

Conclusions

At current levels of exploitation for coho salmon, the bias in the FRAM estimate of the total exploitation rate of unmarked stocks is minimal. The mean and median differences by which FRAM underestimated the total exploitation rate for an unmarked stock were -0.003 in 2009 and -0.002 in 2010. For the 2009 and 2010 FRAM preseason runs, there was only a single instance where the difference between the FRAM estimate of the total exploitation rate for an unmarked stock and the bias-corrected estimates was ≥ -0.01 (-0.015 for Area 12A wild stock in 2009).

However, bias-correction is important when considering exploitation rate guidelines (limits) for coho stocks of concern. Although in the 2009 and 2010 preseason runs there were no instances where the bias-corrected estimate of total exploitation for a stock of concern exceeded its guideline, the potential exists. For example, in 2009 the FRAM estimate for the total exploitation rate on the Upper Fraser River Wild stock (Thompson River coho with an exploitation rate guideline of 0.10 in southern US fisheries) of 0.094 was increased to 0.097 after bias-correction. If FRAM projections of total exploitation rate for a stock of concern are very near a guideline, then there is a very real possibility that the bias-corrected estimate may exceed that guideline.

Also, for the case where a portion of the exploitation rate for a stock is modeled outside of FRAM (as is done for the Lower Columbia River Natural stock aggregate), the underestimate of the exploitation rate for that portion of the impacts occurring in FRAM modeled fisheries must be considered. Because impacts outside the FRAM model are typically projected to be the maximum allowable under the guideline, the guideline will always be exceeded if the bias in the FRAM estimates is not considered.

The current fishery exploitation rate pattern for coho salmon, where stock-specific exploitation rates in the last time step (time step 5) are generally large relative to the ERs in earlier time steps and fisheries in the last time step are almost exclusively non-selective, is largely responsible for keeping the bias in the total FRAM ER introduced by mark-selective fisheries in earlier time steps small. The current version of FRAM usually overestimates the fishery-related mortalities in the last time step, relative to the bias-corrected estimates, because;

- FRAM currently tends to overestimate the cohort size of an unmarked stock entering time step 5 relative to the bias-corrected estimates because it has underestimated fishery mortalities in earlier time steps, and
- Fisheries in the last time step are almost entirely non-selective so the same stock-specific ER is being applied to the starting cohort abundance in time step 5 by both the current FRAM version and the bias-corrected methods.

These additional mortalities in time step 5 in current FRAM then reduce the underestimation of mortalities in earlier time steps and typically result in a relatively small overall bias for the total stock-specific ER estimate.

Recommendations

We recommend that:

1. The bias-correction methodology is incorporated into Coho FRAM.
2. The implementation of bias-correction into FRAM be evaluated by comparing results from bias-corrected FRAM to bias-corrected results calculated outside the model as was done for this report.
3. The effects of adding in additional sources of mortality not included in the bias-correction evaluation to date (i.e., CNR mortality, drop-off mortality, and mark-recognition errors) be evaluated.
4. A process similar to that used to evaluate bias-correction in Coho FRAM be considered for Chinook FRAM.

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Appendix A

Bias-correction calculation example using FRAM PopStat report.

	U-colreh	M-colreh	Derived From	Weighted Release Mortality δ	Derived From	Bias-corrected U-croreh	Derived From
Time Step 1							
Starting Cohort	120,659	516,048	PopStat Report			120,659	PopStat Report
After Nat. Mort	106,481	455,410	PopStat Report			106,481	PopStat Report
After PreTerm	106,310	449,766	PopStat Report			106,309	After NatMort*(1-ER_UM)
Catch	171	5,644	After NatMort - After PreTerm			172	After NatMort - After PreTerm
FRAM TS ER	0.00161	0.01239	Catch/After NatMort	0.12958	(FRAM TS ER_UM)/ (FRAM TS ER_M)	0.00161	ER_UM: 1-(1-ER_M)^δ
Time Step 2							
Starting Cohort	106,310	449,766	PopStat Report			106,309	After PreTerm from previous step
After Nat. Mort	104,118	440,493	PopStat Report			104,117	Starting Cohort * (1- NatMort)
After PreTerm	102,200	390,246	PopStat Report			102,100	After NatMort*(1-ER_UM)
Catch	1,918	50,247	After NatMort - After PreTerm			2,017	After NatMort - After PreTerm
FRAM TS ER	0.01842	0.11407	Catch/After NatMort	0.16149		0.01937	ER_UM: 1-(1-ER_M)^δ
Time Step 3							
Starting Cohort	102,200	390,246	PopStat Report			102,100	After PreTerm from previous step
After Nat. Mort	100,093	382,200	PopStat Report			99,995	Starting Cohort * (1- NatMort)
After PreTerm	96,673	309,807	PopStat Report			96,278	After NatMort*(1-ER_UM)
Catch	3,420	72,393	After NatMort - After PreTerm			3,717	After NatMort - After PreTerm
FRAM TS ER	0.03417	0.18941	Catch/After NatMort	0.18039		0.03717	ER_UM: 1-(1-ER_M)^δ

	U-colreh	M-colreh	Derived From	Weighted Release Mortality δ	Derived From	Bias-corrected U-croreh	Derived From
Time Step 4							
Starting Cohort	96,673	309,807	PopStat Report			96,278	After PreTerm from previous step
After Nat. Mort	94,680	303,420	PopStat Report			94,293	Starting Cohort * (1- NatMort)
After PreTerm	88,546	217,498	PopStat Report			87,378	After NatMort*(1-ER_UM)
Catch	6,134	85,922	After NatMort - After PreTerm			6,915	After NatMort - After PreTerm
FRAM TS ER	0.06479	0.28318	Catch/After NatMort	0.22878		0.07334	ER_UM: 1-(1-ER_M)^δ
Columbia River base period fisheries used for this time step 5 example. Actual time step 5 exploitation rate calculations are currently not modeled in FRAM .							
Time Step 5							
Starting Cohort	88,546	217,498	PopStat Report			87,378	After PreTerm from previous step
After Nat. Mort	86,721	213,014	PopStat Report			85,577	Starting Cohort * (1- NatMort)
After PreTerm	86,721	213,014	PopStat Report			85,577	Same as After Nat Mort
Mature Cohort	86,721	213,014	PopStat Report			85,577	Same as After Nat Mort
Escapement	53,869	132,319	PopStat Report			53,158	Mature Cohort * (1-ER_UM)
Catch	32,852	80,695	After NatMort - After PreTerm			32,419	Mature Cohort - Escapement
FRAM TS ER	0.37882	0.37882	Catch/After NatMort	1.00000		0.37882	ER_UM:1-(1-ER_M)^δ
FRAM (ER)	**0.4524**	**0.6903**	(Catch all time steps)/(Catch all time steps + Escapement)		**Bias-Adjusted ER**	**0.4598**	(Catch all time steps)/(Catch all time steps + Escapement)
Relative Bias	**-1.61%**		(FRAM_ER – Bias Adjusted_ER)/Bias Adjusted_ER				

Appendix B

Regional groupings for unmarked stocks.

Stock Name	Stock Number	Hatchery or Wild	Mark Status	Regional Group
Nooksack R	1	Wild	U	NPS
Kendall Ck	3	Hatch	U	NPS
Skookum Ck	5	Hatch	U	NPS
Lummi Ponds	7	Hatch	U	NPS
Samish R	11	Wild	U	NPS
Area 7/7A	13	Wild	U	NPS
Skagit R	17	Wild	U	NPS
Skagit R	19	Hatch	U	NPS
Baker R	23	Wild	U	NPS
Stillaguamish R	29	Wild	U	NPS
Tulalip	33	Hatch	U	NPS
Snohomish R	35	Wild	U	NPS
Snohomish R	37	Hatch	U	NPS
Port Gamble	43	Wild	U	HC
Area 12/12B	45	Wild	U	HC
Quilcene R	47	Hatch	U	HC
Area 12A	51	Wild	U	HC
Area 12C/12D	55	Wild	U	HC
George Adams	57	Hatch	U	HC
Skokomish R	59	Wild	U	HC
Area 13B Misc.	61	Wild	U	SPS
Deschutes R	63	Wild	U	SPS
South Sound NP	65	Hatch	U	SPS
Nisqually R	67	Hatch	U	SPS
Nisqually R	69	Wild	U	SPS
Minter Ck	73	Hatch	U	SPS
Area 13 Misc.	75	Wild	U	SPS
Area 13A Misc.	81	Wild	U	SPS
Puyallup R	83	Hatch	U	MPS
Puyallup R	85	Wild	U	MPS
Area 11 Misc.	89	Wild	U	MPS
Area 10E Misc.	93	Wild	U	MPS
Green R	95	Hatch	U	MPS
Green R	97	Wild	U	MPS
Lake Wash.	99	Hatch	U	MPS
Lake Wash.	101	Wild	U	MPS
Area 10 Misc.	105	Wild	U	MPS

- continued -

Appendix B

Regional groupings for unmarked stocks.

Stock Name	Stock Number	Hatchery or Wild	Mark Status	Regional Group
Dungeness R	107	Wild	U	SJF
Dungeness R	109	Hatch	U	SJF
Elwha R	111	Wild	U	SJF
Elwha R	113	Hatch	U	SJF
East JDF Misc.	115	Wild	U	SJF
West JDF Misc.	117	Wild	U	SJF
Makah Coastal	125	Hatch	U	WAC
Quillayute R Summer	127	Wild	U	WAC
Quillayute R Fall	131	Wild	U	WAC
Quillayute R Fall	133	Hatch	U	WAC
Hoh R	135	Wild	U	WAC
Queets R	139	Wild	U	WAC
Queets R	141	Hatch	U	WAC
Quinault R	145	Wild	U	WAC
Quinault R	147	Hatch	U	WAC
Chehalis R	149	Wild	U	WAC
Chehalis R	151	Hatch	U	WAC
Humptulips R	153	Wild	U	WAC
Humptulips R	155	Hatch	U	WAC
Grays Harbor Misc.	157	Wild	U	WAC
Grays Harbor NP	159	Hatch	U	WAC
Willapa Bay	161	Wild	U	COR
Willapa Bay	163	Hatch	U	COR
Columbia R Early	165	Hatch	U	COR
Youngs Bay	167	Hatch	U	COR
Columbia R OR Early	169	Wild	U	COR
Columbia R WA Early	171	Wild	U	COR
Columbia R WA Late	173	Wild	U	COR
Columbia R Late	175	Hatch	U	COR
Oregon North Coast	179	Wild	U	ORC
Oregon No. Mid Coast	183	Wild	U	ORC
Oregon So. Mid Coast	187	Wild	U	ORC
Oregon South Coast	193	Hatch	U	ORC
Oregon South Coast	195	Wild	U	ORC
California North Coast	197	Hatch	U	ORC
California North Coast	199	Wild	U	ORC
California Central Coast	201	Hatch	U	ORC
California Central Coast	203	Wild	U	ORC

Appendix C

FRAM and bias-corrected estimates of time-step specific exploitation rates[a] for stocks of concern.

Upper Fraser River Wild (Thompson R.) – 0.100 ER guideline:

Time Step	FRAM	BC	$\hat{B}$	Δ
2009				
1	0.000128	0.000128	-0.02%	0.000000
2	0.020853	0.021165	-1.47%	-0.000312
3	0.030164	0.031266	-3.52%	-0.001102
4	0.037944	0.039097	-2.95%	-0.001153
5	0.005102	0.005104	-0.05%	-0.000003
Final	0.094190	0.096760	-2.66%	-0.002570
2010				
1	0.000136	0.000136	-0.01%	0.000000
2	0.012999	0.013115	-0.89%	-0.000116
3	0.026089	0.026746	-2.46%	-0.000657
4	0.036088	0.036798	-1.93%	-0.000711
5	0.003909	0.003904	0.11%	0.000004
Final	0.079220	0.080700	-1.83%	-0.001480

Lower Columbia River Natural (three stock aggregate) – The ER guideline was 0.200 in 2009 and 0.150 in 2010:

Time Step	FRAM	BC	$\hat{B}$	Δ
2009				
1	0.001857	0.001865	-0.45%	-0.000008
2	0.024019	0.025483	-5.75%	-0.001464
3	0.040153	0.044074	-8.90%	-0.003921
4	0.050595	0.055297	-8.50%	-0.004702
2010				
1	0.001448	0.001452	-0.33%	-0.000005
2	0.015068	0.015629	-3.59%	-0.000561
3	0.030465	0.032649	-6.69%	-0.002185
4	0.027043	0.027947	-3.24%	-0.000904

Guideline measured by combining FRAM impacts with mainstem Columbia River fishery impacts (most of which occur in time step 5) outside the model.

[a] Cohort size for the time step, after natural mortality, used as the denominator for the exploitation rate calculation.

Oregon Coastal Natural (three stock aggregate) – 0.150 ER guideline:

Time Step	FRAM	BC	$\hat{B}$	Δ
2009				
1	0.001659	0.001666	-0.43%	-0.000007
2	0.016572	0.017375	-4.62%	-0.000804
3	0.020944	0.022175	-5.55%	-0.001231
4	0.033072	0.033999	-2.72%	-0.000926
5	0.021180	0.021109	0.33%	0.000070
Final	0.093427	0.096324	-3.01%	-0.002898
2010				
1	0.001005	0.001007	-0.24%	-0.000002
2	0.008965	0.009187	-2.42%	-0.000222
3	0.015575	0.016230	-4.03%	-0.000654
4	0.006590	0.006621	-0.46%	-0.000031
5	0.010442	0.010433	0.09%	0.000009
Final	0.042577	0.043477	-2.07%	-0.000900

Southern Oregon | Northern California Coast (two stock aggregate) – 0.130 ER guideline:

Time Step	FRAM	BC	$\hat{B}$	Δ
2009				
1	0.001232	0.001236	-0.33%	-0.000004
2	0.006104	0.006230	-2.02%	-0.000126
3	0.004480	0.004540	-1.31%	-0.000060
4	0.005432	0.005434	-0.03%	-0.000001
5	0.005488	0.005473	0.29%	0.000016
Final	0.022737	0.022913	-0.77%	-0.000175
2010				
1	0.000372	0.000372	-0.09%	0.000000
2	0.002417	0.002436	-0.76%	-0.000019
3	0.004091	0.004141	-1.22%	-0.000051
4	0.000372	0.000372	-0.10%	0.000000
5	0.005485	0.005485	0.01%	0.000000
Final	0.012737	0.012807	-0.54%	-0.000070

Causes and Effects of Bias in Anticipated Mark Rates in Mark-Selective Fisheries for Coho Salmon.

Robert Kope
National Marine Fisheries Service
Northwest Fisheries Science Center
2725 Montlake Blvd E
Seattle, WA 98112

One issue that has been raised regarding mark-selective fisheries is that observed mark rates for coho have consistently been lower than predicted mark rates. As a result of this, more unmarked fish are encountered per marked fish landed in mark-selective fisheries than is expected preseason. One possible consequence of unexpectedly high encounter rates is that more unmarked fish may be killed as a result of incidental mortality in mark-selective fisheries than is being projected pre-season. To evaluate this potential problem, we examine the magnitude of the bias in preseason mark-rate projections, look for possible explanations of the bias, and evaluate the post-season impacts of mark-selective fisheries on unmarked coho.

Each year the STT reports anticipated and observed mark rates for coho as well as quotas or catch expectations, expected incidental mortality, observed catches and estimates of unmarked coho released and incidental mortality in mark-selective fisheries. These values are reported in Table I-9 of the 2010 Review of Ocean Salmon Fisheries (STT 2011a) and in table I-8 in the same document for prior years back to 2000. In most years, post-season estimates of mark rates in commercial mark-selective fisheries are not available, so we focus here primarily on recreational fisheries.

Magnitude of the bias in projected mark rates

Comparison of preseason anticipated mark rates with observed rates does reveal evidence of bias (Table 1). This bias appears to vary by year and by area, ranging from a 6% relative error in 2006 to 51% relative error in 2003, and from 7% in the buoy 10 fishery to 58% in the recreational fishery in LaPush. The average bias across all years and catch areas amounts to 24% of the post-season observed mark rates. However, this simple average rate is not very meaningful. The LaPush area, with the highest relative error, typically has the smallest recreational fishery of all the catch areas.

Likely Explanations

The expected mark rates in coho fisheries are calculated using the coho Fishery Regulation Assessment Model (FRAM). Each coho stock is divided into marked and unmarked components. Forecast abundance of natural stocks is all unmarked and the marked component is zero. For hatchery stocks, the forecast is divided into marked and unmarked components using the proportion of hatchery releases that were marked.   The base period contribution rates of individual stocks are multiplied by the ratio of forecast abundance to base period abundance, and these are summed over marked and unmarked components.

Observed mark rates are either based on at-sea observers who ride along on commercial passenger fishing vessels (CPFVs), or they are based on dockside sampling by port samplers who interview recreational anglers from both CPFVs and private boats.

Observed mark rates would be less than expected mark rates if: 1) the abundance of hatchery fish is less than forecast, 2) the abundance of unmarked fish is greater than forecast, or 3) the survival of marked fish is less than that of unmarked fish.

Hatchery Forecasts

Hatchery forecasts in the time period when we have had fairly extensive mark-selective recreational fisheries have tended to underestimate hatchery abundance (Table 2). Overall, in the years from 2000 to 2010, forecasts have underpredicted the post-season abundance estimate for aggregate hatchery abundance in the Oregon Production Index (OPI) area by 7%. However, the bias this prediction error would cause in mark rates is in the wrong direction; it would tend to produce observed mark rates higher than expected rather than lower. So this can be ruled out as a source of bias.

Differential Survival

There is relatively little information available to evaluate this potential source of bias. The Pacific Salmon Commission’s Selective Fishery Evaluation Committee reviewed available studies and concluded that the difference in survival between unmarked fish and fish marked with an adipose fin clip and coded-wire tag was negligible (SFEC 2006). If there is a difference in survival, it is unlikely that it would be sufficient to compensate for the apparent bias in forecasting hatchery abundance, much less be responsible for the apparent bias in anticipated mark rates.

Natural Abundance Forecasts

If forecasts of natural coho abundance tend to underestimate natural coho abundance, then expected mark rates would be higher than observed mark rates. Preseason Report I (STT 2011b) presents forecasts and post-season reconstructions of abundance for OPI, Washington Coast, and Puget Sound stocks. Average aggregate abundance of all three of these stock groups has been underestimated during the 2000-2010 time period (Table 3). The apparent bias ranges from 9% for the Washington coastal coho stocks, to 37% for Puget Sound coho stocks. For all hatchery stocks combined, the abundance has been underpredicted by an average of 29%. This bias is of the comparable magnitude to the bias in forecasting mark rate, and is in the direction that would produce the observed bias. Thus is seems that underpredicting the abundance of natural coho is likely responsible for most of the observed bias in predicting mark rates for coho.

This explanation is supported by the patterns of correlations between errors in forecasting mark rates and errors in forecasting abundance components (Table 4). The patters of correlations in forecasting error is generally what is expected: correlations are predominantly negative between mark rate and natural stock abundance, and positive between mark rate and OPI hatchery stock abundance. While these patterns are simply a reflection of the structural relationship between mark rate and the abundance of hatchery and natural stock components, it is somewhat reassuring that both forecasts have sufficient precision that observed correlations bear this out. Because these correlations are for year-to-year variability, they do not directly bear on the question of the source of bias. However, the strongest correlations are between mark rate errors and natural stock forecasting errors.

The anomalies to the expected patterns are somewhat surprising. Errors in forecasting OCN coho do not appear to be correlated with errors in forecasting mark rates, and the errors in forecasting OPI hatchery coho (which are primarily from the Columbia River) do not appear to be well correlated with mark rates in the Columbia River catch area. Possible explanations for these anomalies are that OCN coho do not contribute much to fisheries north of the Columbia River, and that in the Columbia Catch area, the majority of fish encountered in the fishery are of hatchery origin, so errors in predicting natural and hatchery components are less important than forecasting the mark rate of the hatchery component.

Consequences of Biased Mark Rates

The major concern about bias resulting in observed mark rates being less than anticipated on average is that more unmarked fish are encountered than anticipated, and consequently more fish may die from incidental mortality in mark-selective fisheries than anticipated during the preseason planning process. While this is a valid concern, the record does not suggest that this has been a problem in mark-selective coho fisheries.

Encounter rates of unmarked to marked fish in mark-selective coho fisheries have, on average, been higher than anticipated (Table 5). In some cases, this bias has been substantial. For example, in the LaPush recreational fishery, averaged over all years, more than one additional unmarked fish has been encountered for every marked fish than was anticipated. However, most of these unmarked fish survive, and as pointed out earlier, this fishery has the smallest landings of any of the coastal fishing areas.

In addition, in most years mark-selective fisheries do not land their preseason expected catch or quota. Comparison of preseason expected catches and incidental mortalities in mark-selective coho fisheries with post-season observations reveals that both catches and incidental mortalities have been less than expected on average (Table 6). The difference between expected catches and observed catches has been sufficiently large that, despite higher than anticipated encounter rates of unmarked coho, post-season estimates of incidental mortalities have been less on average than anticipated. The lower mortalities of unmarked fish, coupled with higher than predicted abundance means that incidental mortality rates have been lower still.

Table 1. Expected and observed mark rates in mark-selective recreational coho fisheries reported in Table I-8 from Review of Ocean Salmon Fisheries reports from 2000 through 2009, and Table I-9 in Review of 2010 Ocean Salmon Fisheries (STT 2011a). Expected values are based on forecasts of hatchery and natural production, and hatchery mark rates. Observed values are based on at-sea observations where available and on shore-based catch sampling where at-sea observations are unavailable.

	Year											
Area	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	Average
	Expected Mark Rates											
Neah Bay	0.48	0.58	0.49	0.51	0.39	0.38	0.45	0.53	0.48	0.57	0.51	0.49
La Push	0.75	0.73	0.48	0.59	0.45	0.40	0.49	0.59	0.53	0.60	0.54	0.56
Westport	0.77	0.80	0.61	0.74	0.57	0.52	0.57	0.65	0.56	0.67	0.59	0.64
Columbia River	0.87	0.86	0.76	0.85	0.68	0.66	0.69	0.72	0.64	0.72	0.67	0.74
Falcon to Humbug	0.81	0.82	0.68	0.72	0.58	0.50	0.58	0.51	0.50	0.56	0.46	0.61
Strait of Juan de Fuca	0.38	0.54	0.49	0.40	0.40	0.33	0.38	0.48	0.48	0.51	0.47	0.44
Buoy 10	0.87	0.83	0.70	0.81	0.58	0.67	0.69	0.74	0.68	0.72	0.66	0.72
	Observed Mark Rates											
Neah Bay	0.34	0.39	0.39	0.39	0.36	0.30	0.40	0.36	0.41	0.39	0.36	0.37
La Push	0.51	0.32	0.28	0.31	0.28	0.31	0.43	0.30	0.35	0.48	0.42	0.36
Westport	0.70	0.57	0.56	0.53	0.46	0.46	0.55	0.51	0.58	0.54	0.51	0.54
Columbia River	0.86	0.78	0.58	0.57	0.58	0.62	0.65	0.61	0.60	0.61	0.50	0.63
Falcon to Humbug	0.74	0.68	0.56	0.44	0.48	0.50	0.52	0.52	0.53	0.41	0.46	0.53
Strait of Juan de Fuca	0.43	0.36	0.36	0.27	0.42	0.45	0.39	0.38	0.43	0.32	0.39	0.38
Buoy 10	0.83	0.79	0.74	0.61	0.66	0.68	0.70	0.60	0.64	0.57	0.69	0.68
	Relative Error (expected - observed)/observed											
Neah Bay	0.41	0.49	0.26	0.31	0.08	0.27	0.13	0.47	0.17	0.46	0.42	0.31
La Push	0.47	1.28	0.71	0.90	0.61	0.29	0.14	0.97	0.51	0.25	0.29	0.58
Westport	0.10	0.40	0.09	0.40	0.24	0.13	0.04	0.27	-0.03	0.24	0.16	0.18
Columbia River	0.01	0.10	0.31	0.49	0.17	0.06	0.06	0.18	0.07	0.18	0.34	0.18
Falcon to Humbug	0.09	0.21	0.21	0.64	0.21	0.00	0.12	-0.02	-0.06	0.37	0.00	0.16
Strait of Juan de Fuca	-0.12	0.50	0.36	0.48	-0.05	-0.27	-0.03	0.26	0.12	0.59	0.21	0.19
Buoy 10	0.05	0.05	-0.05	0.33	-0.12	-0.01	-0.01	0.23	0.06	0.26	-0.04	0.07
Average Error	0.15	0.43	0.27	0.51	0.16	0.07	0.06	0.34	0.12	0.34	0.19	0.24

Table 2. Forecasting error for Oregon Production Index hatchery abundance. Data from Preseason Report I, Table III-1 (STT 2011b). Errors are expressed as relative error (forecast - observed)/observed.

Year	Forecast	Observed	Error
2000	671.4	677.1	-0.01
2001	1,707.6	1,395.5	0.22
2002	361.7	660.1	-0.45
2003	863.1	952.5	-0.09
2004	623.9	634.6	-0.02
2005	389.9	443.1	-0.12
2006	398.8	440.6	-0.09
2007	593.6	476.5	0.25
2008	216.1	565.4	-0.62
2009	1,073.1	1,066.2	0.01
2010	408.0	551.3	-0.26
Total	7,307.2	7,862.9	-0.07

Table 3. Forecasting errors for natural coho stocks. Data from Tables III-1, III-3, and III-4 in Preseason Report I (STT 2011b). Totals for Washington coast and Puget Sound are calculated for 2000-2009 only. Errors are expressed as relative error.

	Oregon Coast			Washington Coast			Puget Sound			Combined		
Year	Forecast	Observed	Error	Forecast	Observed	Error	Forecast	Observed	Error	Forecast	Observed	Error
2000	55.9	69.0	-0.19	58.2	78.6	-0.26	90.1	146.0	-0.38	204.2	293.6	-0.30
2001	50.1	163.2	-0.69	85.4	156.3	-0.45	105.5	307.4	-0.66	241.0	626.9	-0.62
2002	71.8	304.5	-0.76	85.9	173.0	-0.50	120.0	182.2	-0.34	277.7	659.7	-0.58
2003	117.9	278.8	-0.58	103.5	141.3	-0.27	150.8	364.1	-0.59	372.2	784.2	-0.53
2004	150.9	197.0	-0.23	140.1	105.8	0.32	235.8	444.6	-0.47	526.8	747.4	-0.30
2005	152.0	150.1	0.01	115.1	83.7	0.38	170.3	150.5	0.13	437.4	384.3	0.14
2006	60.8	116.4	-0.48	86.0	41.3	1.08	166.9	74.3	1.25	313.7	232.0	0.35
2007	255.4	60.0	3.26	89.2	54.6	0.63	104.6	195.0	-0.46	449.2	309.6	0.45
2008	60.0	170.9	-0.65	67.7	70.5	-0.04	98.3	73.2	0.34	226.0	314.6	-0.28
2009	211.6	257.0	-0.18	119.4	141.0	-0.15	73.5	145.7	-0.50	404.5	543.7	-0.26
2010	148.0	266.8	-0.45	132.0	-	-	95.6	-	-	375.6	-	-
Total	1,334.4	2,033.7	-0.34	950.5	1,046.1	-0.09	1,315.8	2,083.1	-0.37	3,452.7	4,896.1	-0.29

Table 4. Correlations between errors in expected mark rates in Mark-selective coho fisheries and errors in forecasting stock abundance. Reported values are simple correlation coefficients between relative error in forecasting mark rate and error in forecasting stock abundance over 2000-2010 for OCN and OPI hatchery stocks, and over 2000-2009 for the other series.

	Forecast Error				
	Natural				Hatchery
Catch Area	OCN	WACO	PS	Combined	OPI
Neah Bay	0.36	-0.40	-0.60	-0.10	0.52
La Push	0.23	-0.45	-0.65	-0.41	0.39
Westport	0.19	-0.28	-0.72	-0.29	0.71
Columbia River	-0.08	-0.34	-0.44	-0.40	-0.13
Falcon to Humbug	-0.32	-0.42	-0.43	-0.54	0.19
Strait of Juan de Fuca	-0.03	-0.52	-0.51	-0.47	0.15
Buoy 10	0.36	-0.16	-0.35	0.02	0.32
All Areas	0.12	-0.53	-0.72	-0.47	0.40

Table 5. Errors in expected encounter rates. Reported values are the difference between the expected and observed ratio of unmarked to marked fish based on the mark rates reported in Table 1.

	Year											
Area	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	Average
Neah Bay	0.86	0.84	0.52	0.60	0.21	0.70	0.28	0.89	0.36	0.81	0.82	0.63
La Push	0.63	1.76	1.49	1.53	1.35	0.73	0.28	1.64	0.97	0.42	0.53	1.03
Westport	0.13	0.50	0.15	0.54	0.42	0.25	0.06	0.42	-0.06	0.36	0.27	0.28
Columbia River	0.01	0.12	0.41	0.58	0.25	0.10	0.09	0.25	0.10	0.25	0.51	0.24
Falcon to Humbug	0.12	0.25	0.32	0.88	0.36	0.00	0.20	-0.04	-0.11	0.65	0.00	0.24
Strait of Juan de Fuca	-0.31	0.93	0.74	1.20	-0.12	-0.81	-0.07	0.55	0.24	1.16	0.44	0.36
Buoy 10	0.06	0.06	-0.08	0.40	-0.21	-0.02	-0.02	0.32	0.09	0.37	-0.07	0.08
Average	0.21	0.64	0.51	0.82	0.32	0.14	0.12	0.58	0.23	0.57	0.36	0.41

Table 6. Landings and incidental mortality in mark-selective fisheries. Reported values are tabulated from Table I-8 in Review of Ocean Salmon Fisheries reports from 2000 through 2009, and Table I-9 in Review of 2010 Ocean Salmon Fisheries (STT 2011a).

	Landings		Incidental Mortality		
Year	Expected	Observed	Expected	Observed	Difference
Recreational					
2000	193,969	152,610	18,916	17,719	-1,197
2001	420,072	416,590	22,378	42,189	19,811
2002	200,533	148,838	48,558	29,218	-19,340
2003	384,693	345,651	34,144	84,748	50,604
2004	327,931	231,455	37,080	61,967	24,887
2005	207,515	98,476	61,746	24,264	-37,482
2006	134,692	64,779	24,721	14,073	-10,648
2007	189,325	167,716	40,026	38,089	-1,937
2008	49,588	45,809	13,570	9,688	-3,882
2009	425,751	295,417	81,944	72,551	-9,393
2010	123,902	65,069	40,316	15,647	-24,669
Commercial					
2000	21,000	17,294	1,943	2,368	425
2001	75,000	17,445	2,363	4,286	1,923
2002	5,000	1,695	21,200	20,600	-600
2003	75,000	15,668	19,552	6,045	-13,507
2004	67,500	9,805	28,800	6,711	-22,089
2005	23,200	4,064	14,232	3,615	-10,617
2006	6,800	2,679	6,208	1,183	-5,025
2007	22,400	17,441	8,462	3,976	-4,486
2008	22,400	2,084	3,256	632	-2,624
2009	33,600	32,743	10,521	7,263	-3,258
2010	11,800	3,142	11,044	856	-10,188
Totals					
Recreational	2,657,971	2,032,410	423,399	410,153	-13,246
Commercial	363,700	124,060	127,581	57,535	-70,046
Combined	3,021,671	2,156,470	550,980	467,688	-83,292

References

Salmon Technical Team (STT) 2011a. Review of 2010 ocean salmon fisheries. Pacific Fishery Management Council, Portland, OR. February 2011. 335p.

Salmon Technical Team (STT) 2011b. Preseason Report I: Stock abundance analysis and environmental assessment part 1 for 2011 ocean salmon fishery regulations. Pacific Fishery Management Council, Portland, OR. March 2011. 123p.

Selective Fisheries Evaluation Committee (SFEC) 2006. Special assignment review of coded-wire-tags and mark-selective fishing. Pacific Salmon Commission, Vancouver, BC. Report SFEC (06)-1. 32p.

MODEL EVALUATION WORKGROUP REPORT ON 2011 SALMON METHODOLOGY REVIEW

Members of the Model Evaluation Workgroup (MEW) attended the Salmon Methodology Review meeting in October and offer comments on the following topics:

1) Abundance-based management framework for Lower Columbia River tule fall Chinook; presented by Ray Beamesderfer.
2) Cohort reconstruction and harvest impact model for Sacramento Winter run Chinook; presented by Mike O'Farrell.
3) Application of bias-corrected methods, for estimating mortality in mark-selective fisheries, to coho Fishery Regulation Assessment Model (FRAM); presented by Bob Conrad.
4) Causes and effects of bias in anticipated mark rates in mark-selective fisheries for coho salmon; presented by Robert Kope.
5) Update on potential use of a Visual Studio version of FRAM for 2012 pre-season modeling; presented by Andy Rankis.

Lower Columbia River tule fall Chinook Abundance Based Management

The tule Fall Chinook component of the Lower Columbia Chinook ESU is spread over a wide geographic area, spawning in small to large rivers. The status of individual populations varies widely, as does the data available for analysis. The report from the Tule Chinook Workgroup (TCW) acknowledges these data limitations and relies upon the aggregate abundance of hatchery and wild production to evaluate the merits of an Abundance Based Management for this ESU. The report describes fishery management plans for coho and Chinook stocks for which the Council is already using variable exploitation rates (ER) responsive to stock abundance levels. This approach provides additional protection at low abundances and allows for higher harvest levels when the stocks are more abundant. The MEW endorses this management approach. The exploitation rate (ER) values modeled by the TCW provided a reasonable range to evaluate the relative stock risk and fishery benefits from a variable ER approach compared to the current fixed ER approach (37 percent in 2011).

Cohort reconstruction and harvest impact model for Sacramento Winter run Chinook

The cohort reconstruction and harvest model presented at the Methodology Review was well documented, had technical merit, and used available data in an appropriate manner. The MEW supports this work.

Application of mark-selective fisheries bias-corrected methods, for FRAM estimation of coho mortalities

Work on this topic has been presented at the Methodology Review for several years now, demonstrating that bias exists in FRAM estimation of unmarked mortality levels associated with coho mark-selective fisheries (MSF). This year's report is unique in presenting results from a practical application of bias correction methods appropriate for the FRAM model. The work was done using Excel spreadsheets populated with detailed FRAM output for the 2009 and 2010 pre-season coho models. The bias correction equations were applied to each unmarked stock's mortality through all Time Steps. The cumulative bias in total exploitation rates was demonstrated to be less than the earlier theoretical approaches indicated. The effect on individual stock ERs varied, dependent upon each stock's migration patterns through areas implementing MSF and full retention (non-MSF) fisheries. The MEW agrees that the bias correction algorithms developed during this multi-year analysis properly account for the bias in unmarked coho mortalities that MSF introduced to FRAM calculations. At this time Jim Packer (Washington Department of Fish and Wildlife) is coding these equations into a version of FRAM for further evaluation, potentially for use in 2013 Council area coho FRAM modeling.

Causes and effects of bias in anticipated mark rates in mark-selective fisheries for coho salmon

Mark rates in Council area mark selective coho fisheries were compared between the rates in FRAM preseason model runs and the observed mark rates for 2000 to 2010. In general, the modeled mark rates were shown to be biased high, indicating that in the actual fisheries there were more unmarked coho for each marked coho than what the model had predicted. This report suggested that the underestimation of wild coho abundance is likely contributing to the biased modeled mark rates. It has been hypothesized that the relatively consistent underestimation of the number of unmarked coho present in Council area fisheries is the compounded result of errors in stock enumeration and consequently forecasts of the naturally produced component of coho subject to these fisheries.

The report noted that observed coho mortality in ocean MSF fisheries (recreational combined with commercial) was less than modeled pre-season and consequently the number of unmarked mortalities has been less than predicted. But we should not ignore that if the MSF fishery quotas were reached then the unmarked mortalities would be greater than modeled.

Additional sources of the mark rate bias were also discussed. Further analysis to identify the specific reasons for the mark rate bias could involve more refined run reconstruction methods and/or comparisons of genetic stock identification with FRAM based stock composition estimates. A closer look at stock specific Base Period wild and hatchery stock contribution rates to the individual ocean fisheries was also mentioned as relevant to this issue. Identifying the causes of the mark rate bias may be difficult considering the annual variability in survival/productivity of hatchery and natural stocks in Council fisheries.

Update on potential use of a Visual Studio version of FRAM for 2012 pre-season modeling

The FRAM has been recoded from a Visual Basic (VB) application into a Visual Studio (VS) program for both coho and Chinook fishery modeling. The FRAMVS uses a Microsoft Access database to hold model input and output values, rather than an assortment of independent text files that supported the older version. The goal is to use the new VS version for 2012 Council area modeling. The FRAM modelers are still testing the new version, as they learn to use it. Although the VS version is potentially a better modeling tool, the modelers need to be confident in the new version and in their collective ability to use this version. If progress is slower than expected, then the FRAMVB version could be used for 2012 modeling. However, present and future changes to the FRAM model are being applied only to the VS version.

PFMC
10/12/11

SALMON TECHNICAL TEAM REPORT ON 2011 SALMON METHODOLOGY REVIEW

The Salmon Technical Team (STT) and the Salmon Subcommittee of the Scientific and Statistical Committee (SSC) met in Portland, Oct 4 and 5, to review methodology changes and updates for implementation in the 2012 management season.

Abundance Based Management framework for Lower Columbia River tule fall Chinook.
Ray Beamesderfer presented work done by the Tule Chinook Workgroup on evaluating abundance-based management strategies for Lower Columbia natural tule Chinook. The approach employed a population viability model for natural tules with variable exploitation rates determined by the abundance of hatchery tules. Though the model focuses on a hypothetical natural tule population of intermediate productivity, and incorporates a number of simplifying assumptions, it is a reasonable and defensible approach to evaluate the relative risks and benefits of variable harvest rates compared to those of the fixed harvest rate ceiling that is currently the ESA constraint on fishery impacts for tule fall Chinook. The model demonstrated that variable exploitation rate strategies can increase the expected fishing opportunity while reducing risks to natural tule populations. The STT endorses the use of this approach to develop a new consultation standard incorporating abundance-based variable exploitation rates.

Cohort reconstruction and harvest impacts model for Sacramento winter-run Chinook salmon.
The NMFS biological opinion for ocean salmon fisheries with respect Sacramento River winter-run Chinook resulted in a jeopardy opinion. The reasonable and prudent alternative included development of analytical tools to assess and forecast fishery impacts. To help implement this alternative, the NMFS Southwest Fishery Science Center has developed a winter-run cohort reconstruction model and winter-run harvest model (WRHM).

Dr. Michael O'Farrell presented the WRHM, and the cohort reconstruction of hatchery winter-run Chinook used to parameterize the model. The cohort reconstruction confirmed that winter-run Chinook impacts in ocean fisheries are negligible north of Point Arena, and that this stock has a very high age-3 maturation rate with few age-4 fish remaining in the ocean. The WRHM has a structure consistent with the Klamath Ocean Harvest Model and the Sacramento Harvest Model, but utilizes information gained from the cohort reconstruction to limit the spatial extent of the model to the Monterey and San Francisco port areas, and restrict the age composition within the model to the age-3 cohort. Both the cohort reconstruction and the WRHM appear to be technically sound, and together provide the capability to assess ocean fishery impact rates on this listed ESU. The STT endorses the use of the WRHM in 2012 if this capability is needed.

Application of bias-corrected methods for mark-selective fisheries to Coho FRAM.
Dr. Bob Conrad presented work done to evaluate the magnitude of fishery impacts in 2009 and 2010, and bias in recent pre-season Coho Fishery Regulation Assessment Model (FRAM) runs arising from mark-selective fishing and the computational structure of Coho FRAM. Bias in FRAM exploitation rates exhibit a general pattern of increasing in time steps 1 through 3, and decreasing in time steps 4 and 5 due to a combination of fewer mark-selective fisheries and bigger non-selective fisheries in the later time steps. Though biases were generally small, bias corrected estimates of fishery impacts for Council adopted management measures would have exceeded the allowable limits on upper Fraser coho in 2009, and lower Columbia natural coho in 2009 and 2010.

Angelika Hagen-Breaux discussed WDFW efforts to incorporate bias correction into Coho FRAM. The algorithms have been coded, but there is still a bug in the code. Washington Department of Fish and Wildlife will present the modifications to FRAM when the code has been debugged. The STT recommends that current FRAM be used for 2012 season planning, pending review of the bias-corrected Coho FRAM next year.

Review and evaluation of preseason and postseason mark-selective coho fisheries.
Dr. Robert Kope presented a review of mark-selective fisheries for coho from 2000 through 2010, comparing preseason expectations of mark rates and impacts to postseason observations. The review documented bias in preseason expectations of mark rates, with forecast mark rates being consistently higher than values observed during the fisheries. Though this bias has resulted in higher contact rates for unmarked fish that expected, because coho landings have frequently been less than the quotas, this has not resulted in greater incidental mortality of unmarked fish than preseason projections. The bias in preseason projected mark rates appears to be the result of under-forecasting and under-accounting of natural coho abundance. However, further work is recommended to identify the causes of bias and correct it.

Update on conversion of FRAM from Visual Basic to Visual Studio.
Andy Rankis of the Model Evaluation Workgroup (MEW) gave an update on the conversion of FRAM from Visual Basic (VB) to Visual Studio (VS). The VB version of FRAM has been converted to a VS FRAM application for both coho and Chinook pre-season modeling, and the VS version is being tested and evaluated. The VS version uses the same algorithms as the VB version, but has some clear advantages. Inputs and outputs to the VS version are through a Microsoft Access database instead of the strictly formatted files that the VB version uses, and the VS version has better debugging capabilities. State and tribal technical staffs have been using the new version for evaluation purposes and are still encountering a few problems with the VS version. The MEW is hopeful that the VS version will be ready for use in the 2012 preseason process.

PFMC
10/12/11

Agenda Item C.1.b
Supplemental NMFS Report
November 2011

NATIONAL MARINE FISHERIES SERVICE REPORT ON 2011 SALMON METHODOLOGY REVIEW

The National Marine Fisheries Service (NMFS) met with representatives of the Salmon Advisory Subpanel (SAS) after the methodology review meeting in October and discussed, among other issues, an evaluation of a Lower Columbia River (LCR) Tule Chinook abundance based management (ABM) approach. Based on those discussions and on the specific recommendation of the SAS, NMFS recommends the Council conduct a periodic assessment of any ABM approach that is implemented.

The first assessment of the LCR Tule Chinook ABM program should occur after the third year of implementation with additional assessments every three years thereafter. The assessments should include, but not be limited to:

- Forecast Results
    - Compare preseason forecast of lower river hatchery (LRH) Chinook to post season run size estimates
    - Did forecast predict the correct exploitation rate tier?
- Harvest Results
    - Compare preseason exploitation rate limit to post season exploitation rate
    - Compare distribution of harvest between fisheries to those anticipated during the development process (see Table 10 of Agenda Item C.1.a, Attachment 1).
    - Compare quotas and other southern U.S. preseason fishery management provisions with post season results
- Have hatchery production levels changed significantly? Are the changes such that they might affect the results of the preseason forecast?
- Is there new information that might improve the forecast for LRH Chinook? Is there new information that might allow more direct forecasts of LCR tule natural Chinook abundance?
- Provide a narrative summary of management actions taken to meet the year specific circumstances. How much more or less opportunity resulted from implementation of the abundance based scheme relative to what might have occurred under a fixed exploitation rate limit of 37%?

PFMC
11/2/11

SALMON ADVISORY SUBPANEL REPORT ON 2011 SALMON METHODOLOGY REVIEW

Members of the Salmon Advisory Subpanel (SAS) have held a number of discussions regarding the tule fall Chinook matrix developed by Mr. Ray Beamesderfer and the Tule Chinook Workgroup (TCW) (Agenda Item C.1.a, Attachment 1) and would like to provide the following recommendations to the Council.

First, the SAS endorses the five principles under which the matrix was developed. These are:

1. Abundance based management approaches are widely employed in salmon management.
2. Lower River Hatchery (LRH) tule forecasts provide a mechanism to develop an abundance-based approach for natural tule management
3. Small changes in exploitation rates can result in substantial changes for fisheries opportunities.
4. Scenarios that reduce risks to natural populations and provide fishery benefits do exist
5. There is flexibility in developing abundance-based scenarios to achieve desired results.

The SAS recognizes that the National Marine Fisheries Service (NMFS) recommends a risk level reduction of no less than 3.5 percent, and accepts that as a minimum. We also are in agreement with the concept of matching risk reduction with harvest improvements, to achieve a Win/Win scenario. We want to go on record as strongly recommending a check-in by the Council at the end of three years to assess what has worked and what has not, where the model could be refined, and what lessons have been learned from its implementation. We also recommend that the list of monitoring objectives provided by NMFS be adopted (Agenda Item C.1.b, Supplemental NMFS Report). It is entirely possible that the matrix may need to be revised at this point and we want to leave that possibility open.

At this time, we are requesting the Council recommend NMFS use alternative number 68h2 from Attachment 1 in developing Endangered Species Act guidance for salmon fisheries in 2012 and beyond. We point out that the risk column shows a 3.7 percent reduction in risk, which is more conservative than the 3.5 percent minimum recommended by NMFS. This same scenario provides a 5.9 percent benefit to fisheries, and puts the risk/harvest in the Win/Win category. There are numerous scenarios that might provide more for harvest, or reduce risk still further, but this is a reasonable and prudent scenario that accomplishes both goals and we believe it should be implemented in 2012.

We would like to point out that this matrix will only work if hatchery production for tule fall Chinook remains the same. Further reductions in hatchery production will necessitate re-examination of the matrix.

PFMC
11/02/11

SCIENTIFIC AND STATISTICAL COMMITTEE REPORT ON 2011 SALMON METHODOLOGY REVIEW

The Salmon Subcommittee of the Scientific and Statistical Committee (SS-SSC), the Salmon Technical Team (STT), and the Model Evaluation Workgroup (MEW) met at the Council office in Portland on October 4 and 5, 2011, to review the four salmon methodology issues identified by the Council at the September meeting:

- Abundance-based management framework for Lower Columbia River (LCR) tule fall Chinook,
- Cohort reconstruction and harvest impact model for Sacramento winter run Chinook,
- Examination of the potential bias in Coho Fishery Regulation Assessment Model (FRAM) from mark-selective fisheries, and
- Review and evaluation of preseason and postseason mark-selective fisheries north and south of Cape Falcon.

Abundance-based management framework for Lower Columbia River tule fall Chinook

Mr. Ray Beamesderfer presented the work of the Tule Chinook Work Group (TCW) on evaluating the relative risk and relative change in fishing opportunities presented by alternative abundance-based management (ABM) approaches (Agenda Item C.1.a Attachment 1). ABM could provide more protection for weak runs, and more fishing opportunity on large runs.

The model was similar to the one used to evaluate Klamath River Fall Chinook *de minimis* fisheries which has been reviewed by the SSC. Data are limited for LCR wild tule Chinook (LRN) so LCR hatchery tule Chinook (LRH) were used as a proxy for predicting the status of the stocks. The Population Viability Model developed included both hatchery and natural stocks in a single model. Results are dependent on the current mix of hatchery and natural stocks.

The model evaluated conservation risk and harvest benefits under a variety of ABM scenarios. Conservation risk was expressed as the probability of natural stocks falling below a critical threshold in 20 and 100 years. Harvest benefit was expressed as change in average harvest numbers over 100 years. "Win/win" scenarios with reduced risk and increased benefits were recommended for further consideration. Consecutive years of restricted fishing are especially damaging to the viability of fisheries. The SSC recommends evaluating the probability of multi-year closures and the median length of closures as additional criteria for comparing scenarios.

The analysis assumes that hatchery production remains constant. If hatchery production changes, then the tier structure will need to be reevaluated. Furthermore, tier frequency of occurrence is modeled on recent past environmental conditions but will be dependent on patterns of future environmental conditions and may not match model expectations in the near future. The SSC considers the methods to be reasonable for addressing the relative risks and benefits. With the addition of a closure analysis, the results will give insights into social and economic effects and be adequate for setting harvest policy.

Cohort reconstruction and harvest impact model for Sacramento winter run Chinook

Dr. Mike O'Farrell (STT) gave presentations on the cohort reconstruction for Sacramento winter run Chinook (Agenda Item C.1.a Attachment 2) and the harvest impact model developed for Sacramento winter run Chinook (Agenda Item C.1.a Attachment 3).

Cohort reconstructions were performed for ten broods (1998–2007) of hatchery-origin Sacramento winter run Chinook (SRWC) using coded-wire tag data. The results of the cohort reconstruction indicated that the majority of ocean fishery impacts came from recreational fisheries south of Point Arena, California. For complete broods 1998–2005, the number of potential SRWC spawners was reduced by an estimated 11 to 28 percent due to ocean salmon fisheries. In the future, consideration of genetic stock identification (GSI) data may help to more closely define the distribution of SRWC in the area south of Point Arena.

The winter run cohort reconstruction was reviewed by the Center for Independent Experts (CIE) in March 2010 and its comments were incorporated into the analyses presented for Council review. The SSC considers this cohort reconstruction to provide the best available estimates of:

a) past SRWC fishery impacts, by time and area, and
b) parameters needed for the winter run Chinook harvest impact model.

The Winter Run Ocean Harvest Model (WRHM) is similar to the Klamath Ocean Harvest Model (KOHM) and Sacramento River Harvest Model (SHM) have been previously reviewed by the SSC and STT and approved for Council use. The three ocean harvest models treat age structure differently. The KOHM is fully age-structured, the SHM combines all ages and is not age structured, and the WRHM models only age 3 fish. A size-at-age model is incorporated into both the KOHM and WRHM in order to forecast release mortality incurred by sublegal size fish. In contrast to the KOHM and SHM, the WRHM does not account for in-river fisheries, as winter run Chinook are rarely harvested in the Sacramento River.

The SSC considers the WRHM a significant improvement in the Council's ability to model and project harvest impacts on Sacramento winter run Chinook, and endorses the model for Council use. The SSC compliments the authors for providing thorough and comprehensive documents, which greatly facilitated the review process.

Examination of the potential bias in Coho FRAM from mark-selective fisheries

Mr. Robert Conrad and Ms. Angelika Hagen-Breaux presented an evaluation of the bias in Coho FRAM estimates of the mortalities for unmarked stocks when mark-selective fisheries operate during a FRAM time step (Agenda Item C.1.a Attachment 4). This has been a difficult issue because the calculations needed to make a rigorous bias adjustment cannot be implemented in the current FRAM. The authors have developed and tested an alternative method to estimate the bias within the FRAM framework.

The analysis compared stock-specific fishery exploitation rates (ER) for unmarked stocks using the standard FRAM to bias-corrected estimates calculated from FRAM output for the years 2009 and 2010. The average differences by which FRAM underestimated the total exploitation rate for unmarked stocks were very low: -0.003 in 2009 and -0.002 in 2010.

In the standard FRAM model, the bias increases with the number and intensity of mark-selective fisheries. Bias in this analysis was low because mark-selective coho fisheries in 2009 and 2010 tended to be relatively low in intensity and concentrated in earlier time periods. In the final time step fisheries are typically more intensive and non-selective. The nature of the FRAM model is to overestimate unmarked mortalities in these terminal fisheries, partially balancing the underestimation of mortalities in earlier mark-selective fisheries. As long as the pattern of fisheries is similar to those in 2009 and 2010, overall bias in the FRAM model is expected to be low.

Although bias was generally low, accounting for bias could be important for stocks that are managed for exploitation rate guidelines. Without bias correction, ER guidelines could be exceeded. This appeared to be a risk for Fraser River Coho and Lower Columbia River Coho. The differences between the FRAM and bias-corrected ERs in time step 4 (September) were large enough so that these stocks may have exceeded ER guidelines due to lack of bias accounting.

The current testing excluded several significant sources of mortality including catch non-retention (e.g., coho mortality in Chinook fisheries), drop-off mortality, and mark recognition errors. For this reason the total mortality rates reported in these analyses are generally lower than rates that were modeled by the STT. The bias correction results reported could not be compared with more analytically rigorous bias estimates. However, the degree of bias is consistent with the theoretical modeling that the SSC reviewed in 2010.

The SSC recommends that the proposed bias-correction methods be implemented and tested in FRAM. Testing should include code evaluation and verification of results under a variety of fisheries scenarios and with the full set of mortality factors. This implementation should be available for methodology review in 2012 prior to adoption for use in 2013 fisheries modeling. For 2012 fisheries modeling, the SSC recommends continuing to use their interim guidance, including a pre-season evaluation of impacts. The Council may choose to include a precautionary buffer for stocks with exploitation rate guidelines.

Review and evaluation of preseason and postseason mark-selective fisheries north and south of Cape Falcon

Dr. Robert Kope (STT) presented an evaluation of causes and effects of bias in anticipated mark rates in the ocean recreational mark-selective fisheries for coho salmon in 2000 – 2010 (Agenda Item C.1.a Attachment 5). More unmarked fish are typically encountered per marked fish landed in the ocean mark-selective fisheries than expected pre-season, raising the concern that more unmarked fish may be killed as a result of incidental mortality than is projected pre-season.

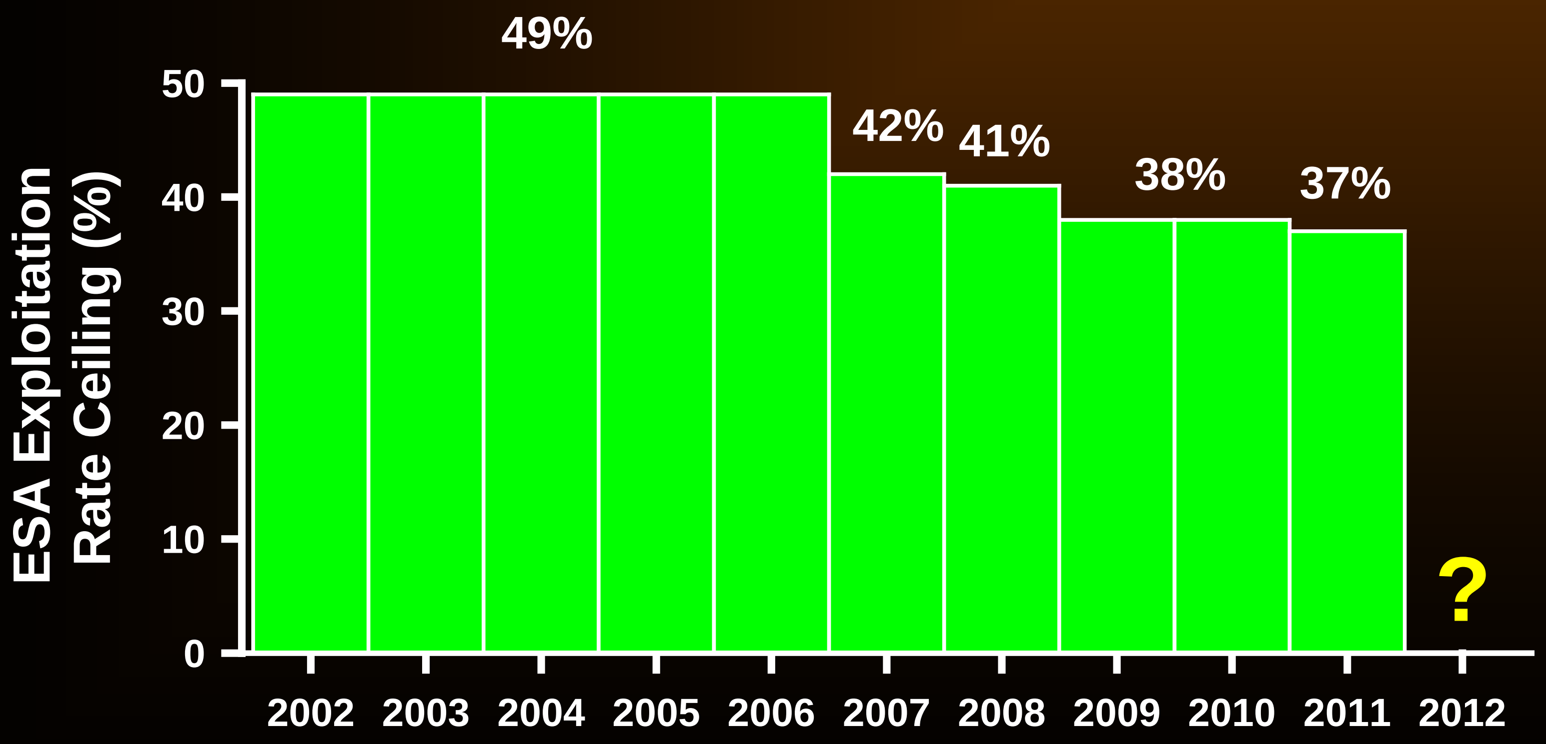
Bias was apparent in the expected mark rates, and varied by year and by management area. Several possible causes of the bias were investigated, including: over-predicting marked hatchery fish abundance; under-predicting unmarked fish abundance; and a differentially lower survival of marked fish relative to that of unmarked fish. The report concluded that under-predicting natural coho abundance was the most likely cause of much of the observed bias in expected mark rates. The report also noted that post-season estimates of incidental mortalities due to the release of coho in mark-selective fisheries have been less on average than predicted pre-season because mark-selective fisheries generally have not landed their pre-season expected catch or quota. The SSC notes that mark recognition errors and incorrectly reported hatchery mark rates could also contribute to the bias.

The SSC recommends that this issue continue to be examined.

PFMC
11/02/11

Agenda Item C.1.b
Supplemental TCW Report
November 2011

Abundance-based Management
LCR Tule Chinook



Progress Checklist

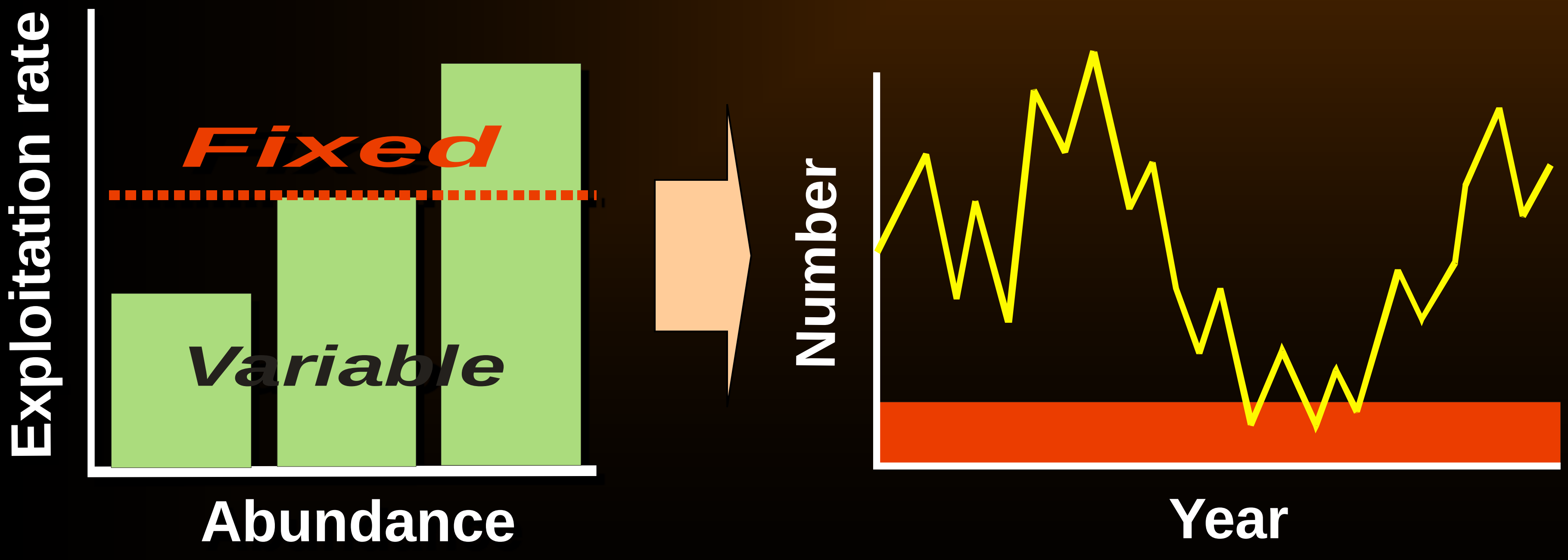
- ✓ 2010 Tule Work Group
- ✓ Sept Council Update
- ✓ SSC Review
- ✓ SAS Alternatives
- ❑ Supplemental Analysis
- ❑ Council Action
- ❑ Future Evaluations

Abundance Based Management

Win/Win Scenarios

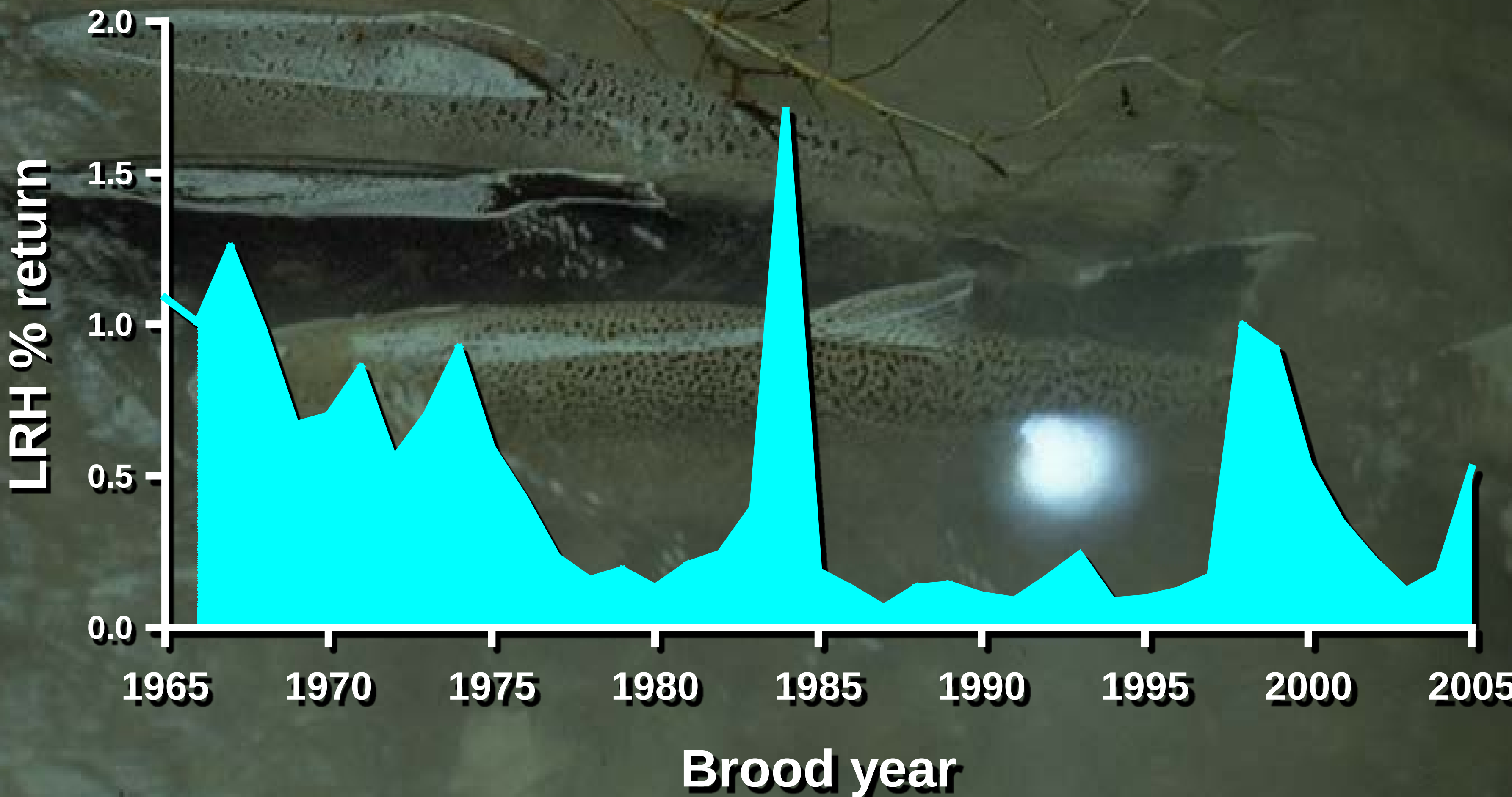
Risk = -3.5%

Harvest = +3.0%

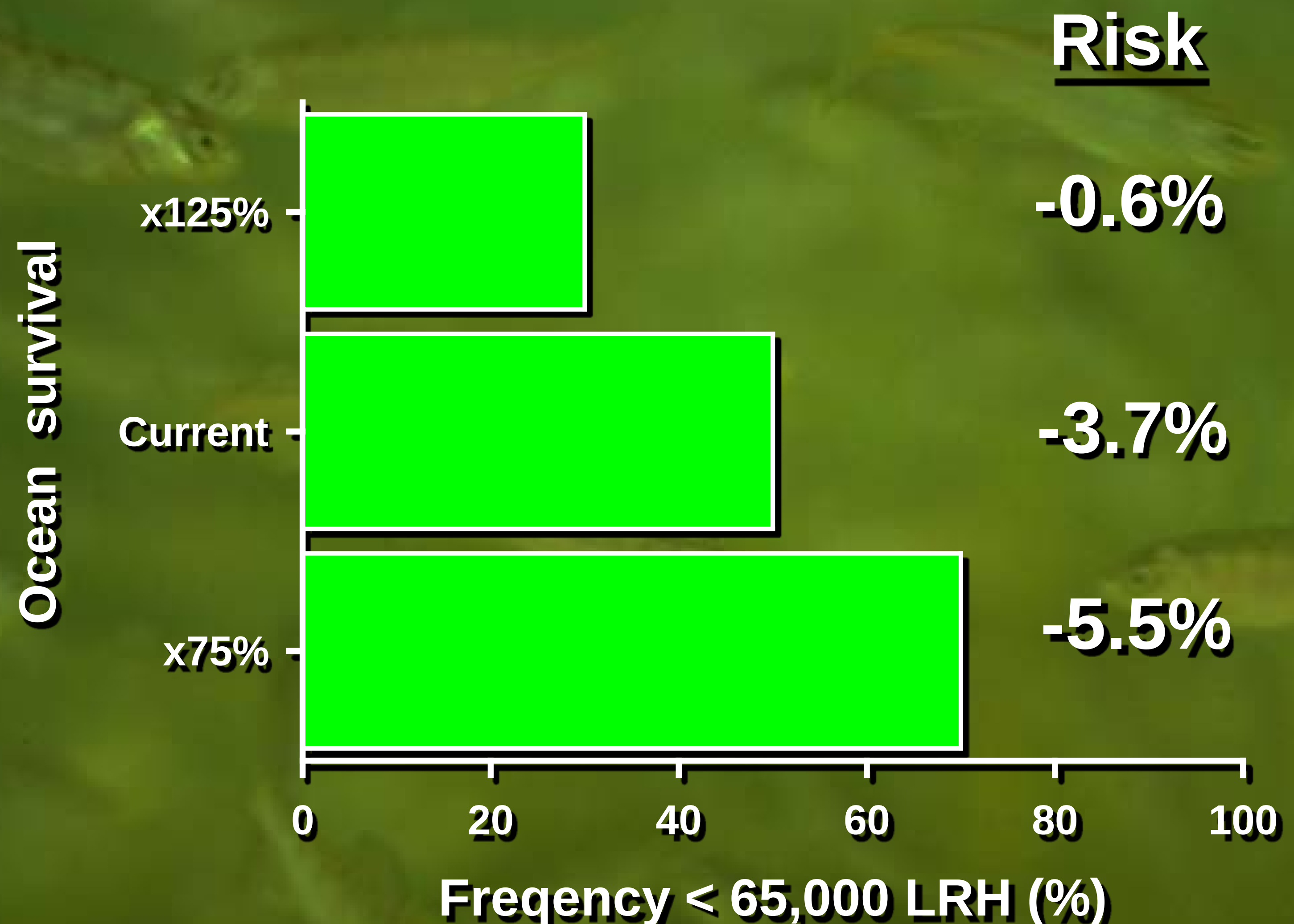


#1: Forecast Errors?

#2: Serial bad years?



#3: Future Ocean?



Scenario Examples

	tiers				Rates (by tier)					Change		
#	1	2	3	4	1	2	3	4	Frequency	Risk	Harv	Category
68g	<30	30-40	40-80	>80	30%	36%	38%	41%	11/11/42/35	-3.1%	6.4%	Fishery ↑
68f	<30	30-40	40-80	>80	30%	36%	38%	40%	11/11/42/35	-3.2%	4.8%	Fishery ↑
68i	<30	30-50	50-100	>100	30%	36%	39%	41%	11/24/42/23	-3.5%	5.9%	Win/Win
68h1	<30	30-40	40-80	>80	30%	35%	38%	41%	11/11/42/35	-3.5%	6.2%	Win/Win
68j	<30	30-50	50-110	>110	30%	36%	39%	42%	11/24/47/19	-3.5%	6.6%	Win/Win
68h2	<30	30-40	40-85	>85	30%	35%	38%	41%	11/11/46/32	-3.7%	5.9%	Win/Win
68d	<30	30-40	40-80	>80	30%	35%	38%	40%	11/11/42/35	-3.8%	4.6%	Win/Win
68h3	<30	30-40	40-90	>90	30%	35%	38%	41%	11/11/49/29	-3.8%	5.6%	Win/Win
68c	<30	30-40	40-100	>100	30%	35%	38%	40%	11/11/55/23	-4.1%	3.9%	Win/Win

**Excerpts from Table 23, pg. 58*

Agenda Item C.2
Situation Summary
November 2011

PRESEASON SALMON MANAGEMENT SCHEDULE FOR 2012

To plan, announce, and meet *Federal Register* deadlines for public hearing sites and the entire preseason salmon management process, staff needs to confirm details of the process prior to the end of November. The proposed 2012 process and schedule are contained in Agenda Item C.2.a, Attachment 1.

For 2012, Council staff recommends one salmon management option hearing per coastal state, the same schedule as in 2011. The hearings would be:

March 26, 2012 Westport, Washington and Coos Bay, Oregon
March 27, 2012 Eureka, California

In 2012, the March Council meeting will occur in Sacramento, California and the April Council meeting in Seattle, Washington. Therefore, the public comment period on Monday of the April meeting in Seattle also serves as a public comment opportunity. If the states desire to have additional hearings, we suggest they organize and staff them as was done in past years. The table below provides the public attendance at the hearing sites since 1998 for Council reference.

Hearing Site Location[1/]	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Westport	4	18	24	30	11	16	16	25	26	34	20	27	21	54
Astoria		14												
Tillamook	28		13	16 [2/]	18 [2/]									
Coos Bay	15	31	36	18	40	26	26	105	146	43	60	108	60	19
Eureka	16	18	37	12	25	46	-				167	65	34	41
Ft. Bragg							27	38						
Sacramento	13													
Santa Rosa			4						500	35				
Moss Landing[2/]	100	51	50	33	14									

1/ Sites in bold are proposed for Council staffing in 2012.
2/ Hearing staffed by state personnel.

Council Action:

1. **Confirm Council-staffed hearing sites and state intentions for additional hearings.**
2. **Approve staff's overall proposed schedule and process for developing 2012 ocean salmon management measures.**

Reference Materials:

1. Agenda Item C.2.a, Attachment 1: Pacific Fishery Management Council Schedule and Process for Developing 2012 Ocean Salmon Fishery Management Measures.
2. Agenda Item C.2.b, STT Report: Salmon Technical Team Statement on the Preseason Management Schedule for 2012.

Agenda Order:

a. Agenda Item Overview — Chuck Tracy
b. Reports and Comments of Advisory Bodies and Management Entities
c. Public Comment
d. **Council Action**: Adopt a 2012 Preseason Management Schedule

PFMC
10/12/11

PACIFIC FISHERY MANAGEMENT COUNCIL SCHEDULE AND PROCESS FOR DEVELOPING 2012 OCEAN SALMON FISHERY MANAGEMENT MEASURES

Nov 1-7, 2011	The Council and advisory entities meet at the Hilton Orange County, Costa Mesa, California, to consider any changes to methodologies used in the development of abundance projections or regulatory options.
Jan. 17-20, 2012	The Salmon Technical Team (STT) and National Marine Fisheries Service (NMFS) economist meet in Portland, Oregon to draft *Review of 2011 Ocean Salmon Fisheries*. This report summarizes seasons, quotas, harvest, escapement, socioeconomic statistics, achievement of management goals, and impacts on species listed under the Endangered Species Act. (February 7 print date, available on-line February 10.)
Feb. 21-24	STT meets in Portland, Oregon to complete *Preseason Report I Stock Abundance Analysis and Environmental Assessment Part 1 for 2012 Ocean Salmon Fishery Regulations*. This report provides key salmon stock abundance estimates and level of precision, harvest and escapement estimates when recent regulatory regimes are projected on 2012 abundance, and other pertinent information to aid development of management options (February 29 print date, March 1 mailed to the Public and available on-line).
Feb. 25 through Mar. 1	State and tribal agencies hold constituent meetings to review preseason abundance projections and range of probable fishery options.
Mar. 2-7	Council and advisory entities meet at the DoubleTree Hotel Sacramento, CA to adopt 2012 regulatory alternatives for public review. The Council addresses inseason action for fisheries opening prior to May 1 and adopts preliminary alternatives on March 4, adopts tentative alternatives for STT analysis on March 5, and final alternatives for public review on March 7.
Mar. 12-16	The STT completes *Preseason Report II: Proposed Alternatives and Environmental Assessment Part 2 for 2012 Ocean Salmon Fishery Regulations* (March 19 print date, March 20 available to the public).
Mar. 12-31	Management agencies, tribes, and public develop their final recommendations for the regulatory alternatives. North of Cape Falcon Forum meetings are tentatively scheduled for March 13-15 and March 27-29.
Mar. 20	Council staff distributes *Preseason Report II: Proposed Alternatives and Environmental Assessment Part 2 for 2012 Ocean Salmon Fishery Regulations* to the public. The report includes the public hearing schedule, comment instructions, alternative highlights, and tables summarizing the biological and economic impacts of the proposed management alternatives.

Mar. 26-27 Sites and dates of public hearings to review the Council's proposed regulatory options are: Westport, Washington (March 26); Coos Bay, Oregon (March 26); and Eureka, California (March 27). Comments on the options will also be taken during the Council meeting on April 2 in Seattle, Washington.

Apr. 1-6 Council and advisory entities meet to adopt final regulatory measures at the Sheraton Seattle Hotel, Seattle, Washington. *Preseason Report II: Proposed Alternatives and Environmental Assessment Part 2 for 2012 Ocean Salmon Fishery Regulations*, results from the public hearings, and information developed at the Council meeting is considered during the course of the week. The Council will tentatively adopt final regulatory measures for analysis by the STT on April 2. Final adoption of recommendations to NMFS is tentatively scheduled to be completed on April 6.

Apr. 7-20 The STT and Council staff completes *Preseason Report III: Analysis of Council-Adopted Management Measures for 2012 Ocean Salmon Fisheries* (April 16 print date, mailed to the Council and available to the public April 17). Council and NMFS staff completes required National Environmental Policy Act documents for submission.

Apr. 17 Council staff distributes adopted ocean salmon fishing management recommendations, and *Preseason Report III* is made available to the public.

May 1 NMFS implements Federal ocean salmon fishing regulations.

PFMC
10/12/11

Agenda Item C.2.b
STT Report
November 2011

SALMON TECHNICAL TEAM REPORT ON PRESEASON SALMON MANAGEMENT SCHEDULE FOR 2012

The Salmon Technical Team (STT) recommends that their meeting to draft the Review of 2011 Ocean Salmon Fisheries be held January 17-20, 2012, and their meeting to draft Preseason Report I be held February 21-24, 2012. The timing of the February meeting will be too late to have Preseason Report I completed before the deadline for 2012 March Council briefing book materials, but several key members of the STT will be at the Pacific Salmon Commission meeting the week of February 13-17, and abundance forecasts will not be available for several stocks the week of February 6-10.

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10/12/11