

FISHERY RESEARCH



IDAHO NATURAL PRODUCTION MONITORING AND EVALUATION

Annual Progress Report
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IDAHO NATURAL PRODUCTION MONITORING AND EVALUATION

Project Progress Report

2003 Annual Report

- Part 1—Monitoring age composition of wild adult spring and summer Chinook salmon returning to the Snake River basin in 2003 to predict smolt-to-adult return rates**
- Part 2—Development of a stock-recruitment relationship for Snake River spring/summer Chinook salmon to forecast natural smolt production**
- Part 3—Improve the precision of smolt-to-adult survival rate estimates for wild steelhead trout by PIT tagging additional juveniles**

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To

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**PART 1—MONITORING AGE COMPOSITION OF WILD ADULT SPRING/SUMMER
CHINOOK SALMON IN THE SNAKE RIVER BASIN TO ESTIMATE SMOLT-TO-ADULT
RETURN RATES**

ABSTRACT

Accurate determination of the ocean-age proportions of wild/natural adult spring/summer Chinook salmon is important for monitoring status and trends. This report covers research to determine the age composition of wild adult spring/summer Chinook salmon returning to the Snake River Basin in 2003. Chinook salmon carcasses were sampled from representative spawning areas throughout the Snake River basin. I determined proportions of carcasses in each ocean-age category for each 5-cm fork length group. I estimated the number of wild adult returns in each ocean-age group by applying the ocean-age proportions by length group from carcasses to the length frequency distribution of wild Chinook salmon adults observed at Lower Granite Dam. Finally, aggregate smolt-to-adult return rates for migratory years 1996-2003 were calculated by dividing adult return estimates to basin-wide smolt production estimates reported in the stock-recruitment section of this report.

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INTRODUCTION

Age information is an important tool for management and recovery of Snake River spring/summer Chinook salmon *Oncorhynchus tshawytscha*. Accurate age data is essential to assign returning adults to a specific brood year and, therefore, to estimate smolt-to-adult return rates. Many agencies use scales to age Pacific salmon *Oncorhynchus sp.* However, when Pacific salmon leave the ocean for their spawning migration, they cease feeding, and scale material is resorbed. Resorption results in the loss of annuli on the periphery of scales, making accurate age determination difficult for salmon with long spawning migrations such as Snake River Chinook salmon (Chilton and Bilton 1986). Scales may be used for determining the freshwater age; however, freshwater age in wild Snake River Chinook salmon is typically one year and, therefore, age data is not necessary for determining out-migration year. This project began in 1998 to use fin rays collected from carcasses to determine ocean ages. Based on known-age adult Chinook salmon that were tagged as juveniles with Passive Integrated Transponder (PIT) tags, ocean ages of adult Chinook salmon were >97% accurate in any given year (Kiefer et al. 2002). In this report, I present the ages of carcasses collected in 2003 throughout the Snake River basin, wild Chinook salmon lengths calculated from Lower Granite Dam videotapes, proportions of age at length, and smolt-to-adult return rates for brood years 1996-2002.

STUDY AREA

The study area encompasses streams in the Snake River basin upstream of Lower Granite Dam (LGD) that are currently accessible to wild/natural spring/summer Chinook salmon. Field personnel sampled carcasses from representative spawning areas throughout the study area (Figure 1).

METHODS

Sampling

To promote quality and consistency of samples and data collected in the field, a spawning survey manual was produced and distributed that illustrated the proper techniques for collecting aging structures and recording data. Interagency redd count training took place at Marsh Creek near Stanley, Idaho. Idaho Department of Fish and Game (IDFG) trainers demonstrated techniques for collecting aging structures and recording data.

Field personnel recorded stream name, fish species, types of samples collected, date, marks, tag numbers, sex, fork length (to nearest cm), mid-eye to hypural length (to nearest cm), collector name, and any noteworthy comments about the fish or samples. Dorsal fins were collected using a serrated knife. The fin was held at a 90° angle to the body and removed by making a cut level with the body of the fish while pulling upward on the fin. The fin was inserted into a coin envelope with base exposed and fin rays aligned perpendicular to the fin base. A small pencil-eraser-size piece (approximately 16 mm²) of fin tissue with good color (excluding the adipose fin) was removed and placed in a test tube filled with 95% ethanol. These tissue samples were catalogued and stored for future genetic analysis.

All samples were placed in uniquely numbered, pre-labeled bags. The identification number contained two digits to identify year, a dash, and four other digits. The latter allowed identification of up to 9,999 unique carcasses sampled in a year. For example, sample containers for the 199th packet assembled for samples collected in 2003 would be labeled 03-0199. Samples were transported to the IDFG Fisheries Research Office in Nampa, Idaho, and were stored in a freezer until preparation for aging.

The majority of samples were collected on spawning grounds from carcasses of wild adults that died naturally. A few samples were collected from wild adult carcasses that floated down to adult trapping weirs. Hatchery personnel collected 30 dorsal fins from known-age hatchery adults at Rapid River, McCall, and Dworshak hatcheries. I used known-age samples to estimate accuracy in the current year. The archive of past known-age fin rays is used as training material for new personnel and as a reference to help identify split annuli and other abnormalities. I used a Microsoft[®] Access database to store data and track samples.

Fin Preparation

Dorsal fins were stored in the freezer until use. Laboratory technicians removed dorsal fins from the freezer and placed them in specially designed wooden racks to dry. The fin rays were adjusted so the base of the fin was perpendicular to the rays. After drying 24 hours, the technicians removed excess material (i.e. bones, loose skin, and flesh), then epoxied and placed them on waxed paper to harden overnight under a fume hood. After hardening, the respective sample number was written on each fin.

The laboratory technicians cut each fin into thin sections by making eight to ten 1.2 mm slices in each individual fin using a water-cooled, Bronwill diamond-grit saw. Cross-sections were placed on a microscope slide to dry by the order in which they were sliced. Sections were affixed onto microscope slides under the fume hood using Flo-texx[™], a clear liquid mounting medium to improve resolution and preserve the sample.

Fin Aging

Mounted fin ray cross sections were aged with the use of a compound microscope and green, filtered, transmitted light. Light passing through the individual fin ray sections illuminated wide translucent zones alternating with narrower opaque zones (Figure 2). Translucent zones represent material deposited during the summer period of rapid growth, while opaque zones represent material deposited during the winter period of slow growth (Ferreira et al. 1999). The opaque zones are annual marks. Technicians counted these annuli to age the fish. Annuli develop from the center outward as the fish and the fin ray grow, adding rings much the way trees do. Wild spring/summer Chinook salmon in the Snake River basin usually spend one winter in freshwater rearing areas before smolting and migrating to the ocean. The bright freshwater annulus is near the center of the fin ray. It is kidney-shaped and has two curved lines underneath. Ocean annuli are broader and not as bright as the freshwater annulus, because ocean winters are not as cold as those spent in freshwater, permitting some growth (Figure 2). Many fin rays exhibit split annuli. Undetected split annuli could cause the technician aging the sample to overestimate the true age. I assembled a reference collection of fin rays from known-age Chinook salmon to help personnel determine when there is a split and what characteristics to look for. This reference collection was comprised of Chinook salmon tagged as smolts with PIT tags and recovered as returning adults. Thus, known-age samples and at least one experienced professional to age fins are critical to the project's success.

All samples were aged independently by at least two technicians trained in fin aging techniques. I used the reference collection of known-age fish to train the laboratory technicians. Fins were read again in a referee session if there was disagreement in age determination, or if the determined age did not match what was expected for the fish's length. During a referee session, a camera was attached to the microscope to display the image on a television screen. Three trained personnel then viewed the fin together and arrived at a consensus age. In some cases, a consensus could not be reached.

Ocean Age Proportions at Lower Granite Dam

I used video images to determine the length frequency distribution of wild adult spring/summer Chinook salmon passing LGD in 2003. A video camera recorded adults passing the viewing window at LGD for 24 hours every third day from April through August. I randomly selected the initial start day for recording videos from the anticipated first week of the migration. The random start established the video recording schedule for the rest of the season. At the end of the adult migration season, the videotapes were shipped to the IDFG Nampa Fisheries Research Office for analysis. I estimated the number of viewing minutes necessary to obtain approximately 400 length samples, based on the number of adults passing LGD in the 2003 migration season and the number of minutes of videotape.

I randomly selected two 26 min segments of each videotape to measure images of adults passing the viewing window. The video images were examined for an adipose fin. Fish with a full adipose fin were assumed to be of wild origin. Chinook salmon missing all or part of the adipose fin were assumed to be of hatchery origin. The images of wild adults were digitized for length analysis. A small percentage of adults with an adipose fin were actually hatchery fish (missed clips or fish experimentally stocked by Idaho Supplementation Studies), but this percentage could not be determined from the video images.

I estimated the fork length of each selected fish image based on the number of pixels between the nose and caudal fin fork. Several vertical lines were placed on the LGD viewing window. The distance between the outer two lines is known; therefore, the value of an image pixel is also known. To correct for image distortion by the water column, sticks of known lengths (62 cm, 85 cm, and 100 cm) were passed through the viewing window and recorded on the video tape. The ratio of stick lengths (in pixels) to the known length between the lines on the window was calculated. I used this ratio to estimate the actual fork length of the digitized images. A length frequency distribution for all unmarked adult Chinook salmon passing LGD was constructed using these estimated fork lengths. The proportion of each ocean age per 5 cm length group was estimated from the carcass age data. To estimate the ocean-age proportions of the run at large, I multiplied the proportion of the video image sample in each length group by the proportion of that length in each age category based on the carcass sample. I applied the resulting ocean-age proportions of the run at large to the number of wild Chinook salmon adults passing LGD provided by the United States v. Oregon Technical Advisory Committee (TAC) to estimate the total number of adult returns for each ocean-age group in 2003.

There were several problems with the videotapes in 2003. Measuring lines were missing from the viewing window. Fish migrating prior to April 14, 2003 were not recorded on videotape. There were also video camera malfunctions. The following steps were taken to deal with these issues: 1) Measurements of the white backboard in the viewing chamber were taken and compared with the white measuring lines on the window; these backboard measurements were

then compared to video with no measuring lines and found to be the same. The window line measurements were then applied to the images with no lines to calculate lengths. 2) The number of fish migrating early were determined to be <1% of the entire run; therefore, they should have a negligible effect on the overall age at length calculations. 3) Hours of missing video were compensated for by randomly selecting alternate sample times from available video minutes.

Estimating Aggregate Smolt-to-Adult Return Rates

I combined the results of the aging research with data from a stock-recruitment analysis (see Part 2 of this report) to estimate an aggregate smolt-to-adult return rate (SAR) estimate for Snake River wild Chinook salmon (Table 1). For a particular smolt year, I used the estimate of wild smolts arriving at LGD from Part 2 as the denominator in the SAR estimates. The numerator was the sum of one-ocean adults that returned the year after the smolt year (smolt year +1), two-ocean adults that returned (smolt year +2), three-ocean adults that returned (smolt year +3), and four-ocean adults that returned (smolt year +4). I reported smolt-to-adult return rates as a percentage (%). Smolt-to-adult-returns were reported with and without one-ocean returns (jacks). Since 1998 was the first year for which this project had aging results, SAR estimates were calculated for smolt year 1996 without jacks. Smolt-to-adult return rate estimates were completed only through three-ocean returns for smolt year 2000 and two-ocean returns for smolt year 2001.

RESULTS AND DISCUSSION

Known Age Adults

Field technicians and hatchery personnel collected 40 known-age unmarked fish and 43 known-age hatchery fish in 2003. Age could not be independently determined for one fish. Laboratory technicians aged 81 of the 82 (99%) known-age fish correctly. In the known-age sample, there were nine one-ocean fish, 20 two-ocean fish, and 52 three-ocean fish. No four-ocean fish with PIT tags returned to collection points.

Wild Adult Carcass Age Determinations

Technicians examined 700 fin ray samples. Ages could not be assigned to 14 samples. The majority of the carcasses aged in 2003 were three-ocean returns (Figure 3). Ocean-age proportions of carcasses collected and aged (Table 3) may not be the same as the ocean-age proportions of the entire population passing Lower Granite Dam. I believe that collectors were more likely to find larger carcasses, because they are easier to see and a larger body is more likely to be caught on rocks.

Estimated Ocean Age Proportions of Wild Adults Passing LGD

The fork length frequency of the fish observed at LGD was similar to that of the carcasses aged (Figure 4). The largest proportion of the run was in the 95-99 cm group (Table 2). I multiplied the estimated ocean age proportions of each 5 cm length group among fish aged from visual examination of the rays (Table 3) by the estimated proportion at LGD for each length group (Table 2) to estimate the overall ocean-age proportions for wild adults

passing LGD (Table 4). I multiplied the 2003 TAC estimate for the number of wild adults passing LGD (37,157; Scott Marshall, IDFG, personal communication) by the estimated ocean-age proportions at LGD to estimate the number of wild adult returns for each ocean age (Table 4). I separated the total 2003 return into spring and summer runs (Table 5). Chinook salmon with unknown run designations were not included; therefore, numbers sampled are slightly different from the numbers in Table 3.

Aggregate Smolt-to-Adult Return Rates

Estimated SAR values of wild Snake River spring/summer Chinook salmon (excluding jacks) ranged from a low of 0.35% for smolt year 1996 to a high of 3.56% for smolt year 2000 (Table 1). Estimated SAR values were comparable to those determined for wild Chinook salmon using PIT tag data for these same four smolt years (Kiefer et al. 2001).

Length at Age Differences Between Spring and Summer Chinook

Sample sizes were not evenly distributed among ages and years or even between run types (Table 6). Few jacks have been sampled over the years due to small jack returns or under-sampling. Little emphasis has been placed on collection of jacks, because there is very little overlap in length between jacks and two-ocean fish.

Before statistically removing the effects of annual fluctuations, there were some obvious differences in mean lengths between spring- and summer-run Chinook salmon during 1999-2003 (Figure 5). Two-ocean spring Chinook salmon were significantly smaller than two-ocean summer Chinook salmon. For one-ocean fish, the difference was significant only in 2003. I did not detect a significant length difference between spring and summer runs for three-ocean Chinook salmon.

Spring Chinook salmon were smaller than summer Chinook salmon as jacks, once the year effects were accounted for (Figure 6). The overall fork length in each year (runs combined) showed no significant difference until fork length dropped in 2003 (Figure 7). These trends may not be representative of the entire Snake River population, as smaller sample sizes may not provide enough data to make an accurate determination.

Two-ocean summer Chinook salmon were larger than two-ocean spring Chinook salmon (Figure 8). After run-type differences were accounted for statistically, two-ocean fish did not show large annual fluctuations in length, although lengths were greater in 2003 over 2002 (Figure 9).

As in the younger ages, spring Chinook salmon in the three-ocean age group tend to be smaller than summer Chinook salmon after annual differences were accounted for (Figure 10). With years separated and runs combined, mean length of three-ocean fish were highest in 2003 (Figure 11). I speculate that two- and three-ocean Chinook salmon showed increased body size in 2003 due to greater ocean productivity.

Data from four-ocean fish were too sparse to allow meaningful statistical analysis (Table 6). It is worth noting that four-ocean Chinook salmon lengths are wholly within the three-ocean length groups and are not on the larger end of the three-ocean Chinook salmon length distribution curve (Figure 3).

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Table 1. Estimated smolt-to-adult return rates (SAR) of the aggregated Snake River wild spring/summer Chinook salmon stock.

Smolt Year	1996	1997	1998	1999	2000	2001	2002
# Smolts	419,826	161,157	599,159	1,560,298	1,344,382	500,700	1,173,566
Adults							
Smolt year +1	—	189	235	1,496	1,227	463	2,624
Smolt year +2	997	2,155	6,925	28,168	20,219	7,406	—
Smolt year +3	456	408	833	17,228	26,401	—	—
Smolt year +4	0	22	390	726	—	—	—
SAR w/ jacks	0.35%	1.72%	1.40%	3.05%	3.56%	1.57%	0.22%
SAR w/out jacks	0.35%	1.60%	1.36%	2.96%	3.47%	1.48%	—

Table 2. Estimated proportion of wild adult spring/summer Chinook salmon passing Lower Granite Dam during 2003 by 5 cm fork length group.

Fork Length (cm)	N	Proportion
<50	2	0.00
50-54	5	0.01
55-59	13	0.03
60-64	10	0.02
65-69	3	0.01
70-74	11	0.03
75-79	18	0.04
80-84	42	0.10
85-89	49	0.12
90-94	59	0.14
95-99	89	0.22
100-104	63	0.15
>104	45	0.11

Table 3. Estimated proportion of each ocean age by 5 cm fork length group of wild Chinook salmon carcasses collected on spawning grounds in 2003.

Fork Length (cm)	N	1-Ocean	2-Ocean	3-Ocean	4-Ocean
<50	7	1.00	0.00	0.00	0.00
50-54	6	1.00	0.00	0.00	0.00
55-59	2	1.00	0.00	0.00	0.00
60-64	1	1.00	0.00	0.00	0.00
65-69	8	0.14	0.86	0.00	0.00
70-74	22	0.00	0.91	0.09	0.00
75-79	62	0.00	0.90	0.10	0.00
80-84	51	0.00	0.81	0.19	0.00
85-89	91	0.00	0.24	0.76	0.00
90-94	136	0.00	0.07	0.92	0.01
95-99	136	0.00	0.00	0.96	0.04
100-104	83	0.00	0.00	0.98	0.02
>104	76	0.00	0.00	1.00	0.00

Table 4. Estimated proportion of wild Chinook salmon adults passing Lower Granite Dam for each ocean age group in each 5 cm fork length group, 2003.

Fork Length (cm)	N	1 Ocean	2 Ocean	3 Ocean	4 Ocean
<50	2	0.000	0.000	0.000	0.000
50-54	5	0.010	0.002	0.000	0.000
55-59	13	0.032	0.000	0.000	0.000
60-64	10	0.024	0.000	0.000	0.000
65-69	3	0.002	0.006	0.000	0.000
70-74	11	0.000	0.024	0.002	0.000
75-79	18	0.000	0.040	0.004	0.000
80-84	42	0.002	0.087	0.014	0.000
85-89	49	0.000	0.029	0.091	0.000
90-94	59	0.000	0.009	0.132	0.003
95-99	89	0.000	0.002	0.205	0.011
100-104	63	0.000	0.000	0.150	0.004
>104	45	0.000	0.000	0.109	0.001
Total	409	0.070	0.198	0.707	0.019
# Adults		2624	7406	26401	726

Table 5. Estimated ocean age proportion by 5 cm fork length groups for Snake River wild Chinook salmon carcasses sampled, 2003.

Fork Length (cm)	N	Spring and Summer Combined				Spring				Summer				
		One	Two	Three	Four	Ocean Years				One	Two	Three	Four	
						One	Two	Three	Four					
<50	7	1.00	0.00	0.00	0.00	1.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
50-54	6	1.00	0.00	0.00	0.00	1.00	0.00	0.00	0.00	1.00	0.00	0.00	0.00	0.00
55-59	2	1.00	0.00	0.00	0.00	1.00	0.00	0.00	0.00	1.00	0.00	0.00	0.00	0.00
60-64	1	1.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.00	0.00	0.00	0.00	0.00
65-69	7	0.14	0.86	0.00	0.00	0.00	1.00	0.00	0.00	1.00	0.00	0.00	0.00	0.00
70-74	22	0.00	0.91	0.09	0.00	0.00	0.95	0.05	0.00	0.00	0.67	0.33	0.00	0.00
75-79	62	0.00	0.90	0.10	0.00	0.00	0.92	0.08	0.00	0.00	0.86	0.14	0.00	0.00
80-84	52	0.00	0.81	0.19	0.00	0.00	0.73	0.27	0.00	0.00	0.91	0.09	0.00	0.00
85-89	90	0.00	0.24	0.74	0.00	0.00	0.21	0.74	0.05	0.00	0.31	0.69	0.00	0.00
90-94	137	0.00	0.07	0.92	0.01	0.00	0.06	0.94	0.00	0.00	0.08	0.90	0.03	0.00
95-99	136	0.00	0.00	0.96	0.04	0.00	0.00	0.98	0.02	0.00	0.00	0.89	0.11	0.00
100-104	83	0.00	0.00	0.98	0.02	0.00	0.00	1.00	0.00	0.00	0.00	0.97	0.03	0.00
>104	76	0.00	0.00	1.00	0.00	0.00	0.00	1.00	0.00	0.00	0.00	1.00	0.00	0.00

Table 6. Total number of Chinook sampled by run, age, and year.

Sample Year	1:1		1:2		1:3		1:4	
	Spring	Summer	Spring	Summer	Spring	Summer	Spring	Summer
1998	0	0	7	1	134	12	17	0
1999	15	13	68	128	18	24	3	3
2000	12	18	149	97	18	15	4	2
2001	3	11	502	307	39	35	3	0
2002	6	7	225	135	226	85	4	2
2003	11	6	80	75	334	165	6	4

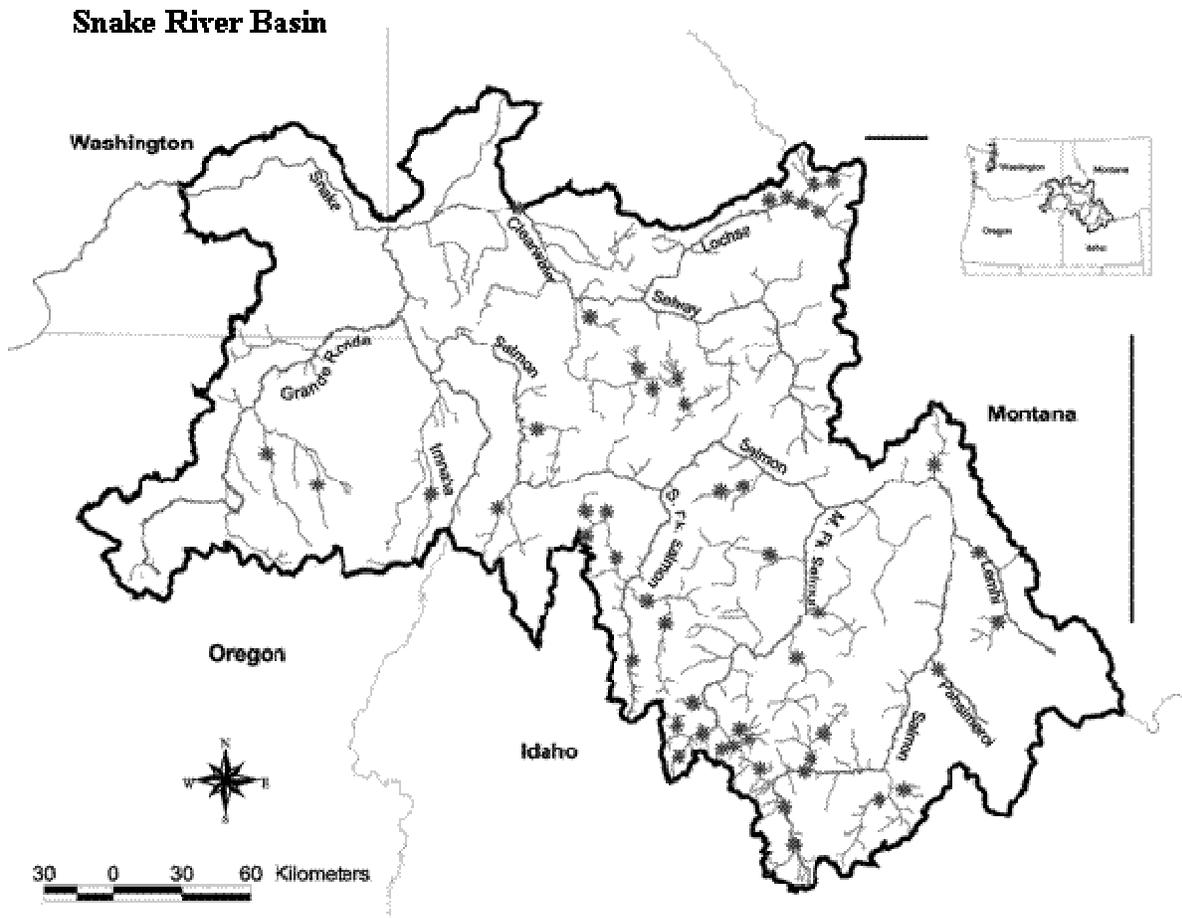


Figure 1. Spawning streams denoted with an * indicate where wild/natural spring/summer Chinook salmon adult carcass aging fins were collected in 2003.

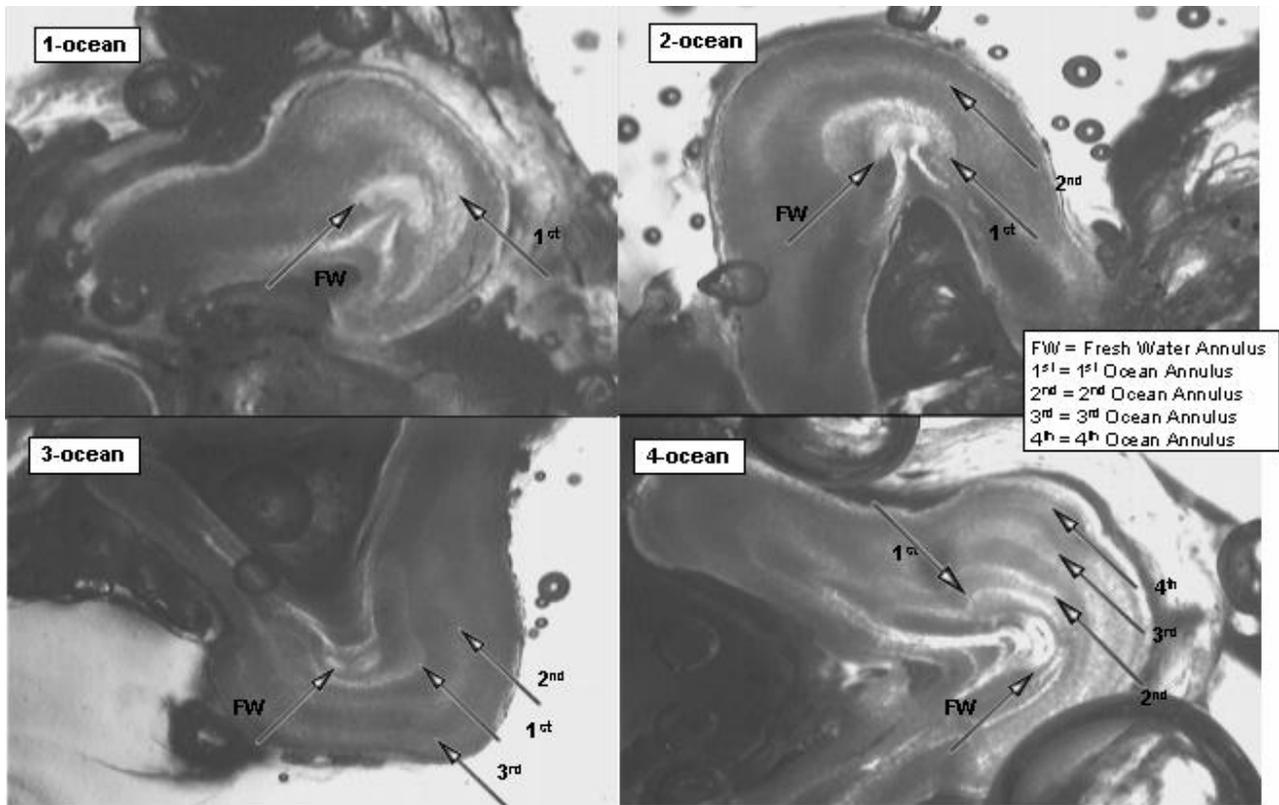


Figure 2. Representative dorsal fin ray cross sections from the four different ocean ages observed for Snake River wild/natural spring/summer Chinook salmon adult returns.

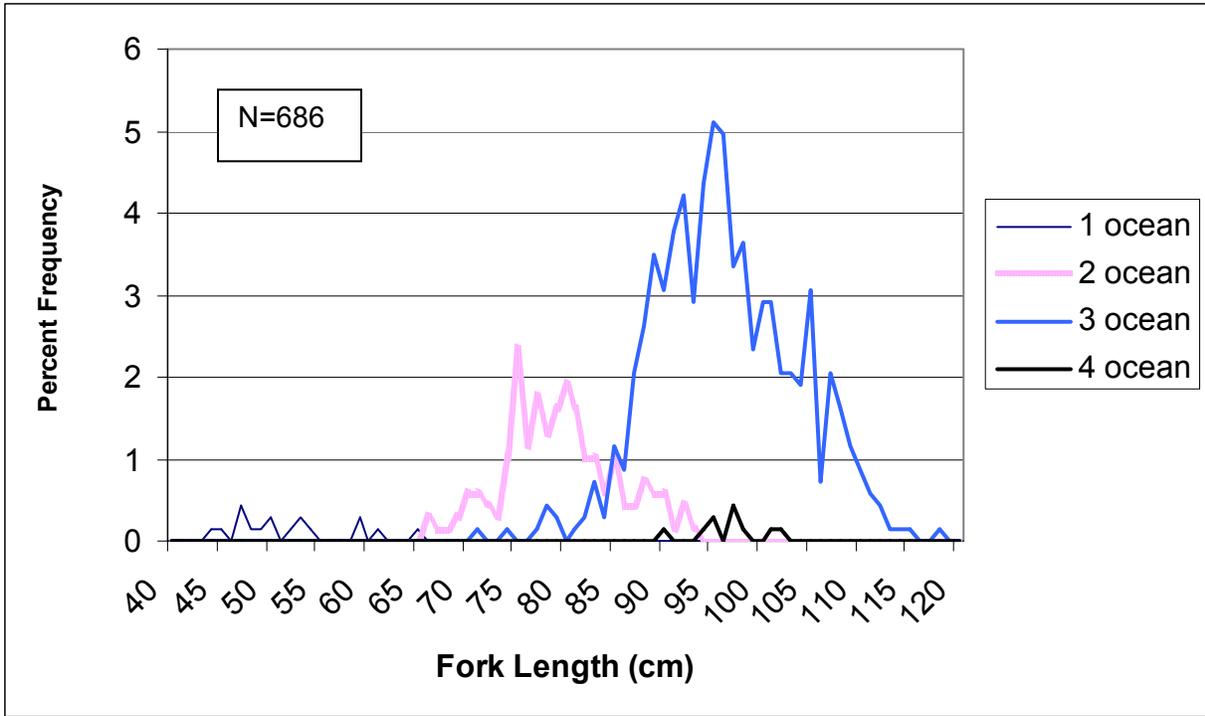


Figure 3. Wild Snake River Chinook salmon carcass fork lengths and ocean ages determined from fin cross sections, 2003.

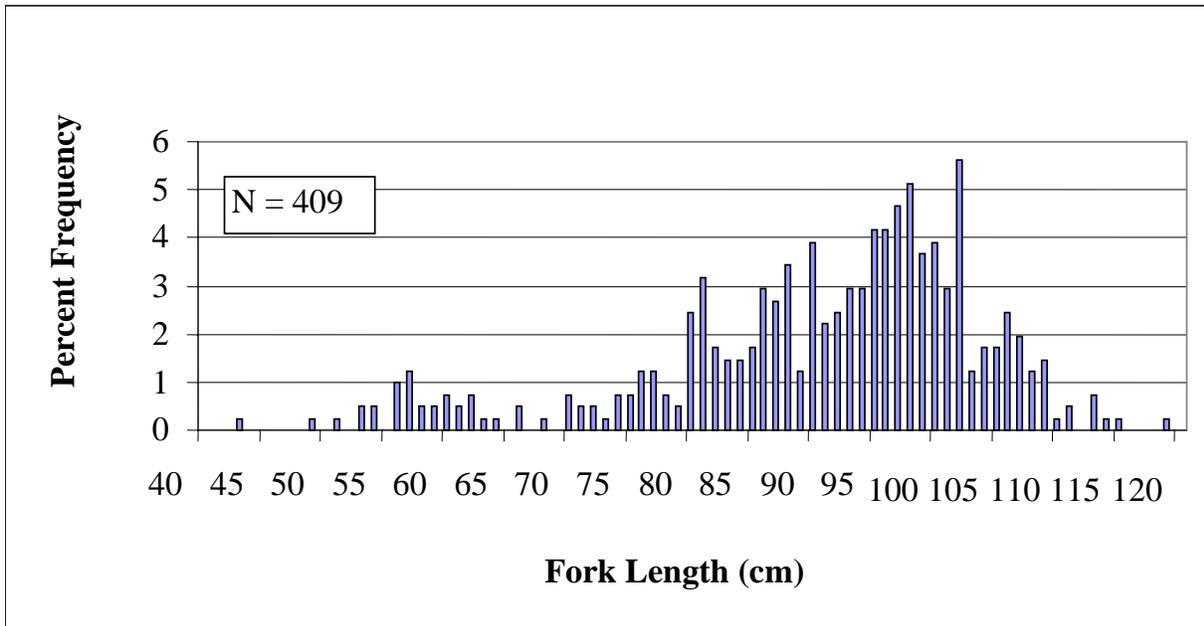


Figure 4. Fork length distribution of wild Snake River spring/summer Chinook crossing Lower Granite Dam, 2003.

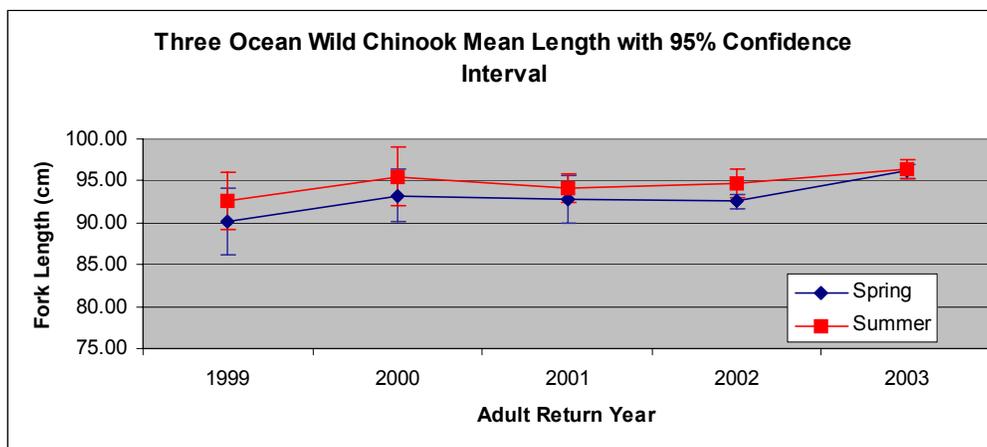
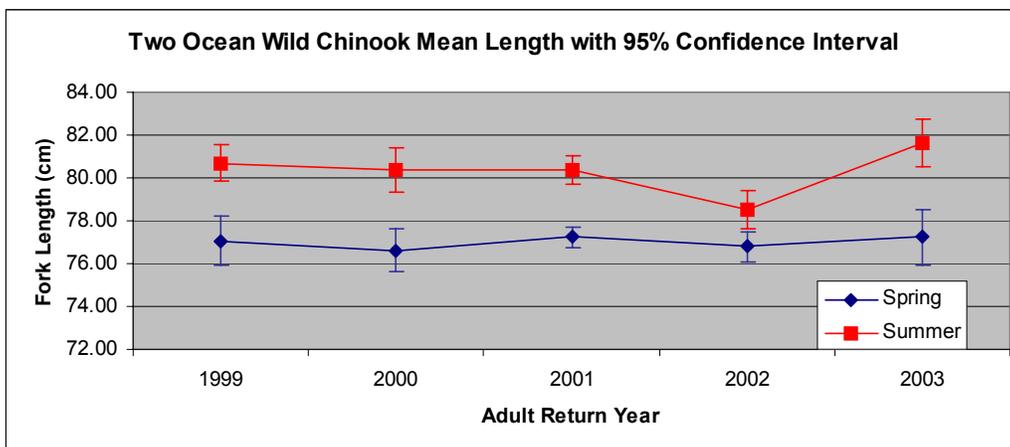
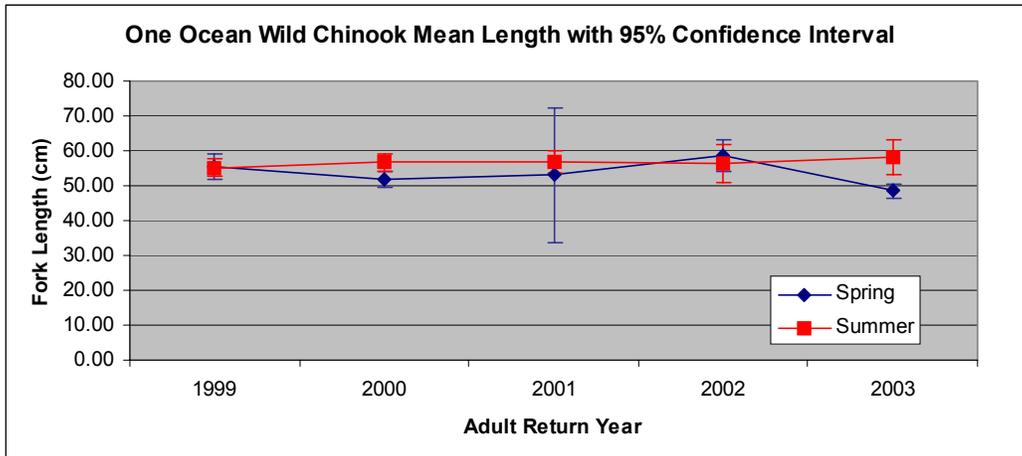


Figure 5. Wild Chinook mean lengths with 95% confidence intervals for each ocean age group, 1999-2003.

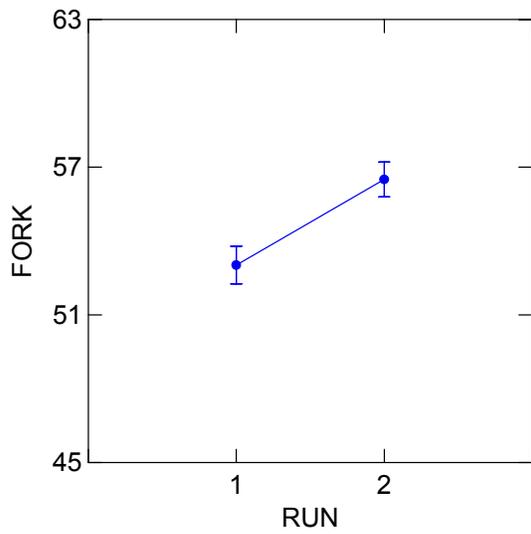


Figure 6. One-ocean Chinook salmon length by run, 1999-2003. Spring (1) vs. summer (2) run.

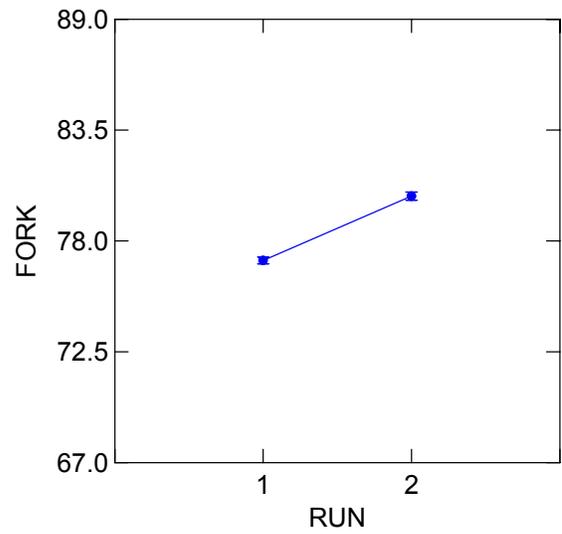


Figure 8. Two-ocean Chinook salmon length by run, 1999-2003. Spring (1) vs. summer (2) run.

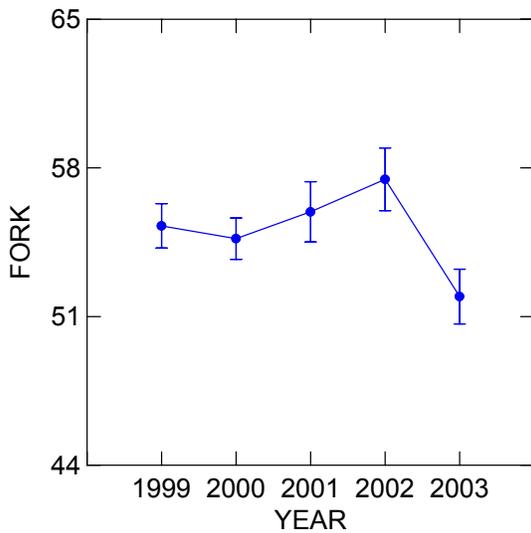


Figure 7. One-ocean Chinook salmon fork length during 1999-2003, runs combined.

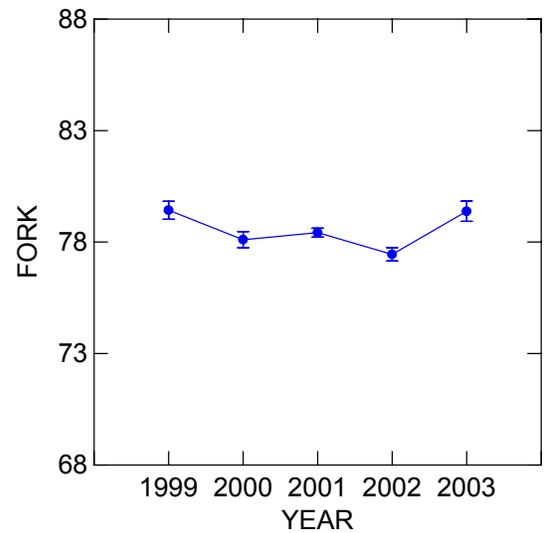


Figure 9. Two-ocean Chinook salmon length during 1999-2003, runs combined.

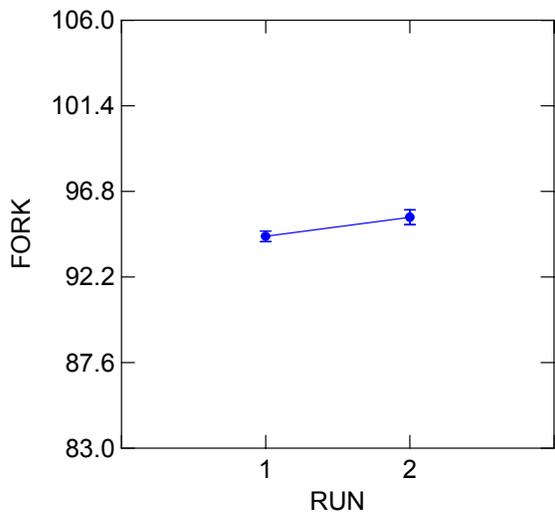


Figure 10. Three-ocean Chinook salmon fork length by run, 1999-2003. Spring (1) vs. summer (2) run.

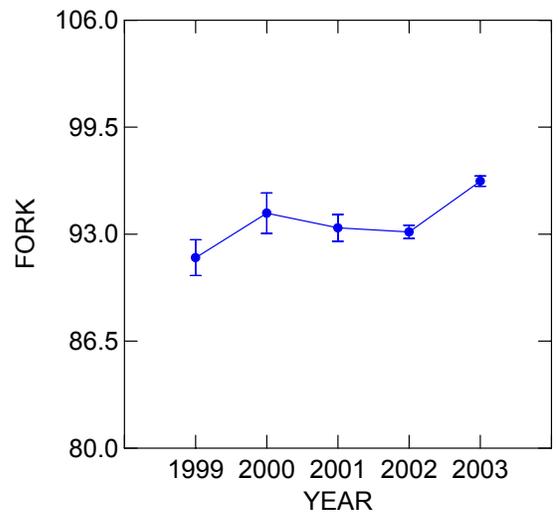


Figure 11. Three-ocean Chinook fork length by year, runs combined.

PART 2—DEVELOPMENT OF A STOCK-RECRUITMENT RELATIONSHIP FOR SNAKE RIVER SPRING/SUMMER CHINOOK SALMON TO FORECAST NATURAL SMOLT PRODUCTION

ABSTRACT

Stock-recruitment relationships are important to understanding how density-dependent forces affect abundance. My objectives were 1) to update a previously constructed Beverton-Holt (BH) curve for naturally spawning Snake River spring/summer Chinook salmon, 2) to use the model to predict future smolt production, and 3) to compare the BH model to a Ricker curve. In 2002 and 2003, numbers of females available for natural reproduction were 31,415 and 26,126, respectively. The 2003 smolt migration (brood year [BY] 2001) was 1,455,848. All model fits to BY 1990-2001 data were significant ($F > 25.0$, $P < 0.001$). The inverse BH model provided the most precise smolt production predictions. Neither the Ricker model nor its log transformation fit the data as well as the inverse BH model ($r^2 = 0.69-0.72$ versus 0.95). Maximum recent productivity (402.5 smolts/female) accords better with density-independent productivity predicted by the BH model (481.0 smolts/female) than by the Ricker model (295.9 smolts/female). Compared to historic data (1962-1974), the BH model suggests reduced system capacity, while the Ricker model does not. However, while the BH model has superior predictive ability at this scale, it is not possible to definitively corroborate or reject the Ricker model with these or similar data. The BH model is a reasonable alternative for examining freshwater Chinook salmon production in the Snake River basin. The model should be validated further by continued refinement and mechanistic investigations. Model choice has practical implications for management of density-dependent factors in Idaho.

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INTRODUCTION

The relationship between parental abundance and subsequent recruitment of progeny is the focus of a significant portion of fisheries research and management efforts. A stock-recruitment analysis describes the demographic ability of a population to sustain itself, assuming nondemographic factors remain constant. This analysis is typically an empirical process simplifying the many intervening stages by aggregating life history stages (Hilborn and Walters 1992). The goal is to produce a predictive model, which is a description of the observed pattern, i.e. the regularities of the system under consideration (Rigler 1982). A simple theoretical model is chosen to which data are fit, but the stock-recruit relationship usually shows extreme annual variability (Hall 1988). Underlying relationships are easiest to detect when stocks are very low or very high, because controlling mechanisms become evident at the extremes (Quinn and Deriso 1999).

Sources of variation in survival of Pacific salmon *Oncorhynchus sp.* are split between freshwater and saltwater phases in approximately equal magnitudes (Bradford 1995). For threatened Snake River spring/summer Chinook salmon *Oncorhynchus tshawytscha*, survival during both freshwater and saltwater life stages must be understood for decision makers to effectively select measures to promote recovery. An important tool to understand survival of anadromous fish is a stock-recruitment relationship spanning the critical period of freshwater residence when density-dependent mortality defines the shape of the relationship (Solomon 1985). Such an analysis will reflect the balance of good and suboptimal habitat in the basin (Crozier and Kennedy 1995).

Stock-recruitment relationships for Columbia River Basin Chinook salmon are assumed to be in the form of a Beverton-Holt (BH) function (NPPC 1986) or a Ricker function (Petrosky et al. 2001). In a BH function, the relationship is regulated by density-dependent mortality and is hyperbolic in shape, with the asymptote representing carrying capacity (Beverton and Holt 1957). In a Ricker function, some regulatory mechanism causes declines in recruitment at higher stock densities (Ricker 1954). In the past, most data sets produced very poor fits to stock-recruitment relationships (Hall 1988). The most serious problem in a stock-recruitment analysis is error in estimation of adult and recruit abundance (Hilborn and Walters 1992). The Columbia River hydrosystem presents an opportunity to estimate the stock-recruitment inputs using the efficient counting systems present at the dams in the system. Here, I updated the BH stock-recruit model of Kiefer et al. (2004) with data from the 2003 smolt migration, used the model to predict smolt production for brood years (BY) 2002 and 2003 based on estimated female escapement, and compared it to alternative formulations of the BH and Ricker functions.

METHODS

I derived an estimate of the number of spring/summer Chinook salmon females available for natural reproduction (FANR) upstream of Lower Granite Dam (LGD) by duplicating the procedure used previously (Kiefer et al. 2004). The estimated number of adults (excluding jacks) passing LGD in 2002 and 2003 was obtained from the annual reports of the Fish Passage Center (FPC 2003, 2004). I obtained the total of spring/summer Chinook salmon (excluding jacks) captured at all Snake River hatchery traps and the number of females taken into hatcheries from unpublished Idaho Department of Fish and Game (IDFG) hatchery reports, the Oregon Department of Fish & Wildlife (Pat Kinery, personal communication), and the U.S. Fish & Wildlife Service (Ralph Roseburg, personal communication). I computed the percentage

of females for all adults identified to sex at the weir by hatchery regional groups (Clearwater basin, McCall, Oregon, Pahsimeroi, Rapid River, Sawtooth). I also grouped females by run type (spring or summer) and mark (adipose and ventral clips versus unmarked). The total number of females taken for each hatchery (spawned, culled, or prespawning mortalities) was also noted. For each run type, the percentage of females, regardless of mark, was applied to the LGD counts to estimate the total number of female Chinook salmon passing LGD. The number of females taken by the hatcheries was adjusted for 20% migration mortality, i.e. by dividing the raw estimates by 0.8. I obtained the total harvest estimates upstream of LGD from the IDFG Bureau of Fisheries (Scott Marshall, personal communication). Female harvest was estimated by multiplying run-specific total harvest by run sex ratio and adjusted for 10% migration mortality based on telemetry studies (Chris Peery, University of Idaho, personal communication), i.e. by dividing the total estimate by 0.9. To compute FANR, the adjusted hatchery female number and the adjusted number of females harvested upstream of LGD were subtracted from the estimated number of females passing LGD. Spring and summer FANR estimates were combined to estimate total FANR.

Smolt production in 2003 was estimated using fish passage data collected at LGD. Passage data consisted of daily counts of wild smolts collected and estimated daily collection efficiencies. Daily smolt migration number was estimated by dividing the daily count by estimated collection efficiency for that day. I obtained the daily numbers of wild Chinook salmon smolts collected at LGD from the Fish Passage Center website (<http://www.fpc.org/smoltqueries/CurrentDailyData.asp>). The estimated daily smolt collection efficiencies at LGD were provided by the Northwest Fisheries Science Center (Steve Smith, personal communication). Daily estimates were summed for the year.

Abundance of the progeny may be predicted from parental abundance with a stock-recruit model. I assumed that the adult-to-smolt stock-recruitment relationship for Snake River spring/summer Chinook salmon would be in the form of a BH function (Beverton and Holt 1957). To develop a stock-recruitment relationship, I regressed FANR for brood years 1990-2001 against the associated smolt production using data from Kiefer et al. (2004) and the 2003 migration. I used the Beverton-Holt formula (Ricker 1975):

$$R = \frac{1}{\alpha + \beta / P}$$

where P = parent year spawning escapement (i.e. FANR),
 R = recruits (smolts) produced by parent year spawning escapement (P),
 α = a fitted parameter indicative of maximum reproductive rate for the population, and
 β = a fitted parameter indicative of compensatory mortality as a function of stock size.

Model parameters were estimated several ways. I fitted the model directly using iterative nonlinear regression and by using Paulik's transformation (Ricker 1975):

$$\frac{P}{R} = \beta + \alpha P$$

in which the dependent variable (P/R) is now a productivity rate.

The more traditional way to model productivity is by regressing parental abundance on the natural logarithm of smolts per female ($\ln(R/P)$, e.g., Petrosky et al. 2001). Because this

variation is usually used in the context of a Ricker stock-recruit relationship, I fitted this model, too. I duplicated the model fitting procedures for the Ricker model for the nonlinear (untransformed) model, $R = \alpha P e^{-\beta P}$, and the linear transformation, $\ln(R/P) = \ln \alpha - \beta P$.

Future smolt production for the 2002 and 2003 brood years was predicted directly from the BH equation and indirectly from the transformed model. For the latter, the model generates a females-per-smolt prediction by which I divided the estimated FANR to get an absolute number. The 90% prediction intervals for each year were estimated from:

$$\hat{y}_0 \pm t_{\alpha/2, n-2} \sqrt{MSE \left(1 + \frac{1}{n} + \frac{(x_0 - \bar{x})^2}{S_{xx}}\right)} \quad (\text{Montgomery and Peck 1992}),$$

where y_0 is the predicted value,
 x_0 is the 2002 and 2003 FANR,
 \bar{x} is the 1990-2001 mean FANR,
MSE is the residual error from the regression,
 n is calibration sample size (12), and
 S_{xx} is the sum of the squared deviations of the previously observed FANR values.

The estimated brood year 2002 and 2003 FANR values were applied to this regression to forecast a mean and 90% confidence interval of migratory year 2004 and 2005 smolt production.

RESULTS

The estimated number of adult spring and summer Chinook salmon crossing LGD, excluding jacks, was 97,184 in 2002 and 87,031 in 2003 (Table 7). Proportions of females were close to 50% except for the 2002 spring run (56.6%). Estimated losses above LGD totaled 22,616 in 2002 and 17,077 in 2003. Subtraction yielded FANR estimates of 31,415 in 2002 and 26,126 in 2003.

The estimated number of smolts exiting the system via LGD during Migratory Year (MY) 2003 was 1,455,848. This estimate covers the period March 26 to August 29, 2003. I assume these fish were produced by the 2001 spawners. Kiefer et al. (2004) previously estimated FANR in brood year 2001 at 51,902. Applying this value to the estimated level of recruitment gives 28 smolts per female for BY 2001.

The smolt estimate from MY 2003 completes the data for the 1990-2001 brood years. The untransformed (nonlinear) BH model fit the data very well (Figure 12; $F = 272.11$, $P < 0.001$, $r^2 = 0.667$). The Ricker function also fit well ($F = 119.80$, $P < 0.001$, $r^2 = 0.685$). The logarithmic transformation of the Ricker model was also highly significant ($F = 25.56$, $P < 0.001$, $r^2 = 0.719$), but the inverse BH model fit the data most precisely ($F = 180.28$, $P < 0.001$, $r^2 = 0.947$). The nonlinear and inverse BH models produced similar parameter estimates (Table 8).

The inverse BH model was used to predict the number of smolts that will be produced by the 2002 and 2003 brood years. The FANR estimates for 2002 and 2003 were inserted into the regression to predict the inverse of smolts/female expected (Figure 12). Back-transforming the predictions yielded 44.6 smolts/female (39.4–51.5 smolts/female, 90% prediction limits) for 2002

and 52.6 smolts/female (45.7–62.0 smolts/female, 90% prediction limits) for 2003. Applying the predicted productivity to the parental abundance gives a total predicted abundance of 1,401,109 (1,237,751–1,617,873, 90% prediction limits) for BY 2002 and 1,374,228 (1,193,958–1,619,812, 90% prediction limits) for BY 2003.

DISCUSSION

Estimated female escapements were similar in magnitude for 2002 and 2003 (Table 7). The number of adult returns in 2002 and 2003 were 60% and 50% of that in 2001, respectively, but still well above the 1990-2000 returns estimated by Kiefer et al. (2004). Smolt abundance for BY 2001 was less than that estimated for BY 1993 and 1997 by Kiefer et al. (2004) but was also above average for recent years.

Adult abundance was input into the model as a point estimate, but there is considerable uncertainty around each estimate. Use of dam counts to estimate escapement assumes that net upstream movement is accurately counted, and losses to in-river mortality and all other sources are accounted for (Dauble and Mueller 2000). These considerations can be problematic. I tried to duplicate procedures used in earlier reports, but attempts to repeat earlier analyses inevitably resulted in slightly different numbers. Differences resulted from updates of previous estimates provided to investigators as well as changes to estimation method, data updates (Russ Kiefer, IDFG, personal communication), and uncertainty in input variables. This uncertainty calls for scrutiny of FANR estimation procedures.

Female percentages were applied to the entire run assuming that weir counts were a random draw from the population or, alternatively, that weir sites were representative of all other sites. However, female percentages varied between years, among locations, and between marked hatchery fish (adipose clip) and unmarked fish, which presumably were mostly naturally-spawned individuals (Table 9). The effects of this variation have not been considered, yet estimates have been lumped by run timing. For instance, proportion of females in 2003 appeared positively correlated to the percentage of jacks that returned in 2002, although the relationship was different for marked and unmarked fish (Figure 13).

Other sources of uncertainty cannot be evaluated with the data in hand. Migration losses are estimated based on unpublished guesses. Migratory loss rates are assumed constant based on comments in Beamesderfer et al. (1997), yet those results also remain unpublished. Peery and Keefer (2003) speculated that a larger proportion reached spawning grounds in 2001 than typical of other years. Large errors are likely in harvest estimates, especially for tribal harvest, which was often disregarded in previous run reconstructions (e.g. Beamesderfer et al. 1997).

Perhaps more troubling is potential incompatibility of FANR estimates with the spawner abundances (males and females) reported by Petrosky et al. (2001). In general, the recent data (1990-1999) presented by Petrosky et al. were about twice the FANR estimates of Kiefer et al. (2004), as would be expected with an approximately 50% female ratio. However, Petrosky et al.'s estimate for BY 1997 was almost equal to Kiefer et al.'s estimated FANR. In addition, Petrosky et al. (2001) estimated past peak spawner abundance (males and females) near 53,000 in 1969 using methods similar to those in this report. The 2001 FANR estimated by Kiefer et al. (2004) was 51,902. Given that there may be serious discrepancies associated with single data points, I caution about the use of the 2001 FANR to represent upper capacity of the system. Unfortunately, this is the only data point in that region of parameter space, and it had a

high influence on model fitting. Dauble and Mueller (2000) speculate that overestimation of escapement is likely in the Columbia hydrosystem. Inputs into the 1997 and 2001 FANR estimates need to be scrutinized further.

In contrast to the adult estimates, it is possible to get very precise, presumably accurate, smolt estimates because of the counting systems installed at LGD. The method employed here involves summation of corrected counts over the migratory season. As long as there is no consistent bias in the way the system is operated, errors should cancel out over the season. Previous smolt abundance estimates were consistent with those reported by Petrosky et al. (2001), theirs being approximately $\pm 10\%$ except for BY 1997. Compared to Petrosky et al. (2001), BY 1997 abundances were high for both smolts and adults. Direct comparisons of productivity between the two studies are complicated because Petrosky et al. (2001) included all adults, whereas this research considered only females. However, a rough comparison of the present data to Petrosky et al.'s data, plotted similarly to Figure 12A, revealed a comforting correspondence in general. The most aberrant data point was 1997, but it was closer than the point estimated by Kiefer et al. (2004) to the predicted trend (data not shown). It would be instructive to see how the aggregated data presented here and in Kiefer et al. (2004) track the trends from the subpopulations studied by the Idaho Supplementation Studies (ISS).

Stock-recruit models are used in the literature often, but few authors explicitly evaluate model performance. Despite concerns about data quality, the BH model fit the data very well. Surprisingly, all fits were sufficient for predictive resolution, ($r^2 > 0.65$), which increases greatly at $r^2 > 0.90$ (Prairie 1996). Using Snake River data, Deriso et al. (2001) found that stock-recruit models were insensitive to errors in spawner measurements. Hilborn and Walters (1992) recommend against using the linear transformation of the BH curve, primarily because the implicit error structure is not lognormal. However, Bradford et al. (1997) found that variance in smolt production is not always lognormal. Regardless, the linear BH transformation fit the data very well and provided precise predictions. Nonlinear fitting did not allow an easy way to calculate confidence intervals. If such are desirable in the future, iterative methods such as jack-knifing or bootstrapping will have to be used.

Other published stock-recruit models for Snake River data use Ricker curves (e.g., Schaller et al. 1999; Petrosky et al. 2001). Kiefer et al. (2004) chose to use a BH model but did not compare fit to that of the Ricker curve. Schaller et al. (1999) stated that the Ricker model fit their baseline data (pre-1970) better than the BH model but did not provide an evaluation of model fits. Much of the acceptance of the Ricker model seems to stem from the implicit incorporation of lognormal error structure in its linearization (Hilborn and Walters 1992). Both versions of the Ricker model fit the data very well (Table 8), although not as well as the linearized Beverton-Holt model. The linearized forms of the Ricker and Beverton-Holt models can be viewed as alternative transformations of a productivity model, i.e. the logarithmic and inverse (or reciprocal) transformations. The more powerful inverse transformation was better at linearizing the data (Figure 12, panels B and C).

Models should be validated in relation to purpose, context, and performance (Rykiel 1996). Smolt forecasts offer a means to evaluate past model performance and the relative usefulness of the current models. Based on the data presented here and in Kiefer et al. (2004), the past predictions of the BH model (Kiefer et al. 2001, 2002, 2004) have performed well, even though BY 2001 represents a large extrapolation (Table 10). On the other hand, the Ricker model does rather poorly at prediction. The linearized Ricker model fit the 1990-2000 data well ($F = 11.67$, $P = 0.008$, $r^2 = 0.565$) but its prediction of the 2001 data point was extremely low and imprecise (1.21 smolts/female, 90% prediction interval 0.16-8.99). Other points of reference

are the predictions for smolt production by BY 2002 and BY 2003. Predictions by the linearized Beverton-Holt model are much more precise than the linearized Ricker model (Table 11). The model predictions differed most at FANR values of 15,000 to 40,000. Smolt returns from BY 2002 and BY 2003 will provide a good test of the BH predictions. However, the fitted Ricker model is so imprecise that it will not be possible to conclusively test model performance. Perhaps a Bayesian perspective can provide a posterior probability for both models. The prediction intervals overlap too broadly for traditional frequentist statistical techniques to provide definitive information.

The usefulness and interpretation of stock-recruit curves depend on the assumptions implicit in the models. Several assumptions are common to both Ricker and BH models (see Hilborn and Walters 1992). Both assume density-dependence during the pre-recruit life stages. This assumption is supported by the 2001 data. However, the form and timing of density-dependent mortality underlying each model are different. Another key assumption is that the relationship is at equilibrium and, therefore, the curve is stationary. Although this is unlikely over a long period of time, it should be close enough for the time period considered here (1990 to present). Most major alterations in the habitat of major production areas took place prior to 1970 (Petrosky et al. 2001). Given that likely errors in the data should not greatly affect model fitting and the precision of the fit, I believe the observed relationships will have great utility. However, recruitment is greatly influenced by stochastic processes. Average predictions, such as made here, should be approached with caution (Hilborn and Walters 1992). Future emphasis should be placed on understanding the factors that affect data variability and curve shape.

Model parameters for both Ricker and BH models have similar interpretations regarding maximum productivity and maximum production. The BH models in Table 9 indicate a maximum productivity of 351.7 to 481.0 smolts/female. The Ricker models in Table 12 estimate maximum productivity of 213.4 to 269.1 smolts/female. Correcting for underestimation by linear fitting increased the productivity estimate to 295.9 smolts/female. Because of greater precision of the inverse linear fit, a similar correction had little effect on the BH estimate. System capacity predicted by the BH models varied from 1,575,000 to 1,842,000 smolts. Perhaps the increasing number of naturally spawning hatchery fish seen in the last decade has reduced the number of smolts that can be produced in aggregate. Breeding success of hatchery females can be less than that of naturally produced fish (Fleming and Gross 1993), so the proportion of naturally spawning hatchery fish may lower the apparent system productivity. Alternatively, years of low escapements may have decreased system productivity via nutrient shortage (Achord et al. 2003). Smolt capacity for the Ricker models ranged from 1,988,000 for the nonlinear fit to 2,306,000 for the corrected linear fit (Figure 14). The adjusted Ricker capacity accords much better with smolt production in the 1960s as estimated by Raymond (1979). On the other hand, recent observed productivity has ranged up to 402.5 smolts/female in 1994, which accords better with the BH predictions. Because the effectiveness of habitat restoration programs, such as are currently underway in Idaho (e.g., ISCC 1995), depends upon the mechanism of population density dependence (Greene and Beechie 2004), it is important to distinguish between models. The scale of density dependence can have a large effect on the success of any habitat restoration/mitigation program.

RECOMMENDATIONS

The Beverton-Holt curve is a viable model to use to study freshwater production by the Snake River spring/summer Chinook salmon stock. The utility of the data presented here and in

Kiefer et al. (2004) will be increased as more data are added and model structure becomes more precise. Several steps can be taken to critically examine model assumptions. Hilborn and Walters (1992) recommend assessment of input variances, comparison of alternative models (such as begun here), examination of model residuals for nonstationarity, and simulation exercises. In addition, historic data can be used to corroborate model shape, and bootstrapping can be used to estimate precision of nonlinear fitting. Particular attention should be given to the effect of naturally spawning hatchery fish on system capacity. Departures from model predictions may be associated with a large percentage of naturally spawning hatchery females. Lastly, data generated by ISS can be used to examine timing of density dependence, thus providing independent data to distinguish among models. If the BH model is supported, the practical implication is that habitat management in Idaho should consider all potential presmolt rearing areas, from the spawning grounds downstream. Knowledge of habitat use or suitability in many of these areas is currently lacking.

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Table 7. Estimated returns to Lower Granite Dam, percentage of females, losses to hatcheries and harvest, escapement, and females available for natural reproduction (FANR). Hatchery take was adjusted by 20% and harvest by 10% to account for migratory losses.

Estimate	Year and run type			
	2002		2003	
	Spring	Summer	Spring	Summer
Dam count	75,025	22,159	70,609	16,422
% females	56.6	52.2	49.0	52.4
Females (#)	42,464	11,567	34,598	8605
Hatchery	6095	1556	4991	1769
Harvest	10,056	4909	6298	4019
Escapement	26,313	5102	23,309	2817
Total FANR	31,415		26,126	

Table 8. Comparison of estimates of the parameters of the Beverton-Holt (BH) and Ricker curves (α , β) from nonlinear and linear models fit with 1990-2001 data and parameters estimated by Kiefer et al. (2004) using 1990-2000 data. The 95% confidence limits are in parentheses. Parametric confidence intervals were not estimable in the nonlinear models (na). The β parameters are not comparable between BH and Ricker models.

Model	α	β
Nonlinear BH	481.0 (na)	1,658,375 (na)
Inverse BH	411.8 (242.7–1344.1)	1,574,803 (1,351,351–1,890,359)
Kiefer et al. (BH)	351.7 (183.7–4115.2)	1,841,621 (1,020,408–9,433,962)
Nonlinear Ricker	213.4 (na)	0.0000395 (na)
Linear Ricker	269.1 (192.8–375.5)	0.0000471 (0.0000264–0.0000680)

Table 9. Proportions of females (%) at hatchery weirs by year, location, and mark (adipose clip versus unmarked).

Location	2002		2003	
	Adipose Clip	Unmarked	Adipose Clip	Unmarked
Clearwater	59.1	45.1	50.5	56.2
Little Salmon	58.1	50.7	44.1	46.2
Upper Salmon	64.7	41.8	58.6	57.9
Oregon	56.2	48.9	51.5	51.1
South Fork Salmon	53.3	47.1	48.7	54.2
Pahsimeroi	62.3	46.2	57.0	62.4

Table 10. Comparison of smolt abundance predicted by the Beverton-Holt model versus observed abundance. Differences were converted to percentages by division with observed values.

Prediction Source	Brood Year	Predicted	Observed	% difference
Kiefer et al. (2001)	1999	478,200	490,534	2.5
Kiefer et al. (2002)	2000	1,196,421	1,128,549	6.0
Kiefer et al. (2004)	2001	1,696,787	1,455,848	16.5

Table 11. Predictions and 90% prediction intervals of the linearized Beverton-Holt and linearized Ricker models for brood years (BY) 2002 and 2003. Model predictions are given in both relative (smolts/female) and absolute abundance. Prediction intervals (90%) are in parentheses.

Model	BY	Smolts/female	Smolts
Beverton-Holt	2002	44.6	1,401,109
		(37.0—56.2)	(1,162,355—1,765,523)
Beverton-Holt	2003	52.6	1,374,228
		(42.7—68.6)	(1,115,580—1,792,244)
Ricker	2002	67.2	2,111,978
		(27.1—166.5)	(852,743—5,230,712)
Ricker	2003	86.3	2,254,095
		(36.1—206.5)	(941,720—5,395,387)

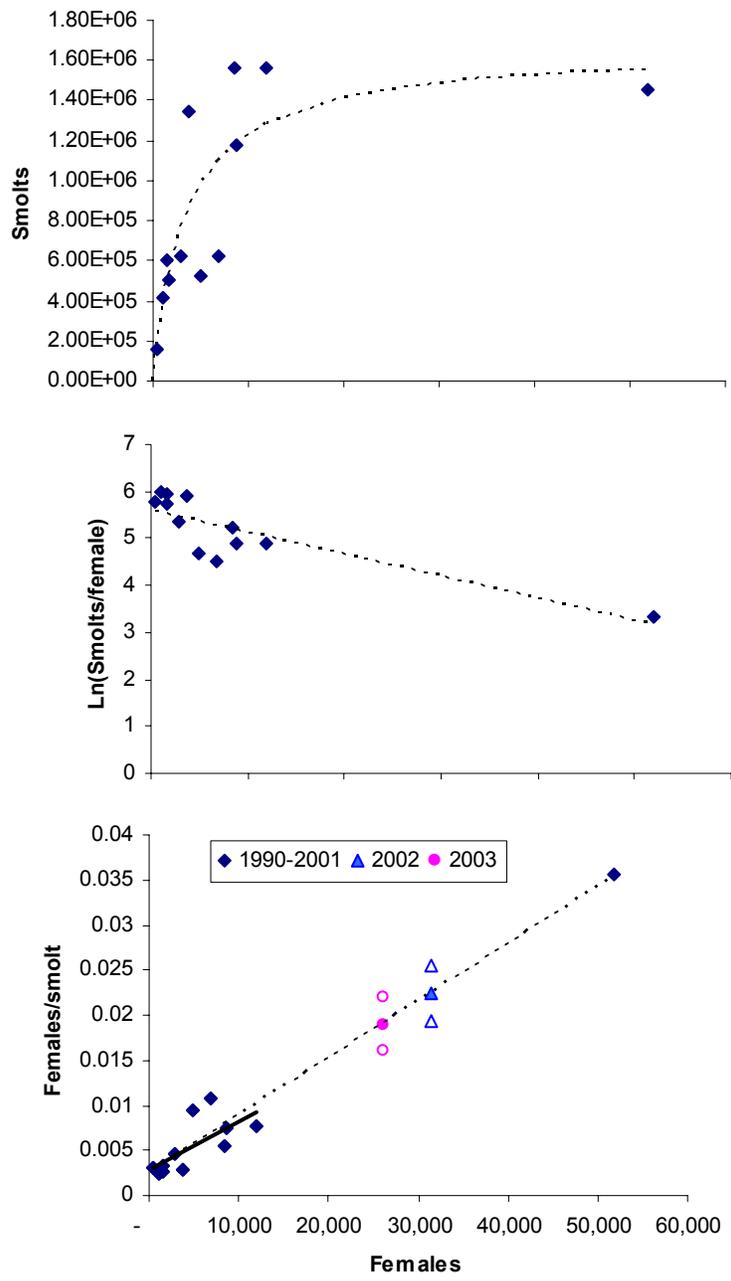


Figure 12. Comparison of observed data to model predictions for the nonlinear Beverton-Holt (top panel), logarithmic Ricker (middle panel), and inverse Beverton-Holt models (bottom panel). Model predictions are shown as a dotted line. The solid line in Panel C represents the regression for 1990-2000 (Kiefer et al. 2004). Predicted smolts per female for brood years 2002 and 2003 are plotted in the bottom panel along with accompanying 90% prediction limits (hollow symbols).

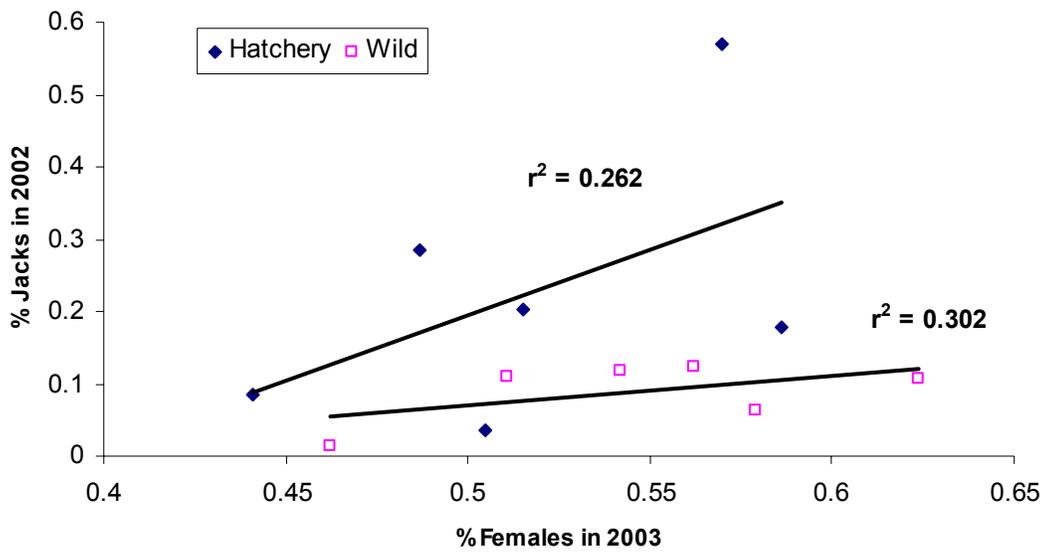


Figure 13. Relationship of proportion of females at hatchery weirs in 2003 to proportion of jacks in 2002.

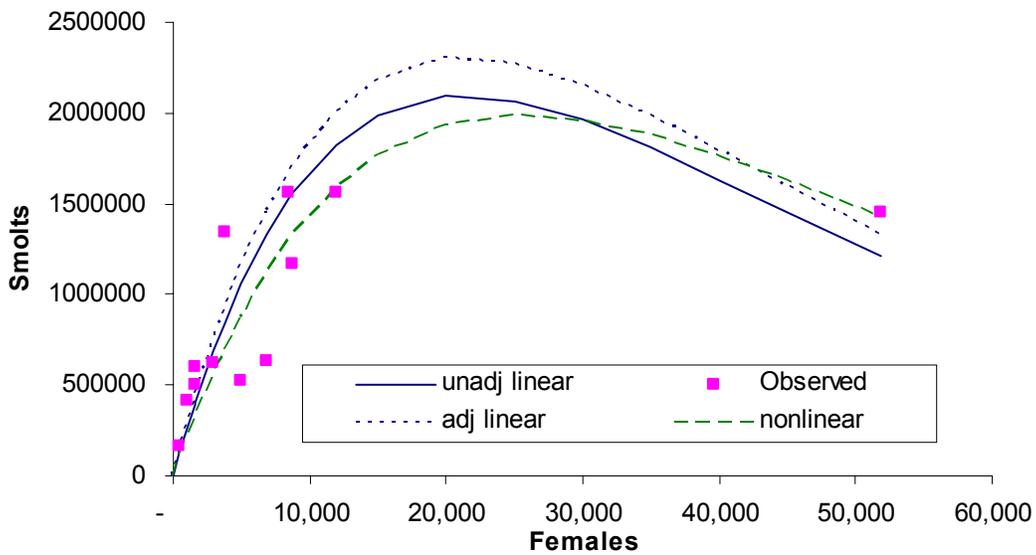


Figure 14. Model predictions for the Ricker model. Predictions are for the nonlinear fit, the unadjusted linear transformation, and the adjusted linear transformation. Observed data are plotted for reference.

PART 3—IMPROVE THE PRECISION OF SMOLT-TO-ADULT SURVIVAL RATE ESTIMATES FOR WILD STEELHEAD TROUT BY PIT TAGGING ADDITIONAL JUVENILES

ABSTRACT

Passive integrated transponder (PIT) tags are used to study the survival of anadromous salmonids in the Columbia River hydrosystem. The goal of this research is to increase the precision of survival estimates by placing additional PIT tags in Idaho's wilderness, using angling as a collection method. We tagged 7040 steelhead *Oncorhynchus mykiss* parr in 2002 and 7819 in 2003. Detection rates at Lower Granite Dam (LGD) in 2003 ranged from 6.0% to 21.8%, depending on population. Migration period was similar among populations but median arrival date differed up to a month. We placed 14%-35% of the annual PIT tag output in the Snake River basin during 1999-2003. However, fish tagged by this project were half as likely to be detected at LGD as the average for all methods. Addition of data from this project did not increase the precision of basin-wide smolt-to-adult return (SAR) rates. The SAR estimates of angled fish appear higher than estimates of fish collected by other methods. We recommend future angling efforts be scaled back unless the objective is to increase the representation of wilderness areas in hydrosystem survival studies or to study populations in rugged terrain. Sources of methodological bias in SAR estimates should be evaluated.

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INTRODUCTION

Much research on protected anadromous salmonid stocks in the Columbia River basin has focused on downstream juvenile survival and smolt-to-adult return (SAR) rates (e.g., Raymond 1988). To facilitate this research, the National Marine Fisheries Service developed the passive integrated transponder (PIT) tag during the 1980s (Prentice et al. 1990a). These tags allow the individual identification of tagged fish as they migrate past the dams of the Columbia River hydrosystem without subsequent handling (Prentice et al. 1990b). Using this information, researchers can compute valid and reliable survival estimates for both juveniles (Skalski et al. 1998) and adults (Newman 1997).

Fish passage downstream through a dam in the Columbia hydrosystem can be accomplished by several routes with differing hazards (Muir et al. 2001). To properly assess these hazards, survival estimates must be calculated separately for each route. Subdividing detections by route has reduced sample size, thus decreasing the precision and usefulness of subsequent survival estimates. Hence, there is a need to increase the number of PIT-tagged smolts, especially for wild steelhead trout *Oncorhynchus mykiss* (Newman 1997).

One of the anadromous salmonid stocks of concern is the Snake River summer-run steelhead. There is very little specific information about the populations that comprise the Snake River stock (ICBTRT 2003). In Idaho, many of the spawning and rearing stream reaches for wild steelhead are in rugged and remote wilderness areas that must be sampled with light and mobile gear such as electrofishing or angling. Most headwater streams in Idaho are very infertile and nonconductive, rendering electrofishing ineffective (Petrosky and Holubetz 1987). Previous work by project personnel has shown that hook-and-line sampling with artificial flies can be effective at collecting juvenile steelhead (Russ Kiefer, Idaho Department of Fish and Game, personal communication).

In 1998, the Idaho Natural Production Monitoring and Evaluation Project (NPM) initiated efforts to increase the number of PIT-tagged steelhead smolts available to estimate SARs and downstream smolt survival. Efforts were directed at important production streams not sampled by other research efforts. Additionally, tagging in streams not sampled by other projects made the overall sample more representative of the entire Snake River wild steelhead stock. Our objectives in this report were two-fold: 1) to summarize the results of fieldwork in 2002 and 2003, and 2) to assess preliminarily the effectiveness of NPM efforts in increasing precision of survival estimates. These objectives should lay the groundwork for a more comprehensive program assessment.

METHODS

The study area included streams in the Salmon and Clearwater basins. Sampling was concentrated in important wild steelhead production areas: Middle Fork Clearwater River basin, Middle Fork Salmon River basin, Salmon River Canyon tributaries, and Salmon River tributaries downstream from the Salmon River Canyon (Figure 15). All streams were believed to have minimal hatchery influence. Scouting and snorkeling of streams occurred prior to the tagging field season. Streams sampled offered the best combination of access, presumed wild steelhead trout densities (age-2 and older juveniles), and stream size to permit efficient collection by angling. Fourteen streams were sampled for this report, thirteen in 2002 and eight in 2003; not all streams were sampled in both years (Table 12).

Wild steelhead trout juveniles were captured by angling with artificial flies from June through August 2002 and 2003. Each angler carried a five-gallon bucket half filled with water to store captured fish temporarily while fishing. Water in the bucket was changed at least every 15-20 min when <10 fish were in the bucket and about every 10 min when >10 fish were in the bucket. Anglers transferred fish from buckets to submerged perforated plastic live-boxes (1.0 m x 0.5 m x 0.7 m) placed at approximately 1 km intervals throughout the stream.

We held captured fish in live-boxes overnight and tagged them the following morning. The delay allowed the fish to recover from collection stress and provided the coolest water temperatures for tagging. We anesthetized the fish with MS-222 or clove oil, and injected PIT tags into the body cavity using a 12-gauge hypodermic needle and modified syringe. Needles and PIT tags were sterilized by soaking them in a 70% alcohol solution for at least 10 min before tagging. Wild steelhead trout 65-249 mm fork length (FL) were tagged, while all others were released. Steelhead <65 mm FL were too small to tag. Wild steelhead trout >250 mm FL were most likely nonmigratory resident fish (Partridge 1985). After tagging, fish were returned to a live-box and allowed to recover at least 1 h before release. At the completion of fieldwork, project personnel uploaded PIT tag data to the Columbia River Basin PIT Tag Information System (PTAGIS).

Unique detection records from adult and juvenile detector sites in the Columbia River hydrosystem were obtained from the PTAGIS database (<http://www.psmfc.org/pittag/>) in March 2004. The PTAGIS reports provided information on tagging dates, capture method, FL at release, release site, interrogation site, and date detected. To maintain continuity with past reports, interrogation reports from the four main smolt collection facilities (Lower Granite [LGD], Little Goose, Lower Monumental and McNary dams) were used to determine detection rates for migratory year (MY) 2003. For the preliminary program evaluation (Objective 2), we obtained data from all tagging of wild steelhead done upstream of LGD and detections anywhere in the Columbia hydrosystem.

To summarize fieldwork in 2002 and 2003, we reported numbers tagged by stream. For fish tagged in 2002, the detection rate for each stream was calculated by dividing the number detected by the total number of fish tagged in that stream. Few fish tagged in 2003 (MY 2004) had migrated by the time we queried PTAGIS, so we present detection rates for MY 2003 only. Because smaller juvenile steelhead trout often rear another year before smolting (Kiefer and Lockhart 1997), we calculated detection rates for juveniles ≥ 125 mm FL at tagging as well as a total rate. We also presented data on population-specific arrival times at LGD during 2003. For each of these streams, the dates were calculated when the 10th, 50th, and 90th percentile of total PIT tag detections occurred.

Our evaluation of the NPM steelhead tagging program focused on juvenile detection rates and smolt-to-adult return (SAR) rates for the 1999-2002 migratory years. We compared tagging efforts by NPM personnel (all sites combined) versus all other tagging operations directed at wild steelhead upstream of LGD. Detection rates were estimated as above but based on anticipated MY for comparability. Detection rate was adjusted for smolts that migrated after anticipated MY. Smolt-to-adult return rates were estimated for each MY by dividing the number of adults detected from that MY by the total number of unique juvenile detections from anywhere in the hydrosystem, including detectors downstream of McNary Dam. Adult returns from smolts not detected were not included. All SAR estimates were based on year of smolt detection, i.e. migratory year. To evaluate the NPM contribution to the precision of the basinwide SAR estimates, we calculated two basinwide SAR estimates, with and without NPM data. Confidence

intervals (CIs) were computed based on the binomial distribution with a finite-population correction (Zar 1999) with $\alpha = 0.10$. The correction factor was based on number of individuals tagged for that MY. Our measure of precision was the width of the 90% CI.

RESULTS

The NPM crew conducted tagging operations during the periods June 28-August 20, 2002 and June 27-August 12, 2003. A total of 7040 parr were tagged in 2002, and 7819 were tagged in 2003. Of the parr tagged in 2002, 892 were detected as smolts in 2003. Detection rates in MY 2003 varied from 6.0% to 21.8%, depending on population (Table 13). Median detection rate for MY 2003 was 14.3%. If parr <125 mm FL are excluded, detection rates increased by approximately 1.9%.

Arrival dates at LGD had a bimodal distribution in 2003 (Figure 16). Migration reached the first peak on April 18. Only 13 detections were recorded prior to April 15. The final peak was on May 25, after which emigrant numbers quickly dropped. Only 11 tagged individuals were detected after May 31. Median date of arrival varied among populations (Figure 17). Dates of the median arrival date ranged from April 20 for Moose Creek to May 20 for Loon Creek. The 10th percentile dates ranged from April 16–27, and 90th percentile dates ranged from May 17-27.

For wild juvenile steelhead trout in the Snake River basin, NPM personnel have done the majority of PIT tagging using angling as a capture method for MY 1999-2003 (Table 14). Project output represents 14%–35% of the PIT tags placed in the basin by any method. Of the wild smolts detected at LGD during MY 1999-2003, 7%-14% were tagged by NPM personnel. The detection rate of NPM-tagged fish was about half the average of all methods.

Smolt-to-adult return rates for wild steelhead detected MY 1999–2002 varied from 0.238% to 2.411% (Table 15). Exclusion of NPM-tagged individuals from these calculations lower SAR estimates consistently but not significantly. Including NPM-tagged fish actually increased the widths of the 90% confidence intervals in most years, thus decreasing precision slightly (Table 15). However, the actual absolute effect size was very small (<0.02%).

DISCUSSION

Detection rates varied among populations and years. Fish tagged in Whitebird Creek had the lowest detection rate, while individuals tagged in Camas Creek had the highest detection rate. However, detection rates can vary over short distances. Although Yellowjacket Creek is contiguous with the section of Camas Creek sampled, detection rates were almost as low as in Slate and Whitebird Creeks. Thus, there are likely local differences in parr survival or incidence of the resident life history. Regional differences may also exist. Byrne (2003) suggested that detection rates in the Clearwater basin are generally higher than from the Salmon basin. Differences among streams appear to be temporally stable; detection rates from Whitebird and Slate Creek have been the lowest for each year (Kiefer et al. 2001, 2002, 2004). Detection rates also vary with each migratory year. The highest overall detection rates (FL >125 mm) were observed during MY 2001 (24%, Kiefer et al. 2002) and lowest in MY 2002 (14%, Kiefer et al. 2004). However, annual differences should be interpreted cautiously because only Bargamin, Chamberlain, and Slate Creeks have been sampled every year (1999-2003).

Commencement and cessation of migration seemed relatively synchronous among the populations sampled, but median arrival dates at LGD differed among populations. Kiefer et al. (2002) proposed that differences in median arrival dates were inversely related to elevation of the tagging sites. This speculation was not supported by the current data; migration was later at both the higher elevation sites in the MFSR basin and the lower elevation sites (Whitebird and Slate Creeks). Achord et al. (1996) speculated that migration timing of wild Chinook salmon *O. tshawytscha* in the study area was controlled by environmental conditions. In this context, elevation may be important as it influences prevailing local weather conditions. However, as with detection rates, there may be regional effects as well. Byrne (2001, 2002) found that median arrival date at LGD was generally earlier for Clearwater basin populations compared to Salmon basin populations. It is difficult to evaluate the importance of regional or local factors on detection rate or arrival timing, because most of the sites we sampled were in the Salmon basin, and stream selection has not been consistent among years.

Migration timing past LGD was bimodal with early and late peaks. This overall pattern has been consistent among years, although the date of the modes changes. The modes are typically 3-4 weeks apart. This pattern is consistent with Achord et al.'s (1996) observations of spring Chinook smolt migration timing. The 2003 pattern was slightly different than observed in past years. Kiefer et al. (2002, 2004) found that hatchery smolts predominate in the later mode and more wild fish migrate in the earlier mode. In 2003, the second mode of wild fish was slightly greater than the first mode. However, we did not examine hatchery fish, so the present results should be compared to Kiefer et al.'s (2002, 2004) results cautiously.

The goal of the NPM steelhead PIT tagging program was to increase the precision of SAR estimates. On the coarsest (basinwide) scale, inclusion of NPM-tagged fish did not increase SAR precision; instead, variation increased slightly in three of four years. That is because SAR estimates from NPM data are consistently higher than estimates from all other data (Figure 18). This bias appears to be methodological; the SAR estimate of fish collected by angling over all years was higher than for other collection methods (Table 16). We concluded that differences in collection methodologies offset gains in precision caused by increased sample size. This conclusion depends on assumptions concerning comparability of the methods, e.g., migration timing and tagging locations and, therefore, should be evaluated more closely.

Angling as a collection tool has both benefits and disadvantages. Properly trained crews can collect fish in streams that are logistically difficult to sample with more efficient, passive methods, e.g. screw traps. Seines are difficult to use in rugged stream channels, where obstacles and currents impair efficiency (Hayes et al. 1996). Electrofishing is inefficient in the study area (Petrosky and Holubetz 1987) and injurious to salmonids in low-conductivity waters (Hollender and Carline 1994). Angling is a flexible, low-cost, and efficient alternative method to collect juvenile salmonids. However, detection rate of fish tagged by angling crews is lower than for most other collection methods (Table 16). Steelhead passively collected by screw trap in White Sands Creek during the summer of 1999 were 3.5 times more likely to be detected in MY 2000 than individuals collected by angling (Kiefer et al. 2001). Passive methods typically collect actively migrating individuals, while active methods also collect residents or individuals delaying migration. Angling is a good collection method if the objective is to study remote populations or to increase their representation in hydrosystem studies. However, it is a relatively inefficient means to increase the number of PIT-tagged smolts in the hydrosystem. Because the NPM angling SAR values were high, angling may be a good way to increase the number of PIT-tagged adults.

RECOMMENDATIONS

Given the inefficiency of angling as a means of increasing the number of smolt tags, we recommend that NPM sampling be scaled back. The effort in 2004 was reduced to four streams. Steelhead Supplementation Studies personnel operated two additional screw traps in 2004, so the number of tags placed should not have changed greatly. However, given the flexibility and low logistic cost of angling crews, future efforts may be increased as conditions mandate.

There is still the question of methodological bias that this research has uncovered. Method evaluation will be better with another year of data. We recommend that a rigorous five-year evaluation be conducted when MY 2003 data can be included.

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Table 12. Streams sampled by drainage area and year.

Drainage Area	Stream	Year	
		2002	2003
Middle Fork Clearwater River basin	Brushy Fork Creek	X	
	North Fork Moose	X	X
Lower Salmon River tributaries	Slate Creek	X	
	Whitebird Creek	X	X
Salmon River Canyon tributaries	Bargamin Creek	X	X
	Chamberlain Creek	X	X
	Horse Creek	X	
Middle Fork Salmon River basin	Big Creek	X	
	Camas Creek	X	X
	Loon Creek	X	
	Middle Fork	X	
	Rapid River		X
	Sulphur Creek	X	X
	Yellowjacket Creek	X	X

Table 13. Number of parr tagged and percent detected by year of tagging. Detections are from all years. Detections are reported only from Lower Granite, Lower Goose, Lower Monumental, and McNary dams to be comparable to older reports.

Stream	# tagged	2002		2003
		% detected	%125-249 det	# tagged
Brushy Fork	462	14.3	16.5	
NF Moose	955	10.4	12.3	1428
Slate Creek	185	6.5	7.7	
Whitebird	960	6.0	5.5	1753
Bargamin	703	9.1	10.6	500
Chamberlain	1334	14.8	17.9	1360
Horse Creek	459	14.6	18.0	590
Big Creek	330	18.2	21.2	
Camas Creek	514	21.8	23.6	767
Loon Creek	380	17.1	19.7	
Middle Fork	298	13.4	18.5	
Rapid River				450
Sulphur Creek	243	14.4	22.7	181
Yellowjacket	217	7.4	8.6	790

Table 14. Number of wild steelhead tagged upstream of Lower Granite Dam and subsequent detection rate (%) by migratory year (MY) for all capture methods (ALL), all angling captures (ANGLE), and angling captures by this project (NPM). Fish tagged includes individuals tagged the previous year that were anticipated to migrate in the given MY.

MY	Fish tagged			% detections	
	ALL	ANGLE	NPM	ALL	NPM
1999	20,777	9776	7195	41.6	16.7
2000	35,328	8305	6525	40.9	20.4
2001	34,253	10,277	7430	41.3	23.0
2002	38,865	7620	5395	32.4	15.4
2003	38,110	8245	7040	28.0	14.0

Table 15. Comparison of estimated smolt-to-adult return rates (90% confidence limits in parentheses) and the difference in precision of the estimates for all PIT-tagged wild steelhead originating upstream of Lower Granite Dam and excluding fish tagged by this project for the 1999-2002 migratory years. Change in precision was calculated as width of the 90% confidence interval as computed from all observations minus the width of the 90% confidence interval as computed excluding fish tagged by this project.

Migratory Year	Smolt-to-adult return (%)		Change in precision
	All data	Excluding NPM	
1999	0.946 (0.820-1.099)	0.963 (0.841-1.102)	-0.018
2000	2.411 (2.223-2.417)	2.286 (2.114-2.292)	-0.016
2001	0.238 (0.190-0.296)	0.231 (0.183-0.291)	0.002
2002	0.594 (0.503-0.698)	0.585 (0.497-0.687)	-0.005

Table 16. Comparison of average detection and smolt-adult return (SAR) rates for selected collection methods in the Snake River basin during Migratory Years 1999-2002.

Method	Tags detected (%)	SAR (%)
Angling	18.2 (16.1-21.4)	1.34 (0.32-3.66)
Dip trap	77.9 (68.0-90.1)	1.02 (0.38-1.88)
Electrofishing	2.8 (0.0 ^a -5.6)	0.78 (0.0-3.03)
Scoop trap	75.6 (67.4-84.8)	1.07 (0.11-2.70)
Screw trap	47.2 (36.3-60.3)	1.00 (0.21-2.24)

^a Only one fish tagged in 1999 was captured by electrofishing.

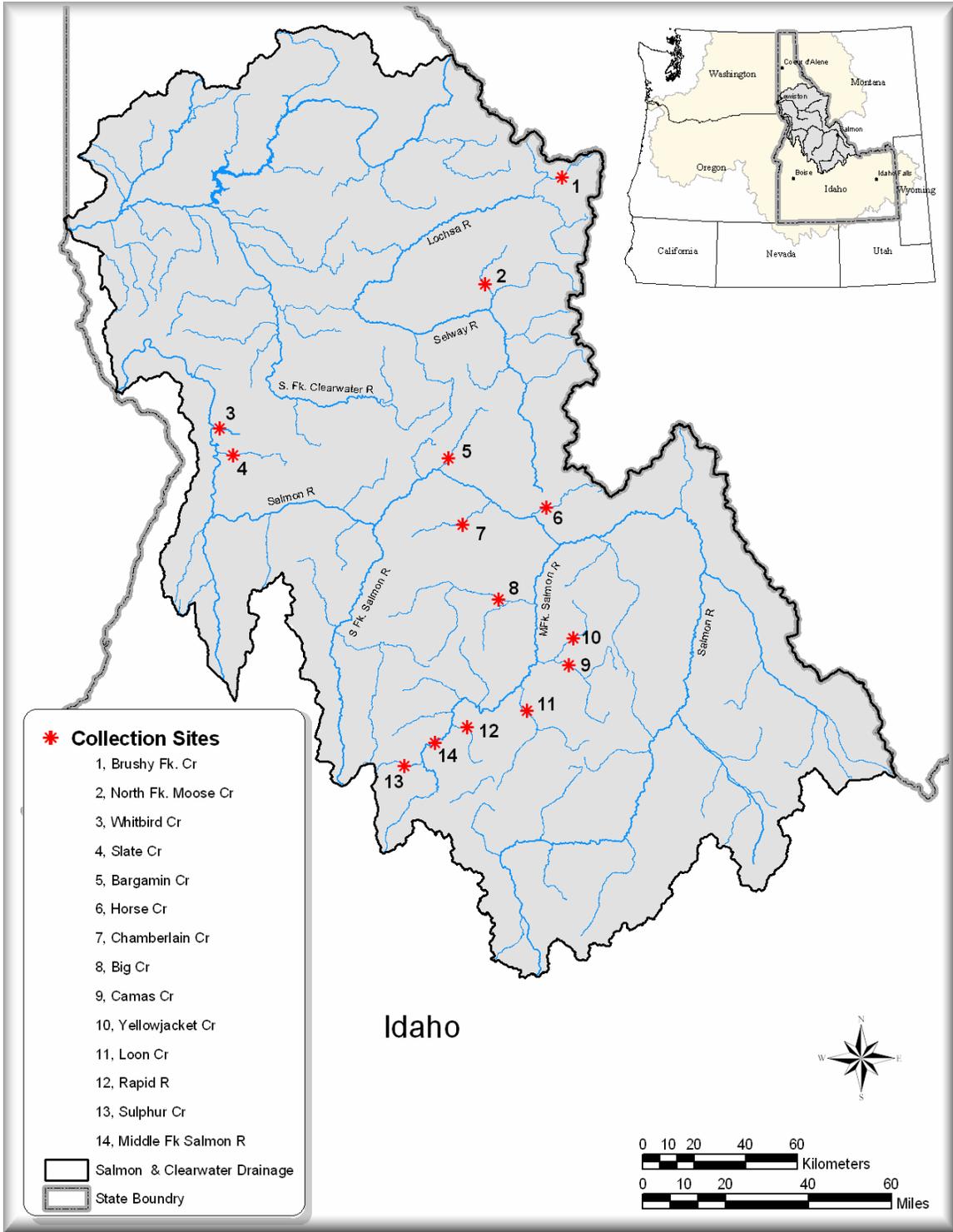


Figure 15. A map of the Snake River basin that is accessible to anadromous salmonids. Sites where NPM personnel collected steelhead parr by angling in 2002-2003 are indicated by asterisks.

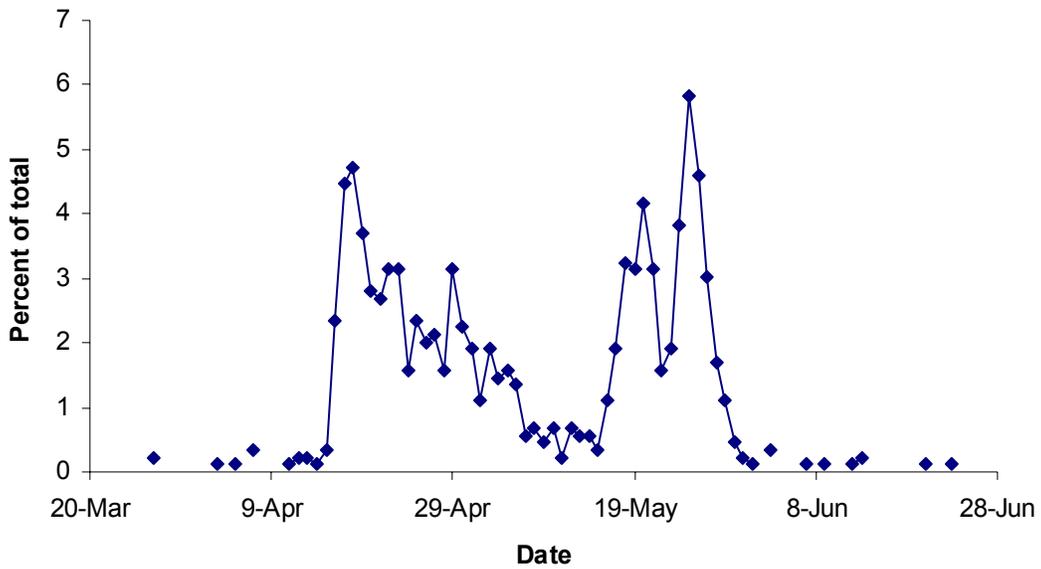


Figure 16. Arrival times at Lower Granite Dam in 2003 of wild steelhead smolts tagged by project personnel.

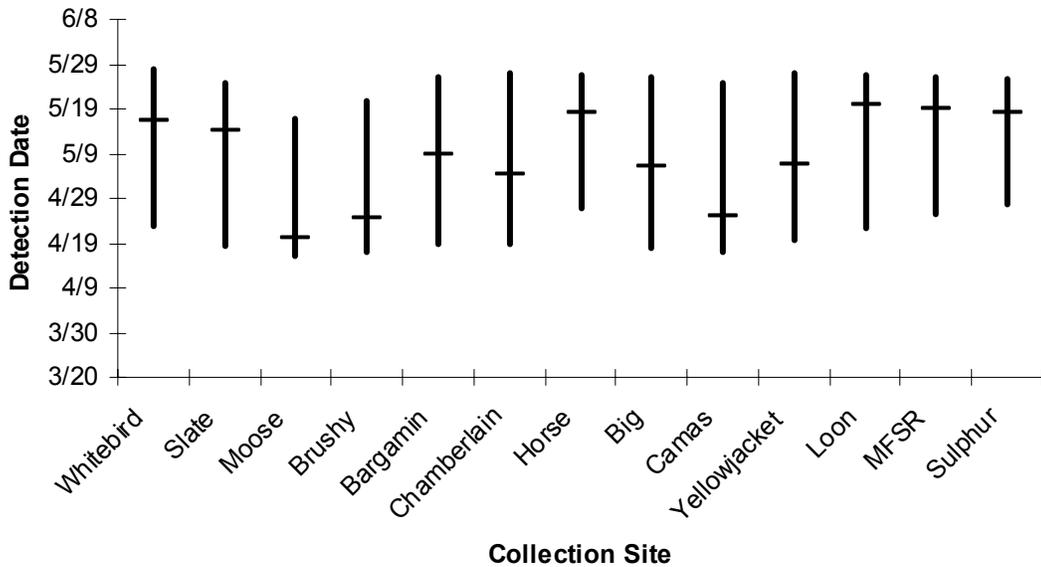


Figure 17. Range of arrival times of wild steelhead smolts at Lower Granite Dam in 2003 by collection site. The bar spans the 10th and 90th percentiles. The hash mark denotes median arrival time. Sites are arranged from left to right by distance from Lower Granite Dam. MFSR = Middle Fork Salmon River.

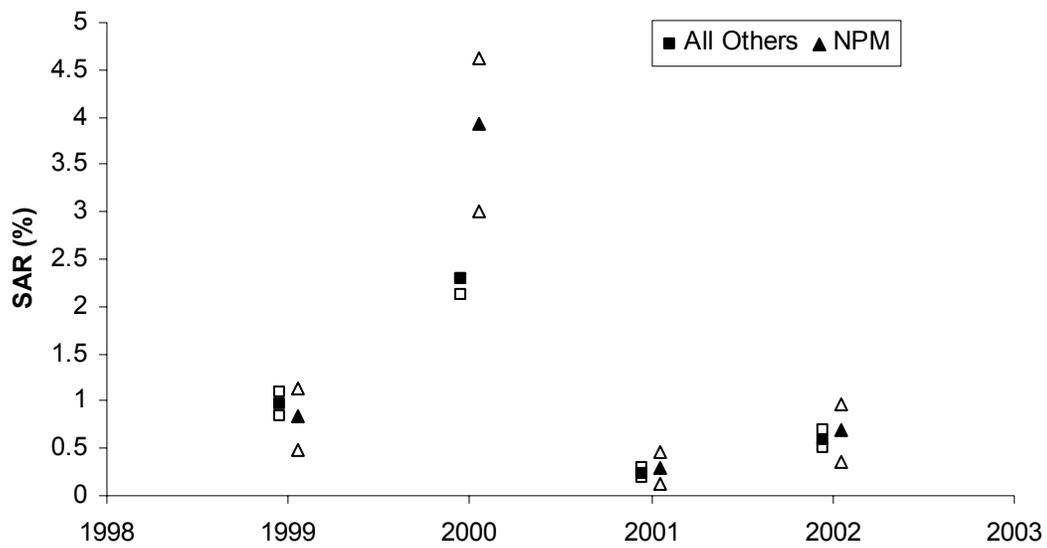


Figure 18. Comparison of smolt-adult return rate (SAR) of individuals collected by angling by this project (NPM) versus all other investigators in the Snake River basin, 1999–2002. Hollow symbols represent the 90% confidence limits. The 2000 estimate based on all other investigators is very close to the upper limit.

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