



**IDAHO NATURAL PRODUCTION  
MONITORING AND EVALUATION**

**Annual Progress Report  
July 1, 2004—June 30, 2005**



Photo credit: June Johnson

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# **Idaho Natural Production Monitoring and Evaluation**

## **Project Progress Report**

### **2004 Annual Report**

**Part 1—Monitoring age composition of wild adult spring/summer Chinook salmon in the Snake River basin in 2004 to estimate smolt-to-adult return rates**

**Part 2—The stock-recruitment relationship for naturally produced spring/summer Chinook salmon in the Snake River basin**

**Part 3—Improve the precision of smolt-to-adult survival rate estimates for wild steelhead trout by PIT tagging additional juveniles**

**Part 4—Monitoring trends in the abundance of anadromous salmonid parr in Idaho**

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## TABLE OF CONTENTS

|   | <u>Page</u> |
|---|-------------|
| PART 1—MONITORING AGE COMPOSITION OF WILD ADULT SPRING/SUMMER<br>CHINOOK SALMON IN THE SNAKE RIVER BASIN IN 2004 TO ESTIMATE SMOLT-TO-<br>ADULT RETURN RATES..... | 1           |
| ABSTRACT .....  | 1           |
| INTRODUCTION .....  | 2           |
| STUDY AREA .....  | 2           |
| METHODS .....   | 2           |
| Sampling .....  | 2           |
| Fin Preparation .....   | 3           |
| Fin Aging .....   | 3           |
| Ocean Age Proportions at Lower Granite Dam .....  | 4           |
| Estimating Aggregate Smolt-to-Adult Return Rates .....  | 5           |
| Population Differences in Length at Maturity .....  | 5           |
| RESULTS AND DISCUSSION .....  | 5           |
| Ocean Age Proportions in 2004 .....   | 5           |
| Aggregate Smolt-to-Adult Return Rates .....   | 6           |
| Factors Affecting Length at Age .....   | 7           |
| Growth Differences Among Spawning Populations .....   | 9           |
| CONCLUSIONS AND RECOMMENDATIONS.....  | 9           |
| ACKNOWLEDGMENTS .....   | 11          |
| LITERATURE CITED .....  | 22          |
| APPENDICES.....   | 26          |
| PART 2—THE STOCK-RECRUITMENT RELATIONSHIP FOR NATURALLY PRODUCED<br>SPRING/SUMMER CHINOOK SALMON IN THE SNAKE RIVER BASIN.....                                    | 29          |
| ABSTRACT .....  | 29          |
| INTRODUCTION .....  | 30          |
| METHODS .....   | 30          |
| RESULTS .....   | 31          |
| DISCUSSION.....   | 32          |
| LITERATURE CITED .....  | 37          |
| PART 3—IMPROVE THE PRECISION OF SMOLT-TO-ADULT SURVIVAL RATE<br>ESTIMATES FOR WILD STEELHEAD TROUT BY PIT TAGGING ADDITIONAL<br>JUVENILES.....                    | 39          |
| ABSTRACT .....  | 39          |
| INTRODUCTION .....  | 40          |
| METHODS .....   | 40          |
| RESULTS .....   | 42          |
| DISCUSSION.....   | 42          |
| Survival by Capture Method .....  | 43          |
| RECOMMENDATIONS.....  | 44          |
| LITERATURE CITED .....  | 45          |

**Table of Contents, continued.**

|   | <u>Page</u> |
|---|-------------|
| PART 4—MONITORING TRENDS IN ABUNDANCE OF ANADROMOUS SALMONID PARR<br>IN IDAHO ..... | 51          |
| ABSTRACT .....  | 51          |
| INTRODUCTION .....  | 52          |
| METHODS .....   | 52          |
| Data Description .....  | 52          |
| Trend Exploratory Analyses .....  | 53          |
| Influences on Detection Probability of Chinook Parr .....                           | 53          |
| Trend Site Selection .....  | 54          |
| RESULTS .....   | 55          |
| Trend Exploration .....   | 55          |
| Influences on Detection Probability of Chinook Parr .....                           | 55          |
| Trend Site Selection .....  | 56          |
| DISCUSSION.....   | 57          |
| Trend Interpretation.....   | 57          |
| Detection Probability, Bias, and Methodology .....                                  | 58          |
| Site Selection .....  | 60          |
| CONCLUSIONS AND RECOMMENDATIONS.....  | 60          |
| LITERATURE CITED.....   | 62          |
| ACKNOWLEDGEMENTS .....  | 76          |
| APPENDICES.....   | 77          |

**LIST OF TABLES**

|          |   |    |
|----------|---|----|
| Table 1. | Estimated proportion of each ocean age by 5 cm fork length group of wild Chinook salmon carcasses collected on spawning grounds in 2004. ....   | 12 |
| Table 2. | Estimated proportion by 5 cm fork length group of wild adult spring/summer Chinook salmon passing Lower Granite Dam during 2004.....  | 12 |
| Table 3. | Estimated proportion by ocean age and 5 cm fork length group of wild Chinook salmon adults passing Lower Granite Dam in 2004. ....  | 13 |
| Table 4. | Number of smolts produced, number of adult returned, and estimated smolt-to-adult return rate (SAR) of the aggregated Snake River wild spring/summer Chinook salmon stock by migratory year. .... | 13 |
| Table 5. | Total number of Chinook carcasses sampled and aged by run, age, and year. ....  | 13 |

**List of Tables, continued.**

|   | <u>Page</u> |
|---|-------------|
| Table 6. Analysis of variation in lengths at two- and three-ocean ages as functions of sex, population, and return year. The results were from a general linear model on the unweighted means using data from the 1998-2004 return years. ....  | 14          |
| Table 7. Comparison of estimates of the parameters of the Beverton-Holt curves ( $\alpha$ , $\beta$ ) from nonlinear and linear models fit with 1990-2002 data. The 95% confidence limits are in parentheses. Parametric confidence intervals were not estimable in the nonlinear model (na). ....  | 34          |
| Table 8. Stream-specific detection rates (%) for steelhead parr tagged by this project 2001-2003 and percent of fish tagged in 2002 that migrated in 2004 (holdovers). Estimates from previous reports (2001 and 2002) were updated with detections from 2004. Data for Chamberlain Creek included fish tagged in West Fork Chamberlain Creek. .... | 47          |
| Table 9. Arrival timing of PIT-tagged steelhead smolts at Lower Granite dam by tagging method and migratory year. Ranges given were the 10 <sup>th</sup> and 90 <sup>th</sup> percentile arrival dates. ....  | 47          |
| Table 10. Number of sites in the GPM database by year sampled and major drainage. ....  | 65          |
| Table 11. Selected percentiles of wetted width, water temperature, and visibility as measured at time of sampling at 2,437 sites. ....  | 65          |
| Table 12. Slope estimates used for references for regressions during 1985-1997 and 1995-2003. ....  | 65          |
| Table 13. Candidate trend monitoring sites and estimated regression parameters for the period 1985-1997 and 1995-2003. Values in parentheses are the 95% confidence limits. ....  | 66          |

**LIST OF FIGURES**

|   |    |
|---|----|
| Figure 1. Location of sites where wild spring/summer Chinook salmon carcasses were collected in 2004. ....  | 15 |
| Figure 2. Representative dorsal fin ray cross sections illustrating the four different ocean ages observed for Snake River spring/summer Chinook salmon adult returns. .... | 16 |
| Figure 3. Length distribution by ocean age as determined from fin cross sections of wild Snake River spring/summer Chinook salmon carcasses collected in 2004. ....         | 16 |

**List of Figures, continued.**

|   | <u>Page</u> |
|---|-------------|
| Figure 4. Length distribution (cm) of wild Snake River spring/summer Chinook crossing Lower Granite Dam in 2004 as measured from video images.....  | 17          |
| Figure 5. Mean length at age by return year of wild spring/summer Chinook carcasses. Error bars are the 95% confidence intervals. ....  | 18          |
| Figure 6. Adjusted mean lengths with error bars (2x standard deviation) for wild spring/summer Chinook aged as two and three ocean, 1998-2004. Years (within age group) with same letter are not significantly different. ....  | 19          |
| Figure 7. Adjusted mean lengths with error bars (2x standard deviation) for two- and three-ocean age groups for wild Snake River spring/summer Chinook populations (see Appendix A for abbreviations), 1998-2004. Clearwater populations are considered spring runs. The spring/summer category includes populations that contain both runs. .... | 20          |
| Figure 8. Annual mean lengths at ocean age by sex. Lines connect the grand means of each age. Females are offset for comparison. Smolt length (ocean age-0) was arbitrarily set at 10 cm. ....  | 21          |
| Figure 9. Comparison of observed data (BY 1990 to BY 2002) to model predictions for the linear fit of the Beverton-Holt model. The BY 2002 point is hollow. ....  | 35          |
| Figure 10. Comparison of the BY 2002 estimate to the Beverton-Holt and Ricker models fit by Copeland et al. (2004). ....  | 35          |
| Figure 11. Number of smolts required to replace spawning females at four survival rates compared to the Beverton-Holt and Ricker stock-recruit functions previously estimated. Smolt sex ratio was assumed to be 1:1. Equilibria are located where a replacement line intersects a stock-recruit curve. ....                                      | 36          |
| Figure 12. A map of the Clearwater and Salmon River drainages. Locations where NPM personnel collected steelhead parr by angling in 2002-2004 are indicated by asterisks. ....  | 48          |
| Figure 13. Date of arrival at Lower Granite Dam in 2004 of steelhead smolts tagged by project personnel.....  | 49          |
| Figure 14. Population-specific arrival dates at Lower Granite Dam in 2004. The vertical bar spans the 10 <sup>th</sup> and 90 <sup>th</sup> percentiles. The hash mark denotes the median arrival date. Streams are arranged from left to right by distance from Lower Granite Dam. MFSR = Middle Fork Salmon River. ....                         | 49          |
| Figure 15. Probability of apparent survival to Lower Granite Dam by capture method and migratory year for PIT-tagged steelhead smolts. Error bars are plus and minus one standard deviation. ....   | 50          |
| Figure 16. Length at tagging by method summed for migratory years 1998-2003.....  | 50          |

## List of Figures, continued.

|   | <u>Page</u> |
|---|-------------|
| Figure 17. The location of all General Parr Monitoring snorkel sites in Idaho.....  | 67          |
| Figure 18. Annual means and selected percentiles of age-0 Chinook parr density. Quadratic trend lines were drawn for each measure except the median. The trend line for means is dashed. ....   | 68          |
| Figure 19. Comparison of trends in annual 75 <sup>th</sup> percentile Chinook parr density between the Clearwater and Salmon basins. ....   | 68          |
| Figure 20. Trends in 75 <sup>th</sup> percentile density by year for the Upper Middle Fork Salmon and Upper Salmon watersheds (fourth field hydrologic unit codes).....   | 69          |
| Figure 21. Percentage of sites sampled in which Chinook parr were observed by year. Sites from the Lower Clearwater HUC, Hells Canyon, and Panther Creek were excluded. ....  | 70          |
| Figure 22. Selected stream characteristics at time of sampling versus number of Chinook parr observed at 2437 GPM sites, 1985-2003. ....  | 71          |
| Figure 23. Proportion of sites where Chinook parr were detected by year and by percentile category (low, medium, and high) based on stream width, temperature, and visibility at time of sampling. Percentiles are defined in Table 11. Trend lines are shown; the medium trend line is dashed..... | 72          |
| Figure 24. Comparison of percentage of sites where Chinook parr were observed by Rosgen channel type. ....  | 73          |
| Figure 25. Seventy-fifth percentile densities by year of 209 trend sites. The quadratic trend line and coefficient of multiple determination are shown. ....  | 73          |
| Figure 26. Trends in 72 selected sites compared to the reference 75 <sup>th</sup> percentile values from Figure 25. Quadratic trend lines are shown. The reference trend is in bold; the trend in median values is dashed.....  | 74          |
| Figure 27. Trends in 12 selected sites compared to the reference 75 <sup>th</sup> percentile values from Figure 25. Quadratic trend lines are shown. The reference trend is in bold; the trend in median values is dashed.....  | 74          |
| Figure 28. Locations of the 12 candidate trend sites.....   | 75          |

## LIST OF APPENDICES

|   | <u>Page</u> |
|---|-------------|
| Appendix A. Table 1. Major population groups, populations and streams where Chinook carcasses were collected. Populations were delineated by the Interior Columbia Basin Technical Recovery Team (2005). Populations for spring Chinook in the Clearwater basin were not delineated by the TRT; we used their delineations for steelhead to define local populations in that basin. Population abbreviations given in parentheses were used in Figure 7. .... | 27          |
| Appendix B. Figure 1. Trends in 75 <sup>th</sup> percentile density during 1985-2003 for each fourth field HUC. Trends were drawn using multiple linear regression with a quadratic term. Note that differences in y-axis scale distort comparisons of slopes. ....   | 78          |
| Appendix B. Table 1. Number of general parr monitoring sites sampled by year and fourth-field hydrologic unit code. CW = Clearwater, SR = Salmon River, MF = Middle Fork, SF = South Fork, MSC = Middle Salmon/Chamberlain, MSP = Middle Salmon/Panther. ....   | 87          |

**PART 1—MONITORING AGE COMPOSITION OF WILD ADULT SPRING/SUMMER  
CHINOOK SALMON IN THE SNAKE RIVER BASIN IN 2004 TO ESTIMATE SMOLT-TO-  
ADULT RETURN RATES**

**ABSTRACT**

Accurate determination of the ocean-age proportions of wild adult spring/summer Snake River Chinook salmon is important for monitoring status and trends. We determined age composition of the 2004 return and updated the run reconstructions and smolt-to-adult returns (SARs) calculated with these data. The bulk of the 2004 return (84.2%) had spent two years in the ocean. Length distributions of all ocean-age classes overlapped adjacent ages; lengths of four-ocean fish were wholly contained within the length distribution of three-ocean fish. We observed a number of two-freshwater salmon and other growth anomalies. Run reconstructions were completed for migratory years (MY) 1996-2000; data for MY 2001-2003 were incomplete. SAR values (excluding jacks) varied from 0.29% to 3.69%. We modeled length for two- and three-ocean ages as a function of return year, sex, and local population. Population and year were significant for two-ocean salmon ( $F > 15.0$ ,  $P < 0.001$ ) but sex was not ( $F = 0.58$ ,  $P = 0.45$ ). All factors were significant for three-ocean fish ( $F \geq 4.37$ ,  $P \geq 0.037$ ). Fish returning in 2003 were longest at both ages. Mean adjusted length at age were not sharply distinct between spring and summer populations; there was a continuum in which mixed-run populations were intermediate. Differences in length between populations may result from local adaptations or environmental effects related to migration and are an aspect of biodiversity relevant to management and recovery planning. We recommend precision and bias of SAR estimates be examined.

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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 are essential to assign returning adults to a specific brood year and to estimate smolt-to-adult return (SAR) rates. Many agencies use scales to age Pacific salmon *Oncorhynchus* species. 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 (Chilton and Bilton 1986), such as the migrations made by Snake River salmon. Frequently, scales are damaged or regenerated, which makes them impossible to read. Inaccurate age data can lead to serious errors in the management of populations (Zabel and Levin 2002).

Project personnel began to collect fin rays from carcasses to determine ocean ages in 1998. The original motivation for this effort was the mismatch of scale ages with known ocean ages of fish tagged with passive integrated transponder (PIT) tags as juveniles. Ocean ages based on fin rays have been >97% accurate in any given year based on known-age adults (Kiefer et al. 2001, 2002, 2004). These data then allow accurate reconstruction of the age structure of a run, assignment of fish back to the appropriate smolt year, and calculation of survival rates.

In this report, we present the age structure of spring/summer Chinook salmon carcasses collected in 2004 throughout the Snake River basin upstream of Lower Granite Dam (LGD). These data were used to assign the run-at-large to age categories and to update and extend SAR rates previously calculated for migratory years 1998-2003. We also examined the effects of sex, run, and population on the length at age.

## STUDY AREA

The study area encompasses streams in the Snake River basin upstream of LGD that have spawning populations of spring/summer Chinook salmon. Field personnel sampled carcasses from 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 on August 11-12, 2004 at Marsh Creek near Stanley, Idaho. Idaho Department of Fish and Game (IDFG) trainers demonstrated techniques for collecting aging structures and recording data.

Collection techniques were the same as used in past years (Kiefer et al. 2004). Carcasses were measured to fork length (FL). Dorsal fins were collected and inserted into a coin envelope. A small pencil-eraser-size piece (approximately 16 mm<sup>2</sup>) of fin tissue with good color (excluding the adipose fin) was removed and placed in a test tube filled with 95% ethanol

for future genetic analysis. Samples were transported to the IDFG Fisheries Research Office in Nampa, Idaho and were stored in a freezer until preparation for aging.

Most 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. We used known-age samples to estimate accuracy in the current year. The archive of past known-age fin rays was used as training material for new personnel and as a reference to help identify split annuli and other abnormalities.

### **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 aligned 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.

Laboratory technicians cut each fin into cross-sections by making eight to ten thin slices (1.2 mm) in each fin using a water-cooled high-speed sectioning saw. Cross-sections were placed on a microscope slide to dry in the order in which they were sliced. Sections were affixed onto microscope slides under the fume hood using a clear liquid mounting medium to improve resolution, adhere the sample to the slide, and preserve the sample.

### **Fin Aging**

Mounted fin ray sections were aged using a compound microscope and green, filtered, transmitted light. Light passing through sections illuminated wide opaque zones alternating with narrower translucent 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 winter zones were 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 (Petrosky et al. 2001). 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 (Figure 2), because ocean winters are not as cold as those spent in freshwater, permitting some growth.

We assembled a reference collection of known-age Chinook salmon to train readers and provide guidance when age criteria are unclear. This reference collection was comprised of Chinook salmon tagged with PIT tags as smolts and recovered as returning adults.

All samples were independently aged by at least two technicians trained in fin aging techniques. 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 and the fin ray was classified as not readable.

### **Ocean Age Proportions at Lower Granite Dam**

Size and sex influence carcass recovery rates, so carcass samples may not represent the true composition of the population (Zhou 2002). We used video images to determine the length frequency distribution of wild adult spring/summer Chinook salmon passing LGD in 2004 to correct for any length biases in the carcass sample. A video camera recorded adults passing the viewing window at LGD for 24 hours every third day from April through August. We randomly selected the initial start day for recording videos from the anticipated first week of the migration. The start date 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.

The length frequency distribution of the total run was calculated based on samples from the videotapes. We estimated the number of viewing minutes necessary to obtain approximately 400 images of Chinook salmon. We randomly selected a 104 min segment of each videotape for viewing. Fish with a full adipose fin were assumed to be of natural origin. Individuals missing all or part of the adipose fin were assumed to be of hatchery origin. The images of natural-origin adults were digitized and FL was estimated from 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 videotape. The ratio of stick lengths (in pixels) to the known length between the lines on the window was calculated. We used this ratio to estimate the actual FL of the digitized images.

We assigned the total 2004 run passing LGD to smolt years. First, we constructed a length frequency distribution by 5 cm increments using FLs estimated from video images:

$$p_i = \frac{v_i}{V}$$

where  $p_i$  is the proportion of the run in length category  $i$ ,  $v_i$  is the number of images in length category  $i$ , and  $V$  is the number of images measured. Similarly, the age distribution of each length group was calculated based on the carcass samples:

$$a_{ij} = \frac{m_{ij}}{M_j}$$

where  $a_{ij}$  is the proportion of carcasses of length  $i$  at ocean age  $j$ ,  $m_{ij}$  is the number of carcasses of length  $i$  at ocean age  $j$ , and  $M_j$  is the total number of carcasses of length  $j$ . The age distribution of the carcass sample was expanded to the total run by multiplying  $a_{ij}$  by  $p_i$ . These proportions were then summed for each age and multiplied by the number of wild spring/summer Chinook salmon passing LGD as estimated by the U.S. v. Oregon Technical Advisory Committee (TAC; S. Marshall, IDFG, personal communication):

$$n_j = N \sum_{i=1}^4 p_i a_{ij}$$

where  $n_j$  is the number of fish at ocean age  $j$  and  $N$  is the TAC escapement estimate.

### **Estimating Aggregate Smolt-to-Adult Return Rates**

We combined the age assignments with estimates of out-migrating smolts from a stock-recruitment analysis (see Part 2 of this report) to estimate an aggregate SAR estimate for wild Snake River spring/summer Chinook salmon. For a particular smolt year, we used the estimated number of wild smolts arriving at LGD for smolt abundance and the sum of returns from the ocean of that migratory cohort for adult abundance:

$$SAR_k = \frac{\sum_{l=1}^4 r_{k+l}}{S_k}$$

where  $SAR_k$  is the SAR rate of migratory year  $k$ ,  $r_{k+l}$  is the return from that cohort in year  $k+l$ , and  $S_k$  is the estimate of smolts migrating in year  $k$ . The maximum value of  $l$  is 4 because that is the maximum ocean age we have observed in the past (Copeland et al. 2004). We reported smolt-to-adult return rates as a percentage. SARs were reported with and without one-ocean returns (jacks).

### **Population Differences in Length at Maturity**

We investigated differences in length at age among populations within the Snake River stock using a general linear model based on unweighted means. Population groupings followed the Interior Columbia Basin Technical Recovery Team's (2005) delineations (Appendix A). We also entered sex and return year into the model to help explain variation. Two- and three-ocean Chinook salmon were analyzed separately because the data were most abundant at those ages, and length differences between those ages are of trivial importance. Two-freshwater fish were omitted. Tukey's post hoc test was used to detect differences among years and sexes.

## **RESULTS AND DISCUSSION**

### **Ocean Age Proportions in 2004**

Field technicians and hatchery personnel collected three known-age natural-origin fish and 66 known-age hatchery fish in 2004. Laboratory technicians aged 68 of the 69 (98.6%) known-age fish correctly. There were 11 one-ocean fish, 54 two-ocean fish, and 4 three-ocean fish. No four-ocean fish with PIT tags were collected, although two passed LGD (data not shown).

Ages could not be determined for nine of the 588 fin rays examined. Some unusual individuals were sampled in 2004. We collected a three-ocean fish that was only 59 cm FL and a five-ocean fish (96 cm FL). These outliers were removed from further analysis. There were

also 60-70 two-freshwater Chinook salmon, which seems high. We were uncertain in identification of the second freshwater annulus, and these classifications should be verified. Two-freshwater individuals were included in subsequent analyses, because we focused on ocean age.

The majority of the carcasses aged in 2004 were classified as two-ocean returns (Table 1, Figure 3). The length distributions of one- and two-ocean ages overlapped (Table 1). The overlap between two- and three-ocean ages was more substantial (Figure 3). Lengths of four-ocean fish were encompassed within the length range of three-ocean fish (Figure 3). Four-ocean fish were slow growers that stayed longer in the ocean before maturing (Ricker 1981; Quinn 2005). The overlap among age groups is consistent with past results (Kiefer et al. 2001, 2002, 2004; Copeland et al. 2004).

We measured 312 video-recorded salmon as they passed LGD (Table 2). The overall length distribution of the run at large was unimodal with extended tails at both extremes (Figure 4). Over half of the fish measured were within the 70-79 cm groups (Figure 4). Unfortunately, video recording did not begin until May 5, 2004. The peak of the daily run count for adults (two-ocean and older Chinook) was missed by the video record. The peak of the daily jack count (one-ocean males) was not missed; however, relatively few jacks returned in 2004. Additional minutes were added to video sampling time to compensate for the days of missing videotapes.

We used lengths of randomly selected fish at Lower Granite Dam to estimate length structure for the entire run. Collectors are more likely to find larger carcasses, because they are easier to see and more likely to catch on rocks and woody debris instead of flushing downstream (Zhou 2002). Therefore, caution should be used when using carcass samples to estimate population parameters. However, the length frequency of the fish observed at LGD (Figure 4) qualitatively was similar to that of the carcasses (Figure 3). After expanding the carcass data to reflect the total run, we found 84.2% of the 2004 return was in the two-ocean age class (Table 3).

### **Aggregate Smolt-to-Adult Return Rates**

Run reconstructions were complete through migratory year (MY) 2000. Estimated SAR values (excluding jacks) ranged from 0.29% for MY 1996 to 3.69% for MY 1999 (Table 4). Since 1998 was the first year for which we had aging results, estimates were calculated for MY 1996 SAR without one-ocean returns. Reconstructions were completed through three-ocean returns for MY 2001 and two-ocean returns for MY 2002. Run reconstruction SAR estimates for wild Chinook salmon have been comparable to those estimated with PIT tag data for past smolt years (Kiefer et al. 2001). In 2004, TAC revised their previous escapement estimates (S. Marshall, IDFG, personal communication). Incorporating these updated escapements did not change SAR estimates for 1996-1998 very much ( $\leq 0.2\%$ ) but did increase the MY 1999 estimate from 3.05% as reported by Copeland et al. (2004) to 3.79%.

Misclassification of hatchery fish could introduce a large bias in SAR estimates for wild fish because the hatchery returns were much larger than the wild escapement. Origin of salmon passing the viewing window at LGD was classified based on presence of an adipose fin. However, the 'wild' category included unclipped hatchery fish, partially clipped fish with regenerated fins, and supplementation fish, which deliberately were not marked. Because both adults and smolts were classified based on fin clip, misclassified fish were in both numerator

and denominator of the SAR estimate. If their survival were equivalent to actual naturally produced fish, then the SAR is unbiased. However, hatchery fish typically do not survive as well as wild fish (Raymond 1988; Bouwes et al. 2002), which would cause the estimate to be lower than actual survival. We believe misclassified individuals were a small percentage of the 'wild' run; therefore, any bias should be small. This assumption should be carefully evaluated in the future.

### **Factors Affecting Length at Age**

Samples were not evenly distributed among ages, years, or even between run types (Table 5). Few jacks have been sampled over the years due to low jack returns or undersampling. Little emphasis has been placed on collection of jacks (<70 cm FL), because it was thought incorrectly that there is very little overlap in length between jacks and two-ocean fish (Kiefer et al. 2002). Although field personnel were instructed to collect carcasses <70 cm FL, smaller carcasses are harder to find (especially males) and were generally a small proportion of the run at large (Table 3).

Two aspects of the data were apparent. Mean length at age fluctuated among return years (Figure 5). Also, summer-run salmon (in aggregate) tended to be longer at age than the aggregated spring runs (Figure 5). Two-ocean spring Chinook salmon were smaller than two-ocean summer Chinook salmon for every year. The 95% confidence intervals around mean length overlapped between spring and summer for one-ocean fish except in 2003, but summer-run fish tended to be larger. Similarly, 95% confidence intervals overlapped for three-ocean Chinook salmon.

Run designations are somewhat arbitrary. Stream sections were classified as spring or summer-run based on when Chinook spawning there cross Bonneville Dam. These designations were based on tag recoveries by Horton and Wallace (1966) during a single year. In some cases, both spring and summer runs exist in a single stream. There were relatively few recoveries in most streams with which to make a clear designation from this one-year's study (C. Petrosky, IDFG, personal communication). Variation within run category is likely and could be important in characterizing each population.

We examined the apparent length differences between the runs at a finer scale (i.e. among populations) while accounting for important influences on length at age. The general linear model incorporating return year, gender, and population explained a significant proportion of the variance in length at two- and three-ocean fish (Table 6). The model for three-ocean fish was a better fit than that for two-ocean fish ( $R^2 = 0.41$  and  $0.23$ , respectively).

In two-ocean fish, return year and population had significant effects on length (Table 6). The adjusted mean length was largest in 2003 and smallest in 2004 (Figure 6). Only eight two-ocean fish were collected in 1998, so this year was omitted from the analysis. Length tended to be smaller in spring populations, particularly those in the Clearwater basin, while summer populations were longer (Figure 7, bottom). However, differences among populations overlapped; i.e. some spring run populations had adjusted mean lengths similar to summer populations. Mixed spring/summer populations occupied a position between spring and summer runs.

In three-ocean fish, all factors had significant effects (Table 6). There were not as many differences among years, although adjusted mean length was largest in 2003. Three-ocean

male Chinook were, on average, 8 cm larger than three-ocean females (Figure 8). This difference should be checked against mid-eye to hypural length to verify that it is not due merely to greater development of secondary sexual characteristics in older males (e.g., Hendry and Berg 1999). There were more similarities among populations at length for three-ocean salmon, although fish from the Lochsa and South Fork Clearwater drainages were smaller than other populations (Figure 7).

Results of the two- and three-ocean models were consistent in several ways. Year effects explained the greatest portion of variation in length at age as measured by the mean sums of squares (Table 6). Sex explained the least amount of variation and was not significant for two-ocean fish. Interestingly, population effects were almost as large as annual effects. However, there were some obvious differences between the models. Sex was significant in the three-ocean model, yet there were fewer differences among years and populations in this model. Length begins to approach an asymptote after three years in the ocean, while the sexes begin to separate (Figure 8). If this asymptote represents the growth potential for Chinook salmon from the Snake River, then one would expect faster growing groups to slow as they approach it and slower growing groups to catch up in length; thus, differences should disappear with time in the common environment. This hypothesis would also explain the similarity in length of three- and four-ocean salmon, although there were not enough four-ocean samples to conduct a formal test.

The statistical model we used had several features to consider during interpretation of the results. Comparisons were made using means of each factor corrected for the effects of the other factors. For example, values shown in Figure 7 are population means adjusted for the effects of annual and gender variation. Each year was weighted equally, even though samples were not evenly distributed among years (Table 5). Error bars were wider for populations with few years of data. A more serious concern was that both models had a linear trend in residuals such that they overpredicted at shorter lengths and underpredicted at longer lengths. Transformation did not correct this trend. We believe there was some factor not measured by carcass samples that systematically affected ocean growth. Nevertheless, differences detected by the models were conservative and likely were larger than shown. We did not test for specific differences among populations, because the large number of comparisons would greatly increase the risk of a Type 1 error.

Length at age is an important consideration in fish biology, because growth integrates the effects of environmental and intrinsic factors on fish performance (Devries and Frie 1996). Maturity in fish is linked to size or growth rate early in life (Roff 1984). In salmon, age and length data from carcasses are *de facto* length and age at maturity because immature salmon do not return from the ocean to spawn. Fast growing (larger) fish mature early and slow growing (smaller) juvenile salmon mature later (Quinn 2005; Scheuerell 2005). In particular, proportion of jacks increases with size and growth rate of smolts (Heath et al. 1996; Vøllestad et al. 2004). Additionally, poor ocean growth is associated with poor survival (Beamish et al. 2004a). Thus, growth can influence age structure and size of the run.

There is considerable variation from year to year in ocean growth of salmon (Quinn 2005). Annual effects were the largest influence on variation in length at age in our data. There are several potential causes of this variation: climatic shifts (Beamish et al. 2004b), changes in community composition of the food chain (Welch and Parsons 2003), density-dependent marine growth (McKinnell 1995), and harvest (Bigler et al. 1996). The presence of other salmonids in the ocean can cause density dependent effects on growth (Ruggerone et al. 2003). Most of these effects are thought to be mediated through ocean zooplankton (Peterman 1987). While

Snake River stocks have been studied intensely, stock-specific influences on ocean growth have yet to be identified.

### **Growth Differences Among Spawning Populations**

Salmon populations often differ in growth and life history characteristics even within the same river basin (e.g., Peterman et al. 1986). Our analysis demonstrated that Snake River Chinook populations differed in length after two years in the ocean (Figure 7). The Salmon River Lower Mainstem population was consistently longest. The Clearwater populations tended to be shorter. Other spring populations were slightly longer than the Clearwater spring populations. Thus, there are significant differences in ocean growth among the local populations within the Snake River basin. There are two groups of hypotheses to explain population differences in length at age: environmental or parental influences (Peterman 1985). Environmental influences on growth during residence in freshwater and juvenile migration timing could have important influences on eventual ocean growth, as well as adult ocean performance. Inherited traits could influence adult size on the spawning grounds. We will discuss each, but note that no hypothesis excludes the others.

Population differences could be related to the timing of juvenile entry into the ocean. The first year at sea is critical and cohort survival is linked to growth (Beamish and Mahnken 2001). Populations within the Snake River basin differ in migration timing (Achord et al. 1996), and faster growing juveniles migrate sooner than smaller smolts of the same age (Quinn 2005). If some populations consistently enter the ocean earlier or during more optimal growing conditions, they will reach a larger size than later migrating populations. Summer-run populations in Idaho are found lower in elevation and in larger rivers than spring-run populations. These factors could increase size of smolts produced and promote earlier migration. Indeed, summer Chinook in the Snake River basin migrate earlier than spring Chinook (Achord et al. 1996). Larger Chinook smolts tend to retain a size advantage after exiting freshwater (Neilson and Geen 1986). There is also some evidence that fast freshwater growth may promote fast ocean growth (Johnsson et al. 1997; Snover et al. 2005). Fish that stay in the ocean longer attain greater growth, even if it is only for a few months; therefore, summer Chinook salmon should be larger than spring Chinook salmon (note this possibility complements early juvenile migration). However, the early marine environment, rather than late, is more likely to be the critical period (Peterman 1985).

The intrinsic, inherited potential for adult size could differ among populations. Migratory costs and spawning ground characteristics can influence adult characteristics such as size and shape (Quinn et al. 2001; Kinnison et al. 2003; Crossin et al. 2004). When factors affecting body size vary spatially, discrete populations should evolve different body sizes to suit local selective regimes (Quinn et al. 2001). For example, summer Chinook tend to spawn in larger rivers and over larger substrate than spring Chinook and tend to be longer at age. Salmon spawning in larger rivers tend to be larger than other stocks within the same region, partially the result of selection for conditions in the natal river (Beacham et al. 1988).

### **CONCLUSIONS AND RECOMMENDATIONS**

Estimated SAR for the aggregated Snake River spring/summer populations varied over an order of magnitude during our study period (0.29% to 3.69%, excluding jacks). We are

confident that run reconstruction was founded on precise and accurate age information. However, the effect of uncertainty in other inputs on SAR precision has never been assessed. We recommend calculation of 1<sup>st</sup> order variance estimates for SAR values directly (e.g., confidence limits for proportions based on the binomial distribution), as well as uncertainty in the numerator and denominator values derived from sampling error. The magnitude and direction of bias caused by assignment errors (hatchery misclips or supplementation) also should be investigated. A rigorous assessment of SAR estimation will increase their usefulness to managers.

Variation in length at age in wild Chinook was related to interannual and population differences. Gender differences also were important but did not explain as much variation as the former two factors. Other factors may be important but were not measured in association with carcass collection. We used the population differences reported for qualitative exploratory purposes. Further analyses should consider more general groups (e.g., major population groups, Appendix A) to cluster and so increase sample sizes in streams/populations of interest.

Growth influences age structure in Snake River Chinook salmon (Scheuerell 2005). Given the length-at-age differences among populations observed, differences in age structure may also occur. The correlation of ocean conditions to run age structure should be explored. Given the link between growth, survival, and age at maturity, our results have implications for salmon management (Peterman 1985).

It is unclear if differences in length at age were due to genotypic or phenotypic factors. However, the presence of such differences suggests differential management may be appropriate so unique adaptive characteristics are not lost (Levin and Schiewe 2001). Loss of a locally adapted population will result in an immediate and lasting drop in regional productivity (Adkison 1995). Understanding adaptive characteristics is therefore important for conservation and recovery efforts (Zabel and Williams 2002). There are several ways to assess differences in selective pressures among populations. For example, morphometry (body shape) may be more sensitive to population-specific selective pressures than body size. A morphometric comparison among populations may help us understand the evolution and adaptations of stocks to a particular stream over time (e.g., Beacham et al. 1988; Kinnison et al. 1998).

The 2004 return was lower than expected. We coincidentally observed slow ocean growth and several growth-related anomalies in these fish. Identification of two-freshwater fish should be verified for 2004 and the fin ray archive should be checked for more of these fish. Density-dependence in the freshwater phase (see Part 2) may cause slower growth and delayed smoltification. This may lead to decreased ocean growth and survival (Beamish et al. 2004a) and influence adult fitness and escapement (Nicieza and Braña 1993). A large number of two-freshwater fish, which grow more slowly than one-freshwater fish, may change the maturity schedule and survival assumptions on which run forecasts are based.

## **ACKNOWLEDGMENTS**

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Table 1. Estimated proportion of each ocean age by 5 cm fork length group of wild Chinook salmon carcasses collected on spawning grounds in 2004.

| <b>Fork Length Group (cm)</b> | <b>1-Ocean</b> | <b>2-Ocean</b> | <b>3-Ocean</b> | <b>4-Ocean</b> | <b>N</b> |
|-------------------------------|----------------|----------------|----------------|----------------|----------|
| <50                           | 1.00           | 0.00           | 0.00           | 0.00           | 2        |
| 50-54                         | 1.00           | 0.00           | 0.00           | 0.00           | 3        |
| 55-59                         | 1.00           | 0.00           | 0.00           | 0.00           | 4        |
| 60-64                         | 0.06           | 0.94           | 0.00           | 0.00           | 16       |
| 65-69                         | 0.04           | 0.96           | 0.00           | 0.00           | 57       |
| 70-74                         | 0.01           | 0.99           | 0.00           | 0.00           | 144      |
| 75-79                         | 0.00           | 0.99           | 0.01           | 0.00           | 154      |
| 80-84                         | 0.00           | 0.93           | 0.07           | 0.00           | 75       |
| 85-89                         | 0.00           | 0.34           | 0.66           | 0.00           | 38       |
| 90-94                         | 0.00           | 0.08           | 0.89           | 0.03           | 36       |
| 95-99                         | 0.00           | 0.07           | 0.86           | 0.07           | 29       |
| 100-104                       | 0.00           | 0.00           | 1.00           | 0.00           | 7        |
| >104                          | 0.00           | 0.00           | 0.92           | 0.08           | 12       |

Table 2. Estimated proportion by 5 cm fork length group of wild adult spring/summer Chinook salmon passing Lower Granite Dam during 2004.

| <b>Length Group</b> | <b># of occurrences</b> | <b>Proportion</b> |
|---------------------|-------------------------|-------------------|
| <50                 | 4                       | 0.012821          |
| 50-54               | 6                       | 0.019231          |
| 55-59               | 5                       | 0.016026          |
| 60-64               | 15                      | 0.048077          |
| 65-69               | 49                      | 0.157051          |
| 70-74               | 86                      | 0.275641          |
| 75-79               | 83                      | 0.266026          |
| 80-84               | 29                      | 0.092949          |
| 85-89               | 15                      | 0.048077          |
| 90-94               | 13                      | 0.041667          |
| 95-99               | 4                       | 0.012821          |
| 100-104             | 2                       | 0.00641           |
| >104                | 1                       | 0.003205          |

Table 3. Estimated proportion by ocean age and 5 cm fork length group of wild Chinook salmon adults passing Lower Granite Dam in 2004.

| <b>Fork Length (cm)</b> | <b>1-Ocean</b> | <b>2-Ocean</b> | <b>3-Ocean</b> | <b>4-Ocean</b> |
|-------------------------|----------------|----------------|----------------|----------------|
| <50                     | 0.013          | 0.000          | 0.000          | 0.000          |
| 50-54                   | 0.019          | 0.000          | 0.000          | 0.000          |
| 55-59                   | 0.016          | 0.000          | 0.000          | 0.000          |
| 60-64                   | 0.003          | 0.045          | 0.000          | 0.000          |
| 65-69                   | 0.006          | 0.152          | 0.000          | 0.000          |
| 70-74                   | 0.002          | 0.274          | 0.000          | 0.000          |
| 75-79                   | 0.000          | 0.264          | 0.002          | 0.000          |
| 80-84                   | 0.000          | 0.087          | 0.006          | 0.000          |
| 85-89                   | 0.000          | 0.016          | 0.032          | 0.000          |
| 90-94                   | 0.000          | 0.003          | 0.037          | 0.001          |
| 95-99                   | 0.000          | 0.001          | 0.011          | 0.001          |
| 100-104                 | 0.000          | 0.000          | 0.006          | 0.000          |
| >104                    | 0.000          | 0.000          | 0.003          | 0.000          |
| <b>Total</b>            | <b>0.059</b>   | <b>0.842</b>   | <b>0.097</b>   | <b>0.002</b>   |
| <b># Adults</b>         | <b>2207</b>    | <b>31770</b>   | <b>3659</b>    | <b>87</b>      |

Table 4. Number of smolts produced, number of adult returned, and estimated smolt-to-adult return rate (SAR) of the aggregated Snake River wild spring/summer Chinook salmon stock by migratory year.

|                 | <b>Migratory Year</b>  |             |             |             |             |             |             |             |
|-----------------|------------------------|-------------|-------------|-------------|-------------|-------------|-------------|-------------|
|                 | <b>1996</b>            | <b>1997</b> | <b>1998</b> | <b>1999</b> | <b>2000</b> | <b>2001</b> | <b>2002</b> | <b>2003</b> |
| Smolts          | 419826                 | 161157      | 599159      | 1560298     | 1344382     | 500700      | 1125819     | 1457418     |
|                 | <i>Adults Returned</i> |             |             |             |             |             |             |             |
| Smolt year +1   | —                      | 174         | 255         | 1670        | 1884        | 366         | 2438        | 1421        |
| Smolt year +2   | 758                    | 2309        | 7732        | 43259       | 15963       | 6880        | 20661       | —           |
| Smolt year +3   | 472                    | 456         | 1279        | 13601       | 24525       | 245         | —           | —           |
| Smolt year +4   | 0                      | 34          | 308         | 674         | 61          | —           | —           | —           |
| SAR w/jacks     | —                      | 1.84        | 1.60        | 3.79        | 3.16        | 1.50        | 2.05        | 0.10        |
| SAR w/out jacks | 0.29                   | 1.74        | 1.56        | 3.69        | 3.02        | 1.42        | 1.84        | —           |

Table 5. Total number of Chinook carcasses sampled and aged by run, age, and year.

| <b>Sample Year</b> | <b>1-ocean</b> |               | <b>2-ocean</b> |               | <b>3-ocean</b> |               | <b>4-ocean</b> |               |
|--------------------|----------------|---------------|----------------|---------------|----------------|---------------|----------------|---------------|
|                    | <b>Spring</b>  | <b>Summer</b> | <b>Spring</b>  | <b>Summer</b> | <b>Spring</b>  | <b>Summer</b> | <b>Spring</b>  | <b>Summer</b> |
| 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             |
| 2004               | 4              | 9             | 275            | 179           | 42             | 69            | 2              | 2             |

Table 6. Analysis of variation in lengths at two- and three-ocean ages as functions of sex, population, and return year. The results were from a general linear model on the unweighted means using data from the 1998-2004 return years.

| <b>Source</b>      | <b>Sums of squares</b> | <b>df</b> | <b>Mean squares</b> | <b>F</b>  | <b>P</b> |
|--------------------|------------------------|-----------|---------------------|-----------|----------|
| <i>Two-Ocean</i>   |                        |           |                     |           |          |
| Grand mean         | 3014315.0              | 1         | 3014315.0           | 113996.09 | <0.001   |
| Sex                | 15.3                   | 1         | 15.3                | 0.58      | 0.446    |
| Population         | 8931.5                 | 22        | 406.0               | 15.35     | <0.001   |
| Year               | 3447.9                 | 5         | 689.6               | 26.08     | <0.001   |
| Error              | 56692.2                | 2144      | 26.4                |           |          |
| <i>Three-Ocean</i> |                        |           |                     |           |          |
| Grand Mean         | 2761021.5              | 1         | 2761021.5           | 90263.6   | <0.001   |
| Sex                | 133.7                  | 1         | 133.7               | 4.37      | 0.037    |
| Population         | 5338.9                 | 20        | 266.9               | 8.73      | <0.001   |
| Year               | 1733.6                 | 6         | 288.9               | 9.45      | <0.001   |
| Error              | 35513.2                | 1161      | 30.6                |           |          |

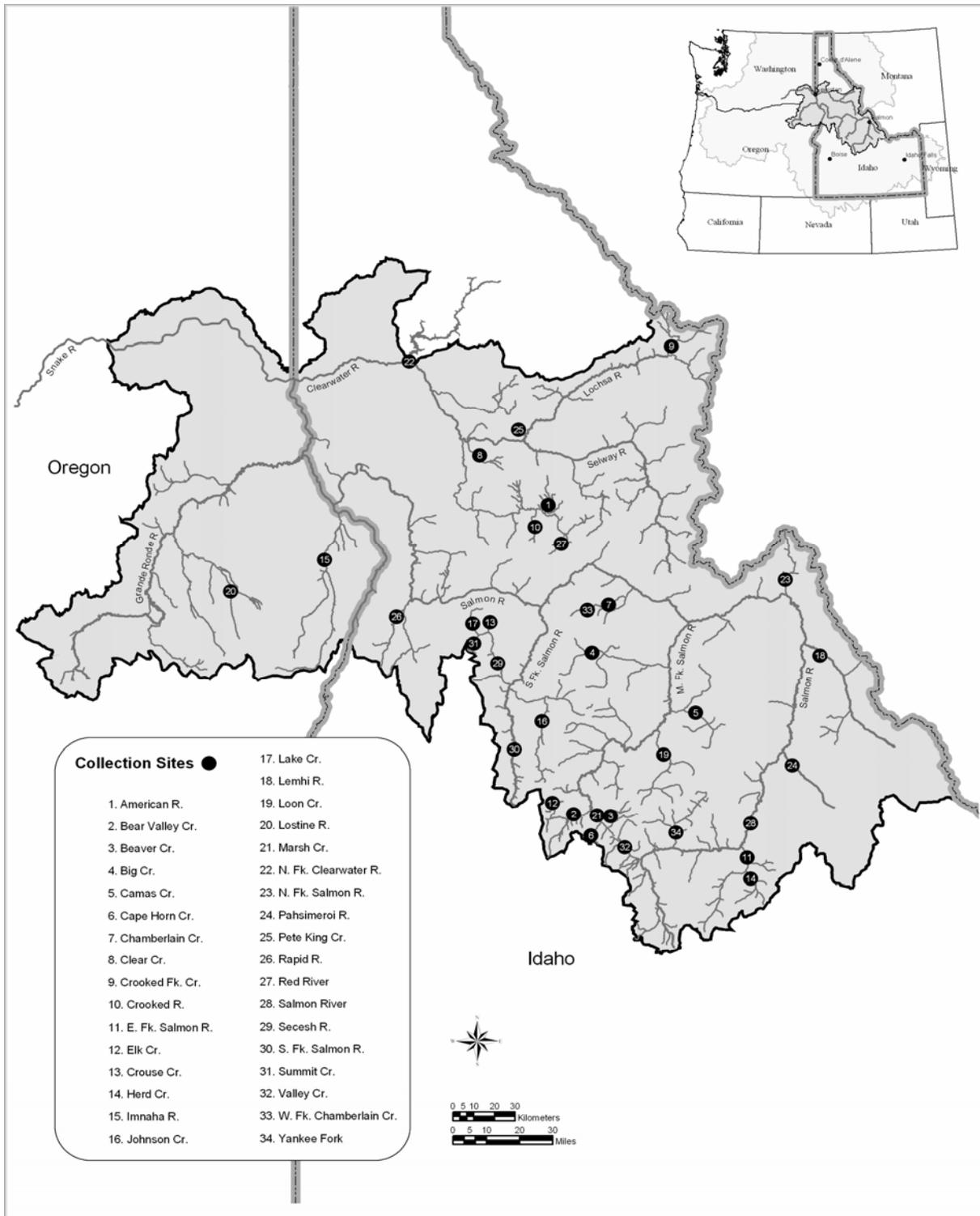


Figure 1. Location of sites where wild spring/summer Chinook salmon carcasses were collected in 2004.

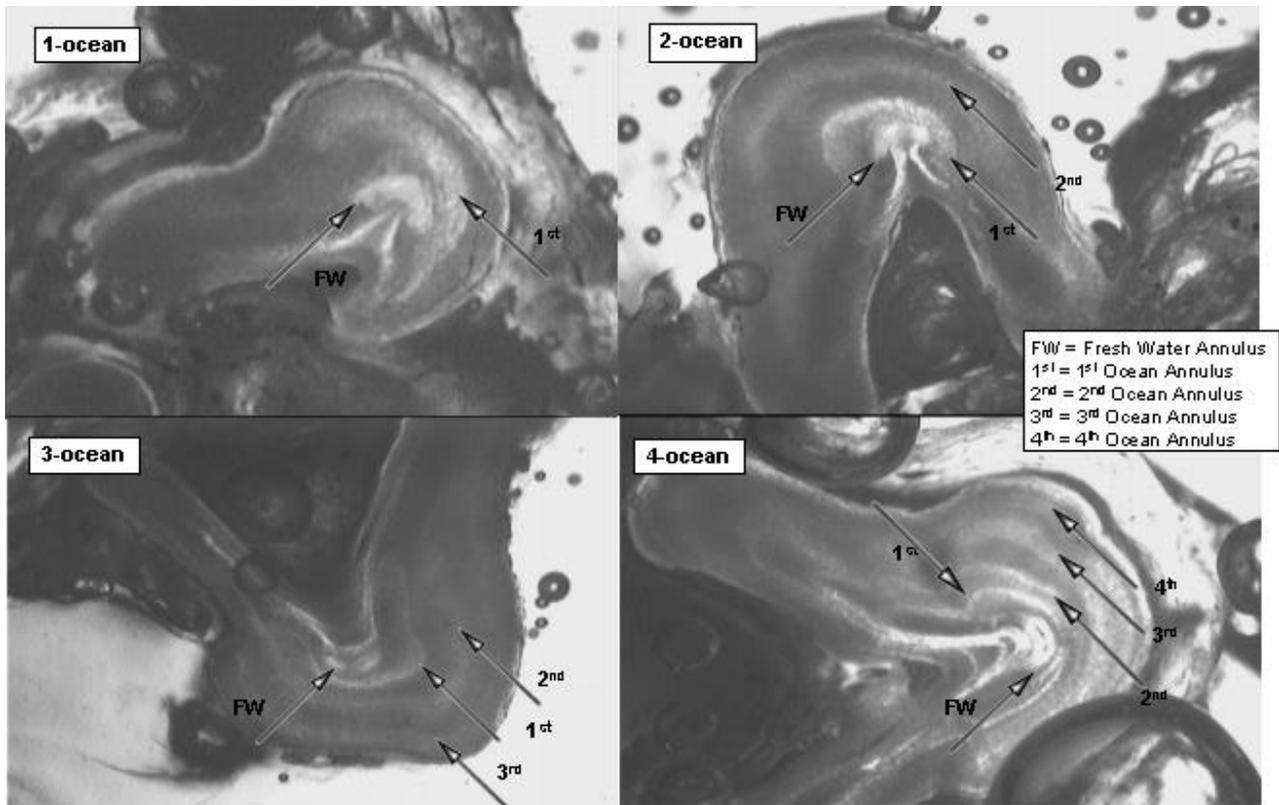


Figure 2. Representative dorsal fin ray cross sections illustrating the four different ocean ages observed for Snake River spring/summer Chinook salmon adult returns.

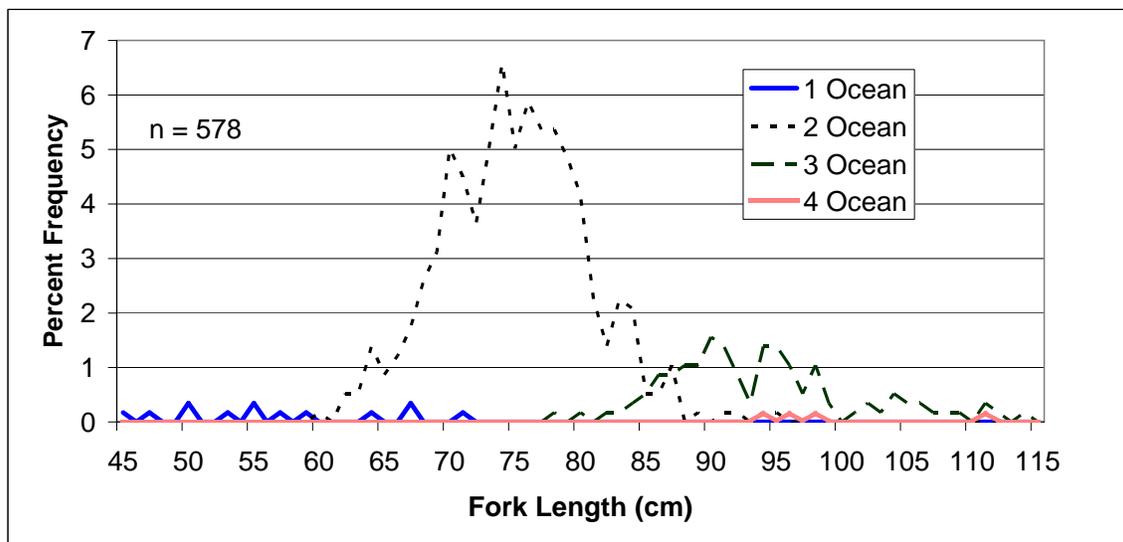


Figure 3. Length distribution by ocean age as determined from fin cross sections of wild Snake River spring/summer Chinook salmon carcasses collected in 2004.

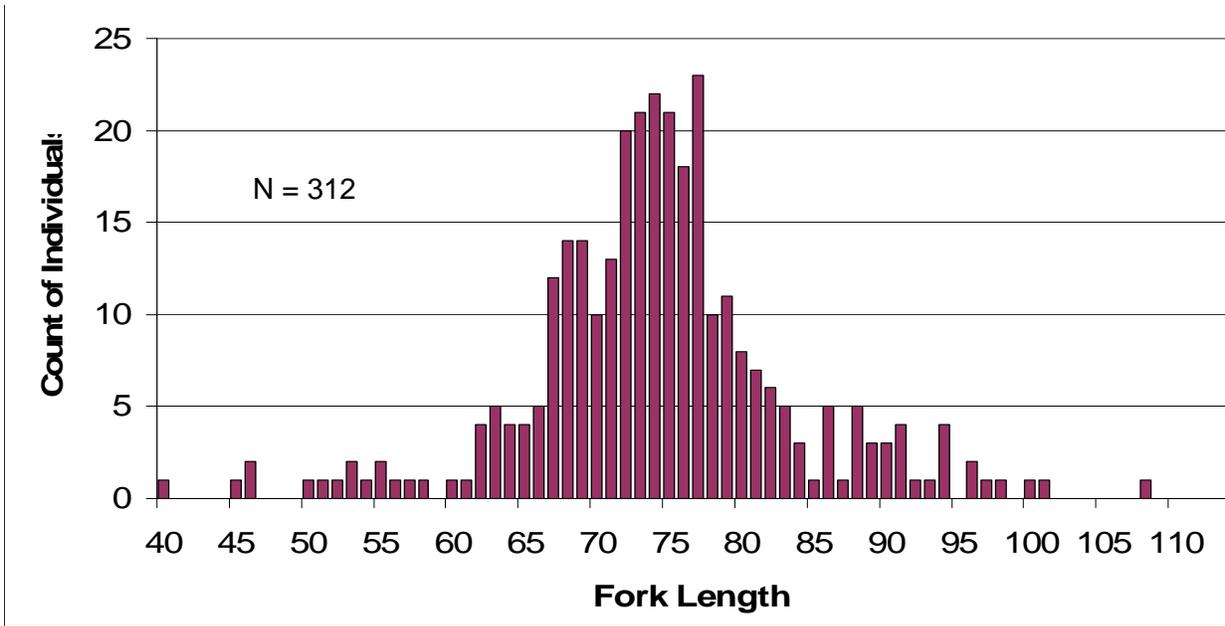


Figure 4. Length distribution (cm) of wild Snake River spring/summer Chinook crossing Lower Granite Dam in 2004 as measured from video images.

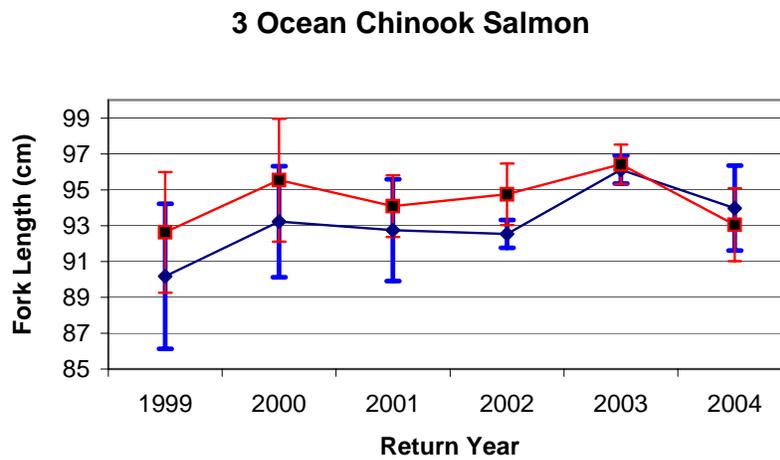
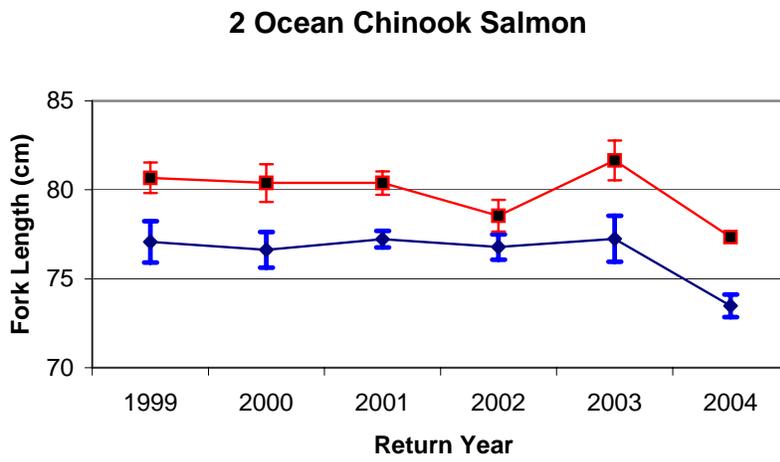
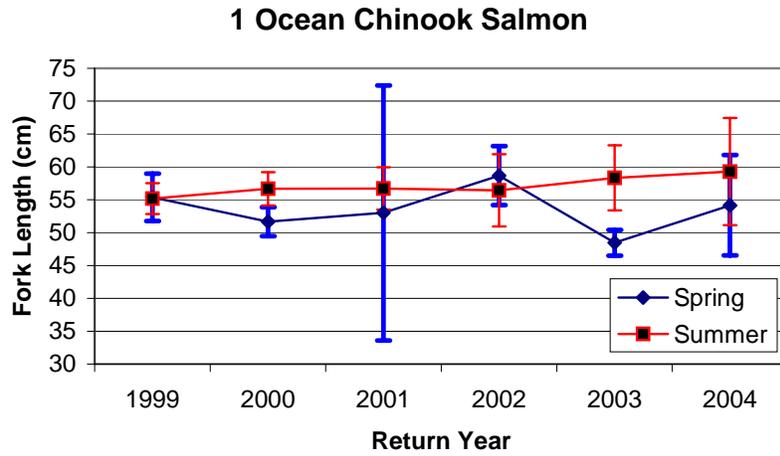


Figure 5. Mean length at age by return year of wild spring/summer Chinook carcasses. Error bars are the 95% confidence intervals.

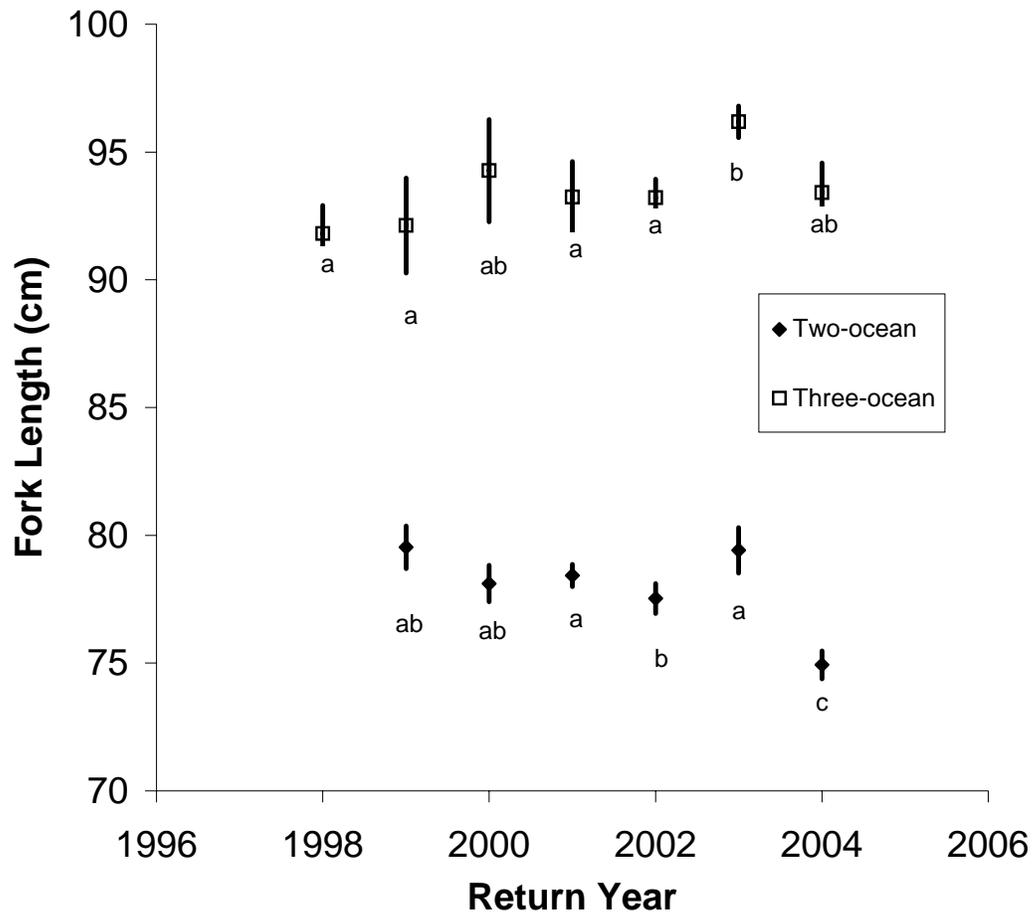


Figure 6. Adjusted mean lengths with error bars (2x standard deviation) for wild spring/summer Chinook aged as two and three ocean, 1998-2004. Years (within age group) with same letter are not significantly different.

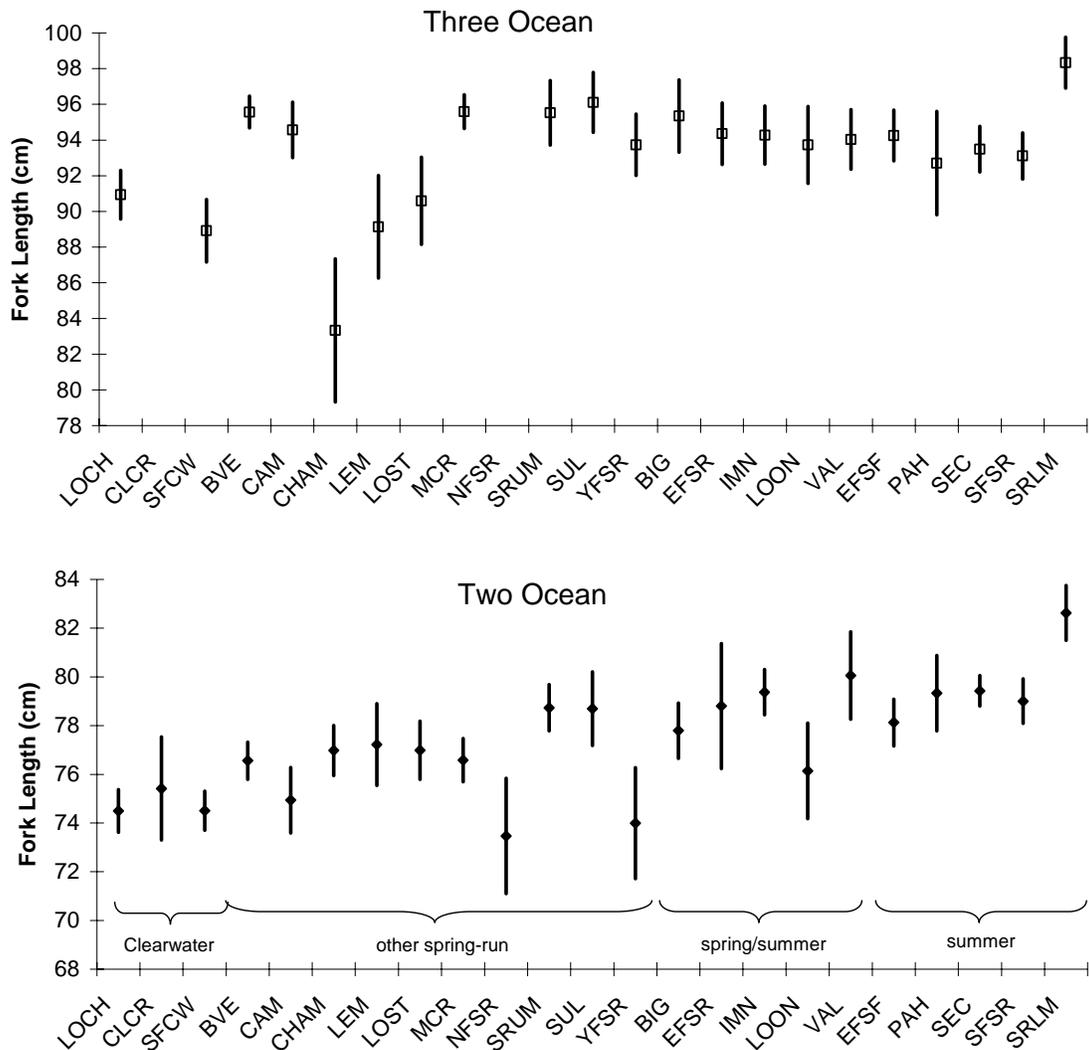


Figure 7. Adjusted mean lengths with error bars (2x standard deviation) for two- and three-ocean age groups for wild Snake River spring/summer Chinook populations (see Appendix A for abbreviations), 1998-2004. Clearwater populations are considered spring runs. The spring/summer category includes populations that contain both runs.

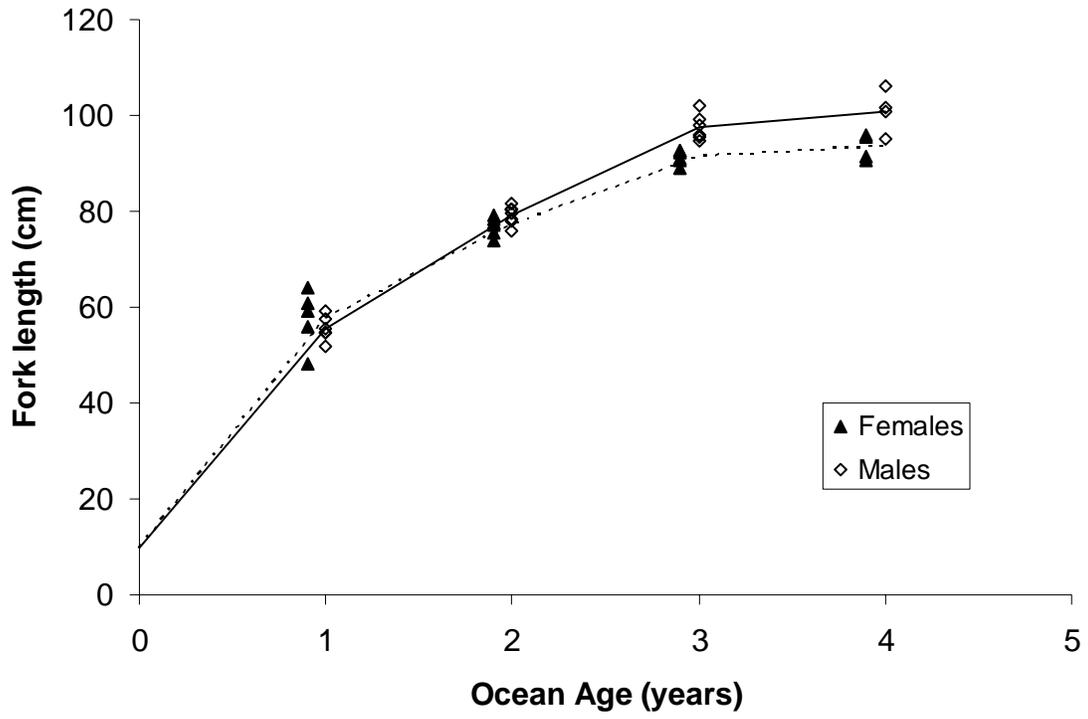


Figure 8. Annual mean lengths at ocean age by sex. Lines connect the grand means of each age. Females are offset for comparison. Smolt length (ocean age-0) was arbitrarily set at 10 cm.

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## **APPENDICES**

Appendix A. Table 1. Major population groups, populations and streams where Chinook carcasses were collected. Populations were delineated by the Interior Columbia Basin Technical Recovery Team (2005). Populations for spring Chinook in the Clearwater basin were not delineated by the TRT; we used their delineations for steelhead to define local populations in that basin. Population abbreviations given in parentheses were used in Figure 7.

| <b>CLEARWATER GROUP</b>  | <b>MIDDLE FORK SALMON RIVER GROUP</b>   |
|--|---|
| <u>South Fork Clearwater River (SFCW)</u><br>American River<br>Crooked River<br>Red River  | <u>Bear Valley Creek (BVE)</u><br>Bear Valley Creek<br>Elk Creek                          |
| <u>Lochsa River (LOCH)</u><br>Brushy Fork<br>Colt Kill Creek<br>Crooked Fork Lochsa River<br>Papoose Creek<br>Pete King Creek<br>Squaw Creek | <u>Marsh Creek (MCR)</u><br>Beaver Creek<br>Cape Horn Creek<br>Knapp Creek<br>Marsh Creek |
| <u>Lower Clearwater (CLCR)</u><br>Clear Creek  | <u>Loon Creek (LOON)</u><br>Loon Creek  |
| <b>GRAND RONDE/IMNAHA GROUP</b>  | <u>Sulphur Creek (SUL)</u><br>Sulphur Creek   |
| <u>Imnaha River (IMN)</u><br>Imnaha River  | <u>Big Creek (BIG)</u><br>Big Creek   |
| <u>Lostine River (LOST)</u><br>Lostine River   | <u>Camas Creek (CAM)</u><br>Camas Creek   |
|  | <u>Chamberlain Creek (CHAM)</u><br>Chamberlain Creek<br>West Fork Chamberlain Creek       |
| <b>SOUTH FORK SALMON RIVER GROUP</b>   |   |
| <u>Secesh River (SEC)</u><br>Grouse Creek<br>Lake Creek<br>Secesh River<br>Summit Creek  | <u>Little Salmon River (LSR)</u><br>Rapid River<br>Slate Creek                            |
| <u>East Fork South Fork Salmon River (EFSF)</u><br>Johnson Creek   | <u>South Fork Salmon River (SFSR)</u><br>Cabin Creek<br>South Fork Salmon River           |

**UPPER SALMON RIVER GROUP**

East Fork Salmon River (EFSR)

East Fork Salmon River  
Herd Creek

Salmon River Lower Mainstem (SRLM)

Salmon River (downstream of Redfish Lake Creek)

Salmon River Upper Mainstem (SRUM)

Salmon River (upstream of Redfish Lake Creek)

Valley Creek (VAL)

Valley Creek

Yankee Fork Salmon River (YFSR)

West Fork Yankee Fork Salmon River  
Yankee Fork Salmon River

Lemhi River (LEM)

Lemhi River

North Fork Salmon River (NFSR)

North Fork Salmon River

Pahsimeroi River (PAH)

Pahsimeroi River

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**PART 2—THE STOCK-RECRUITMENT RELATIONSHIP FOR NATURALLY PRODUCED  
SPRING/SUMMER CHINOOK SALMON IN THE SNAKE RIVER BASIN**

**ABSTRACT**

Stock-recruitment relationships are important for understanding how density-dependent forces affect population abundance. Previously, I fit Beverton-Holt (BH) and Ricker curves to estimates of female spring/summer Chinook salmon *Oncorhynchus tshawytscha* available for natural reproduction above Lower Granite Dam during 1990-2001 versus the number of smolts produced. Here, I updated the BH stock-recruit model with data from the 2004 smolt migration and more recent estimates of detection efficiencies at Lower Granite Dam (LGD) in 2002 and 2003. The estimate of the 2004 smolt migration was compared to predictions based on the most recent version of the BH and Ricker models. The estimated number of naturally-produced smolts migrating past LGD in 2004 was 1,502,253. The updated estimates for 2002 and 2003 were within 5% of previous values. The inverse transformation of the BH model fit the data most precisely ( $r^2 = 0.954$ ). Addition of the 2004 data did not greatly change previously estimated model parameters. The prediction for 2004 from the BH model was 7% lower than the actual estimate. The prediction based on the Ricker model was 40% higher, although within the 90% prediction interval. I concluded that the BH model was stable and offered more precise and accurate predictions compared to the Ricker model. However, the model needs additional corroboration and testing in order to make it useful for management of salmon and their habitat 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. 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 regularities of the system under consideration (Rigler 1982). A simple theoretical model is chosen to which data are fit, but stock-recruit relationships often have poor explanatory power (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). Results of such an analysis will reflect the quality of habitat in the basin (Crozier and Kennedy 1995).

Stock-recruitment relationships for Columbia River Basin Chinook salmon have been described using 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 during the juvenile stage and is asymptotic 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 smolt data from the 2004 migration and more recent estimates of detection efficiencies at Lower Granite Dam (LGD) in 2002 and 2003. The estimate of the 2004 smolt migration was compared to predictions based on the most recent version of Kiefer et al.'s (2004) BH model and a concurrently estimated Ricker function (Copeland et al. 2004).

## METHODS

Smolt production in 2004 was estimated using fish passage data collected at LGD. 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 (S. Smith, personal communication). Daily estimates were summed for the migratory year. Because the daily efficiencies for 2002 and 2003 had been re-estimated by the Northwest Fisheries Science Center, I recomputed smolt numbers for those years.

Abundance of 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). The number of females available for natural reproduction (FANR) for the brood years 1990-2002 and the number of smolts produced by brood years (BY) 1990-2001 previously were estimated by Kiefer et al. (2004) and Copeland et al. (2004). To these data, I added the smolt estimate from the 2004 migration (BY 2002) and updated the smolt estimates for BY 2000 and 2001. The stock-recruit model was re-fit using 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),  
 $\alpha$  = a fitted parameter indicative of maximum reproductive rate for the population, and  
 $\beta$  = a fitted parameter indicative of compensatory mortality as a function of stock size.

Model parameters were estimated in two 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. Following transformation, the model parameters were estimated using linear regression and are referred to as the inverse BH model below.

## RESULTS

The estimated number of smolts exiting the system via LGD during Migratory Year (MY) 2004 was 1,502,253. This estimate covered the period March 26 to September 6, 2004. I assumed these fish were produced by the 2002 spawners. Copeland et al. (2004) previously estimated FANR in brood year 2002 at 31,415. Applying this value to the estimated level of recruitment gave 48 smolts per female for BY 2002.

The estimated smolt migrations in MY 2002 and MY 2003 were 1,125,819 and 1,457,418, respectively. The previous estimate from MY 2003 (Copeland et al. 2004) was within 0.1% of the value given here; however, the previous MY 2002 smolt estimate (Copeland et al. 2004) was 4% higher than the present value.

The smolt estimate from MY 2004 completes the data for the 1990-2002 brood years. The untransformed (nonlinear) BH model fit the data very well (Figure 9;  $r^2 = 0.702$ ). The inverse BH model fit the data more precisely ( $r^2 = 0.954$ ). The nonlinear and inverse BH models produced similar parameter estimates (Table 7).

## DISCUSSION

The 2004 smolt migration (BY 2002) was the third highest in this data series (see Kiefer et al. 2004 and Copeland et al. 2004). More smolts were produced in BY 1993 and BY 1997. I concluded that the BY 2002 estimate was higher than average but within the range of recent production (after 1989). The effect of updating the estimated smolt numbers in 2002 and 2003 was barely discernible in a comparison of Figure 9 to the previous version (Figure 12a in Copeland et al. 2004). Two clusters were apparent in the aggregated data: low smolt years ranging 161,000 to 628,000 and high smolt years ranging 1,125,000 to 1,560,000. BY 1995 (the point closest to the origin in Figure 9) may be a low outlier; the next lowest production was 491,000. The current system may have two possible states, low versus high productivity.

Previously, I proposed that the production by BY 2002 and BY 2003 would provide a valuable test of model predictions and utility (Copeland et al. 2004). The previous predictions for BY 2002 were 1,401,109 by the BH model (90% prediction interval 1,162,355–1,765,523) and 2,111,978 by the Ricker model (90% prediction interval 852,743–5,230,712). The actual BY 2002 estimate was much closer to the value predicted by the BH model (Figure 10). The BH prediction was 7% lower than observed, whereas the Ricker prediction was 40% higher. However, even though the Ricker model greatly overpredicted, the observed point was not statistically distinguishable from the model prediction. I believe the BH model is more useful by virtue of its more precise predictions, although the Ricker model cannot be definitively rejected. The virtues of each may depend on scale; Milner et al. (2003) proposed that dome-shaped curves are more likely to be observed at reach scales (e.g., <50 m stream length) and flat-topped curves are more likely at tributary/river scales.

I re-estimated the parameters of the BH after including the new data point. Compared to the previous estimates (Copeland et al. 2004), the linear parameterization indicated lower intrinsic productivity (403.6 smolts/female versus 411.8 smolts/female) and a higher maximum production (1,607,300 versus 1,574,803). Model fit improved slightly ( $r^2 = 0.954$  versus 0.947). The new estimates were within the 95% confidence intervals of the old ones, so I concluded there were no significant changes to model structure. The effect of adding the new data point was to increase precision of the estimates.

A deterministic model, such as presented here, implies an equilibrium under steady-state conditions. To examine the range of equilibria possible under likely survival rates, I calculated the number of smolts required to replace the spawning population (assuming equal sex ratio in the progeny) under 1-4% SARs and overlaid these lines on the stock-recruit curves (Figure 11). Assuming a SAR of 1% and a BH function, the spawning population should stabilize near 5,000 females. At this SAR, it would be easy for stochastic events affecting survival and recruitment to drive the population to extinction; the population would not be viable at all assuming a Ricker recruitment function. At this low survival, returns greater than 5,000 would not be able to replace themselves. Equilibrium adult population increases to 13,000 at SAR = 2%, 20,000 at SAR = 3%, and 30,000 at SAR = 4%. At survival rates greater than 2%, it would take a greater catastrophe to push the population to the steep part of the recruit curve where extinction due to stochastic events becomes more likely. Additionally, the gains in equilibrium adult population accelerate as SAR rises. Despite a recent rise in SARs (see Part 1), values greater than 2% have not been observed regularly since the 1960s (Petrosky et al. 2001). Given the observed productivity of the stock and recent survival rates, there is still danger of extinction due to stochastic causes. Equilibrium predictions assume that the basic productivity and capacity of the habitat do not change in the future. Successful habitat restoration, especially in

terms of capacity, would increase the equilibrium at a given survival, while degradation would reduce it. However, it is not clear what relative level of effort would be required to shift equilibria via changes to basic system productivity/capacity versus improvements to SARs. Such an assessment would require evaluation of the present and potential habitat present throughout the Snake River basin accessible to salmon, as well as identification of likely limiting factors.

The stock-recruit data presented here have important implications for management. The Columbia Basin Fish and Wildlife Authority proposed to clarify habitat and population objectives for anadromous salmonid stocks employing a model (the All-H Analyzer) that uses BH curves and the corresponding productivity and capacity estimates (ISRP/ISAB 2005; Suzumoto 2005). Ideally, results will inform efforts to develop population-level recovery goals (Shurts 2005). The parameter estimates presented here can be used to characterize natural production in AHA for the basin aggregate stock. However, managers may want to develop plans at population levels. Aggregate stock-recruit curves can be thought of as a weighted average of many smaller curves (Wyatt and Barnard 1997a & b, cited in Milner et al. 2003). Therefore, the data presented here should be compared with similar population-specific data gathered by Idaho Supplementation Studies. This exercise would yield data allowing managers the flexibility to plan at several scales. Corroboration would validate the use of the aggregate relationship for unstudied populations, although this extrapolation should be done with caution.

In summary, the addition of the BY 2002 data corroborated the previous conclusion of density dependence within the freshwater production of Chinook salmon upstream of LGD. The BH model was stable and offered more precise and accurate predictions compared to the Ricker model. However, the model needs additional corroboration and testing in order to make it useful for on-the-ground management of salmon and their habitat in Idaho. Consideration of likely SAR values allows some prediction of equilibrium adult population, which is useful for planning.

Table 7. Comparison of estimates of the parameters of the Beverton-Holt curves ( $\alpha$ ,  $\beta$ ) from nonlinear and linear models fit with 1990-2002 data. The 95% confidence limits are in parentheses. Parametric confidence intervals were not estimable in the nonlinear model (na).

| <b>Model</b> | <b><math>\alpha</math></b> | <b><math>\beta</math></b>          |
|--------------|----------------------------|------------------------------------|
| Nonlinear    | 474.8<br>(na)              | 1,658,375<br>(na)                  |
| Inverse      | 403.6<br>(244.5–1155.2)    | 1,607,300<br>(1,402,048–1,882,955) |

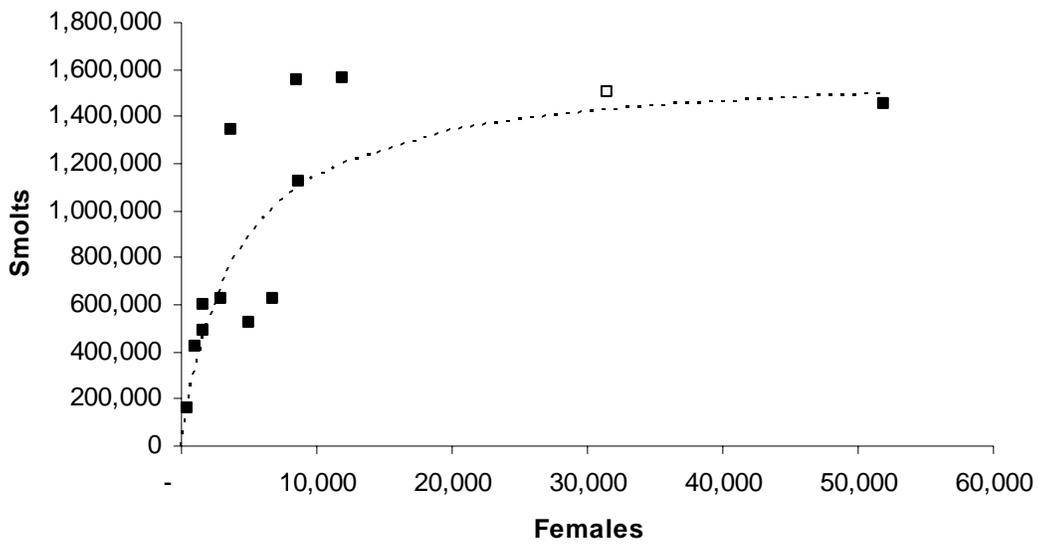


Figure 9. Comparison of observed data (BY 1990 to BY 2002) to model predictions for the linear fit of the Beverton-Holt model. The BY 2002 point is hollow.

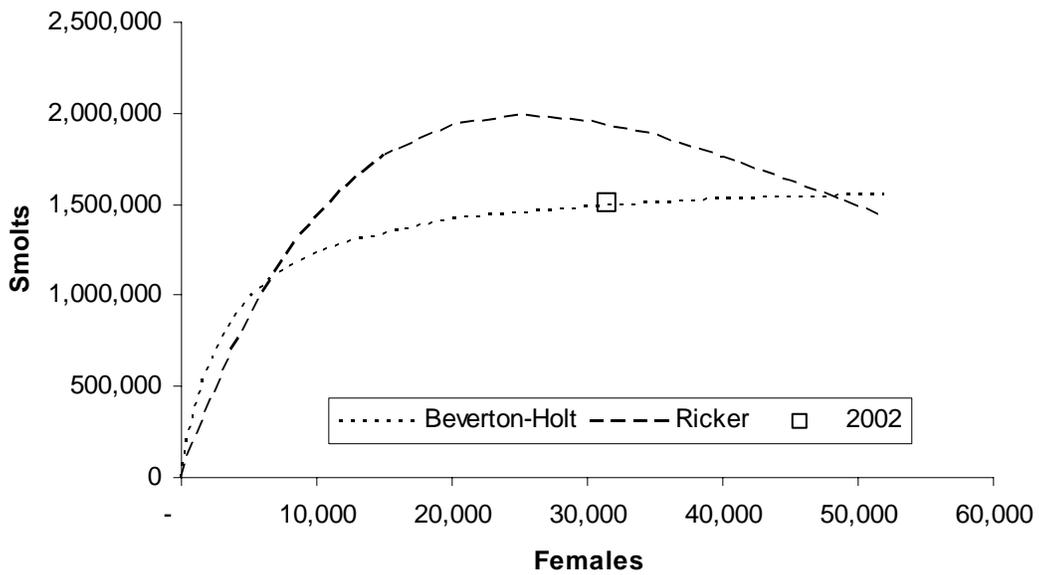


Figure 10. Comparison of the BY 2002 estimate to the Beverton-Holt and Ricker models fit by Copeland et al. (2004).

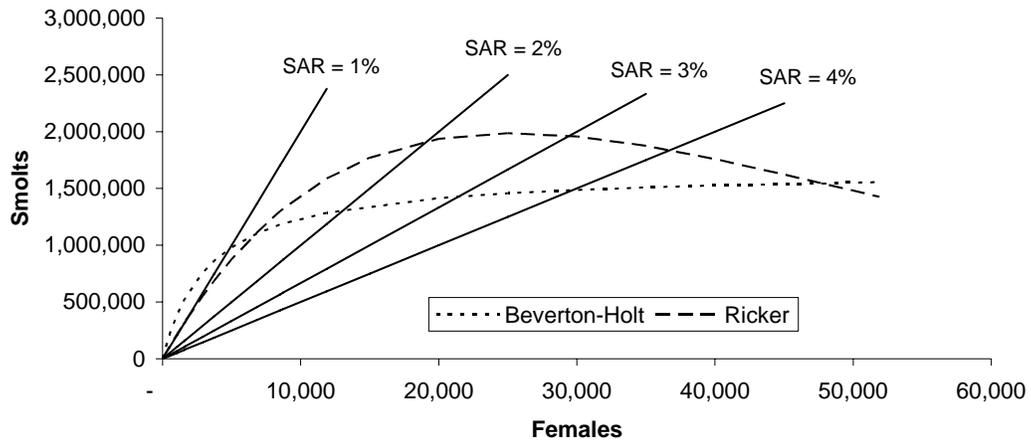


Figure 11. Number of smolts required to replace spawning females at four survival rates compared to the Beverton-Holt and Ricker stock-recruit functions previously estimated. Smolt sex ratio was assumed to be 1:1. Equilibria are located where a replacement line intersects a stock-recruit curve.

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

**ABSTRACT**

There is a need to increase the number of passive integrated transponder (PIT) tags in wild steelhead trout *Oncorhynchus mykiss* smolts in Idaho, particularly in remote drainages. In 1998, this project began work to increase the number of wild steelhead smolts available to estimate smolt-to-adult survival and hydrosystem passage mortality. Subsequently, we found that detection rates at Lower Granite Dam (LGD) and survival of PIT-tagged steelhead varied by capture method. Our objectives in this report were: 1) to summarize the results of fieldwork in 2004, and 2) to more rigorously assess the influence of tagging method on subsequent juvenile survival. Project personnel tagged 3,547 steelhead juveniles from four streams in 2004. Of the fish tagged in 2003, 13.4% were detected at LGD in 2004. Fish tagged prior to 2003 were also detected, and this percentage could be a substantial proportion of detections in some populations (maximum = 38.9%). The migration in 2004 was unimodal; consequently, differences in migration timing among populations were small. Estimated survivals to LGD of fish tagged by each method were remarkably similar among years. Averaging across years, scoop (0.89) and dip trap (0.91) survivals were higher than screw trap (0.59), angling (0.25), and electrofishing survival (0.09). Median length at capture was also lowest in electrofishing samples. Increased effort using downstream methods (dip and scoop traps) will be most effective at placing PIT tags for mainstem passage studies. Screw traps will be most effective for population-specific studies. Where fish collection is logistically difficult, we recommend angling. In conclusion, tagging effects on juvenile salmonids should be evaluated in the long term (e.g., on a seasonal basis), and methodological biases should be considered when estimating survival using PIT-tagged individuals. We recommend that tagging continue at the current sites but with better coordination with the Steelhead Supplementation Studies.

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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). PIT tags allow 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 and adults (Newman 1997; Skalski et al. 1998).

Fish passage downstream past 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 reduces sample size, thus decreasing the precision and usefulness of subsequent route-specific 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 little specific information about the populations that comprise the Snake River stock (ICBTRT 2003). In Idaho, many stream reaches used by wild steelhead for spawning and rearing 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 have low conductivity, 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 for collecting juvenile steelhead (R. 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 downstream smolt survival and SAR rates. Previously, we found that detection rates of NPM-tagged steelhead at Lower Granite Dam (LGD) was much less than for individuals tagged by other methods but that SAR apparently varied with capture method (Copeland et al. 2004). This methodological variation appeared to offset any gains in precision; therefore, we reduced tagging efforts conducted in 2004 to four streams. Our objectives in this report were two-fold: 1) to summarize the results of fieldwork in 2004 and update our previous detection estimates, and 2) to more rigorously assess the influence of tagging method on subsequent juvenile survival.

## METHODS

Fieldwork in 2004 was conducted in four streams: North Fork Moose Creek, Chamberlain Creek, Camas Creek, and its major tributary Yellowjacket Creek (Figure 12). Previous work (Copeland et al. 2004) had found these areas had high catch rates of young steelhead trout and that detection rates of fish tagged in these streams were >10% (except Yellowjacket Creek). All streams were believed to have minimal hatchery influence.

Wild steelhead trout juveniles were captured by angling with artificial flies from June through August 2004. 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 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 in a 70% alcohol for 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. Tagging data were uploaded 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 February 2005. 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) 2004. For the assessment of methodological effects (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 2004, we reported numbers tagged by stream. We also report the detections in MY 2004. The majority of these fish were tagged in 2003, but some were tagged in 2002 and 2001. Therefore, we updated all detection rates for populations previously sampled by this project. The detection rate for each stream was calculated by dividing the number detected by the total number of fish tagged in that stream in a particular year:

$$d_{ij} = \frac{\sum_{k=1}^3 D_{i,j+k}}{T_{ij}}$$

for stream  $i$  sampled in year  $j$ , where  $d$  is the detection rate,  $D$  is the number of fish detected in the hydrosystem in year  $j+k$ , and  $T$  is the number of fish tagged. Because smaller juvenile steelhead trout (<125 mm FL) often rear another year before smolting (Kieffer and Lockhart 1997), we calculated detection rates for juveniles  $\geq 125$  mm FL at tagging as well as a total rate. We also presented data on population-specific arrival times at LGD during 2004. For each of these streams, the dates were calculated when the 10<sup>th</sup>, 50<sup>th</sup>, and 90<sup>th</sup> percentile of total PIT tag detections occurred.

Our evaluation of the effects of capture method focused on juvenile detection rates for the 1999-2003 migratory years. Detections for this analysis included unique juvenile detections of individuals  $\geq 125$  mm FL at tagging from anywhere in the hydrosystem, including detectors downstream of McNary Dam. Data consisted of capture histories (detected or not detected) at the dams within the Columbia hydrosystem equipped with juvenile PIT tag detection systems. Care was taken to ensure that detections were from juvenile and not adult detectors. Data were

grouped by capture method (dip trap, scoop trap, angling, electrofishing, and screw trap) and by MY. Capture histories were entered into a Cormack-Jolly-Seber mark-recapture model (SURPH 2.2a; Lady et al. 2004) to estimate method- and year-specific probabilities of survival and detection.

## RESULTS

The NPM crew conducted tagging operations during July 2004. A total of 3,547 PIT tags were placed in wild *O. mykiss* juveniles. In Chamberlain Creek, 1,324 fish were tagged. There was one individual captured that was tagged in a previous year. There were 781 fish tagged in North Fork Moose Creek. Fifty previously-tagged individuals were also captured. In Camas Creek, 930 individuals were tagged. Only seven were recaptures from previous years. Lastly, 481 fish from Yellowjacket Creek were tagged. Twenty-nine fish were recaptures.

Juvenile detections for NPM-tagged fish in 2004 totaled 1,289. Overall detection rate of fish tagged in 2003 was 13.4%. Some individuals (19%) detected in 2004 were tagged prior to 2003. Of these, 13 were tagged in 2001 and 227 were tagged in 2002. We used the latter values to update detection rates we previously estimated (Copeland et al. 2004). Median detection rate was 17.4% for fish tagged in 2002 and 14.7% for fish tagged in 2003 (Table 8). Rates were about 2% higher if only individuals 125-249 mm FL were considered. For the tagging done in 2002, the rate of delayed detections (holdovers) ranged from 0% in Slate Creek to 38.9% in Brushy Fork Creek (all lengths considered).

The 2004 juvenile migration was unimodal in timing (Figure 13). The migratory peak was May 8. The timing of individual populations was extremely consistent during 2004; median arrival dates ranged from May 2 to May 10 (Figure 14). Only 16 detections were recorded prior to April 12. After May 24, emigrant numbers quickly dropped, and only 17 tagged individuals were detected. There was more variability among populations about the beginning and end of the migration; the 10<sup>th</sup> percentile dates ranged from April 14–May 4, and 90<sup>th</sup> percentile dates ranged from May 7-23.

Probability of survival to LGD was different among capture methods but strikingly similar among years for each method (Figure 15). Averaging across years, mean scoop and dip trap survivals were highest, 0.89 and 0.91, respectively. Mean screw trap survival was 0.59 and mean angling survival was 0.25. Mean electrofishing survival was lowest, 0.09. Arrival timing at LGD was similar for fish tagged by all methods (Table 9).

## DISCUSSION

Detection rates of tagged steelhead juveniles varied among populations. Overall detection rate for MY 2004 was lower than previously observed (14%; Kiefer et al. 2002). However, not all populations have been sampled every year; therefore, annual changes in the aggregate detection rate could be confounded by population differences. Fish tagged in Whitebird Creek in 2003 had the lowest detection rate, while detection rate was highest in Camas Creek. Therefore, relative rank of populations in terms of detection rate did not change much between MY 2003 and MY 2004 for the first potential year for migration (see Copeland et al. 2004 for comparison). We detected a number of fish that held over past their anticipated migratory year (i.e., angled and tagged in 2002 but migrated in 2004). These fish were an

important component of the total detections from Brushy Fork and North Fork Moose Creek. Byrne (2005) found that steelhead populations in the Clearwater drainage tend to take longer to smolt than in the Salmon drainage. However, holdovers were a sizable minority of many populations in the Salmon drainage.

Migration timing did not vary greatly among populations in 2004. Overall, the migration was temporally constrained, forcing all populations to migrate together. Initiation of migration was earlier for Brushy Fork and North Fork Moose Creek (Figure 14), similar to our previous findings (Copeland et al. 2004), but other population differences were likely masked. Previously, a multimodal distribution was evident when several populations were considered together (e.g., Copeland et al. 2004; Mensik et al. undated). Occurrence of a unimodal distribution may be caused by poor snow pack and low flows in early spring 2004.

### **Survival by Capture Method**

The effect of capture method on survival of juvenile steelhead to LGD was fairly constant among years. Most of the observed annual variation in detection rates was due to variations in detection probability at the dam. We concluded that the survival differences we surmised previously (Copeland et al. 2004) were real. Note that this was apparent survival because survival of residualized, resident, or delayed migrants was not measured, i.e. survival was implicitly defined as arrival at LGD. We further assumed that there was no geographic or temporal pattern in site biases that may be generated as a result of sampling different populations with each capture method. The latter part of this assumption seems justified because of the broad overlap in migration timing of fish tagged by the various methods (Table 9).

Several mechanisms can explain differences in survival by capture method. Length frequencies of the fish tagged by each method differ (Figure 16). Differences in length at tagging could be related to differences in survival, but we controlled this by only analyzing survival for fish 125-249 mm FL. There were differences in methods regarding distance from tagging locations from LGD. Distance from dip traps from LGD ranged 52-61 rkm. Scoop trap sites were further upstream (100-747 rkm). Field locations sampled by the other methods fell within this range. Dip traps and scoop traps sample from larger rivers downstream of major production areas, while the other methods typically sample smaller channels just below or within production reaches. Survival may be higher because there is less distance to travel to LGD. The methods differ in how they collect fish. Dip traps, scoop traps, and screw traps are passive gears that collect fish as they move downstream through the sample site. The process of smoltification involves behavioral changes including downstream orientation (Quinn 2005, p. 222). Hence, passive gears were more likely to collect individuals that have begun smoltification, or are more likely to smolt in the near future, by virtue of gear *modus operandi* and position relative to rearing areas. Dip traps and scoop traps were operated only during the spring migration period (March-June) when flows are higher and travel time to LGD shorter. Screw traps typically operate spring through fall. In contrast, angling and electrofishing take place within rearing areas and collect fish regardless of their potential for migration, such as individuals likely to hold over more than one year or never migrate. These latter methods typically were employed during summer base flow conditions. Lastly, handling by the gears may cause delayed mortality. Evaluations of tagging effects typically consider only short-term mortality (e.g., 24 h). Delayed effects from dip traps may not be seen until fish have passed LGD. Electrofishing is injurious to salmonids, particularly in low conductivity waters typical of headwater streams (Hollender and Carline 1994). The distribution of lengths tagged by seining was similar to length distribution of electrofished individuals (data not shown); therefore, analysis of survival by fish tagged via

seining will help separate electrofishing injury effects from size-selective survival effects. Regardless of cause, the consistently low survival of fish tagged by electrofishing show that long-term survival effects may bias conclusions based on these individuals.

## **RECOMMENDATIONS**

The optimal method for increasing the number of PIT-tagged fish will depend on study objectives. Increased effort using downstream methods (dip and scoop traps) will be most effective at placing PIT tags for mainstem passage studies. Screw traps will be most effective for population-specific studies. Where fish collection is difficult logistically, we recommend angling. Tagging effects on juvenile salmonids should be evaluated in the long term and methodological biases should be considered when estimating survival using PIT-tagged individuals (e.g., by including gear type as a variable in models estimating survival).

NPM efforts have resulted in information on previously unsampled steelhead populations, especially in the Salmon basin. These data provide a valuable complement to data from the Steelhead Supplementation Studies (SSS), which has greater emphasis in the Clearwater basin. However, populations sampled by NPM changed frequently, which impairs the utility of the data. The currently sampled sites now have several seasons of data collected. We recommend this limited effort continue, but with better coordination with the SSS fieldwork.

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Table 8. Stream-specific detection rates (%) for steelhead parr tagged by this project 2001-2003 and percent of fish tagged in 2002 that migrated in 2004 (holdovers). Estimates from previous reports (2001 and 2002) were updated with detections from 2004. Data for Chamberlain Creek included fish tagged in West Fork Chamberlain Creek.

| Stream             | All fish |      |      | Fish 125 – 249 mm FL |      |      | Holdovers (%) |
|--------------------|----------|------|------|----------------------|------|------|---------------|
|                    | 2001     | 2002 | 2003 | 2001                 | 2002 | 2003 |               |
| Bargamin Creek     | 7.8      | 11.7 | 13.4 | 8.1                  | 12.9 | 15.7 | 22.0          |
| Big Creek          |          | 20.6 |      |                      | 22.6 |      | 11.8          |
| Brushy Fork Creek  | 22.3     | 23.4 |      | 23.8                 | 24.8 |      | 38.9          |
| Camas Creek        |          | 23.7 | 22.2 |                      | 25.3 | 23.8 | 8.2           |
| Chamberlain Creek  | 15.4     | 17.3 | 11.4 | 19.1                 | 19.5 | 15.2 | 14.3          |
| Horse Creek        | 11.7     | 17.4 | 13.9 | 13.1                 | 19.6 | 16.5 | 16.3          |
| Loon Creek         |          | 21.6 |      |                      | 23.3 |      | 20.7          |
| MF Salmon River    |          | 18.5 | 17.5 |                      | 23.1 | 22.5 | 27.3          |
| NF Moose Creek     | 22.7     | 16.1 | 17.7 | 23.3                 | 16.8 | 20.6 | 35.7          |
| Rapid River        |          |      | 16   |                      |      | 18.5 |               |
| Slate Creek        | 5.7      | 6.5  |      | 6.6                  | 7.7  |      | 0.0           |
| Sulphur Creek      | 18.2     | 19.8 | 12.2 | 20.8                 | 26.0 | 16.4 | 27.1          |
| Whitebird Creek    |          | 6.1  | 4.1  |                      | 5.5  | 3.6  | 1.7           |
| Yellowjacket Creek |          | 8.3  | 15.4 |                      | 9.7  | 16.6 | 11.1          |

Table 9. Arrival timing of PIT-tagged steelhead smolts at Lower Granite dam by tagging method and migratory year. Ranges given were the 10<sup>th</sup> and 90<sup>th</sup> percentile arrival dates.

| Method         | Migratory Year |           |           |           |           |
|----------------|----------------|-----------|-----------|-----------|-----------|
|                | 1999           | 2000      | 2001      | 2002      | 2003      |
| Angling        | 4/22-5/24      | 4/16-5/11 | 4/29-5/16 | 4/16-5/22 | 4/17-5/21 |
| Dip trap       | 4/22-5/25      | 4/19-5/21 | 5/4-5/20  | 4/17-5/25 | 4/20-5/27 |
| Electrofishing | 4/23-5/26      | 4/14-5/16 | 5/2-5/20  | 4/15-5/24 | 4/23-5/29 |
| Scoop trap     | 4/22-5/26      | 4/14-5/15 | 5/1-5/21  | 4/15-5/31 | 4/21-6/1  |
| Screw trap     | 4/22-5/25      | 4/15-5/22 | 4/29-5/18 | 4/16-5/25 | 4/18-5/26 |

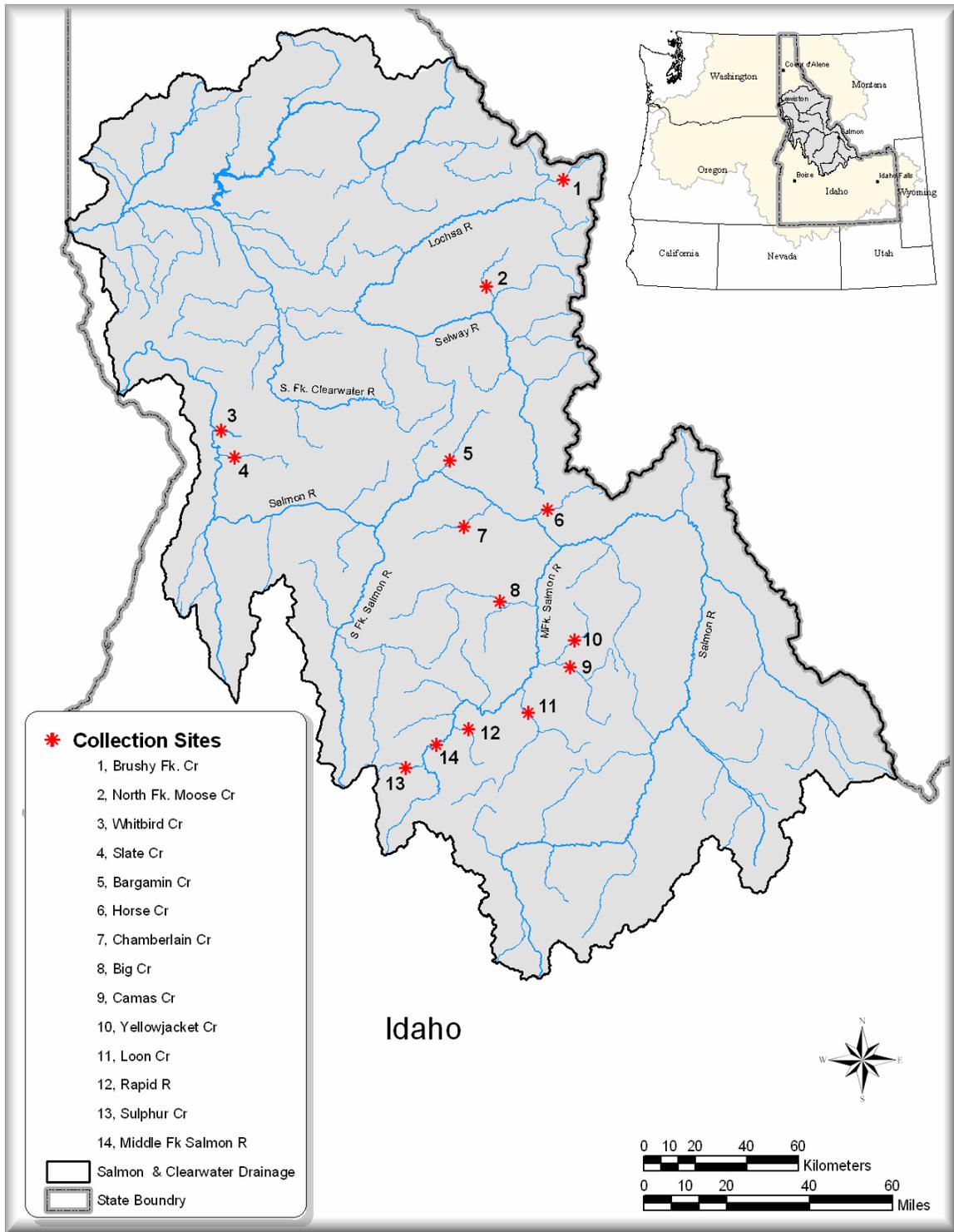


Figure 12. A map of the Clearwater and Salmon River drainages. Locations where NPM personnel collected steelhead parr by angling in 2002-2004 are indicated by asterisks.

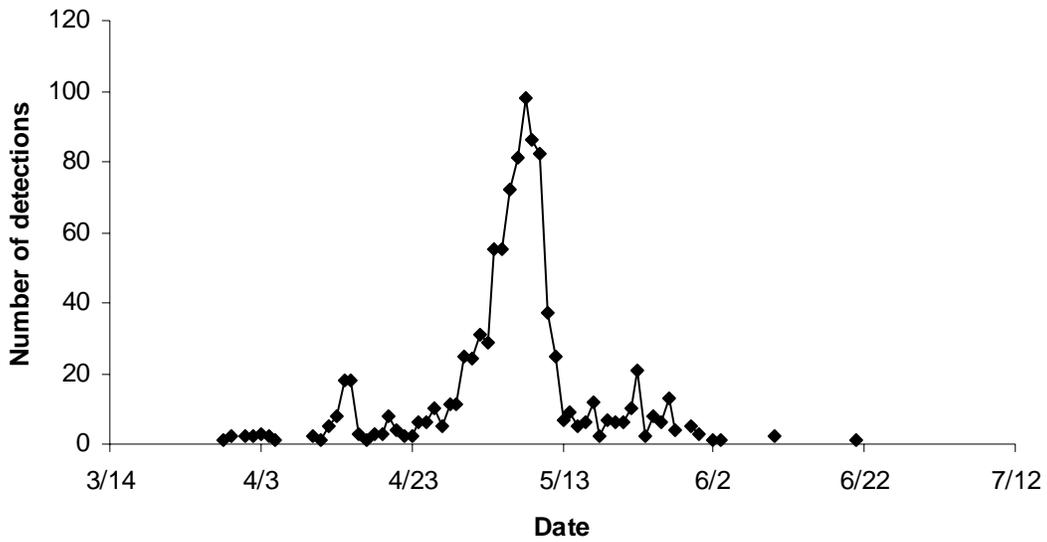


Figure 13. Date of arrival at Lower Granite Dam in 2004 of steelhead smolts tagged by project personnel.

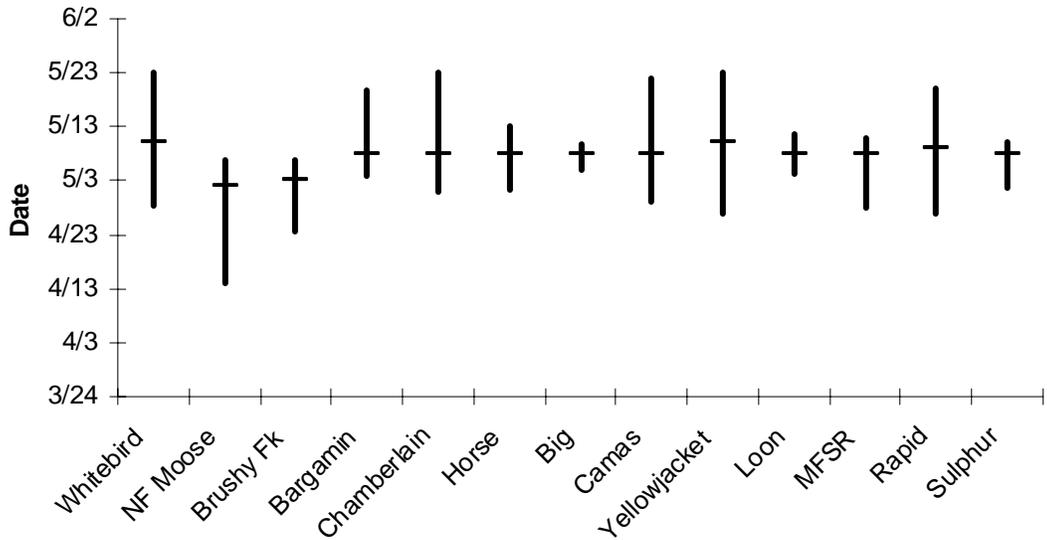


Figure 14. Population-specific arrival dates at Lower Granite Dam in 2004. The vertical bar spans the 10<sup>th</sup> and 90<sup>th</sup> percentiles. The hash mark denotes the median arrival date. Streams are arranged from left to right by distance from Lower Granite Dam. MFSR = Middle Fork Salmon River.

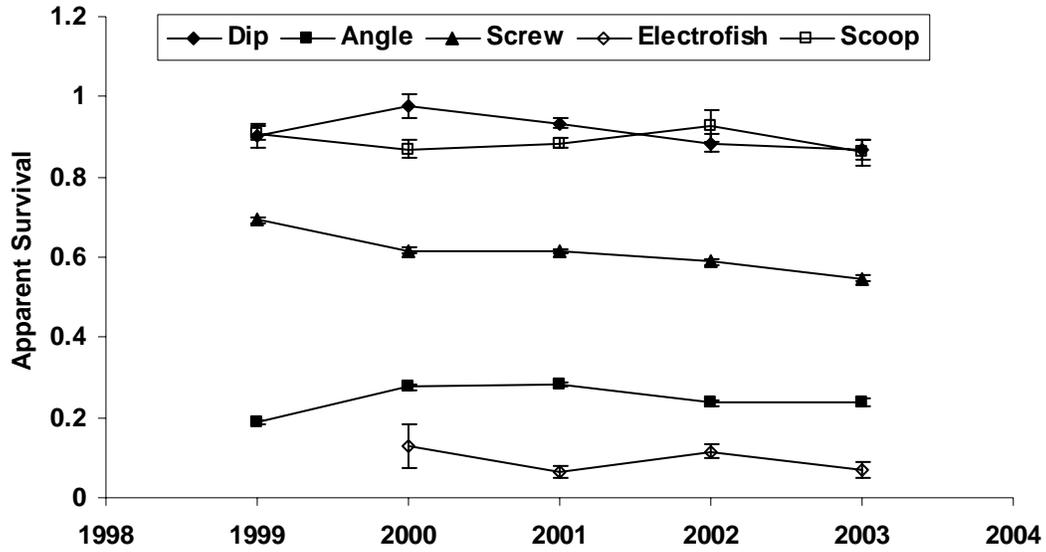


Figure 15. Probability of apparent survival to Lower Granite Dam by capture method and migratory year for PIT-tagged steelhead smolts. Error bars are plus and minus one standard deviation.

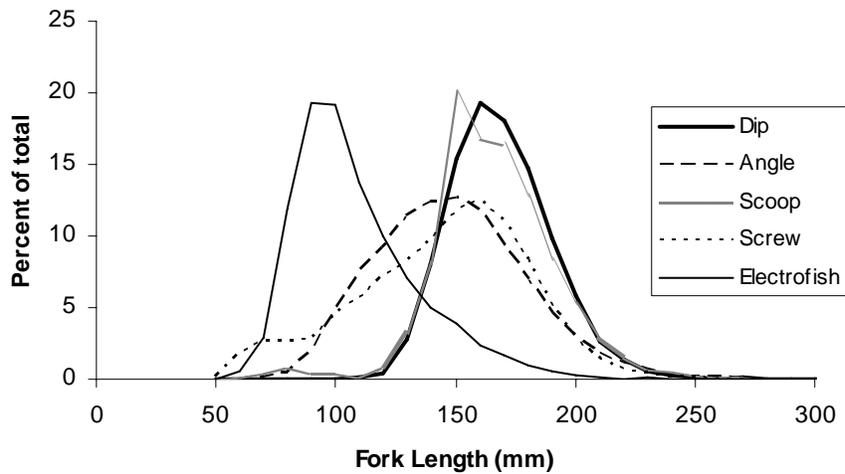


Figure 16. Length at tagging by method summed for migratory years 1998-2003.

**PART 4—MONITORING TRENDS IN ABUNDANCE OF ANADROMOUS SALMONID PARR  
IN IDAHO  
ABSTRACT**

Abundance of juvenile anadromous salmonids has been monitored in central Idaho since 1985. These data have been valuable in assessing regional population trends. However, sample sites were chosen subjectively and hence are not suitable for estimating total population abundance. The primary purpose of this analysis was to identify sites with the best series of trend data for Chinook salmon *Oncorhynchus tshawytscha* for retention in a probabilistic sampling design. Secondly, I wanted to verify the time trend in the data and identify important factors influencing the ability to detect this trend. Densities of age-0 Chinook parr were lowest during 1995-1997, then rebounded. This qualitative trend was apparent at all spatial scales and in the percentage of sites where Chinook parr were observed (frequency of occurrence). Stream width, channel type, water temperature, and visibility all significantly influenced the probability that Chinook parr were observed at a site ( $|t| > 2.75$ ,  $P < 0.006$ ). Seventy-two sites had adequate data for trend detection. Only 12 sites had a time-trend slope not significantly different from the overall trend during both the period of decline (1985-1997) and recovery (1995-2003). Regression slopes appeared greater in more pristine drainages. Because of the large amount of data, regional trend detection should be quite powerful. However, for finer-scale analyses, site-level biases must be assessed, especially in regards to the detection probability of parr. Potential trend sites should be verified against independent data sets, e.g., local redd counts.

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## INTRODUCTION

The Idaho Department of Fish and Game (IDFG) has monitored abundance of juvenile anadromous salmonids since 1985 as part of the Idaho Natural Production Monitoring and Evaluation Project (NPM). This part of NPM is a long-term, general parr monitoring (GPM) program covering a broad geographic area. The original intent of the project was to track natural production of anadromous salmonids and so verify the effectiveness of off-site mitigation projects funded by the Bonneville Power Administration (Petrosky and Holubetz 1985). Since the listing of Idaho's anadromous salmonid stocks under the Endangered Species Act (ESA) in the early 1990s, GPM focus shifted to tracking population trend. A formal sampling plan was developed in 1994 to coordinate and prioritize fieldwork for the purpose of developing trend information (Leitzinger and Holubetz 1994, cited in Hall-Griswold et al. 1995). Essentially, this plan designates 1) a core set of sites to sample annually, 2) sites sampled by other projects but producing data suitable for GPM, and 3) sites sampled occasionally or opportunistically. Observations of all species encountered were recorded. As a result, the GPM database is the most comprehensive source of information on all salmonid species in central Idaho streams. GPM data were used by various agencies to support or dispute ESA listing decisions for Snake River spring/summer Chinook salmon *Oncorhynchus tshawytscha*, summer steelhead trout *O. mykiss gairdneri*, westslope cutthroat trout *O. clarkii lewisi*, and bull trout *Salvelinus confluentus*. The Clearwater Subbasin Inventory (NPCC 2003a; p. 60) mentioned GPM data as the most requested information by other agencies and consultants. Clearly, the data generated by GPM are of great utility; however, they are not optimal for all purposes.

Recovery planning requires data on population abundance (NMFS 2000). At first view, the GPM data appear well suited for such a purpose. However, sites often were selected subjectively (Hall-Griswold and Petrosky 2002), and the program's focus has been at regional rather than population scale. The Independent Scientific Review Panel and other entities have stressed the need for population-level investigations that incorporate probabilistic sampling. Further, such data must be linked to historical trend data sets in order to make valid inferences about unsampled areas. NPM is uniquely positioned to fulfill this need. However, current data should be examined and rigorously evaluated before such a program can be designed.

My primary objective in this report was to identify a set of GPM sites with the best series of trend data. The intent was to retain these sites in a rotating-panel probabilistic sampling design (*sensu* Larsen et al. 2001) for future use. A subset of the current core sites will be retained to be sampled annually. The remaining sites will be evenly distributed between a five-year panel and stratified random sampling. This analysis focused on observations of Chinook salmon parr. Once the site selection process is clearly defined and verified, it can be repeated for other species. My secondary objectives were to verify the time trend present in the data and to identify important influences on the ability to detect this trend. The purpose of the former was to clearly define the standard for comparison, while the latter bears on the need for modification to data collection and analysis.

## METHODS

### Data Description

IDFG biologists sampled streams widely in central Idaho during the period 1985-2003 (Figure 17). Most of the pertinent fieldwork involved snorkel surveys (see Petrosky and Holubetz

1986 for methodology). Sites were snorkeled by divers working upstream, with the number of divers varying according to stream width and visibility. Observers snorkeled slowly, identifying all fish seen and counting salmonids observed by length or age category. For this analysis, I used counts of age-0 Chinook salmon parr (45-99 mm FL; Scully et al. 1990). Use of age-0 observations removed the possibility of counting the same fish in different years so that consecutive years were independent. Counts were standardized to number per 100 m<sup>2</sup> for comparison among streams of different size. Sampling events without records of area sampled were dropped from the data set.

Additional explanatory information was associated with each sampling event. Stream channel type (Rosgen 1985) at each site was determined. Sites sampled were defined as either a B channel (moderate gradient, mostly riffles, narrow valley) or C channel (low gradient, meandering thalweg, riffle/pool morphology, alluvial valley). Average wetted width (m), water temperature (°C), and visibility (m) were recorded during sampling events.

### **Trend Exploratory Analyses**

Before site contributions toward trend detection can be considered, the trend must be identified and verified. I took two complementary approaches towards this objective. The first approach was to examine the generality of any trend that might be apparent. Parr densities were plotted over time for the entire data set. However, zero observations were frequent; in some years, these comprised the majority of observations. Because parametric statistics such as means are biased estimators under such circumstances, I used the 75<sup>th</sup> and 90<sup>th</sup> percentiles of each year. The trend appeared curvilinear, so polynomial regression was used to describe the trend visually. I repeated this analysis by subbasin (Salmon versus Clearwater) and by 4<sup>th</sup> field hydrologic unit code (HUC4). Four tributaries to the Snake River in Hells Canyon were excluded from the subbasin analysis but included in the HUC4 analysis.

The decision to use percentiles was somewhat subjective. To verify the observed trend in parr density percentiles another way, I examined the percentage of sites sampled each year where at least one age-0 Chinook parr was observed. Polynomial regression was used again to describe the trend visually.

### **Influences on Detection Probability of Chinook Parr**

Previous analyses of GPM data implicitly assumed that the probability of detection of parr was 100%. This is rarely true, if ever, and the use of uncorrected counts may cause biased conclusions (Thurow et al. in press). For example, the Idaho Supplementation Studies ceased snorkel surveys because summer parr population estimates were imprecise and often less than the number of fall emigrants counted at screw traps (Leitzinger et al. 1996). Therefore, it is important to identify major influences on probability of detection. I conducted a post-hoc analysis on the GPM data using logistic regression. The dependent variable was the probability of detecting age-0 Chinook parr, which should avoid the gross problems associated with the effect of zero observations on density estimates. Model selection procedures were used to examine the influence of channel type, stream width, water temperature, and visibility. Sampling events where not all of these variables were recorded were dropped from the analysis. The base model incorporated a year and a quadratic term to account for the curvilinear trend in time. All combinations of the other regressors were considered, including two-way interactions. Stream width and visibility appeared non-normally distributed so those values were log-transformed

before entering into the model. I used bivariate plots to depict the main effects visually (percentage of sites where Chinook parr were detected versus year). To do this, I separated the continuously distributed regressors into high, low, and medium categories as delimited by the 25<sup>th</sup> and 75<sup>th</sup> percentiles. Models were evaluated using Akaike's Information Criterion (AIC) and compared to each other relative to the lowest AIC value of the set under consideration (Franklin et al. 2001). AIC was calculated as:

$$AIC = -2 \ln L + 2K$$

and model weights ( $w_i$ ) as:

$$w_i = \frac{\exp(-\Delta_i/2)}{\sum_{m=1}^M \exp(-\Delta_i/2)}$$

where  $L$  is the model likelihood,  $K$  is the number of parameters (including variance),  $\Delta_i$  is the difference between the AIC value of model  $i$  and the minimum AIC, and  $M$  is the number of models in the candidate set.

The full set of models was evaluated to determine the model with greatest support. The effect of each variable was considered by determining the most likely model including only year effects and one other variable and by dropping each variable in turn from the full model (no interactions).

### **Trend Site Selection**

The purpose of this analysis was to determine the set of sites that yielded the best trend information. Potential trend sites were defined as having at least two sampling events in the first and last five years of the data series (1985-1989 and 1999-2003). Sites at which no Chinook parr were ever detected were eliminated.

Candidate sites were evaluated using three criteria. First, I regressed observed parr density as a quadratic function of year for each site. Sites without a statistically significant relationship were removed from consideration. Given the nature of the data and the emphasis on detecting a relationship, I set  $\alpha = 0.10$  for this and the other filters. Because a site may have a significant quadratic regression and still not track the overall trend, two other criteria were used. The lowest Chinook parr abundance was during the period 1995-1997. A piecewise regression approach was used to examine the trends on either side of this period. Parr density was regressed against year for the period 1985-1997 and again for 1995-2003. The regressions on the 75<sup>th</sup> percentile values over all sites for these periods were used as the standard for comparison. Sites were eliminated if the regression was not significant ( $\alpha = 0.10$ ) or if the 95% confidence interval of the year coefficient did not overlap that of the standard. The final list of sites met the criteria for both periods.

## RESULTS

### Trend Exploration

The GPM database had 8,654 observations. Deleting records without data for area left 8,298 observations from 2,141 sites on 219 streams during 1985-2003. Sampling intensity increased in the early 1990s (Table 10). There were more sites in the Salmon River drainage than in the Clearwater drainage; the database also included a handful in Snake River tributaries in Hells Canyon.

The overall trend in densities of age-0 Chinook parr was parabolic, reaching a low in the mid 1990s (Figure 18). The median trend was flat because there were so many zero observations in all years; over half of the observations were zeros in seven of 19 years. The 75<sup>th</sup> and 90<sup>th</sup> percentiles were more sensitive indicators of trend. I reported all other results as 75<sup>th</sup> percentile densities.

The density trend was apparent at several spatial scales. The basic trend was similar in both the Clearwater and Salmon subbasins, although densities were higher in the Clearwater in the late 1980s (Figure 19). The pattern continued at the smaller HUC4 scale as well, although the relative magnitude of the decline and recovery varied among HUC4s (e.g., Figure 20; see Appendix B for all HUC4 results). In general, the trend was most dramatic in wilderness drainages and flatter in more developed drainages.

The trend in frequency of occurrence of Chinook parr mirrored that in the density data (Figure 21). Because much of the Lower Clearwater HUC4, Panther Creek, and the Hells Canyon tributaries do not support spawning populations, I excluded those sites from Figure 21. The exclusions did not change the trend but shifted it up slightly. For instance, the lowest annual percentage of sites where Chinook parr were observed was in 1996. With all the data, only 15% of all the 1996 sites had Chinook, whereas 19% of the pared-down list of sites had Chinook parr in 1996.

### Influences on Detection Probability of Chinook Parr

For this analysis, I excluded all observations from the previous set that did not have a complete record of the explanatory variables (channel type, visibility, width, temperature). This left 2,437 observations recorded from 1986 to 2003. Within these data, the relationship of parr density to the continuous variables was right-skewed in width and visibility and dome-shaped for temperature (Figure 22). Therefore, I log-transformed width and visibility.

All variables in the full logistic model were significant ( $|t| > 2.75$ ,  $P < 0.006$ ), which was not surprising given the large number of observations. However, the model had poor fit (McFaddens  $\rho^2 = 11.7\%$ ). The model correctly predicted 58.1% of the observations. Sensitivity was 62.5% and specificity was 52.5%; therefore, the full model predicted absences slightly better than the presences. Inclusion of the interactions considered did not increase the percentage of observations correctly predicted.

I considered the strength of evidence for the null model (no effects), the base model (year effects only), the full model, and all subsets of the full model that included year effects. In total, 17 models were compared. The full model was the most likely (AIC weight = 94.6%). The only other model with an AIC weight  $> 1\%$  was the full model less visibility (AIC weight = 5.3%). I

compared the models that added a single variable to the year effect and those that subtracted a single variable from the full model. Within each comparison, only one model had an AIC weight >1%. The best variable to add was width; the best to subtract was visibility.

To visually illustrate the influence of each variable, I grouped sites based on the 25<sup>th</sup> and 75<sup>th</sup> percentiles for each continuous variable (Table 11) and by channel type. The effect of width was apparent; percentage of sites with a parr observed was lowest for the narrowest group of sites (Figure 23, top panel). Percentage of detections was lowest in B channel sites (Figure 24). The percentage of detections was lowest at low temperatures (Figure 23, middle panel). Visibility did not have a consistent or easily distinguishable effect (Figure 23, bottom panel).

### **Trend Site Selection**

The purpose of this exercise was to identify a subset of sites that had trend information over time. The majority of sites was eliminated for not having a sufficient time series of data. Of the remainder, I removed five from consideration because no Chinook parr had ever been observed there. The remaining set included 209 sites on 72 streams.

To describe the trend present in those data, I plotted the 75<sup>th</sup> percentile observations by year for all 209 sites (Figure 25). I used these data as the standard for comparison in order to identify the sites contributing the most information towards this trend. The fit of the quadratic regression was significant ( $F = 5.910$ ,  $P = 0.012$ ;  $R^2 = 42.5\%$ ). The linear regression of the 1985-1997 series was significant ( $F = 10.300$ ,  $P = 0.008$ ;  $r^2 = 48.4\%$ ), as was that of the 1995-2003 series ( $F = 9.302$ ,  $P = 0.019$ ;  $r^2 = 57.1\%$ ). The slopes of those regressions showed the magnitude of recovery was similar to that of the decline (Table 12). These values were used to evaluate individual sites for inclusion in the candidate trend set.

Many sites did not exhibit a distinct trend in Chinook parr density as evidenced by lack of a significant fit to the quadratic function. Elimination of these sites left a subset of 72 sites, 56 of which had  $P$ -values <0.05. The average of these sites in time was still somewhat controlled by outliers, but a curve fit through median values was similar to the reference curve (Figure 26).

The remaining 72 sites were put through the criteria of the piece-wise linear regressions. For the 1985-1997 period, 32 sites met criteria. Most were rejected for lack of a detectable slope, although slopes were too extreme at four sites, and no Chinook were observed during this period at one site. For the 1995-2003 period, 30 sites met criteria. Again, most were rejected because of insignificant slope estimates, but two were rejected for extreme slopes and no Chinook were observed at four sites. Only 12 sites met criteria for both periods (Table 13). The trend in means and medians was a little more extreme than the reference data (Figure 27). Of the 12 sites, only Red River and Crooked River had an uninterrupted series. Boulder Creek was missing six years of data; Brushy Fork was missing four years; the others had 16-18 years of data. Most sites exhibited a stronger trend for the 1995-2003 period than the 1985-1997 period (Table 13), despite having fewer surveys during the former.

## DISCUSSION

### Trend Interpretation

The basic trend in Chinook parr abundance declined from the mid 1980s to the mid 1990s, then numbers rebounded through the end of the time series. While it was not surprising that a trend was detected, one must consider whether the pattern is real or an artifact. My verification was not rigorous, because all analyses were done using the same data. However, the trend appeared quite general and was detectable at all spatial scales examined: statewide, major subbasin, and local (HUC4). The trend was apparent not only in observed density but in frequency of occurrence. The mechanistic basis for the trend seems obvious; in the past mean Chinook parr abundance tracked adult escapement (Hall-Griswold and Petrosky 2002). Redd counts in Idaho have declined and rebounded since 1985 (IDFG, unpublished data). Chinook abundance correlates with trends observed in other salmonids in the region (High et al. 2005; Thurow et al. in press). Finally, the trend was found when a fixed group of sites was used (Figure 25). Inconsistent sampling of sites over years can introduce statistical “noise” with each change in the group of sites under consideration. Use of a consistent set of sites removes this possibility so that power to detect a trend is enhanced (Archer et al. 2004). Therefore, I believe the trend in these data is real. However, it should be verified further by comparison to independent data sets, e.g., adult escapement and smolt production, preferably at several scales.

A key result of this work was that the basic pattern at the largest scale carried through to the smallest scale examined. The universality of the pattern is evidence that the population synchrony in the Middle Fork Salmon River drainage detected by Isaak et al. (2003) extends to a much larger geographic scale. However, the magnitude of the decline and recovery varied among HUC4s. At first glance, slope magnitude appeared to be related qualitatively to development in the watershed. For example, the Upper Salmon and Upper Middle Fork Salmon HUC4s are adjacent watersheds that historically were major production areas for Chinook salmon. However, while the Middle Fork is largely contained within a statutory wilderness, the Upper Salmon has been subject to irrigation, road building and other development (NPCC 2004). Recent parr density in the Upper Salmon HUC4 was not as high as in the Upper Middle Fork; consequently, the observed trend is flatter (Figure 20). Human development and land-use patterns have been negatively correlated to Chinook survival and recruitment in the Snake River basin (Paulsen and Fisher 2001; Thompson and Lee 2000), so the contrast in trend between HUC4s is not surprising. Where stream degradation was likely, there were fewer fish to start with; subsequently, the recovery has been less. Vice versa, populations in pristine areas were more resilient. The trend also suggests that the decline in the mid 1990s was not related to local habitat, because all watersheds were affected. However, intact habitat enabled some local populations to recover from the demographic catastrophe.

Another obvious geographic difference was that densities were much higher in the Clearwater subbasin during the 1980s than in the Salmon subbasin, although densities were similar during the recovery period (Figure 19). Hatchery production in the Clearwater is less constrained by ESA issues, which adds a confounding influence to the analysis if the overall hatchery contribution was much greater there. However, the magnitude of hatchery operations in the Clearwater has not declined since the 1980s. The Lochsa and South Fork Clearwater HUC4s, which contain the bulk of sampling in the Clearwater (see Appendix), are relatively pristine watersheds without major water withdrawal issues (NPCC 2003b); therefore, initial densities were high. Why the Clearwater populations should be less resilient than their cousins

in the Salmon basin is unclear. Information on the Clearwater populations is needed to address this and similar issues. The Chinook populations in the Clearwater drainage have received less attention from salmon scientists because Chinook in the Clearwater are not protected under the ESA. Biological information on naturally spawned fish in the Clearwater drainage will be required for continuing subbasin planning (Suzumoto 2005). The Clearwater represents a successful recolonization experiment and may yield valuable information for salmon management (Young 1999).

There were several problems inherent in the data that necessitate caution for analysis and interpretation. First, there were many zero observations, which complicate and bias estimates made with standard parametric statistics (Tu 2002). Zero observations are a common problem in conservation biology, but most investigators seem to work around them rather than directly deal with them (e.g., Taylor and Gerrodette 1993). I attempted to circumvent the problem by considering 75<sup>th</sup> percentile values, but this approach ignores site-level variation that might be important. Percentile regression may offer a way to incorporate this variation (Cade et al. 1999), or zero observations may be modeled with non-zero observations in a two-step mixed distribution process (Tu 2002).

There are also two interpretational issues. First, not all sites were selected with Chinook as the main focus. For example, spawning Chinook populations do not inhabit Panther Creek, the Lower Clearwater HUC, or the Hells Canyon tributaries (I speculate any Chinook parr observed in these areas were hatchery strays). Elimination of these data did not alter conclusions (e.g., Figure 21). The second problem is more pernicious. Sites were not selected randomly or with local populations (as defined by ICBTRT 2003) in mind. Thus, past site selections could lead to a lack of representativeness, and not all sites may be independent because they sample the same population at the same time. Analysis of the former is straightforward: define the expected spatial limits of Chinook parr distribution and plot the coverage of the GPM database on this area. Analysis of the latter may shed light on the nature of site-level biases. If sites are not independent in terms of the fish that are sampled, this may also lead to a loss of real degrees of freedom and the associated dangers of pseudoreplication (Hurlbert 1984). In this regard, the degree of temporal autocorrelation should also be assessed.

The GPM design was intended to track the statewide trend in density, not to assess abundance at finer scales. However, population-level abundance will become more important in future recovery planning. As the data are subdivided, site-level variation and bias will become more important. In effect, fewer degrees of freedom are available with a concomitant loss of statistical power. Conclusions at scales smaller than a HUC4 will be tenuous in most cases; indeed, some HUC4s currently have poor representation in the data. Issues of site-level bias and small sample sizes must be dealt with.

### **Detection Probability, Bias, and Methodology**

Raw counts, such as the GPM snorkel surveys, are neither censuses nor density estimates, and their use to estimate relative abundance requires careful statement and rigorous evaluation of assumptions (Link and Sauer 1998). The trend analyses and interpretations above implicitly assume that the probability of detecting a Chinook parr (given it is present) is constant at all sites and times. All sites were therefore weighted equally in the analysis. However, site-specific probabilities likely were not equivalent; therefore, we must consider how violations of this assumption affect conclusions.

I examined the influence of several variables on the likelihood that Chinook parr were observed during a sampling event (i.e. occurrence) as a surrogate to direct estimates of detection probability. Stream width had the most important influence on probability of observation. The influence of channel type was not as large. Mean widths of each channel type were not greatly different relative to the range (data not shown). The effect of temperature was also not consistent through time (Figure 23). Visibility was the least important of the variables examined, but that does not mean that it is unimportant. The use of *P*-values was misleading in this case because sample sizes were large and *P*-values are influenced by sample size (Johnson 1999). I believe that biologists were not likely to sample when visibility was poor. The use of multiple snorkelers as mandated in the GPM protocol (Holubetz and Petrosky 1986) probably reduced the influence of visibility. However, it is clear that physical site attributes had more influence on detection probability in this data set.

Given the GPM protocol is rigorously applied, site differences in detection probability should not be large enough to affect conclusions, if sample size is sufficiently large. However, as scale and sample size decrease, site-level biases will increase in influence. Methodology must be altered to identify and estimate this bias for rigorous population-level analyses. Bias estimation will facilitate tracking of abundance, because probability of detection must be estimated for population estimation (Pollock et al. 2002). Differences in detection probability will not bias trend estimation as long as there is no spatial or temporal pattern to this variation (Link and Sauer 1998). Estimating biases at a representative set of sites would be a good way to evaluate this assumption.

Qualitatively, the results of my analysis agree with the findings of Thurow et al. (in press), although the streams they sampled were smaller headwaters. Thurow et al. measured 13 habitat variables. Water temperature, visibility, and percentage of site length as pools were retained in the best model predicting probability of detecting *Oncorhynchus* sp. Detection probability was nonlinearly related to visibility in their data, with maximum at 2.5-3.5 m visibility. The GPM Chinook parr densities showed a peak in 3.0-4.5 m visibility, then gradually declined (Figure 22, bottom panel). In Thurow et al.'s data, water temperature was positively and linearly related to detection probability for *Oncorhynchus* sp., but the maximum temperature was 16.8°C. In the GPM data, parr density peaked at 12-17°C and then decreased (Figure 22, middle panel). The two data sets are similar within the more limited range of temperatures observed by Thurow et al. Given these similarities, I concluded that duplicating Thurow et al.'s analysis on GPM sites will be fruitful, and other potential explanatory variables beyond those currently measured should be considered.

This analysis was observational and focused on a surrogate for the parameter of interest. Probability of observing Chinook parr at a site is a product of occurrence and detection probabilities. In my analysis, I assumed that all sites had high and equivalent probabilities of Chinook occurrence; therefore, most of the variation observed was due to site-level biases in detection probability. There was no quantitative way to assess the appropriateness of this assumption. Another problem was that many variables are likely to influence detection probability (e.g., presence of large woody debris), but only three were measured and recorded with the available data. There was much unexplained variation in the model, indicating that consideration of other variables would be worthwhile in future analyses.

## Site Selection

My primary goal for this report was to define a group of candidate sites that would give the best picture of the trend in Chinook parr abundance during 1985-2003. I assumed that monitoring the candidate sites would enable detection of future changes at a regional level. However, prior to adoption, the candidate sites need to be vetted for logical consistency, representativeness, and presence of confounding influences. The majority of them are in the Salmon River subbasin, but three sites are in the Clearwater subbasin (Figure 28). Three sites are in drainages where Chinook salmon have not been stocked. Possible influence by hatchery releases should be considered for the other sites. Several sites are in relatively undeveloped drainages. Sites in more impacted areas should be carefully inspected. Finally, some sites fall within the same population (as defined by ICBTRT 2003). The ability of these sites to reflect local productivity in an independent manner should be investigated.

The use of fixed sites for population trend assessment seems inevitable because of the nonrandom site selection process used in the past. Methods for trend analysis of fixed-site data are most developed for the North American Breeding Bird Survey (BBS). Regional abundance trends are estimated by an average of the trends at each site (in the BBS, transects or routes are sampled; the analysis of resultant data is termed route regression). Contributions by each site should be weighted by population abundance and survey precision (or inverse of model variance; Sauer and Droege 1992; Thomas 1996; Link and Sauer 1998). Use of fixed sites helps maintain constant site biases (Archer et al. 2004), eliminating a source of confounding variation. Another advantage of using the candidate sites, given the selection criteria used, is that parametric statistics may be appropriate for analyses. However, nonlinear and nonparametric methods may be more appropriate, especially as the data set grows in temporal scope (e.g., James et al. 1996; Link and Sauer 1998; Fewster et al. 2000).

Any selection criteria will be arbitrary to some degree, so other approaches need to be considered to supplement and verify the selection of the candidate trend sites. I am concerned that only 12 sites may lack enough power to detect changes in trend within a reasonable amount of time. A power analysis may shed light on the number of sites required. The outcome should indicate if the core group of trend sites needs to be much larger or not, but there are two stumbling blocks to this direction. First, power analysis usually assumes randomly chosen sites. Secondly, criteria for precision and magnitude of change must be chosen. Depending on the results of the power analysis, selection criteria may be relaxed so that some of the 72 sites rejected in the last step could be included in the final group of candidates. Number of sites per year is estimable from a power analysis (e.g., Gerrodette 1987) but, if interannual variation about the trend is real, the number of years monitored controls the equation (Hatch 2003). Therefore, annual monitoring of 12 sites may be sufficient if the annual mean is estimated with useful precision. Alternatively, power analyses may be conducted for each site. Simulations are the preferable means of conducting a power analysis because the available data are often but a single realization of a stochastic process (Link and Hatfield 1990). Several free software packages are available (Hatch 2003).

## **CONCLUSIONS AND RECOMMENDATIONS**

The GPM data showed a decline and subsequent recovery in spring/summer Chinook stocks in Idaho during 1985-2003. This trend was very general but needs additional corroboration, e.g., by comparison to local redd counts. Furthermore, the methods I used were

not very sophisticated. I recommend adapting methods developed for BBS trend analysis to the GPM data to obtain more rigorous and defensible estimates of trend parameters.

Site-level effects on detection probabilities will be a concern for the smaller scale analysis likely in the future. These effects must be dealt with to maintain the usefulness of the GPM data. Important variables should be identified and measured at a set of representative sites. The list of potential explanatory variables should include more than just those typically measured in the past (see Thurow et al., in press, for examples).

This analysis was just the first step towards incorporating probabilistic sampling into the GPM program. The core set of trend sites should be evaluated further. The extent of potential sites also needs definition (the ICBTRT may be able to provide guidance). This definition will enable random selection in the future as well as an assessment of representation by the current set of sites. A subset of the current core sites (Priority 1, see Hall-Griswold and Petrosky 2002) will be retained to be sampled annually. The exact number will be based on a power analysis of the existing data. The remaining sites will be evenly distributed between a five-year panel and stratified random sampling. We will designate three sampling frames based on expected distributions of SRSS Chinook salmon, steelhead trout, and bull trout. We will also develop appropriate strata for each species. Sites selected for one species will generate data applicable for other species (as adjusted by the known statistical framework); thus, sites and data can be efficiently used to track status of all salmonids.

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Table 10. Number of sites in the GPM database by year sampled and major drainage.

| Year | Drainage   |        |              |
|------|------------|--------|--------------|
|      | Clearwater | Salmon | Hells Canyon |
| 1985 | 62         | 122    | 6            |
| 1986 | 86         | 205    | 8            |
| 1987 | 125        | 194    | 8            |
| 1988 | 131        | 229    | 6            |
| 1989 | 136        | 223    | 8            |
| 1990 | 212        | 232    | 8            |
| 1991 | 338        | 276    | 9            |
| 1992 | 359        | 348    | 4            |
| 1993 | 232        | 301    | 5            |
| 1994 | 328        | 482    | 5            |
| 1995 | 238        | 217    | 4            |
| 1996 | 321        | 174    | 4            |
| 1997 | 274        | 183    | 4            |
| 1998 | 191        | 138    | 4            |
| 1999 | 225        | 213    | 4            |
| 2000 | 219        | 184    | 4            |
| 2001 | 159        | 191    | 4            |
| 2002 | 146        | 177    | 2            |
| 2003 | 122        | 164    | 2            |

Table 11. Selected percentiles of wetted width, water temperature, and visibility as measured at time of sampling at 2,437 sites.

| Percentile       | Wetted width<br>(m) | Water Temperature<br>(C) | Visibility<br>(m) |
|------------------|---------------------|--------------------------|-------------------|
| 25 <sup>th</sup> | 7.3                 | 12                       | 2.7               |
| 75 <sup>th</sup> | 13.4                | 17                       | 5.2               |

Table 12. Slope estimates used for references for regressions during 1985-1997 and 1995-2003.

| Period    | Slope  | 95% Confidence Interval |             |
|-----------|--------|-------------------------|-------------|
|           |        | Lower Limit             | Upper Limit |
| 1985-1997 | -1.090 | -1.837                  | -0.342      |
| 1995-2003 | 1.528  | 0.343                   | 2.712       |

Table 13. Candidate trend monitoring sites and estimated regression parameters for the period 1985-1997 and 1995-2003. Values in parentheses are the 95% confidence limits.

| Stream, Site              | 1985-1997                   |                | 1995-2003                 |                |
|---------------------------|-----------------------------|----------------|---------------------------|----------------|
|                           | Slope                       | r <sup>2</sup> | Slope                     | r <sup>2</sup> |
| Alturas Lake Creek, 2-2B  | -2.183<br>(-4.317, -0.049)  | 34.2%          | 0.708<br>(0.307, 1.108)   | 71.4%          |
| Boulder Creek, Below-3    | -1.366<br>(-2.956, -0.224)  | 32.9%          | 5.240<br>(1.491, 8.990)   | 86.8%          |
| Brushy Fork Creek, 3-2    | -1.800<br>(-2.955, -0.645)  | 61.8%          | 1.479<br>(0.279, 2.680)   | 60.2%          |
| Crooked River, 2-Treat2   | -2.485<br>(-4.616, -0.355)  | 37.5%          | 1.748<br>(-0.304, 3.799)  | 36.7%          |
| Marsh Creek, 5-A          | -5.773<br>(-10.425, -1.120) | 46.7%          | 5.417<br>(-1.148, 11.982) | 47.4%          |
| Monumental Creek, Mon5    | -2.422<br>(-4.246, -0.599)  | 46.7%          | 8.514<br>(1.537, 15.490)  | 59.8%          |
| Red River, 4-Treat2       | -4.270<br>(-7.803, -0.738)  | 39.2%          | 6.504<br>(-0.984, 13.956) | 37.8%          |
| Salmon River, 3-3B        | -1.892<br>(-3.603, -0.180)  | 41.0%          | 0.920<br>(0.251, 1.588)   | 60.2%          |
| Salmon River, 4-4B        | -2.037<br>(-3.731, -0.343)  | 45.1%          | 0.429<br>(0.005, 0.854)   | 50.5%          |
| SF Salmon River, Stolle2  | -4.394<br>(-9.506, -0.718)  | 26.8%          | 10.190<br>(0.781, 19.599) | 48.4%          |
| Valley Creek, 3-A         | -2.890<br>(-5.574, -0.206)  | 33.8%          | 1.575<br>(-0.135, 3.285)  | 45.8%          |
| WF Monumental Creek, Mon4 | -4.978<br>(-9.671, -0.284)  | 33.1%          | 11.187<br>(2.482, 19.892) | 62.2%          |

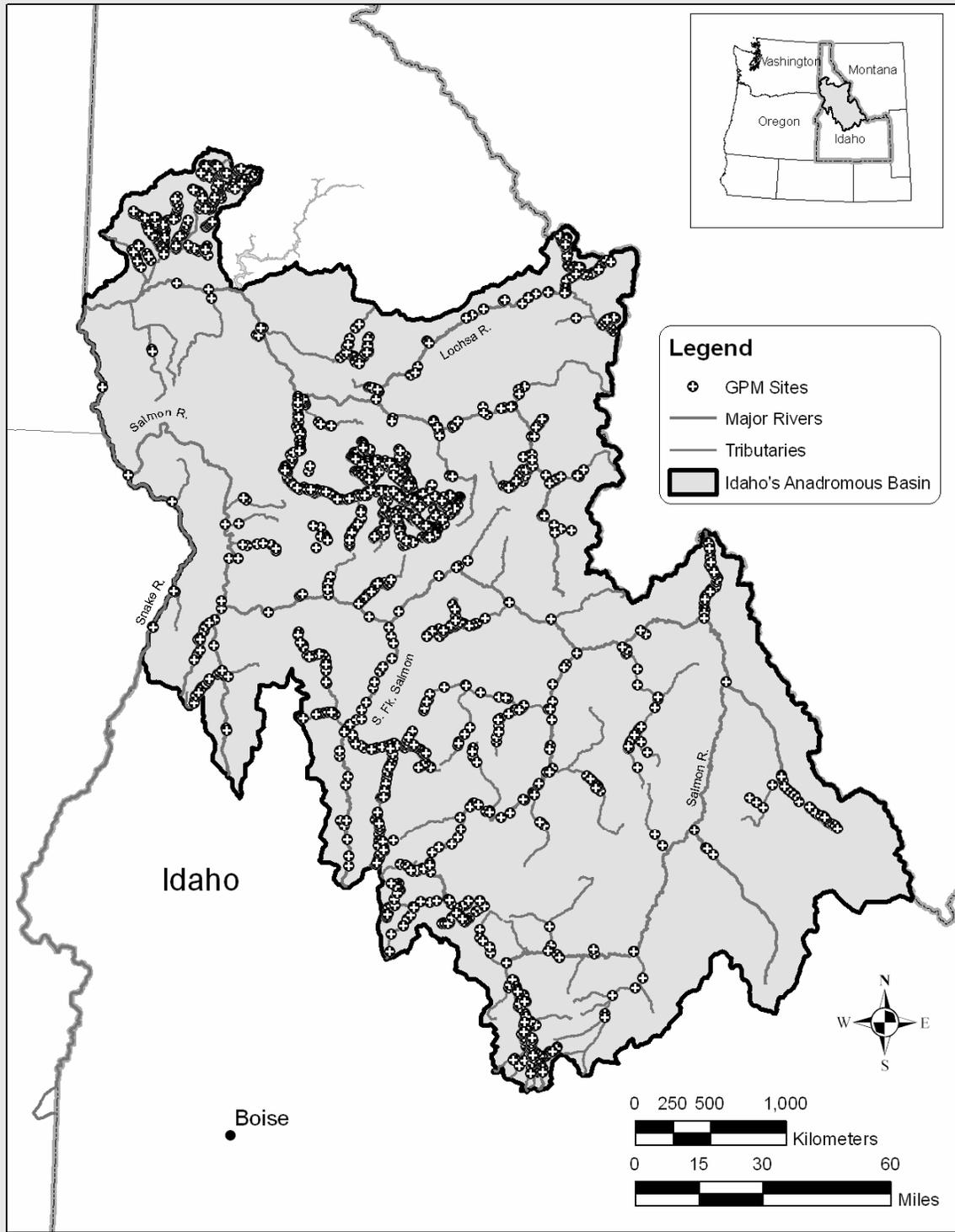


Figure 17. The location of all General Parr Monitoring snorkel sites in Idaho.

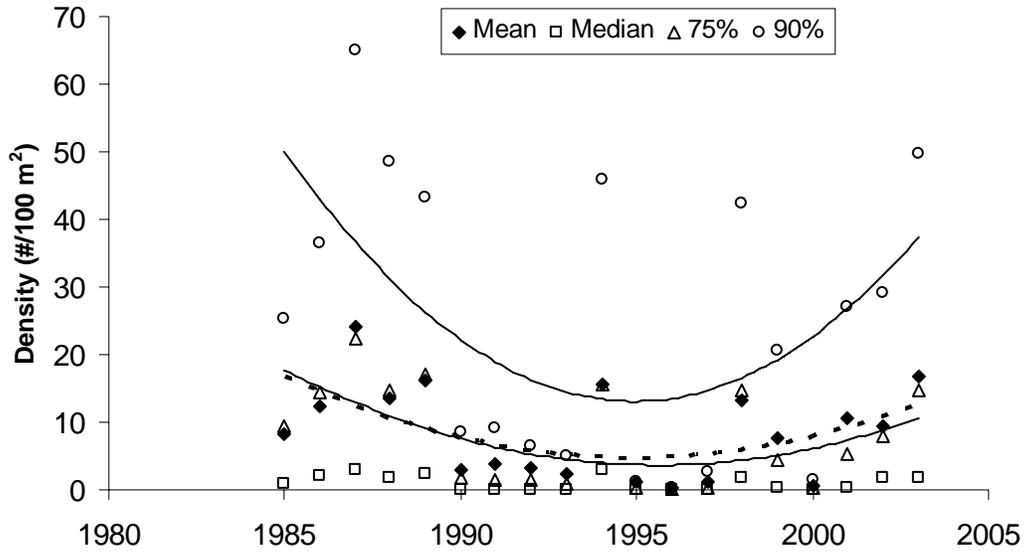


Figure 18. Annual means and selected percentiles of age-0 Chinook parr density. Quadratic trend lines were drawn for each measure except the median. The trend line for means is dashed.

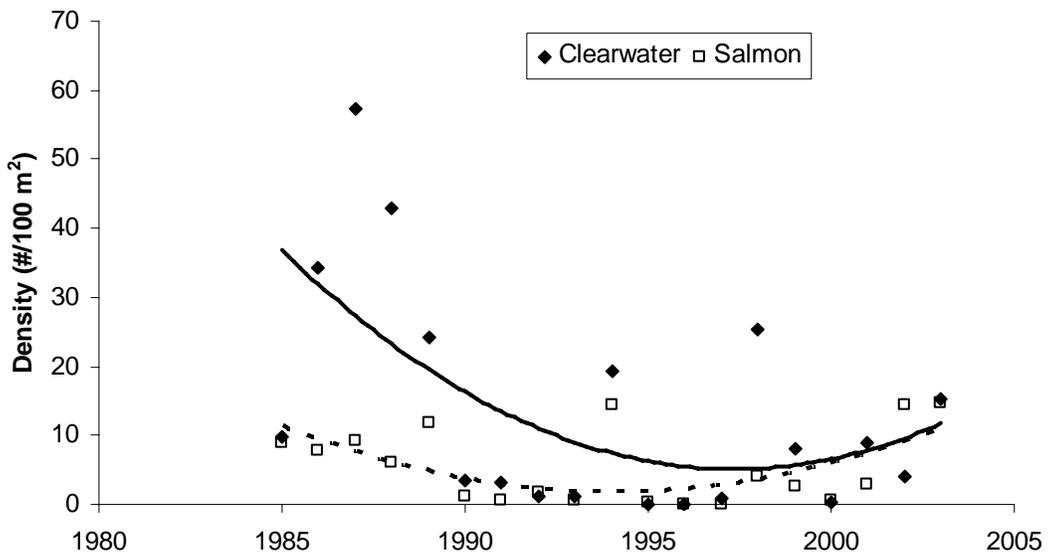


Figure 19. Comparison of trends in annual 75<sup>th</sup> percentile Chinook parr density between the Clearwater and Salmon basins.

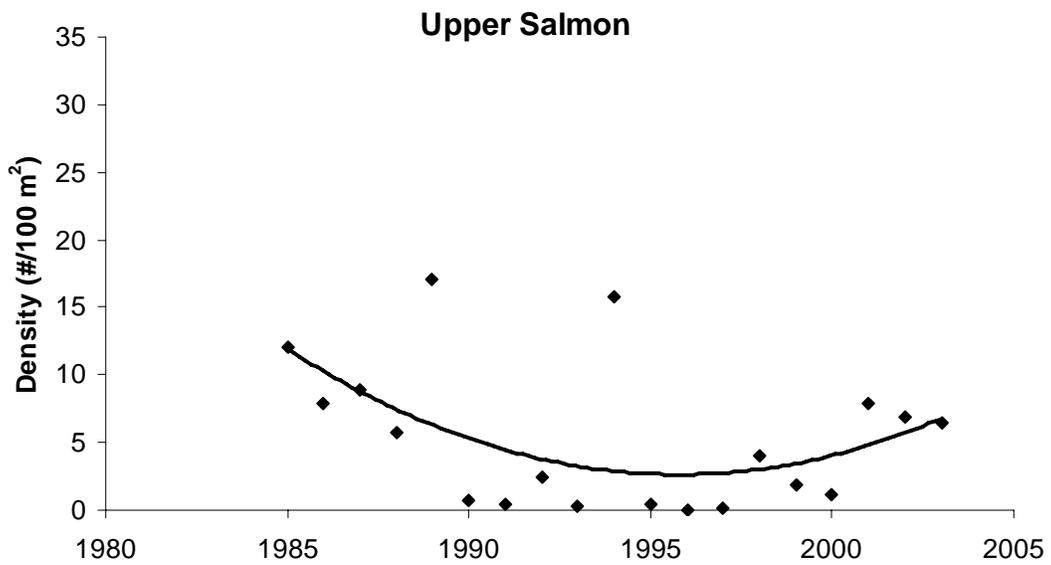
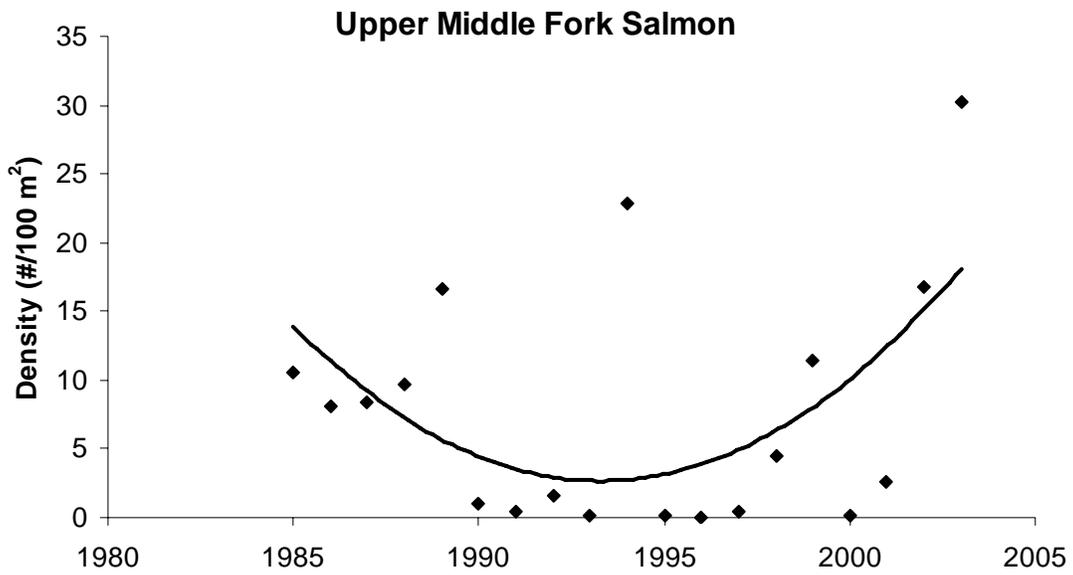


Figure 20. Trends in 75<sup>th</sup> percentile density by year for the Upper Middle Fork Salmon and Upper Salmon watersheds (fourth field hydrologic unit codes).

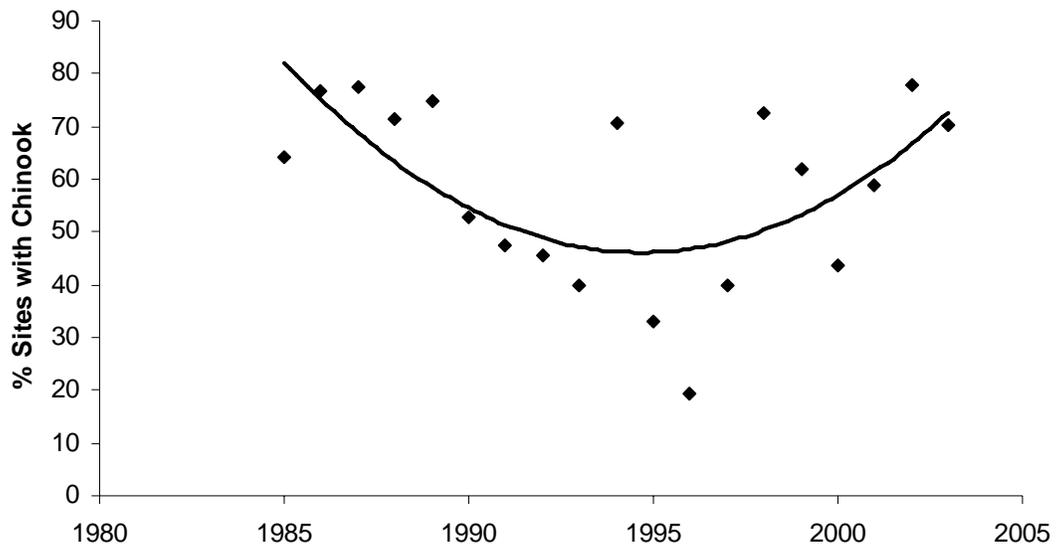


Figure 21. Percentage of sites sampled in which Chinook parr were observed by year. Sites from the Lower Clearwater HUC, Hells Canyon, and Panther Creek were excluded.

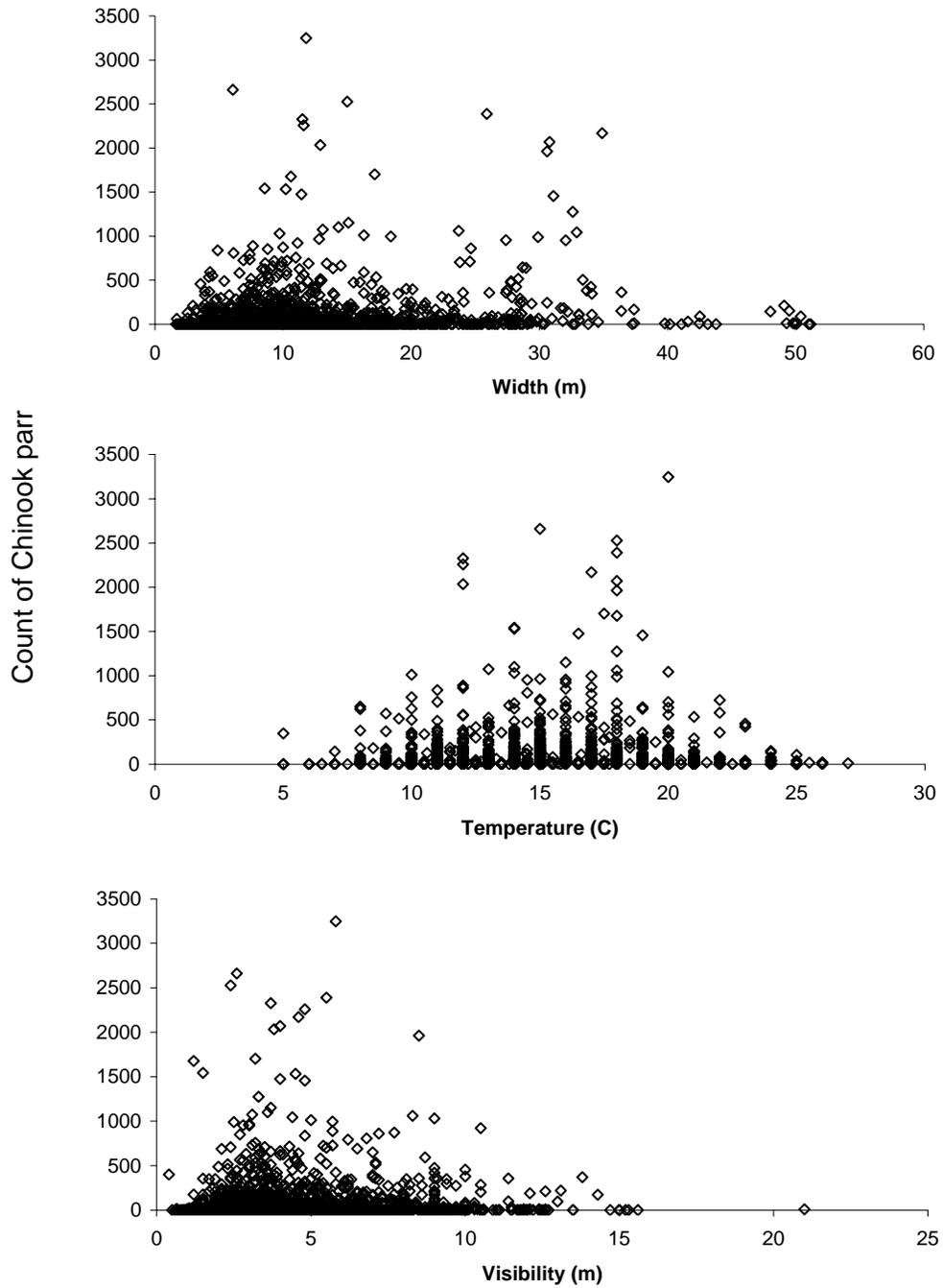


Figure 22. Selected stream characteristics at time of sampling versus number of Chinook parr observed at 2437 GPM sites, 1985-2003.

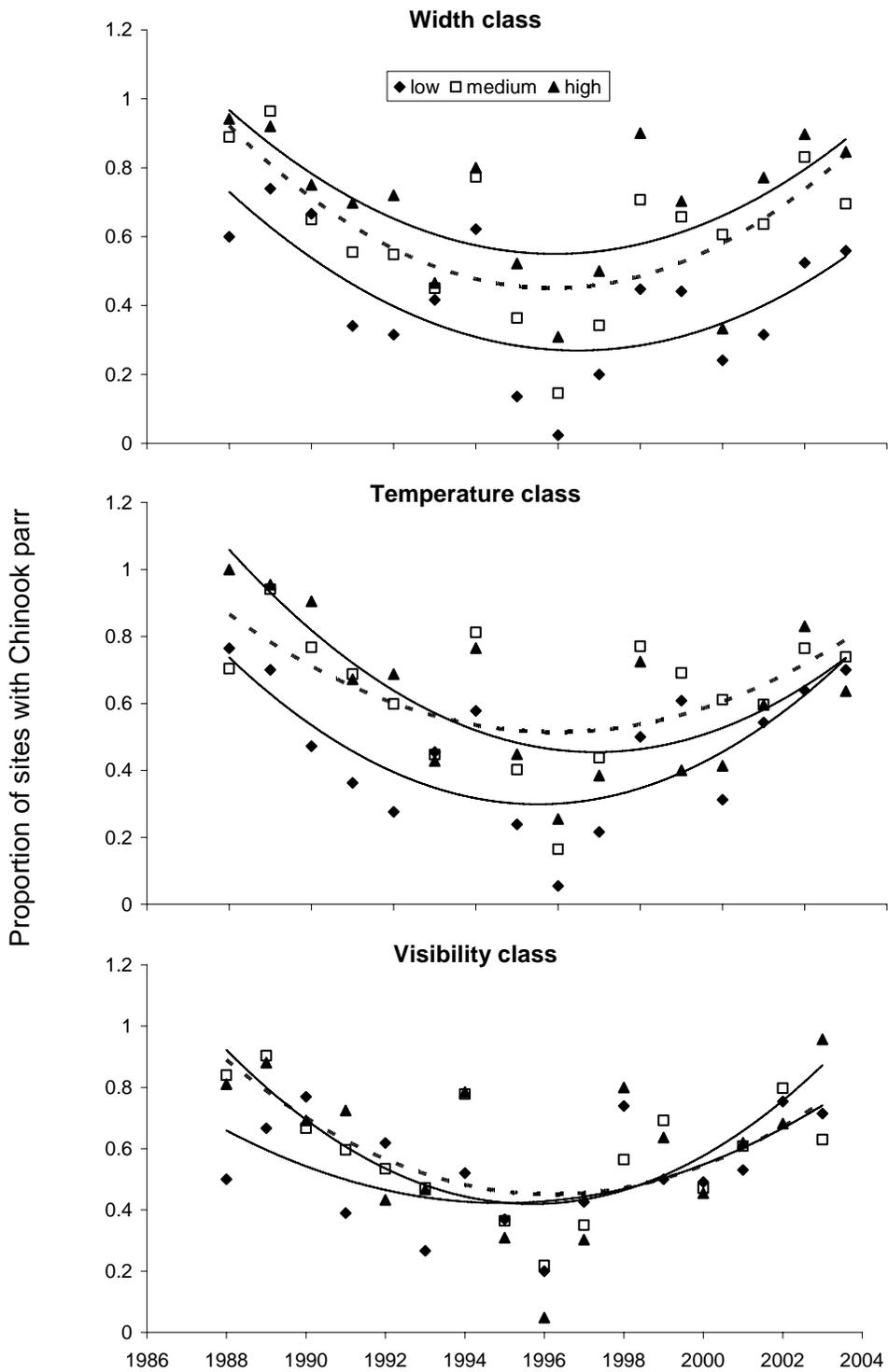


Figure 23. Proportion of sites where Chinook parr were detected by year and by percentile category (low, medium, and high) based on stream width, temperature, and visibility at time of sampling. Percentiles are defined in Table 11. Trend lines are shown; the medium trend line is dashed.

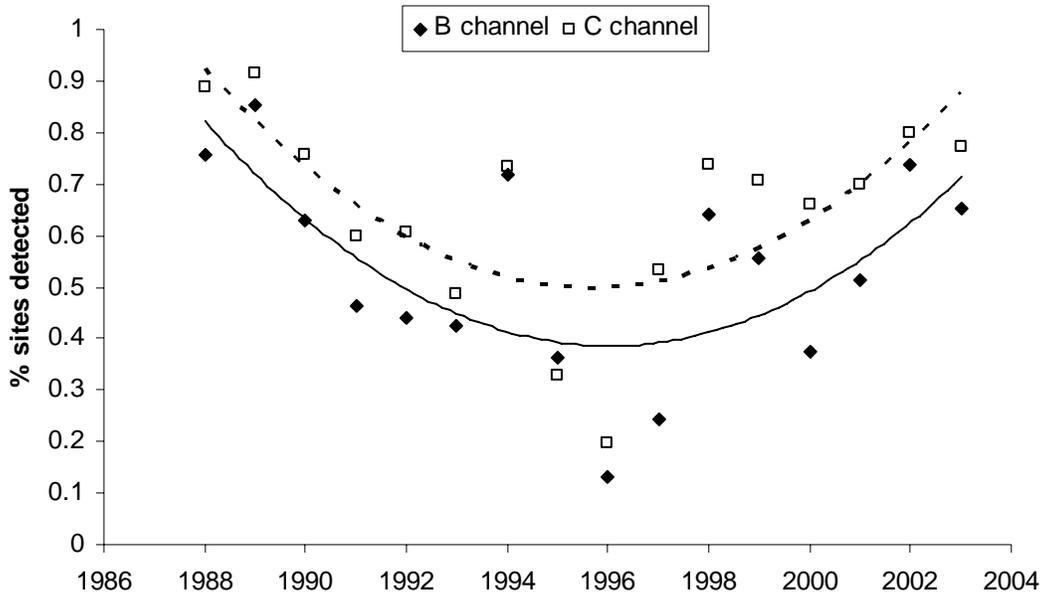


Figure 24. Comparison of percentage of sites where Chinook parr were observed by Rosgen channel type.

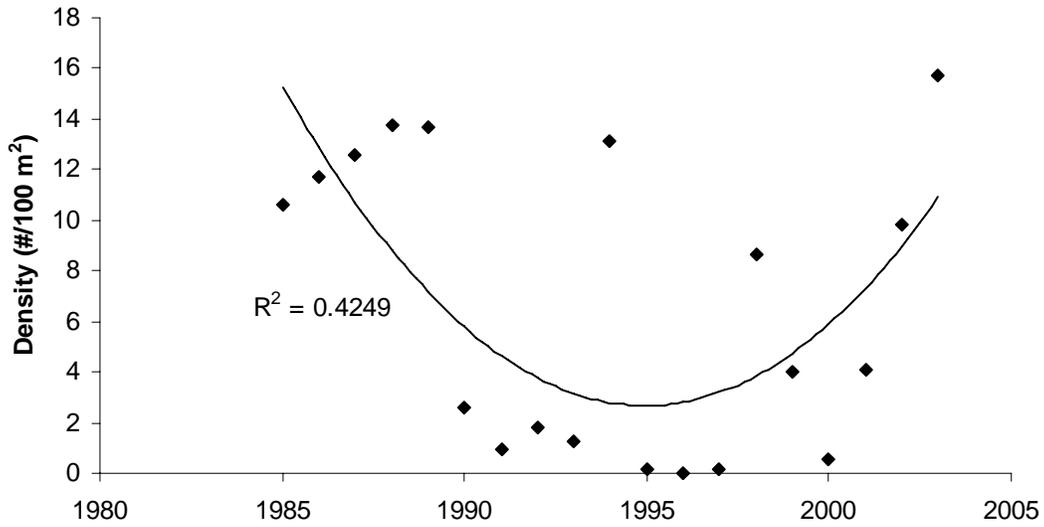


Figure 25. Seventy-fifth percentile densities by year of 209 trend sites. The quadratic trend line and coefficient of multiple determination are shown.

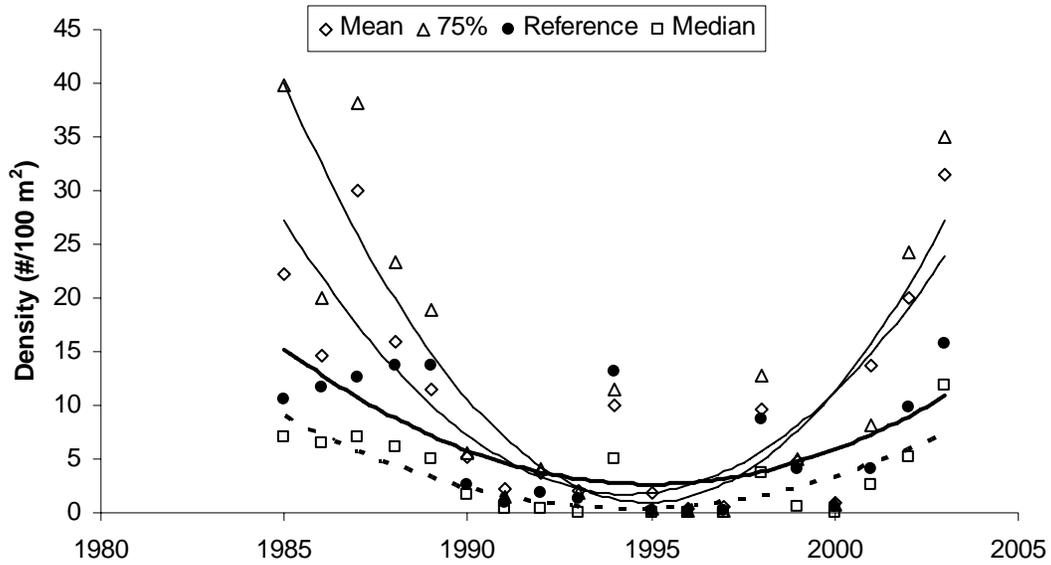


Figure 26. Trends in 72 selected sites compared to the reference 75<sup>th</sup> percentile values from Figure 25. Quadratic trend lines are shown. The reference trend is in bold; the trend in median values is dashed.

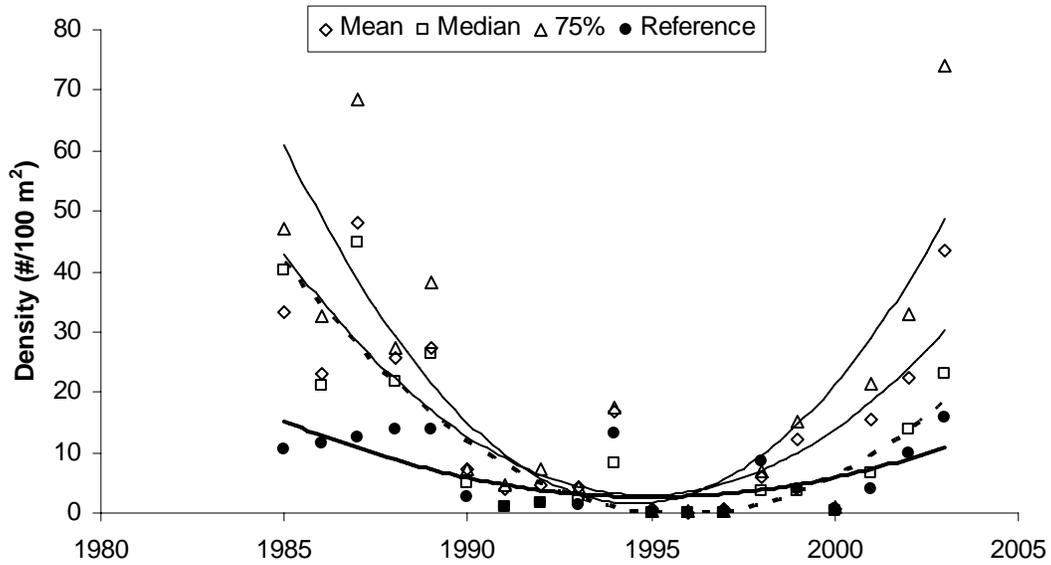


Figure 27. Trends in 12 selected sites compared to the reference 75<sup>th</sup> percentile values from Figure 25. Quadratic trend lines are shown. The reference trend is in bold; the trend in median values is dashed.

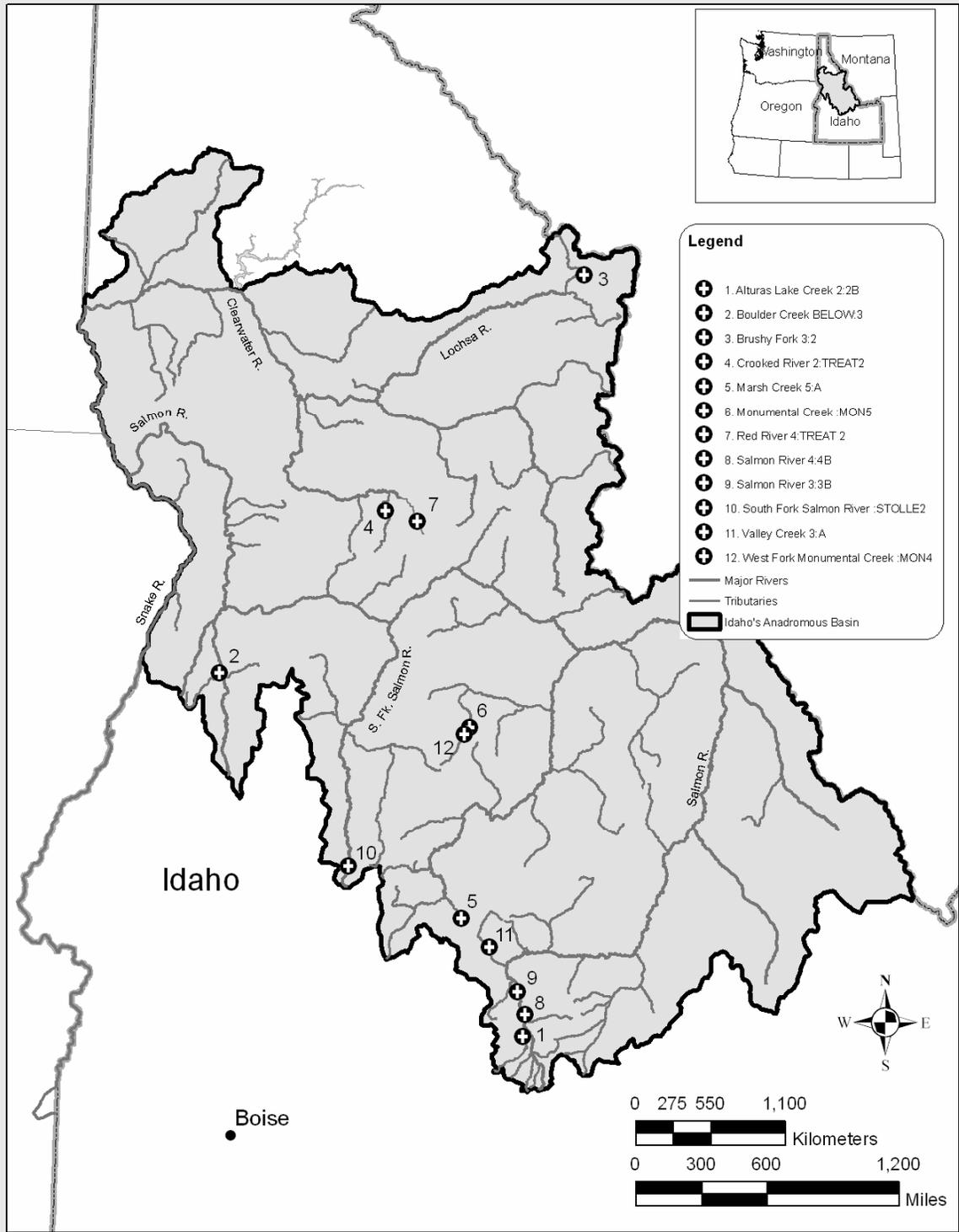


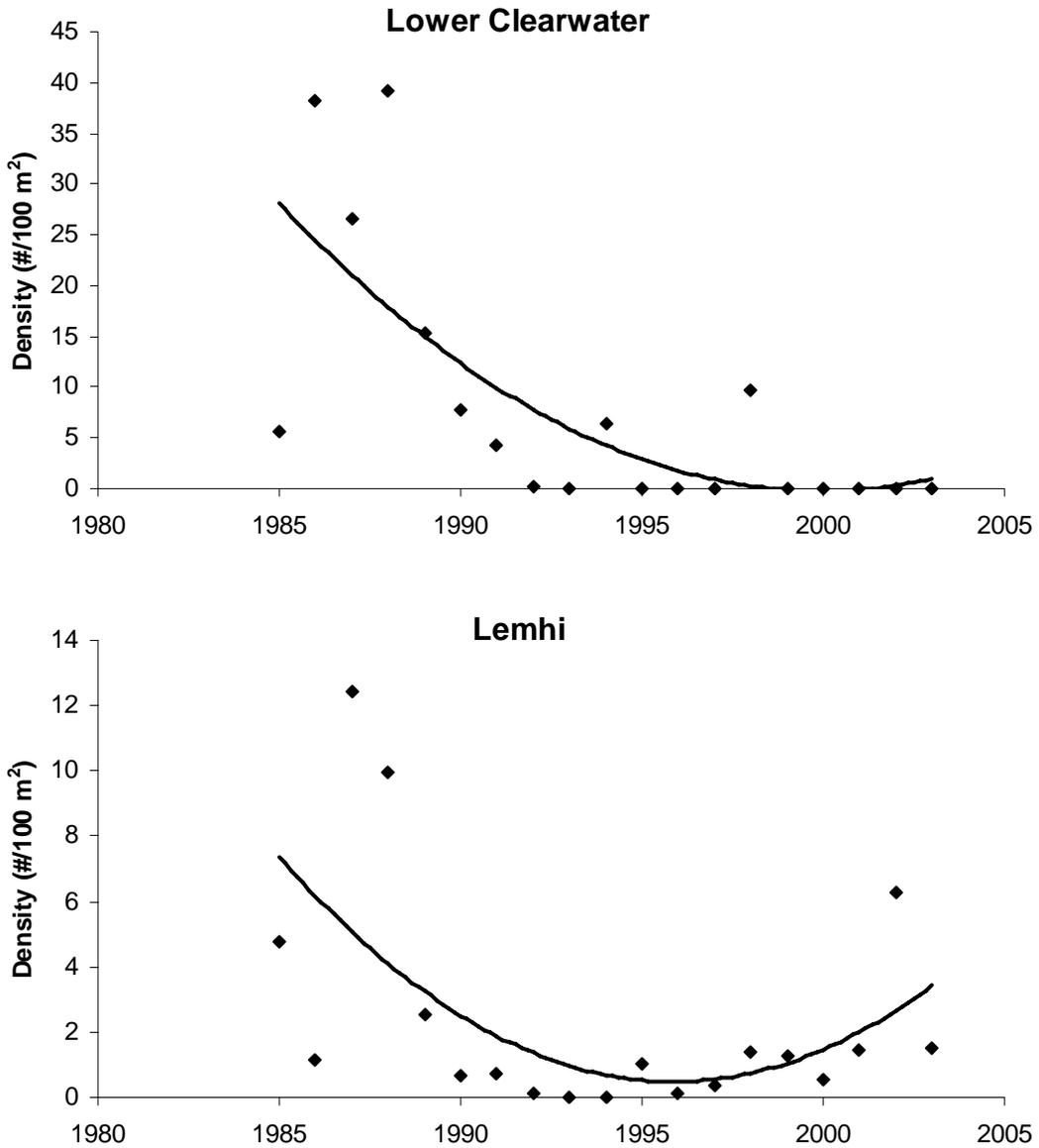
Figure 28. Locations of the 12 candidate trend sites.

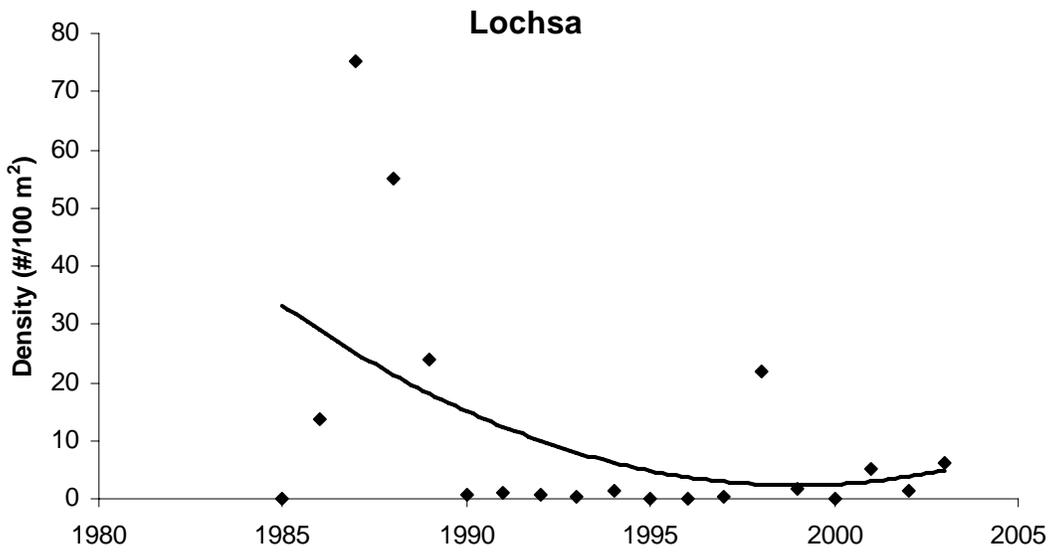
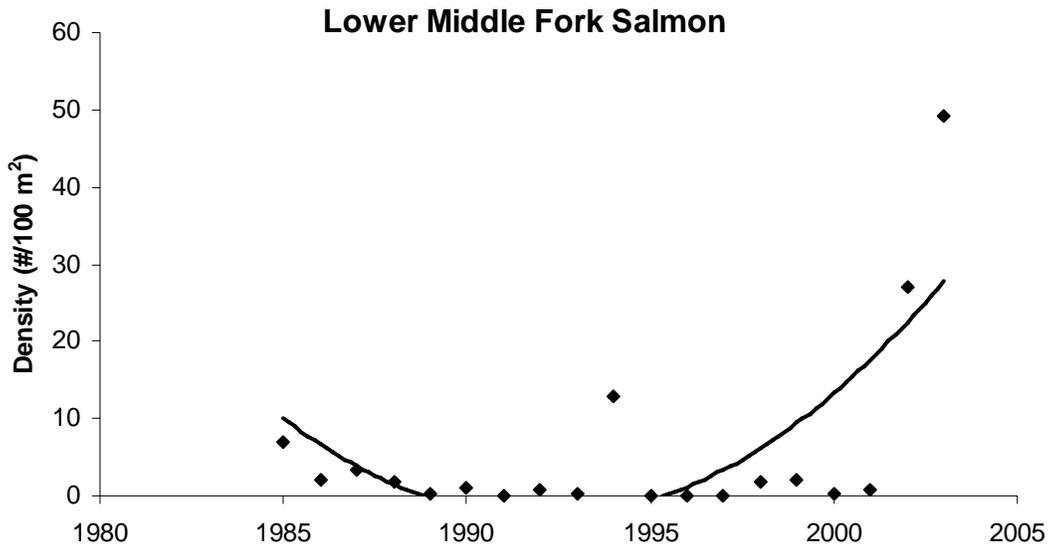
## **ACKNOWLEDGEMENTS**

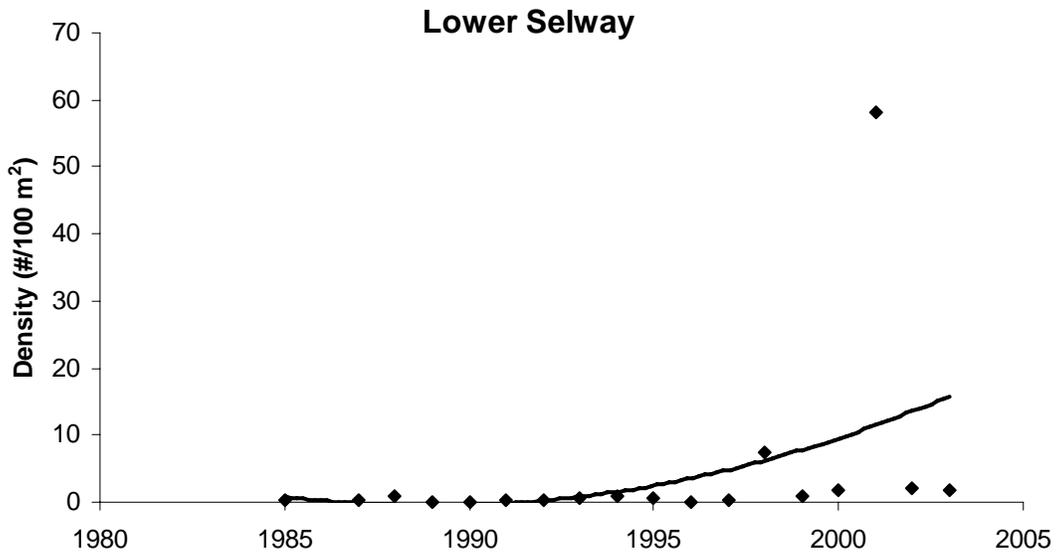
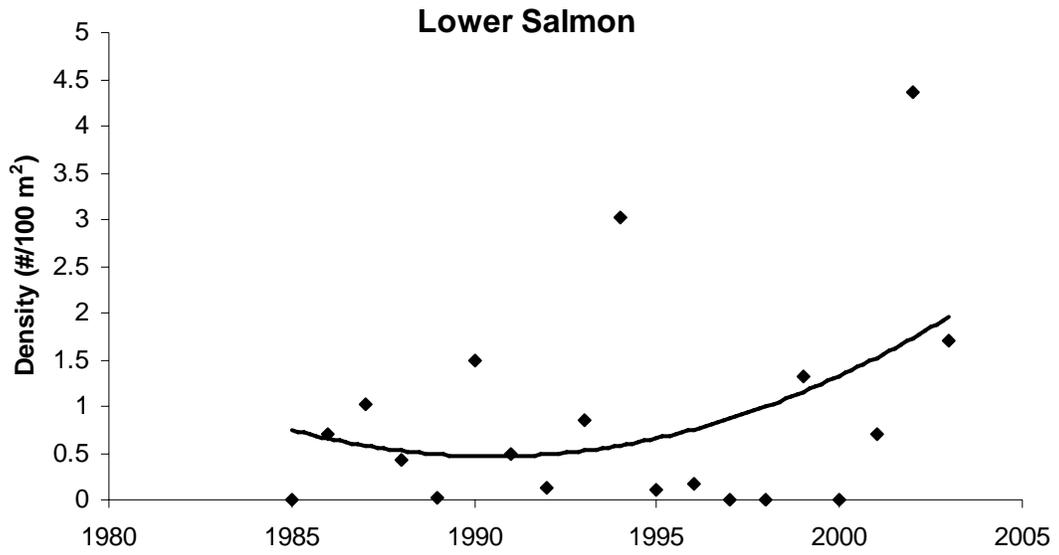
Charlie Petrosky and Judy Hall-Griswold patiently answered many questions regarding the purpose and history of the GPM program. Any errors of interpretation are my own.

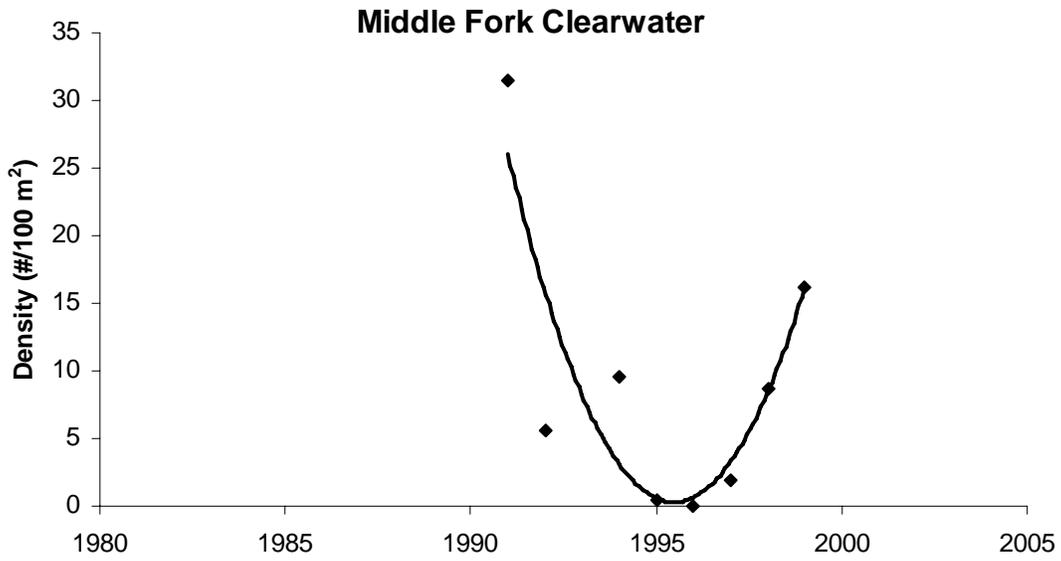
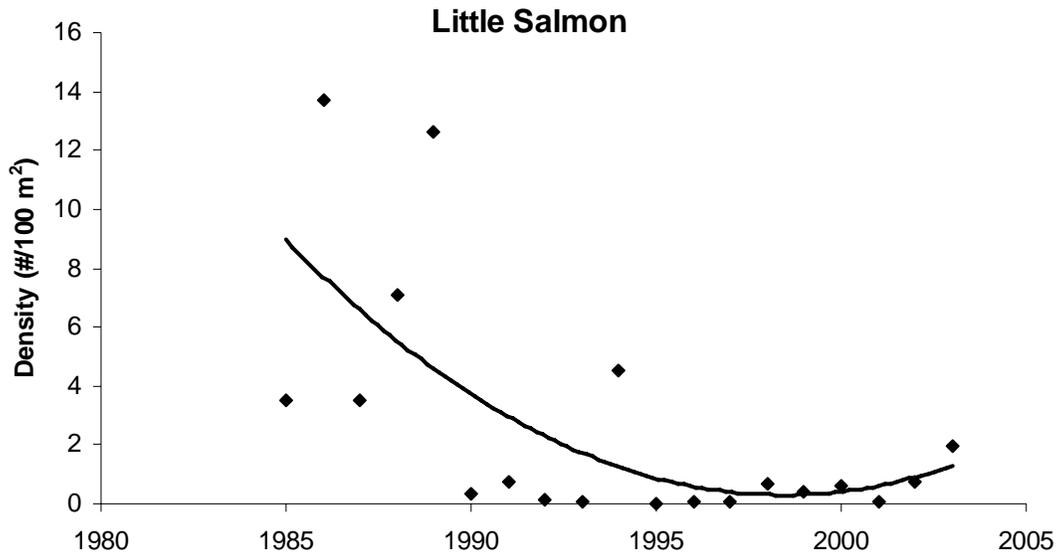
## **APPENDICES**

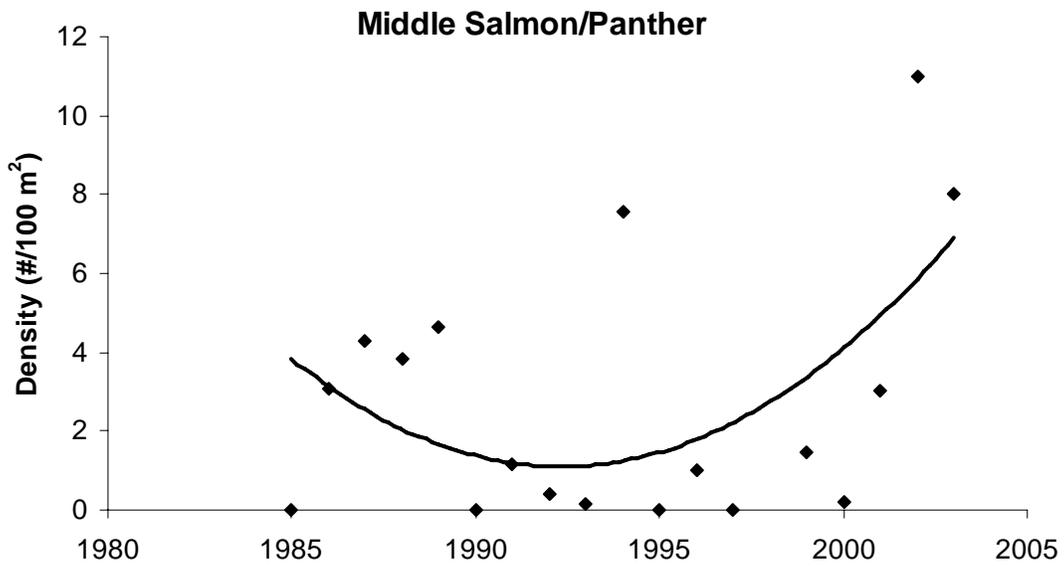
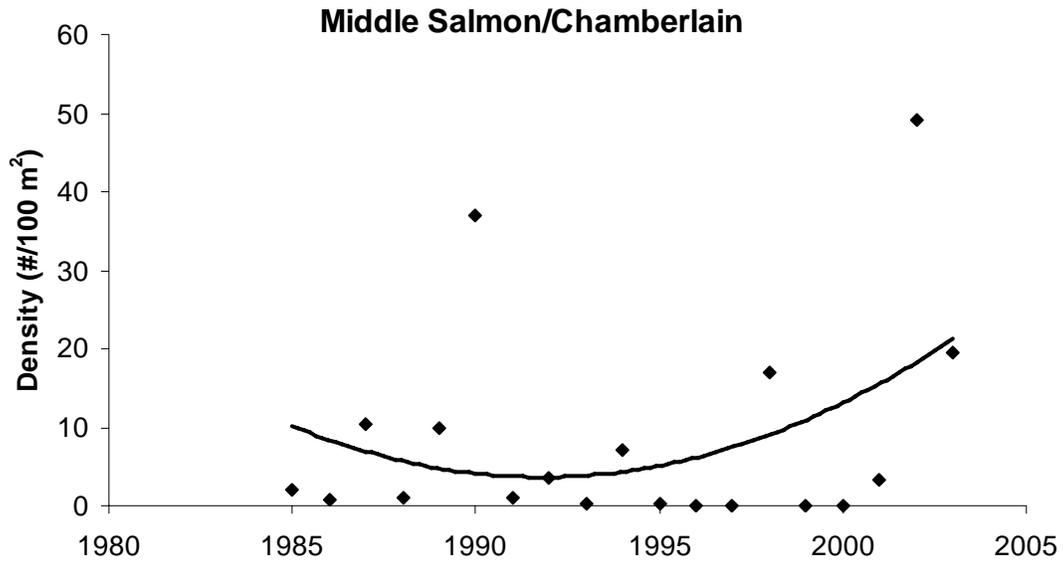
Appendix B. Figure 1. Trends in 75<sup>th</sup> percentile density during 1985-2003 for each fourth field HUC. Trends were drawn using multiple linear regression with a quadratic term. Note that differences in y-axis scale distort comparisons of slopes.

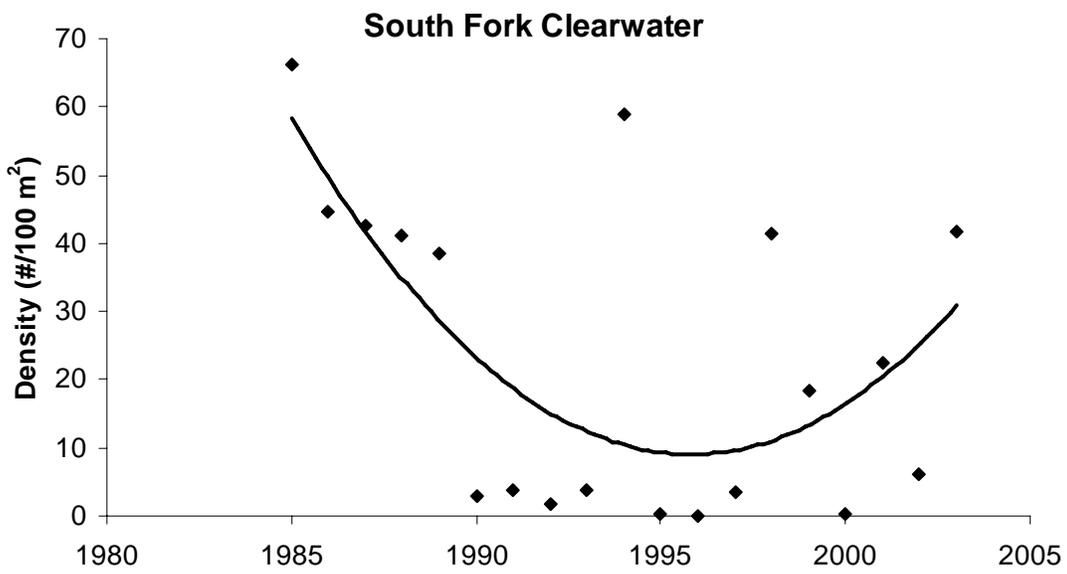
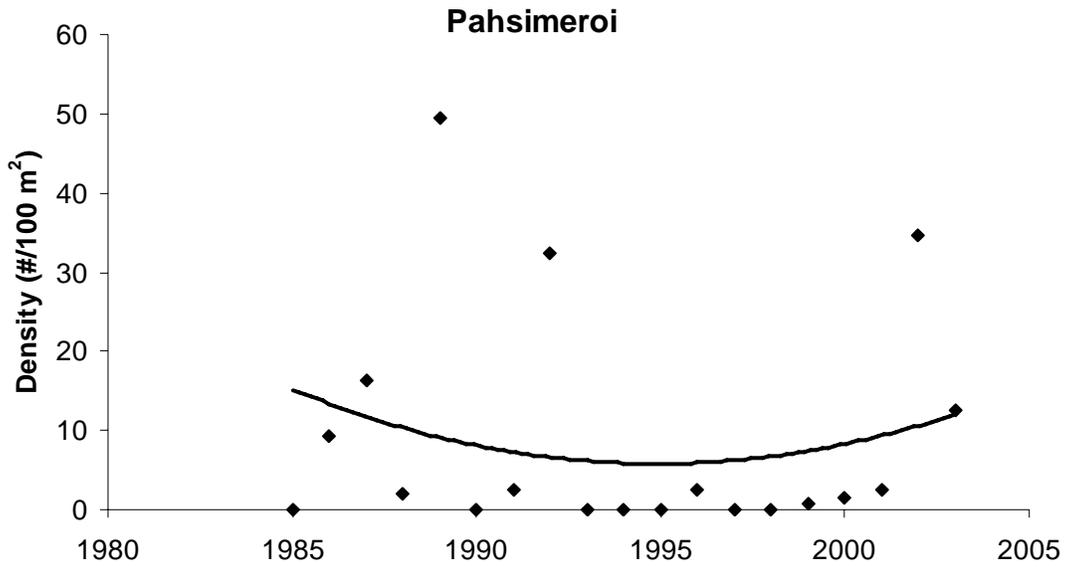


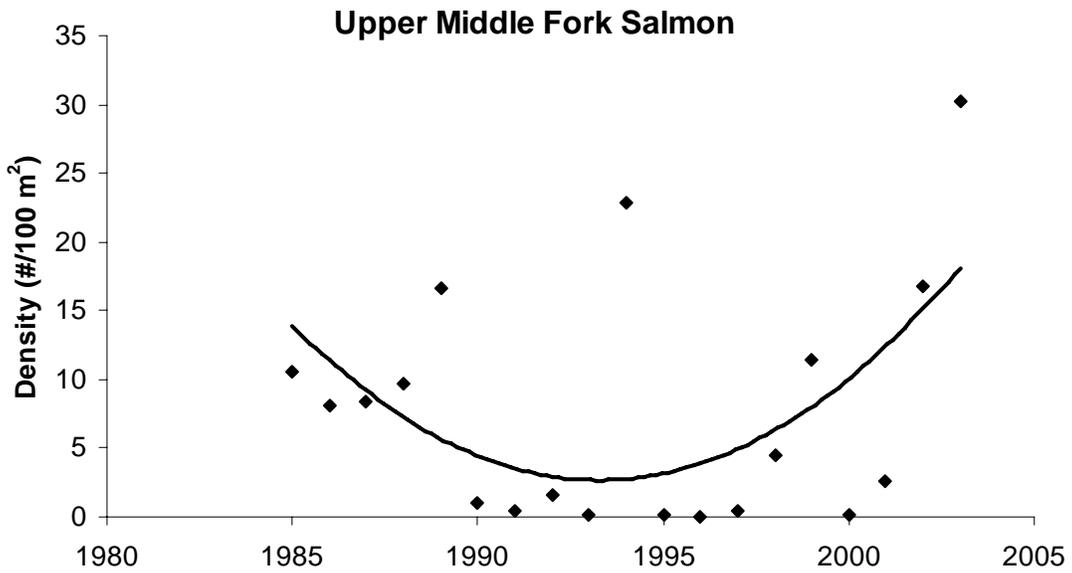
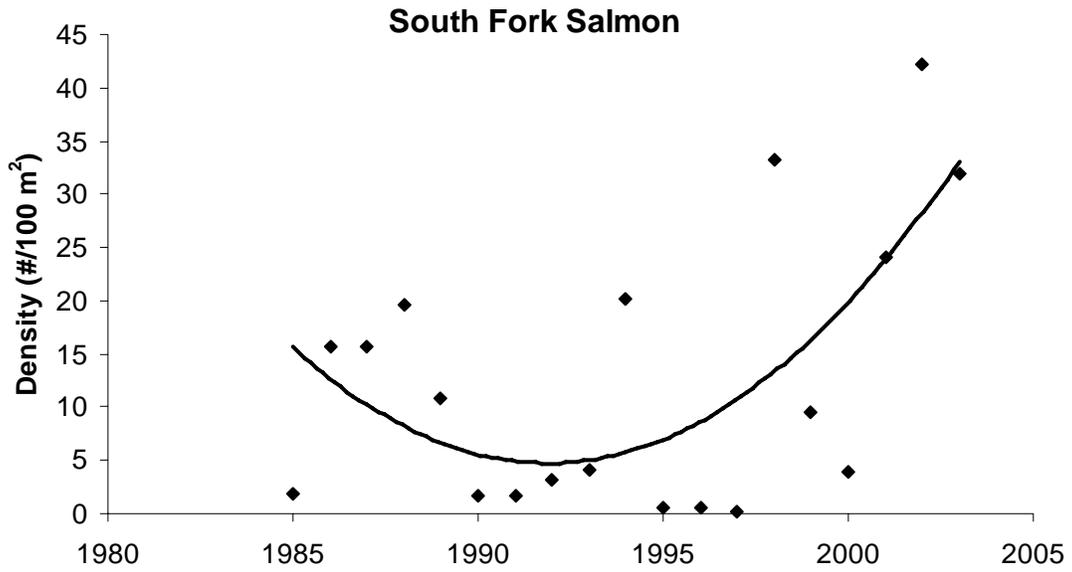


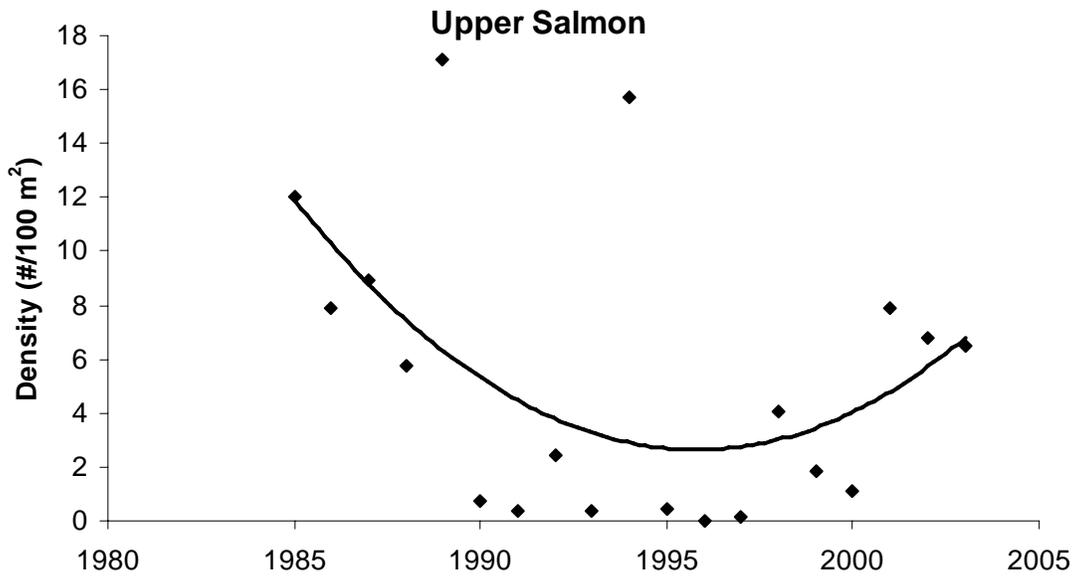
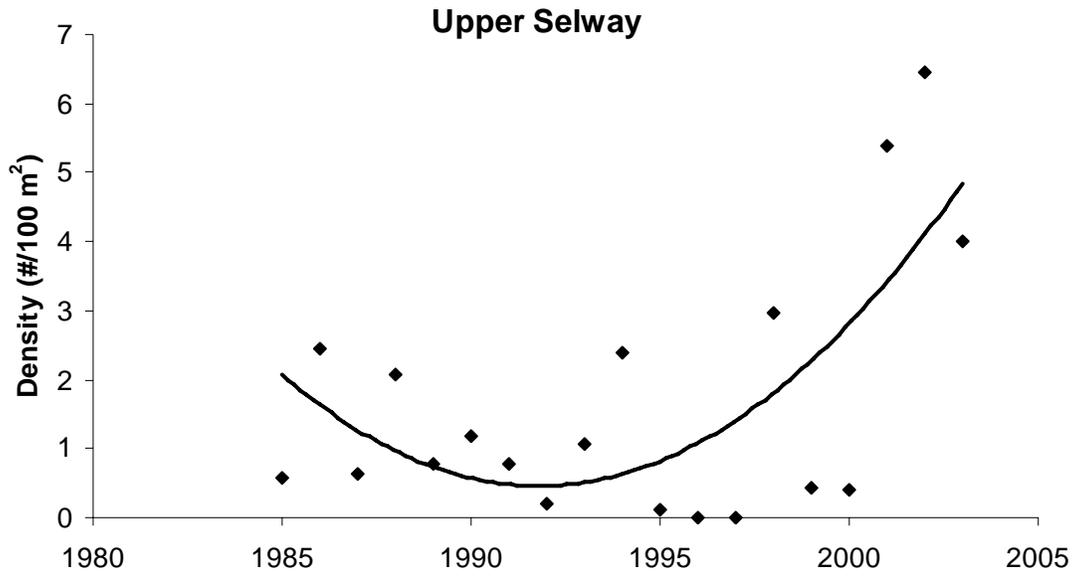


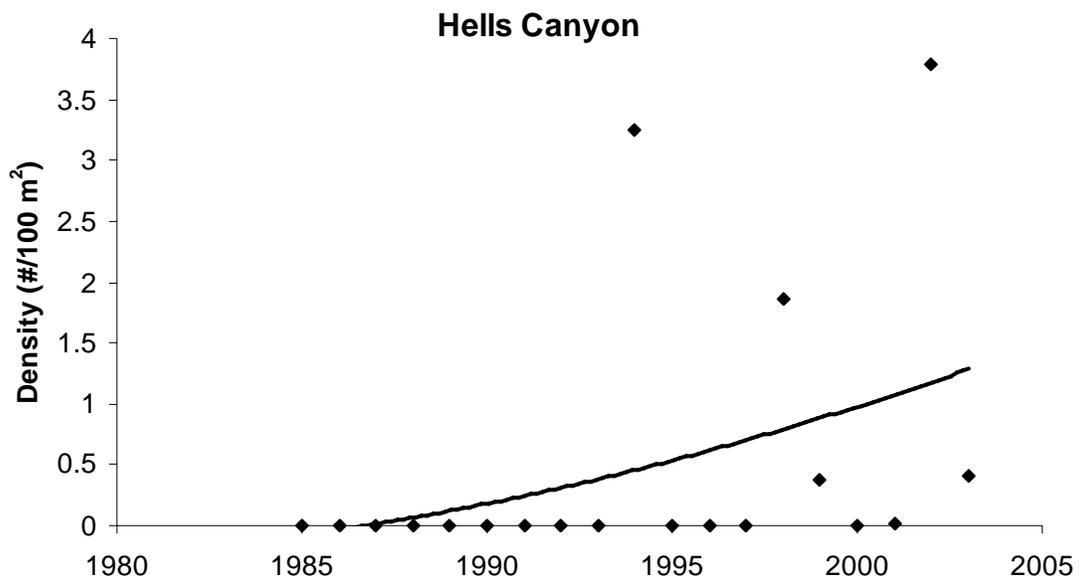












Appendix B. Table 1. Number of general parr monitoring sites sampled by year and fourth-field hydrologic unit code. CW = Clearwater, SR = Salmon River, MF = Middle Fork, SF = South Fork, MSC = Middle Salmon/Chamberlain, MSP = Middle Salmon/Panther.

| Year | Hells Canyon | Lemhi | Little Salmon | Lochsa | Lower CW | Lower MFSR | Lower Selway | Lower Salmon | MFCW | MSC | MSP | Pahsimeroi | SFCW | SFSR | Upper MFSR | Upper Salmon | Upper Selway |
|------|--------------|-------|---------------|--------|----------|------------|--------------|--------------|------|-----|-----|------------|------|------|------------|--------------|--------------|
| 1985 | 6            | 10    | 9             | 14     | 24       | 10         | 2            | 3            |      | 7   | 1   | 5          | 20   | 8    | 76         | 35           | 2            |
| 1986 | 8            | 6     | 20            | 19     | 25       | 22         |              | 7            |      | 13  | 3   | 6          | 34   | 35   | 61         | 32           | 8            |
| 1987 | 8            | 6     | 13            | 59     | 20       | 15         | 2            | 7            |      | 12  | 5   | 10         | 34   | 36   | 36         | 54           | 19           |
| 1988 | 6            | 6     | 13            | 55     | 18       | 21         | 5            | 7            |      | 9   | 8   | 15         | 39   | 34   | 55         | 61           | 14           |
| 1989 | 8            | 6     | 14            | 56     | 14       | 20         | 5            | 7            |      | 13  | 3   | 7          | 47   | 25   | 62         | 67           | 14           |
| 1990 | 8            | 6     | 24            | 29     | 53       | 25         | 26           | 4            |      | 5   | 5   | 5          | 88   | 48   | 47         | 63           | 16           |
| 1991 | 9            | 6     | 15            | 70     | 22       | 33         | 6            | 7            | 19   | 13  | 4   | 7          | 184  | 18   | 65         | 108          | 37           |
| 1992 | 4            | 5     | 22            | 128    | 20       | 32         | 10           | 11           | 20   | 37  | 6   | 7          | 141  | 67   | 56         | 105          | 40           |
| 1993 | 5            | 6     | 22            | 76     | 20       | 25         | 10           | 12           |      | 19  | 7   | 7          | 95   | 53   | 52         | 98           | 31           |
| 1994 | 5            | 37    | 14            | 58     | 38       | 8          | 10           | 14           | 20   | 34  | 39  | 7          | 149  | 85   | 104        | 140          | 53           |
| 1995 | 4            | 12    | 10            | 32     | 78       | 9          | 9            | 13           | 18   | 10  | 16  | 5          | 82   | 33   | 24         | 88           | 19           |
| 1996 | 4            | 19    | 18            | 36     | 121      | 11         | 9            | 10           | 20   | 23  | 2   | 4          | 105  | 25   | 18         | 44           | 37           |
| 1997 | 4            | 35    | 15            | 82     | 28       | 15         | 9            | 8            | 18   | 13  | 6   | 5          | 112  | 31   | 24         | 31           | 25           |
| 1998 | 4            | 9     | 14            | 45     | 29       | 9          | 9            | 8            | 5    | 13  |     | 1          | 91   | 26   | 31         | 27           | 12           |
| 1999 | 4            | 12    | 20            | 44     | 38       | 26         | 6            | 14           | 5    | 28  | 4   | 8          | 113  | 31   | 43         | 27           | 19           |
| 2000 | 4            | 11    | 21            | 15     | 11       | 17         | 4            | 13           |      | 9   | 3   | 8          | 187  | 27   | 53         | 22           | 2            |
| 2001 | 4            | 11    | 22            | 46     | 16       | 12         | 4            | 12           |      | 13  | 6   | 8          | 80   | 32   | 42         | 33           | 13           |
| 2002 | 2            | 11    | 22            | 25     | 17       | 15         | 9            | 9            |      | 12  | 9   | 13         | 76   | 29   | 27         | 30           | 19           |
| 2003 | 2            | 11    | 21            | 23     | 19       | 11         | 4            | 10           |      | 11  | 6   | 9          | 65   | 34   | 22         | 29           | 13           |

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