

Bull trout population assessment in northeastern Oregon: a template for recovery planning

Annual Progress Report for 2004

by

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28 September 2005

TABLE OF CONTENTS

	<u>Page</u>
ACKNOWLEDGMENTS	vi
PREFACE.....	vii
EXECUTIVE SUMMARY.....	viii
CHAPTER 1: Monitoring and evaluation of bull trout populations in northeastern Oregon.....	1
INTRODUCTION	1
STUDY AREA.....	3
METHODS	6
RESULTS	12
DISCUSSION.....	16
FUTURE.....	18
LITERATURE CITED	19
CHAPTER 2: Understanding the demography and significance of redd counts for bull trout.....	47
INTRODUCTION	47
METHODS.....	48
RESULTS.....	52
DISCUSSION.....	53
LITERATURE CITED	55
CHAPTER 3: Development of a microhabitat model for bull trout with explicit consideration of the transferability of habitat preferences across three streams....	58
INTRODUCTION	58
METHODS.....	60
RESULTS.....	64
DISCUSSION.....	67
LITERATURE CITED	70
APPENDIX 1: Appended figures.....	77
APPENDIX 2: Original Study Plan objectives and tasks specified to meet the overall 5-year project goals.....	85

TABLE OF CONTENTS

APPENDIX 3: Proposal submitted for extended work on the North Fork John Day and South Fork Walla Walla rivers. The focus of this research is two-fold: (1) assessment of demography, movement, and population dynamics of bull trout, and (2) evaluation of the importance of salmon in terms of bull trout consumption, growth, and survival.....	86
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LIST OF TABLES

		<u>Page</u>
Table 2.1	Estimates and methodologies used to estimate the number of bull trout per redd from the Columbia River Basin. An average of these four values (2.68 bull trout per redd) was used to expand redd count data in comparisons to population estimates from mark-recapture data.....	51
Table 2.2	Population estimates (with 95% confidence intervals in parentheses) of bull trout based on different sources and calculation methods for four streams and different years. Asterisks indicate inadequate data.....	53
Table 3.1	Summaries of microhabitat use and availability data for the NFUM, SFWEN, and SFWW rivers and a <i>Subset</i> (described in text). Means (with standard deviation in parentheses) are reported for each parameter. Cover was combined and reported as the percent of cells, either used or available, that had cover. Microhabitat use is separated by bull trout size classes: < 220 mm and > 220 mm. Sample size (<i>n</i>) corresponds to each analysis.....	64
Table 3.2	Parameter estimates with standard error, odds ratio point estimates, and <i>P</i> -values for explanatory variables from logistic regression analysis (<i>Full model</i>).	66
Table 3.3	Table 3.3. Parameter estimates with standard error, odds ratio point estimates, and <i>P</i> -values for explanatory variables from the <i>Subset</i> logistic regression analysis.	66

APPENDIX

Table A1	Summary of all fish captured, tagged, and counted (sighted during snorkeling surveys) in sampled reaches of the South Fork Walla Walla, North Fork Umatilla, and South Fork Wenaha rivers, June to August 2004. All sizes combined. No capture activities (<i>na</i>) were conducted in the SF Wenaha River.	78
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LIST OF FIGURES

		<u>Page</u>
Figure 1.1	Map of the South Fork Walla Walla River showing the study reaches (dots).....	23
Figure 1.2	Map of the North Fork Umatilla River showing the study reaches (squares).....	24
Figure 1.3	Length frequency distribution of bull trout captured and handled in the South Fork Walla Walla River and North Fork Umatilla River, 2004. Note changes in y-axis scales.....	25
Figure 1.4	Length-weight relationship for bull trout tagged in the South Fork Walla Walla River, 2004 (open circles) and North Fork Umatilla River, 2004 (black squares). Regression equations and sample sizes are given.....	26
Figure 1.5	Condition (Fulton's $K \pm 1$ SE) of bull trout by size class sampled in the South Fork Walla Walla River (2002 - 2004) and North Fork Umatilla River (2003 - 2004). Sample sizes are given by error bars. $K = 1$ is reference line.....	27
Figure 1.6	Average condition (Fulton's $K \pm 1$ SE) of bull trout (all sizes combined) sampled in the South Fork Walla Walla River (2002 - 2004) and North Fork Umatilla River (2003 - 2004). Sample size is given below error bars.	28
Figure 1.7	Number of bull trout tagged by reach in the South Fork Walla Walla River, 2004. Reaches are numbered from bottom to top of the study site.	29
Figure 1.8	Length frequency distribution of bull trout tagged (black bars) and observed (via snorkel counts; gray bars) in the South Fork Walla Walla River, 2004.....	30
Figure 1.9	Number of bull trout by reach observed during snorkel surveys in the South Fork Walla Walla River, 2004. Reaches are numbered from bottom to top of the study site.	31
Figure 1.10	Number of bull trout tagged by reach in the North Fork Umatilla River, 2004. Reaches are numbered from bottom to top of the study site.	32
Figure 1.11	Length frequency distribution of bull trout tagged (black bars) and observed (via snorkel counts; gray bars) in the North Fork Umatilla River, 2004.....	33
Figure 1.12	Number of bull trout by reach observed during snorkel surveys in the North Fork Umatilla River, 2004. Reaches are numbered from bottom to top of the study site.....	34
Figure 1.13	Pooled mark-resight population estimates across the three temporally different sampling occasions on the North Fork Umatilla River, 2004. Night snorkel surveys are denoted by black bars and day snorkel surveys are represented by white bars.	35

LIST OF FIGURES

Figure 1.14	Total number of bull trout for each sampling occasion (different colored bars) that were marked (<i>M</i>), captured from day surveys (<i>C</i> -day), captured from night surveys (<i>C</i> -night), resighted from day snorkel surveys (<i>R</i> -day), and resighted from night snorkel surveys (<i>R</i> -night) from the three temporal reaches on the North Fork Umatilla River, 2004.	36
Figure 1.15	Mark-resight population estimates across spatial (3 reaches) and temporal (3 sampling occasions) differences in the North Fork Umatilla River, 2004. White bars denote estimates from day snorkeling, and black bars denote night snorkeling efforts.....	37
Figure 1.16	Instantaneous growth of two size classes of bull trout in the South Fork Walla Walla River over two annual periods.....	38
Figure 1.17	Monthly PIT-tag detections (recaptures) of small (120-320 mm; top panel) and large (> 320 mm; bottom panel) bull trout made at the Bear Creek detector, 2004.....	39
Figure 1.18	Monthly PIT-tag detections (recaptures) of small (120-320 mm; top panel) and large (> 320 mm; bottom panel) bull trout made at the Harris Park Bridge detector, 2004.....	40
Figure 1.19	Monthly movement (daytime and nighttime) of bull trout based on PIT-tag directional-detections (recaptures) made at the Harris Park Bridge and Bear Creek PIT-tag detectors, 2004.....	41
Figure 1.20	Monthly movement (daytime and nighttime) of small (120-320 mm) and large (> 320 mm) bull trout based on PIT-tag detections (recaptures) made at the Harris Park Bridge (top panels) and Bear Creek (lower panels) PIT-tag detectors, 2004.	42
Figure 1.21	Monthly directional movement (daytime and nighttime) of small (120-320 mm) and large (> 320 mm) bull trout based on PIT-tag detections (recaptures) made at the Harris Park Bridge detector, 2004..	43
Figure 1.22	Monthly directional movement (daytime and nighttime) of small (120-320 mm) and large (> 320 mm) bull trout based on PIT-tag detections (recaptures) made at the Bear Creek detector, 2004.	44
Figure 1.23	Daily temperatures (maximum, average, and minimum) recorded at Harris Park Bridge on the South Fork Walla Walla River, 2004..	45
Figure 1.24	Survival estimates (± 1 SE) for two size classes of bull trout in the South Fork Walla Walla River over the period 2002 to 2004.....	46
Figure 2.1	Population estimates for the SFWW, NFUM, LIC, and BSC rivers for years 2002, 2003, and 2004. The POPTOT, POP220, and POP370 are calculated from mark-resight data. Confidence intervals are not shown for estimates where sample size was too low. POPREDDS refers to the number of redds for each corresponding year and expanded where the lower error bound is for 1.2 bull trout per redd, the upper error bound is for 4.3 bull trout per redd, and the average is for 2.68 bull trout per redd (see Methods for more details).....	57

LIST OF FIGURES

Figure 3.1	A regional map of northeast Oregon and southeast Washington. Microhabitat use and availability data were collected on the South Fork Walla Walla (SFWW), North Fork Umatilla (NFUM), and the South Fork Wenaha (SFWEN) rivers.....	74
Figure 3.2	Habitat preference curves for depth, substrate, bottom water velocity, and cover type for the SFWW, SFWEN, and NFUM rivers, all bull trout combined, 2004.	75
Figure 3.3	Habitat preference curves for depth, substrate, bottom water velocity, and cover type for different size classes (< 220 mm; > 220 mm) of bull trout in the South Fork Walla Walla River, 2004.....	76
Figure 3.4	Proportion of optimal, useable, and unsuitable habitat using transferability criteria from the SFWW (2004), and tested on the NFUM and the SFWEN.....	77

APPENDIX

Figure A1	Length frequency distribution of bull trout captured in the South Fork Walla Walla River (2002 and 2003) and North Fork Umatilla River (2003); both streams, two years combined.....	79
Figure A2	Combined length-weight relationship for bull trout sampled in the South Fork Walla Walla and North Fork Umatilla rivers, 2002 - 2004. Regression equation, R ² -value, and sample size are given.....	80
Figure A3	Density of <i>O. mykiss</i> spp., Chinook salmon, and mountain whitefish estimated by snorkel counts in various reaches of the South Fork Walla Walla River, summer 2004. Note changes in y-axis scales.....	81
Figure A4	Proportion of <i>O. mykiss</i> spp. and mountain whitefish observed in each size class during snorkeling surveys on the South Fork Walla Walla River, summer 2004.....	82
Figure A5	Density of <i>O. mykiss</i> spp., Chinook salmon, and mountain whitefish estimated by snorkel counts in various reaches of the North Fork Umatilla River, summer 2004. Note changes in y-axis scales.....	83
Figure A6	Proportion of <i>O. mykiss</i> spp. and mountain whitefish observed in each size class during snorkeling surveys on the North Fork Umatilla River, summer 2004.....	84

Suggested citation:

Budy, P., R. Al-Chokhachy, K. Homel, and G.P. Thiede. 2005. Bull trout population assessment in northeastern Oregon: a template for recovery planning. 2004 Annual Progress Report to US Fish and Wildlife Service. UTCFWRU 2005(3):1-93.

ACKNOWLEDGMENTS

Funding was provided by the US Fish and Wildlife Service and the US Geological Survey, Utah Cooperative Fish and Wildlife Research Unit.

We would like to thank Howard Schaller for securing funding and providing guidance and support. Jim Gasvoda and Gayle Zydlewski of USFWS Abernathy Research Station contributed technical oversight and support. Don Anglin, Darren Gallion, Tad Kisaka, and Paul Sankovich of the USFWS CRFPO provided support. Temperature data were provided by the Walla Walla Basin Watershed Council (Bob Bower) and the USFWS CRFPO. Mary Conner provided biometric advice and is help with ongoing modeling efforts. Tim Whitesel provided assistance, guidance and support. Wayne Wurtsbaugh, Mary Conner, and Edward Evans reviewed previous drafts of manuscripts included as chapters. Dave Crabtree (USFS) and Jackie Dougan (BLM) provided information and support. We thank Tim Bailey (ODFW), Steve Starcevich (ODFW), Gretchen Sausen (USFWS), Craig Contor, Brian Mahoney, and Jesse Schwartz (CTUIR) for contributing information, data, support, and logistics. Dale and Judy Johnson provided parking, kindness, hospitality, and support, and Dale was also the caretaker of the lower Harris Park Bridge PIT-tag detector. George Ehmer maintained the upper Bear Creek detector. The North Fork Umatilla detector was aided by Jim Gasvoda, Paul Sankovich, and USFS personnel. Dwayne Wright provided mule transport. Thanks to Travis Sanderson, Douglas Martin, Andy Dean, Holly Campbell, Daniel Cullen, David Lillard, Keith Hofer, Nathaniel Charbonneau, and James Hearsey for summer technical support. Sarah Carothers, Travis Sanderson, and Andy Dean provided lab support and data entry.

Scientific Take Permit (OR2004-2059) was obtained with assistance from Laura Tesler and Mary Hanson of ODFW and Darren Gallion of USFWS CRFPO. Special Use Permit (WAL0162) was obtained from the USFS with the assistance of Jeff Bloom and Mary Gibson. Research was conducted under Utah State University IACUC Protocol 1082.

PREFACE

This report consists of three separate chapters. Chapter 1 contains results from the monitoring and evaluation of bull trout populations in northeastern Oregon, which have been monitored since 2003. This chapter also contains preliminary work by Kris Homel as part of her MS thesis. Chapter 2 is a manuscript that was submitted to the *North American Journal of Fisheries Management*, and is currently being revised for publication. Chapter 3 is a manuscript that was recently submitted to *Transactions of the American Fisheries Society*. Both Chapter 2 and Chapter 3 are part of Robert Al-Chokhachy's PhD dissertation research; therefore, we ask that data or information not be reproduced without permission from the author.

EXECUTIVE SUMMARY

Within the overall framework of conservation and recovery planning for threatened bull trout, this project provides critical information on abundance, trend, vital rates, habitat needs, and information on the potential for improving survival at one or more life stages. In addition, the project gathers information related to population structure (age, life history, and genetic components). We provide a template against which different strategies for monitoring and evaluation can be evaluated in terms of accuracy, precision, and cost/effort. The data and conservation assessment tools provided by this project will ultimately help guide the US Fish and Wildlife Service in determining the necessary courses of action and management actions for recovery of bull trout populations throughout this as well as other provinces. The project was initiated in 2002 and has continued through 2004, with plans to continue work through 2005. To meet our goals, we have developed and implemented each year, a comprehensive mark-recapture program including two tag types, multiple capture techniques (both passive and active) and systematic sampling of two large study areas (South Fork Walla Walla and North Fork Umatilla rivers) with a high degree of effort. We also study habitat needs of bull trout within a hierarchical arrangement of spatial scales and are evaluating the environmental and biological cues for migration as well as the potential for genetic distinction between life-history types.

We summarize our annual information on the population abundance, trend, and vital rates of bull trout in the South Fork Walla Walla River (SFWW) and the North Fork Umatilla River (NFUM) in Chapter 1. In 2004, all bull trout population-based research for the SFWW and NFUM was continued. We sampled 48% of the SFWW study area (22% re-sampled, 26% new), 410 bull trout were tagged; we sampled 40% of the NFUM study area, and 64 bull trout were tagged. All the baseline demographic and population assessment data were collected, analyzed, and synthesized. Mark-recapture population estimates in the SFWW and NFUM have generally remained stable across the three years of the study, with estimates of the total number of bull trout ranging from 7,000 to 10,000 for the SFWW and from 2,000 to 3,000 for the NFUM (2-years only; see Chapter 2).

In 2004, bull trout condition (Fulton's K) was similar as compared to previous years for the SFWW; however, in the NFUM, we observed a dramatic decrease in bull trout condition. Growth rates in the SFWW were substantially higher in 2004 as compared to 2003 with smaller bull trout growing at a much higher rate as compared to bull trout > 320 mm. In addition, survival rates for small and large bull trout are high (over 40%, 2002 through 2004). In the SFWW, most movement of larger fish (> 320 mm)

occurred between July and October, and most movement of small fish (< 320 mm) occurred between May and September. On a diel basis, more movement occurs at night as compared to during the day, especially for smaller fish < 320 mm. The NFUM PIT-tag detector was not completed and operating long enough in 2004 to collect data on bull trout movement.

Our comprehensive mark-recapture program and resulting robust population estimates provide a template against which to compare the accuracy and precision of various sampling procedures and to gain a better understanding of the contribution of different life stages and/or life-history forms to the overall population. In Chapter 2, we do this type of comparisons for redd counts in an attempt to gain a better understanding of the demography and significance of redd counts for bull trout. We use mark-resight population estimates as a comparison to annual redd counts for bull trout in three streams of eastern Oregon. Across basins there appears to be inconsistency between mark-recapture population estimates for different size classes of bull trout and population estimates based on expanded redd count data. In some systems, it is only the larger, potentially migratory fish that are represented in redd counts, and in others it appears to be some combination of both small, resident and large, potentially migratory fish. Our data suggest that trends between redd counts and mark-resight population estimates may be similar within basins across years, but not across different basins. The disparity between redd counts and population estimates for the reproductive population suggests that that caution should be invoked when choosing the monitoring techniques to be used to set recovery and/or monitoring goals for salmonid populations. All sample data are currently being used to develop a model for evaluating sampling variability and our ability to detect trend under various sampling protocols.

In Chapter 3, we take our first step towards developing a hierarchical framework for understanding bull trout habitat needs and both built and validated a microhabitat model for bull trout. We assessed the transferability of bull trout microhabitat relationships using multiple analytical approaches in three streams in northeastern Oregon. We established bull trout microhabitat preference curves for depth, bottom water velocity, substrate, and cover, and assessed the transferability of these preferences across systems. Transferability was assessed based on composite microhabitat classifications and chi-squared tests. To corroborate this approach, we also used logistic regression techniques to model bull trout presence/absence at the microhabitat scale, and validated this model across systems. Our results suggest that bull trout habitat preferences are generally consistent across systems and size classes of fish, as bull trout prefer deeper, slower-moving habitats with cover. However, preferences for substrate and cover types varied across both systems and

size classes of fish. Similar to the habitat preference results, our logistic regression analyses indicated that depth, velocity, and cover were significant parameters in predicting the presence/absence of bull trout. The validation process for the logistic model suggests that the model is effective in predicting both bull trout presence and absence. Overall, our results suggest that bull trout microhabitat use is consistent across systems. Ultimately, these results, and our validation of their general transferability, can help direct recovery and restoration efforts for bull trout across its native range. Future analyses will move from the microhabitat scale up to the channel type (e.g., pool), reach, stream, and watershed in an attempt to evaluate the contribution of key factors at these different spatial scales in explaining bull trout habitat preference.

In 2004, we also completed a pilot study of the feasibility of expanding our research into the John Day Subbasin. Based on a review of the gray literature and fish distribution, in combination with discussions with local biologists, the North Fork John Day River was proposed as the future study reach, and we completed a preliminary field assessment of this area.

CHAPTER 1: Monitoring and evaluation of bull trout populations in northeastern Oregon

INTRODUCTION

When species are in decline or listed under conservation status across a large spatial area, estimates of population abundance and trend are critical for understanding the present and future status of the population (Soule 1987). In addition, the quantification of key demographic parameters (e.g., survival, growth) is an important part of the process of identifying factors that potentially limit the population and understanding the role of these vital rates in determining overall trend. However, for many protected species, estimation of population abundance and demographic parameters is extremely difficult due to (1) their protected status, which limits estimation techniques that may be applied legally, (2) low numbers, (3) high variability, (4) the differential effects of environmental stochasticity at low abundance, (5) the immediate, short-term need for information that typically requires years to collect, and (6) logistical limitations in agency personnel time and/or funding. Nevertheless, population structure (including genetics), abundance, trend, and demographic characteristics are key components required for the recovery planning of any species.

In 1998, bull trout (*Salvelinus confluentus*) were officially listed as a Threatened Species under the 1973 Endangered Species Act (USFWS 1998). Bull trout are native to the northwestern United States and western Canada and are primarily an inland species distributed from the southern limits in the McCloud River in California and the Jarbridge River in Nevada to the headwaters of the Yukon River in Northwest Territories (Cavender 1978). Resident and migratory populations exist within this range and can coexist, representing a diverse population structure (Goetz 1991; Rieman and McIntyre 1993; see also Chapter 2). Habitat degradation (Fraley and Shepard 1989), barriers to migration (Rieman and McIntyre 1995; Kershner 1997), and the introduction of nonnatives (Leary et al. 1993) have all contributed to the decline in bull trout populations of the Columbia River Basin and the Klamath River Basin. Today, bull trout exist only as subpopulations over a wide range of their former distribution (Rieman et al. 1997), and several local extirpations have been documented.

The goal of bull trout recovery planning by the US Fish and Wildlife Service (USFWS) is to describe courses of action necessary for the ultimate delisting of this species

under the Endangered Species Act, and ensure the long-term persistence of self-sustaining, complex interacting groups of bull trout distributed across the species's native range (Lohr et al. 1999). To meet this overall goal, the USFWS has identified several objectives which require the type of information that will be provided by this project: (1) maintain current distribution of bull trout within core areas in all recovery units and restore distribution where needed to encompass the essential elements for bull trout to persist, (2) maintain stable or increasing trends in abundance of bull trout in all recovery units, and (3) restore and maintain suitable habitat conditions for all bull trout life-history stages and strategies. Further, the USFWS recovery-planning document (Lohr et al. 1999) embraces the idea of core areas. Conserving respective core areas within conservation units is intended to preserve genotypic and phenotypic diversity and allow bull trout access to diverse habitats. The continued survival and recovery of individual core area populations is thought to be critical to the persistence of conservation units and their role in overall recovery of the Columbia River distinct population segment (Whitesel et al. 2003).

Despite the growing body of knowledge on bull trout biology and distribution (e.g., presence/absence information or patch occupancy), there are still critical gaps in our information at the population level (Porter and Marmorek 2005). First, little is known about the structure of these populations (e.g., migratory versus resident and age) and/or the importance of different life-history forms to both the status of the population overall and the pattern of genetic variation. Further, the distinction between the two life-history forms, whether genetic or behavioral, has rarely been demonstrated, and recovery goals are vague as to which component of the population is included (Marmorek and Porter 2004). These population-structure issues may be especially problematic, as migratory bull trout have likely experienced greater declines, as compared to resident forms, due to their attempts to move through fragmented landscapes that often exist below their higher elevation, spawning and rearing tributaries (Rieman and McIntyre 1993). Second, while there are an increasing number of PVA's that have been completed (e.g., IDFG 2004), there are still relatively few cases where the status (e.g., trend) of the population has been estimated based on field-derived abundance data and/or evaluated with an acceptable level of certainty, usually because the data simply are not available or uncertainty in trend estimation is high when using relatively short time series of data (Schaller et al. 2005). And finally, there are even fewer cases where the factors that may limit the population, natural or anthropogenic, have been fully identified (Rieman and McIntyre 1993; Buchanan et al. 1997; USFWS 1998). These limitations are important to overcome, in order for proper status evaluation and for identifying management actions aimed at recovery (Nielsen and Spruell 2001; Meffe et al. 1997).

Within this overall framework of conservation and recover planning for threatened bull trout, this project provides critical information on bull trout population abundance, trends in abundance, vital rates, habitat needs, and information on the potential for improving survival at one or more life stages. In addition, the project gathers information related to population structure (age, life history, and tissue for genetic information). Further, we provide a template against which different strategies for monitoring and evaluation can be evaluated in terms of accuracy, precision, and cost per effort. The data and conservation assessment tools provided by this project will ultimately help guide the USFWS in determining the necessary courses of action and management actions for recovery of bull trout populations throughout this as well as other provinces; preliminary data from 2002-2004 are currently being used by the USFWS Bull Trout Recovery, Monitoring, and Evaluation Technical Group (RMEG).

The South Fork Walla Walla River was initially selected as the comprehensive study area due to its potential as a core area for bull trout in the Columbia River Basin, complex and potentially contentious water management issues associated with fish protection, a diversity of habitat types, and an abundance of fish. In addition, selected project goals (e.g., validation of habitat modeling) have required the extension of monitoring and evaluation into other nearby watersheds (e.g., North Fork Umatilla River), where smaller-scale evaluations are also underway.

This chapter covers fish sampling including assessment of sampling efficiency, along with bull trout demographic parameters and dynamics including growth; emigration, immigration, and movement; genetics; migration cues and timing, population abundance and trends, and survival.

STUDY AREA

South Fork Walla Walla River

The Walla Walla River in northeastern Oregon and southeastern Washington is a tributary of the Columbia River that drains an area of 4,553 km² (Walla Walla Subbasin Summary Draft 2001). The tributaries of the Walla Walla River originate in the Blue Mountains at elevations near 1800 m. The mainstem Walla Walla flows for approximately 16 km in Oregon before splitting into the NF Walla Walla and the SF Walla Walla rivers.

The Walla Walla River historically contained a number of anadromous and resident, native salmonid populations including: spring and fall Chinook salmon (*O. tshawytscha*), chum salmon (*O. keta*), and coho salmon (*O. kisutch*), redband trout (*O. mykiss* subpopulation), bull trout, mountain whitefish (*Prosopium williamsoni*), and summer steelhead (*O. mykiss*; the extent of fall chinook, chum, and coho salmon is not known; Walla Walla Subbasin Summary Draft 2001). Today, steelhead represents the only native anadromous salmonid still present in the Walla Walla River system. However, since 2000 there has been annual supplementation of adult chinook in the SF Walla Walla River by the Confederated Tribes of the Umatilla Indian Reservation (CTUIR). Populations of native redband trout, bull trout, and mountain whitefish still persist in the Walla Walla River, as well as introduced brown trout (*Salmo trutta*).

Little documentation exists on the historical distribution of bull trout in the Walla Walla Subbasin prior to 1990. Anecdotal evidence suggests that large fluvial bull trout were found to utilize the Columbia River. Telemetry studies in the mid-Columbia River region have shown bull trout have to use both primary and secondary tributaries for spawning (FERC Project 2145 Draft 2002). Therefore, it is presumed that bull trout had access to the Columbia River and all of its tributaries prior to the impoundment of the Columbia River (Buchanan et al. 1997). Today, resident and fluvial forms of bull trout exist in the Walla Walla (Walla Walla Subbasin Summary Draft 2001), and both populations spawn in the tributaries and headwaters of the Walla Walla River. However, recent telemetry studies with large (> 350 mm) bull trout have not confirmed use of the Columbia River (Mahoney 2001, 2002).

Within the Walla Walla River Basin, bull trout are arbitrarily divided into four populations based on geography: North Fork Walla Walla River, South Fork Walla Walla River, Mill Creek, and the Touchet River (Buchanan et al. 1997). Ratliff and Howell (1992) described the population status of bull trout as “low risk” in the SF Walla Walla River and Mill Creek, and “of special concern” in the NF Walla Walla River. Since that report, the status of the SF Walla Walla population has remained at low risk, but both the NF Walla Walla River and Mill Creek populations have been upgraded to “high risk” and “of special concern” respectively (Buchanan et al. 1997). Alterations to migratory corridors linking these populations have occurred, but the degree of genetic, geographical isolation is unknown.

The study site on the SF Walla Walla River spans nearly 21 km in length. The upper boundary was set at the confluence with Reser Creek (Reach 103), and the lower boundary was set above Harris Park Bridge (on public, county land; Budy et al. 2003,

2004). In order to account for spatial variation of the study site and the distribution of bull trout, the study site was divided into 102 reaches, 200-m each, using Maptech mapping software (Figure 1.1).

An initial site was randomly selected from the list of reaches, and thereafter every fifth reach (an approximate 20% sample rate) was systematically designated for sampling in 2002. The UTM coordinates from the mapping software were used to locate the general location of the bottom of each reach, and the closest pool tail to the coordinates was set as the true reach boundary. The reach continued upstream for at least 200 m and the top was set at the first pool-tail above the 200-m mark. Total length was recorded for each reach. Location coordinates (UTM using GPS) were recorded at the boundaries of each reach.

North Fork Umatilla River

The Umatilla River Basin drains an area of approximately 6,592 km². The Umatilla River is 143 km long from mouth (at Columbia River RK 440) to where it divides into the NF and SF Umatilla rivers, each fork adding another 16 km in length. The Umatilla mainstem originates in Blue Mountains at 1289 m and descends to 82 m at confluence with Columbia River. Earliest documentation of bull trout in Umatilla basin is from ODFW creel reports dating from 1963. The mainstem Umatilla River is artificially confined for much of its length. Spawning occurs in the NF and SF Umatilla rivers, and in NF Meacham Creek. Along with being an important tributary for rearing and migration activities, redd counts indicate that the majority of redds in the Umatilla basin occur in the NF Umatilla River between Coyote and Woodward creeks. Peak spawning generally occurs between mid September and mid October over at least a two-month period (ODFW 1995, 1996) when daily average water temperatures ranged from 6-10 °C (ODFW 1996). Habitat in the NF Umatilla River is fairly complex with low levels of bedload movement, moderate levels of large organic debris, and relatively minimal flow events. Other species occurring in the basin include *O. mykiss* subspecies, sculpin (*Cottus* spp.), Chinook salmon, shiners, suckers (*Catostomus* spp.), dace (likely *Rhinichthys* spp.), and pikeminnow (*Ptychocheilus oregonensis*). Two populations were recognized in the Umatilla basin: the NF Umatilla River rated “Of Special Concern” and the SF Umatilla River rated at “High Risk” (Buchanan et al. 1997).

The study site on the NF Umatilla River spans nearly 5 km in length. The upper boundary was set at the confluence of Johnson, Woodward, and Upper NF Umatilla creeks (416053 E, 5065070 N), and the lower boundary was set at the confluence of

NF and SF Umatilla rivers (110407763 E, 5064070 N). In order to account for spatial variation of the study site and the distribution of bull trout, the study site was divided into 21 reaches, approximately 200-m each, using Maptech mapping software (Figure 1.2).

An initial site was randomly selected from the list of reaches, and thereafter every fifth reach (an approximate 20% sample rate) was systematically designated for sampling in 2003. The UTM coordinates from the mapping software were used to locate the general location of the bottom of each reach, and the closest pool tail to the coordinates was set as the true reach boundary. The reach continued upstream for at least 200 m and the top was set at the first pool-tail above the 200-m mark. Total length was recorded for each reach. Location coordinates (UTM using GPS) were recorded at the boundaries of each reach.

Multiple agencies are conducting coordinated research in our study area. Oregon Department of Fish and Wildlife (ODFW) and Washington Department of Fish and Wildlife (WDFW) currently conduct research on the distribution and movement of bull trout within the lower sections of the Walla Walla River and Mill Creek (a tributary of the Walla Walla River), and limited movement and status monitoring occurs in the other tributaries. In addition, the USFWS is evaluating habitat limitations for juvenile and adult fishes in the migratory corridor below our study area.

METHODS

Fish Sampling

Capture—Multiple sampling techniques were used to capture bull trout including angling, electroshocking down to a seine, trap netting, and minnow trapping. All captured bull trout were weighed (nearest 0.1 g), measured (nearest mm total length, TL), and condition (K_{TL}) was calculated (Fulton's $K_{TL} = W / L^3 * 100,000$). Scales were taken from a subsample of live, released fish. A small subsample of adults was taken for fecundity and sex ratio estimates. We also obtained information from mortalities (non-project related) found in each stream. From these subsamples, stomachs and hard parts (e.g., otoliths) were removed for age, growth, and diet analyses.

Marking—Bull trout (> 120 mm TL) were marked with unique PIT tags and T-bar anchor tags (Floy tags), and subsequently recaptured using a combination of passive PIT-tag array antennae (see below) and snorkeling resights. Prior to tagging, bull

trout were anesthetized until they exhibited little response to stimuli. A 23-mm PIT tag was then placed into a 7-mm surgical incision on the ventral side of the fish, anterior to the pelvic fins. No sutures were required for closure of the incision. In addition, an external T-bar anchor tag, unique to year and stream, was inserted adjacent to the dorsal fin. After surgery, scales were taken from the right side at the base of the dorsal fin for aging and growth information, and fish were placed in a flow-through recovery container within the channel, and monitored until full equilibrium was restored. All fish were returned to slow-water habitat near individual capture locations.

Resighting—To resight Floy-tagged fish, we conducted daytime bull trout snorkel surveys in 41 reaches (mean reach length = 240 m) of the South Fork Walla Walla River (SFWW), and 15 reaches (mean = 212 m) of the North Fork Umatilla River (NFUM) in 2004. To avoid double-counting fish, snorkeling surveys started at the highest reaches working downstream to the bottom of the study site, because many fish were migrating to the headwaters for spawning. This approach likely minimized the incidence of double counts. Water temperature, start, and end times were all recorded for each snorkeling session. All bull trout (tagged and untagged) and were enumerated and placed into 50-mm size classes, and all *O. mykiss* spp. and juvenile Chinook salmon were enumerated but not delineated by size. Accurate identification of fish species and size estimation was emphasized. In each channel unit snorkeled, two observers proceeded in an upstream direction while scanning for fish across their assigned lane, such that the entire channel was surveyed. Incidental juvenile steelhead trout were counted and classified as age-1+ (76 – 127 mm) and age-2+ (> 127 mm), according to the size classes of the Idaho Department of Fish and Game General Parr Monitoring program. Snorkel-sample reach lengths were measured so that fish density (number per 100 m²) could be determined.

Recapture—Tagged bull trout were recaptured one month after PIT tagging; recaptures will continue for the duration of the study (minimum of two years). We began recapturing tagged and untagged individuals using a combination of techniques: seining, trap netting, and pass-through PIT-tag technology described below. Recaptured fish were passed over a handheld PIT-tag detector, and all information about each individual fish was retained electronically. In addition, tagged bull trout were and will be recaptured (and resighted during snorkeling surveys) and released for the duration of the study to provide annual estimates of survival, annual population estimates, and to parameterize the Pradel mark and recapture model. Recapture location will also provide information about movement and subpopulation versus metapopulation structure (see also below). Again, all captured bull trout were

weighed and measured before release, to obtain information about annual growth rates and the effects of fish size on survival.

Passive fish detection—PIT-tag passive antennae arrays (detectors) have been installed in-stream and continuously collect information on tagged bull trout from two locations within the SFWW. One detector is located at Harris Park Bridge (UTM coordinates: 110408261 E, 5076370 N) at the bottom of the study site, and the second detector is located just above the confluence with Bear Creek (approximately 7 km upstream; UTM coordinates: 110414281 E, 5077108 N). The Harris Park Bridge tag detector (WW1) has been running since mid-September 2002, and the Bear Creek detector (WW2) has been operational since mid-October 2002. Both detectors are linked either through phone or satellite, and data is uploaded to the PTAGIS website (<www.psmfc.org/pittag/Data_and_Reports/index.html> under "*Small-scale Interrogation Site Detections -Query*").

The NFUM tag detector (UM1) was constructed on US Forest Service land under a road bridge (UTM coordinates: 110407659 E, 5064089 N) near the confluence with the Umatilla River. Construction was finished in July 2004, and the detector was collecting data by late autumn 2004.

Assessment of sampling efficiency

We also investigated how spatial and temporal differences affected our mark-resight sampling efficiency. Specifically, we examined the potential biases associated with mark-resight data when sampling different reaches at different times of the year. To measure the spatial differences among reaches, we selected three reaches in the NFUM (Reach numbers 26, 35, and 42) which were approximately 4.5, 6, and 7 km from the bottom of the study site, and represented areas of low, moderate, and high bull trout densities. The lowest reach (26) was near the bottom of the known spawning area, while reaches 35 and 42 were both within the core spawning area in the NFUM. To measure the temporal differences across and within reaches, we sampled each reach three times throughout the 2004 field season (early July, the end of July, and the middle of August). We also selected reaches in the SFWW; however, due to high flows our efforts failed (see Results for details).

For each sampling occasion, we double block-netted both the top and bottom of the reaches prior to sampling, and all nets remained situated until the completion of snorkeling surveys. Block-nets were approximately 2 m high, and contained a 6-mm

(1/4-inch) mesh size. To minimize the number of bull trout escaping the reaches, the block-nets were buried within the substrate, and spanned the entire wetted width of the stream.

Sampling procedures within each reach were similar to efforts in both the SFWW and NFUM, where all captured bull trout larger than 120 mm were tagged with both a 23-mm PIT tag and an external anchor tag. Snorkeling surveys for resights began approximately 24 hours after the completion of the sampling efforts, and were performed during both daylight and nighttime hours to examine potential biases associated with day-night snorkeling efforts. Block-nets were removed at the end of each sampling occasion.

Growth

Growth information was obtained from SFWW bull trout tagged in 2003 and recaptured in 2004. Length and weight gains were determined between initial tagging and subsequent capture events. These length and weight gains were evaluated based on annual growth, summer peak season growth, and instantaneous growth per size class. Instantaneous growth rate (g/g/day) was calculated as

$$G = [(\ln W_{final} - \ln W_{initial}) / t] \times 100$$

where W is the weight (mass) in grams and t is trial length in days, and averaged (± 1 SE) by size class. Growth estimates were not made for NFUM bull trout due to low recapture rates.

Emigration, Immigration, and Movement

Bull trout movement information from in-stream passive PIT-tag detectors was obtained from two locations within the SFWW. One tag detector is located at Harris Park Bridge at the bottom of the study site, and the second detector is located just above the confluence with Bear Creek (approximately 7 km upstream). These PIT-tag detectors record movement date and time, and with multiple detections, we can infer movement direction. While the direction of fish movement cannot be detected with a single detector in the SFWW there are two detectors about four miles apart, so direction of movement can be inferred for fish that swim past both detectors. Day versus night movement was defined based on presence of visible light. The times associated with day versus night changed in monthly increments.

Genetics

Determining life-history form is a complicated process because different forms coexist (Jakober 1992), may interbreed (through “sneaking”) and potentially give rise to one another (Balon 1984; Gross 1991), exist in intermediate forms (Schrank and Rahel 2004), or exhibit low levels of genetic variation within populations (Leary et al. 1993, Spruell et al. 1999; Kanda and Allendorf 2001). Despite these difficulties, it is critical to understand the proportional composition of life-history forms in a population so that recovery efforts can be life-history specific. Sensitive genetics techniques (i.e., microsatellite nuclear DNA studies) provide a fine scale method to analyze within population variation occurring on a relatively short time scale. Because microsatellites are highly variable, they are an excellent tool for assessing variation between life-history forms.

Genetic fin clip samples were collected from both the main stem of the SFWW, and the two main tributaries (Skihorton and Reser creeks). A small anal fin clip (4 -25 mm²) was removed from all tagged bull trout (> 120 mm TL) sampled in the mainstem SFWW, along with 41 small fish (< 120 mm TL) sampled in Skihorton and Reser creeks (tributaries of the SFWW) for genetic processing. These fin clips were stored in 95% ethanol in individual vials labeled with a unique code. By sampling on a broad spatial scale, we minimized encounters with sibling groups (which provide little useful life-history genetic information). We also targeted areas that would be more likely to support either a resident (tributary samples) or migratory (mainstem samples) life-history form. From a collection of 1196 genetic fin-clip samples, 60 samples from the SFWW were selected, given an *a priori* life-history designation, and sent to the USFWS Abernathy Fish Technology Center for genetic processing. The life-history designations were broken into three groups: 20 known migratory, 15 likely residents, and 20 tributary samples. An additional 5 samples from euthanized fish (“takes”) were included in order to compare diet analysis and microchemistry to life history. The known migratory group consisted of samples from our largest fish that were detected swimming downstream past the Harris Park Bridge PIT-tag detector. As these fish were leaving the study area and making a discrete shift in habitat type, we considered them migratory. Likely resident fish were those that had been recaptured in multiple years (often in a proximate location), had never been detected at a PIT-tag detector, and were between 150 and 450 mm. While resident fish rarely exceed 320 mm (Goetz 1989), these larger-sized fish had been recaptured twice since 2002 and had not been detected at the detector antennae. The tributary samples may represent the resident life-history form, because the tributaries are quite small and would be unlikely to support larger migratory fish. However, since

these fish were not tagged (as the tributaries are not part of our tagging operation) no life-history designation was made. These fifteen samples, along with the five “takes”, will be considered as an unknown group.

All samples will be evaluated at 13 microsatellite loci that were developed for bull trout at the USFWS Abernathy Fish Technology Center. Currently, genetic processing is in its incipience and results will be forthcoming.

Migration Cues and Timing

In addition to quantifying any genetic structure between migratory and resident life-history forms, we are also evaluating potential behavioral plasticity as demonstrated by variable migration timing, in response to environmental and/or biological migration cues. Environmental cues include, but are not limited to: 1) stream flow, 2) water temperature and degree days, 3) ambient air temperature, 4) day length, and 5) precipitation. Biological cues include fish body size at migration time, and presence of spawning adults. Many of these cues (e.g., water temperature, day length, flow, and hydrologic events) have been associated with diel or annual migrations or spawn time (Quinn and Adams 1996; Fraley and Shepard 1989; Bjorn 1971) and may also be correlated with seasonal migration timing. In order to minimize the number of confounding variables that may affect migration, we will focus on first time downstream migration of subadults.

Temperature data were collected at the Harris Park Bridge and were provided by the Walla Walla Basin Watershed Council and the USFWS (*unpublished data*).

Survival

Survival estimates were calculated using a Cormack-Jolly-Seber model from mark-recapture data collected from 2002-2004. This is an open mark-recapture model, which incorporates the number of marked and recaptured fish in different time intervals. For this model, we selected the following specific time intervals: (1) the first time interval corresponds to the 2002 summer field-season period; (2) the second interval corresponds to fish detected at either of the in-stream PIT-tag detectors or actively recaptured during the 2003 summer field season; (3) the third interval corresponds to the fish detected at either of the in-stream detectors or actively recaptured during the 2004 summer field.

We partitioned our data into the following three size classes in order to better understand how survival varies by age/stage: 120 - 220 mm fish, representing

subadult bull trout; 220 - 320 mm fish, representing smaller, potentially resident bull trout; and fish > 320 mm, representing larger, potentially fluvial bull trout. Survival estimates and recapture probabilities were calculated using Program MARK software.

RESULTS

Fish Sampling

Bull trout were captured or observed in almost all sampled reaches. Length frequency distributions of captured bull trout in 2004 were similar in both streams, but with a greater proportion of larger fish in the SFWW (Figure 1.3). All captured bull trout were weighed and measured, and a separate length-weight relationship was calculated for each stream based on all measured bull trout (Figure 1.4). Additionally, bull trout > 120 mm TL were tagged with both an external Floy tag and a 23-mm PIT tag. Prior to release, scales were removed from the right, posterior position adjacent to the dorsal fin. Bull trout smaller than 120 mm were simply measured, weighed, and immediately released.

Condition—Condition (Fulton's K) of bull trout varied by size class and year (Figure 1.5). In the SFWW, average condition was similar in 2003 and 2004, with condition values ranging from 0.85 (± 1 SE = 0.01) for < 120 mm fish to 0.93 (± 0.01) for > 320 mm fish. The condition of bull trout captured in the NFUM decreased significantly from 2003 to 2004 (mean K = 0.92 and 0.85, respectively; $t = 5.43$, $df = 359$, $P < 0.001$; Figure 1.6). In the NFUM, condition was also highest for the largest bull trout (1.07 \pm 0.01) and lowest for bull trout > 120 mm (0.83 \pm 0.01; Figure 1.5).

South Fork Walla Walla River

We sampled 41 reaches during the 2004 field season, which accounted for approximately 48% of the study site. Over the summer, a total of 771 bull trout were captured of which, 410 were tagged, with the number tagged varying by sample reach (0 - 29 per reach; Figure 1.7; Table A1). The average bull trout captured was 164 mm (± 1 SE = 3.3) and 93.6 g (± 8.4). The smallest bull trout captured was 50 mm (0.9 g) and the largest bull trout caught was 608 mm (2038 g); however, the greatest proportion of bull trout captured or observed were in the 100 to 200 mm size range (Figure 1.8).

Snorkel surveys—Snorkeling surveys were performed in 41 reaches in 2004. Observations were biased toward fish > 120 mm due to the cryptic nature of small fishes (Figure 1.9). In 2004, bull trout observed ranged from 50 to 700 mm, similar to 2002 and 2003 surveys.

North Fork Umatilla River

We sampled 15 reaches in 2004 which accounted for 40% of the study site. Bull trout were captured or observed in all sampled reaches. Over the summer, a total of 64 bull trout were tagged, with the number tagged varying by sample reach (1 to 12 per reach; Figure 1.10; Table A1). The average bull trout captured was 130 mm (± 1 SE = 3.8) and 34.8 g (± 7.27). The smallest bull trout captured was 32 mm (0.7 g) and the largest bull trout caught was 471 mm (1163 g); however, the greatest proportion of bull trout captured or observed were in the 100 to 200 mm size range (Figure 1.11). Comparisons of the size-frequency distribution of bull trout captured and tagged to the size frequency distribution of bull trout observed during snorkeling surveys indicates we had an equal probability of sampling bull trout by both methods (Figure 1.11).

Snorkel surveys—Snorkeling surveys were performed in 15 reaches. Bull trout were observed in all of the sampled reaches. Observations were biased toward fish > 120 mm due to the cryptic nature of small fishes. A similar size distribution of bull trout was observed as in the SFWW, although 2.5-times fewer fish were seen (Figure 1.12).

Assessment of sampling efficiency

North Fork Umatilla River—The results from the NFUM suggested that there are no consistent temporal patterns across reaches. In particular, no trends were evident in population estimates from pooled mark-resight data across sites (Figure 1.13). In addition, there does not appear to be a pattern evident in the number of marked, captured (visually during snorkeling surveys), and resighted bull trout temporally (Figure 1.14). However, within sites, there appear to be different temporal effects across the study area. In particular, we found that through time, mark-resight population estimates decreased in Reach 26, increased in Reach 35, and remained relatively constant in Reach 42. Across the three reaches, there were little or no differences between day-night snorkeling surveys, which would suggest that no day-night patterns exist in the NFUM (Figure 1.15).

South Fork Walla Walla River—Due to the size and nature of the stream, efforts to quantify temporal and spatial patterns failed in the SFWW. In particular, efforts to block-net reaches failed as a result of high base flows, accumulation of debris, and the mesh-size used in block-nets (size was selected to prevent fish larger than 120 mm from escaping reaches by swimming through nets). As a result, (1) proper tie-offs of the block-net were not possible, (2) lack of rigidity in netting material forced the nets to sag and reduce the height of the nets over the water, and (3) it was not possible to properly bury the block-nets within the substrate. Due these inabilities, larger bull trout were witnessed jumping upstream and downstream into and out of the block-netted reaches, and smaller bull trout were observed escaping into and around the substrate. Ultimately, our efforts were unsuccessful over several failed attempts on different reaches and throughout the summer even as flows decreased slightly. Our efforts to quantify small bull trout (< 120 mm), and the use of the mesh size needed to act as a barrier to these fish, were the ultimate causes of failure in a system as large as the SFWW (average width of approximately 9 m). Future efforts on a stream this large would need to use block-nets with a larger mesh size, which would sacrifice the ability to quantify smaller, potentially fecund bull trout.

Growth

Tagged fish—Annual growth of tagged bull trout in the SFWW varied by size class. Bull trout in the 220 – 320 mm size class grew more in length than bull trout > 320 mm, but these results were not significant due to high variance associated with a small sample size; in addition, bull trout > 320 mm gained less weight than bull trout 220 - 320 mm. On average, big bull trout (> 320 mm) gained 0.15 g/day versus 0.3 g/day for medium-sized (220 – 320 mm) bull trout. Instantaneous growth was significantly greater for medium-sized bull trout than for the largest bull trout ($t = 2.64$, $df = 3$, $P = 0.04$; Figure 1.16).

Emigration, Immigration, and Movement

Number of fish detections (i.e., recaptures) at the PIT-tag detectors and movement of bull trout in the SFWW varied by size-class and month. More bull trout were detected at the Bear Creek (WW2) detector (Figure 1.17); over twice as many detections as at the Harris Park Bridge (WW1) site (Figure 1.18). Most detections (and therefore movement) of small fish (120 – 320 mm) occurred from May through September while large fish (> 320 mm) were primarily detected from July through October 2004. Significantly more fish movements occurred at night at both detectors ($n = 160$) than day ($n = 70$, $t = -6.43$, $df = 229$, $P < 0.001$; Figures 1.19 and 1.20). In the SFWW, 74 bull trout were detected at the Harris Park Bridge PIT-tag detection array, of which 46

were moving downstream and 10 were moving upstream (Figure 1.21). At the Bear Creek detector, 156 bull trout were detected; of which 58 moved downstream and 19 moved upstream (Figure 1.22). Large bull trout (> 320 mm) moved into the study section above Harris Park Bridge during May to June. Smaller bull trout moved downstream out of the study area primarily from May to September although a few movements were also detected in November and December. Large bull trout moved out of the study area from September to November, likely the period after spawning. Detection data for the NFUM site has not yet been summarized.

Migration Cues and Timing

A preliminary analysis of potential environmental and biological cues for migration revealed that average daily temperatures were highly and positively correlated with the number of migrants per day. Based on a regression between average daily temperature and mean number of subadult downstream migrants, average daily temperature explained 98% of migration ($R^2 = 0.98$, $df = 4$, $P = 0.001$). In addition, once average temperatures reached 12 °C (Figure 1.23), downstream migration ceased. When considered independently, precipitation, day length, body size at migration, and presence of spawning adults did not appear to explain the timing of migration, but these analyses should be considered extremely preliminary. Future analysis will include flow data, which has not been synthesized at this time.

Survival

Survival estimates and recapture probabilities were calculated using the mark-recapture data in the Cormack-Jolly-Seber model. Data from 2002-2004 allowed us to estimate survival and recapture probabilities for two size classes of bull trout in the SFWW (Figure 1.24). For bull trout between 220-320 mm, survival was estimated at 0.447 (± 1 SE = 0.146), with a recapture probability of 0.421 (± 0.161). For bull trout >320 mm, survival was estimated at 0.569 (± 0.318), and recapture probability was estimated at 0.072 (± 0.047). No valid estimates for bull trout < 220 mm were possible due to the low number of recapture events for this size class. Additional years of data will allow for more robust estimates of survival by age and life-stage classes and with incorporation of covariates.

DISCUSSION

In 2004, research from previous years (initiated in 2002) was continued and expanded to meet overall project objectives; all fish population-based research for the SFWW and NFUM was continued. We sampled 48% of the SFWW study area (20% re-sampled, 20% new), 410 fish were tagged; we sampled 40% of the NFUM study area, and 64 fish were tagged. All the baseline demographic and population assessment data were collected, analyzed, and synthesized. Habitat studies were continued and nearly completed. In addition, in 2004 we initiated the genetic component of our study and collected and prepared a set of fin clip samples representing three potentially different components of the population, migratory, likely resident, and tributary bull trout. A subset of samples, representing a pilot study group, were genetically analyzed through microsatellite DNA techniques by the USFWS Abernathy Fish Technology Center in May 2005. Future analyses will be determined based on the results from this pilot group with 1196 fin clips from individually tagged bull trout collected to date.

As observed and discussed in more detail in previous years, annual population estimation based on mark-resight appears to be a robust technique well suited to this question and for bull trout sampled in the SFWW. Comparison of size frequency distributions of fish observed snorkeling (resighted) versus those caught sampling (marked) overlap and demonstrate an extremely similar pattern in 2004, for both systems. Further, a more rigorous evaluation of diel effects demonstrates again (similar results were observed in 2003) that there is no consistent difference in abundance or size distribution between day versus night sampling in these systems; a pattern inconsistent with that observed by Dunham et al. (2001). Further, using this approach, we are able to systematically and effectively sample a large proportion (20 to 40%) of these study areas each year. The combination of these results, and their consistency across three years of study, indicates that mark-recapture techniques provide a robust population assessment methodology for bull trout in these types of systems. All sample data are currently being used as a template for evaluating sampling variability and our ability to detect trend under various sampling protocols (Al-Chokhachy et al., *in prep*).

In 2004, bull trout condition (Fulton's K) was similar as compared to previous years for the SFWW; however, in the NFUM, we observed a dramatic decrease in fish condition. This dramatic decline is worthy of future investigation, and abiotic and biotic factors that may have caused this decline should be considered. As observed in 2003, condition was the most variable for the intermediate adult size class (220 –

330 mm); a pattern that may be explained by an ontogenetic diet switch (from invertebrates to fish) at this intermediate size. Growth rates in the SFWW were substantially higher in 2004 as compared to 2003 with smaller fish growing at a much higher rate as compared to fish > 320 mm. In addition, survival rates for small and large bull trout are high (over 40%, 2002 through 2004), and although there are few published field estimates of survival, ours are similar to values reported for other salmonids (Rieman and Apperson 1989; Rieman and McIntyre 1993) and within the range observed by Rieman and Allendorf (2001) for bull trout.

Consistent with patterns observed elsewhere (e.g., Mill Creek; Hemmingsen et al. 2001), in the SFWW, most movement of larger fish (> 320 mm) occurred between July and October, and most movement of small fish (< 320 mm) occurred between May and September. Preliminary analyses suggest temperature explains a large portion of the variation in migration timing with downstream movement of subadults ceasing after temperature reach 12 °C. On a diel basis, more movement occurs at night as compared to during the day, especially for smaller fish < 320 mm (Salow 2004). The NFUM detector was not completed and operating long enough in 2004 to collect data on bull trout movement.

In 2004, we also completed a pilot study of the feasibility of expanding our research into the John Day Subbasin. We completed a review of the grey literature and evaluated fish distribution information, and met with local biologists. Based on these evaluations and discussions, the North Fork John Day River (NFJDA) was proposed as the future study reach, and we completed a preliminary field assessment of this area.

FUTURE

In 2005, all fish population monitoring, tagging, and demographic analyses will continue in the SFWW and in the NFUM, along with monitoring of abiotic variables (e.g., flow, temperature). Genetic samples will continue to be taken from bull trout; in 2004, tissues samples (fin clips) were taken from approximately 400 bull trout in the SFWW. In spring 2004, a subsample of these fin clips were evaluated for genetic variation at 13 microsatellite loci. Post hoc comparison of known migrant versus known resident fish (based on tagging and encounter history) will be further evaluated as PIT-tagged fish detections increase and life-history designations are determined. Based on the results of this initial genetic analysis, additional samples will be evaluated accordingly.

We will also expand our bull trout recovery and conservation program into the John Day River Subbasin. This expansion will allow a basin-level comparison of demographic variables like growth and survival, biotic drivers of diet, food availability, and marine derived nutrients (MDN), in a system with low salmon and steelhead abundance (SFWW) compared to a system with high salmon and steelhead abundance (NFJDA). See Appendix 3 for details of this upcoming work.

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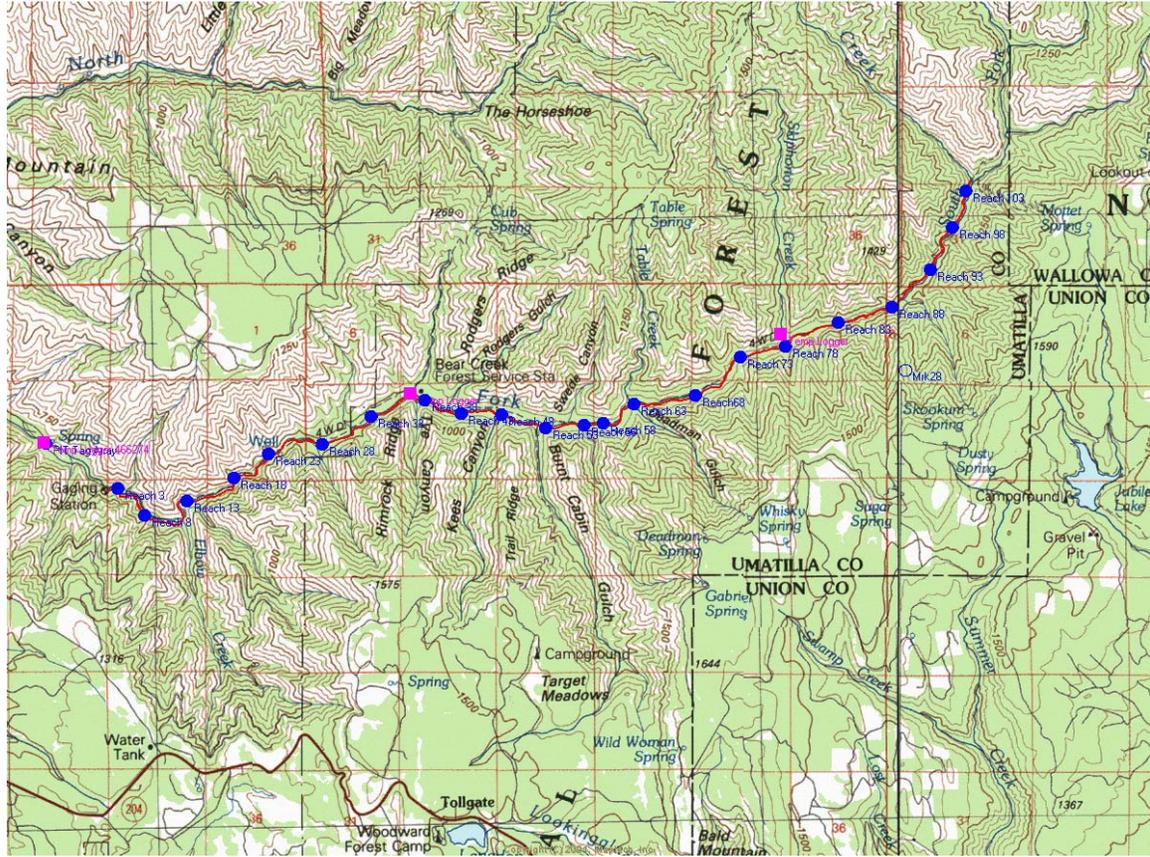


Figure 1.1. Map of the South Fork Walla Walla River showing original 22 study reaches (dots).

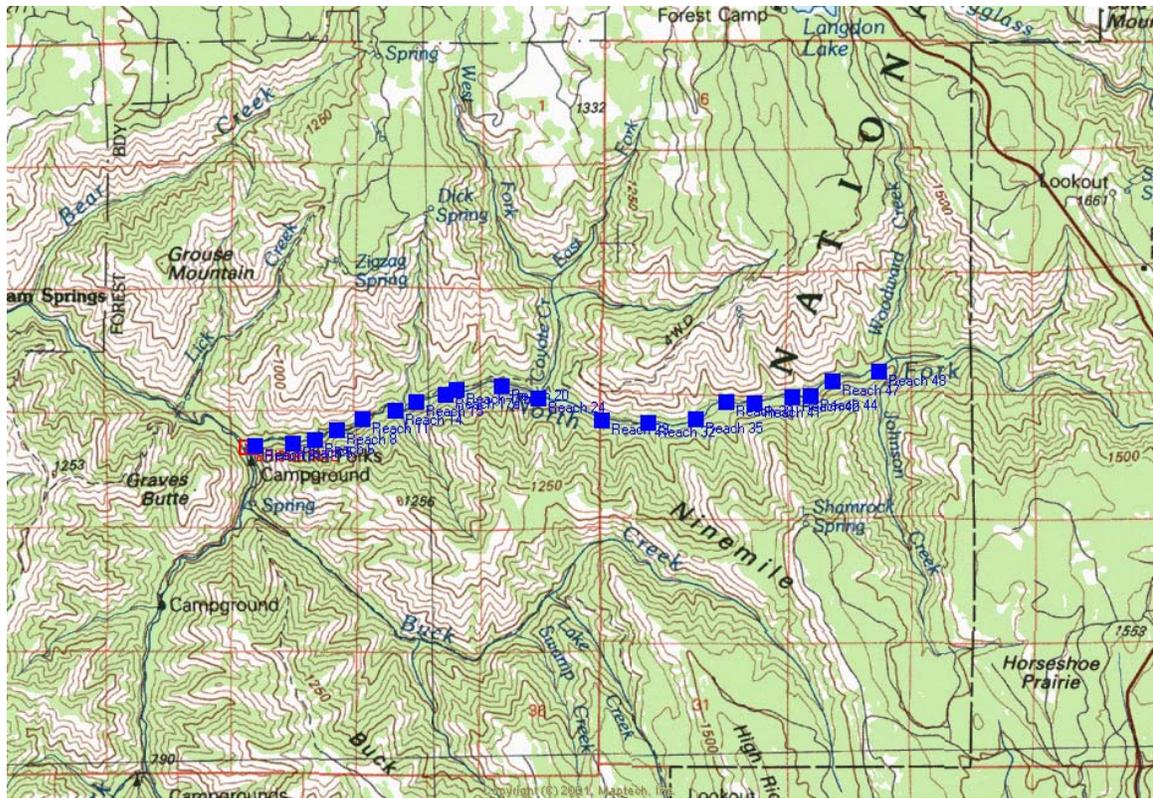


Figure 1.2. Map of the North Fork Umatilla River showing the 20 study reaches (squares).

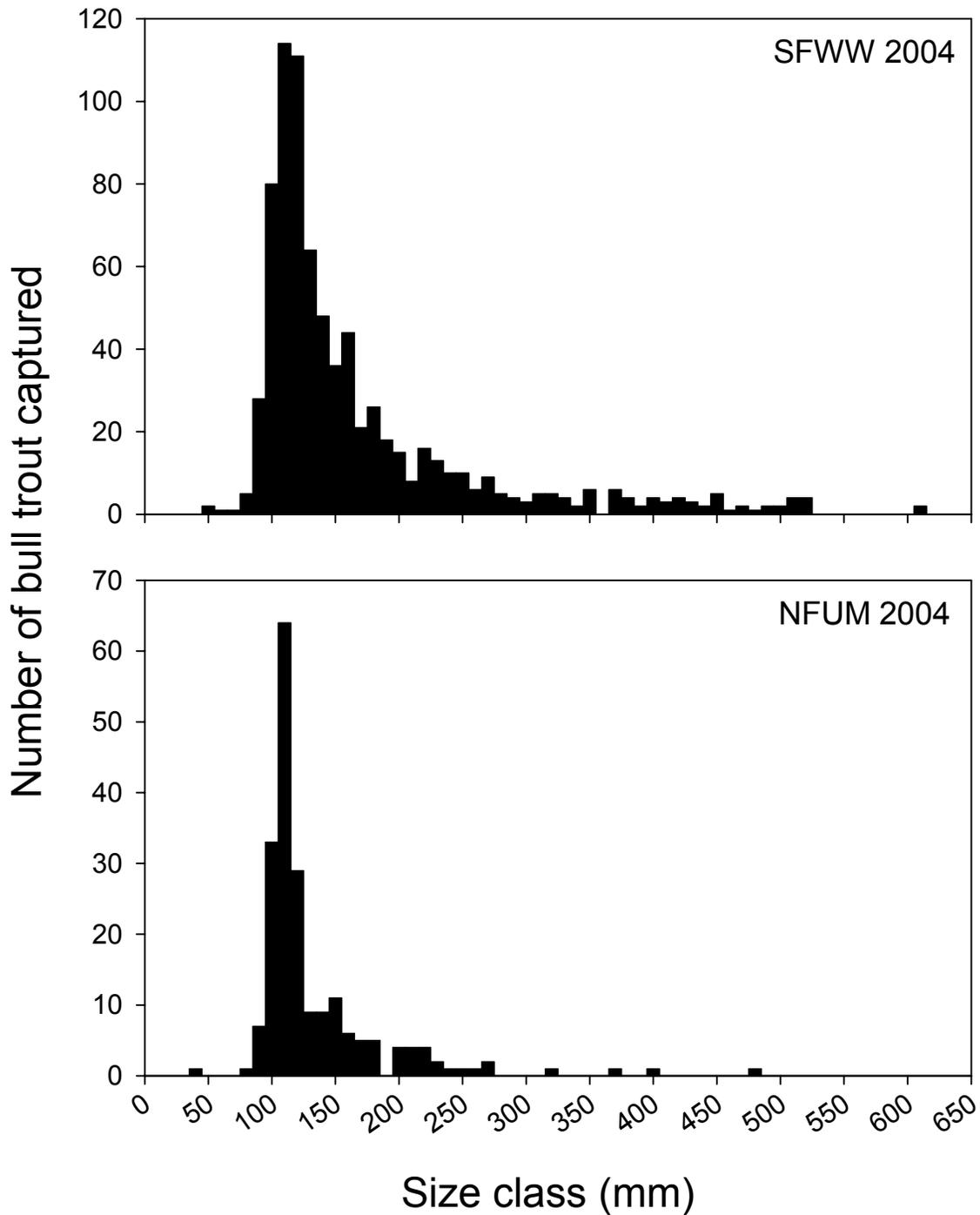


Figure 1.3. Length frequency distribution of bull trout captured and handled in the South Fork Walla Walla River and North Fork Umatilla River, 2004. Note changes in y-axis scales.

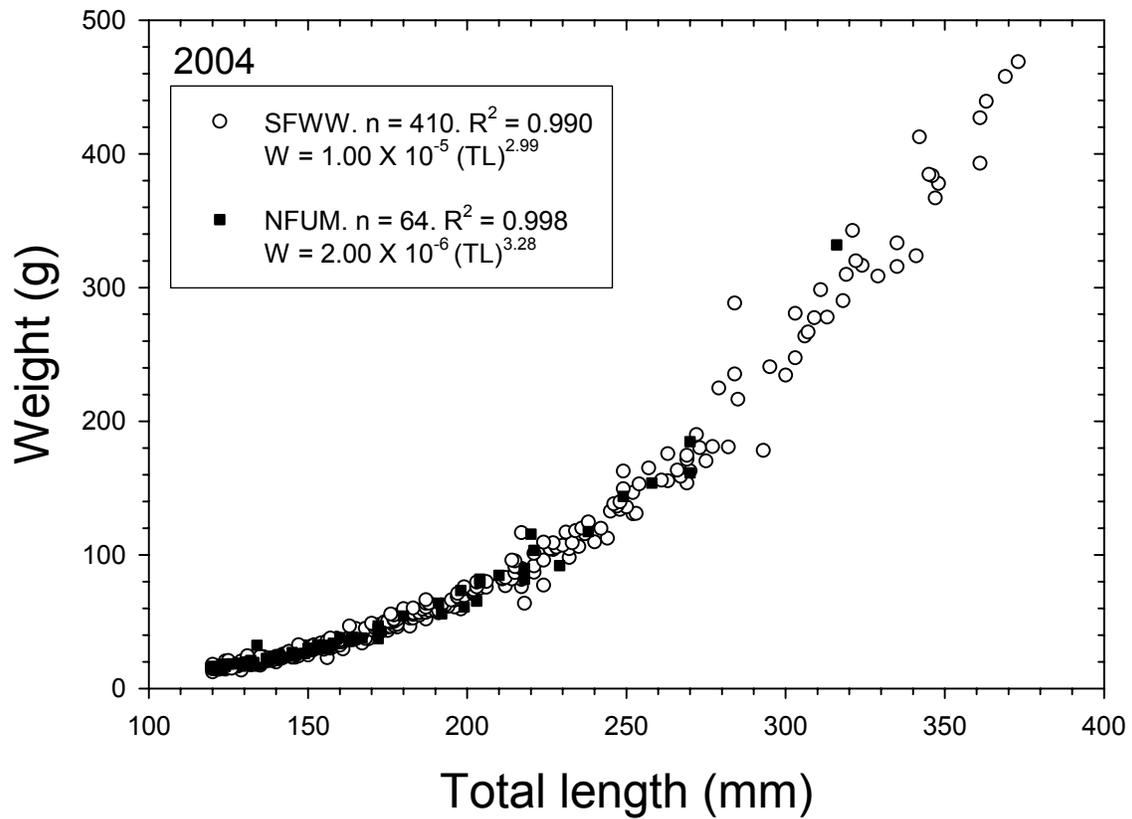


Figure 1.4. Length-weight relationship for bull trout tagged in the South Fork Walla Walla River, 2004 (open circles) and North Fork Umatilla River, 2004 (black squares). Regression equations and sample sizes are given.

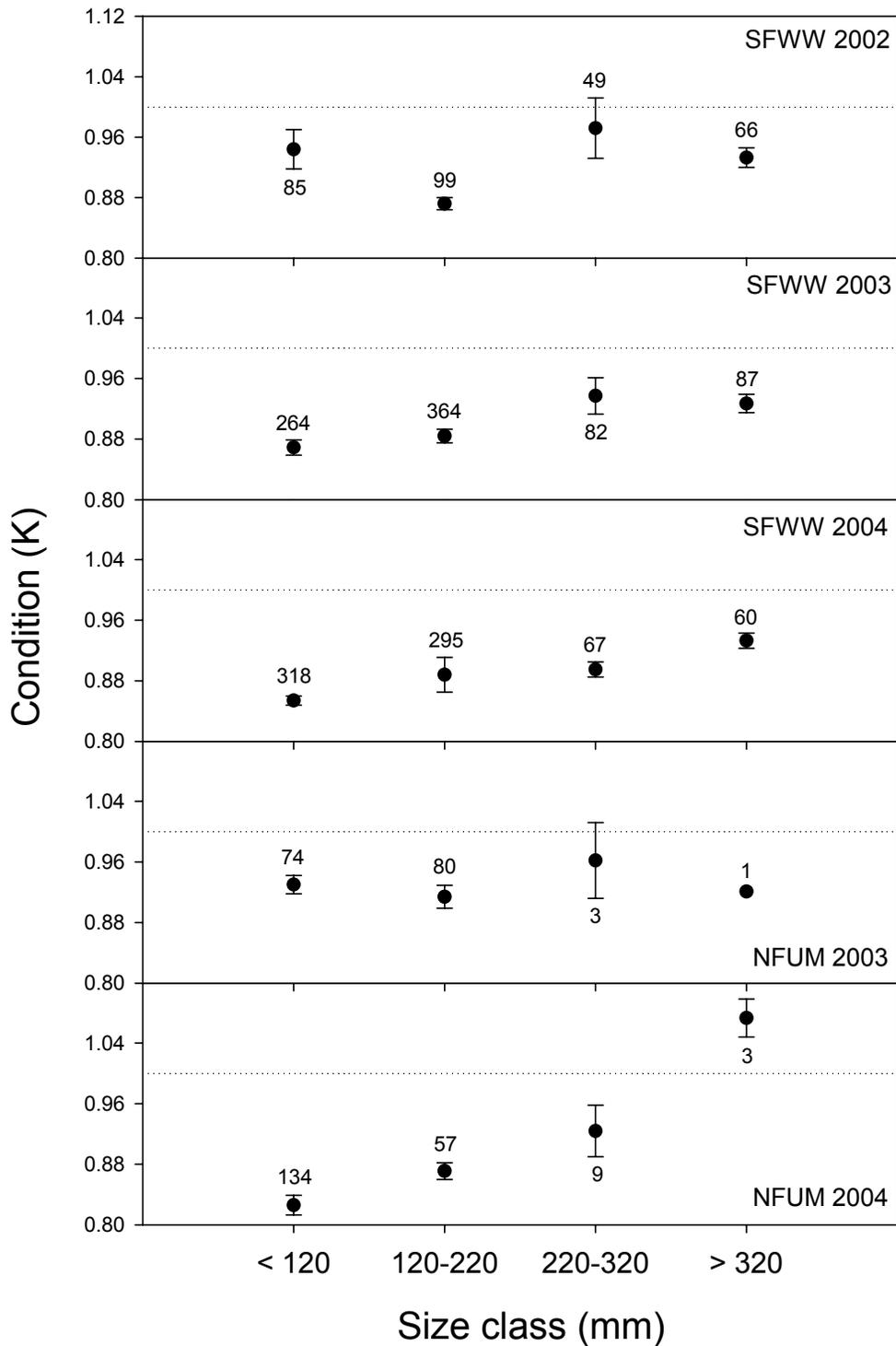


Figure 1.5. Condition (Fulton's $K \pm 1$ SE) of bull trout by size class sampled in the South Fork Walla Walla River (2002 - 2004) and North Fork Umatilla River (2003 - 2004). Sample sizes are given by error bars. $K = 1$ is reference line.

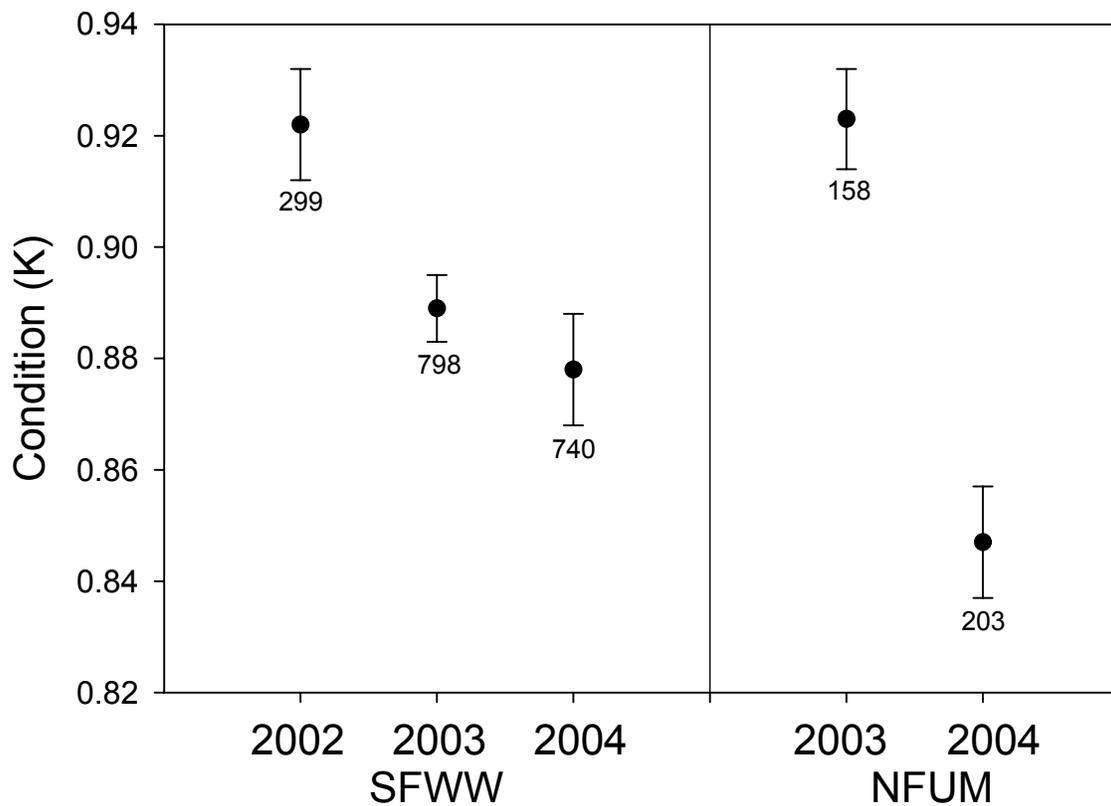


Figure 1.6. Average condition (Fulton's $K \pm 1$ SE) of bull trout (all sizes combined) sampled in the South Fork Walla Walla River (2002 - 2004) and North Fork Umatilla River (2003 - 2004). Sample size is given below error bars.

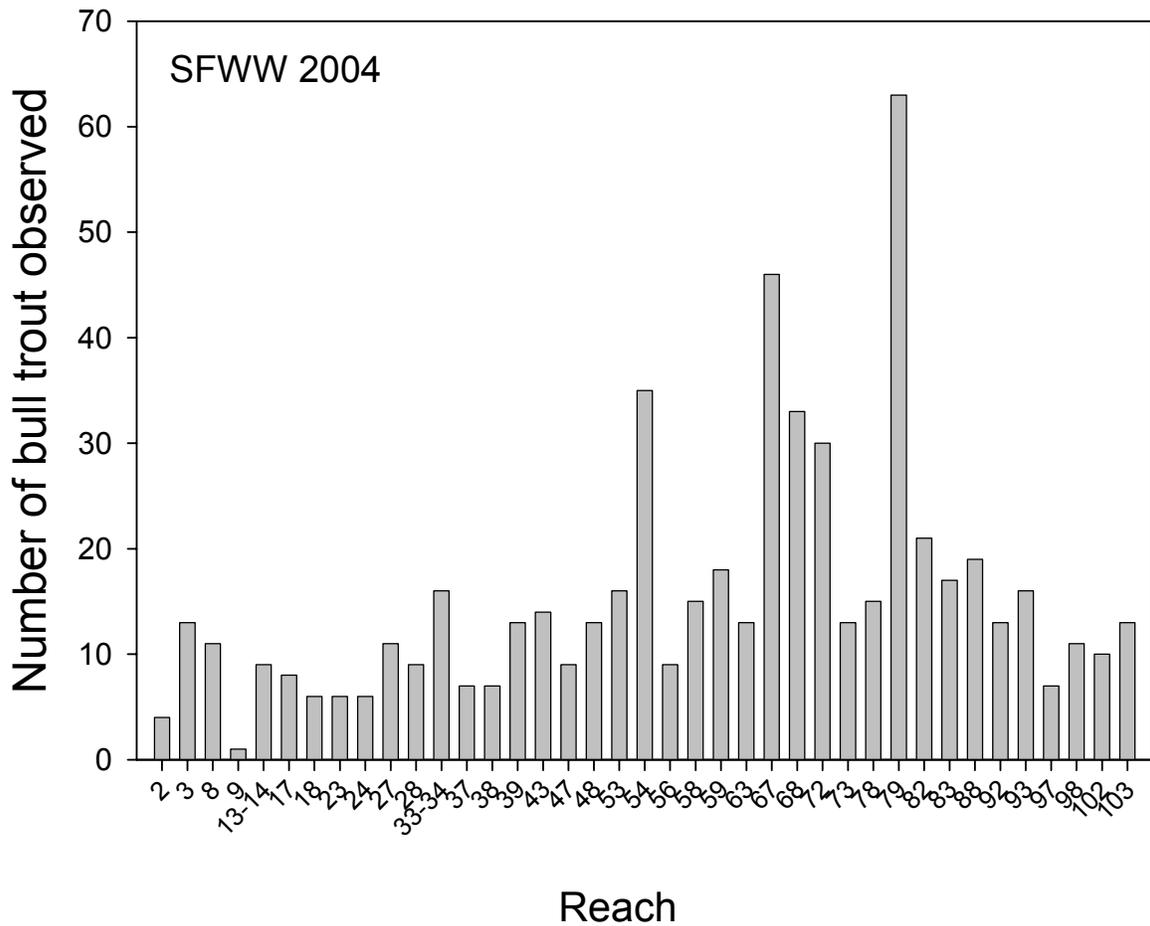


Figure 1.7. Number of bull trout by reach observed during snorkel surveys in the South Fork Walla Walla River, 2004. Reaches are numbered from bottom to top of the study site.

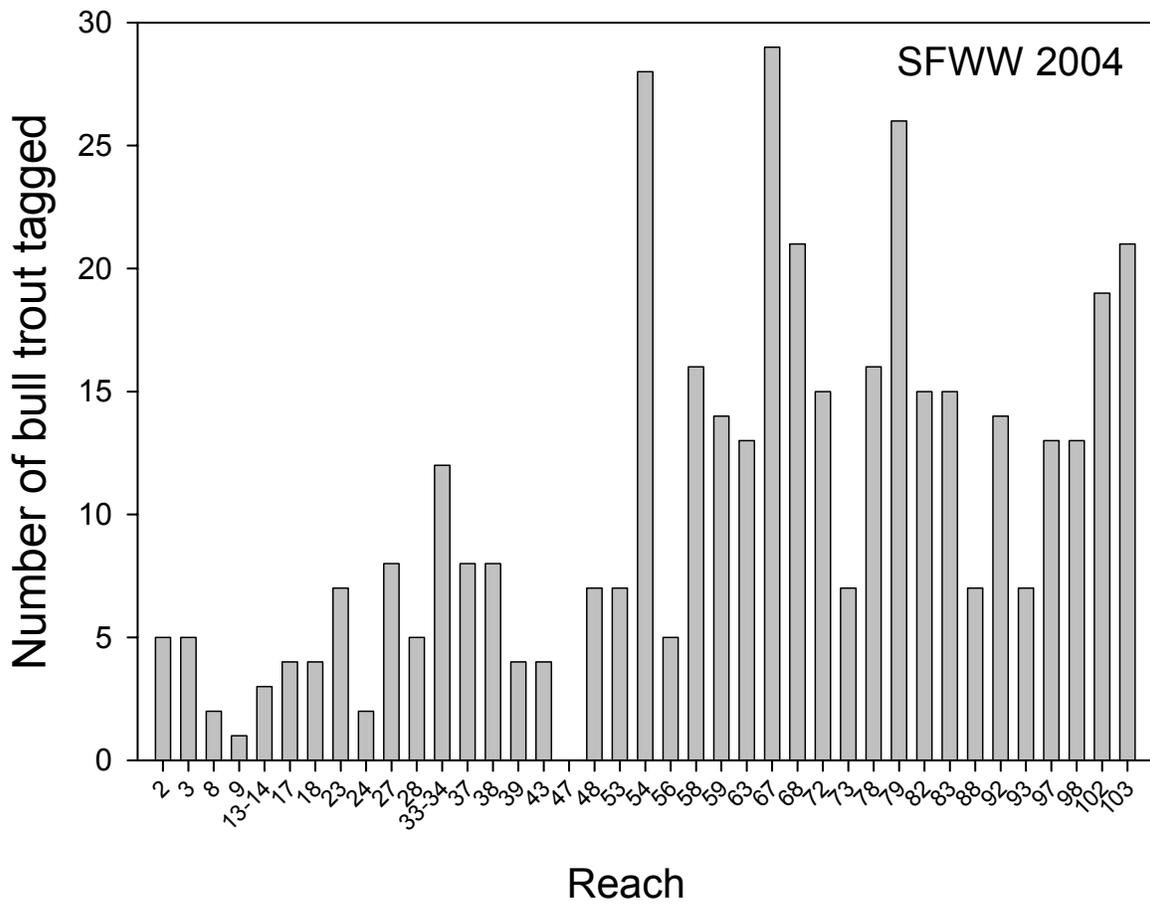


Figure 1.8. Number of bull trout tagged by reach in the South Fork Walla Walla River, 2004. Reaches are numbered from bottom to top of the study site.

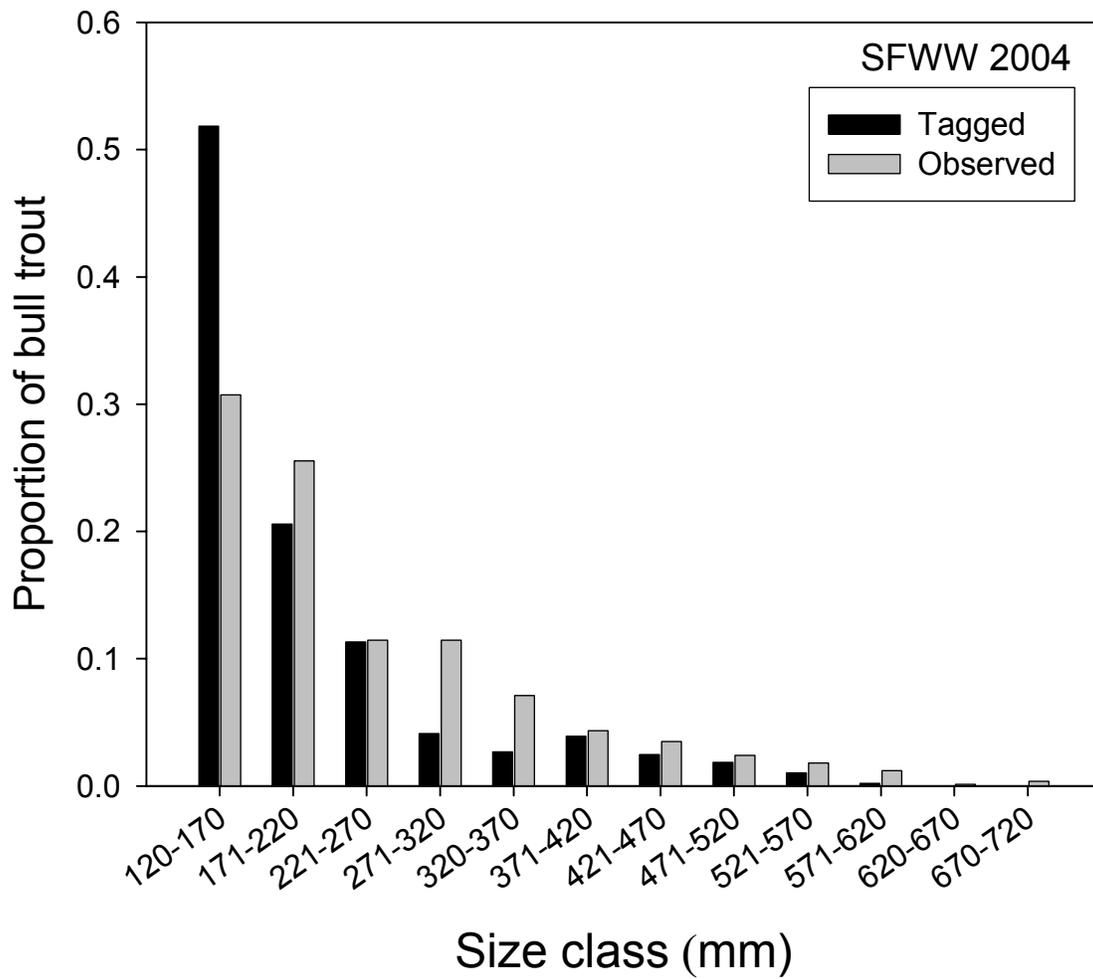


Figure 1.9. Length frequency distribution of bull trout tagged (black bars) and observed (via snorkel counts; gray bars) in the South Fork Walla Walla River, 2004.

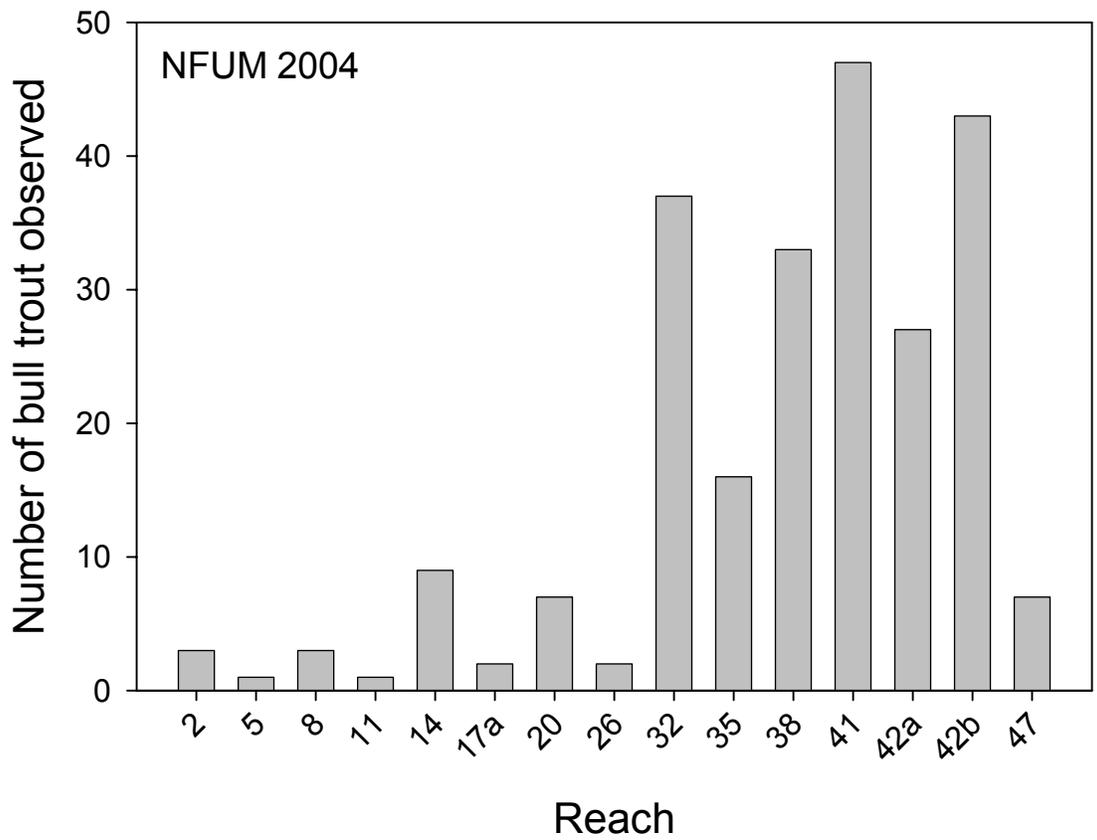


Figure 1.10. Number of bull trout by reach observed during snorkel surveys in the North Fork Umatilla River, 2004. Reaches are numbered from bottom to top of the study site.

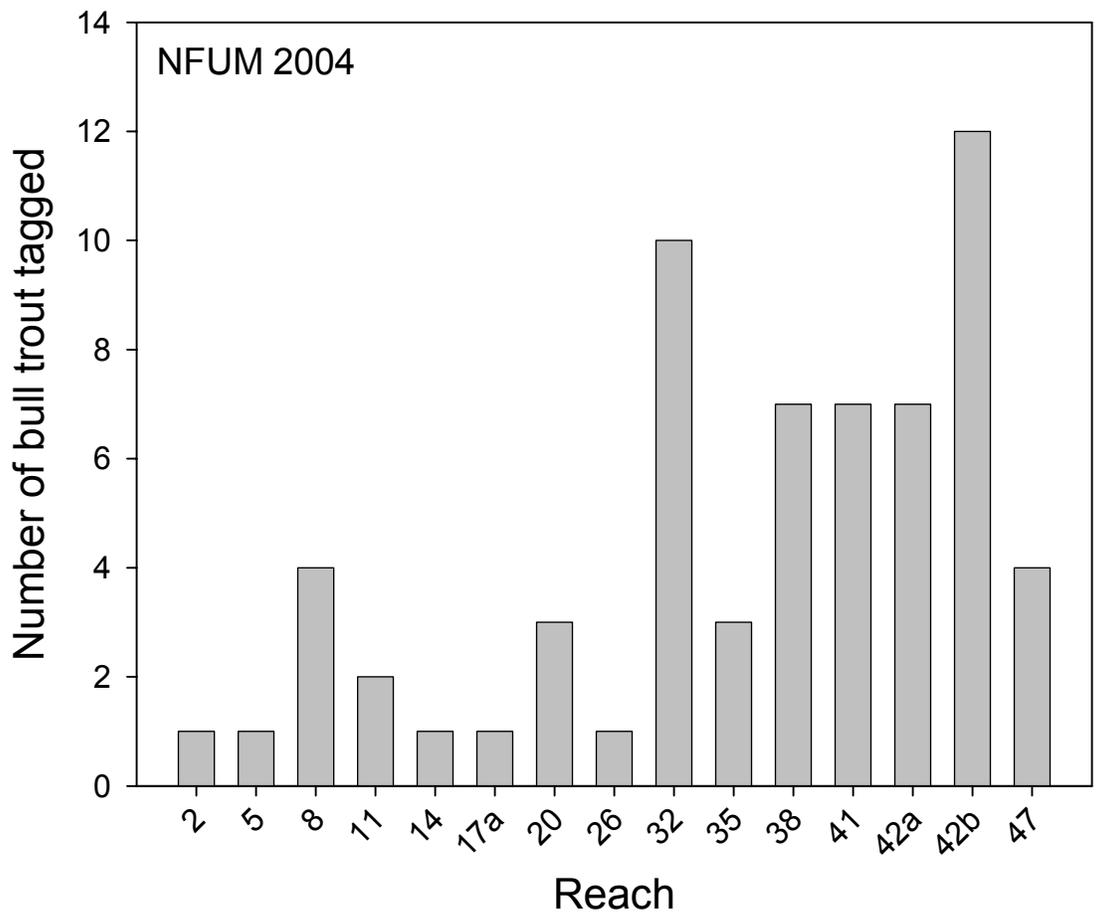


Figure 1.11. Number of bull trout tagged by reach in the North Fork Umatilla River, 2004. Reaches are numbered from bottom to top of the study site.

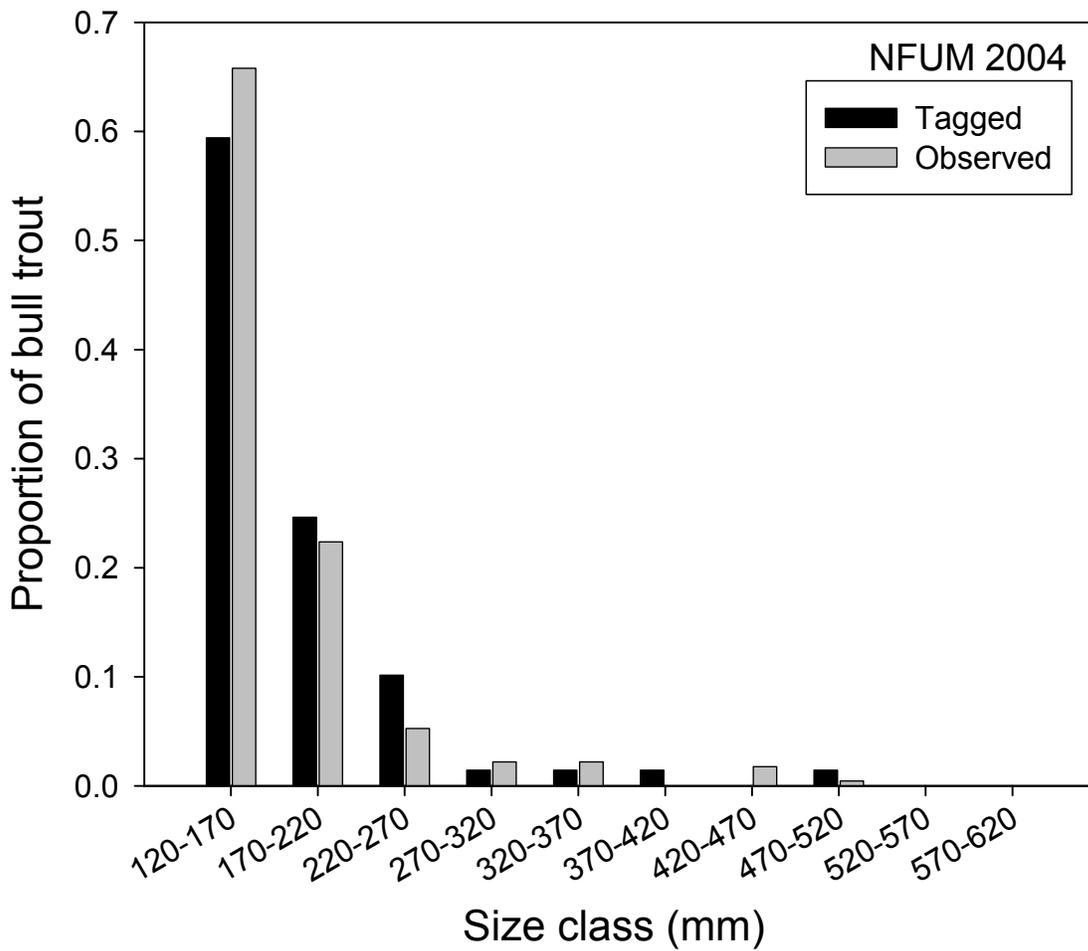


Figure 1.12. Length frequency distribution of bull trout tagged (black bars) and observed (via snorkel counts; gray bars) in the North Fork Umatilla River, 2004.

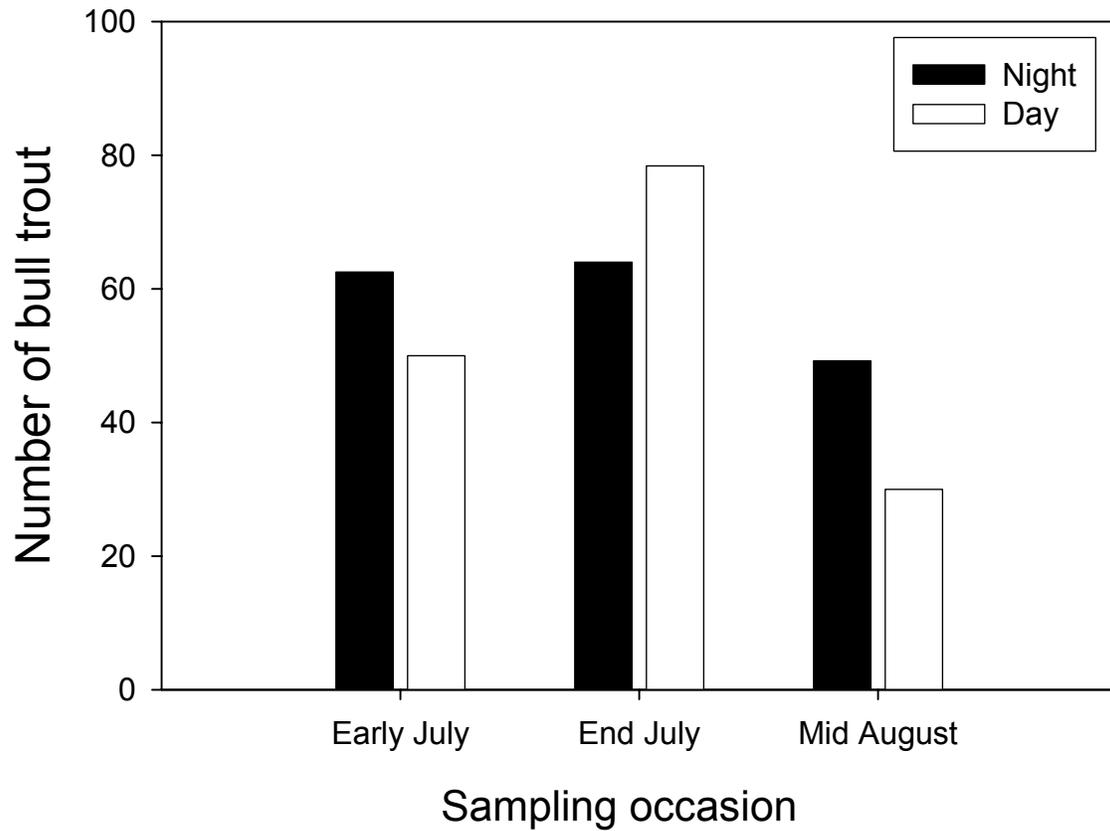


Figure 1.13. Pooled mark-resight population estimates across the three temporally different sampling occasions on the North Fork Umatilla River, 2004. Night snorkel surveys are denoted by black bars and day snorkel surveys are represented by white bars.

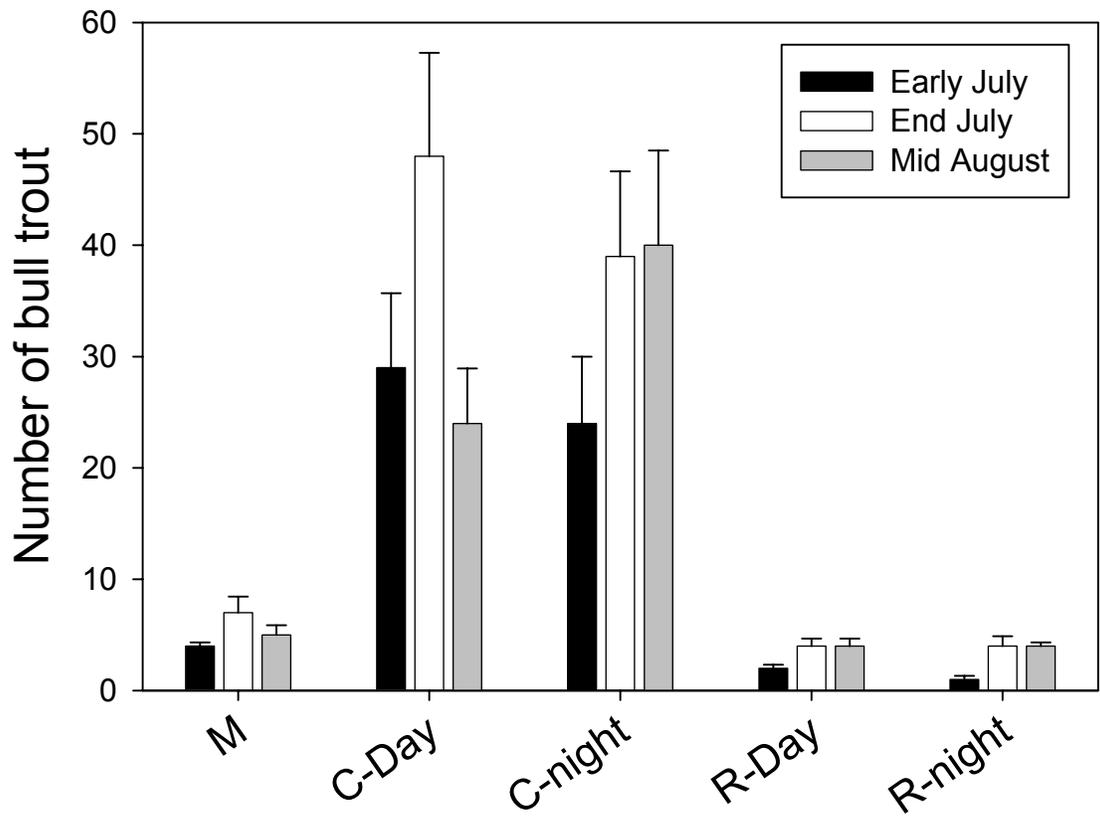


Figure 1.14. Total number of bull trout for each sampling occasion (different colored bars) that were marked (M), captured from day surveys (C-day), captured from night surveys (C-night), resighted from day snorkel surveys (R-day), and resighted from night snorkel surveys (R-night) from the three temporal reaches on the North Fork Umatilla River, 2004.

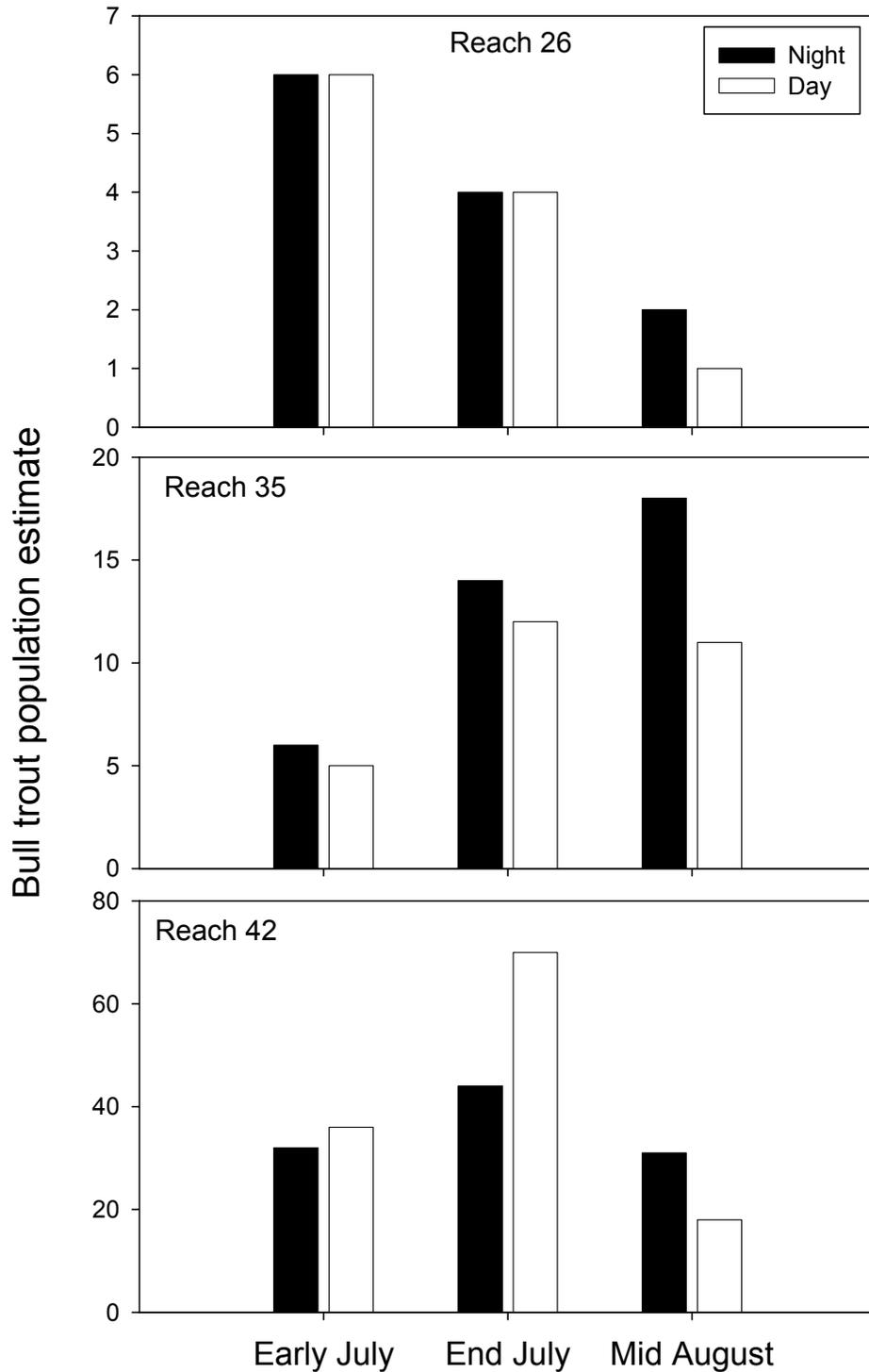


Figure 1.15. Mark-resight population estimates across spatial (3 reaches) and temporal (3 sampling occasions) differences in the North Fork Umatilla River, 2004. White bars denote estimates from day snorkeling, and black bars denote night snorkeling efforts.

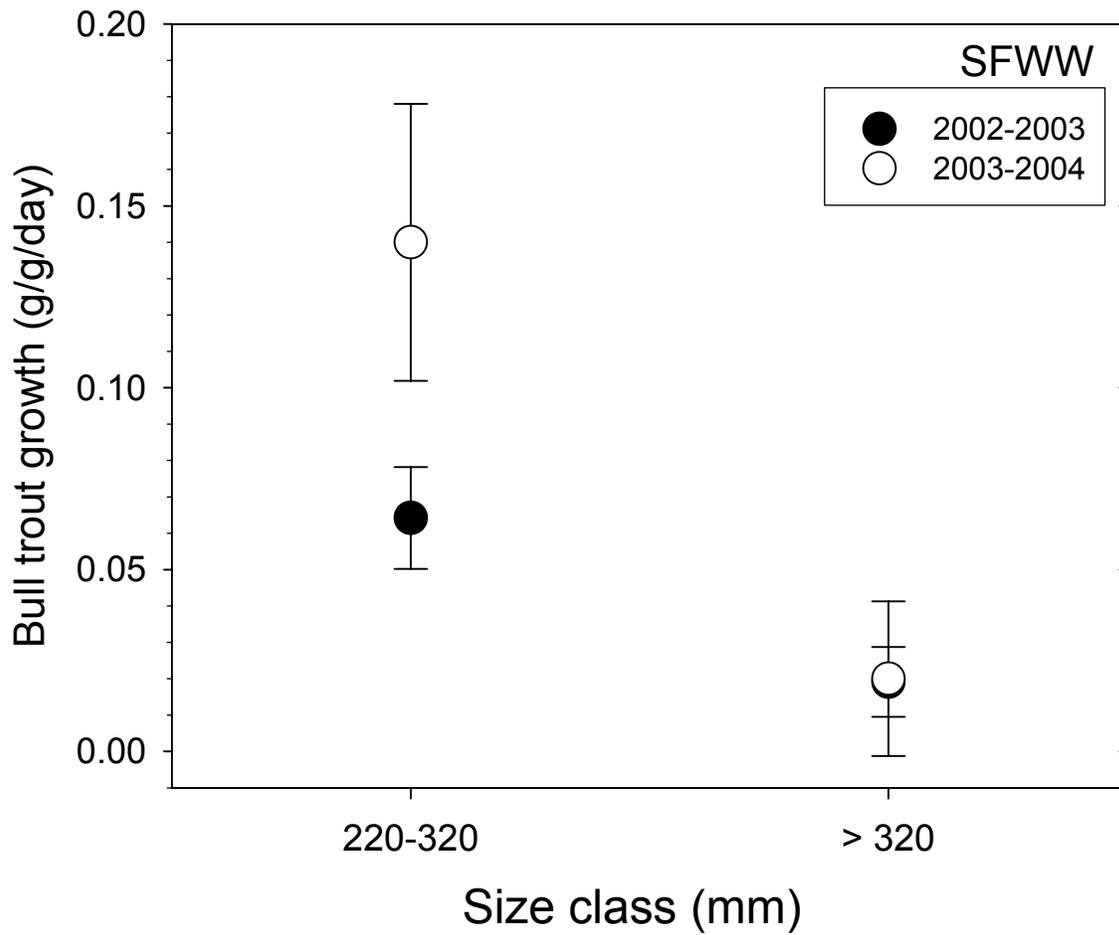


Figure 1.16. Instantaneous growth of two size classes of bull trout in the South Fork Walla Walla River over two annual periods.

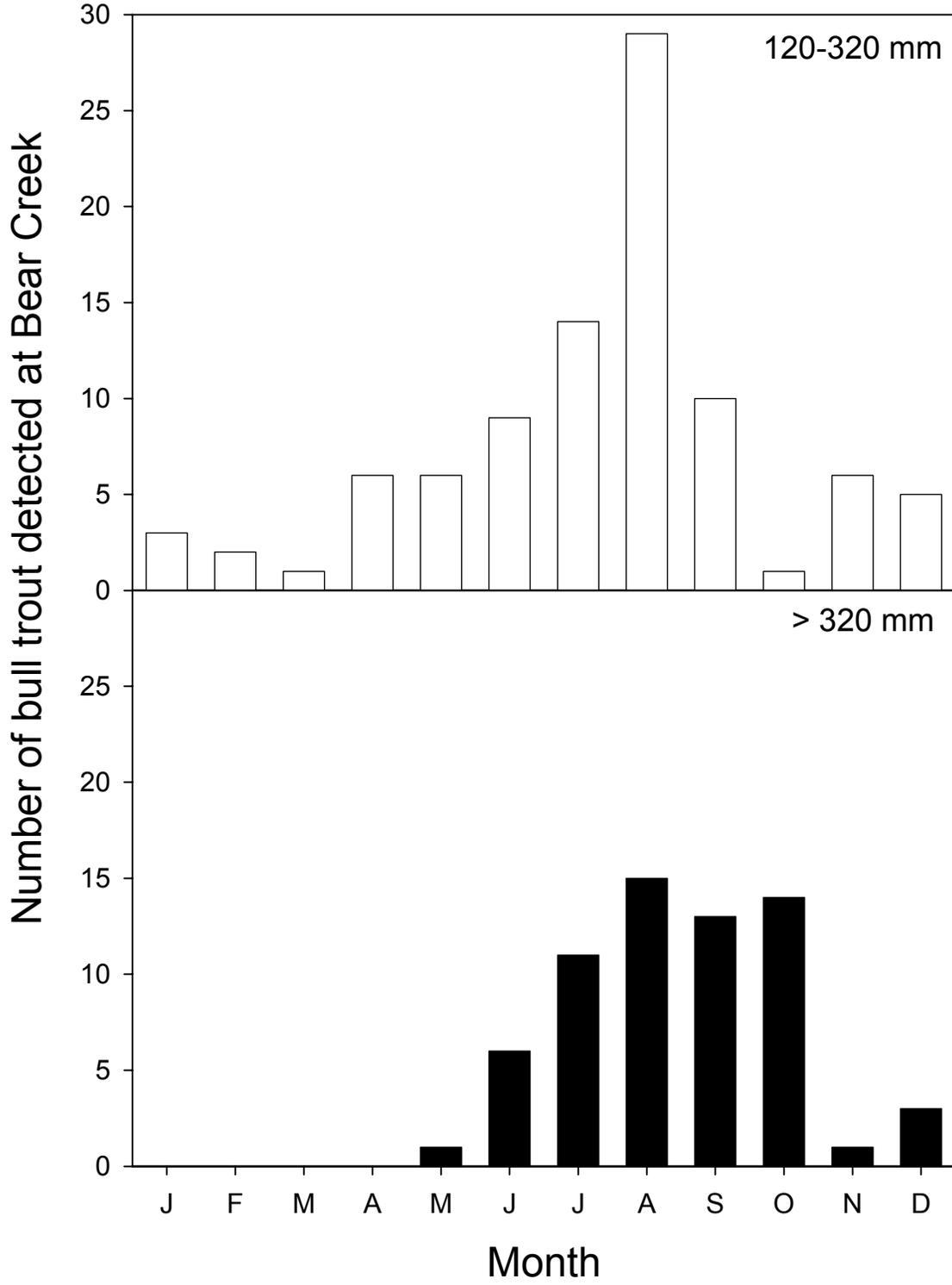


Figure 1.17. Monthly PIT-tag detections (recaptures) of small (120-320 mm; top panel) and large (> 320 mm; bottom panel) bull trout made at the Bear Creek PIT-tag detector, 2004.

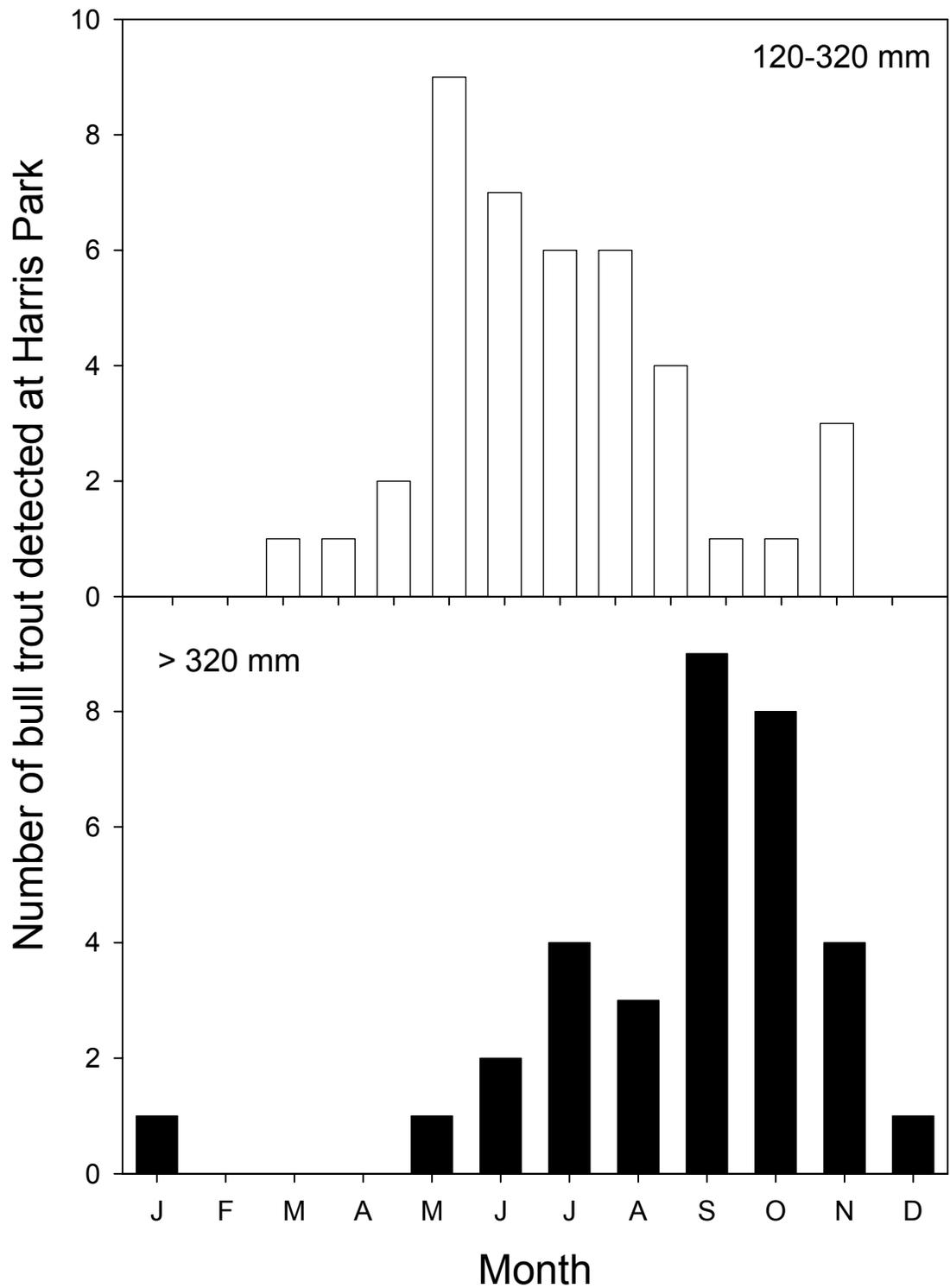


Figure 1.18. Monthly PIT-tag detections (recaptures) of small (120-320 mm; top panel) and large (> 320 mm; bottom panel) bull trout made at the Harris Park Bridge PIT-tag detector, 2004.

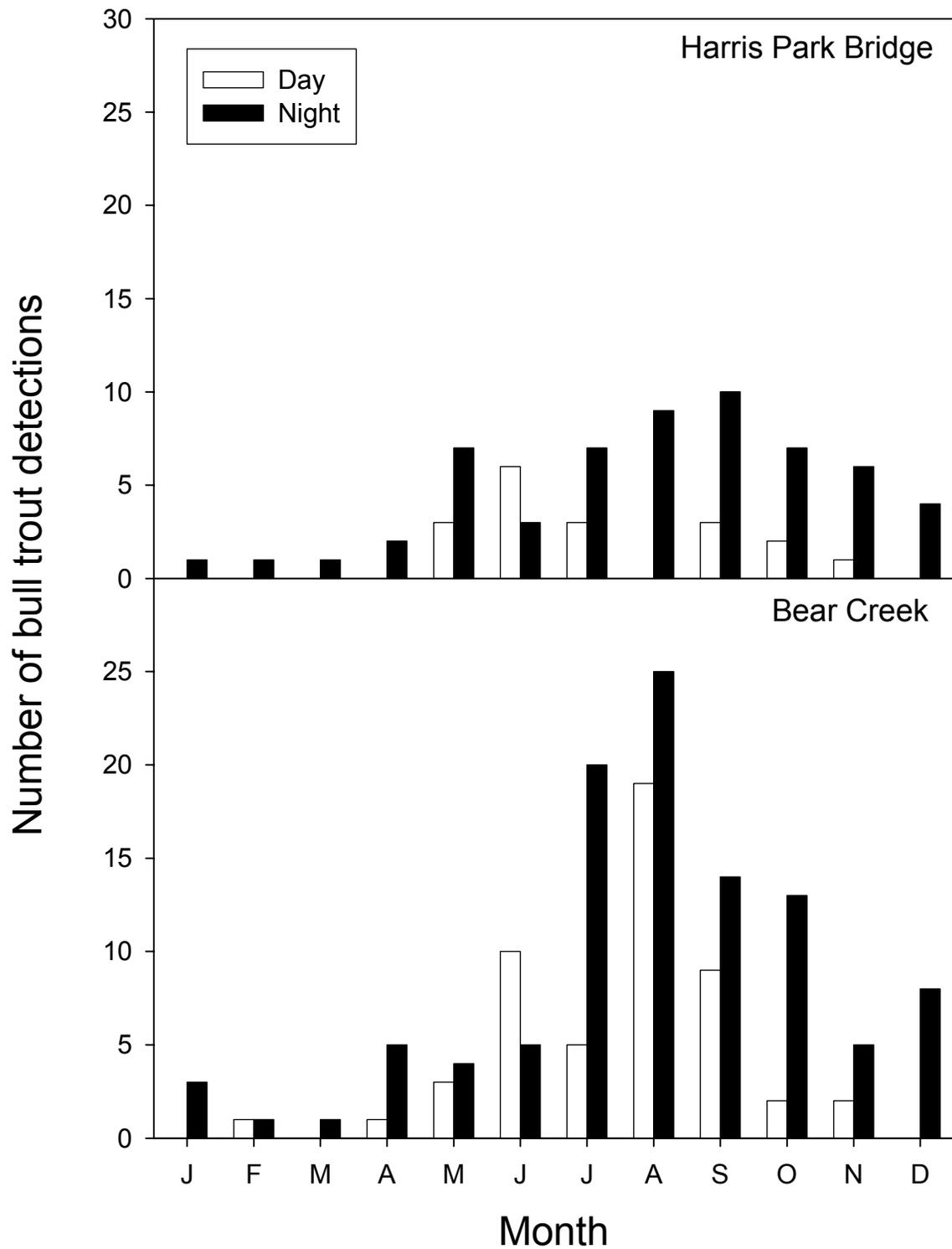


Figure 1.19. Monthly movement (daytime and nighttime) of bull trout based on PIT-tag directional-detections (recaptures) made at the Harris Park Bridge and Bear Creek PIT-tag detectors, 2004.

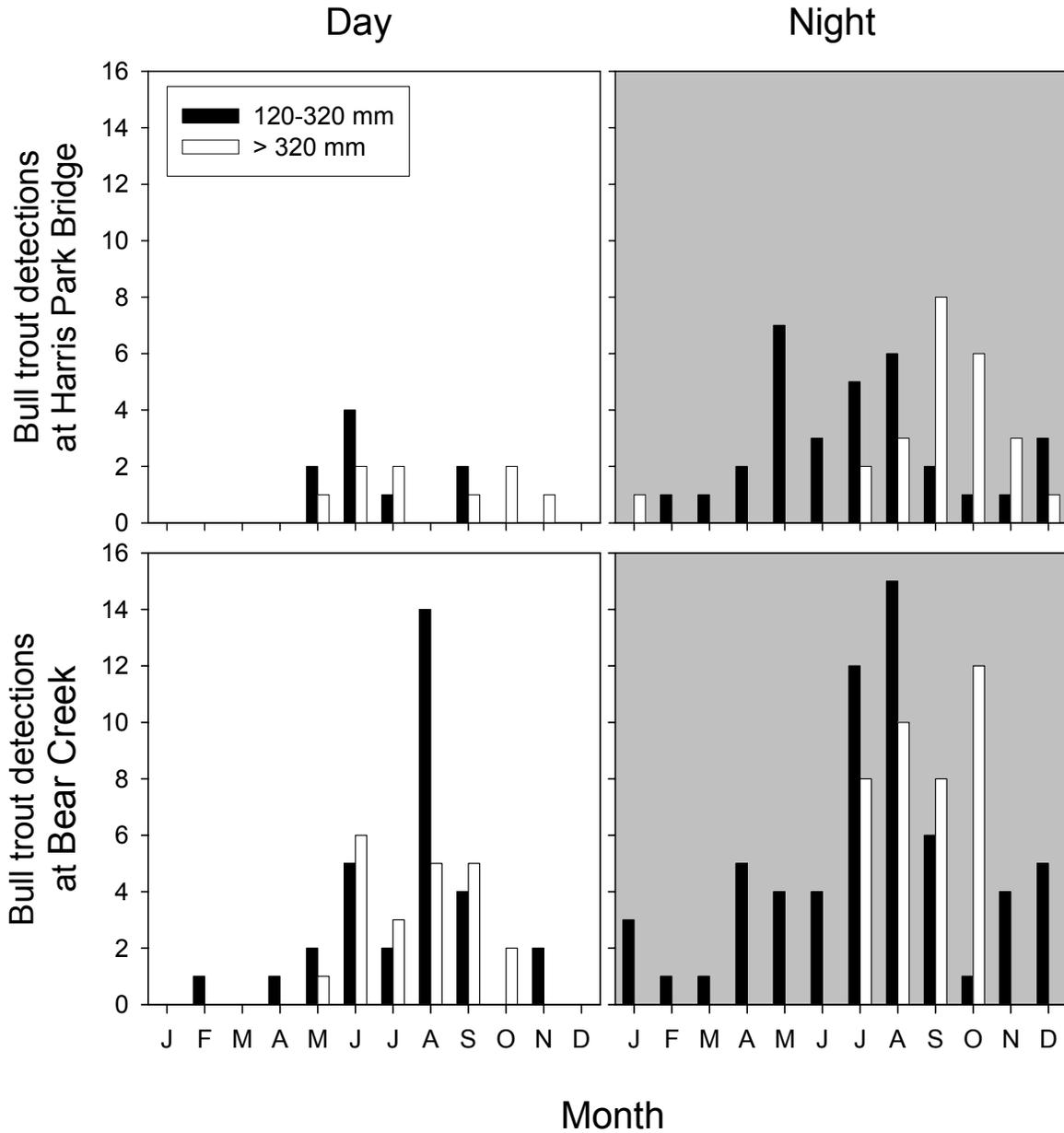
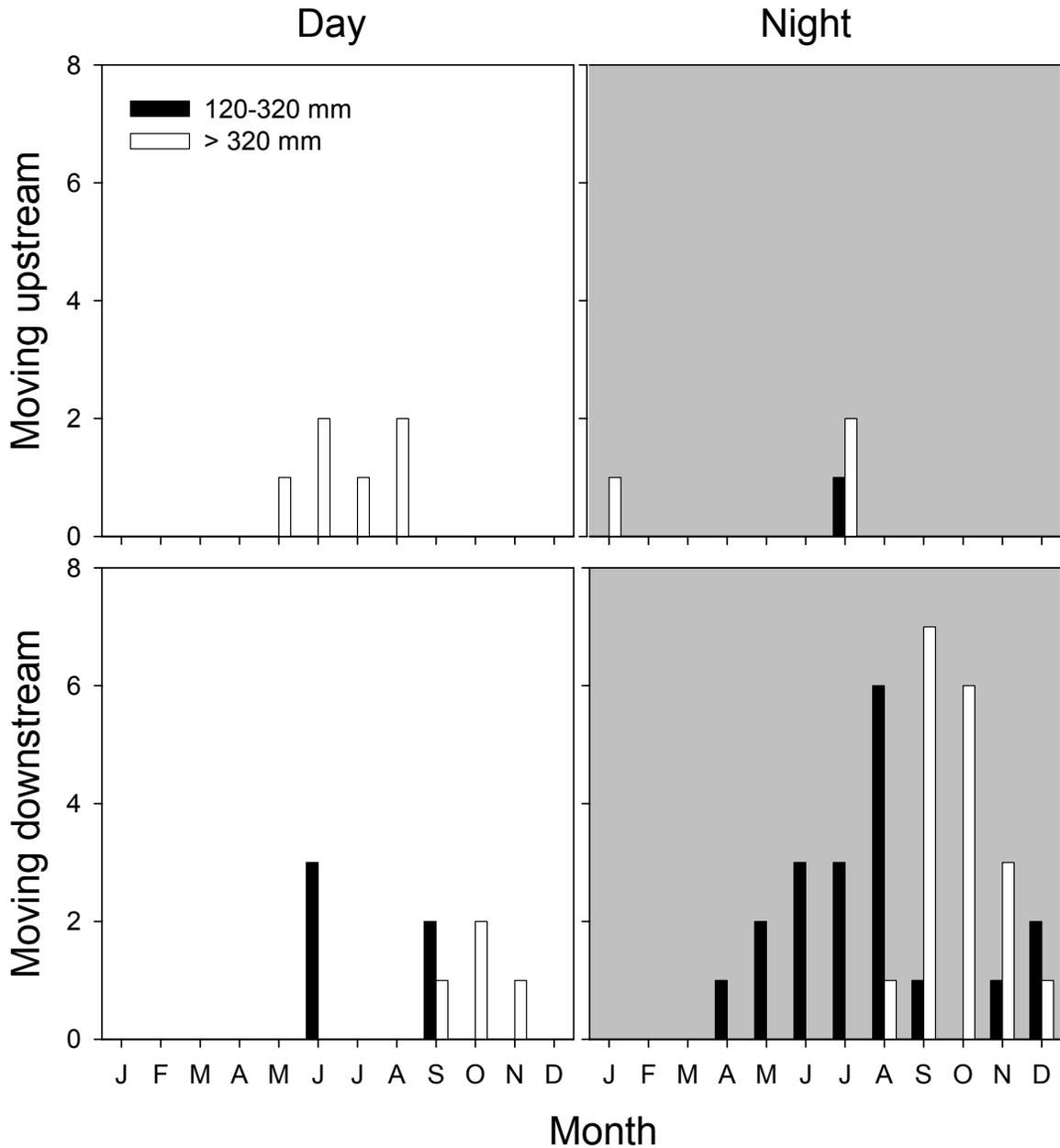


Figure 1.20. Monthly movement (daytime and nighttime) of small (120-320 mm) and large (> 320 mm) bull trout based on PIT-tag detections (recaptures) made at the Harris Park Bridge (top panels) and Bear Creek (lower panels) detectors, 2004.



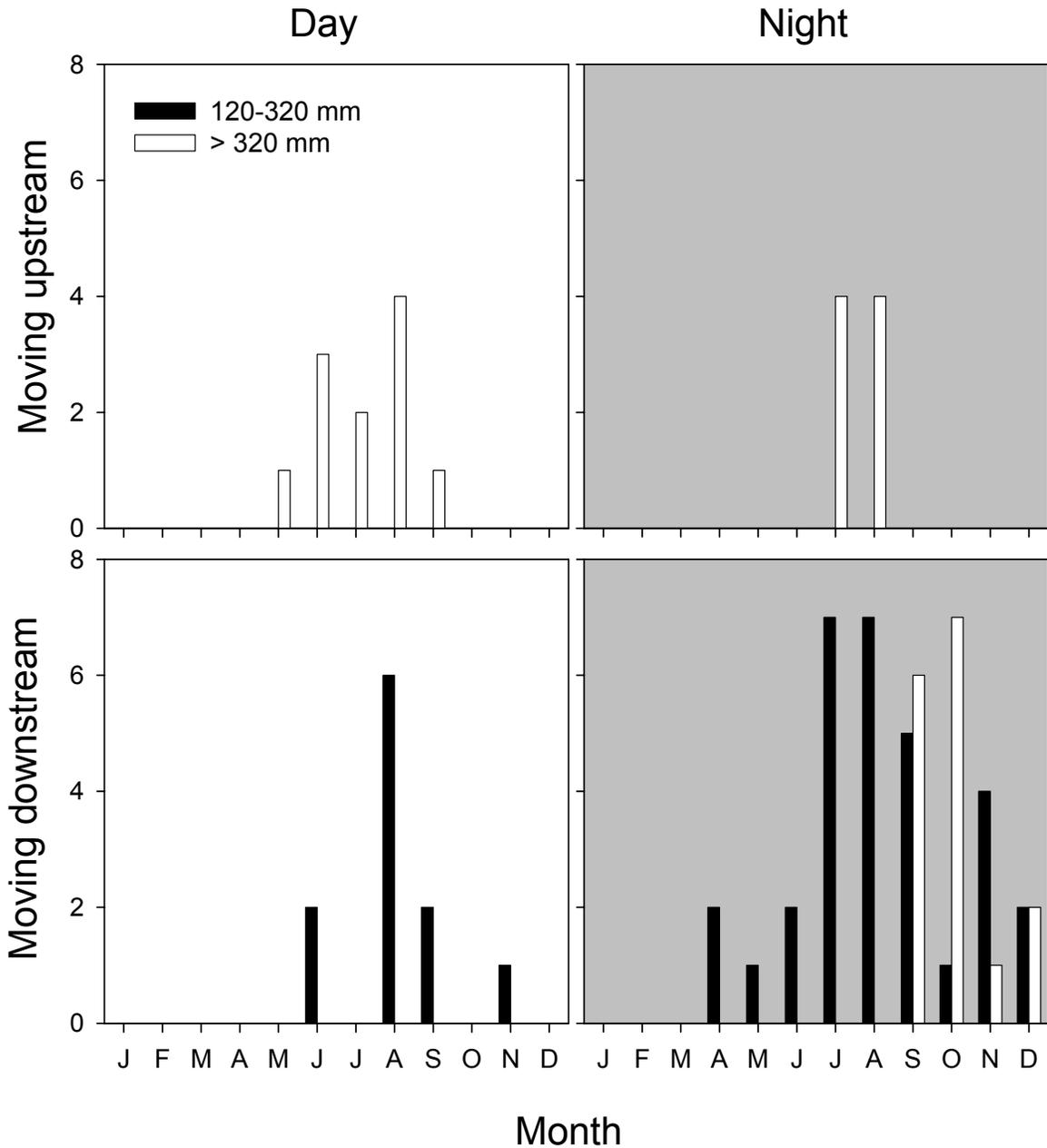


Figure 1.22. Monthly directional movement (daytime and nighttime) of small (120-320 mm) and large (> 320 mm) bull trout based on PIT-tag detections (recaptures) made at the Bear Creek detector, 2004.

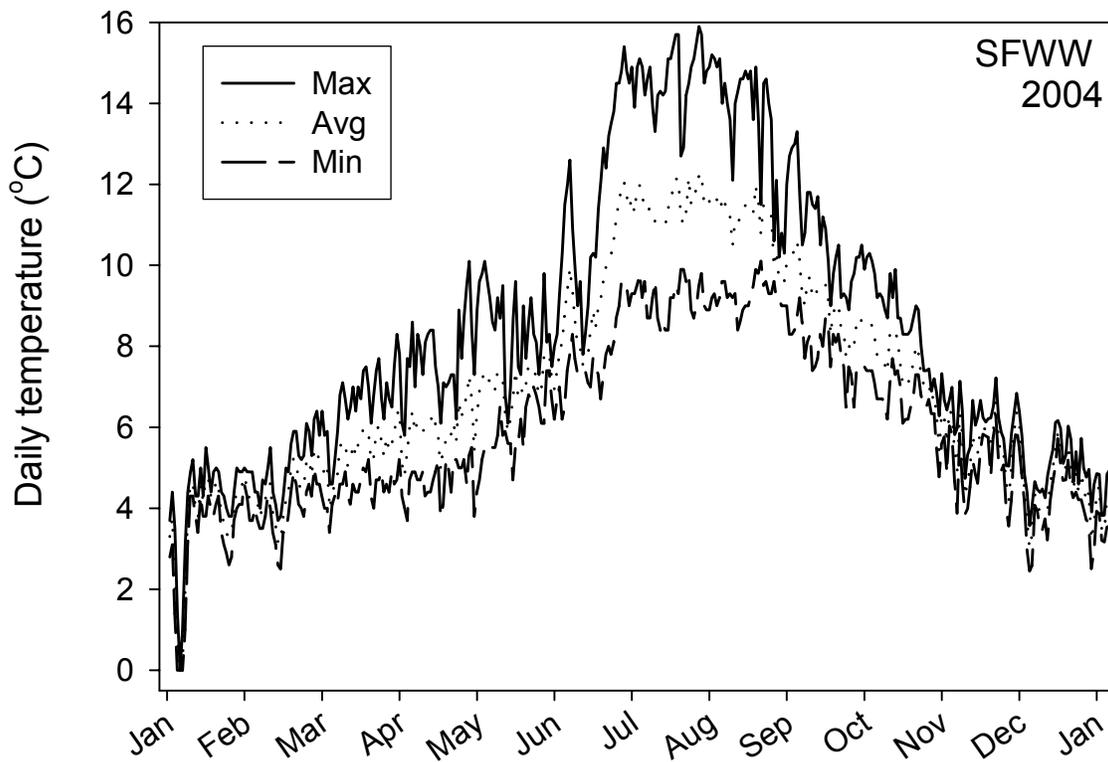


Figure 1.23. Daily temperatures (maximum, average, and minimum) recorded at Harris Park Bridge on the South Fork Walla Walla River, 2004.

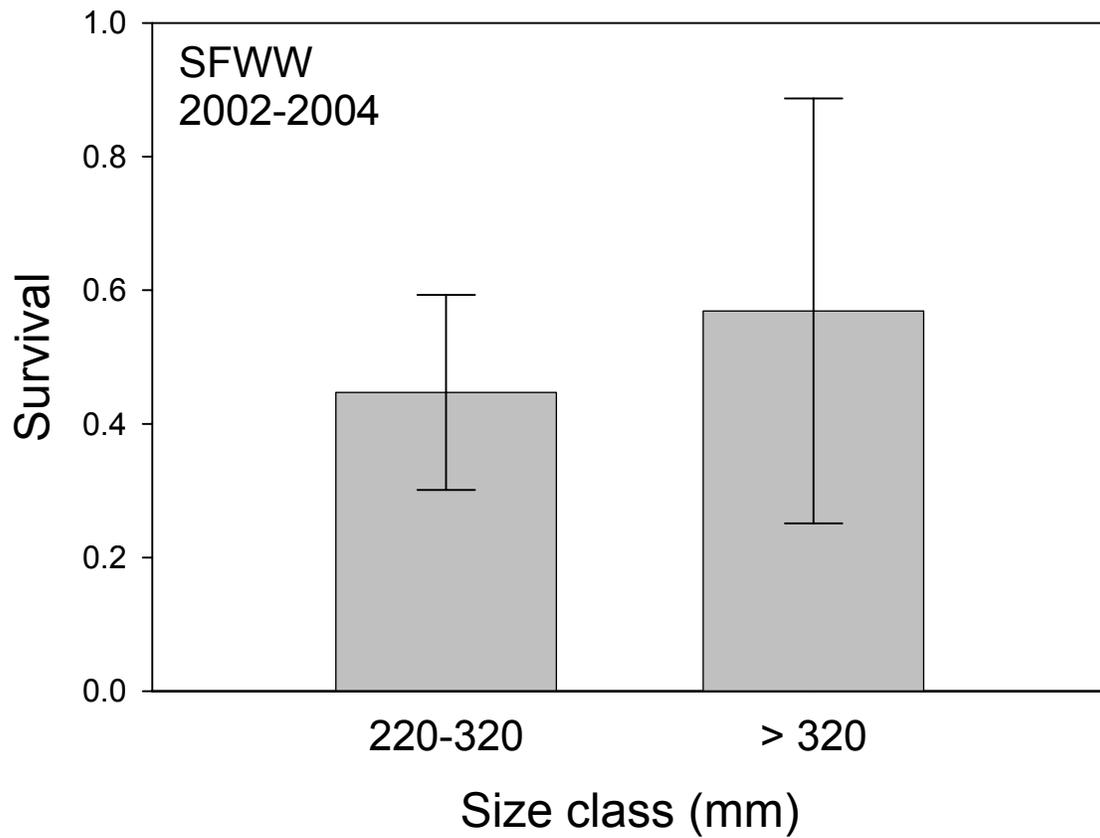


Figure 1.24. Survival estimates (± 1 SE) for two size classes of bull trout in the South Fork Walla Walla River over the period 2002 to 2004.

CHAPTER 2: Understanding the demography and significance of redd counts for bull trout

INTRODUCTION

The trend of the reproductive portion of a population is often one of the most important characteristics in the recovery and conservation of a species, yet the estimation of a population trend can be a deceptively difficult task. This task is especially complicated for species that contain multiple life history forms that coexist within a single population unit and are behaviorally cryptic (e.g., bull trout, *Salvelinus confluentus*; Maxell 1999). In addition to species-specific challenges, managers are often resource limited, and thus must rely on monitoring techniques that are cost and time effective, are specifically focused on the demographic portion of the population of interest (e.g. reproductive adults), and have not been adequately validated with alternative approaches. Given these limitations, the accuracy and precision of population monitoring techniques must be critically evaluated, as these factors can significantly affect our ability to assess the impacts of management and recovery actions implemented on long-term species persistence.

A considerable number of salmonid populations have experienced significant declines throughout their native ranges (Frissell 1993; Thurow et al. 1997). Bull trout, a species of char native to the Pacific Northwest, are no exception, and have been listed as “Threatened” in the United States since 1998 and “Of Special Concern” since 1995 in Canada. Bull trout require cold water temperatures (Selong et al. 2001, Dunham et al. 2003), and are often associated with complex habitats (Rieman and McIntyre 1993; Watson and Hillman 1997), which can limit the accuracy and precision of population-level monitoring. Like many other salmonid species (e.g. coastal cutthroat), bull trout are known to exhibit multiple life-history forms, including resident and migratory, and multiple life-history forms can coexist within a single population (Rieman and McIntyre 1993; Rieman and McIntyre 1996).

Bull trout population monitoring is often based on annual redd count surveys, where biologists visit the spawning grounds of the fish once or several times over the duration of the spawning event and count redds visually based on conditions such as the disturbance of gravel and nest structure. Bull trout return to natal headwater systems and spawn over a concentrated time period from about mid-August through

the end of October, making redd counts a cost and time effective monitoring tool for managers. Despite the feasibility of redd counts, observer variability can reduce the accuracy and precision (Dunham et al. 2001) of redd counts and the ability to detect changes in population trends based on redd counts (Maxell 1999). In addition to observer variability, redd counts may not provide an effective tool for monitoring bull trout populations where both small, resident and large, potentially migratory fish coexist within a single population unit. Finally, factors such as size differences in redd scour sites between small, resident and large, potentially migratory fish, redd superimposition, and delineation between test dig sites and redd sites can reduce the level of certainty in monitoring populations using redd counts (Maxell 1999; Dunham et al. 2001). Therefore, it is necessary to better understand what portion of the population is represented by historical and current redd count data, and how useful this information is for gauging the achievement of target recovery goals.

The purpose of this paper is to evaluate the use of redd counts as a population monitoring tool for bull trout. In particular, we are interested in examining which components of the population are represented in redd counts where multiple life-history forms are present, and if these components are consistent across basins and years. The majority of research that has investigated the robustness of bull trout redd counts has occurred in populations dominated by migratory fish (e.g., Dunham et al. 2001). To achieve our objectives, we used mark-resight techniques to provide robust population estimates and contrasted these estimates with expanded redd counts for several different bull trout populations. Mark-resight data, which has been firmly established as an accurate and precise population estimation technique (Minta and Mangel 1989) for salmonids (Zubik and Fraley 1988), allows for comparisons between different size-classes of bull trout and redd counts. This approach allows us to better understand whether potential biases occur, and if there are consistencies of these biases across different bull trout populations.

METHODS

We collected data in four separate streams in Eastern Oregon, including the South Fork Walla Walla River (SFWW; 2002 through 2004) and the North Fork Umatilla River (NFUM; 2003 and 2004), located within the Umatilla National Forest, and Big Sheep Creek (BSC; 2002) and Lick Creek (LIC; 2002), located within the Wallowa-Whitman National Forest. The elevation and size of streams (average width) differed across the study sites. The SFWW and NFUM study sites occur at relatively low elevations (610 – 1000 m) and are larger streams on average (10 m and 6 m, respectively). In contrast, BSC and LIC occur at significantly higher elevations (1370

– 1830 m), and are smaller systems (5 m and 3.5 m, respectively). Each study site was divided into reaches of at least 200 m in length, and reaches were selected for sampling using a systematic sampling design. A systematic sampling design allowed us to effectively sample across different habitat types and to account for spatial heterogeneity of bull trout distributions. Sampling rates differed across systems based on logistical constraints, yet across all years and systems, we achieved or exceeded a minimum 20 percent sampling rate.

Each year, we began sampling in mid-June and continued until the first week of August. We used a variety of sampling techniques, including corralling fish into a trapnet with snorkelers, angling, minnow traps, and scaring fish down into a seine with a backpack electroshocker, to capture bull trout. The combination of these methods has prevented potential sampling biases and enabled effective sampling across all habitat types (Budy et al. 2003). When captured, all bull trout larger than 120 mm were anesthetized, tagged with both a year-specific external anchor tag (colored Floy tag) and a surgically implanted 23-mm PIT tag, and released at the point of capture.

After sampling was completed, daytime snorkeling surveys were performed in each of the sampled reaches to collect mark-resight data each year. The time interval between marking surveys and snorkeling surveys for each reach ranged from approximately one week to one month. Although Bonneau et al. (1995) suggested that night surveys may be more appropriate for bull trout, pilot studies in 2003 and 2004 suggested no significant differences in day/night snorkel surveys (Budy et al. 2004), which may be the result of cold water temperatures (Thurow and Schill 1996). Therefore, all snorkel surveys were conducted during daylight hours (beginning two hours after sunrise and completed two hours before sunset).

To avoid potential biases that could result from double counting fish as they migrate upstream to spawn, snorkeling surveys in each study site were started at the uppermost reach and continued down to the bottom of the study site. At each reach, snorkelers proceeded upstream from the bottom of the reach and continued without stopping, estimating and enumerating all marked and unmarked bull trout into 50-mm size categories beginning with 120 mm.

Within each site, marking and snorkeling data from reach units were pooled and used to calculate annual population estimates for all tagged fish (hereafter referred to as POPTOT). We used a modified Petersen mark-recapture estimator (Seber 1982):

$$N_{hat} = ((M+1) (C+1) / (R+1)) - 1$$

where M is the number of marked fish, C is the total number of fish sighted in snorkeling surveys, and R is the number of tag resights. Normal approximation confidence intervals (95%; Krebs 1999) were calculated for each population estimate, but were not possible for population estimates with low numbers of resighted fish. We used the sampling efficiency (% of study area sampled) to expand the mark-resight population estimates and corresponding confidence intervals to the entire study area.

The Petersen mark-recapture model is a closed-population estimator, and violations of the closure assumption may significantly bias annual estimates. First, marked animals emigrating from the study site between marking and snorkeling surveys may lead to positive biases in the population estimates. We have been able to monitor the number of emigrated individuals during this period in the SFWW (see Budy et al. 2004), and have estimated emigration to have little or no effect during this period. However, we are uncertain of emigration rates in the other three systems. However, juvenile bull trout (< 200 mm) comprise the majority of bull trout outmigration during this period, while upstream migration is dominated by adult bull trout (Fraley and Shepard 1989; Hemmingsen et al. 2001; Nelson et al. 2002), which would be quantified in our snorkel surveys. In addition, there may be biases associated with fish moving between the individual reaches. However, by pooling our mark-resight data we do not violate assumptions at the study-site level.

Each population was further delineated into different size categories based on demographic information regarding the size at sexual maturity and the size range of fish considered to be migratory. Sized-based population estimates were calculated by separating the number of marked, sighted, and resighted fish into the corresponding size categories. Resident bull trout in the Pacific Northwest can reach sexually maturity at 150 mm (Hemmingsen et al 2001); data from sacrificed fish in the SFWW suggests that bull trout are sexually mature at or below 200 mm (Budy et al. 2004); limited data from NFUM, BCS, and LIC suggests similar (200 mm) or smaller sizes at sexual maturity (P. Budy, *unpublished data*). We therefore calculated a second, conservative, population estimate for all bull trout larger than 220 mm (hereafter referred to as POP220), which includes both small, resident and large, potentially migratory fish. Finally, it has also been proposed that bull trout larger than approximately 300 mm are likely to be migratory in fluvial systems (Goetz 1989; Rieman and McIntyre 1993; Nelson et al. 2002). Therefore a third, conservative population estimate was calculated for bull trout larger than 370 mm (hereafter referred to as POP370). Breaking down the total population unit into different size classes enabled us to better assess the contribution of each age or size class to the

overall population and to compare these compartmentalized and total population estimates to expanded, annual redd counts for reproductive adults.

Annual redd counts were performed by state, federal, and tribal biologists in each of these systems using streamside surveys. For the SFWW and the NFUM, redd counts began in mid-August and continued until the end of October. Counts were performed biweekly in the areas with the highest density of redds and monthly otherwise. In BSC and LIC, redd surveys were conducted twice during the spawning season in index areas of historically high spawning densities. Newly encountered redd sites were flagged to avoid the potential of double counting redds in subsequent surveys. No spawner per redd estimates were available for the systems of interest, therefore for the comparisons of redd count data to the mark-recapture population estimates that follow, annual redd counts were expanded by the number of spawners per redd based on: 1) an average of the various basin values (from the Columbia River Basin) themselves (2.68 bull trout per redd, hereafter referred to as POPREDDS; Table 2.1), and 2) the range of values available (1.2 - 4.3; Table 2.1), which were used to calculate lower and upper confidence bounds. While we acknowledge that there is great uncertainty in converting the number of redds to the number of reproductive fish, we have chosen a wide range of values for expansion to account for this uncertainty and capture the variability expected across systems.

Table 2.1. Estimates and methodologies used to estimate the number of bull trout per redd from the Columbia River Basin. An average of these four values (2.68 bull trout per redd) was used to expand redd count data in comparisons to population estimates from mark-recapture data.

Bull trout per redd estimate	Reference	Method for quantifying the number of bull trout
1.2	Baxter and Westover (2000)	Weir Counts
2.1	Sankovich et al. (2003)	Weir counts
2.3	Ratliff et al. (1996)	Weir counts
3.5	Taylor and Reasoner (1999)	Weir with fish counter
4.3	Taylor and Reasoner (1999)	Weir with fish counter
2.68	<i>Average</i>	

RESULTS

Sampling rates varied by location and year: SFWW (2002, 2003, and 2004) were 22%, 46%, and 48%, respectively; BSC and LIC (2002) were 20%, and NFU (2003 and 2004) were 52% and 37%, respectively. Total population estimates, based on mark-resight data, for each system were substantially larger than the corresponding POPREDDS (Table 2.2). This is expected given that these estimates (POPTOT) include subadult and adult bull trout, whereas redd counts presumably include only adult, sexually-mature fish. However, when we further delineated the populations into POP220 and POP370, the redd-based population estimate was not consistently lower, or higher, across the four basins.

Table 2.2. Population estimates (with 95% confidence intervals in parentheses) of bull trout based on different sources and calculation methods for four streams and different years. Asterisks indicate inadequate data.

Stream	POPTOT	POP220	POP370	POPREDDS
NFU				
2003	1983 (1331 - 5099)	145 (51 - 758)	23 (*)	59 (131 - 211)
2004	2434 (1705 - 5045)	343 (121 - 3017)	22 (8 - 549)	150 (67 - 241)
SFWW				
2002	7287 (6243 - 8895)	2695 (2444 - 3456)	1460 (1009 - 3180)	884 (396 - 1881)
2003	8861 (7579 - 10,853)	2523 (2024 - 3485)	1011 (653-3272)	965 (432 - 1548)
2004	9156 (7415 - 12,420)	1959 (1565 - 2732)	911 (591-2854)	1002 (449 - 1608)
LIC				
2002	1729 (1256 - 3026)	30 (*)	5 (*)	46 (20 - 73)
BSC				
2002	2342 (1681 - 4388)	155 (*)	10 (*)	48 (22 - 77)

For the SFWW, POPREDDS is most similar to population estimates for bull trout larger than 370 mm (POP370); this pattern is similar across the three years within the SFWW (Figure 2.1). Further, POPREDDS is well below the lower bound for the POP220 group for each year, suggesting that across years, smaller resident adult

bull trout may not be included in annual redd counts. The average annual population estimate for this group (2392) is considerably larger than POPREDDS regardless of which spawner per redd ratio is used. In addition, the total annual redd counts in the SFWW include tributaries of the SFWW, which we know contain resident bull trout, while the mark-recapture population estimates do not include resident fish from the tributaries. Thus, our population estimates from mark-resight techniques are potentially biased low at the basin level (although consistently between years), and the discrepancy between redd-count-based and mark-resight-based estimates is likely even greater than what is presented here.

In the NFUM, POPTOT estimates are substantially larger than POPREDDS, which is similar to the SFWW (Figure 2.1). However, unlike the SFWW, POPREDDS are consistently larger than POP370 estimates across the two years, and appear to include part of the smaller, resident population. These results suggest that redd counts represent both a portion of the smaller, resident fish and larger, migratory fish in the NFUM, a pattern inconsistent with the SFWW.

The data for BSC and LIC are similar to the SFWW and the NFUM (Figure 2.1). In BSC, POPREDDS is most similar to POP370, and like the SFWW, it does not appear that expanded redd counts include smaller, resident bull trout (e.g., POP220 is substantially different than POPREDDS). However, in LIC, which is in the same basin as BSC, POPREDDS is most similar to POP220, suggesting that redd counts in LIC included smaller, resident and larger, migratory fish. These results again suggest that there is considerable inconsistency in the portion of the reproductive population that is represented in redd counts data across basins.

DISCUSSION

Although redd counts provide a cost and time efficient method to monitor salmonid populations, there is significant variability between basins regarding which components of the population are best represented by redd counts. In some systems, it is only the larger, potentially migratory fish that are represented in redd counts, and in others it appears to be some combination of small, resident and large, potentially migratory fish. Ultimately, our data suggest that trends between redd counts and population estimates may be similar within systems, suggesting that redd counts may be a viable monitoring tool once validated; however, the differences in the patterns we observed across systems indicate that these trends may be basin specific.

Together, these results illustrate the need to evaluate and validate population estimation techniques for salmonid species containing multiple life-history forms. In particular, we suggest that alternative-monitoring approaches (i.e., in a subset of streams within basins) may be necessary to accurately determine the best overall strategy for effectively monitoring fish populations with redd counts. While redd counts may provide a cost-effective technique for estimating population size in some systems, a significant portion of the population (e.g., smaller, potentially resident, mature fish) may be grossly underrepresented when there is a high proportion of larger migratory fish. Further, because the differences between adult population estimates and redd counts varies in direction across different systems (i.e., sometimes they are greater than expanded redd counts and other times they are smaller), we must use caution when using these redd count data to set targets for recovery goals or in assessing the status of a population, relative to a recovery target. Ultimately, it is necessary to understand which components of the population are represented in the redd count data, and how those components relate to a specified recovery goal or criteria.

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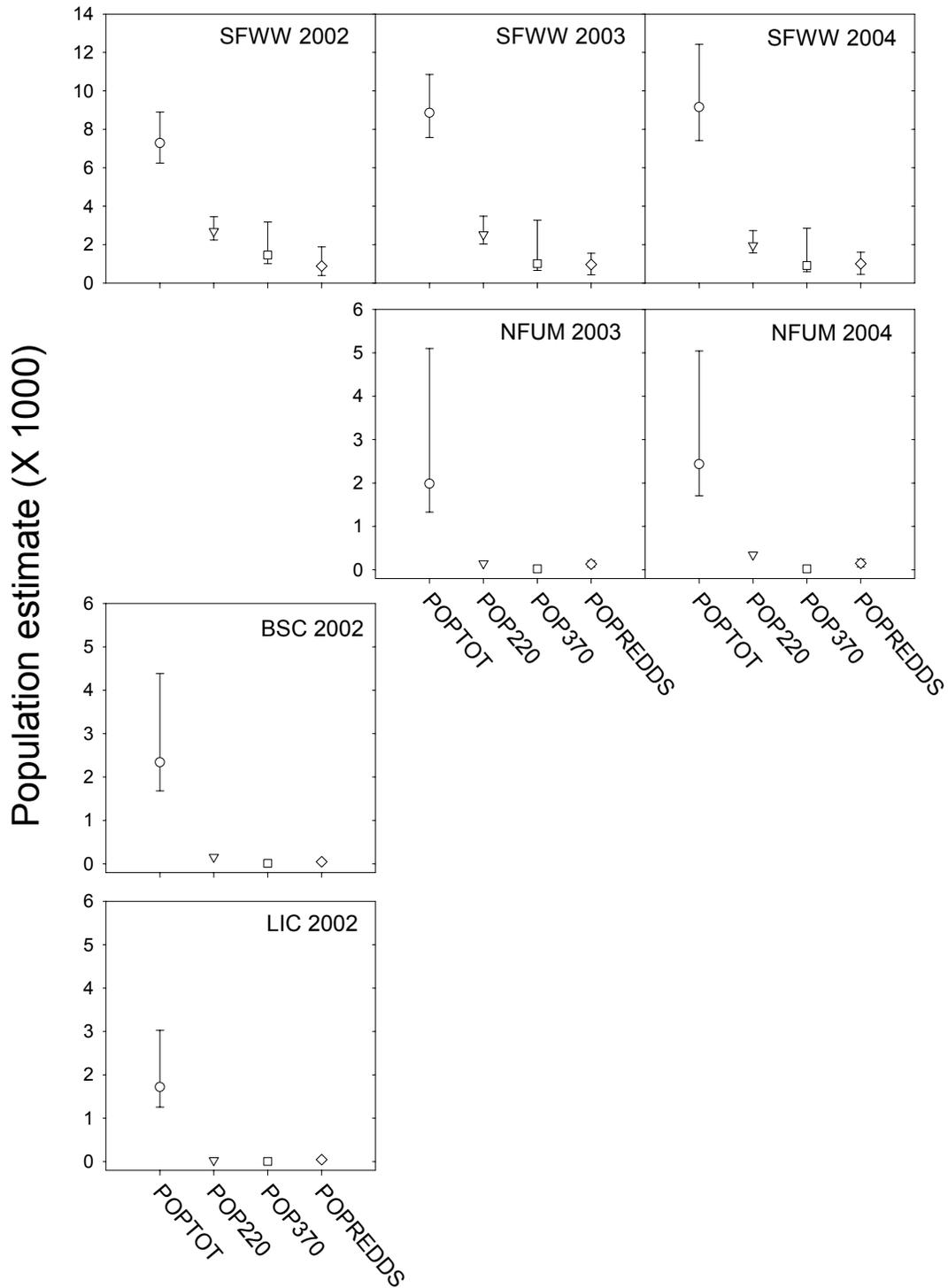


Figure 2.1. Population estimates for the SFWW, NFUM, LIC, and BSC rivers for years 2002, 2003, and 2004. The POPTOT, POP220, and POP370 are calculated from mark-resight data. Confidence intervals are not shown for estimates where sample size was too low. POPREDDS refers to the number of redds for each corresponding year and expanded where the lower error bound is for 1.2 bull trout per redd, the upper error bound is for 4.3 bull trout per redd, and the average is for 2.68 bull trout per redd (see Methods for more details).

CHAPTER 3:
**Development of a microhabitat model for bull trout with explicit
consideration of the transferability of habitat preferences
across three streams**

INTRODUCTION

Fisheries research and management have increasingly relied on physical habitat models to aid in complex decision-making processes (Rosenfeld 2003). A variety of approaches have been used to better understand the relationship between a species and its habitat requirements. In particular, these approaches attempt to link an animal's habitat requirements with presence or presence/absence (e.g., Guay et al. 2000), density data (e.g., Horan et al. 2000), and/or individual-based information (IBM, e.g., Railsback and Harvey 2002). More specifically, microhabitat suitability-type models have been used extensively as tools to estimate and predict the amount of suitable/useable and unsuitable/unusable habitat under changing flow regimes (e.g., allocated instream flows, PHABSIM, Bovee 1982). With these models, fish are assumed to select microhabitats based on the quality of multiple parameters, including water velocity, depth, substrate, and cover (Bovee 1996). However, this use of microhabitat models has been criticized due to concern over the validity of these models with respect to the analytical approach used (Vadas and Orth 2001), the criteria used to designate habitat (Thomas and Bovee 1993), erroneous use of terminology (e.g., suitability vs. preference; Rosenfeld 2003), and the disconnect of this approach with individual fitness (Garshelis 2000).

Despite the many controversies associated with microhabitat models, they are widely applied and offer empirical insight into the habitat use requirements for species or guilds (Freeman et al. 1997). In particular, microhabitat relationships may be more appropriate for understanding fish-habitat relationships at small scales (e.g., within reach) as compared to the channel unit scale (e.g., pools and riffles), where arbitrary designation of individual habitat units made by biologists may be less pertinent to the requirements of the individual or species of interest (Baxter 2002). Given these limitations, the incorporation of fine-resolution habitat relationships into a hierarchical arrangement of a species' habitat requirements, from riverscapes down to microhabitats (e.g., Fausch et al. 2002), may provide the most biologically relevant information for understanding and managing the habitat of riverine fishes.

Bull trout (*Salvelinus confluentus*) are a species of char native to the Pacific Northwest; they have been listed as “Threatened” in the United States since 1998 and “Of Special Concern” since 1995 in Canada. Recovery planning efforts for bull trout have identified the need for comprehensive information on population size and structure, demographic characteristics, dispersal and movement, and habitat requirements for at least a sub-set of bull trout populations, such that we may better design the most effective and economical monitoring, evaluation, and ultimately recovery plan (Whitesel et. al 2004). We have been monitoring several index populations of bull trout in the Blue Mountains of Oregon in response to this need. Within that context, the objective of this component of our research was to evaluate the microhabitat use of bull trout across multiple rivers. Research and monitoring have demonstrated that bull trout require cold water temperatures (Selong et al. 2001, Dunham et al. 2003) and are often associated with complex habitats (Rieman and McIntyre 1993). However, the majority of this work has occurred at large spatial scales (e.g., patch and watershed scale, Rieman and McIntyre 1995; Rieman et. al. 1997; e.g., reach and stream scale, Watson and Hillman 1997). At a smaller spatial scale (e.g., pool/riffle), Banish (2003) observed greater use of pool-type habitats by bull trout in the Cascades. In contrast, based on habitat surveys in eastern Washington (P. Sankovich, USFWS, *personal communication*) and Oregon (Budy et al. 2004), there appear to be no strong habitat use relationships at the channel unit scale (pool/riffle), while at the smaller scale within these units there are more distinct similarities in habitat use. While microhabitat studies for parr and juvenile bull trout exist (Baxter 1997; Polacek and James 2003), we are unaware of a comprehensive and validated microhabitat study for adult bull trout. Specifically, we examine the microhabitat relationships for bull trout across three relatively pristine systems, in order to better understand the natural requirements of this species.

In this study, we use multiple statistical techniques to document the microhabitat needs of bull trout, and evaluate the transferability of subadult and adult bull trout microhabitat relationships. Specifically, we use microhabitat preference curves, chi-square analyses for transferability of preferences, and logistic regression techniques to assess the consistency (across analytical techniques) of the microhabitat relationships of bull trout. By using multiple techniques, we attempt to avoid potential interpretation conflicts associated with microhabitat data (e.g., see above) and terminology associated with these conflicts, and focus on the species-habitat relationships and the interpretation of our research. Our assessment of these microhabitat relationships for bull trout, and our analysis of the transferability of microhabitat models will be useful in the Pacific Northwest, where bull trout habitats are directly and/or indirectly affected by altered flow regimes (e.g., dams and irrigation). Ultimately this greater understanding of bull trout habitat needs can be

used along with population and demographic data to guide recovery and restoration efforts, as well as to designate critical habitat.

METHODS

Study Sites—Sampling occurred in three pristine streams in northeastern Oregon during the months of June, July, and August in 2003 and 2004 (Figure 3.1). The South Fork Walla Walla (SFWW) and the North Fork Umatilla (NFUM) rivers are tributaries of the Columbia River and occur in the Umatilla National Forest; the SFWW and the NFUM study areas are approximately 21 km and 8 km in length, respectively. The South Fork Wenaha River (SFWEN), located in the Wenaha-Tucannon National Forest, is a tributary of the Grande Ronde River, and is approximately 11 km in length. Each stream is known to contain relatively large populations of bull trout (Ratliff and Howell 1992; Budy et al. 2004), and maximum summer water temperatures are not limiting within each study area (Baxter 2002; Budy et al. 2004). Finally, brook trout (*Salvelinus fontinalis*) are not present in any of these systems, offering a chance to better understand natural bull trout habitat needs.

Reaches (approximately 150 to 200 m in length) were systematically selected (with a random start) and sampled within each study site. This sampling design enabled us to quantify the range of habitat conditions available, and for habitat use, to quantify use for both rearing and small resident bull trout found in the headwater reaches, as well as larger, adult bull trout found throughout each area.

Habitat Use—Snorkel surveys were performed to quantify habitat use, and were completed prior to habitat availability measurements to minimize disturbances to fish. Although research in the Columbia River Basin has suggested that night surveys may be more appropriate for bull trout (Bonneau et al. 1995), a pilot study in 2003 suggested no significant differences in day-night microhabitat use. Therefore, all snorkel surveys were conducted during daylight hours (beginning two hours after sunrise and completed two hours before sunset).

We used techniques similar to Guay et al. (2000), where snorkelers began at the bottom of each reach, and progressed upstream, marking the locations of undisturbed bull trout with either painted rocks or large metal washers. Snorkelers also estimated both the length and focal elevation of each observed fish, which were reported to a third person on the stream bank. We categorized fish lengths into 50-mm categories beginning with 70 mm, as few age-0 fish have been observed in

previous surveys (Budy et al. 2004). To better understand the appropriate depth for focal water velocity measurements, we categorized fish focal elevation into four categories including bottom $\frac{1}{4}$, $\frac{1}{4}$ to $\frac{1}{2}$ depth, $\frac{1}{2}$ to $\frac{3}{4}$ depth, and $\frac{3}{4}$ to water surface.

At each marked focal position, we measured water depth, bottom and average water column velocities, cover and cover type, and dominant substrate. Habitat characteristics were measured to represent average values within a one-meter square; measurements were taken at the center of each square. Water velocity was measured with an electromagnetic flow meter, and the 20-second running average was reported for each location. Average water column velocity was measured at 60 percent of the total water depth (taken from the water surface), and bottom water velocity was measured at approximately 2 cm off of the substrate to prevent measurements within the substrate.

We considered multiple cover types in our data collection including vegetation (VEG), large woody debris (LWD), undercut bank (UC), boulder (BD), turbulence (TB), and depth. Any cover found within one meter of the original focal position of the fish was classified as cover. Undercuts and boulders were considered as cover if the depth of an undercut area was at least 5 cm deep, 10 cm in length, and 5 cm in height (Kershner et al. 2004). Pieces of wood that were at least 1 m in length and 10 cm in diameter were classified as LWD. Overhanging vegetation was classified as present when it occurred within one meter of the water surface, and protruded from the bank at least 0.5 m. Since turbulence and increasing depth (Cunjak 1996) are considered to be surrogates for physical cover for instream fishes, we classified turbulence as cover when it prevented the observer from accurately viewing the stream substrate with a plexiglass viewer, and classified depth as cover when greater than 0.7 m. Finally, substrate was classified ocularly with a plexiglass viewer. We recorded the dominant substrate according to Geist et al. (2000) within a one-meter square; substrate was classified as: 1 (0-6 mm), 2 (7-25 mm), 3 (26-50 mm), 4 (51-75 mm), 5 (76-125 mm), and 6 (>126 mm). When depths permitted, ocular estimates were validated with measurements of the dominant substrate particles with a hand ruler.

Habitat Availability—Within each reach, we used systematically spaced transects to measure habitat availability. A systematic design enabled us to account for habitat heterogeneity across different channel types (e.g., pool, riffle). Transects were sampled throughout each reach, and the distance between transects was approximately 10 to 15 m. At each transect, we sampled ten equidistant points perpendicular to the thalweg flow, with intervals between each sampling point based on the wetted width at the transect location. To account for near-bank habitats often selected by bull trout (*personal observation*), we also sampled two additional points

at 10 cm off each of the wetted stream boundaries. Similar to the habitat use locations, we measured water depth, bottom and average water column velocity, cover and cover type, and dominant substrate within one square meter at each point.

Data Analysis

Habitat Preference Curves—Habitat use and availability data were used to calculate bull trout habitat preference curves. Within each stream, preference curves were created for: 1) all observed bull trout, 2) bull trout < 220 mm (to represent juvenile and subadult fish), and 3) bull trout > 220 mm (to represent both resident and migratory adult fish). Habitat preference for each microhabitat variable was calculated as the percent of used habitat / percent available habitat according to Baltz (1987), and scaled to a range of zero to one. The continuous variables, depth and velocity, were grouped into six intervals: 0-20 cm, 21-40 cm, 41-60 cm, 61-80 cm, 81-100 cm, and > 100 cm. Habitat preferences for cover and substrate were calculated for each category of the respective variable.

To assess the transferability of habitat preference curves, we used the SFWW data set as the base model against both the NFUM and SFWEN data. Specifically, we classified the central 50 percent and 95 percent of the frequency distributions from microhabitat use data in the SFWW as optimal and useable habitat, respectively, and anything outside of the 95 percent central distribution as unsuitable (Thomas and Bovee 1993). The use and availability data for each microhabitat parameter (depth, cover, etc.) were classified into three categories as optimal, useable, and unsuitable. Within this framework, we used a composite index of the microhabitat characteristics of each cell, where cells were classified as “optimum” if all habitat characteristics were optimum (as defined above), “useable”, if all characteristics were optimum and/or usable, and “unsuitable” if any of the measured characteristics were considered unsuitable.

Use and availability data from the NFUM and the SFWEN were subsequently classified as either optimum, useable, or unsuitable based on the central 50 % and 95% of the SFWW frequency distributions. Unlike Thomas and Bovee (1993), we used a two-tailed chi-squared (χ^2) test to determine the transferability of the model (Maki-Petays et al 2002). With this approach, we tested the null hypotheses that composite optimum, useable, and unsuitable cells were used in the same proportion as available in all three systems. The model was considered transferable if the null hypothesis for each test was rejected at $\alpha = 0.05$. Although the χ^2 test can be a weak test statistic for transferability tests (Williams et al. 1999), it can provide additional supporting evidence for consistent habitat preferences across streams

(Freeman et al. 1999) when used with additional (both qualitative and quantitative) measures.

Logistic Regression—We also evaluated the influence of microhabitat parameters on bull trout occurrence using logistic regression. In our model, depth, substrate, and water velocity were modeled as continuous variables, and cover was grouped and modeled as a dummy variable (yes if any cover was present, and no otherwise) to minimize the number of parameters in the model. Because of the potential effects of density on habitat selection (Hayes et al. 1996) and the model, we also included average bull trout density (reach level; from snorkeling surveys) for each system as an explanatory variable.

We also investigated the effects of an unequal number of response cases on the interpretation of the logistic regression analysis, as this inequality can affect both the significance of the explanatory variables, and the validity of the validation analyses. Within our SFWW data set, the number of “absences” (n = 1722) greatly exceeded the number of “presences” (n = 120). To examine the effects of these inequalities, we randomly subsampled 120 “absence” observations (equal to the number of “presences”) from the total number of “absence” cases, and reexamined the influence of the explanatory variable (see above) on the presence/absence of bull trout using logistic regression (hereafter referred to as the “subset” model). Prior to the final analyses, we checked the model for multicollinearity among the explanatory variables

The logistic regression models from the SFWW (both full model and subset model) were validated both internally and externally, with data from the NFUM and the SFWEN. Internally, we used cross-validation techniques, and measured the success of the model by the sensitivity, percent of correctly classified presences, and specificity, percent of correctly classified absences, values. We externally validated the logistic model from the SFWW by predicting presence/absence in both the NFUM and the SFWEN, and then comparing the presence/absence predictions to actual field observations. The probability of presence or absence was then calculated as:

$$P(Y_i = 1) = e^{g(x)} / (1 + e^{g(x)})$$

where Y_i is the response variable, e is the natural logarithm, and $g(x)$ is a linear model of the explanatory variables. The transferability of the model was evaluated by calculating sensitivity and specificity values for each system. All statistical analyses were conducted with SAS software (SAS Institute 2000), and statistical significance was assessed at $\alpha = 0.05$.

RESULTS

Across systems, bull trout used relatively deep, slow-water habitats with cover (Table 3.1). Substrate size classes appeared relatively consistent across systems, with the exception of the SFWEN, where bull trout < 220 mm used smaller substrate sizes. Density estimates for each system were very low, suggesting that these populations are not currently approaching densities where saturation of optimal habitats may lead to the use of suboptimal habitats.

Table 3.1. Summaries of microhabitat use and availability data for the NFUM, SFWEN, and the SFWW rivers and a *Subset* (described in text). Means (with standard deviation in parentheses) are reported for each parameter. Cover was combined and reported as the percent of cells, either used or available, that had cover. Microhabitat use is separated by bull trout size classes: < 220 mm and > 220 mm. Sample size (*n*) corresponds to each analysis.

	Depth (m)	Average velocity (m/s)	Bottom velocity (m/s)	Substrate (Size class)	Percent of cells with cover	<i>n</i>	Bull trout average density (per 100 m ²)
NFUM							0.0126 (0.0118)
Available	0.17 (0.14)	0.42 (0.29)	0.27 (0.25)	4.49 (1.57)	0.18	419	
< 220 mm	0.35 (0.18)	0.19 (0.16)	0.10 (0.10)	4.43 (1.73)	0.91	23	
SFWEN							0.0106 (0.0078)
Available	0.23 (0.14)	0.58 (0.38)	0.30 (0.26)	4.36 (1.57)	0.41	527	
< 220 mm	0.37 (0.12)	0.22 (0.19)	0.08 (0.08)	2.28 (1.6)	1.00	18	
> 220 mm	0.44 (0.21)	0.44 (0.30)	0.17 (0.17)	4.50 (1.58)	0.90	10	
SFWW							0.0081 (0.0053)
Available	0.33 (0.28)	0.59 (0.47)	0.28 (0.28)	4.60 (1.40)	0.21	1722	
< 220 mm	0.39 (0.21)	0.24 (0.23)	0.12 (0.14)	4.70 (1.62)	0.75	44	
> 220 mm	0.53 (0.29)	0.24 (0.24)	0.12 (0.13)	4.17 (1.56)	0.77	29	
Subset							
Available	0.34 (0.21)		0.30 (0.29)	4.85 (1.28)	0.28	120	

The majority of bull trout observed during snorkeling surveys was on or associated with the stream bottom. Specifically, 88% of observed bull trout in the SFWW (the remaining 12 percent were within the bottom ½ of the water column) and 100% of observed in both the NFUM and the SFWEN, were classified as occupying the bottom ¼ of the water column. Therefore, while both average and bottom water

column velocity measurements were collected originally, bottom water velocity was used in all subsequent analyses.

Habitat Preference Curves—Habitat preference curves suggested similar use patterns across the three systems (Figure 3.2). For each system, it appears that smaller substrate was the most preferred, and that the degree of preference for larger substrate classes varied across systems, although consistently. Specifically, preference values were consistently the highest for all size classes in the NFUM (except for substrate class two where no observations were recorded), and consistently the lowest for larger size classes in the SFWEN. For bottom water column velocity and depth, bull trout preferred slower water velocities and deeper habitat. Finally, the preference curves were consistent in demonstrating the general avoidance of habitats without cover; however, the type of cover and preference for cover type varied across systems.

The SFWW had the largest sample size of observed bull trout ($n = 83$) across the three systems that we evaluated, and therefore we compared habitat preferences separately for small (< 220 mm) and large (> 220 mm) bull trout in this system (Figure 3.3). Similar to the general patterns observed across systems, both groups preferred habitat with lower bottom water velocities, increased depths, cover, and small substrate. However, these results also suggest that compared to smaller bull trout, larger bull trout more strongly avoided shallow habitats and typically preferred the use of LWD as cover, while smaller bull trout preferred the use of boulders as cover.

Based on the χ^2 transferability analyses, habitat preferences measured in the SFWW transferred adequately to both the NFUM and the SFWEN. All three χ^2 tests were significant ($P < 0.0001$, $df = 2$), rejecting the null hypotheses that microhabitats were used in similar proportion to what was available. In each of the systems, optimal and usable habitats were used at a significantly greater proportion than available habitats, and unsuitable habitats were used significantly less than available habitats (Figure 3.4).

Logistic Regression—Preliminary diagnostics demonstrated no multicollinearity among the explanatory variables; therefore, we ran the logistic models with cover, depth, bottom water velocity, substrate, and density as explanatory variables. The full model resulted in a reasonable fit with a max-rescaled R^2 of 0.221 ($P < 0.0001$, $df = 5$). Depth, bottom water velocity, and cover were all significant ($P < 0.001$) in predicting the presence/absence of bull trout at the microhabitat scale, while substrate and density were insignificant parameters (Table 3.2). The “subset” logistic

regression model demonstrated a better fit overall, with a max-rescaled R^2 of 0.463 ($P < 0.001$, $df = 5$). As with the full model, depth, bottom water velocity and cover were all significant ($P < 0.008$) in predicting the bull trout presence/absence (Table 3.3). Descriptive statistics of the subset “absence” data indicated adequate representation of the total “absence” data (Table 3.1). Thus, results from both logistic regression analyses were conceptually identical. Similar to preference curve analyses (above), logistic regression indicated that bull trout presence at the microhabitat scale appeared to increase with depth and cover, and decrease as bottom water velocities increased.

Table 3.2. Parameter estimates with standard error, odds ratio point estimates, and P -values for explanatory variables from logistic regression analysis (*Full model*).

Parameter	Estimate	Standard error	Odds ratio point estimates	$Pr >$ Chi-square
Cover	0.818	0.129	2.266	< 0.0001
Bottom water velocity	-5.221	0.820	0.005	< 0.0001
Depth	2.608	0.393	13.573	< 0.0001
Substrate	0.001	0.069	1.001	0.988
Density	0.392	0.209	1.479	0.061

Table 3.3. Parameter estimates with standard error, odds ratio point estimates, and P -values for explanatory variables from the *Subset* logistic regression analysis.

Parameter	Estimate	Standard error	Odds ratio point estimates	$Pr >$ Chi-square
Cover	1.4539	0.3178	4.280	< 0.0001
Bottom water velocity	-4.1707	1.0256	0.015	< 0.0001
Depth	3.1710	0.8040	23.832	< 0.0008
Substrate	0.00497	0.1009	1.005	0.9607
Density	-2.4171	35.134	0.089	0.9452

For the full model, internal and external validation of the model suggested that depth, bottom velocity, and cover accurately predicted bull trout absence at the microhabitat level, but that the current model is less effective at predicting bull trout presence. Sensitivity values were all less than 1%, which suggested that the current logistic model could not accurately predict bull trout presence at the microhabitat scale. However, specificity values for the SFWW, SFWEN, and NFUM were 99.6%, 100%,

and 99.7%, respectively, indicating that depth, bottom velocity, and cover parameters can be used to accurately predict bull trout absence.

Both internal and external validation of the subset model, however, suggested that depth, bottom velocity, and cover were effective at predicting bull trout presence and absence. Sensitivity and specificity values from the jackknife validation of the model were 74.6% and 81.5%, respectively. External validation suggested that the model transferred well to both the SFWEN and NFUM for both sensitivity tests (96.5% and 91.3%, respectively) and specificity tests (61.0% and 69.3%, respectively).

DISCUSSION

Habitat Use and Preference

We used multiple analytical approaches to illustrate consistent patterns of microhabitat use and preference for bull trout across three streams in northeastern Oregon. Bull trout appear to be selecting and avoiding specific habitats based on the microhabitat parameters of depth, bottom water velocity, and cover. The corroboration of our results with work at similar and larger scales suggests that these patterns are transferable across systems. For example, cover, which is often a result of habitat complexity, demonstrated to be an important parameter affecting the distribution of bull trout at both intermediate scales (Watson and Hillman 1997) and larger scales (Rieman and McIntyre 1993; Rieman et al. 1997). Water depth has been demonstrated to be an important variable at the intermediate scale in both Oregon (Banish 2003) and central Idaho (Zurstadt 2000). Across scales, the consistency of bull trout associations with deeper, habitats with cover is indicative of their basic biology. Throughout their range, subadult and adult bull trout are considered to be largely piscivorous (Fraley and Shepard 1989; Rieman and McIntyre 1993), and their habitat selection and physiology (e.g., large head; Markle 1992) are consistent with the biology of an ambush predator. Similar to depth and cover, bull trout preferences for slower water habitats were consistent across systems. Our results are supported by recent physiological work by Mesa et al. (2003), which illustrated that bull trout have low critical swimming velocities relative to other salmonids. Bioenergetically, this would suggest that bull trout should use slower water habitats to maximize their fitness.

Substrate preferences varied across systems, and substrate was not a significant variable for predicting the presence/absence of bull trout. However, substrate has been illustrated to be important for bull trout at the microhabitat scale in other

systems (Thurow 1997). There are, however, inherent difficulties in quantifying intra-substrate microhabitats, which are commonly used by smaller bull trout (Jakober 1995; Thurow 1997; Zurstadt 2000). These sampling difficulties may have resulted in an underestimation of the importance of substrate at the microhabitat scale in our study. Further, our research was conducted in relatively pristine watersheds, which generally do not experience excessive sediment loading (Budy et al. 2004). Under these pristine conditions, distinct substrate preferences (or avoidances) may not exist because the substrate remains largely suitable throughout the study areas.

The primary limitation to this type of approach (or many habitat modeling efforts) is the effect of density on habitat use and availability. In particular, high density can lead to the use of suboptimal habitats, and at lower densities, much of the optimal habitat may go unused (Power 1984; Rosenfeld 2003). Both of these scenarios can increase the difficulty in understanding and quantifying species-habitat relationships. In the SFWW, NFUM, and SFWEN, bull trout densities were extremely low (0.0081-0.0126 bull trout per 100 m²), suggesting that optimal habitat was not saturated, despite the fact that total abundances within each basin are relatively large (Baxter 2002; Budy et al. 2004). Although bull trout (>120 mm; 2003) population estimates based on mark-resight techniques in the SFWW and NFUM were 8,533 (95% CI: 7,839 - 11,470) and 1,741 (1,145 - 2,337), respectively, the densities (given above) of bull trout are substantially lower than those reported for small central Idaho streams (snorkel surveys; 0.10 - 3.22 bull trout per 100 m²; Zurstadt 2000). Ideally, we would measure habitat selection in a system where optimal habitats were saturated (Greene and Stamps 2001; Rosenfeld 2003), which would allow us to evaluate changes in habitat use across a range of bull trout densities. However, this approach may be difficult to implement for a species such as bull trout, where densities, even in relatively pristine watersheds, are typically relatively low (Rieman and McIntyre 1993).

Management Implications

Through the use of multiple analytical methods, we have demonstrated consistent bull trout habitat preferences at the microhabitat scale. Our findings are pertinent across disciplines, and in highly managed systems for restoration activities, construction of passage facilities, and the designation of critical habitat for bull trout. For example, in systems where flows are managed and habitat alterations have reduced the complexity of the systems, knowledge of specific bull trout habitat preferences may be integrated into the decision process regarding the timing and magnitude of release events. In particular, this could be critical in summer months when migrations of fluvial and adfluvial bull trout occur (Rieman and McIntyre 1993;

Nelson et al. 2002), and water temperatures are relatively high, resulting in shallow, warm-water stream channels. In addition, in systems where large releases occur in months that are atypical of ambient flow regimes, it is necessary to consider the effects of high flows on the habitat use and ultimately the fitness of migrating and resident bull trout. Finally, the decisions surrounding the designation of critical habitat and the designs of restoration projects need to consider bull trout preferences/requirements at smaller spatial scales, which may be more relevant than processes at larger scales (e.g., channel unit and reach level). Ultimately, the consistency of our findings across systems and with studies at larger spatial scales suggests that our results are widely applicable to bull trout across the northwest.

The management of a species often requires explicit knowledge of habitat relationships at multiple scales (Dunham and Vinyard 1997; Fausch et al. 2004). While we have illustrated consistent patterns of habitat use and preference at the microhabitat scale, additional research is necessary to better understand other factors that may be driving the distribution of this species. When looking across scales, it may be important to investigate the role of biotic factors (Orth 1987; Maki-Petays et al. 1999), such as consumable resources (e.g., forage) and interactions with other species, and how these factors affect bull trout distribution and habitat preferences. For example, an understanding of the behavioral and competitive effects of brook trout, a non-native with similar habitat requirements (Gunckel et al. 2002), may provide additional insight into bull trout habitat and foraging requirements, and ultimately, fitness. Furthermore, research is needed at the intermediate scale (e.g., pool or riffle) to better understand why bull trout habitat relationships at this scale are inconsistent. The consistency of bull trout habitat use and preferences that we have illustrated can ultimately be used as a physical template to guide the appropriate measures (e.g., habitat protection or restoration) leading to the recovery of this species. However, the recovery of a sensitive species, such as bull trout, will ultimately require additional consideration of how habitat and biotic factors interact and affect the fitness of an individual throughout its life cycle.

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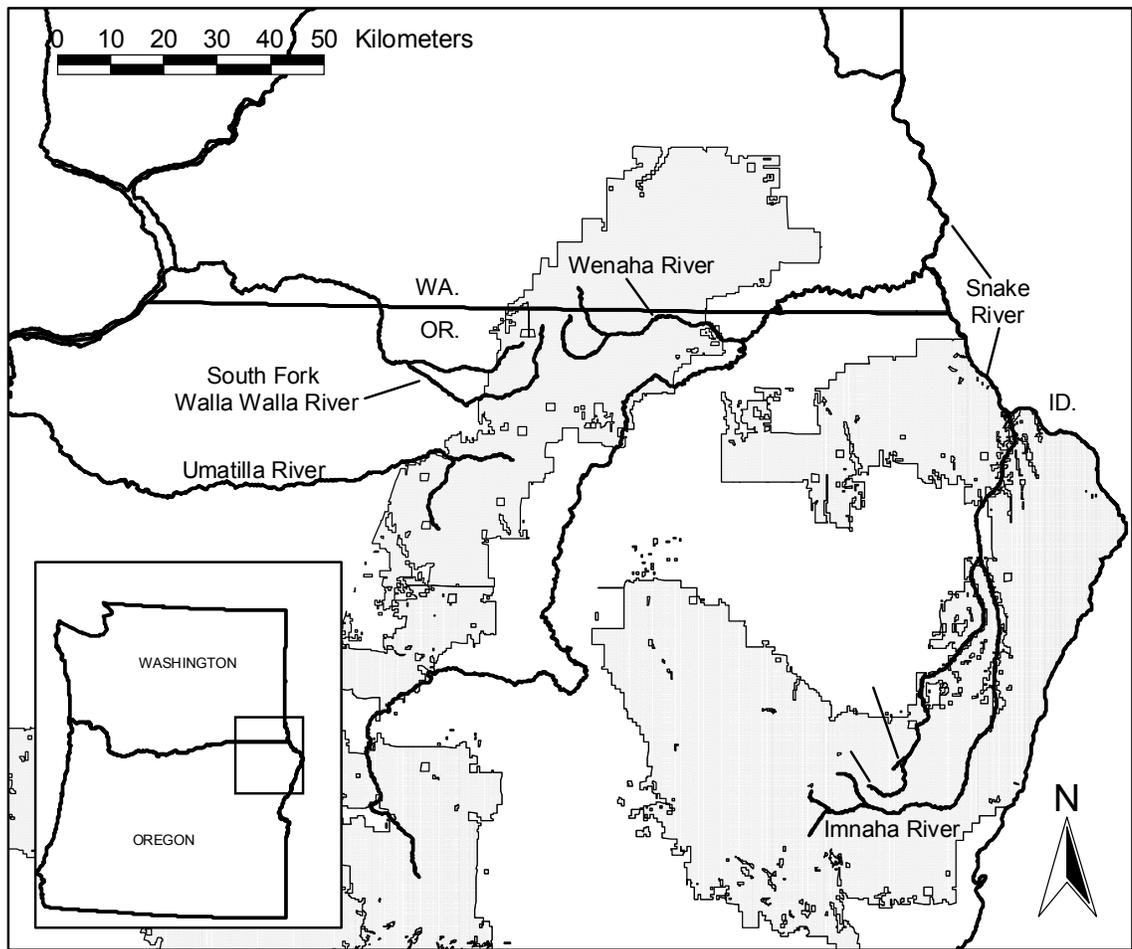


Figure 3.1. A regional map of northeast Oregon and southeast Washington. Microhabitat use and availability data were collected on the South Fork Walla Walla (SFWW), North Fork Umatilla (NFUM), and the South Fork Wenaha (SFWEN) rivers.

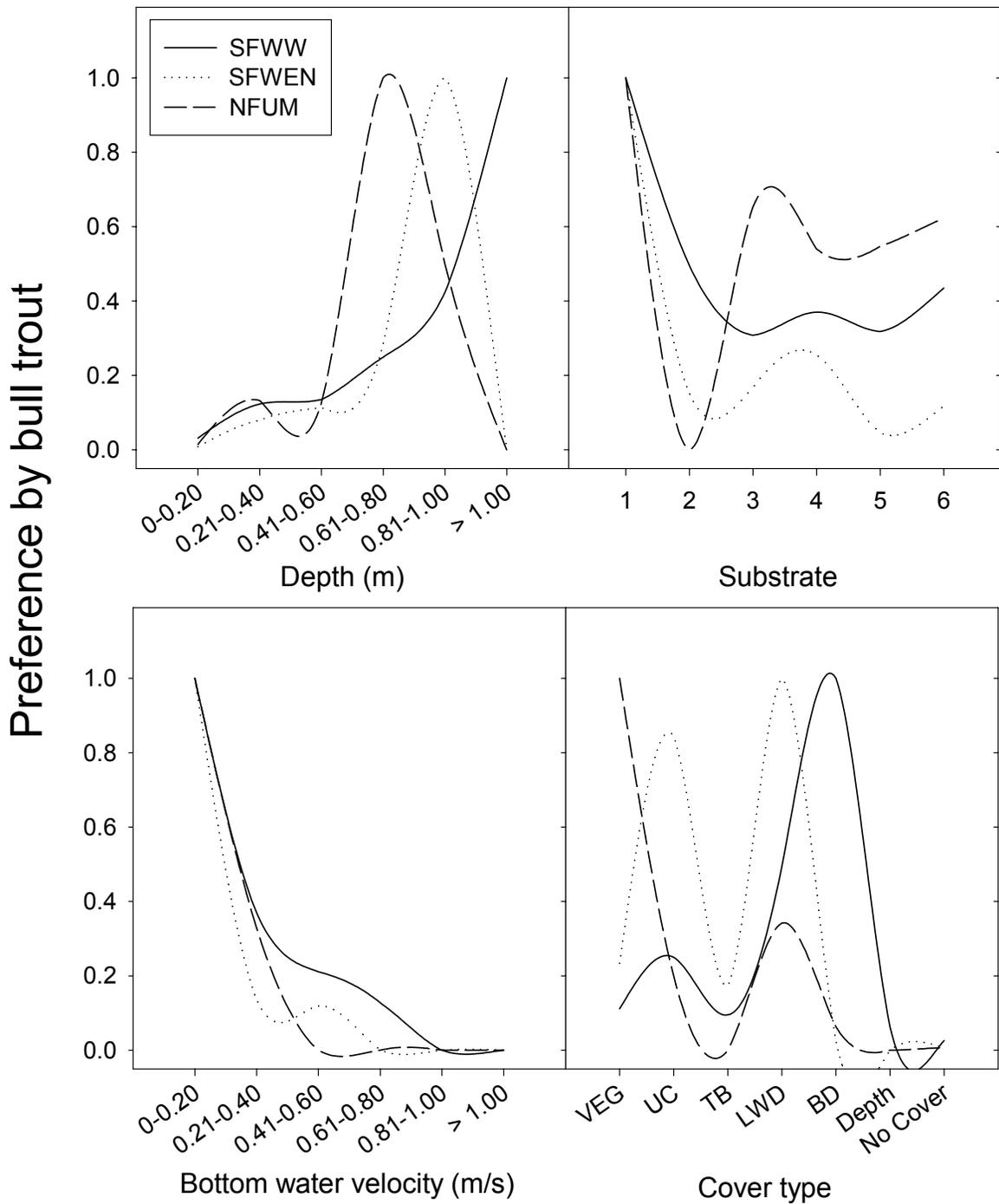


Figure 3.2. Habitat preference curves for depth, substrate, bottom water velocity, and cover type for the SFWW, SFWEN, and NFUM rivers, all bull trout combined, 2004.

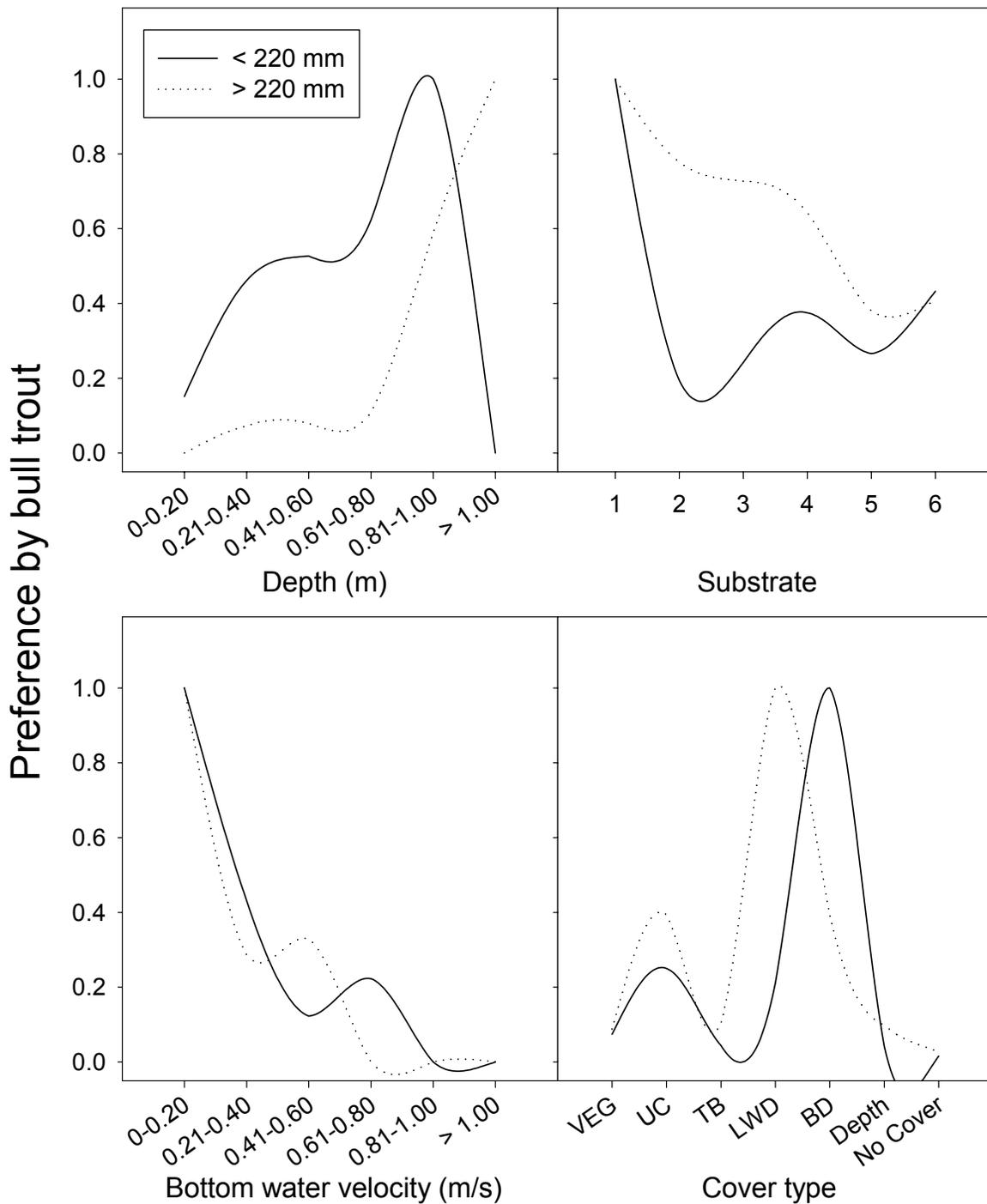


Figure 3.3. Habitat preference curves for depth, substrate, bottom water velocity, and cover type for different size classes (< 220 mm; > 220 mm) of bull trout in the South Fork Walla Walla River, 2004.

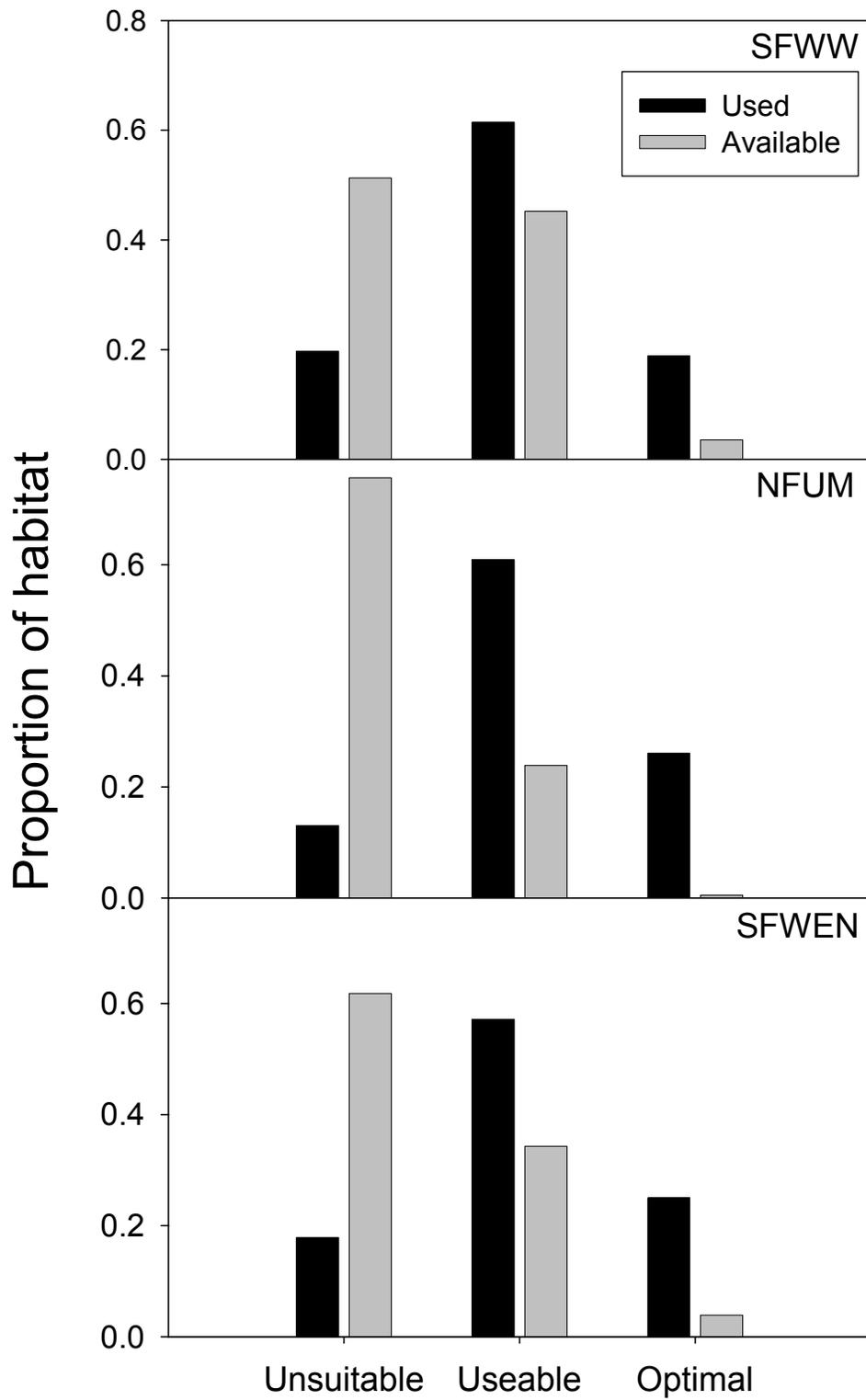


Figure 3.4. Proportion of optimal, useable, and unsuitable habitat using transferability criteria from the SFWW (2004), and tested on the NFUM and the SFWEN.

APPENDIX 1

Table A1. Summary of all fish captured, tagged, and counted (sighted during snorkeling surveys) in sampled reaches of the South Fork Walla Walla, North Fork Umatilla, and South Fork Wenaha rivers, June to August 2004. All sizes combined. No capture activities (*na*) were conducted in the SF Wenaha River.

Species sampled	Activity	SF Walla Walla	NF Umatilla	SF Wenaha
Bull trout				
	Captured	754	190	<i>na</i>
	Tagged	413	60	<i>na</i>
	Counted	731	346	177
<i>O. mykiss</i> spp.				
	Captured	324	482	<i>na</i>
	Counted	3357	1810	2136
Chinook salmon				
	Captured	87	106	<i>na</i>
	Counted	1450	800	3079
Mountain whitefish				
	Captured	3	17	<i>na</i>
	Counted	67	150	0
Sculpin				
	Captured	8	0	<i>na</i>
	Counted	27	25	0

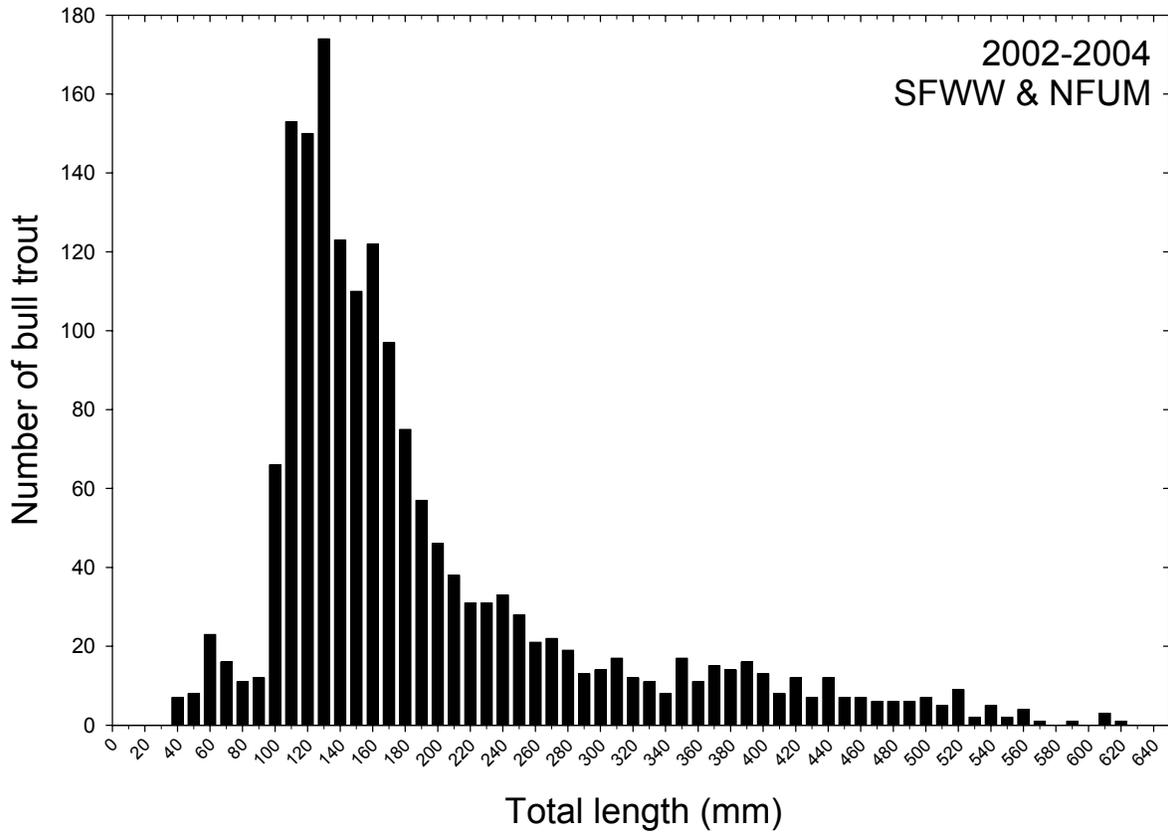


Figure A1. Length frequency distribution of bull trout sampled (n = 1745) in the South Fork Walla Walla River (2002 - 2004) and North Fork Umatilla River (2003 - 2004); both streams and all years combined.

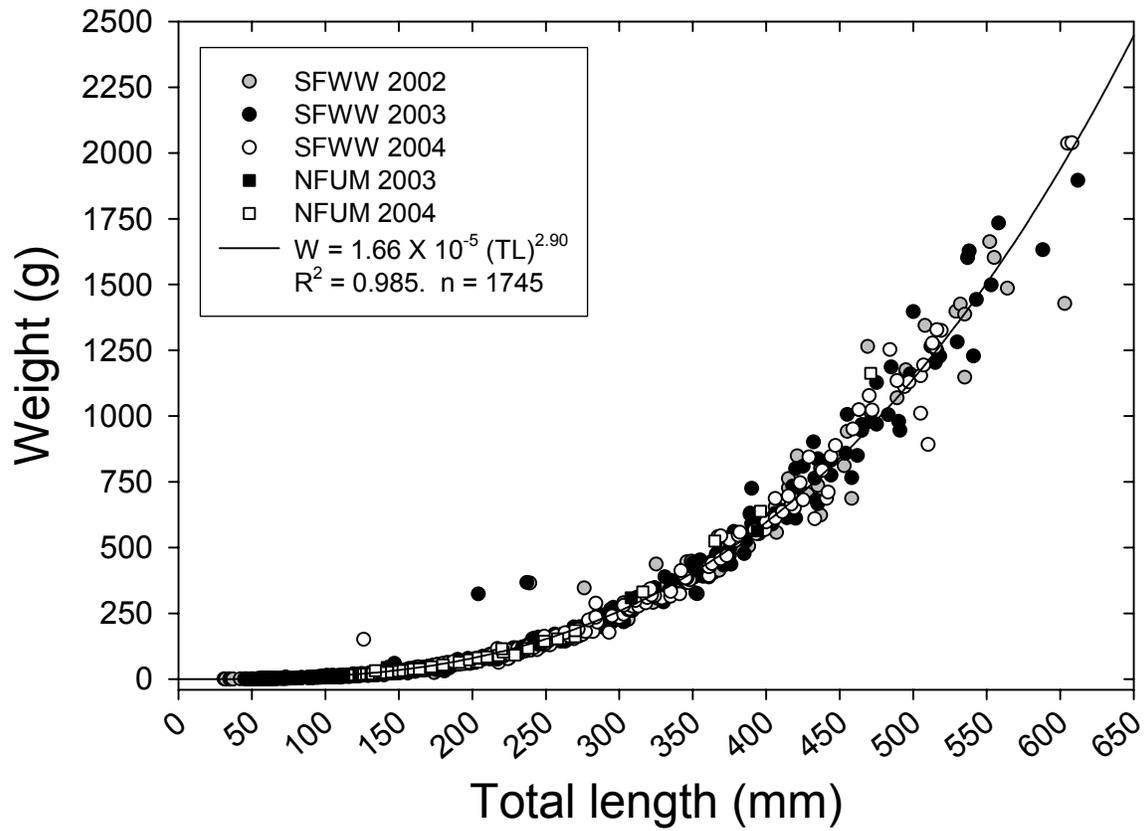


Figure A2. Combined length-weight relationship for bull trout sampled in the South Fork Walla Walla and North Fork Umatilla rivers, 2002 - 2004. Regression equation, R^2 -value, and sample size are given.

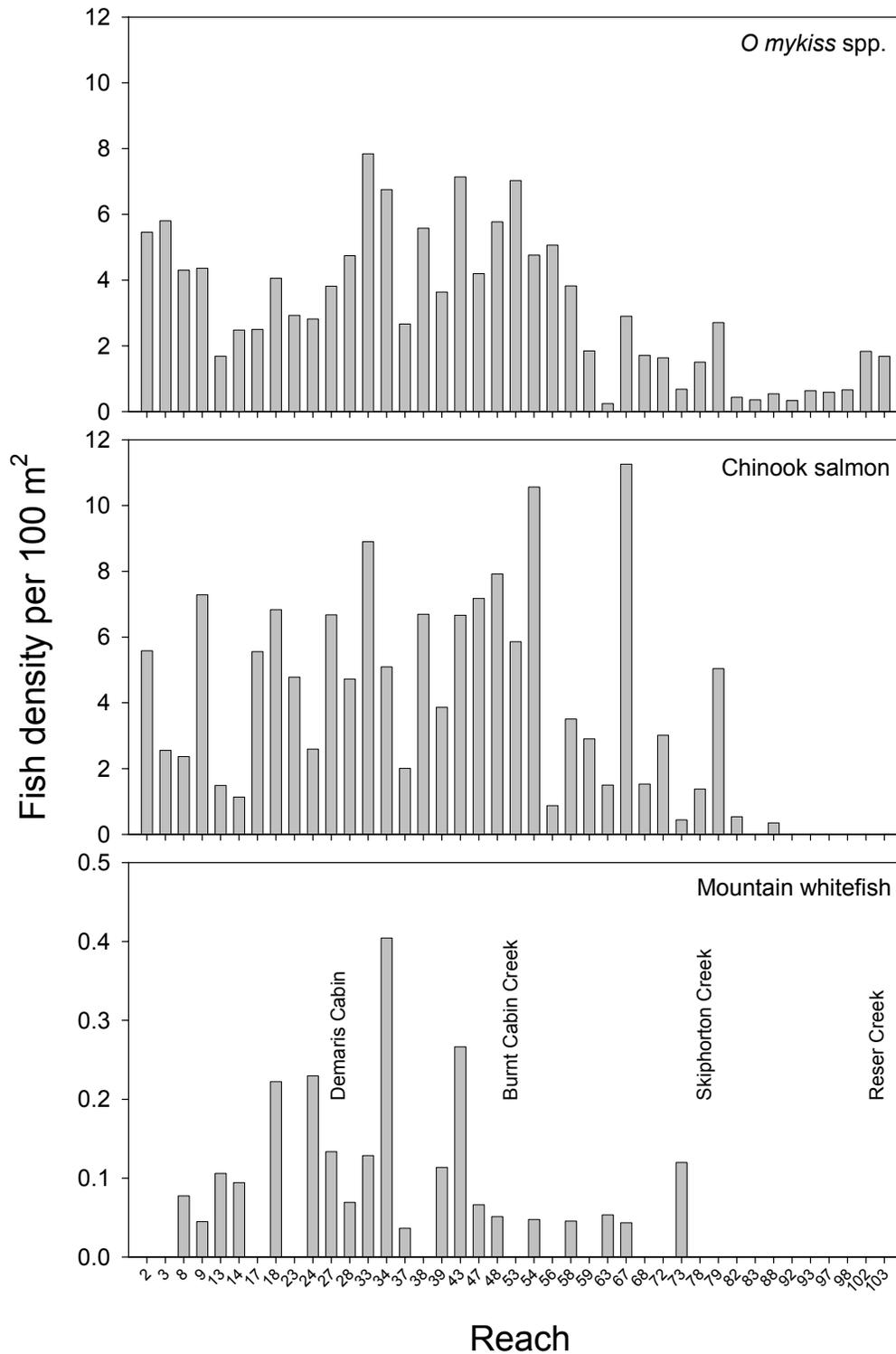


Figure A3. Density of *O. mykiss* spp., Chinook salmon, and mountain whitefish estimated by snorkel counts in various reaches of the South Fork Walla Walla River, summer 2004. Note changes in y-axis scales.

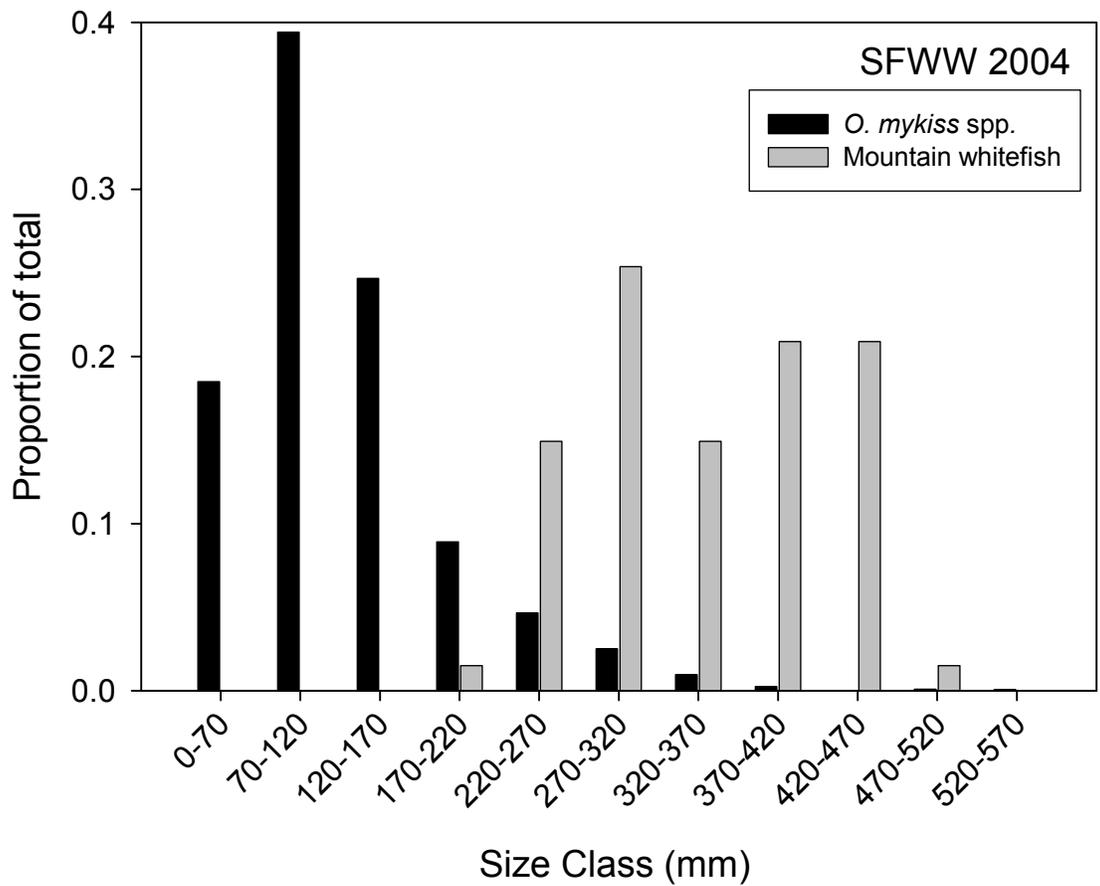


Figure A4. Proportion of *O. mykiss* spp. and mountain whitefish observed in each size class during snorkeling surveys on the South Fork Walla Walla River, summer 2004.

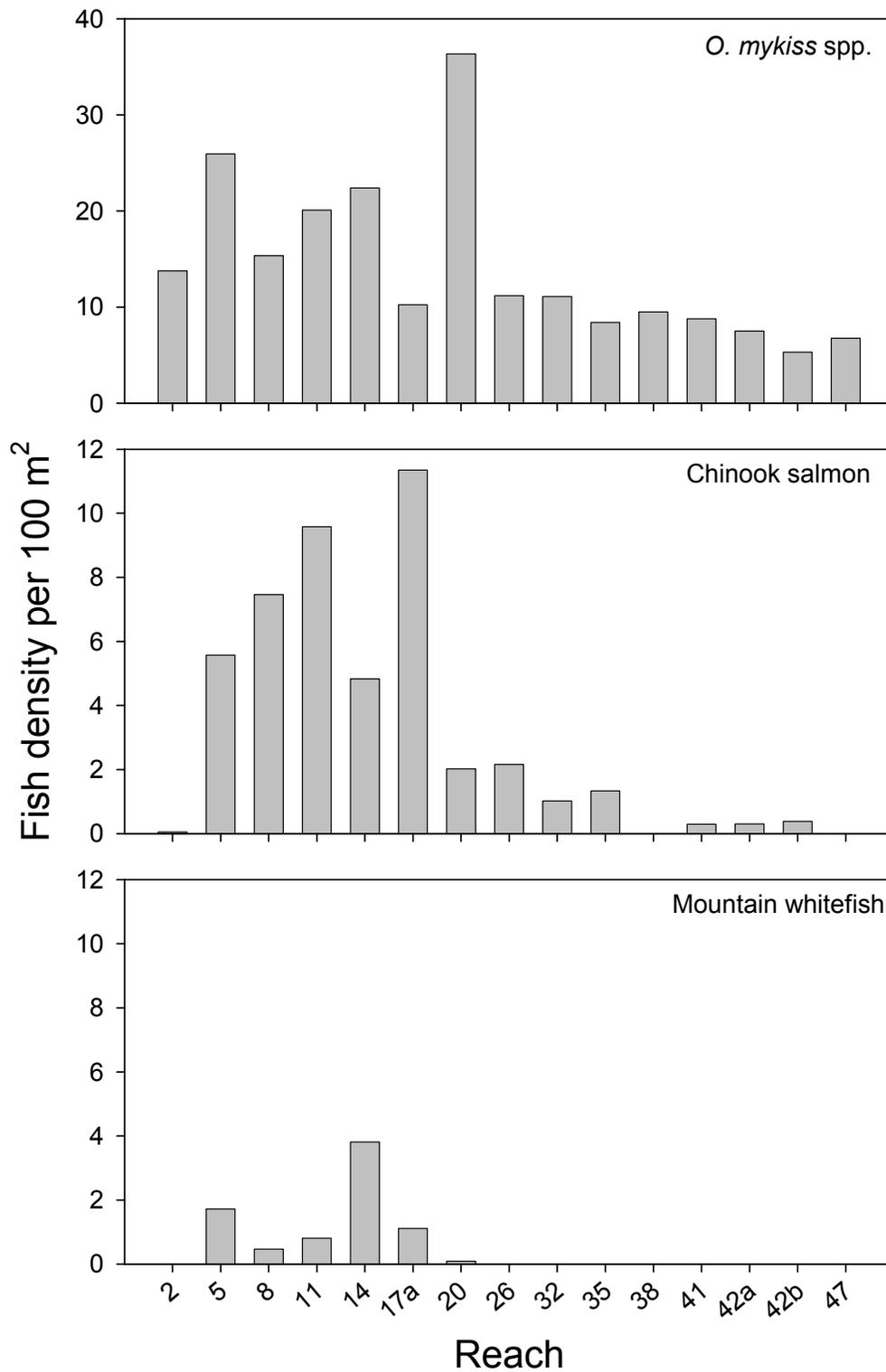


Figure A5. Density of *O. mykiss* spp., Chinook salmon, and mountain whitefish estimated by snorkel counts in various reaches of the North Fork Umatilla River, summer 2004. Note changes in y-axis scales.

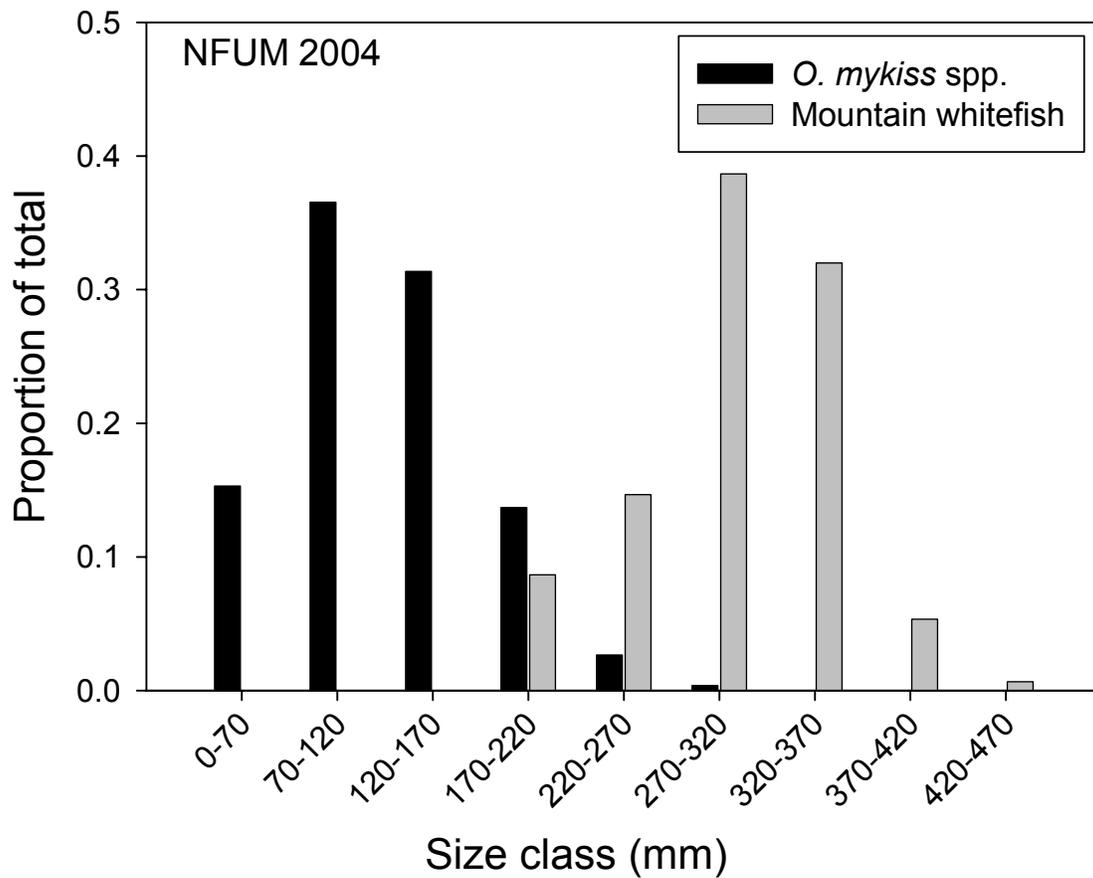


Figure A6. Proportion of *O. mykiss* spp. and mountain whitefish observed in each size class during snorkeling surveys on the North Fork Umatilla River, summer 2004.

APPENDIX 2

Original objectives and tasks specified to meet the overall 5-year project goals.

Objective 1. Comprehensive bull trout population assessment and monitoring.

- Task 1.1** Marking.
- Task 1.2** Recapture.
- Task 1.3** Snorkel surveys for juvenile densities.
- Task 1.4** Adult and egg information, egg-to-parr survival.

Objective 2. Comprehensive stream and riparian habitat assessment and monitoring.

- Task 2.1** Habitat assessment.

Objective 3. Innovative pass-through PIT-tag monitoring system.

- Task 3.1** Tagging, detection, and fish movement.

Objective 4. Data analysis.

- Task 4.1** Analysis of mark-recapture data: population estimates and movement.
- Task 4.2** Analysis of snorkel data: parr density and habitat use.
- Task 4.3** Analysis of adult and egg data: egg-to-parr survival.
- Task 4.4** Analysis of habitat attributes in relation to fish survival and density.

Objective 5. Summarizing available information into a simple population model.

- Task 5.1** Assemble and summarize all existing bull trout population and life-history data for the selected tributaries of the Walla Walla Subbasin.
- Task 5.2** Building the population life-cycle model.

Objective 6. Describe current habitat conditions and land use patterns as they relate to bull trout survival and growth.

- Task 6.1** Summarize and quantify all available habitat data.
- Task 6.2** Exploring the relationship between habitat and bull trout population status indicators.
- Task 6.3** Model calibration and validation.

APPENDIX 3

This proposal was submitted on November 2004 for research to assess and evaluate bull trout in northeastern Oregon to aid in recovery planning. The focus of this research is two-fold: (1) assessment of demography, movement, and population dynamics of bull trout, and (2) evaluation of the importance of salmon in terms of bull trout consumption, growth, and survival.

INTRODUCTION

When species are in decline or listed under conservation status across a large spatial area, as with bull trout (*Salvelinus confluentus*) in the Columbia Basin, estimates of population abundance are critical for understanding the status of the population as well as for recovery planning. In addition, the quantification of key demographic parameters (e.g., survival) is an important component in the process of identifying factors that potentially limit population growth rates overall. However, for many protected species, estimation of population abundance and demographic parameters is extremely difficult due to (1) their protected status, which limits estimation techniques that may be applied legally, (2) low numbers, (3) high variability, (4) the differential effects of environmental stochasticity at low abundance, and (5) the immediate, short-term need for information that typically requires years to collect.

The goal of bull trout recovery planning by the United States Fish and Wildlife Service (USFWS) is to describe courses of actions necessary for the ultimate delisting of this species under the Endangered Species Act, and ensure the long-term persistence of self-sustaining, complex interacting groups of bull trout distributed across the species's native range (Lohr et al. 1999). To meet this overall goal, USFWS has identified several objectives which require the type of information that will be provided by this project: (1) maintain current distribution of bull trout within core areas in all recovery units and restore distribution where needed to encompass the essential elements for bull trout to persist, (2) maintain stable or increasing trends in abundance of bull trout in all recovery units, and (3) restore and maintain suitable habitat conditions for all bull trout life-history stages and strategies.

The USFWS recovery-planning document (Lohr et al. 1999) embraces the idea of core areas. Conserving respective core areas and their habitats within recovery units is intended to preserve genotypic and phenotypic diversity and allow bull trout access to diverse habitats. The continued survival and recovery of individual core area populations is thought to be critical to the persistence of recovery units and their role

in overall recovery of the Columbia River distinct population segment. However, for most threatened populations of bull trout in the Columbia River Basin, little is known about the structure of the population (e.g., migratory versus resident and age), the status (e.g., abundance and trend), or the factors that may limit the population, naturally or anthropogenically (Rieman and McIntyre 1993; Buchanan et al. 1997; USFWS 1998). These limitations are important to overcome, in order for proper status evaluation and for identifying management actions aimed at recovery (Meffe et al. 1997).

In addition to or in combination with habitat degradation and fragmentation, bull trout have likely been affected by changes in the productivity and food availability of their tributary spawning and rearing areas. Given the direct overlap in distribution across much of their historical range, anadromous salmon species (chinook and steelhead) would have played an important role in the productivity and functioning of freshwater bull trout ecosystems before the widespread decline of salmon. There are direct and indirect pathways for marine-derived nitrogen (MDN) to pass from adult anadromous salmon species to bull trout. Direct pathways include bull trout feeding on salmon carcasses, eggs, and juvenile salmon parr and smolts, and indirect pathways include invertebrates scavenging salmon carcasses, which are then eaten by bull trout, and the uptake of nutrients by algae that is grazed on by invertebrates then eaten by bull trout (see review in Schindler et al. 2003). The lack or decline of an abundant and energy-rich food source like eggs and/or juvenile salmon species and fewer aquatic invertebrates would be expected to decrease growth and survival rates of bull trout, which over the long term, could decrease population growth rates and thus their chances of persistence and recovery.

The South Fork Walla Walla River (SFWW) was initially (2002) selected as a comprehensive study area for population assessment monitoring and evaluation in combination with research on habitat needs. This stream was selected by USFWS and Utah State University (USU) due to its potential as a core area for bull trout in the Columbia River Basin and complex and potentially contentious water management issues associated with fish protection. In addition, with an abundance of fish and diverse habitats, the Walla Walla River subbasin offers a unique opportunity to study bull trout population abundance and structure and to identify potentially limiting factors for survival and persistence.

For the study proposed here, we propose to continue the work in the SFWW and expand into the John Day River Basin, allowing a basin-level comparison of demographic variables like growth and survival, and biotic drivers of diet, food availability, and MDN, in a system with low salmon and steelhead abundance

(SFWW) compared to a system with high salmon and steelhead abundance (North Fork John Day River). Bioenergetics modeling based on field observations described here (diet, growth, temperature) will allow estimation of historical contribution of MDN to bull trout growth and consumption as well as spatial expansion of our modeling results to estimate the role of MDN contributions to bull trout populations across their historical range.

STUDY AREA

South Fork Walla Walla River

The Walla Walla River in northeastern Oregon and southeastern Washington is a tributary of the Columbia River that drains an area of 4,553 km² (Walla Walla Subbasin Summary Draft 2001) originating in the Blue Mountains at elevations near 1800 m. The Walla Walla River historically contained a number of anadromous and resident, native salmonid populations including: spring and fall chinook (*Oncorhynchus tshawytscha*), chum (*O. keta*), and coho (*O. kisutch*) salmon, redband trout (*O. mykiss* subpopulation), bull trout, mountain whitefish (*Prosopium williamsoni*), and summer steelhead (*O. mykiss*; the extent of fall Chinook, chum, and coho salmon is not known; Walla Walla Subbasin Summary Draft 2001). Today, steelhead trout represents the only native anadromous salmonid still present in the Walla Walla River system. However, since 2000 there has been annual supplementation of adult chinook in the SFWW by the Confederated Tribes of the Umatilla Indian Reservation (CTUIR). Detailed information on the demography and population size and structure can be found in Budy et al. 2004.

North Fork John Day River

The John Day River in northeastern Oregon is also a tributary of the Columbia River, and drains an area of 12,875 km² (John Day Subbasin Draft Plan 2004) originating at elevations of 2438 m in the Blue Mountains. The North Fork of the John Day River (NFJDA) is the largest tributary and flows westerly for 180 km. The John Day River historically supported large populations of Chinook salmon and steelhead trout and currently demonstrates relatively high (as compared to other subbasins) abundances of both species. Steelhead are in excess of their interim recovery target (1999-2003 NFJDA average = 6,120; spring Chinook salmon are currently estimated to be at about half of their historical abundance for the basin overall; 1999-2001 NFJDA average = 2095). According to the USFWS and local biologists, there are no reliable population estimates for bull trout in the NFJDA; however, both resident and migratory fish and redds are known to be present. Redband trout, rainbow trout,

westslope cutthroat trout (*O. clarki lewisi*), and brook trout (*Salvelinus fontinalis*) also occupy the subbasin with less known about their abundance and distribution.

METHODS

The study site on the SF Walla Walla River spans nearly 21 km in length. The upper boundary was set at the confluence with Reser Creek (Reach 103), and the lower boundary was set above Harris Park Bridge (on county land). The study site was divided into 102 reaches, 200-m each, with 20, 40, and 40 reaches sampled each year for 2002, 2003, and 2004, respectively. Twenty reaches (20% sample rate) will be sampled in 2005.

The proposed study site on the NFJDA spans ~ 80 km from the confluence of Desolation Creek on the NFJDA up to the confluence with Granite Creek (43 km), continuing up the NFJDA to the confluence with Baldy Creek (29 km), totaling 72 km. An initial site will be randomly selected from the list of reaches (each 200 m long), and thereafter every 10th to 15th (sample rate will depend on field time available and logistical limitations) reach will be systematically designated for sampling in 2005. The general location of the bottom of each reach will be located based on UTM coordinates and mapping software, and the closest pool-tail to the coordinates will be set as the reach boundary. Reaches will continue upstream for at least 200 m.

Demography, movement, and population assessment of bull trout

Capture and tagging—Capture activities will begin in early (May or June) summer as soon as water levels permit. Multiple sampling techniques will be used to capture bull trout including angling, electroshocking down to a seine, trap netting, and minnow trapping. All captured bull trout will be weighed (nearest 0.1 g) and measured (nearest mm total length, TL). Bull trout (> 120 mm TL) will be marked with unique PIT tags and T-bar anchor tags (Floy tags), and subsequently recaptured using a combination of passive PIT-tag array antennae (see below) and snorkeling resights. Prior to tagging, bull trout will be anesthetized until they exhibit little response to stimuli. A 23-mm PIT tag will be then placed into a 7-mm surgical incision on the ventral side of the fish, anterior to the pelvic fins. No sutures are required for closure of the incision. In addition, an external T-bar anchor tag, unique to year and stream, will be inserted adjacent to the dorsal fin. Based on local expertise, we expect to sample 80-150 bull trout including resident and migratory forms.

Resighting—Immediately after sampling and marking has been completed, we will conduct daytime bull trout snorkel surveys in the same sample reaches described above. To avoid double-counting fish, snorkeling surveys will be started at the highest reaches working downstream to the bottom of the study site (migrating fish should be moving to the headwaters to spawn). All bull trout and salmon species (tagged and untagged) will be enumerated and placed into 50-mm size classes.

Recapture and passive fish detection—Tagged bull trout can be recaptured anytime after PIT tagging during our sampling in additional reaches or years, as part of other efforts downstream (e.g., seining), and/or with a pass-through PIT-tag detector (see below). All information about each recaptured individual fish is retained electronically; recapture location will provide information about movement. Pass-through PIT-tag detectors will be installed at one or two locations deemed critical for assessing migration and collecting recaptures. Detectors continuously collect information on tagged bull trout; and ideally one detector will be located in the middle of the study area (~ near the North Fork Campground), and one will be located near the bottom of the study area (near the confluence with Desolation Creek). If the lower detector is not possible, we may be able to supplement recapture information based on the trapping and seining being completed by others.

Redd counts—Redd counts will be completed in spawning areas in collaboration with the Oregon Department of Fish and Wildlife and USFWS. Redd counts will begin in August and continue until the end of October. Counts will be weekly in the highest density areas and monthly otherwise. Newly encountered redd sites will be flagged to avoid the potential of double counting redds in subsequent surveys. Redd diameter will be measured and recorded. For comparison to mark-recapture based population assessment, redd counts will be expanded by the number of spawners per redd (see Al-Chokhachy et al., *in revision*).

Tagging information will be used to estimate individual age, growth and survival, population abundance and trend, and life-history type (i.e., migratory versus resident). Detailed analytical methods are available in Budy et al. 2004.

An evaluation of the importance of salmon for bull trout consumption, growth, and survival

Diet analysis—Stomach contents of all handled bull trout will be removed using gastric lavage techniques while fish are anesthetized for tagging as described above. Based on local expertise, we hope to sample 80 – 150 bull trout. These gastric lavage techniques were used on threatened sturgeon by Brosse et al. (2002), who observed

no mortality or injuries due to the lavage process. Stomach contents will be preserved in 95% ethanol and analyzed later in the lab. Stomach contents will be identified to species of prey fish (when possible) using vertebral keys. Aquatic invertebrates will be identified to order with terrestrial invertebrates classified explicitly. Prey fish will be counted and weighed (blot-dry wet weights to nearest 0.001 g); intact prey fish will be measured to the nearest mm (backbone and standard length). Invertebrates will be weighed en masse by classification.

Isotopic analysis of marine derived nitrogen—A small subsample of adult bull trout will be taken for variety of samples (see below) including isotopic analyses of marine derived nitrogen. In addition, we will take non-lethal samples from a subset of bull trout for isotope samples. A non-lethal sample can come from either ~200-500 μ L blood taken from the caudal vein with a 20-gauge hypodermic needle or a small tissue plug (~2 mm diameter x 8 mm long) taken with a biopsy needle. The plasmid portion of the blood will provide short-term changes in stable isotope chemistry, whereas tissue samples will provide longer-term (month-long time scale) isotopic chemistry. Isotope samples and diet samples will be collected on the same fish, and individual information will be retained for direct comparison and validation.

Analyses of stable isotope chemistry from plant and animal tissues will allow us to measure the extent that salmon resources permeate the aquatic food web (e.g., Kline et al. 1990). The ratio of the heavy carbon isotope ($\delta^{13}\text{C}$) offers information on an organism's energy source (e.g., benthic vs. pelagic) while the nitrogen ratio ($\delta^{15}\text{N}$) indicates the relative trophic level (e.g., producer vs. consumer; Peterson and Fry 1987). Anadromous salmon provide an enriched source of these heavier isotopes of carbon and nitrogen relative to other taxa in freshwater ecosystems, and these elements can therefore be traced through the various trophic levels. In particular, $\delta^{15}\text{N}$ has been used to quantify the amount, and hence the relative contribution, of marine-derived nitrogen to the nutrient budgets of plants and animals. Essentially every trophic level in ecosystems with anadromous salmon shows enrichment in marine nitrogen compared to ecosystems without salmon or those upstream from a fish migration barrier (Mathisen et al. 1988, Kline et al. 1990, Bilby et al. 1996). Kline et al. (1998) used stable isotopes to distinguish the contribution of marine-derived resources to the energy budget of Dolly Varden char (*Salvelinus malma*), a species similar to bull trout, and therefore we have every reason to believe it will work here.

To consider indirect pathways of energy transfer in the two subbasins, we will measure the ratio of carbon and nitrogen stable isotopes in the dominant invertebrate organisms in the food web and on diet items where possible. Invertebrate samples will be collected at a subset of our total sample reaches (~ 30 samples across space

and time), always including, but not restricted to, the sites where fish samples were also taken for isotope analyses and paired diets. Dominant taxa in each functional group will be defined as any species that accounts for 10% of the biomass of that functional group (herbivore, collector, or predator). In addition to isotopic analysis of dominant taxa, total invertebrate samples will be analyzed for total identification, counts and biomass at each site. A small subsample of anadromous salmon will be required to calibrate isotope results for the contribution of marine-derived nitrogen; those samples will be taken based on the non-lethal techniques described above. We will use a probabilistic mixing model (Phillips and Gregg 2003) to determine the relative importance of each diet item to the energy budget of bull trout.

All isotope samples will be dried at 60 °C for 24 hours, grinding the resultant tissue into a fine powder, and encapsulating the samples in tin containers. One mg of each dried sample will be analyzed for total nitrogen content and nitrogen isotopic composition using an ANCA-SL elemental analyzer which produces clean gas samples for a 20-20 or GEO-series isotope ratio mass spectrometer (UC Davis).

Gonads and otoliths will also be collected and analyzed on all fish being sacrificed for other indices as discussed above. These additional measurements will provide information on age, growth, fecundity, maturity, and sex ratio.

Survival and growth—Growth and survival will be calculated for individual fish as well as for groups or cohorts of fish, based on mark-recapture data, as described in Budy et al. 2004.

Bioenergetics—Bioenergetics modeling of consumption requires three field-collected inputs: 1) thermal experience, 2) diet composition, and 3) growth. For temperature, temperature loggers will be placed systematically and in geomorphologically important (e.g., stream confluences) throughout the study area (n = 8 - 10). Temperature data and modeling from previous TMDL (total maximum daily load) and FLIR (forward looking infra-red) flight information in both basins will also contribute more detailed spatial information on thermal experience. Inputs for diet and growth will be available based on field estimates described above. We will use bioenergetics-model parameters recently completed for bull trout respiration (Mesa et al. 2004). For evaluations of likely dietary contribution, long time series of Chinook salmon and steelhead abundance are available for both subbasins discussed here as well as for many other subbasins across the historical range of bull trout. The bioenergetics model may also allow us evaluate temporal changes in the stable isotope chemistry (Harvey et al. 2002).

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