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

Annual Progress Report for 2005

by

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21 September 2006

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Suggested citation:
Budy, P., R. Al-Chokhachy, K. Homel, and G.P. Thiede. 2006. Bull trout population assessment in northeastern Oregon: a template for recovery planning. 2005 Annual Progress Report to US Fish and Wildlife Service. UTCFWRU 2006(6):1-102.

ACKNOWLEDGMENTS

Funding for this project 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 of USFWS Abernathy Research Station contributed technical oversight and support. Don Anglin, Darren Gallion, Courtney Newlon, Marshall Barrows, and Paul Sankovich of the USFWS Columbia River Fisheries Program Office 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 help with ongoing modeling efforts. Tim Whitesel provided assistance, guidance and support. Wayne Wurtsbaugh, Mary Conner, and Edward Evans reviewed previous drafts of a manuscript included as a chapter. Dave Crabtree (USFS) and Jackie Dougan (BLM), and Bob Bower (WWBWC) 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 River detector operation was aided by Jim Gasvoda, Paul Sankovich, and USFS personnel. Dwayne Wright provided mule transport. Thanks to our hard working 2005 summer technical support crews: Benjamin Naumann, Timothy Ricks, Anthony Grubb, David McPhie, Tawny Nye, Ryan Rasmussen, Susanna Scott, and Travis Varney.

Scientific Take Permit (OR2005-2412P) was obtained with assistance from Laura Tesler and Mary Hanson of ODFW and Darren Gallion and Paul Sankovich of the USFWS CRFPO. Special Use Permit (WAL0162) was obtained from the USFS with the assistance of Jeff Bloom. Research was conducted under Utah State University IACUC Protocol 1082.

PREFACE

This report consists of three separate chapters. Chapter 1 describes results from the monitoring and evaluation of bull trout populations in several streams in northeastern Oregon, which have been monitored annually since 2003. Chapter 2 describes an investigation of the tradeoff between the costs and different levels of precision associated with increasing sampling efforts, given spatial variability and differences in precision across techniques and is a manuscript that currently in review for the journal *Conservation Biology* as part of Robert Al-Chokhachy's PhD dissertation research, and is currently being revised for publication. Chapter 3 describes an assessment of the movement patterns of subadult bull trout based on both active and passive techniques over the period 2002 – 2005, and is a manuscript in preparation as part of Kristen Homel's MS thesis research. We request that data or information not be reproduced without permission from the authors.

EXECUTIVE SUMMARY

Within the overall framework of conservation and recovery planning for threatened bull trout, we provide 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, we gather information related to population structure (e.g., 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, cost/effort, and limiting factors. Our goal is to provide the data and conservation assessment tools to aide in efforts of the US Fish and Wildlife Service, to determine 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 2005, with plans to continue work through 2006. 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 three large study areas (South Fork Walla Walla, North Fork Umatilla, and North Fork John Day rivers) with a high degree of effort. In addition, we study movement patterns and migratory cues, as well as assess genetic structure within and across populations.

In Chapter 1, we summarize our annual monitoring and evaluation and highlight key results from our mark-recapture program including annual estimates of population abundance, size and growth information, and estimates of condition, survival, and diet. In 2005, we sampled 28 reaches (or 33% of the study area) in the South Fork Walla Walla River (SFWW), 15 reaches (or 40% of the study area) in the North Fork Umatilla River (NFUM), and 30 reaches (or 39% of the study area) in the North Fork John Day River (NFJDA; including Baldy Creek). Bull trout were captured or observed in almost all sampled reaches, and we tagged a total of 417 fish in the SFWW, 149 fish in the NFUM, and 123 fish in the NFJDA. In all three systems, the largest portion of sampled fish were in the 100 - 150 mm size range. In the SFWW and NFUM, we captured more fish in upper reaches, and in the NFJDA, we captured proportionally more fish in Baldy Creek. In both the SFWW and NFJDA, where inter-annual comparisons are possible, we have observed a marked and consistent decrease in fish condition (Fulton's K) across time, a pattern worthy of further exploration. In addition, indices of fish condition were similar and higher in the SFWW and NFJDA as compared to the NFUM, and we caught more large fish in the SFWW as compared to the NFUM or NFJDA. While the population abundance of bull trout (> 220 mm) in the SFWW has remained generally stable over the 4-year time period, small bull trout (120 - 220 mm) appear to be increasing in abundance, albeit with high variance for this size class; the total estimated population size for the SFWW was 9,506 (95% CI = 7,952 - 12,102) in 2005. Population abundance continues to be much lower in the NFUM with a total population size of 1,667 (1,237 – 2,802); abundance estimates for the > 120 mm size category demonstrated high variability, and there was no discernable increasing or decreasing trend for any size class. Our preliminary population estimates for the NFJDA demonstrate a low abundance of bull trout in that system as well (about 1,000 each for both NFJDA and Baldy Creek). And finally, Cormack-Jolly Seber survival estimates for bull trout in the SFWW are generally high, ranging from a low of 40% for the 170 - 220 mm size class to a high of more than 60% for large bull trout (> 270 mm); all of the top models contained either one or both of the individual covariates of movement and condition, with fish that moved below Harris Park demonstrating higher survival as compared to fish that did not move below this point.

In Chapter 2, we present a method for optimizing the efficiency of monitoring designs to detect population change across relevant time intervals. We evaluate the tradeoff between power and sampling effort/cost, using Monte Carlo simulations of commonly-collected field data, both mark-recapture-resight and count data, and estimated the power to detect a declining trend across different time intervals. In addition, we assess the effects of stratification, grouping estimates across different age/stage classes, and the use of shrinkage estimators to reduce sampling variability of mark-resight population estimates; our analysis and results are based largely on spatial variation. For bull trout, precision varied significantly across abundance estimators and indices, with coefficient of variation ranging from 0.40 to 0.16 (similar sampling efforts). Grouping estimates by age/stage class increased the precision of estimates; spatial stratification of sampling units, however, did not increase precision. The use of shrinkage estimators significantly improved precision, but across techniques, detecting a 25% decline in abundance after five years was not possible (power = 0.80), even with very high sampling efforts. Detecting this modest decline (25%) was possible over longer time intervals (15 years), but still required relatively high sampling efforts (power = 0.80; > 15 sample units). When considering the cost associated with each technique, mark-resight techniques require nearly twice the sampling effort as compared to snorkel counts; however, mark-recapture approaches can also provide estimates of key vital rates for the population of interest. Based on this work, we recommend an *a priori* evaluation of the efficiency of monitoring designs using data collected for the species of concern, with consideration of sampling techniques and stratification scenarios, and, if necessary, adaptively adjusting the design to meet management goals. Finally, whenever abundance is monitored, we recommend that sampling variation be removed from the estimate, via a shrinkage (empirical Bayes) estimator, to increase precision and ability to detect trends.

In Chapter 3, we consider the extensive and varied movement patterns of bull trout. While much is known about the migration patterns of adult fish (primarily adfluvial), much less is known about the annual movement patterns of subadult fluvial fish. We tagged 1507 bull trout with passive integrated transponder tags (PIT tags) from 2002 - 2005, and monitored seasonal and diel movement patterns using active recapture methods and two passive PIT-tag detectors. We used this information to 1) evaluate and quantify subadult (120 – 300 mm) migration timing, magnitude, and distance, and 2) evaluate relationships between seasonal and diel movement, and both environmental (i.e., discharge, water temperature, photoperiod, and precipitation) and biological cues (i.e., body size at migration, presence of spawning adults, fish density). Subadults migrated downstream throughout the entire year, peaking in August, and migrated a continuum of distances. In addition, seasonal timing of downstream migration was positively associated with minimum temperature, though minimum temperature was not a strong predictor of migration timing. Finally, most downstream migrations occurred at night. As evidenced by the varied migration timing and distance moved, we suggest that management must address the occupancy of habitat/ migration corridors across a broad temporal and spatial scale.

CHAPTER 1:
Monitoring and evaluation of bull trout populations in the South Fork Walla Walla, North Fork Umatilla, and North Fork John Day rivers, 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 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. 2004).

Despite our growing body of knowledge on bull trout (*see* Budy et al. 2003, 2004, 2005 for these populations), there are still critical gaps in our information that potentially limit our ability to effectively manage bull trout and ensure their continued persistence (Porter and Marmorek 2005). These gaps include basic biological and demographic information for bull trout, detailed population assessment data (e.g., abundance, trend) for all but a few populations, life-history-specific information (e.g., migration timing and contributions of migratory versus resident fish), as well as the relative role of biotic interactions (e.g., competition with non-natives, food availability and declining salmonids). Within the overall framework of conservation and recovery planning for threatened bull trout, this overall 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), and most recently, the role of declining salmon in the parallel decline of bull trout. 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 - 2005 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, such as comparisons of population structure between large and small populations, have required the extension of monitoring and evaluation into the nearby watershed of the North Fork Umatilla River). Further, our most recent work, emphasizing biotic interactions and the decline of salmon, required the addition of a watershed with high salmon abundance (e.g., North Fork John Day River, NFJDA). Thus our complete work includes four years of data to date (2002 - 2005) from one intensively monitored stream and two additional streams where smaller-scale, yet still long-term and continuous, population assessment evaluations are also underway.

STUDY AREAS

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 km^2 (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 (*Oncorhynchus 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, 2005). 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 km^2. 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 ^{o}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 8 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 41 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.

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 km^2 (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 trout 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.

The NFJDA study site spanned 13.5 km from the confluence of NFJDA with Trail Creek (11038402 E 4974314 N) up to the confluence of the NFJDA with Cunningham Creek (UTM 0399908 E 4974038 N). We sampled 22 reaches on this stretch totaling 5.3 km (Figure 1.3). We also sampled sites on Baldy Creek from the confluence of Baldy Creek (UTM 110396075 E 4973524 N) up to (UTM 110395860 E 4969685 N) covering 4.6 km. On Baldy Creek, we sampled 8 reaches encompassing 1.8 km. Reaches continued upstream for at least 200 m. Currently, there are no passive PIT-tag detector stations in the NFJDA.

METHODS

<u>Fish Sampling</u>

Capture.—We used multiple sampling techniques 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.—In all study streams, 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 in-stream PIT-tag antennae (hereafter detector; *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 small 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 23 reaches (mean reach length = 238 m) of the SFWW, 15 reaches (mean = 212 m) of the NFUM, 22 reaches in the NFJDA (mean = 240 m), and 8 reaches (mean = 226 m) in Baldy Creek in 2005. 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), O. *mykiss* spp., and mountain whitefish were enumerated and placed into 50-mm size classes, and all 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.

Recapture.—Tagged bull trout were recaptured one month after PIT tagging; recaptures will continue for the duration of the study. 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 detectors were 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 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 lone NFUM detector (UM1) is located on US Forest Service land under a road bridge (UTM coordinates: 110407659 E, 5064089 N) near the confluence with the Umatilla River. The detector has been collecting data since autumn 2004.

<u>Growth</u>

Growth information was obtained from SFWW bull trout tagged in 2003 - 2004 and recaptured during 2004 - 2005. Length and weight gains were determined between initial tagging and subsequent capture events. These length and weight gains were evaluated based on annual growth per size class. Growth estimates were not made for NFUM bull trout due to low recapture rates.

<u>Population Estimates</u>

We used snorkeling and tagging data to parameterize mark-resight population estimates using a Lincoln-Petersen bias-adjusted estimator (Chapman 1951), and estimated the overall population size for three size groupings of bull trout: > 120 mm, > 220 mm, and > 370 mm. We estimated the standardized population sizes for each reach using tagging and snorkeling data for each individual reach, calculated the average number of bull trout per 200 m across reaches, and multiplied this average by the total number of reaches in the site. To standardize the number of bull trout per 200 m for each reach, we divided each reach estimate by the actual reach length and multiplied this estimate by 200.

<u>Survival</u>

Survival estimates were calculated using a Cormack-Jolly-Seber model from mark-recapture data collected from 2002 - 2005. This is an open mark-recapture model, which incorporates the number of marked and recaptured fish in different time intervals. We used eight encounter occasions for the 2002 - 2005 data; these occasions corresponded to either the summer field season, where active recaptures and passive PIT-tag detector recaptures took place, and the interval between the summer field seasons, where only passive-detector recaptures occurred.

The recapture intervals were analytically adjusted according to the length of each period. We incorporated average growth rates into the analyses, which we calculated from individual recapture data, to create a stage-based model with four life stages representing 120 -170, 170 - 220, 220 - 270, and > 270 mm size classes. Survival estimates and recapture probabilities were calculated using Program MARK software.

<u>Diet Analysis</u>

Sampling design.—Within the SFWW and the NFJDA, we sampled two locations (upper and lower) to quantify spatial differences in available prey in each system. In the SFWW, the lower site was located just upstream of the confluence with Bear Creek (UTM 110414389 E 5077168 N) and the upper site samples were collected just downstream of the confluence with Reser Creek (UTM 110432618 E 5080344 N). In the NFJDA, the lower site was located approximately 3.5 km upstream of the confluence with Trail Creek (UTM 110391942 E 4973634 N) and the upper site was located upstream of the confluence of Baldy Creek (UTM 110396097 E 49773596 N). To quantify temporal differences in available prey, samples were collected in these sites during two separate times of the summer (mid-June and mid-August). Further, within the SFWW and NFJDA, we evaluated bull trout dietary preferences to better understand bull trout trophic status and interactions (i.e., predation) within these communities.

Stomach content collection and analysis.—During each sampling occasion at each site, we captured eight adult bull trout, and used gastric lavage techniques to collect stomach contents. All stomach contents were preserved in 95% ethanol for further prey identification in our laboratory. We identified aquatic macroinvertebrates found in bull trout stomachs to the genus level (BLM Bug Lab, Utah State University), and all fish prey to the species level. Prey fish were counted and weighed (blot-dry wet weights to nearest 0.001 g), while macroinvertebrate prey were weighed *en masse* by classification. Intact prey fish were measured to the nearest mm (backbone and standard length). Unidentified fish prey were apportioned into identified prey categories based on a weighted average of identified fish prey.

Isotope collection and analysis.—In conjunction with stomach content collection, we used the same eight adult bull trout for isotope analysis. We anesthetized each fish, removed a 5-mm tissue sample (dermal plug) from the muscular tissue posterior to the dorsal fin, and collected at least 0.3 mL samples of blood from the caudal vein (22-gauge, 1.5 inch syringe). We placed all tissue and blood samples on ice in the field and subsequently froze all samples. Prior to fish sampling at each site, we sampled periphyton and aquatic macroinvertebrates. Starting at the bottom of each site, we established a transect at one third the distance from the bottom of each of the first four channel units (pool-riffle). Along each transect, four rocks, equidistantly sampled were collected, scrubbed with a nylon brush into a reservoir to collect periphyton, and filtered through a 25-mm GF/F filter. We collected available macroinvertebrates with two kick-net samples along each transect and placed each sample into storage jars filled with water. In the field, we placed both filter samples and macroinvertebrate samples on ice, and subsequently froze all samples for isotopic analyses. In the laboratory, macroinvertebrates for isotope analysis were identified into the following functional feeding groups: filter-feeders, herbivores, and carnivores. For isotopic analyses, we dried all bull trout tissue, bull trout blood, periphyton filter, and macroinvertebrate samples in an oven at 70°C for 48 hours. Subsequent to drying, we encapsulated each sample in 8.5-mm tin capsules. All isotopic samples were analyzed at the University of California, Davis Stable Isotope Facility.

<u>Temperature</u>

We measured in-stream temperature every 90 minutes using temperature loggers at five sites in the SFWW (from Reser Creek to Harris Park bridge) and two sites in the NFUM (Coyote Creek and Campground). We summarized temperature as daily maximum, average and minimum for ease of assessment.

RESULTS and DISCUSSION

<u>Fish Sampling</u>

All captured bull trout were weighed and measured, and a separate length-weight relationship was calculated for each stream in each year based on all measured bull trout (Figures 1.4, 1.5, and 1.6).

South Fork Walla Walla River

We sampled 28 reaches during the 2005 field season, which accounted for approximately 33% of the study site. Over the summer, a total of 644 bull trout were captured of which, 417 were tagged, with the number tagged varying by sample reach (1 – 52 per reach; Figure 1.7). In 2005, as in years since 2003, most bull trout were tagged upstream of Burnt Cabin Creek (Figure 1.7). In 2005, the smallest bull trout captured was 78 mm (4.3 g) and the largest bull trout caught was 720 mm. Length-frequency distributions of captured bull trout in the SFWW have varied little from 2002 through 2005, with most captured fish in the 100 – 150 mm size range (Figure 1.8). More large (> 500 mm) bull trout were captured in the SFWW compared to both the NFUM (Figure 1.9) and the NFJDA system (Figure 1.10).

Condition.—Condition (Fulton's K) of bull trout captured in the SFWW varied by size class and year; in general, condition was lowest for small (< 120 mm) bull trout (4-year mean = 0.87) and highest for larger bull trout (4-year mean = 0.92; Figure 1.11). From 2002 to 2005, average condition has declined greatly for juvenile (< 120 mm) bull trout (Figure 1.11). When all size classes are combined, it appears that average condition has declined steadily in the SFWW from 2002 (K ± 1 SE = 0.93 ± 0.011) through 2005 (0.86 ± 0.005; Figure 1.12). However, bull trout condition over the course of the study in the SFWW was generally higher than in both the NFUM and NFDJA (Figure 1.12). Average condition for these populations was lower than that exhibited by Metolius River (Deschutes River basin, Oregon) adfluvial bull trout (mean K_{TL} range: 1.02 – 1.65; Thiesfeld et al. 1999) and bull trout from southeast Washington (K_{FL} range: 1.00 – 1.23; Underwood et al. 1995).

Snorkel surveys.—Snorkeling surveys were performed in 23 reaches in the SFWW in 2005. As with numbers tagged, more bull trout were observed in the study reaches upstream of Burnt Cabin Creek (Figure 1.13). Observations were likely biased toward fish > 120 mm due to the cryptic nature of small fishes (Figure 1.14). In 2005, bull trout observed in the SFWW ranged from 50 to 570 mm, similar to past surveys (Figure 1.14; Budy et al. 2004, 2005).

North Fork Umatilla River

We sampled 15 reaches in 2005 which accounted for 40% of the study site. Bull trout were captured or observed in all sampled reaches. Over the summer, a total of 223 bull trout were captured and 149 were tagged, with the number tagged varying by sample reach (1 to 35 per reach; Figure 1.15). Most bull trout captured in the NFUM (2003 – 2005) were also in the 100 – 150 mm size range, and the largest bull trout captured in 2005 was a 510 mm (1.07 kg) tagged in Reach 2, while the smallest bull trout captured was 94 mm (7.3 g; Figure 1.1.5).

Condition.—Across years, condition (Fulton's K) of bull trout in the NFUM varied little by size class, ranging from 0.83 (± 1 SE = 0.013 for < 120 mm fish in 2004) to 1.07 (± 0.043 for > 370 mm fish in 2004; Figure 1.16). As in the SFWW, when we combined all size classes, average condition has decreased greatly from 2003 to 2005, ranging from 0.92 (± 0.01) in 2003 to 0.83 (± 0.004) in 2005 (Figure 1.12). In 2005, average condition of all fish combined was lower than in both the SFWW and NFJDA (Figure 1.12).

Snorkel surveys.—Snorkeling surveys were performed in 15 reaches, and bull trout were observed in most sampled reaches, except in Reach 8 (Figure 1.17). As with the number tagged, most bull trout (93% of total) were observed in stream reaches upstream of Coyote Creek (Figure 1.17). As in other streams, observations were biased toward fish > 120 mm (Figure 1.14). A very similar size distribution of bull trout was observed in NFUM and Baldy Creek (Figure 1.14). Observed numbers of bull trout were 3-times less than in SFWW (Figure 1.14).

North Fork John Day River

We sampled 22 reaches in the NFJDA in 2005 plus 8 reaches on Baldy Creek. Bull trout were captured or observed in most sampled reaches. This was our first year tagging bull trout in the NFJDA including Baldy Creek. Over the summer, a total of 33 bull trout were tagged in the NFJDA and 90 were tagged in Baldy Creek, with the number tagged varying by sample reach (1 to 20 per reach; Figure 1.18). More bull trout were captured in Baldy Creek, but fish in Baldy Creek were generally smaller than bull trout captured in the NFJDA (Figure 1.10). The largest fish captured in the NFJDA (Reach 4) was 461 mm long (weight = 967 g), while the smallest was 98 mm (8.3 g). The largest fish captured in Baldy Creek was 209 mm (71.6 g), and the smallest was 71 mm (3.1 g). As in other streams, the greatest proportion of bull trout captured or observed were in the 100 - 150 mm size range (Figure 1.10).

Condition.—In both the NFJDA and Baldy Creek, average condition was the same for juvenile (< 120 mm) bull trout (K = 0.82) and 120 – 370 mm bull trout (K = 0.86; Figure 1.19). Condition was substantially higher for larger bull trout (> 370 mm; K = 0.99, based on one large fish; Figure 1.19). In 2005, average condition of bull trout (all sizes combined) from NFJDA and Baldy Creek was similar to condition of bull trout from SFWW, and higher than condition of bull trout from NFUM (Figure 1.12).

Snorkel surveys.—We performed snorkel surveys in 22 reaches in the NFJDA plus 8 reaches on Baldy Creek in 2005. Bull trout were observed in all sampled reaches in Baldy Creek, but were absent from five sampled reaches in the NFJDA (Figure 1.20). Bull trout observed ranged from 50 – 270 mm TL (Figure 1.14). A similar size distribution of bull trout was observed as in the SFWW, although over 3-times fewer fish were seen in NFJDA and Baldy Creek (Figure 1.14). We also calculated densities of bull trout based on snorkel counts and sampled stream area; densities ranged from 0 – 0.65 fish per 100 m^2 in the NFJDA, and ranged from 0.5 – 2.6 fish per 100 m^2 in

Baldy Creek (Figure 1.21). Snorkel-count densities of bull trout were similar to densities of brook trout in the NFDJA (Figure A5), and were 16- and 32-times lower than *O. mykiss* and juvenile Chinook salmon densities, respectively (Figure A5). Conversely, bull trout densities in Baldy Creek were up to 2.5-times higher than brook trout densities, and only slightly lower than *O. mykiss* densities overall (Figure A5).

Growth

Tagged fish.—Average annual growth of tagged bull trout in the SFWW varied by size class; the > 370 mm size class grew slightly more in length compared to the 120 – 220 mm and 220 – 370 mm size classes; however, these results were not significant due to high variance associated with a small sample size (Figure 1.22). In terms of body mass, small subadult (120 – 220 mm) and medium-sized adult (220 – 370 mm) bull trout gained significantly more weight per annum compared to large adults (> 370 mm; Figure 1.22). Subadult fish gained 60.0 g (± 2 SE = 9.03) while large adults gained only 17.6 g (± 3.74) annually.

Population Estimates

South Fork Walla Walla River.—Estimated abundance of bull trout in the SFWW depends greatly on size grouping. Over a 4-year period, the average abundance of bull trout > 120 mm has ranged from 7,287 (95% CI = 6,243 – 8,895) in 2002 up to 9,506 (95% CI = 7,952 – 12,102) in 2005 (Figure 1.23). The average abundance of bull trout > 220 mm has ranged from 2,700 in 2002 down to 1,800 in 2005, and the average abundance of bull trout > 370 mm (supposedly the migratory component) has ranged from 1,500 in 2002 down to about 900 in 2004 (Figure 1.23). Whereas the population abundance of bull trout (> 220 mm) in the SFWW has remained generally stable over the 4-year time period, small bull trout (120 - 220 mm) appear to be increasing in abundance, albeit with high variance for this size class (Figure 1.23).

North Fork Umatilla River.—Similar to population abundance trends observed in the SFWW, estimated abundance of bull trout in the NFUM also depends greatly on size grouping. Over a 3-year period, the average abundance of bull trout > 120 mm has ranged from a high of 2,434 (95% CI = 1,705 – 5,045) in 2004 to a low of 1,667 (95% CI = 1,237 – 2,802) in 2005 (Figure 1.24). The average abundance of bull trout > 220 mm has ranged from 343 in 2004 down to 61 in 2005, and the average abundance of bull trout > 370 mm (supposedly the migratory component) has ranged from 23 in 2003 down to about 5 in 2005 (Figure 1.24). Overall, abundance estimates for the > 120 mm size category demonstrated high variability, and there was no discernable increasing or decreasing trend for any size class. As in the SFWW, the population abundance estimates for > 220 mm and > 370 mm bull trout are analogous; however, when considering all bull trout > 120 mm the population abundance estimate becomes 5- to 200-times greater.

North Fork John Day River.—In 2005, we initiated our first assessment of bull trout abundance in the NFJDA and Baldy Creek. Considering all bull trout > 120 mm, average abundance estimates were comparable in NFJDA (95% CI range = 390 - 2052) and Baldy Creek (range = 545 - 998; Figure 1.25). Although these estimates were similar to estimates in the NFUM, they were considerably lower (up to 7000 fewer bull trout) than population estimates in the SFWW during the same year (Figure 1.25).

Survival

Survival estimates for the four life stages of bull trout in the SFWW ranged from 40% (± 1 SE = 12) for 170 – 220 mm bull trout up to 64% (± 14) for > 270 mm bull trout (Figure 1.26). These estimates are similar to estimates reported by Budy et al. (2004) in the SFWW over the period 2002 – 2004.

All of our top CJS survival models contained either one or both of the individual covariates (movement and condition) and time for estimates of survival (Al-Chokhachy 2006). Although our most global age model contained six size classes, our top models included four separate size classes for bull trout survival estimates (> 80% of the AICc weights; 120-170, 170-220, 220-270, and >270 mm). Although present in the top models, we found condition (at time of tagging) had little influence on bull trout survival (Beta = - 0.058, SE = 0.66). In contrast, movement had a substantial positive effect on all top models, when modeled as an additive term (Beta = 0.192, SE = 0.077) as bull trout that exhibited movement patterns below the Harris Park detector had higher survival rates than fish that did not migrate. Across years, time (year) had no effect on survival estimates (Beta = 4.66, SE = 349.2). The low emigration rates calculated from movement data had little effect on model-averaged survival estimates; however, these emigration rates may be biased as a result of the incomplete detection efficiencies at the Harris Park PIT-tag detector, which were estimated at approximately 50% (Al-Chokhachy 2006)

Diet Analysis

Using gastric lavage techniques and dissected stomachs of sacrificed fish, we quantified diet information from 26 bull trout from the SFWW and 12 bull trout from the NFJDA in 2005. The primary prey items in both streams in June and August were aquatic macroinvertebrates, which represented 55 – 90% of diets (Figure 1.27). Aquatic macroinvertabrates included chironomids, plecopterans, dipterans, trichopterans, ephemperopterans, and coleopterans. Fish prey included sculpin, *O. mykiss*, and salmonids, and represented 0 – 40 % of diets depending on stream and time period (Figure 1.27). Rare prey included terrestrial insects, fish eggs, worms, and gastropods. In both the upper reaches (near Reser Creek) and lower reaches (near Bear Creek) of the SFWW in June 2005, aquatic invertebrates composed 82 – 88% of diets; however, prey fish (primarily *O. mykiss*) increased in importance (or likely availability) in August, representing 38 – 44 % of diets (Figure 1.28). Due to a low number of bull trout captured in lower section of the NFJDA, we were not able to make an upper reach versus lower reach comparison.

We also compared the diet of bull trout captured in the SFWW in 2003 (n = 16) to current diets. In 2003, fish represented 40% (including cannibalism) of summer combined diets, but represented only 23% of diets in 2005 (Figure 1.29). This corresponds to a marked decrease in *O. mykiss* densities observed in snorkel surveys from 2003 to 2005 (Figure A2) and a decrease in the number of bull trout < 100 mm captured in the SFWW from 2003 to 2005 (Figure 1.8).

Using stable isotope data, we can infer the trophic position (^{15}N) of different organisms as well as assess the extent of dietary overlap among organisms within trophic levels (^{15}C). In both the SFWW and NFJDA in 2005, isotope composition analysis showed the clear trophic distinction between periphyton, macroinvertebrates, and fish (Figure 1.30). In the SFWW, there was little trophic-level distinction between fish, and a high level of dietary overlap in ^{13}C for sculpin, *O. mykiss*, Chinook salmon, and bull trout (Figure 1.30). Similarly, in the NFJDA, there appeared to be equally low trophic-level distinction; however, there was a lower level of dietary overlap (regarding ^{13}C signatures), especially for bull trout and *O. mykiss* (Figure 1.30).

<u>Temperature</u>

We measured temperature using temperature loggers at five sites in the SFWW and two sites in the NFUM from June 2004 to June 2005. In the SFWW, daily temperatures varied little across the year in upper reaches (three sites above Skiphorton Creek; annual range = 2.3 – 9.5 ºC), but varied greatly near Bear Creek and at Harris Park Bridge (annual range = 0.8 – 15.9 ºC; Figure 1.31). In the NFUM from June 2004 to June 2005, temperature ranged from 0.4 – 15.4 ºC near the Coyote Creek sample site, and ranged from 4.6 – 15.1 ºC at the NFUM Campground (Figure 1.32).

LITERATURE CITED

Al-Chokhachy, R. 2006. An assessment of the population dynamics and limiting factors of a fluvial bull trout population. PhD Dissertation, Utah State University, Logan, Utah

Buchanan, D.V., M.L. Hanson, and R.M. Hooten. 1997. Status of Oregon's bull trout. Oregon Department of Fish and Wildlife. October 1997. Portland, Oregon.

Budy, P., R. Al-Chokhachy, and G.P. Thiede. 2003. Bull trout population assessment and life-history characteristics in association with habitat quality and land use in the Walla Walla River Basin: a template for recovery planning. 2002 Annual Progress Report to US Fish and Wildlife Service, Vancouver, Washington. 41 pages.

Budy, P., R. Al-Chokhachy, and G.P. Thiede. 2004. Bull trout population assessment and life-history characteristics in association with habitat quality and land use: a template for recovery planning. 2003 Annual Progress Report to US Fish and Wildlife Service, Vancouver, Washington. 70 pages.

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.

Cavender, T.M. 1978. Taxonomy and distribution of the bull trout, *Salvelinus confluentus* from the American Northwest. California Fish and Game 64:139-174.

Chapman, D.G. 1951. Some properties of the hypergeometric distribution with applications to zoological censuses. University of California Publications in Statistics 1:131-160.

FERC Project 2145 Draft. 2002. Movement of bull trout within the Mid-Columbia and its tributaries 2001-2002. Bioanalysts.

Fraley, J.J., and B.B. Shepard. 1989. Life history, ecology, and population status of migratory bull trout (*Salvelinus confluentus*) in the Flathead Lake and river system, Montana. Northwest Science 63:163-143.

Goetz, F. 1991. Bull trout life history and habitat study. U.S. Department of Agriculture, Forest Service, Deschutes National Forest, Bend, Oregon.

Kershner, J.L., C.M. Bischoff, and D.L. Horan. 1997. Population, habitat, and genetic characteristics of Colorado River Cutthroat trout in wilderness and non-wilderness stream sections in the Uinta Mountains of Utah and Wyoming. North American Journal of Fisheries Management 17:1134-1143.

Leary, R.F., F.W. Allendorf, and S.H. Forbes. 1993. Conservation genetics of bull trout in the Columbia and Klamath River drainages. Conservation Biology 7:856-865.

Lohr, S., T. Cummings, W. Fredenberg, and S. Duke. 1999. Listing and recovery planning for bull trout. U.S. Fish and Wildlife Service publication. Boise, Idaho.

Mahoney, B. 2001. Walla Walla basin summer steelhead and bull trout radio telemetry project: 2001 Annual Report. Prepared by the Confederated Tribes of the Umatilla Indian Reservation for the Bonneville Power Administration. Portland, Oregon.

Mahoney, B. 2002. Walla Walla basin summer steelhead and bull trout radio telemetry project: 2002 Annual Report. Prepared by the Confederated Tribes of the Umatilla Indian Reservation for the Bonneville Power Administration. Portland, Oregon.

ODFW (Oregon Department of Fish and Wildlife). 1995. Bull trout spawning ground surveys Umatilla and Walla Walla basins. September 1, 1995 to October 31, 1995. Progress Report (Preliminary). Umatilla Fishery District, Pendleton, Oregon.

ODFW (Oregon Department of Fish and Wildlife). 1996. Bull trout spawning ground surveys. Umatilla and Walla Walla basins. September 1, 1996 to October 31, 1996. Progress Report. Umatilla Fish District, Pendleton, Oregon.

Porter, M., and D.R. Marmorek. 2005. Bull Trout Recovery Monitoring and Evaluation Technical Working Group (RMEG) Workshop 3 (Dec. 16-17, 2004)- USFS Regional Office, 333 SW First Ave., Portland, Oregon – Workshop Report, Report prepared by ESSA Technologies Limited., Vancouver, BC for the US Fish and Wildlife Service, Vancouver, Washington. 113 pages.

Ratliff, D.E., and P.J. Howell. 1992. The status of bull trout populations in Oregon. *In* Proceedings of the Gearhart Mountain Bull Trout Workshop. Oregon Chapter of the American Fisheries Society.

Rieman, B.E., and J.D. McIntyre. 1993. Demographic and habitat requirements for conservation of bull trout. USDA General Technical Report INT-302.

Rieman, B.E., and J.D. McIntyre. 1995. Occurrence of bull trout in naturally fragmented habitat patches of varied size. Transactions of the American Fisheries Society 124:285-296.

Rieman, B.E., D.C. Lee, and R.F. Thurow. 1997. Distribution, status, and the likely future trends of bull trout within the Columbia River and Klamath River basins. North American Journal of Fisheries Management 17:1111-1125.

Soule, M.E. 1987. Viable populations for conservation. Cambridge University Press.

Thiesfeld, S.L., J. C. Kern, A.R. Dale, M.W. Chilicote, and M.A. Buckman. 1999. Lake Billy Chinook sockeye salmon and kokanee research study 1996-1998. Contract Completion Report. FERC Number 2030. 153 pages.

Underwood, K.D., S.W. Martin, M.L. Schuck, and A.T. Scholz. 1995. Investigations of bull trout (*Salvelinus confluentus*), steelhead trout (*Oncorhynchus mykiss*), and spring chinook salmon (*O. tshawytscha*) interactions in southeast Washington streams. BPA report. Project Number 90-053. Contract Number DE-B179-91BP17758. 173 pages.

USFWS (US Fish and Wildlife Service). 1998. Endangered and threatened wildlife and plants; determination of Threatened Status for the Klamath River and Columbia River Distinct Population segments of bull trout.

Walla Walla Subbasin Summary *Draft*. 2001. Prepared for the Northwest Power Planning Council.

Whitesel, T.A. and 7 coauthors. 2004. Bull trout recovery planning: a review of the science associated with population structure and size. Science Team Report # 2004-01, U.S. Fish and Wildlife Service, Regional Office, Portland, Oregon. Website: *http://columbiariver.fws.gov/scienceteam.htm#product*

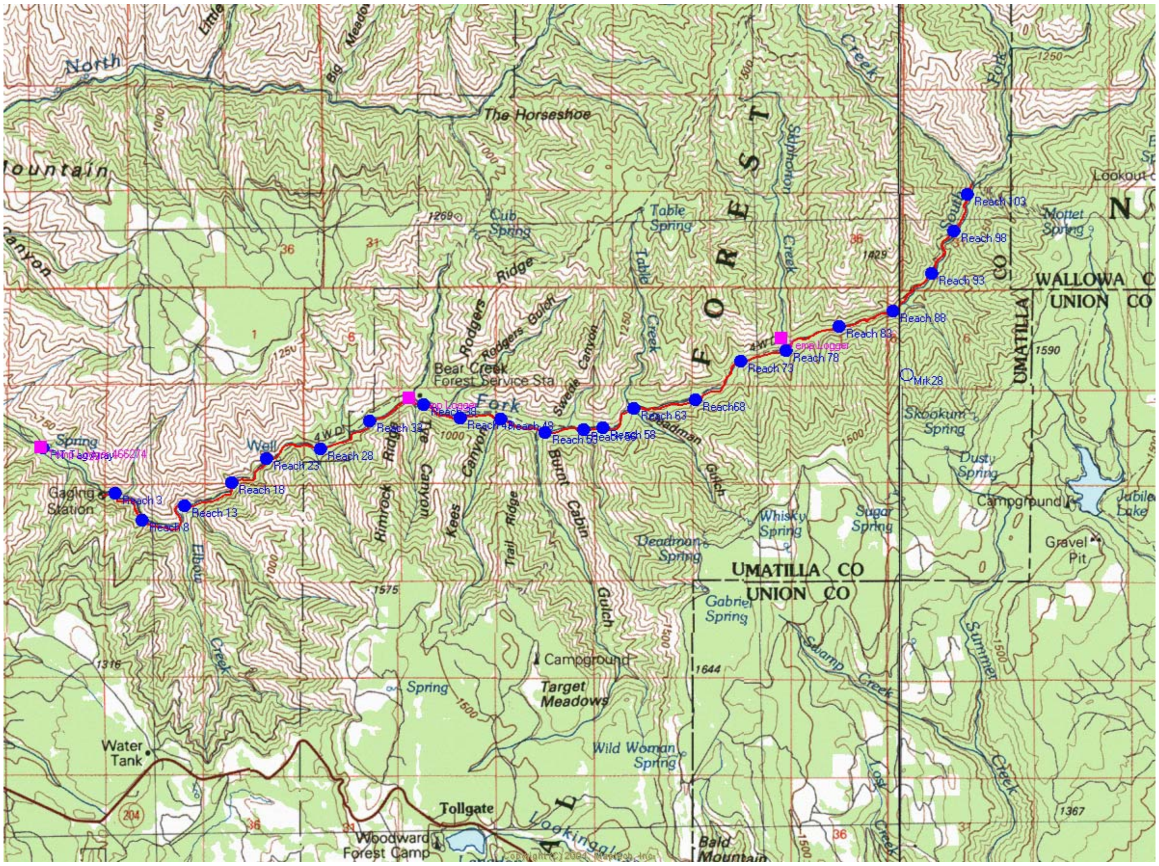

Figure 1.1. Map of the South Fork Walla Walla River showing original 22 study reaches (dots).

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

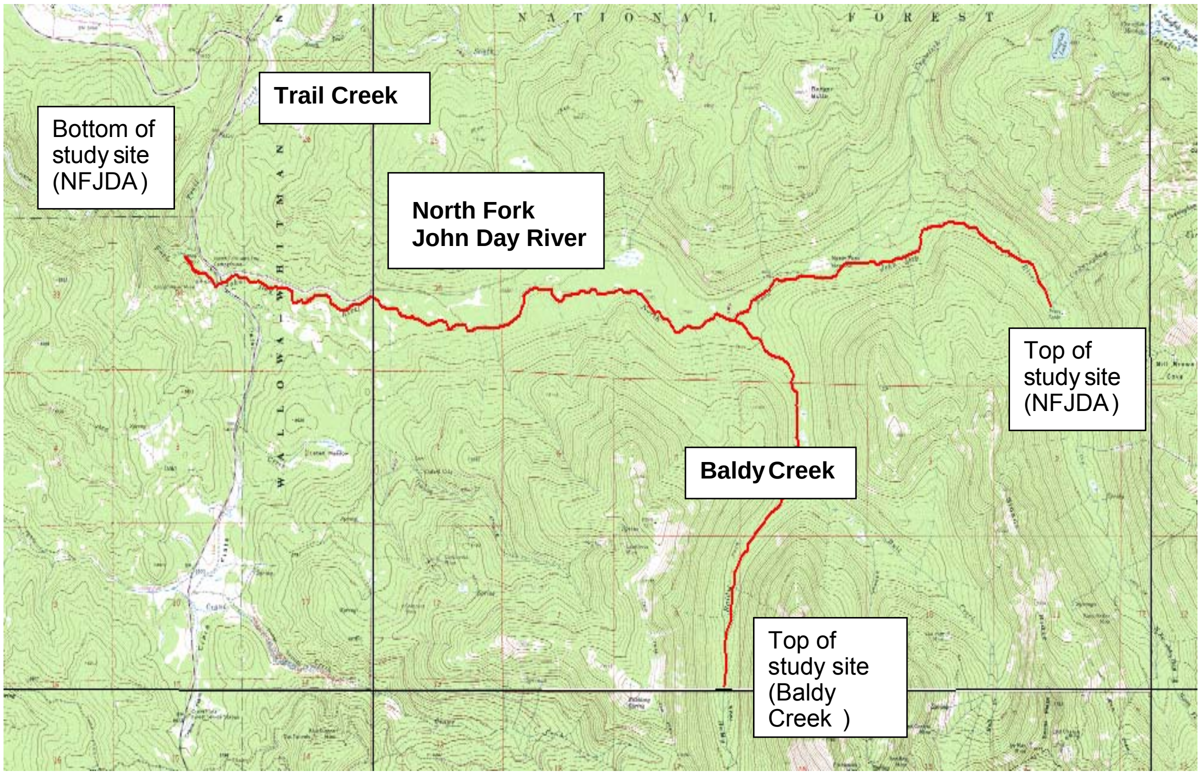

Figure 1.3. Map of the North Fork John Day River (NFJDA) including Baldy Creek.

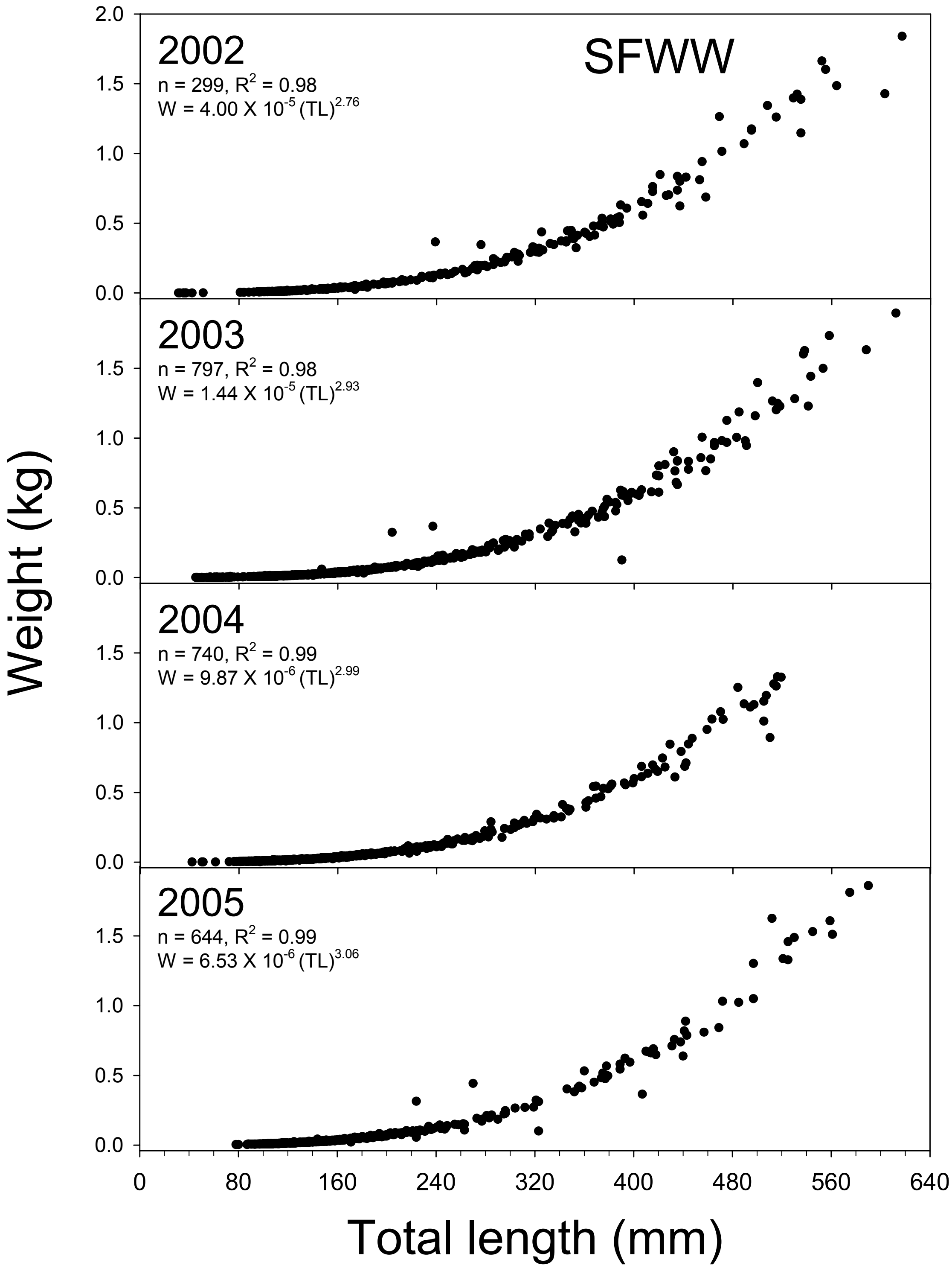

Figure 1.4. Length-weight relationship for all bull trout captured and handled in the South Fork Walla Walla River, 2002 - 2005. Regression equations and sample sizes are given.

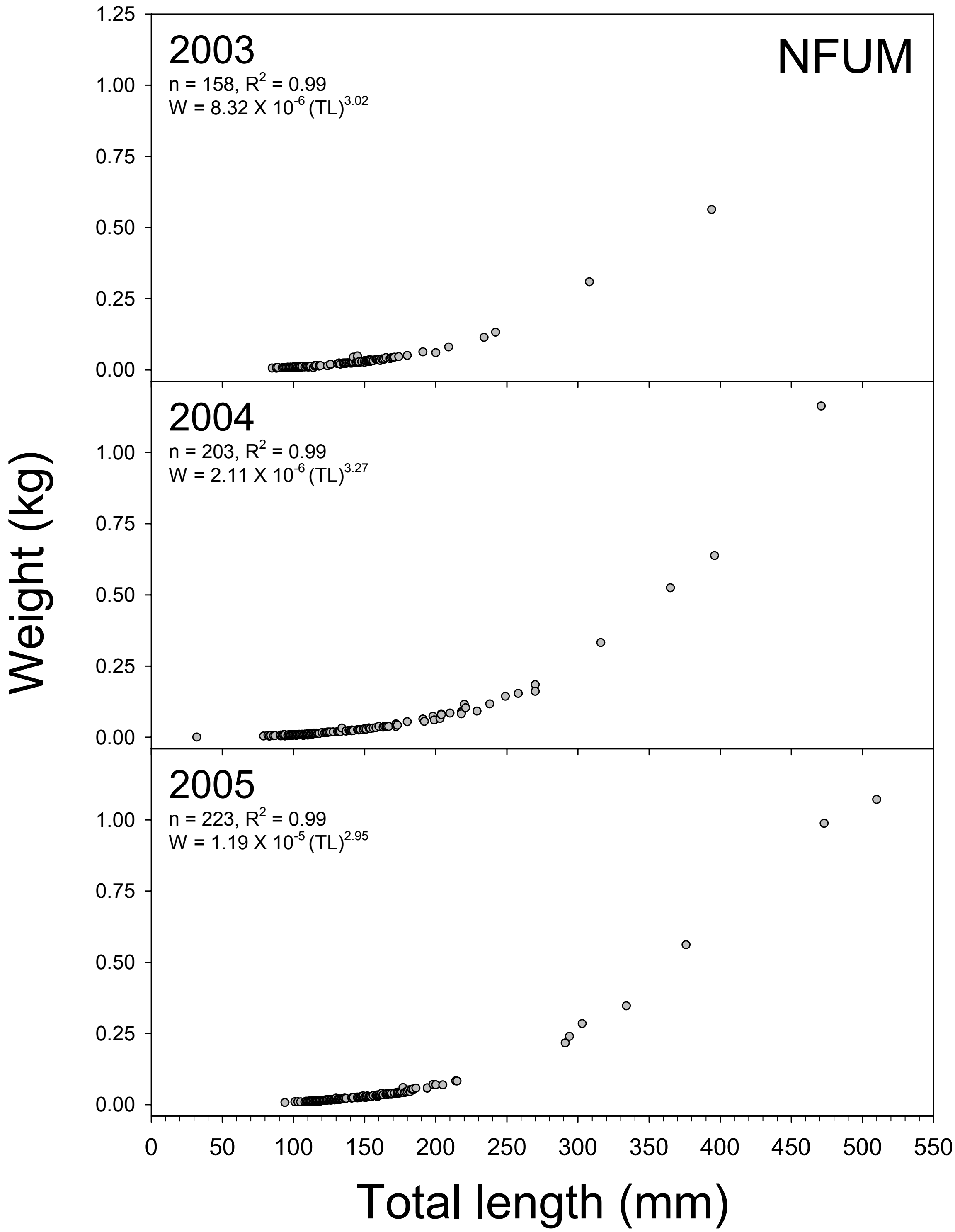

Figure 1.5. Length-weight relationship for bull trout captured and handled in the North Fork Umatilla River, 2003 - 2005. Regression equations and sample sizes are given.

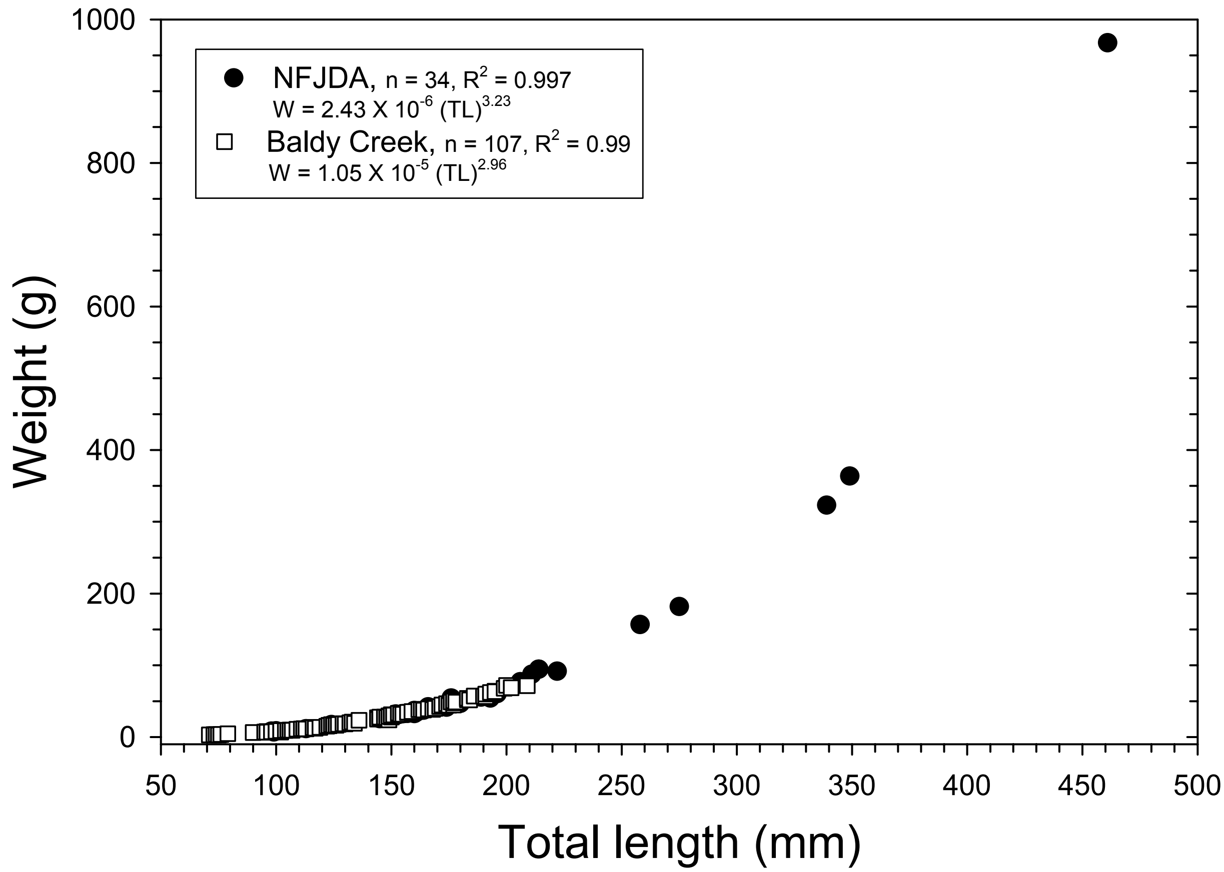

Figure 1.6. Length-weight relationship for bull trout captured and handled in the North Fork John Day River (black circles) and Baldy Creek (open squares), 2005. Regression equations and sample sizes are given.

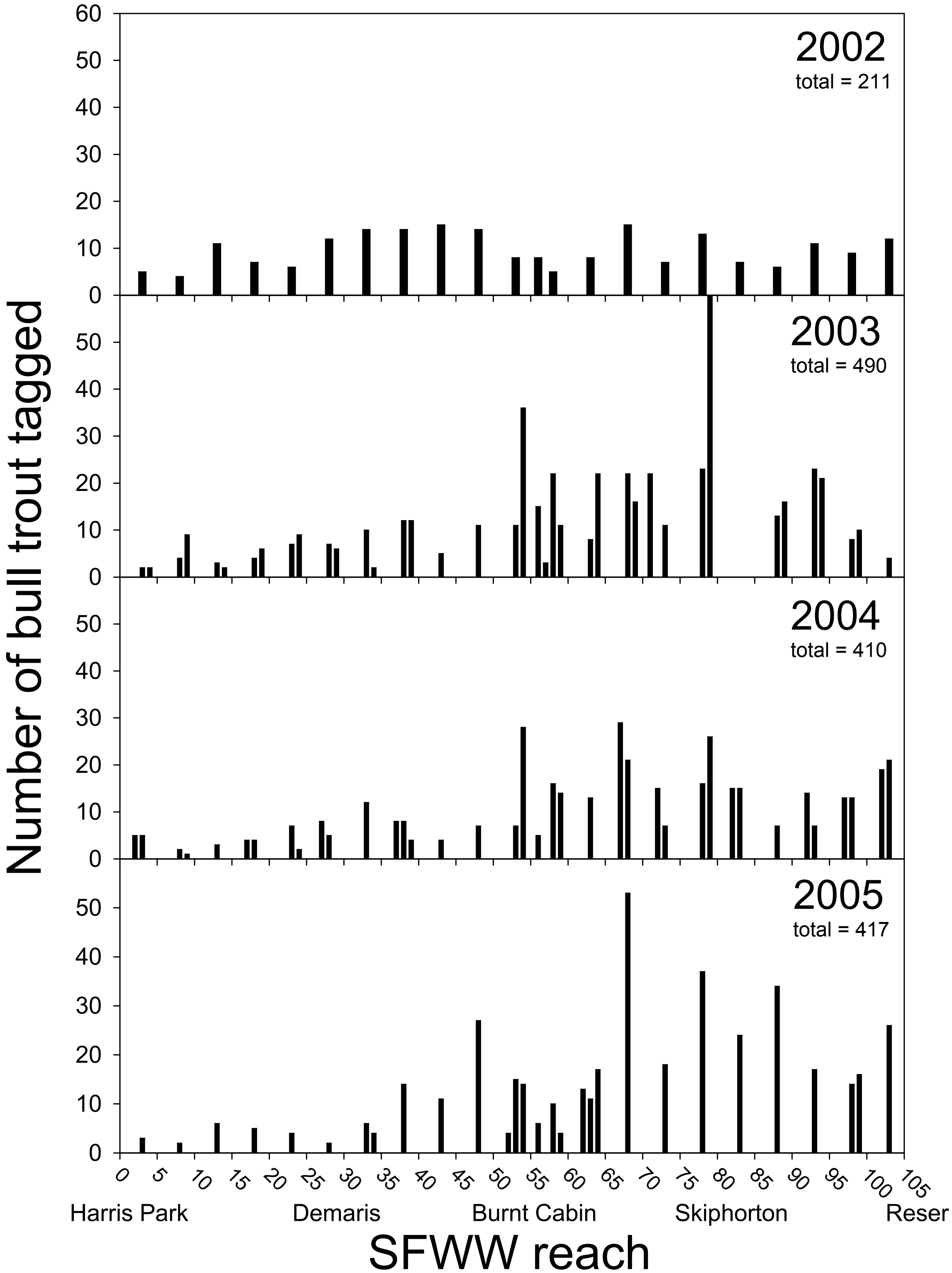

Figure 1.7. Number of bull trout tagged by reach in the South Fork Walla Walla River, 2002 - 2005. Reaches are numbered from bottom to top of the study site. Total numbers tagged are given below sample year.

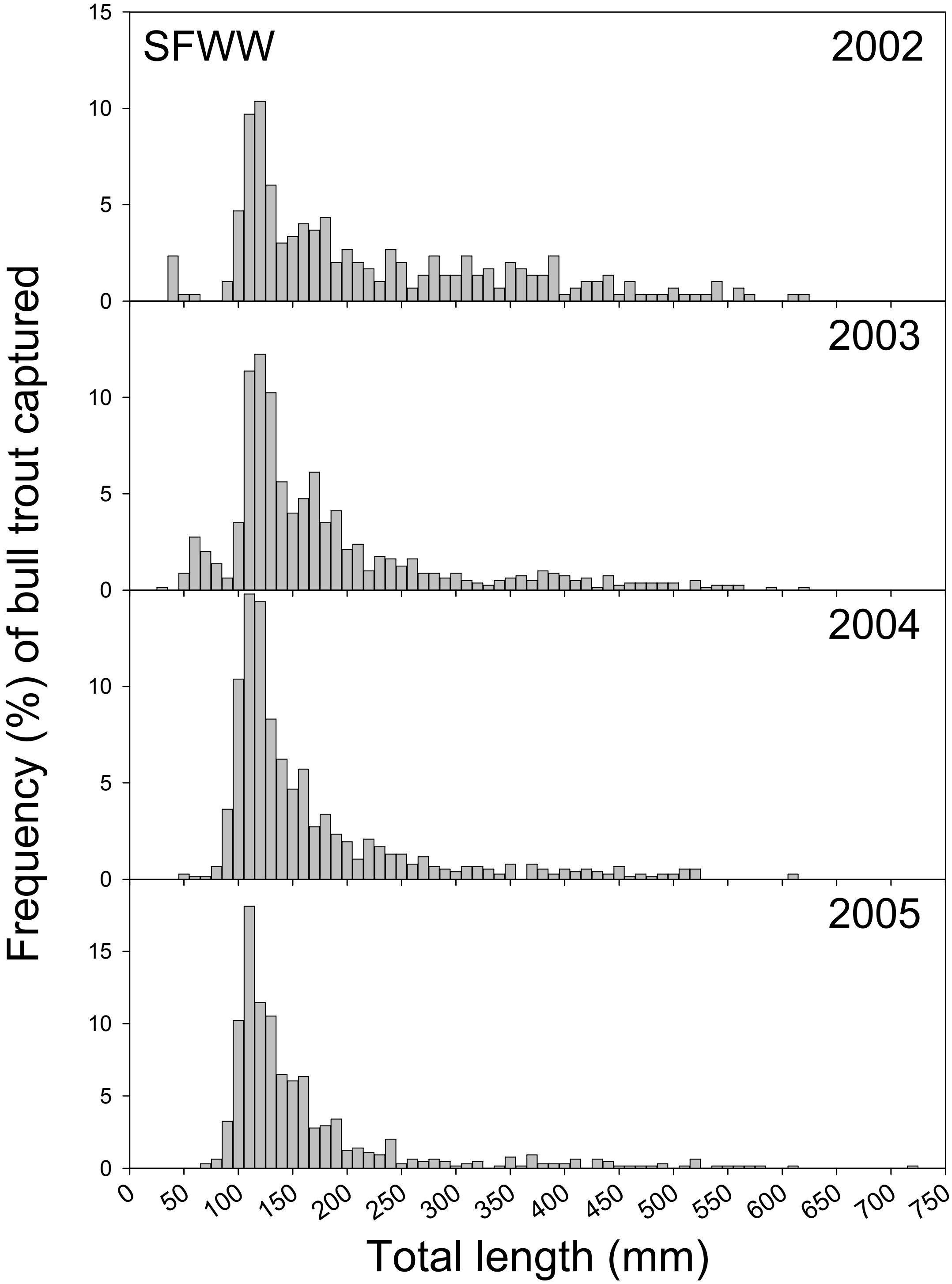

Figure 1.8. Length frequency (% of catch) distribution of bull trout captured and handled in the South Fork Walla Walla River, 2002 - 2005.

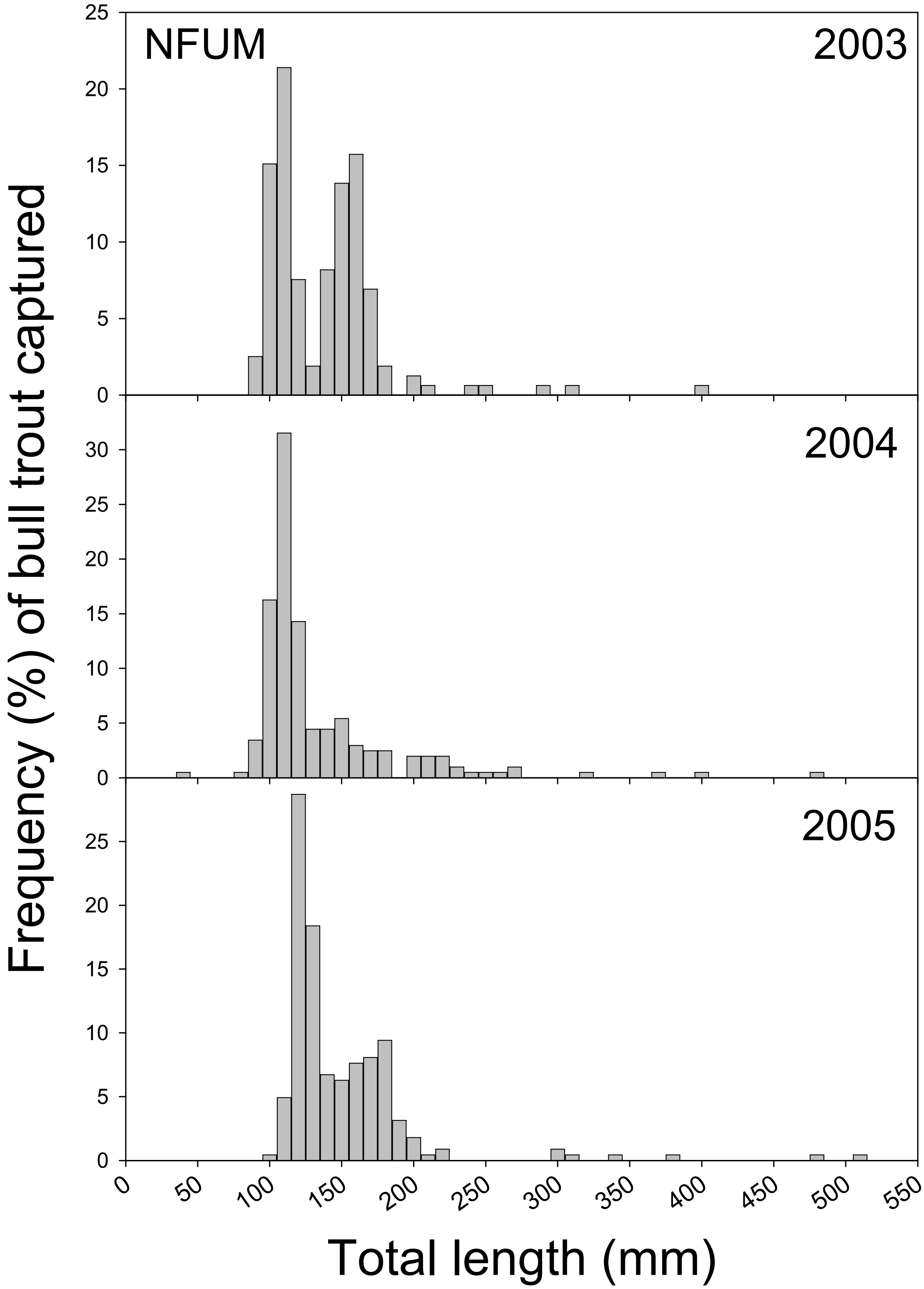

Figure 1.9. Length frequency (% of catch) distribution of bull trout captured and handled in the North Fork Umatilla River, 2003 - 2005.

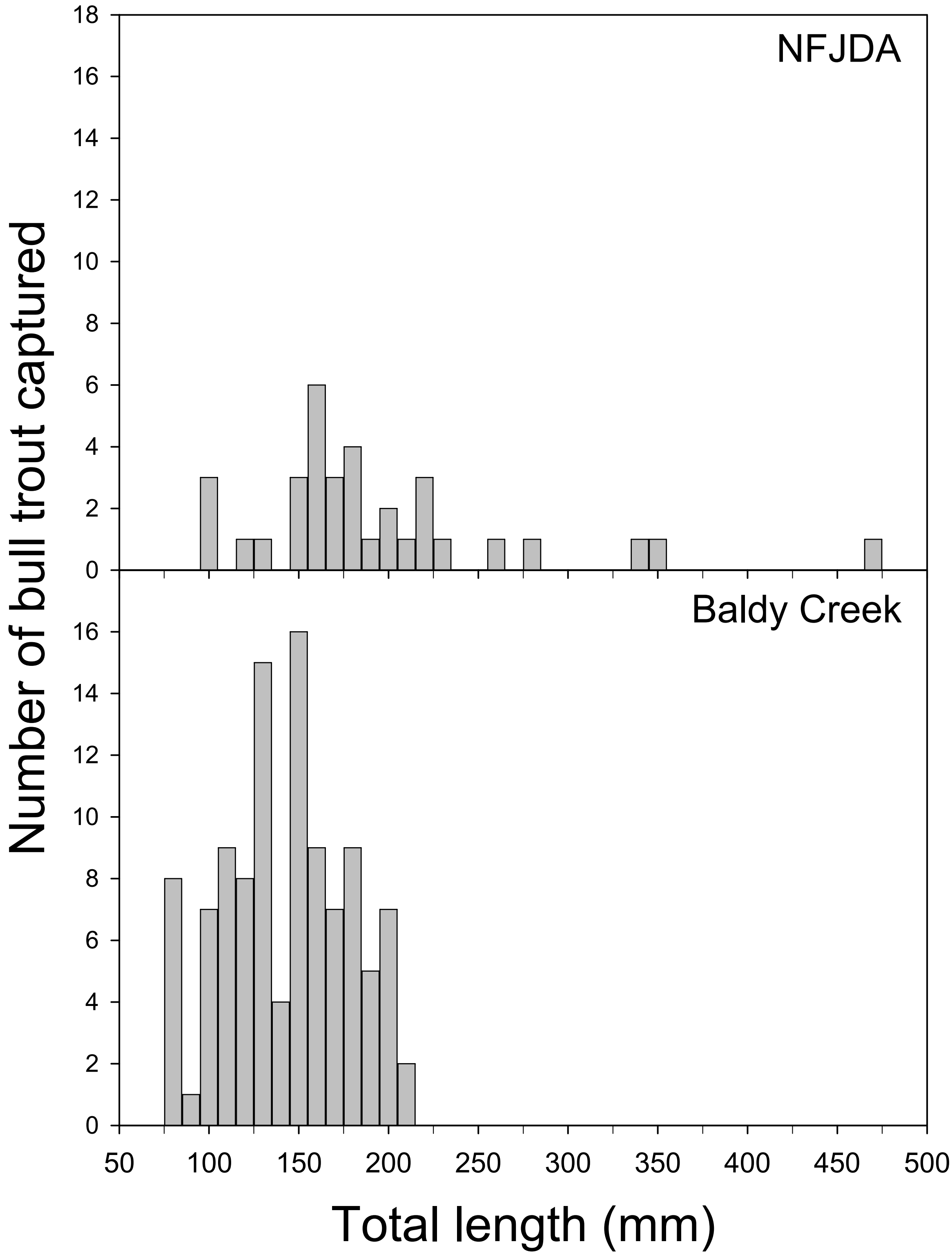

Figure 1.10. Length frequency (numbers captured) distribution of bull trout handled in the North Fork John Day River and Baldy Creek, 2005.

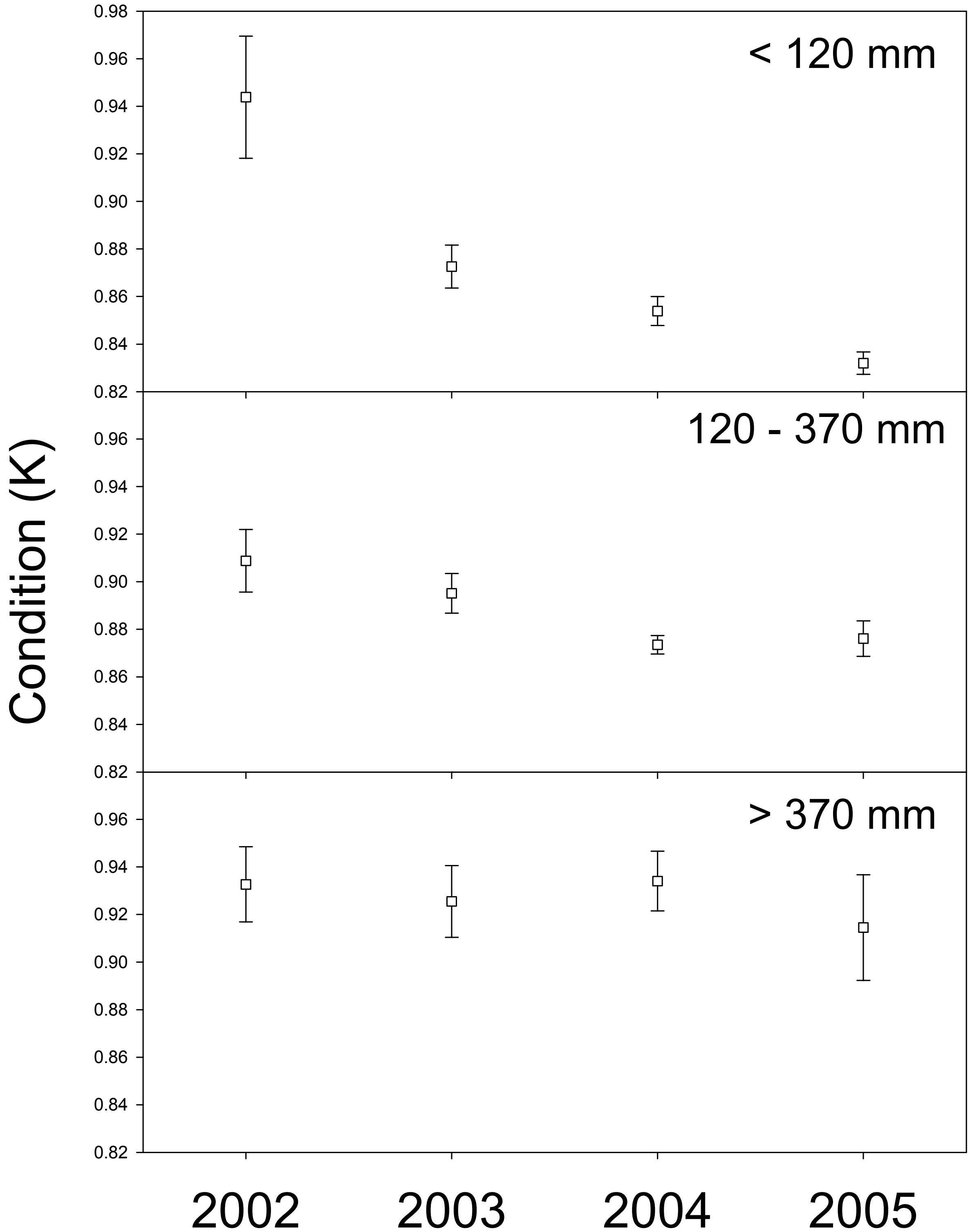

Figure 1.11. Condition (Fulton's K $\pm$ 1 SE) of three different size classes of bull trout sampled in the South Fork Walla Walla River, 2002 - 2005.

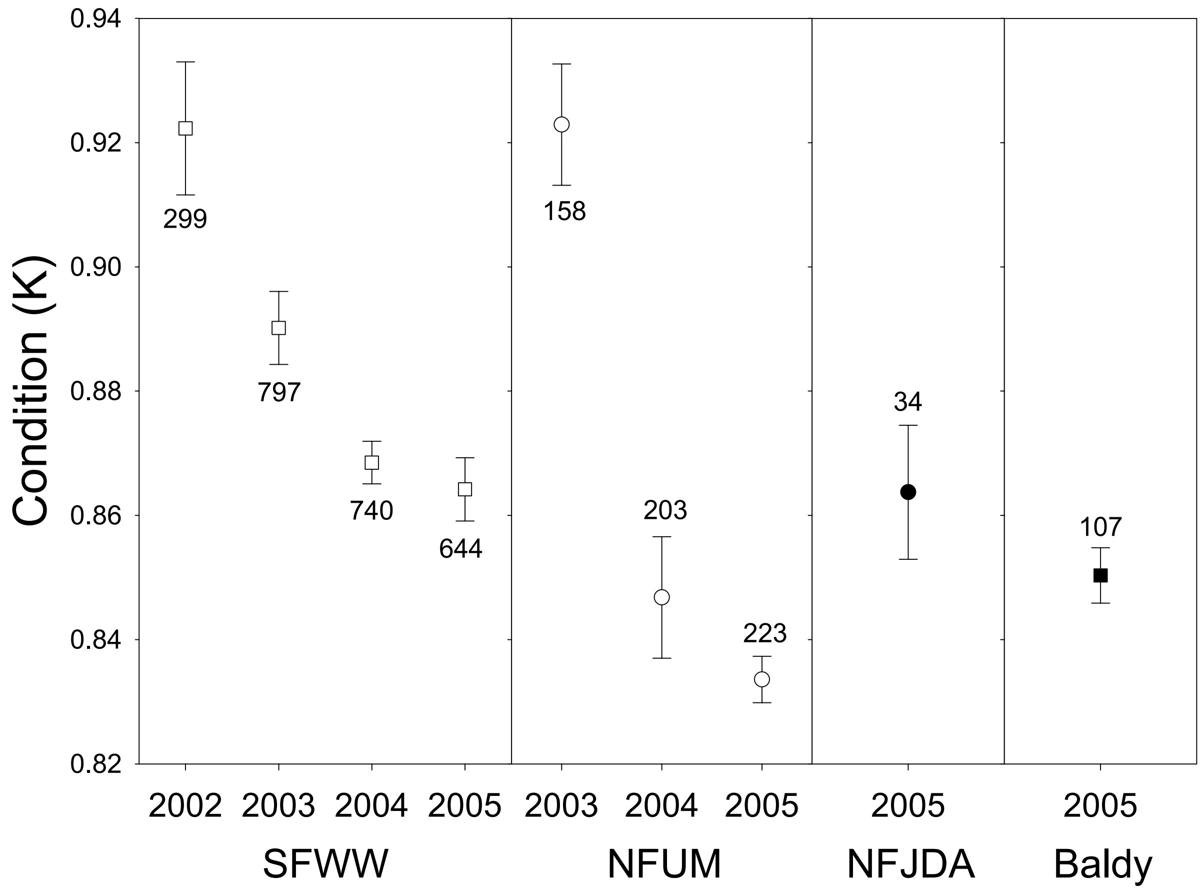

Figure 1.12. Average condition (Fulton's K $\pm$ 1 SE) of bull trout (all sizes combined) sampled in the South Fork Walla Walla River (2002 - 2005), North Fork Umatilla River (2003 - 2005), North Fork John Day River (2005), and Baldy Creek (2005). Sample size is given by error bars.

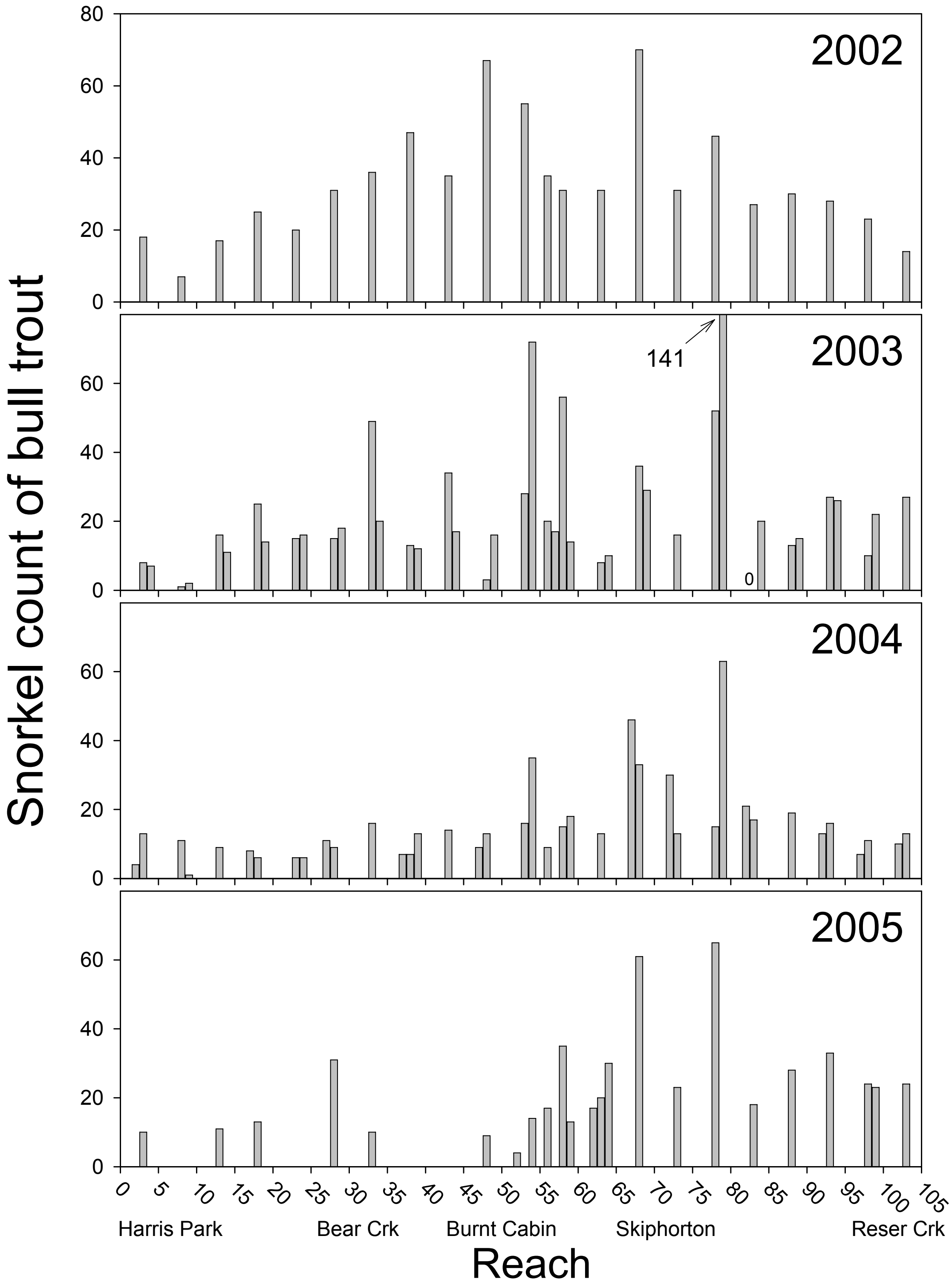

Figure 1.13. Number of bull trout by reach counted during snorkel surveys in the South Fork Walla Walla River, 2002 - 2005. Reaches are numbered from bottom to top of the study site. Note zeros where no fish were observed when sampling was conducted. No bar implies that no sampling was conducted in a particular reach.

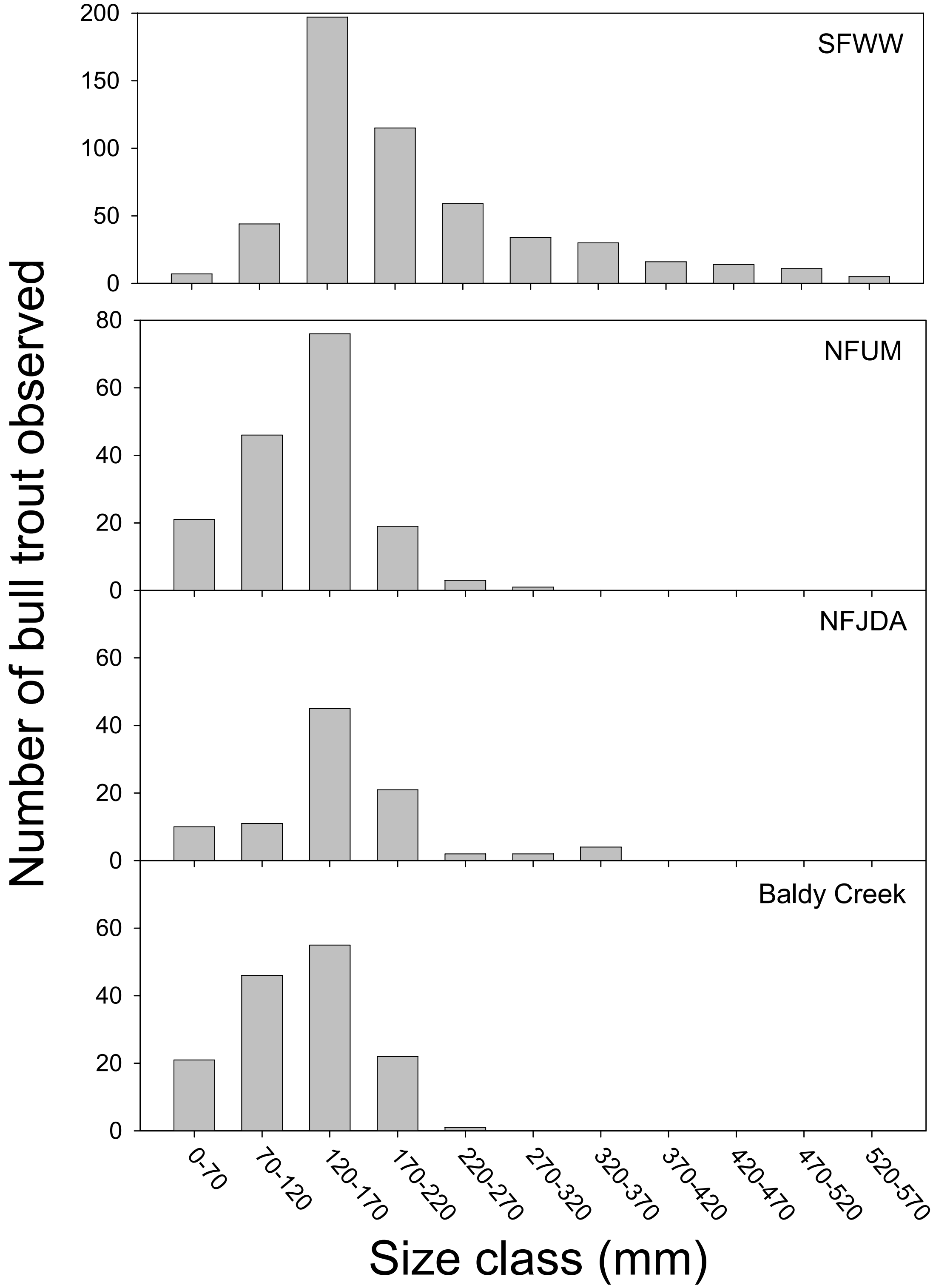

Figure 1.14. Number of bull trout in 70-mm size bins observed during snorkel surveys in the South Fork Walla Walla River, North Fork Umatilla River, North Fork John Day River, and Baldy Creek, 2005. Note changes in y-axis scales.

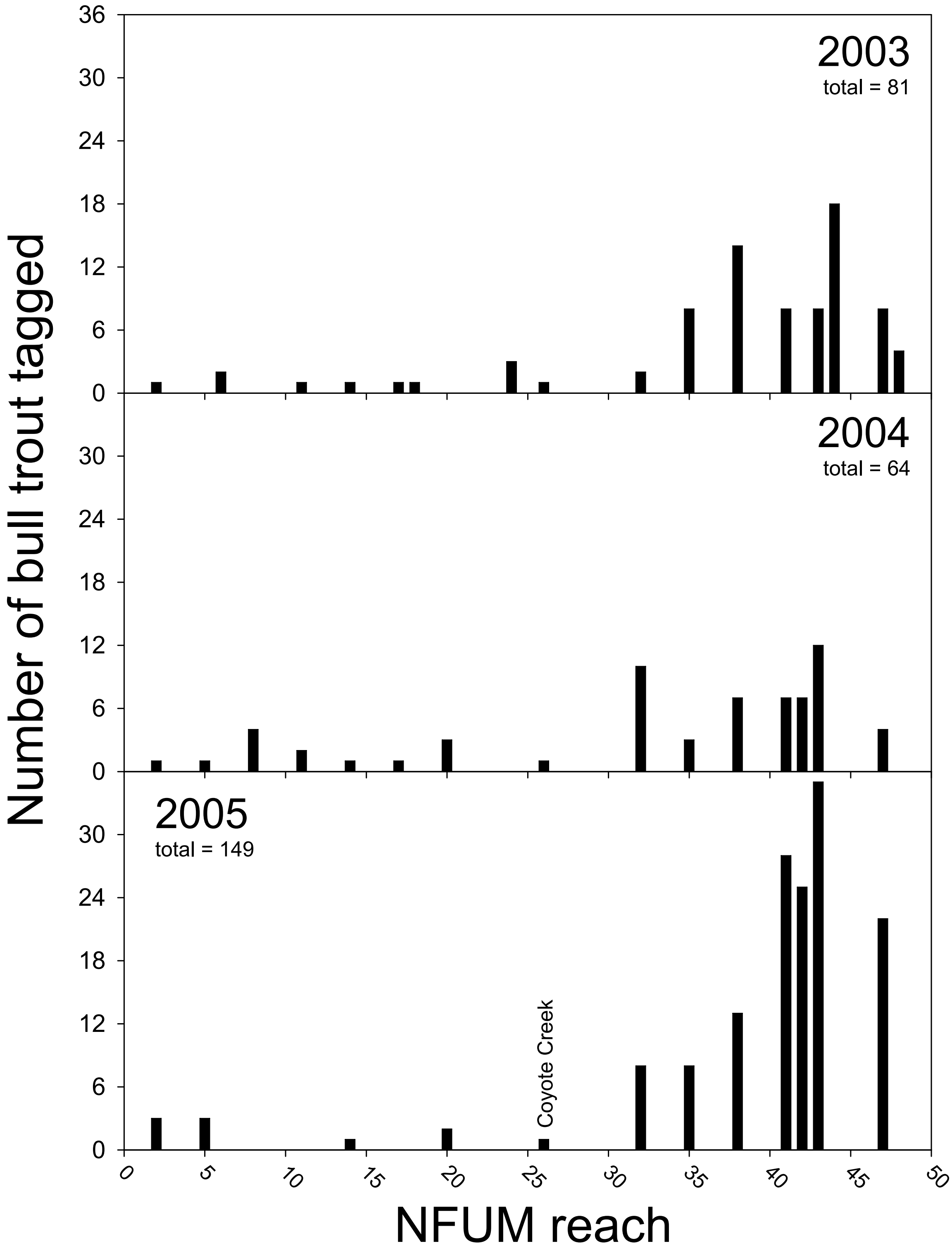

Figure 1.15. Number of bull trout tagged by reach in the North Fork Umatilla River, 2003 - 2005. Reaches are numbered from bottom to top of the study site. Total numbers tagged are given below sample year.

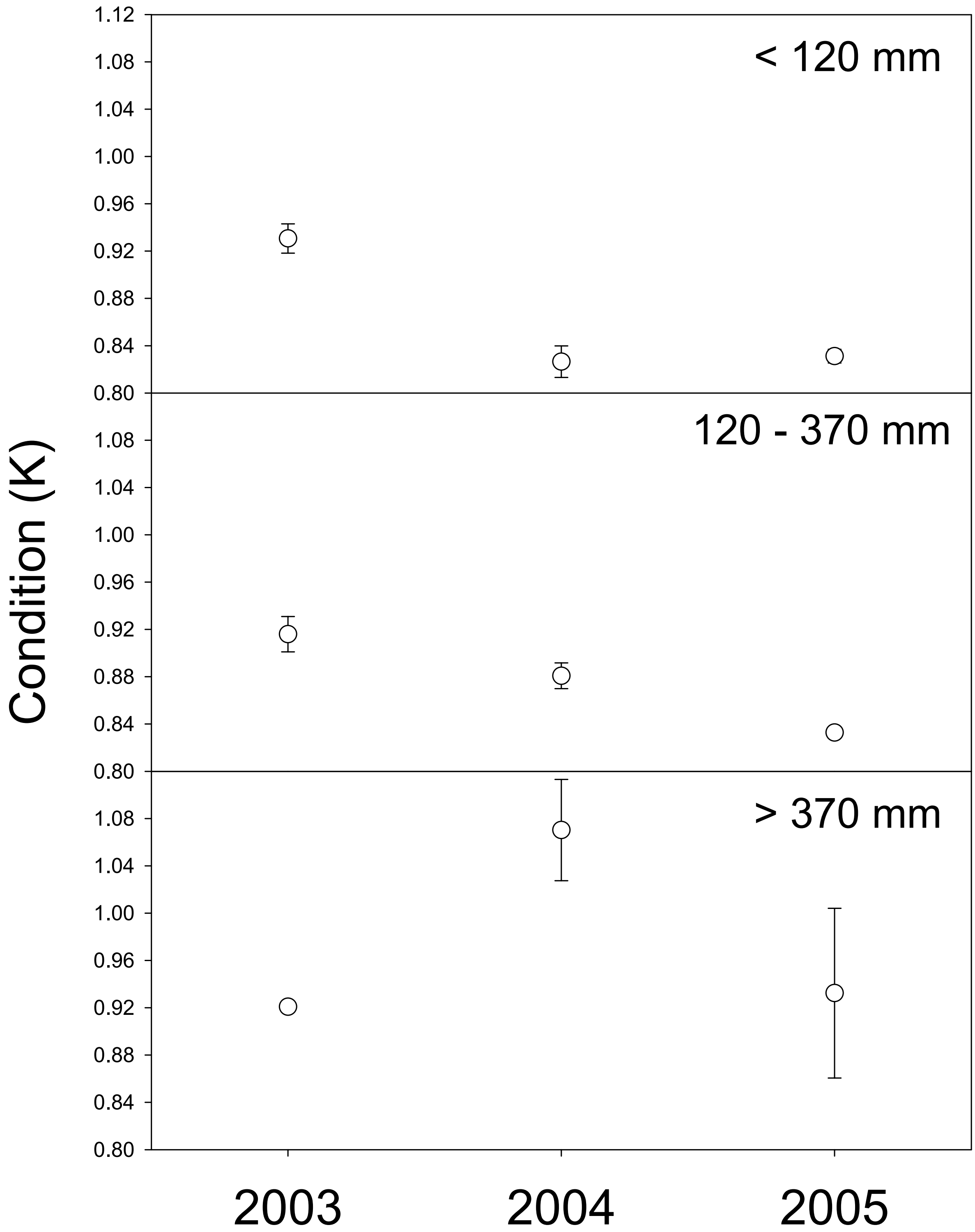

Figure 1.16. Condition (Fulton's K ± 1 SE) of three different size classes of bull trout sampled in the North Fork Umatilla River, 2003 - 2005.

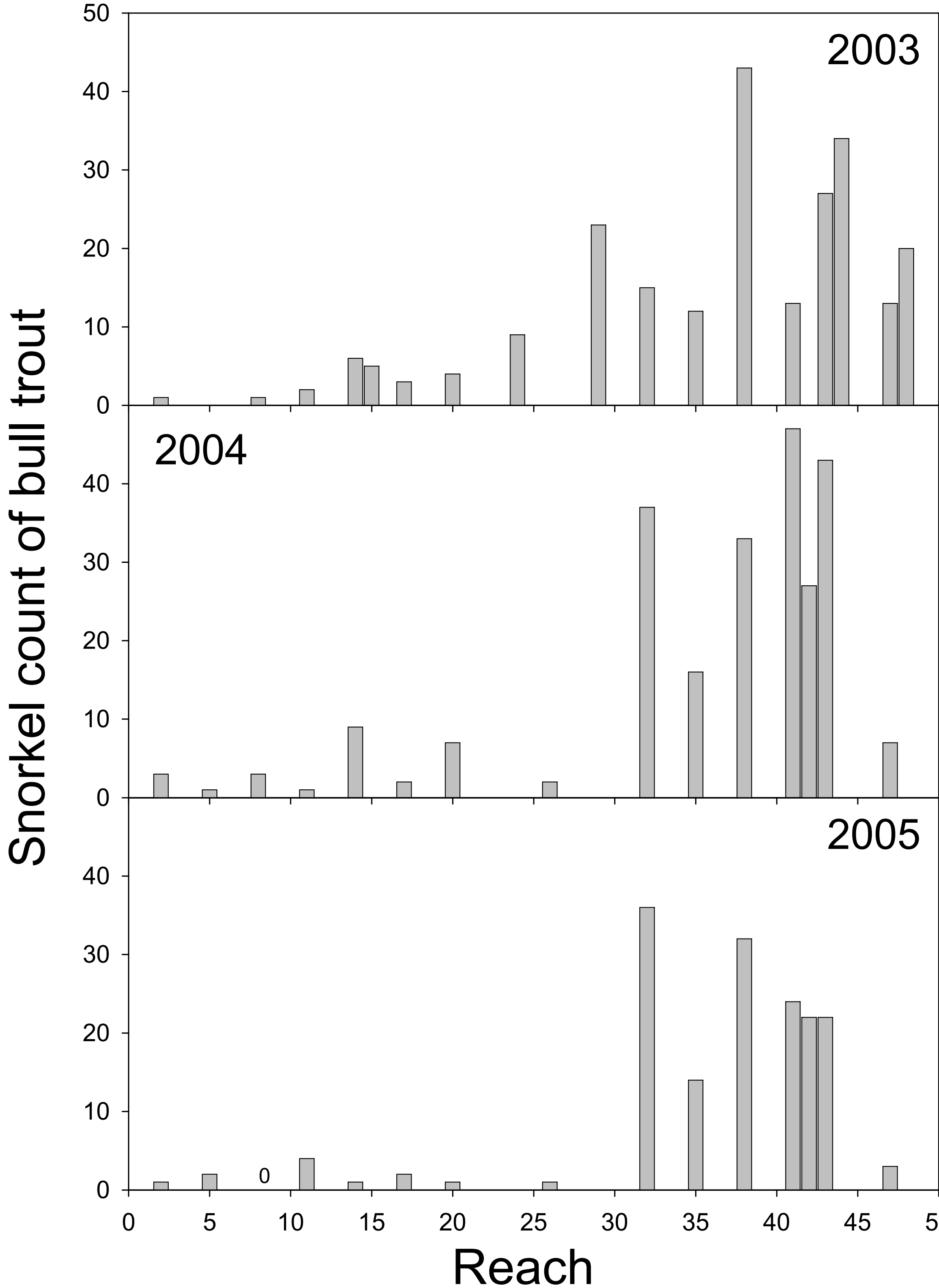

Figure 1.17. Number of bull trout counted by reach during snorkel surveys in the North Fork Umatilla River, 2003 - 2005. Reaches are numbered from bottom to top of the study site. Note zeros where no fish were observed.

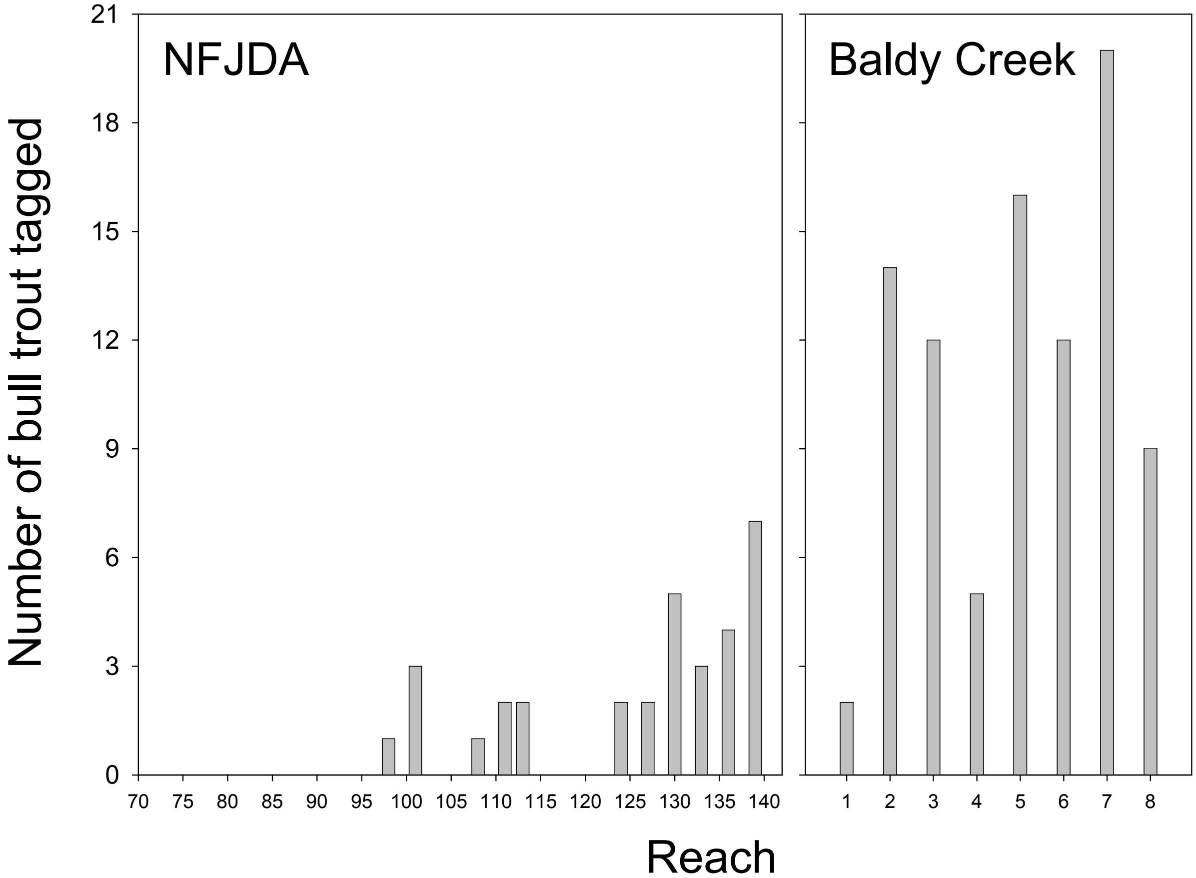

Figure 1.18. Number of bull trout (> 120 mm) tagged by reach in the North Fork John Day River and Baldy Creek, 2005. Reaches are numbered from bottom to top of the study site.

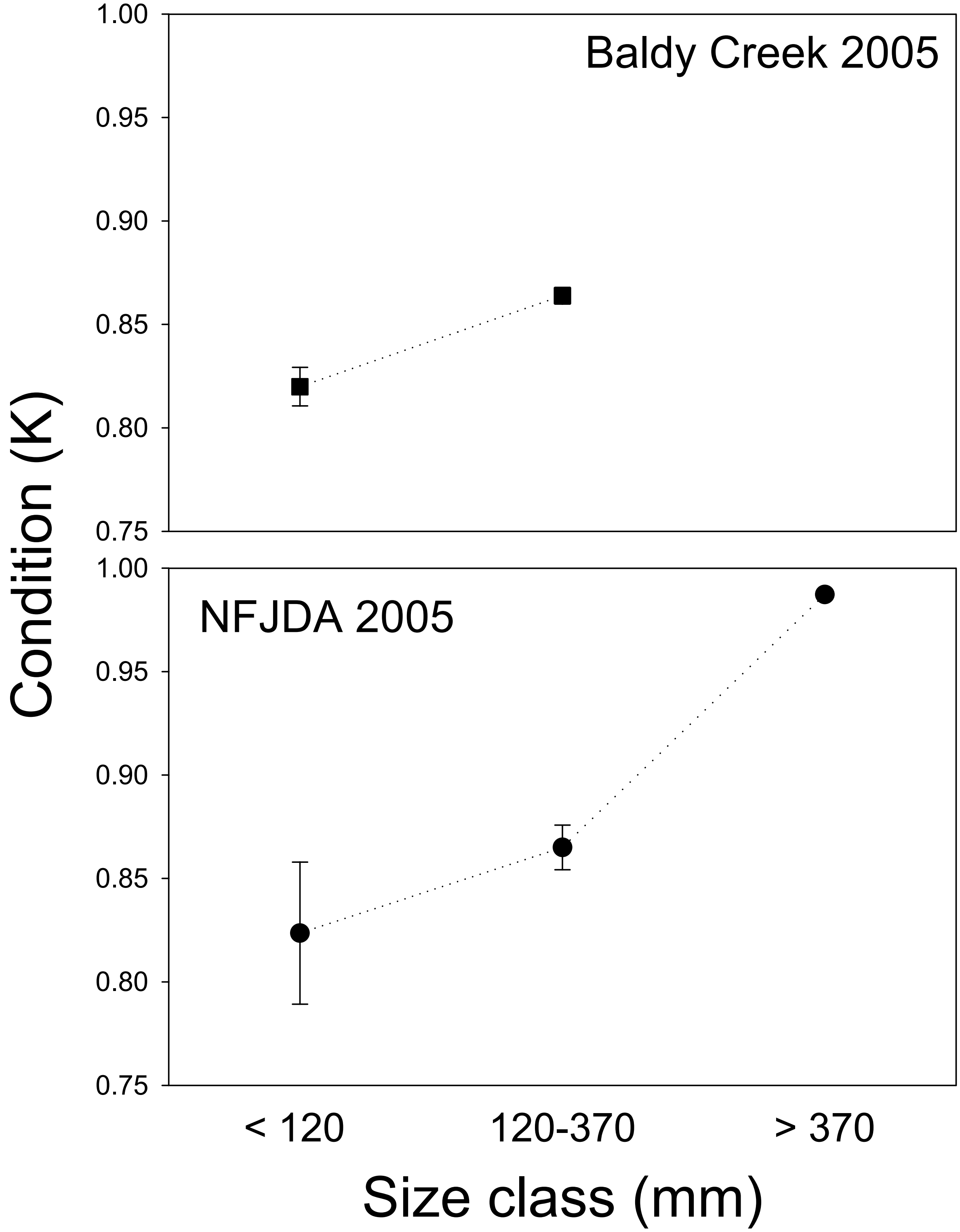

Figure 1.19. Condition (Fulton's K ± 1 SE) of bull trout by size class sampled in the North Fork John Day River and Baldy Creek, 2005. Sample sizes are given by error bars.

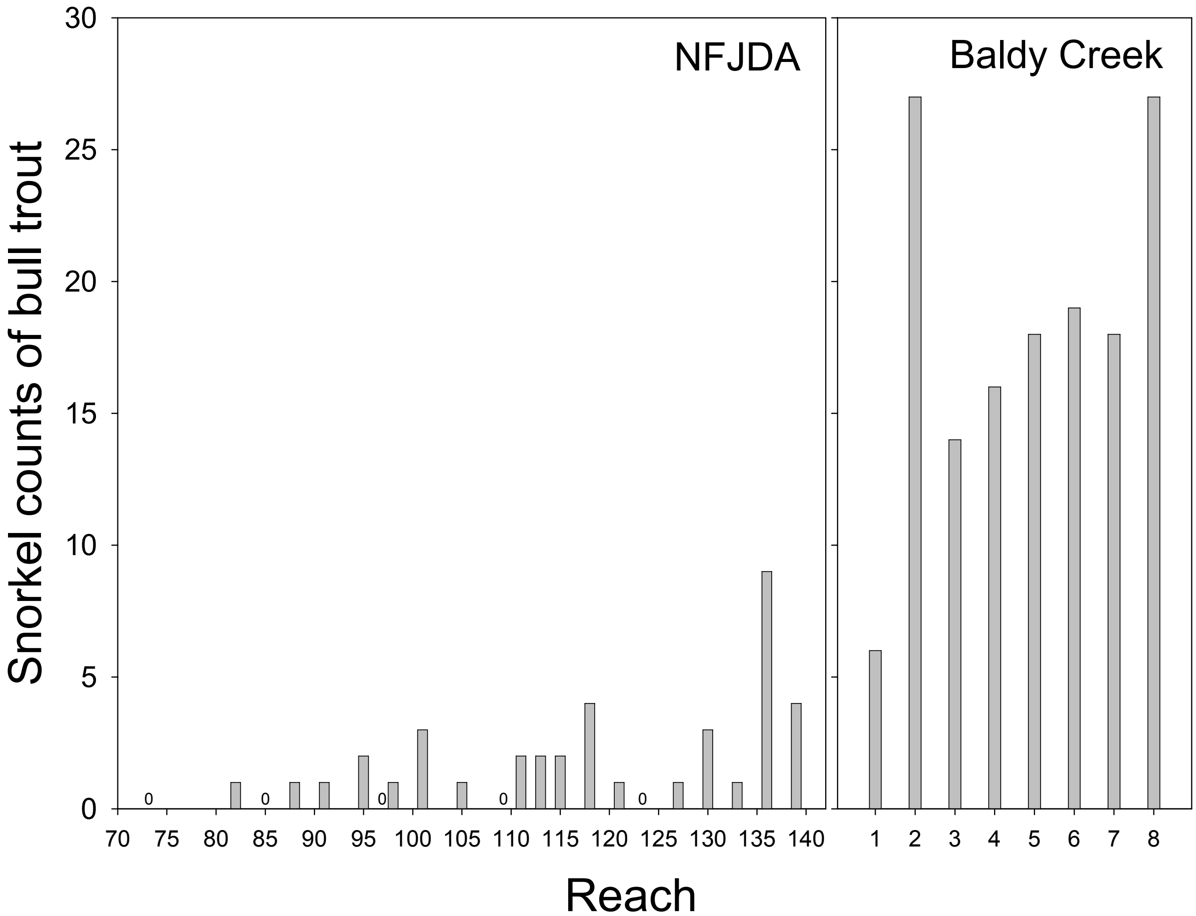

Figure 1.20. Number of bull trout by reach counted during snorkel surveys in the North Fork John Day River and Baldy Creek, 2005. Reaches are numbered from bottom to top of the study site. Note zeros where no fish were observed.

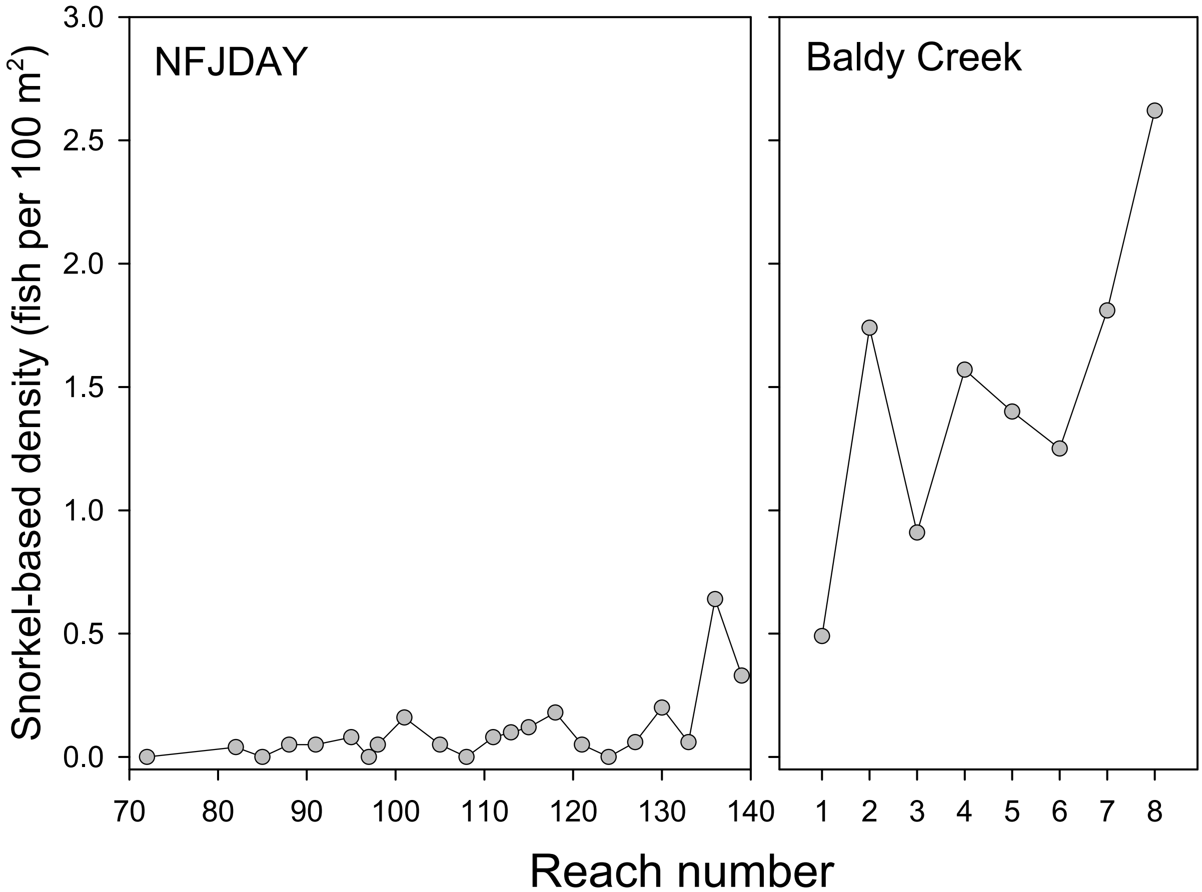

Figure 1.21. Density of bull trout by reach observed during snorkel surveys in the North Fork John Day River and Baldy Creek, 2005. Reaches are numbered from bottom to top of the study site.

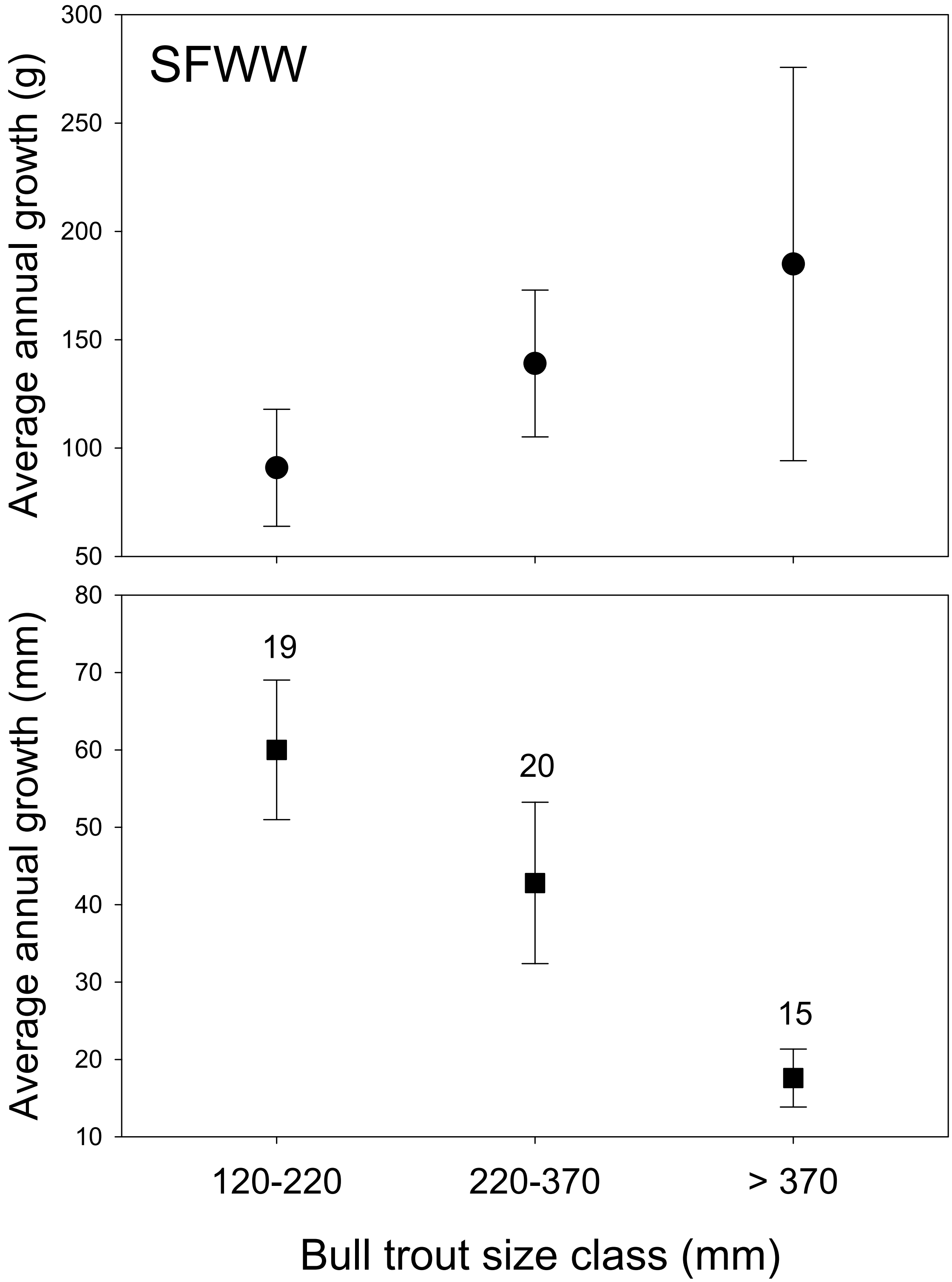

Figure 1.22. Average annual growth (± 2 SE) in weight (g, top panel) and length (mm, bottom panel) for three size classes of tagged and recaptured bull trout in the South Fork Walla Walla River, 2002 - 2005. Sample sizes are given above error bars.

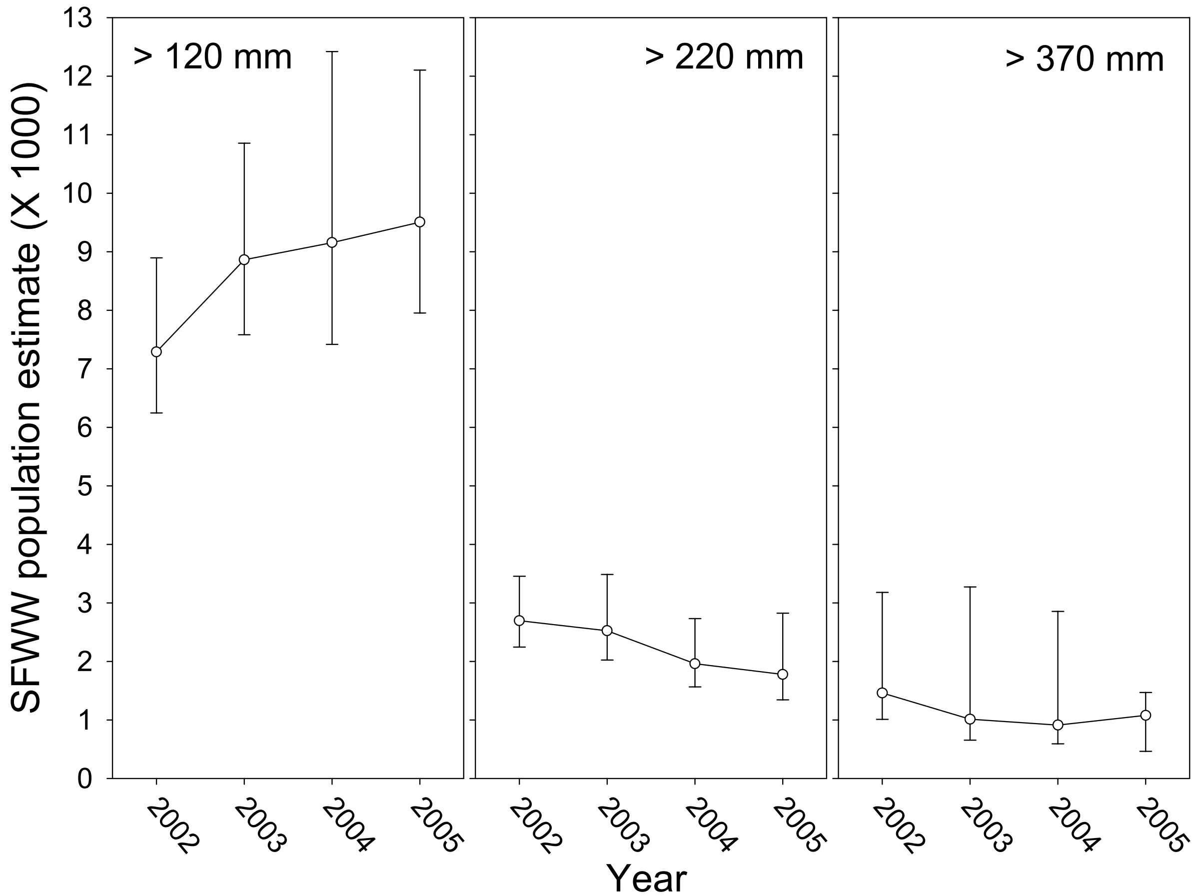

Figure 1.23. Yearly population estimates (± 95% CI) for three size groupings of bull trout in the South Fork Walla Walla River, 2002 - 2005. Bull trout > 370 mm are likely migratory.

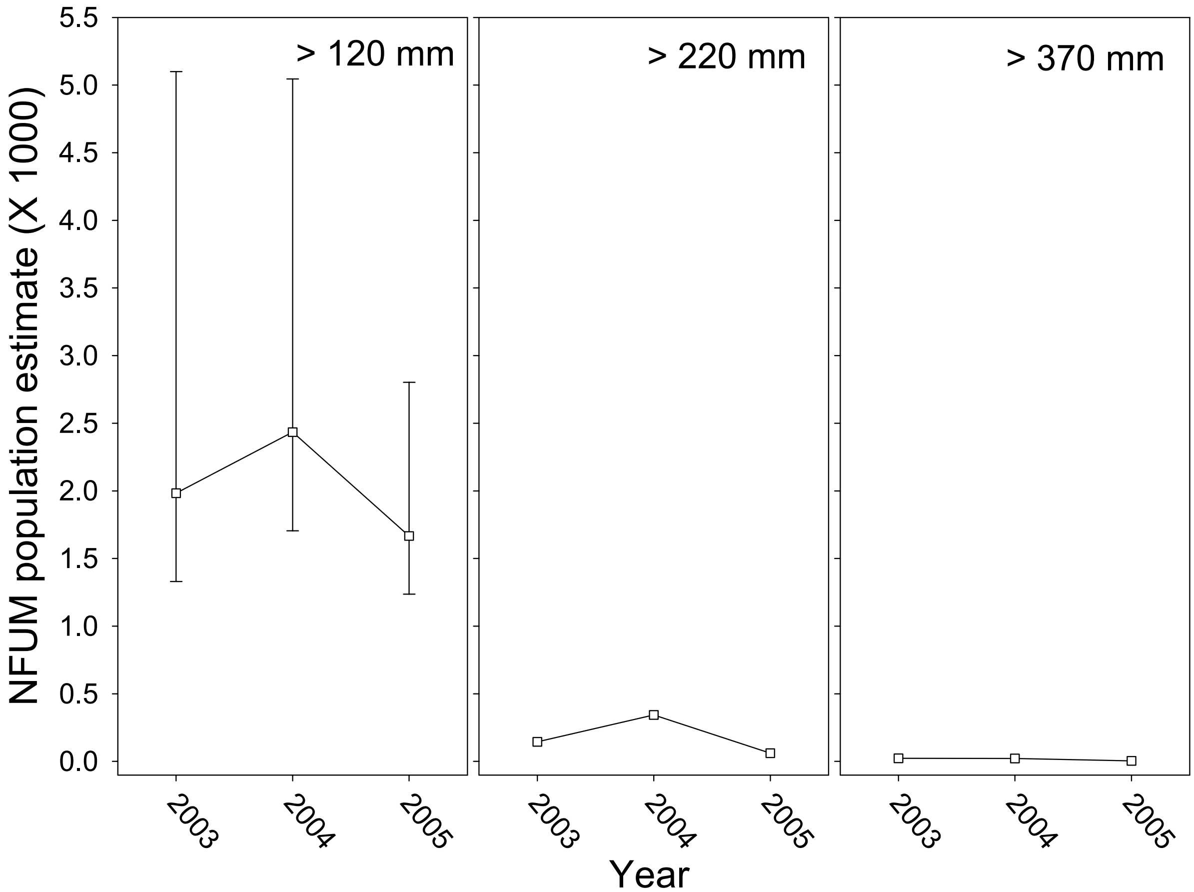

Figure 1.24. Yearly population estimates (± 95% CI) for three size groupings of bull trout in the North Fork Umatilla River, 2003 - 2005. Bull trout > 370 mm are likely migratory. No confidence intervals are obtainable for the bull trout population component > 220 mm or > 370 mm TL.

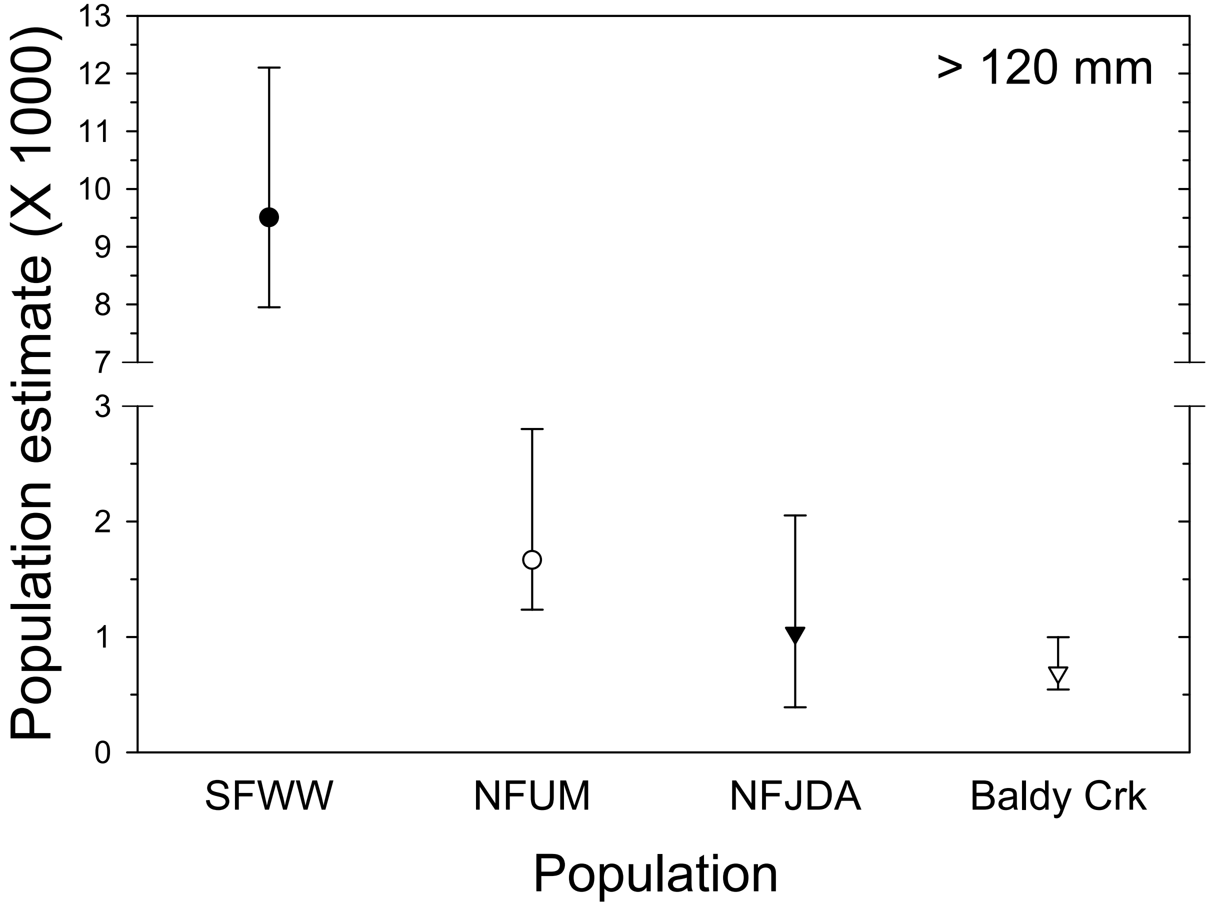

Figure 1.25. Yearly population estimates (± 95% CI) for four populations of bull trout (> 120 mm TL), 2005. Note break of 4000 units in y-axis.

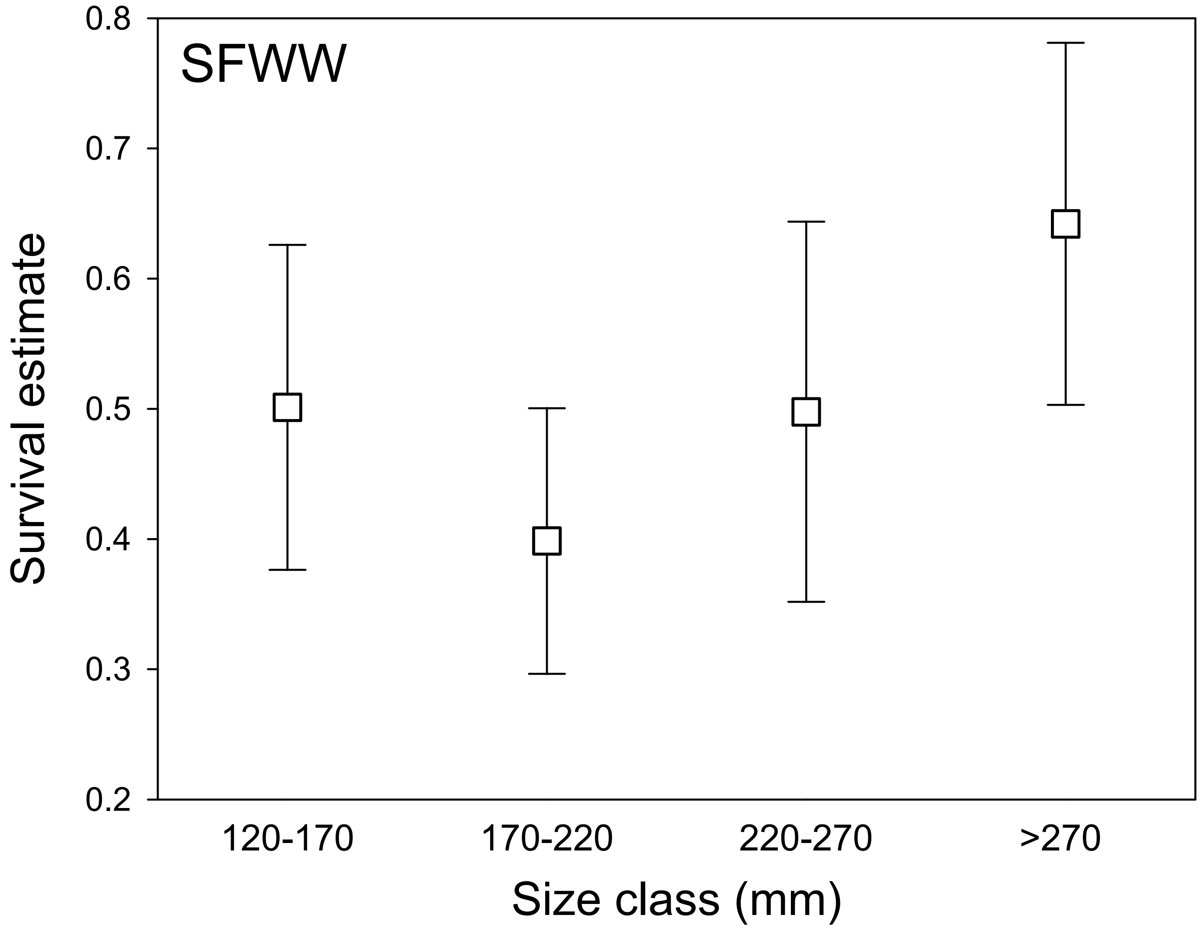

Figure 1.26. Survival estimates (± 1 SE) for four size classes of bull trout in the South Fork Walla Walla River over the period 2002 to 2005.

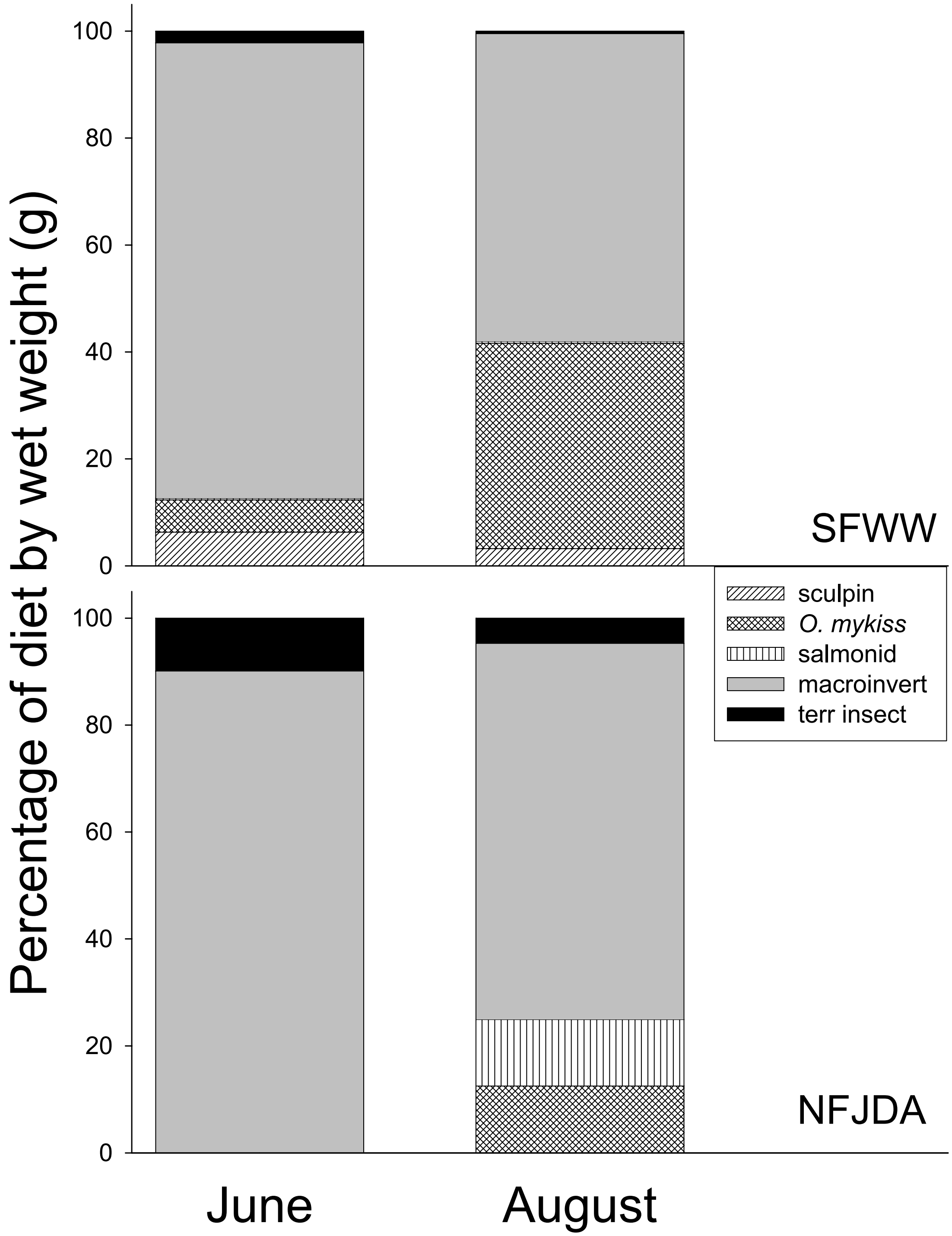

Figure 1.27. Diet composition (% of diet by wet weight) of bull trout taken from the South Fork Walla Walla River (top panel) and North Fork John Day River (bottom panel), June and August 2005. “Salmonid” includes all unidentifiable salmonid species. “Macroinvert” includes all aquatic invertebrates. “Terr insect” includes all terrestrial insects.

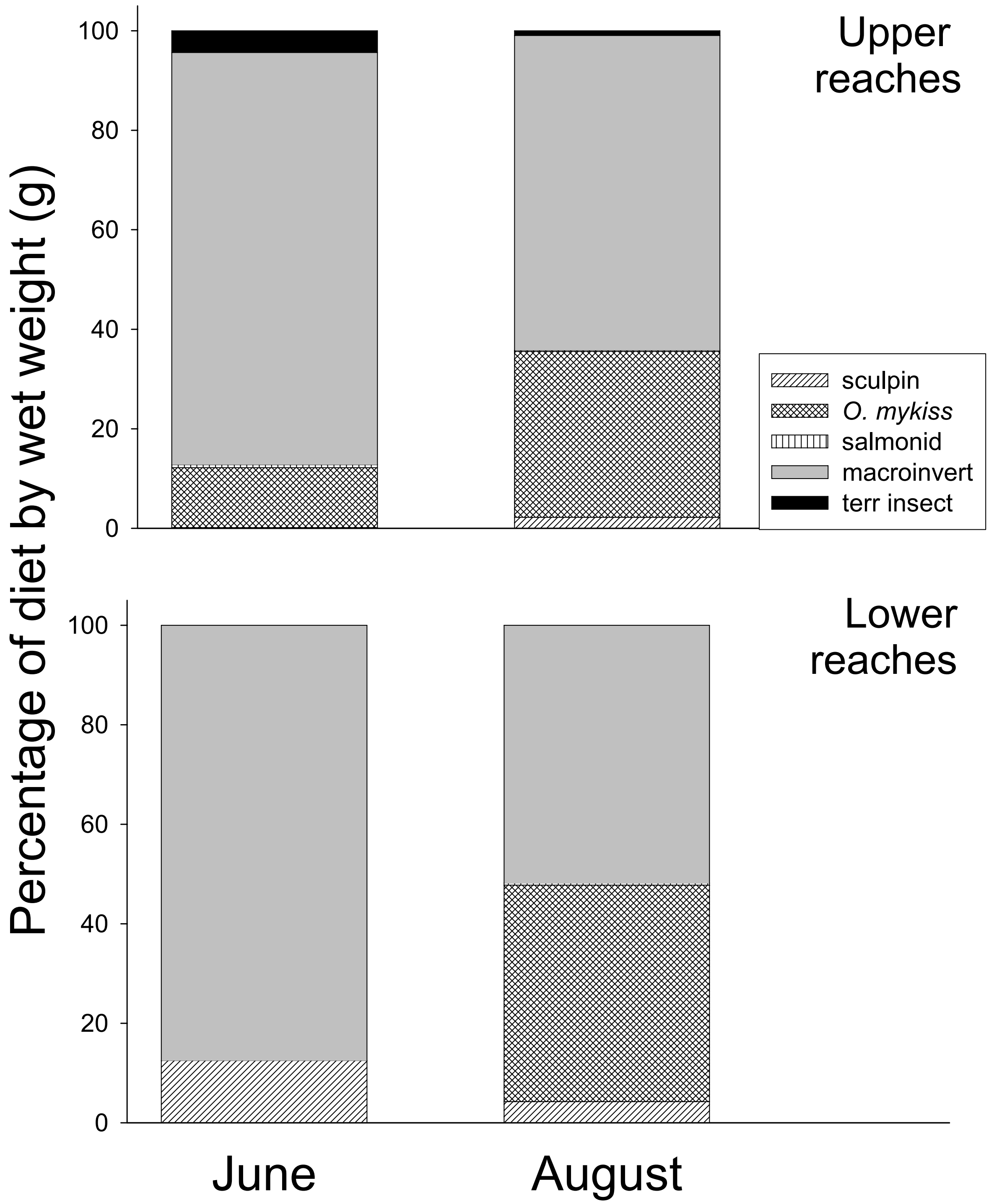

Figure 1.28. Diet composition (% of diet by wet weight) of bull trout captured in the upper (near Reser Creek) and lower (near Bear Creek) reaches of the South Fork Walla Walla River, June and August 2005. "Salmonid" includes all unidentifiable salmonid species. "Macroinvert" includes all aquatic invertebrates. "Terr insect" includes all terrestrial insects.

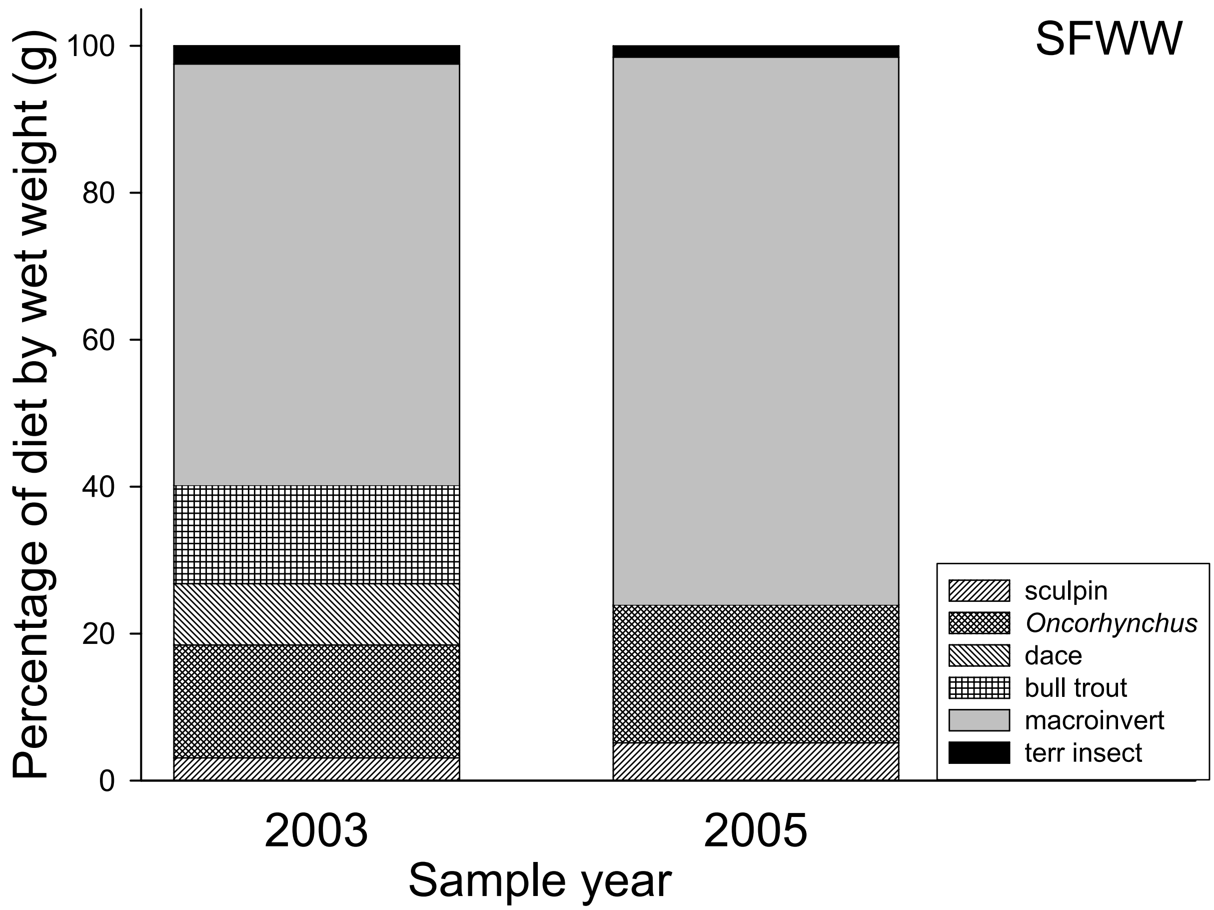

Figure 1.29. Diet composition (% of diet by wet weight) of bull trout captured in the South Fork Walla Walla River in 2003 and 2005. "*Oncorhynchus*" includes all salmonid species, except bull trout. "Macroinvert" includes all aquatic invertebrates. "Terr insect" includes all terrestrial insects.

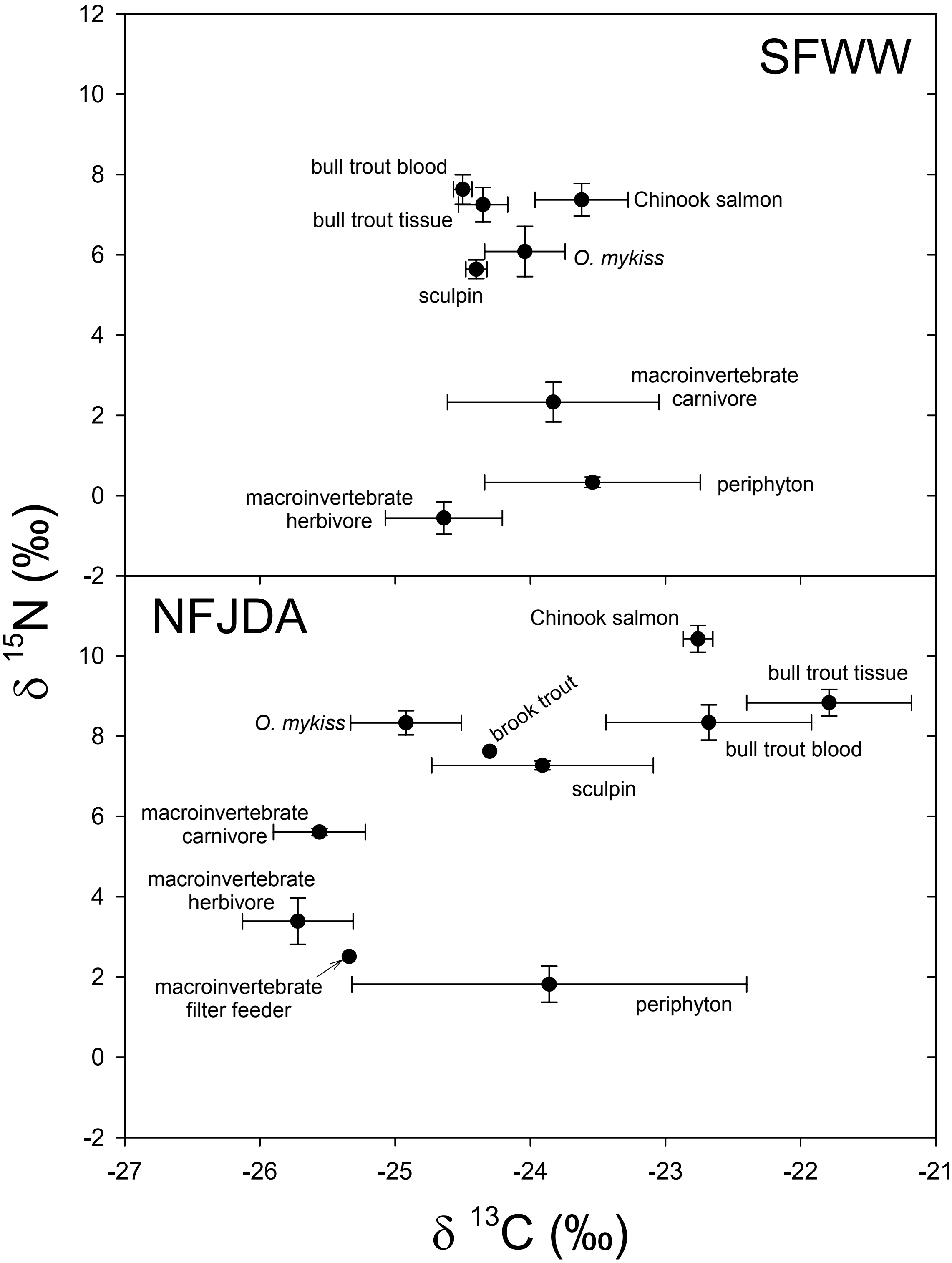

Figure 1.30. Stable isotope (^{15}N and ^{13}C, mean ± 1 SE) composition (‰) of bull trout (blood and tissue samples; n = 8), other resident fish (as named) macroinvertebrates (classified as carnivore, herbivore, and filter-feeder taxa), and periphyton in the South Fork Walla Walla and North Fork John Day rivers, June samples only, 2005.

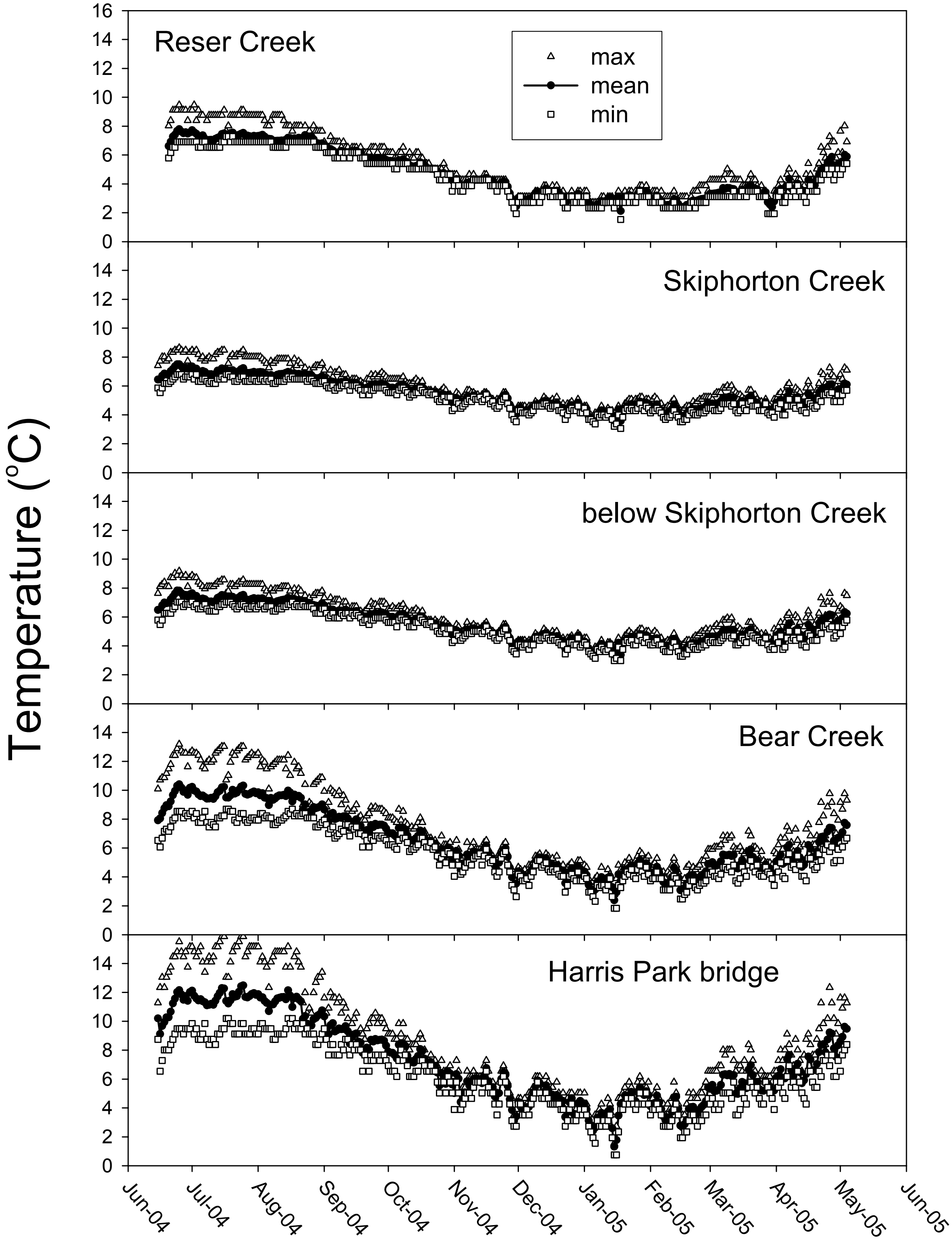

Figure 1.31. Daily temperatures (maximum, mean, minimum) recorded at five locations (Reser Creek is top and Harris Park is bottom of study area) on the South Fork Walla Walla River, Oregon, June 2004 – June 2005.

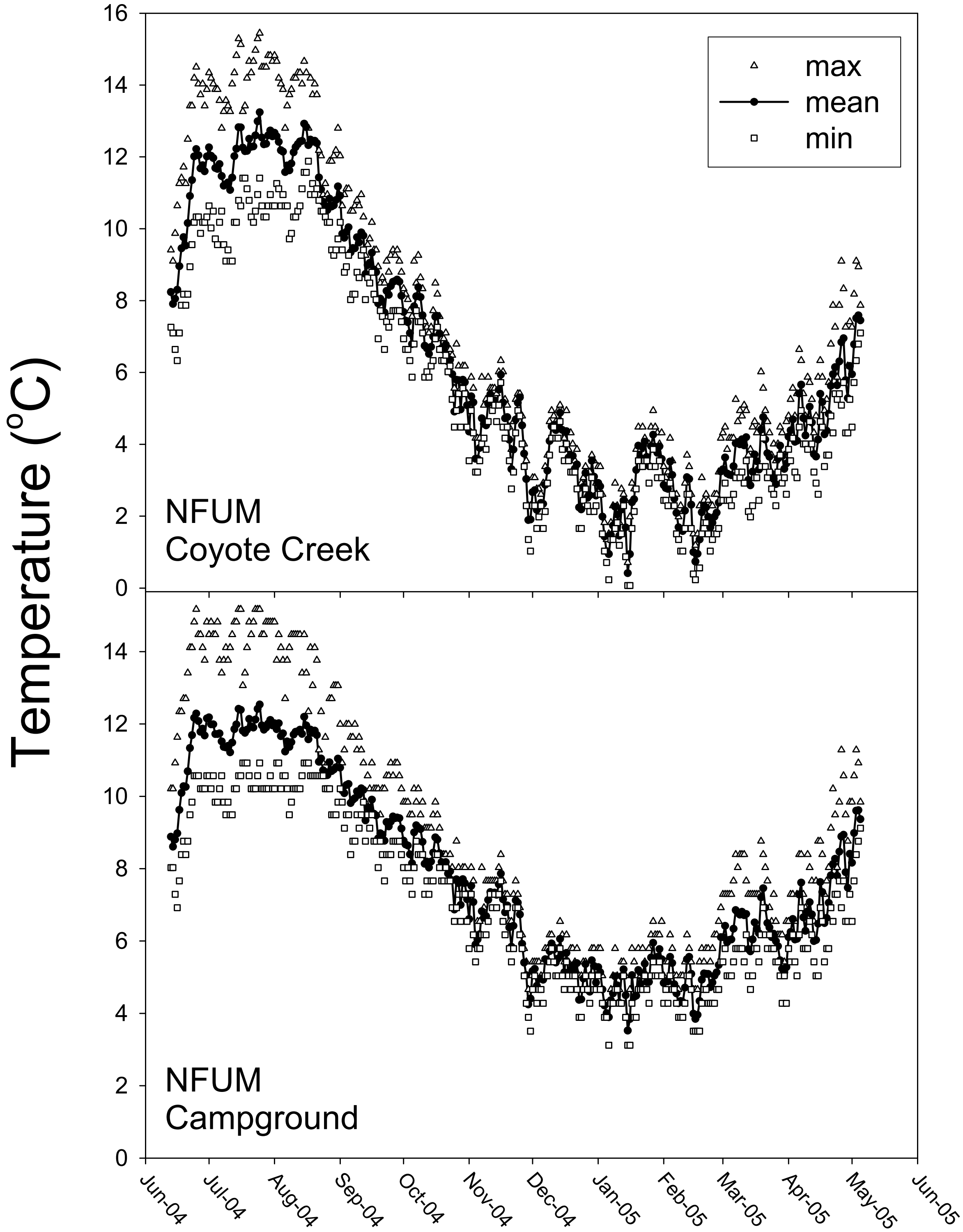

Figure 1.32. Daily temperatures (maximum, mean, minimum) recorded at two locations (Coyote Creek is at top and campground is at bottom of study area) on the North Fork Umatilla River, Oregon, June 2004 – June 2005.

CHAPTER 2:
Detecting changes in population abundance of a threatened species: understanding the accuracy, precision, and costs of our efforts

INTRODUCTION

The conservation and management of species requires explicit information regarding the abundance and trend of populations (Gibbs et al. 1998; Williams et al. 2002) and their response to management actions (Hilborn & Walters 1992; Crowder et al. 1994). This information is especially important for species listed under the Endangered Species Act (ESA). Typically, population trends are monitored using density or count data (e.g., Dennis et al. 1991), stock recruitment data (e.g., Wada & Jacobson 1998), and/or mark/recapture techniques (e.g., Seber 1982), all of which provide an abundance estimate ($\hat{N}$) for a population. While more elaborate approaches for monitor population growth are available (e.g., temporal symmetry model; Pradel 1996), abundance estimates are most widely used due to the availability of existing data sets and standard methodology (Ham & Pearsons 2000).

For each abundance estimation technique, there are costs, both monetary and in terms of harassment (e.g., handling the species of interest), and desired levels of accuracy and precision (Kendall et al. 1992; Wilson et al. 1999). As such, the design of a population monitoring plan mandates explicit consideration of the number of sample units required to obtain a desired accuracy and precision associated with each abundance estimate (Zielinski & Stauffer 1996; Gibbs et al. 1998). With greater levels of variation, it can be difficult or impossible to detect significant changes in population abundance (Taylor & Gerrodette 1993; Maxell 1999). In general, increased sampling effort results in more precise population estimates (Kendall et al. 1992). However, managers must balance the tradeoff between resource expenditure and estimate precision (Wilson et al. 1999). Ideally, the minimum number of sample units required to meet a particular statistical power, allowing for the detection of desired population change over a particular time period, would be explicitly evaluated before widespread implementation of a monitoring plan (Taylor & Gerrodette 1993). Unfortunately, however, these *a priori* evaluations rarely occur.

In conjunction with sample size, there are numerous sources of variation, which also reduce our ability to monitor population trends. In particular, these include: 1) spatial variability, which is a result of the patchy distributions of animals in space (Link &

Nichols 1994); 2) temporal variability, which refers to the interannual (or time frame of interest) variability or changes in the population of interest at a particular location, and occurs as a function of environmental differences between years (Morris & Doak 2002); and 3) sampling variability, which is a function of both the inherent differences in the randomly selected sample units (Williams et al. 2002) and different sources of observation error (Dennis et al. 1991; Staples et al. 2004).

Despite these sources of variability, the precision of abundance estimates can be increased through methodological and analytical techniques. For example, high spatial variation can often be reduced through analytical and design techniques such as stratification of sampling units (Thompson et al.1998; Krebs 1999) and/or the blocking by different ages/stages of interest (Pimm & Redfearn 1988; Zielinski & Stauffer 1996). In addition, sampling variability can be reduced through increased sampling rate, which may not be monetarily feasible, and/or via analytical techniques, which can potentially reduce the sampling effort required to achieve a particular power (Burnham et al. 1987; Morris & Doak 2002). However, there are few examples of studies which have rigorously analyzed the effects of applying statistical techniques (e.g., shrinkage estimators) to reduce sampling variability within the context of detecting changes in population abundance, using actual abundance data collected in the field (Johnson 1989; Ver Hoef 1996).

In lieu of these sources of variability and strategies to limit this variability, the ESA requires formal assessments of population abundance over relatively short time intervals for species listed as Threatened or Endangered. Status reviews, which are updated approximately every five years, are required to determine if the current listing of a species is warranted based on readily-available population information (Section 4(c) ESA). However, historical monitoring data are frequently sparse for many populations, such that status determinations must often rely on limited data. Given natural variability and limitations in the availability and precision of data, it may require a considerable amount of time before new time series of abundance data will provide the ability to detect population change, an important consideration for the conservation of imperiled species.

Bull trout (*Salvelinus confluentus*) is a species of char native to the Pacific Northwest that has experienced significant decline across its range (Rieman et al. 1997), and are currently listed as Threatened under the Endangered Species Act. Habitat degradation (Fraley & Shepard 1989), barriers to migration (Rieman & McIntyre 1995), and the introduction of non-natives (Leary et al. 1993) have led to the decline of bull trout populations across their native range. Today, bull trout exist only as subpopulations over a wide range of their former distribution (Rieman et al. 1997), which historically extended from northern California and Nevada at the southern limits, to the headwaters of the Yukon River in Northwest Territories.

The biology and complex life history strategies of bull trout make them especially challenging to sample. In particular, bull trout are behaviorally cryptic, naturally occur at low densities (Rieman and McIntyre 1993), and are known to exhibit multiple life-history forms that can coexist within a single population (Rieman and McIntyre 1993). As bull trout recovery plans are being drafted, strong consideration should be given to the appropriate sampling rate, the level of precision, sources of variability, costs associated with different monitoring techniques, and the ability to effectively detect changes in population abundance over specific time intervals.

In this paper, we examine several key components related to monitoring animal populations with abundance estimates, using data collected as part of a comprehensive effort to form a template for bull trout recovery. First, we bootstrap our data to investigate the tradeoff between the costs and different levels of precision associated with increasing sampling efforts, given spatial variability and differences in precision across techniques. We compare the results from the mark-resight data, which is typically more robust than density estimates (Minta & Mangel 1989), with alternative monitoring strategies including: 1) count data, an index of population abundance and a simpler population technique; 2) stratification of sampling units, and the effects on both the number of units required to achieve a target level of precision; and 3) grouping the population into biologically relevant components, in order to examine how sampling effort and precision around abundance estimates changes with different age/stage classes. Next, we determine the number of sample units associated with each monitoring technique required to detect a specified decline in population size for a given time frame and statistical power. Finally, we evaluate the use of analytical techniques (e.g., shrinkage estimators) as a method for reducing sampling variability and improving the effectiveness of our sampling.

STUDY AREA

The data for our analyses were collected in the South Fork Walla Walla River (hereafter SFWW), a single, local population of the Walla Walla core area in eastern Oregon (Whitesel et al. 2004). The SFWW study site (21 km in length), was contained primarily within the Umatilla National Forest and contained a variety of habitat conditions ranging from simple channel conditions with little structural complexity to braided, complex channels that were structurally diverse. The SFWW is dominated by cold, groundwater, with summer maximum temperatures at the bottom of the site not exceeding 14° C. The SFWW is known to contain a relatively large population of both smaller (< 370 mm), resident and larger (> 370), potentially migratory bull trout (Al-Chokhachy et al. 2005).

METHODS

Sampling Design and Methodology

We divided the SFWW site into 102 reaches of approximately 200 m in length. To avoid potential problems associated with non-discrete habitat units (Hankin 1984), we began all reaches at pool tail features and continued to the first pool tail beyond 200 m, resulting in reaches of slightly unequal lengths. To sample across the population and different habitat types, we systematically sampled (random start) 39 and 41 reaches in 2003 and 2004, respectively.

We initiated sampling near the end of spring runoff using a variety of techniques to capture bull trout. When captured, we anesthetized all bull trout > 120 mm, tagged each fish with an external anchor tag, and returned fish to the approximate location of capture. We initiated snorkel surveys immediately after the completion of the sampling period. Snorkel data were used to: 1) provide tag resights for mark-resight population estimates, and 2) provide an index of abundance based solely on counts of fish for comparison. Because bull trout exhibit natal site fidelity and migrate upstream during this period to spawn (Fraley & Shepard 1989), we started snorkel surveys at the highest reach within each year and continued downstream to avoid double-counting migratory fish. All tagged (i.e., marked individuals that were resighted; R) and unmarked bull trout were enumerated and assigned into 50-mm size classes beginning at 120 mm in total length. We conducted marking and snorkeling efforts in 41 reaches, totaling over 9.6 km and 46% of the study site. Across reaches, the average number of marked (M), counted in snorkeling surveys (both marked and unmarked; C), and resighted (R) bull trout per reach were 12.2, 19.5, and 2.4, respectively (Table 2.1).

Mark-Resight Population Estimates

We used snorkeling and tagging data to parameterize mark/resight population estimates using a Lincoln-Petersen bias-adjusted estimator (Chapman 1951: hereafter referred to simply as the Lincoln-Petersen estimator and abbreviated as LP), and estimated the overall population size. We estimated the standardized population sizes for each reach using tagging and snorkeling data for each individual reach, calculated the average number of bull trout per 200 m across reaches, and multiplied this average by the total number of reaches in the site. To standardize the number of bull trout per 200 m for each reach, we divided each reach estimate by the actual reach length and multiplied this estimate by 200.

In order to better understand the relative contribution of different components of the population to the overall abundance, we simulated and estimated all statistics for three separate size categories (*see* Table 2.1): > 120 mm (hereafter > 120), > 220 mm (hereafter > 220), and > 370 mm (hereafter > 370). The > 120 size class includes both sexually mature and immature fish, the > 220 size class includes sexually mature fish, and the > 370 size class includes large, potentially migratory individuals (Rieman & McIntyre 1993). Estimates for each category were calculated by grouping the marked (M), both marked and unmarked (counted via snorkeling, C), and resighted (R) data by the specific size criteria (*see* Table 2.1).

Table 2.1. Descriptive statistics for marked (M), resighted (R), and counted in snorkel surveys (both marked and unmarked; C) bull trout in three size classes from 2003 field data.

	M	C	R
> 120 mm			
Totals	499	799	97
Average per reach	12.2	19.5	2.4
Standard Deviation	11	17.3	2.7
Coefficient of Variation	0.90	0.89	1.13
> 220 mm			
Totals	137	353	41
Average per reach	3.3	8.6	1
Standard Deviation	2.5	4.5	1.1
Coefficient of Variation	0.76	0.52	1.1
> 370 mm			
Totals	43	105	9
Average per reach	1	2.6	0.2
Standard Deviation	1.1	2.5	0.5
Coefficient of Variation	1.1	0.96	2.5

Variance, Strata, and Shrinkage Estimator

We used a theoretical variance estimator (Seber 1970) to estimate the sampling variance for each reach; large variances, primarily due to spatial variation across reaches, made it difficult to detect population trends. To reduce this variance component, we evaluated the effect of stratification. Stratification, when applied appropriately, nearly always leads to increases in precision, even when based on little prior information (Cochran 1977), and is especially important when individuals are clumped (Thompson et al. 1998). Thus, we stratified the study site using two strategies, each of which relied on some prior knowledge of bull trout habitat use or distribution. First, we divided the study site into three strata based on the geographic location of the sampled reach within the study site. This stratification approach partitioned reaches into an upper site stratum (5.5 km; hereafter Strata 1), where the majority of spawning activity occurs, a mid-site stratum (8 km; hereafter Strata 2), where very high densities of bull trout have been reported, and finally, a low-site stratum (7.5 km; hereafter Strata 3), where the lowest densities of bull trout occur. For the second stratification scenario, we used professional opinion of habitat complexity, to stratify the reaches into two categories (high and low). We designated reaches based on knowledge of the physical habitat present in each reach, and other relevant research on bull trout habitat association patterns across their range (e.g., large woody debris; Rieman & McIntyre 1993). For each stratification strategy, we emulated the frequent situation where there is little or no fish distribution or abundance information available *a priori*.

In addition to stratification, we evaluated the effect of using a random-effects model and associated shrinkage estimator, also called an empirical Bayes estimator. This approach partitions the overall variance into process variation, which can include both spatial and temporal components, and sampling variation; furthermore, the random-effects model removes this sampling variation from the overall variance (Ver Hoef 1996; Link 1999; Burnham & White 2002). For temporal variation, random-effects models that treat the temporal variation as random with an average value ($E(\varepsilon^2) = \sigma^2$) have demonstrated excellent performance with high accuracy and precision (Link & Nichols 1994; Burnham & White 2002); this technique has also been applied to spatial variation (Johnson 1989; Ver Hoef 1996). Our model for spatial variation was:

$$\tilde{N}_j = \hat{E}(N) + \sqrt{\frac{\hat{\sigma}^2}{\hat{\sigma}^2 + \hat{\text{var}}(\hat{N}_j \mid N_j)}} \times \left[\hat{N}_j - \hat{E}(N)\right],$$

where,

$\tilde{N}_j$ = shrinkage estimator for population size based on the sample from reach j,

$\hat{E}(N)$ = expected value of population size based on estimates from all reaches,
$\hat{N}_j$ = estimate of population size based on the sample from reach j,
$\hat{\sigma}^2$ = estimated random spatial variance, and
$\hat{\mathrm{var}}(\hat{N}_j \mid N_j)$ = sampling variance of $\hat{N}_j$.

The estimate of the random spatial variation ($\hat{\sigma}^2$) derives from the expected value of the theoretical total variance:

$$E\left[\hat{\mathrm{var}}(\hat{\bar{N}})\right] = \frac{\hat{\sigma}^2 + E\left[\mathrm{var}(\hat{N} \mid N)\right]}{n},$$

where,
$\hat{\bar{N}}$ = estimate of $\hat{E}(N)$.

We followed Burnham et al. (1987) to estimate the spatial variance for unequal sample variances.

Conceptually, this approach assumes that a weighted average of two or more estimators may be better than any of the estimates alone (Johnson 1989). The strength of the weights are based on the relative values of variances of the estimators; the lower the variance, the greater the weight. To better estimate true spatial variation and increase power to detect temporal trends, we applied a random-effects model, in which process variation represents spatial variation (no annual temporal variation). The random-effects model leads to shrinkage estimates (hereafter LPSE) with improved precision, as a result of reduced mean square error and equal or improved coverage when compared to maximum likelihood estimators (Burnham et al. 1987; Link 1999; Burnham & White 2002), count-based abundance estimators (Johnson 1989; Link & Nichols 1994), and Jolly-Seber abundance estimators (Link & Nichols 1994), and have been successfully used for the LP bias-adjusted estimator (Ver Hoef 1996).

Count Data

In addition to the LP estimator, we ran all analyses using count data from the snorkeling surveys (hereafter snorkel counts), as well as tagging totals (e.g., one-pass removal; hereafter capture counts) to represent alternative approaches used in situations where mark-resight is not practical or possible. However, count data do not include the additional sampling variation in recapture probability associated with mark/resight population estimates (Thompson et al. 1998), and are not estimates of population size, but rather are indices of abundance (Williams et al. 2002). The count data allowed us to compare the ability to detect population trends using less costly abundance indices, based on snorkeling and capture surveys, with LP estimates based on mark-recapture data.

Simulations

We drew samples for each simulation using a non-parametric bootstrap procedure with replacement. For non-stratified random sampling, we drew a reach randomly from all possible reaches, and then estimated the overall population size as: (study area stream length/reach length) $\times$ reach population size. Next, we estimated the associated sampling variance as: (entire stream length/reach length)^2 $\times$ reach variance. We repeated this process until the specific number of reaches to be sampled was achieved; for these non-stratified samples, we estimated the "true" population size as the mean of the total population size estimates of the sampled reaches across the bootstrap samples. We estimated the variance, which included spatial and sampling variation, as the variance of the estimates. We repeated this process 10,000 times across a range of the number of reaches sampled (10 to 41 for 2003; 10 to 39 for 2004).

We used 10 as a minimum number of reaches sampled in our simulations because preliminary simulation estimates and variances had poor stability when fewer than 10 reaches were sampled. Concomitantly, the coefficient of variation was > 1.0, which is high for use in trend detection. Thus, all simulations included a minimum of 10 reaches.

For the stratified random sampling simulations, we drew samples proportionately from each strata. We calculated the mean population size and variance within strata as described for non-stratified samples, but the overall mean and variance were weighted by strata. For each year, the strata weights were calculated as the number reaches in a strata, divided by the total number of reaches. For these samples, we estimated "true" population size and variance as the mean and variance of the total population size estimates across bootstrap samples.

One of the major goals of a monitoring design is to obtain adequate statistical power to detect a declining trend in population size or vital rate (Taylor & Gerrodette 1993; Thompson et al. 1998). Therefore, we used the bootstrapped population estimates (and sampling variances) and indices (snorkel and capture counts), to evaluate, given our data, the power to detect a 25, 50, and 75% decline in population size or index over a 5, 15, and 30-year period. To simulate the decline in population size over a given time frame, we randomly selected (i.e., bootstrapped) a given number of reaches; for each selected reach, we decreased the total estimated population size by a fixed amount such that the overall drop, in a deterministic world, would be linear and equal the specified 25, 50, or 75% drop for the given time frame. For example, a 0.05 decline per year would be required to achieve a 25% decline over 5 years; thus we multiplied the bootstrapped sample estimate by 0.95 for Year 1, 0.90 for Year 2, 0.85 for Year 3 and so on. To estimate the power, we used PROC REG (SAS Institute 1999), regressed the estimated population size by year (time), and recorded how many times the slope parameter was negative and significant ($p \leq 0.10$; one-sided test). Thus, we calculated power, or probability of detecting a population decline, as the number of times a negative trend was detected divided by the number of simulations. For each time frame, we estimated power for the population estimators and indices versus number of reaches sampled.

RESULTS

For comparison, we present the coefficient of variation (CV) to standardize results (Figure 2.1). Across years, the CVs of the LPSEs, which we considered the most reliable, for the > 120 and > 370 groups were higher than the CVs for the > 220 mm size class (Figure 2.1). We observed similar patterns in the CVs for capture counts and snorkel counts. The lower CVs for these estimates likely represented the tradeoff between the number of marked and resighted fish in each reach and reduced variation due to “blocking” by size (e.g., less variability in the distribution of fish > 220 mm). When all fish were used (Figure 1a), relatively high sample sizes (number of reaches) were required, likely as a result of the variable distribution in the numbers of small fish from reach to reach. Conversely, estimates for the > 370 group demonstrated relatively low variability across reaches; however, low numbers of marked and resighted fish (an order of magnitude less than the > 220 group across years) resulted in a high variance and CV (Figure 2.1c). Because variance of the population estimates in 2003 and 2004 for the >220 group demonstrated the greatest precision across all estimators and indices used, and to simplify presentation of our results, we present all subsequent results for only the >220 group (2003).

Stratification by geographic location did not increase the precision of the estimates (Figure 2.1), primarily because the variation in the middle strata (Strata 2) contained all the variation and the lowest sample size (number of reaches). Strata 1 and 3 did show reduced variation and lower CVs under stratification, but not enough to result in reduced variation overall. Stratification based on habitat complexity categories also failed to increase the precision of the estimates for all cases (not shown). Because stratification by neither geographic location (Figure 2.1) nor complexity (not shown) improved the precision of population estimates or indices, we present all subsequent results only for non-stratified samples.

The LPSEs demonstrated increased precision as compared to LP estimates (Figure 2.2). We also observed less variation in capture and snorkel counts as compared to the LP estimates (see below). In general, LPSEs and snorkel counts demonstrated the highest precision (i.e., lowest CV; Figure 2.2).

Table 2.2. Population estimates and indices, coefficient of variations (CV), and standard errors (SE) versus number of un-stratified sampled stream reaches for each size class of un-stratified sampled stream reaches (10,000 simulations) using a Lincoln-Petersen bias-adjusted estimator (LP), shrinkage Lincoln-Petersen bias-adjusted estimator (LPSE), counts of marked fish (M), and snorkeling counts of marked and unmarked (C) fish.

Method	Size class (mm)	No. of reaches sampled	Estimate	SE	CV	No. of reaches sampled	Estimate	SE	CV
Capture counts (M)	>120	10	1,092	315	0.29	41	1,090	156	0.14
Snorkel counts (C)	>120	10	1,748	489	0.28	41	1,750	242	0.14
LP[a]	>120	10	8,156	2,196	0.36	41	8,178	1,450	0.18
LPSE[b]	>120	10	5,644	1,232	0.22	41	5,650	609	0.11
Capture counts (M)	>220	10	294	71	0.24	41	293	35	0.12
Snorkel counts (C)	>220	10	767	128	0.16	41	768	61	0.08
LP[a]	>220	10	2,277	882	0.39	41	2,288	439	0.19
LPSE[b]	>220	10	1,286	203	0.16	41	1,287	100	0.08
Capture counts (M)	>370	10	92	31	0.33	41	92	15	0.17
Snorkel counts (C)	>370	10	225	67	0.30	41	225	33	0.15
LP[a]	>370	10	476	175	0.37	41	476	87	0.18
LPSE[b]	>370	10	297	70	0.23	41	298	34	0.12

[a] Lincoln-Petersen population estimate using the average population size across standardized reaches.
[b] Lincoln-Petersen population estimate similar to [a] but also using a random effects model (*see text*).

Estimated population sizes and abundance indices varied widely between the estimators and counts (Table 2.2). Because capture and snorkel count population indices do not account for resighting probability, all LP estimates (LP and LPSE) were greater than count data estimates. For example, for the > 120 group, the average SC estimate was 69% smaller than the average LPSE (1748 and 5644, respectively); however, for the > 370 group, this difference was only 25% (225 and 297, respectively). We attributed this result to size-dependent detection probability. That is, while most of the large bull trout were detected in the snorkel counts, a smaller portion of the > 120 group were detected. There were large differences between the LPSEs and the LP estimates (Table 2.2); LPSEs were much lower than the LP estimates. One reach had a population size of 16,836 fish (> 220 mm), with the next largest estimate being 8,687 fish, and 90% of all estimates were less than 3,700 fish. The sampling variance for the largest estimate was 29.8 times larger than 90% of all estimates, indicating very low precision for this estimate. The high value of this reach pulled up the mean for the LP estimates; in contrast, LPSEs are more robust to extreme values with high sampling variance and "shrink" these estimates toward the overall mean (Ver Hoef 1996; Burnham & White 2002), accounting for the lower population estimates.

Concordant with their increased precision, we observed the greatest power to detect declining trends with the LPSEs and snorkel counts (Figures 2.3, 2.4, and 2.5). Across all cases, we had consistently lower power to detect population declines with LP estimates and capture counts than when we used LPSE and snorkel counts. Overall, we found low power (< 80%) to detect a 25% decline in population size over 5 years for all population estimates and indices, even when 41 reaches were sampled (Figure 2.3). However, with the LPSE and snorkel counts, we were able to detect a 50% decline (with > 80% power) with a minimal number of reaches (10) sampled (Figure 2.3). With all methods, we observed high power (> 90%) to detect ≥ 75% declines.

When a longer, 15-year window was used, we achieved >90% power to detect a 75% population decline for all estimates (Figure 2.4). With the LPSEs, snorkel counts, and capture counts, we were able to detect a 50% decline with >90% power when a minimal number of reaches (10) were sampled; however, >15 reaches were required with the LP estimator (90% power). Using the LPSEs and snorkel counts, we were able to detect a 25% decline with > 80% power when 15 - 20 reaches were sampled, but still required ≥ 32 reaches with capture counts to achieve this power level. With LPs, we lacked the power to detect this decline, regardless of the sampling rate (10 - 41 reaches).

For the longest time frame, a 30-year window, our power to detect a 50 - 75% decline was high across all techniques (≥ 90%). When LPSEs or snorkel counts were used, there was ≥ 90% power to detect a 25% decline for any number of reaches sampled (Figure 2.5). With capture counts, we achieved > 90% power to detect a 25% decline when 24 reaches were sampled, while power to detect this drop remained low (< 80%) for the LP, even when 41 reaches were sampled.

DISCUSSION

Monitoring animal populations with abundance data can be difficult, as multiple sources of variability can reduce the precision of population estimates; consequently, the power to detect changes in population abundance is reduced (Boyce 2001; Williams et al. 2002) and the accuracy of persistence/extinction probability assessments (e.g., Staples et al. 2005). In this paper, we examined the use and efficacy of various techniques to monitor population abundance, the sampling variability component associated with abundance data and its effects on population monitoring, the statistical and design methodologies that can reduce the sampling variance component, and the sampling efforts that are required to detect changes in population abundance over time. Using empirical field data, we illustrated that both high levels of sampling effort and long temporal commitments to monitoring may be required to accurately detect changes in population abundance through time, and that formal assessments of population trends over short time intervals may not be possible for some species.

Variation for all population estimates and abundance indices of bull trout was relatively high (CV > 10%), with or without stratification, until more than 25 reaches were sampled. In addition to high spatial variation in bull trout abundance, high overall variation was also due, in part, to sampling variation. The precision of Lincoln-Petersen estimates is almost solely dependent on recapture rate (Seber 1982), which was relatively low for this study despite a high degree of effort (sampling rate of 46%). Recapture rate can also vary significantly among individuals (Zabel et al. 2005) and size classes (e.g. Thurow et al. 2006), which may have increased overall variation; as such, fine-resolution stratification may reduce this aspect of sampling variation. Finally, we suspect the closure assumption of the Lincoln-Petersen estimator was at least mildly violated by movement of bull trout in and out of reaches, which may have increased the sampling variation.

Shrinkage estimators performed well for this study, as the variance of the LPSEs was, on average, approximately 60% less than for the standard LP estimates. These results are consistent with the few other applications of shrinkage estimates available. Shrinkage estimators showed similar reductions of variance for count-based population estimates of eight of 10 species of waterfowl (Johnson 1989) and for Lincoln-Petersen population estimates of harbor seals (Ver Hoef 1996). Additionally, general theory has been developed for the application of shrinkage estimates to any set of maximum likelihood estimates for capture-recapture data (Burnham & White 2002). Shrinkage estimates are most advantageous when process (spatial in this case) and average sampling variation are approximately equal (Burnham & White

2002), but can be effective even when the proportion of the total variation attributed to sampling variation fluctuates significantly (Link & Nichols 1994). Generally, process variance, as extracted via the shrinkage technique, is the appropriate variance estimate to use for any population viability analysis (PVA) or predictive models (Link & Nichols 1994; Link 1999).

For the reasons above, we advocate the use of shrinkage estimators, while acknowledging two drawbacks. First, a minimum of at least five estimates of abundance are required to use a shrinkage estimator, and simulations suggest they become increasingly reliable when there are more than 10 estimates (Burnham & White 2002). Second, outside of the capture-recapture models included in Program MARK (White et al. 2002), the lack of available user-friendly computer programs remain a stumbling block (Ver Hoef 1996) for general use of shrinkage estimators. Nevertheless, our work suggests the shrinkage estimator, by accounting for sampling variation, allows for both increased precision of abundance estimates and reduction in the sampling effort required to detect changes in population abundance.

Focusing on certain size classes increased the precision of the abundance estimates for each monitoring technique. We found that the patchy distribution of both immature (< 220 mm) and large (> 370 mm) bull trout, the low numbers of large bull trout, and differences in sampling variability associated with each group reduced the overall precision of the estimates. The greatest precision occurred when the > 220 mm group was used to estimate population abundance; this class represents the sexually mature portion of the population (Al-Chokhachy et al. 2005), the size/age class of greatest concern for recovery efforts due to the importance of this class to the overall population growth (Caughley 1994; Shrimpton & Heath 2003). The differences in spatial variability across life stages illustrate that different effort levels and/or techniques may be necessary to for precise abundance estimates when monitoring different components of the population.

For our study area, stratification of sample units (reaches) did not increase the precision of our estimates, a surprising pattern contrary to statistical theory (Krebs 1999; Williams et al. 2002). However, this lack of improvement may be the result of the information used to designate reaches into different strata and high levels of variability within each stratum. We emulated a typical situation where little area-specific distribution information is available, and relied on qualitative assessments of habitat characteristics. A more robust approach would include a pilot study that assessed the variability in the distribution of the species and detailed habitat assessments of each reach, to guide subsequent stratification efforts and increase the precision of each abundance estimate (e.g., adaptive cluster sampling; Smith et al.

1995). Additionally, historical abundance data should not be ignored, as this information may be valuable to minimize variance through an adaptive sampling framework (e.g., Khaemba and Stein 2002) and maximize sampling efficiency within a monitoring plan.

Overall, despite the improved precision obtained with shrinkage estimates and by delineating the population into different size classes, both the LPSE and the count indices, still showed relatively high variation. This pattern likely reflects the high spatial variation (and some sampling variation) of bull trout in the SFWW; unremarkably, bull trout were clumped in space (e.g., Brown et al. 1995). Because of this variation, an extensive sampling effort is required to monitor the SFWW population effectively. For example, at longer time frames (e.g., 15 years), detecting a 25% population decline would only be possible if more than 15 reaches per year were sampled (18% of study area). For shorter time frames (e.g., five years), obtaining a 0.80 power of detecting a 25% population decline would require sampling well over 40 reaches (more than 45% of the study site). Similarly for Chinook salmon (*Oncorhynchus tshawytscha*), Petrosky et al. (2001) estimated a power of only 50% of detecting a 31% drop in productivity (measure of recruits/spawner) over 12 years, but when productivity dropped by 50%, their power to detect this change increased to over 80%. In contrast, for the bottlenose dolphin (*Tursiops* spp.), Wilson et al. (1999) estimated a 34% decline could be detected in only 8 years (0.90 power). These examples illustrate the difficulties associated with formal status evaluations performed over short time periods, if a large amount of natural variability exists in abundance across space and time (e.g., bull trout). Furthermore, our analysis only pertained to a single, local population of the Walla Walla core area, yet many species are monitored at a multiple-population level, which may further reduce our ability to detect changes in abundance across relevant time intervals.

In general, power analyses can be used to examine the tradeoff between high variation and increased sample size (Taylor & Gerrodette 1993). However, increased effort requires additional allocation of resources, and managers need to consider which methodology yields the most relevant estimate for their monitoring purposes. We estimate that mark-resight estimates require an average of 18 technician hours per reach, an effort nearly twice the average effort of snorkel counts (10 technician hours per reach). Clearly, snorkel counts are more economical, yet count data is often biased, provides only an index of population size (Link & Sauer 1998; Thurow et al. 2006), and often does not describe the magnitude of the population or provide additional population-level information (e.g., vital rates), which are often critical for viability analyses (Boyce 1992; Caughley 1994). On the contrary, mark/recapture methods, albeit more costly here, can provide estimates of abundance and ultimately population trends, and allow for estimates of vital rates (e.g., Lebreton et al. 1992). Ultimately, both variation in field data and sampling costs must be considered when determining the most efficient sampling and monitoring design for a species.

Based on our observations, a robust, yet economically feasible approach for sampling this bull trout population would include some combination of techniques. A high degree of effort could be used to obtain mark-recapture data and snorkel counts for one population in a basin, thus providing information regarding the bias between the index data and population size, and minimal efforts of the more affordable technique (e.g., snorkel counts) could be used for the other populations in the basin. This approach would be particularly effective for basins where spatial synchrony exists among populations (e.g., Isaak et al. 2003). Clearly this approach would require some understanding of the temporal variation among populations, but where possible, may allow for widespread monitoring of populations using more affordable techniques.

Effective population monitoring requires precise abundance estimates or indices through time, which can require a significant amount of effort and monetary resources (Williams et al. 1999). Our analyses for bull trout demonstrated that the precision of abundance estimates, and subsequently power, were substantially reduced due to natural, process variation, which here only included spatial variability; with the added effect of temporal variability, the power to detect changes in abundance would likely be further reduced. For any species, robust monitoring with abundance data may not be possible over short time intervals, and alternative monitoring techniques, including the temporal symmetry model (Pradel 1996), which can increase the precision of estimates of population change (Sandercock & Beissinger 2002), and risk-based monitoring, which allows for more proactive population monitoring (Staples et al. 2005), should be considered as viable alternatives when appropriate. Regardless of the technique, historical and/or pilot-study data should be integrated into formal sampling-efficiency analyses, as a means to maximize the information gained with limited resources, and monitoring and evaluation programs for imperiled species should be carefully evaluated before full implementation.

LITERATURE CITED

Al-Chokhachy, R., P. Budy, and H. Schaller. 2005. Understanding the significance of redd counts: a comparison between two methods for estimating the abundance of and monitoring bull trout populations. North American Journal of Fisheries Management **25**:1505-1512.

Boyce, M.S. 1992. Population viability analysis. Annual Review of Ecology and Systematics **23**:481-506.

Boyce, M.S. 2001. Population viability analysis: development, interpretation, and application. Pages 123-136 *in* T. Shenk and A. Franklin, editors. Modeling in natural resource management. Island Press, Washington, D.C.

Brown, J.H., D.W. Mehlman, and G.C. Stevens. 1995. Spatial variation in abundance. Ecology **76**:2028-2043.

Burnham, K. P., D. R. Anderson, G. C. White, C. Brownie, and K. H. Pollock. 1987. Design and analysis methods for fish survival experiments based on release-recapture. American Fisheries Society Monograph 5. American Fisheries Society, Bethesda, Maryland.

Burnham, K. P., and G. C. White. 2002. Evaluation of some random effects methodology applicable to bird ringing data. Journal of Applied Statistics **29**:245–264.

Caughley, G. 1994. Directions in conservation biology. Journal of Animal Ecology **63**:215-244.

Chapman, D. G. 1951. Some properties of the hypergeometric distribution with applications to zoological censuses. University of California Publications in Statistics **1**:131–160.

Cochran, W. G. 1977. Sampling techniques. Third edition. John Wiley & Sons, Inc., New York, New York.

Crowder, L.B., D.T. Crouse, S.S. Heppell, and T.H. Martin. 1994. Predicting the impact of turtle excluder devices on loggerhead sea turtle populations**.** Ecological Applications **3**:437-445.

Dennis, B., P. L. Munholland, and M. Scott. 1991. Estimation of growth and extinction parameters for endangered species. Ecological Monographs **61**:115-143.

Fraley, J.J. and B.B. Shepard. 1989. Life history, ecology, and population status of migratory bull trout (*Salvelinus confluentus*) in the Flathead Lake and River system, Montana. Northwest Science **63**:133-143.

Gibbs, J.P., S.Droege, and P. Eagle. 1998. Monitoring populations of plants and animals. Bioscience **48**: 936-940.

Ham, K.D. and T.N. Pearsons. 2000. Can reduced salmonid population abundance be detected in time to limit management impacts? Canadian Journal of Fisheries and Aquatic Sciences **57**:17-24.

Hankin, D.G. 1984. Multistage sampling designs in fisheries research: applications in small stream. Canadian Journal of Fisheries and Aquatic Sciences **41**:1575-1591.

Hilborn, R. and C.J. Walters. 1992. Quantitative fisheries stock assessment. Chapman and Hall. New York, New York.

Isaak, D.J., R.F. Thurow, B.E. Rieman, and J.B. Dunham. 2003. Temporal variation in synchrony among chinook salmon (*Oncorhynchus tshawytscha*) redd counts from a wilderness area in central Idaho. Canadian Journal of Fisheries and Aquatic Sciences **60**:840-848.

Johnson, D. J. 1989. An empirical Bayes approach to analyzing recurring animal surveys. Ecology **70**:945–952.

Kendall, K.C., L.H. Metzgar, D.A. Patterson, and B.M. Steele. 1992. Power of sign surveys to monitor population trends. Ecological Applications **2**:422-430.

Khaemba, W.M. and A. Stein. 2002. Improved sampling of wildlife populations using airborne surveys. Wildlife Research **29**:269-275.

Krebs, C.J. 1999. Ecological methodology. Benjamin-Cummings. Menlo Park, California.

Leary, R.F., F.W. Allendorf, and S.H. Forbes. 1993. Conservation genetics of bull trout in the Columbia and Klamath River drainages. Conservation Biology **7**:856-865.

Lebreton, J.D., K.B. Burnham, J. Clobert, and D.R. Anderson. 1992. Modeling survival and testing biological hypotheses using marked animals: a unified approach with case studies. Ecological Monographs **62**:67-118.

Link, W. A. 1999. Modeling pattern in collections of parameters. Journal of Wildlife Management **63**:1017–1027.

Link, W. A., and J. D. Nichols. 1994. On the importance of sampling variance to investigations of temporal variation in animal population size. Oikos **69**:539–544.

Link, W A., and J. R. Sauer. 1998. Estimating population change from count data: Application to the North American Breeding Bird Survey. Ecological Applications **8**:258-268.

Maxell, B.A. 1999. A power analysis on the monitoring of bull trout stocks using redd counts. North American Journal of Fisheries Management **19**:860-866.

Minta, S. and M. Mangel. 1989. A simple population estimate based on simulation for capture-recapture and capture-resight data. Ecology **70**:1738-1751.

Morris, W.M. and D.F. Doak. 2002. Quantitative conservation biology. Sinauer Associates Inc. Sunderland, Massachusetts.

Petrosky, C.E., H.A. Schaller, and P. Budy. 2001. Productivity and survival rate trends in the freshwater spawning and rearing stage of Snake River chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences **58**:1196-1207.

Pimm, S.L., and A. Redfearn. 1988. The variability of animal populations. Nature **338**:628.

Pradel, R. 1996. Utilization of mark-recapture for the study of recruitment and population growth rate. Biometrics **52**:703-709.

Rieman, B.E. and J.D. McIntyre. 1993. Demographic and habitat requirements for conservation of bull trout. USDA General Technical Report INT-302.

Rieman, B.E. and J.D. McIntyre. 1995. Occurrence of bull trout in naturally fragmented habitat patches of varied size. Transactions of the American Fisheries Society **124**:285-296.

Rieman, B.E., D.C. Lee, and R.F. Thurow. 1997. Distribution, status, and the likely future trends of bull trout within the Columbia River and Klamath River basins. North American Journal of Fisheries Management **17**:1111-1125.

Sandercock, B.K., and S.R. Beissinger. 2002. Estimating rates of population change for a neotropical parrot with ratio, mark-recapture, and matrix methods. Journal of Applied Statistics **29**:589-607.

SAS Institute. 1999. SAS OnlineDoc®, Version 8. SAS Institute Inc., Cary, North Carolina.

Seber, G. A. F. 1970. The effects of trap response on mark-recapture estimates. Biometrika **26**:13–22.

Seber, G. A. F. 1982. The estimation of animal abundance and related parameters. Second Edition. Edward Arnold, London, England.

Shrimpton, J.M. and D.D. Heath. 2003. Census vs. effective population size in chinook salmon: large- and small-scale environmental perturbation effects. Molecular Ecology **12**:2571-2583.

Staples, D.F., M.L. Taper, and B. Dennis. 2004. Estimating population trend and process variation for PVA in the presence of sampling error. Ecology **85**:923-929.

Staples, D.F., M.L. Taper, and B.B. Shepard. 2005. Risk-based viable population monitoring. Conservation Biology **19**:1908-1916.

Smith, D.R., M.J. Conroy, and D.H. Brakhage. 1995. Efficiency of adaptive sampling for estimating density of wintering waterfowl. Biometrics **51**:777-788.

Taylor, B.L, and T. Gerrodette. 1993. The uses of statistical power in conservation biology: the vaquita and northern spotted owl. Conservation Biology **7**:489-500.

Thompson, W. L., G. C. White, and C. Gowan. 1998. Monitoring vertebrate populations. Academic Press, San Diego, California.

Thurow, R.F., J.T. Peterson, and J.W. Guzevich. 2006. Utility and validation of day and night snorkel counts for estimating bull trout abundance in first- to third-order streams. North American Journal of Fisheries Management **26**:217-232.

Ver Hoef, J. M. 1996. Parametric empirical Bays methods for ecological applications. Ecological Applications **6**:1047–1055.

Wada, T., and L.D. Jacobson. 1998. Regimes and stock-recruitment relationships in Japanese sardine (*Sardinops melanostictus*), 1951-1995. Canadian Journal of Fisheries and Aquatic Sciences **55**:2455-2463.

White, G. C. 2000. Population viability analysis: data requirements and essential analysis. Pages 288–331 *in* L. Boitani and T. K. Fuller, Editors, Research Techniques in Animal Ecology: Controversies and Consequences. Columbia University Press, New York, New York.

Whitesel, T.A. and 7 coauthors. 2004. Bull Trout Recovery Planning: A review of the science associated with population structure and size. Science Team Report # 2004-01, U.S. Fish and Wildlife Service, Regional Office, Portland, Oregon, USA.

Williams, B. K., J. D. Nichols, and M. J. Conroy. 2002. Analysis and management of animal populations. Academic Press, San Diego, California.

Wilson, B., P.S. Hammond, and P.M. Thompson. 1999. Estimating size and assessing trends in a coastal bottlenose dolphin population. Ecological Applications **9**:288-300.

Zabel, R.W., T. Wagner, J.L. Congleton, S.G. Smith, and J.G. Williams. 2005. Survival and selection of migrating salmon from capture-recapture models with individual traits. Ecological Applications **15**:1427-1439.

Zielinski, W.J. and H.B. Stauffer. 1996. Monitoring *Martes* populations in California: survey design and power analysis. Ecological Applications **6**:1254-1267.

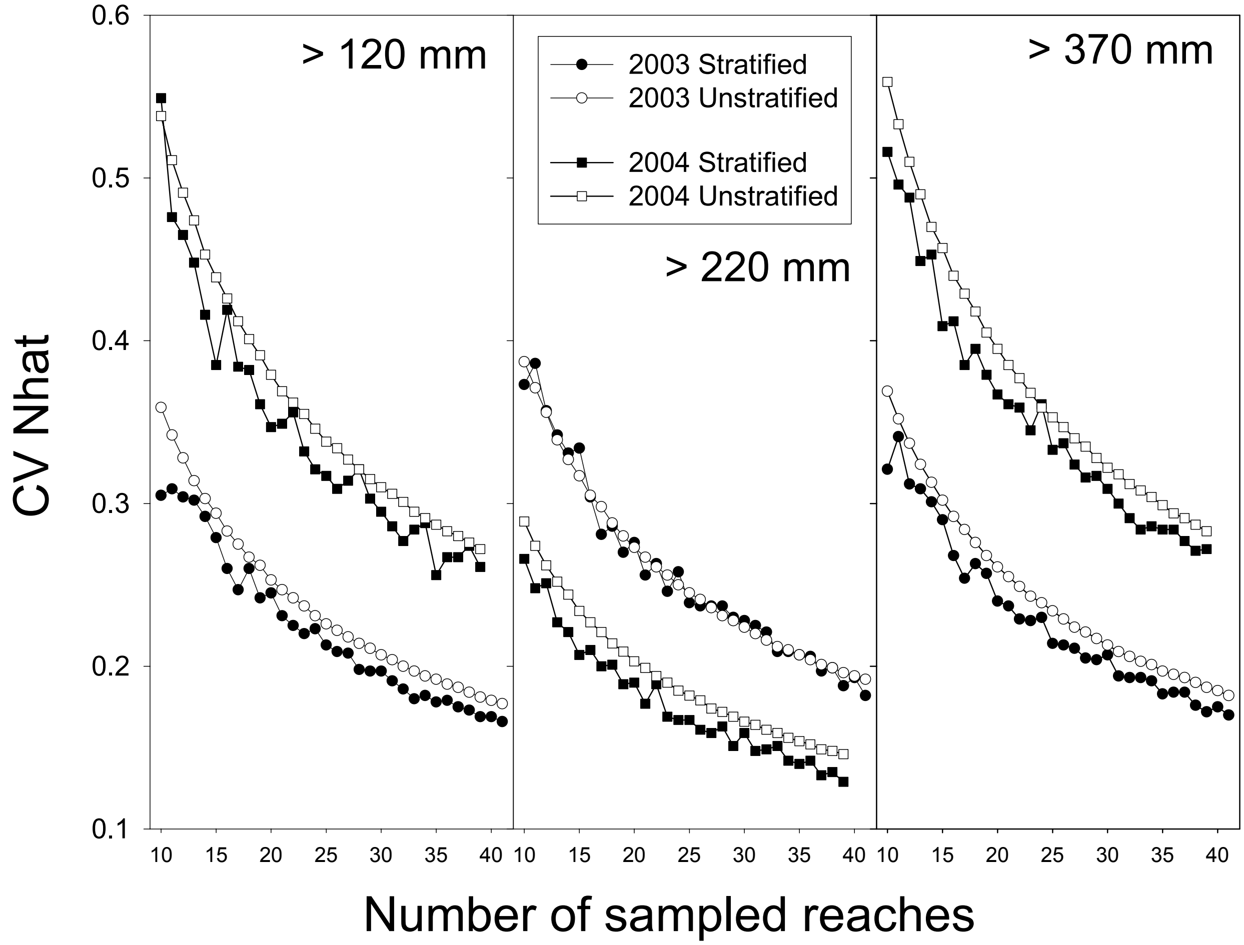

Figure 2.1. Estimated coefficient of variation versus number of sampled stream reaches for Lincoln-Peterson bias-adjusted population size estimators for bull trout of (1) all size classes (> 120 mm), (2) > 220 mm bull trout, and (3) > 370 mm bull trout in the South Fork Walla Walla River, Oregon, 2003 (circles) and 2004 (squares). Coefficient of variation estimated from the mean of 10,000 simulations for unstratified reaches and variance weighted means for the stratified reaches. Reaches were stratified by river elevation which was classified as upper, middle, and lower.

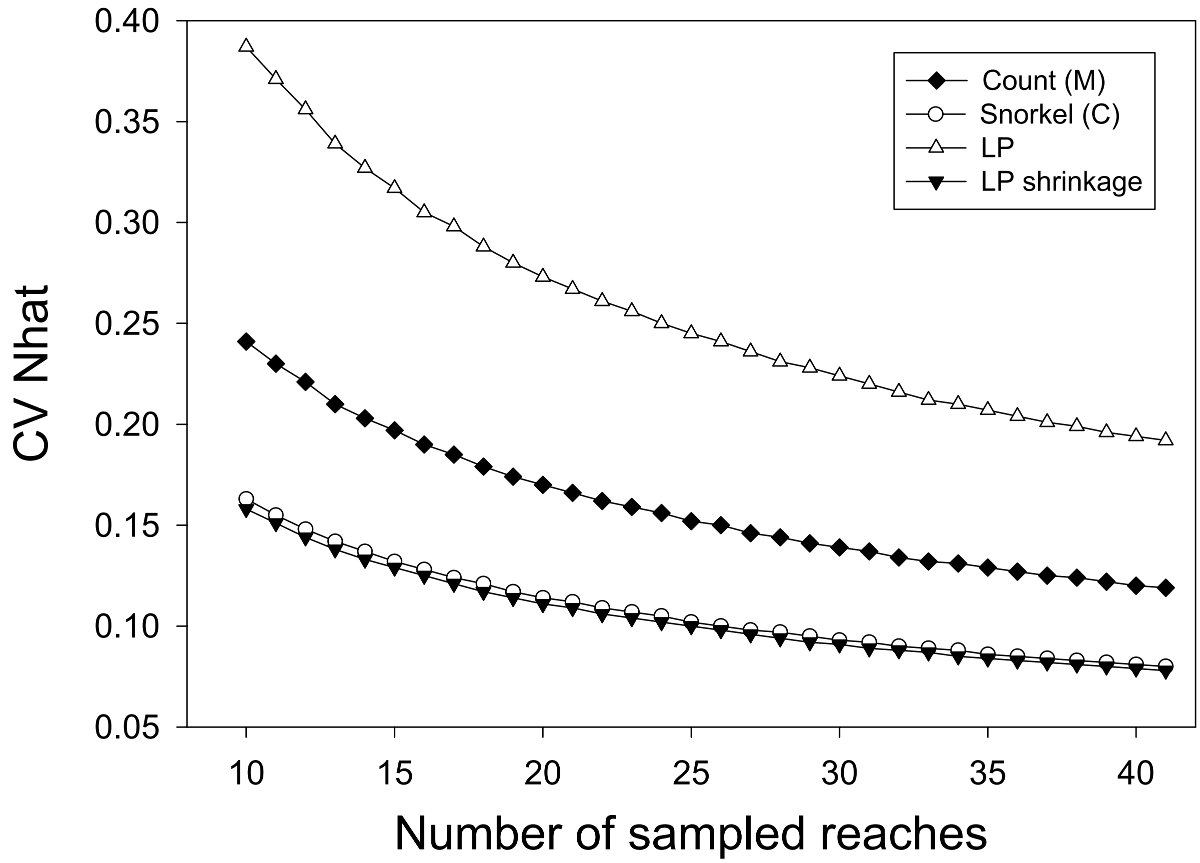

Figure 2.2. Estimated coefficient of variation versus number of unstratified sampled stream reaches for >220 mm bull trout using a Lincoln-Petersen bias-adjusted estimator (LP), shrinkage Lincoln-Petersen bias-adjusted estimator (LP shrinkage), counts of marked (Count M) fish, and snorkeling counts of marked and unmarked (Snorkel C) fish. Data from South Fork of the Walla Walla River, Oregon, 2003. Coefficient of variation estimated from the mean of 10,000 simulations.

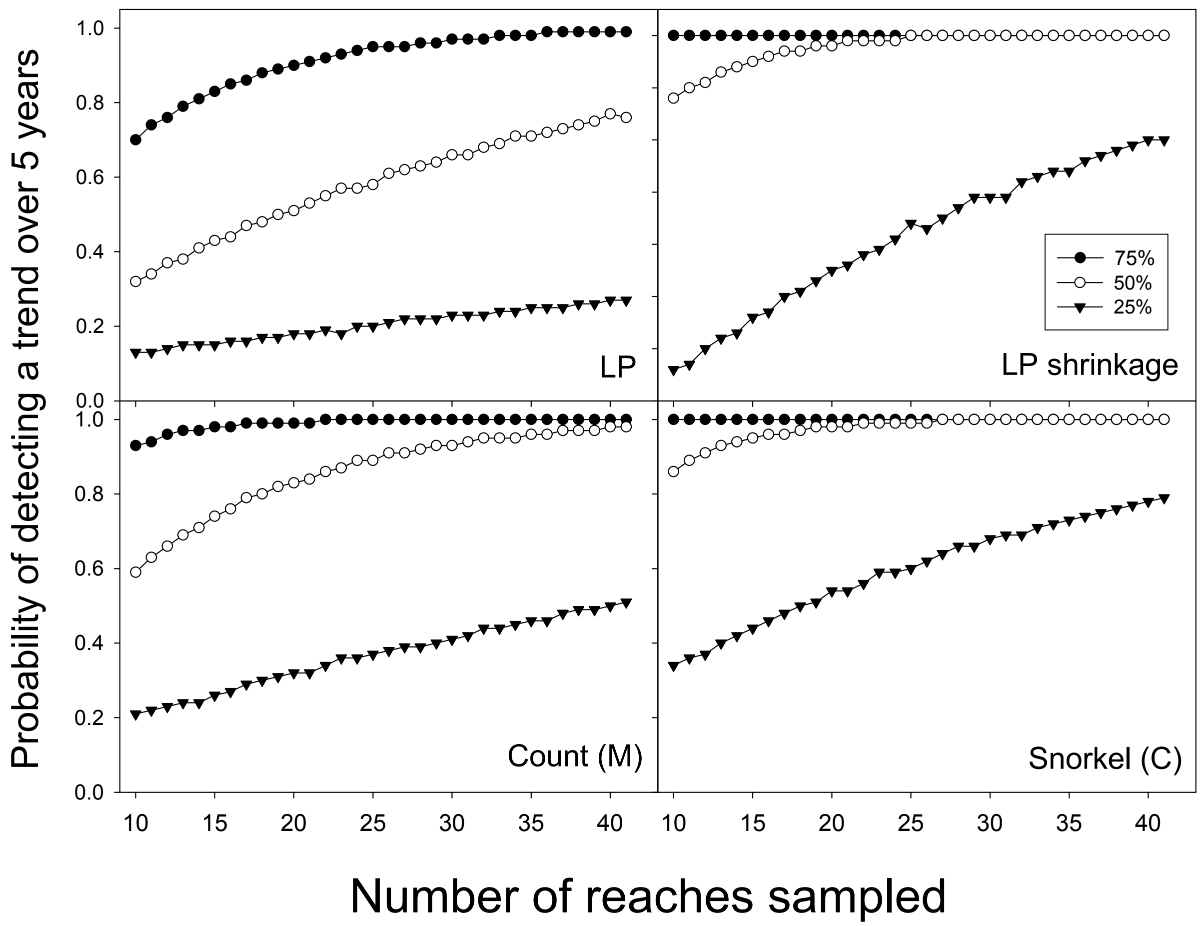

Figure 2.3. Probability of detecting a declining trend over 5 years for a 75%, 50%, and 25% decline in bull trout population size. Results based on the mean of 10,000 simulations for unstratified stream reach samples using a Lincoln-Peterson bias-adjusted population size estimator (LP), shrinkage Lincoln-Peterson bias-adjusted population size estimator (shrinkage LP), counts of marked fish (Count M), and snorkel counts of marked and unmarked fish (Snorkel C). Data from South Fork Walla Walla River, Oregon, 2003.

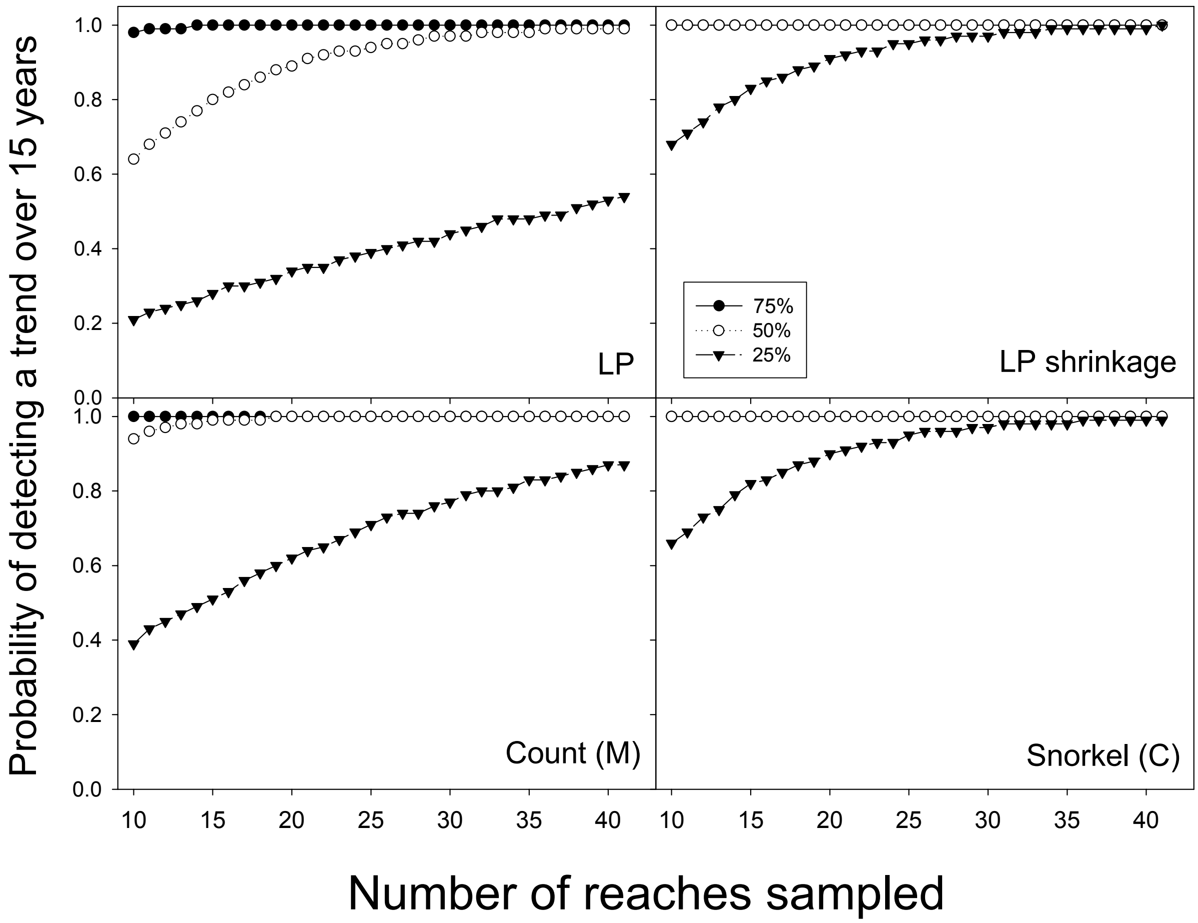

Figure 2.4. Probability of detecting a declining trend over 15 years for a 75%, 50%, and 25% decline in bull trout population size. Results based on the mean of 10,000 simulations for unstratified stream reach samples using a Lincoln-Peterson bias-adjusted population size estimator (LP), shrinkage Lincoln-Peterson bias-adjusted population size estimator (shrinkage LP), counts of marked fish (Count M), and snorkel counts of marked and unmarked fish (Snorkel C). Data from South Fork Walla Walla River, Oregon, 2003.

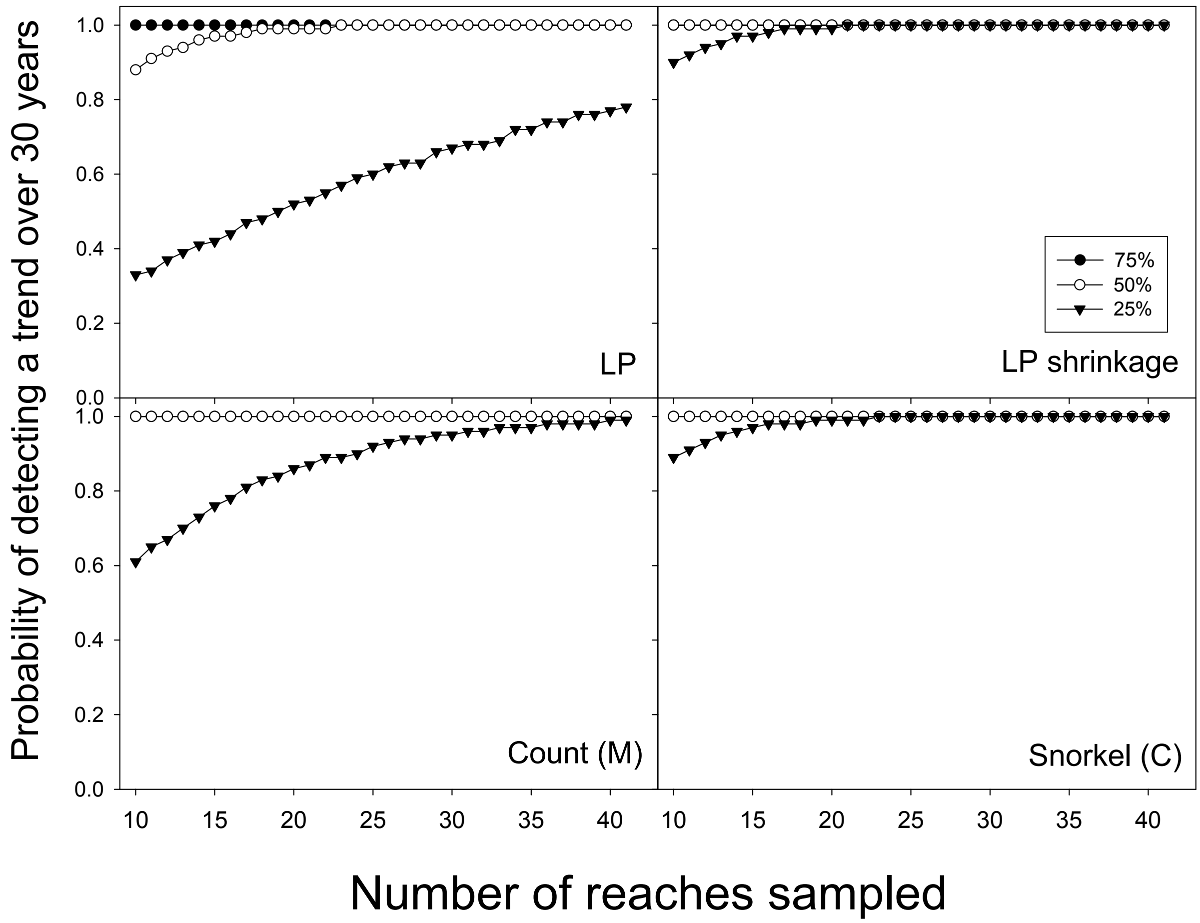

Figure 2.5. Probability of detecting a declining trend over 30 years for a 75%, 50%, and 25% decline in bull trout population size. Results based on the mean of 10,000 simulations for unstratified stream reach samples using a Lincoln-Peterson bias-adjusted population size estimator (LP), shrinkage Lincoln-Peterson bias-adjusted population size estimator (shrinkage LP), counts of marked fish (Count M), and snorkel counts of marked and unmarked fish (Snorkel C). Data from South Fork Walla Walla River, Oregon, 2003.

CHAPTER 3:
The movement continuum: evaluating migration patterns of subadult bull trout in northeast Oregon

INTRODUCTION

In an environment characterized by instability or degradation, populations that contain both resident and migratory individuals are better able to persist in the face of change (Northcote 1992). These movement strategies are of particular importance as they represent evolutionary diversity that has allowed fish to adapt to, and take advantage of, various resources in the environment (Dingle 1996), and furthermore may be negatively impacted by changes to that environment (Schlosser 1991; Quinn and Adams 1996). For imperiled species in particular, it is critical to identify these multiple behavioral strategies in order to determine the patch size and connectivity requirements of different behavioral or life-history forms. For example, highly mobile fish (e.g., anadromous sockeye salmon *Oncorhynchus nerka*) may move distances greater than 900 km, utilizing disparate habitat patches and migratory corridors, while resident forms of the same species (e.g., kokanee salmon, *O. nerka*) may spend their entire life in a single lake (Groot and Margolis 1991). In the case of morphologically indistinguishable forms, it can be difficult to determine what proportion of a population is mobile, and the extent of that mobility, both factors that affect recovery planning. Ultimately, these diverse life history forms are important to population persistence as they 1) disperse population level mortality risk via occupation of multiple habitat patches through time, 2) act as a vehicle for gene flow and 3) can re-found unoccupied habitat patches (Gross 1991; Jackson et al. 2001).

The movement patterns of salmonids have been widely studied. Historic migration patterns of Pacific salmon are believed to have occurred on a spatial and temporal continuum before populations were severely exploited, and impoundments altered flow regimes and decreased connectivity. In contrast, current Pacific salmon migrations tend to occur during discrete time periods (e.g., seasons), are stock specific (e.g., spring Chinook salmon *O. tshawytscha*), and of duration related to the specific strategy employed (fluvial, adfluvial, anadromous; Groot and Margolis 1991). Other salmonids (e.g., char species, Nordeng 1983; or cutthroat trout *O. clarkii*, Schrank and Rahel 2004) demonstrate much more variable migration patterns in terms of the timing and distance of migration. In addition, these fish may switch seasonally or annually from a migratory tactic to a resident one (Hilderbrand and Kershner 2000).

Within the family Salmonidae, not only are the patterns of migration varied, but also the cues that prompt migration. Temperature has been associated with the downstream dispersal of fry (Whalen et al. 1999), and spring and autumn movements (Swanberg 1997; Jonsson and Jonsson 2002; but see also Bjornn 1971), while discharge has been associated with movement timing for multiple age classes (Downs et al. 2006; Quinn and Adams 1996; but see also Bjornn 1971). The seasonal and diel timing of smolting (Byrne et al. 2003; Thorpe and Morgan 1978), and migration (Riley et al. 2002; Muhlfeld et al. 2003) have been associated with photoperiod, and additionally, photoperiod may define the time period in which migration occurs (McCormick et al. 1998). Finally, many of these studies have also alluded to the possibility that precipitation may cue movement (Salow 2005). This diverse array of migration cues illustrates how the role of environmental variability or heterogeneity may result in differential migration responses across the range of a species, or between species, and further necessitates a detailed understanding of these migration cues in order to thoroughly identify movement patterns.

Bull trout are a species of char native to the Pacific Northwest that exhibits a complex array of movement patterns. Throughout their range, bull trout co-occur in resident and migratory (e.g., fluvial, adfluvial, anadromous) forms (Rieman and Dunham 2000). Adult resident fish may be 150 - 300 mm, while adult migratory fish may grow to well over 600 mm (Fraley and Shepard 1989). Bull trout are behaviorally cryptic, require cold, clean water, and have been associated with complex habitat (Rieman and McIntyre 1993). Many factors such as habitat degradation and fragmentation, the introduction of non-native fish, and migration barriers have contributed to range-wide declines, particularly to the migratory form (Nelson et al. 2002). In 1999, bull trout were listed as threatened in the conterminous United States (DOI 1999).

Migratory bull trout (fluvial and adfluvial) exhibit movement patterns across broad temporal and spatial scales (2 - 250 km, Fraley and Shepard 1989, Swanberg 1997, Baxter 2002), and in association with many cues. Both adfluvial and fluvial adults initiate spawning migrations to their natal stream in the late spring or summer as temperatures approach 10 - 12 °C, and the hydrograph decreases (Goetz 1989; Elle and Thurow 1994), and migrate out of the system (post-spawn) as temperatures decrease in the autumn (Fraley and Shepard 1989; Flatter 2000; Hostettler 2004). Some adult fish may holdover in the natal stream and migrate out the following spring (personal observation), but generally little movement is observed during the winter unless the presence of anchor ice, or harsh river conditions, displace fish (Jakober et al. 1998; Hostettler 2003). These migrations tend to occur over discrete time periods that vary across basins (e.g., Fraley and Shepard 1989; Swanberg 1997).

In contrast to adult movement patterns, the migratory patterns and associated cues of bull trout < 300 mm TL (both fluvial and adfluvial) are much less understood. Most subadult bull trout migrate at age-2, although fry or age-3 fish may also migrate (Pratt 1992). However, the distance and rate of those migrations may vary considerably both with body size (Hostettler 2004), and changes in discharge or temperature (Salow 2005; Downs et al. 2006). In addition, these variables may affect young-of-year or juvenile fish differentially (Downs et al. 2006). While studies employing radio telemetry or weirs have been able to identify some specific movement patterns of subadult fish, typically the associated small sample size (limited to fish large enough to carry a transmitter) and sample season (for telemetry and weirs respectively) have precluded the formal testing of cues associated with annual patterns of migration. As these cues are of particular conservation interest, it is critical to identify how these may vary both within and across seasons.

In this study, our goal was to evaluate and understand the movement patterns of migratory subadult bull trout, in order to better understand the time frame in which that movement occurs, the extent of movement, and potential factors that may prompt the initiation of downstream movement. While the distinction between migration and movement has been discussed extensively (e.g., Dingle 1996), that is not the focus of this research. Therefore, for the purpose of this study we define migratory movements (migration hereafter) as movements greater than 2 km (Jakober 1992) in a downstream direction. We combined active mark-recapture techniques with passive PIT-tag detection to: 1) monitor the daily, seasonal, and annual movements of subadult bull trout, 2) determine the timing of first downstream migration, and 3) identify potential cues that may prompt this migration.

STUDY AREA

The South Fork Walla Walla River (SFWW) is a second-order stream that drains out of the Blue Mountains of Northeast Oregon. It originates at elevations near 1800 M and flows roughly 40 km before approaching Milton-Freewater, Oregon. Within the SFWW, habitat conditions are generally high quality, with little forest management, and limited recreational activity (particularly in the headwaters). At base flow, the average width of the SFWW is approximately 10 m. Downstream of the confluence with the North Fork Walla Walla River, the habitat conditions degrade with respect to increased water temperature, simplified channel and habitat, impoundments, and irrigation withdrawals that severely deplete flow. The SFWW confluences with the Columbia River just upstream of McNary Dam. This is an arid climate, and most precipitation falls as snow. Bear Creek, Skiphorton Creek, Reser Creek, and the upper SFWW are the major tributaries to the SFWW. Most observed spawning activity occurs in proximity to these tributaries.

The SFWW contains wild populations of steelhead (*Oncorhynchus mykiss*), redband trout (*O. mykiss* subspecies), mountain whitefish (*Prosopium williamsoni*), sculpin (*Cottus* spp.), and bull trout (*Salvelinus confluentus*). Since 2000, the Confederated Tribes of the Umatilla Indian Reservation has annually supplemented populations of adult Chinook salmon (*O. tshawytscha*).

METHODS

<u>Study design and overall project</u>

This work is part of a larger research effort aimed at creating a general template for recovery planning of bull trout across their range (see Chapter 1). For this larger study, we conducted a mark-recapture-resight study to evaluate population size and structure. To this end, we set the lower bound of our study site at Harris Park, and the upper bound 21 km upstream at the confluence with Reser Creek (Figure 3.1), and divided the study site into 103 reaches of approximately 200 m each. These reaches were adjacent to each other and their location was geo-referenced (e.g., the distance between reach 10 and reach 20 is the same distance as between reach 30 and 40). Each year (2002 - 2005), we systematically sampled 20 index reaches and up to 20 additional reaches from a random starting point selected from within the first five reaches (Budy et al. 2004).

Fish capture and marking.—As part of a larger study, we used multiple techniques to capture fish including: electrofishing to a seine, trap netting, baited minnow traps, angling, and snorkeling to a seine. All captured fish were weighed, measured, and scanned for PIT tags. We anesthetized fish > 120 mm, and once fish were unresponsive to stimuli, we made a 3-mm incision in the ventral cavity and implanted a 23-mm PIT tag, and inserted an external T-bar anchor tag adjacent to the dorsal fin for mark-resight analysis (Budy et al. 2005). Post implantation, we held fish in a flow-through recovery tank until they reached full equilibrium. We released fish in slow water close to the point of capture. From 2002 - 2005, we captured and tagged 1636 bull trout.

Quantifying movements.—Along with using active recapture to annually locate fish, we monitored movement timing and direction of our tagged fish via passive PIT-tag detectors on the SFWW; one at the bottom of our study reach (Harris Park) and one approximately 7 km upstream (Bear Creek; Figure 3.1). These PIT-tag detectors were installed in 2002, and recorded the PIT-tag number of marked fish that swam through the detector. Based on these detections, we inferred movement direction for all fish that swam through both detectors, or for fish that swam through a single detector after active capture. These detections were the basis for determining which component of the population was migratory, and establishing the time frame and distance over which that migration occurred. We used information from active recapture and both detectors to determine the minimum distance that a fish moved (based on the location that a fish was initially captured and the furthest point that it was eventually detected).

Environmental and biological variables.—We measured potential environmental and biological variables at both detector sites that could serve as potential cues for movement. We recorded daily stream temperature from 2002 - 2004 using temperature loggers (set to record every 90 minutes) and from 2004 - 2005 based on data from a gauging station at the Harris Park detector site. The gauging station recorded stream temperature continuously from November 2002 every hour. We obtained precipitation and photoperiod data from local USGS gauging stations (High Ridge SNOTEL site), and validated the photoperiod in-stream with a light meter). For this study, we defined "day" as the hours of visible light (approximately one hour before sunrise to one hour after sunset). In addition, we measured stream discharge at the lower detector site in 2004 and 2005 using a magnetic flow meter; this information corroborated gauge height measurements recorded electronically and continuously at the same site. Finally, we considered whether the presence of spawning adults might cue migration. We monitored the upstream and downstream (spawning-related) migrations of large fish in our system to determine the time frame in which they could potentially influence the migration timing of subadults. When we evaluated environmental and biological correlates with migration timing, we included only migration past Harris Park bridge, the location where these variables were measured.

Data Analysis

We constructed twelve models based on *a priori* hypotheses of variables thought to cue movement. These models quantified the number of migrants per unit time (10 days) in response to a combination of environmental and biological variables, and interactions between variables (Table 3.1). We chose a 10-day unit of time to maximize differences in environmental conditions between time bins while simultaneously maintaining the most precise level of measurement of migrants per unit time, and performing the least complicated statistical transformations to normalize the data. These models were formally tested using linear regression techniques and ranked according to Akaike Information Criteria (AIC) based on parsimony, where more-complicated models are penalized for the inclusion of additional parameters (Burnham and Anderson 2002; SAS Institute 2005). Once we determined our top five annual models, we evaluated these models by season and determined the top model for each season. Seasonal models were only compared to other models from the same season (e.g., each winter model was compared with only winter models) due to sample size variability across seasons (Table 3.1).

Table 3.1. Top five annual models and top four seasonal models of the influence of environmental and biological cues on the number of subadult bull trout migrants per 10-day period. Min = minimum temperature, Q = discharge, and Precip = precipitation as rainfall. NA means not applicable.

Model	Intercept	Min	Q	Precip	Adults	Model *P*	*F*-Value	Adj. R^2	AIC	Δ AIC
Annual										
1	0.42	0.07	-	-	-	0.11	2.60	0.03	-41.98	0
2	0.30	0.09	-	-	-0.11	0.13	2.11	0.04	-41.65	0.33
3	-670.79	0.08	0.34	-	-	0.22	1.58	0.02	-40.59	1.39
4	0.90	-	-	-0.51	-	0.34	0.94	0.00	-40.32	1.66
5	0.50	0.06	-	-0.26	-	0.26	1.39	0.02	-40.22	1.76
Seasonal										
Autumn	-3748.23	0.22	1.9	-	-	0.02	4.88	0.31	-18.00	NA
Winter	1.62	-0.32	-	-	-	0.06	5.02	0.31	-17.13	NA
Spring	2.87	-0.29	-	-	-	0.05	5.31	0.35	-12.40	NA
Summer	-4.01	0.64	-	-	-0.22	0.13	2.61	0.23	- 7.59	NA

In addition to identifying which variables might predict migration timing, we were also interested in understanding how the number of migrants might fluctuate differentially (but not predictively) with certain levels of variables. Thus, we performed a chi-squared test on all variables that were insignificant in our formal model testing. We tested the alternative hypothesis that the number of migrants detected per environmental bin was disproportionate to the availability of that bin in the environment, against a null hypothesis that migrants used all environmental bins in proportion to their availability.

We binned all environmental variables by level (e.g., discharge levels in 1/10th of a foot, ~ 3 cm, gauge height), and evaluated the number of migrations that occurred in each bin. All variables had unequal bin size (e.g., precipitation), so we identified the proportion of time that each level of the variable occurred in the environment, and used that proportion to determine our expected values (e.g., the proportion of time that low discharge occurs in the system throughout the year).

RESULTS

Movement timing.—Bull trout expressed annual and diel variation in movement patterns. Adult fish migrated upstream into the study area in July, and downstream, post-spawn, in September (Figure 3.2). Conversely, subadult fish outmigrated throughout the entire year, with an initial downstream pulse detected in the spring, and a larger pulse detected in August (Figure 3.3). In addition, most fish (97%) migrated at night (Night = 143, Day = 9; Figure 3.4).

Movement distance.—We detected bull trout movements through active recapture during the sampling season, and annual passive detection at the Bear Creek and Harris Park detectors. In addition, some of our tagged fish were detected at the Nursery Bridge Dam detector (Figure 3.1), 24 km below Harris Park. Minimum movement distances ranged from 0 - 45 km (Figure 3.5). Eighteen of our actively recaptured fish were also detected passively at a detector site. When we evaluated the minimum distance these dual- detected fish moved, based solely on active recapture, ten fish moved 0 - 2 km, three moved 2 - 5 km, three moved 5 - 10 km and two moved > 10 km. When those same movement distances were described only using passive detection (in conjunction with initial capture site), two fish moved 0 - 2 km, seven moved 2 - 5 km, three moved 5 - 10 km, and six moved more than 10 km. In general, active recapture alone underestimated the minimum distance these fish moved.

Environmental and biological variables.—Maximum water temperatures in the study area over all years was 15.7 ºC, and minimum temperatures approached, but rarely fell below, 0 ºC. Flows peaked in the late spring (602.4 m gauge height - corresponds to ~160 cfs or 4.5 m^3/s), concurrent with snow-melt run-off, and had a smaller peak in December (602.1 m) related to precipitation. In 2004, the system reached base flow in late July (601.9 m gauge height - corresponds to ~90 cfs or 2.5 m^3/s) while in 2005, the system reached base flow in early June. The SFWW received 44.19 cm of precipitation (primarily in January and February) in 2004, and 86.86 cm (primarily in March and April) in 2005.

Influence of cues on movement.—Prior to running our models, we assessed normalcy and homoskedasticity; we used a square root transformation to meet our assumptions of normalcy. While our top four models were not significantly different based solely on

AIC criteria, our simplest migration cue model (migrants per 10-day period = min temp) was our top model according to significance (R^2 = 0.05, p = 0.11; Figure 3.6); we subsequently re-ran our top five models with migration patterns separated into four seasons. Our top model remained the same, but the significance and explanatory power were improved for three of four seasons (Winter R^2 = 0.38, p = 0.05, Spring R^2= 0.43, p = 0.05, Autumn R^2= 0.17, p = 0.08). Seasonal partitioning changed the nature of the relationship between min temp and migration timing from a positive relationship in the annual model and in the spring, to a negative relationship in the autumn and winter.

Association of environmental variables and movement.—When we tested the null hypothesis that the number of migrants detected per environmental bin was disproportionate to the availability of that bin in the environment, we found that migrants were not distributed evenly across photoperiod bins, but that distribution was variable and there was no clear preference for any level of photoperiod (binned in 10-minute increments, χ^2 = 48.07, df = 42, P < 0.05). While most movements occurred in association with low discharge and low precipitation, fish moved during these conditions in the same proportion that they occurred in the environment.

DISCUSSION

We monitored the movements of over 1500 individually tagged bull trout using both passive and active techniques in order to gain a better understanding of bull trout movement patterns and the factors that cue migration. Subadult bull trout (< 300 mm TL) in the SFWW demonstrated continuous downstream migration throughout the year, culminating in a peak outmigration in August. Not only did fish migrate on a temporal continuum, but also on a spatial one, occupying habitats throughout the entire SFWW. While these migrations occurred in association with several variables in the environment, only minimum temperature was significantly correlated with the seasonal timing of migration. Furthermore, initiation of first downstream migration occurred primarily at night.

Historically, bull trout migration patterns have been described as occurring in discrete time frames (e.g., Fraley and Shepard 1989; Swanberg 1997). However, we observed a much broader temporal continuum of migration in our population. This continuous time frame of subadult bull trout migrations is not unique to this system. Hemmingsen et al. (2000) observed a similar movement pattern of fluvial subadult bull trout in nearby Mill Creek (with peaks in spring and autumn), and Downs et al. (2006) noted continuous migration of adfluvial subadult bull trout in Idaho (with spring and autumn peaks as well). It appears that first time migrations are more flexible (potentially as a result of being influenced by myriad factors) than the more discrete migrations of adult fish.

Our understanding of migration has evolved from first mention of the Restricted Movement Paradigm (Gowan et al. 1994), to a broader understanding of variable movement patterns (Gowan and Fausch 1996). In our system, we found the bull trout migrated anywhere from 0 - 45 km (at a minimum), and there was no clear demarcation between the movement distance of resident fish, and that of migratory fish. In addition, our description of movement differed depending on whether it was based on information from active recapture, passive detection, or both. Based solely on our mark-recapture information, it appeared that several of our fish did not move (as they were recaptured interannually in the same reach). But when we combined our active recapture information with passive detection, we found that a lot of movement occurred between active recapture events. Active recapture was more useful for describing homing, while passive detection was a better indicator of annual movement distances. Similar to Schrank and Rahel (2005), we found that this disparity in information illustrates the need to use multiple techniques when identifying movement patterns.

Not only did our migration patterns vary temporally and spatially, but also between day and night. As in other studies (e.g., Fraley and Shepard 1989), we found that most migrations occurred at night, particularly in the hours after sunset, and just before sunrise. Night time movements are important to bull trout as they allow smaller bull trout to escape predation from larger bull trout and other predators. Along with commencing migrations at night, bull trout also display a distinct diel habitat shift (Muhlfeld et al. 2003) into shallower water. This strategy may allow bull trout to prey on smaller conspecifics (Muhlfeld et al. 2003). The combination of diel movements and habitat shift reflect an evolutionary adaptation that allows bull trout to maximize their foraging while minimizing mortality risk, variables that contribute to increased overall fitness (Werner and Hall 1988)

Our model clearly showed that seasonal patterns in movement exist, and that abiotic variables may differentially cue migration across seasons. Bull trout are a cold water species and when we typically identify limiting factors, we consider maximum stream temperatures (Selong et al. 2001). However, we found that minimum temperatures were a far better predictor of the number of migrants per unit time across three seasons (Autumn, Winter, and Spring). As both maximum and minimum temperatures increased and decreased in a similar magnitude during the spring and autumn, it was the ‘below zero’ temperatures in the winter that contributed to the explanatory power of this variable. Likely, the physical and physiological mechanism by which temperature exerts influence on movement differs across seasons. For instance, in winter when stream temperatures fall below 0 ºC, the presence of anchor ice may force fish to move downstream (Jakober 1998). While temperature may be a stimulus, the low overall explanatory power of this model suggests that other factors likely influence migration patterns. Thus, while this model is statistically significant, its biological inference is limited.

While some variables (e.g., photoperiod) did not appear to cue migration, they may prepare the individual to respond to subsequent cues (Dingle 1996). In temperate zones, changes in photoperiod signal a seasonal change, and have been associated with physiological changes related to smolting (Byrne et al. 2003), and migration (Muhlfeld et al. 2003). As such, for some seasons, photoperiod may set the time frame in which migration could occur, and in other seasons (e.g., winter), other cues may be differentially important (e.g., temperature). Along with photoperiod changes, we found that movement occurred in association with low discharge and low precipitation. However, as fish moved during these conditions in the same proportion that they occurred in the environment, this suggests that fish were not differentially migrating during these conditions in the environment.

There were two primary limitations to our study: 1) variable detection probability and 2) we only monitored movements of fish > 120 mm TL. First, detection probability was variable, primarily due to limited times of inoperation (due to electrical outages and a fire in the study area in 2005), and not actual detector efficiency. Nevertheless, our data set of known migrants may have been larger if the detectors were operating 100% of the time. Our upper detector was one, if not the first, remote solar-powered detectors installed via helicopter deposits of equipment into a roadless area. While this approach allowed us to monitor movements and obtain recaptures in an upper headwater area of a bull trout stream rarely studied at this scale, we were limited by the logistics and technology available. We inferred movement direction based on the known capture location, and the eventual passive detection location (either the Harris Park or Bear Creek detectors), while recent approaches using passive PIT-tag detectors have addressed this issue with the installation of multiple detectors at a single location. Second, as we only tagged and monitored the movements of fish larger than 120 mm TL, our inference about subadult movement patterns does not apply to smaller fish. Fish smaller than 120 mm TL (i.e., young-of-year or age-1) may express alternate movement patterns in differential association with cues that may or may not have been important for larger fish. Despite these detector limitations, our use of passive detectors allowed us to identify annual movement patterns of multiple size classes of fish (minimum size = 120 mm TL) in all discharge conditions.

Our study represents the first multiple-year study of a fluvial bull trout population using active mark-recapture and passive PIT-tag detection techniques. This intense sampling effort allowed us to identify which individuals expressed migratory movements, assess the timing of that migration, statistically evaluate multiple environmental and biological cues that may prompt migration, and determine the distribution of migration distances throughout the population. Rather than select large fish to monitor a priori (as is done in a telemetry study), our mark-recapture technique allowed us to acquire "whole-population" movement information without sample bias. Finally, this study combined multiple sampling and monitoring techniques which, in concert, provided a thorough and detailed description of the continuum of migratory behavior displayed within our population of fish > 120 mm.

Ultimately, in defining bull trout migration patterns, it is critical to balance out the need for semantic simplicity with the need to accurately describe the variety of patterns within a system. It's been suggested that resident bull trout do not move distances greater than 2 km, while migratory fish make movements in excess of 2 km (Jakober 1992; Fraley and Shepard 1989). In the SFWW, migrant fish move between 2 - 80 km. Clearly, a fish that "migrates" 2 km is not functionally the same as one that migrates 80 km, and may have different management needs. Rather than simply categorize fish based on their propensity to move, a more accurate method may be one that describes the degree of movement. In such a manner, biological variability (both in terms of movement patterns and local stream productivity) is not obscured by categorical semantics.

Migratory bull trout represent an important behavioral component within populations. Not only can they interact with other populations genetically (in the case of a meta-population; Rieman and McIntyre 1993) but they may also occupy multiple habitats concurrently, and diffuse population level mortality risk. For example, in the Boise River Basin, widespread fires in the early 1990's extirpated resident bull trout populations, but spared the migratory fish that were farther downstream when the fires occurred. When the migratory fish returned to their natal streams to spawn, they gave rise to both resident and migratory fish that re-occupied the habitat (Dunham et al., *in press*). The long-term persistence of bull trout populations will require the conservation and recovery of both resident fish, and wide-ranging migratory fish, in conjunction with their associated local habitat patches and the disparate patches and corridors used by migratory fish.

MANAGEMENT IMPLICATIONS

The year-round temporal and spatial migration continuum of subadult bull trout observed here has some important management implications. While previous discussions of migration patterns suggested that fish use migratory corridors during discrete time intervals and move in association with various cues in the environment, our study demonstrates that fish 1) migrate continuously, 2) likely respond to a combination of complex cues (including temperature), and 3) utilize “migratory corridors” as both year-round habitat for some fish, and purely as migratory corridors for others. Therefore, management must focus on maintaining or recreating the natural conditions to which these fish have adapted (e.g., thermal and flow regimes) year round in order to insure that populations have the best possible chance to persist.

LITERATURE CITED

Baxter, C.V. 2002. Fish movement and assemblage dynamics in a Pacific Northwest riverscape. Doctoral dissertation. Oregon State University. Corvallis, Oregon.

Bjornn, T.C. 1971. Trout and salmon movements in two Idaho streams as related to temperature, food, stream flow, cover, and population density. Transactions of the American Fisheries Society 100:423-438.

Budy, P., R. Al-Chokhachy, and G.P. Thiede. 2004. Bull trout population assessment and life-history characteristics in association with habitat quality and land use: a template for recovery planning. 2003 Annual Report to US Fish and Wildlife Service. 70 pages.

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 Report to US Fish and Wildlife Service. UTCFWRU 2005(3):1-93.

Burnham, K.P., and D.R. Anderson. 2002. Model selection and multimodel inferences: A Practical Information-Theoretic Approach, 2nd edition. Springer, New York.

Byrne, C.J., R. Poole, G. Rogan, M. Dillane, and K.F. Whelan. 2003. Temporal and environmental influences on the variation in Atlantic salmon smolt migration in the Burrishoole system 1970-2000. Journal of Fish Biology 63:1552-1564.

Department of the Interior (DOI). 1999. Determination of Threatened Status for Bull Trout in the Coterminous United States; Final Rule, Notice of Intent to Prepare a Proposed Special Rule Pursuant to section 4(d) of the Endangered Species Act for the Bull Trout; Proposed Rule. U. S. Fish and Wildlife Service. Federal Register November 1, 1999. Vol. 64, No. 210. Rules and Regulations pages 58909-58933.

Downs, C.C., D. Horan, E. Morgan-Harris, and R. Jakubowski. 2006. Spawning demographics and juvenile dispersal of an adfluvial bull trout population in Trestle Creek, Idaho. North American Journal of Fisheries Management 26:190-200.

Dunham, J.B., M.K. Young, R.E. Gresswell, and B.E. Rieman. *In press.* Effects of fire on fish populations: landscape perspectives on persistence of native fishes and nonnative fish invasions. *Submitted to* Forest Ecology and Management.

Elle, S. and R. Thurow. 1994. Rapid river bull trout movement and mortality studies. Job 1 *in* Elle, S., R. Thurow, and T. Lamansky. Job Performance Report: Rivers and Streams Investigations. Idaho Department of Fish and Game, Boise, Idaho.

Flatter, B.J. 2000. Life history and population status of migratory bull trout in Arrowrock Reservoir, Idaho. Master's thesis. Boise State University, Boise, Idaho.

Fraley, J. J., and B. B. Shepard. 1989. Life history, ecology and population status of migratory bull trout (*Salvelinus confluentus*) in the Flathead Lake and River System, Montana. Northwest Science 63:133-143.

Goetz, F. 1989. Biology of the bull trout, *Salvelinus confluentus*, a literature review. Eugene, Oregon, U.S. Department of Agriculture, Forest Service, Willamette National Forest. 53 pages.

Gowan, C., M.K. Young, K.D. Fausch, and S.C. Riley. 1994. Restricted movement in resident stream salmonids: a paradigm lost? Canadian Journal of Fisheries and Aquatic Sciences 74: 2626-2637.

Gowan, C., and K.D. Fausch. 1996. Mobile brook trout in two high elevation Colorado streams: re-evaluating the concept of restricted movement. Canadian Journal of Fisheries and Aquatic Sciences 53:1370-1381.

Groot, C., and L. Margolis. 1991. Pacific Salmon Life Histories. UBC Press. Vancouver, B.C., Canada.

Gross, M.R. 1991. Salmon Breeding Behavior and Life History Evolution in Changing Environments. Ecology 72:1180-1186.

Hemmingsen, A.R., S.M. Gunckel, P.M. Sankovich, and P.J. Howell. 2000. Bull trout life history, genetics, habitat needs, and limiting factors in central and northeast Oregon. 2000 Annual Report. Project Number 199405400, Bonneville Power Administration, Portland, Oregon.

Hilderbrand, R.H., and J.L. Kershner. 2000. Movement patterns of stream-resident cutthroat trout in Beaver Creek, Idaho-Utah. Transactions of the American Fisheries Society 129: 1160-1170..

Hostettler, L. 2003. Migration of juvenile bull trout (*Salvelinus confluentus*) from the North Fork Boise River. 2001 Juvenile bull trout radio tagging interim report.

Hostettler, L. 2004. Influence of fish size and environmental factors on the extent and rate of autumn downstream migration by bull trout in the Boise River basin, Idaho. Final Report: Juvenile bull trout movement study, USBR-Boise State University Cooperative Grant Agreement #1425-01fc107320.

Jackson, D.A., P.R. Peres-Neto, and J.D. Olden. 2001. What controls who is where in freshwater communities- the role of biotic, abiotic, and spatial factors. Canadian Journal of Fisheries and Aquatic Sciences 58:157-170.

Jakober, M. 1992. Influence of stream size and morphology on the seasonal distribution and habitat use of resident bull trout and westslope cutthroat trout in western Montana. Unpublished progress report on file at: U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Boise, Idaho. 7 pages.

Jakober, M.J., T.E. McMahon, R.F. Thurow, C.C. Clancy. 1998. Role of stream ice on fall and winter movements and habitat use by bull trout and cutthroat trout in Montana headwater streams. Transactions of the American Fisheries Society 127:223-235.

Jonsson, N, and B. Jonsson. 2002. Migration of anadromous brown trout *Salmo trutta* in a Norwegian river. Freshwater Biology 47:1391-1401.

McCormick, S.D., L.P. Hansen, T.P. Quinn, and R.L. Saunders. 1998. Movement, migration, and smolting of Atlantic Salmon (*Salmo salar*). Canadian Journal of Fisheries and Aquatic Sciences 55(suppl 1):77-92.

Muhlfeld, C.C., S. Glutting, R. Hunt, D. Daniels, B. Marotz. 2003. winter diel habitat use and movement by subadult bull trout in the upper Flathead River, Montana. North American Journal of Fisheries Management 23:163-171.

Nelson, M.L., T.E. McMahon, R.F. Thurow. 2002. Decline of the migratory form in bull char, *Salvelinus confluentus*, and implications for conservation. Environmental Biology of Fishes 64:321-332.

Nordeng, H. 1983. Solution to the “char problem” based on Arctic char (*Salvelinus alpinus*) in Norway. Canadian Journal of Fisheries and Aquatic Sciences 40:1372-1387.

Northcote, R.G. 1992. Migration and residency in stream salmonids- someecological considerations and evolutionary consequences. Nordic Journal of Freshwater Research 67:5-17.

Pratt, K.L. 1992. A review of bull trout life history. In: Howell, P.J.; Buchanan, D.V., eds. Proceedings of the Gearhart Mountain bull trout workshop; 1992 August; Gearhart Mountain, OR. Corvallis, Oregon: Oregon Chapter of the American Fisheries Society:5-9.

Quinn, T.P., and D.J. Adams. 1996. Environmental changes affecting the migratory timing of American shad and sockeye salmon. Ecology 77:1151-1162.

Rieman, B.E., and J.B. Dunham. 2000. Metapopulations and salmonids: a synthesis of life history patterns and empirical observations. Ecology of Freshwater Fish 9:51-64.

Rieman, B.E. and J.D. McIntyre. 1993. Demographic and habitat requirements for conservation of bull trout. USDA General Technical Report INT-302. September.

Riley, W. D., M. O. Eagle, and S. J. Ives. 2002. The onset of downstream movement of juvenile Atlantic salmon, *Salmo salar L*., in a chalk stream. Fisheries Management and Ecology 9:87-94.

Salow, T. 2005. Inferences from weir counts of population size and environmental influence on migration timing for adfluvial bull trout (*Salvelinus confluentus*) in the North and Middle forks of the Boise River, Idaho: Summary report for the Boise River Cooperative Bull Trout Monitoring project.

SAS Institute. 2005. SAS Version 9.01. SAS Institute, Cary, North Carolina.

Schlosser, I.J. 1991. Stream fish ecology: a landscape perspective. Bioscience 41:704-712.

Schrank, A.J., and F.J. Rahel. 2004. Movement patterns in inland cutthroat trout (*Oncorhynchus clarki utah*): management and conservation implications. Canadian Journal of Fisheries and Aquatic Sciences 61:1528-1537.

Schrank, A.J., and F.J. Rahel. 2005. Factors influencing summer movement patterns of Bonneville cutthroat trout (*Oncorhynchus clarkii utah*). Canadian Journal of Fisheries and Aquatic Sciences 63: 660-669.

Selong, J.H., and T.E. McMahon. 2001. Effect of temperature on growth and survival of bull trout, with application of an improved method for determining thermal tolerances in fishes. Transactions of the American Fisheries Society 130:1026-1037.

Swanberg, T.R. 1997. Movements of and Habitat Use by Fluvial Bull Trout in the Blackfoot River, Montana. Transactions of the American Fisheries Society 126:735-746.

Thorpe, J.E., and R.I.G. Morgan. 1978. Periodicity in Atlantic salmon, *Salmo salar L*. smolt migration. Journal of Fish Biology 12:541-548.

Werner, E.E., and D.J. Hall. 1988. Ontogenetic habitat shifts in bluegill: the foraging rate-predation risk trade-off. Ecology 69:1352-1366.

Whalen, K.G., D.L. Parrish, and S.D. McCormick. 1999. Migration timing of Atlantic salmon smolts relative to environmental and physiological factors. Transactions of the American Fisheries Society 128:289-301.

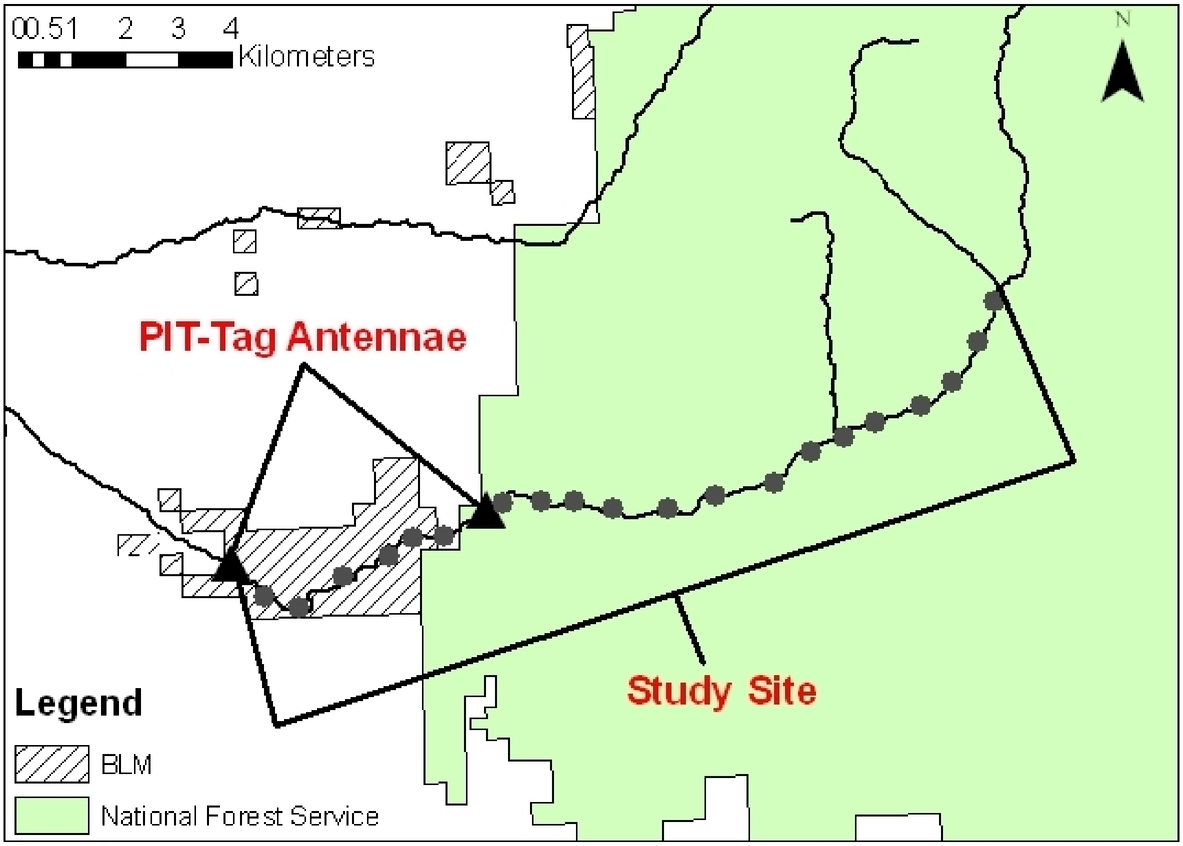

Figure 3.1. Map of the study site on the South Fork Walla Walla River with locations of sample reaches (gray circles) and passive PIT-tag antennae (detectors, black triangles).

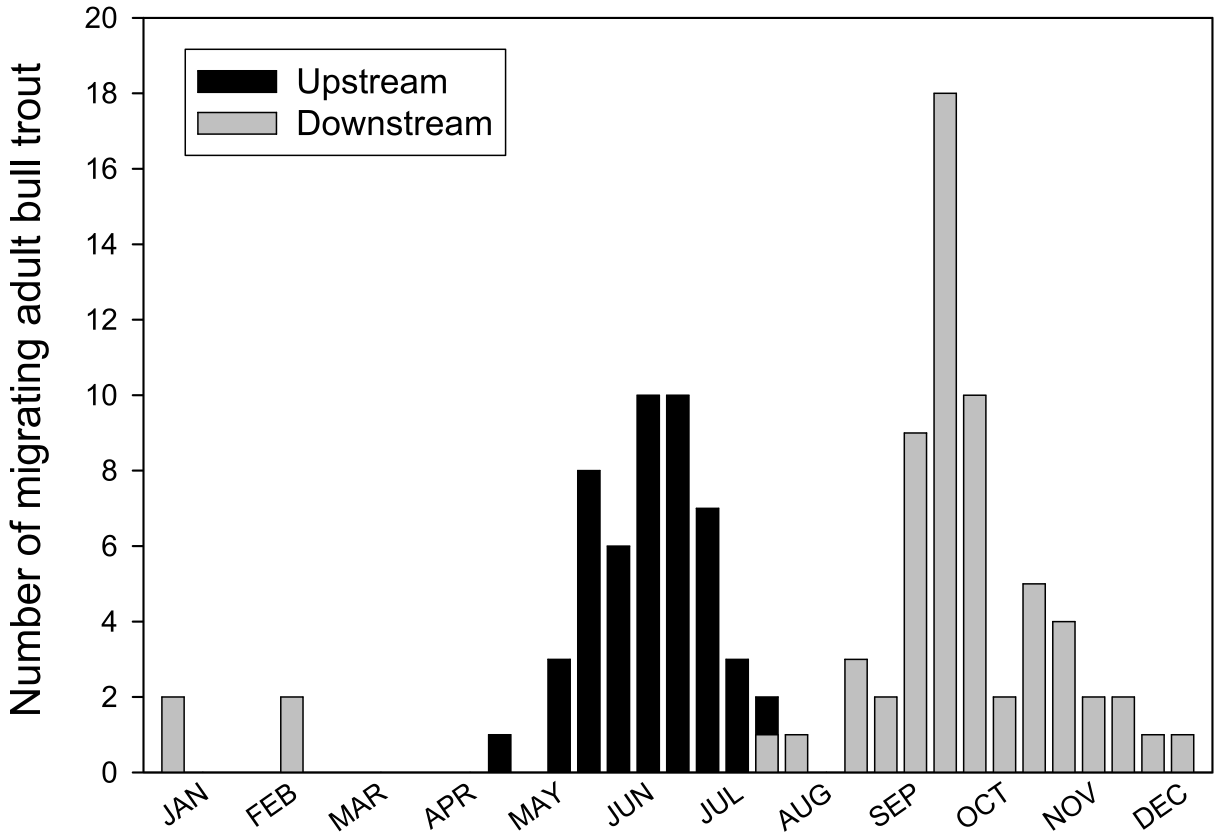

Figure 3.2. Timing of upstream and downstream migration (i.e., date of detection at Harris Park) of adult bull trout (> 300 mm TL) in the South Fork Walla Walla River, 2002 - 2005 combined.

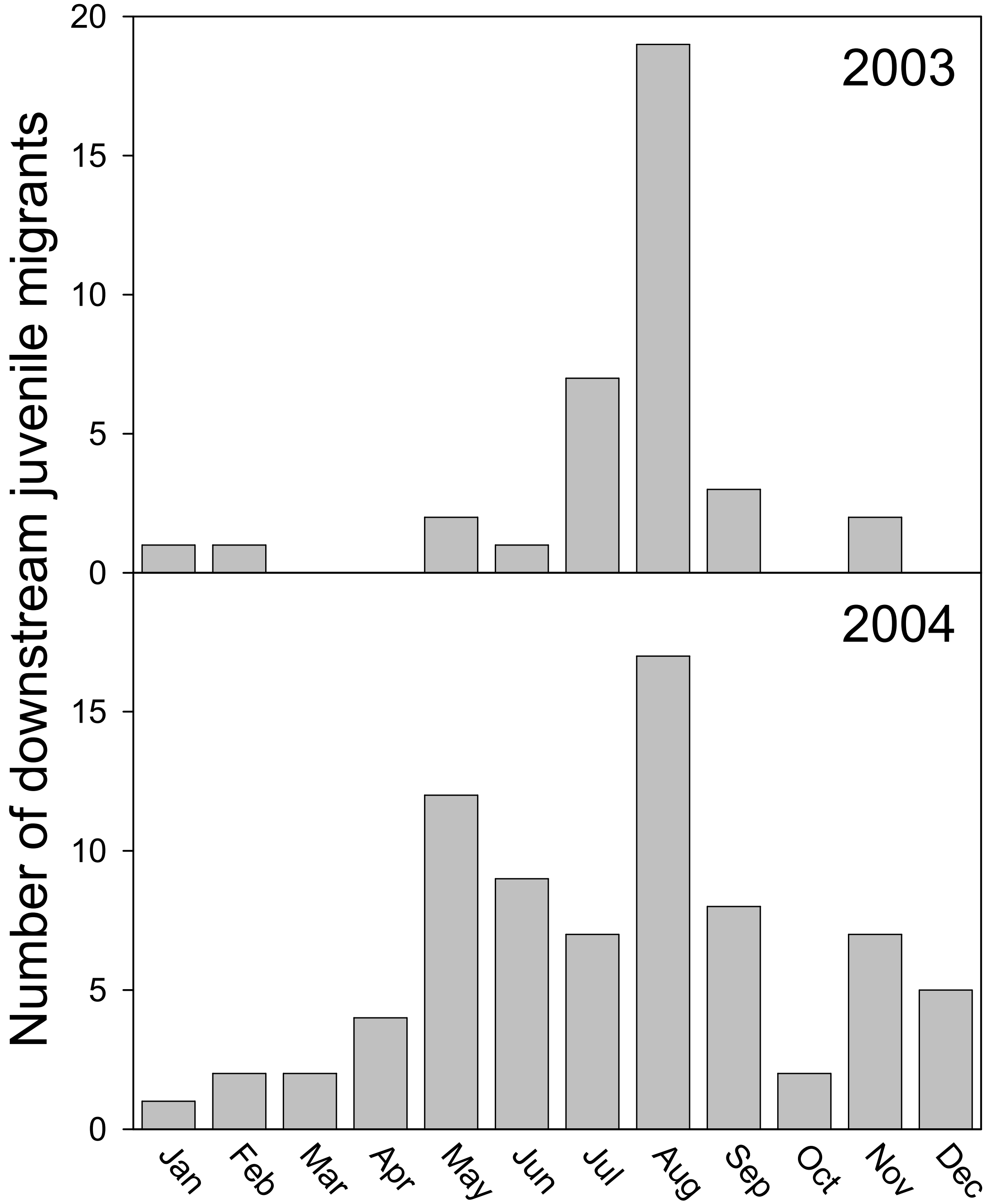

Figure 3.3. Timing of downstream migration (i.e., date of detection at Harris Park) of subadult bull trout (120 - 300 mm TL) in the South Fork Walla Walla River, 2003 (top panel) and 2004 (bottom panel).

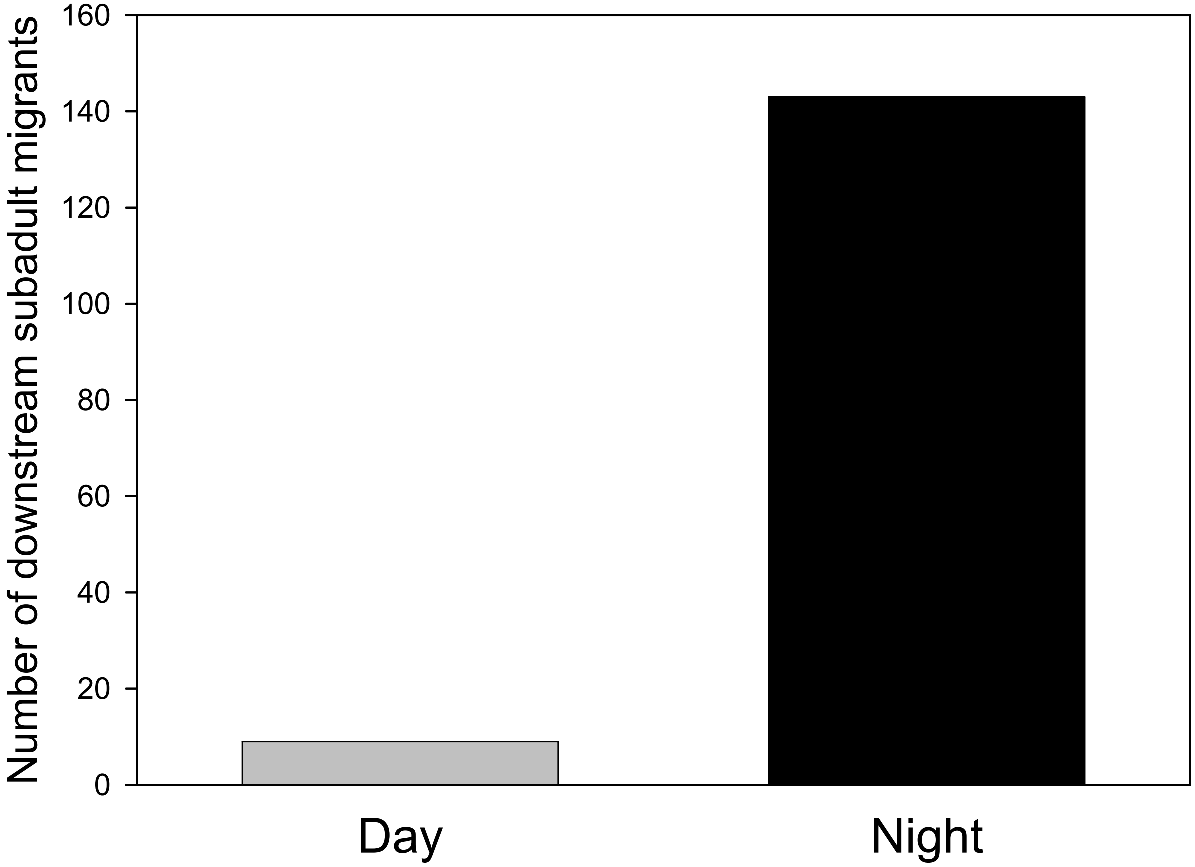

Figure 3.4. Day and night detections of downstream migrating subadult bull trout at the Harris Park detector, South Fork Walla Walla River, 2002 - 2005.

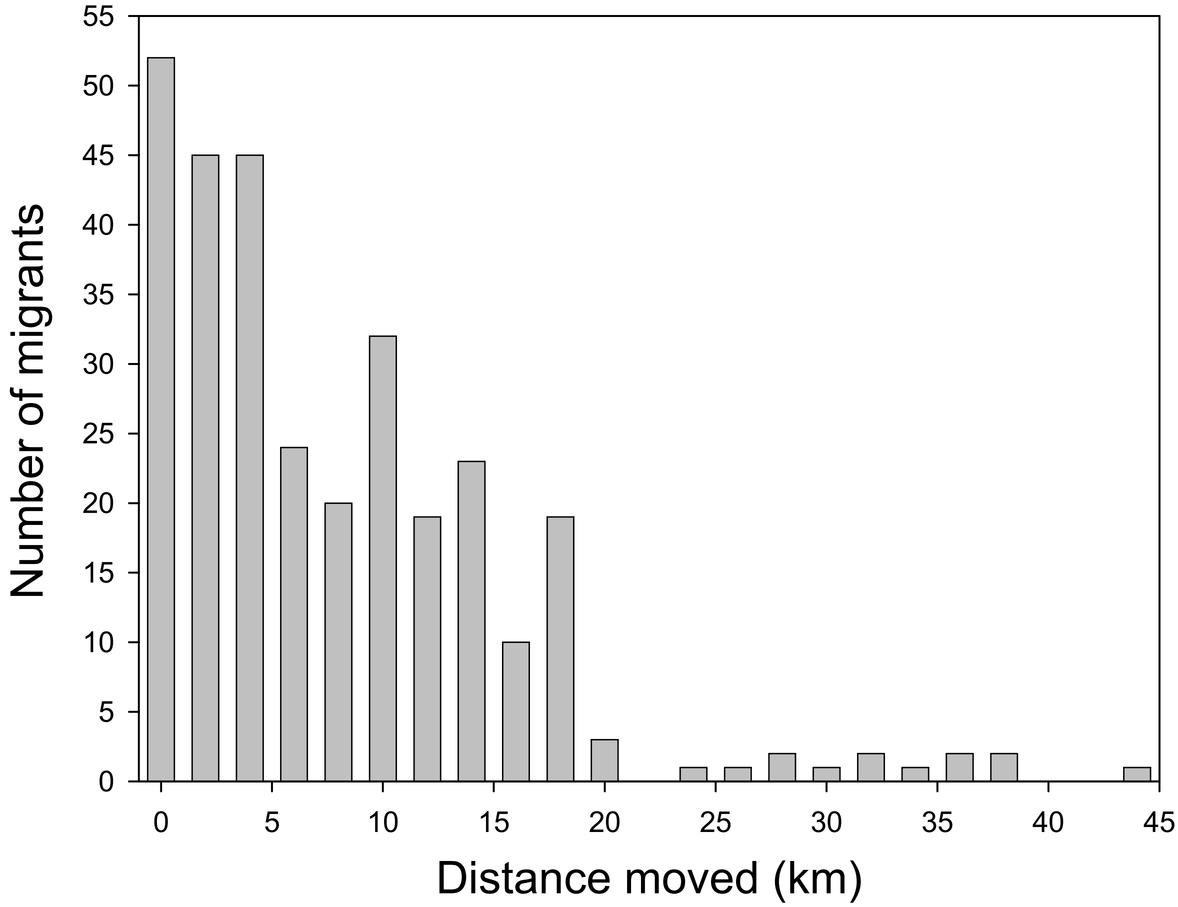

Figure 3.5. Minimum distance moved for all size classes of bull trout based on both annual active recaptures, and daily passive detections at the Harris Park, Bear Creek, and Nursery Bridge dam detectors on the South Fork Walla Walla River, 2002 - 2005.

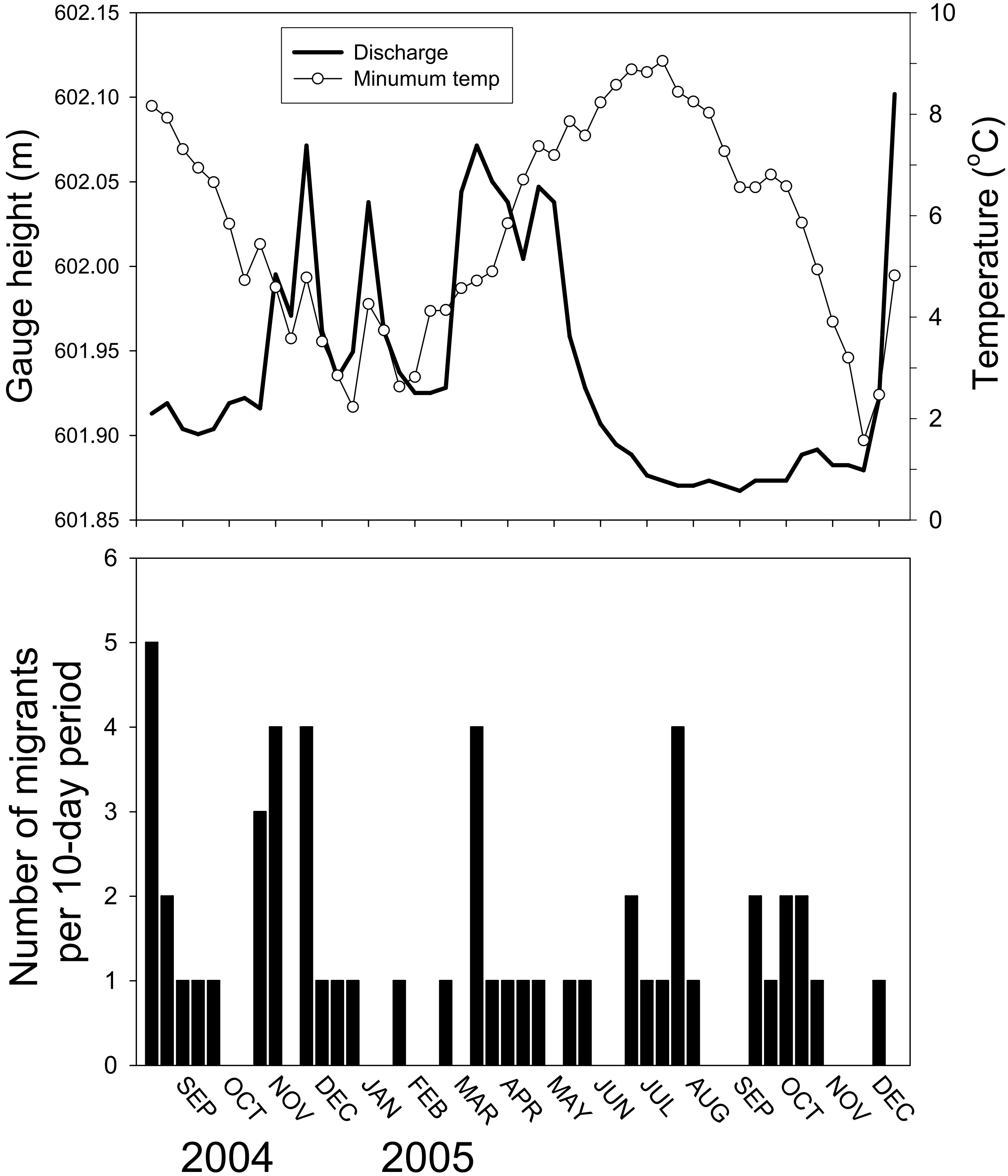

Figure 3.6. Number of juvenile migrants detected at the Harris Park detector (per 10-day period) in the South Fork Walla Walla River, September 2004 - December 2005 (bottom panel) as it relates to two environmental conditions (minimum temperature and discharge) recorded at Harris Park (top panel). Discharge is reported as gauge height (m) as the gauging station is not yet rated.

APPENDIX 1

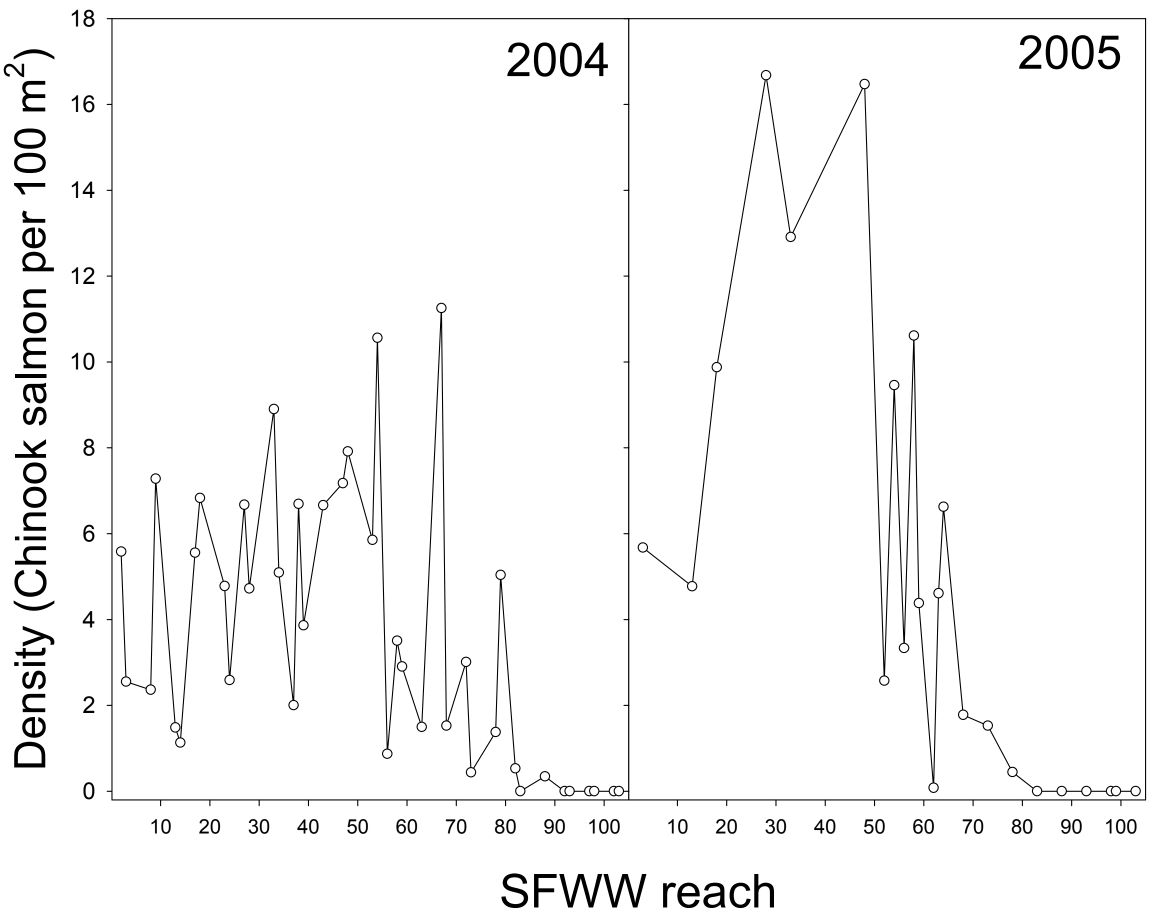

Figure A1. Density of Chinook salmon estimated by snorkel counts in various reaches of the South Fork Walla Walla River, 2004 and 2005.

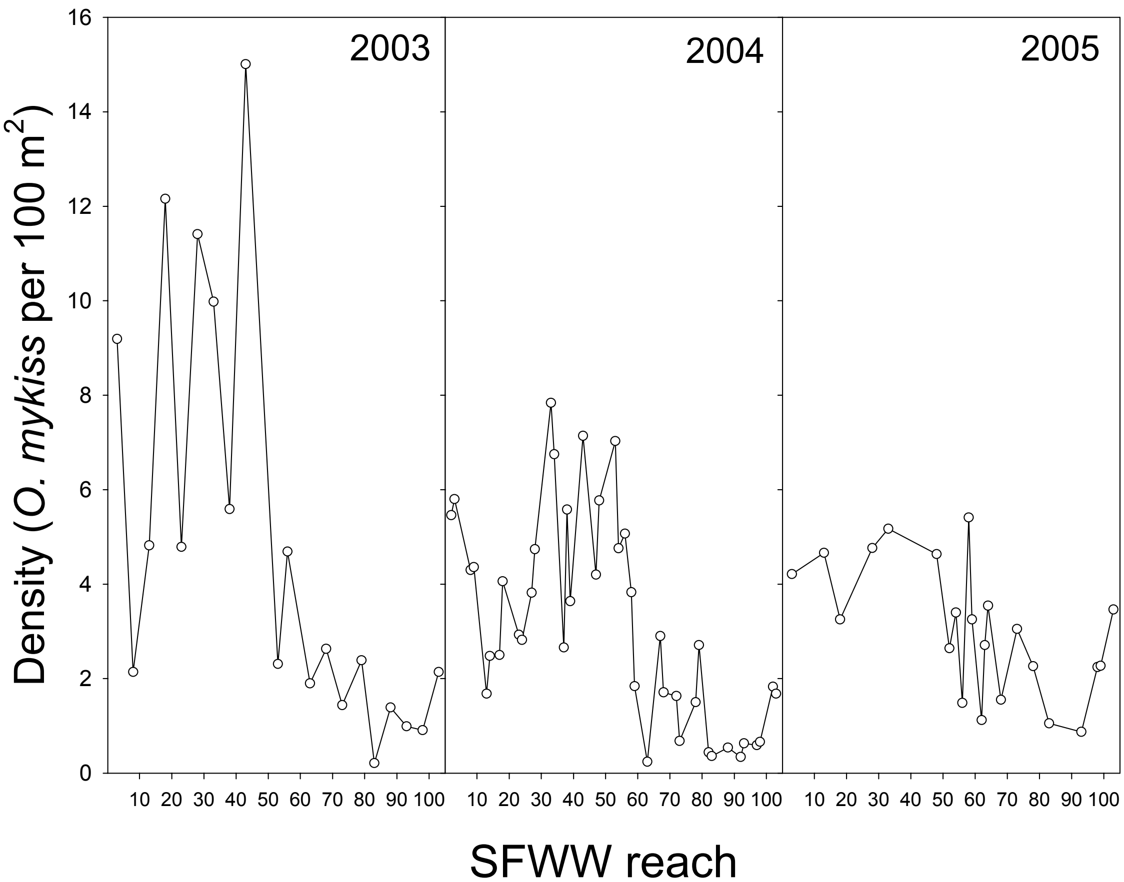

Figure A2. Density of *O. mykiss* spp. estimated by snorkel counts in various reaches of the South Fork Walla Walla River, 2003, 2004, and 2005.

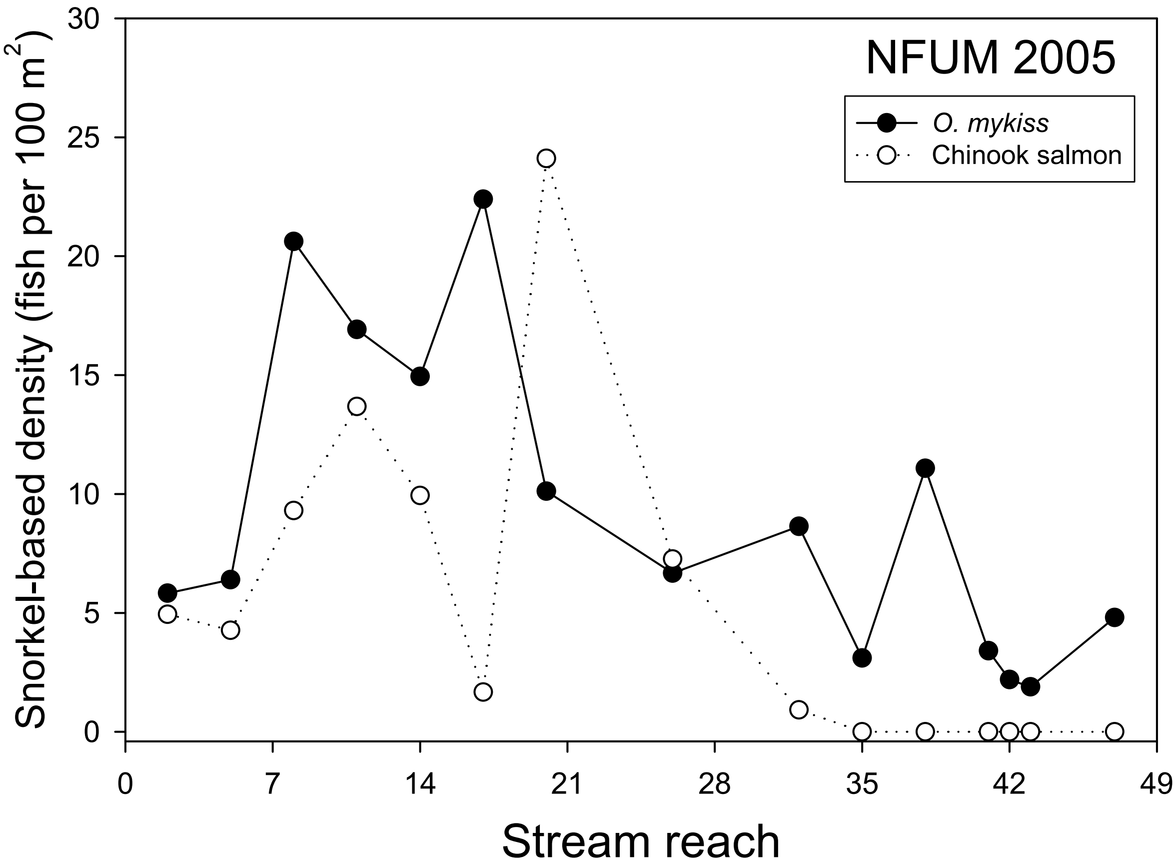

Figure A3. Density of *O. mykiss* spp. and Chinook salmon estimated by snorkel counts in various reaches of the North Fork Umatilla River, 2005.

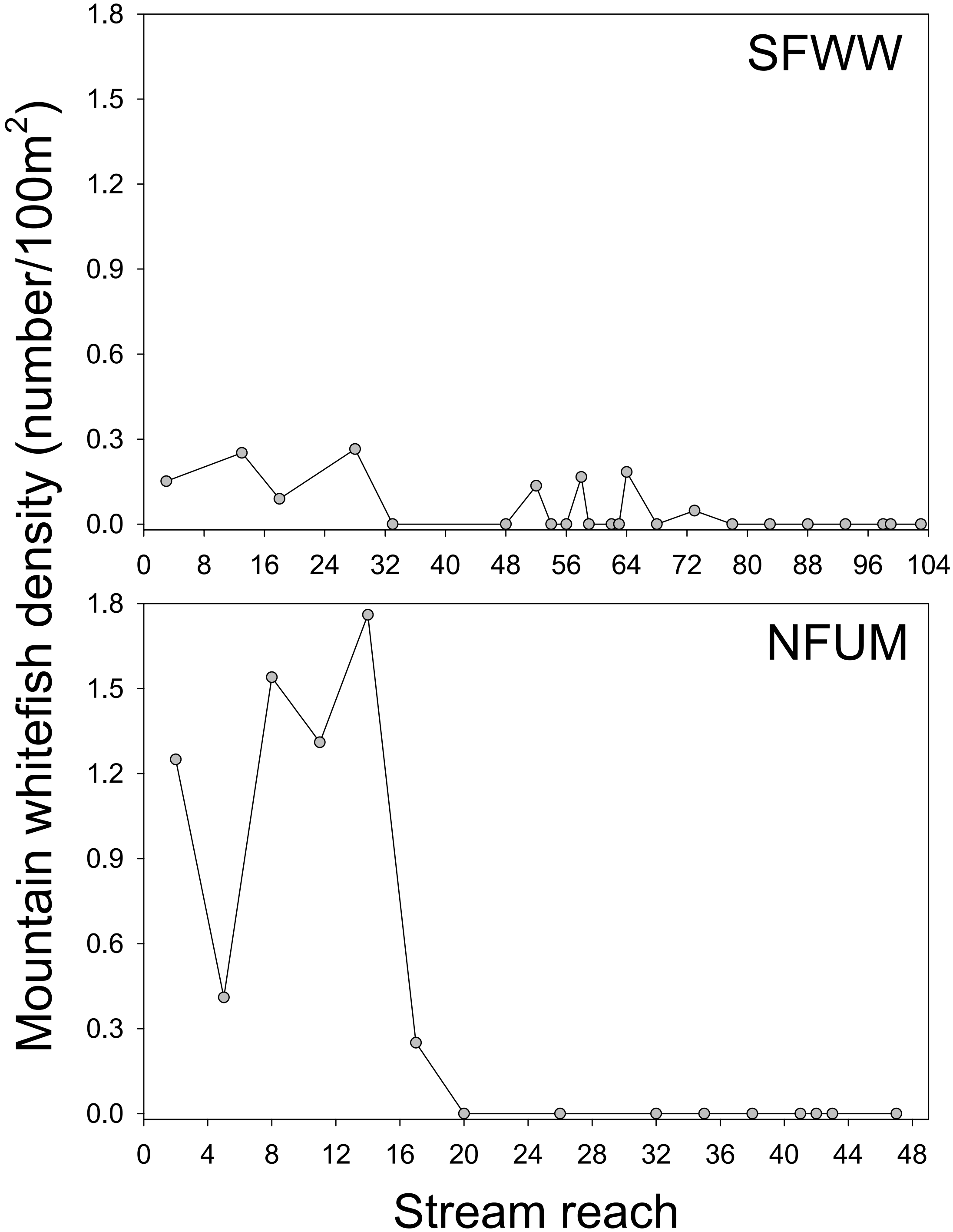

Figure A4. Densities (number per 100 m^2) of mountain whitefish observed during snorkeling surveys in the South Fork Walla Walla River and North Fork Umatilla River, summer 2005.

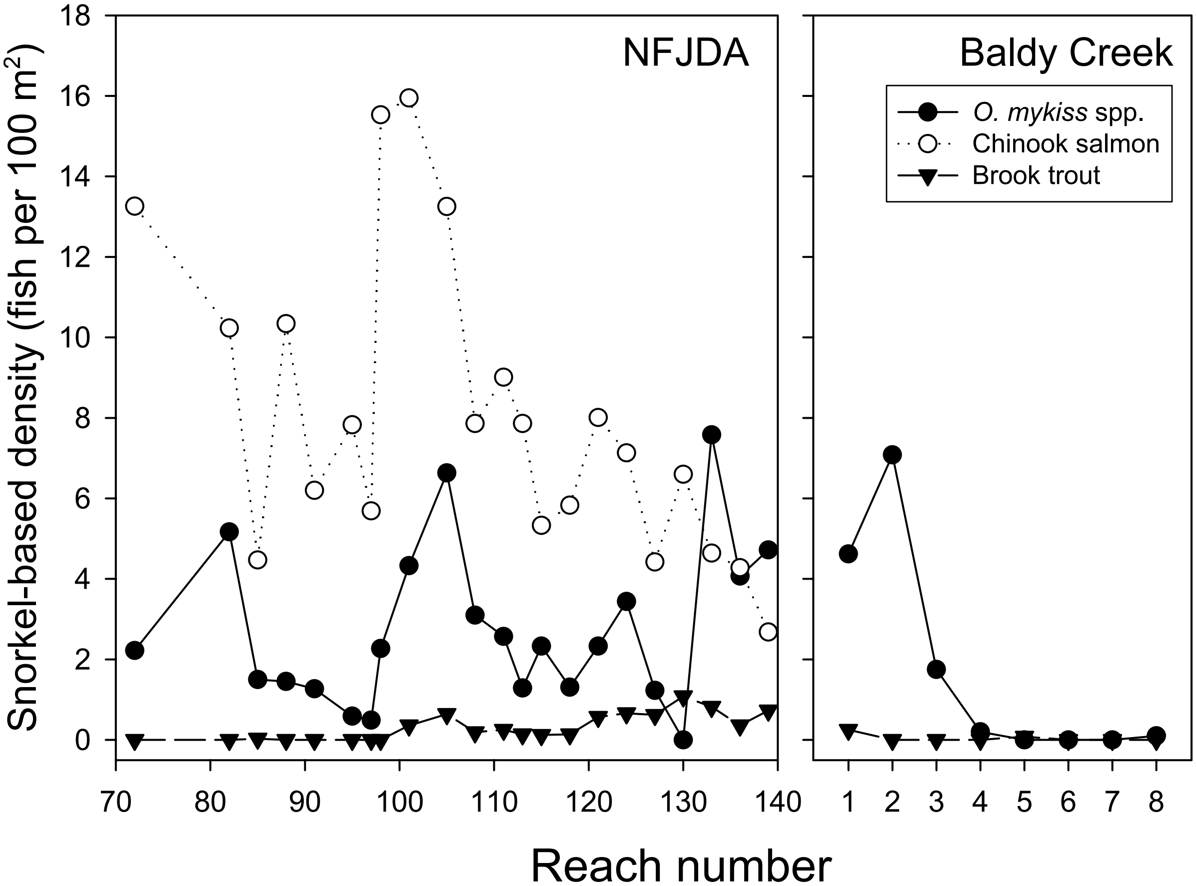

Figure A5. Densities of *O. mykiss* spp., Chinook salmon, and brook trout estimated by snorkel counts in various reaches of the North Fork John Day River and Baldy Creek, summer 2005.

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.