

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

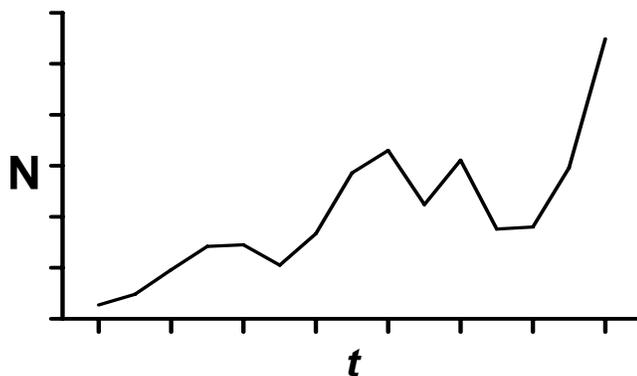
Annual Progress Report for 2006

by

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PREFACE

This report consists of two separate chapters. Chapter 1 describes the results from the monitoring and evaluation of bull trout populations in several streams in northeastern Oregon, which have been monitored annually since 2002. Chapter 2 evaluates the use of mark-recapture methods to monitor bull trout population trends and the trends of different components of these populations, and compares these results with more commonly-used methods. Chapter 2 is a manuscript currently under development and internal review for the journal *Transactions of the American Fisheries Society*. 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 2006, with plans to continue work through 2007. 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.

The efforts of this project have been part of a PhD dissertation (Al-Chokhachy 2006) and master's thesis (Homel 2007) conducted through Utah State University. Together, our efforts have resulted in six peer-reviewed manuscripts addressing issues related bull trout conservation and management. First, we examined the efficacy of current bull trout population monitoring efforts, and illustrated potential bias in our ability to monitor bull trout populations using redd counts (Al-Chokhachy et al. 2005). Next, we evaluated bull trout habitat relationships, and despite consistency in habitat-use patterns across streams, we found the inherently low densities of bull trout populations prevented the effectiveness of formal models to predict bull trout habitat use (Al-Chokhachy and Budy, *In press*). Next, we evaluated the movement patterns and abiotic/biotic cues associated with the migration timing of fluvial juvenile and subadult bull trout and observed distinct seasonal patterns in migration timing for these age-classes (Homel and Budy, *In review*). Next, we used 5 years of mark-recapture data to quantify the first field estimates of bull trout survival and used this information in simple population model to evaluate bull trout population structure and the effects of various management actions on population growth rates. Our results suggested changes to the juvenile components of bull trout populations can have large impacts on population growth rates; additionally, we found adult bull trout exhibited the highest amount of migratory behavior, highlighting the evolutionary importance of these fish to

the overall persistence of bull trout (Al-Chokhachy and Budy, *In review*). Next, assessed genetic differentiation in bull trout between resident and migratory forms, as well as among tributaries, and found a complete lack of neutral genetic structure (both between behavioral groups, and among tributaries) within the SF Walla Walla population (Homel et al. *In review*). Finally, we investigated the tradeoff between the costs and levels of precision associated with different bull trout monitoring techniques; our results indicated the low levels of precision associated with population abundance estimates inhibit our ability to detect modest changes in bull trout abundance over relatively short time intervals (e.g., 5 years; Al-Chokhachy et al. *In review*). Overall, these documents add substantial contribution to our understanding of the ecology of bull trout and our abilities to effectively monitor and evaluate bull trout populations. Our information is being used by USFWS Recovery Monitoring and Evaluation workgroup to help guide large scale monitoring efforts to assess abundance, trends in abundance, and distribution of bull trout within and among core area populations.

This current document related to our most recent monitoring is delineated into two chapters. 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 2006, we sampled 22 reaches (or 26% of the study area) in the South Fork Walla Walla River (SFWW), 16 reaches (or 43% of the study area) in the North Fork Umatilla River (NFUM), and 28 reaches (or 29% 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 221 fish in the SFWW, 77 fish in the NFUM, and 110 fish in the NFJDA. In all three systems, the largest portion of sampled fish was 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 2006, bull trout condition (Fulton's K) in the SFWW and NFUM, was significantly higher than in the NFJDA and Baldy Creek, and we observed the first increase in condition in the SFWW and NFUM since the onset of this study in 2002. Interestingly, this increase in condition did not correspond with any substantial change in bull trout diet, as bull trout diets continued to be dominated by aquatic invertebrates. Bull trout survival in the SFWW appeared to vary significantly across years and size classes, and individual condition had a significant positive effect on bull trout survival (Beta = 0.45, SE = 0.05). In the NFUM, bull trout survival was consistent with estimates from corresponding size classes in the SFWW, but small sample sizes prevented any estimates across different size classes. While the population abundance of bull trout (> 220 mm) in the SFWW has remained generally stable over the 5-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 10,600 (95% CI = 8,080 – 16,598) in 2006. Population abundance continues to be much lower in the NFUM with a total population size of 2,088 (1,134 – 6,046). Abundance estimates for juvenile and small adult bull trout (120 – 370 mm) demonstrated high annual-variability since 2003, but no apparent increasing or decreasing trend in abundance; however, we observed a consistent decrease in large bull trout (>370 mm) over this time period. Similar to 2005, our population estimates for the NFJDA demonstrate a low abundance of bull trout with estimates of fish >120 mm at 437 (95% CI = 274 – 752) and 1,193 (95% CI = 825 – 2509) in NFJD and Baldy Creek, respectively.

In Chapter 2, we use the temporal symmetry open mark-recapture model (2002 – 2006) and both active and passive mark-recapture techniques to estimate bull trout (*Salvelinus confluentus*) population trends for different size classes and life-history forms in the SFWW River. We compare the precision of these results with trend analyses from two commonly used techniques in salmonid monitoring, including snorkel surveys and redd count data. With these analyses, we considered the population trend (λ) stable if λ or the 95% confidence intervals (CI) overlapped with one, declining if λ (and 95% CI) <1, and increasing if λ (and 95% CI) >1. Our mark-recapture results suggest bull trout population trends are stable for juvenile (120 – 220 mm; $\lambda = 0.91$, 95% CI = 0.79 – 1.05) and small adult bull trout (221 – 370 mm; $\lambda = 0.88$, 95% CI = 0.72 – 1.05), but large bull trout declined over this period (>370 mm; $\lambda = 0.83$, 95% CI = 0.75 – 0.91). Across techniques, we found snorkel surveys exhibited the lowest precision with coefficient of variation (CV) values ranging from 0.27 – 0.50. Redd counts trend estimates exhibited significantly higher precision than snorkel surveys (CV = 0.17), and most closely corresponded with bull trout >370 mm, but with slightly reduced precision ($\lambda = 0.90$, 0.78 – 1.02). We observed evidence of declining trends in the migratory component of bull trout in the SFWW ($\lambda = 0.81$, 95% CI = 0.72 – 1.05), but with relatively high annual variability. Overall, we found that open population-trend mark-recapture methods can provide managers with an ability to monitor different components of a population, and with higher precision than more commonly-used salmonid monitoring techniques. With higher precision, temporal symmetry mark-recapture methods may be more appropriate for detecting population change in response to different management actions, restoration activities, and for short-term monitoring and evaluations required for different regulatory actions.

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 across stage classes and life-history forms (e.g., survival, growth) is an important part of the process of identifying factors that potentially limit the population, evaluating the importance of these vital rates on overall trend, and ultimately directing future recovery and restoration activities. 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). 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. Furthermore, 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, and 2006 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, robust evaluations of different monitoring techniques, 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 (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 - 2006 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 abundance of both resident and migratory fish, complex water management issues associated with fish protection, and a diversity of habitat types.

Furthermore, expansion of research into multiple additional watersheds has allowed for comparisons of critical population-level metrics (e.g., population structure) across ecosystems and varying levels of bull trout abundance. Thus our complete work includes five years of data to date (2002 - 2006) from one intensively monitored stream, and two additional streams where smaller-scale, yet still long-term and continuous, population assessment evaluations are also underway. Collecting monitoring data in these additional streams allows us to investigate some key questions in greater detail and across a range of biotic and abiotic conditions.

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 km² (Walla Walla Subbasin Summary Draft 2001). The tributaries of the Walla Walla River originate in the Blue Mountains at elevations near 1800 m. The mainstem Walla Walla flows for approximately 16 km in Oregon before splitting into the NF Walla Walla and the SF Walla Walla rivers.

The Walla Walla River historically contained a number of anadromous and resident, native salmonid populations including: spring and fall Chinook salmon (*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, mountain whitefish, sculpin (*Cottus* spp.), and dace (*Rhinichthys* spp.) 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 km². The Umatilla River is 143 km long from mouth (at Columbia River RK 440) to where it divides into the NF and SF Umatilla rivers, each fork adding another 16 km in length. The Umatilla mainstem originates in Blue Mountains at 1289 m and descends to 82 m at confluence with Columbia River. Earliest documentation of bull trout in Umatilla basin is from

ODFW creel reports dating from 1963. The mainstem Umatilla River is artificially confined for much of its length. Spawning occurs in the NF and SF Umatilla rivers, and in NF Meacham Creek. Along with being an important tributary for rearing and migration activities, redd counts indicate that the majority of redds in the Umatilla basin occur in the NF Umatilla River between Coyote and Woodward creeks. Peak spawning generally occurs between mid September and mid October over at least a two-month period (ODFW 1995, 1996) when daily average water temperatures ranged from 6-10 °C (ODFW 1996). Habitat in the NF Umatilla River is fairly complex with low levels of bedload movement, moderate levels of large organic debris, and relatively minimal flow events. Other species occurring in the basin include *O. mykiss* subspecies, sculpin (*Cottus* spp.), Chinook salmon, shiners (*Richardsonius balteatus*), suckers (*Catostomus* spp.), dace (likely *Rhinichthys* spp.), and northern 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 km² (John Day Subbasin Draft Plan 2004) originating at elevations of 2438 m in the Blue Mountains. The North Fork of the John Day River (NFJDA) is the largest tributary and flows westerly for 180 km. The John Day River historically supported large populations of Chinook salmon and steelhead trout and

currently demonstrates relatively high (as compared to other subbasins) abundances of both species. Steelhead 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 few reliable population estimates for bull trout in the NFJDA (see Budy et al. 2006); 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 8.8 km from the confluence of NFJDA with Baldy Creek (110395981 E 4973793 N) up to the headwaters of the NFJDA where the stream gradient increased substantially (UTM 110401330 E 4970212 N). We sampled 16 reaches on this stretch totaling 6.0 km (Figure 1.3). We also sampled 12 sites on Baldy Creek from the confluence of Baldy Creek (UTM 110396075 E 4973524 N) up approximately 7 km (UTM 11395782 E 4969691 N), 3 reaches on Bull Creek up approximately 3.4 km (UTM 11398336 E 4971072 N), and 2 reaches up Limber Creek (UTM 11397589 E 4969276 N). Overall, we sampled 2.4 km of the 11.9 km in the Baldy Creek complex. Reaches continued upstream for at least 200 m. Currently, there are no passive PIT-tag detector stations in the NFJDA.

METHODS

Fish Sampling

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 in the SFWW 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 in the NFJDA and Baldy Creek fin clips (approximately 25 mm²) were collected for genetic analyses. All fish were placed in a flow-through recovery container within the channel, monitored until full equilibrium was restored, and returned to slow-water habitat near individual capture locations.

Resighting.—To resight Floy-tagged fish, we conducted daytime bull trout snorkel surveys in 22 reaches (mean reach length = 244 m) of the SFWW, 16 reaches (mean = 212 m) of the NFUM, 16 reaches in the NFJDA (mean = 225 m), and 12 reaches (mean = 201 m) in Baldy Creek in 2006. 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.—We recaptured previously tagged bull trout (2002 – current) using a combination of techniques: seining, trap netting, and pass-through PIT-tag technology described below. All actively captured bull trout were passed over a handheld PIT-tag detector and checked for anchor tags from previous years. When recaptured, all bull trout were weighed and measured for estimates of annual growth, and we recorded information regarding location of recapture. Recapture events also provided critical information for estimates of bull trout survival, annual population estimates, and to parameterize the temporal symmetry mark and recapture model.

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/pitttag/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 South Fork Umatilla River. The detector has been collecting data since autumn 2004.

Growth

Growth information was obtained from bull trout previously tagged in the SFWW (2002-2006), NFUM (2003-2006), and NFJDA (2005-2006) and recaptured during the 2006 summer field season. Length and weight gains were determined between initial tagging and subsequent capture events. These length and weight gains were evaluated based on annual growth, and delineated by size class at initial tagging.

Population Estimates

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.

Survival

SFWW.—In the SFWW, we estimated survival using the Barker model (e.g., Buzby and Deegan 2004) with five years of mark-recapture data (2002-2006). The Barker model is an open mark-recapture model, which is similar to the Cormack-Jolly-Seber, incorporates the number of marked and recaptured fish in sampling events (June – August); however, the Barker model also incorporates recapture events that may occur between annual sampling events (e.g., detected at PIT-tag detectors).

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, 270 - 320, 320 -370, and > 370 mm size classes. Survival estimates and recapture probabilities were calculated using Program MARK software.

NFUM.—In the NFUM, where we have a significantly smaller sample size of marked and recaptured bull trout, we used four years of mark-recapture data (2003 – 2006) to estimate survival with a Cormack-Jolly-Seber model (CJS). The CJS model is a simpler model than the Barker model, and does not require large sample sizes. Since the majority of fish captured in the NFUM are typically 120-170 mm, we did not delineate the NFUM into different size categories. Survival estimates and recapture probabilities were calculated using Program MARK software.

Diet Analysis

Sampling design.—In the SFWW, 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, we selected only one sampling location due to the limited distribution of bull trout in 2005 surveys. The majority of the sampling occurred just upstream of the confluence of Baldy Creek (UTM 110396097 E 49773596 N), and continued upstream. 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).

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.

Temperature

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

Movement

We measured bull trout movement patterns in the SFWW and NFUM using mark-recaptured data and passive instream antennae. Movement information for the SFWW has been previously described in Budy et al. 2006 and furthermore, in Homel and Budy (*In review*), and therefore this document will focus on movement patterns in the NFUM. We summarize movement information in the NFUM by month and across diel periods.

RESULTS and DISCUSSION

Fish Sampling

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 22 reaches during the 2006 field season, which accounted for approximately 29% of the study site. Over the summer, a total of 445 bull trout were captured of which, 221 were tagged, with the number tagged varying by sample reach (1 – 26 per reach; Figure 1.7). In 2006, as in years since 2003, most bull trout were tagged upstream of Burnt Cabin Creek (Figure 1.7). In 2006, the smallest bull trout captured was 35 mm (0.3 g) and the largest bull trout caught was 574 mm. Length-frequency distributions of captured bull trout in the SFWW have varied little from 2002 through 2006, with most captured fish in the 100 – 150 mm size range (Figure 1.8). More large (> 400 mm) bull trout were captured in the SFWW compared to both the NFUM (Figure 1.9) and the NFJDA systems (Figure 1.10).

Condition.—Condition (Fulton’s K) of bull trout captured in the SFWW varied by size class and year. In 2006, we observed significant increases in condition of juvenile (<120 mm) and large bull trout (>370 mm) as compared to 2005 estimates. Juveniles and small adults (120-370 mm) exhibited similar condition as 2005 (Figure 1.11). When all size classes are combined, it appears that average condition in 2006 (Mean = 0.88 95% CI = 0.86 – 0.90) has increased since 2005, and is more similar to overall condition values for 2004 (Figure 1.12). On average, condition in the SFWW was similar to values in the NFUM, but significantly higher than average condition in the NFJDA. 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 22 reaches in the SFWW in 2006. As with numbers of fish 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 (Thurrow 1997; Figure 1.14). In 2006, bull trout observed in the SFWW ranged from 50 to 720 mm, similar to past surveys (Figure 1.14; Budy et al. 2004, 2005).

North Fork Umatilla River

We sampled 16 reaches in 2006 which accounted for 43% of the study site. Bull trout were captured or observed in all sampled reaches. Over the summer, a total of 179 bull trout were captured and 77 were tagged, with the number tagged varying by sample reach (1 to 17 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 2006 was a 256 mm fish (173.3 g), while the smallest bull trout captured was 49 mm (1 g; Figure 1.1.5).

Condition.—Similar to previous years, condition (Fulton’s K) in the NFUM varied little across size classes in 2006, as condition values for bull trout <120 mm and >120 mm were nearly identical (Figure 1.16). As observed in the SFWW, bull trout condition of all fish combined increased in the NFUM in 2006 ($K = 0.87$; 95% CI = 0.86 – 0.89; Figure 1.12).

Snorkel surveys.—Snorkeling surveys were performed in all 16 reaches, and bull trout were observed in most sampled reaches, except in Reach 2, which occurs just upstream from the confluence with the South Fork Umatilla River (Figure 1.17). As with the number of fish tagged, most bull trout (78% 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 NFJDA and Baldy Creek (Figure 1.14); observed numbers of bull trout were substantially lower than in SFWW (Figure 1.14).

North Fork John Day River

We sampled 16 reaches in the NFJDA in 2006 plus 12 reaches on Baldy Creek. Bull trout were captured or observed in most sampled reaches. This was our second year sampling in the NFJDA basin. Based on very sparse encounters with bull trout in 2005, we eliminated sampling areas below the confluence with Baldy Creek, and extended our sampling to the headwaters in both the NFJDA and Baldy Creek (see *methods*). We tagged a total of 37 bull trout in the NFJDA and 73 bull trout in Baldy Creek, with the number tagged varying by sample reach (0 to 31 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 247 mm long (weight = 115.4 g), while the smallest was 89 mm (5.5 g). The largest fish captured in Baldy Creek was 446 mm (761.5 g), and the smallest was 46 mm (0.8 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.—We found little variation in average condition for bull trout <370 mm between Baldy Creek and the NFJDA (figure 1.19). Average condition for large bull trout (>370 mm) was substantially higher in NFJDA ($K = 0.96$) than in Baldy Creek ($K = 0.88$); however, sample sizes were small. In Baldy Creek, there was little change in condition across years for all size classes. In NFJDA, average condition of bull trout 120 – 370 mm dropped slightly from 2005 to 2006, but condition varied little across juvenile (<120 mm) and large bull trout (>370 mm). In 2006, average condition of bull trout (all sizes combined) from NFJDA and Baldy Creek was similar to condition of bull trout from NFJDA, but lower than condition of bull trout from the SFWW (Figure 1.12).

Snorkel surveys.—We performed snorkel surveys in 16 reaches in the NFJDA plus 12 reaches on Baldy Creek in 2006. We observed bull trout in all sampled reaches in Baldy Creek and Bull Creek, but did not observe any fish in the two reaches surveyed in Limber Creek. In the NFJDA, we observed bull trout in all but one of the reaches surveyed in 2006 (Figure 1.20). Bull trout observed ranged from 50 – 350 mm in NFJDA and from 50 – 450 mm in Baldy Creek (Figure 1.14), which was similar to the distribution observed in NFJDA; the overall size distribution and numbers of bull trout observed was substantially less than SFWW. We also calculated densities of bull trout

based on snorkel counts and sampled stream area; densities ranged from 0 – 0.65 fish per 100 m² in the NFJDA, and ranged from 0.5 – 2.6 fish per 100 m² in Baldy Creek (Figure 1.21).

Growth

Tagged fish.—Since 2002, we recaptured seventy bull trout for estimates of annual growth. Average annual growth of tagged bull trout varied across size classes and systems. In the SFWW, small bull trout (120-220 mm) had significantly larger annual growth in length than small adults (220-370 mm) and large adults (>370 mm; Figure 1.22). In terms of body mass, we found no significant differences in growth across size classes, due to the large variability in annual growth for bull trout >370 mm (95% CI = 89 – 264 g). Across systems, small bull trout in the SFWW and NFUM had significantly higher growth rates in length than similar sized fish from the NFJDA. Small bull trout in the SFWW had significantly higher annual growth rates in body mass than the NFUM and NFJDA. Since no bull trout >220 mm were captured, we were unable to make comparisons for larger fish.

Population Estimates

South Fork Walla Walla River.—Across all systems, the SFWW bull trout population was significantly larger than the NFUM and NFJDA populations (Figure 1.25). Estimated abundance of bull trout in the SFWW depends greatly on size grouping. Over a 5-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 10,600 (95% CI = 8,080 – 16,598) in 2006 (Figure 1.23). The abundance of bull trout > 220 mm has ranged from 2,700 in 2002 down to 1,800 in 2005, with the 2006 estimate at 2,056 (95% CI = 1,544 – 3,237). In 2006, we estimated the abundance of large bull trout (> 370 mm) at 1,113 (95% CI = 521 – 2,085), which was similar to the 2005 estimate. 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. Since 2003, the 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, with an estimate for 2006 of 2,088 (95% CI = 1,134 – 6,046; Figure 1.24). The abundance of bull trout > 220 mm has varied substantially over this period, from 343 in 2004 down to 61 in 2005, with the 2006 estimate of 216 fish. The abundance

estimate of large bull trout (> 370 mm) for 2006 was approximately 2 fish, which is the lowest since we began research in the NFUM (Figure 1.24). Overall, abundance estimates for the > 120 mm size category demonstrated high variability, and there was no discernable increasing or decreasing trend.

North Fork John Day River.—In 2006, we estimated abundance of bull trout >120 mm as 437 (95% CI = 274 – 752) in NFJDA and 1193 (95% CI = 825 – 2,509) in Baldy Creek (Figure 1.25). Estimates of all adult bull trout (<220 mm) were relatively low in both NFJDA (52 fish) and Baldy Creek (35 fish). Combining both the NFJDA and Baldy Creek estimates, the abundance of all bull trout was substantially smaller than SFWW estimate and relatively similar with the NFUM estimate (Figure 1.25).

Survival

SFWW.—Since 2002, we have marked 1782 bull trout >120 mm for mark-recapture survival analyses. Based on model selection, survival appeared to vary significantly across years, and bull trout condition had a significant positive effect on bull trout survival (Beta = 0.45, SE = 0.05). Average annual survival estimates for the six life stages of bull trout in the SFWW ranged from 0.16 (1 SE = 0.04) for 170 – 220 mm bull trout up to 0.49 (SE = 0.08) for 270 – 320 mm bull trout (Figure 1.26). Since 2002, there has been substantial variability in survival across years.

NFUM.—Since, 2003, we have marked 376 bull trout for mark-recapture survival analyses. We found annual survival in the NFUM (0.34, SE = 0.23) to be generally similar to estimates for bull trout in the SFWW of similar size (120 – 170 mm; 0.28, SE = 0.06); however, there was very high variance associated with the NFUM estimates, which resulted from generally low capture probabilities (<0.10).

Diet Analyses

Using gastric lavage techniques and dissected stomachs of sacrificed fish, we quantified diet information from 30 bull trout from the SFWW and 13 bull trout from the NFJDA in 2006. The primary prey items in both streams in June and August were aquatic macroinvertebrates, which represented 59 – 78% of diets (Figure 1.27). Aquatic macroinvertebrates included chironomids, plecopterans, dipterans, trichopterans, ephemeropterans, and coleopterans. Fish prey included sculpin, *O. mykiss*, and salmonids, and represented 12 – 24 % of diets depending on stream and time period (Figure 1.27). Rare prey included terrestrial insects, fish eggs, worms, spiders, and gastropods. In both the upper reaches (near Reser Creek) and lower reaches (near Bear Creek) of the SFWW in June 2006, there was significant variability

in the diet composition: aquatic invertebrates composed 46 – 85% of diets, prey fish (primarily *O. mykiss*) composed 0 – 19% of diets, and terrestrial invertebrates composed 0 – 35% of diets (Figure 1.28). In the upper reaches, there was a temporal increase in prey fish consumption and a decrease in aquatic invertebrate consumption from June to August. Bull trout diets from the lower reaches suggested an increase in invertebrate consumption (both aquatic and terrestrial), and a decrease in prey fish consumption.

We also compared the diets of bull trout captured in the SFWW in 2003, 2005, and 2006 (Figure 1.29). Over this period, we found bull trout diets were consistently composed of macroinvertebrates (range = 57 – 75%). Similar to 2005, we observed no evidence of cannibalism in bull trout diets in 2006. In 2006, bull trout diets consisted of 20% prey fish, which was lower than 2003 (37%) and 2005 (23%) diet analyses.

We observed little annual difference in bull trout diets across years (2005 – 2006) in the NFJDA. Macroinvertebrates were the most prevalent dietary item (76 – 78%), while prey fish (14 – 17%) and terrestrial invertebrates (7 – 8%) were significantly less prevalent in bull trout diets over this time period.

Temperature

We measured temperature using temperature loggers at four sites in the SFWW, two sites in the NFUM, and two sites in the NFJDA from June 2005 to June 2006. Vandalism to the Reser Creek temperature logger resulted in incomplete temperature data for 2005 – 2006 (logger found in campfire near stream). In the SFWW, daily min/max temperatures varied little across the year in the SFWW below Skiphorton Creek (annual range = 2.4 – 8.7 °C), but varied greatly at the bottom of the study at the Harris Park Bridge (annual range = 0.1 – 16.1 °C; Figure 1.31). In the NFUM from June 2004 to June 2005, min/max temperature ranged from 0.2 – 15.2 °C above the confluence with Coyote Creek, and ranged from 1.9 – 15.0 °C at the NFUM Campground (Figure 1.32). In the NFJDA, there was substantial variation in min/max temperatures above the confluence with Baldy Creek (annual range = -0.1 – 16.5 °C) and near the North Fork John Day Campground (annual range = 0.2 – 20.7 °C; Figure 1.33).

Movement

We quantified movement in the NFUM (2005 – 2006) using mark-recapture techniques and passive PIT-tag antennae. In 2005, we detected 30 bull trout at the NFUM

antenna (UM1), and in 2006, we detected 13 bull trout (Figure 1.34a). Across years, we observed bull trout movement in most months, but the majority of movement appears to occur during spring and fall months. Similar to the SFWW (Budy et al. 2006; Homel 2007), the majority of fish movement in 2005 (90%) and 2006 (77%) occurred during nighttime hours (i.e., before sunrise and after sunset). Finally, we found considerable periods of time where the antenna was not in operation due to power outages (Figure 1.34), and suggest future interpretations of movement patterns incorporate these gaps in data collection.

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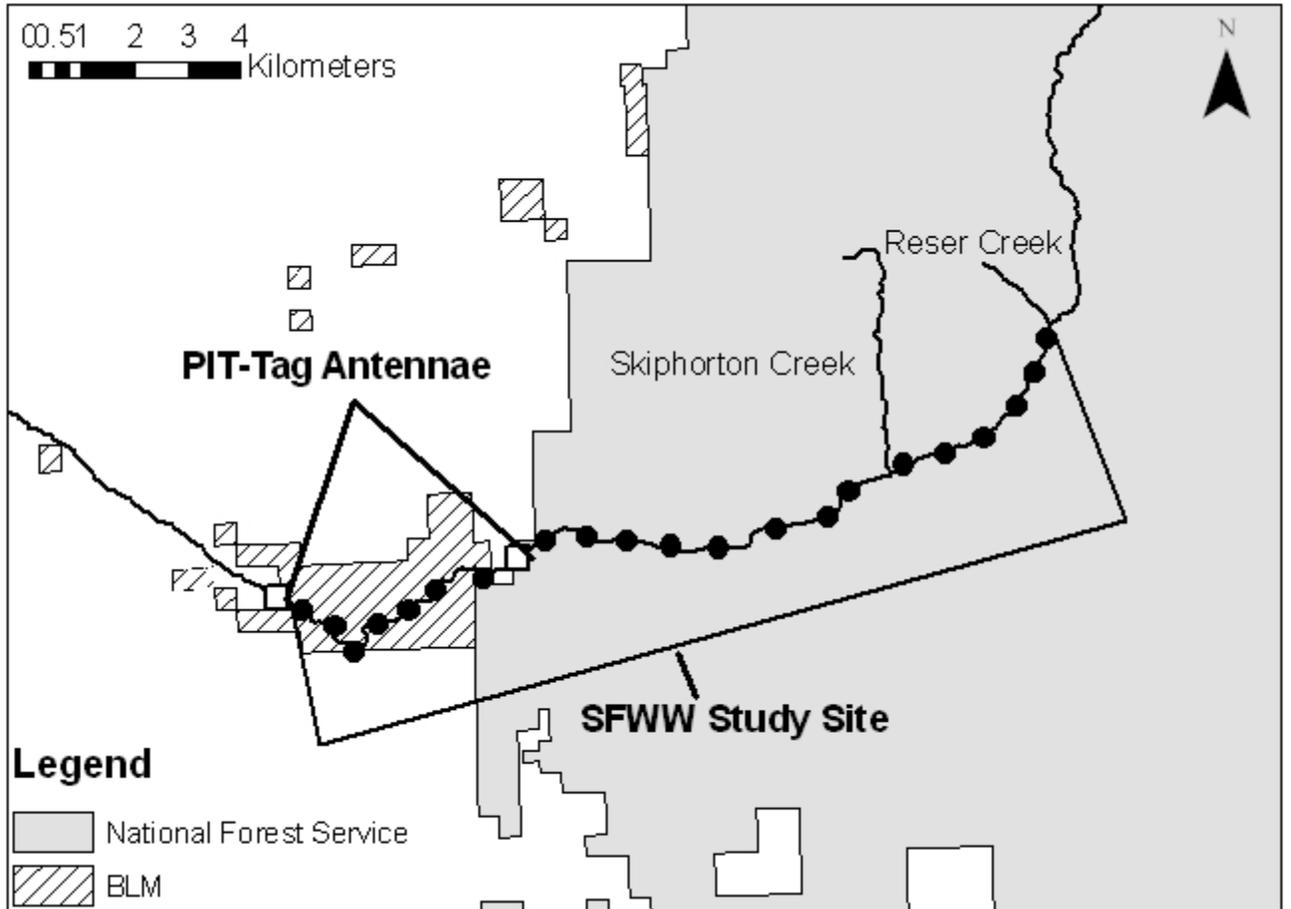


Figure 1.1. Map of the South Fork Walla Walla River showing original 22 study reaches (dark circles) and antennae locations (white squares).

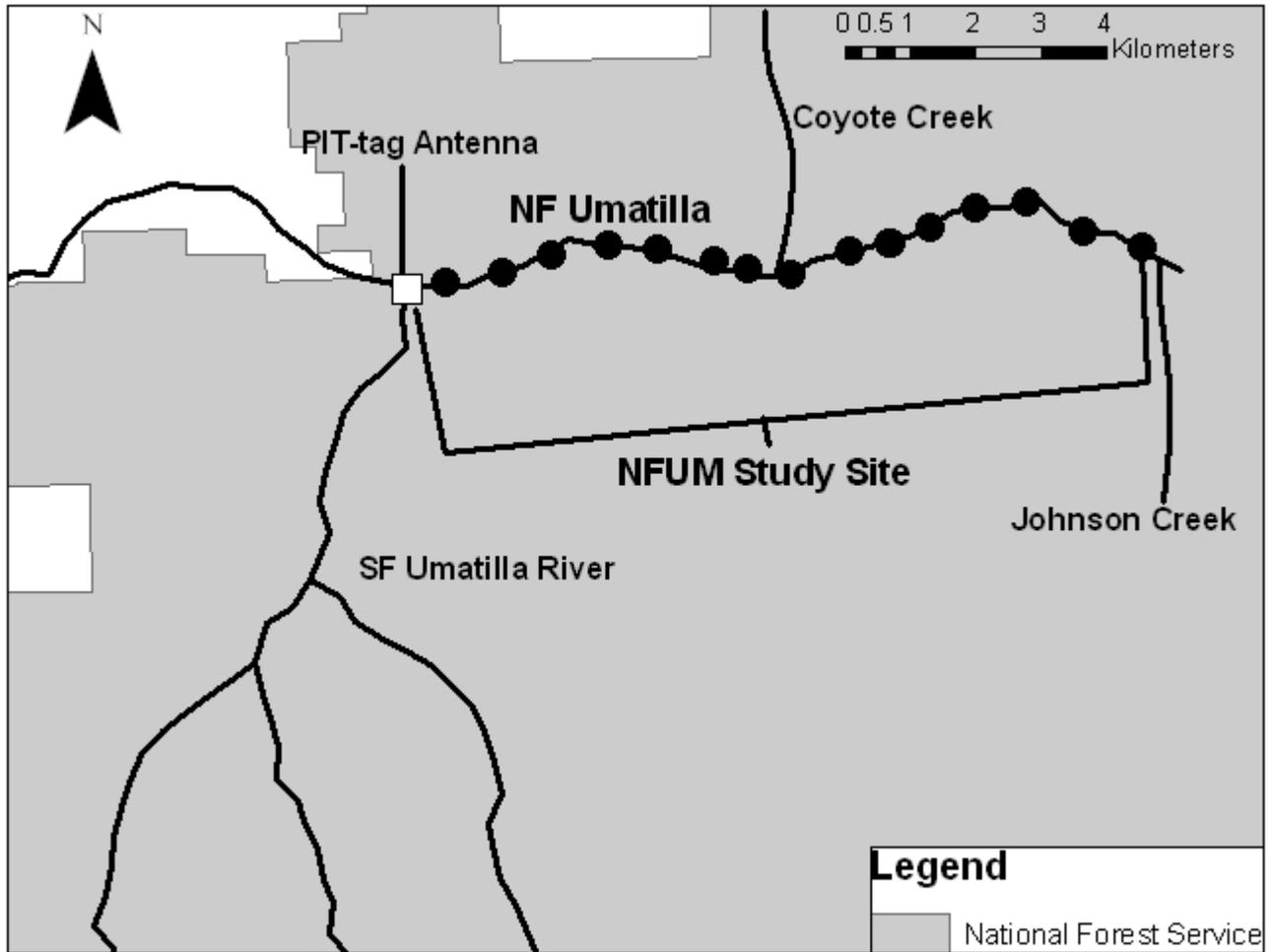


Figure 1.2. Map of the North Fork Umatilla River showing the 15 study reaches (dark circles) and antenna location (white square).

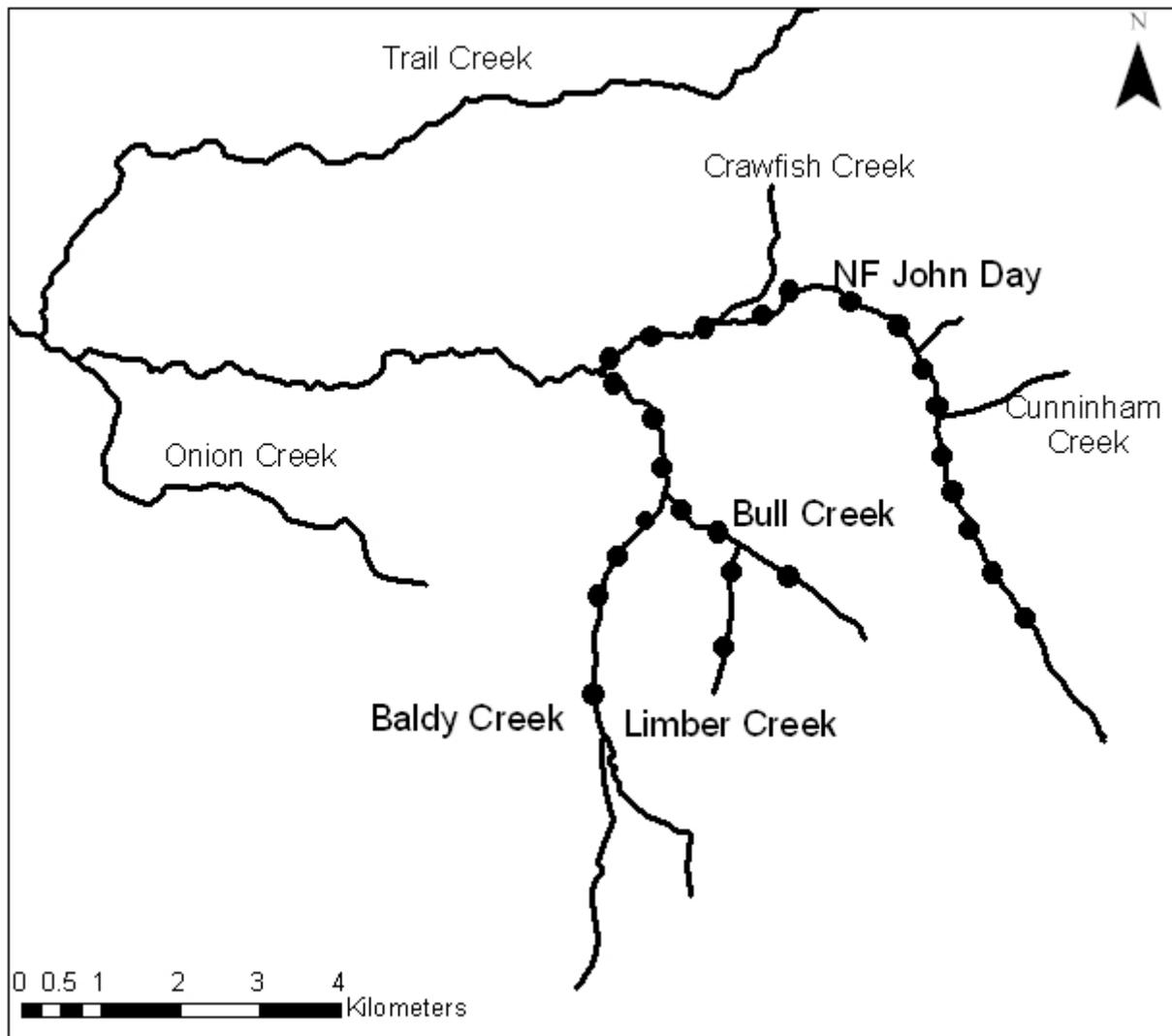


Figure 1.3. Map of the North Fork John Day River (NFJDA) and Baldy Creek showing the 2006 sampling locations (dark circles).

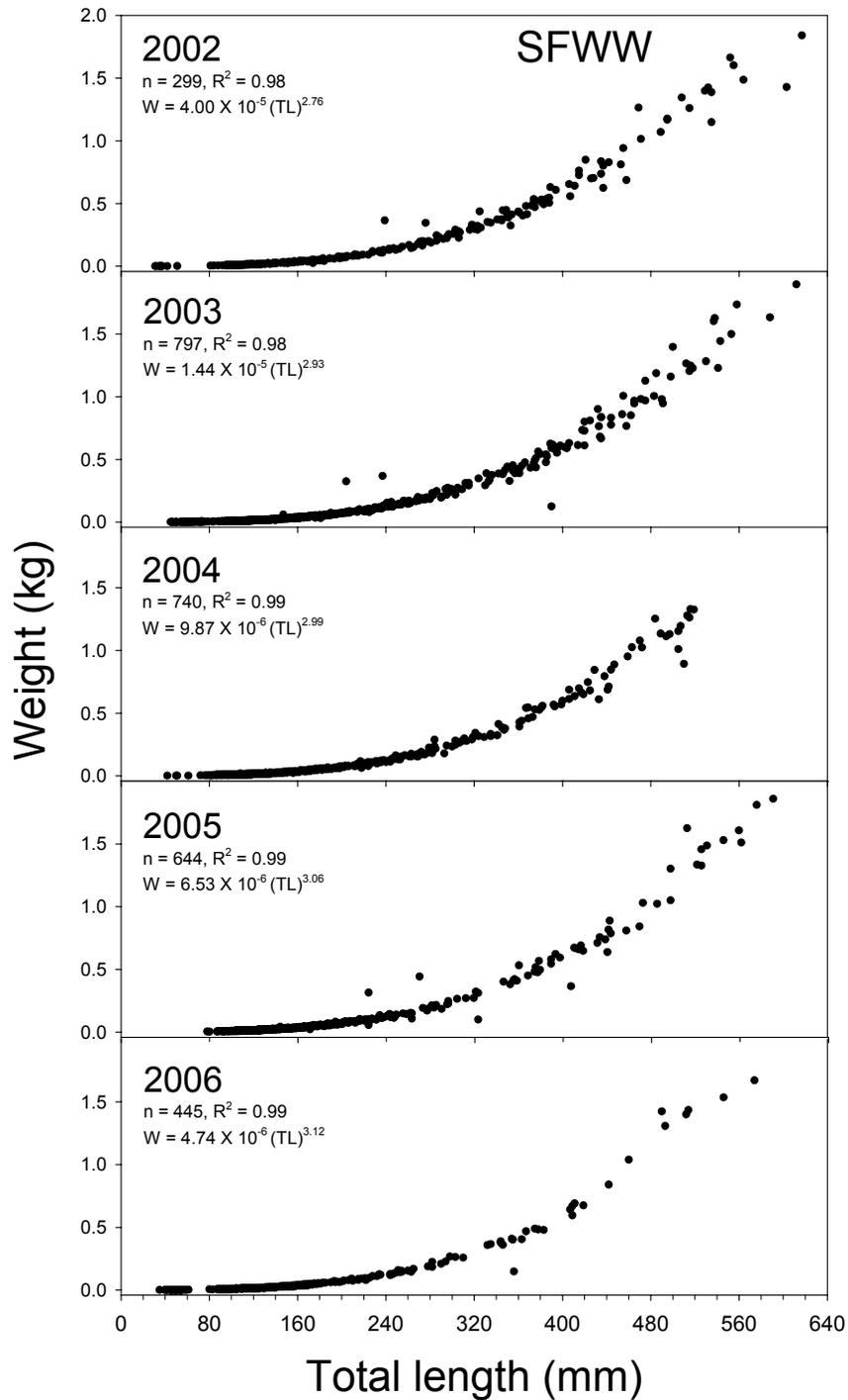


Figure 1.4. Length-weight relationship for all bull trout captured and handled in the South Fork Walla Walla River, 2002 - 2006. Regression equations and sample sizes are given.

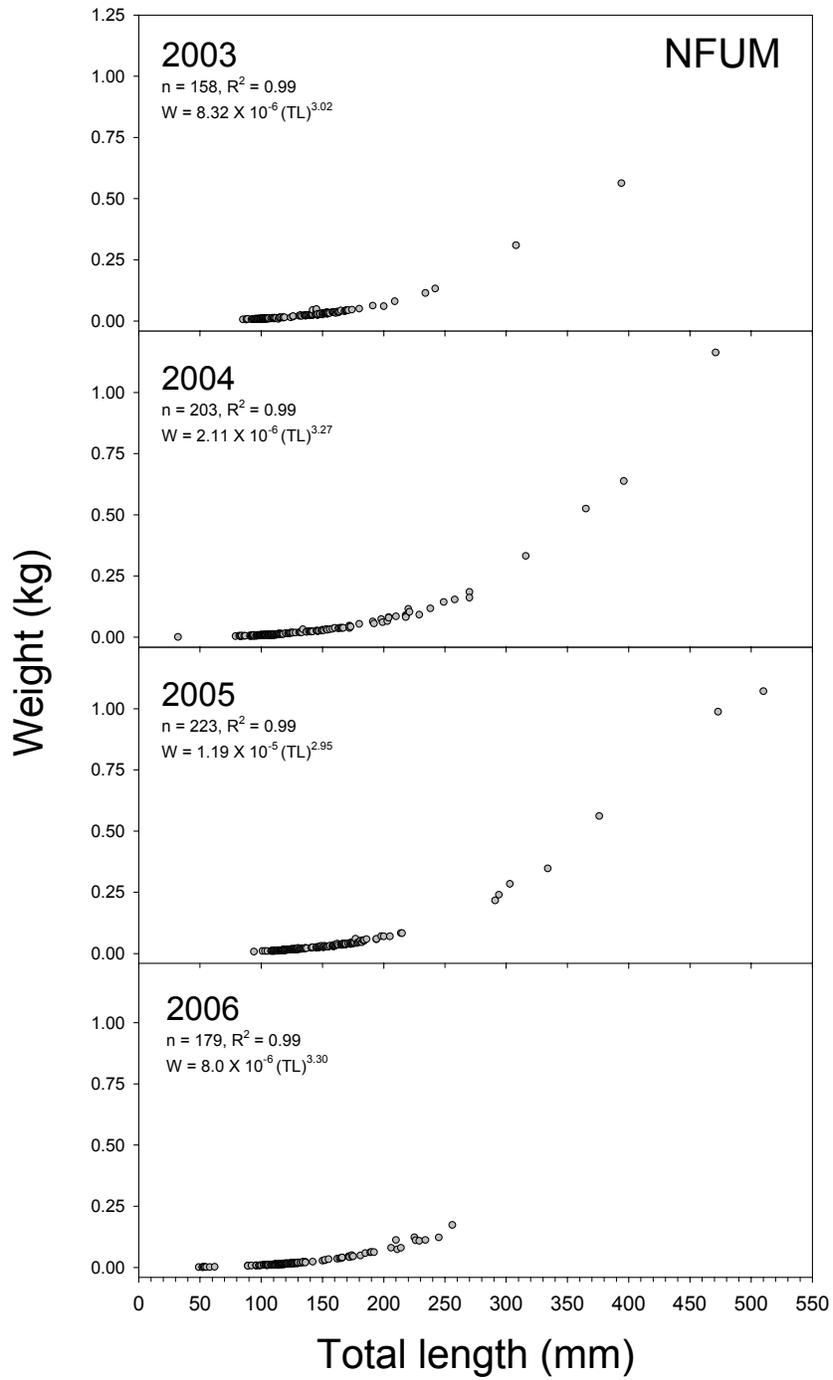


Figure 1.5. Length-weight relationship for bull trout captured and handled in the North Fork Umatilla River, 2003 - 2006. Regression equations and sample sizes are given.

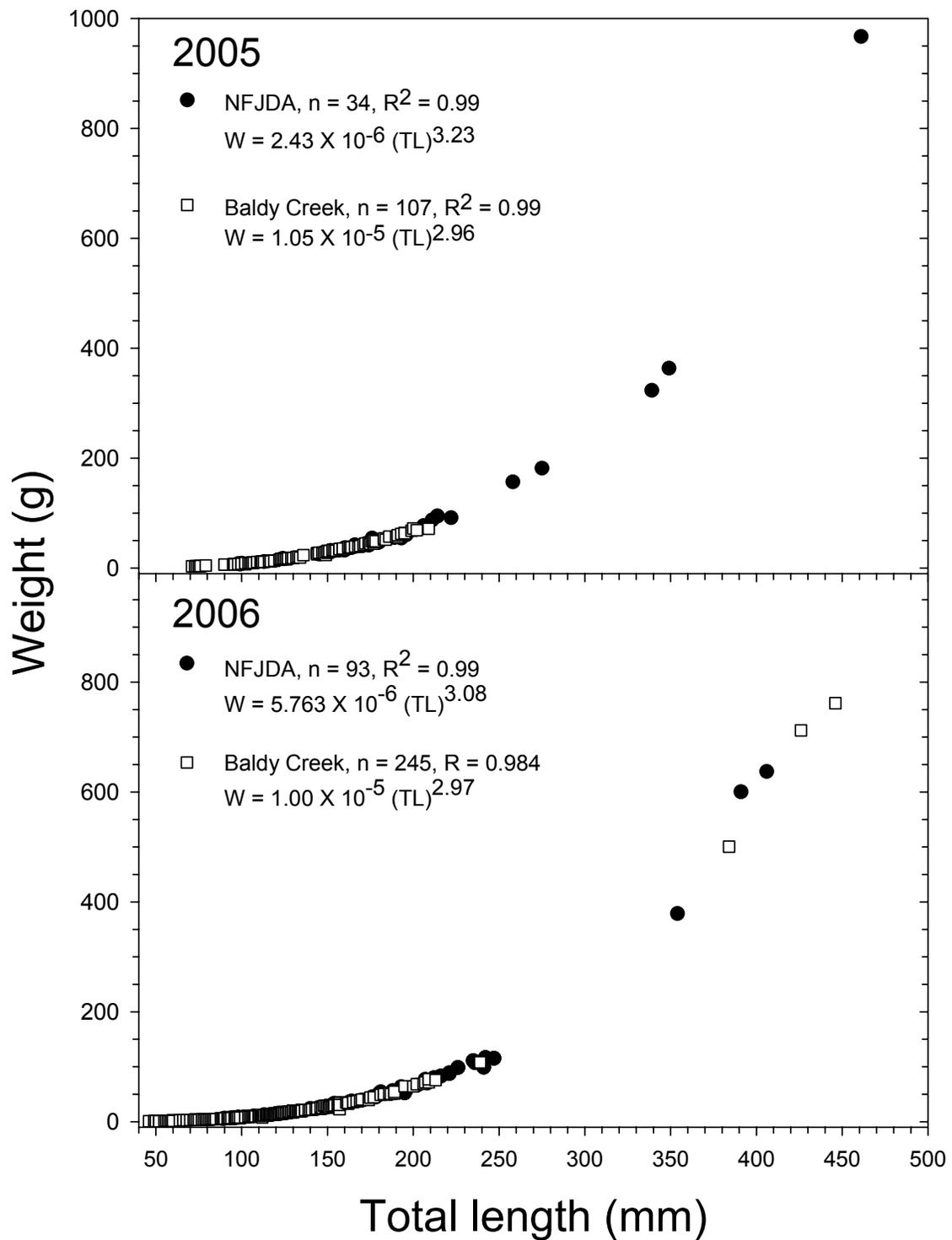


Figure 1.6. Length-weight relationship for bull trout captured and handled in the North Fork John Day River (black circles) and Baldy Creek (open squares), 2005 – 2006.. Regression equations and sample sizes are given.

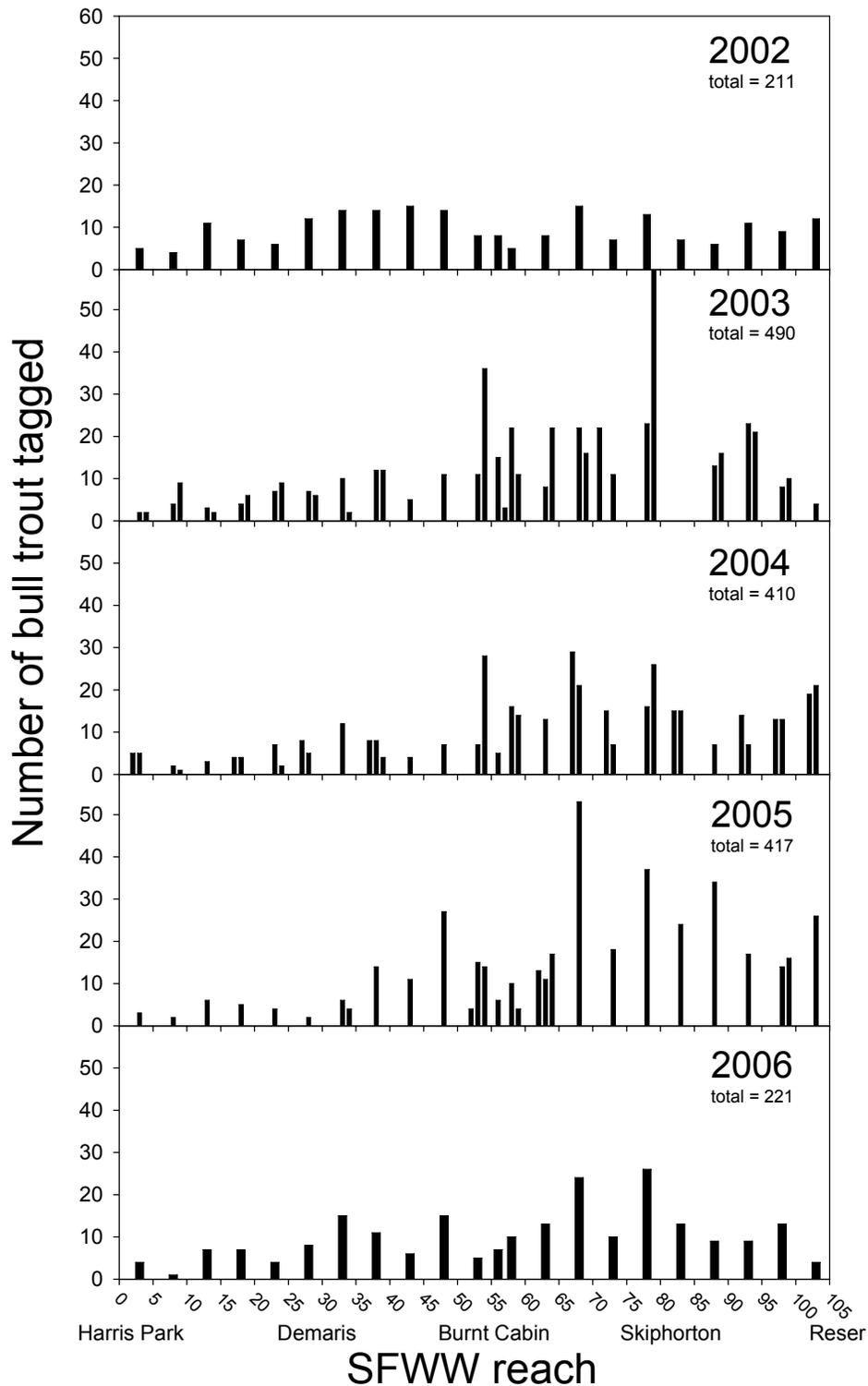


Figure 1.7. Number of bull trout tagged by reach in the South Fork Walla Walla River, 2002 - 2006. Reaches are numbered from bottom to top of the study site. Total numbers tagged are given below sample year.

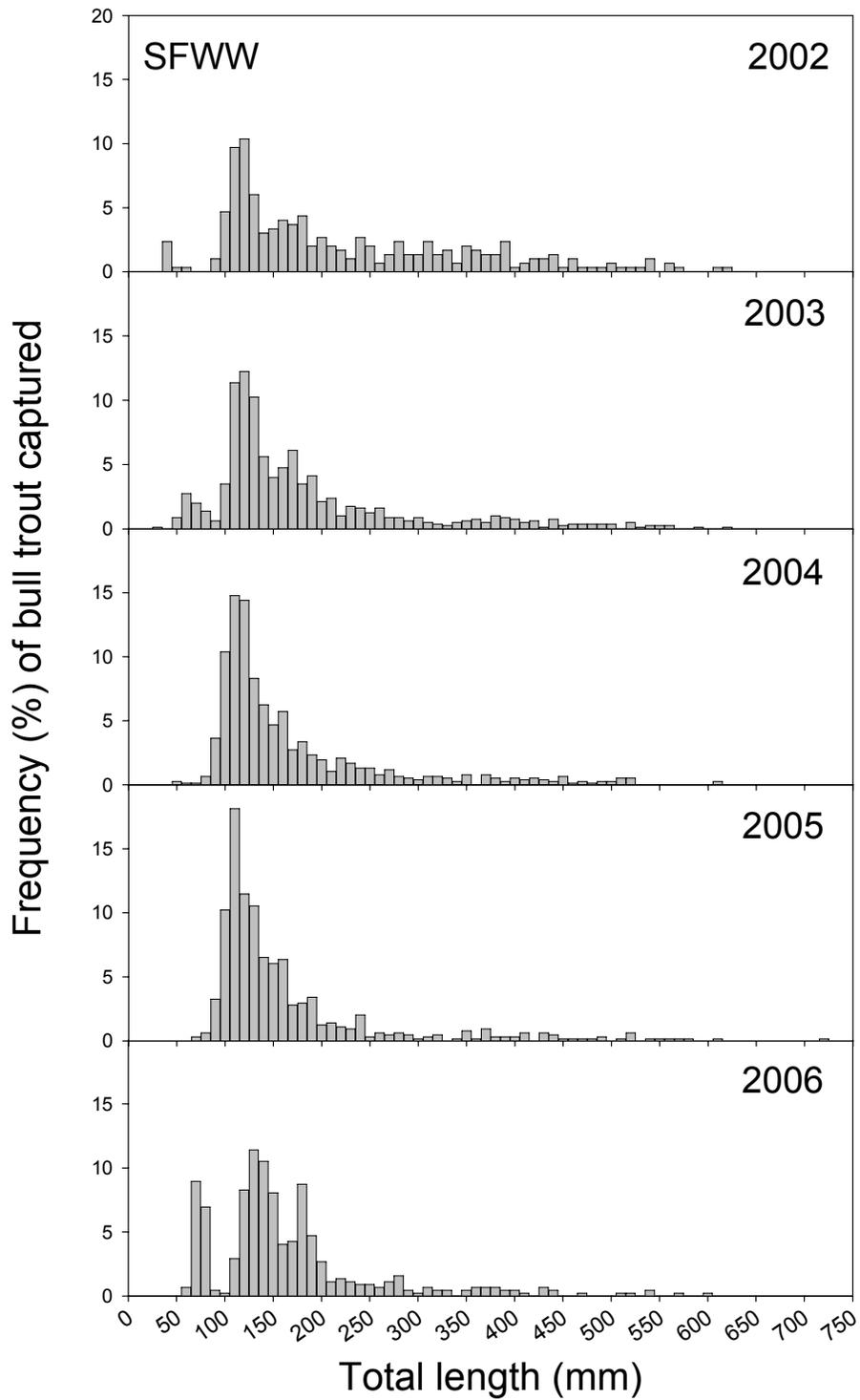


Figure 1.8. Length frequency (% of catch) distribution of bull trout captured and handled in the South Fork Walla Walla River, 2002 - 2006.

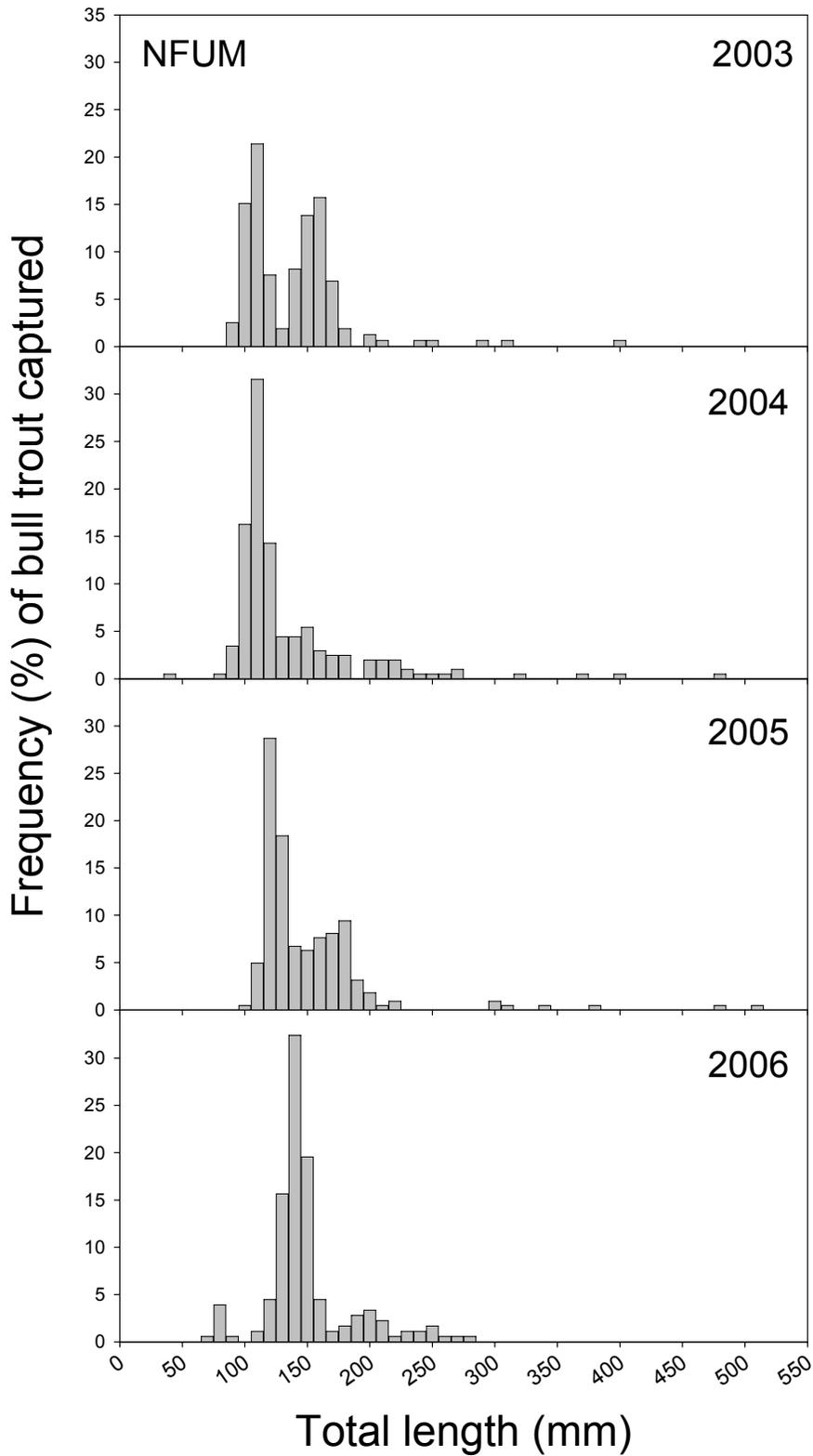


Figure 1.9. Length frequency (% of catch) distribution of bull trout captured and handled in the North Fork Umatilla River, 2003 - 2006.

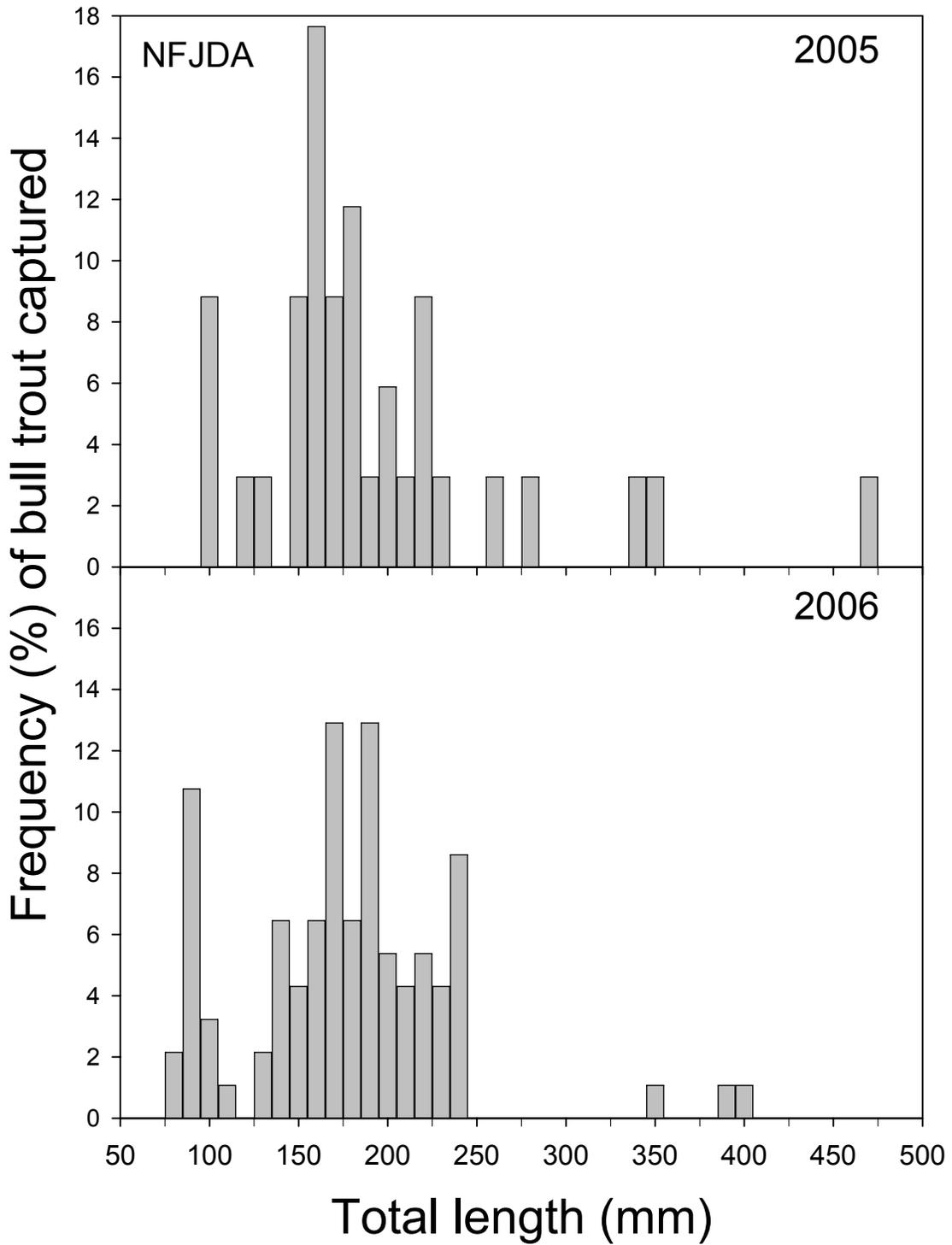


Figure 1.10. Length frequency (%) distribution of bull trout handled in the North Fork John Day River in 2005 and 2006.

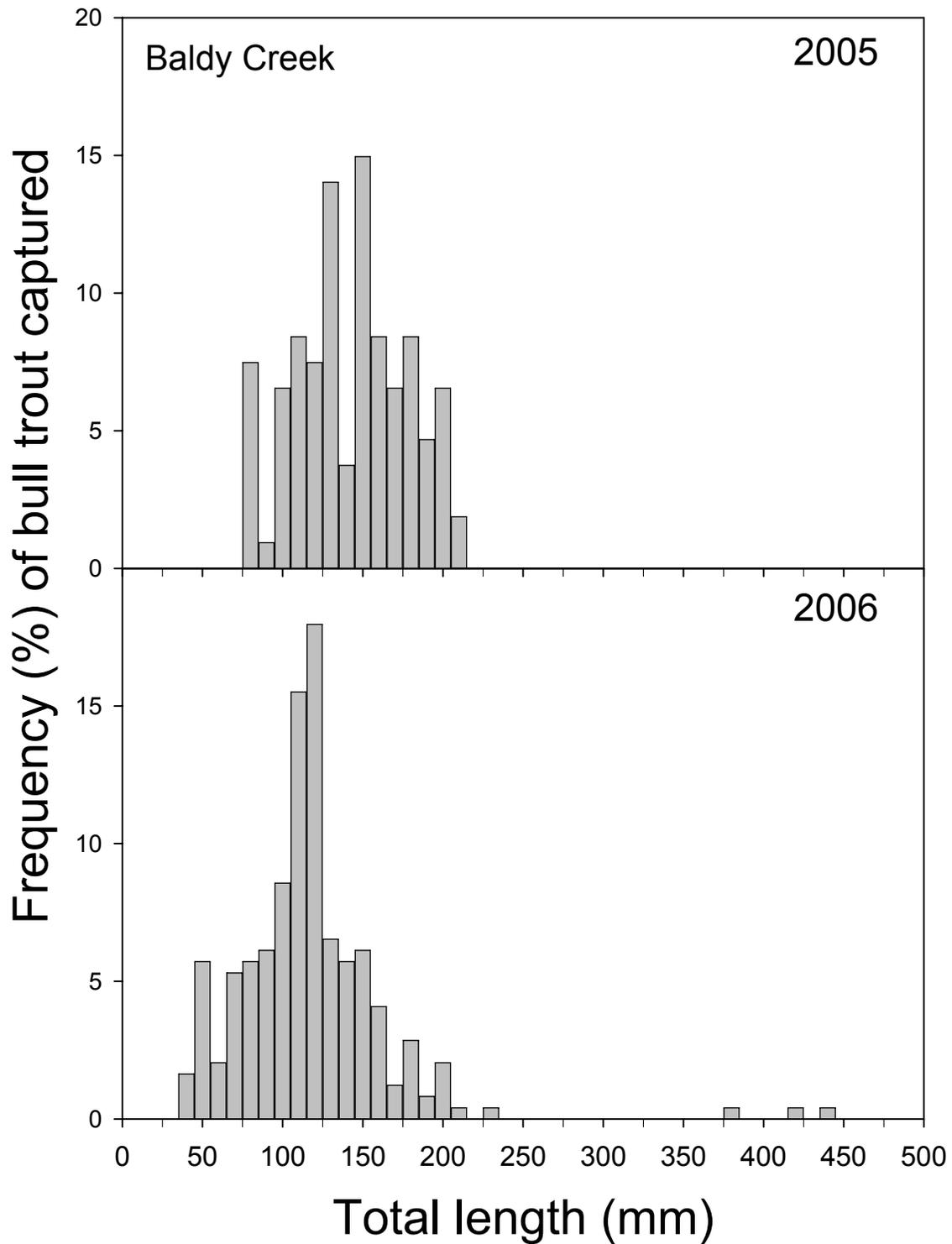


Figure 1.11. Length frequency (%) distribution of bull trout handled in Baldy Creek in 2005 and 2006. In 2006, Baldy Creek data includes fish captured in Bull and Limber creeks.

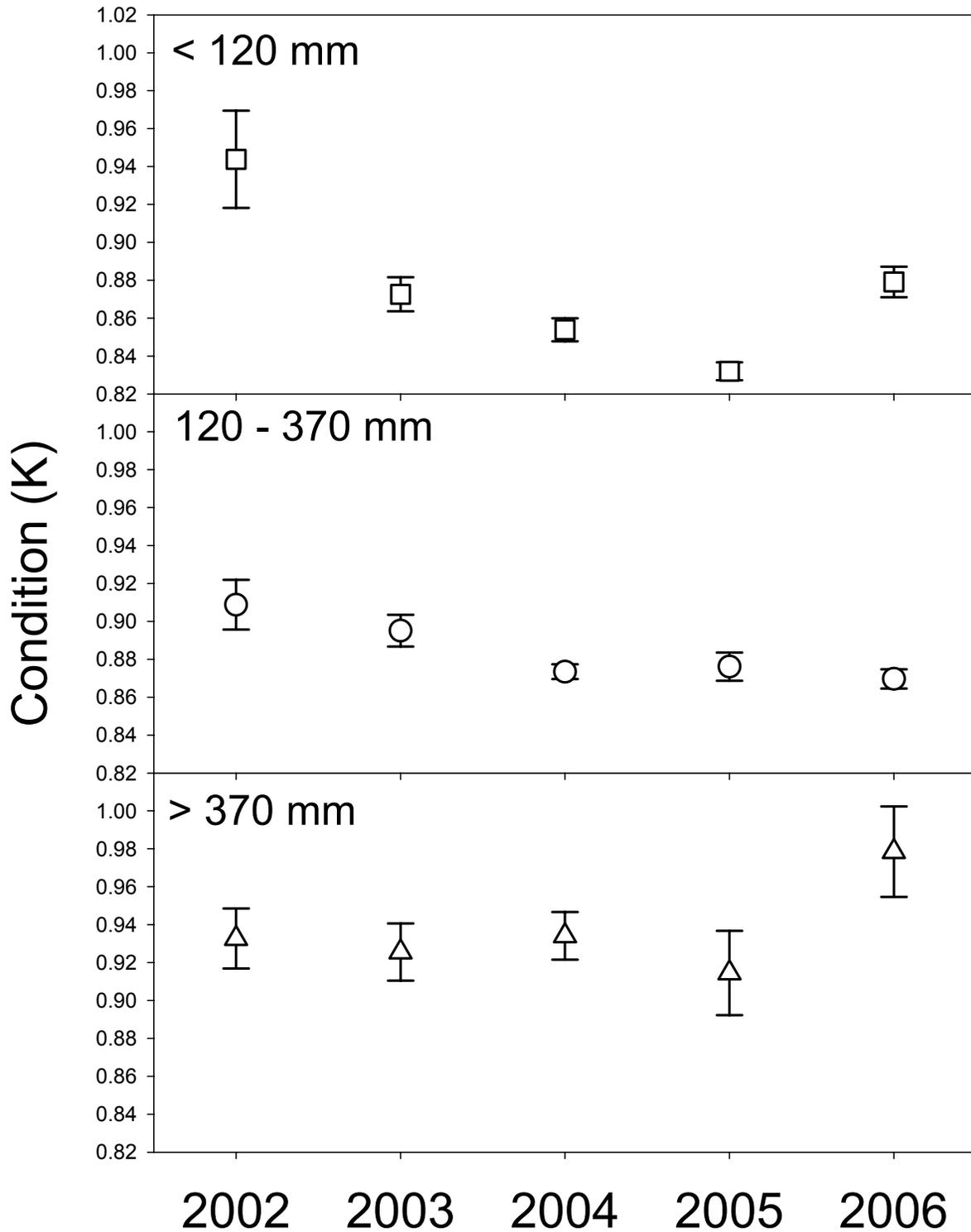


Figure 1.12. Condition (Fulton's $K \pm 1$ SE) of three different size classes of bull trout sampled in the South Fork Walla Walla River, 2002 - 2006. Note changes in scale on y-axis for the three plots.

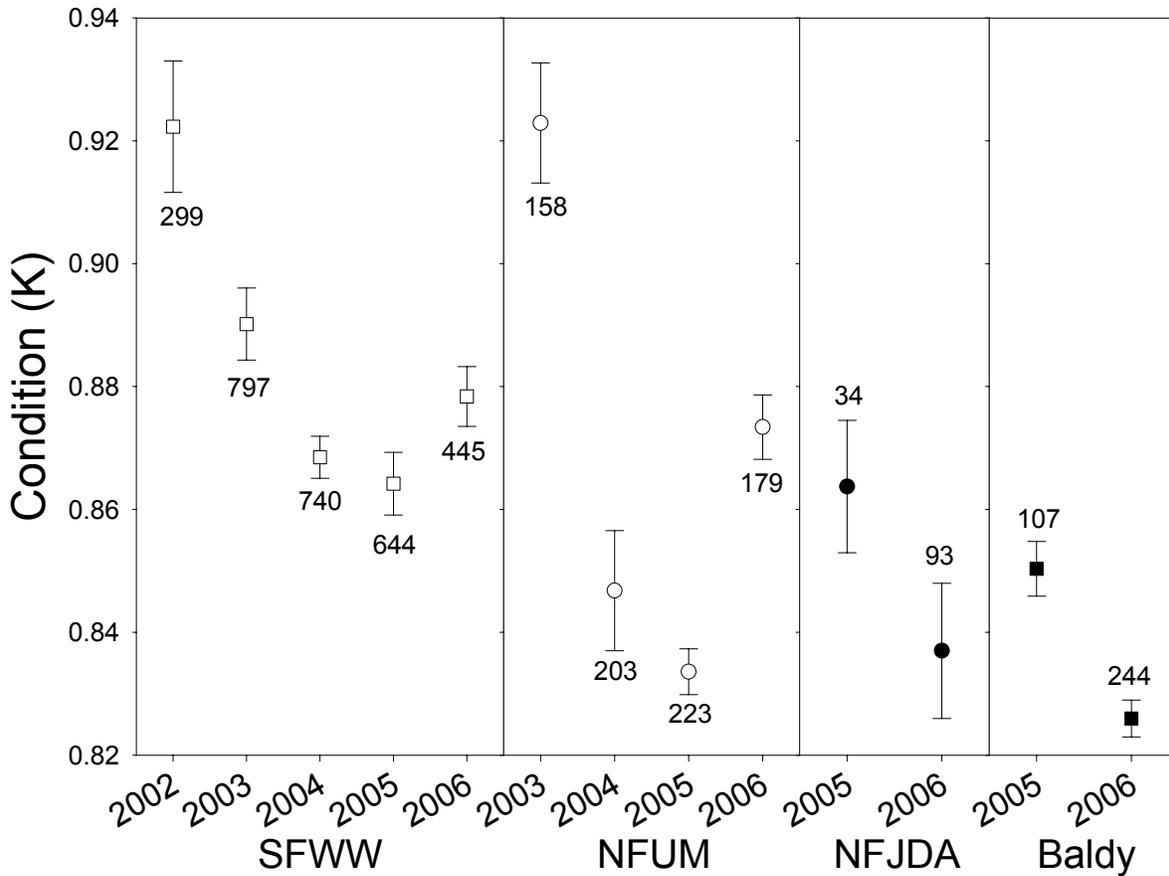


Figure 1.13. Average condition (Fulton's $K \pm 1$ SE) of bull trout (all sizes combined) sampled in the South Fork Walla Walla River (2002 - 2006), North Fork Umatilla River (2003 - 2006), North Fork John Day River (2005 – 2006), and Baldy Creek (2005 – 2006). Note: Baldy includes bull trout from Baldy and Bull creeks within the Baldy Creek Basin. Sample size is given by error bars.

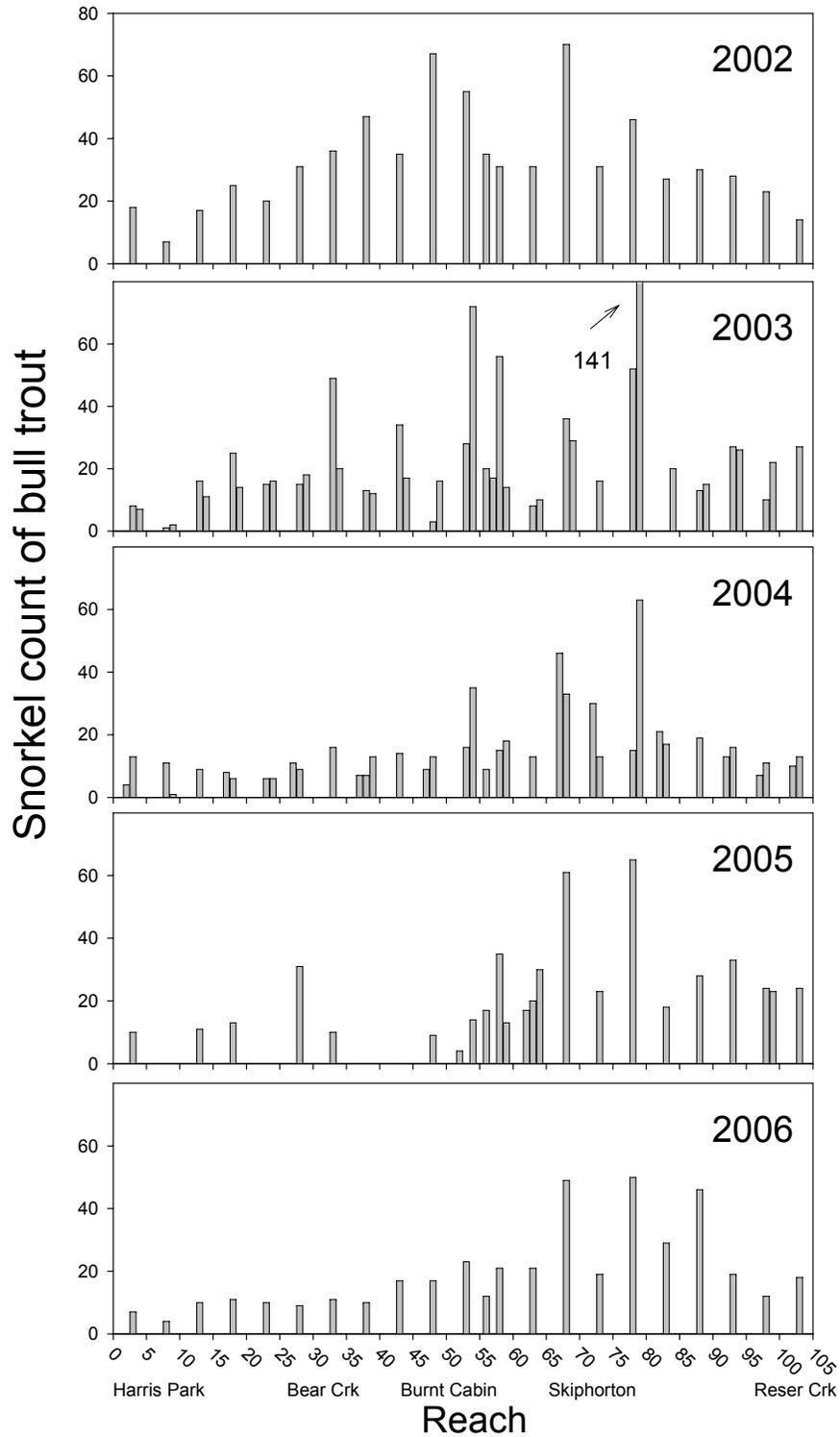


Figure 1.14. Number of bull trout by reach counted during snorkel surveys in the South Fork Walla Walla River, 2002 - 2006. Reaches are numbered from bottom to top of the study site. No bar implies that no sampling was conducted in a particular reach.

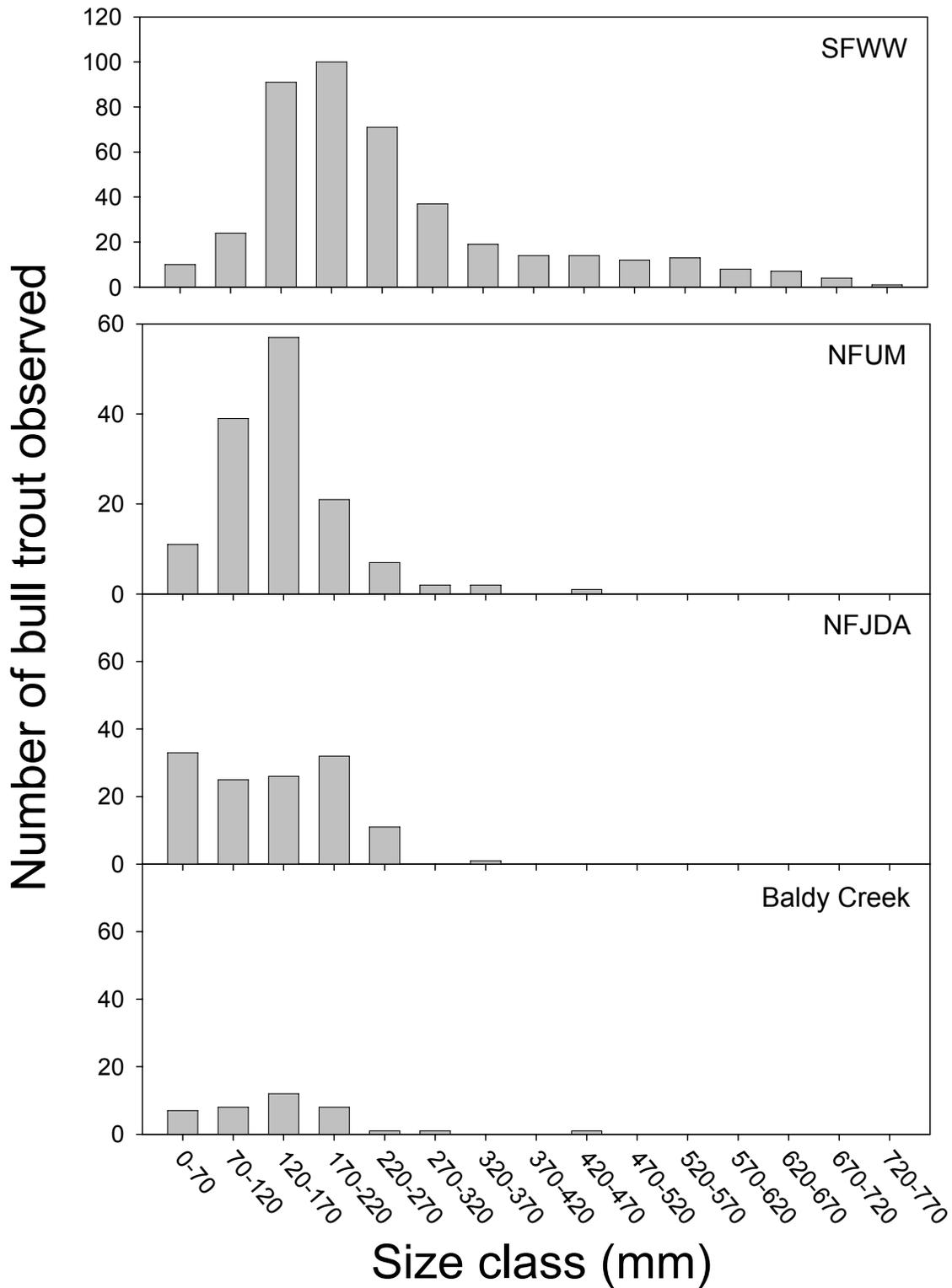


Figure 1.15. Number of bull trout in 70-mm size bins observed during snorkel surveys in the South Fork Walla Walla River, North Fork Umatilla River, North Fork John Day River, and Baldy Creek, 2006. Note changes in y-axis scales.

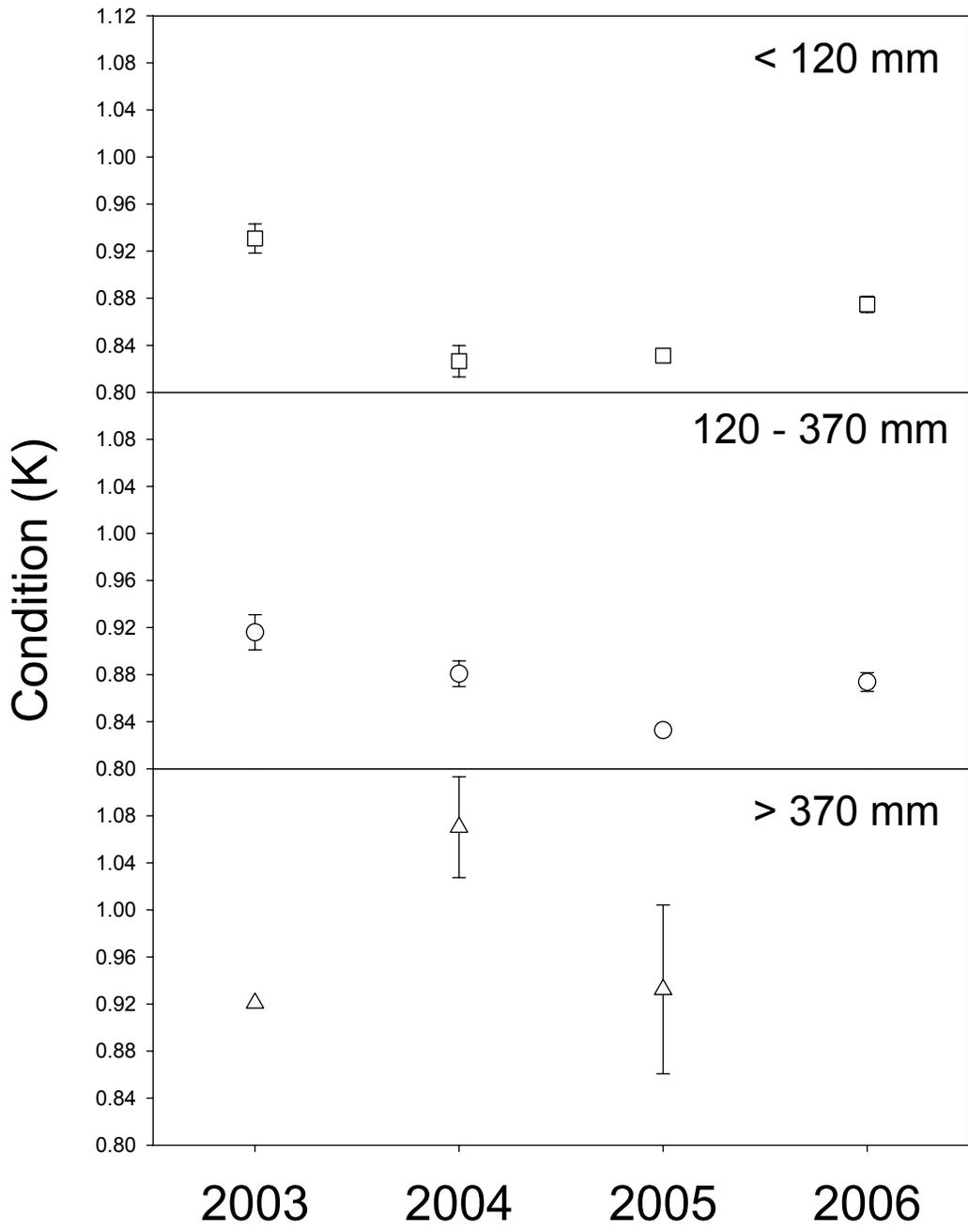


Figure 1.17. Condition (Fulton's $K \pm 1$ SE) of three different size classes of bull trout sampled in the North Fork Umatilla River, 2003 - 2006. Note: no bull trout >370 mm were capture in 2006.

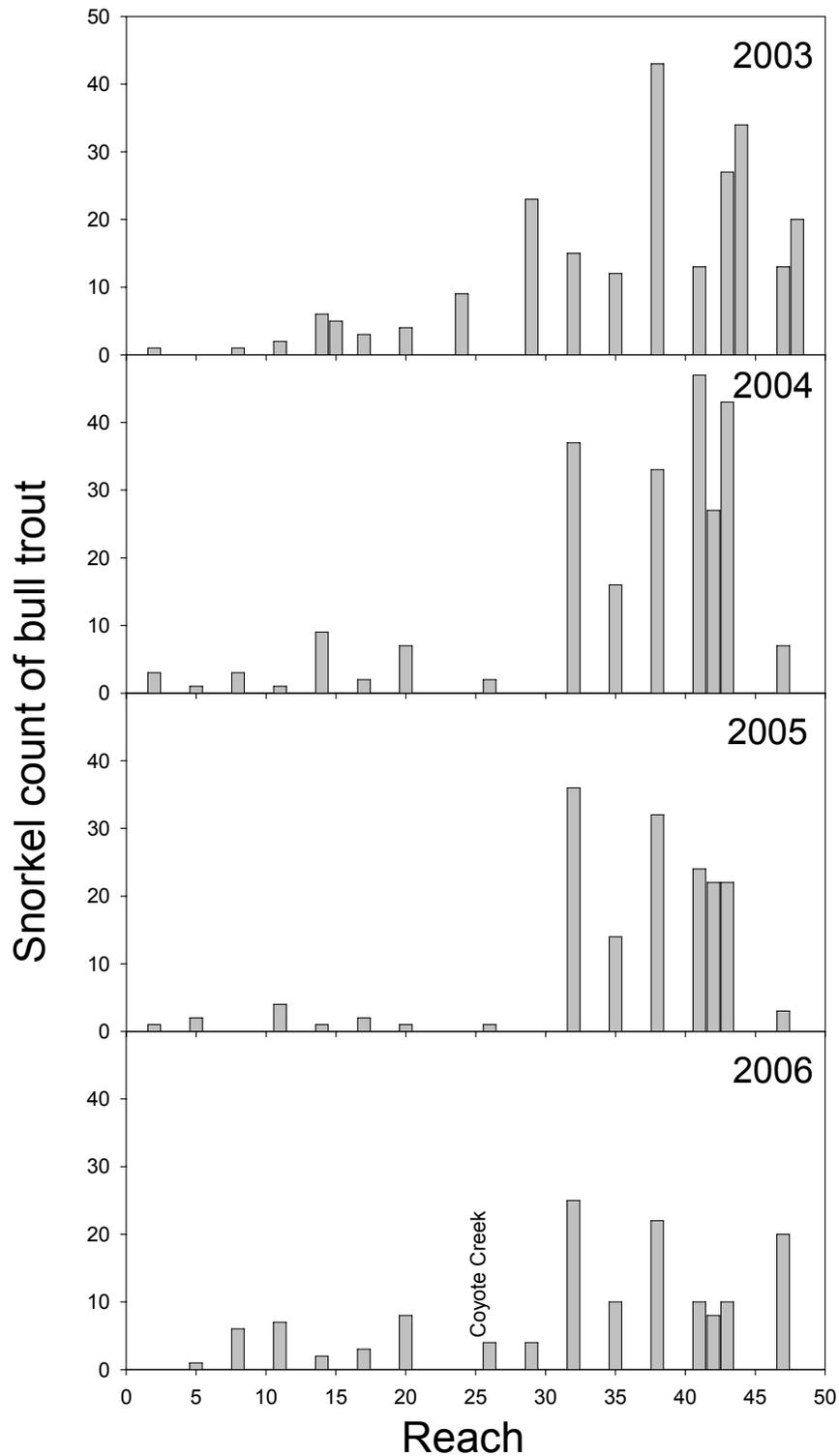


Figure 1.18. Number of bull trout counted by reach during snorkel surveys in the North Fork Umatilla River, 2003 - 2006. Reaches are numbered from bottom to top of the study site

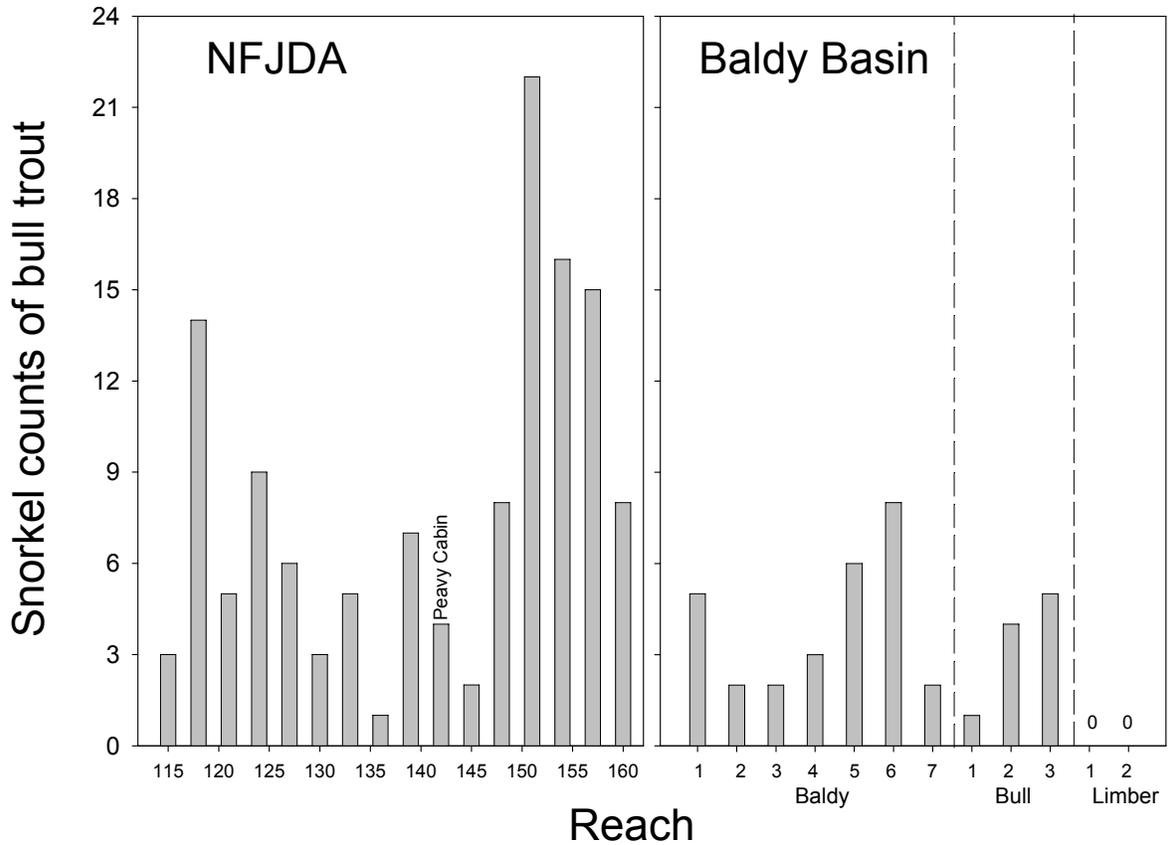


Figure 1.19. Number of bull trout (> 120 mm) tagged by reach in the North Fork John Day River and Baldy Basin, 2006. Reaches are numbered from bottom to top of the study site, and the dashed lines delineate the Baldy, Bull, and Limber creeks within the Baldy basin. Note zeros in Limber Creek.

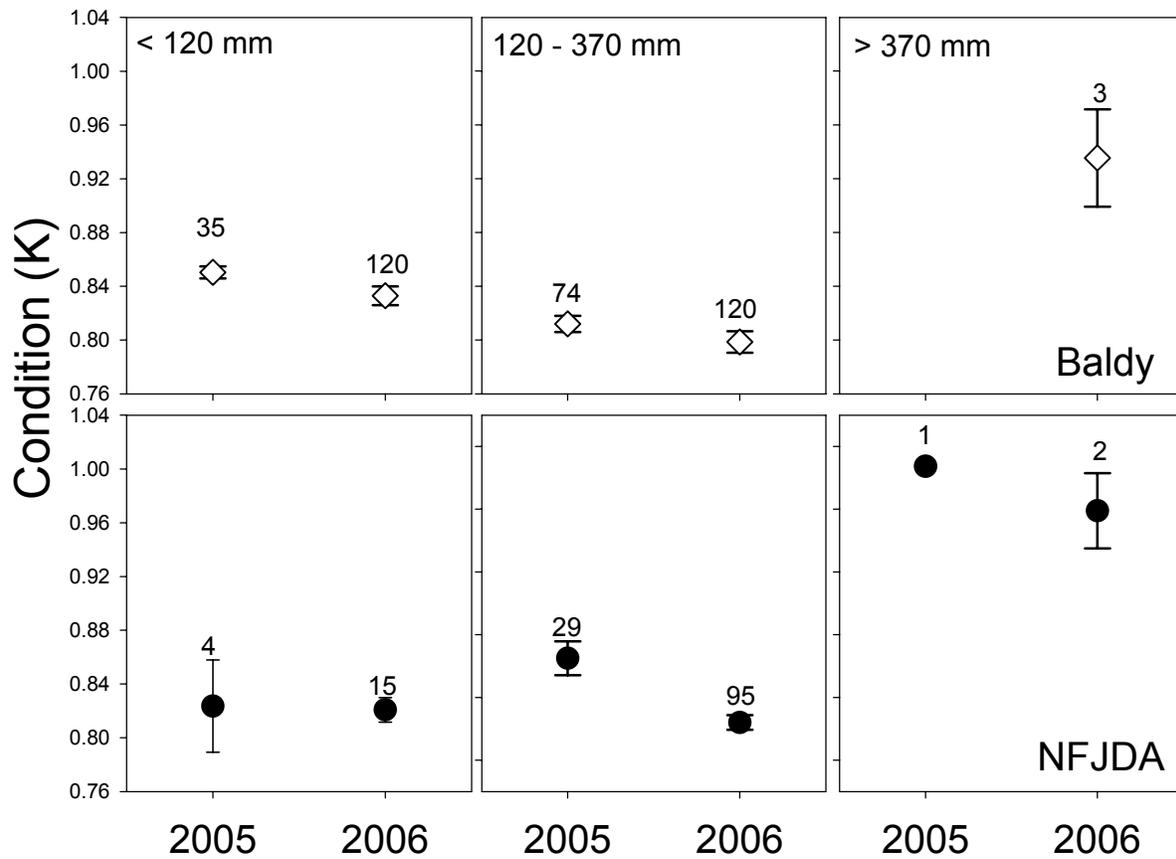


Figure 1.20. Condition (Fulton's $K \pm 1$ SE) of bull trout by size class sampled in the North Fork John Day River and Baldy Creek (2005 – 2006). Sample sizes are given by error bars.

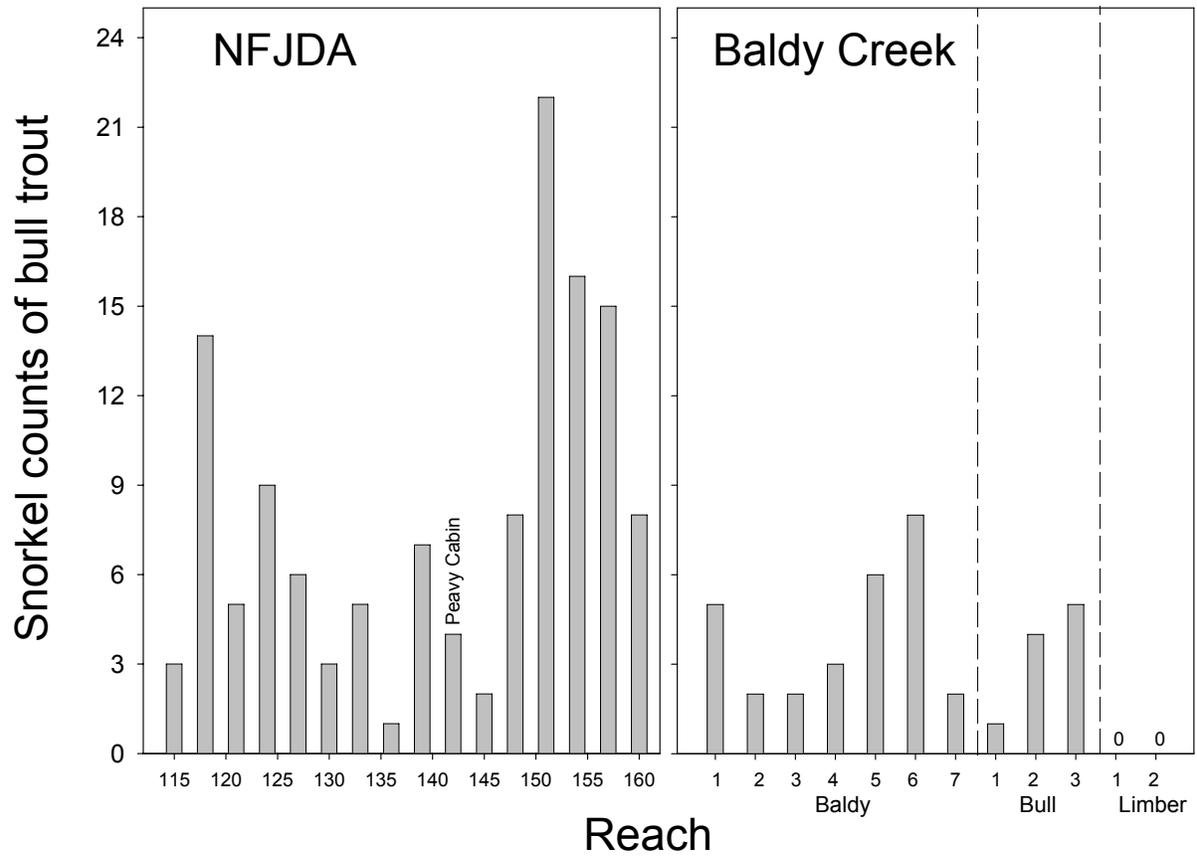


Figure 1.21. Number of bull trout by reach counted during snorkel surveys in the North Fork John Day River and Baldy Creek, 2006. Reaches are numbered from bottom to top of the study site, and Baldy Creek is delineated in Baldy, Bull, and Limber Creeks. Note zeros in Limber Creek.

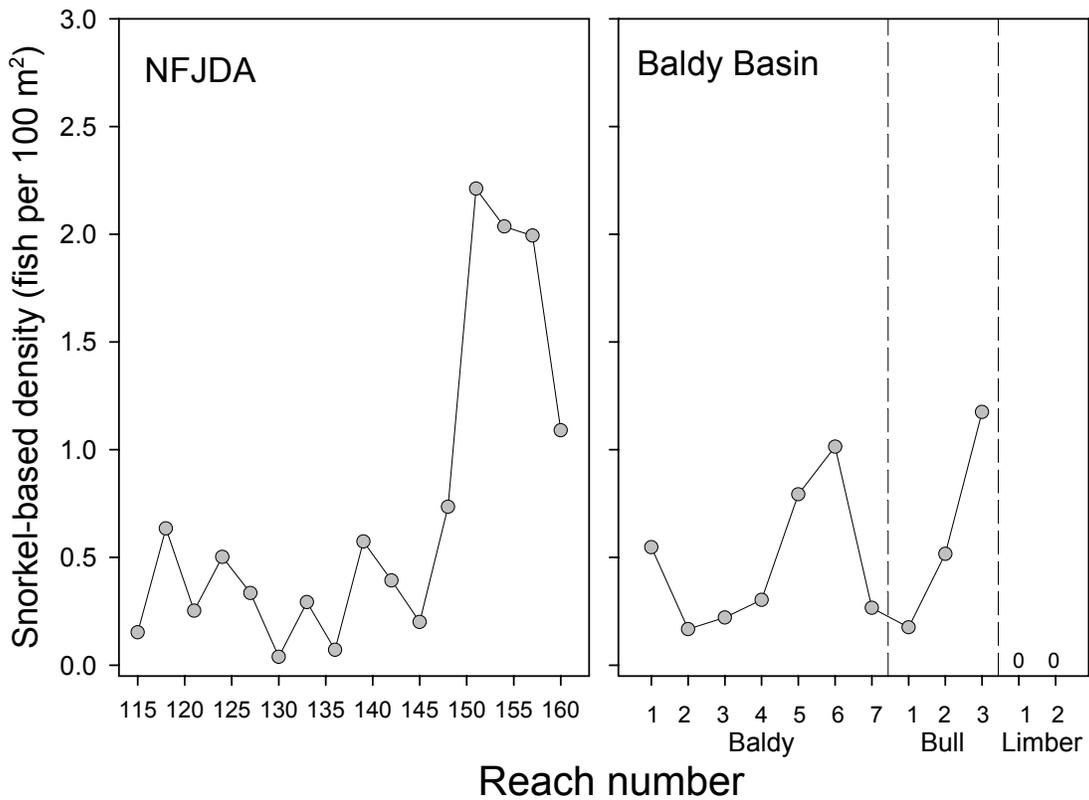


Figure 1.22. Density of bull trout by reach observed during snorkel surveys in the North Fork John Day River and Baldy Creek, 2006. Reaches are numbered from bottom to top of the study site, and the dashed lines delineate the Baldy, Bull, and Limber creeks within the Baldy basin. Note zeros in Limber Creek.

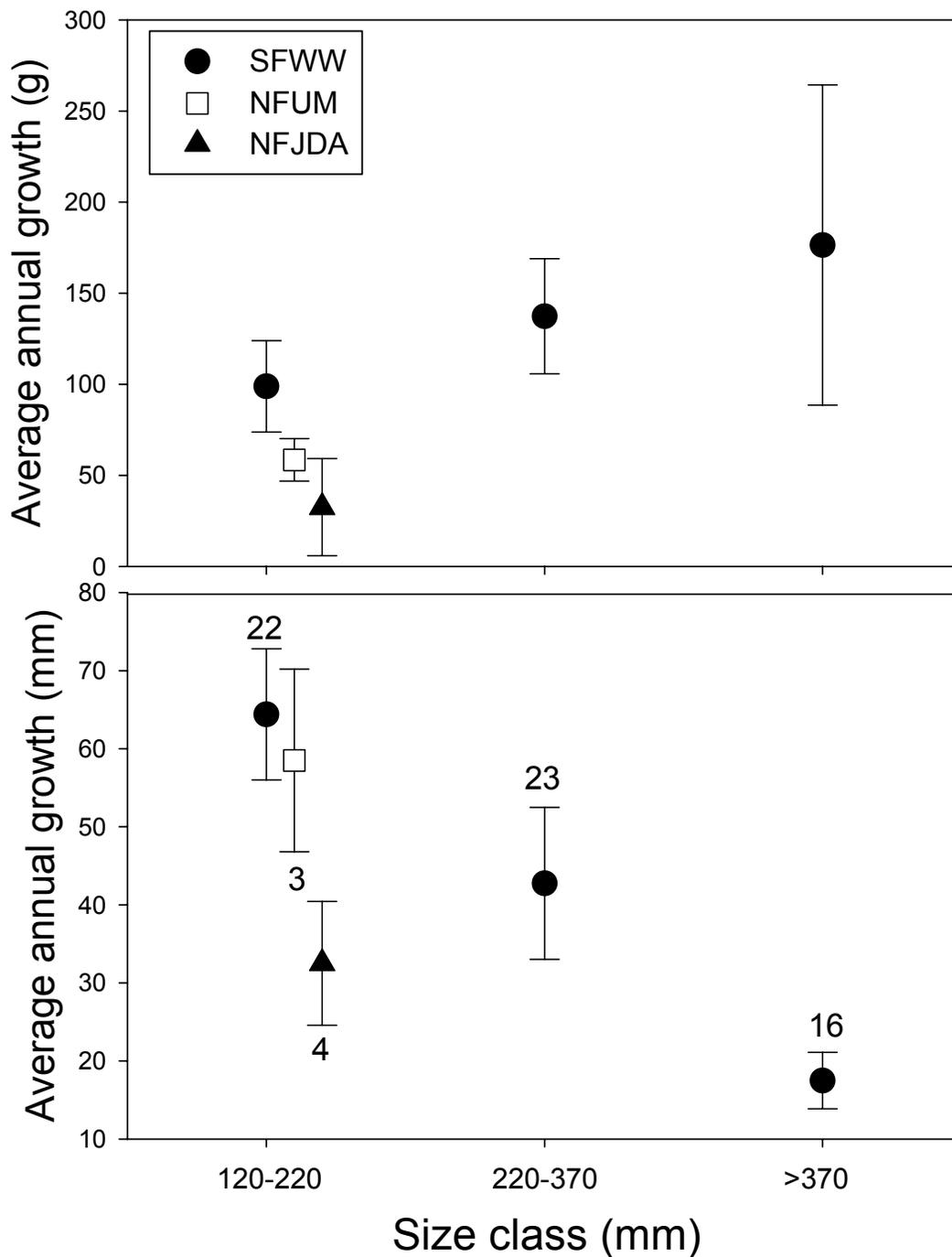


Figure 1.23. Average annual growth (± 2 SE) in weight (g, top panel) and length (mm, bottom panel) for three size classes of tagged and recaptured bull trout in the South Fork Walla Walla (2002 – 2006), North Fork Umatilla (2003 – 2006), and North Fork John Day rivers (2005 – 2006). Sample sizes are given above error bars. Note: no bull trout > 220 mm have been recaptured for growth estimates in the NFUM and NFJDA.

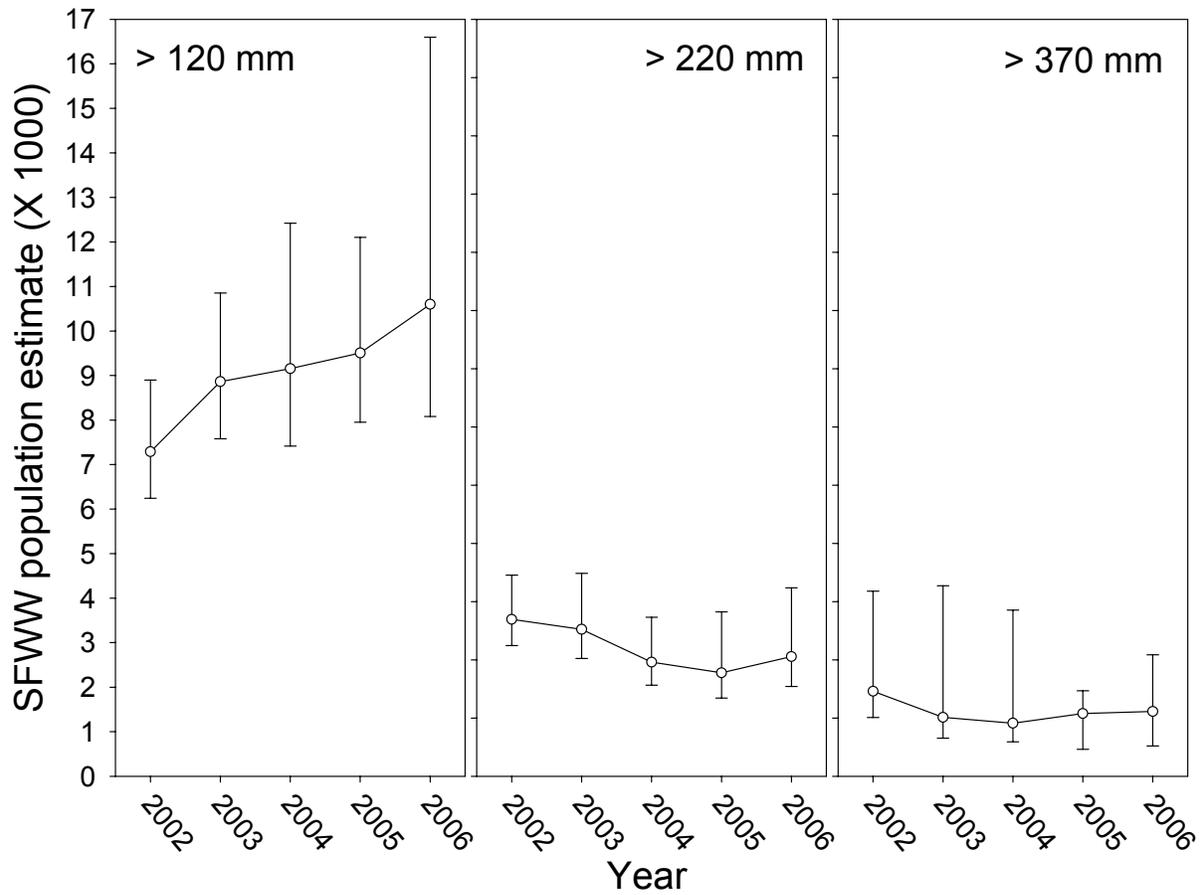


Figure 1.24. Yearly population estimates (\pm 95% CI) for three size groupings of bull trout in the South Fork Walla Walla River, 2002 - 2006.

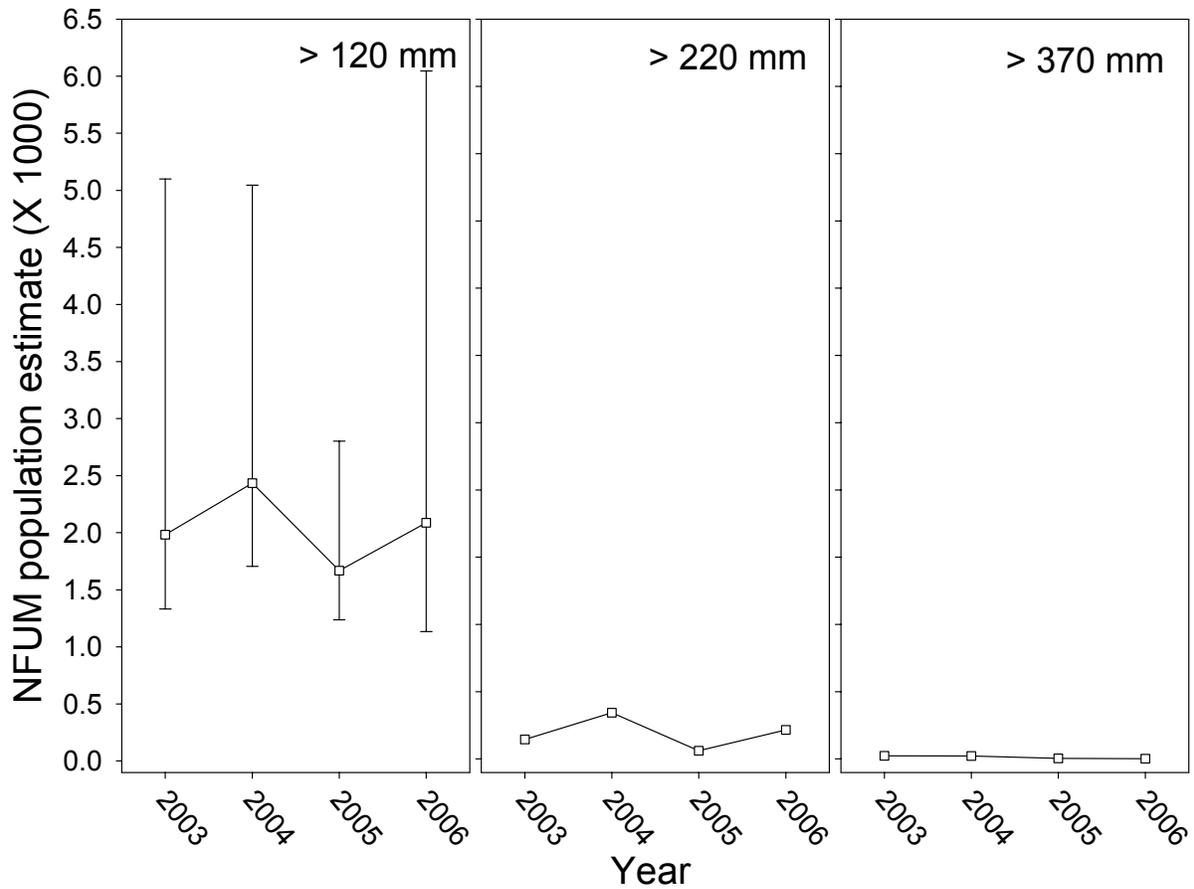


Figure 1.25. Yearly population estimates (\pm 95% CI) for three size groupings of bull trout in the North Fork Umatilla River, 2003 - 2006. Due to low sample sizes, no confidence intervals were obtainable for the bull trout population component $>$ 220 mm or $>$ 370 mm TL.

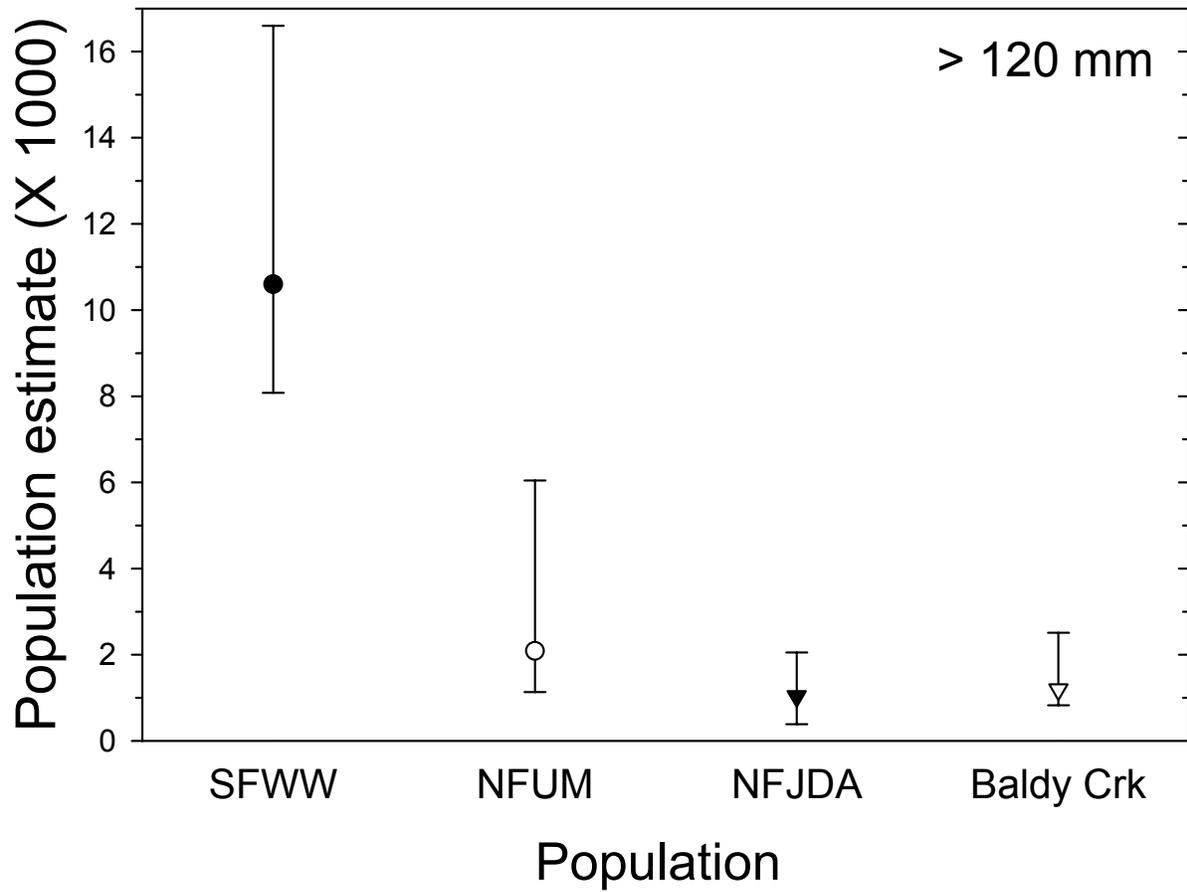


Figure 1.26. Yearly population estimates (\pm 95% CI) for four populations of bull trout (> 120 mm TL), 2006.

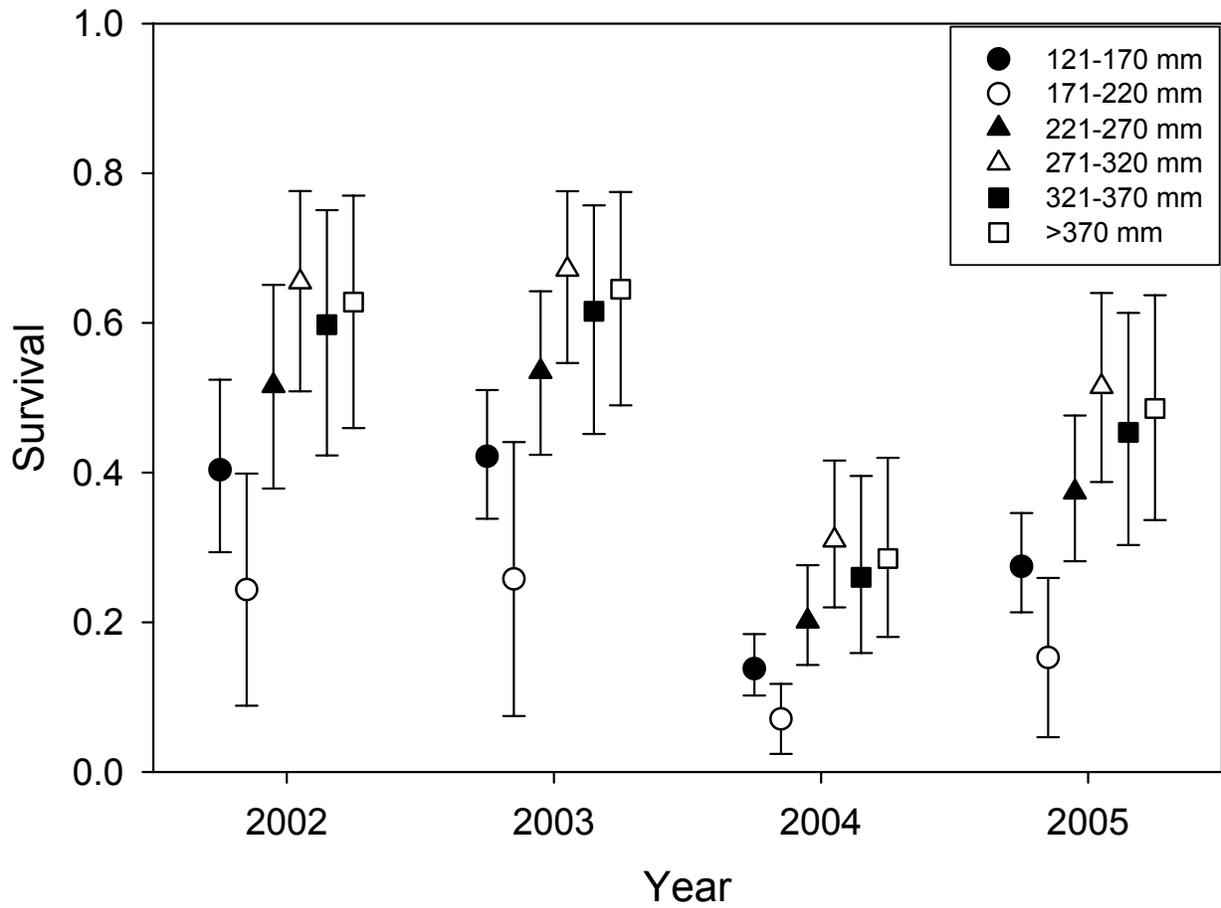


Figure 1.27. Survival estimates (± 2 SE) for six size classes of bull trout in the South Fork Walla Walla River over the period 2002 to 2005.

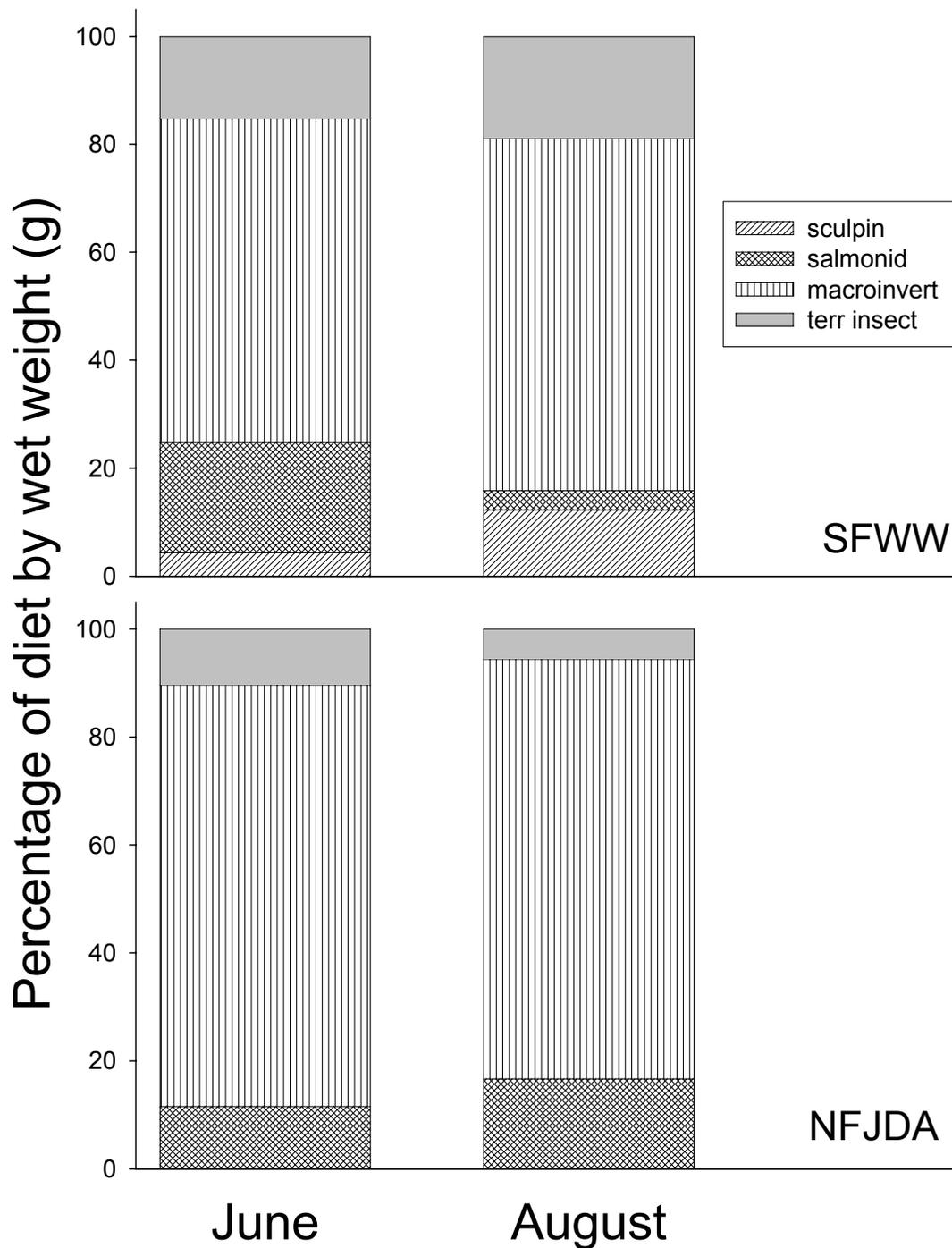


Figure 1.28. Diet composition (% of diet by wet weight) of bull trout taken from the South Fork Walla Walla River (top panel) and North Fork John Day River (bottom panel), June and August 2006. “Salmonid” includes all unidentifiable salmonid species. “Macroinvert” includes all aquatic invertebrates. “Terr insect” includes all terrestrial invertebrates.

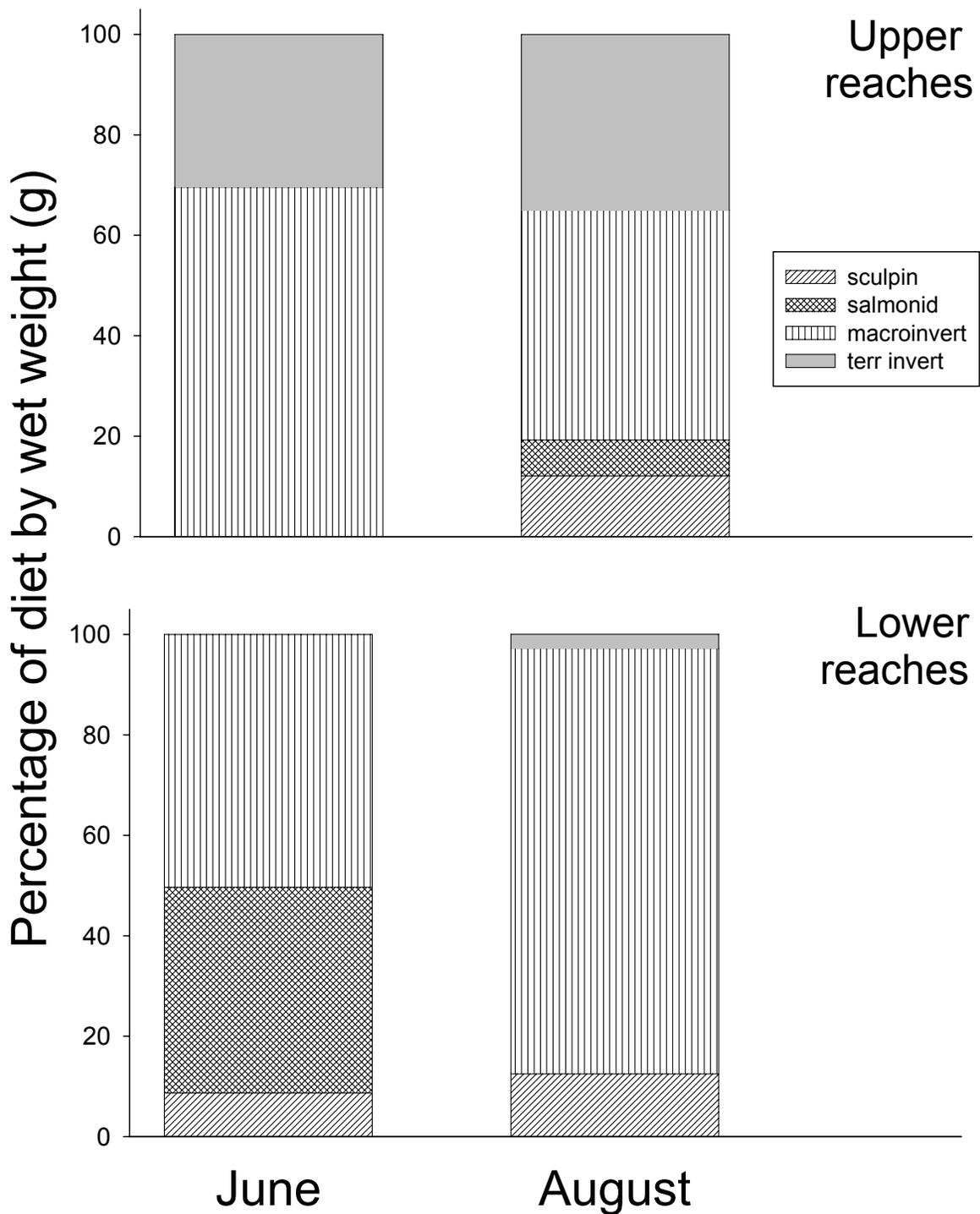


Figure 1.29. Diet composition (% of diet by wet weight) of bull trout captured in the upper (near Reser Creek) and lower (near Bear Creek) reaches of the South Fork Walla Walla River, June and August 2006. “Salmonid” includes all unidentifiable salmonid species. “Macroinvert” includes all aquatic invertebrates. “Terr insect” includes all terrestrial invertebrates.

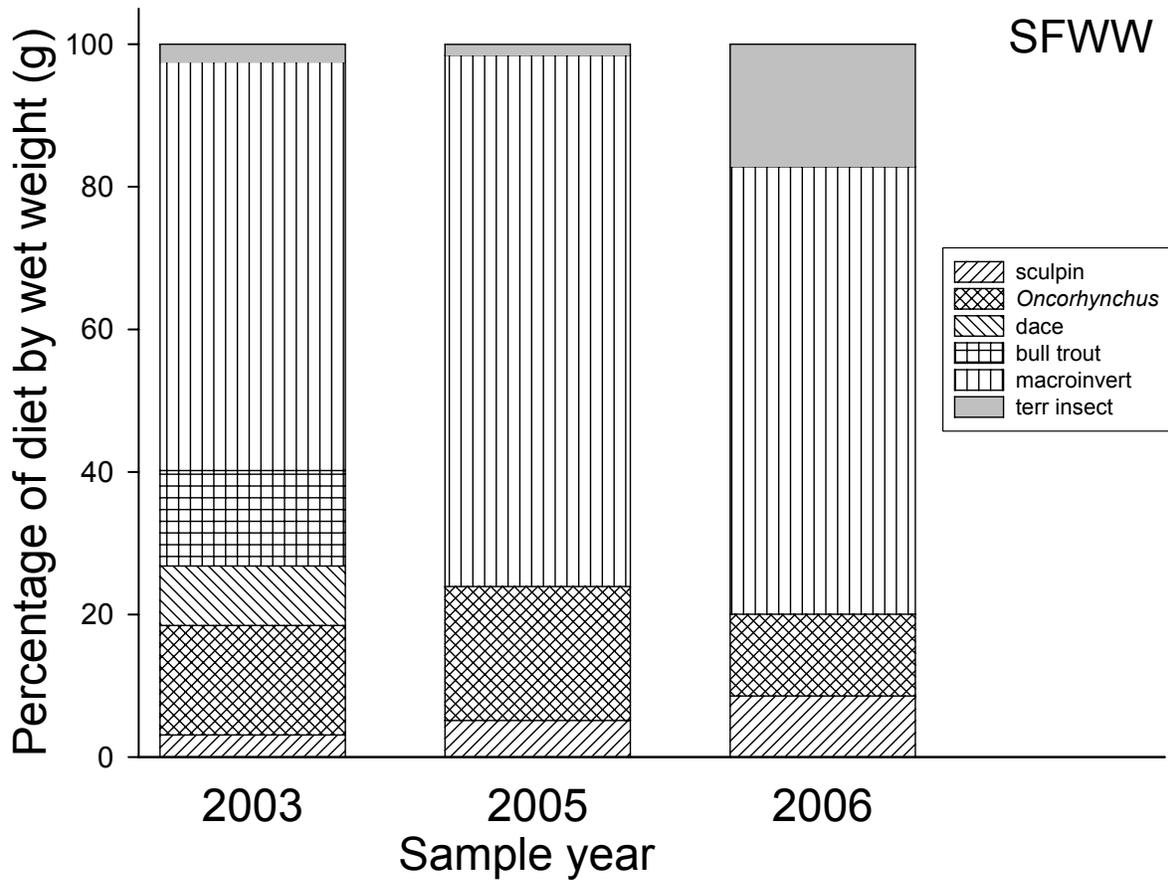


Figure 1.30. Diet composition (% of diet by wet weight) of bull trout captured in the South Fork Walla Walla River in 2003, 2005, and 2006. “*Oncorhynchus*” includes all salmonid species, except bull trout. “Macroinvert” includes all aquatic invertebrates. “Terr insect” includes all terrestrial invertebrates.

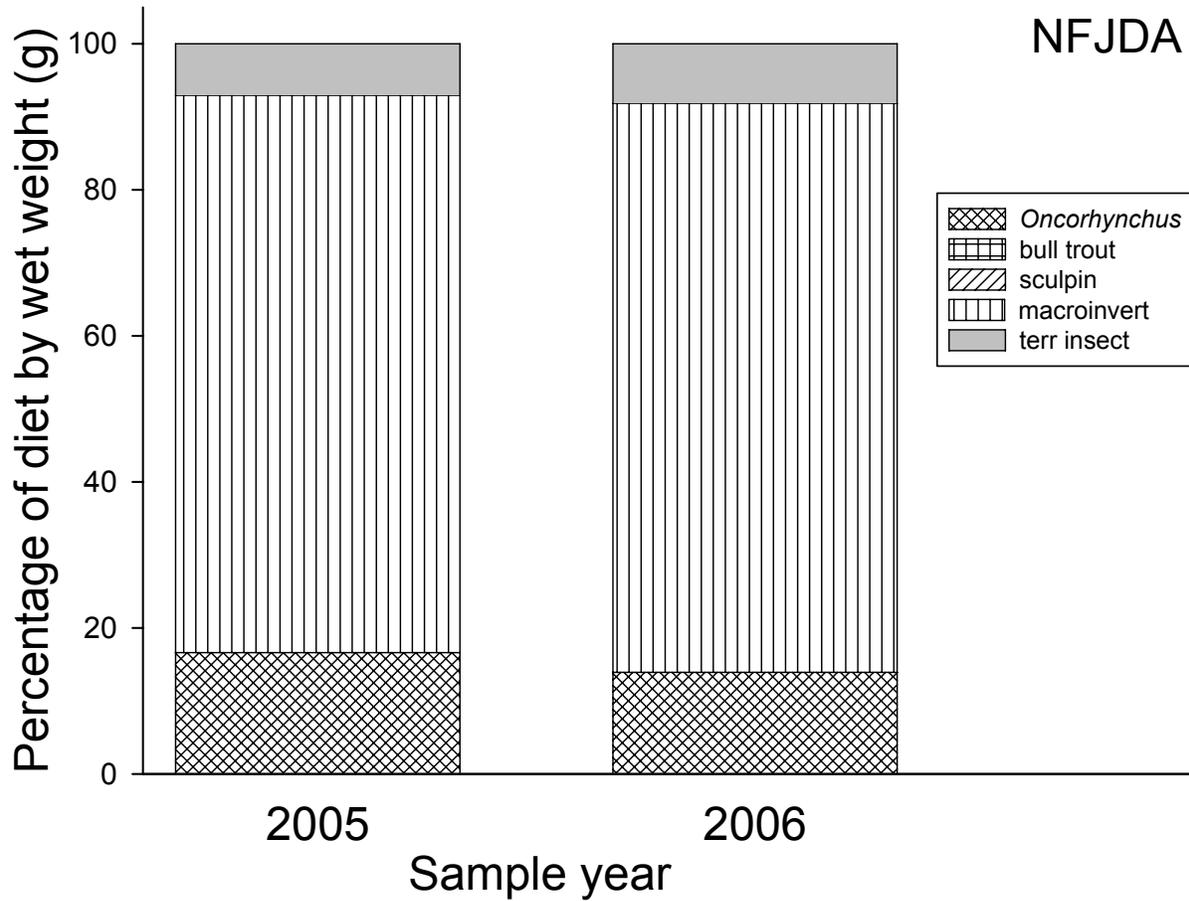


Figure 1.31. Diet composition (% of diet by wet weight) of bull trout captured in the North Fork John Day River in 2005 and 2006. “*Oncorhynchus*” includes all salmonid species, except bull trout. “Macroinvert” includes all aquatic invertebrates. “Terr insect” includes all terrestrial invertebrates.

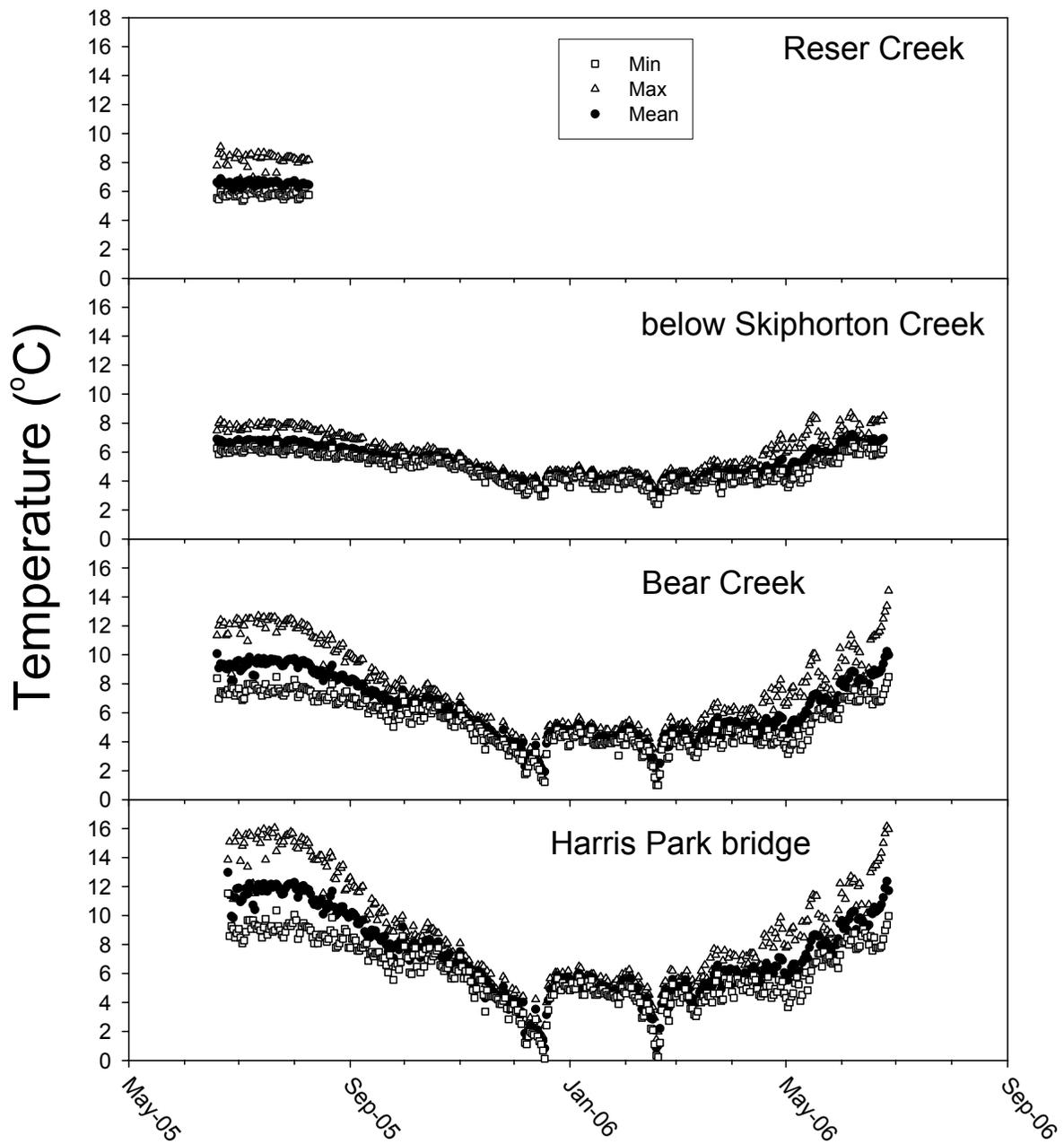


Figure 1.32. Daily temperatures (minimum, maximum, mean) recorded at four locations (Reser Creek is top and Harris Park is bottom of study area) on the South Fork Walla Walla River, Oregon, June 2005 – June 2006. Note: data for Reser Creek temperature logger is incomplete, as logger was found in campfire.

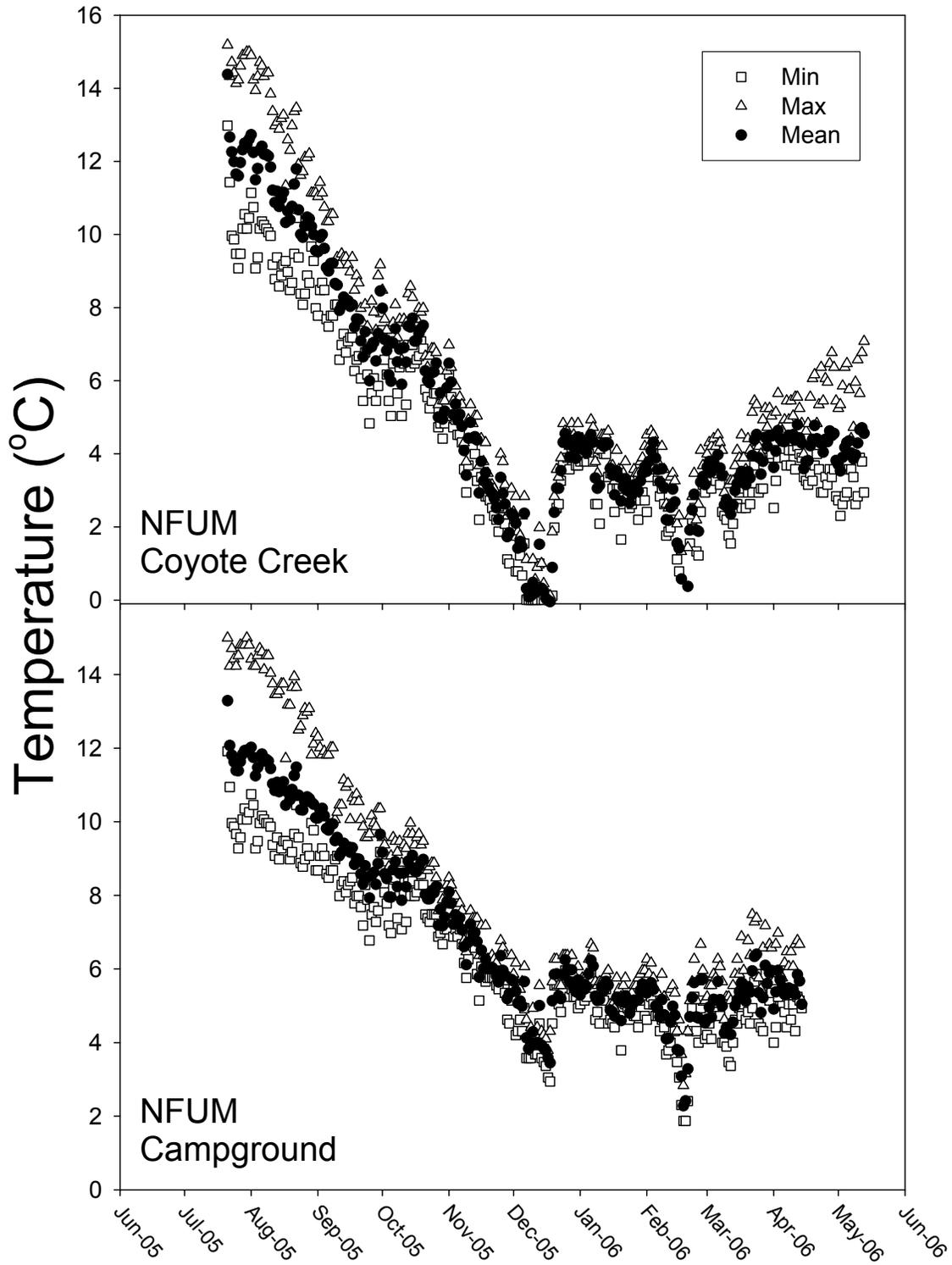


Figure 1.33. Daily temperatures (minimum, maximum, mean) recorded at two locations (Coyote Creek is near the middle and campground is at bottom of study area) on the North Fork Umatilla River, Oregon, 2005 – 2006.

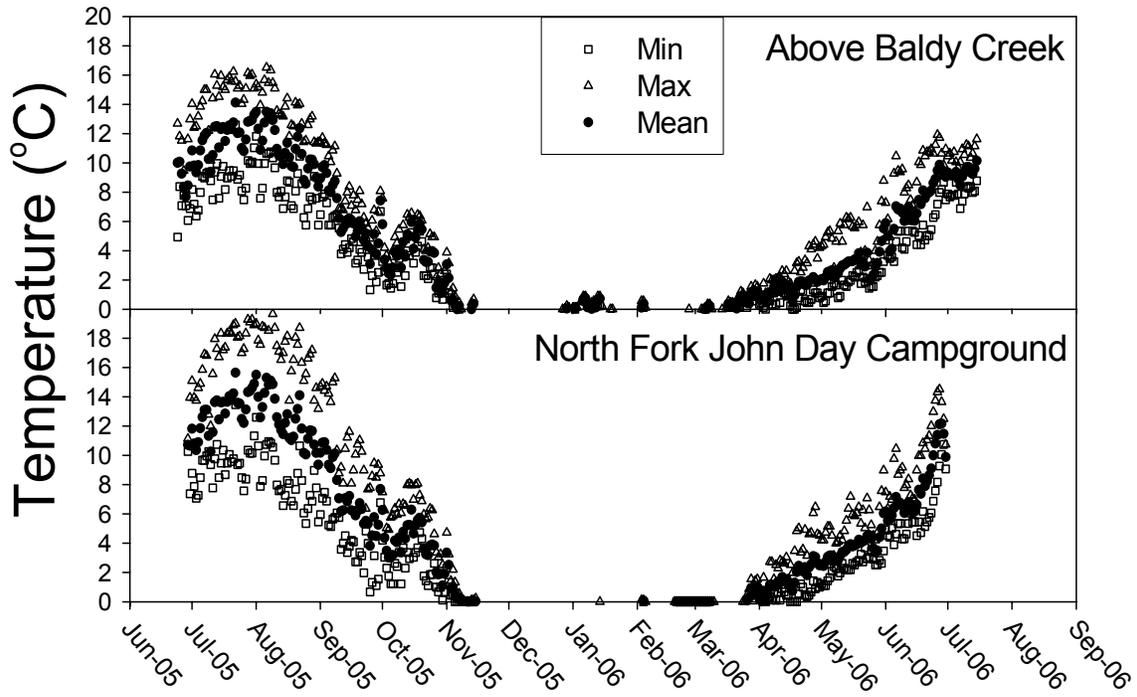


Figure 1.34. Daily temperatures (minimum, maximum, mean) recorded at two locations (Above the confluence with Baldy Creek and adjacent to the North Fork John Day campground) on the North Fork John Day River, Oregon, 2005 – 2006.

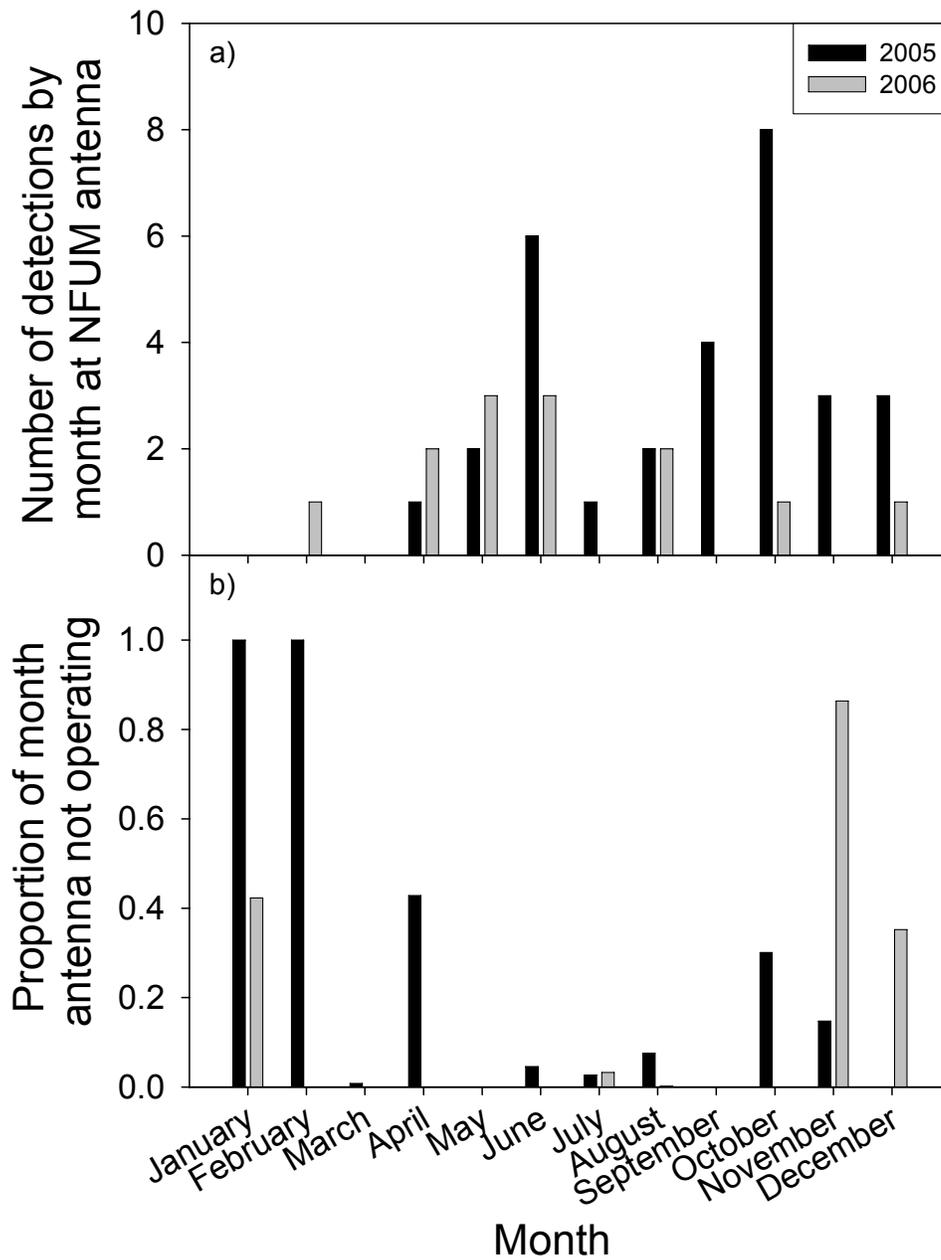


Figure 1.35. A monthly summary of bull trout detections (a) and percent of each month the antenna was in operation (b) at the NFUM antenna during 2005 and 2006.

CHAPTER 2:
**Monitoring freshwater salmonid population trends using
mark-recapture methods**

INTRODUCTION

Monitoring population trends continues to be one of the most important components in the management and conservation of freshwater salmonids (McElhany et al. 2000). However, this can be a challenging endeavor due to high levels of variability (Maxell 1999; Ham and Pearsons 2000), observer error (Dunham and Rieman 1999), and limited resources. Furthermore, many freshwater salmonid populations contain multiple life history forms, which present additional challenges as a result of inherent differences in capture probabilities (Thurow et al. 2006), observer error (Al-Chokhachy et al. 2005), and the demographics between life-history forms (Al-Chokhachy 2006). For these populations containing multiple life-history forms, there are also additional data needs as managers may need to evaluate the trends of the different life-history components within a population independently. For salmonids in particular, assessing the trend of the migratory component may be necessary to evaluate potential isolation and/or connectivity to other populations (e.g., metapopulation structure; Dunham and Rieman 1999). Despite these challenges, assessments of population trend over relatively short time intervals (e.g., 5 – 10 years) can be critical for evaluating the effectiveness of different restoration actions (Bradford et al. 2005), response to management actions (Holmes and York 2003), and in determining proper regulatory actions (Maxell 1999).

Recent theoretical advances in mark-recapture methods have resulted in applications for evaluating population trends (Pradel 1996), and over the past decade, there has been a substantial increase in the number of projects using passive integrative transponder tags (PIT tags), passive instream antennae, and mark-recapture techniques to quantify various population-level metrics for salmonids (Zydlewski et al. 2006). Passive instream antennae, which detect fish tagged with PIT-tags as they move through the antennae are similar those used in major hydropower facilities throughout the Pacific Northwest, are becoming increasingly common in fisheries research in small and medium sized streams. These antennae can provide managers with a means for additional recaptures of previously PIT-tagged individuals without further harassment through increased sampling efforts (e.g., electroshocking), and continuous sampling, which can provide insight into diel behavioral patterns (Horn 2007). Ultimately, the individual-specific mark-recapture information can be used to estimate growth, movement, and survival across multiple size classes and life-history forms within and across populations (Al-Chokhachy 2006).

However, despite the widespread use of PIT-tag technology in monitoring and evaluating salmonid movement patterns and vital rates, there has been little effort to use long-term mark-recapture information to monitor population trends. Temporal symmetry models utilize individual encounter histories of mark-recapture data to estimate population trend (Pradel 1996), and do not rely on the precision of abundance estimates (e.g., Dennis et al. 1991) or stage-specific demographic estimates and oft-biased projection matrix estimates (Nichols and Hines 2002). Furthermore, temporal symmetry models can account for the reduced capture probabilities generally associated with cryptic species and potential differences in capture probabilities across groups within populations. Thus, unlike count data, which only provides an index of population size without accounting for differences in capture probabilities, or abundance estimates, which generally have low precision (Ham and Pearsons 2000), the temporal symmetry model can provide more precise estimates of population trend (Sandercock and Beissinger 2002). To date, temporal symmetry models have proven effective for monitoring terrestrial animal populations (Sandercock and Beissinger 2002; Boulanger et al. 2004) and for long-lived, large-river species (Pine et al. 2001); however, formal comparisons of the precision and applications of temporal symmetry models with current monitoring techniques for salmonids or other similar species have not been evaluated.

Bull trout are a species of char native to the Pacific Northwest that have exhibited significant declines, and are currently listed as Threatened under the Endangered Species Act (ESA) in the United States and 'Of Special Concern' in Canada. The sensitivity of bull trout to water temperatures above 16°C (Selong et al. 2001) and habitat degradation and fragmentation has resulted in the restriction of many populations to primarily headwater reaches (Rieman et al. 1997), making them difficult and expensive to monitor effectively (Al-Chokhachy 2006). Furthermore, bull trout populations can contain multiple life-history forms (Rieman and McIntyre 1993; Nelson et al. 2002), are behaviorally cryptic (Thurow et al. 2006), and naturally occur at low densities (Al-Chokhachy and Budy, *In press*). Overall, these inherent attributes present managers with unique challenges for monitoring bull trout populations, yet accurate and precise population trend assessments are required for effective monitoring and evaluation of this Threatened species, for prioritizing future management actions, and for understanding our conservation and management actions.

In this paper, we use five years of bull trout mark-recapture data to evaluate the use of temporal symmetry models to monitor salmonid population trends. Al-Chokhachy et al. (*In review*) previously illustrated the low precision of bull trout population

abundance and index data resulted in low statistical power to detect moderate changes (25%) in population abundance. These results suggest detecting bull trout population trends with abundance data may be extremely difficult, highlighting the need to evaluate alternative trend estimation methods. Within the framework, the objectives of this paper are to assess: 1) the precision of population trend estimates from temporal symmetry models; 2) the ability to monitor population trends of different life-history forms and components of the population; 3) comparisons of the precision of trend estimates from temporal symmetry models and traditional count data (redd counts and snorkel counts); and 4) the sensitivities of the temporal symmetry model to variable detection probabilities using passive instream antennae.

METHODS

As part of a larger study (Al-Chokhachy 2006; Homel 2007), we collected comprehensive mark-recapture data in the South Fork Walla Walla River (SFWW; 2002-2006), a tributary of the Walla Walla River in Northeastern Oregon (Figure 1). The SFWW originates in the Blue Mountains, which exist at the eastern boundary of the arid steppe of the Columbia River Basin, and are characterized by hot, dry summers and cold, wet winters. Despite the relatively low elevation of the SFWW (610 – 1000 m), it is a system dominated by cold, groundwater influences where stream temperatures generally do not exceed the suggested upper limit for bull trout (16° C; Selong et al. 2001).

The SFWW study site was contained primarily in the Umatilla National Forest and was approximately 21 km in length. Habitat conditions within the SFWW basin can generally be described as high quality, with few forest management activities, but several recreational activities do occur in the basin (e.g., hiking). Within the study site, the SFWW is a medium-sized stream (average width = 10 m), and stream habitat conditions ranged from confined-channel reaches with steep or bedrock hillslopes to relatively unconfined channels with more complex, meandering reaches. Downstream of the SFWW site, habitat conditions degrade in a longitudinal, downstream direction with respect to water temperature, simplification of habitat, channelization, and barriers to migration.

The fish assemblage within the SFWW was dominated by salmonids, consisting primarily of rainbow and steelhead trout (*Oncorhynchus mykiss* spp.), Chinook salmon (*O. tshawytscha*), mountain whitefish (*Prosopium williamsoni*), sculpin (*Cottus* spp.), and bull trout. The SFWW contains a relatively large population of both small,

potentially resident and large, potentially migratory bull trout (Al-Chokhachy et al. 2005; but see Homel et al. *In review*).

Mark-recapture data

We initiated mark-recapture efforts in the SFWW in 2002, and divided the study site into reaches of at least 200 m in length (102 reaches). We used a systematic sampling design (random start), based on an annual 20% minimum sampling rate, to achieve spatial balance in sampling throughout each site (Stevens and Olsen 2004). Each year (2002-2006), we began sampling in mid-June and consistently sampled the same 21 reaches for marking and recapturing bull trout; sampling continued through the middle of August.

We used multiple techniques, including minnow traps, trap nets with snorkelers, electroshocking fish down to a seine, and angling, to actively capture bull trout. We combined multiple sampling techniques to avoid potential sampling bias (Budy et al. 2003) and effectively sample across all habitat types. When captured, all bull trout larger than 120 mm were anesthetized, tagged with both a year-specific external anchor tag and a surgically implanted 23-mm PIT-tags (full duplex; 134.2 kHz), weighed and measured, and released at the point of capture. Recaptured individuals were anesthetized, checked for tag loss, weighed and measured, and released. In 2002, we installed passive instream antennae (hereafter referred to as antennae) in the SFWW to provide additional recaptures and quantify bull trout movement; one antenna was located at the bottom of our study site (WW1), and one approximately 6 km upstream from the bottom of the study site (WW2; Figure 1). At each location, the antenna consisted of rectangular polyvinyl chloride (PVC) detectors that spanned the entire width of the stream. The antennae operated continuously from November 2002 through the duration of the study; however, there were short time periods where a particular antenna was shut down due to technical difficulties.

For all mark-recapture analyses, we used a sampling period from June 1st through October 31st. We selected this period based on previous movement information in the SFWW (Homel and Budy *In review*), as this time period included the majority of juvenile and adult seasonal migrations. Within this annual sampling period, we included all bull trout captured and recaptured via active sampling methods (e.g., electrofishing) as well as recapture events which occurred at the antennae.

Snorkeling Data

After the completion of mark-recapture sampling, we performed annual snorkel surveys in each of the 21 reaches as an index of bull trout abundance (2002 – 2006). We began snorkel surveys in early August, and surveys typically continued for two to three weeks. Based on previous work in the SFWW (Al-Chokhachy et al. 2005), we considered three different size classes, including juvenile bull trout (120-220 mm), small adults (220-370 mm), and large adults (>370 mm). We conducted all surveys during the daytime hours of 2 hours after sunrise and 2 hours prior to sunset (for more detail regarding snorkeling methods see Al-Chokhachy et al. 2005). To expand the annual snorkel counts to the study site, we summed the snorkel counts across all reaches and divided this total by the sampling rate (length of reaches sampled/length of study site).

Redd data

Annual redd counts in the SFWW were consistently conducted by an experienced group of state, federal, and tribal biologists. Redd surveys began in early September and continued until the end of October. Annual surveys included an index site, which occurs from the confluence with Skiphorton Creek and upstream to the second tributary above Reser Creek (Length = 5.3 km; Figure 1), where the majority of bull trout spawning occurs, and a comprehensive survey, which included the entire river. The number of surveys for the index site (range 3 – 6) and comprehensive survey (range 1 – 4) varied each year based on available personnel. Surveyors flagged and recorded the location (GPS coordinates) of each redd to avoid the chance of double-counting redds in subsequent surveys.

Trend analyses

Mark-recapture.—We used the temporal symmetry model in Program MARK (White and Burnham 1999) to evaluate bull trout population trend (λ) in the SFWW (Pradel 1996). The temporal symmetry model simultaneously models the capture history of each individual using both forward-time and reverse-time modeling (Nichols and Hines 2002). Forward-time mark-recapture data is used to estimate apparent survival (Φ), defined as the probability that an animal tagged in occasion t was alive in occasion $t + 1$, and capture probability (p). In addition to these parameters, the temporal symmetry model uses reverse-time mark-recapture data (i.e., interprets the encounter history backwards) is used to estimate seniority probability (γ), defined as the probability that an individual did not enter the SFWW population between $t - 1$ and t . With the temporally symmetry model, population growth rates are calculated as:

$$\lambda = \frac{\Phi}{\gamma} \quad \text{Equation 1}$$

and values of $\lambda > 1$, $\lambda < 1$, or $\lambda = 1$ suggest the population is increasing, decreasing, or stable, respectively.

We established a set of *a priori* models where we included group, time effects, and considered two individual covariates, movement and condition. We calculated condition at time of tagging for each fish using relative condition, which we calculated as the difference in an individual's condition relative to average condition values for the SFWW (Al-Chokhachy 2006), and modeled condition as a continuous variable. We modeled movement as a binary variable, which included fish that moved below WW1 (1) and those that did not move below WW1 (0). We incorporated individual covariates to account for variability in estimates of Φ and p , but did not include these covariates when we modeled λ due to interpretation limitations on the effects of individual covariates on overall population growth (Cooch and White 2006).

We performed two separate analyses to evaluate the trends of different size classes of bull trout in the SFWW. First, we modeled population trends using nested analyses, where we considered the following three groups: 1) bull trout >120 mm, which includes both juvenile and adults; 2) bull trout > 220 mm, which includes only sexually-mature adults; and 3) bull trout >370 mm, which includes only large adults. Secondly, we modeled the SFWW population using non-nested groups, including juveniles (120 – 220 mm), small adults (220 – 370 mm), and large adults (>370 mm), to evaluate the population trends of distinct components of the population. Together, these analyses allowed us to consider the population trends of different size classes of fish when combined (nested) and the trends of specific size-classes of bull trout when delineated (non-nested).

We also evaluated the trends of migratory bull trout over this time period. We considered any bull trout that migrated past WW1, a distance of approximately 12 km from the bottom of the core spawning area, as migratory (Hornel 2007). We included two groups in these analyses, migratory and resident (fish that did not move beyond WW1) and estimated λ for each group.

Previous analyses suggested that the detection probability at WW1, defined as the probability that a PIT-tagged fish that travels through an antenna was detected, was less than one (approximately 50%). To account for this reduced detection efficiency, which was largely due to power outages and environmental variability (e.g., high flow

events); we randomly assigned recapture events to avoid underestimating population growth rates in the SFWW. For these analyses, we quantified how many bull trout moved passed WW1 each year (m), expanded this total by 50%, and randomly assigned a recapture event to ($m/0.5$) individuals. Since one of our objectives involved evaluating the effects of reduced detection probabilities at PIT-tag antennae on temporal symmetry model estimates of population growth rate, we also modeled the SFWW population growth rates with the current detection probability (e.g., ambient detection probability; 50%).

We used Program MARK to generate the likelihood function value and estimate the appropriate Akaike Information Criterion (AIC_c ; bias adjusted for small sample size) value for each model. We ranked models according to the lowest AIC_c score, and used the difference in AIC_c values (ΔAIC_c) between models to calculate an Akaike weight for each model ($\exp(-0.5\Delta AIC_c)/\text{sum of } \exp(-0.5 \Delta AIC_c)$ all models). Similar to Franklin et al. (2004), we first modeled those parameters of less biological interest to these analyses (e.g., p and Φ), and then maintained the model structure of these parameters with the lowest AIC_c score while modeling λ .

Count data.—We used a count-based method to estimate bull trout population trends in the SFWW (2002 – 2006; e.g., Morris and Doak 2002). We performed separate analyses for snorkel data and redd data, and under this approach, we used linear regression analyses of the log-transformed annual changes in population growth as a function of time step (Dennis et al. 1991). As with mark-recapture trend analyses, we considered both nested and non-nested size groups (see above) for the trend analyses of snorkel count data. For all trend analyses, we estimated population trend (λ) and 95% confidence intervals. We considered population trends increasing ($\lambda > 1$) or decreasing ($\lambda < 1$) when 95% confidence intervals did not overlap with one, and otherwise stable ($\lambda = 1$).

RESULTS

Each year from 2002 through 2006, we sampled 21 reaches or approximately 5.1 km of the 21 km study site (25%), and marked 1140 bull trout >120 mm with PIT tags (Table 2.1). We performed annual snorkel surveys in all 21 reaches in 2002 – 2004, and 2006; however, a forest fire on the study site in 2005 resulted in a reduced sampling effort of only 16 reaches (19%). There were similar limitations for the comprehensive redd count surveys due to the 2005 forest fire, and therefore we only included redd count data from index surveys.

Table 2.1. Coefficient of variation (CV), the annual number of PIT-tagged bull trout and snorkel counts for three size classes of bull trout (120-220 mm, 221-370 mm, and >370 mm), and annual redd count data (index site) from South Fork Walla Walla River (2002 – 2006).

Year	Number marked by size class (mm)			Snorkel counts by size class (mm)			Redd count
	120-220	221-370	>370	120-220	221-370	>370	
2002	92	69	41	1741	712	382	198
2003	153	45	18	818	525	224	186
2004	133	36	21	533	350	195	198
2005	257	36	28	956	460	142	170
2006	156	39	16	687	451	334	126
CV	0.385	0.309	0.409	0.497	0.269	0.390	0.171

Trend estimates

We observed considerable differences by both method and size class in the precision (95% CIs) and point values of population trend estimates in the SFWW (Figure 2.2). Trend estimates based on snorkel data suggested that the population in the SFWW is stable across all size classes (>120 mm, $\lambda = 0.99$, 95% CI = 0.58 – 1.70), all adults (>220 mm, $\lambda = 0.98$, 95% CI = 0.71 – 1.34), and large adults (>370 mm, $\lambda = 1.07$, 95% CI = 0.70 – 1.66). Snorkel surveys exhibited substantial annual variation with high coefficients of variation (CV; range = 0.330 – 0.428). Trend estimates from the temporal symmetry model exhibited the highest precision across methods. The trend for all bull trout (>120 mm) indicated the population was relatively stable over this period ($\lambda = 0.94$, 95% CI = 0.84 – 1.05). However, trend estimates for all adult bull trout (>220 mm) from the temporal symmetry model suggested a declining population trend ($\lambda = 0.89$, 95% CI = 0.79 – 1.00), with large bull trout (>370 mm) exhibiting the sharpest decline over this period ($\lambda = 0.83$, 95% CI = 0.75 – 0.91). Redd count trend estimates ($\lambda = 0.90$, 95% CI = 0.78 – 1.02) were most similar to temporal symmetry model estimates for bull trout >220 mm, but with slightly reduced precision; however, redd count data exhibited significantly lower CV (0.171) than snorkel counts.

Delineating the population into distinct size classes (non-nested) yielded similar results for snorkel surveys but with overall lower precision than mark-recapture analyses (Figure 2.3). Trend estimates from snorkel surveys indicated the bull trout population

is stable across juvenile (120 – 220 mm, $\lambda = 0.94$, 95% CI = 0.53 – 1.65), small adult (221 – 370 mm, $\lambda = 0.99$, 95% CI = 0.65 – 1.50), and large adults (see above). Similar to the nested analyses, snorkel surveys exhibited very high CVs for population monitoring (Table 2.1). The results from temporal symmetry model analyses suggested both juvenile ($\lambda = 0.91$, 95% CI = 0.79 – 1.05), and small adults ($\lambda = 0.88$, 95% CI = 0.72 – 1.05) exhibited stable population trends, and again large bull trout exhibited declining population trends (see above).

Although point estimates of λ were substantially lower for the migratory portion of the population, we observed no significant differences in population trends of resident and migratory bull trout due to high levels of variability in the trend estimates (Figure 2.4). There was some evidence of declining population trends for migratory bull trout ($\lambda = 0.81$, 95% CI = 0.64 – 1.03). Population trends for resident bull trout appear to be relatively stable ($\lambda = 1.07$, 95% CI = 0.90 – 1.26) over this time period.

Reduced detection probabilities at PIT-tag antennae had substantial effects on the interpretation of population trend estimates when using mark-recapture methods (temporal symmetry model; Figure 2.5). In particular, when detection probabilities were not accounted for, trend estimates of all bull trout >120 mm in the SFWW indicated the population had decreased over this time period ($\lambda = 0.86$, 95% CI = 0.77 – 0.97), and were substantially lower than analyses which incorporated reduced detection probabilities (see above for estimates). Although there were no significant differences in the interpretation of population trends of small adults (221-370 mm; $\lambda = 0.86$, 95% CI = 0.77 – 0.97) and large adults (>370 mm; $\lambda = 0.82$, 95% CI = 0.74 – 0.92) under these reduced detection probabilities, we did observe increases in population trend estimates and corresponding CIs.

DISCUSSION

Estimating population trends over relatively short time intervals can be extremely challenging due to the inherent biological attributes of the species of interest (e.g., various life-history forms) and potential sampling limitations (e.g., resources). The precision of these estimates can be critical for proper evaluation of the effectiveness of different restoration actions, management strategies, and provide critical information for future regulatory actions (e.g., 5-year status reviews under the Endangered Species Act). In this paper, we used 5 years of bull trout sampling to compare the effectiveness of mark-recapture, temporal symmetry models to estimate population trend with methods commonly used for salmonid monitoring (e.g., snorkel surveys). We found the temporal symmetry model exhibited higher precision for monitoring

population trends than both snorkel surveys and redd count data, thus indicating higher potential for accurate decision-making regarding population responses to different management actions.

Monitoring population trends

Count data.—Across size classes we found snorkel counts exhibited the lowest precision for population trend estimates. In particular, we observed the highest variability in population trends for juvenile (120 – 220 mm) and large adult (>370 mm) size classes of bull trout. Juvenile bull trout are extremely cryptic as they spend much of their time within the substrate and are commonly associated with cover (Thurrow 1997; Al-Chokhachy and Budy, *In press*). While adult bull trout can be more easily detected in snorkel surveys due to their larger body size (Thurrow et al. 2006), the relatively low densities of large bull trout can result in patchy distribution and subsequently high variability in annual count data (Thompson et al. 1998). Snorkel surveys for small adults (221 – 370 mm) exhibited the highest precision among size classes; however, the precision of population trend estimates for this size class was significantly lower than both redd count and mark-recapture estimates. Overall, these results suggest a long time series of data would be required to detect true population trends using snorkel data, a pattern consistent with similar analyses in the Columbia River Basin (Ham and Pearsons 2000).

Over this time period, we observed relatively high precision for the trend estimate based on redd count data in the SFWW, suggesting redd counts may be effective at detecting changes in abundance of adults for this population. The relatively high precision of our redd count data is consistent with recent analyses for bull trout populations comprised entirely of large, adfluvial individuals (Muhlfeld et al. 2006). However, many populations of bull trout exhibit high interannual variability in redd data, thus limiting the precision and ability of redd data to accurately detect population trends (Rieman and Myers 1997; Maxell 1999). Furthermore, previous analyses suggested redd counts in the SFWW only account for large bull trout, but underestimate the numbers of small adults in this population (Al-Chokhachy et al. 2005); thus, potentially limiting the inference of redd counts when used to estimate population trends where multiple life-history forms coexist.

Mark-recapture trend analyses.—Using active and passive mark-recapture data, we were able to quantify bull trout population trends for specific components of the population with relatively high levels of precision. When compared with snorkel counts and redd counts, we consider temporal symmetry models more appropriate for estimating trend over short time intervals for the following reasons. First, mark-

recapture approaches can account for reduced capture probabilities, which may vary temporally and across relevant size classes. For bull trout, which are difficult to capture and/or detect (Thurow et al. 2006), this may be particularly important for monitoring as demonstrated by the low precision of the snorkel surveys. Secondly, mark-recapture methods can allow for specific and/or aggregate estimates of population trend and enable managers to evaluate the effects of different regulatory actions (e.g., angling regulations) on the population as a whole, or for specific size classes of fish. Third, using a combination of active and passive recapture data can provide information regarding trends of different life-history components. Many salmonid populations have been isolated due to habitat fragmentation and loss in downstream river reaches (Thurow et al. 1997), and monitoring trends of migratory components may be critical for understanding the levels of isolation, gene flow, and potential recolonization within and across populations.

Finally, although not a main consideration of this paper, temporal symmetry models can provide important information regarding the relative contributions of different components of the population to population growth rates. Specifically, encounter histories, when reversed (see Cooch and White 2005), can be used to estimate seniority probability (γ). Furthermore, $1 - \gamma$ provides an estimate of population gains, either through births or immigration, and interpretation of γ can provide insight into the relative importance of different components to the overall population growth rate. Seniority values greater than 0.5 suggest that survival has a greater influence than recruitment on λ , and conversely, values less than 0.5 indicate that recruitment, either through births or immigration, has a larger effect (Nichols et al. 2000). Thus, the results from the temporally symmetry model can provide managers with information regarding the sensitivity of population growth rates to different parameters without relying on survival, fecundity, and transition probabilities across multiple stages (e.g., matrix models; Sandercock and Beissinger 2002). For the SFWW, we estimated seniority (γ) as 0.401 (SE = 0.023), which suggests recruitment to the population has overall larger effects on bull trout population growth rates than seniority effects. With little chance of immigration from other populations in the Walla Walla basin (B. Mahoney, personal communication), these results indicate early life-history survival as having a large effect on the SFWW population growth rates; a result similar to sensitivity analyses from more complicated matrix models for this population (Al-Chokhachy 2006).

Future considerations

We acknowledge two main considerations when using the temporal symmetry model for estimating population trend for salmonids. First, one of the main assumptions of

the temporal symmetry model is that there is no expansion of study site during the period of the survey, as increases/decreases in sampling efforts can bias the estimates of population growth rates (see Pradel 1996 for complete list of assumptions). Under these circumstances, the estimates of population growth would correspond to changes in the number of marked/recaptured individuals as result of the changes in sampling effort, and not actual increases/decreases in population size (Sandercock 2006). These changes in sampling effort may occur through direct actions (i.e., more sample units) or indirectly (i.e., sampling ability). In particular for salmonids, changes in variation in environmental characteristics (e.g., water flows) and field skills (e.g., handling equipment) may dramatically change our ability to capture fish for mark-recapture analyses, and these changes may indirectly act as expansions or contractions of study areas (Cooch and White 2006). In the SFWW, we acknowledge that these indirect changes in study area may have affected our estimates of λ , but consider these effects minimal due to the consistent base-flow conditions in the SFWW and our efforts to maintain consistent sampling methods across years.

Secondly, PIT-tag antennae detection probabilities can have significant effects on the interpretation of population trend. In our analyses, accounting for reduced detection probability resulted in stable population trend estimates for the SFWW (>120), and we found declining population trends when we did not incorporate detection efficiency into our analyses (Figure 4). For temporal symmetry models, reduced detection probabilities can result in lower estimates of population trend as apparent survival is biased low (see Equation 1). Thus, for populations containing migratory individuals, evaluating the effects of varying detection probabilities at key migratory points may be necessary.

Finally, we acknowledge that our analyses were conducted in a relatively large bull trout population (Al-Chokhachy et al. 2005), and further analyses are needed to evaluate the precision of the temporally symmetry model in small populations. Monitoring population trends for these populations can be extremely challenging as a result of clumped or patchy distribution, resulting in low precision and statistical power (Morrow et al. 1999; Al-Chokhachy 2006). However, unlike traditional abundance estimates or count data, low fish densities or uneven distribution can be accounted for in estimates of capture probability; thus allowing for more precise estimates of population trend. When using mark-recapture methods in these small populations, we urge managers to consider *a priori* simulation analyses (e.g., Peterman and Bradford 1987) to estimate the required number of tags, recapture rates, and time needed to achieve adequate statistical power.

Management considerations

Fisheries managers must often make critical decisions over relatively short time intervals (e.g., ESA 5-year Status Review), and correct decision-making generally relies on the precision of the information gathered. Across different monitoring techniques there can be different levels of precision (Sandercock and Beissinger 2002), and selecting the most appropriate method can be critical for robust assessments of population status and trend. We conducted our analyses on a population of bull trout, a species which can be challenging to monitor due to its behavior and basic biology; our results suggest recent advances in PIT-tag technology and temporal symmetry models can be effective at monitoring different components of these and other salmonid populations. In addition, this approach can provide insight into the relative importance of these components to population growth rates, which can be necessary, but difficult information to obtain from traditional approaches that rely on stage-specific demographic and vital rate information. Finally, we acknowledge that the relatively high costs associated with mark-recapture methods may not be possible for managing numerous populations across large spatial scales (Al-Chokhachy 2006). However, in many instances this cost may be offset by the information gained from mark-recapture methods (e.g., vital rates; see introduction), the ability to evaluate potential bias in alternative monitoring techniques, and the relatively high precision generally associated with temporal symmetry models.

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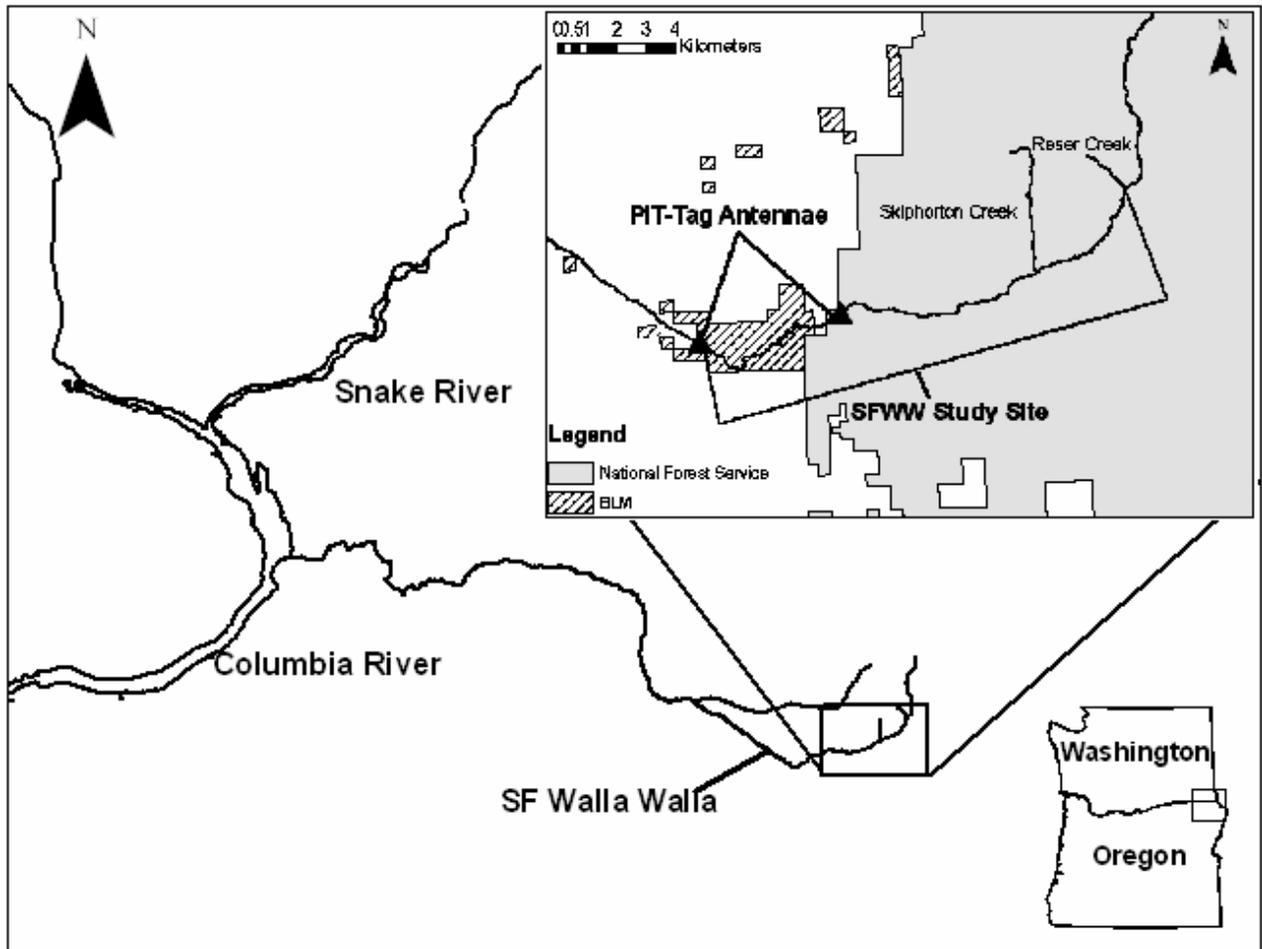


Figure 2.1. Map of the SF Walla Walla River illustrating the study site boundaries, location of passive PIT-tag antennae, and major tributaries.

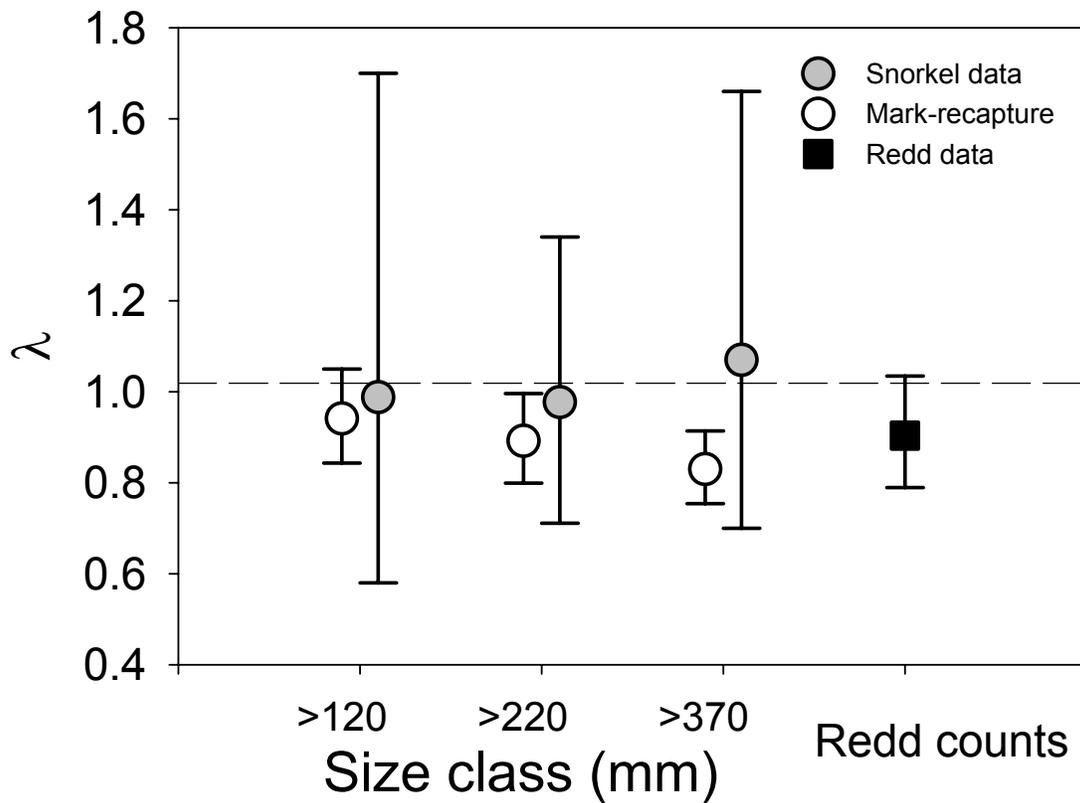


Figure 2.2. A comparison of bull trout population trend estimates in the SF Walla Walla River for annual redd count data and using nested analyses of snorkel data and mark-recapture data for three size classes of bull trout (>120 mm, >220 mm, and >370 mm). A dashed line is shown to represent a stable population growth rate of 1.00.

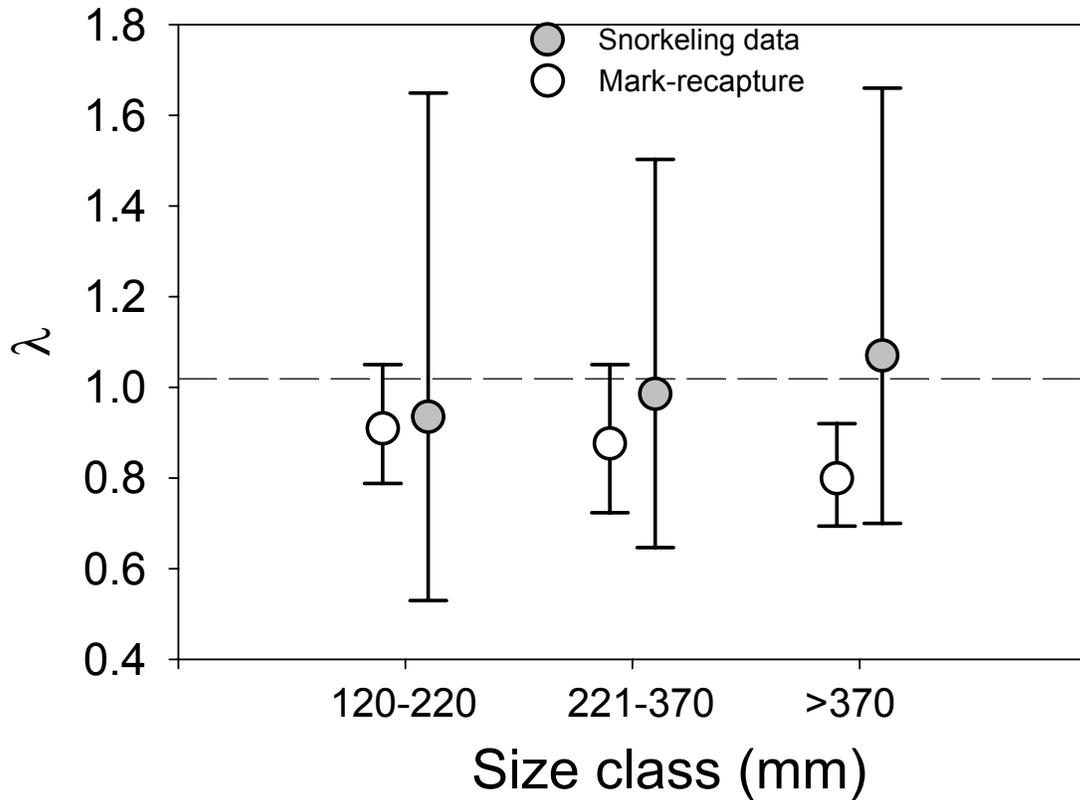


Figure 2.3. Bull trout population trend estimates for the SF Walla Walla River across three non-nested groups of bull trout (120 – 220 mm, 221 – 370 mm, and >370 mm) using snorkel data and mark-recapture data. A dashed line is shown to represent a stable population growth rate of 1.00.

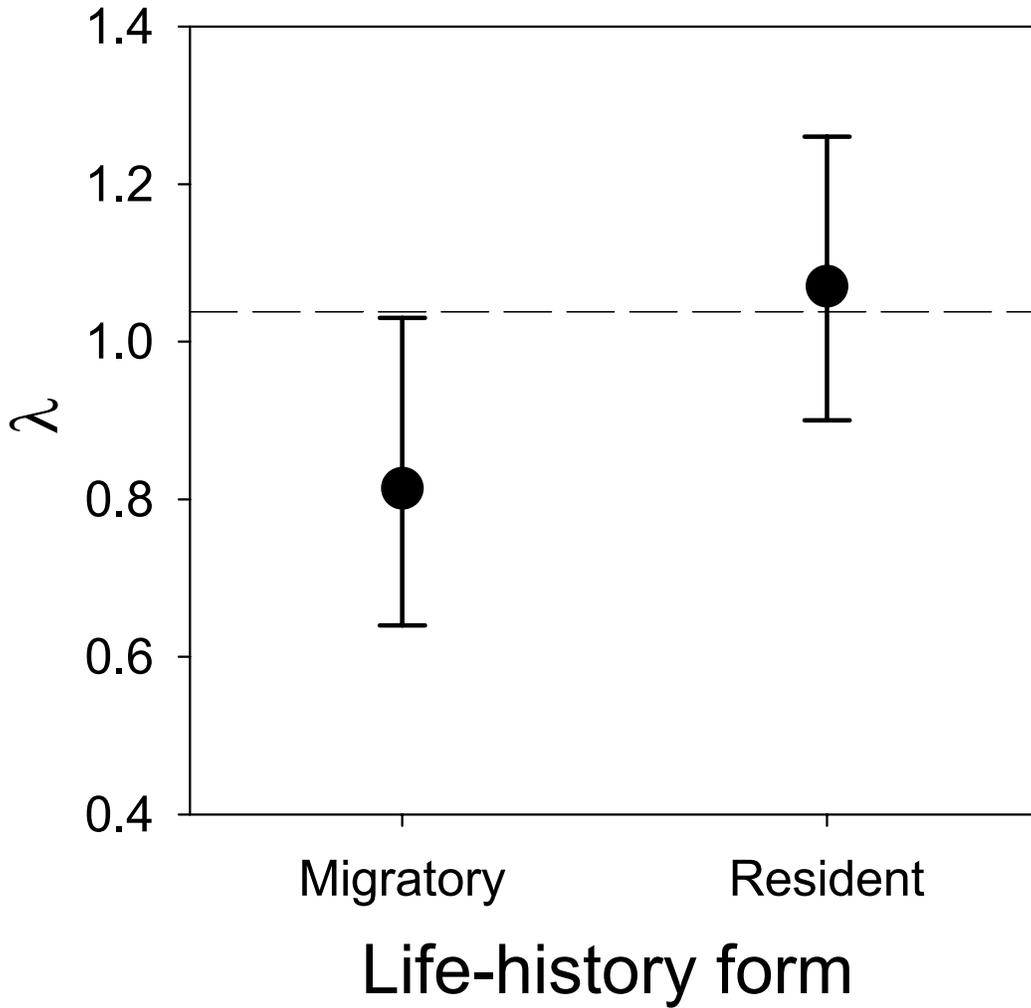


Figure 2.4. A comparison of trend analyses for resident and migratory bull trout in the SF Walla Walla River using mark-recapture data (2002 – 2006). Fish were considered migratory if they traveled below the bottom of the study site, a distance approximately 12 km from the bottom of the core spawning area.

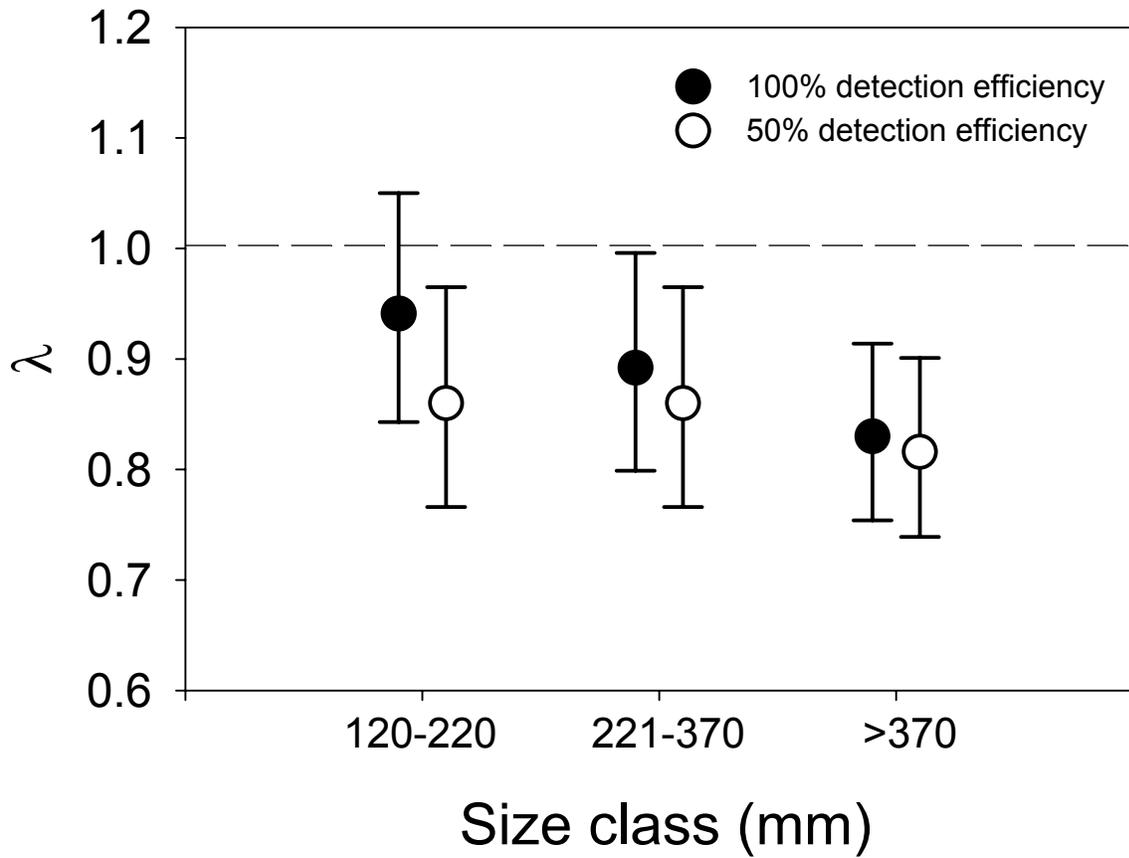


Figure 2.5. A comparison of population trend estimates in the SF Walla Walla River with and without accounting for reduced detection efficiencies at PIT-tag antennae across the three non-nested size classes of bull trout.

APPENDIX 1

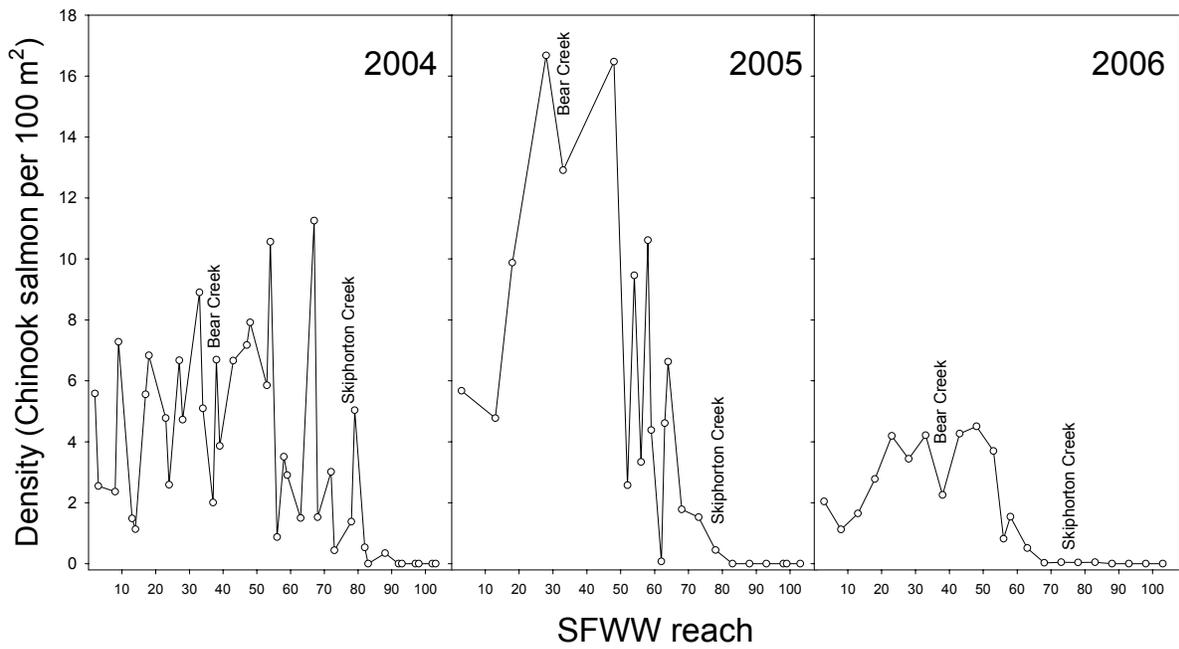


Figure A1. Density of Chinook salmon estimated by snorkel counts in various reaches of the South Fork Walla Walla River (2004 – 2006).

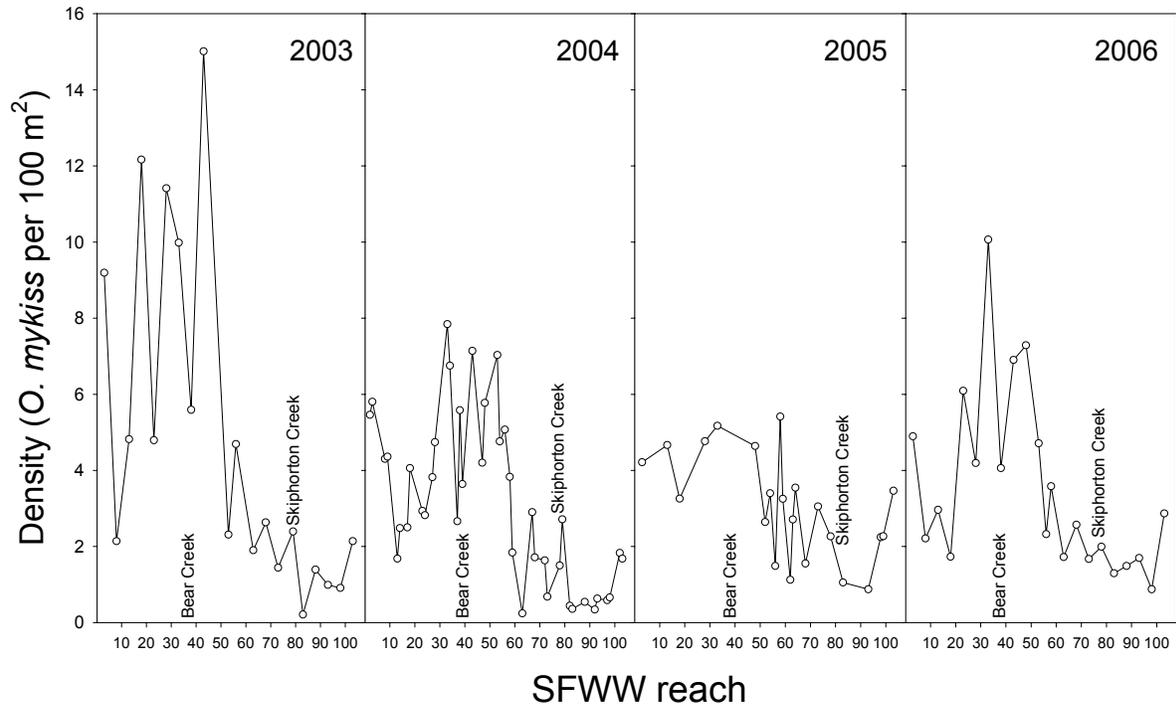


Figure A2. Density of *O. mykiss* spp. estimated by snorkel counts in various reaches of the South Fork Walla Walla River (2003 – 2006).

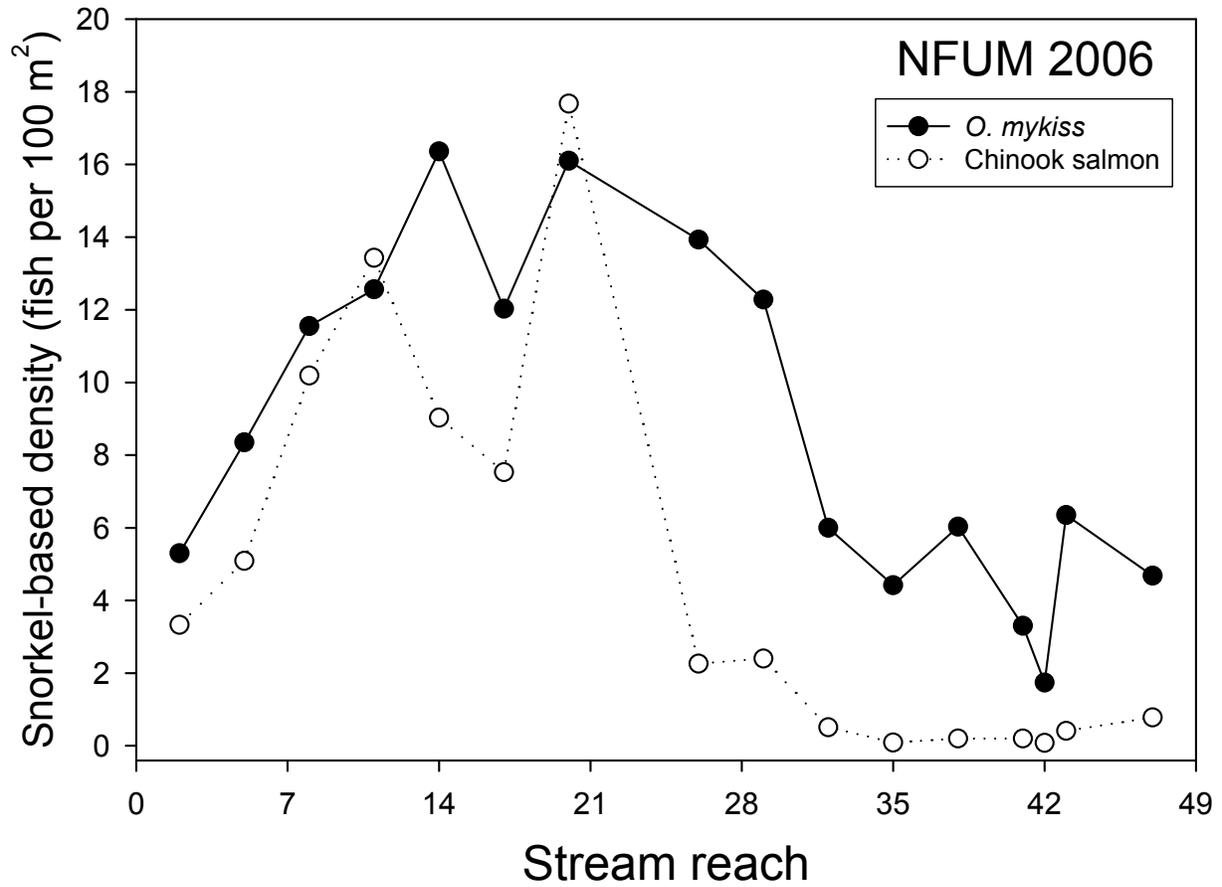


Figure A3. Density of *O. mykiss* spp. and Chinook salmon estimated by snorkel counts in various reaches of the North Fork Umatilla River, 2006.

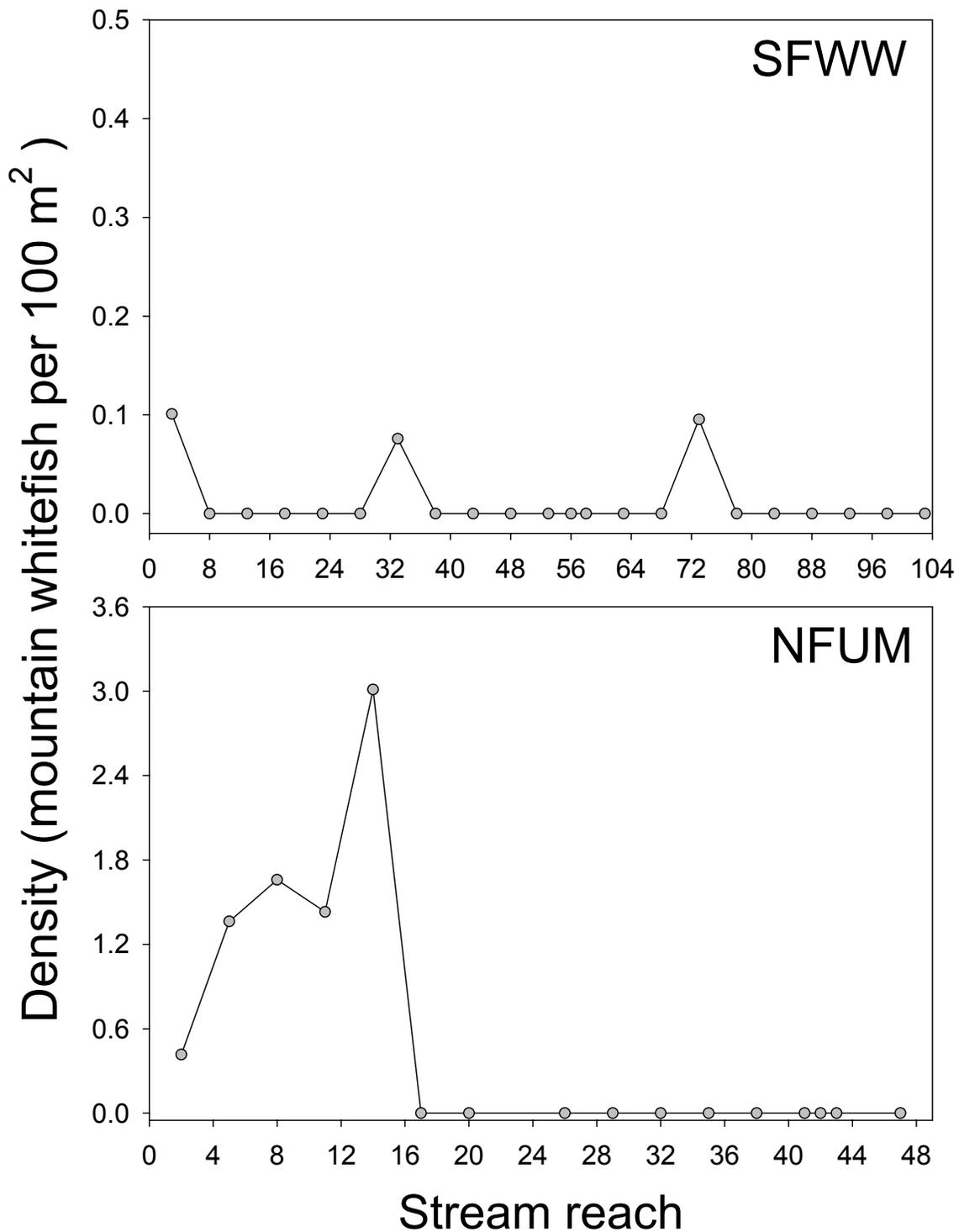


Figure A4. Densities (number per 100 m²) of mountain whitefish observed during snorkeling surveys in the South Fork Walla Walla River and North Fork Umatilla River, summer 2006. Notice change of scale on y-axis between systems.

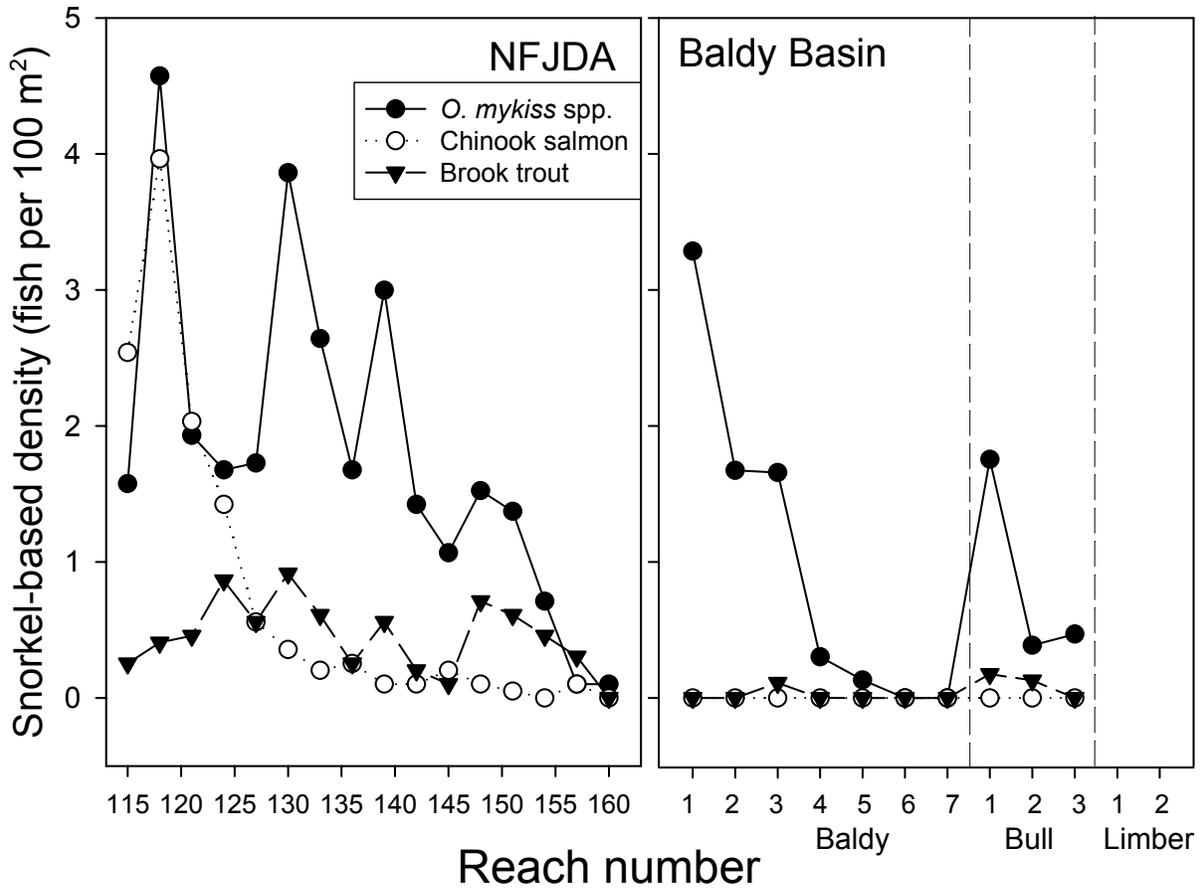


Figure A5. Densities of *O. mykiss* spp., Chinook salmon, and brook trout estimated by snorkel counts in sampled reaches of the North Fork John Day River and Baldy Creek, summer 2006. Baldy Creek, Bull Creek, and Limber Creek are delineated by dashed lines in the Baldy Basin.

APPENDIX 2

Original objectives and tasks specified to meet the original 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.