

**Ecological Interactions Between Non-target Taxa of Concern
and Hatchery Supplemented Salmon
Yakima/Klickitat Fisheries Project Monitoring and Evaluation**

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Executive Summary

Species interactions research and monitoring was initiated in 1989 to investigate ecological interactions among fish in response to proposed supplementation of salmon and steelhead in the upper Yakima River basin. This is the fifteenth of a series of progress reports that address species interactions research and supplementation monitoring of fishes in the Yakima River basin associated with the Yakima/Klickitat Fisheries Project. Data have been collected before and during supplementation to characterize the ecology and demographics of non-target taxa (NTT) and target taxon, and to monitor interactions and supplementation success. Major topics of this report are associated with implementing NTT monitoring prescriptions for detecting potential impacts of hatchery supplementation. This report is organized into four chapters, with a general introduction preceding the first chapter. This annual report summarizes data collected primarily by the Washington Department of Fish and Wildlife (WDFW) between January 1, 2006 and December 31, 2006 in the Yakima River basin, however these data were compared to data from previous years to identify preliminary trends and patterns. Summaries of each of the chapters included in this report are described below.

Chapter 1. Hatchery and wild fish advocates often engage in spirited communication about the ecological interactions between hatchery and wild fish. Many arguments about ecological interactions between hatchery and wild fish have been fueled by a variety of myths, scientific uncertainties, and differences of unstated objectives. In order to reduce the frequency of unfruitful discussions, it is important to expose a variety of myths about ecological interactions and risk. In addition, it is necessary to synthesize what is currently known about ecological interactions. Fifteen myths, seven realities, and five of the most important scientific uncertainties are described. I conclude that ecological interactions between hatchery and wild fish will occur but whether those interactions are biologically significant or socially acceptable, and whether any impacts are statistically detectable are probably dependant upon the hatchery program, the natural environment where hatchery fish are released, and how interactions are evaluated.

Chapter 2. Salmon supplementation and reintroduction programs have the potential to negatively impact other valued fish taxa, which are not the target of enhancement (non-target taxa). We evaluated the impacts of spring Chinook salmon supplementation and coho salmon reintroduction (hereafter supplementation) to non-target fish taxa after eight years of stocking approximately one million yearling smolts annually in the upper Yakima Basin between 1999 and 2006. Field methods included backpack electrofishing and snorkeling in tributaries, and drift-boat electrofishing in the main stem. We used three sequential steps in our evaluation: First, we determined if spatial overlap occurred between supplementation fish and non-target taxa. Second, if overlap occurred, we determined if a change in abundance, size, or biomass occurred during supplementation. Lastly, if a change occurred we determined if the change could be reasonably attributed to supplementation. Spatial overlap and changes in abundance, size, or biomass were determined to be significant if they exceeded containment objectives. Salmon rarely overlapped cutthroat and bull trout in tributaries, but some overlap of cutthroat occurred in relatively high elevations of the main stem, and considerable overlap with rainbow trout occurred in tributaries and the main stem. Salmon overlapped mountain whitefish and

sucker species in the main stem, and dace and sculpin species in tributaries. With the exception of steelhead and mountain sucker, the lower 90% confidence limit of abundance, size, and biomass was above the containment objective for non-target taxa that overlapped significantly with salmon. We used rainbow trout as an analog for steelhead, and subadult suckers as an analog for mountain sucker. The lower 90% confidence limit of rainbow trout size in tributaries, and size and biomass in the main stem, and the lower 90% confidence limit for mountain sucker abundance were below our containment objectives. Comparisons of rainbow trout size in tributaries, size and biomass in the main stem, and mountain sucker abundance in main stem sections with relatively high and low salmon abundance revealed that these change were unlikely to be the result of supplementation (BACIP $P > 0.05$). Our data indicate that early stages of salmon supplementation have not impacted most valued species in the upper Yakima Basin beyond predetermined containment objectives, but the containment objective for steelhead abundance in the Teanaway Watershed has been exceeded (see Chapter 3).

Chapter 3. We used a Before-After-Control-Impact-Paired design to evaluate impacts of hatchery salmon reintroduction and supplementation on rainbow trout *Oncorhynchus mykiss* in two contiguous rivers. Trout and salmon were sampled with backpack electrofishers and abundance, size, and biomass was estimated in treatment and control sites before (1990-1998) and during hatchery releases (1999-2006). We detected impacts to rainbow trout abundance and biomass ($P < 0.05$), but not size ($P > 0.05$) from production scale releases of coho salmon *O. kisutch* (1999) and spring Chinook salmon *O. tshawytscha* (2000-2006) in the Teanaway Basin. Impacts were largest in the stream closest to the release site. The impacts to rainbow trout were likely the result of cumulative impacts from hatchery released Chinook salmon smolts, residualized spring Chinook salmon, and an increase in naturally produced parr. Each one of these life-stages was significantly correlated with differences in trout abundance in at least one treatment location ($P < 0.05$). We also observed that the combined biomass of rearing rainbow trout and spring Chinook salmon was impacted by supplementing spring Chinook salmon ($P < 0.05$). This study highlights the importance of evaluating hatchery programs from a long-term and multi-species perspective.

Chapter 4. The objective of our study was to determine the magnitude of data entry error present in a typical fisheries field station, how those errors might affect common fisheries estimates if undetected, and what method of data entry would be most proficient in preventing errors for the lowest cost. We evaluated four methods of data entry and compared common fisheries estimates derived from each to estimates from verified data. We found no significant difference between estimates generated from any of the four data entry methods and estimates from our verified data. This suggests that a single data entry, or the use of a personal digital assistant (PDA) is sufficient for similar data entry applications. This study provides information that should lower concerns surrounding the use of data collected under different data entry protocols.

Chapter 5. Long-term trend monitoring of resident rainbow trout *Oncorhynchus mykiss* in the Yakima River Basin is a vital component of the Yakima Klickitat Fisheries Project (YKFP) non-target taxa of concern (NTTOC) monitoring and evaluation program. Concerns that supplementation may negatively affect the resident rainbow trout population in the upper Yakima River contributed to the development and implementation of the program. Information on the abundance, size, and distribution of rainbow trout in the Yakima River are collected

annually in five sections of the Yakima River between Roza Dam and the Cle Elum River confluence. The information collected in these sections is presented under the risk containment monitoring framework (Chapter 2) and can be difficult to interpret outside of that context. The purpose of this report chapter is to provide managers, anglers, special interest groups, and all interested readers a summarized version of Yakima River rainbow trout data collected under the YKFP's NTTOC monitoring program in the Yakima Basin. These data should be considered preliminary until published in a scientific journal.

All findings in this report should be considered preliminary and subject to further revision unless they have been published in a peer-reviewed technical journal (i.e., see General Introduction).

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General Introduction

This report is intended to satisfy two concurrent needs: 1) provide a contract deliverable from the Washington Department of Fish and Wildlife (WDFW) to the Bonneville Power Administration (BPA), with emphasis on identification of salient results of value to ongoing Yakima/Klickitat Fisheries Project (YKFP) planning, and 2) summarize results of research that have broader scientific relevance. This is the fifteenth of a series of progress reports that address species interactions research and supplementation monitoring of fishes in response to supplementation of salmon and steelhead in the upper Yakima River basin (Hindman et al. 1991; McMichael et al. 1992; Pearsons et al. 1993; Pearsons et al. 1994; Pearsons et al. 1996; Pearsons et al. 1998; Pearsons et al. 1999; Pearsons et al. 2001a, Pearsons et al. 2001b, Pearsons et al. 2002, Pearsons et al. 2003, Pearsons et al. 2004; Pearsons et al. 2005; Pearsons et al. 2006). Journal articles and book chapters have also been published from our work (McMichael 1993; Martin et al. 1995; McMichael et al. 1997; McMichael and Pearsons 1998; McMichael et al. 1998; Pearsons and Fritts 1999; McMichael et al. 1999a; McMichael et al. 1999b; Pearsons and Hopley 1999; Ham and Pearsons 2000; Ham and Pearsons 2001; Amaral et al. 2001; McMichael and Pearsons 2001; Pearsons 2002; Fritts and Pearsons 2004; Major et al. 2005; Fritts and Pearsons 2006; Temple and Pearsons 2006; Pearsons et al. 2007a; Pearsons et al. 2007b; Pearsons and Temple 2007; Temple and Pearsons 2007). This progress report summarizes data collected between January 1, 2006 and December 31, 2006. These data were compared to findings from previous years to identify general trends and make preliminary comparisons. Interactions between fish produced as part of the YKFP, termed target species or stocks, and other species or stocks (non-target taxa) may alter the population status of non-target species or stocks (Figure 1). This may occur through a variety of mechanisms, such as competition, predation, and interbreeding (Pearsons et al. 1994; Busack et al. 1997; Pearsons and Hopley 1999). Furthermore, the success of a supplementation program may be limited by strong ecological interactions such as predation or competition (Busack et al. 1997). Salmon are so influential in the flow of energy in aquatic systems, that it is unlikely that changes in their abundance will not influence others species (Figure 2).

Our work has adapted to new information needs as the YKFP has evolved. Initially, our work focused on interactions between anadromous steelhead and resident rainbow trout (for explanation see Pearsons et al. 1993), then interactions between spring Chinook salmon and rainbow trout, and recently interactions between spring Chinook salmon and highly valued non-target taxa (NTT; e.g., bull trout); and interactions between strong interactor taxa (e.g., those that may strongly influence the abundance of spring Chinook salmon; e.g., smallmouth bass) and spring Chinook salmon. The change in emphasis to spring Chinook salmon has largely been influenced by the shift in the target species planned for supplementation (Bonneville Power Administration et al. 1996; Fast and Craig 1997). Originally, steelhead and spring Chinook salmon were proposed to be supplemented simultaneously (Clune and Dauble 1991). However, due in part to the uncertainties associated with interactions between steelhead and rainbow trout, spring Chinook and coho salmon were supplemented before steelhead. This redirection in the species to be supplemented has prompted us to prioritize interactions between spring Chinook and rainbow trout, while beginning to investigate other ecological interactions of concern. Pre-facility monitoring of variables such as rainbow trout density, distribution, and size structure was continued and monitoring of other NTT was initiated in 1997.

This report is organized into five chapters that represent major topics associated with monitoring stewardship, utilization, and strong interactor taxa. Chapter 1 reports on common myths associated with interactions between hatchery and wild fish. Chapter 1 will be submitted for publication in Fisheries. Chapter 2 reports the results of non-target taxa monitoring after the eighth release of hatchery salmon smolts in the upper Yakima River Basin. Chapter 3 documents impacts to rainbow trout in the Teanaway River basin associated with salmonid supplementation. Chapter 4 presents an evaluation of data entry error and data proofing techniques and the potential influence data errors have on common fisheries population metrics. Chapter 4 will be submitted for publication in Fisheries. Chapter 5 reports the abundance, and size of Yakima River rainbow trout outside of the framework of risk containment monitoring.

The chapters in this report are in various stages of development and should be considered preliminary unless they have been published in a peer-reviewed journal. Additional field-work and/or analysis is in progress for topics covered in this report. Throughout this report, a premium was placed on presenting data in tables so that other interested parties could have access to the data. Readers are cautioned that any preliminary conclusions are subject to future revision as more data and analytical results become available.

Except where otherwise noted, the methods and general site descriptions are the same as described in previous reports (Hindman et al. 1991; McMichael et al. 1992; Pearsons et al. 1993; Pearsons et al. 1994; Pearsons et al. 1996; Pearsons et al. 1998; Pearsons et al. 1999; Pearsons et al. 2001a; Pearsons et al. 2001b; Pearsons et al. 2002; Pearsons et al. 2003; Pearsons et al. 2004; Pearsons et al. 2005; Pearsons et al. 2006; Temple and Pearsons 2007).

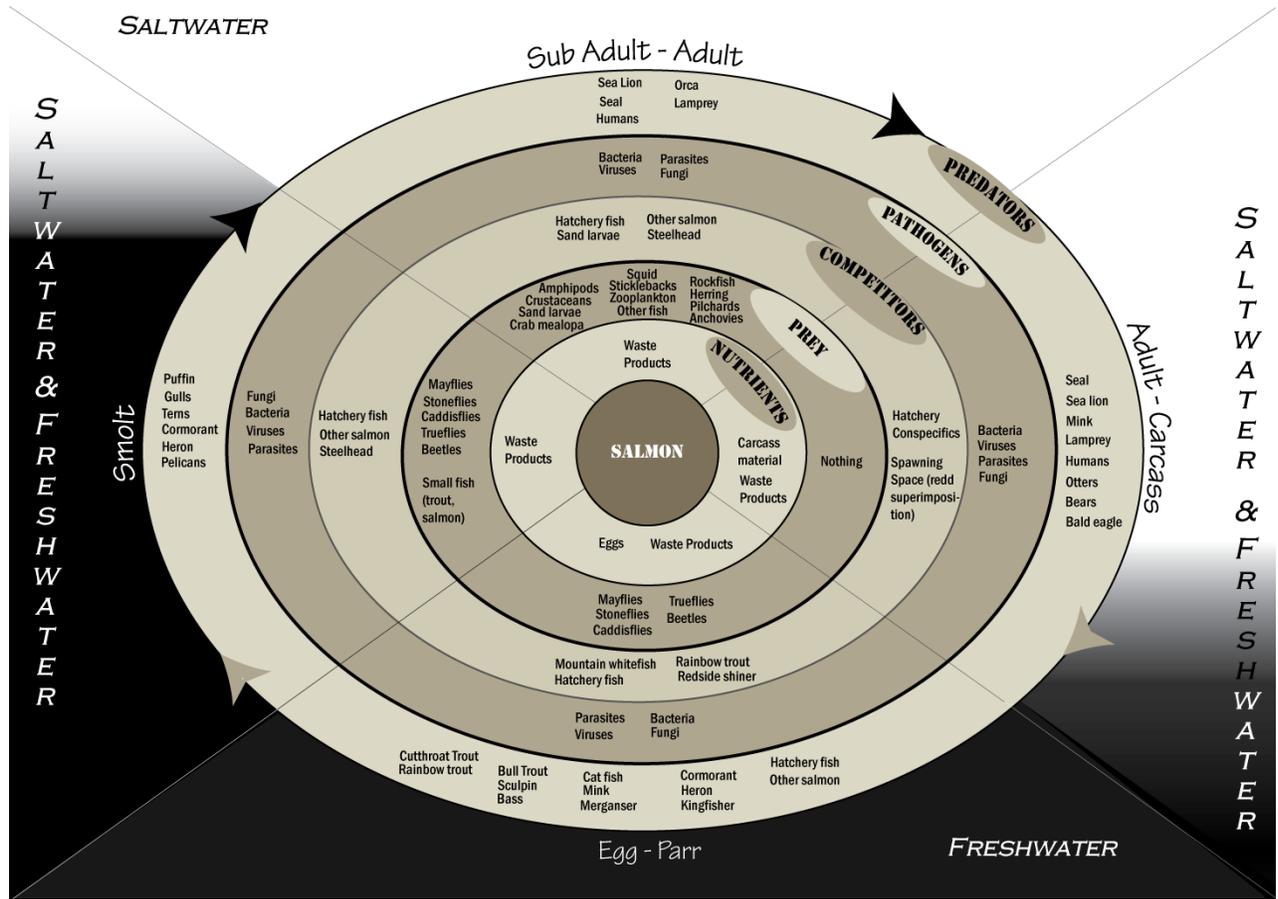


Figure 1. Schematic showing some of the large number of species that interact with salmon throughout their life-cycle.

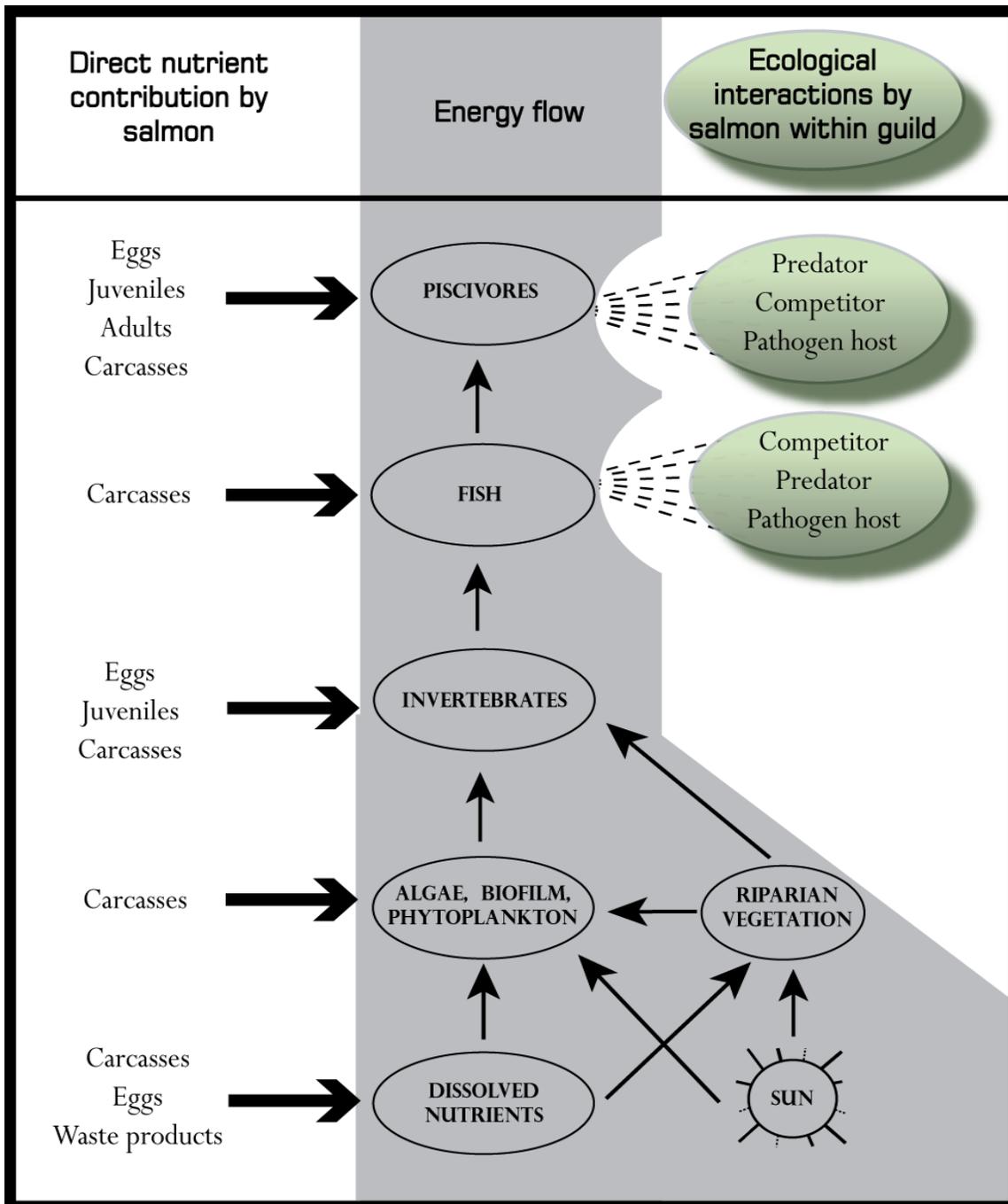


Figure 2. Schematic showing the influence that salmon have on the energy flow within the food chain.

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Chapter 1

Myth, Reality, and Uncertainty About Ecological Interactions and Risks Between Hatchery and Wild Salmonids

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Abstract

Hatchery and wild fish advocates often engage in spirited communication about the ecological interactions between hatchery and wild fish. Many arguments about ecological interactions between hatchery and wild fish have been fueled by a variety of myths, scientific uncertainties, and differences of unstated objectives. In order to reduce the frequency of unfruitful discussions, it is important to expose a variety of myths about ecological interactions and risk. In addition, it is necessary to synthesize what is currently known about ecological interactions. Fifteen myths, seven realities, and five of the most important scientific uncertainties are described. I conclude that ecological interactions between hatchery and wild fish will occur but whether those interactions are biologically significant or socially acceptable, and whether any impacts are statistically detectable are probably dependant upon the hatchery program, the natural environment where hatchery fish are released, and how interactions are evaluated.

Hatchery and wild fish advocates often engage in lively communication about the ecological interactions between hatchery and wild fish (Meffe 1992; Hilborn 1992; White et al. 1995; Brannon et al. 2004). Some communications can even degrade into passionate pseudo-scientific arguments that have the flavor of scientific credibility (Lackey 2004; 2006; 2007) but do not advance the true evaluation of costs and benefits of hatchery operations (Waples 1999). There have been pleas for improved civility among fisheries professionals (Martin et al. 1992; Stickney 1994). One way to improve civility and increase objectivity of discussion is to conduct formal risk assessments. Disciplined risk assessment procedures have been developed to improve predictions of ecological costs and benefits associated with hatcheries (Pearsons and Hopley 1999; Ham and Pearsons 2001; Busack et al. 2005). However, even with disciplined procedures, biased information, viewpoints, and interpretation can taint the assessment of costs and benefits (Lackey 2006; 2007). This bias in outlook is the subject of this paper.

What is “known” about ecological interactions are often wrong, and what is “not known” about ecological interactions is often unrecognized. It is critically important to productive future dialog to clarify in general terms what is known about ecological interactions involving hatcheries. This would complement the work of Waples (1999), who addressed myths about hatchery programs primarily from a genetic perspective, but intentionally avoided ecological issues. This avoidance was not because “ecological considerations are unimportant; rather, the topic is too complex and my expertise in that area too limited to do it justice.”

During the many years that I have engaged in assessing ecological risks of hatchery operations, I have encountered many partial truths and falsehoods that are uncritically accepted as truth. These “myths” may not be found in citable documents, but are part of the “scientific” dialogue that influences local decisions, and part of the way that the scientific literature is interpreted and applied. The myths may be so persuasive that formal risk assessments are perceived as a waste of time. My purpose in writing this paper is to expose these myths so that ecological risk assessments will be conducted and the quality of assessments improved. In many cases there is not empirical data to conclusively falsify a myth. However, the converse is also true – there is also insufficient empirical data to support acceptance of a myth. My goal is to provide sufficient evidence to question the veracity of a myth so that the burden of proof for acceptance of a myth is shifted to the proponent of the myth. Some of these myths may seem

ridiculous and unworthy of text, but I have encountered them repeatedly so I have opted to include them. In some cases, a myth may not be presented as bluntly as I have portrayed, but the essence of a variety of versions is contained within my portrayal of the myth.

I identified 15 myths and qualitatively assessed 1) the frequency with which I have encountered it and 2) the risk of negative consequences to non-target fish if the myth is accepted (Table 1). The list of myths, realities, and uncertainties should not be considered exhaustive, but illustrative of the important issues that confront risk assessors and managers. Myths are loosely arranged into categories of theory, statistics, interpretation, risk assessment and management. I will refer to hatchery fish as fish that are released from a hatchery and wild fish as natural origin con-specifics and other species that aren't the target of enhancement.

Table 1. List of ecological interaction myths, frequency of myth occurrence, and the risk of negative consequences to wild fish if the myth is accepted.

#	Myth	Frequency	Risk
1	Low densities of wild fish means that interactions will be inconsequential	High	High
2	Competitive superiority of wild fish means that wild fish aren't impacted	High	Med
3	Release of hatchery fish downstream of species of concern means no impact will occur	High	Med
4	Lack of a statistically significant impact means that an impact has not occurred	High	High
5	Statistically significant impacts are biologically and socially significant	Low	Low
6	Statistically significant differences in microhabitat use means that impacts will be negligible	Med	Med
7	Statistically significant changes in abundance means that it was caused by hatcheries	Low	Low
8	The published literature is a representative sample of ecological interactions and lack of a documented impact in the literature means that impacts don't occur	Med	High
9	Observation of an interaction means that an impact has occurred	High	Low
10	Results from interaction studies can be directly applied to an area of interest	High	Low
11	Interactions from hatchery fish are always negative	Med	Low
12	Historical coexistence translates into present coexistence	High	High
13	Acceptable impacts to wild fish (e.g., containment objectives) shouldn't be any less than what is scientifically detectable	High	High
14	Adaptive management can contain risks associated with ecological interactions within acceptable levels	High	High
15	Ecosystems are so complicated that we can't predict what interactions will occur	Med	High

Myths

Myth #1: Low densities of wild fish means that interactions will be inconsequential

It is plausible that lower densities of wild fish will decrease the total number of interactions that will occur between hatchery and wild fish. However, the proportion of the wild population that engages in interactions may be higher than at higher densities. In other words, the number of interactions that occur when a population is at low abundance may be low but the proportional impact to the wild population may be high. For instance, stocking one million smolts is more likely to negatively impact (expressed as a proportion of the population) a wild population of a thousand than a wild population of one million. In addition, the resiliency of a wild population with low abundance may be lower than populations that have high abundance. Some evidence indicates that density-dependent growth of stream-dwelling salmonids is stronger at low population densities rather than at high densities, probably due to exploitative competition (Grant and Imre 2005). Density dependent survival of juvenile Chinook salmon was documented in the Snake River; a region with such depressed abundance levels that the population is protected by the federal government (Achord et al. 2003). Furthermore, smolt to adult survival rates of Snake River Chinook salmon was negatively associated with hatchery steelhead releases (Levin and Williams 2002).

Myth #2: Competitive superiority of wild fish means that wild fish aren't impacted

Some studies have shown that wild fish behaviorally dominate hatchery fish in some circumstances (Metcalf 2003; Weber and Fausch 2003). These studies are used to support the contention that wild fish will dominate hatchery fish in the natural environment. In some cases, wild fish will dominate hatchery fish, but domination of a competitor is not without cost to the dominant. Dominant fish may have to interfere with subordinates (e.g., threat, bite, chase) and lose feeding opportunities in the process. Dominant fish may also subject themselves to greater predation risk while engaging in behavioral contests (Bachman 1984; Johnsson and Abrahams 1991; Yamamoto and Reinhardt 2003). In short, although wild fish may be competitively superior in some instances, the cost of superiority may be high and result in lower survival or growth in the presence of hatchery fish. Furthermore, even when wild fish are more dominant than hatchery fish, dominance is rarely complete and hatchery fish are dominant in some contests. For instance, a study may find that wild fish dominate 60% of the contests with hatchery fish. Although wild fish may be more dominant than hatchery fish, 40% of the contests are dominated by hatchery fish and thereby may impact wild fish.

Myth #3: Release of hatchery fish downstream of species of concern means no impact will occur

It is appealing to assume that impacts will not occur if hatchery fish are released below species of concern. However, hatchery fish may swim far upstream of their release site and overlap with species of concern (Hume and Parkinson 1987; McMichael and Pearsons 2001). Wild fish may also swim downstream into areas containing hatchery fish. Hatchery fish may also leave a shadow of reduced food abundance that may impact later migrating, later emerging, or resident salmonids. Exploitative competition, may be particularly important in areas such as

estuaries and near shore marine environments that experience large numbers of fish at different times or in freshwater environments when fish densities are low (Grant and Imre 2005; Williams 2006).

Myth #4: Lack of a statistically significant impact means that an impact has not occurred

When studies do not detect a statistically significant impact, they are usually used to support the conclusion that impacts did not occur. Detection of a statistically significant impact may not be found because of a lack of impact or because of inadequate statistical power (Peterman and Bradford 1987; Ham and Pearsons 2000). In many cases, detection of impacts to population abundance of less than 20% is highly unlikely using current methods (Ham and Pearsons 2000). This illustrates the difficulty of detecting impacts to population abundance in all but very large impacts and therefore the likelihood of falsely accepting the interpretation of no impacts.

Myth #5: Statistically significant impacts are biologically and socially significant

The statistical power to detect small impacts in some studies may be very high. For example, a study may be able to detect an impact to fish size of 0.1%. Although this impact may have been caused by hatchery fish the impact may be biologically insignificant when compared to other factors that influence fish size. Furthermore, even statistically and biologically significant impacts may be socially acceptable. In other words, managers may be willing to accept certain levels of ecological impacts in order to gain the expected benefits of artificial propagation.

Myth #6: Statistically significant differences in microhabitat use means that impacts will be negligible

Some have interpreted statistical differences in microhabitat use between hatchery and wild fish as reason to believe that impacts to wild fish will be negligible. Many studies have presented statistically different microhabitat use between hatchery and wild fish (Weber and Fausch 2003). These differences may be the result of interactive or selective segregation (Chapman 1966, Everest and Chapman 1972, Bisson et al. 1988). If microhabitats differ because of interactive segregation, then both hatchery and wild fish may suffer impacts to growth or survival. Even if fish select different microhabitats because of inherently different habitat preferences (Ross 1986; Bisson et al. 1988), statistically significant differences in the means does not mean that overlap does not occur. For example, the mean focal point velocity of hatchery and wild fish may differ, but 80% of the population may have overlapping focal point velocities. Ability to detect differences in microhabitat means is a function of statistical power. A large sample size could result in the power to detect very small difference in microhabitat means.

Myth #7: Statistically significant changes in abundance means that it was caused by hatcheries

Sometimes studies are conducted without adequate controls and detections in changes are interpreted as impacts caused by hatchery operations. Before-after designs or trend monitoring

are helpful to detect changes and trends but are insufficient, in and of themselves, to attribute causation unless it can be demonstrated that other factors varying with time are accounted for. Annual changes that occur after hatchery operations start may be due to changes in environmental conditions or a host of other factors.

Myth #8: The published literature is a representative sample of ecological interactions and lack of a documented impact in the literature means that impacts don't occur

When insufficient evidence of ecological impacts in the scientific literature occurs, then this lack of evidence is sometimes used to support the conclusion that impacts do not occur. However, the lack of literature may simply be because studies have not been conducted or that they have been conducted but haven't been published (Moller and Jennions 2001). The scientific literature is likely to be biased in a number of ways. First, many journals are uninterested in publishing results that fail to reject the null hypothesis (Moller and Jennions 2001). In other words, studies that do not demonstrate a statistically significant interaction or impact are less likely to be published. Second, those studies that are successful in publishing statistically insignificant ($\alpha > 0.05$) data may be committing a type II statistical error because of a lack of statistical power (myth #4). The implication of this bias is that the literature cannot be viewed as a representative sample in syntheses.

Myth #9: Observation of an interaction means that an impact has occurred

Studies may reveal that interaction mechanisms such as competition or predation occur. However, without evaluating the impact on variables such as abundance, it is unclear whether observed mechanisms will manifest into population level impacts. In addition, compensatory mechanisms may undo the impacts of the mechanisms studied. Sometimes the only information that is available is anecdotal. For example, an angler may have observed hatchery fish chasing wild fish or eating a wild fish. Anecdotal information about an interaction mechanism (e.g., predation) is sometimes magnified into concluding that hatchery fish are impacting wild fish abundance.

Myth #10: Results from interaction studies can be directly applied to an area of interest

Ecological interaction studies are rarely sufficient to draw accurate and precise inferences to a specific area of interest. As such, insufficient study results are applied to a study area. For example, the following sub-myths can occur; results from one stream apply to all others; laboratory studies can be directly applied to the natural environment; small-scale study results apply to larger scales; results from one impacted life-stage (e.g., parr) will apply to a later life-stage (e.g., adult); and impacts observed during one stage of supplementation apply to other stages. Due to the lack of basin-specific information, many will use information produced from other areas and apply those results to their program. However, many hatchery operations and stream systems differ substantially from one another. The results from one stream may not be transferable to another unless it can be supported with suitable information.

Results from laboratory studies are frequently more available than field studies. As such, they are frequently used to support the presence or absence of interactions that will occur in the natural environment. In some cases, this may be appropriate, but often the laboratory conditions

are so different than the natural conditions that the results cannot be applied to the natural environment very well.

Results from studies conducted at the territory size or channel unit scale may not be directly transferable to scales such as the watershed (Levin 1992). For example, a reduction in growth within an experimentally enclosed pool or enclosure may not be what occurs at larger scales where greater movement and partitioning of resources are possible.

Managers are usually most concerned about impacts to adult population size. However, most interactions studies are conducted during early life stages. Impacts that occur to early life-stages may be compensated for by compensatory growth or survival and result in no impact to the adult population or be amplified (Zabel and Achord 2004). Conversely, lack of an impact at an early life-stage does not mean that impacts at later life-stages don't occur.

The strength and outcome of interactions could be dramatically different among different stages of supplementation (Pearsons 2002). Different stages of supplementation can be characterized by the relative number of target fish that spawn in the river and hatchery environments. Pearsons (2002) characterized successful supplementation programs as having Baseline, Broodstock, Building, and Boundary stages. If impacts occur they are most likely to be detected late in the Building stage.

Myth #11: Interactions from hatchery fish are always negative

Discussions about ecological interactions between hatchery and wild fish usually center on negative interactions to the exclusion of positive ones. Negative interactions such as predation, competition, and disease are usually highlighted. Positive interactions such as nutrient addition and predator swamping are usually ignored. A true assessment of risk can only be completed after both positive and negative interactions are considered (Pearsons and Hopley 1999).

Myth #12: Historical coexistence translates into present coexistence

Many species that are the focus of artificial propagation are released into places that they historically occurred. Some suggest that historical coexistence of species means that wild (conspecifics and non-target species) and hatchery fish will be able to coexist in the present. This assumes that environmental conditions and species assemblages of the past are functionally similar to the present. This assumption is likely to be false in many if not most areas where hatchery programs exist. If environmental conditions, such as habitat complexity or water temperature has changed over time then we cannot assume that the interactions will be the same. Studies have shown that outcomes of interactions can change when environmental conditions change (Reeves et al. 1987; Li et al. 1994; Tait et al. 1994). In addition, establishment of non-native species may also change the functional relationships among species or races (Li et al. 1987; Fritts and Pearsons 2004; 2006). Furthermore, reintroduction or enhancement of species in areas where other non-target species have been released from competition are likely to result in a reduction in abundance of non-target species (Pearsons 2002).

Myth #13: Acceptable impacts to wild fish (e.g., containment objectives) shouldn't be any less than what is scientifically detectable

Containment objectives are the levels of impacts to non-target taxa that are acceptable in order to achieve the predicted benefits of a hatchery program and are derived based on the perceived values of different species in the ecosystem (Pearsons and Hopley 1999). They are the management targets that scientists assess risks against and attempt to detect with a monitoring plan. Monitoring plans vary in their ability to detect different levels of impacts (Ham and Pearsons 2000). Variation in impact detection can be driven by non-biological factors such as funding. Containment objectives should be compared to predicted impact detection to determine the level of risk containment that is possible (Ham and Pearsons 2001). However, the value based containment objectives should not be adjusted by the level of impact that is scientifically detectable. If this were the case, then a species that has low societal value and low level of detectable impacts would have a lower containment objective than a Federally listed species that has a high level of detectable impact. It is important for managers to know whether a monitoring plan can or cannot detect and contain risks at the desired level. This information could have implications about the potential benefits and costs of a hatchery (Pearsons and Hopley 1999).

Myth #14: Adaptive management can contain risks associated with ecological interactions within acceptable levels

Due to the desire to stock hatchery fish and the high uncertainties associated with ecological risk assessments, many propose to adaptively manage hatchery programs. Adaptive management is the change of management approach that is guided by new results from carefully designed and executed monitoring and evaluation plans (Lee 1993). Adaptive management may not be able to contain risks adequately because of inability to detect impacts before containment objectives are exceeded, or because actions to contain risks are not available (Ham and Pearsons 2000; Ham and Pearsons 2001). Furthermore, the concept of adaptive management, though laudable, has been more useful as a concept than as a practical means of management (Lee 2000).

Myth #15: Ecosystems are so complicated that we can't predict what interactions will occur

While it is true that ecosystems are very complicated and no one has ever really studied all of the interactions within an ecosystem, we know enough about interactions to make less mistakes than if we acted as if we knew nothing (e.g, the just do it approach). We know enough about the potential interactions that could occur and some of the factors that will influence the strength of these interactions, that we can make reasonable predictions. For instance, most people would agree that higher spatial-temporal overlap of hatchery and wild fish and higher number of hatchery fish increases interaction risk (McMichael et al. 1999a).

Realities

While it is true that there are many myths about ecological interactions and risks, there are also a number of realities. Realities can come from a variety of sources such as the presence or absence of empirical studies, ecological theory, or risk assessment tools. These realities could be used in support of conducting risk assessments and identifying critical knowledge gaps.

Although many other realities besides the ones that I present are possible, I attempt to focus on ones that provide the most heuristic value.

Reality #1: Ecological interactions between hatchery and wild fish will occur

Releasing thousands or millions of fish into natural watersheds will result in direct or indirect interactions with wild fish. Fish released from a hatchery will eat a variety of prey, occupy space, be eaten by predators, influence the flow of nutrients through their carcasses, and potentially serve as amplifiers of pathogens. Furthermore, many other con-specifics and non-target species of wild fish share similar food sources, space, predators, and susceptibility to pathogens as hatchery fish. Ecological communities are too interdependent for hatchery fish to live within an ecological vacuum. In addition, theoretical support for ecological interactions between hatchery and wild fish is strong. The following are examples of theoretical support for the occurrence of ecological interactions:

- Ecological resources (e.g., food and space) are finite and many populations, including fishes, have been demonstrated to be regulated through density-dependent mechanisms (Chapman 1966; Achord et al. 2003; Grant and Kramer 1990). Adding additional animals (hatchery fish) to the natural environment has the potential to limit the resources to wild animals.
- Hatchery fish that are large enough, have a tendency towards piscivory, and have available prey will eat other fish (Pearsons and Fritts 1999; Hawkins and Tipping 1999). Adding piscivores into waters containing prey-sized fish has the potential to increase mortality of wild fish.
- Susceptibility to disease is theorized to be an interaction between the environment, host, and the pathogen (Snieszko 1974). The presence of hatchery fish may increase stress levels to wild fish and increase susceptibility of wild fish to disease. An increase in fish density has been shown to increase stress and susceptibility of disease to rainbow trout (Peters et al. 1988; 1991). Cases of disease transfer between fishes in the natural environment has been reported (Mitchum 1981, Goede 1986; McVicar 1997).
- Salmon carcasses can have a dramatic influence on food web productivity. Addition of hatchery coho salmon carcasses increased growth of natural origin salmonids (Bilby et al. 1998). Addition or reduction in carcasses caused by hatchery operations can increase or decrease the food available to fish.
- Many fish and bird species have been shown to consume salmonids (Wood 1987a; b; Fritts and Pearsons 2004, 2006). Animals will move to areas of high prey abundance (e.g., release of hatchery fish; Collis et al. 1995; Shively et al. 1996 a,b; Major et al. 2005) and will switch their diet to prey of high abundance. This has the potential to increase or decrease impacts to wild fish (Peterman and Gatto 1978; Nickelson 2003).
- Animals have a limit to what they can eat and it is possible that hatchery fish could swamp predators and reduce impact on wild fish (Peterman and Gatto 1978; Wood 1987 a; b).
- Reducing spatial and temporal overlap decreases most interactions and therefore it is likely to reduce ecological impacts (McMichael et al 1999a).

Reality #2: Many mechanisms of ecological impacts have been observed in the lab and the field

There have been many review papers that have been published about the various interactions mechanisms that could occur between hatchery and wild fish (Marnell 1986; Fresh

1997; Weber and Fausch 2003). It has been conclusively demonstrated that hatchery fish do eat wild fish in some situations (Sholes and Hallock 1979; Hawkins and Tipping 1999; Sharpe et al. in press) and will use agonistic interactions against wild fish in competitive contests (Rhodes and Quinn 1998; McMichael et al. 1999b; Weber and Fausch 2003). Other mechanisms such as disease, indirect predation, behavioral anomalies, and nutrient dynamics are less well studied but theoretically possible (Hillman and Mullan 1989; Pearsons and Hopley 1999; Nickelson 2003; see Reality #1).

Reality #3: Few population level impacts have been studied and conclusively linked to smolt programs

Although many ecological mechanisms of impact have been demonstrated (e.g., hatchery fish eating wild fish), few studies have been published that evaluate the impacts of a production scale hatchery in natural environments (e.g., percent of population consumed, or decrease in abundance; Weber and Fausch 2003; Levin and Williams 2002). Only a handful of studies have attempted to measure impacts and most of these have focused on impacts to naturally produced conspecifics (Nickelson et al. 1986; Chilcote 2003; Nickelson 2003) and/or stocking salmon before the smolt stage (Bjornn 1978; Tripp and McCart 1983; Nickelson et al. 1986). Pearsons and Temple (2007) evaluated smolt release impacts to three trout species during the early stages of salmon supplementation and reintroduction programs. However, it did not evaluate the long-term effects of later stages of hatchery programs. I am not aware of any study that has conclusively evaluated the impacts of hatchery smolt programs on valued non-target taxa in natural environments.

Reality #4: Population level impacts to abundance are difficult to detect

Population level impacts from hatchery programs are difficult to detect because of the high natural variability in abundance (Ham and Pearsons 2000). This variability may be further enhanced by variation in impacts associated with different stages of supplementation (Pearsons 2002). The implications of these challenges are that monitoring programs will likely be long in duration and therefore expensive. Furthermore, some rivers are very difficult to sample because of access issues, hydrology, temperature, or turbidity. Finally, absence of adequate control streams can also increase or preclude the ability to produce conclusive results.

Reality #5. Abundance monitoring will rarely be sensitive and timely enough to detect small impacts that are important to managers

Detecting impacts of less than 20% to population abundance may be desirable from a management perspective, but it is likely that the probability to do this is low (Ham and Pearsons 2001). For example, managers may be interested in containing the impacts of a hatchery program to a species protected by the Endangered Species Act. However, impacts between 5% and 20% may be socially unacceptable but also scientifically undetectable. Furthermore, impacts are unlikely to be detectable within the time that a manager may want to contain further impacts (e.g., 5 years). As such, risk containment measures cannot be triggered in the event that unacceptable impacts occur (Ham and Pearsons 2001). This may mean that more risk averse

strategies or monitoring more powerful variables (e.g., size structure, interactions mechanisms) will be necessary to contain risks within acceptable levels.

Reality #6: Ecological interactions are context specific

Outcomes of ecological interactions can vary depending upon the ecological conditions of the environment (Southwood 1977) and the characteristics of hatchery and wild fish. No two river systems or hatcheries are exactly the same. For example, changes in ecological conditions such as water temperature has been shown to change outcomes of ecological interactions (Reeves et al. 1987; Tait et al. 1994). Furthermore, differences in size at release, time of release, and degree of domestication could influence competitive dominance (Berejikian et al. 1996, Metcalfe et al. 2003, Pearsons et al. in press). Hatchery operations vary in a variety of important characteristics such as the species cultured, number of fish released, release strategy (e.g., volitional vs. direct stream releases), time and location of release, size at release, and disease history. Natural streams also vary in a number of important characteristics such as discharge, habitat complexity, water temperature, carrying capacity, productivity, and species composition.

Reality #7: The sources of many hostile arguments are political

The most difficult exercise of conducting a risk assessment or applying the results of risk assessments is the valuing of different benefits and costs associated with a hatchery. A negligible or unimportant impact to one stakeholder may be excessive and important to another stakeholder (Lackey 2006). Many stakeholders think that they are arguing about differing scientific interpretations when in actuality they are arguing about what constitutes a significant impact (Lackey 2004). Others mask personal policy preferences with science to increase credibility of an argument (Lackey 2006; 2007). A significant impact is determined based on stakeholder values as opposed to science. Said in another way, stakeholders will differ in weighting how much impact they are willing to endure in order to get the predicted benefits of a hatchery program. For example, wild steelhead advocates might be unwilling to accept any impacts to get the benefits of a hatchery coho salmon program, but hatchery coho salmon advocates may be willing to accept a rather large impact in order to get the benefits of a hatchery program. It is important to acknowledge that some species, and the people that champion them, will be winners and others will be losers (Lackey 2006).

Key Uncertainties

The realities about ecological interactions illustrate the high level of uncertainty associated with hatchery impacts. The fact is that ecological interactions are often complicated and the most relevant information difficult to obtain. However, some uncertainties are bigger and potentially have more management implications than others. I have attempted to identify the most important uncertainties relative to the lack of information and the importance of the information at the highest geographic scale. However, in particular locations (e.g., watersheds) a different prioritization could easily be supported. These uncertainties could be used to guide or prioritize evaluation programs of ecological interactions, particularly at the State and Federal levels.

Uncertainty #1 Cumulative impacts of multiple hatchery programs

Many wild fish interact with fish from multiple hatchery programs. For example, wild steelhead originating in the Yakima Basin may interact with hatchery coho, fall Chinook, and spring Chinook salmon that are released into the Yakima basin, the many other hatchery fish from other locations in the Columbia basin (migration corridor and estuary), and hatchery fish from outside of the Columbia basin (ocean). Cumulative impacts are most likely to occur in areas where many hatchery stocks overlap those of wild fish such as in main stem migration corridors, estuaries, and oceans. Unfortunately, these locations are the least studied locations with respect to ecological interactions (Zimmerman and Nielsen 2004). There is mounting evidence that ocean carrying capacity is limited for anadromous salmonids and that density dependent impacts may occur from the large number of hatchery fish that are released from multiple locations (Bisbal and McConnaha 1998; Wertheimer et al. 2004; Ruggerone and Nielsen 2004). Impacts could be particularly important in areas such as the Far East of Russia where large numbers of fish are released, but where hatchery impact evaluations are relatively scarce (Zaporozhets and Zaporozhets 2004). It is possible that hatchery impacts to a wild population may not be detectable, but impacts of multiple programs may be detectable.

Uncertainty #2 Community/Population level impacts of hatchery smolt programs throughout the duration of hatchery stages

As stated in reality #3 there are relatively few studies that have investigated this important topic. The advantage of evaluating impacts to the population of interest is that it incorporates all of the potential mechanisms of impacts. Due to the expense and duration of this kind of work, it is important to design experiments at fairly large scales so that inferences can be made across large geographic areas. Experimental designs might benefit by testing predictions of ecological risk assessments (e.g., Busack et al. 2005) across a range of outputs. Furthermore, it is important to quantify the magnitude of the impact, not just whether impacts were statistically significant. If impacts are detected, then more detailed studies might be conducted to determine the specific mechanisms causing impacts.

Uncertainty #3: Unstudied interaction mechanisms and locations of interaction

Some interaction mechanisms and the interactions that occur in some locations are poorly understood. Interaction mechanisms, such as disease and indirect predation have rarely been investigated, but have the potential to be very important. In addition, interactions in certain locations, such as the migration corridor, estuary and ocean are also very underrepresented in the literature. Most literature about ecological interactions has focused on the freshwater environment.

Uncertainty #4: Situations that cannot be evaluated with a monitoring program

There will be some locations that will not be amenable to sampling because of physical constraints such as remoteness and many locations where money will not be available to adequately monitor and evaluate effects of interactions. In these cases, risk assessment models or approaches may be the best that can be done to provide the best balance of benefits and costs.

There are currently tools available to assess risks, but improvement and expansion of these tools would be beneficial. An expert based approach for risk assessment has been developed and has been used in a variety of programs to assess risks (Pearsons and Hopley 1999). In addition, an individual based model called PCD Risk 1 is available to assess predation, competition, and disease risks of smolt programs to freshwater salmonids (Busack et al. 2005). Similar models that include unstudied mechanisms and locations of interactions (Uncertainty #3) should be developed. Furthermore, other models that incorporate direct and indirect interactions that occur throughout a community can be applied to risk assessment (Dambacher et al. 1999, Dambacher 2001).

Uncertainty #5: Valuation of different species that share resources

Different cultures, organizations, and generations assign different and potentially conflicting values to different fish species. One group may prize Chinook salmon above all other species; another group values native trout, and still another values non-native smallmouth bass. Impacts, short of those contributing to extinction of native species, cannot be interpreted by managers as acceptable without determining what impact to other species is acceptable in order to achieve the benefits of a hatchery program (Pearsons and Hopley, 1999; Ham and Pearsons 2001). Policy makers have rarely made these difficult species valuations (although see Pearsons and Temple 2007) and often manage by inaction or lean on scientists to provide the answer (Lackey 2007). Other times, scientists step outside of their domain and advocate particular preferences in the guise of science (Lackey 2004; 2006; 2007).

Conclusion

Ecological interactions between hatchery and wild fish will occur but whether those interactions are biologically significant or socially acceptable, and whether any impacts are statistically detectable are probably dependant upon the hatchery program, the natural environment where hatchery fish are released, and how interactions are evaluated. There are many things that we know and don't know about ecological interactions between hatchery and wild fish. It is important to distinguish between myth, reality, and uncertainty so that expected benefits and costs of interactions can be accurately compared and evaluated. One of the reasons that we have so many myths and substantial uncertainties associated with ecological interactions is because there has been limited investment in obtaining information and those investments have not been spent on resolving uncertainties that will allow for broad application of results. In order to improve our ability to assess ecological risks we will need to build upon our current tools and conduct large-scale experiments addressing the most critical uncertainties.

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Chapter 2

Impacts of early stages of salmon supplementation and reintroduction programs on non-target fish taxa

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Abstract

Salmon supplementation and reintroduction programs have the potential to negatively impact other valued fish taxa, which are not the target of enhancement (non-target taxa). We evaluated the impacts of spring Chinook salmon supplementation and coho salmon reintroduction (hereafter supplementation) to non-target fish taxa after eight years of stocking approximately one million yearling smolts annually in the upper Yakima Basin between 1999 and 2006. Field methods included backpack electrofishing and snorkeling in tributaries, and drift-boat electrofishing in the main stem. We used three sequential steps in our evaluation: First, we determined if spatial overlap occurred between supplementation fish and non-target taxa. Second, if overlap occurred, we determined if a change in abundance, size, or biomass occurred during supplementation. Lastly, if a change occurred we determined if the change could be reasonably attributed to supplementation. Spatial overlap and changes in abundance, size, or biomass were determined to be significant if they exceeded containment objectives. Salmon rarely overlapped cutthroat and bull trout in tributaries, but some overlap of cutthroat occurred in relatively high elevations of the main stem, and considerable overlap with rainbow trout occurred in tributaries and the main stem. Salmon overlapped mountain whitefish and sucker species in the main stem, and dace and sculpin species in tributaries. With the exception of steelhead and mountain sucker, the lower 90% confidence limit of abundance, size, and biomass was above the containment objective for non-target taxa that overlapped significantly with salmon. We used rainbow trout as an analog for steelhead, and subadult suckers as an analog for mountain sucker. The lower 90% confidence limit of rainbow trout size in tributaries, and size and biomass in the main stem, and the lower 90% confidence limit for mountain sucker abundance were below our containment objectives. Comparisons of rainbow trout size in tributaries, size and biomass in the main stem, and mountain sucker abundance in main stem sections with relatively high and low salmon abundance revealed that these change were unlikely to be the result of supplementation (BACIP $P > 0.05$). Our data indicate that early stages of salmon supplementation have not impacted most valued species in the upper Yakima Basin beyond predetermined containment objectives, but the containment objective for steelhead abundance in the Teanaway Watershed has been exceeded (see Chapter 3).

Introduction

Despite the long history of stocking hatchery salmon into streams, few large-scale evaluations of impacts to the status of taxa that are not the target of enhancement (non-target taxa; NTT) have been conducted that apply to current stocking practices. Many mechanisms of impacts have been documented (Marnell 1986; McMichael et al. 1999; Hawkins and Tipping 1999), but impacts to NTT growth and abundance have generally not been conclusively demonstrated at scales larger than experimental reaches (Fresh 1997; Weber and Fausch 2003). Most large-scale evaluations of hatchery and wild fish interactions have addressed impacts to naturally produced conspecifics (Nickelson et al. 1986; Chilcote 2003; Nickelson 2003) and/or stocking salmon before the smolt stage (Bjornn 1978; Tripp and McCart 1983). Although these studies are illuminating, most contemporary hatchery salmon programs release smolts, and these smolts are released into areas containing species of concern other than wild conspecifics. Releasing smolts is appealing because they provide the highest adult returns and potentially minimize ecological interactions in the freshwater environment. In order to assess risks of contemporary programs, information about the impacts of smolt releases is needed. This is particularly true in watersheds containing threatened or endangered NTT, such as bull trout *Salvelinus confluentus* and steelhead trout *Oncorhynchus mykiss*.

Although it is assumed that releasing smolts poses less ecological risk than stocking earlier life stages, this assumption has not been tested and the practice is clearly not without risk. It is believed that smolts pose lower risks than earlier life stages because they spend less time interacting with NTT. However, hatchery smolts can interact with wild fish during downstream migration and during periods when they residualize in rearing environments. In addition, some hatchery-released yearlings swim upstream of release locations into areas containing listed species (McMichael and Pearsons 2001). Ecological interactions that can occur during migration include competition, predation, behavioral anomalies, and pathogenic interactions (Pearsons and Hopley 1999). If competition occurs, it may be of short duration because hatchery smolts generally move downstream and feed as they migrate or during brief “resting” periods. It is during these “resting” periods that competition may be most intense (Coutant and Whitney 2006).

Chinook *O. tshawytscha* and coho salmon *O. kisutch* are the most commonly cultured Pacific salmon released as yearlings and are the species of consideration in this paper. Studies have demonstrated the potential for stocked salmon to impact wild target and NTT. Hatchery spring Chinook smolts were observed to behaviorally dominate wild smolts and secure the most food and best habitat in laboratory experiments (Pearsons and Ham 2001). Predation by Chinook and coho salmon smolts on naturally produced salmon has also been demonstrated (Sholes and Hallock 1979; Hawkins and Tipping 1999). The release of large numbers of hatchery smolts can change the functional and numerical response of predators to mixed groups of hatchery and wild fish (Peterman and Gatto 1978; Wood 1987; Collis et al. 1995). Depending upon the predator response, the releases can either benefit or harm naturally produced species. Large numbers of hatchery fish can also alter the behavior of wild fish and influence susceptibility to predators or food acquisition (Hillman and Mullan 1989; McMichael et al. 1999). Finally, hatchery fish have the potential to directly transmit pathogens or increase the

susceptibility to pathogens by wild fish (Goede 1986; Bucke 1993; McVicar 1997). Similar interactions can occur if “smolts” residualize, although the intensity or manifestation of the interaction may differ. For example, competition is likely to be more potent locally when fish residualize because they remain in an area, as opposed to more temporal occupation of areas during downstream migration. Increased natural production of the target taxa translates into potentially increased interactions in the freshwater rearing area (Pearsons 2002).

Naturally produced offspring of hatchery coho and Chinook salmon have the potential to impact trout and other species of concern. Coho salmon dominate cutthroat trout *O. clarki* in pool habitats and many cutthroat trout are displaced to riffle habitats in the presence of coho salmon, particularly at warmer temperatures (Glova 1984; 1986; 1987). Sabo and Pauley (1997) suggested that size was equally important as species identity in determining competitive dominance among stream-dwelling cutthroat trout and coho salmon. Coho salmon displace some steelhead trout from pools or alter habitat use within pools that they would occupy in the absence of coho (Hartman 1965; Allee 1974; 1981). Growth of steelhead was lower in experiments with high coho densities than low ones (Fraser 1969). Despite overlap in several key habitat and food variables (Dolloff and Reeves 1990; Nakano and Kaeriyama 1995), Dolly Varden *Salvelinus malma*, who are ecologically similar to bull trout, generally occupy microhabitats close to the bottom whereas coho occupy the water column (Bugert et al. 1991; Nakano and Kaeriyama 1995).

Growth and abundance of rainbow trout in high elevation streams of the upper Yakima Basin were not affected when salmonid densities were doubled with naturally produced Chinook salmon parr (McMichael and Pearsons 1998). These controlled experiments were conducted in small to moderate size enclosures. The current study extends the findings of McMichael and Pearsons (1998) by evaluating production scale supplementation of spring Chinook salmon throughout tributary and main stem waters. In addition, impacts from coho salmon reintroduction were also evaluated. Furthermore, this study includes evaluation of interactions from hatchery produced smolts, residuals, and naturally produced offspring of hatchery fish and extends the findings of Pearsons and Temple (2007). In that study, impacts of the early stages of supplementation (5 years) were within containment objectives for *O. mykiss*, cutthroat trout, and bull trout.

In addition to the aforementioned trout species, several non-trout fish taxa occupy areas in sympatry with both natural and supplementation origin Chinook and coho salmon in the Upper Yakima Basin. Taxa of interest in this study include mountain whitefish, dace spp. Sucker spp. and sculpin spp. These taxa have the potential to be impacted through direct and indirect interactions with salmon, however interactions with non-trout fish taxa has received little attention in the literature. The literature that has reported interactions between non-trout fish taxa and salmon has generally focused on impacts to salmon (Fritts and Pearsons 2004; Murdoch et al. 2005; Fritts and Pearsons 2006).

In this paper, we examine the impacts to trout and NTT during the early-middle stages of a spring Chinook supplementation program and the reintroduction of coho salmon in the Yakima Basin, Washington (Figure 1). It was unknown whether impacts would occur from smolts that migrated downstream of acclimation sites, residuals that distributed themselves up- and downstream of acclimation sites, and/or from an increase in the distribution and abundance of naturally produced offspring of supplemented adults. Thus, NTT that were distributed upstream of acclimation sites were not considered immune from risks because residuals might migrate upstream and overlap with NTT, such as has been observed with hatchery steelhead (McMichael

and Pearsons 2001), or an increase in distribution and abundance of naturally produced Chinook salmon could result in overlap that was not observed prior to supplementation. Certainly one of the goals of supplementation is to increase the distribution and abundance of target species.

Concerns about the possibility of hatchery fish having negative impacts on valued NTT in the Yakima Basin prompted the development and implementation of a risk containment monitoring program (Bonneville Power Administration 1996; Busack et al. 1997; Ham and Pearsons 2001). Our primary management interest was to determine if changes in the status of NTT exceeded “specified biological limits” (Regional Assessment of Supplementation Project 1992) or “containment objectives” (Ham and Pearsons 2001) relative to the baseline period in which no stocking occurred. The containment objectives are: 0% impact for bull and steelhead trout; less than 5% impact for mountain sucker *Catostomus platyrhynchus*; less than 10% impact for rainbow and cutthroat trout in the main stem; less than 40% impact for rainbow and cutthroat trout in the tributaries (Pearsons and Hopley 1999; original cutthroat trout containment objective was modified by Yakima/Klickitat Fisheries Project 2004); less than 40% impact for mountain whitefish *Prosopium williamsoni*; 90% impact for sucker species (*Catostomidae*) in the main stem; 65% and 95% for longnose dace *Rhinichthys cataractae* and speckled dace *R. osculus*, respectively, in tributaries; and 90% impact for sculpin species (*Cottidae*) in tributaries. These containment objectives were developed based on the status (e.g., abundance) of the NTT, their value to fisheries, and their relative value compared to the expected benefits of supplementation. Containment objectives for NTT are evaluated relative to baseline abundance, size, biomass, and distribution. If containment objectives are exceeded for any one of these population parameters, then further action is warranted.

We believe that it is important to report results within the context of the life-span of a supplementation program. An early stage of supplementation, such as the “broodstock” stage, is less likely to produce impacts than a later stage such as the late “building” stage (Pearsons 2002). However, waiting to report results until a supplementation program has matured can delay the presentation of important information for up to 30 years, and therefore limit information that could be used to help improve current management decisions (Pearsons 2002). Indeed, many scientific reviews about hatchery and wild fish interactions identify the paucity of information that is available to evaluate these risks (Fresh 1997; Weber and Fausch 2003). Therefore, we present the findings of the early to middle stages of supplementation knowing that impacts could change during later stages.

We acknowledge that impacts in areas of overlap cannot be definitively evaluated without adequate controls. Unfortunately, spatial controls for most large river systems are difficult to find and we could not find an adequate spatial control for trout in the upper Yakima River. However, achievement of management objectives can be evaluated relative to fixed standards, such as containment objectives, and performance of NTT relative to containment objectives can be used to prioritize impact evaluations. For example, if the status of an NTT has not changed after stocking, then it would not be a high priority to evaluate farther. In contrast, if the status of an NTT decreases below a certain level then further evaluation should be initiated. We believe that the containment objectives are suitable triggers for prioritizing evaluation. Furthermore, if environmental conditions do not change appreciably before and during supplementation, then impacts can be evaluated using temporal controls. The results presented in this paper are the early stages of risk containment monitoring associated with the Yakima/Klickitat Fisheries Project (YKFP).

Methods

Study Area

The Yakima River Basin is a large river system that drains into the Columbia River near Richland, Washington. The upper Yakima River Basin, which is the subject of this paper, is located upstream of Roza Dam (Figure 1). Historically large numbers of salmon and steelhead returned to the upper Yakima Basin (Bonneville Power Administration 1996). Coho salmon were extirpated by the early 1980s and spring Chinook salmon have been severely depressed. Steelhead (mid-Columbia Evolutionarily Significant Unit) and bull trout are currently listed as threatened (National Marine Fisheries Service 1999; U.S. Fish and Wildlife Service 1998). Rainbow trout in the Yakima River provide one of the best wild trout fisheries in Washington (Krause 1991; Probasco 1994) and westslope cutthroat trout are present in many high elevation tributaries. Mountain whitefish and sucker species are widely distributed in main stem areas, although mountain suckers are rarely observed. Sculpin species are widely distributed in tributary areas, while speckled dace inhabit low elevations and longnose dace inhabit mid-to high elevation areas of tributary streams.

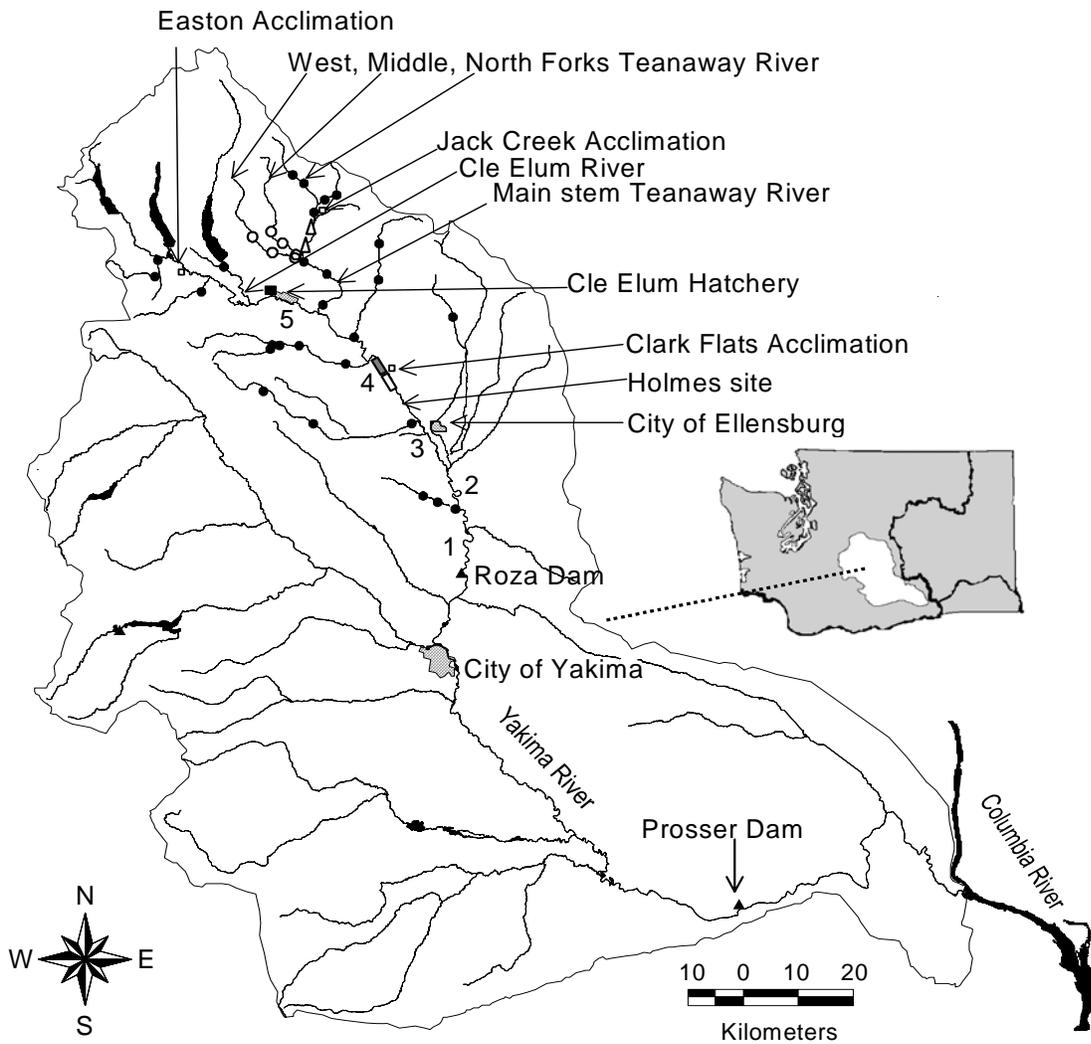


Figure 1. Map of the Yakima River Basin. Darkened circles are tributary NTT distribution sampling sites. Teanaway Basin tributary BACIP control and treatment sites are represented by open circles and triangles, respectively. Bold numbers represent Main stem Yakima River non-target taxa (NTT) monitoring sites where 1, 2, 3, 4, and 5 represent the LCYN, UCYN, EBURG, THORP, and CELUM sections, respectively. Main stem BACIP control and treatment sites are represented by shaded and open rectangles, respectively.

The main stem and tributaries of the Yakima River differ in their physical properties and the way that they are managed. The flows in the main stem are regulated by three dams. Peak flows during the spring have been truncated and flows are artificially high during the summer. The main stem is heavily fished for trout and a catch-and-release regulation has been in effect since 1990. Tributaries have less flow regulation, but lower portions of tributaries can experience low flows from irrigation withdrawals. Anglers are allowed to retain two trout greater than 203 mm in tributaries.

Chinook and coho salmon hatchery programs

Spring Chinook and coho salmon yearlings were released into the upper Yakima Basin for the first time during spring 1999 as part of the YKFP. The goal for both of these species is to increase natural production and to provide harvest opportunity using artificial propagation while keeping adverse impacts within specified biological limits (Bonneville Power Administration 1996; Fast and Craig 1997; Bosch 2004). Approximately one million Chinook salmon smolts have been released annually in the upper Yakima River from 1999 to 2006 (Table 1). Broodstock for the spring Chinook program were natural origin upper Yakima stock collected at a trapping facility at Roza Dam. Chinook were spawned and juveniles were reared at a hatchery facility in Cle Elum, Washington (Knudsen et al. 2006). Spring Chinook salmon yearlings were transported to acclimation sites during January and February and volitionally released into the Yakima River from the Easton and Clark Flats acclimation sites, and from the Jack Creek acclimation on the North Fork of the Teanaway River (Figure 1). Fish were permitted to volitionally migrate between March 15 and May 31 and averaged 120 mm fork length (FL) when released. At the end of May all fish were forced out of the acclimation sites into the river. Mean travel time of migrants from acclimation sites to a detector near Prosser Washington (river kilometer [RKM] 75.6 measured from the confluence with the Columbia River) is about 24.3 days. However, a large number of Chinook salmon residualize in the upper Yakima Basin (Larson et al. 2004; Pearsons et al. 2005).

Table 1. Numbers and location of yearling spring Chinook salmon released in the upper Yakima River annually.

Brood Year	Release Year	Clark Flats	Easton	Jack Creek	Total
1997	1999	229,290	156,758		386,048
1998	2000	221,460	230,860	137,363	589,683
1999	2001	232,563	269,502	256,724	758,789
2000	2002	285,954	263,061	285,270	834,285
2001	2003	80,782	39,106	250,348	370,236
2002	2004	284,151	261,234	274,466	819,851
2003	2005	265,242	261,239	278,508	804,989
2004	2006	271,817	256,929	219,110	747,856

Broodstock for the coho salmon program were largely from lower Columbia hatchery sources that have been under culture for many generations. Coho salmon were volitionally released into the Yakima River from sites near the Cle Elum Hatchery (hatchery slough 1999, 2000, and 2001), from the Easton spring Chinook acclimation site (1999-2003), from the Holmes site (2003-2006) approximately 8.2 rkm upstream from from the city of Ellensburg, Washington, from the Boone Pond acclimation site (2004-2006) near Yakima River kilometer 290.5, and into the North Fork of the Teanaway River from the Jack Creek spring Chinook acclimation site (1999; Table 2). An additional 3,937, 10,504, and 10,255 coho fry were planted in the Hanson Ponds near Cle Elum, WA, the Boone Pond, and at the Holmes Site, respectively, during July of 2004 (Table 2). Mean travel time of migrants from the Holmes site to a detector near Prosser Washington in 2003 (RKM 75.6) was 41.9 days. Very few coho salmon have been observed to residualize in the upper Yakima River. Coho salmon sizes were not recorded at the time of release, however, two to three months after release, hatchery origin coho smolt fork lengths averaged 157 mm in the migration corridor. Prior to 1999, coho salmon were released in the middle portion of the Yakima River, a considerable distance below Roza Dam. Coho salmon observed in the upper Yakima River prior to 1999 were likely the result of downstream releases. More detail about the study area and background of the supplementation project has been previously described (Busack et al. 1997; Pearsons and Hopley 1999; Ham and Pearsons 2000). Early findings indicate that Chinook and coho hatchery programs are increasing the abundance and distribution of spawners in the upper Yakima Basin (Bosch 2004; Bosch et al. 2007).

Table 2. Numbers and location of yearling coho salmon released in the upper Yakima River annually.

Brood Year	Release Year	Easton	Jack Creek	Hatchery Slough	Boone Pond	Holmes	Fry Plants	Total
1997	1999	48,000	240,000	210,000				498,000
1998	2000	247,153		247,523				494,676
1999	2001	233,076		233,388				466,464
2000	2002	314,450						314,450
2001	2003	228,000				100,000		328,000
2002	2004				139,000	264,369	24,696	428,065
2003	2005				52,500	261,207		332,000
2004	2006					288,127		

General approach

We used three sequential steps in our evaluation of impacts to trout and NTT. First we determined if overlap occurred. Second, if overlap occurred we determined if a temporal change in abundance, size, or biomass occurred after supplementation. Finally, if a change occurred, we determined if the change could be reasonably attributed to supplementation (Table 2). Increases in distribution of the target species can result in spatial overlap with trout resulting in the potential for impacts. If overlap is less than or equal to containment objectives, then impacts are assumed to be acceptable. However, if overlap exceeds containment objectives, then changes to abundance, size, and biomass were evaluated. Change was evaluated by comparing the abundance, size, and biomass of trout and abundance and size structure of other NTT before and after salmon were released into the upper Yakima Basin (1999). A change in abundance, size, or biomass does not demonstrate that the hatchery caused the impact. Changes in abundance, size, or biomass can be used to trigger further investigation to identify the causes of changes in monitoring variables. We used a conservative approach to trigger further investigation because of the high inter-annual variability of population parameters (Ham and Pearsons 2000). We used the lower 90% confidence limit (CL) of the response variables (abundance, size, and biomass) as the trigger to initiate further investigation. This ensured that there was a 95% probability that the true value was above the lower CL. If the lower 90% CL was lower than the containment objective, then we attempted to determine causation. Confidence limits associated with parameter estimates were computed based on a *t*-statistic with *n*-1 degrees of freedom (Sokal and Rohlf 1981).

Table 2. Field sampling location (Loc.) and abundance estimation methods used for the following tributary (Trib) and main stem (Main) species (Spp.); bull trout (B), cutthroat trout (C), rainbow trout (R), dace species (D), sucker species (K), sculpin species (P), mountain whitefish (W), naturally produced spring Chinook and coho salmon (S, O), and hatchery origin spring Chinook salmon (H). Additional abbreviations are as follows: Snork = Snorkeling; BP Efish = backpack electrofishing; DB Efish = drift boat electrofishing; Surrogate Spp. Eff. Exp. = site and size specific surrogate species efficiency expansions; NA = not applicable, MR = mark-recapture; WMI = weighted mean generated from index sites; ORCO = overlap judged relative to containment objectives; 90%CLCO-*t* = 90% confidence limit judged relative to containment objectives; *t* = two-sample student's *t*-test; BACIP = before/after control/impact paired test.

Spp.	Loc.	Field Method	Spp. Effic. Derived from	Estimation Method	Analysis
Overlap					
B	Trib	Snork	NA	% Overlap	ORCO
C, R, D, K, P	Trib	BP Efish	NA	% Overlap	ORCO
S, H, O	Trib	Snork / BP Efish	NA	% Overlap	ORCO
R, C, S, H, O	Main	DB Efish	NA	% Overlap	ORCO
Before / After					
R, S, H, O	Trib	BP Efish	R, S, H, S	Removal Based WMI	90%CLCO- <i>t</i>
R, C	Main	DB Efish	R	MR Based WMI	90%CLCO- <i>t</i>
S, H, C	Main	DB Efish	R	Surrogate Spp. Eff. Exp.	90%CLCO- <i>t</i>
M	Main	DB Efish	-	Visual count	90%CLCO- <i>t</i>
K	Main	DB Efish	-	Visual count	90%CLCO- <i>t</i>
Causation					
R	Trib	BP Efish	R	Treatment / Control	BACIP
R	Main	DB Efish	R	Treatment / Reference	BACIP
K	Main	DB Efish	-	Treatment / Reference	BACIP

To determine causation we compared abundance, size, and biomass in control and treatment sites (e.g. tributaries). Where this was not possible (e.g., main stem), we compared abundance, size, and biomass of NTT upstream and downstream of a hatchery acclimation facility. We reasoned that the magnitude of interactions between released salmon and NTT would invariably be larger downstream of a release site.

We used abundance and size of age 1 rainbow trout in the main stem Yakima River and all ages of rainbow trout in the tributaries as an analog for evaluating impacts to steelhead trout. We did this because of the difficulty of monitoring impacts to steelhead in the upper Yakima Basin and the similarities in genetics and pre-smolt ecologies of the two forms of *O. mykiss* (Pearsons et al. 2007). Difficulties of monitoring included; low abundance of steelhead, distinguishing juvenile rainbow and steelhead non-lethally, and sampling upper Yakima steelhead smolts. The spawning populations of rainbow and steelhead trout overlap considerably in space and time, evidence of gene flow has been documented (Pearsons et al. 2007), and it is

believed that the juveniles rear together and share similar ecologies prior to smoltification. Smoltification in the Yakima Basin generally occurs between ages 1 and 3. Using rainbow trout as an analog does not address impacts that would occur during or after the smolt stage.

Similarly, we used juvenile sized suckers in the main stem Yakima River as an analog for evaluating changes in status of mountain suckers. Mountain suckers are rare in the basin and very few have been observed during field sampling. The low abundance and low containment objective for mountain sucker makes detecting impacts to their status difficult. The use of surrogate species (all subadult suckers) greatly improves our ability to detect impacts, although we must assume mountain suckers and surrogate species respond to supplementation activities similarly. We used subadult suckers because mountain suckers are considerably smaller than the other dominant sucker species, bridgeline and largescale sucker (Wydoski and Whitney 2003).

Environmental variables were compared before and during supplementation to determine if key environmental factors changed between the two periods. In the main stem Yakima River, water temperature and flow were recorded daily at U.S. Bureau of Reclamation (USBOR) gauging stations located throughout the basin (available at <http://www.usbr.gov/pn/hydromet/yakima/yakwebaread.html>). We evaluated stream flow and temperature recorded for the months October through September at three gauging stations on the Yakima River including one near the city of Cle Elum (YUMW), one near the city of Ellensburg (ELNW), and one in the Yakima River canyon (UMTW) approximately 22 rkm downstream from the city of Ellensburg, Washington. We calculated mean, maximum and minimum daily flows and temperature for each station annually. We recorded wetted stream width, during the time of trout sampling, at 100 m intervals within our main stem sites and used the standard deviation of the widths as an index of stream morphology. We used two-sample student's t-tests to determine if there were differences in these variables before (1990-1998) and during (1999-2006) the supplementation period.

In upper Yakima Basin tributaries, we measured stream wetted width and stream flow at the time of sampling and computed total stream discharge as described by Gallagher and Stevenson (1999). We also generated a longitudinal streambed profile in each site by recording water depth at 1 m intervals along the stream thalweg as described by Kaufmann (1987). Variability of thalweg measurements, calculated as the standard deviation (SD) of thalweg depths, were calculated to index habitat complexity and residual pool volume. Thalweg measurements were discontinued in 2005, so before-after comparisons were limited to the period 1993-2004. Wetted width and stream flow measurements in tributaries during the years 1990-1992 were not the same as the period 1993-2006 so we excluded this time period in our analysis. To describe mean, minimum and maximum daily tributary stream flow that was of a similar nature as the main stem dataset, we compiled flow data collected at the USBOR Teanaway River (TNAW) gauging station. Direct measures of water temperature were not available for both time periods at this gauging station so we used daily air temperatures to generate an index of tributary water temperature from a SNOTEL gauging station located in the general vicinity of the Yakima River headwaters (Stampede Pass, Washington). The SNOTEL gauging station was operated by the United States Department of Agriculture's Natural Resources Conservation Service both before and during supplementation (available at <http://www.wcc.nrcs.usda.gov/snow>). In tributaries, daily discharge and temperature was evaluated for the year prior to field sampling (e.g., August through July). Finally, we used two-sample student's t-tests to determine if there were differences in the tributary environmental variables between the baseline and supplementation period.

Field Methods

The spatial overlap between NTT and spring Chinook and coho salmon during supplementation was quantified as the mean annual percent of the NTT distribution in sympatry with salmon (Table 2). Spatial overlap was determined annually at the sites indicated in Figure 1 and Table 3. These sites were used because they consistently had NTT in the years prior to supplementation. Annual NTT distribution was calculated from the sum of the rkm that NTT were present. The annual percent overlap was calculated as the number of rkm used by sympatric salmon divided by the number of rkm used by NTT. The mean percent overlap was the mean of the annual overlaps from 1999-2006 and was compared to the containment objectives for NTT.

The spatial overlap between bull trout and supplemented salmon in the North Fork of the Teanaway River was inventoried by snorkeling and electrofishing. Bull trout that had the potential to overlap with salmon in the upper Yakima Basin were primarily located in the North Fork of the Teanaway River. The distribution of bull trout was determined by snorkeling and electrofishing conducted throughout the North Fork of the Teanaway River prior to 1999 (WDFW, unpublished data). The entire 9.3 km rearing area of bull trout (1999) or a systematic sample of nine 1 km reaches (2000-2006) were snorkeled at night (Thurow et al. 2006) to determine if any salmon were present. The nine index sites sampled during 2000-2006 were 200 m long and were generally situated at 800 m intervals up the stream channel. During September, two divers with underwater lights, moved upstream and counted all fish observed. Additional snorkel surveys were conducted during the spring and summer to determine the maximum upstream distribution of spring Chinook and the presence of other species (see residuals methods). We also supplemented our snorkeling surveys with backpack electrofishing described below.

Abundance Indices

The presence, and population abundance indices of salmonids in upper Yakima tributary sites were generated from single-pass, backpack electrofishing capture efficiency expansions (Temple and Pearsons 2004; Temple and Pearsons 2007; Table 2). In tributary streams, a crew of three to six people electrofished 200 m long index sites during the day with a backpack electrofisher during summer base flows (Table 3). A single upstream electrofishing pass was performed and attempts were made to net all visible fish. Netted fish were held in perforated buckets in the stream margins until they were processed. All fish were anesthetized, identified to species, and the lengths and weights of salmonids were recorded. We assumed, and later verified, that electrofishing efficiencies would be poor for small fish and restricted our evaluation to salmonids greater than 79 mm FL (Reynolds 1983).

An index of salmonid abundance was calculated by expanding the first pass count by a median capture ratio established for each site during the baseline monitoring phase. Median capture ratios were calculated by dividing the number of fish captured on the first electrofishing pass by a multiple-removal maximum likelihood estimate of the number of fish in the site (Zippen 1958). A minimum of two and a maximum of six electrofishing removal passes were used to generate capture efficiencies at each site once annually during the baseline phase. Median efficiencies were based on between four and eight annual baseline replicates depending on the year each site was established. On average, removal based efficiencies indicated that 75%

of the trout greater than 79 mm FL were captured during the first electrofishing pass. Approximately 49% of the sites were sampled with multiple removal methods during both periods because insufficient numbers of annual capture efficiency estimates during the baseline period were available to generate expansions.

Table 3. Distribution monitoring site locations in the upper Yakima Basin. A sub-set of distribution sites are used for rainbow trout (R) or cutthroat trout (C) abundance and size before and after supplementation (B-A) comparisons. Before-After-Control-Impact-Paired (BACIP) abundance and size monitoring control (1) and impact (2) site designations are presented as well. Latitude and longitude positions are presented in degrees, minutes, and seconds (D°M'S"). Main stem Yakima River sampling site locations represent the downstream boundary of each site.

Stream / Site	B-A	BACIP	Years	Latitude	Longitude
Yakima River Tributaries					
Cabin Creek 1			90-06	47° 14' 08.72"	121° 13' 8.72"
Domerie Creek A			97-03	47° 14' 12.73"	121° 04' 6.83"
Jungle Creek A			00-06	47° 20' 47.43"	120° 52' 36.08"
Manastash Creek 1			92-06	46° 59' 39.45"	120° 35' 26.81"
Manastash Creek 3			92-06	47° 2' 21.09"	120° 57' 36.41"
Manastash Creek A			98-06	46° 59' 30.35"	120° 50' 57.30"
M.F. Teanaway 1	R	1	90-06	47° 15' 53.54"	120° 53' 53.19"
M.F. Teanaway 2	R	1	90-06	47° 16' 51.06"	120° 55' 50.37"
M.F. Teanaway 3	R	1	90-06	47° 17' 57.47"	120° 57' 42.06"
M.S. Teanaway 1			94-06	47° 10' 58.40"	120° 49' 29.80"
M.S. Teanaway 2			94-06	47° 13' 28.32"	120° 48' 15.61"
M.S. Teanaway 3			94-06	47° 15' 6.65"	120° 52' 27.53"
N.F. Teanaway 1	R	2	90-06	47° 16' 53.10"	120° 51' 53.86"
N.F. Teanaway 1.5			01-06	47° 17' 24.67"	120° 51' 35.38"
N.F. Teanaway 2	R	2	90-06	47° 18' 41.97"	120° 51' 31.40"
N.F. Teanaway 2.5			99-06	47° 19' 36.74"	120° 51' 21.15"
N.F. Teanaway 2.75			02-06	47° 19' 56.28"	120° 51' 22.71"
N.F. Teanaway 3	R		90-06	47° 24' 18.24"	120° 55' 56.68"
N.F. Teanaway A			97-04	47° 22' 51.58"	120° 53' 11.52"
N.F. Teanaway B			98-04	47° 24' 54.67"	120° 56' 20.50"
^a N.F. Teanaway			97-02	47° 24' 60.24"	120° 52' 48.68"
Rkm 18.2-27.2					
Stafford Creek A			97-06	47° 21' 20.08"	120° 50' 0.84"
Stafford Creek B			97-04	47° 21' 48.82"	120° 48' 32.18"
Swauk Creek 1	R		92-06	47° 7' 58.30"	120° 44' 51.39"
Swauk Creek 2	R		92-06	47° 13' 45.90"	120° 41' 46.96"
Swauk Creek 3	R		92-06	47° 19' 15.08"	120° 41' 9.65"
Taneum Creek 1	R		90-06	47° 5' 7.71"	120° 46' 8.35"
Taneum Creek 2	R		90-06	47° 6' 46.99"	120° 52' 58.95"
Taneum Creek 3	R		90-06	47° 6' 37.20"	120° 56' 9.09"
Taneum Creek A			97-06	47° 6' 43.34"	120° 55' 45.11"

Taneum Creek B			97-06	47° 6' 30.69"	120° 56' 11.71"
Umtanum Creek 1			92-06	46° 51' 27.63"	120° 29' 49.55"
Umtanum Creek 1.5			92-04	46° 51' 57.13"	120° 32' 4.26"
Umtanum Creek 2			92-04	46° 52' 27.65"	120° 33' 58.07"
W.F. Teanaway 1	R	1	90-06	47° 15' 25.52"	120° 53' 56.00"
W.F. Teanaway 2	R	1	90-06	47° 15' 51.79"	120° 57' 11.25"
W.F. Teanaway 3	R	1	90-06	47° 16' 11.37"	120° 58' 36.13"
Wilson Creek A			97-03	47° 9' 54.93"	120° 30' 38.79"

Mainstem Yakima River

Lower Canyon	R		91-06	46° 47' 32.32"	120° 27' 23.94"
Upper Canyon	R		91-06	46° 53' 42.55"	120° 30' 10.93"
Ellensburg	R		91-06	46° 58' 47.39"	120° 34' 9.24"
Thorp	R / C	1 / 2	91-06	47° 5' 58.73"	120° 42' 8.48"
Cle Elum	R / C		91-06	47° 10' 24.96"	120° 51' 36.48"

^aLatitude and longitude position refers to the lowest reach boundary at river kilometer (RKM) 18.2 (measured from the confluence with the Yakima River).

In the main stem of the upper Yakima River, a crew of two people electrofished 4.2–7.4 km long index sites at night with a driftboat mounted electrofisher as described by Temple and Pearsons (2007). During the electrofishing passes, all fish were identified visually and attempts were made to net all trout. Trout greater than 99 mm fork length were marked with a fin clip and released. One week later, another electrofishing pass was made to determine the proportion of marked and unmarked salmonids. Three types of abundance measures were made based on the abundance of the non-target taxa (Table 2). One type, which was used for rainbow trout, was generated from mark-recapture methods. Main stem rainbow trout were grouped into 25 mm size intervals and mark-recapture abundance estimates for each 25 mm group were generated using the maximum log-likelihood estimator as computed by the computer software program Mark-Recapture for Windows (Mark-Recapture for Windows 1997, Version 5.0 Beta, Montana Fish, Wildlife, and Parks). The general form of the estimator was:

$$\hat{N} = \frac{M}{effic}$$

where \hat{N} was the estimated abundance for each 25 mm size class, M was the number fish marked, and $effic$ was the log-likelihood model estimated capture efficiency. The sum of the abundance estimates generated for each 25 mm grouping represented total abundance per index section. The assumptions associated with the log-likelihood model included; 1) the proportions of marked and unmarked fish remained equal between mark and recapture sampling, 2) marked and unmarked fish were equally catchable, and 3) fish marks were permanent for the duration of the sampling and were not overlooked during the recapture sample. We used a one-week interval between mark and recapture sampling because; 1) it allowed adequate time to redistribute, and 2) it provided adequate time for marked fish to recover (Mesa and Schreck 1989; Peterson et al. 2004).

The abundance and size of age 1 main stem rainbow trout was calculated for each index site each year using a mixture analysis of the length frequency distributions from each individual site (MacDonald and Pitcher 1979). We used R statistical software (R development core team 2005) and the add in package mixdist (Du 2002) to determine the proportion of age 1 rainbow trout in each index site and their mean fork length. Age 1 rainbow trout abundance was calculated by multiplying the total mark-recapture abundance estimate for each index site by the estimated proportion of age 1 rainbow trout present in each index site. We weighted the age 1 rainbow trout abundance estimates for each index section by the amount of stream each index site represented and then generated a mean abundance estimate of age 1 rainbow trout per km for each year. Confidence limits (95%) for the parameter estimates were calculated by incorporating both the spatial and within site variance components (Bohlin et al. 1989).

To evaluate our age 1 rainbow trout parameter estimates in the main stem, we compared them against known main stem rainbow trout population parameters. In 2003, we systematically collected scales from rainbow trout in our main stem index sites. Scales were mounted on gummed cards in the field and acetate impressions of each scale were made in the lab. Scales were projected with a microfiche reader and were aged by counting annuli (Jearld 1983) by a recognized expert with over 25 years experience. We used a chi square test to compare the known proportion of rainbow trout in each age class in each index site versus the proportions predicted by the mixture analysis. We also used a student's t-test to compare the mean length of known age 1 main stem rainbow trout versus the mean length predicted from the mixture analysis.

Efficiency Expansions

The second type of abundance index, used for main stem cutthroat trout, was based on efficiency expansions. An abundance index for cutthroat trout was generated from the number of individual fish netted and was expanded by size specific rainbow trout recapture efficiencies (r/c). We calculated an index of cutthroat trout abundance in the THORP and CELUM main stem sections. The low abundance of cutthroat trout prevented performing valid mark-recapture estimates due to insufficient recaptures of marked fish. Thus, we calculated an abundance index of cutthroat trout (<250 mm) by expanding the observed number of cutthroat trout captured during electrofishing, by our capture efficiencies for similar sized rainbow trout (Peterson and Zhu 2004). Abundance indices were extrapolated to the reach scale based on reach length.

The next type of abundance index, used for natural origin spring Chinook, was a visual estimate that was expanded by capture efficiencies. The numbers of natural origin spring Chinook visually enumerated during the electrofishing marking runs were expanded by maximum log-likelihood model recapture efficiencies for the smallest sizes of rainbow trout observed (generally between 100 mm and 126 mm). These efficiencies may have been overestimated because naturally produced spring Chinook are slightly smaller than the rainbow trout used to establish the capture efficiencies at this time of year. Thus, the resulting abundance index is likely an underestimate. However, in sites where we could generate capture efficiencies for spring Chinook, our observed rainbow trout electrofishing capture efficiency was within the 95% confidence interval (CI) of the hatchery origin spring Chinook electrofishing efficiency. Thus, we believe that size based efficiencies are reasonable ways of indexing relative abundance because fish size is one of the most important factors that influences electrofishing efficiency (Anderson 1995; Buttiker 1992).

The last type of abundance index, used for mountain whitefish and sucker species (including mountain sucker) in the main stem, was a visual estimate based upon visual counts taken during boat electrofishing surveys. Visual counts appear to provide an adequate index for abundance. Comparisons between mainstem rainbow trout visual counts correlated significantly with rainbow trout abundance estimates generated from mark-recapture electrofishing. Similarly, visual counts of spring Chinook fall parr correlated significantly with smolt counts at Prosser dam the following year. This information was used as support for the use of visual counts to index abundance for mountain whitefish and sucker species.

Five index sites in the main stem Yakima River were used to represent five contiguous study reaches and the index sites comprised approximately 28% of the upper Yakima River between Roza Dam and the Cle Elum River confluence. Each index site was selected to be representative of a larger reach. Selection criteria included river constraint and elevation. Estimates were generated from abundance and size data collected in the site and were extrapolated to the reach scale based on reach length. The Lower Canyon (LCYN) site was 4.8 km long, the Upper Canyon (UCYN) site was 5.2 km long, the Ellensburg (EBURG) site was 4.2 km long, the Thorp (THORP) site was 5.7 km long, and the Cle Elum (CELUM) site was 7.4 km long. The reach descriptions are as follows: LCYN extends 19.2 km upstream from Roza Dam to Umtanum Creek; UCYN extends 13.4 km upstream from Umtanum Creek to the Ringer Road access; EBURG extends 21.2 km upstream from the Ringer Road access to the Ellensburg Dam; THORP extends 24.1 km upstream from the Ellensburg Dam to the Teanaway River; and CELUM extends 16.2 km upstream from the Teanaway River to the Cle Elum River.

Size of an NTT was quantified using the mean length of fish collected in our main stem and tributary sites. All rainbow trout that were captured were measured to the nearest mm FL. Mean length of rainbow trout in each tributary site was calculated and then the mean of the site means was calculated for each year. Length of age 1 main stem rainbow trout was calculated for each index site each year using a mixture analysis of the length frequency distributions (MacDonald and Pitcher 1979) from each individual site and then weighted by abundance and section length. Biomass estimates were generated from the product of mean estimated abundance and mean weight. Mean weight of rainbow trout in each tributary site was averaged and then the mean of the averages was calculated for each year. The mean annual weight of age 1 main stem rainbow trout was computed for each index site using the log-transformed length/weight regression from mean lengths of fish in individual annual index sites and then weighted by abundance and section length.

Size structure for mountain whitefish, sucker species, and mountain suckers were based upon visual counts of fish in each size class. Since fish were generally not handled while visual counts were performed, an index of size was based upon the proportions of size classes of fish observed during electrofishing. For mountain whitefish, we used the proportion of subadults (≤ 250 mm TL) relative to the total whitefish observed for evaluating changes to size. For sucker species, we used the proportion of adults (≥ 250 mm TL) relative to the total suckers observed during electrofishing. For our mountain sucker size index, we used the proportion of subadult suckers (≤ 250 mm TL) relative to the total suckers observed during electrofishing. Proportions of fish for the size groupings were arcsine square-root transformed prior to analysis. Estimates of biomass for visually observed species could not be accurately calculated using the data we collected.

Residualization of Hatchery Smolts

Abundance estimates and the presence of residualized hatchery spring Chinook and coho salmon present in the Yakima River from mid September to mid October for release years 1999-2006 were calculated using observed abundance and boat electrofishing recapture efficiencies calculated for similar sized rainbow trout. We calculated recapture efficiencies of similar sized rainbow trout utilizing mark-recapture methods in main stem Yakima River electrofishing index sites. The rainbow trout recapture efficiencies were applied to the number of hatchery residuals netted during the mark runs in each index section. A final estimate of hatchery residual abundance was expanded to the reach scale based on reach length.

Distribution and relative abundance of residuals was also estimated by snorkeling the North Fork Teanaway and main stem reaches. The farthest upstream presence of residuals was estimated in the North Fork Teanaway by snorkeling upstream of the acclimation site until no residuals were observed in three sequential pools. Snorkel surveys occurred during the time of maximum distribution from June through September. Snorkeling observations to determine presence or absence were also supplemented by electrofishing.

Natural Production

Spring Chinook salmon natural production occurs primarily in the main stem Yakima River upstream from the city of Ellensburg, Washington (Figure 1). The most intense spawning activity has been observed upstream from the Cle Elum hatchery facility, and in the general vicinity of the Easton acclimation facility (Figure 1). Some spawning activity has also been observed in the Cle Elum and main stem Teanaway Rivers although redd counts indicate these tributaries generally contribute a relatively small proportion of the total redds counted in the upper Yakima River Basin annually.

Abundance indices (fish/km) and 95% CL of naturally produced spring Chinook and coho salmon were generated for the main stem Yakima River and tributary index monitoring sites for the baseline (1990-1998) and during supplementation periods (1999-2006). Spring Chinook and coho salmon visually enumerated during boat electrofishing were expanded by rainbow trout electrofishing efficiencies as previously described. In the tributary index monitoring sites, spring Chinook abundance indices were generated from multiple removal capture efficiencies established during the baseline phase. Due to the low abundance of juvenile coho salmon and their similar size with Chinook salmon, we applied spring Chinook salmon capture efficiencies to the number of coho salmon captured in the first electrofishing pass to obtain an index of coho abundance. We used a two-sample student's t-test to compare differences in the abundance of salmon before and during the supplementation period.

Before-After and Causation Analysis

Changes in NTT abundance, size, and biomass after stocking began were expressed as percent changes prior to stocking and were judged to warrant further attention if the percent change in before-after samples decreased beyond the containment objectives. We did not confine ourselves to have statistically significant results to trigger further attention because of the chances of committing a type II statistical error from a test with low power (Ham and Pearsons

2000). Impacts to NTT are difficult to detect because of high interannual variation of response variables and the low number of annual surveys available to isolate the impacts that occur during the initial stages of supplementation (Ham and Pearsons 2000; Ham and Pearsons 2001; Pearsons 2002). For example, prospective power analyses indicated that abundance impacts of <19% were not statistically detectable after 5 annual surveys (Ham and Pearsons 2000). Based on these constraints, only large impacts will be statistically detectable. However, one-tailed student's t-tests were used to determine if the results were statistically significant and 90% CL were calculated to aid with interpreting the magnitude of the results and triggering the next phase of evaluation.

We used an analysis of co-variance (ANCOVA) to test the hypothesis that the log length-log weight relationships of main stem age 1 rainbow trout in the main stem, and rainbow trout in tributaries did not differ between the baseline (1990-1998) and supplementation (1999-2006) periods. For main stem rainbow trout, a homogeneity of slopes test was performed first to determine if the slopes were significantly different. The log length-log weight relationships for tributary rainbow trout before versus during the supplementation period did not meet the assumptions of equal slopes for the ANCOVA, so we tested the relationship using a similar test (separate slopes model) that does not require this assumption.

Although age 1 rainbow trout are used for evaluating impacts to rainbow trout and serve as an analog for steelhead trout in the main stem, we also evaluated changes in the response variables of catchable sized main stem rainbow trout. Since large trout are valued as a fishery resource, we calculated mean size, abundance, and biomass of all main stem rainbow trout greater than 249 mm FL. The field methods and calculations for these larger fish were the same as was described for age 1 rainbow trout in the main stem.

When NTT exhibited decreases in abundance, size, or biomass beyond the lower 90% CL, we attempted to determine if the change was caused by supplementation. In the Teanaway Basin and in the main stem near Thorp, we used a before-after-control-impact-paired site analysis (BACIP; Stewart-Oaten et al. 1986) to evaluate changes to the abundance, size, or biomass of NTT. This test evaluates the annual paired differences in control and impact sites before and after stocking. Paired differences before and after stocking were compared with a two-sample student's t-test or ANOVA. We used a subset of sites and field methods for the BACIP (Table 2). The treatment sites were based on their proximity to acclimation sites and the control sites were selected based on their geographic proximity to treatment sites. Two sites located downstream from the Jack Creek acclimation facility on the North Fork Teanaway River and three sites in the main stem Teanaway River were used as treatment sites and three sites on each of the Middle and West Forks of the Teanaway River were the paired controls (Figure 1). The "before" period was from 1990-1998 and the "after" period was from 1999-2006. When BACIP results were significant, we investigated relationships between core variables using simple linear regression.

The data from the Thorp main stem site was divided into control and impact sites after 1992 and the "before" and "after" periods were the same as for the Teanaway River. The area downstream from the Clark Flats acclimation site was the impact site and the area upstream from the acclimation site was the control. Abundance estimates and associated 95% CL generated for the Thorp site were partitioned by the proportions of fish marked in either the treatment or control section during the electrofishing marking runs. Rainbow trout located immediately downstream from the Clark Flats acclimation facility had the potential to interact with all upstream spring Chinook and coho smolt releases as well as residual salmon that did not migrate

to the ocean. Rainbow trout immediately upstream from the Clark Flats acclimation facility had the potential to interact with migrating smolts and residualized Chinook salmon from the other upstream release locations in the upper Yakima Basin. In essence, we tested if the fish released from the Clark Flats acclimation site caused additional impact to rainbow trout beyond what occurred from releases farther upstream. No controls were available for cutthroat trout in the main stem.

Results

General approach

Comparisons of environmental variables measured in main stem and tributary areas before and during supplementation indicated that there were no significant differences in the variables evaluated (Table 4).

Table 4. Environmental variables and results from two-sample student's t-tests of before versus during supplementation periods in the Yakima Basin. Main stem variables were based on water year (October through September of the following year as recorded at three gauging stations) and tributary mean, minimum (Min.), and maximum (Max.) discharge and mean air temperature (based on August through July of the following year) recorded at a single tributary gauging station or weather station. Main stem standard deviation of stream width was recorded at the time of sampling. Tributary mean summer discharge, mean wetted width, mean thalweg depth, and standard deviation of thalweg depth were recorded at the time of sampling. Degrees of freedom (df) were 15 for all comparisons unless otherwise noted.

Environmental Variable	Before Mean	During Mean	<i>t</i>	<i>P</i>
Main stem				
Mean Daily Discharge (m ³ /s)	61.20	52.52	1.10	0.29
Min. Daily Discharge (m ³ /s)	13.42	15.80	-1.41	0.18
Max. Daily Discharge (m ³ /s)	239.39	166.83	1.31	0.21
Mean Daily Water Temperature (°C)	8.69	8.81	-0.41	0.69
Min. Daily Water Temperature (°C)	0.30	0.56	-0.85	0.20
Max. Daily Water Temperature (°C)	17.81	18.46	-1.17	0.26
Standard Deviation of Stream Width (m) ^a	12.28	12.77	-0.46	0.65
Tributary				
Mean Daily Discharge (m ³ /s)	10.13	8.45	0.85	0.41
Min. Daily Discharge (m ³ /s)	0.29	0.35	-1.49	0.16
Max. Daily Discharge (m ³ /s)	100.62	66.75	1.28	0.22
Mean Summer Discharge (m ³ /s) ^a	0.22	0.24	-0.25	0.80
Mean Daily Air Temperature (°C)	3.93	4.02	-0.27	0.79
Mean Wetted Width (m) ^a	6.17	6.41	-0.72	0.48
Mean Thalweg Depth (m) ^b	0.30	0.29	0.59	0.57
Standard Deviation of Thalweg Depth (m) ^b	0.15	0.13	1.32	0.22

^a Degrees of freedom (df) was 12 for these tests.

^b Degrees of freedom (df) was 10 for these tests.

Field Methods

Results from statistical tests indicated that using the mixture analysis to determine the parameter estimates from the length frequencies of main stem age 1 rainbow trout were valid. Chi squared tests of known versus predicted proportions of main stem rainbow trout in the population were not significantly different in the LCYN ($X^2 = 0.14$; 4 df; $P > 0.99$), UCYN ($X^2 = 0.03$; 2 df; $P > 0.99$), EBURG ($X^2 = 0.10$; 4 df; $P > 0.98$), THORP ($X^2 = 0.12$; 4 df; $P > 0.99$), or CELUM index sites ($X^2 = 0.10$; 4 df; $P > 0.99$) during 2003. In addition, student's t-tests of the known versus model predicted mean length of age 1 main stem rainbow trout were not significantly different in the LCYN ($t = -2.96$; 404 df; $P > 0.99$), UCYN ($t = -0.85$; 270 df; $P > 0.60$), EBURG ($t = -0.44$; 262 df; $P > 0.34$), THORP ($t = -0.89$; 174 df; $P > 0.63$), or CELUM ($t = -0.16$; 200 df; $P > 0.13$) index sites.

Residualization of Hatchery Smolts

Many spring Chinook salmon did not migrate to the ocean after release (residuals) and may have interacted with trout. In contrast, very few coho salmon residuals were observed. Residuals were most concentrated below acclimation sites during the spring and summer, but were found in all reaches of the main stem that we sampled. Residuals were observed approximately 1-2 km upstream of the acclimation site in the North Fork Teanaway during most years and also migrated upstream into the Cle Elum River. Many residuals were observed in the main stem as late as September and October. For instance, we estimated an annual average of 408, 1,195, 12,009, 554, 1,804, 2,498, 1,077, and 8 residual spring Chinook salmon during September and October from 1999 to 2006 respectively. We estimated that 54, 57, 341, 533, and 101 coho salmon residuals were in the Yakima River main stem during the fall of 1998, 2001, 2004, 2005, and 2006 respectively.

Natural Production

We did not observe coho salmon in our tributary index monitoring sites before the supplementation period (0 coho/33 sites). During the supplementation period (1999-2006) we observed coho in 5 of our index sites (between 25 and 36 sites sampled annually) averaging 3 ± 3 per km (mean \pm 95%). This increase was statistically significant ($t = -2.31$; 15 df; $P=0.04$). Our coho abundance index calculated for the main stem Yakima River averaged 0.10 ± 0.25 (mean \pm 95%) coho per river km before supplementation releases. During supplementation, we estimated 1.64 ± 1.73 coho per km (mean \pm 95%). The increase in main stem coho salmon abundance was not statistically significant ($t = -1.81$, 12 df, $P= 0.10$).

Before supplementation, our abundance index was 23 ± 17 (mean \pm 95%) spring Chinook salmon per km in our tributary monitoring sites and during the supplementation period it was 41 ± 24 per km (mean \pm 95%). The observed increase during this period was not statistically significant ($t=-1.44$; 15 df; $P=0.17$). In the main stem Yakima River, our abundance index for spring Chinook salmon indicated there were approximately 1987 ± 843 fish per km during the baseline period while there were 2045 ± 1226 fish per km during the supplementation period. The observed increase was not significant ($t=-0.86$, 12 df; $P = 0.93$).

Overlap

The degree of trout overlap with salmon was highest in main stem areas, intermediate for cutthroat and rainbow trout in tributaries, and absent for bull trout (Figure 2). There was no overlap of salmon and bull trout in our index sites. In fact, the shortest distance between the uppermost distribution of Chinook salmon and the lowermost distribution of bull trout was 8 km. Cutthroat trout and supplemented spring Chinook overlapped in distribution in both tributary and main stem Yakima River areas. The distributional overlap in tributary streams was approximately 12%, confined to relatively moderate elevations, and was less than the 40% containment objective (Figure 2). Salmon overlapped 100% of the main stem distribution of cutthroat trout (Figure 2). In tributaries, salmon overlapped 40% of the distribution of rainbow trout. Overlap was predominately confined to lower portions of tributaries (e.g., Swauk Creek 1

and Umtanum Creek 1) and farther upstream in the North Fork Teanaway River. However, salmon did not overlap rainbow trout in high elevation portions of tributaries.

There was also extensive overlap between rainbow trout, sucker species, and mountain whitefish and salmon in the main stem (100%; Figure 2). Salmon overlapped in distribution with longnose dace (53%) and speckled dace (70%) in tributaries, although mean overlap was less than the containment objectives for both species. Salmon overlapped sculpin species 33% in tributaries but this was less than the containment objective. Finally, there was 71% overlap in distribution between sucker species and salmon in tributary streams, although this was also less than the containment objective.

Data that were collected at similar times and sites by snorkeling and electrofishing methods were consistent with each other. For example, in areas that we found salmon, rainbow trout, cutthroat trout or bull trout, they were detected with both electrofishing and snorkeling methods. In addition, we did not capture any salmon when we electrofished areas where bull trout were present.

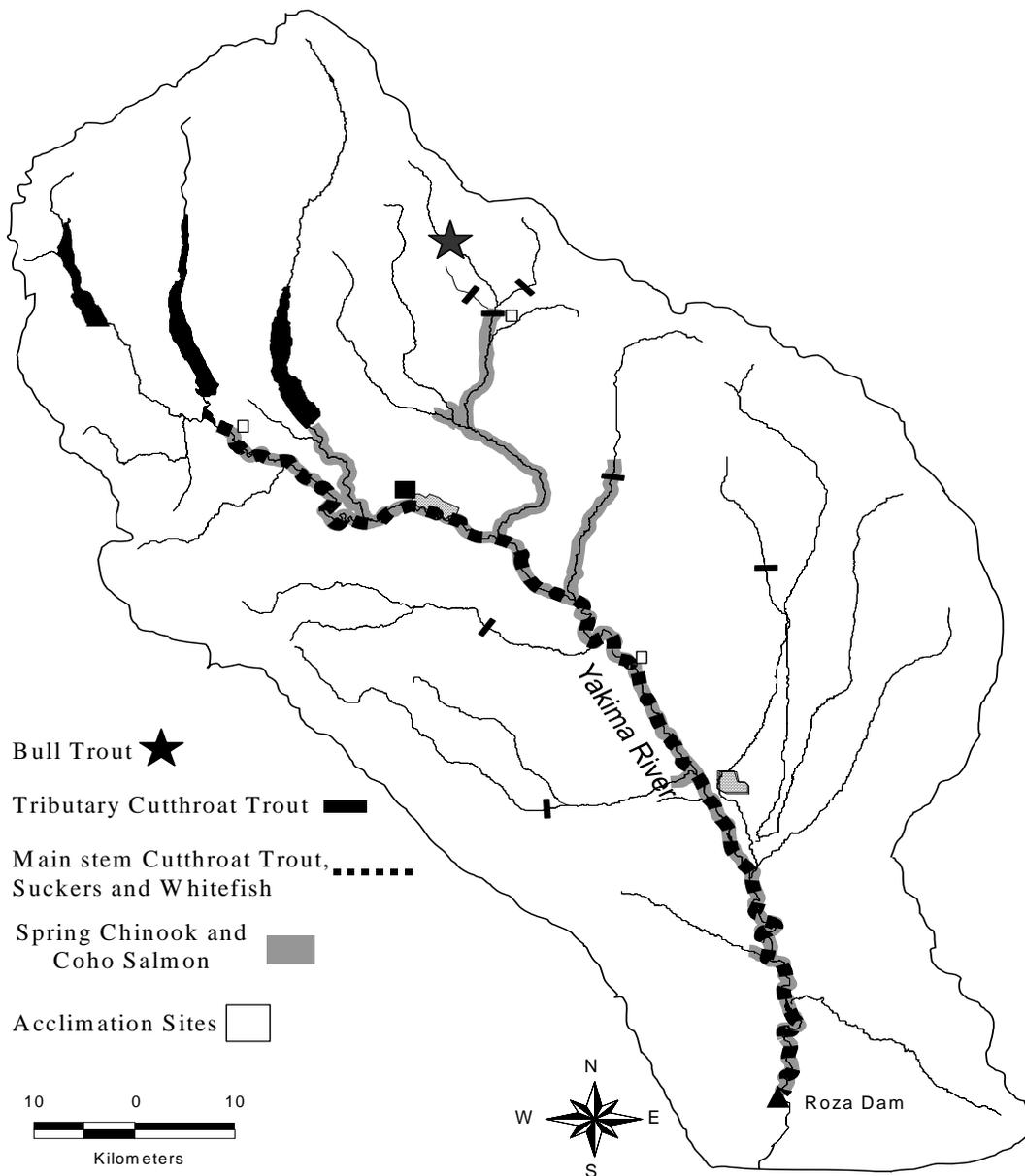


Figure 2. Map of species distributions in the upper Yakima Basin. Spring Chinook and coho salmon distributions are shaded grey. The lowest elevation observations of bull trout and cutthroat trout in tributary streams are marked with stars and bars, respectively. Cutthroat trout, suckers and mountain whitefish distribution in the main stem is marked as a dashed line. The Cle Elum hatchery facility is marked with a black square and hatchery acclimation sites are marked with open squares. Rainbow trout are widely distributed throughout the basin and are not marked on the map.

Before-After Analysis

Rainbow and cutthroat trout, mountain whitefish, and sucker species in the main stem, and rainbow trout in tributaries (analog for steelhead) exhibited overlap with salmon that were outside the containment objectives and therefore we compared their abundance, size, and biomass (salmonids) before and after stocking began. The mean abundance and 90% CL of sympatric rainbow trout was $18 \pm 18\%$ higher in the tributaries and $29 \pm 26\%$ higher in the main stem in the years when supplementation occurred than during the baseline phase (Tables 5 and 6; Figure 3). The mean abundance of cutthroat trout was $108 \pm 60\%$ CL higher in the main stem during supplementation than during the baseline phase (Table 5; Figure 3). The mean abundance of subadult mountain whitefish increased $44 \pm 17\%$ CL during supplementation period, while the mean abundance of sucker species adults decreased $44 \pm 12\%$ CL and the decrease was significant ($P=0.008$), although it was within our containment objectives (Figure 4). Finally, we observed a $9 \pm 16\%$ CL increase in subadult sized sucker abundance (analog for mountain sucker) during supplementation, however, the lower 90% CL exceeded our containment objective by approximately 1.5% (Figure 4).

During the supplementation period, the mean and 90% CL of rainbow trout size in the main stem indicated that size decreased by $2 \pm 2\%$ (Table 5; Figure 3). Slopes between log length-log weight of age 1 rainbow trout in the main stem were not significantly different before and during supplementation ($P=0.59$) and the mean weights at each length were also not different ($P=0.23$, Figure 5). In addition, biomass increased by $21 \pm 24\%$ CL. Similarly, the mean and 90% CL of cutthroat trout size in the main stem also indicated a $1 \pm 1\%$ CL decrease in size and an increase in biomass of $103 \pm 52\%$ CL (Table 5; Figure 3). The size of rainbow trout in the tributaries was similar during both periods ($1 \pm 3\%$ CL; Table 6; Figure 3). Slopes between log length-log weight for rainbow trout in tributaries were significantly different before and during supplementation ($P=0.0005$). The test of the log length-log weight relationship based upon the separate slopes model indicated the mean weights at each length were significantly greater during the supplementation period than the before period ($P=0.0005$, Figure 5). Additionally, tributary rainbow trout biomass increased by $17 \pm 14\%$ CL (Table 6; Figure 3). Our index of mountain whitefish size indicated that the proportions of subadults observed increased $14 \pm 3\%$ CL during the supplementation period (Figure 4). Our index of sucker species size indicated that the proportion of adults decreased $20 \pm 9\%$ during supplementation, and although the decrease was significant ($P=0.02$), it was still well within our containment objectives (Figure 4). Our index of mountain sucker size indicated an $18 \pm 8\%$ CL increase in the proportion of of subadults during the supplementation period (Figure 4).

The only NTT with parameter estimates outside of the containment objectives were steelhead, which uses rainbow trout as an analog, and mountain sucker, which uses subadult suckers in the main stem as an analog. The lower 90% CL for age 1 rainbow trout size and biomass in the main stem and size in tributaries, and the lower 90% CL for mountain sucker abundance were outside the containment objectives, so we tested whether the decrease was caused by supplementation.

The mean abundance, size, and biomass of catchable sized main stem rainbow trout (>249 mm) did not decrease during supplementation. The mean abundance of rainbow trout greater than 249 mm increased by $34 \pm 16\%$ (mean \pm 90%), mean size increased by $3 \pm 1\%$, and biomass increased by $64 \pm 19\%$ during supplementation when compared to baseline conditions.

Table 5. Annual abundance (fish/km), size (mm, FL), and biomass (kg/km) estimates and associated 95% confidence intervals of age 1 rainbow trout and cutthroat trout less than 250 mm fork length in the main stem Yakima River.

Year	Abundance		Size		Biomass	
	RBT	CUT	RBT	CUT	RBT	CUT
1990			210 ± 33	237 ± 5		
1991	189 ± 67	11 ± 14	205 ± 27	237 ± 11	19 ± 14	1.6 ± 3.2
1992	151 ± 28	1	217 ± 31	242	18 ± 7	0.1
1993	193 ± 48	6 ± 17	232 ± 36	238 ± 3	27 ± 11	0.8 ± 3.5
1994	180 ± 33	2 ± 1	217 ± 32	225 ± 17	21 ± 8	0.3 ± 1.4
1995	190 ± 54	6 ± 17	235 ± 34	239 ± 6	28 ± 12	0.9 ± 3.5
1996	182 ± 27	5 ± 11	217 ± 32	239 ± 10	22 ± 7	0.7 ± 2.4
1997	272 ± 49	10 ± 44	203 ± 35	239 ± 5	27 ± 10	1.4 ± 8.9
1998	130 ± 20	16 ± 84	212 ± 34	230 ± 5	15 ± 6	2.0 ± 16.8
1999	182 ± 25	12 ± 25	217 ± 33	236 ± 5	22 ± 7	1.8 ± 5.1
2000	214 ± 40	13 ± 1	210 ± 36	227 ± 13	24 ± 10	1.8 ± 1.4
2001	384 ± 81	18 ± 85	206 ± 32	238 ± 7	41 ± 16	2.5 ± 17.1
2002	207 ± 39	7 ± 42	203 ± 31	232 ± 6	20 ± 9	0.9 ± 8.4
2003	230 ± 41	10 ± 34	207 ± 30	234 ± 7	24 ± 9	1.3 ± 7.0
2004	275 ± 19	16 ± 34	223 ± 32	234 ± 5	35 ± 15	2.3 ± 6.9
2005	272 ± 20	28 ± 142	213 ± 32	229 ± 5	30 ± 9	3.4 ± 28.6
2006	150 ± 12	16 ± 11	216 ± 34	235 ± 5	17 ± 7	2.1 ± 2.5

Table 6. Annual abundance (fish/km), size (mm, FL), and biomass (kg/km) estimates and associated 95% confidence intervals for rainbow trout in Yakima River Basin tributary streams.

Year	Abundance	Size	Biomass
1990	241 \pm 129	136 \pm 8	8 \pm 13
1991	204 \pm 102	131 \pm 8	6 \pm 8
1992	375 \pm 240	130 \pm 5	11 \pm 24
1993	317 \pm 158	131 \pm 7	9 \pm 17
1994	328 \pm 129	132 \pm 8	11 \pm 15
1995	213 \pm 118	139 \pm 8	7 \pm 14
1996	165 \pm 109	133 \pm 8	5 \pm 11
1997	294 \pm 119	132 \pm 5	8 \pm 11
1998	442 \pm 174	138 \pm 7	15 \pm 25
1999	288 \pm 175	135 \pm 8	12 \pm 27
2000	318 \pm 135	144 \pm 8	11 \pm 21
2001	464 \pm 178	129 \pm 3	12 \pm 17
2002	321 \pm 131	132 \pm 6	10 \pm 15
2003	291 \pm 142	132 \pm 5	8 \pm 14
2004	243 \pm 135	142 \pm 5	9 \pm 15
2005	349 \pm 163	127 \pm 5	9 \pm 16
2006	434 \pm 171	134 \pm 5	13 \pm 20

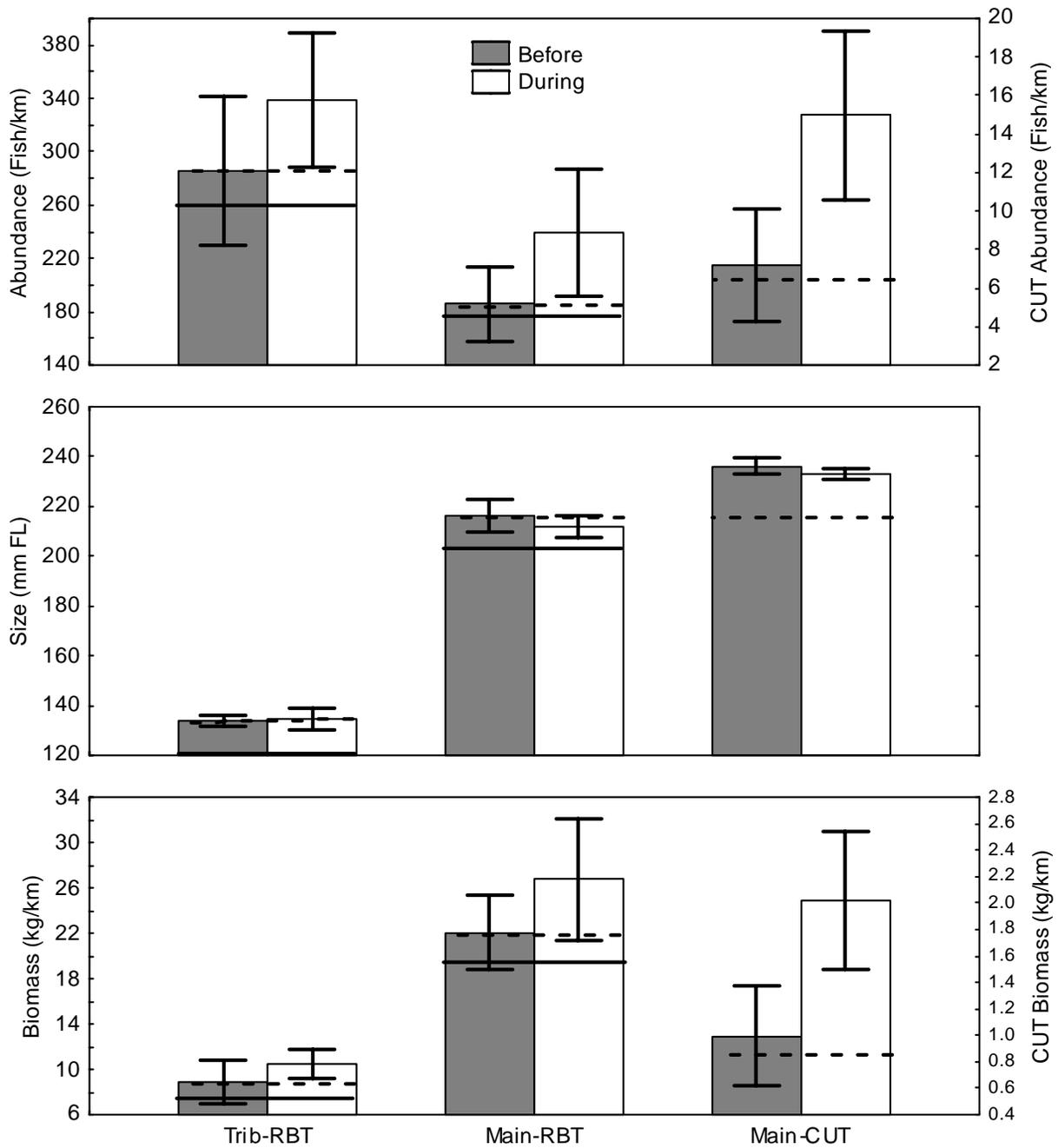


Figure 3. Abundance (n/km), size (FL mm), and biomass (kg/km) of tributary rainbow trout, main stem Yakima River rainbow trout (age 1) and cutthroat trout (<250 mm) before and during supplementation. Main stem cutthroat trout abundance and biomass are associated with the right y-axis in the abundance and biomass panels. The horizontal dashed line represents the 0% containment objective (CO) for steelhead in the main stem and tributaries, and the 10% CO for mainstem cutthroat trout. The solid horizontal line represents the 10% CO for main stem rainbow trout and 40% CO for tributary rainbow trout. Error bars represent 90% confidence intervals.

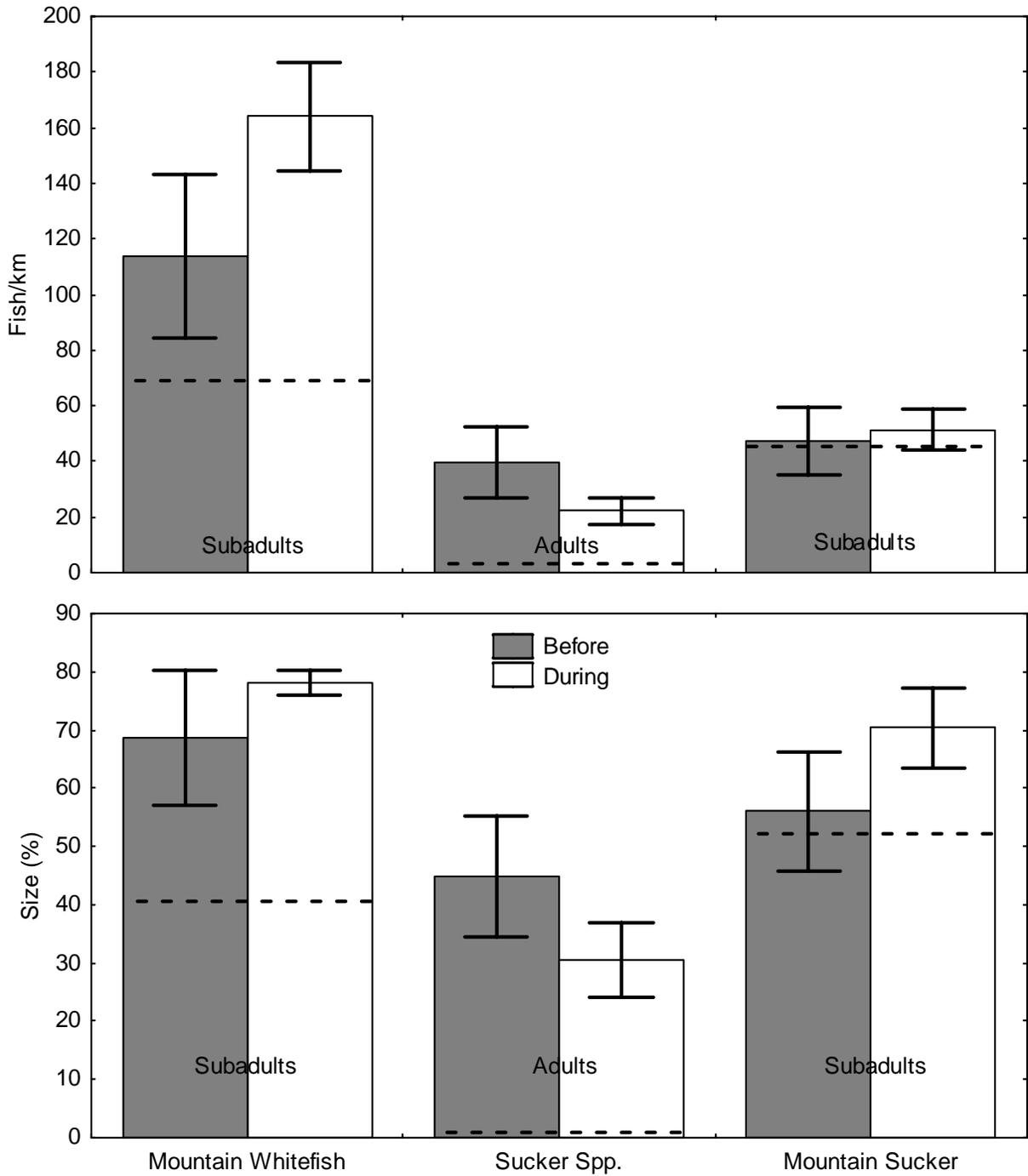


Figure 4. Abundance (fish/km) and size (percent by size class) of mountain whitefish, suckers, and mountain suckers before and during supplementation. Error bars represent the 90% confidence interval. Dashed lines represent the 40% containment objectives for mountain whitefish, 90% for sucker species (Spp), and 5% for mountain suckers.

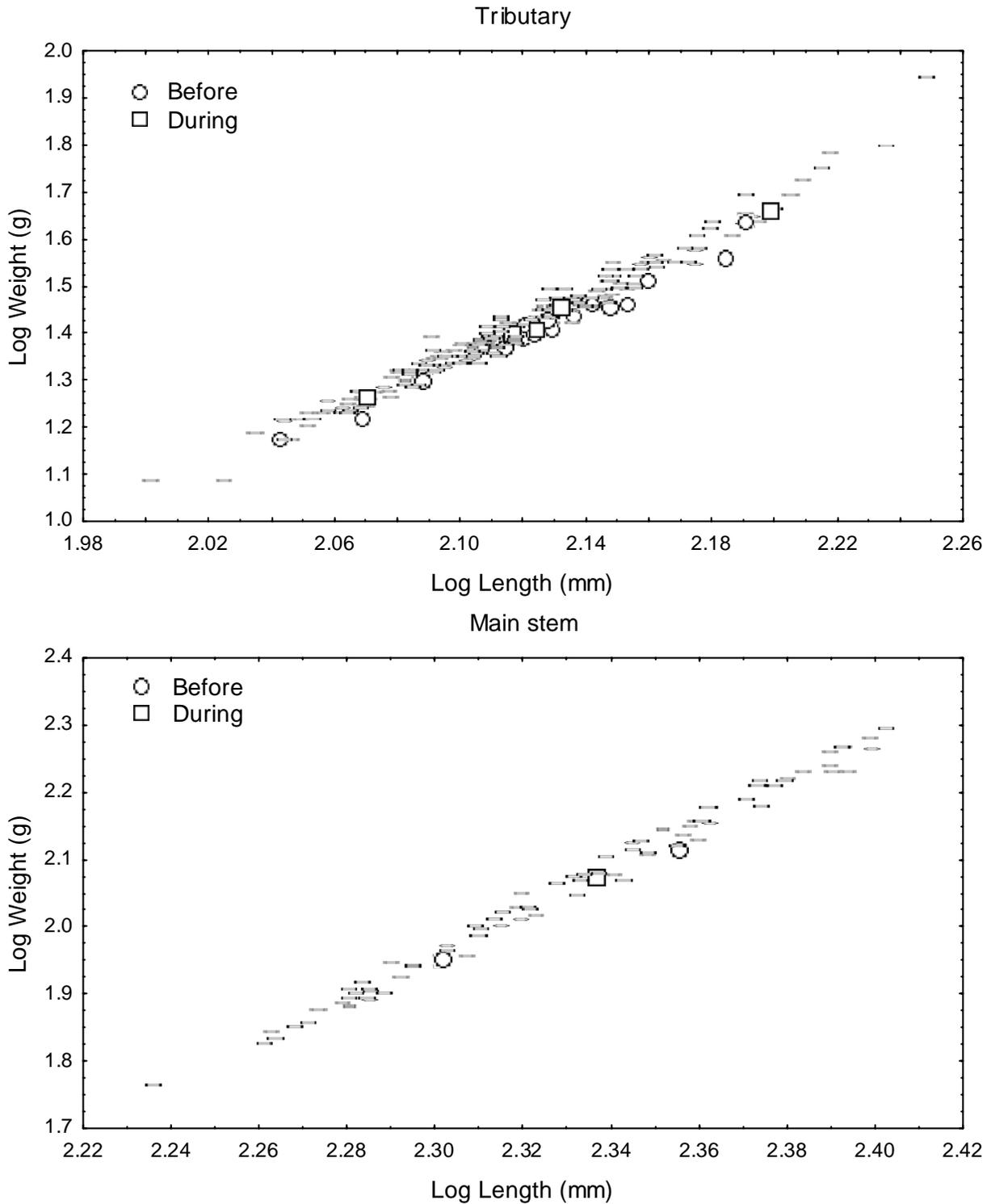


Figure 5. Mean length-weight relationships of tributary and age 1 main stem Yakima River rainbow trout before (1990-1998) and during (1999-2006) the supplementation period. Each data point represents the mean from a sample site.

Causation

We did not detect a decrease in size (BACIP; $P=0.38$), or biomass (BACIP; $P=0.24$) of rainbow trout in treatment areas of the North Fork Teanaway when compared to control sites (Figure 6). Furthermore, we did not detect a decrease in the abundance (BACIP; $P=0.18$), size (BACIP; $P=0.21$), or biomass (BACIP; $P=0.07$) of rainbow trout in the main stem below the Clark Flats acclimation site (Figure 7). We also evaluated changes in the status of mountain suckers (using subadult suckers as an analog) in areas of relatively high and low densities of salmon in the main stem upstream and downstream from the Clark Flats acclimation release site. We did not detect a significant decrease in our abundance index (BACIP; $P=0.50$) or size index (BACIP; $P=0.14$) downstream from the release site (Figure 8).

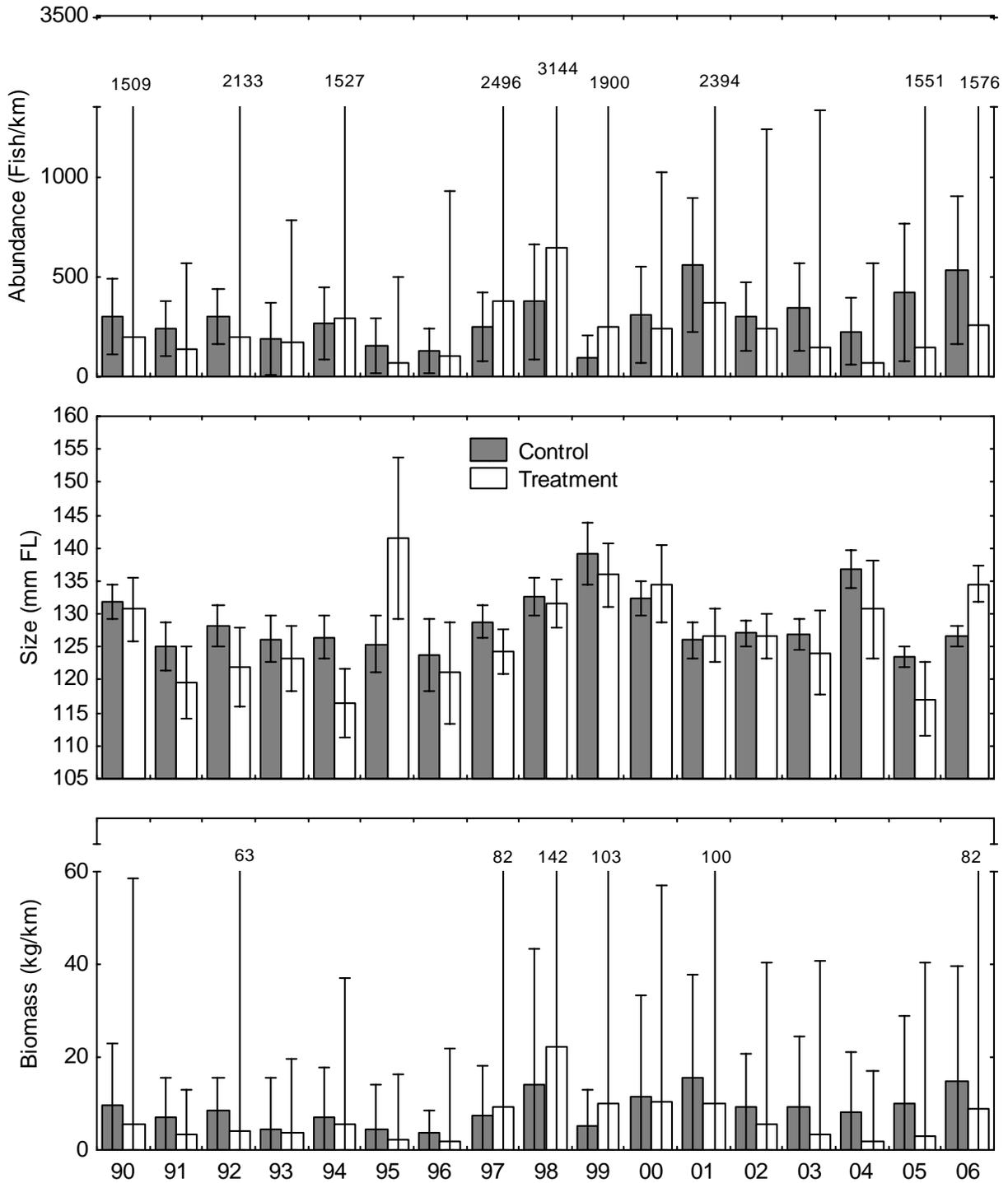


Figure 6. Abundance (fish/km), size (mm, FL), and biomass (kg/km) of rainbow trout in treatment and control areas in the Teaway River Basin. Error bars represent the 95% confidence interval and negative lower limits were truncated to prevent negative values. Numbers above error bars are the upper limit associated with large confidence intervals.

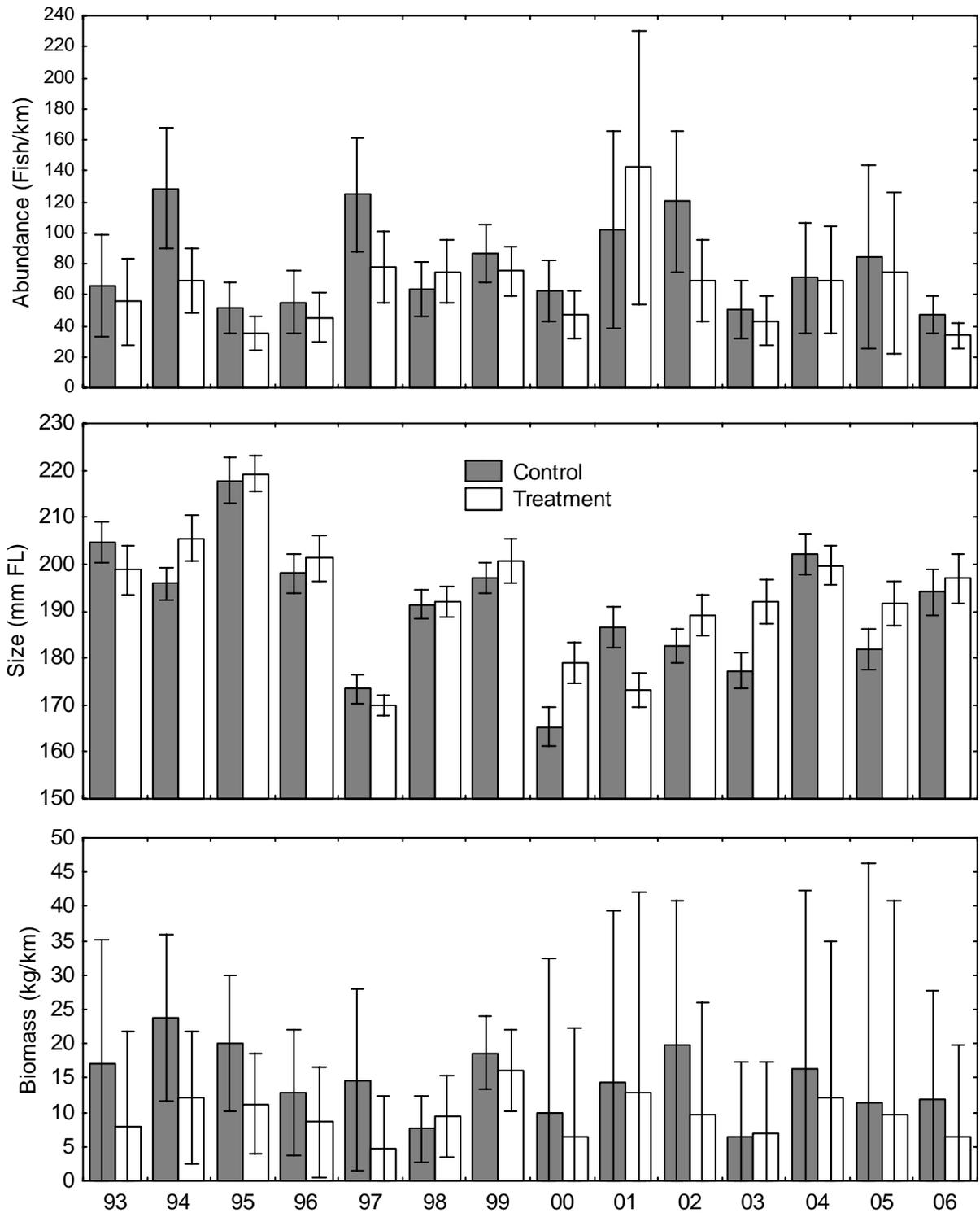


Figure 7. Abundance (fish/km), size (mm, FL), and biomass (kg/km) of rainbow trout in treatment and control areas in the main stem Yakima River. Error bars represent the 95% confidence interval and negative lower limits were truncated to prevent negative values.

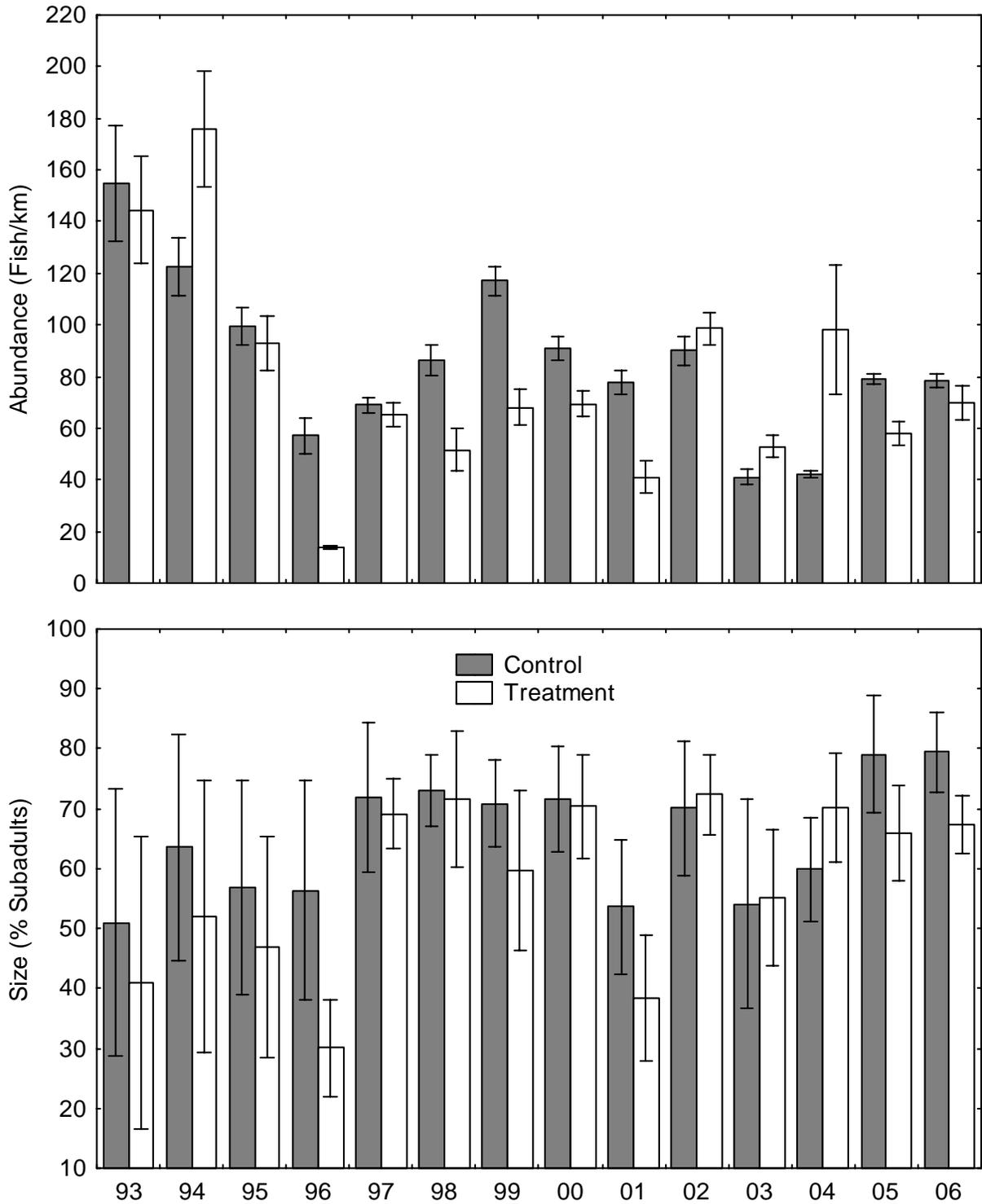


Figure 8. Abundance (fish/km) and size index of mountain suckers in treatment and control areas in the mainstem Yakima River. Error bars represent the 95% confidence interval.

Discussion

We failed to reject the hypothesis that early-middle stages of salmon supplementation have impacted valued trout species in the upper Yakima Basin beyond predetermined containment objectives. There were no impacts of supplementation activities on bull and cutthroat trout that inhabited tributary streams because limited or no overlap with hatchery or naturally produced salmon occurred. However, the potential existed for much overlap between salmon and bull and cutthroat trout in the tributaries of the upper Yakima Basin. For example, hatchery steelhead that were released in 1994 very close to the release site in the North Fork of the Teanaway River, migrated upstream into areas containing bull trout and cutthroat trout (McMichael and Pearsons 2001). Hatchery spring Chinook also migrated upstream of the acclimation site in the North Fork of the Teanaway River, but not nearly as far as hatchery steelhead. This finding is consistent with our earlier work and extends the findings into later stages of supplementation (Pearsons and Temple 2007).

It is possible that some overlap occurred at times and places when/where we did not sample. However, substantial overlap was unlikely because we sampled at times and places that overlap was most likely. There are certainly areas outside the upper Yakima watershed where overlap occurs at the times that we sampled. Furthermore, overlap has been detected using the methods we used (e.g. snorkeling). Salmon and bull and cutthroat trout overlap during the summer in another large tributary in the Yakima Basin that parallels the upper Yakima River. In the Naches Basin, which merges with the upper Yakima River near the city of Yakima, substantial overlap exists between bull and cutthroat trout and naturally produced Chinook salmon (T. Pearsons, unpublished data). Hatchery coho salmon are released into that basin and undoubtedly overlap with bull and cutthroat trout. Other studies have also documented overlap between salmon and cutthroat and bull/Dolly Varden trout (Glova 1984; Bisson et al. 1988; Nakano and Kaeriyama 1995; Thurow et al. 1997).

There are a variety of possible reasons why overlap was not detected in tributaries of the upper Yakima River. First, all but one of the acclimation sites for salmon were located in the main stem and the acclimation site in the tributary was located downstream of bull and cutthroat trout. Risks to bull and cutthroat trout were one of many factors that contributed to acclimation site placement. Second, the distribution of juvenile salmon has not increased substantially even though the abundance of adult salmon has increased. We had expected that the distribution of juvenile salmon would have increased with increasing abundance of spawners. Third, high abundance of rainbow trout in lower elevation portions of tributaries may competitively exclude cutthroat and bull trout to higher elevations that salmon do not occupy. Relaxation of competition could result in broader distributions of bull and cutthroat trout and the possibility of greater overlap with salmon. Fourth, salmon, bull trout, and cutthroat trout have different habitat preferences. Salmon typically occupy streams of lower gradient, lower elevation, and warmer water temperatures than cutthroat and bull trout (Glova 1987; Dunham and Rieman 1999). Glova (1987) concluded that impacts to cutthroat trout could be reduced by stocking coho in areas with gradients greater than 1% and ample fast water habitats. Faster water velocities allow for more resource partitioning and competitive dominance by trout. Most of the tributaries in the upper Yakima Basin met these criteria. We did observe overlap between salmon and cutthroat trout in the main stem, where water temperatures were more suitable for both of these species.

Contrary to our previous findings (Pearsons and Temple 2007), we did detect a significant impact in the abundance of rainbow trout in treatment areas in the North Fork Teanaway River and main stem Teanaway River (Chapter 3). During the past three years we have observed some of the largest differences between the control and treatment areas. With each additional year of sampling we will have increased power to detect smaller differences (Ham and Pearsons 2000). However, it is important to note that our “Before-After” detection plan would not have triggered the “Causation” analysis that was used to detect the impact (Chapter 3).

Although we observed decreases in the size of rainbow trout during the post-supplementation period, the decline is unlikely to have been caused by supplementation. If supplementation had changed the size structure or growth of the steelhead size index, we would expect to detect this change in areas with high densities of salmon. We did not detect a reduction in the size of rainbow trout in the high-density areas of the target taxa below the Clark Flats acclimation site or below the release site in the North Fork Teanaway River. These areas are likely to be the highest risk of detecting an impact. One potential explanation for the observed decrease in main stem rainbow trout size is that intraspecific density dependent mechanisms have altered the size of main stem Yakima River rainbow trout. The abundance of rainbow trout increased by nearly 30% (29% increase of age 1 fish, and 35% increase of fish greater than 249 mm) after stocking began. This information and results from small-scale enclosure experiments (McMichael et al. 1997) leads us to believe that the decline in rainbow trout lengths is most likely the result of intraspecific competition.

With the exception of the BACIP results presented in Chapter 3, the lack of detectable impacts to rainbow trout were consistent with results that were derived from smaller scale enclosure experiments between naturally produced spring Chinook salmon and rainbow trout in high elevation tributaries (McMichael and Pearsons 1998). In these experiments, growth and abundance of rainbow trout were not impacted when the density of salmonids was doubled by the addition of naturally produced spring Chinook salmon parr. However, growth of rainbow trout was suppressed when the density was doubled with rainbow trout (McMichael et al. 1997), which supports the previously mentioned idea of intraspecific impacts to rainbow trout growth in the main stem. Impacts to growth of PIT tagged fish was also not detected (Chapter 3). The current results extend the findings of McMichael and Pearsons (1998) to smolts, residuals, coho salmon, and to lower elevation waters such as the main stem. Our ability to detect impacts with the BACI design and the longer experimental period in this study (higher statistical power) may explain the differences among the studies. Opportunities for cumulative impacts to manifest and larger sample sizes may be necessary to detect impacts where high natural variation occurs.

It is possible that our abundance estimates in the main stem and tributaries and the size estimates in the tributaries were influenced by the size breaks that we used in our analysis. The lower size breaks were necessary (e.g., 80 mm in tributaries and 100 mm in the main stem), because we capture very few of these fish due to our low electrofishing efficiencies on small fish and hence cannot calculate valid estimates on these fish. This could result in varying proportions of age 0 and 1 fish in our estimate if the length at age varied across years or sites. However, we do not believe that length truncations significantly affected our conclusions. For example, if fish length was negatively impacted then the distribution of fish size would have become smaller, and more age 1 fish could have been pushed below 100 mm. Regardless of how many fish may have been shifted below 100 mm, if the impact occurred to the whole age class then we should have detected a decrease in size for fish above 100 mm (e.g., the whole length frequency curve would

be shifted to smaller sizes). Similarly, if many fish were impacted so that they were less than 100 mm then the abundance of age 1 fish would have been negatively biased. In other words, we would expect to detect less fish than we did prior to supplementation. If we had concluded that impacts had occurred, then our length truncations would be a more serious issue.

We did not detect impacts to non-trout NTT that could be attributed to supplementation. In the tributaries, this was because none of the non-trout NTT in tributaries overlapped salmon at high enough levels to exceed the CO. All non-trout NTT in the main-stem overlapped completely, but only mountain sucker exceeded the lower 90% CL for abundance. The lower 90% CL for our mountain sucker abundance index was below our containment objective by approximately 1.5%. Since we observe so few mountain suckers, we used all subadult suckers observed as an analog for evaluating impacts to mountain sucker. The potential liability of using analogs is that we assume the impact to the analog is the same to the NTT. In the case of mountain suckers, we assume the decline in all subadult sucker abundance reflects declines in mountain sucker abundance. Interestingly, although the 90% CL for mountain sucker abundance was exceeded, the proportion of subadult suckers observed increased during the supplementation period. We would assume that if substantial impacts to our abundance index were occurring, we would see a decrease in the proportion of subadult suckers in the population. However, we have not observed a decrease in proportions of subadult suckers in the population. In addition, we did not see a decrease in our mountain sucker abundance index in areas with high densities of salmon. This leads us to believe that although we have exceeded the lower 90% CL for abundance, it is unlikely to be related to supplementation activities.

We did not detect significant changes in any of the environmental variables that were measured, which supported the use of time as a control for both main stem and tributary areas. Flow and temperature in the main stem were regulated by upstream dams, which provided us an opportunity to evaluate impacts in a relatively stable environment. Although discharge in tributaries is unregulated, summer baseflows have not differed drastically during the time of sampling from year to year. This supports the use of time as a control for tributary streams as well.

Management Implications

There are a number of management decisions that likely contributed to the small number of impacts in our study. For example, acclimation site placement and the release of fish at the smolt stage reduced the potential spatial and temporal overlap with NTT. Implementation of strategies to limit the number of precociously mature male salmon entering the natural environment would further decrease the risk of failing to meet containment objectives in the future. By reducing the number of these precociously mature salmon, both direct and indirect undesirable interactions with NTT will be reduced. Strategies to reduce the production and release of precociously mature salmon are currently being implemented.

We are implementing the approach described by Ham and Pearsons (2001) to contain risks to NTT throughout the life span of salmon supplementation programs in the Yakima Basin (Pearsons 2002). According to this risk containment approach, if we detect a decrease in NTT status that is greater than a containment objective, then we attempt to determine if the decrease was caused by the supplementation program. Only changes that are due to supplementation warrant risk containment action specific to the supplementation program. The only NTT that were outside of their containment objectives were steelhead and mountain sucker. Steelhead was

monitored using age 1 rainbow trout as an analog of juvenile steelhead in the main stem and all rainbow trout in the tributaries, and mountain sucker were monitored using all observed subadult suckers as analogs. The declines in mean steelhead trout size and the surpassing of the lower 90% CL of the containment objective for abundance for both steelhead and mountain sucker were unlikely to be due to supplementation and therefore do not require risk containment actions. However, the impact to rainbow trout abundance in the North Fork Teanaway is cause for concern because it exceeds the CO for steelhead (Chapter 3). Finally, the building stage of supplementation began in 2002 when juveniles from the first return of age 4 hatchery fish were available to interact with wild fish. This stage is likely to be the one where the risk of impacts is the highest because the number of salmon is potentially very high due to a combination of hatchery and naturally produced fish (Pearsons 2002).

We recommend that our results be placed into context of supplementation dynamics and the ecological context of the Yakima Basin. Our evaluation addressed impacts that occur during the early-middle stages of supplementation. These stages correspond to the Broodstock stage and the Building stage described by Pearsons (2002). The Building stage, provides the greatest opportunity for interactions between hatchery fish, naturally produced offspring of hatchery fish, and NTT. Large numbers of hatchery smolts, residuals, and the offspring from returning hatchery adults increases interaction potentials between hatchery and wild fish in the freshwater migration corridor and freshwater rearing area. The Yakima main stem is a highly altered system, and this alteration could change the outcomes of ecological interactions. For example, the flows in the upper Yakima main stem are heavily regulated by the United States Bureau of Reclamation's water storage and delivery programs and are highly unnatural. Artificially high flows during the summer might decrease interspecific interactions among species because of the large volume of water and ability for species to partition resources. In contrast, flows in the tributaries are more normative and might be more reflective of tributaries in other regions. Application of our results should be tempered within these important contexts.

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Chapter 3

Hatchery supplementation of salmon impacts rainbow trout abundance and reduces salmonid biomass

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Abstract

We used a Before-After-Control-Impact-Paired design to evaluate impacts of hatchery salmon reintroduction and supplementation on rainbow trout *Oncorhynchus mykiss* in two contiguous rivers. Trout and salmon were sampled with backpack electrofishers and abundance, size, and biomass was estimated in treatment and control sites before (1990-1998) and during hatchery releases (1999-2006). We detected impacts to rainbow trout abundance and biomass ($P < 0.05$), but not size ($P > 0.05$) from production scale releases of coho salmon *O. kisutch* (1999) and spring Chinook salmon *O. tshawytscha* (2000-2006) in the Teanaway Basin. Impacts were largest in the stream closest to the release site. The impacts to rainbow trout were likely the result of cumulative impacts from hatchery released Chinook salmon smolts, residualized spring Chinook salmon, and an increase in naturally produced parr. Each one of these life-stages was significantly correlated with differences in trout abundance in at least one treatment location ($P < 0.05$). We also observed that the combined biomass of rearing rainbow trout and spring Chinook salmon was impacted by supplementing spring Chinook salmon ($P < 0.05$). This study highlights the importance of evaluating hatchery programs from a long-term and multi-species perspective.

Introduction

To determine if hatchery augmentation of salmon produce net ecosystem benefits, it is important to know whether increasing salmon abundance causes impacts to other species that are not the target of enhancement. In some of our earlier work, we found that rainbow trout *Oncorhynchus mykiss* were not impacted beyond management objectives by supplementation of spring Chinook salmon *O. tshawytscha* and restoration of coho *O. kisutch* salmon in the Yakima Basin after 5 years of stocking (Pearsons and Temple 2007). However, this study assessed only the early stages of supplementation and impacts could accumulate through time (Pearsons 2002). We continued to monitor impacts to rainbow trout, and found that after 8 years of stocking, an impact to rainbow trout in the Teanaway Basin was detected. We also investigated whether the impacts to rainbow trout exceeded the increase in biomass provided by enhancing salmon.

Hatchery supplementation of salmon can unintentionally impact other species that are highly valued and are not the target of enhancement (non-target taxa; NTT). For example, hatchery releases of steelhead trout *O. mykiss* were negatively associated with smolt-to-adult survival of wild Chinook salmon in the Snake River (Levin and Williams 2002). Hatchery program managers do not intentionally seek to harm other species in the course of enhancing another, but the fact that salmonids are strong interactors with a variety of other species suggests that the potential to impact other species is highly probable. Salmon interact with other species through mechanisms such as competition, predation, disease, and nutrient supply (Pearsons and Hopley 1999; Pearsons 2002).

The management paradigm used by hatchery programs has traditionally been one of single species management. However, without an understanding of the net ecosystem benefits and costs, the success of a program may be incorrectly evaluated. This could lead to the continuance or expansion of a management strategy that reduces abundance of other valued species and also contributes to reduced ecosystem efficiency (e.g., conversion of resources into biomass).

Restoring species to historical ranges has the potential to increase ecological efficiency resulting in combined biomass of valued species. It is generally believed that multiple species will be able to produce higher levels of biomass than single species because of the opportunity to use a greater variety of the resources that are available (Everest and Chapman 1972). Multiple species can partition food and space resources (Ross 1986; Bisson et al. 1988) and potentially use resources that are unused by single species (McMichael and Pearsons 1998). Alternatively, if species that are the target of restoration are able to convert shared resources at the same rate as existing species, then biomass among species could be exchanged but no changes in combined biomass might occur. Lastly, if ecological interactions between species reduces multispecies energy conversion efficiency, or if targeted species are less efficient at converting resources to biomass, then combined biomass may actually be reduced. (Nickelson et al. 1986; Weber and Fausch 2005). Thus, depending upon the energy conversion efficiency of the target species and it's impact on conversion efficiency of other valued species, combined biomass of multiple species could increase, decrease, or remain the same.

Growth and abundance of rainbow trout in high elevation streams of the upper Yakima River Basin were not affected when salmonid densities were doubled with naturally produced

Chinook salmon parr (McMichael and Pearsons 1998). These controlled experiments were conducted in small to moderate size enclosures. The current study extends the findings of McMichael and Pearsons (1998) by evaluating production scale supplementation of spring Chinook salmon throughout a large tributary. In addition, impacts from one year of coho salmon reintroduction were also evaluated. Coho salmon displace some steelhead trout from pools or alter habitat use within pools that they would occupy in the absence of coho (Hartman 1965; Allee 1974; 1981). Growth of steelhead was lower in experiments with high coho densities than with low densities (Fraser 1969). Furthermore, this study includes an evaluation of interactions from hatchery-produced smolts, hatchery-produced residuals, and naturally produced offspring of hatchery fish and extends the findings of Pearsons and Temple (2007). In that study, impacts of the early stages of supplementation (5 years) were within management containment objectives for *O. mykiss*, cutthroat trout *O. clarki*, and bull trout *Salvelinus confluentus*.

In this paper, we examine the impacts to rainbow trout during the early-middle stages of a spring Chinook supplementation program and the reintroduction of coho salmon in the Teanaway Basin, Washington (Figure 1). It was unknown whether impacts would occur from smolts that migrated downstream of the acclimation site, residuals that distributed themselves up- and down-stream of acclimation sites, and/or from an increase in the distribution and abundance of naturally produced offspring of supplemented adults. Certainly one of the goals of supplementation is to increase the distribution and abundance of target species.

Methods

Study Area

The Teanaway River is the largest tributary that enters the upper Yakima River near the city of Cle Elum, Washington (Figure 1). There are three primary forks that flow into the Main stem Teanaway River (Figure 1). Salmon are released into the North Fork Teanaway annually, but no fish are released into the Middle and West forks. Smolts that are released into the North Fork also migrate through the main stem Teanaway River. Adults return to spawn in the Teanaway Basin and their progeny rear there too.

Historically large numbers of salmon and steelhead returned to the upper Yakima Basin, including the Teanaway Basin (Bonneville Power Administration 1996). Coho salmon were extirpated by the early 1980s (Bosch et al. 2007) and spring Chinook salmon have been severely depressed particularly in the Teanaway Watershed. Steelhead (mid-Columbia Evolutionarily Significant Unit) and bull trout are currently listed as threatened (National Marine Fisheries Service 1999; U.S. Fish and Wildlife Service 1998).

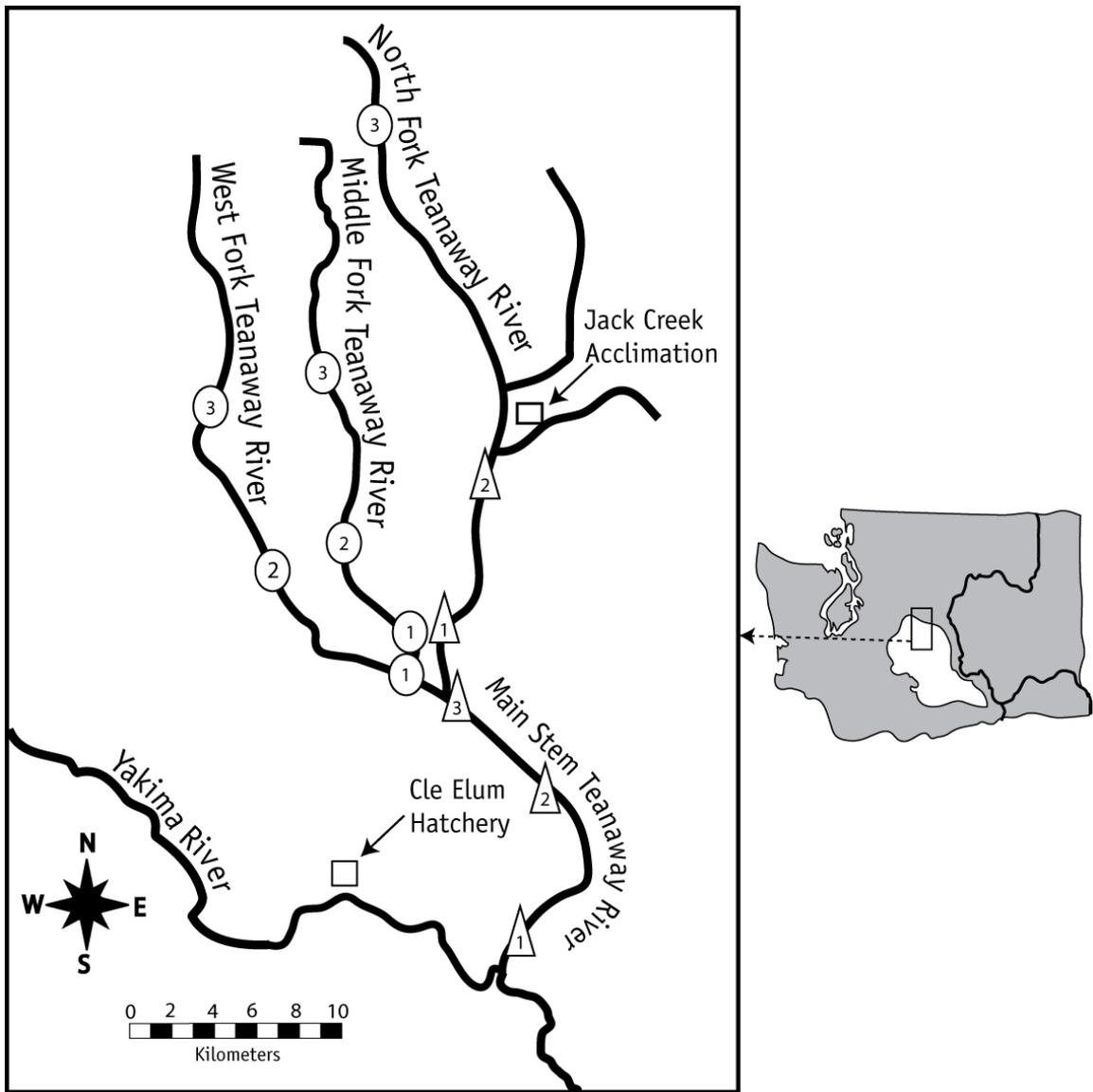


Figure 1. Map of the Teanaway River Basin and the locations of different sample sites. Control sites are represented by circles and treatment sites are represented by triangles.

Chinook and coho salmon hatchery programs

The fish that are released into the North Fork Teanaway are part of a larger program to enhance and restore salmon in the Yakima Basin. Spring Chinook and coho salmon yearlings were released into the upper Yakima Basin for the first time during spring 1999 as part of the Yakima/Klickitat Fisheries Project (YKFP). The goal for both of these species is to increase natural production and to provide harvest opportunity using artificial propagation while keeping adverse impacts within specified biological limits (Bonneville Power Administration 1996; Fast and Craig 1997; Bosch 2004). Approximately one million Chinook salmon smolts have been released annually in the upper Yakima River from 1999 to 2006, and approximately 33% of the annual releases occur in the North Fork Teanaway River (Table 1). Broodstock for the spring Chinook program were natural origin upper Yakima stock collected at a trapping facility at Roza Dam. Chinook salmon adults were spawned and juveniles were reared at a hatchery facility in Cle Elum, Washington (Knudsen et al. 2006). Spring Chinook salmon yearlings were transported to acclimation sites during January and February and volitionally released into the Yakima River from the Easton and Clark Flats acclimation sites, and from the Jack Creek acclimation on the North Fork of the Teanaway River (Figure 1). Fish were permitted to volitionally migrate between March 15 and May 31 and averaged 120 mm fork length (FL) when released. At the middle to end of May, all remaining fish were forced out of the acclimation sites into the river. Mean travel time of PIT-tagged migrants from acclimation sites to a PIT-tag detector near Prosser Washington (river kilometer [RKM] 75.6 measured from the confluence with the Columbia River) is about 24.3 days. However, a large number of Chinook salmon residualize in the upper Yakima Basin (Larson et al. 2004; Pearsons et al. 2005).

Table 1. Numbers of yearling spring Chinook and coho salmon released in the North Fork Teanaway River annually. Coho salmon were released in the North Fork Teanaway River during 1999 and spring Chinook salmon were released between 2000 and 2006.

Brood Year	Release Year	Release Number
1997	1999	240,000
1998	2000	137,363
1999	2001	256,724
2000	2002	285,270
2001	2003	250,348
2002	2004	274,466
2003	2005	278,508
2004	2006	219,110

Broodstock for the coho salmon program were largely from lower Columbia hatchery sources that have been under culture for many generations. Coho salmon were volitionally released into the Yakima River from sites near the Cle Elum Hatchery (hatchery slough 1999, 2000, and 2001), from the Easton spring Chinook acclimation site (1999-2003), from the Holmes

site (2003-2006) approximately 8.2 rkm upstream from the city of Ellensburg, Washington, from the Boone Pond acclimation site (2004-2006) near Yakima River kilometer 290.5, and into the North Fork of the Teanaway River from the Jack Creek spring Chinook acclimation site (1999; Table 1). Mean travel time of PIT-tagged migrants from the Holmes site to a PIT-tag detector near Prosser Washington in 2003 (RKM 75.6) was 41.9 days. Very few coho salmon have been observed to residualize in the upper Yakima River. Coho salmon sizes were not recorded at the time of release, however, two to three months after release, hatchery origin coho smolt fork lengths averaged 157 mm in the migration corridor. Prior to 1999, coho salmon were released in the middle portion of the Yakima River, a considerable distance below Roza Dam. Coho salmon observed in the upper Yakima River prior to 1999 were likely the result of downstream releases. More detail about the study area and background of the supplementation project has been previously described (Busack et al. 1997; Pearsons and Hopley 1999; Ham and Pearsons 2000). Early findings indicate that Chinook and coho hatchery programs are increasing the abundance and distribution of spawners in the upper Yakima Basin and in the Teanaway Basin (Bosch 2004; Bosch et al. 2007).

Field Methods

Environmental variables were compared before and during supplementation to determine if key environmental factors changed in the treatment sites relative to the control sites between the two periods. We measured stream wetted width and stream flow at the time of sampling and computed total stream discharge as described by Gallagher and Stevenson (1999). We also generated a longitudinal streambed profile in each site by recording water depth at 1 m intervals along the stream thalweg as described by Kaufmann (1987). Variability of thalweg measurements, calculated as the standard deviation (SD) of thalweg depths, were calculated to index habitat complexity and residual pool volume. Thalweg measurements were discontinued in 2005, so before-after comparisons were limited to the period 1993-2004 in North Fork Teanaway River index sites, and 1994-2004 for main stem Teanaway River index sites. Wetted width and stream flow measurements in tributaries during the years 1990-1992 were not the same as the period 1993-2006 so we excluded this time period in our analysis.

The population abundance indices of salmonids in index monitoring sites in the Teanaway Basin were generated from single-pass, backpack electrofishing capture efficiency expansions (Temple and Pearsons 2004; Temple and Pearsons 2007; Pearsons and Temple 2007). A crew of three to six people electrofished 200 m long index sites during the day with a backpack electrofisher during summer base flows (Table 2). A single upstream electrofishing pass was performed and attempts were made to net all visible fish. Netted fish were held in perforated buckets in the stream margins until they were processed. All fish were anesthetized, identified to species, and the lengths and weights of salmonids were recorded. We assumed, and later verified, that electrofishing efficiencies would be poor for small fish and restricted our evaluation to salmonids greater than 79 mm FL (Reynolds 1983; Anderson 1995).

An index of salmonid abundance was calculated by expanding the first pass count by a median capture ratio established for each site during the baseline monitoring phase. Median capture ratios were calculated by dividing the number of fish captured on the first electrofishing pass by a multiple-removal maximum likelihood estimate of the number of fish in the site (Zippen 1958). A minimum of two and a maximum of six electrofishing removal passes were used to generate capture efficiencies at each site once annually during the baseline phase.

Median efficiencies were based on between four and eight annual baseline replicates depending on the year each site was established. On average, removal based efficiencies indicated that 75% of the trout greater than 79 mm FL were captured during the first electrofishing pass.

Approximately 36% of the sites were sampled with multiple removal methods during both periods to validate the single pass expansions. Due to the low abundance of juvenile coho salmon and their similar size with Chinook salmon, we applied spring Chinook salmon capture efficiencies to the number of coho salmon captured in the first electrofishing pass to obtain an index of coho abundance.

Table 2. Salmon and rainbow trout distribution monitoring site locations in the Teanaway Basin. Before-After-Control-Impact-Paired (BACIP) abundance and size monitoring control (C) and impact (I) site designations are presented as well. Latitude and longitude positions are presented in degrees, minutes, and seconds (D°M'S").

Stream / Site	BACIP	Years	Latitude	Longitude
M.F. Teanaway 1	C	90-06	47° 15' 53.54"	120° 53' 53.19"
M.F. Teanaway 2	C	90-06	47° 16' 51.06"	120° 55' 50.37"
M.F. Teanaway 3	C	90-06	47° 17' 57.47"	120° 57' 42.06"
W.F. Teanaway 1	C	90-06	47° 15' 25.52"	120° 53' 56.00"
W.F. Teanaway 2	C	90-06	47° 15' 51.79"	120° 57' 11.25"
W.F. Teanaway 3	C	90-06	47° 16' 11.37"	120° 58' 36.13"
M.S. Teanaway 1	I	94-06	47° 10' 58.40"	120° 49' 29.80"
M.S. Teanaway 2	I	94-06	47° 13' 28.32"	120° 48' 15.61"
M.S. Teanaway 3	I	94-06	47° 15' 6.65"	120° 52' 27.53"
N.F. Teanaway 1	I	90-06	47° 16' 53.10"	120° 51' 53.86"
N.F. Teanaway 2	I	90-06	47° 18' 41.97"	120° 51' 31.40"

Size of rainbow trout and salmon was quantified using the mean length of fish collected in our tributary sites. All rainbow trout that were captured were measured to the nearest mm fork length (FL) and weighed to the nearest gram. Mean length of rainbow trout in each tributary site was calculated and then the mean of the site means was calculated for each year. Mean weight of rainbow trout in each tributary site was averaged and then the mean of the averages was calculated for each year. Biomass estimates were generated from the product of mean estimated abundance and mean weight.

To improve our ability to detect potential impacts to rainbow trout growth resulting from supplementation releases, we tagged rainbow trout with Passive Integrated Transponder (PIT) tags in treatment and control areas in the Teanaway basin during 2006. Fish were collected in the North Fork Teanaway River downstream from the Jack Creek Acclimation Facility (treatment) and in the Middle Fork Teanaway River (control) with a backpack electrofisher and were measured, tagged, and released near their point of capture. Tagging collections were performed primarily during June, and recapture sampling during September-October.

Analysis

We used a before-after-control-impact-paired site analysis (BACIP; Stewart-Oaten et al. 1986) to evaluate changes to abundance, size, or biomass of rainbow trout; abundance of spring Chinook salmon, combined biomass of rainbow trout and Chinook salmon, and environmental variables. This test evaluates the annual paired differences in control and impact sites before and after stocking. Paired differences before and after stocking were compared with a two factor ANOVA with the North Fork Teanaway River and main stem Teanaway River as site factors (stream) and time (before vs. during; period) as another factor. We analyzed our treatment locations as separate factors because the ecological conditions, fish densities, and distance from release location in these locations were different. The treatment sites were based on their proximity to acclimation sites and included the migration corridor of released smolts and the control sites were selected based on their geographic proximity to treatment sites. Two sites located downstream from the Jack Creek acclimation facility on the North Fork Teanaway River and three sites in the main stem Teanaway River were used as treatment sites and three sites on each of the Middle and West Forks of the Teanaway River were the paired controls (Figure 1). We used a three factor ANOVA to test for differences in environmental variables (treatment-control). Stream, environmental variable (mean wetted width, discharge, mean thalweg depths, and SD of thalweg depths), and period (before versus during supplementation), were the three factors included in the analysis. The “before” period was from 1990-1998 and the “after” period was from 1999-2006.

We investigated relationships between the abundances of different Chinook salmon life stages and rainbow trout abundance to determine life-stages that were directly associated with rainbow trout abundance. More specifically, annual hatchery Chinook smolt abundance, annual difference in parr in treatment streams minus that in control streams, and annual difference in hatchery residuals in treatment streams minus that in control streams were compared to rainbow trout in treatment streams minus that in control streams using simple linear regression.

Instantaneous growth was calculated for recaptured PIT tagged fish in treatment and control areas following Ricker (1975). Instantaneous growth of fish in treatment and control areas were compared using a student’s t test. We used an analysis of co-variance (ANCOVA) to test the hypothesis that the log length-log weight relationships of rainbow trout did not differ between treatment and control areas in the Teanaway Basin.

Results

Comparisons of environmental variables measured in main stem and North Fork Teanaway index sites before and during supplementation indicated that there were significant differences in the variables evaluated relative to the control sites (Table 3; BACIP; $P=0.006$; Figure 2). The main difference was that there was an increase in mean wetted stream widths in treatment sites relative to the control sites during the supplementation period (Figure 2). Although mean wetted stream width increased during the supplementation period, mean stream discharge and mean thalweg depth did not (Figure 2).

Table 3. Analysis of variance results for environmental variables in treatment sites relative to control sites before and during supplementation.

Source of Variance	SS	DF	MS	F	p
Intercept	101.87	1	101.87	37.58	0.00
Stream	0.05	1	0.05	0.02	0.89
Period	33.44	1	33.44	12.34	0.00
Environmental Variable	242.17	3	80.72	29.78	0.00
Stream*Period	13.68	1	13.68	5.05	0.03
Stream*Environmental Variable	0.31	3	0.10	0.04	0.99
Period *Environmental Variable	111.91	3	37.30	13.76	0.00
Stream*Period*Environmental Variable	35.33	3	11.78	4.34	0.01
Error	265.65	98	2.71		

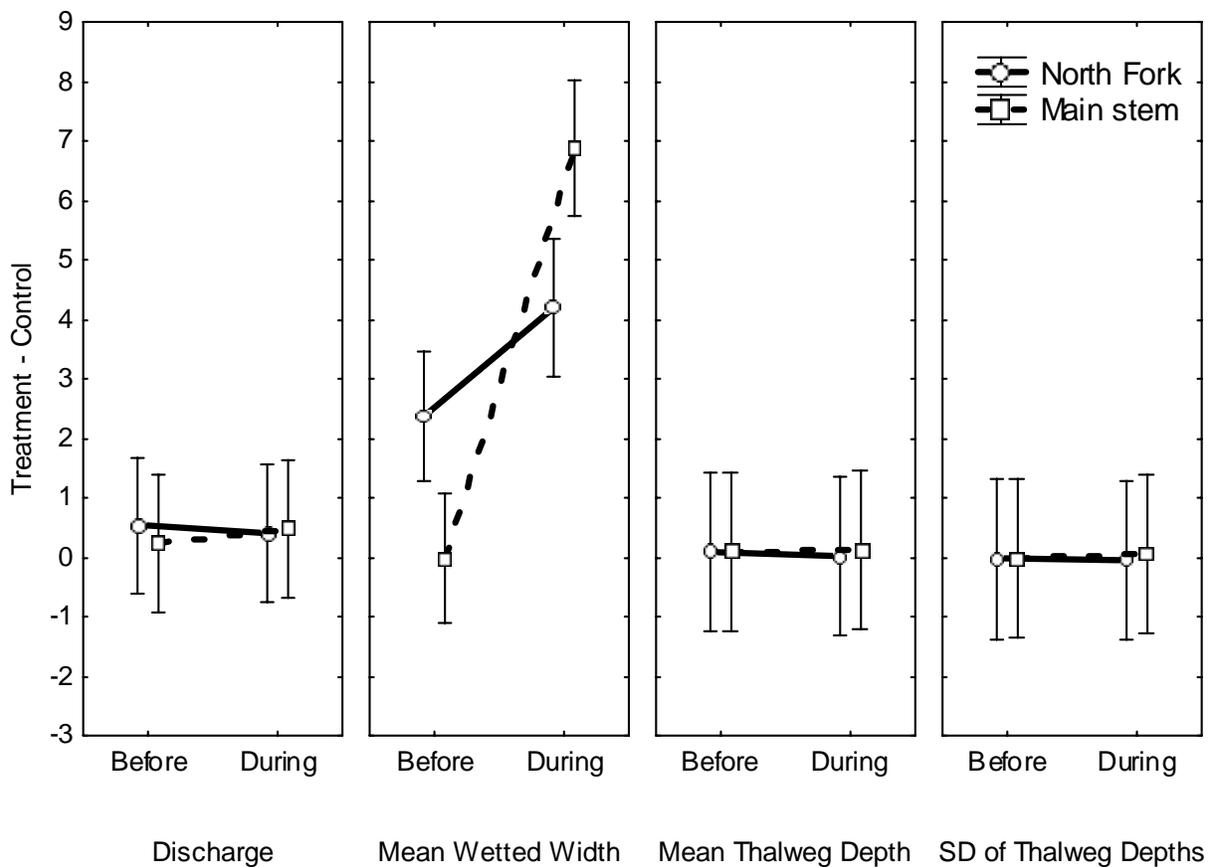


Figure 2. Comparison of environmental variables in North Fork and main stem Teanaway index monitoring sites before versus during the supplementation period relative to environmental conditions in control sites.

Many spring Chinook salmon did not migrate to the ocean after release (residuals) and may have interacted with trout. In contrast, no coho salmon residuals were observed. Residuals were most concentrated below the acclimation site during the spring and summer. Residuals were observed approximately 1-2 km upstream of the acclimation site in the North Fork Teanaway during most years.

Natural production of spring Chinook salmon parr in treatment locations increased relative to control sites during supplementation, but the difference was marginally significant ($P=0.09$; Table 4). There was not a significant difference in differences in natural origin spring Chinook salmon abundance in the treatment sites (Table 4). Biomass of juvenile spring Chinook salmon significantly increased during the supplementation period ($P=0.03$).

Table 4. Analysis of variance of paired differences in spring Chinook salmon abundance.

Source of Variance	SS	DF	MS	F	p
Intercept	44943	1	44943	13.67	0.001
Stream	2516	1	2516	0.77	0.39
Period	10193	1	10193	3.10	0.09
Stream*Period	5940	1	5940	1.81	0.19
Error	85426	26	3285		

We detected a significant decrease in the paired differences (treatment minus control) in abundance of rainbow trout before and during supplementation in both the North Fork and Main stem Teanaway (Table 5; Figure 3). Differences in fish abundance between treatment and control areas in the NFT during supplementation were 102 times less than the quantity before supplementation (paired differences of treatment-control were -1.3 fish/km before and -133.2 fish/km during) and in the MST they were nearly 2 times less (paired differences of treatment-control were -126.5 fish/km before and -226.1 fish/km during). The five biggest differences between the abundance of rainbow trout in control streams and the North Fork Teanaway treatment stream occurred during supplementation. Similarly, four of the six years during the supplementation period had the biggest differences in abundance in the Main stem Teanaway relative to control streams. Sites in control areas exhibited a 42% increase in abundance during supplementation, sites in the North Fork Teanaway lost 12%, and sites in the Main stem Teanaway increased 14% when compared to baseline conditions.

We did not detect a decrease in size (Table 5; Figure 4), but we did detect a decrease in biomass (Table 5; Figure 5) of rainbow trout in these areas during the supplementation period relative the baseline period. Differences in the length-weight relationship of rainbow trout in treatment and control streams were not detected (ANCOVA; $P=0.41$; Figure 6). In addition, we did not detect a difference in the instantaneous growth of individual PIT-tagged rainbow trout downstream from the acclimation facility release site in the North Fork Teanaway River relative to control sites located in the Middle Fork Teanaway River ($P=0.33$; Figure 7).

Table 5. Analysis of variance results of paired differences in rainbow trout abundance, size, and biomass.

Source of Variance	SS	DF	MS	F	p
Abundance					
Intercept	422731	1	422731	30.63	0.00
Stream	84714	1	84714	6.14	0.02
Period	95474	1	95474	6.92	0.01
Stream*Period	1859	1	1859	0.13	0.72
Error	358827	26	13801		
Size					
Intercept	213.00	1	213.00	3.43	0.08
Stream	1175.30	1	1175.30	18.95	0.00
Period	50.14	1	50.14	0.81	0.38
Stream*Period	181.43	1	181.43	2.92	0.10
Error	1612.75	26	62.03		
Biomass					
Intercept	416.93	1	416.93	38.72	0.00
Stream	62.41	1	62.41	5.80	0.02
Period	53.83	1	53.83	5.00	0.03
Stream*Period	0.13	1	0.13	0.01	0.91
Error	279.91	26	10.77		

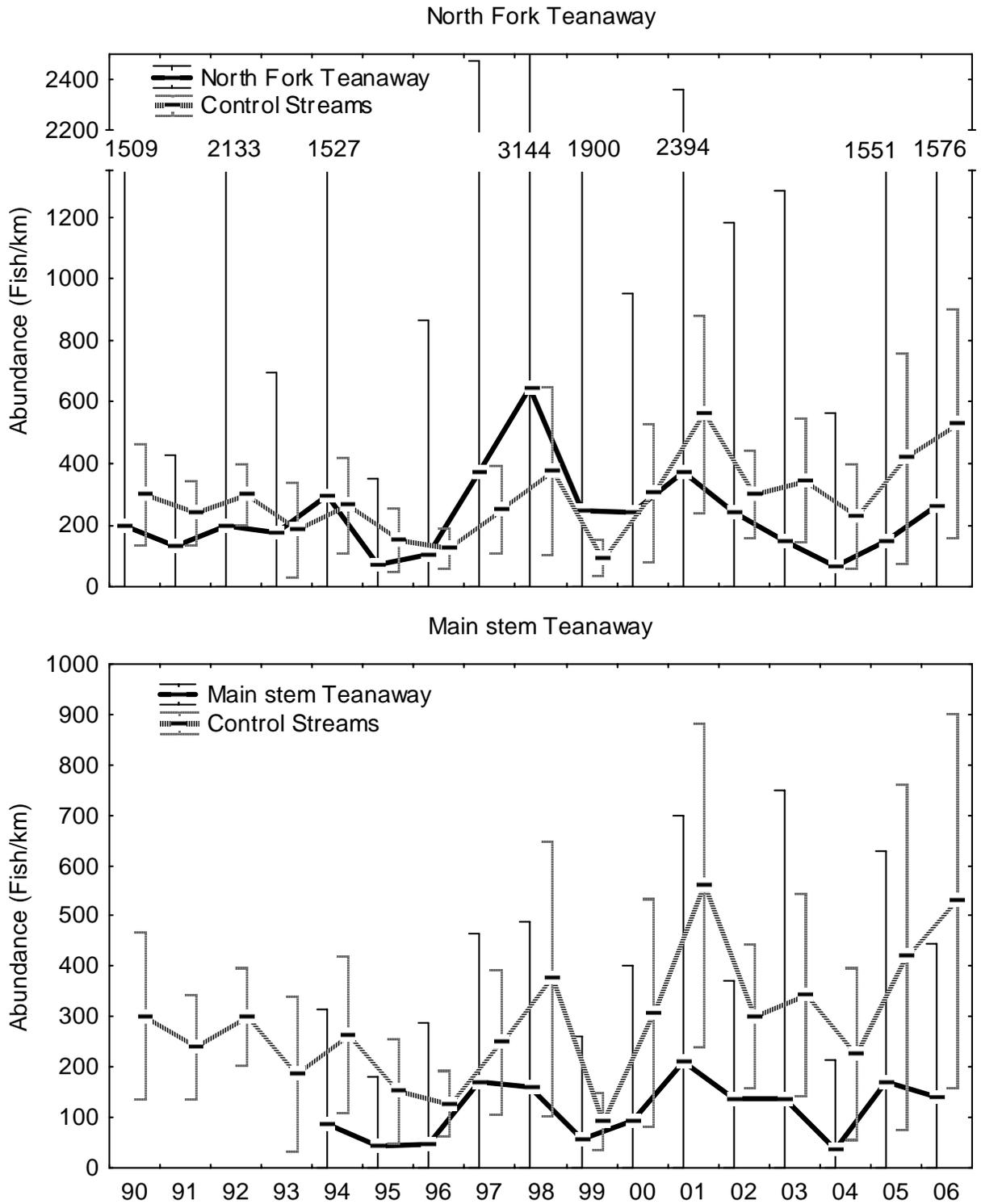


Figure 3. Abundance (fish/km) of rainbow trout in treatment and control areas in the Teanaway River Basin. Error bars represent the 95% confidence interval and negative lower limits were truncated to prevent negative values. Numbers above error bars are the upper limit associated with large confidence intervals.

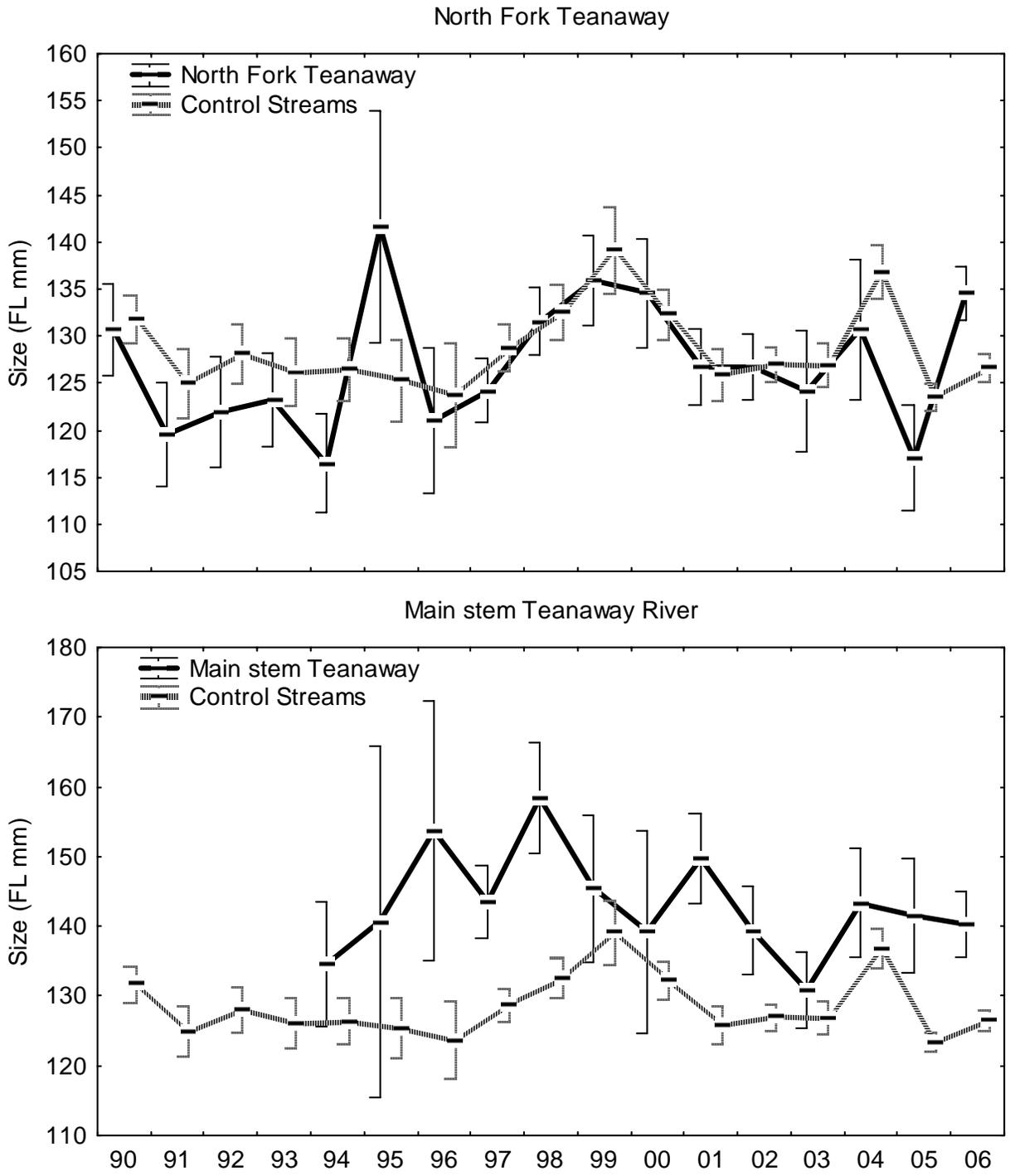


Figure 4. Size (FL mm) of rainbow trout in treatment and control areas in the Teanaway River Basin. Error bars represent the 95% confidence interval.

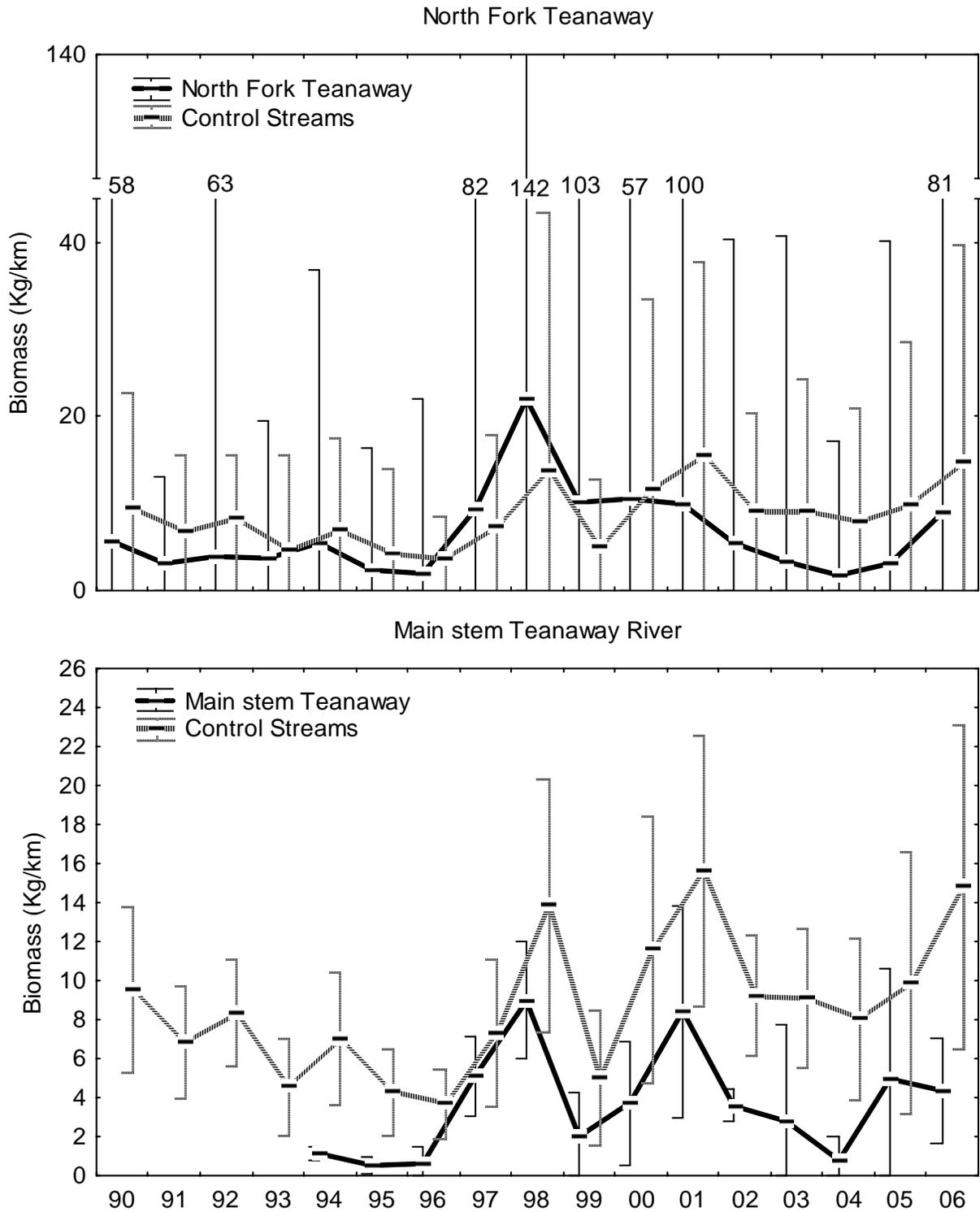


Figure 5. Biomass (kg/km) of rainbow trout in treatment and control areas in the Teanaway River Basin. Error bars represent the 95% confidence interval and negative lower limits were truncated to prevent negative values. Numbers above error bars are the upper limit associated with large confidence intervals.

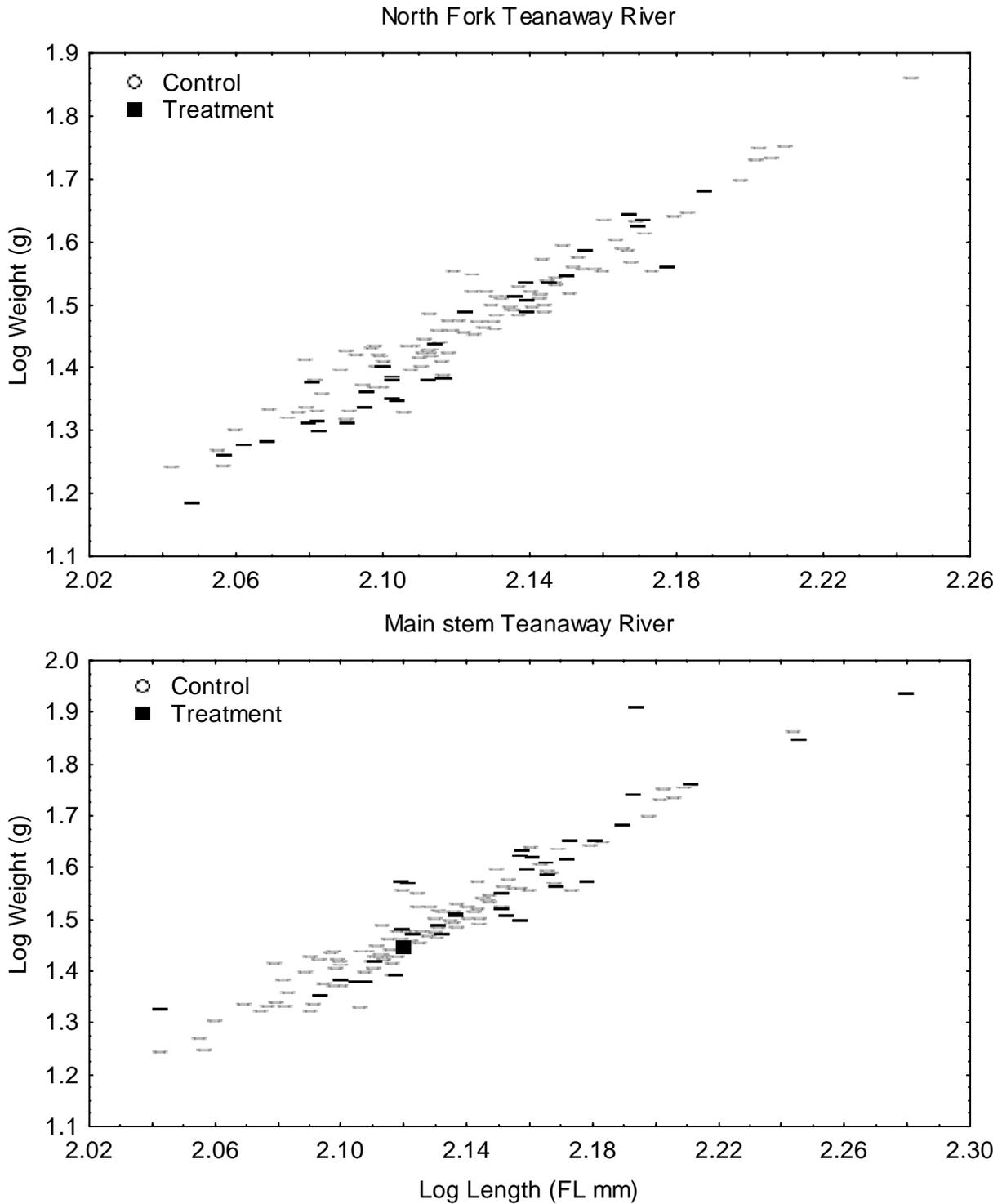


Figure 6. Mean length-weight relationships of rainbow trout in treatment and control sites Teanaway Basin. Each data point represents the mean from a sample site.

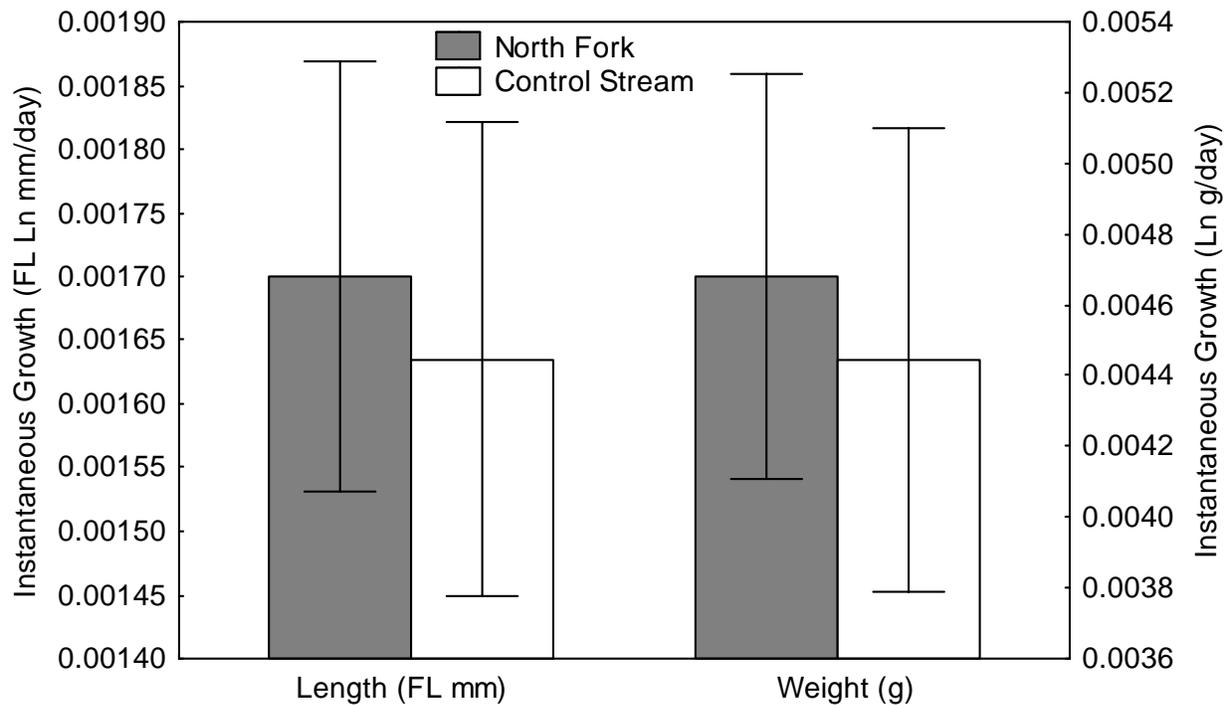


Figure 7. Instantaneous growth (per day) in fork length (FL) and weight (g) of PIT tagged rainbow trout in the North Fork Teanaway River and in an adjacent control stream (Middle Fork Teanaway River). Values presented were natural logarithm (ln) transformed. Instantaneous growth in weight is associated with the right y-axis and error bars represent the 95% confidence interval.

Negative correlations were detected between each measured spring Chinook life-stage and annual paired differences of rainbow trout in either the North Fork or main stem Teanaway River (Figure 8). There was a significant negative correlation between the annual paired differences of spring Chinook parr abundance and rainbow trout in treatment and control streams for the North Fork Teanaway River (Figure 8; $R^2=0.41$; $P=0.005$). There was also a marginally significant negative correlation between the annual paired differences of Hatchery origin spring Chinook smolts released from the Jack Creek Acclimation Facility and rainbow trout in the North Fork Teanaway River (Figure 8; $R^2=0.23$; $P=0.05$), but there was not a significant correlation between residual hatchery Chinook salmon abundance and annual paired differences of rainbow trout in treatment and control streams (Figure 8). In contrast, in the main stem Teanaway River, the only significant negative correlation was between the annual paired differences of spring Chinook residual abundance and rainbow trout (Figure 8; $R^2=0.32$; $P=0.04$). However, one data point (in the low left corner of Figure 8) kept parr from being significantly correlated with differences in rainbow trout abundance in the main stem.

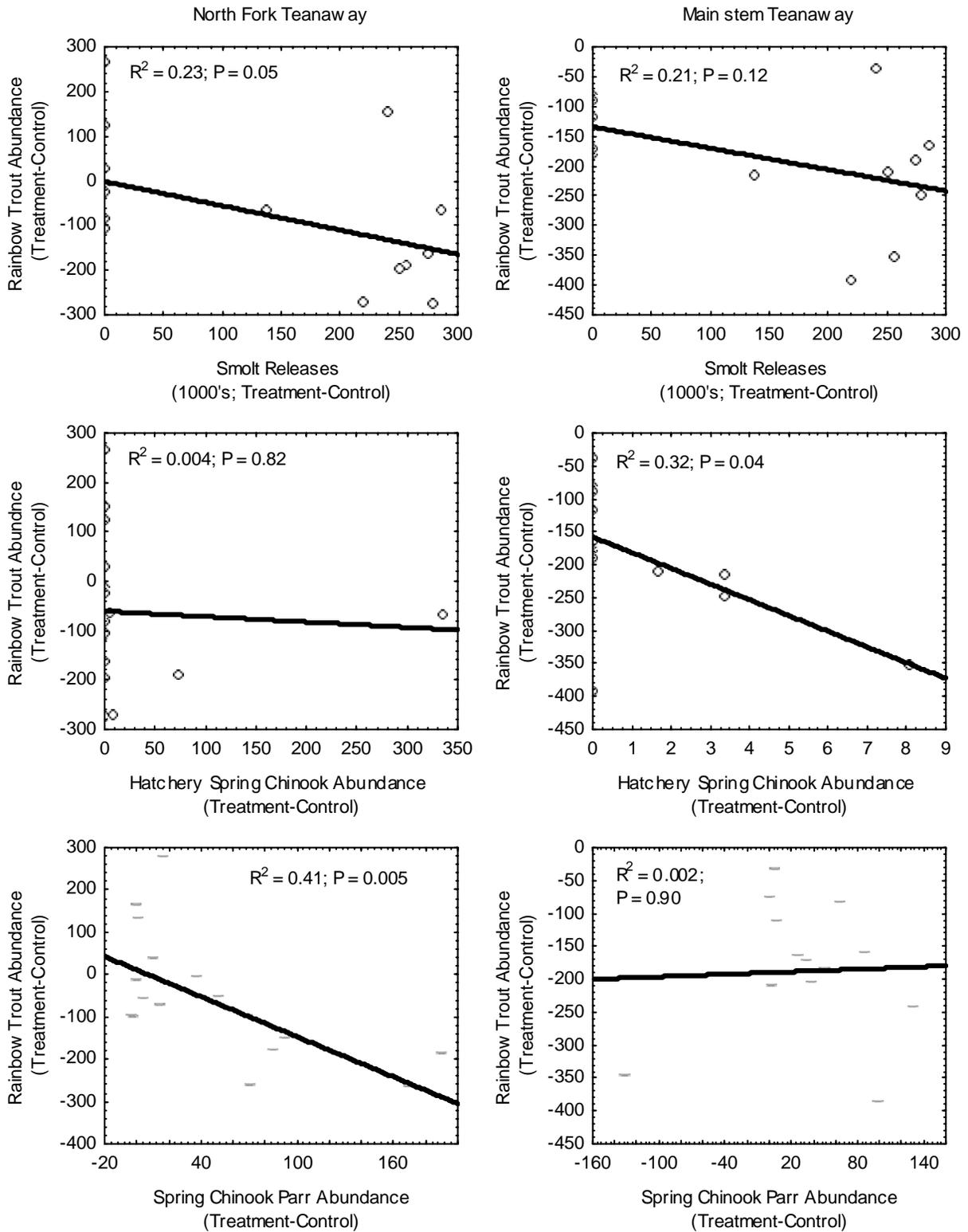


Figure 8. Correlations between annual differences in treatment minus control abundance of hatchery smolts, residuals, and parr versus annual differences in treatment minus control abundance of rainbow trout.

Combined biomass of spring Chinook salmon parr and rainbow trout in treatment locations decreased relative to control sites during supplementation (Table 6). There was also a significant difference between treatment locations of the North Fork Teanaway and Main stem Teanaway (Table 6). Differences in salmonid biomass between treatment and control areas in the North Fork Teanaway during supplementation were 3.39 times less than the quantity before supplementation (paired differences of treatment-control were -709.8 g/km before and -3113.6 g/km during) and in the main stem Teanaway they were 69% less (paired differences of treatment-control were -3814.7 g/km before and -6439.4 g/km during).

Table 6. Analysis of variance results for paired differences of combined biomass of spring Chinook parr and rainbow trout.

	SS	DF	MS	F	p
Intercept	3.53E+08	1	3.53E+08	3.53E+01	2.88E-06
Stream	4.51E+07	1	4.51E+07	4.50E+00	4.36E-02
Period	7.37E+07	1	7.37E+07	7.36E+00	1.17E-02
Stream*Period	8.70E+04	1	8.70E+04	8.69E-03	9.26E-01
Error	2.60E+08	26	1.00E+07		

Discussion

The impacts to rainbow trout that we observed were likely the result of cumulative impacts from hatchery released Chinook smolts, residuals, and an increase in naturally produced parr. Each one of these life-stages was significantly correlated with differences in trout abundance in at least one treatment location. Furthermore, coho salmon parr, residuals, or adults were rarely observed in the Teanaway Basin and were unlikely to have posed more than a single year of interaction during the one year of release. Age 1 rainbow trout were likely the age class that was impacted most because they were the most abundant age of fish that we sampled and they were the smallest fish that we sampled. Age 0 fish aren't sampled until the following year because they are too small to sample effectively with our gear. Impacts to age 0 fish would be reflected in the abundance of age 1 fish the following year of sampling.

Impacts in the site closest to the release site (North Fork Teanaway) were impacted more than those that were far downstream (main stem Teanaway River). Differential impacts were likely the result of stronger dosages of the treatment. The residence time of hatchery smolts was likely to be much longer in the North Fork Teanaway than in the main stem. Furthermore, it is likely that some smolts died before entering the main stem Teanaway and thus did not have the potential to interact with fish in this area.

Previous competition experiments in the Teanaway and nearby tributaries indicated that Chinook salmon parr did not impact growth or survival of rainbow trout (McMichael and Pearsons 1998). Our current evaluation of growth impacts is consistent with previous work, but abundance impacts are contradictory with previous work. Conflicting results between this study and previous work are likely to be due to the differences in abundance and life-stages of spring Chinook salmon during treatments and the length of time that we were able to evaluate

interactions. In the previous work the additional density of salmon matched the density of rainbow trout (e.g., 1 to 1) and was restricted to parr. In this study, the ratio of naturally produced spring Chinook parr to rainbow trout was lower (North Fork Teanaway River; 0.11 before to 0.59 during; main stem Teanaway River; 0.26 before to 0.61 during, and the control sites 0.05 before to 0.13 during) during the supplementation period. Our previous small-scale studies were primarily designed to evaluate growth, our experiments were less than four months long, and did not include effects of abundant smolts or residuals.

The lack of detectable impact to growth and the impact to abundance was consistent with the hypothesis reported by Grant and Imre (2005). They provided support for the hypothesis that density-dependent growth of stream-dwelling salmonids is stronger at low population densities rather than at high densities, probably due to exploitative competition. Furthermore, they speculated that density-dependent mortality and emigration was strongest at high densities, probably due to interference competition. Although our findings of impacts match their predictions, the density of salmonids in our study was relatively low (<1 fish m^2) except during the time of smolt emigration. It is possible that low food abundance and adequate habitat may limit the portions of the stream that are suitable for rearing and limit the carrying capacity. In short, the percent habitat saturation of our streams may be high.

Detection of results after 8 years of salmon releases was likely due to increased statistical power and increases in the impacts of Chinook salmon. In this study, years served as replicates, and increasing the number of replicates is one way to increase statistical power (Cohen 1988). Similarly, it is easier to detect significant differences with large effect sizes as opposed to small ones (Cohen 1988). Earlier study in the Teanaway did not reveal significant impacts after five years of salmon releases (Pearsons and Temple 2007). This study reveals the importance of studying impacts throughout the life-stages of a supplementation program (Pearsons 2002) and of a sufficient duration to have adequate statistical power to detect impacts (Ham and Pearsons 2000; 2001). If we had stopped our evaluation after the initial stage of supplementation, then we would have falsely concluded that impacts would not occur in later stages of supplementation.

The combined efficiency of rainbow trout and spring Chinook salmon to use resources in the Teanaway appeared to be hampered by supplementing spring Chinook salmon. We observed that the combined biomass of rearing rainbow trout and spring Chinook salmon was impacted by supplementing spring Chinook salmon. We do not know whether this combined inefficiency is due to poorer efficiency of spring Chinook salmon to convert resources into biomass (e.g., it takes more resources to support Chinook salmon than rainbow trout) or whether increased density of Chinook salmon results in higher rates of rainbow trout emigration. Domesticated Chinook salmon may be more inefficient at converting resources into biomass than natural origin Chinook. For example, progeny of Chinook salmon that spent a portion of one generation in the hatchery have been shown to survive at 2.2% lower rates than wild Chinook salmon when exposed to predators (Fritts et al. 2007). Furthermore, in competitive dominance experiments, progeny of wild fish grew 143% better in contest competition than progeny of fish that spent a portion of one generation in the hatchery (from Table 5 in Pearsons et al. 2007).

Recapture of tagged fish suggests that emigration of rainbow trout was unlikely to explain our results. Although movement of rainbow trout in the Teanaway Basin is likely to occur, we have evidence that most resident fish movement is local. Data collected during trapping operations revealed low levels of largescale movement (Pearsons et al. 1996). Fish that we tag are rarely captured far from the tagging location and never collected in other tributary sites. An unknown proportion of *O. mykiss* in the Teanaway are anadromous. We have detected

some PIT tagged fish at downstream dams during the spring following tagging. However, differential proportions of migrants does not explain our results because the ratios of migrant to tagged fish is similar between the treatment and control sites. We do not have similar data for Chinook salmon, but if they do migrate into the Yakima River, then it is unlikely that they contribute much to smolt production following years of high adult escapement. This is likely because survival is strongly density dependent at the escapements that have occurred during the years of supplementation (Pearsons et al. 2007). Furthermore, the proportion of natural origin fish that spawn in the Teanaway has been low suggesting that survival of naturally produced fish to the adult stage is very low. Wild Chinook salmon also did not emigrate at differential rates when exposed to hatchery Chinook salmon in enclosures (Weber and Fausch 2005). Regardless of whether fish emigrated from the Teanaway or died, the ability of the Teanaway to support salmonid biomass during the summer was decreased during supplementation.

We examined three alternative hypotheses, besides supplementation, that might explain the observed impacts in treatment and control areas. In order for these hypotheses to be reasonable, they had to explain the differences in differences that we observed. First, we examined whether environmental variables changed differently in the treatment and control areas. The only variable that we measured that was significantly different was mean stream width. This variable might be particularly important if it was associated with an increase in stream discharge or an increase in depth. Neither of these scenarios occurred. Furthermore, we compared the differences in mean width between treatment and control streams to differences in rainbow trout abundances, similar to our methods for evaluating Chinook salmon life-stage impacts, and we did not detect any significant correlations. We also looked at historic and current photos to determine any major changes in riparian vegetation or channel structure. Most photos were remarkably similar between the two periods. In short, environmental differences did not appear to explain the differences in rainbow trout abundance that we observed.

In contrast, it is likely that environmental variables that we did not measure had a positive effect on rainbow trout abundance. Rainbow trout abundance increased 42% in the control streams, in 2 of 3 other unsupplemented tributary streams that we monitor in the upper Yakima Basin, and in the main stem Yakima River. Furthermore, cutthroat trout abundance increased 90% in a site in the North Fork Teanaway that is upstream of the salmon release location. These data suggest that environmental conditions at a scale larger than our study area are positively impacting trout abundance.

The second alternative hypothesis that we examined was differential angling pressure in the treatment and control streams. The angling regulations in all of our study streams were identical prior to 1998 and the only difference from 1998 to the present was that only artificial lures and flies (no bait) with a single barbless hook could be used in treatment streams. Bait and multiple hooks were allowed in control streams. The duration of the fishing season was between June 1 and October 31 during all years in both treatment and control streams. Prior to 1994, 8 trout could be harvested (not more than 2 above 12 inches, no minimum size). Beginning in 1994, only 2 trout could be harvested and the minimum size for retention was 8 inches. The most dramatic change in harvest regulations, which occurred in both control and treatment streams during 1994, was temporally unassociated with increases in abundance or changes in size. This suggests that the angling regulations have not had much impact on rainbow trout abundance. Unfortunately, angler surveys have not been conducted during the study period. The only data that we have that pertains to angling pressure is the percent of fish in our streams that have scars that are caused by hooking (e.g., holes in the mouth, maxillary damage, presence of a

hook). These percentages were remarkably similar among the different streams (Figure 9). In summary, evidence indicates that differential angling pressure do not explain the impacts we observed.

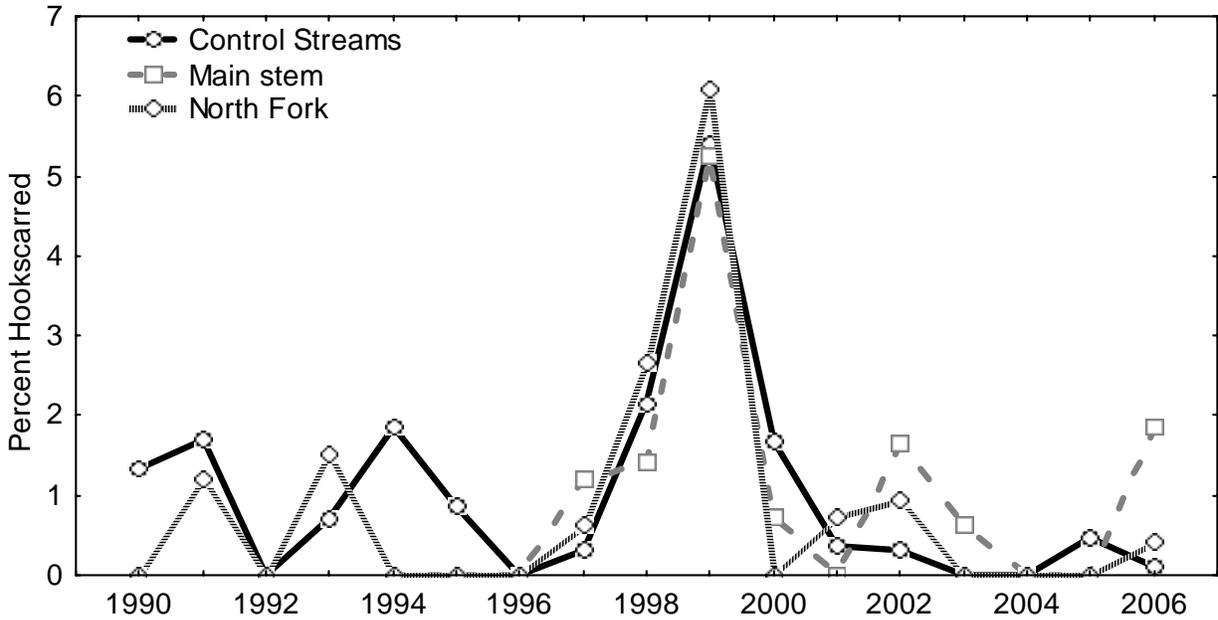


Figure 9. Percent of rainbow trout with visible hookscarring observed during sampling in treatment (Main stem and North Fork Teanaway Rivers) and control streams (Middle Fork and West Fork Teanaway Rivers) in the Teanaway Basin.

Evaluating the supplementation program using the two most valued species in the study area provided conflicting views of the success of the program, than if we had only evaluated the species supplemented. If we had focused our evaluation exclusively on the spring Chinook salmon, we would have concluded that supplementation was successful in increasing natural production of summer parr; a positive result consistent with the management goal. This result would suggest that no changes in management strategy would be necessary. However, the large impacts to rainbow trout and the decrease in combined salmonid abundance suggests that the current supplementation program in the Teanaway basin is not successful. Rainbow trout in the Teanaway are a combination of resident rainbow trout and juvenile steelhead (Pearsons et al. 2007). Steelhead are listed as threatened in the Mid-Columbia (includes the Yakima Basin) and because of their low abundance, individuals could be considered more valuable than the more numerous spring Chinook salmon. If steelhead individuals are more valued than Chinook salmon, then the cost to steelhead could be viewed as higher than the benefit to Chinook salmon. Furthermore, the reduction in salmonid biomass would be viewed as a detriment even if the value of the two species were identical.

This study highlights the importance of evaluating hatchery programs from a multi-species perspective. Single species management approaches have been reported to be ineffective in other resource management disciplines such as forestry for reasons similar to the one that we have identified in this study. We recommend that a broader taxonomic and ecological perspective be used when evaluating and managing hatcheries.

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Chapter 4

Comparison of error rates between four methods of data entry; Implications to precision of fish population metrics

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Abstract

The objective of our study was to determine the magnitude of data entry error present in a typical fisheries field station, how those errors might affect common fisheries estimates if undetected, and what method of data entry would be most proficient in preventing errors for the lowest cost. We evaluated four methods of data entry and compared common fisheries estimates derived from each to estimates from verified data. We found no significant difference between estimates generated from any of the four data entry methods and estimates from our verified data. This suggests that a single data entry, or the use of a personal digital assistant (PDA) is sufficient for similar data entry applications. This study provides information that should lower concerns surrounding the use of data collected under different data entry protocols.

Introduction

Fisheries practitioners are often required to provide estimates (e.g., abundance, size structure, assemblage structure) of the highest possible quality, and for the lowest possible cost. This involves reducing the amount of sampling error so that the “true” value of the population can be approximated. Sampling error can occur at a variety of stages such as: during the planning of the sampling design, field sampling, and in the process of data entry (Figure 1). Books have been written on how to improve sampling designs (Schreck and Moyle 1990; Murphy and Willis 1996) and on how to improve field-sampling protocols through standardization of field methods (Johnson et al. 2007). But the subject of data entry error and how it may affect these estimates has been largely overlooked. Protocols to reduce error typically involve the implementation of error checking methods such as range-checking procedures, or more time consuming and presumably more effective measures that involve comparing each entry to the original (Brown and Austen 1996). One of the most frequently used methods is the “read-aloud” method. In the read-aloud method, a single computer entry is printed to hard copy and then read out loud. A second technician simultaneously follows along comparing those values to the original filed data. Another is the “double-data-entry” method, where two separate entries of the same field data are entered into the computer, and then compared using a spreadsheet function. However, the relative benefits of these proofing methods, and their respective costs, have not been addressed.

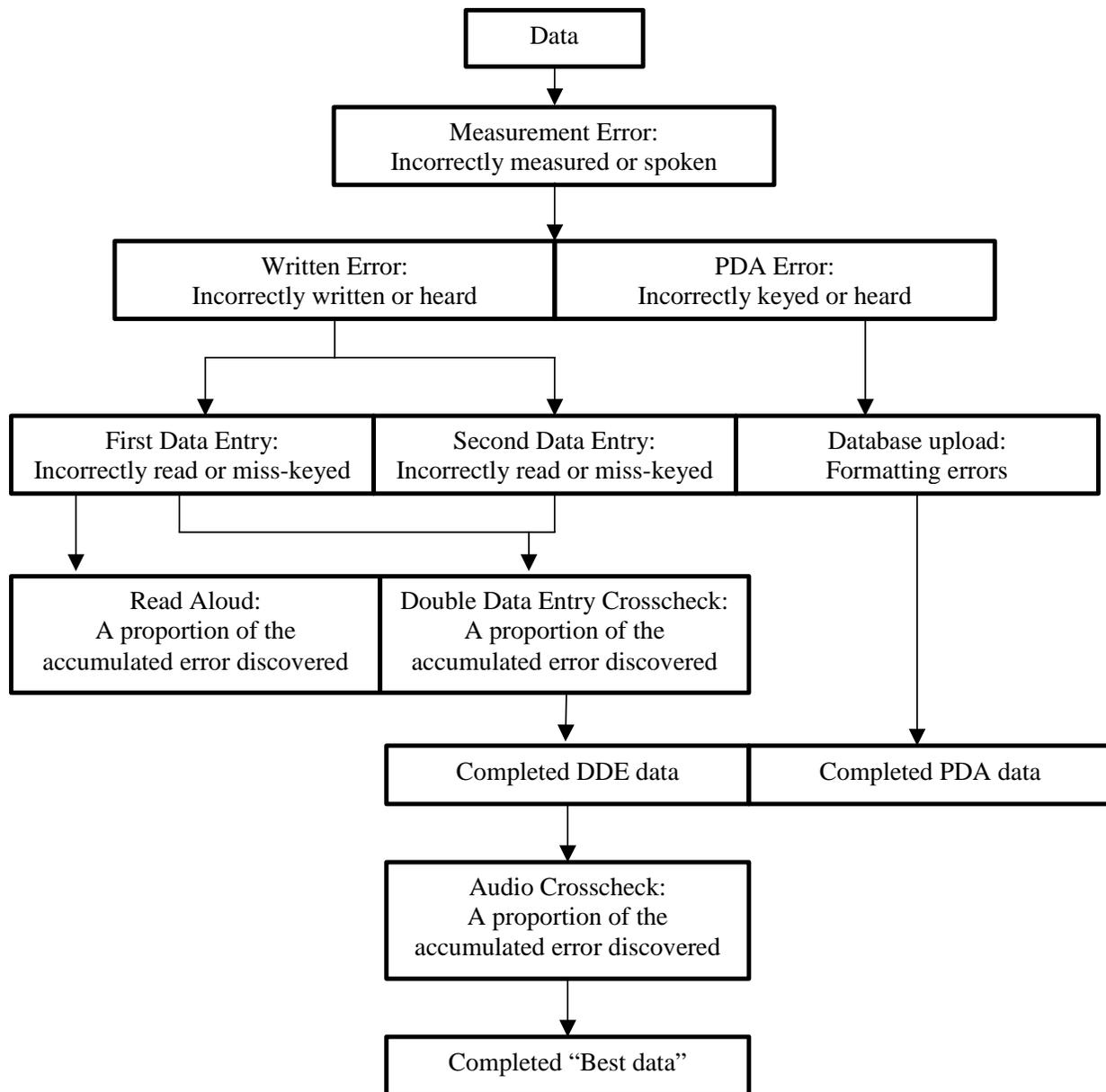


Figure 1. Flow of data and sources of error from the field to analysis. PDA = Personal Digital Assistant, DDE = the Double Data Entry method of error checking.

In contrast, medical scientists have realized the implications of data entry error and have published articles reporting rates of data entry error, evaluating methods of error checking, and assessing the potential impact of undiscovered errors that remain in their datasets. Kawado et al. (2003) found that double-data-entry was superior to read-aloud checks but reported error rates of only 0.37 percent in their unchecked data. Gibson et al. (1994) and Day et al. (1998) both acknowledged that double-data-entry can be effective in discovering entry error, but found that error rates were low enough in some clinical trials that double-data-entry was unnecessary. Further, Neaton et al. (1990) and Shenker (1989) reported error rates as low as 0.10 percent; and

0.07 percent respectively, which further supports the notion that data error rates may be so low that they are inconsequential in the desired analyses.

The results of such studies may not be directly applicable to the field of fisheries, where data are often collected under conditions that are far less controlled. For instance, inclement weather, inhospitable sampling locations, and crew fatigue may all affect the fisheries practitioner's ability to record data in the field, or efficiently enter data after returning to the office. Furthermore, estimates that require modeled expansions may pose larger risks than calculations of means.

There are many trade-offs to designing and implementing studies to produce the best available estimates of population parameters (Temple and Pearsons 2007). Most if not all fisheries practitioners are limited by the amount of time and money that is available to collect data. Excessive effort eliminating data entry errors directly impacts productivity and reduces effort that may be better spent in other areas (e.g., increasing sample size). Conversely, failing to reduce data entry error when warranted may limit the researcher's ability to draw valid conclusions from the field data. Further, the level of effort required depends upon the level of accuracy required in the estimate. For instance, a mark-recapture abundance estimate is more likely to be adversely affected by a missing mark code, than an estimate of average fish size is by incorrectly recording a fish length.

Some fisheries practitioners have adopted the use of personal digital assistants (PDAs) as an alternative to more traditional data entry protocols (Hollowell 2003; Lang et al. 2006). These compact, battery-operated devices allow field data to be entered directly into electronic form, thereby avoiding the data entry process necessary with hand-written data. Advantages of PDA use include the elimination of errors resulting from poor penmanship, reduction of transcription and measurement error through the use of real-time range checks or data entry restraints, and immediate availability of the data for analysis. However, short battery life, difficulty in viewing the computer screen under varying light conditions, the propensity for data loss, inflexibility to enter different data types, and steep learning curves, may be perceived as drawbacks that hinder the effectiveness of PDAs in fisheries fieldwork. Despite potential drawbacks, the use of PDAs in the field has been advocated in a variety of scientific applications (Green 2001; Fletcher et al 2003; Ice 2004). Again, the results from these studies may not apply directly to fisheries research, where sampling environments are less controlled. Hollowell (2003) presented a preliminary assessment of PDAs for use in recording creel data. However, a further evaluation of how and to what degree data obtained from a PDA differ from hand-written data, and how those data may affect fisheries estimates is needed.

The objective of our study was to determine the magnitude of error present in a typical fisheries field station. Furthermore, we wanted to determine where within our dataset errors occur most often, and how they might affect commonly generated estimates of abundance, size structure, and species richness if they were not prevented through our standard data checking protocols. We wished to know what methods of data entry would be most proficient in preventing errors, and for the lowest cost. To address these questions, we evaluated four common methods of data entry; a single entry of the hard-copy field data (SDE), a read aloud check of that single entry (RA), an independent second entry and computer crosscheck of those data (DDE), and lastly, those same data recorded in the field using a personal digital assistant (PDA). We determined the level of error present in data, after each error checking method had been performed, by direct comparison with data validated using audio recordings. We then compared estimates of abundance, size, and species richness from data proofed by each method

to those from the validated data. Finally, we created a series of logical tests in the form of VBA macros, and evaluated their effectiveness in providing minimum levels of data quality after only a single entry of the data or a single entry onto a PDA. We provide a framework for others to use in evaluating data entry error, and recommend a protocol that provides sufficient data entry error detection for the lowest cost for similar type datasets.

Methods

Data were collected from July through September of 2004 and 2005 as part of a long-term evaluation program, which provides salmonid population estimates in tributaries of the upper Yakima River in central Washington State. Detailed collection protocol is outlined in Temple and Pearsons (2007). Briefly, salmonids were netted in one of two electrofishing passes within 100m long tributary sites. Captured salmonids were anesthetized, examined, measured, marked, and subsequently released after being provided time to recover. A portion of those that were captured in the first electrofishing pass, were recaptured in the second pass and the presence of a mark was recorded. A single technician made all necessary morphological measurements, conducted observations of fish condition, and then clearly communicated the information to a second technician. The second technician then transcribed those data onto a hard-copy datasheet (Figure 2). In addition to the standard protocol, data were also concurrently entered into a Palm[®] personal digital assistant (PDA) (Figure 3), and recorded with a compact audiocassette recorder. The types of data recorded and transcribed varied from yes or no check-fields, to lengths and weights of fish, and comments documenting infrequent events and/or specimen condition. A list of data collected is presented in Table 1. Throughout the duration of the study, technicians and biologists of varying field experience and computer skills took equal turns collecting and entering data.

Table 1. Data collected during 2004 and 2005 electrofishing surveys “Non-target” refers to non-salmonid fish species, from which only rudimentary data were collected

Electrofishing Pass	Green	Eye Injury Left
Species Type	Ripe	Eye Injury Right
Mark Code	Spent	External Parasites Fins
Mortality	Eroded Caudal Upper	External Parasites Gills
E-fishing Time (seconds)	Eroded Caudal Lower	Site Segment
Age (Non-target)	Injury Hooking	Length
# (Non-target)	Injury Other	Weight
Bruised	Hook Scar	Electrofishing Injury
Orange Slash	Bird Scar	Field Comments
Black Spot	Other Scar	

The image shows two side-by-side screenshots of a PDA datasheet interface. Both screens have a title bar labeled "TribZ".

Left Screen:

- STREAM**: A dropdown menu with a downward arrow and the text "Select one..."
- DATE**: A field containing "- Set Date -" with a dotted border.
- PASS**: A row of five numbered buttons (1, 2, 3, 4, 5).
- SECTION**: A dropdown menu with a downward arrow and the text "Select one..."
- SEGMENT**: A dropdown menu with a downward arrow and the text "Select one..."
- At the bottom are three buttons: "End", "Previous", and "Next".

Right Screen:

- SPECIES**: A dropdown menu with a downward arrow and the text "Select one..."
- LENGTH**: A field with a dotted border.
- WEIGHT**: A field with a dotted border.
- Below the WEIGHT field is a numeric keypad:

+ -	7	8	9
Del	4	5	6
00	1	2	3
	000	0	.
- At the bottom are three buttons: "End", "Previous", and "Next".

Figure 3. Example of a PDA datasheet. Down arrows are drop down menus.

Field data were returned to the office daily. Hardcopies of the data were organized and stored within binders before they were hand-entered into Microsoft Excel workbooks as time allowed throughout the season. Data from the PDA were uploaded by "fire wire" to a Microsoft Access 2000 database. No attempt was made to evaluate the potential for errors within the dataset specifically due to the uploading of data from the PDA to an Access database file. We make the assumption that such error is insignificant when compared to that of human error. The uploaded data were then exported to Microsoft Excel. Audio recordings were stored on their original cassette tapes and in their original cases until the end of the field season, when they could be reviewed.

As time allowed, technicians possessing varying computer skills entered data from the original hardcopy into electronic format (single data entry). Technicians were often fatigued as a result of having worked a number of hours in the field, distracted by responsibilities that often interrupted their data entry tasks, or allowed only short periods for data entry before the end of their shift; typical conditions within an operating field station. Data were entered into Microsoft Excel spreadsheets that had been formatted to closely resemble field datasheets (Figure 2). As is common when entering data into Microsoft Excel, our standard template spreadsheet contained rudimentary restrictions for species code and check box fields, (i.e. the user could only enter one of several predetermined species codes, and could only enter "check" or "no check" entries into cells that indicated fish condition). Users also had access to features that allowed "dragging" of values in order to replicate them down a column or across a row, and "auto-completion," which allowed the user to enter only the first letter of a recently used text value before moving to the

next cell in the spreadsheet. These data, now in both original hardcopy and electronic form, were then used as base data for the error check methods listed below.

The read aloud method compared the original hardcopy field datasheet to a printed copy of a single electronic entry. One technician read the data hard copy aloud, while another checked it visually against the electronic print out. Discovered errors were highlighted on the print out and subsequently corrected in the electronic version. Read aloud checks were performed in a closed and quiet room where distractions were minimal.

The double data entry method of error checking compared directly, with the aid of a spreadsheet function, cell by cell differences between two independent computer entries of the same data. Data were entered a second time into the worksheet template. We required that different technicians perform the original and second computer data entry. This was done to reduce bias and/or repeated error. A third worksheet containing a crosschecking function directly compared the cells within the initial and second entry worksheets and placed a highlighted error code where there were differences between the two. Discovered errors are normally checked against the field copy and corrected. For the purposes of this study, the address of each cell containing an error was recorded before corrections were made.

We used our audio recordings to create a benchmark dataset against which we could compare our other methods of data entry error. Using the corrected data from our double-data entry as the base data, each cell entry in the dataset was then compared to its respective audio recording. This allowed us to further refine the dataset by discovering and correcting both data entry errors not previously discovered, and to a lesser degree, transcription errors made in the field. Audiotapes were reviewed in their entirety while the reviewer followed along concurrently viewing the electronic entry. In order to reduce the likelihood of misinterpretation, only experienced technicians served as reviewers. When areas of the tape were inaudible, corresponding sections within the electronic copy were highlighted in grey to prevent their inclusion in subsequent comparisons. Discovered errors were highlighted in yellow, their position within the dataset noted, and then corrected. Corrections were only made when the errors discovered were indisputable. These data in their final form were considered our “best” data.

Data were organized by year, sampling location, and error check method. Each sampling location within a year was considered a replicate. There were a total of 24 replicates in 2004, with the number of cell entries ranging from 210 to 1336, ($X: 540.9$, $SD: 300.9$); and 25 replicates in 2005, with the number of cell entries ranging from 100 to 1548, ($X: 769.2$, $SD: 433.5$).

We performed a series of automated logical tests and outlier regression routines on our single entry and PDA datasets to evaluate the effectiveness of automated checking procedures in providing data that meet minimum data quality standards. Tests were designed to target specific error types known to have occurred within our datasets in previous years. These tests were then incorporated into a series of Excel Visual BASIC for Applications (VBA) macros. Lastly, a single macro was assembled to run the various check routines and generate a list of results directing the user to any areas of concern in the datasets. Descriptions of the various check routines, and the error types they were designed to prevent are provided in Table 2.

It would have been favorable to measure the elapsed time of data entry during our study so that we could calculate effort. However, due to the sporadic opportunities allowed for data entry throughout the field season, no definitive measure of effort was made for the

implementation of the various data entry methods. As an alternative, estimates of required effort for each method of data entry were made through post entry interviews.

Table 2. Description of the computer automated data checks performed by the VBA error checking macro.

Module	Error check	Description
1	Formatting errors	Checked datasheet format
2	Misplacement & unique circumstances	Checked comment fields against a compiled list of keywords that might have signified movement between sections, mark mortality, misplaced mark codes, etc
3	Missing data	Checked for missing data in critical areas within the dataset
4	Drag errors	Checked pass, time and segment columns for inconsistencies
5	Categorical errors	Checked for multiple initial marks in the same pass and segment
6	Numerical errors	Flagged salmonid length and weight outliers

Analysis

We evaluated the magnitude of error remaining in the data after implementing each of the four methods of data entry. This was accomplished by comparing the data from each method to that of our audio-reviewed, “best” data. The number of outstanding errors in each replicate was recorded, and that number then divided by the total number of entries in the dataset. This proportion was then subtracted from the proportion of total error discovered through our review of the audio recordings. This new value, the proportion of error still remaining in the data, was then multiplied by one hundred in order to present the value as a percentage of the dataset.

In order to determine the data types that resulted in the highest percentage of error, data were categorized into six distinctive entry types specific to what was required of the technician to enter the data into the computer. “Drag” type data were those that only required one physical entry, which was then dragged down a number of cells using the auto-fill function within Microsoft Excel. “Missing data” were blank entries where a value should have been present. “Numerical” and “Categorical” data types required either a numeric or alphabetic entry respectively. “Misplacement” errors were those in which data were transposed, often resulting in two errors within the dataset. Lastly, “Other” errors were those which fell outside the range of the preceding categories such as missed check boxes, missing comments in a text field, and other miscellaneous errors. For the PDA datasets, entries that resulted in broad errors, over multiple entries (e.g. pass, time, section) were considered Drag-type errors.

We evaluated the use of a PDA as a possible alternative to standard data entry methods. Because a PDA can electronically upload field data, the opportunity for introduced error during the hard copy to electronic copy conversion is eliminated. However, it was unknown what level of error existed in the form of transcription error while in the field. We compared data collected with the PDA to our audio-reviewed data in order to establish the percentage of otherwise detectable errors that would have remained in the dataset had we used a PDA exclusively in our

data collection efforts. Unlike the other methods of data entry, PDA entry had the potential of showing either higher, or lower levels of error when compared directly to our benchmark dataset.

We used two common methods of fish population abundance estimation, multiple-removal (Zippen 1958), and Petersen-type mark recapture (Chapman 1951), to evaluate what effect, if any, undiscovered errors in the dataset would have on our estimates. Because small portions of our audio recordings were inaudible, and because complete datasets by sampling site are required to generate a population estimate, we restricted our analysis to those sites that had the majority of the data backed by audio recordings. All replicates we chose to retain had greater than 93 percent of the data backed by audio recordings. In addition, only those that generated valid abundance estimates were used in the analysis. Valid estimates for multiple species within a given dataset (e.g. Rainbow and Cutthroat trout, Chinook salmon) were considered independent for the purposes of this evaluation.

Multiple removal estimates were generated for each replicate set, for each of the two years using the Microfish computer program (Van Deventer and Platts 1989). The number of replicates for each data entry method was 23 in 2004 and 29 in 2005. In order to maintain the highest level of consistency when generating estimates, and also to expedite the process of generating the many estimates, we wrote a Microsoft Visual Basic (VBA) macro to organize, send data to, and retrieve data from the Microfish program. This macro was tested extensively prior to its application in the study. Petersen-type mark-recapture estimates were also calculated for each of the 29 replicates in 2005. There were no mark-recapture surveys conducted in 2004. A Visual Basic macro was also written to organize our data and to generate the estimates.

We generated metrics for both size and species richness to evaluate the potential effect undiscovered errors in the dataset might have on these common estimates. To avoid potential sampling bias in the estimates, we restricted our data to fish captured on the first electrofishing pass. Fish were categorized by sampling section, and species type. A count of species present and mean length for each was then generated. No attempt was made to segregate communities by age class prior to our calculations. Further, for the purpose of comparing datasets, we have considered each species present as an independent replicate. A Microsoft Visual Basic (VBA) macro was written to facilitate the large number of calculations necessary, and to maintain consistency in both size: 2004, $n = 179$; 2005, $n = 105$ and species composition calculations: 2004, $n = 185$; 2005, $n = 127$. This macro was tested extensively prior to its application in the study.

Statistical tests

Mann-Whitney U-tests were used to test for year-to-year differences in the error remaining in our data after each of the four data entry methods. Tests were considered significant when $P < 0.05$. When no difference was detected, data from the two years were pooled. Wilcoxon matched-pairs tests were then used to test for differences in error between data entry methods. We used a G-test to test the assumption that errors in our single entry and PDA data were equal between categorical types. When differences in a method's remaining error were discovered (described above), a G-test was used to compare the categorical distribution of error between years. Tests were considered significant when $P < 0.05$. We used paired t-tests to test for differences between population estimates generated from the four data entry methods and our audio-reviewed data. Data from 2004 and 2005 were pooled when

appropriate. These methods were also used in evaluating size and community structure from each of the evaluated data entry methods.

We calculated our statistical power to detect differences. We used the *Statistica Power Analysis* module for dependant sample t-tests (StatSoft 1999) to calculate our power in detecting differences between mean population estimates generated from our audio-reviewed data and each of the four data entry methods. Power was considered low when below 80 percent ($1-\beta < 0.80$). We used the following equation from (Zar 1999) to estimate the difference in means we could detect given $\alpha = .05$ and $\beta = .10$, and our current sample size:

$$\delta = \sqrt{\frac{s_d^2}{n}}(t_{\alpha, v} + t_{\beta(1), v})$$

Where s_d^2 is the difference in variance between means of a paired sample.

Results

We detected no year-to-year difference in the level of error remaining in the dataset using single, double, or read aloud methods of data entry ($P > 0.05$). These data were pooled in subsequent analyses. A small, but statistically significant difference was detected between the 2004 and 2005 PDA datasets ($\bar{X}_{2004} = 1.40$, $\bar{X}_{2005} = 1.43$), (Mann-Whitney U -test: $Z = -2.30$; $df = 23, 24$; $P = 0.02$). When compared to single entry ($\bar{X}_{04-05} = 1.16$), significant reductions in remaining error were detected in the read aloud ($\bar{X}_{04-05} = 0.89$) (Wilcoxon matched-pairs test: $Z = 5.01$; $df = 48$; $P < 0.001$) and double-data-entry methods of error detection ($\bar{X}_{04-05} = 0.44$; Wilcoxon matched-pairs test: $Z = 5.97$; $df = 48$; $P < 0.001$). Further, the double-data-entry method showed a significantly greater reduction in error than the read-aloud method (Wilcoxon matched-pairs test: $Z = 3.38$; $df = 48$; $P < .001$) (Figure 4).

We did not detect a significant difference in categorical error in our best data when compared to an equal distribution (G-test: $X^2 = 0.75$; $df = 5$, $P = 0.98$). Likewise, the data collected using the PDA did not differ significantly from an equal distribution in either year (2004 G-test: $X^2 = 8.94$; $df = 5$, $P = 0.11$), (2005 G-test: $X^2 = 1.27$; $df = 5$, $P = 0.94$), or from our benchmark dataset (2004 G-test: $X^2 = 8.48$; $df = 5$, $P = 0.13$), (2005 G-test: $X^2 = 0.67$; $df = 5$, $P = 0.98$) (Figure 5).

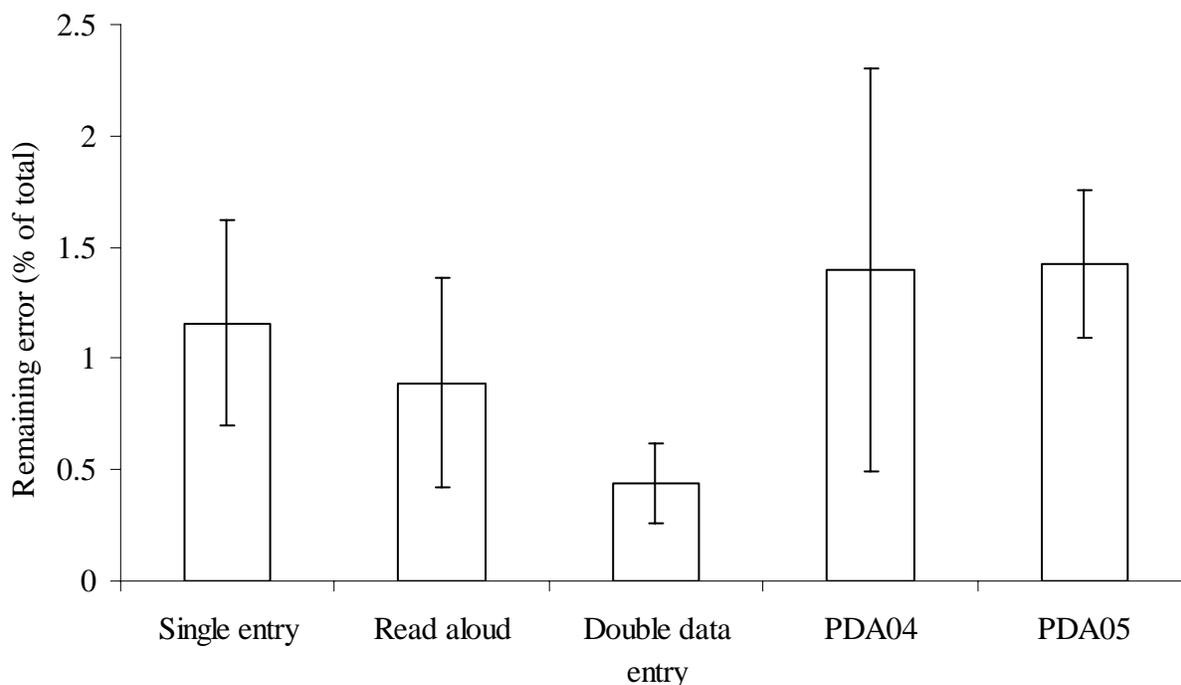


Figure 4. Percent of the dataset identified as entry error.

Multiple removal estimates generated from the four methods of data entry fell outside of Audio-review estimate confidence intervals (95% CI) in six of the fifty-two replicates (Figure 6) PDA data accounted for four of the six deviations, three due to missing species entries, and one due to species misidentification. Of the remaining two deviations, one can be attributed to writer error while in the field (hardcopy error), and the other, a single entry error, originated from a drag error during data entry. None of the estimates generated from the four evaluated methods of data entry differed significantly from those generated from our best data: SDE, DDE, RA (Paired t-test: $df = 52$, $P > 0.11$), PDA 2004 (Paired t-test: $df = 23$, $P = 0.16$), PDA 2005 (Paired t-test: $df = 29$, $P = 0.42$). Power to detect differences between estimates generated from our best data and those from each method of data entry was low ($1-\beta < 0.05$). However, we estimate that we had sufficient power ($\alpha = 0.05$, $1-\beta = 0.90$) to detect differences of 3.6 percent when comparing estimates between a single data entry and our audio-reviewed data, and 0.8 percent when comparing estimates generated from either read-aloud or double-data-entry datasets. Power to detect differences was also low when comparing estimates generated from PDA data to those from our audio reviewed data for both years ($1-\beta < 0.29$). We estimate that we had power ($\alpha = 0.05$, $1-\beta = 0.90$) to detect differences of 12.3 percent in 2004 and 5.2 percent in 2005. None of the Peterson-type estimates generated from the four methods of data fell outside the confidence intervals of the AR estimates.

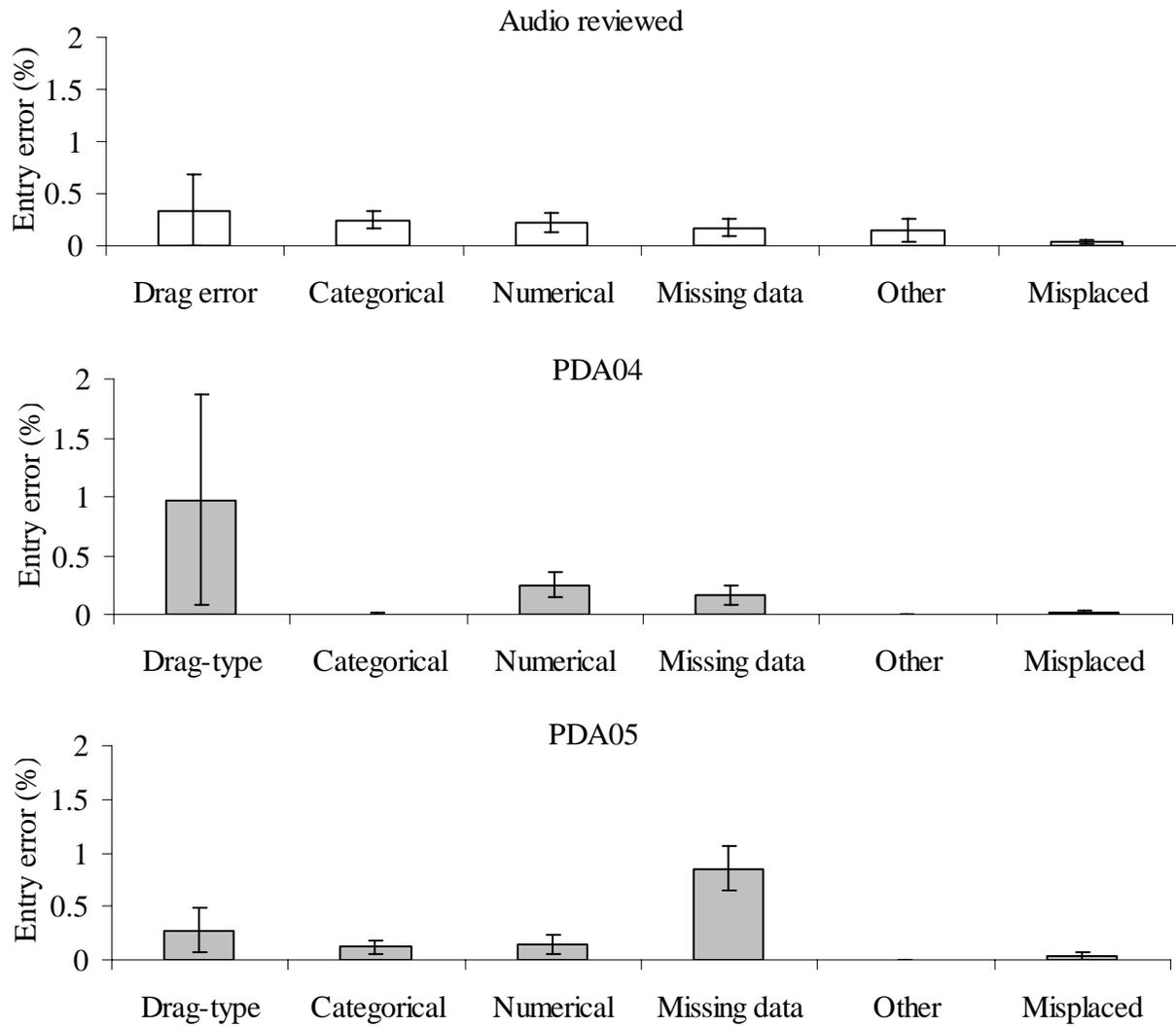


Figure 5. Percent ($\pm 2SE$) of the dataset identified as data entry error, by category for both audio reviewed data and data entered in the field using a PDA.

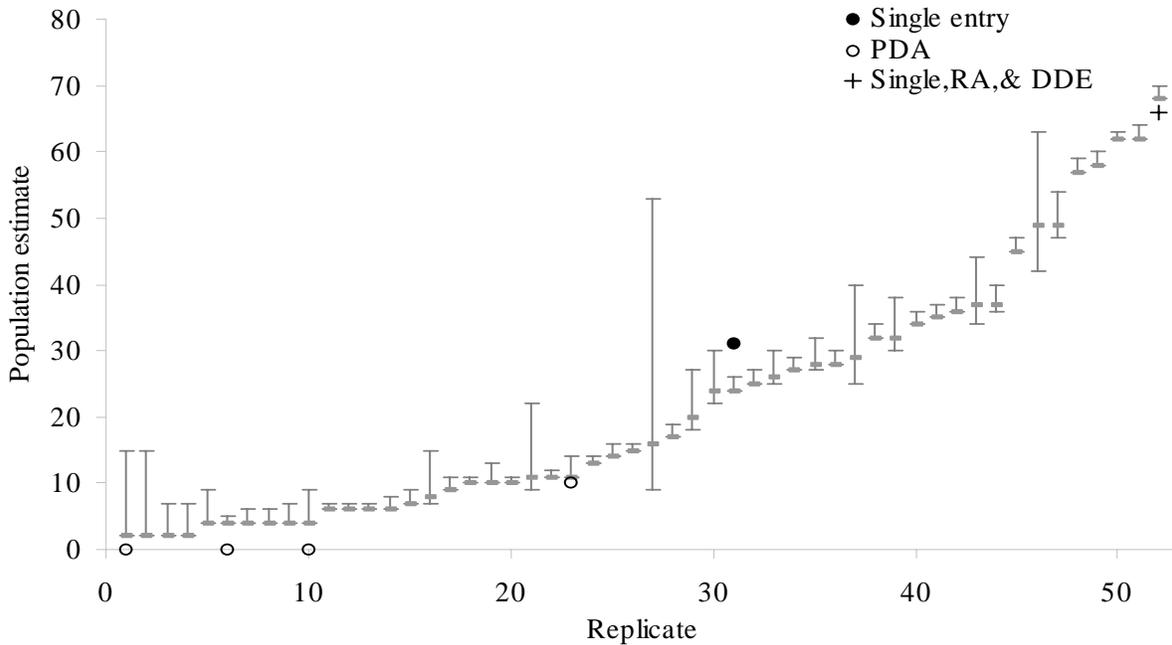


Figure 6. Multiple removal population estimates from 2004 and 2005. Estimates calculated from audio-reviewed data are shown in grey with ninety-five percent confidence intervals surrounding each. Deviations from the audio-reviewed data estimate, from any of the four entry methods are presented in black.

Estimates of fish size generated from the four methods of data entry fell outside of AR estimate confidence intervals (95% CI) in only 6 of 301 replicates (Figure 7). Four of the six deviations can be attributed to drag-type error in the PDA dataset. One of the two remaining deviations, single entry, resulted from a categorical error. The remaining deviation was common in all but the PDA datasets. This error was actually a written error in the field (an unrecorded length) and cannot be categorized as a data entry error. No significant differences were detected in mean species length generated from single entry, read-aloud, or double-data-entry methods and our audio-reviewed data (Paired t-test: $df = 280$, $P > 0.37$). Likewise, size estimates generated from the PDA data did not differ significantly from those generated from the audio-reviewed data in either year ($P < 0.44$).

Estimates of species richness generated from the four methods of data entry fell outside of AR estimate confidence intervals (95% CI) in 6 of 66 replicates (Figure 8). Five of the six deviations can be attributed to the PDA data. Two as a result of the same miss-entered sampling times, and three resulting from categorical errors when entering species type. The remaining deviation, from the single entry, also resulted from a categorical error when entering species type. No significant differences were detected in species richness generated from the single entry and those from the audio-reviewed data (Paired t-test: $df = 65$; $P = 0.32$), or between the PDA estimates in 2004 and those from the audio-reviewed data (Paired t-test: $df = 40$; $P = 0.58$). PDA and audio-review data species richness estimates were identical in all 2005 replicates ($n=25$).

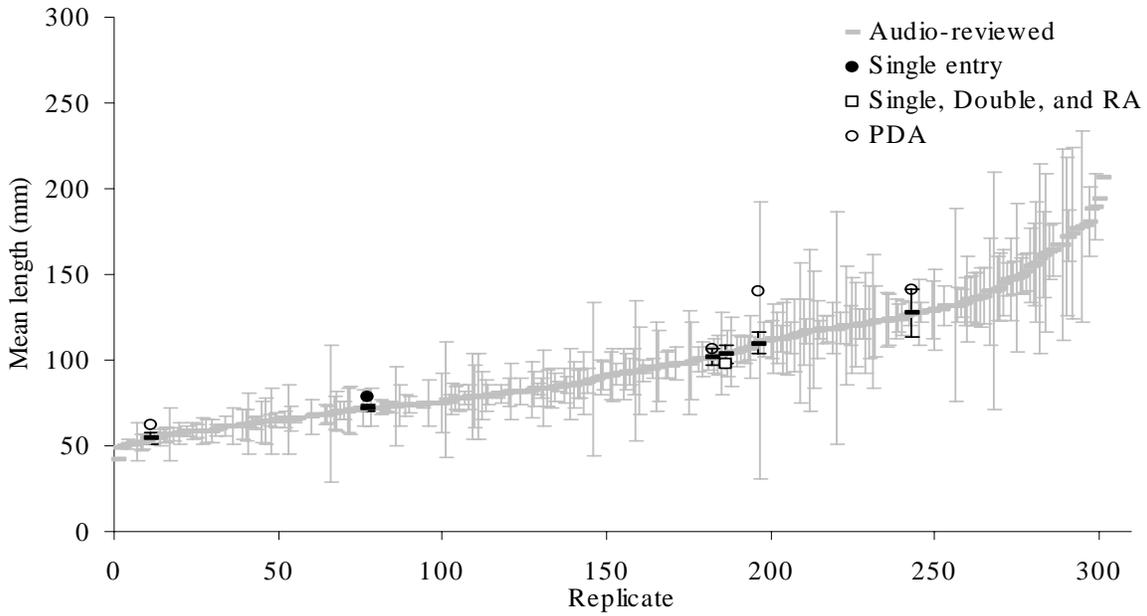


Figure 7. Multiple species length estimates generated from all four methods of data entry. Estimates generated from audio-reviewed data are presented in grey unless an estimate generated from one of the four evaluated methods fell outside of the confidence interval surrounding the estimate.

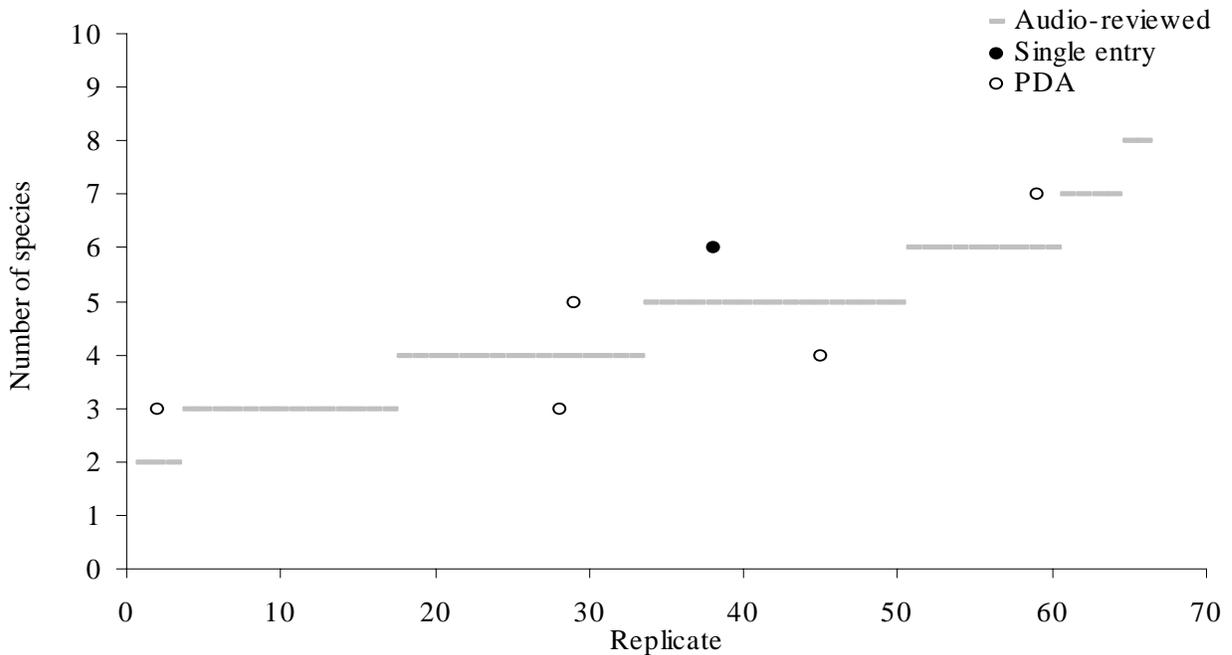


Figure 8. Deviations in the number of species present during electrofishing surveys. Read aloud and double data entry methods did not differ from the audio reviewed data in any of the replicates.

Of the seventeen total errors that caused discrepancies within our three estimates, seven (forty-one percent), were captured through the use of VBA error-checking macros (Table 3.). Errors were typically drag-type or numerical. Errors not identified using automated routines were either categorical or missing species entries.

From post data entry interviews, we determined that read aloud checking procedures took approximately 1.5 times as long as single entry alone. With the addition of a second entry and the effort required to crosscheck the new entry to the original, double data entry took approximately 2.5 times longer than single entry alone.

For our data entry needs we required two PDAs, which were purchased for \$200 per unit in the spring of 2004. The forms software cost us another \$200, and an additional \$60 for licensing costs. Additionally, although the software was fairly strait forward, some time was required to create forms specific to our data entry needs. We approximate initial cost associated with PDA use in 2004 to be \$500. The same units, software, and licenses were used in 2005 at no additional cost.

Table 3. Data errors responsible for estimate deviations outside of the confidence intervals around audio-reviewed “best data”. “Detected” errors are those that were discovered by automated error checking routines. NFT = North Fork Teanaway River, STF = Stafford Creek, TAN = Taneum Creek, UMT = Umtanum Creek, WFT = West Fork Teanaway River. CUT = Cutthroat trout, HYB = Cutthroat-Rainbow hybrid, LND = Long nose dace, PSC = Piute sculpin, RBT = Rainbow trout, TSC = Torrent sculpin.

Method	Type	Description	Fault	Site	Section	Spp	Year	Detected
Abundance								
SDE	Drag	Incorrect pass	Entry	STFA	0-100	CUT	2004	Yes
PDA	Missing	Missing spp	Entry	NFT2.5	0-100	HYB	2005	No
PDA	Missing	Missing spp	Entry	TANB	100-200	CUT	2005	No
PDA	Missing	Missing spp	Entry	TANB	100-200	RBT	2005	No
PDA	Categorical	Incorrect spp	Entry	TANB	100-200	RBT	2005	No
Size								
PDA	Numerical	Incorrect time	Entry	WFT3	100-200	PSC	2004	Yes
PDA	Numerical	Incorrect time	Entry	WFT3	0-100	LND	2004	Yes
PDA	Numerical	Incorrect time	Entry	WFT3	100-200	RBT	2004	Yes
PDA	Numerical	Incorrect time	Entry	WFT3	0-100	TSC	2004	Yes
SDE	Categorical	Incorrect spp	Entry	WFT2	0-100	PSC	2004	No
SDE/RA/DDE	Missing	Missing length	Field	MFT2	0-100	TSC	2005	No
Number of species								
PDA	Categorical	Incorrect spp	Entry	TAN1	0-100	RBT	2004	No
PDA	Numerical	Incorrect spp	Entry	UMT2	100-200	LND	2004	No
PDA	Numerical	Incorrect time	Entry	WFT3	0-100	HYB	2004	Yes
SDE	Categorical	Incorrect spp	Entry	NFT2	100-200	RBT	2004	No
PDA	Numerical	Incorrect time	Entry	WFT3	100-200	HYB	2005	Yes
PDA	Categorical	Incorrect spp	Entry	WFT2	0-100	RBT	2005	No

Discussion

For estimates used by fisheries practitioners, the use of extensive error checking methods such as read-aloud or double-data-entry in lieu of a single entry appears unnecessary. Although we did observe higher levels of error than those reported in other studies, this is to be expected in the field of fisheries, where data are often collected under conditions that are less than ideal such as inclement weather (e.g. temperature, wind, rain), fatigue (e.g. data entry after a full days work), diverse location (e.g. on riverbank, in a drift boat, underwater). Despite the higher data entry error levels, our estimates of abundance, size, and species richness, were not substantially affected. Implications of this may be considerable when considering cost savings (both monetary and in terms of time) may be substantial. By limiting data to only a single entry or a

single PDA entry, fisheries practitioners can expect to reduce their current or projected data entry costs by as much as 150 percent. This could be a savings when the quantity of recorded data is large. Perhaps of even greater importance is the potential savings to smaller programs, where funding is often the most sensitive.

Resource organizations are increasingly using large regional databases as a means of communicating data in a standardized way and also to discover broad scale trends and patterns (Johnson et al. 2007). One of the concerns about the use of large databases that rely upon different groups to supply data is the unevenness of data error checking methods. In short, some data sets may be error free, whereas others may be riddled with error. It is our impression that most data sets have minimal disciplined protocols for data proofing. This could have serious consequences to the utility of the database if errors are frequent. Our study provides information that the concerns for these errors should be low compared to other sources of error such as field techniques and study design.

We found that from a data entry perspective, the PDA is a viable alternative to the more traditional methods of data entry. Although we did detect a year-to-year difference in error rate within the 2004 and 2005 datasets, this difference is likely attributable to a somewhat steeper learning curve in the first year. By the second year some of the crew were familiar enough with the PDAs to instruct new crewmembers in their use. Additionally, our second year of data collection included extensive recording of mark codes, which may have increased complexity of the data entry and the frequency of the data entry error. Regardless, neither year's estimates of abundance, size, or species richness differed significantly from those of our best data. This suggests that their use in the field should be further considered as an alternative to the standard hard-copy method of data recording.

There are inherent advantages to using PDAs over other methods of data entry. Data are available for analysis immediately, and it may be possible to run error-checking macros and outlier rejection routines when data is being collected. This would enable the user to catch error while in the field, where the greatest opportunity for correction exists and also allow practitioners to run timely analyses on the data before returning to the field the next day. From a cost perspective, PDAs have a higher initial cost than hand written data. The cost of each unit, the data entry software, and the time to develop survey specific entry forms add up to a substantial initial investment. However, when weighed against the cost of data entry, particularly with larger datasets or long-term programs, the long-term gains could be substantial. We speculate that PDAs will likely replace hand written data as the primary data-recording device in many areas of fisheries research in the near future.

However, there are still issues surrounding their sole use that are beyond the scope of our study. Foremost is the possibility, however remote, of losing data due to a dead battery or unit malfunction. Fisheries data are often event data, and cannot be recreated if lost. In our experience we found that the PDA units were reliable, durable, and retained their data when their batteries did die. Further, the PDA gave warnings to the user, providing them ample time to replace batteries before the unit actually turned off. Still, fisheries practitioners should use caution when considering the use of PDAs for data collection, until data loss of this type is shown to be so infrequent as to be equated with a lost clipboard.

For our estimates, we based the independence of our replicates on our ability to generate valid estimates with accurate data. In other words, there were instances ($n=2$) in which an entry error was responsible for bias in our estimate in more than one replicate. For this reason, the levels of estimate error presented are higher than what we would have experienced had we

limited our analysis to only one target species per dataset. However, the possibility for multiple errors due to a single data entry error is real, and our analysis provides an illustration of how these errors may materialize in other datasets. Further, our technicians might have had a lower propensity for entry error than those in other programs. Although the majority of our technicians were temporary, all held or were in the process of attaining degrees in the sciences from a local university, and most desired a career in fisheries. Some programs may rely on the efforts of untrained seasonal laborers or volunteers in order to keep costs low, and might therefore encounter higher rates of entry error, particularly if their staffs have not used PDAs. Consequently, practitioners are unlikely to accept a single entry, or a PDA entry, without some assurance that the resulting data meet minimum quality standards. There is a need for low cost data checking methods such that data of reasonable quality and consistency can be assured.

We found that VBA macros designed to catch data entry errors filled our need for a low cost data checking method. Visual Basic macros are easily created and revised, cost effective to implement, and provide an effective method for discovering both errors and outliers within a dataset. Although our macros did not catch all of the errors that were responsible for discrepancies in our individual estimates, further revision of the macros over time would allow the capture of most. The use of such macros reduces error levels and can prevent gross discrepancies arising from unforeseen circumstances (e.g. inappropriate electronic data handling, inexperienced data entry, inattentive employees). In addition, thoughtfully constructed macros can further reduce costs by pointing the user directly to the location of potential errors. Most importantly, the use of automated error checking provides fisheries practitioners with data that meet a minimum level of quality with little additional cost. However, as with any method of error checking, opportunities for error exist with the implementation of automated macros (e.g. formatting errors could result in sorting errors).

We suggest that fisheries practitioners consider the use of PDAs in their collection efforts because of their potential to decrease entry error, improve efficiency, and lower costs. PDA datasets averaged, at most, 1.43 percent error, 0.27 percent greater than single entry alone. This difference may be marginal, but the long-term benefits are greater than conventional single data entry, particularly when well-designed, automated checking procedures are incorporated as a data proofing method to assure standard levels of data quality. For data collection efforts in which PDAs are considered too costly, or for which the risks of data loss are too high, a single data entry with rudimentary checking procedures will suffice for most common fisheries programs.

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Chapter 5

Abundance, size, and distribution of main stem Yakima River rainbow trout

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Abstract

Long-term trend monitoring of resident rainbow trout *Oncorhynchus mykiss* in the Yakima River Basin is a vital component of the Yakima Klickitat Fisheries Project (YKFP) non-target taxa of concern (NTTOC) monitoring and evaluation program. Concerns that supplementation may negatively affect the resident rainbow trout population in the upper Yakima River contributed to the development and implementation of the program. Information on the abundance, size, and distribution of rainbow trout in the Yakima River are collected annually in five sections of the Yakima River between Roza Dam and the Cle Elum River confluence. The information collected in these sections is presented under the risk containment monitoring framework (Chapter 2) and can be difficult to interpret outside of that context. The purpose of this report chapter is to provide managers, anglers, special interest groups, and all interested readers a summarized version of Yakima River rainbow trout data collected under the YKFP's NTTOC monitoring program in the Yakima Basin. These data should be considered preliminary until published in a scientific journal.

Introduction

Long-term trend monitoring of resident rainbow trout *Oncorhynchus mykiss* in the Yakima River Basin is a vital component of the Yakima Klickitat Fisheries Project non-target taxa of concern (NTTOC) monitoring and evaluation program. The rationale for monitoring rainbow trout was initiated in response to salmon supplementation in the upper Yakima River Basin. Concerns that supplementation may negatively affect the resident rainbow trout population in the upper Yakima River contributed to the development and implementation of the NTTOC monitoring program. Data from main stem Yakima River rainbow trout have been collected since 1990 under the program. These data has generally been presented under the framework of risk containment monitoring and has been difficult for managers and resource groups to interpret under this context. The purpose of this report is to provide managers, anglers, and other interested readers a summarized version of the status of Yakima River rainbow trout collected under the YKFP's NTTOC monitoring program in the Yakima Basin. In addition, specific information that is of interest to anglers is also presented. The specific details and evaluation of this data in the context of risk containment monitoring is presented in chapter 2 of this report (Pearsons et al. 2007). The data presented in this report supercedes any previously reported Yakima River rainbow trout population data but should be considered preliminary and is subject to change as new analysis and evaluations are performed.

Methods

In the main stem Yakima River, the abundance and size structure of rainbow trout is estimated in 5 index sites during the fall of every year, generally after the river returns to baseline discharge levels following unnaturally high flows (flow management termed “flip-flop”). Electrofishing is conducted after dark to increase capture efficiencies on trout. A drift boat mounted electrofishing unit is used for mark-recapture sampling to estimate the abundance of rainbow trout and simple visual counts taken during sampling is used to estimate the relative abundance of all other species. During sampling, all trout are captured, fork length and weight are measured, inspected for general condition, and released. Notable comments such as visible injuries from hook scarring, parasites, or fish in abnormal condition are recorded. A more detailed description of sampling protocols is presented in Temple and Pearsons (2007). Fish collection data is analyzed using computer software to estimate abundance, abundance at age, and size structure information.

Five index sites in the main stem Yakima River were used to represent five contiguous study reaches and the index sites comprised approximately 28% of the upper Yakima River between Roza Dam and the Cle Elum River confluence (Figure 1). Each index site was selected to be representative of a larger reach. Selection criteria included river constraint and elevation. Estimates were generated from abundance and size data collected in the site and were extrapolated to the reach scale based on reach length. The Lower Canyon (LCYN) site was 2.98

river miles (rm) long, the Upper Canyon (UCYN) site was 3.22 rm long, the Ellensburg (EBURG) site was 2.61 rm long, the Thorp (THORP) site was 3.54 rm long, and the Cle Elum (CELUM) site was 4.60 rm long. The reach descriptions are as follows: LCYN extends 11.92 rm upstream from Roza Dam to Umtanum Creek; UCYN extends 8.32 rm upstream from Umtanum Creek to the Ringer Road access; EBURG extends 13.17 rm upstream from the Ringer Road access to the Ellensburg Dam; THORP extends 14.97 rm upstream from the Ellensburg Dam to the Teanaway River; and CELUM extends 10.06 rm upstream from the Teanaway River to the Cle Elum River. The data collected in these reaches are presented in the following tables and figures.

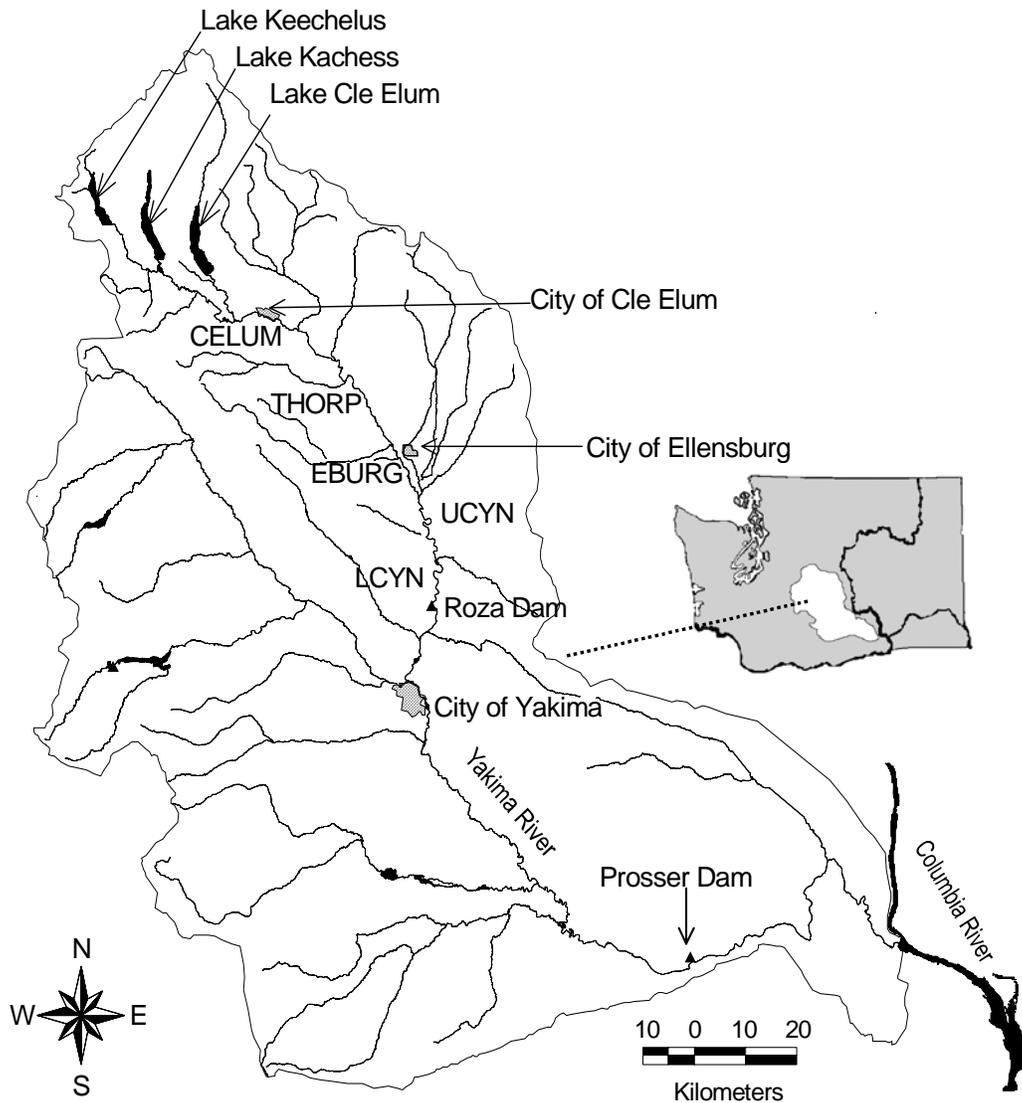


Figure 1. Map showing the Yakima River Basin. Main stem Yakima River sampling sections are shown with capital letters.

We have developed the following equation that allows the weight in pounds of Yakima River rainbow trout to be calculated from their measured fork length in inches. The fork length is measured from the tip of the snout to the notch, or “V” in the caudal fin (tail).

$$\text{Weight in pounds} = 0.0005 \times (\text{measured fork length in inches})^{2.9298}$$

Most calculators contain function buttons that provide an easy way to perform the calculation. Calculators containing a button labeled Y^x can be used by first entering the measured fork length, pressing the Y^x button once, entering the coefficient 2.9298, then the = button. Finally, multiply the result by the coefficient 0.0005. The result is weight in pounds. For example, using the above formula, a 14.5 inch rainbow trout weighs approximately 1.3 pounds.

Using the equation to calculate fish weight can help minimize handling stress of caught fish because they do not need to be removed from the water to determine their size. This is beneficial because it is generally thought that there is a direct relationship between excessively exercised (hooking, fighting, handling, and air exposure) trout with lethal and sub-lethal effects on survival. Ferguson and Tufts (1992) reported 38% and 72% mortality within 12 hours for rainbow trout for that had been held out of water for 30 and 60 seconds after exhaustive exercise (perhaps similar to being hooked, played, and removed from the water for hook removal and held for a photo). Thus, when participating in catch-and-release fisheries, it is prudent to quickly land hooked fish, avoid over playing them, minimize handling, and remove the hook without removing the fish from the water (Cassleman 2005). When taking photos, it is best to do so quickly, and preferable to hold the fish just above the water surface. When fish are deep hooked and must be removed from the water for hook removal, exposure times to the air should be minimized (< 120 seconds) to prevent impairments to swimming performance (Schreer et al. 2005).

Results

The results from sampling this year were within the range we have seen in previous years. The rainbow trout abundance index (including all fish over 4 inches in length) was less than observed the previous few years, although they were of similar size and condition. The average abundance and size of main stem Yakima River rainbow trout for all years sampled are presented in Table 1. The total estimated annual abundance of rainbow trout in the Yakima River between Roza Dam and the Cle Elum River confluence is presented in Figure 2.

Table 1. Average annual abundance (fish/mile), average size (fork length in inches, and weight in pounds), and total abundance of upper Yakima River rainbow trout greater than 4 inches in length between Roza Dam and the Cle Elum River confluence.

Year	Fish / River Mile	Avg. Fork Length (in)	Avg. wt (lbs)	Total Fish (Roza to Cle Elum River)
1990		10.7	0.530	
1991	546	10.3	0.463	32108
1992	373	10.2	0.422	21963
1993	463	10.7	0.501	27264
1994	419	10.2	0.459	24645
1995	402	10.5	0.486	23632
1996	426	9.6	0.397	25049
1997	534	9.2	0.351	31423
1998	443	10.5	0.500	26084
1999	543	10.6	0.542	31972
2000	641	10.1	0.471	37701
2001	852	9.6	0.428	60863
2002	638	10.4	0.485	37518
2003	659	10.3	0.461	38767
2004	635	10.3	0.466	39531
2005	650	10.1	0.452	38221
2006	425	10.2	0.574	25017

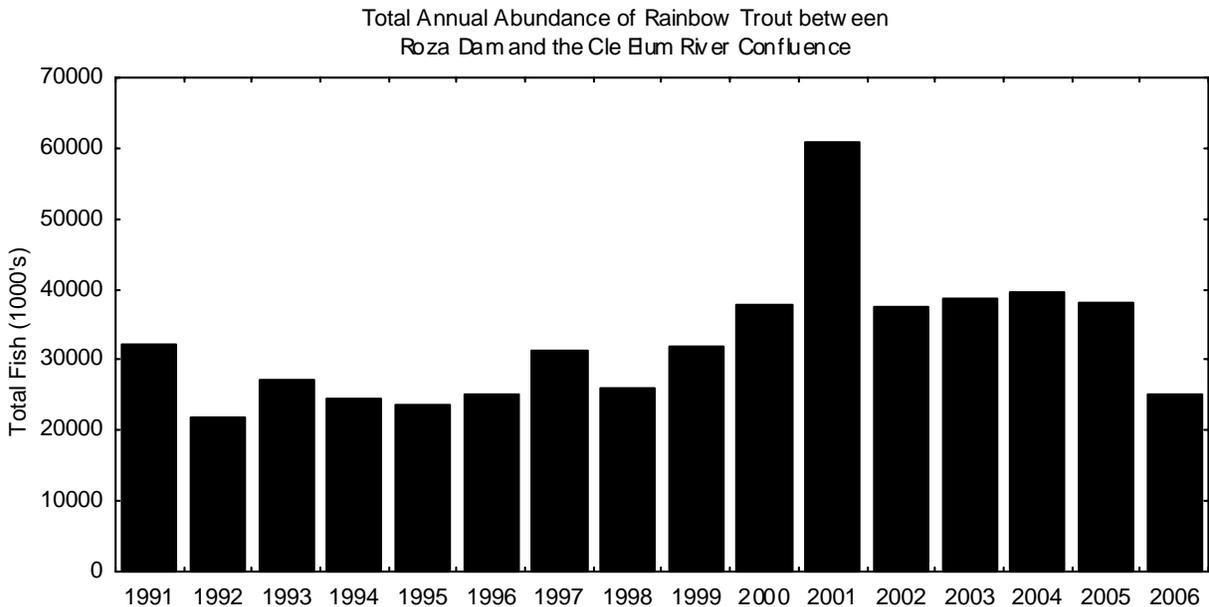


Figure 2. Total estimated annual abundance of rainbow trout greater than 4 inches in length in the Yakima River between Roza Dam and the Cle Elum River confluence.

The annual estimated abundance and the average catchable sized rainbow trout (over 10 inches) per river mile in each sampled section is presented in Figure 3.

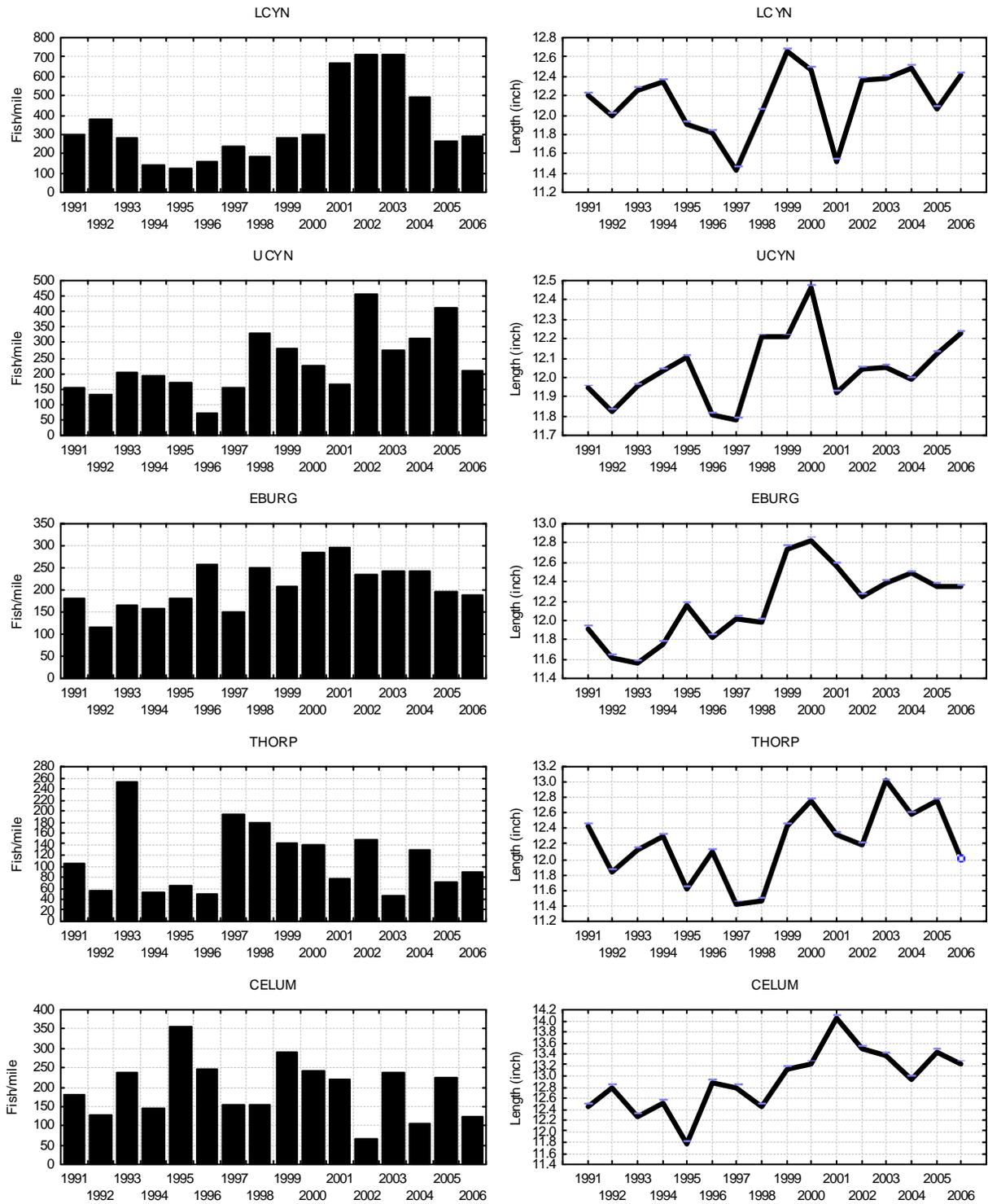


Figure 3. Abundance (Fish/mile) and size (fork length in inches) of catchable sized (over 10 inches) Yakima River rainbow trout in main stem sampling sections.

The largest resident rainbow trout captured during sampling activities are presented in Table 2. The longest and heaviest fish have all been captured since 2000 (Table 2).

Table 2. Largest Yakima River rainbow trout recorded annually in main stem sampling sites. The longest and heaviest fish are in bold type. TL is total length in inches and WT is weight in pounds.

Year	LCYN		UCYN		EBURG		THORP		CELUM	
	TL	WT								
1990	18.6	1.7	18.7	2.2	16.9	1.6	18.4	2.1	19.2	2.4
1991	16.4	2.0	16.3	1.4	17.6	1.4	18.1	1.7	18.9	2.4
1992	17.8	1.9	15.5	1.4	16.5	1.3	18.6	2.1	20.3	2.7
1993	18.2	1.9	17.6	1.8	16.3	1.4	18.0	1.9	19.5	2.3
1994	17.9	2.0	16.2	1.4	15.6	1.4	19.6	2.8	20.3	3.1
1995	18.6	2.5	19.2	2.1	19.2	2.4	17.7	2.0	19.0	2.4
1996	18.1	2.2	16.9	1.8	18.6	2.0	18.4	2.0	19.2	2.6
1997	17.6	1.9	16.7	1.9	17.8	1.9	18.0	2.0	20.0	3.1
1998	19.8	2.9	18.0	2.1	17.4	1.9	18.9	2.2	20.3	2.6
1999	21.3	3.6	18.4	2.0	18.8	2.3	19.0	2.2	21.1	3.4
2000	21.4	3.8	17.8	2.1	18.3	2.0	20.2	3.6	21.7	3.3
2001	19.1	2.6	20.0	3.1	18.1	2.3	18.9	2.6	20.2	3.0
2002	19.4	2.4	18.2	2.2	18.0	2.0	19.9	2.7	19.9	2.7
2003	18.8	2.6	17.1	1.9	17.2	1.7	18.7	2.4	21.4	3.6
2004	17.4	1.8	17.7	1.9	18.9	2.5	19.2	2.7	20.4	3.4
2005	17.2	1.9	17.0	1.7	17.3	1.7	18.4	2.5	20.7	3.4
2006	18.9	2.6	17.4	1.9	19.6	2.5	21.4	3.0	20.0	2.9

The length-at-age of Rainbow trout in the Yakima River appears to be quite variable within each age class based on age information collected by reading their scales (Figure 4).

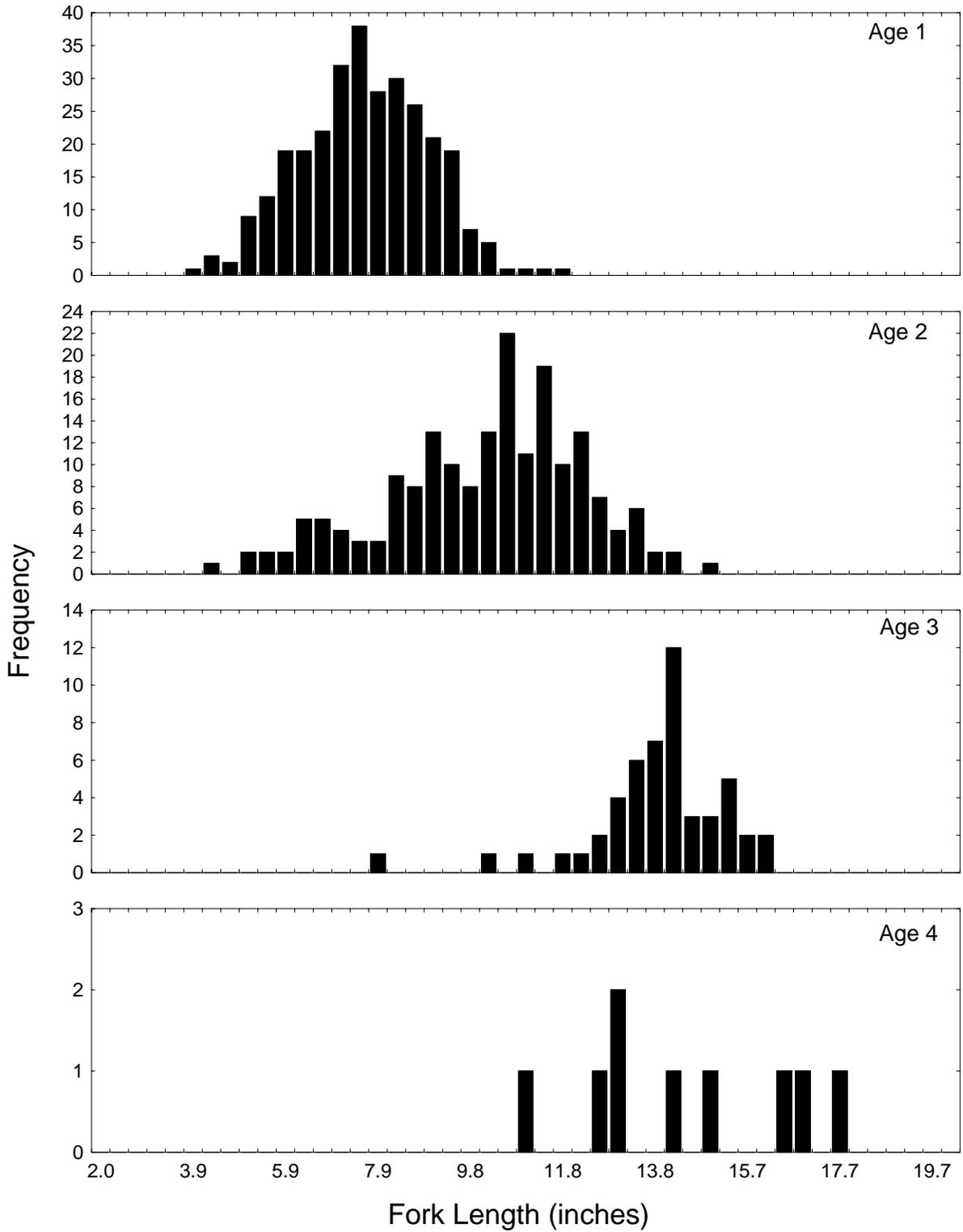


Figure 4. Length at age of Yakima River rainbow trout from scale data collected 1991-1995.

In addition to rainbow trout, several other fish species inhabit the upper Yakima River. The relative abundance of the fish species observed in 2006 is presented in Figure 5. We observe cutthroat trout regularly, but generally in the higher elevation sections (THORP and CELUM). Fish with hyoid red-slashes are also routinely observed. These red-slashes are characteristic of rainbow trout (redband) that live east of the Cascades (Phelps 1993). Over the course of the NTTOC monitoring program, we have also documented several other fish species in the river that are generally low in number and are not presented in Figure 5. Several of these species are benthic, living on or near the stream bottom, and therefore are difficult to observe. Sculpin species that have been documented include torrent and mottled sculpin. Dace species include longnose dace and speckled dace. Sucker species include bridgelip sucker, largescale sucker, and an occasional, although rare, mountain sucker. We have also documented the presence of western brook lamprey, although they are often buried in the stream bottom and not often observed, although they are likely present. We have also observed coho salmon, kokanee, lake trout, burbot, bull trout, brook trout, common carp, largemouth bass, yellow perch, chiselmouth minnow, pumpkin-seed, brown trout, threespine stickleback, smallmouth bass, and bluegill during sampling.

Relative Abundance of Upper Yakima River Fishes 2006

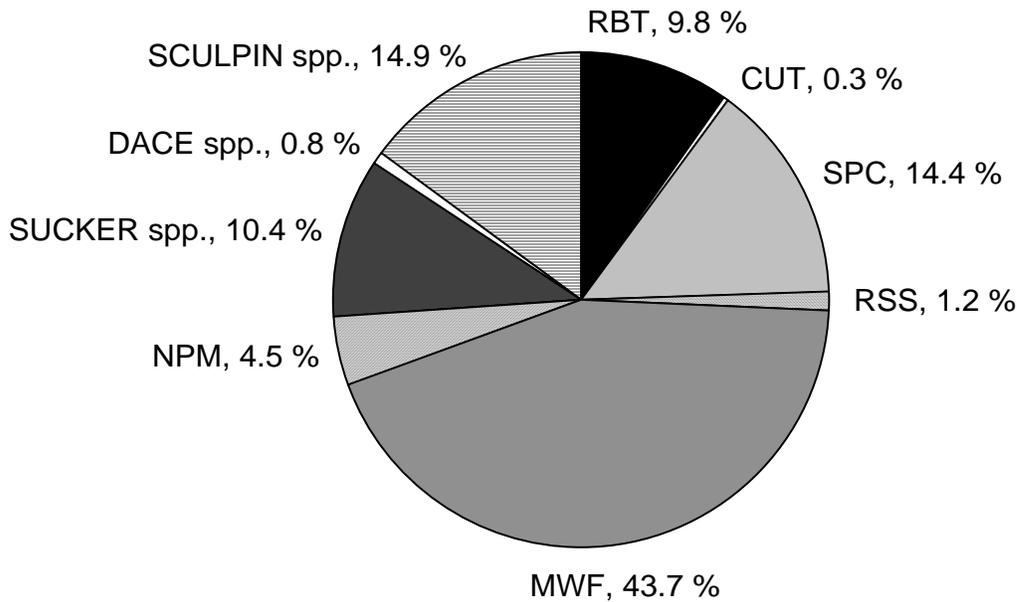


Figure 5. Species composition of fish in the upper Yakima River observed during electrofishing. Species codes are: Rainbow trout (RBT); Cutthroat trout (CUT); spring Chinook salmon (SPC); reidside shiner (RSS); mountain whitefish (MWF); northern pikeminnow (NPM); sucker species (SUCKER spp.); dace species (DACE spp.); and sculpin species (SCULPIN spp.).

We observed spatial and temporal trends in the percent of fish with visible hook scars. The LCYN and UCYN sections had the highest proportion of fish with visible hook scarring while the THORP and CELUM sections had the least in 2006 (Figure 6, Panel A). This may coincide with fishing pressure because the lower elevation river sections generally have the highest fishing pressure (McMichael 1998). However, despite the assumed increase in fishing pressure through time, there has been a general decline in the number of hook scarred trout observed annually (Figure 6, Panel B). This may be due to improved techniques of removing hooks.

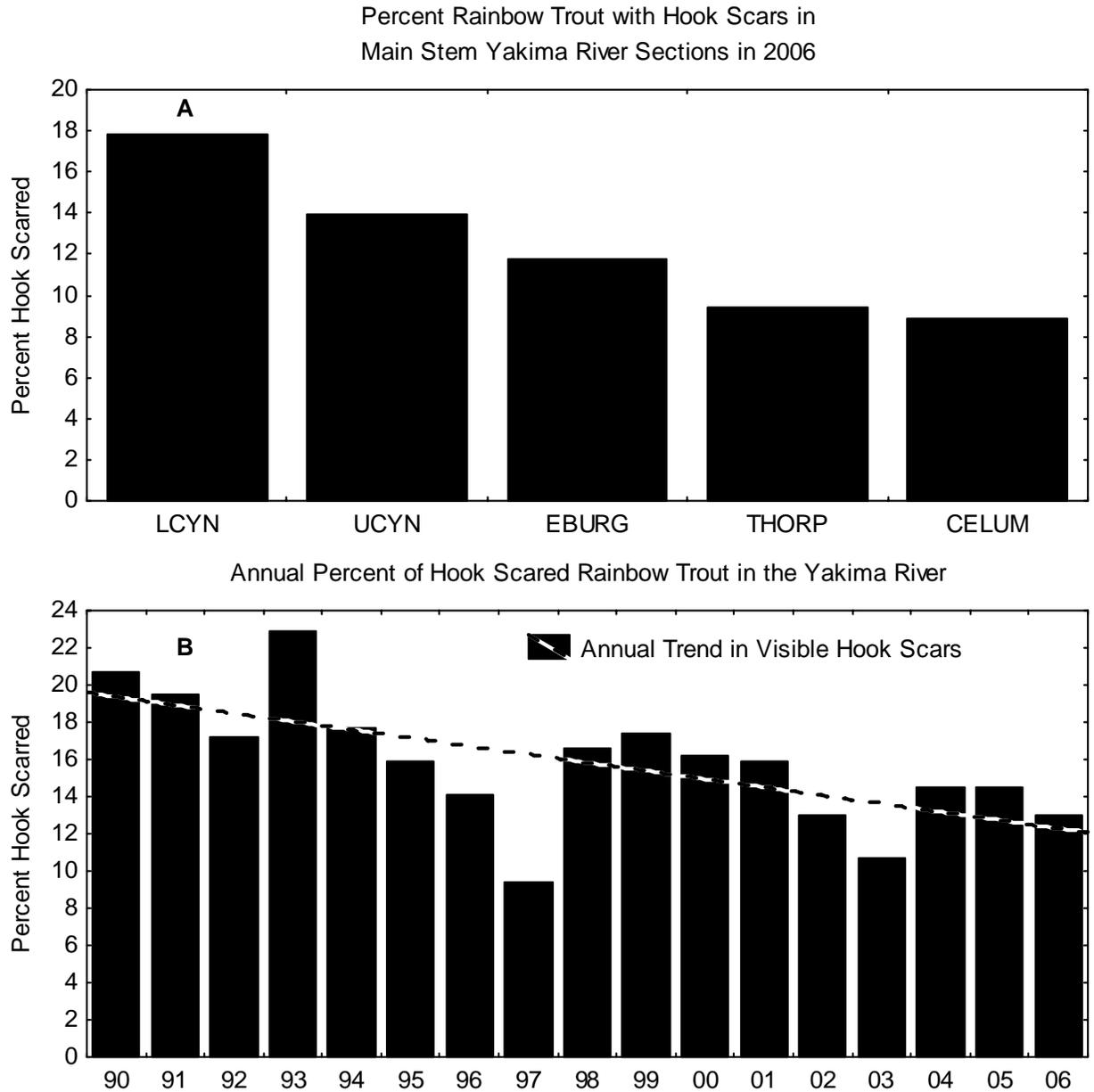


Figure 5. Percent of rainbow trout in main stem Yakima River sections containing visible hook scars during the fall of 2006 (Panel A) and the percent of rainbow trout with visible hook scars observed annually (Panel B).

A comparison of the density and size at age of rainbow trout in several Pacific Northwest streams and rivers is presented in Tables 3 and 4. The number of fish per mile is lower in the Yakima River than the other Rivers in Table 3. This may be due to a variety of factors such as stream size. Wider streams are more likely to produce more trout because there is more area available for fish to use. However, with the exception of age 1, the size at age of fish in the Yakima is similar to the streams listed in Table 4. The size of fish at age 1 is smaller than most of those in other Rivers in Table 4.

Table 3. Density of Yakima River rainbow trout versus other Pacific Northwest rivers.

Year	River	Fish per mile	Reference
1986	E. Gallatin River, MT	2518	Vincent 1987
1985	E. Gallatin River, MT	2326	Vincent 1987
1984	E. Gallatin River, MT	3802	Vincent 1987
1987	Big Horn River, MT	949	Fredenberg 1988
1982	Madison River, MT	1563	Vincent 1984
1983	Madison River, MT	1229	Vincent 1984
1986	Madison River, MT	1539	Vincent 1987
1986	Madison River, MT	1174	Vincent 1987
1970's	Deschutes River, OR	1757	Schroeder and Smith 1989
1970's	Deschutes River, OR	1618	Schroeder and Smith 1989
1991	Yakima River, WA	546	This Report
1995	Yakima River, WA	402	This Report
2000	Yakima River, WA	641	This Report
2005	Yakima River, WA	650	This Report

Table 4. Average length at age (inches) of rainbow trout in several Northwest streams.

River	Age					Reference
	1	2	3	4	5	
Deschutes R., OR						
Nena Creek	4	7	11	12	13	Schroeder and Smith 1989
N. Junction	5	9	11	13	13	Schroeder and Smith 1989
Jones Canyon		9	11	12	14	Schroeder and Smith 1989
Snake River ID	5	10	13	18	19	Irving and Culpin 1956
Montana Streams	3	7	10	13	14	Carlander 1977
Yakima River	3	7	10	13	14	Martin and Pearsons 1993
Yakima River	5	8	11	13	15	WDFW unpublished (2003)

Discussion

Results from the YKFP NTTOC monitoring program indicate that there has not been any significant decline in the abundance, size, or distribution of main stem Yakima River rainbow trout after approximately 1 million supplementation origin salmon have been released in the river annually beginning in 1999. Thus, data collected under the risk containment program indicate that there has not been any significant negative effects of salmon supplementation on the rainbow trout available to the fishery. In fact, the abundance and size of catchable sized rainbow trout have increased during the period salmon have been released in the Yakima River and their distribution has remained unchanged. The largest rainbow trout that we have captured since 1990 have been collected after the year 2000. However, in some years such as 2001, the quality of the fishery may be degraded because of the large numbers of spring Chinook salmon residuals (Chinook salmon that do not migrate to the ocean). The YKFP has implemented strategies to reduce the number of these fish and we have had relatively few complaints from the angling community since 2001. Pending funding, we will continue to closely monitor the status of rainbow trout in the Yakima River as the YKFP salmon supplementation program progresses so that managers are informed about adverse effects on main stem rainbow trout that are beyond acceptable levels.

Many people have expressed concerns about the potential for electrofishing to harm fish populations (Snyder 1995; Nielsen 1998). It is clear that electrofishing can harm individual fish, particularly large individuals or susceptible life stages or species (McMichael et al. 1998). Injuries can range from bruising to broken backs. However, in the Yakima River and in many other areas, the proportion of individuals within populations that are actually sampled is small (generally between 5% and 15%) and even large injury rates at the sample scale may be acceptable for monitoring and conservation oriented studies (McMichael et al. 1998). Furthermore, electrofishing induced injury rates evaluated from the population level may be negligible when considering natural mortality rates of stream fish (Schill and Beland 1995). Repeated electrofishing from long-term monitoring did not appear to have detrimental population level effects on salmonids in small Colorado streams (Kocovsky et al. 1997) or in our sampling sites.

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