Yakima River Species Interactions Studies

Yakima/Klickitat Fisheries Project Monitoring and Evaluation Report 7 of 7

Annual Report
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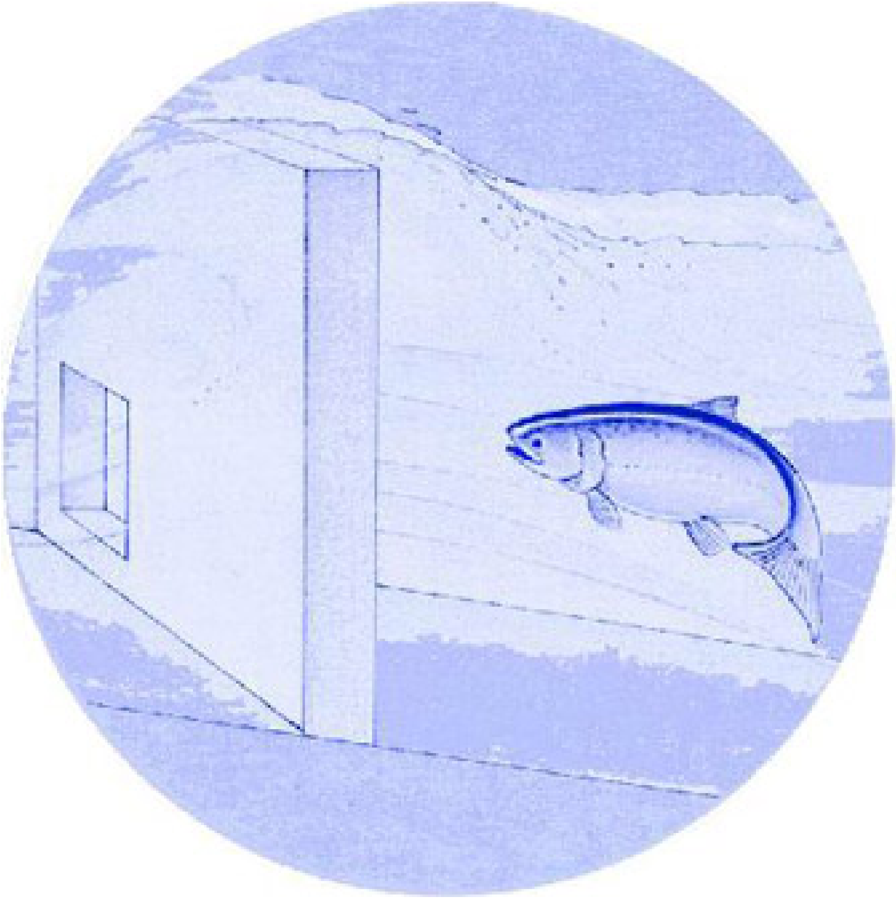

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Yakima River Species Interactions Studies
Yakima/Klickitat Fisheries Project Monitoring and Evaluation

Annual Report 2003

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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 twelfth 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, and monitoring fish predation indices. This report is organized into two 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, 2003 and December 31, 2003 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.

Release of large numbers of hatchery origin salmon has the potential to negatively impact other taxa (non-target taxa, NTT). To determine changes in NTT status that could be related to hatchery smolt releases, we compared the abundance, size structure, and distribution of 14 non-target taxa before and five years after annual spring releases of about 1 million yearling smolts (coho and chinook) in the Yakima River, Washington. We compared any observed changes in status to predetermined containment objectives that were judged to reflect acceptable levels of impact. We utilized detection strategies that would balance our ability to detect changes and the chances of falsely associating a change with supplementation. With the exception of steelhead size, all of the changes we observed were within the containment objectives established for the project. The mainstem Yakima River steelhead size index has decreased through the post-supplementation period although the decrease was not significant with the addition of the 2003 data (-1%, $P>0.05$). Our analysis suggests that the depressed size of the steelhead index was not related to supplementation activities. For instance, we could not detect any differences in the sizes of rainbow trout between areas of high and low spring chinook abundance. Our results suggest that any impacts that might have been caused by releasing hatchery smolts into areas containing NTT were balanced or exceeded by the benefits (e.g., ecological release) of reducing the progeny of naturally produced fish or by the increase in nutrients provided by the hatchery and returning adults. The reduction of naturally produced fish in the river was the result of taking fish that would have spawned in the river into the hatchery. Results from status monitoring of 14 NTT after five years of hatchery releases suggest that risk containment actions are not necessary at this time.

We endeavored to evaluate whether increasing site length and keeping sampling effort constant could improve our ability to detect changes in the abundance, size structure, and species richness of fish in tributaries to the Yakima River, Washington. The efficacy of using a single electrofishing pass to quantify stream salmonids was evaluated by comparing the abundance estimates generated from the first electrofishing pass in block-netted stream sections to traditional maximum likelihood estimates for the same sites and years. The numbers of fish captured during the first electrofishing pass in multiple-removal estimates were significantly correlated with the resulting maximum likelihood estimates for those sites ($r=0.98$, $P<0.001$). Our data indicated that our capture probabilities generated annually in mid-elevation multiple-removal sites between the years 1999 and 2003 have not significantly differed from median capture probabilities established for those sites during the years 1990-1998 ($G=3.21$, $df=99$, $P>0.05$). Removing block-nets from a sub-set of electrofishing index sites between 1998 and 2003 did not bias the estimates from those sites when they were compared to estimates generated from traditional multiple-removal estimates from adjacent sites during the same period ($F=0.05$, $df=1$, $P>0.05$). Reallocating our sampling effort from multiple-pass electrofishing in 100 meter long index sites to single-pass electrofishing in 200 meter long index sites provided a 20% decrease in the annual variation around our abundance estimates while not significantly altering the population estimates themselves ($t=1.65$, $df=124$, $P>0.05$). We conducted a power analysis and found that we could decrease our detectable effect size to population abundance by 26% after five years of monitoring. Extending our site lengths did not appear to provide detection benefits when evaluating impacts to the size structure of rainbow trout, *Oncorhynchus mykiss*. However, we improved our ability to detect rare species such as bull trout, *Salvelinus confluentus*, spring chinook salmon parr, *Oncorhynchus tsawytscha*, and cutthroat trout, *Oncorhynchus clarki*, by 75%, 19%, and 17%, respectively. We recommend that multiple-removal sampling be used when stream fish monitoring programs intend to monitor for less than five years and that long term programs consider adopting single-pass electrofishing methods.

Predation on hatchery and wild salmonids by non-native smallmouth bass *Micropterus dolomieui* was examined in the Yakima River, Washington from 1998 to 2001. Smallmouth bass were sampled weekly in two sections of the lower Yakima River from March through June using a drift boat electrofisher. Average abundance estimates of smallmouth bass $\geq$150 mm fork length for the four years sampled ranged from 3,347 in late March to 19,438 in early June. Abundance estimates from 1998 to 2001 all showed a similar trend of increasing abundance throughout the spring. Salmonids were identified in the guts of smallmouth bass throughout the sampling period and were most prevalent during the month of May. Ocean type chinook salmon *Oncorhynchus tshawytscha* constituted 47% of all the fish species identified in the gut samples. Smallmouth bass were estimated to have consumed an average of 200,405 salmonids yearly from March 22 to June 16, 1998 to 2001; of these only 3,176 were yearling salmonids (primarily spring chinook salmon). Smallmouth bass predation on all yearling salmonids never exceeded 0.6% of the annual production of hatchery and wild fish combined. We estimated that 85% of the ocean type chinook salmon consumed by smallmouth bass in a given year were of natural origin. Estimated smallmouth bass consumption of hatchery ocean type chinook salmon has only comprised up to 4% of production in a single year. Our estimates of consumption on ocean type chinook salmon are likely to be underestimates because we did not sample throughout the entire rearing and emigration period of these fish. Our results indicate that smallmouth bass can have negative impacts on ocean type chinook, particularly naturally produced chinook, which are generally smaller and available longer than hatchery fish.

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 twelfth 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). 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. 1999; McMichael et al. 1999; 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, Pearsons et al. in press, Major et al. in press). This progress report summarizes data collected between January 1, 2003 and December 31, 2003. 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. 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).

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 three chapters that represent major topics associated with monitoring stewardship, utilization, and strong interactor taxa. Chapter 1 reports the results of non-target taxa monitoring after the fifth release of hatchery salmon smolts in the upper Yakima

River basin. Chapter 2 describes our tributary sampling methodology for monitoring the status of tributary NTT. Chapter 3 describes predation on juvenile salmonids by smallmouth bass and channel catfish in the lower Yakima River.

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).

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Chapter 1

Results of non-target taxa monitoring after the fifth release of hatchery salmon smolts in the upper Yakima Basin

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Abstract

Release of large numbers of hatchery origin salmon has the potential to negatively impact other taxa (non-target taxa, NTT). To determine changes in NTT status that could be related to hatchery smolt releases, we compared the abundance, size structure, and distribution of 14 non-target taxa before and five years after annual spring releases of about 1 million yearling smolts (coho and chinook) in the Yakima River, Washington. We compared any observed changes in status to predetermined containment objectives that were judged to reflect acceptable levels of impact. We utilized detection strategies that would balance our ability to detect changes and the chances of falsely associating a change with supplementation. With the exception of steelhead size, all of the changes we observed were within the containment objectives established for the project. The mainstem Yakima River steelhead size index has decreased through the post-supplementation period although the decrease was not significant (-1%, $P>0.05$). Our analysis suggests that the depressed size of the steelhead index was not related to supplementation activities. For instance, we could not detect any differences in the sizes of rainbow trout between areas of high and low spring chinook abundance. Our results suggest that any impacts that might have been caused by releasing hatchery smolts into areas containing NTT were balanced or exceeded by the benefits (e.g., ecological release) of reducing the progeny of naturally produced fish or by the increase in nutrients provided by the hatchery and returning adults. The reduction of naturally produced fish in the river was the result of taking fish that would have spawned in the river into the hatchery. Results from status monitoring of 14 NTT after five years of hatchery releases suggest that risk containment actions are not necessary at this time.

Introduction

Despite the long history of stocking hatchery salmon into streams, few evaluations of impacts to non-target taxa (NTT) have been conducted. Many mechanisms of impacts have been documented (Marnell 1986; Nielsen 1994; Hawkins and Tipping 1999), but impacts to NTT population size, growth, or distribution generally have not been conclusively demonstrated at scales larger than experimental reaches (Fresh 1997). Exceptions include the relatively large-scale evaluations of stocking salmon before the smolt stage (Bjornn 1978; Nickelson et al. 1986). Although these studies are illuminating, most contemporary hatchery salmon programs release smolts. In order to evaluate impacts of contemporary programs, information about the impacts of smolt releases is needed.

Ecological interactions resulting from smolt releases should be evaluated throughout the lifespan of a hatchery supplementation program because the type and strength of ecological interactions differ during stages of hatchery supplementation dynamics (Pearsons 2002). This paper will address impacts that occur during the early stages of supplementation termed the Broodstock and Building stages by Pearsons (2002). When a supplementation program is initiated, wild broodstock are collected, spawned, and then their progeny are released as smolts. During this initial stage, interactions between naturally produced target species and NTT are reduced but interactions between hatchery produced target species and NTT are potentially high (Pearsons 2002). In essence, rearing of fish in a hatchery is an ecological tradeoff between lower interactions with wild fish before the smolt stage, with higher interactions from the smolt to adult stages. A reduction in the interactions among naturally produced fish occurs because target species that would normally rear in the wild are reared in the hatchery. In contrast, the higher survival of fish reared in the hatchery translates into greater number of smolts than would have occurred naturally. The next stage of supplementation, 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 and the offspring from returning hatchery adults increases interaction potentials between hatchery and wild fish in the freshwater migration corridor, freshwater rearing area (e.g., if hatchery fish residualize), estuary, and ocean.

Hatchery yearling smolts released and the progeny from returning adult hatchery fish can interact with NTT. Type I interactions are those that occur between hatchery fish (e.g., smolt, residual, or adult) and wild fish (Pearsons and Hopley 1999). If Type I impacts are less than benefits produced from ecological release (reduced interspecific competition), then non-target species will benefit, the converse is also true. Type I interactions can be non-natural because humans artificially rear and release the fish. Type II interactions occur between NTT and naturally produced offspring of hatchery fish (Pearsons and Hopley 1999). Type II interactions may be more natural than Type I interactions because the behaviors of the target fish are not altered in a hatchery environment. While Type I interactions occur during the Broodstock stage, both Type I and Type II interactions occur during the Building stage of supplementation. Hatchery fish are typically more numerous, more concentrated, larger, and in some instances more aggressive than wild fish (Ruzzante 1994; White et al. 1995). These differences can confer dominance status to hatchery fish (McMichael et al. 1997; Rhodes and Quinn 1998; McMichael et al. 1999), decrease the size refuge of wild fish to predation by hatchery fish (Pearsons and Fritts 1999), and 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). If smolts actively migrate after release, then the interactions with NTT in the freshwater migration corridor are likely to be relatively low. However, increased natural production of the target taxa translates into potentially increased interactions in the freshwater rearing area.

Hatchery smolts can interact with wild fish during downstream migration and during periods when they residualize in rearing environments. Ecological interactions that can occur during migration include competition, predation, behavioral anomalies, and pathogenic interactions (Pearsons and Hopley 1999). If competition occurs, it is likely to be intense but of short duration because hatchery smolts generally move downstream and feed as they migrate or during brief "resting" periods. It is during the "resting" periods that competition might be most intense. 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). As mentioned before, 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, which has the potential to influence susceptibility to predators or food acquisition (Hillman and Mullan 1989; McMichael et al. 1999). Finally, hatchery fish have the potential to transmit or increase the susceptibility of pathogens to wild fish (Goede 1986; Bucke 1993; McVicar 1997). The same aforementioned interactions can occur during the periods when "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.

In this paper, we examine the impacts to NTT during the Broodstock and early Building stages of a spring chinook supplementation program and the reintroduction of coho salmon in the Yakima Basin, Washington (Figure 1). Concerns about the possibility of hatchery fish having negative impacts on valued non-target taxa (NTT) in the Yakima Basin prompted the development and implementation of a risk containment monitoring program (Bonneville Power Administration 1996; Busack et al.1997). Spring chinook and coho salmon were released in 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 using artificial propagation (supplementation). Approximately one million salmon smolts have been released annually in the upper Yakima River from 1999 to 2003 (Table's 1 and 2). Spring chinook salmon were volitionally released into the Yakima River from sites near the cities of Easton, Thorp, and near Jack Creek on the North Fork of the Teanaway River (Figure 1). Coho salmon were volitionally released into the Yakima River from sites near the city of Cle Elum (hatchery slough 1999, 2000, and 2001), near Jack Creek on the North Fork of the Teanaway River (1999), below Easton Dam (1999-2003), and at the "Holmes site" at Yakima River mile 160. 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).

Table 1. Numbers and location of yearling spring chinook salmon released in the upper Yakima River 1999-2003.

		Spring Chinook Salmon			
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	70,132	31,614	241,956	373,702

Table 2. Numbers and location of yearling coho salmon released in the upper Yakima River 1999-2003.

		Coho Salmon				
Brood Year	Release Year	Easton	Jack Creek	Hatchery Slough	Holmes	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

Methods

We monitored the changes in status of 14 NTT that have the potential to be impacted by the supplementation of spring chinook salmon and coho salmon in the Yakima Basin. Status is defined as the abundance, distribution, and size structure of an NTT and change in status as a deviation from baseline conditions (prior to supplementation). A change in status does not indicate causation, but a decline in status must occur if supplementation did have a negative impact. Therefore, changes in status can be used to trigger further studies to identify the causes of changes in monitoring variables. In some cases, changes in status and whether a change occurred from supplementation can be determined simultaneously. This occurs when control sites are available and are currently monitored. Based upon baseline data, the most statistically powerful and economically feasible techniques were assembled into monitoring prescriptions.

Monitoring prescriptions were developed to maximize our sensitivity to detect changes. Previous work identified the difficulty in detecting changes using abundance monitoring alone (Ham and Pearsons 2000). Subsequent work identified improvements in detecting changes by using alternative measures (Ham and Pearsons 2001). These newer measures include spatial overlap, analogs, predation indexing, and modeling (Table 7). Each of these measures can improve the detectability of changes in NTT status although each may have shortcomings.

Spatial overlap is used for species that are located upstream of target species acclimation sites during the baseline period (e.g., bull trout and cutthroat trout). Increases in distribution of the target species can result in spatial overlap with NTT resulting in the potential for impacts. If overlap never occurs, then impacts are assumed to be negligible. However, if overlap does occur, then changes to status must be investigated. NTT that have similar ecological responses to interactions are used as analogs if they significantly improve the ability to detect changes. The use of analogs is particularly useful when NTT are rare and dispersed, and therefore difficult to sample. The potential liability of using analogs is that one must assume that impacts to the analog are the same as to an NTT. Finally, modeling of flow can be used to reduce the amount of unexplained inter-annual variation in an NTT response variable. If the parameters used in the model are not actually causing the changes observed in the status of NTT (e.g., spurious correlations), then the model may give a false interpretation. We follow the risk containment approach for detecting and protecting NTT described by Ham and Pearsons (2001).

The wide range in life cycles of the NTT, river conditions and flow necessitate the use of sampling techniques ranging from snorkeling, backpack electrofishing, dam counts, and trapping to boat electrofishing. Abundance, size structure, and distribution (status) are determined annually at the sites indicated in Figure 1 and Tables 4, 5, and 6. Techniques have been previously described by Ham and Pearsons (2000), but are briefly described here for completeness.

The spatial overlap between bull trout and supplemented salmon in the North Fork of the Teanaway River is inventoried by snorkeling. The entire rearing area of bull trout is snorkeled at night to determine if any salmon are present. Night snorkeling is recommended as the best low impact sampling strategy for bull trout. During September two divers, equipped with underwater lights, move upstream and count all fish observed and estimate the length of all bull trout encountered.

Population estimates in upper Yakima tributary sites are based on single-pass backpack electrofishing. In tributary streams, a crew of three to six people electrofish 200-m long index sites during the day with a backpack electrofisher (Table's 5 and 6). A single electrofishing pass is performed and attempts are made to net all visible fish. Netted fish are held in perforated buckets in the stream. All fish are anesthetized, identified to species, and the lengths and weights of salmonids are recorded. For other taxa, the fish are counted, grouped into age classes (adult, juvenile, age 0+), weighed as a group, and an average weight calculated. An estimate of salmonid abundance is calculated by expanding the first pass count by the median capture ratio established for each site during the baseline monitoring phase. The capture ratio is the number of fish captured on the first pass divided by a multiple-removal estimate of the number of fish in the site (Zippen 1958).

In the mainstem of the upper Yakima River, a crew of two people electrofish 4.2 –7.4 km long sites at night with a driftboat electrofisher (Table's 4 and 5). Two types of abundance measures are made: one type is generated from mark-recapture methods (rainbow trout) and the other is a visual estimate intended to index abundance (e.g. mountain whitefish, suckers). During the electrofishing passes, all fish are identified visually and trout are netted. Trout are marked and released. One week later another electrofishing pass is made to determine the proportion of marked and unmarked salmonids. Visual estimates during electrofishing are analogous to snorkel counts because the fish are only observed and never handled. An estimate of salmonid abundance is determined by maximum likelihood estimators using standard mark-recapture techniques (Mark-Recapture for Windows 1997, Version 5.0 Beta, Montana Fish, Wildlife, and Parks).

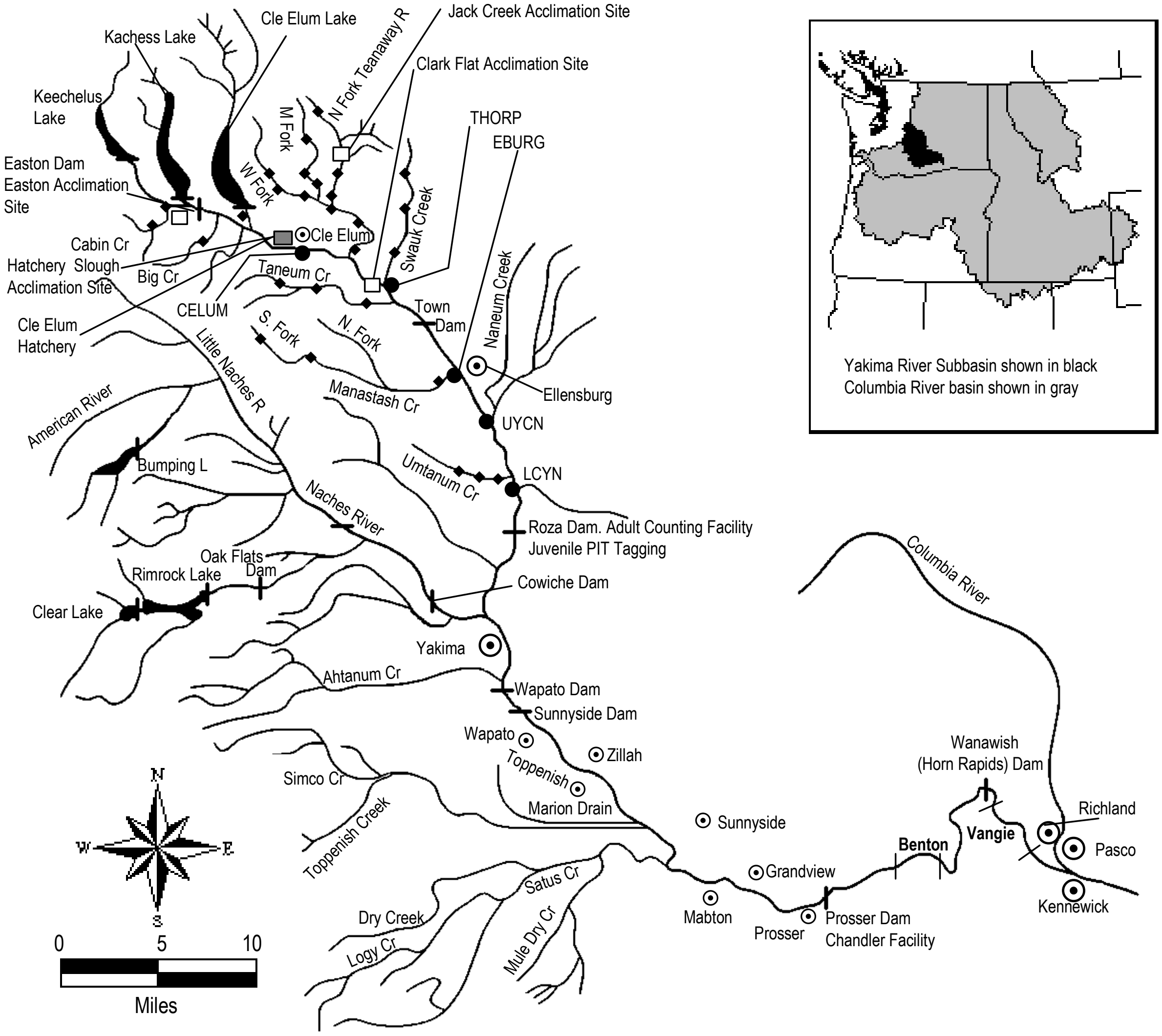

Figure 1. Yakima River Basin. Tributary survey sites (•), upper river mainstem survey sites (●), lower river mainstem survey sites (|~|), and major cities (⊙).

Smolt counts of spring and fall Chinook salmon are made at the Chandler facilities and are provided by the Yakama Nation (Fast et al. 1991). Estimates of the total number of fish passing Prosser Dam are made by expanding the number of fish collected in the trap by a flow/entrainment relationship. Additionally, migratory lamprey species are enumerated passing the Chandler facilities and are recorded by the Yakama Nation.

Size structure of an NTT was quantified as the mean length (salmonids), weight (non-salmonids in tributaries), or percent of fish visually observed that are adults (mountain whitefish and suckers), of fish collected in sites used to describe abundance. We accepted the convention that electrofishing efficiencies for small fish were generally poor (Reynolds 1983) and for salmonids, focused our analysis on fish greater than 79 mm fork length. Non-salmonids in the tributaries are grouped into life-stages and weighed as separate groups.

Distribution of an NTT is quantified as the weighted area of index sites that contain a minimum number of an NTT (Table 8). Index sites are weighted based on the length of stream that they represent. Most of the sites that are used to determine distribution are the same as those used to describe abundance. However, some exceptions do occur (Table's 7 and 8). These exceptions are included to provide a greater area in which to assess distributional changes.

Abundance estimates for residualized hatchery spring chinook salmon present in the Yakima River from mid September to mid October for release years 1999-2003 were calculated utilizing boat electrofishing capture efficiencies. We calculated our capture efficiencies of similar sized rainbow trout in mainstem Yakima River electrofishing index sites utilizing mark-recapture methods. The rainbow trout capture efficiencies were applied to the number of hatchery spring chinook netted during the mark runs in each index section. A final estimate of the hatchery spring chinook residual abundance was expanded to the reach scale based on reach length (Table 3).

Table 3. Estimated abundance of hatchery origin spring chinook salmon residuals in Upper Yakima River mainstem reaches.

	Yakima River Reach					
Year	LCYN	UCYN	EBURG	THORP	CELUM	Total
1999	87	127	98	69	0	381
2000	168	127	26	714	89	1,124
2001	6,581	1,594	736	1,665	0	10,576
2002	294	0	131	64	0	489
2003	1,008	290	245	57	0	1,601
Avg.	1,628	428	247	514	18	2,834
SD	2,793	660	284	703	40	4,356

Analysis

Changes in NTT status or surrogate measures were detected with a one-tailed t-test and results were expressed as log percent changes from baseline (Tables 9, 10, and 11). The numerical values for abundance, size, and distribution are also presented for interpretation of changes and comparison with historical values. The statistical power was calculated to determine the probability of committing a type II statistical error with the one-tailed t-test using the program Statistica (StatSoft, Inc., 2001).

Since we have observed a decline in steelhead size that is outside of the containment objectives, the decline triggered further investigations to determine if the observed change is related to supplementation activities. We compared the average size of rainbow trout in treatment and reference streams both before and after supplementation releases in the Teanaway basin. We removed the years 1991-1995 from the baseline data set to remove any effect that hatchery steelhead releases in the area may have had on rainbow trout size during the baseline period. We used a student's t-test to determine if there was any difference in rainbow trout size between treatment and reference streams during both periods.

Results

Status monitoring of NTT, after five years of supplementation releases, indicated that many of the parameters we measure increased slightly and all, except steelhead size were within predetermined containment objectives (Table 12). Rainbow trout in the mainstem, which is also the analog for steelhead, increased in abundance, decreased slightly in size, and remained unchanged in distribution. The slight decrease in size (-1%) is outside of the containment objective for steelhead but was within the containment objective for rainbow trout. However, comparisons of the rainbow trout size in index areas that were stocked and those that were not stocked indicated that supplementation was not the cause of the decline in size (Figure 2). Differences between the average size of rainbow trout in treated and untreated areas near the Jack Creek acclimation facility in the Teanaway Basin were not significantly different from baseline conditions (Figure 3, $P>0.05$). The status of rainbow trout in the tributaries was similar to baseline conditions. This result is expected because the spatial overlap of salmon and trout was low in all of the tributaries except the North Fork of the Teanaway River. The primary impact detection strategy for bull trout and cutthroat trout is overlap in the distribution between these species and supplemented salmon (Table 7). There was no overlap between salmon and bull trout in our index sites, which indicated that supplementation activities did not negatively change the status of this species. Additionally, there was no overlap between supplemented salmon and cutthroat trout in tributary index monitoring sites during 2003. However, cutthroat trout and supplemented spring chinook exhibited overlap in distribution in mainstem Yakima River areas. The extent of the overlap in the mainstem Yakima River decreased with decreasing elevation (figure 4) and response variables have not declined in the overlap areas ($P>0.05$). Speckled dace abundance remained below baseline levels ($P<0.05$), but was still well within our containment objectives. Similarly, sculpin spp. abundance in tributary streams remained below baseline levels (-12%, $P<0.05$), although this difference was also within our predetermined containment objectives.

Since the predation index used to monitor interactions with Pacific lamprey, fall chinook, leopard dace, and sandroller was discontinued in 2003, status monitoring for leopard dace and sandroller was discontinued and secondary monitoring strategies were adopted for Pacific lamprey and fall chinook. The status of Pacific lamprey and fall chinook salmon has not declined during the post-supplementation period. Simple abundance monitoring for Pacific lamprey indicated there was no significant decline in abundance (P>0.05). Pacific lamprey size was not monitored or reported on. Finally, fall chinook salmon abundance and size have remained above baseline levels (Table's 9 and 10).

Statistical tests of monitoring prescriptions before and after supplementation are presented in Tables 9, 10, and 11. Actual values (unmodelled and untransformed) are presented for abundance (Table 13), size (Table 14), and distribution (Table 15).

Table 4. Latitude and longitude positions in degrees, minutes, seconds (D°M'S") and Universal Transverse Mercator (UTM) of mainstem Yakima River monitoring sites. Sites are represented by upstream (top) and downstream (bottom) boundaries.

Site Name	Lat. (D°M' S")	Long. (D°M'S")	UTM E	UTM N
LCYN top	46° 48' 48.32"	120° 27' 7.59"	10 0694374 E	5187583 N
LCYN bottom	46° 47' 32.32"	120° 27' 23.94"	10 0694103 E	5185226 N
UCYN top	46° 55' 34.76"	120° 30' 57.68"	10 0689101 E	5199971 N
UCYN bottom	46° 53' 42.55"	120° 30' 10.93"	10 0690200 E	5196540 N
EBURG top	47° 0' 24.13"	120° 35' 50.40"	10 0682637 E	5208710 N
EBURG bottom	46° 58' 47.39"	120° 34' 9.24"	10 0684866 E	5205790 N
THORP top	47° 7' 8.01"	120° 43' 37.97"	10 0672402 E	5220882 N
THORP bottom	47° 5' 58.73"	120° 42' 8.48"	10 0674351 E	5218799 N
CELUM top	47° 11' 19.15	120° 55' 56.81"	10 0656629 E	5228202 N
CELUM bottom	47° 10' 24.96"	120° 51' 36.48"	10 0662153 E	5226677 N

Table 5. Monitoring site names and abbreviations used in text and locations.

Site Name	Abb.	Location
Upper Yakima Tributaries		
Cabin Creek	CAB-1	4.4 km up Cabin Creek Rd. from junction with Railroad Av. (Easton)
Domerie Creek	DOM-A	0.9 rkm above Cle Elum River
Manastash Creek	MAN-3	Buck Meadows Campground at Old Quartz Mountain Trailhead
Middle Fork	MFT-1	Middle/West Fork Teanaway Rd. 1.6 km above junction with Teanawa
Teanaway River	MFT-2	Middle/West Fork Teanaway Rd. 5.1 km above junction with Teanawa
	MFT-3	Middle/West Fork Teanaway Rd. 8.5 km above junction with Teanawa
North Fork.	NFT-1	Teanaway Rd., km 13.5
Teanaway River	NFT-2	Teanaway Rd., km 19.3
	NFT-3	Teanaway Rd., km 33.1
	NFT-A	Bottom of site is 30 m below trail #1383 bridge
	NFT-B	350 m above Eldorado Creek (near Camp Wahoo)
Stafford Creek	STF-A	Bottom of site is 50 m above Standup Creek
	STF-B	Bottom of site is 200 m below confluence with Bear Creek
Swauk Creek	SWK-1	First bridge crossing on private road. at Milepost 95.6 on Highway 10
	SWK-2	Highway 97, Milepost 151.75
	SWK-3	Highway 97, Milepost 158
Taneum Creek	TAN-1	On West Taneum Rd. 1.9 km above Thorp Cemetery Rd.
	TAN-2	On West Taneum Rd. 11.9 km above Thorp Cemetery Rd.
	TAN-3	N. Fork Taneum Rd. 0.7 km above S. Fork Meadows junction
	TAN-A	10.2 road miles up West Taneum Road, 650 m below Forks
	TAN-B	10.2 road miles up West Taneum Road, 1550 m above Forks
Umtanum Creek	UMT-1	0.4 rkm above confluence with Yakima River
	UMT-	3.4 rkm above confluence with Yakima River
	UMT-2	0.4 km downstream from Umtanum Creek/Durr Road crossing
West Fork	WFT-1	Confluence with Middle Fork Teanaway
Teanaway River	WFT-2	On West Fort Teanaway Rd. 5.6 km above junction with Teanaway Rd.
	WFT-3	400 km below West Fork Trailhead Rd.
Upper Yakima Mainstem		
Cle Elum	CELUM	Swift Water Campground to 300 m above the Teanaway game ramp
Ellensburg	EBURG	Top of the riffles below the Ellensburg KOA to 200 m above Reinhart
Lower Canyon	LCYN	Road mile 11.7 on Highway 821 to 200 m upstream of the Slab takeout
Thorp	THORP	Anderson Homestead to 200 m above the Thorp highway bridge
Upper Canyon	UCYN	150 m above Wilson Creek to 150 m above Bighorn takeout

Table 6. Latitude and longitude positions in degrees, minutes, seconds (D°M’S”) and Universal Transverse Mercator (UTM) of tributary monitoring sites.

Site Name	Lat. (D°M' S")	Long. (D°M'S")	UTM E	UTM N
CAB-1	47° 14' 08.72"	121° 13' 8.72"	10 0637264 E	5232889 N
DOM-A	47° 14' 12.73"	121° 04' 6.83"	10 0646186 E	5233296 N
JUN-A	47° 20' 47.43"	120° 52' 36.08"	10 0660376 E	5245857 N
MAN-A	46° 59' 30.35"	120° 50' 57.30"	10 0663534 E	5206494 N
MAN-1	46° 59' 39.45"	120° 35' 26.81"	10 0683178 E	5207346 N
MAN-2	46° 59' 5.70"	120° 50' 10.93"	10 0664534 E	5205759 N
MAN-3	47° 2' 21.09"	120° 57' 36.41"	10 0654967 E	5211538 N
MFT-1	47° 15' 53.54"	120° 53' 53.19"	10 0659003 E	5235343 N
MFT-2	47° 16' 51.06"	120° 55' 50.37"	10 0656493 E	5238451 N
MFT-3	47° 17' 57.47"	120° 57' 42.06"	10 0654094 E	5240439 N
MST-1	47° 10' 58.40"	120° 49' 29.80"	10 0664791 E	5227783 N
MST-2	47° 13' 28.32"	120° 48' 15.61"	10 0666222 E	5232454 N
MST-3	47° 15' 6.65"	120° 52' 27.53"	10 0660842 E	5235343 N
NFT-1	47° 16' 53.10"	120° 51' 53.86"	10 0661460 E	5238648 N
NFT-1.5	47° 17' 24.67"	120° 51' 35.38"	10 0661821 E	5239633 N
NFT-2	47° 18' 41.97"	120° 51' 31.40"	10 0661839 E	5242021 N
NFT-2.5	47° 19' 36.74"	120° 51" 21.15"	10 0662008 E	5243718 N
NFT2.75	47° 19' 56.28"	120° 51' 22.71"	10 0661959 E	5244321 N
NFT-3	47° 24' 18.24"	120° 55' 56.68"	10 0655994 E	5252251 N
NFT-A	47° 22' 51.58"	120° 53' 11.52"	10 0659528 E	5249669 N
NFT-B	47° 24' 54.67"	120° 56' 20.50"	10 0655465 E	5253363 N
STF-A	47° 21' 20.08"	120° 50' 0.84"	10 0663605 E	5246955 N
STF-B	47° 21' 48.82"	120° 48' 32.18"	10 0665439 E	5247894 N
SWK-1	47° 7' 58.30"	120° 44' 51.39"	10 0670811 E	5222390 N
SWK-2	47° 13' 45.90"	120° 41' 46.96"	10 0674379 E	5233233 N
SWK-3	47° 19' 15.08"	120° 41' 9.65"	10 0674862 E	5243417 N
TAN-1	47° 5' 7.71"	120° 46' 8.35"	10 0669340 E	5217078 N
TAN-2	47° 6' 46.99"	120° 52' 58.95"	10 0660600 E	5219901 N
TAN-3	47° 6' 37.20"	120° 56' 9.09"	10 0656601 E	5219492 N
TAN-A	47° 6' 43.34"	120° 55' 45.11"	10 0657101 E	5219695 N
TAN-B	47° 6' 30.69"	120° 56' 11.71"	10 0656551 E	5219290 N
UMT-1	46° 51' 27.63"	120° 29' 49.55"	10 0690785 E	5192389 N
UMT-1.5	46° 51' 57.13"	120° 32' 4.26"	10 0687904 E	5193210 N
UMT-2	46° 52' 27.65"	120° 33' 58.07"	10 0685466 E	5194076 N
WIL-A	47° 9' 54.93"	120° 30' 38.79"	10 0688655 E	5226535 N
WFT-1	47° 15' 25.52"	120° 53' 56.00"	10 0658967 E	5235875 N
WFT-2	47° 15' 51.79"	120° 57' 11.25"	10 0654842 E	5236577 N
WFT-3	47° 16' 11.37"	120° 58' 36.13"	10 0653043 E	5237135 N

Table 7. Primary monitoring detection strategy, sampling method, abundance, and size structure index sites, and if environmental models were used to assess changes to NTT.

NTT	Detection Strategy/Method	Index Sites	Model[4]
Bull trout	Spring chinook salmon spatial overlap/Snorkeling	North Fork Teanaway River, river km 8.0 to 14.2 from the confluence of Jungle Creek	No
Cutthroat trout	Spring chinook salmon spatial overlap/Electrofishing	DOM-A, MAN-3, NFT-3, NFT-A, NFT-B, STF-A, STF-B, SWK-2, SWK-3, TAN-2, TAN-3, TAN-A, TAN-B, WIL-A	No
Pacific lamprey	Status/Trapping	Chandler juvenile facility annual counts	No
Steelhead	Status (Year 1 rainbow trout as analogs)/Electrofishing	CELUM, THORP, EBURG, UCYN, LCYN	Yes[4]
Fall chinook salmon	Status/Trapping	Chandler juvenile facility annual counts	No
Mountain sucker	Status: all suckers as analogs/ Visuals during Electrofishing	CELUM, THORP, EBURG, UCYN, LCYN	Yes[2]
Rainbow trout-mainstem	Status/Electrofishing	CELUM, THORP, EBURG, UCYN, LCYN	Yes[3]
Spring chinook salmon	Status/Trapping	Chandler juvenile facility annual counts	No
Mountain whitefish	Status (subadult)/Visuals during Electrofishing	CELUM, THORP, EBURG, UCYN, LCYN	Yes[2]
Rainbow trout – tributaries	Status/Electrofishing	MFT-1,2,3; NFT-1,2,3; SWK-1,2,3; TAN-1,2,3; and WFT-1,2,3	No
Longnose dace	Status/Electrofishing	MFT-1, MFT-2, NFT-1, SWK-2	Yes[1]
Speckled dace	Status/Electrofishing	SWK-1, UMT-1, UMT-1.5, UMT-2	Yes[2]
Sculpins	Status/Electrofishing	MFT-1,2,3; NFT-1,2,3; SWK-1,2,3; TAN-1,2,3; UMT-1,1.5,2; and WFT-1,2,3	No
Suckers	Status/Visuals during Electrofishing	CELUM, THORP, EBURG, UCYN, LCYN	Yes[2]

Based on Bureau of Reclamation flow data from stations at the; [1]Teanaway River near Cle Elum, WA., [2]Yakima River near Umtanum, WA., and [3]Yakima River near Cle Elum, WA.
[4]Models are only applied to abundance estimates, not size or distribution.

Table 8. Index sites and threshold values for distribution monitoring of NTT.

NTT	Distribution Index Sites	Threshold for Use
Bull trout	North Fork Teanaway River, river km 8.0 to 14.2 from the confluence of Jungle Creek	≥ 1 fish/site
Cutthroat trout	NFT-3; TAN-3	≥ 10 fish/km
Steelhead	Year 1 rainbow trout in CELUM, THORP, EBURG, UCYN, LCYN	≥ 100 fish/km
Rainbow trout – mainstem	CELUM, THORP, EBURG, UCYN, LCYN	≥ 100 fish/km
Mountain whitefish	CELUM, THORP, EBURG, UCYN, LCYN	≥ 40 fish/km
Rainbow trout – tributaries	CAB-1; MFT-1,2,3; NFT-1,2,3; SWK-2,3; TAN-1,2,3; UMT-1,2 and WFT-1,2,3	≥ 25 fish/km
Longnose dace	CAB-1; MFT-1,2,3; NFT-1,2; SWK-2,3; WFT-1,2,3	≥ 30 fish/km
Speckled dace	MFT-1; SWK-1; UMT-1, 1.5, 2; WFT-1	≥ 60 fish/km
Sculpins	CAB-1; MFT-1,2,3; NFT-1,2,3; SWK-1,2,3; TAN-1,2,3; UMT-1,1.5,2 and WFT-1,2,3	≥ 100 fish/km
Suckers	CELUM, THORP, EBURG, UCYN, LCYN	≥ 40 fish/km
	SWK-1; UMT-1,1.5,2	≥ 10 fish/km

Table 9. Monitoring prescription abundance baseline mean, standard deviation, number of baseline survey years, post-supplementation average (n=5, 1999 - 2003 surveys), t-statistic, p-level, and power analysis where α is set to 0.05 or 0.10. Significant declines in abundance (P<0.05) are identified with an asterisk.

NTT	Baseline	(n)	Post	t	p	0.05	0.10
Bull trout	2.00 ± 0.00	(3)	2.00 ± 0.00	--	1.000		
Cutthroat trout	2.02 ± 0.38	(9)	2.21 ± 0.51	-0.79	0.222		
Pacific lamprey	1.96 ± 0.63	(6)	2.30 ± 0.35	-0.10	0.158		
Steelhead	1.99 ± 0.11	(8)	2.15 ± 0.21	-1.83	0.047		
Fall chinook salmon	4.85 ± 0.44	(16)	5.14 ± 0.66	-1.14	0.133		
Mountain sucker	1.92 ± 0.07	(6)	1.84 ± 0.10	1.57	0.075	51	67
Rainbow trout-main	1.99 ± 0.11	(8)	2.15 ± 0.21	-1.83	0.047		
Spring chinook salmon	5.14 ± 0.24	(16)	5.21 ± 0.31	-0.49	0.316		
Mountain whitefish	1.65 ± 0.11	(6)	1.80 ± 0.02	-2.72	0.012		
Rainbow trout – tribs.	2.44 ± 0.14	(9)	2.53 ± 0.09	-1.36	0.101		
Longnose dace	1.99 ± 0.10	(7)	2.01 ± 0.15	-0.30	0.385		
Sculpins	1.98 ± 0.14	(7)	1.80 ± 0.16	2.18	0.027*		
Speckled dace	1.98 ± 0.15	(6)	1.65 ± 0.18	3.40	0.004*		
Suckers	1.92 ± 0.07	(6)	1.84 ± 0.10	1.57	0.075	51	67

Table 10. Monitoring prescription size baseline mean, standard deviation, number of baseline survey years, post-supplementation average (n=5, 1999 - 2003 surveys) t statistic, p-level, and power analysis where α is set to 0.05 or 0.10.

NTT	Baseline	(n)	Post	t	p	0.05	0.10
Bull Trout	2.00 ± 0.00	(3)	2.00 ± 0.00	--	1.000		
Cutthrout trout	1.87 ± 0.09	(9)	1.85 ± 0.06	0.43	0.337	10	18
Steelhead	2.08 ± 0.03	(9)	2.06 ± 0.02	1.62	0.065	40	55
Mountain sucker	1.64 ± 0.13	(6)	1.52 ± 0.13	1.63	0.115	45	61
Fall chinook salmon	1.92 ± 0.03	(8)	1.93 ± 0.01	-0.93	0.185		
Rainbow trout-main	2.08 ± 0.03	(9)	2.06 ± 0.02	1.62	0.065	40	55
Spring chinook-salmon	1.78 ± 0.02	(8)	1.76 ± 0.07	0.93	0.187	53	68
Mountain whitefish	1.45 ± 0.27	(6)	1.34 ± 0.09	0.88	0.201	16	27
Rainbow trout – tribs.	2.13 ± 0.01	(9)	2.13 ± 0.02	-0.27	0.396		
Longnose dace	0.87 ± 0.09	(6)	1.00 ± 0.01	-3.08	0.006		
Sculpins	0.76 ± 0.05	(6)	0.88 ± 0.04	-4.45	0.001		
Speckled dace	0.53 ± 0.10	(6)	0.60 ± 0.06	-1.53	0.081		
Suckers	1.64 ± 0.13	(6)	1.52 ± 0.13	1.63	0.115	45	61

Table 11. Monitoring prescription distribution baseline mean, standard deviation, number of baseline survey years, post-supplementation average (n=5, 1999-2003 surveys) t statistic, p-level, and power analysis where α is set to 0.05 or 0.10. Significant declines in distribution (P<0.05) are identified with an asterisk.

NTT	Baseline	(n)	Post	t	p	0.05	0.10
Bull trout	2.00 ± 0.00	(3)	2.00 ± 0.00	-	1.000		
Cutthroat trout	1.79 ± 0.00	(2)	1.18 ± 0.01	-1.12	0.157		
Rainbow trout-main	2.00 ± 0.00	(8)	2.00 ± 0.00	-	1.000		
Mountain whitefish	2.00 ± 0.00	(6)	2.00 ± 0.00	-	1.000		
Rainbow trout – tribs.	4.99 ± 0.02	(7)	4.99 ± 0.03	-0.19	0.423	8	15
Longnose dace	1.89 ± 0.06	(7)	1.86 ± 0.07	0.77	0.230	20	32
Sculpins	1.96 ± 0.02	(6)	1.84 ± 0.10	2.79	0.010*		
Speckled dace	1.94 ± 0.09	(6)	1.89 ± 0.02	1.25	0.121	21	34
Suckers	4.56 ± 0.06	(6)	4.53 ± 0.07	0.69	0.252	17	28

Table 12. Percent change in post-supplementation NTT status relative to baseline for monitoring prescriptions. Values were calculated as a percentage for each year, rounded, and the average taken. The minimum and maximum percent change for the post-supplementation period is also listed. The containment objective (CO) is listed for each non-target taxa.

	Post Supplementation Change, (%)									
	CO	Abundance			Size			Distribution		
		Ave.	Min	Max	Ave.	Min	Max	Ave.	Min	Max
Bull trout	0	0	0	0	0	0	0	0	0	0
Cutthroat trout	0	7	-29	22	0	-7	8	5	4	6
Pacific lamprey	0	17	-2	41						
Steelhead	0	8	0	27	-1	-2	0	0	0	0
Fall chinook	-5	6	-5	28	1	0	1			
Mtn. sucker	-5	-4	-11	0	35	-22	65	-1	-3	0
Rainbow – main	-10	8	0	27	-1	-2	0	0	0	0
Spring chinook	-10	2	-6	9	-1	-6	4			
Mtn. whitefish	-40	8	7	9	12	4	20	0	0	0
Rainbow – tribs	-40	5	1	9	0	-1	2	0	0	0
Longnose dace	-65	1	-7	13	15	13	17	-2	-8	2
Speckled dace	-85	-17	-25	-2	15	2	32	-3	-3	-1
Sculpins	-90	-12	-19	-9	16	10	22	-6	-13	0
Suckers	-90	-4	-11	0	35	-22	65	-1	-3	0

Table 13. Actual values for abundance (fish/km, unless otherwise indicated). N is the number of baseline samples. Mountain sucker are too rare for quantitation.

NTT	Baseline	(n)	Post Supplementation
Bull trout	22 ± 19[1]	(3)	28 ± 24 fish
Cutthroat trout	138 ± 90	(9)	226 ± 121 /km
Pacific lamprey	198 ± 241[2]	(6)	254 ± 201 migrants
Steelhead	63,247 ± 38,259[3]	(16)	35,484 ± 6,210 smolts
Fall chinook salmon	108,973 ± 102,976[3]	(16)	409,664 ± 711,572smolts
Rainbow trout-main	147 ± 43	(8)	242 ± 130 age 1/km
Spring chinook-salmon	158,355 ± 75,216[3]	(16)	195,508 ± 122,175 smolts
Mountain whitefish	116 ± 34	(6)	157 ± 22 subadult/km
Rainbow trout – tribs.	287 ± 89	(9)	345 ± 73 /km
Longnose dace	59 ± 22[4]	(7)	55 ± 13 /site
Sculpins	63 ± 27[4]	(7)	36 ± 5 /site
Speckled dace	104 ± 45[4]	(6)	52 ± 27 /site
Suckers	158 ± 42	(6)	146 ± 34 /km

[1] Number of fish, [2] Number of migrants, [3] Number of smolts, [4] Number/site

Table 14. Actual values for size. N is the number of baseline samples. Mountain sucker are too rare for quantitation. Size of Pacific lamprey is not determined.

NTT	Baseline	(n)	Post Supplementation
Bull trout	275 ± 134 mm	(3)	213 ± 90 mm
Cutthroat trout	155 ± 15 mm	(9)	151 ± 11 mm
Steelhead	166 ± 30 mm	(6)	158 ± 40 mm
Fall chinook salmon	83 ± 5 mm	(8)	86 ± 2 mm
Rainbow trout-main	201 ± 8 mm	(9)	195 ± 5 mm
Spring chinook-salmon	128 ± 3 mm	(8)	125 ± 10 mm
Mountain whitefish	32 ± 15% subadults	(6)	22 ± 4 % subadults
Rainbow trout – tribs.	133 ± 3 mm	(9)	134 ± 6 mm
Longnose dace	8 ± 2 g	(7)	10 ± 0 g
Sculpins	6 ± 1 g	(7)	8 ± 0 g
Speckled dace	3 ± 1 g	(6)	4 ± 1 g
Suckers	45 ± 13 % adults	(6)	33 ± 12 % adults

Table 15. Actual values for percent distribution. N is the number of baseline samples.

NTT	(n)	Baseline	Post Supplementation
Bull trout	(3)	26 ± 17 %	26 ± 10 %
Cutthroat trout	(2)	66 ± 32 %	77 ± 2 %
Rainbow trout-main	(8)	100 ± 0 %	100 ± 0 %
Rainbow trout - tribs.	(9)	95 ± 4 %	95 ± 6 %
Longnose dace	(7)	79 ± 10 %	74 ± 12 %
Sculpins	(7)	91 ± 5 %	71 ± 16 %
Speckled dace	(6)	89 ± 16 %	78 ± 3 %
Suckers	(6)	80 ± 11 %	78 ± 15 %

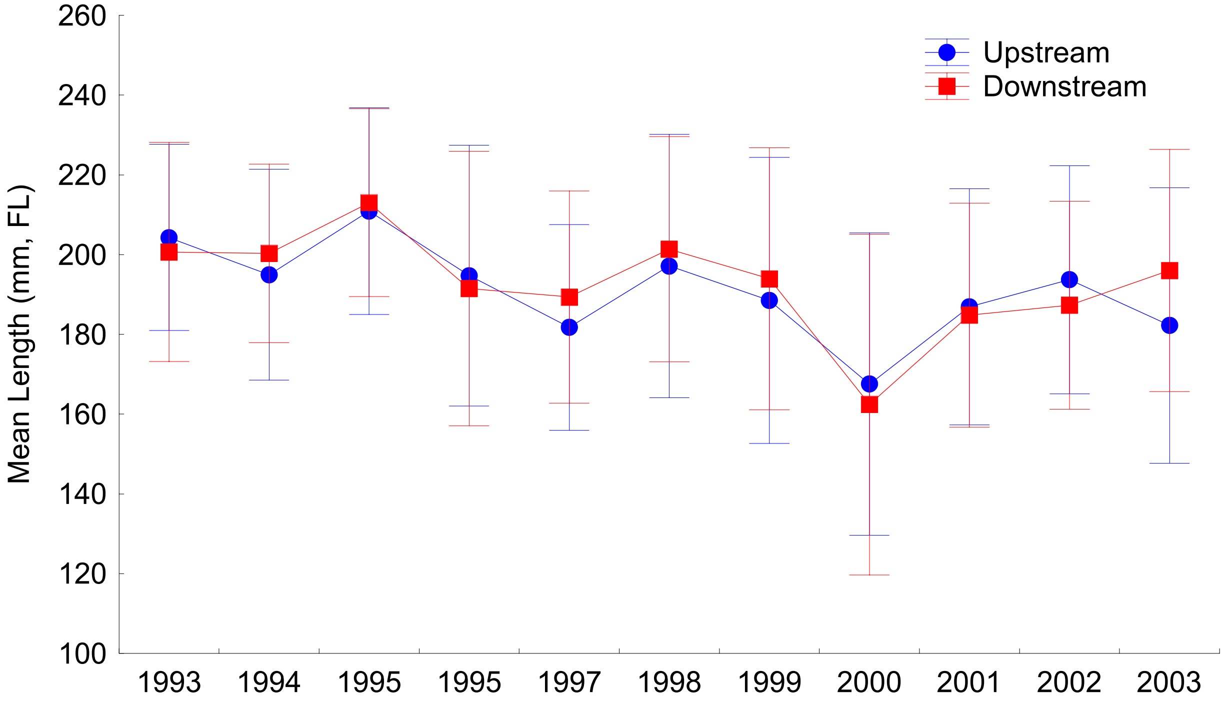

Figure 2. Mean fork length (FL) of Yakima River rainbow trout (<250mm) above and below the Clark Flats acclimation site discharge channel. Error bars represent 1 standard deviation.

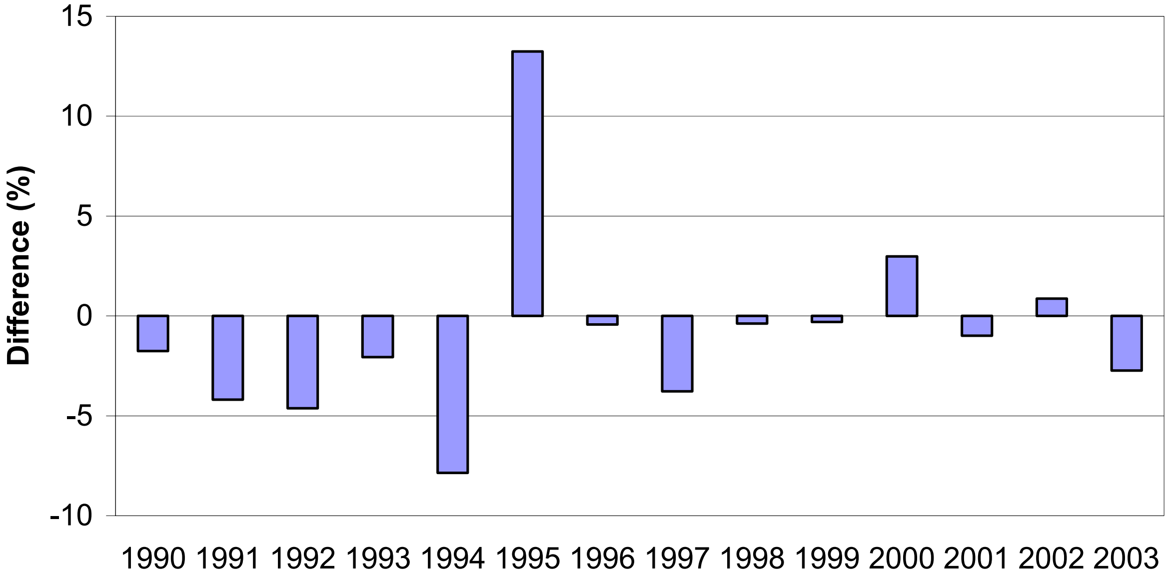

Figure 3. Difference in rainbow trout lengths between treatment and reference streams in the Teanaway Basin.

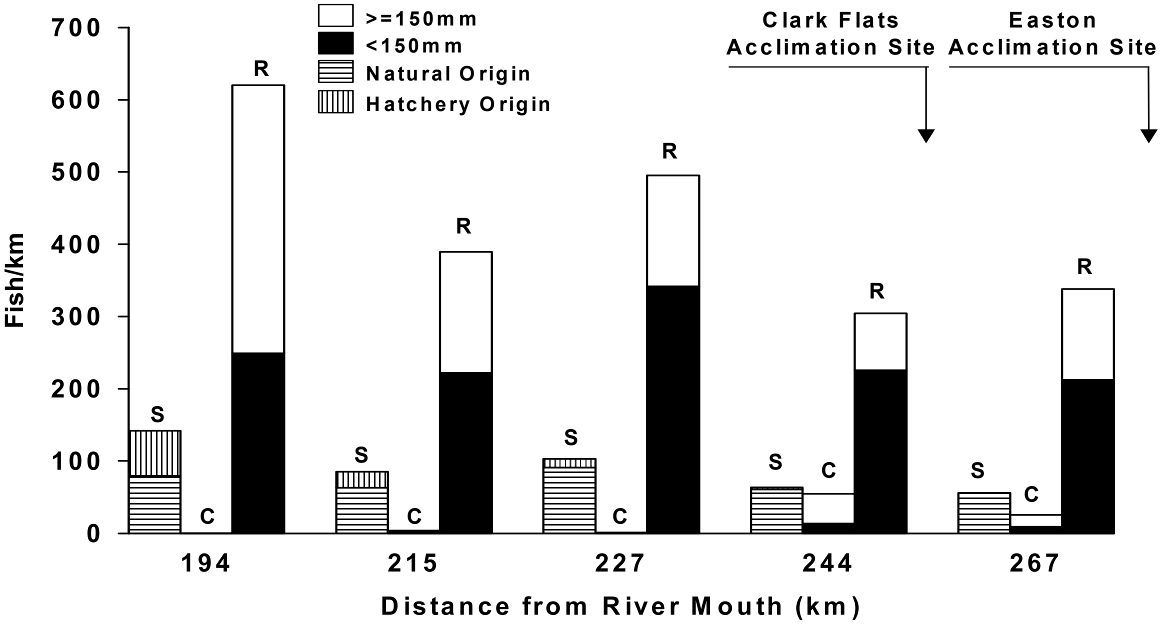

Figure 4. Cutthroat trout (C), spring chinook (S), and rainbow trout (R) overlap in distribution in the mainstem Yakima River during the post-supplementation period, 1999-2003. River kilometers are measured to the middle of a sampling section.

Discussion

The detection of few negative impacts to NTT status that could be related to supplementation is likely due to: 1) the lack of spatial overlap between salmon and NTT; 2) the impacts of hatchery yearlings were balanced or exceeded by the benefits (e.g., ecological release) of reducing the progeny of naturally produced fish or increased nutrients provided by hatchery effluent or higher numbers of adult salmon; 3) benign interaction or density dependent benefits of higher numbers of smolts, and; 4) the low statistical power of our tests. Six of 14 NTT had limited or no overlap with hatchery salmon (bull trout, tributary cutthroat trout, tributary rainbow trout, longnose dace, speckled dace, and sculpins). However, the opportunity for overlap existed. For example, hatchery steelhead that were released in 1994 into the North Fork of the Teanaway River migrated upstream into areas containing bull and cutthroat trout (McMichael and Pearsons 2001). Steelhead were released into the river very close to the area where salmon were released. Hatchery spring chinook were not observed upstream from the release site in the North Fork of the Teanaway River during 2002, but were observed upstream 2.5 km in 2000,1.4 km in 2001, and 0.5 km in 2003. However, none were observed in index areas containing bull trout and very few were observed in tributary index areas containing cutthroat trout. We assume that minimal or no spatial and temporal overlap precludes significant ecological interactions.

In areas where overlap occurred, impacts that might have been caused by releasing hatchery smolts were balanced or exceeded by the benefits (e.g., ecological release) of reducing the progeny of naturally produced fish or increased nutrients provided by hatchery effluent or higher numbers of adult salmon. The NTT that likely fit into this category are cutthroat trout and rainbow trout in the mainstem, steelhead, mountain whitefish, and suckers. Most of the NTT that spatially overlapped salmon showed positive or no changes in status and all of the NTT, except steelhead, were within the containment objectives. The reduction of naturally produced target fish in the river was the result of taking fish that would have spawned in the river into the hatchery. However, the large return of wild fish in 2000 and the combination of supplementation and wild returns in 2001 produced progeny that had ample opportunity to interact with NTT. Thus, the proportion of the run that was taken for broodstock was relatively high in 1997, 1998, 1999, and low in 2000, 2001, and 2002. During years when high proportions of the run were taken for broodstock, more ecological release was likely to occur. We expected that impacts would be most noticeable during the period 2001-2002 because of the large numbers of salmon released (Type I interactions) and the increased natural production of supplementation origin salmon (Type II interactions).

Although we have observed decreases in the size of steelhead (rainbow trout as an analog) 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. 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 would primarily interact with smolt releases from Jack Creek and

Easton acclimation facilities, the Cle Elum Hatchery slough coho releases, and lower numbers of residualized chinook 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. In addition, we would expect that the size of rainbow trout below the release site in the North Fork Teanaway River would be smaller than those in comparable sites where target fish were not released. We did not however, detect any significant decreases in the size of rainbow trout in these high-density areas suggesting that the decreased size is not related to supplementation. Furthermore, the steelhead size index began to decline in the baseline period before hatchery fish were released. Finally, the length of steelhead smolts measured at the Chandler Juvenile facility have increased during the post-supplementation period, although these are a mix of stocks, originating from the Naches and upper Yakima River, that may not be representative of upper Yakima steelhead. This information leads us to believe that the decline in steelhead lengths is most likely the result of natural variation or some other factor.

Although our trend monitoring suggests that the decline in the steelhead size index is not related to supplementation, a more rigorous evaluation of the size index should be implemented. During 2003, we conducted a prospective power analysis and sample size calculations to determine the sample size necessary to detect impacts to rainbow trout growth based on back-calculated length-at-age data collected from scales. We collected scale samples from 677 mainstem Yakima River rainbow trout and sent them to the Washington Department of Fish and Wildlife's scale lab where they were aged and scale measurements were recorded. To date, approximately 50% of the scales have been analyzed. Preliminary observations suggest that the length of age 1+ rainbow trout (analog for steelhead) were different during the baseline and supplementation periods. However, the final length at age analysis from rainbow trout scales collected during 2003 will be completed and reported on at a later date.

Cutthroat trout in the mainstem Yakima River have exhibited overlap with both naturally produced and hatchery released salmon. The greatest overlap occurred at higher elevations and decreased with decreasing elevation. Large sized cutthroat trout in these areas could benefit from supplementation if they eat hatchery smolts or naturally produced salmon, or utilize food produced from returning hatchery adults. Large cutthroat trout have been shown to be piscivorous in other lotic systems in the Pacific Northwest (Lowry 1966). However, smaller size classes of cutthroat trout may not have the predatory size advantage of their larger counterparts. Thus, the smaller fish may not directly consume hatchery produced salmon. In addition, smaller cutthroat trout may not be able to compete for resources as well as the larger cutthroat trout due to their smaller size. The low abundance of cutthroat trout in mainstem index areas makes it difficult to evaluate their status. Additionally, our cutthroat trout parameter estimates in the Yakima Basin have exhibited high amounts of natural variation complicating rapid and sensitive impact detection for this species (Ham and Pearsons 2000).

Large numbers of spring chinook salmon did not migrate to the ocean after release (residuals) and may have interacted with NTT (Table 3). Approximately 22% of the total spring chinook salmon production precocially matured and likely residualized in the river (Larsen et al. 2004). Larsen et al. (2004) estimated that 85,640, 133,141, 166,815, and 184,398 precocially mature yearlings were released into the upper Yakima Basin during 1999, 2000, 2001, and 2002 respectively. Residual chinook have been concentrated below the Clark Flats acclimation site and some were observed below the Easton acclimation site during 1999 and 2000. Other high concentrations were observed below the acclimation site in the North Fork Teanaway River during 2000. During 2001, precocials were more abundant and were more evenly distributed throughout the Yakima River and the North Fork Teanaway River than in the two previous years. Fewer residualized spring chinook salmon were observed in 2002 and 2003 than in previous years. However, the observed residuals were larger than wild conspecifics and modal sized rainbow trout which could confer dominance status. They also ate similar prey items, and food appeared to be limiting growth to rainbow trout and wild conspecifics (James et al. 1999; WDFW unpublished data). Previously, we found that residual hatchery spring chinook salmon negatively impacted the growth of wild spring chinook salmon in small enclosures in the Teanaway Basin (WDFW unpublished data).

Since the predation index was discontinued during 2002, we initiated secondary monitoring strategies that were previously identified for fall chinook and Pacific lamprey during 2003 (Ham and Pearsons 1999). Briefly, simple status monitoring replaced interactions monitoring for these species. Lamprey abundance, fall chinook abundance, and fall chinook size have been recorded by the Yakama Nation at the Chandler facility and will now be used as the primary monitoring strategy for these species. The liability in shifting our monitoring focus to secondary strategies is the resulting reduction in our impact detection ability for these NTT. Ham and Pearsons (1999) noted that the predation index provided large benefits in monitoring fall chinook interactions but only marginal improvements in detecting impacts to Pacific lamprey abundance. Therefore, discontinuing the predation index may inhibit our ability to detect impacts to fall chinook status but may not substantially reduce our monitoring ability for Pacific lamprey. After the first year of evaluating the status of these species using the secondary impact detection strategies, it did not appear that there was any decline in the response variables measured. Finally, we have no economically feasible alternative for monitoring the status of leopard dace and sandroller, and will not be monitoring their status in the future.

The discussion of impacts should be tempered by a realistic view of the natural variability of most indicators of impact. This variability limits the ability to detect impacts, even after 5 years of stocking (Ham and Pearsons 2000). The lack of impacts to NTT that spatially overlap salmon is, at this stage, insufficient evidence to draw conclusions about what interactions are or are not important.

Management Implications

We are using 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 change in status that is greater than a containment objective, then we attempt to determine if the change 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 is outside of the containment objectives is steelhead size. The decline in steelhead size is unlikely to be due to supplementation and therefore do not require risk containment actions. If substantive declines continue, then more refined methods of determining causation should be implemented. Monitoring prescriptions described in Table 7 appear to be working as they were designed and should continue to be implemented during 2004. As previously described, monitoring prescriptions related to the predation index will not be implemented in future years. As a result, sandroller and leopard dace status will not be monitored while secondary impact detection strategies will continue to be implemented for fall chinook salmon and Pacific lamprey. The monitoring prescriptions appear, thus far, to be relatively insensitive to impacts that were caused by factors other than supplementation. For example, bull trout abundance and size has decreased after the onset of supplementation in the Yakima Basin. However, because distributional overlap between bull trout and hatchery fish has not been observed, the decrease was not attributed to supplementation. Finally, the Building Stage of supplementation began in 2002. This stage is likely to be the one where the risk of impacts is the highest (Pearsons 2002).

Implementation of strategies to limit the number of precocially mature salmon entering the natural environment would decrease the risk of failing to meet containment objectives, including those for steelhead. By reducing the number of these precociously mature salmon, both direct and indirect undesirable interactions with NTT will be reduced.

Acknowledgements

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Chapter 2

Comparison of Single vs. Multiple Pass Electrofishing Effort to Monitor Fish Populations in Wadeable Streams

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Abstract

We endeavored to evaluate whether increasing site length and keeping sampling effort constant could improve our ability to detect changes in the abundance, size structure, and species richness of fish in tributaries to the Yakima River, Washington. The efficacy of using a single electrofishing pass to quantify stream salmonids was evaluated by comparing the abundance estimates generated from the first electrofishing pass in block-netted stream sections to traditional maximum likelihood estimates for the same sites and years. The numbers of fish captured during the first electrofishing pass in multiple-removal estimates were significantly correlated with the resulting maximum likelihood estimates for those sites ($r=0.98$, $P<0.001$). Our data indicated that our capture probabilities generated annually in mid-elevation multiple-removal sites between the years 1999 and 2003 have not significantly differed from median capture probabilities established for those sites during the years 1990-1998 ($G=3.21$, $df=99$, $P>0.05$). Removing block-nets from a sub-set of electrofishing index sites between 1998 and 2003 did not bias the estimates from those sites when they were compared to estimates generated from traditional multiple-removal estimates from adjacent sites during the same period ($F=0.05$, $df=1$, $P>0.05$). Reallocating our sampling effort from multiple-pass electrofishing in 100 meter long index sites to single-pass electrofishing in 200 meter long index sites provided a 20% decrease in the annual variation around our abundance estimates while not significantly altering the population estimates themselves ($t=1.65$, $df=124$, $P>0.05$). We conducted a power analysis and found that we could decrease our detectable effect size to population abundance by 26% after five years of monitoring. Extending our site lengths did not appear to provide detection benefits when evaluating impacts to the size structure of rainbow trout, *Oncorhynchus mykiss*. However, we improved our ability to detect rare species such as bull trout, *Salvelinus confluentus*, spring chinook salmon parr, *Oncorhynchus tsawytscha*, and cutthroat trout, *Oncorhynchus clarki*, by 75%, 19%, and 17%, respectively. We recommend that multiple-removal sampling be used when stream fish monitoring programs intend to monitor for less than five years and that long term programs consider adopting single-pass electrofishing methods.

Introduction

Biologists are often faced with designing and implementing monitoring plans to detect changes in resident salmonid populations and their associated species. Frequently, monitoring programs will be constrained by the amount of money and time available. In such situations, monitoring programs should use sampling strategies that maximize the ability to detect changes in population and community parameters while minimizing expense and effort.

The abundance, size structure, and distribution of fish in lotic systems are often monitored annually using electrofishing removal-depletion techniques (Zippin 1958). Multiple electrofishing passes through a site are performed to produce a declining trend in catch with each additional pass. The obvious benefit of this technique is the ability to determine the capture efficiency of the targeted fish and, ultimately, an estimate of their abundance through maximum likelihood estimation procedures. However, removal-depletion techniques exhibit relatively poor qualities when expense and effort are evaluated (Lobon-Cervia and Utrilla 1993; Kruse et al. 1998).

Several authors have commented on the increasing demand for efficient and reliable methods to quantify stream fish populations (Crozier and Kennedy 1994; Jones and Stockwell 1995; Mitro and Zale 2000) and to characterize stream fish communities (Meador et al. 2003; Reynolds et al. 2003). Kruse et al. (1998) followed similar methods proposed by Lobon-Cervia and Utrilla (1993) to enumerate stream trout populations when a relationship between a single electrofishing pass catch could be related to a three-pass abundance estimate. Lobon-Cervia and Utrilla (1993) cautioned against employing an unknown capture probability to expand single-pass capture data into population estimates. Mitro and Zale (2000) endorsed the use of a mean capture probability model to estimate rainbow trout, *Oncorhynchus mykiss,* abundance in large geographical areas from several individual electrofishing events. Several innovative sampling schemes have been proposed to produce the best quantitative information (in terms of data quantity and quality) for the effort expended in small stream fish data collection (Crozier and Kennedy 1994; Jones and Stockwell 1995; Patton et al. 2000). Furthermore, others have commented on the tradeoff between collecting precise data at a limited number of sampling sites, and collecting less precise data over a larger geographic area when a fixed amount of resources are available (Hankin and Reeves 1988; Kruse et al. 1998). Simonson and Lyons (1995) suggested that reduced monitoring effort could be beneficial for monitoring temporal trends in fish abundance at single stations. Reallocating effort from intensive sampling at few sites to sampling at many sites may increase uncertainty in point estimates at the site level but may increase the utility of the information collected at the basin-wide scale. Some of the benefits provided by these truncated sampling schemes may include the ability to sample larger areas with restricted financial support.

In this investigation, we assess a single-pass electrofishing protocol used to enumerate fish in small tributaries of the Yakima River, Washington. We evaluate the results associated with reallocating sampling effort from intensive multiple-removal sampling in 100-meter (m) long index sites to single-pass electrofishing in 200-m long sites. We evaluated our methods with respect to abundance and size of rainbow trout, species richness, and detection of rare species. Our objectives were to determine (1) if single-pass electrofishing provided a reliable indicator of true rainbow trout abundance, (2) if single-pass sampling of doubled lineal stream distances decreased the variability of our annual estimates of rainbow trout abundance without introducing bias, (3) any effect that wetted channel widths may have on rainbow trout abundance and size, and estimates of species richness, (4) if sampling longer stream sections increased our species detections, (5) the effect of removing block-nets, and (6) if doubling our lineal sampling distances provided increased statistical power in detecting impacts to the abundance or size structure of rainbow trout and in detecting impacts to species richness.

Methods

Abundance, size, and distribution information of rainbow trout were collected at 29 monitoring sites in 13 tributary streams in the Yakima Basin, Washington, annually beginning in 1990. Although most sites were sampled once annually, some sites were discontinued and new sites added such that a maximum of 36 sites in 11 tributaries were sampled annually during the duration of this study. Twelve index sites were established between 1997 and 2003 for monitoring cutthroat trout *Oncorhynchus clarki* and spring chinook salmon *Oncorhynchus tshawytscha* populations but also provided abundance and size information for rainbow trout and additional species richness data (Table 1). Lineal sampling distances were increased from 100-m to 200-m between 1997 and 2003. The additional 100-m were added to the top of the original 100-m index site when possible to maintain the original 100-m data series. Most tributaries sampled contained at least two 200-m sites. Criteria used for site selection has been previously described by McMichael et al. (1992). Multiple-removal electrofishing techniques were used in all sites prior to 1998. In 1998, low- and high-elevation sites were sampled using a single electrofishing pass. However, the mid-elevation site in each tributary was sampled via multiple-removal methods during both periods to evaluate temporal changes in capture probability. Thus, we sampled approximately 33% of our sites using multiple-removal methods. A minimum of two-pass removal sampling was used in all removal sites in most years. All sites were sampled between the months of June and September when stream flows were reduced to summer base flow levels. The physical attributes of each site are presented in Table 2.

Table 1. Yakima Basin tributary monitoring sites and response variables measured in each site. Stream sections are numbered sequentially upstream: 1 = lowest elevation, 3 = highest elevation; A and B sections are cutthroat trout monitoring sections that provide rainbow trout information. Stream codes are as follows: CAB = Cabin Creek, JUN = Jungle Creek, MAN = Manastash Creek, MFT = Middle Fork Teanaway River, MST = Mainstem Teanaway River, NFT = North Fork Teanaway River, STF = Stafford Creek, SWK = Swauk Creek, TAN = Taneum Creek, UMT = Umtanum Creek, WFT = West Fork Teanaway River.

Stream section	Abundance	Size	Block-net vs. no net	Capture Probability	Species Richness	Rare Fish Distribution
CAB 1	X	X		X	X	X
JUN A	X	X			X	
MAN 1	X	X		X	X	
MAN 2	X	X			X	
MAN A	X	X			X	
MANAA	X	X			X	
MFT 1	X	X	X		X	X
MFT 2	X	X	X	X	X	X
MFT 3	X	X	X		X	X
MST 1	X	X			X	X
MST 2	X	X		X	X	X
MST 3	X	X			X	X
NFT 1	X	X	X		X	X
NFT 1.5	X	X			X	
NFT 2	X	X	X	X	X	X
NFT 2.5	X	X			X	
NFT 2.75	X	X			X	
NFT 3	X	X	X		X	X
NFT A	X	X			X	
NFT B		X			X	
STF A	X	X			X	
STF B	X	X			X	
SWK 1	X	X			X	X
SWK 2	X	X		X	X	X
SWK 3	X	X			X	X
TAN 1	X	X	X		X	X
TAN 2	X	X	X	X	X	X
TAN 3	X	X	X		X	X
TAN A	X	X			X	
TAN B	X	X			X	
UMT 1	X	X			X	X
UMT 1.5	X	X		X	X	X
UMT 2	X	X			X	X
WFT 1	X	X	X		X	X
WFT 2	X	X	X	X	X	X
WFT 3	X	X	X		X	X

Table 2. Physical descriptions of tributary rainbow trout index monitoring sites in the Yakima Basin, Washington. Physical measurements were recorded once annually on the day of sampling and are presented as the average for the period 1998-2003. Cutthroat trout monitoring sites are not described. Stream sections are defined in Table 1.

Stream Section	Elevation (m)	Gradient (%)	Q (m^3/s)	Mean width (m)	Pool area (%)	Mean pool depth (m)	Riffle area (%)	Mean riffle depth (m)	Run area (%)	Mean Run depth (m)
CAB1	2360	2.5	0.41	6.3	7	0.25	69	0.21	24	0.18
MAN1	1700	2.0	0.14	3.9	27	0.41	63	0.14	10	0.13
MAN3	4400	2.6	0.16	4.4	23	0.33	65	0.17	12	0.13
MFT1	2340	2.8	0.08	5.5	26	0.43	47	0.14	27	0.24
MFT2	2500	5.0	0.17	5.8	20	0.25	56	0.21	25	0.24
MFT3	2760	3.0	0.31	6.3	14	0.31	71	0.24	15	0.22
MST1	1800	1.5	0.49	10.1	15	0.15	56	0.17	29	0.30
MST2	2000	2.0	0.64	10.5	11	0.22	32	0.14	57	0.22
MST3	2100	1.5	0.95	14.1	27	0.11	39	0.18	34	0.27
NFT1	2340	2.3	0.60	10.4	10	0.22	49	0.24	41	0.34
NFT2	2560	3.0	0.73	8.2	8	0.24	54	0.27	39	0.32
NFT3	3620	3.3	0.41	5.9	7	0.20	86	0.25	8	0.18
SWK1	1900	3.0	0.05	4.6	20	0.21	52	0.10	28	0.10
SWK2	2400	2.6	0.09	4.5	21	0.38	58	0.19	21	0.23
SWK3	2960	2.9	0.05	3.3	18	0.31	57	0.38	25	0.18
TAN1	2040	2.4	0.27	5.4	18	0.50	61	0.21	21	0.29
TAN2	2660	1.9	0.27	5.3	15	0.35	66	0.21	19	0.27
TAN3	3000	1.9	0.17	3.9	19	0.34	65	0.20	16	0.17
UMT1	1540	3.1	0.02	2.3	23	0.29	63	0.10	14	0.15
UMT1.5	1760	2.1	0.02	2.6	18	0.20	57	0.11	25	0.18
UMT2	2040	2.5	0.01	2.4	40	0.38	37	0.12	23	0.18
WFT1	2340	3.0	0.15	5.7	35	0.72	42	0.14	23	0.15
WFT2	2460	2.1	0.14	5.7	22	0.29	44	0.14	34	0.23
WFT3	2620	3.5	0.20	6.9	9	0.30	76	0.20	15	0.18

The single-pass sampling method started with measuring the site length (two contiguous 100-m reaches) and the distribution of 19 L buckets at every 25-m interval within the site. Lineal site lengths were measured along the thalweg. The upstream ends of the sampling sites were extended or retracted slightly such that the uppermost boundary of the site was situated at a natural break between habitat units (typically in a shallow riffle area). A three or four member crew used a Smith Root model 12 electrofisher, set at either 300 or 400 volts straight DC depending on water conductivity, to capture stream fishes in the site. Two crewmembers with dip nets remained downstream from the electrofishing anode and netted stunned fishes. All netted fish were transferred into buckets within each 25-m interval until the upstream end of the site was reached.

Multiple-removal/depletion protocols used in all sites from 1990 to 1997 and mid-elevation sites from 1998 to 2003 required the additional steps of installing block-nets and conducting multiple electrofishing passes in the site. First, 6.4 mm knotless nylon block-nets were installed at the downstream and upstream ends of each 100-m site. Multiple electrofishing passes were then performed in each 100-m reach until at least 50% of the trout population was depleted from the previous pass or fewer than 2 fish were captured. The site was considered successfully depleted in two passes if the 50% removal goal was met in the field. We accepted the convention that electrofishing efficiencies for small fish were generally poor (Reynolds 1983) and focused our removals and analysis on salmonids greater than 79 mm fork length. We attempted to net all fish that were observed when the electric current was applied to the water. All captured fish were returned to the stream near their point of capture at the end of sampling.

For both single-pass and multiple-removal estimates, the abundance of the target taxa in each site was determined by expanding the number of fish captured on the first electrofishing pass by a species and site-specific median capture ratio that was established between 1990 and 1998. Median capture ratios were calculated by dividing the first pass catch by a maximum likelihood estimate of abundance for a particular site. They were assumed to reflect the range of temporal variability in our capture efficiencies in our sites during the period 1990 to 1998. We have used the abbreviation CREST to identify estimates generated from median capture ratios. The median capture ratios for each site were compared to capture probabilities generated from two- and three-pass removal sampling to validate their use. Multiple-removal maximum likelihood estimates of abundance were calculated using the computer software program MicroFish 3.0 (Van Deventer and Platts 1985).

We periodically conducted three-pass multiple-removal sampling to test the assumption that trout catchability remained constant during a sampling event. The computer program CAPTURE (White et al. 1982) was used to identify violations of catchability assumptions. Although White et al. (1982) recommend evaluating catchability at the $P<0.20$ level for precision purposes, we considered declines in capture probabilities significant when $P<0.05$. We discarded parameter estimates from sampling that failed to produce a declining catch of fish and considered them unreliable (3% of three-pass removals). We excluded parameter estimates that had significantly different capture probabilities among electrofishing passes as were identified by the program CAPTURE (White et al. 1982). Thus, 13% of three-pass removals were excluded from the analysis of abundance but were used for the analysis of size, species richness, and rare species detections.

The utility of a single electrofishing pass to predict trout abundance was evaluated by comparing the number of rainbow trout greater than 79 mm fork length captured in the first pass of a multiple-removal estimate against the resulting maximum likelihood estimate for that site. We used a $\log_{10}$ transformation to correct for the heteroscedastic nature of the data. Log_{10} transformed maximum likelihood abundance estimates were also compared to $\log_{10}$ transformed CREST estimates to evaluate the use of median capture ratios. Comparisons and relationships were made using a simple least squares analysis. To evaluate the effects of using blocknets, we compared the CREST estimates from low- and high-elevation sites for the period 1990 to 1997 (block-nets) versus the period 1998 to 2003 (no block-nets) to the CREST estimates from sites that maintained multiple-removal sampling protocols during the latter period. We used a two-factor analysis of variance (ANOVA) to test the assumption that CREST estimates would not differ significantly in the years when block-nets were employed and when they were not.

We evaluated the efficacy of a 200-m sampling reach to reflect population level trends in abundance and size structure of rainbow trout, species richness, and distribution of rainbow trout, bull trout, *Salvelinus confluentus*, cutthroat trout, and spring chinook salmon parr. First, the benefits associated with doubling our lineal sampling distances in 1997 were evaluated by comparing the mean annual abundance and variation in CREST estimates for both the original 100-m and the 200-m section with a modified variance ratio test for dependent samples (Snedecor and Cochran 1980). Next, we compared the size structure of trout in the 100-m and 200-m sites with a students t-test for dependent samples. Lastly, species accumulation curves for a single 100-m pass, a single 200-m pass, and 100-m double pass sampling were constructed to determine the species detection rates associated with increasing stream sampling distances. A Wilcoxon matched pairs test was used to determine if sampling extended site lengths provided substantial benefits in detecting rare or disbursed species over 100-m site lengths.

We calculated the minimum detectable impacts to rainbow trout abundance and size, and species richness that could be detected when either 100-m or 200-m were used for long-term monitoring within established index sites. Detectable impacts were calculated using the equations presented by Zar (1999) for two-sample hypotheses. We followed the procedures of Ham and Pearsons (2000) and calculated the percent impact to parameter estimates that could be detected with increasing numbers of post impact surveys. Acceptable Type I and Type II statistical error rates (a and β) were set at 0.1 (power = $1-\beta$). Critical values used for the impact calculations were based on a one tailed students t-test. Detectable impact curves were constructed to facilitate comparisons between 100-m and 200-m long sampling reaches. The difference between the detectable impact curves generated for the 100-m and 200-m sites were standardized for the percent impact levels.

The influence of stream size, as measured by average wetted channel width, on estimates of rainbow trout abundance and size, and species richness was evaluated. During the period 1997 to 2003, two contiguous 100-m reaches were sampled within index monitoring sites. We illustrate the difference in our parameter estimates for the two contiguous reaches versus stream size in scatter plot diagrams. To evaluate the dispersion of each response variable with increasing stream size (as measured by channel width), we lumped the parameter estimates into bins based on stream size and determined the variation of each bin. The bins were based on 20 channel width increments and variability was reported as the standard deviation of the values in each bin. Least squares regression techniques were used to judge the effect of stream size on the response variables measured.

We used our 200-m long sampling sites to evaluate changes in the accuracy of our parameter estimates with 25-m incremental increases in lineal sampling distances. The parameter estimates generated in each 200-m site were defined as being the “true” estimates and were set as benchmark values. Finally, parameter estimates were generated for each 25-m interval and were reported as a percent of the “true” 200-m estimate.

Results

Overall, removal efficiencies in multiple-removal sites sampled between 1999 and 2003 were not significantly different from the median capture ratios established between the years 1990 and 1998 as indicated by a goodness of fit test (Sokal and Rolf 1995; G=3.21, df=99, P>0.05). On average, removal efficiencies from all multiple-pass removal sampling (1990 to 2003) were high with 75% of rainbow trout removed in the first electrofishing pass. Evaluation of three-pass catchability assumptions indicated violations in 13% (12/89) of removals when evaluated at P<0.05. Six of the 12 estimates that failed catchability assumptions resulted from no fish captured in the third electrofishing pass.

The number of rainbow trout captured during the first electrofishing pass was significantly correlated with maximum likelihood estimates generated in 200-m multiple-removal sites (r=0.98, P<0.001, Figure 1). Annual CREST estimates did not change when block-nets were removed from our sampling sites when compared to mid-elevation sites that maintained block-nets and removal techniques (F=0.05, df=1, P>0.05) as indicated in Figure 2.

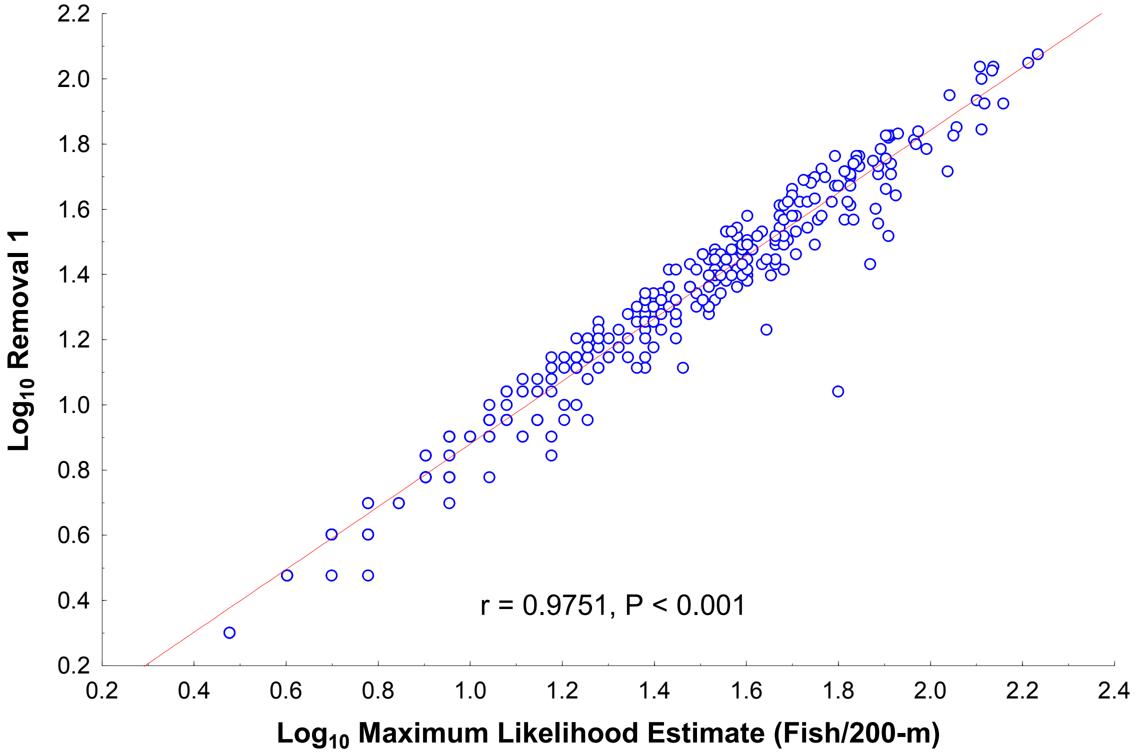

Figure 1. Maximum likelihood estimate generated from multiple-removal sampling versus the number of rainbow trout removed during the first electrofishing pass in 200-m sites. Data has been $\log_{10}$ transformed.

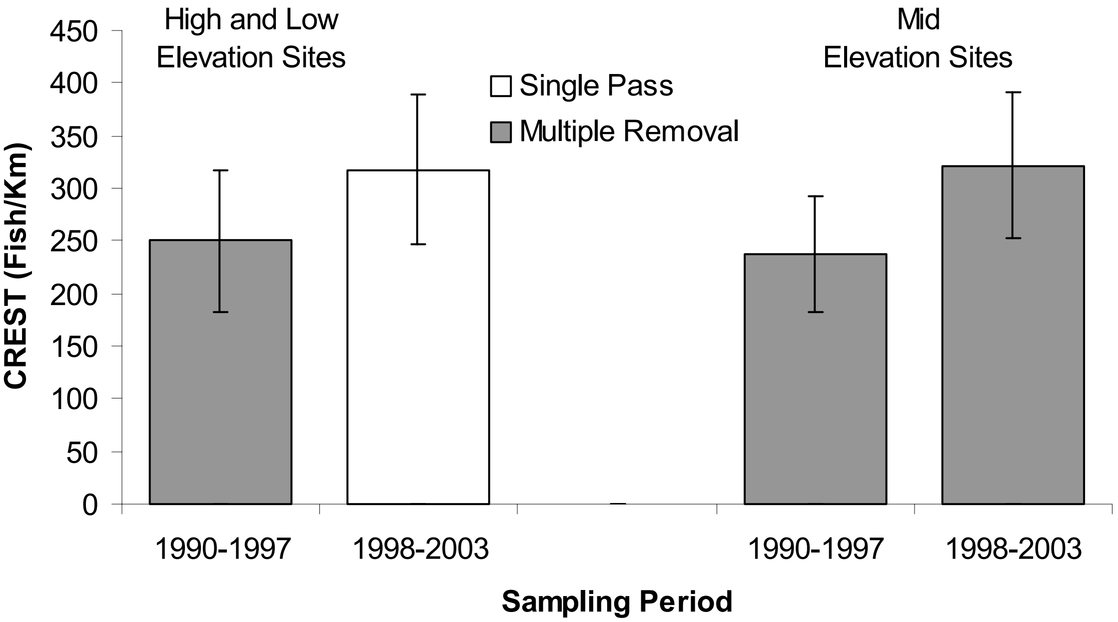

Figure 2. Comparison between single-pass and multiple-removal estimates. The high and low-elevation sites were sampled via multiple-removal methods from 1990 to 1997 (colored bar) and single pass methods thereafter (transparent bar). Mid-elevations sites were sampled via multiple-removal techniques during both periods. Error bars represent the 95% confidence interval.

Sampling extended 200-m sites with both single-pass and removal methods decreased the annual variation of the population estimates. Results from a modified variance ratio test indicate that the variation in the population estimates from 200-m sites was decreased by 20% over the estimates generated for the 100-m sites (r_{ds}=0.23, df=124, P<0.05). The estimates of abundance generated for the 100-m and 200-m reaches were not significantly different when evaluated with a students t-test for dependent samples (t=1.65, df=124, P>0.05).

Stream size, as measured by multiples of wetted channel width, did not significantly affect estimates of rainbow trout density (fish/m^2) or size, but did affect estimates of species richness. The difference in estimated densities between the 100-m and 200-m reach were not significantly correlated with wetted channel widths (r=0.12, P>0.05, Figure 3). However, the variation in the density estimates was correlated with wetted channel width such that variability significantly increased with decreasing stream size (r=0.92, P<0.05). Estimates of trout size did not appear to be related to wetted channel widths (r=0.05, P>0.05, Figure 4). Additionally, the variability associated with our size estimates was not correlated with wetted channel widths (r=0.81, P>0.05). Stream size appeared to be a significant factor affecting estimates of species richness (r=0.17, P<0.05, Figure 5). Generally, variation in species richness estimates decreased with decreasing stream size although the highest levels of variability were observed within the site lengths that measured 80-100 times the mean wetted width.

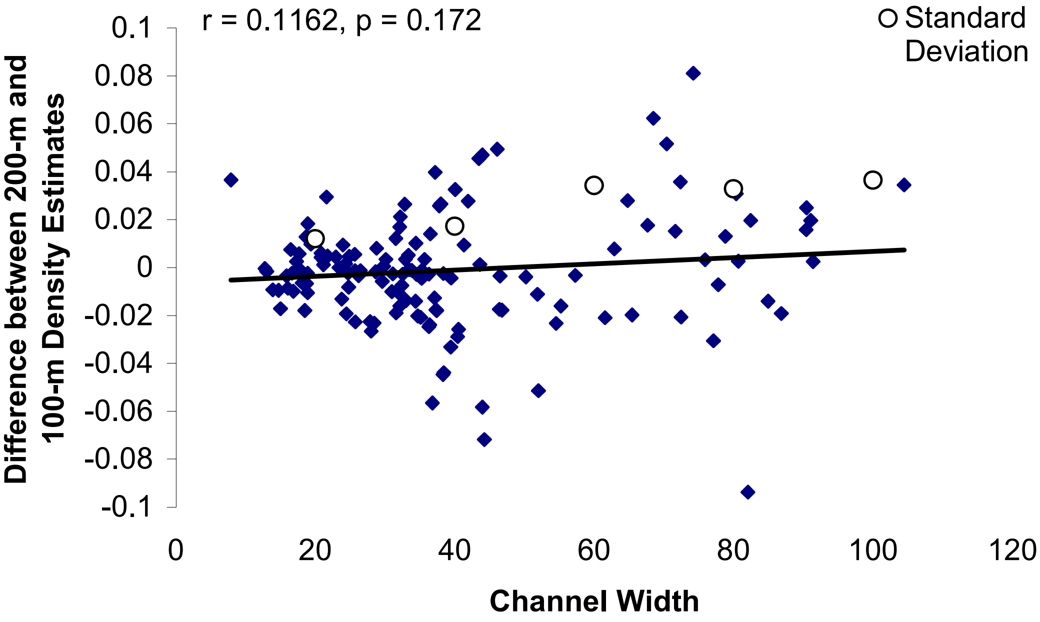

Figure 3. Difference between estimated rainbow trout density (200-m – 100-m, fish/m^2) versus channel width. The differences were grouped into 20 channel width increments (bins) and the standard deviations calculated for the bins are presented as open circles.

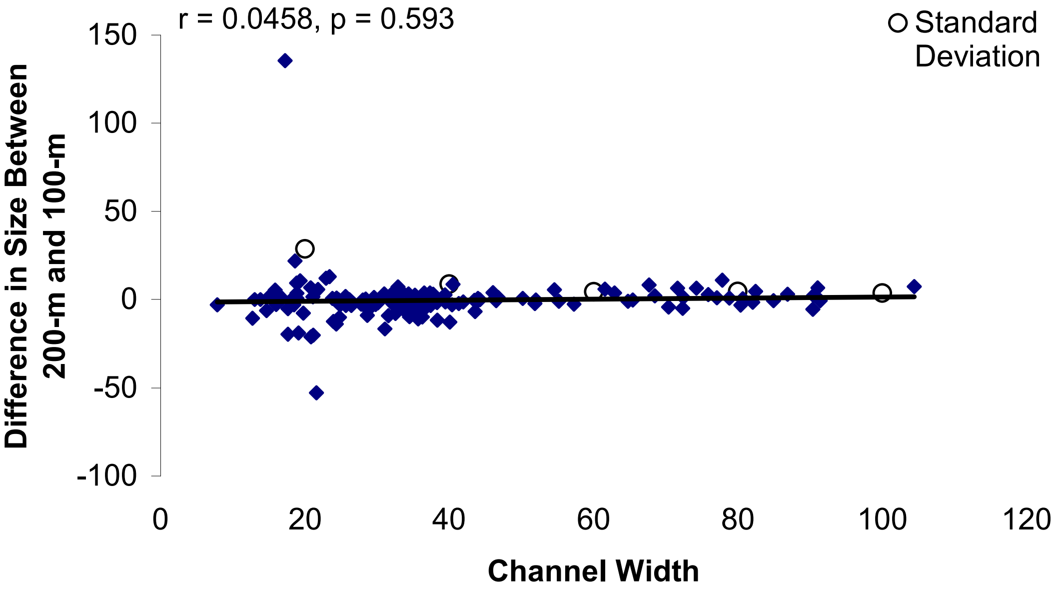

Figure 4. Difference between estimated rainbow trout size (200-m – 100-m, mm fork length) versus channel width. The differences were grouped into 20 channel width increments (bins) and the standard deviations calculated for the bins are presented as open circles.

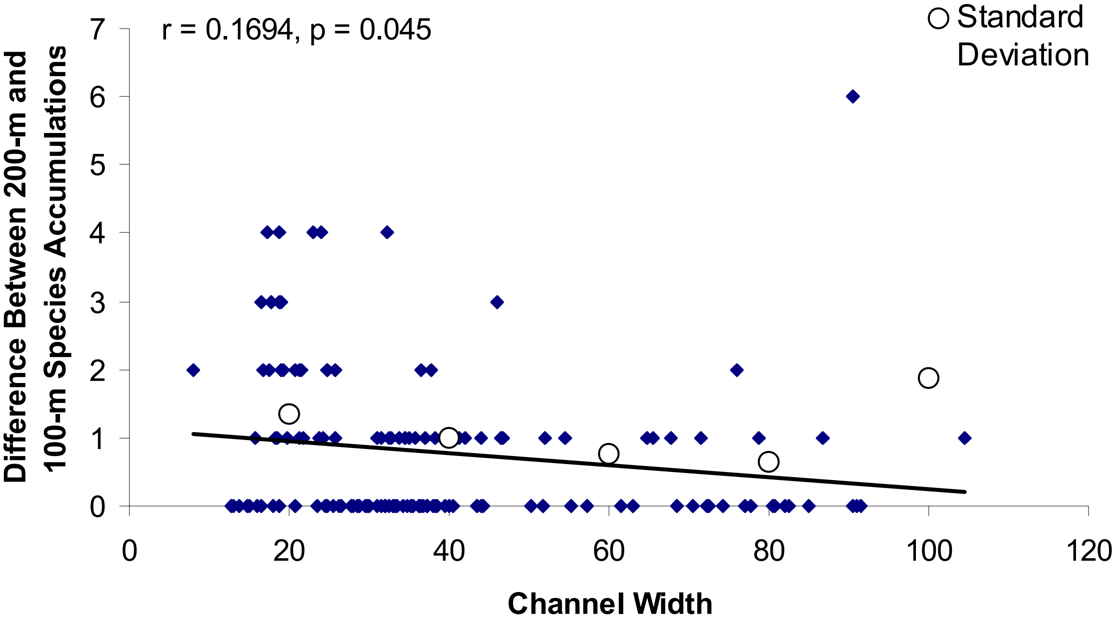

Figure 5. Difference between species accumulations (200-m – 100-m species counts) versus channel width. The differences were grouped into 20 channel width increments (bins) and the standard deviations calculated for the bins are presented as open circles.

By using single-pass sampling methods and extending the length of our sampling sites, we were able to decrease the detectable changes to rainbow trout abundance and size, and species richness that we could detect after 5 years of monitoring. The largest improvement in impact detection was related to abundance monitoring. Monitoring 200-m sites as opposed to 100-m sites provided a 26% increase in our ability to detect impacts to rainbow trout abundance (Figure 6). Generally, small impacts to the size structure of rainbow trout could be detected, although, our impact detection ability for rainbow trout size was negligible with increased lineal sampling. As a result of increased sampling, we gained a 0.1% advantage in our impact detection ability for size related impacts (Figure 7). Impact detection related to species richness showed only marginal benefits from increased lineal sampling on the basin-wide scale although we increased our ability to detect impacts by 4% (Figure 8). This marginal improvement may provide substantial increases in the ability to detect impacts to the distribution of highly valued fish taxa such as bull trout, cutthroat trout and spring chinook salmon at the site scale (Figure 9).

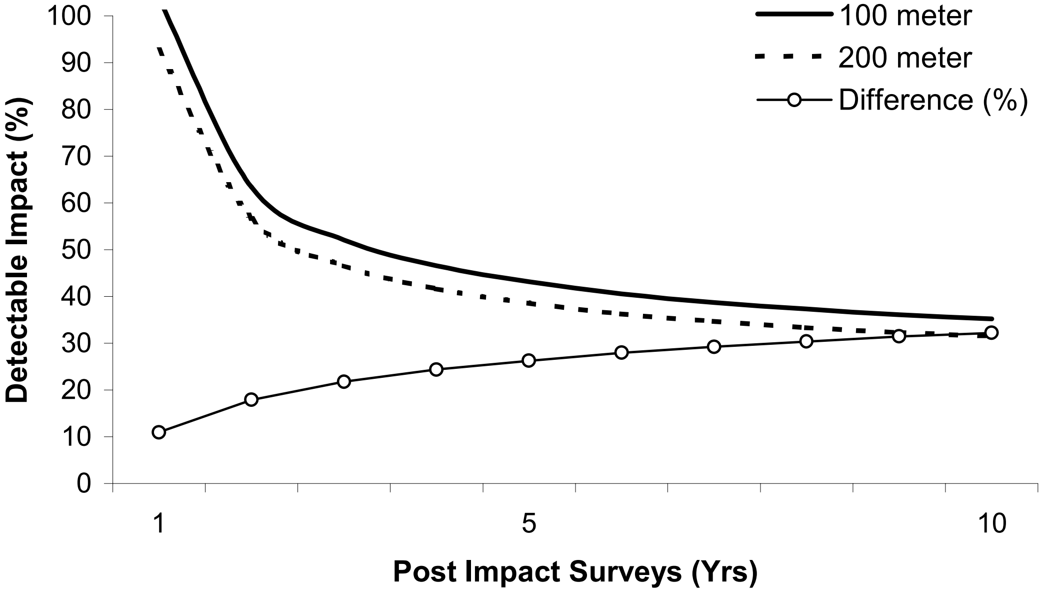

Figure 6. Minimum detectable difference in rainbow trout abundance resulting from 100-m and 200-m lineal stream sampling distances when *a* and β are set at 0.1, (power = 1-β). The difference (%) between 100-m and 200-m sampling was standardized for the percent detectable impact level and is displayed as the difference line.

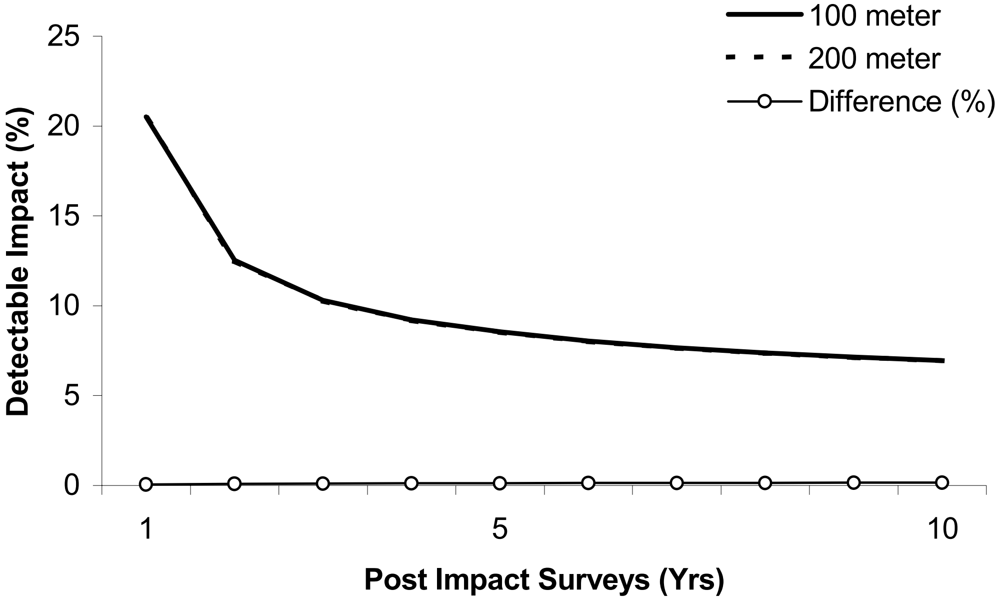

Figure 7. Minimum detectable difference in rainbow trout size resulting from 100-m and 200-m lineal sampling distances when *a* and β are set at 0.10 (Power = 1 - β). The difference (%) between 100-m and 200-m sampling was standardized for the percent detectable impact level and is displayed as the difference line.

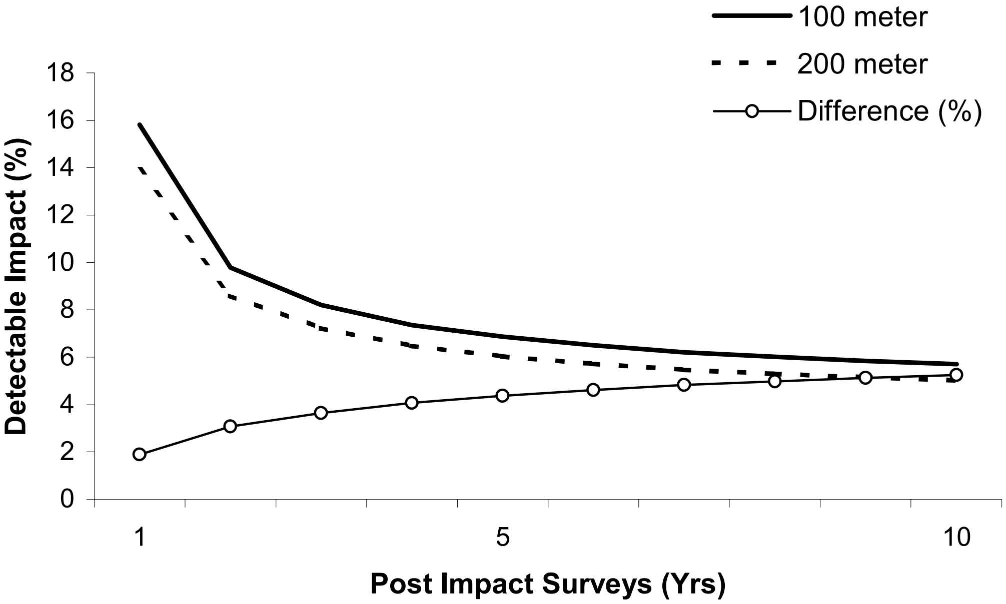

Figure 8. Minimum detectable difference in species richness resulting from 100-m and 200-m lineal sampling distances when *a* and β are set at 0.10 (Power = 1 - β). The difference (%) between 100-m and 200-m sampling was standardized for the percent detectable impact level and is displayed as the difference line.

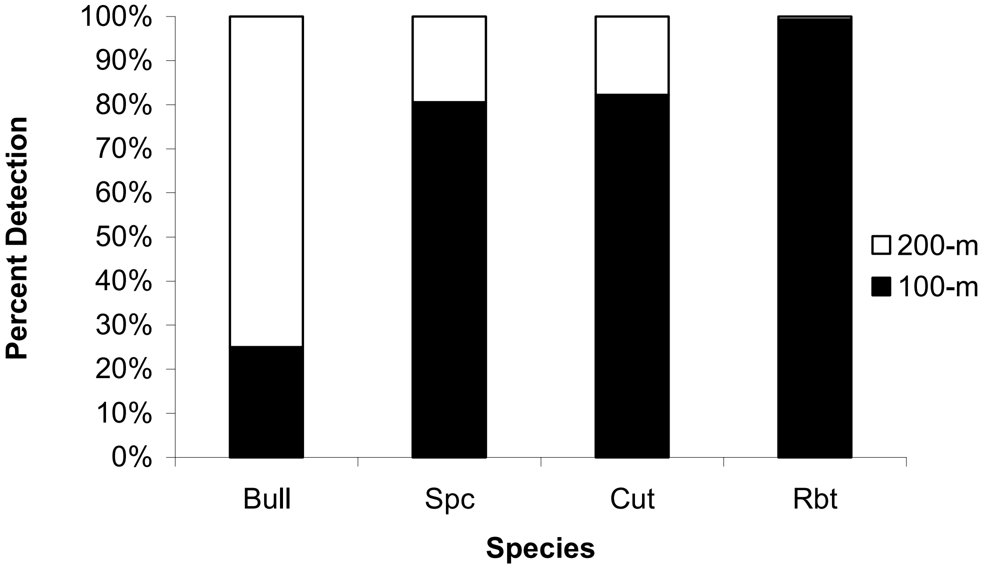

Figure 9. Percent detection of selected sensitive species for either 100-m or 200-m lineal sampling distances 1997-2003. Species codes are as follows: Bull = Bull trout, Spc = spring chinook salmon (parr), Cut = cutthroat trout, Rbt = rainbow trout.

By increasing our lineal sampling distances, we increased our detections of fish species and increased our detections of rare fish distributions. A normal approximation to the Wilcoxon matched pairs test indicated that extended sampling distances supported the detection of more species (Z=8.3, n=211, P<0.001). However, if the effort expended in the additional 100-m sites were reapportioned to sampling a second 100-m removal pass in the same 100-m reach, additional species would still be accumulated (Z=4.3, n=211, P<0.001). Two-pass electrofishing in 100-m sites detected a 3% increase in fish species richness over a single electrofishing pass in the same 100-m reach. Single-pass electrofishing in extended 200-m reaches provided an 18% increase in the number of species detected on the basin-wide scale over all years sampled. We would not have detected the presence of our most rare and threatened salmonid, the bull trout, 75% of the time if only 100-m lineal sampling distances had been repeatedly sampled in lieu of 200-m sites (Figure 9). Additionally, we observed a 19% increase in our detections of chinook salmon parr by sampling extended 200-m site lengths (Figure 9).

The variability of our rainbow trout density, size, and species richness estimates decreased with increasing lineal stream sampling distances. It also appeared that the response variables were temporally variable suggesting that accurate estimates could be generated with differing site lengths in different years. Density estimates appeared to be the most temporally variable parameter estimates when evaluated at short lineal distances. In most years, rainbow trout density estimates did not begin to stabilize when less than 175-m of stream had been sampled (Figure 10). Our estimates of rainbow trout size appeared to be the least variable of our parameter estimates, both temporally and spatially within the sites. On average, the size estimates generated were within 1% of the benchmark value estimated for the entire 200-m reach when only 50 lineal meters had been sampled (Figure 11). Species richness estimates displayed consistent reductions in temporal variation with each 25-m sampled. When annual species richness estimates generated for each 25-m increase in sampling distance were judged against the 200-m estimates, 150-m of stream had to be sampled to capture 90% of the species present in our streams (Figure 12). On average, 88% (range 83%-95%) of the species present would be encountered within100-m of sampling. Finally, there was no consistent trend in the parameter estimates within any given year such that accurate estimates could be generated for all parameters by sampling distances less than the benchmark 200-m.

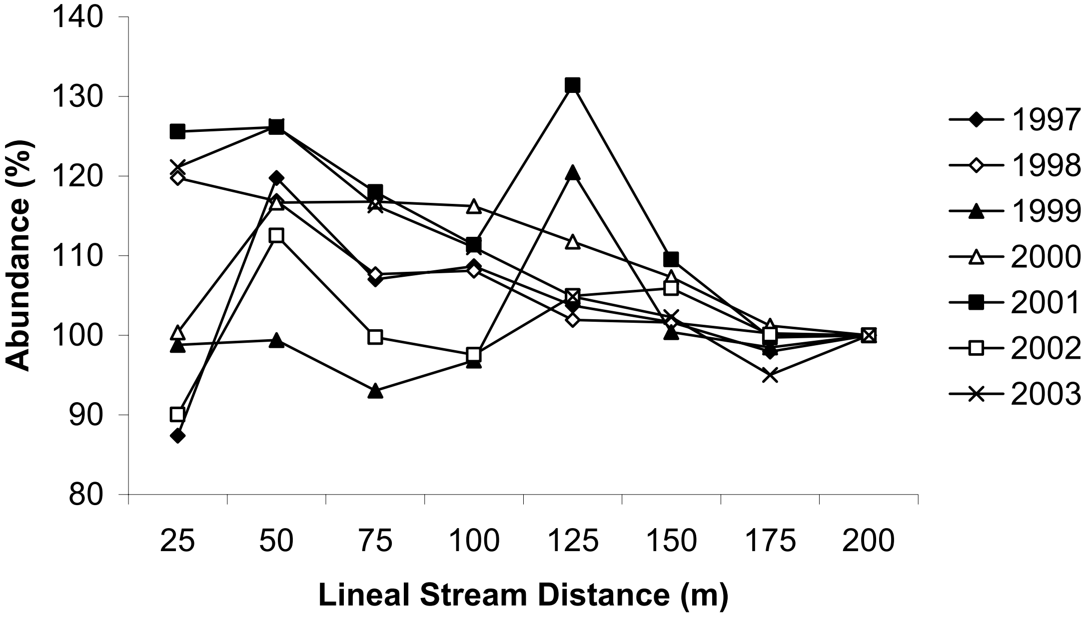

Figure 10. Annual mean density (fish/m^2) of rainbow trout with increased lineal sampling distance as a percent of the density estimate generated in 200-m.

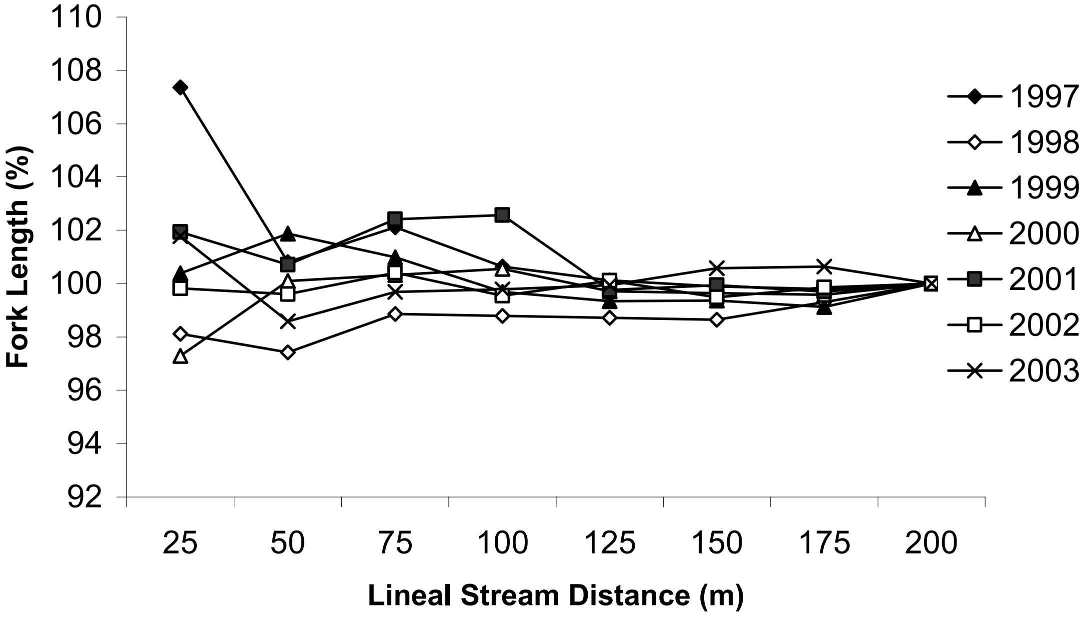

Figure 11. Annual mean length (mm, fork length) of rainbow trout with increased lineal sampling distance as a percent of the mean fish length from all fish sampled in 200-m.

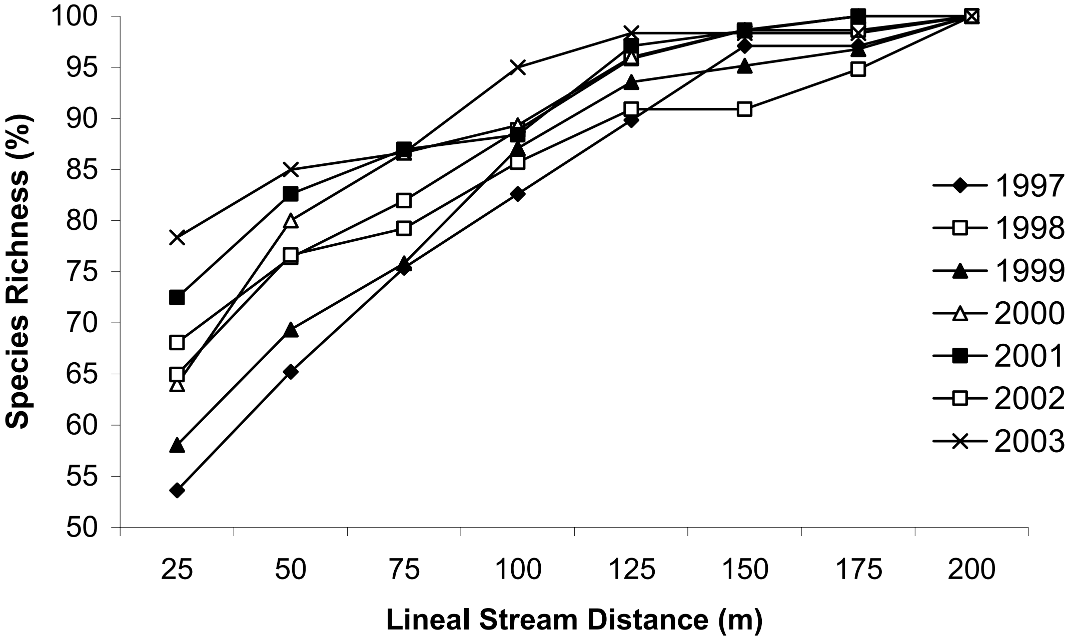

Figure 12. Annual species accumulation curves per 25 meters sampled as a percent of the 200-m total for the year.

Discussion

The application of single-pass electrofishing methods may only be advantageous to monitoring programs that are relatively long and abundance monitoring and rare species detections are important. The detectable impact curve for abundance indicated that adopting single-pass methods will benefit impact detections and the benefits will be most pronounced when monitoring programs last greater than 5 years. Single-pass sampling methods also provided species detection benefits for fish that were relatively rare and dispersed in our streams. The benefits provided to species richness and size monitoring were marginal over all time periods. Short-term monitoring programs (e.g. 1-4 years) would receive little or no benefits from single-pass methods because several years of multiple-removal sampling would be required to establish capture efficiencies for all sites monitored and to get a fair representation of their temporal variation. With this limitation, the decision to utilize single-pass expansion methods should be made after several years of multiple-pass electrofishing has been performed. Our impact detection curves show that for short-duration monitoring programs, the difference in impact detection for abundance and species richness estimates is marginal between 100-m and 200-m sampling sites. Additionally, it appears that size structure information collected in either the 100-m or 200-m site is similar. Therefore, whichever abundance or species richness sampling routine is employed, the size information collected typically will be sufficient to detect relatively small size related impacts. We recommend multiple-removal sampling for programs lasting less than 5 years and programs lasting more than 5 years should consider adopting single-pass electrofishing methods.

We found that rainbow trout density and size estimates were independent of stream width suggesting that sites based on set lineal stream distances were appropriate for monitoring these parameters. Our observations of cumulative abundance and size estimates of rainbow trout and the temporal variability associated with 25-m incremental increases in sampling distances suggested that longer sampling distances generally increased the accuracy of the estimates. However, abundance estimates were more variable than size estimates with increasing stream area. We found sampling 200-m sites provided an acceptable balance between effort requirements and estimate precision for long-term monitoring of these variables.

We found our species richness estimates were not independent from stream size. This suggests that species accumulations will reach an asymptotic level at shorter lineal stream distances in smaller streams than in large streams. Large streams will require longer sampling sections than small streams. Sampling effort requirements have been typically reported as the multiple channel widths that must be sampled in order to collect a large proportion of the species present in a given stream reach at some predetermined level of accuracy (Lyons 1992; Angermeier and Smogor 1995; Patton et al. 2000; Reynolds et al. 2003). We found 27-31 channel widths was the minimum sampling distance required to detect 90% of the species present in our streams. For generating accurate species richness estimates, Lyons (1992) recommended sampling stream lengths comprised of a minimum of 35 channel widths. Patton et al. (2000) found that stream lengths of 12-50 times the mean wetted stream width should be sampled to capture 90% of species when electrofishing small streams in the Great Plains region of the United States. In western Oregon streams, Reynolds et al. (2003) found that electrofishing lineal stream distances of 40 times the mean channel width captured 90% of the species present. Based upon this information, it may be most effective for programs to monitor species richness by establishing lineal sampling site lengths based upon multiples of the mean wetted stream width.

Since abundance and size monitoring can be effectively monitored using standard lineal distances and species richness is more effectively monitored based upon variable site lengths, monitoring programs may need to adopt different sampling effort strategies for monitoring different response variables. When trout abundance or size variables are of interest, sampling effort requirements may be based on predetermined sampling distances. When species richness is the primary variable of interest, lineal sampling distances should be based upon multiples of the mean wetted width. When all variables are of interest, a hybrid approach may be used in which set lineal distances are sampled for abundance information, and additional stream sampled for species richness estimates. The appropriate amount of effort will be a balance between the program objectives and a fixed budget. We attempted to maximize our sampling to collect abundance, size, species richness, and rare fish distribution using single-pass electrofishing methods in 200-m long index monitoring sites.

One potential liability of single-pass electrofishing to estimate fish population size may be the use of an assumed capture probability to expand the first pass catch of fish into an estimate of their abundance. Our research shows that the assumed median capture ratios have not differed significantly from the capture probabilities established from annual multiple-removal sampling. Thus, it appears that the use of a single-pass catch expanded by a median capture ratio may be a suitable substitute for removal sampling for monitoring trends in rainbow trout populations under the conditions in this study. It would be prudent, however, for other researchers in other field settings to perform routine multiple-removal sampling periodically to verify the consistency of the implied capture efficiencies. Riley and Fausch (1992) have shown that increasing electrofishing efficiencies between removal passes have the potential to severely bias population estimates. However, this bias is likely negligible when the first pass efficiency is high and a relatively large proportion of the population is removed in the first electrofishing pass (Cross and Stott 1975). Thus, multiple-removal estimates are not without potential expansion flaws either.

Single-pass electrofishing provides stewardship benefits by reducing the exposure of missed fish to multiple electroshocking episodes and reducing the time captured fish are held captive. Stream fish have been shown to have an adverse response to electroshocking (Dalby et al. 1996; Habera et al. 1996). Mesa and Schreck (1989) showed that multiple-removal electrofishing altered cutthroat trout physiology and behavior and the stress response of fish subjected to multiple-pass removal methods was more severe than a single electroshock alone. Although we potentially sample a larger proportion of the population by sampling extended site lengths, electrofishing induced injury judged on the stream scale is negligible (McMichael et al. 1998) and fatal injuries are likely even less common.

Our results indicate that single-pass expansion estimates approximate removal estimates and are beneficial for monitoring temporal trends in rainbow trout abundance. However, several studies have shown that multiple-removal based estimates are typically underestimates of true population sizes in stream fish (Riley and Fausch 1992; Rodgers et al. 1992; Peterson et al. 2004). In light of this bias, removal-based expansion estimates should, perhaps, be viewed as biased indices of abundance as opposed to estimates of true abundance (Peterson et al. 2004). However, single-pass expansions are, perhaps, no more biased than multiple-pass expansions. By evaluating population abundance trends in several small streams throughout the Yakima Basin, we believe that our estimates adequately reflect population level trends in abundance over the temporal and spatial range of conditions that we sampled. Our abundance estimates from the streams we sampled exhibited similar trends through time suggesting that our estimates were tracking the trends in the rainbow trout tributary population on the basin-wide scale.

In conclusion, our data indicate that single-pass electrofishing methods can be reliably employed for long-term monitoring of trout population trends over the range of environments encountered in our streams over the last decade. Our research is based on many years of sampling over a large spatial scale and has helped shed light on the costs and benefits associated with stream sampling effort. We believe the benefits associated with adopting this method outweigh the costs when abundance and rare fish distribution monitoring will last greater than five years. Although our data indicate that the precision of trout population point estimates can be increased with each additional removal pass in block-netted stream sections, the increase would be of little value evaluating temporal trends in salmonid abundance over the basin-wide scale. Additionally, our long-term data shows that sampling effort requirements for the response variables we evaluated will be temporally variable. In some years rainbow trout parameter estimates can be generated and fishes can be monitored effectively by sampling short stream sections. Other years, very long sections will be required. Our data shows that there is no temporal or spatial pattern to this variability and thus, it would be difficult to make real-time corrections to sampling effort. Thus, reallocating sampling effort to sample extended sites when monitoring long-term trends in rainbow trout abundance, size, species richness, and the distribution of rare fish, provides substantial monitoring benefits.

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Chapter 3

Smallmouth Bass Predation on Hatchery and Wild Salmonids in the Yakima River, Washington

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Abstract

Predation on hatchery and wild salmonids by nonnative smallmouth bass *Micropterus dolomieui* was examined in the Yakima River, Washington from 1998 to 2001. Smallmouth bass were sampled weekly in two sections of the lower Yakima River from March through June using a drift boat electrofisher. Average abundance estimates of smallmouth bass $\geq$150 mm fork length for the four years sampled ranged from 3,347 in late March to 19,438 in early June. Abundance estimates from 1998 to 2001 all showed a similar trend of increasing abundance throughout the spring. Salmonids were identified in the guts of smallmouth bass throughout the sampling period and were most prevalent during the month of May. Ocean-type Chinook salmon *Oncorhynchus tshawytscha* constituted 47% of all the fish species identified in the gut samples. Smallmouth bass were estimated to have consumed an average of 200,405 salmonids yearly from March 22 to June 16, 1998 to 2001; of these only 3,176 were yearling salmonids (primarily spring Chinook salmon). Smallmouth bass predation on all yearling salmonids never exceeded 0.6% of the annual production of hatchery and wild fish combined. We estimated that 85% of the ocean-type Chinook salmon consumed by smallmouth bass in a given year were of natural origin. Estimated smallmouth bass consumption of hatchery ocean-type Chinook salmon has only comprised up to 4% of production in a single year. Our estimates of consumption on ocean-type Chinook salmon are likely to be underestimates because we did not sample throughout the entire rearing and emigration period of these fish. Our results indicate that smallmouth bass can have negative impacts on ocean-type Chinook, particularly naturally produced Chinook, which are generally smaller and available longer than hatchery fish.

Introduction

Predation by nonnative introduced species in the Columbia River Basin has been suggested as a contributing factor for the declines of the native Pacific salmon *Oncorhynchus spp*. (Li et al. 1987; Bennett et al. 1991; Poe et al. 1991; Rieman et al. 1991; Tabor et al. 1993; Zimmerman and Parker 1995; Zimmerman 1999). In the late nineteenth century, very little was known about the effects of introduced species on the native fish faunas of the Northwest. This is evidenced by the following statement taken from Lampman (1946); the bass would "prove himself, if given the opportunity, the best friend of our salmon and trout". Even David Starr Jordan, a noted early ichthyologist, approved of the introduction of bass in Oregon believing they would confine their diets to minnows, suckers, and chubs (Lampman 1946).

By the late 1800's, the abundance of the native trout and salmon was already declining in localized areas and settlers arriving to the Pacific Northwest wanted to fish for the species they were accustomed to in the East such as bass. Smallmouth bass *Micropterus dolomieui* are a top predator native to the Eastern and Midwest United States and Southeast Canada (Wydoski and Whitney 1979). One of the earliest introductions of smallmouth bass in Washington State occurred in 1925 when 5000 juvenile fish were planted in the Yakima River by state game protector N. E. Palmer and again in 1934 (Lampman 1946). By the early 1940's, smallmouth bass were reported to be plentiful in the lower 68 km of the Yakima River, within the adjacent Columbia River, and up into the Snake River (Lampman 1946). Some researchers have theorized that the introduction of smallmouth bass to Northwest rivers has caused a shift in the trophic dynamics of riverine systems (Poe et al. 1994). Where northern pikeminnow *Ptychocheilus oregonensis* was once the keystone predator of a system, smallmouth bass may have displaced them by competition or direct predation (Fletcher 1991; Shrader and Gray 1999). Although smallmouth bass have been shown to feed heavily on other fishes (Poe et al. 1991; Zimmerman 1999), there have been mixed reports of smallmouth bass preying on salmonids in lotic environments of the Northwest. Shrader and Gray (1999) and Summers and Daily (2001) reported no predation on salmonids in the John Day River, Oregon and very low predation on salmonids in the Willamette River, Oregon respectively. The John Day River study was conducted in areas where there are no salmonids rearing and salmonids are only available during their spring outmigration when they are relatively large yearling smolts, discharge and turbidity are typically high and water temperatures are generally low. The Willamette River study was conducted in a reach where there is thought to be few salmonid spawners and salmonids are, for the most part, only available during their outmigration. Poe et al. (1991) reported that smallmouth bass diets in the John Day Reservoir of the Columbia River increased from almost no salmonids in April to 6% by weight in August. This increase over time was attributed to the increase in spatial overlap of subyearling Chinook salmon with smallmouth bass. Tabor et al. (1993) found that the diets of smallmouth bass ≥200 mm fork length (FL) consisted of 59% salmonids by weight and that salmonids were present in 65% of the samples in the Columbia River at the interface of the Hanford Reach and the McNary Pool near Richland, Washington. Tabor et al. (1993) attributed the high rates of predation to smallmouth bass consumption on subyearling Chinook salmon from the Hanford Reach population. None of the aforementioned studies conducted rigorous estimates of predator abundance in a free-flowing river environment, so estimates of salmonid consumption in these habitats could not be estimated.

The objective of our study was to investigate predation of a nonative predator, smallmouth bass, on salmonids in a riverine environment. We endeavored to discover the overall loss of salmonids due to predation by smallmouth bass and specifically what impact, or percentage of the population of salmonids, were affected. Furthermore, we sought to compare predation on yearling salmonids versus subyearling salmonids and predation of naturally produced subyearling salmonids versus hatchery produced subyearling salmonids.

Methods

Study Area

The Yakima River is a large tributary to the Columbia River in central Washington State. The Yakima River enters the Columbia River at river kilometer (rkm) 539 near Richland, Washington (Figure 1). There are two dams in the lower portion of the Yakima River that are low head diversions and have ladders to provide passage for migrating fish. These dams divert water for irrigation but do not significantly affect the amount of water in our study area until later in the summer, although they may contribute to lower flows and increased water temperatures in years with lower than average runoff; thus patterns of discharge and water temperatures in the lower Yakima River during the spring are not unlike other large tributaries to the Columbia River. During our study, the mean secchi depth was 81 cm (range, 14-153 cm). Mean water temperatures and discharges were relatively similar during all years of our study except for 2001, which was an extreme drought year (Table 1). Smallmouth bass spawn and rear in the Yakima River from the mouth to Prosser Dam (rkm 68). Introductions of nonative species have changed the fish species assemblages markedly in the lower Yakima River. Of the 30 species of fish we have identified in the lower Yakima River, 12 are not native. We found the top seven species in order of abundance during the period from March to June to be: sucker (*Catostomus spp*.), smallmouth bass, common carp (*Cyprinus carpio*), ocean-type Chinook salmon, chiselmouth (*Acrocheilus alutaceus*), mountain whitefish (*Prosopium williamsoni*), and dace (*Rhinichthys spp*.). These are based on visual estimates of relative abundance taken while electrofishing and are undoubtedly underestimates of smaller sized fishes relative to larger fishes. We use the terms ocean-type Chinook salmon for the fall Chinook salmon populations and stream-type Chinook salmon for the spring Chinook salmon populations.

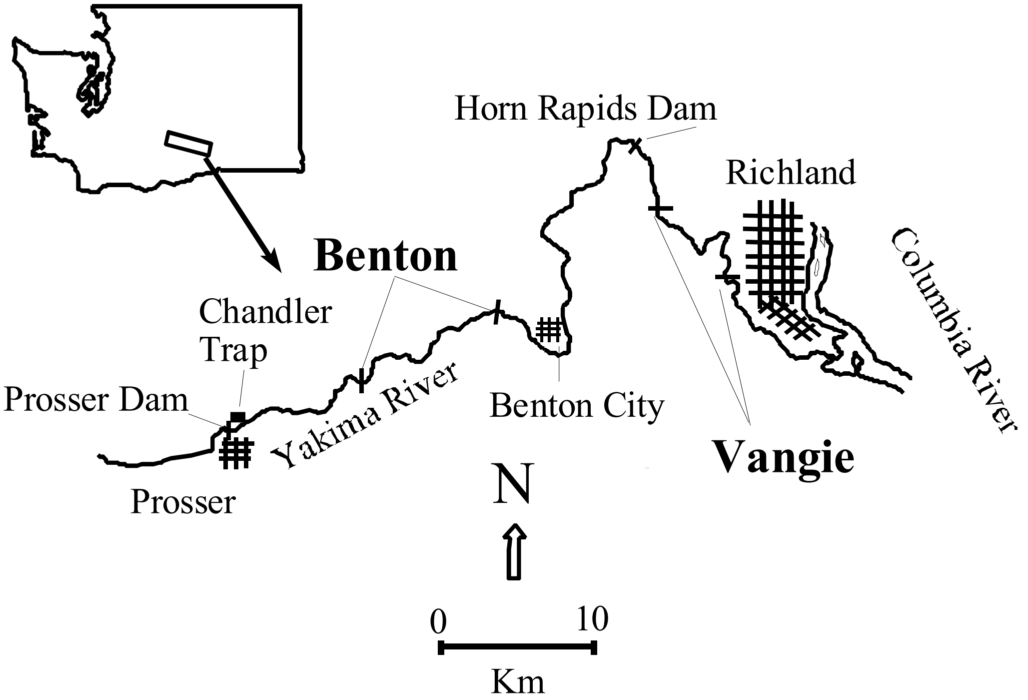

Figure 1. Map of the study area in the lower Yakima River showing index sections in bold type.

Table 1. Daily average temperature and discharge by month between 1998 and 2001 and by year during March through June with upper and lower ranges.

Dates	Temperature (ºC)	Range (ºC)	Discharge (m^3/s)	Range (m^3/s)
March, 1998-2001	8.3	4.7-12.1	136	53-257
April, 1998-2001	12.2	7.7-19.3	139	28-294
May, 1998-2001	15.8	10.8-23.8	159	28-418
June, 1998-2001	18.4	12.4-23.9	145	25-377
March-June, 1998	12.9	4.7-20.4	182	52-438
March-June, 1999	12.2	5.2-19.3	215	113-411
March-June, 2000	13.6	6.7-23.6	164	52-308
March-June, 2001	15.3	6.2-23.9	47	25-87

The lower Yakima River is utilized for spawning and rearing as well as a migration corridor for anadromous salmonids. Stream-type Chinook salmon, coho salmon *O. kisutch*, and steelhead *O. mykiss*, which spawn in the upper reaches of the system migrate through our study area during the brief spring emigration period. Ocean-type salmonids such as fall Chinook salmon spawn and rear in our study area and are available prey for smallmouth bass from the time they emerge from the gravel until the time they emigrate from the Yakima River. Based on passage estimates of naturally produced ocean-type Chinook salmon that are spawned upstream of Prosser Dam, the average peak of migration for 1998 to 2001 was June 20 (range, May 26 to July 1).

For our study area we divided the lower 68 km of the Yakima River into two reaches bisected by Wanawish (or Horn Rapids) Dam (a low head irrigation diversion) (Figure 1). The lower reach, termed the "Vangie" reach, is approximately 28.1 km long and is located between the mouth of the Yakima River and Wanawish Dam. The upper reach, termed the "Benton" reach, is approximately 39.9 km long and is located between Prosser Dam and Wanawish Dam. A section in each reach was chosen to represent the entire habitat available within that respective reach. The Vangie section is about 8 km in length, from rkm 13 to rkm 21, and was used to extrapolate to the Vangie reach. The Benton section is about 7.8 km in length, from about rkm 49 to about rkm 57, and was used to extrapolate to the Benton reach. We will refer to the sampled area as the "section" and the area it represents as the "reach".

Abundance Estimates

Mark-recapture population estimates were performed twice annually for smallmouth bass ≥150 mm FL in each section in 1998 and 2000 and once in each section in 1999 using a drift boat electrofisher (McMichael et al. 1998) (Table 2). Two boats were used in order to sample both banks simultaneously during the population estimates. Electrofisher settings were generally 400 V pulsed DC (PDC; Coffelt's CPS setting) at between 2 and 5 Amps, depending on water conductivity. Smallmouth bass and juvenile salmonids were netted by a person standing at the front of the boat and subsequently placed into a holding vessel. The electrofishing unit was generally positioned along the banks, especially during high flows; this was where we were able to get our highest electrofishing efficiencies. Smallmouth bass tended to be associated with the bank and along the high/low velocity interfaces near the bank, especially during the spring when water discharge was relatively high and water temperatures were relatively cool. The numbers of non-target species of fishes that were electrofished, but not netted, were visually assessed and recorded by the person netting. Fish were processed every kilometer during all electrofishing runs. Processing consisted of recording length (mm, FL) and weight (g) for all fish netted. During the mark runs all smallmouth bass ≥150 mm FL were given a fin clip for identification in the recapture run. In addition, all smallmouth bass ≥200 mm FL were given a serially numbered anchor tag. The recapture runs followed one day after the mark runs and all smallmouth bass ≥150 mm FL were interrogated for marks.

Estimates were calculated using the modified Peterson's method (Vincent 1971) (Table 3). We pooled all sizes of fish (150 mm FL and larger) because in some cases there were not enough recaptures to get valid estimates in multiple size classes (at least 3 recaptures for each size class). We believe this is a valid approach even though smallmouth bass <200 mm FL were not marked with an anchor tag because the sum of the estimates of smallmouth bass <200 mm FL and >200 mm FL when we had valid estimates for both size classes did not significantly differ from the pooled estimate (paired t-test, P=0.18, df=4). If there were a significant difference in mortality or probability of recapture between the two size classes due to marking differences, then we would expect to have seen a significant directional difference between the two sizes. We chose to perform our mark and recapture runs only one day apart to decrease the risk of violating the assumption of no movement in or out of the study area.

During weeks that we did not perform a mark recapture estimate, we electrofished the left bank of each section to gather diet data and generate a catch per unit of effort (CPUE) for smallmouth bass ≥150 mm FL. A significant relationship between catch per unit of effort (CPUE) of the left bank sample and mark-recapture estimates that were generated using data from 1998 to 2000 was observed (P<0.001) (Figure 2). This relationship was used to estimate weekly abundance of smallmouth bass ≥150 mm FL for the years 1998 to 2001. We had to rely solely on the regression for weekly abundance estimates in 2001 because we were unable to generate valid abundance estimates due to a low number of recaptured fish.

Table 2. Weeks sampled for diet data (D), single bank catch per unit effort (C), and mark-recapture abundance estimates (A) for the years 1998 to 2001.

Week	1998	1999	2000	2001
Mar, 1-7			D	
Mar, 8-14				
Mar, 15-21			D	D
Mar, 22-28		D		
Mar, 29-Apr, 4		D, C	D, C	D, C
Apr, 5-11		D, C	D, C	
Apr, 12-18		D, C	D, C	D, C
Apr, 19-25	D, A	D, C	D, C	D, C
Apr, 26-May, 2		D, A	D, A	D, C
May, 3-9		D, C	D, C	D, C
May, 10-16	D, A	D, C	D, C	D, A
May, 17-23		D, C	D, A	D, C
May, 24-30		D, C	D, C	D, C
May, 31-Jun, 6	D, C	D, C	D, C	D, C
Jun, 7-13		D, C	D, C	D, C
Jun, 14-20			D, C	D, C

Table 3. Results of mark recapture population estimates for smallmouth bass performed in the Benton and Vangie sections during 1998, 1999, and 2000. Included is the catch per unit effort (CPUE) used to develop the relationship with abundance.

Date	Section	Estimate	Confidence interval	Efficiency	CPUE (fish/min)
April 21, 1998	Benton	1,338	523-2,153	6.6%	0.294
April 23, 1998	Vangie	2,925	1,357-4,493	6.1%	0.498
May 12,1998	Benton	2,928	1,353-4,503	5.5%	0.658
May 14, 1998	Vangie	4,026	2,653-5,398	9.6%	0.917
April 28, 1999	Benton	1,410	451-2,370	6.4%	0.327
April 30, 1999	Vangie	2,030	1,036-3,023	9.5%	0.404
April 26, 2000	Benton	713	374-1,051	14.6%	0.303
April 28, 2000	Vangie	2,263	945-3,581	6.1%	0.394
May 17, 2000	Benton	1,827	651-3,003	6.0%	0.306
May 19, 2000	Vangie	1,067	570-1,563	12.6%	0.276

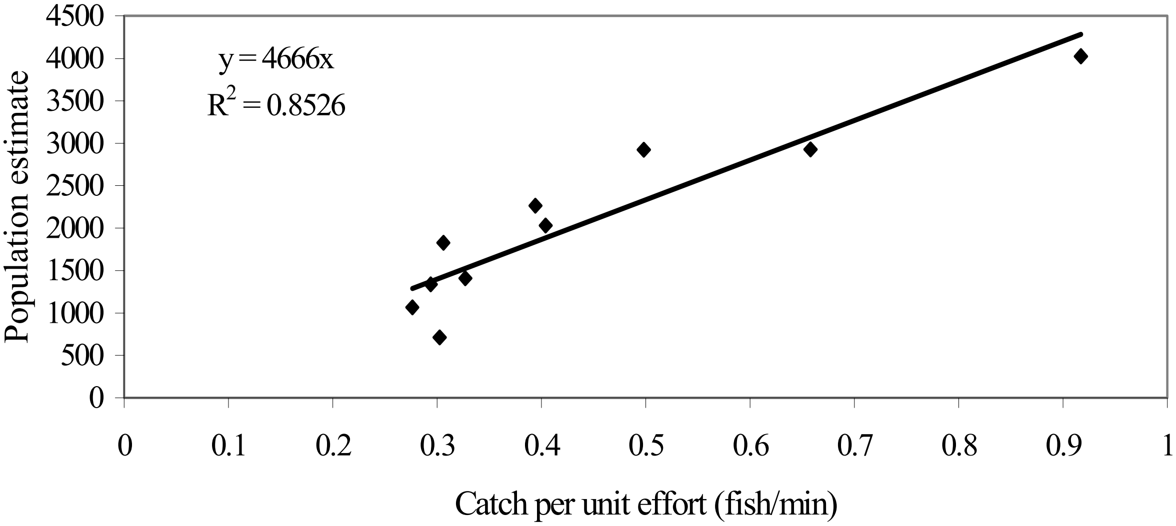

Figure 2. Relationship between catch per unit effort and mark-recapture population estimates in the Benton and Vangie sections during 1998, 1999, and 2000.

Diet Samples

Diet samples from smallmouth bass captured during weekly electrofishing of the left bank were obtained by pulsed gastric lavage (Light et al. 1983). Diet samples were generally collected once weekly in each section with a few exceptions (Table 2). A systematic subsample (i.e. every other or every third fish, depending on CPUE, in order to examine a minimum of twenty samples throughout the section) of all smallmouth bass $\geq$150 mm FL was examined for stomach contents except when CPUE of fish was low, then all predatory-sized fish were examined. This is a departure from Tabor et al. (1993) who only sampled smallmouth bass $\geq$200 mm FL. We found in our study area that an average of 12% of the smallmouth bass 150 to 199 mm FL that were sampled had consumed salmonids. All diet samples were placed in whirl-paks with 10 ml of saturated sodium bicarbonate solution and marked with date of collection, stomach number, length, weight, and the section where the fish was captured and then placed on dry ice. Samples were kept frozen for later examination in the laboratory (1 to 3 months).

In the lab, the diet samples were weighed to the nearest 0.1 g, then immersed in a porcine pancreatin solution consisting of 1 g pancreatin powder, 65 ml lukewarm tap water, and 35 ml saturated borate solution as a buffer. These samples were placed in a drying oven and allowed to sit for 2 to 24 hours at 40° C depending on the size of the fish. Diagnostic bones were then picked out of the digested tissue and analyzed. The analysis of the contents consisted of placing a single stomach sample into a petri dish and counting and identifying fish to the lowest possible taxonomic classification based on diagnostic bones. For bone identification, a series of keys and sketches produced and provided by the United States Geological Survey Columbia River Research Laboratory located in Cook, Washington, were used. Standard equations presented by Hansel et al. (1988), as well as some equations that we developed, were used to estimate the original lengths of fishes in the stomach samples based on dimensions of bones measured to the nearest 0.05 mm with an ocular micrometer. Length-weight regressions based on live fish we collected concurrently with the predatory fishes, as well as equations presented by Vigg et al. (1991), were then used to calculate an estimated weight of each prey fish at the time of ingestion.

Digestion time (DT) was used to reveal the time(s) of day that predators were eating salmonid prey items and the length of time that they were present in the gut. This was then used to reveal the temperature regime to use for calculating the evacuation rate. We used the average temperature of that time span for our calculations in order to more closely mimic the actual temperatures encountered by the predators. Temperature (T) was obtained from thermographs placed in each section and set to record the water temperature each hour. Using an equation derived from Rogers and Burley (1991) we back-calculated the average time since ingestion of a single salmonid prey by smallmouth bass (*DT*).

$$DT = -200\ln(-E^{0.513}S^{-0.513}+1)S^{0.29}e^{-0.15T}W^{-0.23} \quad [1]$$

E = amount of prey evacuated or digested by predator (g)[back-calculated weight at time of ingestion – weight of stomach contents sampled],
S = prey meal weight [back-calculated weight at time of ingestion](g),
T = water temperature (C)[24 hour mean from midnight to midnight for sampling day], and
W = predator weight (g)

Based on these results (Fritts et al. 2001) we then elected to use the average temperature for the 24-hour period starting from the mean time that samples containing single salmonid prey were eaten (1100 hours). This new temperature variable will be called *T2* and is used in our consumption equations.

Consumption

To calculate consumption we followed four basic steps:

1. Calculate evacuation time for each smallmouth bass containing at least one salmonid;
2. Calculate the average daily consumption for smallmouth bass containing salmonids;
3. Extrapolate the average daily consumption for each day for each reach;
4. Sum the daily consumptions of each reach to obtain consumption for the time period being investigated.

These steps are described in further detail below.

We used the equation presented by Rogers and Burley (1991) to calculate evacuation time (*ET90*; days) for smallmouth bass and modified it to solve for *ET90* in hours. This is the number of hours for a given meal to be 90% evacuated at a given temperature and predator weight. We calculated 90% evacuation as opposed to 100% evacuation because indigestible parts such as bones remain in the gut for extended periods (Rogers and Burley 1991):

$$ET90 = (24.542S^{0.29}e^{-0.15T2}W^{-0.23})x(24) \quad [2]$$

This equation was used to obtain daily evacuation times by using daily *T2* data and the *S* and *W* values obtained from our weekly sample. For example, the *S* and *W* we get on our Friday sample is used to calculate Tuesday through Monday's daily evacuation times along with the actual *T2* for each day.

To calculate estimated consumption rate C (salmonids per predator per day) we used the equation presented by Ward et al. (1995):

$$C = n(24 / ET90) \qquad [3]$$

n = number of salmonids observed in predator gut samples per day, and
$ET90$ = daily evacuation time for a salmonid meal (includes all salmonids and nonsalmonids present in sample) in hours from equation 2.

Extrapolations

Weekly population estimates of smallmouth bass $\geq$150 mm FL were generated by the regression equation based on the relationship between mark-recapture population estimates and CPUE for the Benton and Vangie study sections. To estimate the daily number of salmonids eaten within each study section by smallmouth bass (*SE*), we used the following equation:

$$SE = PExFxC \qquad [4]$$

PE = weekly population estimate of smallmouth bass $\geq$150 mm FL within the study section,
F = fraction of smallmouth bass stomachs examined that contained at least one salmonid, and
C = estimated daily consumption rate per predator from equation 3.

To estimate the daily number of salmonids consumed by smallmouth bass in the lower 68 km of the Yakima River (smallmouth bass abundance sharply decreases upstream of Prosser Dam) (S_{tot}), we added the number of salmonids consumed in the Benton and Vangie reaches. We used the following equation to estimate consumption in each of the reaches:

$$S_{tot} = (SE / SL)xRL \qquad [5]$$

SL = length of the study section (km), and
RL = length of reach being extrapolated to (km).

Frequency of occurrence

We calculated the frequency of occurrence of prey items found in the guts of smallmouth bass in the Benton and Vangie sections by dividing the number of each prey species found in the guts of smallmouth bass each week by the number of smallmouth bass stomachs examined that week in that section.

Consumption of yearling and subyearling salmonids

We partitioned consumption of yearling and subyearling salmonids based on size. For the time period prior to the first hatchery release, we assumed that all salmonids $\geq$100 mm FL were yearling salmonids. For the time period after the first hatchery release, we assumed that all salmonids $\geq$120 mm FL were yearling salmonids unless a smaller salmonid contained a tag identifying it as a yearling.

Consumption of hatchery and naturally produced salmonids

We partitioned total consumption of ocean-type Chinook salmon into consumption of hatchery and naturally produced Chinook salmon based on their size when ingested and the timing of hatchery releases. Each year roughly two million-hatchery ocean-type Chinook salmon are released at the Chandler Juvenile Monitoring Facility below Prosser Dam. About 150,000 local origin hatchery fish are released in late April. Approximately 1,800,000 are released in late May of which approximately 150,000 are from local broodstock and the remainder are from an out of basin hatchery (Table 4). The size data for the local broodstock origin fish (Table 5) is based on lengths of roughly 2000 passive integrated transponder (PIT)-tagged fish that are measured 1 to 3 weeks prior to release each year. This is assumed to be a representative sample because far less than 1% were culled (not tagged or measured due to small size) during the tagging process (Yakama Nation Fisheries, unpublished data).

We used the PIT-tag data to represent the size distribution of all hatchery fish released in May because the mean size of the out of basin fish was similar to or greater than the mean length of the PIT-tagged fish. The size data for the out of basin fish (Table 5) is based on the number of fish per pound. This was converted to the number of fish per gram and the number of fish divided by the number of grams yielded the number of grams per fish. We then used a length weight relationship to estimate length in mm (FL) from the weight to get a mean length of these fish. We had to assume that the size distribution of the out of basin fish was similar to the local origin fish because we only had data for the mean size of the out of basin fish.

For 1999, we assumed that all ocean-type Chinook salmon <65 mm FL consumed after the April release and <55 mm FL consumed after the May release were of natural origin because only 0.2% of the PIT-tagged fish released in April were <65mm FL and 0.7% of the PIT-tagged fish released in May were <55mm FL (Figure 3). During 2000 and 2001, only 0.1% of the PIT-tagged fish released in April and May were <65mm FL so we assumed that all ocean-type Chinook salmon <65mm FL that were consumed were of natural origin. All ocean-type Chinook salmon ingested before the first release in April of each year were also categorized as being naturally produced. The size data for the naturally produced fish (Table 5) is based on length measurements taken on fish that were beach seined near Benton City and Wanawish Dam around the time of the April release and within one week prior to the May release of hatchery reared fish during 2000 and 2001 (Yakama Nation Fisheries, unpublished data). We compared the length histograms of hatchery origin and naturally produced fish during April and May to help us judge whether our size cutoffs were reasonable (Figures 4, 5). No seining data was available for 1999 so we cannot be sure of the degree of size overlap of naturally produced and hatchery fish for that year.

Table 4. Release dates for local origin (L) and out of basin (O) hatchery ocean-type Chinook salmon in the Yakima River during 1999, 2000, and 2001.

Year	Origin	April release date	Number	May release date	Number
1999	L	Apr, 25	113,000	May, 25	78,000
	O			May, 25	1,700,000
2000	L	Apr, 21	146,000	May, 26	160,750
	O			May, 25	1,700,000
2001	L	Apr, 20	145,400	May, 17	215,000
	O			May, 7[a]	700,000
				May, 16	1,000,000

[a] Unintentional early release due to pond failure.

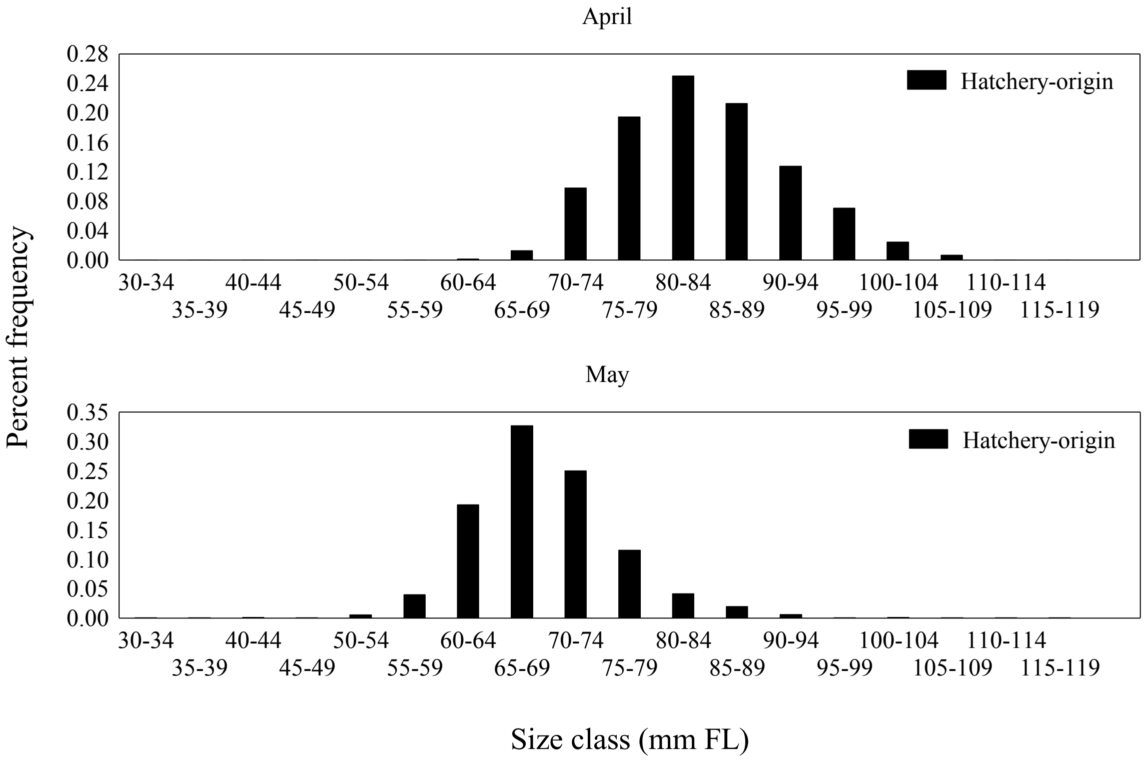

Figure 3. Percent frequency of hatchery origin ocean-type Chinook salmon measured during late April and mid May, 1999 in each size class.

Table 5. Mean lengths of local origin hatchery (L), out of basin origin hatchery (O), and naturally produced (N) ocean-type Chinook salmon released in the Yakima River from 1999 to 2001.

		April		May	
Year	Origin	Mean length (mm FL)	Range (mm FL)	Mean length (mm FL)	Range (mm FL)
1999	L	84 (N=1998)	44-110	78 (N=5,018)	36-117
	O			83[a]	
	N				
2000	L	83 (N=999)	64-106	91 (N=999)	66-117
	O			88[a]	
	N	46 (N=251)	34-91	58 (N=120)	41-89
2001	L	77 (N=2022)	64-94	84 (N=2021)	67-104
	O			75[a]	
	N	45 (N=321)	32-72	58 (N=191)	41-77

[a] Out of basin lengths are based on the number of fish per pound, therefore there is only one measurement per year with no range to present.

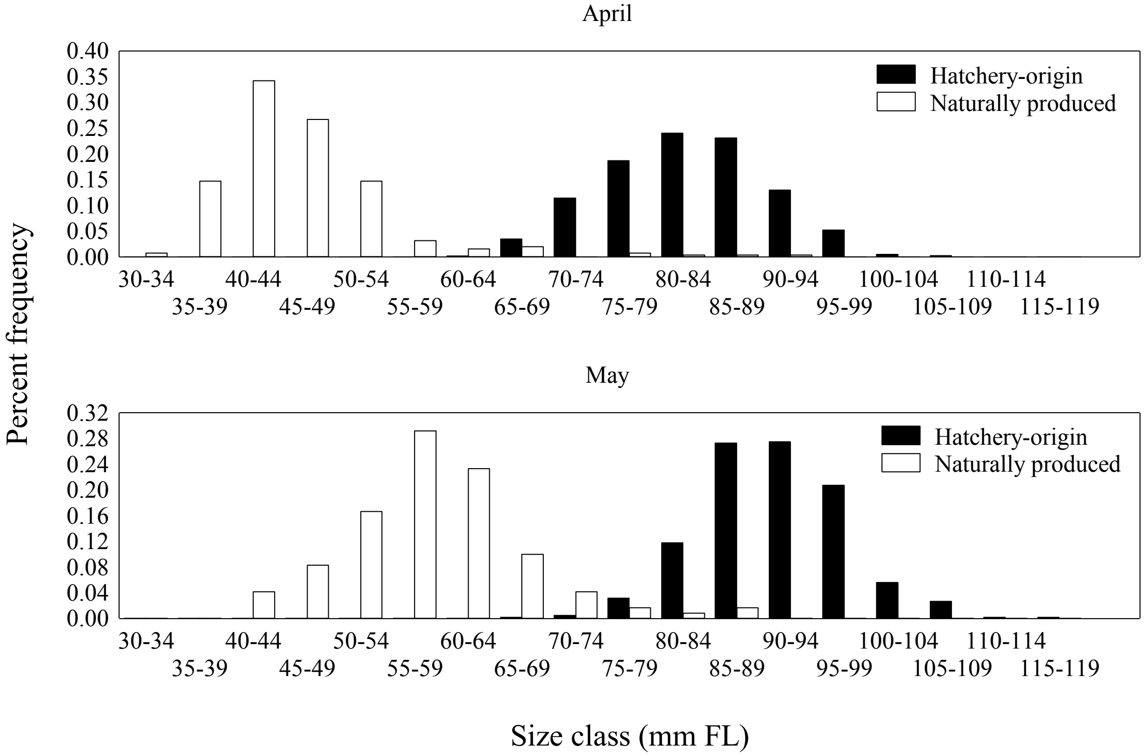

Figure 4. Percent frequency of hatchery origin and naturally produced ocean-type Chinook salmon measured during late April and mid May, 2000 in each size class.

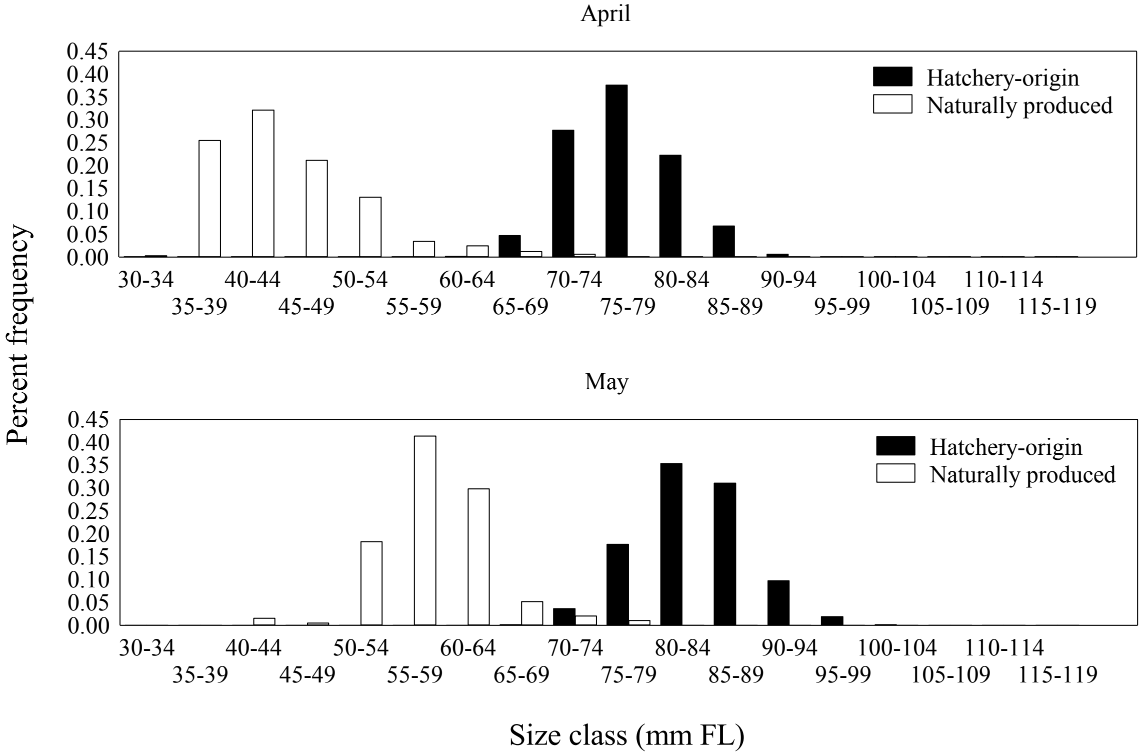

Figure 5. Percent frequency of hatchery origin and naturally produced ocean-type Chinook salmon measured during late April and mid May, 2001 in each size class.

Downstream salmonid passage numbers at Prosser Dam were estimated by Yakama Nation Fisheries staff using entrainment relationships developed at the Chandler Juvenile Monitoring Facility (Neeley 2001) (Yakama Nation Fisheries, unpublished data). Passage timing at Prosser Dam was obtained from daily estimates at the Chandler Juvenile Monitoring Facility (Yakama Nation Fisheries, unpublished data).

Results

Abundance Estimates

Abundance estimates from 1998 to 2001 all showed a similar trend of increasing abundance throughout the spring (Figure 6). Abundance of smallmouth bass ≥150 mm FL in the lower 68 km of the Yakima River increased from an annual average of 3,347 in late March to 19,438 in June (Figure 6).

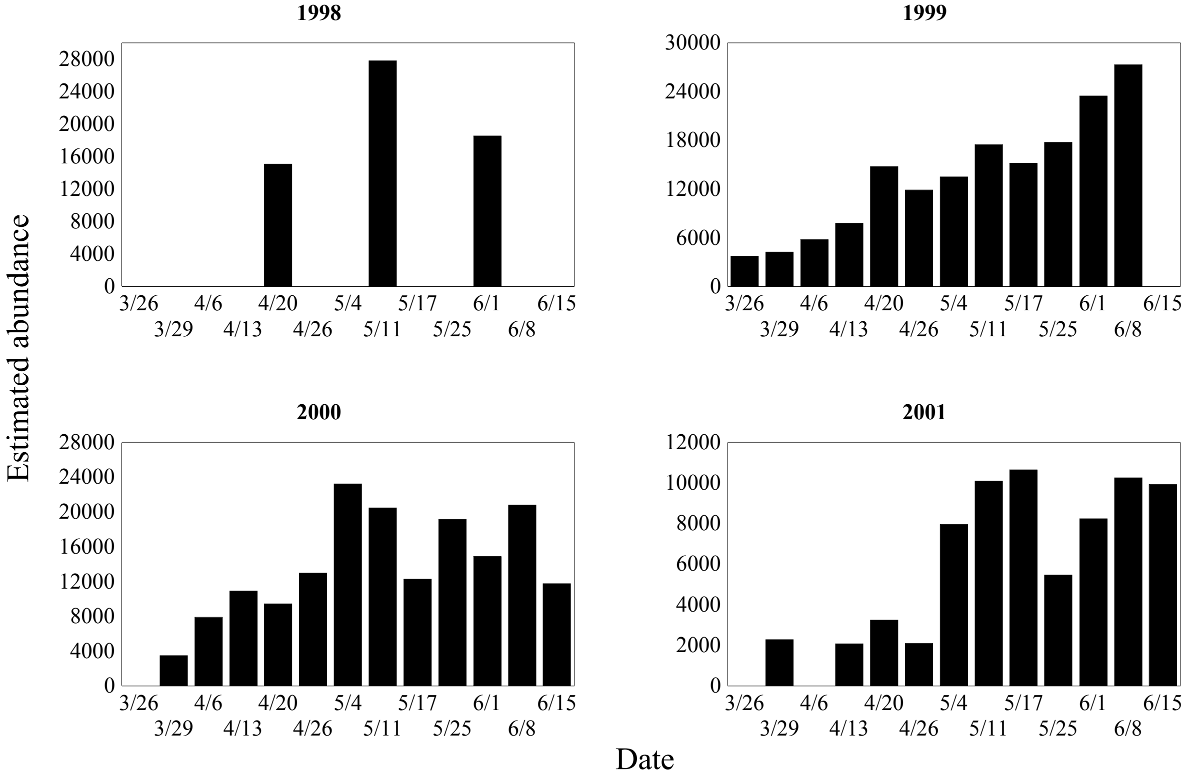

Figure 6. Estimated abundance of smallmouth bass ≥150 mm FL in the lower 68 km of the Yakima River, 1998 to 2001.

Diet

We examined the gut contents of 3,160 smallmouth bass from 1998 to 2001. These smallmouth bass had a mean fork length of 242 mm (range, 150-484 mm FL). Ocean-type Chinook salmon were found in the guts of smallmouth bass throughout the majority of the sampling period and peaked in the month of May (Tables 6,7). The percentage of stomachs that contained fish and salmonids rose steadily from late March to a peak in late May and then sharply decreased (Tables 6,7). The mean length of all ocean-type Chinook salmon found in the guts was 57 mm FL and was 127 mm FL for yearling salmonids (Table 8). There were a couple of extremely small ocean-type Chinook that were recently ingested so that we were able to measure them directly. They may have been some kind of abnormality or were flushed out of the gravel early. Only one ocean-type Chinook salmon and one yearling salmonid were identified in the samples from March 22 to March 31. Sixteen fish taxa were identified in the guts of smallmouth bass. Ocean-type Chinook salmon, mountain whitefish, and dace spp. were the dominant fish species consumed constituting 77% of the fish found in the guts with ocean-type Chinook salmon making up 47% of the fish found in the guts alone.

Table 6. Summary results of diet analyses by week for smallmouth bass (≥150 mm FL) sampled in the Benton section from March 1 to June 20, 1998 to 2001. The number of stomachs examined (N) and the frequency of occurrence (%) of samples that were empty, or contained at least one of each prey item is shown. The invertebrate category does not include crayfish.

Week	N	Empty	Invertebrates	Crayfish	Fish	OT[a]	Mountain whitefish
Mar, 1-7	5	40.0	0.0	20.0	40.0	20.0	0.0
Mar, 8-14	0	0.0	0.0	0.0	0.0	0.0	0.0
Mar, 15-21	12	27.1	41.4	7.1	17.1	0.0	0.0
Mar, 22-28	15	15.6	4.4	11.1	6.7	0.0	2.2
Mar, 29-Apr, 4	51	39.1	22.1	27.7	23.2	3.3	5.2
Apr, 5-11	78	32.6	12.9	8.2	9.0	1.8	0.9
Apr, 12-18	86	33.0	36.0	29.8	15.6	8.6	1.7
Apr, 19-25	160	29.5	30.5	22.4	32.1	16.6	5.3
Apr, 26-May, 2	161	31.1	17.7	5.4	29.2	15.7	11.5
May, 3-9	144	13.0	44.5	9.9	21.1	11.1	1.0
May, 10-16	210	16.0	60.4	19.1	39.8	28.1	1.6
May, 17-23	182	8.8	43.8	13.1	25.3	17.3	2.8
May, 24-30	101	19.8	29.2	6.4	31.1	16.2	8.6
May, 31-Jun, 6	208	25.1	47.8	13.5	22.7	7.3	1.2
Jun, 7-13	179	22.3	46.6	30.6	15.1	2.2	0.6
Jun, 14-20	47	9.7	51.5	32.6	16.8	3.6	0.0

[a] Ocean-type Chinook salmon.

Other species found in stomach contents listed in order of frequency of occurrence: dace spp., smallmouth bass, sucker spp., unidentified nonsalmonid, channel catfish (*Ictalurus punctatus*), tadpole, chiselmouth, yearling salmonid, lamprey, frog, unidentified salmonid, northern pikeminnow, *Lepomis* spp., sculpin spp., mammal, mosquitofish (*Gambusia affinis*), redside shiner (*Richardsonius balteatus*), common carp, and peamouth (*Mylocheilus caurinus*).

Table 7. Summary results of diet analyses by week for smallmouth bass (≥150 mm FL) sampled in the Vangie section from March 1 to June 20, 1998 to 2001. The number of stomachs examined (N) and the frequency of occurrence (%) of samples that were empty, or contained at least one of each prey item is shown. The invertebrate category does not include crayfish.

Week	N	Invertebrates	Empty	Fish	Crayfish	OT[a]	Mountain whitefish
Mar, 1-7	1	100.0	0.0	0.0	0.0	0.0	0.0
Mar, 8-14	0	0.0	0.0	0.0	0.0	0.0	0.0
Mar, 15-21	7	33.3	50.0	0.0	16.7	16.7	0.0
Mar, 22-28	14	9.5	11.9	0.0	4.8	0.0	0.0
Mar, 29-Apr, 4	43	65.7	19.9	0.0	10.8	0.0	0.0
Apr, 5-11	39	34.7	16.4	0.0	9.5	3.2	0.0
Apr, 12-18	90	42.5	41.7	1.68	26.4	13.9	10.0
Apr, 19-25	165	50.2	26.6	5.2	25.6	9.6	8.7
Apr, 26-May, 2	237	29.8	18.4	0.2	30.2	12.3	13.3
May, 3-9	86	21.8	31.1	3.6	23.6	12.2	8.5
May, 10-16	231	26.9	37.9	3.3	40.7	20.0	16.4
May, 17-23	137	24.6	28.7	1.92	26.8	16.4	2.2
May, 24-30	96	16.3	28.9	1.7	35.3	20.6	13.0
May, 31-Jun, 6	179	27.7	46.8	3.8	23.5	14.2	7.2
Jun, 7-13	143	34.0	49.3	1.6	21.8	6.3	3.2
Jun, 14-20	53	22.5	60.1	5.8	11.7	4.5	0.0

[a] Ocean-type Chinook salmon.

Other species found in stomach contents listed in order of frequency of occurrence: dace spp., smallmouth bass, sucker spp., unidentified nonsalmonid, channel catfish (*Ictalurus punctatus*), tadpole, chiselmouth, yearling salmonid, lamprey, frog, unidentified salmonid, northern pikeminnow, *Lepomis* spp., sculpin spp., mammal, mosquitofish (*Gambusia affinis*), redside shiner (*Richardsonius balteatus*), common carp, and peamouth (*Mylocheilus caurinus*).

Table 8. Estimated mean lengths of ocean-type Chinook salmon (OT) and yearling salmonids (Y) (Chinook and coho) consumed by smallmouth bass by month from 1998 to 2001.

	OT		Y	
Month	Mean length (mm FL)	Range (mm FL)	Mean length (mm FL)	Range (mm FL)
April	46 (N=143)	21-97	130 (N=7)	120-138
May	59 (N=289)	29-101	111 (N=3)	101-124
June	65 (N=54)	48-92	139 (N=5)	127-153

Consumption

Consumption of salmonids by smallmouth bass followed the same general trend in all of the years we have sampled (Figure 7). Between March and early May, consumption was relatively low and gradually increased as smallmouth bass abundance, available prey, and temperatures increased. In early May, consumption quickly rose to a peak in late May and then began to decline through mid June. Between March 22 and June 16, 1998 to 2001, we estimated that smallmouth bass consumed an average of 200,405 salmonids per year of which 3,176 were yearling salmonids. The highest estimated consumption of ocean-type Chinook salmon and lowest estimate of yearling salmonids was in 1998 (Table 9).

The majority of ocean-type Chinook salmon were estimated to have been consumed in the month of May during all years of sampling (Table 10). Consumption of yearling salmonids was highest in April during 1998 and 1999 and June during 2000 and 2001 (Table 10).

We estimated that smallmouth bass consumed more naturally produced ocean-type Chinook salmon than hatchery origin fish in all three of the years that we were able to estimate them separately. Of the ocean-type Chinook salmon consumed by smallmouth bass, naturally produced fish were estimated to be most commonly consumed during 1999 and 2000 consisting of 66% (78,671) and 85% (136,700) of the total consumption of ocean-type Chinook salmon respectively (Table 11). In 2001, naturally produced ocean-type Chinook salmon accounted for 55% (94,559) of the total. The year with the highest percentage consumption of hatchery ocean-type Chinook salmon production was 2001 where we estimate 78,707 were consumed which amounts to 4% of the hatchery fish released (Table 11). We were unable to estimate the percentage of hatchery versus naturally produced fish for 1998 due to a lack of hatchery fish measurements.

Ocean-type Chinook salmon passage at Prosser Dam was relatively low until the last week of May for the years 1999 to 2001 (Figure 8).

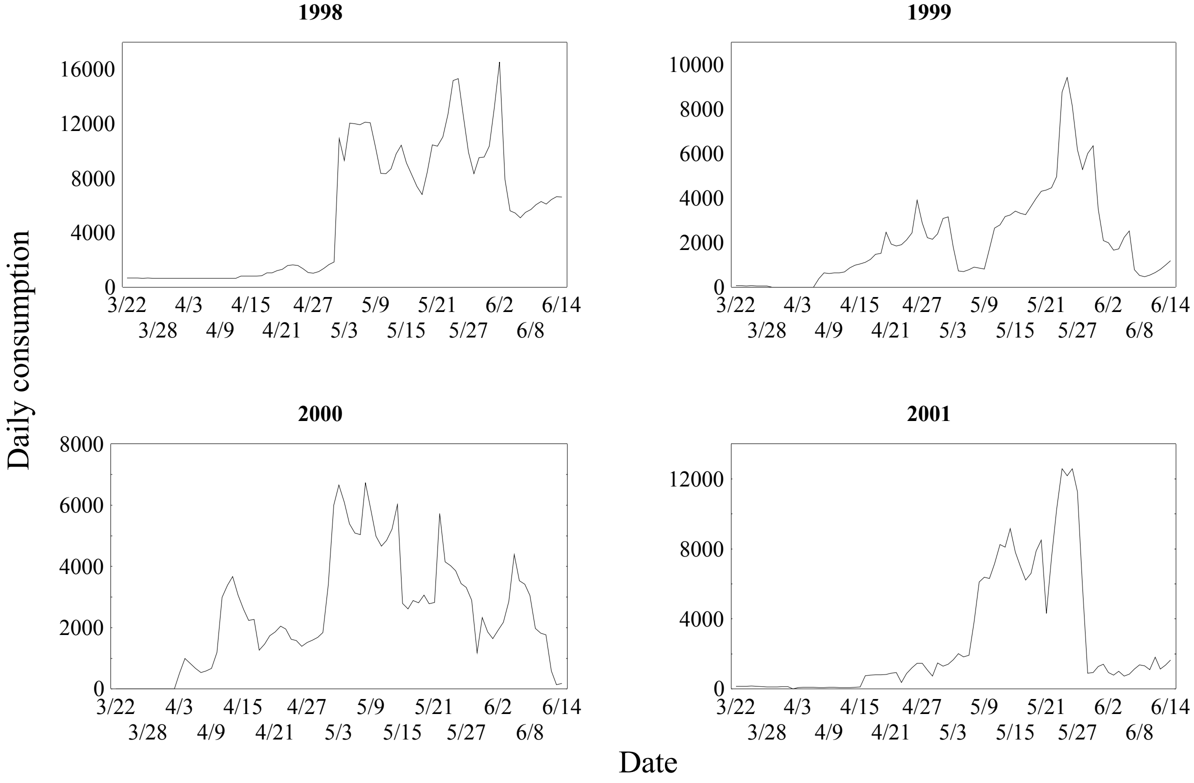

Figure 7. Estimated daily salmonid consumption by smallmouth bass between March 22 and June 16, 1998 to 2001 in the Yakima River between Prosser Dam and the confluence of the Columbia River.

Table 9. Estimated smallmouth bass consumption of all salmonids between March 22 and June 16, 1998 to 2001 and the percent of consumed salmonids that were yearlings. The salmonid category includes yearling (Chinook and coho) and subyearling salmonids combined.

Year	Salmonids	Yearling salmonids	Percent of salmonids consumed
1998	335,626	2,236	0.7
1999	120,922	2,744	2.3
2000	166,544	2,315	1.4
2001	178,526	5,407	3.0

Table 10. Estimated smallmouth bass consumption of ocean-type Chinook salmon (OT) and yearling salmonids (Y) by month between March 22 and June 16, 1998 to 2001.

	March		April		May		June	
	OT	Y	OT	Y	OT	Y	OT	Y
1998	4,966	351	21,806	1,581	244,459	304	62,157	0
1999	0	195	26,122	663	78,050	1,072	14,007	813
2000	0	0	36,355	572	105,063	717	22,811	1,026
2001	1,011	0	12,142	52	147,490	357	12,477	4,999

Table 11. Population size, estimated number consumed, and percent of population consumed by smallmouth bass for salmonid species between March 22 and June 16, 1999, 2000, and 2001. Population sizes are estimated passage at the Chandler Juvenile Monitoring Facility from March, 1 to July, 31 (Yakama Nation Fisheries, unpublished data) and hatchery ocean-type Chinook salmon released downstream of Prosser Dam.

	Species[a]				
	WOT	HOT	WY	HY	WSH
		1999			
Population size	39,453	1,891,000	245,019	253,381[b]	38,266
Number consumed	78,671	39,876	2,744	0	0
Percent consumed	NA[d]	2	1	0	0
		2000			
Population size	198,002	2,012,135	94,352	390,064	42,696
Number consumed	136,700	25,005	2,315	0	0
Percent consumed	NA[d]	1	2	0	0
		2001			
Population size	1,677,543[c]	2,076,000	137,300	894,000	28,428
Number consumed	94,559	78,707	3,785	1,622	0
Percent consumed	NA[d]	4	3	0.2	0

[a] WOT-wild ocean-type Chinook salmon, HOT-hatchery ocean-type Chinook salmon, WY-wild spring Chinook salmon and wild coho salmon, HY-hatchery spring Chinook salmon and hatchery coho salmon, WSH-wild steelhead.

[b] All coho passing Chandler in 1999 assumed to be hatchery origin.

[c] Estimates of passage at Chandler may be inflated due to higher than average entrainment rates caused by extremely low discharges.

[d] Estimates of WOT below Prosser Dam are not included, therefore percent consumption is not realistic.

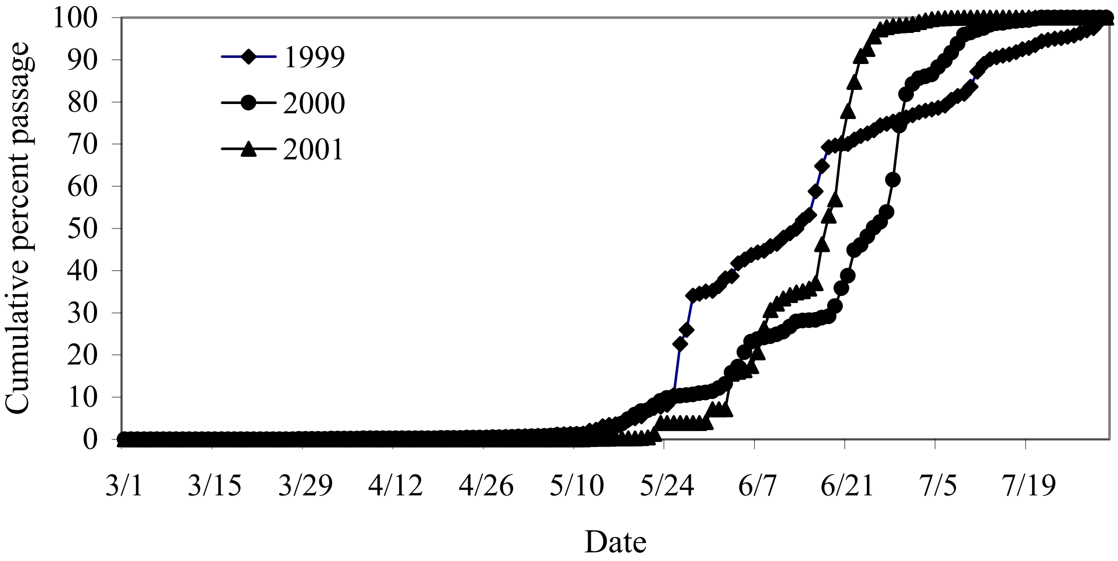

Figure 8. Cumulative percent passage of naturally produced ocean-type Chinook salmon estimated at Prosser Dam between March 1 and July 31, 1999 to 2001.

Discussion

Predation by smallmouth bass has undoubtedly contributed to lowered survival of naturally produced ocean-type Chinook salmon in the lower Yakima River, but is less likely to have contributed substantially to declines in survival of offspring of wild and hatchery stream-type Chinook salmon, hatchery coho salmon, or wild steelhead (Table 11). Smallmouth bass primarily ate the smallest salmon available, and the smallest salmon were offspring of naturally spawning ocean-type Chinook salmon. We estimated that up to 85% of the ocean-type Chinook salmon consumed were of natural origin, whereas the highest estimate of ocean-type hatchery fish consumed was 45% of the total (Table 11). In contrast, smallmouth bass consumed relatively few yearling salmonids. Predation on all yearling salmonids by smallmouth bass never exceeded 3% of the annual production (Table 11). It should be noted that our estimates of consumption of yearling salmonids may be inflated because we used the abundance of all smallmouth bass $\geq$150 mm FL to extrapolate consumption whereas the smallest smallmouth bass we found to contain a yearling sized salmonid was 206 mm FL. Others have also observed that smallmouth bass rarely ate yearling salmonids but readily consumed subyearlings (Poe et al. 1991; Tabor et al. 1993; Zimmerman 1999). The 1998 estimates were based on the extrapolation of only three weeks of sampling (Table 2) so they may not be as accurate as are the 1999 to 2001 estimates.

We believe that our estimates of total consumption and consumption of naturally produced ocean-type Chinook salmon are likely to be underestimates. For instance, we do not sample through the end of June even though ocean-type Chinook salmon are still passing Prosser Dam in relatively high numbers into July (Figure 8). In addition, measurements of hatchery fish were taken weeks before they were released, during times of high growth. Thus we are likely underestimating the size of hatchery fish. Finally, in order to be sure we were not overestimating the number of naturally produced ocean-type Chinook salmon consumed from late May to June 16, we used the lower end of the hatchery origin ocean-type Chinook salmon size range to classify hatchery and naturally produced fish. There was an overlap in the length frequencies of hatchery ocean-type Chinook salmon released in May and naturally produced ocean-type Chinook salmon collected by seining a week prior to the May release (Figures 4,5). This overlap makes it more likely that we are classifying large naturally produced Chinook salmon as being of hatchery origin during the time that consumption is the highest rather than classifying small hatchery fish as natural origin. Despite all of these factors that contribute to underestimates, our estimates of consumption of naturally produced ocean-type Chinook salmon were still high.

Our consumption estimates are valuable because we have weekly estimates of smallmouth bass abundance that allows us to extrapolate total consumption of salmonids in a large tributary to the Columbia River. Although some of our abundance estimates are extrapolated outside of the lower limits of the regression, the impact to the majority of the consumption estimates is minimal. Most of these low estimates occurred before mid April when water temperatures were low, causing the smallmouth bass to be less active and the rate of predation on salmonids to be low. If we were to exclude all predation before April 20, the average total consumption for 1998 to 2001 would still be over 187,000 salmonids. All but three of the abundance estimates during 2001, a drought year, were below our regression. Turbidity and water velocity were much lower and water temperatures were higher that year than in other years. Even with the low abundance estimates, consumption estimates were still very high which we believe is mainly due to the higher water temperatures and increased availability of salmonids from a large return of spawning adults in 2000.

We found some evidence that naturally produced ocean-type Chinook salmon were more susceptible to predation than hatchery produced ocean-type Chinook salmon. Hatchery fish are typically thought to be more susceptible to predators because of maladaptive behavior and inappropriate coloration (Maynard et al. 1995 and references therein; White et al. 1995 and references therein). Fish size and availability appeared to be more influential than behavior or coloration in determining susceptibility of Chinook salmon in the lower Yakima River to predation. We found that smallmouth bass consumed more smaller sized ocean-type Chinook salmonids and very few of the larger yearling salmonids. Hillman and Mullan (1989) also found that smaller sized wild salmon were more susceptible to rainbow trout predators than larger hatchery fish. Wild ocean-type Chinook salmon were available to smallmouth bass throughout March, April and May when there were no or relatively few hatchery fish available. Hatchery fall Chinook salmon are also less likely than naturally produced fall Chinook salmon to inhabit nearshore areas (Dauble et al. 1989) where smallmouth bass are most commonly found.

It is likely that the warmer water temperatures and low water conditions in 2001 increased the growth rate of naturally produced ocean-type Chinook salmon. If their growth was increased enough, then it would help to explain the higher consumption of hatchery versus naturally produced ocean-type Chinook salmon that we estimated that year. More naturally produced fish could have been $\geq$55 mm FL that year than in 1999 and 2000 causing us to classify them as hatchery produced fish thus inflating the estimated number of hatchery fish consumed. We believe it is also possible that an early release of hatchery ocean-type Chinook salmon on May 7, 2001 (Table 4) that would normally have been released later in May could explain some of the difference in the increased consumption of hatchery compared to naturally produced ocean-type Chinook salmon in 2001 because this would provide a longer window of availability. These higher water temperatures would also increase smallmouth bass feeding and growth by increasing their metabolism. Lower turbidity caused by lower discharges could also give smallmouth an advantage because they are a sight-feeding predator. These two factors could explain how overall consumption of salmonids by smallmouth bass in 2001 was comparable to previous years even though our abundance estimates were lower.

We attempted to get an idea of the magnitude of smallmouth bass predation on the naturally produced ocean-type Chinook salmon by estimating production below Prosser Dam. Estimates of ocean-type Chinook salmon redds below Prosser Dam were 376 in 1998, 662 in 1999, and 984 in 2000 (Watson and LaRiviere 1999; Watson and Cummins 2000; Watson et al. 2001). If we assume an average fecundity of 5,000 eggs and egg to fry survival of 10%, rough estimates of fry production below Prosser Dam would be 188,000, 331,000, and 492,000 for 1999, 2000, and 2001 respectively. If we add these numbers to the estimated number of wild ocean-type Chinook salmon passing Prosser Dam (Table 11), predation by smallmouth bass would account for 35%, 26%, and 4% of the estimated number of naturally produced ocean-type Chinook salmon produced in the Yakima River in 1999, 2000, and 2001 respectively. If we assumed 100% egg to fry survival, then estimates of percent consumption would be 4% in 1999, 4% in 2000, and 1% in 2001. When we assume an unrealistically high egg to fry survival rate of 100% the proportion of the population consumed is still higher than that of yearling salmonids for 1999 and 2000 (Table 11).

The pattern of abundance that we have seen during our weekly sampling has remained consistent for all years of sampling. We believe the increase in abundance throughout our sampling is attributable to movement and to a lesser degree, recruitment of smaller fish into the ≥150 mm FL size class. We have collected substantial evidence that smallmouth bass are moving into the Yakima River from the Columbia River in the spring and moving back to the Columbia River later in the summer based on the patterns of movement of tagged smallmouth bass recaptured during our sampling and by anglers (Fritts et al. 2001).

The pattern of consumption has been relatively similar for all years sampled. The low consumption in the early stages can be explained by a combination of low water temperatures that decreases the metabolism of smallmouth bass (Rogers and Burley 1991) and low abundance of smallmouth bass. As water temperatures, smallmouth bass abundance, and availability of ocean-type Chinook salmon from upriver spawning areas and hatchery releases increases, consumption increases to a peak in late May. From late May through mid June consumption decreases despite increasing water temperatures, high smallmouth bass abundance, and high availability of ocean-type Chinook salmon according to passage estimates at the Chandler Juvenile Monitoring Facility (Figure 8). It may be that the smaller naturally produced ocean-type Chinook salmon are becoming less available/desirable because of mortality or growth into less desirable sizes. While the fish passing through the Chandler Juvenile Monitoring Facility are relatively large smolts that are likely less susceptible to predation. Smallmouth bass also seem to be switching from a diet composed of a high percentage of fish to a diet composed of a higher percentage of invertebrates and crayfish in June (Tables 6,7). We did not have the data to compare the stomach content weights of smallmouth bass not containing fish to evaluate if they were maintaining a similar caloric intake in June as in May.

Our data indicates that smallmouth bass can adversely impact native salmonids through predation when there is spatial overlap with small sized salmonids such as in areas where salmonids spawn and rear (Tabor et al. 1993). Fisheries managers should consider this before any enhancement or stocking effort is initiated in watersheds where salmonids occur or where there is the possibility that smallmouth bass could extend their range into salmonid spawning and rearing areas. In the John Day River, Oregon, smallmouth bass were stocked upstream of a passage barrier in 1971 that had had until that time, limited the distribution of smallmouth bass to the lower river. The range of smallmouth bass has since expanded over 47 kilometers upstream of the release site (Schrader and Gray 1999). In addition, resource actions such as habitat alterations, flow manipulations, or temperature modifications, while they may be intended to enhance salmonid populations, could have indirect impacts on salmonids by enhancing the functional or numerical response of smallmouth bass predators.

Possible methods of reducing the impacts on salmonids by smallmouth bass have been considered. These include, direct removal by electrofishing or trapping, a bounty program, regulation changes, decreasing water temperatures with changes in the irrigation system and reservoir releases, disruption of spawning by manipulating flows, and swamping predators with releases of hatchery fish. Fishery managers decided that the most feasible method to implement, in terms of public acceptance and cost, was to change the angling regulations. Beginning in 2002 the regulations for smallmouth bass were changed from a five fish retention limit, only one longer than 43 cm, to no limit with a protected slot from 30.5 cm to 43 cm and only one longer than 43 cm. This regulation is intended to reduce the numbers of smallmouth bass less than 30.5 cm which have been found to be the most predaceous on salmonids in the Yakima River. Creel surveys were begun in 2002 to assess angler exploitation on smallmouth bass and will continue for two more years, at which time the regulation will be re-evaluated.

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