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YAKIMA RIVER SPECIES INTERACTIONS STUDIES
PROGRESS REPORT 1995-1997

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Yakima River Species Interactions Studies

Progress Report 1995-1997

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Executive Summary

Species interactions research 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 sixth of a series of progress reports that address species interactions research and pre-supplementation monitoring of fishes in the Yakima River basin. Data have been collected prior to supplementation to characterize the ecology and demographics of non-target taxa (NTT) and target taxon, predict the potential interactions that may occur as a result of supplementation, and develop methods to monitor interactions and supplementation success. Major topics of this report are associated with the ecology and life history of NTT, interactions experimentation, methods for sampling, risk assessment, and risk minimization. This report is organized into 14 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, 1995 and December 31, 1997 in the Yakima basin, however these data were compared to data from previous years to identify preliminary trends and patterns. Major preliminary findings from each of the chapters included in this report are described below.

o Objectives for highly valued non-target taxa (NTT) were described as acceptable impacts to their current status that could be attributed to spring chinook salmon supplementation. Impacts to a NTT's distribution, abundance, or size structure in excess of the acceptable impact would indicate a failure to achieve an objective. The following objectives for NTT are recommended: accept no impact - bull trout, westslope cutthroat trout, Pacific lamprey; accept a maximum of 5% impact - Marion Drain fall chinook salmon, upper Yakima steelhead, mountain sucker, leopard dace, sand roller; accept a maximum of 10% impact - resident rainbow trout in the mainstem Yakima River, Naches steelhead, Satus steelhead, Toppenish steelhead, Naches spring chinook salmon, American River spring chinook salmon; accept a maximum of 40% impact - mountain whitefish, resident rainbow trout in tributaries; and accept impacts up to a maximum that maintains all native species at sustainable levels. These objectives reflect the goal of having all native species at sustainable levels including populations robust enough to support harvest and recreation.

o Certain species or guilds within an ecosystem may interact strongly with spring chinook salmon resulting in failure to achieve desired numerical goals of spring chinook salmon through supplementation. Monitoring the influence of strong interactor taxa relative to spring chinook salmon will aid in interpreting factors that may influence YFP success. Recommended objectives for strong interactors are of two types: 1) identify and reduce impacts of <u>unnaturally</u> high predatory, pathogenic, and competitory interactions that limit spring chinook salmon productivity, and 2) identify, conserve and enhance mutualist and prey taxa function at or above baseline levels to support maintenance or enhancement of ecosystem capacity for spring chinook salmon.

o Preliminary results suggest that Pacific lamprey, leopard dace, and the North Fork Teanaway bull trout are at extremely low abundances and are narrowly distributed, sand roller may be extirpated, mountain sucker are not abundant but widely distributed, and mountain whitefish are very abundant and widely distributed. Umatilla dace (identification is tentative until genetic analysis is complete), which had not been identified in the Yakima basin previously, were found in the lower Yakima River.

o Predictive models reduced the number of years to detect a 10% impact to rainbow trout abundance in the mainstem Yakima River from 24 or more for raw data to a more practical one year. Much of the variation in rainbow trout abundance and biomass during the baseline period can be accounted for by variation in river discharge, resulting in predictable estimates against which potential impacts of spring chinook supplementation can be judged.

o Predictive models reduced the number of years to detect a 40% impact to rainbow trout abundance in tributaries to the Yakima River from nine or more (raw data) to one year. Predictive models and raw data for cutthroat and bull trout are currently not sufficient to rapidly or reliably detect small impacts to abundance, but monitoring is sufficient to reveal important changes in cutthroat trout distribution. Bull trout cannot be efficiently monitored within the present tributary salmonid monitoring program, however modification of the monitoring program will increase the potential for successful monitoring.

o Electrofishing-induced injuries in rainbow trout, juvenile steelhead, and juvenile spring chinook salmon were low when examined at monitoring index reach (0.7-11.2%) and stream (0.1-2.1%) scales because of the low proportion of fish that are electrofished compared to those present and/or the low injury rates. However, 28% of *Oncorhynchus mykiss* ≥ 250 mm fork length that were captured were injured, but less than 6% of *O. mykiss* < 250 mm in the Yakima River, *O. mykiss* in tributaries, or juvenile spring chinook salmon were injured.

o Anglers who fished for rainbow trout in the Yakima River in the summer of 1995 were predominantly western Washington residents, but some anglers were from Oregon, Idaho, Wyoming, California, Arizona, Colorado and Texas. Catch rates during the study period were relatively low and were affected by sample week, angler experience, fishing location, and whether anglers were fishing with a hired guide. Stream flows and fishing pressure were the top two factors anglers felt were limiting the trout population.

o Rotary screw traps located and operated below Roza Dam were insufficient for determining production of upper Yakima spring chinook salmon or steelhead or even to capture a sufficient number of fish for proposed survival studies. Low capture efficiency, high flows, large debris, and ice reduced or eliminated the effectiveness of screw traps. The juvenile bypass trap at Roza Dam was more effective at capturing fish than the screw traps and was operable during a greater variety of environmental conditions. Trapping revealed that the fall-winter migrant is the dominant life-history form of upper Yakima spring chinook salmon.

o The presence of non-migrant (residual) steelhead negatively impacted growth of wild rainbow trout but did not impact growth of spring chinook salmon in small screened enclosures placed in creeks. The cortisol levels, a measure of physiological stress, of fish paired with residual steelhead did not differ from those fish that were alone. However, the food habits of paired and unpaired fish differed.

o Juvenile spring chinook salmon did not impact the growth or abundance of age-0 or -1 rainbow trout in controlled experiments conducted in three small high elevation streams. It appears that rainbow trout and spring chinook salmon partitioned the resources so that impacts were not detected and that rainbow trout have an interactions refuge in small streams above 700 m.

o Gene flow between resident and anadromous *Oncorhynchus mykiss* has occurred in the Yakima basin. Naturally produced rainbow trout were genetically indistinguishable from naturally produced steelhead when collected in sympatry. In addition, estimates of hatchery and wild fish admixtures in naturally produced *O. mykiss* indicated that hatchery rainbow trout had previously spawned with steelhead; and hatchery steelhead had previously spawned with rainbow trout. Spawn distribution and timing of rainbow trout and steelhead also overlapped considerably.

o The maximum size of fall chinook salmon consumed by coho salmon smolts in laboratory trials was 74 mm and the largest relative size (fall chinook salmon length/coho salmon length) consumed was 46%. Initially (3 days; the approximate time it takes for actively migrating coho salmon smolts to exit the Yakima River), coho salmon consumed the smaller fall chinook salmon, generally less than 40% of the coho salmon's body length.

o Sufficiently large numbers of northern squawfish, smallmouth bass, and channel catfish were captured in the lower Yakima River to warrant the calculation of predation indices in future years. Northern squawfish and smallmouth bass had similar diets, with 23 to 29% of the stomachs of these species containing salmonids. Censuses of fish predators that potentially consume spring chinook salmon smolts should be conducted in the spring as opposed to the summer because predatory fish in some portions of the river migrate between the Yakima and Columbia rivers.

o Potential for adverse impacts resulting from ecological interactions among wild salmonids and hatchery steelhead was greatest when; (1) hatchery fish did not emigrate quickly; (2) water temperatures were over 8^{o} C; (3) hatchery fish were the same species as the wild salmonids; (4) hatchery fish were larger than the wild salmonids; (5) habitat and/or food were limiting; and 6) numbers of fish released was over about 30,000 per stream. Ecological interactions with wild salmonids could be reduced or minimized by releasing; (1) only actively migrating smolts (no residuals); (2) hatchery fish of a size that minimizes interaction potential (smaller than wild fish); (3) the minimum number necessary to meet management objectives; (4) fish that do not exhibit counter-productive and inappropriate behaviors (e.g., less likely to engage wild fish in agonistic encounters); and (5) when water temperatures are relatively cold (less than 8^{o} C) .

o A practical approach for assessing ecological risks associated with stocking anadromous salmonids is described. This risk assessment approach requires the completion of five tasks: 1) determine non-target taxa objectives in areas of overlap (e.g., acceptable impact of 10% to a species distribution, abundance, and size structure), 2) determine potential spatial-temporal overlap of NTT life stages, 3) determine potential strong ecological interactions, 4) determine ecological risk, and 5) determine scientific uncertainty.

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 Fisheries Project (YFP) planning, and 2) summarize results of research that have broader scientific relevance. This is the sixth of a series of progress reports that address species interactions research and pre-supplementation monitoring of fishes in response to proposed 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). This progress report summarizes data collected between January 1, 1995 and December 31, 1997. 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 Yakima Fisheries Project (YFP) 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 (reviewed in Pearsons et al. 1994; Busack et al. 1997). 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 YFP 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 salmon will be 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. In addition, concerns that stocking coho salmon may increase predation pressure to fall chinook populations was investigated. 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 fourteen chapters which represent major topics associated with NTT monitoring, spring chinook salmon monitoring, ecological interactions, and ecological risk. Chapters 1-6 address various topics of NTT monitoring including: draft containment objectives for NTT, feasibility monitoring of NTT that have previously received little attention, pre-supplementation monitoring of rainbow trout in the upper Yakima River and it's tributaries, angler surveys of the upper Yakima River, and the impacts of monitoring rainbow trout using electrofishing methods. Chapters 6 and 7 address methods for monitoring spring chinook salmon including injury assessment of electrofishing methods and monitoring migrants at or near Roza Dam. Chapters 8-12 address ecological interactions between residual hatchery-reared steelhead and wild rainbow trout and spring chinook salmon, wild spring chinook salmon and wild rainbow trout, rainbow trout and steelhead, coho salmon and fall chinook salmon, and predatory fish and spring chinook salmon. Finally, chapters 13 and 14 address methods to assess and minimize ecological risk. Chapter 14 also serves as a synthesis of the previous chapters. One task (assisting in the development of the YFP monitoring plan) that we helped to complete during the contract period is not represented as a chapter in this report. Results of this task have been described in: "Yakima Fisheries Project Spring Chinook Supplementation Monitoring Plan" by Busack et al. (1997).

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. Many of the chapters in this report are submitted, accepted, in press, or already published, in peer-reviewed journals or proceedings. Chapter 8 has been published and chapter 9 is in press in the journal "Transactions of the American Fisheries Society". Chapter 13 is in press in the "Proceedings of the Sustainable Fisheries Conference". Chapter 5 has been accepted and will be published in the "North American Journal of Fisheries Management". Chapter 10 has been submitted to the "Proceedings of the Inland Rainbow Trout Workshop" and chapter 11 has been submitted to "North American Journal of Fisheries Management". 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).

Acknowledgments

We would like to thank Bonneville Power Administration for financially supporting this work. In addition, we could not have completed this work without the help and support of many individuals during 1995-1997. Most of these individuals are recognized on title pages or in acknowledgments of various chapters in this report. We are also appreciative of those individuals who have had long-term positive impacts on this project such as Steve Leider, Steve Martin, and Greg Strom.

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Chapter 1

Draft Objectives for Non-target Taxa of Concern Relative to Supplementation of Upper Yakima Spring Chinook Salmon

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Abstract

Non-target taxa in the Yakima basin may be impacted by the supplementation of upper Yakima River spring chinook salmon (target taxa) or non-target taxa may limit the success of the supplementation program (i.e., Yakima Fisheries Project (YFP)). Supplementation of upper Yakima River spring chinook salmon may impact or be impacted by non-target taxa through a variety of ecological mechanisms including competition, predation, mutualism, altered behavior, pathogenic interactions, nutrient mining, and prey limitation. Monitoring appropriate attributes of non-target taxa will allow detection of impacts to non-target taxa, as well as identify biotic interactions that limit YFP success. Before a monitoring plan can be developed, specific objectives must be defined or identified. The different reasons for monitoring these two classes of ecological interactions necessitates different monitoring designs. Impacts to non-target species should be monitored in the context of ecological risk containment. Impacts to upper Yakima spring chinook salmon should be monitored in the context of ecological risk limitations to project success. To reflect ecological risk containment and ecological risk limitations differences, this report has been divided into two sections. The purposes of this report are to identify containment objectives for management of 1) fish taxa of concern that might be impacted by spring chinook salmon supplementation; and 2) taxa that have the potential to influence the success of the YFP. The overall goal is to achieve sustainable, productive and diverse native communities.

Objectives for highly valued non-target taxa of concern (NTTOC) were described relative to their current status. To achieve this endpoint three steps were taken. First, all taxa in the areas of presumed overlap with upper Yakima spring chinook salmon were identified. Second, taxa that were extraordinarily important with respect to stewardship and utilization values were identified. Stewardship values were prioritized higher than utilization values, regionally rare taxa were prioritized over locally rare taxa, and high use of utilization taxa was prioritized over low use. Third, containment objectives were proposed that reflected NTTOC prioritization. The status, and therefore the variables that impacts will be compared to, are distribution, abundance, and size structure. Impacts to a NTTOCs distribution, abundance, or size structure in excess of the containment objective would indicate a failure to achieve that objective.

The following containment objectives for NTTOC are proposed: **no impact - bull trout, westslope cutthroat trout, Pacific lamprey, upper Yakima redband steelhead, Naches redband steelhead, Satus redband steelhead, Toppenish redband steelhead; very low impact - Marion Drain fall chinook salmon, mountain sucker, leopard dace, sand roller; low impact - resident redband rainbow trout in the mainstem Yakima River, Naches spring chinook salmon, American River spring chinook salmon; moderate impact - mountain whitefish, resident redband rainbow trout in tributaries; and impacts up to a maximum that maintains all native species at sustainable levels**. These objectives reflect the goal of having all native species at sustainable levels including populations robust enough to support harvest and recreation.

Certain species or guilds within an ecosystem may interact strongly with spring chinook salmon resulting in failure to achieve desired numerical goals of spring chinook salmon through

supplementation. These taxa are referred to as strong interactor taxa (SIT). Monitoring the influence of SIT relative to spring chinook salmon will aid in interpreting factors that may influence YFP success. Furthermore, if interactions are demonstrated to limit project success, then corrective actions might be possible if detected early enough. Species or guilds, such as predators, competitors, or pathogens may negatively affect spring chinook salmon. Conversely, mutualists and prey may positively affect production of spring chinook salmon. Increases in the abundances or functions of predators, competitors, or pathogens, (negative interactors) or decreases in the abundance of mutualists and prey (positive interactors) may reduce the possibility of a successful supplementation program. Objectives for strong interactors are of two types: **1) identify and reduce impacts of unnaturally high predatory, pathogenic, and competitory interactions that limit spring chinook salmon productivity, and 2) identify, conserve and enhance mutualist and prey taxa function at or above baseline levels to support maintenance or enhancement of ecosystem capacity for spring chinook salmon**.

Identification of hypothetical strong interactors was accomplished by reviewing scientific literature, using available ecological information in the Yakima basin, and using ecological theory. Hypothetical strong interactors that were identified are as follows: **predators - northern pikeminnow, rainbow trout/steelhead, torrent sculpin, mottled sculpin, shorthead sculpin, smallmouth bass, channel catfish, gulls, common merganser, great blue heron, double-crested cormorant, common loon, western grebe, and otter; pathogens - viruses, bacteria, fungi, parasites; competitors - redside shiner, rainbow trout/steelhead, and mountain whitefish; mutualists - riparian vegetation, and beaver; prey - ephemeroptera, plecoptera, trichoptera, diptera, and coleoptera**. Baseline characterization of strong interactions is necessary in order to detect changes in impacts to spring chinook salmon.

The benefits of managing multiple taxa within the Yakima ecosystem are obvious and yet rarely has it been accomplished in other basins. The success of spring chinook supplementation within the YFP may depend upon knowledge of critical ecological interactions. In addition, further governmental regulations, such as those associated with the endangered species act, may not be necessary if populations within the Yakima are healthy. Finally, knowledge about ecological interactions in riverine environments is scarce, and the kinds of ecological information and procedures that are developed in the Yakima basin will be invaluable to others working in the Columbia basin.

We acknowledge that selection of objectives for NTTOC is based on values. Furthermore, we do not presume that the values represented by the objectives proposed in this report are representative of WDFW as a whole or the YIN. We recommend that the WDFW and YIN negotiate a common set of objectives that can be used to guide ecological risk containment monitoring in the Yakima basin. Acceptance of a set of objectives that are mutually acceptable to co-managers in the Yakima basin will provide the impetus for development of ecological risk containment and project limitation monitoring plan development.

General Introduction

The purpose of this report is to recommend ecological interactions objectives that promote 1) conservation of highly valued fish taxa; and 2) rebuilding of the upper Yakima spring chinook salmon population, relative to a spring chinook supplementation program in the upper Yakima basin (Figure 1). These conservation and production objectives are part of a larger effort to successfully implement and test the concept of supplementation. The Yakima Fisheries Project's (YFP) definition of supplementation is: "the use of artificial propagation in an attempt to **m aintain or increase natural production** while maintaining the long term fitness of the target population, and **keeping the ecological** *and genetic* **impacts on non-target populations within specified biological limits**" (RASP 1992; BPA 1996). Clearly, this definition highlights the goals of maintaining or increasing natural production of the target population and conserving non-target populations. Other goals of the YFP are to test the concept of supplementation (learning benefits), increase harvest, conserve genetic resources of non-target populations, and maintain the long term fitness of the target population. Objectives for containment of ecological interactions must be established to clearly determine whether the YFP is succeeding in managing ecological impacts. Once established, the objectives will be the basis for the development of an ecological interactions monitoring plan (Busack et al. 1997).

This report is divided into two parts. Part 1 is devoted to developing a proposed set of impact containment objectives for non-target taxa of concern (NTTOC). Part 2 is devoted to describing objectives that would contribute to attaining project production objectives.

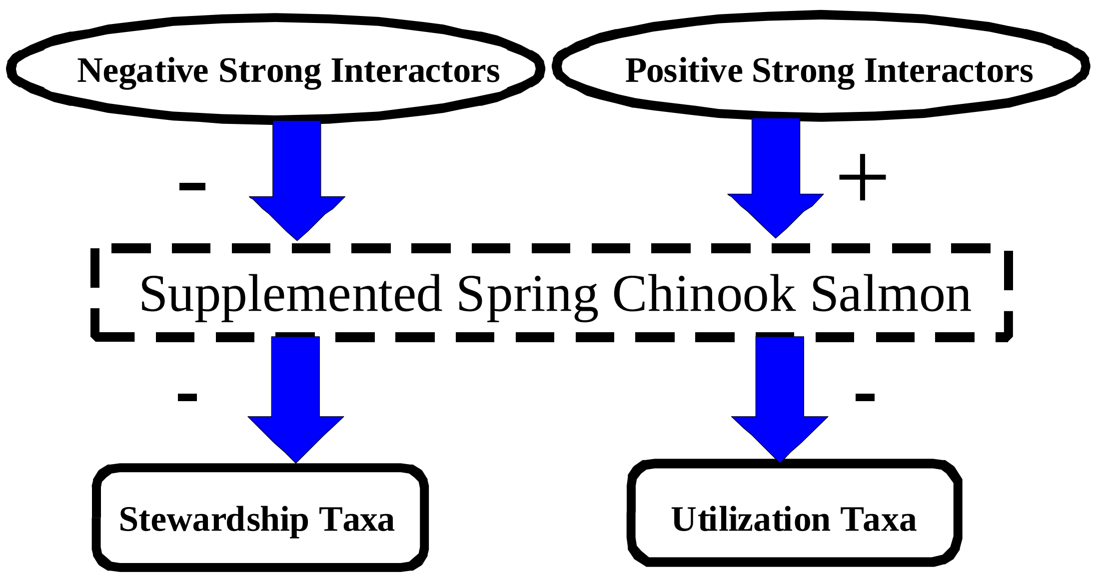

Figure 1. Critical ecological interactions affecting the success of the Yakima Fisheries Project. The top half of the figure represents topics covered in Part 2 of this report and the lower half represents topics covered in Part 1.

Part 1: Ecological risk containment objectives for non-target taxa of concern (stewardship and utilization)

Introduction

Inherent within the Yakima Fisheries Project's (YFP; a.k.a Yakima/Klickitat Fisheries Project or Upper Yakima Spring Chinook Supplementation Project) definition of supplementation is the concept that trying to increase natural production of a target species, using artificial technologies, may impact non-target taxa (RASP 1992; BPA 1996). Indeed, hatcheries have been implicated for contributing to the decline of both target and non-target populations (Marnell 1986; Miller et al. 1990; Steward and Bjornn 1990; Krueger and May 1991; Nehlsen 1991; White et al. 1995; Pearsons et al. 1996; McMichael et al. 1997). Supplementation may impact non-target taxa through a variety of ecological mechanisms including competition, predation, behavioral anomalies, nutrient mining, and pathogenic interactions. Unfortunately, in many instances little or no attention has been directed at containing ecological risks associated with hatchery or supplementation projects. This will not be the case for the YFP. The Yakima Fisheries Project Environmental Impact Statement says "The project managers will define or identify objectives for management of the key non-target species before the project is implemented, so that an effective monitoring plan can be developed and implemented" (BPA 1996, page 164). That is, the YFP plans to supplement upper Yakima spring chinook salmon and conduct ecological risk containment monitoring to ensure healthy populations of native fishes.

The goal of ecological risk containment monitoring in the Yakima basin is to provide an indication of the change in status of non-target taxa in relation to ecological interactions that result from the YFP. This information can then be used to take appropriate management actions to contain the risks to non-target taxa. However, a monitoring plan to contain or manage the risks of adverse ecological interactions resulting from the YFP can only be developed after specific objectives for non-target species have been defined or identified. The purpose of this report is to propose objectives for containing biological impacts to non-target taxa in the Yakima basin in response to supplementation of upper Yakima spring chinook salmon. This is the necessary first step towards development of meaningful ecological risk containment monitoring plans by the YFP's Monitoring Implementation Planning Team (Busack et al. 1997).

Unfortunately, monitoring all non-target taxa in the ecosystem is impractical if not impossible. Therefore, objectives will be identified and monitored for a prioritized subset of taxa in the affected ecosystem. Monitoring will be used to determine achievement of objectives. Selection of the appropriate subset of taxa and appropriate monitoring effort will result in appropriate risk containment monitoring for the fish community in the Yakima basin.

An appropriate subset of non-target taxa that should be monitored and managed should reflect societal values such as stewardship (e.g., social, cultural, religious, scientific) and utilization (e.g., food, recreation, economic, religious ceremonies). Stewardship values can be represented as maintaining native populations at or above sustainable levels. Stewardship values would not be represented well if native populations were managed (inadvertently or otherwise) into extinction or if the genetic or ecological capacity of the taxa was significantly impaired. Utilization values can be represented through productive populations of desirable taxa whose abundance is sufficient enough to allow use (recreational angling, harvest). These values are reflected in expressions of various management entities in the region. For instance, the legislative mandate of the Washington State Department of Fish and Wildlife (WDFW) is to preserve, protect, perpetuate, and manage fish and wildlife and to provide commercial and recreational uses of animal resources (RCW 77.12.010; RCW 75.08.012; RCW 75.08.013); the WDFWs draft wild salmonid policy gives highest protection to critical and endangered status salmonid stocks and secondarily gives higher priority to those stocks that provide the greatest level of benfits or value (i.e., economic, social, ecological, cultural, and others; WDFW 1997); and the U.S. Endangered Species Act was enacted to minimize extinction of a species, subspecies, or distinct population segment of a species throughout a significant portion of its range (Waples 1995).

Prioritization of societal values should be reflected in the levels of protection afforded to non-target taxa. For instance, those taxa that are rare should have the most conservative containment objectives and those taxa that are highly utilized for food or recreation should have lower containment objectives than those taxa that are robust and rarely utilized. Populations that are severely depressed may go extinct if additional stresses to the populations are encountered. In contrast, stresses to healthy populations may decrease their productivity and subsequent utilization, but long term harm to the population would be negligible.

This document only refers to NTTOC and interactions that occur in the Yakima basin. Other NTTOC and interactions with NTTOC from anadromous fish originating from the Yakima basin may occur in the Columbia River, Columbia estuary, and ocean.

Methods

The methods that were used to develop proposed objectives for non-target species of concern are outlined in Table 1. Three steps were used to identify NTTOC. First, those non-target taxa that will presumably overlap with the target species were identified. It is assumed that only those taxa that overlap with spring chinook salmon will interact significantly. Second, those taxa that might overlap with spring chinook salmon and that are extraordinarily important with respect to societal values were identified. Third, quantitative goals for non-target taxa of concern were described. Details of these three steps are described below.

Species lists, step I, were developed from Patten et al. 1970, Mongillo and Faulconer 1980, McMichael 1991, and Jim Cummins (WDFW, personal communication). A taxon was predicted to overlap with expanding populations of spring chinook salmon in the future if the taxa was below an unladdered barrier in the Yakima River (i.e., Keechelus Dam) or in the tributaries of the upper Yakima River (i.e., Cle Elum Dam). Although pygmy whitefish is a state candidate species, it was not considered further because it is distributed above barriers and would not overlap with spring chinook salmon. Stocks of steelhead and salmon were determined electrophoretically (Craig Busack, WDFW, personal communication). Resident rainbow trout in the upper Yakima River were split into two groups - those that inhabit the mainstem and those that inhabit the tributaries. This was done for three reasons: 1) rainbow trout in tributaries are generally genetically different than those in the mainstem (Phelps and Baker 1994); 2) angling regulations for the tributaries are different than those in the mainstem; and 3) angling pressure in the mainstem is much higher than in the tributaries. Finally, stocks of bull trout were determined using available life history information and geographic location data (WDFW 1997).

Table 1. Outline of the methods that are used to determine risk containment objectives for non-target taxa of concern.

I. Identify non-target taxa in areas of presumed overlap with target taxa
    A. Develop complete taxa list - species, stocks, management types
    B. Determine hypothetical overlap
        1. Current and historical distribution in basin
        2. Environmentally suitable areas
        3. Relative distributions in other basins

II. Identify non-target taxa of concern
    A. Stewardship
        1. ESA list, state list
        2. Local information - trends and current status
        3. Native taxa
    B. Utilization
        1. Angler use
        2. Harvest

III. Identify quantitative goals for non-target taxa of concern
    A. Use existing regulations for guidance (e.g., ESA)
    B. Inventory resource users
    C. Other taxa not covered by A and B, are prioritized and goals are set relative to those taxa with goals identified
        1. Prioritize taxa with respect to societal values
            a. Rare - regionally (across a taxons range)
            b. Rare - locally (within a taxons range; e.g., Yakima basin)
            c. Very important utilization taxa
            d. Important utilization taxa
            e. Common native species

IV. Negotiate objectives with appropriate parties
    A. WDFW
    B. YIN

Selection of NTTOC that might be affected by supplementation of spring chinook salmon (Step II) was accomplished by determining those taxa that were inordinately important because of stewardship and utilization values. All native fish taxa were identified as taxa that should be stewarded. Furthermore, those taxa that were rare regionally or locally (within the Yakima basin) were extraordinarily valued. A fish was deemed to be regionally rare if it was listed as a category 1 or 2 species on the Federal Register. A fish was designated to be locally rare if few individuals have been collected during the past decade. Those taxa that are used for food or recreation were termed utilization taxa. These taxa were determined through consultation with WDFW biologists and by examining the WDFW fishing regulation pamphlet. Native game or food fish were judged to be very important (as opposed to important) based upon perceived or actual use. Exotic species were not included as NTTOC because the targeted species is of native origin and is therefore more highly valued. In other words, because the target species (upper Yakima spring chinook salmon) is of native origin, any impact to exotic species is acceptable. In summary, five categories of NTTOC were identified: rare regionally, rare in-basin, very important game or food fish, important game or food fish, and common native taxa.

Quantitative containment objectives for NTTOC were framed in terms of the acceptable impact to a taxon's baseline status or health (prior to supplementation; Step III). The statuses of taxa were described with the following three attributes: distribution, abundance, and size structure. For example, a containment objective of 5% refers to a decline in the status of a taxon as reflected by baseline characteristics of it's distribution, abundance, and size structure. If any one of these variables are impacted by 5% or more, and the impact can be attributed to supplementation, then the project would fail to meet the objective. Conversely, if the impact could not be attributed to supplementation, then the project would not fail with respect to the objective. Impacts might be attributed to supplementation by the use of small scale experiments within the treated area or other methods (Pearsons et al. 1993). In cases where objectives for one taxon limit achievement of objectives for another taxon, highest protection will be given to the highest priority taxon. Taxa that could be included in multiple categories were placed in the most conservative category. For instance, westslope cutthroat trout could be included as a stewardship or utilization taxa, but because it is a category 2 species on the Federal Register it should be placed in the category that receives the highest protection (i.e., lowest impact levels).

Containment objectives were set relative to a combination of existing regulations (e.g., Endangered Species Act), inventories of resource users, and relativistically - according to prioritized importance. The containment objectives were assigned to a category of stewardship and utilization (e.g., rare regionally) that included many taxon. The Endangered Species Act was used as a reference to set an acceptable containment objective for those taxa that are rare regionally. Anglers and sport clubs were surveyed to determine the containment objective for rainbow trout density, distribution, growth, size structure, and catch rate from supplementing spring chinook salmon. Surveys were conducted during 1995. Members from the Cascade Field and Stream Club (9), Northwest Flyanglers (21), and the Yakima flyfishers (11) were surveyed (club members). One hundred anglers that were fishing the Yakima River (anglers) were also surveyed (chapter 6). The question that was asked was:

Assuming that spring chinook supplementation occurs and that it results in increased *natural production of adult spring chinook salmon; What maximum level of impact (according to variables such as abundance, growth, size, distribution, and catch rate) to the resident rainbow trout population would you be willing to accept? (Please check the level of reduction that you would be willing to accept)*

___ none ___ 10% ___ 25% ___ 50% ___ 75% ___100%.

Answers to the questions were mathematically averaged and the results summarized graphically. All taxa that fit within the same category as Yakima River rainbow trout (very important native game or food fish) received the same containment objective. Other taxa that were categorized differently (e.g., rare in the Yakima basin or important game or food fish), were assigned containment objectives that were relative to previously assigned levels. These relative impact levels were assigned values according to their prioritized importance. The following priorities were identified in order of importance; stewardship over utilization, rare over common, native over non-native, use is very important over important.

Results

Forty eight fish species have been identified in the Yakima basin (Table 2). Twenty nine of these species are of native origin. Eighteen stocks of chinook salmon, *O. mykiss*, and bull trout were identified (Table 3). Most of the non-target fish taxa in the Yakima Basin were predicted to overlap with supplemented spring chinook salmon (Table 2 and 3). However, only 9 out of 48 species (19%) met the criteria necessary to have containment objectives identified (Table 4). Of course, one non-specific objective is to maintain all native species at sustainable levels which includes 29 out of 48 species (60%). Excluding the non-specific objective, approximately half of the NTTOC were selected because of stewardship and the other half because of utilization values.

Quantitative objectives ranged from no impact to a NTTOC distribution, abundance, or size structure to impacts up to a maximum that maintains NTTOC at sustainable levels. The objective proposed for NTTOC that were rare regionally was no impact. This determination was made from language in the Endangered Species Act which prohibits the "take [of] any such species within the United States or the territorial sea of the United States" (Section 9,1,B). A "take" means to "harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct" (Section 3,19). NTTOC that were classified as rare-regionally were bull trout, westslope cutthroat trout, steelhead, and Pacific lamprey. Of these four species, bull trout is the only one that is listed as threatened, but conservative management of westslope cutthroat trout, steelhead, and Pacific lamprey in the Yakima basin may prevent listing of these species in the future.

Table 2. List of species found in the Yakima basin and species that might overlap with supplemented upper Yakima basin spring chinook salmon. Information was compiled from 1) Patten et al. (1970), 2) Mongillo and Faulconer (1980), 3) McMichael (1991), and 4) Jim Cummins (Personal communication, WDFW). Origin: N = native, H=current hatchery programs or known hatchery ancestry, E = exotic or non-native.

Common Name	Scientific Name	Overlap	Origin	Reference
Western brook lamprey	*Lampetra richardsoni*	X	N	1
Pacific lamprey	*Lampetra tridentata*	X	N	1
White sturgeon	*Acipenser transmontanus*	X	N	3
Coho salmon	*Oncorhynchus kisutch*	X	N,H	1
Chinook salmon	*Oncorhynchus tshawytscha*	X	N	1
Sockeye salmon	*Oncorhynchus nerka*	X	N,H	1
Rainbow trout	*Oncorhynchus mykiss*	X	N,H	1
Cutthroat trout	*Oncorhynchus clarki*	X	N,H	1
Brown trout	*Salmo trutta*	X	E,H	1
Bull trout	*Salvelinus confluentus*	X	N	1
Brook trout	*Salvelinus fontinalis*	X	E	1
Lake trout	*Salvelinus namaycush*	X	E	2
Mountain whitefish	*Prosopium williamsoni*	X	N	1
Pygmy whitefish	*Prosopium coulteri*		N	2
Northern pikeminnow	*Ptychocheilus oregonensis*	X	N	1
Chiselmouth	*Acrocheilus alutaceus*	X	N	1
Peamouth	*Mylocheilus caurinus*	X	N	1
Redside shiner	*Richardsonius balteatus*	X	N	1
Speckled dace	*Rhinichthys osculus*	X	N	1
Longnose dace	*Rhinichthys cataractae*	X	N	1
Leopard dace	*Rhinichthys falcatus*	X	N	1
Common carp	*Cyprinus carpio*	X	E	1
Goldfish	*Carassius auratus*	X	E	3
Bridgelip sucker	*Catostomus columbianus*	X	N	1
Largescale sucker	*Catostomus macrocheilus*	X	N	1
Mountain sucker	*Catostomus platyrhynchus*	X	N	1
Sand roller	*Percopsis transmontana*	X	N	1

Burbot	*Lota lota*		N	2
Threespine stickleback	*Gasterosteus aculeatus*	X	N	2
Black bullhead	*Ictalurus melas*	X	E	1
Brown bullhead	*Ictalurus nebulosus*	X	E	3
Channel catfish	*Ictalurus punctatus*	X	E	2
Flathead catfish	*Pylodictis olivarus*	X	E	4
Smallmouth bass	*Micropterus dolomieui*	X	E	1
Largemouth bass	*Micropterus salmoides*	X	E	1
Black crappie	*Pomoxis nigromaculatus*	X	E	1
White crappie	*Pomoxis annularis*	X	E	3
Green sunfish	*Lepomis cyanellus*	X	E	3
Bluegill	*Lepomis macrochirus*	X	E	1
Pumpkinseed	*Lepomis gibbosus*	X	E	1
Yellow perch	*Perca flavescens*	X	E	1
Walleye	*Stizostedion vitreum*	X	E	3
Mosquitofish	*Gambusia affinis*	X	E	3
Mottled sculpin	*Cottus bairdi*	X	N	1
Torrent sculpin	*Cottus rhotheus*	X	N	1
Piute sculpin	*Cottus beldingi*	X	N	1
Shorthead sculpin	*Cottus confusus*	X	N	3
Prickly sculpin	*Cottus asper*	X	N	1

Table 3. List of stocks and management types found in the Yakima basin. References for the stock identification were provided by the following WDFW employees Craig Busack (1), Steve Phelps (2), and Eric Anderson (3).

Stock	Overlap	Origin	Reference
Fall chinook salmon			
Marion Drain	X	N	1
Lower Yakima River	X	N,H	1
Spring chinook salmon			
Upper Yakima River	X	N	1
Naches	X	N	1
American River	X	N	1
Summer Steelhead[a]			
Upper Yakima	X	N,H	2
Naches	X	N,H	2
Satus	X	N	2
Toppenish	X	N	2
Resident rainbow trout			
Upper Yakima River mainstem	X	N,H	2
Upper Yakima River tributaries	X	N,H	2
Bull trout			
Mainstem Yakima River	X	N	3
Ahtanum Creek		N	3
Naches River	X	N	3
Rimrock Lake		N	3
Bumping Lake		N	3
North Fork Teanaway River	X	N	3
Cle Elum/Waptus Lakes		N	3
Kachess Lake		N	3
Keechelus Lake		N	3

[a] Electrophoretic differences between upper Yakima and Naches stocks were inconclusive presumably because the samples were contaminated with resident rainbow trout

A containment objective of less than or equal to 10% was proposed for very important native game or food fish (Table 4). This objective was determined by examining averages and modes of responses from anglers fishing the Yakima River (anglers) and angling club members (club members). The most common response from anglers was that no impact was acceptable (Figure 2). The most common response from club members was that a maximum of 10% impact was acceptable (Figure 2). On average, anglers would accept a maximum of 8% impact and club members a 5% impact. Values were rounded up to 10% to account for resource users who do not fish for rainbow trout. The 10% objective was assigned to all native game or food fish of high importance.

A containment objective of less than or equal to 5% was proposed for NTTOC that were rare in the Yakima basin because they were prioritized between rare-regional NTTOC (no impact) and very important game or food fish (10%). A containment objective of 40% was proposed for important native game or food fish because the amount of use was considered to be at least four times less than that for very important game or food fish (10%) and they are extremely abundant (Chapter 2). Finally, as a general objective, conservation objectives for all other native species was proposed to be up to a maximum that maintains all native species at sustainable levels. This objective is intended to reflect the value that no native species should be allowed to go extinct in the Yakima basin as a result of spring chinook salmon supplementation.

All native rainbow trout and steelhead in the Yakima basin are classified as redbands, which are primitive forms of *O. mykiss* (Behnke 1992). In this report, *O. mykiss* will generally be referred to as rainbow or steelhead trout.

Table 4. Proposed containment objectives for non-target taxa of concern in the Yakima Basin relative to supplementing the upper Yakima stock of spring chinook salmon. Objectives refer to negative impacts upon one or more of a taxas distribution, abundance or size structure relative to pre-supplementation levels.

NTTOC	Containment Objective
Rare - species, stock, or regionally	no impact
Bull trout	
Westslope cutthroat trout	
Pacific Lamprey	
Naches steelhead	
Satus steelhead	
Toppenish steelhead	
Upper Yakima steelhead	
Rare - in basin	Very low impact (≤ 5%)
Marion Drain Fall chinook	
Mountain sucker	
Leopard dace	
Sand roller	
Native game or food fish - very important	Low impact (≤ 10%)
Resident rainbow trout in the mainstem Yakima River	
Naches spring chinook salmon	
American River spring chinook salmon	
Native game or food fish - important	Moderate impact (≤ 40%)
Mountain whitefish	
Resident rainbow trout in tributaries	
Common	≤ maximum impact that maintains all native species at sustainable levels
Other native species	

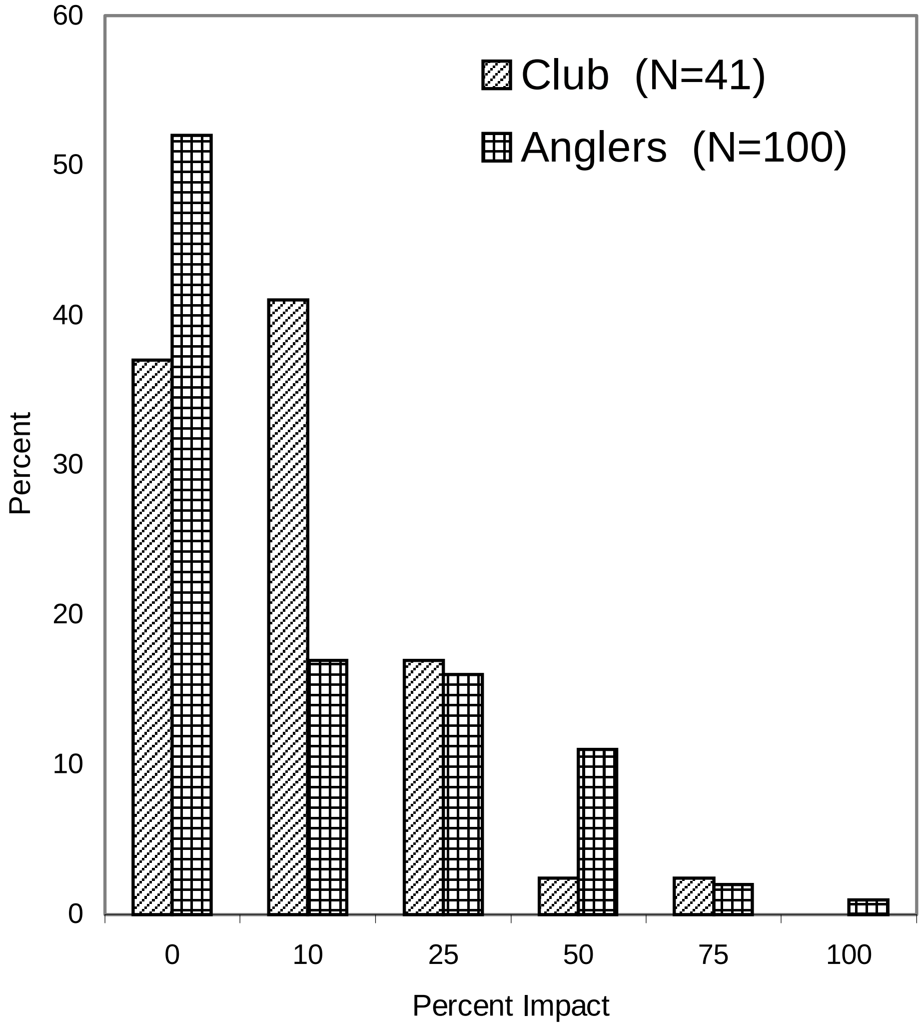

Figure 2. Acceptable impact levels of resident rainbow trout (mainstem) relative to spring chinook salmon supplementation. Survey results were from angling clubs and anglers on the upper Yakima River.

A brief rationale for selecting objectives for each of the NTTOC is described below. The NTTOC are arranged in order of increasing containment objective levels.

Bull trout

Bull trout in the Columbia Basin have been listed as "threatened" by the U.S. Fish and Wildlife Service. Bull trout have experienced broad scale declines throughout the state of Washington (Mongillo 1993, Brown 1994, WDFW 1997). In the Yakima basin, bull trout are most abundant in reservoirs and tributaries that drain into reservoirs. A few resident populations are restricted to high elevation portions of streams and a few large individuals have been collected in the mainstem of the Yakima River. Stock status reports indicate that most stocks in the Yakima basin are in depressed or critical condition (WDFW 1997). The only known stock in the upper Yakima basin, below barriers, where bull trout have consistently been observed is in the North Fork of the Teanaway River (Pearsons et al. 1996). Spawning and feeding ecology of adfluvial bull trout in tributaries to Rimrock Reservoir are described by Sexauer (1994) and James (1997).

Westslope cutthroat trout

Westslope cutthroat trout are currently recognized as a category 2 candidate species as listed by the U.S. Fish and Wildlife Service. Westslope cutthroat trout are widely distributed throughout the Yakima basin but are primarily found in high elevation portions of tributaries although some large individuals are regularly found in the mainstem of the Yakima River (Pearsons et al. 1996).

Pacific Lamprey

Pacific Lamprey are currently recognized as a category 2 candidate species as listed by the U.S. Fish and Wildlife Service. Pacific Lamprey are declining in most, if not all, areas of the Columbia basin (Close et al. 1995). Historically, native Americans fished for lamprey in the Yakima basin, which suggests that they were quite abundant (Hunn, 1990). Currently, lamprey are not harvested in the Yakima River because of their scarcity. Although fish counting facilities at Prosser and Roza dams are not equipped properly for counting adult lamprey, some have been observed during the past few years. Five adult Pacific lamprey have been observed on video tapes between 1992 and 1996 at Prosser Dam and one at Roza Dam (Joel Hubble, YIN, personal communication). In 1995, five adults were collected, and in 1996 one adult was collected at the Chandler Juvenile Fish Facility (Mark Johnston, YIN, personal communication). Furthermore, the following number of juvenile lamprey (western brook or Pacific lamprey - not identified to species) were collected at the Chandler Juvenile Fish Facility between 1993 and 1996: 1993 - 613; 1994 - 102; 1995 - 367; 1996 - 27 (Mark Johnston, YIN, personal communication).

Upper Yakima River steelhead

Steelhead in the middle Columbia basin have been proposed as threatened and are likely to be listed in 1999. Little is known about the spawning escapement of upper Yakima steelhead. Rough estimates of adult steelhead at Roza Dam are 70, 78, and 126 (mean 91) for brood years 1990, 1991, and 1992 (YIN, 1996). Despite intensive trapping and spawning survey efforts between 1991 and 1996 few adult steelhead spawners or redds have been observed in the upper Yakima basin (Hindman et al. 1991, McMichael et al. 1992, Pearsons et al. 1993, Pearsons et al. 1994, Hockersmith et al. 1995, Pearsons et al. 1996).

Naches steelhead

Steelhead in the middle Columbia basin have been proposed as threatened and are likely to be listed in 1999. Steelhead escapement into the Naches in 1990, 1991, and 1992 was 232 (all wild), 312 (288 wild, 24 hatchery, and 545 (523 wild, 22 hatchery) respectively (YIN, 1996). Movement of adult Naches steelhead is described in Hockersmith et al. (1995).

Satus Creek steelhead

Steelhead in the middle Columbia basin have been proposed as threatened and are likely to be listed in 1999. The average estimated escapement into Satus Creek between 1988 and 1995, excluding 1992, was 384 (YIN 1996). The biology of Satus Creek steelhead was described by Hubble (1992) and Hockersmith et al. (1995).

Toppenish Creek steelhead

Steelhead in the middle Columbia basin have been proposed as threatened and are likely to be listed in 1999. Population abundances are unknown for Toppenish steelhead. However, 47, 26 and 30 redds were observed in Toppenish Creek in 1989, 1990, and 1991 respectively (Mark Johnson, personal communication, Yakama Indian Nation, 1996). Movement of adult Toppenish Creek steelhead is described in Hockersmith et al. (1995).

Marion Drain fall chinook salmon

Although the spawning escapement of Marion Drain fall chinook salmon has averaged 341 between 1991 and 1995, most of these fish are jacks (YIN, 1996). An average of only 36 redds have been observed in Marion Drain between 1991 and 1995. Between 1983 and 1989 an average of 82 redds were observed. Biology of Marion Drain fall chinook salmon is described by Hollowed (1984).

Mountain sucker

Mountain sucker are classified as a state "monitoring" species (Geist 1995). They have been collected in the Yakima River from rkm 48 to 266 during surveys conducted between 1957 and 1958 (Patten et al. 1970). During sampling from 1990 to 1995 in the upper Yakima basin, a few mountain sucker have been collected in the mainstem of the upper Yakima River and in the lower portions of the Teanaway River. Life history of the mountain sucker in Montana is described by Hauser (1969).

Leopard dace

Leopard dace are only found in the Columbia and Fraser river basins and have been collected in the Yakima River from rkm 24 to 250 during surveys conducted between 1957 and 1958 (Patten et al. 1970). In addition, leopard dace were collected in the Yakima River near Zillah on December 16, 1930 (UWFC). During sampling from 1990 to 1995 in the upper Yakima basin no leopard dace have been collected by the WDFW in the mainstem or tributaries (Pearsons et al. 1996). However, in 1994, 18 unconfirmed leopard dace were collected in two creeks (Cooke and Dry Creeks; UWFC) that had previously been sampled cursorily. The biology of the leopard dace inhabiting the Fraser River is described by Gee and Northcote (1963).

Sand roller

Sand roller are endemic to the Columbia River basin and have been collected in the Yakima River between rkm 145 and 161 during surveys conducted between 1957 and 1958 (Patten et al. 1970). Sand roller are a state "monitoring" species (Geist 1995). No collections of sand roller have been reported in the Yakima basin since 1958. Biology and distribution of the sand roller in the central Columbia River is described by Gray and Dauble (1979).

Resident rainbow trout in the upper mainstem Yakima River

Rainbow trout that inhabit the mainstem of the upper Yakima basin provide the best naturally produced stream trout fishery in the state of Washington (Krause 1991; Probosco, 1994). To enhance the fishery in the mainstem of the upper Yakima River, a catch and release regulation was instituted in 1990. Average densities of rainbow trout in index sites exceeded 300/km between 1991 and 1994 in the reach between Roza Dam and the Cle Elum River confluence (Pearsons et al. 1996). Biology of resident rainbow in the upper mainstem Yakima River can be found in (Mongillo and Faulconer 1980; Campton and Johnston 1985; Fuller, 1990; Hindman et al. 1991, McMichael et al. 1992, Pearsons et al. 1993, Pearsons et al. 1994, Murdoch 1995; Hockersmith et al. 1996, Pearsons et al. 1996).

Naches River spring chinook salmon

Chinook salmon have long been an important food fish for native Americans (Hunn, 1990) and others. Between 1982 and 1995, the mean number of Naches spring chinook salmon returns to the mouth of the Yakima River was 924 and the mean spawning escapement was 802 (YIN, 1996). The biology of Naches spring chinook salmon can be found in Major and Mighell (1969), Fast et al. (1991) and Hockersmith et al. (1994).

American River spring chinook salmon

Chinook salmon have long been an important food fish for native Americans (Hunn, 1990) and others. Between 1982 and 1995, the mean number of American River spring chinook salmon returns to the mouth of the Yakima River was 491 and the mean spawning escapement was 427 [YIN 1996]. The biology of American River spring chinook salmon can be found in Major and Mighell (1969), Fast et al. (1991), and Hockersmith et al. (1994).

Mountain whitefish

Mountain whitefish are among the most numerous species in the upper Yakima River (Pearsons et al. 1996). They provide a significant recreational harvest opportunity for anglers during times when other fishing opportunities are minimal.

Resident rainbow trout in tributaries

Rainbow trout in tributaries are used for recreation and food primarily during the summer. Tributaries that are used frequently are Wilson Creek, Taneum Creek, Swauk Creek, Manastash Creek, and the three forks of the Teanaway River. Rainbow trout are distributed throughout most tributaries of the upper Yakima basin excluding some high elevation portions (Pearsons et al. 1996). Densities of rainbow trout in tributaries generally range from 0.01 to 0.20 and are generally highest in Taneum and Swauk creeks (Pearsons et al. 1996). Biology of resident rainbow in tributaries to the upper Yakima River can be found in (Mongillo and Faulconer 1980; Campton and Johnston 1985; Hindman et al. 1991, McMichael et al. 1992, Pearsons et al. 1993, Pearsons et al. 1994, Murdoch 1995; Hockersmith et al. 1996, Pearsons et al. 1996).

Other native species

Native species other than those described above are generally in good condition (Pearsons et al. 1996). Catostomids, Cottids, and *Rhinichthys* are generally the most abundant taxa in the upper Yakima basin (Pearsons et al. 1996).

Discussion

Non-target taxa risk containment objectives proposed in this report can be used to guide the development of a monitoring plan for non-target taxa. The goal of the ecological risk containment monitoring plan will be to determine if the objectives are achieved and, if they are not, the cause for failing to achieve the objectives. In some cases, the deficiency of baseline data, small sample sizes, or high natural variation will preclude determination of an objective at the specified containment level. In such cases, the ability to contain ecological risks is relatively low and the merit of the project should be weighed against the risk of failing to meet the objective. An assessment of the ecological risks can also be used to assign a probability of failing to meet an objective (Table 5). If the risk is deemed acceptable, then the taxa of concern will be monitored until more severe consequences (with respect to the stated objective) are detected. For example, the proposed objective for Pacific lamprey is no impact, but impacts may not be statistically detectable until they reach 75%. This should not result in a relaxation of the objective (e.g. changing acceptable impact levels to 75%) but rather an acknowledgment that monitoring will not allow the detection of an impact until it has far surpassed the specified objective. An objective is based on societal values but the ability to detect achievement of an objective is a scientific issue. Therefore, scientific limitations should not change what society deems as important. However, the scientific deficiencies should be fully acknowledged and assessed with respect to ecological risk.

A two-phased monitoring approach can be used to minimize the cost of monitoring and provide adequate ecological risk containment. If the status of a NTTOC begins to decline then this would trigger the need to determine the cause of the decline. In the first phase, achievement of objectives will be determined. If impacts are well within containment objectives then further monitoring is unnecessary. However, if impacts are being detected, then controlled field experiments are necessary to determine if supplementation is the cause of the impacts unless impacts are obviously due to other factors such as flooding. In some cases, rarity of the NTTOC would preclude attempting formal experiments with them as subjects. In such cases a surrogate species might be used to determine the cause of an impact. For example, resident rainbow trout could be used as a surrogate for upper Yakima steelhead or western brook lamprey could be used as a surrogate for Pacific lamprey. Deviations from this general approach should be considered if there is a high probability of failing to meet an objective or if there is a need for scientific information that might be used for other applications.

An additional way to prioritize monitoring effort is to focus effort on those taxa that are at highest risk of failing to meet an NTTOC objective. For example, high risk NTTOC should be monitored more intensively than low risk NTTOC. A preliminary risk assessment is provided in Table 5 as a means to facilitate monitoring prescriptions for NTTOC. Examination of Table 5 suggests that upper Yakima steelhead, Pacific lamprey, resident rainbow trout (mainstem), bull trout, westslope cutthroat trout, and Marion Drain fall chinook should receive the highest priority for monitoring effort. A more formal risk assessment and monitoring prescription, as described in chapter 14, should be completed in the future to increase accuracy.

Despite the importance of the NTTOC, very little is known about most of their distribution, abundance, and size structure in the Yakima basin. Therefore, it is extremely important to begin baseline monitoring of these taxa as soon as possible. Without baseline data it will be difficult to determine achievement of NTTOC objectives. Furthermore, this report focussed on objectives for fish but objectives should also be developed for other taxa that could be impacted by the YFP such as amphibians (e.g., tailed frog), reptiles (e.g., sharp tailed snake), birds (e.g., bald eagle) and mammals (e.g., bear).

We acknowledge that selection of objectives for NTTOC is based on values. Furthermore, we do not presume that the values represented by the objectives described in this report are representative of WDFW as a whole or the YIN. We recommend that the WDFW and YIN negotiate a common set of objectives that can be used to guide ecological risk containment monitoring in the Yakima basin.

Table 5 . Preliminary ecological interaction potentials, types, and risks to NTTOC associated with supplementing upper Yakima River spring chinook salmon. Interaction potential is the probability that supplemented spring chinook salmon will interact with the NTTOC. Interaction types include: competition (C), direct and indirect predation (P), behavioral anomalies (B), nutrient enhancement or mining (N), prey for piscivores (F), and pathogenic (D). Ecological risks are associated with the probability of failing to meet objectives listed in table 4. "Interactions" do not include potentials and risks associated with genetics or mixed stock fishing. Additional distributional and behavioral information is needed for many of the NTTOC to increase accuracy of predictions.

NTTOC	Interaction			
	Potential	Type	Risk	Uncertainty
Bull trout	Low	C,D,F	Low	Moderate
Westslope cutthroat trout	Low	C,D,F	Low	Moderate
Pacific Lamprey	Low	P,D,N	High	High
Marion Drain Fall chinook	Low	C,P,B,D	Low	Moderate
Upper Yakima steelhead	Moderate	C,P,B,D,N	High	Moderate
Mountain sucker	Low	C,P,D,N	Low	Low
Leopard dace	Low	C,P,D,N	Low	Low
Sand roller	Low	C,P,D	Low	Low
Resident rainbow trout (mainstem)	High	C,P,D,N,F	Moderate	Moderate
Naches steelhead	Low	C,P,B,D	Low	Low
Satus steelhead	Low	C,P,B,D	Low	Low
Toppenish steelhead	Low	C,P,B,D	Low	Low
Naches spring chinook salmon	Low	C,P,B,D	Low	Low
American River spring chinook salmon	Low	C,P,B,D	Low	Low
Mountain whitefish	Moderate	C,P,D,N	Low	Low
Resident rainbow trout (tributaries)	Moderate	C,P,D,N	Low	Low
Other native species	High	C,P,D,N	Low	Low

Part 2: Biotic limitations to project production objectives (Strong interactors)

Introduction

The purposes of Part 2 of this report are to develop objectives and identify those species of plants and animals (strong interactors) that may limit the production of upper Yakima spring chinook salmon. Objectives for strong interactors will be used to guide the development of ecological monitoring (Busack et al. 1997). Understanding and managing key ecological interactions between spring chinook salmon and strong interactors are critical to achieving production goals of the YFP. For instance, most hatchery populations experience lower survival than wild fish in natural environments. This difference in survival may be related to inferior predator avoidance, difficulty in procuring food, inferior competitive ability, superfluous activity, decreased disease resistance, inappropriate use of habitat, and other ecological interactions shortfalls (White et al. 1995). Certain species or guilds within an ecosystem may interact strongly with spring chinook salmon resulting in failure to achieve desired numerical goals for a supplemented population. Species or guilds, such as predators, competitors, or pathogens may negatively affect spring chinook salmon. Conversely, mutualists and prey may positively affect production of spring chinook salmon. Increases in the abundances of predators, competitors, or pathogens, (negative interactors) or decreases in the abundance of mutualists and prey (positive interactors) may limit the success of a supplementation program.

Unnatural increases in the impacts of predators, competitors, and pathogens to spring chinook salmon can occur from human induced changes in environmental conditions. For example, increased water temperature can increase digestion rates of predators which allows them to consume more salmon (Vigg and Burley 1991); can change dominance relationships among competitors (Reeves et al. 1987) resulting in competitive inferiority of cold water species such as chinook salmon; and can increase susceptibility and impact of chinook salmon to pathogens (Snieszko 1974; Bucke 1993). Thus human actions that increase water temperatures, such as riparian vegetation removal and water withdrawal, can potentially influence the interactions that occur between negative interactors and spring chinook salmon (Steedman 1991). Other manipulations to the system such as introduction of exotic species, construction of dams, channel simplification, and creating conditions favorable to predator species can unnaturally increase negative biotic interactions with spring chinook salmon (Li et al. 1987). Conversely, unnatural decreases in the effects of positive interactors (e.g., riparian vegetation which provides shading, hydraulic cover, cover from predators, food, and nutrient retention) may be caused by flow manipulation, logging, grazing, stream simplification, and beaver trapping (Li et al. 1987). Furthermore, these relationships indicate the need to examine abiotic conditions in the environment in concert with biotic monitoring (Busack et al. 1997). The focus of this report is entirely on biotic interactions, however, abiotic conditions will also be examined simultaneously in the field.

The goal of ecological limitations monitoring is to provide an indication of whether ecological interactions are limiting the success of the spring chinook salmon supplementation program. This information can then be used to understand the conditions under which supplementation may or may not succeed (i.e., application within and between other river systems) and to make appropriate management actions that will promote success of the YFP.

Methods

Taxa that may strongly affect the abundance of upper Yakima spring chinook salmon were identified using available interactions information that was collected in the Yakima basin, distributional information, information from research outside of the Yakima basin, and ecological theory. Taxa that have the potential to strongly affect, either positively or negatively, the abundance of spring chinook salmon were termed strong interactors. In general, strong interactors had to meet two criteria: 1) hypothesized or demonstrated ability to interact with spring chinook salmon, 2) sufficiently abundant to impact the spring chinook salmon population. Taxa were classified as predators, pathogens, competitors, mutualists, and prey. A taxon could be classified in more than one interaction category. Furthermore, the life-stages of spring chinook salmon that might be influenced by strong interactors were identified.

Objectives for strong interactors were framed in a way that would facilitate the success of spring chinook supplementation. For instance, negative interactions might be reduced and positive interactions conserved and enhanced. A distinction was made between unnaturally high negative interactions and natural interactions. The goal of the objectives is not to eliminate all negative interactions; only to eliminate negative interactions that are amplified by anthropogenic actions, such as introduction of exotic species or increasing water temperatures. In addition, confirmation or refutation of hypothesized interactions must be demonstrated before management actions are taken. This is necessary because resource managers have a long history of acting on hunches which later turned out to be fruitless or detrimental. For example, no small amount of effort was devoted to pulling wood from streams in an effort to help fish. During the past few decades, wood in streams has been shown to be extremely beneficial to fish. If appropriate ecological studies had been done prior to wood removal, then fish populations would have been the better for it. Finally, an understanding of the influences of strong ecological interactions will be beneficial for other managers in the Columbia basin.

Results

Objectives for strong interactors are of two types: **1) identify and reduce impacts of unnaturally high predatory, pathogenic, and competitory interactions that limit spring chinook salmon productivity, and 2) identify, conserve and enhance mutualist and prey taxa function at or above baseline levels to support maintenance or enhancement of**

ecosystem capacity for spring chinook salmon.

Thirty taxa of plants and animals were identified as strong interactors with upper Yakima spring chinook salmon (Table 1). Potentially all life stages could be impacted by strong interactors (Table 1). Brief justifications for selecting taxa as strong interactors are described below.

Table 1. Non-target taxa of concern (strong interactors) whose abundance may strongly influence the success of the spring chinook supplementation program.

Guild	Species	Life stage influenced
Predators	Northern pikeminnow	parr-smolt
	Rainbow trout/steelhead	fry-smolt
	Torrent sculpin	fry
	Mottled sculpin	fry
	Shorthead sculpin	fry
	Smallmouth bass	migr[a]-smolt
	Channel catfish	migr-smolt
	Gulls	migr-smolt
	Common Merganser	parr-smolt
	Great blue heron	parr-smolt
	Double-crested cormorant	migr-smolt
	Common Loon	smolt
	Western Grebe	smolt
	Tern	smolt
	Otter	smolt, adult
	Other[b]	
Pathogens	Viruses	smolt-adult
	Bacteria	smolt-adult
	Fungi	smolt-adult
	Parasites	smolt-adult
Competitors	Redside shiner	parr
	Rainbow trout/steelhead	parr
	Mountain whitefish	parr
Mutualists	Riparian vegetation (eg. cottonwood, willow, alder)	egg-adult
	Beaver	fry-parr, adult
Prey	Ephemeroptera	fry-smolt
	Plecoptera	fry-smolt
	Trichoptera	fry-smolt
	Diptera	fry-smolt
	Coleoptera	fry-smolt

[a] Fall or winter migrant
[b] Crappie, largemouth bass, bullheads, walleye, hooded merganser, belted kingfisher, common bittern, osprey

Predators

Various species of fish, birds, and mammals have been shown to be significant predators on salmonids (White 1939; Ricker 1941; Alexander 1979). Northern pikeminnow (Rieman et al. 1991; Poe et al. 1991; Ward et al. 1995) rainbow trout/steelhead (Hillman and Mullan 1989; Beauchamp 1995), sculpins (Ricker 1941; Hunter 1959; Patten 1975; Hillman 1989a; Berejikian 1995), smallmouth bass (Tabor et al. 1993), channel catfish (Vigg et al. 1991), gull (Ruggerone 1986), common merganser (Wood 1987a,b), great blue heron (Krebs 1974), double-crested cormorant (Modde et al. 1996), common loon, tern, western grebe (Modde et al. 1996), and river otter (Dolloff 1993; Reid et al. 1994) are known to prey substantially on salmonids. They are also quite abundant in the Yakima River basin (Patten et al. 1970; Pearsons et al. 1996; WDFW unpublished information). In addition, unnaturally high predation rates may occur in the Yakima basin because of high water temperatures and low water flows below Sunnyside Dam during spring migration, the presence of six dams and associated fish bypasses, and the presence of exotic fishes. Warm water temperatures in the lower Yakima River can increase gastric evacuation which can result in higher predation capacities (Beyer et al. 1988, Vigg and Burley 1991). In addition, low water flows resulting from irrigation withdrawals can increase smolt travel time and increase their susceptibility to predators. Dams and fish bypasses can concentrate and disorient spring chinook salmon during migration making them more susceptible to predators (Brown and Moyle 1981; Ruggerone 1986; Mesa et al. 1994; Ward et al. 1995). Furthermore, smallmouth bass and channel catfish, which were historically absent from the Columbia basin, are abundant in the lower Yakima River. Smolt survival data indicates that survival is poor below Sunnyside Dam presumably because of high predation pressure (Fast et al. 1991). Predation on the fry-parr life stages may also be accentuated because unnaturally high flows and simplified stream channels may disorient or concentrate spring chinook making them more vulnerable to bird and fish predators. Piscivorous birds in the Yakima basin appear to be increasing in abundance due to federal protection and reduced affects of pesticides such as DDT. Other predators, such as walleye, largemouth bass, bullheads, flathead catfish, crappie, bull trout, cutthroat trout, hooded merganser, belted kingfisher (White 1936), common bittern, osprey, and mink have been observed in the Yakima basin but because of there current low abundance or ability to consume large quantities of spring chinook salmon were not included as strong predators.

Competitors

Redside shiners are known to interact with spring chinook salmon in areas of sympatry and dominate spring chinook salmon at warm temperatures. Spring chinook salmon in the upper Yakima River were associated with redside shiners, particularly during the fall (Pearsons et al. 1996). In addition, agonistic interactions were observed between the two species (Pearsons et al. 1996). In laboratory experiments, redside shiners dominated steelhead at temperatures above 18^{o}C but were dominated at cooler temperatures (Reeves et al. 1987). Hillman (1989b) observed agonistic interactions between redside shiner and chinook salmon in the Wenatchee River. In that study, redside shiners displaced chinook salmon from favorable locations resulting in higher chinook salmon densities in allopatry than in sympatry.

Rainbow trout/steelhead are commonly associated with spring chinook salmon in the mainstem Yakima River, particularly in side-channels and during the late summer and early fall (Pearsons et al. 1996). Furthermore, when associated together they frequently interact sometimes resulting in displacement of spring chinook salmon. Larger rainbow trout dominate spring chinook salmon in behavioral contests (Pearsons et al. 1996).

Mountain whitefish are one of the most abundant fishes in the upper Yakima River (Pearsons et al. 1996). However, they are rarely observed near juvenile spring chinook salmon (Pearsons et al. 1996). Mountain whitefish may exploit food resources that spring chinook salmon utilize thereby restricting growth and survival of spring chinook salmon (Daily 1971). Comparisons of diets between the two species indicate that they eat a variety of aquatic invertebrates (Laakso 1951; Daily 1971; Healey 1991; Northcote and Ennis 1994).

Mutualists

Riparian vegetation is known to affect the densities of salmonids by providing cover from predators, instream structure, shading, and as a food chain base (Platts and Nelson 1989; Li et al. 1994; Tait et al. 1994). The abundance, type, and function of riparian vegetation has decreased dramatically throughout the Yakima basin as a result of resource uses such as logging, channelization, grazing, mining, water management, and housing development (Pearson 1985; Johnson 1994; Leland 1995). Furthermore the role of riparian vegetation as hydraulic, temperature, and predator cover has been reduced because of the way water is managed in the upper Yakima basin.

Beaver construct dams which can increase nutrient retention and habitat structure for rearing salmonids (Naiman et al. 1988). Positive relationships between beaver density and salmonid abundance have been observed in small tributary streams (Gard 1961). High densities of juvenile spring chinook salmon in the upper Yakima River have been observed in areas that were created by beavers (Pearsons, unpublished data, WDFW). Beavers appear to have been quite abundant in the Yakima basin historically but were trapped extensively prior to 1850 for their fur (Johnson and Chance 1974; Glauert and Kunz 1976).

Pathogens

Viruses, bacteria, fungi, and parasites can negatively affect the survival of spring chinook salmon (Bucke 1993). Similar to other salmonids, spring chinook salmon are particularly susceptible to pathogens when their immune systems are compromised (Wedemeyer 1970; Pickering and Duston 1983; Pottinger and Pickering 1992). This may occur when a fish is stressed or when a fish is undergoing physiological changes. In the Yakima basin, smolts can experience severe stress when they encounter poor water quality in the lower river. In addition, adults shunt energies away from the immune system making them very vulnerable to pathogens. Exotic pathogens such as those that cause whirling disease can have devastating impacts to rainbow trout populations and may impact anadromous salmonids.

Prey

The availability of prey can have a strong influence on the abundance and growth of spring chinook salmon. Reduction of prey can potentially increase intraspecific competition of spring chinook salmon. For example, the diet of chinook salmon rearing in streams is primarily larval and adult insects (Healey 1991). During the winter, juvenile chinook salmon consumed mainly Diptera, Trichoptera, and Plecoptera in the Fraser River basin (Levings and Lauzier 1991). Furthermore, the diet of chinook salmon during downstream migration in the Snake River was dominated by dipterans (Muir and Coley 1996). In addition, 26 to 38% of these fish had empty stomachs. Stomach fullness of spring chinook salmon collected in the Yakima River varied with location and time of year (B. Beckman, unpublished data, NMFS). Alteration of stream flows, such as occur in the Yakima basin, can significantly impact prey abundance. Furthermore, decreases in the availability of salmon carcasses can decrease the food base for spring chinook salmon in the Yakima basin. This could occur if carcasses from YFP broodstock are not released back into the Yakima basin in ways that simulate natural carcass distributions.

Influence of non-YFP hatchery releases

Hatchery releases of fish that are not part of the YFP may impact spring chinook salmon. For instance, release of coho salmon (additional to YFP releases) may decrease the survival of spring chinook salmon through competition, predation, or pathogenic interactions (WDFW 1995). Hatchery coho salmon that have been released in the Yakima River have the potential to consume chinook salmon that are up to 45.3% of their body length (chapter 11). Furthermore, chinook salmon that are too large to eat may be attacked and killed (chapter 11). Presently, hatchery coho salmon, fall chinook salmon, and small numbers of rainbow trout are introduced into the Yakima River and/or tributaries. The YIN and WDFW are attempting to balance the costs and benefits of hatchery releases outside of the YFP. For instance, the timing of coho salmon releases in the Naches and middle Yakima River have been managed to minimize impacts to the YFP.

Discussion

To accomplish the objectives that were described for strong interactors, monitoring and implementation of strategic management actions are essential. Five steps can be used to accomplish strong interactor objectives: first, interactions indices should be developed; second, interactions indices should be monitored; third, significant changes in indices should trigger more intense research to determine causation; four, causation should trigger strategic management actions to achieve objectives; and five, monitor effectiveness of strategic managment actions relative to objectives. This five step approach will allow scientists to focus monitoring and management effort most appropriately.

Interactions indices should be developed for the five types of interactions (e.g., competition, predation) and should indicate changes in strong interactions relative to baseline conditions. Prioritization of index development should reflect which interactions are hypothesized to be strongest. At current conditions, it is believed that predation is the strongest interaction that might limit project success (Table 2), in particular, predation on spring smolts and fall or winter migrants. Therefore, development of a predation index for migrating spring chinook should be initiated immediately. Other indices should be developed soon because the opportunity for collecting baseline data is diminishing. The intensity and scope of monitoring strong interactors is dependant upon the survival rates of spring chinook salmon. Low survival rates would prompt the need for a high monitoring effort. Adequate survival rates of spring chinook salmon could limit the need for monitoring substantially.

Some management actions can be instituted immediately which will help to achieve objectives. For instance, salmon carcasses from hatcheries can be naturally distributed into the Yakima basin, riparian vegetation can be protected and enhanced, and beavers can be protected or enhanced in suitable spring chinook salmon habitat. Furthermore, release strategies for non-YFP hatchery releases can be performed to minimize impacts to spring chinook salmon production and YFP monitoring plan. Other management actions are contingent upon results from interactions monitoring and evaluation such as; predator removal, dam modifications to decrease predation (Shively et al. 1996), or water temperature lowering.

Table 2. Preliminary ecological risk assessment for accomplishment of strong interactor objectives at current system conditions.

Interactor	Life-stage influenced	Risk
Predators	smolt	H
	fall/winter migrant	H
	fry	M
	parr	M
	adult	L
Prey	parr	M
	fry	M
	smolt	M
	fall/winter migrant	L
Mutualists	parr	M
	fry	M
	adult	M
	fall/winter migrant	L
	smolt	L
Competitors	parr	L-M
Pathogens	smolt	L-M
	adult	L

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Chapter 2

Feasibility of Monitoring Non-Target Taxa of Concern in the Yakima Basin

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Abstract

Highly valued non-target taxa of concern (NTTOC) have recently been identified and are proposed for monitoring as part of a spring chinook salmon (*Oncorhynchus tshawytscha*) supplementation project in the upper Yakima basin. Unfortunately, many NTTOC are relatively unstudied and methods for monitoring these taxa are not obvious. The purpose of this chapter is to investigate the feasibility of monitoring certain NTTOC. These include bull trout *(Salvelinus confluentus*), Pacific lamprey (*Lampetra tridentata*), mountain sucker (*Catostomus platyrhynchus*), sand roller (*Percopsis transmontana*), leopard dace (*Rhinichthys falcatus*), and mountain whitefish (*Prosopium williamsoni*). We sampled in areas where NTTOC were historically collected and other areas where we thought they might be found. Sampling in 1997 included backpack and drift boat electrofishing and snorkeling. We were successful in locating all species of interest except for sand roller. In addition, we collected Umatilla dace (*R. umatilla*; identification is tentative pending definitive genetic analysis), which have not been identified in the Yakima basin prior to this study. With the exception of mountain whitefish, all of the NTTOC appear to have very low population sizes. Despite considerable sampling effort, only 80 leopard dace (identification is tentative pending definitive genetic analysis), no sand roller, 29 mountain sucker, and 2 Pacific lamprey were observed during this study. A population of 10 bull trout was estimated in the North Fork of the Teanaway River and a population of 1,031 mountain whitefish was estimated in a 3.2 km reach above the Ellensburg town irrigation diversion. Preliminary results from our surveys indicate that North Fork Teanaway River bull trout, Pacific lamprey, and leopard dace are at extremely low abundances and are narrowly distributed, sand roller may be extirpated, mountain sucker are not abundant but widely distributed, and mountain whitefish are extremely abundant. Preliminary monitoring prescriptions are described for each of the NTTOC. We will be able to detect a 40% impact to mountain whitefish abundance or relative abundance in one or two years. However, our ability to detect acceptable impact levels with high statistical power (e.g., alpha =0.10 and beta=0.10 resulting in a power of 0.90) will be low for NTTOC that have low acceptable impact levels (e.g., 5%) related to spring chinook salmon supplementation (chapter 1) and have few years of baseline data. Therefore, we recommend that statistical power be adjusted to provide greater protection at the expense of increased probability of erroneously detecting an impact.

Introduction

The goal of YKFP supplementation is to increase natural production of a target taxa (e.g., spring chinook salmon, *Oncorhynchus tshawtscha*) while keeping impacts to non-target taxa within acceptable limits (BPA 1996). Non-target taxa of concern (NTTOC) have recently been identified in relation to supplementation of upper Yakima River spring chinook salmon (chapter 1). These NTTOC are: bull trout, westslope cutthroat trout (*O. clarki*), Pacific lamprey, Marion Drain fall chinook, upper Yakima steelhead (*O. mykiss*), mountain sucker, leopard dace, sand roller, resident rainbow trout in the mainstem Yakima River (*O. mykiss*), Naches steelhead, Satus steelhead, Toppenish steelhead, Naches spring chinook, American River spring chinook, mountain whitefish, and upper Yakima tributary rainbow trout. A monitoring plan is necessary to evaluate impacts to NTTOC relative to supplementation of spring chinook salmon. The current conceptual monitoring plan (Busack et al. 1997) calls for monitoring the distribution, density, and size structure of NTTOC. Before a monitoring plan can be implemented, basic information about the distribution, abundance, and methods for collecting NTTOC is necessary. Unfortunately, with the exception of rainbow trout and some other salmonids (chapters 3 and 4), little is known about other NTTOC in the Yakima basin (chapter 1). The NTTOC that are relatively unstudied are the subject of this chapter. These include bull trout, Pacific lamprey, mountain sucker, sand roller, leopard dace, and mountain whitefish. Westslope cutthroat trout monitoring will be treated in chapters 3 and 4.

Patten et al. (1970) sampled the mainstem Yakima River from the confluence with the Columbia River to its headwaters during 1957 and 1958. During those surveys, leopard dace were collected in 16 sites, mountain sucker in 20 sites, and sand roller in 3 sites. Lamprey were not identified to species in those surveys. Voucher specimens from these surveys were submitted to the University of Washington Fish Collection (UWFC). In addition, specimens of leopard dace were submitted to the UWFC by other collectors between 1930 and 1994. To our knowledge, no sand roller have been collected in the Yakima basin since 1958. Bull trout have been observed sporadically from 1990 till the present in the North Fork Teanaway River (Pearsons et al. 1996, chapter 3 and 4 of this report).

The purpose of this work is to investigate the feasibility of monitoring the density, distribution and size structure of bull trout, Pacific lamprey, mountain sucker, sand roller, leopard dace, and mountain whitefish. In addition, where data is available, comparisons to historical distributions will be performed. All data, analyses, and conclusions should be considered preliminary pending future work and verification of voucher specimens.

Methods

We sampled all locations within the study area where NTTOC of low abundance were known to have occurred (Figure 1). In addition, other sampling occurred in areas that we

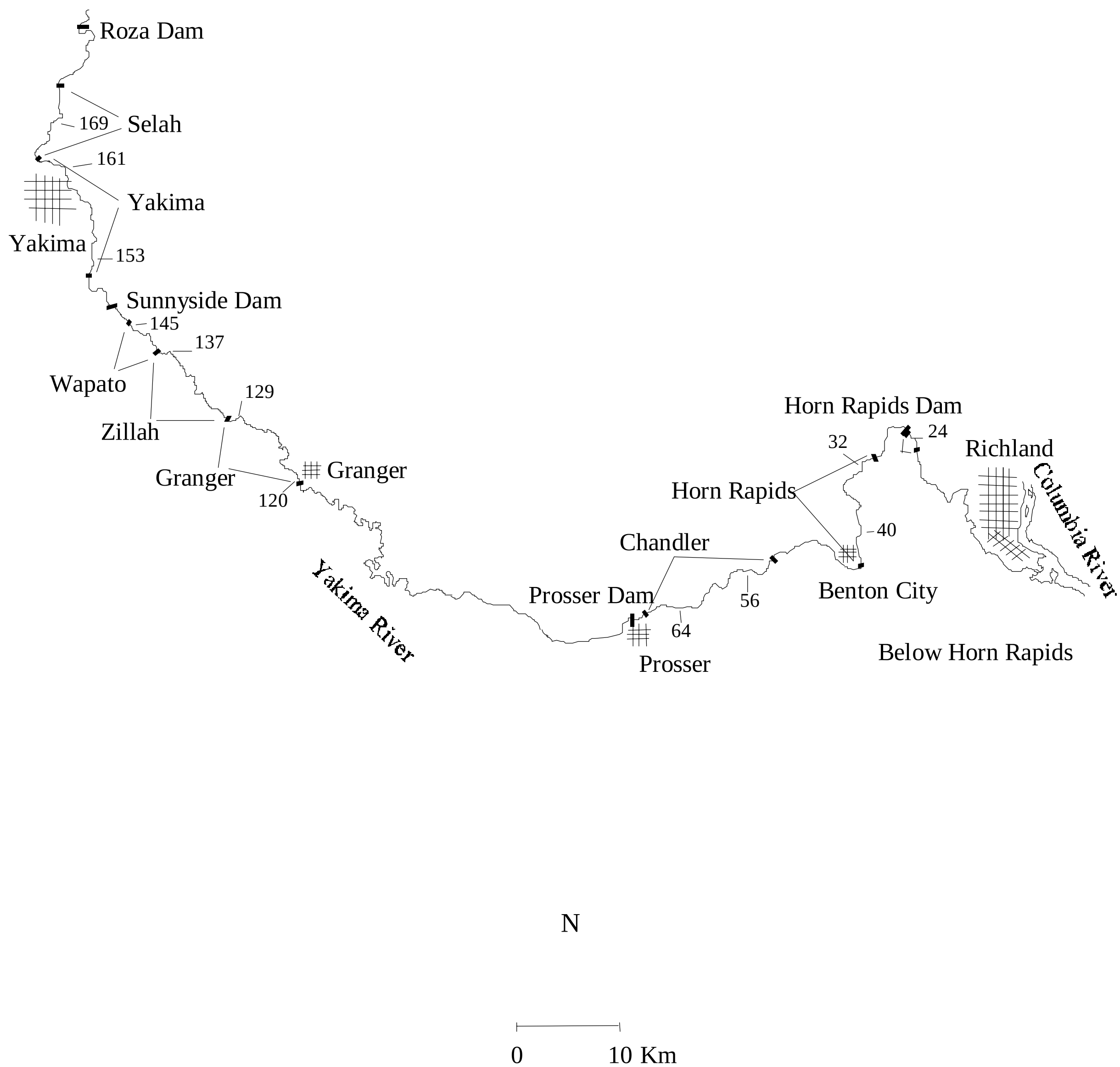

Figure 1. Map of the study area. River reaches that were sampled are identified and river kilometers used in the Patten et al. (1970) study are also displayed.

suspected might contain NTTOC. Our intent was to sample locations that historically contained NTTOC. In addition, we attempted to sample these sites with at least the effort that they had originally been sampled. We measured sampling effort by the total number of fish collected. In previous work we determined that the number of fish collected is a better indicator of sampling effort for determining the presence of rare species than sampling time or distance sampled. Fishes were sampled using electrofishing and snorkeling. Driftboat or backpack electrofishing units were used to collect NTTOC other than bull trout. Driftboat electrofishing occurred as follows: a rower would position the boat so the anode passed through areas representing the suite of habitats encountered, then a worker on the bow would net all fish smaller than 200 mm and visually identify fish over 200 mm. When the holding tank contained a sufficient number of fish, the fish would be identified, weighed and measured. Backpack electrofishing occurred as follows: two persons would drift downstream in an inflatable raft, when good habitat was encountered (both shallow enough to shock and representative of habitat types encountered e.g., side channels, sloughs, mainstem margins) the raft would be anchored and the site electrofished on foot until approximately 200 fish, or a minimum of 100 m was electroshocked. Electrofishing was conducted at sites throughout the reach that was floated.

Surveys were conducted during three time periods (e.g., spring (3/17-3/19), summer 1 (7/2-8/28), and summer 2 (9/2-9/29)) that corresponded to times that the Patten et al. (1970) surveys were conducted (3/3-3/21, 7/15-8/7, 9/18-10/18). During our summer 2 sample we concentrated our sampling effort on reaches and sites that contained leopard dace to increase our sampled size of individual fish. All individuals collected during backpack and drift boat electrofishing were identified and NTTOC were measured to the nearest mm fork length (FL) and weighed to the nearest gram. Voucher specimens of NTTOC were retained for later taxonomic analysis. Voucher specimens were placed in 95% ethyl alcohol so that DNA analysis could be conducted. Dace specimens are being analyzed by Gordon Haas and will be catalogued into the University of British Columbia’s Fish Collection.

We also collected NTTOC during other surveys that were not directed primarily toward NTTOC feasibility monitoring. These incidental collections of NTTOC were useful in describing the distribution of NTTOC. In particular, visual estimates of mountain whitefish were conducted during rainbow trout population estimates in five upper Yakima River index sites from 1993 to 1997 (Pearsons et al. 1996) and lower river sampling was supplemented during fish predator surveys described in chapter 12.

In order to determine the documented historical distribution of NTTOC, we searched published and unpublished literature, and the University of Washington’s Fish Collection (UWFC) for occurrences of NTTOC in the Yakima basin. We were suspicious of fish identified as leopard dace in some of the recent collections because we had sampled in similar areas and had not found any. In addition, speckled and leopard dace are difficult to differentiate. For these reasons, we had an expert on dace taxonomy (Gordon Haas, University of British Columbia) reexamine the historical collections. To minimize misidentification of dace during our surveys we developed a basin specific field key that was generated using an amalgamation of field keys (Bond 1973; Scott and Crossman 1973; Wydoskyi and Whitney 1979; Sigler and Sigler 1987; Peden and Hughes 1988) and previous and current experience with these fishes. Field keys were also developed for mountain sucker and Pacific lamprey.

The two characters that were primarily used to differentiate leopard and speckled dace during our study were pigmentation and caudal fin shape. Leopard dace have large spots or blotches that are generally larger in size than five individual scales, whereas speckled dace have dark pigment that generally covers a single scale. Leopard dace have a deeply forked caudal fin with lobes that are “pointy”, whereas speckled dace caudal fins are not deeply forked and the lobes are rounded. Other secondary characteristics were also used for distinguishing leopard and speckled dace including: the concavity of the dorsal and anal fin, the size of the barbels, the size of the pelvic fin stays, and the overall shape of the fish particularly as it related to the caudal peduncle (i.e., thin peduncle in leopard dace). Because of the difficulty in distinguishing leopard and Umatilla dace, our results should be considered preliminary until voucher specimens have been verified.

Suckers were distinguished using primarily mouth shape, and secondarily body shape, body pigmentation and scale size. Mountain suckers have an incomplete cleft of the lower lip, the posterior edge of the lower lip is squared off, and have deep notches at the corner of the mouth. They are generally slender and have small scales. Finally, they have five darkly pigmented vertical bars, three of which are easily visible. However, some specimens have been captured without any pigmentation.

Pacific lamprey were distinguished using caudal fin pigmentation and tooth configuration. Pacific lamprey have a thin line of black pigment that is located above and below the tip of the caudal fin. More specifically, the pigment is located at the base of the fin adjacent to the body. Myomere counts were also used to verify the accuracy of field identifications. If the lamprey specimen had eyes, the teeth were examined. Pacific lamprey have three sharp supraoral cusps.

The population size of mountain whitefish in a 3.2 km reach of the upper Yakima River was estimated using mark-recapture methodologies similar to those used to calculate rainbow trout abundance (chapter 3). An estimate was attempted during the fall of 1996, but too many whitefish were being injured so we discontinued the survey (i.e., close to spawning time). A second survey was conducted in March of 1997. Fish were marked on March 5 and 6 and recaptured on March 12 and 13. Visual estimates of all species electrofished, including mountain whitefish, were also done on these sample dates using methods described by Pearsons et al. (1996). In addition, visual estimates of mountain whitefish abundance and relative abundance were conducted during sampling for rainbow trout population estimates between 1993 and 1997 (Pearsons et al. 1996; chapter 3 of this report).

The abundance and distribution of bull trout was investigated in 1997 using snorkeling methods in the North Fork Teanaway (NFT) River basin. Snorkeling was used to avoid electrofishing injuries and to sample large areas in a relatively short time. Snorkeling surveys were conducted in 18 x 200-m long sites during the day and night. These sites were arranged systematically throughout the upper North Fork of the Teanaway River and were spaced 1 km apart. The lowest site was located 100 m below the downstream fence at 29 Pines Campground. The highest site was located 50 m above the parking lot for the trail into Mount Stuart. A population estimate was calculated by multiplying the number of bull trout observed by five (the number of 200 m sections within a 1 km section). Four other sites were added to increase sample area and to increase bull trout sample size. Three of these sites were located between the mouth of the North Fork of the Teanaway River and Dickey Creek Bridge (below 29 Pines Campground) and one 600-m long site was located between DeRoux Creek Campground and DeRoux Creek trail bridge. The latter site was chosen because bull trout were incidentally observed in this site. Two snorkelers looked for bull trout in the water column and under rocks, wood, and undercut stream banks. During night snorkeling, underwater dive lights were used.

The number and size of each bull trout was visually estimated and recorded on PVC dive cuffs.

Results

Most of the historical collections of purported leopard dace, including those from the Patten et al. (1970) study, were actually speckled dace (Table 1). Thus most of what we knew about the historical abundance and distribution of leopard dace in the Yakima basin is now in question.

Table 1. Voucher specimens of purported leopard dace catalogued in the University of Washington Fish Collection that were later reexamined[a].

Location	Collector	Date	Museum #	Identification
Yakima R. near Zillah	Royal	12/16/1930	UW001820	Not checked
Yakima R.	Patten	summer, 1957	UW014122	Speckled dace?
Yakima R.	Patten	summer, 1957	UW014386	Dried out
Yakima R.	Patten	1958-1959	UW014582	Not checked
Near Cle Elum	Patten	12/10/1962	UW020273	Speckled dace
Dry Creek	WADOE	5/18/1994	UW029566	Speckled dace
Dry Creek	WADOE	5/18/1994	UW029572	Speckled dace
Cooke Creek	WADOE	5/31/1994	UW029570	Speckled dace
Teanaway River	WADOE	6/14/1995	UW041234	Speckled dace

[a] Identifications were made by Gordon Haas and usually corroborated by J.D. McPhail, University of British Columbia.

Except for mountain whitefish, all of the NTTOC appear to have very low population sizes. Despite considerable sampling effort, only 80 leopard dace, no sand roller, 29 mountain sucker, and two Pacific lamprey were observed during this study (Table 2). Relative to historical surveys, the relative abundance (% of total fish collected) of leopard dace was lower in all of the surveys except the summer 2 survey which targeted leopard dace sites; mountain sucker was higher during all surveys, mountain whitefish higher in one survey, and sand roller lower during all surveys (Table 3). The number of fish that we collected during the spring, summer 1, and summer 2 surveys was higher than the Patten et al. (1970) surveys (Table 3), which suggests that we had a higher probability of capturing rare species. Mountain sucker (Table 4), Pacific lamprey and leopard dace were also collected during incidental surveys. One Pacific lamprey that was 90 mm long was collected on August 19, 1997 in the Yakima River near Richland. Leopard dace and Pacific lamprey were most abundant in the lower river below Chandler, mountain sucker in the middle to upper Yakima basin, and mountain whitefish in the upper Yakima basin (Tables 2 and 5).

Only two bull trout were observed in the 18 x 200-m sites that were sampled in the North Fork Teanaway River. Both of the fish that were seen were observed at night in site 16 (approximately 15 km above 29 Pines Campground). The population estimate for the North Fork Teanaway River was ten bull trout. In the site between DeRoux Creek Campground and trail bridge, five and seven bull trout were observed during two night surveys, and four bull trout were seen during the day. All of the bull trout that were observed during 1997 were large and appeared to be adults. The size of fish observed in the DeRoux Creek site were 225, 250, 300, 300, 350, 370, and 530 mm FL. These fish were considerably larger than all but two bull trout that have been observed in the North Fork Teanaway River since 1990 (Table 5). In fact, the size of bull trout observed in the NFT during 1997 are very similar to the size of bull trout collected in the mainstem Yakima River (Table 5).

Mountain whitefish were very abundant in the upper Yakima basin and widely distributed. In the upper Yakima River they accounted for 29 to 39% of the fish that we observed during our rainbow trout population estimates (Table 6). The population estimate of mountain whitefish in the THORPb section was 1,031 fish and the visual estimate was 118.5. The mean length of mountain whitefish that we captured during the population estimate was 264 mm FL, with a standard deviation of 52.4 mm.

Table 2. Numbers of NTTOC (LPD=leopard dace, MNS=mountain sucker, PAL=Pacific lamprey, MWF=mountain whitefish, SND=sand roller) found during present and historical studies (Patten et al. 1970). Only those historical surveys that have comparable data to present surveys are reported. No sandroller were collected during the present surveys.

Location	RKM	Present				Historical			
		LPD	MNS	PAL	MWF	LPD	MNS	MWF	SND
Below Horn Rapids	24	23	0	1	0	0	0	3	0
Horn Rapids	32,40	9	0	1	0	0	0	28	0
Chandler	56,64	46	0	0	0	1	0	8	0
Granger	120,129	1	2	0	2	26	0	38	0
Zillah	137	0	1	0	17	3	0	3	0
Wapato	145	0	14	0	114	16	0	31	7
Yakima	153,161	1	12	0	15	3	3	33	18
Selah	169	0	0	0	109	0	0	11	0
Total		80	29	2	257	49	3	155	25

Table 3. Relative abundance (% of total fish collected) of NTTOC (LPD=leopard dace, MNS=mountain sucker, PAL=Pacific lamprey, MWF=mountain whitefish, SND=sand roller) for present surveys and historical surveys (Patten et al. 1970).

	% of total fish collected					
Time	LPD	MNS	PAL	MWF	SND	TOTAL
			Present			
Spring	0.00	0.35	0.00	0.52	0.00	576
Summer 1	0.11	0.36	0.05	5.71	0.00	4466
Summer 2	6.43	0.94	0.00	0.00	0.00	1167
			Historical			
Spring	0.98	0.20	NA	2.95	0.98	459
Summer 1	0.81	0.00	NA	3.38	0.41	1230
Summer 2	0.97	0.04	NA	2.59	0.31	779

Table 4. Collections of mountain sucker made between 1994 and 1997 (excluding this study) by the ecological interactions team.

Location	Date	Number	Length	Weight	Survey-type
			Yakima River		
Milepost 19-20	9/10/97	1	274	264	Drift boat electrofishing
UCYN	9/21/94	1	239	154	Drift boat electrofishing
UCYN	9/24/97	1	juvenile		Drift boat electrofishing
UCYN	9/24/97	1	adult		Drift boat electrofishing
UCYN	9/25/97	1	juvenile		Drift boat electrofishing
EBURG	9/25/97	1	adult		Drift boat electrofishing
EBURG	9/30/97	1	218	153	Drift boat electrofishing
THORP	10/2/97	1	220	127	Drift boat electrofishing
			Tributaries		
Caribou Cr.	10/17/96	1	110	19	Backpack electrofishing
Caribou Cr.	10/17/96	1	106	17	Backpack electrofishing

Caribou Cr.	10/17/96	1	55	2	Backpack electrofishing
Caribou Cr.	10/17/96	15	juveniles		Backpack electrofishing
Caribou Cr.	10/17/96	2	adults		Backpack electrofishing
Caribou Cr.	11/4/96	1	89	4	Backpack electrofishing
Coleman Cr.	10/29/96	1	153	43	Backpack electrofishing
Coleman Cr.	10/29/96	1	188	75	Backpack electrofishing
Teanaway R.	8/94	3	adults		Backpack electrofishing

Table 5. Observations of bull trout made between 1990 to 1997 by the ecological interactions team. River kilometers (rkm) is the distance above Jungle Creek. "Up" or "down" refers to which direction a fish was moving when it was captured in a trap.

Location	Date	Number	Length	Weight	Survey-type
		North Fork Teanaway Basin			
NFT - 200 m upstream of NFT Bridge	5/31/91	1	149	NA	Fyke net - down
NFT - same as above	4/17/92	1	137	21	Screw trap
	4/27/92	1	141	28	Screw trap
	5/5/92	1	140	25	Screw trap
	5/30/92	1	151	33	Screw trap
	4/14/94	1	132	28	Screw trap
	4/30/94	1	130	21	Screw trap
	4/30/94	1	140	24	Screw trap
	5/11/94	1	146	29	Screw trap
	5/13/94	1	130	20	Screw trap
	5/18/94	1	137	23	Screw trap
	5/18/94	1	157	35	Screw trap
	5/21/94	1	137	24	Screw trap
	6/1/94	1	132	20	Screw trap
Jungle Creek	5/20/94	1	126	17	Panel weir trap - down
Jungle Creek	6/8/94	1	124	15	Panel weir trap - down
Jack Creek	6/6/95	1	122	15	Panel weir trap - down
NFT- 0.8 km up from Dickey Creek Br	7/5/94	1	133	21	Backpack electrofishing
NFT Pop. E -rkm 7.8	8/10/93	1	59	2	Backpack electrofishing
		1	58	2	Backpack electrofishing
		1	109	13	Backpack electrofishing

NFT Pop. 3 - rkm 8.2	9/26/90	1	113	12	Backpack electrofishing
		1	101	11	Backpack electrofishing
		1	146	30	Backpack electrofishing
		1	236	173	Backpack electrofishing
	8/6/92	1	59	2	Backpack electrofishing
		1	63	2	Backpack electrofishing
	8/3/93	1	120	16	Backpack electrofishing
		1	120	17	Backpack electrofishing
		1	125	17	Backpack electrofishing
		1	110	12	Backpack electrofishing
	7/20/94	1	101	5	Backpack electrofishing
		1	92	7	Backpack electrofishing
		1	84	6	Backpack electrofishing
	8/11/95	1	132	27	Backpack electrofishing
	7/31/97	1	258	213	Backpack electrofishing
NFT- rkm 10.88	7/12/94	4	100-200	NA	Snorkeling
NFT-rkm 11.52	7/12/94	1	180	NA	Snorkeling
NFT-rkm 12.16	7/12/94	1	100	NA	Snorkeling
		1	100	NA	Snorkeling
NFT-rkm 12.8	7/12/94	1	180	NA	Snorkeling
NFT - rkm 13.44	7/12/94	1	150	NA	Snorkeling
		1	125	NA	Snorkeling
NFT-rkm 13.5	9/9/97	1	225	NA	Snorkeling
		1	250	NA	Snorkeling
		1	300	NA	Snorkeling
		1	300	NA	Snorkeling
		1	350	NA	Snorkeling
		1	370	NA	Snorkeling
		1	530	NA	Snorkeling
NFT-rkm 14	9/26/96	1	330	NA	Snorkeling
NFT-rkm 14	10/7/96	1	330	NA	Snorkeling
NFT-rkm 14	10/15/96	2	NA	NA	Snorkeling
NFT-rkm 14	7/30/97	1	ca. 250	NA	Snorkeling
NFT-rkm 14	7/30/97	1	ca. 250	NA	Snorkeling

Yakima River

Above Benton City	6/19/97	1	278	173	Drift boat electrofishing
EBURG	6/16/92	1	340	415	Drift boat electrofishing
CELUM	10/21/92	1	474	1252	Drift boat electrofishing
CELUM	10/24/94	1	205	72	Drift boat electrofishing
CELUM	10/16/95	1	433	951	Drift boat electrofishing

Other Areas

Swauk Creek - 200 m above mouth	5/13/93	1	304	317	Panel weir trap - up
Naches at confluence with Yakima River	3/9/93	1	ca. 350	NA	Angling

Table 6. Visual estimates of the number and % composition of mountain whitefish (MWF) in five index sections of the upper Yakima River combined. "SPC" refers to age-0 spring chinook salmon.

Year	Total number	% MWF/total fish	% MWF/total fish without SPC
1993	5811.5	36.6	47.2
1994	7299.5	36.9	43.9
1995	5146	39.4	46.5
1996	2900	31.5	33.9
1997	3737	28.9	38.3
Mean	4978.8	34.7	42.0
C.V.	34.7	12.4	13.6

Discussion

Preliminary results from our surveys indicate that Pacific lamprey, leopard dace, and North Fork Teanaway bull trout are at extremely low abundances and are narrowly distributed, sand roller may be extirpated, mountain sucker are not abundant but widely distributed, and mountain whitefish are abundant. Other sampling in the Yakima basin supports our findings. For example, to our knowledge, sand roller have not been observed in the Yakima basin for over 30 years. Less than 15 adult Pacific lamprey have been observed passing Prosser or Roza dams, or have been collected at the Chandler Juvenile Fish Facility (CJFF) since 1992 (M. Johnston and J. Hubble, YIN, personal communication). Between 1993 and 1996, an annual average of 277 lamprey have been counted at the CJFF (M. Johnston, YIN, personal communication). An unknown proportion of these fish are Pacific lamprey since they have not been identified to species. Mountain whitefish are typically 30-40% of the total fishes observed during rainbow trout population estimates in the upper Yakima River and together with bridgelip and largescale suckers are the most numerous species that we observe during our rainbow trout estimates (Pearsons et al. 1996).

Despite considerable sampling effort, few bull trout have been observed in the North Fork of the Teanaway River. Only ten fish were estimated to be present in the North Fork of the Teanaway during 1997 and these fish were limited to a small area. Age-0 and juvenile fish were not observed during 1997 which indicates that very few fish spawned in 1996 and/or that mortality of young fish was high. Our data cannot eliminate either of these hypotheses. For example, only two bull trout redds were observed in the North Teanaway basin during 1996 (E. Anderson, WDFW, Personal communication). Both of these redds were observed in DeRoux Creek. In addition, large discharge events occurred in 1996 and 1997 which could have scoured redds and killed young bull trout. Potential recruitment of age-0 fish in 1998 does not appear to be very good because only 10 adult fish were estimated to be present in 1997 and no redds were observed in the North Fork Teanaway or DeRoux Creek in 1997 (Dave Burgess, personal observation).

Most of the bull trout observed in 1997 may have migrated into the North Fork Teanaway River from the Yakima River and may represent a fluvial life-history form. Unusually high discharges during 1997 may have improved migration success through the lower Teanaway River which typically has very low flows during the irrigation season. Prior to 1996, we had not observed a bull trout larger than 258 mm in the NFT basin, whereas five of the seven bull trout observed in the DeRoux Creek site were 300 mm or greater. Furthermore, the size of bull trout observed in the NFT during 1997 was similar to those captured in the Yakima River. In addition, small bull trout (e.g., 130-157 mm) were captured moving downstream in the NFT basin during the springs of 1991-1995 in traps. These fish may have been migrating to the Yakima River. It was previously thought that bull trout in the NFT basin exhibited only a resident life-history (WDFW 1997). These new data may complicate future monitoring and place tremendous significance on passage for juveniles and adults in the Teanaway River.

Due to the rarity of most NTTOC, monitoring can be extremely difficult. Preliminary recommendations for monitoring each species is as follows:

Mountain whitefish - visual estimates of abundance and age structure in five index sections of the Yakima River that are used for rainbow trout population estimates,

Leopard dace - population and size structure in six 400 m long index sites below Chandler during September (this might be accomplished by mark-recapture techniques or using catch per unit effort at times that experience similar environmental conditions from year to year),

Pacific lamprey - juvenile counts at Chandler and ladder adult counts at Prosser and Roza,

Sandroller - if no individuals are found during the next year they should be eliminated from the monitoring plan and possibly reintroduced,

Mountain sucker - collections during incidental surveys (due to their scarcity over a large area, electrofishing might cause more deleterious impacts than the spring chinook supplementation program - thus no specific risk containment monitoring is planned unless future sampling identifies aggregations of mountain sucker that are sympatric with spring chinook salmon)

Bull trout - snorkeling counts and estimates of size, during August and September at night, from rkm 10 above Jungle Creek up to impassable water falls, redd surveys (number and size) during October.

These recommendations will be tested during 1998 to determine their efficacy.

Preliminary power analyses suggest that it will take between one to two years to detect an impact of 40% to mountain whitefish abundance or relative abundance. This assumes that an appropriate model is used to explain much of the variation in abundance that might be caused by water discharge (e.g., minimum average monthly flow). Otherwise, if a model is not used, it would take about two to ten years to detect an impact of 40% to mountain whitefish abundance or relative abundance.

Our ability to detect acceptable impact levels with high statistical power (e.g., alpha =0.10 and beta=0.10 resulting in a power of 0.90) will be low for NTTOC that have low acceptable impact levels related to spring chinook salmon supplementation (chapter 1) and have few years of baseline data. These NTTOC include bull trout, Pacific lamprey, leopard dace, mountain sucker, and sand roller. Requiring that detection of impacts achieve a high statistical power may result in damage or extinction to NTTOC. In other words, an NTTOC may go extinct before we are able to detect a statistically robust violation of acceptable impact level. Therefore, we recommend that statistical power be relaxed when dealing with risk containment issues such as described above. For instance, we might want to reduce statistical power levels enough to detect acceptable impact levels after only one or two years. In short, the statistical power for NTTOC acceptable impact levels may be determined by the minimum time that we need to detect impacts in order to avoid irreparable damage to a population. Using this approach we sacrifice statistical confidence that the impact is real for risk containment. These concepts will be developed further in a future report.

It is difficult to determine how the status of NTTOC has changed from times past because so little historical information is available and what is available is either not detailed enough or unreliable. For instance, the Patten et al. (1970) surveys were the best source of historical information of NTTOC distribution and abundance. Unfortunately, upon closer examination, the study provides less information than we originally had hoped. For example, lamprey were not identified to species so the historical abundance and distribution of Pacific lamprey cannot be determined and leopard dace were frequently misidentified, making abundance and distribution information irretrievable. However, mountain sucker appear to be more abundant and sandroller less abundant than they were during the Patten et al. (1970) surveys in 1957 and 1958.

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Chapter 3

Rainbow Trout Population Abundance and Variation in the Upper Yakima River and Implications for Monitoring to Detect Impacts

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Abstract

The Yakima Fisheries Project (YFP) will attempt to increase natural production of spring chinook salmon *Oncorhynchus tshawytscha* through supplementation while limiting impacts to mainstem rainbow trout *Oncorhynchus mykiss* populations to less than or equal to 10% of baseline levels. Detection of impacts to fish population abundance is difficult because natural variation can be high even in the absence of an impact. If the YFP monitoring program is to detect an impact of supplementation activities on resident rainbow trout, sources of variation unrelated to supplementation must be resolved and taken into account. Since 1991, rainbow trout population abundance has been monitored in the upper Yakima River to establish abundance and biomass levels and their variation prior to supplementation of spring chinook salmon. Trout were collected by driftboat electrofishing and population estimates were calculated by maximum-likelihood mark-recapture methods. Results are reported for the sixth and seventh year of monitoring. In addition, estimates previously reported (1991-1994) have been revised due to new data-quality requirements and changes in analysis or presentation and are tabulated with the new results. Variation among years during baseline monitoring was greater (CV =14.6%) than the proposed acceptable impact level of 10 percent. Detection of a 10% impact will require that much of the temporal variation be accounted for by other sources. Predictive models were constructed to account for the effects of flows during critical periods in the life history of trout on the variation in estimates of abundance and biomass. Variation in river discharge explained the majority of temporal variation in rainbow trout abundance and biomass. Potential impacts to rainbow trout populations can be judged by comparing the population with its predicted state assuming no impact. The statistical power of the monitoring program to detect potential impacts of spring chinook supplementation was evaluated by comparing the magnitude of unexplained yearly variation in estimates of population density and biomass during the baseline period with the magnitude of acceptable impacts. The required number of years to detect an impact was computed for raw data and for data standardized to a predictive model. Predictive models reduced the number of years required to detect an impact from 24 or more for raw data down to a more practical 1 (density) to 3 (biomass) years. Much of the variation in rainbow trout abundance and biomass during the baseline period can be accounted for by variation in discharge, resulting in predictable estimates against which potential impacts of spring chinook supplementation can be judged.

Introduction

Among the most important goals of the Yakima Fisheries Project (YFP) are to increase natural production of spring chinook salmon *Oncorhynchus tshawytscha* and contain impacts to non-target taxa within acceptable levels. Rainbow trout *Oncorhynchus mykiss* in the upper Yakima River support a popular fishery, and have long been recognized as a taxa that should be monitored and protected relative to the supplementation of spring chinook salmon (Chapter 1). The conceptual monitoring plan of the YFP includes risk containment monitoring of mainstem Yakima River rainbow trout (Busack et al. 1997). The proposed acceptable impact level for mainstem rainbow trout is ≤ 10% decrease as a result of activities related to supplementation.

The commencement of operations at the Cle Elum hatchery signals the impending end of the baseline period. Once spring chinook salmon smolts are released into the Yakima River in 1999, the baseline period ends and the evaluation of potential impacts of supplementation on resident fish populations begins. Baseline data are now receiving additional scrutiny to prepare for use in evaluating potential impacts. Appropriate data-quality requirements have been implemented to assure that data reflect the status of the index sites without concern for errors in data entry or transcription. All new and revised (relative to that reported in 1990-1994) estimates reported in this chapter are based on data that have met the data-quality requirements.

Population monitoring methods and goals remain unchanged from previous years (McMichael et al. 1992; Pearsons et al. 1993; Martin et al. 1994; Pearsons et al. 1996). Estimates of rainbow trout density and biomass are reported for 1995 and 1996 sample years. Corrected estimates for 1991 through 1994 are reported to replace previously reported values following completion of new data quality requirements. Estimates for 1990 are not included due to methodological differences, but 1990 data are included in a plot of mean length. Variation among years for these estimates during the baseline period will be a primary determinant of the ability of the monitoring program to detect impacts. Whatever variation remains unexplained is the background level above which the impact must be detected.

The development of acceptable impact levels for non-target taxa of concern (chapter 1) provide a standard for evaluating the statistical power of the present monitoring efforts to detect impacts exceeding those levels. For a yearly monitoring program and a predetermined impact level, one key goal of power analysis is to determine how many years must pass before an average impact equal to the acceptable level is detected. Ideally, an impact of lesser magnitude than the acceptable level could be detected at the first sample analysis following the onset of impact to provide a margin of safety. In actual practice, natural sources of variation may overshadow the acceptable impact, requiring that additional variables be measured to account for non-impact sources of variation. We report estimates of sample sizes needed to detect impacts at the acceptable level, given the variation among raw estimates to date and for variation of estimates standardized to a predictive model of abundance or biomass.

The goals of this work were to 1) establish the level and variability of rainbow trout population abundance and biomass in the upper Yakima River 2) form models to explain temporal variation unrelated to potential impacts, and 3) to determine if sufficient statistical power is available to rapidly and reliably detect impacts exceeding acceptable levels established for the supplementation program. Results are preliminary pending further data collection and analyses.

Methods

Rainbow Trout Abundance and Biomass

From 1991 through 1996, trout populations were sampled in five sections of the Yakima River with mark-recapture methods (Ricker 1975) using a drift boat electrofishing unit. The river was divided into five geographical study sections as described in Hindman et al. (1991) and McMichael et al. (1992). An index site approximately 4-6 km long was sampled within each of five study sections. Index site characteristics are described in Table 1. Section locations are marked on the map in Figure 1. Each index site was floated after dark on two successive nights (one bank each night) to mark fish and repeated one week later to recapture marked fish. Methods are described in detail in McMichael et al. (1992) and Pearsons et al. (1993). Beginning with this report, estimates are computed using maximum likelihood methods that provide more accurate estimates than the modified Peterson method that was used previously. Beginning in 1996, work-up stations were standardized to 1 per kilometer and spring chinook redds were marked with painted rocks so the electroshocker could be turned off to minimize electrofishing injury. In this report, we have chosen to include only rainbow trout in the estimates, rather than all trout. Fish previously identified in the field as possible hybrid rainbow x cutthroat trout were changed to rainbow trout to reflect the results of genetic analyses on purported hybrids. In the future, we expect to estimate other species abundance and biomass relative to rainbow trout estimates by multiplying them by a species relative abundance and biomass in mark-recapture samples. Trout less than 80 millimeters are not included in estimates because capture efficiency is typically very low.

Fish density and biomass are reported as fish number and weight for the entire section as well as number and weight per kilometer to facilitate comparisons among sites and among years. Mean lengths are plotted to compare size variation among sites and years. Results are reported for the sixth and seventh year of monitoring. Estimates previously reported (1991-1994) have been revised due to new data-quality requirements and updated estimates computed with the maximum-likelihood method are provided in this chapter.

Table 1. Yakima River rainbow trout population monitoring section names and dimensions.

Section	Abbreviation	Length (km)
Lower Canyon	LCYN	4.5
Upper Canyon	UCYN	4.5
Ellensburg	EBURG	4.0
Thorp	THORP	5.8
Cle Elum	CELUM	6.3

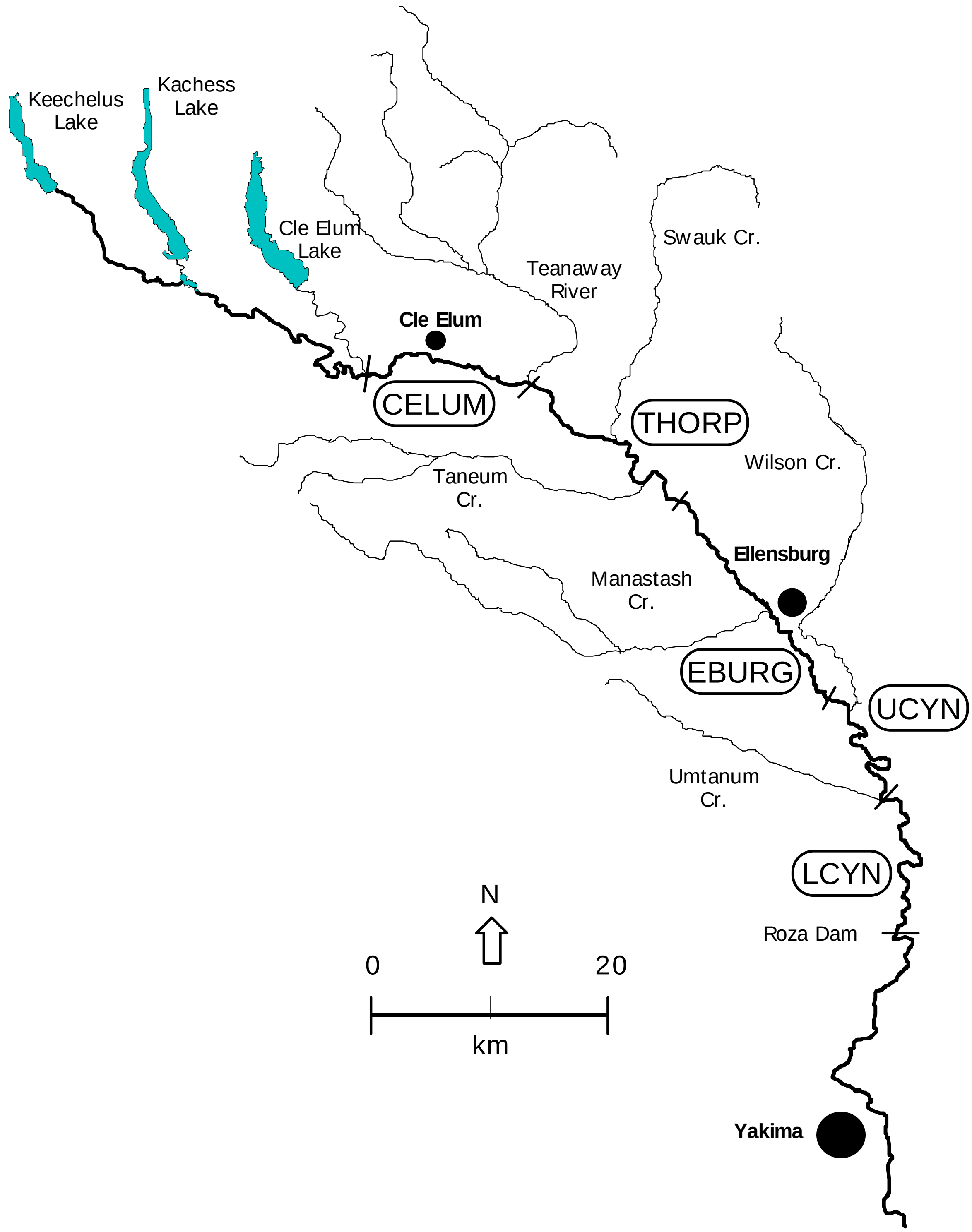

Figure 1. Locations of Upper Yakima River index sections. Study area includes the Yakima River between Roza Dam and the Cle Elum River. Index sections are separated by heavy lines and are labeled with their abbreviation.

Predictive Models of Abundance and Biomass

A regression model of population abundance with respect to river discharge was formed to explain temporal variation in rainbow trout abundance estimates. Several potential indices of flow magnitude and variation were constructed for critical periods in the life history of trout. One type of index reflected the variation in discharge from February through October. This index is intended to capture the effects of high spring flows that might influence the estimate through changes in distribution or survival. Another index reflected the discharge in May relative to April. This index is intended to capture the effects of changes in flow on survival in the redds, affecting year class strength in later sample years when those fish reach a size that can be effectively sampled by electrofishing. In addition, yearly minimums and maximums were used as indices to reflect the influences of extreme flows on available habitat for a range of life stages. Each index was computed for the sample year and for the three years previous to the sample year, encompassing the life span of the most abundant age groups sampled (Martin and Pearsons 1994). Models were constructed by choosing two indices by stepwise regression. One regression model predicts population abundance on the basis of flow indices. Another regression model predicts biomass from flow indices.

Implications for Monitoring to Detect Impacts

Yearly variation in population abundance data during the baseline period was evaluated relative to acceptable impact levels detailed in chapter 1. We examined whether detection of the maximum acceptable impact (10%) was feasible and, if so, how rapidly detection could be achieved. Setting $\alpha = 0.10$ and $\beta = 0.10$ to control power to 0.90, we computed the number of years it would take to detect an impact of 10% both for raw estimates and for estimates standardized to the predictive model. The formula for sample size at a given power for a two-sample comparison (Zar 1996, equation 8.22, page 133, for one-tailed test per page 134) is:

$$n \geq 2\left(\frac{\sigma}{\delta}\right)^2 \left\{ t_{\alpha[v]} + t_{(1-P)[v]} \right\}^2$$

Where:
n = number of years to detect an impact
σ = standard deviation
δ = acceptable impact level
α = significance level
v = degrees of freedom, df = $2(n - 1)$
P = power to detect an impact equal to δ. $P = (1 - \beta)$
$t_{\alpha[v]}$ = value from a one-tailed t table with probability α and v df
$t_{(1-P)[v]}$ = value from a one-tailed t table with probability 1- P (=β) and v df.

The values of σ and δ must be expressed in the same units (percentage of the mean are most convenient for our purposes) so that units cancel out in the formula above. The formula is solved for n iteratively by starting with an estimated n. After each iteration, the formula is recalculated with the previous solution as the new estimate until the estimate of n stabilizes.

Results

Rainbow Trout Abundance and Biomass

Rainbow trout population abundance estimates within individual index sections differed widely among years in the Yakima River (Table 2). We chose to compare sites and years using the estimates on a lineal (per kilometer) basis, to standardize for site length while avoiding variation among years that would be the case with estimates on an areal basis. Two years stand out as having a relatively high mean number of fish per kilometer overall, 1991 and 1993. The very high density estimate for the Cle Elum section in 1991 explains the high overall estimate in 1991, but in 1993, most of the sections exhibited relatively high abundance. In other years, mean estimates for all sites combined differed much less among years. Individual sections were much more variable among years, but a pattern of interactions among sections is not obvious.

Lineal biomass estimates also differed widely among years, and a cyclic pattern was evident in the mean biomass estimates (Table 3). Odd-numbered years have relatively high biomass estimates while even-numbered years have relatively low biomass. A pattern of interactions among sections is not obvious.

Mean length differed little among sites or among years. Variation in THORP and EBURG sections was similar among years (Figure 2). Other sections varied among years in less structured ways, but all sections exhibit a low mean length for the 1996 sample year, relative to previous years (Figure 2).

Table 2. Rainbow trout density in index sections of the upper Yakima River from 1991 through 1996. Sections are listed from lowest to highest elevation.

Section	Units	1991	1992	1993	1994	1995	1996
LCYN	no.	1371	1652	1098	835	693	1497
	no./km	305	367	244	186	154	333
UCYN	no.	1222	1183	1504	1549	1524	735
	no./km	272	263	334	344	339	163
EBURG	no.	1212	865	1593	1320	1160	1071
	no./km	303	216	398	330	290	268
THORP	no.	1316	870	1384	1484	988	955
	no./km	227	150	239	256	170	165
CELUM	no.	5371	1962	2335	2005	3156	3521
	no./km	853	311	371	318	501	559
Mean	no.	2098	1306	1583	1439	1504	1556
	no./km	392	262	317	287	291	297

Table 3. Rainbow trout biomass in index sections of the upper Yakima River from 1991 through 1996. Sections are listed from lowest to highest elevation.

Section	Units	1991	1992	1993	1994	1995	1996
LCYN	kg	347	468	362	197	150	227
	kg/km	77	104	81	44	33	50
UCYN	kg	208	230	307	295	271	137
	kg/km	46	51	68	66	60	30
EBURG	kg	187	124	250	219	242	125
	kg/km	47	31	63	55	60	31
THORP	kg	224	104	252	173	160	112
	kg/km	39	18	44	30	28	19
CELUM	kg	696	351	502	346	866	550
	kg/km	110	56	80	55	137	87
Mean	kg	332	256	335	246	338	230
	kg/km	64	52	67	50	64	44

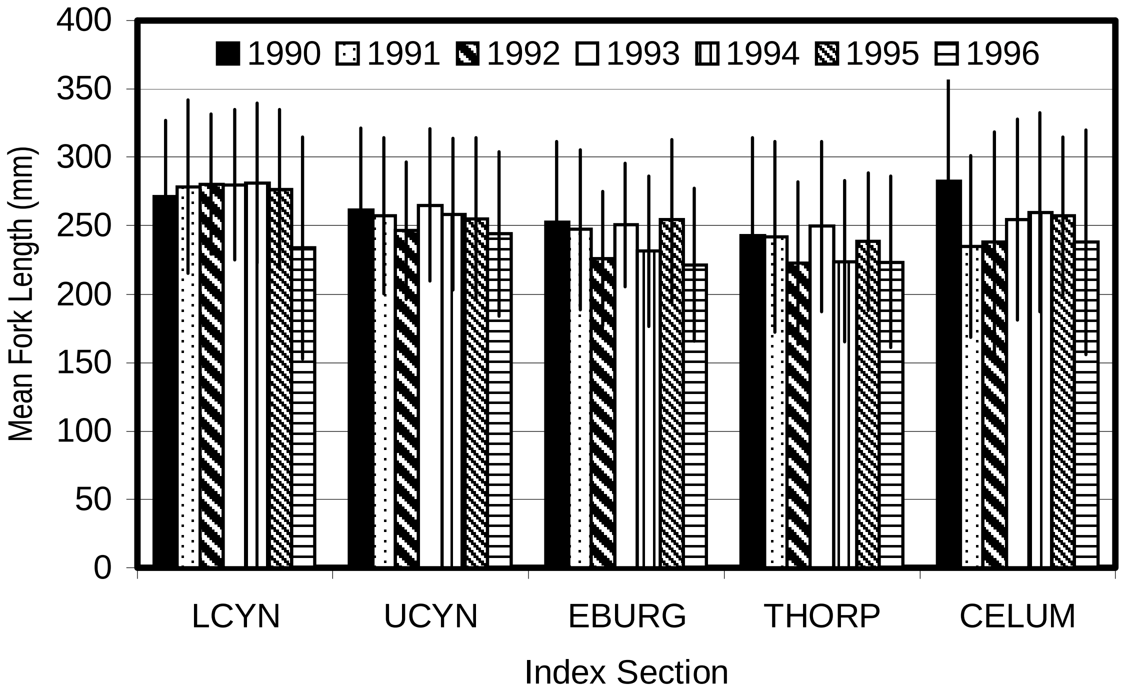

Figure 2. Mean fork length of rainbow trout in index sections of the upper Yakima River from 1990 through 1996. Error bars are means ± 1 standard deviation.

Predictive Models of Abundance and Biomass

The predictive models were linear regression models of the form:

$$\text{Predicted Estimate} = \text{Intercept} + B_1(\text{index}_1) + B_2(\text{index}_2)$$

Where B_1 and B_2 are regression coefficients and index_1 and index_2 discharge indices. For abundance the most important index was the yearly minimum of the minimum monthly flows lagged 3 years. The other index was the relative difference in minimum flows in April versus May during the water year prior to the estimate (not lagged). The predictive model for biomass included an index of the variability of maximum flow from October through February lagged 1 year and an index of the yearly maximum of the minimum monthly flows lagged 3 years. We speculate that these time periods are important in the growth of fish, though for now these are empirical, not mechanistic, models. River discharge explained 83% of the temporal variation in baseline rainbow trout population density and 72% of the temporal variation in biomass estimates. Estimates standardized by dividing them by the model prediction were therefore much less variable than raw estimates.

Implications for Monitoring to Detect Impacts

The ability to detect impacts of a given magnitude decreases with increasing unexplained variation in baseline data. Table 4 compares the acceptable impact levels with coefficients of variation and shows the resulting number of years of sampling required to detect an impact at that level. Raw estimates are variable among years, relative to acceptable impact levels, and that results in low power to detect impacts, requiring a large sample size for detection. Estimates standardized to the predictive models are much less variable among years, providing greater power and requiring much lower sample sizes to achieve detection of potential impacts at or exceeding the acceptable level.

Table 4. Sample sizes required to detect impacts at the acceptable impact level for a given significance level and power for raw estimates of density and biomass compared to the same estimates standardized to a predictive model.

Measurement	Type	Acceptable Impact (%)	α	Power ($1-\beta$)	Temporal Variation CV (%)	Years to Detect
Density (no./km)	Raw Estimates	≤ 10	0.10	0.90	14.62	29
	Predictive Model	≤ 10	0.10	0.90	2.43	1
Biomass (kg/km)	Raw Estimates	≤ 10	0.10	0.90	16.65	37
	Predictive Model	≤ 10	0.10	0.90	4.63	3

Discussion

The low acceptable impact levels for resident rainbow trout abundance and biomass in the upper Yakima River, relative to variation among years in raw estimates, makes detection of impacts challenging. In raw form, estimates of abundance and biomass are too variable to allow for detection of impacts at or below the acceptable level within a reasonable span of years. By modeling abundance and biomass with respect to discharge data, it is possible to account for much of the temporal variation in estimates of abundance or biomass, greatly improving the power of the monitoring program to detect potential impacts. If these predictive models can be validated and refined with 1997 and 1998 data, detection of potential effects of spring chinook salmon supplementation should be rapid and powerful.

Standardizing data to the predictive model removes much of the temporal variation. The remaining variance in the standardized estimates is a combination of lack of fit of the predictive model and pure error. Together these sources compose the unexplained error, which has been reduced by the process of fitting the predictive model. This reduction of unexplained variation is the goal, but it should be made clear how the use of the models changes how acceptable impact levels are evaluated. Without the model, a 10% impact could be stated to be detected if an estimate is less than or equal to 90% of the baseline mean. Unfortunately, the baseline variation is high enough that even some of the yearly estimates during the baseline period would fall below this cutoff. By standardizing estimates to the predictive model, a 10% impact would be detected if an estimate fell more than 10% below the predicted value. The variability around the predictive model is much lower, and the test is powerful, but the comparison is less straightforward. For example, an estimate that would be 10% or more below the baseline mean might not be judged to be evidence of a 10% or greater impact if the predicted value for that estimate is also low. The model will predict an acceptable range for a year on the basis of discharge conditions, rather than a single range for any and all years that does not account for the influence of natural phenomena. As we have stated previously, we wish to detect potential impacts of chinook salmon supplementation greater than an acceptable level. In addition to being statistically detected, impacts must also be related to spring chinook salmon supplementation through other lines of evidence that help establish the most likely cause and effect relationships.

Few monitoring programs have the advantage of such a large baseline data set. Because estimates of population abundance and biomass have been made over a number of years including a range of discharge conditions, there is data available to construct models. Raw estimates for individual index sections are highly variable among years. Raw estimates for the combined sections are much less variable among years and provide a much better basis for monitoring for potential impacts. Averaging across sites is consistent with the acceptable impact level for mainstem rainbow trout which are based on the upper Yakima River population as a whole. Whatever temporal variation in population abundance and biomass estimates can be accounted for by discharge reduces the unexplained variation and increases the power of statistical tests. Although the discharge indices included in the predictive models are intended to represent potential mechanistic pathways by which discharge variation might affect fish abundance or biomass, the model should be considered largely empirical at this time. We intend to refine the predictive models to be as mechanistic as practical, as time permits. We feel that pursuing a mechanistic model will increase our understanding of rainbow trout population dynamics in addition to providing robust predictions across a broad range of inputs. The few remaining years of baseline data will be used to test and refine the model and progress toward mechanistic understanding of the factors controlling abundance and biomass in the absence of impacts.

Acknowledgements

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Chapter 4

Salmonid Distribution and Population Abundance Variation in the Upper Yakima River Tributaries and Implications for Monitoring to Detect Impacts

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Abstract

The Yakima Fisheries Project (YFP) will attempt to increase natural production of spring chinook salmon *Oncorhynchus tshawytscha* in the upper Yakima River through supplementation while limiting impacts to natural salmonid populations. Tributaries to the upper Yakima River support populations of several salmonids of concern including rainbow trout *Oncorhynchus mykiss*, cutthroat trout *Oncorhynchus clarki*, and bull trout *Salvelinus confluentus*. One of the acclimation sites for spring chinook salmon smolts is located along a tributary, and comparisons with other tributaries that differ in abundance of spring chinook salmon will provide evidence of potential impacts of outmigrating smolts or residuals. Since 1990, salmonid distribution and rainbow trout population abundance have been monitored in tributaries to the upper Yakima River to establish baseline conditions prior to supplementation of spring chinook salmon. Multiple-removal sampling methods were used to estimate abundance and distribution of salmonids. Results are reported for the fifth, sixth, and seventh year of baseline monitoring. In addition, estimates for previous years (1990-1994) that have been revised due to more stringent data-quality requirements are tabulated. The number of index sites was expanded in 1997 to focus more effort on cutthroat trout status, and site length was doubled to increase the accuracy of estimates. Temporal variation in estimates of salmonid density or biomass was high. Predictive models of population density and biomass with respect to stream discharge were formed to explain a portion of the temporal variation for rainbow and cutthroat trout. The statistical power of the monitoring program to detect potential impacts of spring chinook salmon supplementation was evaluated by comparing the magnitudes of yearly variation in estimates of population density, biomass, and mean fork length during the baseline period with the acceptable impact levels. Temporal variation in raw estimates of rainbow trout density or biomass during baseline monitoring makes detection of the maximum acceptable impact level of 40% highly unlikely within less than 9 (density) or 5 (biomass) years. Standardizing estimates to model predictions accounted for much of the temporal variation and detection of a 40% impact in 1 year is likely. Baseline monitoring to date has concentrated on rainbow trout and has not provided sufficient statistical power to rapidly or reliably detect impacts to density or biomass of cutthroat trout or bull trout, but monitoring is sufficient to reveal important changes in distribution. Expansion of index site reaches will increase the accuracy of future estimates and will improve the ability to detect impacts to less abundant species. Without an impractical increase in effort, the monitoring program is unlikely to achieve rapid and reliable detection of smaller impacts that more closely approximate the desired level of no impact. Predictive models for cutthroat trout density and biomass with respect to discharge have also proven useful for explaining temporal variation, but statistical power is still insufficient for detecting small impacts of supplementation activities. Bull trout cannot be efficiently monitored within the present tributary salmonid monitoring program. The monitoring program has sufficient power to detect impacts to rainbow trout abundance and to cutthroat trout or spring chinook salmon distribution. Refinement of the sampling design will increase power available to detect impacts to cutthroat abundance.

Introduction

Knowledge of salmonid distribution and abundance in the tributaries of the upper Yakima River is vital to the accomplishment of the Yakima Fisheries Project (YFP) goals of increasing natural production of spring chinook salmon, *Oncorhynchus tshawytscha*, and containing impacts to non-target taxa of concern (NTTOC) within acceptable levels (Busack et al. 1997; chapter 1, of this report). First, tributaries support populations of non-target species that have been recognized as taxa that should be monitored so that appropriate management actions can be taken during the supplementation phase. Second, an acclimation site to distribute spring chinook salmon smolts will be located on the North Fork Teanaway River (NFT), making this stream an important area for evaluating supplementation success and potential interactions among outmigrating stocked chinook salmon smolts and NTTOC. Third, supplementation may result in greater numbers of juvenile salmonids moving into tributaries, increasing the potential for interactions of spring chinook salmon with NTTOC. The monitoring design for tributaries simultaneously addresses the varied aspects of the YFP by sampling index sites that address both basin-wide and tributary-specific objectives and overlap the distributions of many species. Though many important non-salmonid species are captured in tributary multiple-removal passes, this chapter focuses on salmonids of concern. Non-salmonid taxa of concern are discussed in chapters 1 and 2.

The level and variation of salmonid population estimates in tributaries are being determined in baseline monitoring that was begun in 1990. When spring chinook salmon smolts are released from YFP facilities in 1999, monitoring for impacts will commence. Baseline data are being scrutinized to optimize our ability to detect potential impacts of YFP activities to populations of salmonids of concern in tributaries. New data-quality requirements have been implemented to assure that estimates reflect the status of populations at the index sites without concern for errors in data entry or transcription. All new and revised estimates reported in this chapter are based on data that have met the new data-quality requirements.

In combination with baseline estimates, the recently proposed acceptable impact levels for NTTOC (chapter 1) provide a basis for evaluating the statistical power of the present sampling design to detect impacts exceeding those levels. For an annual monitoring program and a predetermined impact level, one way to approach the question is to determine how many years must pass before a constant impact equal to the acceptable level is detected. Ideally, an impact below the acceptable level by some margin of safety could be detected at the first sampling following the onset of impact. In actual practice, the variability of population estimates can make detection of impacts challenging. We report estimates of sample sizes needed to detect impacts to rainbow trout *Oncorhynchus mykiss*, westslope cutthroat trout *Oncorhynchus clarki*, or bull trout *Salvelinus confluentus* at or greater than the acceptable level for each species, given the variation among raw population estimates to date. Spring chinook salmon are also detected in index sites, but the results will be used to evaluate supplementation success and for evaluating impact potential or mechanisms. Brook trout *Salvelinus fontinalis* and other salmonids are also infrequently encountered in tributary monitoring but, at present, those species are not of critical concern to the YFP.

Predictive models of population density (number of fish/km) or biomass(kg/km) for rainbow or cutthroat trout with respect to stream discharge were created to account for some of the temporal variation in estimates. We report sample sizes needed to detect impacts to rainbow or cutthroat trout using estimates standardized to predictive models to account for non-impact sources of temporal variation. No predictive models were formed for bull trout, because estimates are less reliable for species of very low abundance in index sites.

The goals of this work were to 1) determine the level and variability of salmonid population abundance and distribution in tributaries to the upper Yakima River, 2) form models to explain temporal variation unrelated to potential impacts, and 3) determine if sufficient statistical power is available to detect impacts exceeding acceptable levels for salmonid NTTOC with respect to the spring chinook salmon supplementation program. Results should be considered preliminary pending further data collection and analysis.

Methods

Tributary Salmonid Population Estimates

100-m sections: 1990 – 1997

From 1990 through 1997, removal-depletion methods (Zippin 1958; McMichael et al. 1992; Pearsons et al. 1993; Martin et al. 1994; Pearsons et al. 1996) were used to estimate density and biomass of rearing salmonids in several tributaries of the upper Yakima River to evaluate their spatial and temporal variation prior to supplementation activities. Estimates were performed in 100-m index sites to evaluate salmonid abundance and distribution (McMichael et al. 1992; Pearsons et al. 1993; Martin et al. 1994; Pearsons et al. 1996). The number of tributaries sampled increased from five in 1990 to 10 in 1992 and to 12 in 1997. Overall, 35 index sites on 14 tributaries have been sampled (Table 1 and Figure 1). In 1997, Site 1 in Swauk creek was sampled in early December instead of the planned sampling time in early August because permission to access the property could not be obtained until after the normal sampling season. Three sites (MAN1, MAN2, and CAB2) were dropped because their past performance suggested they would contribute little to monitoring for impacts. Estimates exclude individual salmonids less than 80mm fork length, due to typically poor efficiency for collecting small individuals by electrofishing.

Salmonid species of concern found in tributary index sites include rainbow trout, cutthroat trout, and bull trout. In the field we had identified purported rainbow trout x cutthroat hybrids on the basis of external characteristics, but genetic analysis indicated that few, if any, were actually hybrids. Hybrids have therefore been changed to rainbow trout or cutthroat trout, whichever was more abundant at the site of capture. Spring chinook salmon were also monitored, not to detect impacts of supplementation, but to evaluate supplementation success. The monitoring program was originally intended to evaluate rainbow trout abundance and distribution, but only the distribution of other salmonids. In spite of the limited effort expended toward cutthroat trout and bull trout, we also evaluate the ability to monitor abundance of these salmonids of concern.

Table 1. Table of abbreviations for tributaries.

Tributary	Abbreviation	Sites
Big Creek	BIG	1
Cabin Creek	CAB	1,2
Domerie Creek	DOM	A,
Jungle Creek	JUN	1
Main Stem Teanaway River	MST	1,2,3
Manastash Creek	MAN	1,2,3
Middle Fork Teanaway River	MFT	1,2,3
North Fork Teanaway River	NFT	1,2,3,A
Stafford Creek	STF	A,B
Swauk Creek	SWK	1,2,3
Taneum Creek	TAN	1,2,3,A,B
Umtanum Creek	UMT	1,1.5,2
West Fork Teanaway River	WFT	1,2,3
Wilson Creek	WIL	A

Estimates of density for individual species were computed by maximum likelihood estimation using the Microfish program (Van Deventer and Platts 1983, 1985). Where removal patterns fail to meet the assumptions of the method for an individual species, the program substitutes a maximum limit (1.5 x total number of fish caught) if the pattern is non-descending, or substitutes a minimum limit (total number of fish caught) if all fish were caught on the first pass or if less than 2 fish were caught in all passes.

Estimates are reported on a per kilometer basis to facilitate comparisons among different sections and among tributaries. We sought to increase the accuracy of the estimates and decrease variability by data proofing, consistent scheduling of sampling across years, and, beginning in 1997, adding an additional 100-m sample reach at each site.

200-m index sites: 1997

A study conducted during the 1996 sampling season to evaluate whether 100-m index sites provide data that is sufficiently representative of the larger stream sections indicated that doubling index site length would provide increased accuracy. Single-pass electrofishing collections were performed in 400-m reaches including the standard 100-m index sites at MAN1, MFT1, NFT1, SWK1, and UMT1. The first pass of the multiple removal sampling in the index site served as the single pass for the first 100-m reach, the next three 100-m reaches were sampled in a single pass without block nets, but otherwise by the same methods as for multiple removal sampling. Improvements in stability with length were compared with marginal effort to arrive at 200-m as the optimum for our goals.

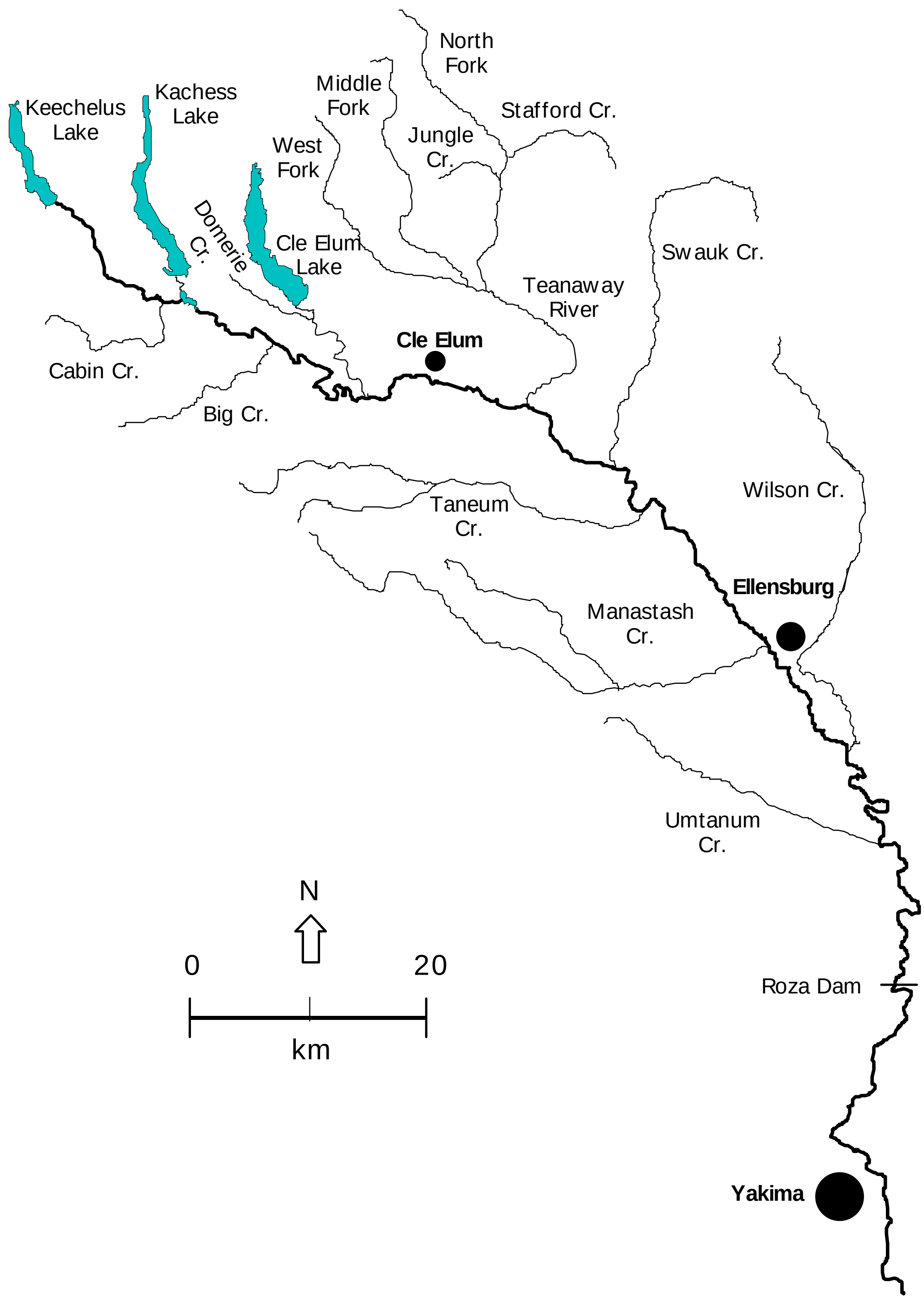

Figure 1. Select tributaries of the upper Yakima River. One or more index sites is located on each tributary named above.

In 1997 sampling, the original 100-m index site was maintained but an additional 100-m reach was located contiguous to the original, where possible, for a total of 200 meters. Each 100-m reach was sampled to provide an independent multiple removal pattern. This separate sampling and analysis preserves and continues the original monitoring data set while improving accuracy from 1997 forward. Estimates are expressed on a lineal basis to facilitate comparisons across years. Where the number of passes was equal for both 100-m portions, a combined multiple-removal estimate was computed. If the number of passes was unequal the section estimates were added or averaged, as needed, to provide a combined estimate.

The tributary monitoring program for salmonids of concern was also being refined and refocused in 1997 to reflect the latest monitoring goals and temporal variation in past sampling. To increase our ability to monitor abundance of westslope cutthroat trout, seven new sites were added in 1997 in locations where cutthroat trout abundance would reliably permit removal estimates for comparison across years. Sites were placed where proportions of cutthroat and rainbow trout would be roughly equivalent to monitor the advance or retreat of the distribution of either species in relation to the other. The sites added were DOMA, NFTA, STFA, STFB, TANA, TANB and WILA. Additional sites targeted at cutthroat trout will be added in 1998.

Predictive Models of Density and Biomass

A regression model of population density with respect to tributary stream discharge was formed to account for a portion of the temporal variation in density or biomass estimates in the original 100-m sites. River discharge information was available from the Bureau of Reclamation for the Teanaway River, and that was used as an index of discharge for all tributary streams included in the models. Water-years are recorded from October through September, conveniently ending just beyond the end of our tributary sampling season. Water-years are used for the purpose of modeling. The predictive models were simple linear regression models of the form:

$$\text{Predicted Estimate} = \text{Intercept} + B_1(\text{index}_1) + B_2(\text{index}_2)$$

Where B_1 and B_2 are regression coefficients and index_1 and index_2 are discharge indices. Several potential indices of flow magnitude and variation were constructed for critical periods in the life history of trout. The index AM attempts to capture flow variation that may alter egg survival. AM is calculated as the April flow minus the May flow divided by the average of the two. The index CVFO is intended to capture flow variation that may alter survival or distribution of juvenile or adult fish. CVFO is the coefficient of variation of monthly flows from February through October. CVFO is intended to capture highly variable flows which may affect growth or survival of a range of sizes of fish. The index MIN is the minimum of the monthly flows for the water year and is intended to capture critical flow levels that may limit resource or habitat availability. These indices were computed using monthly means, minimums, or maximums and each index was computed for the sample year and for the three years previous to the sample year (for lag analyses, relative to the sample year), encompassing the life span of the most abundant age groups sampled (Martin and Pearsons 1994). The two best indices were chosen by stepwise regression. Models were limited to two parameters and an intercept to prevent over-parameterization relative to the amount of data available.

To facilitate modeling of rainbow trout density and biomass, analysis was limited to a select group of tributaries near the Teanaway River area (MFT, NFT, TAN, and WFT) that have been monitored since 1990 and have similar temporal trends in abundance. Sites within tributaries were averaged and then tributaries were averaged to compute an overall estimate for each sample year. A model was formed for both rainbow trout density and biomass estimates. Similar models were constructed for westslope cutthroat trout density and biomass. Cutthroat trout models are based on the only two sites (NFT3 and TAN3) where cutthroat trout have consistently been sampled in numbers sufficient to compute accurate estimates of abundance.

Implications for Monitoring

Yearly variation in population density data during the baseline period was evaluated relative to acceptable impact levels detailed in chapter 1. We examined whether detection of an effect on rainbow trout at the acceptable impact level (≤40%) was feasible and, if so, how rapidly detection could be accomplished. For mean fork lengths of all species, 80 mm was subtracted from the mean before computing the coefficient of variation to correct for inefficiency in catching small fish. Setting $\alpha = 0.10$ and $\beta = 0.10$ to control power to 0.90, we computed the number of years it would take to detect an impact of 40% both for raw rainbow trout density and biomass estimates and for estimates standardized to the predictive model. An analysis of our ability to detect impacts to cutthroat trout and bull trout was also conducted. Though the acceptable level is no impact for both species, an impact of 10% was used to evaluate the relative power of the monitoring program to detect impacts to density, biomass and mean fork length. We also calculated the number of years to detect a 10% impact to cutthroat trout density or biomass on the basis of estimates standardized to the predictive models. The formula for sample size for a given power for a two-sample comparison (Zar 1996, equation 8.22, page 133, adapted for one-tailed test per page 134) is:

$$n \geq 2\left(\frac{\sigma}{\delta}\right)^2 \left\{ t_{\alpha[v]} + t_{(1-P)[v]} \right\}^2$$

Where: n = number of years to detect an impact
σ = standard deviation
δ = acceptable impact level
α = significance level
v = degrees of freedom, df = $2(n - 1)$
P = power to detect an impact equal to δ. $P = (1 - \beta)$
$t_{\alpha[v]}$ = value from a one-tailed t table with probability α and v df
$t_{(1-P)[v]}$ = value from a one-tailed t table with probability 1- P (=β) and v df.

The values of σ and δ must be expressed in the same units (percentage of the mean is most convenient for our purposes) so that units cancel out in the formula above. The formula is solved iteratively by starting with an estimated value for n. After each iteration, the formula is recalculated with the previous solution as the new estimate until the estimate stabilizes. The estimated sample size is the number of years required to detect a mean sustained impact equal to the maximum acceptable level.

Results

1990 through 1997: 100-m index sites

Rainbow trout were the most abundant and widely distributed salmonids in index sites on tributaries to the upper Yakima River. Rainbow trout density and biomass estimates (Tables 2 and 3) varied greatly among tributaries and sites but less so among years within sites. Consistent trends across years were found for density and biomass for a group of tributaries clustered near the Teanaway River. This group includes NFT, WFT, MFT, MST, SWK and TAN. These consistent trends suggest that some regional processes are consistently affecting abundance within this group of sites. Temporal trends in density or biomass in other tributaries such as CAB or UMT, were sometimes markedly different than those of the Teanaway area group. Rainbow trout mean fork lengths (Table 4) were not highly variable among sites or among years. Distribution changed little among years.

Cutthroat trout density also varied greatly among sites and years (Table 5) as did biomass (Table 6). Cutthroat trout were found in fewer sites and typically at lower densities than were rainbow trout. Low densities reduced the accuracy of estimates. Only the NFT 3 and TAN 3 sites consistently had sufficient numbers of cutthroat trout to accurately estimate density and biomass. The low numbers of cutthroat trout collected at many index sites made estimates of mean fork length unreliable and highly variable (Table 7). Distribution was consistent among years.

Bull trout were encountered in only one index site, NFT3 and densities were generally very low. The estimated density (number of fish/km) for bull trout for 1990, 1993, 1994, and 1995 were 55, 35, 30 and 10, respectively. Biomass estimates (kg/km) for the same years were 3.12, 0.54, 0.18, and 0.28, respectively. No bull trout were collected in 1991, 1992, 1996, or 1997 in the original index sites. Bull trout mean fork lengths (mm) were 149.0, 118.8, 92.3, and 132.0 for 1990, 1993, 1994, and 1995, respectively. The low numbers of bull trout collected at many index sites made estimates of mean fork length unreliable and highly variable. Densities of bull trout are so low that distribution cannot be reliably estimated.

Spring chinook salmon were encountered in a limited number of sites and often at low densities (Tables 8 and 9). Density and biomass estimates were highly variable among years and among sites. The low numbers of spring chinook salmon collected at many index sites made estimates of mean fork length unreliable (Table 10). Distribution was patchy, but spring chinook salmon were more often found in the lower elevation sites that are nearest the main stem Yakima River, and rarely found at higher elevation sites within a tributary.

Table 2. Rainbow trout density (number of fish/km) in original 100-m index sites for fish >79mm in tributaries of the upper Yakima River. Tributaries are arranged in descending order of average elevation of index sites.

	Year									
Site	1990	1991	1992	1993	1994	1995	1996	1997	Mean	SD
MAN1			138		130	204	770			
MAN2			95		170					
MAN3			0		0			0		
Mean			78		100	204	770	0	230	310
NFT1	163	226	271	160	306	42	140	770		
NFT2	516	181	51	278	157	95	30	160		
NFT3	28	30	62	44	80	31	20	0		
Mean	236	146	128	161	181	56	63	310	160	85
JUN		53	18	609	20	64			153	256
TAN1	455	1991	1472	1180	630	400	267	417		
TAN2	339	298	568	492	504	440	390	708		
TAN3	26	183	119	282	357	164	80	144		
Mean	273	824	720	651	497	335	246	423	496	216
MFT1	378	337	248	201	240	112	240	736		
MFT2	502	220	258	162	80	110	99	570		
MFT3	485	307	418	438	560	310	130	440		
Mean	455	288	308	267	293	177	156	582	316	141
WFT1	207	109	129	214	93	30	75	223		
WFT2	212	288	387	192	354	96	40	359		
WFT3	113	254	264	268	280	251	100	240		
Mean	177	217	260	225	242	126	72	274	199	70
CAB1	22	40	73	0	70	198	40	60		
CAB2	75	10	228	70	40	90	116			
Mean	49	25	150	35	55	144	78	60	75	48
MST1					80	42	97	190		
MST2					110	30	0	150		
MST3					40	50	30	347		
Mean					77	41	42	229	97	89
SWK1				443	787	150	79	280		
SWK2			495	558	250	480	327	640		
SWK3			265	371	240	240	209	423		
Mean			380	458	426	290	205	448	368	101
BIG1			235						235	
UMT1			183	157	69	69	360	296		
UMT15				50	30	190	507	666		
UMT2			29	28	0	189	734	684		
Mean			106	78	33	150	533	548	241	235

Table 3. Rainbow trout biomass (kg/km) in original 100-m index sites for fish >79mm in tributaries of the upper Yakima River. Tributaries are arranged in descending order of average elevation of index sites.

	Year									
Site	1990	1991	1992	1993	1994	1995	1996	1997	Mean	SD
MAN1			4.64		2.72	17.86	45.76			
MAN2			4.09		5.64					
MAN3			0		0			0		
Mean			2.91		2.79	17.86	45.76	0.00	13.86	19.16
NFT1	3.45	5.68	5.68	3.72	6.07	2.40	4.03	18.33		
NFT2	15.81	5.17	1.19	6.07	3.32	4.70	0.75	4.65		
NFT3	0.98	2.17	1.45	1.17	4.05	2.83	1.83	0		
Mean	6.75	4.34	2.77	3.65	4.48	3.31	2.20	7.66	4.40	1.90
JUN		0.64	0.18	7.26	0.19	1.39			1.93	3.02
TAN1	17.92	25.94	47.14	33.98	18.14	20.89	8.99	13.27		
TAN2	13.87	12.23	22.83	16.53	16.19	14.80	12.38	18.24		
TAN3	2.46	6.52	4.64	10.21	9.98	7.79	3.87	5.00		
Mean	11.42	14.90	24.87	20.24	14.77	14.49	8.41	12.17	15.16	5.20
MFT1	9.66	9.66	6.99	6.87	7.94	8.60	15.26	26.76		
MFT2	16.65	5.83	7.82	5.45	3.08	2.76	3.77	14.70		
MFT3	17.12	15.92	14.22	9.19	18.22	9.79	4.67	11.07		
Mean	14.48	10.47	9.68	7.17	9.74	7.05	7.90	17.51	10.50	3.70
WFT1	8.31	5.61	3.28	3.47	2.15	1.22	2.37	6.79		
WFT2	8.50	10.57	9.98	5.71	12.01	3.41	1.28	13.51		
WFT3	5.74	7.47	6.63	5.50	6.27	6.49	4.44	7.28		
Mean	7.52	7.88	6.63	4.89	6.81	3.70	2.69	9.19	6.17	2.21
CAB1	0.82	1.65	1.44	0	2.95	7.56	2.78	0.81		
CAB2	4.87	0.92	6.99	3.81	5.22	9.89	12.39			
Mean	2.85	1.28	4.21	1.91	4.09	8.72	7.58	0.81	3.93	2.89
MST1					4.52	2.05	6.59	7.91		
MST2					8.15	4.20	0	6.77		
MST3					1.53	1.65	1.82	12.75		
Mean					4.73	2.63	2.80	9.14	4.83	3.03
SWK1				20.61	47.15	9.35	5.83	33.70		
SWK2			17.83	17.97	11.23	23.35	13.73	18.57		
SWK3			6.65	9.28	7.49	6.47	7.06	10.97		
Mean			12.24	15.95	21.96	13.06	8.87	21.08	15.53	5.17
BIG1			6.83						6.83	
UMT1			2.88	3.46	2.05	2.81	7.78	6.14		
UMT15				10.70	5.38	4.92	10.18	17.90		
UMT2			1.93	6.32	0	8.00	23.25	20.18		
Mean			2.40	6.83	2.48	5.24	13.74	14.74	7.57	5.44

Table 4. Rainbow trout mean fork length (mm) in original 100-m index sites for fish >79mm in tributaries of the upper Yakima River. Tributaries are arranged in descending order of average elevation of index sites.

	Year									
Site	1990	1991	1992	1993	1994	1995	1996	1997	Mean	SD
MAN1			93.6		103.5	156.7	133.1			
MAN2			132.5		136.2					
MAN3										
Mean			113.1		119.9	156.7	133.1		130.7	19.2
NFT1	120.1	127.5	120.9	126.9	117.0	150.4	123.1	125.3		
NFT2	134.4	120.6	130.0	120.7	115.3	136.7	111.7	129.2		
NFT3	113.7	148.3	111.0	108.2	134.6	162.3	166.5			
Mean	122.7	132.2	120.6	118.6	122.3	149.8	133.7	127.2	128.4	10.2
JUN		96.0	100.5	100.3	95.0	120.0			102.4	10.2
TAN1	141.9	139.3	133.9	128.1	124.9	144.5	136.4	132.7		
TAN2	136.9	138.3	137.5	138.4	130.6	131.6	134.5	124.6		
TAN3	159.3	122.0	138.8	132.1	120.7	125.8	125.1	123.1		
Mean	146.1	133.2	136.7	132.9	125.4	134.0	132.0	126.8	133.4	6.3
MFT1	127.0	122.8	131.4	138.1	138.7	159.0	138.1	144.4		
MFT2	140.2	124.9	131.0	134.1	129.8	125.8	131.6	120.1		
MFT3	134.7	145.7	140.8	120.7	129.0	128.1	133.5	123.8		
Mean	133.9	131.1	134.4	131.0	132.5	137.6	134.4	129.4	133.0	2.6
WFT1	145.7	135.0	119.4	142.7	124.3	130.7	117.2	138.3		
WFT2	143.8	134.2	130.6	135.2	139.3	143.5	120.0	142.4		
WFT3	135.6	110.3	125.8	113.8	114.7	113.5	123.0	133.2		
Mean	141.7	126.5	125.3	130.6	126.1	129.2	120.1	138.0	129.7	7.1
CAB1	130.5	128.0	122.6		130.4	133.1	150.8	105.0		
CAB2	169.1	186.0	131.5	152.4	185.7	147.4	186.2			
Mean	149.8	157.0	127.1	152.4	158.0	140.3	168.5	105.0	144.8	20.3
MST1					146.4	141.5	154.5	149.1		
MST2					128.0	190.3		158.0		
MST3					129.0	110.2	150.7	139.6		
Mean					134.5	147.3	152.6	148.9	145.8	7.9
SWK1				157.5	163.6	160.7	156.9	160.5		
SWK2			142.9	135.6	152.4	148.5	142.7	132.6		
SWK3			125.7	126.0	137.8	126.0	130.2	123.0		
Mean			134.3	139.7	151.3	145.1	143.3	138.7	142.0	5.9
BIG1			123.0						128.5	
UMT1			107.2	119.2	135.3	146.6	116.6	120.1		
UMT15				228.8	212.3	122.7	117.4	123.6		
UMT2			151.7	216.7		146.5	134.0	128.6		
Mean			129.4	188.2	173.8	138.6	122.7	124.1	146.1	28.0

Table 5. Cutthroat trout density (number/km) in original 100-m index sites for fish >79mm in tributaries of the upper Yakima River. Tributaries are arranged in descending order of average elevation of index sites.

	Year									
Site	1990	1991	1992	1993	1994	1995	1996	1997	Mean	SD
MAN1			0		0	0	0			
MAN2			26		20					
MAN3			312		539			250		
Mean			113		186	0	0	250	110	111
NFT1	0	0	0	0	0	0	0	0		
NFT2	0	0	0	0	0	0	0	0		
NFT3	404	138	114	237	200	375	140	150		
Mean	135	46	38	79	67	125	47	50	73	37
JUN		0	9	0	0	0			2	4
TAN1	20	0	10	20	30	20	0	0		
TAN2	0	0	0	0	10	11	0	0		
TAN3	206	64	50	64	63	55	0	10		
Mean	75	21	20	28	34	29	0	3	26	23
MFT1	0	0	0	0	10	0	0	9		
MFT2	7	8	0	0	0	0	0	0		
MFT3	10	0	10	0	0	10	0	0		
Mean	6	3	3	0	3	3	0	3	3	2
WFT1	0	0	0	0	0	0	0	0		
WFT2	0	7	0	0	0	9	0	0		
WFT3	0	0	0	0	10	0	0	0		
Mean	0	2	0	0	3	3	0	0	1	2
CAB1	0	0	0	0	0	0	0	0		
CAB2	47	0	60	0	13	0	32			
Mean	24	0	30	0	7	0	16	0	9	12
MST1					10	0	0	0		
MST2					0	0	0	0		
MST3					0	0	0	0		
Mean					3	0	0	0	1	2
SWK1				0	0	0	0	0		
SWK2			0	10	0	0	0	0		
SWK3			47	67	20	30	0	9		
Mean			24	25	7	10	0	3	11	11
BIG1			18						18	
UMT1			0	0	0	0	0	0		
UMT15				0	0	0	0	0		
UMT2			0	0	0	0	0	0		
Mean			0	0	0	0	0	0	0	0

Table 6. Cutthroat trout biomass (kg/km) in original 100-m index sites for fish >79mm in tributaries of the upper Yakima River. Tributaries are arranged in descending order of average elevation of index sites.

	Year									
Site	1990	1991	1992	1993	1994	1995	1996	1997	Mean	SD
MAN1			0		0	0	0			
MAN2			1.05		0.23					
MAN3			7.02		16.06			9.29		
Mean			2.69		5.43	0.00	0.00	9.29	3.48	3.95
NFT1	0	0	0	0	0	0	0	0		
NFT2	0	0	0	0	0	0	0	0		
NFT3	21.75	10.24	7.20	11.75	10.23	18.72	6.24	9.63		
Mean	7.25	3.41	2.40	3.92	3.41	6.24	2.08	3.21	3.99	1.82
JUN		0	0.11	0	0	0			0.02	0.05
TAN1	2.37	0	1.09	1.04	1.62	4.03	0	0		
TAN2	0	0	0	0	0.89	1.68	0	0		
TAN3	8.52	3.93	2.98	0.96	1.92	2.05	0	0.31		
Mean	3.63	1.31	1.36	0.67	1.48	2.58	0.00	0.10	1.39	1.23
MFT1	0	0	0	0	0.65	0	0	2.21		
MFT2	0.57	0.07	0	0	0	0	0	0		
MFT3	0.30	0	0.25	0	0	0.25	0	0		
Mean	0.29	0.02	0.08	0.00	0.22	0.08	0.00	0.74	0.18	0.25
WFT1	0	0	0	0	0	0	0	0		
WFT2	0	0.24	0	0	0	0.25	0	0		
WFT3	0	0	0	0	0.19	0	0	0		
Mean	0.00	0.08	0.00	0.00	0.06	0.08	0.00	0.00	0.03	0.04
CAB1	0	0	0	0	0	0	0	0		
CAB2	2.04	0	0.80	0	0.32	0	2.04			
Mean	1.02	0.00	0.40	0.00	0.16	0.00	1.02	0.00	0.33	0.45
MST1					2.13	0	0	0		
MST2					0	0	0	0		
MST3					0	0	0	0		
Mean					0.71	0.00	0.00	0.00	0.18	0.36
SWK1				0	0	0	0	0		
SWK2			0	0.44	0	0	0	0		
SWK3			1.66	1.65	0.46	0.52	0	0.28		
Mean			0.83	0.70	0.15	0.17	0.00	0.09	0.32	0.35
BIG1			0.33						0.33	
UMT1			0	0	0	0	0	0		
UMT15				0	0	0	0	0		
UMT2			0	0	0	0	0	0		
Mean			0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table 7. Cutthroat trout mean fork length (mm) in original 100-m index sites for fish >79mm in tributaries of the upper Yakima River. Tributaries are arranged in descending order of average elevation of index sites.

	Year									
Site	1990	1991	1992	1993	1994	1995	1996	1997	Mean	SD
MAN1										
MAN2			156.0		103.5					
MAN3			120.4		132.7			145.0		
Mean			138.2		118.1			145.0	133.4	13.9
NFT1										
NFT2										
NFT3	160.6	188.0	169.9	149.0	154.4	156.8	156.8	170.7		
Mean	160.6	188.0	169.9	149.0	154.4	156.8	156.8	170.7	163.2	12.5
JUN			107.0						107.0	
TAN1	229.5		222.0	163.0	174.0	269.5				
TAN2					203.0	241.0		0.0		
TAN3	140.5	167.9	155.0	109.4	133.4	140.4		141.0		
Mean	185.0	167.9	188.5	136.2	170.1	217.0		70.5	162.2	47.2
MFT1					196.0			282.0		
MFT2	195.0	92.0								
MFT3	142.0		136.0			134.0				
Mean	168.5	92.0	136.0		196.0	134.0		282.0	168.1	65.9
WFT1										
WFT2		148.0				138.0				
WFT3					124.0					
Mean		148.0			124.0	138.0			136.7	12.1
CAB1										
CAB2	147.0		110.5		133.0		181.3			
Mean	147.0		110.5		133.0		181.3		143.0	29.7
MST1					266.0					
MST2										
MST3										
Mean					266.0				266.0	
SWK1										
SWK2				165.0						
SWK3			138.6	132.6	135.0	117.3		136.0		
Mean			138.6	148.8	135.0	117.3		136.0	135.1	11.4
BIG1			118.0						118.0	
UMT1										
UMT15										
UMT2										
Mean										

Table 8. Spring chinook salmon density (number/km) in original 100-m index sites for fish >79mm in tributaries of the upper Yakima River. Tributaries are arranged in descending order of average elevation of index sites.

Site	1990	1991	1992	1993	1994	1995	1996	1997
MAN1			35		140	253	0	
MAN2			0		0			
MAN3			0		0			0
NFT1	343	0	0	70	19	34	0	90
NFT2	0	0	0	0	0	0	0	10
NFT3	0	0	0	0	0	0	0	0
JUN1		0	0	0	0	0		
TAN1	0	0	0	0	0	0	0	0
TAN2	0	0	0	0	0	0	0	0
TAN3	0	0	0	0	0	0	0	0
MFT1	312	0	9	0	0	31	0	27
MFT2	37	0	0	0	0	0	0	10
MFT3	0	0	0	0	0	0	0	0
WFT1	180	0	0	0	0	0	0	19
WFT2	25	0	8	0	0	0	0	0
WFT3	0	0	0	0	0	0	0	0
CAB1	22	0	0	0	0	0	10	0
CAB2	9	0	10	0	0	0	0	
MST1					70	42	0	360
MST2					0	0	0	100
MST3					10	0	0	13
SWK1				93	745	140	10	280
SWK2			10	38	0	160	0	0
SWK3			0	0	0	0	0	0
BIG1			0					
UMT1			0	433	10	30	0	26
UMT15				0	0	0	0	0
UMT2			0	0	0	0	0	0

Table 9. Spring chinook salmon biomass (kg/km) in original 100-m index sites for fish >79mm in tributaries of the upper Yakima River. Tributaries are arranged in descending order of average elevation of index sites.

Site	1990	1991	1992	1993	1994	1995	1996	1997
MAN1			0.33		1.26	2.72	0	
MAN2			0		0			
MAN3			0		0			0
NFT1	2.72	0	0	0.46	0.10	0.30	0	0.66
NFT2	0	0	0	0	0	0	0	0.08
NFT3	0	0	0	0	0	0	0	0
JUN1		0	0	0	0	0		
TAN1	0	0	0	0	0	0	0	0
TAN2	0	0	0	0	0	0	0	0
TAN3	0	0	0	0	0	0	0	0
MFT1	2.48	0	0.06	0	0	0.30	0	0.24
MFT2	0.32	0	0	0	0	0	0	0.08
MFT3	0	0	0	0	0	0	0	0
WFT1	1.44	0	0	0	0	0	0	0.13
WFT2	0.19	0	0.05	0	0	0	0	0
WFT3	0	0	0	0	0	0	0	0
CAB1	0.19	0	0	0	0	0	0.07	0
CAB2	0.08	0	0.09	0	0	0	0	
MST1					0.47	0.35	0	2.88
MST2					0	0	0	0.94
MST3					0.05	0	0	0.12
SWK1				0.73	4.63	1.00	0.12	2.87
SWK2			0.07	0.32	0	1.27	0	0
SWK3			0	0	0	0	0	0
BIG1			0					
UMT1			0	3.38	0.06	0.19	0	0.22
UMT15				0	0	0	0	0
UMT2			0	0	0	0	0	0

Table 10. Spring chinook salmon mean fork length (mm) in original 100-m index sites for fish >79mm in tributaries of the upper Yakima River. Tributaries are arranged in descending order of average elevation of index sites.

Site	1990	1991	1992	1993	1994	1995	1996	1997
MAN1			93.0		89.3	92.4		
MAN2								
MAN3								
NFT1	90.8			83.9	80.5	89.8		84.0
NFT2								89.0
NFT3								
JUN1								
TAN1								
TAN2								
TAN3								
MFT1	91.2		81.0			86.7		87.7
MFT2	93.6							86.0
MFT3								
WFT1	90.0							81.5
WFT2	88.0		80.0					
WFT3								
CAB1	90.0						83.0	
CAB2	89.0		82.0					
MST1					81.9	85.8		86.7
MST2								92.6
MST3					85.0			90.0
SWK1				85.8	84.7	85.6	107.0	97.3
SWK2			86.0	88.3		89.3		
SWK3								
BIG1								
UMT1				86.8	86.0	82.7		91.0
UMT15								
UMT2								

Predictive models

Regression models of rainbow trout and cutthroat trout density and biomass with respect to stream discharge explained a large amount of variation in those estimates among years. Table 11 details the flow indices selected for the models and the model statistics. The regression models chosen by stepwise regression for rainbow trout density and biomass both included AM as the most important index. Rainbow trout density and biomass in the sample year increased as the maximum April flow increased relative to the maximum May flow 3 years prior to the sample year. Rainbow trout density decreased as mean flows from February through October became more variable, and biomass increased as the minimum of maximum monthly flows for the water year previous to the sample year increased. The models with respect to stream flow explained 70% of the temporal variation in baseline rainbow trout population density and 83% of the temporal variation in biomass estimates.

The regression models for cutthroat trout density and biomass chosen by stepwise regression both included MIN as the most important index (Table 11). Cutthroat trout density and biomass increased as the minimum of maximum monthly flows for the water year decreased. Cutthroat trout density also decreased as mean flows from February through October became more variable for the water year 2 years prior to the sample year, but not significantly. Biomass increased as the minimum April flow increased relative to the minimum May flow 3 years prior to the sample year. The models with respect to stream flow explained 65% of the temporal variation in baseline cutthroat trout population density and 66% of the temporal variation in biomass estimates.

Table 11. Parameter estimates and statistics for predictive models of density and biomass with respect to indices of monthly stream flow characteristics.

Model	Tributaries	Sites	Parameter	Estimate	Index	Flow type	Lag (yrs)	p	R^2
Rainbow trout									
Density	4	12	Intercept	398.123				<0.001	0.90
			B1	1.259	AM	MAX	3	0.011	
			B2	-2.005	CVFO	MEAN	0	0.025	
Biomass	4	12	Intercept	6.947				<0.001	0.95
			B1	0.042	AM	MAX	3	<0.001	
			B2	0.006	MIN	MAX	1	0.004	
Cutthroat trout									
Density	2	2	Intercept	300.685				0.001	0.88
			B1	-8.606	MIN	MAX	1	0.006	
			B2	0.874	AM	MIN	3	0.012	
Biomass	2	2	Intercept	14.669				<0.001	0.94
			B1	-0.405	MIN	MAX	1	0.001	
			B2	0.043	AM	MIN	3	0.002	

1997 additions: 200 meter sites

Single pass electrofishing samples of 400-m reaches in 1996 indicated that doubling the length of the index sites would considerably increase the accuracy of estimates (a 45% reduction in coefficient of variation for 200-m sites relative to 100-m) with moderate impact on manpower and equipment required, while sampling 300 meters or more would provide smaller marginal gains (a 67% reduction in coefficient of variation for 300-m sites relative to 100-m) and could not be completed in one day with available equipment, greatly increasing required resources. This information was used to justify sampling an additional 100-m reach at each index site in 1997.

Increasing index site length increased the accuracy of estimates of density, biomass, and mean fork length. For rainbow trout, sampling an additional 100 meters sometimes resulted in very different estimates of density for each 100-m reach, with the 200-m estimate usually falling somewhere between the two 100-m estimates (Table 12). Variation among sites within a tributary was generally reduced by increasing site length, indicating that 200-m estimates were more accurate and representative of the entire stream. Doubling site length had little effect on estimates of mean fork length, except where densities were low. In only one site (CAB1) were rainbow trout found in one but not both 100-m reaches, so increased index site length had a negligible effect on estimates of rainbow trout distribution.

Table 12. Rainbow trout density, biomass, and mean fork length for 1997 200-meter index sites for fish >79mm in tributaries of the upper Yakima River.

Site	Density (number/km)			Biomass (kg/km)			Mean fork length (mm)		
	0 – 100	100 – 200	0 – 200	0 – 100	100 – 200	0 – 200	0 - 100	100 - 200	0 – 200
CAB1	60	0	20	0.81	0	0.41	105.0		105.0
DOMA	0	0	0	0	0	0			
MAN3	0	0	0	0	0	0			
MFT1	736	619	600	26.76	18.87	22.91	144.4	136.6	140.3
MFT2	570	30	300	14.70	0.53	7.61	120.1	115.0	119.8
MFT3	440	360	335	11.07	9.34	10.20	123.8	127.0	125.6
MST1	190			7.91			149.1		
MST2	150	190	150	6.77	4.60	5.68	158.0	125.8	136.9
MST3	347	137	203	12.75	9.59	10.59	139.6	150.0	144.2
NFT1	770	190	385	18.33	5.10	11.72	125.3	127.2	125.8
NFT2	160	580	370	4.65	16.44	10.55	129.2	124.3	125.8
NFT3	0	0	0	0	0	0			
NFTA	30	30	25	1.19	1.75	1.47	150.5	167.0	160.4
STFA	180	364	241	6.07	7.55	6.43	132.9	118.8	127.1
STFB	30	20	20	0.86	0.56	0.71	131.0	131.5	131.3
SWK1	280	591	416	33.70	19.11	27.35	160.5	132.4	143.5
SWK2	640	500	570	18.57	13.42	15.99	132.6	127.6	130.3
SWK3	423	376	402	10.97	7.72	9.49	123.0	115.8	119.9
TAN1	417	500	427	13.27	17.35	15.31	132.7	139.2	136.5
TAN2	708	659	652	18.24	15.06	16.85	124.6	120.2	122.6
TAN3	144	350	305	5.00	10.62	7.85	123.1	127.5	126.4
TANA	290	240	235	7.75	7.54	7.65	127.3	130.4	128.8
TANB	47	30	43	2.37	1.03	1.72	156.7	135.0	145.8
UMT1	296	450	354	6.14	11.03	8.15	120.1	125.5	122.9
UMT15	666	493	514	17.90	12.10	14.91	123.6	122.8	123.2
UMT2	684	583	637	20.18	22.65	21.22	128.6	139.7	132.9
WFT1	223	60	143	6.79	1.64	4.25	138.3	132.0	136.9
WFT2	359	169	229	13.51	5.78	9.03	142.4	141.9	142.1
WFT3	240	210	210	7.28	3.97	5.62	133.2	117.1	125.4
WILA	0	0	0	0	0	0			

A greater benefit of increasing site length was realized in less abundant species, such as cutthroat trout. At lower densities, the additional 100-m reach increased the probability that a species would be found at an index site and increased the probability that a descending multiple removal pattern would result from sampling (Table 13). Variation among sites and among tributaries was decreased by increasing index site length. There were five sites where cutthroat trout were found in only one of the 100-m reaches, but not both. The increased index site length should improve the reliability and repeatability of estimates of distribution. Cutthroat monitoring also benefited from new sites added in 1997 in areas of cutthroat trout abundance (DOMA, NFTA, STFA, STFB, TANA, TANB, and WILA). All of these sites, with the exception of NFTA, were found to have sufficient numbers of cutthroat trout to allow reliable and repeatable multiple removal estimates.

The distribution of spring chinook salmon within our study sites was patchy, and densities were often low (Table 14), especially at greater distances from the main stem Yakima River. Increasing site lengths increased the chances that they would be detected within a site. Increased site length should improve the accuracy of estimates of distribution. To a lesser extent, the increased site length increased the number of sites where multiple removal estimates could be conducted and should result in greater ability to accurately quantify abundance.

Bull trout were found only at site NFT3. No fish were encountered in the original 100-m reach, but 1 fish was found in the second 100-m reach. Therefore, increased index site length improved the ability of index sites to contribute information on bull trout distribution. Densities (number of fish/km) were estimated to be 0 for the 0 – 100 reach, 10 for the 100 – 200 reach and 5 for the entire 200-m site. Biomass (kg/km) was estimated to be 0, 2.13, and 1.04 for the first 0 – 100 reach, 100 – 200 reach, and the entire site, respectively. Mean fork length was estimated to be 258 mm except for the 0 – 100 reach where no bull trout were caught.

Implications of Variability for Monitoring

The number of years required to detect impacts at the acceptable level for raw estimates of density or biomass for any salmonid species were impractical for rapid detection (Table 15). An acceptable impact level of less than or equal to 40% has been proposed for rainbow trout in tributaries. Using raw estimates, only mean fork length would allow detection of a 40% impact in two years or less. Though rapid detection of impacts is a goal of most monitoring programs, detection in two years or less is important for this supplementation program because adults will begin returning after 2 years, adding additional potential mechanisms for impacts. The goal is to evaluate potential impacts of smolts, before confounding factors increase. Westslope cutthroat trout and bull trout have a proposed acceptable impact level of no impact (0%). The sample size equation can not be solved for $\ast = 0\%$ impact, so we computed sample sizes using a 10% impact level to compare the relative power available for detecting impacts. Even at a 10% impact level, raw estimates of density or biomass for cutthroat trout or bull trout were far too variable to allow detection within two years (Table 15).

Table 13. Cutthroat trout density, biomass, and mean fork length for 1997 200-meter index sites for fish >79mm in tributaries of the upper Yakima River.

Site	Density (number/km)			Biomass (kg/km)			Mean fork length (mm)		
	0 – 100	100 – 200	0 – 200	0 – 100	100 – 200	0 – 200	0 - 100	100 - 200	0 – 200
CAB1	0	0	0	0	0	0			
DOMA	96	71	83	10.76	2.85	6.45	177.5	144.9	162.1
MAN3	250	28	160	9.29	1.56	7.35	145.0	171.0	146.2
MFT1	9	0	5	2.21	0	1.13	282.0		282.0
MFT2	0	0	0	0	0	0			
MFT3	0	0	0	0	0	0			
MST1	0			0					
MST2	0	0	0	0	0	0			
MST3	0	0	0	0	0	0			
NFT1	0	0	0	0	0	0			
NFT2	0	0	0	0	0	0			
NFT3	150	38	78	9.63	2.17	5.81	170.7	162.8	168.4
NFTA	10	0	5	0.60	0	0.30	172.0		172.0
STFA	40	0	30	1.59	0	1.20	139.0		139.0
STFB	180	170	165	5.03	4.21	4.62	129.2	125.4	127.5
SWK1	0	0	0	0	0	0			
SWK2	0	0	0	0	0	0			
SWK3	9	0	5	0.28	0	0.15	136.0		136.0
TAN1	0	0	0	0	0	0			
TAN2	0	11	5	0	1.69	0.74		239.0	239.0
TAN3	10	30	25	0.31	2.24	1.29	141.0	191.5	174.7
TANA	80	20	50	2.40	1.34	1.87	132.8	175.0	141.2
TANB	84	100	97	3.21	3.97	3.58	147.0	146.3	146.6
UMT1	0	0	0	0	0	0			
UMT15	0	0	0	0	0	0			
UMT2	0	0	0	0	0	0			
WFT1	0	0	0	0	0	0			
WFT2	0	0	0	0	0	0			
WFT3	0	0	0	0	0	0			
WILA	311	204	234	16.95	10.57	13.83	164.8	168.4	166.4

Table 14. Spring chinook salmon density, biomass, and mean fork length for 1997 200-meter index sites for fish >79mm in tributaries of the upper Yakima River.

Site	Density (number/km)			Biomass (kg/km)			Mean fork length (mm)		
	0 – 100	100 – 200	0 – 200	0 – 100	100 – 200	0 – 200	0 - 100	100 - 200	0 – 200
CAB1	0	0	0	0	0	0			
DOMA	0	0	0	0	0	0			
MAN3	0	0	0	0	0	0			
MFT1	27	19	23	0.24	0.17	0.20	87.7	91.0	89.0
MFT2	10	0	5	0.08	0	0.04	86.0	0.0	86.0
MFT3	0	0	0	0	0	0			
MST1	360			2.88			86.7		
MST2	100	70	85	0.94	0.64	0.79	92.6	92.3	92.5
MST3	13	12	13	0.12	0.09	0.10	90.0	84.5	86.3
NFT1	90	0	45	0.66	0	0.33	84.0	0.0	84.0
NFT2	10	0	5	0.08	0	0.04	89.0		89.0
NFT3	0	0	0	0	0	0			
NFTA	0	0	0	0	0	0			
STFA	0	0	0	0	0	0			
STFB	0	0	0	0	0	0			
SWK1	280	57	183	2.87	0.66	1.91	97.3	101.0	98.0
SWK2	0	0	0	0	0	0			
SWK3	0	0	0	0	0	0			
TAN1	0	0	0	0	0	0			
TAN2	0	0	0	0	0	0			
TAN3	0	0	0	0	0	0			
TANA	0	0	0	0	0	0			
TANB	0	0	0	0	0	0			
UMT1	26	138	77	0.22	1.16	0.60	91.0	89.7	90.0
UMT15	0	0	0	0	0	0			
UMT2	0	0	0	0	0	0			
WFT1	19	10	15	0.13	0.08	0.10	81.5	83.0	82.0
WFT2	0	7	4	0	0.07	0.04	0.0	91.0	91.0
WFT3	0	0	0	0	0	0			
WILA	0	0	0	0	0	0			

Table 15. Number of sample years required to detect maximum acceptable impacts to tributary salmonid populations. Data is from 100-m index sites sampled from 1990 through 1997. Tributaries MFT, NFT, TAN, and WFT are included in rainbow trout analyses. Sites NFT3 and TAN3 are included in cutthroat trout analyses. Site NFT3 is included for bull trout analyses.

Taxa	Measurement	Data	Acceptable Impact	CV (%)	Years to detect acceptable impact
Rainbow trout	Density	Raw	≤ 40%	31.44	9
		Model	≤ 40%	9.35	1
	Biomass	Raw	≤ 40%	22.39	5
		Model	≤ 40%	3.72	1
	Mean length	Raw	≤ 40%	7.47	1
Cutthroat trout	Density	Raw	No impact	57.43	434 (10% impact)
		Model	No impact	20.07	53 (10% impact)
	Biomass	Raw	No impact	52.21	359 (10% impact)
		Model	No impact	17.73	42 (10% impact)
	Mean length	Raw	No impact	19.39	50 (10% impact)
Bull trout	Density	Raw	No impact	129.83	2215 (10% impact)
	Biomass	Raw	No impact	207.44	5654 (10% impact)
	Mean length	Raw	No impact	61.55	498 (10% impact)

Predictive models for density or biomass with respect to discharge characteristics accounted for much of the variation among years for rainbow and cutthroat trout estimates. The reductions in coefficients of variation (CV) from raw to data standardized to the predictive model are evident in Table 15. By standardizing estimates to model predictions, enough variation was accounted for to allow rapid detection of impacts to rainbow trout density or biomass. Sample sizes for detection of impacts to cutthroat trout density or biomass decreased with standardization to model predictions, but were still too high for practical detection of impacts near the acceptable level.

Discussion

The high temporal variation in population estimates during baseline monitoring of salmonid species of concern in tributaries to the upper Yakima River is the greatest obstacle to detection of potential impacts of spring chinook salmon supplementation. Rapid, reliable detection requires low unexplained variation, relative to the magnitude of impact we wish to detect. Where this is not the case, there are two ways to reduce unexplained variation: 1) account for non-impact sources of variation and 2) change the sampling design to reduce variation that is an artifact of sampling methods or timing. Both of these methods are being used to improve our ability to detect potential impacts to salmonids of concern in tributaries to the upper Yakima River.

Rainbow trout in tributaries have relatively high acceptable impact levels (40%) and are relatively abundant across many index sites, which minimizes sampling error. In spite of these advantages, raw estimates of rainbow trout density and biomass were still too variable to allow rapid detection of impacts at the acceptable level. Raw estimates of mean fork length were much less variable, and rapid detection of a 40% impact could be achieved in 1 year or less. By using a predictive model to account for variation with respect to stream discharge, unexplained variation was reduced enough to allow 40% impacts to density or biomass to be detected in the first sample year. The use of 200-m index sites should further reduce sampling error, especially where rainbow trout are least abundant, but the benefits to rainbow trout alone might not justify the additional effort. The increased index site lengths will be important in evaluating small scale impacts limited to a portion of a tributary stream. The sites added in 1997 to focus on cutthroat trout are of limited benefit for rainbow trout monitoring, except that they may be nearer the limits of rainbow trout distribution, providing a better indicator of changes in distribution. No modifications to the tributary salmonid monitoring program design are needed to provide for rapid, reliable detection of potential impacts of spring chinook salmon supplementation to rainbow trout at the current maximum acceptable level.

Cutthroat trout have a low acceptable impact level (no impact) and were not abundant in many of the index sites. Low abundance increases sampling error, inflating unexplained variation. Predictive models with respect to stream discharge accounted for the majority of temporal variation in estimates of density or biomass, but rapid detection of impacts even at the 10% level was still not possible. Increased index site lengths improved the accuracy of estimates in 1997, and reduced unexplained variation. Increased site lengths did not result in great improvements in the ability to estimate cutthroat trout density or biomass in index sites where they were not already abundant. Most sites added in 1997 had sufficiently high densities of cutthroat to allow estimates to be made with minimal sampling error. We plan to add more sites in the 1998 sampling season to increase the number of index sites in areas of high cutthroat trout abundance. These changes will improve the ability to detect impacts to cutthroat trout in tributaries to the upper Yakima River. The improved coverage should allow detection of important changes in distribution. Without an impractical increase in effort, however, the monitoring program is unlikely to achieve rapid and reliable detection of smaller impacts to cutthroat trout density, biomass, or mean length that would closely approximate the desired level of no impact. In spite of the difficulty in detecting small impacts, monitoring by the present methods, with modifications to improve coverage in areas of cutthroat trout abundance, will likely be the most efficient way to monitor cutthroat trout for impacts.

Bull trout were found in only one index site and were absent there in almost half the sample years. The low abundance of bull trout results in high variation due to sampling error. The low acceptable impact level and high level of variation result in very low power to detect impacts. Power to detect impacts will remain low in spite of the benefits of an expanded number of index sites and increased site length in 1997 and future monitoring. There is too little information to model the influence of non-impact sources of variation. The tributary monitoring design is not well suited to monitoring changes in density, biomass, or distribution of bull trout. While sightings of bull trout in tributary multiple-removal sampling will continue to provide a small amount of information, it will be necessary to rely on other methods (chapter 2) to effectively monitor changes in bull trout abundance or distribution.

Spring chinook salmon were most often found in index sites nearest the main stem Yakima River, but abundance was generally low. Increased site lengths improved the accuracy of multiple removal estimates of spring chinook salmon. Spring chinook salmon were not found in any of the sites added in 1997. The monitoring program samples many sites near the present edges of spring chinook salmon distribution and should easily be able to detect expanding spring chinook salmon distributions following supplementation. The small number of sites where spring chinook are found, and the variability among years would make it far more difficult to determine that the distribution of spring chinook salmon was contracting. Changes in distribution may be especially informative in evaluating causation or mechanisms of potential impacts of high densities of stocked spring chinook salmon on salmonid species of concern as well as for evaluating the success of the supplementation program.

The monitoring program for salmonids of concern in tributaries of the upper Yakima River provides sufficient power to detect impacts to rainbow trout, but not to other species. Increased site length and additional sites will improve the power to detect impacts to cutthroat trout, but will almost surely fall short of detecting small impacts near the acceptable level of no impact. Bull trout are best monitored by a sampling program directed toward areas of greatest abundance (chapter 2). Sampling effort applied to monitoring salmonid species of concern in tributaries to the upper Yakima River will also provide the ability to detect increases in the abundance or distribution of spring chinook salmon in the same areas.

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Chapter 5

Electrofishing Injury to Stream Salmonids; Injury Assessment at the Sample, Reach, and Stream Scales

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Abstract

Electrofishing injury rates in rainbow trout and juvenile steelhead *Oncorhynchus mykiss* and juvenile spring chinook salmon *O. tshawytscha* were quantified in samples collected in four tributaries to, and one reach of, the Yakima River, Washington. Estimated electrofishing injury rates at the reach and stream scales were generated by using sample injury rates, derived from this study, multiplied by capture probabilities and fraction of habitat sampled. Sample injury rates in small *O. mykiss* and juvenile spring chinook salmon were low. Mean electrofishing injury rate in *O. mykiss* samples captured in tributaries was 5.1%. Only 2.0% of the juvenile spring chinook salmon captured by electrofishing in the Yakima River were injured. Larger *O. mykiss* ($\geq$ 250 mm fork length (FL)) were injured at a higher rate (27.7%) than their smaller counterparts (1.2%; P = 0.023) in the Yakima River sample. Electrofishing injury rates decreased with increasing scale; from the sample to the reach and stream scales. Injury rates for index reaches that we use for long-term monitoring were 4.9% for *O. mykiss* in tributaries, 0.7% for *O. mykiss* <250 mm FL in the Yakima River, and 11.2% for *O. mykiss* $\geq$ 250 mm FL in the Yakima River. Although the injury rate at the reach scale for larger *O. mykiss* was relatively high, we do not believe it affects our long-term monitoring data because annual mortality of these fish is high (>60%) and a small proportion of the population is captured by electrofishing (18 to 21%). Stream-scale injury rates were very low in tributaries (0.1%) and in the Yakima River for smaller *O. mykiss* (a mixture of juvenile steelhead and resident rainbow trout <250 mm FL; 0.1%). Estimated stream-scale injury rate for larger *O. mykiss* ($\geq$ 250 mm FL) was 2.1%. Stream-scale injury rates for all groups examined were below levels that we would expect to affect our long-term monitoring data. The distribution, conservation status, size structure, life-span and annual mortality of the population, the fraction of the habitat sampled, sampling frequency, and availability, effectiveness, and cost of alternative sampling methods, must all be balanced against the need for data when establishing research or monitoring efforts that use electrofishing.

Introduction

Electrofishing has been widely used by fisheries researchers and managers for over 50 years to collect fish and monitor fish populations. During the course of electrofishing history there have been many advancements in gear and methods that have increased the capture probability of fish in various environments (Taylor et al. 1957; Bird and Cowx 1993; Burkhardt and Gutreuter 1995; Cunningham 1995; Miranda et al. 1996). Deleterious effects of electrofishing on the sampled fishes were recognized by some researchers over 40 years ago (Hauck 1949; Pratt 1955), but it was not until relatively recently that gear and methods have been refined in attempts to decrease electrofishing injury and mortality in fish exposed to this method of collection (Sharber et al. 1994). With some native North American fish populations at critically low levels (e.g., Williams et al. 1989; Nehlsen et al. 1991), managers must weigh the potential management or scientific benefits of obtaining fish data by electrofishing against the risks of harming fish populations.

As certain stocks of Pacific salmonids (*Oncorhynchus* species) become more imperiled in the northwest United States (Nehlsen et al. 1991), the potential adverse effects of sampling these populations with any potentially harmful method must be evaluated. Injuring or killing fish by sampling them is considered a 'take' under the Federal Endangered Species Act. Electrofishing has been singled out as an activity that will not be granted a research exemption in areas containing threatened or endangered coho salmon *O. kisutch* (Federal Register 1997). In addition to potential effects on rare or sensitive species, injuring fish that are sampled may affect long-term population monitoring programs, such as the one we are involved with in the Yakima River basin. If electrofishing injury significantly affects one or more response variables being monitored in a given population, then the value of the data collected with this method will be markedly reduced for determining the effects of specific management actions.

The majority of the existing literature on electrofishing injury focuses on injury at the individual fish (sample) scale, with some notable exceptions (Schill and Beland 1995; Kocovsky et al. 1997). Social values associated with electrofishing injury at the sample scale are primarily ethical and perhaps aesthetic relative to the appearance of an injured fish. Electrofishing injury at the monitoring site or reach scale (all fish within the total area sampled) may affect scientific research results that managers use to make resource decisions. Population scale (e.g., all fish in a stream) injury due to electrofishing becomes more a conservation and stewardship issue. Schill and Beland (1995) recommended viewing electrofishing injury from a population perspective. However, all three scales -- sample, reach, and stream -- are important. The stock status, value to humans, and value to the ecosystem of a particular fish population as well as the learning/management value of the sampling of that population, should all be considered when determining acceptable electrofishing injury rates.

As part of an ongoing research effort in the Yakima River basin, Washington, that relies heavily on data collected by electrofishing, we have examined the effects of certain electrofisher settings on the potential to adversely affect the fish we sample (McMichael 1993). To further examine the possibility that our data collection efforts were affecting fishes in various environments within the Yakima basin, we initiated the current study in 1995. We had concerns about electrofishing injuries related to our monitoring program because we observed “bruises” (surface hemorrhages) attributed to electrofishing in an average of 2.6% of the rainbow trout *O. mykiss* we collected from the Yakima River between 1990 and 1995 (WDFW, unpublished data). Some of the bruised fish were unable to swim when released. Further, data presented by Kocovsky et al. (1997) showed that external bruises provided an underestimate of actual spinal injury due to electrofishing.

Our primary objective in this work was to determine if our electrofishing methods cause significant harm to upper Yakima River juvenile steelhead, resident rainbow trout, or juvenile spring chinook salmon *O. tshawytscha* at the sample, reach or stream scales. From a resource stewardship perspective, we were concerned that the critically low population level of upper Yakima River steelhead might be affected by electrofishing. We were also concerned about potential effects to current monitoring programs for *O. mykiss* and spring chinook salmon. Individual effects to larger resident rainbow trout were also of concern due to the popularity of the recreational fishery and the potential for these larger fish to be exposed to electrofishing multiple times. Specifically, we wanted to learn what percentages of smaller rainbow trout or juvenile steelhead (*O. mykiss* < 250 mm fork length (FL)), adult resident rainbow trout (≥ 250 mm FL), and juvenile spring chinook salmon (< 150 mm FL) were injured as a result of being collected with the electrofishing equipment and settings we typically use in the Yakima River and its tributaries. This data was then combined with existing data to assess the rates of electrofishing injury in our annually-sampled index reaches and in entire streams.

Methods

Study Area

All sampling was conducted in the Yakima River basin, a tributary to the Columbia River, upstream from Roza Dam (Figure 1). Samples were collected from four tributaries (West Fork of the Teanaway River, and Swauk, Taneum, and Naneum creeks) and one section of the Yakima River (from Umtanum Creek to Wymer (6 km) in the canyon between the cities of Ellensburg and Yakima, Washington). Tributaries were selected to represent the range of environmental conditions present in the upper Yakima River basin (Table 1).

Field and Laboratory Procedures

We used a control-treatment design to determine the effects of electrofishing on spinal injury in juvenile steelhead, rainbow trout and juvenile spring chinook salmon. In tributaries and the Yakima River, control fish were captured by angling with artificial flies and lures prior to the collection of treatment fish with electrofishing equipment. We assumed that angling would not produce spinal injuries. To minimize our chances of collecting fish we had electrofished in

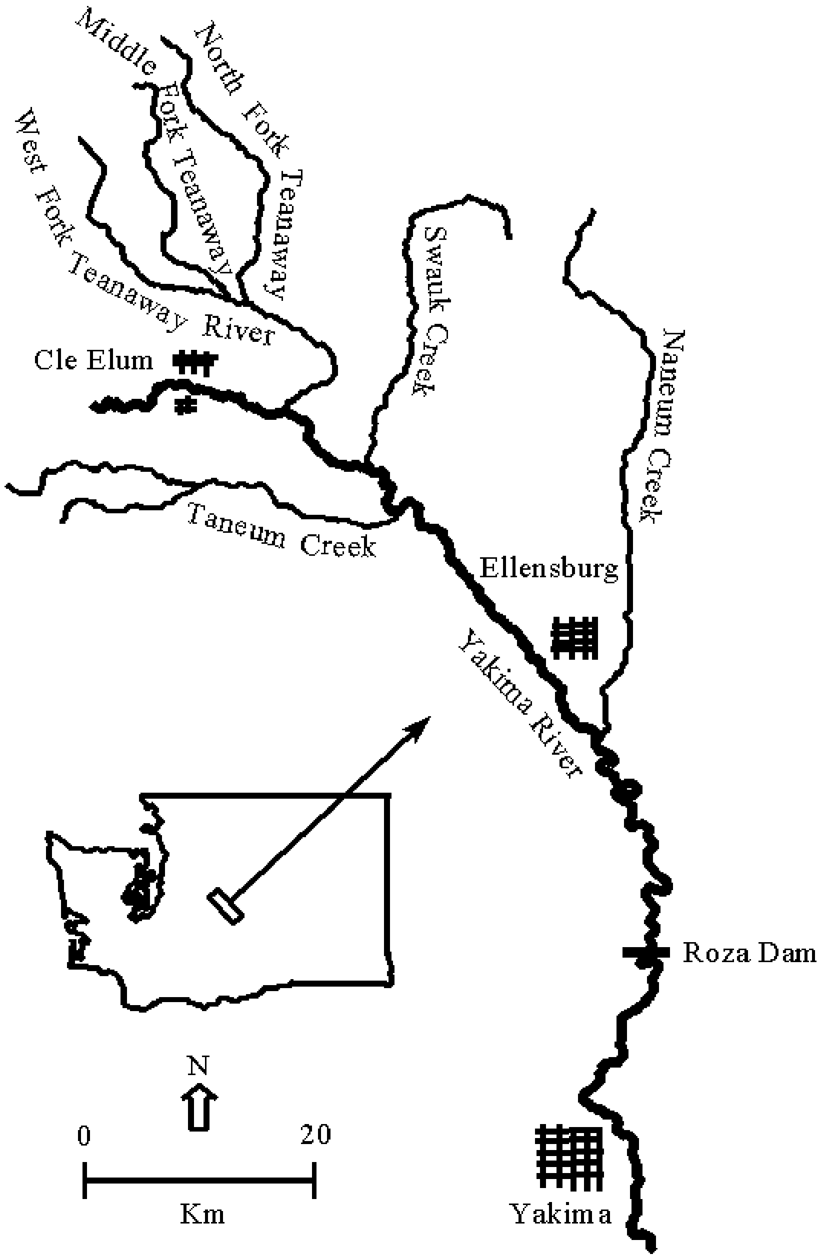

Figure 1. Map of the study area showing the Yakima River and associated tributaries where electrofishing injury research was conducted.

Table 1. Physical parameters for four tributaries and one section of the Yakima River and dates when treatment and control samples for electrofishing-injury research were collected.

Stream	Length	Aspect (km)	Water temp. (°C)	Conductivity (*mS*/cm)	Discharge (m^3/s)	Date (mo/d/y)
Naneum Creek	52	S		70		9/21/95
Swauk Creek	34	S	16	140	0.05	9/19/95
Taneum Creek	39	NE	11	150	0.28	9/20/95
W. Fork Teanaway R.	24	E	18	100	0.10	9/18/95
Yakima River	315	SE	18	130	39.80	10/18/95

previous surveys, all samples were collected in areas where we had not electrofished within the previous four years (the typical life span of *O. mykiss* in these areas (Martin and Pearsons 1994)). The electrofishing equipment and methods used in this work were the same as those used in our routine sampling in the tributaries and the Yakima River. Tributaries were sampled during daylight using a battery-powered Smith Root model 12 backpack electrofisher with a 28-cm-diameter aluminum ring anode and a 305-cm-long cable cathode. Settings for all tributary sampling were 300 V, 30 Hz pulsed DC (12.5% duty cycle). Fish collected in this study were removed on only one pass, while our typical population estimates through 100-m-long index sites require two or three passes (estimate is based on ≥ 50% depletion, Zippin 1958).

The Yakima River was sampled using a 5.1-m-long fiberglass drift boat with a stationary "Wisconsin ring" anode (102-cm-diameter with 20 46-cm long cable droppers, each 0.6-cm in diameter) suspended from the bow by a boom, and a 30-cm x 3.7-m aluminum cathode attached to the hull. The boat electrofisher was powered by a 3500 W generator and a Coffelt Mark XXII rectifier on the CPS setting (see Sharber et al. (1994) for details on the output of this setting) . Electrical output from the boat electrofisher, as registered on the meters, was 450 V and 7 A. This is the same sampling protocol that we use to conduct annual mark-recapture population estimates (methods similar to Vincent 1971) in the Yakima River. All samples collected in the Yakima River for this study were captured on one pass, while actual population estimates require two passes (one marking pass and one recapture pass a week later).

All fish were kept alive in holding vessels until we were finished sampling. Control fish were subjected to the same degree of post-capture handling as treatment fish. Fish were killed in a lethal dosage (0.5 mg/L) of tricaine methanesulfonate (MS-222), weighed to the nearest 1 g , measured to the nearest 1 mm FL, and visually examined for external electrofishing bruises (Horak and Klein 1967). Each fish was placed in a re-sealable plastic bag, which was labeled with fish length and weight, notes, and a unique number, and held on ice in a cooler. Upon returning from the field, the fish were immediately frozen until they were X-rayed.

Partially-thawed fish were X-rayed using a MinX-ray model 300 X-ray unit set at 30/60 kilovolts, at a distance of 61 cm from the plate for a 1.5 s exposure using 3M no. 1413 standard veterinary X-ray film. Each fish was X-rayed both laterally and dorso-ventrally with an X-ray marker that displayed a randomly assigned number. All fish were frozen again after being X-rayed for later necropsies.

The X-ray plates were shuffled and then skeletal images were examined for the presence of spinal injuries using an X-ray reader and an 8X magnification loupe. All examinations of X-ray plates and necropsies were done without the knowledge of whether the fish was a control or treatment fish. Spinal injuries detected on X-ray plates were rated by severity (0 = none apparent; 1 = compression of vertebrae; 2 = misalignment and compression of vertebrae; 3 = fracture of one or more vertebrae or complete separation of 2 or more vertebrae; see Reynolds (1996) for details). Number of vertebrae, location, severity, and whether the injury was visible on dorsal, lateral, or both views were recorded. All fish that showed any possibility of spinal injury on X-rays were partially thawed and necropsied by filleting musculature away from the spine on both sides and examining the spinal column and surrounding tissue to determine whether the injury appeared to have been caused by the sampling event in which the fish was captured, or whether it was an old healed injury or natural spinal abnormality (McCrimmon and Bidgood 1965; Gill and Fisk 1966; Sharber and Carothers 1988). Necropsies were also used to evaluate the extent of hemorrhaging in musculature surrounding the spinal column (McMichael 1993). When X-rays alone were inconclusive, necropsy results were the final determinant on whether we considered a fish was injured by our sampling. If a fish showed displacement, compression, or fusion of vertebrae on the X-ray but did not show any related hemorrhages when filleted, then it was assumed to be either an old injury or a natural deformity and therefore was not classified as an sampling injury caused during this experiment (McCrimmon and Bidgood 1965; Sharber and Carothers 1988). Hemorrhages were also rated by apparent severity where 0 = no hemorrhage; 1 = wounds separate from spine; 2 = wounds on spine $\leq$ width of 2 vertebrae; 3 = hemorrhages on spine > width of 2 vertebrae (Reynolds 1996). Spinal injury ratings were used unless the X-rays were equivocal, then hemorrhage ratings from necropsies were used. Injury rates are therefore a combination of the results from both techniques. However, each fish was classified as either injured by our sampling or not).

Injury Rate Estimations - Tributaries

Our first step was to calculate injury rate (percentage) at the sample scale using individual fish captured at each site (sample). To determine the rate of injuries in treatment fish that were the result of our electrofishing, we subtracted the rate of injuries observed in each control group from the rate of injuries observed in each treatment group. These differences between the injury rates in control and treatment groups are estimates of electrofishing injury rates at the sample scale.

To determine injury rates at the reach and stream scales, we used data we have collected on capture probabilities (from multiple-removal population estimates) and fraction of habitat we sample annually in our monitoring program. Specifically, we wanted to know what percentage of all *O. mykiss* within our index reaches were likely to be injured as a result of our electrofishing efforts. To estimate injury rates among fish within index reaches in tributaries we used the following equation:

$$I_t = E_t(1-(1-C)^n) \qquad [1]$$

I_t = percent of *O. mykiss* within the tributary index reach injured during a single multiple-removal population estimate, E_t = sample injury rate in tributaries, C = capture probability, and n = number of electrofishing passes. Capture probabilities were available from several index reaches in the same or adjacent tributaries to the ones sampled in this study (1990 through 1995) and were calculated by the Microfish program for multiple-removal population estimates (Van Deventer and Platts 1985) .

To provide a stream-scale injury rate estimate, we multiplied I_t by the fraction of habitat that we sample in tributaries. The fraction of habitat sampled was calculated for each tributary by dividing the stream length (km) sampled each year by the length of the tributary. Assumptions for the tributary models include: 1) all injured fish are captured in the pass they are injured (i.e., exposed but not captured fish are uninjured), 2) additional exposures do not increase E_t, 3) capture probability (of the remaining fish) is the same on all passes, and 4) fish density is the same inside the index reach as it is outside the index reach. If our assumptions are true, this produces what we believe is a high ('worst-case') estimate of electrofishing injury we might cause in tributary *O. mykiss* populations as a result of sampling. If assumptions 1 or 2 are wrong, then a low estimate would be produced. If assumptions 3 or 4 are wrong, the direction of the bias would be influenced by the way in which the assumptions were violated.

Injury Rate Estimations - Yakima River

To estimate electrofishing injury rates in the Yakima River we used different models to account for the differences in sampling methodology and size classes of *O. mykiss* collected. In index reaches that are sampled annually, we estimated annual injury rates for fish divided into two size classes. The following equation estimates the percentage of a specific size class of *O. mykiss* that are injured within an index reach of the Yakima River in one season:

$$I_s = n(E_s C_s) \qquad [2]$$

I_s = percent of *O. mykiss* of size class *s* in the main stem index reach that are injured in one season, n = the number of exposures (i.e., one mark pass and one recapture pass = 2), E_s = percent of *O. mykiss* of size class s that are injured on one electrofishing pass, and C_s = capture probability for *O. mykiss* of size *s* in each pass (as determined by mark-recapture methodology, Vincent (1971)). This equation assumes that two electrofishing passes (one mark pass and one recapture pass) double the proportion of injured *O. mykiss*. As opposed to the tributary model where fish are removed from the site after they are captured and not susceptible to electrofishing in subsequent passes, all fish captured in the Yakima River index reaches are marked and released back into the same section and are present during the subsequent electrofishing pass. This is the reason why the *EC* doubles in Yakima River reaches instead of changing the way it does in tributary index reaches (equation 1). Population estimates and capture probabilities for each size class were available from our monitoring program mark-recapture population estimates for five sections of the Yakima River (1991 through 1995). To estimate the percentage of all *O. mykiss* within an index reach in the Yakima River that would be injured by one year's electrofishing (I_m) we used the following equation:

$$I_m = \sum_{s=1}^{n} (I_s P_s) \qquad [3]$$

I_s = reach injury rate for fish of size class s; P_s = proportion of the population made up by fish of size class *s*.

To expand the injury rate estimate to the stream-scale in the Yakima River, we multiplied I_m by the fraction of habitat we sample each year. The fraction of habitat we sampled for the Yakima River was calculated by dividing the total number of km we sample annually (22 km, in five discrete sections) by the distance between Roza and Easton dams (119 km). The habitat-based expansion assumes that mean *O. mykiss* density outside the index reaches equals the mean density within the index reaches.

Statistical Procedures

Fork lengths of fish in control and treatment groups were compared using two-sample t-tests. Standard deviations for injury rates of experimental groups were calculated by applying the binomial distribution procedure for determining assumed standard deviations described by Sokal and Rohlf (1981). Significance of differences between control and treatment injury rates were determined through the use of one-sided t-tests for differences in proportions. Logistic regressions were performed independently on control and treatment groups of *O. mykiss* to plot the relationship between injury and fish length. We used a chi-square test based on the logistic regression model to test for effects of fish length group (<250 mm FL or $\geq$250 mm FL) and control-treatment group on injury on *O. mykiss* collected in the Yakima River.

Results

A total of 396 fish (95 control, 301 treatment) were X-rayed. Table 2 shows the sample sizes and lengths of *O. mykiss* collected in each tributary and in the Yakima River and the spring chinook salmon collected in the Yakima River. Mean fork length of control samples were longer in five of seven experimental groups, and were significantly longer in four of those groups (Table 2).

Table 2. Sample sizes (N), mean fork lengths in mm (FL), ranges, and standard deviations of fork length (SD) for salmonids collected in tributaries to, and the main stem of, the Yakima River in 1995. Treatment samples were collected by electrofishing, control samples were collected by angling. *P*-values for t-tests comparing lengths of control and treatment groups are also shown (asterisk denotes significant difference at $P < 0.05$). WFT = West Fork of the Teanaway River, SPC = spring chinook salmon.

Stream/ Group	Control			Treatment			
	N	Mean FL(range)	SD	N	Mean FL(range)	SD	*P*
Tributaries							
Naneum	10	205(161-250)	27	25	148(108-227)	34	<0.001*
Swauk	13	171(121-250)	34	25	145(119-207)	24	0.008*
Taneum	19	163(121-195)	20	25	156(108-207)	31	0.383
WFT	10	187(126-261)	37	25	141(95-196)	25	<0.001*
Yakima River							
O. mykiss (<250 mm)	11	232(198-245)	14	78	191(98-247)	48	0.005*
O. mykiss (≥250 mm)	23	299(251-356)	37	22	315(253-376)	39	0.144
SPC	9	111(104-120)	6	101	117(98-138)	7	0.017*

Injury rates in treatment groups were higher in six of seven experimental groups, however only in *O. mykiss* 250 mm FL and longer was this difference significant (Figure 2). Electrofishing injury rates were relatively low in tributaries, where *O. mykiss* were generally shorter than 250 mm FL (mean difference between control and treatment = 5.1%), and in smaller *O. mykiss* (mean difference = 1.2%) and juvenile spring chinook salmon (mean difference = 2.0%) collected in the

Yakima River (Figure 2). However, a relatively high proportion (mean difference = 27.7%) of the larger *O. mykiss* collected in the Yakima River were injured. Regardless of the collection method or species, injury rates were relatively low in small fish (< 250 mm FL) and higher in larger fish ($\geq$ 250 mm FL). When all *O. mykiss* samples from treatment groups were pooled to examine the relationship between fish length and incidence of injury, we found that larger *O. mykiss* ($\geq$ 250 mm FL) were injured at a higher rate than their smaller (<250 mm FL) counterparts ($\chi^2 = 7.55$, P = 0.023). Similarly, fish length was positively related to incidence of injury in treatment *O. mykiss* (Figure 3; P <0.001). In control samples, injury also increased with increasing fish length but not in a statistically significant manner (Figure 3, P = 0.265).

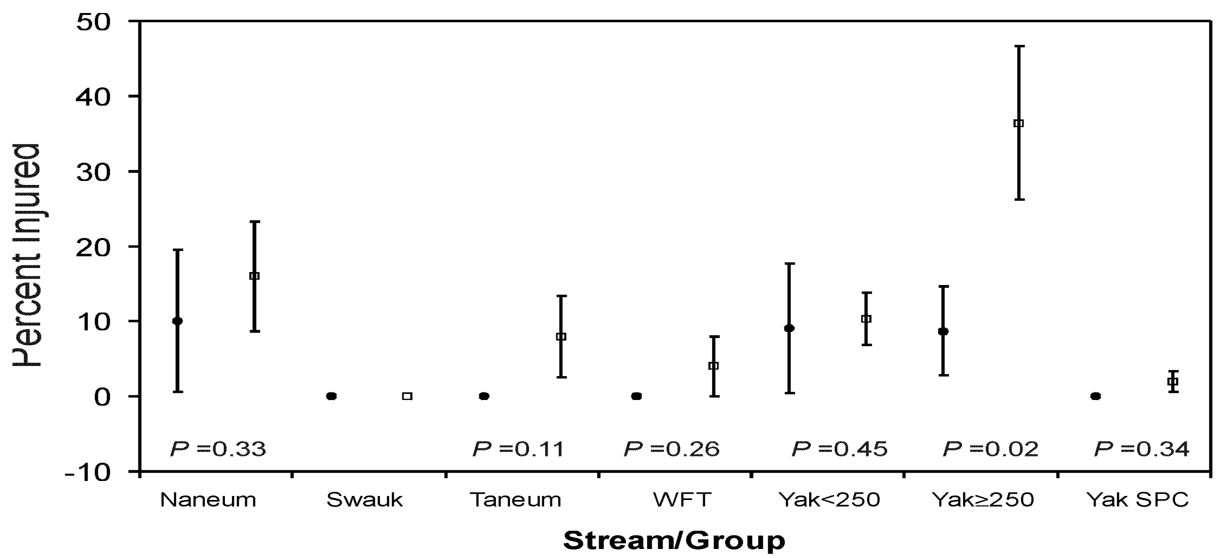

Figure 2. Percent of electrofishing-induced spinal injuries ($\pm$ SD) detected in control (closed circle) and treatment (open square) *O. mykiss* and spring chinook salmon captured by electrofishing in tributaries to, and the main stem of, the Yakima River in 1995. *P* values for t-tests between treatment and control groups are shown below each experimental pair. Yak<250 = *O. mykiss* < 250 mm FL from the Yakima River, Yak$\geq$250 = *O. mykiss* $\geq$ 250 mm FL from the Yakima River, Yak SPC = juvenile spring chinook from the Yakima River.

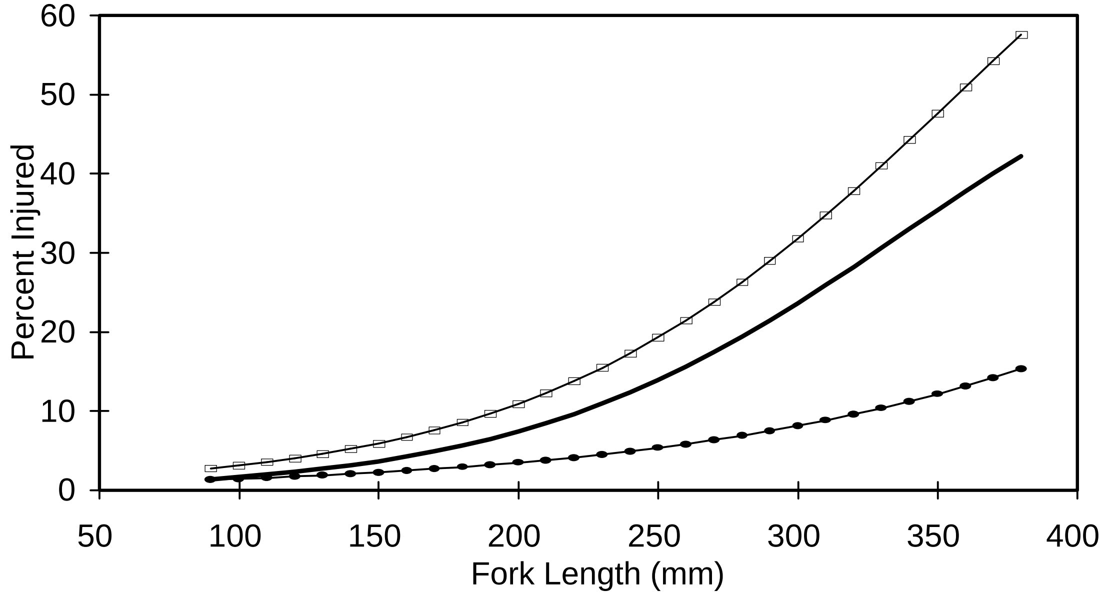

Figure 3. Percent spinal injuries of *O. mykiss* versus fork length (mm) in the Yakima River and tributaries in 1995. Treatment (open rectangle), control (closed circle), and the difference between treatment and control (heavy solid line) lines were fitted to the data by logistic regression.

Spinal injuries occurred with greater frequency in treatment fish (Figure 4A) than in control samples (Figure 4B). The majority of the spinal injuries that were detected in treatment samples were classified as class 2 (misalignment and compression of vertebrae). Similarly, hemorrhages occurred with greater frequency in fish that had been exposed to electrofishing (Figure 4C) that those that were captured by angling (Figure 4D). All of the hemorrhages that were observed in control samples were classified as class 2 (wounds on spine ≤ width of 2 vertebrae), while the most common classification in treatment samples was class 3 (wounds on spine > width of 2 vertebrae).

Spinal injuries were not observed in control fish in three of the four tributaries sampled nor in the small control sample of juvenile chinook salmon from the Yakima River. Both size groups of control *O. mykiss* collected in the Yakima River, however, showed minor spinal injuries in about 9% of each sample (Figure 3).

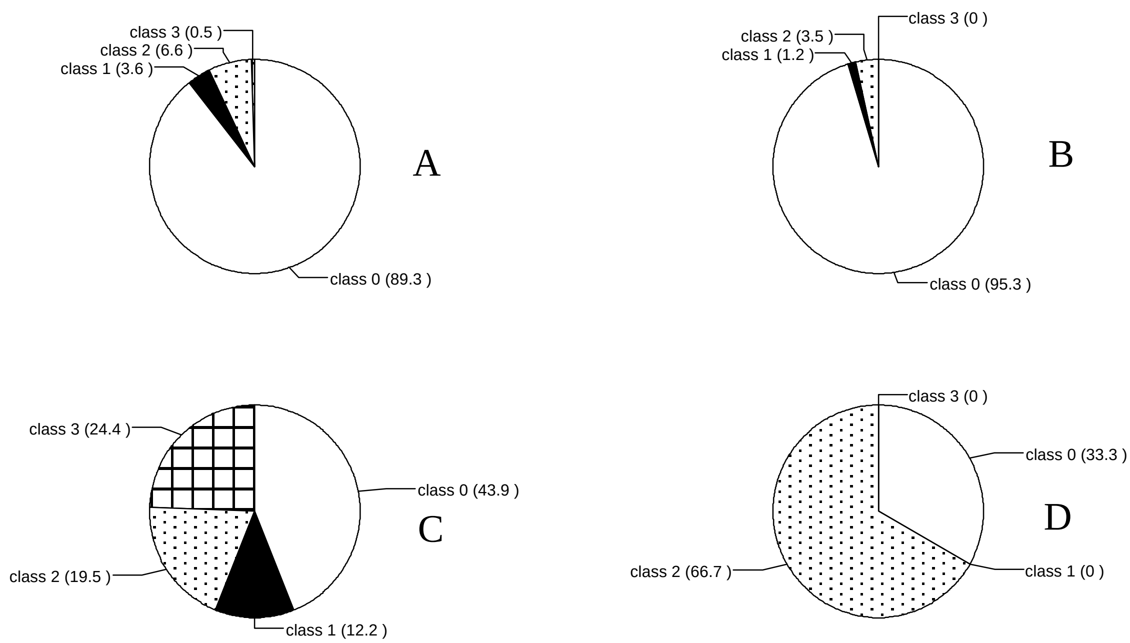

Figure 4. Percent frequency (in parentheses) of different class ratings of spinal skeletal injuries in treatment (A) and control (B) *O. mykiss* in the Yakima River and its tributaries. Also shown, class ratings (percent) of hemorrhages observed in treatment (C) and control (D) samples. Injury ratings; 0 = no injury, 1= slight injury, 2 = moderate injury, 3 = severe injury (see text for specific criteria for classifications).

Electrofishing injury rates to *O. mykiss* sampled decreased with increasing spatial scale (Table 3). In tributaries, an average of 5% of the *O. mykiss* sampled were injured by electrofishing. We used the model to estimate the injury rate within the index reach (equation 1) by incorporating the average capture probability in multiple-removal population estimates within our 100-m-long index reaches between 1990 and 1995 (68.7%; Washington Department of Fish and Wildlife (WDFW), unpublished data). Based on this capture probability data, predicted annual electrofishing injury rates were 4.6% for two-pass estimates, and 4.9% for three pass estimates. Further expansions of tributary data are based on three-pass estimates. Estimated annual electrofishing injury rate at the tributary scale for three pass estimates was 0.1% (Table 3), because only 1.1% of the tributary habitat was electrofished.

In the Yakima River, the only significant difference between control and treatment samples was in the group of larger *O. mykiss*. However, the differences between control and treatment samples of fish collected in tributaries and the smaller *O. mykiss* from the Yakima River were consistent. In the Yakima River, 1.2% of the small *O. mykiss* sampled were injured by our single electrofishing pass. Our capture probability on small *O. mykiss* averaged 18.1% in the same area where our fish were collected. Using equation 2, we calculated the index reach electrofishing injury rate for smaller trout to be 0.4% as a result of two electrofishing passes per year. Higher individual injury rate in larger *O. mykiss* ($\geq$ 250 mm FL) contributed to higher electrofishing injury estimates at the index reach scale. These larger individuals were injured at a rate of 27.7% following a single electrofishing pass and our capture probability (from annual mark-recapture population estimates in five sections) for fish of that size in that area has averaged 20.7% between 1990 and 1995. Therefore, the estimated annual injury on these larger fish within the index reach was 11.2% (Table 3). To estimate the percentage of all *O. mykiss* within a Yakima River index section that would be injured in one season, we used equation 3 and multiplied the size-specific injury estimates by their respective proportions in the population estimate (small, 59.7%; large, 40.3%). The result is an estimated injury rate of all *O. mykiss* within an index reach of 4.9%.

Because we do not conduct population estimates on juvenile spring chinook salmon, we do not have estimated capture probability data to expand individual injury rates to index reach and stream scales. However, visual estimates of juvenile spring chinook salmon abundance are higher than those for small *O. mykiss* in the Yakima River and injury rates for the two species are similar. If we assume similar capture probabilities for the two species, it suggests that reach and stream scale injury rates are equal to or lower than those for small *O. mykiss* in the Yakima River.

Table 3. Annual electrofishing injury rate projections for *O. mykiss* in tributaries to, and the main stem of, the Yakima River. Location, mean length (mm FL), sample injury rates (Sample), estimated injury rates within index reaches (Reach), and estimated injury at the stream scale (Stream) are shown. Tributary estimates are based on a three-pass multiple-removal sampling protocol.

Location	Size class	Sample(%)	Reach(%)	Stream(%)
Tributaries	all (148 mm)	5.1	4.9	0.1
Yakima River	small (191 mm)	1.2	0.7	0.1
Yakima River	large (315 mm)	27.7	11.2	2.1
Yakima River	all (218 mm)	11.9	4.9	0.9

Discussion

Expanding sample electrofishing injury rate estimates to the reach and stream scales provides a useful context for evaluating potential magnitudes of sampler-induced effects. High injury rates at the reach scale may affect long-term monitoring data and any subsequent management decisions that might rely upon those data. Stream-scale (population) injury estimates would typically be more important from a conservation/stewardship perspective. Electrofishing injury rates, as calculated using our models, decreased as spatial scale increased.

In the upper Yakima River basin, wild steelhead are very rare, while wild resident rainbow trout populations are relatively healthy. Wild steelhead juveniles are not visually distinguishable from resident rainbow trout prior to the smolt stage, which typically occurs before the fish reaches about 250 mm in length (Peven et al. 1994). Therefore, in order to minimize our sampling effects on the critically low steelhead population, we must have a very low effect on all *O. mykiss* less than about 250 mm in length.

Based on this study, injury due to our routine sampling on the small size group of *O. mykiss* is low. In tributaries, 5.1% of the individuals we captured in this size group were injured as a result of electrofishing. After accounting for multiple passes and capture probabilities, estimates of electrofishing injury rates within 100-m-long index reaches in tributaries was 4.9% annually. The tributary stream-scale electrofishing injury effect on smaller *O. mykiss* in the Yakima River was also very low (0.1%). The electrofishing injury rate among larger trout was much higher than it was for smaller fish, and resulted in an estimated annual stream-scale injury of 2.1 %. However, when we accounted for the proportions of all *O. mykiss* in Yakima River index sections that were above and below the 250 mm FL size, we predicted an overall stream-scale injury rate for all *O. mykiss* of 0.9%. Even if we assume all of the larger rainbow trout that were injured (2.1% at the stream scale) died each year, it would still be only about one-thirtieth of the average estimated annual mortality rate for these fish in that river reach between 1991 and 1995 (61.5% average annual mortality, WDFW, unpublished data). Electrofishing injury studies that have examined delayed mortality have typically concluded that even in cases where injury rates were high (upwards of 50%), short-term delayed mortality rates have been low (McMichael 1993; Habera et al. 1996) or have not significantly affected long-term survival (Dalbey et al. 1996). Even though injury rates were very low at the stream or population scale, effects of injuries on larger rainbow trout within index reaches needs to be considered when designing long-term monitoring programs. Cumulative electrofishing injury rates could be relatively high for longer-lived and larger salmonids in sites that are sampled multiple times during the fish's life-span and in tributaries where a high proportion of the fish present are captured. For example, large, long-lived rainbow trout living within an index reach that is sampled annually have an increasing probability of being injured by electrofishing as their size increases and the number of years they are exposed to electrofishing increases. In situations where the injury rate of these older and larger individuals is high, annual mortality is low, and the fraction of habitat sampled is large, it may be advisable to limit sampling to every other year to decrease the potential for electrofishing injuries to affect response data (e.g., growth) for research or management (Gatz et al. 1986; Thompson et al. 1997).

It is desirable to determine the cumulative effects (multiple years) of electrofishing injuries so that total electrofishing effects could be evaluated. Determining cumulative electrofishing injury rates in annually-sampled reaches is complex. Without a great amount of data, many assumptions must be made. To calculate cumulative injury rates, one would ideally have age-specific injury rates, and annual survival rates and population estimates. Accounting for the multiple years fish within a tributary index reach might be exposed to electrofishing (3 years), and for annual mortality of all fish (assumed to be 60% annual mortality after reaching age-1), we calculated the cumulative electrofishing injury rate within the reach to be 10.2% (in comparison to 4.9% injury in one year). Extrapolated to the stream scale, cumulative injury rate in tributaries was 0.1%, which was the same as the annual injury rate at that scale since only 1.1% of the habitat is sampled. In the Yakima

River where *O. mykiss* live longer and attain a larger size, the potential for higher cumulative impacts resulting from annual sampling increases if annual mortality is relatively low. However, in situations where annual mortality is high, the cumulative impacts would not increase much over the annual injury rates because injured fish (and uninjured fish) survival is low between sampling years (i.e., injuries do not accumulate in the population if annual survival is low).

In tributaries, where we captured high proportions of the fish present, the estimated injury rate within index reaches (4.9%) was similar to the injury rate at the sample scale (5.1%). The potential to affect large populations of smaller fish, or migratory fish that spend a year or less in the index area, is much lower due to the lower injury rates at a small size combined with the shorter residence within the index reaches. Juvenile spring chinook salmon, for example, are relatively small when they are shocked and spend only one year rearing in freshwater. Frequency of using electrofishing to sample these populations of smaller fish with shorter residence times need not be restricted to less than once per year.

Although juvenile spring chinook salmon in this study were relatively resistant to electrofishing injury (perhaps due to their small size), they appear more susceptible to mortality from other handling stresses, such as crowding and oxygen depletion in the holding vessel (Strange et al. 1978), than the *O. mykiss* we capture. Biologists must be aware of effects of collection and handling stress on fish that are sampled. Lower fish densities in holding vessels, combined with supplemental air and/or regular water changes may do more to reduce sampling impact on fish like juvenile spring chinook salmon than reducing electrofishing injury rates.

Field sampling that involves electrofishing a large fraction of the available habitat and uses injurious electrofishing methods has the potential to injure a relatively large portion of the population being sampled. For example, if a hypothetical wild resident rainbow trout population were restricted to a 10-km reach of stream, and samplers used mark-recapture methods (2 runs) to electrofish 80% of the habitat in that reach with a capture probability of 25%, and injured 25% of the fish they captured, then the stream-scale electrofishing injury resulting from such an effort would be 10% (2 x 0.8 x 0.25 x 0.25). Depending on managers' and the public's valuation of the resource, the relationship between injury and survival, the need for the data, and the frequency of sampling, a 10% injury rate at the stream scale may or may not be acceptable.

All area-based expansions of the individual injury data assume that we captured all fish that were injured. It is possible that some injured fish are not netted and therefore went undetected by the samplers. The result of this would be that our estimates of individual injury rates, as well as their subsequent area-based estimates, would underestimate actual injury rates. The degree to which this may happen would be affected by environmental variables such as turbidity, water velocity, water depth, and habitat complexity. Without using lethal sampling, such as sodium-cyanide poisoning following electrofishing, the true injury rate in fish not captured would be difficult to determine in natural streams. Collecting fish from an electrofished area by using a non-injurious method such as seining may be helpful in some reaches, but one would have to assume that injured fish were not more likely to be captured than uninjured fish. We did not attempt to estimate injury in fish that were exposed to electrofishing but not captured, but we believe that it is generally less than for captured fish because fish that are not captured seldom experience galvanonarcosis and tetany that is commonly observed in close proximity to the anode.

Other assumptions made in tributary injury rate calculations were, 1) additional exposures do not increase the sample scale injury, 2) capture probability is the same on all electrofishing passes, and 3) fish were equally distributed along the length of the stream. If assumptions 1 or 2 were violated it would have increased the projected rate minimally at the reach scale due to the high percentage of fish that are captured in small tributary multiple-removal estimates. If assumption 3 were violated, the relative densities between the sampled reach and the stream influence the direction of the bias. If the density of fish in the sampled reach was high relative to the rest of the stream, then the projected stream-scale injury rate would be underestimated. Conversely, if fish density was higher outside the sampled reach, the injury rate estimate would be too high.

The control-treatment experimental design we used provided a baseline context for the electrofishing injuries. Had we examined only fish captured by electrofishing (i.e., no controls), our estimates of electrofishing injury rates would have been substantially higher in three of our seven experimental groups. Minor injuries were observed in some of the fish in control samples we captured by angling. Hollender and Carline (1994) observed similar incidence of injuries (7%) in brook trout *Salvelinus fontinalis* they captured by angling. Other researchers have also noted spinal abnormalities in salmonids that they concluded were not the result of electrofishing (McCrimmon and Bidgood 1965; Gill and Fisk 1966).

We inadvertently collected control fish that were significantly larger than treatment fish in five of seven experimental groups. This size disparity, though small, could have caused us to underestimate injury at the sample scale (and the subsequent reach and stream scales) for some experimental groups. The effects of this would be most apparent at the sample and reach scales in tributaries due to the large portion of fish in those areas that are exposed to electrofishing.

The dorso-ventral and lateral X-rays combined with follow-up necropsies provided a good system for detecting injuries around the spinal column. In addition, this combination allowed us to make what we felt were accurate determinations on whether an injury was caused by our capture. If we had used only one X-ray view we would have underestimated injury rates. The lateral view provided the best injury detection. The majority of the injuries (53.7%) were detected on lateral view plates alone. Dorso-ventral plates alone showed 13% of the injuries, while 33.3% of the injuries were seen on both views. We may have missed some hemorrhages because we only necropsied fish that looked irregular on the X-rays. If we did miss some hemorrhages, this would have caused us to underestimate injury.

We focused our attention on two highly-valued salmonid species that are commonly associated with other fish species that may also be injured as a result of our electrofishing. Although effects on the two species we examined were low at the stream scale, injury rates in non-target species were not evaluated. We observed some mountain whitefish *Prosopium williamsoni* bleeding from the opercula when they have been exposed to electrofishing in the Yakima River. We recommend that future electrofishing injury research take a community perspective to truly evaluate electrofishing impacts to ecosystems.

Electrofishing injury rates on the fish we examined in the upper Yakima River basin appeared to be small relative to natural annual mortality rates at the stream scale and moderate at the index reach scale. The distribution, status, size structure, and life-span of the population, the proportion of the available habitat sampled, the sampling frequency, and the availability, effectiveness, and cost of alternative sampling methods, must all be balanced against the need for the data when establishing research or monitoring efforts that use electrofishing. Even though individual fish, especially larger rainbow trout, were injured at relatively high rates by electrofishing, the combination of high annual mortality and the low percentage of the population that is captured by electrofishing minimizes the effect of injury on our monitoring data.

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Chapter 6

Upper Yakima River Angler Survey
June 21 to August 13, 1995

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Abstract

Surveys were conducted in four sections of the upper Yakima River to determine angler characteristics, catch rates, and angling and non-angling use between June 21 and August 13, 1995. Anglers who fished in the study area were predominantly western Washington residents, had much prior experience fishing the Yakima River, and fished with flies. Catch rates during the study period were relatively low and were affected by sample week, angler characteristics such as experience and fishing location, and whether anglers were fishing with a hired guide. Within the range of catch rates observed during the study period, the number of trout caught per hour was higher early and late, and lower in the middle of the study period. In general, anglers with more experience fishing the Yakima River caught fish at a higher rate than those with less experience. Boating anglers had higher catch rates than anglers fishing from the bank or wading. Guided anglers, though few in number (8% of the total 242 anglers contacted), enjoyed higher catch rates than anglers fishing without a hired guide. With one exception, catch rates generally corresponded with estimated trout abundance. Most anglers felt the trout population was stable (63%), while some thought it was decreasing (11%), and still others thought it was increasing (5%). Stream flows and fishing pressure were the top two factors anglers felt were limiting the trout population. Few anglers noticed hook-scars on trout they caught, and those who did notice a hook-scar did not mind that the fish was scarred. Regarding the supplementation of spring chinook salmon in the study area, most anglers said they would accept no impact to the trout population resulting from increased natural production of salmon following supplementation. Mean acceptable impact was 8.5%. Anglers also showed little interest in participating in a sport fishery for salmon in the study area if one were to open. Angling and non-angling pressure was much higher on weekends than on weekdays, and was also higher in the Canyon sections than in the areas around Ellensburg and Thorp. Non-angling use in the Canyon sections was very high on weekends.

Introduction

The Washington Department of Wildlife began studying the rainbow trout populations in the upper Yakima basin in 1990 in response to concerns that a proposed supplementation hatchery might impact the trout fishery. Trout spawning, abundance, distribution, size structure, and movement have been examined annually since 1990 (Hindman et al. 1991; McMichael et al. 1992; Pearsons et al. 1993; Pearsons et al. 1994; Pearsons et al. 1996; Chapter 3, this report). These data have given us some insight into the biological characteristics of the trout populations but we have gathered little information on the trout fishery and angler characteristics. Angler catch rates, satisfaction, and perceptions are important measures to fishery managers. Periodic examination of these factors allows managers to monitor and evaluate effects of management actions, such as a regulation change or implementation of a supplementation program, on the people participating in the resident trout fishery.

The Yakima River above Roza Dam has become a very popular recreational fishery for resident rainbow trout in the last decade. With a change in angling regulations to catch and release only in 1990 and a surge in the popularity of flyfishing in the mid-1990's, the angling pressure on the upper Yakima River has increased dramatically. The objectives of this work were to; 1) determine the catch rates (fish caught per hour of angling, hereafter referred to as fish/hr) for anglers fishing the upper Yakima River; 2) examine relationships between catch rates and angler characteristics/environmental factors (e.g., water temperature and discharge); 3) examine angler perceptions and values regarding the trout population and its management, with emphasis on perceptions and values as they relate to YFP supplementation activities and acceptable impacts of those activities; and 4) estimate summer angling and non-angling use within the study reach.

Methods

Study Area

This survey was conducted in four sections of the upper Yakima River between Roza Dam (on State Route 821 between Yakima and Ellensburg) and the mouth of the Teanaway River (near the town of Cle Elum). The sections surveyed were defined by Hindman et al. (1991) as the Lower Canyon, Upper Canyon, Ellensburg, and Thorp sections. The boundaries for each section are as follows; Lower Canyon - from Roza Dam to Umtanum Creek; Upper Canyon - from Umtanum Creek to Ringer Road access area; Ellensburg - from Ringer Road access area to the Town Diversion Dam; and Thorp - from the Town Diversion Dam to the mouth of the Teanaway River. See McMichael et al. (1992) for a more detailed description of the study area.

Data Collection

Angler counts and on-site interviews were conducted two to three days per week from June 21 through August 13, 1995. The survey effort was systematically divided to cover approximately equal numbers of weekend days and week days in each section. Time of day was also systematically chosen to encompass the range of angling hours per day.

Anglers were asked a series of questions and their responses were recorded on an angler survey form (Table 1). The surveyor also drove the length of each section at the beginning and end of the survey period and performed counts of anglers and non-anglers in various categories and recorded the data on a pressure survey form (Table 2).

Catch rates were determined for each angler by dividing their total catch by the number of hours they had fished. Angling pressure estimates were based on an assumed angling day length of 15 hours (7:00 am to 10:00 pm PDT) and non-angling pressure estimates assumed a day length of nine hours (10:00 am to 7:00 pm PDT). Expansion factors were calculated for each survey day by dividing the number of survey hours by 15 for angling pressure and nine for non-angling pressure. Daily expansion factors were then applied to actual daily counts to estimate pressure of each group per day. Weekend days and weekdays were analyzed separately due to differences in types of pressure at those times. All pressure estimates were conducted on the lower two sections combined (Lower and Upper Canyon), and the upper two sections combined (Ellensburg and Thorp) due to similarities in the angling pressure in those reaches.

Data sets were entered, analyzed and summarized on a personal computer. Most data are presented graphically to aid in the interpretation of the results.

Table 1. Data form used to collect angler information in on-site angler interviews on the upper Yakima River, June 21 - August 13, 1995.

SECTION: LCYN UCYN EBURG THORP CELUM OTHER:__________
MILEPOST/LOCATION:___________________________
DATE:___________________
****

DEMOGRAPHICS:
RESIDENCY: EAST. WA____ WEST. WA______ or STATE_________________
NUMBER OF PEOPLE IN GROUP: 1 2 3 4 5 6
JUVENILE: 1 2 3 4 5 6
ADULT: 1 2 3 4 5 6
GUIDED: YES NO NOT SURE
TIME FISHED: START_______________ QUIT__________________
COMPLETED: YES NO
ANGLER TYPE:_______________
GEAR CODE:_________________
TARGET SPECIES:_____________________

CATCH COHORT:
CATCH BY: GROUP INDIVIDUAL
SPECIES:____________________
ORIGIN:_____________________
MARKS:______________________
NUMBER OF FISH: KEPT:________ RELEASED:___________
RELEASE CODE:_______________

QUESTIONS:
1) What is the estimated size range of the fish you have caught today? MIN:______ MAX:_________ AVG:_______

2) Have you noticed any hookscars on fish you have caught today? YES NO, If yes: Do you mind catching fish that are hookscarred? YES, NO, I dont care

3) Have you caught anything other than trout? YES NO
If yes, How many of each species?_______________________________
____

4) Are you satisfied with your catch today? YES NO

5) How many times have you fished the Yakima River? 1st 2-5 6-20 >20

6) [If they have fished the Yakima more than 6 times] Do you think the trout population is STABLE INCREASING or DECREASING in this section of the river?

7) Do you agree with the current catch-and-release regulation on this section of river? AGREE DISAGREE

8) What do you think is the main factor limiting trout production in this part of the Yakima River? I DONT KNOW FLOWS POACHING SEDIMENT NOT ENOUGH FOOD HABITAT
FISHING PRESSURE POLLUTION WATER TEMPERATURE HATCHERY STOCKING PREDATORS COMPETITION
NOTHING OTHER:_________________________

9) Assuming spring chinook supplementation occurs and results in increased natural production of adult spring chinook salmon, what maximum level of impact to the resident rainbow trout population would you be willing to accept? NONE 10% 25% 50% 75% 100%

10) Would you participate in angling for spring chinook salmon if a fishery was open in the Yakima River above Roza Dam? YES NO

Table 2. Pressure data form used to collect angling and non-angling pressure information on the upper Yakima River, June 21 - August 13, 1995.

DATE:____________

SECTION:_________________

WEATHER: CLEAR PARTLY CLOUDY CLOUDY CALM WINDY RAIN

TIME ENTERED SECTION:_____

NUMBER OF ANGLERS SEEN:_______

NUMBER OF ANGLING BOATS SEEN:__________

NUMBER OF BOAT TRAILERS AT TAKE-OUTS/PUT-INS:________

NUMBER OF NON-ANGLING BOATS SEEN(INCLUDE TUBES):________

TIME LEFT SECTION:_______

NUMBER OF SAME ANGLERS SEEN:_______

NUMBER OF NEW ANGLERS SEEN:________

TOTAL NUMBER OF ANGLERS SEEN:______

NUMBER OF SAME BOATS SEEN:__________

NUMBER OF NEW BOATS SEEN:___________

TOTAL NUMBER OF BOATS SEEN:__________

NUMBER OF BOAT SAME TRAILERS AT TAKE-OUTS/PUT-INS:________

NUMBER OF NEW BOAT TRAILERS AT TAKE-OUTS/PUT-INS:________

TOTAL NUMBER OF BOAT TRAILERS AT TAKE-OUTS/PUT-INS:________

Results

Angler Characteristics

Anglers who fished the upper Yakima River were generally from the western half of Washington State, used fly fishing equipment, and had fished the Yakima River many times. Of the 242 anglers interviewed 59% were from the western portion of Washington State, while 36% were from eastern Washington, and the remaining 5% were from outside of Washington State. Angling regulations precluded the use of bait, but allowed use of artificial lures with single barbless hooks. The majority of anglers contacted were using flyfishing equipment (94%) while the other six percent were using artificial lures on spinning equipment. Anglers who had previously fished the Yakima River at least 20 times comprised 37% of the anglers interviewed (Figure 1). Only 16% of the anglers contacted were fishing the Yakima River for the first time. Over a third (35%) of the anglers on the Yakima River for the first time were fishing with a guide, while only seven percent of all anglers were being guided. More anglers fished from boats (60%) than from the bank or wading (40%). Most anglers were contacted in the Lower Canyon section (57%), followed, in decreasing order, by the Upper Canyon (21%), Ellensburg (17%), and Thorp (5%) sections.

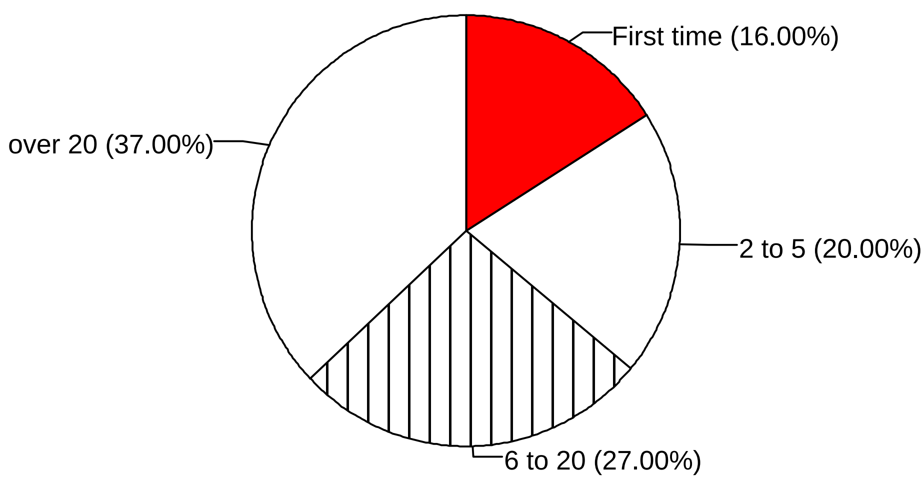

Figure 1. Experience level (number of times anglers had previously fished the Yakima River) of anglers contacted on the upper Yakima River between June 21 and August 13, 1995. A total of 242 anglers were contacted.

Catch Data

Catch rates (fish/hr) during the summer of 1995 were relatively low and were affected by sample week, angler characteristics, water temperature, and section of river fished. The overall mean catch rate between June 21 and August 13, 1995, was 0.154 fish/hr, compared to a mean catch rate of 0.304 fish/hr between April 1 and May 30, 1993 (Tonseth 1993). Catch rates decreased from June 21 to mid-July, and then increased toward the end of the sample period (Figure 2). Angler experience on the Yakima River affected catch rates. With the exception of anglers fishing on the Yakima River for their first time, the higher the number of previous times an angler had fished the Yakima River, the greater their respective catch rates were (Figure 3, R = 0.117, P = 0.070). Anglers fishing from boats also experienced higher catch rates (0.187/h) than those anglers fishing from the bank or wading (0.107/h)(t = -2.19, df = 240, P = 0.029). Guided anglers caught fish at a significantly higher rate (0.294/h) than anglers who did not fish with a guide (0.143/h)(t = -2.20, df = 240, P = 0.029).

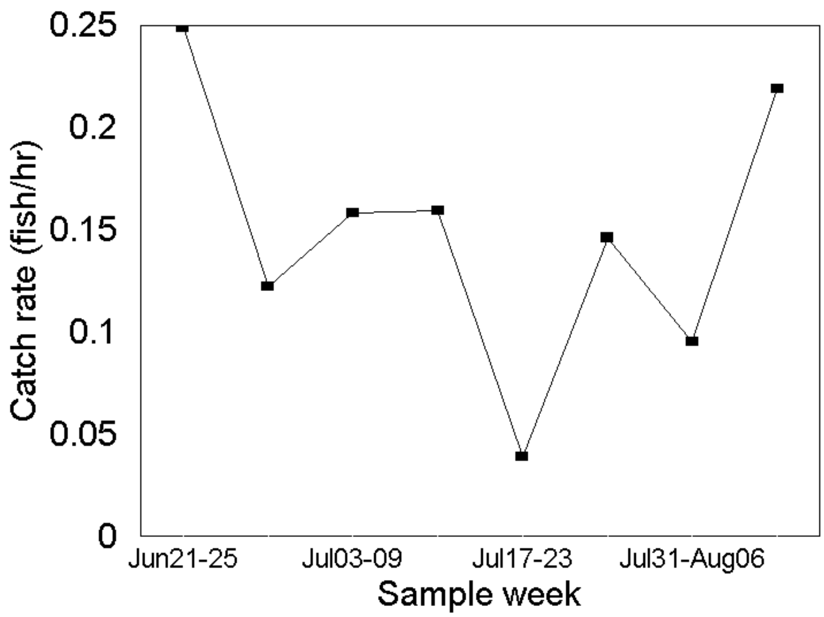

Figure 2. Overall mean catch rate (fish/hr) versus sample week for anglers fishing the upper Yakima River between June 21 and August 13, 1996. A total of 242 anglers were contacted.

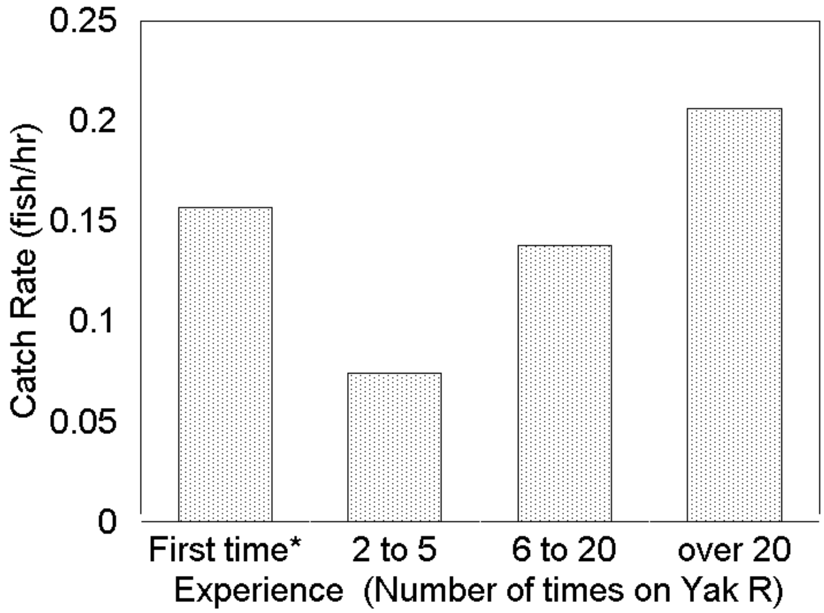

Figure 3. Mean catch rate (fish/hr) versus angler experience (number of times they had previously fished the Yakima River) for 242 anglers contacted between June 21 and August 13, 1995. Note: 35% of the first time anglers were fishing with a guide.

Catch rates generally paralleled estimated rainbow trout population abundance (number of trout/km, from fall 1995 estimates, Chapter 3, this report), with the exception of the Upper Canyon section where catch rates were very low (Figure 4). Catch rate was negatively correlated with water temperature ($R = -0.192$, $df = 241$, $P = 0.003$). In other words, anglers caught fish at a higher rate when water temperatures were lower (the range during the period was 8.2 to 15.7 C, mean = 13.6 C). Water temperature was also likely correlated with discharge. Other variables such as cloud cover and wind were not measured but may have affected catch rates.

Of the anglers who caught at least one trout ($N = 86$), those who said they were not satisfied with their catch had significantly lower catch rates (mean = 0.27 fish/hr) than those who responded with a satisfactory rating (mean = 0.51 fish/hr)($t = -3.62$, $df = 84$, $P < 0.001$).

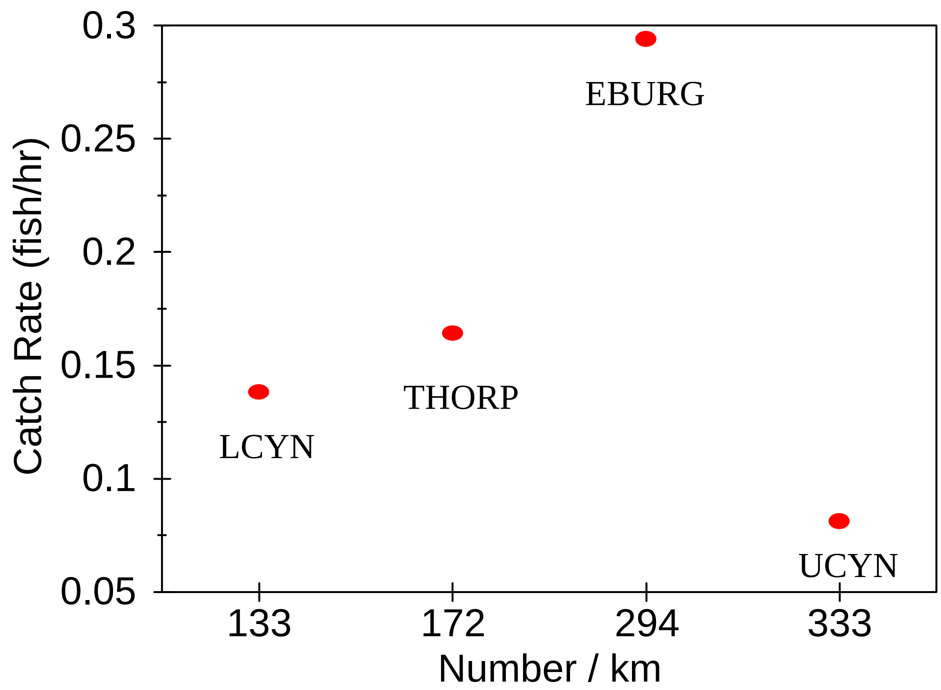

Figure 4. Mean catch rate (fish/hr) between June 21 and August 13, 1995, versus estimated number of rainbow trout/km (from fall, 1995, WDFW estimates, Chapter 3, this report) in four sample sections of the upper Yakima River.

Angler Perception

Anglers' perception of the Yakima River trout population and management provided much insight into the types of anglers fishing the upper Yakima River and their interests. Of anglers who had fished the Yakima River at least six times, most thought the trout population in the upper Yakima River was stable (63%), while 11% thought it was decreasing and 5% thought it was increasing, and 21% did not know. When asked what they thought was limiting the trout population in the study area, many anglers felt that river flows, fishing pressure, water temperature, and sedimentation were important factors (Figure 5). Most anglers supported the current catch and release regulations on the upper Yakima River (97% in favor; 3% opposed). Only eight percent of the anglers who had caught at least one fish reported noticing a hook-scar on at least one fish. Of the anglers who noticed a hook-scar, 86% reported that they did not mind that the fish was scarred, 14% had no opinion about it, and none minded that the fish was scarred.

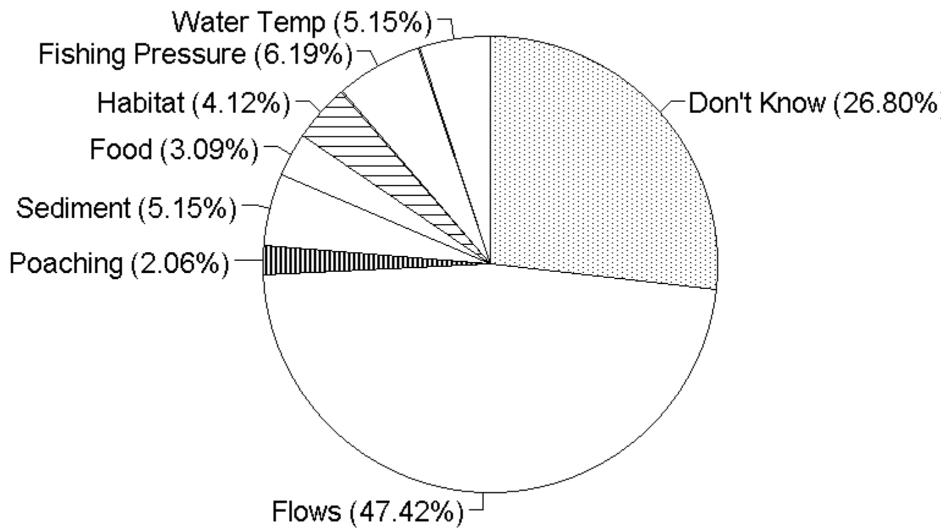

Figure 5. Percentages of anglers' responses to the question of what they thought was currently limiting the trout population in the upper Yakima River. Only anglers who had fished the Yakima River at least six times were asked (N = 155).

When asked what level of impact to the trout population they would accept due to spring chinook salmon supplementation in the upper Yakima basin, over half of the respondents said they would accept no impact (Figure 6). A mean acceptable impact, calculated from curve-fitting of the data, was 8.5%. There was a general trend toward less acceptance as the hypothetical impact increased. Finally, when anglers were asked whether they would participate in a sport fishery for spring chinook salmon in the upper Yakima River if supplementation of that stock were successful, most (58%) indicated they would not (24% would, and 18% were undecided).

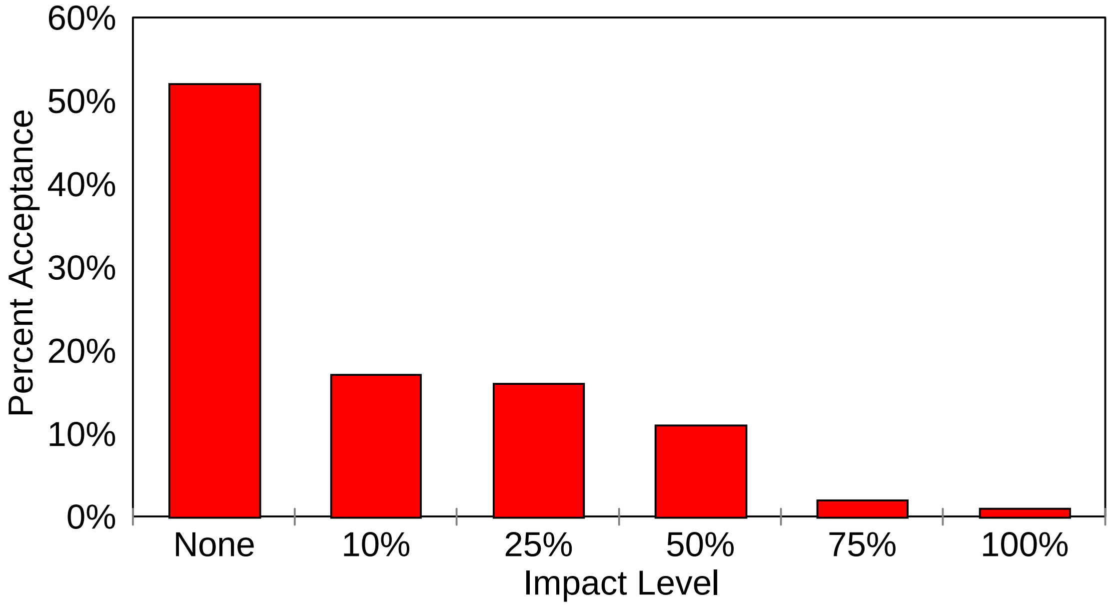

Figure 6. The percentage of anglers contacted on the upper Yakima River between June 21 and August 13, 1995, that would accept various levels of impact to the trout population as a result of spring chinook salmon supplementation.

Angling and Non-angling Use

Estimated angling and non-angling use in the Yakima Canyon sections was much higher than in the Ellensburg and Thorp sections and pressure in all areas was much higher on weekends than on weekdays. Between June 21 and August 13, 1995, the estimated number of anglers per day was 80 (68% of the total) on weekend days and 53 (32%) on weekdays in the Canyon sections (Figure 7). An estimated 40 angling boats used the Canyon sections each weekend day, while an estimated 23 angling boats floated those waters each weekday (Figure 7). Non-angling floaters were much more abundant in this area on weekends. An estimated 193 non-angling boats (includes tubes) used the Canyon sections each weekend day, while only 18 were estimated to have been present each weekday (Figure 7).

Similar relationships between day of week and use were observed in the Ellensburg and Thorp sections combined, where there were more users per day on weekends than on weekdays. Overall pressure was also much lower on these upper two sections. An estimated 34 anglers fished these sections each weekend day, while 12 were estimated to have fished there each weekday (Figure 8). Nine angling boats and 42 non-angling boats were estimated to have used this area each weekend day, while four angling and two non-angling boats were estimated for each weekday (Figure 8).

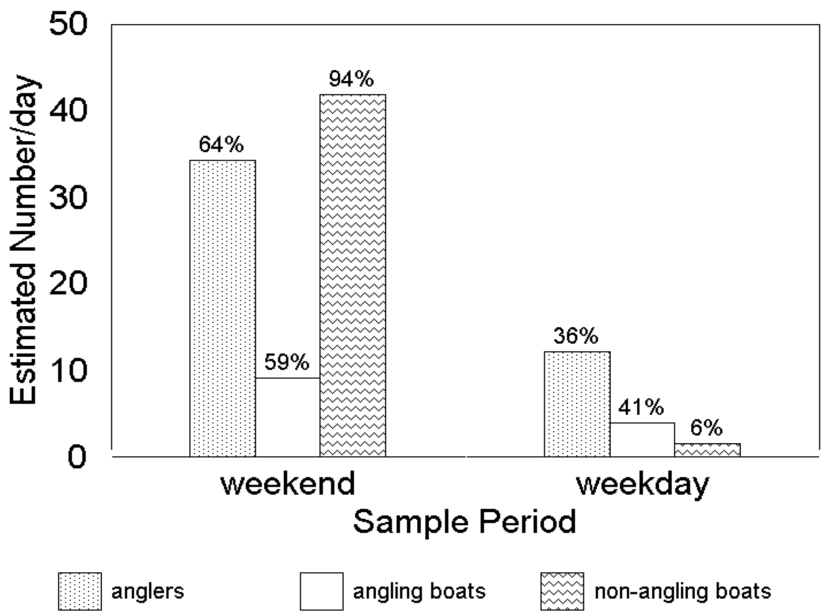

Figure 7. Estimated angler, angling boat, and non-angling boat (includes tubes) numbers per day on weekends and weekdays on the Canyon sections (Upper and Lower combined) of the upper Yakima River between June 21 and August 13, 1995**.** Percentages for weekend/weekday breakdowns are presented for each group above the bars.

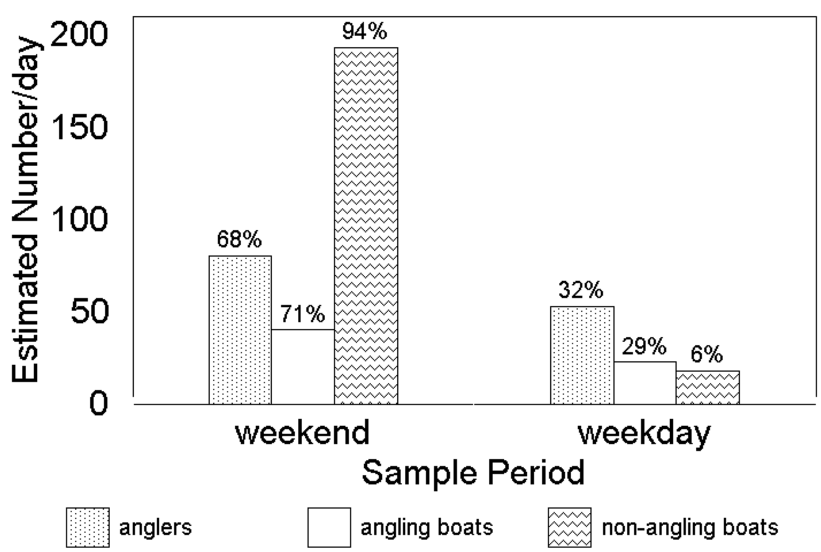

Figure 8. Estimated angler, angling boat, and non-angling boat (includes tubes) numbers per day on weekends and weekdays on the Ellensburg and Thorp sections (combined) of the upper Yakima River between June 21 and August 13, 1995**.** Percentages for weekend/weekday breakdowns are presented for each group above the bars.

Discussion

Most of the anglers who were contacted on the upper Yakima River were from western Washington, were using flyfishing equipment, and had fished the Yakima River several times. Tonseth (1993) reported similar residency figures of 55.7% from western Washington and 39.6% from eastern Washington for the anglers he contacted in the spring of 1993.

Catch rates on the upper Yakima River were relatively low during the study period. With a mean catch rate of 0.154 fish/hr, it took anglers an average of about 6.5 hours to catch one fish. Mean catch rates in April and May, 1993, reported for the Canyon sections of the Yakima River by Tonseth (1993) were about twice as high as those seen in this study. For comparison, mean catch rates for rainbow trout reported by McMichael and Kaya (1991) for the upper Madison River, Montana, were 0.79 and 0.99 fish/hr in the summers of 1987 and 1988, respectively. Catch rates in this study may have been lower than typical catch rates on the Yakima River during early spring and late fall. This study was conducted between June 21 and August 13, during the irrigation season. Catch rates are generally higher during periods of low flow that occur prior to irrigation flows in the spring, and again after the irrigation season in the upper Yakima River ends in early September.

Anglers who had more experience on the Yakima River generally caught fish at a higher rate than their counterparts with less experience. One exception to this was anglers who were experiencing their first trip on the Yakima River. First time anglers caught an average of about 0.16 trout/hr, which was higher than mean catch rates for anglers who had previously fished the Yakima River between 2 and 20 times, but 35% of these first time anglers were fishing with a guide. Higher mean catch rates for guided anglers explain this diversion from the general trend of increasing catch rates with increasing experience levels.

Boat anglers had mean catch rates that were significantly higher than anglers fishing from the bank or wading. High river discharge during the summer irrigation season seems to decrease the effectiveness of most bank/wading anglers and favor anglers casting toward banks from boats.

Hiring a guide appeared to be a worthwhile way to increase catch rates on the upper Yakima River. Only seven percent of the anglers contacted were fishing with a guide, but they caught fish at a significantly higher rate than anglers not fishing with a guide. These findings are contrary to those reported by McMichael and Kaya (1991) for the Madison River, Montana, where catch rates for guided and non-guided anglers were not different.

Interestingly, it appeared that anglers caught fish at a higher rate in sections where the estimated trout abundance was higher. However, catch rates in the Upper Canyon section were very low, and the estimated trout abundance in that section was relatively high. Reasons for this deviation from the apparent relationship are unclear.

Anglers who had some prior experience on the Yakima River had a fairly good feel for the status of the trout population as well as some factors that might be limiting it. Estimated overall rainbow trout abundance in the upper Yakima River has been fairly stable, with a slight decrease in the past two years (Chapter 3, this report). Flow was one of the primary factors anglers thought might be limiting the trout population in the study sections. It is reasonable to speculate that high velocities associated with the artificially high flows during the summer irrigation season may reduce survival of age-0 trout that emerge from the gravel during the high flow period (Pearsons et al. 1996). High flows may also reduce the amount of low velocity rearing areas which are critical to the survival of trout fry during their first year of life.

It is surprising that only eight percent of the anglers who caught fish reported noticing any hook-scars on the fish they caught, when our data shows that 7 to 36% of the trout in the Yakima River show a scar indicative of a past hooking event (WDFW, unpublished data). Also surprising was the fact that none of the anglers who noticed a hook-scar on a trout minded that the trout was scarred.

Anglers contacted during the study were generally not in support of spring chinook salmon supplementation in the upper Yakima River. To assist managers in setting objectives for the trout population level in the upper Yakima River as it relates to the supplementation of spring chinook salmon, anglers were asked what level of impact they would accept on the resident trout population. Over half of the anglers said they would accept no impact, while 12 to 15% said they would accept up to a 50% impact. Less than a fourth of the anglers contacted said they would participate in a spring chinook salmon sport fishery in the upper Yakima River, further supporting the general resistance to salmon supplementation.

There has been a tremendous increase in angling pressure on the upper Yakima River in the past six years. However, with no angling pressure data from previous years it is difficult to quantify this increase. Estimated numbers of users per day was very high on weekends for all groups, and lower but still considerable on weekdays, especially for anglers. Angling and non-angling pressure was more intense in the Yakima Canyon sections than in the Ellensburg and Thorp sections. This difference may be largely due to access limitations in the upper sections. It is interesting that trout population estimates have generally declined in the Lower Canyon section over the past five years, while they appear to have remained stable or increased in the upper sections where pressure is lower (Chapter 3, this report). It is possible that some of the decrease in trout abundance may be related to angler-induced mortality, even though the regulations have not allowed harvest of trout in this area since 1989. Results from a study conducted on the New Haven River, Vermont, indicated that catch and release regulations did not meet the objective of increasing trout abundance and size (Anon. 1989). Ferguson and Tufts (1992) reported that the combination of exercise and air exposure (similar to that experienced by a fish that is hooked, played, held out of the water while the hook is removed, and possibly held up for a photo), together reduced survival of rainbow trout substantially. In their research, rainbow trout that were held out of water for 30 and 60 seconds following exhaustive exercise showed 38% and 72% mortality rates within 12 hours, respectively (Ferguson and Tufts 1992). Therefore, it is not unreasonable to speculate that decreases in direct angling harvest mortality may have been off-set, or even out-weighed, by increases in unintentional mortality associated with catch and release fishing in light of the recent rapid increase in angling pressure.

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Chapter 7

Roza Dam Smolt Trapping Feasibility Study: Fall 1995-Spring 1996

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Abstract

We operated rotary screw fish traps and the juvenile bypass trap during portions of the fall of 1995, the winter of 1995-96, and the spring of 1996 at Roza Dam to evaluate the feasibility of using mobile smolt trapping equipment at that site to monitor spring chinook salmon *Onchorynchus tshawytscha* and steelhead *O. mykiss* outmigration from the upper Yakima River basin. Environmental conditions during all seasons were often prohibitive for operating mobile gear, and were often also restrictive relative to the use of the juvenile bypass trap. Large numbers of fall-winter migrant spring chinook salmon were captured with both gear types when conditions allowed the operation of those gear types during the fall and winter. Very few spring chinook salmon smolts were captured during the spring season. Catch rates in the juvenile bypass trap were generally very low when less than about 15% of the river flow was diverted into Roza Canal. Environmental and operational conditions that might limit the usefulness of both gear types were identified. High flow generally reduces or eliminates the effectiveness of both the screw traps and the juvenile bypass trap. Weather and operational issues, and their subsequent effects on human safety, greatly limit the usefulness of the juvenile bypass trap in its current configuration. The combination of mobile (screw traps) and fixed (juvenile bypass trap) smolt sampling equipment at Roza Dam does not appear to be sufficient for determining the production of upper Yakima River spring chinook salmon smolts and/or fall-winter migrants. With some modification to existing structures, the juvenile bypass facility could be used as a collection point for marking salmon smolts during the spring season for relative survival comparisons between different groups of fish. Based on our data, and that of others who have worked at Roza Dam, it appears that the dominant life-history form of upper Yakima River spring chinook salmon exhibit a fall-winter migration from the Yakima River above Roza Dam. Many other species of fish were captured at Roza Dam, especially during the fall season when the reservoir above Roza Dam was lowered. The spring chinook salmon smolts captured in the screw traps were generally shorter than their counterparts captured in the juvenile bypass trap, though the difference was not statistically significant. The salmon captured in the bypass trap did, however, have significantly lower condition factors than those captured in screw traps. There was a significant positive relationship between the entrainment of flow into Roza Canal and the total number of salmonids captured in the juvenile bypass trap.

Introduction

The Yakima Fisheries Project (YFP) Monitoring Implementation and Planning Team (MIPT) identified Roza Dam as a potential monitoring site for spring chinook salmon *Onchorynchus tshawytscha* smolts, both hatchery-origin and naturally-produced, emigrating from the upper Yakima River basin. Previous attempts to quantify anadromous salmonid smolts at this facility had not been successful in developing smolt passage estimates that might be useful in a monitoring program (Fast et al. 1991; D. Seiler, WDFW, personal communication). Earlier efforts, utilizing both the fixed juvenile bypass trap (JUV) as well as a mobile screw trap, during the spring outmigration period resulted in conclusions that the two gear types were biased and did not capture representative portions of the population migrating past that point in the river. Seiler (WDFW, personal communication) believed that the fish captured in the JUV were less fit, or stressed, individuals, while the screw trap captured fish more representative of the population passing that point. This conclusion was based on the finding that the JUV captured a higher proportion of marked fish (presumably less fit due to handling stress) than did the screw trap. In addition, no canal flow entrainment-fish capture relationships were established to allow for the determination of trap capture efficiency information upon which production estimates might be based.

In the fall, 1995, winter, 1995-96, and spring, 1996, we attempted to determine the feasibility of using mobile trapping equipment to monitor upper Yakima River spring chinook salmon and steelhead *O. mykiss* smolts and presmolts passing Roza Dam. Our primary objective was to install and operate two rotary screw traps below Roza Dam and operate them during the fall-winter spring chinook salmon migration, as well as during the spring smolt migrations of both salmon and steelhead. Specifically, we wanted to determine under what environmental conditions (e.g.s discharge, water temperature, debris loads) we could operate the screw traps and the JUV trap. In addition, we operated the JUV trap, and compiled and analyzed juvenile salmonid trapping data that had been collected at Roza Dam by others in 1988- 1991.

Methods

Study Area

The Yakima River, a tributary to the Columbia River, drains the eastern portion of the Cascade Mountains in central Washington State (see McMichael et al. 1992 for more details). Roza Dam is an irrigation dam located on the main stem Yakima River between the cities of Yakima and Ellensburg (Figure 1). The dam has two spill gates, an adult salmonid ladder with entrances on both banks, and a large juvenile screening/bypass facility that was updated in 1987 (Hosey & Associates and Fish Management Consultants 1989). The Roza Canal typically diverts about 15 to 70% (up to 63 m^3/s) of the water passing that point between late winter and the following fall. The juvenile evaluation facility consists of a 10 m-deep cement pit with a vertical ladder into it. Juvenile fish must be captured from the bypass by placing a 6.7 m-long inclined plane into the bypass to direct fish and a reduced volume of water into a trap and holding box. No permanent fish capture devices are installed, requiring samplers to collect and/or fabricate their own terminal trapping and holding devices.

The screw traps were installed approximately 500 m downstream of Roza Dam at the upstream margin of a large scour pool. The JUV trap was located in the juvenile bypass system and was accessed by climbing down a 10 m vertical ladder into a concrete lined box located adjacent to the bypass just downstream of the dewatering portion of the system. The bypass is dewatered either by a gravity return to a diffuser box on the river bank, or by pumping the excess water into the canal.

Equipment

The two screw traps we used had rotary cones that were 1.52 m in diameter and were constructed by E.G. Solutions, Inc. (Corvallis, Oregon). Each screw trap was constructed of aluminum and included a wheel-driven drum intended to make the live box self-cleaning. Screw traps were assembled on Roza Dam and lowered into the Yakima River by crane. There was no road access to the trapping area, thus the traps were floated downstream into position 500 m below the dam. Due to a lack of sufficient cable attachment points on both banks at the trapping site, we used a Hilti rock drill to drill several holes in the basalt cliffs on the west bank where we glued (using Hilti C 100 epoxy) 23 cm-long x 2.5 cm diameter galvanized steel shoulder bolts into the cliff. On the east bank, where there were no cliffs or trees, we had a helicopter fly in a 3 m tall tripod constructed with 10 cm diameter steel pipe. We then poured 450 kg (dry weight) of concrete into 0.6 m^3 plywood-formed footings on each of the three legs of the tripod. In addition, we flew in four concrete ecology blocks (1725 kg each) for lateral and terminal supports for the main highline cable. A 1.3 cm diameter highline cable spanned the entire river channel and went through a pipe welded to the top of the tripod and was attached to two ecology blocks on the east bank. On the west bank the highline cable was run through three 13 cm diameter snatch block pulleys (which were in turn attached to bolts in the rock), and then run upstream 50 m and attached to a 9 m^3 rock-filled gabion. A person then went out on the highline and attached back-to-back snatch block pulleys (one set of two for each trap) and secured them on both sides by placing cable clamps on the highline. Cables from screw traps were then run through the remaining open pulley and then run to their respective banks where they were attached to the same anchor points as the highline cable. This setup allowed for upstream/downstream repositioning of the traps from the bank, but did not allow for lateral adjustments of trap position. To move traps laterally a person had to go out on the highline cable and reposition the cable clamps manually.

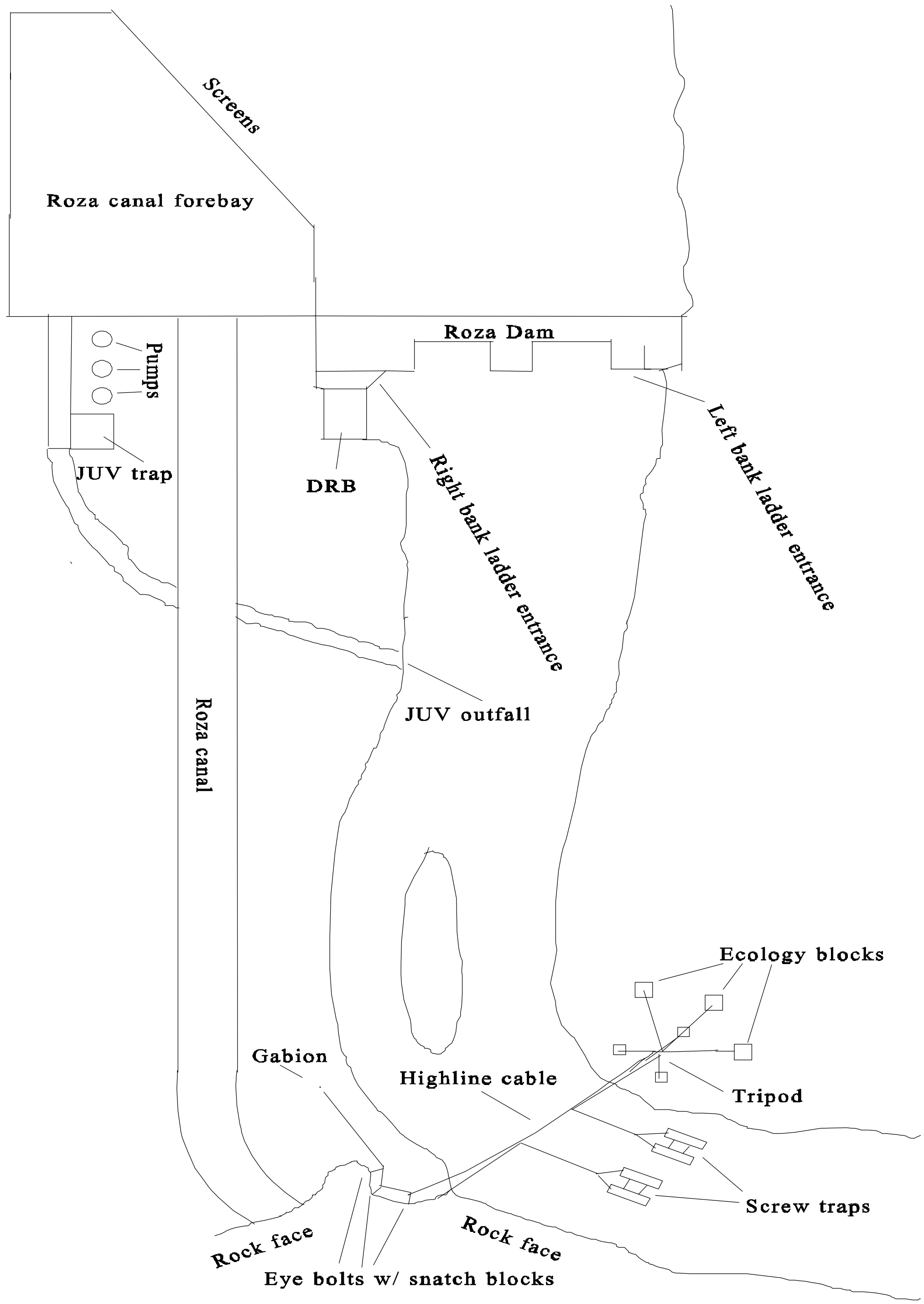

Figure 1. Map of the study area, showing Roza Dam juvenile fish trap (JUV) and screw trap locations on the Yakima River.

The trap used in the juvenile bypass was located at the downstream edge of a dewatering ramp. The discharge in the bypass at the trap location was typically about 0.7 m^3/s. The dewatering ramp passed approximately 0.6 m^3/s through the ramp, and the fish and the remaining 0.1 m^3/s of water passed through the trap. The trap, borrowed from the National Marine Fisheries Service, was constructed of aluminum and had a u-shaped throat that narrowed to an opening that was placed above a 1.5 m^2 x 0.6 m deep plastic fish tote with screened openings to allow water to exit. The trap had a pneumatic gate in the floor of the sluiceway that when opened allowed all fish to pass, and when closed directed all fish into the holding box.

Protocols

The screw traps were accessed by motorized raft from the west bank. All fish and debris in the live box were netted out and all fish were placed in buckets. Traps were checked daily during high periods of fish movement and every other day when catch rates were low. All salmonids were weighed, measured, assigned a smoltification classification based on external characters, and all non-salmonids were counted by species and life stage. Some salmonids were given partial fin-clips and placed in holding boxes upstream of the screw traps for trap efficiency releases. Fish that were not held for efficiency releases were released on the west bank 50 m downstream of the screw traps. Efficiency releases for the screw traps were conducted by releasing marked fish near the train trestle, approximately 500 m upstream of the screw traps.

The JUV trap was operated for several 24 h cycles during fall, winter, and spring seasons. When we fished the trap it ran continuously for 2 to 6 h, then the sluiceway gate was opened, the fish were netted out of the live box and placed in a bucket, the dewatering ramp and live box screens were cleaned, and the sluiceway gate was closed. It typically took about 10 - 30 min to complete the above sequence one time, depending on the number of fish captured. Fish were processed in the same manner as described above. Fish not used for trap efficiency releases were released in the Yakima River at the downstream end of the juvenile bypass system. Marked fish were released nightly for the efficiency releases for the JUV trap by driving a container of marked fish upstream to the Roza boat launch area and releasing the fish there during hours of darkness.

Results

Poor access, record high flows and ice severely affected our ability to operate mobile trapping equipment below Roza Dam during the fall-winter period. Screw traps were operated for the first time on October 27, 1995. One or two traps were then operated until a fall flood began on November 8. During the fall of 1995, discharge averaged 72 m^3/s, while the average of the previous five years was 19 m^3/s. Fall flows peaked at about 470 m^3/s in mid-November, 1995. Ice problems, extremely high flows, and trap damage delayed the reinstallation of traps until April 4, 1996, after which the traps were only fished four days before high flows forced us to halt sampling. More equipment failures and high flow problems prevented us from fishing screw traps again until May 3. Screw traps were then operated until May 17. The juvenile bypass trap was operated periodically between January 3 and 23, 1996, and again between April 8 and May 24, 1996. Flow during the spring of 1996 averaged 79 m^3/s, which was nearly three times the average spring flow of the previous five years (27 m^3/s).

Large numbers of juvenile upper Yakima River spring chinook salmon were migrating downstream past Roza Dam during the fall of 1995. A total of 260 juvenile spring chinook salmon and 13 rainbow trout were captured in the 13 days the screw traps were operated during the fall (Table 1). This movement occurred at a time when the reservoir elevation above Roza Dam was lowered to the bottom of the spill gates. The low pool level prevents the use of the juvenile bypass trap (no water passes through the bypass), which thereby eliminated our ability to compare data collected in the JUV trap and the screw traps during the fall period. We conducted one screw trap efficiency release during the fall period. At sunset on November 4, we released 150 marked juvenile spring chinook, that had been previously captured in the screw traps. The fish were released near the outfall of the juvenile bypass, about 400 m upstream of the screw trap. Two marked fish (1.3%) were recaptured the following day. Expanding the catch data by dividing total number of juvenile spring chinook salmon captured by the 1.3% capture efficiency provides an estimated passage of 19,846 fish during the 13 day period between October 27 and November 8, 1995. Many other species were also captured in the screw traps during the fall period (Table 2). The abundance of warm-water species may have been related to the recent lowering of the reservoir above Roza Dam. In our work, fall and winter catch rates of juvenile spring chinook salmon were much higher than catch rates during the spring (Table 1). Catch rates in earlier studies were also higher during the fall-winter period than during the spring. Fast et al. (1991) reported catching 4,626 juvenile spring chinook salmon at Roza Dam between December, 1989 and, February, 1990, and only 542 between March and April, 1990. We did not recapture any of the marked fish in the juvenile trap that were released at the Roza boat launch in the winter and spring, therefore we did not calculate any JUV trap capture efficiencies.

Table 1. Sampling dates in 1995 and 1996, and number of juvenile spring chinook salmon (SPC), wild steelhead (WSH), and rainbow trout (RBT) captured in two types of downstream migrant traps at Roza Dam. Data from 1989-1991 is provided for comparison (1989-90 data from Fast et al. (1991); 1991 JUV trap data provided by Bill Sharp, YIN; 1991 Screw trap data provided by D. Seiler, WDFW).

Sample Time	Dates	Days On	Trap	Num SPC	Num WSH	Num RBT
Fall 1995	10/27-11/8	13	Screw	260	0	13
Winter 1996	1/3-1/23	7	JUV	312	0	0
Spring 1996	4/4-5/17	15	Screw	27	10	6
Spring 1996	4/8-5/24	22	JUV	94	66	76
Winter 1989	12/12-2/28	46	JUV	4626	NA	NA
Spring 1990	3/1-4/17	47	JUV	542	NA	NA
Spring 1991	4/1-5/31	60	JUV	1239	441	NA
Spring 1991	4/1-5/26	49	Screw	519	221	NA

Table 2. Other fish species captured in screw traps and the juvenile bypass trap at Roza Dam, fall 1995-spring 1996. MWF=mountain whitefish, SUK=sucker species (approximately 70% bridgelip, and 30% largescale), NSF=Northern squawfish, STK=three-spine stickleback, YLP=yellow perch, PMK=pumpkinseed, SCU= sculpin species, LND=longnose dace, SPD=speckled dace, RSS=redside shiner, CHM=chiselmouth, BBH=brown bullhead. One brook lamprey was also captured in the screw trap in the spring of 1996.

Sample Time/ Trap	MWF	SUK	NSF	STK	YLP	PMK	SCU	LND	SPD	RSS	CHM	BBH
Fall 1995/ Screw	5	104	62	70	23	2	0	1	0	107	27	0
Winter 1996/ JUV	6	32	1	1	14	2	1	0	0	0	0	0
Spring 1996/ JUV	1	264	25	8	119	64	3	2	4	4	4	3
Spring 1996/ Screw	0	3	0	0	0	0	2	0	0	1	0	0

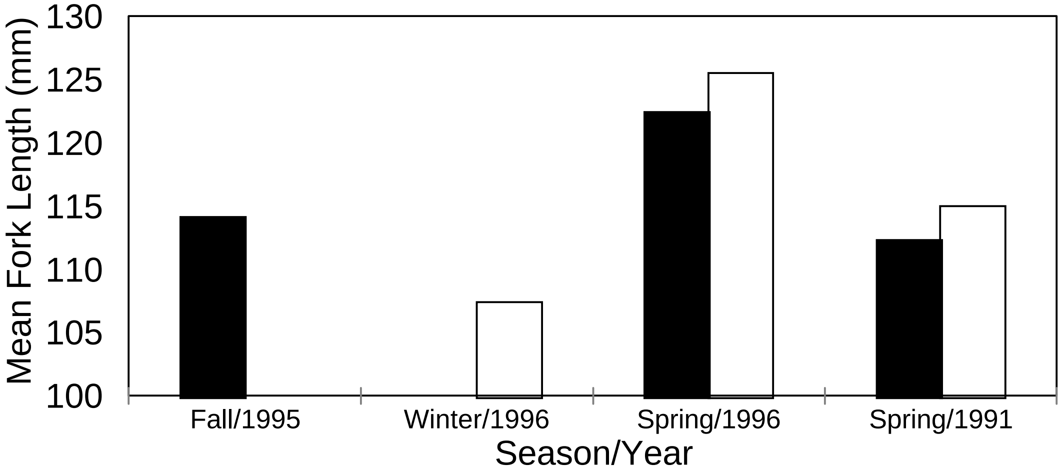

Figure 2. Mean lengths of juvenile spring chinook salmon captured at Roza Dam in the juvenile bypass trap (open bars) and in screw traps (solid bars). Data from 1991 provided by B. Sharp, YIN, and D. Seiler, WDFW.

The spring chinook salmon captured in the juvenile bypass trap were generally longer than those captured in the screw traps during the spring time period (Figure 2). However, differences in length were not significant for the spring period in 1996 ($t = 1.09$, $P = 0.278$). The condition factor (CF) of spring chinook salmon smolts captured in the JUV during the spring was significantly lower than the CF of spring chinook salmon captured in the screw traps ($t = -2.35$, $P = 0.020$). This may have been an artifact of fish length, however, with shorter fish being caught in the screw trap. The mean length of spring chinook salmon smolts captured in a screw trap the spring of 1991 was also shorter than the mean length of those captured in the JUV trap (Figure 2).

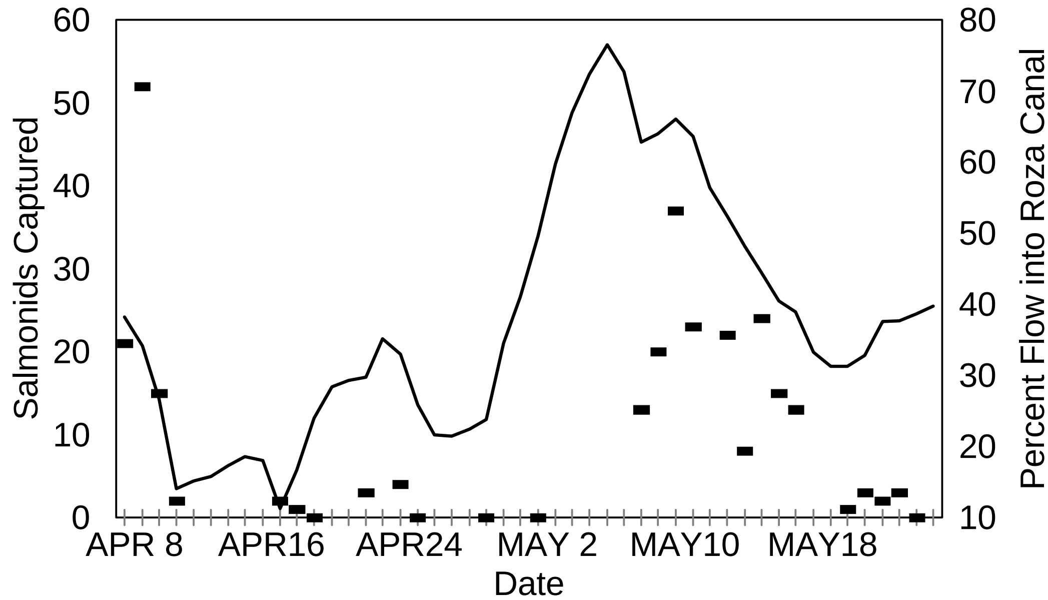

Figure 3. Percent of Yakima River flow entrained in Roza Canal (line) and total daily catch of salmonids in the juvenile bypass trap (rectangles) at Roza Dam during spring, 1996.

We had a small amount of data to use to compare catch rates of salmonids with flow entrainment into the Roza Canal. Between April 8 and May 22, 1996, there was a significant positive correlation (R = 0.51, P = 0.008) between the number of salmonids captured in the JUV trap and the percent of the flow at the Umtanum station (immediately upstream of Roza Dam) that was entrained into the Roza Canal (Figure 3). When examined by species, the relationship was stronger for rainbow trout (R = 0.40, P = 0.042), and steelhead smolts (R = 0.61, $P \leq$ 0.001), than it was for spring chinook salmon smolts (R = 0.37, P = 0.062).

During the course of our work at Roza Dam we identified conditions under which each type of downstream migrant trapping equipment might be used. We experienced extraordinary flow and winter conditions, which provided a good 'reality check' of what may be possible in terms of the types of conditions the juvenile bypass trap and the screw traps might be operable. Table 3 summarizes the conditions when each type of equipment would and would not be operable. In general, high flows are not conducive to high catch rates with either type of equipment. Screw traps do not perform well when water velocity exceeds about 2.5 m/s, and the juvenile bypass trap does not appear to entrain fish well when canal entrainment rates are below about 15%. In addition, the juvenile bypass trap is not functional when the reservoir upstream of Roza Dam is lowered for maintenance each fall. Further, icing conditions in the river below the dam as well as in the trap area of the juvenile bypass facility present samplers with very dangerous work environments. Facility changes at the trap site on the juvenile bypass facility could increase the number of days that the trap may be used by providing workers with a safer work environment, especially at low temperatures. Finally, when the juvenile bypass system is being dewatered by pumping, the trap area is unsafe due to the risk of rapid filling of the trap pit in the event of a pump failure. Only when the juvenile bypass is being dewatered by gravity feed back to the river is it safe for workers to be down in the trap pit. Unfortunately, use of the gravity return system poses problems for adult spring chinook salmon attempting to migrate upstream past Roza Dam. On May 31, 1996, we found five dead female adult spring chinook salmon (69-80 cm FL), on the ground outside the diffuser return box (DRB) where the gravity dewatering system empties into the river. Upon looking into the DRB we saw 10-20 more adult salmon jumping at the walls. This system was immediately shut down and all remaining salmon volitionally returned to the river. The attraction of this structure is great when the DRB discharges approximately 47 m^3/s of water immediately below the dam, where flows at that time of the year may be as low as 30 m^3/s.

Table 3. Summary of probable limitations to enumerating juvenile salmonids at Roza Dam with a combination of fixed (juvenile bypass trap) and mobile (screw traps) gear. A 'yes' indicates the gear could be operated (or would provide sufficient catch) under that condition, whereas a 'no' indicates it could not.

Condition	JUV Trap	Screw Trap	Reason, if No
Roza pool full	Yes	Yes	
Roza pool not full	No	Yes	bypass only draws at full pool
Discharge >70 m^3/s	Yes	No	excessive velocity for screws
Discharge <15 m^3/s	Yes	No	too shallow/slow for screws
Very high debris load	Yes	No	screws sink when cone stops
Air temp < -5° C	No	No	metal becomes coated in ice, unsafe for workers
Frazzle ice in water	No	No	traps clog
Bypass pumps on	No	Yes	Unsafe in case of pump failure
High outmig. numbers	Yes	Yes	
Low outmig. numbers + high canal entrainment	Yes	No	Low capture effic. in screws

Discussion

The objectives of this work shifted during the course of the work as new information was incorporated and as the MIPT identified new monitoring objectives for the Roza facility. The ultimate objective for smolt trapping at the Roza facility became capturing enough spring migrating spring chinook salmon smolts of both wild and hatchery origin (when hatchery releases begin in 1999) to compare relative survival of various hatchery and wild groups from Roza to Chandler, and points downstream (Busack et al. 1997). In light of this 'updated' objective, the discussion section focuses first on a general examination of the current equipment at the facility, and its anticipated ability to capture sufficient numbers of representative (unbiased samples) of spring smolts passing Roza Dam. Discussion relative to the initial objective of determining whether a combination of mobile and fixed gear at Roza would be useful in determining total salmon smolt production estimates for the upper Yakima basin as well as other interesting biological and life-history information will be presented later in the discussion.

Screw traps were effective at capturing large numbers of migrating smolts under only very limited environmental conditions. Screw traps were most effective during the fall period and were not very productive during the spring portion of the smolt outmigration. The juvenile trap was successful in capturing larger numbers of spring outmigrating smolts under a wider range of environmental conditions. There were circumstances that did reduce the effectiveness of the juvenile trap too, however. The screens associated with the juvenile facility at Roza appear to be quite effective at guiding smolts down the river (and through the dam spill), and preventing them from entering the canal. The juvenile bypass trap was effective at capturing migrating smolts only when entrainment into the canal was high (>50%). There is, however, little chance that existing facilities at Roza Dam will enable samplers to capture enough salmon smolts during the spring-migrating portion of the run during low production years to obtain sufficient sample sizes at Roza Dam without modifications to the existing juvenile bypass facility and trap. During high production years, either gear type would probably capture sufficient numbers of fish, but during low production years, the juvenile facility probably has the best chance to capture large enough numbers of fish to allow for valid survival comparisons between groups.

To effectively use the juvenile bypass trap as a monitoring facility would require several modifications to the existing conditions at Roza Dam. First, the proportion of fish entrained into the canal must be increased to allow for larger sample sizes and capture of a representative portion of the migrants. In addition, the trap area must be modified to increase the safety of the samplers. The trap pit should be enlarged and covered by a heated building, with a stair case down to the trap replacing the existing vertical ladder. Finally, a solution to the potential pit flooding problem (when the bypass is dewatered by pumping) must be implemented.

Similar to the findings of past researchers (Fast et al. 1991), we found that more juvenile salmonids were moving past Roza Dam during the fall and winter periods than during the spring. It appears that the dominant life history type of this species in the upper Yakima River is the fall-winter migrant. Unfortunately, the value of these fish in the monitoring program for the YFP is somewhat limited. Many assumptions are necessary in order to compare survival rates of fall-winter migrants and hatchery-reared spring-migrating smolts (Busack et al. 1997). This large portion of fall-winter migrants also necessitates the monitoring of basin production of spring chinook salmon smolts to be located as far downstream in the system as possible. Thus, Chandler, near the town of Prosser, has been selected as the production monitoring site (Busack et al. 1997).

To best compare performance of different hatchery treatments as well as wild and hatchery fish performance measures at the smolt stage, fish must be captured in the spring as they embark upon their seaward migration. Our results, similar to those of earlier work, indicate that a combination of mobile and fixed smolt trapping gear would be ineffective at Roza Dam in determining the production of spring chinook salmon smolts above Roza Dam. It also appears that the two gear types may capture different sizes of spring chinook salmon. Though the length differences were not significant, it is probable that there may be sampling biases associated with both gear types. Seiler (personal communication) postulated that the juvenile bypass trap captured fish that were injured or stressed. He based his conclusion on the higher proportion of marked fish that were recaptured in the juvenile bypass (26%), than in the screw trap (8%). In our study, both gear types injured low percentages of salmon captured. One spring chinook captured in the screw trap died (0.3%), while six others (2.1%) showed some sign of trap-induced injury or descaling. Four spring chinook salmon smolts (1.0%) were killed in the juvenile trap, while an additional seven were injured (1.7%). While our data also showed a lower condition factor in spring chinook smolts captured in the juvenile bypass trap, this may have been due to the fact that the fish captured there were longer than those captured in the screw traps. Longer fish typically have a lower condition factor than their shorter counterparts. Also, screw traps tend to be selective for shorter individuals, as longer fish have higher burst speeds that may make them better able to escape from the cone of a screw trap than a shorter fish of the same species. It may be that the fish captured in the bypass trap were moving along the bank, while the fish captured in the screw traps were migrating in the thalweg. The opening into the juvenile fish facility is very large, has low approach velocities, and is located along the outside bend of the reservoir in an impounded area. This question of whether the juvenile bypass trapping facility is capable of capturing an unbiased sample of spring chinook smolts during the spring period requires further evaluation.

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Chapter 8

Effects of Residual Hatchery-reared Steelhead on Growth of Wild Rainbow Trout and Spring Chinook Salmon

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Abstract

We investigated the effects of non-migrant (residual) juvenile hatchery steelhead *Oncorhynchus mykiss* on growth of wild rainbow trout *O. mykiss* and juvenile spring chinook salmon *O. tshawytscha* to examine how increased densities of residual hatchery steelhead might affect the growth of preexisting wild rainbow trout and chinook salmon. We used screened enclosures in a natural stream to examine food utilization and physiological stress, mechanisms which may have affected fish growth. The presence of residual hatchery steelhead negatively impacted growth of wild rainbow trout (1993: $P = 0.019$; 1994: $P = 0.020$) but did not impact growth of spring chinook salmon ($P = 0.360$). Enclosures did not reduce the total number of food items available but did influence the species composition of aquatic and terrestrial invertebrates. The food habits of paired and unpaired fish differed; however, the power of those tests was low. A measure of physiological stress, cortisol levels, did not differ between paired and unpaired fish held in enclosures. Cortisol levels in fish confined for 42 d were significantly lower than levels in wild fish outside the enclosures at the termination of the experiment. Our results suggest that adverse impacts to wild rainbow trout growth resulting from high densities (a doubling) of residual juvenile steelhead from hatchery releases may be significant.

Introduction

During the past decade concerns have increased regarding the potential for releases of fish from hatcheries to impact naturalized and indigenous fish populations (Bachman 1984; Vincent 1987; Goodman 1990; Waples 1991; Schramm and Piper (eds.) 1995). Hatchery-reared salmonids may compete with wild fish (Bachman 1984; Nickelson et al. 1986; Vincent 1987). Releases of large numbers of hatchery-reared fish increase the total density of fish in certain areas for varying lengths of time and competition for limited resources increases when fish density increases (Li and Brocksen 1977; Kennedy and Strange 1986; Heggenes 1988; Christiansen et al. 1992). Social interactions between hatchery and wild fish may negatively impact the wild fishes. For example, increased competitive behavioral interactions following releases of hatchery fish may increase stress levels in wild fish (Noakes and Leatherland 1977; Ejike and Schreck 1980) and reduce feeding opportunities (Abbott and Dill 1989).

Knowledge of competition between wild salmonids is more fully developed than understanding of competition between hatchery-reared and wild salmonids. Many studies have focused on the mechanisms of competition among salmonids such as agonistic interactions (Abbott et al. 1985; Huntingford et al. 1990; Hughes 1992) and niche separation (Griffith 1972; Hearn and Kynard 1986). For example, Everest and Chapman (1972) found that juvenile steelhead *Oncorhynchus mykiss* and chinook salmon *O. tshawytscha* utilized different habitats in streams and suggested that competition was limited.

The anadromous (steelhead) and resident (rainbow trout) forms of *O. mykiss* represent two life history forms of the same species that would be expected to be strong interactors if a common resource were limiting, because they have similar ecological requirements during a substantial part of their freshwater life history. Kennedy and Strange (1986) showed that fish of the same species compete strongly. Releases of hatchery-reared steelhead juveniles often result in relatively prolonged increases in salmonid density and biomass when a portion of the released fish fail to migrate seaward and become residuals (Viola and Shuck 1995). Residual hatchery steelhead are defined as those not emigrating from the release area prior to June 1.

Four years of underwater observations showed that residual hatchery steelhead and rainbow trout occupied similar habitat types and engaged in agonistic interactions throughout the summer period (McMichael et al. 1992; Pearsons et al. 1993; McMichael et al. 1994). To better understand the impacts of residual hatchery steelhead on wild rainbow trout and spring chinook salmon, we conducted a series of experiments in small enclosures in a natural stream in the upper Yakima River basin, Washington. Our intent was to provide insight to policy makers regarding the potential for residual hatchery-origin steelhead (resulting from releases of hatchery steelhead smolts) to impact the growth of wild rainbow trout and spring chinook salmon. The objectives of these experiments were to: 1) investigate the effects of the presence of hatchery steelhead residuals on growth of wild rainbow trout and spring chinook salmon, and 2) explore mechanisms contributing to any observed impacts. Our results have implications for current as well as future artificial propagation programs where fish are released into areas that have populations of wild salmonids.

Methods

Study Area

We conducted growth experiments in the North Fork of the Teanaway River, a tributary which enters the Yakima River, Washington, 282 km upstream from the confluence of the Yakima and Columbia rivers. The North Fork of the Teanaway River is 29 km long and drains a portion of the eastern slope of the Cascade Mountains covering a basin area of 246 km^2. Our 2.5 km study reach ranged in elevation from 750 to 780 m above sea level. Streamside vegetation was composed of conifers and deciduous trees and shrubs. Substrate composition was dominated by cobbles and areas of sandstone bedrock. Water temperatures measured during the study periods ranged from 7 to 20.5 C.

Both wild rainbow trout and residual hatchery steelhead were present in the study reach. Hatchery fish were identified by an excised adipose fin. Natural production of steelhead in the study area was extremely low during the study (McMichael et al. 1992). Wild resident rainbow trout are not visually discernable from juvenile steelhead prior to the smolt stage, thus all naturally-produced *O. mykiss* were classified as resident rainbow trout. Other fish species observed in the study reach during the study period included, in order of decreasing abundance, shorthead sculpin *Cottus confusus*, torrent sculpin *C. rhotheus*, longnose dace *Rhinichthys cataractae*, mountain whitefish *Prosopium williamsoni*, bridgelip sucker *Catostomus columbianus*, and eastern brook trout *Salvelinus fontinalis*. Our study reach overlapped the area of previous study of interactive behavior between wild rainbow trout and hatchery steelhead (McMichael et al. 1992; Pearsons et al. 1993; McMichael et al. 1994).

Experimental Design

Our experiment utilized a control-treatment design focusing on differences in growth of paired and unpaired fish. Growth experiments between 1) residual hatchery steelhead and wild rainbow trout, and 2) hatchery steelhead residuals and wild spring chinook salmon were performed from July 7 to August 19, 1993. The residual hatchery steelhead and wild rainbow trout tests were repeated from July 5 to August 17, 1994. A solitary fish (control) was placed in one chamber of the enclosure and treatment and response fish were placed in the other. The control and response fish were of the same species group for a given test, while the treatment fish were residual hatchery steelhead in all cases. The combinations used in this experimental design were intended to ascertain effects on response fish. The terms control, response, and treatment fish distinguish the different groups of fish in each test; control and unpaired are used interchangeably as are response and paired. Each of the tests was replicated 10 times in 1993 and test 1 was replicated 20 times in 1994.

In the area where we conducted these tests, residual hatchery steelhead are typically about 40% longer than wild rainbow trout and nearly twice the length of wild age 0+ spring chinook salmon during the summer rearing season. This experiment was not designed to determine which species groups were the most dominant when fish sizes were equal. It was instead designed to determine if the presence of a treatment fish influenced the growth of the response fish. We designed the experiment in this manner in an attempt to determine what effect a doubling in the density of salmonids (due to presence of residual steelhead from hatchery releases) would have on the growth of preexisting wild salmonids.

Twenty Enclosures were constructed with 5 cm x 5 cm wood frame members enclosed with galvanized wire (0.95 cm mesh) on all sides and the bottom. The inside dimensions of each enclosure were 91 cm high by 91 cm long and 99 cm wide. Each enclosure was divided into two equal-sized (0.46 m^2) chambers by a plywood barrier. Four large cobbles (20 to 30 cm diameter) were collected from the wetted stream channel and positioned in each chamber of each enclosure to coarsely simulate natural conditions and to provide substrate for benthic organisms. A plywood lid was attached to the top of each enclosure to increase fish security.

The 20 enclosure sites used in 1993 were selected on June 29. Each site was assigned randomly to either a pool or run habitat type with a depth of 0.35 to 0.70 m, and a velocity of 0.12 to 0.42 m/s. These criteria were developed from observations of fish-habitat relationships from prior sampling (McMichael et al. 1992). Enclosures were randomly placed among the selected sites on July 6. In 1994, site selection criteria were the same as in 1993 and enclosures were placed in pool and run habitats ranging in depth from 0.35 to 0.66 m, and velocities from 0.20 to 0.43 m/s.

In both 1993 and 1994, all wild rainbow trout were collected from the North Fork of the Teanaway River. Rainbow trout between 100 and 150 mm fork length (FL) were collected on July 7, 1993, and on July 5, 1994, using battery-powered backpack electrofishers (settings: pulsed direct current (PDC) 300 V, 30 Hz and 300 V, 60 Hz) in the area immediately surrounding (within 50 m) each enclosure. This range of trout length was targeted because it represented the modal length of trout ($\pm$ 25 mm) previously observed in the study reach at that time of year (McMichael et al. 1992). The relative sizes of the groups of fish used in this experiment (Table 1) were those typically found during the summer rearing period in streams in the upper Yakima River basin. Though the trout used in this experiment were not individually aged using scale analyses, available age and size information from the North Fork of the Teanaway River suggest that trout between 100 and 150 mm FL are predominantly age 1+ and 2+ (Martin and Pearsons 1994).

On the day of collection, test fish were anesthetized (using approximately 0.1 g/l MS-222), measured to the nearest mm FL, weighed to the nearest 0.1 g, and the external appearance of each fish (e.g. fin condition) was recorded. Age 0+ spring chinook salmon were not present near the study area when this experiment began, necessitating their collection by electrofishing from the mainstem of the Yakima River near the town of Cle Elum, Washington on July 7, 1993. These fish were immediately transported in aerated vessels to the study area (approximately 30 km) where they were distributed into the appropriate enclosures (for test 2) in the same manner as previously described for rainbow trout. The rainbow trout and spring chinook salmon control and response fish were then allowed to acclimate to, or establish 'prior residence' in, the enclosures for

Table 1. Mean fork length (mm) and mean weight (g), standard deviation (SD), and range of test groups at the beginning of the growth experiments for all tests, 1993 and 1994. RBT = wild rainbow trout, HSH = residual hatchery steelhead, and SPC = spring chinook salmon. Sample sizes were ten for each group in 1993 and 20 for each group in 1994.

	Control			Response			Treatment		
Test No. (year)	Mean	SD	Range	Mean	SD	Range	Mean	SD	Range
Length									
		RBT			**RBT**			**HSH**	
1 (1993)	114.4	14.6	101-143	117.0	13.5	102-140	169.4	25.1	140-204
(1994)	115.9	9.2	108-136	118.4	15.0	102-138	168.5	10.1	156-183
		SPC			**SPC**			**HSH**	
2 (1993)	76.1	10.6	64-92	70.1	3.9	64-76	155.9	38.4	117-213
Weight									
		RBT			**RBT**			**HSH**	
1 (1993)	18.0	8.5	11.5-36.0	18.9	6.0	12.8-29.6	51.0	23.2	26.9-88.7
(1994)	18.8	6.0	14.4-32.6	21.1	5.9	13.3-28.9	47.2	9.4	37.0-61.4
		SPC			**SPC**			**HSH**	
2 (1993)	5.7	2.4	2.9-9.7	4.0	0.9	2.5-5.2	43.5	29.8	15.1-90.6

two days before the introduction of a residual hatchery steelhead (treatment fish) into one of the chambers in each enclosure. This period of prior residence approximated conditions where hatchery fish are released into areas occupied by wild fish. For a variety of reasons, prior residence has been shown to afford an advantage to stream salmonids in competitive situations (Allee 1982; Heggenes 1988; Metcalfe and Thorpe 1992).

Treatment fish were placed in the enclosures on July 9, 1993, and on July 7 in 1994. In both years, these hatchery steelhead residuals were collected from Jungle Creek, a tributary to the study stream, using backpack electrofishing gear and placed into a holding vessel. Hatchery steelhead were then removed from the holding vessel and sampled following the protocol described for other test fish. A hatchery steelhead residual was then placed in one of the chambers (assigned randomly) in each of the enclosures containing rainbow trout (test 1) and spring chinook salmon (test 2).

Enclosures in which one or more mortalities occurred prior to the end of the study period were discarded from the final analyses. This occurred in three of the replicates for both test 1 and test 2 in 1993. Ten of the 20 replicates were not used in 1994 due to one or more fish missing at the termination of the experiment. Most missing fish were assumed to have died prior to the termination of the experiment although some may have escaped.

On August 19, 1993, 42 days after the control and response fish were placed in the enclosures, all fish were collected from the enclosures, euthanized in a lethal concentration (>200 mg/l) of MS-222, measured to the nearest mm FL, weighed to the nearest 0.1 g, and bled for physiological analyses.

Capture protocol consisted of approaching an enclosure from the downstream side, quickly removing the plywood top and cobbles, and electrofishing within the enclosure with the same equipment and settings used in the initial collections. The amount of time to capture all fish within each enclosure averaged 1 min 54 s (range; 1 to 3 min). The reason fish were collected by electrofishing, instead of simply netting them, was to provide samples collected from inside and outside enclosures by like means for physiological testing at the conclusion of the experiment. In 1994, all fish were netted from the enclosures on August 17 (after 43 days in the enclosures), anesthetized in MS-222, measured to the nearest mm fork length, and weighed to the nearest 0.1 g.

In 1993, we compared food habits of control and response rainbow trout and spring chinook salmon by examining stomach contents from a subsample of all fish. Stomachs were extracted and preserved in 10% buffered formalin at the termination of the experiment. The contents of a subset of the rainbow trout stomachs (6 control and 6 response) and spring chinook salmon stomachs (5 control and 5 response) were examined using binocular dissecting microscopes. Food items were enumerated and identified to order using head capsules.

Between August 26 and 28, 1993, a test was conducted to determine whether the presence of the mesh screen influenced food availability within the enclosures. In run habitat downstream of a riffle in the middle of the study reach, six screens (0.95 cm square galvanized wire mesh) were attached to a 13 mm diameter metal re-bar that had been pounded into the substrate perpendicular to the water surface. These screens were then not cleaned for 48 h to simulate the accumulated debris load typically found on our experimental enclosures. Drift nets (46 cm x 31 cm, 363 micron mesh size) were then attached to the downstream side of the screens, and six additional unscreened nets were also located along the same transect in an alternating pattern. Drift nets were deployed for 22 to 24 h. Samples were preserved in 70% isopropyl alcohol. Using a binocular dissecting microscope, we identified and enumerated insects in each order observed.

In 1993, we also examined potential enclosure effects on stress physiology (plasma cortisol levels) and condition factor in test fish. Fish from inside the enclosures were collected on August 19 and those outside the enclosures were collected on August 20. Blood samples for stress physiology data were collected from anesthetized fish by severing the caudal peduncle and collecting the blood in ammonium-heparinized capillary tubes. After centrifugation, the plasma was frozen for later analysis. Plasma cortisol levels were obtained by radioimmunoassay using the protocol developed by Foster and Dunn (1974) as modified by Redding and Schreck (1983).

Data Analyses

Specific growth rate (SGR) was calculated using the following equation (Fausch 1984):

$$SGR = (\log_e W_t - \log_e W_0)/t$$

Where W_t = weight (g) at the end of the period, W_o = weight (g) at the beginning of the period, and t = time (days).

One-tailed paired t-tests were performed to test for differences in SGR among control and response fishes. For stress physiology samples, paired t-tests were performed for differences among means. Sample distributions of cortisol levels were normalized by log transformation. Screened and unscreened drift samples were compared using two-tailed paired t-tests. Paired t-tests were also used to compare the numbers and types of food items ingested by control and response rainbow trout and spring chinook salmon. Statistical significance was based on alpha levels of less than 0.05. Statistical power analyses (Snedecor and Cochran 1981; Peterman 1990) for t-tests involving control and response fish growth, food habits, and physiological stress were performed following data collection to aid in the interpretation of these results.

Results

Residual hatchery reduced the SGR of wild rainbow trout but not age 0+ spring chinook salmon. In 1993, the mean SGR of unpaired rainbow trout (controls) in test 1 was greater than the SGR of trout paired with residual hatchery steelhead (P = 0.0019; Table 2). The mean SGR of treatment fish in test 1 was also negative in 1993 (Table 2). Again in 1994, SGRs of control rainbow trout were higher than SGRs of response rainbow trout (P = 0.0020; Table 2). Residual hatchery steelhead in the 1994 test exhibited negative SGRs comparable to those observed in both tests conducted in 1993 (Table 2).

Table 2. Mean specific growth rates (SGR) of control (C) and response (R) rainbow trout and spring chinook salmon in growth experiments in 1993 and 1994. Standard deviations (SD), sample size (N), and range are also presented. Residual hatchery steelhead were the treatment (T) fish in all cases. Species groups: RBT = rainbow trout, SPC = spring chinook salmon, HSH = residual hatchery steelhead.

Test No.	Year	Species group	Test group	SGR	SD	N	Range
1	1993	RBT	C	-0.0016	0.0029	7	-0.0053 to +0.0016
1	1993	RBT	R	-0.0060	0.0020	7	-0.0094 to -0.0032
1	1993	HSH	T	-0.0020	0.0015	7	-0.0042 to -0.0002
1	1994	RBT	C	-0.0039	0.0020	10	-0.0071 to -0.0017
1	1994	RBT	R	-0.0061	0.0021	10	-0.0109 to -0.0034
1	1994	HSH	T	-0.0023	0.0021	10	-0.0064 to -0.0003
2	1993	SPC	C	-0.0004	0.0040	7	-0.0049 to +0.0071
2	1993	SPC	R	-0.0006	0.0048	7	-0.0051 to +0.0087
2	1993	HSH	T	-0.0018	0.0013	7	-0.0036 to 0.0000

In test 2, spring chinook salmon paired with hatchery steelhead residuals did not exhibit significantly different SGRs from their unpaired counterparts (P = 0.360; Table 2). The statistical power of the SGR comparison for test 2, however, was low (0.109). Average decreases in SGR for hatchery steelhead residuals in test 2 were similar to those in test 1 (Table 2).

In 1993, control and response fish did not ingest significantly different numbers of food items. In test 1 conducted in 1993, control rainbow trout contained an average of 27.5 items while response fish averaged 6.0 items (Table 3). These differences were not significant (P = 0.129); however, the power of that test was low (0.357). Control fish in test 1 did, however, contain a significantly greater number of orders of insects than their paired counterparts (P = 0.006). Paired and unpaired spring chinook salmon did not contain significantly different numbers of items (P = 0.109) or numbers of orders (P = 0.115)(Table 3).

Table 3. Mean number of food items and orders identified in stomachs of control and response fish from inside experimental enclosures at the termination of the experiments, August 19, 1993. Ranges are shown in parentheses. Test number, species (RBT = rainbow trout, SPC = spring chinook salmon), test group, control (C) or response (R), and number of stomachs sampled (N) are shown.

Test No.	Species group	Test group	N	Mean number	
				Food items	Orders
1	RBT	C	6	27.5 (6-107)	4.2 (3-5)
1	RBT	R	6	6.0 (1-15)	2.3 (1-4)
2	SPC	C	5	53.0 (19-68)	4.2 (3-5)
2	SPC	R	5	42.2 (1-76)	3.2 (1-6)

Insects of the orders Ephemeroptera, Hymenoptera, Trichoptera, and Hydracarina were numerous in the stomachs of both species (Table 4). Trichoptera made up a larger percentage of the items eaten by trout than by spring chinook salmon while trout diets contained a lower percentage of dipterans. Hymenoptera adults (terrestrials) appeared with greater frequency in stomach samples than in drift samples.

Mesh screen similar to that used on the experimental enclosures did not affect the total number of food items passing through, but did appear to affect the relative occurrence of two orders. The total numbers of food items in unscreened drift samples (mean = 204.8 items, range 117 - 387) were similar to those in screened samples (mean = 244.5 items, range = 31 - 416) (P = 0.64) (Table 4). Furthermore, diversity of insect orders in unscreened and screened samples were similar (unscreened mean = 7.3, range = 6 - 9, screened mean = 7.2, range = 4 - 10)(P = 0.88). Interestingly, unscreened samples contained a higher percentage of Trichoptera larvae while the

Table 4. Total numbers of food items, and percents, listed by order, found in drift samples (screened (N=6) and unscreened (N=6)) and stomachs (rainbow trout (RBT); N=12; spring chinook salmon (SPC); N=10) in 1993 tests.

	Drift samples (%)		Stomachs (%)	
Order	screened	unscreened	RBT	SPC
Ephemeroptera	492 (33.5)	100 (8.1)	46 (12.4)	60 (12.6)
Plecoptera	10 (0.7)	10 (0.8)	1 (0.3)	2 (0.4)
Diptera	752 (51.3)	781 (63.5)	259 (69.6)	384 (80.5)
Trichoptera	136 (9.3)	233 (19.0)	11 (3.0)	3 (0.6)
Coleoptera	8 (0.5)	7 (0.6)	13 (3.5)	1 (0.2)
Hemiptera	7 (0.5)	5 (0.4)	4 (1.1)	5 (1.0)
Hymenoptera	6 (0.4)	10 (0.8)	30 (8.1)	13 (2.7)
Neuroptera	0 (0.0)	1 (0.1)	0 (0.0)	0 (0.0)
Odonata	0 (0.0)	1 (0.1)	2 (0.5)	0 (0.0)
Collembola	3 (0.2)	3 (0.2)	0 (0.0)	0 (0.0)
Hydracarina	50 (3.4)	77 (6.3)	6 (1.6)	8 (1.7)
Crustacea	0 (0.0)	0 (0.0)	0 (0.0)	0 (0.0)
Lepidoptera	2 (0.1)	0 (0.0)	0 (0.0)	0 (0.0)
Araneae	1 (0.1)	1 (0.1)	0 (0.0)	1 (0.2)
Total	1467	1229	372	477

screened samples included a higher percentage of Ephemeroptera nymphs. Diptera was the most abundant order in both unscreened and screened drift samples, making up 63.5 and 51.3 percent of the total number of insects respectively. In contrast to the large number of adult Diptera in stomach samples, nearly all (98%) of the Diptera in the drift samples were larvae. Ephemeroptera, Trichoptera, and Hydracarina were the other orders of insects that were most abundant in the drift samples.

It did not appear that the enclosures had an effect on the mean condition factor of rainbow trout. The condition factors of trout inside the enclosures (mean = 1.20) were not significantly different from those captured outside the enclosures (mean = 1.18) at the termination of the experiment in 1993 ($P = 0.320$).

Titers of circulating cortisol did not differ significantly between control and response rainbow trout ($P = 0.740$) and or chinook salmon ($P = 0.510$; Figure 1). However, rainbow trout that were confined for 42 d had significantly lower circulating levels of cortisol than rainbow trout captured outside the enclosures at that time ($P = 0.010$; Figure 1).

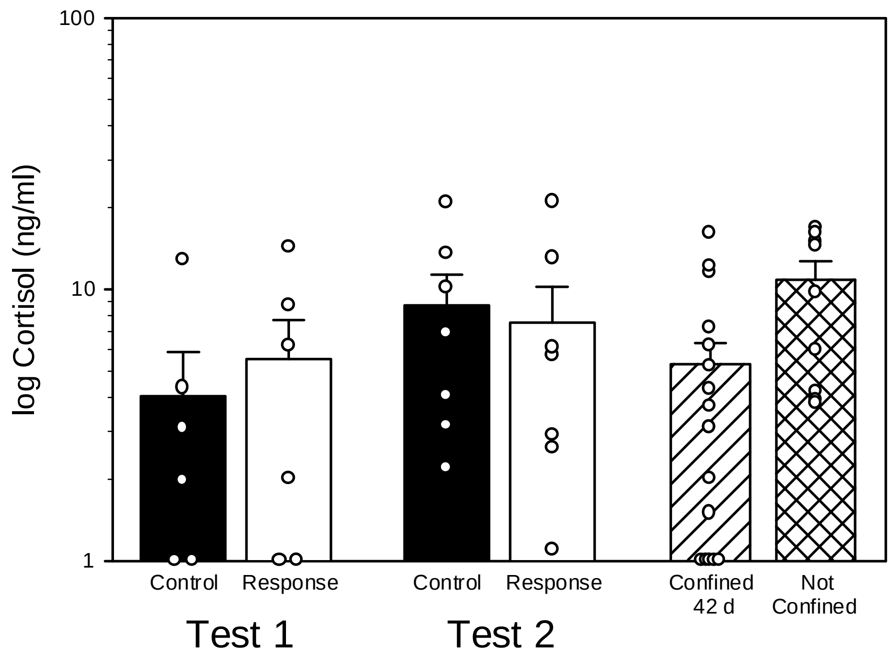

Figure 1. Mean cortisol levels of control and response fish for test 1, test 2, and mean cortisol levels in unconfined and confined rainbow trout at the termination of the experiment in 1993. Capped lines represent 1 standard error of the mean. Individual data points are provided. Y-axis is on a log scale.

Discussion

Our results suggest that the occurrence of hatchery steelhead residuals reduced the growth of wild resident rainbow trout during the summer. When hatchery steelhead become residuals, thus increasing densities of fish inhabiting similar areas for long periods of time, the potential for reduced wild rainbow trout growth is increased. A reduction in size, due to slower growth during the summer, could decrease over-winter survival (Hunt 1969; Toneys and Coble 1979, 1980; Oliver and Holeton 1979), resulting in decreased population size (Cunjak et al. 1987).

Fish size and species played important roles in the relationship between dominance and growth. The greatest differences in growth between control and response fish were seen in test 1 in 1993 and test 1 in 1994 in which the treatment fish were considerably larger than the conspecific response fish. Our results are consistent with existing literature on competition among salmonids which suggests that larger fish generally dominate smaller fish (Griffith 1972; Abbott et al. 1985; Hearn 1987; Chandler and Bjornn 1988; Huntingford et al. 1990; Hughes 1992). To the extent that a fish in our study chambers was dominant, its feeding success might have been increased (Helfrich et al. 1982) thereby reducing the amount of food available to the smaller subordinate fish (Li and Brocksen 1977). Relatively small differences in size (a weight advantage of 5% or more) have been shown to assure dominant status for salmonids (Abbott et al. 1985). In addition, these socially dominant fish are known to exhibit greater mobility and feeding success than smaller subordinates (Helfrich et al. 1982). Subordinate fish in our study were not allowed to emigrate from the test enclosures. Under natural conditions, subordinate fish may emigrate, thereby reducing negative growth impacts in environments that are below carrying capacity.

In our study, the only significant growth differences between control and response fish were seen among conspecifics (*O. mykiss*, test 1 in both years). Fish of the same species might be expected to compete more than fish of different species due to the commonality in their ecological requirements at similar life stages (Allee 1982; Kennedy and Strange 1986).

There are two primary explanations for our failure to detect significant growth impacts in the test involving spring chinook salmon (test 2). First, competition may not have occurred because spring chinook occupied different niches than the residual hatchery steelhead. In cases where fish species are different and the difference in fish size is very large, competitive impacts may be reduced by niche partitioning (Lister and Genoe 1970; Everest and Chapman 1972; Dolloff and Reeves 1990). The spring chinook salmon we observed were generally higher in the water column than the treatment fish (steelhead) (McMichael, unpublished data). Secondly, the sample sizes used in these tests were smaller than would have been preferable for statistical considerations given the actual variation in our results. The resulting power of this test was relatively low (0.109), thereby increasing our chances of making type II errors. Given the observed differences in mean SGRs between control and response spring chinook salmon, and the relatively high standard deviation in this test, we would have had to deploy 8,241 enclosures to achieve a statistical power of 0.9. This number of trials would obviously be impractical, but it does illustrate that this method may not be well suited for examination of growth impacts for some species combinations. It is likely, however, that standard deviation would decrease as sample size increased, thereby reducing the number of required trials.

Mean SGR for all groups of fish in our experiments was negative. Some individuals did gain weight, but most (81%) lost weight during the six week study periods. Growth impacts were then necessarily determined by comparison of negative SGRs.

Weight loss in stream salmonids has been reported for fish placed in enclosures. For example, Miller (1952) found that hatchery cutthroat trout *O. clarki* lost weight during the first 40 days after being placed in enclosed stream sections with wild trout. Fausch and White (1986) reported negative SGRs for individual brown trout *Salmo trutta* and brook trout in competition experiments with coho salmon *O. kisutch* in an artificial stream. Our enclosures prevented fish inside from foraging directly off the substrate (with the exception of the four cobbles in each chamber). It is conceivable that this feeding restriction may have contributed to the negative SGRs.

Factors related to the enclosures may have individually or interactively influenced our results due to effects on fish behavior and movement. Stream salmonids have been shown to move in response to fluctuating environmental factors such as daylight, invertebrate drift, and water temperature. For example, Edmundson et al. (1968) observed juvenile steelhead occupying inshore, low velocity areas during darkness and areas with moderate current during daylight hours. The fish inside our enclosures were unable to move naturally as could fish outside the enclosures. Fish outside our enclosures were able to adjust their positions in response to water temperature fluctuations (behavioral thermoregulation), changes in the intensity of sunlight, the periodicity of insect emergence and drift, flow, and density of other fishes and potential predators. Nonetheless, all fish in our tests were subjected to a standardized confinement protocol, making relative comparisons between control and response fish inside the enclosures meaningful.

The uniformly low cortisol levels among confined fish suggest that the fish may have completely acclimated to their new environment (enclosures). The significantly lower cortisol levels in confined fish at the termination of the experiment may, in part, be explained if conditions were less stressful inside the enclosures than outside with respect to the availability of overhead cover (plywood top) and protection from predation. However, the experiment was not designed to characterize the physiological changes that were certainly associated with the capture and handling of the fish immediately before their confinement, nor was the statistical power of that test sufficiently high to detect low levels of variation. Given that other results in this study indicate that negative, and presumably stressful, interactions did occur, a more powerful experimental design (e.g., larger sample sizes and/or shorter time intervals between sampling) to explore the physiological basis of the negative aspects of competitive interactions is warranted.

Our enclosures did not affect the amount of food available to fish inside, though composition of invertebrate orders was affected. The mesh size we used (0.95 cm) was considerably larger than the mesh sizes that Cooper et al. (1990) found to significantly influence the immigration and emigration of invertebrates in enclosures they used in trout streams. Our results were similar; mesh size did not affect invertebrate movement into enclosures, even when the screens went uncleaned for 24 to 48 h. It is possible that some invertebrate orders, such as Ephemeroptera, may have been more common in screened samples due to their affinity for the vegetative matter that accumulated on the screens. The effect on SGR of the shift in relative proportions of invertebrate orders available to fish inside our enclosures is unclear, though probably small.

Our results may be useful in the evaluation of risks associated with behavioral interactions resulting from high residual densities following releases of hatchery fishes into areas containing wild populations. The application of these results to hatchery programs falls into two general categories; 1) assessment of the potential impacts of hatchery residuals on wild fish populations, and 2) alternatives in the operational aspects of hatchery management that affect the incidence and density of residuals.

If hatchery-reared salmonids compete with wild fish to the detriment of the latter, decreased productivity of the wild population could result. In areas such as the northwestern United States, where many wild stocks of salmonids are at critically low levels (Nehlsen et al. 1991, Washington Department of Fisheries et al. 1993), the impact of hatchery-produced fish may be serious enough to warrant program review and modification. Our study suggests that the species and size of the hatchery fish influences the potential for impacts on the growth of wild salmonids. In cases where hatchery residuals are larger than their wild conspecifics, the impacts would be expected to be greatest. It is conceivable however, that very large size differences could reduce competition through differential habitat segregation (Pearsons et al. 1994). However, there is concern that if very large hatchery steelhead (over 250 mm in length) become residuals, the potential for predation on wild salmonid fry would increase (Cannamela 1992). However, research in Washington state has found predation by hatchery steelhead residuals on wild salmonids to be low (Martin et al. 1993; Pearsons et al. 1993).

Hatchery release strategies that minimize the occurrence of non-migrating residuals and that minimize the spatial and temporal overlap of residuals with wild rainbow trout populations would be expected to have the least impact. For instance, in areas where large numbers of hatchery steelhead smolts become residuals, the impacts from these residual fish on wild rainbow trout could be acute. Where most hatchery fish emigrate quickly, their short-term impacts should be relatively minor. As an indication of the potential seriousness of this issue, the National Marine Fisheries Service (1995) has drafted hatchery steelhead smolt target size criteria for releases to be made in the area encompassed by the draft Snake River chinook Recovery Plan. Research on methods to achieve hatchery program objectives while maintaining wild stocks (e.g., Viola and Schuck 1995; McMichael et al. in review) is urgently needed.

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Chapter 9

Effects of Wild Juvenile Spring Chinook Salmon on Growth and Abundance of Wild Rainbow Trout

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Abstract

We investigated some of the ecological impacts to rainbow trout *Oncorhynchus mykiss* that could occur by supplementing spring chinook salmon *O. tshawytscha* in the upper Yakima River basin, Washington. Controlled field experiments conducted in three different streams indicated that presence of wild juvenile spring chinook salmon did not adversely affect growth of wild rainbow trout in high elevation tributaries. Experiments at two spatial scales, habitat-subunit and stream-reach scale, were used to detect impacts. In small enclosure experiments conducted in two tributaries to the Yakima River in 1993 and 1994, specific growth rates (SGRs) of wild rainbow trout paired with wild juvenile spring chinook salmon were not significantly lower than SGRs of their unpaired counterparts (1993: $P = 0.360$; 1994: $P = 0.190$). Stream-reach experiments in another Yakima River tributary in 1995 also indicated that introductions of wild juvenile spring chinook salmon into 100 m-long enclosures, at a numerical density equal to the pre-existing wild trout, did not adversely affect wild rainbow trout growth or abundance. The mean fork length and instantaneous growth rate (IGR) of age-0 wild rainbow trout in stream-reach enclosures was unaffected by introduced spring chinook salmon after seven (FL: $P = 0.318$) and 14 weeks (FL: $P = 0.387$, IGR: $P = 0.265$) in sympatry. Mean fork lengths and IGRs of age-1 rainbow trout were also unaffected by the addition of the spring chinook salmon after seven (FL: $P = 0.553$, IGR: $P = 0.124$) and 14 (FL: $P = 0.850$, IGR: $P = 0.084$) weeks of cohabitation. Furthermore, the stream-reach experiment showed that spring chinook salmon introduction did not affect rainbow trout abundance ($P = 0.298$) or biomass ($P = 0.538$). Site elevation in the stream-reach tests appeared to influence trout size more than the addition of juvenile spring chinook salmon. Site elevation was negatively correlated with length of wild age-0 ($P < 0.001$) and age-1 ($P < 0.001$) rainbow trout in October, 1995. It appears that rainbow trout and spring chinook salmon partitioned the resources so that impacts were not detected. Our work suggests that rainbow trout have a refuge from interactions with juvenile spring chinook salmon in high elevation portions of tributaries (e.g., over 700 m).

Introduction

In efforts to restore or enhance anadromous salmonid stocks, managers often consider supplementation as a strategy to increase natural production through artificial propagation (Clune and Dauble 1991; Cuenco et al. 1993; Winton and Hilborn 1994). A goal of supplementation programs is to increase the abundance of spawning adult salmonids so that natural production may increase. A truly successful supplementation program would be terminated after succeeding in boosting natural production. One desirable result of these efforts would be an increased abundance of naturally produced juvenile salmonids occupying freshwater rearing areas. An undesirable result of any artificial propagation program might be that non-target species would be adversely affected (Vincent 1987; Steward and Bjornn 1990; Krueger and May 1991; Garman and Neilsen 1982; Fossa et al. 1994).

Concerns about impacts to rainbow trout *Oncorhynchus mykiss* relative to planned supplementation of spring chinook salmon *O. tshawytscha* in the upper Yakima River basin, prompted us to examine potential interactions between these species. Even though these species have evolved in sympatry, the possibility exists that spring chinook salmon and rainbow trout could compete for resources in the freshwater rearing area because spring chinook salmon spend over a year in freshwater, emerge earlier than rainbow trout, and are found in similar habitats. The earlier emergence of spring chinook salmon could give them a size advantage that would increase their ability to dominate the later-emerging rainbow trout (Abbott et al. 1985). For example, we have observed age-0, -1, and -2 rainbow trout in close proximity to age-0 spring chinook salmon in the main stem of, and tributaries to, the Yakima River. We have also observed behavioral contests between these two species in which spring chinook salmon dominated age-0 rainbow trout (McMichael et al. 1992; Pearsons et al. 1996). However, other studies of sympatric wild populations of chinook salmon and rainbow trout have indicated that the two species utilize different microhabitats (Everest and Chapman 1972; Hillman et al. 1989), which may be related to their different sizes resulting from disparity in emergence times. Furthermore, McMichael et al. (1997) found that residual hatchery steelhead (non-migrant progeny of anadromous *O. mykiss*) did not adversely impact growth of wild spring chinook salmon even when the steelhead were nearly twice as large as the salmon. There is, however, a lack of information on the potential effects of experimentally increased densities of spring chinook salmon on rainbow trout in natural settings.

The Yakima Fisheries Project (YFP) will test the strategy of supplementation using different strategies to rear and release spring chinook salmon into the upper Yakima River basin (Clune and Dauble 1991; BPA et al. 1996). Because salmon from the YFP facilities will be released at the smolt stage, and these fish will inhabit the freshwater rearing environment for a relatively brief time period, the risk of competition for food and space to other fish species from these artificially-reared smolts is expected to be low (McMichael et al. 1994). However, the greatest risks for ecological interactions impacts to other non-target salmonids could occur if natural production of spring chinook salmon increases. This would result in increased densities of age-0 spring chinook salmon in freshwater rearing environments. Concerns about these competitive impacts are especially acute in areas with highly valued wild salmonid populations, such as the Yakima River. The upper Yakima River supports one of Washington State's best wild resident rainbow trout fisheries (Krause 1991; Probasco 1994).

The purpose of this study was to determine whether an increase in density and distribution of juvenile spring chinook salmon in tributaries to the Yakima River would decrease the growth or abundance of rainbow trout. Small-scale enclosure experiments were used in two tributaries to the upper Yakima River in 1993 and 1994 to investigate the effects of juvenile spring chinook salmon presence on growth of juvenile rainbow trout. Larger-scale experiments were conducted in another upper Yakima River tributary in 1995 to further define the potential for juvenile spring chinook salmon to affect rainbow trout growth and abundance.

Methods

Study Area

The Yakima River originates on the eastern slope of the Cascade Mountains and flows east and southeast to enter the Columbia River near Richland, Washington. Experiments were conducted in three different tributaries to the upper Yakima River; the North and Middle forks of the Teanaway River and Cooke Creek (Figure 1).

Experiments in small enclosures: 1993 and 1994

We conducted growth experiments in small enclosures in the North (1993) and Middle (1994) forks of the Teanaway River, which join and then enter the Yakima River, Washington, 282 km upstream from the confluence of the Yakima and Columbia rivers. The North Fork of the Teanaway River is 29 km long and drains a portion of the eastern slope of the Cascade Mountains covering a basin area of 246 km^2. Our 2.5 km study reach in the North Fork of the Teanaway River ranged in elevation from 750 to 780 m above sea level. Rainbow trout present in the study area are primarily resident forms with some smaller unknown portion being juvenile anadromous steelhead. For the purpose of this report all *O. mykiss* are referred to as rainbow trout unless otherwise stated. Density of rainbow trout >79 mm FL in a 100-m long section within the study reach on the North Fork of the Teanaway River was 0.031 rainbow trout/m^2 on August 2, 1993, however the distribution of rainbow trout within the 100-m long index sites was not uniform. Densities of rainbow trout observed in some areas exceeded 10 trout/m^2 (G.A.M., unpublished data). We observed juvenile spring chinook salmon within the North Fork of the Teanaway River in 1993 and 1994 5 km downstream of the study reach, though their abundance was very low (1993, 0.014/m^2; 1994, 0.002/m^2 (Pearsons et al. 1996)). The study reach was located immediately downstream of a proposed acclimation facility where spring chinook salmon smolts would be released as part of the YFP (BPA et al. 1996). Water temperatures measured in the North Fork of the Teanaway River during the study period ranged from 7 to 20.5^o C. Stream discharge near the lower boundary of the North Fork of the Teanaway River study section was 0.489 m^3/s on August 18, 1993.

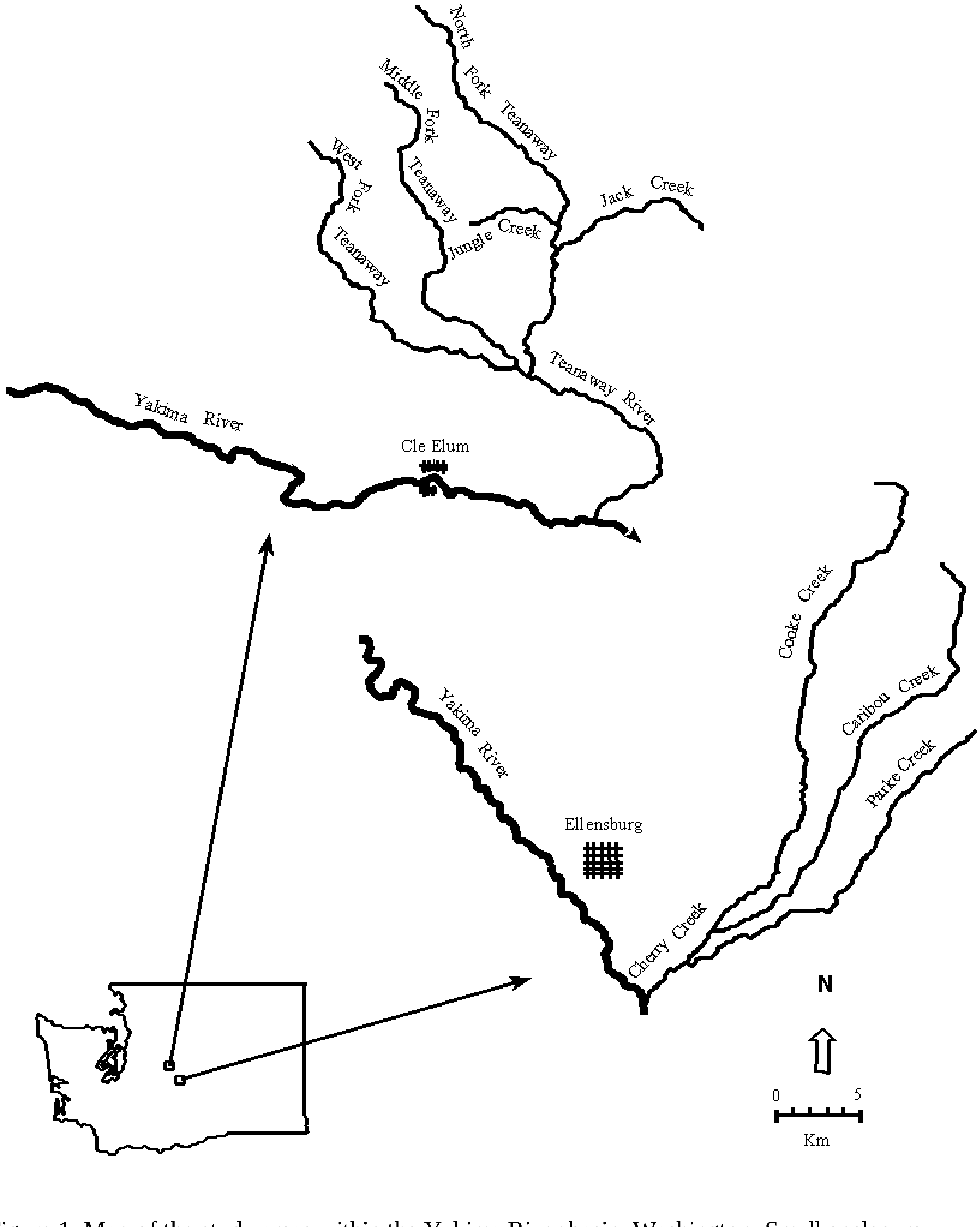


Figure 1. Map of the study areas within the Yakima River basin, Washington. Small enclosure experiments were conducted in the North and Middle forks of the Teanaway River. The stream-reach experiment was conducted in Cooke Creek.

The Middle Fork of the Teanaway River is 24 km long and drains an area of 221 km^2. The study reach in the Middle Fork of the Teanway River ranged from 687 to 695 m in elevation. Rainbow trout (>79 mm FL) density in a 100-m long section immediately below the study reach in the Middle Fork of the Teanaway River was $0.043/m^2$ on August 9, 1994. Wild juvenile spring chinook salmon were observed in the study reach in the Middle Fork of the Teanway River in 1992 and 1994, though their abundance was low (1992, $0.002/m^2$; 1994, $0.004/m^2$ (Pearsons et al. 1996)). Water temperatures measured in the Middle Fork of the Teanaway River ranged from 11.7 to 20.9^o C and stream discharge was 0.062 m^3/s on August 9, 1994. Other fish species observed in the both forks of the Teanaway River study reaches included, shorthead sculpin *Cottus confusus*, torrent sculpin *C. Rhotheus*, piute sculpin *C. beldingi*, longnose dace *Rhinichthys cataractae*, speckled dace *R. osculus*, mountain whitefish *Prosopium williamsoni*, bridgelip sucker *Catostomus columbianus*, and eastern brook trout *Salvelinus fontinalis*. Sculpins and dace were the most abundant non-salmonids in both streams (Pearsons et al. 1996). Streamside vegetation in both areas was composed of conifers and deciduous trees and shrubs. Land use in both areas was primarily undeveloped forest lands with some evidence of past timber harvests. Substrate composition was dominated by cobbles with areas of exposed sandstone bedrock.

Experiment in stream-reach enclosures: 1995

Experiments in stream-reaches were conducted in Cooke Creek, a tributary which enters Cherry Creek, which subsequently enters the Yakima River 211 km upstream from its confluence with the Columbia River (Figure 1). Cooke Creek is 38 km long and flows southward off the Colockum Ridge into the Kittitas Valley. The drainage area of Cooke Creek is 93 km^2. The 4.2 km reach where we worked ranged from 910 to 1113 m above sea level. Mean density of rainbow trout (>79 mm FL) in six 100-m long sites was $0.142/m^2$ on July 10 and 11, 1995. The only fish species captured in the Cooke Creek study reach were rainbow trout, piute sculpin *C. beldingi*, shorthead sculpin, and western brook lamprey *Lamptera richardsoni*. Spring chinook salmon do not currently inhabit the area of Cooke Creek where we worked, as it is above numerous human-made barriers (irrigation dams). Streamside vegetation was primarily deciduous trees and shrubs in the lower four sites and shifted to conifers in the upper two sites. Canopy coverage was not measured, but was nearly complete at most sites. Substrate within study sites was dominated by cobble and gravel-sized particles. Water temperatures measured during the study period ranged from 8 to 17^o C. Stream discharge was 0.099 m^3/s on July 13, 1995.

Experimental design

Experiments in small enclosures: 1993 and 1994

Experiments to determine the effects of increased natural production of spring chinook salmon on wild rainbow trout were conducted in small enclosures in the North Fork of the

Teanaway River from July 7 to August 19, 1993, and in the Middle Fork of the Teanaway River from August 2 to September 12, 1994. These experiments used a control-treatment design that focused on growth differences in groups of paired and unpaired fish placed in enclosures (McMichael et al. 1997). We placed age-0 spring chinook salmon with age-1 and -2 rainbow trout in the tests conducted during 1993, while in the tests conducted during 1994 we used age-0 fish of both species.

Enclosures were wood frame constructed of 5-cm X 5-cm stock and enclosed with galvanized wire mesh (0.95 cm) on all sides and the bottom. Enclosures used in 1993 were 91 cm high by 91 cm long by 99 cm wide. Each enclosure was divided into equal-sized (0.46 m^2) chambers by a plywood barrier. For more details on the methods see McMichael et al. (1997). In 1994, we used smaller enclosures (61 cm X 61 cm X 61 cm) of the same design and mesh size as the enclosures used in 1993. Smaller enclosures were used in the 1994 test due to the relatively small size of the test fish used in this experiment.

A solitary rainbow trout (control) was placed in one chamber of an enclosure and a treatment fish (age-0 spring chinook salmon) and a response fish (rainbow trout) were placed in the other. The combinations used in these experiments were intended to ascertain effects on response fish. The terms control, response, and treatment fish distinguish the different groups of fish in each test; “control” and “unpaired” are used interchangeably, as are “response” and “paired”. The test in 1993 was replicated 10 times and the test in 1994 was replicated 20 times.

In 1993, enclosures were randomly placed in previously selected sites within pool or run habitats in the North Fork of the Teanaway River with depths of 0.35-0.70 m and water velocities of 0.12-0.42 m/s (see McMichael et al. 1997 for more detail). These criteria were derived from fish-habitat relationships previously studied by McMichael et al. (1992). In 1994, enclosures were placed in pool and run habitats in the Middle Fork of the Teanaway River with depths of 0.14-0.38 m and water velocities of 0.06-0.43 m/s. These enclosures were placed in locations where we had previously observed juvenile rainbow trout and spring chinook salmon in close proximity (McMichael et al. 1994; Pearsons et al. 1996).

Rainbow trout used in 1993 were collected on July 7 within 50 m of the enclosures in which each was placed. We used backpack electrofishers set for pulsed direct current (PDC), 300 V, and either 30 or 60 Hz, to capture the rainbow trout. We targeted fish of 100-150 mm in fork length (FL), which bracketed the modal length (125 mm) previously observed in the study reach at that time of year (McMichael et al. 1992). We did not age the trout used in this experiment, but available age and size information from the North Fork of the Teanaway River suggests that trout between 100 and 150 mm FL are predominantly age-1 and -2 (Martin and Pearsons 1994). On the day of collection, rainbow trout were anesthetized in a 0.1-g/L solution of tricaine (MS-222), measured to the nearest mm FL and weighed to the nearest 0.1 g. Following recovery from the anesthetic, trout were placed in enclosures. Rainbow trout were allowed to acclimate to, or establish “prior residence” in, the enclosures for 2 d before a spring chinook salmon (treatment fish) was placed (on July 9, 1993) into one chamber of each enclosure. Spring chinook salmon were captured by backpack electrofishing (300 V, DC) in the main stem Yakima River near the town of Cle Elum and were immediately transported in aerated vessels approximately 30 km to the study area.

In 1994, age-0 rainbow trout were collected in Jungle Creek, a tributary of the North Fork of the Teanaway River, with a backpack electrofisher (200 V, DC) and the age-0 spring chinook salmon were collected with the same equipment and settings in the main stem Teanaway River approximately 3.2 km upstream of its mouth and transported in aerated containers to the enclosures. Rainbow trout and spring chinook salmon were placed in the enclosures on August 2, 1994. No “prior residence” was given to rainbow trout in the 1994 experiment because both species were age-0 and spring chinook salmon typically emerge before rainbow trout in our study area (Pearsons et al. 1996). Table 1 shows the relative mean sizes of the fishes deployed in these tests.

Debris was cleaned off enclosure screening every 24-48 h. Enclosures in which fish either died or escaped were discarded from the analyses. Fish were missing in one of the replicates in 1993 and in nine of the enclosures in 1994. We assumed most missing fish had died, although some may have escaped.

On August 19, 1993, 42 d after the control and response fish had been placed in enclosures in the North Fork of the Teanaway River, all fish were collected from the enclosures, killed in a lethal concentration of MS-222, measured to the nearest mm FL, and weighed to the nearest 0.1 g. Fish were recovered from the enclosures in the Middle Fork of the Teanaway River on September 12, 1994, 41 d after they had been put into enclosures. Capture protocol were described by McMichael et al. (1997) for the experiment in 1993. In 1994, fish were simply netted from the enclosures at the termination of the experiment.

Table 1. Beginning mean lengths and weights (SDs and ranges in parentheses) of test fish used in enclosure experiments with wild rainbow trout and spring chinook salmon. Sample sizes were 10 fish per group in 1993 and 20 fish per group in 1994. In 1995, there were 72 age-1 rainbow trout each in control and response groups and 156 chinook salmon.

Year	Group	Species	Fork length (mm)	Weight (g)
1993	Control	Rainbow trout	123 (12.1,108-145)	20.7 (6.8,14.6-34.1)
	Response	Rainbow trout	125 (15.1,106-149)	23.5 (8.5,14.2-37.5)
	Treatment	Chinook salmon	68 (4.6,61-77)	4.1 (0.9,3.1-6.1)
1994	Control	Rainbow trout	84 (7.4,71-98)	6.0 (1.7,3.2-9.6)
	Response	Rainbow trout	79 (7.7,74-93)	5.1 (1.8,3.5-8.2)
	Treatment	Chinook salmon	81 (8.1,72-99)	5.7 (1.5,3.7-8.6)
1995	Control	Rainbow trout	94 (10.6,73-115)	9.3 (3.2,4-17)
	Response	Rainbow trout	98 (8.3,80-115)	10.3 (2.9,5-18)
	Treatment	Chinook salmon	75 (8.7,54-96)	5.3 (1.9,2-10)

Experiment in stream-reach enclosures: 1995

In an effort to reduce possible enclosure effects on experimental results (McMichael et al. 1997) and to investigate potential competitive impacts in larger areas, stream reach enclosures were used in 1995. Six 100-m long reaches of Cooke Creek were enclosed on up- and downstream boundaries by 6-mm square mesh galvanized hardware cloth weirs supported by 12-mm diameter rebar posts pounded into the substrate. The hardware cloth was bent resulting in a 90 degree angle to form a 30 to 50 cm wide apron that was placed on the upstream side of the weir along the substrate. These aprons were then covered with gravel and cobbles to minimize fish emigration or immigration. Enclosed reaches were spaced approximately 400-600 m apart.

Initial multiple removal population estimates (Zippin 1958) were conducted in these six sites on July 10 and 11, 1995. All salmonids were anesthetized, measured to the nearest mm FL, weighed to the nearest g, given a section-specific partial fin clip, and placed in a perforated bucket in the stream channel for recovery prior to release back into the study section. All non-salmonids (Cottids) were grouped into species and age class groups, weighed, and released back into the enclosures.

Following the initial population estimates, age-0 wild spring chinook salmon were collected with backpack electrofishers (300 V 60 Hz PDC) in the Yakima River near Cle Elum and transported in aerated vessels and released into three (A, C, and E) of the six study sites. Stocking densities of spring chinook salmon were equal to the estimated number of rainbow trout in each site to reflect the differential capacity of each reach. This effectively doubled the numerical density of salmonids in the treatment sites. Population estimates were repeated in these sites seven weeks later, on August 28, and again after about another seven weeks on October 9 and 10. Hardware cloth weirs were cleaned and maintained for the first seven weeks and were removed on August 30. Thus fish movement was not restricted between August 30 and the termination of the experiment on October 10. This seven week period was chosen to be consistent with the timing and duration of the experiments conducted in 1993 and 1994 in the small enclosures. The weirs were intended to restrict movement of experimental fishes during the first seven weeks of the experiment, thereby maintaining stocking densities while forcing all fish to locate microhabitats within the enclosed areas (Fausch 1984).

Rainbow trout size structure, growth, length-weight relationships, and abundance were compared between groups of fish in control and treatment stream reaches. Emphasis was placed on analyses of size/age-groups of rainbow trout (those fish that were similar in size or smaller) we expected to interact most intensely with the introduced salmon. Length frequency histograms were used to visually differentiate between apparent age groups for each sample site and date. Age-0 and -1 age groups of rainbow trout were analyzed separately to increase our abilities to detect growth differences. No age-0 rainbow trout were captured on the July 10 and 11 sampling dates and, based on length frequency histograms, age-1 fish were considered to be those less than 115 mm FL at that time. Age-0 rainbow trout on the August 28 sample date were considered to be those less than 80 mm FL, while rainbow trout between 80 and 120 mm FL were considered age-1. On October 10, rainbow trout less than 90 mm FL were assumed to be age-0, while those between 90 and 130 mm were considered to be age-1. Table 1 shows the mean fork lengths and weights for age-1 rainbow trout as well as spring chinook salmon at the beginning of the stream-reach experiment. No age-0 rainbow trout were captured prior to the introduction of the juvenile spring chinook salmon into treatment sites, so pre-treatment length and weight data for the age-0 trout were not collected. Abundance estimates of age-0 rainbow trout were not made due to very low capture efficiency on these small fish.

Data Analyses

Specific growth rate (SGR) for fish in small enclosure experiments in 1993 and 1994 was calculated using the following equation (Ricker 1975):

$$SGR = (\log_e W_t - \log_e W_o)/t;$$

W_t = weight (g) at the end of the period, W_o = weight (g) at the beginning of the period, and t = time (days).

One-tailed paired t-tests were performed to test for differences in SGR among control and response fishes in the small enclosure experiments in 1993 and 1994. Statistical power analyses (Snedecor and Cochran 1981; Peterman 1990) for t-tests involving control and response fish growth were performed to aid in the interpretation of the results.

Data from 1995 stream-reach experiments in Cooke Creek were analyzed by using analysis of covariance (ANCOVA), with treatment being the main effect and elevation being a covariate, to determine the effect of added juvenile spring chinook salmon on wild rainbow trout fork length seven and 14 weeks after the introduction of the juvenile salmon. Instantaneous growth rate (IGR) for treatment and control groups of rainbow trout and spring chinook salmon were calculated using the following equation (Ricker 1975):

$$IGR = (\log_e (FL_{t2}) - \log_e (FL_{t1})) / t_2 - t_1;$$

FL_{t1} and FL_{t2} are mean fork lengths (mm) at time 1 (t_1) and time 2 (t_2). One-tailed paired t-tests were used to compare IGRs between treatment and control groups of rainbow trout during each time period.

Length-weight relationships of age-1 rainbow trout in control and treatment sites were also compared by using ANCOVA, with treatment being the main effect and elevation and length being covariates. Length-weight relationships were not examined for age-0 rainbow trout due to the relatively low weights of these fish (often less than 1 g) and the associated rounding errors resulting from rounding up to the nearest gram. To determine if regression line slopes were homogenous, we tested for interactions between treatment and elevation for all comparisons and found none. Effects of the treatment on abundance and biomass estimates of rainbow trout were tested for significance using multi-factor analysis of variance (ANOVA) with treatment and sample week as the main effects. The critical level for all statistical tests was $P < 0.05$.

Results

Experiments in small enclosures: 1993 and 1994

Wild rainbow trout (age-1 and -2) paired with wild age-0 spring chinook salmon did not grow more slowly than unpaired rainbow trout in 1993 (Table 3; t = 0.37, df = 8, P = 0.360). Similarly, in the 1994 experiment examining effects of age-0 spring chinook salmon on age-0 rainbow trout, SGRs of paired rainbow trout were not significantly lower than those of their unpaired cohorts, (t = -0.91, f = 10, P = 0.190; Table 2). However, the power of tests in both years was low (1993, 0.109; 1994, 0.232). On average, fish of all groups lost weight during the tests.

Table 2. Mean specific growth rates (SDs and ranges in parentheses) of wild rainbow trout and spring chinook salmon in small stream enclosures. Samples sizes (number of replicates) were 9 fish per group in 1993 and 11 fish per group in 1994.

Year	Group	Species	Specific growth rate (d^{-1})
1993	Control	Rainbow trout	-0.0019 (0.0027,-0.0046-0.0041)
	Response	Rainbow trout	-0.0023 (0.0018,-0.0050-0)
	Treatment	Chinook salmon	-0.0012 (0.0021,-0.0047-0.0016)
1994	Control	Rainbow trout	-0.0017 (0.0041,-0.0128-0.0035)
	Response	Rainbow trout	-0.0000[a] (0.0028,-0.0030-0.0056)
	Treatment	Chinook salmon	-0.0046 (0.0019,-0.0068--0.0018)

[a] actual value = -0.000029

Experiments in stream-reach enclosures: 1995

Growth of age-0 and age-1 rainbow trout was not significantly reduced by the introduction of juvenile spring chinook salmon in stream-reach experiments in Cooke Creek in 1995. Mean lengths of age-0 rainbow trout were not significantly different in treatment and control sites after seven or 14 weeks (week 7, F-ratio = 1.022, P = 0.318; week 14, F-ratio = 0.786, P = 0.387; Figure 2). Differences in IGR of age-0 rainbow trout in control and treatment sites were also not statistically significant (t = -0.751 , df = 2 , P = 0.265; Figure 3). The power of this test was low (0.19), however. Mean FLs of age-1 rainbow trout in sites where spring chinook salmon were stocked (treatment sites) were not significantly affected by the treatment after seven or 14 weeks (week 7, -ratio = 0.364, P = 0.553; week 14, F-ratio = 0.037, P = 0.850;

Figure 4). Instantaneous growth rates of age-1 rainbow trout in treatment sites were not significantly different from control groups between week 0 and 7, and between week 7 and 14 (week 0 - 7, t = -1.612 , df = 2 , P = 0.124; week 7 - 14, t = -2.117 , df = 2 , P = 0.084; Figure 3). Power of these tests were higher (week 0 - 7, 0.49; week 7 - 14, 0.68) than for the age-0 test. In addition, length-weight relationships for age-1 rainbow trout were unaffected by the addition of juvenile spring chinook salmon in treatment sites (week 7, F-ratio = 0.517, P = 0.481; week 14, F-ratio = 2.260, P = 0.135).

Elevation of experimental site appeared to influence size of trout more than the treatment effect of juvenile spring chinook salmon presence. In general, a negative relationship existed between trout length and site elevation. The strength of the relationship between fish length and elevation was greatest during the October sample period (week 14) for both age-0 and age-1 rainbow trout. Figure 2B shows the relationship between site elevation and mean length of age-0 rainbow trout on October 9 and 10 (R = -0.665, df = 83, $P \leq 0.001$). Length of age-1 rainbow trout was also significantly negatively correlated with site elevation in October (Figure 4B, R = -0.298, df = 175, $P \leq 0.001$).

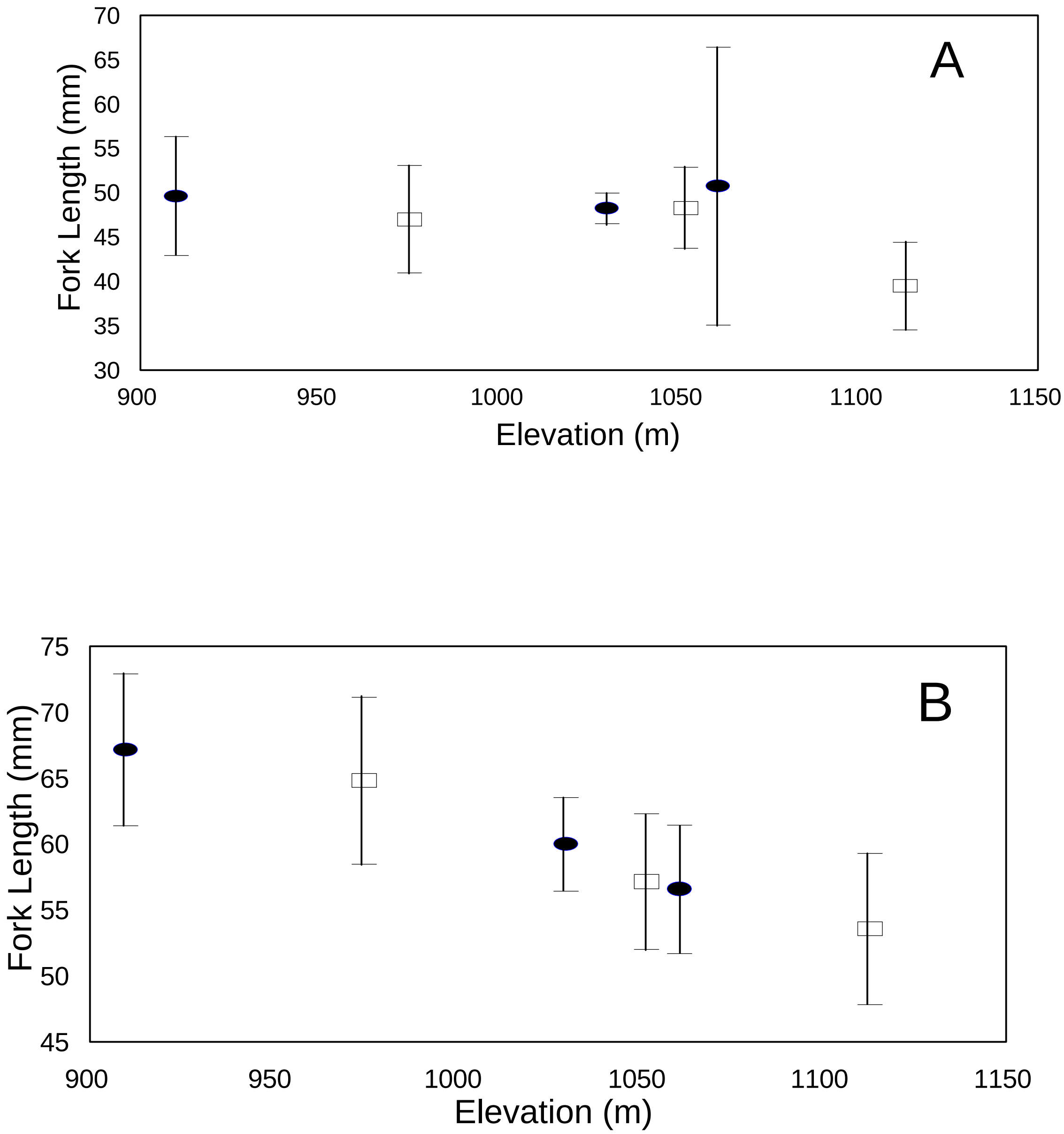


Figure 2. Mean fork length (mm) of age-0 rainbow trout versus site elevation (m) in treatment and control stream-reach enclosures in Cooke Creek seven (A) and 14 (B) weeks after the introduction of juvenile spring chinook salmon in treatment sites. Treatment site means are represented by closed circles, while control site means are marked with open squares. Vertical bars represent $\pm$ 1 SD.

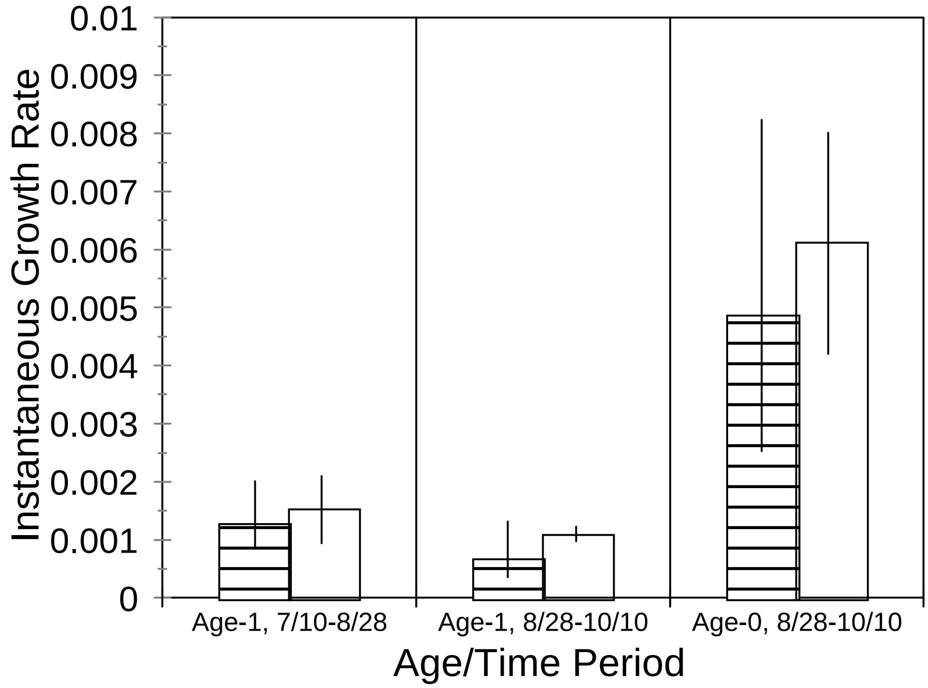
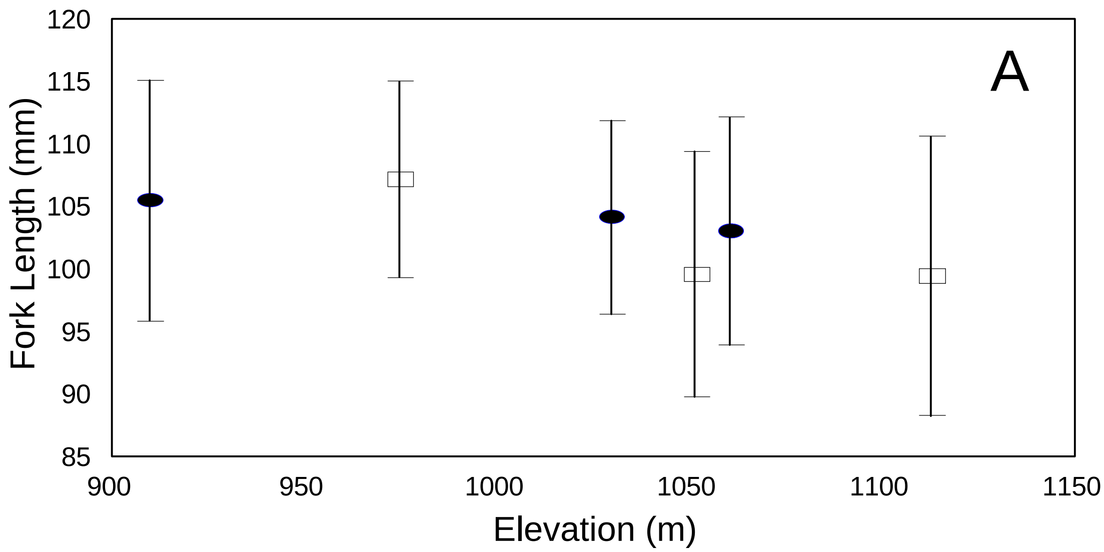


Figure 3. Mean instantaneous growth rate of age-0 and age-1 rainbow trout in treatment and control sites in the stream-reach experiment. Treatment groups are represented by shaded bars, while control sites are open bars. Vertical bars represent ± 1 SD.



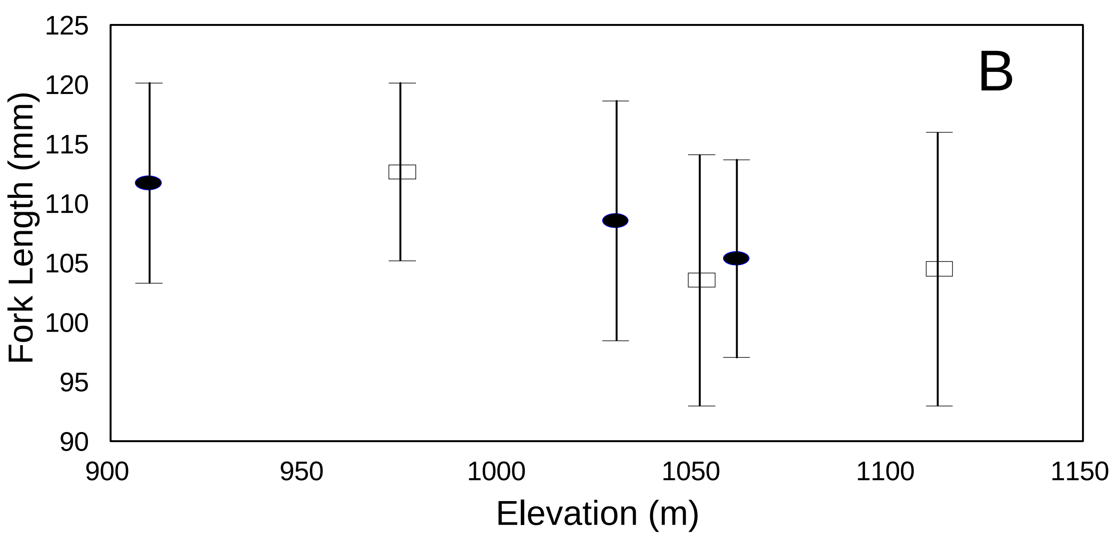


Figure 4. Mean fork length (mm) of age-1 rainbow trout versus elevation (m) in treatment and control stream-reach enclosures in Cooke Creek seven (A) and 14 (B) weeks after the introduction of juvenile spring chinook salmon in treatment sites. Treatment site means are represented by closed circles, while control site means are marked with open squares. Vertical bars represent ± 1 SD.

Estimated abundance of rainbow trout was not affected by the addition of juvenile spring chinook salmon in treatment sites over the course of the experiment (F-ratio = 1.171, P = 0.298). Abundance of age-1 and older rainbow trout in control and treatment sites was generally stable between July (week 0) and October (week 14), with a slight increase in four of the six sites in the late August samples (week 7)(Figure 5A). Exceptions to this were the upper two sites (sites E and F), where trout abundance in the uppermost treatment site (E) decreased 11% between week 0 and week 7 and 20% between week 0 and week 14. In contrast, the uppermost control site (F) showed 21% and 27% increases in trout abundance by the week 7 and 14 sample dates, respectively. The estimated biomass of rainbow trout was also unaffected by the introduction of juvenile spring chinook salmon (F-ratio = 0.413, P = 0.538; Figure 5B).

Spring chinook salmon densities decreased in all sites as the experiment progressed (Figure 6). Numbers of salmon present seven weeks after their release ranged from 34 to 38% of the number released. By the termination of the experiment, estimates of spring chinook abundance ranged from nine to 14% of the numbers released. One chinook salmon was captured in a control site during week 7, and four individuals were captured during week 14 in sites where they had not been released. Thus not all loss within treatment sites was attributable to mortality, as some fish moved out of our sites both before and after weirs were removed. Spring chinook salmon present in treatment sites after seven and 14 weeks appeared to be growing. Mean IGR of spring chinook salmon was 0.0016 between July 13 and August 28 and 0.0019 between August 28 and October 10, 1995.

Sculpin species (piute and shorthead sculpins) biomass did not appear to be affected by the addition of juvenile spring chinook salmon. Abundance and biomass for these species was not rigorously estimated and data presented are total weight of all sculpin species netted for each date and site. Visual examination of the data indicates that the biomass of sculpins in the sites A, B, and D declined from week 0 to 14 (Figure 7), while all sites showed a decrease in biomass between the week 7 and 14. By week 14, sculpin biomass was positively, though not significantly, related to site elevation R = 0.799, P = 0.057; Figure 7).

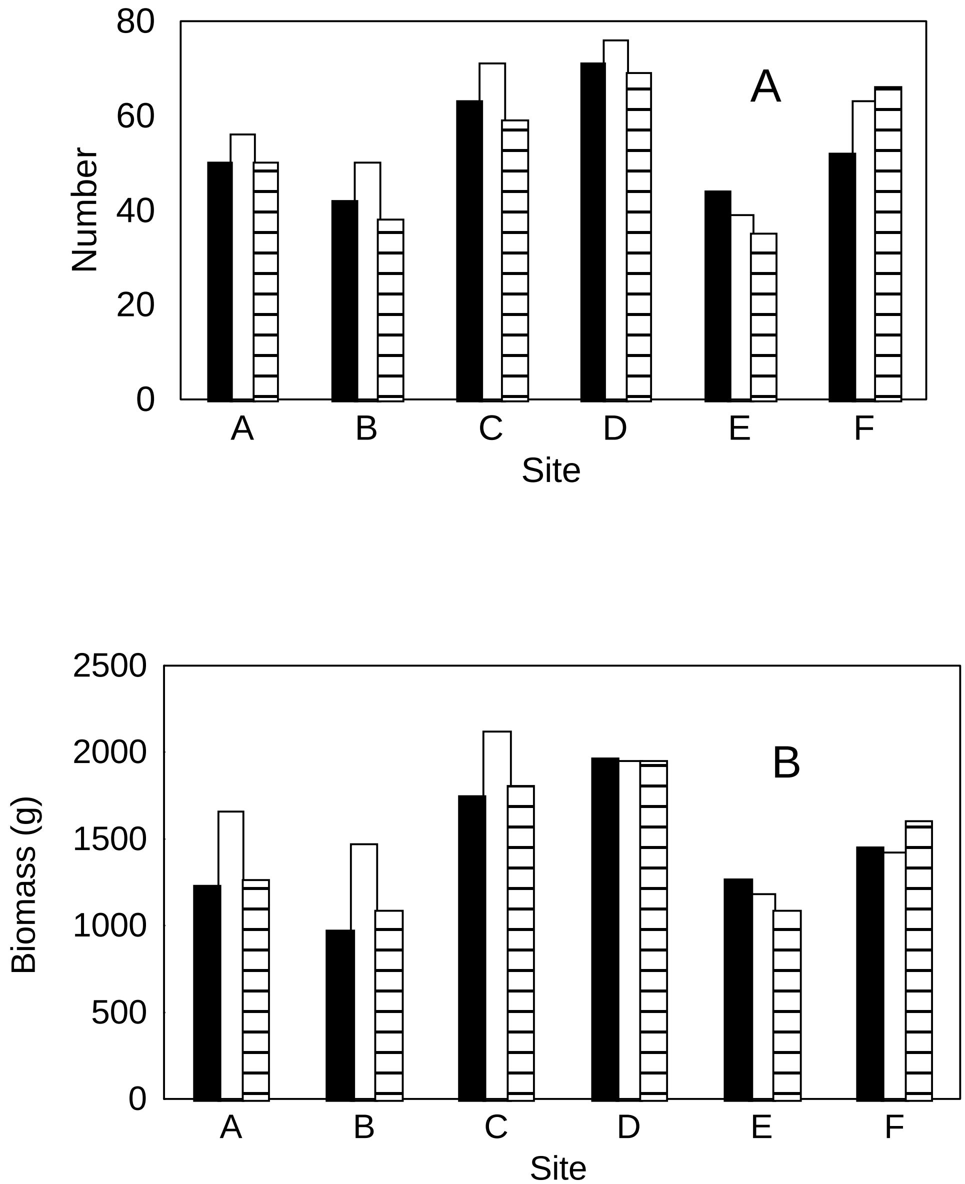


Figure 5. Estimated abundance (A) and biomass (B) of rainbow trout > 79 mm FL in stream-reach experiment sites on July 10-11 (solid bars), August 28 (open bars), and October 9-10 (cross-hatched bars), 1995.

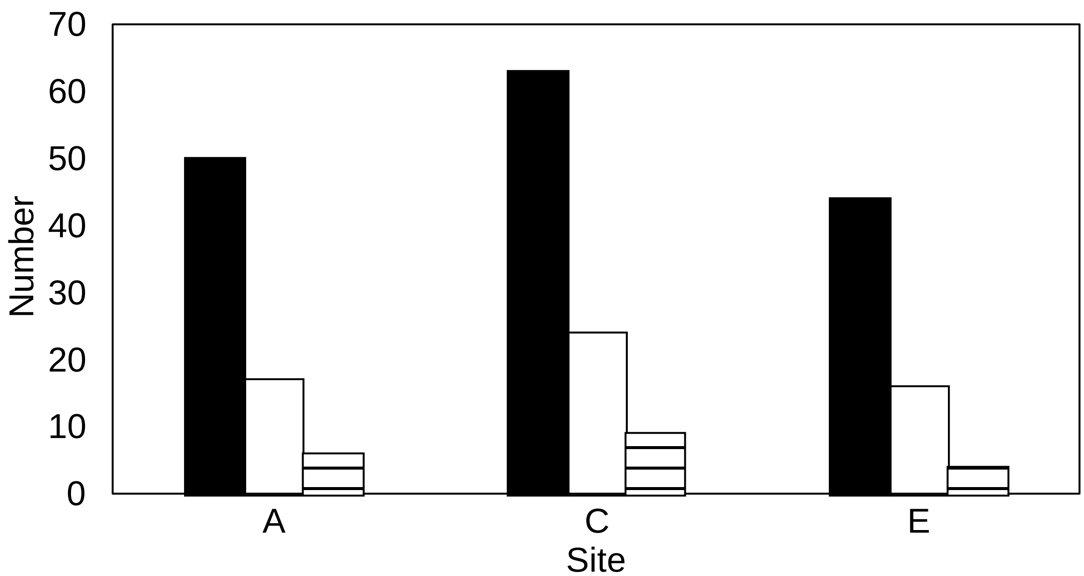


Figure 6. Number of juvenile spring chinook salmon (SPC) released into stream-reach enclosures in Cooke Creek on July 13 and 14, 1995 (solid bars), and subsequent population estimates of spring chinook salmon on August 28 (week 7; open bars)) and October 9 and 10 (week 14; cross-hatched bars)), 1995.

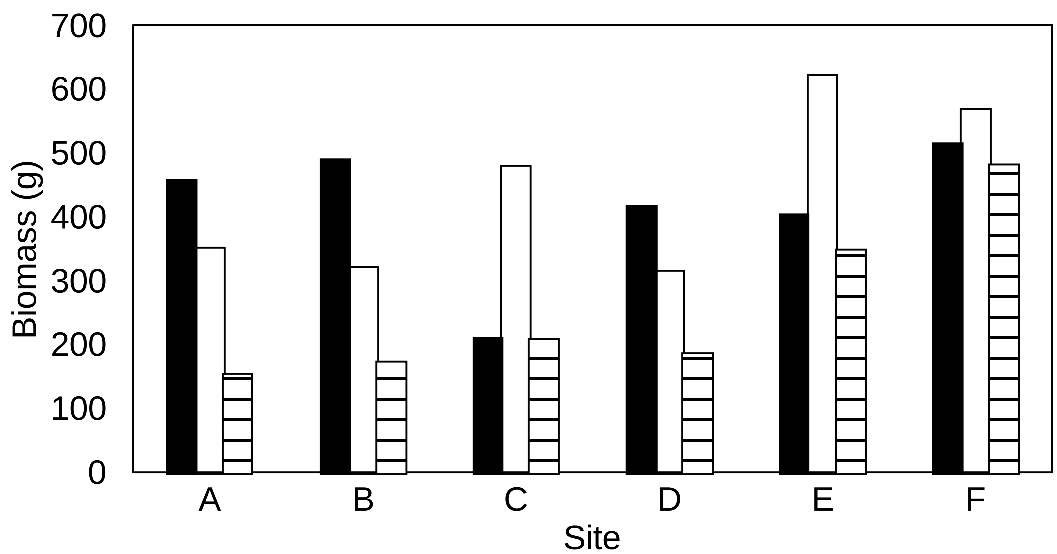


Figure 7. Sculpin species biomass (piute and shorthead sculpins combined) in stream-reach enclosure sites on July 10-11 (solid bars), August 28 (open bars), and October 9-10 (cross-hatched bars), 1995.

Discussion

Experiments we conducted in response to our concern that successful supplementation of spring chinook salmon in the upper Yakima basin might adversely affect the wild rainbow trout populations in tributaries, indicated that rainbow trout growth was not affected. These experiments were conducted over three years, at two spatial scales, and in three different streams which suggests our results can be applied to a variety of streams. All of our experiments tested for impacts to trout growth and/or abundance when spring chinook salmon density was numerically equal to trout density. However, densities of juvenile spring chinook salmon in the stream-reach experiments decreased by 62 to 66% during the first half of the study period. It is difficult to predict whether we would have observed impacts on trout if we had introduced or maintained much higher numerical densities of salmon. The intensity of competition should be affected by the density of fish in a given area (Li and Brocksen 1977; Harvey and Nakamoto 1996). It is possible that higher initial, or more stable, spring chinook salmon densities may have influenced the results of our experiments. The potential for competition to occur in the small enclosure experiments should have been accentuated by the small dimensions of the enclosures used in 1993 and 1994. Fish densities in the enclosures were 4.35 to 10.8 /m^2, while mean densities of rainbow trout in the stream areas where the small enclosures were used was between 0.031 and 0.043/m^2. However, distribution of trout within our natural stream reaches was not uniform. The densities resulting from the patchy distribution of trout, with larger concentrations in the most suitable habitat (e.g., the upstream margin of a pool), were similar to the densities within the small enclosures (G.A.M., personal observation). Therefore, the densities within our small enclosures may not have been high when viewed at the meso- or microhabitat scale. Other studies have shown that stream salmonids are patchily distributed. Grant and Kramer (1990) presented information in a review article supporting the concept that stream salmonids are not distributed uniformly throughout stream reaches. They suggested that juvenile salmonids follow a more patchy distribution pattern dependent on factors such as territory size, and food and habitat availability. While fish in natural environments can respond to changing conditions by moving, fish in small enclosures cannot. This lack of freedom to move may have affected fish growth in small enclosures.

Mean SGR for all groups of fish in small enclosure experiments was negative, which suggest the quantitative results observed in our small enclosure study should not be extrapolated to natural populations. Some individuals did gain weight, but most (78%) lost weight during those experiments. Weight loss by stream salmonids has been reported for fish placed in enclosures (Miller 1952; Fausch and White 1986; Harvey and Nakamoto 1996; McMichael et al. 1997). Enclosures may restrict natural fish behavior and movement related to environmental factors which may in turn affect their growth (Edmundson et al. 1968; McMichael et al. 1997). Nonetheless, all fish in our tests were subjected to standardized confinement protocol, making comparisons between control and response groups meaningful.

The most likely reason for our observance of no impact to rainbow trout and sculpins was resource partitioning. Resource partitioning between species may have decreased the frequency and intensity of interspecific behavioral interactions, and subsequently reduced the impact of such competitive interactions on performance measures such as growth or biomass.

These results are in agreement with earlier studies that suggested that rainbow trout and chinook salmon minimize competitive effects through resource partitioning (Everest and Chapman 1972; Hillman et al. 1989; McMichael et al. 1997). Co-evolved salmonid species have often been shown to select different niches (Lister and Genoe 1970; Everest and Chapman 1972; Allee 1982; Dolloff and Reeves 1990), precluding or reducing the potential for behavioral interactions. Underwater observations of both species in our experiments supported this thesis. Snorkeling observations revealed that juvenile spring chinook salmon held higher in the water column and over deeper water than rainbow trout in the North Fork of the Teanaway River and in Cooke Creek (G.A.M., unpublished data).

Another possible explanation for our failure to detect significant impacts on growth of rainbow trout when forced to exist with spring chinook salmon juveniles in the stream-reach tests may be that the environment where we conducted this part of our research was possibly more suited to rainbow trout than to spring chinook salmon. The site elevation, gradient, and habitat types (predominantly riffles) may have favored the rainbow trout (Edmundson et al. 1968; Everest and Chapman 1972; Roper et al. 1994). It is possible that the Cooke Creek study reach was outside of the normal historical distribution of spring chinook salmon. However, spring chinook salmon have been studied at elevations up to at least 1487 m (determined from study area information presented by Edmundson et al. 1968), which is over 350 m higher than our highest elevation site. The same types of experiments in lower elevation areas in the Yakima basin, where the habitat may be more suitable for spring chinook salmon, may have produced different results. However, the survival and growth of the spring chinook salmon used in the stream-reach tests indicate that those fish remaining in the study sections were able to grow at rates similar to spring chinook salmon found in several lower elevation tributary sites within the upper Yakima basin (Pearsons et al. 1996).

In the stream-reach experiment, we found that site elevation explained more of the variation in trout size on a given date than did the treatment effect. Apparently, small differences in stream temperature or other factors that are expressed along an elevational gradient, may have affected growth of age-0 and -1 trout in Cooke Creek. Other researchers have also observed differential growth in salmonids relative to elevation; where salmonids at lower elevations grow faster than those at higher elevations (Mullan et al. 1992; Peven et al. 1994). These elevational differences may have confounded our results in the stream-reach experiment. On average, treatment sites were lower (mean, 1000 m) than control sites (mean, 1047 m), even though the sites were alternated. Due to differences in rainbow trout size at the beginning of the experiment, any growth impacts in treatment sizes may have been undetectable by simply examining fish length at a point in time. When we standardized for differences in length at the beginning of the experiment by calculating IGRs for the rainbow trout, we found that mean IGRs for rainbow trout in treatment sites were consistently lower than in control sites. These differences were not significant, but the power of one of the three tests was low.

Estimated biomass of sculpin species was also unaffected by the introduction of the juvenile spring chinook salmon in our study. Niche separation between benthic species such as sculpins and mid-water species such as juvenile chinook salmon would be expected to preclude most agonistic interactions between these two species. However, an exception to this might occur around the time of spring chinook salmon emergence, when sculpin prey upon newly hatched fry (Ricker 1941; Hunter 1959; Patten 1975; Hillman 1989). In this case, increased abundance of spring chinook salmon resulting from supplementation activities might be expected to have a positive effect on sculpin growth.

The primary implication of our findings for ongoing and planned anadromous salmonid stocking programs is that resident rainbow trout stocks may have a refuge from competitive interactions impacts from juvenile spring chinook salmon in high elevation areas (e.g., over 700 m). Increases in abundance of naturally produced juvenile spring chinook salmon (as might be expected in a successful supplementation program) up to a point that is equal to the numerical density of wild rainbow trout in a given area did not impact trout growth or abundance in high elevation areas of tributaries. Further research is necessary to be able to assess ecological risks to resident and anadromous rainbow trout relative to spring chinook salmon supplementation and hatchery programs in areas that may be outside of high elevation refugia. For example, increased densities of spring chinook salmon may impact rainbow trout growth or abundance in main stem rivers where high agonistic interaction rates between these two species have been observed (Pearsons et al. 1996).

As hatchery programs increasingly come under fire for having played a role in the declines of Pacific salmon (Lichatowich and McIntyre 1987; Meffe 1992; Nehlsen et al. 1991; Flagg et al. 1995; White et al. 1995), and as they may be utilized in the future to try to rebuild or restore depressed stocks (Clune and Dauble 1991; Cuenco et al. 1993; BPA et al. 1996), more attention must be payed to the potential impacts of hatchery programs on pre-existing wild fish populations. Only by increasing our understanding of the ecological interactions resulting from artificial propagation programs, will we be able to strategically manage hatchery operations to achieve management objectives while maintaining wild stocks.

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Chapter 10

Gene flow between resident and anadromous *Oncorhynchus mykiss* in the Yakima basin: ecological and genetic evidence

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Abstract

We examined ecological and genetic evidence to determine the potential for gene flow between resident rainbow trout and anadromous steelhead trout (*Oncorhynchus mykiss*) in the Yakima basin. Electrofishing, trapping, radio telemetry, redd surveys, and snorkeling methods were used to determine the spatial and temporal distribution of *O. mykiss* spawning. Steelhead had a smaller spatial spawning distribution than rainbow trout, but it was entirely within the range of rainbow trout spawning areas. Furthermore, the spawn timing of rainbow and steelhead trout was positively related to elevation and no differences in timing were detected between forms ($P>0.05$). In addition, we documented many instances of interbreeding between rainbow and steelhead trout. Genetic evidence, using starch gel electrophoresis, also indicated that rainbow trout and steelhead interbreed. Rainbow trout were genetically indistinguishable from sympatric steelhead collected in the North Fork of the Teanaway River. In addition, estimates of hatchery and wild fish admixtures in naturally produced *O. mykiss* indicated that hatchery rainbow trout had previously spawned with steelhead; and hatchery steelhead had previously spawned with rainbow trout. We speculate that the magnitude of gene flow between rainbow and steelhead trout may vary spatially and temporally, depending in part, on the number of anadromous steelhead that spawn within an area or year and the number of steelhead offspring that rear and mature entirely within freshwater. Our work suggests that aboriginal rainbow trout should be included within a steelhead ESU because the two forms are not reproductively isolated when in sympatry.

Introduction

Oncorhynchus mykiss may possess the most diverse life history patterns of any of the Pacific salmonid species. Much of the variation can be attributed to the migrational tendencies that have been observed within the species. One dominant life history form of *O. mykiss* rears in fresh water, migrates to the ocean, and then returns to spawn in freshwater (Shapovalov and Taft 1954; Withler 1966, Peven et al. 1994). This anadromous life history form of *O. mykiss* is called steelhead (Wydoski and Whitney 1979; Behnke 1992). The other dominant life history form of *O. mykiss* spends its entire life in freshwater and is commonly referred to as rainbow trout (Wydoski and Whitney 1979; Behnke 1992). Within each dominant life history form there is additional variation in migrational tendencies. For instance, some steelhead will migrate between the ocean and freshwater up to five times in order to spawn (Shapovalov and Taft 1954); some steelhead may rear in freshwater for up to seven years before migrating to the ocean (Peven et al. 1994), and some "steelhead" may mature in freshwater without migrating to the ocean (Shapovalov and Taft 1954, Mullan et al. 1992a, Viola and Schuck 1995). In addition, some rainbow trout are extremely sedentary whereas others vary in the amount they migrate - some migrating considerable distances particularly as juveniles during the spring and as adults before spawning (Moring 1978; Moring 1993, Bartrand et al. 1994). It is presently unclear whether ecological or genetic factors are responsible for the expression of the variation in migrational tendencies within *O. mykiss*, but both factors are probably influential. One mechanism that may influence the expression of intermediate or a mixture of migrational tendencies is interbreeding between steelhead and rainbow trout (Moring 1978).

There is considerable uncertainty about whether resident and anadromous *O. mykiss* interbreed despite being the same species (Busby et al. 1996). Both forms are found together throughout much of their range and spawn primarily during the spring (Rounsefell 1958; Scott and Crossman 1973; Wydoski and Whitney 1979). However, rainbow trout do spawn during other times, particularly in spring-fed streams and in streams that have had hatchery stocking (Biette et al. 1981). Aboriginal rainbow and steelhead trout spawn in clean, cold, fast moving streams and rivers from northern Mexico to southeastern Alaska (MacCrimmon 1971; Scott and Crossman 1973; Wydoski and Whitney 1979). Thus, although rainbow and steelhead trout generally spawn at similar times and places little is known about fine scale spatial and temporal differences (see Neave 1944; Shapovalov and Taft 1954). High spatial and temporal overlap in spawning among rainbow and steelhead trout increases the potential for these forms to interbreed. Examination of genetic data can also be used to determine past interbreeding.

Genetic comparisons of rainbow and steelhead trout collected within a basin reveal differences in *O. mykiss* which are correlated with geography. In general, rainbow and steelhead trout collected from similar geographic locations are more genetically similar to each other than rainbow and steelhead trout collected from different geographic locations (Busby et al. 1996). For example, rainbow and steelhead trout collected in the Columbia basin east of the Cascade mountains are more similar to each other than to rainbow trout and steelhead collected west of the Cascades (Allendorf 1975). At a smaller scale, in the Deschutes River basin, Currens et al. (1990) found that there were significant differences in allozyme frequencies between rainbow trout collected above a barrier falls and *O. mykiss* collected below the barrier which were probably a combination of rainbow and steelhead trout. Somewhat contrary to the theory that rainbow and steelhead trout share the same gene pool, Campton and Johnston (1985) found that rainbow trout in the upper Yakima basin were primarily an admixture of hatchery and native rainbow trout despite the stocking of hatchery steelhead into the basin. Furthermore, Neave (1944) found significant differences between average scale counts of rainbow and steelhead trout collected from the same place in the Cowichan river basin, B.C. Neave (1944) believed that these differences were inherited. Unfortunately, we know of no published studies that have compared the genetics of wild rainbow and steelhead trout that were collected from populations spawning in sympatry.

Determination of interbreeding between rainbow and steelhead trout has important management implications. As early as 1944, and most likely before, Neave (1944) suggested that rainbow and steelhead trout should be managed as different species in order to protect both forms. More recently, the National Marine Fisheries Service is deciding whether to include rainbow trout as part of a steelhead evolutionarily significant unit (ESU; National Marine Fisheries Service 1996; Busby et al. 1996). If interbreeding between rainbow and steelhead trout occurs frequently then rainbow trout might be considered part of a steelhead ESU. Furthermore, the abundance of rainbow trout may influence whether a steelhead ESU is worthy of protection by the federal government. Thus, depending on whether rainbow and steelhead trout interbreed and the abundance of rainbow trout, an ESU may or may not receive federal protection.

The purpose of our study was to determine the potential for gene flow between resident and anadromous forms of *O. mykiss* in the Yakima River basin. We did this by determining the spatial and temporal overlap of spawning, comparing the genetic similarity of rainbow and steelhead trout collected from the same area, and by examining allozyme data for resident x anadromous matings using hatchery fish as genetic markers.

Methods

Study Area

The Yakima basin drains an area of 15,941 km^2 and enters the Columbia River near Richland, Washington. Major sub-basins of the Yakima River include the Satus, Toppenish, Naches, and upper Yakima (Figure 1). It is estimated that almost 100% of *O. mykiss* spawners in the Satus and Toppenish sub-basins are steelhead (Hubble 1992; Joel Hubble, YIN, personal communication), and less than 1% of *O. mykiss* spawners in the upper Yakima basin are steelhead (WDFW, unpublished data). The proportion of *O. mykiss* that spawn in the Naches sub-basin as steelhead is believed to be intermediate between Satus/Toppenish and the upper Yakima River (based on reports from anglers). The lower portion of the upper Yakima basin is bordered by Roza Dam which was constructed in 1939. The adult fish ladder at Roza Dam was impassable during much of the winter migration of steelhead prior to 1987, but winter passage was improved

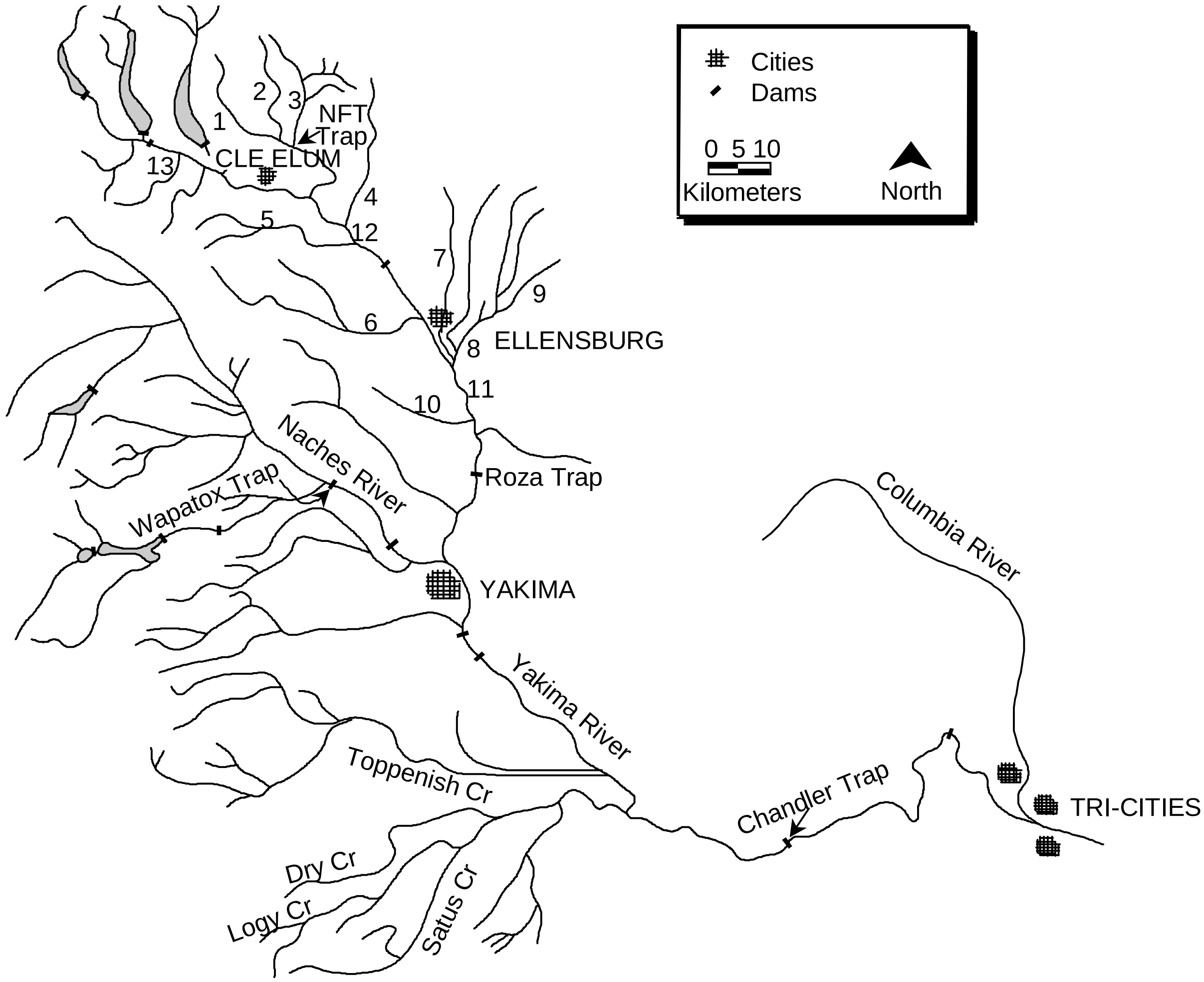

Figure 1. Map of the study area. Numbers refer to study locations in the upper Yakima basin (1=West Fork of the Teanaway River, 2=Middle Fork of the Teanaway River, 3=North Fork of the Teanway River, 4=Swauk Creek, 5=Taneum Creek, 6=Manastash Creek, 7=Wilson Creek, 8=Cherry Creek, 9=Badger Creek, 10=Umtanum Creek, 11=sections 1-3 of the Yakima River, 12=section 4-6 of the Yakima River, 7=section 7 of the Yakima River).

during 1987. Rainbow trout located upstream of Roza Dam are abundant and provide Washington's best wild stream-trout fishery (Krause 1991; Probasco 1994). The Naches basin has several low head irrigation diversions which span the entire channel width. The Satus basin has no dams.

Non-local rainbow trout and steelhead have been stocked in the Yakima basin since the early 1940's (Campton and Johnston 1985). The stocking of hatchery rainbow and steelhead in the Yakima basin has been prolific. Between 1950 and 1980, 3,400,000 rainbow trout and 830,000 juvenile steelhead were stocked into the Yakima basin from non-native hatchery populations (Campton and Johnston 1985). Between 1981 and 1994, about 966,060 rainbow trout and 1,268,525 steelhead were stocked into the Yakima basin from native and non-local hatchery sources (Washington Department of Fish and Wildlife stocking records). Most of the hatchery rainbow trout stocked into the Yakima basin were derived from a few sources in northern California and hatchery steelhead smolts stocked into the Yakima basin were derived mainly from fish native to the Washougal and Klickitat rivers (Crawford 1979; Campton and Johnston 1985). These hatchery populations are genetically distinguishable from each other and from *O. mykiss* in the Yakima basin (Campton and Johnston 1985; Pearsons et al. 1994). Non-local hatchery populations contain alleles that are not found in aboriginal Yakima River populations and alleles common to hatchery and aboriginal populations are found in different frequencies (Phelps and Baker 1994).

Ecology

Research occured in seven sections of the mainstem of the Yakima River between Roza Dam and Easton Dam and in 35 study sections in 13 tributaries of the upper Yakima River (Figure 1). The seven mainstem sections were numbered from 1 to 7 with 1 the lowest and 7 the highest elevation. Elevations above sea level (referred to as “elevation”) at the midpoint of the sections ranged from 390 m to 701 m in the mainstem and 446 m to 847 m in the tributaries. Electrofishing was used to locate sexually mature rainbow trout in the upper Yakima River and it's tributaries from February through June from 1990 to 1993. Sample sizes were usually 10 to 30 adult-sized rainbow trout per section per survey for a total of 30 to 90 rainbow trout per tributary per survey. In each tributary, a sample was collected using a back-pack electrofisher in a low, middle, and high elevation stream section. In the mainstem, a driftboat electrofisher was used to collect fish at least once per month from February until June. Regardless of gear type, collected fish were anesthetized and checked for spawning condition by gently squeezing the abdomen with thumb and forefinger to see if ova, milt or resorbing fluids could be extruded. Sexually mature fish were defined to be those that exuded either milt or ova and were further classified as ripe or spent.

Using electrofishing techniques, the peak time of rainbow trout spawning was determined by calculating the time at which the greatest percentage of adult rainbow trout were sexually mature. The percentage of sexually mature rainbow trout was calculated for each electrofishing survey by first determining the minimum size of rainbow trout spawners for each tributary stream and mainstem section. The minimum adult size for each tributary stream and mainstem section was based on the fork length (mm FL) of the smallest sexually mature rainbow trout collected during the spring sampling period (Martin et al. 1994). All others that were equal to, or greater than that length were considered adult size and were defined as "potential" adults for each location. The percentage of sexually mature rainbow trout was calculated by dividing the number of mature rainbow trout in a given sample by the number of "potential" adults. The peak of spawning activity was identified using an Open Quasi Cubic Spline method (Manugistics Corporation 1992) and then locating the highest points on the curve. Peaks were not interpreted if sample size was small or if the percentage of sexually mature rainbow trout was less than 16%. Two criteria were used for assigning a peak: more than 15% of the sample had to be sexually mature and the sample size had to be at least seven adult size fish. In other words if we collected eight fish and two were ripe (25%), then the sample met the minimum requirements and the data was used. The estimated peak time of spawning for each stream section was then compared to elevation at the midpoint of the section to determine if a relationship existed between rainbow trout spawn timing and elevation.

The spawn timing of steelhead was determined using a variety of methods between 1990 and 1993 such as redd sitings, bank side observations, snorkeling, trapping, electrofishing, and radio telemetry. Panel or picket weir traps were used to trap fish in Wilson, Cherry, Umtanum, Swauk, and Taneum creeks. Spawn times and locations of radio tagged steelhead were determined from Hockersmith et al. (1995). Redd sitings with fish on or near them were conducted by helicopter on April 30, 1991. Bank side and snorkeling observations of steelhead were made throughout the study period. If an adult steelhead trout was collected by electrofishing or observed in a tributary after March 1, it was defined as a spawner and information on spawn time and location was recorded. We assumed that steelhead collected in a tributary after March 1 would subsequently spawn within that tributary.

Simple regression was used to describe the relationship between spawn timing and elevation for both rainbow and steelhead trout. Multiple regression and t-tests were used to compare the slopes and intercepts of regressions for rainbow and steelhead trout.

Hatchery steelhead were released into the North Fork Teanaway subbasin annually between the spring of 1991 and 1994 and were examined for sexual maturity and gender. Approximately 150 steelhead from an average release number of 31,155 were examined at the time of release. In addition, hatchery steelhead were captured throughout the spring and summer in the North Fork Teanaway subbasin and examined for sexual maturity and gender. These fish were collected by electrofishing and trapping.

Genetics

Three genetic analyses were performed to determine if gene flow occurred between rainbow and steelhead trout. First, naturally-produced sympatric rainbow and steelhead trout were collected from the North Fork of the Teanaway River and their allele frequencies were compared. Similarities in allele frequencies are an indication that the rainbow and steelhead trout represent an interbreeding population. Rainbow trout were collected between 1991 and 1993 by electrofishing and angling. Steelhead smolts were collected in a downstream migrant trap during

1991. Second, naturally- produced steelhead trout were collected throughout the Yakima basin and were genetically examined for past spawning with hatchery rainbow trout. Third, naturally produced rainbow trout were collected throughout the upper Yakima basin and were genetically examined for past spawning with hatchery steelhead trout.

Naturally produced steelhead smolts that were used to determine past spawning with hatchery rainbow trout were collected between 1989 and 1994. Downstream migrant traps were used to collect steelhead from the Naches basin, upper Yakima basin, and from the Yakima basin at Prosser. Steelhead smolts were collected by electrofishing in the Satus and Toppenish basins. Very few, if any, naturally produced rainbow trout have been collected in Satus or Toppenish Creeks despite extensive sampling (Hubble 1992; Joel Hubble, pers. communication). Therefore, collections were assumed to be steelhead in the Satus and Toppenish sub-basins. Naturally produced rainbow trout were collected between 1990 and 1993 in seven mainstem sections of the upper Yakima River and ten of its tributaries. Rainbow trout were collected by electrofishing, angling, and trapping. To avoid mixes of steelhead and rainbow trout in the samples, specific guidelines were used to classify naturally produced rainbow and steelhead trout at the time of collection. Only fish that were documented as rainbow or steelhead trout were used in the analyses.

Fish were classified as rainbow trout if they were less than 51 cm, had characteristic coloring and morphology, and if they were sexually mature at the time of collection. Hatchery-produced rainbow trout were excluded from the samples. Rainbow trout were classified as "hatchery" if they had eroded fins and wavy fin rays. Juvenile fish were classified as steelhead if they had typical smolt characteristics and if they were captured in a downstream migrant trap or by electrofishing in areas of known steelhead dominance, such as in the Satus or Toppenish basins. Typical smolt charactistics include: silvery coloration, lack of visible parr marks, easy loss of scales, torpedo shape (relatively thin), and dark band on the caudal fin. Hatchery produced steelhead were excluded from the samples if they had clipped adipose or ventral fins, eroded fins, or wavy fin rays.

The collected fish were either dissected in the field (most adult specimens) or frozen whole at ultra-low temperatures (-80^{o}C) and transported on dry ice to the WDFW genetic laboratory. Muscle, heart, eye, and liver tissues were dissected from each fish and placed into 12 x 75 mm plastic culture tubes. Electrophoresis followed the methods of Aebersold et al. (1987). The electrophoretic protocol, enzymes screened, and alleles observed during this study are listed in Phelps et al. (1994). Genetic nomenclature follows the conventions of Shaklee et al.(1990).

A chi-square goodness of fit test was used to compare allozyme data between rainbow trout and steelhead collected in the North Fork of the Teanaway River. Hatchery admixtures were calculated using the program ADMIX (Long 1991; Jeffrey Long, National Institute of Health, supplied us with the ADMIX program). The ADMIX programs calculates maximum likelihood estimates of the percent of the genes that are contributed from hypothetical ancestral parental sources. Stated another way, ADMIX yields admixture proportions that are the best estimates of combinations of hypothetical parental sources. Results can indicate the degree of past breeding events if the parental source information is correct. Because, wild and non-local hatchery fish in the Yakima basin are genetically so different, intermediate allele frequencies should indicate past breeding events. Three potential parental sources were used in the analysis -

wild *O. mykiss*, hatchery rainbow trout, and hatchery steelhead trout (Appendix 1). "Wild" is defined as the naturally produced population that would be present in the Yakima basin in the absence of interbreeding with non-local hatchery fish. Potential parental sources for wild *O. mykiss* were determined by examining allele frequencies of two sets of collections, those from the Satus and Teanaway basins, that were presumed to be representative of wild *O. mykiss*. Allele frequencies for steelhead produced at the Skamania hatchery were used for the potential parental source of hatchery steelhead. Allele frequencies for rainbow trout produced at the Goldendale Hatchery were used for the potential parental source of hatchery rainbow trout. Some allele frequencies of potential parental sources were adjusted up or down if the "unknown population" was outside of the ranges of potential parental sources. This "adjustment" was necessary in order for the program to run. To determine whether admixture estimates were the results of samples containing single or multiple gene pools we used gametic disequilibrium analysis.

Gametic disequilibrium analysis was used to determine whether rainbow trout collections contained a mixture of distinct gene pools or recent mixtures of gene pools (Waples and Smouse 1990; Phelps et al. 1994). In our samples, we could find significant gametic disequilibrium due to: the presence of "pure" hatchery and "pure" aboriginal rainbow trout, recent interbreeding of hatchery and aboriginal rainbow trout or mixtures of rainbow trout cohorts that were genetically dissimilar (our samples were pooled among years). Failure to detect significant disequilibrium in collections suggests that parental gene pools have introgressed and represent one gene pool. We did not examine the steelhead collections because some of these were from traps which collected multiple stocks of steelhead such as at the Chandler trap. Otherwise, gametic disequilibrium analysis was performed on the same data as was used in the ADMIX analysis.

Results

Ecology

The spatial and temporal overlap of rainbow and steelhead trout spawning was very high. Except for one steelhead that spawned in a high elevation tributary, sexually mature rainbow trout were collected in all areas that steelhead spawned. However, rainbow trout spawned over a much larger geographic area than steelhead. Rainbow trout spawned throughout all sampled reaches of the upper Yakima River basin with the possible exception of the highest elevation portions of some tributaries. Rainbow trout spawned in tributaries and mainstem areas and sexually mature individuals were collected at elevations between 375 to 1,061 m. Steelhead also spawned in tributaries and mainstem areas but spawned at elevations between 375 and 896 m. The one steelhead that spawned outside of the documented range of sexually mature rainbow trout was probably within the range of rainbow trout spawning, but we cannot confirm this because we did not sample there.

Rainbow and steelhead trout spawned at similar times during the spring. Spawning generally began in February and continued through June. The earliest date that sexually mature rainbow trout were collected was February 1 and the latest date was June 28, although some sexually mature fish were also collected during fall sampling (Pearsons et al. 1996). Steelhead spawning occurred between February 28 and July 2. Spawn timing of both rainbow and steelhead trout was positively related to elevation (Figure 2).

PSTRBT = 0.103 x (E) + 34.8 (N=30, r^2=0.37, P<0.05)

STSTH = 0.098 x (E) + 61.1 (N=28, r^2=0.30, P<0.05)

Where PSTRBT is the peak of spawn timing of rainbow trout measured in Julian days, STSTH is the time of steelhead spawning in Julian days, and E is the elevation measured in meters. There was no statistical difference in the intercepts (t=0.142, P=0.89) or slopes (t=0.978, P=0.33) of regressions between the peak of rainbow trout spawning and steelhead spawning.

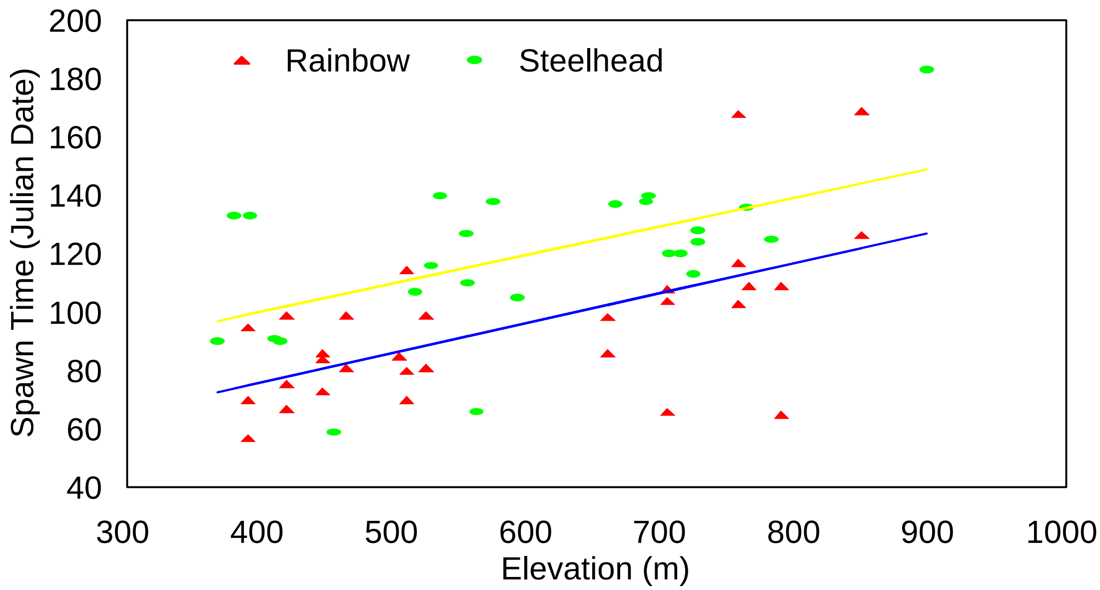

Figure 2. Peak of spawn time for rainbow trout and spawn time of steelhead relative to elevation in the upper Yakima River basin. Regression lines are also presented (top line=steelhead, bottom line=rainbow trout).

It is probable that we underestimated the duration of time that sexually mature rainbow trout were present because sexually mature fish were often collected during the first or last sampling period. However, sexually mature rainbow trout were rarely collected during other sampling activities in the summer, although some sexually mature rainbow trout have been collected in Badger, Wilson, and Cherry creeks and the Middle Fork of the Teanaway River during the fall.

We have directly documented many instances of suspected interbreeding between rainbow and steelhead trout. In 1992, a ripe female steelhead was trapped while migrating into Umtanum Creek and later was observed exiting the stream spent. This occurred near the peak of rainbow trout spawning activity in the creek. No other steelhead were observed to have entered the creek through the trap that year. In 1990, a spent female steelhead was collected adjacent to her redd in association with several ripe male rainbow trout. No other steelhead were collected in Umtanum Creek during 1990 despite intensive sampling. In 1995, one female steelhead was observed with mature rainbow trout on a redd in Buckskin Creek. Again, no male steelhead was captured in an adult migrant trap located below the redd. Lastly, many precocial hatchery steelhead have been observed in the North Fork Teanaway subbasin. Up to 4.0% of hatchery steelhead released were sexually mature males. Considerably less than 0.1% of precocial hatchery steelhead encountered during all sampling activities were females. Some precocial hatchery steelhead have been collected after they have spawned and others have been observed spawning with rainbow trout.

Genetics

Genetic data indicated that rainbow and steelhead trout were interbreeding where they were found in sympatry. Rainbow trout were genetically indistinguishable from steelhead collected in the North Fork of the Teanaway River (X^2=54.99, df=43, P=0.104). However, the test had low statistical power because of small sample sizes. The low statistical power of the test and the small P value suggest that some reproductive isolation among rainbow and steelhead trout did occur. Except for the Satus and Toppenish creek basins, steelhead trout were an admixture of hatchery rainbow trout, hatchery steelhead, and wild steelhead (Table 1). This result indicates that some steelhead have spawned with hatchery rainbow trout sometime in the past. Conversely, with the exception of Wilson and Cherry creeks and the mainstem Yakima River above the Ellensburg Town Diversion, rainbow trout were an admixture of hatchery rainbow trout, hatchery steelhead trout, and wild rainbow trout (Table 2). Again, this result further indicates that some rainbow trout and steelhead spawned together.

The percent of hatchery admixture among *O. mykiss* varied with location and life history type. The percent of hatchery rainbow trout alleles detected in rainbow trout samples was negatively related to elevation (Figure 3). However, the percent of hatchery steelhead alleles detected in rainbow trout was unrelated to elevation. The highest percent of hatchery rainbow trout admixture in steelhead samples in subbasins of the Yakima basin was found in the upper Yakima basin (16%), followed by the Naches (6%), Satus (2%), and Toppenish (0%) basins (Table 1). Rainbow trout had higher percentages of hatchery admixtures than steelhead (Tables 1 and 2).

Gametic disequilibrium analyses generally indicated that gene pools of parental sources were mixed through interbreeding. In other words, most (8 of 12 rainbow trout) of the collections were not separate populations of non-local hatchery and aboriginal fish but rather mixed populations of hatchery and aboriginal fish. Those collections that were in disequilibrium were Wilson Creek, Cherry Creek, Manastash Creek, and the Middle Fork of the Teanaway River ($P<0.05$). These collections were probably not in equilibrium because they contained samples

Table 1. Admixture analysis of parental source (Long 1991) for Yakima basin steelhead stocks. Smolts were collected by electrofishing or trapping.

Location	Years	N	Potential Parental Sources %		
			Hatchery		Wild Yakima River
			Rainbow	Steelhead	
		Satus Creek basin			
Dry Creek	1989,1991	153	2±.5	0	98±.5
Logy Creek	1990,1991	186	1±.3	0	99±.3
Satus Creek	1990,1991,1994	263	2±.7	0	98±1
		Toppenish Creek basin			
Toppenish Creek	1990,1994	172	0	0	100±.4
		Naches River basin			
Wapatox trap	1989,1990,1991	366	6±1	3±2	91±2
		Upper Yakima basin			
Roza trap	1989,1990	175	16±2	6+3	78+3
N.F. Teanaway trap	1991	25	10±4	6+6	84+7
		Yakima River (conglomerate)			
Chandler trap	1989,1994	373	11±2	2±2	87±2
Yakima Hatchery	1991	40	4±3	44±10	52±10

Table 2. Admixture analysis of parental source (Long 1991) for upper Yakima basin rainbow trout. Mature (ripe or spent) adults were collected by electrofishing.

Location	Years	N	Potential Parental Sources %		
			Hatchery		Wild Yakima River
			Rainbow	Steelhead	
		Tributaries			
Wilsonb	1990-1993	74	39+3	0	61+3
Cherryb	1990-1991	12	37+5	0	63+5
Badger	1991-1992	45	44+7	2+5	54+8
Umtanum	1990-1993	56	41+7	4+6	55+8
Taneum	1990-1992	59	11+4	2+4	87+5
Swauk	1990-1992	64	11+2	2+3	87+3
Manastash	1991-1992	69	9+3	5+4	86+5
M.F. Teanaway	1991-1993	86	5+2	10+6	85+6
W.F. Teanaway	1991-1993	72	9+4	9+9	82+9
N.F. Teanaway	1991-1993	55	6+3	6+6	88+7
		Mainstem			
Sections 1-3	1990-1992	108	38+4	10+6	52+6
Sections 4-6b	1990-1992	52	42+4	0	58+4
Section 7	1991-1992	14	15+3	0	85+3

[b]=did not use PEPD for the estimate

with recent hatchery admixtures or were from pooled samples. Wilson Creek is the last stream location in the upper Yakima basin that is still stocked with non-local rainbow trout and some of these hatchery rainbow trout probably disperse into nearby Cherry Creek.

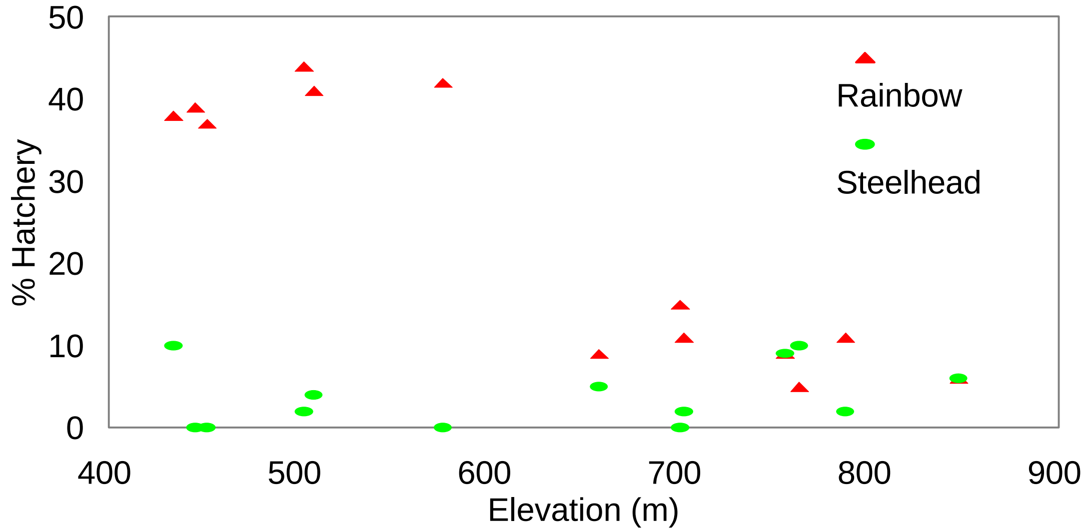

Figure 3. Percent of non-local hatchery and rainbow trout alleles found in naturally produced rainbow trout collected in the upper Yakima River basin.

Discussion

Ecological and genetic evidence indicate that rainbow and steelhead trout in the Yakima basin interbreed when in sympatry. Interbreeding between the two life history forms may occur in a variety of ways. For example, the following crosses may occur: female steelhead x male rainbow trout, male steelhead x female rainbow trout, residual female steelhead (offspring of steelhead x steelhead mating) x male rainbow trout, residual male steelhead (offspring of steelhead x steelhead mating) x female rainbow trout, female steelhead x anadromous male rainbow trout (offspring of rainbow x rainbow trout mating) and male steelhead x anadromous female rainbow trout (offspring of rainbow x rainbow trout mating). The most common modes of interbreeding between rainbow and steelhead trout appears to occur between female steelhead and male rainbow trout and between precocial male steelhead and female rainbow trout. All of the interbreedings that we observed that involved an anadromous adult were between female steelhead and male rainbow trout. In Waddell Creek, California, Shapovalov and Taft (1954) observed that female steelhead were very often accompanied by male rainbow trout during spawning. Only rarely were resident females observed with steelhead during spawning.

Furthermore, as in other salmonids, male rainbow trout may successfully spawn with female steelhead, even in the presence of male steelhead, by sneak spawning (Shapovalov and Taft 1954; Hutchings and Myers 1985; Foote and Larkin 1988; Wood and Foote 1996). The sex ratio of anadromous steelhead may also be skewed towards females when large proportions of male offspring residualize (Thorpe 1987; Peven et al. 1994). A high proportion of anadromous females would increase the potential for female steelhead to spawn with male rainbow trout particularly if anadromous male steelhead were scarce. This has been shown to occur in other salmonids (Jonsson 1985; Myers and Hutchings 1987).

Precocial male steelhead that do not migrate to the ocean may also spawn with female rainbow trout. We know of no studies that have definitively documented residualized precocial steelhead in natural populations below barriers. However, Mullan et al. (1992a) provide evidence to suggest that "steelhead" at high elevations are thermally fated to a resident life history. In addition, humans have created self-sustaining populations of residualized steelhead in locations above impassable barriers created by humans. Furthermore, in addition to our study, it is well documented that hatchery reared populations of steelhead produce numerous precocial male steelhead (Tipping 1995; Viola and Schuck 1995), some of which spawn with female steelhead (Viola and Schuck 1995). Spawning between precocious males and anadromous females has also been documented in other salmonids (Myers and Hutchings 1987; Mullan et al. 1992b).

The amount of gene flow in the upper Yakima may be artificially high due to low escapement of steelhead and a high number of rainbow trout. For example, three instances of gene flow between rainbow trout and steelhead occured when only one female steelhead had ascended a stream that contained many mature rainbow trout. If the steelhead was to spawn in that stream it had to spawn with a rainbow trout or with another species. In addition, steelhead in the upper Yakima basin collected at Roza Dam had the highest percentage of hatchery rainbow trout ancestry which further supports the contention that interbreeding may be artificially high in the upper Yakima basin. Other explanations for the relatively high percent ancestry of hatchery rainbow trout in steelhead in the upper Yakima basin are also possible such as: the number of hatchery rainbow trout stocked, or their survival and reproductive success, may have been higher in the upper Yakima basin than in other areas of the basin. Unfortunately, we do not have the data that could be used to eliminate either of the explanations. Roza Dam was a probable contributor to the reduction in steelhead abundance, and the ecological release of rainbow trout in the absence of strong anadromous fish runs (Campton and Johnston 1985). The high proportion of spawning steelhead found in Satus and Toppenish creeks might be more representative of a natural population in the Yakima basin.

Interbreeding between rainbow and steelhead trout has also been documented outside of the Yakima basin. In surveys conducted between March 15 and August 21, 1995, in the Deschutes River, Oregon; 48 (3%) steelhead redds, 1,430 (97%) rainbow trout redds, and two steelhead x rainbow trout redds (0.1%) were counted (Zimmerman and Reeves, 1996). The steelhead x rainbow trout redds were confirmed by the observation of steelhead and rainbow trout spawning together (Zimmerman and Reeves, 1996). The low percentage of rainbow and steelhead trout that spawned together could be attributed to differences in the spawn timing of these fish in the Deschutes River. Steelhead trout spawning peaked approximately ten weeks earlier than rainbow trout spawning. The two interbreeding events occurred after 85% of the steelhead already spawned and when few steelhead were available for spawning. During the time the interbreeding events occured, 2 of 9 (22%) redds that had at least one steelhead participant were also accompanied by rainbow trout (Zimmerman and Reeves, 1996). As mentioned earlier, Shapovalov and Taft (1954) frequently observed mature steelhead with mature rainbow trout on redds in Waddell Creek, California. Most of the time the rainbow trout were present with other male steelhead and probably sneak spawned. However, occassionally single male rainbow trout were found with female steelhead that were unaccompanied by male steelhead.

We speculate that the magnitude of gene flow between rainbow and steelhead trout may vary spatially and temporally. If steelhead are in low enough numbers that they have difficulty finding steelhead mates then there is a high probability that they will spawn with rainbow trout if rainbow trout are present in spawning condition. Thus, in times or locations that experience low steelhead spawning escapements and that have sympatric rainbow trout populations, per capita interbreeding could be relatively high. Whereas, in times or locations with high steelhead spawning escapements and/or low numbers of sympatric rainbow trout, per capita interbreeding would be relatively low. In addition, interbreeding may be high when conditions that promote maturation of steelhead in freshwater are good. This might occur when growing conditions in freshwater are poor (Mullan et al. 1992a) or good (Thorpe 1987).

Results from our work suggest that aboriginal rainbow should be included within a steelhead ESU because the two forms are not reproductively isolated when in sympatry. In fact, rainbow trout may be a good source of natural genes if steelhead are extirpated or if steelhead are excessively admixed with non-local steelhead. Indeed, rainbow trout located in high elevation areas of the upper Yakima basin could have a more natural complement of genes than steelhead spawning in the upper Yakima basin. However, breeding between rainbow trout and steelhead poses genetic risks to steelhead. For instance, interbreeding between hatchery admixed rainbow trout and steelhead may decrease the long term fitness of steelhead. In the upper Yakima basin, this may occur at relatively low elevations where hatchery admixtures of rainbow trout are high.

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Appendix 1. Allele frequencies of potential parental sources used in admixture analyses.

Locus and Allele	Allele Frequency of Parental Source		
	Wild *O. mykiss*	Hatchery Rainbow	Hatchery Steelhead
ADA-1 A	100	25	100
sAH A	80	100	97
ALAT A	96	100	94
CK-A1 A	100	94	100
GAPDH-3	96	100	100
bGLUA A	100	67	100
mIDHP-2 A	100	63	99
sIDHP-2 D	30	5	2
LDH-B2 A	50	100	82
LDHC A	100	90	100
sMDH-B2 D	0	55	1
MPI A	90	100	100
mSOD A	90	100	100
sSOD-1 B	5	36	24
sSOD-1 C	5	0	0
TPI-3 A	96	100	99
ADH D&E	100	100	96
GPI-A C	100	100	94
G3PDH-1 B	100	99	84
sMEP-1 D	100	100	94
PEPA C	100	100	98
PEPD C	100	100	91

Chapter 11

Maximum Size of Fall Chinook Salmon Prey Consumed by Juvenile Coho Salmon in Freshwater

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Abstract

To facilitate the assessment of hypothesized hatchery coho salmon *Oncorhynchus kisutch* stocking impacts on wild fall chinook salmon *O. tshawytscha*, we conducted predation experiments in partitioned fiberglass troughs and in outdoor enclosures where yearling coho salmon were being acclimated prior to release into the Yakima River. Our goal was to determine the maximim size of fall chinook salmon that juvenile coho salmon could, or would attempt to, consume. In one experiment, one coho salmon (135-171 mm) and six fall chinook salmon (41-69 mm) were stocked in each of eight cells (41 cm x 41 cm x 32 cm deep) with flowing water for 29 days. Two cells had six fall chinook salmon and no coho salmon to serve as controls. In a second experiment, twelve coho salmon (129-149 mm) and six fall chinook salmon (40-76 mm) were stocked into a cell 328 cm x 41 cm x 32 cm deep on two occassions. In a third experiment, imitation fall chinook salmon lures ranging in size from 35 to 115 mm were cast and retrieved through areas containing coho salmon that were being acclimated prior to release into the Yakima River. The number of strikes on each lure was recorded. In the first experiment, coho salmon consumed 37 of 49 (76%) fish presented to them during the 29 day study. Initially (3 days; the approximate time it takes for actively migrating coho salmon smolts to exit the Yakima River), coho salmon consumed the smaller chinook salmon, generally less than 40% of the coho salmon's body length. The maximum prey size consumed by coho salmon was 74 mm and the largest relative size (fall chinook salmon length/coho salmon length) consumed was 46%. In the second experiment, coho salmon consumed fall chinook salmon a maximum of between 41% and 47% of their body length. Some chinook salmon that were too large to eat were attacked and killed by coho salmon. Most of the fall chinook salmon that were killed by coho salmon were greater than about 45% of the coho salmon's body length. In the third experiment, the number of strikes per cast decreased as lure size increased. However, coho salmon struck at lures that were 115 mm long (approximately 64% of the length of the largest coho salmon we measured). Our results suggest that risks of predation on fall chinook salmon caused by stocking coho salmon can be reduced by stocking fewer, smaller fish that emigrate quickly and at times when fall chinook have a size refuge from predation.

Introduction

Efforts to restore coho salmon to mid-upper Columbia River tributaries may pose ecological risks to other species currently inhabiting those areas, particularly those that are endangered or at critically low abundances. Wild coho salmon were once widely distributed and very abundant in the Columbia River above Bonneville Dam (Fulton 1970; Chapman 1986). However, due to a variety of factors, wild coho salmon are now extinct in the mid- to upper Columbia basin (Mullan 1984; Wahle and Pearson 1984; Flagg et al. 1995). Recent efforts to restore coho salmon into mid- Columbia tributaries involve the stocking of millions of coho salmon reared in lower Columbia basin hatcheries. There are concerns about how current methods of restoration may affect other species.

The effects of predation should be considered when assessing the ecological risks associated with stocking coho salmon. Coho salmon have been, and continue to be, stocked into areas with salmonid species that are nearing extinction. For example, coho salmon have been stocked into the Snake River basin which has four races of salmon and steelhead, including fall and spring chinook salmon, that are protected under the Endangered Species Act. Fall and spring chinook salmon might be most vulnerable to predation by coho salmon smolts because they emerge slightly before coho salmon migrate to the ocean.

Coho salmon generally eat aquatic invertebrates in freshwater but salmonid fry supplement the diet considerably at times (Pritchard 1936; Sandercock 1991). For example, sockeye salmon *O. nerka* fry were the principal food item for juvenile coho salmon in Cultus Lake, British Columbia (Ricker 1941; Foerster and Ricker 1953). In the Chignik Lakes, Alaska, coho salmon were estimated to consume 24 to 78 million sockeye salmon fry annually which represented approximately 59% of the average population of sockeye salmon fry for the three years of study (Ruggerone and Rogers 1992). Hunter (1959) estimated that 23 to 85% of pink *O. gorbuscha* and chum *O. keta* salmon fry production were eaten annually by predators, most notably coho salmon and sculpins *Cottus* species. Coho salmon $\geq$ 75 mm fork length (FL) ate chum salmon fry that were between 35 to 44 mm in Big Beef Creek, Washington (Fresh and Schroder 1987). Finally, the species of concern in this study, fall chinook salmon, were found in the guts of juvenile coho salmon in Washington (Thompson 1966).

Proposals to supplement coho salmon in areas of the Columbia basin with critical or depressed fall chinook salmon populations prompted us to study the potential for ecological risks to fall chinook salmon. Our goal was to determine the maximum size of fall chinook salmon that coho salmon could, or would attempt to, consume. We postulated that if we knew the maximum size of fall chinook consumed by coho salmon and the spatial-temporal distribution of fall chinook size then we could determine maximum predation risk to a fall chinook population.

Methods

Two types of studies were conducted to evaluate the maximum fall chinook salmon prey size of coho salmon. The first type was conducted in free-flowing laboratory enclosures and the second type was conducted in a large enclosure used as a coho salmon acclimation site.

Laboratory Experiments

Experiments were conducted at the Chandler Juvenile Fish Facility (CJFF) on the Yakima River near Prosser, Washington. The main purpose of the CJFF is to capture and enumerate fish as they migrate downstream in the Yakima River. Experiments were initiated on April 22, 1996 and concluded on May 21, 1996. A rectangular fiberglass trough was partitioned into 10 experimental cells. Cells were 41 cm long, 41 cm wide, and 32 cm deep and partitioned with perforated metal screens. Well water was introduced at a rate of 54 l/min and the temperature averaged 14.5^{o}C. Fall chinook salmon from Priest Rapids Hatchery broodstock were reared at CJFF. Coho salmon were collected at the facility and were offspring of Cascade or Lewis River hatchery broodstock and had been introduced into the Yakima River as smolts approximately 60 km upstream.

To determine the potential maximum size of fall chinook that coho salmon might consume in the natural environment, various sizes of fall chinook salmon and coho salmon were placed together. The basic experimental design was to place one coho salmon with six fall chinook salmon in each of eight cells of a partitioned hatchery incubation trough. Coho salmon were not added to two cells which served as controls. Two fall chinook salmon from three size classes (41-50, 51-60, and 61-70 mm FL) were used in each cell. Prior to stocking the cells all fish were anesthetized and measured to the nearest mm FL. Fish were checked daily to determine the number of fall chinook that died from direct predation and for other reasons. Dead fish were removed and measured. Fall chinook salmon were fed satiation diets of Biodry^{R} daily. Except for a small quantity of Biodry which they may have consumed, coho salmon were only presented with fall chinook salmon for food. On days 3, 18, and 29 all fish were anesthetized, measured, and restocked. On day 18, one additional fall chinook salmon, that was quite large, was added to cell 5 because all of the smaller fish had been eaten.

An additional experiment (experiment 2) was conducted in a larger vessel where behavioral observations were made. Twelve coho salmon were placed in a trough that was equivalent in size to 8 cells (328 cm x 41 cm x 32 cm deep). Six fall chinook salmon were selected based on aforementioned criteria and stocked into the trough. Cursory behavioral observations were made on day 1 and day 29 to determine predator-prey interactions.

The size of fish consumed by coho salmon was expressed as the quotient of the prey size and coho salmon size. The size of fishes used to calculate the percent body length of prey consumed by coho salmon was the size of fish measured closest to the time of consumption. Coho salmon lengths measured at the start of the experiment were used as the denominator for day 3. Otherwise the length taken on day 29 was used. The relative size of fish consumed or killed during experiment 2 was expressed as a range to reflect size differences in coho salmon used in the experiment.

Angling

To determine the size of salmonid that coho salmon smolts in the Yakima basin might prey upon, imitation salmonid lures of nine different sizes (35, 45, 55, 65, 75, 85, 95, 105, and 115 mm) were presented to coho salmon in acclimation areas in Roza wasteway. Roza wasteway is approximately 10-15 m wide and contained approximately 250,000 hatchery coho salmon smolts. Two anglers each cast 90 times at each of three locations within the wasteway on April 23, 1996, for a total of 540 casts. At each location each angler would cast and retrieve each sized lure ten times through an area. Lure sizes were haphazardly selected to avoid size related temporal bias. Anglers tried to cast and retrieve each lure with equal effort (cast length = 8 m, cast and retrieval time = 20 seconds). One replicate of 90 casts could not be used because the recorded lure sizes on the data sheet were questionable.

Results

Experiment 1

Coho salmon consumed 37 of 49 (76%) of the fall chinook salmon presented to them during the 29 day experiment. Coho salmon ate between one and six of the fall chinook salmon in each cell. The largest fish that was eaten was 74 mm long and it was eaten by the largest coho salmon used in the experiment (Table 1). The largest relative size of fall chinook consumed by a coho salmon ranged between 40 and 46% of the coho salmon's body length (Figure 1). A significant positive correlation ($P=0.05$, $N=8$) was found between coho size and the maximum length of fall chinook consumed. However, with the exception of the smallest coho, all of the other coho salmon ate the largest fall chinook salmon that was presented. Thus, the significant relationship was partly an artifact of the size of fall chinook salmon presented to coho salmon.

Those chinook salmon consumed during the first three days were generally less than 40% of the coho salmon's body length (Figure 1). Eleven of twelve fish (92%) that were consumed before day 3 were < 40% of the coho salmon's body length. The average of the maximum relative size of fish that was consumed by coho salmon before day 3 was 35%. Later in the experiment, all coho salmon but one ate fall chinook salmon that were at least 40% of their body length (Figure 1). In many instances, larger fall chinook salmon were eaten before smaller ones which may be related to observed differences in fall chinook salmon behavior. Some of the fall chinook salmon were observed orienting themselves vertically in the water column or positioning themselves in the lower corner of the tank when coho salmon were nearby.

The proportion of fall chinook salmon that died, and were not eaten by coho salmon, was similar in controls (8%) and treatments (6%) which suggests that mortalities in experiment 1 were due to factors other than the presence of coho salmon.

Experiment 2

Except for one questionable mortality, all of the fall chinook salmon presented to coho salmon were eaten or killed. Coho salmon ate four and killed two fall chinook salmon after only 3 days during trial 1 and killed five fall chinook salmon after one day during trial 2. The maximum size of fall chinook eaten was 61 mm. The maximum size fish eaten corresponded to a relative size of 41% body length if the largest coho (149 mm) ate it, or 47% body length if the smallest coho (129 mm) ate it. The fish that were killed by the coho salmon ranged in size from 59 to 76 mm. Eight of the nine (89%) fish that were killed were the largest fish presented to the coho salmon. Most of the fish that were killed were greater than 45% of the coho salmons body length.

On the first day of trial 1, we observed coho salmon acting as a group to surround the fall chinook salmon. A few unsuccessful predation attempts were observed. On the first day of trial 2, coho salmon again worked in groups presumably to facilitate prey capture. Fifteen predatory strikes were observed during 10 min of observation. Two of these strikes resulted in capture of the fall chinook salmon. However, after a second or two of struggling, the fall chinook salmon swam free of the coho salmon's mouth. Within an hour of beginning trial 2, a 70 mm fall chinook salmon was captured but the coho salmon that captured it could not swallow it. After the coho salmon released it, many other coho salmon attempted to swallow it but they could not. Typically, the coho salmon would strike at the middle of the fall chinook's body and then manipulate the fall chinook with its mouth so that the head would go down the throat first. If the fish was too large to ingest, it would be expelled.

Experiment 3

The number of strikes per cast decreased as lure size increased (Figure 2). All lures were struck, including ones that were 115 mm long. The smallest lure that was used was struck an average of 4.3 times per cast. The largest lure was struck an average of 0.3 times per cast. Differences in the movement action of certain sized lures (55 and 85 mm) may have resulted in deviations from the general pattern (Figure 2).

Table 1. Sizes of coho and fall chinook salmon used in experiment 1 and 2.

Cell #	Coho length (FL mm)		Chinook salmon length (FL mm)						
	Day 3	Day 29							
			Experiment 1						
1	162	163	41	43	51	56	62	65	
2	150	152	43	47	51	56	63	64	
3	Control		44	47	51	54	60	66	
4	155	160	48	49	52	54	63	65	
5	171	174	43	50	53	56	63	67	74
6	143	150	46	50	52	55	65	67	
7	Control		46	50	53	59	68	66	
8	157	159	46	48	53	59	62	69	
9	140	144	47	49	54	59	62	64	
10	135	136	46	50	55	55	61	65	
Trial #			**Experiment 2**						
1	129-149		40	44	52	59	61	62	
2	131-149		67	70	71	71	72	76	

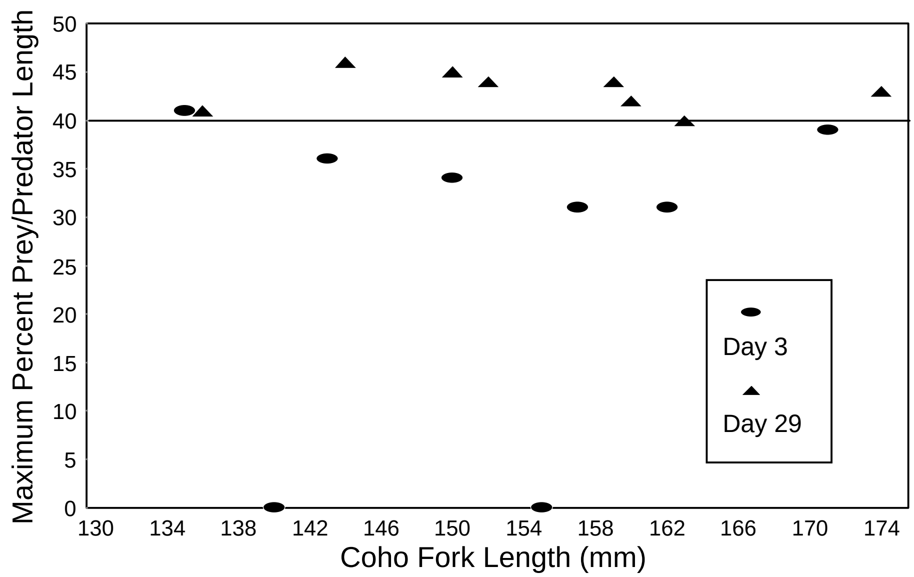

Figure 1. Maximum prey length (fall chinook salmon) relative to predator length after 3 and 29 days.

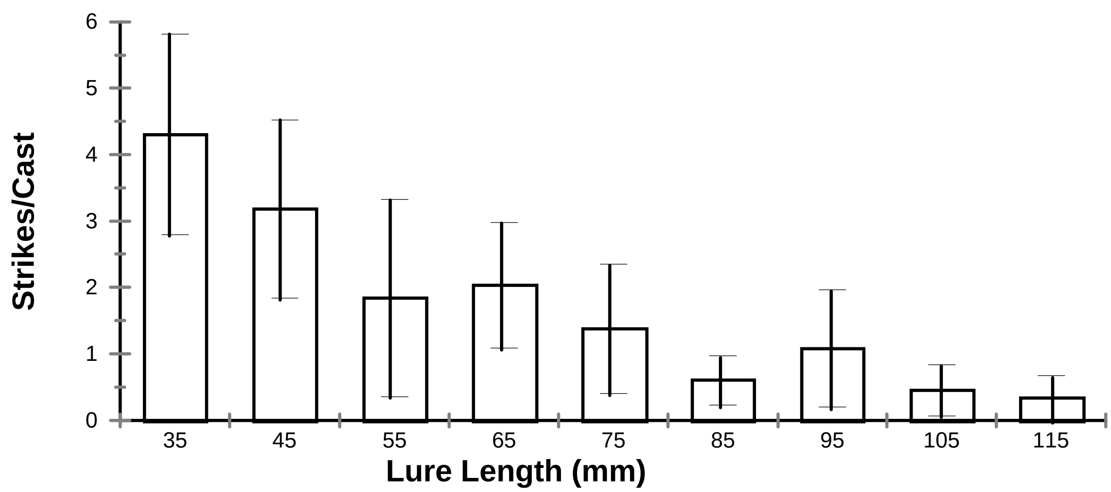

Figure 2. The average numbers ± 1 standard deviation of coho salmon strikes per cast on imitation salmon lures of different sizes.

Discussion

Although, coho salmon ate the largest fall chinook presented to them in seven of eight cells, we believe that the maximum size of fall chinook salmon that juvenile coho salmon can eat in freshwater riverine environments is close to 46% of their body length. First, coho salmon did not eat the largest individuals until relatively late in the experiment. Secondly, coho salmon in experiment 2 could not consume fish that were about 46% or greater than their body length. Preliminary examination of prey fish found in coho salmon smolts released into the Yakima basin revealed that prey fish were considerably smaller than 46% of the coho length (McConnaughey 1998). More specifically, the maximum relative size of prey fish observed in 200 coho salmon smolts containing fish was 32%. Others have also demonstrated that salmonids consume fish in freshwater environments up to 46% of their body length. For example, Jonasson et al. (1995) found that hatchery residual steelhead ate juvenile salmonids up to 44% of their body length in controlled predation trials. In addition, hatchery steelhead in the Tucannon River, Washington consumed salmonids up to 42% of their own body length (Martin et al. 1993). In contrast, yearling coho salmon in marine environments are capable of consuming relatively larger fish than those in freshwater. For example, yearling coho salmon (80-140 mm, FL) in Masset Inlet, British Columbia ate juvenile chum salmon up to 75% of their own body length, although usually they do not eat pink or chum salmon greater than 50% of the coho's length (<140 mm FL) (Hargreaves and LeBrasseur 1986).

Even though juvenile coho salmon in freshwater may not consume fish greater than about 46% of their body length, they did attempt to eat larger fish - often killing them in the process. Fall chinook salmon that were up to 51 to 58% of the coho salmon's length were killed when coho salmon unsuccessfully tried to eat them. In addition, coho salmon struck at imitation salmon lures that were 115 mm in length which corresponds to 64% of the length of the largest coho salmon we measured at CJFF. Prey fish larger than 115 mm in length may also be struck at, but with increasing size behavioral interactions are probably related more to competition than to predation. Hargreaves and LeBrasseur (1986) observed coho salmon harrassing chum salmon which may have decreased the growth of chum salmon in their experiments. Despite the possibility of coho salmon preying on, or killing prey larger than 40% of their body length, most predatory interactions (if they occur) are likely to occur to fish less than 40% of the coho salmon's body length. For instance, coho salmon primarily ate fish smaller than 40% of their body length during the first three days of the experiment. In addition, the strike rate on lures that were smaller than 75 mm was considerably higher than on lures larger than 75 mm. The mean size of fish prey found in stomachs of coho salmon collected in the Yakima basin (17%) was also small relative to the length of the coho salmon (McConnaughey 1998). Others outside of the Yakima basin have also observed prey size preferences for juvenile coho salmon (Parker 1971, Hargreaves and LeBrasseur 1986).

We acknowledge that the three experiments were conducted in arenas and conditions that would accentuate the possibility of detecting predation or predatory behavior. Indeed, that was our intent. We confined predators and prey in small areas, we provided no obvious cover for prey, and we restricted the prey type that was available to the predators (e.g., fall chinook salmon and biodry). Application to natural environments, however, should be limited to the maximum fall chinook size that coho salmon could potentially eat or try to eat. This experiment does not demonstrate that coho salmon will eat fall chinook salmon in the natural environment (e.g., Muir and Emmett 1988), although Thompson (1966) found fall chinook salmon in the guts of coho salmon in natural environments and fish were commonly found in stomachs of coho salmon in or around irrigation diversions in the Yakima basin (McConnaughey 1998). Our results simply indicate that coho salmon have the potential to eat relatively large fall chinook salmon (46% of their body length) but generally eat fish 40% of their length or smaller.

Relationship to Risk Assessment

Risks of predation caused by stocking coho salmon can be reduced by stocking fewer and smaller fish that emigrate quickly and at times when the relative size of wild salmonids provide a size refuge from predation. Fish that emigrate quickly (e.g., less than 3 days), may only consume fish that are considerably less than 40% of their body length (McConnaughey 1998). The average maximum size of fish that was consumed during the first three days of the experiment was 35% and the maximum was 41%. Fish that migrate slowly or that residualize, may eat fish up to 46% of their body length.

Knowledge of the spatial and temporal size distribution of fall chinook salmon and the size distribution of the coho salmon to be stocked can aid in the determination of release timing. For example, if one wants to be confident that 20% of the fall chinook salmon population will have a size refuge from coho salmon predators, then 20% of the population should be larger than 46% of the body length of the coho salmon to be stocked. However, the population at risk should be monitored to determine if the risk assessment hypothesis is correct. Finally, monitoring should not be limited to stomach content analysis because predatory interactions on large prey fish may occur but may not result in ingestion and therefore cannot be confirmed by inspection of the gut contents.

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Chapter 12

Lower Yakima River Predatory Fish Census: Feasibility Study 1997

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Abstract

In response to concerns that native and non-native predator fishes might limit spring chinook salmon *Oncorhynchus tshawytscha* smolt survival in the lower Yakima River, a predator study was initiated in April, 1997. Feasibility work was conducted in the lower Yakima River to attempt to conduct spring and summer population estimates of piscivorous fish known to consume juvenile salmonids. We also examined stomach contents of predatory fishes, developed and refined methods for sampling all large predatory species, and tagged a large number of fish to elucidate life history and movement information. Smallmouth bass *Micropterus dolomieui* and northern squawfish *Ptychocheilus oregonensis* were captured primarily by electrofishing. Channel catfish *Ictalurus punctatus* were collected in drifting gill nets, hoop nets, traps, and by electrofishing and angling. Stomach samples were collected during the spring when availability of salmonid smolts was estimated to be highest. We examined stomach contents in 270 fish. A total of 79 stomachs containing food items were analyzed in the lab. Northern squawfish and smallmouth bass had similar diets, with 23 to 29% of the stomachs of these species containing salmonids. Over 13% of the channel catfish stomachs analyzed contained fish, though the primary food items were insects and crustaceans. The one walleye captured in the study contained two spring chinook salmon smolts. Electrofishing was not effective for capturing channel catfish during the spring period. Drifting gill nets proved useful in capturing catfish for diet analysis during the spring period. Cheese-baited hoop nets and slat traps were very effective in capturing channel catfish after water temperatures rose above 14^{o} C. A large number of smallmouth bass (N=1234) and channel catfish (N=1126) were tagged during the study. Recaptures of tagged fish as well as seasonal changes in length distributions indicated that there is a large exchange of adult smallmouth bass between the Yakima and Columbia rivers. In general, smallmouth bass moved upstream in the spring, and downstream and out of the Yakima River in late spring and summer. Channel catfish movements were generally more limited and appeared less directed, though sample size was small. Only one of the 16 recaptured northern squawfish had moved. High spring flows and low water temperatures impacted capture rates and delayed population estimates until after the estimated peak of spring chinook salmon smolt outmigration. Population estimates were generally invalid or had very large confidence intervals. Due to fish movement and low capture efficiencies, only one of the five spring population estimates was valid. Four of the seven summer population estimates were also invalid. Confidence intervals on population estimates were generally very large due to low capture efficiency. The size structure of smallmouth bass captured in the lower Yakima River changed between spring and summer. About 64% of the smallmouth bass we sampled in April and May were 300 mm fork length (FL) or longer. In contrast, only 2% of the smallmouth bass sampled in August were 300 mm FL or longer. Northern squawfish were more abundant in the Yakima River above Prosser than below. During June, the size distribution of northern squawfish showed more large fish in the tail water area below Sunnyside Dam than in the free-flowing reaches downstream. This size disparity was reduced by August. Even though our efforts to accurately assess population sizes of predatory fish were generally unsuccessful, we successfully identified methods that may enable us to develop fish predation indices in the lower Yakima River in the future.

Introduction

The development of draft objectives for non-target taxa of concern relative to supplementation of spring chinook salmon *Oncorhynchus tshawytscha* in the Yakima basin identified predatory fish as strong interactors that could potentially limit the success of supplementation efforts (see Chapter 2, this report). Predatory fishes have been implicated as a source of smolt mortality throughout the mid- and lower Columbia and Snake rivers (Vigg et al. 1991; Tabor et al. 1993; Ward et al. 1995). To date, little predatory fish work has been conducted in the Yakima River. Low smolt survival through the Yakima River, especially between the city of Yakima and the confluence with the Columbia River, have been attributed to predation by high densities of native and non-native piscivorous fishes in the lower reaches of the river. Northern squawfish *Ptychocheilus oregonensis*, smallmouth bass *Micropterus dolomieui*, and channel catfish *Ictalurus punctatus* are the primary piscivorous fish species that are present in the lower Yakima River. High concentrations of northern squawfish have been observed below irrigation dams on the Yakima River during the spring smolt migration period (Bruce Watson, Yakama Indian Nation, personnel communication, G.A.M., personal observation). Densities of both smallmouth bass and channel catfish are high enough to support popular recreational fisheries in the lower Yakima River. The abundance of these predatory fishes combined with the low salmonid smolt survival rates within the Yakima basin prompted our study to assess and develop methods that would be capable of determining the abundance of predatory fishes that might consume migrating spring chinook salmon smolts.

Busack et al. (1997) outlined the specific need for determining the abundance of predators and their consumption rates of spring chinook salmon smolts in the spring chinook salmon monitoring plan for the Yakima Fisheries Project. The goals of our study were to: 1) determine if population estimates could be done, 2) compare spring and summer population estimates and, 3) determine if predator abundances were high enough to warrant further work. The specific objectives of this study were to: 1) attempt to conduct population estimates of smallmouth bass, channel catfish, and northern squawfish during the spring and again in the summer in index sections of the lower Yakima River, 2) compare daylight and darkness boat electrofishing catch rates, 3) examine the stomach contents from a portion of the predatory fish captured, 4) explore other methods of sampling predatory fish populations in the lower Yakima River, and 5) tag predatory fish in an attempt to elucidate movement patterns. The field work for this study was conducted by the WDFW and the Yakama Indian Nation (YIN). The work on this project is ongoing. This report briefly summarizes the data collected to date, results should be considered preliminary and subject to change with further analyses. More complete analyses and interpretation will be presented in a future report.

Methods

Study Area

The lower Yakima River flows through irrigated farm land in an otherwise arid area in central Washington State. Crops produced in the area are dominated by hops, wine grapes, hay, and fruits. Much of the water in the lower Yakima River has been utilized by irrigators and then returned to the river. Irrigation dams within the study area Include Sunnyside Dam (river kilometer (rkm) 167.4), Prosser Dam (rkm 74.9), and Horn Rapids Dam (rkm 28.1). Summer water levels can be extremely low below Prosser Dam (Figure 1). Water temperatures in the lower Yakima River often exceed the upper lethal limits for salmonids (> 25^{o} C; Bidgood and Berst 1969). Non-native warm and cool water species such as smallmouth bass, channel catfish, pumpkinseed *Lepomis gibbosus,* bluegill *L. machrochirus*, yellow perch *Perca flavescens,* walleye *Stizostedion vitreum,* largemouth bass *Micropterus salmoides*, black crappie *Pomoxis nigromaculatus*, brown bullhead *Ictalurus nebulosus,* carp *Cyprinus carpio*, and goldfish *Carassius auratus* are present in the lower Yakima River. Many of the native species previously found in this lower reach, such as sandroller *Percopsis transmontana* and Pacific lamprey *Lampetra tridentata* (Patten et al. 1970), are now either rare or extinct (see Chapter 2, this report).

The habitat in the lower Yakima River corridor has been influenced by irrigation diversions and bank stabilization. Riparian vegetation is dominated by black cottonwood, alder, and Russian olive. The gradient of the river decreases as the river nears its confluence with the Columbia River between the cities of Richland and Kennewick, Washington. The lower 6.4 km of the Yakima River are influenced by the pool elevation behind McNary Dam on the Columbia River.

Population estimates were conducted by boat electrofishing in five sections. The two sections sampled by WDFW with an electrofishing drift boat were; 1. Twin Bridges (Grosscup Road) to Van Geisen Road (Vangie), and 2. Chandler Power House to Benton City (Benton). The Vangie section was 8.0 km long, while the Benton section was 7.8 km long. The YIN used a jet boat electrofisher to sample three areas: 1. from Horn Rapids Dam upstream for 12.9 km (Horn), 2. approximately 2.1 km upstream of the Granger boat ramp to a point 2.0 km downstream of the boat ramp (Granger), 3. a small area 0.18 km long immediately below Sunnyside Dam (Sunnyside). Additional locations that were sampled by electrofishing were between Van Geisen Bridge and Duportail Road, and Duportail Road to the Highway 240 bridge. Catfish trapping and gill netting were conducted between the I-182 bridge and the mouth of the river. Most of the gill netting of catfish occurred in the lower 5 km of the river, while catfish traps were operated primarily between rkm 5 and 10, between the I-182 and highway 240 bridges. The three general categories of data collection for this work were stomach sampling, methods refinement/tagging, and population estimates/species composition.

Stomach Sampling

Stomach samples were collected from northern squawfish, smallmouth bass, and channel catfish. Stomach samples taken from Northen squawfish and smallmouth bass were collected by drift boat electrofishing. Most channel catfish collected for stomach samples were captured in drifting gill nets. Gill nets were monofilament with a 12.7 cm stretch (6.3 cm bar mesh) and were 15.2 m long and 3 m deep, with a lead line along the bottom edge and high floatation buoys along the top edge. Drifting gill nets were stretched out perpendicular to the river bank in 1.8 to 4.3 m of water and allowed to float downstream for 5 to 30 min. A net was retrieved when it appeared to have entangled a fish, became snagged in debris, or the drift exceeded 30 minutes. Two nets were fished simultaneously by one crew of three people.

Initial stomach samples were collected by excising the stomach and placing the stomach and its contents in whirl-paks with a 10% formalin solution. After using gastric lavage (garden sprayer with river water) on several catfish and bass, and then subsequently removing the stomach, we concluded that gastric lavage worked well enough (i.e., no large food items remained in the gut following lavage) to collect all subsequent samples with the gastric lavage technique. All stomach samples were bagged separately and tagged with date, stomach number, species, length, weight, and the section where the fish was captured.

The stomach samples were brought back to the lab where the stomach content samples were transferred from formalin to alcohol, and placed in various size glass and nalgene containers. During the transfer process, the whole stomachs were dissected and the contents removed. The analysis of the contents consisted of placing the contents of a single sample into a petri dish and first separating the vegetation from the animal material. With fish, insects, Crustacea, and a few other miscellaneous animal types (Mollusca, Amphipoda, Diplopoda, Annelida, Gastrapoda, and Arachnida), more detailed information was taken. The miscellaneous and vegetation categories had the sub-category "other" for materials which only showed up once or twice over the course of all of the samples. The specifics on these materials, if known, were noted in the comments. For identification purposes, a series of keys were used. For insects, *Aquatic Insects of Montana* and *Peterson Field Guide - Insects* were used. For fish, identification method depended on the state of decomposition of the consumed fish. A recently consumed fish was often identified by looking at the skin or fins. More completely decomposed fish required using certain bones for identification. The cleithrum proved to be the most helpful identification bone when present and in good condition. The dentary and pharyngeal arch also proved to be of some assistance, although they were best used for confirmation. In several cases, only the vertebral column of the consumed fish remained. Although the shape of the individual vertebrae was noted to be an indicator of salmonid vs. non-salmonid, it proved difficult to get a look at a clean, undamaged vertebrae. Acid removal of the material surrounding the vertebra ended up damaging the vertebra, as did physical removal of this material (scraping with a scalpel). Except in a few cases where a clear examination of a solid vertebrae was possible, identifications made using this method were regarded as extremely tentative and the fish was entered into the database as unidentified with a comment made regarding the tentative identification. For bone identification, a series of keys and sketches produced and provided by the National Biological Survey station (formerly U.S. Fish and Wildlife Service) located in Cook, Washington, were used.

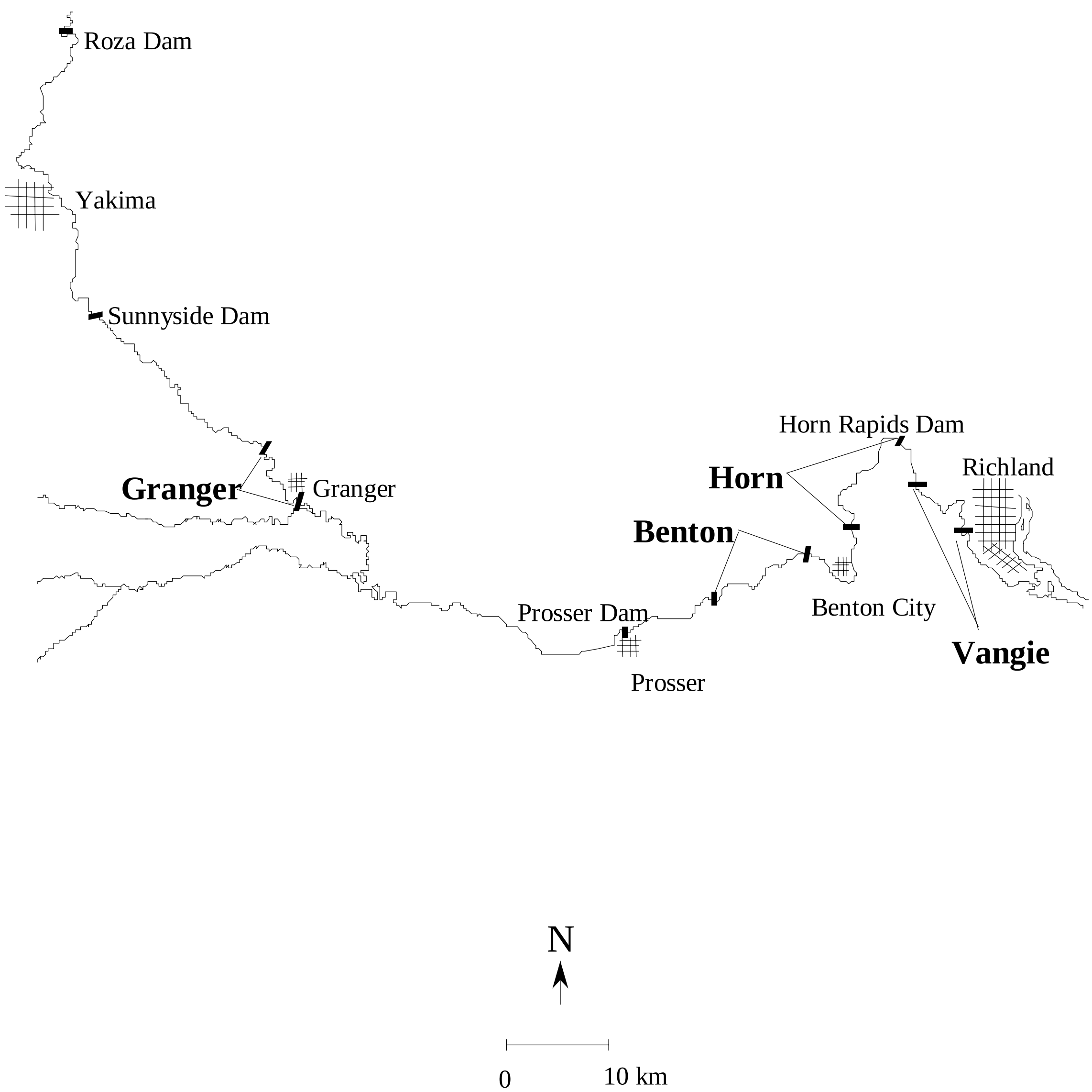

Figure 1. Map of the study area in the lower Yakima River showing index sections in bold type.

Methods Refinement/Tagging

Electrofishing was selected as the primary method for collecting smallmouth bass and northern squawfish, while gill netting and trapping were necessary for capturing sufficient numbers of channel catfish. To determine whether we could capture enough smallmouth bass and northern squawfish to conduct population estimates during daylight electrofishing, we conducted two tests wherein we compared catch per unit effort (CPUE) of electrofishing during pre-dawn hours of darkness to CPUE during daytime electrofishing.

Channel catfish are not very susceptible to capture by electrofishing (cites), therefore we experimented with different trapping devices (and baits), drifting gill nets, angling, limb-lines, jug-lines, and trot-lines to determine the best method for capturing channel catfish. We used two sizes of hoop nets with 2.5 cm mesh; small = 60 cm downstream hoop, big = 88 cm downstream hoop. We also used round wooden 'slat-box' traps with a 38 cm outside diameter. All traps had two throats. We used several different substances and mixtures as bait in catfish traps. Trout pellets, commercial fish attractant, herring, chicken liver, soy beans, and cheese trimmings were all used alone or in combination. Bait was placed in mesh bags that were tied into the upstream end of each trap. Traps were tied to trees and shrubs on the bank and were placed 1 to 10 m out from the bank in depths ranging from 2 to 4 m. Traps were generally checked every 48 to 72 hours. All catfish were measured to the nearest mm FL and tagged with an anchor tag and released. Weights (g) were taken on some catfish. In addition, some catfish and smallmouth bass were captured by angling.

Population Estimates/Species Composition

Mark-recapture population estimates were conducted on smallmouth bass and northern squawfish captured by boat electrofishing (Vincent 1987). All captured predatory fish over 100 mm FL were marked with a partial fin clip and/or tag on successive runs down each bank. Recapture runs followed the same sequence 7 days after the marking runs. Fish were processed every 1 km during both marking and recapture runs. The species composition was visually assessed and recorded by the netter. Due to high water and turbidity during the spring smolt outmigration period, the beginning of the first set of population estimates was delayed until June 16. Spring population estimates were completed in the Vangie, Benton, and Horn sections by June 24. The Sunnyside section was sampled between June 18 and June 25. The Granger section was sampled between June 26 and July 2. Population estimates were conducted in all sites again in August to compare spring and summer abundance and size structure data. The Vangie and Benton sections were sampled between August 12 and 20. Horn was sampled between August 11 and 19. Sunnyside was sampled between August 13 and 20 and again on September 5 when a multiple removal population estimate was attempted for comparative purposes. Granger was sampled August 13 and 20. An additional section in the upper Yakima River was sampled for northern squawfish on September 10 and 11. Sampling occurred on other dates in the Horn, Sunnyside, and Granger sections, but it was outside of the established protocol. Data from these additional runs was used for size structure, species composition, CPUE, and movement information.

Results

Stomach Sampling

We examined stomach contents collected from 270 fish. Stomach contents from 98 fish were analyzed in the laboratory, while stomach contents from 172 fish were examined in the field. Of the 98 stomach samples examined in the lab, 19 were empty (16 smallmouth bass and 3 northern squawfish). A summary of the contents from the remaining 79 stomachs are presented in Table 1. Data from stomach contents examined in the field during the spring are shown in Table 2. Stomach contents examined in the field were not analyzed as completely as those examined in the lab.

Table 1. Stomach content data from fishes collected in the lower Yakima River between April 10 and June 11, 1997, that were analyzed in the lab. Species of fish, sample size (N), and percent of the stomachs containing each type of general item are shown.

		Percent of Stomachs Containing Item					
Species	N	Fish	Salmonid	Chinook	Insects	Crustacea	Other[a]
Channel Catfish	15	13.3	0.0	0.0	100	73.3	40.0
Northern Squawfish	7	57.1	28.6	14.3	28.6	14.3	14.3
Smallmouth Bass	56	58.9	23.2	7.1	46.4	16.1	12.5
Walleye	1	100	100	100	0.0	0.0	0.0

[a] Includes molluscs, amphipods, diplopods, annelids, gastropods, and arachnids.

Table 2. Stomach content data from fishes examined in the field between April 10 and June 11, 1997. Species of fish, sample size (N), and percent of the stomachs containing each type of general item are shown.

		Percent of Stomachs Containing Item						
Species	N	Fish	Salmonid	Chinook	Insects	Crustacea	Other[a]	Empty
Channel Catfish	19	0.0	0.0	0.0	84.2	0.0	26.3	15.8
Northern Squawfish	2	0.0	0.0	0.0	0.0	0.0	50.0	50.0
Smallmouth Bass	150	20.0	0.0	0.0	22.6	1.3	3.3	71.3
Largemouth Bass	1	0.0	0.0	0.0	0.0	0.0	0.0	100

[a]Includes molluscs, amphipods, diplopods, annelids, gastropods, and arachnids.

With the exception of two spring chinook salmon smolts in the walleye (593 mm FL), all chinook salmon identified in stomach samples were presumed to be fall chinook salmon due to their small size. Spring chinook salmon smolts measured in our electrofishing in the lower Yakima River were between 116 and 147 mm FL while fall chinook salmon were between 38 and 80 mm FL. Capture method appeared to influence stomach content data. Channel catfish captured in gill nets and by hook and line tended to have non-empty stomachs, while those captured by electrofishing had empty guts. In addition, a 420 mm channel catfish that was captured by angling on July 2, 1997, regurgitated 14 age-0 carp *Cyprinus carpio* between 40 and 60 mm long.

Methods Refinement/Tagging

Electrofishing for predatory fish during pre-dawn hours of darkness did not increase catch rate. Tests conducted on April 11 and 18 in the Horn and Vangie sections, respectively, showed no increase in CPUE during pre-dawn electrofishing. The average CPUE for smallmouth bass during hours of darkness was 0.22 bass/minute, while CPUE after first light was 0.24 bass/minute. All

electrofishing after the April 18 test was conducted during daylight hours.
Few channel catfish were captured by electrofishing during the spring sample period. Gill nets were effective during the early period when water temperatures were cool and lost effectiveness as time passed and water temperature increased (Table 3, Figure 2).

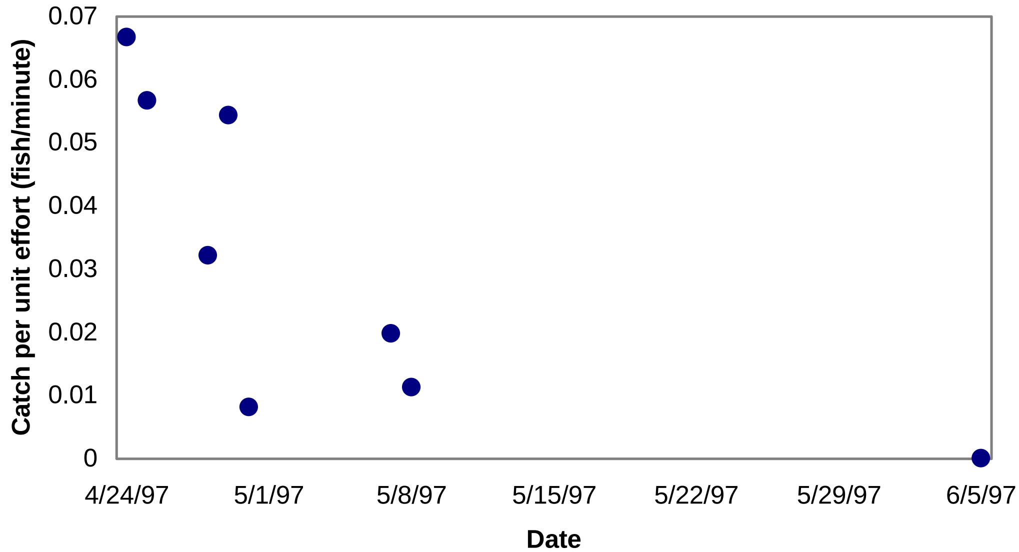

Figure 2. Catch per unit effort for channel catfish captured with drifting gill nets in the lower Yakima River versus date.

Table 3. Catch per unit effort (CPUE) for channel catfish in the lower Yakima River using different gear types during spring and summer, 1997. Note: CPUE for hoop nets and slat traps is the number of fish/trap/night.

Gear Type	Total Time	Catch	CPUE	Dates	Water Temp.
Gill net	33.9 h	43	1.27/h	4/24 - 6/5	10.5 - 14.5
Angling	32.5 h	41	1.26/h	4/29 - 7/2	12.0 - 20.0
Hoop nets	16 nights	1	0.06/night	4/24 - 5/23	10.5 - 12.0
Hoop nets	60 nights	186	3.10/night	6/5 - 7/2	17.0 - 20.0
Hoop nets	83 nights	315	3.80/night	7/2 - 8/9	19.0 - 27.0
Slat traps	174 nights	5	0.03/night	5/7 - 6/5	12.0 - 14.0
Slat traps	156 nights	303	1.94/night	6/5 - 7/2	17.0 - 20.0
Slat traps	210 nights	171	0.81/night	7/2 - 8/9	19.0 - 27.0
Electrofishing	17.1 h	3	0.18/h	4/10 - 5/29	7.0 - 14.0
Electrofishing	52.6 h	33	0.63/h	6/10 - 6/30	15.0- 20.0
Electrofishing	25.9 h	72	2.78/h	8/12 - 8/20	20.5 - 27.0

The channel catfish captured in gill nets were restricted to a narrow size range; most fish captured were between 450 and 600 mm FL (Figure 3A). Angling for catfish produced a mean CPUE that was very similar to the gill net CPUE (Table 3) and sampled fish from a wider size range than the gill nets (Figure 3B). Gill nets seemed to provide the best stomach samples, but the three catfish captured by angling that were sampled for stomach contents also contained food items. Early attempts (April 24 to June 5) at trapping were unsuccessful (Table 3). Water temperature during this early period was generally below 12^{o} C which may have reduced catfish movement. Also, during this early period we used various baits in hoop nets, with the exception of cheese. Subsequent comparisons found the cheese to be far more effective in attracting catfish than the other bait substances we tried. Further, the early trapping attempts were all one or two nights long and then the traps were removed. Leaving the traps in place longer may increase catch by allowing the bait scent to attract fish from a greater distance downstream as well as the additional attraction of large female catfish that had already entered the traps. Beginning on June 5, we baited all traps with spoiled cheese trimmings and left the traps out continuously. Water temperature in early June increased above 14^{o} C and our hoop net and slat trap CPUEs increased dramatically. Hoop nets captured 186 channel catfish between June 5 and July 2, 1997 (Table 3). The length distribution of catfish captured in hoop nets was skewed toward smaller individuals, but several larger fish (600 - 750 mm FL) were also captured (Figure 3C). Slat traps captured 303 channel catfish during the period, and captured more medium and large fish than the hoop nets (Table 3, Figure 3D). Between July 2 and when traps were removed on August 9, hoop nets had nearly five times higher catch rates than the slat traps due to the presence of larger numbers of small catfish present (Table 3). More channel catfish were captured by electrofishing as water temperature increased (Table 3). The sizes of channel catfish captured by electrofishing was similar to the sizes collected in slat traps, with large portions in the 275 and 475 mm size groups (Figure 3E).

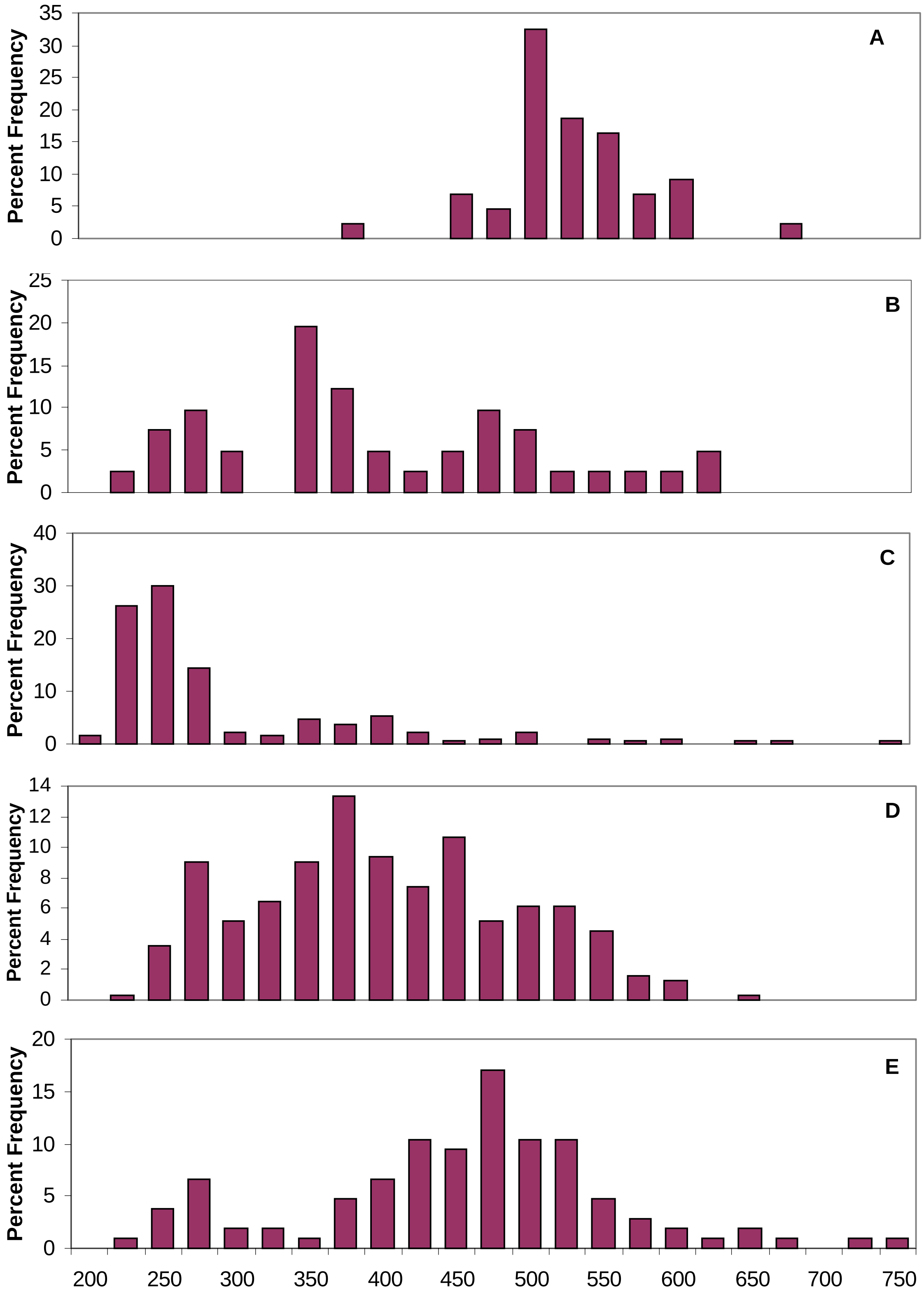

Figure 3. Channel catfish length distributions for fish captured via gill nets (A, N=43), angling (B, N=41), hoop nets (C, N=187), slat traps (D, N=308), and electrofishing (E, N=106) in the lower Yakima River during spring and summer, 1997.

A large number of smallmouth bass and channel catfish were tagged during the spring and summer of 1997. A total of 1234 smallmouth bass, 1126 channel catfish, and 471 northern squawfish were tagged during the study period. Most smallmouth bass and some of the channel catfish appeared to move downstream and/or out of the Yakima River between spring and summer. Our methods were biased towards capturing tagged fish that had not moved, because we sampled the same areas repeatedly. Smallmouth bass tagged in the Yakima River in the spring were predominantly recaptured in the Columbia River. Recaptures reported by anglers, as well as fish we recaptured through our work, showed a general upstream movement of smallmouth bass in the early spring, followed by a downstream movement, often down into the Columbia River, beginning in late June (Table 4). Seven of the nine (78%) smallmouth bass that moved upstream were recaptured before July 15, while all 15 of the bass that moved downstream were recaptured after June 20. All 15 of the bass that moved downstream left the Yakima River and entered the Columbia River. Ten of the 15 bass that migrated down out of the Yakima River moved upstream in the Columbia River. Of the 93 tagged smallmouth bass that were reported, 39 (42%) were caught by anglers and 54 (58%) were captured by electrofishing. Twenty-three tagged channel catfish were reported. Of those, more than half of them had moved over 1 km (Table 4). Of the 12 fish that moved, seven (58%) moved downstream and five (42%) moved upstream. All but one of the catfish that moved upstream did so before the end of July. Eight (35%) of the tagged channel catfish were recaptured by anglers, while the remaining 15 (65%) were recaptured in traps. Only one (6%) of the 16 tagged northern squawfish that were recaptured had moved (Table 4). This individual moved 33 km downstream in a period of nine days. All tagged northern squawfish were recaptured by electrofishing.

Table 4. Movement summary data for predatory fish tagged in the lower Yakima River between April and September, 1997. Number of recptured fish, mean distance moved, the percentage of fish that did not move (recaptured within the same sample reach; %None), moved upstream (%Up), and moved downstream (%Down) are shown for each species. Negative values indicates downstream movement while positive values correspond to upstream movement.

		Mean Distance Moved (km)					
Species	Recaps	Overall	Up	Down	%None	%Up	%Down
Smallmouth bass	93	-2.79	14.7	-26.1	74.2	9.7	16.1
Channel catfish	23	-0.96	3.66	-4.26	47.8	21.8	30.3
Northern squawfish	16	-2.06[a]	0.00	-33.0	93.8	0.0	6.2

[a]Only one fish moved, mean based on total movement divided by number of recaptured fish.

Population Estimates/Species Composition

Population estimates were attempted in five river sections in mid-June to early July and again in mid- to late August. Spring population estimates were generally invalid with the exception of the smallmouth bass estimate in the Vangie section (Table 5). Tag recapture information during the spring population estimate period showed that marked fish were moving out of the study reaches, thereby violating an assumption of the mark-recapture methodology. Smallmouth bass were the only species which were present in high enough numbers and were susceptible enough to our collection methods to conduct population estimates using boat electrofishing in the Vangie, Horn, and Benton sections. The smallmouth bass estimates during the summer period were mathematically valid in all three sections, even though capture efficiencies were very low (Table 5). The Granger and Sunnyside reaches were sampled specifically for northern squawfish. Spring and summer mark-recapture population estimates for northern squawfish were not valid. Table 5 presents all relevant population estimate data, however caution should be used regarding the accuracy of the population estimates due to the low number of marked fish recovered in the recapture samples, and the subsequent effects of that on the confidence intervals and the validity of the estimates.

Table 5. Population estimate data for smallmouth bass (SMB) and northern squawfish (NSF) 100 mm FL or larger in sections of the Yakima River. Dates, species, estimate, 95% confidence intervals, capture efficiency (Effic.), and validity of the estimate are shown for each river section.

Dates	Species	Section	Estimate	CI	Effic.	Valid
6/18&25	NSF	Sunnyside	338	37-1415	8.3%	no
6/26&7/1,2	NSF	Granger	no est.[a]	-	0.0	no
6/16,17&23,24	SMB	Horn	6716	163-29727	1.1%	no
6/18,19&25,26	SMB	Benton	4522	267-26264	3.0%	no
6/16,17&23,24	SMB	Vangie	6954	374-43224	2.4%	yes
8/13&20	NSF	Sunnyside	1260	69-5502	2.9%	no
9/5	NSF	Sunnyside[b]	130	88-181	24.0%	no
9/10&11	NSF	Upper Yak.	1175	95-5731	4.1%	no
8/13&20	NSF	Granger	no est.[a]	-	0.0%	no
8/11,12&18,19	SMB	Horn	26033	1546-282194	3.0%	yes
8/13&20	SMB	Benton	9502	1275-130690	5.3%	yes
8/12&19	SMB	Vangie	11902	930-115875	4.4%	yes

[a] No marked fish were observed in the recapture sample; 58 were marked on 6/26 and 59 were marked on 8/13.
[b] Data from a four pass multiple-removal methodology presented for comparison.

The size structure of the smallmouth bass present in the lower Yakima River during the spring period differed markedly from the size structure observed there during the summer. Specifically, the larger bass (300 mm FL and larger) were nearly absent during the summer surveys. In April and May, 1997, 63.7% of the bass 100 mm FL and larger sampled by electrofishing were 300 mm FL or larger. This figure dropped to 33.4% during June, and to 2.1% during August (Figure 4). This reduction in the number of larger smallmouth bass present in the lower Yakima River following the spring period is well supported by our movement data presented previously. The movement data illustrated a movement of larger smallmouth bass out of the Yakima River and into the Columbia River during late June and July.

Catch per unit effort for smallmouth bass $\geq$ 100 mm FL increased between spring and summer. High and turbid water during spring, coupled with the fact that fewer of the age-1 bass were larger than 100 mm FL at that time may have reduced CPUE in the spring. In April and May, a total of 342 smallmouth bass $\geq$ 100 mm FL were captured in 1028 minutes of electrofishing (0.33/min). The CPUE for smallmouth bass during June was the same (0.33/min: 1038 bass in 3153 minutes). During the summer surveys, the increased water temperature and visibility, and perhaps the growth of age-1 bass into the size range over 100 mm FL increased our CPUE to 1.19 bass/min (1848 bass in 1556 minutes).

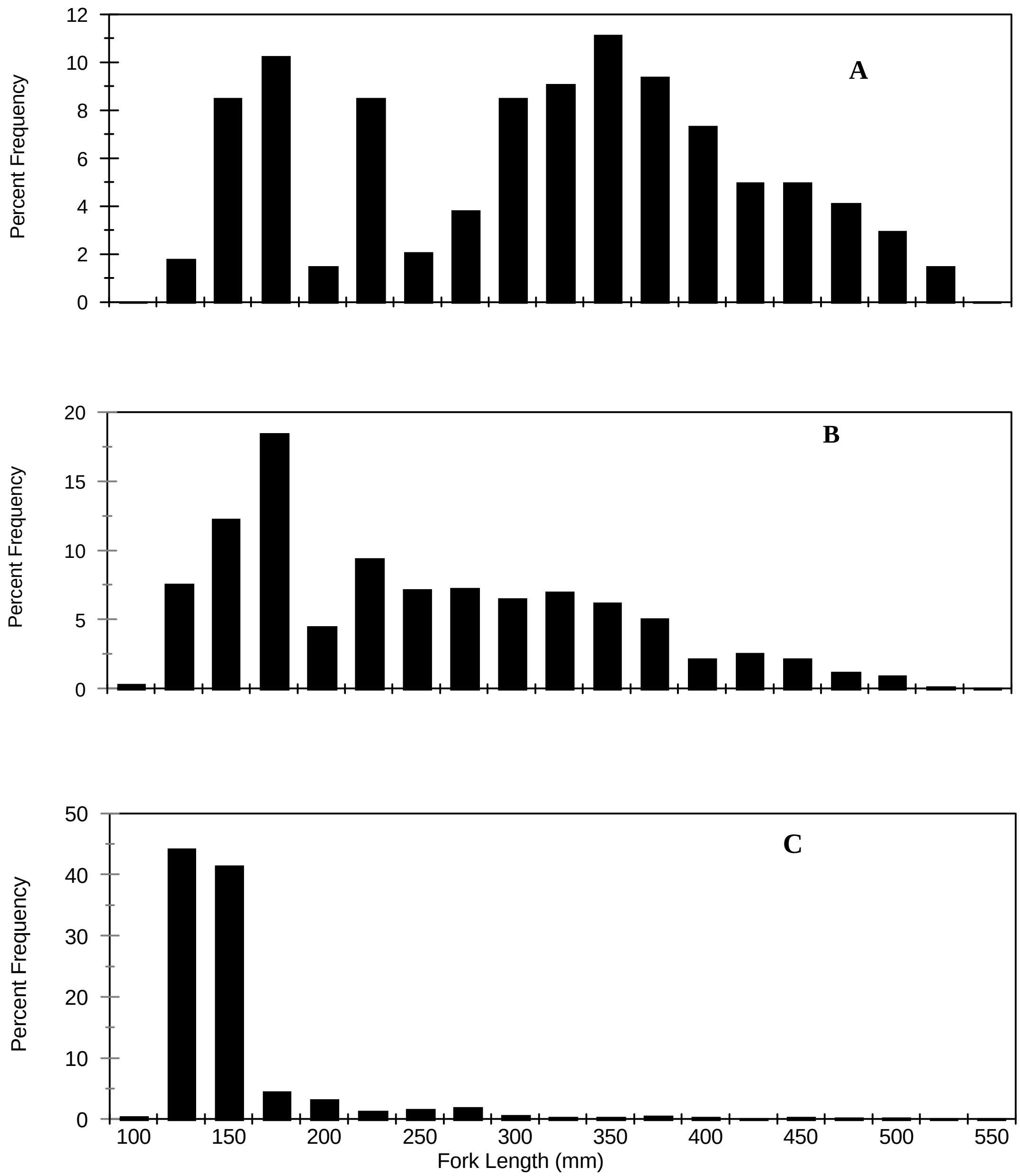

Figure 4. Length distributions of smallmouth bass greater than or equal to 100 mm FL captured by electrofishing in the lower Yakima River during April and May (A, N=342), June (B, N=1038), and August (C, N=1848), 1997.

Northern squawfish were relatively rare in free-flowing portions of the lower Yakima River below the town of Prosser. None of the mark-recapture population estimates on northern squawfish were valid due to low numbers of marked fish in the recapture samples. Catch per unit effort of northern squawfish ≥ 100 mm FL was very low in comparison to smallmouth bass in the free-flowing reaches of the lower Yakima River. During April and May we captured only 22 northern squawfish in 1028 minutes of electrofishing (0.021/min); this was nearly 16 times lower than the CPUE for smallmouth bass. During June, 144 northern squawfish were sampled in 3153 minutes (0.046/min), this was about seven times lower than the CPUE for bass during that period. The CPUE for northern squawfish in August was very similar to the CPUE in June (0.049/min, 76 northern squawfish in 1556 minutes), which was 24 times lower than the CPUE for bass during that time period. Northern squawfish catch per unit effort values were higher in the Sunnyside, Granger, and upper Yakima River reaches.

Northern squawfish captured in free-flowing river reaches were generally smaller than those captured in the tail water below Sunnyside Dam during June (Figure 5). This may have

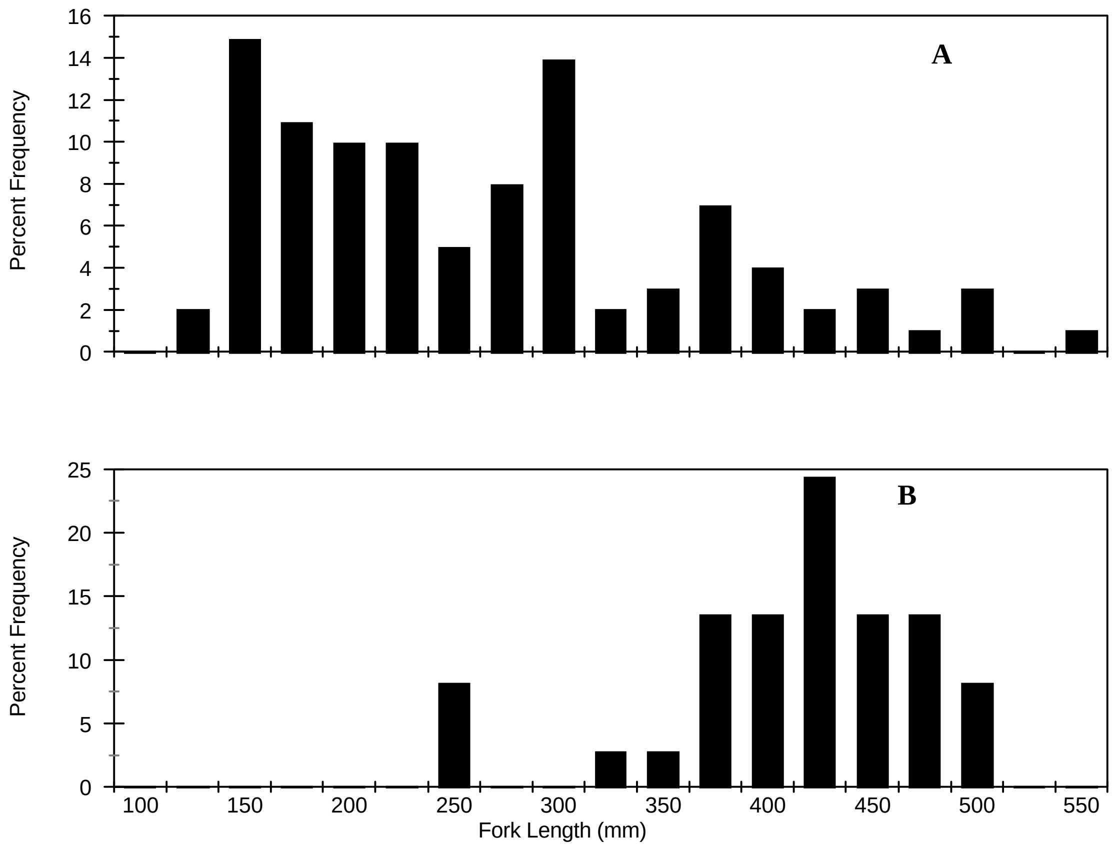

Figure 5. Length distribution of northern squawfish ≥ 100 mm FL in free-flowing river reaches (A, N=101) and in the tailwater below Sunnyside Dam (B, N=37) during June, 1997.

been due to the spawning movement of adult northern squawfish upstream to the area where they encountered a barrier to free upstream movement (even though the dam has fish ladders). It is also possible that these larger fish were in this area to prey upon juvenile fish returned to the river through the juvenile bypass system. In August, the length distribution of northern squawfish was less disparate between the free-flowing reaches and the tailwater area below Sunnyside Dam (Figure 6), further supporting the seasonal movement into the tailwater area hypothesis. Furthermore, CPUE for northern squawfish in the free-flowing Granger reach was about half what it was in the tailwater below Sunnyside Dam (Granger, 0.41/min; Sunnyside, 0.79/min).

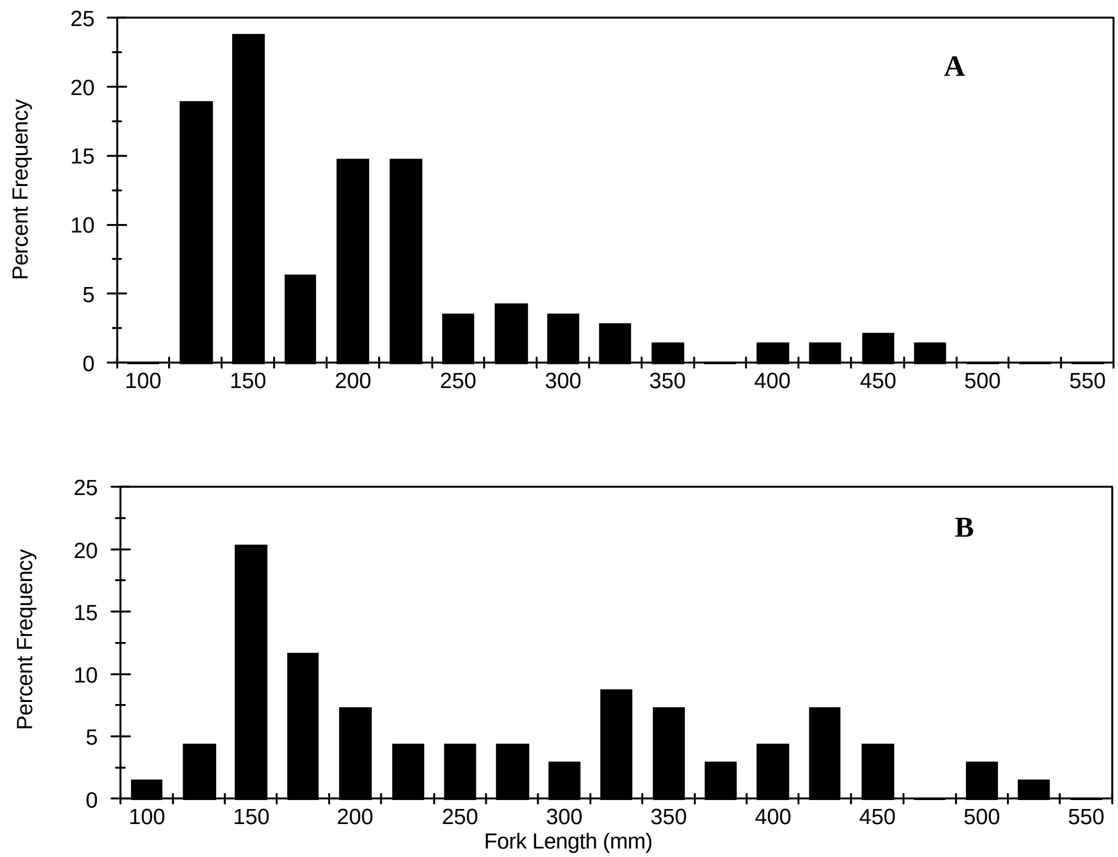

Figure 6. Length distribution of northern squawfish ≥ 100 mm FL in free-flowing river reaches (A, N=143) and in the tailwater below Sunnyside Dam (B, N=69) during August, 1997.

A total of 28 species of fish were observed in the lower Yakima River during sampling for predacious fishes. Suckers (largescale *Catostomus macrocheilus* and bridgelip *C. columbianus* were common, mountain *C. platyrhynchus* were rare) and chiselmouth *Acrocheilus alutaceus* were present and relatively abundant in all sample sections during all sample periods (Tables 6, 7, and 8). Smallmouth bass were abundant in the lower four sections and absent or nearly absent in the Granger and Sunnyside sections. Conversely, northern squawfish were relatively rare in the lower four sections and much more abundant in the Granger and Sunnyside sections. It should be reiterated that channel catfish are not as susceptible to capture by electrofishing as many other species, therefor the percent composition of channel catfish is probably underestimated.

Table 6. Visually estimated fish species composition in total number and percent composition (in parentheses) for April - May, 1997, in lower Yakima River study sections (river km from mouth are shown in parentheses below section names). Data were collected by boat electrofishing.

Species.[a]	Duportail (7.6-12.0)	Vangie (12.2-20.2)	Horn (28.1-41.0)	Benton (49.3-57.1)	Granger (130.0-134.4)	Sunnyside (167.4)
BBH	1(0.03)		0(0)			
BLG	0(0)		0(0)			
BUL	0(0)		0(0)			
CCF	1(0.03)		0(0)			
CCP	586(17.41)		545(24.61)			
CHM	568(16.88)		132(5.96)			
COH	0(0)		0(0)			
DAC	0(0)		9(0.41)			
FCH	26(0.77)		432(19.5)			
LMB	0(0)		0(0)			
LMP	0(0)		0(0)			
LND	10(0.3)		15(0.68)			
MWF	18(0.54)		31(1.4)			
NSF	16(0.48)		22(0.99)			
PMK	0(0)		0(0)			
PMO	19(0.56)		2(0.09)			
RBT	0(0)		1(0.45)			
RSS	11(0.33)		2(0.09)			
SCU	0(0)		0(0)			
SMB	605(17.97)		167(7.54)			
SPC	25(0.74)		18(0.81)			
SPD	0(0)		0(0)			
STB	0(0)		0(0)			
SUK	1467(43.58)		832(37.56)			
WCR	1(0.03)		0(0)			
WSH	10(0.3)		7(0.32)			
WST	1(0.03)		0(0)			
YLP	1(0.03)		0(0)			

[a] BBH(brown bullhead), BLG(bluegill), BUL(bull trout), CCF(channel catfish), CCP(common carp), CHM(chiselmouth), COH(coho salmon), DAC(dace spp.), FCH(fall chinook), LMB(largemouth bass), LMP(lamprey spp.), LND(longnose dace), MWF(mountain whitefish), NSF(northern squawfish), PMK(pumkinseed), PMO(peamouth), RBT(rainbow trout), RSS(redside shiner), SCU(sculpin spp.), SMB(smallmouth bass), SPC(spring chinook), SPD(speckled dace), STB(stickleback), SUK(sucker spp.), WCR(white crappie), WSH(wild steelhead), WST(white sturgeon), YLP(yellow perch).

Table 7. Visually estimated fish species composition in total number and percent composition (in parentheses) for June, 1997, in lower Yakima River study sections. Data were collected by boat electrofishing.

Species.[a]	Duportail	Vangie	Horn	Benton	Granger	Sunnyside
BBH	0(0)	26(0.74)	0(0)	2(0.05)	0(0)	0(0)
BLG	2(0.1)	9(0.26)	0(0)	0(0)	0(0)	0(0)
BUL	0(0)	0(0)	0(0)	1(0.02)	0(0)	0(0)
CCF	4(0.2)	43(1.23)	2(0.08)	4(0.09)	0(0)	0(0)
CCP	335(16.95)	1038(29.68)	746(30.85)	828(19.31)	35(3.44)	2(0.76)
CHM	392(19.84)	363(10.38)	175(7.24)	605(14.11)	90(8.83)	40(15.27)
COH	0(0)	0(0)	0(0)	1(0.02)	0(0)	0(0)
DAC	0(0)	0(0)	0(0)	1(0.02)	0(0)	0(0)
FCH	129(6.53)	100(2.86)	17(0.7)	74(1.73)	0(0)	0(0)
LMB	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)
LMP	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)
LND	3(0.15)	123(3.52)	0(0)	1212(28.27)	40(3.93)	0(0)
MWF	32(1.62)	22(0.63)	34(1.41)	6(0.14)	130(12.76)	70(26.72)
NSF	9(0.46)	12(0.34)	10(0.41)	25(0.58)	60(5.89)	38(14.5)
PMK	2(0.1)	26(0.74)	0(0)	4(0.09)	1(0.1)	0(0)
PMO	13(0.66)	0(0)	10(0.41)	0(0)	10(0.98)	0(0)
RBT	0(0)	1(0.03)	0(0)	0(0)	0(0)	0(0)
RSS	0(0)	1(0.03)	27(1.12)	0(0)	160(15.7)	0(0)
SCU	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)
SMB	386(19.53)	1114(31.86)	242(10.01)	677(15.79)	2(0.2)	0(0)
SPC	0(0)	0(0)	5(0.21)	2(0.05)	1(0.1)	2(0.76)
SPD	0(0)	2(0.06)	0(0)	71(1.66)	40(3.93)	0(0)
STB	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)
SUK	668(33.81)	617(17.64)	1150(47.56)	773(18.03)	450(44.16)	110(41.99)
WCR	0(0)	0(0)	0(0)	1(0.02)	0(0)	0(0)
WSH	1(0.05)	0(0)	0(0)	0(0)	0(0)	0(0)
WST	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)
YLP	0(0)	0(0)	0(0)	0(0)	0(0)	0(0)

[a] BBH(brown bullhead), BLG(bluegill), BUL(bull trout), CCF(channel catfish), CCP(common carp), CHM(chiselmouth), COH(coho salmon), DAC(dace spp.), FCH(fall chinook), LMB(largemouth bass), LMP(lamprey spp.), LND(longnose dace), MWF(mountain whitefish), NSF(northern squawfish), PMK(pumkinseed), PMO(peamouth), RBT(rainbow trout), RSS(redside shiner), SCU(sculpin spp.), SMB(smallmouth bass), SPC(spring chinook), SPD(speckled dace), STB(stickleback), SUK(sucker spp.), WCR(white crappie), WSH(wild steelhead), WST(white sturgeon), YLP(yellow perch).

Table 8. Visually estimated fish species composition in total number and percent composition (in parentheses) for August, 1997, in lower Yakima River study sections. Data were collected by boat electrofishing.

Species.[a]	Duportail	Vangie	Horn	Benton	Granger	Sunnyside
BBH		82(2.6)	0(0)	0(0)	0(0)	0(0)
BLG		4(0.13)	0(0)	0(0)	0(0)	0(0)
BUL		0(0)	0(0)	0(0)	0(0)	0(0)
CCF		73(2.31)	8(0.14)	4(0.11)	0(0)	0(0)
CCP		738(23.36)	295(4.98)	522(15.28)	105(2.79)	27(0.9)
CHM		159(5.03)	640(10.81)	582(15.28)	305(8.1)	245(8.13)
COH		1(0.03)	0(0)	1(0.03)	1(0.03)	0(0)
DAC		0(0)	0(0)	0(0)	0(0)	0(0)
FCH		0(0)	0(0)	0(0)	2(0.05)	2(0.07)
LMB		30(0.95)	1(0.02)	26(0.68)	29(0.77)	0(0)
LMP		1(0.03)	0(0)	1(0.03)	0(0)	0(0)
LND		2(0.06)	26(0.44)	148(3.89)	40(1.06)	26(0.86)
MWF		10(0.32)	255(4.31)	14(0.37)	900(23.91)	900(29.85)
NSF		79(2.50)	65(1.1)	66(1.73)	220(5.85)	185(6.14)
PMK		310(9.81)	0(0)	28(0.74)	16(0.43)	1(0.03)
PMO		1(0.03)	0(0)	0(0)	0(0)	0(0)
RBT		0(0)	0(0)	0(0)	1(0.03)	0(0)
RSS		6(0.19)	165(2.79)	2(0.05)	1070(28.43)	500(16.58)
SCU		0(0)	0(0)	0(0)	0(0)	0(0)
SMB		1191(37.7)	3156(53.3)	1357(35.62)	7(0.19)	0(0)
SPC		0(0)	0(0)	0(0)	20(0.53)	75(2.49)
SPD		0(0)	10(0.17)	5(0.13)	10(0.27)	0(0)
STB		0(0)	0(0)	0(0)	0(0)	0(0)
SUK		451(14.28)	1300(21.96)	1052(27.61)	1030(27.37)	1050(34.83)
WCR		5(0.16)	0(0)	2(0.05)	7(0.19)	0(0)
WSH		0(0)	0(0)	0(0)	1(0.03)	3(0.1)
WST		0(0)	0(0)	0(0)	0(0)	0(0)
YLP		16(0.51)	0(0)	0(0)	0(0)	1(0.03)

[a] BBH(brown bullhead), BLG(bluegill), BUL(bull trout), CCF(channel catfish), CCP(common carp), CHM(chiselmouth), COH(coho salmon), DAC(dace spp.), FCH(fall chinook), LMB(largemouth bass), LMP(lamprey spp.), LND(longnose dace), MWF(mountain whitefish), NSF(northern squawfish), PMK(pumkinseed), PMO(peamouth), RBT(rainbow trout), RSS(redside shiner), SCU(sculpin spp.), SMB(smallmouth bass), SPC(spring chinook), SPD(speckled dace), STB(stickleback), SUK(sucker spp.), WCR(white crappie), WSH(wild steelhead), WST(white sturgeon), YLP(yellow perch).

Discussion

The lower Yakima River is not what it used to be. Humans have influenced this environment to favor non-native over native fish taxa by simplifying habitat and increasing summer water temperature and decreasing water quality through irrigation system development and operations. Historically, northern squawfish were probably the primary predatory fish in the lower Yakima River. Currently, non-native channel catfish appear to dominate the fish predator community in the lower Yakima River from the mouth to about West Richland. Smallmouth bass appear to be the dominant predatory fish between West Richland and Prosser Dam. Between Prosser Dam and the city of Yakima, northern squawfish are the most abundant piscivorous fish species. Our efforts were aimed at establishing methods which would enable us to monitor piscivorous fish abundance and distribution through time.

We were relatively unsuccessful in establishing methods that would allow us to accurately estimate the number of piscivorous fish present in index reaches of the lower Yakima River. In 1997, we were faced with anomalous river conditions due to an above average snow pack in the mountains. The effects of this snow pack on our efforts were related to river discharge, temperature, and turbidity. The water temperature during the spring smolt outmigration period (April-May) in 1997 was 8 - 10^{o} C cooler than normal. Further, discharge was about two to five times above average. These high flows also increased water turbidity. These conditions combined to reduce our electrofishing capture efficiency substantially and delayed the initiation of our spring population estimates by several weeks. Because the environmental conditions we experienced in 1997 were so unusual, one should be cautioned that our results may not reflect ‘normal’ conditions in the lower Yakima River. Our low capture efficiency combined with the behavior of the fish effectively reduced our chances of being able to accurately assess the population abundances of the predatory fishes. However, these conditions might favor the species of concern and reflect a more normal historical river condition.

Smallmouth bass were moving upstream during the spring period, which violates an assumption of the mark-recapture methodology (no movement in - no movement out between mark and recapture samples). One bass we tagged on our initial marking run below Horn Rapids Dam was recaptured in the index section above the dam six days later. Mark-recapture estimates under more favorable to this effort environmental conditions and with a shorter delay between mark and recapture samples may provide somewhat more precise and accurate population estimates of smallmouth bass and northern squawfish. Summer population estimates and size structures of predatory fishes were quite different from those in the spring period. The comparison between spring and summer data as well as the tag return information confirmed that smallmouth bass move freely between the Yakima and Columbia rivers. Most of the large bass in the Yakima River during the spring appear to migrate back down to the Columbia River in late spring or early summer. Therefore, future attempts to characterize piscivorous fish populations that may reduce salmonid smolt survival through the lower Yakima River should focus on the spring period around the time of peak smolt outmigration.

Channel catfish populations were not estimated. Because channel catfish were not very susceptible to capture by electrofishing, especially during the spring, we were not able to estimate their abundance using the techniques we used on northern squawfish and smallmouth bass. Based on the large numbers of channel catfish we captured in gill nets, traps, and by angling and electrofishing (N=1126), and the low number (N=15) of tagged catfish we recovered, we assume that the density of channel catfish in the lower Yakima River is extremely high. If , for discussion purposes, we apply our capture efficiency (1.3%) to the total number we captured (1126/0.013) we would estimate 86,600 catfish in the area where we worked. Our catch rates of channel catfish in hoop nets and traps after the water temperature exceeded 14^{o} C (0.81 to 3.80/net-day) were higher than most values presented in the literature for studies where similar equipment was used (0.59/net-day, Starrett and Banickol 1955; 0.55/net-day, Pierce et al. 1981; 0.75-0.85/net-day, Gerhardt and Hubert 1989). It may be possible to better estimate channel catfish abundance by stratifying sampling and employing a multiple mark-recapture methodology (Everhart and Youngs 1981). We know little about the movement patterns of these fish in the Yakima River, but it is likely that there is exchange with the Columbia River. One 505 mm FL channel catfish that was tagged in the lower Yakima River on July 15 was caught by an angler at the confluence of the Snake and Columbia rivers on August 8, after swimming nearly 21 km.

The predatory species we examined all ate fish. Stomach contents of smallmouth bass and northern squawfish were quite similar; about a quarter of the stomachs from both species contained salmonids. Channel catfish ate fish but we did not identify any salmonids in catfish stomachs possibly because most channel catfish stomachs were collected when water temperatures were relatively cold (below 15^{o} C). Most of the salmonids found in stomachs were fall chinook parr. The YIN released about 1.7 million fall chinook during the study period in the Yakima River upstream of where we were collecting samples. Spring chinook salmon smolts were relatively rare in 1997 due to very low natural production of fry in the spring of 1996. Until we examine stomach contents of predatory fishes when both fall (60-80 mm FL) and spring chinook (120-150 mm FL) are available, we will be unable to elucidate any predator preferences between the two life history forms of the species.

We found that numbers of predatory fishes are high enough in the lower Yakima River to warrant long-term monitoring of a salmonid predation index. We were successful in determining which methods were effective for capturing the three primary piscivorous fish species present in the lower Yakima River. We also learned a great deal about the life history and movement patterns of these fishes. Further, we found that absolute population estimates may not be feasible during spring flows. Incorporating these findings into a plan for future efforts to monitor salmonid smolt predation by fish in the lower Yakima River will be useful. Developing a predation index that incorporates both a measure of predator density and consumption information appears to be the best approach to monitoring fish predation on salmonids in the lower Yakima River over the long term (Busack et al 1997; Ward et al. 1995). Predator density may best be monitored by considering both CPUE and estimated abundance based on mark-recapture population estimates with a short interval (24-48 h) between the mark and recapture runs (to minimize chance of fish movement between samples). Consumption indices should be developed by incorporating stomach sample data and predator and prey fish size structure data. Furthermore, experiments to determine thermal thresholds for channel catfish predation on salmonid smolts may be useful in determining whether channel catfish need to be considered in the long term monitoring of a predation index for fishes in the lower Yakima River.

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Chapter 13

Minimizing Ecological Impacts of Hatchery-reared Juvenile Steelhead on Wild Salmonids in a Yakima Basin Tributary

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Abstract

Increasingly, adverse ecological effects on wild fish resulting from releases of hatchery-reared fish are being scrutinized and balanced against benefits afforded by hatchery programs. Various measures to limit the risks of ecological interactions have been applied for some time, but there is room for better information and approaches. In an attempt to improve understanding of the potential ecological effects of hatchery steelhead (anadromous form of *Oncorhynchus mykiss*) on wild trout (resident *Oncorhynchus* species) populations, we released 23,000 to 38,000 hatchery-reared steelhead smolts into a tributary to the Yakima River, Washington, annually from 1991 through 1994. We snorkeled in control and treatment streams to observe behavioral interactions between hatchery steelhead and wild salmonids. Movement of residual hatchery steelhead was examined through the use of traps and direct underwater observation, and abundance was estimated by electrofishing. Instream enclosures were used to determine whether residual hatchery steelhead impacted growth of wild rainbow trout or spring chinook salmon (*O. tshawytscha*). Potential for adverse impacts resulting from ecological interactions among wild salmonids and hatchery steelhead was greatest when; (1) hatchery fish did not emigrate quickly; (2) water temperatures were over 8^{o} C; (3) hatchery fish were the same species as the wild salmonids; (4) hatchery fish were larger than the wild salmonids; (5) habitat and/or food were limiting; and 6) numbers of fish released was over about 30,000. Ecological interactions with wild salmonids could be reduced or minimized by releasing; (1) only actively migrating smolts (no residuals); (2) hatchery fish of a size that minimizes interaction potential (smaller than wild fish); (3) the minimum number necessary to meet management objectives; (4) fish that do not exhibit counter-productive and inappropriate behaviors (e.g., less likely to engage wild fish in agonistic encounters); and (5) when water temperatures are relatively cold (less than 8^{o} C) . To minimize risks of adverse ecological interactions, hatchery steelhead should only be released in areas where; (1) coexisting wild salmonid populations are either absent or abundant/healthy; (2) limitations to wild populations exist due to a density-independent pre-smolt stage bottleneck; and (3) habitat diversity is complex. Management actions that encourage angler harvest of residuals while protecting coexisting wild species may also help minimize ecological impacts of hatchery steelhead releases on their wild counterparts. Implementing these strategies may reduce the number of returning hatchery-origin adults of the target group but will help reduce risk to the sustainability of wild fish populations.

Introduction

Conservation of endemic wild fish populations in the Pacific Northwest has received increased emphasis in recent years (Nehlsen et al. 1991; NMFS 1995, Huntingford et al. 1996; NRC 1996; Allendorf et al. 1997). Potential adverse impacts of hatchery programs on wild salmonid populations has been one of many conservation issues. Managers have acknowledged that competitive interactions with wild stocks as a result of hatchery stocking can have adverse consequences, and new management guidelines are beginning to emerge in efforts to reduce these effects (IHOT 1995). Due largely to the failure of many anadromous fish hatchery programs to meet expectations, and the continued decline of wild fish populations (Meffe 1992), the public is becoming more involved in decision-making processes regarding the use of hatchery fish (WDFW 1997). Most research on hatcheries has focused on ways to alter hatchery methods to increase post-release survival of hatchery fish (e.g., Banks 1994; Jarvi and Uglem 1993; Wiley et al. 1993). Few studies have directly examined the effects of hatchery fish on wild populations.

The Yakima River has one of the premier wild resident rainbow trout (*Oncorhynchus mykiss*) fisheries in Washington State (Krause 1991; Probasco 1994). Concerns that proposed steelhead supplementation (Clune and Dauble 1991) might negatively impact this fishery prompted a five year study, beginning in 1991, of the potential ecological effects of releasing hatchery steelhead juveniles (judged to have reached the smolt stage) on coexisting wild salmonids in streams (McMichael et al. 1992; 1994; 1997; Pearsons et al. 1993; 1994; 1996). Results from this work helped us form hypotheses regarding the ecological impacts of hatchery steelhead releases on wild salmonids, particularly on resident rainbow trout. Specifically, our objectives were to (1) determine whether releases of hatchery steelhead affected wild salmonids, and (2) develop guidelines to reduce ecological impacts of hatchery steelhead releases on wild salmonids. For our uses, ecological impacts are defined as any or a combination of the following: behavioral dominance of hatchery fish over wild fish, displacement of wild fish by hatchery fish, reduced distribution of wild fish due to the presence of hatchery fish, decreased abundance of wild fish due to the presence of hatchery fish, and decreased growth of wild fish when hatchery fish are present.

Methods

We used a control-treatment approach, in which streams not receiving hatchery steelhead were considered controls, while those that were stocked with steelhead were the treatment streams. We released 23,000 - 38,000 hatchery-reared smolt-sized steelhead annually from 1991 through 1994 into Jungle Creek, a tributary of the North Fork of the Teanaway River, north of the Town of Cle Elum, Washington (Figure 1). Most of the hatchery steelhead smolts released into Jungle Creek (small treatment stream) moved downstream into the North Fork of the Teanaway River (large treatment stream). Hatchery steelhead that did not emigrate during the time when wild steelhead did were defined as residuals. Residuals may take up residence in streams or may outmigrate in later years (Pearsons et al. 1993). The hatchery steelhead were progeny of adult steelhead collected lower in the Yakima River basin. Table 1 shows the relevant data on adult steelhead used to produce fish for these experimental releases. Jack Creek, a small tributary to the North Fork of the Teanaway River, was blocked off to prevent hatchery steelhead from moving up into it. Jack Creek was used as a small control stream. The large control stream, the Middle Fork of the Teanaway River, is in an adjacent basin and is similar in size and other physical and biotic makeup to the large treatment stream.

Snorkeling was used to (1) observe behavioral interactions between hatchery steelhead and wild salmonids in index sites in two streams where hatchery steelhead were present (treatments) in comparison with two streams where no hatchery fish were released (controls) in 1991-1994, and (2) determine the distribution of residual hatchery steelhead within the North Fork of the Teanaway River in 1994. Direct underwater observation of fish behavior was performed by snorkeling in control and treatment streams as described by McMichael et al. (1992) and Pearsons et al. (1993). Each agonistic interaction was classified into one of the following five groups, threat, crowd, chase, nip, or butt. We defined threats as overt signs of aggression, such as fin-flares and body arching (Taylor and Larkin 1986; Holtby et al. 1993). Crowds occurred when fish moved toward other fish laterally, causing a subordinate fish to move out of the way (Helfrich et al. 1982; Taylor and Larkin 1986; Holtby et al. 1993). Chases occurred when one fish slowly pursued another fish for several body lengths without making physical contact (Keenleyside and Yamamoto 1962; Helfrich et al. 1982; Taylor and Larkin 1986). Nips were classified as physical contact in which one fish actively bit another fish (Stringer and Hoar 1955; Helfrich et al. 1982; Taylor and Larkin 1986). Physical contact made between two fish in which the mouth of the attacking fish was closed was classified as a butt. A contest may have included multiple interactions. For example, a hatchery steelhead and a wild rainbow trout could chase and nip each other several times during one contest. A fish was considered to be dominant if it displaced (defined below) its opponent in a contest. Interaction rates standardized for each observation period by dividing the number of interactions observed by the number of fish observed and the number of minutes spent observing (i.e., interaction rate = interactions/fish/minute).

To determine the extent of spatial overlap between residual hatchery steelhead and resident rainbow, cutthroat (*O. clarki*) and bull trout (*Salvelinus confluentus*) in the North Fork of the Teanaway River, we snorkeled pools and runs at 0.6 km intervals from the mouth of Jungle Creek upstream to a point 13.4 km upstream of the mouth of Jungle Creek. The reason we sampled in the North Fork of the Teanaway River above the confluence with Jungle Creek was because that is where the cutthroat and bull trout are present. Snorkeling took place on June 24, June 30, and July 12, 1994. A total of 22 sites were sampled. Each site was sampled by two snorkelers, each covering approximately 100 m of stream separated by about 100 m. Snorkeling was repeated at the upper nine sites on the night of July 12, 1994, to better estimate bull trout presence. No bull trout were observed during daylight snorkeling in these areas, therefor data collected at night was used for the upper nine sites. The relative abundances of bull trout, cutthroat trout, rainbow trout, and hatchery steelhead were expressed as percentages of all salmonids observed at each site.

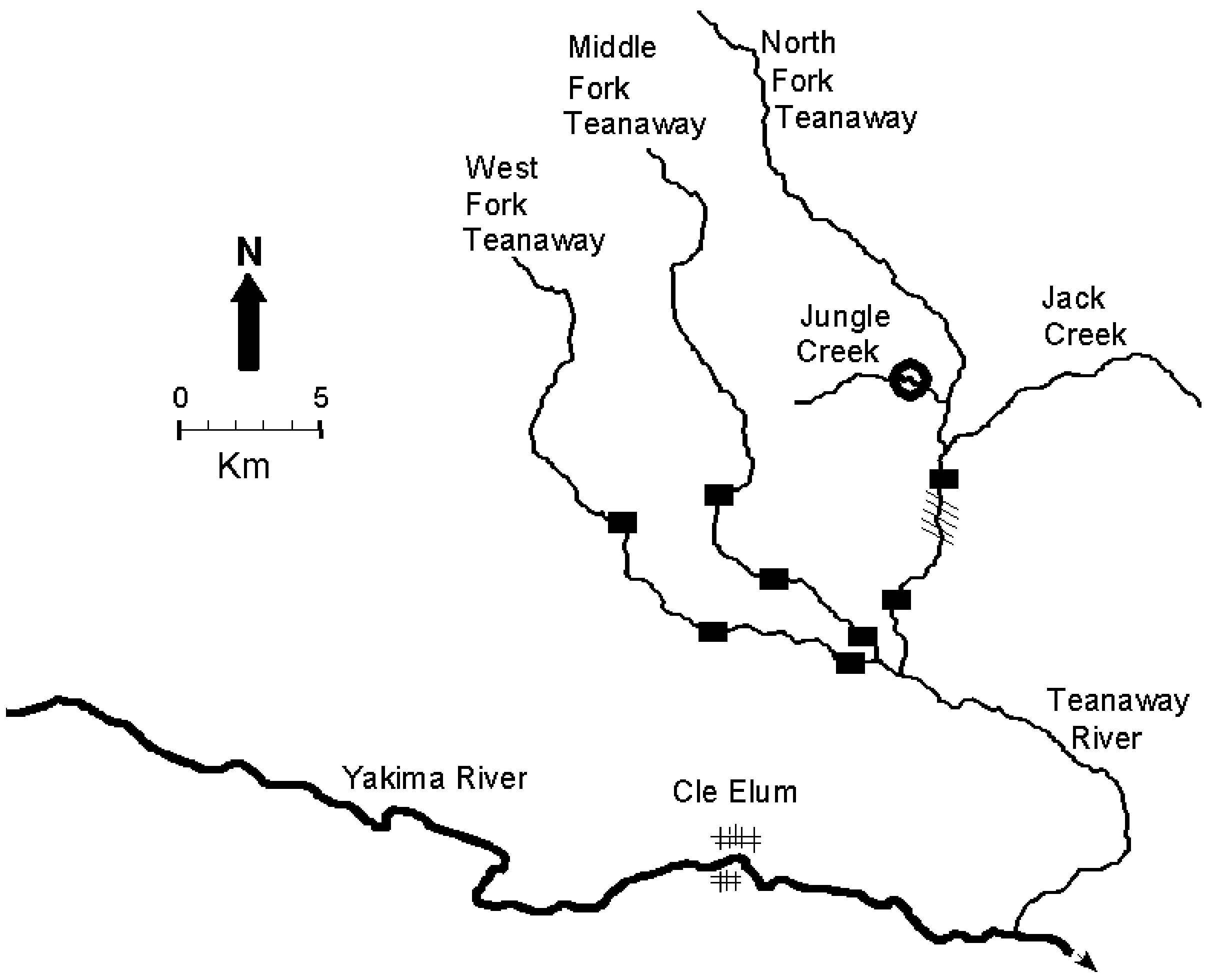

Figure 1. Map of the upper Yakima River basin showing study streams in the Teanaway River basin. The hatchery steelhead release location on Jungle Creek is marked with an open circle. Filled squares indicate 100 m-long population estimate sites. The cross-hatched area on the North Fork of the Teanaway River is where small enclosure growth experiments were conducted.

Table 1. Parentage (number, % hatchery and % wild) and loading density of smolts (kg of fish/l/min in the raceway immediately prior to release) for hatchery steelhead smolts released into Jungle Creek in 1991, 1992, 1993, and 1994.

		Parents			Progeny
Brood Year	Release Year	Number	Percent Hatchery	Percent Wild	Loading Density
1990	1991	106	0	100	0.46
1991	1992	24	63	37	0.19
1992	1993	26	100	0	0.16
1993	1994	25	24	76	0.20

To determine whether juvenile hatchery-reared steelhead displaced wild fish we examined displacement at three spatial scales using trapping and snorkeling methods described by McMichael et al. (1992) and Pearsons et al. (1993). We defined a displacement as one fish causing another fish to move at least two body lengths away from a preferred feeding or holding site (Brown (1975) as cited *in* Helfrich et al. 1982). Small-scale displacements were those that occurred within a channel unit of stream, such as a pool. Wild fish movement out of the release stream (determined by captures in a downstream migrant trap at the mouth of Jungle Creek) concurrent with large numbers of hatchery fish was considered a mid-scale displacement. Large-scale displacement was monitored at a downstream migrant trap near the mouth of the North Fork of the Teanaway River, approximately 11 km downstream of the release site in Jungle Creek. Determination of small-scale displacements was more direct (because they were observed) and involuntary than mid- and large-scale displacements which could be termed voluntary and had to be inferred from fish emigration information obtained by trapping. Downstream migrant traps, in combination with electrofishing, were also used to assess the proportion of *Saprolegnia* infection in hatchery and wild salmonids.

To determine the influence of hatchery steelhead releases on rainbow trout abundance, population estimates were conducted in three study streams. Multiple-removal population estimates were conducted in index sites in the North (N = 2), Middle (N = 3), and West (N = 3) forks of the Teanaway River (1990-1995).

Growth experiments were performed in in-stream enclosures in 1993 and 1994 in the North Fork of the Teanaway River to determine the effects of residual hatchery steelhead on growth of wild rainbow trout and spring chinook salmon. Enclosures were wood-framed boxes, a cubic meter in size, divided in the middle and wrapped with 0.95 cm hardware cloth on the sides and bottom to allow free passage of invertebrates in and out of the enclosures (Cooper et al. 1990; McMichael et al. 1997). Enclosures were deployed with one wild rainbow trout or salmon on one side (control) and one wild rainbow trout or salmon with a residual hatchery steelhead (treatment) on the other side for a six week period in mid-summer (McMichael et al. 1997). Specific growth rates of trout and salmon with and without residual hatchery steelhead were compared to determine impacts of residual hatchery steelhead on growth of wild rainbow trout and spring chinook salmon.

Results

The ecological impacts from releases of hatchery steelhead smolts on wild salmonids were greatest when; (1) hatchery fish did not emigrate quickly; (2) water temperatures were above 8^{o} C; (3) wild salmonids were the same species as the hatchery fish; (4) wild salmonids were smaller than hatchery fish; (5) habitat and/or food were limited, and; (6) numbers of fish released were at the high end of the range we investigated.

Large numbers of residual steelhead were present in our treatment streams in three of the four years we released hatchery steelhead (Figure 2). Abundance and biomass of residual steelhead varied but often exceeded those of wild rainbow trout in the release stream during the summer rearing period (Table 2; McMichael et al. 1994). Residual hatchery steelhead were also observed over 12 km upstream from their release point in the North Fork of the Teanaway River, in an area containing populations of bull and cutthroat trout in addition to rainbow trout (Figure 3).

When smolt-sized (over 175 mm FL) hatchery steelhead failed to promptly migrate toward the sea and instead remained in freshwater as residuals, they behaviorally dominated the typically smaller coexisting wild salmonids (McMichael et al. 1992; 1994; Pearsons et al. 1993;1996). Larger salmonids typically dominated smaller ones, regardless of hatchery or wild origin or species. The wild rainbow trout within our study area averaged around 125 mm FL while hatchery steelhead we released averaged between 179 and 201 mm FL (McMichael et al. 1994). Despite this size disparity, rainbow trout and residual hatchery steelhead were often observed occupying the same or similar habitats. In 84% of the contests we observed between 1991 and 1994, larger fish were judged to be dominant. Hatchery steelhead also dominated over 75% of the contests between steelhead and wild rainbow trout (McMichael et al. 1994a; Pearsons et al. 1996). Interactions among all salmonids observed in 1994, in streams where residual hatchery steelhead were present, more often involved physical contact (nips and butts; 25%) than those observed in control streams (14%)(Pearsons et al. 1996). The most common type of behavior observed in contests in control streams was a threat (47%), while only 20% of the behaviors displayed by fish in treatment streams were classified as threats (Pearsons et al. 1996).

Interaction rates were highest when stream temperatures were over 8^{o} C. When water temperatures were below 8^{o} C we often had a difficult time locating wild salmonids to observe, presumably because they were holding within the substrate (Hillman et al. 1992). In many locations throughout the upper Yakima basin we have seen far fewer fish when water temperatures were below 8^{o} C than when temperatures were higher. When wild salmonids were not in the water column behavioral interactions with hatchery steelhead were precluded, as hatchery fish were not observed utilizing interstices at lower temperatures. It is possible that wild trout may have emerged from the interstices during hours of darkness, but we did sample during darkness as lights were found to alter the behavior of hatchery steelhead (McMichael et al. 1992). In all four years, interaction rates in the stream where the hatchery fish were released were higher after June 1 than before (McMichael et al. 1994; Pearsons et al. 1996). Water temperatures generally exceeded 8^{o} C by June 1 of each year. June 1 is also the date after which we defined all hatchery steelhead present to be residuals.

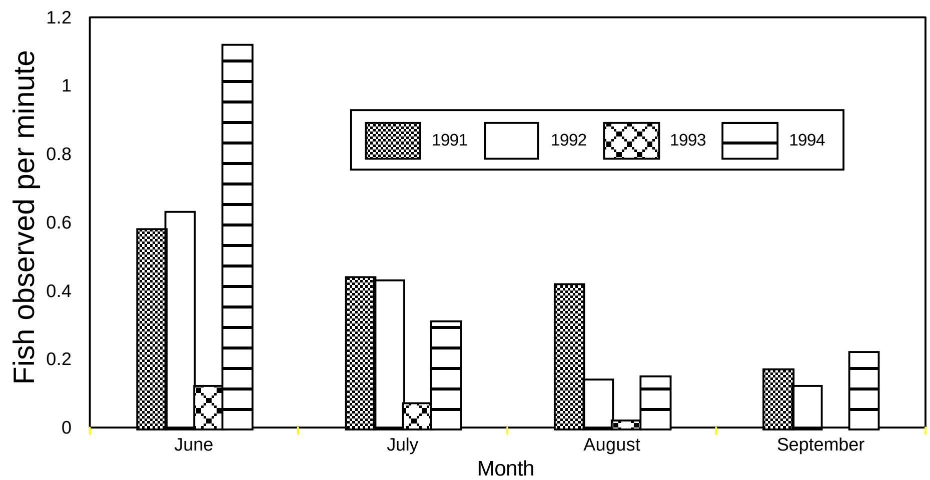

Figure 2. Frequency of residual hatchery steelhead observed during snorkeling activities in treatment streams during the summer and early fall of 1991, 1992, 1993, and 1994.

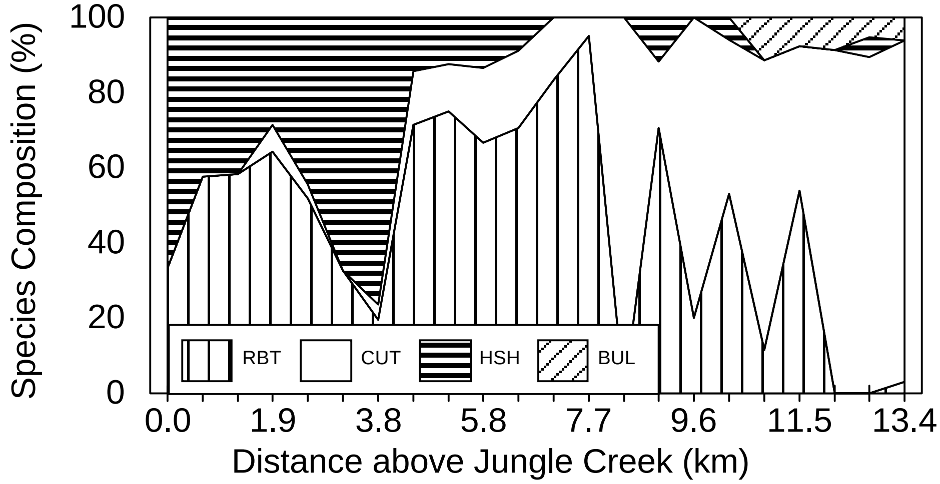

Figure 3. Linear distribution of salmonids in the North Fork of the Teanaway River upstream of the mouth of the release stream on June 24, 30, and July 12, 1994. RBT = rainbow trout, CUT = cutthroat trout, BUL = bull trout, and HSH = residual hatchery steelhead. An average of 27 salmonids were observed at each of 22 sites (0.6 km apart). Sites from 8.3 to 13.4 km were snorkeled at night, while all others were snorkeled during daylight.

Table 2. Estimated wild rainbow trout and residual hatchery steelhead number and biomass in Jungle Creek, 1991-1994. Index site was 100 m long and located immediately upstream of the confluence of Jungle Creek and the North Fork of the Teanaway River.

	Number		Biomass (g)	
Year	rainbow trout	steelhead	rainbow trout	steelhead
1991	5	9	47	530
1992	2	13	21	468
1993	40	11	477	453
1994	2	5	19	220

In addition to higher interaction rates, the incidence of disease in both residual hatchery steelhead and wild fish was higher when water temperature was higher. In 1991, when water temperatures were very high in late May and June (often exceeding 20^{o} C), *Saprolegnia* infections were commonly observed on hatchery fish and, though less common, were also observed on wild fish within the release stream (McMichael et al. 1992). The incidence of *Saprolegnia* on hatchery steelhead emigrating from Jungle Creek between May 29 and June 13, 1991, was 13.2% (N = 53). A sample of fish collected by electrofishing in Jungle Creek on June 25, 1991, showed 32.1% of the hatchery steelhead (N= 28) and 16.7% of the wild resident trout (N = 6) were infected. We did not directly sample in control streams to determine infection rates, but none were observed in those streams during underwater observations. Infection sites were generally on the lateral body surface near the base of the dorsal fin which corresponds with the area where most of the violent agonistic attacks we observed were targeted (McMichael et al. 1992). It is likely that the combination of high densities of residual hatchery steelhead, elevated rates of agonistic interactions, and warm water temperatures combined to increase stress levels in many salmonids to the point that they were vulnerable to fungal infection by *Saprolegnia*.

Wild trout were displaced from apparently preferred microhabitats by hatchery steelhead, but were not generally displaced over larger spatial scales (0.2 to 11.2 km) (Pearsons et al. 1996). Small-scale displacement of subordinate fish occurred in 47-59% of the contests observed in treatment streams in 1993 and 1994 (McMichael et al. 1994; Pearsons et al. 1996). Displaced fish did not return to their previous locations prior to the end of the observation period (typically 20 minutes).

Hatchery steelhead residuals had a greater impact on conspecifics than on other species, such as spring chinook salmon. Growth experiments revealed that rainbow trout paired with hatchery steelhead residuals had significantly lower specific growth rates than their unpaired counterparts (1993: $P = 0.019$; 1994: $\underline{P} = 0.020$; McMichael et al. 1997). When spring chinook salmon were paired with residual hatchery steelhead however, their growth was not significantly reduced (P = 0.360; McMichael et al. 1997).

Abundance estimates of wild rainbow trout in sites within the large treatment stream (the North Fork of the Teanaway River) where hatchery steelhead were released were significantly lower than in sites in adjacent control streams ($P = 0.02$; McMichael et al. 1994; Pearsons et al. 1996)(Figure 4). A complex relationship existed between the host of variables we examined. For example, in some years residualism was very high (1991 and 1994; Figure 2), and interaction rates were low (1991) to high (1994)(Pearsons et al. 1996). The interrelatedness of these variables (e.g., interaction rates, proportions of residuals, displacement), and their combined effects on response variables such as the population estimates shown in Figure 4 were beyond the scope of this study.

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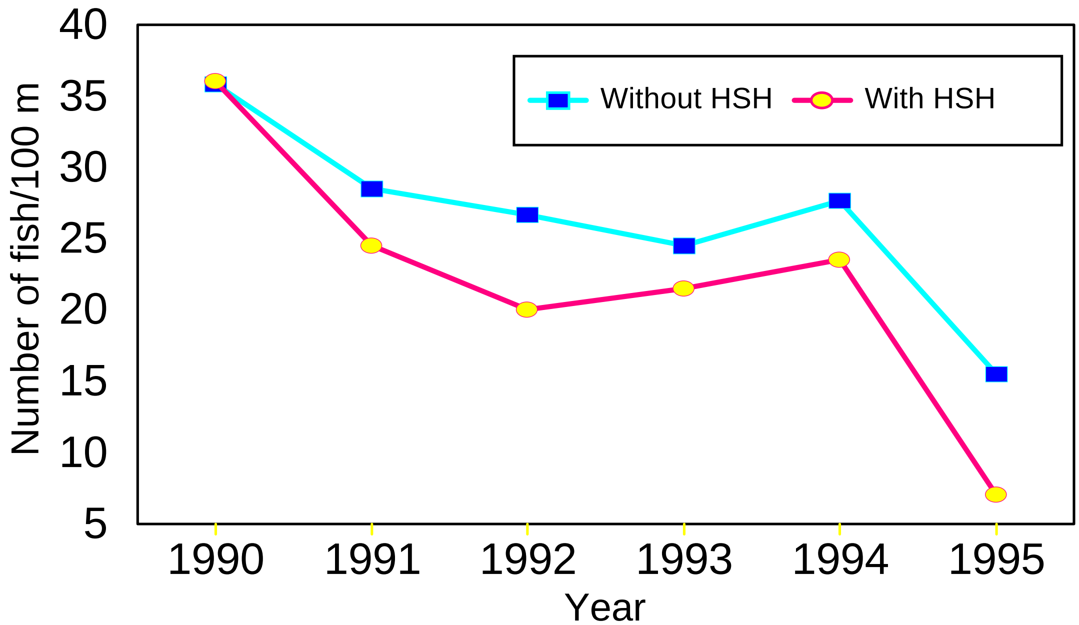

Figure 4. Mean wild rainbow trout population abundance estimates (number of fish per 100 m) for streams without hatchery steelhead and with hatchery steelhead between 1990 and 1995. Index sites were 100 m in length and were sampled by electrofishing, using the multiple -removal methodology (Zippin 1958). Hatchery steelhead smolts were released in 1991, 1992, 1993, and 1994.

Discussion

Hatchery steelhead may adversely interact with wild salmonids under certain conditions. We fully appreciate that wild fish populations face a variety of risks besides those from ecological interactions with hatchery fish (e.g., direct or incidental harvest, extreme environmental variation, habitat degradation and loss, genetic introgression and hybridization). However, various measures exist that could be applied to reduce or minimize undesirable ecological effects of hatchery steelhead on wild fish. Those measures should reflect the information available about conditions under which interactions might be reduced. There are four general categories of actions culturists and managers can control in situations where hatchery steelhead will be released which will reduce ecological effects of hatchery steelhead on wild salmonids, including; (1) the hatchery product - *what* is released; (2) location - *where* the fish are released; (3) timing - *when* the fish are released, and; (4) *how* the fish are released. Guidelines to reduce the impacts of released hatchery steelhead on wild salmonids are presented below.

The Hatchery Product - what is released

The hatchery product slated for release should be an active migrator, resulting in no residuals. Tools are becoming available with potential for reducing the number of hatchery steelhead that are introduced into streams and later residualize. For example, Viola and Schuck (1995) successfully reduced the number of residual steelhead in the Tucannon River in southeastern Washington by periodically examining the juveniles in an acclimation pond prior to release and closing the exit gate when most emigrants no longer exhibited smolt characteristics and were predominantly males; thus potential residuals were retained. Hatchery practices affecting rearing densities, size at release, and parentage may influence residualism in steelhead. In our study, hatchery steelhead that were offspring of hatchery-origin parents and reared at low densities residualized less and survived better than offspring of wild parents that were reared at higher densities (Tables 1 and 3; McMichael 1994; Pearsons et al. 1996). We did not conduct paired tests of these factors and are therefore unable to discern the relative effects of parentage and rearing density on smolt survival or residualism.

Increased growth rates and large size at release of fish reared in hatcheries have often been implicated in the production of residual anadromous salmonids (e.g., Rowe and Thorpe 1990; Mullan et al. 1992). In a study on the upper Salmon River, Idaho, Partridge (1986) found that hatchery steelhead released at a ‘large-size’ (mean FL = 265 mm; number released = 40,322) constituted 92% of the residual steelhead catch by anglers, while steelhead smolts released at ‘normal-size’ (mean FL = 202 mm; number released = 39,763) represented less than 8% of the catch.

The number of residuals expected for any release group may vary considerably. We estimated that 26 - 39% of the fish we released (23,000 - 39,000) did not emigrate from the study area within a month after release. Other researchers have reported non-migrant portions of hatchery steelhead releases between 19.8% (Tipping et al. 1995) and 65.5% (Tipping and Byrne 1996). However, non-migrant steelhead smolts do not always become residual steelhead, many are lost to predation or other sources of mortality and some may emigrate a year or more later as age 2 or 3 smolts (Pearsons et al. 1993). In our work, if we assume all non-migrants became residuals, then an estimated 5,980 -15,210 residual hatchery steelhead were in an area supporting an estimated 3,800 wild rainbow trout (Figure 5). Even applying a conservative residualism rate of 15% (derived from a non-migrant rate of 30%, only half of which survived and remained as residuals) and our lowest release number, we would have expected nearly as many residual steelhead (3,450) as rainbow trout (3,800) in the 10 km reach of the North Fork of the Teanaway River below where the steelhead were released. It should be pointed out that Figure 5 only projects residualism estimates for releases of up to 50,000 fish. Most steelhead releases involve larger numbers of smolts. For example, the average number of smolt-size steelhead released from State and tribal hatcheries in Washington watersheds in 1994 was 87,780 (range = 4,000 to 794,700)(WDFW 1995).

Social interactions between actively emigrating hatchery steelhead smolts and wild salmonids may have only short term adverse consequences for wild fish. One such mechanism is the indirect effect of predation on wild fish through forced habitat shifts resulting from dominance of hatchery fish over wild fish (e.g., Abrahams and Healey 1993; Walters and Juanes 1993). Even when hatchery smolts emigrate promptly, the combined effects of higher social interaction rates and longer duration of spatiotemporal overlap associated with the occurrence of those residuals that do exist increases the risk of adverse outcomes. Therefore, fish culturists and managers should carefully address the potential for a residual component in any hatchery steelhead release group.

Table 3. Number of hatchery steelhead estimated to have passed Prosser Dam during May, June, and July in 1991, 1992, 1993, and 1994. Data are expressed as estimated total number, and the percentage of the number released into Jungle Creek that were estimated to have passed Prosser Dam within three months of their release (M. Kohn and M. Johnston, Yakama Indian Nation, personal communications).

Outmigration Year	Est. Total (number)	Percentage of number released
1991	648	1.9%
1992	575	1.5%
1993	5,592	24.9%
1994	837	2.6%

It has been well established that a positive correlation exists between fish size and social dominance for salmonids (e.g., Abbott et al. 1985; Huntingford et al. 1990). Abbott et al. (1985) reported that a weight advantage of as little as five percent was sufficient to assure dominant status for larger steelhead juveniles. Behavioral data from our field studies indicated that in most contests (84%) socially dominant fish were larger than subordinates. To minimize the potential for hatchery fish to socially dominate wild fish, the size of hatchery steelhead released should be smaller than their wild counterparts. This smaller size may conflict with size criteria that maximize emigration, and should therefore be secondary to tactics that result in few or no residuals. In many, if not most, instances it may be difficult to meet competing objectives of smaller hatchery fish size and effective smoltification. For example, in our study area, where wild rainbow trout averaged 125 mm FL during the summer rearing period (McMichael et al. 1994), it would be difficult to produce steelhead juveniles in the hatchery that became smolts at a size smaller than 125 mm FL. Tipping et al. (1995) reported that hatchery-reared winter steelhead over 190 mm in FL migrated at higher rates than smaller hatchery fish, suggesting there may be conflicts between objectives aimed at producing hatchery smolts that will emigrate quickly, yet will not dominate smaller wild fish in behavioral interactions. It is conceivable that hatchery steelhead smolts that were much larger than their wild counterparts would occupy different habitats than the smaller wild fish (Roper et al. 1994), thereby reducing interactions and their effects. There is a tradeoff in this approach, however, in that hatchery steelhead that are much larger than the wild fish may become predators of age 0 wild salmonids (Cannamela 1992; Martin et al. 1993). It is noteworthy that, even with the size disparity between the hatchery fish and wild fish (a mean difference of about 50-70 mm) we observed, many hatchery and wild fish occupied the same habitats.

The number of hatchery fish released is a key influence on the level of ecological interactions experienced by wild fish. We suggest that only the minimum number of hatchery steelhead smolts necessary to meet management objectives should be released in any given situation. The likelihood of adverse ecological interactions resulting from hatchery steelhead releases is positively correlated with hatchery fish abundance. When factors affecting residualism rates are equal (e.g., parentage, size at release, precocity, rearing density), hatchery steelhead releases of larger magnitude result in more residual steelhead in the freshwater rearing environment. In addition, hatchery managers are finding that fish reared at relatively low densities often produce the greatest percentage of adult hatchery fish returning (Banks 1994; Ewing and Ewing 1995), suggesting that reducing production levels from facilities might not result in reduced returns but could reduce ecological impacts.

Hatchery fish should not exhibit counter-productive and inappropriate behaviors following release into the natural environment. Inappropriate behaviors would include agonistic tendencies with no perceptible benefit to either fish in the interaction (Ruzzante 1994). Some innovative fish culture strategies may help reduce the undesirable behaviors expressed by many hatchery fish after release. Maynard et al. (1995) reviewed the use of seminatural culture strategies for anadromous salmonids and concluded that some of these innovative techniques may improve behavior with respect to increasing survival of hatchery fish. These authors recommended such strategies be used in both enhancement and conservation hatcheries, even though the efficacy of the strategies remains to be tested at those scales of production. While most refinements to standard hatchery protocols have focused on producing hatchery fish that survive at a higher rate in the wild (e.g., Banks 1994; Olla et al. 1994; Jarvi and Uglem 1993; Wiley et al. 1993), little research has been conducted on means of producing hatchery fish that, following release, are ecologically compatible with wild fish. Cuenco et al. (1993) stated that, "There is a paucity of information on the potential competition between the supplemented salmon population and other fish populations inhabiting the target stream." Certainly, this is an area deserving further research.

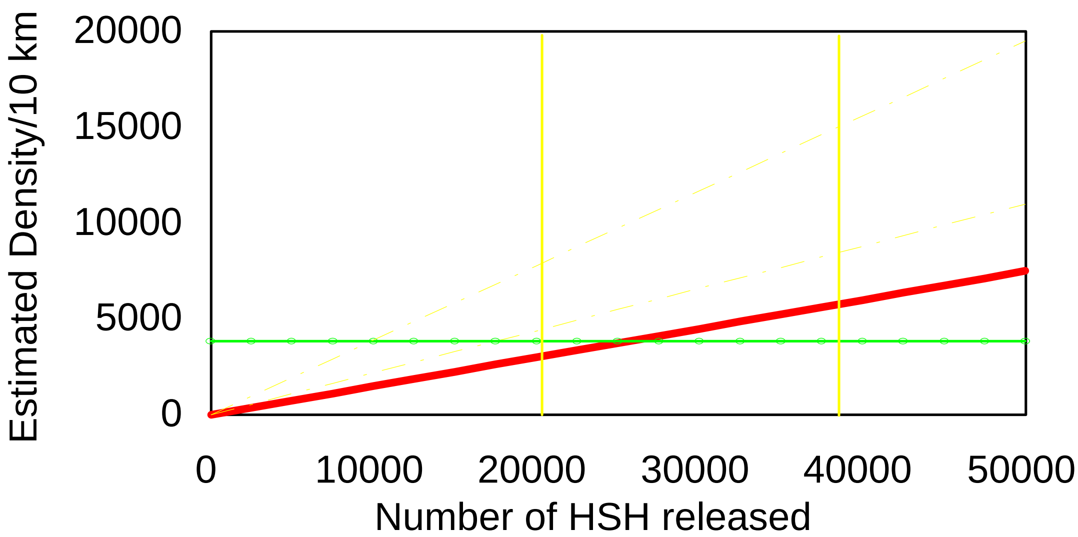

Figure 5. Estimated number of residual hatchery steelhead in a 10 km reach of the treatment stream versus total number of hatchery steelhead released. Dashed lines represent our observed residualism rates (26 to 39%), solid line represents a residualism rate of 15%, and the line with circles represents the estimated number of wild rainbow trout in that reach. Vertical lines show the range of numbers of hatchery steelhead we released annually between 1991 and 1994.

Location - where the fish are released

Viability of wild stocks would be impacted least if hatchery smolts were only released where recipient wild salmonid populations were either absent or abundant and healthy (e.g., reaching escapement goals). An exception to this may be in cases where there are very few healthy stocks of a species remaining (Huntington et al. 1996). Different management objectives may dictate when hatchery fish are to be released into a particular area. In areas containing depressed or critical stocks managers may use supplementation to restore populations, while in areas that are considered to have healthy populations, they may use hatchery fish solely to augment harvest. Hatchery releases in areas containing depressed or critical stocks are not advisable simply because they may add ecological stress to precarious stock status situations. In addition, in those circumstances, any impacts that do occur to wild populations will have greater per capita effects than for healthy populations. Even in areas where wild salmonid stocks are not present, potential effects on other species (e.g., non-salmonid fishes, aquatic invertebrates, and amphibians) should be recognized. Some might argue that hatchery fish should be excluded from systems with abundant or healthy wild salmonid populations to avoid any ecological risk to wild stocks (Huntington et al. 1996). The merits of this argument will increasingly deserve attention, especially when faced with decisions regarding new or proposed hatchery projects to augment fishery harvests where healthy wild stocks exist. For example, in areas that are known to have a density-dependent bottleneck after the smolt stage, it would not be advisable to release large numbers of hatchery steelhead smolts (Royal 1972, cited by Lichatowich and McIntyre 1987; Peterman 1984), because this would only exacerbate stresses on wild stocks, with negligible benefit in the form of returning adult hatchery fish.

Releases of hatchery steelhead smolts into areas that have complex habitat (e.g., a wide variety of depths, velocities, substrates and cover) would be expected to have the lowest ecological impact on wild salmonids. Steelhead follow diverse and often intertwining life-history trajectories that require many types of stream habitat during their freshwater rearing phase. Therefore, areas with diverse and abundant habitat provide an opportunity for wild fish to segregate themselves from their hatchery counterparts. Partitioning the habitat enables fish to reduce the intensity of competition through differential resource use and/or visual isolation (Allee 1981; Hearn and Kynard 1986; Roper et al. 1994; Raddum and Fjellheim 1995). Conversely, an argument could be made that hatchery smolts should be released only in simple habitat (very little variation in habitat features). This approach might avoid wild salmonid "strongholds" and encourage hatchery fish to emigrate more rapidly. However, our field studies have shown that hatchery steelhead smolts and residuals dominated wild fish in all habitat types examined. Further study of this issue is necessary before establishing firm guidelines.

Timing - when the fish are released

Release timing can have important effects on the extent to which adverse behavioral interactions occur between hatchery steelhead and wild salmonids. To minimize the chance for interactions, hatchery smolts should be released when they are naturally ready to migrate. After a certain date, or when smolt characteristics decrease and sex ratios heavily favor males in volitional release facilities (e.g., >75% male), access to streams should be precluded, to avoid release of non-migrant individuals. Non-migrant fish could be planted into lakes where ecological risks are not deemed unacceptable (Evenson and Ewing 1992), or the public could even be invited to fish in the acclimation pond itself (Viola and Schuck 1995).

Adverse interactions can be reduced by releasing hatchery steelhead smolts when water temperatures are relatively cold (less than 8^{o} C). When water temperatures were below about 8^{o} C, we found most wild salmonids were on or in the substrate, in contrast to hatchery fish that remained in the water column. This spatial segregation tends to reduce the likelihood of adverse interactions. In work done on the Wenatchee River, Washington, Hillman et al. (1992) observed less than 20% of the salmonids present when water temperatures were less than 9^{o} C. Cooler water also reduces the possibility for transmission and outbreak of diseases (e.g., *Saprolegnia*).

It appears that adverse interactions can be reduced if hatchery fish are released at dusk or shortly thereafter. Over 90% of the hatchery steelhead from treatment streams were captured in outmigrant traps during hours of darkness (McMichael et al. 1992). This suggests less active emigration in the day time. Releasing fish in the dark should minimize competitive interactions with wild salmonids in the immediate release area because most hatchery fish should have moved downstream by morning, and should be less densely distributed compared to day time releases.

How - fish are released and other management actions affecting densities of residuals

Under certain conditions, other management actions may help minimize adverse effects of hatchery steelhead releases on wild salmonids. If non-migrant hatchery steelhead were not forced out of raceways and ponds and into the streams, perhaps dramatic reductions in the number of residual steelhead in streams would result. The use of acclimation ponds or raceways with volitional exit capabilities can greatly enhance probabilities that released fish will be true smolts that do not residualize (Viola and Schuck 1995). Furthermore, any fish not leaving of their own volition should not be allowed to enter the stream. In most cases, existing artificial propagation vessels may not be equipped with volitional release capabilities. However, they can often be modified to include traps at the downstream ends for collection of actively emigrating smolts. These fish could then be released while other fish are retained until they move down into the trap. This would help sort out non-migrant from emigrant fish.

In situations where hatchery residuals do exist in streams, there may be ways of removing some in ways that do not adversely impact wild fish. Angling regulations could be adopted that encourage the harvest of hatchery steelhead residuals. A key consideration is that such activities should not inadvertently increase risks to wild fish. In our study, where hatchery steelhead were much larger than their wild salmonid counterparts, a minimum size limit of 18 cm (7 inches) might have been an effective way to allow for the harvest of residual hatchery steelhead while protecting against harvest of the smaller wild fish. In areas where size ranges of wild and hatchery fish overlap, and where all hatchery steelhead have clipped adipose fins, regulations could allow the retention of fish with clipped fins. In areas where minimum size limits might adversely impact the larger size classes of wild salmonids, the fin clip regulation would be required. In addition to angling regulations targeting residual hatchery steelhead, Martin et al. (1993) also recommended releasing hatchery steelhead smolts in locations that were easily accessible to anglers. In any case, incidental mortality to wild fish is possible when undersized or unmarked fish are hooked and released (Ferguson and Tufts 1992). This factor should be seriously considered in any circumstance where angling is used to reduce the abundance of residual hatchery steelhead.

As with any general guidelines, decisions must be made at each step regarding the importance of each factor in the final management decisions. Furthermore, implementation of guidelines should be coordinated with a monitoring program that is designed to determine whether the guidelines are effective in minimizing impacts. The guidelines presented in this paper were developed for our specific location and species groups, and while they may be useful in other basins with the same or other species, some may not. Although these guidelines do not represent official Washington Department of Fish and Wildlife policy nor do they address other risk factors (e.g., genetic concerns) associated with hatchery steelhead releases, we believe they will be of value to those attempting to reconcile ecological risks in integrating hatchery and wild stock management programs. It is important to note that these guidelines pertain only to ecological interactions resulting directly from hatchery fish released in the freshwater rearing environment. These interactions may result from outmigrating smolts or from released smolt-size fish that fail to emigrate (residuals) and remain in freshwater for a considerable period of time (e.g., days to years).

Conservation of wild fish resources is a fundamental goal shared by fisheries management entities charged with stewardship responsibilities. However, in contrast to other factors (e.g., genetics - Busack and Currens 1995; Waples and Do 1995), the risk of adverse ecological interactions arising from the use of hatchery fish has received little attention. In situations where conservation of wild salmonid stocks is a high priority, explicit decisions must be made and steps taken to reduce the deleterious ecological effects of hatchery fish in the context of the many other threats to wild stocks. For the specific conditions of our study, the guidelines offered here should help reduce the impact of hatchery programs on wild salmonid populations, but by themselves, will not guarantee the viability of the wild salmonids. Squarely addressing various types of costs and benefits in the context of often-times-contradictory management goals and objectives is necessary in order to make responsible management decisions. In many cases, such decisions will likely translate into reductions in the number of returning hatchery adults to the target area, or harvest opportunities, but will increase protection for wild stocks and the chances of their sustainability.

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Chapter 14

A Practical Approach for Assessing Ecological Risks Associated with Stocking Anadromous Salmonids

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Abstract

As wild fish populations continue to decline, there are increased concerns about how hatchery operations might be contributing to declines of highly valued wild populations. A practical approach for assessing ecological risks to non-target taxa of concern (NTTOC) associated with stocking anadromous salmonids is described to facilitate decision making. This risk assessment approach requires the completion of five tasks: 1) determine non-target taxa objectives in areas of overlap (e.g., acceptable impact of 10% to a species distribution, abundance, and size structure), 2) determine potential spatial-temporal overlap of NTTOC life-stages, 3) determine potential strong ecological interactions, 4) determine ecological risk, and 5) determine scientific uncertainty. These tasks are completed, or a least initiated, by gathering the input of scientists, managers, and policy makers. A decision analysis is also described which recommends specific actions that are triggered by permutations of ecological risk and scientific uncertainty. These actions include different levels of uncertainty resolution, risk minimization strategies, risk containment monitoring, and stocking proposal implementation.

Introduction

The stocking of anadromous salmonids has come under increased scrutiny during the past decade largely because of the ecological and genetic impacts that stocked salmonids are believed to have on wild fish (Marnell 1986; Nickelson et al. 1986; Krueger and May 1991; Waples 1991; Nehlsen et al. 1991; Flagg et al. 1995; White et al. 1995; McMichael et al. 1997). Genetic risk to wild populations has received considerable attention (Waples and Do 1994; Busack and Currens 1995; Currens and Busack 1995) often resulting in genetic guidelines for state and federal hatchery operations. However, ecological risks have received less attention despite the potential for effecting a greater number of taxa in the ecosystem. For example, genetic risks associated with interbreeding are primarily restricted to a single stock or species. In contrast, ecological risks may extend to many classes of plants and animals such as fishes, birds, mammals, amphibians, trees, and insects.

Anadromous salmonids are known to play key ecological roles in freshwater environments where they exist (Willson and Halupka 1995; Bilby et al. 1996; Fresh 1997). Ecological interactions between hatchery and wild fish may be intensified because hatchery fish are typically larger than conspecifics (the target taxon), are released in huge numbers, and are behaviorally different than wild fish (Nielsen 1994; White et al. 1995; McMichael et al. In Press). Taxa that are stocked are termed target taxa and those that are not, non-target taxa. Stocking anadromous salmonids may impact non-target taxa of concern (NTTOC) through a variety of ecological mechanisms including competition, predation, behavioral anomalies, and pathogenic interactions. These interactions can occur at two distinct times. Interactions can occur as a result of direct releases of hatchery fish (Type 1 interaction) and through an increase in naturally-produced offspring of hatchery fish (Type 2 interaction). In addition, NTTOC may be impacted if nutrients are mined from the ecosystem during broodstock collection.

Despite the large potential for stocked salmonids to impact other species of concern, we are aware of no practical approaches for assessing ecological risks associated with stocking anadromous salmonids. Most assessments of hatcheries have focused on the benefit to target taxa, irrespective of impacts to NTTOC. With increasing numbers of fish species and stocks nearing extinction (Williams 1989; Nehlsen et al. 1991; Warren and Burr 1994), ecological risk assessments are desperately needed to make sound resource decisions. We develop a practical approach to assess ecological risks associated with stocking anadromous salmonids in the Columbia Basin to provide decision makers with relevant ecological information. Furthermore, we provide a decision analysis that can be used to select management and research actions contingent upon ecological risk tolerance levels.

Methods

The risk assessment approach used in this work requires completion of five tasks (Table

1). These tasks are best accomplished by a gathering of scientists, managers, and policy makers who systematically work through a risk template worksheet (Table 2). The idea is to bring together people that have local knowledge about the ecosystem of interest, ecological interactions specialists, and those that have management responsibility. The input of policy makers is needed for the first task, but scientists and managers should complete the final four tasks in the absence of policy makers. In order to complete the risk assessment, some assumptions may need to be made. The assumptions and sources of information used in the risk assessment should be documented so that conclusions can be supported and reviewed. Methods for accomplishing each of the five tasks are described below. For the purpose of this paper, taxa that are stocked are termed target taxa and those that are not, non-target taxa.

Table 1. Tasks required to perform an ecological risk assessment of an anadromous salmonid stocking program.

Tasks
I. Determine non-target taxa objectives
A. Identify non-target taxa of concern (NTTOC)
B. Determine status of NTTOC
C. Determine acceptable impact level (e.g., 10% impact in abundance and distribution)
II. Determine or hypothesize spatial-temporal overlap of target taxa with NTTOC life-stages
A. Determine Type 1 overlap of target taxa and NTTOC by life stage
B. Determine Type 2 overlap of target taxa and NTTOC by life stage
III. Determine or hypothesize strong ecological interactions
A. Determine the potential kinds of Type 1 and 2 ecological interactions that might occur
B. Circle the kinds of interactions that are hypothesized to be strong
IV. Determine ecological risk
A. Assess ecological risk for each NTTOC (the risk of failing to meet an objective for a NTTOC) by summing the positive and negative interactions that might occur
V. Determine scientific uncertainty
A. Determine the level of scientific uncertainty of risk assessment

Table 2. Template for conducting ecological risk assessments to NTTOC relative to hatchery stocking or supplementation programs. An example of one NTTOC is provided relative to a hypothetical stocking program.

Proposed Stocking Program: Target taxon______________________ Size at release___________________

Date of release________ Number and Location____

		Overlap[a]		Interaction Strength[b]			
NTTOC[c]	Status[d]/ Impact	Type 1 Life-stage	Type 2 Life-stage	Type 1 Interaction	Type 2 Interaction	Risk[e]	Uncertainty[f]
Example - species or stock	D/10%	Fry	All	C,P,B,D N	C,P,B,D N,S	LOW	LOW
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[a] Type 1 - spatial and temporal overlap between released hatchery salmonids, residuals, and returning adults, and NTTOC
Type 2 - spatial and temporal overlap between all life history stages of naturally produced offspring of returning hatchery adults and NTTOC
Life-stage - fry, parr, smolt, adult (salmonids); age 0, juvenile, adults (other species) or all if overlap occurs for all life-stages or none if no overlap occurs.

[b] Ecological interactions that could occur between stocked anadromous salmonids and NTTOC

<u>Negative interactions</u>
competition (C)- the presence of hatchery salmonids limiting the availability of resources that NTTOC would use in the absence of hatchery salmonids. This occurs when stocked salmonids and NTTOC utilize common resources, the supply of which is short (i.e. exploitative or indirect competition); or if the resources are not in short supply, competition occurs when hatchery salmonids limit access of NTTOC that are seeking a desired resource (i.e., interference or direct competition; Birch 1957).
predation (P)- the direct consumption of NTTOC by hatchery salmonids (direct predation; **Pd**) or the increase in predation by other predator species resulting from the presence of hatchery salmonids (indirect predation; **Pi**). Indirect predation can occur through the following mechanisms 1) hatchery salmonids displace NTTOC from preferred habitat - making NTTOC more vulnerable to predators; 2) the increased abundance of hatchery salmonids attracts predators or increases population densities of predators which can increase consumption of NTTOC, particularly if NTTOC are preferred.
behavioral anomalies (B) - the presence and behavior of hatchery salmonids alters the natural behavior of NTTOC. For example, migrating hatchery salmonids may cause premature migration of NTTOC (e.g., pied-piper effect; Hillman and Mullan 1989) or may cause NTTOC to become less active.
pathogenic interactions (D) - the transfer of a pathogen from hatchery salmonids to NTTOC (direct pathogenic interaction) or the increased susceptibility of NTTOC to pathogens (indirect pathogenic interaction).
nutrient mining (M) - the carcasses of fish that would normally reproduce naturally are collected for hatchery broodstock and are not distributed back into the natural environment or are distributed inappropriately. This results in a loss of nutrients/food that would ordinarily be available to NTTOC.

<u>Beneficial interactions</u>
Nutrient enrichment (N) - increase in nutrients available to NTTOC because of an increase in marine derived nutrients from increased salmonid returns (e.g., salmon carcasses).
Prey (F) - increased availability of prey for piscivorous NTTOC.
Predator swamping (S) - the survival of NTTOC is enhanced due to swamping of predators by hatchery fish.
[c] NTTOC - non-target taxa of concern - highly valued non-target taxa

[d]Status - H=healthy, D=depressed, C=critical (or other status descriptors); Impact - acceptable impact level to the NTTOC (e.g., 10% impact to abundance, distribution, and size structure)

[e]Risk - probability of failing to meet an objective for NTTOC

[f]Uncertainty - scientific uncertainty of risk assessment due to lack of information or variability of ecological interaction outcomes

Task 1 is to identify (NTTOC) and describe acceptable impact levels which establishes a management intent. These “non-target taxa objectives” serve as the benchmark against which ecological risk will be assessed. These objectives must be the starting point for the assessment or else consensus of a risk assessment will be difficult because people will be assessing risk against different goals (Lackey 1994). The selection of non-target taxa objectives is largely a decision based on values, so policy makers should work with scientists to determine them. For instance, policy makers can determine an acceptable impact level based on how a scientist hypothesizes that impact will harm the NTTOC. A method for selecting NTTOC is described in Chapter 1 of this report. Briefly, these actions include identifying non-target taxa in areas of presumed overlap with target taxa, identifying those species that are highly valued (e.g., economically, socially, religiously, recreationally, etc.), identifying the status of the NTTOC, and identifying acceptable impact levels for each taxa. Acceptable impact levels might be influenced by the status and the relative value of a taxa. For example, an acceptable impact level for the status of an endangered species may be zero but, for a healthy taxa of moderate value it may be 40%.

Task 2 is to determine which life-stages of NTTOC will overlap with the target taxon. Two types of overlap may occur. Type 1 overlap occurs when an NTTOC life-stage overlaps with that of the hatchery fish (e.g., residual, smolt, or adult). Type 2 overlap occurs when an NTTOC life-stage overlaps with the naturally-produced offspring of hatchery fish. Knowledge of the movement patterns of the target taxon and the distribution of NTTOC is essential in predicting the hypothesized overlap. If this information is unavailable, then a conservative approach is to assume that overlap will be complete unless barriers restrict movement of hatchery fish.

Task 3 is to postulate which ecological interactions may occur between NTTOC life-stages that overlap with the target taxon and which interactions are strongest. All possible types of ecological interactions should be listed and the ones that are likely to be strong should be highlighted on the worksheet (Table 2). Potential interactions can be determined by examining the available literature and by examining basin specific information relating to interactions. Prediction of interactions between hatchery and wild fish should not rely only on historical information or on studies of wild fish in pristine environments. The habitat of most river systems has changed substantially which can change the outcomes of ecological interactions (Reeves et al. 1987, Ward et al. 1995, Fresh 1997) and can reduce the potential for fish to partition resources or avoid predators. In addition, hatchery fish are often larger, behave differently, and are present in larger numbers than wild fish. Similar to task 2, interactions resulting from hatchery practices can occur as the result of hatchery fish (type 1 interaction) or naturally-produced offspring of hatchery fish (type 2 interaction). Deleterious and beneficial interactions with NTTOC may occur as a result of hatchery practices. Deleterious interactions include: competition for food or space, predation on NTTOC by target taxa, decreased survival of NTTOC due to predator attraction or increased susceptibility to predation, pathogen transfer to NTTOC, increased susceptibility to pathogens due to high stress, behavioral alterations such as pied-piper, and reduction in nutrients available to NTTOC through broodstock mining. Beneficial interactions include an increase in prey available for piscivorous NTTOC, an increase in nutrients available through increased spawning escapement, and increased survival of NTTOC due to swamping or satiating predators.

After tasks 1, 2, and 3 are completed, the ecological risk of stocking the target taxon can be determined. Ecological risk is here defined as the risk of failing to meet non-target taxa objectives. To determine the risk, the ecological interactions, particularly the strong ones, should be qualitatively summed. If the positive interactions outweigh the negative ones, then the risk would be low. Conversely, if negative interactions outweigh positive ones, then the risk would be high.

Finally, the scientific uncertainty of the risk assessment should be determined. If there is a lot of uncertainty about what kind and how strong ecological interactions are, then the uncertainty would be high. For example, it is generally uncertain whether releases of hatchery fish will increase or decrease predation on wild smolts. If this is potentially a strong interaction then there would be much uncertainty about the risk assessment.

Discussion

After an ecological risk assessment has been completed, decisions must be made whether or not to proceed with a stocking proposal or to proceed pending certain strategic risk minimization strategies, uncertainty resolution research, and risk containment monitoring. These decisions would preferably be made by managers who represent the agencies' values as opposed to scientists. We describe a method that could be used by managers to decide what actions are appropriate given certain levels of ecological risk and scientific uncertainty. We suggest that permutations of ecological risk and scientific uncertainty can result in different levels of "ecological comfort" which can be used to influence decision making (Table 3). Ecological comfort is defined as the degree of ease or tolerance that is felt with a fish stocking proposal or hatchery operation. It is assumed that ecological comfort decreases with increasing risk and uncertainty. In other words, we are uneasy about those things that are suspected to be detrimental and those things that are unknown. Levels of ecological comfort can then be used to trigger certain kinds of management, monitoring, or research actions (Table 4). Some examples of these actions are described below.

Table 3. Ecological comfort levels associated with different levels of ecological risks and scientific uncertainty.

	Low Risk	High Risk
Low Uncertainty	High	Low
High Uncertainty	Medium	Low

Table 4. Action items based on ecological comfort levels. All actions assume that the stocking provides a cost efficient method of providing production benefits and that genetic risks have been deemed acceptable.

Ecological Comfort	Uncertainty Resolution	Risk Minimization Strategies	Risk Containment Monitoring	Stocking Proposal
High	None	None	Low effort	Proceed
Medium	Some	Some	Moderate effort	Proceed with caution
Low	Little or Much	Extensive	High effort	Do not stock unless adequate risk containment measures are in place or uncertainties are resolved

Uncertainty resolution can be accomplished by literature reviews or by performing new studies in the basin of interest. The highest priority for uncertainty resolution (e.g., high risk and high uncertainty) could be determined from the risk worksheet. For example, we were uncertain about how spring chinook salmon supplementation in upper Yakima basin tributaries would impact wild rainbow and steelhead trout. There was relatively little published information on interactions between these taxa so we conducted a variety of field studies and experiments that addressed critical ecological interaction uncertainties such as competition (McMichael et al. 1997, McMichael and Pearsons In press). These studies significantly reduced the scientific uncertainty of interaction impacts and increased the ecological comfort of proceeding with spring chinook salmon supplementation in the upper Yakima basin.

Ecological risk can also be reduced by employing strategies that minimize negative and maximize positive interactions between hatchery fish and NTTOC. Risk minimization strategies fall under six categories: what is released, where, when, and how fish are released, broodstock distribution management, harvest management, and habitat enhancement. Brief examples for each of these categories is described below, but fuller treatments can be found in WDFW (1995) and McMichael et al. (In press). To reduce ecological risk, fish should be stocked that are active migrators, appropriately sized (e.g., small - so that predation potential and competitive dominance are low), appropriately behaved (e.g., not too aggressive), in minimum numbers, and disease free. Furthermore, they should be released at times when NTTOC are relatively inactive, absent, or invulnerable to impacts (e.g., large enough to avoid predation) and in places where NTTOC are absent and have adequate interactions refugia. Finally, management actions that benefit NTTOC, such as distributing carcasses used for broodstock into natural spawning areas, restrictive harvest regulations, and habitat enhancements can help to reduce or offset ecological risk.

Finally, risk containment monitoring such as monitoring NTTOC status for evidence of impacts, can provide information that will allow for changes in hatchery practices to minimize or eliminate undesirable impacts. Risk containment monitoring is particularly important when ecological risks are high because it is highly probable that objectives for non-target taxa will not be met (Table 4). The focus of risk containment monitoring is most appropriate if it is targeted at NTTOC status (e.g., distribution, abundance, size structure), however in some cases, monitoring a critical ecological interaction may be more feasible. For example, an objective for rainbow trout in the upper Yakima River relative to spring chinook salmon supplementation is to allow a maximum impact of a 10% decrease of the baseline distribution, abundance, and size structure (Chapter 1). Failure to achieve this objective, with statistical reliability, will be detectable in one or two years (Chapter 3). Similar to monitoring rainbow trout in the upper Yakima River, 40% impacts to rainbow trout in tributaries to the upper Yakima River can be detected in one year (Chapter 4). Finally, the impacts of monitoring methods, such as electrofishing and trapping, should also be considered when evaluating ecological risks to NTTOC (Chapter 5, 7).

We acknowledge that ecological interactions are only one of a number of different potential benefits or detriments related to stocking hatchery fish. Genetic and harvest risk assessments should be performed in conjunction with ecological risk assessments to provide a more complete perspective of risk associated with hatchery operations and stocking. These risks should also be balanced with the benefits afforded by hatchery production such as stock restoration or harvest enhancement.

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