December 1999

YAKIMA RIVER SPECIES INTERACTIONS STUDIES

Annual Report 1998

DOE/BP-64878-5

This report was funded by the Bonneville Power Administration (BPA), U.S. Department of Energy, as part of BPA's program to protect, mitigate, and enhance fish and wildlife affected by the development and operation of hydroelectric facilities on the Columbia River and its tributaries. The views of this report are the author's and do not necessarily represent the views of BPA.

This document should be cited as follows:

Pearsons, Todd N., Kenneth D. Ham, Geoffrey A. McMichael, Eric L. Bartrand, Anthony L. Fritts, Charles W. Hopley - Washington Department of Fish and Wildlife, Contributor: (BPA Report DOE/BP-64878-5)

This report and other BPA Fish and Wildlife Publications are available on the Internet at:

http://www.efw.bpa.gov/cgi-bin/efw/FW/publications.cgi

For other information on electronic documents or other printed media, contact or write to:

Bonneville Power Administration
Environment, Fish and Wildlife Division
P.O. Box 3621
905 N.E. 11th Avenue
Portland, OR 97208-3621

Please include title, author, and DOE/BP number in the request.

Yakima River Species Interactions Studies

Annual Report 1998

Prepared by:

Todd N. Pearsons
Kenneth D. Ham
Geoffrey A. McMichael
Eric L. Bartrand
Anthony L. Fritts
Charles W. Hopley

Contributor:

Jim Dunnigan (Yakama Indian Nation)

Washington Department of Fish and Wildlife
600 Capitol Way North
Olympia, Washington 98501-1091

Prepared for:

U.S. Department of Energy
Bonneville Power Administration
Division of Fish and Wildlife
P.O. Box 3621
Portland, Oregon 97283-3621

Project Number 95-064-02

December 1999

Executive Summary

Species interactions research and monitoring was initiated in 1989 to investigate ecological interactions among fish in response to proposed supplementation of salmon and steelhead in the upper Yakima River basin. This is the seventh 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, and develop methods to monitor interactions and supplementation success. Major topics of this report are associated with monitoring potential impacts to support adaptive management of NTT and baseline monitoring of fish predation indices on spring chinook salmon smolts. This report is organized into three 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, 1998 and December 31, 1998 in the Yakima basin, however these data were compared to data from previous years to identify preliminary trends and patterns. Summaries of each of the chapters included in this report are described below.

● We examined variability in abundance of 16 native fish taxa to determine if rapid, sensitive detection of change is possible for native fish populations in the Yakima River basin. Prospective power to detect impacts was estimated from 2 to 16 annual baseline surveys conducted by electrofishing, trapping, or snorkeling. Detectable impacts decreased with greater quantity and quality of baseline surveys, but high temporal variability prevented detection of small impacts for most taxa. For 10 taxa, models of environmental and biological influences accounted for between 19 and 61% of temporal variation, increasing our ability to detect impacts of other influences. Detectable impacts were computed for both biological and environmental detection time limits and compared among taxa. Detectable impacts for a t-test with $\alpha = 0.1$ and $\beta = 0.1$ were greater than 19% for all 16 taxa and > 50% for 7 of 16 taxa using both biological and environmental detection time limits.

● We examined the potential to adaptively manage ecological impacts to wild fishes using data from the Yakima River basin. Sensitivity and speed of impact detection through status or interactions monitoring of NTT were evaluated. We defined status of an NTT population as its distribution, abundance, and size structure. Interactions monitored include predation and spatial overlap with target species. Monitoring options, alone or in combination, often failed to achieve adequate power to detect impacts equal to the containment objective (CO) for some or all interaction types. Impact detection and containment at or below the CO was only rarely possible for rare or valuable taxa (CO = 0 - 10% reduction relative to baseline status). Inadequate feedback will prevent the adaptive management approach from assuring that ecological impacts to NTT that exceed the CO are quickly corrected. However, some NTT could be monitored well enough to facilitate risk containment management and monitoring will provide the potential of some risk containment for all of the NTT.

● We calculated predation indices (PI) during 1998 for the three primary fish predators in the lower Yakima River; smallmouth bass, northern pikeminnow, and channel catfish. Bass and

pikeminnow were captured primarily by electrofishing. Channel catfish were collected in drifting gill nets, hoop nets, traps, and by electrofishing and angling. Stomach samples were collected during the spring when emigration of spring chinook salmon smolts was estimated to be at its peak (mid-late April), and again during the last quartile (mid-May) of their emigration. Most of the smallmouth bass predation on salmonids was on fall chinook salmon parr and smolts. The smallmouth bass PI for spring chinook salmon was seven to 63 times lower than the PI for fall chinook salmon. 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. We were unable to generate population estimates of northern pikeminnow due to low capture efficiency. Northern pikeminnow rarely consumed salmonids during the April sampling period, but during May, 21-29% of the northern pikeminnow stomachs contained at least one salmonid. During this period northern pikeminnow consumed both yearling and subyearling salmonids. We captured large numbers of channel catfish, and 2.9% of the stomachs examined contained at least one salmonid. One channel catfish contained 76 fall chinook salmon, and several other fish species in its gut. By extrapolating smallmouth bass numbers from the mouth of the Yakima River upstream to Prosser Dam, we estimated that smallmouth bass could consume about 18,840 salmonid smolts in the lower 68 km of the Yakima River daily during the smolt emigration period. Estimates of the number of salmonids consumed by northern pikeminnow above Prosser ranged from 35-390 salmonids/1000 predators/day throughout the emigration period. Predator control options are discussed, with the most promising being a 2 C decrease in water temperature in the lower Yakima River.

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).

Table of Contents

List of Figures

Figure Page

Chapter 1

Chapter 2

Chapter 3

List of Tables

Table Page

Chapter 1

Chapter 2

Chapter 3

General Introduction

This report is intended to satisfy two concurrent needs: 1) provide a contract deliverable from the Washington Department of Fish and Wildlife (WDFW) to the Bonneville Power Administration (BPA), with emphasis on identification of salient results of value to ongoing Yakima/Klickitat Fisheries Project (YKFP) planning, and 2) summarize results of research that have broader scientific relevance. This is the seventh 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; Pearsons et al. 1998). This progress report summarizes data collected between January 1, 1998 and December 31, 1998. These data were compared to findings from previous years to identify general trends and make preliminary comparisons. Interactions between fish produced as part of the YKFP, termed target species or stocks, and other species or stocks (non-target taxa) may alter the population status of non-target species or stocks. This may occur through a variety of mechanisms, such as competition, predation, and interbreeding (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 YKFP has evolved. Initially, our work focused on interactions between anadromous steelhead and resident rainbow trout (for explanation see Pearsons et al. 1993), then interactions between spring chinook salmon and rainbow trout, and recently interactions between spring chinook salmon and highly valued non-target taxa (NTT; e.g., bull trout); and interactions between strong interactor taxa (e.g., those that may strongly influence the abundance of spring chinook salmon; e.g., smallmouth bass) and spring chinook salmon. The change in emphasis to spring chinook salmon has largely been influenced by the shift in the target species planned for supplementation (Bonneville Power Administration et al. 1996; Fast and Craig 1997). Originally, steelhead and spring chinook salmon were proposed to be supplemented simultaneously (Clune and Dauble 1991). However, due in part to the uncertainties associated with interactions between steelhead and rainbow trout, spring chinook 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. Pre-facility monitoring of variables such as rainbow trout density, distribution, and size structure was continued and monitoring of other NTT was initiated in 1997.

This report is organized into three chapters which represent major topics associated with monitoring stewardship, utilization, and strong interactor taxa. Chapter 1 reports the baseline abundances of stewardship and utilization species and the statistical power that is possible to detect potential impacts from supplementation. Chapter 2 explores the potential of adaptive management to contain impacts to NTT and describes a decision framework for developing and implementing risk containment monitoring plans. Finally, chapter 3 describes the development and refinement of the fish predation index. 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. All of the chapters in this report are submitted or published in a peer reviewed journal, or already published in a conference proceeding. Chapter 1 is in published in an altered version in the "Canadian Journal of Fisheries and Aquatic Sciences" (Ham and Pearsons, 2000). Chapter 2 has been submitted in a different form to the same journal (Ham and Pearsons, submitted). We have chosen to include the original versions of chapters 1 and 2 that were submitted to the journal because they represent the full complement of species that we are monitoring and topics of importance to the YKFP. The version of chapter 1 that is published represents only salmonids and the submitted version of chapter 2 does not include the monitoring framework. Chapter 3 has been published in the proceedings of the workshop "Management implications of Co-occurring Native and Introduced Fishes" (McMichael et al. 1999). Additional field work and/or analysis is in progress for topics covered in this report. Throughout this report, a premium was placed on presenting data in tables so that other interested parties could have access to the data. Readers are cautioned that any preliminary conclusions are subject to future revision as more data and analytical results become available.

Except where otherwise noted, the methods and general site descriptions are the same as described in previous reports (Hindman et al. 1991; McMichael et al. 1992; Pearsons et al. 1993; Pearsons et al. 1994; Pearsons et al. 1996; Pearsons et al. 1998).

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 1998. Most of these individuals are recognized on title pages or in acknowledgments of various chapters in this report.

References

Bonneville Power Administration, Washington Department of Fish and Wildlife and Yakama Indian Nation. 1996. Yakima Fisheries Project - Final Environmental Impact Statement. Bonneville Power Administration, Portland, Oregon. DOE/BP-2784.

Busack, C., B. Watson, T. Pearsons, C. Knudsen, S. Phelps, and M. Johnston. 1997. Yakima Fisheries Project Spring Chinook Supplementation Monitoring Plan.

Clune, T., and D. Dauble. 1991. The Yakima/Klickitat fisheries project: a strategy for supplementation of anadromous salmonids. Fisheries 16(5):28-34.

Fast, D. E., and C. Craig. 1997. Innovative hatchery project working to rebuild wild salmon populations. Hydro Review XVI:30-33.

Ham, K. D., and T. N. Pearsons. 2000. Can reduced salmonid population abundance be detectedin time to limit management impacts? Canadian Journal of Fisheries and Aquatic Sciences 57:1-8.

Ham, K. D., and T. N. Pearsons. Submitted. What level of ecological impact can be detected in salmonid populations? Canadian Journal of Fisheries and Aquatic Sciences.

Hindman, J. N., G. A. McMichael, J. P. Olson, and S. A. Leider. 1991. Yakima River Species Interactions Studies. Annual Report FY 1990 submitted to Bonneville Power Administration, Portland, Oregon. DOE/BP-01483-1. 75 pp.

McMichael, G. A., J. P. Olson, E. L. Bartrand, M. Fischer, J. N. Hindman, and S. A. Leider. 1992. Yakima River Species Interactions Studies. Annual Report FY 1991 submitted to Bonneville Power Administration, Portland, Oregon. DOE/BP-01483-2. 177 pp.

McMichael, G. A., A. L. Fritts, T. N. Pearsons, and J. L. Dunnigan. 1999. Lower Yakima River predatory fish monitoring: progress report 1998. Pages 93-124 *In* ODFW and NMFS. Management implications of co-occurring native and introduced fishes: Proceedings of the workshop. Portland, Oregon.

Pearsons, T. N., G. A. McMichael, E. L. Bartrand, M. Fischer, J. T. Monahan, and S. A. Leider. 1993. Yakima Species Interactions Study. Annual Report FY 1992 submitted to Bonneville Power Administration, Portland, Oregon. DOE/BP-01483-3. 98 pp.

Pearsons, T. N., G. A. McMichael, S. W. Martin, E. L. Bartrand, M. Fischer, and S. A. Leider. 1994. Yakima River Species Interactions Studies. Annual Report FY 1993 submitted to Bonneville Power Administration, Portland, Oregon. DOE/BP-99852-2. 247 pp.

Pearsons, T. N., G. A. McMichael, S. W. Martin, E. L. Bartrand, J. A. Long, and S. A. Leider. 1996. Yakima River Species Interactions Studies. Annual Report FY 1994 submitted to Bonneville Power Administration, Portland, Oregon. DOE/BP-99852-3. 150 pp.

Pearsons, T. N., G. A. McMichael, K. D. Ham, E. L. Bartrand, A. L. Fritts, and C. W. Hopley. 1998. Yakima River Species Interactions Studies. Progress Report FY 1995-1997 submitted to Bonneville Power Administration, Portland, Oregon.

Chapter 1

Can changes in native fish abundance be detected in time to avert unacceptable impacts of fish stocking?

Kenneth D. Ham

and

Todd N. Pearsons

Washington Department of Fish and Wildlife
600 Capitol Way North
Olympia, Washington 98501

Abstract

We have examined variability in abundance of 16 native fish taxa to determine if rapid, sensitive detection of change is possible for native fish populations in the Yakima River basin, Washington, where a large-scale test of hatchery supplementation is being conducted. Prospective power to detect impacts was estimated from 2 to 16 annual baseline surveys conducted by electrofishing, trapping, or snorkeling. Detectable impacts decreased with greater quantity and quality of baseline surveys, but high temporal variability prevented detection of small impacts for most taxa. For 10 taxa, models of environmental and biological influences accounted for between 19 and 61% of temporal variation, increasing our ability to detect impacts of other influences. Detectable impacts were computed for both biological and environmental detection time limits and compared among taxa. Detectable impacts for a t-test with $\alpha = 0.1$ and $\beta = 0.1$ were greater than 19% for all 16 taxa and > 50% for 7 of 16 taxa using both biological and environmental detection time limits. Even thorough preparation does not assure that detection will be sensitive enough to provide feedback sufficient to avoid unacceptable impacts, especially for populations with low acceptable impacts.

Introduction

Stocking of native and exotic fishes has been used for over a hundred years to enhance fisheries or restore fish populations (Wahle and Pearson 1984; Stroud 1986; Schramm and Piper 1995), but the potential for ecological impacts to other fish populations is often neglected. This neglect has resulted in undesirable consequences such as species endangerment and extinction (Nehlsen et al. 1991; Lassuy 1995). Recently, however, stocking of hatchery reared anadromous salmonids to supplement wild populations has been identified as a major strategy to restore, maintain, and conserve imperiled anadromous salmonids in the Pacific Northwest (National Research Council 1996; Independent Scientific Group 1996, Fast and Craig 1997). Supplementation, as defined by Regional Assessment of Supplementation Planning (1992), incorporates both the enhancing of the target species and limiting impacts to non-target taxa. Supplementation has been defined as "the use of artificial propagation in an attempt to maintain 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" (Regional Assessment of Supplementation Planning 1992; Bonneville Power Administration 1996). This strategy is untested and is not without controversy (Independent Scientific Group 1996; Busack et al.1997). Part of the controversy reflects the concern of how cultured fish (target taxa) may impact wild fishes that are not being supplemented (non-target taxa) (White et al. 1995; McMichael et al. 1997). Ecological risk assessments performed prior to fish stocking help determine if ecological risks are of concern and to identify strategies that could be used to reduce risk (Pearsons and Hopley 1998; McMichael et al. 1999). However, in many cases there will still be considerable scientific uncertainty regarding ecological risks. Supplementation has frequently been coupled with adaptive management because of the high scientific uncertainty of outcomes (Regional Assessment of Supplementation Planning 1992; Bonneville Power Administration 1996; Independent Scientific Group 1996).

Adaptive management is founded upon the ability to design management actions as experiments and learn from mistakes so that corrections can be made (Walters and Hilborn 1976; Walters 1986). As experiments, adaptive management actions must employ clear experimental design and strong statistical inference to resolve uncertainty and guide future management actions. Rapid, sensitive impact detection is a crucial element of adaptive management because corrective actions, if needed, must be implemented before impacts become harmful to the population of interest or before they exceed an acceptable level. If impacts are not detected, the sensitivity of impact detection determines how certain we are that an impact did not occur.

As with other ecological experiments, monitoring to detect impacts to native fish abundance faces difficulty in establishing sufficient replication and controls (Hurlbert 1984), sampling the component of interest (Eberhardt and Thomas 1991), and overcoming high levels of variation unrelated to the imposed treatment. Frequently, environmental monitoring plans are designed with insufficient statistical power (Peterman and Bradford 1987; Peterman 1990). If insufficient power is not recognized, it can lead to a failure to reject a false null hypothesis (i.e., an impact goes undetected), a Type II statistical error (Peterman 1990). A Type II error results in underestimation of impacts. Underestimation of impacts undermines adaptive management, because uncertainties resolved in error could mask the need for corrective action. Controlling or recognizing the probability of a Type II error is essential to evaluating the potential risk of a management alternative, especially where conservation of a species is involved (National Research Council 1995). Therefore, the need exists to evaluate whether impact detection is rapid and sensitive enough to allow impacts to be detected and corrective actions to be implemented.

Impact detection is affected by temporal variation in population status in the absence of the impact of interest, identification of sources of variation that explain non-impact variation, and baseline data quantity and quality. These factors are most crucial when no experimental controls are available, as is frequently the case when working in large river systems, and impacts must be determined primarily from data collected before and after a management action (Green 1979, Eberhardt and Thomas 1991; Stewart-Oaten et al. 1992). This is the case in the Yakima River basin where supplementation of spring chinook salmon *Oncorhynchus tshawytscha* has been in the planning stages for over a decade (Clune and Dauble 1991; Fast and Craig 1997).

Baseline monitoring surveys of native fish abundance conducted to prepare for spring chinook salmon supplementation in the Yakima basin provide ample material to evaluate the limits of population abundance monitoring in detecting impacts. First, a tremendous amount of data on a variety of species has been collected (Fast et al. 1991; Bonneville Power Administration 1996; Busack et al. 1997; Pearsons et al. 1998). Second, the Yakima Fisheries Project (YFP) was designed to produce information and methodologies that could be used in other areas of the Columbia basin (Busack et al. 1997). Third, the baseline sampling period has ended, and detectable impacts can now be estimated. In this paper, we evaluate how biological properties of native fish taxa combine with baseline data quantity and quality to determine whether population status estimates will provide adequate impact detection to support adaptively managing the potential impacts of fish-stocking programs. The fish-stocking project in the Yakima River will be used as a case study to illustrate a realistic range of detectable impacts for ecological impacts to wild fish populations.

Methods

Study area and background

The Yakima River is the longest river located entirely within Washington and flows in a north to south direction before entering the Columbia River near the city of Richland (Figure 1). The Yakima River drainage basin has an area of 15,900 km^2 and ranges from over 2440 m in elevation at the headwaters to about 104 m at the river mouth. Forty-eight fish species have been identified in the Yakima basin, of which only 29 are of native origin (Pearsons 1998). Estimates of historical, anadromous salmon and steelhead runs are between 600,000 and 960,000 adults per year, but currently average only about 7,000 adults per year (Fast and Craig 1997). Pre-1900 runs of spring chinook salmon have been estimated to be about 200,000 adults returning per year (Bonneville Power Administration 1996), while the most recent 5-year average was about 2,300 adults returning per year (Yakama Indian Nation (YIN) unpublished data). Similar to other parts of the Columbia basin, degradation of the biotic resources in the Yakima basin have been attributed to a variety of factors including irrigation, dams, overfishing, channelization, introduction of exotic fishes, mining, logging practices, overgrazing, beaver trapping, pollution, and urbanization (Li et al. 1987; Leland 1995; Bonneville Power Administration 1996; Cuffney et al. 1997).

Hatchery supplementation was proposed as a means to increase natural production of spring chinook salmon in the upper Yakima basin (Clune and Dauble 1991; Bonneville Power Administration 1996). Operation of a supplementation hatchery near the town of Cle Elum has begun, and the first smolt release is scheduled for spring of 1999. The first adults to return from hatchery releases in 1999, will be in 2000 (jacks), but the vast majority from that release will return in 2001 (Bonneville Power Administration 1996). Naturally produced progeny from the

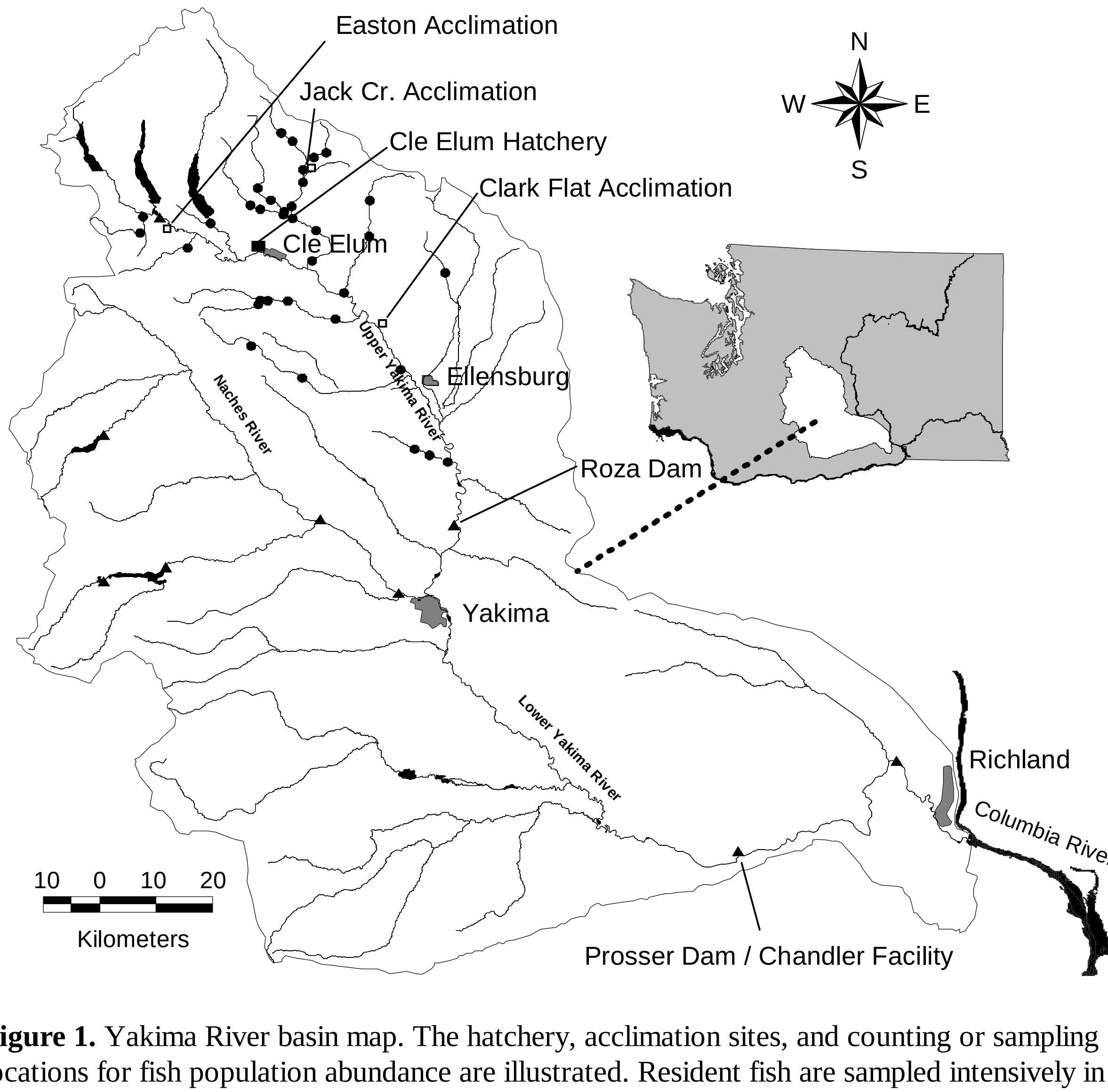

Figure 1. Yakima River basin map. The hatchery, acclimation sites, and counting or sampling locations for fish population abundance are illustrated. Resident fish are sampled intensively in the upper Yakima River basin, including tributaries. Less intensive sampling is conducted in the main stem of the lower Yakima River. Anadromous fish are counted during emigration at Prosser dam/ Chandler juvenile facility, and at Roza dam.

fish that return in 2001 would not emerge and potentially interact with native fish until 2002. At the time of this writing, baseline surveys have been completed for taxa that reside below spring chinook salmon acclimation sites (e.g., steelhead *O. mykiss*). As many as three more annual baseline surveys can be conducted for taxa that reside above acclimation sites (e.g., bull trout *Salvelinus confluentus*) where spring chinook salmon smolt density will not increase until after the first generation of hatchery adults return and spawn in the wild. The greatest potential for some types of impacts may not be realized until after 5 or 6 generations of successful fish stocking when natural production has increased to near the projected maximum but hatchery smolts are still being released. At the extreme, pre-impact baseline surveys may be separated from the time of potentially highest impact by two decades or more.

Non-target fish taxa of concern that have the potential to overlap with supplemented spring chinook salmon were identified. Certain species were selected because of stewardship-related concerns for the long-term survival of the population in the Yakima River basin. Some fish are rare across the region. For instance, bull trout are federally listed as threatened and mid-Columbia steelhead are proposed as threatened under the Endangered Species Act (Fish and Wildlife Service 1998; National Marine Fisheries Service 1998). Westslope cutthroat trout *O. clarki* have been petitioned for listing. In addition, Pacific lamprey *Lampetra tridentata* in the Columbia basin have severely restricted abundance and distribution relative to historic conditions (Close et al. 1995; Thurow et al. 1997). Mountain sucker *Catostomus platyrhynchus*, sand roller *Percopsis transmontana*, leopard dace *Rhinichthys falcatus*, and the Marion Drain stock of fall chinook salmon are rare within the Yakima basin (Patten et al. 1970; Pearsons et al. 1998), though other populations or stocks of these species are more common in other parts of the region. Other species were selected because of concerns about impacts to people that utilize them for recreation, food, science, livelihood, culture, or religion. Some species are very important for fishery utilization. For example, rainbow trout *O. mykiss* in the mainstem Yakima River provide one of the best resident wild trout fisheries in the state of Washington (Krause 1991; Probasco 1994) and are currently managed as a catch and release fishery. Naches and American river stocks of spring chinook salmon are also taken in Yakima-basin subsistence fisheries, and are utilized in recreational and commercial fisheries in the Columbia River and Pacific Ocean (Hunn 1990). Other species receive less attention but are still important for utilization. Species groups such as mountain whitefish *Prosopium williamsoni*, and tributary rainbow trout support recreational fisheries. Other native species that are not currently rare and have little obvious utilization value have a stewardship objective of maintaining sustainable population sizes. Species such as longnose dace *R. cataractae*, speckled dace *R. osculus*, sculpin *Cottus spp.*, and sucker *Catostomus spp.*, are examples of species common throughout the Yakima River basin and the region.

The status of a native fish taxon can be characterized as the combination of distribution, abundance, and size structure. This study concentrates on detection of impacts to abundance, for simplicity. Competition and predation are the ecological interaction types of greatest concern across all species, although nutrient mining, pathogenic interactions, and behavioral interaction may be important in some cases (Pearsons 1998). The challenge is to detect impacts from any of these impacts with minimal effort. Abundance is the least equivocal evidence of an impact and has the potential of responding to the full range of interaction types. These characteristics, in combination with standardization of methods due to common use, make abundance estimates an attractive candidate as a response variable for detecting and reporting impacts of fish stocking on native fish species.

To determine detectable impacts for each of 16 taxa, we needed to complete 3 major tasks. First, we determined the feasibility of surveying taxa abundance and conducted baseline monitoring. Second, we attempted to identify important biological and environmental factors that account for some of the temporal variation in population abundance. Finally, we examined statistical power in terms of what impacts were detectable for increasing numbers of post-impact surveys and how many years were available to detect impacts before possible permanent harm to the population.

Feasibility sampling and baseline surveys

After native fish taxa were identified, preliminary sampling determined which survey methods were effective and helped evaluate the feasibility of quantifying population abundance. Directed feasibility sampling included locations throughout the Yakima River basin in areas where there is a perceived risk of ecological impacts of proposed spring chinook salmon supplementation. Resident taxa were surveyed between 1990 and 1998 in locations where they were historically present (Patten et al. 1970; Wydoski and Whitney 1979) and in other locations where they might be present. A survey of the Yakima River conducted by Patten et al. (1970) in 1957 and 1958 was used to initially estimate the distribution and abundance of species prior to sampling. Our survey methods included boat and backpack electrofishing as well as snorkeling. The YIN has been counting salmon and steelhead at various permanent structures in the Yakima River basin for over 30 years (Major and Mighell 1969; Fast et al. 1991), and additional information is available from redd surveys and harvest statistics. Information from past surveys, initial feasibility surveys and incidental encounters was used to evaluate the feasibility of sampling each taxon, to indicate the limits of distribution, and to help determine the potential to annually survey abundance. When annual surveys were judged feasible, a sampling plan was created to quantify baseline abundance and temporal variability.

Effort was not distributed equally among taxa during baseline investigations, and differences are reflected in data quality or quantity. The level of effort for each taxon was the result of long-term status monitoring of salmon and steelhead, flyfishers concerns about potential impacts of hatchery operations on the rainbow trout fishery, incidental encounters, and refinement of YFP objectives. Unfortunately, separation of anadromous stocks is difficult at the smolt stage (counts often include a mixture of different stocks). Pacific lamprey counts of emigrants may also include western brook lamprey *L. richardsoni*. Impacts to individual stocks or species will be harder to detect when counts are not separated. Until recently, rainbow trout was the only species directly monitored specifically for impact detection relative to the YFP. Bull trout, westslope cutthroat trout, mountain whitefish, and mountain sucker and other native species have been monitored incidental to rainbow trout surveys with varying degrees of success. Finally, feasibility monitoring of leopard dace, sand roller, and mountain sucker was initiated within the past two years, and has yet to lead to collection of a series of baseline estimates.

Modeling

To minimize unexplained variation in fish population abundance estimates, we attempted to account for some of the baseline temporal variation with environmental factors likely to impact survival during critical periods within the life history of a species. Empirical regression models were constructed of the relationships between the estimated abundance of a native fish taxon and environmental factors such as water temperature, flow magnitude, or flow variability. The predictive models were simple linear regression models of the form:

$$\text{Predicted estimate} = \text{Intercept} + B_1(\text{index}_1) + \ldots + B_n(\text{index}_n)$$

where B_1 through B_n are regression coefficients and index_1 through index_n are indices representing quantitative measures of important sources of variation. Models were usually limited to two or fewer parameters and an intercept because most data series included fewer than 10 baseline surveys. No models were constructed if few (< 5) baseline surveys had been conducted. The indices include factors such as water flows and temperatures, which are measured by the U. S. Bureau of Reclamation at many sites throughout the basin. Yearly variation in river flow and temperature was summarized for each of several reporting stations along the Yakima River by computing the first two principal components of the group of monthly mean flows and temperatures. Factor scores of stations located in the range of a taxon were used as possible indices in the model. Other indices included status of a prior life stage such as using redd counts or escapement as one index in a model to predict smolt counts. Models were chosen from the available indices for each taxa by stepwise multiple regression, with models generally limited to two or fewer indices to prevent over-parameterization of the model, given the limited number of annual surveys available to generate the models.

Power analyses and detectable impacts

Detectable impacts were computed for each taxon, given the baseline variation in estimates of abundance and models of other sources of variation. Testing for impacts is to be by a one-tailed t-test of baseline abundance estimates versus abundance estimates during the potential impact period. We assumed variance is proportional to mean abundance and incorporated this assumption into testing by testing for differences in log-transformed abundance. Detectable impacts computed with log-transformed abundances were converted by subtracting the transformed detectable impact from the transformed mean, taking the antilog, and expressing the result as a percentage of baseline mean abundance. Setting $\alpha = 0.1$ and $\beta = 0.1$ (power = 0.9), we computed what impacts would be detectable with increasing numbers of post-impact surveys. The detectable impact can be computed with the following formula from Zar (1996, page 135 equation 8.23):

(1) $$\delta \geq \sqrt{\frac{2s_p^2}{n}}\left(t_{\alpha,v} + t_{\beta(1),v}\right)$$

where: n = number of years to detect an impact; s_p^2 = pooled sample variance; δ = detectable impact; α = significance level; v = degrees of freedom, df = 2(n - 1); β = probability of a Type II error (power to detect an impact equal to δ is 1-β); $t_{\alpha,v}$ = value from a one-tailed t table with probability α and v df; $t_{\beta(1),v}$ = value from a one-tailed t table with probability β and v df. The single value of n in Equation 1 assumes equal numbers of baseline and post-impact surveys. Since we are interested in the number of surveys required following the management action, it is necessary to rearrange equation 8.21 in Zar (1996) to compute the value of n to use in equation 1 as follows:

(2) $$n = \frac{2n_1n_2}{n_1+n_2}$$

where: n = number of years to detect an impact calculated in equation 1 above; n_1 = number of baseline surveys completed before fish stocking begins; n_2 = number of post-impact surveys. Equation 2 corrects for the reduction in efficiency that is associated with unequal sample sizes in the baseline and potential impact periods. Equation 1 and 2 were used to plot detectable impact over a number of surveys following the onset of an impact. The plotted power relationships were compared among taxa to illustrate how temporal variation and the number of samples influence detectable impact.

An important limit to detectable impact is how rapid detection must occur to prevent impacts from causing permanent harm or unacceptably altering other valued attributes of the population. For each taxon, we determined the number of cohorts that were potentially susceptible to the interaction types of most concern and subtracted them from the age of the normal maximum reproductive cohort. This difference is the number of cohorts buffered from impacts. For example, interference competition among salmonids is most intense when the species of concern is smaller or similar in size to competing species (Abbott et al. 1985, McMichael et al. 1997). Furthermore, prey fish can experience a size refuge from predators if they grow sufficiently large (Pearsons and Fritts 1999). Once the size range of low vulnerability is determined for a taxon, then the age at that length was determined using information collected in the Yakima basin (Fast et al. 1991; Martin and Pearsons 1994; Bonneville Power Administration 1996; Washington Department of Fish and Wildlife (WDFW) and YIN unpublished data) and elsewhere (Carlander 1969; Wydoski and Whitney 1979; Close et al. 1995). Maximum reproductive ages were determined from age work that has been done in the Yakima basin unless age information was unavailable (Fast et al. 1991; Martin and Pearsons 1994; Bonneville Power Administration 1996; WDFW and YIN unpublished data). Other sources outside of the Yakima basin were used to fill in information gaps (Carlander 1969; Wydoski and Whitney 1979; Close et al. 1995). We also determined how much time might pass between the earliest possible onset of an impact and its appearance in cohorts that can be effectively surveyed. The time available to detect impacts was calculated by subtracting the time lag, if any, from the number of buffered cohorts. The resulting time limit provides a minimal margin of safety in assuring continued spawning recruitment if nearly 100% of individuals in susceptible cohorts are lost until detection, but provides a greater margin of safety for more moderate impact intensities. Longer detection times would be acceptable if there is no risk of rapid, large impacts, if impacts are not additive, or if compensatory mechanisms are effective.

Where biological detection time limits are not critical, environmental factors can still limit the time available to detect impacts. Changes in the environment over time may lead to significant differences in abundance in the absence of the impact of interest. Climatological cycles, anthropogenic changes to habitat or flow regimes, and pollution are some of the factors outside the test of supplementation that could alter fish abundance. To limit the influence of these types of changes within a test series, we have chosen to set an environmental detection time limit of five annual surveys. Longer or shorter environmental detection time limits might be more suited to the dynamics characteristic of other river systems.

Detectable impacts were computed for each taxon using both the biological and environmental detection time limit as the number of post-impact surveys (n_2). In addition to evaluating detectable impacts at the detection time limit at the desired level of power (β = 0.1, power = 0.9), we examined detectable impacts for a lower level of power (power = 0.5). At this minimum level of power, it is possible to statistically detect smaller impacts, but with a much greater probability that the impact will go undetected, even if it is of the specified size. To provide a comparison at a uniform detection time, detectable impacts were computed for 5 post-impact surveys at both levels of power.

Results

Feasibility sampling and baseline surveys

Feasibility of sampling to quantify fish abundance differed widely among resident taxa. Three major factors influencing feasibility of status surveys for these taxa were extent and uniformity of distribution across survey sites, abundance at survey sites, and sampling efficiency. These factors determine how difficult it is to enumerate fish. Table 1 compares attributes that influenced feasibility among resident taxa. Feasibility of sampling to quantify status differed only slightly among anadromous taxa. The availability of opportunities to count juvenile migrants, the abundance of individuals being counted, and the ability to differentiate among stocks or species determined the quality of status estimates. Because traps capture only a fraction of emigrants, less abundant taxa can have low daily counts that yield lower precision in estimates of total passage. If smolt trap counts are high, as they are for fall chinook salmon and spring chinook salmon, sampling feasibility is high. For lamprey, emigrant counts are low and sampling feasibility is low, even before considering that counts may include brook lamprey. Steelhead smolt counts are moderate and sampling feasibility is moderate.

The number of baseline surveys differed widely among taxa (Table 2). The number of surveys conducted influenced feasibility when population estimation methods improved rapidly in early sampling, as it did with mainstem rainbow trout. Sampling methods in 1991 and following were improved over methods in effect during 1990, so 1990 data is excluded from mainstem rainbow trout analyses. In the case of leopard dace, mountain sucker, and sand roller, too few samples have been collected to fully evaluate potential feasibility, but the lack of information results in low feasibility. The number of baseline surveys is generally high for anadromous species, and did not limit feasibility.

Table 1. Resident native fish sampling feasibility as determined by distribution, abundance, and sampling efficiency.

Taxon	Distribution	Abundance[a]	Sampling efficiency[b]	Feasibility[a]
Bull trout	Limited-uniform	L	H	H
Cutthroat trout	Limited-uniform	M	H	H
Leopard dace	Limited-patchy	L	L	L
Longnose dace	Wide-uniform	H	M	H
Mountain sucker	Wide-uniform	L	ML	L
Mountain whitefish	Wide-uniform	H	L	H
Rainbow trout - mainstem	Wide-uniform	H	M	H
Rainbow trout - tributaries	Wide-uniform	H	H	H
Sand roller	Limited-patchy	L	L	L
Sculpin	Wide-uniform	H	M	H
Speckled dace	Wide-uniform	H	M	H
Sucker	Wide-uniform	H	L	H

[a]L=Low. M=Moderate. H=High.
[b]For sampling efficiency, L indicates capture efficiency is less than 5% in a given pass, M indicates efficiency is between 5 and 30%, and H indicates efficiency is greater than 30%. ML indicates a value that is near the boundary between the M and L categories.

Table 2. Baseline population abundance and variation over time.

Taxon	units	Year																	n	Mean	CV
		57	83	84	85	86	87	88	89	90	91	92	93	94	95	96	97	98			
Bull trout	number/km												3			1	1	1	4	1	76
Cutthroat trout	number/km									251	98	53	102	166	182	39	15	142	9	117	65
Fall chinook	Smolts (1000s)		104	44	68	33	154	76	28	111	55	253	149	196	33	7	36	397	16	109	94
Leopard dace	number/km	10															5		2	8	43
Longnose dace	number/km											850	590	650	290	368	218	625	7	513	44
Mountain sucker	number/km	5															50		2	28	116
Mountain whitefish	number/km												276	348	239	137	273	357	6	272	30
Pacific lamprey	Emigrant count												613	102	367	27	15	48	6	195	124
Rainbow trout - mainstem	number/km										131	117	129	149	101	134	223	126	8	139	26
Rainbow trout - tributary	Number/km									177	150	207	168	213	124	95	182	272	9	176	29
Sand roller	Number/km	45															0		2	23	141
Sculpin	Number/km											89	43	77	53	31	24	54	7	53	44
Speckled dace	number/km												1188	1585	1033	948	270	470	6	915	52
Spring chinook salmon	Smolts (1000s)		214	146	232	192	215	214	71	161	101	103	76	191	170	364	142	315	16	182	44
Steelhead	Smolts (1000s)		12	35	33	50	43	17	14	7	6	10	4	3	2	2	3	10	16	16	101
Sucker	number/km												208	277	179	140	197	223	6	204	22

Modeling

Models have been successfully applied to 10 of 16 taxa. Variation in flow, temperature and spawning escapement accounts for much of the variation in abundance for some taxa. Models accounted for between 19-61% of variation in abundance estimates for these taxa (Table 3). Models have yet to be constructed for other taxa due to a lack of sufficient series of baseline status surveys or a need for explanatory information. It will not be possible to construct models for some taxa. Leopard dace, mountain sucker, and sand roller are examples of taxa with little baseline data and incomplete life history information. Although many surveys have been completed for some anadromous species, such as spring chinook salmon, modeling efforts using available information were not successful.

Table 3. Model parameters and variation reduction. Models have not been constructed for taxa not listed.

Taxa	Index 1	Index 2	Index 3	Variation reduction (%)
Cutthroat trout	Flow PC1[a] (lagged 1 yr)	Flow PC1		42
Fall chinook salmon	Temperature PC2	Brood year escapement	Escapement >1200 (crosstab)	46
Longnose dace	Flow PC1			57
Mountain whitefish	Flow PC1			48
Rainbow trout - mainstem	Flow PC1			49
Rainbow trout - tributaries	Flow PC1	Flow PC2		22
Sculpin	Flow PC1			59
Speckled dace	Flow PC1			19
Steelhead	Temperature PC1	Redd count		61
Sucker	Flow PC1			39

[a]PC indicates a principal component factor score. PC1 indicates the first principal component factor score, PC2 indicates the second principal component factor score.

Power analyses and detectable impacts

Detectable impacts varied widely among native fish taxa with respect to baseline temporal variation. Figure 2 illustrates the relationship among the number of post-impact surveys and the detectable impact at a power level of 0.9 for each taxa, given the baseline level of temporal variation and the number of baseline surveys.

For anadromous species, detection of impacts less than 36% is unlikely, even after 10 surveys (Figure 2a). Lamprey have the highest detectable impacts, in part due to the difficulty of enumerating relatively rare emigrants. For resident salmonids, detection of impacts less than 15% is unlikely, even after 10 surveys (Figure 2b). Bull trout have the highest detectable impact because their small numbers result in greater relative sampling error. Mainstem rainbow trout have the lowest detectable impacts of resident salmonids, reflecting low temporal variability and the large amount of effort directed toward the taxa. The group of resident non-salmonids display a wide range of detectable impacts (Figure 2c). Leopard dace, mountain sucker, and sand roller have been surveyed only once and historical collections have been used to compute a rough estimate of maximum variability for use in plotting the power relationship. Only leopard dace achieve power to detect less than a 65% impact within ten surveys in the impact period. More abundant resident salmonids, that also received more sampling effort than the former group, achieved much lower detectable impacts. Detectable impacts between 18 and 25% were found for sucker, sculpin, and longnose dace after 10 surveys.

Biological detection time limits ranged from zero to seven years. Table 4 shows, for each taxon, which cohorts are susceptible, the earliest cohort that can be monitored, and the maximum number of cohorts within a generation. The resulting lag, buffered cohorts and detection time limit are also reported. The detection time limits for large, long-lived fish were generally longer than for smaller, shorter-lived fish. Detection time limits indicate the number of post impact surveys that can be conducted to accomplish detection before the impact reaches all cohorts (Table 4). Most of the time limits for the 16 taxa were four years or less (88%) and 29% of the taxa had time limits of zero years (i.e., the year of impact). Pacific lamprey differed from other species because the time of vulnerability was hypothesized to be limited to the year of emigration. Cohorts before and after the year of emigration were believed to be relatively isolated from interactions with spring chinook salmon due to unique life history characteristics.

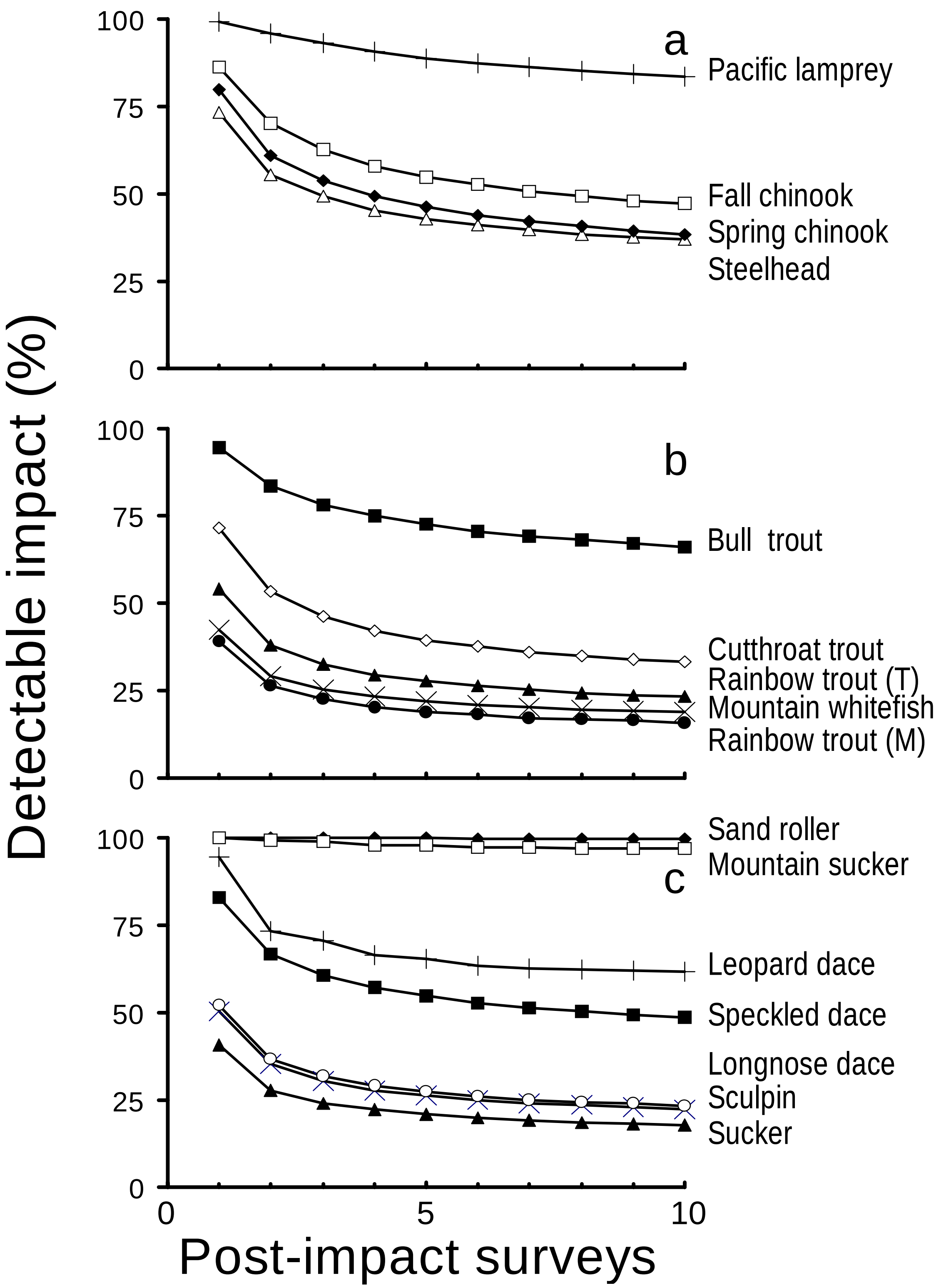

Figure 2. Detectable impact versus number of post-impact survey years for all taxa. The number of post impact surveys is counted beginning at the first survey after the impact is imposed at a given level. Detectable impact computations assume a constant impact throughout the detection period.

Short detection time limits increased detectable impacts above the potential minima illustrated in Figure 2. Table 5 indicates how detection time limits from Table 4 combine with power curves from Figure 2 to determine detectable impacts. Detectable impacts do not approach the minimum until after five post-impact surveys (Figure 2) for most taxa, but detection time limits are usually less than four years (Table 4). Therefore, the biological characteristics of a taxon affect what

Table 4. Susceptible cohorts and detection time limits. Susceptible cohorts are those potentially impacted by spring chinook salmon smolts (average spring chinook salmon length is assumed to be 130mm). Monitoring lag is the difference between earliest potential cohort impacted and the earliest cohort that can be monitored following an impact (e.g., impacts to age 1 steelhead would not be evident until they appear in monitoring surveys at age 3 (2+) = 2 year lag). Detection time limits are computed by subtracting the lag from the number of buffered cohorts.

Taxon	Susceptible cohorts	Earliest cohort monitored	Lag	Maximum cohorts	Buffered cohorts	Detection time limit
Bull trout	1,2	2	1	7	5	4
Cutthroat trout	1,2	2	1	6	4	3
Fall chinook salmon	1	1	0	5	4	4
Leopard dace	1,2,3,4,5	2	1	5	0	0
Longnose dace	1,2,3,4,5	2	1	5	0	0
Mountain sucker	1,2,3,4	2	1	9	5	4
Mountain whitefish	1,2	1	0	9	7	7
Pacific lamprey	4,5,6	4	0	9	6	6
Rainbow trout – mainstem	1,2	2	1	5	3	2
Rainbow trout – tributaries	1,2	2	1	4	2	1
Sand roller	1,2,3,4,5,6	2	1	6	0	0
Sculpins	1,2,3,4,5	2	1	5	0	0
Speckled dace	1,2,3	2	1	3	0	0
Spring chinook salmon	1,2	2	1	6	4	3
Steelhead	1,2	3	2	6	4	2
Suckers	1,2,3,4	2	1	11	7	6

impacts are detectable. Detectable impacts are also shown for a minimal power level (power = 0.5) that results in a nearly 50% reduction of detectable impact, but with a higher probability of failing to detect an impact of the smaller size. Small detection time limits have a great impact on the detectable impact, as is evident in Figure 2, but unexplained temporal variation of baseline estimates is still the primary determinant of what level of impacts are detectable.

Table 5. Detection time limits and detectable impacts for full and minimum levels of power for both biological and environmental detection time limits.

	Detectable impact (%)				
	Biological time limit			Environmental time limit (5 years)	
Taxon	Detection time limit (years)	Power = 0.9	Power = 0.5	Power = 0.9	Power = 0.5
Bull trout	4	75	50	73	48
Cutthroat trout	3	46	27	40	22
Fall chinook salmon	4	58	35	55	33
Leopard dace	0(1)	94	76	65	41
Longnose dace	0(1)	52	31	27	15
Mountain sucker	4	98	86	98	85
Mountain whitefish	7	20	11	22	12
Pacific lamprey	6	87	64	89	67
Rainbow trout – mainstem	2	26	14	19	10
Rainbow trout – tributaries	1	54	32	28	15
Sand roller	0(1)	100	100	100	96
Sculpin	0(1)	50	30	26	14
Speckled dace	0(1)	83	59	55	33
Spring chinook salmon	3	54	32	46	27
Steelhead	2	55	33	43	24
Sucker	6	20	10	21	11

Discussion

To answer the question that we posed in our title, "Can fish stocking be adaptively managed to limit impacts to native fish abundance?" we must determine whether impacts can be detected in time to allow management actions to be implemented. Impacts less than an acceptable level must be detectable so that management actions can prevent impacts from exceeding an acceptable level. In addition, detection must be rapid so that, if required, management actions can effect a reversal or mitigation of the impacts before they progress beyond the acceptable level. Our results indicate that impacts to population abundance less than 20% are generally not detectable within biologically or environmentally based time limits (Figure 3). Although impacts greater than 20% may be acceptable for abundant species with low utility value (e.g., speckled

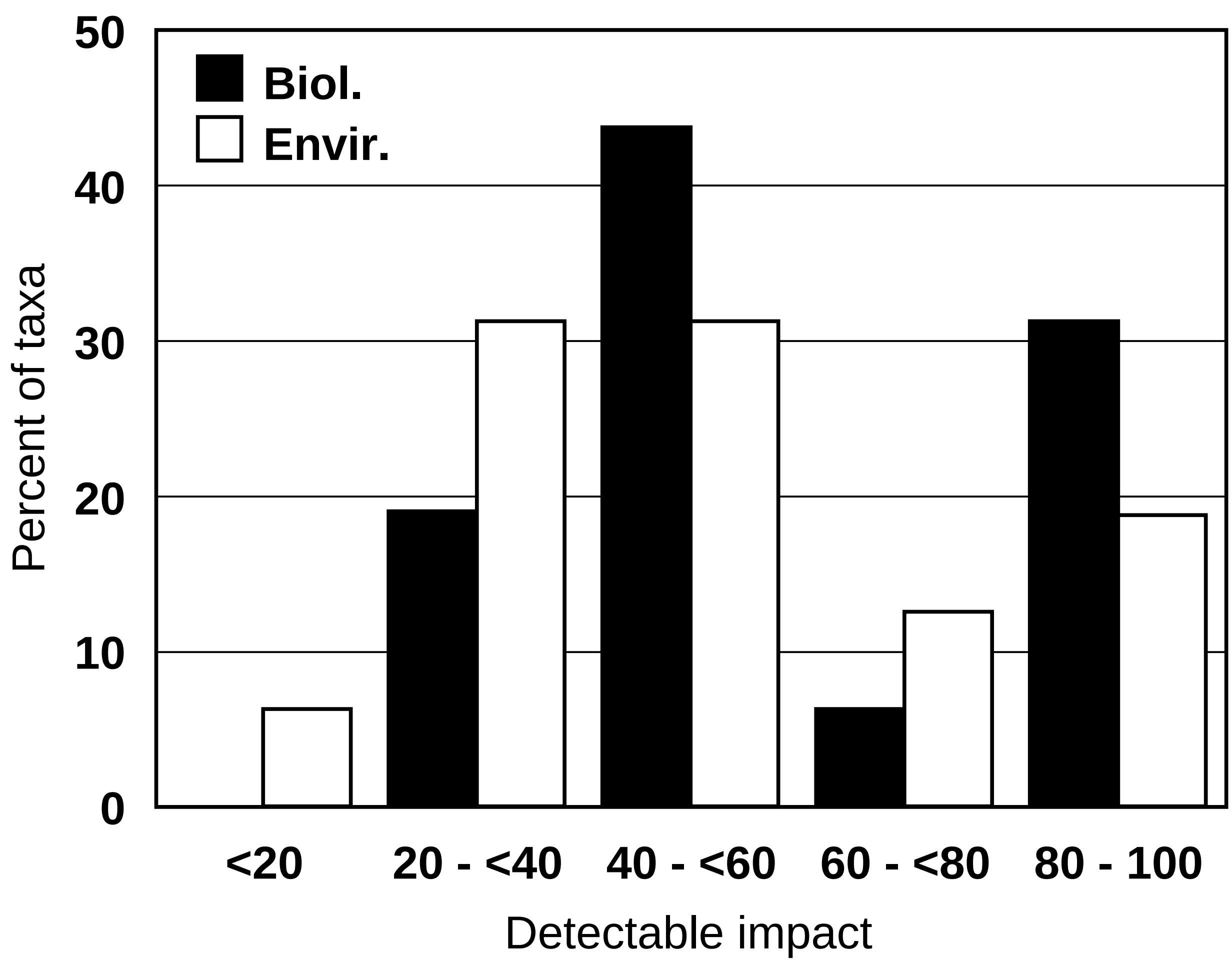

Figure 3. Percent of taxa in detectable impact categories for biologically- and environmentally-based detection time limits.

dace, longnose dace, sculpins, suckers), they would be unacceptable for many other species that are rare (e.g., ESA; bull trout, steelhead) or have high utility value (e.g., rainbow trout). These rare and important taxa are usually the ones that are the focus of adaptive management attention – yet our data indicate that abundance estimates are too variable for effective adaptive management of these taxa, even within our most liberal criteria for detection (e.g. environmental detection time limit of 5 years and power = 0.5). Our results are consistent with studies of single populations that found high temporal variation often makes detecting impacts difficult, even if longer times are available for detection (Van Winkle et al. 1981; Vaughan and Van Winkle 1982; Peterman and Bradford 1987). Our results illustrate that rapid, sensitive detection of impacts to rare or important species is unlikely to occur under natural conditions, even when considerable effort is expended in preparing to detect impacts through status monitoring. That conclusion leads us to pose the question: "could anything be done differently to achieve rapid sensitive impact detection to abundance?"

The primary ways to increase rapid, sensitive impact detection to abundance are to 1) minimize variation of estimates among years, and 2) produce predictive models that explain large amounts of variation in annual abundance estimates. We attempted to minimize the variation in our data sets by conducting many annual surveys for many taxa and by using state-of-the-art census techniques. Twelve of 16 taxa that we studied had between six and 16 years of systematic surveys. This abundance of data can rarely be expected to be available in most other locations. The amount of variation that we observed in the Yakima basin was generally consistent with published studies and few studies have reported variations in multiple annual surveys of abundance that are much lower than what we observed for our least variable taxa, rainbow trout in the mainstem (Figure 4). Although it might be possible, through increased effort, to considerably reduce the variation associated with study design and sampling error, the results of other studies do not suggest that our least variable taxa can benefit enough to foster sufficient impact detection. However, high natural variation does not indicate that rapid, sensitive impact detection is impossible. To the contrary, predictive models that explain much of the variation in abundance estimates have the potential to accomplish our goal.

We attempted to construct simple, predictive models that would explain the maximum amount of variation in our abundance estimates, but none of our models could explain enough of the variation to meet our goal. Our models explained between 19 and 61% of the variation in abundance estimates. For comparison, we computed how much variation would need to be accounted for by a model to allow detection of a 5% impact. Our least variable anadromous species, spring chinook salmon, would require a model that accounted for 92% of variation to achieve detection of 5% impact at power = 0.9 within 5 years. The least variable resident salmonid, mainstem rainbow trout, would require a model that accounted for 87% of variation to achieve the same detection ability. The least variable resident non-salmonid, suckers, would require 77% of variation to be accounted for to achieve the same detection ability. Other taxa in these groups would require models that accounted for an even greater proportion of variation to achieve detection of 5% impact at power = 0.9 within 5 years. These scenarios suggest that a combination of very low variation and a very good model could support adaptive management of impacts to even rare and important species. The effort required to create a model that explains enough variation in abundance estimates may be difficult to justify. It seems unlikely that low variation and a very good model will occur in combination for other than a few well-studied and economically important species.

Adaptively managing the impacts of fish stocking on native taxa will be more effective if indicators of impact that respond more rapidly than population abundance can be identified and utilized. Alternatives to abundance monitoring, such as monitoring mechanisms of interaction or conducting controlled experiments to evaluate the potential for impacts (e.g. McMichael et al. 1997; McMichael and Pearsons 1998), can increase the rapidity of impact detection or eliminate the need for impact detection. For example, interactions indices such as spatial overlap and predation can be detected in a single year. Although interactions mechanisms can be easily detectable, it is difficult to interpret how an interactions index reflects an impact to population abundance (McMichael and Pearsons 1998). In addition, interactions indices generally focus on a specific interaction type that is hypothesized to be strong and may not account for the effects of different interaction types or cumulative effects. Because population abundance monitoring

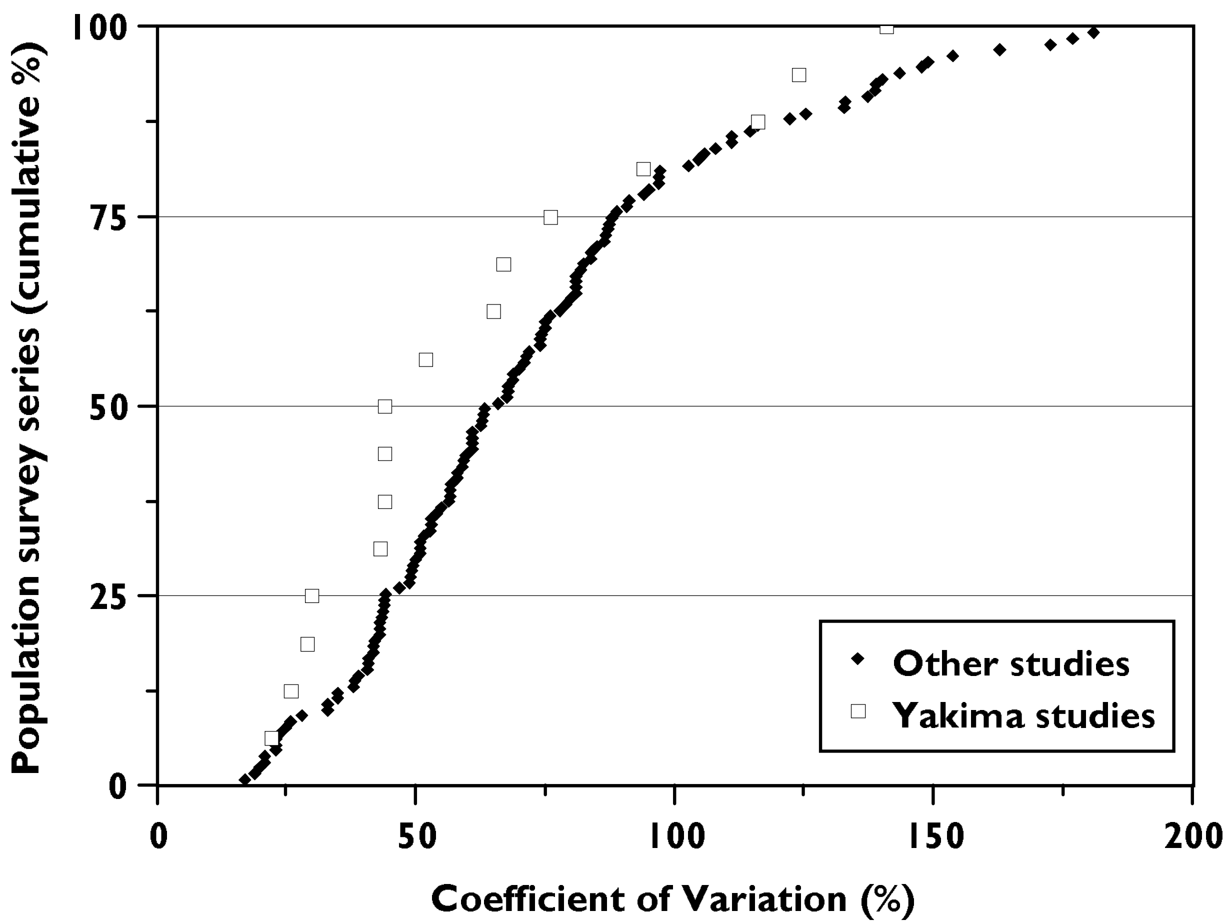

Figure 4. Cumulative percent of population abundance survey variation compared among the taxa surveyed in this study and 131 population abundance survey series from other studies. Open squares indicate individual taxa abundance survey series collected in this study. Closed diamonds indicate population abundance survey series collected in 131 other studies. Other studies compiled from Streamnet database (1998) and Gibbs (2000).

integrates impacts from all interaction types, it may have a role in containing impacts that might otherwise be impossible to monitor. Interactions index monitoring and abundance monitoring can be used together to overcome some shortfalls of each technique in isolation.

Conclusions

Fish abundance monitoring will generally not be sufficient to detect small impacts and hence will not meet the requirements of adaptive management for rare or highly valued fish taxa. Inability to adaptively manage using abundance monitoring suggests that we must be more risk averse, or use other measures of impacts that can provide the rapid, sensitive impact detection that will allow adaptive management to succeed. For example, more easily measured interactions indices (e.g., spatial overlap, predation index) could be used to detect mechanisms of impacts well in advance of changes in abundance. More thorough risk assessment and uncertainty resolution could direct effort toward impacts of greatest concern or likelihood. Where these approaches fail to enable adaptive management, taking a risk-averse approach may be the only option that is sufficient for management of risks to rare and important taxa.

Acknowledgments

We thank Mark Johnston who provided the lamprey data, Bruce Watson and Joel Hubble who provided the data on anadromous salmonids. Many members of the Washington Department of Fish and Wildlife's Ecological Interactions Team were instrumental in the collection of data and in discussions about monitoring. Of particular note are Eric Bartrand, Anthony Fritts, Geoff McMichael, Steve Martin, and Gabe Temple. Craig Busack, Kenny Rose, and Geoff McMichael provided helpful reviews of the draft manuscript. Steve Leider, Bill Hopley, and Keith Wolf provided the technical/policy interface that allowed us to stay focussed and funded. We are indebted to Tom Vogel and David Byrnes of the Bonneville Power Administration who helped secure long-term funding for the Yakima Species Interactions Studies that made this work possible.

References

Abbott, J.C., Dunback, R.L., and Orr, C.D. 1985. The interaction of size and experience in dominance relationships of juvenile steelhead trout (*Salmo gairdneri*). Behaviour **92:** 241-253.

Bonneville Power Administration. 1996. Yakima fisheries project - Final environmental impact statement. Bonneville Power Administration, Portland, Oreg. DOE/EIS-0169.

Busack, C., Watson, B., Pearsons, T., Knudsen, C., Phelps, S., and Johnston, M. 1997. Yakima fisheries project spring chinook supplementation monitoring plan. Bonneville Power Administration, Portland, Oreg. DOE/BP-64878-1.

Carlander, K.D. 1969. Handbook of freshwater fishery biology, Volume 1. Iowa State University Press, Ames, Iowa.

Close, D.A., Fitzpatrick, M., Li, H., Parker, B., Hatch, D., and James, G. 1995. Status report of the Pacific lamprey (*Lampetra tridentata*) in the Columbia River Basin. Bonneville Power Administration, Portland Oregon. DOE/BP-39067-1.

Clune, T., and Dauble, D. 1991. The Yakima/Klickitat fisheries project: a strategy for supplementation of anadromous salmonids. Fisheries **16(5)**: 28-34.

Cuffney, T.F., Meador, M.R., Porter, S.D., and Gurtz, M.E. 1997. Distribution of fish, benthic invertebrate, and algal communities in relation to physical and chemical conditions, Yakima River Basin, Washington 1990. U.S. Geological Survey Water Resources Investigations Report 96-4280. Raleigh, North Carolina.

Eberhardt, L.L., and Thomas, J.M. 1991. Designing environmental field studies. Ecological Monographs **61**: 53-73.

Fast, D.E., and Craig, C. 1997. Innovative hatchery project working to rebuild wild salmon populations. Hydro Review **14**: 30-33.

Fast, D., Hubble, J., Kohn, M., and Watson, B. 1991. Yakima River spring chinook salmon enhancement study. Final report submitted to Bonneville Power Administration, Portland, Oreg. DOE/BP-39461-9.

Fish and Wildlife Service. 1998. Endangered and threatened wildlife and plants; determination of threatened status for the Klamath River and Columbia River distinct population segments of bull trout. Federal Register [June 10, 1998] **63**(111): 31647-31674

Gibbs, J.P. 2000. Monitoring populations. *In* Research techniques in animal ecology: controversies and consequences. *Edited by* T. Fuller and L. Boitani. Columbia University Press, New York. In press.

Green, R.H. 1979. Sampling design and statistical methods for environmental biologists. John Wiley and Sons, New York.

Hunn, E.S. 1990. Nch'i-Wana "The Big River" Mid-Columbia Indians and their land. University of Washington Press, Seattle and London.

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs **54**: 1877-211.

Independent Scientific Group. 1996. Return to the river: Restoration of salmonid fishes in the Columbia River ecosystem. Northwest Power Planning Council, Portland, Oreg.

Krause, T. 1991. The Yakima. Fly Fisherman **22**: 40-43,76-78.

Lassuy, D.R. 1995. Introduced species as a factor in extinction and endangerment of native fish species. American Fisheries Society Symposium **15:** 391-396.

Leland, H.V. 1995. Distribution of phytobenthos in the Yakima River basin, Washington, in relation to geology, land use, and other environmental factors. Canadian Journal of Fisheries and Aquatic Sciences **52**: 1108-1129.

Li, H. W., Schreck, C.B., Bond, C.E., and Rexstad, E. 1987. Factors influencing changes in fish assemblages of Pacific Northwest streams. *In* Community and evolutionary ecology of North American stream fishes. *Edited by* W. J. Matthews and D. C. Heins. University of Oklahoma Press, Norman, Okla. Pp. 193-202.

Major, R.L., and Mighell, J.L. 1969. Egg-to-migrant survival of spring chinook salmon (*Oncorhynchus tshawytscha*) in the Yakima River, Washington. Fishery Bulletin U.S. **67**: 347-359.

Martin, S.W., and Pearsons, T.N. 1994. Age and growth of rainbow trout in the upper Yakima River basin. *In* Yakima River species interactions studies. Annual report for FY 1993. *Prepared by* T.N. Pearsons, G.A. McMichael, K.D. Ham, E.L. Bartrand, A.L. Fritts, and C.W. Hopley. Bonneville Power Administration, Portland, Oreg. pp. 105-124.

McMichael, G.A., Sharpe, C.S., and Pearsons, T.N. 1997. Effects of residual hatchery-reared steelhead on growth of wild rainbow trout and spring chinook salmon. Transactions of the American Fisheries Society **126**: 230-239.

McMichael, G.A., and Pearsons, T.N. 1998. Effects of wild juvenile spring chinook salmon on growth and abundance of wild rainbow trout. Transactions of the American Fisheries Society **127**: 261-274.

McMichael, G.A., Pearsons, T.N., and Leider, S.A. 1999. Minimizing ecological impacts of hatchery-reared juvenile steelhead trout on wild salmonids in a Yakima basin tributary. Pages 365-380 in Eric Knudson et al., editors. Sustainable fisheries management: Pacific salmon. CRC Press, Boca Raton, FL.

National Marine Fisheries Service. 1998. Endangered species: Proposed threatened status for two ESUs of steelhead in Washington and Oregon. Federal Register [Docket No. 980225046-8046-01, 10 March 1998] **63**(46): 11797-11809.

National Research Council. 1995. Science and the endangered species Act. National Academy Press, Washington, D.C.

National Research Council. 1996. Upstream: salmon and society in the Pacific Northwest. National Academy Press, Washington D. C.

Nehlsen, W.J., Williams, J.E., and Lichatowich, J.A. 1991. Pacific salmon at the crossroads: stocks at risk from California, Oregon, Idaho, and Washington. Fisheries **16(2)**: 4-21.

Patten, B.H., Thompson, R.B., and Gronlund, W.D. 1970. Distribution and abundance of fish in the Yakima River, Washington, April 1957 to May 1958. U. S. Dept. Int. Fish Wildl. Serv. Fish Bull. No. 603.

Pearsons, T.N. 1998. Draft objectives for non-target taxa of concern relative to supplementation of upper Yakima spring chinook salmon. *In* Yakima River Species Interactions Studies. Progress report 1995-1997. *Prepared by* T.N. Pearsons, G.A. McMichael, K.D. Ham, E.L. Bartrand, A.L. Fritts, and C.W. Hopley. Bonneville Power Administration, Portland, Oreg. pp. 4-44.

Pearsons, T.N., and Fritts, A.L. 1999. Maximum size of chinook salmon consumed by juvenile coho salmon. North American Journal of Fisheries Management 19:165-170.

Pearsons, T.N., and Hopley, C.W. 1998. A practical approach for assessing ecological risks associated with stocking anadromous salmonids. *In* Yakima River Species Interactions Studies. Progress report 1995-1997. *Prepared by* T.N. Pearsons, G.A. McMichael, K.D. Ham, E.L. Bartrand, A.L. Fritts, and C.W. Hopley. Bonneville Power Administration, Portland, Oreg. pp. 278-291.

Pearsons, T.N., McMichael, G.A., Ham, K.D., Bartrand, E.L., Fritts, A.L., and Hopley, C.W. 1998. Yakima River Species Interactions Studies. Progress report 1995-1997. Bonneville Power Administration, Portland, Oreg.

Peterman, R.M. 1990. Statistical power analysis can improve fisheries research and management. Canadian Journal of Fisheries and Aquatic Sciences **47**: 2-15.

Peterman, R.M., and Bradford, M.J. 1987. Statistical power of trends in fish abundance. Canadian Journal of Fisheries and Aquatic Sciences **44**: 1879-1889.

Probasco, S. 1994. River Journal, Volume 2; Yakima River. Frank Amato Publications, Incorporated. Portland, Oreg.

Regional Assessment of Supplementation Planning. 1992. Supplementation in the Columbia Basin. Final Report submitted to Bonneville Power Administration, Portland, Oreg. DOE/BP-01830-14.

Schramm, H.L., Jr., and Piper, R.G., editors. 1995. Uses and effects of cultured fishes in aquatic ecosystems. American Fisheries Society Symposium 15.

Stewart-Oaten, A., Bence, J.R., and Osenberg, C.W. 1992. Assessing effects of unreplicated perturbations: No simple solutions. Ecology **73**: 1396-1404.

StreamNet Database (Version 98.3) [database downloaded to disk]. Gladstone (OR) : StreamNet, April 1998. [16 March 1998] . URL: <http://www.streamnet.org/accesstable.html>.

Stroud, R.H. 1986. Fish Culture in Fisheries Management. Proceedings of a symposium on the role of fish culture in fisheries management. American Fisheries Society, Bethesda, Maryland.

Thurow R.F., Lee, D.C., and Rieman, B.E. 1997. Distribution and status of seven native salmonids in the interior Columbia River basin and portions of the Klamath River and Great basins. North American Journal of Fisheries Management **17:** 1094-1110.

Van Winkle, W., Vaughan, D.S., Barnthouse, L.W., and Kirk, B.L. 1981. An analysis of the ability to detect reductions in year-class strength of the Hudson River white perch (*Morone americana*) population. Canadian Journal of Fisheries and Aquatic Sciences **38**: 627-632.

Vaughan, D.S., and Van Winkle, W. 1982. Corrected analysis of the ability to detect reductions in year-class strength of the Hudson River white perch (*Morone americana*) population. Canadian Journal of Fisheries and Aquatic Sciences **39**: 782-785.

Wahle, R.J., and Pearson, R.E. 1984. History of artificial propagation of coho salmon *Oncorhynchus kisutch,* in the mid-Columbia River system. Marine Fisheries Review **46**: 34-43.

Walters, C.J. 1986. Adaptive management of renewable resources. MacMillan, New York.

Walters, C.J., and Hilborn, R. 1976. Adaptive control of fishing systems. Journal of the Fisheries Research Board of Canada **33**: 145-159.

White, R.J., Karr, J.R., and Nehlsen, W. 1995. Better roles for fish stocking in aquatic resource management. American Fisheries Society Symposium **15**: 527-547.

Wydoski, R.S., and Whitney, R.R. 1979. Inland fishes of Washington. University of Washington Press. Seattle, Washington.

Zar, J.H. 1996. Biostatistical Analysis. Prentice Hall, Upper Saddle River, New Jersey.

Chapter 2

Can native fishes be monitored well enough to support adaptive management of ecological impacts?

Kenneth D. Ham

and

Todd N. Pearsons

Washington Department of Fish and Wildlife
600 Capitol Way North
Olympia, Washington 98501

Abstract

We have examined the potential to adaptively manage ecological impacts to wild fishes using data from the Yakima River basin, Washington, where a large-scale test of hatchery supplementation is being conducted. Non-target taxa (NTT; n = 16) have been identified, impact containment objectives (CO) determined, and up to 16 annual baseline surveys completed. Sensitivity and speed of impact detection through status or interactions monitoring of NTT were evaluated. We defined status of an NTT population as its distribution, abundance, and size structure. Interactions monitored include predation and spatial overlap with target species. Monitoring options, alone or in combination, often failed to achieve adequate power to detect impacts equal to the CO for some or all interaction types. Impact detection and containment at or below the CO was only rarely possible for rare or valuable taxa (CO = 0 - 10% reduction relative to baseline status). For rare or valuable taxa, which have the greatest need for impact containment, monitoring is unlikely to provide the rapid feedback necessary to assure that unacceptable or irreversible loss of valued population characteristics does not occur. Inadequate feedback will prevent the adaptive management approach from assuring that ecological impacts to NTT that exceed the CO are quickly corrected.

Introduction

We pose a question in the title of our paper that, if answered, can have important implications for the future of fisheries management. The core issue is whether the potential for ecological impacts must be shown as acceptably low prior to a management action or whether monitoring can provide feedback sufficient to keep impacts acceptably low during implementation of the action. The latter of these two scenarios is referred to as adaptive management (Walters and Hilborn 1976; Walters 1986). Adaptive management is designed to evaluate risks as they occur and to make mid-course adjustments to keep risks within acceptable limits. This approach is appealing because it allows a proposed action to proceed without certainty of its outcomes and it promises protection of those organisms that might be adversely affected by the management action—a true win-win scenario. However, a critical assumption about the efficacy of adaptive management is that it is possible to detect and respond to changes if they exceed an acceptable level and, more importantly, before irreversible loss of the population or its valued characteristics occurs. That is, feedback from monitoring can adjust management actions before it is too late. This assumption has rarely been tested with multiple populations of native fish, and lack of knowledge of the robustness of the assumption can result in a false sense of security that may lead to a failure to recognize when impacts exceed acceptable levels and, in extreme cases, could allow irreversible harm to the population to occur (Waples 1999). It is this assumption that we intend to test in this paper.

In a previous study, we found that changes in abundance could not often be detected rapidly enough to facilitate adaptive management of rare and economically important populations with low acceptable impacts (Ham and Pearsons 2000). Often these are the populations society is most interested in and are often the primary focus of monitoring. If the valued characteristics of these species are to be protected through an adaptive management process, more rapid and sensitive indicators of impact are needed. Furthermore, population abundance is only one characteristic of a population that could be impacted by a management action. For instance, the size structure or distribution of a population could be unfavorably altered while abundance remains unchanged. Thus, multiple characteristics may be needed to describe the status of a population, any one of which could be adversely impacted. In contrast to our previous paper where we examined just abundance monitoring, in this paper we address abundance, size structure, and distribution monitoring for 16 non-target fish taxa. In addition, we explore other ways of measuring the potential for impacts to these population characteristics that are independent of status monitoring. We also describe the process of evaluating impact detection strategies and a decision framework for designing impact detection plans. The risk containment effort for the supplementation project in the Yakima River will be used as a case study to illustrate factors that may limit the effectiveness of monitoring and adaptive management of ecological impacts to wild fish populations.

Methods

Study area and background

The study area and background of this work was previously described by Ham and Pearsons (2000). Briefly, the work that is presented here was conducted in the Yakima Basin as part of an adaptively managed spring chinook salmon *Oncorhynchus tshawytscha* supplementation program (Clune and Dauble 1991; Busack et al. 1997; Fast and Craig 1997). The first hatchery-reared spring chinook salmon will be released into the upper Yakima basin during the spring of 1999. There was concern that supplementing spring chinook salmon could unintentionally adversely impact non-target taxa (NTT). In order to protect the basin's fishery resources from unforeseen, adverse impacts, project managers adopted an adaptive management policy (Bonneville Power Administration 1996).

Non-target fish taxa with a potential to be adversely impacted by supplementation activities were identified and containment objectives (CO) were developed to establish goals for limiting loss of valuable population characteristics attributable to supplementation, (Pearsons 1998). All of the NTT that were identified have the potential to overlap and interact with supplemented spring chinook salmon. Certain non-target populations were selected because of stewardship-related concerns for long-term survival. For instance, bull trout *Salvelinus confluentus* is federally listed as threatened and the mid-Columbia steelhead *O. mykiss* has recently been listed as threatened under the Endangered Species Act (U. S. Fish and Wildlife Service 1998; National Marine Fisheries Service 1999). In addition, Pacific lamprey *Lampetra tridentata* and westslope cutthroat trout *O. clarki* in the Columbia basin have severely restricted abundance and distribution relative to historic conditions (Close et al. 1995; Thurow et al. 1997). Mountain sucker *Catostomus platyrhynchus*, sand roller *Percopsis transmontana*, leopard dace *Rhinichthys falcatus*, and the Marion Drain stock of fall chinook salmon are rare within the Yakima basin (Patten et al. 1970; Pearsons et al. 1999). Other taxa were selected because of concerns about impacts to people that utilize them for recreation, food, science, livelihood, culture, or religious purposes. For example, rainbow trout *O. mykiss* in the Yakima River provide one of the best resident trout fisheries in the state of Washington and are currently managed as a catch and release fishery (Krause 1991; Probasco 1994). Naches and American river stocks of spring chinook salmon also provide significant recreational, subsistence, and commercial fisheries (Hunn 1990). Taxa such as mountain whitefish *Prosopium williamsoni* and tributary rainbow trout also support recreational fisheries. COs for these taxa reflect their societal value and are consistent with other management practices. Many native taxa, such as longnose dace *R. cataractae*, speckled dace *R. osculus*, sculpin *Cottus spp.*, and sucker *Catostomus spp.*, are common throughout the Yakima River basin and the region. CO for native taxa without exceptional stewardship or utilization concerns are intended to protect their sustainability. CO for these common native taxa were set by computing the maximum impact that maintains approximately 10,000 reproductive females.

The status of each NTT was characterized as the combination of distribution, abundance, and size structure. A decrease in any one of these characteristics beyond the CO, as a percent of baseline status, would constitute a failure to achieve the objective. We used an approach to analyze risks that was similar to that described by Pearsons and Hopley (1999). Potential interaction types and their potential risk vary among NTT (Table 1). Competition and predation are the ecological interaction types of greatest concern across all taxa. The level of risk and uncertainty relative to the CO prior to supplementation are categorized as high, medium, or low for each NTT (Table 1). Greater risk and uncertainty increase the need for effective impact containment to assure impacts do not exceed the CO. This need is summarized as a priority rank. Decreasing priority rank indicates increasing need for containment.

Data collection and analysis

The methods that we used to estimate baseline abundance of NTT are described in detail by Ham and Pearsons (2000). Briefly, populations were annually surveyed by methods appropriate to each taxa and stream size. Survey methods included electrofishing, passage counts at dams, visual counts while electrofishing, and snorkel counts. Population abundance was estimated through mark-recapture, removal, or expansions of other counts of individuals. Mean population abundance and temporal variation was estimated from annual surveys. Where possible, abundance surveys were also used to quantify size structure and distribution.

Table 1. Non-target taxa, primary concern, containment objective (CO), potential ecological interaction types, predicted risk, uncertainty, and priority rank associated with supplementing upper Yakima River spring chinook salmon.

Non-target Taxa	Primary concern	CO (%)	Interaction type					Risk	Uncertainty	Priority Rank
			Competition	Predation	Behavior	Nutrient mining	Disease			
Bull trout	Stewardship – regionally rare	0	$\boxed{\text{X}}$[a]				X[b]	L	M	4
Cutthroat trout	Stewardship – regionally rare	0	$\boxed{\text{X}}$				X	L	M	4
Pacific lamprey	Stewardship – regionally rare	0		$\boxed{\text{X}}$		X	X	H	H	2
Steelhead	Stewardship – regionally rare	0	$\boxed{\text{X}}$	$\boxed{\text{X}}$	$\boxed{\text{X}}$	X	X	H	M	1
Fall chinook salmon	Stewardship – rare in basin	5	$\boxed{\text{X}}$	$\boxed{\text{X}}$	X		X	L	M	4
Leopard dace	Stewardship – rare in basin	5	X	$\boxed{\text{X}}$			X	LM	L	5
Mountain sucker	Stewardship – rare in basin	5	X	$\boxed{\text{X}}$		X	X	LM	L	5
Sand roller	Stewardship – rare in basin	5	X	$\boxed{\text{X}}$			X	LM	L	5
Rainbow trout - mainstem	Utilization – very important	10	$\boxed{\text{X}}$	X	$\boxed{\text{X}}$	X	X	M	M	3
Spring chinook salmon	Utilization – very important	10	$\boxed{\text{X}}$	$\boxed{\text{X}}$	X		X	L	M	4
Mountain whitefish	Utilization – important	40	$\boxed{\text{X}}$	X		X	X	L	L	6
Rainbow trout - tributaries	Utilization – important	40	$\boxed{\text{X}}$	X	X	X	X	L	L	6
Longnose dace	Sustainability	65	X	X		X	X	L	L	6
Speckled dace	Sustainability	85	X	X		X	X	L	L	6
Sculpins	Sustainability	90	X	X		X	X	L	L	6
Suckers	Sustainability	90	X	X		X	X	L	L	6
Other common native species	Sustainability	Sufficient spawners	X	X	X	X	X	L	L	6

[a]$\boxed{\text{X}}$ indicates interaction types of greatest concern. [b]X indicates interaction types of concern. L = low. LM = low to medium. M = medium. H = high.

Size structure of populations was quantified as the mean length, if fish could be measured directly, or as the proportion of adults, judged visually by size. Mean size and temporal variation were estimated from annual survey results. Size structure was not determined for taxa with few individuals in survey collections. Spatial distribution was quantified as the proportion of surveyed habitat that contained more than one individual of the taxa of interest. Only sites where multiple individuals of the species had been collected in at least one annual survey were included in computing the proportion. Spatial distribution was not determined for anadromous taxa, due to their temporal habitat use, lack of sampling in important locations and difficulty in sampling. Distribution was not determined for resident taxa where survey locations did not encompass a large proportion of their range.

Surveys were also conducted to establish baseline magnitude and variability for several indicators of interaction (Busack et al. 1997). One interaction indicator was the predation index (McMichael et al. 1999). One purpose of the predation index is to reveal if increased numbers of individuals of the target taxon or other stocking activities result in increased predation on NTT. Additional survey locations or times will be added to index predation effects on NTT that are not adequately represented in predation index samples, but these will not have baseline samples for comparison. A second interaction indicator was spatial overlap. This indicator uses information from status surveys. If the target taxon does not occur within the range of a NTT, then interactions are considered unlikely. If the target taxon expands its range into that of the NTT, the likelihood of interaction increases, and more intensive monitoring may be needed.

Where possible, modeling of the influence of external factors on temporal variation in abundance or size was used to reduce unexplained variation in baseline status. Methods are described in Ham and Pearsons (2000). The models were empirical models of the influence of environmental factors, such as stream flows and water temperatures, on variation in annual estimates of the status of a taxon. These models were applied to abundance and size, but not to distribution. If few baseline surveys were conducted or environmental factors were not well characterized during the baseline period, no model was created for that NTT.

Preparing to detect impacts

The process of evaluating and selecting impact containment strategies is iterative, and progresses as information increases throughout the baseline period (Figure 1). If neither risk nor uncertainty warrant monitoring, incidental monitoring provides minimal, low-cost impact containment for the NTT. If risk or uncertainty warrant monitoring, the feasibility of status or interaction surveys must be evaluated. If neither is feasible, it may be possible to conduct experiments that more fully assess the risk of an impact exceeding the CO. If the results of experiments indicate with sufficient certainty that the impact is likely to always be less than the CO, then incidental monitoring will again be sufficient. If experiments do not indicate that impacts are unlikely to exceed the CO, then policy makers will have to decide whether the level of risk is acceptable, in spite of the lack of detection and containment options. If the risk is acceptable, incidental monitoring will be sufficient. If the risk is not acceptable, managers may find a way to alter activities to minimize or mitigate impacts to NTT while, ideally, maintaining the benefits of the program.

When it is feasible to quantify status or an interaction, a series of baseline surveys establishes the variation among years. If variation is low enough, detection of the desired effect at the desired power level within the time available for containment is possible and a plan can be designed to detect impacts less than or equal to the CO. If power is lacking but time and resources are available, better sampling or an explanatory model may help reduce unexplained variation and increase power. If sufficient power results, it is possible to design a plan that will have a high probability of detecting impacts equal to the CO. When time or resources become limiting and sufficient power has not been achieved, it may be useful to evaluate whether impacts might be detected at a level only slightly greater than the CO. If detection is possible at a slightly higher impact level, it will be necessary to make a policy decision whether the marginal risk is acceptable. If so, a plan can be designed to detect impacts at some level greater than the CO that still provides for some containment. If the risk is not acceptable, or if detection is far short of the CO, it will be necessary to employ experiments to attempt to reduce uncertainty and assess risk. As we stated above, if the results of experiments indicate with sufficient certainty that the impact is likely to always be less than the CO, then incidental monitoring will again be sufficient. If experiments do not indicate impacts are unlikely to exceed the CO, then proceed as above by deciding whether the level of risk is acceptable with only incidental monitoring or, if the risk is not acceptable, find a way to alter activities to minimize or mitigate impacts to NTT.

There are several possible outcomes, providing more or less impact containment by balancing acceptable level of risk or uncertainty and the rapidity and sensitivity of impact detection that can be achieved with available time and resources. In the first case, only incidental monitoring is required due to an acceptable level of risk and uncertainty and little effort is expended to detect or contain impacts. In the second case, monitoring achieves adequate power to provide impact containment consistent with the CO with little risk to NTT. Third, monitoring falls short of containment at CO, but is still the best option for containing impacts. This outcome results in unavoidable, but acceptable, risks to NTT. Fourth, risks and uncertainties cannot be resolved, and risks are deemed unacceptable and the most effective impact containment option is to alter activities or to mitigate impacts such as by predator management or habitat enhancement. The diagram (Figure 1) is simplified to show the process of determining what is detectable for a single interaction type and a single monitoring strategy, and it may be necessary to work through the diagram independently for each interaction type and each monitoring strategy. By following the diagram for all potential interaction types and monitoring strategies for each NTT, each containment option can be evaluated relative to the CO and relative to other strategies.

For taxa that are difficult to survey due to their low numbers or life-history traits, we sought more easily monitored taxa that were similar in their susceptibility to impact. The best example of this is for steelhead, which are difficult to enumerate because they are at such low densities at the time of these surveys. Rainbow trout, a different life-history strategy of the same species, are much more abundant and easily monitored. Because these two groups are spatially intermingled during the time of greatest potential for impacts from spring chinook salmon supplementation and are ecologically very similar, the occurrence of impacts to rainbow trout should provide a good indication of impacts to steelhead (McMichael and Pearsons 1998). Detectable impacts were reevaluated by using the decision framework (Figure 1) for the analog taxa within limits imposed by the life history traits or status of the non-target taxa that it is intended to represent.

After evaluating individual strategies for their ability to detect impacts, an impact detection plan is devised by combining the most effective detection strategies for each NTT. These strategies are chosen to address each interaction type of importance to the NTT. Each plan has a primary detection strategy that provides detection across most important interaction types and for the greatest number of characteristics. Secondary strategies provide an additional benefit for a subset of interactions or characteristics, or may provide a backup where the primary strategy is uncertain or limited in time or space. Additional strategies are included without requiring any real-time detection ability, because they are intended to provide additional information that may address risks or uncertainties over longer time scales. This combination of strategies constitutes an impact detection plan. These plans have detection ability equal to that of the best strategies included.

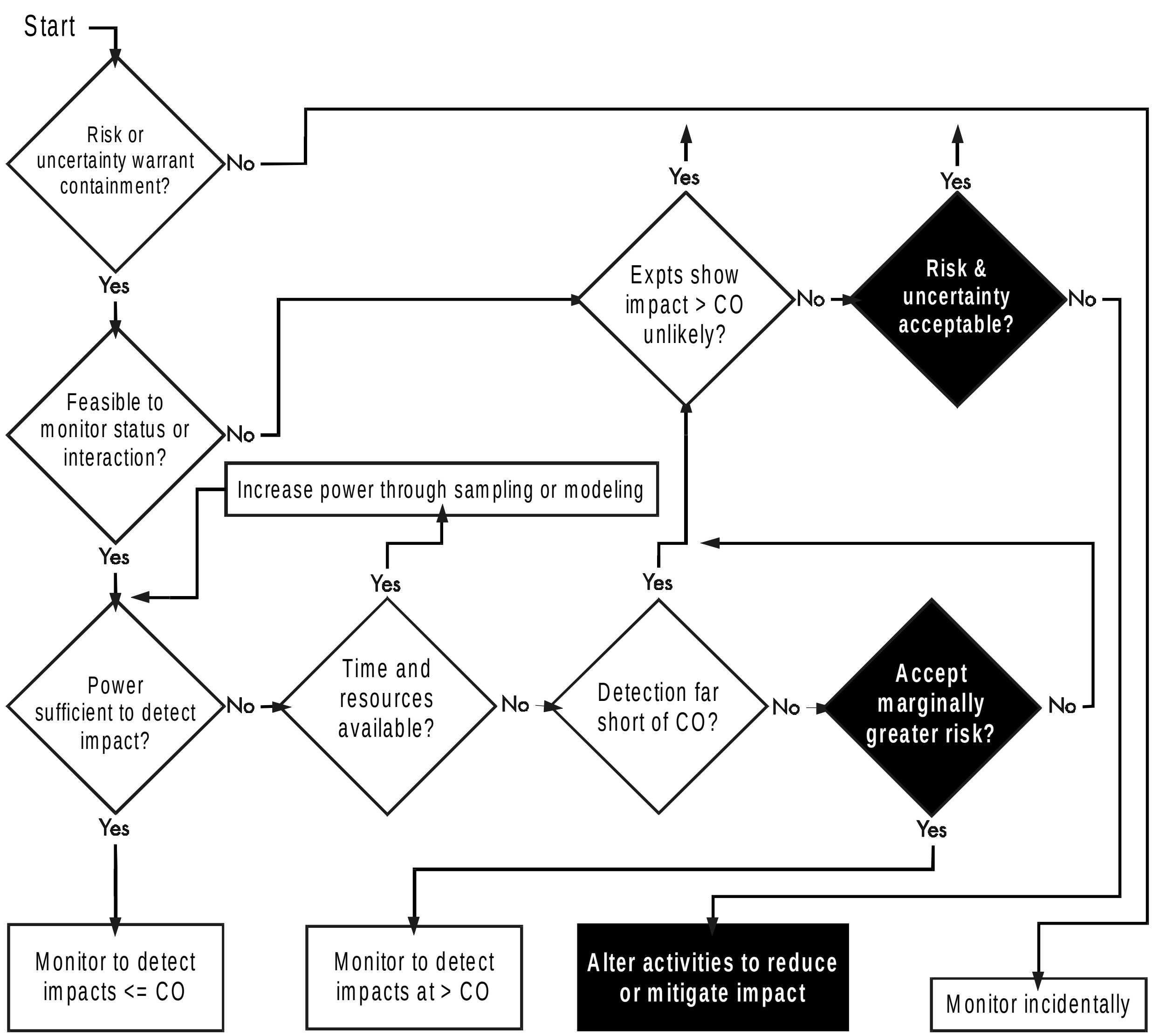

Figure 1. Decision framework for evaluating impact detection strategies. Diagram proceeds from the symbol in the upper left corner. Black symbols indicate decisions that may require a change in policy.

Results

Distribution and size structure revealed a lower degree of temporal variation (Table 2) than was found for abundance surveys reported in Ham and Pearsons (2000). Gaps are evident in the data that limit our ability to establish a baseline condition against which impacts to distribution or size structure can be compared. Distribution was not surveyed for anadromous populations, but other gaps are a result of either a lack of resources, low or changing priority of the NTT, or an inability to monitor certain characteristics of rare populations. The coefficient of variation (CV) for size should be considered underestimates of variability because fork length of fish does not begin at 0, but rather at the length at which scales originate. In evaluating detectable impacts, lower bounds of fish size were estimated. These lower bounds then act as the "true" zero and have the effect of reducing the mean. This in turn would increase the CV used in computing detectable impacts. CV's for distribution are not fully representative of the variation in distribution possible for these NTT populations, because survey sites do not evenly cover the potential range of the population. Little effort was made to correct for this limitation, but detectable impacts were restricted to be no lower than the smallest index unit on a proportional basis.

Table 2. Baseline annual survey results for NTT distribution and size structure. Distribution was not recorded for anadromous NTT.

NTTOC	Characteristic	units	Year 88	89	90	91	92	93	94	95	96	97	98	n	Mean	cv
Bull trout	Distribution	% occupied							100			40	40	3	60	58
	Size														nd	
Cutthroat trout	Distribution	% occupied										43	100	2	71	56
	Size	Fl			167	182	131	131	146	144	171	161	156	9	154	11
Pacific lamprey	Distribution	% occupied													na	
	Size														nd	
Steelhead	Distribution	% occupied													na	
	Size	Fl	184			171	183		126			200		5	173	16
Fall chinook salmon	Distribution	% occupied													na	
	Size	Fl				89	83	79	84	86	91	81		7	85	5
Leopard dace	Distribution	% occupied													nd	
	Size														nd	
Mountain sucker	Distribution	% occupied													nd	
	Size														nd	
Sand roller	Distribution	% occupied													nd	
	Size														nd	
Rainbow trout – mainstem	Distribution	% occupied				100	100	100	100	100	100	100	100	8	100	0
	Size	Fl			259	253	250	255	258	250	249	250	259	9	254	2
Spring chinook salmon	Distribution	% occupied													na	
	Size	Fl				122	128	126	130	133	126	129		7	128	3
Mountain whitefish	Distribution	% occupied							100	100	100	100	100	5	100	0
	Size	adults/juv						41	40	38	40	11	13	6	31	47
Rainbow trout – tributary	Distribution	% occupied					100	94	92	100	93	90	96	7	95	4
	Size	Fl			140	131	132	128	127	135	130	131	137	9	132	3
Longnose dace	Distribution	% occupied					82	72	78	90	59	84	84	7	79	13
	Size	Fl						7	5	8	7	9	9	6	8	19
Speckled dace	Distribution	% occupied						95	100	100	89	95	58	6	89	18
	Size	Fl						3	2	3	3	3	4	6	3	15
Sculpins	Distribution	% occupied						90	90	100	86	87	93	6	91	6
	Size	Fl						6	5	5	6	6	8	6	6	16
Suckers	Distribution	% occupied							100	100	100	100	100	5	100	0
	Size	% Adults						65	51	46	44	36	27	6	45	28

nd = insufficient data. na = not applicable.

Status monitoring alone was adequate for containing impacts within CO for those NTT that had CO of greater than 40% (Figure 2 and Ham and Pearsons 2000). Status monitoring was sometimes adequate for NTT with a CO of 10% to 40%, but wasn't adequate for all characteristics. Status monitoring was clearly inadequate for all NTT with CO of 5% or less. Using analog taxa for detecting impacts to status greatly improved estimated detection ability for steelhead and mountain sucker (Figure 3), but did not result in estimated detectable impacts at or below the CO for these taxa.

Interactions monitoring, of spatial overlap with the target species, provided greatly improved impact detection for bull trout and cutthroat trout (Figure 3). We have estimated detectable impacts for bull trout and cutthroat trout at 5%, which is consistent with the quality of our distribution surveys. Smaller impacts are potentially detectable if target taxa distribution increases are not rapid. The predation index provided a large gain for detecting impacts to fall chinook salmon abundance and a marginal gain for detecting impacts to Pacific lamprey abundance, relative to status monitoring alone. Using the predation index and an analog taxon for leopard dace greatly improved the detection ability for interactions monitoring of impacts to abundance. Low numbers of sand roller or its analogs encountered in predation index sampling make it difficult to demonstrate any improvement over status monitoring, but this strategy is

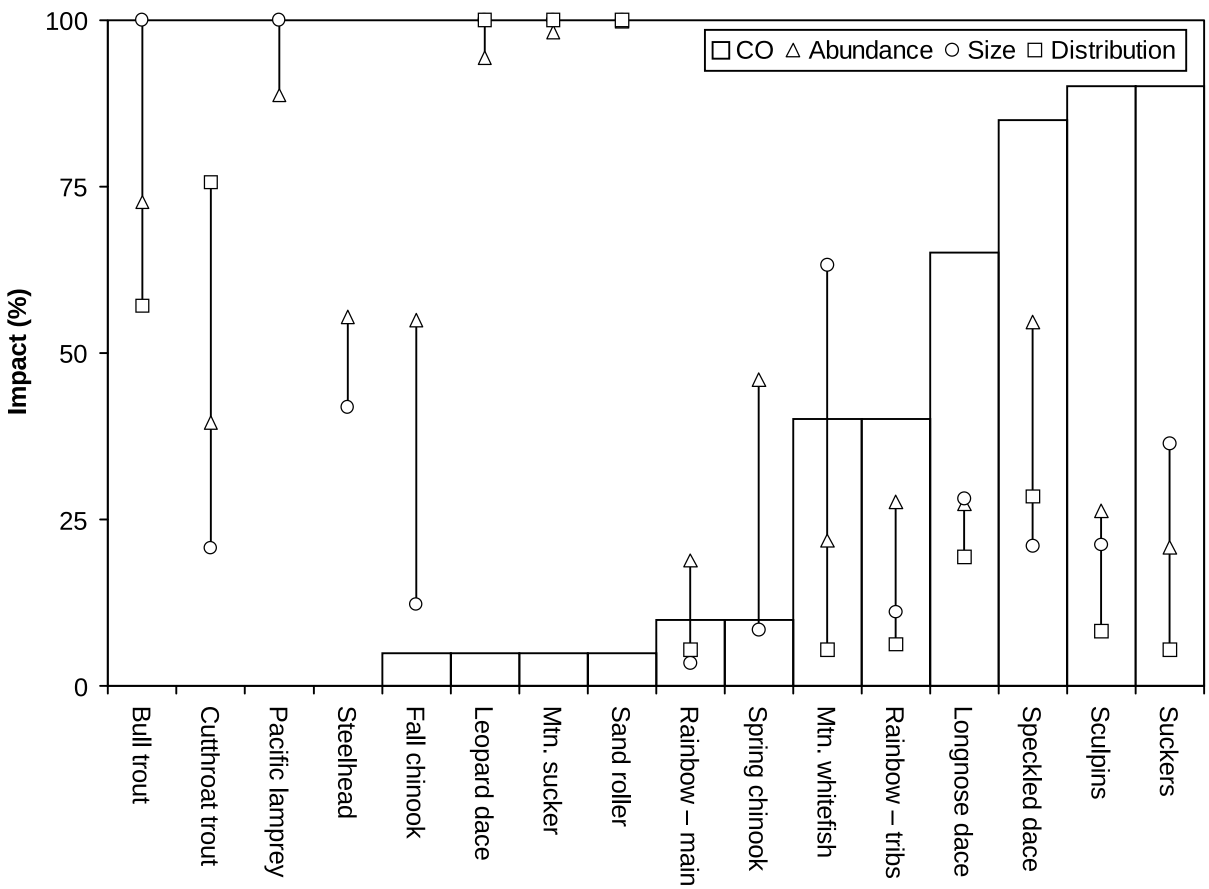

Figure 2. Detectable impacts to non-target taxa using status monitoring alone.

considerably less costly than status monitoring for this NTT. Interactions monitoring did not result in an estimated detectable impact equal to or less than the CO for any NTT where status monitoring alone was inadequate.

Table 3 details impact detection plans that were formed from a combination of detection strategies. Status monitoring was the most commonly used primary containment strategy (69%), with interactions monitoring making up the remainder (31%). Only one NTT, sand roller, had no status monitoring included in the impact detection plan. Two types of interactions monitoring were included as primary or secondary strategies in detection plans, spatial overlap and a predation index. Monitoring spatial overlap promises to greatly improve impact detection for bull trout and cutthroat trout, relative to status monitoring. The predation index was included as a primary strategy for Pacific lamprey and sand roller, and as a secondary strategy for several other taxa. The estimated detectable impacts for these plans are evident in Figure 2 or 3, depending upon the inclusion of interactions monitoring and the use of analog taxa. The lesser detectable impact for each of abundance, size, and distribution represent the prospective power for impact

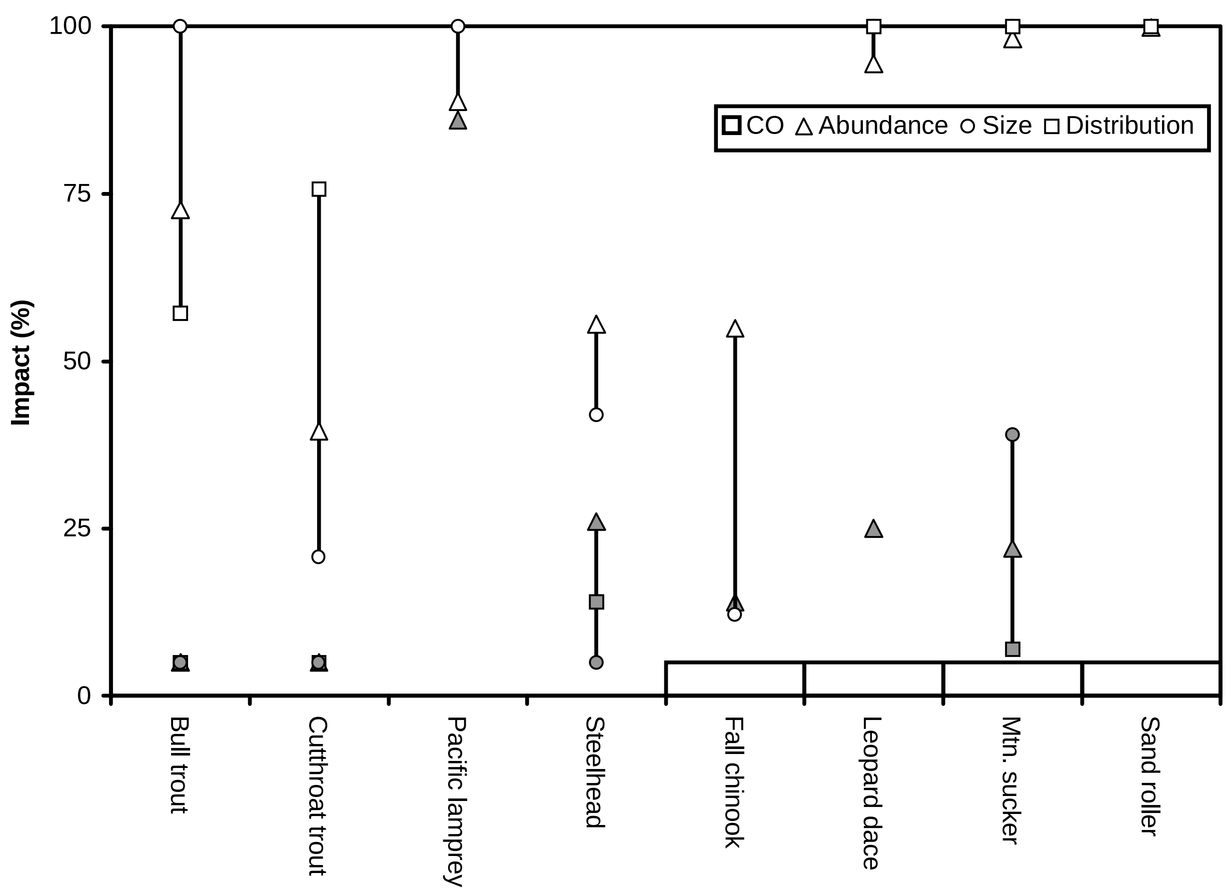

Figure 3. Detectable impacts including using analog taxa and interactions monitoring. Open symbols indicate detectable impacts for status monitoring. Shaded symbols indicate detectable impacts using either analog taxa or interactions monitoring, or both.

detection plans containing multiple strategies. Though many strategies were included in the additional categories to illustrate the breadth of strategies that are being pursued, these are not intended to provide real-time detection ability, and are most useful for longer-term evaluation of impacts.

Impact detection plans combining status and interactions monitoring sometimes achieved

Table 3. Impact detection plans for NTT in the Yakima basin.

	Impact detection strategy		
NTT	Primary	Secondary	Additional
Bull trout	Spring chinook salmon spatial overlap	Status	Status: Redd surveys; Incidental monitoring
Cutthroat trout	Spring chinook salmon spatial overlap	Status	Incidental monitoring
Pacific Lamprey	Predation index (Fall chinook salmon as analog)	Status: juvenile counts	Status: Adult counts
Steelhead	Status: (small rainbow trout as analogs)	Status: smolt counts	Status: Redd surveys; Predation index; Pied-piper index
Fall chinook salmon	Status	Predation index	Status: redd surveys
Leopard dace	Predation index with all dace as analogs		Status: Longnose dace as analogs
Mountain sucker	Status: all suckers as analogs	Predation index with all suckers as analogs	Incidental monitoring
Sand roller	Predation index (sand roller or chiselmouth <100 mm analogs)		Incidental monitoring
Rainbow trout-mainstem	Status		
Spring chinook salmon	Status	Predation index, treatment-reference comparison of smolts per spawner	Status: stock specific redd surveys
Mountain whitefish	Status		
Rainbow trout – tributaries	Status		
Longnose dace	Status		Predation index
Speckled dace	Status		Predation index
Sculpins	Status		Predation index
Suckers	Status		Predation index
Other native species			Incidental monitoring

much lower estimated detectable impacts for an NTT, relative to either status or interactions monitoring alone. These improvements in detection and containment still fell short of the CO for rare and important species at greatest risk of exceeding CO. The impact detection that can be achieved often falls short of the CO for an NTT, but may allow impacts to be contained at a level above CO, or at a lower level over many years. This detection ability is inadequate for containing impacts within the CO, but may be useful for containing ecological impacts at a higher level, as long as the increased risk is acceptable. This limited detection ability may also be useful in demonstrating that even greater impacts did not occur.

Discussion

Populations of native fishes at greatest risk of exceeding CO could not be monitored well enough to provide the feedback necessary to assure that irreversible impacts do not occur. The critical feedback required for adaptively managing actions while containing potential impacts is difficult to achieve, even with exemplary monitoring effort. Waples (1999) stated that simply monitoring genetic impacts of hatchery programs does not result in ability to quickly and surgically intervene to correct undesirable outcomes. Our results show that the same is true for ecological impacts, that monitoring does not guarantee impacts can be contained within acceptable, or even reversible, bounds for taxa at greatest risk. We find that the monitoring is most effective at providing the feedback required for containing impacts through adaptive management where it is needed least, for NTT that are at low risk and uncertainty and with high acceptable impacts. Where it is needed most, for NTT with relatively high risk or uncertainty and with low acceptable impacts, the feedback from monitoring is likely to be inadequate to assure that ecological impacts are contained within acceptable and reversible limits. These results indicate that adaptively managing potential ecological impacts to NTT will rarely be able to eliminate the potential for unacceptable impacts, and even that may only be in trivial cases where risks are already acceptably low. There is a danger that the monitoring and evaluation required by the adaptive management approach will be seen as a substitute for thorough risk assessment. Instead, managers should recognize the process as a tool for balancing of risks and benefits of actions through monitoring and feedback, not as a tool for eliminating risks. To achieve the required balance of risks and benefits, managers must choose between taking a risk averse approach, assessing risks more adequately prior to a management action, accepting risks to non-target populations, or a combination of these approaches.

Detecting impacts to a fish population is challenging, even if baseline data are plentiful (Van Winkle et al. 1981; Vaughan and Van Winkle 1982; Peterman and Bradford 1987, Ham and Pearsons 2000). We do not claim that we have achieved the maximum potential for impact detection for all 16 NTT in this study. Because impact containment priorities continued to develop during our baseline period, resource distribution among NTT does not entirely reflect current values, risks, and uncertainties. While greater resources could benefit nearly any monitoring program, few NTT in the present study might improve from inadequate containment to marginal or adequate containment through increased effort. Of taxa with CO > 5%, the best potential for improvement is for mountain whitefish size distribution to be measured more precisely. For taxa with CO of 5% or less, effort is not lacking in surveying anadromous salmon, though there is hope that environmental influences on abundance can be modeled and that stocks can be separated for better detection of localized impacts. Pacific lamprey and sand roller have the most potential for improvement, but it is hard to imagine how low numbers and difficulty in sampling could be practically overcome. The potential for interactions monitoring of sand roller or Pacific lamprey to greatly improve detectable impacts is also remote, because many of the characteristics that thwart status monitoring also would apply to interactions monitoring. It is possible to suppose that monitoring trends or other untested strategies could achieve better detection of impacts than we predict for the strategies we have evaluated. Other studies have demonstrated that it is more difficult to detect change without baseline information (Gerrodette 1987). In some cases, our detectable impacts may not approach the ideal, but they represent a range of containment potential that is likely to be possible wherever multiple, dissimilar species are of interest.

Although impact detection failed to match the CO for most NTT, impacts of 50% would be detectable for nearly all taxa. Many types of impacts could be detected if they exceed 25% for many NTT. A fundamental objective of management, adaptive or otherwise, is to avoid irreversible harm to native species (Washington Department of Fish and Wildlife 1997). Where containment cannot protect against impacts exceeding the CO, it may be useful to ask whether containment efforts would prevent irreversible loss of the population or its value. This fundamental objective is met for most taxa, with the exception of sand roller, Pacific lamprey, and steelhead, fall chinook salmon, and leopard dace. Risks to sand roller, fall chinook salmon, or leopard dace are not excessive, but Pacific lamprey and steelhead are the two taxa at greatest risk. Impacts to these taxa cannot be effectively contained through monitoring and feedback. Containing impacts to these taxa will require risk-averse actions that do not rely on feedback from monitoring to determine their effectiveness.

Impact detection plans are just one element of effective adaptive management. Adaptive management is an iterative cycle of setting objectives, monitoring, evaluation, and decision making that repeats until critical uncertainties are resolved (Figure 4). Many projects are adopting an adaptive management approach, but examples of successful application of this approach are limited. It is easy to agree to change management actions to adapt as information accrues or conditions change. It is difficult to assure unacceptable risks or impacts are not imposed before adaptive measures can correct the problem. The danger is that the adaptive approach, carelessly applied, could allow unacceptable risks or impacts to develop with no hope of detecting and correcting them.

For effective application of the adaptive management approach, it is necessary to complete each step in the cycle (Figure 4) with a rigor that we have shown, in part, will be difficult to achieve. First, quantitative objectives that reflect stakeholder values must be set. Objectives can change as new information becomes available, as the system being managed changes, or as stakeholder values change. Once objectives have been set, critical uncertainties in meeting the objectives are identified. Monitoring is then needed to evaluate whether objectives are being met and to resolve critical uncertainties. Monitoring will be effective only within a rigorous experimental design that focuses effort on critical uncertainties and assures appropriate biological indicators are selected for the objectives. Biological indicators must provide sufficiently rapid and sensitive detection of changes before they become unacceptable or irreversible or corrective action will not be taken when needed.

Regular evaluation is needed to direct adaptation. Monitoring results must be evaluated regularly to assess whether objectives are being met. Actions must be evaluated to assure that they are being implemented as designed and that the desired outcomes are being produced. Evaluation should also determine whether any critical uncertainties have been resolved. Decision-making follows evaluation. There may be a decision to modify management actions if objectives are not being met, if the state of the system has changed, if stakeholder values have changed, or if a critical uncertainty has been resolved. These decisions are the means of adapting management actions to increase the probability of achieving project goals. Under constant conditions, or while waiting for uncertainties to be addressed, management actions may continue unaltered through more than one management cycle. After decisions have been made and the management actions have been adapted to any new information or changes that arose, the cycle begins again with setting of objectives.

If all critical uncertainties have been resolved, it is possible to make decisions that can meet project goals and objectives with minimal need for monitoring and evaluation. This underscores an important distinction between adaptively managing an ecosystem and adaptively managing a project within that ecosystem. Adaptive management applied to an ecosystem includes many uncertainties that are unlikely to be resolved within a practical period. Adaptive management applied to a project addresses a more limited number and scope of critical uncertainties associated with project goals. If these critical uncertainties can be resolved within the life of the project, the need for adaptive management ends, because uncertainty no longer limits the achievement of project goals. Decisions can be made that impose a known, acceptable risk. While it is not wise to think that all uncertainty can be addressed, it is likely that consistently successful actions will be accepted as standard practice, and will then be applied with little effort toward adaptive management.

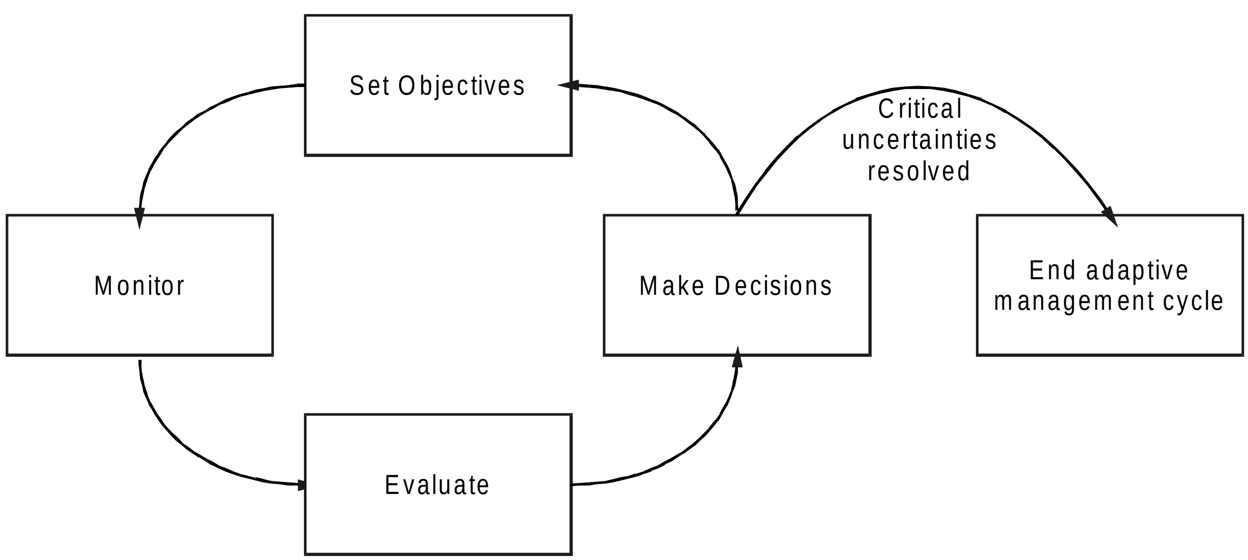

Figure 4. Adaptive management cycle for specific management actions. Diagram proceeds from the uppermost symbol, with annual cycles until adaptive management ends.

Conclusions

Critical to answering the question "Can native fishes be monitored well enough to support adaptive management of ecological impacts?" is determining whether deferring risk assessment until after a management action has begun poses unacceptable risks. Our results show that potentially irreversible impacts have a high likelihood of going undetected for rare or highly valued NTT at high risk of exceeding CO. We feel that monitoring will rarely result in detection and containment of impacts adequate to protect rare and important taxa from unacceptable risks. If a management action poses a risk of unacceptable impact to rare or important NTT, it would be irresponsible to assume adequate containment is possible unless that can be demonstrated through preliminary surveys. Adaptive management is only as good as the weakest function in the cycle (Figure 4), and will not be effective without full awareness of the risk of unacceptable impacts and associated uncertainties. Risks may sometimes be especially difficult to assess prior to an action, but inadequate effort at this stage will place a greater burden on other mechanisms for reducing uncertainty and containing risks, potentially increasing their cost and reducing their effectiveness. Managers have always had a responsibility to balance the risks and benefits of management actions. It is important to recognize that the adaptive management approach does not remove the responsibility for identifying and balancing risks and benefits but, when properly applied, can help meet that responsibility.

Acknowledgments

Anadromous fish information was provided by the Yakama Indian Nation Fisheries Program: Mark Johnston and his crew provided the lamprey data and Bruce Watson and Joel Hubble provided the data on anadromous salmonids. Many members of the Washington Department of Fish and Wildlife's Ecological Interactions Team were instrumental in the collection of data on the other species and in discussions about monitoring. Of particular note are Eric Bartrand, Anthony Fritts, and Geoff McMichael. Kenny Rose, and Craig Busack provided helpful reviews of the draft manuscript. Steve Leider, Bill Hopley, and Keith Wolf provided the technical/policy interface that allowed us to stay focussed and funded. We are indebted to Tom Vogel and David Byrnes of the Bonneville Power Administration who helped secure long-term funding for the Yakima Species Interactions Studies that made this work possible.

References

Bonneville Power Administration. 1996. Yakima Fisheries Project Final Environmental Impact Statement. Bonneville Power Administration, Portland, Oreg. DOE/EIS-0169.

Busack, C., B. Watson, T. Pearsons, C. Knudsen, S. Phelps, and M. Johnston. 1997. Yakima fisheries project spring chinook supplementation monitoring plan. Bonneville Power Administration, Portland, Oreg. DOE/BP-64878-1.

Close, D.A., M. Fitzpatrick, H. Li, B. Parker, D. Hatch, and G. James. 1995. Status report of the Pacific lamprey (*Lampetra tridentata*) in the Columbia River Basin. Bonneville Power Administration, Portland Oregon. DOE/BP-39067-1.

Clune, T., and D. Dauble. 1991. The Yakima/Klickitat fisheries project: a strategy for supplementation of anadromous salmonids. Fisheries **16(5)**: 28-34.

Fast, D.E., and C. Craig. 1997. Innovative hatchery project working to rebuild wild salmon populations. Hydro Review **14**: 30-33.

Gerrodette, T. 1987. A power analysis for detecting trends. Ecology 68(5):1364-1372.

Ham, K.D. and T.N. Pearsons. 2000. Can reduced salmonid abundance be detected in time to limit management impacts? Canadian Journal of Fisheries and Aquatic Sciences **57**: 1-8.

Hunn, E.S. 1990. Nch'i-Wana "The Big River" Mid-Columbia Indians and their land. University of Washington Press, Seattle and London.

Krause, T. 1991. The Yakima. Fly Fisherman **22**: 40-43,76-78.

McMichael G.A., A.L. Fritts, T.N. Perasons, and J.L. Dunnigan. 1999. Lower Yakima River predatory fish monitoring: Progress report 1988. Pages 93-124 *In* ODFW and NMFS. 1999. Management Implications of Co-occuring Native and Introduced Fishes: Proceedings of the Workshop. October 27-28, 1998, Portland, Oregon. 243 pgs. Available from: National Marine Fisheries Service, 525 N.E. Oregon St., Suite 510, Portland, OR 97232.

McMichael, G.A., and T.N. Perasons. 1998. Effects of wild juvenile spring chinook salmon on growth and abundance of wild rainbow trout. Transactions of the American Fisheries Society **127**: 261-274.

National Marine Fisheries Service. 1999. Endangered and Threatened Species: Threatened Status for Two ESUs of Steelhead in Washington and Oregon. Federal Register [Docket No. 980225046–9070–03; I.D. 021098B] 64(57) Rules and Regulations14517-14528

Patten, B.H., R.B. Thompson, and W.D. Gronlund. 1970. Distribution and abundance of fish in the Yakima River, Washington, April 1957 to May 1958. U. S. Dept. Int. Fish Wildl. Serv. Fish Bull. No. 603.

Pearsons, T.N. 1998. Draft objectives for non-target taxa relative to supplementation of upper Yakima spring chinook salmon. *In* Yakima River Species Interactions Studies. Progress report 1995-1997. *Prepared by* T.N. Pearsons and five co-authors. Bonneville Power Administration, Portland, Oreg. pp. 4-44.

Pearsons, T.N., and C.W. Hopley. 1999. A practical approach for assessing ecological risks associated with fish stocking programs. Fisheries 24(9): 16-23.

Pearsons, T.N., G.A. McMichael, K.D. Ham, E.L. Bartrand, A.L. Fritts, and C.W. Hopley. 1998. Yakima River Species Interactions Studies. Progress report 1995-1997. Submitted to Bonneville Power Administration, Portland, Oreg.

Peterman, R.M., and M.J. Bradford. 1987. Statistical power of trends in fish abundance. Canadian Journal of Fisheries and Aquatic Sciences **44**: 1879-1889.

Probasco, S. 1994. River Journal, Volume 2; Yakima River. Frank Amato Publications, Incorporated. Portland, Oreg.

Thurow R.F., D.C. Lee, and B.E. Rieman. 1997. Distribution and status of seven native salmonids in the interior Columbia River basin and portions of the Klamath River and Great basins. North American Journal of Fisheries Management **17:** 1094-1110.

U. S. Fish and Wildlife Service. 1998. Endangered and threatened wildlife and plants; determination of threatened status for the Klamath River and Columbia River distinct population segments of bull trout. Federal Register [June 10, 1998] **63**(111): 31647-31674

Van Winkle, W., D.S. Vaughan, L.W. Barnthouse, and B.L. Kirk. 1981. An analysis of the ability to detect reductions in year-class strength of the Hudson River white perch (*Morone americana*) population. Canadian Journal of Fisheries and Aquatic Sciences **38**: 627-632.

Vaughan, D.S. and W.Van Winkle. 1982. Corrected analysis of the ability to detect reductions in year-class strength of the Hudson River white perch (*Morone americana*) population. Canadian Journal of Fisheries and Aquatic Sciences **39**: 782-785.

Walters, C.J. 1986. Adaptive management of renewable resources. MacMillan, New York.

Walters, C.J., and R. Hilborn. 1976. Adaptive control of fishing systems. Journal of the Fisheries Research Board of Canada **33**: 145-159.

Waples, R.S. 1999. Dispelling some myths about hatcheries. Fisheries 24(2):12-21.

Washington Department of Fish and Wildlife. 1997. Wild salmonid policy, draft environmental impact statement. Washington Department of Fish and Wildlife, Olympia, Washington.

Chapter 3

Lower Yakima River Predatory Fish Monitoring:
Progress Report 1998

Geoffrey A. McMichael

Anthony L. Fritts

Todd N. Pearsons

Washington Department of Fish and Wildlife
600 Capitol Way North
Olympia, Washington 98501

and

James L. Dunnigan

Yakama Indian Nation
PO Box 151
Toppenish, Washington 98948

Abstract

We began an effort to examine predation by fish on salmonids in the lower Yakima River in 1997. Based on the findings from 1997, we initiated a program in 1998 to determine predation indices for the three primary fish predators in the lower Yakima River; smallmouth bass, northern pikeminnow, and channel catfish. Bass and pikeminnow were captured primarily by electrofishing. Channel catfish were collected in drifting gill nets, hoop nets, traps, and by electrofishing and angling. Stomach samples were collected during the spring when emigration of spring chinook salmon smolts was estimated to be at its peak (mid-late April), and again during the last quartile (mid-May) of their emigration. Population estimates of smallmouth bass increased between April and May, as did the proportion of larger (> 200 mm) fish. A higher percentage of the bass sampled during May contained salmonids (16.9-33.3%) than during the April sampling (4.2-4.5%). Most of the smallmouth bass predation on salmonids was on fall chinook salmon parr and smolts. Only one spring chinook salmon smolt was found in a smallmouth bass. Smallmouth bass predation indices (PI) on all salmonids (predominantly fall chinook salmon) were five to ten times higher in May than in April. The smallmouth bass PI for spring chinook salmon was seven to 63 times lower than the PI for fall chinook salmon. A large number of smallmouth bass (N=2645) and channel catfish (N=2694) were tagged in 1997 and 1998. 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. We were unable to generate population estimates of northern pikeminnow due to low capture efficiency. Northern pikeminnow rarely consumed salmonids during the April sampling period, but during May, 21-29% of the northern pikeminnow stomachs contained at least one salmonid. During this period northern pikeminnow consumed both yearling and subyearling salmonids. We captured large numbers of channel catfish, and 2.9% of the stomachs examined contained at least one salmonid. One channel catfish contained 76 fall chinook salmon, and several other fish species in its gut. By extrapolating smallmouth bass numbers from the mouth of the Yakima River upstream to Prosser Dam, we estimated that smallmouth bass could consume about 18,840 salmonid smolts in the lower 68 km of the Yakima River daily during the smolt emigration period. Estimates of the number of salmonids consumed by northern pikeminnow above Prosser ranged from 35-390 salmonids/1000 predators/day throughout the emigration period. Predator control options are discussed, with the most promising being a 2 C decrease in water temperature in the lower Yakima River.

Introduction

Predatory fish have been identified as strong interactors that could potentially limit the success of spring chinook salmon *Oncorhynchus tshawytscha* supplementation efforts in the Yakima basin (Busack et al. 1997; Pearsons et al. 1998). 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 (McMichael et al. 1998). 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 large numbers of native and non-native piscivorous fishes in the lower reaches of the river (B. Watson, personal communication). Northern pikeminnow *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 (McMichael et al. 1998).

Proliferation of non-native predators, exacerbated by alterations of the physical environment may have contributed to unnaturally high predation impacts to anadromous salmonids in the Yakima River. Introductions of non-native predatory fishes were done at times when anadromous fish populations were relatively high and the risks of such introductions to salmonids were unknown. Smallmouth bass were introduced into the Yakima River in 1925 by the Benton County Game Commissioners to increase angling opportunities (Lampman 1947). Channel catfish were introduced into the Boise River, Idaho, in 1893 (Lampman 1947) and probably dispersed into the Snake and Columbia rivers and then the Yakima River. Densities of both smallmouth bass and channel catfish are high enough to support popular recreational fisheries in the lower Yakima River.

High concentrations of the native piscivore, northern pikeminnow, have been observed below irrigation dams on the Yakima River during the spring smolt migration period (McMichael et al. 1998). Salmonids that migrate past dams may be particularly susceptible to predation because they are frequently concentrated in small areas and disoriented. In addition, unnaturally high water temperatures, caused by irrigation withdrawals and riparian vegetation removal can also increase digestion rates of predators, resulting in higher consumption. 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 in 1997. Findings from the work conducted in 1997 showed that large numbers of large smallmouth bass migrated from the Columbia River into the Yakima River prior to the emigration of most salmonid smolts. We also observed salmonids in the guts of predatory fishes. Methods were developed that allowed us to capture sufficient quantities of the predatory species to attempt to develop predation indices for the three primary predatory species; smallmouth bass, northern pikeminnow, and channel catfish.

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 overall goal of our study was to calculate predation indices for the main predatory fish species during the peak and the last quartile of spring smolt emigration in the lower Yakima River.

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. During the late spring and summer, 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 76), and Horn Rapids Dam (rkm 28.1)(Figure 1). Summer water levels can be extremely low below Prosser Dam. Water temperatures in the lower Yakima River often exceed the upper lethal limits for salmonids during summer (> 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. macrochirus,* yellow perch *Perca flavescens,* walleye *Stizostedion vitreum,* largemouth bass *M. salmoides*, black crappie *Pomoxis nigromaculatus*, brown bullhead *I. 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 very rare.

The habitat in the lower Yakima River corridor has been influenced by irrigation diversions and bank stabilization. Riparian vegetation is dominated by grasses, dogwoods, willows, 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 and the discharge out of Priest Rapids 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. Grosscup Road to Van Geisen Road bridge (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), and 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. Most of the gill netting of catfish occurred in the lower 5 km of the river, while catfish traps were operated in three reaches between rkm 2.5 and 19.0, between the I-182 and Grosscup Road bridges. The Kennewick traps were operated between rkm 2.5 and 4.1. The Duportail traps were located between rkm 7.1 and 12.0. The traps in the Vangie section were placed between rkm 15.2 and 19.0. Additional northern pikeminnow sites were sampled immediately below Roza Dam (WDFW) and near the outfall of the juvenile bypasses below Sunnyside and Prosser dams (YIN).

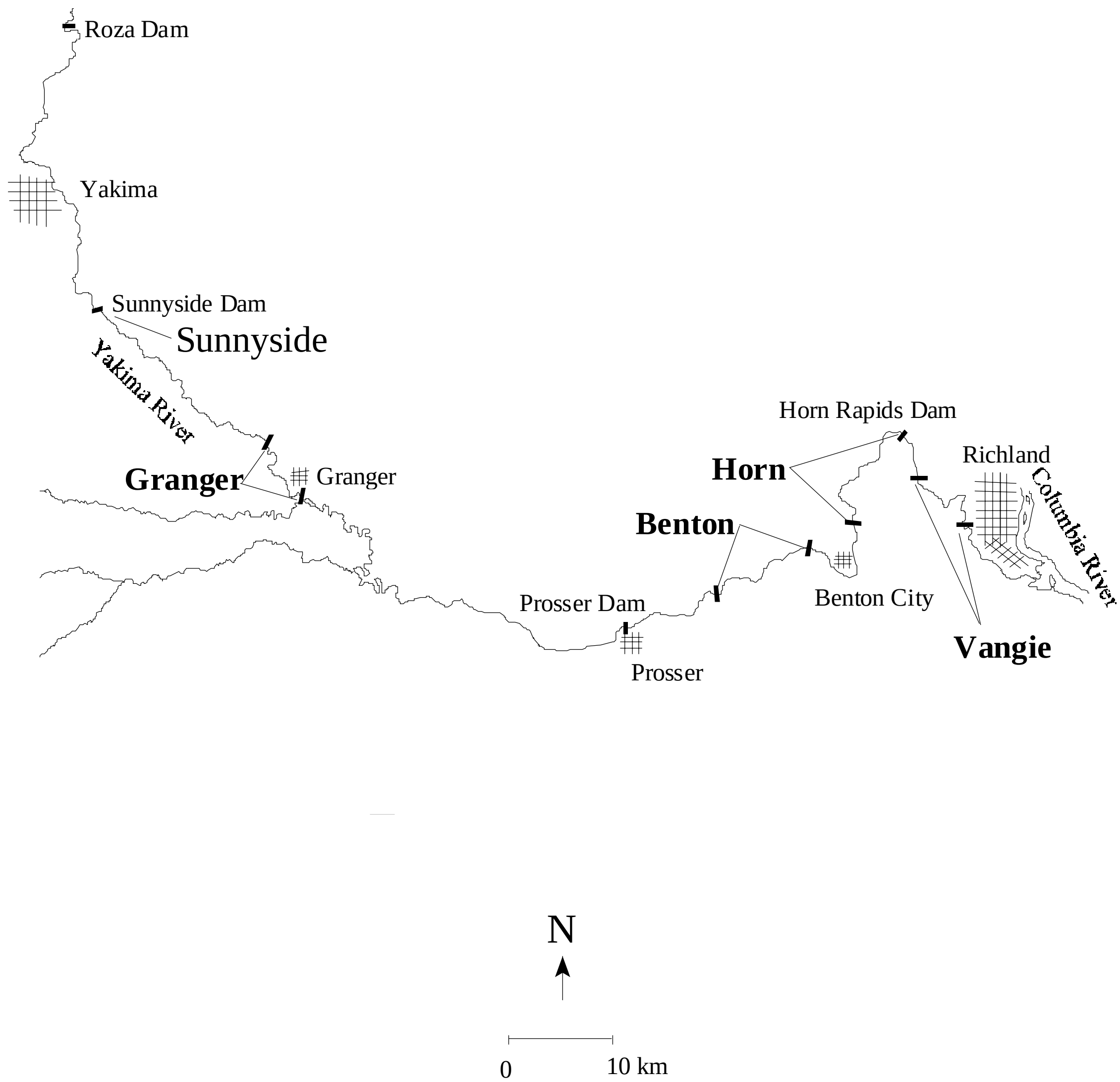

Figure 1. Map of the study area in the lower Yakima River showing index sections in bold type.

Population Estimates/Movement

Mark-recapture population estimates (Vincent 1971) were conducted on smallmouth bass and northern pikeminnow captured by boat electrofishing. Electrofisher settings were about 400 V pulsed DC (PDC; Coffelt's CPS setting) at between 2 and 5 Amps during spring sampling (through June) and 400-500 V PDC at 60Hz and 4-6 Amps during summer. All captured predatory fish over 100 mm FL were marked with a partial fin clip and/or tag (predatory fishes $\geq$ 200 mm) on successive runs down each bank. Recapture runs followed the same sequence 1 day (1998) or 7 days (1997) after the marking runs. Fish were processed every 1 km during both marking and recapture runs. The electrofishing runs were generally along the banks, especially during high water. The driftboat was often operated closer to the bank than the jet boat due to difficulty maneuvering the jet boat in swift water. The species composition was visually assessed and recorded by the netter. An additional 133 smallmouth bass captured by tournament anglers fishing in the Columbia River were tagged September 20-21, 1997, and released in the Columbia River at Richland. Correlations between water temperature and discharge in the Yakima River were examined and compared to water temperatures in the Columbia River to help explain fish movement.

Electrofishing was conducted during the estimated peak (April 20-23) and last quartile (May 11-14) of spring chinook salmon smolt emigration. The peak and last quartile were estimated by examining smolt emigration data collected at the Chandler juvenile fish facility between 1983 and 1996. Additional single electrofishing runs were conducted in early June (2-5) and August (19-20) to obtain additional catch per unit effort, diet, and movement data.

Catfish captured in baited slat traps and hoop nets were marked with anchor tags in three sections of river with 6 traps each (*see* McMichael et al. 1998 for details on traps). Catfish traps were operated from mid-April through June and were located primarily along deep outside bends in the river that contained some wood or rock structure. A jet boat was used to travel the section of river where the traps were located. Catfish traps were baited with rotten cheese and were checked every 24-72 h between April 6 and June 30, 1998.

Diet Samples

Diet samples were collected from northern pikeminnow, smallmouth bass, and channel catfish. Northen pikeminnow and smallmouth bass sampled for diet were collected by drift and jet boat electrofishing. Most channel catfish collected for stomach samples were captured in drifting gill nets. Gill nets were made of 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 (smaller mesh nets were tried and abandoned in favor of the larger mesh). 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. A total of 59 sets averaging 12.4 minutes each were performed between April 7 and May 21, 1998.

Diet samples for smallmouth bass were obtained by gastric lavage (Light et al. 1983). Samples of lavaged bass were sacrificed to validate efficiency of the lavage technique. Digestive tracts were excised from channel catfish and northern pikeminnow. All diet samples were placed in whirl-paks with 10 cc of buffered solution and tagged with date, stomach number, species, length, weight, and the section where the fish was captured and then placed on dry ice. Samples were kept frozen until lab analyses (1 to 3 months).

In the lab, the diet samples were weighed to the nearest 0.1 g, then transferred into a pancreatin solution to digest soft tissues, revealing only bones, and finally placed in various size glass and nalgene containers. The analysis of the contents consisted of placing the contents of a single sample into a petri dish and identifying fish to lowest possible taxonomic classification based on diagnostic bones. For bone identification, a series of keys and sketches produced and provided by the Biological Resources Division station located in Cook, Washington, were used. Standard equations were used to calculate estimated length of each fish in the stomach samples based on dimensions of diagnostic bones (Hansel et al. 1988). Length-weight regressions based on live fish we collected concurrently with the predatory fishes, as well as equations presented by Vigg et al. (1991), were then used to calculate estimated weight of each prey fish at the time of ingestion.

We then used the equation presented by Vigg et al. (1991) to calculate digestion time (*DT*; hours) for smallmouth bass:

$$DT=268.529(E+0.01)^{0.696}S^{-0.363}e^{-0.138T}P^{-0.175} \tag{1}$$

E = prey mass evacuated [amount pumped out of gut](g),
S = prey meal weight [at time of ingestion](g),
T = water temperature (C), and
P = predator weight (g).

For northern pikeminnow we used the equation presented by Vigg et al. (1991) to calculate digestion time (*DT*; hours):

$$DT=1330.753E^{1.081}S^{-0.469}T^{-1.606}P^{-0.273} \tag{2}$$

For channel catfish, we calculated digestion time by the following equation (derived from data presented by Shrable et al. (1969)):

$$DT=4.93525+e^{4.07303-0.02289T-1.535966D} \tag{3}$$

D = % of prey weight digested.

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

$$C=n(24/DT) \tag{4}$$

n = number of salmonids observed in predator's gut, and
DT = digestion time for a salmonid meal (hours) from equations 1 - 3.

Extrapolations

Population estimates of smallmouth bass $\geq$ 150 mm FL (the minimum size found to contain salmonids) were generated by mark-recapture techniques within the Benton and Vangie study sections during the April and May sample periods. To estimate the daily number of salmonids eaten within each study section by smallmouth bass (*SE)* we used the following equation:

(5) $$SE=PE x F x C$$

PE = population estimate of smallmouth bass $\geq$ 150 mm FL within the study section,
F = fraction of smallmouth bass stomachs examined that contained at least one salmonid, and
C = estimated daily consumption rate per predator from equation 4.

To estimate the number of salmonids consumed daily by smallmouth bass in the lower 68 km of the Yakima River (the range of high bass densities) (S_{tot}) we used the following equation:

(6) $$S_{tot}=(PE/SL) x RL x F x C$$

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

Extrapolations were not performed on predation estimates for northern pikeminnow or channel catfish due to the lack of predator abundance data. Prey preference for smallmouth bass was examined by subtracting the percentage of a given prey species observed while electrofishing (availability) from the percentage of that species observed in smallmouth bass guts (use).

Predation Indices

The predation indices were calculated based on modified versions of the equations presented by Ward et al. (1995):

(7) $$PI=AI \times CI$$

PI = predation index,
AI = abundance index, and
CI = consumption index.

We used a modified form of predator abundance that we felt better reflected the abundance of predators in the lower Yakima River and that we also believed was more sensitive to changes in capture efficiency that might be related to differing environmental conditions from year to year. Our measure of abundance was the catch per unit effort (CPUE; fish/minute) divided by the capture probability plus one. In cases where no fish were recaptured (thus no capture probability could be calculated), we used straight CPUE. Our measure of area differed from that used by Ward et al. (1995) also, in that we used a linear measurement of the river reach length that our index site represented multiplied by our CPUE data to arrive at the abundance index (AI) for each river reach:

(8) $$AI_{ij}=D_i x S_j$$

D = CPUE of predatory-sized fish of species *i*, and
S = length of section j in km [the length of river that the index reach data is extrapolated to].

To calculate consumption indices we used *C* from equation 4 multiplied by the fraction of predatory fish that contained at least one salmonid:

(9) $$CI_{ik}=C_{ik} x F_{ik}$$

CI_{ik} = consumption index by predator *i* of prey species *k*,
C_{ik} = consumption rate of prey species *k* by predator *i* (from equation 4), and
F_{ik} = fraction of predator species *i* found to contain at least one of prey species *k*.

Results

Population Estimates

Population estimates were attempted in five river sections in the third week of April, 1998, (peak of spring chinook salmon smolt emigration) and again in the second week of May (last quartile of smolt emigration).

Smallmouth Bass

Populations of smallmouth bass were generally higher during the second sampling period (May) and also in the lower section (Vangie). Data for smallmouth bass in the Benton and Vangie sections are presented in Table 1. Population estimates of smallmouth bass in the Horn section were not possible because no marked fish were recaptured during the recapture period. Future surveys will not use the Horn section due to these low capture efficiencies.

Table 1. Population estimate data for smallmouth bass (SMB) in two sections of the Yakima River. Dates (1998), species/size class (mm FL), estimate, 95% confidence intervals (CI), capture efficiency (Effic.), and validity of the estimate are shown for each river section/date.

Dates	Species/size	Section	Estimate	CI	Effic.	Valid
4/20-21	SMB/≥100	Benton	3528	419-29243	4.7%	yes
4/20-21	SMB/≥150	Benton	1528	221-10838	6.4%	yes
4/20-21	SMB/≥200	Benton	499	98-2970	10.6%	yes
4/22-23	SMB/≥100	Vangie	5092	569-46959	5.8%	yes
4/22-23	SMB/≥150	Vangie	3210	374-25580	6.1%	yes
4/22-23	SMB/≥200	Vangie	2952	152-14511	2.8%	no
5/11-12	SMB/≥100	Benton	5534	578-50024	4.7%	yes
5/11-12	SMB/≥150	Benton	3177	373-25395	5.5%	yes
5/11-12	SMB/≥200	Benton	1313	203-9244	7.9%	yes
5/13-14	SMB/≥100	Vangie	6048	949-76929	9.4%	yes
5/13-14	SMB/≥150	Vangie	4176	701-48307	9.6%	yes
5/13-14	SMB/≥200	Vangie	2490	456-25133	9.7%	yes

Larger bass were caught at a higher rate during May sampling than during the April samples (Figure 2). Figure 3 shows the proportional stock density (PSD) of the smallmouth bass in the Benton and Vangie sections between April and August, 1998.

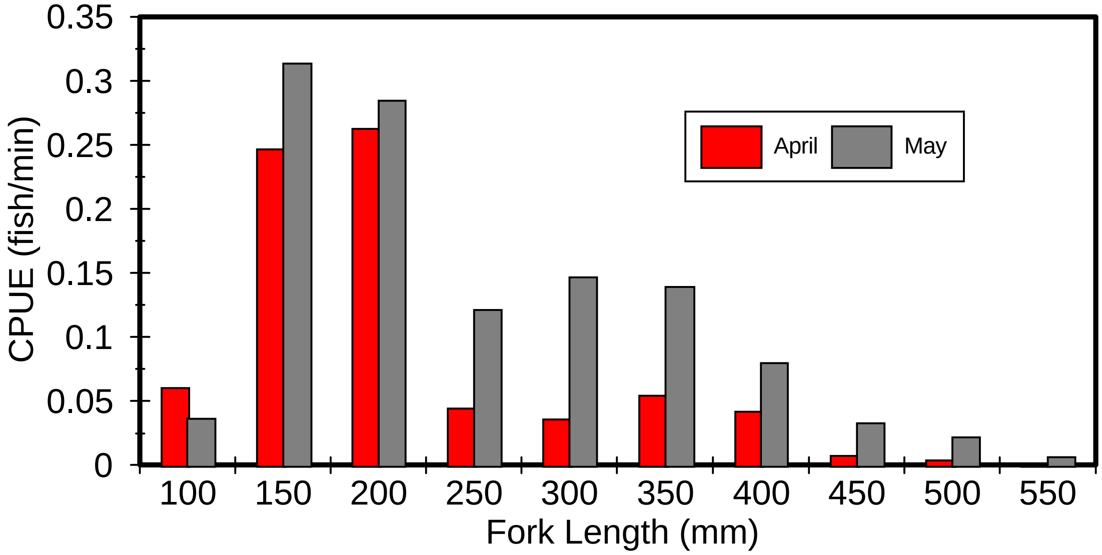

Figure 2. Catch per unit effort (fish/min) of smallmouth bass captured by electrofishing versus length in the Vangie section of the Yakima River in April and May, 1998.

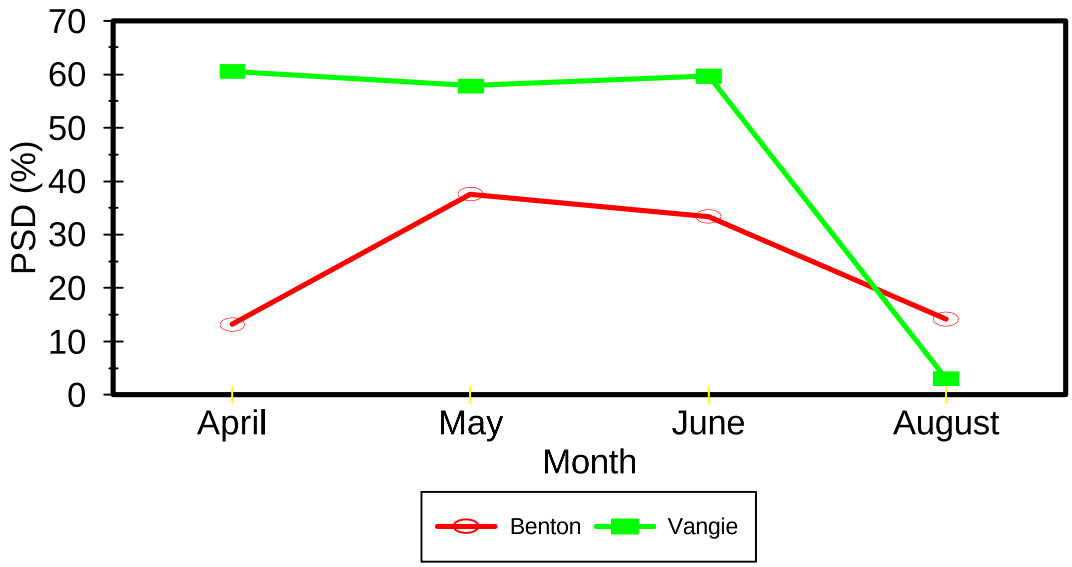

Figure 3. Proportional stock density (PSD; % > 280 mm/> 180 mm) of smallmouth bass captured in the Benton and Vangie sections of the lower Yakima River between April and August, 1998.

The smallmouth bass population estimates in 1997 were generally invalid due to low capture efficiencies related to high waters (McMichael et al. 1998). However, one of the valid estimates for the Vangie section in June, 1997 (6,954 smallmouth bass 100 mm and longer), was similar to the population estimates generated in 1998. Further, in 1997 similar relationships were observed between time period and the number and size structure of smallmouth bass captured.

Northern Pikeminnow

Low catch rates and lack of recaptures for northern pikeminnow prevented us from being able to calculate population estimates for this species in the Horn Rapids and Granger sections. A total of 18 and 57 northern pikeminnows were marked in the Horn Rapids and Granger sections for the season respectively; no marked fish were recaptured. The CPUE (fish > 150 mm fork length per minute) for smallmouth bass at the Horn Rapids section was approximately five times higher than for northern pikeminnow. However, the CPUE for northern pikeminnow at the Granger section was approximately five times higher than the CPUE for northern pikeminnow at the Horn Rapids section. These trends in relative abundance were similar between sampling periods, suggesting that northern pikeminnow abundance is higher in the Yakima River above Prosser than below Prosser throughout the spring chinook salmon smolt emigration period. This observation was also corroborated by visual estimates of the species assemblage for multiple sampling sections throughout the Yakima Basin (see Species Composition Section).

Channel Catfish

Channel catfish were difficult to capture by electrofishing during the spring period; only 2 were captured by electrofishing in 1997 and 27 were captured in 1998. Traps proved much more effective for capturing large numbers of fish, however, the stomach samples from trapped fish were usually unusable due to long periods of holding in traps prior to being removed. Also, the number of recaptures of tagged catfish were very low (Table 2). Channel catfish that were captured in drifting gill nets provided excellent stomach samples, but catch rates were low with this method (43 fish in 1997; 14 fish in 1998). Gill net catches appeared to have been better in 1997 when the river was higher, more turbid, and colder.

The catch of channel catfish in traps generally was higher in the lower traps early in the season, and was higher in the middle and upper section later in the spring (Figure 4), indicating a movement of channel catfish into the Yakima River form the Columbia River or the Yakima River delta. Most of the channel catfish captured in traps were between 200 and 350 mm (Figure 5), similar to the size frequency observed in 1997 (McMichael et al. 1998). Similar to 1997, hoop nets captured a wider size range than the slat traps.

Table 2. Channel catfish trapping data from the lower Yakima River in 1997 and 1998. The number of catfish captured in traps, number of tagged fish recaptured, and population estimates are presented. The length of river (km) where traps were located, and estimated number of channel catfish per km are also presented. Population estimates are crude and are not statistically valid, they are provided for discussion purposes only.

Year	No. Tagged	No. '97 recaps	No. '98 recaps	Pop. Est.	km	CCAT/km
1997	981	15	-	64,157	6.0	10,693
1998	1498	10	12	224,400	15.8	14,203

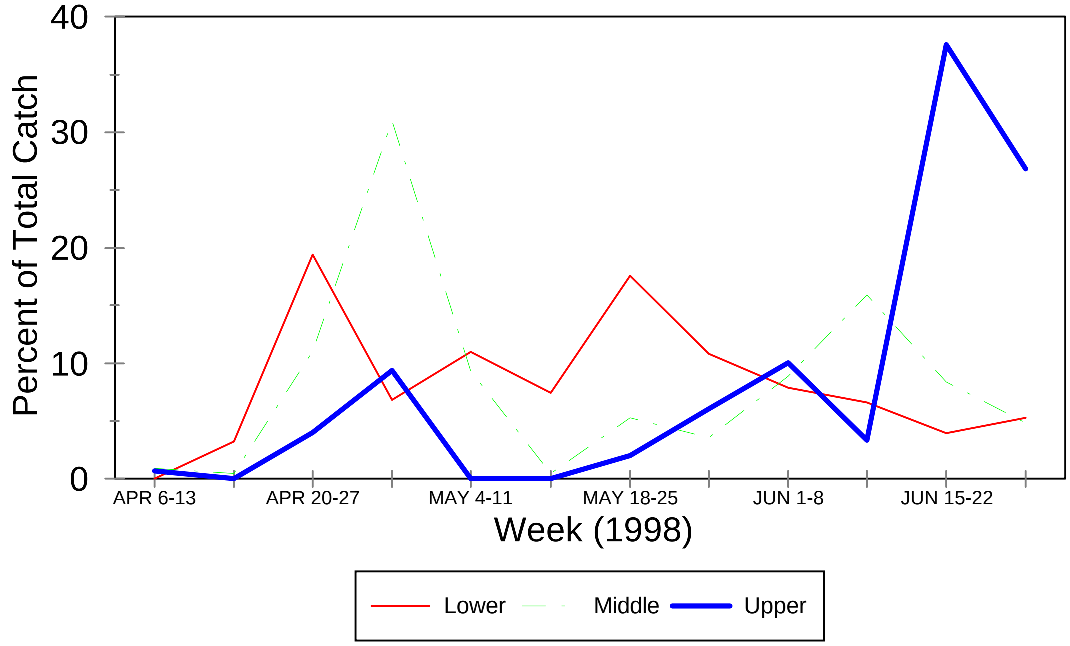

Figure 4. Percent of the total trap catch of channel catfish versus sample week in the lower Yakima River in 1998.

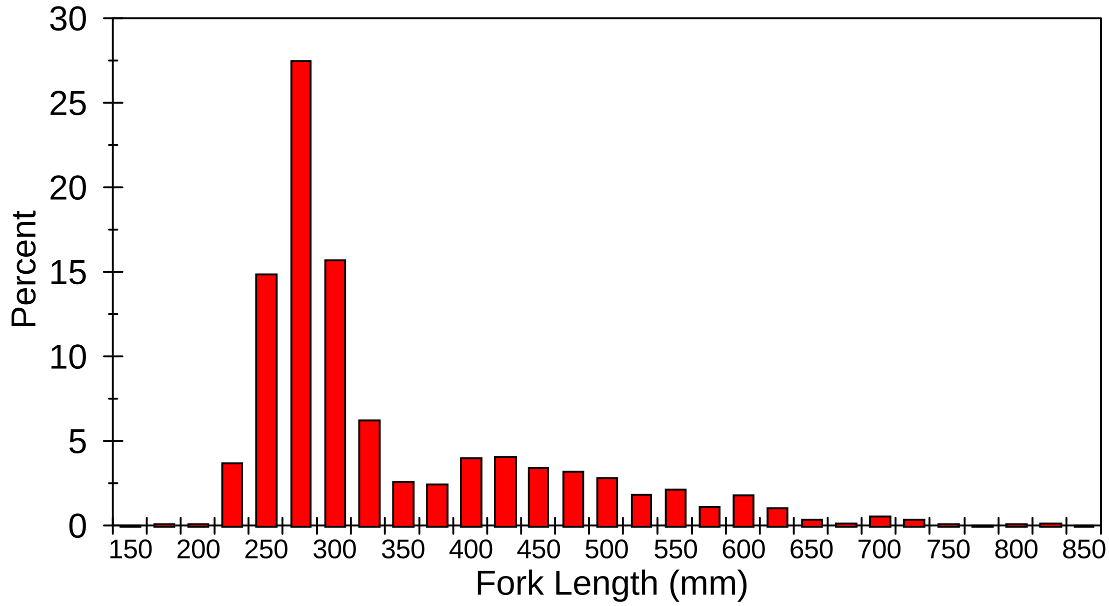

Figure 5. Length frequency of channel catfish captured in traps in the lower Yakima River between April 6 and June 30, 1998. Total sample size was 1,352.

Fish Movement

Tag recaptures and size structure information indicate that smallmouth bass migrated into the Yakima River from the Columbia River during the early spring and migrated out in the summer or fall. Tagged smallmouth bass that have been recaptured by anglers fishing in the Columbia River in the summer and fall indicate that bass tagged in the Yakima River in the spring moved out of the Yakima River in the late spring and early summer. A total of 28 smallmouth bass tagged in the Yakima River (over 1% of the number of fish tagged) have been recaptured by anglers fishing in the Columbia River. Further, five of 133 (3.8%) smallmouth bass tagged in the Columbia River in the fall of 1997 were recaptured in the Yakima River in the spring of 1998. A higher percentage of tagged smallmouth bass captured during spring months were recaptured upstream of the location where they were tagged than during the summer and fall months (Figure 6). Figure 7 illustrates the decrease in the presence of larger smallmouth bass in the Vangie section of the Yakima River between June and August, 1998. The percentage of smallmouth bass in the Vangie section that were greater than or equal to 300 mm FL decreased from 40.7% to 3.3% between June and August, indicating that most large fish present during the spring had emigrated from the section prior to August. This shift in size structure was not due to increased numbers of age-0 smallmouth bass, as we only included fish 100 mm FL or longer.

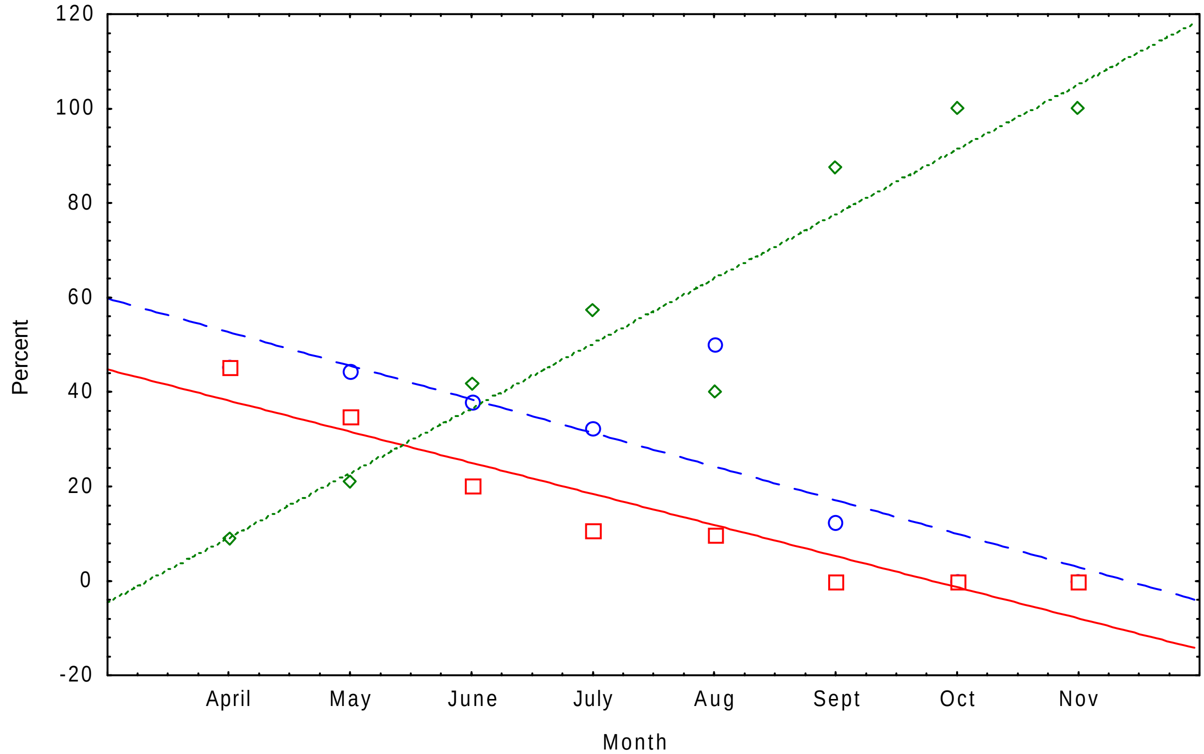

Figure 6. Frequency (percent) of recaptured smallmouth bass moving downstream (dotted line and diamonds), upstream (dashed line and circles) and showing no movement (solid line and squares) in the lower Yakima River in 1997 and 1998 combined. Only smallmouth bass recaptured between 2 and 250 days after tagging were used.

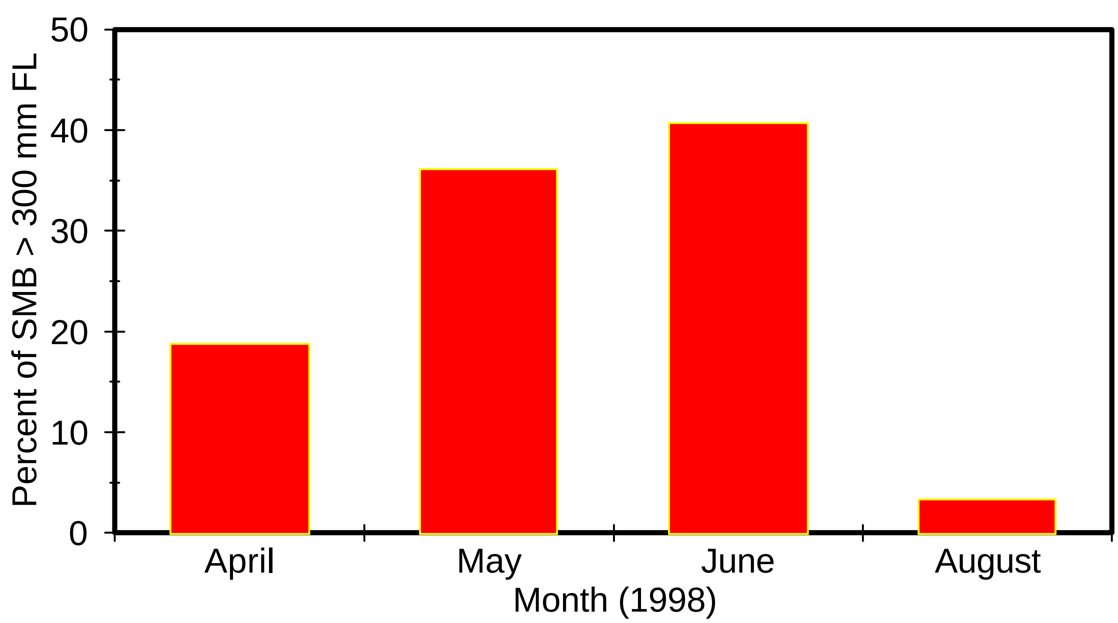

Figure 7. Percent of smallmouth bass captured by electrofishing in the Vangie section of the Yakima River that were greater than or equal to 300 mm FL versus month in 1998.

Warmer water and less drastic diel fluctuations in discharge in the Yakima River than in the Columbia River may be two reasons why adult smallmouth bass move into the Yakima River from the Columbia River in the early spring. The water temperature in the lower Yakima River was generally 1 to 4 C higher than the Columbia River during the early spring period (Figure 8). Within the Yakima River, there was a significant inverse relationship between discharge and water temperature ($R = -0.44$, $P < 0.001$). Increases in flow during the spring period coincided with drops in water temperature.

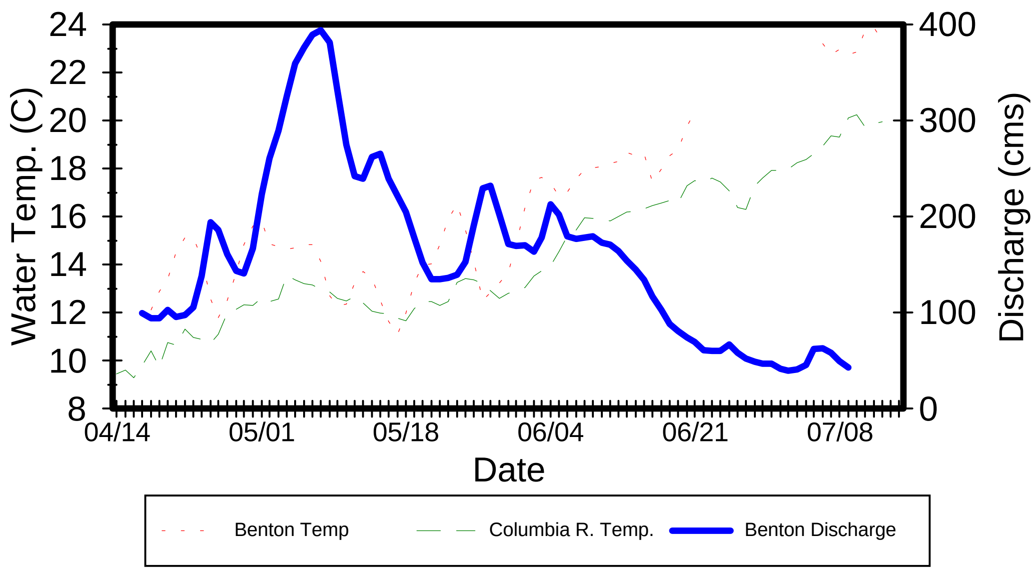

Figure 8. Daily mean water temperature and discharge (m^3/s) in the Benton section of the Yakima River and water temperature in the Columbia River above McNary Dam versus date during the spring and early summer of 1998.

Movement of northern pikeminnow and channel catfish did not exhibit as clear a pattern as smallmouth bass. We recaptured 41 northern pikeminnows in 1997 and 1998 ranging from 1-327 days after they were tagged. Most northern pikeminnows were recaptured in the Sunnyside and Granger sections (N = 18 and 11 respectively). Most northern pikeminnow (N = 37; 90.2%) were recaptured in the same section that they were initially marked. The average number of days between marking and recapture was 66.5 days. Three fish were recaptured at locations away from where they were originally marked. Three of these fish moved downstream after they were initially marked at Sunnyside, Granger and Benton (8/29/97, 8/20/97 and 8/13/97 respectively). The fish marked at Sunnyside and Granger were recaptured at the Chandler Juvenile Monitoring Facility at Prosser Dam in December, 1997. The fish tagged in the Benton section was recaptured near the Yakima River/Columbia River confluence by an angler.

With a few exceptions, most channel catfish were typically captured in the same general area where they were tagged. Two channel catfish tagged in the lower Yakima River were recaptured in the lower Snake River. Trapping data (Figure 4) and a limited number of recaptures of tagged fish weakly indicate channel catfish initiate an upstream migration from the Columbia River or Yakima River delta into the Yakima River as the water warms in the spring. Trap catches peaked April 19 in the Kennewick section (rkm 2.5-4.1), May 4 in the Duportail section (rkm 7.1-12.0), and June 15 in the Vangie section (rkm 15.2-19.0).

Species Composition

A total of 30 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. Introduced warmwater species such as smallmouth bass and common carp were abundant in the lower four sections and rare above Prosser Dam (Figure 9; Table 3). Conversely, native cool/coldwater species such as northern pikeminnow and mountain whitefish were relatively rare in the lower four sections and much more abundant above Prosser Dam. Two sandrollers, a rare species that had not been documented in the Yakima River for at least 30 years, were captured in the Vangie section in April, 1998.

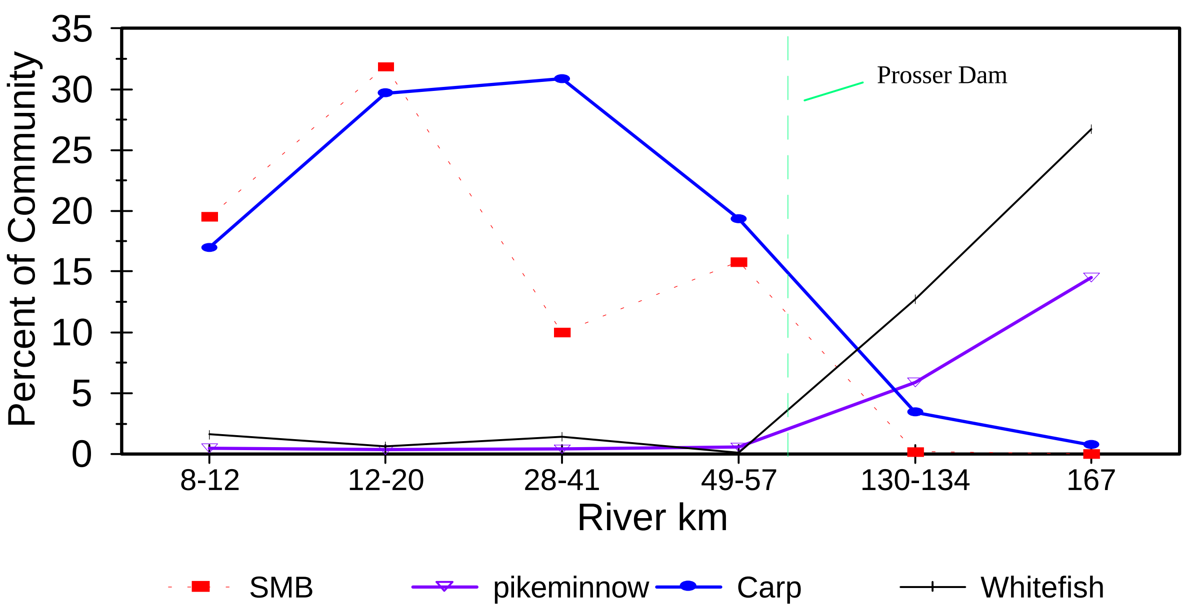

Figure 9. Species composition of smallmouth bass (SMB), northern pikeminnow, common carp, and mountain whitefish versus distance from the mouth of the Yakima River in 1997.

Table 3. Visually estimated fish species composition in total number and percent composition (in parentheses) for April 1998, 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]	Vangie	Horn	Benton	Granger	Sunnyside
	(12.2-20.2)	(28.1-41.0)	(49.3-57.1)	(130.0-134.4)	(167.4)
CCF[b]	5(0.12)	0(0.00)	0(0.00)	0(0.00)	0(0.00)
CCP	1347(31.84)	632(25.34)	939(31.20)	156(4.04)	3(0.92)
CHM	566(13.38)	343(13.75)	178(5.91)	214(5.54)	0(0.00)
COH	3(0.07)	55(2.21)	26(0.86)	56(1.45)	0(0.00)
DAC	2(0.05)	20(0.68)	92(3.06)	0(0.00)	2(0.62)
FCH	64(1.51)	18(0.72)	35(1.16)	18(0.47)	0(0.00)
MWF	21(0.50)	279(11.87)	92(3.06)	755(19.55)	240(73.85)
NPM	16(0.38)	31(1.24)	26(0.86)	59(1.53)	8(2.46)
PMK	0(0.00)	0(0.00)	2(0.07)	0(0.00)	0(0.00)
PMO	4(0.09)	0(0.00)	0(0.00)	0(0.00)	0(0.00)
RSS	19(0.45)	1(0.04)	2(0.07)	41(1.06)	0(0.00)
SCU	1(0.02)	0(0.00)	0(0.00)	0(0.00)	1(0.30)
SMB	887(20.96)	169(6.78)	723(24.02)	1(0.02)	0(0.00)
SND	2(0.05)	0(0.00)	0(0.00)	0(0.00)	0(0.00)
SPC	9(0.21)	231(9.26)	110(3.65)	42(1.09)	0(0.00)
SUK	1241(29.33)	689(27.63)	695(23.09)	2517(65.17)	70(21.54)
WCR	0(0.00)	0(0.00)	1(0.03)	0(0.00)	0(0.00)
WSH	16(0.38)	28(1.12)	71(2.36)	3(0.03)	1(0.31)
YLP	0(0.00)	0(0.00)	1(0.03)	0(0.00)	0(0.00)

[a] CCF(channel catfish), CCP(common carp), CHM(chiselmouth), COH(coho salmon), DAC(dace spp.), FCH(fall chinook), MWF(mountain whitefish), NPM(northern pikeminnow), PMK(pumpkinseed), PMO(peamouth),RSS(redside shiner), SCU (prickly sculpin), SMB(smallmouth bass), SND (sandroller), SPC(spring chinook), SUK(sucker spp.), WCR(white crappie), WSH(wild steelhead), YLP(yellow perch).

[b]Channel catfish are relatively unsusceptible to capture by electrofishing, therefore, they represent a larger but unknown proportion of the total fish community than is represented by these data.

Table 4. Visually estimated fish species composition in total number and percent composition (in parentheses) for May 1998, 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]	Vangie (12.2-20.2)	Horn (28.1-41.0)	Benton (49.3-57.1)	Granger (130.0-134.4)	Sunnyside (167.4)
CCF[b]	18(0.33)	0(0.00)	3(0.06)	0(0.00)	0(0.00)
CCP	756(14.02)	900(22.86)	518(11.12)	151(4.00)	13(6.53)
CHM	616(11.42)	613(15.57)	703(15.09)	228(6.04)	20(10.05)
COH	35(0.65)	52(1.32)	52(1.12)	436(11.55)	40(20.10)
DAC	39(0.72)	0(0.00)	367(7.88)	1(0.02)	0(0.00)
FCH	1222(22.66)	432(10.97)	962(20.65)	221(5.86)	0(0.00)
MWF	250(4.64)	176(4.47)	135(2.90)	745(19.74)	50(25.13)
NPM	20(0.37)	37(0.93)	72(1.55)	127(3.37)	12(6.03)
PMK	3(0.06)	0(0.00)	1(0.02)	0(0.00)	0(0.00)
PMO	4(0.07)	0(0.00)	0(0.00)	0(0.00)	0(0.00)
RSS	0(0.00)	1(0.02)	1(0.02)	188(4.50)	0(0.00)
SMB	1266(23.48)	214(5.44)	773(16.59)	0(0.00)	0(0.00)
SPC	6(0.11)	63(1.60)	13(0.28)	21(0.56)	0(0.00)
STG	0(0.00)	1(0.02)	0(0.00)	0(0.00)	0(0.00)
SUK	1155(21.42)	1447(36.68)	1058(22.71)	1656(43.88)	64(32.16)
WCR	1(0.02)	1(0.02)	0(0.00)	0(0.00)	0(0.00)
WSH	1(0.02)	0(0.00	0(0.00)	0(0.00)	0(0.00)
YLP	0(0.00)	0(0.00)	1(0.02)	0(0.00)	0(0.00)

[a] CCF(channel catfish), CCP(common carp), CHM(chiselmouth), COH(coho salmon), DAC(dace spp.), FCH(fall chinook), MWF(mountain whitefish), NPM(northern pikeminnow), PMK(pumpkinseed), PMO(peamouth),RSS(redside shiner), SMB(smallmouth bass), SPC(spring chinook), STG(sturgeon),SUK(sucker spp.), WCR(white crappie), WSH(wild steelhead), YLP(yellow perch).

[b]Channel catfish are relatively unsusceptible to capture by electrofishing, therefore, they represent a larger but unknown proportion of the total fish community than is represented by these data.

Diet Sampling

The modified lavage technique worked well for obtaining food habits information for smallmouth bass. The relatively small sample (N = 12) of smallmouth bass examined to estimate lavage efficiency for fish remains revealed no fish remains were missed by this technique. Consumption of salmonids by smallmouth bass was much lower in April than it was during May (Tables 5 and 6). The May samples for the Benton and Vangie sections showed that smallmouth bass consumed large numbers of salmon smolts, primarily fall chinook salmon. Only one of the salmonids in the April samples was identified as a spring chinook salmon, while the remainder were fall chinook salmon. All of the salmonids identified in the guts of smallmouth bass in the Benton and Vangie sections in May were fall chinook salmon.

Table 5. Summary results of diet analyses for smallmouth bass ($\geq$ 150 mm FL) sampled in the Benton, Horn and Vangie reaches on April 21-23, May 12-14 and June 4-5, 1998. The number of stomachs examined (N), the number (percent) of fish's guts in each sample that were empty, or contained invertebrates, fish, anadromous salmonids, and/or spring chinook salmon (SPC). The fish category includes salmonids.

Date	Section	N	Empty(%)	Invert.(%)	Fish(%)	Salmonids(%)	SPC(%)
4/21	Benton	48	21(43.8)	17(35.4)	12(25.0)	2(4.2)	0(0)
4/22	Horn	39	23(59.0)	14(35.9)	1(2.6)	0(0)	0(0)
4/23	Vangie	67	40(59.7)	16(23.9)	12(17.9)	3(4.5)	1(1.5)
5/12	Benton	93	25(26.9)	35(37.6)	33(35.5)	31(33.3)	0(0)
5/12	Horn	25	8(25.0)	5(20.0)	10(40)	4(16.0)	0(0)
5/14	Vangie	118	62(52.5)	24(20.3)	35(29.7)	20(16.9)	0(0)
6/5	Horn	92	34(37.0)	39(42.4)	17(18.5)	5(5.4)	0(0)

Table 6 shows the estimated predation rates for the Benton and Vangie sections during April and May, 1998. Table 7 shows the species of fish consumed by smallmouth bass in both sections and time periods. Table 8 shows the predation indices smallmouth bass for the river reaches and times when both abundance and consumption data were available.

Table 6. Estimated consumption rates of anadromous salmonids (primarily fall chinook salmon) within a section, based on smallmouth bass ($\geq$ 150 mm FL) population estimates, and gut analyses performed on samples collected during April and May, 1998. Total daily consumption refers to the estimated number of salmonids that would be eaten by all the smallmouth bass of predatory size ($\geq$ 150 mm FL) in one day. The estimated daily number of anadromous salmonids that would be consumed by 1000 smallmouth bass (S/1000) is also provided for comparison to the other predator species.

Section/Month	Pop. Est.	%w/ salmon	daily consumpt. rate	total daily consumpt.	S/1000
Benton/April	1528	0.042	3.79	243	159
Vangie/April	3210	0.045	2.80	404	126
Lower Yak/April [a]	19088	-	-	4085	-
Benton/May	3177	0.333	2.59	2740	862
Vangie/May	4176	0.169	1.95	1376	330
Lower Yak/May [a]	27676	-	-	18840	-

[a] Total average for the time period, calculated by using Vangie data for the Yakima River below Horn Rapids Dam and the Benton data for the reach between Horn Rapids and Prosser dams.

The relative abundance of spring chinook salmon smolts based on visual counts made by the electrofishing crew showed that they were most abundant during the April sampling period, but were still quite rare in comparison with other fish species/stocks (Figure 10, Table 3). Fall chinook were relatively abundant during the May and June sampling dates (Figure 10, Table 4).

Table 7. Species composition of fish found in smallmouth bass stomachs collected in the lower Yakima River April through August, 1998. Total number of prey fish in sample (N), and number of each prey species are presented for each date in each section. Numbers in parentheses represent the number of fish placed in that category based on size and bone information but are not positively identified using diagnostic bones and are included in the total number. The number in parentheses in the SMB and FAC species groups represents fish that were not positively identified to be in those groups but were placed in them by a weight of evidence approach. The numbers in parentheses are included in the total number for the species groups.

			Prey Species[a]														
Date	Section	N	DAC	SUC	NPM	CHM	SMB	CCF	YLP	FAC	SPC	MWF	CCP	PMK	MOS	SAL	NSA
4/21	Benton	21	4	2		2	3(1)	2		3(1)		5					
4/22	Horn	1														1	
4/23	Vangie	14	1			1	1(1)	1		3	1	6					
5/12	Benton	51	2	2			2			44(7)							1
5/12	Horn	19		1				1		10		4					3
5/14	Vangie	41				1	1	1		22(6)		9		1	3		3
6/3	Benton	17	8	1			1(1)	1		6(3)							
6/4	Horn	22	1	2	1			1		6			1			1	9
6/4	Vangie	10					1(1)			7(3)		2					
8/19	Benton	9	2		1	1	1						4				
8/20	Vangie	17	5				2	6	1							2	1

[a] DAC = dace spp., SUC = sucker spp., NPM = northern pikeminnow, CHM = chiselmouth, SMB = smallmouth bass, CCF = channel catfish, YLP = yellow perch, FAC = fall chinook salmon, SPC = spring chinook salmon, MWF = mountain whitefish, CCP = common carp, PMK = pumkinseed, MOS = mosquitofish, SAL = salmonid spp., NSA = non-salmonid spp.

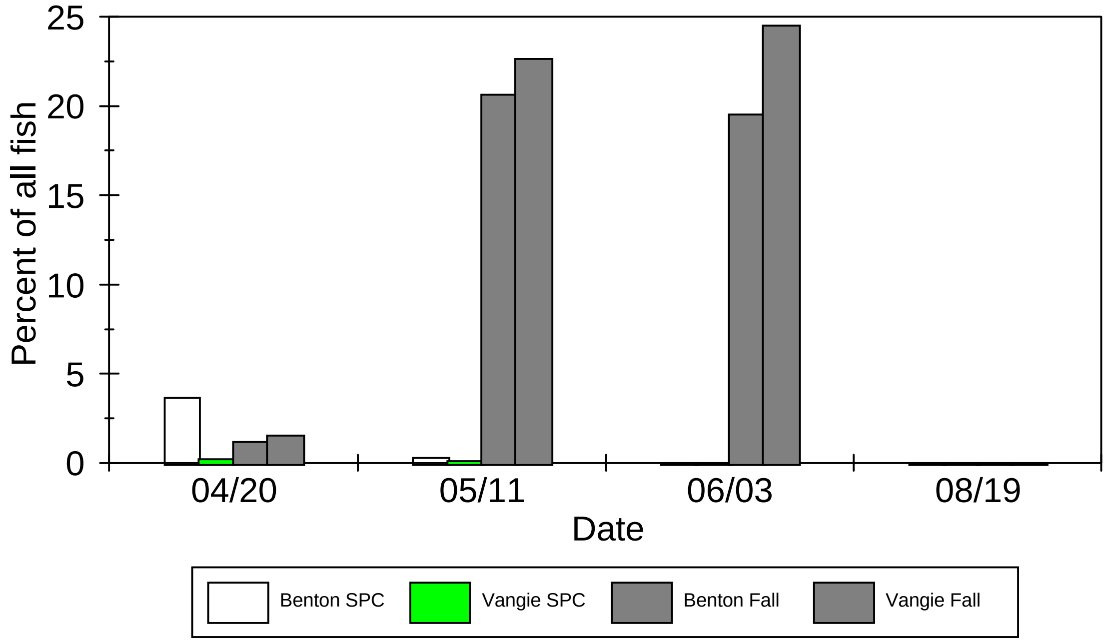

Figure 10. Relative abundance (percent of all fish observed) of spring chinook salmon smolts in the Benton and Vangie sections of the lower Yakima River versus sample date (1998). Relative abundance of fall chinook salmon parr and smolts is also shown for the Benton and Vangie sections.

Smallmouth bass appeared to show preference for fall chinook salmon and mountain whitefish (Figure 11). The appearance of preference for channel catfish is probably incorrect due to the low capture efficiency for channel catfish. It also appeared that sucker species and chiselmouth were not preferred prey species. The lack of preference for sucker species and chiselmouth may be explained by the large size of the juveniles (that were included in the 'availability' estimates); as they may have been too large for most smallmouth bass to prey upon.

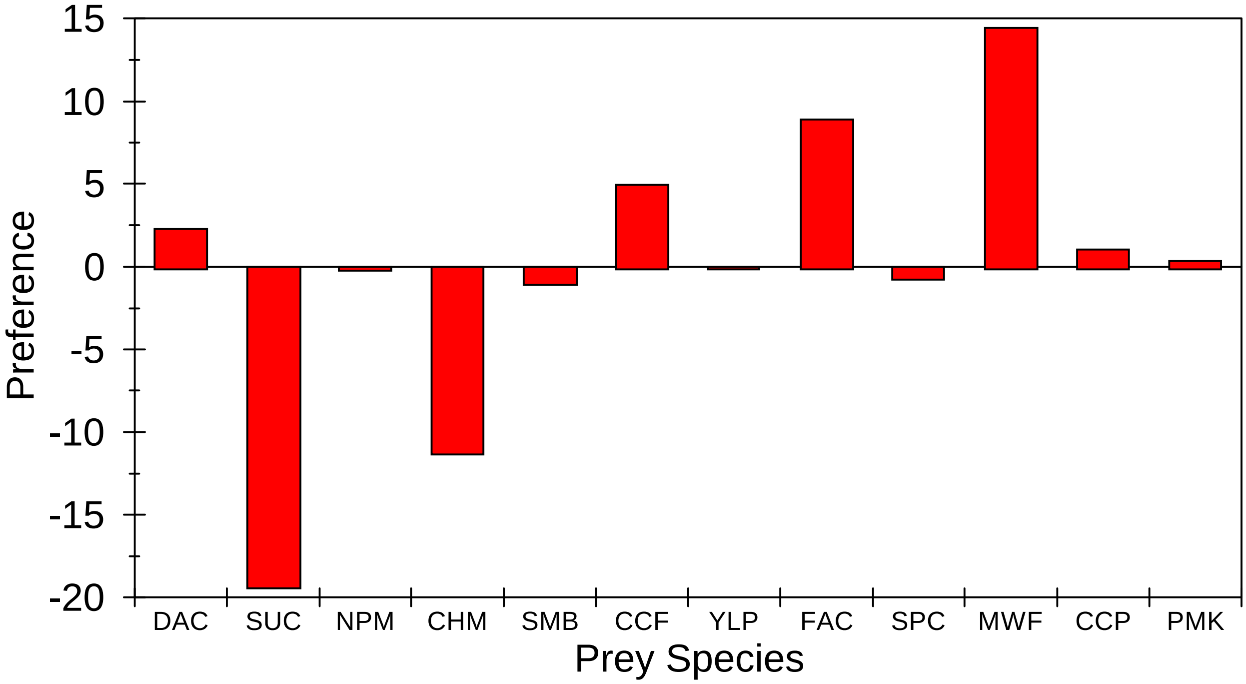

Figure 11. Prey species preference of smallmouth bass in the lower Yakima River during the spring of 1998. Preference was determined by subtracting the percent of a given prey species observed during electrofishing (availability) from the percent of that prey species observed in smallmouth bass guts (use). A positive value suggests a preferred prey species, while negative values suggest prey items that were not preferred.

Table 8. Predation index data for smallmouth bass predation on anadromous salmonids in the lower Yakima River reaches sampled during the estimated peak and last quartile of spring chinook salmon smolt emigration in 1998. The catch per unit effort (fish/min) of smallmouth bass ≥ 150 mm (D), the length of the extrapolated river reach (S) and the abundance index (AI) are shown for date and reach. The mean digestion time (DT), the mean number of salmonids/gut (SAL), the fraction of gut samples that contained at least one salmonid (F), the consumption index (CI), and the predation index (PI) are also shown for each reach and sample date. The predation index data for consumption of spring chinook salmon are also shown.

Date	Section	D	S	AI	DT	SAL	F	CI	PI
					Total Salmonids				
					Peak				
4/21	Benton	0.24	39.9	9.58	10.2	1.50	0.042	0.15	1.44
4/23	Vangie	0.43	28.1	12.08	17.76	1.33	0.045	0.08	0.97
4/24	Horn	0.15	12.9	1.97	N/A	0	0	0	0
					Last quartile				
5/12	Benton	0.56	39.9	22.34	19.29	1.41	0.333	0.58	12.96
5/14	Vangie	0.77	28.1	21.64	18.21	1.11	0.169	0.25	5.41
5/12	Horn	0.15	12.9	1.31	13.90	2.25	0.160	0.66	1.31
					Spring chinook salmon				
					Peak				
4/21	Benton	0.24	39.9	9.58	N/A	0	0	0	0
4/23	Vangie	0.43	28.1	12.08	33.18	1.00	0.015	0.01	0.12
4/24	Horn	0.15	12.9	1.97	N/A	0	0	0	0
					Last quartile				
5/12	Benton	0.56	39.9	22.34	N/A	0	0	0	0
5/14	Vangie	0.77	28.1	21.64	N/A	0	0	0	0
5/12	Horn	0.15	12.9	1.31	N/A	0	0	0	0

When we use the Vangie data to extrapolate consumption by smallmouth bass to the portion of the Yakima River below Horn Rapids Dam (28.1 km) and the Benton data to extrapolate to the portion of the Yakima River between Horn Rapids and Prosser dams (39.9 km), the total daily consumption of juvenile fall chinook salmon during the May sampling was estimated to be 18,840/day. The data from the Horn section was not used in expansions for 2 reasons; 1) low capture efficiencies give us low confidence in the abundance estimates, and 2) future sampling will not be conducted in the Horn section. When we incorporate the entire data set (both sections for both time periods, i.e., four data sets) into these calculations, we estimate that a total of 524,300 salmon smolts were consumed between April 15 and May 30, 1998. The total estimated number of fall chinook salmon pre-smolts and smolts emigrating past Prosser Dam in 1998 was 486,573 (B. Watson, YIN, Personal communication). If we assume 75% of the fall chinook salmon production in the Yakima River occurs below Prosser Dam, then a crudely expanded estimate of wild fall chinook salmon juveniles available to predators in the lower Yakima River would be about 1.9 million. So, smallmouth bass may have consumed about 27% of the wild fall chinook produced in the Yakima basin in 1998. An additional 1.2 million hatchery-reared fall chinook salmon were released by the Yakama Indian Nation Fisheries Program below Prosser Dam beginning on the evening of May 29, however, all of the sampling used to conduct these extrapolated consumption estimates was conducted prior to the release of the hatchery fish. The average number of adult fall chinook salmon passing Prosser Dam between 1983 and 1996 was 1,251/yr. Again, if we assume 75% of the Yakima basin fall chinook salmon production is below Prosser Dam, the total estimated return of fall chinook salmon to the Yakima River might have averaged slightly over 5,000 fish. If this is the case, the smallmouth bass may be consuming enough juvenile salmon to reduce the return of adult fall chinook salmon to the Yakima River by about 1,350 adults (0.27 x 5000)(however, hatchery production has accounted for some unknown portion of the adult returns of fall chinook salmon to the Yakima River since 1984).

When we used the formulas to estimate daily consumption of spring chinook salmon smolts we predicted that 206/d were eaten in the lower 68 km. If we assume 30 days of relatively high availability of spring chinook salmon smolts, the expanded consumption estimate would be 6,180 (4.3% of the estimated spring emigration numbers (142,821; March 15 - June 30)).

Salmonid consumption by northern pikeminnow was higher during the period of sampling that was intended to coincide with the last quartile of the spring chinook salmon emigration than during the period of the estimated peak of emigration (Tables 9 and 10). The percentage of northern pikeminnow that contained salmonids ranged from 21 to 25% for the Horn and Granger sections during May. Northern pikeminnows consumed yearling and subyearling salmonids in the Granger section, however, northern pikeminnow in the Horn section consumed only subyearling fall chinook. During the peak and last quartile sampling periods, 82% of all salmonids consumed were subyearling salmonids (Table 9). The only northern pikeminnow stomach samples that contained spring chinook were collected at Granger during the period of sampling intended to coincide with the last quartile of the spring chinook emigration. Based on diagnostic bones, we identified 11 species of prey fish that were consumed by northern pikeminnow in the Horn and Granger sections from April to June (Table 11). Most (87.5%) of the fish prey items consumed by northern pikeminnow were soft-rayed species (Table 11).

The predation indices for northern pikeminnow predation on salmonids were generally much lower than the predation indices for smallmouth bass (Tables 10 and 8 respectively). The predation indices for northern pikeminnow were highest during the sampling period that was intended to coincide with the last quartile of the spring chinook emigration for all sampling periods for the Sunnyside, Granger and Horn sections. The predation indices for northern pikeminnow during the last quartile of spring chinook emigration for the Horn and Granger sections were similar (Table 10), although, the abundance index was higher for the Granger section and the consumption index was highest for the Horn section (Table 10). The only sampling period that produced a predation index for spring chinook salmon was the period intended to coincide with the last quartile of the spring chinook emigration (Table 10). We found no evidence of spring chinook predation by northern pikeminnow during any other sampling period or location. The estimated number of salmonids consumed per 1000 northern pikeminnows was highest for the Horn section during the period of the last quartile of spring chinook emigration (Table 10). However, lack of estimates of northern pikeminnow abundance for all sampling periods and locations preclude estimates of total number of salmonids consumed by northern pikeminnow.

Table 9. Summary results of diet analyses for northern pikeminnow (≥ 150 mm FL) sampled in the Granger and Horn reaches on April 17-24, May 12-19, and June 4-5, 1998. The number of stomachs examined (N), the number (percent) of fish's guts in each sample that were empty, or contained invertebrates, fish, anadromous salmonids, and/or spring chinook salmon (SPC). The fish category includes salmonids.

Date	Section	N	Empty(%)	Invert.(%)	Fish(%)	Salmonids(%)	SPC(%)
4/17	Granger	19	8(42.1)	2(10.5)	4(21.1)	1(5.3)	0(0)
4/22	Horn	15	9(60.0)	4(26.7)	2(13.3)	0(0)	0(0)
5/12	Horn	8	2(25.0)	3(37.5)	2(25.0)	2(25.0)	0(0)
5/19	Granger	33	12(36.4)	16(48.5)	14(42.4)	7(21.2)	2(6.1)
6/5	Horn	13	5(38.5)	6(46.2)	2(15.4)	1(7.7)	0(0)

Table 10. Predation index data for northern pikeminnow on anadromous salmonids in the lower Yakima River reaches sampled during the estimated peak and last quartile of spring chinook salmon smolt emigration in 1998. The catch per unit effort of northern pikeminnow ≥ 150 mm (fish/min; D), the length of the river reach (S) and the abundance index (AI) are shown for date and reach. The mean digestion time (DT), the mean number of salmonids/gut (SAL), the fraction of gut samples that contained at least one salmonid (F), the consumption index (CI), the predation index (PI), and the estimated number of salmonids consumed per 1000 northern pikeminnow per day (S/1000), are also shown for each reach and sample date. The predation index data for consumption of spring chinook salmon are also shown.

Date	Section	D	S	AI	DT	SAL	F	CI	PI	S/1000
					Total Salmonids					
					Peak					
4/16	Sunnyside	0.33	0.18	0.06	N/A	0	0	0	0	0
4/17	Granger	0.13	4.1	0.52	17.8	1	0.05	0.04	0.02	35
4/24	Horn	0.15	12.9	1.97	N/A	0	0	0	0	0
					Last quartile					
5/28	Sunnyside	0.55	0.18	0.10	48.2	2	0.29	0.37	0.04	376
5/19	Granger	0.28	4.1	1.14	34.6	2.5	0.21	0.39	0.44	390
5/12	Horn	0.04	12.9	0.46	11.2	1.5	0.25	1.06	0.48	1058
					Spring chinook salmon					
					Peak					
4/16	Sunnyside	0.33	0.18	0.06	N/A	0	0	0	0	0
4/17	Granger	0.13	4.1	0.52	N/A	0	0	0	0	0
4/24	Horn	0.15	12.9	1.97	N/A	0	0	0	0	0
					Last quartile					
5/28	Sunnyside	0.55	0.18	0.10	N/A	0	0	0	0	0
5/19	Granger	0.28	4.1	1.1.4	61.9	2.0	0.06	0.12	0.14	80
5/12	Horn	0.04	12.9	0.46	N/A	0	0	0	0	0

Table 11. Species composition of fish found in northern pikeminnow stomachs collected in the lower Yakima River April through June, 1998. Total number of prey fish in sample (N), and number of each prey species are presented for each date in each section.

			Prey Species[a]											
Date	Section	N	DAC	SUC	CHM	SMB	RSS	FAC	SPC	MWF	COH	CCP	STB	NSA
4/17	Granger	4								1	1	2		
4/22	Horn	2					2							
5/12	Horn	4				1		3						
5/19	Granger	21	1	2	1		2	3	4	2	2		1	3
6/4	Horn	1						1						

[a] DAC = dace spp., SUC = sucker spp., CHM = chiselmouth, SMB = smallmouth bass, RSS = redside shiner, FAC = fall chinook salmon, SPC = spring chinook salmon, MWF = mountain whitefish, COH = coho salmon, CCP = common carp, STB = stickleback, NSA = non-salmonid spp.

Sample size and quality of channel catfish stomach samples was relatively low in comparison to the data for smallmouth bass and northern pikeminnow. A small percentage of the catfish guts examined contained salmonids (Table 12), however, one catfish that consumed a large number of salmonids (76 fall chinook salmon), which increased the mean number of salmonids consumed per catfish (Table 12; mean number of salmonids/catfish = 20, for the catfish that contained at least one salmonid).

Table 12. Diet composition of channel catfish stomachs collected in the lower Yakima River, April through June 1998. Total number of stomachs in sample (N), and number of times (percentage) each category was found in a stomach is presented. Anadromous salmonids are included in the fish category. The invertebrate (Invert.) category includes crayfish.

	Food Category							
N	Empty	Fish	Salmonid	Invert.	Crayfish	Seeds	Bird	Rodent
137	70 (51.0)	26 (19.0)	4 (2.9)	43 (31.3)	31 (22.6)	21 (15.3)	3 (2.2)	2 (1.5)

Table 13. Species composition of fish found in channel catfish stomachs collected in the lower Yakima River April through June 1998. Total number of fish in stomachs (N), and number (and percentage in parentheses) of prey species is presented.

	Prey Species[a]												
N	CCF	CCP	CHM	DAC	FAC	SUC	MWF	NSA	NPM	SAL	SCU	SMB	WSH
121	8 (6.6)	3 (2.5)	2 (1.7)	1 (0.8)	77 (63.6)	8 (6.6)	3 (2.5)	7 (5.8)	2 (1.7)	2 (1.7)	1 (0.8)	6 (5.0)	1 (0.8)

[a]CCF = channel catfish, CCP = common carp, CHM = chiselmouth, DAC = dace spp., FAC = fall chinook salmon, SUC = sucker spp., MWF = mountain whitefish, NSA = non-salmonid spp., NPM = northern pikeminnow, SAL = salmonid spp., SCU = sculpin spp., SMB = smallmouth bass, WSH = wild steelhead.

The potentially large population of channel catfish could consume a substantial number of salmonids annually if even a small portion of the population consumes salmonids at the rate we observed. For every 1,000 channel catfish in the lower Yakima River we estimate that 580 salmonids/d would be eaten. If we assume the same period of high availability for fall chinook salmon juveniles that we used for smallmouth bass extrapolations (April 15-May 30) we estimate that 26,100 salmonids would be consumed for every 1,000 channel catfish in the lower Yakima River. Our data indicates there was a minimum of 2,664 channel catfish between the mouth and Horn Rapids Dam in the spring of 1998 (based on raw capture data). Our mark-recapture ratios from trapping data suggest the numbers may be much higher (Table 2; 10,693-14,203/km). Thus, our wide ranging estimates of seasonal salmonid consumption by channel catfish extend from 69,530/year to 10.4 million/year.

"Hot Spot" Sampling

Success at capturing northern pikeminnows at 'hot spots' was very low. Hook and line sampling for northern pikeminnow immediately below Roza Dam yielded low catch rates, and none of the pikeminnows examined had eaten any salmonids (Table 14). Mean catch rate (fish/min) was much lower during the sampling period intended to coincide with the peak of spring chinook salmon smolt emigration past Roza Dam than it was in the period intended to cover the last quartile of emigration. Most of the northern pikeminnows captured below Roza Dam were sexually mature adults (sample 1: 86%; sample 2: 88%).

Mean catch rates (fish/min) using jet boat electrofishing gear were higher at Sunnyside Dam than at Roza Dam due to sampling gear (Table 14). However sample sizes during the mark and recapture efforts were low, and never exceeded 7 northern pikeminnow. Only the April sampling period which was intended to coincide with the peak of the spring chinook salmon smolt emigration at Sunnyside Dam yielded a population estimate of the total number of northern pikeminnows present (25; 95% CI = 6-49). Two of the seven pikeminnows sampled during the period intended to coincide with the last quartile of the spring chinook salmon emigration had consumed coho salmon, although none of the northern pikeminnow sampled during the earlier sampling period had consumed salmonids. Most (79%) northern pikeminnows captured at Sunnyside Dam were >300 mm fork length. Sampling efforts at the outfall of the Chandler juvenile fish facility did not yield any predatory fish.

Table 14. Data from northern pikeminnow 'hot spot' sampling in the Yakima River during 1998. Mean catch per unit effort (CPUE (fish/min)), number marked (M), number recaptured and number of marked fish (R) in the recapture sample (C), and summary of gut contents of fish in the recapture sample are presented for Roza and Sunnyside sites. Data collected at Roza is based on hook and line sampling and data collected at Sunnyside Dam is based on jet boat electrofishing.

Dates	Site	CPUE	M	R/C	%Empty	%Invert.	%Fish	%Salmonids
3/31-4/2	Roza	0.013	16	0/14	50	36	43[a]	0
4/27-29	Roza	0.043	37	0/16	63	13	25[b]	0
4/14-16	Sunnyside	0.330	6	1/6	33	0.5	50[c]	0
5/27-28	Sunnyside	0.550	5	0/7	71	0	29	29[d]

[a] All sculpins.
[b] Two suckers, one sculpin, and one unknown, non-salmonid.
[c] All whitefish.
[d] Two coho salmon based on length frequency data.

Discussion

Our results indicate that predation on juvenile salmonids by predaceous fishes in the lower Yakima River is substantial. Smallmouth bass alone are estimated to be able to consume about half a million smolts per year, resulting in the annual loss of an estimated 1,350 adult salmon. Channel catfish and northern pikeminnow also consume large numbers of salmonids, however our data for those species is not readily expandable due to our difficulty in assessing their abundance. We were, however, able to estimate the number of anadromous salmonids that might be consumed by every 1000 northern pikeminnow and channel catfish. Based on the estimated mean numbers of anadromous salmonids that might be consumed by every 1000 predators of each species, it appears that the predatory species' potential to impact fall chinook salmon might be ranked as follows; 1) smallmouth bass, 2) northern pikeminnow, and 3) channel catfish. Because we found very few spring chinook and coho salmon and steelhead in guts, our ability to rank the predatory species by their potential impact on those prey species is limited. If we consider timing and distribution of both predators and prey, as well as the limited diet information we have regarding spring chinook salmon, we would rank the predators' potential to impact spring chinook salmon as follows; 1) northern pikeminnow, 2) smallmouth bass, and 3) channel catfish.

Some sampling adjustments will be made, beginning in 1999. We will sample weekly throughout the spring chinook salmon smolt emigration period (April 1 to May 21). We will no longer sample the Horn section (primarily a smallmouth bass section), and will instead focus more effort on northern pikeminnow in the section of the river between Sunnyside and Prosser dams. We will not operate traps for channel catfish and will rely instead on a limited amount of data we can obtain by incidental electrofishing captures and a limited number of gillnet sets. We will discontinue hook and line sampling at the Sunnyside and Chandler hot spot areas.

Smallmouth bass abundance and the consumption rates in the lower Yakima River are substantially higher than other studies have reported for large rivers and reservoirs in the Columbia basin. Figure 12 shows the estimated density of smallmouth bass 200 mm long and larger in the Yakima River sites and in the John Day Reservoir on the Columbia River (Beamesderfer and Rieman 1988). The smallmouth bass we sampled in the lower Yakima River also consumed salmonids more frequently than most other smallmouth bass populations that have been sampled in the Northwest. Figure 13 illustrates the variability of the incidence of predation on salmonids by smallmouth bass within the Yakima River, as it is affected by sampling date, as well as the relation of the incidence of predation in the Yakima River to that which has been reported for other waters in the area (data from Bennett et al. 1991).

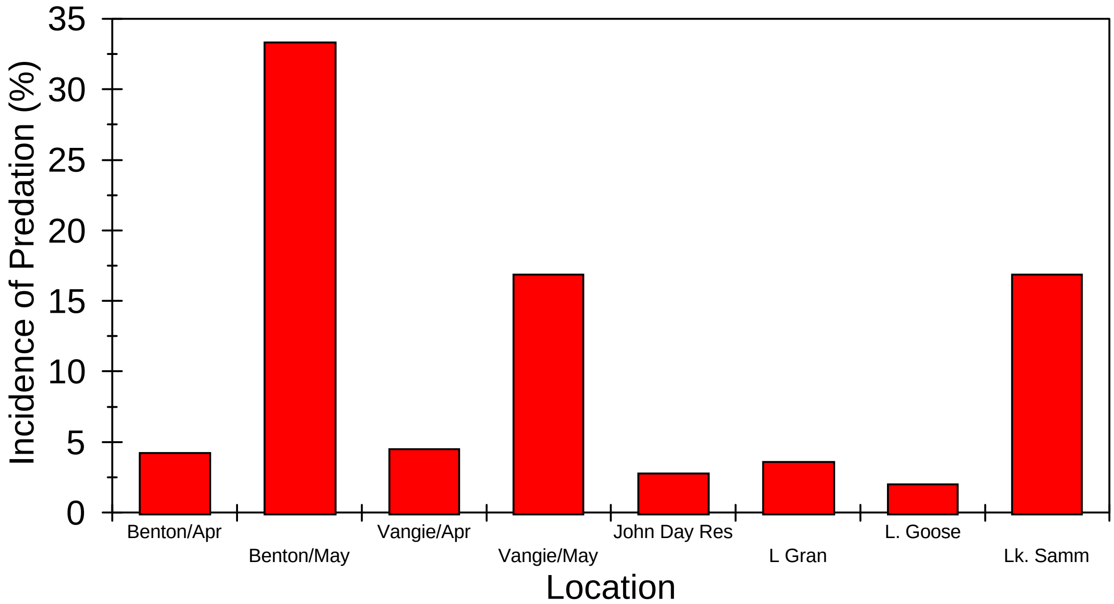

Figure 12. Density of smallmouth bass (estimated number 200 mm and longer per hectare) in two sections of the lower Yakima River during two time periods in 1998 and in the John Day Reservoir (spring samples, data from Beamesderfer and Rieman 1988).

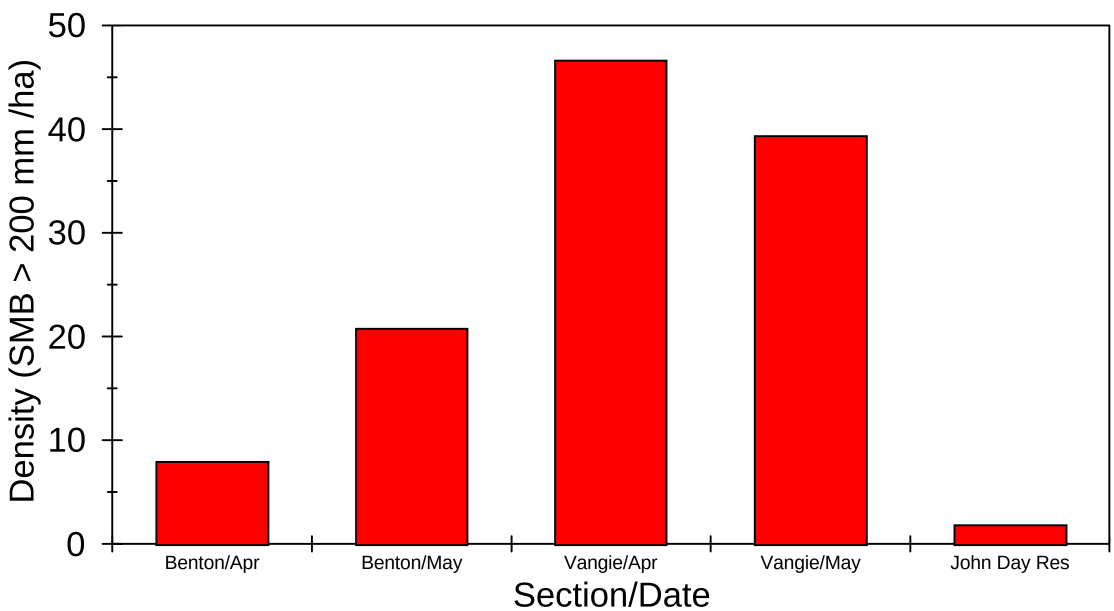

Figure 13. Incidence of predation (% of sampled predators containing at least one salmonid) by smallmouth bass in two sections of the lower Yakima River during two time periods in 1998 as well as spring data from four other sites in the northwest (data from Bennett et al. 1991; John Day Res = John Day Reservoir on the Columbia River, L Gran = Lower Granite Reservoir on the Snake River, L Goose = Little Goose Reservoir on the Snake River, Lk. Samm = Lake Sammamish, Washington).

The combined effect of high predator densities and high consumption rates of salmonids in the Yakima River indicates a significant annual loss of salmonid production to piscine predators. This is particularly true when predation impacts are unnaturally high due to the introduction of non-native species and the alteration of the physical habitat (e.g., dams and irrigation effects) that favors these introduced fishes. When managers are presented with this information, many will begin to consider what types of management actions could be undertaken to reduce this loss of anadromous fish production to predators. There are a wide variety of actions that have the potential to reduce unnaturally high predation impacts (Table 15).

Table 15. Management actions intended to reduce losses of salmonids to predation in the lower Yakima River. Each action listed is qualitatively rated by its potential to result in the desired effect as well as its potential to produce undesired ecological effects.

Management Action	Desired effect	Undesired effects (risks)
Trapping migrants/removal	Low	Low-Moderate
Electrofishing/removal	Moderate	Moderate
Capture/relocation	Moderate	Low-moderate
Removal of Angling regs.	Low	Low
Bounty Program	Low-Moderate	Low
Decrease water temp. 2 C	High	Low

Each of the actions listed in Table 15 has some potential to decrease losses of juvenile salmonids to fish predators within the time frame that the action is applied. Long term effects of these actions are uncertain, but unlikely to produce much of the desired effect over time, unless the action is maintained at a high level of effort. Trapping predatory fish migrating into the Yakima River from the Columbia River could remove some of the larger adult smallmouth bass and channel catfish. The benefits of this are that within a given year, many fish could be removed, thereby reducing predation within that year. Drawbacks of trapping migrants are that it only reduces the migratory portion of the populations, which could be compensated for by resident populations. Also, it targets only the larger/reproducing individuals which do not appear to be as selective for salmonid smolts as prey items (i.e., they eat larger prey, e.g., chiselmouth). Further, any potential benefits, in terms of reduced predation, will accrue only upstream of the trap and the trapping would have to be conducted every year to produce the survival benefits for smolts, unless spawning by predators was impacted, in which case future reductions in predation could result. Finally, disposal of the fish that were removed could be difficult if they are found to be contaminated. Mountain whitefish and bridgelip suckers collected by the USGS in the lower Yakima River between 1989 and 1991 were found to be highly contaminated with DDT (Washington State Department of Health 1995). Smallmouth bass and channel catfish samples have been provided to the EPA and the data should be released in mid-1999.

Using electrofishing to remove predators could be effective at reducing the number of predators within the area that is covered by the electrofishing crews. Repeated passes would have to be done to achieve appreciable depletions in the predator population due to the generally low capture efficiency of this gear (2-4% per day per boat for smallmouth bass; a lower, but unknown effectiveness would be expected for northern pikeminnow; and an extremely low effectiveness would be expected for channel catfish). Electrofishing has the added benefit of being selective, in that it can be applied to areas and size ranges that appear most critical. Electrofishing also has moderate potential for producing undesired effects such as injuring or impairing the reproductive capacities of returning adult salmonids (Snyder 1995). In addition, electrofishing injuries to emigrating smolts may reduce their long-term fitness and survival (McMichael et al. 1998). Finally, electrofishing would only affect the area where it was conducted, and it would have to be applied multiple times annually to be effective. Similar to the trapping/removal effort, fish would have to be disposed of, which may be problematic if they are found to be contaminated.

Capture and relocation of fish could reduce the number of predators within the areas sampled if a host of gear types were employed to capture predatory fishes. This would only be viable if the fish are found to be fit for human consumption (see previous paragraph). Undesired effects on other species would depend on the types of methods used to capture the fish and the level of effort (e.g., the number of electrofishing trips) to capture the fish. Again, the effects of this approach would generally only be apparent in the reaches sampled and during the year of sampling unless future reproduction of predatory fishes were reduced.

Removal of the angling regulations would not be likely to produce any appreciable effect on the predator populations. There are currently no restrictions on gear or harvest for channel catfish and northern pikeminnow in the Yakima or Columbia rivers. Tagged smallmouth bass were captured by anglers at a mean rate of 3.15%, which if we assume a non-reporting rate of 50%, translates into an exploitation rate of about 6.3%/year. However, anglers released 42% of the bass they captured, yielding a harvest estimate of about 3.7%. It appears unlikely that eliminating the 5 fish limit on smallmouth bass would have any substantial effect on the density of predators.

A bounty program for predatory fishes, if it achieved harvest rates similar to those reported by Beamesderfer et al. (1996), could be effective in decreasing predation loss to fishes during the years that the bounty program operated. The annual removal of 9-16% (the range reported by Beamesderfer et al. (1996)) of the predatory fishes in the lower Yakima River could potentially increase adult returns by that amount. Undesirable impacts to salmonids would likely be fairly low, as the adult salmonids are not present in large numbers when most of the angling for predatory fishes would occur. However, most of the angling for predatory fishes would probably occur after the spring emigrating smolts were out of the Yakima River, thereby reducing the potential to positively influence survival rates within the year the predatory fish were removed. Disposal of predatory fish may be complicated by the issues of contamination mentioned above.

Decreasing the river temperature during the spring could substantially reduce the loss of salmonids to predation by fish. The effect of water temperature on the metabolic rates and subsequent meal turnover times in fishes is large. A 2 C drop (from 13.2 to 11.2 C) in water temperature would decrease the losses of salmonids to smallmouth bass by 23% in the Benton reach during the May sample period. Digestion rates of northern pikeminnow and channel catfish would be similarly affected. This could possibly be achieved by changes in management or configuration of the irrigation system in the Yakima basin. Vaccaro (1986) projected a 2 to 3 C decrease in water temperature in April and May, 1981, for unregulated flow conditions (102 to 181 m^3/s) versus regulated flow (42 to 57 m^3/s). It may also be possible to restructure irrigation returns to pass water through the ground to clean and cool it prior to entering the river channel via groundwater. Reducing water temperature could potentially affect the entire reach of the Yakima River where predation by fish is most prevalent. Also, it could potentially reduce the influx of predators from the Columbia River if the temperature of the Yakima River was reduced to the point where it was cooler than the Columbia River. Cooler water would potentially benefit returning adult spring chinook salmon and steelhead as well as decreasing the chances for smolts and adults to be affected by pathogens. There are some drawbacks/potential roadblocks to this action. To decrease the temperature in some years would be difficult due to the distance between storage reservoirs and the river reach where most predation occurs, as well as natural variability of the hydrograph (low flow and high flow years) and weather (hot weather) patterns. It would also be expensive and possibly politically difficult to modify system operations and/or structure to accomplish this task.

It is possible that the loss of salmonids to predaceous fishes in the lower Yakima River could be reduced by a combination of the aforementioned management actions. If, for example, the river were cooled 2 C during smolt emigration, predatory fishes were removed on four electrofishing runs early in the season (before smolt emigration), and a bounty program were instituted that reduced predator populations to a level similar to the reductions reported for the sport reward program for northern pikeminnow (Beamesderfer et al. 1996), then the cumulative survival advantage (reductions in losses to predation) of these actions might be close to 40% (23% + 4% + 12.5%; for smallmouth bass). A multiple-method approach such as this would have the largest cumulative benefits for salmonid survival but would also carry the highest risks to non-target species and highest economic costs. Ultimately, it will be a public policy decision to determine what, if any, management approach will be taken to reduce unnaturally high predator impacts on anadromous salmonids in the lower Yakima River.

Acknowledgments

Jeff Jackson, Vernon Bogar, Dave Burgess, Brenda James, Pat Kaelber, Scott Monds, Jim Cummins, Roger McKeel, Eric Anderson, Kenneth Ham, and Eric Bartrand all assisted with the field work. Paul Wagner and his field crew also assisted in the field. Joe Jay Pinkham III, Eugene Billy, and Linda Lamebull, YIN, assisted in the collection of most of the northern pikeminnow data and also sampled the Horn index section. Pat Monk provided some useful insight on stream temperatures. Staff at the Cook, Washington, BRD lab provided training for analyzing gut contents by diagnostic bones. This work was funded by the Bonneville Power Administration as part of the Yakima/Klickitat Fisheries Project.

References

Beamesderfer, R. C., and B. E. Rieman. 1988. Predation by resident fish on juvenile salmonids in a mainstem Columbia River reservoir: Part III. Abundance and distribution of northern squawfish, walleye, and smallmouth bass. Pages 211-248 *in* T. P. Poe and B. E. Rieman, editors. Predation by resident fish on juvenile salmonids in John Day Reservoir, 1983-1986. Final Report to Bonneville Power Administration, Portland, Oregon.

Beamesderfer, R. C. P., D. L. Ward, and A. A. Nigro. 1996. Evaluation of the biological basis for a predator control program on northern squawfish (*Ptychocheilus oregonensis)* in the Columbia and Snake rivers. Canadian Journal of Fisheries and Aquatic Sciences 53:2898-2908.

Bennet, D. H., J. A. Chandler, and L. K. Dunsmoor. 1991. Smallmouth bass in the Pacific Northwest: benefit or liability. Pages 126-135 *in* D. C. Jackson, editor, The first international smallmouth bass symposium. Mississippi State University, Mississippi State, Mississippi.

Bidgood, B. F., and A. H. Berst. 1969. Lethal temperatures for Great Lakes rainbow trout. Journal of the Fisheries Research Board of Canada 26:456-459.

Busack, C., and five coauthors. 1997. Yakima fisheries project: spring chinook supplementationmonitoring plan. Bonneville Power Administration, Portland, Oregon. DOE/BP-64878-1.

Hansel, H. C., S. D. Duke, P. T. Lofy, and G. A. Gray. 1988. Use of diagnostic bones toidentify and estimate original lengths of ingested prey fishes. Transactions of the American Fisheries Society 117:55-62.

Lampman. B. H. 1947. The coming of the pond fishes. Metropolitan Press, Portland, Oregon.

Light, R. W., P. H. Alder, and D. E. Arnold. 1983. Evaluation of gastric lavage for stomach analyses. North American Journal of Fisheries Management 3:81-85.

McMichael, G. A., A. L. Fritts, and J. L. Dunnigan. 1998. Lower Yakima River predatory fish census: feasibility study 1997. Pages 229-254 *in* Pearsons, T. N, and five coauthors. 1998. Yakima River species interactions studies, progress report 1995-1997. Bonneville Power Administration, Portland, Oregon.

McMichael, G. A., A. L. Fritts, and T. N. Pearsons. 1998. Electrofishing injury to stream salmonids; injury assessment at the sample, reach, and stream scales. North American Journal of Fisheries Management 18:894-904.

Patten, B. G., R. B. Thompson, and W. D. Gronlund. 1970. Distribution and abundance of fish in the Yakima River, Wash., April 1957 to May 1958. United States Fish and Wildlife Service Special scientific report No. 603. Washington D.C.

Pearsons, T. N., and five coauthors. 1998. Yakima River species interactions studies, progressreport 1995-1997. Bonneville Power Administration, Portland, Oregon.

Shrable, J. B., O. W. Tiemeier, and C. W. Deyoe. 1969. Effects of temperature on rate of digestion by channel catfish. The Progressive Fish Culturist 13:131-138

Snyder, D. E. 1995. Impacts of electrofishing on fish. Fisheries 20:26-27.

Tabor, R. A., R. S. Shively, and T. P. Poe. 1993. Predation on juvenile salmonids by smallmouth bass and northern squawfish in the Columbia River near Richland, Washington. North American Journal of Fisheries Management 13:831-838.

Vaccaro, J. J. 1986. Simulation of streamflow temperatures in the Yakima River Basin, Washington, April-October, 1981. U. S. Geological Survey Water Resources Investigations report 85-4232. Tacoma, Washington. 91 pages.

Vigg, S, T. P. Poe, L. A. Prendergast, and H. C. Hansel. 1991. Rates of consumption of juvenile salmonids and alternative prey fish by northern squawfish, walleyes, smallmouth bass, and channel catfish in John Day Reservoir, Columbia River. Transactions of the American Fisheries Society 120:421-438.

Vincent, R. 1971. River electrofishing and fish population estimates. Progressive Fish-Culturist33:163-170.

Ward, D. L., J. H. Petersen, and J. J. Loch. 1995. Index of predation on juvenile salmonids bynorthern squawfish in the lower and middle Columbia River and in the lower Snake River. Transactions of the American Fisheries Society 124:321-334.

Washington State Department of Health. 1995. DDT in bottom fish from the Yakima River. Informational brochure 12/95. Olympia, Washington