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YAKIMA RIVER SPECIES INTERACTIONS STUDIES

Annual Report 1993

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Executive Summary

Species interactions research was initiated in 1989 to investigate ecological interactions among fish in response to proposed supplementation of salmon and steelhead in the upper Yakima River basin. Data have been collected prior to supplementation to characterize the rainbow trout population, predict the potential interactions that may occur as a result of supplementation, and develop methods to monitor interactions. Major topics of this report are associated with the life history of rainbow trout, interactions experimentation, and methods for sampling. This report is organized into nine chapters with a general introduction preceding the first chapter and a general discussion following the last chapter. This annual report summarizes data collected primarily by the Washington Department of Fish and Wildlife (WDFW) between January 1 and December 3 1, 1993 in the upper Yakima basin above Roza Dam, however these data were compared to data from previous years to identify preliminary trends and patterns. Major preliminary findings from each of the chapters included in this report are described below.

- Rainbow trout generally spawned from February through June throughout the upper Yakima basin, with the peak of spawn timing positively related to elevation (peak of spawn timing was earlier at low elevations sites than at high ones). Steelhead spawned at similar times and within the range and habitat conditions as those used by spawning rainbow trout.

- Rainbow trout survival, growth, gonad development, and general health were not significantly impacted by radio transmitters that were surgically implanted into the intraperitoneal cavity of test fish shortly before the time of spawning.

- The movement distances of large rainbow trout (longer than 174 mm) that were anchor tagged were between 0 and 194 km. A total of 59% of the recaptured trout moved less than 5 km in the mainstem whereas 89% moved less than 5 km in the tributaries. Movement distances were generally longer in upper mainstem areas than in lower ones and longer during the winter and spring than during summer and fall.

- Using electrofishing techniques, densities of rainbow trout ranged from 0 to 0.24 fish/m^2 from 1990 through 1993 in index sites within eight tributaries. Spatial and temporal' variation of trout densities appearred to be related to elevation and habitat conditions/variability. Rainbow trout densities in five mainstem sections averaged 325 fish/km and were not as variable as densities in tributary index sites.

- Length-at-age of rainbow trout was inversely related to elevation and was generally greater in the mainstem than in the tributaries. Patterns in length-at-age also tended to correspond to geographic patterns of genetic variation. Most of the trout sampled from the tributaries spawned at age 1+ and 2+ whereas most that were sampled from the mainstem spawned at age 2+ and 3+. Very few of the rainbow trout sampled lived longer than five years.

Three major types of fish assemblages associated with rainbow trout were identified in the upper Yakima basin and these assemblage types could be distinguished using elevation/temperature and stream size. Fish species that characterized assemblages in sites that were relatively high in elevation and within small streams were bull, cutthroat, and brook trout. Assemblages inhabiting relatively low elevation sites in small streams were characterized by a high proportion of speckled dace. Assemblages inhabiting relatively low elevation sites in large streams were characterized by northern squawfish, chiselmouth, suckers, redside shiners, longnose dace, mountain whitefish, and juvenile spring chinook salmon.

- As a result of four successive annual experimental releases of approximately 33,000 hatchery steelhead into a tributary of the North Fork Teanaway River, no impacts to the sizes or densities of sympatric wild trout, or large scale displacements of trout were detected. However, agonistic interactions and small scale displacements were observed between hatchery steelhead and wild rainbow trout, with hatchery steelhead behaviorally dominating most contests presumably because of their larger size.

- Results from competition experiments performed in small enclosures within the North Fork Teanaway River suggested that: competition between hatchery-reared steelhead and naturally produced age 1+ and 2+ rainbow trout adversely impacted rainbow trout growth; the presence of age 0+ spring chinook salmon did not impact the growth of age 1+ and 2+ rainbow trout; and the presence of hatchery steelhead did not impact age 0+ spring chinook growth.

- Superior performance of hatchery-reared steelhead, reflected by in-river emigration rates, rates of precocialism, and incidence of residuaiism, was observed when their parents were hatchery broodstock as opposed to wild broodstock, were reared at lower densities, and were released at smaller sizes.

The results presented in these chapters were used to develop a preliminary recommendation regarding the appropriate spatial and temporal scale necessary for monitoring ecological interactions and provide some strategies that might be used to minimize undesirable ecological interactions between hatchery and wild fish. A genetic analyis of resident and anadromous *Oncorhynchus mykiss* collected from the Yakima basin is presented in an Appendix. In addition, future research needs are identified. All findings in this report should be considered preliminary and subject to further revision.

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General Introduction

Species interactions research was initiated in 1989 to investigate interactions among fish 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). Interactions between fish produced as part of the Yakima Fisheries Project (YFP) termed target species or stocks (hereafter referred to as target species), and other species or stocks (hereafter referred to as non-target species) may alter the population status of non-target species. This may occur through a variety of mechanisms. For instance, target species may consume non-target species (Cannamela 1992, Martin et al. 1993), alter habitat utilization thereby making non-target species more susceptible to predators, alter movement patterns of non-target fish (Hillman and Mullan 1989), compete with non-target species for food and space (Bachman 1984, Vincent 1987, Irvine and Bailey 1992), increase transmission and susceptibility to disease of non-target fish (Krueger and May 1991, Pearsons et al. 1993), and interbreed with non-target fish (Krueger and May 1991, Pearsons et al. 1993). These interactions may result from releases of first generation hatchery fish (type 1) and/or increases in the numbers of naturally produced progeny of hatchery fish (type 2) (Pearsons et al. 1993).

Work to date has focused on interactions between anadromous steelhead and resident rainbow trout (for explanation see Pearsons et al. 1993), however during the past year increased emphasis has been directed at investigating interactions between spring chinook salmon and rainbow trout. The change in emphasis to spring chinook salmon has largely been influenced by the shift in species planned for supplementation (DEIS 1994). 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, steelhead may be supplemented at a later date than spring chinook salmon. This redirection in the species to be supplemented has prompted us to prioritize interactions between spring chinook and rainbow trout, while continuing essential work on steelhead and rainbow trout interactions. Pre-facility monitoring of variables such as rainbow trout density, distribution, and size structure was continued and should be an important part of monitoring the effects of interactions, regardless of which species is supplemented.

This report is organized into nine chapters which represent major topics associated with the life history of rainbow trout, interactions experimentation, and methods for sampling. In contrast to previous reports (Hindman et al. 1991, McMichael et al. 1992, Pearsons et al. 1993) major topics are divided into chapters to provide broad treatment of each topic. The main topic of Chapter 1 is the description of the spatial and temporal spawning distribution of rainbow trout and steelhead. Chapter 2 presents findings from a field experiment designed to assess whether surgical implantation of radio tags affected gonad development of resident rainbow trout and the feasibility of using radio telemetry as a tool to determine spawning migrations of rainbow trout in the Yakima River. The actual field study to determine spawning migrations using radio telemetry was conducted collaboratively between the National Marine Fisheries Service (NMFS) and Washington Department of Fish and Wildlife (WDFW) during 1993 and a separate report will be prepared by NMFS. Movements that occurred throughout the year, as investigated using primarily anchor tags and traps, was the topic of Chapter 3. The abundance and biomass of salmonids rearing in index sites in tributary and mainstem areas of the upper Yakima basin is presented in Chapter 4, as well as identification of hypotheses that might explain the spatial and temporal variation of salmonid abundance observed. Chapter 5 presents size-at-age relationships for rainbow trout rearing and spawning in different locations throughout the upper Yakima basin. Description of the assemblage structure of fishes associated with rainbow trout and identification of the factors that might influence assemblage structure is the topic of Chapter 6. Chapter 7 attempts to quantify how the release of hatchery steelhead affected behavioral interactions, movement patterns, and population densities of wild trout. Results from experiments designed to investigate competition between three groups of fish: 1) hatchery steelhead and rainbow trout, 2) hatchery steelhead and spring chinook salmon, and 3) spring chinook salmon and rainbow trout are reported in Chapter 8. Finally, Chapter (9) examines the effects of parentage, rearing density, and size-at-release on instream performance of hatchery steelhead.

Information from these chapters has been synthesized into the following "General Discussion" section. The discusssion describes practical implications for planning in two general topic areas; 1) factors to consider for monitoring the ecological status of rainbow trout, and 2) potential strategies to reduce undesirable ecological interactions.

A genetic analysis of rainbow trout was conducted by WDFW in support of our studies and is presented in Appendix A. This appendix addresses the stock structure of rainbow trout in the upper Yakima basin, the influence of past hatchery stocking of rainbow trout on the genetic structure of rainbow trout populations, and the genetic delineation of steelhead and rainbow trout.

Except for Chapter 2, which has been submitted for publication, all of the chapters are in various stages of development and should be considered preliminary. Additional field work and/or analysis is in progress for topics covered in all other chapters. Additional field work will be contingent upon the availibility of funds. Readers are cautioned that any preliminary conclusions are subject to future revision as more data and analytical results become available.

This report is intended to satisfy two concurrent needs: 1) provide a contract deliverable from WDFW to the Bonneville Power Administration (BPA), with emphasis on identification of salient results of value in ongoing YFP planning, and 2) summarize results of research that have broader scientific relevance.

This annual report summarizes data collected between January 1, and December 31, 1993. These data were compared to findings from previous years to identify general trends and make preliminary comparisons. This study was conducted in the upper Yakima basin between Roza and Keechelus dams. Most of the work was conducted in seven sections of the mainstem of the Yakima River and in twelve tributaries of the Yakima River. 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).

Chapter 1

Rainbow and steelhead trout temporal and spatial spawning distribution in the upper Yakima River basin

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Abstract

The temporal and spatial spawning distribution of resident rainbow trout and anadromous steelhead trout was determined in the upper Yakima River basin by trapping, radio telemetry, electrofishing, redd surveys, and snorkeling. In addition, steelhead trout redds in the Naches River and Satus Creek were physically characterized to see if differences existed between the redds of the two forms of *Oncorhynchus mykiss* in allopatry. Rainbow trout spawned in all sampled reaches of the upper Yakima River basin from February through June, with the peak of spawn timing positively related to elevation. The spawn timing of steelhead trout overlapped that of rainbow trout, however steelhead trout spawned over a more restricted geographic area than that of rainbow trout. Rainbow trout redds in the mainstem Yakima River were significantly smaller than steelhead trout redds measured in two small tributaries of the Yakima River. Differences in redd sizes between locations may have been due to differences of fish size or differences in the physical conditions of the streams. Although the two forms of *O. mykiss* have unique life histories, they spawn at similar times and similar geographic locations in the upper Yakima basin. Furthermore, it appears that the two forms utilize similar spawning habitat. As a result of these similarities, the potential for interbreeding between the two forms of *O. mykiss* appears probable.

Introduction

Rainbow trout (*Oncorhynchus mykiss*) inhabit rivers from Mexico to Alaska (Wydoski and Whitney 1979) and can migrate to the sea (anadromous) as steelhead trout or complete their entire life infreshwater as rainbow trout (Neave 1943). Steelhead trout migrate to the ocean to grow before returning to spawn in their natal stream, while rainbow trout complete their entire life cycle in fresh water (Wydoski and Whitney 1979). Although there are differences in the life history patterns, the temporal and spatial spawning habits of the two forms of this species in sympatry are similar (Neave 1943). Both forms spawn in the spring and spawning may be related to water temperature and discharge (Orcutt et al. 1968; Erman and Hawthorne 1976; Thurow and King 1994). Within an individual river it is suggested that spawning occurs earlier in low elevation and later in high elevations which may be attributed to water temperature (Pearsons et al. 1993). Although the two forms may spawn in both mainstem river sections and in tributaries (Erman and Hawthorne 1976), large salmonids typically spawn in faster, deeper water with larger substrate than small salmonids (Crisp and Carling 1989; Ottaway et al. 1981). Furthermore, it is well documented that various redd attributes of the Salmonidae are related to the size of the spawning fish (Crisp and Carling 1989; Ottaway 1981) but that habitat availability may influence the dimensions of individual redds (Ottaway 1981).

The primary goal of this study was to determine the spawn timing and location of *O. mykiss* in the Yakima River basin, Washington. Our hypothesis was that rainbow trout temporal spawning distribution was related to elevation, julian date and water temperature. A secondary goal was to describe rainbow trout spawning within the mainstem river and tributaries. Additional research questions were: Are physical characteristics of redds different among resident and anadromous life history forms, and can a model be developed to discern the redds of the two forms spawning in sympatry?

Due to annual differences in sampling designs and techniques the methods and results sections of this chapter have been combined. The results are complementary but the method of data collection has varied between years. Results and analyses contained within this chapter should be considered preliminary.

Study Area

Spawn timing and location of *O. mykiss* was investigated in seven sections of the upper mainstem Yakima River between Roza Dam(rkm 180) and Easton Dam (rkm 326), and in thirty five study sections in 13 tributaries of the upper Yakima River (Refer to Figure 1 in chapter 4) (Hindman et al. 1991, McMichael et al. 1992, Pearsons et al. 1993). Elevations above sea level (referred to as "elevation") at the midpoint of the sections ranged from 390 m to 695 m in the mainstem and 451 m to 975 m in the tributaries. Study sections reflected the length of stream available to anadromous *O. mykiss* (chapter 1).

In addition to the spawn time and location, rainbow trout redds were characterized in the mainstem of the upper Yakima River and steelhead trout redds were characterized in Satus and Buckskin creeks. Buckskin Creek is a second-order tributary of the Naches River that enters the Yakima River at rkm 161. In the section of Buckskin Creek that redds were measured in, elevation was 500 m, gradient was less than 1%, and mean stream width was 1.9 m. Satus Creek is a fourth-order tributary to the Yakima River and enters the Yakima River below the town of Yakima at approximately rkm 140. The section that redds were measured in was 610 m above sea level, mean stream width was 5.6 m, and gradient was estimated to be 2%.

Methods and Results

Electrofishing

Electrofishing was used to locate sexually mature *O. mykiss* in the upper Yakima River and it's tributaries from February through June during 1990, 1991, 1992, and 1993. In each tributary, a sample was collected using a backpack-electrofisher in a low, middle and high elevation stream section. Sample sizes were usually 10 to 30 adult-sized rainbow trout per section (Hindman et al. 1991; McMichael et al. 1992). In the mainstem, a driftboat electrofisher was used to collect fish at least once per month (McMichael et al. 1992; Pearsons et al. 1993). Regardless of gear type, collected fish were anesthetized and checked for spawning condition by gently squeezing the abdomen with thumb and forefinger to see if ova, milt or resorbing fluids could be extruded (McMichael et al. 1992). Sexually mature fish were defined to be those that exuded either milt or ova.

If an adult steelhead trout (*O. mykiss* > 51 cm) was collected by electrofishing in a tributary after March 1, or in the mainstem Yakima River after March 30, it was defined as a spawner and information on spawn time and location was recorded. We assumed that steelhead collected in a tributary after March 1 would subsequently spawn within that tributary. March 30 was used as a cut-off date for the mainstem because our previous work revealed no sexually mature (ripe) steelhead in the Yakima River before March 30 (Hindman et al 1991; McMichael et al. 1992; Pearsons et al. 1993).

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

Rainbow trout spawned throughout all sampled reaches of the upper Yakima River basin with the possible exception of the highest elevation portions of some tributaries (Figure 1). Between 1990 and 1993, the earliest date that sexually mature rainbow trout were collected was February 1 and the latest date was June 28, although some were collected during fall sampling. The peak and range of rainbow trout spawn timing varied between years and locations (Figures 2 and 3). The peak of rainbow trout spawn timing based on electrofishing was positively related to elevation in tributaries and mainstem sections:

$$PST_r = 0.101 \times (E) + 34.5 \quad (N = 30,\ r^2 = 0.45,\ P = 0.00006)$$

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

Steelhead spawned throughout the upper basin although 69% (N=20) of the observations were in tributaries (Figure 4). Steelhead spawned from February 28 to May 21 with most activity occurring during April and May (Table 1). The spawn timing for individual steelhead in the mainstem and tributaries of the Yakima River from 1990 through 1993 was positively related to elevation:

$$PST = 0.033 \times (E) + 53.2 \ (N = 27,\ r^2 = 0.20,\ P = 0.0172)$$

Where PST_s is the time of individual steelhead spawn timing measured in Julian days and E is the elevation measured in meters.

Based on samples collected by electrofishing, spatial and temporal overlap of rainbow trout and steelhead spawning distributions was high. Except for one steelhead observation in Iron Creek, a tributary of upper Swauk Creek (Marc Divens, United

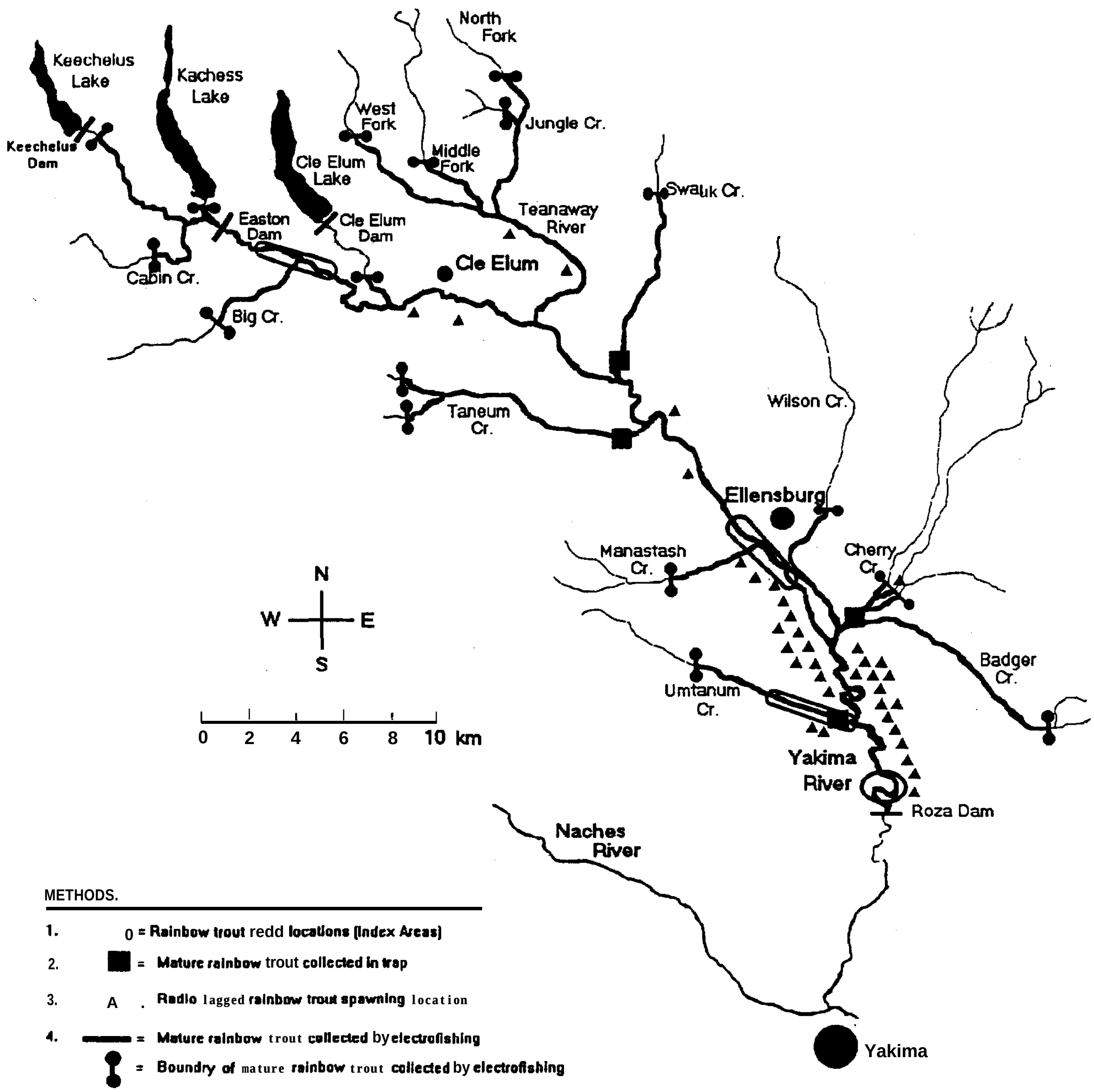

Figure 1. Spawning distribution of rainbow trout within the upper Yakima River basin. Data were collected from 1990 through 1993.

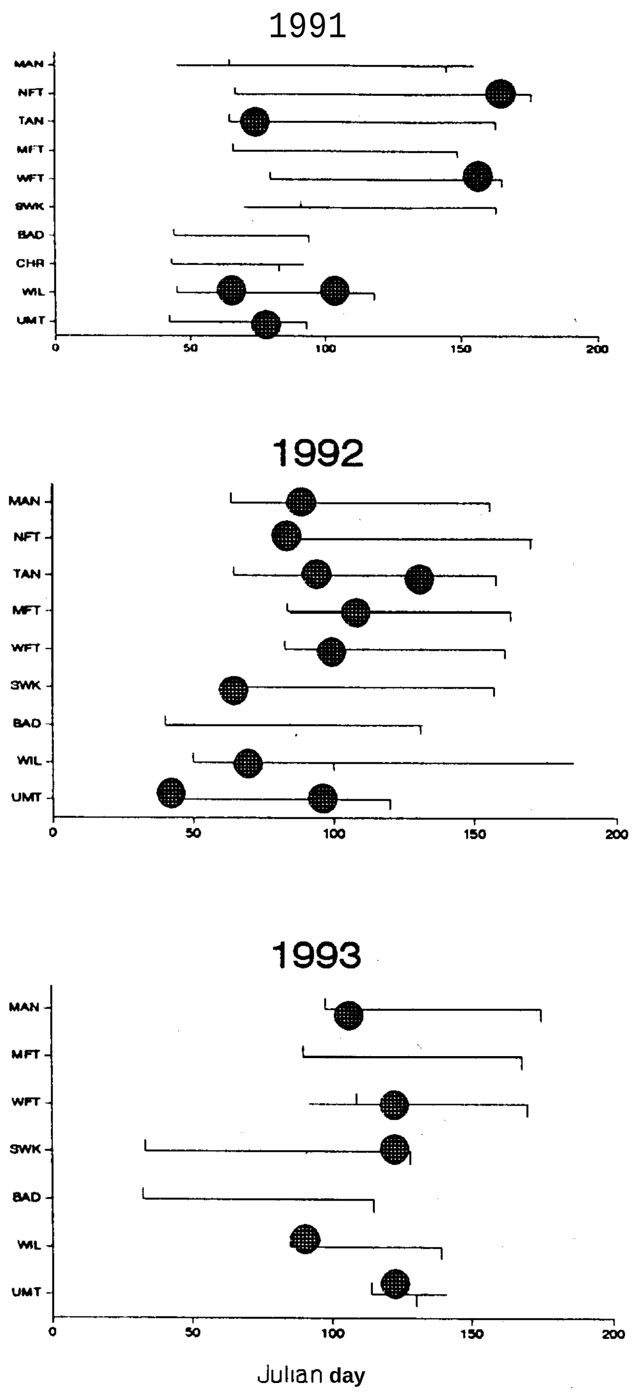

Figure 2. Estimated range and peak of rainbow trout spawn timing in tributaries of the upper Yakima River from 1991 to 1993. Horizontal bar represents entire sampling period, vertical bars represent range of days that mature fish were collected, and solid circles represent estimated peak of spawning for each tributary. Tributaries are arranged on the Y-axis from low (bottom) to high (top) elevation. UMT=Umtanum, WIL=Wilson, CHR=Cherry, BAD=Badger, and SWK=Swauk, TAN=Taneum, and MAN=Manastash creeks; WFT=West Fork, WFT=West Fork, and NFT=North Fork of the Teanaway River.

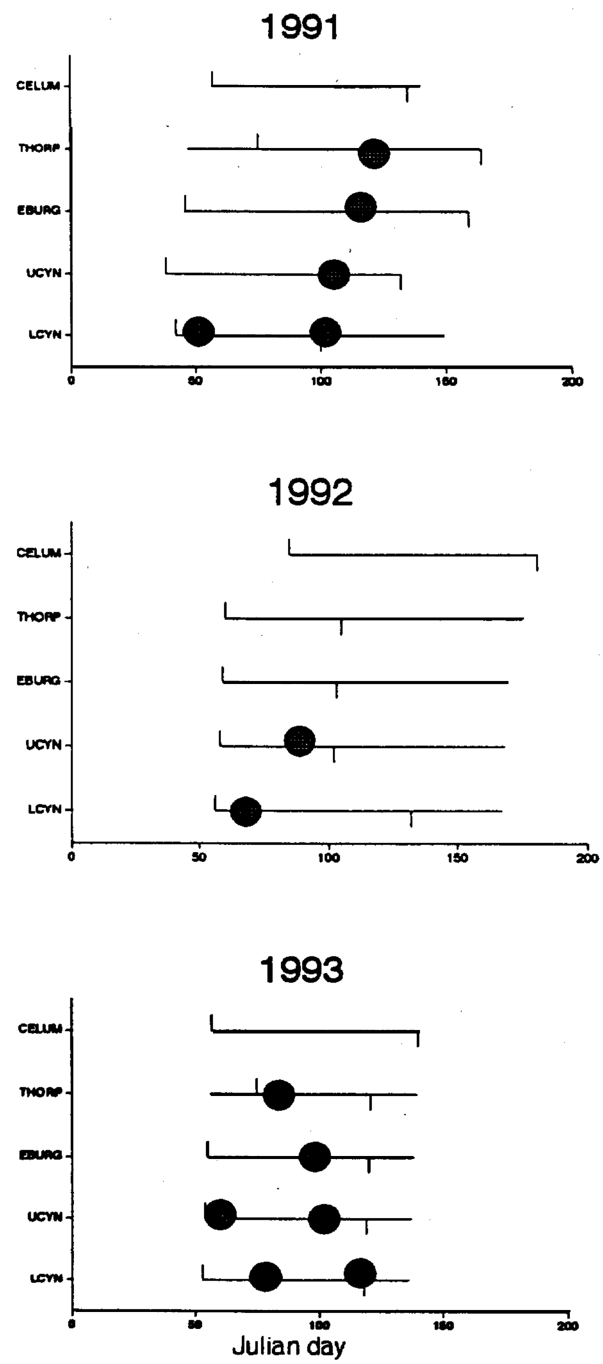

Figure 3. Estimated range and peak of rainbow trout spawn timing in the mainstem of the upper Yakima River from 1991 to 1993. Horizontal bar represents entire sampling period, vertical bars represent range of days that mature fish were collected, solid circles represent estimated peak of spawning for each mainstem section. Sites are arranged on the Y-axis from low (bottom) to high (top) elevation. LCYN=Lower Canyon, UCYN=Upper Canyon, EBURG=Ellensburg, and CELUM=Cle Elum section.

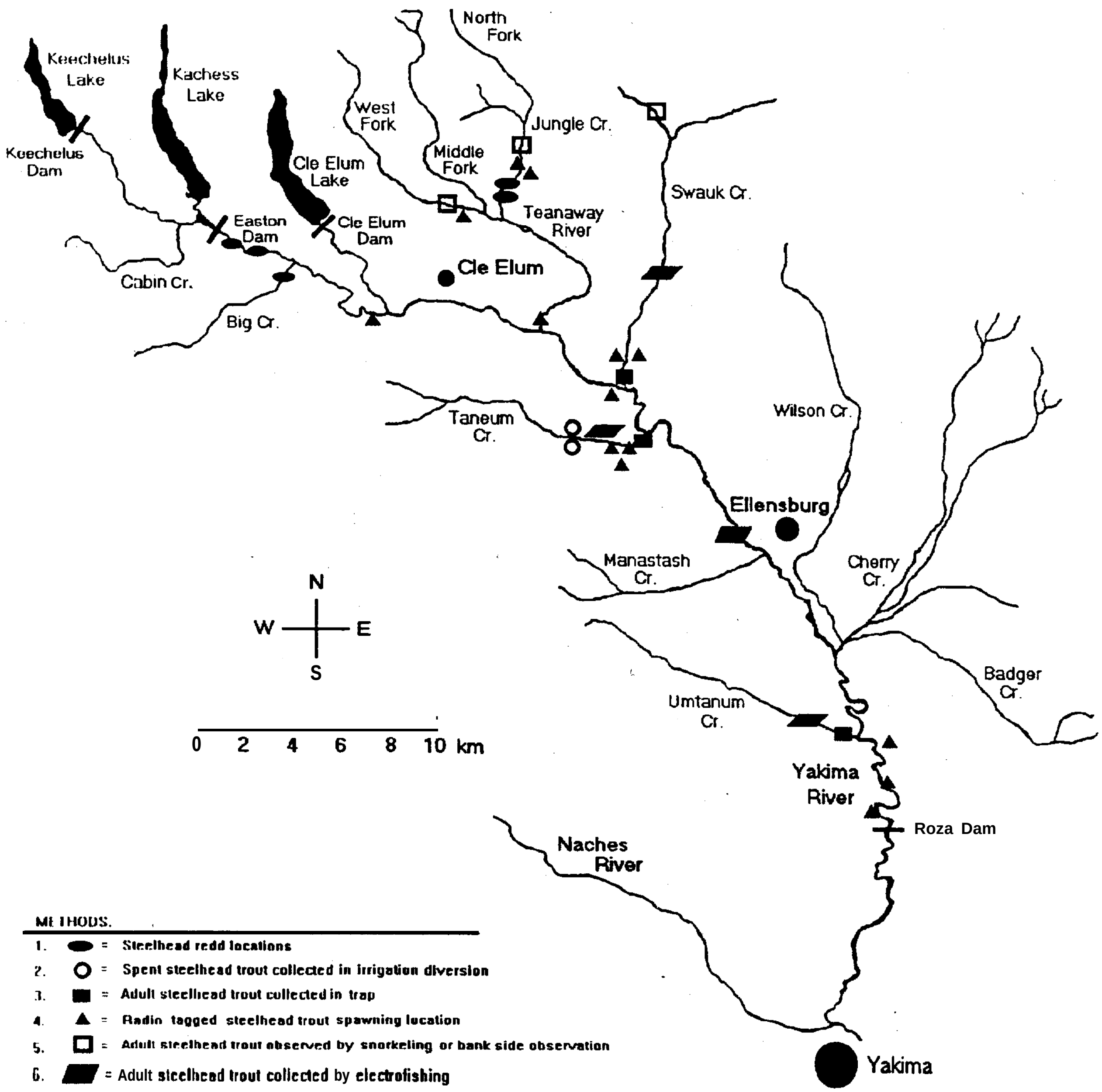

Figure 4. Spawning distribution of steelhead trout within the upper Yakima River basin. Data were collected from 1990 through 1993.

Table 1. Temporal spawning distribution of steelhead trout in the upper Yakima River Basin from 1990 to 1993. Data were collected by snorkeling, redd sitings, bank side observations, radio telemetry, and indirectly by electrofishing and traps,

Year	Steelhead Spawning Location	Method	Estimated Spawning Date	Julian Date	Elevation(m)
1990	Yakima River Lower Canyon (LCYN)	Radio telemetry	5/8	129	436
	Yakima River Lower Canyon (LCYN)	Radio telemetry	5/9	128	436
	Umtanum Creek	Electrofishing	3/21	90	470
	North Fork Teanaway River	Radio telemetry	5/15	135	714
	West Fork Teanaway River	Bank side observation	5/20	140	714”
				Mean = 124	
1991	Yakima River (CELUM section)	Redd siting	4/30	120	,641
	Yakima River (CELUM section)	Redd siting	4/30	120	641
	North Fork Teanaway River	Redd siting/snorkeling	5/8	128	714
	North Fork Teanaway River	Redd siting	5/8	128	714
				Mean = 124	
1992	Yakima River Lower Canyon (LCYN)	Radio telemetry	3/31	91	436
	Yakima River (CELUM section)	Radio telemetry	4/15	106	610
	Yakima River (CELUM section)	Radio telemetry	3/31	91	610
	Yakima River (EBURG section)	Electrofishing	2/28	59	506’
	Umtanum Creek	Trapping	3/31	91	470
	Swauk Creek	Electrofishing	3/6	66	580
	Iron Creek	Bank side observation	7/2	183	996[a]
	Taneum Creek	Electrofishing	5/6	127	622[a]
	North Fork Teanaway River	Snorkeling	5/13	124	714
	Big Creek	Redd siting	5/16	137	677
				Mean = 108	
1993	Yakima River (THORP section)	Radio telemetry	4/23	113	555
	Taneum Creek	Trapping	5/21	141	622
	Taneum Creek	Trapping	5/21	141	622
	Taneum Creek	Trapping	5/21	141	622’
	Swauk Creek	Trapping	5/20	140	580[a]
	Swauk Creek	Trapping	5/20	140	580[a]
	Teanaway River	Radio telemetry	4/23	113	686’
	North Fork Teanaway River	Radio telemetry	4/23	113	686
	North Fork Teanaway River	Snorkeling	5/5	125	686
	West Fork Teanaway River	Radio telemetry	5/21	141	714
				Mean ▫ 131	

‘Estimated spawn time and elevation based on the collection of steelhead trout that were not ripe, All other estimates are based on redd sitings, radio telemetry, or the collection of ripe steelhead by electrofishing or trapping.

States Forest Service, personal communication), the entire observed spawning distribution of steelhead was within that of rainbow trout, although rainbow trout did spawn over a much larger area. Steelhead spawning also occurred at times when rainbow trout were sexually mature.

Trapping

Two-way "V", "W" or picket weir traps were used to trap upstream and downstream migrating fish in Umtanum, Cherry, Wilson, Swauk and Taneum creeks. Trapping occurred in 1992 and 1993 in Umtanum Creek, in 1992 in Cherry and Wilson creeks, and in 1993 in Swauk and Taneum creeks. Traps were located within 1 rkm of the mouth of each tributary to reduce the possibility that fish spawning in lower reaches were undetected. Water temperature (°C), stream gauge height (mm), and date were recorded daily at each trap. In addition, fish length (mm FL), weight (g), direction of travel, maturity and sex if it could be determined, were recorded for each salmonid captured. Each rainbow trout greater than 175 in length was tagged with a serially numbered anchor tag (McMichael et al. 1992; Pearsons et al. 1993). In 1992, two sexually mature rainbow trout were captured in the Cherry Creek trap during the last week of February. One sexually mature male rainbow trout was collected in the Wilson Creek trap on February 25, 1992.

Except for the Umtanum Creek trap in 1992, each of the traps was either destroyed by high water discharge and debris loads, or was otherwise not effective for up to 10 days due to water flowing over and around them. As a result, we were unable to census the entire migrating rainbow trout population into these tributaries in most years. Therefore, little information about the temporal spawning distribution, except a range of dates that sexually mature rainbow trout were captured, can be interpreted from the trap data. The only complete data set is from Umtanum Creek in 1992. Trapping efficiency for steelhead trout, however was much higher. Adult steelhead trout were captured migrating into Umtanum Creek in 1992, and in Taneum and Swauk creeks in 1993.

Radio Telemetry

A substantial data set on steelhead and rainbow trout spawning has been provided by the National Marine Fisheries Service (NMFS) and this subsection of the report is an abbreviated summary of their work. NMFS has been studying movements associated with spawning in the upper Yakima River since 1989 with the use of oral radio transmitters (Eric Hockersmith, NMFS personal communication). All results of the radio telemetry study were provided by Eric Hockersmith, and will be reported in more detail by NMFS. In 1993, we and NMFS used radio telemetry in a collaborative effort to determine where and when rainbow trout tagged in the canyon section of the upper Yakima River spawned. Fifty rainbow trout greater than 300 g were captured in the two canyon sections of the Yakima River (rkm 209 and 225, respectively) on March 5 and March 8, surgically implanted with a 10 g radio transmitter and tracked for eight months (surgical procedures are reported in Chapter 2). Briefly, if a trout moved some distance from it's initial tagging location, remained stationary for more than 2 days and then returned to its tagging location, that date and location was defined as its spawn time and location (Eric Hockersmith, NMFS personal communication).

A total of 10 steelhead trout were also radio tracked in the upper Yakima River basin from 1990 to 1993 (Eric Hockersmith NMFS, personal communication). Spawning by steelhead trout, based on radio telemetry, occurred between March 31 and May 21, 1989 through 1993. Sixty percent of the radio tagged steelhead spawned in the mainstem of the Yakima River while the remaining 40% spawned in the Teanaway Basin (refer to Table 1). Based on radio tracking from 1989 to 1993, all steelhead spawning in the mainstem river occurred before May 10, while 75% (3 of 4) steelhead spawned in the Teanaway Basin after May 1. Similarly, radio tagged rainbow trout spawned between April 1 and May 8, 1993. The majority of radio tagged rainbow trout spawned in the canyon section (65%) of the mainstem Yakima River, while 28% migrated upstream out of the canyon section, presumably to spawn elsewhere in the mainstem river or it's tributaries. Five percent of the radio tagged rainbow trout spawned in Umtanum Creek. The remaining fish (2%), spawned in Cherry Creek. Of the 13 (28%) trout that migrated upstream out of the canyon, two spawned within the Teanaway Basin and 11 spawned in the mainstem of the Yakima River.

Redd surveys

In 1993, redd surveys were conducted in index sites within each of three elevational strata of the mainstem Yakima River to determine the spatial spawning distribution of rainbow trout. Index sites ranged from 6 to 9 km long and were established in locations that were presumed to have high quality spawning habitat. Surveys were conducted weekly by canoeing or rafting the river along the bank that appeared to be most suitable for rainbow trout spawning based on visual inspection of gravel composition and water velocity. Surveys began on April 15 and continued through May 24. Redds were identified by the presence of clean substrate, a corresponding depression in the stream bottom (pot) and a mound of substrate behind the depression (mound) (Ottaway et al. 1981). Although some of the redds may have been constructed by steelhead, only 30 adult steelhead were estimated to be present in the entire Yakima River above Roza Dam in 1993 (Joel Hubble YIN, personal communication). As a result of the small number of steelhead present, we assumed that all redds observed were constructed by rainbow trout. In support of this assumption, no steelhead were observed during any redd survey.

The peak of rainbow trout spawn timing was calculated using the temporal distribution of redd construction in three index sites. By dividing the number of new redds by the number of days elapsed since the previous survey, the number of redds per day was determined. Since the length of each index site was different, the number of redds per day was then divided by the survey site length. The resultant number was the number of redds per day per kilometer. Because redds could only be identified for 14 days after they were constructed, 14 was used as the divisor if more than 14 days elapsed since the previous survey. The peak of spawn timing based on redd surveys was determined by visual inspection of plotted data.

A total of 172 redds were identified in six sections of the mainstem Yakima River between April 15 and May 24, 1993 The peak time of rainbow trout spawning based on redd surveys in three index sites of the Yakima River was positively related to elevation.

In the lowest section (section 1) the peak of spawn timing occurred on April 20, in the middle section (section 3) the peak occurred on April 23, and in the highest section (section 6), the peak occurred on April 27 (Figure 5). High turbidity on April 24 precluded further surveys in the lowest elevation section.

On April 30, 1991, an incidental spawning survey was conducted by helicopter over the mainstem Yakima River in an attempt to locate steelhead redds. Steelhead redds were identified as described above but were generally larger. In addition, before recording it as a steelhead redd, we required that a steelhead must also have been observed on or near the redd (McMichael et al. 1992). Only two redds were seen and both were in the Yakima River near the town of Cle Elum.

To describe the temporal and spatial spawning distribution of rainbow trout within Umtanum Creek, redd surveys were conducted by students from Central Washington University (CWU) in 1991, 1992 and 1993. Surveys were conducted weekly over an average of six weeks and redds were identified by the presence of clean substrate and typical morphology as described above. Information from these surveys are reported here with reference and permission from their authors and CWU (Paul James CWU, personal communication).

The temporal distribution of rainbow trout spawning in Umtanum Creek was from March 25 through April 18 in 1991, from March 13 through April 21 in 1992, and from April 8 through April 28 in 1993. The peak of spawn timing was April 8 in 1991 and March 31 in 1992. In 1993, no peak was identified because redds could not be observed due to high discharge and turbidity from the first of March through April 5.

Redd characterization

In addition to describing the temporal and spatial distribution of rainbow trout spawning, we recorded the physical characteristics of 73 redds that did not have fish located on or adjacent to them. The redd measurement and characterization terminology we followed was similar to that of Ottaway et al. (1981) but the locations at which some of the measurements were made were different (Figure 6). Length and width of the redds (disturbed gravel), water depth over the tail of the redd, at the deepest point in the pot and at one point adjacent to the redd were measured. Adjacent water depth was used to estimate the water depth at the site prior to redd construction.

Surface water velocity in meters per second (m/sec) at the transition point between the pot and mound, water velocity at 60% of the water depth, and bottom water velocity were measured with a Marsh McBirney model 201D portable water current meter. Velocity was also recorded at 60% of the water depth at the leading edge of the redd which was assumed to provide a reasonable estimate of water velocity at the site prior to redd construction.

In addition to the physical measurements, the following descriptive information for each redd was recorded:

1) size composition of gravel in the pot and tail;
2) distance to a)stream bank, b)overhead cover, c) turbulence and d)boulders;
3) habitat type (run, riffle, tail of pool) that redd was located in;
4) presence in side channel or main channel;
5) minimum distance to nearest redd;
6) redd maturity (0 = rainbow trout present on redd, 1 = no fish present but redd possessed typical morphology and was composed of clean gravel);
7) distance to main channel/distance to side channel; and,
8) channel width.

Redd locations were marked by attaching biodegradable orange flagging on a tree adjacent to the redd. The redd number and date were written on the flag to prevent remeasurements of previously measured redds.

No confirmed steelhead trout redds were observed in the upper Yakima River presumably because the steelhead population in the upper Yakima River was low, averaging less than 50 adults per year since 1988; the number of adult steelhead that passed Roza Dam (RK 180) for run year 1992-1993 was 30 (Joel Hubble, personal communications, Yakima Indian Nation). Because of the low number of steelhead present in the upper Yakima River, steelhead redds were characterized in the lower Yakima Basin (Satus and Buckskin creeks) where steelhead were more abundant. Satus and Buckskin creeks are small-order tributaries of the lower Yakima River. Redd characterization measurements were the same as those reported for rainbow trout redds. These data were used to compare the physical attributes of rainbow(trout) redds in the mainstem Yakima River versus(known) steelhead trout redds. All steelhead redd measurement data is reported in Appendix 1A.

A total of 73 rainbow trout redds were measured and characterized in three index and four additional sections of the mainstem Yakima River (measurement data is reported in Appendix 1B). The average area of these redds was 1.8 m^2. The adjacent water depth was an average depth of 0.30 m and the average water velocity upstream of the redds was 0.60 cm/s (Table 2).

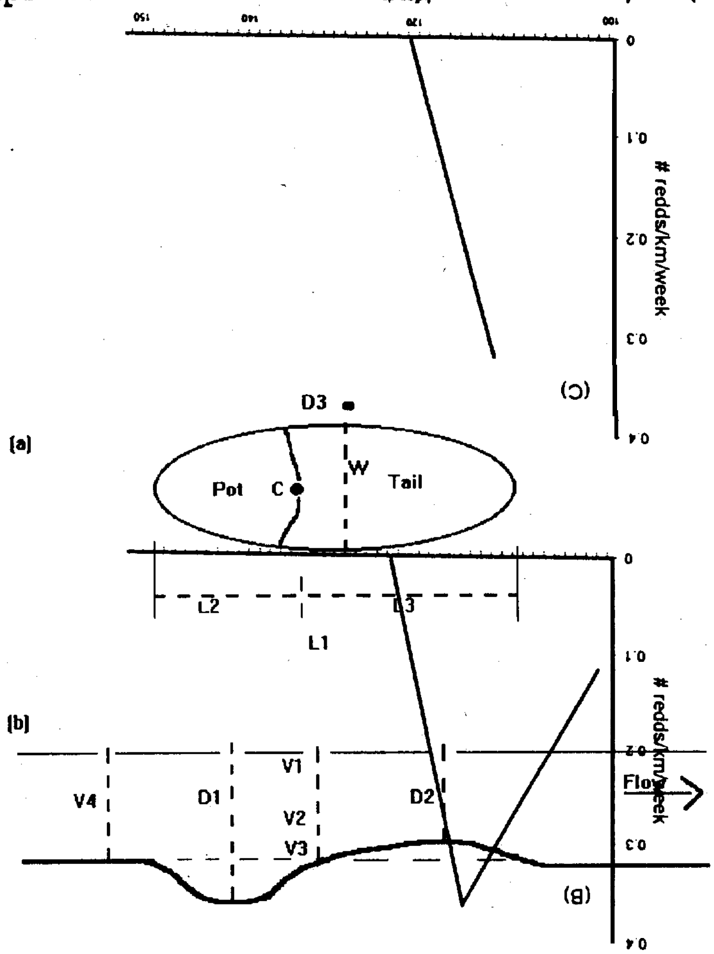

Figure 6. (a) Plan and (b) longitudinal section of a redd, showing the dimensions measured and positions of measurements . L1 = total length of the redd(m), L2 = pot (m) and L3 = tail (m) ; W = width of the redd (m); D1 = water depth in the pot (m), D2 = tail and D3 = adjacent depth (m); V1 = water velocity at surface(m/s), V2 = 60% depth, V3 = bottom and V4 = leading edge; and C = center of redd from which distance to hiding cover and bank were measured from (m).

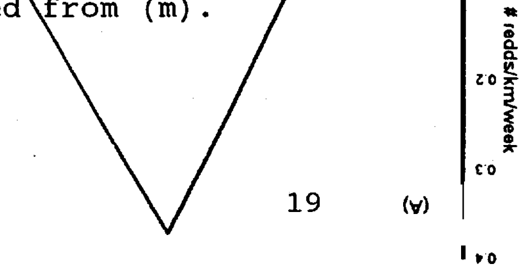

Table 2. Average physical measurements of rainbow trout redds in the mainstem Yakima River, 1993 (N = 73). All measurements are in meters unless specified otherwise.

	Area	Length	Width	Water Depth (m)			Velocity (m/s) @ 0.6 depth		% Substrate (Pot) by size class'			% Substrate (Tail) by size class'		
				bowl	tail	side	surface	head	1	2	3	1	2	3
Avg.	1.8	1.94	0.89	0.4	0.3	0.3	0.7	0.6	12.5	12.0	55.0	6.6	17.1	67.4
(SD)	(0.6)	(1.0)	(0.3)	(0.1)	(0.1)	(0.1)	(0.2)	(0.2)	(11.7)	(11.5)	(21.9)	(5.4)	(15.1)	(21.2)

* 1< 3 mm diameter; 2 = 3 mm to 1.3 cm diameter; 3 = 1.3 cm to 6.4 cm diameter
The remaining substrate was greater than 6.4 cm diameter.

Redds were often constructed near organic debris greater than 25 cm long, and 99% of the redds were constructed in run habitat. Although side channel habitat was not as abundant as main channel habitat, 53% of the rainbow trout redds were located in or within 25 m of a side channel. The majority of rainbow trout redds were within 6 m of other redds, and in several cases redd superimposition occurred.

Comparisons were made between the 73 rainbow trout redds measured in the mainstem Yakima River and 38 rainbow trout redds that were measured and characterized in Umtanum Creek. In addition, the 73 rainbow trout redds in the mainstem Yakima River were compared to 15 steelhead trout redds that were measured and characterized in Satus and Buckskin creeks.

The length and width of steelhead redds in Satus and Buckskin creeks were significantly greater than rainbow trout redds in the mainstem Yakima River, and rainbow trout redds in Umtanum Creek were significantly smaller than rainbow trout redds measured in the Yakima River (Table 3). Based on these comparisons, there appeared to be a gradient of redd sizes from large to small for steelhead, rainbow trout in the Yakima River, and rainbow trout in Umtanum Creek, respectively (Table 3). In addition, water depth over the mound of steelhead redds was less than that of rainbow trout redds, suggesting that steelhead excavate a deeper pot, which results in a taller mound.

Snorkeling

Snorkeling methods were used incidently in all years to locate steelhead and rainbow trout redds and spawning activity in the mainstem of the Yakima River and the Teanaway River basin. Observations of adult steelhead in the mainstem after March 31 were assumed to reflect spawning dates and locations. In addition, incidental observations of steelhead redds in tributaries were made during other field activities (e.g. trapping, electrofishing, behavioral observations).

Snorkeling surveys in the mainstem Yakima River resulted in few observations of steelhead spawning. The only steelhead observed while snorkeling were in the North Fork of the Teanaway River on April 13, 1992, and again in the same river on April 5, 1993 (refer to Table 1).

Table 3. Physical measurements of rainbow trout redds in the mainstem of the Yakima River (N=73) and Umtanum Creek (N=38), and steelhead trout redds in Buckskin and Satus creeks (N=15) and results of t-tests among groups.

Parameter	Redd Group			
	Yakima River	Umtanum Cr.	t	P
Velocity at 60% depth in center of redd (m/s)	0.56	0.26	10.8	0.00’
Area of redd (m)	1.76	0.46	8.4	0.00’
Redd length (m)	1.94	0.96	9.2	0.00’
Redd width (m)	0.89	0.45	8.4	0.00*
Adjacent water depth (m)	0.29	0.19	6.0	0.00*
	Yakima River	Buckskin/Satus		
Redd area (m^2)	1.76	3.1	5.0	0.00’
Redd width (m)	0.89	1.26	4.4	0.00’
Redd length (m)	1.94	2.41	2.9	0.00’
Water depth in pot (m)	0.37	0.38	-0.16	0.88
Water depth at mound (m)	0.26	0.18	2.68	0.01'
Adjacent water depth (m)	0.29	0.28	0.35	0.73
Water surface velocity (m/s)	0.66	0.70	-0.83	0.41
Water velocity at 60% depth in center of redd (m/s)	0.56	0.55	0.06	0.95
Water velocity at bottom (m/s)	0.33	0.34	-0.15	0.88
Water velocity at leading edge of redd (m/s)	0.59	0.55	0.77	0.44

* significant difference ($P < 0.05$).

Discussion

Based on findings from five distinct, complementary methods, complete temporal and nearly complete spatial overlap appear to have occurred between resident and anadromous *O. mykiss* in the upper Yakima River basin (Figure 7). The collection of sexually mature rainbow trout from all areas sampled, and the presence of redds in all mainstem sections sampled indicated that the spatial spawning distribution of rainbow trout in the upper Yakima River basin was large. The spatial distribution of spawning steelhead trout however, appeared more limited. Sexually mature and spawning steelhead trout were collected primarily in tributaries, which may be a result of sampling and detection capability. Of 29 sexually mature steelhead observed, six (21%) were captured by trapping, and these traps were used only in tributaries, thus biasing the results towards steelhead spawning in tributaries. As a result, the percentage of steelhead spawning in the mainstem of the river may be greater than the number presented. This sampling bias is supported by the results of radio telemetry studies which indicated 60% of the radio tagged steelhead spawned in the mainstem of the river. Radio telemetry may be less biased than trapping methods because it can discern spawning activity and location independent of stream size.

Spawn timing of both rainbow and steelhead trout in the Yakima River basin was positively correlated to elevation. Because spawn timing is related to stream temperature (Thurow and King 1994), elevation may be a surrogate for stream temperature. This within-year relationship between spawn timing and stream temperature was substantiated by between-year spawn timing data. The peak of rainbow trout spawn timing was generally earlier in 1992 than in 1991 or 1993. This may be attributed to variation in environmental conditions. The winters of 1990-91 and 1992-93 were colder than the winter of 1991-92, and colder water temperatures resulted in later rainbow trout and steelhead spawn timing in the springs of 1991 and 1993.

Identifying the peak of rainbow trout spawn timing based on electrofishing methods was difficult in some tributaries due to the extended time in which sexually mature fish were observed. Within several tributaries, numbers of sexually mature male rainbow trout were collected on every survey, resulting in temporal spawning distributions with no discernable peak. Bernier et al. (1993) reported that in the genus *Oncorhynchus*, precocious maturation of male parr is common. In addition, Berglund et al. (1992) reported that spermatogenesis for the genus may be renewed over the summer, resulting in repeated maturation. Because male rainbow trout were sexually mature throughout the spring and summer it was difficult to identify a definite peak of spawn timing based solely on the collection of sexually mature males. On the other hand, determination of spawning time based on the collection of mature females was not possible because of the short period of time in which an individual females will exude gametes. As a result we observed very few sexually mature females. One way to circumvent this problem would have been to sample fewer tributaries more intensively to increase sample sizes of sexually mature females. This was not done because of the possible negative effects that

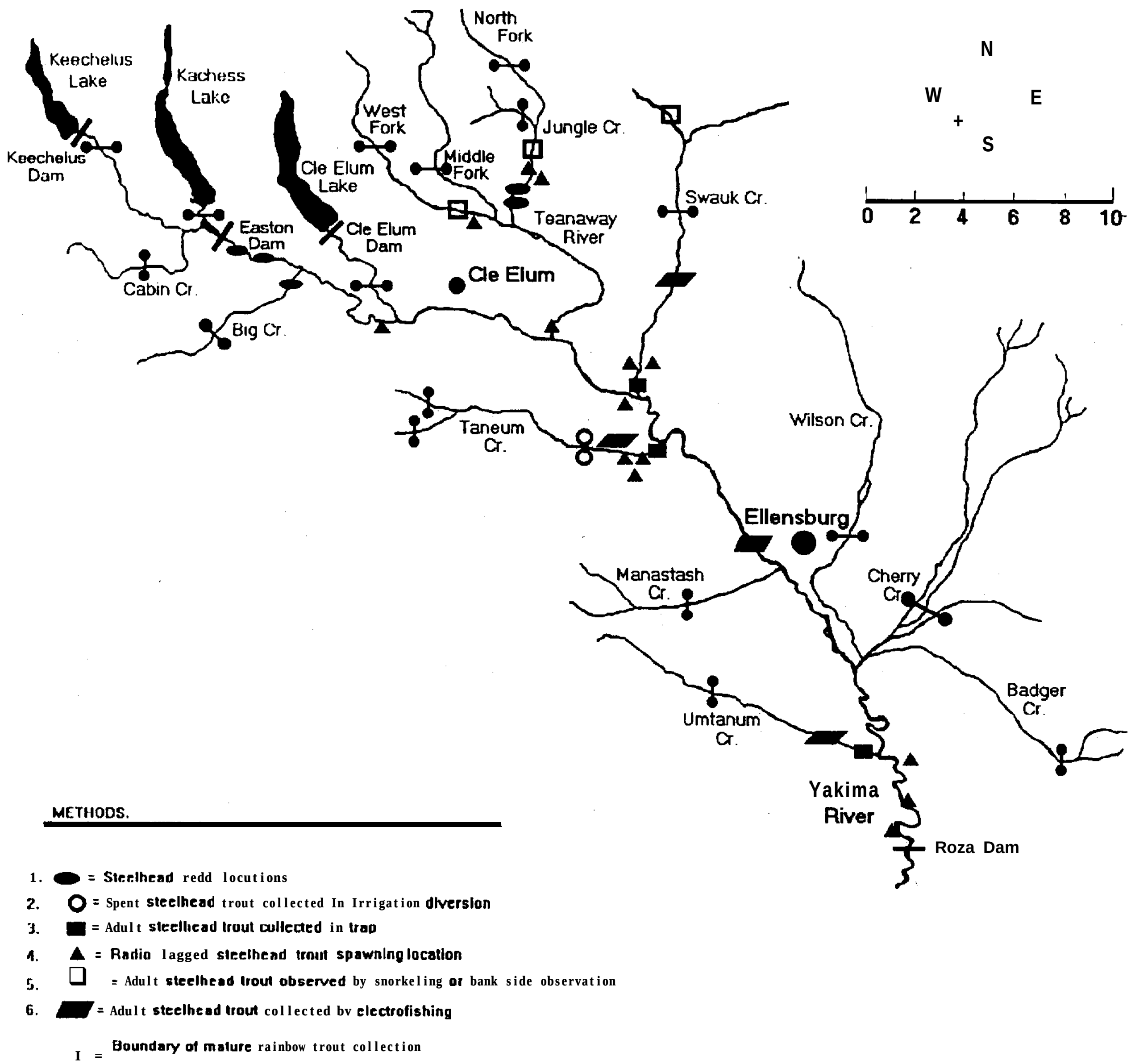

Figure 7. Spatial spawning distribution of rainbow and steelhead trout in the upper Yakima basin. Steelhead spawning data were from 1990 through 1993, while rainbow trout data represents 1991 through 1993.

electrofishing may have on fish (McMichael 1993) or on gametes, zygotes (Godfrey 1957) or alevins. Also, samplers wading in the stream may damage eggs in the gravel (Roberts and White 1992).

With the exception of Umtanum Creek, determining peak spawn timing in tributaries based on the collection of sexually mature fish in traps was not possible. During the spring, stream flows can increase dramatically in a single day and may be accompanied by debris, resulting in unknown but obviously poor trapping efficiency. Poor trapping efficiencies were further exacerbated by the fact that rainbow and steelhead trout typically move upstream into tributaries in the spring to spawn as discharge increases (Erman and Hawthorne 1976). Although traps were ineffective in capturing the entire spawning population that moved into individual tributaries, trapping did identify rainbow trout from the mainstem of the Yakima River that migrated into tributaries to spawn. In addition, steelhead trout spawners were trapped in Umtanum, Swauk and Taneum creeks, suggesting the importance of these creeks to the steelhead population and possibly also to the rainbow trout population.

Even when the composite of results from all methods used were reviewed, identifying the peak spawning time for steelhead was not possible. This was due to the small number of spawning or sexually mature adults observed. However, the dates reported or estimated for each individual fish, demonstrated that steelhead spawned earliest in lower and latest in higher elevation locations. This relationship may be attributed to water temperature. In general, steelhead spawned earlier in 1992 than in 1991 and 1993. A similar trend was observed for rainbow trout.

Rainbow trout redd surveys in the mainstem of the Yakima River seemed to provide more definitive spawn timing information than electrofishing methods. However, in the mainstem of the Yakima River rainbow trout and steelhead spawned in sympatry and confident differentiation of their redds was difficult. To circumvent this problem, a redd classification scheme was developed to assess the feasibility of differentiating redds based on physical measurements. Results indicated that 62% of the rainbow trout redds measured in the mainstem of the Yakima River were smaller in area, length, and width than the smallest steelhead redd measured in Satus and Buckskin creeks. In contrast, depth and water velocity were not significantly different between steelhead and rainbow trout redds. Thus, steelhead spawned in sites that were similar to sites used by rainbow trout, but their redds were generally larger.

The only difference between steelhead and rainbow trout redds, other than size was the depth of water over their mounds. This finding may have been influenced by the limited spawning habitat in Buckskin and Satus creeks resulting in redds with a deep pot and a tall (shallow) mound. In Satus and Buckskin creeks the amount of water that was deeper than 30 cm with a velocity greater than 0.2 m/s was limited. Or, in fact deep pots and shallow mounds may be a characteristic of steelhead redds. Although steelhead appear to construct redds near cover, this finding, again, may be attributed to stream size. The chance of a redd being located adjacent to cover is much greater the smaller the stream.

Although most of the physical attributes of steelhead and rainbow trout redds we measured in the Yakima River basin had similar characteristics, Ottaway et al. (1981) and Crisp and Carling (1989) documented that redd size and various redd dimensions were related to spawner length. If these relationships between fish length and redd size were applicable to both forms of *O. mykiss* in the Yakima River, then an approximation of fish size may be derived from redd size. Because rainbow trout were smaller than steelhead it may be possible to differentiate steelhead and rainbow trout redds based on redd size. This would be useful for determining the temporal and spatial distribution of steelhead spawning in the upper Yakima River based solely on redd measurements.

Physical characteristics of steelhead redds in Buckskin and Satus creeks were similar to those reported by Smith (1978) in the Deschutes and Rogue rivers of Oregon, by Orcutt (1968) in the Clearwater and Salmon rivers of Idaho, and by Coble (1961) in Oregon's Alsea River basin (Table 4). The most notable differences between characteristics of Satus and Buckskin creek steelhead redds and those from other studies are size and water velocity; steelhead redds in Satus and Buckskin creeks were considerably smaller than those reported in Table 4. This may be due to differences in the size of steelhead in Satus and Buckskin creeks compared to those reported in Table 4. The average size of steelhead in Satus and Buckskin creeks was 61 cm (Joel Hubble, YIN, personal communication), which is smaller than the steelhead reflected in the Clearwater, Deschutes and Rogue river studies. Although redd size can be related to fish size (Ottaway et al. 1981; Crisp and Carling 1989), the utilization of slower water velocity in Satus and Buckskin creeks may be attributed to limited availability of slow water habitat, although we did not measure this physical parameter.

Rainbow trout in the upper Yakima River mainstem spawned in water depths and velocities that were similar to those of rainbow trout in the mainstem of the Deschutes River. In the Deschutes River, (Smith 1978) reported that rainbow trout spawned in an average water velocity of 0.70 m/s and in an average water depth of 0.34 m. These values were similar to those we measured for rainbow trout in the Yakima River.

Table 4. Measurements of steelhead redds in Satus and Buckskin creeks in 1993. (see Appendices 1A and 1B for complete measurement data) compared to those for steelhead from other rivers in the Pacific Northwest.

Parameter	Average	Stream	Reference
Water Velocity			
	0.70 m/s	Deschutes R., Oregon	Smith 1978
	0.68 m/s	Rogue R., Oregon	Smith 1978
	0.70 - 0.76m/s	Clearwater & Salmon R., Idaho	Orcutt 1961
	0.55 m/s	Satus & Buckskin Cr., Washington	Current Study
Average depth			
	0.41 m	Deschutes R., Oregon	Smith 1978
	0.22 m	Rogue R., Oregon	Smith 1978
	>0.21 m	Clearwater R., Idaho	Orcutt 1968
	0.28 m	Satus & Buckskin Cr., Washington	Current study
Average size			
	5.4 m^2	Clearwater & Salmon R., Idaho	Orcutt 1968
	range (2.4 - 11.2 m^2)		
	3.1 m^2	Satus & Bucksin Cr., Washington	Current study
Predominant substrate (diameter)			
	1.3 < x < 10 cm	Clearwater & Salmon R., Idaho	Orcutt 1968
	silt < x < 7.5 cm	Alsea R., Oregon	Coble 1961
	1.3 < x < 6.4 cm	Satus & Buckskin Cr., Washington	Current study

Conclusion

In the upper Yakima River basin, interbreeding between rainbow trout and steelhead trout appears to be highly probable. The distribution of spawning rainbow trout and steelhead overlapped in both space and time, and incidental observations of interbreeding occurred. Except for one steelhead observation in Iron Creek, the entire spawning distribution of steelhead overlapped that of rainbow trout, although rainbow trout did spawn over a much larger area (Figure 7). Steelhead spawning also occurred at times when rainbow trout were ripe.

We have directly documented two instances of suspected interbreeding in Umtanum Creek. In 1992, a ripe female steelhead was trapped while migrating into Umtanum Creek and later was observed exiting the stream spent. This occurred near the peak of rainbow trout spawning activity in the creek. No other steelhead were observed to have entered the creek through the trap that year. In 1990, a spent female steelhead was collected adjacent to her redd in association with ripe male rainbow trout. No other steelhead were collected in Umtanum Creek during 1990.

Although it appeared that rainbow and steelhead trout spawned at similar times and within the same geographic area in the upper Yakima River basin, there were differences in physical attributes of redds among groups. Differences in redd size and habitat utilization between rainbow and steelhead trout may be related to differences in fish size or habitat availability between streams. Therefore, to improve comparisons of spawning habitat utilization, steelhead redds should all be measured within the upper Yakima River to standardize habitat availability for both rainbow and steelhead spawners. Until further work is completed, the results presented here should be considered preliminary and subject to change.

Acknowledgements

Steve Leider reviewed this manuscript, was involved in establishing the original sampling design and has been fundamental in the refining process that resulted in this study. We thank Eric Hockersmith for providing information on radio tagged fish, and the Yakama Indian Nation for providing information on steelhead redd locations. Geoff McMichael, Jim Olson, Marcia Fischer, Greg Strom, Andrew Murdoch, Nick Hindman, Karin Weiland, and John Long contributed significantly in the field collection of the data and refinement of field methods. We also thank Paul James and students of Central Washington University, including Ernie McKenzie, Chuck Wooley, Karin Weiland, Marc Divens and David Harper for information about rainbow trout redds in Umtanum Creek.

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APPENDIX **1A. Physical** measurements and characterization of steelhead trout redds in Buckskin and **Satus** creeks, 1993.

	REDD NUMBE	REDO DIMENSION (m)			WATER DEPTH (m)			WATER VELOCITY (m/sec.)				DISTANCE TO COVER (m) *					
DATE	redd # - cmk	length	width	area	pot	mound	side	surface	0.6 depth	bottom	herd	T	0	s	0	U	B
5/17	1-Buckskin	3.6	1.7	6.12	0.50	0.12	0.25	0.63	0.48	0.27	0.52	18.0	2 5	1.0	3.0	3.0	
5/17	2-Buckskin	3 0	1.1	3.30	0.48	0.12	0.27	0.79	0.63	0.41	0.72	2.0	3.0	1.0	5.0	8.0	10.0
5/17	3-Buckskin	2.8	1.6	5.04	0.46	0.16	0.26	0.63	0.60	0.46	0.63	3.0		3.0	0.1		
5/17	4-Buckskin	2.6	11	2.86	0.40	0.16	0.26	0.60	0.61	0.35	0.42	2 0		0.2	1.0	0.7	
Y17	5-Buckskin	1.8	1.0	1.80	0.20	0.17	0.21	1.30	0.71	0.55	0.45	0.0	4.0		2.0	2.7	
Y1?	6-Buckskin	2.1	1.3	2.73	0.32	0.22	0.28	0.78	0.57	0.37	0.69	1.0	3.0		5.0		
Y17	7-Buckskin	2.1	1.4	2.94	0.30	0.17	0.27	0.52	0.43	0.22	0.57	3.0	2 0		7.0		
Y17	8-Buckskin	2.0	1.0	200	0.25	0.10	0.20	0.49	0.48	0.42	0.47	0.5		2.0	5.0		
5/17	9-Buckskin	2.1	1.3	2.73	0.36	0.20	0.24	0.62	0.60	0.18	0.59		3.0	0.5	4.0	1.0	
5/18	1-Satus	2.6	1.1	2.86	0.32	0.15	0.28	0.36	0.40	0.31	0.22	1.5	5.0	4.0	0.7		3.0
5/18	2-Satus	2.3	1.0	2.30	0.40	0.32	0.36	0.76	0.57	0.31	0.62	1.5	3.5	4.5	7.0		8.0
5/18	3-Satus	2.6	1.4	3.64	0.50	0.32	0.44	0.70	0.70	0.40	0.30	1.5	1.8	1.5	7.0		9.0
5/18	4-Satus	2.3	1 4	3.22	0.28	0.14	0.28	0.56	0.45	0.18	0.47	1.5			0.0	4.0	1.5
5/18	5-Satus	2 0	1.1	2.20	0.46	0.28	0.32	0.91	0.66	0.35	0.90	3.0	3.0	0.0	3.7	4.0	0.0
5/18	6-Satus	2.3	1 2	2.76	0.38	0.18	0.3	0.88	0.7	0.3	0.72	2.8	3	11	6	4	20

. **T = turbulence, D = depth (>70 cm), S = submerged organic material (able to conceal 50 cm fish), 0 = overhead cover (able to conceal a 50 cm fish), U = undercut bank, B = boulder**

no entry indicates that the cover type was not available near the redd

APPENDIX 1 A (Continued). Physical measurements and characterization of steelhead trout redds in Buckskin and **Satus** creeks, 1993. All measurements are in meters unless stated otherwise.

REDD NUMBER	SUBSTRATE IN POT (%)				SUBSTRATE IN MOUND (%)				DISTANCE	DISTANCE TO	DISTANCE TO	HABITAT	SIDE	DISTANCE
redd # creek	[illegible]	5.13	1.3-6.4	>6.4 cm	<.5 cm	5-1.3	1 343.4	>6.4 cm	OTHER REDD	TO SIDE CHAN.	MAIN CHAN.	TYPE	CHAN.	TO SANK
1-Buckskin	5	10	20	65	10	10	80	0	17.0			Run	N	1.2
2-Buckskin	5	5	20	70	10	20	70	0	17.0			Run	N	1.6
3-Buckskin	10	10	30	50	5	15	70	10				Run	N	1.0
4-Buckskin	15	10	20	55	10	10	80	0				Run	N	0.6
5-Buckskin	0	10	70	20	0	10	70	20	6.0			Riffle	N	1.5
6-Buckskin	0	10	10	60	0	10	80	10	3.0			Riffle	N	1.1
7-Buckskin					10	10	80	0	0.1			Run	N	1.1
8-Buckskin	5	5	70	20	10	10	80	0				Riffle	N	1.0
0-Buckskin	15	5	10	70	15	10	70	5				Run	N	0.6
1-Satus	10	30	40	20	5	10	80	5				Run	N	1.0
2-Satus	5	5	30	60	5	5	65	25				Run	N	1.3
3-Satus	5	5	90	0	0	10	60	10	3.0			Run	N	0.5
4-Satus	10	10	50	30	5	15	60	20				Run	N	1.3
5-Satus	10	10	50	30	5	15	60	20				Run	N	2.0
6-Satus	5	5	60	30	5	5	70	20				Run	N	3

APPENDIX **1B.** Physical measurements and characterization of rainbow trout redds in the **mainstem** Yakima River, 1993. Only redds with complete measurements are reported.

	REDD NUMBER	REDD DIMENSION (m)			WATER DEPTH (m)			WATER VELOCITY (m/s)				DISTANCE TO COVER (m) *					
DATE	section-survey-redd	length	width	area	pot	mound	side	surface	0.6 depth	bottom	head	T	D	S	O	U	B
4/20	1-1-1	1.9	0.8	1.52	0.44	0.35	0.24	0.87	0.64	0.42	0.71			6.7		3.8	
4/20	1-1-2	1.8	0.7	1.23	0.32	0.19	0.24	0.69	0.57	0.26	0.62					0.3	
4/20	1-1-3	1.4	0.0	1.26	0.35	0.25	0.25	0.87	0.60	0.38	0.73					1.1	
4/20	1-1-4	1.7	0.7	1.19	0.35	0.26	0.32	0.86	0.50	0.46	0.74					0.2	
4/20	1.18	1.3	0.5	0.65	0.32	0.27	0.26	0.90	0.78	0.33	0.77					1.3	
4/20	1-1-6	1.7	0.4	0.68	0.31	0.24	0.27	0.99	0.76	0.46	0.77					1.4	
4/20	1-1-?	1.1	0.4	0.44	0.31	0.27	0.24	0.90	0.70	0.54	0.62					1.4	
4/22	1.18	1.8	0.8	1.44	0.36	0.25	0.30	0.55	0.69	0.46	0.64			1.1	1.5	1.5	
4/22	1-1-Q	2.4	0.9	2.04	0.32	0.12	0.18	0.54	0.46	0.37	0.58			1.2	2 5	1.2	
4/22	1-1-10	1.6	0.8	1.20	0.27	0.20	0.22	0.57	0.51	0.30	0.31			1.7	1.0	1.7	
4/22	1-1-11	1.3	0.7	0.85	0.20	0.07	0.10	0.22	0.52	0.69	0.57			1.1	0.8	1.3	
4/22	1-1-12	1.7	1.3	2.21	0.17	0.08	0.09	0.53	0.43	0.26	0.04			2 0	0.8	2 4	
4/22	1-1-13	1.5	1.3	1.95	0.26	0.16	0.20	0.57	0.47	0.34	0.52			0.8	1.0	2.3	
4/22	1-1-14	2.3	1.1	2.53	0.40	0.26	0.36	0.60	0.65	0.46	0.53			1.1	3.0	3.0	
4/22	1-1-15	1.2	0.7	0.84	0.38	0.28	0.30	0.58	0.57	0.36	0.50			0.6	0.6	0.6	
4/22	1-1-16	1.0	0.0	1.62	0.60	0.45	0.54	0 . 0	0.54	0.40	0.50			3.0	2 0	2.2	
4/14	2-1-1	2.3	0.8	1.84	0.46	0.25	0.30	0.66	0.64	0.48	0.81	7.6			2 4	1.7	
4/14	2-1-2	1.9	0.8	1.43	0.44	0.29	0.30	1.00	0.65	0.22	1.12	5.3			1.8	1.6	
4/14	2-1-3	1.3	0.5	0.64	0.00	0.32	0.43	0.59	0.47	0.35	0.40			5.9	1.4		
4/14	2-1-4	3.8	0.0	3.42	0.42	0.27	0.30	0.56	0.38	0.28	0.74			3.1		3.5	
4/13	51-1	1.0	0.6	0.96	0.35	0.29	0.30	0.46	0.34	0.29	0.31				0.5	2.0	
4/13	3-1-2	1 5	0.8	1.13	0.42	0.28	0.38	0.46	0.46	0.13	0.46						
4/13	3-1-3	1.4	0.8	1.12	0.45	0.38	0.34	0.75	0.74	0.22	0.76		5.0		5.0		
4/13	3-1-4	1.9	0.9	1.71	0.34	0.29	0.34	1.25	0.72	0.46	0.62		10.0				
4/13	3-1-5	2.0	0.0	1.70	0.24	0.14	0.23	0.69	0.51	0.37	0.48		15.0				
4/13	3-1-6	1.8	0.8	1.44	0.48	0.32	0.41	0.71	0.61	0.38	0.55	3.0	1.0	0.0	0.0		
4/13	3-1-7	2.0	0.8	1.60	0.40	0.26	0.32	0.53	0.43	0.35	0.52	1.5	1.5		3.0		
4/13	3-1-8	1.6	0.8	1.20	0.45	0.38	0.38	0.66	0.60	0.42	0.51	5.0			3.0		
4/13	3-1-9	3.5	0.7	2.45	0.58	0.44	0.42	0.58	0.57	0.25	0.54	19.0					
4/13	3-1-10	1.0	0.7	0.70	0.37	0.29	0.27	1.06	0.95	0.43	0.88		20.0				
4/23	3-2-11	2.0	0.5	1.00	0.46	0.38	0.42	0.63	0.38	0.30	0.51		13.0	0.7	3.6		
4/23	3-2-12	0.8	2.3	1.73	0.51	0.38	0.45	0.65	0.65	0.38	0.70		9.0	6.4	10.1		
4/23	3-2-13	2.7	0.8	216	0.34	0.24	0.28	0.64	0.54	0.30	0.60		12.8	6.1	10.9		
4/23	3-2-14	1.8	0.8	1.44	0.43	0.37	0.42	0.69	0.65	0.40	0.59		6.4	4.0	6.2		
4/23	3-2-15	2 1	0.8	1.58	0.28	0.10	0.24	0.69	0.69	0.47	0.72		14.1	10.6	10.1		
4/23	3-2-16	1.4	0.6	0.84	0.50	0.28	0.36	1.10	1.00	0.17	1.04	25.0	40.0	25.0	25.0		
4/23	3-2-17	3.0	1.0	3.00	0.26	0.08	0.14	0.43	0.38	0.27	0.26	1 2 0	1 2 0	4.6	4.0		
4/23	3-2-18	1.4	0.7	'0.98	0.27	0.18	0.25	0.67	0.68	0.40	0.70		1 2 7	a.1	28.9		
4/23	3-2-19	1.4	0.7	0.91	0.50	0.40	0.42	0.80	0.52	0.21	0.48			1.8	2 3		
4/23	3-2-20	2.3	0.9	1.96	0.47	0.31	0.34	0.34	0.32	0.19	0.25	4.0		0.6	0.6		
4/23	3-2-21	3.2	0 9	2.88	0.48	0.34	0.38	1.02	0.87	0.54	1.02	6.0	5.0	6.0	6.0		
4/23	3-2-22	1.6	0.8	1.20	0.40	0.30	0.34	0.58	0.52	0.44	0.61		0.1	4.1	5.1		
4/23	3-2-23	2.4	0.7	1.68	0.59	0.48	0.56	0.63	0.44	0.28	0.66		6.5	4.0	4.3		
4/23	3-2-24	2.4	0.8	1.92	0.44	0.36	0.40	0.69	0.50	0.26	0.59		a.0	2 0	4.5		
4/23	3-2-25	1.0	0.7	0.70	0.36	0.16	0.28	0.49	0.44	0.15	0.46	4.6		1.0	0.0	5.0	
4/23	32.26	1.7	1.0	1.62	0.20	0.14	0.15	0.50	0.54	0.35	0.48	8.0	8.0	1.2	1.2		
4/23	3-2-27	2.0	1.0	200	0.54	0.42	0.47	0.57	0.55	0.26	0.54		6.5	4.1	4.2		
5/8	S1.1'	1.5	0.8	0.87	0.35	0.30	0.33	0.62	0.59	0.36	0.45				16.0		
4/15	6.1-1	I S	1.0	1.90	0.38	0.25	0.32	0.59	0.49	0.26	0.59			33.0		10.0	
4/15	51.2	1.7	0.9	1.40	0.42	0.22	0.31	.0.90	0.82	0.48	0.85		9.4	6.3		11.7	
4/15	6-1-3	2.6	1.1	3.08	0.34	0.20	0.33	0.70	0.42	0.26	0.50		11.6	4.3	11.6		
4/15	C1-4	1.5	0.9	1.35	0.33	0.16	0.23	0.53	0.45	0.26	0.51				6.0		
4/15	6-1-5	1.4	0.0	1.26	0.32	0.20	0.23	0.64	0.45	0.24	0.54			20.0			
4/27	6-2-6	1.8	0.9	1.82	0.32	0.22	0.16	0.56	0.26	0.02	0.39			10.0	10.0		
4/27	6.2-7	2.4	1.2	2.88	0.38	0.32	0.28	0.54	0.40	0.16	0.50			1.5			
4/27	6.26	1.7	0.8	1.36	0.48	0.35	0.30	0.73	0.63	0.39	0.66			7.0		5.5	
4/27	6-2-9	1.5	0.7	1.05	0.42	0.22	0.30	0.70	0.47	0.26	0.57			3.3		3.2	
4/27	6-2-10	1.2	0.8	0.96	0.30	0.24	0.18	0.39	0.36	0.09	0.32			2 0			
5/8	6-3-12	1.4	0.7	0.98	0.27	0.17	0.25	0.54	0.53	0.33	0.47			3.0	1.0		
5/8	6-3-13	2.3	1.0	2.30	0.24	0.17	0.22	0.59	0.56	0.34	0.62		6.0	3.8	4.0		
5/8	6-3-14	2.3	0.0	1.84	0.35	0.17	0.25	0.73	0.71	0.46	0.72		4.0	4.8	5.0		
Ya	6-3-15	2.6	1.4	3.92	0.26	0.25	0.16	0.21	0.22	0.12	0.25			1.5	1.6		
5/8	6-3-16	2.3	1.2	2.76	0.22	0.10	0.25	0.40	0.40	0.17	0.36			3.0			
5/8	6-3-17	2.4	1.0	2.40	0.20	0.09	0.16	0.40	0.37	0.15	0.46			1.0			1.5
5/8	6-3-18	3.0	1.7	5.10	0.50	0.28	0.30	0.83	0.83	0.62	0.76		25.0	8.0	11.0		8.0
5/8	6-3-19	2.0	1.4	3.64	0.50	0.30	0.36	0.67	0.64	0.37	0.61	20.0	20.0	20.0	20.0		
5/8	6-3-20	2.2	1.0	2.20	0.30	0.21	0.22	0.61	0.62	0.39	0.58	17.0	17.0	17.0	17.0		

. T = turbulence, D = depth (>70 cm), S = submerged organic material (able to conceal 20 cm fish), 0 = overhead cover (able to conceal a 20 cm fish), U = undercut bank, B = boulder

no entry indicates that the cover type was not available near the redd

APPENDIX 1 B (Continued). Physical measurements and characterization of rainbow trout redds in the **mainstem Yakima River, 1993. All measurements are in meters unless stated otherwise.**

REDD NUMBER	SUBSTRATE IN PDT (%)				SUBSTRATE IN MOUND (%)				DISTANCE TO	DISTANCE TO	DISTANCE TO	HABITAT	SIDE	DISTANCE
section-survey-redd	<.5 cm	.5-1.3	1.3-6.4	>6.4 cm	<.5 cm	.5-1.3	1.3-6.4	>6.4 cm	OTHER REDD	SIDE CHAN.	MAIN CHAN.	TYPE	CHAN.	TO BANK
1-1-1	0	50	20	30	20	45	25	10				Run	N	3.6
1-1-2	35	10	10	45	5	25	50	20				Run	N	0.3
1-1-3	40	15	3	40	20	10	30	40	3.5			Run	N	1.1
1-1-4	10	10	79	1	5	10	0	85	1.0			Run	N	0.2
1-1.5	35	5	20	40	2	30	63	5	1.6			Run	N	1.3
1.18	40	5	5	50	5	10	80	5	0.8			Run	N	1.4
1-1.7	40	25	25	10	2	23	75	0	1.1			Run	N	1.4
1-1-8	I0	10	70	10	5	10	60	5	1.5			Run	N	1.1
1-1-e	5	10	40	45	5	5	80	10	2.4		15	Run	Y	1.2
1-1-10	10	10	50	30	10	10	80	0	7.5		15	Run	Y	1.7
1.1.11	10	10	70	10	5	10	75	10	0.9		18	Run	Y	1.3
1-1-12	10	10	70	10	5	15	80	0	0.0		20	Run	Y	2.4
1-1-13	10	10	40	40	10	10	60	20	5.0		20	Run	Y	2.3
1-1-14	5	5	40	50	10	15	75	0	1.6			Run	N	1.3
1-1.15	10	10	30	50	5	10	a5	0	6.2			Run	N	0.6
1-1-16									6.2			Run	N	2.2
2-1-1	5	10	80	5	0	25	75	0				Run	N	1.7
2-1-2	5	5	al	0	0	10	80	IO				Riffle	Y	1.6
2-1-3	5	5	30	60	25	30	45	0				Run	N	1.0
2-1-4	5	10	80	5	2	5	88	5				Run	Y	3.5
3-1-1	0	5	30	65	5	20	60	15				Run		
3-1-2	10	5	70	15	5	10	80	5				Run	N	6.0
3-1-3	5	20	75	0	10	20	70	0				Run	Y	
3-1-4	10	10	80	0	5	10	85	0				Run	Y	3.0
3-1-5	2	5	70	23	a	25	57	10				Run	Y	2.5
3-1-6	10	5	35	50	0	10	90	0				Run	Y	
3-1-7	15	10	85	0	5	20	75	0				Run	Y	
3-1-8	10	5	45	40	10	10	70	10				Run	Y	
3-1-Q	5	20	75	0	5	20	75	0				Run	Y	
3-1-10	5	10	80	5	5	15	75	5				Run	Y	
3-2-1 1	IS	15	65	5	10	15	75	0	0.0			Run	N	4.1
3-2-12	5	5	60	30	10	10	65	15	4.4			Run	N	7.6
3-2-13	10	10	60	20	5	5	80	10	5.3			Run	N	4.1
3-2-14	10	10	50	30	10	10	60	lo	6.0			Run	N	5.3
3-2-15	5	5	60	30	3	5	92	0	13.7			Run	N	2.4
3-2-16	5	5	80	10	5	5	90	0		25.0		Run	N	2.0
3-2-17	10	10	40	40	5	5	80	10	3.4		6	Run	Y	6.0
3-2-18	10	10	60	20	5	5	80	10	13.7			Run	N	6.1
3-2-19	10	10	30·	50	30	lo	40	10	4.8	lo.9		Run	N	6.9
52.20	10	10	70	11	5	10	85	0	1.7	16.1		Run	N	6.0
3-2.21	5	10	55	25	10	5	80	5				Run	N	2.5
3-2-22	3	5	67	5	10	40	40	10	1.4			Run	N	4.3
3-2-23	5	5	80	10	5	5	90	0	5.7			Run	N	5.3
3-2-24	10	15	65	10	5	10	85	0	0.0			Run	N	5.0
3-2-25	10	5	45	40	5	5	90	0	20.0			Run	N	1.0
32.26	5	5	70	20	5	10	60	25	3.4		4	Run	Y	4.0
32-27	5	5	85	5	5	5	90	0	2.9			Run	N	5.2
51.1	10	10	50	30	10	10	80	0				Run	N	1.0
6-1-1	20	30	40	10	10	25	55	10			70	Run	Y	15.4
Cl-2	10	40	48	2	5	60	30	5			20	Run	Y	1.6
6-1-3	60	15	22	3	2	35	62	1				Run	N	0.3
6-1-4	15	15	45	25	20	50	28	2			150	Run	Y	5.0
6-1-5	5	5	70	20	7	35	54	4			130	Run	Y	6.0
6-2-6	20	0	70	10	0	10'	80	10			50	Run	Y	1.0
6-2-7	10	43	42	5	3	15	82	0				Run	N	3.0
6-2-8	5	10	55	30	5	10	85	0			115	Run	Y	5.5
6-2-9	50	15	15	20	5	10	85	0			120	Run	Y	3.2
6-2-10	25	10	60	5					1.0	10.0		Run	N	1.0
6-3-12	5	20	75	0	5	40	55	0				Run	N	1.4
6513	10	30	60	0	5	15	80	0	1.0		16	Run	Y	1.0
6-3-14	10	10	80	0	5	10	80	5	1.0		18	Run	Y	1.5
6-3-15	5	5	50	40	5	5	60	30				Run	N	1.5
6.316	5	5	60	30	0	5	80	15	30.0			Run	N	4.0
6-3-17	5	5	80	IO	5	5	80	IO	45.0			Run	N	1.3
6-3-18	10	10	60	20	5	5	80	10	8.0			Run	N	8.0
6-3-19	10	5	60	25	5	10	70	15	4.0			Run	N	3.8
6-3-20	10	10	70	10	5	5	80	10	4.0			Run	N	1.9

Chapter 2

Comparison of survival, gonad development, and growth between rainbow trout with and without surgically implanted dummy radio transmitters

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Abstract

The use of radio telemetry to determine fish movement patterns associated with spawning has proliferated in recent years. However, little is known about how surgically implanted radio transmitters affect spawning behavior or gonad development of fish collected near the time of spawning. We compared survival, gonad development and growth of wild rainbow trout, *Oncorhynchus mykiss,* with and without dummy radio transmitters implanted prior to their spawning period. Wild rainbow trout (mean FL = 351 mm) were collected from the Yakima River, Washington on February 5, 1993, using a driftboat electrofisher. Ten fish that met selection criteria had dummy radio transmitters surgically implanted into the intraperitoneal cavity, while ten other fish were retained as controls. All 20 fish were released into a nearby pond and fed daily. After 47 days all fish were measured and weighed, and gonad development and general health was assessed. All fish survived and there were no transmitter expulsions by treatment fish. There was no significant difference in weight, condition factor, or gonad development between treatment and control fish. These results suggest that wild rainbow trout may be used for telemetry studies even when the collection of fish and transmitter implantation occurs close to the time of spawning.

Introduction

Radio telemetry has been used extensively to investigate movement and migrations of salmonids (McCleave et al. 1978; Couturier et al. 1986; Lorenz and Eiler 1989) but the effects that radio transmitters have on spawning behavior and gonad development has received little attention (Mellas and Haynes 1985). Research on growth, tissue reaction, and behavior in response to transmitter implantation has been conducted on several Orders of fish in hatcheries and laboratory settings (Summerfelt and Mosier 1984; Chisolm and Hubert 1985; Mellas and Haynes 1985; Lucas 1989), however little is known of the effects that transmitters have on gonad development and growth of wild fish. Although implanted transmitters appear ideal for long-term tracking studies, they may alter a fish's behavior and growth as well as their ability to spawn (Stasko and Pincock 1977; Mellas and Haynes 1985). We surgically implanted dummy transmitters into 10 adult wild rainbow trout to assess the effects of implanted transmitters on survival, gonad development and growth of wild rainbow trout. Surgery occurred on February 5th, approximately two months before the early April spawning period of rainbow trout in the Yakima River. Ten additional wild rainbow trout served as control fish.

Methods

Thirty wild rainbow trout were captured from the Yakima River by driftboat electrofishing on February 5, 1993, approximately 25 km south of the city of Ellensburg, Washington. Electrofishing was conducted from a 5 m long driftboat from 0400 h until one hour after sunrise using 400 volt direct current (Coffelt, complex pulse system). The fish were transported in a 1,000 L holding tank to a small man-made pond located 20 km east of Ellensburg, Washington. The pond had a surface area of 80 m^2 and a maximum depth of 3 m. The pond had a large deciduous tree at one end; the remainder was without riparian cover. No other fish were present in the pond during this study.

To avoid problems associated with implanting dummy transmitters into small fish or fish that were sexually mature, only green rainbow trout weighing more than 300 g were used. Fish were classified as green if they did not expel gametes upon slight pressure to the abdomen. It was assumed that all fish used in this study would spawn in the spring of 1993.

Two fish were netted from the holding tank and anesthetized in a 20 L bucket containing a 100 mg/L tricaine methanesulfonate (MS-222) solution. Fish were considered to be fully anesthetized when they had a total loss of equilibrium (3-4 minutes) (Summerfelt and Smith 1990). The first of the two fish which met the weight and sexual maturity requirements was surgically implanted with a dummy radio transmitter. Dummy radio transmitters were made by Advanced Telemetry Systems (ATS), set in epoxy, and were the size and weight of an actual radio transmitter (13 mm x 50 mm, 10 g).

Anesthetized treatment fish were placed on their dorsum in a trough lined with neoprene to restrict their movement during surgery. The fish's head and gills were continuously submerged in water containing a dilute solution of MS-222 (20 mg/L). Prior to implantation, all tags were sterilized in Zephiran and rinsed with water. Following the techniques of Lucas (1989), a 2.5 cm incision was made lateral to the ventral midline of the fish, just anterior of the right pelvic fin. A 20 cm long x 2 mm diameter cannula was then inserted into the body cavity through the incision, and pushed toward the caudal end of the fish in such a way as to avoid contacting any internal organs. When the cannula contacted the body wall, it was pushed through the myomeres until it exited the skin along the caudal peduncle below the lateral line. The dummy transmitter antenna was then threaded through the cannula. While holding the dummy transmitter, the cannula was removed by pulling it out through the exit wound. The dummy radio transmitter was then placed in the intraperitoneal cavity. The incision was closed with 4 or 5 absorbable sutures (polyglycolic acid) using a 19 mm cutting edge needle. The incision was then covered with Betadine, a topical antiseptic solution and then coated with Baciguent, an antibiotic ointment. Total surgery times varied from 9 to 22 min. All surgery was conducted on the same day. The fish were allowed to recover fully from the anesthesia (approximately 8 min) and then were released into the pond. One treatment fish was accidently transported back to the Yakima River before the study was terminated.

The second fish that met the above requirements served as a control fish; it was handled in the same manner as the treatment fish but no incision was made and it was not implanted with a dummy transmitter. Sampling continued until ten treatment and ten control fish had been processed. In addition to weighing and measuring each treatment and control fish, each fish was marked with a numbered anchor tag, and scales were removed for age analysis.

Surface water temperature was recorded daily at 0800 h. Observations from the pond shoreline were made daily in an attempt to document feeding behavior, and to locate and remove any dead fish. Fish were fed BioDry 5 mm fish food pellets at a rate of 1% of their body weight daily (Piper et al. 1983). In addition to the artificial food, aquatic insects were present in the pond. Surface water dissolved oxygen concentration was determined by titration one d prior to the termination of the study.

On March 20, 1993, 47 d after initiation of the study, the pond was drained and the fish were sacrificed to obtain information on survival, gonad development, growth, incision condition, and external appearance. The fish were later autopsied to assess gonad development and the condition of the incision and antenna exit wound.

Results and Discussion

None of the 19 rainbow trout died during the experiment and it appeared that gonad development and gamete production was not affected by the implantation of dummy transmitters. The only exception was one female in the treatment group that was resorbing eggs. Three of five males and three of four females in the treatment group possessed fully developed gonads (Table 1) and were likely to spawn in 1993. The remaining two treatment males showed no signs of gonad development and were not likely to spawn in 1993 regardless of transmitter implantation. Three of the four female treatment fish had fully developed eggs present in unbroken ovaries. However, eggs in the fourth fish were small and several were opaque and thus assumed to be resorbing. In addition to small size and opaque coloration, the ovaries of this fish contained cloudy fluid that occupied the space between individual eggs. At the start of the study, this female was comparable in length (361 mm) and weight (467 g) to other treatment fish (mean length and weight 354 mm, 487 g).

Among the seven males in the control group, six exuded milt at the termination of the study. The remaining male showed no signs of gonad development and was not likely to spawn that year. Sexual maturity of the three female control fish was as follows; one had developed eggs in unbroken ovaries, one had loose eggs, and one had small eggs that would probably have remained immature until the following year. All eggs appeared to be viable in both groups, except for the one treatment fish with resorbing eggs.

We observed no statistical differences in weight gain between the two groups (t-test; df = 18, P = 0.30), although the control group gained weight during the experiment and the treatment group lost weight (Table 1). Weight loss was not significantly correlated with sex, although the sample size was small. The percentage of females in the treatment and control groups that lost weight was 75% and 67%, respectively, while the percentage of males in the treatment and control groups that lost weight was 60% and 57%, respectively. Also, condition factors between the treatment and control groups did not differ significantly (t-test; df = 18, P = 0.90). At study termination, only three treatment and two control fish had empty stomachs (Table 1), suggesting that the dummy transmitters probably did not

TABLE I. Comparison of characteristics associated with dummy tagged (treatment) and untagged (control) rainbow trout.

Treatment Group

			At Study Start (day 0)					At Study Termination (day 47)						
Fish No.	Age	Sex	Surgery Time	Gonad Status (a)	Length (mm)	Weight grams	K (b)	Gonad Status (a)	Length (mm)	Weight grams	K(b)	Suture Cond (c)	Antenna Wound(d)	Gut Contents(e)
1	4	M	12:03	2	365	481	0.99	3	365	481	0.99	2	1	4
2	3	M	13:oo	2	322	383	1.15	3	318	353	1.10	1	1	7
3	3	M	13:05	2	360	501	1.07	1	365	526	1.08	3	1	1
4	3	M	22:02	2	360	511	1.10	1	360	508	1.09	1	1	7
5	3	M	16:01	2	338	580	1.50	1	339	558	1.43	3	1	3.4
6	3	F	11:02	2	321	382	1.15	2	318	365	1.14	3	1	3
7	3	F	10:02	2	361	467	0.99	4	358	479	1.04	2	1	7
8	3	F	16:01	2	368	560	1.12	2	364	532	1.10	2	1	3.4
9	4	F	14:03	2	357	473	1.04	2	351	464	1.07	2	2	1
10	---	F	09:04	2	328	429	1.22	(f)	---	---	---	---	---	---
Mean:	3.2		13:30	2	348	477	1.13	Mean:	349	474	1.12			

Control Group

			At study Start (day 0)					At Study Termination (day 47)						
Fish NO.	Age	sex	Surgery Time	Gonad Status (a)	Length	Weight grams	K (b)	Gonad Status (a)	Length (mm)	Weight grams	K(b)	Suture Cond. (c)	Antenna Wound(d)	Gut Contents(e)
II	3	M	N/A (g)	2	361	500	1.06	1	360	594	1.27	N/A (g)	N/A (g)	1
12	3	M	"	2	361	498	1.06	1	363	561	1.17	"	"	5
13	4	M	"	2	354	462	1.04	3	350	435	1.01	"	"	3
14	3	M	"	2	362	477	1.01	1	368	594	1.19	"	"	1
15		3.M	"	2	341	440	1.11	1	340	416	1.06	"	"	1,2
16	3	M	"	2	344	488	1.20	1	345	467	1.14	"	"	2,3,6
17	3	M	"	2	400	680	1.06	1	398	629	1.00	"	"	7
18	3	F	"	2	320	313	0.96	3	310	293	0.98	"	"	7
I9	4	F	"	2	385	668	1.17	1	386	677	1.18	"	"	6
20	3	F	"	2	313	341	1.11	2	310	328	1.10	"	"	3
Mean:	3.2			2	354	487	1.08	Mean:	353	499	1.11			

a Gonad status: 1 = exuding gametes, 2 = green (fully developed gonads; assumed at study start), 3 = immature, and 4 = resorbing

b Condition Factor

c Suture condition: 1 = 0% infected. 2 < 25% infected, 3 < 50% infected, and 4 > 50% infected

d Antenna wound: 1 = 0% infection, 2 < 25% infection, 3 < 50% infection, and 4 > 50% infection

e Gut contents: 1 = commercial pellets. 2 = Tricoptera, 3 = Hemiptera, 4 = Diptera, 5 = Amphipoda, 6 = Debris, and 7 = empty

f This fish was accidently transported from the pond and released into the Yakima River before study termination.

g N/A = not applicable

affect the tagged fish's feeding ability.

Although no fish died and the differences in weight between treatment and control wild rainbow trout in this study was low, results from other studies concerning survival and growth of fish implanted with transmitters have been variable in both field (Minor and Crossman 1978) and laboratory settings (Chisolm and Hubert 1985; Summerfelt and Mosier 1984). Chisolm and Hubert (1985) reported that eight hatchery rainbow trout tagged in the spring died during a 175 d experiment, and five of the eight died within the first 30 d of the experiment. The cause of death was not reported; however, they used hatchery fish and dummy transmitters coated with paraffin wax. Wax is more porous than epoxy and may have led to tissue reaction and adhesion, expediting expulsion or infection which may have led to mortality. This is supported by Helm and Tyus's (1992) finding in which 13% of rainbow trout surgically implanted with transmitters coated with paraffin (wax) were expelled. Generally, survival and growth appears to be most strongly associated with careful surgical procedures (Lucas 1989) and environmental conditions at the time of tag implantation. We believe that conducting this type of experiment when water temperatures and fish metabolic rates are low may reduce the chance for infection and subsequent negative growth or death.

No dummy transmitter expulsions occurred during this experiment and there was no evidence of transmitter encapsulation. This finding is contradictory to other researcher's findings (Chisolm and Hubert 1985, Summerfelt and Mosier 1984, Helm and Tyus 1992). Chisolm and Hubert (1985) reported that 13 out of 22 surviving fish expelled their dummy transmitters by encapsulation and passage through the anus during a 175 d experiment.

In this study, all fish exhibited similar behavior following release into the study pond. Treatment and control fish moved slowly to the bottom and within one minute swam to the center of the pond. This non-erratic behavior has also been noted for rainbow trout released into a swimming chamber following surgical transmitter implantation (Mellas and Haynes 1985). Although inspections were conducted on a daily basis, none of the fish were observed until they were removed from the pond on d 47.

The general health of the fish upon recapture from the pond appeared to be good. Three of the nine treatment fish had more than 25% of the length of the incision infected, and only one fish had infection associated with the antenna exit wound. Infection was determined by fungus or bacterial growth in or adjacent to the incision or exit wound. One of the nine treatment fish had lost two of its four sutures, and although the incision was completely closed, 50% of the incision was infected (Table 1). Our results are similar to Chisolm and Hubert's (1985) findings in which they observed no external lesions or rupturing of the incision.

Furthermore, they reported that during their 175 d study, the study fish grew and maintained good body condition.

Upon recovery from the pond, only one fish had external infection (fungus) in locations other than the incision site or antenna exit wound. However, four control and three treatment fish had eroded or frayed fins. The presence of fungus and eroded fins may be attributed to poor water quality in the pond, or digging activity associated with spawning. When the study was terminated on March 20, surface dissolved oxygen was very low (1.6 mg/L dissolved oxygen at 7.8 °C). Piper (et al. 1983) state that fish subjected to extended oxygen concentrations below 5 mg/L may experience reduced growth and survival. We speculate that the low dissolved oxygen concentration of the surface water on March 20 was temporary, and not indicative of the actual oxygen distribution in the pond. Wetzel (1983) states that in productive waters, surface water oxygen concentrations can vary markedly on a diel basis, fluctuating between afternoon super-saturation and early morning undersaturation. As a result of the natural diel fluctuations and distribution of dissolved oxygen in lakes and ponds, the low dissolved oxygen concentration for the surface water of the pond at 0800 h on March 20 may have been a result of when the sample was taken.

Our results suggest that wild rainbow trout and possibly other resident salmonids, can be surgically implanted with epoxy-coated radio or dummy transmitters shortly before the spawning period without adverse effects on fish survival, gonadal development, and growth. However, to minimize risks to the population under study, we recommend that the surgical implantation be completed well before the spawning period.

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Chapter 3

Movement of resident rainbow trout within the upper Yakima River basin

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Abstract

Compared to the migrations of the anadromous form of rainbow trout (*Oncorhynchus mykiss*), the movements of the non-anadromous or resident form are poorly understood. We tagged 7,235 rainbow trout larger than 175 mm fork length with numbered external tags from 1990 to 1993 to determine their movement patterns. Fish were collected for tagging by electrofishing and migrant trapping, and were subsequently recaptured by electrofishing, migrant trapping, and angling. Movements of tagged rainbow trout were determined by comparing the locations of their previous capture with the locations where they were recaptured. Trout emigration from tributaries was also detected by trapping. Preliminary results from the four-year study showed that the net movements of tagged rainbow trout previously captured within the mainstem Yakima River ranged from 0 to 149 km with 59% moving a distance less than 5 km. Fish previously captured in tributaries moved between 0 and 194 km with 89% moving less than 5 km. The average movement distance of tagged fish previously captured in upper study sections of the mainstem were significantly greater than those by fish that were previously captured in lower sections. In addition, tagged rainbow trout in the mainstem generally moved greater distances than those in tributaries. Rainbow trout moved more during the winter and spring than during the summer and fall. The distance that rainbow trout moved appeared to be related to elevation, stream size, and time of year.

Introduction

Movements made by stream-resident rainbow trout (*Oncorhynchus mykiss*) are not completely understood. Although much is known about their time of spawning and general habitat and environmental preferences, volitional shifts in location, particularly those not associated with reproduction, and those within shorter time frames, have not been widely reported. The anadromous form of *O. mykiss*, steelhead trout, are noted for lengthy seaward migrations as juveniles and extensive migrations while at sea (Hart 1973). After spending a large share of their lives at sea, most steelhead return to their natal streams as adults (Shapovalov and Taft, 1953 and Slatick et al. 1981). In contrast, rainbow trout spend their entire lives in freshwater and do not migrate as far as steelhead. Often, the juveniles of both forms are present in sympatry within streams and are difficult to differentiate from each other prior to the steelhead smolt transformation. The reported movements of juvenile *O. mykiss* often represent both forms in their early life-history stages.

Most large rainbow trout in streams have been shown to move less than 15 km during a wide range of time frames. Cargill (1980) characterized rainbow trout in a spring-fed Minnesota stream as being "quite sedentary". Schroeder and Smith (1989) found that approximately 90% of the rainbow trout they tagged in the Deschutes River, Oregon, and one of its tributaries were recaptured within 1.6 km of the tagging site. Vincent (1987) reported that 71% of the tagged rainbow trout in the Madison River and O'Dell Creek, Montana, that were later recaptured, had moved 0.0 to 0.6 km, with 29% moving more than 0.6 km and 8% having moved more than 1.6 km. Downstream movements constituted fifty-five percent of the observations in those streams. In a Montana stream, Stefanich (1952) found that of 52 tagged rainbow trout that were recovered 16 were caught in the sections where they had been tagged, 12 were reobserved upstream and 24 were reobserved downstream of where they had been tagged. Of 32 rainbow trout recaptured by anglers, 28% moved downstream on average between 2.0 and 14.6 km and 16% moved upstream an average of 7.1 km.

Rainbow trout movement appears to vary with fish age and the time of year. In streams, the greatest movement from natal areas to other rearing areas normally occurs at ages 1, 2, and 3 (Stauffer 1972; Alexander and MacCrimmon 1974; Erman and Hawthorne 1976). Erman and Leidy (1975) observed the emigration of juvenile rainbow trout from Sagehen Creek, California, primarily in the spring and early summer. Similarly, Alexander and MacCrimmon (1974) found emigrations of rainbow trout juveniles from two Great Lakes tributaries in the spring. Bjornn (1971) found movements of non-smolt trout and salmon in two Idaho streams in the fall, winter, and spring and that their preferences for alternative rearing sites were best correlated with the presence of large rubble substrate. He theorized that energetic limitations in colder water presumably stimulated the migrations to velocity refuges.

Adult rainbow trout travel the farthest during the time of spawning activities which are usually in the spring. (Northcote 1992). Pearsons et al. (1993) found that most of the resident trout that had been tagged in the mainstem of the Yakima River-at any time of the year, and were subsequently recaptured in a tributary, were recaptured in the spring and were often in spawning condition. This suggests that they were moving into tributaries to spawn.

The purpose of this study was to determine the distances resident rainbow trout moved, including an examination of differences in movement related to the time of year and the various stream environments present. We used externally visible tags to investigate the movements of rainbow trout in streams within the upper Yakima River drainage. This study was conducted incidentally to the original intent of qualifying the exchange of resident trout between the Yakima River mainstem and tributaries, and trout emigration from the study area. Rainbow trout recaptures documented after October 31, 1993 are not a part of the analysis. Recaptures documented through October 31, 1994 shall be integrated within findings of a subsequent report. Interpretation of results from this study should be considered preliminary and subject to further revision.

Study Area

This study was conducted on the Yakima River and its tributaries in Kittitas County, Washington. The study area is located between Roza Dam and Keechelus Dam, river kilometer (rkm) 180 and 305. Most major and many minor tributaries in the upper Yakima basin are included within this area (Figure 1).

The Yakima River originates in the Cascade Mountains of central Washington above Keechelus Lake (elevation 767 m). As the river flows southeast to its confluence with the Columbia River, it passes through climatic transitions ranging from relatively cool and moist in the mountains to arid in the Yakima Valley.

Three reservoirs (Keechelus, Kachess, and Cle Elum) that lie above the study area and one (Easton) that lies within it provide water storage to accommodate regulated flows for irrigation and flood control in the Yakima River. Compared to the pre-regulation hydrograph, the reservoirs greatly affect instream flows within the Yakima River. Flows tend to be artificially low in late winter and early spring and artificially high during the summer with flows as high as 113 m^3/s or more in some river reaches within the study area.

For this study, the mainstem of the upper Yakima River was divided into seven sections based on broad geographical characteristics. Section numbers, names and lengths in kilometers, from lowest to highest elevation, are: 1 Lower Canyon

(16.2), 2 Upper Canyon (13.3), 3 Ellensburg (20.6), 4 Thorp (21.9), 5 Cle Elum (26.1), 6 Nelson (16.8), and 7 Crystal (17.7).

Similarly, most tributaries were divided into three study sections, but in a range of 2-4 sections per tributary. Section boundaries were selected based on variation in landforms and barriers to migration. Uppermost study boundaries were set at the presumed end of the potential spawning area for steelhead. As a result, upper elevation study sections in both the Yakima River mainstem and its tributaries were normally of higher gradient and often terminated at barriers, such as falls or dams.

Roza Dam, the lower boundary for this study, is passable in both directions by older age classes of salmonids. Fish may pass the dam via a fish ladder that was modified in 1988, or through spillways. Similarly, the existing fish ladder at Easton dam was remodeled in 1989, primarily to improve the access of adult salmon and steelhead to upstream spawning areas. Fish ladders have not been constructed at the regulating dams of the three upper storage reservoirs (Anonymous 1990).

The trout fishery in the Yakima River above Roza Dam is presently managed for catch-and-release, requiring the use of single barbless hooks and artificial lures only.

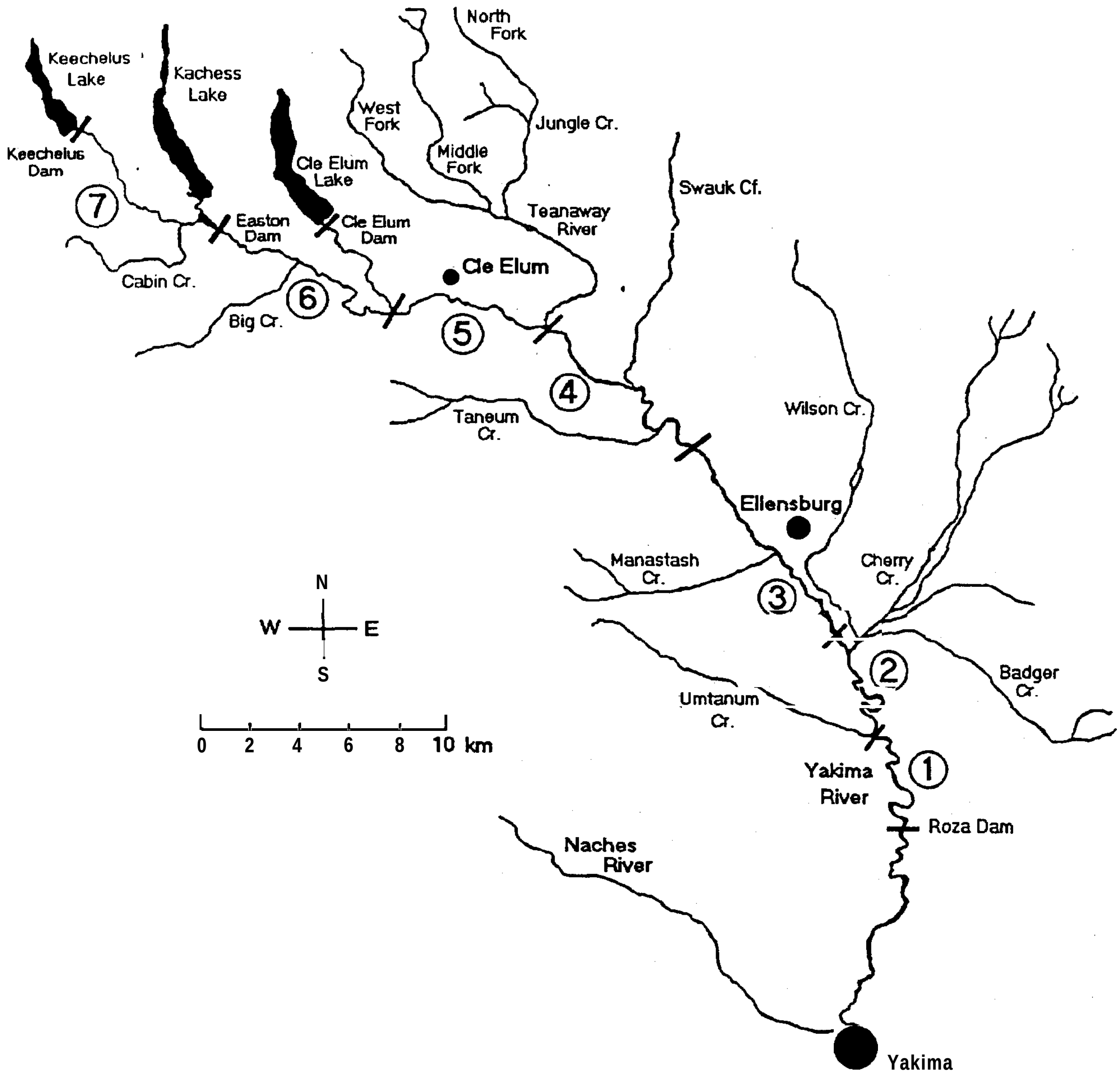

Figure 1. Map of study area within the upper Yakima River basin. Mainstem study sections numbers (1-7) are shown in circles.

Methods

Tagging

Rainbow trout were captured and tagged from February through October, 1990 through 1993, in the mainstem and tributaries of the Yakima River above Roza Dam using electrofishing and trapping methods (Hindman et al. 1991; McMichael et al. 1992; Pearsons et al. 1993). Three types of serially-numbered, externally visible tags were applied: T-Bar anchor, Visible Implant (VI), and Floy fingerling "dangler" style. Dangler and VI tags were used on trout between 120 and 175 mm fork length (FL), and were used mostly within the Teanaway River basin. Less than 30 VI tags were applied to trout of the same size range in Umtanum Creek. Anchor tags were used on trout of 175 mm FL and greater. Trout less than 120 mm FL were not tagged. All fish lengths in this chapter are expressed as fork length. Anchor tags were inserted into the muscle tissue on the left side of the fishes back, at the base of the dorsal fin two or three radials from its posterior edge. Visible Implant tags were inserted beneath adipose tissue just posterior to the orbit of the left eye. A hollow needle with an internal plunger was used to deliver individual tags. Dangler tags were applied by sewing them through a fishes back at the same relative location on the fish that anchor tags were applied. Both free ends were tied together to discourage shedding.

Tagged fish were reobserved during electrofishing and trapping activities within the study area. In addition, anglers reported the locations and observation dates of tagged trout on self-addressed postcards that had been distributed to sport shops, angling clubs, and individuals that were fishing. The cards contained information about our tagging study and instructions for recording recapture information. Additional information was obtained from various fishery workers handling fish passing dams along the Yakima River.

Trapping

Various trapping methods and trap designs were used to obtain information about trout movements associated with tributaries (McMichael et al. 1992; Pearsons et al. 1993; Chapter 1, this report). Weirs built from screened panels and arranged in "W" and "V" configurations, and picket weir fish traps were installed near the mouths of several tributaries (Umtanum, Cherry, Wilson, Taneum, Swauk, Jack, and Jungle creeks) to monitor the movements of fish during spring and early summer in 1991, 1992, and 1993 (see Chapter 1, this report). A traversing fyke net with a square aperture measuring 1.8 m per side was used to sub-sample emigrants in the North Fork of the Teanaway in 1991. We used rotary screw-traps with 1.5 m diameter apertures in the Middle and North Forks of the Teanaway River in the spring of 1992 and 1993. All traps had varying capture efficiencies based on design, site selection, flow conditions, operational problems, and the time of operation (see Chapter 7, this report).

Data Analyses

Rainbow trout movement distances were assessed by calculating the lineal stream distance between the location of capture and the location of subsequent recapture. Where a fish was recaptured multiple times, a previous recapture was treated as a capture. Therefore, the same fish accounted for more than one recorded movement if it was recaptured more than once. Trout capture and recapture locations were converted to stream kilometer (skm). Both lower (downstream) and upper (upstream) boundaries were assigned to each location of a capture and a subsequent recapture that reflected the most specific capture and recapture sites. This was done because most observations were recorded in terms of a location range, that is, a stream section or sub-section, rather than specific point locations. If no section boundary or migration barrier was present upstream of an observation where only section information was available, headwaters, defined by that location where the stream was expected to be seasonally intermittent, were designated as the upper boundaries. The shortest possible distance traveled by a fish between capture and recapture is reported as the minimum distance value and the longest possible net distance traveled is reported as the maximum distance value. These values were based solely on the distances between respective site boundaries. Average movement distance values were the quotient of the sum of minimum and maximum distance values, divided by two, and are presented hereafter unless otherwise stated. If point locations (where upper and lower site boundaries were the same) were available for both the capture and recapture observations, the calculated distance was defined as "actual" net movement.

Many combinations of upstream and downstream movements were possible between the Yakima River mainstem and its tributaries. The mouth of the Yakima River was assigned skm 0. Similarly, the mouth of each Yakima River tributary was assigned skm 0 for that stream. When movement between the mainstem and its tributaries was observed, an adjustment value was integrated into the distance calculation to correct for the different points of reference involved. This value usually reflected the skm of the receiving stream (usually the mainstem) at its confluence with the tributary.

Several analyses were performed to assess possible sources of bias within our recapture data. Records having the greatest probable accuracy regarding net movement were identified by comparing the ratio of the minimum to the maximum calculated distances of movement. We devised an index of quality (IQ) to reflect the quality of movement information associated with each recapture record. This index was calculated using the formula:

$$IQ = \frac{(\text{minimum distance} + 1)}{(\text{maximum distance} + 1)}$$

where the IQ could range from 0 to 1, with unity representing the most accurate measure of the true net distance moved. A stratified sample of recaptured, tagged fish was formed by selecting 30 records with the highest IQ values from each of the Yakima River study sections 1 through 5. This stratified sample was used as a reference to examine potential biases within the complete set of mainstem recapture observations and the set composed of actual movement distance values. Pearson Product-Moment correlations between average movements from the complete mainstem data set and the tributary set versus the respective lengths of the sections of previous capture were performed. This was done to determine the influence of stream section length on estimates of average movement distance values. The amount of time that elapsed between a fishes capture and recapture, or "days-at-large", was calculated to determine its relation to movement distances. Additionally, we investigated the relative movement distances between rainbow trout that were recaptured only once versus those that were recaptured multiple times. This was done to minimally assess the independence of data derived from repeated recaptures.

Observations of fish that were captured and recaptured in the same trap were omitted from our analyses since that circumstance produced erroneous results indicating that no net movement had occurred. T-tests were used in all cases to detect significant differences in mean movement between streams and stream sections ($P \leq 0.05$). Recaptures outside the study area were included in the analyses so that the reported movement distances would not be artificially limited.

During the spring, steelhead smolts were distinguished from resident rainbow trout using external characteristics as outlined by Pearsons et al. (1993). In addition, putative rainbow trout less than 300 mm long that were recaptured at or below Roza Dam between February 15 and June 15, in all years, were assumed to be steelhead smolts. The basis for this assumption was that in 1992, more than 99% of the steelhead smolts trapped at Roza Dam met these timing and length criteria (Yakima Indian Nation 1992, unpublished data). Downstream migrant rainbow trout that were trapped at a tributary mouth and were neither identified as steelhead nor in spawning condition were identified as "recruits". In this context, recruitment is intended to refer to immigration into a larger tributary or mainstem area where individuals may be expected to rear and mature.

Estimating the proportion of adult rainbow trout that rear in mainstem areas, but spawn in tributaries, required that we estimate the number of trout that inhabited the mainstem and spawned each year. Schroeder and Smith (1989), working with resident rainbow trout in the Deschutes River basin, studied different size groups to determine the proportion that exhibited gonad development. We applied their estimate (0.51) as the proportion of mainstem rainbow trout larger than 250 mm that would spawn in any given year. The population size of rainbow trout larger than 250 mm in mainstem sections was obtained from our other concurrent studies using Petersen estimates (see Chapter 4, this report).

Results

The number of rainbow trout tagged during this study totalled 4,977 in the mainstem Yakima River (Table 1) and 2,265 in its tributaries. Of trout tagged in the seven mainstem study sections, 34% of all tags applied were applied to fish in the Lower Canyon section because more fish were captured there. Of the anchor, VI, and dangler tagged rainbow trout released throughout the basin, only anchor-tagged fish were reobserved in the mainstem. However, four steelhead smolts with VI tags were recaptured during their outmigration past Roza Dam. A total of 547 rainbow trout with anchor tags were recaptured in mainstem areas. There were 121 reobservations of fish tagged with one of the three tag types in tributaries. Downstream of the study area, five recaptures of anchor-tagged rainbow trout were recorded.

The distances that tagged rainbow trout moved varied according to the elevation at which they were previously captured. The amount of movement in the mainstem was positively related to elevation. (Figure 2). Based on "actual movement" data of fish previously captured in the Lower and Upper canyon sections, only 41% moved over 1 km. Also, based on the stratified data set, fish previously captured in the Cle Elum section moved farther on average than those previously captured in all other mainstem sections combined, but the difference was not statistically significant ($P = 0.0632$, $t = -1.872$, $df = 145$).

Table 1. Number of wild resident rainbow trout tagged and number of recaptures between 1990 and 1993 in study sections of the upper Yakima River mainstem. The relative percent contributions to the total number tagged and the total number of recaptures are also given. LCYN = Lower Canyon, UCYN = Upper Canyon, EBURG = Ellensburg, CELUM = Cle Elum, NELSN = Nelson, CRSTL = Crystal.

sections	Number tagged	Percent tagged	Number recapt.	Percent recapt.
LCYN	1690	34.0	316	57.8
UCYN	975	19.6	109	19.9
EBURG	845	17.0	56	10.2
THORP	646	13.0	30	5.5
CELUM	800	16.1	28	5.1
NELSN	19	0.4	1	0.2
CRSTL	2	<0.1	2	0.4
outside study area:				
Below Roza Dam	0	0.0	5	0.9
Total	4977	100	547	100

There was, however, a statistically significant difference in the average distance moved between trout previously captured in the Cle Elum section (14.2 km) and those previously captured in the two Yakima Canyon sections combined, using the "stratified set" (5.9 km; P = 0.010, t = -2.621, df = 88), with 69% of the fish from the Cle Elum section moving over 10 km. This may have resulted because rainbow trout previously captured in the Yakima Canyon had the smallest average movement distances. The average distances moved as reflected by all recaptures from fish previously captured in the Lower Canyon and Upper Canyon sections were not statistically different at 6.3 and 6.0 km (P = 0.709, t = 0.373, df = 421).

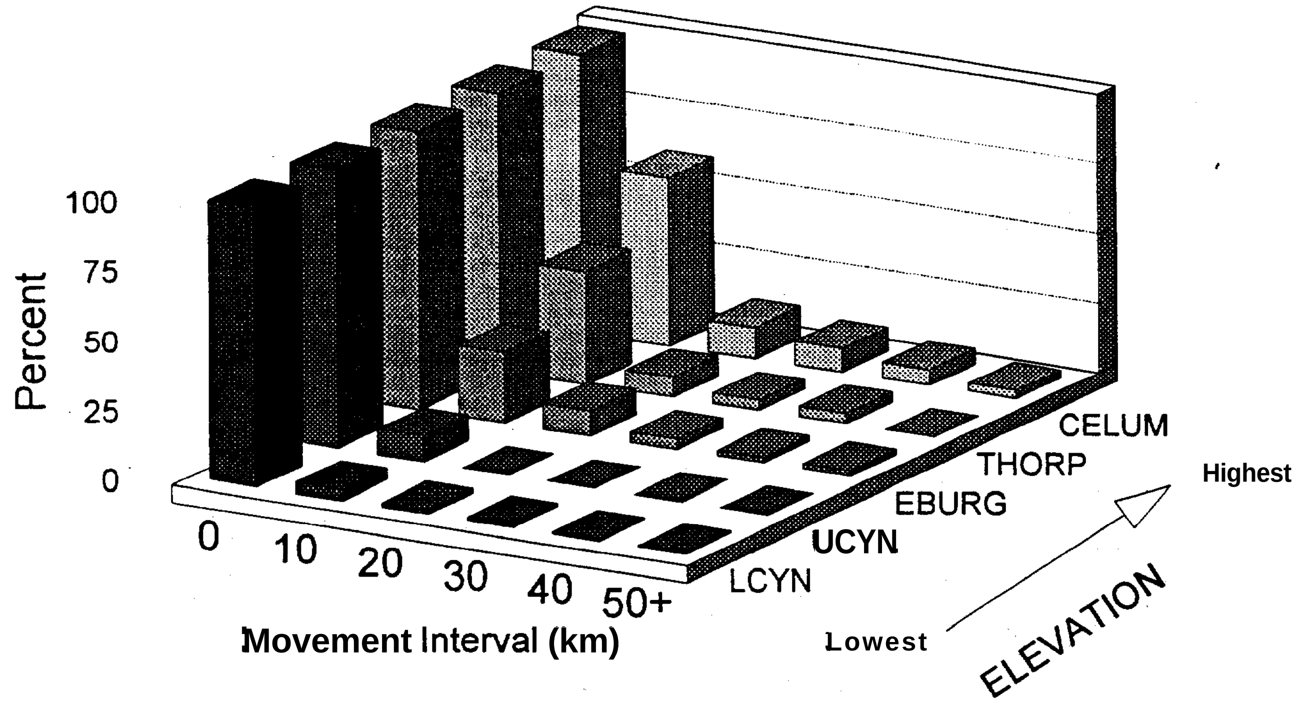

Figure 2. Histogram showing the percentage and the minimum distance moved by rainbow trout captured in different contiguous study sections of the Yakima River ordered from lowest to highest elevation. Movement is based on the average of minimum and maximum calculated movements. LCYN = Lower Canyon (N=324), UCYN = Upper Canyon (N=94), EBURG = Ellensburg (N=55), THORP = Thorp (N=27), CELUM = Cle Elum (N=32).

We did detect some minor differences in movement characteristics between trout in mainstem areas and tributaries. The mean movement of fish previously captured in all mainstem sections combined was 9.0 km based on the stratified data set. The mean movement distance of fish previously captured in tributaries was 6.4 km. This difference was not statistically significant, however ($t = 1.227$, $P = 0.221$, $df = 266$). For average movement distances represented in the stratified data set, 26% of the trout moved a net distance greater than 10 km, if they were previously captured in mainstem areas, and 11% moved more than 10 km ,if they were previously captured in tributaries.

In addition, the distances that tagged rainbow trout moved varied according to which bi-seasonal period they were reobserved. For combined mainstem sections, there was significantly more movement by fish captured and subsequently recaptured in the mainstem if they were recaptured in the winter or spring (December 20 through June 20) versus those recaptured in the summer or fall (June 21 through December 19; $t = 2.187$, $P = 0.029$, $df = 538$).

The interrelationship of tributaries and mainstem areas appears to have a substantial influence regarding the movements of rainbow trout in at least two different life-stages. An exchange of relatively small numbers of sexually mature rainbow trout and juvenile rainbow trout was observed between the mainstem and eight tributaries during the study period. Tributaries apparently provided spawning habitat for fish from all areas, whereas, few sexually mature trout were observed exiting tributaries to spawn in the mainstem. In Umtanum Creek, adult rainbow trout that had formerly immigrated from the mainstem to spawn were reobserved returning to the mainstem river following spawning.

In general, the average of combined minimum and maximum calculated movements appeared to best approximate actual net movements. For trout previously captured in the mainstem Yakima River, stream section boundary information was the most specific information available regarding capture and recapture locations for the calculation of minimum and maximum movement distances in 46% of the computations. Similarly, 57% comprises the utilization of stream section boundaries in the computation of movement distances for trout previously captured in the tributaries. More specific locations were available to calculate movements for the remainder of recapture records. The stratified data set of recapture records for trout previously captured in the mainstem appeared to represent trout movements the most reliably since lower elevation areas were over-represented in the non-stratified set (see Table 1). Observations of rainbow trout previously captured in the Thorp and Cle Elum sections composed 11% of the recapture records. Less than 1% of the recaptured fish were previously captured in the highest study sections, Nelson and Crystal. Averaged values from the stratified and non-stratified data sets were not statistically different ($P = 0.098$, $t = 1.658$, $df = 685$; Figure 3), although the utility of the stratified set was limited by the small sample sizes available from upper elevation study sections.

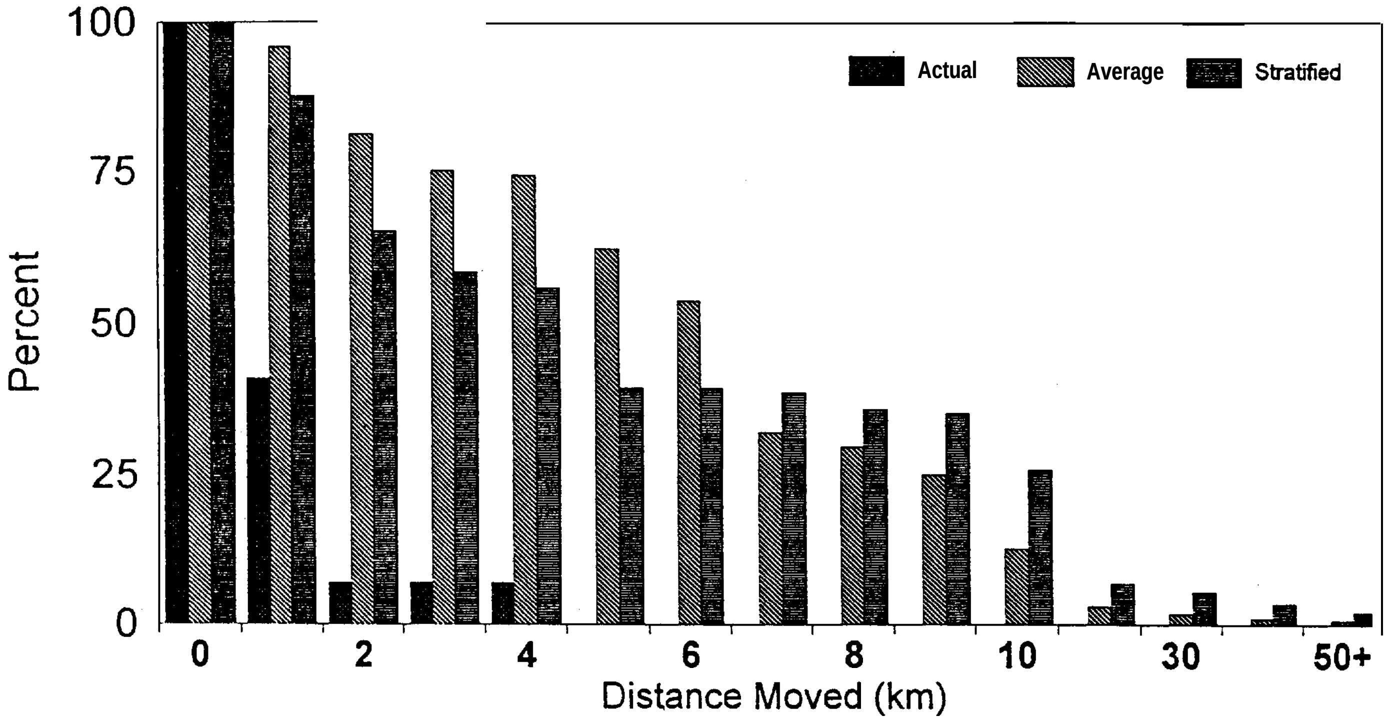

Figure 3. Histogram showing the percentage and the minimum distance moved by rainbow trout in the Yakima River using three different data sets: actual, where point locations were available for capture and recapture (N=27); Average, the average of combined minimum and maximum calculated movements (N=540); and stratified, 30 average values from the Lower Canyon, Upper Canyon, Ellensburg, Thorp (N=27), and Cle Elum section data sets which had the greatest index of data quality (IQ; N=147).

Concerns that unequal study section lengths may have seriously affected the observed trends in movement distances were not supported through regression analyses. Correlation between the lengths of stream-sections and average movement distances yielded a small correlation coefficient, in the case of mainstem data ($r = 0.213$, $P < 0.001$, $df = 539$), and for tributaries, a non-significant relationship ($r = -0.053$, $P = 0.564$, $df = 120$). The maximum differential in mainstem study section lengths was 2:1 (Cle Elum section- 26.1 km and Lower Canyon- 13.3 km). This indicates that section lengths may have affected the outcome of movement distance computations in the mainstem, but did not likely produce a bias greater than the actual movements that were measured.

The analyses of two time-related factors were mixed in results. The relationship between the number of days-at-large and reported movement distances was weak ($r = 0.014$, $P = 0.757$, $df = 539$). However, movement distances of fish that were recaptured multiple times tended to decrease with the number of recaptures, as did the days-at-large (Table 2).

Table 2. Number of observations, means and standard deviations of movement distances (km), and average days-at large by tagged rainbow trout that were recaptured once or multiple times and were previously captured in either the Lower Canyon or the Upper Canyon study sections of the upper Yakima River between 1990 and 1993.

	N	Mean Dist.	SD	Avg. Days
One recapture only	280	6.8	7.5	267
<u>Multiple recaptures</u>:				
1st recapture	65	6.2	3.3	210
2nd recapture	65	4.4	3.7	176
3rd recapture	12	2.6	2.9	164
Combined	143	5.1	3.6	244

The calculation of actual movement distances was possible for 27 mainstem rainbow trout reobservation records. All of the information from fish where point locations were available that pertained to capture and recapture locations was acquired from anglers fishing the lower elevation, Yakima Canyon study sections. Because movement distances tended to increase with elevation in mainstem areas, the actual movement distances were biased toward smaller movements relative to those throughout the mainstem. This inherent sampling bias limited consideration of the actual movement data set as being representative of the entire mainstem by itself. The mean of these actual movement distances was statistically different from the mean of averaged distances for both stratified and complete mainstem data sets.

Actual movement distances calculated for reobserved tributary fish were not significantly different from averaged movement distances ($t = -1.177$, $P = 0.242$, $df = 119$; Figure 4). Averaged mainstem and tributary trout movement distances are most accurate and appropriate for discussion because of the larger sample sizes they represent and because they appear to best represent the available data.

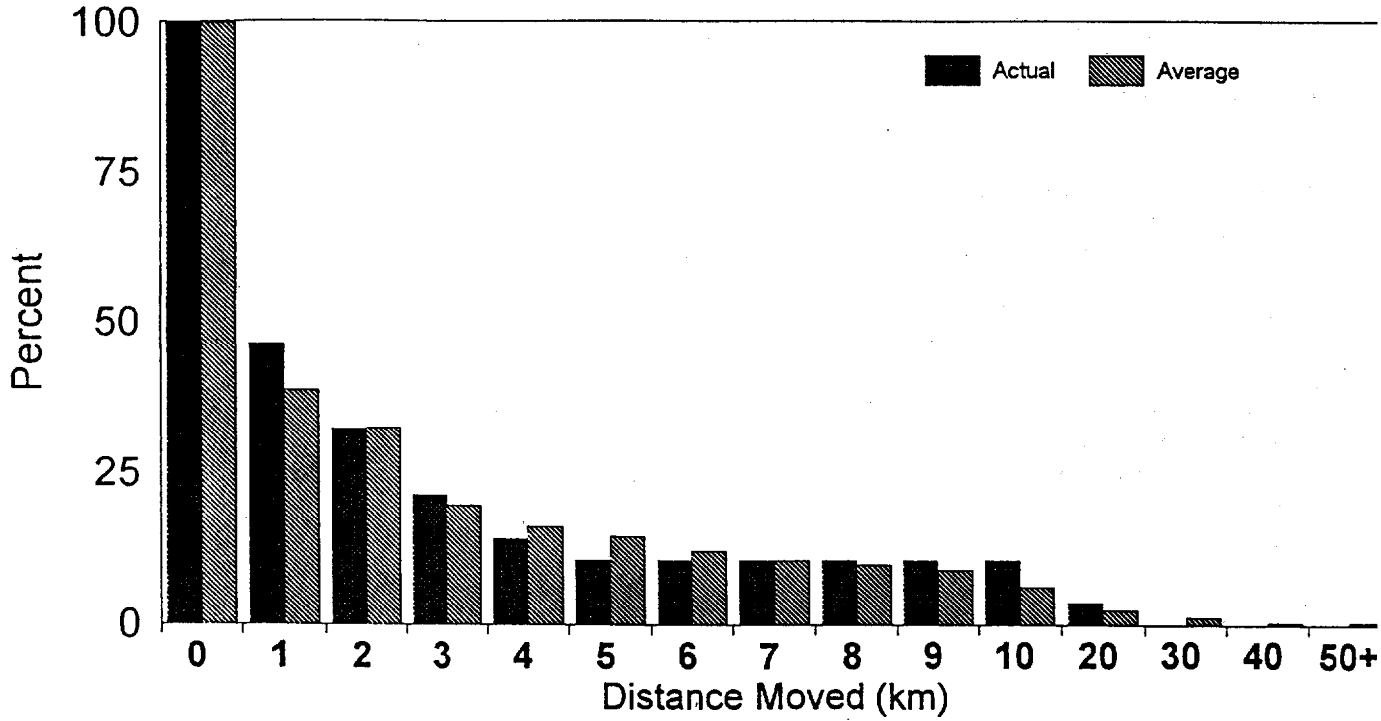

Figure 4. Histogram showing the percentage and the minimum distance moved by rainbow trout previously captured in Yakima River tributaries using two data sets: actual, where point locations were available for capture and recapture (N=28), and average, the average of combined minimum and maximum calculated movements (N=121). Excludes captures and subsequent recaptures if both occurred by trapping in the same trap.

As shown in Table 3, hundreds of juvenile rainbow trout or steelhead pre-smolts were trapped while emigrating in the spring from both lower and upper elevation tributaries. Although these fish were numerous, they presumably constituted a relatively minor source of new individuals to the mainstem population. This is because upper Yakima River redd count data suggest that a large amount of rainbow trout reproduction occurs in areas of the mainstem (see Chapter 1, this report). The mean length of these emigrants was similar between years with the possible exception of fish from the Middle Fork of the Teanaway River.

Table 3. Summary of rainbow trout emigrants trapped in upper Yakima River tributaries during the winter and spring months from 1991 to 1993. Fish that were in spawning condition are not included. Length represents mm fork length. # = actual number trapped. From top to bottom, tributaries are arranged in order of decreasing mean elevation. NFT = North Fork of the Teanaway River, JUN = Jungle Creek, JCK = Jack Creek, TAN = Taneum Creek, MFT = Middle Fork of the Teanaway River, SWK = Swauk Creek, WIL = Wilson Creek, CHR = Cherry Creek, UMT = Umtanum Creek.

		Length			Date	
StreamYr	#	Range	Mean	SD	Captured	Trapping[1]
NFT91[2]	34	65-180	103	25	4/28-5/29	4/22-5/31
NFT92[2]	85	62-177	104	26	4/03-6/01	4/04-5/31
NFT93[2]	171	65-152	100	18	4/02-6/04	4/01-6/04
JUN91[3]	4	58-188	121	59	6/02-6/13	5/29-6/13
JUN92[3]	7	40-126	78	35	5/10-7/29	5/05-8/12
JUN93[3]	419	54-137	77	11	4/30-6/24	4/30-7/13
JCK92[3]	96	42-184	93	23	5/05-7/15	5/05-8/12
JCK93[3]	257	68-162	97	16	4/30-7/02	4/30-7/13
TAN93[4]	6	107-167	141	21	4/11-6/11	3/03-8/11
MFT92[5]	62	67-156	97	22	4/03-5/23	4/04-5/31
MFT93[5]	101	70-178	117	30	3/31-5/31	3/31-6/04
SWK93[4]	5	160-308	235	56	5/13-5/27	3/06-8/11
WIL92[4]	4	127-188	150	27	2/22-3/15	2/17-3/19
CHR92[4]	6	120-214	147	41	3/04-3/24	2/13-3/26
UMT92[3]	322	44-212	98	25	2/16-5/01	2/10-5/01
UMT93[3]	51	53-172	115	23	2/10-5/24	2/10-6/02

[1] Dates from trap installation to removal including intervening dates when trap was out of service
[2] Capture efficiency approximately 3% to 10%
[3] Capture efficiency approaching 100%
[4] Capture efficiency unknown
[5] Capture efficiency approximately 10% to 15%

Mean lengths of trout emigrating from two intermittent feeders entering the North Fork of the Teanaway River, Jack and Jungle creeks, were smaller than those of emigrants trapped at the mouth of the North Fork of the Teanaway River. Despite experiencing a relatively cool summer with above average precipitation during 1991, and in contrast, dry and hot conditions during the summer of 1992, the number of emigrants appeared to be the same or higher in 1993, given uncertain trap capture efficiencies (Figure 5).

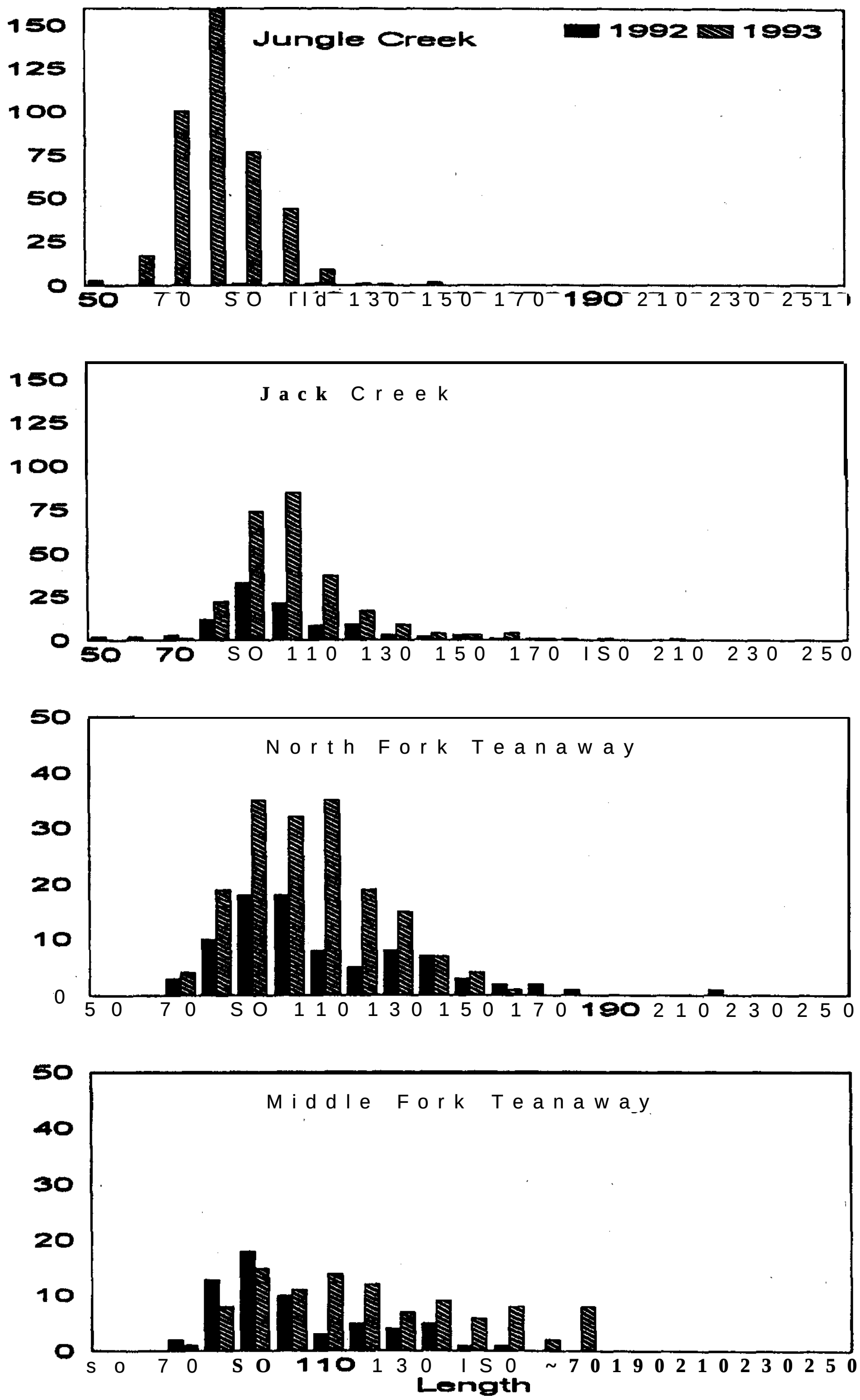

Figure 5. Length frequency histograms of rainbow trout emigrants captured with traps in Jungle Creek and Jack Creek, May through July; and North Fork Teanaway River and Middle Fork Teanaway River, April through June. Length is in mm fork length. Ordinate refers to number of rainbow trout or steelhead pre-smolts.

Only six rainbow trout tagged in tributaries of the Yakima River were recaptured in the Yakima River mainstem and were classified as recruits to the mainstem. The tributaries in which these fish were previously captured and the number of recruits observed were: Umtanum Creek 4, Badger Creek 1, and Manastash Creek 1. The small number of recruits we identified was probably due to the 175 mm minimum length at which anchor tags were applied. If we had applied tags to larger numbers of smaller trout, we may have detected more recruits.

A relatively small proportion of the rainbow trout from the upper Yakima River appeared to spawn in tributaries. Less than nine percent of the mature rainbow trout from the combined Lower Canyon and Upper Canyon mainstem sections spawned in Umtanum Creek during 1992, however, major assumptions should be noted (Table 4). Based on tag reobservations, most of these fish appeared to have migrated upstream in the mainstem before entering a tributary to spawn. For example, 21 of 23 tagged spawners reobserved in Umtanum Creek were previously captured at or below its mouth in the Lower Canyon section of the Yakima River.

Table 4. Estimated utilization by rainbow trout $\geq$250 mm fork length of three upper Yakima River tributaries in the spring for spawning by fish presumed to rear in the mainstem. N1 = number of resident trout $\geq$250 mm caught in upstream traps, N2 = estimated number of resident trout $\geq$250 mm within the presumed rearing area based on information from tagged fish reobservations and Petersen mark-recapture surveys conducted during the previous fall (McMichael et al. 1992; Pearsons et al. 1993). (%) = estimated percent utilization of tributary for spawning by rainbow trout $\geq$250 immigrating from the presumed rearing area after assuming a 0.51 maturity factor (Schroeder and Smith 1989). UMT = Umtanum Creek, SWK = Swauk Creek, TAN = Taneum Creek.

Stream Yr	N1	N2	(%)
UMT 1992[a]	190	4328	8.6
SWK 1993[b]	14	873	3.1
TAN 1993[b,c]	3	873	0.7

[a] Presumed area of origin is the Lower Canyon and Upper Canyon study sections combined.

[b] Presumed area of origin is the Thorp section.

[c] A large number of immigrants may have avoided trapping during high flows that were beyond the trap's working capacity.

Discussion

In general, most rainbow trout in the upper Yakima River basin longer than 175 mm appeared to move less than 5 km. However, fish movement distances in the mainstem of the Yakima River varied with elevation and the time of year. Movement distances there increased with increasing elevation and during the winter and spring period. In addition, movement distances in the mainstem river were generally greater than in the tributaries. Higher environmental stresses, such as greater water velocities and corresponding reductions in refuge habitat, may cause fish to be displaced or move more in upper elevation mainstem sections than in lower sections. Stream discharges are more variable in high elevation sections of the mainstem than in lower ones. In contrast, relative discharges generally fluctuated more in tributaries than in mainstem sections, yet fish movement was less in the tributaries.

Large-scale trout movements in tributaries may be more limited by a stream's natural and unnatural physical features than those in the mainstem. Irrigation diversion dams, beaver dams, stream dewatering, and areas of warm water may serve as complete or partial barriers to fish movement in many tributaries at varying frequencies. At least one of these types of barriers is found in almost every tributary stream we sampled. For example, Umtanum Creek has numerous beaver dams, lower reaches of Swauk Creek became intermittent in some years, and Taneum Creek has numerous diversion dams. Gerking (1950, 1953) suggested that riffles may serve as barriers that fish do not traverse under some conditions. Alternatively, some of these features may provide greater refuge in the form of velocity barriers.

Rainbow trout moved more during the winter and spring than during the summer and fall, presumably because of the migrations associated with spawning activities. Some rainbow trout inhabiting the mainstem made long migrations within the mainstem or migrated into tributaries to spawn. For example, rainbow trout inhabiting the mainstem migrated into Umtanum, Cherry, Wilson, Taneum, and Swauk creeks to spawn (Pearsons et al. 1993).

A collaborative study of rainbow trout movements in the upper Yakima River, using radiotelemetry, between the National Marine Fisheries Service and the Washington Department of Fish and Wildlife tended to support results from the present study. In March, 1993, 50 rainbow trout longer than 300 mm FL were captured in the Lower and Upper canyon study sections, presence and stage of sexual maturity was noted, and radio transmitters were implanted to determine fish spawning locations and movement behaviors (see Chapter 1 of this report). These rainbow trout were tracked for approximately seven months, limited only by the life span of their radio transmitter. Most of the movements observed between March and September occurred in association with spawning activities that peaked in late March and April (Eric Hockersmith, National Marine Fisheries Service, personal communication). Spawning migrations to the presumed spawning site were from 0.3 to 87.2 km long. Less than 18% of the radiotagged fish that appeared to spawn did so in tributaries (10% spawned in Umtanum Creek, 2% in Cherry Creek, and 5% in the Teanaway River basin). Most of the radiotagged fish that spawned in tributaries returned to the mainstem after spawning was complete.

The exchange of rainbow trout ≥175 mm in length between the mainstem and its tributaries appeared to have been relatively minor, with the most exchange occurring during the spring (Pearsons et al. 1993). However, the exchange of smaller fish (<175 mm) appeared to have been considerable. Migrant trapping during the spring months indicated that many small rainbow trout or steelhead pre-smolts exited small tributaries and entered larger water bodies, such as the Yakima River, the Teanaway River, and the North Fork of the Teanaway River. Unfortunately, we were unable to determine movement patterns of small fish with most of the techniques we used. The use of VI or dangler tags did not contribute sufficient results. Reasons for this may have included the relatively small number of tags applied, possible tag application to steelhead pre-smolts, tag loss, or poor identification of tag presence if VI tagged fish were recaptured. We did not use anchor tags on fish smaller than 175 mm because we were concerned that the application procedure would contribute to increased mortality or severely altered behavior (Everhart and Youngs 1981; McFarlane et al. 1990). In short, our knowledge is limited regarding the movement patterns of rainbow trout in smaller size classes.

The patterns of fish movement we interpreted from our study may also be a result of biases in our study design, sampling effort, or analyses. As mentioned previously, this study was not specifically designed to answer many of the questions that are addressed in this chapter. Thus, we tried to determine the potential sources of bias and levels of precision that might have been affected by our study design, sampling effort, and analyses. After identifying sources of bias and making appropriate adjustments, we found no source of bias other than section length that would considerably influence the interpretation of the results.

Unequal catchability of rainbow trout between different study sections and between different seasons may bias results if movement patterns are not similar. Access to the most fish for tagging and recapture occurred in the Yakima Canyon, presumably because the conductivities were relatively high and slower water velocities increased capture success. The effect of this disproportionately high rate of tagging and recaptures within the Canyon sections may have been to bias our description of the mean movements of fish throughout the whole mainstem. For this reason, the stratified data set was assembled and analyzed to describe mean movement trends for the entire mainstem. The index of quality that we used to select the most reliable movement distances for the stratified set introduced a dual bias as it approached unity; one favored trout recapture observations with "actual" movement distances that underestimated mean movement, and another that favored fish displaying longer migration distances.

In upper elevation sections of the mainstem, conditions for fish capture using electrofishing equipment were poor in the spring and were fair to good in the fall. This reduced the possibility of recapturing rainbow trout in upper elevation sections that had been previously captured in lower elevation sections. This possibility may have accentuated differences in the amount of movement that was suggested to have occurred between lower and upper elevation sections. However, conductivities in Cle Elum section were roughly equal to those of the Thorp and Ellensburg sections. Yet, the percentage of rainbow trout previously captured in the Cle Elum section (N=35) that moved distances of over 10 km was 81% higher than comparable movements in the Thorp (N=27) and Ellensburg sections (N=55), when both were combined.

Exclusion of data that were suspected to have been juvenile steelhead as determined by size, timing, and their presence below Roza Dam may have also inadvertently excluded some resident rainbow trout data. This may have reduced our detection and consideration of individuals that exited the study area and consequently reduced our estimates of total movement distance.

The precision of our movement distance estimates (particularly where section boundaries were used to estimate the average distance moved) would have been improved if more and shorter study sections had been established (Funk 1956, Hill and Grossman 1987). Funk (1956) reviewed the limitations of estimating movement using tagging methods, many of which were described above.

We included multiple data points from fish that were recaptured more than once to augment sample sizes and for reasons identified by Gatz and Adams (1994). They felt that alternatives such as using distances between all possible pairs of captures or using only the distance between the first capture the last recapture would be less representative than treating each capture and recapture independently. However, preliminary investigation of the information from repeated recaptures indicates that the movement distance recorded decreased with a greater number of recaptures. Further examination of this issue will be treated in a subsequent report.

Other research has shown that most large rainbow trout in streams move considerably less than 15 km, with most moving less than 5 km between capture and recapture (Stefanich 1952, Cargill 1980, Vincent 1987, Schroeder and Smith 1989). Northcote (1992) found that adult rainbow trout moved the most during the spring when they are migrating to spawning areas. Other researchers have also found much fish movement to be associated with high environmental variability (Funk 1955; Pearsons 1994).

Most studies of movement by resident fish have found that the majority of individuals tend to express "limited movement" and that a small fraction are strays (Stefanich 1952,;Funk 1956;

Gerking 1959; Mense 1975; Cargill 1980; Vincent 1987; and Northcote 1992). We could not determine a minimum distance that trout may have moved for them to be referred to as strays, since rainbow trout in the upper Yakima River basin appeared to move over a gradient of distances, rather than displaying separate or distinct modes of movement. We found that the distances rainbow trout moved were comparable to the movements of the salmonid species other researchers have studied.

In summary, our preliminary results and analyses suggest that most of the large rainbow trout observed in this study moved less than 5 km, although fish movement distance varied with environmental conditions and the time of the year. Although small fish were captured emigrating from some tributary streams, the movement distances of small rainbow trout remains largely unknown. We recommend that the information on movement distances provided in this chapter might be used to determine appropriate spatial and temporal scales used for monitoring rainbow trout population sizes (see General Discussion of this report).

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Chapter 4

Rainbow trout distribution and population abundance variation in the upper Yakima River basin

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Abstract

Detection of impacts to fish population abundance and distribution caused by a disturbance is challenging. This is due to high spatial and temporal variability observed for many populations. To determine if supplementation of anadromous salmonids impacts wild salmonid populations, we described the salmonid density variation, both in space and time prior to supplementation in the upper Yakima River basin. In addition, we attempted to identify causal factors that might be responsible for the variation observed. Population densities of rainbow trout were calculated in 100 m long index sites of eight tributaries and five mainstem sections (4.0 - 6.3 km long) in the upper Yakima River from 1990 through 1993. In tributaries, population estimates were conducted using back-pack electrofishing using removal-depletion methodologies. In the mainstem, trout were collected by driftboat electrofishing and population estimates were calculated using Peterson mark-recapture methods. In tributaries, the number of trout in index sites has fluctuated considerably since 1990, ranging from 0 to 0.24 fish/m^2. However, within individual index sites, temporal variation of rainbow trout densities was much lower than the variation observed between index sites. Temporal variability was the highest in an index site in Taneum Creek where the density of rainbow trout was 0.09 fish/m^2 in 1992 and 0.23 fish/m^2 in 1993. Spatial variation between tributaries may be explained by geographic location and habitat quality, and temporal variation may be explained by habitat instability, environmental variation and fish movement. Population variation in the mainstem was not as high as in the tributaries, which may be due to homogeneity in habitat among mainstem sites. Also, interannual and spatial population variation may be contained in mainstem index sites due to the large size of the index sites. The number of trout in five sections of the mainstem Yakima River has averaged 325/km and has fluctuated only moderately from 1991 through 1993. Although we have observed spatial and temporal variation in the upper Yakima River, it appears that population variance may be explained by a combination of physical, environmental and biotic variables. These variables may help explain spatial and temporal variance and thus increase our ability to detect the effects of a supplementation program or other unnatural or natural causes in the Yakima River basin.

Introduction

The spatial and temporal variation in abundance and distribution of plants and animals has interested ecologists for over a century (Cowles 1889; Shelford 1911; Whittaker 1960). Fish populations are dynamic and some may fluctuate both in abundance and distribution considerably over relatively short periods of time (Decker and Erman 1992). Studies of stream fish have shown that abiotic factors such as temperature, current velocity and stream flow, elevation above sea level, and substrate diversity can determine the distribution and abundance of individual species (Gorman and Karr 1978; Scarnecchia 1987; Nelson et al. 1992) and that the quality of stream habitat may limit population levels (Lewis 1969). The temporal and spatial distribution of trout may therefore be directly related to local habitat conditions to which populations must adapt if they are to persist (Nelson et al. 1992). After analysis of 12 years of data, Grossman et al. (1982) concluded that an Indiana stream fish assemblage was regulated by fluctuating environmental conditions, not deterministic factors such as competition. In addition, information exists about the strong influence that environmental factors may have on fish distribution (Gorman 1987; Pearsons et al. 1992). However, it may be argued that trout distributions and habitat utilization are directly related to competitive interactions among species (Chapman 1966; Fausch and White 1981). In addition, studies have suggested that the density of salmonids is closely related to competition for limited resources (Chapman 1966; Kennedy and Strange 1986).

Spatial and temporal variation in trout abundance and distribution can provide challenges to resource managers who must monitor the status of a population. Resource managers frequently try to assess whether a particular management action might be beneficial or detrimental to a species of interest. If spatial and temporal variations are high, effects of a management action may be difficult to detect. Sufficient statistical power, coupled with an understanding of the factors that influence trout density are necessary to be able to determine what effects specific management actions might have on trout. Understanding the predictable biotic and abiotic factors that influence trout densities can aid in the development of models to predict trout densities. Currently, many models exist that were designed to describe relationships between fish species and their habitat (Binns and Eiserman 1979; Lanka et al. 1987; Nelson et al 1992). Modeling efforts however, have met with varying degrees of success in terms of accuracy and precision, and have often had little reliability outside the areas in which they were developed (Fausch et al. 1988). Models that are developed to predict responses of fish populations are more accurate if they are developed and applied within a specific ecoregion (Fausch et al. 1988).

The purposes of this chapter are to (1) describe the spatial and temporal variation of abundance and distribution of salmonids in the mainstem of the Yakima River and it's tributaries above Roza Dam, and (2) assess physical factors that may explain that temporal and spatial variation. Based on the results of this work, a model or criteria can be developed to aid in investigating whether spring chinook salmon (*Oncorhynchus tshawytscha*) and/or steelhead trout (*O. mykiss*) supplementation affect the status of the resident wild rainbow trout (*O. mykiss*) population in the upper Yakima River.

Methods

Tributaries

Salmonid rearing densities were determined from 1990 through 1993 in several tributaries of the upper Yakima River to evaluate their spatial and temporal distribution (McMichael et al. 1992; Pearsons et al. 1993). The number of tributaries sampled increased from five in 1990 (McMichael et al. 1992), to 10 in 1992 (Pearsons et al. 1993). In 1993, eight tributaries with a total of 21 index sites were sampled. The five original tributaries sampled in 1990 and 1991 were selected based on two criteria (1) the tributary was likely to contain resident trout that could be affected by the presence of hatchery-origin anadromous fish, and (2) an acclimation facility was proposed to be developed along the tributary (McMichael et al. 1992). Index sites were selected within three elevational strata of each tributary based on the following criteria, (1) the index site contained a range of habitat types that could be effectively sampled, and (2) access to the site was possible. The index sites were intended to represent each strata within a tributary, so that spatial (between sites) and temporal (interannual, within sites) population abundance variability could be monitored. Tributaries were sampled in 1993 if at least two years of previous data were available, and (1) the tributary represented a different elevation than any of the others sampled in 1993, (2) the tributary might be slated to receive releases of hatchery steelhead and/or salmon in the future, or the tributary serves as a control for a tributary that may receive releases of hatchery steelhead and/or salmon in the future.

The abundance of rainbow trout greater than 79 mm and juvenile spring chinook salmon (all sizes) was estimated using removal-depletion methods (Zippen 1958; McMichael et al. 1992) in eight Yakima River tributaries above Roza Dam (rk 180) in 1993 (Figure 1). Although hatchery-origin steelhead trout were present in some sites within the Teanaway River basin, these fish were not included in the *O. mykiss* population estimates because we wanted to focus on naturally produced fish. Rainbow trout population estimates were based only on those rainbow trout greater than 79 mm because of typically poor electrofishing sampling effeciencies for smaller fish. In addition to rainbow trout and juvenile spring chinook salmon, population estimates were also calculated for eastern brook trout (*Salvelinus fontinalis*), cutthroat trout (*O. clarki*) and bull trout (*S. confluentus*) when encountered. Because our objective was to calculate population estimates for rainbow trout, estimates for salmonid species other than *O. mykiss* were not always valid due to unequal probability of capture on successive electrofishing passes. For instance, to meet our criteria, the number of individuals within a species collected on the second pass must be less than 50% of the number collected on the first pass. If the number collected on the second effort was greater than 50% of the number removed on the first pass, then the estimate was deemed to be invalid.

Each tributary was divided into three elevational strata, each having approximately equal length, from the tributary mouth to the highest elevation of known anadromous salmonid migration (BPA 1990). One index site was located in each of the three strata. In 1993, juvenile spring chinook salmon population estimates were calculated. If salmon were captured, additional passes were conducted to achieve a 50% removal pattern from the first to the second pass. In 1990, 1991 and 1992, rainbow x cutthroat hybrids that were collected were classified as "hybrid", and were evaluated in a separate trout category. Subsequent genetic sampling in the Yakima Basin showed that the majority of these "hybrids" were genetically pure rainbow trout (Pearsons et al. 1993). As a result, in 1993, all putative hybrids were classified as either rainbow trout or cutthroat. Cutthroat trout were visually identified by their spotting pattern, bright orange hyoid slash, and maxillaries that extended beyond the posterior edge of the eye (Wydoski and Whitney 1979). Rainbow trout were visually identified by their lack of a bright orange hyoid slash, spots that were distributed evenly above the lateral line from the operculum to the caudal fin, and maxillaries that did not extend beyond the posterior edge of the eye (Wydoski and Whitney 1979). It was not uncommon for a rainbow trout to have dull-orange hyoid slashes.

In addition to population estimates, habitat area, stream discharge, water temperature, longitudinal streambed profile (thalweg depth) and gradient were recorded for each index site. Within sites, habitat types in 1990, 1991 and 1992 were identified and measured as described in Bisson et al. (1982). In 1993, we used a hierarchical classification system to identify habitat unit types based on the methodology described by Hawkins et al. (1993). These habitat units were classified into either fast or slow water categories. Slow water habitat was then classified as either "slow and uniform" or "slow with at least one deep spot greater than 150% of the mean depth for that unit". The categories fast, slow with uniform depth, and slow with atleast one deep spot, approximately correspond to the commonly used terms "riffle", "run" and "pool", respectively. The use of this hierarchial approach provided a consistent means for identifying habitat type. Identifying channel geomorphic units based on hierarchical classification alleviates problems that some users have with Bisson et al. (1982) original classification scheme (Hawkins et al. 1993). Bisson et al. (1982) classification scheme was based on numerous habitat unit types, requiring the user to classify a unit into a specific type. We found that by using a hierarchical classificaction scheme the user can arrive at the habitat unit type easily, simply by answering a series of yes or no questions.

The surface area of each habitat unit was calculated by multiplying the length by the width for each individual unit type and then summing each area per type. Standard deviation of water depth in the thalweg (thalweg depth) was used as an index of habitat complexity (Kaufmann 1987). Thalweg depth (m) was taken at approximately 1 m intervals from the upstream boundary of the site to the downstream boundary of the site. Stream temperature was recorded at the time of each population estimate. In addition to recording instantaneous stream temperature, in 1993 maximum-minimum thermometers were deployed at the middle elevation site in each tributary to obtain monthly maximum and minimum stream temperatures.

The number of index sites surveyed in each tributary during 1993 was two in Cabin Creek, one in Jungle Creek, three in each of the Middle Fork, North Fork and West Fork of the Teanaway River, three in Swauk Creek, three in Taneum Creek and three in Umtanum Creek. In 1992, only two sites were surveyed in Umtanum Creek. In 1993, an additional site was surveyed to be consistent with sample sizes in other tributaries and to further describe trout density and distribution in this creek. Although Big and Manastash creeks were surveyed in 1992, they were not resurveyed in 1993. A total of 25 different index sites have been used for population estimates since the study began in 1990, including 14 that have been used for all four years.

To determine if population estimates in index sites were representative of the rearing density in each stream reach, five additional sites were selected (termed random sites) in both the Middle and North forks of the Teanaway River in 1993. Population estimates in these random sites were then compared to index sites in those streams. The location of random sites was established by first determining the length of each tributary utilized by anadromous fish, and then dividing this length by five. The result was five sections of equal length. A random site was then located within the lower most section by selecting a random number from a random numbers table. The number served as the number of meters upstream from the mouth that the first site was located. The four remaining sites were then located upstream from the first site, at a distance equal to the section length. Sampling methods used in the random sites were the same as those used in the index sites.

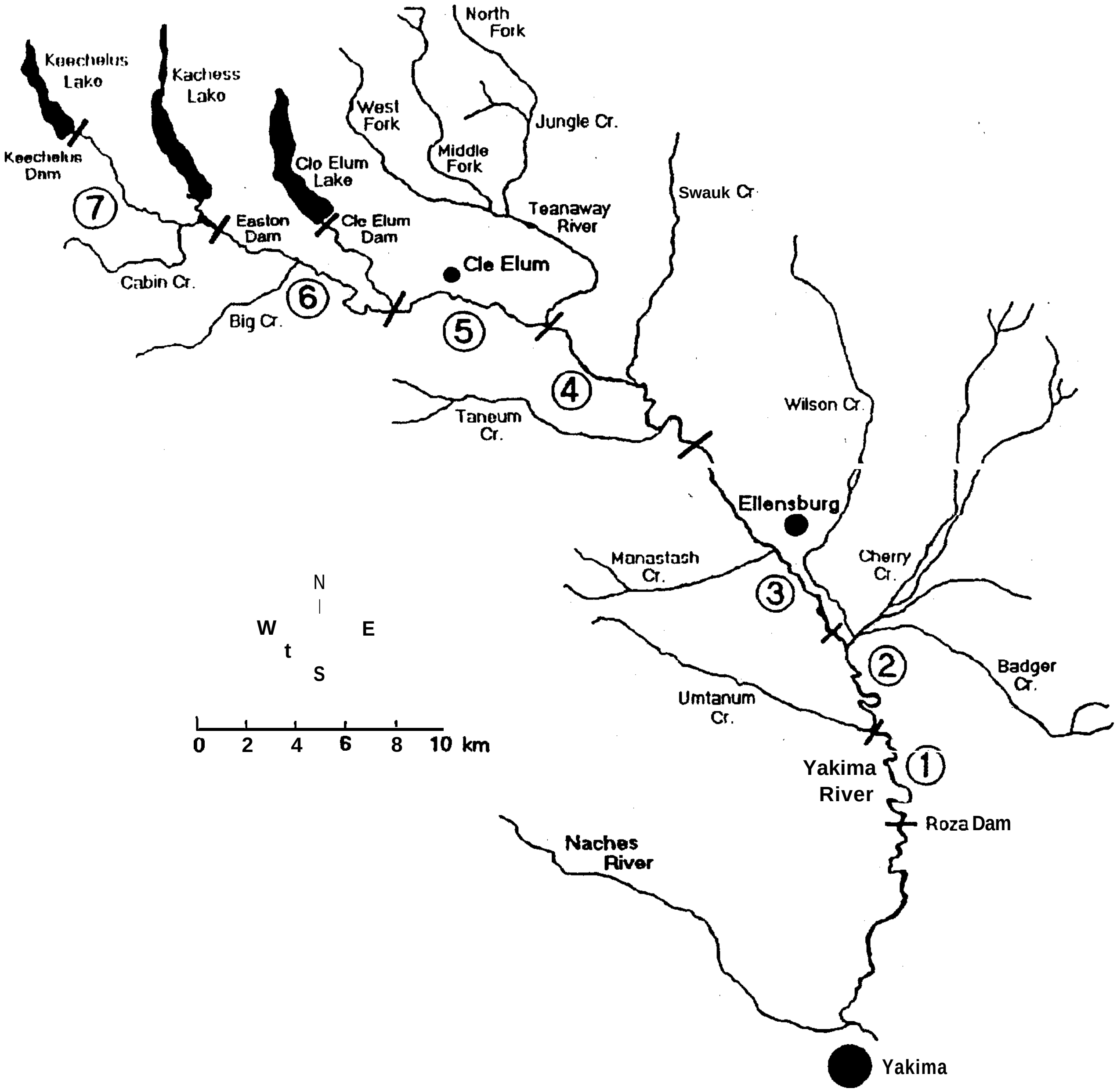

Figure 1. Map of the Yakima River drainage north of the city of Yakima. Study area includes the Yakima River and its tributaries between Roza Dam and Keechelus Dam. Mainstem Yakima River study sections, seperated by heavy lines on the figure (1 - 7), are labeled with circles.

Both within and between-year correlations between biotic and physical variables were examined in each index site from 1990 to 1993 using Pearson product-moment correlations with Statgraphics Plus software (Manugistics Corporation 1992). Data interpretation was assisted by calculating the percent error rate due to our multiple testing of dependent data (Ottenbacher 1989). Percent error (PE) was calculated using the following formula (Ottenbacher 1989):

$$PE = 100\ (c)\ (\alpha)\ /\ (M)$$

Where c = number of correlations run;
α = alpha level; and,
M = number of comparisons that are statistically significant.

Mainstem Yakima River Trout Population Estimates

From 1991 to 1993, trout population estimates were conducted in five sections of the mainstem Yakima River by using mark-recapture methods (Ricker 1975) to assess the spatial and temporal distribution of trout as described by McMichael et al. (1992) and Pearsons et al. (1993). Young-of-year trout and juvenile spring chinook salmon were not included in these population estimates (see chapter 6 for treatment of spring chinook abundance) due to poor electrofishing efficiencies for small fish. Briefly, the mainstem Yakima River was divided into five study sections based on geographical features as described in Hindman et al. (1991) and McMichael et al. (1992) (Figure 1). One index site approximately 5 km long was located within each of the study sections, and sections were numbered sequentially from the lowest elevation to highest elevation. The section numbers and names were as follows: 1, Lower Canyon (LCYN); 2, Upper Canyon (UCYN); 3, Ellensburg (EBURG); 4, Thorp (THORP); and, 5, Cle Elum (CELUM). All salmonids were captured and marked on two successive nights using a driftboat electrofisher. One week later fish were recaptured on two successive nights. Methods were detailed by McMichael et al. (1992).

Lineal fish densities and biomass were calculated by dividing the population and biomass estimates by the length of the index site and are reported as the number and kilograms of trout per kilometer, respectively, for each of the five sites surveyed. In addition to calculating trout population estimates and number of fish per kilometer, areal densities and biomass were also calculated so that comparisons could be made with tributaries in the upper Yakima Basin and other river systems in the western United States. Areal density calculations were made by dividing the total trout population estimate and biomass of trout by the site area. Site area was determined by floating each site and recording the river width at approximately 100 m intervals with an optical range finder. Site area was then calculated by multiplying the average river width by the length of each index site. River discharge information was provided by the United States Bureau of Reclamation. As described in the tributary section of this chapter, putative rainbow x cutthroat hybrids were considered to be a separate trout category in 1991 and 1992. In 1993, all hybrids were classified to either the rainbow or cutthroat trout category only.

Results

Tributaries

In index sites in tributaries of the upper Yakima basin, the densities of rainbow trout generally showed little temporal variation from 1990 through 1993, but did show high spatial variation between index sites and tributaries. Exceptions to this pattern included Taneum and Jungle creeks, within which densities varied considerably from 1990 through 1993 (Table 1, Figure 2). Average densities of rainbow trout in index sites were highest in Jungle, Taneum and Swauk creeks, while Cabin Creek and the North Fork of the Teanaway River were the lowest. Rainbow trout densities in index sites in 1993 were significantly correlated to rainbow trout densities in 1992 ($r=0.91$, $P=0.00001$). Densities were also significantly correlated between 1993 and 1991 index sites ($r=0.80$, $P = 0.0003$) and between 1992 and 1991 ($r=0.89$, $P=0.00001$). However, a significant correlation was not observed between rainbow trout densities in 1990 compared to any other year sampled, which may be attributed to the small number of index sites surveyed in the 1990 season.

Although rainbow trout density varied little between years, rainbow trout biomass variability was high (Figure 3). The only significant rainbow trout biomass correlation was between years 1990 and 1991 ($r=0.86$, $P<0.0001$). No other significant correlations existed for rainbow trout biomass between years.

The overall mean fork length of rainbow trout in the tributary index sites appeared to be similar between years and between tributaries (Table 1, Figure 4). The length-at-age for rainbow trout in tributary index sites, however, was quite different (see chapter 5). Rainbow trout in Umtanum Creek index sites had the largest mean length in 1993, followed in order by Swauk and Cabin creek index sites.

The density and biomass of spring chinook salmon and bull, brook, and cutthroat trout exhibited high spatial variation within sampling years but were similar between years at individual sites (Tables 2 and 3). Of the 72 population estimates in the 25 index sites sampled from 1990 through 1993, we found no spatial overlap between spring chinook salmon and bull trout. Spring chinook salmon did overlap with brook or cutthroat trout in 5 (7%) of the 72 sites sampled. All juvenile spring chinook salmon were observed in sites less than 730 m elevation, while bull trout were observed only in high elevation sites (1,091 m to 1,103 m elevation). Cutthroat and brook trout were observed inhabiting sites covering a wide range of elevations. Rainbow trout, were the most ubiquitous, being observed in 70 of the 72 sites sampled. The two index sites that did not contain rainbow trout were the highest elevation site (1,341 m) in Manastash Creek in 1992, and the lowest elevation (719 m) site in Cabin Creek in 1993. Tributary index sites ranged in elevation from 469 m to 1,341 m. The only sites that contained bull trout were the highest elevation index site and highest elevation random site in the North Fork of the Teanaway River (1,103 m and 1,091 m, respectively). In general, cutthroat and brook trout densities were highest in high elevation index sites. Cutthroat trout were found in 29% of the 72 index sites between 677 and 988 m elevation and also in mainstem Yakima River sections. Brook trout were found in 21% of the index sites between 719 and 988 m elevation. As with cutthroat trout, brook trout were also collected in mainstem Yakima River sections.

There were no significant correlations observed between rainbow trout densities and any other salmonid species densities in tributary index sites within 1993. In addition, rainbow trout abundance was not strongly correlated to that of any other salmonid fish species encountered in 1993. Although, in 1993, there were no significant correlations between spring chinook salmon densities and any other salmonid species densities in tributary index sites, the spatial distribution of spring chinook salmon overlapped completely with that of rainbow trout.

Table 1. Rainbow trout density (#/m²), biomass (g/m²) and mean fork length (mm) of fish > 79 mm, for each index site sampled in each upper Yakima River tributary from 1990 through 1993. The average and standard deviation (SD) are also shown. Tributaries are listed from high to low elevation (measured as the average elevation for the three index sites).

	1990			1991			1992			1993		
Site*	Density	Biomass	Ln	Density	Biomass	Ln	Density	Biomass	Ln	Density	B i i	Ln
MAN1							0.038	0.641	93.6	•		
MAN2							0.039	1.168	132.5			
MAN3							0.000	0.000	0.0			
avg							0.026	0.606	113.1			
SD							0.022	0.582	68.1			
NFT1	0.014	0.346	96.4	0.031	0.746	126.5	0.031	0.600	120.9	0.024	0.535	126.9
NFT2	0.070	2.001	124.1	0.030	0.709	120.3	0.021	0.482	123.8	0.031	0.618	120.7
NFT3	0.005	0.123	82.5	0.006	0.269	148.3	0.013	0.245	111.0	0.00s	0.144	108.2
avg	0.065	0.823	101.0	0.022	0.575	131.7	0.022	0.442	118.6	0.020	0.432	118.6
SD	0.048	1.026	21.2	0.014	0.26s	14.7	0.009	0.181	6.7	0.013	0.253	9.5
JUN				0.020	0.190		0.080	0.110	loo.s	0.150	1.793	loo.3
TAN1		2.368	121.5	0.087	13.466	139.3	0.233	7.062	133.9	0.198	5.596	128.1
TAN2	0.060	0.303	106.5	0.071	3.096	138.3	0.132	4.524	137.5	0.110	3.875	140.8
TAN3	0.060	I.336	74.1	0.025	0.528	113.9	0.028	0.944	138.2	0.033	2.849	128.2
avg	0.060	1.336	100.7	0.061	5.697	130.5	0.132	4.177	136.5	0.114	4.107	132.4
SD	0.00	1.033	24.2	0.032	6.85	14.4	0.103	3.07	2.2	0.083	1.39	7.3
MFT1	0.016	4.047	117.0	0.059	1.438	122.8	0.044	1.090	131.4	0.030	0.911	138.1
MFT2	0.103	3.208	117.2	0.044	1.153	128.7	0.027	0.673	121.1	0.029	0.842	134.1
hln-3	0.080	2.414	109.3	0.050	1.983	145.6	0.074	2.366	140.0	0.061	1.277	120.7
avg	0.066	3.223	114.5	0.051	1.52s	132.4	0.049	1.376	130.8	0.040	1.01	131.0
SD	0.045	0.817	4.5	0.008	0.423	11.8	0.024	0.882	9.4	0.01	0.234	9.1
WFT1	0 051	1.208	92.6	0.020	0.562	135.0	0.016	0.371	121.1	0.026	0.909	142.7
WFT2	0.036	1.107	132.0	0.056	1.720	134.2	0.075	1 .a75	129.9	0.037	1.091	135.2
WFT3	0.020	0.472	111.3	0.033	0.570	110.3	0.039	1.005	128.1	0.025	0.425	113.8
avg	0.036	0.929	112.0	0.036	0.951	126.5	0.043	1.084	126.4	0.029	0.808	130.6
SD	0.015	0.399	19.7	0.018	0.666	140.0	0.030	0.755	4.6	0.001	0.344	ls.o
CAB1	0.016	0.003	90.5	0.008	0.193	128.0	0.013	0.252	122.6	0.000	0.000	0.0
CAB2	0.029	0.507	109.8	0.042	0.251	186.0	0.047	1.210	120.6	0.011	0.586	132.7
avg	0.022	0.275	100.2	0.023	0.222	157.0	0.030	0.731	121.6	0.005	0.293	132.7
SD	0.001	0.328	13.6	0.025	0.041	41.0	0.024	0.671	1.4	0.008	0.414	0.0
SWK1										0.113	5.232	157.3
SWK2							0.242	a.719	142.9	0.128	4.111	135.2
SWK3							0.103	2.557	125.7	0.105	2.630	126.0
avg							0.173	5.638	134.3	0.115	3.991	139.6
SD							0.098	4.357	12.2	0.012	1.30s	16.2
BIG							0.071	1.979	126.5			
UMT1							0.111	1.618	107.2	0.086	1.912	119.2
UMT1.5										0.016	2.113	228.8
UMT2							0.016	0.624	151.7	0.012	1.657	216.7
avg							0.063	1.121	1.9	0.038	1.894	188.2
SD							0.067	0.703	0.2	0.042	0.229	60.1

* MAN = Manastash Creek, NFT = North Fork Teanaway River. JUN = Jungle Creek, TAN = Taneum Creek, MFT = Middle Fork Teanaway River. CAB = Cabin Creek, SWK = Swauk Creek, BIG = Big Creek, UMT = Umtanum Creek.

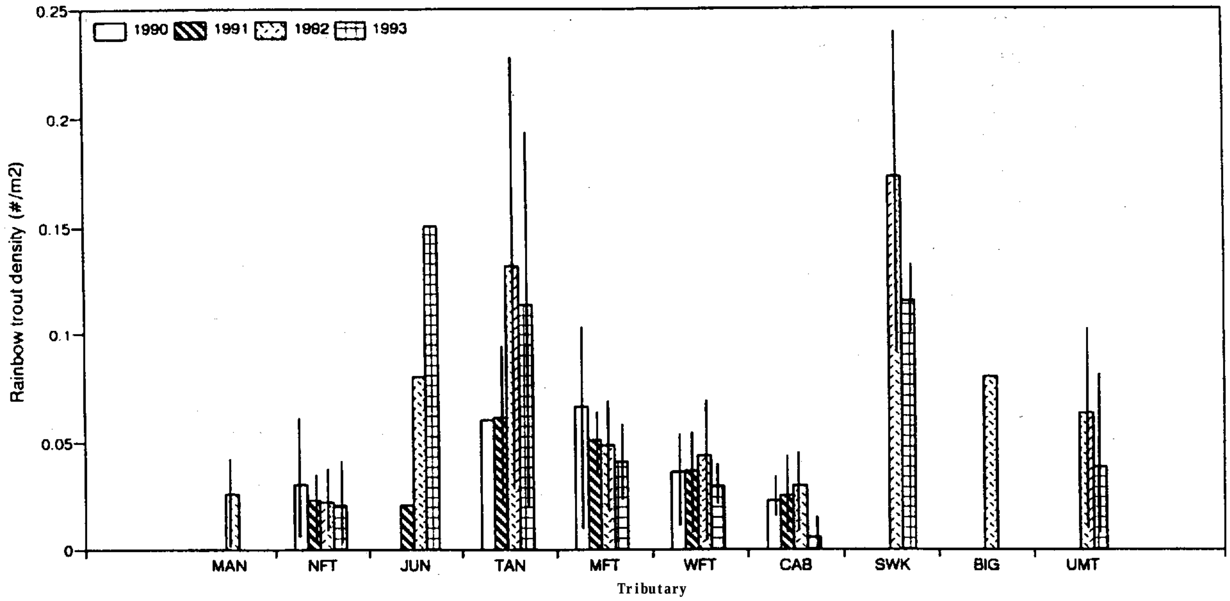

Figure 2. Mean rainbow trout density ($\#/m^2$) in 10 Yakima River tributaries sampled from 1990 through 1993. Vertical lines represent the range between the maximum and minimum densities each year.

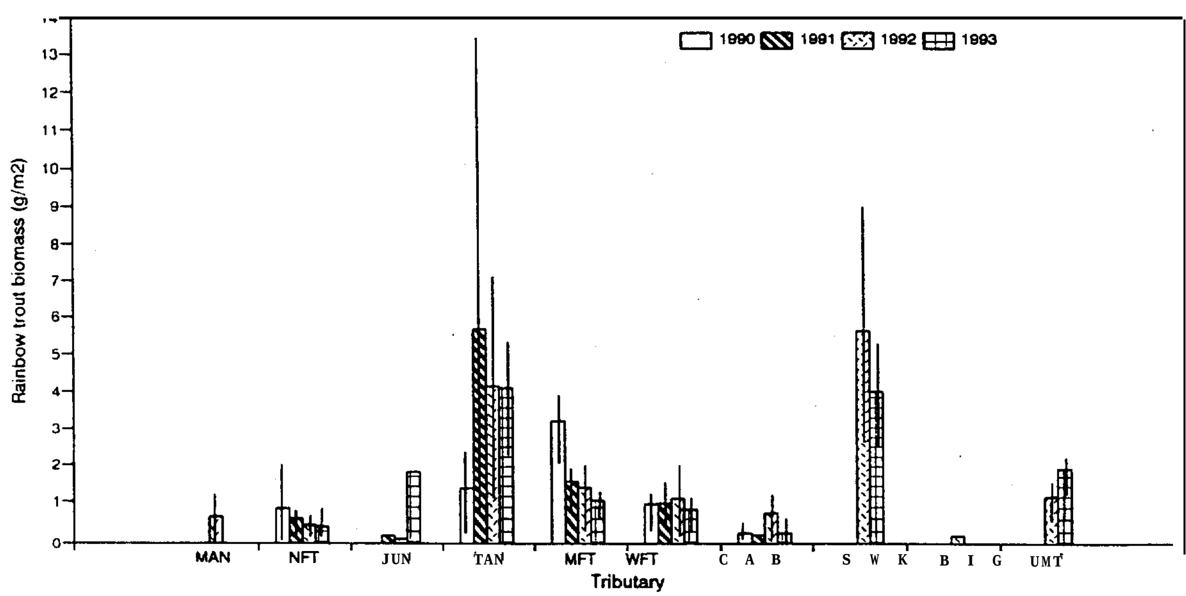

Figure 3. Mean rainbow trout biomass (g/m^2) in 10 Yakima River tributaries sampled from 1990 through 1993. Vertical lines represent the range between the maximum and minimum densities each year.

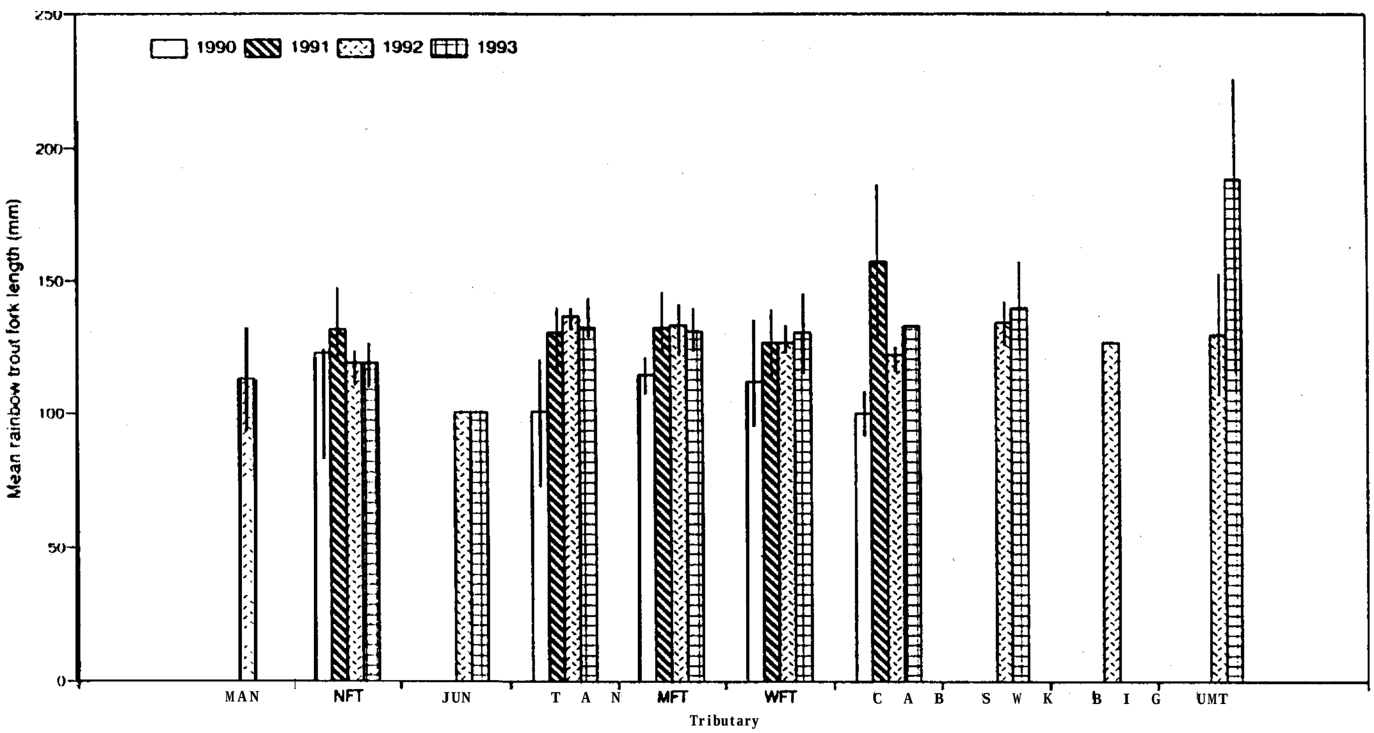

Figure 4. Mean length of rainbow trout colleted in ten tributaries of the Yakima River from 1990 through 1993. Vertical lines represent the range between the mean maximum and mean minimum length of rainbow trout collected from index sites.

Table 2. Density ($\#/m^2$) of juvenile spring chinook salmon and bull, eastern brook and cutthroat trout in each index site surveyed in tributaries of the Yakima River from 1990 to 1993. The average and standard deviation (SD) are also reported. Tributaries are listed from high to low elevation (measured as the average elevation for the three index sites).

	1990				1991				1992				1993			
Site[a]	SCS[b]	BUL	EBT	CUT	SCS	BUL	EBT	CUT	SCS	BUL	EBT	CUT	SCS	BUL	EBT	CUT
MAN1									0.01	0	0	0				
MAN2									0	0	0.018	0.011				
MAN3									0	0	0.046	0.103				
• ⁸									0.01		0.032	0.057				
SD									0		0.020	0.065				
NFT1	o.v.27	0	0	0	0	0	0	0	0	0	0	0	0.014	0	0	0
NFT2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0:001	0
NFT3	0	0.009	0	0.084	0	0	0	0.031	0	0.004	0	0.024	0	0.004	0	0.027
avg	0.009	0.003		0.028				0.010		0.001		0.008	0.005	0.001	0.001	0.009
SD	0	0		0				0		0		0	0	0	0	
JUN					0	0	0	0	0	0	0	0	0	0	0	0
TAN1					0	0	0	0	0	0	0.002	0	0	0	0	0.003
TAN2	0	0	0.002	0	0	0	0.005	0	0	0	0.005	0	0	0	0.002	0
TAN3	0	0	0.010	0.014	0	0	0.009	0.013	0	0	0.008	0.012	0	0	0.014	0.018
avg			0.006	0.005			0.007	0.004			0.005	0.004			0.008	0.011
SD			0.006	0 ,			0.003	0			0.003	0			0.008	0.011
MFT1	0.044	0	0	0.001	0	0	0	0	0.002	0	0	0	0	0	0	0
Mm2	0.005	0	0	0.001	0	0	0	0.002	0	0	0	0	0	0	0	0
MFT3	0	0	0	0.001	0	0	0	0	0	0	0	0.002	0	0	0	0
avg	0.025			0.001				0.001	0.001			0.001				
SD	0.028			0				0	0			0				
WFT1	0.017	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFT2	0.003	0	0	0	0	0	0	0	0.002	0	0	0	0	0	0	0
WFT3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
avg	0.010								0.001							
SD	0.010								0							
CAB1	0.001	0	0.013	0	0	0	0.004	0	0.005	0	0	0	0	0	0	0
CAB2	0.001	0	0.004	0.006	0	0	0	0	0	0	0	0.002	0	0	0	0
avg	0.003		0.009	0.003			0.002		0.003			0.001				
SD	0.002		0.006	0			0		0			0				
SWK1													0.260	0	0	0
SWK2									0.005	0	0	0	0.008	0	0	0.002
SWK3									0	0	0	0.018	0	0	0	0.019
avg									0.003			0.009	0.134			0.011
SD									0			0	0.178			0.012
BIG									0	0	0	0.005				
UMT1									0	0	0	0	0.304	0	0	0
IJMTl.5													0	0	0	0
UMT2									0	0	0	0	0	0	0	0
avg													0.101			
SD													0			

[a]Sites: MAN = Manastash Creek, NFT = North Fork Teanaway River. JUN = Jungle Creek, TAN = Taneum Creek, MFT = Middle Fork Teanaway River. WFT = West Fork Teanaway River, CAB = Cabin Creek, SWK = Swauk Creek, BIG = Bii Creek and UMT = Umtanum Creek.

[b]SCS = juvenile spring chinook salmon, BUL = bull trout, EBT = eastern brook trout, and CUT = cutthroat trout . [b]SCS = juvenile spring chinook

Table 3. Biomass (g/m²) of juvenile spring chinook salmon and bull, eastern brook and cutthroat trout in each index site in tributaries of the Yakima River surveyed from 1990 to 1993. The average and standard deviation (SD) are also reported. Tributaries are arranged from high to low elevation (measured as the average elevation for the three index sites).

	1990				1991				1992				1993			
Site[a]	SCS[b]	BUL	EBT	CUT	SCS	BUL	EBT	CUT	SCS	BUL	EBT	CUT	SCS	BUL	EBT	CUT
MAN1									0.093	0	0	0				
MAN2									0	0	0.799	0.437				
MAN3									0	0	2.160	2.347				
avg									0.003		1.480	1392				
SD									0		0.962	1.351				
NFT1	0.287	0	0	0	0	0	0	0	0	0	0	0	0.092	0	0	0
NFT2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.018	0
NFT3	0	0	0	0	2.110		0	0.009	0	1.495	0	0	0	1.453		
avg	0.091	0.022		1.414				0.703		0.003		0.498	0.031		0.006	0.484
SD	0	0		0				0		0		0	0		0	0
JIM					0	0	0	0	0	0	0	0	0	0	0	0
TAN1					0	0	0	0	0	0	0.173	0	0	0	0	0.174
TAN2	0	0	0.010	0	0	0	0.499	0	0	0	0.664	0	0	0	0.039	0
TAN3	0	0	0.011	1.107	0	0	0.361	0.823	0	0	0.042	0.473	0	0	0.652	0.272
[illegible]			0.011	0.554			0.430	0.274			0.293	0.158				0.223
SD			0.001	0			0.098	0			0.328	0			0.433	0.069
MFT1	0.988	0	0	0	0	0	0	0	0.010	0	0	0	0	0	0	0
MFT2	0.064	0	0	0.113	0	0	0	0.01s	0.002	0	0	0	0	0	0	0
MFT3	0	0	0	0.048	0	0	0	0	0	0	0	0.047	0	0	0	0
'"I	0.526			0.08 1				0.005	0.006			0.016				
SD	0.653			0.016				0	0.006			0				
WFT1	0.236	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFT2	0.028	0	0	0	0	0	0	0.016	0.010	0	0	0	0	0	0	0
WFT3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
avg	0.132							0.015	0.003							
SD	0.147							0	0							
CAB1	0..036	0	0.650	0	0	0	0.588	0	0.017	0	0	0	0	0	0	0
CAB2	0.013	0	0.370	0.320	0	0	0	0	0	0	0	0.020	0	0	0	0
avg	0.025		0.510	0.016			0.294		0.009			0.010				
SD	0.016		0.198	0			0		0			0				
SWK1													1.617	0	0	0
SWK2									0.034	0	0	0	0.064	0	0	0.090
SWK3									0	0	0	0.646	0	0	0	0.440
avg									0.017			0.323	1.129			0.265
SD									0			0	0.691			0.247
BIG									0	0	0	0.094				
UMT1									0	0	0	0	2.710	0	0	0
UMT1.5													0	0	0	0
UMT2									0	0	0	0	0	0	0	0
avg													0.903			
SD													0			

[a]Index sites: MAN = Manastash Creek, NFT = North Fort Teanaway River. JUN = Jungle Creek. TAN = Taneum Creek, MFT = Middle Fork Teanaway River. WFT = West Fork Teanaway River. CAB = Cabin Creek, SWK = Swauk Creek, BIG = Bii Creek and UMT = Umtanum Creek.

[b]SCS = juvenile spring chinook salmon, BUL = bull trout, EBT = eastern brook trout, and CUT = cutthroat trout.

In 1993, rainbow trout were collected in 20 of 21 (95%) of the tributary index sites. These sites represented a wide array of habitat conditions (Appendix 4A). Rainbow trout densities varied between sites but there were no significant correlations between rainbow trout densities and any physical variable measured in 1993. However, in 1992, a strong positive correlation was observed between rainbow trout densities and habitat complexity (as measured by the standard deviation of thalweg depth). Also in 1990 and 1991, a strong positive correlation between rainbow trout densities and the amount of pool habitat in index sites existed (Table 4). These relationships may be an artifact due to repeated comparisons of dependent data; however, the percent error was low, with values ranging from 12 to 45% (Table 4).

Rainbow trout densities appeared to be loosely correlated with stream size, elevations, quantity of pool habitat, and habitat complexity ($P<0.10$) within years (Table 4). Elevation could be associated with fish distribution through its relationship to stream temperature. We could not confirm this relationship directly because temperature measurements were taken at non-standardized times during the summer. The typically large diel fluctuations in stream temperature (eg. high temperature in afternoon and low temperature in the morning) precluded effective analysis. In addition to time of day, stream temperature may also have been affected by solar radiation, which in turn was affected by the amount of cloud cover. To circumvent these problems, we examined the relationship between the mean maximum and mean minimum stream temperatures for the 10 middle elevation tributary index sites. A strong negative correlation was found between elevation and mean minimum stream temperature ($r=-.8641$, $P = 0.0057$, $df=8$). This suggests that rainbow trout density may be inversely related to temperature as mediated by elevation.

Few strong correlations existed between rainbow trout density and nine physical habitat variables recorded during the summer sampling period (Table 4). However, it is likely that rainbow trout density is more affected by physical habitat and environmental variability than by the presence of other salmonids. It appears that rainbow trout density was independent of other salmonids (Figure 5). In fact, rainbow trout density was found to be highest in sites that contained high densities of other salmonids. This is consistent with our failure to find significant correlations between rainbow trout densities and other salmonid densities in index sites during 1993.

Table 4. Table of correlation coefficients (r) between upper Yakima Basin rainbow trout densities (#/m^2), and nine physical variables measured at each tributary index site (N = 72) by year. Standard deviations (SD) of thalweg depth were recorded in 1992 and 1993 (N = 44).

Physical variables	1990	1991	1992	1993
Site elevation (m)	-0.03	-0.46*	-0.37	-0.11
Site area (m^2)	-0.03	0.05	-0.14	-0.43*
Mean site width (m)	-0.11	-0.04	-0.05	-0.41"
Thalweg depth (SD)	No data	No data	0.59**	0.16
Gradient	0.08	No data	-0.34	0.28
Discharge (m^3/sec.)	-0.23	0.18	-0.25	-0.41*
Total pool area (m^2)	-0.03	0.77**	0.34	-0.11
Number of pools	-0.04	0 . 2 8	-0.01	0.12
Maximum site depth (m)	-0.33	0.56**	0.25	-0.11
Percent error rate				
a = 0.01	N/A[a]	12	45	N/A[a]
a = 0.05	40	7	45	15

* P < 0.10; ** P < 0:05

[a]Not applicable because no significant relationships were found at this a level

Rainbow trout and total salmonid densities were higher in randomly selected sites than in index sites in two forks of the Teanaway River (Table 5). Rainbow trout densities in random sites of the North Fork of the Teanaway River were significantly higher than in the index sites in the same tributary (t = 3.724, P = 0.0098, df = 6). Rainbow trout densities were also significantly higher in random sites in the Middle Fork of the Teanaway River than in the index sites in the same tributary (t= 2.603, P = 0.0405, df = 6).

Rainbow trout densities were generally two times higher in random sites than in index sites. Because rainbow trout and total salmonid densities were correlated with site elevation, mean stream width, thalweg depth, discharge and percent pool area (Table 4), we compared the index sites to the random sites within tributaries to determine if the sites were physically similar. These comparisons showed that there were no significant differences between the index and random sites for any of the five physical variables measured.

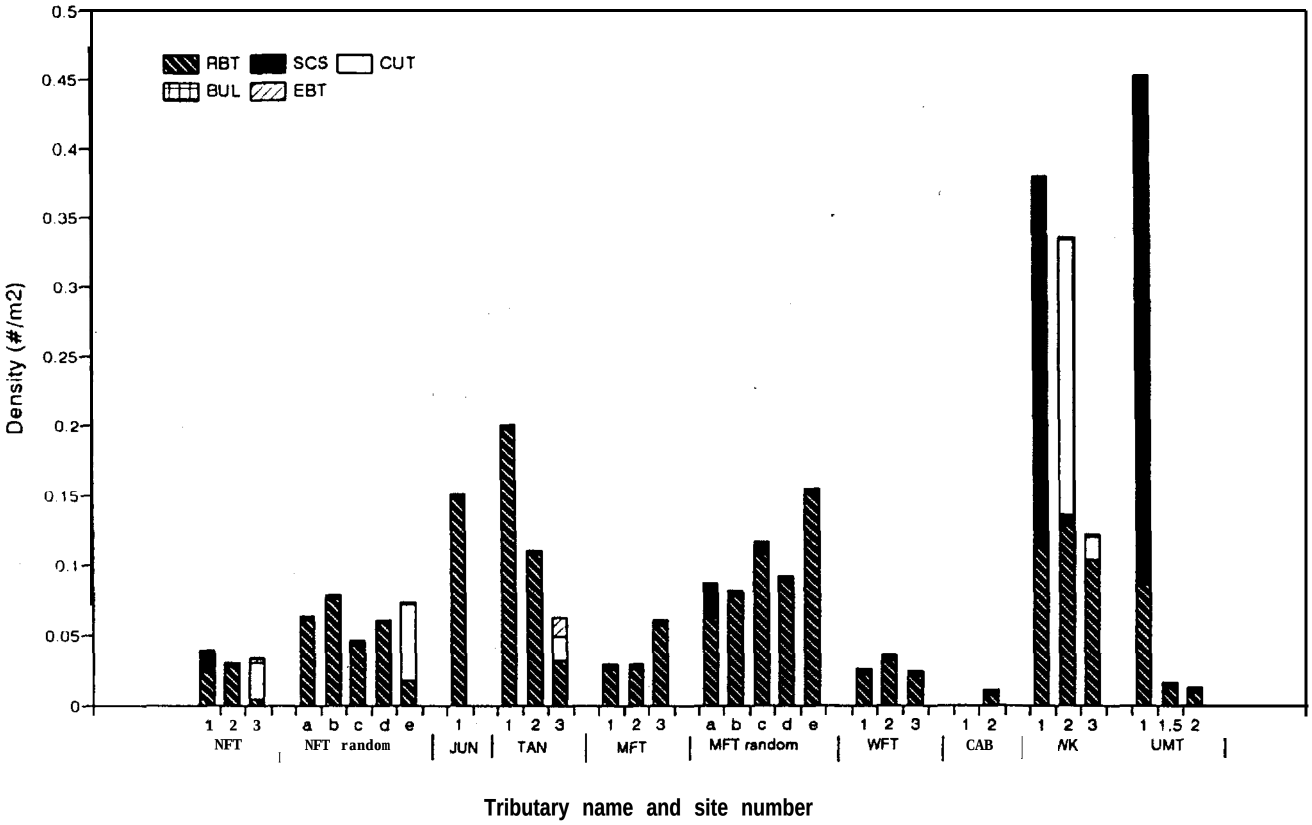

Figure 5. Rainbow trout (RBT), juvenile spring chinook salmon (SCS), cutthroat trout (CUT), bull trout (BUL), and eastern brook trout (EBT) densities (#/m^2) in index and random sites sampled in 1993.

Table 5. Rainbow trout and total **salmonid** densities in six index sites and 10 randomly selected sites in the Middle Fork and the North Fork of the Teanaway River, 1993. The random sites are listed adjacent to the index site that they were closest to.

Site type*		Rainbow trout density (#/m^2)		Total salmonid density (#/m^2)	
Index	Random	Index	Random	Index	Random
NFT 1	NFT A	0.0244	0.6310	0.3800	0.6590
	NFT B		0.0790		0.0850
NFT 2	NFT C	0.0106	0.0464	0.0330	0.0464
	NFT D		0.0608		0.0624
NFT 3	NFT E	0.0054	0.0190	0.0358	0.0782
avg		0.0201	0.1672	0.1496	0.1862
SD		0.0131	0.2602	0.1995	0.2647
MFT 1	MFT A	0.0301	0.0631	0.0301	0.0801
	MFT B		0.0816		0.0816
MFT 2	MFT C	0.0294	0.1018	0.3294	0.1115
	MFT D		0.0918		0.0918
MFT 3	MFT E	0.0611	0.1514	0.0611	0.1540
avg		0.0402	0.0979	0.0402	0.1045
SD		0.0181	0.0331	0.0181	0.0310

*NFT = North Fork of the Teanaway River; MFT = Middle Fork of the Teanaway River

Mainstem Yakima River

Trout density ($\#/km^2$) within mainstem Yakima River index sections varied among 1991, 1992 and 1993 (Figure 6). In 1993, trout density exhibited a normal distribution with respect to section location. In other words, density was low in sections at both elevational ends (LCYN and CELUM) of the river and high in the middle section (EBURG). In 1992 however, trout density was distributed opposite to that in 1993; density was low in the middle sections (EBURG and THORP), and high at sections at either end (LCYN and CELUM). In 1991, trout density appeared to be distributed evenly throughout the four sections surveyed. In 1993 the highest trout density was observed in the EBURG section. Moreover, the estimated density for the EBURG section in 1993 was 150% greater than the next highest population estimate that year. Not only did the spatial distribution of trout differ between 1992 and 1993, but the trout population estimate and biomass of trout of the five sites pooled was also higher in 1993 than in 1992.

The trout population estimate increased from 7,101 in 1992, to 8,939 in 1993, and the estimated biomass of trout increased from 1,357 kg in 1992, to 1,827 kg in 1993 (Table 6). Because the population estimate for the CELUM section was not valid in 1991, the total combined estimated number of trout in this section was calculated to be the average of the 1992 and 1993 estimates (2,200 fish; this number is not intended to be used as an actual estimate).

Using this estimated number (2,200), it appears that the trout abundance decreased from 1991 to 1992, and then increased from 1992 to 1993. In general, the increase from 1992 to 1993 can be attributed to an increase in the number of trout in upper sections (EBURG, THORP and CELUM sections).

The estimated biomass of trout per kilometer varied between sections and years, but in all years was greatest in LCYN (Figure 7). This can be attributed to the larger size and the high density of trout in the LCYN section. Although trout biomass was variable, it markedly increased in the three higher sections (EBURG, THORP, and CELUM) in 1993, compared to 1991 and 1992. This may be attributed to an increased number of fish in these sections, rather than an increase in the size of the fish.

Spatial and temporal variation in mean fork length of trout captured in the five mainstem sites was minimal. In general, the mean fork length was longest for trout in the LCYN section, followed by UCYN, and the trout with the shortest mean fork length were captured in the EBURG or THORP sections (Figure 8). Large trout (>250 mm) were captured in the canyon sections of the Yakima River in greater percentages than other sections sampled in most years (Figure 9).

Trout densities ($\#/m^2$) in the mainstem sections in 1993 was distributed similarly to the number of trout per kilometer (refer to Figure 6) even though site area was different between sections (Table 7). In 1993, the biomass of trout (kg) per kilometer in the mainstem (Figure 7) was not distributed the same as the biomass of trout (g) per square meter (Table 7). The difference in biomass distributions reported in Figure 7 and Table 7, is that Table 7 shows the biomass of trout per square meter to be highest in CELUM, while Figure 7 shows that it is highest in LCYN. This discrepancy is not due to fish size but rather to site area; the CELUM section had a much smaller area than the LCYN section, resulting in a higher biomass (g) per square meter than biomass (kg) per kilometer.

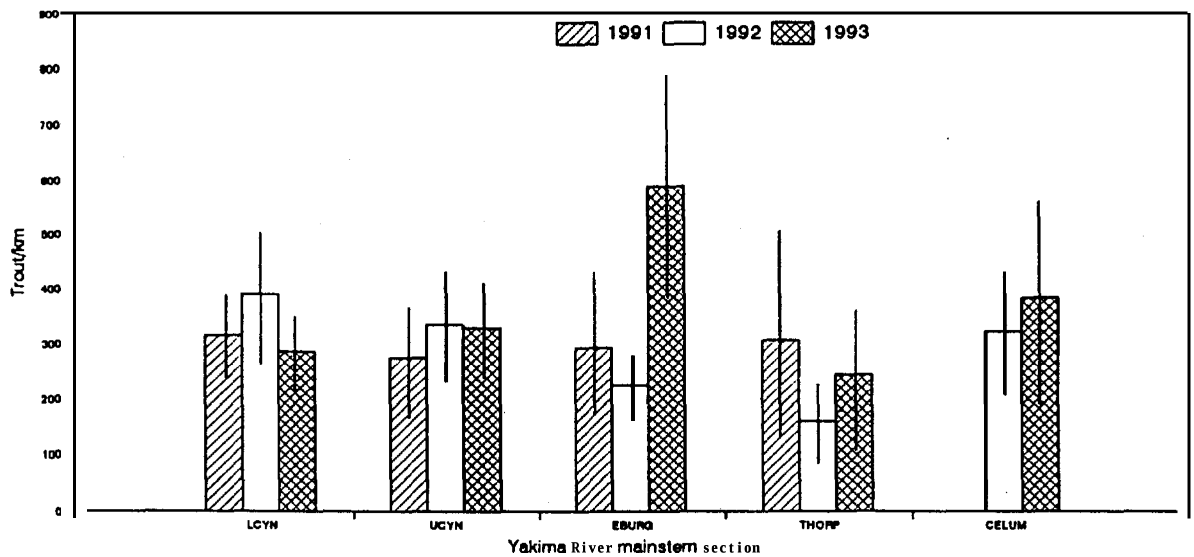

Figure 6. Trout population estimates (#/km) in the five sections of the mainstem Yakima River sampled in 1991, 1992 and 1993. Vertical lines are 95% confidence intervals around the population estimate.

Table 6. Mainstem Yakima River site length (km), total site trout population estimate and biomass, and trout density and biomass per kilometer in each site from 1991 through 1993.

		1991				1992				1993			
Section	Site Length	Number	Biomass	#/km	kg/km	Number	Biomass	I/km	kg/km	Number	Biomass	#/!&q/km	
.1 (LCYN)	4.5	1,414	355	314	79	1,754	527	390	117	1,280	475	284106	
(UCY(EBURG)	4.5 4.0	1,232 1,167	238 191	274 292	53 40	1,503	236	334	52	1,480	304	329	68
						894	124	224	31	2,349	315	587	79
4 (THORP)	5.9	1,774	305	306	53	927	132	160	23	1,413	259	244	45
5 (CELUM)	6.3	(2,200)'				2,023	338	323	54	2,417	474	384	75
TOTAL		(7,807)				7.101	1,357			8,939	1,827		

• Estimated number because 1991 estimate was not valid. Figure in parentheses is the average of 1992 and 1993 trout population estimates for the CELUM section.

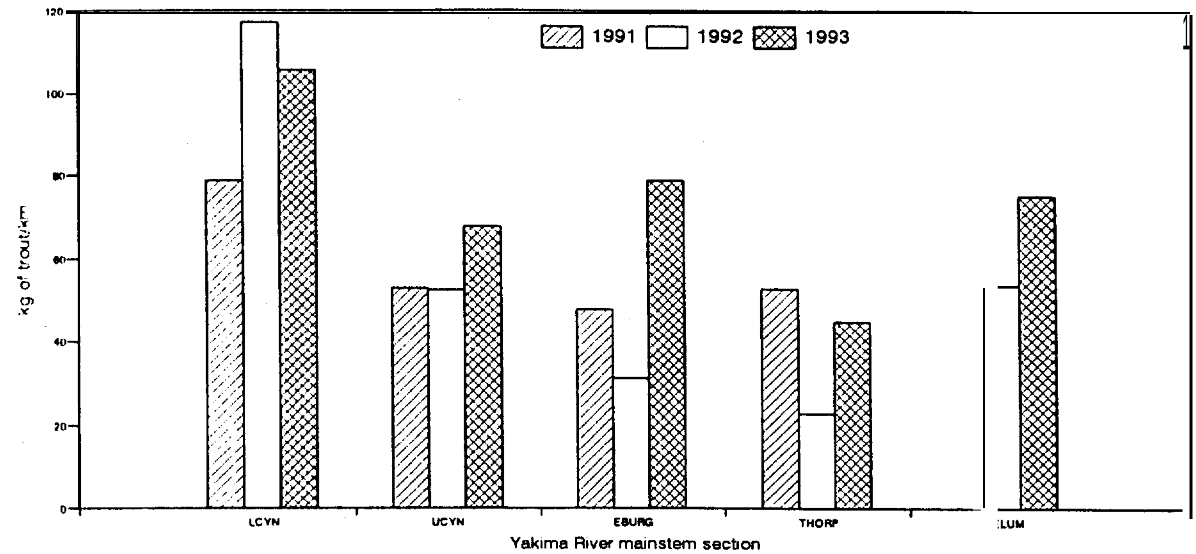

Figure 7. Estimates of trout biomass (kg/km) in the five sections of the mainstem Yakima River sampled in 1991, 1992 and 1993.

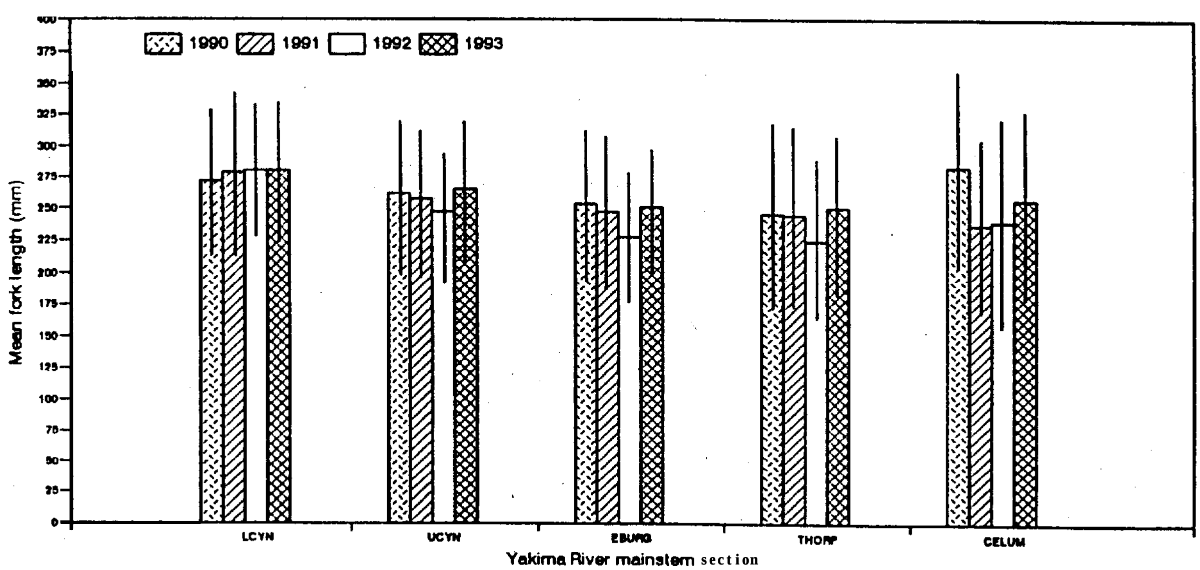

Figure 8. Mean fork length (mm) of rainbow trout captured in the five sections of the mainstem Yakima River from 1990 through 1993. Vertical lines represent 1 SD.

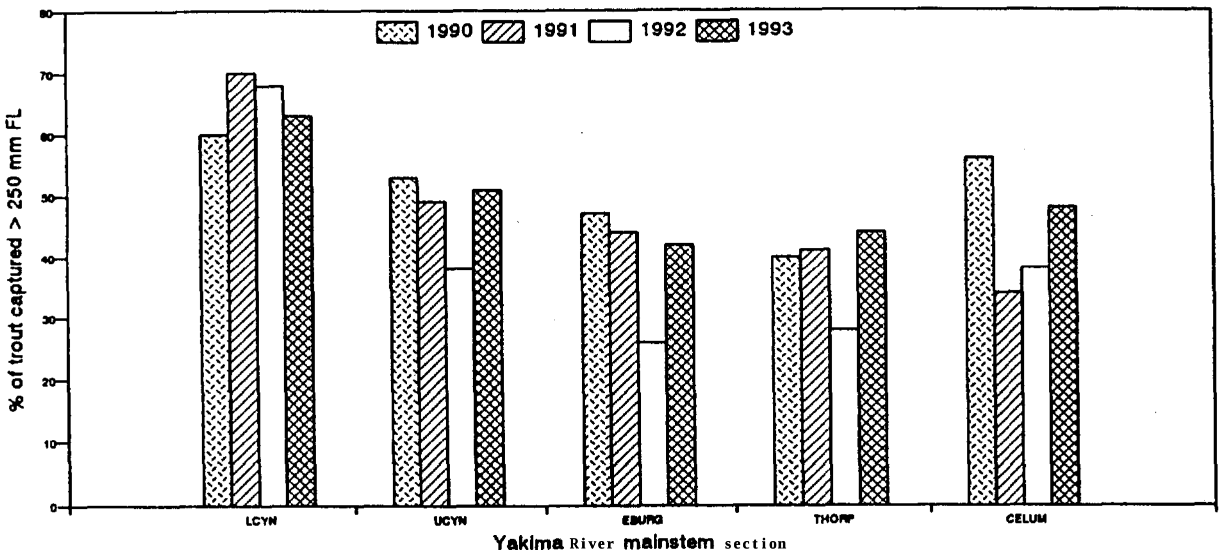

Figure 9. Percent of trout captured whose fork length was greater than 250 mm in five sections of the mainstem Yakima River from 1990 through 1993.

Table 7. Rainbow trout density (#/m^2) and biomass (g/m^2) during 1993 in five mainstem Yakima River index sites.

Site	Site area (m^2)	Density (#/m^2)	Biomass (g/m^2)
1 (LCYN)	243,508	0.005	1.95
2 (UCYN)	202,854	0.007	1.50
3 (EBURG)	174,195	0.013	1.81
4 (THORP)	260,437'	0.005	1.00
5 (CELUM)	213,116	0.018	2.22

The percent composition of rainbow and eastern brook trout in the mainstem Yakima River varied little among years. There was higher interannual variation however, for cutthroat trout (Table 8). As mentioned previously in this report, a hybrid trout category was used in 1990, 1991 and 1992. This category was not included in Table 8 because of potential classification bias among observers and because all hybrids were classified as rainbow trout beginning in 1993. In general, the percentages of cutthroat and eastern brook trout increased from low elevation to high elevation (Table 8).

Areal densities (#/m^2) of rainbow trout in sections of the mainstem Yakima River (0.005 to 0.018 fish/m^2) were lower than those in index sites of seven tributaries (0.00 to 0.210 fish/m^2) in 1993 ($T = -2.271$, $P = 0.0442$, $df = 11$). However, lineal densities were considerably higher in the mainstem than in seven tributaries sampled.

Table 8. Trout species percent composition for each mainstem Yakima River site surveyed from 1991 through 1993. Totals for some sites do not equal 100% because hybrid trout were not included.

	Percent composition											
	1991				1992				1993			
Site	RBT	CUT	BUL	EBT	RBT	CUT	BUL	EBT	RBT	CUT	BUL	EBT
1 (LCYN)	99.6	0.0	0.0	0.0	98.6	0.7	0.0	0.0	99.5	0.5	0.0	0.0
2 (UCYN)	100	0.0	0.0	0.0	99.4	0.3	0.0	0.0	99.4	0.6	0.0	0.0
3 (EBURG)	Y9.6	0.0	0.0	0.0	99.1	0.0	0.0	0.0	99.7	0.3	0.0	0.0
4 (THORP)	91.5	5.6	0.0	0.0	98.1	1.4	0.0	0.0	97.9	2.1	0.0	0.0
5 (CELUM)	95.4	0.9	0.0	0.3	97.7	1.4	0.2	0.5	94.0	5.4	0.0	0.6

RBT = rainbow trout, CUT = cutthroat trout, BUL = bull trout, EBT = eastern brook trout

Discussion

The spatial variability we observed in rainbow trout density appears to be partially explained by five physical variables: (1) standard deviation of thalweg depth, (2) elevation, (3) stream width, (4) percentage pool area, and (5) discharge. Although the percent error we calculated indicated that these data should be intertpreted with caution we believe that the above relationships have biological importance and are not simply an artifact of the analysis. Other studies have also demonstrated relationships between trout densities and physical habitat parameters. For example, cover, substrate size, wetted width, average depth, and stream order were significantly correlated with trout densities in tributaries of the Flathead River in Montana (Graham 1981). Nelson et al. (1992) found that in the North Fork Humbolt River drainage, trout presence in a site was negatively correlated with elevation and the occurrence of pool habitat, and positively correlated with stream width and stream flow. Scarnecchia and Bergersen (1987) found that in small streams in Colorado total trout number and biomass were negatively correlated with elevation. Rahel and Hubert (1991) found that elevation was negatively correlated with site scores (trout abundance). Although there is not complete agreement in the literature, trout density appears to be negatively correlated with elevation, stream width and stream order, and positively correlated with pool habitat. We believe that a contributing factor to inconsistent findings is that identifying and measuring habitat units is subjective and rarely reproducible. This problem is especially true for pool habitat. Nelson et al. (1992) reported that the accuracy of identifying pool habitat was poor and that classification lacked consistency. For this reason we used a hierarchical method of identifying habitat unit type (Hawkins et al. 1993). Continuation of this method should improve our accuracy and precision over time and result in more reproducible habitat measures.

For individual tributaries, variability in rainbow trout densities were low through time, except for Umtanum, Taneum and Jungle creeks. Some of the temporal variability in Umtanum Creek may be explained by a 100 year flood on June 4, 1993. Based on the presence of mud and debris the following day on a railroad bridge abutment in Umtanum Creek, we estimated that the discharge on the day of this event was between about 13 cms and 26 cms which was nearly 200 times the discharge the day before the flood (0.08 m^3/s on June 3). As a result of this flood we believe that juvenile, and to a lesser extent, adult rainbow trout, were displaced out of the stream or experienced high mortality. On June 6, several dead juvenile spring chinook salmon and rainbow trout, as well as other non-salmonid species were seen observed in the streambed as well as along the stream banks. Elwood and Waters (1969) found that adult brook trout were nearly eliminated following a severe flood, while Pearsons et al. (1992) stated that young-of-year fishes may be particularly vulnerable to floods because of their poor swimming ability and small size.

Although we observed more small fish carcasses than large ones we feel that the flood probably reduced the number of adult trout as well. As a result, the density of rainbow trout in Umtanum Creek in 1993 was substantially lower than in 1992. Temporal variability in Taneum and Jungle creeks may also be attributed to stochastic events, although we can present no data to support this speculation at this time. In all years, Cabin Creek had the lowest density of rainbow trout among all tributary index sites sampled. Cabin Creek experiences frequent and severe fluctuations in discharge that not only temporarily reduce fish numbers, but may also reduce instream habitat complexity by removing instream structure such as woody debris and boulders (Elwood and Waters 1969; Pearsons et al. 1992).

The tributary sites we selected at random to test for the representativeness of our index sites had significantly higher rainbow trout densities than the index sites in the North and Middle forks of the Teanaway River. One explanation for this is the occurrence of repeated electrofishing in index sites. Our index sites have been sampled annually since 1990. McMichael (1993) found that electrofishing had a deleterious effect on rainbow trout in a hatchery environment, but that short-term mortality was negligible. Impacts of electrofishing on trout survival and growth have also been documented by Gatz et al. (1986) and Sharber (1988). A second possible reason for lower densities in index sites may be that the index sites we used were not good estimators of the rainbow trout population (due to their selection by non-random methods). Platts (1983) stated that bias often results from a lack of randomness in the selection of sample sites, and therefore, can influence the accuracy of the data generated.

Abiotic factors such as temperature and stream size appear to be more influential in affecting rainbow trout densities on a large scale than the influence of salmonid species interactions. Salmonid species interactions may be particularly important in influencing trout densities at a local scale. Although Nelson et al. (1992) found that trout densities at specific stream sites may be related to physical variables, they suggested that competition may influence trout densities at specific sites.

Comparisons of rainbow trout areal densities between the mainstem and tributaries indicated that density was significantly higher in tributaries of the upper Yakima River than in the mainstem river. This may be due to the amount of living space available associated with the requirements at different life stages. Much literature exists, as reported above, that describes trout density and its positive relationship to lower order and higher elevation streams. Platts (1979) found that the amount of fish habitat (water space) increased as stream order increased in an Idaho River drainage. Therefore, fish biomass and total numbers tend to increase with increasing stream order. The findings of Platts (1979) may seem contradictory to our findings, but, we also found numerically more trout in the mainstem than in the tributaries. However, when the numbers of trout were standardized by site area, density was higher in tributaries than in the mainstem river.

Trout abundance increased and their distribution changed in the mainstem of the Yakima River from 1991 to 1993. The increase in their numbers might be attributed to greater numbers of small trout in the EBURG and THORP sections and an overall increase in the CELUM section. This overall increase may be explained by one or more of several factors: good spawning success or an increased number of spawners in 1990 and 1991 (possibly as a result of catch and release regulations that were implemented in 1990), a shift in spawning towards higher sections, population recovery after the basin-wide flood in November 1990, or lower flows in upper sections which have improved capture efficiencies of smaller fish. Discharge was lower in the upper sections in 1993 than in 1991 or 1992, however, capture efficiency did not mirror discharge. In 1993, capture efficiency of smaller fish (<250 mm) was lower than in 1991 or 1992, which leads us to believe that recruitment through good spawning success or other factors must be the reason for the increase in trout numbers in the upper sections of the Yakima River in 1993.

Based on three annual estimates it appears that the abundance of rainbow trout (> age 0) in the upper Yakima River is quite stable. Population stability may imply that the system is at carrying capacity and that there is some factor, such as water velocity, that limits production. Chapman (1966), Edmundson et al. (1968) and others have demonstrated that the carrying capacity of a stream is greatly influenced by water velocity that is, in turn, a function of streamflow. Seegrist and Gard (1972) report that in Sagehen Creek survival for spring spawning rainbow trout was low in years with spring floods. In the upper Yakima River, we speculate that one limiting factor to rainbow trout production is flow fluctuations during the first month of their life followed by high summer discharge that continues until the second week of September. Our hypothesis is that trout experience high mortality between the time of emergence and the summer rearing period. This speculation may be substantiated by the large number of redds observed in the mainstem river (refer to chapter 2) and the scarcity of young-of-year captured during night electrofishing surveys, miscellaneous day electrofishing surveys, or observed during snorkeling surveys. We would expect, even if survival from egg to fry was less than 0.01, that there would be substantially greater numbers of young-of-year captured or observed in the mainstem river. This preliminary speculation about young-of-year survival should be viewed cautiously, further research needs to be conducted prior to any statements about carrying capacity or limiting factors can be adequately addressed or discussed.

A comparison of the mainstem Yakima River trout densities to trout densities reported in the published literature indicated that rainbow trout densities in the mainstem Yakima River are lower than other large rivers in the western United States (Table 9). Rainbow trout densities shown in Table 8 for the mainstem Yakima River reflect the mean density for the five sections we have surveyed. Values in the table for the tributaries are the mean densities for all tributaries sampled in 1992 and 1993. A further analysis of these kinds of comparisons will be provided in a future report or publication.

Although the densities of rainbow trout in the mainstem Yakima River are lower than rainbow trout densities reported for other rivers this difference may be explained by the relatively low productivity in the Yakima River. Productivity is directly related to total dissolved solids (TDS) (Rinella et al. 1992). TDS is related to the concentration of major ions dissolved in water, and therefore is also directly related to conductivity. Conductivity readings in the upper Yakima River ranged from 60 to 180 us/cm. In contrast, conductivity in the Montana rivers reported in Table 9 were about 350 us/cm (Geoff McMichael Washington Department of Fish Wildlife, personal communication).

Lower trout densities in the Yakima River than in other stream shown in Table 9 may also be attributed to artificial flow fluctuations which result in high summer discharge and low winter discharge. Underwood and Bennett (1994) found that in the Spokane River, Idaho, year-class strength of rainbow trout was associated with constant flows between 1 April and 25 June. In the Yakima River Subbasin Plan (NPPC 1989) it is noted that there are rapid daily flow fluctuations below storage reservoirs and that these have adverse impacts on spawning and rearing habitat. Vincent (1987a) found that high winter flows resulted in increased biomass of trout age 2 and older in the Gallatin River, Montana. After four years of research, Vincent (1987a) concluded that two major physical factors that can alter trout biomass in Montana streams were variations in habitat and volume of flow. We speculate that in the Yakima River, high summer discharge results in loss of velocity refugia for YOY rainbow trout which probably limits survival from fry to juvenile.

Table 9. Comparison of rainbow trout rearing densities (> age 0) in the upper Yakima River and in other rivers of similar size in the western United States.

Year	Enumeration method	River	Density	Reference
Mainstem				
1986	Mark-Recapture	E Gallatin River, MT	1,574/km	Vincent, E. et al. 1987
1985	Mark-Recapture	E Gallatin River, MT	1,454/km	Vincent, E. et al. 1987
1984	Mark-Recapture	E Gallatin River, MT	2,376/km	Vincent, E. et al. 1987
1987	Mark-Recapture	Big Horn River, MT	593/km	Fredenberg, W. 1988
7992	Mark-Recapture	Madison River, MT	977/km	Vincent, E. 1984
1983	Mark-Recapture	Madison River, MT	768/km	Vincent, E. 1984
1906	Mark-Recapture	Madison River, MT	962/km	Vincent, E. 1987
1986	Mark-Recapture	Madison River, MT	734/km	Vincent, E. 1987
1970'3	Mark-Recapture	Deschutes River, OR	1,098/km	Schroeder, 1989
1980's	Mark-Recapture	Deschutes River, OR	1,011/km	Schroeder, K. 1989
1991	Mark-Recapture	Yakima River, WA	310/km	McMichael, G. et al. 1992
1992	Mark-Recapture	Yakima River, WA	292/km	Pearsons, T. et al. 1993
1993	Mark-Recapture	Yakima River, WA	351/km	This report
Tributaries				
1985	Snorkeling	Tributaries of the John Day River, OR	6 to 24/100m^2 [a]	Li, H.W. et al., 1985
1991	Multiple removal	Tributaries of the Lower Snake River, WA	3.5 to 14.3/100m^2 [a]	Martin, S. et al. 1992
1992	Multiple removal	Yakima River Tributaries, WA	7.36/100m^2 [a]	This report
1993	Multiple removal	Yakima River Tributaries, WA	7.05/100m^2 [a]	This report

[a] mean value of all tributaries sampled. Values may reflect composite of rainbow trout with steelhead present.

Past stocking with hatchery-reared fish in the Yakima River may also have some influence on relativelylow trout densities. Vincent (1987b) found that in the Madison River and O'dell Creek, Montana, wild rainbow trout densities increased 800% and their biomass increased 1000% following the cessation of stocking with catchable-sized hatchery rainbow trout into the river. The release of hatchery catchable-size rainbow trout has occurred from the early 1900's through the late 1980's in the upper Yakima River, which may also be a causative factor for the low rainbow trout density in the Yakima River. If rainbow trout in the Yakima River respond as did the population in the Madison River (Vincent 1987b), then an increase in the numbers and biomass of rainbow trout may occur in future years. However, the response in the Yakima River, if any, may be less dramatic because of the low productivity of the Yakima River and/or flow fluctuations that may limit the rainbow trout abundance or carrying capacity during the rearing life history phases.

Conclusion

The production of salmonid fishes in streams shows great natural variability in time and space (Hall and Knight 1982) and it is agreed that biotic dynamics are strongly linked to variation in abiotic factors (Power et al. 1988). We have described rainbow trout density and its correlation to five physical habitat characteristics, and have suggested possible reasons why high spatial variability in rainbow trout density exists. Although we may suggest reasons for high spatial variability, at this time we can not place quantifiable bounds on the natural temporal variation of rainbow trout density. Temporal variation must be understood prior to any disturbance so that the effects of the disturbance on data interpretations are minimized. Trout populations are not necessarily stable entities and should not be regarded as such (Platts and Nelson 1988).

In the future, sites will be partitioned into low, moderate, or high variance sites based on the interannual coefficient of variation of estimated abundance. Sites that have low variability may be very useful in detecting impacts on trout because abundance may be at or near equilibrium making any impacts easier to detect. Sites that have higher variability may not be as useful for monitoring population abundance status due to the difficulty in detecting impacts. However, if trout abundance or biomass can be predicted by environmental or biotic variables, then sites prone to higher variability may also be useful as monitoring sites.

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APPENDIX 4A.

Tributary index site physical habitat data. All measurements are in meters unless specified otherwise. Blank cells indicate that'data is not available.

SITE	YEAR	SITE ELEVATION	SITE AREA	MEAN WIDTH	GRADIENT (%)	THALWEG DEPTH (S.D.)	POOL % OF AREA	POOL VOLUME	FOOL AREA	# OF POOLS	MAX. SITE DEPTH	DISCHARGE (c.m.s.)
CAB1	1993	719	682	6.8	2.0	20.18	16.7	38.76	114.0	1	0.80	0.71
CA02	1993	766	650	6.5	3.0	8.07	0.0	0.00	0.0	0	0.66	0.67
JUN1	1993	805	266	2.7	3.0	7.30	9.8	6.76	26.0	1	0.35	0.00
MFT1	1993	713	597	6.5	1.6	18.34	36.1	84.14	215.7	1	0.78	0.15
MFT2	1993	762	570	5.5	1.2	5.31	0.0	0.00	0.0	0	0.36	0.19
MFT3	1993	041	720	7.3	24	9.76	0.0	0.00	0.0	0	0.08	0.21
NFT1	1993	713	738	74	2.5	11.96	0.0	0.00	0.0	0	0.83	0.48
NFT2	1993	780	017	0.2	21	14.51	0.0	0.00	0.0	0	0.74	0.65
NFT3	1993	1103	921	8.1	3.0	8.26	0.0	0.00	0.0	0	0.74	0.26
WFT1	1993	713	932	9.3	2.0	7.64	40.8	68.39	380.4	2	0.68	0.22
WFT2	1993	750	628	5.3	2.0	10.84	30.1	67.00	188.9	2	0.66	0.14
WFT3	1993	799	850	8.2	2.5	8.04	0.0	0.00	0.0	0	0.45	0.12
SWK1	1993	579	362	3.9	2.5	13.54	27.4	32.23	104.8	2	0.72	0.00
SWK2	1993	732	512	51	2.6	9.18	6.8	20.32	35.0	1	0.64	0.08
SWK3	1993	902	371	3.5	2.9	7.74	6.3	7.74	23.5	1	0.64	0.04
TAN1	1993	622	597	6.0	2.4	25.74	0.0	0.00	0.0	0	0.97	0.23
TAN2	1993	611	508	5.1	2.0	25.01	42.9	143.68	218.3	3	0.76	0.22
TAN3	1993	914	442	3.a	1.0	12.47	10.5	17.67	46.3	2	0.85	0.11
UMT1	1993	469	197	1.8	3.1	9.48	5.3	207	10.4	1	0.50	0.01
UMT1 5	1993	538	317	3.2	2.1	11.21	13.1	12.90	41.6	1	0.68	0.01
UMT2	1993	622	245	2.5	2.0	28.73	12.4	80.90	30.3	3	1.40	0.01
CAB1	1992	719	632	5.8	2.0		4.0		28.7	1	0.70	0.11
CAB2	1992	768	516	5.2		7.89	13.3		75.5	2	0.60	0.29
JUN1	1992	805	191	4.3		8.65	28.5		54.4	5	0.47	0.00
MFT1	1992	713	607	5.6	1.3	13.22	48.0		292.7	4	0.55	0.03
MFT2	1.992	762	589	5.4	1.5	6.40	0.0		0.0	0	0.41	0.09
MFT3	1992	841	552	5.4	1.0		5.0		27.3	1	0.59	0.07
NFT1	1992	713	878	a.6		11.97	28.0		232.3	1	0.79	0.34
NFT2	1992	760	630	6.1	0.5	9.07	3.0		37.7	1	0.61	0.25
NFT3	1992	1103	466	4.8	2.0	7.45	16.0		74.1	2	0.73	0.15
WFT1	1992	713	551	5.5		3.60	4.0		23.8	1	0.48	0.03
WFT2	1992	750	586	4.7			15.0		89.8	2	0.51	0.04
WFT3	1992	799	597	6.3	3.5		21.0		124.8	4	0.78	0.05
SWK2	1992	732	207	2.1	0.5	12.55	11.0		36.4	2	0.70	0.02
SWK3	1992	902	271	2.6	1.0	8.25	12.0		31.9	2	0.49	0.01
TAN1	1992	622	619	6.8	1.3	34.96	61.0		379.4	3	0.98	0.11
TAN2	1992	a11	431	4.3	0.5	23.m	10.0		42.9	1	1.00	0.10
TAN3	1992	914	425	4.0	0.5	10.48	10.0		43.4	3	0.72	0.08
UMT1	1992	469	208	2.1	1.0	10.13	20.0		42.2	4	0.38	0.01
UMT2	1992	622	192	1.9	1.0	15.90	33.0		63.1	3	0.91	0.01
MAN1	1992	518	313	3.2	1.0	17.70	22.0		88.5	4	0.93	0.07
MAN2	1992	988	270	46		8.17	0.0		0.0	0	0.58	0.19
MAN3	1992	1341	369	3.7	1.5	9.45	21.0		78.8	3	0.60	0.00
BIG1	1992	677	393	3.4			28.0		110.4	2	1.00	0.02

APPENDIX 4A. Continued

SITE	YEAR	SITE ELEVATION	SITE AREA	MEAN WIDTH	GRADIENT (%)	THALWEG DEPTH (S.D.)	POOL % OF AREA	POOL VOLUME	POOL AREA	# OF POOLS	MAX. SITE DEPTH	DISCHARGE (c.m.s.)
CA01	1991	719	664	5.3			14.0		96.4	2	0.64	0.16
CAB2	1991	768	499	5.1			0.0		0.0	0	0.54	0.16
JUN1	1991	805	247	3.8			14.0		33.9	2	0.53	0.01
MFT1	1991	713	576	5.6			10.0		109.4	3	0.68	0.08
MFT2	1991	762	598	5.1			5.0		26.3	1	0.51	0.09
MFT3	1991	841	655	5.9			16.0		106.4	5	0.68	0.08
NFT1	1991	713	863	7.0			0.0		0.0	0	0.80	0.64
NFT2	1991	780	726	6.0			0.0		0.0	0	0.76	0.42
NFT3	1991	1103	491	4.9			7.0		32.3	1	0.70	0.20
WFT1	1991	713	552	7.0	0.5		14.0		86.6	1	0.53	0.15
WFT2	1991	750	699	5.2			16.0		10'9.1	1	0.69	0.14
W m	1991	799	a43	7.7			18.0		148.0	6	0.83	0.14
TAN1	1991	622	600	5.4	2.5		42.0		251.7	3	1.00	0.32
TAN2	1991	611	561	4.2			13.0		71.7	2	0.48	0.39
TAN3	1991	014	521	4.8	2.0		4.0		23.2	1	0.86	0.75
CAB1	1990	719	471	5.3	2.0		21.0		97.1	2	1.45	0.12
CAB2	1990	768	679	6.4			10.0		62.7	4	0.60	0.15
MFT1	1990	713	266	5.4	0.8		51.0		286.8	2	0.82	0.06
MFT2	1990	762	683	5.0	2 0		12.0		84.1	2	0.52	0.08
MFT3	1990	841	627	5.6	2 5		28.0		175.0	4	0.60	0.09
NFT1	1990	713	1164	9.5	2.0		15.0		170.3	2	1.00	0.55
NFT2	1990	780	766	7.6	1.0		5.0		41.0	1	0.50	0.28
NFT3	1990	1103	519	4.8			23.0		121.2	4	a64	0.11
WFT1	1990	713	678	6.1	1.0		5.0		33.3	1	0.69	0.07
WFT2	1990	750	781	6.6	1.0		21.0		166.4	1	0.93	0.08
W m	1990	799	631	5.8	1.8		10.0		64.5	4	0.66	0.10
TAN2	1990	811	608	8.1	3.0		20.0		123.3	3	0.55	0.50
TAN3	1990	014	568	5.6	2 0		6.0		35.3	2	0.67	0.30

Chapter 5

Age and growth of rainbow trout in the upper Yakima River basin

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Abstract

The growth and size of fish in different geographic locales may be influenced by ecological and genetic factors. We attempted to determine some of the factors that are related to rainbow trout growth and length in 12 tributaries and seven sections of the mainstem of the upper Yakima River. Length-at-age of fish was determined from rainbow trout scales using the Dahl-Lea back calculation method. A considerable amount of variation in length-at-age existed between sites in the upper Yakima River basin. Preliminary results suggested that rainbow trout length-at-age is related to both ecological and genetic factors. The relative position of principal component scores of length-at-age data corresponded closely to the genetic stock structure dendogram of rainbow trout in the upper Yakima River basin. Length-at-age was negatively correlated with elevation. Furthermore, the length-at-age of trout in the tributaries was significantly less than in the mainstem of the Yakima River. In addition we characterized the rainbow trout spawning populations in these same sites by determining the spawners' age and the minimum length at which they matured sexually. Most trout spawning in tributaries were age 1+ and 2+, whereas in the mainstem river, most spawning trout were age 2+ and 3+. The minimum size of sexually mature rainbow trout was negatively correlated with elevation. We were unable to confirm repeat spawning based on scale analysis. During their first year of life, growth of rainbow trout in the mainstem of the Yakima River appeared to be low compared to the growth of rainbow trout in other large rivers of the Northwest. Slow first year growth supports the hypothesis that the young of the year life stage is the one limiting rainbow trout production in the mainstem of the Yakima River.

Introduction

The growth rate and length-at-age of rainbow trout *Oncorhynchus mykiss* might be affected by ecological, both biotic and abiotic, and genetic factors. For instance, abiotic factors influencing fish growth include a body of water's elevation and size (McAfee 1966; Purkett 1958), and water temperature (Bridge 1943; Smith and Griffith 1994). Biotic factors that might influence growth include food availability (Bjornn 1966), fish density (Percival 1963; Buss 1960), the presence of predators (Dill 1983; Milinski 1986), nutrient input, and interspecific competition and social dominance (Li and Brocksen 1977). Within species, growth may also have a genetic influence (Gjerde 1986; Gall and Huang 1988), and patterns in growth may be reflected within and among stocks (Ricker 1972; Taylor 1991). In addition, genetic factors may interact with environmental factors to influence growth (McKay et al. 1984). Models have been developed that predict that fish using optimal rearing habitat will have higher growth rates (Fausch 1984; Newman 1993). Furthermore, fish that rear in different environments may mature sexually at different sizes and ages.

Many rainbow trout appear to mature at age 2 and 150 mm length, however this can vary substantially depending on gender and the environment (Scott and Crossman 1973). For instance, male rainbow trout in Sagehen Creek mature at a younger age and at a smaller size than females (Erman 1976). In a British Columbia stream male rainbow trout mature one year earlier than females, and survival is often low following spawning (Scott and Crossman 1973).

Prior to impact assessment studies in the Yakima River basin, an understanding of factors related to growth must be understood. Underwood (1994) reports that any univariate measurement, such as fish growth, can be analyzed and used to detect cause and effect relationships. The primary objective of this chapter is to explain relationships between rainbow trout growth, and ecological and genetic factors in the upper Yakima basin.

Data used in this chapter have been collected from 1990 through 1993. Earlier treatments of the topic are presented in Hindman et al. 1991, McMichael et al. 1992, and Pearsons et al. 1993. Considering that back-calculation of length-at-age was performed on scales collected since 1990, the ages in the data set may be interpreted back to 1987 for some streams. This chapter is a precursor to a full treatment of the data and the results presented are based only on limited statistical analyses of the data. Results and conclusions are subject to revision.

Methods

The ages of rainbow trout in the Yakima basin were estimated using scale analysis. Scales were collected from rainbow trout in the spring, summer and fall from 1990 to 1993 (Hindman et al. 1991; McMichael et al. 1992; Pearsons et al. 1993). Sampling was done selectively in order to represent the full range of rainbow trout lengths and to include two different seasons. As a result of selective sampling methods, the data can not be used to estimate the age composition of the general population (Pearsons et al. 1992). It can, however, be used to describe the age and growth of individual fish which can then be used to characterize the population at specific locations and seasons.

Age was analyzed separately for fish during the spawning and rearing periods. Spawners were defined to be fish that were sexually mature and had spawned or were going to spawn that year. Spent females in general had fungus on an eroded caudal fin and a distended vent. Rearing trout represented a cross section of fish lengths within each study section and were collected in the spring, summer and fall, irrespective of sexual maturity.

A minimum of six scales were collected from above the lateral line and just posterior to the dorsal fin, and placed onto gummed cards in the field or laboratory. Fish length, weight, date of capture, capture location, sexual maturity, and sex if it could be determined, were recorded on each scale card. Scales on the gummed cards were subsequently pressed into acetates for age analysis.

Acetate cards containing rainbow trout scale impressions were delivered to Eastern Washington University for analysis. The acetate cards were viewed using a Micro Design 895A microfiche reader, equipped with a 48X lens. One person viewed scale impressions from all fish while the second person viewed scale impressions from only the first twenty scales at each site and then every fifth for the remainder. In order to maintain observer independence, the wo observers never read the same scale simultaneously. The age estimates made by the second worker were used to determine the level of aging precision.

Fish ages were determined by counting the number of annuli (Jearld 1983) on scales. An annulus was defined as a region of closely spaced circuli where "cutting over" of the annulus across previously deposited circuli was present (Jearld 1983). "Cutting over" occurs in the winter months when water temperatures are colder and feeding and growth of the fish slow down. We followed the accepted convention that a fish's growth year begins on January 1 and ends on December 31. An annulus had to be completely formed to be counted as a full year's growth. Measurements were made along the mid-line of the anterior part of the scale to the nearest millimeter, to determine the distances from the focus to each annulus and from the focus to the edge of the scale. Age was determined for the single, best scale, that is, the scale was mounted correctly and was not damaged or regenerated.

Length-at-age of the fish were determined using the Dahl-Lea method for back calculation (Francis 1990):

$$L_i = (L_c/S_c) * S_i$$

where: L_i = length of fish (mm) at each annulus;
L_c = length of fish (mm) at time of capture;
S_c = distance (mm) from the focus to the edge of the scale; and
S_i = scale measurement to each annulus.

Three statistics were calculated to determine the precision of the estimated ages. Scale reader variation (V) was one of the statistics used to estimate the precision of aging (see Chang 1982 for formula). Calculation of V allows an interpretation of how reproducible the age analysis was. High V's indicate low precision.

The index of precision (D) was used to determine the percent error contributed by each observation to the average age-class. D was calculated by dividing V by R (see Elliott 1977 for formula; Sokal and Rohlf 1969). Low D's indicate high precision.

The determination of an average percent error (APE) in aging the jth fish was done by using Beamish and Fournier's (1981 see reference for formula) formula. The index (APE) can be used to compare age determinations for individual readers aging a fish several times or the APE can be used to compare age determinations between readers. The set of determinations for a particular species with a smaller index is more precise than a larger index; greater precision is achieved as percent error is minimized. By averaging the length-at-age for each cohort the grand mean of length-at-age of rainbow trout for each age was determined for each mainstem section and tributary.

The minimum length of sexually mature fish in each tributary and mainstem section was determined for fish collected in the spring. Fish were collected using electrofishing, trapping, and angling methods. Upon capture, fish were gently squeezed to determine if gametes would be exuded (following methods described in Chapter 1). Length frequency histograms of sexually mature fish were used to determine the minimum size of spawners in each location. If the smallest sexually mature fish was more than 20 mm smaller than the next smallest sexually mature fish, then the smallest fish was rejected (assumed to be an outlier) and the next smallest fish was used as the smallest spawning fish. The rejection of outlier fish was done to avoid including fish that may have been moving through a section or into a tributary from another location, and thus were not representative of the section in which they were collected.

Relationships between elevation and trout length-at-age, growth, and minimum adult size was determined using Pearson-product moment correlations. In addition, relationships between the electrophoretically determined genetic stock structure of rainbow trout, elevation, and length at age was assessed by principal components analysis with length at age 1, 2, and 3 as input variables. Principal components analysis is a technique that can be used to minimize the number of variables in a data set and thus aid in the interpretation of multivariate ecological data (Gauch 1982). The technique finds linear combinations of those variables that explain the most variance in the data set. The combinations of variables that explain the most variance are arranged along principal component axis 1 and subsequent axes are then found perpendicular to the first which accounts for the maximal remaining variance. Because principal component axes are linear combinations of the data, the axes must be interpreted either visually or statistically. Relationships between genetic stock structure and principal component scores were determined by visual inspection of a principal components plot. Circles were drawn around principal component points to indicate levels of genetic organization or clusters, as identified in Appendix 1. Relationships between principal components scores and elevation were examined with Pearson product moment correlations. Differences between the mean length-at-age of trout in the mainstem and tributaries was tested using a t-test. Differences between the mean length-at-age between males and females was also tested with a t-test. A paired t-test was used to test for differences in the mean length-at-age between females and males in tributaries and mainstem of the Yakima River. All statistical tests were conducted using Statgraphics Version 5 (Statistical Graphics Corporation, 1991).

Results

In the upper Yakima River basin, in general, rainbow trout in the seven mainstem river sections grew faster and reached a greater length at a given age than in the 12 tributaries (Table 1). Length-at-ages 2 and 3 were significantly higher in the mainstem of the Yakima River than in the tributaries (age 2 - $t=2.75$, $P=0.014$, $df=17$; age 3 - $t=2.27$, $P=0.037$, $df=17$). Although the mean length of trout at age 1 was also higher in the mainstem of the Yakima River than in the tributaries, the difference was not significantly different ($t=0.255$, $P=0.802$, $df=17$). In contrast to this pattern, length-at-age of trout in three tributaries, Badger, Cherry, and Wilson creeks, was as great or greater than in mainstem sections (Table 1).

The mean annual growth increment of trout was negatively related to elevation (Table 2). Statistically significant correlations were observed between elevation and length-at-age, and between elevation and growth for all sites combined (mainstem and tributaries). There was also a statistically significant correlation observed between elevation and length-at-age for mainstem river sections (Table 3).

Table 1. Grand mean (mean length-at-age for each cohort sampled) of back calculated length-at-age data for rainbow trout in seven sections of the mainstem Yakima River and 12 tributaries arranged from low to high elevation. The number of cohorts (C), standard deviation (SD), range, and sample sizes (N) for each age class are reported.

	Age Class																								
	1					2					3					4					5				
Site[a]	avg	SD	C	range	N	avg	SD	C	range	N	avg	SD	C	range	N	avg	SD	C	range	N	avg	SD	C	range	N
Mainstem Section																									
LCYN	95	(6.8)	5	41-137	95	217	(19.4)	5	100-324	82	296	(5.1)	4	212-373	51	333	(11.0)	3	299-389	9					
UCYN	93	(10.1)	5	54-101	93	220	(29.3)	5	109-361	61	308	(18.8)	4	211-399	22	407	(39.6)	1	379-435	2					
EBURG	63	17.6)	4	45-129	113	199	19.1)	4	93-299	73	300	(12.0)	3	178-350	37	306	(0.0)	1	306-306	4					
THORP	81	(9.6)	6	U-120	90	179	(3.0)	S	94-270	52	262	142.8)	4	193-333	24	343	(16.3)	3	273-402	10	366		1	350-382	2
CELUM	76	(3.0)	6	44-114	101	176	(27.1)	5	88-321	52	248	(30.3)	4	134-336	22	305	(E9.1)	2	242-418	8					
NELSN	76	(8.5)	5	40-145	76	162	(0.7)	4	89-268	35	201	124.7)	3	150-239	5	306	(0.0)	1		1	---				
CRSTL	71	(7.1)	5	41-131	70	173	(52.6)	4	92-234	34	192	(3.0)	3	165-229	6	239	(24.0)	2	222-256	2	---				
Tributary																									
UMT	68	(2.5)	5	35-139	147	155	(26.5)	4	63-329	61	266	(28.1)	3	111-351	35	298	(4.2)	2	265-349	8	330		1	321-338	2
WIL	98	10.6)	6	58-270	97	236	(18.7)	5	110-353	79	320	18.2)	4	219-413	33	397	(15.7)	3	322-442	9	---				
CHR	90	112.7)	5	38-180	58	187	(28.2)	4	89-315	36	258	(53.7)	3	175-340	12	358	(22.6)	2	342-374	2	---				
BAD[b]	116	(0.0)	4	71-165	49	174	(22.6)	3	85-311	7	248	(0.0)	1	131-310	3	---					---				
MAN	79	(5.8)	4	45-150	141	147	(3.5)	3	96-264	58	173	(0.7)	2	133-201	11						---				
TAN	73	(0.7)	4	34-167	177	125	(3.1)	3	58-197	70	164	131.1)	2	89-236	10	---					---				
SWK	73	(1.2)	4	48-125	116	127	(4.5)	3	96-169	51	167	16.4)	2	137-235	14	202		1	202-307	2	---				
NPT	80	(15)	4	36-140	113	123	(0.7)	3	92-169	46	173	(0.7)	2	122-236	12	163	(0.0)	1	---	1	---				
MPT	69	(121)	5	33-123	146	121	(8.2)	4	76-174	70	143	(5.2)	3	114-177	14	175	12.1)	2	174-177	2	---				
WFT	81	(8.7)	4	30-128	157	130	11.4)	3	86-177	71	159	(6.4)	2	125-184	5	---					---				
CAB[c]	75	10.0)	3	59-90	14	137	(0)	2	---	1	168	(0.0)	1	---	1	---					---				
BIG[d]	65	(0.7)	3	46-115	38	113	11.4)	2	103-m	11	164	(0.0)	1	153-174	2	---									

[a]Site Names: Mainstem river sections LCYN = Lower Canyon, UCYN = Upper Canyon, EBURG= Ellensburg, THORP = Thorp, CELUM = Cle Elum, NELSN = Nelson, CRSTL = Crystal, Tributary names: MAN = Manastash Creek, NFT = North Fork Teanaway River, JUN = Jungle Creek, TAN = Taneum Creek, MFT = Middle Fork Teanaway River, CAB = Cabin Creek, SWK = Swauk Creek, BIG = Big Creek, UMT = Umtanum Creek.

[b]1988 through 1991 represented

[c]1988 through 1989 represented

[d]1988 through 1990 represented

Table 2. Mean annual growth increment (GI), standard deviation (SD), and range of rainbow trout collected from the upper Yakima River basin. Mainstem sections and tributaries are arranged from low to high elevation (C = number of cohorts in analysis). Sample sizes are the same as presented in Table 1.

	Age Class																		
	1				2				3				4				5		
Site[a]	GI	SD	C	range	GI	SD	C	range	GI	SD	C	range	GI	SD	C	range	GI	SD	C
Mainstem Section																			
LCYN	95	(6.8)	5	87-100	123	(12.7)	5	109-134	78	(22.5)	4	56-101	37	(18.0)	3	25-54			
UCYN	93	(10.2)	5	84-104	127	(21.5)	5	106-149	88	(22.0)	4	65-109	88	(0.0)	1	--			
EBURG	83	(7.6)	4	74-88	116	(8.7)	4	106-121	106	(14.1)	3	96-116	149	(90.0)	1	--			
THORP	78	(4.0)	5	74-82	101	(5.9)	4	97-108	83	(43.1)	3	34-114	81	(27.6)	2	59-112	25	(0.0)	1
CELUM	76	(3.0)	6	73-79	100	(25.5)	5	74-125	72	(25.0)	4	52-100	95	(0.0)	1	--			
NELSN	76	(8.5)	5	70-82	86	(7.8)	4	80-91	39	(26.2)	3	20-57	88	(0.0)	1	--			
CRSTL	71	(7.1)	5	65-79	102	(45.9)	4	74-155	84	(65.1)	3	38-130	47	(2.8)	2	45-49			
Tributary																			
VMT	68	(2.5)	5	66-71	87	(26.5)	4	66-116	111	(28.1)	3	107-114	47	(4.2)	2	32-62	37	(0.0)	1
WIL	99	(0.6)	6	98-99	138	(18.7)	5	122-159	84	(8.2)	4	72-99	77	(15.7)	3	72-86			
CHR	90	(12.7)	5	81-99	107	(28.2)	4	96-118	62	(53.7)	3	44-79	100	(22.6)	2	78-122			
BAD[b]	116	(0.0)	4	116-116	58	(22.6)	3	42-74	58	(0.0)	1	--							
MAN	79	(5.8)	4	72-82	64	(3.5)	3	62-67	26	(0.7)	2	23-29							
TAN	75	(0.7)	4	72-79	50	(3.1)	3	49-52	39	(31.1)	2	20-58							
SWK	74	(1.2)	4	72-74	54	(4.5)	3	51-58	42	(6.4)	2	39-44	40	(0.0)	1	--			
NFT	80	(1.5)	4	69-97	52	(0.7)	3	50-53	50	(0.7)	2	50-50	-10	(0.0)	1	--			
MFT	69	(3.6)	5	66-73	52	(8.2)	4	44-57	21	(5.2)	3	18-25	34	(2.1)	2	31-37			
WFT	81	(8.7)	4	74-91	53	(1.4)	3	52-55	28	(6.4)	2	25-32							
CAB[c]	75		3	--	63		2	--	30	(0.0)	1	--							
BIG[d]	65	(0.7)	3	64-65	48	(1.4)	2	47-50	50	(0.0)	1	--							

[a]Site Names: **Mainstem** river sections LCYN = Lower Canyon, UCYN = Upper Canyon, EBURG= Ellensburg, THORP = Thorp, CELUM = Cle Elum, NELSN = Nelson, CRSTL = Crystal. Tributary names: MAN = Manastash Creek, NFT = North Fork Teanaway River, JUN = Jungle Creek, TAN = Taneum Creek, MFT = Middle Fork Teanaway River, CAB = Cabin Creek, SWK = Swauk Creek, BIG = Big Creek, UMT = Umtanum Creek.
[b]1988 through 1991 represented
[c]1988 through 1989 represented
[d]1988 through 1990 represented

Table 3. Statistically significant correlations ($P<0.05$) between elevation and length-at-age, and between elevation and growth for all sites combined (mainstem and tributaries). Additionally, correlations between elevation and length-at-age for mainstem river sections are presented.

Site/Comparison	Age	r	P	N
All sites combined	1	-0.56	<0.0130	
Length-at-Age	2	-0.83	<0.0001	
	3	-0.89	<0.0001	17
All sites combined	0-1	-0.56	<0.0130	
Growth	1-2	-0.77	<0.0002	
	2-3	-0.69	<0.0012	19
Mainstem River	1	-0.96	<0.0008	
Length-at-Age	2	-0.92	<0.005	
	3	-0.96	<0.0006	
	4	-0.76	<0.0490	7

It appeared that patterns of genetic variation as determined using protein electrophoresis (genetic stock structure Appendix 1) was correlated with patterns of rainbow trout length-at-age. A close correspondence between the principal components scores and genetic stock structure existed (Figure 1). In addition, the axis 1 principal components scores were also correlated with elevation ($r=-0.85$, $P<0.0001$, $n=19$). Principal component axis 1 explained 82.2% of the variance in the length-at-age data whereas axis 2 explained 15.6% of the variance. Combined, the two axes explained 97.8% of the variance. Patterns in genetic stock structuring, however, also appeared to be correlated with elevation (Appendix 1).

Fish sampled in the tributaries appeared to become sexually mature at a younger age than those from the mainstem of the Yakima River. Fifty-two percent of male and 27% of female spawners sampled from the mainstem were age 1+ or 2+, whereas 89% of males and 66% of females sampled from the tributaries were age 1+ or 2+. No age 1+ female spawners were collected from the mainstem of the Yakima River, whereas 17 were collected in tributaries. In all sites, higher proportions of males also matured sexually at younger ages than females (Table 4). In addition, 74% of the spawners collected from the mainstem Yakima River and 83% collected from the tributaries were males.

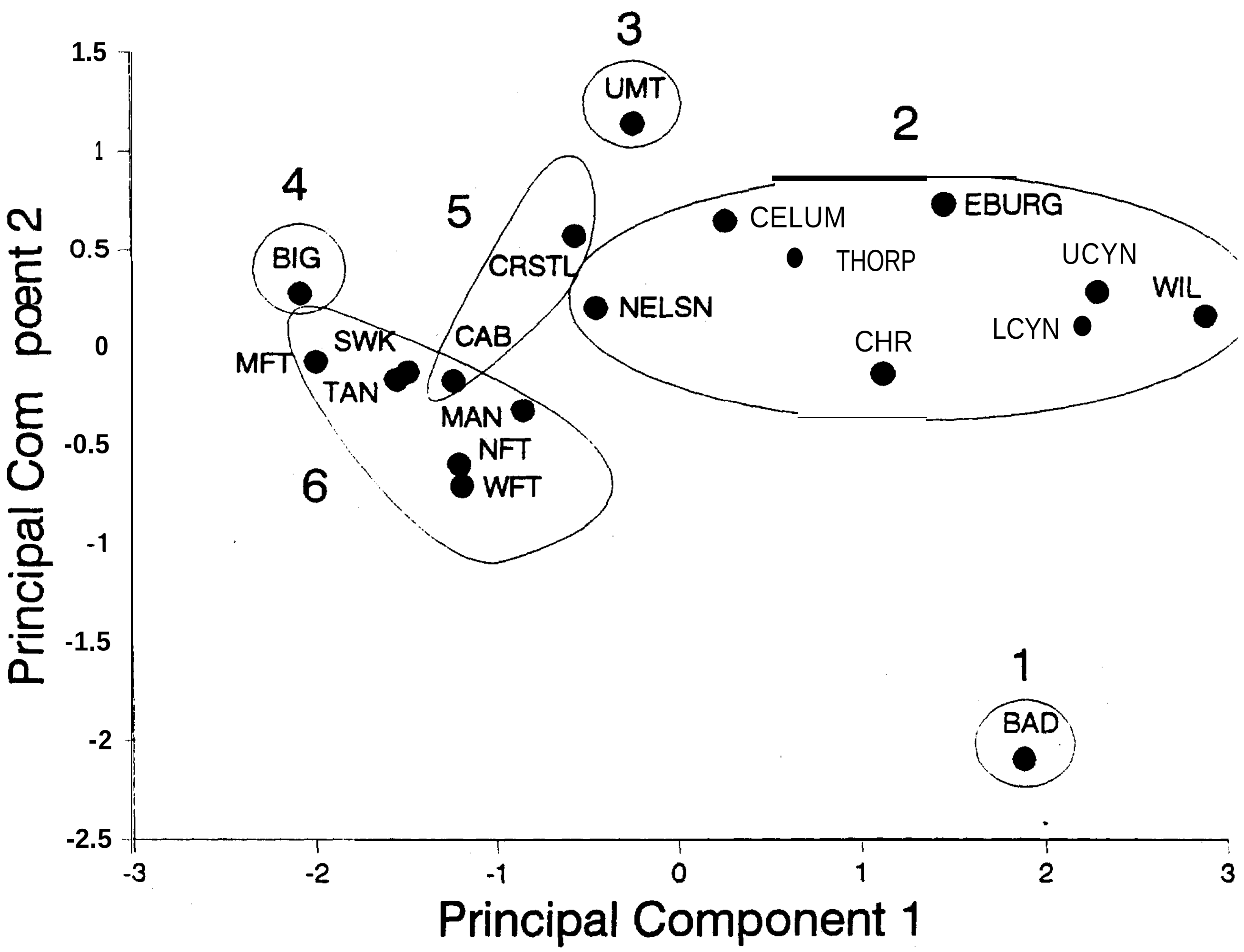

Figure 1. Principal components analysis of size at ages 1, 2, and 3 for rainbow trout in different geographic areas of the upper Yakima basin. Encircled sites or groups of sites were identified using electrophoretic analyses and were more genetically similar than they were different. Numbers above each circle correspond to the clusters resulting from analysis of allele frequencies as reported in Appendix 1.

Table 4. Average length (mm), standard deviation (SD), range, and sample size (N) of male and female rainbow trout spawners collected from the upper Yakima River basin. Both the mainstem sections and the tributaries are arranged from low to high elevation.

	Age Class 0				1				2				3				4			
Site[a]	avg	SD	range	N	avg	SD	range	N	avg	SD	range	N	avg	SD	range	N	avg	SD	range	N
Male																				
Mainstem Section																				
LCYN									310	(35)	253-365	8	332	(21)	283-361	12	356	(10)	349-363	2
UCYN					210			1	324	(35)	273-370	10	338	(20)	300-360	8	347			1
EBURG					213	(11)	205-220	2	384			1	336	(23)	316-374	8				
THORP					193	(18)	180-206	2	263	(17)	251-275	2	334			1	376	(8)	366-385	4
CELUM					203	(18)	186-220	4	308	(58)	250-395	5	335	(14)	325-345	2	438			1
NELSN					262	(79)	172-320	3					385			1				
CRSTL					156	(15)	140-169	3	211	(21)	187-229	4	224			1				
Tributary																				
UMT					134	(32)	94-227	17	220	(92)	139-353	8	268	(87)	165-390	6	339	(35)	300-375	4
WIL	174	(19)	147-190	4	291	(51)	209-349	7	335	(52)	215-393	12	348	(64)	275-430	4				
CHR									355			1								
BAD	141	(16)	110-166	20	160	(32)	100-226	11	234	(31)	215-270	3								
MAN					165	(28)	113-221	30	199	(34)	146-270	10	190	(33)	153-218	3				
TAN					145	(23)	110-191	23	176	(31)	153-276	15	201	(12)	192-221	5				
SWK					141	(21)	112-191	24	170	(26)	124-209	19	208	(41)	151-275	7	221			1
NFT					126	(18)	97-153	11	155	(22)	125-188	13	196	(28)	156-234	5				
MPT					145	(24)	104-211	79	167	(22)	131-216	28	172	(35)	140-216	5	201	(8)	195-206	2
WFT					149	(24)	105-191	20	178	(30)	135-253	33	193	(15)	182-203	2				
CAB					140	(11)	129-154	6												
BIG									184			1								
Female																				
Mainstem Section																				
LCYN													350	(32)	307-386	5	371	(17)	359-383	2
UCYN									346	(23)	330-362	2	340	(10)	331-350	3				
EBURG													372	(34)	333-396	3				
THORP																	363			1
CELUM									212			1	361	(6)	356-365	2	401	(23)	373-423	4
NELSN									274	(33)	250-297	2								
CRSTL									222	(39)	192-266	3	238			1	271			1
Tributary																				
UMT					147	(16)	135-158	2	230	(77)	147-335	7	313	(64)	159-351	8	313	(2)	311-314	2
WIL					258	(38)	223-315	5	330	(57)	239-400	14	398	(19)	380-418	4	437	(26)	408-465	5
CHR									352			1								
BAD	144				190	(18)	177-211	3	264	(68)	208-350	5	374			1				
MAN									203	(34)	165-266	7	200	(26)	182-219	2				
TAN									216	(20)	202-230	2								
SWK									172	(4)	169-175	2								
NFT									152	(1)	151-152	2	198			1	189			1
MFT					183			1	170	(17)	158-182	2	193	(21)	177-216	3				
WFT									168	(19)	150-188	3	223			1				
CAB					160	(17)	142-181	4												
BIG																				

[a] Site Names: Mainstem river sections LCYN = Lower Canyon, UCYN = Upper Canyon, EBURG= Ellensburg, THORP = Thorp, CELUM = Cle Elum, NELSN = Nelson, CRSTL = Crystal. Tributary names: MAN = Manastash Creek, NFT = North Fork Teanaway River, JUN = Jungle Creek, TAN = Taneum Creek, MFT = Middle Fork Teanaway River, CAB = Cabin Creek, SWK = Swauk Creek, BIG = Big Creek, UMT = Umtanum Creek.

Sexually mature rainbow trout collected from high elevation sections were not only young (age 1 and 2), but they were also small (length-at-age). The minimum size of rainbow trout spawners was negatively correlated with elevation ($r=-0.62$, $P<0.009$, $n=17$; Table 5).

The average percent error (APE) of age determinations between readers for all the tributaries and all years, ranged from 0.0 to 0.5, with the majority being 0.0 The coefficient of variation (V) also ranged from 0.0 to 0.5 with the majority being 0.0. The range of the index of precision (D) was from 0.0 to 0.35 with the majority being 0.

Table 5. Mean elevation and minimum length (FL) of mature rainbow trout in different tributaries and sections of the mainstem of the Yakima River.

Site	Elevation (m)	Minimum adult length (mm)
Mainstem Section		
LCYN	390	253
UCYN	417	253
EBURG	463	160
THORP	536	160
CELUM	573	160
NELSN	630	160
CRSTL	694	140
Tributary		
UMT	512	92
WIL	451	147
CHR	451	135
BAD	475	100
MAN	731	113
TAN	694	110
SWK	719	110
NFT	975	97
MFT	780	104
WFT	780	105
CAB	743	---
BIG	658	---

"Site Names: Mainstem river sections LCYN = Lower Canyon, UCYN = Upper Canyon, EBURG= Ellensburg, THORP = Thorp, CELUM = Cle Elum, NELN = Nelson, CRSTL = Crystal. Tributary names: MAN = Manastash Creek, NFT = North Fork Teanaway River, JUN = Jungle Creek, TAN = Taneum Creek, MFT = Middle Fork Teanaway River, CAB = Cabin Creek, SWK = Swauk Creek, BIG = Big Creek, UMT = Umtanum Creek.

Discussion

Study results to date suggest that rainbow trout growth in the upper Yakima basin is related to elevation and may have a genetic component as reflected in patterns of genetic stock structure. Results indicated that fish sampled in the mainstem of the Yakima River grew faster and reached a greater length-at-age than did fish in the tributaries. Length-at-age was negatively correlated with elevation which was most likely due to water temperature, food availability, and habitat quality. Slow growth in tributaries of young (small) fish may be related to one, if not all of the following physical factors: annual and diel water temperature fluctuations, food availability, and habitat quality. In addition, principal component scores were correlated with elevation and were associated with the genetic stock structure of rainbow trout (Appendix 1). Most likely, no single environmental or genetic factor controls fish growth in the upper Yakima basin. For example, length-at-age for trout in Badger Creek is distinctly different than in other streams, even those streams or mainstem sections at similar elevations. This suggests that factors other than elevation, such as genetic constitution, are responsible for differences between length-at-age of trout in Badger Creek versus other areas. It should be noted that correlations or associations between an environmental or genetic factor and length-at-age of trout does not demonstrate a cause and effect relationship. However, they do suggest potential influences.

The length-at-age of rainbow trout in the upper Yakima basin was generally smaller in tributaries and smaller in mainstem sections at age 1 when compared to other Northwest streams and rivers (Table 6). Differences in rainbow trout growth between the different streams and rivers may be related to factors described above for the Yakima basin. However, young of the year trout in the mainstem of the Yakima River appeared to grow much slower, as indicated by their length at age 1, than fish from other rivers of comparable size (Table 6). Unnaturally high flows in the mainstem of the Yakima River during the summer rearing time period may limit rainbow trout growth by forcing rainbow trout to occupy habitats having less than ideal water velocities.

Rainbow trout generally spawned at a younger age and at a smaller length in the mainstem of the Yakima River than in the tributaries. In addition, the minimum size of sexually mature fish was negatively related to elevation. These relationships may be related to the amount of environmental variability in each of the sites. For example, in streams with high annual discharge and high annual temperature variability, such as in the Teanaway River basin, the minimum length of mature fish was shorter and the percentage of fish spawning at age 1+ or 2+ was relatively high. In contrast, in streams with low annual discharge and temperature variability, such as in Cherry, Wilson, or Badger creeks or the canyon sections of the Yakima River, the minimum length of mature fish was relatively large and the percentage of fish spawning at age 1+ or 2+ was relatively low. In theory, animals that live in harsh environments should reproduce at younger ages than animals that live in benign environments (Pianka 1970).

Table 6. Average length-at-age (mm) of rainbow trout in several Northwest streams and rivers and in a representative subsample of areas in the upper Yakima basin.

	Age						
River/Stream	1	2	3	4	5	6	Reference
Mid-Columbia R.							
Tributaries	155	162	171	173	164	--	Peven 1994
Deschutes R.							
Nena Creek	110	190	270	300	340	--	Schroeder 1989
North Junction	130	220	290	320	330	350	Schroeder 1989
Jones Canyon	--	230	290	310	350	380	Schroeder 1989
Snake R. Idaho	124	251	341	450	480		Irving 1956
Spokane R. Id.	154	245	307	354	396		Underwood 1994
Montana streams	84	170	251	323	363		Carlander 1977
Hatchery Cr. Id.	127	244	333	445			Carlander 1977
Veddar R. B.C.	124	198	251	467			Carlander 1977
Mean	126	212	278	349	346		
Yakima River							
Lower Canyon[a]	95	217	296	333			
Yakima River							
upper mainstem[b]	82	190	258	320	366		
Taneum Creek	73	125	164	--			
N.Fk. Teanaway	80	123	173	163			
Swauk Creek	73	127	167	202			
Umtanum Creek	68	155	266	298	330		
Mean[c]	75	144	206	246	348		

[a]Taken from Table 1
[b]The average length-at-age of rainbow trout collected from all mainstem sections
[c]Excluding data reported for Yakima River Lower Canyon because these data are included in the Yakima River upper mainstem data set

The ages reported in this chapter appear to be highly reliable as indicated by our examination of the APE and V. As previously stated, there was low disagreement between readers of the number of annuli on individual scales. This suggests that the results were reproducible, with the exception of the age analysis of rainbow trout in Cherry and Wilson creeks. The circuli of the scales collected from fish in these two creeks were very evenly spaced, making it difficult to distinguish annuli. This same phenomenon has been reported for rainbow trout in a West Virginia stream (Surber 1937). Compared to other areas of the Yakima basin, the lack of discernable winter annuli in these fish might have been due to a more abundant food supply and relatively warmer water temperatures during winter (Surber 1937). In a study by Gray (1931), hatchery rainbows were experimentally fed throughout the winter and no annuli were formed and circuli were widely spaced. Although we may not be able to definitively explain why the annuli were not always discernable, we do believe that the high APE and V values were not generally excessive and thus did not affect reliable data interpretation.

The Lee method, which has been most commonly used in age studies, was not used here since there were not enough older (>4+) and younger (0+) age classes of rainbow trout present in the data for a reliable length-at-age regression. Using Lee's method without a complete spectrum of ages would have resulted in what is referred to as Lee's phenomenon. This phenomenon occurs in situations in which back-calculated lengths at an age are smaller than the source fish from which the lengths are back-calculated. Although we used methods to avoid misrepresentation of the length-at-age, we have few tagged, known age fish, to serve as controls.

In short, considerable variation in length-at-age was observed in the upper Yakima basin, and this variation may be related to a variety of ecological and genetically influenced factors. Among the ecological factors that were measured, elevation and stream size appeared to be important. With the exception of length-at-age 1, length of rainbow trout in the mainstem were similar to those in other rivers of similar size. Further analysis of the information in this chapter is planned, which should result in another report or publication.

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Chapter 6

Assemblage structure of fishes associated with rainbow trout in the upper Yakima River basin

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Abstract

Fish species associated with rainbow trout (*Oncorhynchus mykiss*) in the upper Yakima Basin were sampled to characterize and investigate factors that relate to assemblage structure. Fish assemblage structure was assessed during 1992 and 1993 in tributaries of the upper Yakima River and during 1993 in the upper mainstem of the Yakima River. Identification of assemblage types and the physical variables that influenced assemblage structure were determined using detrended correspondence analysis (DCA) and correlation analysis. Three major assemblage types were identified in the upper Yakima Basin during 1993. These assemblage types could be distinguished using elevation/temperature and stream size. Fish species that characterized assemblages in sites that were relatively high in elevation and within small streams (elevation 2040-3620 m, discharge 0.002-0.713 m^3/s, stream width 2.66-9.32 m) were bull (*Salvelinus confluentus*), cutthroat (*Oncorhynchus clarki*), and brook trout (*Salvelinus fontinalis*). Assemblages inhabiting relatively low elevation sites in small streams (elevation 1540-2040 m, discharge 0.001-0.010 m^3/s, width 1.81-3.94 m) were characterized by a high proportion of speckled dace (*Rhinichthys osculus*). Assemblages inhabiting relatively low elevation sites in large streams (elevation 1430-1960 m, discharge 7.301-29.432 m^3/s, width 33.8-56.6 m) were characterized by northern squawfish (*Ptychocheilus oregonensis*), chiselmouth (*Acrocheilus alutaceus*), suckers (*Catostomus sp.*), redside shiners (*Richardsonius balteatus*), longnose dace (*Rhinichthys cataractae*), mountain whitefish (*Prosopium williamsoni*), and spring chinook salmon (*Oncorhynchus tshawytscha*). Rainbow trout and sculpins (*Cottus sp.*) were ubiquitous and were part of all assemblages. Some DCA tributary site scores were in relatively different positions between 1992 and 1993. Differences in the relative position of DCA site scores might be attributed to stochastic factors, time of sampling related to fish migrations, recruitment success, habitat changes, and differences in sites that were sampled between years.

Introduction

The Yakima River, located in central Washington, is a major tributary of the mid-Columbia River and is host to a variety of native and exotic fish species. Patten et al. (1970) suggested that fish assemblage structure in the mainstem of the Yakima River was influenced primarily by water temperature and secondarily by water velocity. In that study 35 sites were sampled between river kilometers (rkm) 0 and 281 at approximately two month intervals in 1957 and 1958. A total of 23 native and 10 exotic species were collected. Each species of fish collected in the Yakima River during Patten et al's. (1970) field sampling was classified as inhabiting either cold or warm water and low or high water velocity locations. The majority of species classified as "cold water" were primarily caught above the Sunnyside and Wapato irrigation diversions between river kilometers 153 and 281. In contrast, "warm water" species were primarily caught below irrigation diversions between 0 and 145 rkm.

Various studies have shown that fish assemblage structure in tributaries and subbasins of the Columbia River are organized by abiotic variables similar to those mentioned by Patten et al. (1970). Fish assemblages in two large Columbia River tributaries, the John Day River and the Willamette River, were organized using variables similar to those identified by Patten et al. (1970) in the Yakima River. In three subbasins of the John Day River, elevation and stream size were believed to be the most influential factors determining assemblage structure (Leitzinger 1991). In the Willamette basin, Kruse (1988) found assemblages to be structured based primarily on three habitat parameters: channel unit composition, cover, and discharge.

The purposes of this study were to identify fish assemblage types in the upper Yakima River basin above 180 rkm, and identify what environmental factors influenced these types. In addition, temporal comparisons of assemblage structure was evaluated. We acknowledge that fish assemblage structure may be dynamic and that our portrayal of assemblage structure was a "snapshot" of potentially many states of the assemblage. Finally, spatial comparisons of the factors that influence assemblage structure in the upper Yakima basin were compared to those influencing assemblages in two large tributaries of the Columbia River (Willamette and John Day rivers). All results and data interpretations presented in this chapter should be considered preliminary and subject to revision as additional data are collected and/or analyses are performed.

Methods

Study Design

The basic design called for establishment of index study sections in sites representative of habitat in their respective reaches, the number of which happened to vary between sampling years. To the extent possible, the same index sites were sampled both in 1992 and 1993, allowing spatial and temporal contrasts. Index sites in tributaries and the mainstem Yakima River were sampled once during low flow time periods of the summer and early fall when fish assemblages were assumed to be most stable. To investigate the hypothesis that fish movement influenced assemblage structure, fish movement was evaluated using fish traps in three tributary streams.

Tributary Sampling

Index sites in upper Yakima River basin tributaries were sampled once from July through September during 1992 (Pearsons et al. 1993) and 1993. Twenty-three sites were sampled in 1992 and 31 in 1993. Block nets were placed at the top and bottom end of each 100 m long site to prevent fish from moving into or out of the site during sampling. Fish were stunned with a backpack electrofisher, netted, and placed in a holding bucket. After each of two electrofishing passes in each site, fish were identified to species and all individuals counted. Population estimates of rainbow trout were also conducted in these sites (presented in Chapter 4 of this report). Site elevation, gradient, standard deviation of mean thalweg depth, maximum depth, pool area, pool number, width, and discharge were measured following methods described in Chapter 4. Relationships between site elevation and temperature were investigated by placing maximum-minimum recording thermometers within eight tributaries.

Movement of fish in and out of three tributary streams was assessed using traps. Two trap designs were used and methods are described in Chapter 1 of this report. Fish were counted and measured and released unharmed within 1-3 days of their capture, and released in the direction of movement.

Mainstem Sampling

Fishes were sampled in the mainstem of the Yakima River in 1993. Fish were enumerated in five sites by estimating the number and species of fish observed during electrofishing. Sites ranged in length from 4.0 to 6.3 km (McMichael et al. 1992; Pearsons et al. 1993; Chapter 4 of this report). Sampling began at dusk and continued into the night to increase capture efficiency. Fish were sampled using a driftboat electrofishing unit mounted with halogen lights (McMichael et al. 1992).

Resident trout were netted for population estimation (Chapter 4 of this report) while other species were visually identified and individuals were enumerated. Data were recorded at the end of four equal spatial intervals within each site. On two consecutive nights, a single side of the river was sampled in each section. One week later the same protocol was followed. In short, each bank of the river was sampled twice. In two sections (UCYN and EBURG), species abundances was not estimated over approximately 50% of the distance.

Analysis

Relative abundances of fish were expressed in two ways; densities and percent composition. Densities were calculated for fish inhabiting tributaries. Relative densities of fishes in the tributaries were calculated by adding the numbers of fish collected on two electrofishing passes and dividing by the site area. Percent composition of each species was used in the mainstem/tributary comparison in 1993 because sampling methods differred between the areas. Percent composition was calculated for each species by dividing the number of individuals of each species by the total number of fish collected or observed. In the mainstem Yakima River, total counts of each species were averaged for each bank and then the bank averages were totalled.

Detrended correspondence analysis (DCA) using DECORANA (Hill 1979) was used to compare assemblage structure of fishes in the mainstem Yakima River and tributaries in 1993, and in the tributaries in 1992 and 1993. Detrended correspondence analysis is an indirect gradient analysis (ordination) that is superior to other ordination techniques such as principal components analysis and correspondence analysis because it eliminates the arch or horseshoe effect and has very good performance when species have nonlinear and unimodal relationships to environmental gradients (Gauch 1982). Detrended correspondence analysis simultaneously orders sites and species using a weighted averaging ordination technique (Gauch 1982). Analyses result in multiple axes which are dimensionless. All axes must be interpreted. Axes with low eigenvalues relative to other axes were not included in subsequent analyses (Gauch 1982). Percent composition data was arcsin transformed and rare species were downweighted (Gauch 1982). Densities of all species were used in the 1992 and 1993 tributary analysis. Each analysis will be referred to as "mainstem and tributaries 1993" (percent composition), "tributaries 1992" (density), or "tributaries 1993" (density). Interpretation of the DCA axes was aided by correlating DCA axis scores and environmental variables.

Results

Of the physical variables measured and analyzed in this study, elevation and stream size had the greatest correspondence with the observed structure of fish assemblages in the upper Yakima basin. Eigenvalues of the DCA suggested that the first two axes were adequate for describing assemblage structure (Table 1). In tributary, and tributary and mainstem analyses for 1993, axis 1 was interpreted as an elevation (temperature)/stream size axis (Figure 1 a,b; Table 2). In tributaries during 1992 and 1993, and tributaries and mainstem during 1993, axis 2 was interpreted as an elevation axis (Figure 1 a,b,c; Table 2). Although axes 1 and 2 were both correlated with elevation during 1993, the axes may be correlated for different reasons. Sites at different elevations may be intercorrelated with factors such as temperature, stream size, and distance from a colonization source. Elevation was negatively correlated with minimum ($r=-0.86$, $P=0.006$) and average ($r=-0.77$, $P=0.03$) temperatures in July and August 1993 in eight tributary sites which suggests that these variables were associated with one another. Results of the physical habitat measurements are presented in Chapter 4 of this report.

Table 1. Eigenvalues for DCA axes during two years in tributaries and mainstem sites of the upper Yakima River.

	Eigenvalue
<u>1992 tributaries</u>	
Axis 1	0.46
Axis 2	0.24
Axis 3	0.02
Axis 4	0.01
<u>1993 tributaries</u>	
Axis 1	0.78
Axis 2	0.24
Axis 3	0.07
Axis 4	0.04
<u>1993 tributaries and mainstem</u>	
Axis 1	0.58
Axis 2	0.25
Axis 3	0.10
Axis 4	0.04

Table 2. Pearson product-moment correlations between DCA axes and physical variables in tributaries and upper mainstem Yakima River areas during 1992 and 1993.

Physical variable	1992 tributaries		1993 tributaries		1993 trib and main	
	Axis 1	Axis 2	Axis 1	Axis 2	Axis 1	Axis 2
Elevation	-0.13	0.68**	-0.63**	0.67**	-0.78**	0.64**
Gradient	-0.10	0.06	0.04	0.47*		
Thalweg depth SD	-0.00	-0.04	0.32	0.14		
Maximum depth	0.33	0.18	0.26	0.11		
Pool area	-0.24	-0.18	-0.06	-0.12		
Pool number	-0.19	-0.07	0.44*	0.01		
Width	-0.27	-0.15	-0.53*	-0.04	0.72**	-0.16
Discharge	-0.01	0.18	-0.42*	0.09	0.72**	-0.15

* P<0.05
** P<0.001

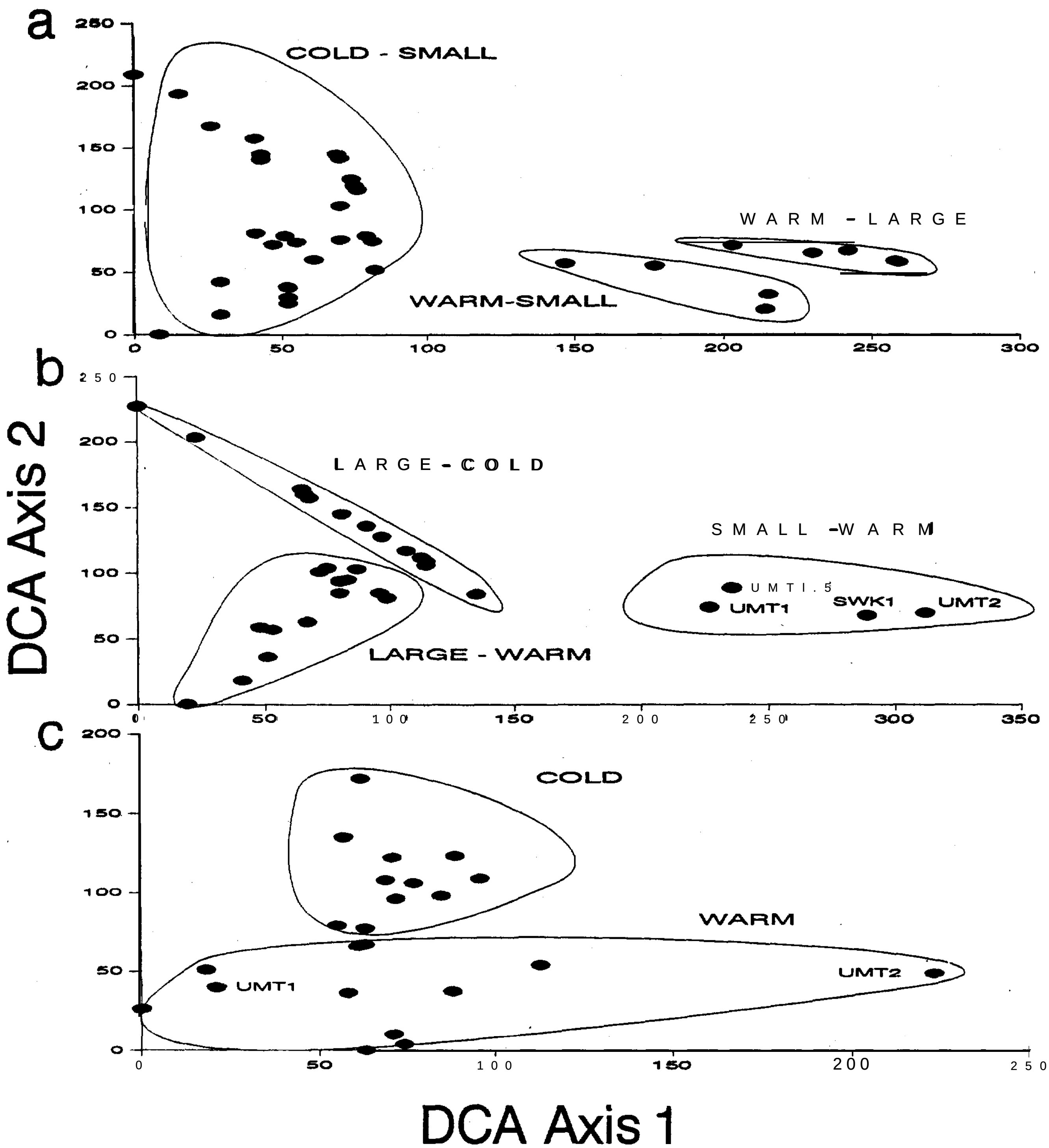

Figure 1. Detrended correspondence analysis of sites in a) the tributaries and mainstem of the upper Yakima River basin during 1993, b) tributaries during 1993, and c) tributaries during 1992. a) Axis 1 is interpreted as elevation (temperature)/stream size and axis 2 as elevation (distance from a colonization source). b) Axis 1 is interpreted as elevation (temperature)/stream size and axis 2 as elevation (distance from a colonization source). c) Axis 1 could not be interpreted and axis 2 is interpreted as elevation (temperature).

Site scores and interpretation of DCA axis 1 differed between 1992 and 1993 in tributary comparisons. In 1992, none of the measured physical variables correlated with axis 1, however in 1993 many of them did (Table 2). SWK1 and UMT 1.5 were not sampled in 1992 and were quite influential in the DCA for 1993 (Figure 1 b,c). UMT1 was at different ends of axis 1 in 1992 and 1993, presumably because of the differences in speckled dace (*Rhinichthys osculus*) densities between the two years.

Three types of stream fish assemblages in the upper Yakima River basin were identified. Each assemblage type occupied sites of distinct ranges of elevation/temperature and stream size. We will refer to these sites as cold-small (elevation 2040-3620 m, discharge 0.002-0.713 m^3/s, stream width 2.66-9.32 m), warm-small (elevation 1540-2040 m, discharge 0.001-0.010 m^3/s, width 1.81-3.94 m), and warm-large (elevation 1430-1960 m, discharge 7.301-29.432 m^3/s, width 33.8-56.6 m)(Figure 1). Sites in the cold-small group were from Taneum, Swauk, Cabin, and Jungle creeks and the North, Middle and West forks of the Teanaway River. In these sites bull (*Salvelinus confluentus*), cutthroat (*Oncorhynchus clarki*), or brook trout (*Salvelinus fontinalis*) were present (Figure 2a, 3). Sites in the warm-small group included Umtanum Creek and the lowest site in Swauk Creek. These sites were characterized by a high proportion of speckled dace (Figure 2a, 3). Sites in the warm-large group were from the upper mainstem of the Yakima River. These sites were characterized by the presence of northern squawfish (*Ptychocheilus oregonensis*), chiselmouth (*Acrocheilus alutaceus*), suckers (*Catostomus sp.*), redside shiners (*Richardsonius balteatus*), longnose dace (*Rhinichthys cataractae*), mountain whitefish (*Prosopium williamsoni*), and spring chinook salmon (*Oncorhynchus tshawytscha*) (Figure 2a, 3). Rainbow trout and sculpins (*Cottus sp.*) were ubiquitous and therefore were part of all assemblages. Certain species or groups of fishes were quite rare, such as hybrid trout (*Oncorhynchus sp.*), unidentified trout (*Oncorhynchus sp.* - age 0+ rainbow or cutthroat trout that were too small to identify in the field), burbot (*Lota lota*), pumpkinseed (*Lepomis gibbosus*) and yellow perch (*Perca flavescens*) and were not useful for classification of assemblage types. Species characterizations in tributaries were similar to patterns described above (Figure 2 b,c, 3).

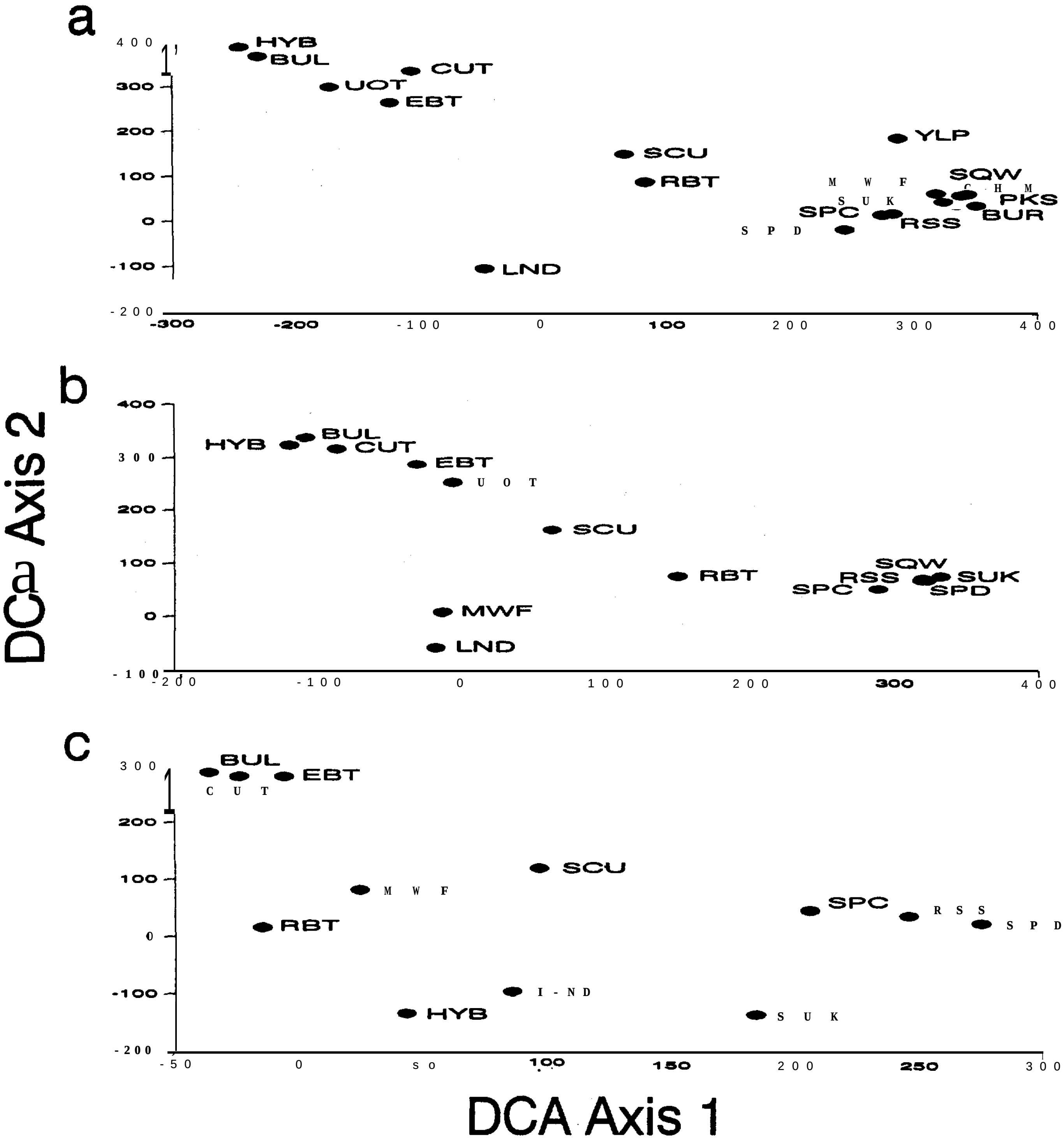

Figure 2. Species scores of fishes in a) the tributaries and mainstem of the upper Yakima River basin during 1993, b) tributaries during 1993, and c) tributaries during 1992 as identified by DCA. HYB=hybrid trout (cuttroat x rainbow), BUL=bull trout, UOT=unidentified age 0+ trout, EBT=eastern brook trout, CUT=cutthroat trout, LND=longnose dace, SCU=sculpin sp., RBT=rainbow trout, SPD=speckled dace, SPC=spring chinook salmon, SUK=sucker sp., YLP=yellow perch, MWF=mountain whitefish, RSS=redside shiner, SQW=northern squawfish, CHM=chiselmouth, PKS=pumpkinseed, BUR=burbot.

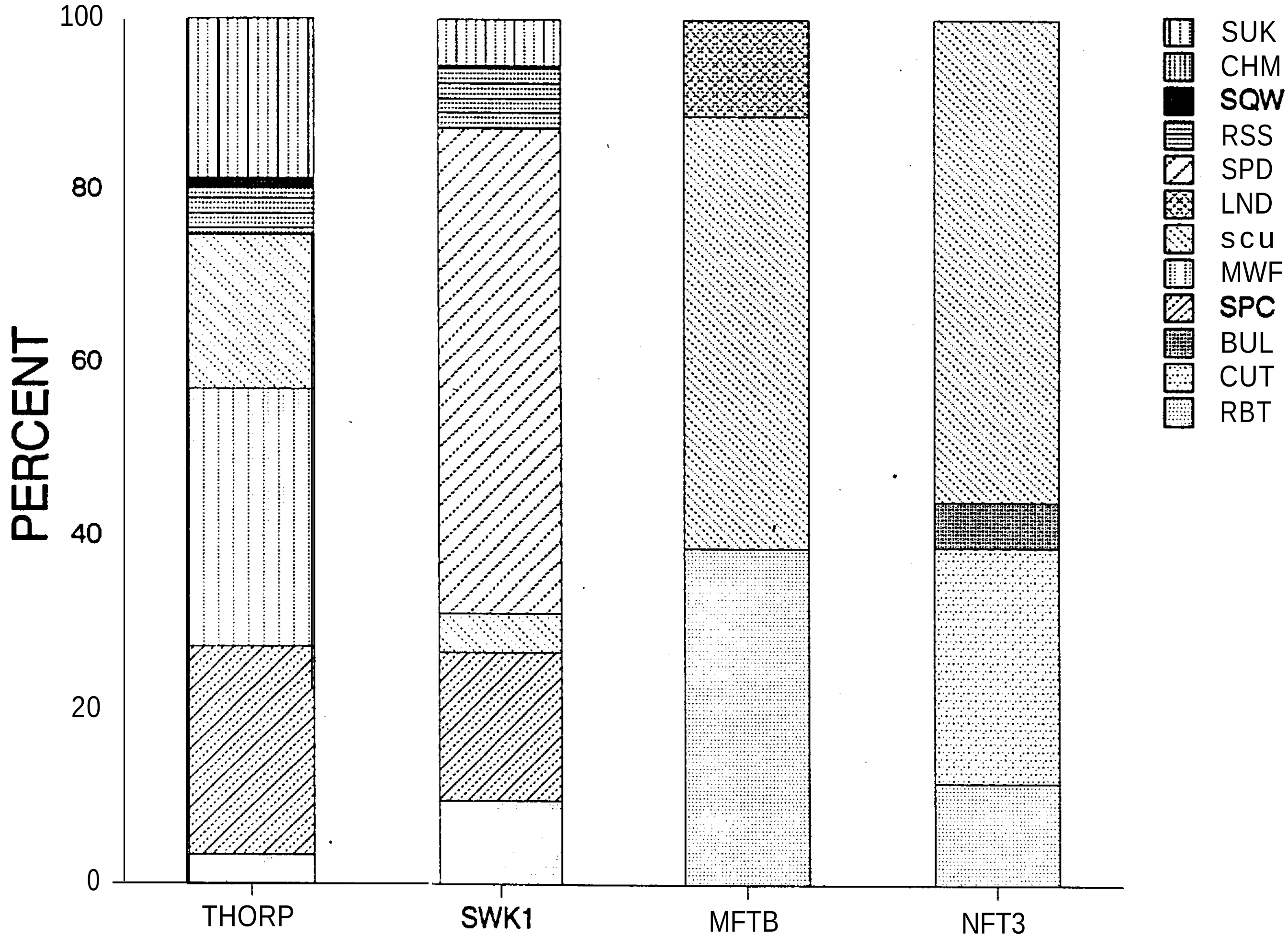

Figure 3. Percent composition of fishes in representative sites of three environmental types in the upper Yakima River basin during 1993. THORP=representative site of the warm - large environmental type, SWK1=representative site of the warm - small environmental type, MFTB and NFT3= representative sites of the cold - small environmental type.

Considerable up and downstream movement of fishes occurred during the spring and summer in Umtanum, Swauk, and Taneum creeks. Movements of rainbow trout in the upper Yakima River is presented in Chapter 3 of this report. Other species collected in traps include sculpins, redside shiner, longnose dace, speckled dace, northern squawfish, chiselmouth, mountain whitefish, juvenile spring chinook, cutthroat trout, bull trout, pumpkinseed, carp (*Cyprinus carpio*), and bridgelip sucker (*Catostomus columbianus*). Because of the large spaces between weir/trap pickets, only the movements of large fishes could be quantified. Large numbers of adult bridgelip suckers migrated into Umtanum, Taneum, and Swauk creeks during the spring and early summer (Table 3). The timing of these migrations partially overlapped with the times that rainbow trout spawn in these creeks (Chapter 4 of this report); however, bridgelip suckers generally migrated into tributaries later than rainbow trout.

Table 3. Number, length (mm) and (standard deviation), and timing of adult bridgelip suckers caught immigrating (up) and emigrating (down) for three tributaries.

Tributary	Number	Length (mm)		Timing	Trapping dates
		Mean	SD		
Umtanum					
up	573	405	35	4/15 - 5/30	2/10 - 6/2[a]
down	465	396	63	4/10 - 6/1	
Swup`					
down	166	40386939		4/21 --6/25 5/13 7/2	2/12 - 8/4[b]
Taneum					
up	58	--------		5/9 - 5/25	2/13 - 8/11[c]
down	386	396	30	5/23 - 6/29	

[a] trap was inoperable 3/11 - 4/6
[b] trap was inoperable 2/16 - 3/4
[c] trap was inoperable 2/16 - 3/2, 4/26, and 5/13 - 5/24

Discussion

Based on data collected to date, the structure of fish assemblages in the upper Yakima basin appears to be correlated primarily with elevation and stream size. Furthermore, there are many variables that are correlated with elevation and stream size that may be associated with the assemblage structure we observed. Temperature, stream order, and distance from sources of fish colonists are variables that may be correlated with elevation. Although all of these variables, and more, are potentially important for structuring assemblages, stream temperature is probably the most important in the present case. Many species such as bull trout and cutthroat trout are believed to be restricted in distribution primarily by temperature (Li et al. 1987). Species such as redside shiner and squawfish rarely overlap the distributions of bull trout and cutthroat trout because they cannot survive or avoid the areas with cold water temperatures. Stream size may also be correlated with other variables such as width, depth, discharge and stream order but it is difficult to know if there is a single variable that is responsible for the correlation between stream size and assemblage structure. Furthermore, variables that were not measured in this study (e.g., discharge variability, food production) may be important factors influencing assemblage structure.

Earlier work in the Yakima River also suggested summer water temperature as the primary factor influencing the distribution of fish (Patten et al. 1970). The classification of fish species as cold or warm water inhabitants used by Patten et al. (1970) was similar to the classification we used. The only species that were classified differently between the two studies were mountain whitefish and spring chinook salmon. We classified mountain whitefish and spring chinook salmon as warm water species and Patten et al. (1970) classified them as cold water species. In part, this difference may be related to the differences in spatial scales of the two studies. Patten et al. (1970) examined fishes throughout the Yakima River basin (excluding tributaries), whereas we examined fishes in the mainstem of the Yakima River and associated tributaries above Roza Dam. Thus, although spring chinook and mountain whitefish may inhabit relatively warm water in the upper Yakima Basin, when compared to the generally warmer water temperatures in the lower Yakima River, the water temperatures in the Yakima River above Roza Dam are relatively cold.

Temporal variation in site scores in the tributaries between 1992 and 1993 may be a result of stochastic factors, time of sampling related to fish migrations, differential recruitment success, and differences in sites that were sampled. For example, in 1992 Umtanum 1 had a very low axis 1 score and was widely divergent from Umtanum 2. In contrast, in 1993 Umtanum 1 had a high axis 1 score and was close to Umtanum 2. A large flood occurred in Umtanum Creek between years (on June 6, 1993).

In addition, large migrations of bridgelip suckers and rainbow trout were documented ascending Umtanum Creek during the spring, and juvenile spring chinook migrated into the creek later in the spring. If migration timing and magnitude varies between years, then differences in assemblage structure may merely reflect when the sample is taken (Decker and Erman 1992, Pearsons 1994). Differential recruitment success of species may also explain differences in assemblage structure between years. Finally, because DCA groups similar entities together relative to each other, inclusion of different sites between years may influence where a site is distributed in multivariate space.

Our study describes assemblage structure during low flow conditions of the summer and early fall, however assemblage structure may be considerably different, in certain sites, during different times of the year such as the winter and spring. The carrying capacity of habitat in stream sections may change between seasons which may affect fish movement, recruitment, or survival. Furthermore, fishes may use different portions of a river basin for different purposes (e.g. spawning and overwintering) and hence influence their local abundance. In short, the structure of fish assemblages reported in this study should be viewed as single positions within potentially many assemblage states both within a year and between many years.

Temporal variation in assemblage structure of fishes in the mainstem Yakima River could not be directly compared between 1957 and the present study because sampling techniques were too different. Patten et al. (1970) sampled fish within 100 m reaches of the river, whereas in this study the site lengths were between 4.0 and 6.3 km. In Patten et al.'s study, fishes were captured using a boat electrofisher which probably functioned similarly to the one used in this study. However, the operation of the boat was quite different. In that study, the boat was pushed upstream along both river banks while waders netted the fish, and in the middle of the river channel the boat drifted downstream while staff in the boat netted fish (R. Thompson, personal communication). Although they sampled fish by wading and floating, they probably underestimated the abundance of large fish which inhabit fast water. The efficiency of capturing fish in the middle of the channel was probably much lower than along the banks because many fish are difficult to catch as the boat is moving downstream. For example, the percent composition of mountain whitefish and rainbow trout caught between 185-258 rkm (similar to our study area) were much lower than in our study (generally less than 10% in Patten et al.'s study versus generally greater than 40% in our study). In addition their sample sizes were quite low (generally between 17 and 175 fish in sample locations between 185-258 rkm in the upper Yakima River) at many stations and assemblage structure was quite variable between adjacent sites. In contrast, we probably underestimated the abundance of small fish such as sculpins and dace (2-28% in our study versus 34-82% in Patten et al.'s study). In addition to sampling biases, differences in the proportions of certain species of fish between 1957 and 1993 may also reflect changes in assemblage structure resulting from changing biotic and abiotic conditions. Despite the differences in sampling techniques and any associated biases, assemblage structure in the mainstem was consistently different than what we observed in the tributaries.

The structure of fish assemblages in other major tributaries of the Columbia River have also been influenced by similar abiotic variables as those found in this study. In the Willamette basin, fish assemblage structure was most closely associated with channel unit type ranging from pools to riffles (Kruse 1988). Although that study included a point estimate of stream temperature in the analysis, and temperature was associated with the first DCA axis, temperature may have been more influential in their analysis if a better representation of temperature was included. Point estimates of stream temperatures are extremely variable and are difficult to compare in a meaningful way even when they are taken at the same time and date. Unfortunately, temperatures were measured on different dates and at different times within the sites of the Willamette basin. Axis 2 of Kruse's analysis was interpreted as habitat cover, and axis 3 as stream discharge.

Assemblage structure in three subbasins of the John Day River was associated with similar physical variables as in the upper Yakima basin. The two major factors explaining assemblage structure in each of the subbasins of the John Day River were elevation and stream size (Leitzinger 1991). Leitzinger (1991) also identified temperature to be the main correlate of elevation that was related to assemblage structure. In the John Day basin, two assemblage types were described: a warmwater type dominated by speckled dace, redside shiner, northern squawfish, and suckers; and a coldwater type dominated by steelhead and chinook salmon. The coldwater type could be further divided into assemblages dominated by steelhead and those dominated by chinook salmon. In short, fish assemblages in the John Day and Willamette river basins appeared to be structured along environmental gradients that were similar to those found in the Yakima basin.

We used abiotic factors to explain assemblage structure in the upper Yakima basin, but biotic factors may also have major influences on fish abundance (Li et al. 1987, Pearsons 1994), particularly rainbow trout and spring chinook salmon abundance. Northern squawfish, redside shiner, speckled dace, and sculpin were identified as non-salmonid species that may interact strongly (sensu Mills et al. 1993) with rainbow trout and chinook salmon in the upper Yakima basin (Pearsons et al. 1993). Bridgelip suckers may also interact strongly with rainbow trout in some tributary streams. Large populations of spawning suckers may destroy, disrupt, or enhance rainbow trout redds, or compete with rainbow trout for spawning habitat. On the other hand, bridgelip sucker larvae may provide a food source for rainbow trout particularly if drifting invertebrate abundance is low as might be the case in dessicating pools (Hubble 1992) or in pools created by beaver dams. Although none of these mechanisms have been documented in the upper Yakima River, the potential for these interactions exist because large migrations of bridgelip sucker have been documented in Umtanum, Swauk, and Taneum creeks during or immediately after rainbow trout spawning.

In summary, preliminary analyses suggest that the structure of fish assemblages in the upper Yakima River was influenced primarily by temperature and secondarily by stream size. The variation in assemblage structure in tributaries we observed between 1992 and 1993 may be related to stochastic factors such as flooding, or differences in time or sites sampled between years, and differential recruitment. Assemblage structure in Columbia River tributaries appear to be influenced by similar abiotic variables as those in the Yakima basin.

Acknowlegements

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Chapter 7

The effects of releases of hatchery-reared steelhead on wild salmonids in natural streams

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Abstract

The potential for hatchery fish to negatively impact wild fish has been identified as a concern for dwindling stocks of naturally-produced anadromous salmonids in the Pacific Northwest. A proposed supplementation project in the Yakima basin in central Washington prompted a multi-year examination of potential impacts of releases of hatchery-produced steelhead on preexisting wild salmonid populations. This investigation called for releases of approximately 33,000 hatchery-reared steelhead smolts (treatment) into an upper Yakima River tributary system in 1991, 1992, 1993, and 1994. This report summarizes preliminary results of work conducted on the 1991 through 1993 releases. Snorkelers conducted behavioral observations in two streams of different sizes that were influenced by the treatment and one control stream for each stream size affected by the treatment. Outmigrant trapping was used to examine mid- and large-scale displacements. Hatchery steelhead, which were generally larger than wild rainbow trout, dominated wild trout in most contests. Larger salmonids typically dominated smaller salmonids. Agonistic interactions observed in treatment streams generally involved more physical contact and more often resulted in the displacement of the subordinate fish than those observed in streams not containing hatchery steelhead. Within-channel unit displacements were documented, however no stream-reach or larger-scale displacements were detected. Predation by residual hatchery steelhead on naturally-produced salmonid fry was not detected. Behavioral interactions between hatchery-reared steelhead and wild resident rainbow trout did not appear to significantly impact the trout populations we examined. Population abundance of wild salmonids did not appear to have been negatively impacted by releases of hatchery steelhead. Downward trends in abundance from 1990 through 1993 were observed in control and treatment streams. The potential exists, however, for negative impacts to occur in situations were large numbers of hatchery-reared steelhead fail to emigrate during the typical smolt outmigration period.

Introduction

Concerns related to impacts of releases of hatchery-produced anadromous salmonids on preexisting wild fish populations have increased recently (Goodman 1990; Waples 1991). Furthermore, releases of hatchery origin fishes have contributed to the decline of populations of wild anadromous salmonids (Nehlsen et al. 1991). Hatchery origin fish may interbreed with, spread disease to, prey upon, or compete with wild fish. Behavioral interactions between stream-dwelling salmonids play an important role in structuring fish communities (Chapman 1966; Stein et al. 1972; Fausch and White 1986; Kennedy and Strange 1986). Interactions between hatchery-reared salmonids and their naturally-produced (wild) counterparts can dramatically affect the abundance and growth of the wild fish (Nickelson et al. 1986; Vincent 1987). Because artificially propagated fish can negatively affect wild salmonid populations, concern was raised about how new artificial propagation techniques (termed supplementation) might impact wild fish in the Yakima River basin. Supplementation is a relatively new strategy in the use of artificially-propagated fish in an attempt to increase the abundance of naturally-producing fish (BPA 1992). Supplementation differs from traditional hatchery programs in that it is not solely intended to increase harvest opportunities. Supplementation of anadromous salmonid species might affect wild populations of rainbow trout in the upper Yakima River basin through behavioral interactions. A large supplementation project has been proposed for the Yakima River basin which would potentially release large numbers of artificially-produced summer steelhead (*Oncorhynchus mykiss*) and spring chinook salmon (*O. tshawytscha*) into areas of the upper Yakima basin (Clune and Dauble 1991). Concern related to potential ecological impacts of fish released from supplementation facilities on preexisting resident rainbow trout (*O. mykiss*) in the upper Yakima River prompted us to examine some mechanisms of competition between steelhead juveniles produced in a hatchery and naturally-produced salmonids.

No Yakima Fisheries Project (YFP) facilities have been constructed, thus we used fish from the nearest available source of hatchery fish that happened to be in the Yakima basin. These fish were not raised using existing YFP guidelines and so may have behaved differently than fish from a proposed YFP facility. Thus, results from this work should be interpreted with this important caveat. We released hatchery-produced summer steelhead smolts from the Washington Department of Wildlife's Yakima Hatchery into a tributary of the upper Yakima River in 1991, 1992, and 1993 and examined the behavioral interactions between the various groups of fishes (McMichael et al. 1992, Pearsons et al. 1993). We will conduct one more set of releases in May, 1994, after which a final analysis will be completed.

Our overall objective was to try to understand some of the probable impacts that might result from interactions between juvenile steelhead from a YFP facility and naturally-produced rainbow trout. Specific objectives were to: 1) determine whether hatchery-produced fish interacted with pre-existing wild trout, 2) determine which group of fish dominated most interactions, 3) examine the differences between behaviors and outcomes in streams with and without hatchery steelhead, 4) determine the frequency and scale of physical displacement as a result of behavioral interactions, 5) examine the effects of releases of hatchery steelhead on the abundance of wild rainbow trout, and 6) determine whether hatchery-produced juvenile steelhead preyed upon juvenile wild salmonids.

This report is an annual progress report covering the period from January through December, 1993, and the information presented should be considered preliminary. A final report on this aspect of our work will be produced following the final year of field work in 1995.

Study Area and Experimental Design

This research was conducted within the Teanaway River drainage north of the town of Cle Elum, Washington. The Teanaway River is a tributary to the upper Yakima River. As described by McMichael et al. (1992), hatchery-produced steelhead were released into Jungle Creek, a tributary to the North Fork of the Teanaway River. Jungle Creek served as the small treatment stream (T_S). The fish released into Jungle Creek (at rkm 0.5) migrated downstream into the North Fork of the Teanaway River, which served as the large treatment stream (T_L). Jack Creek flows into T_L approximately 1.6 km below the mouth of T_S. We considered Jack Creek a small control stream (C_S, no hatchery fish were released there). The Middle Fork of the Teanaway River parallels the large treatment stream (T_L). We did not release hatchery steelhead into the Middle Fork of the Teanaway River and considered it a large control stream (C_L). We also collected population abundance information from index sites within the West Fork of the Teanaway River for comparisons of rainbow trout abundance estimates in T_L and C_L. We considered the West Fork of the Teanaway River as a large reference stream (R_L) for comparisons of trout abundance. McMichael et al. (1992) described the flora and fauna of the study area in greater detail than is presented in this chapter.

Methods

Smolt Releases

Hatchery-reared steelhead smolts (target release number = 33,000 per year) were released into Jungle Creek (T_S) during early May of 1991, 1992, and 1993 in a manner intended to mimic the outmigration pattern expected from an acclimation pond

(McMichael et al. 1992). The methods for the smolt releases were consistent with those described by McMichael et al. (1992).

Behavioral Observations

Direct underwater observation of fish behavior was performed by snorkeling in control (C_S and C_L, no hatchery fish released) and treatment (T_S and T_L, hatchery-reared steelhead smolts released) streams as described by McMichael et al. (1992) and Pearsons et al. (1993). In addition to the categories of information recorded in 1991 and 1992, the types of agonistic interactions were also determined in 1993. Each agonistic interaction was classified into one of the following five groups, threat, crowd, chase, nip, or butt. We defined threats as overt signs of aggression, such as fin-flares and body arching (Taylor and Larkin 1986; Holtby et al. 1993). Crowds occurred when fish moved toward other fish laterally, causing a subordinate fish to move out of the way (Helfrich et al. 1982; Taylor and Larkin 1986; Holtby et al. 1993). Chases occurred when one fish slowly pursued another fish for several body lengths without making physical contact (Keenleyside and Yamamoto 1962; Helfrich et al. 1982; Taylor and Larkin 1986). Nips were classified as physical contact in which one fish actively bit another fish (Stringer and Hoar 1955; Helfrich et al. 1982; Taylor and Larkin 1986). Physical contact made between two fish in which the mouth of the attacking fish was closed was classified as a butt. A contest may have included multiple interactions. For example, a hatchery steelhead and a naturally-produced rainbow trout could chase and nip each other several times during one contest.

Displacement

To determine whether juvenile hatchery-reared steelhead displaced wild fish we examined displacement at three spatial scales using methods described by McMichael et al. (1992) and Pearsons et al. (1993). We defined a displacement as one fish causing another fish to move away from a preferred feeding or holding site (Brown (1975) as cited *in* Helfrich et al. 1982). Small-scale displacements were those that occurred within a channel unit (sensu Frissell et al. 1986) of stream, such as a pool. Wild fish movement out of the release stream (T_S) concurrent with large numbers of hatchery fish was considered a mid-scale displacement. Large-scale displacement was monitored at a downstream migrant trap near the mouth of the North Fork of the Teanaway River (T_L), approximately 11 km downstream of the release site in Jungle Creek (T_S). Determination of small-scale displacements was more direct (because they were observed) than mid- and large-scale displacements which had to be inferred from fish emigration information.

Population Estimates

To determine the influence of hatchery steelhead releases on rainbow trout abundance and size structure, population abundance was assessed in four study streams. Population estimates were conducted in index sites in the North (T_L, N = 2), Middle (C_L, N = 3), and West (R_L, N = 3) forks of the Teanaway River (1990-1993) and in Jungle Creek (T_S, N = 1)(1991-1993) using the electrofishing methods described by McMichael et al. (1992). The third site in T_L was omitted from the analyses this year because it was much higher in elevation (over 260 m higher) than any control sites and the proportion of salmonid abundance that was rainbow trout was low (< 20% of salmonids present, see Chapter 4, this report).

Predation

To determine whether residual hatchery steelhead preyed upon post-emergent wild rainbow trout we collected residual hatchery steelhead from areas with abundant age 0+ rainbow trout (Pearsons et al. 1993). Residual hatchery steelhead were collected using backpack electrofishing equipment in the North Fork of the Teanaway River (T_L) and in Jungle Creek (T_S) on July 6, 1992, and all residual steelhead stomachs were removed and examined for the presence of fish.

Data Analyses

Data from underwater observations were pooled for all sites within each study stream. The variation between observational data from sites within each stream was small enough to allow pooling of the data for each stream. Two mean interaction rates were calculated for each stream and year; one during the smolt outmigration period (May) and one for the summer rearing period (June to October). Time periods were selected that corresponded to two distinctly different freshwater life history periods (steelhead smolts and residuals), each potentially having different interactive potential among fish present. Visual inspections of graphical information was used to examine differences between behaviors (and outcomes) among treatment and control streams.

Population estimate data (number and grams of rainbow trout per 100 m) were averaged for all sites sampled within each stream for each year (1990 to 1993). This yielded average rainbow trout and residual hatchery steelhead (where present) abundance figures for each stream for 1990, 1991, 1992, and 1993. More rigorous statistical analyses of these data will follow completion of data collection in 1995.

Results

Smolt releases

Total numbers, sizes, and smolt quality of hatchery steelhead released into Jungle Creek (T_S) varied among the three years of study. Numbers released were 31,542 in 1991, 38,000 in 1992, and 22,500 in 1993. Approximately 45% of the fish were released on the first Monday in May of each year, with 33% being released two days later, and the final 22% were released nine days after the initial release. The mean fork lengths (± SD) of the hatchery steelhead released each year were 201 mm (± 16) in 1991, 196 mm (± 16) in 1992, and 182 mm (± 21) in 1993. Mean weights were 81 g (± 25), 78 g (± 22), and 64 g (± 23), for 1991, 1992, and 1993, respectively. Mean condition factors for the fish were 1.00, 1.04, and 1.06 for 1991, 1992, and 1993, respectively. Of the fish released in 1991, 4.0% were classified as precocial males. The percentages of precocial males in the two subsequent years were considerably lower (1.0% in 1992 and 0.7% in 1993). Though smolt quality was not directly assessed in 1991, it appeared that very few (< 50%) of the fish released exhibited the external characteristics of steelhead smolts, such as absence of parr marks, dark banding of the caudal fin, and overall silver coloration (Ewing et al. 1984). In 1992, 72 to 76% of the fish released appeared to be smolts. In 1993, 92 to 100% were classified as smolts at the time of release.

Behavioral Observations

Hatchery steelhead generally dominated contests with wild rainbow trout and were also larger. Hatchery steelhead in Jungle Creek (T_S) and the North Fork of the Teanaway River (T_L) dominated preexisting wild trout in 76% of contests observed from 1991 through 1993 (Figure 1). When agonistic interactions among all groups of fish were pooled, larger fish dominated 84% of the contests observed (Figure 1). Hatchery steelhead were significantly larger than the resident trout in the study streams (Figure 2).

Agonistic contests between juvenile spring chinook salmon and resident trout were observed in C_L and T_L sample sites each year between 1991 and 1993 (N = 18). Spring chinook salmon dominated rainbow trout in nine of the contests (50%) between those species. Rainbow trout were also dominant in half of the contests. As stated earlier, each contest often includes multiple interactions (e.g. a reciprocal bout (Newman 1956)). Interestingly, spring chinook salmon dominated rainbow trout in 82% of the interactions observed (N = 38).

Observation rates of resident trout were generally higher after the May smolt emigration period while observation rates of hatchery steelhead were lower (Table 1). Residual hatchery steelhead (those observed between June and October) were

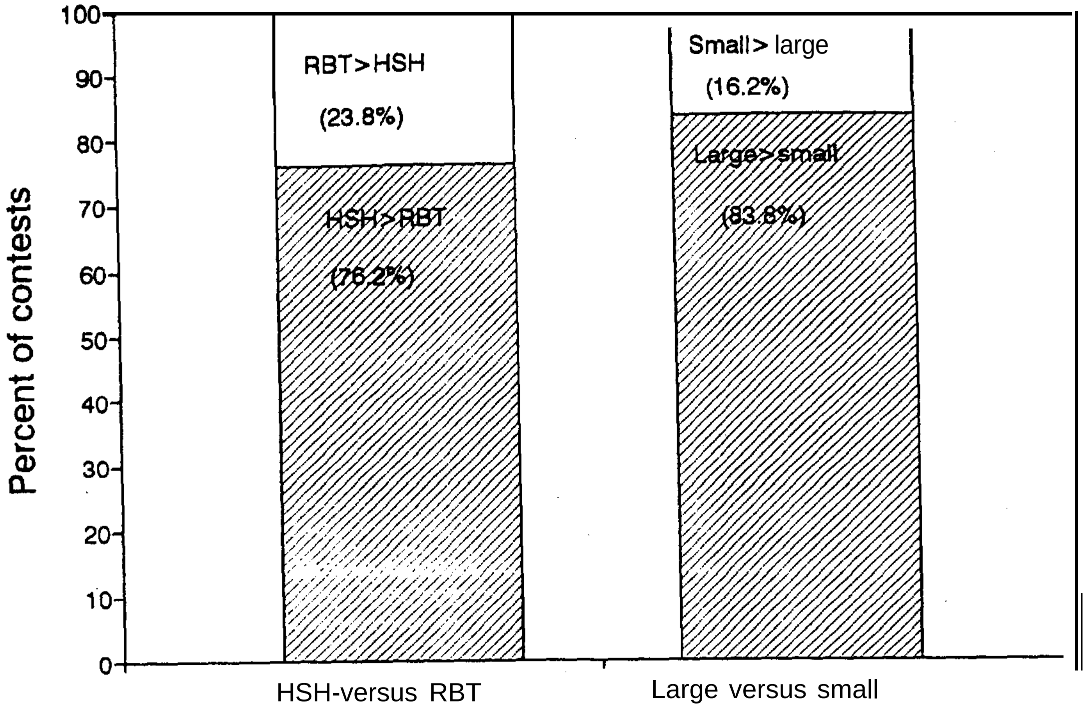

Figure 1. Dominance:subordinance relationships between hatchery steelhead (HSH) and rainbow trout (RBT) and between large and small salmonids as determined by direct underwater observation in the Teanaway River basin. Groups preceding the > symbol were dominant. Data were pooled for all sites in treatment streams from 1991 through 1993.

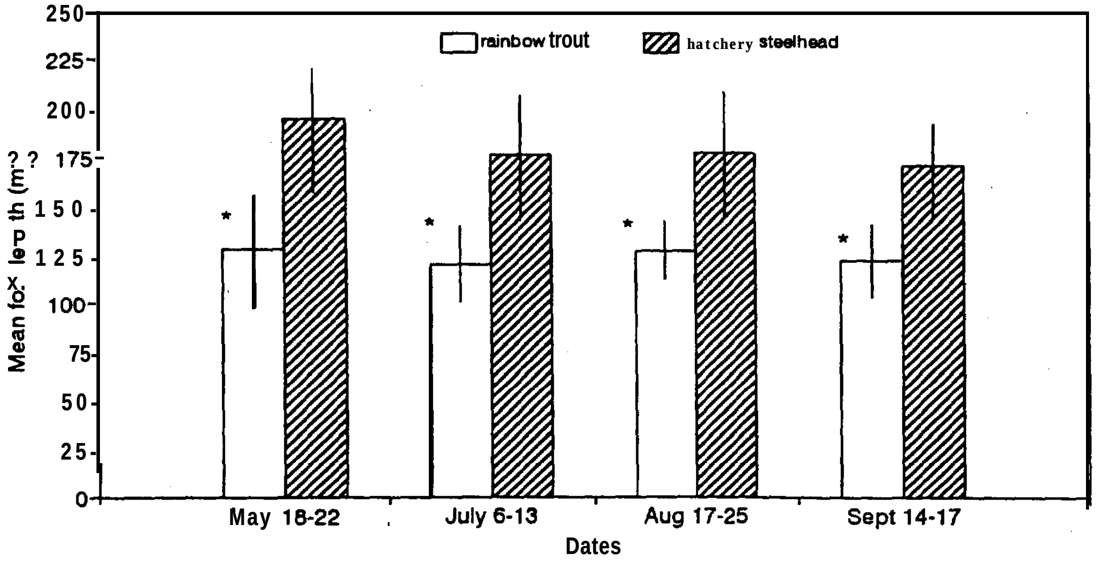

Figure 2. Mean fork length (mm) of naturally-produced rainbow trout and hatchery-reared steelhead during the summer of 1992 in the North Fork of the Teanaway River (T_L). Vertical lines denote $\pm$ 1 standard deviation. Asterisks denote significant differences between groups ($P<0.05$). Sample sizes for each time period ranged from 10 to 30 fish. (Data courtesy of S. Urakawa, Central Washington University)

encountered with higher frequency in 1991 and 1992 than in 1993. Juvenile spring chinook salmon were only observed during the summer months and were generally seen in the lower elevation index sites in T_L and C_L. It did not appear that the presence of hatchery steelhead resulted in an increased rate of behavioral interactions (interaction/fish/min). Within years, interaction rates were generally lower in treatment streams than in control streams (Table 1). Interaction rates also tended to be higher in small streams than in large streams (Table 1). Interaction rates were generally lower during the smolt emigration period than they were during the summer (Table 1). This was particularly evident in Jungle (T_S) and Jack (C_S) creeks.

The types of agonistic interactions observed in 1993 differed between control and treatment streams. Interactions observed in control streams generally involved less physical contact than those observed in streams where hatchery steelhead were present. Interactions in which physical contact was made (nips and butts) accounted for about 32% of the interactions observed in control streams and for almost half of the interactions observed in treatment streams (Figure 3).

Displacement

Hatchery steelhead displaced wild trout from apparently preferred microhabitats within habitat units, but did not displace trout from stream reaches over larger (0.2 to 11.2 km) spatial scales. Twice as many of the agonistic interactions observed in treatment streams (47%) resulted in the displacement (typically within the channel unit) of the subordinate fish than was observed in control streams (23%). Mid-scale displacements were not detected. Most of the trout that did move out of Jungle Creek (T_S) were age 0+ and moved out in greater numbers several days following the emigration of hatchery steelhead. The timing and magnitude of trout outmigration was similar between the release stream (T_S) and the small control stream (C_S), suggesting that the hatchery steelhead did not influence the movement of trout out of the release stream (Figure 4).

Table 1. Observation rates of resident trout (RBT), juvenile hatchery steelhead (HSH), and spring chinook salmon (SPC) in study streams in the Teanaway River basin during (May) and after (June to October) the smolt outmigration period, 1991 through 1993. The number and rate of agonistic interactions among these fish is also shown. T_S = Jungle Creek, T_L = North Fork of the Teanaway River, C_S = Jack Creek, C_L = Middle Fork of the Teanaway River.

Stream/ year	Obs. time (min)	Observation rates			Interactions	
		RBT/min	HSH/min	SPC/min	Number	Int/f/m[a]
			May			
T_S 1991	788	0.34	1.34	0.00	119	11.4
T_S 1992	1559	0.08	2.07	0.00	136	2.6
T_S 1993	640	0.20	2.85	0.00	414	33.2
T_L 1991	986	0.05	1.74	0.00	153	8.8
T_L 1992	419	0.05	0.52	0.00	20	20.1
T_L 1993	83	0.02	0.96	0.00	28	411.4
C_S 1992	520	0.44	0.00	0.00	29	24.1
C_S 1993	372	0.50	0.00	0.00	58	84.3
C_L 1992	467	0.15	0.00	0.00	21	66.1
C_L 1993[b]	5					
			June to October			
T_S 1991	223	0.07	0.32	0.00	5	25.5
T_S 1992	288	0.32	0.40	0.00	50	83.9
T_S 1993	82	0.17	0.18	0.00	15	630.8
T_L 1991	945	0.23	0.39	0.00	21	3.8
T_L 1992	977	0.36	0.37	0.01	68	9.6
T_L 1993	401	0.26	0.03	0.02	116	231.4
C_S 1992	219	0.53	0.00	0.00	15	48.6
C_S 1993	116	1.97	0.00	0.00	55	208.0
C_L 1992	1091	0.69	0.00	0.03	123	14.2
C_L 1993	549	0.61	0.00	0.33	238	83.7

[a] Interactions per fish per minute x 10^5.
[b] Poor snorkeling conditions prevented observations during May.

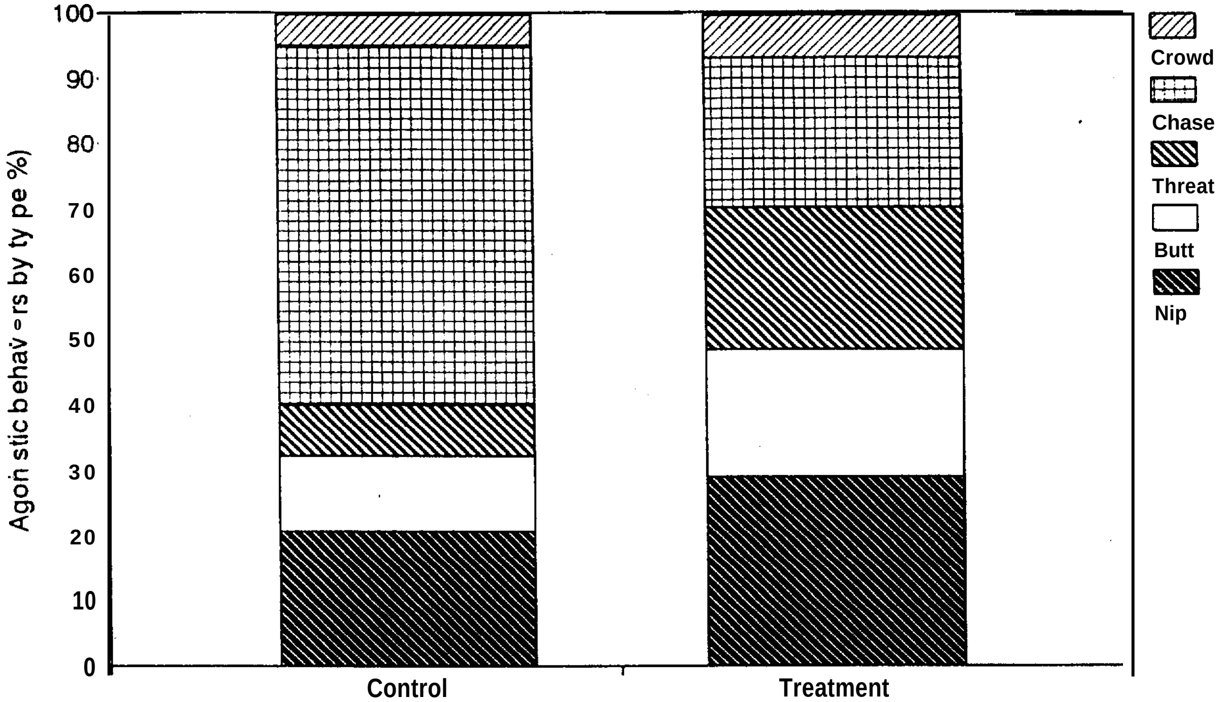

Figure 3. Percent by type of agonistic interactions observed in control streams (C_S and C_L, N = 354) and treatment streams (T_S and T_L, N = 571) during 1993.

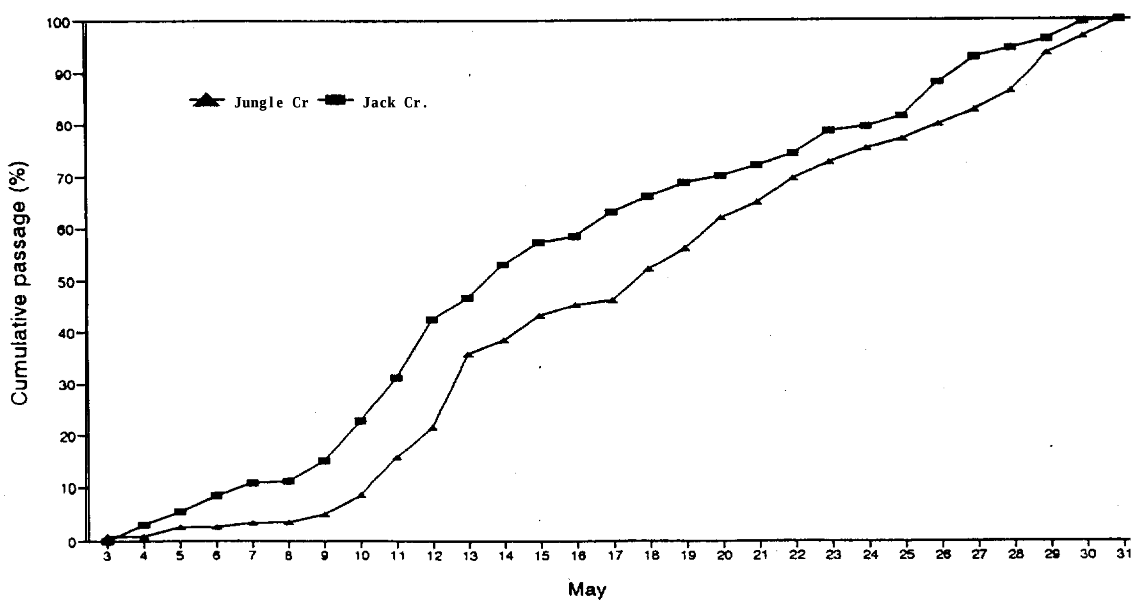

Figure 4. Cumulative outmigration of naturally-produced trout (and/or wild steelhead presmolts) in Jungle (T_S, N = 362) and Jack (C_S, N = 232) creeks during May of 1993.

Outmigration timing of resident trout (and/or wild steelhead presmolts) did not appear to be affected by the magnitude and timing of hatchery steelhead emigration. If large-scale displacements occurred we would have expected to detect large numbers of naturally-produced rainbow trout moving simultaneously with hatchery steelhead. We did not document large outmigrations of naturally-produced rainbow trout occurring concurrently with large outmigration pulses of hatchery steelhead from T_S or T_L *(Figures* 5 and 6). So, while small-scale displacements were observed, mid- and large-scale displacements were not seen in 1991 (McMichael et al. 1992), 1992 (Pearsons et al. 1993), or 1993 (this study).

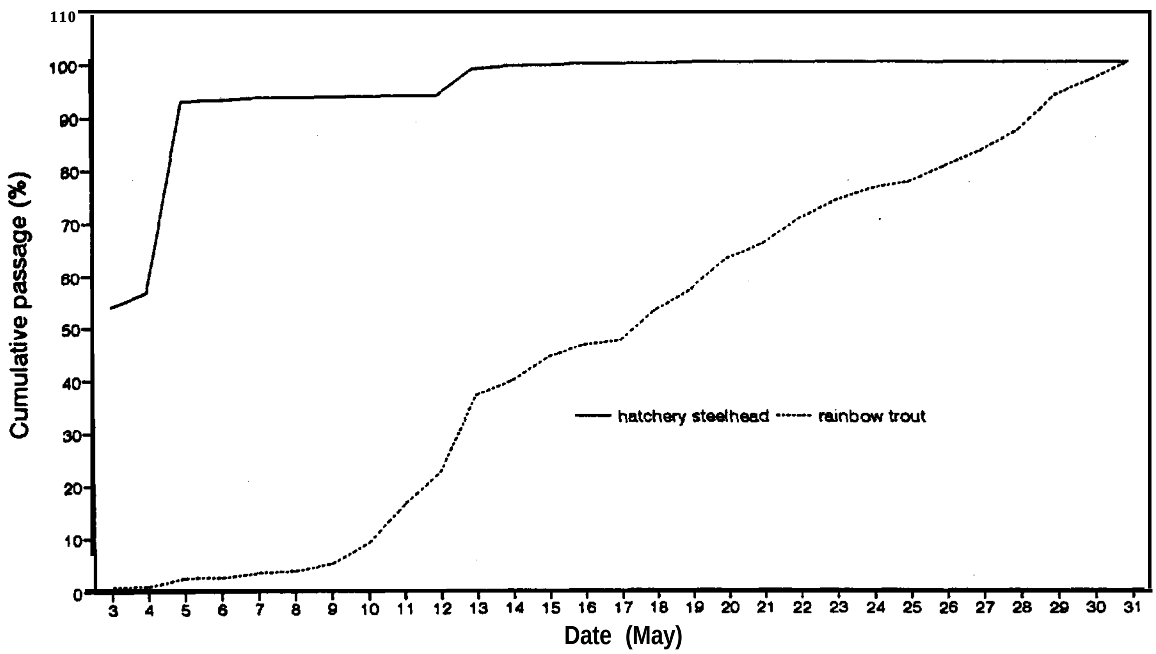

Figure 5. Cumulative emigration of juvenile hatchery steelhead and rainbow trout captured moving downstream out of Jungle Creek (T_S) in May, 1993. Release dates were May 3, 5, and 13.

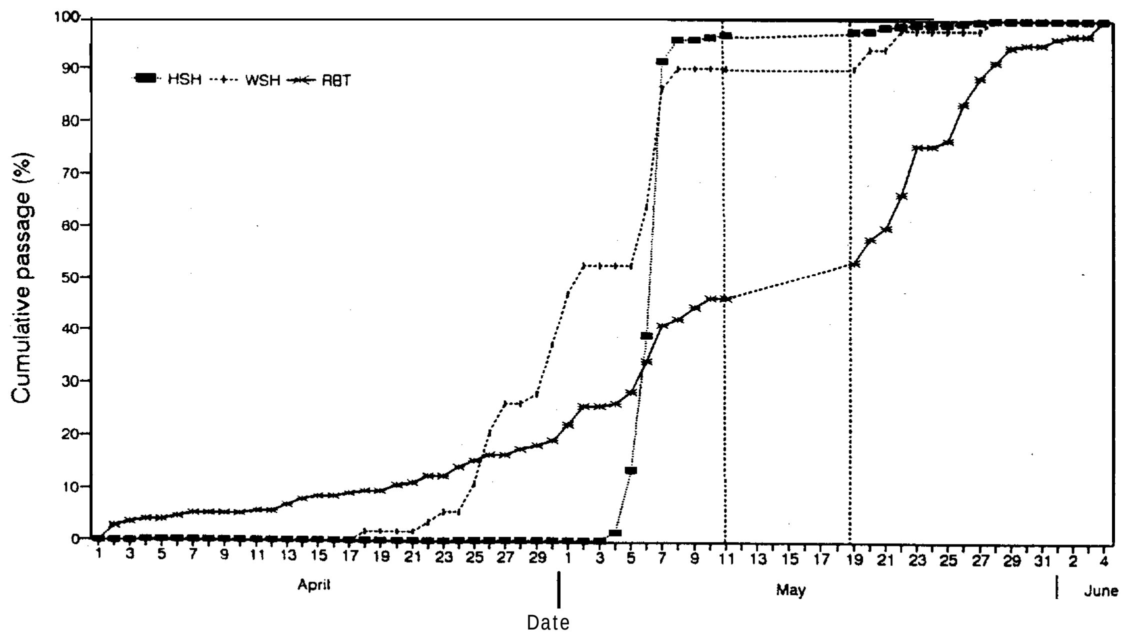

Figure 6. Cumulative outmigration of hatchery steelhead (HSH), wild steelhead smolts (WSH), and naturally-produced trout (and/or wild steelhead presmolts) (RBT) captured in a rotary screw fish trap near the mouth of the North Fork of the Teanaway River (T_L) during 1993. The trap was not operated from May 11 to 18 and lines were interpolated by eye.

Population Estimates

Rainbow trout densities did not appear to be influenced by the release of hatchery steelhead. Mean annual rainbow trout abundance (number/100 m) and biomass (g/100 m) declined in the North (T_L) and Middle (C_L) forks of the Teanaway River and remained quite stable in the West Fork of the Teanaway River (R_L) (Figure 7A). The abundance in R_L appeared to increase slightly, while the biomass appeared to decrease slightly. In contrast to the large streams, rainbow trout abundance and biomass increased in Jungle Creek (T_S) between 1991 and 1993 (Figure 7B). Residual hatchery steelhead were abundant in Jungle Creek (T_S) during all three years of sampling and due to their larger size, they constituted over 90% of the total salmonid biomass in 1991 and 1992, and about half of the total biomass in 1993 (Figure 7B). Because Jack Creek (C_S) became intermittent during 1992 and 1993 prior to the fall population estimate (about September 1), we did not estimate trout abundance there. We assumed that trout abundance in the study reach of Jack Creek at that time was at or near zero.

Predation

Stomachs from 55 hatchery steelhead residuals collected in the North Fork of the Teanaway River (T_L) and Jungle Creek (T_S) in July, 1992 contained no evidence of fish. Newly-emerged age 0+ trout were abundant in the areas where the hatchery steelhead were collected. In over 200 h of underwater observation between 1991 and 1993, no naturally-produced salmonids were consumed by hatchery steelhead and only one predatory attack was observed.

A

B

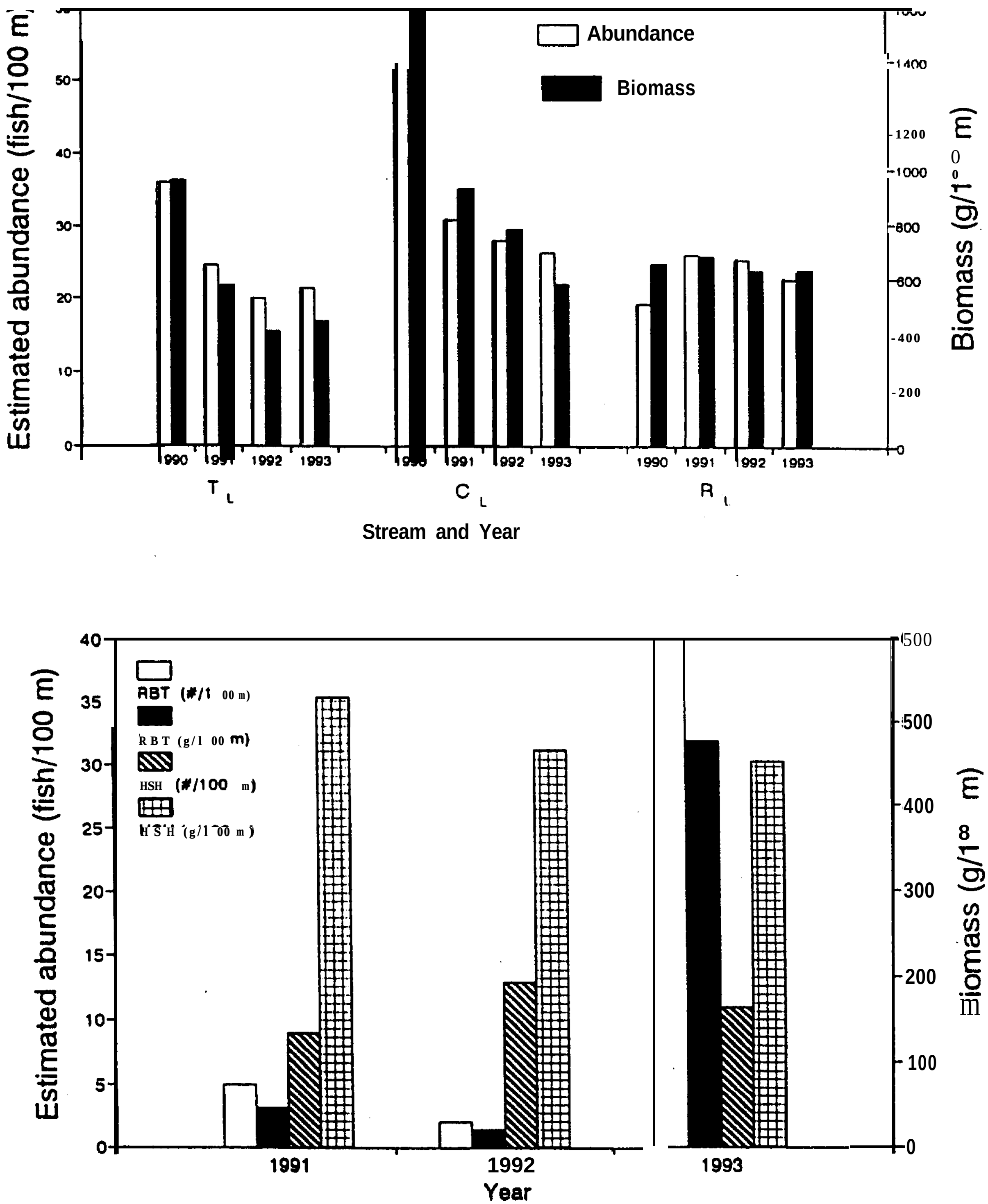

Figure 7. A. Estimated abundance (number of fish/100 m) and biomass (g/100 m) of wild rainbow trout in the North (T_L), Middle (C_L), and West (R_L) forks of the Teanaway River from 1990 to 1993. B. Estimated abundance (fish/100 m) and biomass (g/100 m) of wild rainbow trout and residual hatchery-reared steelhead in Jungle Creek (T_S), 1990 to 1993.

Discussion

Juvenile hatchery steelhead released into the Teanaway River system socially dominated preexisting wild trout in most instances, presumably because of their larger size or aggressive tendencies. The larger relative size of the hatchery steelhead may explain much of their dominance. Many researchers have documented size-related advantages in competition between stream-dwelling salmonids (Griffith 1972; Abbott et al. 1985; Chandler and Bjornn 1988; Huntingford et al. 1990; Hughes 1992). Alternatively, aggressive fish may dominate even in instances in which the subordinate fish is larger (Abbott et al. 1986). It may be that the rearing experience of the hatchery steelhead produced more aggressive fish (Moyle 1969; Fenderson and Carpenter 1971; Bachman 1984; Swain and Riddell 1990). Different stocks of salmonids may have different inherent aggressive tendencies that may be genetically influenced. All of the hatchery steelhead we used as broodstock, however, were of the same stock. Therefore, for our purposes, the genetic effects on aggressiveness of the hatchery steelhead we used can be discounted.

Despite the dominance of hatchery-reared steelhead over naturally-produced rainbow trout, negligible benefits (e.g. increased survival) may have been conferred to the hatchery steelhead. First, it was not apparent what, if any, resource was being competed for by the hatchery steelhead. Most interactions in which a hatchery steelhead was dominant did not result in the dominant fish achieving better access to any scarce resource (e.g. feeding microhabitats). Even though hatchery steelhead often displaced naturally-produced trout, hatchery steelhead rarely remained in positions the trout had occupied for more than one minute. Bachman (1984) reported that hatchery-reared brown trout wandered more and fed less than their wild counterparts. So, while trout may have been forced into less optimal positions (a potentially negative impact), hatchery steelhead did not appear to gain anything by the displacement. In contrast, most interactions we observed between naturally-produced trout did appear to involve benefits for the dominant fish, in terms of a superior feeding or holding location (Fausch 1984).

The types of agonistic interactions observed in the treatment and control streams in 1993 were quite different with respect to their apparent energetic costs. We classified interactions such as nips and butts as energetically costly and crowds, threats, and chases as less costly. This interpretation was justified because nips and butts involve physical contact and require more abrupt movements. The proportion of energetically costly interactions in treatment streams was double that in control streams. It is likely that interactions which require a great deal of energy, but which do not afford the victor better access to a limited resource, could reduce the growth and fitness (e.g. survival) of the fishes involved in those contests. In situations where residual hatchery steelhead are present in large numbers for prolonged periods of time (and they behave as the ones we have studied), the impacts of behavioral interactions on the growth of wild resident trout may be significant (see Chapter 8, this report). The different types of interactions observed in treatment and control streams may have influenced the amount of displacement we observed.

Displacement of subordinate fish within a channel unit occurred with twice the frequency in streams where hatchery steelhead were present than where they were not present. The impacts of these small-scale displacements are yet unclear. However, displacement from a preferred microhabitat may reduce food intake and consequently growth (Fausch 1984; Fausch and White 1986), or increase the susceptibility to predators and hence survival (Werner et al. 1983; Dill and Fraser 1984). We did not, however, document displacements from a stream reach or over a larger spatial scale. Thus, to date we have not observed the pied-piper behavior described by Hillman and Mullan (1989) in any year the hatchery steelhead have been released.

Interaction rates appeared to increase between the smolt outmigration period and the summer rearing period. This may have been due, in part, to the warmer water temperatures and smaller pool volumes during the summer. The interaction rate may also have been higher during the summer because the number of hatchery steelhead observed was lower after May. It also appeared that the interaction rates increased between years (from 1991 through 1993). While this may have in fact occurred, it is likely that the discrimination between different types of agonistic interactions in 1993 increased the number of total interactions that were recorded on data sheets in the field. Even if the number of interactions was recorded differently in 1993 than in the previous two years, the end result of the analyses involving year to year comparisons of the outcome of interactive contests was unaffected. We chose to focus most analyses of behavioral interactions on the outcome of agonistic contests (as opposed to individual interactions), as each contest may include multiple interactions of various types. The end result of each contest is what we felt reflected the greatest ecological importance. However, individual interaction types were important from the perspective of their energetic costs, as previously discussed.

Rainbow trout population abundance in the treatment streams did not appear to be impacted by the releases of hatchery steelhead. Population abundance of rainbow trout in the North Fork (T_L) and Middle Fork (C_L) of the Teanaway River showed a general downward trend from 1990 through 1993, while in the West Fork of the Teanaway River (R_L) it remained stable. With a presumed annual decrease in trout abundance in the treatment and control streams, and a stable population in a reference stream, it did not appear that the decline in the rainbow trout population in T_L could be attributed to the releases of hatchery-reared steelhead. Additional population information in these streams would provide more statistical power to examine the effects of releases of hatchery-reared steelhead on wild rainbow trout populations. Environmental factors may have affected trout abundance to a greater extent than density-dependent factors such as competition. In Jack Creek (C_S) it appeared that the most important factor controlling trout abundance was desiccation, not behavioral interactions. This creek was generally dry in the lower reach from late July through September.

Trout abundance in Jungle Creek (T_S) actually increased from 1992 to 1993, however this may have been related to differential spawning success between years (most of the trout captured in T_S during the population estimates were age 0+ in all years). The index site in T_S was very close to T_L and adult trout may have moved into T_S to spawn. Consequently, our population estimates in Jungle Creek may simply provide a measure of wild trout reproductive success and early rearing survival. If this is so, then reproductive success and early rearing survival do not appear to be adversely influenced by the presence of residual hatchery steelhead in Jungle Creek.

From the rainbow trout abundance data collected thus far, it appears that releases of hatchery steelhead have not adversely impacted wild rainbow trout abundance in the Teanaway drainage. Petrosky and Bjornn (1988) reported that population abundance of wild rainbow and cutthroat trout were not adversely impacted by the release of catchable-size (> 15 cm) rainbow trout in two streams in Idaho. Somewhat contrary to our findings and those of Petrosky and Bjornn (1988), Vincent (1987) found that when releases of catchable-size hatchery rainbow trout were discontinued in two Montana streams, the abundance of wild trout increased dramatically. He attributed the increase in abundance of wild trout to the removal of counterproductive behavioral interactions and competition between the hatchery and wild fish. Both Vincent (1987) and Petrosky and Bjornn (1988) examined the effects of releases of resident forms of hatchery-reared trout on their wild resident counterparts while we studied the effects of releases of an anadromous form of trout on resident trout populations. One could reasonably expect that the competitive effects would be greatest in cases where the hatchery and wild fish occupied overlapping environments for the longest period of time, thus favoring the detection of impacts in cases where resident or residual forms are evaluated.

Hatchery steelhead residuals did not appear to prey upon emergent age 0+ wild trout. Even though hatchery steelhead were collected in areas where trout fry were abundant, no fish were seen in over 50 residual steelhead stomachs. Martin et al. (1993) examined a total of 1,713 hatchery steelhead stomach samples in southeast Washington streams and found only three juvenile salmonids (spring chinook salmon). We suggest that predation by hatchery steelhead on wild trout fry was negligible in our treatment streams during the years of study.

Application of results from this study may not be directly transferable to interactions that may occur between fish produced as part of the YFP and wild fish. The hatchery-reared steelhead we released into Jungle Creek were produced at the Washington

Department of Wildlife's Yakima Hatchery. We realize that the fish we used may not behave identically to those produced in a YFP facility. However, because no YFP facilities have yet been constructed in the Yakima basin, we used the nearest available source of hatchery steelhead.

In summary, we observed hatchery-reared steelhead dominating wild trout in most of the interactions between those groups of fish. The behavioral interactions observed in the streams where hatchery steelhead were present more often resulted in the small-scale displacement of subordinate fish than those interactions observed in streams where hatchery steelhead were not present. No mid- or large-scale displacements were observed. Even though hatchery steelhead dominated and displaced wild trout in most instances, we cannot, at this time, attribute any declines in trout abundance to the releases of hatchery steelhead.

It is important to note that these results are preliminary and subject to further analysis and revision following completion of data collection in 1995. Final results will be presented in a future report or publication.

Acknowledgements

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Chapter 8

Examination of competition among hatchery-reared steelhead, naturally-produced rainbow trout, and spring chinook salmon using small enclosures in a natural stream

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Abstract

We investigated the effects of competition on fish growth to understand how releases of hatchery steelhead might affect the growth of preexisting naturally-produced salmonids. Using screened enclosures in a natural stream, we also examined mechanisms that may have affected fish growth such as food utilization, physiological stress, habitat use, and behavior. Three tests were performed; 1) effects of hatchery-reared steelhead on naturally-produced rainbow trout, 2) effects of naturally-produced spring chinook salmon on naturally-produced rainbow trout, and 3) effects of hatchery-reared steelhead on naturally-produced spring chinook salmon. Competition between hatchery-reared steelhead and naturally-produced rainbow trout negatively impacted rainbow trout growth. The presence of spring chinook salmon did not impact the growth of rainbow trout. Spring chinook salmon paired with hatchery steelhead did not grow at a significantly different rate than their unpaired counterparts in that test. Enclosures did not significantly reduce the amount of food available to the fish inside and the food habits of paired and unpaired fish did not differ greatly in mast cases. Cortisol levels, as a measure of physiological stress, did not differ between paired and unpaired fish. Cortisol levels in fish confined for 42 d were significantly lower than levels in fish outside the enclosures at the termination of the experiment. In situations where hatchery steelhead remain in freshwater with naturally-produced conspecifics for long periods of time, the growth of the naturally-produced fish may be negatively impacted.

Introduction

During the past decade concerns have increased regarding the potential for releases of fish from hatcheries to impact naturally-produced fish populations (Bachman 1984, Vincent 1987, Goodman 1990, Waples 1991). Releases of hatchery-reared salmonids into areas with preexisting populations of salmonids have been suggested to affect wild or naturally-produced fish through competitive interactions (Bachman 1984, Vincent 1987, Irvine and Bailey 1992). Competition between stream salmonids may occur when demand for either food or space exceeds availability (Chapman 1966).

Hatchery programs which release large numbers of fish increase the density of fish in certain areas for varying lengths of time. Competition for limited resources increases when fish density increases (Li and Brocksen 1977, Kennedy and Strange 1986, Heggenes 1988, Christiansen et al. 1992). Irvine and Bailey (1992) reported that hatchery coho salmon fry may have outcompeted naturally-produced coho salmon fry for supplemental food. Competition between hatchery-reared and naturally-produced salmonids has been suggested to affect the growth and abundance of the naturally-produced fish (Nickelson et al. 1986, Vincent 1987). Some research has shown relationships between social interactions and stress within salmonid species (Noakes and Leatherland 1977, Ejike and Schreck 1980).

Knowledge of competition between naturally-produced salmonids is more fully developed than understanding of competition between hatchery-reared and naturally-produced salmonids. Many studies have focused on the mechanisms of competition among salmonids such as agonistic interactions (Abbott et al. 1985, Huntingford et al. 1990, Hughes 1992) and niche separation (Griffith 1972, Hearn and Kynard 1986). For example, Everest and Chapman (1972) found that juvenile steelhead (*Oncorhynchus mykiss*) and chinook salmon (*O. tshawytscha*) utilized different habitats in streams and suggested that competition was limited. The anadromous (steelhead) and resident (rainbow trout) forms of *O. mykiss* represent polymorphisms that would be expected to be strong interactors if a common resource were limiting, because they have similar ecological requirements during a substantial part of their freshwater life history. Kennedy and Strange (1986) showed that fish of the same species compete strongly due to similarities in their requirements. The potential for hatchery-reared steelhead and resident rainbow trout to competitively interact was expected to be high.

It is also possible for specific age classes of different species of salmonids to compete for limited resources. Rose (1986) showed that brook trout (*Salvelinus fontinalis*) grew slower following the emergence of rainbow trout, and concluded that these two species may be competing during their first summer. Naturally-produced offspring of hatchery-reared spring chinook salmon could compete with naturally-produced rainbow trout if resource requirements of the two species were limiting. Therefore, we also examined potential impacts of naturally-produced spring chinook salmon on resident rainbow trout. Finally, we examined the possible competitive impacts of hatchery-reared steelhead smolts on naturally-produced juvenile chinook salmon.

The primary objectives of this experiment were to: 1) determine the extent of competition between specific groups of salmonids, 2) determine what the results of the competition were, and 3) determine what mechanisms were responsible for the competition observed. We wanted to quantify the competitive impacts of hatchery-reared steelhead residuals or naturally-produced juvenile spring chinook salmon on the growth and physiological status of resident rainbow trout. We also examined the potential impacts of increased density of residual hatchery steelhead on the growth and physiological status of naturally-produced spring chinook salmon. The results of these studies have implications for future hatchery or supplementation projects proposed for areas that currently have populations of naturally-produced salmonids.

Materials and Methods

Study Area

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

Criteria used for selection of the study reach included presence of both naturally-produced rainbow trout and past observations of hatchery steelhead that had not emigrated during the typical smolt outmigration period (residuals). Hatchery fish were identified by adipose fin clips. Natural production of steelhead in the study area was extremely low (McMichael et al. 1992). Naturally-produced resident rainbow trout are not visually discernable from juvenile steelhead, thus all naturally-produced *O. mykiss* were classified as resident rainbow trout. Other fish species observed in the study reach included, in relative order of decreasing abundance, shorthead sculpin (*Cottus confusus*), torrent sculpin (*C. rhotheus*), longnose dace (*Rhinichthys cataractae*), mountain whitefish (*Prosopium williamsoni*), bridgelip sucker (*Catostomus columbianus*), eastern brook trout, and spring chinook salmon. This reach was also selected because it overlapped with the area in which past observations of interactive behavior between rainbow trout and hatchery steelhead had been made (McMichael et al. 1992, Pearsons et al. 1993).

Experimental Design

Competition experiments between 1) hatchery steelhead and naturally-produced rainbow trout, 2) naturally-produced groups of spring chinook salmon and rainbow trout, and 3) hatchery steelhead and naturally-produced spring chinook salmon were performed using small enclosures from July 7 to August 19, 1993. Enclosures were constructed with 5 cm x 5 cm wood frame members and were enclosed with 0.95 cm square galvanized wire mesh. Inside dimensions of each enclosure were 91 cm high by 91 cm long and 99 cm wide. Each enclosure was divided into two equal-sized (0.46 m^2) chambers by a plywood barrier. Four large cobbles (20 to 30 cm diameter) were collected within the wetted stream channel and positioned in each chamber of each enclosure to simulate natural conditions and to provide substrate for benthic organisms. A plywood lid was attached to the top of each enclosure.

A total of 30 enclosure sites were selected on June 29, 1993, each of which was assigned randomly to either a pool or run habitat type in a depth 0.35 to 0.70 m, and in a velocity of 0.12 to 0.42 m/s. These criteria were developed from observations of fish-habitat relationships from prior sampling (McMichael et al. 1992). Enclosures were randomly placed at predetermined sites on July 6.

The experimental design required three fish to be placed in each enclosure according to treatment groups shown in Table 1.

Table 1. Experimental design for competition experiments in the North Fork of the Teanaway River, July 7 through August 19, 1993. Each test was replicated 10 times.

Test Number	Control/response	Treatment
1	rainbow trout	hatchery steelhead
2	rainbow trout	spring chinook
3	spring chinook	hatchery steelhead

As illustrated in Figure 1, a solitary fish (control) was placed in one chamber and treatment and response fish were placed in the other. The control and response fish were of the same species group for a given test, while the treatment fish was from a different species group. The combinations used in this experimental design were intended to ascertain effects on response fish. The terms control, response, and treatment fish will be used to distinguish between the different groups of fish in each test. The terms control and unpaired are used interchangeably as are response and paired. Each of the three tests was replicated 10 times.

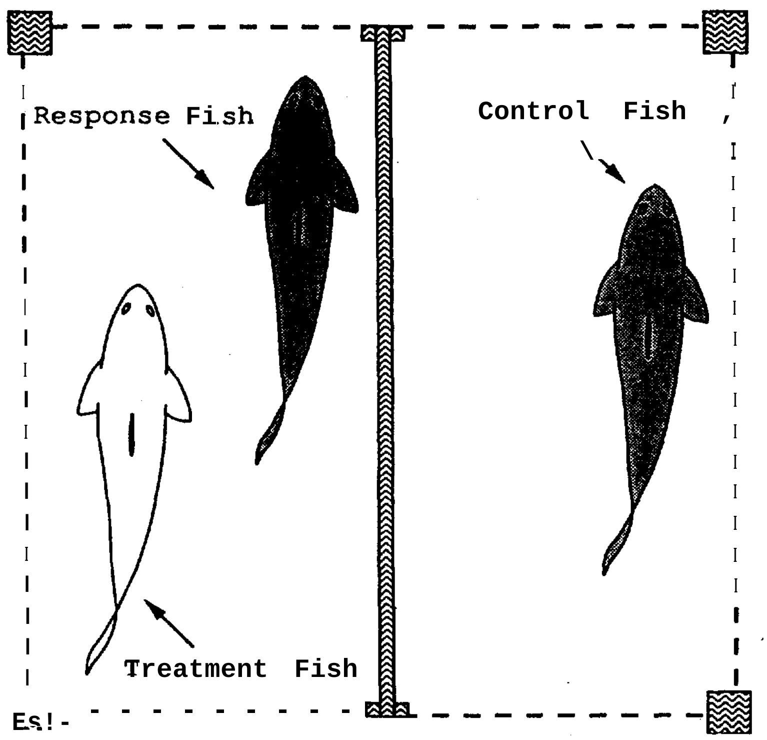


Figure 1. Example of fish placement for small enclosure competition experiments. The figure represents a top view of an enclosure. The control and response fish are of the same group whereas the treatment fish belongs to a different group (eg. test 1, control and response fish were rainbow trout and the treatment was a residual hatchery steelhead).

Fish used in this study were obtained both from the North Fork of the Teanaway River and the mainstem of the Yakima River. Naturally-produced rainbow trout between 100 and 150 mm fork length (FL) were collected on July 7 using battery-powered backpack electrofishers (settings: pulsed direct current (PDC) 300 V, 30 Hz and 300 V, 60 Hz) in the area immediately surrounding each enclosure. This range of trout length was targeted because it represented the modal length of trout (± 25 mm) previously observed in the study reach at that time of year (McMichael et al. 1992). The relative sizes of the groups of fish used in this experiment (Table 2) were those typically found during the summer rearing period in streams in the upper Yakima River basin. Though the trout used in this experiment were not individually aged, available age and size information from the North Fork of the Teanaway River suggest that trout between 100 and 150 mm FL are predominantly age 1+ and 2+ (Pearsons et al. 1993). This experiment was not designed to determine which species groups were the strongest competitors when fish sizes were equal. It was instead designed to determine if the presence of a treatment fish influenced the response fish.

Table 2. Relative fork length (mm) of fish groups at the beginning of the competition experiments. Mean length, standard deviation (SD), and range are shown for both groups in each test.

Test Number	Control/response			Treatment		
	Mean	SD	Range	Mean	SD	Range
1	115.7	13.6	101-143	169.4	25.1	140-204
2	123.8	13.3	106-149	67.4	4.2	63-77
3	73.1	8.3	64-92	155.9	38.4	117-213

On the day of collection, test fish were anesthetized (approximately 0.1 g/l MS-222), measured to the nearest mm FL, weighed to the nearest 0.1 g, and external appearance of each fish was noted. Each fish was associated with an enclosure and test number, species, use in control or treatment group, and the specific chamber of the enclosure it was placed in.

Age 0+ spring chinook salmon were not present in the immediate study area when this experiment began, necessitating their collection from the mainstem of the Yakima River. Test fish were collected near the town of Cle Elum, Washington on July 7 using backpack electrofishers (PDC, 300 V, 60 Hz and 400 V, 30 Hz). These fish were immediately transported in aerated vessels to the study area (approximately 30 km) where they were distributed into the appropriate enclosures (for test 3) in a manner consistent with that previously described for rainbow trout. The rainbow trout and spring chinook salmon control and response fish were then allowed to acclimate to, or establish 'prior residence' in, the enclosures for two days before the introduction of treatment fish. The experimental design utilized this period of prior residence in an attempt to approximate conditions that typically occur when fish are released from hatcheries into areas with preexisting fish. Prior residence has been shown to afford an advantage to stream salmonids in competitive situations (Allee 1982, Heggenes 1988, Metcalfe and Thorpe 1992).

Treatment fish were placed in the enclosures on July 9. Juvenile hatchery steelhead were collected from Jungle Creek, a tributary to the study stream, using backpack electrofishing gear (PDC, 300 V, 30 Hz) and placed into a holding vessel. Hatchery steelhead were then removed from the holding vessel and sampled following the protocol described for the other test fish. A hatchery steelhead was then placed in one of the chambers (assigned randomly) in each of the enclosures containing rainbow trout (test 1) and spring chinook salmon (test 3). Spring chinook salmon treatment fish for test 2 were sampled following the established protocol and placed in randomly assigned chambers of ten of the enclosures containing resident rainbow trout.

Two fish died within the first five days and were replaced with fish collected in the previously described manner. A dead spring chinook was replaced on July 8 and a dead rainbow trout was replaced on July 12. If an enclosure had suffered one or more mortalities by the end of the study period, it was discarded from the final analysis. This occurred in three of the replicates for tests 1 and 3, and in one of the replicates for test 2. Enclosures were cleaned of debris with a wire brush twice each week.

Attempts were made to observe behavioral interactions within the enclosures by snorkeling between 12:00 and 17:00 PDT on July 13 and 27. Each enclosure was observed once on both dates. Snorkelers entered the stream 10 to 20 m downstream of each enclosure and moved slowly upstream until they could observe the fish in the chamber containing the response and treatment fish. Observation periods averaged 7.5 min per enclosure and ranged from 3 to 16 min in duration. Interactive behavior, position of fishes in the water column, their association with substrate, and feeding activity were recorded.

On August 19, 42 days after the study was initiated, all fish were collected from the enclosures, euthanized in a lethal concentration (>200 mg/l) of MS-222, measured to the nearest mm FL, weighed to the nearest 0.1 g, examined for external anomalies, bled for physiological analyses, and examined for gonadal development. Capture protocol consisted of approaching an enclosure from the downstream side, quickly removing the plywood top and cobbles, and electrofishing within the enclosure with the same equipment and settings as were used in the initial collection of most fish (PDC 300 V, 60 Hz). The amount of time to capture fish within each enclosure averaged 1 min 54 s (range; 1 to 3 min).

To compare control and response rainbow trout and spring chinook salmon food habits we removed stomachs from all fish and preserved them in 10% buffered formalin at the termination of the experiment. The contents of 30 rainbow trout stomachs (15 control and 15 response) and 12 spring chinook salmon stomachs (6 control and 6 response) were examined. Binocular dissecting microscopes were used to identify food items to Order. Invertebrate head capsules were identified to Order and counted.

Between August 26 and 28, a test was conducted to determine whether the presence of the mesh screen may have influenced food availability within the enclosures. In a run downstream of a riffle in the middle of the study reach, screens (0.95 cm square galvanized wire mesh) were attached to 13 mm metal rebar that had been pounded into the substrate perpendicular to the water surface. These screens were then unattended for 48 h to simulate the typical debris load found on experimental enclosures. Drift nets (46 cm x 31 cm, 363 micron mesh size) were then attached to six screens, and six additional unscreened nets were also located along the same transect in an alternating pattern. Drift nets were deployed for 22 to 24 h. Samples were preserved in 70% isopropyl alcohol. Using a binocular dissecting microscope, we identified and enumerated insects in each Order observed.

We also examined potential enclosure effects on fish used in this experiment. Condition factors of fish inside and outside enclosures were compared at study termination to determine whether growth of fish inside was affected by confinement. Fish from inside the enclosures were collected on August 19 and those outside the enclosures were collected on August 20. Condition factors were calculated using the following standard equation: $K = W (100,000)/FL^3$, where K = condition factor, W = weight in g, and FL = fork length in mm). The stress physiology of fish inside and outside of the enclosures at the beginning and end of the study period was compared. Blood samples for stress physiology data were collected by severing each fish's caudal peduncle and collecting the blood in ammonium-heparinized capillary tubes. After centrifugation, the plasma was frozen for later analysis. Plasma cortisol levels were obtained by radioimmunoassay using the protocol developed by Foster and Dunn (1974) as modified by Redding and Schreck (1983).

Data Analyses

To test whether the presence of treatment fish affected the growth of response fish, one-tailed paired t-tests were performed on growth differences. To examine differences in growth between fish with and without developing gonads (unpaired rainbow trout from tests 1 and 2 combined) two-tailed two sample t-tests were performed. Physical characteristics such as sex, presence of lesions, and tissue damage on the nose of the fish were recorded binomially based on presence (1) or absence (0) of each condition. Logistic regressions were used to examine the relationships between growth and these characteristics. For stress physiology samples, paired t-tests were performed for differences among means. Sample distributions of cortisol levels were normalized by log transformation. Screened and unscreened drift samples were compared using two-tailed paired t-tests. Paired t-tests were also used to compare the numbers and types of food items ingested by control and response rainbow trout and spring chinook salmon. Condition factors were compared using a two-sample t-test. Ottenbacher's (1986) percent error (PE) method was applied to tests comparing multiple dependent variables to aid in interpretation of the results (PE = 100(c)(a)/M, where c = number of comparisons, a = alpha level, and M = number of comparisons that were found to be significant). Statistical power analyses (Peterman 1990) for t-tests involving control and response fish growth and physiological stress were performed to aid in the interpretation of these results.

Results

Competition between hatchery steelhead and naturally-produced rainbow trout appeared to be more intense than between age 0+ spring chinook salmon and rainbow trout and between hatchery steelhead and spring chinook salmon. Competition between hatchery steelhead and naturally-produced rainbow trout (test 1) negatively impacted growth of the rainbow trout. The unpaired rainbow trout (controls) in test 1 increased an average of 2.4 mm in length while the trout paired with hatchery steelhead lost an average of 1.6 mm (Table 3, Figure 2A).

The percent mean weight loss for the control group in test 1 was 8.9%, while the rainbow trout paired with hatchery steelhead in that test lost an average of 22.8% of their body weight (Table 3, Figure 2A). The difference in length and weight changes between control and response rainbow trout in test 1 were statistically significant (Table 4).

Table 3. Mean fork lengths (mm) and weights (g) of control (C) and response (R) fish at the start and end of the 42 day experimental period. Ampersands (&) denote groups of treatment fish. Total and percent differences in mean lengths and weights are shown. Number of replicates for test group were; test 1 = 7, test 2 = 9, test 3 = 7. Fish species (Spec.): RBT = rainbow trout or steelhead presmolts, SPC = juvenile spring chinook salmon, HSH = residual hatchery-reared steelhead (released as smolts).

			Start		End		Difference (%)	
Test	Spec.	C/R	Length	Weight	Length	Weight	Length	Weight
1	RBT	C	114.4	18.0	116.9	16.4	+2.4 (+2.2)	-1.6 (-8.9)
1	RBT	R	117.0	18.9	115.4	14.6	-1.6 (-1.4)	-4.4 (-22.8)
2	RBT	C	122.8	20.7	119.9	18.8	-2.9 (-2.4)	-1.8 (-9.2)
2	RBT	R	124.8	23.4	122.8	21.0	-1.9 (-1.6)	-2.4 (-10.3)
3	SPC	C	76.1	5.7	78.7	5.4	+2.6 (+3.4)	-0.3 (-5.3)
3	SPC	R	70.1	4.0	70.9	3.8	+0.8 (-1.1)	-0.2 (-5.0)
1	HSH	&	169.4	51.0	167.4	46.8	-2.0 (-1.9)	-4.2 (-9.0)
2	SPC	&	67.8	4.1	71.1	3.9	+3.3 (+4.9)	-0.2 (-5.1)
3	HSH	&	155.9	43.5	153.6	39.9	-2.3 (-1.5)	-3.6 (-9.0)

Table 4. Results of paired t-tests comparing changes in length and weight between control and response fish in three experimental groups. Species (RBT = rainbow trout, SPC = spring chinook salmon), sample size (N), degrees of freedom (df), t statistics (t), probability values (P), and power are shown for changes in length and weight. Asterisks denote significant differences (P<0.05).

		Length				Weight			
Test	Species	df	t	P	power	df	t	P	power
1	RBT	6	2.59	0.02*	0.16	6	2.07	0.04*	0.19
2	RBT	8	-0.42	0.66	0.11	8	0.49	0.32	0.20
3	SPC	6	1.20	0.14	0.16	6	-0.34	0.63	0.71

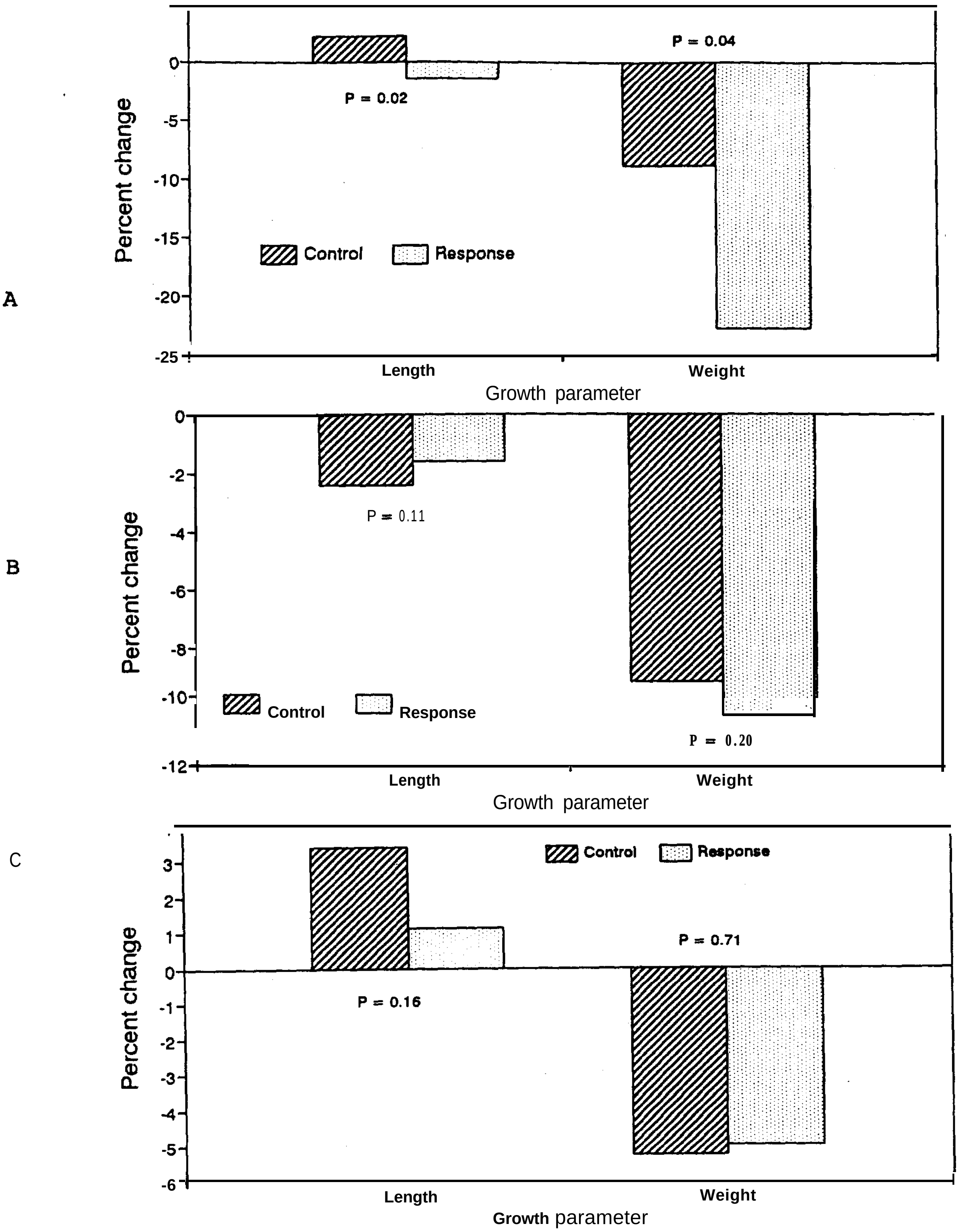


Figure 2. A. Percent change in body length and weight of control and response rainbow trout in a competition experiment with hatchery steelhead (test 1). B. Percent change in body length and weight of control and response rainbow trout in a competition experiment with spring chinook salmon (test 2). C. Percent change in body length and weight of control and response spring chinook salmon in a competition experiment with hatchery steelhead (test 3). P-values for paired t-tests are shown for control/response comparisons.

The treatment fish in test 1 decreased in length and weight (Table 3). Using Ottenbacher's (1986) PE method to help interpret the results revealed that 5% of the significant results found in this test may have been due to chance. The statistical power of the tests performed on length and weight change was relatively low, indicating a high probability of committing a type II error (Table 4).

Wild juvenile spring chinook salmon did not appear to negatively impact the growth of naturally-produced rainbow trout (test 2). Control rainbow trout lost, on average, slightly more length than the trout paired with spring chinook salmon (Table 3, Figure 2B). The difference in length change for this test was not significant (Table 4). Weight change between control and response trout in test 2 was also insignificant (Table 4, Figure 2B). Spring chinook salmon treatment fish, on average, increased in length while they lost weight (Table 3).

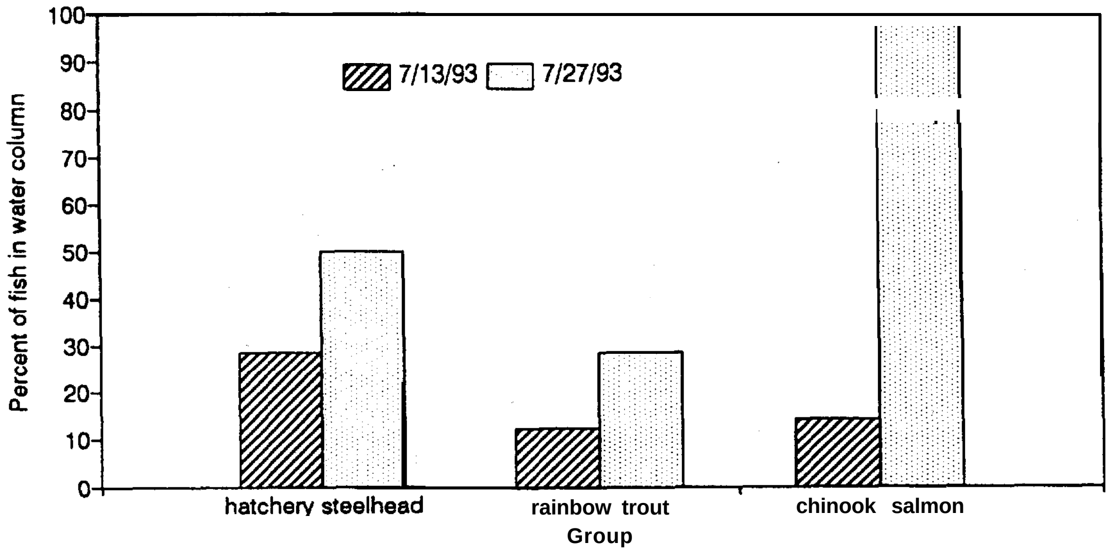
In test 3, spring chinook salmon paired with hatchery steelhead did not exhibit significantly different growth than their unpaired counterparts (Table 3, Figure 2C). Spring chinook salmon in the control group did, however increase in length (mean = +2.6 mm) more than the salmon in that test that were paired with hatchery steelhead (mean = +0.8 mm). Spring chinook in both control and response groups lost, on average, nearly equal amounts of weight (Table 3, Figure 2C). Differences in length and weight between control and response spring chinook in this test were, however, not statistically significant (Table 4). Hatchery steelhead treatment fish in this test showed average decreases in length and weight that were similar to the losses exhibited by treatment hatchery steelhead in test 1 (Table 3). Table 4 shows that the statistical power of the length change comparison for this test was low. This indicates that we may have accepted the hypothesis that hatchery steelhead did not affect spring chinook growth when we should have rejected it.

We suspected that intrinsic variables such as gonad development may have affected growth in unpaired (control) rainbow trout. Only two of nine (22%) trout in the control group for test 1 showed gonad development while seven of nine (78%) trout in the control group in test 2 had developing gonads. All trout with developing gonads were males. Only four of the 18 (22%) fish in these groups were females. Control trout in test 1 grew an average of 2.4 mm while control rainbow trout in test 2 lost an average of 2.9 mm in length. When all rainbow trout from both control groups were pooled, the mean length for trout without developing gonads increased an average of 0.7 mm while fish with developing gonads lost an average of 1.6 mm. Rainbow trout without developing gonads lost, on average, 1.5 g while those with developing gonads lost an average of 1.8 g. However, differences in length ($df = 9.2$, $t = +1.16$, $P = 0.27$) and weight ($df = 11.9$, $t = -0.29$, $P = 0.77$) changes between rainbow trout with and without developing gonads were not significant. No significant relationships were detected between presence or absence of visible abrasion of the nose (presumably resulting from contact with the enclosure screen), presence or absence of lesions, gender of the fish, and length and weight changes. Data were also recorded regarding missing scales and fin condition; however, no control rainbow trout were missing scales nor did any have apparent fin damage.

Underwater observations revealed that most of the fish observed 4 to 6 d after they were introduced into the enclosures occupied areas in or on the substrate, making them difficult to observe. Of the fish observed on July 13, more hatchery steelhead were found in the water column than either rainbow trout or spring chinook salmon (Figure 3A). On July 13 and 27 we observed the majority of the steelhead and trout in microhabitats in the interstices of the cobble substrate. All of the juvenile spring chinook salmon observed on July 27 occupied positions in the water column and did not appear to be directly associated with the substrate. Similarly, fewer fish appeared to be feeding after only 4 to 6 d in the enclosures than after 18 to 20 d (Figure 3B).

Spring chinook salmon appeared to utilize the water column more and feed more than either the hatchery steelhead or rainbow trout after 18 d in the enclosures. Only one agonistic bout was observed during the 450 min of underwater observations, whereby a hatchery steelhead chased and nipped a smaller trout twice, displacing the trout from a fixed location.

A



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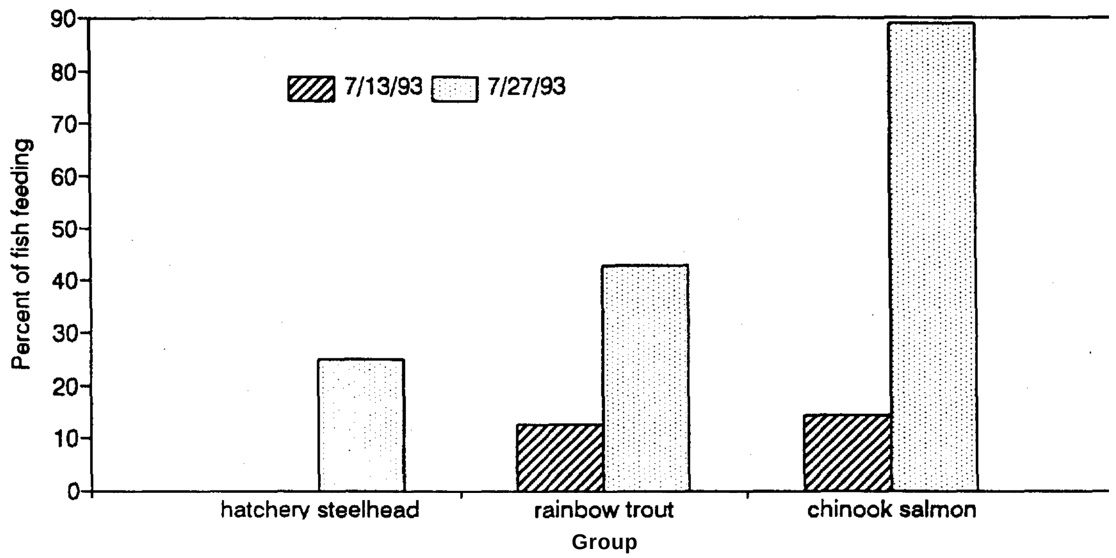

Figure 3. A. Percent of each fish group observed in the water column in experimental enclosures on July 13 and 27, 1993. HSH = hatchery steelhead, RBT = rainbow trout or steelhead parr, and SPC = spring chinook salmon. Sample sizes were, on July 13, HSH = 7, RBT = 8, SPC = 7, on July 27, HSH = 8, RBT = 14, SPC = 9. B. Percent of each fish group observed feeding in experimental enclosures on July 13 and 27, 1993. HSH = hatchery steelhead, RBT = rainbow trout or steelhead parr, and SPC = spring chinook salmon. Sample sizes were, on July 13, HSH = 7, RBT = 8, SPC = 7, on July 27, HSH = 8, RBT = 14, SPC = 9.

In all tests, food habits of control and response fish did not differ significantly with respect to the numbers of food items ingested (Table 5).

Table 5. Mean number of food items and mean number of Orders ingested by control and response fish in experimental enclosures. Ranges are shown in parentheses. Test number, group of fish (RBT = rainbow trout, SPC = spring chinook salmon), control (C) or response (R), and sample size (N) are shown for each group.

Test	Species	C/R	N	Number of Food items	Number of Orders
1	RBT	C	7	25.0 (6-107)	4.1 (3-5)
1	RBT	R	7	5.6 (1-15)	2.4 (1-4)
2	RBT	C	8	46.1 (12-148)	4.1 (2-6)
2	RBT	R	8	54.4 (13-183)	4.4 (2-6)
3	SPC	C	6	48.0 (19-68)	4.2 (3-5)
3	SPC	R	6	39.8 (1-76)	3.2 (1-6)

In test 1, numbers of food items ingested by control and response rainbow trout did not differ significantly (df = 6, t = 1.37, P = 0.22). However, control fish in test 1 did ingest a significantly greater number of Orders of insects than their paired counterparts (df = 6, t = 6.00, P = 0.001) (Table 5). Ottenbacher's (1986) PE test shows that there is a 10% chance that this result was due to chance. Rainbow trout control and response fish in test 2 did not display significant differences in either the total number of food items ingested (df = 7, t = -0.27, P = 0.80) or the total number of Orders ingested (df = 7, t = -0.45, P = 0.67). Diptera (75% adult, 25% larvae) made up about 70% of the total number of food items found in the 30 rainbow trout stomachs and over 80% in the 12 spring chinook stomachs that were examined (Table 6).

Table 6. Total numbers and percents (in parentheses) of food items, listed by Order, found in screened and unscreened drift samples (N = 6 of each) and rainbow trout (RBT; N = 30) and spring chinook salmon (SPC; N = 12) stomachs.

	Drift samples (%)		Stomachs (%)	
Order	screened	unscreened	RBT	SPC
Ephemeroptera	492 (33.5)	100 (8.1)	132 (12.9)	79 (12.5)
Plecoptera	10 (0.7)	10 (0.8)	2 (0.2)	2 (0.3)
Diptera	752 (51.3)	781 (63.5)	709 (69.4)	511 (81.0)
Trichoptera	136 (9.3)	233 (19.0)	41 (4.0)	3 (0.5)
Coleoptera	8 (0.5)	7 (0.6)	16 (1.6)	2 (0.3)
Hemiptera	7 (0.5)	5 (0.4)	14 (1.4)	5 (0.8)
Hymenoptera	6 (0.4)	10 (0.8)	70 (6.9)	15 (2.4)
Neuroptera	0 (0.0)	1 (0.1)	0 (0.0)	0 (0.0)
Odonata	0 (0.0)	1 (0.1)	1 (0.1)	0 (0.0)
Collembola	3 (0.2)	3 (0.2)	0 (0.0)	0 (0.0)
Hydracarina	50 (3.4)	77 (6.3)	35 (3.4)	13 (2.1)
Crustacea	0 (0.0)	0 (0.0)	1 (0.1)	0 (0.0)
Lepidoptera	2 (0.1)	0 (0.0)	0 (0.0)	0 (0.0)
Araneae	1 (0.1)	1 (0.1)	0 (0.0)	1 (0.1)
Total	1467	1229	1021	631

Ephemeroptera, Hymenoptera, Trichoptera, and Hydracarina were also numerous. Trichoptera made up a larger percentage of the items eaten by trout than by spring chinook salmon. Hymenoptera adults (terrestrials) appeared with greater frequency in stomach samples than in drift samples (Table 6). This suggests that trout may have preferentially selected insects of this Order. Parasitic Nematodes were found in eleven of the stomachs, but were omitted from the analyses because they were not considered food items. Paired and unpaired spring chinook salmon did not ingest significantly different numbers of items (df = 5, t =

0.64, P = 0.55) or numbers of Orders (df = 5, t = 1.22, P = 0.28).

The presence of mesh screen on the experimental enclosures did not affect the number of food items available to the fish inside but did appear to affect the relative occurrence of two Orders. The total number of food items in unscreened drift samples (mean = 204.8 items, range 117 - 387) were similar to those in screened samples (mean = 244.5 items, range = 31 - 416) (df = 5, t = 0.50, P = 0.64) (Table 6). Furthermore, unscreened and screened samples contained similar diversity of insect Orders (unscreened mean = 7.3, range = 6 - 9, screened mean = 7.2, range = 4 - 10)(df = 5, t = -0.15, P = 0.88). Unscreened samples contained a higher proportion of Trichoptera larvae while the screened samples included a higher percentage of Ephemeroptera nymphs (Table 6). Diptera was the most abundant Order in both unscreened and screened drift samples, making up 63.5 and 51.3 percent of the total number of insects respectively (Table 6). In contrast to the large number of adult Diptera in stomach samples, nearly all (98%) of the Diptera in the drift samples were larvae. Ephemeroptera, Trichoptera, and Hydracarina were the other Orders of insects that were most abundant in the drift samples.

It did not appear that the enclosures had an effect on the condition factor of rainbow trout. The condition factors of trout inside the enclosures were not significantly different from those of trout captured outside the enclosures at the termination of the experiment (df = 23, t = -1.01, P = 0.32).

Circulating cortisol titers did not differ significantly between control and response fishes (Figure 4A, Table 7). Rainbow trout that were confined for 42 d had significantly lower circulating levels of cortisol than rainbow trout captured outside the enclosures (Figure 4B, Table 7).

A

B

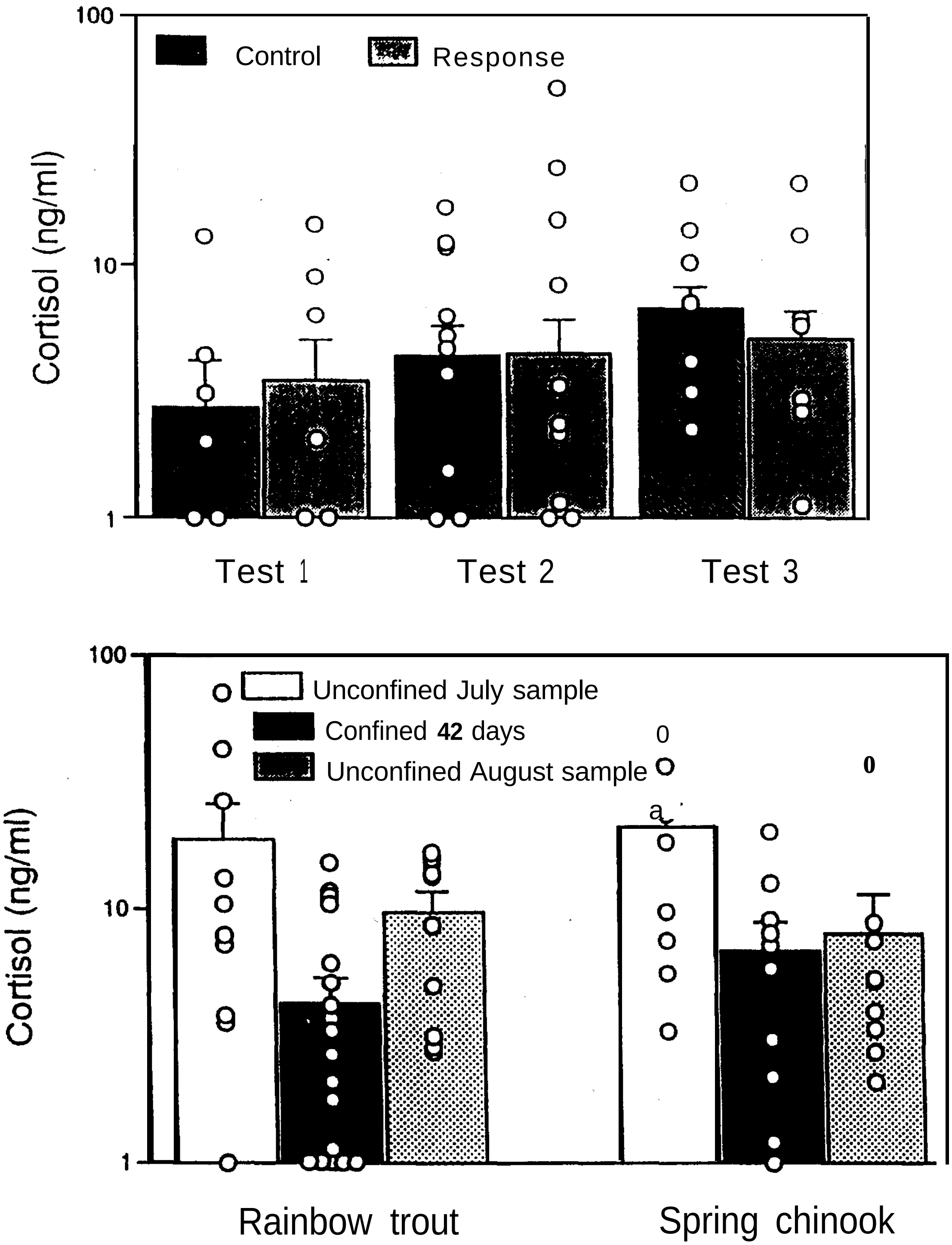

Figure 4. A. Mean cortisol levels of control and response fish for test 1 (rainbow trout without and with hatchery steelhead), test 2 (rainbow trout without and with spring chinook salmon), and test 3 (spring chinook salmon without and with hatchery steelhead). B. Mean cortisol levels in unconfined and confined rainbow trout and spring chinook salmon. Lines represent + 1 standard error of the mean. Individual data points are provided. Y-axes are on a log scale.

Table 7. Results of paired t-tests comparing cortisol levels between control and response fish in three experimental groups. Fish species (RBT = rainbow trout, SPC = spring chinook salmon), sample size (N), degrees of freedom (df), t statistics (t), probability values (P), and power are shown. Test 4 refers to two-sample t-test results for comparisons between cortisol levels in rainbow trout captured outside the enclosures and control rainbow trout from within the enclosures at the termination of the experiment. Asterisk denotes a significant difference ($P<0.05$).

Test	Species	N	df	t	P	power
1	RBT	6	5	-0.36	0.74	0.90
2	RBT	10	9	-0.10	0.92	1.00
3	SPC	7	6	0.69	0.51	0.99
4	RBT	29	24.3	-2.77	0.01*	1.00

Discussion

Our results suggest that releases of hatchery-reared steelhead may adversely affect naturally-produced resident rainbow trout, whose growth may be reduced during the summer. Increased densities of hatchery-reared steelhead may force these two groups of fish to inhabit similar areas for long periods of time, which may increase the potential for competitive impacts. Three years of underwater observations showed that hatchery steelhead and rainbow trout occupied similar habitat types and engaged in agonistic interactions (McMichael et al. 1992, Pearsons et al. 1993). It could be argued that a reduction in size, due to slower growth during the summer, could decrease the over-winter survival and reproductive success of the rainbow trout individuals and provide a mechanism for a negative impact on population size (Cunjak et al. 1987). Small differences in size (a weight advantage of 5% or more) have been shown to assure dominant status for larger salmonids (Abbott et al. 1985). In addition, these dominant fish are known to exhibit greater mobility and feeding success than smaller subordinate fish (Helfrich et al. 1982).

The largest differences in growth between control and response fish were seen in the two tests (1 and 3) in which the treatment fish were considerably larger than the response fish. This is consistent with existing literature on competition among salmonids in which larger fish typically dominate smaller fish (Griffith 1972, Abbott et al. 1985, Chandler and Bjornn 1988, Huntingford et al. 1990, Hughes 1992). The dominance of a fish in a chamber may have increased its feeding success (Helfrich et al. 1982) and enabled it to grow faster and more efficiently (Li and Brocksen 1977) than the smaller subordinate fish. In cases where difference in fish size is very large, competitive impacts may be reduced by differential habitat use of large and small fish (Everest and Chapman 1972). It is still possible, however, for fish occupying different habitats in streams to compete for food. Food items that drift downstream may be intercepted by a fish in a riffle precluding its ingestion by a fish in a pool downstream.

The only significant differences between control and response fish growth were seen in the test in which the treatment and response fish were conspecifics (*O. mykiss*, test 1). It would be expected that fish of the same species might compete to a greater degree than fish of different species for a limited resource due to the similarities in their ecological requirements when at similar life stages (Allee 1982, Kennedy and Strange 1986).

There are at least three primary explanations for our failure to detect significant growth impacts in the tests involving spring chinook salmon (tests 2 and 3). First, competition may not have occurred because spring chinook occupied different niches than the rainbow trout (test 2) and hatchery steelhead (test 3) (Everest and Chapman 1972). The spring chinook salmon we observed on July 27 were generally higher in the water column than either the trout or the steelhead. Secondly, the sample sizes we used in these tests were smaller than would have been statistically preferable given variation in our results. The resulting power of our tests was relatively low, thereby increasing the chance of making a type II error. The fact that we found a significant difference in rainbow trout growth when they were associated with hatchery steelhead, despite the low power of that test, suggests that the biological impact must have been rather substantial. Finally, the selection of size classes used in these experiments may have reduced the potential for chinook salmon to impact rainbow trout. It may be that age 0+ spring chinook salmon and age 0+ rainbow trout would be more likely to compete for limited resources than age (size) classes that were used in this experiment.

To better understand the effects of artificially produced fish on preexisting salmonid populations, it is important to concentrate on life stages where competition would be most likely. It is probable that the earlier emerging fish would have an initial size and prior residence advantage (and thus probably a competitive advantage) for at least the first few weeks after emergence (Everest and Chapman 1972, Fausch and White 1986, Heggeness 1988). Spring chinook salmon generally emerge several weeks earlier than rainbow trout (e.g. the upper Yakima River basin (McMichael, pers. obs.)). It would be during the time immediately following trout emergence that the effect of competition might be greatest. The increased density of naturally-produced offspring resulting from a hatchery program designed to restore or increase natural production may have the greatest potential to negatively impact trout growth. Therefore, efforts should be concentrated on examining the competitive effects of interactions between age 0+ salmon and age 0+ rainbow trout to better discern the potential effects on resident rainbow trout growth. For greatest applicability of the results, such tests should be conducted during all seasons and in a variety of habitats when and where these species would be expected to share resources.

Though we did not detect a significant difference in the growth of maturing and non-maturing control rainbow trout, gonad maturation may have influenced the results of test 2. Rowe and Thorpe (1990) found that maturing male Atlantic salmon (*Salmo salar*) parr fed and grew less than their non-maturing counterparts during the summer rearing season. Therefore, the state of gonadal development of anadromous salmonids should be considered when designing comparative growth studies involving those species.

Factors related to the enclosures may have influenced our results to some extent by affecting fish behavior and movement. Restriction of movement may have decreased the probability of detecting a competitive effect by altering the fishes' behavior to the extent that they spent much of their time in interstitial spaces between the cobbles. Spatial isolation may have precluded the possibility for competitive interactions. The fish inside the enclosures were unable to move as much as fish outside the enclosures. Fish outside the enclosures were able to adjust positions in response to water temperature fluctuations, changes in the intensity of sunlight, and the periodicity of insect emergence and drift.

The physiological samples indicated that the stress levels in fish inside the enclosures at the termination of the experiment were significantly lower than those of fish outside the enclosures. This may, in part, be explained if the fish inside the enclosures experienced less stressful conditions with respect to availability of overhead cover (the plywood top) and protection from the risk of predation. However, stress levels may have been high before the test fish had habituated to their new environment.

The mesh size we used (0.95 cm) was considerably larger than the mesh sizes that Cooper et al. (1990) found to significantly influence the immigration and emigration of invertebrates in enclosures in trout streams. Our results are consistent with theirs, whereby mesh size did not affect invertebrate movement into enclosures when the screens were uncleaned for 24 to 48 h.

The enclosures may have decreased our ability to detect an impact; however, we believe that the overall effect of the enclosures on our findings was minimal.

The application of these results to hatchery or supplementation strategies fall into two general categories; 1) assessment of the potential impacts of hatchery releases on preexisting fish populations, and 2) alternatives in the operational aspects of hatchery management that affect the size of fish released as well as the timing of releases. If hatchery-reared salmonids compete with preexisting naturally-produced fish to the detriment of the latter, decreased productivity could result. In areas such as the northwest United States, where many naturally-produced stocks of salmonids are at critically low levels (Nehlsen et al. 1991), the interactive effects caused by hatchery-produced fish may be serious enough to be deemed unacceptable. Our study suggests that the species and size and of the hatchery fish released influences the potential for competitive impacts. In cases where hatchery-reared fish are larger than their naturally-produced conspecifics, the impacts would be expected to be greatest. It is conceivable however, that very large size differences could reduce competitive impacts through habitat partitioning. Hatchery release practices that minimize spatial and temporal overlap would be expected to have the least impact on preexisting populations. For instance, in areas or times where large numbers of hatchery steelhead smolts are released but high rates of residualism occur, the impacts of these residual fish on preexisting rainbow trout could be acute. Hatchery steelhead are typically larger and will occupy similar habitats as the resident rainbow trout for considerable periods of time. Where most hatchery fish emigrate quickly the short-term impacts on resident rainbow trout growth would be expected to be relatively minor.

The hatchery steelhead we used in this experiment were produced using traditional hatchery technology. They were however, reared at much lower density than is typical for hatchery steelhead (about 30% of traditional permissible loading density (Piper et al. 1983)). Therefore, the results of our experiments are most applicable to traditional hatchery operations. Our findings do however, provide useful information about impacts that might be expected from releases of fish from hatchery facilities intended to increase natural production of target species (e.g. Supplementation project discussed by Clune and Dauble (1991)). These results should be applicable to supplementation programs unless fish from supplementation facilities are found to behave and interact differently after release than fish produced in traditional hatcheries.

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Chapter 9

Effects of parentage, rearing density, and size at release of hatchery-reared steelhead smolts on smolt quality and post-release performance in natural streams

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Abstract

The performance of hatchery-produced salmonids after release into streams is often inferior to that of naturally-produced (wild) salmonids. Several factors may be responsible for this difference. I examined some possible effects of parentage, rearing density, and size at release on hatchery-reared summer steelhead post-release performance in the Yakima River basin in central Washington between 1991 and 1993. Measures of performance that I examined were in-river emigration rates, rates of precocialism, and incidence of residualism. In contrast to general expectations, preliminary results suggest that the offspring of naturally-produced (wild) parents did not perform as well in natural streams as offspring of parents that had been reared for one year in a hatchery environment. Selection for emigration at age 1+ may have occurred rather quickly for Yakima basin hatchery-reared steelhead. Regardless of parentage, smolts reared at the lowest density appeared to successfully emigrate as far as Prosser Dam (242 km downstream of release site) at a much higher rate than smolts reared at traditional densities. The smaller sized smolts (1993) emigrated at higher rates and exhibited lower incidences of precocialism and residualism than larger smolts that were released in 1991 and 1992. In 1991, when the smolts having the largest mean length were released, the incidence of precocial males was the highest of the three years examined. The greatest numbers of residual steelhead were observed in 1991 when the largest smolts were released. The smolts released in 1991 were offspring of exclusively wild Yakima River summer steelhead. Fewer residuals were present when offspring of only hatchery-origin adult steelhead were released at a smaller mean size. Because multiple variables changed each year it is difficult to distinguish among their influences due to the possible interrelationships among variables.

Introduction

Survival rates of hatchery-reared salmonids have typically been lower than those of their naturally-produced (wild; wild and naturally-produced will be used interchangeably hereafter) counterparts (e.g., Ersbak and Haase 1983; Hume and Parkinson 1987). Many factors may contribute to this difference. The hatchery rearing experience has been identified as a major reason that hatchery fish do not perform as well as wild fish in natural environments (Wiley et al. 1993). Wiley et al. (1993) identified factors that may influence the post-release survival of hatchery-reared salmonids. These factors were; unnaturally high rearing densities, a constant environment, lack of cover and predators, and food at regular intervals. Rearing density in hatchery vessels has been shown to influence salmonid behavior (Brown et al. 1992) and post-release survival (Mazur and Iwama 1993; Banks 1994).

In this study several effects of different mating and rearing protocols for hatchery steelhead (*Oncorhynchus mykiss*) on smolt quality and in-stream post-release performance were examined. It was not possible to control variables such as parentage, rearing density, and size at release of hatchery steelhead smolts between years in this study. It would have been preferable to keep these treatment variables constant to better meet our original objectives. The primary purpose for the activities that allowed us to conduct this study was the actual production of test fish to be used to examine the effects of releases of hatchery steelhead smolts on wild rainbow trout in natural streams (see Chapter 7). Therefore, although not the product of a rigorous experimental design, the opportunity existed to examine several possible effects of these variables on smolt quality and post-release in-river performance of hatchery steelhead smolts. The results of this research are primarily a qualitative description rather than quantitative analyses. This report presents results from three of four consecutive smolt releases. The results of this work should be considered preliminary. More complete analyses of these data will be presented following the 1994 release of hatchery steelhead and the final year of data collection (1995) on this aspect of the research.

Methods

In general, study methods consisted of collecting adult steelhead from the Yakima River for use as broodstock, monitoring the offspring of those fish (juvenile hatchery steelhead) in the hatchery prior to release, releasing the juvenile steelhead into a stream, and evaluating the performance of the juvenile steelhead in the field using trapping, snorkeling, and electrofishing techniques (McMichael et al. 1992). Treatment variables that were examined were; parentage (hatchery-origin versus naturally-produced broodstock), rearing density, and size at release of hatchery-produced steelhead smolts. Measures of fish performance in the natural streams (response variables) were: estimated numbers of hatchery steelhead that emigrated from the study area, the percentages of fish that were sexually mature (males) at the time of release, and estimated numbers of these fish that did not emigrate (became residuals) from the study area. Releases of fish were conducted each year between 1991 and 1993. Specific methods are presented in more detail by McMichael et al. (1992) and in Chapter 7 of this report.

Hatchery steelhead smolts used in these experiments were offspring of adult summer steelhead that were collected at an adult trapping facility (denile-type) in the Yakima River at Prosser Dam (rkm 74) and by angling in the Yakima and Naches rivers (McMichael et al. 1992; Pearsons et al. 1993). Our intentions were to use only hatchery-origin adult steelhead for broodstock to minimize impacts on the wild portion of the run, however, agency policy, run size, and/or trap limitations forced us to use some wild steelhead to produce the smolts released in 1990 and 1992. Hatchery-origin steelhead had been fin marked (clipped adipose fin) as juveniles and were the offspring of wild Yakima River summer steelhead parents. Only naturally-produced steelhead were used as broodstock in the WDFW Yakima Hatchery program from 1986 through 1990 (J. Cummins, WDFW, personal communication). Thus, steelhead used as broodstock in 1990 (BY1990) were all naturally-produced steelhead collected at Prosser Dam. Due to low numbers of returning hatchery steelhead in late 1990 and early 1991, angling for wild steelhead was used in 1991 to augment the number of broodstock collected at Prosser Dam. Thus, hatchery-origin and naturally-produced steelhead were used for broodstock in BY1991. Angling efforts in early 1991 were conducted in the Yakima River and Naches River, upstream of Union Gap, to reduce the possibility of capturing Satus Creek steelhead (a genetically distinct stock within the Yakima basin (PSR 1994)). During only one year, BY1992, were enough hatchery origin steelhead collected to produce the number of smolts necessary to meet objectives for the work presented in chapter 7. No naturally-produced steelhead were used as broodstock that year. For all brood years, the resultant smolts were released approximately 12 to 18 months after the parental groups were collected. For example, the smolts released in 1992 were offspring of adult steelhead captured in BY1991.

Hatchery rearing was conducted by WDFW personnel at the Yakima Hatchery generally according to statewide agency standards. Adult steelhead were collected between September and April and then held in round concrete ponds at the Yakima Hatchery until fish began to exhibit external coloration and morphology typical of spawning steelhead. When steelhead began showing external spawning coloration (mid-January), all adults were examined for spawning readiness once per week and gametes were collected from each ripe female and fertilized with milt from two or three ripe males following methods described by Piper et al. (1983). Eggs collected during each week were held in isolation in the hatchery building until results from pathological examinations were returned. Groups of eggs infected with Infectious Hematopoietic Necrosis (IHN) were destroyed (one confirmed case occurred in BY1991). Weekly gamete collection continued until the eggs from the last female available were fertilized (typically in mid-April).

Incubation, hatching and early rearing of juvenile steelhead took place at the Yakima Hatchery. Steelhead fry were reared in indoor troughs (mean water depth was 15 cm, flow ranged from 26 to 38 l/min) at the Yakima Hatchery until they averaged approximately 50 mm in fork length. They were then transferred to concrete outdoor round ponds (mean depth was 91 cm, flow ranged from 208 to 378 l/min) at that facility. Estimated rearing densities at each stage of development were not available. In late winter or early spring, most steelhead were transferred to a spring-fed wooden raceway at Nelson Springs for the final three months of rearing. The Nelson Springs raceway was 61 m long, 2.9 m wide, had a mean depth of 79 cm, and the flow was approximately 8,700 l/min. Smaller fish (less than about 120 mm) were retained at the Yakima Hatchery in an attempt to expedite their growth to the smolt size (about 175 mm, 50 g) targeted by WDFW. In 1991, an attempt was made to hold one group of fish on a slower growth regime for experimental purposes (McMichael et al. 1992). The fish were sorted by size and handled in a manner similar to other traditional steelhead hatcheries operated by WDFW and other agencies in the Northwest. Facility limitations (inadequate rearing chamber barriers) prevented separation between the test groups. Both size groups were released in 1991. Fish released in 1992 were also made up of two different size groups. In 1992, fish for the first release and half of the second release were reared for the final three months at Nelson Springs. Fish for half of the second release and the entire third release in 1992 were reared at the Yakima Hatchery. In 1992, the fish from the Yakima Hatchery were smaller than the fish from Nelson Springs. All fish released in 1993 were reared for the final three months at Nelson Springs.

Loading densities were very different for the hatchery steelhead released in each of the three years. The fish released in 1991 were reared near the traditional permissible loading density (Piper et al. 1983). Permissible loading density is the accepted maximum weight of fish that may be reared in a given vessel with a certain water flow (kg of fish/l/min). The fish released in 1992 were reared at less than half the density of the 1991 release group. The fish released in 1993 were reared at approximately 30% of the 1991 density. In 1991, approximately 500 to 700 large (estimated lengths 300 to 400 mm) hatchery rainbow trout were present in the Nelson Springs raceway during the time that the hatchery steelhead were rearing there.

For the purposes of other studies, steelhead smolts were released into Jungle Creek in a manner intended to roughly mimic the outmigration periodicity and magnitude that might be expected given volitional emigration opportunity from an acclimation pond in that creek (McMichael et al. 1992). Original plans for steelhead supplementation under the YKFP called for an acclimation pond capable of holding 33,000 steelhead smolts adjacent to lower Jungle Creek. The expected target number of fish to be released from that site each year was 33,000. Thus, this target was used for the present study. Each year of the study, the first release was scheduled to occur on the first Monday in May, the second two days later, and the final release following ten days after the first. This release timing was intended to correspond to that expected for naturally-produced steelhead smolts in the study area (McMichael et al. 1992). Approximately 45% of the fish were slated for release in the first group, 33% in the second, and the final 22% in the third.

At the time of release into Jungle Creek steelhead smolts were examined for external physical characteristics (indicating degree of smoltification), sexual maturity (precociousness), and length and weight. In 1991, subsamples (N = 50) of fish from the first and third release groups were sacrificed, measured, weighed, sexed, and inspected for sexual maturity. Degree of smoltification was not directly assessed in 1991. In 1992 and 1993, fish from each release group (N = 50) were measured, weighed, examined for sexual maturity, visually categorized as either smolts *or* non-smolts, and released into Jungle Creek (McMichael et al. 1992). The steelhead released in 1993 were subsampled periodically through the final four months of rearing for two physiological indicators of smolt quality by the National Marine Fisheries Service (NMFS). These subsamples (N = 30) were measured, weighed, visually examined for external smoltification characteristics, and dissected to obtain blood for determination of thyroxin levels (T3 and T4).

Outmigrant traps were used to enumerate hatchery steelhead as well as naturally-produced salmonids emigrating from the North Fork of the Teanaway River and Jungle Creek. Trapping was only conducted for portions of the smolt outmigration period in 1991. For more information on the specific types of traps used see McMichael et al. (1992) and Chapter 7 of this report. In 1992 and 1993, traps were deployed for the majority of the smolt outmigration period. Fish traps (traversing fyke in 1991, rotary screw in 1992 and 1993) in the North Fork of the Teanaway River provided only a sample of all emigrating fish, while "V-weir" traps in Jungle and Jack creeks captured 100% of the outmigrants. Capture efficiency estimates were calculated for the traps used in the North Fork of the Teanaway River. These estimates have an unknown degree of variance with environmental variables, thus outmigration estimates presented for the North Fork of the Teanaway River should be viewed with caution.

Juvenile hatchery steelhead that were present after June 1 were defined as residuals. Most steelhead smolts initiate their seaward migration from headwater rearing areas before June 1 (Wagner 1968; Evenson and Ewing 1992). Hatchery steelhead residuals were enumerated in sites throughout the study area.

Outmigration estimates, population estimates (see Chapter 4 for details on sites and methods), and underwater observation rates (from snorkeling, see Chapter 7 for specific methods) were used to examine the abundance and distribution of residual hatchery steelhead during 1991, 1992, and 1993.

Results

The numbers and origin (hatchery or naturally-produced) of summer steelhead adults used to produce the smolts for 1991, 1992, and 1993 releases varied greatly between years (Table 1). Smolts released in 1991 were progeny of exclusively naturally-produced adults; adults used to produce the smolts for the 1992 releases were approximately two-thirds hatchery origin and one-third naturally-produced origin; and the broodstock used to produce the smolts released in 1993 were all of hatchery origin.

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

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

The loading density of the hatchery steelhead smolts decreased each year between 1991 and 1993 (Table 1). The smolts released in 1991 were reared at approximately 92% the traditionally accepted maximum loading density for summer steelhead (Piper et al. 1983; S. Roberts, WDFW, pers. comm.). In 1992 and 1993, the smolts were reared at about 38% and 32% of the traditionally accepted maximum loading densities, respectively. In 1991, the several hundred larger rainbow trout that were reared with the hatchery steelhead at Nelson Springs were released into Jungle Creek with the steelhead smolts. It is possible that the actual number of hatchery steelhead reared and released was overestimated due to the inclusion of the larger hatchery rainbow trout.

Total numbers, sizes, and degree of smoltification varied among the three years hatchery steelhead were released into Jungle Creek. The mean size of the hatchery steelhead appeared to decrease each year while the percentage classified as smolts increased (Table 2). In addition, the percentage of precocial males was highest in 1991 and decreased each year thereafter (Table 2). Mean condition factors of smolts released during the three years were higher in 1992 and 1993 than in 1991.

In 1993, physiological sampling of smolts indicated that the hatchery steelhead were released at the appropriate time. Sharp increases were seen in mean thyroxin (T3 and T4) levels (ng/ml) in the two weeks prior to the first release (Figure 1A). Figure 1B shows that mean fork length (mm) and the mean degree of smoltification based on external appearance also increased prior to the initial release date.

Table 2. Number released, mean fork length (mm, ± SD), mean weight (g, ± SD), mean condition factor (CF), percent classified as smolts, and percent precocial males for sampled hatchery steelhead released into Jungle Creek from 1991 to 1993. Sample sizes (N) are also presented.

Year	Number Released	Number Sampled	Mean Length	Mean Weight	Mean CF	% Smolts	%Precocial males
1991	31,542	100	201 (± 16)	81 (± 25)	0.98	< 50[a]	4.0
1992	38,000	200	196 (± 16)	78 (± 22)	1.01	72 to 76[b]	1.0
1993	22,500	150	182 (± 21)	64 (± 23)	1.02	92 to 100[b]	0.7

[a] Smolt quality was not assessed directly, however most fish released did **not** exhibit typical external characteristics of steelhead smolts (Wedemeyer et al. 1980; Ewing et al. 1984).
[b] This is the range among the samples from the three different release dates within years.

A

B

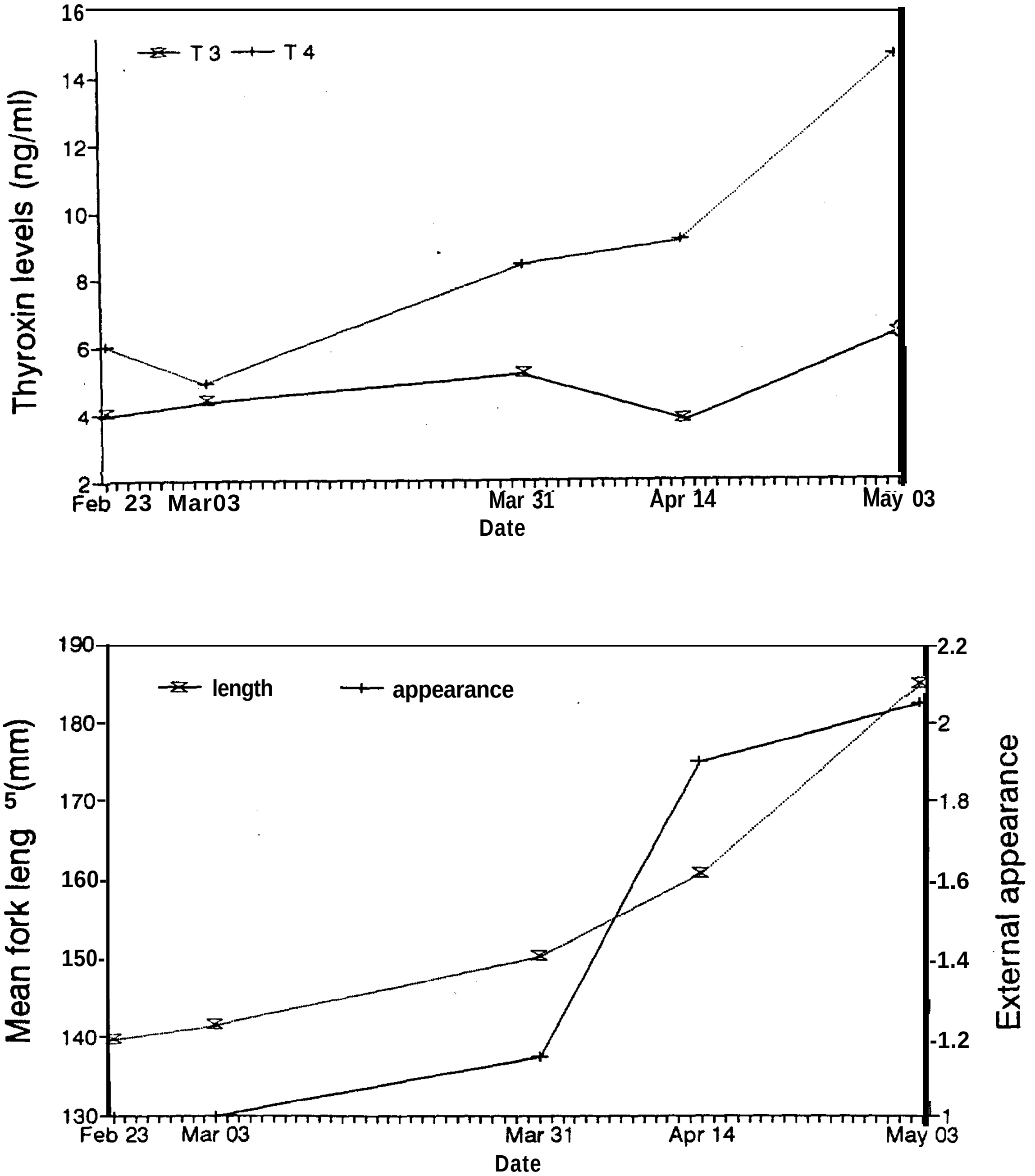

Figure 1.A. Mean thyroxin levels (ng/ml) (T3 and T4) of hatchery steelhead in the Nelson Springs raceway between February 23 and May 3, 1993. B. Mean fork length (mm) and the mean degree of smoltification based on external appearance (scored on a scale of 1 to 3, 1 being not a smolt, 2 being transitional, and 3 being a smolt) of hatchery steelhead sampled from the Nelson Springs raceway between February 23 and May 3, 1993. The first release of fish occurred on May 2, 1993. Twenty fish were sampled on each date labeled on the x axis (Data from D. Larsen, NMFS).

In 1991, large numbers of hatchery steelhead (visually estimated to number over 5,000) were still present in Jungle Creek up to one month after the final release. Many of these fish moved upstream until they reached a waterfall over 2 km upstream from the release site. An outmigrant trap was operated at the mouth of Jungle Creek between May 29 and June 13, 1991, which captured 53 hatchery steelhead that did not appear to be smolts, based on external appearance (McMichael et al. 1992). Many (26%) of the hatchery steelhead captured in this trap in 1991 were precocial males (McMichael et al. 1992). In 1992, the same trap was operated from the date of the third release until August 12, during which time 407 hatchery steelhead were captured (Pearsons et al. 1993). This trap was not operated during the first two releases in 1992. In 1993, we operated the trap from the date of first release (with continual sampling 24 h/day for the first 3 days) through July 13 and captured 19,954 hatchery steelhead smolts. Of 79 hatchery steelhead examined for sexual maturity at this trap in May of 1993, 5 (6%) were sexually mature males.

Emigration rates and timing from the release stream (Jungle Creek) varied widely among years. The hatchery steelhead released in 1991 migrated out of the North Fork of the Teanaway River over a longer period of time and in lower proportions (relative to the number released) when compared to those released in 1992 and 1993 (Figure 2). An estimated 64% of the smolts released in 1991 emigrated out of the North Fork of the Teanaway (11 km) prior to June 1. In 1992 and 1993, emigration estimates were 65% and 71%, respectively.

The Yakima Indian Nation (YIN) enumerated steelhead smolts that passed Prosser Dam. In 1991, a total of 1,781 hatchery steelhead smolts were estimated to have migrated downstream past Prosser Dam during May, June, and July (McMichael et al. 1992). Scale-age information collected by the YIN (M. Kohn, pers. comm.) that year indicated that only 36% (648) of the hatchery fish passing Prosser in 1991 were migrating as age 1+ fish (Table 3). The age 2+ and older hatchery steelhead smolts detected at Prosser Dam were not fish released as a part of these experiments. Instead, they had been released into the Naches River at least a year earlier by WDFW as part of the then ongoing steelhead production program. An estimated 36% of the hatchery steelhead smolts emigrating past Prosser Dam in 1991 were age 1+; 62% were age 2+; and 2% were age 3+. Importantly, all hatchery steelhead released in the Yakima basin above Prosser Dam were released as age 1+ fish. Unfortunately, we were unable to determine the age structure of these fish migrating past Prosser Dam in 1992 and 1993.

Only about 2% of the hatchery steelhead smolts that were released into Jungle Creek passed Prosser Dam in 1991. In 1992, the number of hatchery steelhead smolts estimated to have emigrated past Prosser Dam was similar to the previous year (Table 3). In 1993, however, there was over a ten-fold increase in the number of hatchery steelhead smolts emigrating past

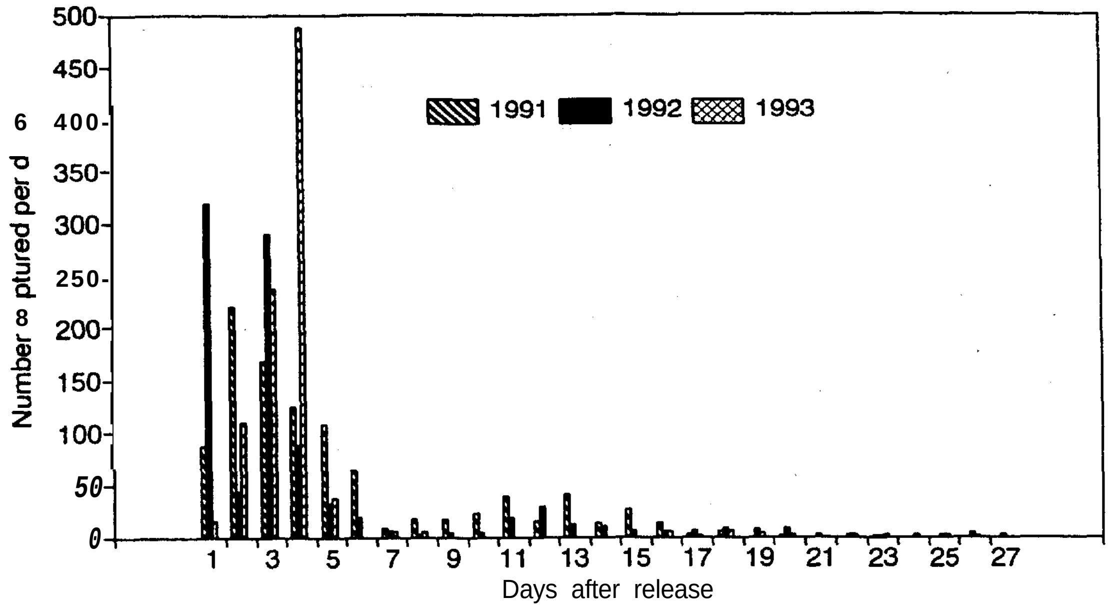

Figure 2. Daily number of hatchery steelhead captured emigrating from the North Fork of the Teanaway River during May of 1991, 1992, and 1993.

Table 3. Number of hatchery steelhead estimated to have passed Prosser Dam during May, June, and July in 1991, 1992, and 1993. Data are expressed as estimated total number, and the percentage of the number released into Jungle Creek that were estimated to have passed Prosser Dam within 2 months of their release (M. Kohn and M. Johnston, YIN, pers. comm.).

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

Prosser Dam over the previous two years. It is possible that some of the smolts passing Prosser Dam in 1992 may have been released in 1991 or 1992 and that fish passing Prosser Dam in

1993 could have been released in 1991, 1992, or 1993. Age 1+ hatchery steelhead detected at Prosser Dam in 1992 and 1993 were all from our releases.

Large percentages of hatchery steelhead smolts released into Jungle Creek did not migrate out of the North Fork of the Teanaway River prior to June 1 of each year. Similar to Wagner (1968), hatchery steelhead that did not exhibit a seaward migration prior to June 1 were defined to be residuals. In 1991, many residual hatchery steelhead were removed from Jungle Creek by sport anglers (McMichael et al. 1992). Numbers of residual steelhead captured in population estimate index sites (see Chapter 5) and observed in underwater observation sites (see Chapter 7) were substantially lower in 1992 than in 1991, and even fewer still in 1993. Very few fish that did not emigrate the year they were released were captured (via trapping or electrofishing) in the Teanaway basin the following year. Many juvenile hatchery steelhead were captured during data collection efforts related to other aspects of research, as well as by anglers fishing in the Yakima Canyon in the area between the cities of Ellensburg and Yakima (McMichael et al. 1992). In 1992, using outmigration trapping methods, 20 hatchery steelhead that were released in 1991 were estimated to have emigrated from the North Fork of the Teanaway River. Similarly, in 1993, an estimated 50 hatchery steelhead from the 1992 releases were estimated to have emigrated from the North Fork of the Teanaway River. Therefore, not all hatchery steelhead that failed to emigrate the year they were released became non-migratory "resident" rainbow trout of hatchery origin. Many of the hatchery steelhead that migrated out of the North Fork of the Teanaway may have reared in the mainstem Yakima River for one or more additional years prior to seaward emigration. As previously mentioned, many of the hatchery steelhead smolts captured at Prosser Dam were age 2+ or older is consistent with this hypothesis.

The number of residual hatchery steelhead observed (per unit time) during behavioral sampling in the North Fork of the Teanaway River showed that residual hatchery steelhead were most abundant in 1991 and least abundant in 1993 (Figure 3). Observations in 1992 were generally intermediate, especially for the late summer period. These data are supported by the emigration estimates, in which the lowest percentage of migrating hatchery steelhead was observed in 1991.

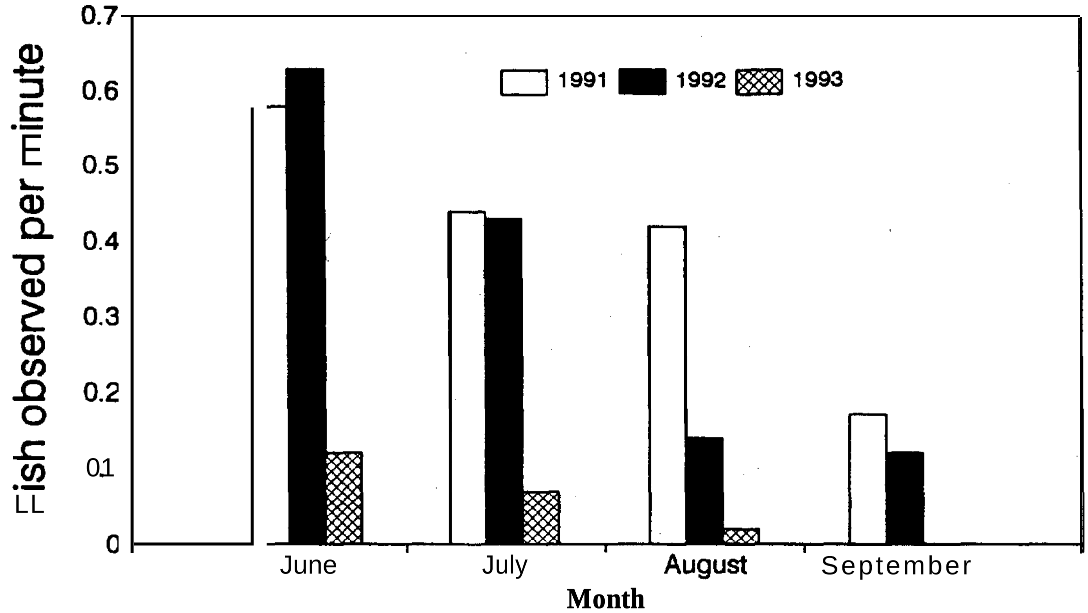

Figure 3. Snorkeling observation rates (fish/min) of residual hatchery steelhead in the North Fork of the Teanaway River during 1991, 1992, and 1993.

Discussion

Observed differences in post-release smoltification and behavioral characteristics for hatchery steelhead between years may be related to: 1) parentage, 2) rearing density, 3) size of fish at release, and 4) timing of the releases. From this study, it is not possible to fully separate influences among these often interdependent factors. Nevertheless trends were observed that qualitatively illustrated relationships between performance of hatchery steelhead smolts and the above factors.

Preliminary results suggest that using traditional hatchery practices, steelhead smolts resulting from naturally-produced parents had lower smolt quality than, and did not perform as well as, hatchery steelhead smolts resulting from artificially-produced parents (F_1 hatchery fish). Under completely natural conditions steelhead smolts typically do not emigrate to sea until they are two or three years old (Withler 1966; Randall et al. 1987). From data collected at Prosser Dam, naturally-produced steelhead emigrate from the Yakima basin as smolts primarily at age 2+ and age 3+ at a length of 130 to 210 mm (J. Hubble, YIN, pers. comm.). As shown in this study, despite hatchery steelhead smolts being of similar size to the naturally-produced steelhead smolts at the time of their release, some hatchery steelhead do not migrate until they are age 2+ or 3+, regardless of size at release.

The tendency for some stream salmonids to migrate may be genetically controlled (Jonsson 1982; Skaala and Naevdal 1989; Northcote 1992). Jonsson (1985), however, reported that environmental conditions that affect growth can alter the degree of residency expressed. When hatchery fish (F_1 offspring of naturally-produced fish) were used as broodstock, smolt quality and performance (as determined by emigration rate, magnitude, and in-stream survival) of their age 1+ smolts were superior to smolt quality and performance of offspring resulting from wild parents. Thus, it would appear that selection for hatchery steelhead smolts that emigrate at age 1+ may occur very rapidly. In the present study, offspring of adult steelhead that had been through one generation in a hatchery (BY1992) outperformed smolts that resulted from naturally-produced adults (BY1990) and progeny of a mixture of hatchery-origin and naturally-produced adults (BY1991). This finding suggests that to successfully increase the number of returning adult steelhead to the Yakima basin by using hatchery propagation, it may be best to collect gametes only from hatchery fish that have successfully returned from the sea following their release as age 1+ smolts (i.e. marked adults). However, this recommendation does not take into account the genetic or ecological consequences of such an action.

Different stocks of steelhead released into the Yakima River have experienced different post-release survival rates. Hatchery steelhead smolt passage from release points to Prosser Dam was higher when the WDW released Skamania stock (a hatchery summer-run strain originating from the Washougal and Klickitat rivers) steelhead smolts. Mean post-release survival of Skamania stock steelhead from the Naches River to Prosser was approximately 30% (range 19% to 41%) for 1983 to 1986. Mean post-release smolt survival after local origin wild adults were used as broodstock (1987 to 1990) was 18% (range 9% to 26%).

In the present study, smolt size appeared to be inversely related to post-release performance. The length of the smaller smolts released was closer to that of the naturally-produced steelhead smolts emigrating from the Teanaway drainage (McMichael et al. 1992; Pearsons et al. 1993). The fish with the higher condition factors appeared to perform better than those with lower condition factors. This finding is contrary to findings of Tipping (J. Tipping, WDFW, personal communication) who found that hatchery steelhead smolts with a mean length of 190 mm and a mean condition factor between 0.90 and 0.99 performed better than other groups he tested. It is probable, however, that because many variables for the current study changed annually (e.g. parentage and rearing density) the results regarding the effects of condition factor on post-release performance may have been due to chance.

Traditional hatchery practices and agency fish cultural standards often have a target size for production of hatchery steelhead smolts that is based largely on cost effectiveness as it pertains to smolt production goals. These targets are generally independent of fish age, though most hatcheries release smolts at age 1+. To efficiently achieve steelhead smolt size (generally over 175 mm FL) in one year, juvenile steelhead are often reared in water that is warmer than water in the surrounding streams (where wild steelhead may be rearing) and are fed maximum food rations. This forcing of fish growth has often been unsuccessful in producing age 1+ smolts that exhibit appropriate post-release behavior and survive to return as adults.

Traditional rearing strategies have been shown to produce high percentages of precocious males (Bjornn and Ringe 1984). In this study larger percentages of sexually mature male steelhead were observed in the groups released in 1991, than in the groups released in 1992 and 1993. Fish released in 1991 were reared at the highest density and had the largest mean size of the fish released during the study. Bjornn and Ringe (1984), in their work with hatchery steelhead in Idaho, found that larger hatchery steelhead smolts showed a greater tendency to become precocious males. Prevost et al. (1992) showed that large size after the first year of life favored sexual maturation in hatchery Atlantic salmon (*Salmo salar*). McClay et al. (1992) reported that sexual maturity for Atlantic salmon reared at lower densities was less likely to occur early than for those reared at higher densities. They attributed this tendency to increased growth rates in treatments where the fish were reared at lower density (McClay et al. 1992). Peven (1990) stated that early maturation due to elevated growth rates during critical periods of development in hatcheries may have profound management implications. Hatchery or naturally-produced steelhead smolts that become sexually mature in freshwater are unlikely to undertake a seaward migration until a subsequent year, if at all, and therefore are less likely to return as adult steelhead. Precocious male steelhead may interbreed with naturally-produced rainbow trout (Pearsons et al. 1993). The genetic and behavioral consequences of interbreeding between resident and precocious anadromous forms of *O. mykiss* could increase the proportion of these 'hybrid' fish that emigrate (Moring and Buchannan 1978; Northcote 1992; Moring 1993). Any emigration of fish that might otherwise be resident trout could result in a loss to the resident trout population in the upper Yakima basin.

In the present study, hatchery steelhead that were reared at the highest density (1991: normal hatchery densities) exhibited a lower degree of smoltification, a higher proportion of precocial males, and a greater incidence of residualism than those that were reared at lower densities (1992 and 1993). Though this study did not determine the number or relative return rate for adult steelhead returning from any of the releases, the effects of rearing density on the in-river performance appeared to be consistent with the findings of Banks (1994). Banks (1994) found that chinook salmon (*Oncorhynchus tshawytscha*) reared at lower densities had greater post-release survival than their counterparts that were reared at higher densities. He reported that a three-fold increase in smolt rearing density did not result in an increase in the number of returning adults (Banks 1994). Fagerlund et al. (1981) reported that stress and associated mortality due to crowding in hatchery coho salmon (*O. kisutch*) were higher in fish reared at high densities than in fish reared at lower densities. Mazur and Iwama (1993) reported that chinook salmon reared at lower densities had significantly longer survival times (in freshwater aquaria) than chinook salmon held at higher densities. Even though only three consecutive brood years of hatchery steelhead were released in this study, and all were reared at different densities, it appears that the lowest rearing densities (fish released in 1993) were associated with the highest in-stream survival.

Hatchery steelhead (smolts) in-river passage varied greatly between 1991 and 1993. Estimated survival of hatchery steelhead smolts to Prosser Dam in 1993 appeared to be about ten times higher than in the previous two years. However, in 1992 and 1993, it was not possible to distinguish between hatchery steelhead smolts released in different years. For example, age 1+ smolts released in 1993 and age 2+ (released in 1992), or age 3+ (released in 1991) were included in counts of emigrating hatchery smolts at Prosser Dam in 1993. Therefore, some of the hatchery steelhead smolts passing Prosser Dam in 1993 may have actually been released in 1991 or 1992, and may have reared in the Yakima basin for an additional year or two prior to being observed in 1993 while emigrating to sea. Assuming similar smolt-to-adult survival for hatchery steelhead for the 1991 and 1993 releases (hatchery smolt-to-adult survival at Prosser Dam from 1987 to 1990 is estimated to have been 0.07%; calculated by the author from data provided by J. Hubble, YIN and J. Cummins, WDFW), less than one adult hatchery steelhead would have returned as a result of the 1991 releases, while four would have returned from the releases in 1993, even though an additional 10,000 smolts were released in 1991.

Based on these preliminary findings and within the constraints of a limited experimental design, it appears that hatchery practices could be altered to improve the in-stream post-release performance of hatchery-reared steelhead smolts in the Yakima basin. Some practices that could produce steelhead smolts that have better survival rates within the Yakima basin include: (1) use of hatchery-origin (marked fish that have experienced at least one generation of hatchery rearing) adult steelhead for broodstock, (2) production of relatively small smolts (1993 mean length was 182 mm FL), and (3) rearing of fish at low density (less than half of traditionally accepted loading densities). With only three years of data it is difficult to confidently elucidate the possible interrelationships among these variables. Analyses following the final year of experimental hatchery smolt releases in the Yakima basin in 1994 should improve understanding of some of these interrelationships.

In areas where hatchery steelhead are released that also have pre-existing naturally-produced steelhead populations, two management objectives typically exist. One objective is to increase the number of returning adult steelhead (to increase natural production and/or harvest). Another objective is to minimize the impacts of the hatchery program on pre-existing fish populations such that the first objective is compromised or otherwise cannot be met. Hatchery steelhead smolts that emigrate quickly, do not residualize to a great extent, and do not become sexually mature in freshwater (precocial males) are the most desirable hatchery products to meet the above objectives. Potential ecological impacts of hatchery steelhead that do not emigrate on naturally-produced salmonids may be a great concern in areas where naturally-producing populations are at critically low levels (see previous chapter, this report).

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General Discussion

Implications for Monitoring Planning

The information that has been derived from the research presented in this report provides an opportunity to contribute to the development of a rainbow trout monitoring plan. Many factors may contribute to variation in the life history of rainbow trout and the structure of fish assemblages associated with rainbow trout. Two of these factors appear to be particularly important. Preliminary analyses suggest that, of the variables measured, elevation and stream size appeared to explain the most variation in rainbow trout spawn timing, size-at-age, population density, movement distance, and assemblage structure of species associated with rainbow trout. The peak of rainbow trout spawn timing occurred early in the spring at low elevations and was progressively later at high elevations (Chapter 1). Trout spawning at high elevations were generally shorter than trout spawning in lower elevation areas (Chapter 5). In addition, rainbow trout that spawned in tributaries did so at shorter sizes and younger ages than in mainstem areas and trout that reared in tributaries were generally shorter at age than trout in the mainstem (Chapter 5). The areal density of rainbow trout in tributary streams was negatively correlated with elevation in every year, however the relationship was statistically significant only in 1991 (Chapter 4). In addition, areal density of rainbow trout was much lower in the mainstem than in tributary streams (Chapter 4). Three assemblage types could be characterized based on elevation and stream size (Chapter 6). One assemblage type occupied mainstem areas which were relatively low in elevation and large in size (elevation 1430-1960 m, discharge 7.301-29.432 m^3/s, stream width 33.8-56.6 m), another in relatively low elevation, small sized tributaries (elevation 1540-2040 m, discharge 0.001-0.010 m^3/s, stream width 1.81-3.94 m) and another in tributaries that were relatively high elevation and small size (elevation 2040-3620 m, discharge 0.002-0.713 m^3/s, stream width 2.66-9.32 m). Other unmeasured variables; such as food abundance and water quality, may also influence biological attributes of rainbow trout, but were beyond the scope of our study.

Because elevation and stream size appeared to influence a variety of important variables, monitoring of variables that reflect the status of rainbow trout should account for site elevation and stream size. Furthermore, variables that are influenced by elevation, such as stream temperature, or are correlated with elevation, such as distance from a source of fish colonists, should be measured. In order to assess the status of the rainbow trout population following supplementation of anadromous salmonids in the Yakima basin, a monitoring plan for non-target species such as rainbow trout needs to be developed.

Effective monitoring of the rainbow trout population in the upper mainstem Yakima River must be conducted at appropriate spatial and temporal scales. The spatial scale of sampling should include the home-range of the population of interest (Connell and Sousa 1983, Grossman et al. 1990). Sampling at scales smaller than the home-range, increases the probability that population variability is related to movement of fish rather than fluctuations in population size (Grossman et al. 1990). Furthermore, sampling should be conducted for at least the amount of time that it takes for all individuals within the population to complete their life cycle (Connell and Sousa 1983). The maximum life span of the oldest individual is the time that it takes for a population to turnover. Sampling for this length of time will reduce the probability that population stability is related to low adult mortality in addition to long adult life spans and limited recruitment (Frank 1968). The spatial and temporal scale criteria outlined above can be applied to monitoring plan development for rainbow trout in the upper Yakima River, and may also be applicable to other areas where similar conditions exist.

The spatial scale necessary for monitoring the rainbow trout population that receives the most angler interest comprises the upper mainstem of the Yakima River from Roza Dam to Easton Dam. This portion of the river provides a high quality trout fishery and has the catch and release angling regulation throughout it's length. Movement distances of rainbow trout in this area may be considerable, necessitating a large spatial scale for population monitoring. Movement distances ranged from 0 to 149 km in the Yakima River (Chapter 3). However, most (59%) movement distances of rainbow trout were less than 5 km and 74% were less than 10 km. Movements of rainbow trout between sections of the mainstem may partially explain the variation of fish densities observed in individual mainstem sections (Chapter 4). Genetic data also supports the selection of the upper mainstem Yakima River as an appropriate spatial scale for monitoring.

Rainbow trout collected from the mainstem of the Yakima River between Roza and Easton dams were genetically similar (Appendix 1). These genetic data suggest that rainbow trout in the upper mainstem breed with each other and should be monitored as a large unit. Monitoring the rainbow trout population in the upper Yakima River might be accomplished by sampling or subsampling the river between Roza and Easton Dams at times when rainbow trout movement distances are least.

Sampling of trout abundance should occur during September and October to minimize the effects of fish movement and to maximize the precision and accuracy of population estimates. Fish moved less from June through December than during other times of the year (Chapter 3). In addition, it appeared that few fish larger than 174 mm moved between tributaries and the mainstem during this time period. In contrast, movements during the spring are frequently associated with spawning migrations and some fish ascended tributaries to spawn (Chapters 1 and 3). Following spawning, adult rainbow trout generally returned to mainstem areas. In addition, small rainbow trout (< 175 mm) also moved considerably during the spring (Chapter 3). Therefore, sampling in the mainstem should occur during September and October if the flow regimes are managed in ways similar to those currently used. Flows are typically high during the summer in upper mainstem areas until the middle of September. Capture efficiency of rainbow trout has been relatively higher at low flows than at high ones. Thus, sampling the abundance of trout should occur during the fall so that population estimates are most accurate and precise.

At least five years of abundance data should be collected prior to supplementation to accomodate for trout that live at least five years and to increase the statistical power necessary to detect changes. Few rainbow trout in the mainstem areas studied lived longer than five years (Chapter 5) and hence a complete life cycle of some of the oldest trout is about five years long. Furthermore, larger sample sizes available from sampling over multiple years would increase statistical power and thus improve our ability to detect changes in the trout population following supplementation (Peterman 1990).

Although a well designed monitoring plan will help managers assess impacts to a population of interest, strategies to avoid undesirable interactions might also be used to minimize the likelihood of undesirable adverse impacts occurring.

Strategies to avoid undesirable interactions

The original intent for most hatcheries built in the Columbia River Basin was to mitigate for losses of anadromous fish production caused by degradation or elimination of habitat important to fishes. Unfortunately, one unexpected consequence of releasing large numbers of fish from hatcheries was their impact on wild fish populations (Waples 1991). Many studies have demonstrated the potential for releases of fish from hatcheries to negatively impact wild stocks (Nickelson et al. 1986, Vincent 1987, Pearsons et al. 1993). Furthermore, many wild fish stocks are in severe need of protection because of their low population size (Nehlsen et al. 1991). Thus, hatcheries must be managed in ways that minimize negative impacts to wild fish to aid wild stocks recovery.

The future of hatchery management might include provisions for minimizing interactions with wild fish in addition to producing fish for harvest and natural production. Furthermore, hatchery success might be judged as a combination of fish production attributes and minimization of undesirable interactions (Figure 1).

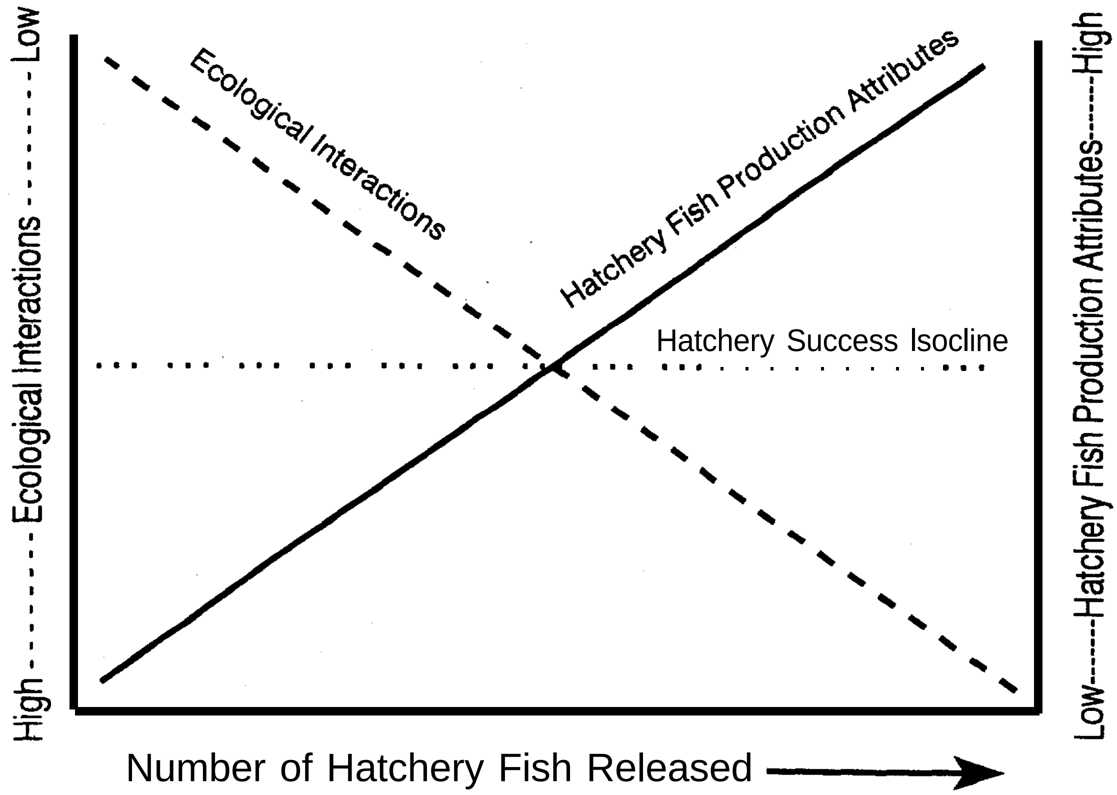

Figure 1. Hypothesized interaction between hatchery fish production attributes and ecological interactions, and a proposed indicator of hatchery success. Hatchery management strategies might vary according to sympatric wild fish populations. For example to the left of the x-axis might represent areas with species of concern (eg. sensitive or rare species), at the middle of the x-axis areas with productive wild fish populations, and at the right of the x-axis hybridized populations and exotic fishes. Diagram assumes that interactions with wild fish does not reduce hatchery fish production attributes.

Results from species interactions research (Chapters 7,8,9 of this report) have provided a foundation to describe methods that might be used to minimize undesirable interactions between hatchery and wild fish. Unfortunately, many of the methods that might be used to minimize undesirable interactions will also decrease the success of hatchery programs (as measured simply by numbers of returning adults). In other words, there will be trade-offs between minimizing undesirable interactions and traditional measures of success of hatchery programs (Figure 1). Despite the possible impracticality of some methods, a description of numerous methods that might be used to reduce or eliminate negative interactions between target and non-target fish is presented here. Where possible, the description of strategies to minimize interactions will be coupled with clarification of potential detriments to hatchery program success measures.

Two classes of methods to minimize interactions can be described: 1) minimize spatial and temporal overlap of hatchery and wild fish, and 2) minimize the ability of hatchery fish to behaviorally dominate wild fish. Minimizing spatial and temporal overlap can be accomplished at a variety of spatial and temporal scales. Overlap at large scales can be reduced by releasing fish far from populations of wild fish or releasing fish at times when wild fish are not present. Overlap at small scales can be reduced by minimizing the time hatchery fish are in sympatry with wild fish. This might be accomplished by producing fish that migrate to the ocean immediately upon release, reducing the number that residualize, and/or releasing hatchery fish at times when wild fish are relatively inactive. The fish used as broodstock, the rearing density, and the size of fish at release may influence residualism and subsequent impacts on pre-existing wild fish. Our preliminary results suggest that fish surviving the hatchery experience and returning as adults might make better broodstock, in some respects, than wild fish (Chapter 9). These fish migrated at a faster rate and residualized less than progeny of wild broodstock (Chapter 9). Fish reared at lower densities also appeared to perform better than fish reared at higher densities. In addition, fish that were large at the time of release residualized more than those that were small at release (Chapter 9). Additional methods that might be used to increase the rate at which hatchery fish migrate and achieve concommitant decreases in residualism, include volitional release of smolts (although for contrary evidence see Evenson and Ewing 1992), with physiological measures indicating when fish should be allowed to enter the stream and proportions of precocial males indicating when fish should no longer be allowed to enter the stream (Viola and Schuck, in press). Direct stream releases might be conducted when environmental conditions are suitable for fish migration, such as high discharges, high turbidity, and appropriate water temperatures. However, minimizing spatial and temporal overlap is not always possible so alternative methods to minimize interactions need to be identified.

Where hatchery and wild fish exist in sympatry, the ability of hatchery fish to socially dominate wild fish must be minimized to avoid undesirable interactions. The ability of hatchery fish to dominate wild fish may be reduced by decreasing the size of the smolts released, increasing the size differential between smolts and wild fish, and reducing the aggressiveness of hatchery fish.

Releases of hatchery fish that were smaller or much larger than the wild fish's most critical life stage may minimize hatchery fish dominance (Figure 2). The life stage that is most critical is one that constrains population productivity the most. Hatchery steelhead behaviorally dominated rainbow trout in treatment streams of the North Fork Teanaway basin presumably because of their larger size (Chapter 7). Larger wild fish also generally dominated smaller ones. Results from other studies have also suggested that large salmonids generally dominate smaller ones (Griffith 1972, Abbott et al. 1985, Hughes 1992). A difference of 5% body weight was enough to assure dominance in rainbow trout (Abbott et al. 1985). However, fishes that are drastically different in size may not interact, because they may select and occupy different habitats (Everest and Chapman 1972, Bisson et al. 1988, Hillman et al. 1989). For example, if age 0+ rainbow trout represent the life-stage limiting production, then release of large smolts may have less negative impact than the release of smaller age 0+ hatchery fish. In contrast, smolt size should not be so large that they are able to prey upon age 0+ fish (Cannamela 1992, Martin et al. 1993).

Determination of whether to produce hatchery fish smaller or larger than wild fish depends upon the size of the life stage of fish that is most limiting to production, the success (number of returning adults) of hatchery programs releasing different sizes of hatchery fish, and the length of time that hatchery fish will stay in freshwater prior to seaward migration. Reducing interactions among life stages of fish that are not limiting the production of the population may not appreciably reduce the impacts of the interactions to the whole population. Therefore, the emphasis on containing risks of interactions should be focussed on the direct and indirect linkages between hatchery fish and the life stage of sympatric wild fish where production of wild fish is most limited.

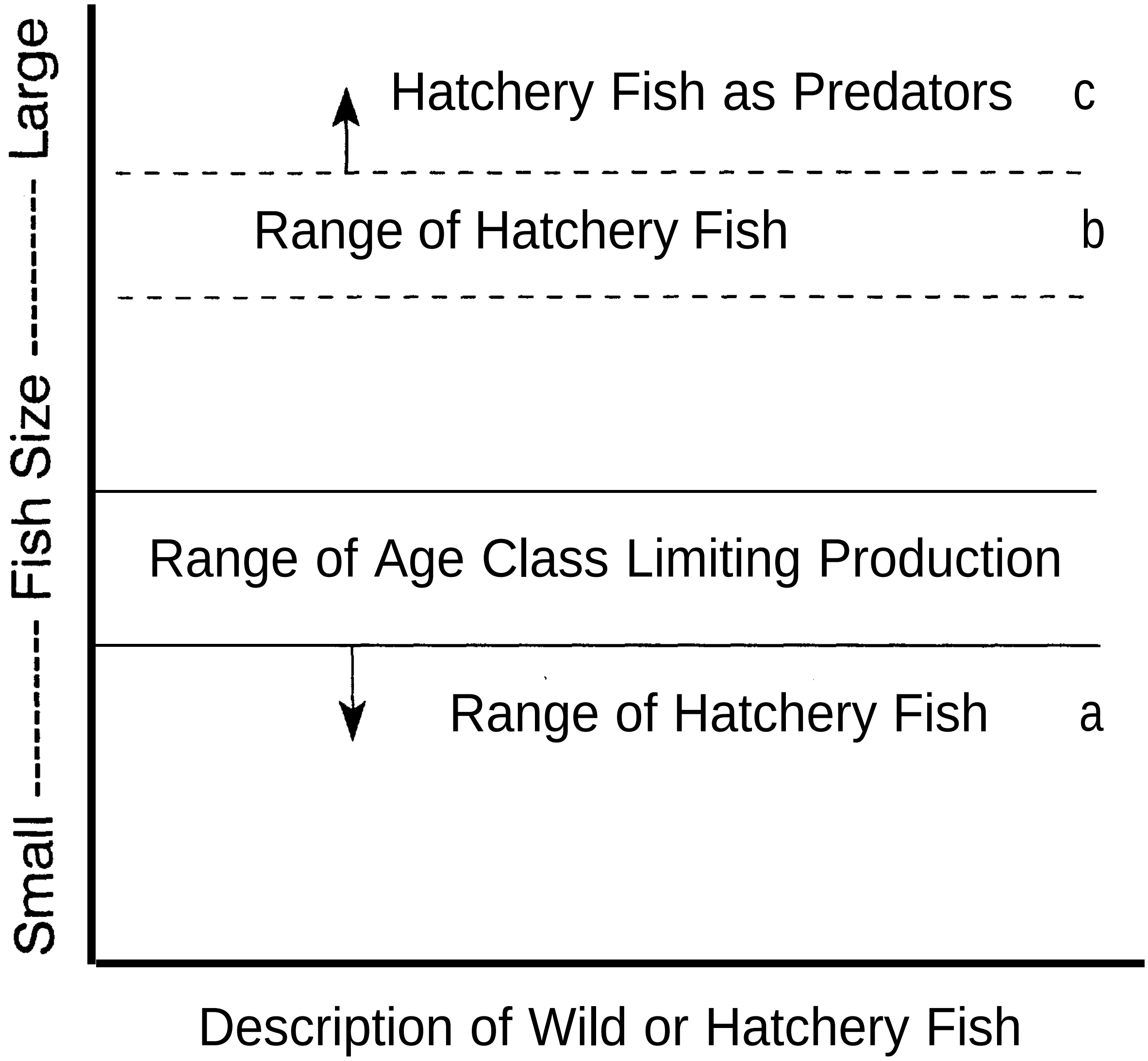

Figure 2. Hypothetical sizes of fish that could be stocked to minimimize adverse impacts to wild fish populations. One strategy (a) is to stock fish at smaller sizes than the age class of wild fish where production is most limited, another (b) is to stock fish at much larger sizes than the sizes of the age class of wild fish where production is most limited, however (c) not so large that the hatchery fish become predators on the age class of wild fish where production is most limited.

Reducing the social aggressiveness of hatchery fish could be accomplished by genetically selecting for passive fish, training fish to be submissive, culling out dominant fish, and providing hatchery rearing conditions that tend to reduce aggression. Aggresiveness may be both genetically determined and learned (Abbott et al. 1985, Huntingford et al. 1990, Metcalfe and Thorpe 1992). Passive fish might be genetically selected by tagging known passive fish and using only tagged fish for broodstock. This technique may be undesirable for numerous reasons including: the time needed to produce passive fish may be quite long, selection for passive fish may result in undesirable genetic qualities of the fish, and selection for passive fish may reduce genetic diversity and fitness of the hatchery fish. Alternatively, ecological strategies might be used to reduce hatchery fish aggressiveness.

Hatchery fish might be trained to be submissive. Fish have the ability to learn to consume novel prey and avoid predators (Suboski and Templeton 1989, Olla et al. 1992). Abbott et al. (1985) demonstrated that experience plays an important role in dominance relationships among juvenile steelhead trout. These authors found that a difference of 5% body weight was sufficient to assure dominant status for the larger fish. However, when fish that lost contests with another fish of similar size were separated, and then fed more food, they still lost contests with the dominant fish even though the subordinate was larger (14-114% larger). These results suggest that it might be possible to train hatchery fish to be submissive to wild fish regardless of difference in fish size. For instance, hatchery fish might be exposed to dominant fish for certain periods of time until the time of release. Subordinate fish may return to normal levels of aggression after only 1 to 14 days (Francis 1983) so the effects of subordinance training may be relatively short lived. Alternatively, dominant hatchery fish could be identified and culled prior to releasing fish, although there could be genetic ramifications for removing all dominant fish. As opposed to directly manipulating fish aggressiveness through breeding, training, or culling, manipulation of the hatchery environment may also reduce aggressiveness.

Producing a hatchery environment in which all positions in the raceway are equally preferable for fish, and with abundant food, should reduce fish aggressiveness. If resources within a hatchery could be distributed uniformly, no position in a raceway would be any better than another and thus territorial behavior would be theoretically unnecessary. Positions within hatchery raceways are typically uniform in depth, velocity, and cover. However, food is generally not distributed homogenously and is probably the resource that is most sought after in hatchery settings. Olla et al. (1992) found that agonistic behavior was reduced if food was introduced rapidly throughout aquaria as opposed to release from a single point source. In addition, chum salmon that were fed high rations were less aggressive than those fed low rations (Olla et al. 1992). In natural streams, food supplementation also reduced territorial behavior (Li et al. unpublished data). In short, passive fish might be produced by distributing food evenly throughout raceways in high amounts.

In summary, a variety of methods might be used in combination or singularly to minimize the interactions between hatchery and wild fish. Most of these methods are untested and should be further developed at a small scale before implementation at the production scale.

At present there are no plans within the YFP to produce fish that minimize interactions with wild fish, despite the goal of the YFP to minimize undesirable ecological interactions. In addition, deviations in YFP treatments to minimize interactions may confound the objectives of the current experimental design. Therefore, we recommend that studies be designed to determine the feasibility of developing a LIT (limited interaction treatment) which may be added to the YFP experimental design. If LIT is successful at reducing impacts to wild fish and returns of adults are deemed satisfactory then application of the rearing methods could be used to minimize impacts to wild fish caused by other hatcheries - particularly in areas with sympatric species of concern (Figure 1).

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Research Recommendations

The following research recommendations were identified using the work presented in this report. Additional research needs that were not covered by the scope of this report were not listed.

- Continue to investigate what locations steelhead trout spawn and rear in so that ecological interaction risks can be assessed and minimized

- Continue to monitor rainbow trout density and size structure in mainstem index sections and in select index sites of tributaries with added emphasis on determining the abundance and distribution of age 0+ rainbow trout in the mainstem - this information could be used to further characterize the rainbow trout population prior to supplementation

- Estimate the total number of rainbow trout redds in the mainstem of the upper Yakima River - this information could be used in addition to or as an alternative to mark-recapture population estimates (techniques that are less stressful to fish populations may become mandatory if certain fish species become protected under the Endangered Species Act)

- Describe the long term temporal variation of fish assemblage structure in the upper Yakima basin by repeating a study that assessed assemblage structure in the Yakima River during 1957 and 1958 - results might be used for characterizing assemblage structure prior to supplementation and for monitoring plan development

- Determine if age 0+ spring chinook salmon compete with age 0+ and 1+ rainbow trout - assessment of competition will help determine the potential effects of spring chinook supplementation to the life history stage of rainbow trout most limiting production and might also provide methods for monitoring interactions

- Determine the feasibility (through experimentation) of minimizing interactions between wild and hatchery fish - outcomes of this work might be used to modify treatments of the YFP

- Perform experimentation to examine the viability and smoltification characteristics displayed by offspring of matings between anadromous rainbow trout (steelhead trout) and resident rainbow trout, steelhead trout and steelhead trout, and resident rainbow trout and resident rainbow trout - results from these experiments will help assess potential impacts to rainbow and steelhead trout as a result of cross breeding

Appendix A

Genetic diversity in Yakima River rainbow trout above Roza Dam: Variation within 43 time/area collections and relationships to other Yakima River and hatchery *Oncorhynchus mykiss*

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Introduction

The electrophoretic analysis of rainbow trout, *Oncorhynchus mykiss*, collected from the mainstem Yakima River and tributaries above Roza Dam is part of the baseline phase of the Yakima River species interactions studies (Pearsons et al. 1993). The goals of this genetic work were: (1) to provide a baseline genetic profile of wild-spawned rainbow trout populations, (2) to determine the patterns of genetic diversity and stock structure among these populations and steelhead from the Yakima River, and (3) identify interbreeding between wild rainbow trout and steelhead, and between these wild populations and WDFW hatchery rainbow trout and steelhead strains that have been stocked into the Yakima River.

Previous reports (McMichael et al. 1992; Pearsons et al. 1993) have described the progress in genetic analyses of Yakima River rainbow trout populations (a group of trout residing at a specific reach of river at the time of collection). The baseline field sampling and laboratory electrophoresis have been completed. This report contains the electrophoretic data and preliminary analyses of within population genetic variation from all the collections for this study.

The specific topics covered in this progress report are: (1) the amount of within population heterozygosity and polymorphic loci within seasonal strata, (2) tests for nonrandom mating (Hardy-Weinberg, gametic phase disequilibrium) within a seasonal stratum at each location, (3) tests for nonrandom mating between spring and fall collections at each location, (4) tests for reproductive isolation among steelhead and rainbow trout, and (5) relationships based on genetic distances among rainbow trout and steelhead in the Yakima River and hatchery strains. All findings should be considered preliminary pending further analysis of the data.

Methods

Rainbow trout were collected from seven mainstem locations in the Yakima River and nine tributaries above Roza Dam during the spring and fall of 1990 through 1993 (Table 1). At many locations spring and fall collections were made during several years. The annual samples collected during these seasons were combined to maximize the number of loci used in the analyses. Sampling was done this way to reduce the impact on the rainbow trout population at each location. Rainbow trout were collected by electrofishing primarily, but some were collected by angling or trapping. Fish were collected during two distinct time periods to characterize the 1) spawning population and 2) rearing population. Fish collected in the spring were sexually mature (ripe or spent) or were large enough to spawn that year. Fish collected in the summer or fall were generally longer than 90 mm and were collected in a manner so that all size classes rearing in a locale would be represented (with the possible exception of age 0+ fish).

The collected fish were either dissected in the field (most adult specimens) or frozen whole at ultra-low temperatures (-80°C) and transported on dry ice to the Washington Department of Fish and Wildlife (WDFW) Genetics Laboratory. Muscle, heart, eye and liver were dissected from each fish and placed into 12 X 75 mm plastic culture tubes. Fork length and weight of each fish were recorded. The fish were photographed and refrozen for storage. Externally identifiable cutthroat trout and hatchery rainbow trout were excluded from the collections. Electrophoresis followed the methods of Aebersold et al. (1987). The electrophoretic protocol, enzymes screened, and alleles observed during this study (and other studies on rainbow trout and steelhead by WDFW) are listed in Phelps et al. (1994). Genetic nomenclature follows the conventions of Shaklee et al. (1990). All electrophoretic data collection used the WDFW computerized data acquisition system (Shaklee and Phelps 1990). Other details of the laboratory process are summarized in Phelps et al. (1994).

Allozyme pattern variation at the loci *mAAT-2**, *sAAT-4**, *AK1**, *DIA1**, *EST-2**, *bGALA**, *GAPDH-4**, *GAPDH-5**, *GDA-1**, *GDA-2**, *G3PDH-3**, *G3PDH-4**, *aMAN**, *mMDH-2**, *mMDH-3**, *PGM-3,4**, and *PNP1-1** were not used for the population genetic analyses of the data from this study because of uncertain genetic basis of the allozyme patterns or insufficient resolution and/or activity to accurately collect data in enough collections. Allozymes that had different variant alleles with similar mobilities were either pooled for analysis or rerun with standards to distinguish among them (The WDFW Genetics Laboratory maintains a tissue bank of standard allozyme mobility variants). Fish with the 116 and 121 alleles at *sIDHP-1,2**, the 37 and 49 alleles at *sMDH-A1,2**, and the 116, 120, and 125 alleles at *sMDH-B1,2** were rerun with standards to accurately score these alleles. At other loci, alleles were pooled for analysis (*bGLUA** E with A; *IDDH-1** B with A; *sIDHP-1,2** G with B; *sMEP-1** B and C with A, and F with D; *sMEP-2** C and D with A; *TPI-3** D with B;). Genotypes data at *ADA-1** with D, E, F alleles were given a score of zero (no data) because we believe that many of these patterns were due to enzyme breakdown.

We also examined the genotype data for the presence of alleles characteristic of cutthroat trout. The inclusion of other species in a collection is easily identified by alleles that occur in one species and do not in the other species (diagnostic alleles). The allelic differences between cutthroat trouts and rainbow trout have been characterized by Leary et al. (1987) and Phelps and LeClair (1993). All pure cutthroat trout were excluded from the analysis, but suspected hybrids were included.

BIOSYS-1 (Swofford and Selander 1981) was used for calculating allele frequencies, average heterozygosity, percentage of loci polymorphic at the 0.99 and 0.95 levels, genotype count agreement to Hardy-Weinberg expectations, genetic distances, and cluster analyses of genetic distances. Because this program can not use isoloci, the loci were split and treated as individual loci with all the allelic variation assigned to one locus for analyses that used isoloci (all except Hardy-Weinberg and gametic disequilibrium). The log likelihood ratio test (G-test) (Sokal and Rohlf 1981) program that was used to test for significant differences in allele frequencies among selected collection pairs was written by R. Waples (NMFS) and revised by C. Busack (WDFW). We tested the hypothesis that the pair of collections being tested differed no more than two random samples taken from the same randomly mating population. Included in the analysis were steelhead collections from the Yakima River that served as reference samples.

We also compared collections taken during the spring and fall at a location for evidence of nonrandom mating and the presence of more than one gene pool. Detecting a mixture of distinct gene pools or evidence of past stock mixing within a collection is important because of the effects on genetic diversity patterns and understanding the effects of past stock transfers. The chi-square goodness-of-fit test for Hardy-Weinberg equilibrium can be used to detect for the presence of more than one gene pool in a collection. When populations with significantly different allele frequencies are collected in the same location, the observed number of heterozygotes will be less than expected. This is called the Wahlund effect. We would expect to find a consistent deficit of heterozygotes in a collection (at loci that differ significantly) when there was: (1) reproductive isolation between rainbow trout and steelhead, or hatchery and wild rainbow trout -- and their juveniles reside together at the same location, (2) the presence of multiple wild stock gene pools in the collection (perhaps due to a spawning migration of trout from the mainstem into tributaries), (3) the presence of cutthroat trout and hybrids at a location, or (4) the sampling of stocked hatchery fish along with the wild populations. A consistent excess of heterozygotes indicates a first generation mixing of two gene pools. The X^2 value is = $F^2 \cdot N$, where F=1 - observed/expected heterozygotes. This statistic is useful for large recent mixtures of gene pools, but for subtle stock structuring, this test is not very sensitive. The agreement of observed genotype proportions to those expected by Hardy-Weinberg equilibrium was tested for all the non isoloci at each collection with at least five copies of a variant allele (627 tests).

Isoloci, loci that have four alleles due to past gene duplication (Allendorf and Thorgaard 1984), can not be tested for agreement to Hardy-Weinberg equilibrium unless some assumptions are made about how the genetic variation is distributed. In this study we did not use the isoloci (*sAAT-1,2**, *mAH-1,2**, *sMDH-A1,2**, *sMDH-B1,2**) for these tests. At another system that has been reported as an isolocus, *sIDHP-1,2**, we believe that the alleles can be associated with the individual loci *sIDHP-1** and *sIDHP-2** because of the differential tissue distribution of alleles, similar to a pattern observed in chinook salmon (Shaklee and Phelps 1992). In these collections, almost all the variation observed at this isolocus occurred at only one of the loci (*sIDHP-2**).

Another method to evaluate the incorporation of hatchery-origin rainbow trout genes into the wild populations within the Yakima River watershed is gametic disequilibrium analysis (Waples and Smouse 1990). While Hardy-Weinberg equilibrium analysis examines the relationship among different alleles at the same locus, gametic disequilibrium analysis examines the relationship among alleles at different loci. Alleles are expected to segregate independently according to Mendelian principles. Deviations from this expected random association of alleles is gametic disequilibrium and may indicate the mixing of two gene pools or the presence of multiple gene pools in a collection. Unlike the Wahlund effect, gametic disequilibrium does not disappear in the next generation. Among unlinked loci, the gametic disequilibrium decays at a rate of 50% per generation. If disequilibrium is found, we can identify which alleles are causing the disequilibrium. A positive relationship indicates that the alleles were found together more than expected, such as we would expect for alleles characteristic of either hatchery or wild populations. A negative relationship indicates that alleles were found in association less than expected, such as we would expect for combinations of hatchery-origin alleles with wild-origin alleles. A computer program, PANMIX, (Waples and Smouse 1990) was initially used to test collections for gametic disequilibrium. Collections with significant departure from equilibrium were tested further (by the program LINKNE) to identify the locus pairs that were significantly correlated (Campton 1987, Bartley et al. 1992).

To determine the power of gametic disequilibrium analysis in detecting mixtures of wild Yakima River rainbow trout and WDFW hatchery rainbow trout, we made some known mixtures of hatchery and wild rainbow trout using genotype data from two collections analyzed for this study. We chose the Goldendale Hatchery (9), which was known to have been used for stocking within the Yakima drainage and rainbow trout from the West Fork Teanaway River (36) which was thought to have minimal hatchery influence. Individual fish from these two collections were combined into five different mixtures of 100 fish total: 50:50 (50% hatchery), 25:75 (25% hatchery), 10:90 (10% hatchery), 5:95 (5% hatchery), and 1:99 (1% hatchery). Loci (except isoloci) that were polymorphic (frequency of common allele ≤ 0.95) and had at least one variant allele present at a frequency ≥ 0.05 were used in the analysis. Alleles present at a frequency < 0.05 were not selected. In addition to analyzing the five mixture samples, the two source collections (hatchery and wild) were analyzed separately. Allele frequencies were calculated for each collection and loci and alleles chosen using the criteria mentioned above.

Results and Discussion

We based this analysis on the products of 65 polymorphic loci including four isoloci, *sAAT-1,2**; *mAH-1,2**; *sMDH-A1,2**; and *sMDH-B1,2**. The nine loci that were monomorphic in all the collections in this study are: *AK*, CK-B*, G3PDH-2*, LDH-A2*, PGDH*, PGK-1*, sSOD-2*, TPI-2*, TPI-4**.

The average heterozygosity (observed and expected) and the two measures of the percentage of polymorphic loci are slightly higher than those reported in past studies (McMichael et al. 1992, Pearsons et al. 1993) due to the exclusion of monomorphic loci from this analysis. A small number of samples collected at a time/area stratum may affect the percentage of polymorphic loci values at the 0.99 level (make it lower), but there was still a wide range of values found. The loss of rarer alleles and hybridization between hatchery-origin and wild-origin fish likely contributed to these results. In general, the highest percentage of polymorphic loci and average heterozygosity occurred in the Umtanum and in the lower Yakima River mainstem collections. The WDFW hatchery rainbow trout strains had the lowest values, especially at the percentage of polymorphic loci at the 0.99 criterion. In contrast, the average heterozygosities were generally similar between the wild and hatchery collections.

The number of significant departures from expected Hardy-Weinberg equilibrium was higher than that expected by random chance at the $P<0.05$ level. Seventy-eight tests were significant out of the 627 total tests for a 12% rejection rate of the null hypothesis. The number of loci out of Hardy-Weinberg equilibrium and the total number of tests for each collections are listed on Table 2.

Gametic disequilibrium

All five mixture proportions of hatchery and wild rainbow trout had significant disequilibrium at the $P<0.05$ level (91 degrees of freedom): 50:50, χ^2=282.4 (50% hatchery), 25:75, χ^2=282.4 (25% hatchery), 10:90, χ^2=267.9 (10% hatchery), 5:95, χ^2=223.7 (5% hatchery), and 1:99, χ^2=267.9 (1% hatchery). The Goldendale Hatchery collection (9) and the West Fork Teanaway River collection (36) (the two source collections) were in gametic equilibrium. Twenty of the collections used in this study had significant gametic disequilibrium. The collection, significant locus pairs, and the relationship (positive +, or negative -) are listed in Table 3.

The results of a greater than expected number of loci out of Hardy-Weinberg equilibrium and the significant gametic disequilibrium in many of the time/area collections indicate the presence of multiple gene pools in many of these collections. Combining annual samples at a location for analysis in addition to genetic drift would contribute to a higher than expected number of significant tests. The nonrandom mating of fish with different genetic origins would produce the same result. Further analysis of this data will try to indicate the relative importance of these two factors.

Genetic differences between collections within streams

We tested all fall and spring collections (17 pairs) at the same location for significant differences in allele frequencies (G-test, Sokal and Rohlf 1981). Five pairs of collections, Cherry Creek (5 & 6), West Fork Teanaway River (36 & 37), Yakima River @ Cle Elum (51 & 52), Yakima River @ Crystal (54 & 55), and Yakima River @ Thorpe (62 & 63) were not significantly different at the $P<0.05$ criterion. In addition, 6 pairs of collections, Middle Fork Teanaway River (30 & 31), North Fork Teanaway River (32 & 33), Yakima River @ Ellensburg (56 & 57), Yakima River @ Lower Canyon (58 & 59), Yakima River @ Nelson Landing (60 & 61) and Yakima River @ Upper Canyon (64 & 65) were not significantly different at the $P<0.01$ criterion. The remaining six pairs of fall and spring collections were significantly different at $P<0.01$.

We also tested for differences in allele frequencies between Yakima River wild steelhead collections and rainbow trout. The combined Wilson River and Naneum Creek steelhead collection (47) was not different from several rainbow trout collections: the fall Cherry Creek collection (5) at $P<0.05$, and spring Cherry Creek (6) and spring Wilson Creek (49) at $P<0.01$. The North Fork Teanaway River steelhead collection (34) was not significantly different from North Fork Teanaway River spring rainbow trout (33) at $P<0.05$. The West Fork Teanaway River steelhead collection (35) was not significantly different from fourteen rainbow trout collections: seven at $P<0.05$ (20, 27, 30, 31, 32, 33, 37) and seven at $P<0.01$ (13, 17, 18, 26, 28, 29, 36). The Yakima River at Cle Elum steelhead (53) were not significantly different from spring Yakima River at Cle Elum rainbow trout (52) at $P<0.05$ and fall Yakima River at Cle Elum rainbow trout (51) at $P<0.01$. We have no evidence that rainbow trout and steelhead collected at the same location were reproductively isolated. However, the rainbow trout collections from the Yakima River and tributaries above Roza Dam were significantly different from wild steelhead in the rest of the Yakima River at $P<0.01$.

Genetic differences among collections

We calculated unbiased genetic distances (Nei 1978) and performed cluster analysis using the unweighted pair group method with arithmetic averaging (UPGMA). Numerous groups are evident (Figure 1). Group 1 consists of the spring and fall collections from Badger Creek. Group 2 consists of mainstem Yakima River and two lower tributaries. Group 3 consists of spring and fall collections from two sections of the Umtanum River. Groups 4 and

5 are from the most upper Yakima River collection sites. Group 6 consists of Yakima River tributary collections above Ellensburg and Naches River and mainstem Yakima River steelhead collections. Groups 7-13 contain collections used for comparison of the genetic characteristics of the Yakima River rainbow trout: 7 is the 1991 WDFW Yakima Hatchery collection, 8 contains wild steelhead from the Satus Creek drainage, 9 is the WDFW Wells Hatchery summer-run steelhead, 10 is Nile Pond Hatchery rainbow trout, 11 is wild Toppenish Creek steelhead, 12 is summer-run and winter-run steelhead from the WDFW Skamania Hatchery, and 13 contains hatchery rainbow trout strains from WDFW and a private business. These relationships will be analyzed further in the final report(s) of the electrophoretic analyses of Yakima River rainbow trout and steelhead.

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Table 1. Rainbow trout and steelhead collections from the Yakima River and WDFW hatcheries from 1989 - 1993.

Number and Abbreviation	Year	Number Collected	Location and collection season
1. BADGFRT	1991	33	Badger Creek rainbow trout, fall
	1992	33	
	1993	33	
2. BADGSRT	1991	32	Badger Creek rainbow trout, spring
	1992	29	
3. BIGFRT	1990	4	Big Creek rainbow trout, spring and fall
	1991	1	
4. CABINFRT	1990	25	Cabin Creek rainbow trout, fall
5. CHERFRT	1990	50	Cherry Creek rainbow trout, fall
	1991	33	
6. CHERSRT	1990	18	Cherry Creek rainbow trout, spring
	1991	6	
7. COWICHST	1991	25	Cowiche Creek steelhead smolts
8. DRYCR	1991	153	Dry Creek (Satus Creek trib.) steelhead smolts
9. GOLD HAT	1990	100	WDFW rainbow trout hatchery, Goldendale strain
10. LNACST	1991	45	Little Naches River steelhead smolts
11. LOGYST	1990	78	Logy Creek (Satus Creek trib.) steelhead smolts
	1991	108	
12. MANAFRT	1990	49	Manastash Creek rainbow trout, fall
	1991	33	
	1992	33	
	1993	33	
13. MANASRT	1990	2	Manastash Creek rainbow trout, spring
	1991	33	
	1992	33	
	1993	15	
14. NILERT	1990	50	Nile Pond Hatchery rainbow trout, Goldendale strain
15. PRSER1	1989	79	Prosser Dam steelhead smolts May 3-11
16. PRSER2	1989	86	Prosser Dam steelhead smolts May 14-18
17. PRSER3	1989	88	Prosser Dam steelhead smolts May 22-30
18. PRSER4	1989	48	Prosser Dam steelhead smolts June 2-14
19. RATYAK	1991	47	Rattlesnake Creek summer steelhead smolts
20. ROZAST	1989	54	Roza Dam summer steelhead smolts
	1991	122	
21. SATUSST	1990	102	Satus Creek summer steelhead smolts
	1991	111	
22. SKAMSRST	1991	95	WDFW steelhead hatchery, Skamania River summer-run strain
23. SKAMWRST	1993	50	WDFW steelhead hatchery, Skamania River winter-run strain
24. SPOK HAT	1990	100	WDFW rainbow trout hatchery, Spokane strain
25. STAC HAT	1990	100	WDFW rainbow trout hatchery, South Tacoma strain
26. SWAKFRT	1990	50	Swauk Creek rainbow trout, fall
	1991	33	
	1992	32	
	1993	18	
27. SWAKSRT	1990	5	Swauk Creek rainbow trout, spring
	1991	33	
	1992	33	

Table 1. continued

28.	TANFRT	1990	51	Taneum Creek rainbow trout, fall
		1991	33	
		1992	33	
		1993	33	
29.	TANSRT	1990	13	Taneum Creek rainbow trout, spring
		1991	33	
		1992	33	
30.	TNWMFFRT	1990	50	Middle Fork Teanaway River rainbow trout, fall
		1991	33	
		1992	33	
		1993	33	
31.	TNWMFSRT	1991	33	Middle Fork Teanaway River rainbow trout, spring
		1992	33	
		1993	33	
32.	TNWNFFRT	1990	50	North Fork Teanaway River rainbow trout, fall
		1991	31	
		1992	32	
		1993	33	
33.	TNWNFSRT	1990	2	North Fork Teanaway River rainbow trout, spring
		1991	33	
		1992	33	
		1993	3	
34.	TNWNFSST	1991	25	North Fork Teanaway River summer steelhead smolts
35.	TNWSST	1991	26	West Fork Teanaway River summer steelhead smolts
36.	TNWWFFRT	1990	51	West Fork Teanaway River rainbow trout, fall
		1991	33	
		1992	33	
		1993	33	
37.	TNWWFSRT	1991	33	West Fork Teanaway River rainbow trout, spring
		1992	33	
		1993	24	
38.	TOKL HAT	1990	100	WDFW rainbow trout hatchery, Tokul Creek strain
39.	TOPPST	1990	15	Toppenish Creek summer steelhead smolts
		1991	110	
40.	UFISH	1990	38	U-fish hatchery rainbow trout (private business)
41.	UMT1FRT	1990	27	Umtanum Creek section 1 (lower), rainbow trout, fall
		1991	17	
		1992	20	
		1993	20	
42.	UMT1SRT	1990	16	Umtanum Creek section 1 (lower), rainbow trout, spring
		1991	19	
		1992	20	
		1993	8	
43.	UMT2FRT	1990	24	Umtanum Creek section 2 (upper), rainbow trout, fall
		1991	16	
		1992	17	
		1993	20	
44.	UMTPSRT	1990	13	Umtanum Creek section 2 (upper), rainbow trout, spring
		1991	14	
		1992	19	
		1993	7	
45.	WAPAST	1987	48	Naches River steelhead smolts, Wapatox collection
		1989	117	
		1990	100	
		1991	105	
46.	WELLSHST	1991	40	WDFW Wells Hatchery steelhead, summer-run strain
		1993	50	
47.	WIL/NST	1991	20	Wilson and Naneum creeks steelhead smolts
48.	WILFRT	1990	17	Wilson Creek rainbow trout, fall

Table 1. continued

		1991	32	
		1992	25	
49.	WILSRT	1990	16	Wilson Creek rainbow trout, spring
		1991	33	
		1992	23	
		1993	22	
50.	YHATST	1991	34	WDFW Yakima Hatchery steelhead, summer-run strain
51.	YMSCEFRT	1990	13	Yakima River mainstem, Cle Elum, rainbow trout, fall
		1991	20	
		1992	22	
		1993	26	
52.	YMSCESRT	1991	20	Yakima River mainstem, Cle Elum, rainbow trout, spring
		1992	22	
53.	YMSCEST	1991	40	Yakima River mainstem, Cle Elum, summer steelhead smolts
54.	YMSCRFRT	1990	18	Yakima River mainstem, Crystal, rainbow trout, fall
		1991	20	
		1992	24	
		1993	25	
55.	YMSCRSRT	1992	25	Yakima River mainstem, Crystal, rainbow trout, spring
56.	YMSELFRT	1990	14	Yakima River mainstem, Ellensburg, rainbow trout, fall
		1991	20	
		1992	25	
		1993	25	
57.	YMSELSRT	1990	1	Yakima River mainstem, Ellensburg, rainbow trout, spring
		1991	20	
		1992	20	
58.	YMSLCFRT	1990	6	Yakima River mainstem, Lower Canyon, rainbow trout, fall
		1991	20	
		1992	25	
		1993	17	
59.	YMSLCSRT	1990		Yakima River mainstem, Lower Canyon, rainbow trout, spring
		1991	32	
		1992	25	
60.	YMSNLFRT	1991	20	Yakima River mainstem, Nelson, rainbow trout, fall
		1992	19	
		1993	25	
61.	YMSNLSRT	1991	10	Yakima River mainstem, Nelson, rainbow trout, spring
		1992	25	
62.	YMSTHFRT	1990	13	Yakima River mainstem, Thorpe, rainbow trout, fall
		1991	20	
		1992	25	
		1993	19	
63.	YMSTHSRT	1990		Yakima River mainstem, Thorpe, rainbow trout, spring
		1991	20	
		1992	15	
64.	YMSUCFRT	1990	15	Yakima River mainstem, Upper Canyon, rainbow trout, fall
		1991	20	
		1992	25	
		1993	25	
65.	YMSUCSRT	1990	8	Yakima River mainstem, Upper Canyon, rainbow trout, spring
		1991	21	
		1992	25	

Table 2. Within population genetic diversity measures and the number of loci in each collection that were tested and found significantly different (P<0.05) from Hardy Weinberg expectations [excess (+) or deficit (-) of heterozygote's] in 65 rainbow trout and steelhead collections from the Yakima River and WDFW hatcheries.

collection	Mean sample size per LOCUS	Mean no. of alleles per locus	Percentage of loci polymorphic+ 0.99	Percentage of loci polymorphic+ 0.95	Mean heterozygosity Direct-count	Mean heterozygosity HdyWbg expected*+	Hardy-Weinberg proportions deviations No. of loci	Hardy-Weinberg proportions deviations Total tests
1. BADGFRT	95.6	1.5	40.0	23.3	0.074	0.078	-2	15
2. BADGSRT	58.1	1.5	40.0	23.3	0.080	0.078	-1	13
3. BIGFRT	50.7	1.7	48.3	30.0	0.084	0.092	-3	19
4. CABINFRT	24.3	1.5	30.3	26.7	0.081	0.082	-1	10
5. CHERFRT	81.8	1.8	53.3	26.7	0.085	0.089	-1	18
6. CHERSRT	22.5	1.5	38.3	23.3	0.084	0.086	+2, -1	10
7. COWICHST	24.9	1.5	41.7	28.3	0.090	0.092	0	11
8. DRYCR	146.0	1.7	48.3	18.3	0.066	0.070	-3	18
9. GOLD HAT	98.0	1.4	26.7	20.0	0.076	0.074	+1	12
10. LNACST	42.6	1.6	35.0	18.3	0.065	0.065	-1	10
11. LOGYST	176.9	1.7	48.3	18.3	0.069	0.071	-1	16
12. MANAFRT	140.9	1.9	61.7	26.7	0.061	0.083	-4	21
13. MANASRT	80.6	1.8	56.7	20.0	0.077	0.079	-1	16
14. NILERT	42.0	1.5	35.0	21.7	0.072	0.070	-1	10
15. PRSER1	72.3	1.8	51.7	28.3	0.077	0.076	-2	15
16. PRSER2	53.0	1.7	51.7	28.3	0.075	0.090	-6	17
17. PRSER3	80.1	1.8	55.0	25.0	0.070	0.076	-5	17
18. PRSER4	44.0	1.8	55.0	28.3	0.077	0.085	-4	17
19. RATYAK	46.2	1.8	51.7	21.7	0.077	0.079	-3	15
20. ROZAST	171.9	2.0	60.0	25.0	0.078	0.081	-4	24
21. SATUSST	198.5	1.7	51.7	20.0	0.074	0.073	+1, -1	20
22. SKAMSRST	92.8	1.5	38.3	20.0	0.068	0.069	-1	15
23. SKAMWRST	49.9	1.5	35.0	15.0	0.059	0.059	-1	9
24. SPOK HAT	95.8	1.4	33.3	18.3	0.088	0.079	t2	17
25. STAG HAT	97.3	1.4	30.0	23.3	0.087	0.086	-1	14
26. SWAKFRT	125.0	2.0	61.7	21.7	0.077	0.076	+1, -3	17
21. SWAKSRT	68.7	1.8	51.7.	21.7	0.072	0.074	+3, -1	16
28. TANFRT	142.4	1.9	61.7	28.3	0.079	0.082	-7	25
29. TANSRT	76.2	1.7	50.0	25.0	0.077	0.077	-1	16
30. TNWMFFRT	144.7	1.9	61.7	20.0	0.078	0.076	+1	20
31. TNWMFSRT	97.0	1.8	53.3	20.0	0.080	0.077	-1	20
32. TNWNFFRT	138.4	1.9	60.0	23.3	0.074	0.078	-3	24
33. TNWNFSRT	68.0	1.7	53.3	25.0	0.083	0.080	+1, -2	19
34. TNWNFSST	24.4	1.5	38.3	21.7	0.074	0.076	0	8
35. TNWSST	24.0	1.5	38.3	20.0	0.070	0.072	-1	7
36. TNWWFFRT	146.0	1.9	61.7	18.3	0.077	0.078	-2	22
37. TNWWFSRT	88.7	1.9	61.7	25.0	0.080	0.080	0	18
38. TOKL HAT	96.9	1.2	21.7	16.7	0.072	0.070	-1	10
39. TOPPST	118.4	1.5	36.7	20.0	0.065	0.066	-1	13
40. UFISH	36.5	1.4	28.3	16.7	0.066	0.069	0	8
41. UMT1FRT	80.3	1.9	58.3	25.0	0.094	0.093	-2	18
42. UMT1SRT	60.5	1.8	56.7	25.0	0.106	0.101	+1, -1	14
43. UMTZFRT	73.5	1.7	46.7	23.3	0.093	0.091	-2	14
44. UMT2SRT	51.3	1.6	41.7	30.0	0.096	0.097	t2, -2	16
45. WAPAST	340.0	2.1	65.0	21.7	0.071	0.073	-6	24
46. WELLSHST	ea.9	1.7	51.7	20.0	0.063	0.067	-2	14
47. WIL/NST	19.6	1.5	33.3	23.3	0.079	0.079	0	7
48. WILFRT	71.7	1.8	51.7	21.7	0.090	0.092	-2	17
49. WILSRT	89.2	1.8	55.0	26.7	0.061	0.084	-1	18
50. YHATST	35.5	1.6	48.3	23.3	0.081	0.079	-1	12
51. YMSCEFRT	19.2	1.9	55.0	26.7	0.088	0.091	-2	17
52. YMSCESRT	40.9	1.6	46.7	23.3	0.089	0.088	-1	13
33. YMSCEST	38.9	1.7	46.7	26.7	0.087	0.084	0	12
54. YMSCRFRT	82.8	1.7	50.0	23.3	0.073	0.074	-1	15
55. YMSCRSRT	24.5	1.4	33.3	20.0	0.067	0.069	-1	10
56. YMSELFRT	00.5	1.8	55.0	23.3	0.088	0.089	-1	18
57. YMSELSRT	40.7	1.7	46.7	23.3	0.092	0.088	Cl, -1	13
58. YMSLCFRT	66.8	1.8	51.7	28.3	0.088	0.087	-2	16
59. YMSLCSRT	64.7	1.8	55.0	28.3	0.090	0.089	0	15
60. YMSNLFRT	62.1	1.7	50.0	23.3	0.090	0.090	-2	16
61. YMSNLSRT	34.2	1.6	41.7	23.3	0.079	0.084	-1	13
62. YMSTHFRT	74.9	1.8	55.0	23.3	0.079	0.082	-3	16
63. YMSTHSRT	34.8	1.6	45.0	28.3	0.088	0.092	-3	13
64. YMSUCFRT	83.3	1.9	53.3	25.0	0.090	0.091	+1, -2	19
63. YMSUCSRT	52.7	1.8	45.0	30.0	0.087	0.089	-1	17

Table 3. Gametic disequilibrium relationships of significant locus pairs (+= positive association of alleles, -= negative association of alleles) from 20 collections with significant gametic disequlibrium (P<0.05).

Collection	Locus pair	Relationship
BIGFRT (3)	ADA-1/GAPDH-3	+
	ADA-1/IDDH-2	
	ADA-1/mIDHP-2	+
	ADA-2/sAH	+
	ADA-2/GPI-A	+
	ADA-2/PEPB-1	+
	ADA-2/PGK-2	
	ADA-2/PGM-2	+
	ADA-2/sSOD-1	+
	ALAT/sSOD-1	+
	GPI-A/IDDH-2	+
	GPI-A/PEPB-1	+
	GPI-A/PGM-2	+
	GPI-A/sSOD-1	+
	mIDHP-2/mSOD	+
	NTP/PEPB-1	
	NTP/PGM-2	+
	NTP/sSOD-1	+
CABINFRT (4)	ADA-1/sAH	
	ADA-1/LDH-C	+
	ADA-1/NTP	+
	ADA-1/sSOD-1	+
	ADA-1/mSOD	+
	ADA-2/IDDH-2	+
	ADA-2/sSOD-1	+
	sAH/ßGLUA	+
	sAH/mIDHP-2	+
	sAH/sSOD-1	+
	ßGLUA/mIDHP-2	+
	ßGLUA/sIDHP-2	+
	sIDHP-2/LDH-B2	
	LDH-B2/PEP-LT	
	LDH-C/NTP	+
	LDH-C/mSOD	+
	NTP/mSOD	+
	PEPB-1/PEP-LT	+
	PGK-2/sSOD-1	+

Table 3. continued

Collection	Locus pair	Relationship
LOGYST (11)	sAH/LDH-B2	+
	ALAT/NTP	+
	ALAT/sSOD-1	-
	MPI/mSOD	-
	PEPA/PGK-2	+
	PGK-2/mSOD	-
	sSOD-1/mSOD	+
	mSOD/TPI-3	+
MANASRT (13)	ADA-1/LDH-B2	-
	sAH/MPI	+
	GAPDH-3/sIDHP-2	+
	GAPDH-3/LDH-B2	+
	GAPDH-3/PEPA	+
	sIDHP-2/NTP	-
	sIDHP-2PEPB-1	-
	LDH-B2/NTP	-
	LDH-B2/PEPA	+
	NTP/PEPA	-
	PEPB-1/sSOD-1	+
NILERT (14)	ADA-1/sIDHP-2	+
	ADA-1/PGM-2	+
	sAH/sIDHP-2	+
	sAH/LDH-B2	+
	GAPDH-3/MPI	+
	ßGLUA/PGM-2	+
	sIDHP-2/LDH-B2	+
	sIDHP-2/NTP	-
	PGM-2/TPI-3	+
PRSER2 (16)	ADA-1/CK-A1	+
	ADA-1/ßGLUA	+
	ADA-1/mIDHP-2	+
	ADA-1/sIDHP-2	
	ADA-1/LDH-B2	-
	ADA-1/NTP	+
	ADA-1/sSOD-1	+
	sAH/mIDHP-2	-
	CK-A1/ßGLUA	+
	CK-A1/mIDHP-2	+
	CK-A1/sIDHP-2	-
	CK-A1/LDH-B2	-
	GAPDH-3/TPI-3	+
	ßGLUA/mIDHP-2	+
	ßGLUA/sIDHP-2	-
	ßGLUA/LDH-B2	
	ßGLUA/NTP	+
	ßGLUA/sSOD-1	+

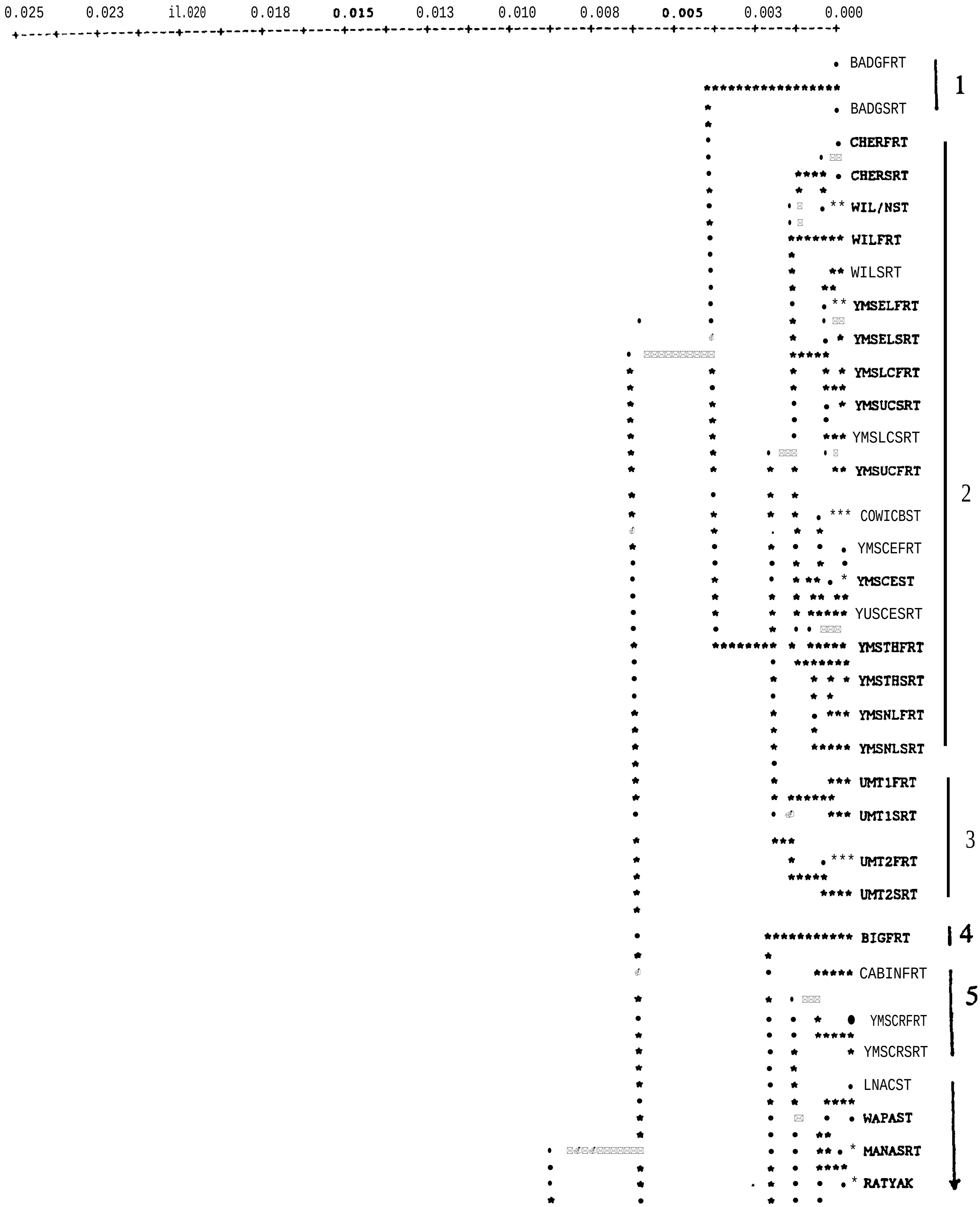
Table 3. continued

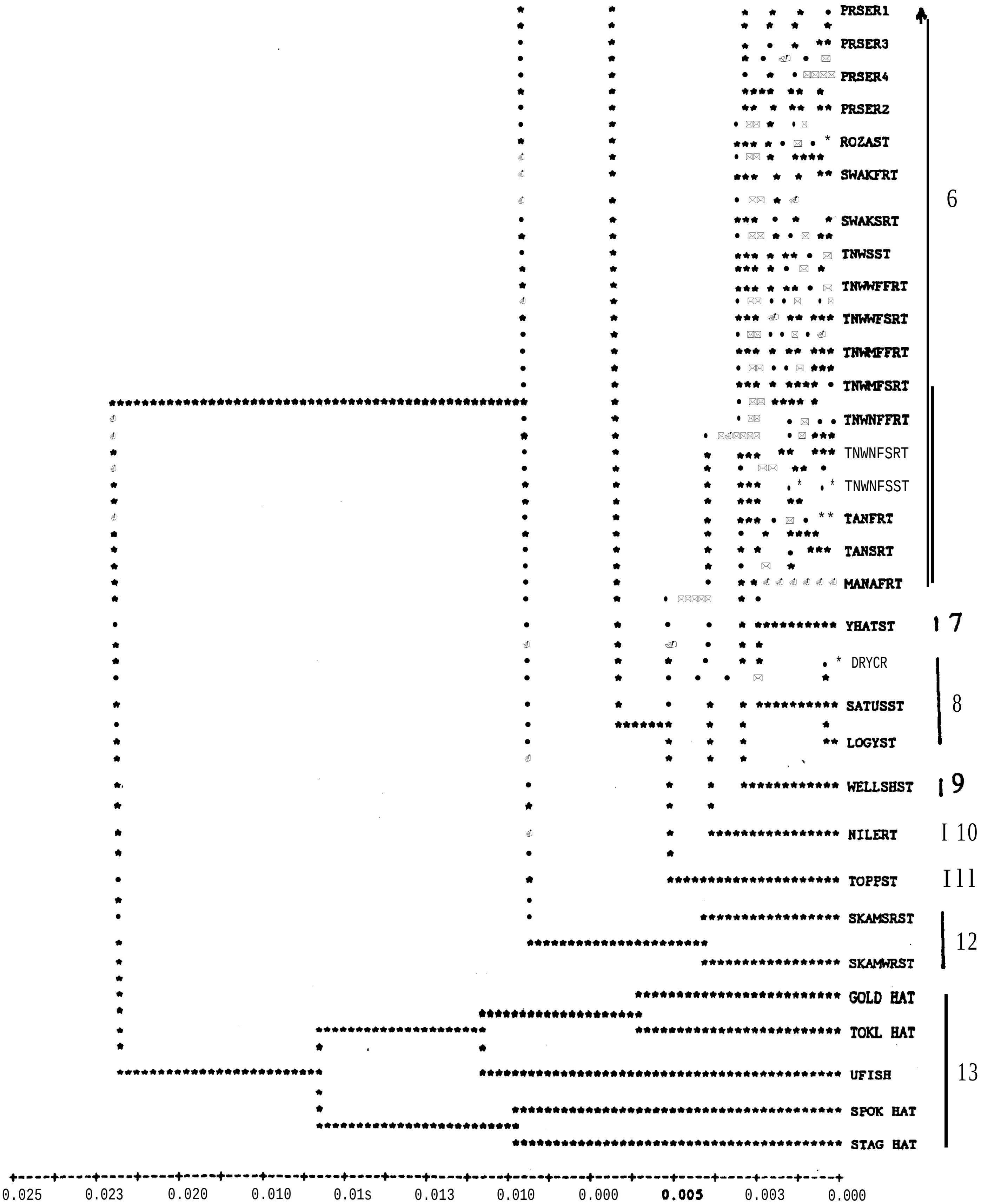
Collection	Locus pair	Relationship
	mIDHP-2/sIDHP-2	-
	mIDHP-2/LDH-B2	-
Collection	**Locus pair**	**Relationship**
	mIDHP-2/NTP	+
	mIDHP-2/sSOD-1	+
	sIDHP-2/MPI	+
	sIDHP-2/NTP	
	sIDHP-2/sSOD-1	
	LDH-B2/NTP	
	PEPA/sSOD-1	
	PEPA/mSOD	+
	PGK-2/TPI-3	
PRSER3 (17)	ADA-1/mIDHP-2	+
	ADA-1/sIDHP-2	-
	ADA-1/LDH-B2	-
	sAH/mSOD	+
	sIDHP-2/MPI	+
	NTP/PEPA	-
SKAMWRST (23)	sIDHP-2/PEPD-1	-
	sIDHP-2/PGK-2	-
	NTP/sSOD-1	-
SPOKHAT (24)	CK-A1/NTP	
	LDH-C/PGK-2	+
SWAKSRT (27)	sAH/PEPB-1	-
TNWMFSRT (31)	sAH/LDH-B2	+
	sAH/sSOD-1	-
	GAPDH-3/MPI	+
	sIDHP-2/NTP	+
	LDH-B2/MPI	+
	MPI/PEPB-1	+
	PEPA/PGK-2	-
TNWNFSST (34)	sAH/LDH-B2	+
	sAH/PEPA	+
	ßGLUA/PGK-2	-
	mIDHP-2/mSOD	+
	mMEP-1/PGM-2	+
UFISH (40)	ADA-1/PGM-2	+
	CK-C1/PGM-2	
	ßGLUA/PGM-2	

Table 3. continued

Collection	Locus pair	Relationship
WAPAST (45)	sAH/PEPB-1	+
	ALAT/PEPA	+
	ALAT/mSOD	+
	GAPDH-3/PEPB-1	+
	sIDHP-2/PGK-2	
	LDH-B2/mSOD	+
WELLSHST (46)	ADA-2/sIDHP-2	+
	ADH/PEPLT	+
	PEPA/PGK-2	
WILFRT (48)	ADA-1/mSOD	
	sAH/PGM-2	+
	ßGLUA/MPI	+
	ßGLUA/NTP	+
	ßGLUA/PGM-2	+
	MPI/PGM-2	+
WILSRT (49)	sAH/LDH-B2	+
	ßGLUA/mIDHP-2	+
	mIDHP-2/sSOD-1	+
	LDH-B2/PEPD-1	+
	MPI/NTP	
	PEPD-1/PGK-2	
YMSCRFRT (54)	sAH/ALAT	+
	GAPDH-3/sIDHP-2	
	GAPDH-3/PGK-2	+
	sMEP-1/sMEP-2	+
YMSELSRT (57)	ADA-1/CK-C1	+
	ADA-1/NTP	+
	sAH/ßGLUA	
	CK-C1/ßGLUA	+
	CK-C1/PEPD-1	+
	mIDHP-2/sIDHP-2	
	sIDHP-2/PEPD-1	+
YMSUCSRT (65)	ADA-1/sIDHP-2	
	sAH/PEPD-1	+
	ALAT/mIDHP-2	+
	ALAT/PGK-2	+
	mIDHP-2/MPI	
	mIDHP-2/NTP	
	sIDHP-1/LDH-B2	+
	NTP/PGK-2	
	PEPA/PGK-2	+

Figure 1. Population **relationships of 65 Yakima River** rainbow **trout** and **steelhead** and WDFW hatchery strains **based** on **Nei's** (1978) unblamed **genetic distance**




PRSER1
PRSER3
PRSER4
PRSER2
ROZAST
SWAKFRT
SWAKSRT
TNWSST
TNWWFFRT
TNWWFSRT
TNWMFFRT
TNWMFSRT
TNWNFFRT
TNWNFSRT
TNWNFSST
TANFRT
TANSRT
MANAFRT
YHATST
DRYCR
SATUSST
LOGYST
WELLSHST
NILERT
TOPPST
SKAMSRST
SKAMWRST
GOLD HAT
TOKL HAT
UFISH
SPOK HAT
STAG HAT
6
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9
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12
13
0.025
0.023
0.020
0.010
0.01s
0.013
0.010
0.000
0.005
0.003
0.000