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YAKIMA RIVER SPECIES INTERACTIONS STUDIES

ANNUAL REPORT 1994

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Executive Summary

Species interactions research was initiated in 1989 to investigate ecological interactions among fish in response to proposed supplementation of salmon and steelhead in the upper Yakima River basin. This is the fifth of a series of annual reports that address species interactions research and pre-facility monitoring of fishes in the upper Yakima River basin. Data have been collected prior to supplementation to characterize the rainbow trout and other fish populations such as steelhead and spring chinook salmon, 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 two chapters followed by seven "updates" with a general introduction preceding the first chapter and a general discussion following the last update. An appendix follows the general discussion. This annual report summarizes data collected primarily by the Washington Department of Fish and Wildlife (WDFW) between January 1 and December 31, 1994 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.

- Age 0+ rainbow trout and spring chinook salmon were closely associated with bank habitats in the Yakima River during the spring, summer, and fall. Few fish were observed in the middle of either mainstem or side channel habitats. Spring chinook salmon and rainbow trout were most commonly found together during the fall when size differences between the species was smallest and the size of the river smallest. Spring chinook salmon dominated rainbow trout in 52% of the behavioral contests examined during underwater snorkeling observations.

- The number of fish and fish species captured in traps in different sites of Swauk Creek decreased with increasing site elevation. However, annual variations in assemblage structure did not appear to be different among sites. Immigration of rainbow trout and spring chinook salmon decreased with site elevation and up to 50% of the individuals collected in an index site originated from outside of the site. Electrofishing at the time when rainbow trout population estimates were conducted did not appear to influence fish movement substantially.

- The spatial distribution of rainbow trout redds in the Yakima River was patchy. Most redds were observed in reaches with unconstrained channels and abundant instream

cover. The length of redd tails was positively related to the length of the fish constructing the redd. Rainbow trout redds constructed in 1994 were significantly smaller than those constructed in 1993, suggesting that spawning fish were smaller in 1994 than in 1993.

- Large rainbow trout (longer than 174 mm) that were anchor tagged tended to move downstream more often than upstream. In addition, low numbers of rainbow trout tagged in 100 m long tributary index sites were recaptured one year later. However, more fish were recaptured at upper elevation sites than at lower ones.

- Temporal variability of rainbow trout abundance in tributary index sites ranged from stable to highly fluctuating. Average rainbow trout density in 1994 ranged from $0.12/m^2$ (Swauk Creek) to $0.01/m^2$ (Cabin Creek). Trout densities in five index sections of the Yakima River averaged 297/km during 1994 and were not as temporally variable as tributary sites. All juvenile spring chinook salmon were observed in sites less than 730 m elevation above sea level.

- Variation in assemblage structure was larger in space than in time in tributary and mainstem index sites. In addition, patterns of assemblage structure documented in 1994 were similar to those reported in previous years. Methods that have been used in tributary and mainstem sites appear to be adequate for describing relative abundances of species.

- Hatchery-reared steelhead released into the North Fork Teanaway sub-basin for test purposes behaviorally dominated rainbow trout in most agonistic contests presumably because of their larger size. Displacement of wild trout by hatchery steelhead was seen within channel units, may have occurred over a short stream reach (500 m) in 1994, and was not detected over large spatial scales. Densities and biomasses of wild rainbow trout appeared to have been negatively influenced by releases of hatchery steelhead. Residual steelhead were relatively abundant in 1994 and were observed over 12 km upstream of the area where they were released in an area containing populations of wild bull and cutthroat trout.

- Results from competition experiments performed in small screened enclosures within the North and Middle forks of the Teanaway River suggested that: 1) the presence of hatchery-reared steelhead negatively impacted growth of naturally-produced rainbow trout, but did not impact the growth of spring chinook salmon; 2) the presence of hatchery-reared spring chinook salmon negatively impacted the growth of wild spring chinook salmon; and 3) the presence of wild spring chinook salmon did not impact the growth of wild rainbow trout. The potential impact of hatchery spring chinook salmon on wild rainbow trout was not examined.

- Superior performance of hatchery-reared steelhead, reflected by more rapid in-river emigration rates, lower rates of precocialism, and lower incidence of residualism, was observed when their parents were hatchery broodstock as opposed to wild broodstock. In addition, performance was also enhanced when hatchery steelhead were reared at lower densities, and were released at smaller sizes. The correlations between parentage and performance were consistent during the four year study, but rearing density and size at release deviated from the general pattern in 1994.

- The condition (length to weight relationship) of rainbow' trout that had previously been hook-scarred by anglers was the same as rainbow trout that had not been hook-scarred. Between 1990 and 1994, the proportion of rainbow trout that had hook-scars ranged from 7 - 36% in five mainstem index sections. An increase in the proportion of fish having hook-scars was observed in the Lower Canyon section between 1990 and 1994.

All findings in this report should be considered preliminary and subject to further revision.

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The names of the lead authors are presented by each chapter, update, and appendix. Their names are presented here to recognize their important contributions and to provide a point of contact if readers would like further information about a specific topic. Each author can be contacted at the address on the title page.

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General Introduction

This report is intended to satisfy two concurrent needs: 1) provide a contract deliverable from the Washington Department of Fish and Wildlife (WDFW) to the Bonneville Power Administration (BPA), with emphasis on identification of salient results of value to ongoing Yakima Fisheries Project (YFP) planning, and 2) summarize results of research that have broader scientific relevance. This annual progress report summarizes data collected between January 1, and December 31, 1994. These data were compared to findings from previous years to identify general trends and make preliminary comparisons. This is the fifth of a series of annual reports that address species interactions research and pre-facility monitoring of fishes in response to proposed supplementation of salmon and steelhead in the upper Yakima River basin (Hindman et al. 1991; McMichael et al. 1992; Pearsons et al. 1993; Pearsons et al. 1994). Interactions between fish produced as part of the YFP termed target species or stocks, and other species or stocks may alter the population status of non-target species or stocks. This may occur through a variety of mechanisms, such as competition, predation, and interbreeding (reviewed in Pearsons et al. 1994).

Initially, our work focused on interactions between anadromous steelhead and resident rainbow trout (for explanation see Pearsons et al. 1993), however during the past few years 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 (Bonneville Power Administration et al. 1996). Originally, steelhead and spring chinook salmon were proposed to be supplemented simultaneously (Clune and Dauble 1991). However, current planning may provide for steelhead to be supplemented at a later date than spring chinook salmon. This redirection in the species to be supplemented has prompted us to emphasize investigations of 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 has been continued and should be an important part of -monitoring the effects of interactions, regardless of which species is supplemented, or when.

This report is organized into two chapters, seven updates of chapters in previous reports, and one appendix. These sections pertain to major topics associated with the life histories of rainbow trout and spring chinook salmon, interactions experimentation, and methods for sampling. In contrast to previous reports (Hindman et al. 1991; McMichael et al. 1992; Pearsons et al. 1993; Pearsons et al. 1994) this revised report' format is intended to provide broad treatment of topics that have not been addressed in previous reports (chapters), and updates of chapters in previous reports (updates). Throughout this report, a premium was placed on presenting data in tables so that interested parties could have access to the data. Chapter 1 presents information about species and habitat associations of spring chinook salmon and rainbow trout, particularly as they relate to limiting factors and species interactions. The main topic of chapter 2 relate= to differential movement and colonization of fishes at three different elevations in Swauk Creek. The next seven topics are presented as updates from chapters presented in our FY 1993 annual report (Pearsons et al. 1994). Topics to be addressed in future status reports and research recommendations are included in a "General Discussion" section that follows the last update. Finally, Appendix A addresses the factors that influence the condition of rainbow trout in the Yakima River; in particular, the influence of mouth and head scars caused by angling.

The chapters, updates, and appendix are in various stages of completion and should be considered preliminary. Additional field work and/or analysis is in progress for topics covered in this report. Readers are cautioned that any preliminary conclusions are subject to future revision as more data and analytical results become available.

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; Pearsons et al. 1994).

Chapter 1

Species and habitat associations of spring chinook salmon and rainbow trout in the upper Yakima River

Abstract

We hypothesized that unnaturally high discharges in the upper Yakima River during the summer would result in unnaturally high habitat overlap of age 0+ rainbow trout and spring chinook salmon in slow water habitats. Habitat utilization, species associations, and behavioral interactions among spring chinook salmon and other species were determined in two sections of the upper Yakima River by snorkeling. One section (Cle Elum) was sampled during the spring, early summer, late summer, and fall, and the other (Lower Canyon) during the late summer and fall. Fishes were also collected with a backpack electrofisher to compare sizes of age 0+ rainbow trout and spring chinook salmon collected from different locations. Age 0+ rainbow trout and spring chinook salmon were closely associated with bank habitats in both **mainstem** and off channel habitats. However, contrary to our hypothesis, spring chinook and rainbow trout were most closely associated with each other during the fall when discharges were lowest. High species associations corresponded to shrinking stream widths and to increasing overlaps in fish sizes. Spring chinook salmon dominated rainbow trout in 52% of all the behavioral contests, rainbow trout dominated spring chinook in 38% of the contests, and 10% of the contests had no apparent winner. Large fish generally dominated small fish and spring chinook salmon were generally larger than age 0+ rainbow trout. These preliminary results suggest that stream discharge and spring chinook salmon and age 0+ rainbow trout sizes interact to influence habitat overlap and behavioral interactions. Lack of correspondence between fish and habitat variables and the patchy distribution of fish we observed suggest that information on fish and habitat relationships at a smaller scale would be a priority area of investigation.

Introduction

Flow regimes can significantly affect stream fish assemblage structure, biotic interactions, and habitat associations (Poff and Ward 1989; Horwitz 1978; Pearsons et al. 1992). As a result, assemblage structure, biotic interactions and habitat associations observed in rivers with natural flow conditions may not be similar to r'ivers with manipulated flows (Bain et al. 1988; Shirvell 1990; 1994). Consequently, observations that have been made among juvenile chinook salmon (*Oncorhynchus tshawytscha*) and *0. mykiss* (resident rainbow trout and steelhead trout; hereafter referred to as rainbow trout) at natural flows may not apply to situations below dams with manipulated flows.

At natural flows, studies have indicated that sympatric populations of juvenile chinook salmon and rainbow trout occupy different microhabitats during the day (Everest and Chapman 1972, Hillman et al. 1989a) and night (Hillman et al. 198933). An explanation for this disparity in habitat use may be differences in the sizes of age 0+ spring chinook salmon and rainbow trout. Spring chinook salmon generally emerge earlier than rainbow trout and thus are also generally larger than rainbow trout that emerged in similar areas. Most authors believe that these species select microhabitats (i.e. depth, velocity) based on their size (Everest and Chapman 1972; Hillman et al. 1989a). For instance, as rainbow trout and chinook salmon grow, they select faster water velocities and deeper water. Differences in habitat selection may also be related to differences in body morphology such as body depth (Bisson et al. 1988). The lack of habitat overlap between spring chinook salmon and rainbow trout, and the lack of agonistic behavioral interactions observed has led some authors to suggest that these two species do not compete for space (Everest and Chapman 1972; Hillman et al. 1989a; 198933).

We hypothesized that unnatural discharges in the upper Yakima River (Johnson 1994) would increase the potential for competition between age 0+ rainbow trout and juvenile spring chinook salmon. Discharges in the upper Yakima River between Roza Dam and the Cle Elum River confluence are typically relatively moderate during the late spring, high during the summer, and low during the fall. In contrast, natural flows would be high during the spring and decrease during the summer and fall. If large releases of water from irrigation dams causes limited areas of slow water (<30 cm/s) then age 0+ salmonids may be forced to occupy the limited amount of areas with slow water. Shirvell (1990) reported that juvenile coho salmon and rainbow trout selected areas below instream root wads because the root wads provided refuge from fast water velocities. If these fishes are forced into close proximity with one another, they may compete for food and or space.

The goals of this study were to 1) determine if age 0+ rainbow trout and juvenile spring chinook salmon occupy slow water habitats, 2) determine if these habitats are in short supply, 3) determine the frequency that these species are associated with one another, and 4) determine if they behaviorally interact and if so who dominates. Results and interpretations should be considered preliminary pending further data collection and analysis.

Methods

Study Area

Three water storage reservoirs in the upper portion of the Yakima basin (above Roza Dam, rkm 180) are managed to supply water to irrigators located primarily below Roza Dam. This management scheme results in unnatural water flows in the river above Roza Dam where spring chinook and rainbow trout commonly occur. Discharges during the summer have been drastically increased as a result of flow management. In addition, the abundance of side channels and sloughs has decreased through time as a result of bank stabilization (rip-rap), filling sloughs and side channels, scouring the main channel by splash damming, and reduction of seasonal high flows that might create multiple channels (Johnson 1994).

Sampling Design

Habitat utilization, species associations, and behavioral interactions among spring chinook salmon and other species were determined in two sections of the Yakima River. These sections were located near the town of Cle Elum and in the Yakima Canyon. These sections were selected because of their high densities of spring chinook salmon and/or rainbow trout (Fast et al. 1991; Martin et al. 1994). The Cle Elum section was sampled at different discharges from the late spring through the fall, whereas the Yakima Canyon section was sampled only during the late summer and early fall because of poor water visibility during the late spring and early summer.

We counted fishes, determined species associations, and observed behavioral interactions while snorkeling. In the Cle Elum section, sampling occured during four periods: spring (May 23, 24, 25, 27), summer 1 (June 29, 30, July 8, 11, 12, 14, 15, 18, 19), summer 2 (August 15, 16, 19), and fall (September 12, 13, 14, 15). In the Lower Canyon section, sampling occurred during one period: summer 2 (August 22, 24, 25). Snorkelers and gear were transported between sites with an inflatable raft. Stream sites to be snorkeled within each reach were selected using random starting points and systematically sampling thereafter. For example, a random number, representing the number of minutes to travel downstream before sampling the first mainstem location was selected prior to rafting. After the first mainstem location was sampled, subsequent locations were sampled at systematic intervals (generally 10-15 minutes). Side channels 'and slough sites were selected and sampled in similar ways except that the number of units were sampled systematically by count as opposed to the amount of time (generally every third unit). All sites that were sampled were flagged to avoid resampling sites between sample periods. If a site was selected that had previously been sampled, a site immediately downstream was sampled. On occassion, side channels and sloughs were sampled during more than one sample period because of their limited abundance.

After a site was selected, both banks and the center of the channel were snorkeled. Sites ranged in length from 13 to 200 m but were generally 50 m. The length of the site was dependent on the length of homogenous habitat at the site. Only fish that were within 1 m of the bank or within a 1 m swath in the center of the channel were counted. Along the banks, fish were counted or observed while the snorkeler moved slowly upstream. In the center of the channel, fish were counted while floating downstream, unless water velocities were slow and water depth shallow enough to snorkel upstream. Multiple snorkeling passes were made in the center because fish were more difficult to count when moving quickly downstream. Multiple counts were averaged.

All fishes were counted and their age class-(adult, juvenile, or age 0+) recorded on white plexiglass slates that were attached to the snorkelers' arm. In addition, pods of fish, defined as two or more fish within 30 cm of another were also recorded. Pods of fish were used to determine species associations. Behavioral interactions between species were recorded when undisturbed pods of fish were observed. We selectively chose pods of fish with more than one species to bolster observations of interspecific interaction. Variables that were recorded during behavioral interactions included; 1) the initiator of the interaction, 2) the relative sizes of the fish that were interacting (large vs. small), 3) the type of interaction (butt, nip, chase, display - described in detail by McMichael et al. 1994), and 4) the outcome of the interaction (which fish was dominant).

To supplement the observations of species associations and behavioral interactions, we drift snorkeled in the lower and upper Yakima Canyon (August 29 and 30, September 22, 27, and 28) and the Cle Elum section (September 20 and 26). Drift snorkeling was conducted by floating downstream and generally occured along the banks of the river. Snorkelers would attempt to sample the best habitat so that the maximum number of fish could be observed. Once fish were observed, the snorkeler would stop and snorkel upstream until the fish could be re-observed. Habitat information was not collected during drift snorkeling.

Age 0+ spring chinook salmon and rainbow trout were collected to determine if size differences were similar among different sites in the upper Yakima River basin. Fishes were collected in the Lower Canyon of the Yakima River, Cle Elum section of the Yakima River, Manastash Creek, and Teanaway River during the early fall (9/6/94 - 9/22/94) using a backpack electrofisher. Rainbow trout that were shorter than 110 mm FL were determined to be age 0+ (Martin and Pearsons 1994).

Habitat measurements were taken at the time of the fish surveys and also during habitat availability surveys. Habitat variables measured or described during snorkeling surveys were unit type (mainstem, side channel, slough), location snorkeled (bank or center), length of site, average velocity, presence or absence of rip-rap, temperature, and discharge. Side channels were defined as flowing water channels that were estimated to be less than 25% of the flow volume of the river. Side or split channels that were greater than 25% of the flow were treated as mainstem units. Sloughs were defined as discontinuities in the river channel which had no flow at the upstream ends and abundant slack water. Average water velocity was determined by assessing the time taken for an orange to float a distance of 10 m through a representative portion of the snorkeled reach (orange time). Orange float times were calibrated by comparing orange times to water velocities taken in the same location using a Marsh-McBirney flow meter. Water velocities were measured at the water surface, 0.6 of water depth (measured from the surface - represents average velocity), and on the bottom. A regression model was developed to determine water velocities from orange float times.

The availability of fast and slow water sites was determined by systematically sampling 100 m long sites in the mainstem in a manner similar to that described in snorkel sites. Side channel and slough habitats were also sampled systematically by count as described for the snorkel sites. Briefly, water velocity at each bank and center location within a site was visually determined as fast or slow. The Cle Elum section was sampled on May 26, August 11, and September 16, 1994. The Lower Canyon section was sampled on August 26, 1994.

Results

Habitat Associations

Age 0+ spring chinook salmon and rainbow trout were closely associated with banks during the spring and summer but as they grew they became less associated with banks. Age 0+ salmon and trout were found in much greater densities along mainstem banks than in the center of the mainstem (Table 1 and 2). With the exception of the fall sample, they were also found in higher densities along banks than in the center in side channel habitats (Table 1 and 2). Juvenile and adult trout were not clearly associated with either banks or the center of the channel (Table 3).

Bank habitats in side channels and sloughs appeared to be more important to age 0+ trout and salmon when they were small than when they were large. Densities of young trout and salmon were generally higher in either side channel or slough bank habitats than **mainstem** ones during the spring and summer and lower during the fall in the Cle Elum section (Table 1 and 2). In the lower Yakima Canyon, where age 0+ rainbow trout and salmon are large relative to those in the Cle Elum section, densities were highest along **mainstem** banks (Table 1 and 2). No clear patterns were detected with other habitat variables measured (Table 1, 2, and 3). Average water velocity was correlated most strongly with orange float times so it was used to construct a regression model. The relationship between orange float time and water velocity is as follows:

$$\text{average water velocity (m/s)} = 8.66\,(\text{orange float s})^{-1.0629}$$

Thus, an orange float time of 30 seconds corresponds to an average water velocity of 0.23 m/s.

Spring chinook salmon distribution within a section was very patchy. For instance, a snorkeler may not see any spring chinook salmon while snorkeling 200 m of the river and then observe 50 salmon in a single pod within one m.

Mainstem habitats and the centers of side channels had primarily fast water velocities (Table 4). Less than 50% of the banks of side channels in the Cle Elum section had fast water velocities (Table 4). Sloughs, by definition, had slow water velocities. In the Cle Elum section, the percentage of banks having slow water increased after discharge reduction in September. Discharge decreased rapidly during the second week of September (Table 4). This rapid decrease in discharge undoubtedly caused some fish to die because they were stranded. Fourteen percent (3/21) of the side channels and sloughs that were sampled prior to the discharge decrease were dry on September 28, 1994. Five percent (1/21) were disconnected from the main river channel and had no water velocity.

Table 1. Average density (#/m^2) of age 0+ and 1+ spring chinook salmon in different habitats (fast > 0.23 m/s) and sections (CELUM = Cle Elum, LCYN = Lower Yakima Canyon) of the Yakima River during different times (SPRG = spring, SUM1 = summer 1, SUM2 = summer 2, FALL = fall), 1994. "None" refers to habitat types that were not sampled because of their rarity.

	CELUM				LCYN
Habitat	SPRG	SUM1	SUM2	FALL	SUM2
Mainstem bank	0.928	0.485	0.079	0.083	0.010
Mainstem center	0.000	<0.001	0.002	0.000	0.000
Side ch. bank	0.204	0.298	0.309	0.023	0.007
Side ch. center	0.000	0.064	0.008	0.033	0.000
Slough bank	1.092	0.364	0.049	0.000	none
Slough center	2.134	0.015	0.044	0.000	none
Fast velocity	0.556	0.285	0.050	0.090	0.007
Slow velocity	0.494	0.258	0.186	0.013	0.004
Rip-rap.	0.260	0.085	none	0.000	0.002
Total fish observed	1684	1414	306	168	15

Table 2. Average density (#/m^2) of age 0+ rainbow trout in different habitats (fast > 0.23 m/s) and sections (CELUM = Cle Elum, LCYN = Lower Yakima Canyon) of the Yakima River during different times (SPRG = spring, SUM1 = summer 1, SUM2 = summer 2, FALL = fall), 1994. "None" refers to habitat types that were not sampled because of their rarity.

	CELUM				LCYN
Habitat	SPRG	SUM1	SUM2	FALL	SUM2
Mainstem bank	0.000	0.062	0.016	0.007	0.008
Mainstem center	0.000	0.000	0.000	0.000	0.000
Side ch. bank	0.002	0.070	0.074	0.002	0.003
Side ch. center	0.000	0.006	0.000	0.004	0.000
Slough bank	0.000	0.005	0.000	0.000	none
Slough center	0.000	0.004	0.000	0.000	none
Fast velocity	0.000	0.027	0.014	0.014	0.005
Slow velocity	0.001	0.063	0.032	0.004	0.004
Rip-rap	0.000	0.062	none	0.000	0.000
Total fish observed	3	218	41	22	9

Table 3. Average density (#/m^2) of juvenile and adult rainbow trout in different habitats (fast > 0.23 m/s) and sections (CELUM = Cle Elum, LCYN = Lower Yakima Canyon) of the Yakima River during different times (SPRG = spring, SUM1 = summer 1, SUM2 = summer 2, FALL = fall), 1994. "None" refers to habitat types that were not sampled because of their rarity.

	CELUM				LCYN
Habitat	SPRG	SUM1	SUM2	FALL	SUM2
Mainstem bank	0.005	0.002	0.008	0.004	0.006
Mainstem center	0.000	0.006	0.004	0.025	0.000
Side ch. bank	0.002	0.007	0.001	0.029	0.000
Side ch. center	0.008	0.010	0.005	0.015	0.000
Slough bank	0.006	0.005	0.000	0.000	none
Slough center	0.000	0.000	0.000	0.000	none
Fast velocity	0.007	0.004	0.006	0.032	0.003
Slow velocity	0.003	0.006	0.001	0.004	0.002
Rip-rap	0.018	0.000	none	0.004	0.000
Total fish observed	19	25	9	45	8

Table 4. Physical parameters of sites that were snorkeled or inventoried during 1994. Available habitat was calculated as the percent of sampled sites containing fast or slow habitats (all slough habitats were slow). Different habitats (fast > 0.23 m/s) and sections (CELUM = Cle Elum, LCYN = Lower Yakima Canyon) of the Yakima River were sampled during different times (SPRG = spring, SUM1 = summer 1, SUM2 = summer 2, FALL = fall), 1994.

Physical parameter	CELUM				LCYN
	SPRG	SUM1	SUM2	FALL	SUM2
Avg. discharge (m^3/s)	22	87	73	13	75
Avg. temperature (°C)	14	16	18	15	17
Length snorkeled (m)	3704	5919	2339	3079	2958
	Number of units sampled				
# mainstem bank	17	34	12	18	24
# mainstem center	10	16	7	9	12
# side ch. bank	23	28	8	10	6
# side ch. center	6	15	4	6	3
# slough bank	6	6	4	4	0
# slough center	3	3	2	2	0
# fast velocity	22	63	23	17	36
# slow velocity	43	39	14	32	9
# Rip-rap	2	4	0	5	7
	Available habitat				
% mainstem bank fast	75		87	69	86
% mainstem bank slow	25		13	31	14
% mainstem center fast	100		93	100	100
% mainstem center slow	0		7	0	0
% side ch. bank fast	42		33	0	75
% side ch. bank slow	58		67	100	25
% side ch. center fast	83		100	0	100
% side ch. center slow	17		0	100	0

Size structure

Despite significant differences in the average size of age 0+ spring chinook salmon and rainbow trout, there was considerable overlap in size of this age class between these two species. Average sizes of spring chinook salmon were significantly larger than age 0+ rainbow trout in mainstem sections (LCYN and CELUM) but not in tributary sections (MAN and MST; Table 5). In both mainstem and tributary sections, the range of fish lengths overlapped (Table 5). The average length of spring chinook salmon and age 0+ rainbow trout was greatest in the Lower Canyon section of the Yakima River (Table 5).

Table 5. Comparisons of size (mm FL) between and among age 0+ rainbow trout (RBT; FL <110 mm) and spring chinook salmon (SPC) collected during the fall sample period. Fish were collected with a backpack-electrofisher on September 20 and 22, 1994 in the Lower Canyon section (LCYN) of the Yakima River, September 20, 1994 in the Cle Elum section (CELUM)of the Yakima River, September 6, 1994 in Manastash Creek section 1 (MAN), and September 22, 1994 in the **mainstem** Teanaway (MST).

Site/Species	Length (mm)					t-test
	Min	Max	Avg	S.D.	N	P
LCYN RBT	64	109	90.9	11.2	80	0.032
LCYN SPC	75	111	96.9	10.2	20	
CELUM RBT	54	101	81.1	12.6	18	0.013
CELUM SPC	75	97	88.3	5.5	26	
MAN RBT	72	95	83.6	6.5	17	0.059
MAN SPC	78	105	88.5	8.1	17	
MST RBT	56	104	78.9	12.7	43	0.361
MST SPC	75	92	81.9	5.0	16	

Species associations

Associations among spring chinook salmon, rainbow trout, and **redside** shiners were generally highest during the fall in both the Cle Elum and the Yakima Canyon sections of the Yakima River (Table 6). Rainbow trout and **redside** shiner were rarely if ever associated with each other except during the fall (Table 6). In contrast, spring chinook salmon were frequently associated with **redside** shiner even during the spring and summer (Table 6).

Table 6. Species associations (%) among juvenile spring chinook salmon (SPC, age 0+), rainbow trout (RBT, all age classes), and redside shiner (RSS, all age classes) in two locations during different times. Percent association is defined as the number of times that SP1 was observed within the same pod as SP2 divided by the number of times that at least one individual of SP2 was oberved. (CELUM = Cle Elum, LUCYN = Lower and Upper Yakima Canyon, SPRG = spring, SUM1 = summer 1, SUM2 = summer 2, FALL = fall).

SP1/SP2	CELUM				LUCYN	
	SPRG	SUM1	SUM2	FALL	SUM2	FALL
RBT/SPC	3	14	6	28	4	23
RSS/SPC	11	6	24	25	4	17
SPC/RBT	0	0	0	24	8	40
RSS/RBT	70	43	44	11	3	20
SPC/RSS				61	14	60
RBT/RSS	0	0	0	32	7	40

Interactions

A total of 47 contests were observed among pods of spring chinook salmon, rainbow trout, and redside shiner. These contests were observed during approximately 225 minutes of sampling between May 24 and September 27, 1994. All but two of the contests resulted in displacement of the subordinate fish, and in all but two contests the fish initiating the contest dominated. The types of interactions observed were; chases, nips, butts, threats, and crowds.

Larger, fish generally dominated smaller fish of the same species. Larger spring chinook salmon dominated smaller ones in 80% (12/15) of the contests. Larger rainbow trout dominated smaller ones in 100% (6/6) of the contests. The proportion of contests observed between conspecifics (49%) was similar to that between spring chinook salmon and rainbow trout (45%) (Table 7), although our sample was probably biased towards observing mixed species interactions because we preferentially selected pods of fish with more than one species.

Spring chinook salmon dominated rainbow trout in 52% of the contests, rainbow trout dominated spring chinook in 38% of the contests, and in 10% of the contests had no apparent winner. In part, this difference in dominance appears to be related to a greater number of observations where spring chinook salmon were larger than rainbow trout (with dominance strongly related to fish size) and the dominance of spring chinook salmon when fish were of equal size. Spring chinook salmon were larger (38%;

8/21), smaller (29%; 6/21), than or equal (33%; 7/21) to rainbow trout in the contests observed. In cases where spring chinook salmon were larger than rainbow trout, spring chinook salmon dominated rainbow trout in 75% (6/8) of the contests. In cases where rainbow trout were larger than spring chinook salmon, rainbow trout dominated 83% (5/6) of the contests. When spring chinook salmon were of approximately equal size to rainbow trout, spring chinook salmon dominated 57% of the contests, rainbow trout dominated 14% of the contests, and there was no clear dominant in 29% of the contests. Contests observed between spring chinook salmon and redside shiner, and between rainbow trout and redside shiner were rare (Table 7). Of the three contests observed the salmonids dominated 67% (2/3) of the interactions.

Table 7. Behavioral interactions observed among fishes in the upper Yakima River. Observations were conducted between May 24, 1994 and September 27, 1994. SPC = spring chinook salmon, RBT = rainbow trout, and RSS = redside shiner, L = larger than other fish, S = smaller than other fish, = is the same size as the other fish. The symbol to the left of the ">" indicates dominance.

Species 1 x 2	#	%	Dominance 1>2	2>1	1=2	L>S	S>L	=>= 1>2	=>= 2>1	=== =
SPC x SPC	16	34	16	0	0	12	3	1	0	0
RBT x RBT	7	15	7	0	0	6	0	1	0	0
SPC x RBT	21	45	11	8	2	11	3	4	1	2
SPC x RSS	2	4	1	1	0	1	1	0	0	0
RBT x RSS	1	2	1	0	0	1	0	0	0	0
Total	47	100				31	7			
Percent						82	18			

Discussion

Contrary to our original hypothesis, preliminary analyses suggest that the association between rainbow trout and spring chinook salmon was lower at high flows (spring and summer) than at low flows (fall). Low-association between rainbow trout and salmon during the spring and summer is probably related to selective segregation of different habitat types by these fishes (Everest and Chapman 1972, Hillman et al.-1989). Spring chinook salmon inhabited faster and deeper water than age 0+ rainbow trout. This difference in habitat use may be because spring chinook salmon were much larger than age 0+ rainbow trout during the spring and summer (Hillman et al. 1989). Larger individuals can maintain positions in faster water velocities than small ones. In addition larger fish may occupy deeper water than small fish because they are less susceptible to aquatic predators (Power 1984).

High species association during the fall was correlated to shrinking habitat and reduced size differences between species. The wetted area of the river was much reduced as a result of reduced discharge. This decrease in area may have forced fish to occupy similar habitats. In addition, although the size of age 0+ salmon and trout were significantly different during the fall in the Yakima River, there was considerable overlap in the sizes of the two species. If habitat selection is determined most by fish size (Everest and Chapman 1972) then reduction in size differences will result in increased overlap in habitat preference; Hillman et al. (1989) also found higher resource overlap between steelhead and spring chinook salmon as the difference in fish sizes decreased during the fall. However, in contrast to our study Hillman et al. (1989 a, b) observed no behavioral interactions between rainbow trout and spring chinook salmon.

Dominance relationships and interaction rates were very similar in the mainstem Yakima River and to those reported in the Teanaway River basin. For example, spring chinook salmon dominated rainbow trout in 58% of the contests in the Yakima River and in 50% of the contests in the Teanaway River drainage (McMichael et al. 1994). In addition, the number of interactions per fish per minute was 0.00062 in the Yakima River and 0.00084 in the Middle Fork of the Teanaway River during 1993 (McMichael et al. 1994).

Redside shiner were found more often with spring chinook salmon than with rainbow trout. Hillman (1989) also found a high degree of resource overlap and interaction between redside shiners and spring chinook salmon. Furthermore, redside shiners may competitively dominate steelhead, particularly at temperatures above 19 °C (Reeves et al. 1987). However, the average water temperature did not exceed 18 °C during our sampling.

In contrast to findings by Fast et al. (1991), this study indicated that spring chinook densities were higher in the Cle Elum section than in the Yakima Canyon. Fast et al. (1991) reported that spring chinook salmon densities were approximately ten times higher in the Yakima Canyon than in the Cle Elum section during the summer. Different sampling methods between the two studies may explain the observed differences. Fast et al. (1991) sampled fish by beach seining while we sampled by snorkeling. Beach seining may be a more biased sampling method in the mainstem Yakima River than snorkeling, because it can only be done affectively in relatively slow water. Indeed, Fast et al. (1991) found fewer fish seining than snorkeling in the Yakima River around Easton. Alternative explanations for differences observed between this study and Fast et al. (1991) also include changes in habitat, migration timing, and spawning sites used during the different years that the studies were conducted.

Decreases in.the density of spring chinook salmon from spring through fall may be the result of a variety of factors including: active or passive migration from the study reach, mortality due to predation, stranding, and/or other factors. Chinook salmon have been shown to move offshore and downstream in response to discharge fluctuations caused by dams (Shirvell 1994). Furthermore, juvenile spring chinook salmon were caught in traps near the mouths of Swauk and Umtanum creeks, indicating movement of spring chinook salmon from the mainstem into these tributaries (Chapter 2). Among other things, spring chinook salmon may migrate into tributaries to avoid high discharges or to avoid predators. Northern squawfish, known piscivores, were on occasion found in close proximity to spring chinook salmon. Finally, stranding may have affected salmon densities during the fall. Stranding of salmon has been demonstrated in the Yakima River (Fast et al. 1991) and in other regulated river systems (Bradford et al. 1995 and references therein). All of the factors described above (and others) appear to contribute to declining densities between seasons of spring chinook salmon in the Cle Elum section.

Bank and off channel habitats appear to be extremely important to age 0+ spring chinook salmon in the mainstem. Most age 0+ trout and salmon were within a few meters of the stream bank within the main channel. Many fish were observed in the middle of side channels presumably because the influences of the banks extended into the centers of the channel (small channel widths). In addition, water velocities in many side channels were lower than in the main channel. Side channel habitats are extremely important to rearing fish because with each additional side channel the effective bank habitat is approximately doubled within the length of stream containing the side channel. Bank and off channel habitats may be important to fish because water velocities are generally lower than in the middle of the channel, hydraulic refuges more frequent, and instream and overhead cover are more abundant which can minimize exposure to predators. Bank and side channel habitats have also been shown to be of importance to spawning rainbow trout (Martin et al. 1994) and age 0+ cutthroat trout (Moore and Gregory 1988). Long-term monitoring of fish populations should include measurement of the quality and quantity of edge and off channel habitats to help explain variations in fish population abundance.

Lack of correspondence between habitat measurements and fish abundance suggests that we may have examined this relationship at a scale that was too coarse, we measured the wrong habitat variables, or there was truly no correspondence. Fish abundance was extremely patchy within a sampled reach. Often the fish were located within a very small area within the reach (< 2 m) and appeared to be selecting micro-locations within the reach. Because fish appear to be selecting habitat at a smaller scale than we examined, we recommend that future habitat and fish relationships be examined at a smaller scale (a few meters). In addition, sampling in the center of the mainstem is probably unnecessary, since almost all of the fish observed were located within a few meters of the river bank.

Chapter 2

Movement of fishes along an elevational gradient within Swauk Creek

Abstract

Contrasting views about how much fishes move may be partially explained by the environmental conditions that the fishes encounter. We tested the hypothesis that fish movement and immigration was equal at three Swauk Creek sites that differed in elevation. In addition, we evaluated how fish immigrations affected assemblage composition estimates in index sites that were sampled annually. Finally, we evaluated whether electrofishing affected fish movement. Fish movement was assessed using panel weirs and traps, from June 14 to October 26, 1994, that were located above and below 100 m long index sites. The number of fish and taxa that were captured moving up or downstream was negatively related to the elevation of the index site. We collected 11,249 individuals and 10 taxa at the lowest elevation site, 3,029 individuals and 8 taxa at the middle elevation site, and 113 individuals and 3 taxa at the highest elevation site. Immigrations of rainbow trout and spring chinook salmon were also negatively related to site elevation. Most of the fishes captured at the low and middle elevation sites were cyprinids. At the highest elevation site, cottids and salmonids were the most prevalent. Despite large differences in numbers of fish moving into sites, annual variations in assemblage compositions measured during three summers did not appear to be different among sites. The differences in the abundance and diversity of fishes migrating within portions of Swauk Creek is related to a suite of environmental variables that are correlated with elevation, such as distance from a mainstem immigration source, discharge, and temperature. Electrofishing did not appear to influence fish movement substantially. Differences in movement that we detected at sites differing in elevation suggest the need to monitor these sites in different ways.

Introduction

The degree to which fishes move in streams has critical implications for how fish assemblages should be monitored and managed. If fishes are sedentary then they can be managed at small spatial scales, whereas if fishes are highly mobile then they must be managed at large spatial scales to provide adequate habitat and free access to move within river systems. In addition, the amount of fish movement affects our perception of fish assemblages in reaches of streams as relatively open or closed biological systems which can affect decisions about management actions such as stream diversions/blockages, hatchery fish release sites, instream flow requirements, and habitat enhancement priorities and scale.

There has been much disagreement about the extent to which fish move in streams. Many authors have suggested that fish movement is relatively small (Bangham and Benington 1939; Gerking 1953; 1959; Hill and Grossman 1987). This viewpoint is exemplified by Gerking 1953, who suggested that in streams with riffle-pool development, riffles are boundaries that fish do not traverse. In contrast, Gowan et al. (1994) suggested that a paradigm shift is occurring toward the view that movement in salmonids is substantial. Others have suggested that whole assemblagesof fish move (Hall 1972; Schlosser 1982; Decker and Erman 1992; Pearsons 1994).

Some have claimed that biases in techniques and analyses have been largely responsible for the differences in study interpretations (Funk 1955; Gowan et al. 1994). While techniques and analyses have differed, ecological conditions may also influence the amount of movement that is expressed (Pearsons 1994). In other words, fishes may move large distances in some areas and may be rather sedentary in other areas. Pearsons (1994) suggested that the diversity and density of fishes in a tributary to the John Day River was strongly influenced by migrations of fishes and that the magnitude and timing of fish migrations was strongly influenced by the distance from an immigration source. In short, fish assemblages close to an immigration source were highly influenced (density and diversity) by fish migrations and these migrations occur primarily during the spring and early summer. In contrast, fish assemblages that are far from an immigration source'are less influenced by fish migrations and these migrations occur later than at areas that are -close to an immigration source.

We wanted to test whether fish movement was equal in three sections of a stream that varied in elevation and distance from a mainstem source of immigrants. In addition, we wanted to determine if fish migration influenced assemblage composition among years in three 100 m long sites. Finally, we wanted to determine if electrofishing influenced fish movement.

Methods

Study area and background

Swauk Creek is a third order tributary to the Yakima River, located in central Washington. Swauk Creek flows through a basalt canyon in its lower reaches, and is surrounded by alders and conifers in the upper reaches. Discharges are typically highest during the spring and lowest during the summer. Various land and water uses have and continue to influence the Swauk Creek basin. Mining, forestry, agriculture, and ranching are the primary resource activities conducted in the basin. Water withdrawals for irrigation have occured since at least 1936, and can cause the lower portions of Swauk Creek to go dry during the summer (Bryant and Parkhurst 1948). Salmonids, Cottids, Cyprinids, Catostomids, and Petromyzontids inhabit Swauk Creek.

Study design

Fish movement at three sites in Swauk Creek was assessed using panel weirs and traps. These sites were located at different elevations (579, 732, 902 m) and at different distances (1.9, 14.3, 22.7 rkm) from the confluence with the Yakima River (a presumed source of fish immigrants). Trapping began at the time when discharges were low enough to install traps (June 14-17, 1994) and terminated when high discharges dismantled them (October 26, 1994). Two-way “V" weir traps were located approximately 50 m above and below three 100-m long index sites in Swauk Creek to determine upstream and downstream fish movement. Panels were 0.9 m tall and were constructed out of 0.63 mm hardware cloth attached to a frame constructed of 5.1 x 5.1 cm lumber. Panels were attached to reinforced iron bar that was pounded into the substrate. An extra 30 cm of hardware'cloth that was attached to the panels was buried in the substrate to prevent fish passage. Both weirs at a location funneled fish into a trap that was partitioned so that fish moving up and downstream could be distinguished.

During periods when most fish movement was observed, traps were cleaned and checked two times per day. As the number of fish captured in traps decreased the frequency of trap checking and cleaning decreased. During August the traps were only checked twice per week. At each checking, fish were anesthetized, identified to species, counted, and the direction of movement recorded. Sculpins (Cottus sp.) were identified to species but were later lumped to genus because of questionable identifications. As time permitted subsamples of fish were measured (FL mm), weighed, and assessed for reproductive condition. Rainbow trout (*Oncorhynchus mykiss*) and spring chinook salmon (0. tshawytscha) were fin clipped with a trap specific clip to determine how many of them moved into 100 m index sites. Fish were released approximately 30 m away from the trap in the direction of their movement.

Analysis of upstream fish movement at each elevation was assessed using data collected from the trap located below each index site. Downstream fish movement at each elevation was assessed using data collected from the trap located above each index site.

Fish assemblage composition and temporal constancy was assessed during the month of August from 1992 to 1994 in three 100 m long index sites (Update 4). Block nets were installed at the up and downstream ends of the site to prevent immigration and emigration. Fishes were collected by netting fish that were stunned by a backpack electrofisher. Two electrofishing passes were conducted through each site. During 1992, most of the stream bed was dry in the lowest elevation site and fish were censused by snorkeling the existing water, which consisted of two pools. Fishes were identified to species except for sculpins which were identified-to genus. Site area, mean width, maximum depth, streambed profile (SD of thalweg depth), stream gradient, % pool, % riffle, % run, and discharge were determined on the date that index sites were sampled using methods presented in Update 3.

Creek discharge and temperature were measured throughout the trapping period in 1994. Thermographs were used to determine temperature at each of the sites. However, our thermograph from site 3 was dislodged and swept away, so no temperature data was available for that site. Discharges were determined by measuring' stream heights on permanently positioned staff gauges that were calibrated with a Marsh-McBirney flow meter.

Results

Number and type of fish

The number of fish that were captured moving up or downstream was negatively related to the elevation of the index site. The most fish captured moving up and downstream were at the lowest elevation site (11,249; Table 1). Furthermore, more fish were caught migrating up and downstream at index site 2 (3,029; Table 1) than at site 3 (113; Table 1).

Most individuals that were captured in sites 1 and 2 were Cyprinids and in site 3 were Salmonids and Cottids (Table 2). Longnose and speckled dace were the most abundant species captured at sites 1 and 2 (Table 1). Speckled dace migrated upstream into or through sites 1 and 2 at similar times as longnose dace were migrating downstream. More rainbow trout and spring chinook salmon were caught migrating upstream than downstream (Table 1). In addition, the lower the site elevation the more rainbow trout moved up or downstream (Table 1).

At least eighty-five percent of the fish that we captured were adults and juveniles (Table 3). Over 85% of fish captured at site 2 were adults. Many of the adults captured at site 1 and 2 exhibited spawning coloration, breeding tubercles, or expelled gametes when pressure was gently applied. The percent composition of age 0+ rainbow trout that were captured was negatively related to elevation (Table 3).

Timing of fish movement

The peak of fish movement at all sites occurred at or before June 20 with the possible exception of site 1 (Figure 1). At site 1, there appeared to be two peaks. The first peak was at or before June 20, and the second peak was between July 19, and August 1 (Figure 1). The second peak was associated with the dewatering of the stream at site 1 (Figure 2). The upstream end of site 1 was dry on August 4, 1994. All of the other sites had flowing water throughout the study period. Initial discharges at site 3 were the lowest (Figure 2). Fish movement continued throughout summer base flow conditions.which began in the middle of July. However, most of **the fish** movement occurred before the end of August (Figure 1). Most rainbow trout movement occurred before August 1 (Figure 3).

Initially, maximum water temperatures and discharges were higher in site 1 than in site 2 (Figure 2). Water temperatures were relatively low and constant between the beginning of August and October in site 1 presumably because the water was influenced by subsurface cooling.

Directionality of fish movement

Patterns in directional movement were found at the two lowest sites but not at the highest site. Most fish that were captured at site 1 were moving downstream, at site 2 upstream, and at site 3 no clear directionality was detected (Table 1). At site 1, the number of fish moving downstream was almost twice as high as the number of fish moving upstream. In contrast, the number of fish moving upstream was more than 10 times as high as the number of fish that were moving downstream at site 2. At site 3, the numbers of fish moving up and downstream were similar.

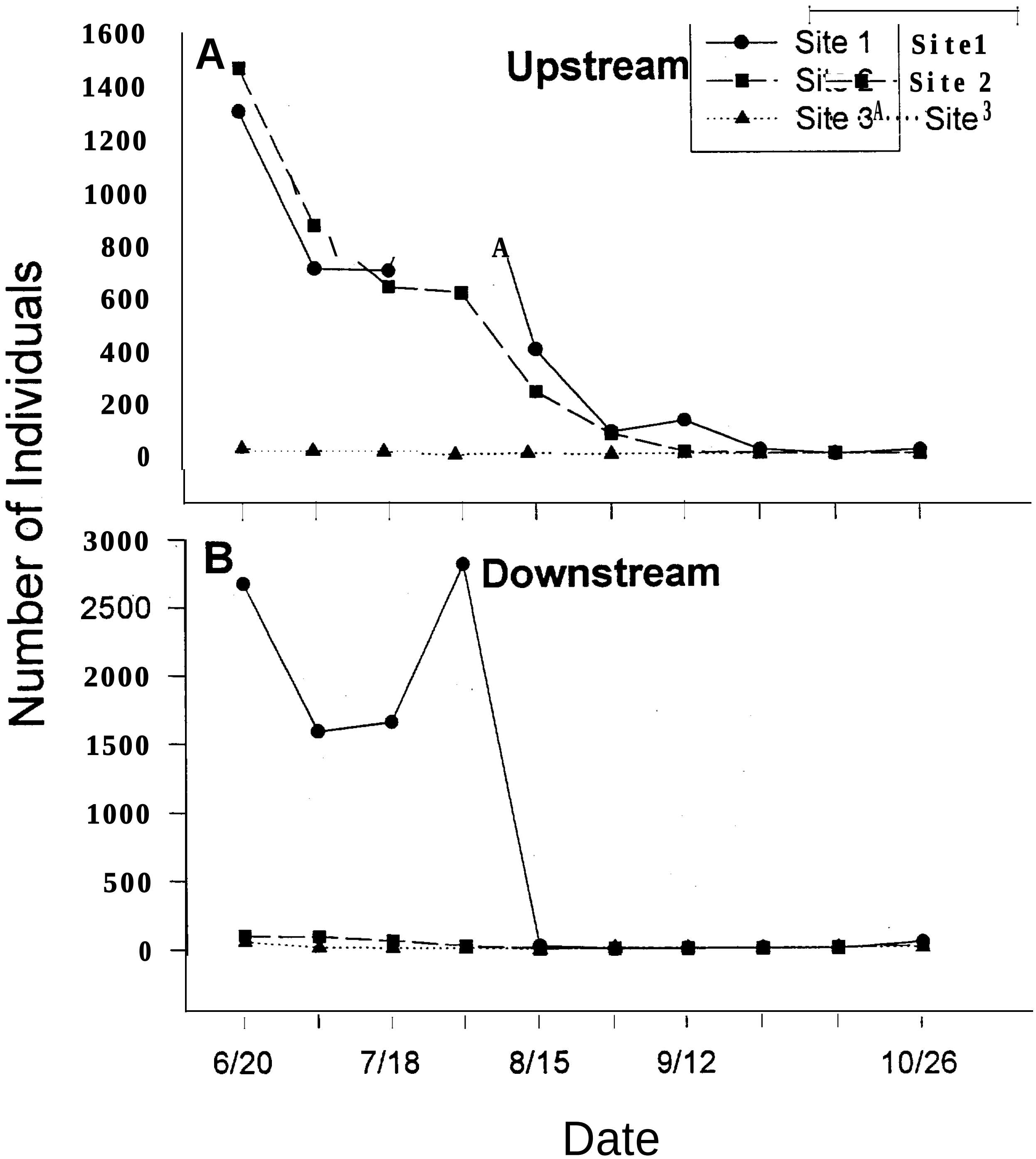

Figure 1. Number of fish migrating A) upstream or B) downstream between June 14-17 to October 26, 1994 in three sites of Swauk Creek. Data presented for June 20 was extrapolated to represent two weeks. Actual data for June 20 was: 559 fish collected moving up (up) and 1147 fish collected moving down (down) at site 1 during 6 days; 314 up and 22 down at site 2 during 3 days; and 6 up and 12 down at site 3 during 3 days.

Table 1. Numbers of fish and **taxa** moving up and downstream at three sites in Swauk Creek, 1994. The following codes were used: RBT = rainbow trout, CUT = cutthroat trout, SPC = spring chinook salmon, SPD = speckled **dace,** LND = **longnose dace,** RSS = **redside** shiner, **SQF** = northern squawfish, SCP = sculpin species, BLS = bridgelip sucker, LSS = largescale sucker, **BRL** = brook lamprey.

Age	RBT	CUT	SPC	SPD	LND	RSS	SQF	**SCP**	BLS	LSS	BRL	Total	**Taxa**
						Site 1 **-** up							
Adult	23	0	0	2030	99	139	0	136	4	0	0	2432[a]	
Juven.	4	0	0	309	387	42	33	22	111	20	0	928	
Age Ot	201	0	329	13	28	**0**	**0**	6	28	2	0	607	
Total	228	0	329	2352	514	181	33	164	143	22	0	3967	9
						Site 1 - down							
Adult	12	0	0	773	2201	157	1	31	0	0	0	3175	
Juven.	1	1	0	213	3737	15	2	**9**	15	4	0	3997	
Age 0+	68	0	10	3	15	**0**	**0**	2	11	**1**	0	**110**	
Total	81	1	10	989	**5953**	172	3	42	26	5	0	7282	10
						Site 2 - up							
Adult	17	3	0	35	2245	**0**	**0**	**259**	0	0	1	2560	
Juven.	8	0	0	5	176	0	0	19	1	0	0	209	
Age 0+	16	0	2	0	0	0	0	1	0	0	0	19	
Total	41	3	2	40	2421	0	0	279	1	0	1	2788	8
						Site 2 - down							
Adult	5	0	0	26	131	**0**	**0**	45	2	0	2	211	
Juven.	4	0	0	1	4	**0**	**0**	7	0	0	0	16	
Age 0+	8	0	0	0	**0**	**0**	**0**	3	3	0	0	14	
Total	17	0	0	27	135	**0**	**0**	55	5	0	2	241	6
						Site 3 - up							
Adult	4	**1**	0	0	**0**	**0**	**0**	17	0	0	0	22	
Juven.	**11**	**3**	0	0	**0**	**0**	**0**	3	0	0	0	17	
Age Ot	7	0	0	0	**0**	**0**	**0**	**0**	0	0	0	7	
Total	22	**4**	0	0	**0**	**0**	**0**	20	0	0	0	46	3
						Site 3 - down							
Adult	4	0	0	0	**0**	**0**	**0**	42	0	0	0	46	
Juven.	3	**7**	0	0	**0**	**0**	**0**	8	0	0	0	18	
Age 0+	2	**1**	0	0	**0**	**0**	**0**	**0**	0	0	0	3	
Total	9	**8**	0	0	**0**	**0**	**0**	50	0	0	0	67	3

[a] includes 1 hybrid individual

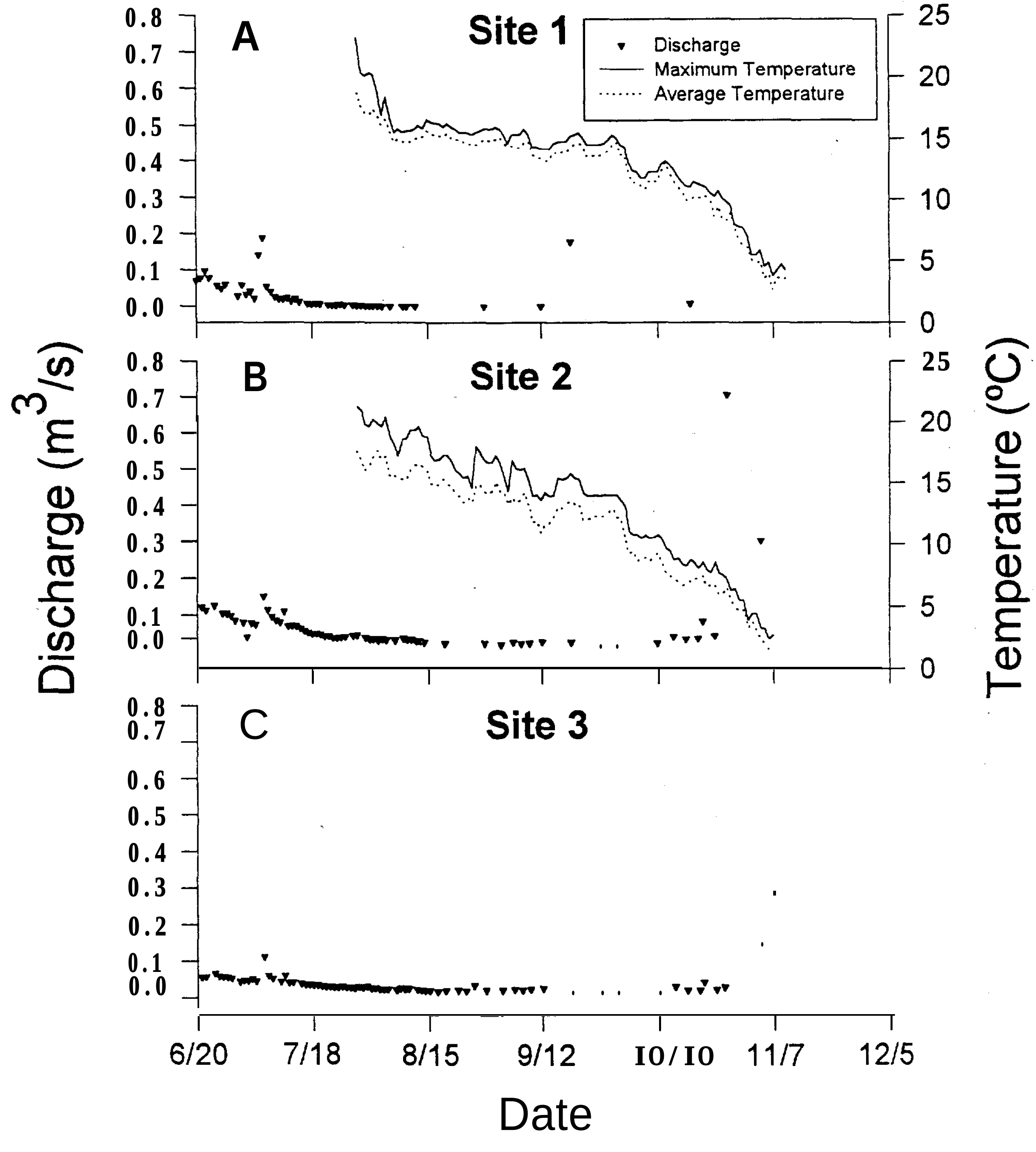

Figure 2. Discharges and temperatures measured during 1994 at three sites in Swauk Creek differing in elevation. No temperature data was available for site 3. Site 1 is located at the lowest elevation and site 3 is at the highest.

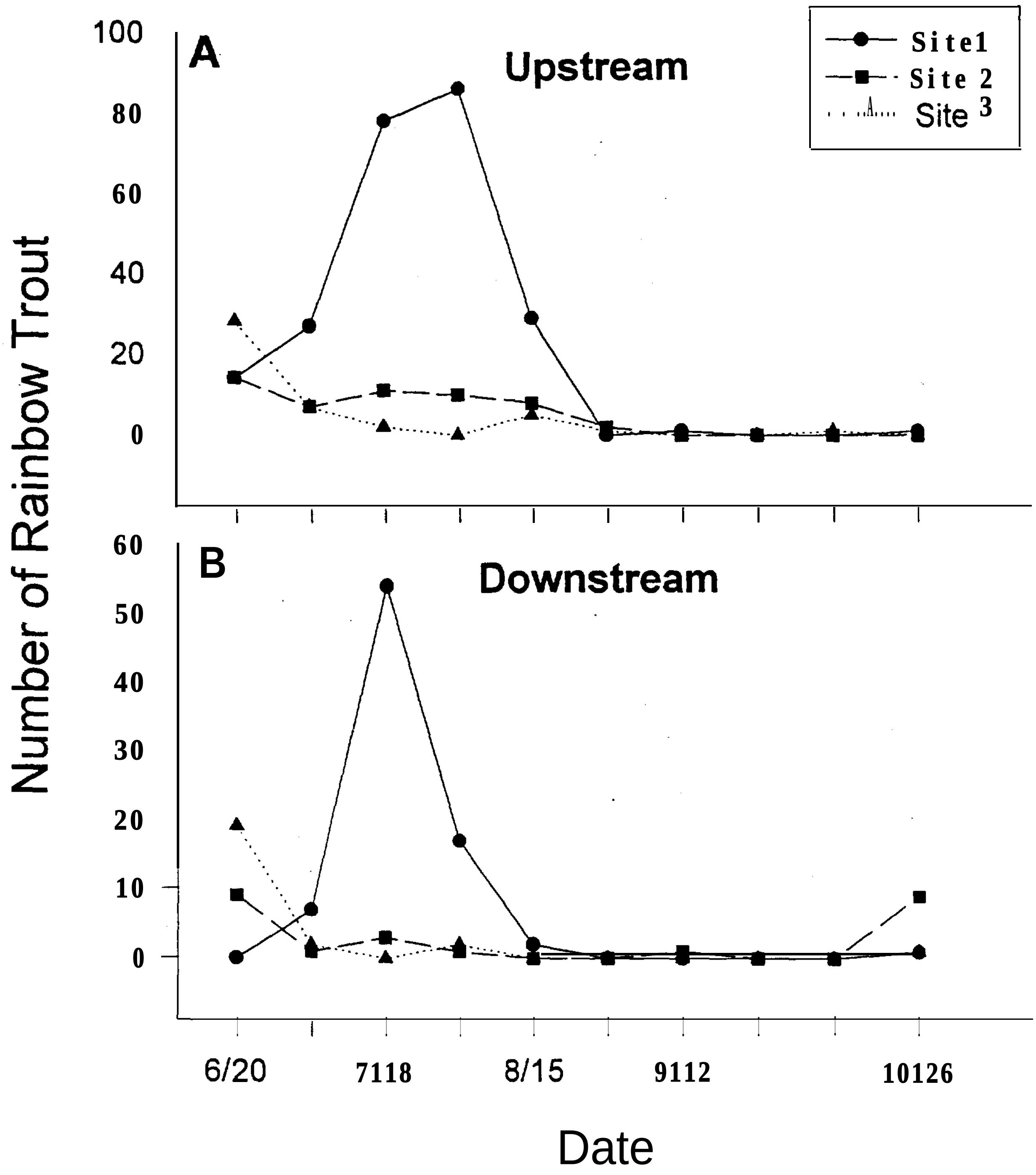

Figure 3. Number of rainbow trout migrating A) upstream or B) downstream between June 14-17 to October 26, 1994 in three sites of Swauk Creek. Data presented for June 20 was extrapolated to represent two weeks. Actual data for June 20 was: 6 rainbow trout collected moving up (up) and no rainbow trout collected moving down (down) at site 1 during 6 days; 3 up and 2 down at site 2 during 3 days; and 6 up and 4 down at site 3 during 3 days.

Number and timing of taxa movement

The number of taxa moving up and downstream was negatively associated with site elevation (Table 1). The only taxa that were captured at all three sites were rainbow trout and sculpins. The number of taxa that were captured moving up or downstream generally decreased during the study (Figure 4). However, no clear patterns were detected in site 3. During October, an increase in taxa richness occurred when discharges increased (Figure 2 and 4). Most species migrated prior to the end of August (Figure 4). The maximum number of species collected in a trap during a two week period was ten.

Table 2. The percent composition of individuals in five fish families that were captured in traps located at three sites in Swauk Creek, 1994.

Site-direc[a]	Salmonidae	Cyprinidae	Catostomidae	Cottidae	Petromyzontidae
l-up	14	78	4	4	0
l-down	1	98	Cl	1	0
2-up	2	88	<1	10	Cl
2-down	7	67	2	23	1
3-up	57	0	0	43	0
3-down	25	0	0	75	0

[a] direction of fish movement

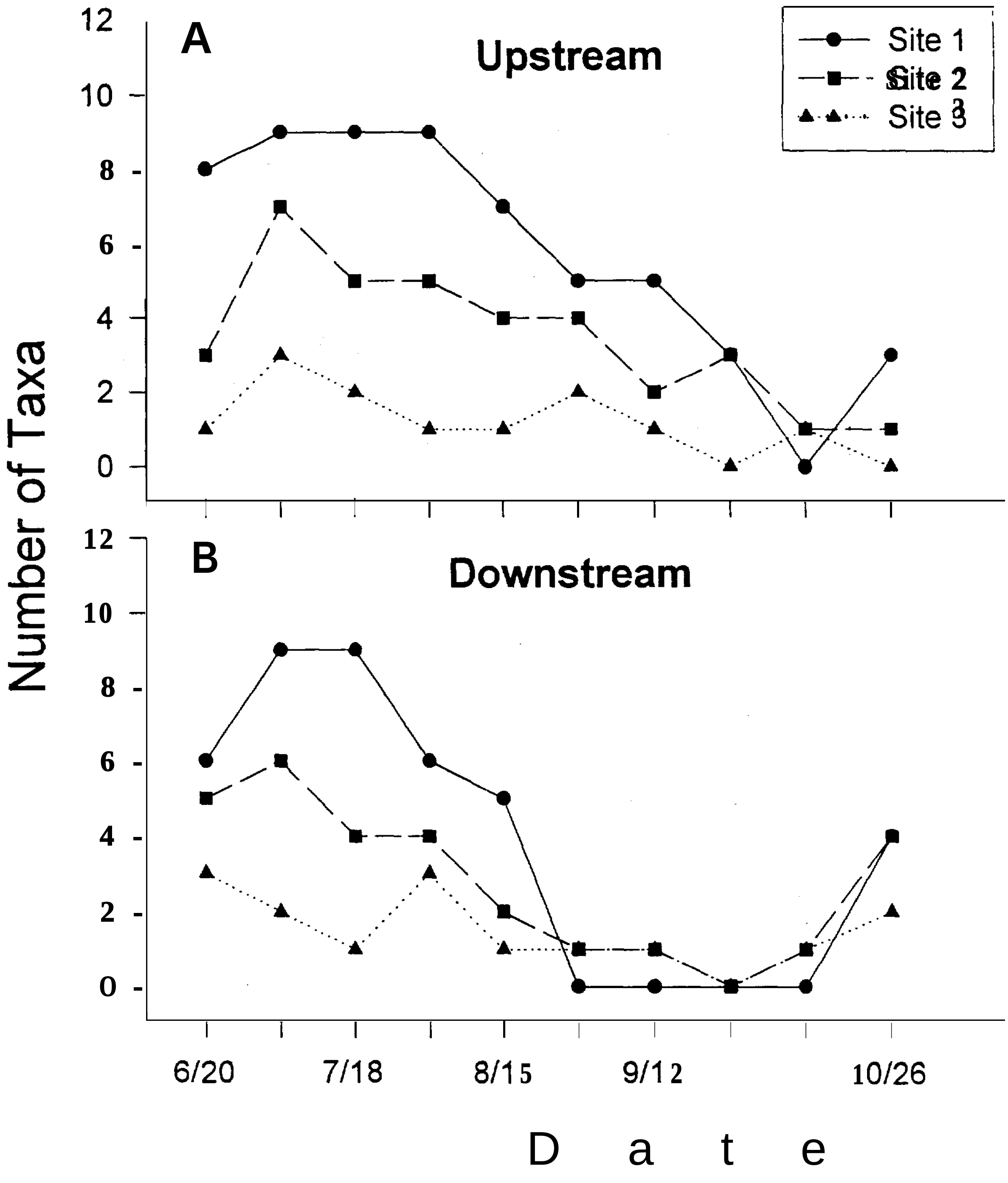

Figure 4. Number of taxa migrating A) upstream or B) downstream between June 14-17 to October 26, 1994 in Swauk Creek. Site 1 was the lowest elevation and site 3 was the highest.

Table 3. Age-class composition of all fish and rainbow trout that were captured moving up or downstream (dn) at three sites in Swauk Creek, 1994.

Age-class	Site 1 Up	Site 1 Dn	Site 2 Up	Site 2 Dn	Site 3 Up	Site 3 Dn
Total						
Adult	61	44	92	88	48	69
Juvenile	23	55	7	7	37	27
Age 0+	15	2	1	6	15	5
Rainbow trout						
Adult	10	15	42	30	18	44
Juvenile	2	1	20	24	50	33
Age 0+	88	84	40	47	32	22

Immigration

Immigration of rainbow trout and spring chinook salmon into sampled sites decreased with increasing site elevation (Table 4). Furthermore, greater than 49% of rainbow trout and spring chinook salmon collected in site 1 originated from outside of the site (Table 4). Most of the rainbow trout that moved into each site originated from downstream of the site. Only one marked rainbow trout that was captured in an index site was marked at a trap that was located kilometers away (marked at site 1 and recaptured at site 2).

Table 4. Percent of rainbow trout and spring chinook salmon that were collected in population index sites and were marked. Fish were marked with unique fin clips at traps surrounding index sites. Mark origins refer to trap location and direction of movement; "A" refers to the trap located below the index site and "B" refers to the trap located above the index site. Totals refer to the percent of fish collected that were marked (N = number of fish collected).

Mark origin	Site 1		Site 2		Site 3	
	%	N	%	N	%	N
Rainbow trout						
A up	83.6	46	67.0[a]	4	66.7	2
down	10.9	6	0	0	33.3	1
B up	1.8	1	16.7	1	0	0
down	3.6	2	0	0	0	0
Total	57.3	96	14.3	42	10.3	29
Spring chinook salmon						
A up	84.6	22	0	0	0	0
down	0	0	0	0	0	0
B up	3.8	1	0	0	0	0
down	11.5	3	0	0	0	0
Total	49.1	53	0	0	0	0

[a] an additional 16.7% of marked fish (N=1) originated from trap 1A up

Variation in assemblage composition

Annual variations in assemblage composition did not appear to be different among sites (Table 5). However, assemblage structure was considerably different among sites. The primary differences in assemblage composition involved the daces, sculpins, and warm water cyprinids (Table 5). Site 1 had a high percent of speckled dace, site 2 had a high percent of longnose dace, and site 3 had a low percent of both dace species. Site 1 had a low percent of sculpins, but the other two sites had very high percents of sculpins. Site 1 was the only site that contained cyprinids such as redside shiners, bridgelip suckers, and northern squawfish. Taxa richness decreased with increased site elevation (Table 5).

Table 5. Percent composition of fish captured during electrofishing surveys in Swauk Creek index sites. Each site was 100 m long and was sampled by conducting two electrofishing passes. The following codes were used: RBT = rainbow trout, CUT = cutthroat trout, SPC = spring chinook salmon, SPD = speckled dace, LND = longnose dace, RSS = redside shiner, SCP = sculpin species, BLS = bridgelip sucker, OTH = other species.

		Percent composition									
Site	Year	RBT	CUT	SPC	SPD	LND	RSS	SCP	BLS	OTH	Taxa
SWK1	1992	2	0	1	53	0	11	0.3	32	0.2[a]	7
SWK1	1993	10	0	17	56	0	7	4	5	0.2[a]	7
SWK1	1994	12	0	6	52	19	5	4	2	0.3[a]	9
SWK2	1992	32	0	'0.3	0.3	26	0	41	0	0	5
sWK2	1993	24	0.4	2	0	18	0	56	0	0	5
SWK2	1994	18	0	0	0	16	0	66	0	0	3
sWK3	1992	10	2	0	0	8	0	62	0	18[b]	4
SWK3	1993	34	6	0	0	0	0	57	0	3[b]	3
sWK3	1994	17	1	0	0	0	0	81	0	0	3

[a] northern squawfish
[b] unidentified age 0+ trout

Electrofishing effects

Electrofishing surveys did not appear to be substantially correlated with fish movement. More movement. occurred four days before electrofishing than four days after electrofishing in site 1 (Table 6). Small numbers of fish were captured before and after electrofishing at sites 2 and 3, although more fish were captured after electrofishing than before at site 2 (Table 6).

Table 6. Number of rainbow trout and total fish that were captured moving out of index sites four days before and after electrofishing surveys. Site 1 **was** electrofished on August 4, site 2 on August 11, and site 3 on August 12, 1994.

Direction-day	Site 1		Site 2		Site 3	
	RBT	Total	RBT	Total	RBT	Total
			Before Electrofishing			
up-l	2	22	0	4	0	0
up-2	a	25	0	3	0	0
up-3	5	20	0	0	0	1
up-4	1	19	0	0	0	0
Total	16	86	0	7	0	1
down-l	0	44	0	2	0	0
down-2	0	4	0	0	1	1
down-3	1	41	0	1	0	0
down-4	0	13	0	1	0	0
Total	1	102	0	4	1	1
			After Electrofishing			
up-l	O	0	0	3	1	1
up-2	O	0	0	0	0	1
up-3	O	0	0	1	0	0
up-4	O	0	0	9	0	0
Total	0	0	0	13	1	2
down-l	O	9	1	6	0	0
down-2	O	0	2	5	0	0
down-3	O	1	0	0	0	0
down-4	O	0	0	4	0	0
Total	0	10	3	15	0	0

Discussion

The difference in the abundance and diversity of fishes migrating within portions of Swauk Creek is related to a suite of environmental variables that are correlated with elevation, such as distance from a mainstem immigration source, discharge, and water temperature. We expect fish movement to be greatest at sites that are closest to a mainstem immigration source, have discharges most suitable for migration, and are relatively warm. Although these variables probably interact to affect fish movement, we will treat each abiotic variable separately to facilitate this discussion. The greatest extent of movement would be expected into sites closest to an immigration source because the energetic costs of swimming to them are the least and the number of barriers associated with blocking or hindering migration would be least. Sites with high discharges should have the most movement because they protect fish during migration and provide adequate depths for fishes that prefer larger water bodies such as whitefish and squawfish from predators (Update 4). Alternatively, drastically low discharges may increase localized fish movement, such as at site 1, but reduce large scale movement. Finally, fish movement into sites with relatively warm temperatures (i.e. 20 - 25 "C) should be the greatest because they provide conditions suitable for the greatest diversity of fishes. For instance, warm water species such as redside shiner, northern squawfish, and largescale sucker were only caught at site 1.

Limited fish movement was observed in another Yakima basin tributary which supports the contention that sites with low discharges have relatively few migrators. We captured very few fish and taxa between June 14 and October 26, 1994 in a trap located near the mouth of Umtanum Creek (WDFW, unpublished data). Although the distance from a mainstem immigration source and water temperatures were similar between Swauk site 1 and Umtanum Creek, the discharges in Umtanum Creek were considerably lower than in Swauk Creek. In late June, 1994, (prior to stream dewatering in Swauk Creek) discharges in Umtanum Creek were approximately one-tenth the discharges of Swauk Creek. Thus, it appears that fish did not move into Umtanum Creek after June 14 because of the low discharges. Many fish may move through sites, but what factors influence what sites they immigrate into?

The variability in immigration of fishes among sites may be due to stream dewatering and species specific habitat suitability and use. Some fishes were trapped in pools when flows became intermittent in site 1 which may explain why such a high proportion of migrating rainbow trout and spring chinook salmon existed at this site. Longnose dace were the most abundant migrators through sites 1 and 2, but their percent of the assemblage composition was relatively low, particularly in site 1, during summer electrofishing surveys (Table 5). This suggests that most longnose dace don't colonize the lower parts of Swauk Creek and that habitat conditions for rearing were more favorable elsewhere. The magnitude of fish migrating into and through sites described in this study conformed well to the models presented by Pearsons (1994) for fishes in the John Day basin, Oregon.

In general, one would think that the occurrence of fish migrations would decrease concordance of assemblage composition between years (Decker and Erman 1992). However, if migrations occur with annual regularity concordance of assemblage composition may occur. This may have been the case in site 1. Concordance of assemblage composition between years was similar among sites despite higher numbers of fish migrating through lower elevation sites. This may have resulted from fish migrating at similar times every year and/or fish migrating through sites at times that they would not be captured. Some species in Swauk Creek, such as longnose dace, migrated through index sites and rarely remained at the site. However, assemblage composition within a year would likely be least concordant in sites with the greatest amount of fish movement (Pearsons 1994).

Despite the large number of fish that we captured, our study underestimated the number of fish moving in Swauk Creek. It appears that we installed our traps near or after peak fish movement occurred. Unfortunately, we were not able to install our weir-traps any earlier than we did because of construction difficulties associated with high discharges. In addition, we underestimated the number of age 0+ fish that moved because of the relatively large mesh size used to construct weir panels. During 1993, we installed a picket weir-trap near the mouth of Swauk Creek that could be operated at high discharges. From April 21 until June 25 we captured 205 adult bridgelip suckers as well as many other adult fish moving upstream (Pearsons and Martin 1994). This information supports the contention that we underestimated the number of fish moving in Swauk Creek in 1994 and that fishes moved during months other than those we sampled.

Monitoring Implications

To minimize annual variations in rainbow trout and juvenile spring chinook population and associated assemblage composition estimates, annual monitoring of fishes in index sites of Swauk Creek. could be conducted after August 29th. This recommendation is contrary to conventional practices which suggest that sampling should be conducted when discharges reach summer base flow. This conventional practice has been based on the assumption that fish movement was typically minimal during summer base flows. We found that fish movement was considerable at the lowest two sites in Swauk Creek until the beginning to middle of August despite summer base flows starting in mid-July.

Annual variations in population and assemblage composition estimates in other Yakima basin tributaries might be minimized by sampling sites that are less than 800 m in elevation and 18 rkm from a large mainstem river late in the summer. If many sites must be sampled during the summer and limited personnel is available to sample during the most opportune times, then the highest elevation sites far from a large mainstem source should be sampled first (e.g. during late July). Sampling should be completed before water temperatures and discharge change substantially in the late summer or fall. Low water temperatures can affect electrofishing efficiency as well as influence fish movement. In this study, electrofishing did not appear to influence fish movement substantially.

Update 1

Rainbow trout temporal and spatial spawning distribution in the upper Yakima River basin, and characterization of their redds

Introduction

This report describes the temporal and spatial spawning distribution of rainbow trout (Oncorhynchus *mykiss*) in the mainstem Yakima River and in two tributaries; Umtanum and Badger creeks. It is part of an on-going study to assess the potential for interbreeding between resident rainbow trout and anadromous steelhead trout, and to describe the temporal and spatial spawning distribution of rainbow trout in the upper Yakima basin. Concern for rainbow trout populations and fisheries exist if the two forms of 0. *mykiss* interbreed, especially if their progeny exhibit migratory tendencies over time. Conversely, if progeny exhibit resident tendencies, the migratory component of the population may be affected, which is also a concern. A complete description of resident and anadromous 0. *mykiss* life histories and the possibilities for interbreeding between them in the Yakima basin, using several field techniques, was presented in last year's annual report (Martin et al. 1994).

The goals of this update are to describe when and where rainbow trout spawn in the Yakima River. In addition, we assessed the feasibility of using redd surveys as-a monitoring tool. The use of redd surveys as a monitoring tool is potentially less harmful to fish than other techniques, such as electrofishing, trapping, and radio telemetry. Redd surveys can be used for monitoring the spawn timing, location, and abundance of spawners. Lastly, a model based on the physical characteristics of rainbow trout redds and their relationship with fish length is presented that may help differentiate between rainbow and steelhead trout redds when the fish spawn in sympatry.

Methods

Redd surveys were conducted in index sites within each of three elevational strata of the mainstem Yakima River to determine the temporal and spatial spawning distribution of rainbow trout. Survey techniques used in 1994 were the same as those used in 1993 (Martin et al. 1994). However, survey index sites were different from 1993. In 1993 we surveyed from the Squaw Creek boat ramp (Rkm 191.4) to the Slab (Rkm 186.8), from Damman Road (Rkm 220.6) to Ringer Road (Rkm 212.2), and from the ponds near Easton (Rkm 296) to the Washington Department of Fish and Wildlife (WDFW) Nelson Siding boat ramp (Rkm 284). In 1994, the lowest two sections were surveyed, but due to difficulties we experienced surveying the highest elevation section in 1993, we established a new high elevation index section in 1994. The new high-elevation section was from the mouth of the Cle Elum River (Rkm 280.8) to 1.5 rkm downstream of the south Cle Elum bridge (Rkm 267.9) (Figure 1).

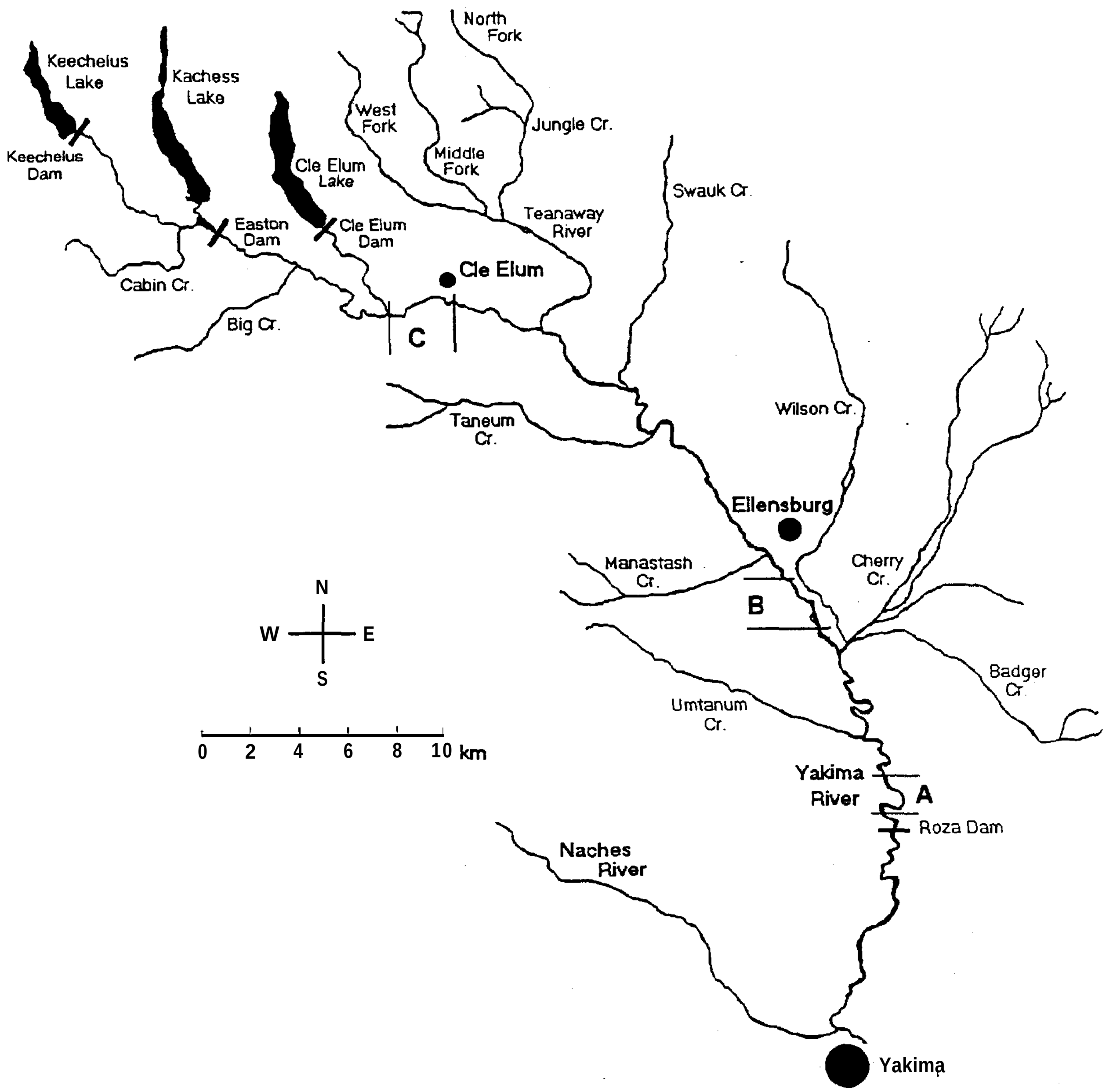

Figure 1. Locations of redd survey index sections in the upper Yakima Basin. Redd index sections were from Rkm 191 to Rkm 187 (A), Rkm 221 to Rkm 212 (B), and from Rkm 281 to Rkm 268 (C).

Surveys were conducted weekly from March 1 through June.

All redds encountered were assumed to be constructed by rainbow trout. We are confident that the majority of the redds counted were rainbow trout because less than 50 adult steelhead were estimated to be present in the entire Yakima River above Roza Dam in 1994 (Joel Hubble YIN, pers. corn.), and because typically 50% of the steelhead spawn in tributaries (Martin et al. 1994).

The peak of rainbow trout spawn timing in each section was calculated using the temporal distribution of redds constructed in index sections. To standardize the data, we divided the number of new redds by the number of days elapsed since the previous survey. Since the length of each index site was different, the number of redds per day was then divided by the survey site length so that magnitude differences of redd abundance between sections could be compared. The resultant number was the number of redds per day per kilometer. Because redds could only be identified for 10 days after they were constructed, 10 was used as the divisor if more than 10 days elapsed between surveys. This differed from 1993 when we divided by 14, if more than 14 days had elapsed between surveys. In 1994, individual redds were marked with in-stream markers so we could determine the number of days that a redd could be located. The peak of spawn timing based on redd surveys was calculated using Q-spline analysis (Manugistics Corporation, 1992).

The river bank with the best spawning habitat was surveyed each week and was referred to as the index bank. Bank selection procedures were described by Martin et al. (1994). Redds located on the index bank were marked by attaching biodegradable flagging to a fixed object, such as a tree, adjacent to the redd. The redd number and date were written on the flag to prevent remeasuring previously measured redds. Additionally, a brightly colored marker was placed in the bowl of completed redds to . provide a more specific mark.

The three index sections represented only 14% of the total length of the Yakima River between Roza Dam and Easton Dam. To determine the spatial distribution and abundance of redds throughout the entire Yakima River between those dams, the remainder of the river was surveyed during the peak of spawning. The surveys were conducted in the same manner as described for index sections.

In addition to describing the temporal and spatial distribution of rainbow trout spawning, the physical characteristics of 130 randomly selected redds in the Yakima River were recorded. To avoid disturbing fish from uncompleted redds, only those redds that did not have fish located on or adjacent to them were measured. Terminology and measurement of redd features was similar to that of Ottaway et al. (1981), but the locations at which some of the measurements were made were different. Measurement techniques were reported in Martin et al. (1994) .

Redd measurement data were used to construct a model to differentiate between rainbow and steelhead trout redds based on redd length. The model was constructed using regression analysis where fish length was the independent variable and redd length was the dependant variable. To further assess the relationship between fish length and redd length, and to increase the usefulness of the model, redds constructed by rainbow trout, bull trout (*Salvelinus* confluentus) steelhead trout, and spring chinook salmon (0. *tshawytscha*) in 10 Columbia Basin rivers were measured in 1993 and 1994. Redd measurement techniques were the same for all redds measured.

An analysis correlating redd length to fish length was based on redds constructed by rainbow trout in 1993 and 1994 in three elevational strata of the Yakima River. The mean length of female rainbow trout in each strata was determined during electrofishing surveys conducted in the spring of 1993 (WDFW, unpublished data), and assumed to represent the mean length of females constructing redds in each of these strata in 1993 and 1994. Redds constructed by rainbow trout in Umtanum Creek in 1993 were measured by students from Central Washington University (CWU). Rainbow trout spawner lengths in Umtanum Creek were measured at a trap located near the mouth of the creek in 1993 and 1994. In 1993, steelhead trout redds were measured in the lower Yakima Basin (Satus and Buckskin creeks). In 1994, steelhead trout redds were measured in Satus Creek and in the Tucannon and Touchet river systems which are tributaries of the Snake and Columbia rivers, respectively. Fish lengths in the Tucannon River and Wolf Fork Creek, a tributary to the Touchet River were provided by Art Viola (WDFW, pers. corn.), while spawner lengths in Satus Creek were provided by Joel Hubble (YIN, pers. corn.). These rivers were used because steelhead redds were relatively abundant and they were easy to observe. Steelhead' length was similar in those rivers and in the upper Yakima Basin (average 60 cm in length). Lengths of bull trout spawners were determined in the Tucannon and Touchet rivers, and Mill Creek, in 1990 and 1991, by Martin et al. (1992). Spring chinook salmon redds were measured in the American and Yakima rivers in 1994 and spawner lengths were provided by Joel Hubble (YIN, pers. corn.).

To compare our data to those from the literature, total redd lengths were multiplied by 0.66 to arrive at an estimated redd tail length. This value was based on the relationship of brown trout total redd length and tail length observed by Ottaway et al. (1981).

To describe the temporal and spatial distribution of rainbow trout spawning within Umtanum Creek, redd surveys were conducted daily from March 12, to April 29, 1994. Redds were identified by the presence of clean substrate and typical morphology as described above (Murdoch 1995).

Biological information about rainbow trout spawners and environmental variables associated with fish movement into Umtanum Creek were evaluated by trapping fish near the mouth of the creek. Fish moving upstream or downstream were trapped in a two-way panel weir in 1993. As in previous years (Martin et al. 1994), the trap was located within 0.2 rkm of the mouth of the creek to reduce the possibility that fish spawning in lower reaches were undetected. Water temperature ("C), water column depth (mm), and date were recorded daily at each trap. In addition, fish length (mm FL), weight (g), direction of travel, sexual maturity, and sex (if it could be determined), were recorded for each salmonid captured.

Lastly, due to the unusual genetic composition of rainbow trout in Badger Creek (Phelps and Baker 1994) and the unknown peak of spawn timing in this creek in 1993 (Martin et al. 1994), we conducted electrofishing surveys in the fall of 1994 to determine if rainbow trout spawned during the fall and winter in this creek. Our hypothesis was that the peak of spawning had occurred prior to our spring spawning surveys in previous years of study. Therefore, we began sampling in November 1994, and continued collecting bi-weekly samples through March, 1995. Fish were collected using electrofishing techniques and the percentage of the sample that was sexually mature was determined. Two criteria were used for assigning a peak of spawning: more than 15% of the sample had to be sexually mature and the sample size had to be at least seven fish of adult size (see Martin et al. 1994 for complete explanation).

Results

A total of 206 rainbow trout redds were observed in three , index sections of the Yakima River in 1994. Rainbow trout spawned in each of the three sections surveyed and spawn timing peaked on April 12 and 14 in the middle and highest elevation sections. No peak was identified in the lowest section due to high turbidity which precluded surveys after April 15 (Figure 2). A total of 357 redds were observed during our survey of the entire Yakima River between Easton and Roza dams during the peak of spawning. Of the 357 redds observed, 137 (38%) were in the three index sections. Within the upper Yakima River redd densities (# redds/km) in our index sections were similar to larger reaches of the river (Figure 3). The percentage of redds observed in the index sections was probably lower than 38 because we did not sample the area from the Teanaway River to Ellensburg Dam (Rkm 255 to 233), and from the Nelson Game Ramp to the Cle Elum River (Rkm 284 to 281). The spatial distribution of rainbow trout redds was patchy, with a high proportion of redds occurring between Rkm 211 and 212 (Cherry Creek to Ringer Road; Figure 4). The remainder of the river exhibited similar densities of rainbow trout redds, ranging from 2 to 20 redds per kilometer in those reaches surveyed.

Rainbow trout spawning in Umtanum Creek began on March 27, and the last new redd was observed on May 6, 1994 (Figure 5). Rainbow trout spawned from the mouth of the creek to Rkm 0.5, where a large beaver dam obstructed further upstream passage (Murdoch 1995). A total of 47 adult rainbow migrated into Umtanum Creek from the Yakima River as identified by trap catches. The average length of these fish was 334 mm and, of five environmental variables included in a stepwise regression, average daily temperature was the most important variable determining when fish entered the creek to spawn (Murdoch 1995).

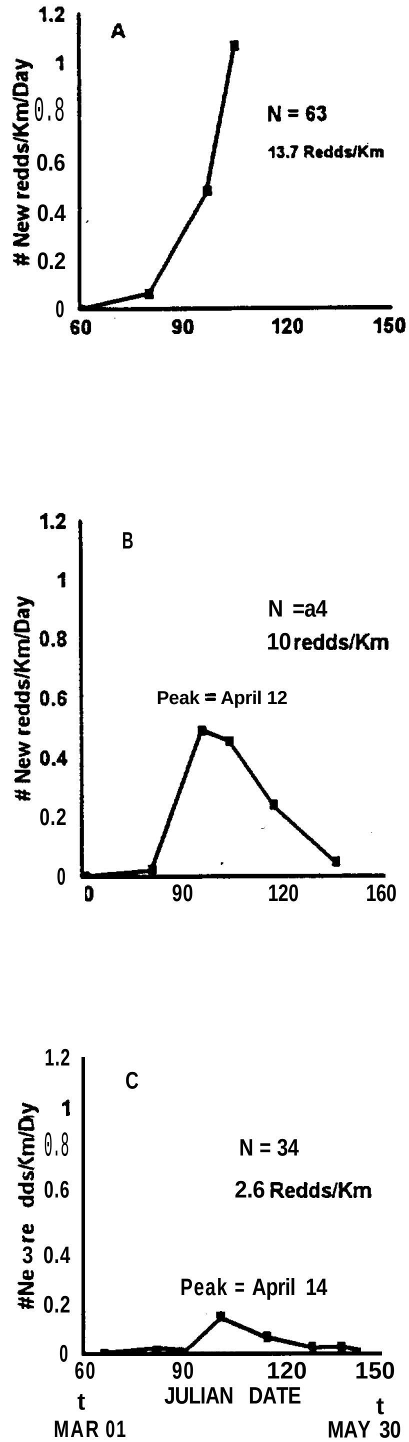

Figure 2. Temporal spawning distribution of rainbow trout based on redd surveys in three index sites of the mainstem Yakima River. A depicts the lowest elevation site (422 m), B depicts the middle elevation site (494 m), and figure C depicts the highest elevation site (575-m). The total number of redds and the number of total redds per kilometer are presented for each section.

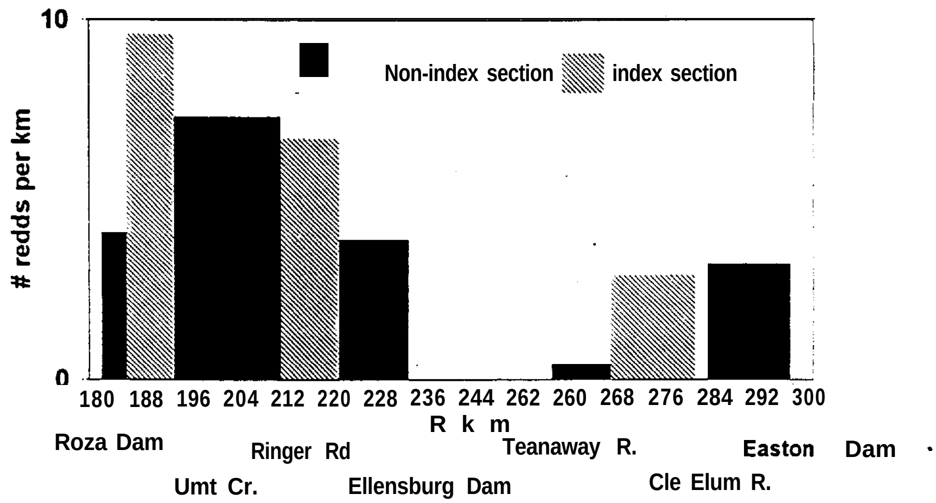

Figure 3. Distribution of the average density (# redds/km) of rainbow trout redds in index and non-index reaches of the upper Yakima River during the peak of spawning (mid-April), 1994. Sections that were not surveyed included reaches from the Teanaway River to Ellensburg Dam (Rkm 255 to 233), and from Nelson game ramp to the Cle Elum River (Rkm 284 to 281).

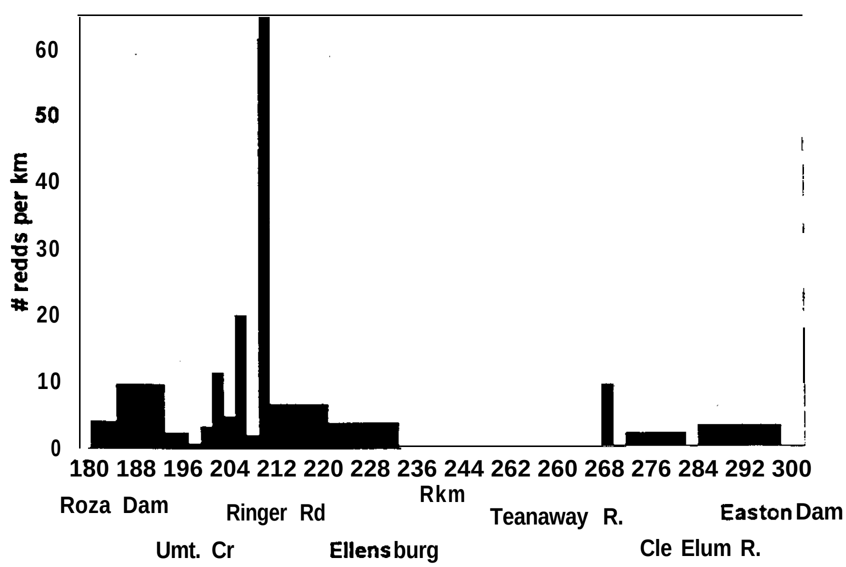

Figure 4. Distribution of rainbow trout redd density (#redds/km) in the upper Yakima River during the peak of spawning (mid-April), 1994. Sections that were not surveyed included reaches from the Teanaway River to Ellensburg Dam (Rkm 255 to 233), and from Nelson game ramp to the Cle Elum River (Rkm 284 to 281).

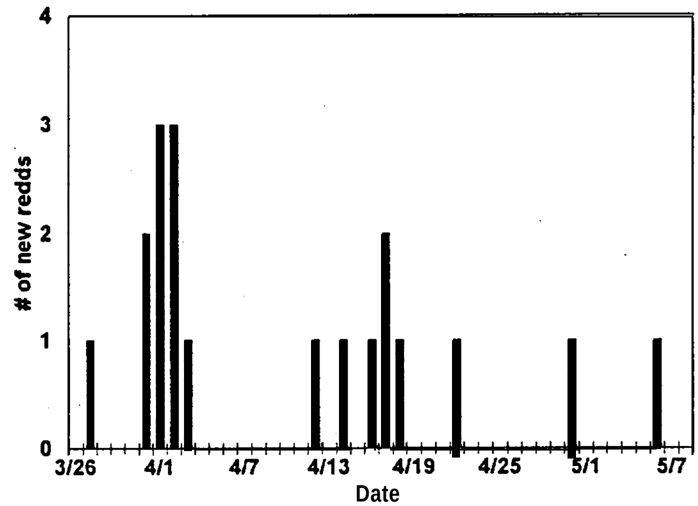

Figure 5. Temporal distribution of rainbow trout redds in Umtanum Creek, 1994.

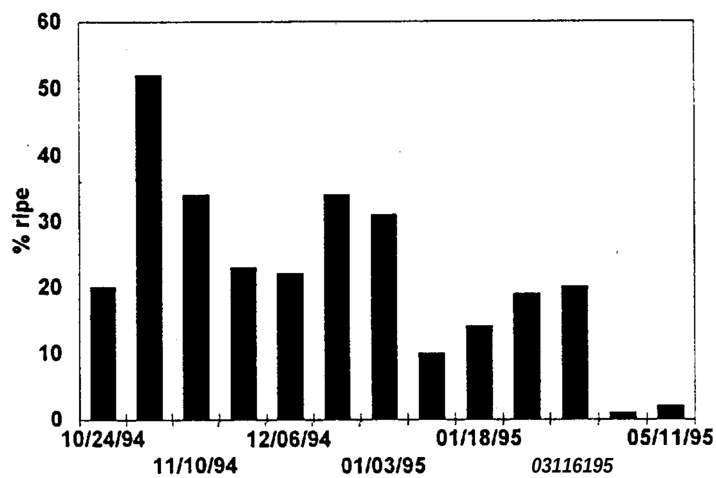

Figure 6. Temporal distribution of sexually mature rainbow trout collected from Badger Creek in 1994 and 1995.

Rainbow trout in Badger Creek appeared to spawn between October and May, with most spawning occurring during the winter (Figure 6). Spawn timing was inferred primarily from observations of sexually mature males; only one sexually mature female was captured during 11 bi-weekly surveys conducted from October 24, 1994 to May 11, 1995. This fish was captured on November 10, 1994.

In the Yakima River, rainbow trout redds measured in 1994 were significantly shorter in length than those measured in 1993 ($t=2.6$, $df=332$, $P=0.01$), and were also smaller in area ($t=2.1$, $df=332$ $P=0.03$). The width of the redds did not differ significantly between years ($t=1.1$, $P>0.26$, $df=332$). In 1994, we observed a significant difference between water depth in the bowl, side, and tail of redds constructed in the main channel versus those constructed in side channels ($t=3.6$, $df=177$, $P=0.0003$; $t= 3.7$, $df=176$ $P=0.0003$; $t=3.6$, $df=177$ $P=0.0005$, respectively), which was probably due to shallower water in side channels than in the main channel. Although depth differed, there were no significant differences for length, width, or area between redds constructed in the main channel verses those constructed in side channels at($P<0.05$).

Our intent was to characterize every rainbow trout redd observed; however, time constraints precluded achieving this objective. Statistics presented in Table 1 include all redds characterized, but not all redds observed.

Rainbow trout utilized in-stream cover and side channel habitat when constructing redds. Redds were often constructed near organic debris greater than 25 cm long, and 91% of the redds were constructed in run habitat (165 of 181) redds. Although side channel habitat was not as abundant as main channel habitat, 45% (82 of 181) of the rainbow trout redds were located in a side channel, and an additional 28% were located within 25 m of a side channel. Only 27% (27 of 99) of the redds constructed in the main channel were greater than 25 m from the nearest side channel. The majority of rainbow trout redds (87%) were within 5 m of another redd, and 10 of 117 (8.5%) were superimposed by other rainbow trout redds.

Table 1. Average physical measurements of rainbow trout redds in the mainstem Yakima River, 1994 (N = 130). All measurements are in meters unless specified otherwise.

				Water Depth (ml			Velocity (m/sec) @ 0.6 depth		% Substrate (Pot) by size class*			% Substrate (Tail) by size class'		
	Area	Length	Width	bowl	tail	side	surface	head	1	2	3	1	2	3
Avg.	1.2	1.5	0.8	0.4	0.3	0.3	0.7	0.6	12.0	32.9	45.2	6.6	17.1	67.4
(s.d.)	(0.6)	(0.4)	(0.2)	(0.1)	(0.1)	(0.1)	(0.21	(0.2)	(9.2)	(25.0)	(26.6)	(6.2)	(31.1)	(31.7)

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.

As in 1993, a significant difference was found between the length ($t = -6.8$, df= 158, $P<0.0001$), width ($t = -10.9$, df= 158, $P<0.0001$), and area ($t = -7.3$, df= 158, $P<0.0001$) of rainbow trout redds constructed in Umtanum Creek and the Yakima River. Comparisons between rainbow trout redds constructed in the Yakima River and steelhead trout redds constructed in the Tucannon River, and Wolf Fork and Satus creeks showed that the rainbow trout redds were significantly shorter ($t = -9.2$, df= 160, $P<0.0001$), narrower ($t = -8.1$, df= 160, $P<0.0001$), and smaller area ($t = -10.3$, df= 160, $P<0.0001$) than steelhead trout redds. Based on these comparisons, there appeared to be a gradient of redd sizes from large to small for steelhead in small tributaries, rainbow trout in the Yakima River, and rainbow trout in Umtanum Creek, respectively (Table 2).

Table 2. Comparison of 1994 redd dimensions(m), for rainbow trout spawning in a small stream and a large river, and steelhead trout spawning in small streams.

	Rainbow trout Umtanum Creek (N=30)	Rainbow trout Yakima River (N=130)	Steelhead trout Tucannon R., Touchet R. and Satus Creek (N=32)
Length	0.94	1.51	2.32
Width	0.42	0.78	1.11
Area	0.40	1.18	2.58

Using redd tail length data for rainbow, bull, and steelhead trout, and spring chinook salmon, a significant relationship was found between the length of the fish and the length of their redd. The formula relating fish length to redd tail length is:

$$Y = -43.0 + 4x \quad r = 0.80, \; P = 0.002,$$

Where: Y = redd tail length (cm), and
X = fish length (cm).

The relationship between spawner length and redd tail length is illustrated in Figure 7.

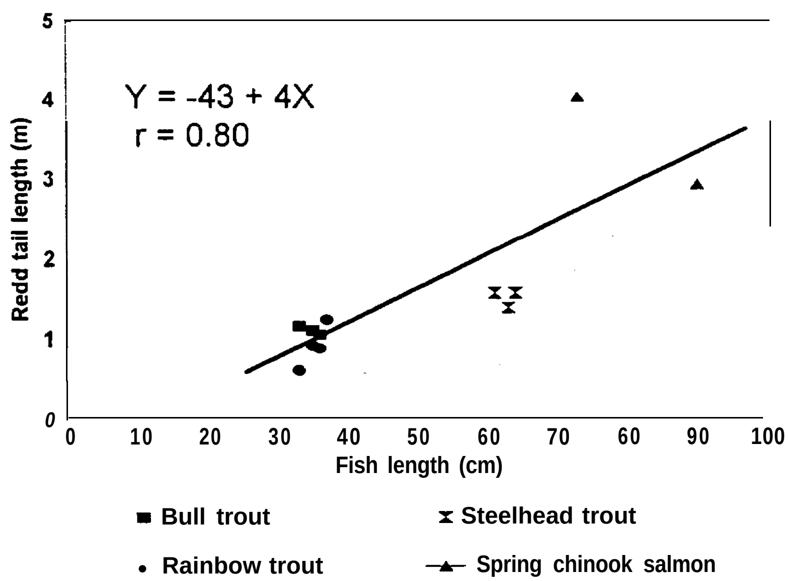

Figure 7. Relationship between redd tail length and spawning fish length for three salmonid species in the Columbia River basin.

Although Figure 7 indicates that redd length was positively related to fish length, this relationship appears to be mediated by factors associated with stream size (eg., flow, substrate type and size). Steelhead trout that spawned in Wolf Fork Creek were the same length as those spawning in the Tucannon River, but they made smaller redds than their Tucannon River counterparts. River discharge at the time of spawning was 2 cubic meters per second (m^3/s) in Wolf Fork Creek but was nearly twice as high in the Tucannon River at 3.7 cms. Spring chinook salmon spawning in the American River were considerably larger than their Yakima River counterparts, but they made smaller redds. River discharge at the time of spawning was 3 cms in the American River but was more than 300% greater in the Yakima River at 10 cms. In the Yakima River, rainbow trout made significantly larger redds than rainbow trout in Umtanum Creek, even though the average length of the fish in the Yakima River was only 19 mm greater. Discharge in Umtanum Creek, at the time of spawning was less than 1 cms, while in the Yakima River, discharge was nearly 20 cms.

Discussion

The peak time of rainbow trout spawning in the Yakima River was April 14 (Julian day 104) based on redd surveys, which was very similar to the peak identified using electrofishing techniques in 1991, 1992, and 1993 (Martin et al. 1994). Due to the similarity of results, we suggest that redd surveys are a viable method to monitor the temporal and spatial spawning distribution of rainbow trout in the upper Yakima River. This technique imposes relatively little harm to fish and provides definitive spawning information with relatively low monetary costs.

Martin et al. (1994) reported that there was no definitive peak time of rainbow trout spawning in Badger Creek in 1991, 1992, and 1993. Even though they did not find a peak time of spawning, sexually mature fish were collected on every survey. The results of the winter and spring surveys conducted in 1994-1995 indicated that rainbow trout were sexually mature from October to May, but that most appeared to spawn in the winter. Due to the extended period of time that sexually mature fish were collected from Badger Creek, we will use redd surveys in 1995 in an attempt improve documentation of spawn timing in this creek. Spawn timing for rainbow trout in this creek is important from a baseline characterization aspect as well as describing the temporal spawning habits of this unusual genetic component of the upper Yakima rainbow trout population. Surveys will begin in December of 1995 and continue through the spring of 1996.

The temporal spawning distribution of rainbow trout in Umtanum Creek in 1994 was very similar to 1991, 1992, and 1993, with redds being.constructed from late March to late April (Martin et al. 1994). In this creek, the continued use of redd surveys to monitor the temporal and spatial distribution, abundance, and size of rainbow trout spawners immigrating from the Yakima River is recommended.

In the mainstem Yakima River, redd characterization techniques may be used to monitor rainbow trout spawner size and abundance. In 1994, rainbow trout redds were significantly smaller than those measured in 1993. Based on these results, and the positive relationship we found between redd length and fish' length, it-is likely that length of spawners in 1994 was smaller than in 1993. Visual examination of Figure 9 in Update 3 of this report indicates that the percent of rainbow trout greater than 250 cm (those large enough to spawn) in 1994 was considerably lower than in 1993. The relationship we observed between mean redd length and mean fish length appears to provide promise for use of redd measurement data to monitor mean spawner size.

The use of redd surveys to monitor spawner abundance has some pitfalls. Extrapolating redd densities to the entire upper river using small index sections is not recommended due to spatial patchiness of redds (refer to Figure 4). To explore improvements in extrapolation approaches, in 1995 we will stratify the upper river into constrained and unconstrained strata. Constrained strata are those in which the valley width is less than two times the active channel width, while unconstrained reaches are those in which the valley width is greater than two times the active channel width. Large reaches of each strata will then be surveyed to contain the observed variability within each strata and allow us to expand the number of redds per kilometer within each strata. By then summing each strata, we can estimate the total number of rainbow trout redds in the river. Specifically, we plan to sample four index sections in 1995; 1) Umtanum Creek to the Slab (Rkm 187), constrained, 2) Reinhart Park to Cherry Creek, unconstrained, 3) Teanaway River to Thorp Bridge, constrained, and 4) Cle Elum River to Teanaway Game Ramp, unconstrained.

There is some extent of error associated with the possibility that some steelhead trout redds may have been classified as rainbow trout redds because steelhead do spawn in the upper river. Our model differentiated steelhead and rainbow trout redds in streams of different size, but we do not know its capability to discriminate redds within a single river. Therefore, in 1995, we plan to measure known steelhead redds constructed in rivers of the northwest that are of similar size to our study reaches in the Yakima River, and to again measure any known steelhead redd(s) observed in the Yakima River.

Lastly, due to problems with visibility, redd surveys may not always allow effective redd detection thus biasing results regarding the temporal spawning distribution of rainbow trout. The lower most survey section, below Wilson Creek (rkm 211), typically experiences annual high turbidity in mid-April as irrigation water is released through the Kittitas Valley. Therefore, it will not be possible to identify the complete temporal and spatial spawning distribution, or the total number of redds in this section of the river. By coordinating survey dates around expected irrigation water releases, we should be able to survey until, and maybe slightly after, the peak of spawning in this section. Using the temporal spawning distribution curves generated for upstream sections, we will be able to construct the descending portion of the spawning curve. This will allow monitoring of the temporal spawning distribution, as well as estimation of the total number of redds constructed in this section.

In conclusion, given current depressed steelhead trout spawner densities in the upper Yakima Basin, redd surveys are an excellent method for describing the temporal and spatial distribution of rainbow trout spawners. This method, coupled with the positive relationship between fish length and redd length, has promise as a spawner size and spawner abundance monitoring technique. The use of rainbow trout redd surveys as a monitoring tool for the Yakima Fisheries Project will be effective if the steelhead trout population remains low, if steelhead redds can be distinguished from those made by rainbow trout, and if redd surveys will satisfy statistical criteria established for the project's monitoring plan (eg. reliable, precise, and accurate). Until further work is completed, the results presented here should be considered preliminary.

Update 2.

Movement of resident rainbow trout within the upper Yakima River basin

Introduction

In a previous report, we reported that a majority of rainbow trout in the upper Yakima Basin appeared to move little between their time of initial capture and subsequent recapture (Bartrand et al. 1994). Only the absolute movement distances of tagged juvenile and adult-sized rainbow trout were presented. However, biases and the lack of sensitivity in our methods, and the generally small fraction of fish that we recaptured limited the conclusions that could be drawn. The orientation of fish movements with regard to stream flow and more precise measures of movement over smaller spatial scales were not included. This "update" attempts to address these more specific facets of rainbow trout movement within the upper Yakima River basin. The results provide an update to the analyses presented in Bartrand et al. (1994). All methods are similar to those presented in Bartrand et al. (1994) unless otherwise noted In addition, specific sources of bias and statistical assumptions within the previous analyses shall be discussed. The results hereinafter are preliminary and subject to further revision.

Methods

This update contains two new analyses of the tagging and recapture data collected between 1990 and 1994 formerly reported by Bartrand et al. (1994). Results here are based on a different data set from that used in the previous report. The new data set contains first-time recapture information, collected from March 13, 1990 to October 18, 1994, and includes the most precise capture locations available from field records. Only first-time recaptures were utilized since independence of the data derived from repeated recaptures was not demonstrated. Greater precision was gained through a rigorous interpretation of the geographic areas described for both tagging and recaptures in field notes, and of the time of year of respective fish collection activities. The direction of fish movements and the occurrence of movements between two or more streams were determined during movement distance calculations and are summarized. The minimum net-movement values associated with each movement type are included. Movement direction was categorized as upstream, downstream, or complex. Complex movements involved a stream and its tributary and sometimes incorporated movements in both upstream and downstream directions (Funk, 1955). Fish were described as static if no movement could be ascertained from their capture and recapture information. This happened if the geographic areas described for both capture and recapture overlapped; We 'also present separate figures that reflect the movements of fish recaptured solely above Roza Dam so that downstream movement data could be more validly compared to the upstream movement data.

Movements were additionally described in association with physical variables to provide some understanding of variations in movement patterns. Small scale movements in tributaries were revealed by examining recapture data from rainbow trout tagged in 100 m population index sites from 1990 to 1993. In these sites, fish were considered to be static if they were reobserved within the same site 300 to 400 days later. Reobservation rates were then compared to 1) a measure of habitat complexity, and 2) the percentage of the site area composed of pool habitat (Kennedy and Strange, 1982), as plotted against site elevation above mean sea-level. Site habitat complexity was measured by the standard deviation of depths taken in the thalweg at 1 m intervals (Martin et al. 1994). These sites were located in Cabin Creek, Taneum Creek, and the North, Middle and West forks of the Teanaway River.

Results

Differences existed in the movement patterns exhibited by rainbow trout depending on the direction they moved with respect to flow and the location of initial capture. Downstream movements appeared to be of greater distance and more prevalent than upstream movements. As was indicated in Bartrand et al. (1994), movement distances in the mainstem river were greater than those in tributaries. Table 1 summarizes minimum net movement distances with respect to the observed movement types and their relationship to tributary or mainstem stream areas.

Table 1. Movements exhibited by tagged rainbow trout (>174 mm FL) upon their first recapture in the Yakima River and its tributaries from 1990 through 1994. Complex refers to movements involving a stream and its tributary (Funk 1955). Static indicates that a lack of information was available to detect movement or that little movement occurred. Numbers in parentheses reflect only tagged fish recaptured above Roza Dam and excludes those observed downstream of that point.

	Minimum Net Movement Distance (km)		
Type	Mean	Median	N
Complex:			
downstream, then upstream	7.8	1.7	23
upstream	9.0	3.5	28
downstream	25.6 (10.0)	8.8 (7.7)	13 (11)
Downstream:			
tributary	1.7	1.0	18
mainstem	14.6 (11.8)	5.2 (5.0)	45 (40)
Uostream:			
tributary	1.3	0.25	10
mainstem	6.0	4.6	35
Static:			
tributary	0	0	153
mainstem	0	0	398
Total	***	***	723 (716)

The immigration and emigration of individual fish to and from specific stream locations between years was large. Reobservation rates of tagged trout within 100 m tributary sites approximately one year later was low; from 0 to 50 percent with a median value of 0. The total number tagged and recaptured there were 129 and 10, respectively. Due to low recapture rates and data showing that the numbers and length frequencies of rainbow trout in these sites were similar between years (Martin et al. 1994), we believe that population levels are maintained through replacement by new individuals from outside these sites.

Although we recaptured only 10 rainbow trout one year after tagging in these tributary sites, it appeared that site fidelity is positively related to site elevation. Figure 1 shows greater site fidelity exhibited at the uppermost elevations. However, habitat complexity and pool area appear to decrease with increasing elevation. Therefore, a negative relationship exists between those variables and the likelihood of an individual's presence one year later.

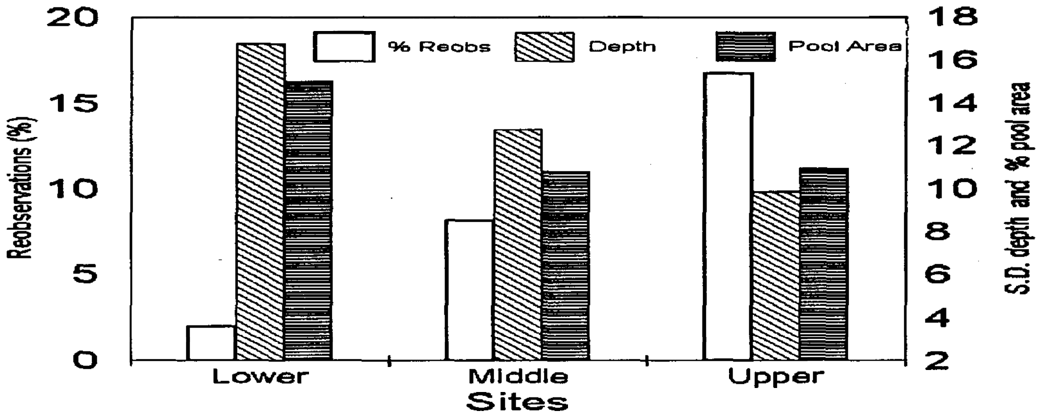

Figure 1. Rainbow trout reobservation rates (% Reobs), standard deviation of thalweg depths, and percentage of pool area of annually surveyed 100 m tributary index sites. Standard deviations of thalweg depths are represented by 1993 data only. Pool areas are means for 1991 through 1993. Sites are arranged in order of increasing elevation relative to their location in respective streams.

Discussion

The observed patterns of rainbow trout movement in this analysis were consistent with the works of other researchers and results from our previous analysis. It is somewhat predictable that downstream movements would exceed upstream movements as trout reproduction generally occurs upstream of rearing areas (Martin et al. 1994) and progeny will tend to replace adults from downstream habitats. Whereas the differences in movement distances by fish in tributary and mainstem areas shown in our previous report were small, the tabulation of movements by direction here shows these differences to be manifold. The large number of fish determined to be static according to minimum net-movement distance is generally a reflection of weaknesses in the data collection procedure and should not be interpreted to mean a majority of the recaptured fish had not moved. The number interpreted to be static in this analysis decreased from the 1994 report due to the availability of more precise capture and recapture locations.

Describing the persistence of tagged individuals in tributary sites provided an excellent alternative means of characterizing movements. This method is discussed in Gowan et al. (1994) as a valid means of verifying other commonly used movement study techniques. Our analysis of the reobservation of individuals in'100 m sites and the figures given for median upstream and downstream movements in tributaries are compatible. However, only a small number were determined to be static in 100 m sites. The overall reobservation rate of eight percent meant that the remaining 92 percent of tagged fish must be accounted for through mortality, movement, or tag losses. Assuming 50 percent annual mortality for taggable-sized trout and negligible tag loss, four of ten tagged fish would have been expected to move from their original locations and then be replaced by untagged fish moving into those vacated sites.

The finding of a negative relationship between site fidelity and habitat complexity was unexpected. An explanation for this outcome may be that quality habitat units are less common and more important for survival in areas with low complexity. However, this negative relationship could support the possibility that stream temperature, as moderated by site elevation, is more important for site fidelity than habitat complexity. The small number of fish tagged in some sites precluded a more rigorous investigation of these results. Conversely, site fidelity may not be directly related to physical variables. Seasonal colonization rates of low elevation stream areas by fishes of other species is much greater compared to higher elevation reaches (Pearsons, 1995, and Chapter 2, this report), possibly triggering more or larger scale displacements of trout, or both. Similarly, colonization rates of rainbow trout tend to decrease with increasing elevation (Chapter 2, this report).

This analysis differs from Bartrand et al. (1994) in that only first-time recaptures and minimum, as opposed to average net-movement distances, are presented here. A lack of independence was demonstrated for the movement distances observed from repeated recaptures, therefore, any repeated recaptures of individual fish were omitted from this analysis. Minimum net-movement distances reflect minimal possible movements, resulting in underestimates of actual movement. However, they are less affected by biases in methodology and provide for stronger comparisons. Increased sample sizes of non-zero minimum movement distances gained after refinement of the tagging and recapture data set made the use of minimum net-movement values a better choice in this analysis.

Statistical tests presented by Bartrand et al. (1994) 'should be interpreted with caution because, upon further-examination, the data were not normally distributed. For that reason, parametric tests were not performed within this analysis. Conformance of the data set to a normal distribution may be achieved through polarizing the movement distances according to upstream or downstream direction and, if necessary, some form of data transformation. Results of further analyses will be provided in a subsequent report.

Tracking and interpreting the movements of fishes requires caution. We have described the movements of rainbow trout in this and our previous report using complementary methods such as trapping, determining the net movements of tagged fish, and measuring the number of fish reobserved in a fixed area. These descriptions were based upon specific linear movements that were the sum, or a part of, many possible movements. Therefore, conclusions the reader draws from these observations should reflect only probabilities of such movements, rather than complete movement characterizations of the population.

Update 3

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

Introduction

Results are reported for the fifth year of an on-going study to describe rainbow trout population abundance and salmonid distribution in the upper Yakima basin. These data will be used to assess potential impacts to the resident rainbow trout population as a result of a proposed spring chinook salmon supplementation program in the upper basin and help guide establishment of a monitoring plan. The objectives of this report remain unchanged from previous reports. Briefly, they are to 1) document annual rainbow trout abundance and distribution in five mainstem sections and 10 tributaries of the Yakima River above Roza Dam, 2) assess biotic and abiotic factors associated with rainbow trout abundance in index sites, and 3) document the abundance and distribution of naturally produced juvenile spring chinook salmon and other salmonid species in tributary index sites. Results should be considered preliminary pending further data collection and analysis.

Methods

Tributary Salmonid Population Estimates

From 1990 through 1994, densities of rearing salmonids in several tributaries of the upper Yakima River were determined to evaluate salmonid abundance as well as their spatial and temporal distribution (McMichael et al. 1992; Pearsons et al. 1993; Martin et al. 1994) (Figure 1). The number of tributaries and index sites sampled has changed over the five years of sampling for reasons described by Martin et al. (1994). In 1994, 10 tributaries and a total of 27 index sites were sampled.

Tributary and site selection criteria were presented by McMichael et al. (1992). Briefly, the 100 m index sites were selected to represent each of three elevational strata within a tributary so that spatial (between sites) and temporal (interannual, within sites) variability could be monitored. The abundance of rainbow trout greater than 79 mm and juvenile spring chinook salmon (all sizes) was estimated with backpack electrofishing using removal-depletion methods (Zippen 1958). Variation in rainbow trout abundance among years was assessed with the coefficient of variation (CV). Coefficient of variation is calculated by dividing the mean population abundance by the standard deviation. The coefficient of variation is used here to describe the amount of variation a population exhibits.

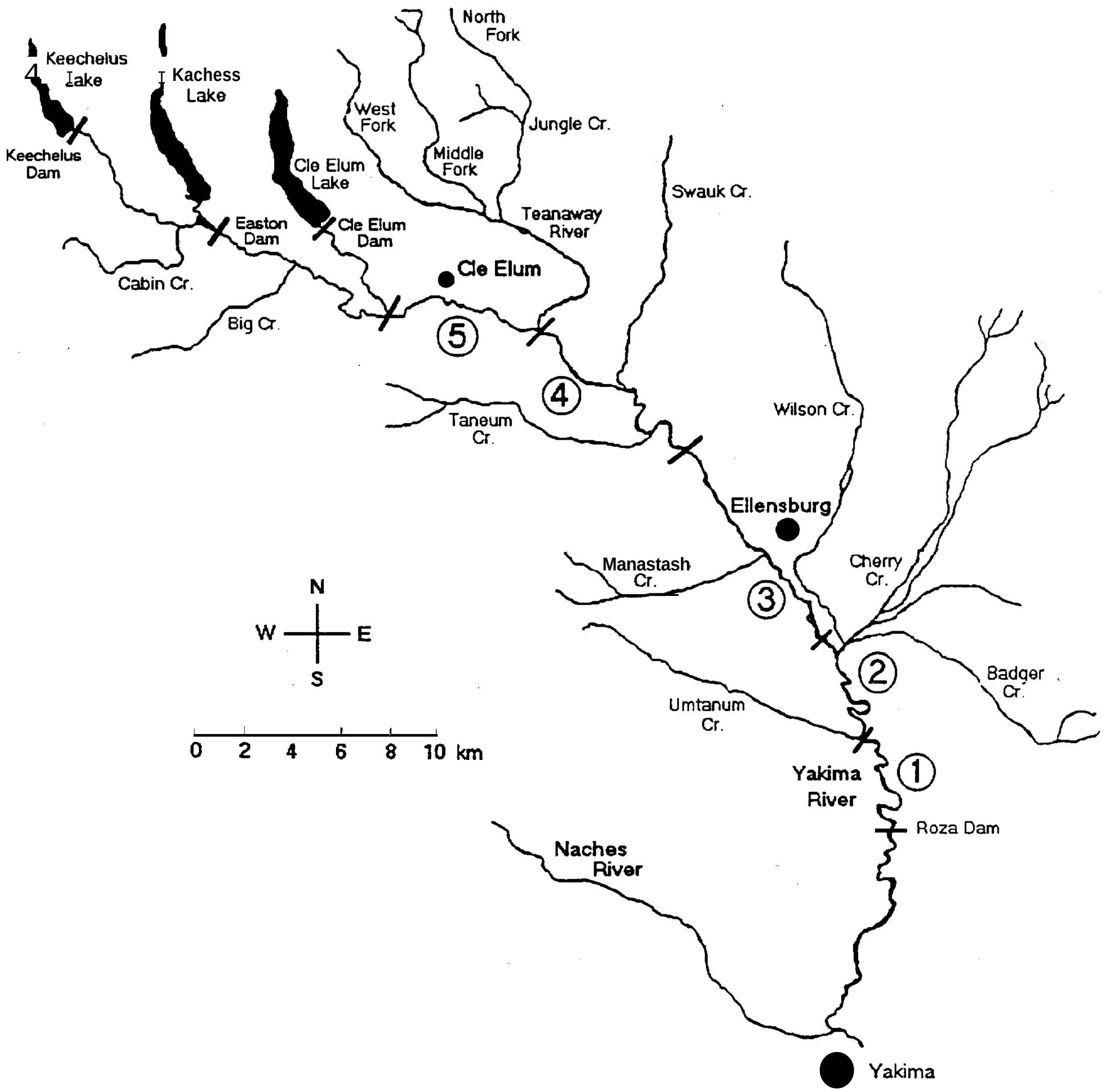

Figure 1. Map of the upper Yakima basin showing locations of tributaries and mainstem sites sampled. Circled numbers indicate mainstem sections; 1=LCYN, 2=UCYN, 3=EBURG, 4=THORP, and 5=CELUM.

The coefficient of variation is independent of abundance, and therefore allows for the comparison of variability across sites with different abundances. In this study we used the CV classification proposed by Freeman et al. (1988), in which CV values less than 25 were considered to represent a highly stable population, values between 25 and 50 represent a moderately stable population, values between 50 and 75 represent a moderately fluctuating population, and values greater than 75 represent a highly fluctuating population. Further description of data collection and analysis is presented in Martin et al. (1994). We used a hand-held GPS (Global Positioning System) to record the coordinates of each tributary index site. In addition to recording the longitude and latitude of each site, we also recorded directions to each site. Site coordinates and directions to each site are presented in Appendix 1A and 1B.

Habitat area, stream discharge, water temperature, longitudinal stream bed profile (thalweg depth) and gradient were also recorded for each index site. Methods for measuring habitat variables were presented in Martin et al. (1994). Continuous reading thermographs were deployed at each of the 27 index sites on July 29 and recovered in October, to obtain stream temperature data during the summer.

We also established three index sites in the mainstem of the Teanaway River to further describe rainbow trout and juvenile spring chinook salmon abundance and distribution in the subbasin. The three index sites previously established in Manastash Creek (Pearsons et al. 1993) were also re-surveyed in 1994. A total of 28 different population index sites have been assessed since the study began in 1990, including 13 that have been surveyed in all four years.

Both within and between-year correlations between biotic and physical variables were examined in each index site from 1990 to 1994. Data analysis and interpretation techniques were presented by Martin et al. (1994).

To determine if index site population estimates in Taneum Creek were representative of the rearing density in the creek, six additional systematically selected sites were sampled in Taneum Creek in 1994. Population estimates in these sites were then compared to index sites in that creek. The location of the systematic sites was determined using methods described by Martin et al. (1994) for systematically chosen sites sampled in the Teanaway Basin in 1993.

In Swauk Creek, we established four 100 m long contiguous sites to assess spatial variability of rainbow trout abundance in the lowest reach of this creek. This creek was chosen because juvenile spring chinook salmon were abundant and a long term data set exists for the creek. The 400 m reach was located 0.5 km downstream from our lowest Swauk Creek index site and 0.5 km upstream from the creek's confluence with the Yakima River. Sampling methods used in the systematic sites in Taneum Creek and the contiguous sites in Swauk Creek were the same as those used in the index sites.

Mainstem Yakima River Trout Population Estimates

From 1991 to 1994, trout populations were estimated in five sections of the mainstem Yakima River with mark-recapture methods (Ricker 1975) using a drift boat electrofishing unit. Estimates were conducted to assess the abundance, and spatial and temporal distribution of trout as described by McMichael et al. (1992). Juvenile spring chinook salmon were not included in these population estimates due to poor electrofishing efficiencies for small fish. One index site approximately 5 km long was electrofished within each of five study sections (Figure 1). 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). In each index area all trout species (trout include rainbow, cutthroat, bull, eastern brook, and hatchery steelhead trout) were captured and marked on two successive nights using a drift boat electrofisher. One week later fish were recaptured on two successive nights. Methods were described in detail by McMichael et al. (1992). We calculated lineal trout densities and biomass for each of the five sites surveyed. Area1 trout density and biomass were calculated by dividing the population and biomass estimates by the area of water within each index site (methods are described in Martin et al. 1994). The percent composition of rainbow trout in each of the five sites is also reported, and was calculated by dividing the total number of rainbow trout collected by the total number of trout captured in each section. The percentage of rainbow trout greater than 250 mm (10") captured within each section is also reported. This length category is reported due to the interest that the public has in "large" rainbow trout available to the catch and release fishery.

Results

Tributaries

Variability of rainbow trout density ranged from stable to highly fluctuating in tributary index sites. In the 13 tributary index sites that have been monitored since 1990, we have observed a wide range of temporal abundance variability with CV values ranging from 15 to 78 (Table 1). The remaining 14 sites (which have been sampled less than 5 years) have exhibited much higher CV with several sites having an inter-annual CV greater than 100. Although the 13 long-term index sites in general, have exhibited moderate temporal variability, spatial variation among index sites and tributaries was high (Table 2, Figure 2).

Average densities of rainbow trout in index sites were highest in Taneum and Swauk creeks, while Cabin Creek, the mainstem of the Teanaway River and the North Fork of the Teanaway River were the lowest (Table 2). Rainbow trout densities in individual index sites in 1994 were significantly correlated to rainbow trout densities in those same sites in 1993 ($r=0.54$, $P=0.0107$, $df=20$). Significant correlations also existed between rainbow trout densities between index sites in 1994 and 1992 ($r=0.66$, $P=0.0080$, $df=21$), and between 1994 and 1991 ($r=0.60$, P=0.0173, df=14). No significant correlations were observed between rainbow trout densities in 1990 and any other year sampled, which may be attributed to the small number of index sites surveyed in 1990.

Although rainbow trout density varied moderately between years, rainbow trout biomass variability within index sites was high (Figure 3). The overall mean fork length of rainbow trout in the tributary index sites appeared to be similar among years and tributaries (Table 2, Figure 4). Rainbow trout in Umtanum Creek index sites had the largest mean length in 1994, followed in order by Cabin and Swauk creek index sites. Average fish length may be an artifact of sampling date. For example, Umtanum Creek was sampled latest of all streams, and therefore the fish had more time to grow than in the other tributaries.

Table 1. Mean rainbow trout density (SD), coefficient of variation of inter-annual rainbow trout density, and stability category for index sites within tributaries of the upper Yakima River that have been sampled since 1990. Sites are arranged in order of stability. In this study, CV values less than 25 were considered to represent a stable population, values between 25 and 50 represent a moderately stable population, values between 50 and 75 represent a moderately fluctuating population, and values greater than 75 represent a fluctuating population.

Site	Density (#/m^2) (mean)	Density (#/m^2) (SD)	Coefficient of Variation	Stability Category
MFT 3	0.068	0.011	15	Stable
NFT 1	0.027	0.006	21	Stable
TAN 2	0.096	0.026	27	Moderately Stable
WFT 1	0.023	0.007	29	Moderately Stable
WFT 2	0.053	0.016	31	Moderately Stable
WFT 3	0.031	0.100	31	Moderately Stable
NFT 3	0.009	0.004	45	Moderately Stable
NFT 2	0.032	0.019	61	Moderately Fluctuating
MFT 1	0.065	0.042	65	Moderately Fluctuating
MFT 2	0.042	0.029	68	Moderately Fluctuating
CAB 2	0.022	0.017	76	Highly Fluctuating
CAB 1	0.007	0.779	78	Highly Fluctuating

[a]CAB=Cabin Creek, NFT=North Fork Teanaway River, MFT=Middle Fork Teanaway River, WFT=West Fork Teanaway River, and TAN=Taneum Creek.

Table 2. Rainbow trout density (#/m²), biomass (g/m²), and mean fork length (Ln; mm) of fish > 79 mm, for each index site sampled in each upper Yakima River tributary from 1990 through 1994. 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			1994		
Site'	Density	Biomass	Ln	Density	Biomass	Ln	Density	Biomass	Ln	Density	Biomass	Ln	Density	Biomass	Ln
MAN1							0.038	0,641	93.6				0.044	0.922	103.5
MN2							0.039	1.168	132.5				0.044	1.446	136.2
MAN3							0.000	0.000					0.000	0.000	
avg							0.026	0.606	113.1				0.029	0.734	119.9
SD							0.022	5.58;:	68.1				0.025	0.732	23.1
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	0.033	0.603	117.0
NFT2	0.070	2.001	124.1	0.030	0.709	120.3	0.021	0,492	123.8	0.031	0.618	120.7	0.011	0.196	115.3
NFT3	0.005	0.123	82.5	0,006	0.269	148.3	0.013	0.245	111.0	0.005	0.144	108.2	0.015	0.510	134.6
avg	0.065	0,823	101.0	0.022	0.575	131.7	0.022	0.44;	118.6	0,020	0.432	118.6	0.020	0,436	120.6
SD	0.048	1.026	21.2	0.014	0.265	14.7	0.009	0.181	6.7	0.013	0.253	9.5	0,012	0.213	12.2
JUN				0.020	0.190	NA	0.060	0.110	100.5	0.150	1.793	100.3	0,013	0.123	95.0
TAN1				0.087	13,466	139.3	0,233	7,061	133.9	0.198	5.596	128.1	0.098	2,466	124.2
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	0.105	3.427	131.4
TAN3	0.060	1.336	74.1	0.025	0.528	113.9	0.026	0.944	136.2	0.033	2.849	128.2	0.115	2.789	120.7
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	0.104	2.694	125.4
SD	0	1.033	24.2	0.032	6.95	14.4	0.103	3.07	2.2	0.083	1.39	7.3	0,011	0.489	5.5
MFT1	0.016	4.04'1	117.0	0.059	1.438	122.8	0.044	1.090	131.4	0.030	0.911	138.1	0.043	1,349	138.7
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	0.015	0,380	129.8
MFT3	0.060	2.414	109.3	0.050	1.983	145.6	0,074	2,366	140.0	0,061	1.277	120.7	0.075	1,961	129.0
avg	0.066	3.223	114.5	0,051	1.525	132.4	0.049	1.376	130.8	0,040	1.01	131.0	0.044	1,230	132.5
SD	0.045	0.817	4.5	0.008	0.423	11.8	0,024	0.882	9.5	0.01	0.234	9.1	0.030	0.797	5.4
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	0.017	0.385	124.3
WFT2	0.036	1.107	132.0	0,056	1.720	134.2	0.075	1.875	129.4	0.037	1.091	135.2	0.063	1.995	139.3
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	0.043	0.818	113.6
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	0.041	1.066	125.8
SD	0.015	0,399	19.7	0.018	0.666	140.0	0.030	0.755	4.5	0.001	0.344	15.0	0.023	0.833	12.P
CAB1	0.016	0.043	90.5	0.008	0.193	128.0	0.013	0.25;	122.6	0.000	0.000		0,015	0.395	128.9
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	0.005	0.404	175.7
avg	0.022	0.275	100.2	0,025	0.222	157.0	0.030	0.731	121.6	0.005	0.293	132.7	0.010	0.400	152.3
SD	0.001	0.328	13.6	0,025	0.041	41.0	0,024	0.677	1.4	0.006	0.414	0,000	0.007	0.006	33.1

Table 2 (Continued)

Site'	1990			1991			1992			1993			1994		
	Density	Biomass	Ln	Density	Biomass	Ln	Density	Biomass	Ln	Density	Biomass	Ln	Density	Biomass	Ln
MST1													0.010	0,374	146.4
MST2													0.012	0.203	128.0
MST3													0.004	0.101	129.0
avg													0.009	0,253	134.8
SD													0,004	0. 139	10.0
SWK1										0.113	5.232	157.5	0,200	15.417	163.6
SWK2							0,242	8.713	142.9	0.120	4.111	135.2	0,070	2.997	152.4
SWK3							0,103	2.557	125.7	0.105	2.630	126.0	0.076	2.363	137.8
avg							0.173	5.638	134.3	0.115	3.991	139.6	0,115	6.926	151.3
SD							0.098	4.357	12.2	0.012	1.305	16.2	0.073	7.361	12.9
BIG							0.071	1.979	126.5						
UMT1							0,111	1.618	107.2	0.086	1,912	119.2	0.060	1.704	135.3
UMT1.5										0.016	2,113	228.8	0.013	1.410	212.3
UMT2							0.016	0.624	151.7	0.012	1,657	216.7	0.000	0.000	
avg							0.063	1.121	129.5	0.038	1,894	188.2	0,024	1.065	173.9
SD							0,067	0.703	31.5	0.042	0,229	60.1	0,032	0.941	54.4

' MAN = Manastash Creek, NFT = North Fork Teanaway River, JUN = Jungle Creek, TAN = Taneum Creek, MFT = Middle Fork Teanauay River, CAB = Cabin Creek, MST = Mainstem of the Teanaway River, SWK = Swauk Creek, BIG = Big Creek, UMT = Umtanum Creek.

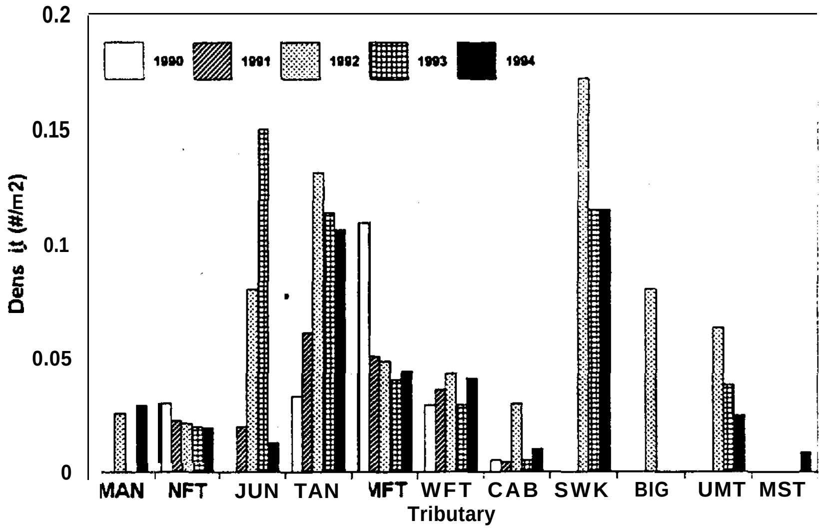

Figure 2. Mean rainbow trout density ($\#/m^2$) in 10 Yakima River tributaries sampled from 1990 through 1994. Vertical lines represent the range between the maximum and minimum index site densities for each year sampled.

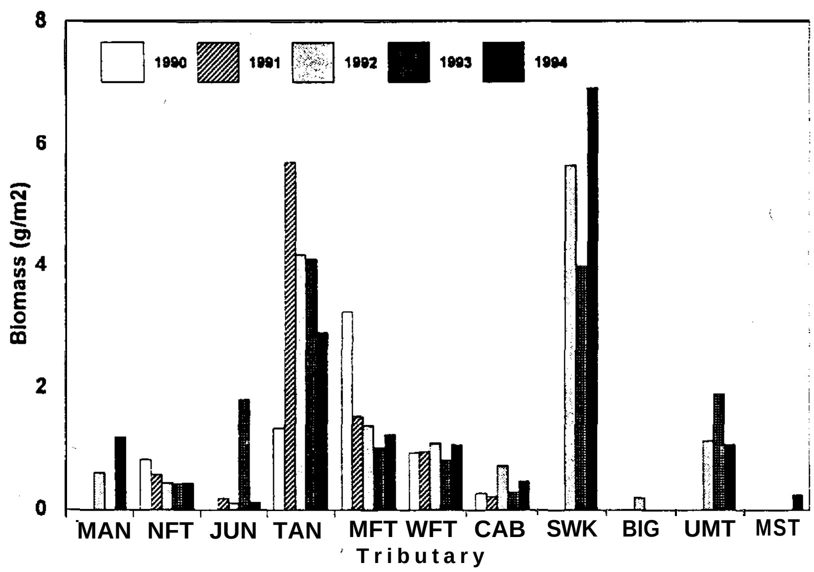

Figure 3. Mean rainbow trout biomass (g/m^2) in 10 Yakima River tributaries sampled from 1990 through 1994. Vertical lines represent the range between the maximum and minimum index site densities each year.

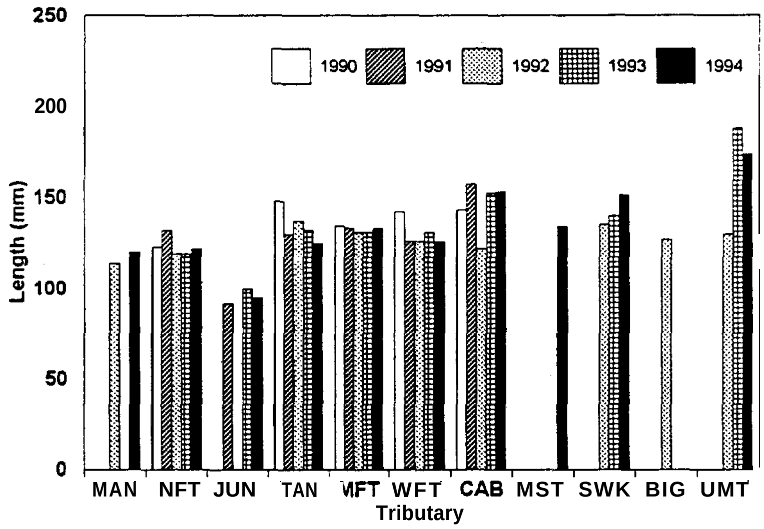

Figure 4. Mean length of rainbow trout collected in ten tributaries of the Yakima River from 1990 through 1994. Vertical lines represent the range between the mean maximum and mean minimum length of rainbow trout collected from index sites.

‘The density and biomass of spring chinook salmon and bull, brook, and cutthroat trout exhibited high spatial variation within sampling years but were similar among years at individual sites (Tables 3 and 4). Of the 99 population estimates conducted in the 28 index sites from 1990 through 1994, we found no spatial overlap between spring chinook salmon and bull trout. Spring chinook salmon did overlap with brook or cutthroat trout in 6 (6%) of the 99 sites. All juvenile spring chinook salmon were observed in sites less than 730 m elevation, while bull trout were observed only in the highest elevation index and systematic sites in the North Fork of the Teanaway River (1,103 m elevation). Tributary index sites ranged in elevation from 469 m to 1,341 m. Cutthroat and brook trout always inhabited sites higher than 677 m elevation. Rainbow trout were the most ubiquitous salmonid species, being observed in 95 of the 99 sites since 1990.

Table 3. Density (#/m²) of juvenile spring chinook salmon (SPC) and bull (BUL) , eastern brook (EBT) and cutthroat trout (CUT) in each Yakima River tributary index site surveyed from 1990 to 1994. 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				1994			
Site'	SPC	BUL	EBT	CUT	SPC	BUL	EBT	CUT	SPC	BUL	EBT	CUT	SPC	BUL	EBT	CUT	SPC	BUL	EBT	CUT
MAN1									0.01	0	0.0	0.0					0.051	0	0	0
MAN2									0	0	0.018	0.011					0	0	0.007	0.007
MAN3									0	0	0.046	0.103					0	0	0.275	0.164
avg									0.01	0	0.032	0.057					0.017	0	0.940	0.056
SD									0.006		0.020	0.065					0.029	0	0.157	0.090
NFT1	0.027	0	0	0	0	0	0	0	0	0	0	0	0.014	0	0	0	0.002	0	0	0
NFT2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.001	0.0	0	0	0	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	0	0.0060		0,038
avg	0.009	0,003	0	0.028	0	0	0	0.010	0	0.001	0	0.008	0.005	0.001	0.000	0.009	0.001	0.0020		0.013
SD	0.016	0,005	0	0.048	0	0	0	0.018	0	0.002	0	0.014	0.008	0.002	0.001	0.016	0.001	0.0030		0,022
JUN					0	0	0	0	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	0	0	0	0.005
TAN2	0	0	0.002	0	0	0	0.005	0	0	0	0,005	0	0	0	0.002	0	0	0	0	0.002
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	0	0	0.029	0.020
avg	0	0	0.006	0.005	0	0	0,007	0,004	0	0	0.005	0.004	0	0	0.008	0.011	0	0	0.010	0.009
SD	0	0	0.006	0.101	0	0	0.003	0.008	0	0	0.003	0	0	0	0.008	0.011	0	0	0.017	0.010
MFT1	0.044	0	0	0.001	0	0	0	0	0.002	0	0	0	0	0	0	0	0.004	0	0	0.002
MFT2	0,005	0	0	0.001	0	0	0	0.002	0	0	0	0	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	0	0	0	0
avg	0,025	0	0	0.001	0	0	0	0.001	0.001	0	0	0.001	0	0	0	0	0,001	0	0	0.001
SD	0.028	0	0	0	0	0	0	0.001	0.001	0	0	0.001	0	0	0	0	0.002	0	0	0,001
WFT1	0.017	0	0	0	0	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	0	0	0	0
WFT3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.001
avg	0.010	0	0	0	0	0	0	0	0.001	0	0	0	0	0	0	0	0	0	0	0.000
SD	0.010	0	0	0	0	0	0	0	0.001	0	0	0	0	0	0	0	0	0	0	0.001
CAB1	0.004	0	0.013	0	0	0.	0.004	0	0.005	0.0	0	a	0	0	0	0	0	0	0,021	0
CAB2	0.001	0	0,004	0.006	0	0	0	0	0	0.0	0	0.002	0	0	0	0	0	0	0	0.002
avg	0.003	0	0.009	0.003	0	0	0.002	0	0.003	0.0	0	0.001	0	0	0	0	0	0	0.011	0.001
SD	0.002	0	0.006	0.0004	0	0	0.003	0	0.004	0.0	0	0.001	0	0	0	0	0	0	0.015	0.001

Table 3 (Continued)

	1990				1991				1992				1993				1994			
Site'	SPC	BUL	EBT	CUT	SPC	BUL	EBT	CUT	SPC	BUL	EBT	CUT	SPC	BUL	EBT	CUT	SPC	BUL	EBT	CUT
MST1																	0,089	0	0	0.001
MST2																	0	0	0	0
MST3																	0.001	0	0	0
avg																				
SD																				
SWK1													0.260	0	0	0	0.509	0	0	0
SWK2									0,005	0	0	0	0.008	0	0	0.002	0	0	0	0
SWK3									0	0	0	0.018	0	0	0	0.019	0	0	0	0.006
avg									0.003	0	0	0.009	0.134	0	0	0.011	0.111	0	0	0,002
SD									0	0	0	0	0.178	0	0	0.012	0.293	0	0	0.003
BIG									0	0	0	0.005								
UMT1									0	0	0	0	0,304	0	0	0	0.009	0	0	0
UMT1.5													0	0	0	0	0	0	0	0
UMT2									0	0	0	0	0	0	0	0	0	0	0	0
avg									0	0	0	0	0.101	0	0	0	0.003	0	0	0
SD									0	0	0	0	0	0	0	0	0.005	0	0	0

*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, MST = Mainstem of the Teanaway River, SWK = Swauk Creek, BIG = Big Creek and UMT = Umtanum Creek.

Table 4. Biomass (g/m^2) of juvenile spring chinook salmon (SPC) and bull (BUL) , eastern brook (EBT) and cutthroat trout (CUT) in each Yakima River tributary index site surveyed from 1990 to 1994, 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				1994			
Site'	SPC	BUL	EBT	CUT	SPC	BUL	EBT	CUT	SPC	BUL	EBT	CUT	SPC	BUL	EBT	CUT	SPC	BUL	EBT	CUT
MAN1									0,093	0	0	0					0.444	0	0	0
MAN2									0	0	0.799	0,437					0	0	0.115	0,059
MAN3									0	0	2.160	2,347					0	0	5.450	4.447
avg									0.003		1.480	1.392					0.148	0	1.855	1.499
so									0		0.962	1.351					0.256	0	3.114	2.553
NFT1	0.287	0	0	0	0	0	0	0	0	0	0	0	0,092	0	0	0	0.011	0	0	0
NFT2	0	0	0	0	0	0	0	0	0	0	0	1.494	0	0	0.018	0	0	0	0	0
NFT3	0	0.653	0	4,241	0	0	0	2.112	0	0.009	0	0	0	0.067	0	1.454	0	0.003	0	0
avg	0.091	0	0	0	0	0	0	0.704	0	0.003	0	0.498	0.031	0.022	0.006	0.484	0	0	0	0
SD	0.166	0	0	0	0	0	0	1.219	0	0.005	0	0.863	0.053	0.039	0.010	0,839	0	0	0	0
JUN					0	0	0	0	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	0	0	0	0.289
TAN2	0	0	0.010	0	0	0	0.499	0	0	0	0.664	0	0	0	0.039	0	0	0	0	0.189
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	0	0	0.783	0,630
avg	0	0	0.011	0,554	0	0	0.430	0.274	0	0	0,293	0.158	0	0	0	0.223	0	0	0.261	0.353
SD	0	0	0.001	0	0	0	0.098	0	0	0	0,328	0	0	0	0.433	0.069	0	0	0	0.252
MFT1	0.988	0	0	0	0	0	0	0	0.010	0	0	0	0	0	0	0	0.018	0	0	0.115
MFT2	0.064	0	0	0.113	0	0	0	0.015	0.002	0	0	0	0	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	0	0	0	0
avq	0,526	0	0	0.081	0	0	0	0.005	0.006	0	0	0.016	0	0	0	0	0.004	0	0	0.038
SD	0.653	0	0	0.046	0	0	0	0	0,006	0	0	0	0	0	0	0	0	0	0	0
WFT1	0,236	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
WFT2	0.028	0	0	0	0	0	0	0,046	0.010	0	0	0	0	0	0	0	0	0	0	0
WPT3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0,818
avq	0.132	0	0	0	0	0	0	0.015	0.003	0	0	0	0	0	0	0	0	0	0	0.273
SD	0.147	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
CAB1	0.036	0	0.650	0	0	0	0.586	0	0.017	0	0	0	0	0	0	0	0	0	0.556	0
CAB2	0.013	0	0.370	0.320	0	0	0	0	0	0	0	0.020	0	0	0	0	0	0	0	0.041
avg	0.025	0	0.510	0.016	0	0	0.294	0	0.009	0	0	0.010	0	0	0	0	0	0	0.278	0.021
SD	0.016	0	0.198	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table 4. (continued)

Site[1]	1990 SPC	BUL	EBT	CUT	1991 SPC	BUL	EBT	CUT	1992 SPC	BUL	EBT	CUT	1993 SPC	BUL	EBT	CUT	1994 SPC	BUL	EBT	CUT
MST1																	0,159	0	0	0.265
MST2																	0	0	0	0
MST3																	0,005	0	0	0
avg																	0,055	0	0	0.088
SD																	0.090	0	0	0
SWK1													1.617	0	0	0	3.389	0	0	0
SWK2									0,034	0	0	0	0.064	0	0	0,090	0	0	0	0
SWK3									0	0	0	0.646	0	0	0	0.440	0.010	0	0	0.145
avg									0.017	0	0	0.323	1.129	0	0	0.265	1.133	0	0	0.048
SD									0.024	0	0	0.457	0,691	0	0	0.247	1.954	0	0	0.084
BIG									0	0	0	0,094								
UMT1									0	0	0	0	2.710	0	0	0	0.052	0	0	0
UMT1.5													0	0	0	0	0	0	0	0
UMTL									0	0	0	0	0	0	0	0	0	0	0	0
avg													1.355	0	0	0	0.017	0	0	0
SD													1.916	0	0	0	0.030	0	0	0

[1]Index 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, MST = Mainstem of the Teanaway River SWK = Swauk Creek, BIG = Big Creek and UMT = Umtanum Creek.

The two index sites that did not contain rainbow trout were the highest elevation site (1,341 m) in Manastash Creek in 1992 and 1994, the highest elevation Umtanum Creek site in 1994, and the lowest elevation (719 m) site in Cabin Creek in 1993. In general, cutthroat and brook trout densities were highest in high elevation index sites. Cutthroat trout were found in 34% of the 99 sites from 1990 to 1994, and only between 677 and 988 meters elevation. Although cutthroat trout were generally collected from high elevation tributary sites, cutthroat trout were also collected from the Yakima River at lower elevations. Brook trout were found in 20% of the sites from 1990 to 1994, only at elevations between 719 and 988 m. As with cutthroat trout, brook trout were also collected in mainstem Yakima River sections.

As in 1993 (Martin et al. 1994), in 1994 it appeared that rainbow trout density was independent of the presence of other salmonids (Figure 5). In 1994, the only statistically significant correlation observed between rainbow trout density and the density of any other salmonid species was between rainbow trout and spring chinook salmon ($r=0.43$, $P=0.0078$, $df=36$). Although in previous years sampling we found no significant correlation between density of spring chinook salmon and any other salmonid species, the spatial distribution of spring chinook salmon overlapped completely with that of rainbow trout. In 1994, rainbow trout were collected in 25 of 27 (93%) of the tributary index sites. These sites represented a wide array of habitat conditions (Appendix 1C).

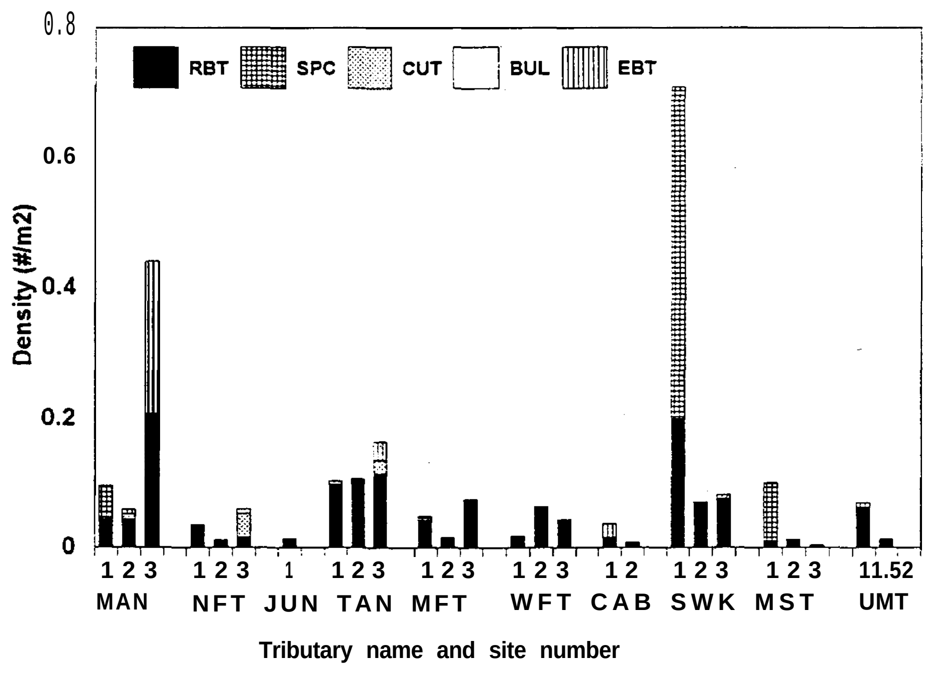

Figure 5. Rainbow trout (RBT), juvenile spring chinook salmon (SPC), cutthroat trout (CUT), bull trout (BUL), and eastern brook trout (EBT) densities (#/m^2) in index and systematically selected sites sampled in 1994.

In 1993 and 1994 we found few physical site habitat variables that correlated with rainbow trout density. In 1993, site area, site width, and discharge were significantly correlated with rainbow trout density (Martin et al. 1994). In 1994, discharge was the only physical site habitat variable that correlated with rainbow trout density.

Rainbow trout densities in 1994 appeared to be loosely correlated with site area, stream width, and channel gradient, but these relationships were not statistically significant ($P<0.10$) (Table 5). In 1991 there was a significant correlation between site elevation and rainbow trout density but we believed elevation could also be associated with fish abundance through its relationship to stream temperature. In 1994 we assessed this relationship with the use of continuous reading thermographs. A significant correlation was found between site elevation and average and maximum stream temperatures ($r=-0.79$, $P=0.0014$, $df=12$ and $r=-0.81$, $P=0.0007$, $df=12$ respectively). Even though rainbow trout density increased with elevation, and stream temperature decreased with elevation, we found no significant correlations between rainbow trout density and maximum, minimum, or coefficient of variation of stream temperature. Mean, maximum, and coefficient of variation of stream temperature is presented in Table 6.

Table 5. 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 = 99) by year. Standard deviations (SD) of thalweg depth were recorded from 1992 to 1994.

Physical variables	1990	1991	1992	1993	1994
Site elevation (m)	-0.03	-0.46*	-0.37	-0.11	-0.08
Site area (m^2)	-0.03	0.05	-0.14	-0.43*	-0.27
Mean site width (m)	-0.11	-0.04	-0.05	-0.41*	-0.20
Thalweg depth (SD)	No data	No data	0.59**	0.16	0.13
Gradient	0.08	No data	-0.34	0.28	-0.20
Discharge (m^3/sec.)	-0.23	0.18	-0.25	-0.41*	-0.40**
Total pool area (m^2)	-0.03	0.77**	0.34	-0.11	0.03
Number of pools	-0.04	0.28	-0.01	0.12	0.10
Maximum site depth (m)	-0.33	0.56**	0.25	-0.11	0.10
Maximum temperature(°C)	No data	No data	No data	No data	-0.04
Average temperature(Y)	No data	No data	No data	No data	-0.07
Percent error rate					
a = 0.10	N/A[a]	12	45	N/A[a]	N/A[a]
a = 0.05	40	7	45	15	55

* P < 0.10; ** P < 0.05
not applicable because no significant relationships were found at this a level

Table 6. Mean, maximum, and coefficient of stream temperature variation (CV) in 15 tributary index sites from July 29 to September 30, 1994.

		Water Temperature (°C)		
Stream[a]	Site	Mean	Max	CV[b]
CAB	1	12.7	19.0	0.16
CAB	2	10.7	17.2	0.20
MFT	1	15.2	20.9	0.12
MFT	2	9.2	23.9	----
MFT	3	13.3	19.0	0.15
NFT	1	15.4	22.9	0.17
NFT	3	9.0	12.5	0.13
WFT	1	15.0	21.1	0.17
WFT	2	11.0	23.1	----
WFT	3	14.1	21.4	0.19
MST	1	16.3	24.3	0.17
MST	2	16.8	23.9	0.17
SWK	1	11.7	22.5	----
SWK	2	14.2	21.2	0.16
TAN	1	15.1	22.7	0.17
TAN	3	11.3	16.7	0.17
UMT	1	16.2	21.6	0.12
UMT	1.5	16.0	20.9	0.12

[a]CAB=Cabin Creek, NFT=North Fork Teanaway River, MFT=Middle Fork Teanaway River, WFT=West Fork Teanaway River, MST=Mainstem Teanaway River, SWK=Swauk Creek, TAN=Taneum Creek, and UMT=Umtanum Creek.
[b] Coefficient of Variation (SD/mean).

Contrary to our findings in the Teanaway basin in 1993, in Taneum Creek we found that rainbow trout densities in systematically selected sites were not significantly higher than in index sites. In fact, the average density in index sites was higher than in the systematically selected sites (0.1063 and 0.0940 rainbow trout per m^2, respectively), although the difference was not statistically significant ($t=0.36$, $P=0.7300$, $df=7$) (Table 7).

Table 7. Rainbow trout and total salmonid densities in three index sites and 6 systematically selected sites in Taneum Creek, 1994. The random sites are listed adjacent to the index site that they were closest to within the stream.

Site type		Rainbow trout density (#/m²)		Total salmonid density (#/m²)	
Index	Systematic	index	Systematic	Index	Systematic
1	A	0.0982	0.1894	0.1036	0.2006
	B		0.1358		0.1409
2	C	0.1054	0.0737	0.1075	0.0753
	D		0.0566		0.0704
3	E	0.1153	0.0660	0.1643	0.1246
	F		0.0407		0.1660

In 1994 we evaluated the variability in abundance of rainbow trout in four contiguous 100 m reaches in lower Swauk Creek located approximately 0.5 km downstream of the lowest Swauk Creek index site. Rainbow trout density varied among these four sites, ranging from 0.0 to $0.0292/m^2$ ($mean=0.0135/m^2$, $SD=0.0126$). Comparing the rainbow trout density in the four contiguous sites to the one index site (rainbow trout density = $0.200/m^2$), in lower Swauk Creek indicated that the observed rainbow trout densities in the four contiguous sites averaged one-tenth of the index site populations. Even though rainbow trout densities in the contiguous sites were much lower than the one'index site, t-test comparisons of the four contiguous sites indicated that the rainbow trout density in these four sites did not differ significantly from one another ($P=0.1211$, $t=2.15$, $df=3$). These results indicate that the index site may not be representative of the average rainbow trout density in lower Swauk Creek. Another factor that may result in high temporal abundance variability is site length. In Swauk Creek, it appears that rainbow trout abundance varies considerably among 100 m contiguous sites, and even among 25 m reaches within the 100 m sites (Figure 6). Our 100 m index sites, may therefore, be too short to encompass the natural spatial variability in tributaries.

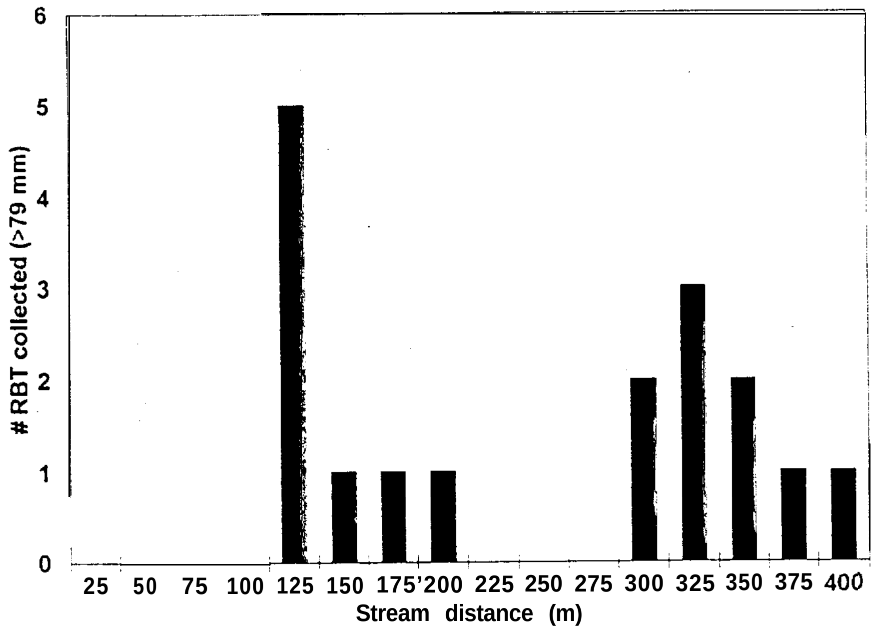

Figure 6. The number of rainbow trout collected in 25 m reaches of four 100 m contiguous sites in lower Swauk Creek, 1994.

Mainstem Yakima River

Trout population estimates within mainstem Yakima River index sections varied among 1991, 1992, 1993, and 1994. Trout population estimates for the five sections combined, increased from 7,101 in 1992, to 8,939 in 1993, and then decreased to 7,466 in 1994 (Table 8). Refer to Martinet al. (1994) for interpretation and analysis of data'prior to 1994.

Table 8. Mainstem Yakima River site length (km), trout population estimate and biomass (kg) in each site from 1991 through 1994.

	1991			1992		1993		1994	
Section	Site Length	Estimate	Biomass	Estimate	Biomass	Estimate	Biomass	Estimate	Biomass
1 (LCYN)	4.5	1,414	355	1,754	527	1,280	475	a44	211
2 (UCYN)	4.5	1,232	238	1,503	236	1,480	304	1,660	343
3 (EBURG)	4.3	1,167	191	894	124	2,349	315	1,293	202
4 (THORP)	5.9	1,774	305	927	132	1,413	259	1,609	202
5 (CELUM)	6.3	(2,200)[a]		2,023	338	2,417	474	2,070	381
TOTAL		(7,807)		7,101	1,357	3,939	1,827	7,466	1,339

[a] Projected number because 1991 population estimate was not valid. Figures in parentheses are the average of 1992 and 1993 trout population estimates for the CELUM section.

The combined estimated biomass of trout in the index sections increased from 1,357 kg in 1992, to 1,827 kg in 1993, and decreased to 1,339 in 1994 (Table 8). Trout density (#/km) within each of the five sections also varied somewhat among years (Figure 7, Table 9). Trout biomass also varied between years, but in general was highest in LCYN (Figure 8, Table 8), which can be attributed to the larger size of trout in the LCYN section (Figure 9).

Table 9. Mainstem Yakima River trout density and biomass per kilometer in each index site from 1991 through 1994.

Year	LCYN	UCYN	EBURG	THORP	CELUM
		# of trout/km			
1991	314	274	292	306	---[a]
1992	390	334	224	160	323
1993	284	329	587	244	384
1994	188	369	369	277	329
Avg.	294	327	368	247	345
		kg of trout/km			
1991	79	53	48	53	---[a]
1992	117	52	31	23	54
1993	106	68	79	45	75
1994	47	76	55	35	61
Avg.	87	62	53	39	63

"Invalid estimate in 1991

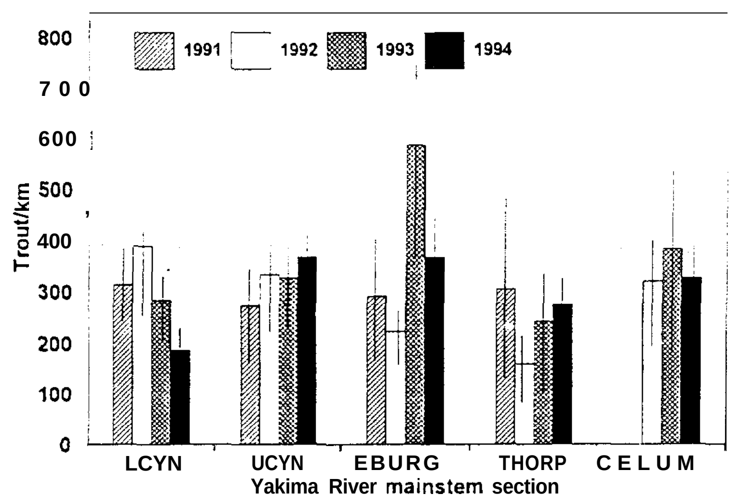

Figure 7. Trout population estimates (P/km) (rainbow, bull, cutthroat, eastern brook trout) in the five index sections of the mainstem Yakima River sampled in 1991, 1992, 1993, and 1994. Vertical lines are 95% confidence intervals around the population estimate.

Spatial and temporal variation in mean fork length of trout captured in the five mainstem sites was minimal. In general, mean fork length was greatest for trout in the LCYN section, followed by the CELUM section. The trout with the shortest mean fork length were captured in the EBURG or THORP sections (Figure 9). Large trout (>250 mm) were captured in the canyon sections of the Yakima River with greater frequency than other sections sampled in most years (Figure 10).

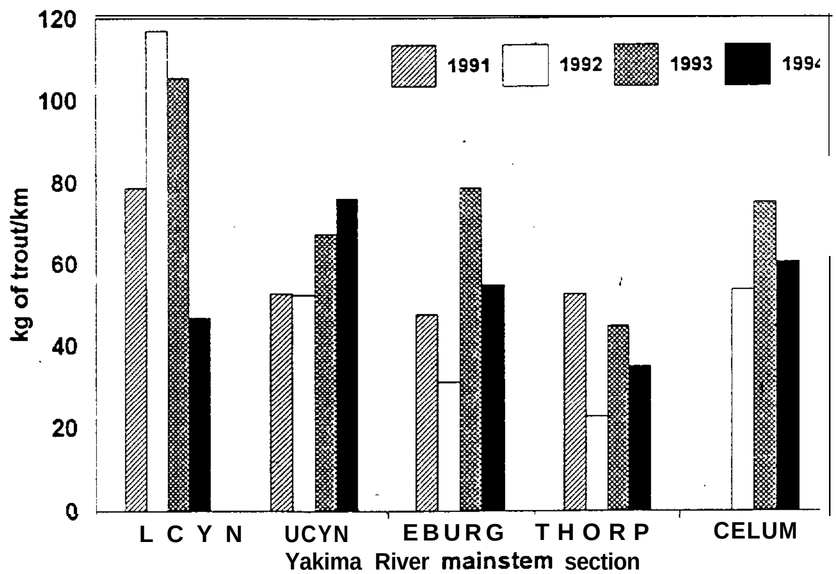

Figure 8. Estimates of trout biomass (kg/km) in the five index sections of the mainstem Yakima River sampled in 1991, 1992, 1993, and 1994.

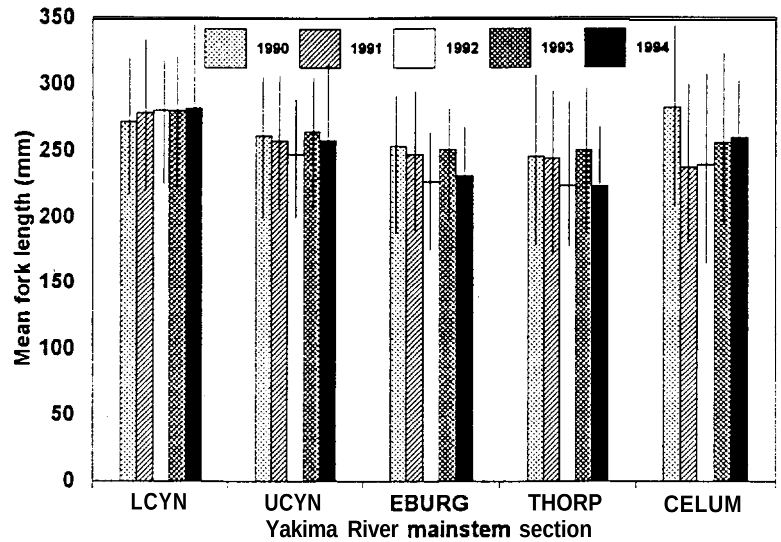

Figure 9. Mean fork length (mm) of rainbow trout captured in the five index sections of the mainstem Yakima River from 1990 through 1994. Vertical lines represent 95% confidence intervals.

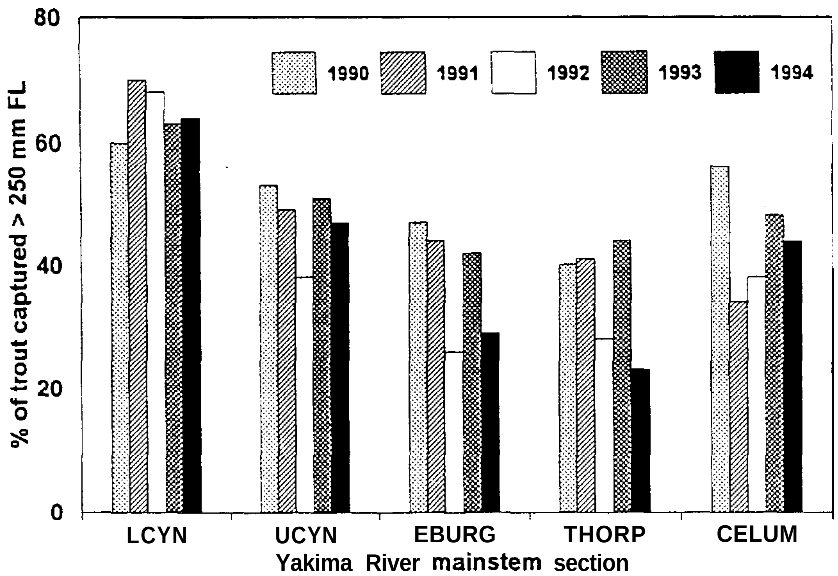

Figure 10. Percent of trout captured whose fork length was greater than 250 mm in five index sections of the mainstem Yakima River from 1990 through 1994.

Although the lineal density of trout was highest in the UCYN section, the area1 density ($\#/m^2$) was highest in the CELUM section. This was also true for biomass, and is due to smaller site areas in the upper elevation sections than in the lower ones (Table 10).

Species composition of rainbow in the mainstem Yakima River varied little among years (Table 11). In general, the percentages of bull, cutthroat and eastern brook trout increased from low elevation to high elevation (Table 11).

Although average area1 density ($\#/m^2$) of trout in index sections of the mainstem Yakima River (0.007 fish/m') was only one-sixth of the density in index sites of seven tributaries (0.0476 fish/m^2) in 1994, the difference was not significant (t = 1.57, P = 0.1275, df = 27). Lineal fish densities, however were considerably higher in the mainstem than in seven tributaries sampled.

Table 10. Trout density (#/m^2) and biomass (g/m^2) in five mainstem Yakima River index sites, 1994.

Site	Site area (m^2)	Density (#/m^2)	Biomass (g/m^2)
1 (LCYN)	243,508	0.0035	0.87
2 (UCYN)	202,854	0.0082	1.69
3 (EBURG)	174,195	0.0074	1.16
4 (THORP)	260,437	0.0062	0.78
5 (CELUM)	213,116	0.0097	1.79

Table 11. Trout species percent composition for each mainstem Yakima River index site surveyed from 1991 through 1994. Totals for some sites do not equal 100% because hybrid trout were not included.

	Percent composition															
	1991				1992				1993				1994			
Site	RBT	CUT	BUL	EBT	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	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	99.6	0.2	0.0	0.0
3 (EBURG)	99.6	0.0	0.0	0.0	99.1	0.0	0.0	0.0	99.7	0.3	0.0	0.0	99.4	0.6	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	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	96.4	3.3	0.3	0.0

RBT = rainbow trout, CUT = cutthroat trout, BUL = bull trout, EBT = eastern brook trout, HSH =juvenile hatchery steelhead

Discussion

In Yakima River tributary index sites, the abundance of rainbow trout exhibited a wide range of temporal variability over the study period. Although we have collected physical habitat data in an attempt to explain observed abundance variability, we remain unable to construct a predictive model. It appears that there is some population regulation mechanism occurring in tributaries that we currently are not measuring. Some mechanisms that might explain the variation observed include stochastic events which result in habitat alterations and changes in productivity. Other factors may include fish movement, species interactions, anthropogenic disturbance, or mortality resulting from repeated electrofishing.

In 1993 we tested the hypothesis that factors associated with repeated electrofishing in index sites affects rainbow trout density. We found that rainbow trout density was significantly lower in index sites that were electrofished annually, than in systematically selected sites that had not been previously electrofished and came to the conclusion that perhaps electrofishing was having an affect on fish abundance (see Martin et al. 1994). This hypothesis was tested in 1994 in Taneum Creek, and contrary to our findings in the Teanaway Basin, we found no significant differences between index and systematically selected, previously unsampled sites. Based on our findings in Taneum Creek, we believe that the index sites in the Teanaway

Basin were not representative of the rainbow trout population (due to their selection by potentially biased 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. This explanation is substantiated by our findings in Swauk Creek where rainbow trout density in the index site was more than 10 times the density in four randomly selected 100 m contiguous sites located only 500 m downstream from the index site. The index site (SWK 1) was chosen in 1990 due to its close proximity to the road and therefore may not be representative of the population abundance in lower Swauk Creek. Due to the potential for habitat alterations to occur between years as a result of flooding or other stochastic events, it may be necessary to increase the length of our tributary index sites so that the small-scale spatial variability of rainbow trout abundance can be accounted for.

As in 1993, in 1994 we found that rainbow trout area1 densities in tributaries were substantially higher than in the mainstem. This can be attributed to differences in rainbow trout habitat use in tributaries verses mainstem sections. In the mainstem, rainbow trout occurred primarily along the banks while in tributaries they were found throughout the site.

Total trout abundance for the five mainstem index sections combined neither increased nor decreased significantly from 1991 to 1994; however, distribution changed markedly. A large decrease in abundance in the LCYN coupled with an increase in UCYN in 1994 and a general increase in the last two years in EBURG and THORP sections has resulted in a distribution of fish whose abundance peaked in the center of our study reach (Figure 7). Although we may only speculate at this time, increased fishing pressure, which may result in unintended hooking mortality, in the LCYN may be one reason for the decrease in abundance there (Appendix A). Upper river sections however, exhibited increased trout abundance in 1993 and 1994. Martin et al. (1994) suggested that an explanation for the increased trout abundance in the EBURG and CELUM sections in 1993 was that fish may be recruiting to the upper river sections to spawn and, as a result, production in these reaches had increased. Although not as pronounced in 1994, greater production in these sections may have occurred in 1994 as well (see Figure 7). This hypothesis may be tested in 1995, as we have conducted redd surveys in each of the index sections since 1993 (see Update 1, this report). Progeny of fish that reproduced in 1993 would be included in our population estimate conducted in 1995. Similarly, fish that reproduced in 1994 will be included in the 1996 population estimate. Knowledge of the number of redds and the subsequent number of 1 year old fish 18 months later in the population, will allow us to test the hypothesis that rearing density is related to redd abundance in each section, assuming that immigration and emigration are equal.

Based on four annual estimates, it appeared that the abundance of trout in the upper Yakima River (all sections combined) was quite stable. As in 1993 (Martin et al. 1994),one limiting factor to rainbow trout production in the upper Yakima

River is flow fluctuations during the first month of life followed by high summer discharge that continues until the second week of September (for a complete treatment of this topic see Martin et al. (1994)):

In conclusion, based on the observed temporal and spatial variability of rainbow trout abundance in tributaries of the -upper Yakima River, we offer the following recommendations. First, to minimize negative impacts to fish due to electrofishing in tributaries, one-pass electrofishing estimates should be used in index sites. Second, in an attempt to contain the spatial variability of rainbow trout abundance, we recommend increasing tributary index site length. Last, temporal and spatial variation in abundance may be related to stochastic events or -stream productivity. Therefore, we recommend that water temperature and discharge variability be monitored throughout the year, and that stream productivity be assessed by-determining macro-invertebrate abundance, water chemistry, and solar input in index sites.

Update 4

Species associated with rainbow trout and juvenile spring chinook salmon in the upper Yakima Basin

Introduction

Information about fish species associated with rainbow trout and juvenile spring chinook salmon has been collected as part of the Yakima Fisheries Project's (YFP) prefacility characterization of the upper Yakima basin. Knowledge about species associated with target species is important because it may help to assess the variability of target species demographics (e.g. age and size structure) and density, and aid in more comprehensive management approaches such as ecosystem management. Information about species associations in the upper Yakima basin has been presented by Hindman et al. (1991), McMichael et al. (1992), Pearsons et al. (1993), and Pearsons and Martin (1994). The objective of this report is to present updated information (through December 31, 1994) on species associated with rainbow trout and spring chinook salmon in index sites of the upper Yakima basin.

Methods

Fish species associated with rainbow trout and spring chinook salmon were determined by electrofishing tributary and mainstem index sites. Except for minor deviations, which are described below, methods were similar to those presented by Pearsons and Martin (1994). Briefly, the numbers of fish collected from two electrofishing passes in 100 m index sites in tributaries were counted by species on various occasions from July to September, 1994. In mainstem index sites, fish were electroshocked and visually identified to species or genus and their numbers were visually estimated. These mainstem surveys were conducted from September to October, 1994.

During '1994, two methods were used to determine if visual estimates of electroshocked fish were accurate. First, numbers of estimated fish were compared to actual counts of fish recorded using a video camera. Second, the number of estimated fish were compared to a running tally of fish numbers that were spoken and recorded on an audio tape recorder. In the first method, one person video-taped the fishes that were electrofished around the anode and, at the same time, a second person estimated the number of fish observed. In the second method, a person verbally recorded the number of fish observed on a continuously recording portable tape-recorder. The same observer estimated the number of fish observed. Estimates of abundance from video and audio tapes were calculated by reviewing tapes and tallying numbers of occurrences in the lab.

Results

The numbers of fish collected in tributary index sites and their percent contribution to the assemblage are provided in Table 1. Coarse level patterns of fish abundance and distribution appear to be similar to previous years (Pearsons and Martin 1994). Sculpins, dace, and rainbow trout were found in almost every index site. Torrent and paiute sculpins were broadly distributed, whereas mottled sculpin were mainly observed in the mainstem and shorthead sculpin in high elevation tributary sites (Table 2). In general, longnose and speckled dace were absent from the highest elevation sites and their occurrences were inversely associated with one another (Table 2). Percent composition and distribution of rainbow trout were similar among years (Table 1). The abundance of mountain whitefish and sucker species was extremely low in the tributaries sampled, despite the high abundance of these taxa in the mainstem (Table 3), and the documentation of spawning migrations by bridgelip suckers into tributaries during the spring (Pearsons and Martin, 1994). Consistent with findings from tributary index sites, assemblage composition within mainstem index sites was similar between 1993 and 1994 (Table 3).

Comparisons of the three methods for determining relative abundance of fish in the mainstem were only partially successful. Counting video images of fish was determined to be unfeasible due to several critical biases including: 1) except for the largest fish, such as suckers, trout and squawfish; and smaller fish tended to be extremely difficult to identify to species; and 2) multiple counting of fish occured which could not be accounted for on the video images. However, keeping a running tally of fish observed using a tape recorder was quite successful. Comparisons of the percent compositions using the two viable methods revealed surprisingly similar results (Table 4). Percent compositions of species using the two different methods were not statistically different (Chi-square = 0.680, P= 1.00, df = 7).

Table 1. Percent composition of fish captured in 1992, 1993, and 1994, during electrofishing surveys in tributary index sites. Each site was 100 m long and was electrofished two times. The following codes were used: RBT = rainbow trout, CUT = cutthroat trout, EBT = eastern brook trout, BUL = bull trout, SPC = spring chinook salmon, SPD = speckled dace, LND = longnose dace, RSS = redside shiner, SCP = sculpin spp., BLS = bridgelip sucker, LSS = largescale sucker, OTH = other species.

	Percent Composition												Total Number
Site	RBT	CUT	EBT	BUL	SPC	SPD	LND	RSS	SCP	BLS	LSS	OTH	of Fish
						1992							
MAN1	52	0	0	0	0.6	9	10	0.4	28	0.2	0	0	466
MAN2	3	0.7	2	0	0	0	0	0	95	0	0	0	406
MAN3	0	16	13	0	0	0	0	0	71	0	0	0	218
NFT1	26	0	0	0	0	0.4	42	0	26	0	0	0	265
NFT2	9	0	0	0	0	0	6	0	84	0	0	1'	136
NFT3	9	16	0	3	0	0	0	0	67	0	0	6[b]	70
JUN1	0.9	0.5	0	0	0	0.5	3	0	18	0	0	77[b]	213
TAN1	58	0.9	0	0	0	0	0	0	30	0	0	11[b]	220
TAN2	21	0	1	0	0	0	0	0	78	0	0	0	292
TAN3	12	5	6	0	0	0	0	0	71	0	0	6[b]	129
MFT1	28	0	0	0	0.5	0.5	14	0	57	0	0	0	208
MFT2	21	0	0	0	0	0	42	0	37	0	0	0	268
MFT3'	24	0.7	0	0	0	0	2	0	60	0	0	13[bc]	152
WFT1	17	0	0	0	0	2	42	0	36	0	1	2[c]	254
WFT2	20	0	0	0	0.4	10	29	0	41	0	0	0	228
WFT3	38	0	0	0	0	1	9	0	52	0	0	0	157
CAB1	3	0	0.7	0	1	0.3	0	0	95	0	0	0	286
CAB2	18	5	0	0	2	0	0	0	71	0	0	4[b]	82
SWK1[a]	2	0	0	0	1	53	0	11	0.3	32	0	0.2[d]	375
SWK2	32	0	0	0	0.3	0.3	26	0	41	0	0	0	328
SWK3	10	2	0	0	0	0	8	0	62	0	0	18[b]	275
BIG1	14	0.7	0	0	0	2	0	0	61	0	0	22[b]	284
UMT1	70	0	0	0	0	4	0	0	26	0	0	0	151
UMT2	1	0	0	0	0	91	0	0	8	0	0	0	279
						1993							
NFT1	21	0	0	0	3	1	47	0	28	0	0	0	188
NFT2	28	0	1	0	0	0	15	0	55	0	0	0	73
NFT3	12	27	0	5	0	0	0	0	56	0	0	0	77
JUN1	77	0	0	0	0	3	3	0	18	0	0	0	88
TAN1	68	1	0	0	0	0	0	0	21	0	0	9[b]	76
TAN2	43	0	6	0	0	0	0	0	48	0	0	4[b]	101
TAN3	28	6	13	0	0	0	0	0	52	0	0	1[b]	111
MFT1	20	0	0	0	0	0	39	0	41	0	0	0	82
MFT2	14	0	0	0	0	0	68	0	18	0	0	0	113
MFT3	52	0	0	0	0	0	0	0	48	0	0	0	99
WFT1	9	0	0	0	0	6	42	1	43	0	0	0	197
WFT2	20	0	0	0	0	6	35	0	40	0	0	0	139

WFT3	38	0	0	0	0	0	16	0	47	0	0	0	45
CAB1	3	0	0	0	0	0	0	0	97	0	0	0	34
CAB2	6	0	0	0	0	0	0	0	94	0	0	0	144
SWK1	10	0	0	0	17	56	0	7	4	5	0	0.2[d]	450
SWK2	24	0.4	0	0	2	0	18	0	56	0	0	0	260
SWK3	34	6	0	0	0	0	0	0	57	0	0	3[b]	109
UMT1	25	0	0	0	47	14	0	0	14	0	0	0	85
UMT2	16	0	0	0	0	61	0	0	23	0	0	0	31
UMT3	2	0	0	0	0	87	0	0	3	9	0	0	188
NFTA	9	40	0	4	0	0	0	0	45	0	0	1.5[c]	67
NFTB	39	0	0	0	0	0	0	0	61	0	0	0	74
NFTC	37	0	0	0	0	0	12	0	45	0	0	6[a]	84
NFTD	52	0	1	0	0	0	12	0	31	0	0	4[a]	87
NFTE	26	0	0	0	1	0	55	0	19	0	0	0	186
MFTA	61	0	0	0	0	0	1	0	38	0	0	0	129
MFTB	60	0	0	0	0	0	0	0	40	0	0	0	169
MFTC	33	0	0	0	2	0	21	0	44	0	0	0	161
MFTD	39	0	0	0	0	0	11	0	50	0	0	0	116
MFTE	36	0	0	0	5	4	18	0	37	0	0	0	132

1994

MAN1	6	0	0	0	6	60	1	7	16	3	0	0	255
MAN2	9	1	2	0	0	0	0	0	88	0	0	0	213
MAN3	0	23	36	0	0	0	0	0	40	0	0	0	262
NFT1	18	0	0	0	1	0	66	0	16	0	0	0	191
NFT2	29	0	0	0	0	0	15	0	56	0	0	0	54
NFT3	10	24	0	3	0	0	0	0	63	0	0	0	99
JUN1	13	0	0	0	0	7	55	0	26	0	0	0	31
TAN1	39	3	0	0	0	0	0	0	58	0	0	0	104
TAN2	22	0.5	0.5	0	0	0	0	0	77	0	0	0	219
TAN3	30	5	17	0	0	0	0	0	48	0	0	0	145
MFT1	16	1	0	0	1	11	18	1	51	0	0	0	165
MFT2	8	0	0	0	0	0	45	0	47	0	0	0	180
MFT3	37	0	0	0	0	0	1	0	62	0	0	0	123
WFT1	11	0	0	0	0	7	20	3	58	1	0	0	333
WFT2	15	0	0	0	0	4	42	0	39	0	0	0	321
WFT3	29	0.6	0	0	0	2	29	0	40	0	0	0	156
CAB1	2	0	3	0	0	0	2	0	93	0	0.3	0	395
CAB2	3	1	0	0	0	0	0	0	96	0	0	0	111
SWK1	12	0	0	0	6	52	19	5	4	2	0.1	0.3[d]	762
SWK2	18	0	0	0	0	0	16	0	66	0	0	0	236
sWK3	17	1	0	0	1	0	0	0	81	0	0	0	168
UMT1	64	0	0	0	1	18	0	0	10	7	0	0	126
UMT2	26	0	0	0	0	63	0	0	11	0	0	0	92
UMT3	0	0	0	0	0	98	0	0	1	1	0	0	158
MST1	14	0.5	0	0	11	33	33	0	10	0	0	0	218
MST2	16	0	0	0	0	5	12	0	66	1	0	1'	152
MST3	3	0	0	0	0.5	0	59	0	37	0	0	0.2'	208
TANA	12	0.2	0	0	0	0	0	0	88	0	0	0	535
TANB	37	1	1	0	0	0	0	0	62	0	0	0	167
TANC	36	1	0	0	0	0	0	0	64	0	0	0	202
TAND	46	2	8	0	0	0	0	0	44	0	0	0	98
TANE	24	9	17	0	0	0	0	0	50	0	0	0	120
TANF	12	24	17	0	0	0	0	0	47	0	0	0	109
SWKA	'18	0	0	0	7	13	54	0	5	4	0	0	56

SWKB	39	0	0	0	12	26	9	1	7	3	1	1[d]	151
SWKC	6	0	0	0	5	67	3	1	1	8	0	10[d]	486
SWKD	8	0	0	0	6	41	7	3	1	15	6	13[de]	240

[a] mountain whitefish
[b] unidentified age 0+ trout
[c] putative cutthroat x rainbow trout hybrid
[d] northern squawfish
[e] unidentified sucker
[s] site was snorkeled

Table 2. Percent composition of species within two genera, *Cottus* and Rhinichthys, collected in 100 m long index sites of upper Yakima River tributaries, 1994. Fish were captured with a backpack electrofisher (two passes). The following codes were used: SPD = speckled dace, LND = longnose dace, TSC = torrent sculpin, PSC = paiute sculpin, SSC = shorthead sculpin, MSC = mottled sculpin.

	Percent Composition							
Site	Rhinichthys			Cottus				
	SPD	LND	Total Number	TSC	PSC	ssc	MSC	'Total Number
MAN1	98	2	157	71	29	0	0	42
MAN2	0	0	0	8	17	75	0	188
MAN3	0	0	0	0	0	100	0	106
NFT1	0	100	123	28	62	10	0	29
NFT2	0	100	8	34	28	38	0	29
NFT3	0	0	0	0	44	55	0	62
JUN1	11	89	19	25	0	75	0	8
TAN1	0	0	0	13	87	0	0	60
TAN2	0	0	0	22	78	0	0	169
TAN3	0	0	0	35	0	65	0	69
MFT1	38	63	48	20	80	0	0	84
MFT2	0	100	81	5	95	0	0	85
MFT3	0	100	1	61	39	0	0	76
WFT1	25	75	89	20	80	0	0	194
WFT2	9	91	148	46	54	0	0	124
WFT3	6	94	48	60	32	8	0	62
CAB1	0	100	9	1	48	52	0	366
CAB2	0	0	0	0	0	100	0	107
SWK1	74	26	541	74	26	0	0	27
SWK2	0	100	38	17	83	0	0	156
SWK3	0	0	0	15	85	0	0	137
UMT1	100	0	23	0	100	0	0	1 2
UMT2	100	0	58	10	90	0	0	10
UMT3	100	0	155	0	100	0	0	1
MST1	50	50	142	27	73	0	0	22
MST2	28	72	25	9	82	9	0	100
MST3	0	100	123	23	70	3	4	77
TANA	0	0	0	36	64	0	0	469
TANB	0	0	0	15	85	0	0	103
TANC	0	0	0	18	82	0	0	129
TAND	0	0	0	30	42	28	0	43
TANE	0	0	0	5	0	95	0	60
TANF	0	0	0	4	0	96	0	51
SWKA	19	81	37	100	0	0	0	3
SWKB	74	26	54	90	10	0	0	10
SWKC	96	4	338	67	33	0	0	3
SWKD	85	15	115	100	0	0	0	3

Table 3. Percent composition of fish observed during electrofishing surveys in the mainstem Yakima River, 1993 and 1994. Rainbow trout (RBT), juvenile spring chinook salmon (SPC), mountain whitefish (MWF), sculpin species (SCU), sucker species (SUK), redside shiners (RSS), northern squawfish (SQW), and others (OTH) were observed. Sculpin species observed include torrent sculpin, mottled sculpin, and paiute sculpin. Sucker species observed include largescale sucker, bridgelip sucker, and mountain sucker.

Section	Percent Composition								Number
	RBT	SPC	MWF	SCU	SUK	RSS	SQW	OTH	
				1993					
LCYN	9	14	39	8	21	<1	9	a	2410
UCYN	6	22	43	2	22	1	5	b	995[k]
EBURG	4	26	39	4	15	0	12	c	1387[k]
THORP	3	24	30	18	19	5	1	d	3678
CELUM	4	23	39	27	5	<1	1	e	4009
				1994					
LCYN	7	12	37	10	21	11	3	f	3617
UCYN	8	10	35	15	25			g	3827
EBURG	7	13	30	10	18	<1	20	h	4028
THORP	6	25	24	21	14	6	5	i	3948
CELUM	6	21	57	12	5	1	<1	j	4386

The following numbers of fish were observed during all four surveys combined:

a 2 cutthroat trout and 1 chiselmouth
b 1 cutthroat trout and 1 chiselmouth
c 1 burbot and 1 pumpkinseed
d 5 cutthroat trout, 1 chiselmouth and 9 dace
e 17 cutthroat trout, 2 brook trout, 74 dace, and 1 yellow perch
f 1 yellow perch
g 1 smallmouth bass; 1 brown trout, 18 dace, 6 brook lamprey, 5 yellow perch and 13 chiselmouth
h 5 brook lamprey, 1 pumpkinseed, 4 yellow perch and 46 chiselmouth
i 6 dace, 5 carp, 1 yellow perch and 4 chiselmouth
j 54 dace, 4 cutthroat trout, 1 brook trout and 1 yellow perch
k averages of 2 station totals for each bank (4 stations were sampled)

Table 4. Comparison of the number of fish and percent composition of taxa using two methods in the mainstem Yakima River (Bighorn to Umtanum) on September 23, 1994. "Tape" refers to estimates that were calculated using a tape recorder and "Visual" refers to estimates that were calculated by visually observing electroshocked fish and estimating the number of fish observed. Rainbow trout (RBT), juvenile spring chinook salmon (SPC), mountain whitefish (MWF), sculpin species (SCU), sucker species (SUK), redside shiners (RSS), northern squawfish (SQW), and others (OTH) were observed. Sculpin species observed include torrent sculpin, mottled sculpin, and paiute sculpin. Sucker species observed include largescale sucker, bridgelip sucker, and mountain sucker.

Species	Total Fish		Percent Composition	
	Tape	Visual	Tape	Visual
RBT	139	100	5.4	5.0
SPC	51	32	2.0	1.6
MWF	950	730	37.0	36.3
SUK	1132	870	44.1	43.2
scu	240	227	9.3	11.3
SQF	53	50	2.1	2.5
RSS	1	0	0.0	0.0
CHM	4	4	0.2	0.2
Total	2570	2013	100	100

Discussion

The percent composition of fish assemblages in tributary and index sites in the upper Yakima basin appeared to be relatively constant during the years sampled (Pearsons and Martin 1994). Despite the large numbers of fish that migrated in, out, or through low elevation sites such as Swauk Creek 1 (Chapter 2), the percent composition of species was as stable there as in some high elevation sites, including those higher up in the same stream. Fish migrations may not have drastically altered inter-annual species composition results because: 1) fish that originated from outside of index sites may have migrated through the sites or fish that originated within index sites did not emigrate from the index sites in large numbers; or 2) fish species may have migrated at approximately the same time and in the same relative proportions every year. Although fish migrations into or out of a site have the potential to influence fish assemblage structure among years, variation in assemblage structure in our index sites appeared to be larger in space than in time.

The percent composition of rainbow trout in tributaries appears to have considerable merit as a parameter to monitor the distribution and relative abundance of this fish. Based on our results to date, this parameter appears to have at least two desirable qualities: 1) it is relatively stable among years (unlike population densities in index sites), and 2) it is sensitive to changes in the abundances of other species which may reflect environmental changes. The utility of this parameter for long-term monitoring should be reviewed following additional data collection and analysis.

The methods that we used to describe the relative abundances of species in tributaries and mainstem sites appear to be accurate. To accurately determine species richness and percent species composition in reaches of tributary streams, our preliminary analyses suggest that a minimum of 200 individuals should be collected from slow and fast water habitat types, or a minimum of 25 individuals from a minimum length of stream equal to 40 channel widths (WDFW unpublished data). Most sampling efforts in tributaries approximated or exceeded these criteria. Criteria for effective sampling of fish assemblages in large rivers, such as the Yakima River have not been developed yet. However, in our studies, we believe that sufficient numbers of individuals and lengths of stream were sampled in the mainstem to estimate assemblage composition accurately. Of more concern is the accuracy and precision of the method we have used to assess assemblage composition (Pearsons and Martin 1994). The precision of our sampling approach appears to be relatively good based on a comparison of annual samples and a comparison of two enumeration techniques. However', a larger number of comparisons should be examined in the mainstem Yakima River before final acceptance of this technique for use by the YFP is adopted.

We recommend that additional work should be directed at understanding the biotic factors that influence rainbow trout and spring chinook salmon abundance and distribution. For instance, an increase in bridgelip sucker abundance may lead to decreased rainbow trout abundance in certain areas such as Umtanum Creek (Murdoch 1995). In addition, an increase in the abundance of northern squawfish may decrease the survival of spring chinook salmon (Pearsons 1994). Monitoring the abundance of species that strongly influence the abundance of target species can help to determine limitations to YFP success.

Update 5:

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

Introduction

Concerns about potential ecological impacts of hatchery fish releases on preexisting resident rainbow trout in the upper Yakima River prompted us to examine some mechanisms of competition between hatchery-reared steelhead juveniles and naturally-produced salmonids.

Because no Yakima Fisheries Project (YFP) facilities have yet been constructed, we used test fish from the nearest available source of hatchery steelhead. 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, 1993, and 1994 and examined behavioral interactions between various groups of coexisting fishes (McMichael et al. 1992; Pearsons et al. 1993; McMichael et al. 1994).

Our overall objective was to try to delineate some of the probable impacts that might result from interactions between juvenile steelhead from a YFP facility and pre-existing naturally-produced rainbow trout, and to develop methods for monitoring the intensity and outcomes of behavioral interactions. Specific objectives of this study 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, 6) determine whether hatchery-produced juvenile steelhead preyed upon juvenile wild salmonids, and 7) document the distribution of residual hatchery steelhead within the North Fork of the Teanaway basin (with emphasis on areas of overlap between hatchery steelhead and cutthroat and bull trout). We define residuals to be hatchery steelhead present in the drainage where they were released after June first of the year they were released.

This annual progress report covers the period from January through December, 1994, and the information presented should be considered preliminary. A status report on this aspect of our work will be produced following completion of field work in 1995.

Methods

This research was conducted within the Teanaway River drainage north of the town of Cle Elum, Washington. The Teanaway River is a tributary of the upper Yakima River. As described by McMichael et al. (1992), hatchery-produced steelhead were released into Jungle Creek, a tributary of the North Fork of the Teanaway River. In our experimental design, Jungle Creek was the small treatment stream (T,) and the North Fork of the Teanaway River, which was the large treatment stream (T_L). The fish released into Jungle Creek (at rkm 0.5) migrated downstream into the North Fork of the Teanaway River. Jack Creek flows into T_L approximately 1.6 km below the mouth of T,. Jack Creek was designated as 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 designated it a large control stream (C,). We also collected population abundance information from index sites within the West Fork of the Teanaway River for comparisons of rainbow trout abundance estimates between stream where hatchery steelhead were (treatment) and were not (control) released. The West Fork of the Teanaway River provided another large control stream (R_L) for comparisons of trout abundance.

Smolt Releases

Hatchery-reared steelhead smolts (target release number = 33,000 per year) were released into Jungle Creek (T,) during early May of 1991, 1992, 1993, and 1994 in a manner intended to mimic the outmigration pattern expected from an acclimation pond (McMichael et al. 1992). In 1994, smolts were released on May 2 (N=14,819), May 4 (N=11,248), and May 11 (N=6,512). The methods for the smolt releases were consistent with those described by McMichael et al. (1992).

Behavioral Observations

Direct underwater observation of fish agonistic behavior was performed by snorkeling in control (C, 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), Pearsons et al. (1993), and McMichael et al. (1994). Each agonistic interaction was classified into one of the following five groups, threat, crowd, chase, nip, or butt. A contest was defined as a discrete interaction or group of interactions between two specific fish without breaks between interactions of more than 1 minute. Many contests 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.

To determine whether juvenile hatchery-reared steelhead displaced wild fish, three spatial scales were examined using methods described by McMichael et al. (1992) and Pearsons et al. (1993). "Small-scale" displacements (< 4 m) were defined as those that occurred within a channel unit of stream, such as a pool. Wild fish movements out of the release stream (T_S) concurrent with large numbers of hatchery fish were defined as "mid-scale" displacements (S-500 m). "Large-scale" displacements (> 10 km) were monitored using similar analyses 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,).

Population Estimates

To determine the influence of hatchery steelhead releases on rainbow trout abundance, population abundance was assessed in four study streams. Population sizes were estimated in index sites in the North (T_L, N = 2), Middle (C,, N = 3), and West (R_L, N = 3) forks of the Teanaway River (1990-1994) and in Jungle Creek (T,, N = 1)(1991-1994) using the electrofishing methods described by McMichael et al. (1992). Sampling dates in 1994 were July 21 (T_L site 2), July 25 (T_L site 1), July 29, (C, site 3), August 10 (C, site 2), August 9, (C, site 1), July 28 (R_L sites 2 and 3), August 8 (R_L site 1), and September 9 (Jungle Creek, T,).

Predation

To determine whether residual hatchery steelhead, wild trout, or shorthead sculpin (*Cottus confusus*) preyed upon post-emergent wild rainbow trout we collected residual hatchery steelhead and sculpins from areas with abundant age 0+ rainbow trout (Pearsons et al. 1993). Fish were collected using backpack electrofishing equipment in the North Fork of the Teanaway River (T_L) and in Jungle Creek (T,) on July 27 and August 8, 1994. Stomach contents from residual steelhead, trout and sculpins were flushed using gastric lavage techniques (Light et al. 1983) and immediately examined for the presence of fish. Stomach contents were visually assessed and generally classified into the following groups: AI=aquatic invertebrates; AI/TR=aquatic invertebrates and terrestrial invertebrates; TR=terrestrial invertebrates.; FSH=fish; and NF=no food items.

Residual Hatchery Steelhead Distribution

To determine the extent of spatial overlap between residual hatchery steelhead and resident rainbow, cutthroat and bull trout in the North Fork of the Teanaway River (T_L), we snorkeled pools and runs' at 0.6 km intervals from the mouth of Jungle Creek upstream to a point 13.4 km upstream from the mouth of Jungle Creek. Snorkeling took place on June 24, June 30, and July 12 at a total of 22 sites. Each site was sampled by two snorkelers, each covering approximately 100 m of stream separated by about

100 lineal meters of stream. Snorkeling was conducted at the upper nine sites on the night of July 12 to better estimate bull trout presence given their nocturnal tendencies (Fraley and Shepard 1989). No bull trout were observed during diel (daylight) snorkeling efforts in these areas. Percentages of each species or group of salmonids were calculated for each site.

Results

Smolt releases

Total numbers, sizes, and smolt condition of hatchery steelhead released into Jungle Creek (T,) varied among the four years of study (Table 1). Mean lengths decreased each year, while mean condition factors were slightly higher each year. In only one year (1993) were more than 90% of the juvenile steelhead released classified as smolts based on external appearance. More complete statistical analyses of these data will be present in the status reports compiled after final data collection in 1995.

Table 1. Number released, sample sizes, mean fork length (mm, ± SD), mean weight (g, ± SD), mean condition factor (CF), mean percent classified as smolts, and mean percent precocial males for sampled hatchery steelhead released into Jungle Creek from 1991 to 1994.

	Number		Mean			Percent	
Year	Released	Sampled	Length	Weight	CF	Smolts	Prec.males
1991	31,542	100	201 (± 16)	81 (± 25)	0.98	< 50	4.0
1992	38,000	200	196 (± 16)	78 (± 22)	1.01	74	1.0
1993	22,500	150	182 (± 21)	64 (± 23)	1.02	97	0.7
1994	32,579	150	179 (± 21)	61 (± 24)	1.03	81	0.0

Behavioral Observations

Hatchery steelhead generally dominated contests with wild rainbow trout and were also larger. Similar to previous findings (McMichael et al. 1992; Pearsons et al. 1993; McMichael et al.

1994), hatchery steelhead in Jungle Creek (T_s) and the North Fork of the Teanaway River (T_L) dominated preexisting'wild trout in 70% of contests observed in 1994. When agonistic interactions among all groups of fish were pooled, larger fish dominated 84% of the contests observed. Hatchery steelhead were significantly larger than the resident trout in the study streams (McMichael et al. 1992; Pearsons et al. 1993; McMichael et al. 1994).

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 1994 (N = 20). Rainbow trout were dominant over spring chinook salmon in 11 (55%) of the contests.

In general, resident trout were observed at higher rates after the May smolt emigration period, whereas observation rates of hatchery steelhead were lower during that time (Table 2). 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 2). Interaction rates were generally higher in large streams than small streams during May, but higher in small streams after May. Interaction rates were typically lower during the smolt emigration period than they were during the summer (Table 2). This was particularly evident in Jungle (T_s) and Jack (C_s) creeks.

Table 2. 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 1994. 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_S	1994	698	0.14	8.16	0.00	635	15.7
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
T_L	1994	236	0.18	0.35	0.00	76	253.6
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_S	1994	526	0.43	0.00	0.00	400	249.3
C_L	1992	467	0.15	0.00	0.00	21	66.1
C_L	1993[b]	5	-	-			
C_L	1994	162	0.16	0.02	0.01	15	298.7

Table 2. continued

				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_S	1994	274	0.03	1.54	0.00	210	177.4
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
T_L	1994	543	0.40	0.26	0.01	122	60.4
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_S	1994	297	1.09	0.00	0.00	148	118.4
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
C_L	1994	594	0.49	0.01	0.07	280	140.3

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

The types of agonistic interactions observed in 1994 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 14% of the interactions observed in control streams and for 25% of the interactions observed in treatment streams (Figure 1).

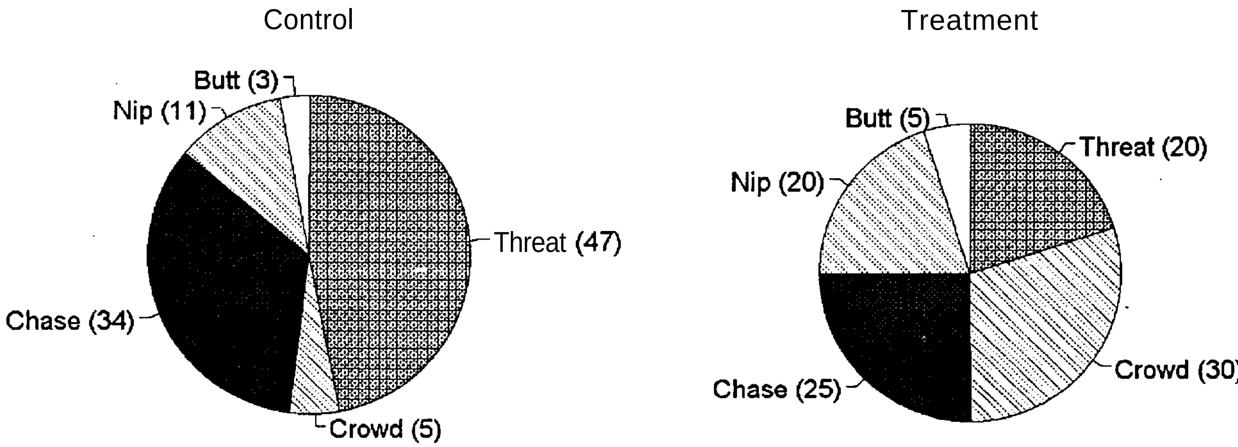

Figure 1. Percent (in parentheses) by type of agonistic interactions observed in control streams (C, and C_L, N = 828) and treatment streams (T_S and T_L, N = 1,254) during 1994.

Displacement

Hatchery steelhead displaced wild trout from apparently preferred micro habitats within habitat units, but did not displace trout from stream reaches over larger (0.2 to 11.2 km) spatial scales. In contrast to results from 1993, more of the agonistic interactions observed in control streams (69%) resulted in the displacement (typically within a channel unit) of the subordinate fish than was observed in treatment streams (59%). Mid-scale displacements were not detected in 1993 but may have occurred in Jungle Creek in 1994. Many of the trout that moved out of Jungle Creek (T,) were age 0+ and moved out in large numbers during the outmigration period of hatchery steelhead. The timing and magnitude of trout emigration was different between the release stream (T,) and the small control stream (C,), suggesting that the hatchery steelhead may have influenced the movement of trout out of the release stream in 1994 (Figure 2).

Large-scale (over 10 km) emigration of resident trout (and/or wild steelhead presmolts) did not appear to be affected by the magnitude and timing of hatchery steelhead outmigration. 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 emigrations of naturally-produced rainbow trout occurring concurrently with large outmigration pulses of hatchery steelhead from the North Fork of the Teanaway River (T_L) (Figure 3).

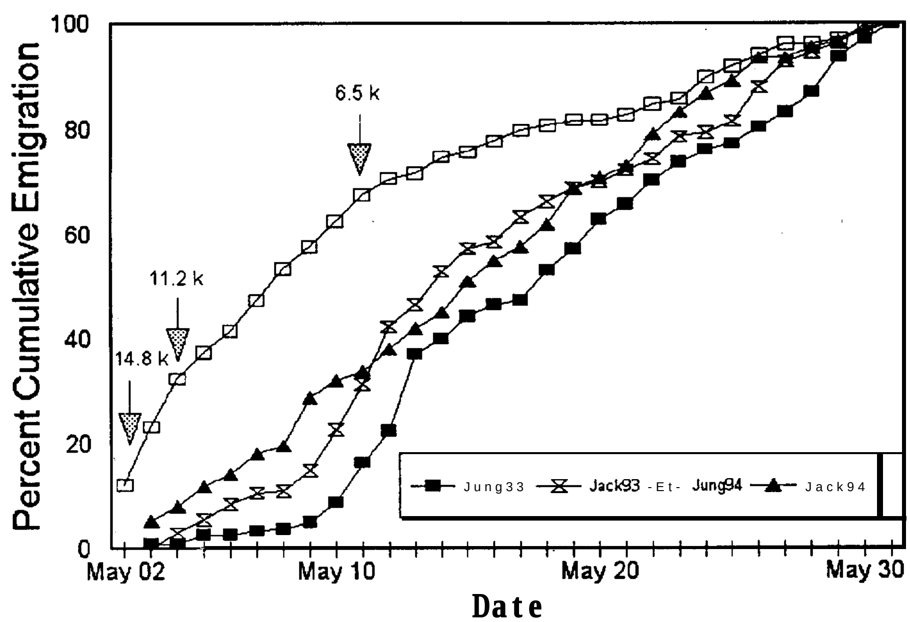

Figure 2. Cumulative emigration of naturally-produced trout (and/or wild steelhead presmolts) in Jungle and Jack creeks during May of 1993 and 1994. Arrows indicate dates and numbers of hatchery steelhead released in 1994.

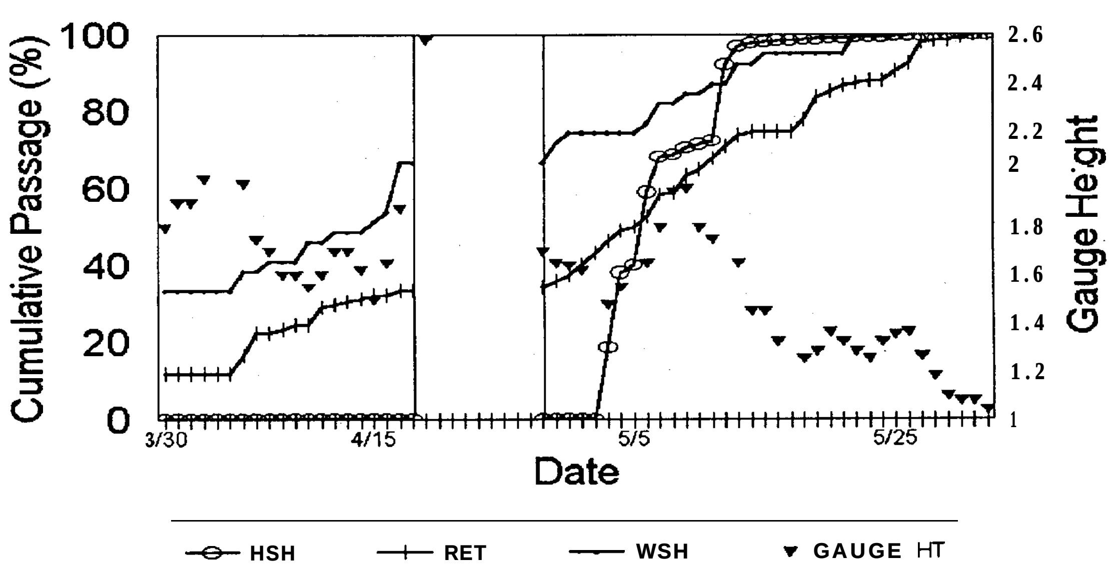

Figure 3. Cumulative outmigration of hatchery steelhead (HSH), wild steelhead smolts (WSH), and naturally-produced trout (and/or wild steelhead presmolts)(RBT) captured in a rotary screw fish trap near the mouth of the North Fork of the Teanaway River (T_L) during 1994. Gauge height (feet) is also shown. The trap was not operated from April 18 to 27.

In summary, while small-scale displacements were observed in all years, and mid-scale displacements may have occurred in 1994, and large-scale displacements were not seen in 1991 (McMichael et al. 1992), 1992 (Pearsons et al. 1993), 1993 (McMichael et al. 1994), or 1994. Also, about 50 percent of the rainbow trout and 75 percent of the wild steelhead we captured had emigrated prior to the release of hatchery steelhead.

Population Estimates

Rainbow trout densities appeared to have been influenced by the releases of hatchery steelhead. Mean annual rainbow trout abundance (number/100 m) and biomass (g/100 m) declined in the North Fork of the Teanaway River (T_L) and in control streams. However, pooled estimates in the North Fork of the Teanaway River were significantly lower than corresponding estimates for control streams (Figure 4). An analysis of covariance with the number of rainbow trout per 100 m as the dependent variable, treatment (hatchery steelhead releases) as main effects variable, and year as a covariate showed that treatment explained a significant ($F = 8.37$, $df = 1$, $P = 0.0201$) amount of the variability in trout abundance. In contrast to the larger streams, rainbow trout abundance and biomass in Jungle Creek (T,) was quite variable between 1991 and 1994 (Figure 5). Residual hatchery steelhead were abundant in Jungle Creek (T,) during all four years of sampling and due to their larger size, they constituted over 90% of the total salmonid biomass in 1991 and 1992, about half of the total biomass in 1993, and 92% of the biomass in 1994. Because Jack Creek (C,) became intermittent during 1992, 1993, and 1994, 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.

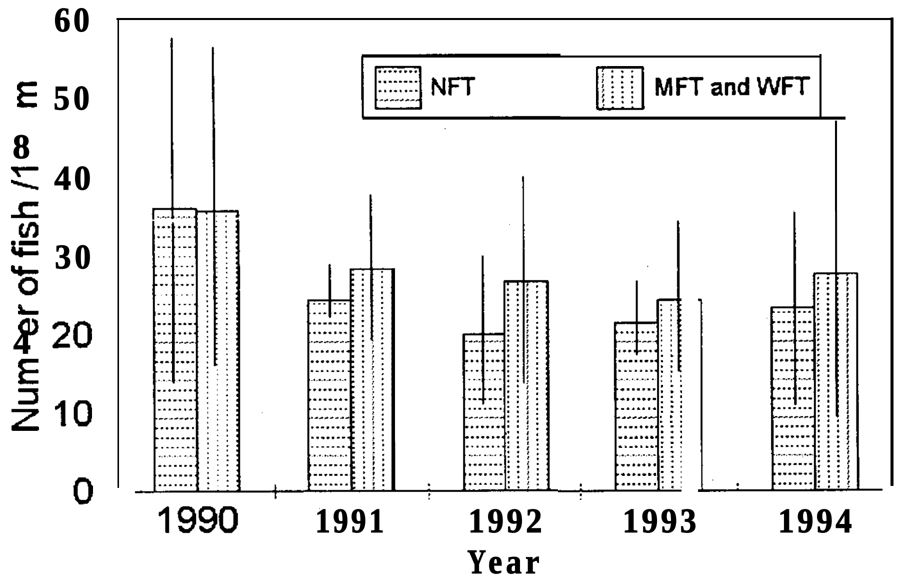

Figure 4. Mean rainbow trout population lineal density (number of fish/100 m) in the North Fork of the Teanaway River (T_L) and the Middle (C,) and West (R_L) forks of the Teanaway River from 1990 through 1994. Bars represent ± 1 standard deviation.

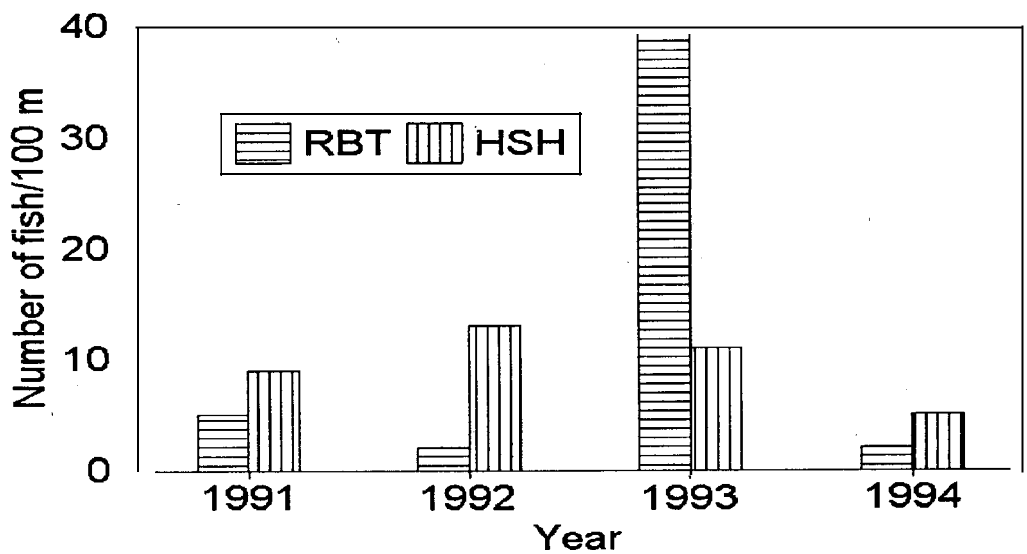

Figure 5. Population lineal density (number of fish/100 m) of rainbow trout (RBT) and residual hatchery steelhead (HSH) in Jungle Creek (T,) from 1991 through 1994.

Predation

Stomachs from 31 residual hatchery steelhead, 53 rainbow trout, two cutthroat trout, and 29 shorthead sculpin were collected in the North Fork of the Teanaway River (T_L) and Jungle Creek (T,) in July and August of 1994. Only one stomach contained a fish (Table 3). The one fish that was found in stomach samples was a small sculpin in the gut of a 215 mm rainbow trout. Newly-emerged age 0+ trout were abundant in the areas where the fishes were collected. No salmonid fry were observed in any of the stomach samples collected from residual hatchery steelhead in 1992 (N=55) or 1994 (N=31). In over 250 h of underwater observation between 1991 and 1994, no naturally-produced salmonids were consumed by hatchery steelhead and only one predatory attack (unsuccessful) was observed.

Table 3. Stomach contents of fishes collected in Jungle Creek (T,) and the North Fork of the Teanaway River (T_L) on July 27 and August 4, 1994, with sample sizes (N), and mean, minimum, maximum, and standard deviation (SD) of fork lengths (mm).

		Fish Length (mm)				Food Items (%)[b]				
Species[a]	N	Mean	Min	Max	SD	AI	AI/TR	TR	FSH	NF
CUT	2	141	127	155	12	50	0	0	0	50
RBT	53	117	62	215	27	21	18	2	2[c]	57
HSH	31	168	128	210	20	35	51	7	0	7
SHS	29	90	71	111	10	66	0	3	0	31

[a] CUT=cutthroat trout; RBT=rainbow trout; HSH=hatchery steelhead; SHS=shorthead sculpin.
[B] AI=aquatic invertebrates; AI/TR=aquat. invertebrates and terrestrial invertebrates.; TR=terrestrial invertebrates.; FSH=fish; NF=no food items.
[c] One rainbow trout (215 mm FL) had consumed a small sculpin.

Residual Hatchery Steelhead Distribution

Residual hatchery steelhead were encountered with higher frequency in 1991, 1992, and 1994 than in 1993 (Figure 6). A systematic snorkel survey of the North Fork of the Teanaway River upstream from the mouth of Jungle Creek in late June and early July, 1994, revealed that rainbow trout and residual hatchery steelhead were most abundant in lower reaches, while cutthroat and bull trout were more prevalent in upper reaches (Figure 7). Residual hatchery steelhead were present in the North Fork of the Teanaway River as far as 12.8 km upstream of the mouth of Jungle Creek. Bull trout were observed in the North Fork of the

Teanaway within 10.9 km of Jungle Creek. Therefore, residual hatchery steelhead had moved downstream (about 600 m) and out of Jungle Creek and then moved upstream in the North Fork of the Teanaway River into areas containing cutthroat and bull trout.

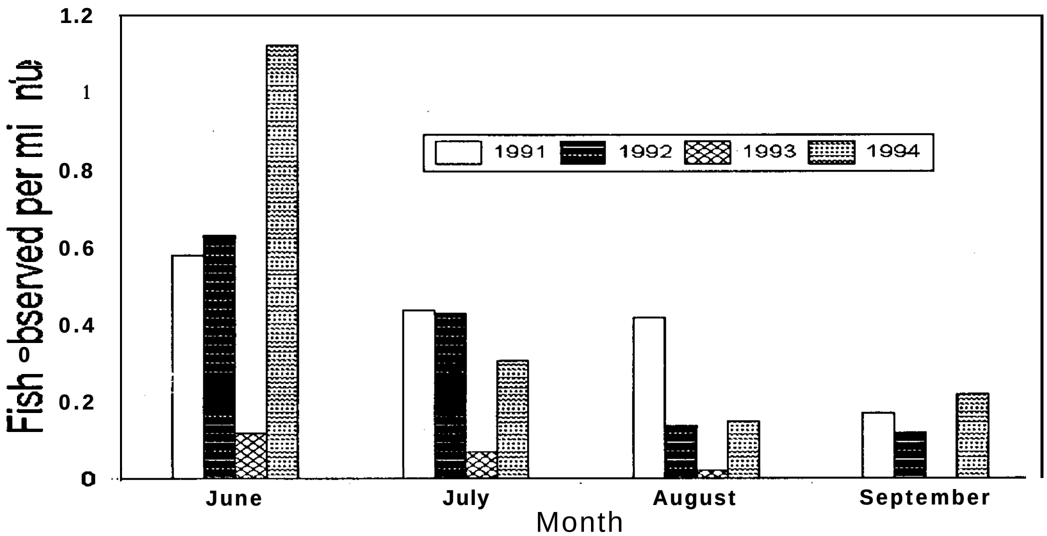

Figure 6. Pooled observations of residual hatchery steelhead (HSH) during snorkeling activities in the summer and early fall of 1991, 1992, 1993, and 1994. Data represent T_S (Jungle Creek) and T_L (North Fork of the Teanaway River) combined.

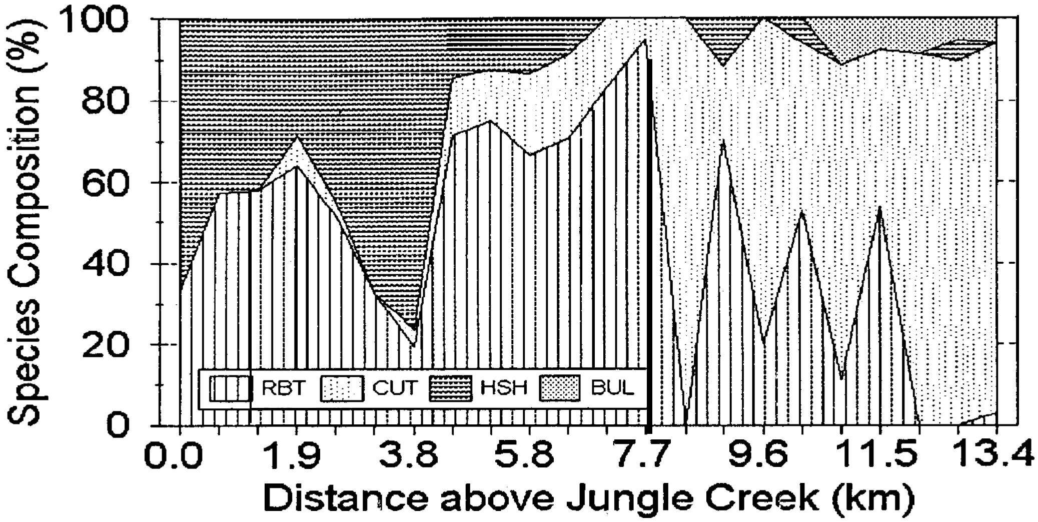

Figure 7. Linear distribution of salmonids upstream of the mouth of Jungle Creek on June 24, 30 and July 12, 1994. RBT = rainbow trout, CUT = cutthroat trout, BUL = bull trout, and HSH = residual hatchery steelhead. An average of 27 salmonids were observed at each of 22 sites (0.6 km apart). Sites from 8.3 to 13.4 km upstream from Jungle Creek were snorkeled at night, all other sites were snorkeled during daylight.

Discussion

The findings presented in this report are consistent with those in our previous reports (McMichael et al. 1992, Pearsons et al. 1993, McMichael et al. 1994) with a few exceptions that will be discussed here. Juvenile hatchery steelhead released into the Teanaway River system typically dominated preexisting wild trout, presumably because of their larger size or aggressive tendencies.

The types of agonistic interactions observed between the treatment and control streams in 1994 were similar, with respect to their apparent energetic costs, to those seen in 1993 (see McMichael et al. 1994 for a discussion of the relationship between type of interaction and its energetic cost). In 1994, the proportion of apparently energetically costly interactions in treatment streams was higher (25% of all interactions were nips and butts) than in control streams (14% were nips and butts). 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 were 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 Update 6, this report). The different types of interactions we observed in treatment and control streams may have influenced the amount of displacement we noted. For discussion purposes, it may be argued however, that because interaction rates were often higher in control streams, the overall effects of interactions may have been greater in control streams than in treatment streams even though interaction types in treatment streams were generally more violent. If we simply multiply the interaction rate by the percentage of interactions that we termed violent (nips and butts; McMichael et al. 1994) to determine the relative overall energetic cost to control and treatment populations we see that control streams, due to their higher interaction rates, often had a higher overall energetic cost due to interactions as a whole than did treatment streams. For example, in the June to October period in 1994, the interaction rate in T_L was 60.4 (x 10^5) and the violent portion of the interactions then was 25%; while in C_L interaction rate was 140.3 (x 10^5) and the violent interaction made up 14% of those observed. The following calculations,

treatment stream = 60.4 x 0.25 = 15.1, and

control stream = 140.3 x 0.14 = 19.6

illustrate that overall energetic costs to the population may, in some circumstances, have been higher in streams containing only wild fish than in those where hatchery steelhead were released. Mechanisms for this apparently lower 'interaction energy cost' in streams where hatchery steelhead were present may include behaviors exhibited by wild fish in which they elect to conceal themselves in the substrate or in other habitat features that could conceivably limit their ability to forage effectively. We have observed this type of behavior in wild trout and it is particularly evident soon after the release of large numbers of hatchery steelhead. So, while overall energy expense may possibly be lower in treatment streams, it is also very likely that overall energy gain (through efficient feeding) may also be lower in streams where hatchery fish are released.

In contrast to findings from 1993, when more agonistic contests in treatment streams resulted in the displacement of subordinate fish, more contests (69%) in control streams resulted in the displacement of subordinate fish than those in treatment streams (59%) in 1994. 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). Also in contrast to relationships seen in 1993, we documented possible displacements from a stream reach in Jungle Creek in 1994. There was a significant correlation between the numbers of trout-emigrating and the numbers of hatchery steelhead emigrating. This does not prove a cause and effect relationship; however, we can not rule out the possibility that the trout were pushed or pulled from Jungle Creek by the movement (or residualism) of large numbers of hatchery steelhead in 1994.

Rainbow trout population density in the treatment streams may have been negatively impacted by the releases of hatchery steelhead. Population abundance of rainbow trout in the North Fork of the Teanaway River (T_L) and in the Middle (C,) and West (R_L) forks of the Teanaway River showed a general downward trend from 1990 through 1992. The differences between linear densities of rainbow trout in control and treatment streams suggest that the release of hatchery steelhead into the North Fork of the Teanaway system reduced its capacity to rear wild rainbow trout. The final year of hatchery steelhead releases for this experiment occurred in 1994. If the decreased trout abundance in the North Fork of the Teanaway was due to the releases of hatchery steelhead, then, all else being equal, we might expect population densities of rainbow trout there to rebuild to levels similar to the control streams within 2 to 4 years. Additional information on these stream populations will allow testing of this hypothesis and will provide more statistical power to examine the effects of releases of hatchery-reared steelhead on wild rainbow trout populations.

Trout abundance in Jungle Creek (T,) was highly variable between 1991 and 1994, suggesting spawning success varied between years (most of the trout captured in T_S during the population estimates were age 0+ in all years). The index site in Jungle Creek was very close to the North Fork of the Teanaway River and adult trout may have moved into Jungle Creek to spawn. Consequently, our population estimates in Jungle Creek may simply provide a measure of wild trout recruitment or reproductive success, and/or early rearing survival in that area.

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 55 residual steelhead stomachs in 1992 nor in 31 stomachs examined in 1994. Martin et al. (1993) examined a total of 1,713 hatchery steelhead stomach samples in southeast Washington streams and found only three juvenile spring chinook salmon and 17 0. *mykiss* fry (S. Martin, pers. comm.). We suggest that predation by hatchery steelhead on wild trout fry was negligible in our treatment streams during the years of study.

Harvest rates on residual hatchery steelhead have been relatively high in the study area in previous years (McMichael et al. 1992). However, in 1994, anglers were not allowed to keep residual hatchery steelhead less than eight inches (203 mm) long due to a new angling regulation. The decreased harvest may have been partially responsible for the relatively high observation rates of residual steelhead throughout the study area in 1994.

Hatchery steelhead residuals moved downstream out of the release stream and migrated up the North Fork of the Teanaway River into areas containing bull and cutthroat trout populations. This illustrates the importance of monitoring outmigration of hatchery-reared smolts at multiple points along their route to the sea as well as in areas upstream of release sites. If we had only. operated the weir-trap at the mouth of Jungle Creek we may have mistakenly believed that all smolts that passed that trap were headed downstream. Nine sites were snorkeled during daylight and at night to better estimate bull trout presence in the salmonid species composition in 1994. No bull trout were seen during daylight hours, while eight were observed during night surveys of the same reaches.

Application of results from this study may not be directly transferable to interactions that may occur between hatchery fish and those produced by the YFP and wild fish. The hatchery-reared steelhead we released into Jungle Creek were produced at a traditionally operated hatchery facility. We realize that the hatchery fish we used may not behave identically to those produced in a more innovative facility.

It is important to note that these results are preliminary and subject to revision following additional data collection and analyses. Final results will be presented in a future report or publication.

Update 6:

Studies of hatchery and wild steelhead, rainbow trout, and chinook salmon paired in instream enclosures

Introduction

In an effort to better understand the impacts of releasing hatchery steelhead and chinook salmon smolts on preexisting rainbow trout and spring chinook salmon in the upper Yakima River basin, Washington, we conducted a multi-part series of experiments in small enclosures in two natural streams. Our intent was to be-able to provide information for resource managers regarding the potential for hatchery-origin smolts to impact the growth of wild salmonids in the release area. In addition, we wanted to determine how increased natural production of spring chinook salmon resulting from supplementation activities might affect the growth of wild rainbow trout. We examined the effects of hatchery-reared steelhead residuals on: wild rainbow trout (part 1), and wild spring chinook salmon (part 2). We also studied the effects of wild age 0+ spring chinook on wild rainbow:trout (part 3), and age 1+ hatchery-reared spring chinook salmon on wild age 0+ spring chinook salmon (part 4). Our methods were generally the same as those presented by McMichael et al. (1994) with a few exceptions. Where methods varied from the previous report, deviations will be explained. The Methods and Results sections of this report are divided into four parts corresponding to the species and groups of salmonids tested. The Discussion section will include all tests.

These experiments were not designed to determine which species were the most dominant given equal fish sizes, nor to determine dominance regardless of size. They were instead designed to determine if the presence of a treatment fish influenced the growth of the response fish. We designed the experiment in this manner in an attempt to determine what effect a doubling in the number of salmonids (due to direct hatchery releases or subsequent increased natural production) would have on the growth of preexisting wild salmonids. The experimental design required three fish to be placed in each enclosure. A solitary fish (control) was placed in one chamber and treatment and response fish were placed in the other. Treatment fish were residual hatchery steelhead for tests 1 and 2, wild age 0+ spring chinook salmon for test 3, and age 1+ hatchery spring chinook salmon for test 4. Control and response fish belonged to the same species/origin/age groups and were wild rainbow trout for tests 1 and 3, age 0+ wild spring chinook salmon for tests 2 and 4.

Methods

Part 1:

For the first part of our experiments, methods used in 1994 were generally the same as those used in 1993 (see McMichael et al. 1994). Briefly, enclosures were constructed with 5 cm x 5 cm wood frame members and were enclosed with 0.95 cm square galvanized wire mesh on all sides and the bottom. 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, separated parallel to stream flow by a vertically-oriented plywood barrier. Four large cobbles (20 to 30 cm diameter) were collected within the wetted stream channel and positioned randomly in each chamber of each enclosure to simulate natural conditions and to provide substrate for benthic organisms. A plywood lid was attached to cover each enclosure.

Sites and selection criteria used were the same as McMichael et al. (1994) used in 1993, with enclosures placed in pool and run habitats ranging in depth from 0.35 to 0.66 m, and velocities from 0.20 to 0.43 m/s.

The control and response fish were wild rainbow trout between 100 and 150 mm in fork length (FL), while the treatment fish were residual hatchery steelhead between 140 and 204 mm. The combinations used in this experimental design were intended to ascertain effects on response fish. As defined earlier, we use the terms control, response, and treatment fish to distinguish between the different groups of fish in each test. The terms control and unpaired are used interchangeably as are response and paired. This test was intended to be replicated 10 times in 1993 and 20 times in 1994.

Fish used in this study were obtained both from the North Fork of the Teanaway River and one of its tributaries. In both years all naturally-produced trout were collected from the North Fork of the Teanaway River. McMichael et al. (1994) described fish collection and handling in detail. The relative sizes of the groups of fish used in this experiment (Table 1) were intended to be similar to those typically found during the summer rearing period in streams in the upper Yakima River basin.

Table 1. Fork length (mm) and weight (g) of fish groups at the beginning of the growth experiments. Mean length and weight, standard deviation (SD), and range are shown for control and response (rainbow trout) and treatment (residual hatchery steelhead) groups in 1993 and 1994 tests.

	Control			Response			Treatment		
Year	Mean	SD	Range	Mean	SD	Range	Mean	SD	Range
				Length					
1993	114.4	14.6	101-143	117.0	13.5	102-140	169.4	25.1	140-204
1994	115.9	9.2	108-136	118.4	15.0	102-138	168.5	10.1	156-183
				Weight					
1993	18.0	8.5	11.5-36-O	18.9	6.0	12.8-29.6	51.0	23.2	26.9-88.7
1994	18.8	6.0	14.4-32.6	21.1	5.9	13.3-28.9	47.2	9.4	37.0-61.4

Control and response fish were placed in the enclosures on July 9, 1993, and on July 7 in 1994. A residual hatchery steelhead was then placed in one of the two chambers (assigned randomly) in each of the enclosures containing rainbow trout.

Enclosures in which one or more mortalities or escapes occurred prior to the end of the study period were discarded from the final analyses. This occurred in three of the replicates in 1993. Ten of the 20 replicates were not used in 1994 due to one or more fish missing at the termination of the experiment. Most missing fish were assumed to have died prior to the termination of the experiment; however, some may have escaped. Enclosures were cleaned of debris with a wire brush twice each week to facilitate passage of drifting invertebrates.

On August 19, 1993, 42 days after the control and response fish were placed in the enclosures, all fish were collected from the enclosures, euthanized in a lethal concentration (>200 mg/l) of MS-222, measured to the nearest mm FL , weighed to the nearest 0.1 g, and bled for physiological analyses [see McMichael et al. (1994) for details on physiology and stomach content sampling conducted in 1993]. In 1994, all fish were netted from the enclosures on August 17 (after 43 days in the enclosures), anesthetized in MS-222, measured to the nearest mm FL, weighed to the nearest 0.1 g. The fish were then placed back in the enclosures and were sampled again on October 5, 1994.

Part 2:

This test, conducted in 1993, was designed to determine the effects of residual hatchery steelhead on wild spring chinook salmon in the same area and in the same manner as described above for Part 1. One exception to this pertained to where the juvenile spring chinook salmon were collected. 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. Juvenile spring chinook salmon were collected in the river near the town of Cle Elum, Washington on July 7, 1993, 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 in a manner consistent with that previously described for rainbow trout. Relative sizes of these fishes are presented in Table 2.

Table 2. Fork length (mm) and weight (g) of fish groups at the beginning of the growth experiments. Mean length and weight, standard deviation (SD), and range are'shown for control and response (spring chinook salmon) and treatment (residual hatchery steelhead) groups in 1993.

	Control			Response			Treatment		
Year	Mean	SD	Range	Mean	SD	Range	Mean	SD	Range
				Length					
1993	76.1	10.6	64-92	70.1	3.9	64-76	155.9	38.4	117-213
				Weight					
1993	5.7	2.4	2.9-9.7	4.0	0.9	2.5-5.2	43.5	29.8	15.1-90.6

Part 3:

Part 3 involved experiments to determine the effects of increased natural production of spring chinook salmon on wild rainbow trout, and were conducted in the North Fork of the Teanaway River in 1993 and in the Middle Fork of the Teanaway River in 1994. The tests conducted during 1993 required placement of age 0+ spring chinook with age 1+ and 2+ rainbow trout, and the tests conducted during 1994 used age 0+ fish of both species. Methods used in 1993 were described by McMichael et al. (1994). In 1994, we used 20 smaller enclosures (61 cm x 61 cm x 61 cm) of the same design and mesh size as the enclosures used in 1993 and placed them in a section of the Middle Fork of the Teanaway River between 1.6 and 1.8 km upstream of its mouth. Smaller enclosures were used in this test due to the smaller size of the fish used in this experiment. The age 0+ rainbow trout were collected in Jungle Creek with a backpack electrofisher (200 V DC) and the age 0+ spring chinook salmon were collected with the same equipment and settings in the mainstem Teanaway River at approximately rkm 3.2. Rainbow trout and spring chinook salmon were placed in the enclosures on August 2, 1994. Fish were recovered from these enclosures on September 12, 1994. Table 3 shows the relative mean sizes of the fishes in these tests.

Table 3. Fork length (mm) and weight (g) of fish groups at the beginning of the growth experiments. Mean length and weight, standard deviation (SD), and range are shown for control and response (rainbow trout) and treatment (spring chinook salmon) groups in 1993 and 1994 tests.

	Control			Response			Treatment		
Year	Mean	SD	Range	Mean	SD	Range	Mean	SD	Range
				Length					
1993	122.8	12.1	108-145	124.8	15.1	106-149	67.6	4.6	61-77
1994	83.8	7.4	71-98	79.4	7.7	74-93	81.2	8.1	72-99
				Weight					
1993	20.7	6.8	14.6-34.1	23.5	8.5	14.2-37.5	4.1	0.9'	3.1-6.1
1994	6.0	1.7	3.2-9.6	5.1	1.8	3.5-8.2	5.7	1.5	3.7-8.6

Part 4:

In the final part of these studies, the impacts of hatchery-reared spring chinook salmon on wild spring chinook salmon were examined via experiments in the North Fork of the Teanaway River in 10 enclosures that were deployed in the same area as those discussed in Part 1. The wild spring chinook salmon for this test were collected in the Teanaway River with a backpack electrofisher (300 V DC) at rkm 2.0 on July 12, 1994. Hatchery-reared spring chinook were obtained on July 14, 1994, from the National Marine Fisheries Service in Seattle, Washington. These hatchery-reared fish were the first generation progeny of wild upper Yakima River spring chinook salmon adults collected in the upper Yakima basin. These fish were older (about 21 months) and larger than typical hatchery-reared spring chinook salmon smolts but were thought to represent residuals that might result from hatchery releases. Relative lengths and weights of test groups are shown in Table 4. These fish were weighed and measured on August 17, 1994 and placed back in the enclosures. On October 5, 1994, we re-captured the fish and anesthetized, weighed and measured them.

Table 4. Fork length (mm) and weight (g) of fish groups at the beginning of the growth experiments. Mean length and weight, standard deviation (SD), and range are shown for control and response (wild spring chinook salmon) and treatment (hatchery-reared spring chinook salmon) groups in 1994.

	Control			Response			Treatment		
Year	Mean	SD	Range	Mean	SD	Range	Mean	SD	Range
				Length					
1994	75.0	7.0	66-91	77.9	8.8	64-94	180.0	19.4	142-209
				Weight					
1994	5.0	1.7	3.1-8.9	6.2	2.2	3.3-11.0	84.7	32.0	35-144

Data Analyses

To test whether the presence of treatment fish negatively affected the growth of response fish in all four tests, one-tailed paired t-tests were performed on specific growth rate (SGR) differences between respective pairs of control and response fishes. In 1993, we examined both length and weight as response variables (McMichael et al. 1994), but this year we examined the data using only the SGR measure. Specific growth rate is superior to our previously used growth measures because it standardizes the data-for variation in fish size and trials of different durations (Fausch 1984). Specific growth rate was calculated using the following equation:

$$SGR = \frac{\ln W_t - \ln W_o}{t}$$

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

Statistical power analyses (Snedecor and Cochran 1981; Peterman 1990) for t-tests involving control and response fish growth were performed to aid in the interpretation of the results of all four tests.

Results

Part 1:

We found the presence of hatchery steelhead negatively impacted growth of naturally-produced rainbow trout in experiments in both 1993 and 1994. In 1993, the mean SGR of unpaired rainbow trout (controls) in test 1 was higher than the SGR of trout paired with hatchery steelhead (even though both groups had negative SGRs)(Table 5). The difference in SGRs between control and response rainbow trout in test 1 was statistically significant (Table 6). The SGR of treatment fish in test 1 also decreased during trials in 1993 (Table 5). In 1994, SGRs of control rainbow trout were relatively higher than SGRs of response rainbow trout that had been paired with hatchery steelhead for the period between July 5 to August 17, 1994 (Table 5). Specific growth rates of control rainbow trout were significantly higher than SGRs for response rainbow trout for the later period in 1994 (Table 6). Mean specific growth rates were also negative for control, response, and treatment fish (Table 5).

Table 5. Mean specific growth rates (SGR) of control © and response (R) rainbow trout (RBT) and residual hatchery-reared steelhead (HSH) treatment (T) fish in growth experiments in 1993 and 1994. Standard deviations (SD) are also presented.

Year	Species	C/R/T	SGR	SD	N	Days	Dates
1993	RBT	C	-0.0016	0.0029	7	42	7/7 to 8/19
1993	RBT	R	-0.0060	0.0020	7	42	7/7 to 8/19
1993	HSH	T	-0.0020	0.0015	7	40	7/9 to 8/19
1994	RBT	C	-0.0039	0.0020	10	43	7/5 to 8/17
1994	RBT	R	-0.0061	0.0021	10	43	7/5 to 8/17
1994	HSH	T	-0.0023	0.0021	10	41	7/7 to 8/17
1994	RBT	C	-0.0001	0.0005	8	49	8/17 to 10/5
1994	RBT	R	-0.0011	0.0012	8	49	8/17 to 10/5
1994	HSH	T	-0.0006	0.0007	8	49	8/17 to 10/5

Table 6. Results of paired t-tests comparing specific growth rates of control and response rainbow trout in growth experiments using hatchery steelhead as treatment fish. Degrees of freedom (df), t statistics (t), probability values (P), and power are shown for differences in specific growth rates. Asterisks denote significant differences ($P<0.05$).

Year	Dates	df	t	P	power
1993	7/7 to 8/19	6	2.66	0.019*	0.838
1994	7/5 to 8/17	9	3.70	0.002*	0.981
1994	8/17 to 10/5	7	2.37	0.025*	0.764

Hatchery steelhead in this test exhibited negative SGRs similar to those observed in the test conducted in 1993 (Table 5). Similar to the first half of the summer, during the late summer/early fall period (August 17 to October 5, 1994) the wild rainbow trout that were paired with the residual hatchery steelhead had significantly lower SGRs than the control rainbow trout (Tables 5 and 6).

Part 2:

Wild spring chinook salmon paired with hatchery steelhead did not exhibit significantly different SGRs than their unpaired counterparts (df = 6, t = 0.09, P = 0.470, Table 7). Specific growth rates of spring chinook in both control and response groups decreased by nearly equal amounts. Hatchery steelhead treatment fish in this test showed average decreases in SGR that were similar to those exhibited by treatment hatchery steelhead in Part 1 (Tables 5 and 7). The statistical power for this test was low (0.058).

Table 7. Mean specific growth rates (SGR) of control © and response (R) wild spring chinook salmon (SPC) and residual hatchery-reared steelhead (HSH) treatment (T) fish in growth experiments in 1993. Standard deviations (SD) are also presented.

Year	Species	C/R/T	SGR	SD	N	Days	Dates
1993	SPC	C	-0.0004	0.0040	7	42	7/7 to 8/19
1993	SPC	R	-0.0006	0.0048	7	42	7/7 to 8/19
1993	HSH	T	-0.0018	0.0013	7	40	7/9 to 8/19

Part 3:

Wild rainbow trout (age 1+ and 2+) paired with wild spring chinook salmon did not grow at different rates than rainbow trout that were not paired (Tables 8 and 9) in 1993. Similarly, in our 1994 test which examined effects of age 0+ spring chinook on age 0+ rainbow trout, we found no significant difference between the SGR of trout with and without spring chinook salmon (Tables 8 and 9).

Table 8. Mean specific growth rates (SGR) of control (C) and response (R) wild rainbow trout and wild spring chinook salmon (SPC) treatment (T) fish in growth experiments in 1993 and 1994. Standard deviations (SD) are also presented.

Year	Species	C/R/T	SGR	SD	N	Days	Dates
1993	RBT	C	-0.0019	0.0027	9	42	7/7 to 8/19
1993	RBT	R	-0.0023	0.0018	9	42	7/7 to 8/19
1993	SPC	T	-0.0012	2.7380	9	40	7/9 to 8/19
1994	RBT	C	-0.0017	0.0041	11	41	8/2 to 9/12
1994	RBT	R	-0.0000[a]	0.0028	11	41	8/2 to 9/12
1994	SPC	T	-0.0046	0.0019	11	41	8/2 to 9/12

[a] actual value = -0.000029

Table 9. Results of paired t-tests comparing specific growth rates of control and response wild rainbow trout in growth experiments using wild spring chinook salmon as treatment fish. Degrees of freedom (df), t statistics (t), probability values (P), and power are shown for differences in specific growth rates.

Year	Dates	df	t	P	power
1993	7/7 to 8/19	8	0.37	0.36	0.109
1994	8/2 to 9/12	10	-0.91	0.19	0.232

Part 4:

Wild age 0+ wild spring chinook salmon paired with hatchery-reared age 1+ spring chinook salmon had significantly lower specific growth rates than unpaired wild chinook between July 12 and August 17, 1994 (Tables 10 and 11). However, there was no significant difference in SGRs of control and response wild spring chinook salmon in this test from August 17 to October 5, 1994 (Tables 10 and 11).

Table 10. Mean specific growth rates (SGR) of control © and response (R) wild spring chinook salmon (SPC) and hatchery-reared age 1+ spring chinook salmon (HSPC) treatment (T) fish in growth experiments in 1994. Standard deviations (SD) are also presented.

Year	Species	C/R/T	SGR	SD	N	Days	Dates
1994	SPC	C	0.0022	0.0016	10	36	7/12 to 8/17
1994	SPC	R	-0.0014	0.0028	10	36	7/12 to 8/17
1994	HSPC	T	-0.0045	0.0018	10	34	7/14 to 8/17
1994	SPC	C	0.0005	0.0031	8	49	8/17 to 10/5
1994	SPC	R	0.0020	0.0035	8	49	8/17 to 10/5
1994	HSPC	T	-0.0017	0.0007	8	49	8/17 to 10/5

Table 11. Results of paired t-tests comparing specific growth rates of control and response wild age 0+ spring chinook salmon in growth experiments using hatchery-reared age 1+ spring chinook salmon as treatment fish. Degrees of freedom (df), t statistics (t), probability values (P), and power are shown for differences in specific growth rates. Asterisk denotes significant difference (P<0.05).

Year	Dates	df	t	P	power
1994	7/12 to 8/17	9	3.03	0.007*	0.912
1994	8/17 to 10/5	7	-0.72	0.754	0.184

Discussion

This is a summary discussion, integrating results across all four parts of this study. Preliminary conclusions drawn by McMichael et al. (1994) were supported by interpretation of data collected in 1994. In 1994, trials involving residual hatchery steelhead and wild resident trout or steelhead presmolts, we again found that hatchery-reared steelhead residuals adversely affected growth of naturally-produced O. *mykiss* during the summer rearing period.

No new trials were conducted in 1994 in which hatchery steelhead residuals were placed with wild spring chinook salmon. However, new analyses of data from 1993 confirmed the interpretation of McMichael et al. (1994).

Results from the trials we conducted in 1993 that examined the effects of wild age 0+ spring chinook salmon on wild age 1+ rainbow trout suggested that spring chinook salmon did not negatively affect trout growth (McMichael et al. 1994). The rainbow trout used in 1993 experiments were larger than the spring chinook salmon and would be expected to be behaviorally dominant because of their larger size. It was our expectation that fishes of similar size would compete most intensely; therefore, trials conducted in 1994 were between age 0+ fish of both species (similar sizes). However, in 1994 we again found no negative effect on growth of rainbow trout due to the presence of wild chinook salmon. In 1994, trials were conducted in the Middle Fork of the Teanaway River as opposed to the North Fork of the Teanaway River because age 0+ fish of these two species often rear together during the summer (McMichael et al. 1992; Pearsons et al. 1993; McMichael et al. 1994).

Trials designed to examine the effects of hatchery-reared spring chinook salmon on wild spring chinook salmon in 1994 used age classes that would rarely be expected to occur together in natural streams. The hatchery fish were age 1+ and would best represent residual spring chinook salmon. Residual spring chinook salmon have been found to occur in hatchery and wild populations but their occurrence appears to be very rare (Mullan et al. 1992; Schreck et al. 1994; Chapman et al. 1995). Nevertheless, when age 1+ hatchery reared spring chinook salmon were paired with naturally produced age 0+ fish of the same species, the growth of the smaller naturally produced fish was significantly reduced. Again, these two groups of fish were the same species and would thus be expected to compete with greater intensity than fish of different species (Allee 1982; Kennedy and Strange 1986).

The reversal in spring chinook salmon growth trends between the early and late sample periods in 1994 might be explained by differences in environmental conditions between the early and late periods. Water temperatures during the early period may have reached high enough levels that were sufficient to limit growth when two fish were present in one chamber, while the growth of the solitary control fish was not influenced. During the later period water temperatures were lower which may have afforded conditions wherein impacts on response fish were lessened. Condition factors of hatchery spring chinook steadily decreased during these tests while condition factors of response and control fish generally decreased from the date the experiment began to the mid-point (8/17/94) and then increased until the end of the experiment (10/5/94) (McMichael, unpublished data). This supports the hypothesis that: 1) conditions for growth improved as water temperatures decreased during the latter half of these trials, and/or 2) that continual deterioration in hatchery spring chinook body condition throughout the period may have provided some competitive release (e.g., a reduced level of dominance in the treatment fish) for the response fish, thereby enabling them to grow more during this later interval.

We anticipate that competitive impacts on pre-existing salmonids due to direct releases of hatchery spring chinook smolts will be minimal unless the hatchery fish residualize to the extent that densities of hatchery and wild fish near equivalent levels. If fish released are true smolts which readily outmigrate from the basin, the spatial and temporal overlap with pre-existing fishes will be minimal. The greatest potential for releases of hatchery reared spring chinook salmon to impact naturally produced chinook salmon would be following the successful return and reproduction of hatchery origin fish. Progeny of hatchery origin adults may compete with naturally produced wild salmon if resources are limited. The most appropriate life stage to examine the impacts of hatchery-wild interactions between spring chinook salmon and other wild salmonids would be at the age 0+ or presmolt stages. This presmolt rearing phase is when growth-impacts due to competition would be expected to be greatest.

Differences in growth between control and response fish were largest in tests in which the treatment fish were considerably larger than the response fish, were the same species as the response fish, and were reared in a hatchery environment. This is consistent with findings reported by McMichael et al. (1994) and other 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).

Mean SGR for most groups of fish in our experiments was negative. Some individuals did gain weight, but most lost weight during the study periods. Growth impacts were then necessarily determined by comparison of negative specific growth rates. Weight loss in stream salmonids has also been reported from other studies that examined fish in enclosures. For example, Miller (1952) found that hatchery cutthroat trout lost weight during the first 40 days after being placed in enclosed stream sections with naturally-produced trout. In addition, Fausch (1984) reported negative SGRs for individual brown trout and brook trout in competition experiments with coho salmon in an artificial stream.

Enclosing fish in the manner we did may have influenced our results by affecting fish behavior and movement. Obviously, the physical confinement of fish inside the enclosures limited the range of possible movements by test fish. However, it is not clear to what extent this confinement may have inhibited movement patterns that might otherwise have been naturally expressed. Implications of enclosure effects were discussed by McMichael et al. (1994).

Impacts on pre-existing fish populations would be expected to be minimized when spatial and temporal overlap with released hatchery fish is also minimized (e.g., Viola and Schuck 1995; McMichael et al. In Press). For instance, in areas or times when large numbers of hatchery steelhead smolts are released and high rates of residualism occur, the impacts of these residual fish on pre-existing salmonids could be acute. Released hatchery-origin anadromous salmonids are typically larger and will occupy similar habitats as their wild conspecifics, increasing the likelihood of competitive impacts. The Yakima Fisheries Project proposes to volitionally release hatchery spring chinook salmon from acclimation ponds located in three areas of the upper Yakima basin. These fish are expected to expeditiously leave the ponds and emigrate seaward quickly, thereby minimizing spatial and temporal overlap with wild fishes in the Yakima basin. If this expectation is correct, then the short-term direct impacts on growth (due to competitive interactions) of preexisting salmonids' would be expected to be minor.

Update 7:

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

Introduction

This study was conducted to investigate post-release performance of hatchery steelhead in relation to various physical parameters. Data were collected as a byproduct of activities associated with experimental steelhead smolt releases that were conducted in the Teanaway River basin from 1991 through 1994 (see Update 5). The specific objectives of this analysis were to examine the effects of parentage, rearing density, and size at release on post-release in-stream performance of juvenile hatchery steelhead. Performance, as we define it, is a combination of survival, outmigration tendencies, and residualism. For example, a group of hatchery steelhead smolts exhibiting high survival as they migrate seaward, a strong tendency to migrate seaward, and a low propensity for remaining in freshwater as residuals, would be considered to perform better than a group that shows poor survival, little seaward movement, and a high degree of residualism. Similar to Wagner (1968), we defined hatchery steelhead that did not exhibit a seaward migration prior to June 1 to be residuals.

Methods

In general, study methods consisted of compiling available information from adult steelhead from the Yakima River for use as broodstock at the Yakima Hatchery; monitoring the offspring of those fish (juvenile hatchery steelhead) in the hatchery (Yakima Hatchery and Nelson Springs raceway) prior to release; releasing the juvenile steelhead into a test stream (Jungle Creek); and evaluating various performance measures of the juvenile steelhead in the field using trapping, snorkeling, and electrofishing techniques (McMichael et al. 1992). See McMichael (1994) for detail on specific methods.

Results

The numbers and origin (hatchery or naturally-produced) of summer steelhead adults used to produce the smolts for 1991, 1992, 1993, and 1994 releases varied greatly between years (Table 1).

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, 1993, and 1994.

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

The loading density of the hatchery steelhead smolts decreased each year between 1991 and 1993 and was-only slightly higher in 1994 than in the previous two years (Table 1).

Total number released, sizes, and degree of smoltification varied among the four years. The mean size at release of the hatchery steelhead decreased each year while the percentage classified as smolts increased through- 1993 and then was lower in 1994 (Table 2). In addition, the percentage of precocial males was highest in 1991 and decreased each year thereafter (Table 2). The mean condition factors of smolts released during the last three years were slightly over 1.00, whereas mean condition factor in 1991 was less than 1.00.

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, 1993, and 1994 (see Update 5). We know from the data collected at the Jungle Creek weir trap that only 22,499 hatchery steelhead emigrated from Jungle Creek into the North Fork of the Teanaway River before July 11, 1994.

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 1994. Sample sizes are also presented.

Release Year	Number Released	Number Sampled	Mean (± SD) Length	Mean (± SD) 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
1994	32,579	150	179 (± 21)	61 (± 24)	1.03	74 to 86[b]	0.0

[a] Smolt quality was not quantitatively assessed (it was estimated post-hoc based on the memory of staff present at release times), 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 from the three different release dates within years.

The Yakima Indian Nation (YIN) enumerated steelhead smolts that outmigrated past Prosser Dam. The percentages of hatchery steelhead that were released into Jungle Creek and later passed Prosser Dam were uniformly low in 1991, 1992, and 1994 but was relatively high in 1993 (Table 3). The performance of hatchery steelhead smolts that were offspring of solely hatchery-origin parents (release year 1993) was an order of magnitude better than smolts produced by wild or a mixture of hatchery and wild broodstock. It is possible that some of the smolts passing Prosser Dam after 1991 may have been from releases in previous years. For example, hatchery steelhead passing Prosser Dam in 1993 could have been released in 1991, 1992, and/or 1993. After 1991, no hatchery steelhead were released in the Yakima River basin besides those used for research. Thus, age 1+ hatchery steelhead detected at Prosser Dam in 1992, 1993, and 1994 were from our releases.

Table 3. Juvenile hatchery steelhead estimated to have passed Prosser Dam during May, June, and July in 1991, 1992, 1993, and 1994. The percentages represent portion of fish released into Jungle Creek that were estimated to have passed Prosser Dam within three months of their release (M. Kohn and M. Johnston, YIN, pers. comm.).

Release Year	Est. Total number at Prosser	Percentage of number released obs. at Prosser
1991	648	1.9
1992	575	1.5
1993	5,592	24.9
1994	837	2.6

Large percentages of hatchery steelhead smolts released into Jungle Creek did not emigrate out of the North Fork of the Teanaway River prior to June 1 of each year. Residual hatchery steelhead (those observed between June and October) were encountered with higher frequency in 1991, 1992, and 1994 than in 1993 (see Update 5). These data are corroborated by emigration estimates, where the lowest percentages of emigrating hatchery steelhead were observed in 1991 and 1994. The distribution of residual hatchery steelhead was documented in 1994 (see Update 5) and revealed that many residuals migrated upstream in the North Fork of the Teanaway River during the summer following release.

Discussion

Similar to findings presented by McMichael (1994), data from 1994 further support the conclusion that 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, hatchery fish). All hatchery steelhead we released were released as age 1+ smolts. 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). In the present study, offspring of adult steelhead that had been through one generation in a hatchery (release year 1993) outperformed smolts that resulted from naturally-produced adults (release year 1991) and progeny of a mixture of hatchery-origin and naturally-produced adults (release years 1992 and 1994).

Contrary to findings presented in last year's report (McMichael 1994) in which smolt size appeared to be inversely related to post-release performance for 1991 through 1993, this tendancy was not seen in 1994 when the smolts with the shortest mean length were released. The smolts released in 1994 had the shortest mean length of the four years of study yet performed relatively poorly. Smolts between 180 and 200 mm performed best during the four years of study. Smolts released at sizes over 200 mm and under 180 mm tended to show poorer overall performance.

The fish with the higher condition factors (between 1991 and 1993) appeared to perform better than those with lower condition factors. The 1994 release group, however, had the highest condition factor and performed poorly. It is possible that the negative relationship between size and condition factor may mask actual effects associated with condition factor on performance. This is particularly probable given the small differences in condition factor between years. However, Martin et al. (1993) also observed a greater incidence of residualism in groups of hatchery steelhead having lower condition factors.

Residual hatchery steelhead were relatively abundant for a prolonged period in 1994. Harvest rates on residual hatchery steelhead have been relatively high in the study area in previous years (McMichael et al. 1992). Anglers were not allowed to keep residual hatchery steelhead less than eight inches (203 mm) long in 1994 due to a new angling regulation. The decreased harvest may have been partially responsible for the relatively high observation rates of residual steelhead throughout the study area in 1994.

Because many variables in the current study changed annually (e.g. parentage and rearing density), the results regarding the effects of size at release and condition factor on post-release performance may have been due to chance. In the present study, hatchery steelhead reared at the highest density (release year 1991) 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). It is worth noting that the rearing densities used in the production of the smolts in for the 1991 release were considered standard by the Washington Department of Wildlife. Even though only four 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 instream survival.

Based on these preliminary findings and within the constraints of a limited experimental design, it appears that the one variable we examined that consistently influenced instream smolt performance was parentage. Smolts produced by exclusively hatchery-origin parents performed better than those produced by exclusively wild parents and those produced by combinations of hatchery-origin and wild parents. Other factors that appeared to affect in-steam performance were rearing density (three of four years) and size at release (three of four years).

Environmental conditions may have influenced post-release performance of the hatchery steelhead smolts released in this study. This report does not consider the possible relationships between abiotic factors and performance. However, to the extent that environmental data are available, these examinations will be included in a later report.

General Discussion

Except for preliminary conclusions presented in Chapters 1 and 2, which represents work not previously addressed in our progress reports, most of the major preliminary conclusions presented in this report were consistent with those reported by Pearsons et al. (1994). Of particular importance to the Yakima Fisheries Project is our finding that spring chinook salmon immigrate up into the lower portion of tributaries shortly after emergence. If the number of spring chinook increases following supplementation, their use of tributaries may increase. This will complicate monitoring juvenile spring chinook salmon density and distribution and increase the area in which spring chinook salmon may interact with trout species. Final conclusions from topics presented in this report will be published in journal publications and/or future progress or status reports.

It is important to understand the extent to which the techniques we used to sample fishes might have adverse consequences on those fishes. Obviously, an adverse impact would not only affect important resources, but would also influence our ability to monitor the effects of supplementation. Of most concern is the effect that electrofishing may have on fish populations. Electrofishing is one of the most common techniques that we have used to collect data on fishes (Updates 2, 3, 4). Many studies have indicated that electrofishing may injure fishes (Dwyer et al. 1993; McMichael 1993; Mitton and McDonald 1994) but little information is available about how it might affect a population of fish (Schill and Beland 1995). Because the utility of electrofishing is high and the potential ecological risks of its use are also high, we recommend that the methods used to capture fish be evaluated with respect to fish populations in the upper Yakima basin.

The chapters and updates presented in this and our previous progress reports provide the requisite information for the development of an ecological risk assessment and risk containment plan associated with supplementing steelhead trout and spring chinook salmon in the upper Yakima basin. The ecological risk approach we propose includes five steps. First, management objectives for non-target species would be determined. Second, ecological risks would be assessed with respect to non-target species objectives. Third, protocols of hatchery operations would be reviewed and refined (if necessary) to minimize undesirable interactions. Fourth, a monitoring plan would be developed and implemented to measure the extent to which non-target species objectives are achieved. Finally, alterations to existing operations would be recommended if deviations from non-target species objectives were detected. Because ecological assessment of risks and monitoring plan development are dependant upon non-target species objectives, assessment of ecological risks will be deferred until objectives for non-target species are delineated. Non-target species objectives will also be reported in a future document.

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Appendix A

Temporal and spatial variation in the condition of hook-scarred rainbow trout in the Yakima River

Abstract

Catch and release fishing regulations were implemented on the Yakima River in 1990 with expectations of increasing rainbow trout size and density. During the five years after regulation change, fish size and density remained stable while angling pressure appeared to increase. We examined the length to weight relationship between hook-scarred and non hook-scarred rainbow trout to explore potential associations between angling and the condition of rainbow trout. Furthermore we examined this relationship in five sections of the Yakima River from 1990 to 1994. No statistical differences were detected between the length-to-weight relationship of rainbow trout with and without hook-scars. However, there was a significant spatial relationship between rainbow trout condition and study section. Although the relationship of length to weight varied between sections, the proportion of hook-scarred fish appeared to be greatest in the lowest elevation sections below the input of agricultural run off. Also, the proportion of hook-scarred fish appeared to be increasing in the lowest elevation section which received the heaviest angling pressure. Incidence of hook-scar has the potential to be a valuable tool for indirectly monitoring fishing pressure in catch and release fisheries.

Introduction

Restrictive fishery regulations, such as catch and release, are frequently instituted to help maintain or maximize size and abundance of fish in areas with production limitations, heavy angler pressure, or having quality management objectives. Ever since the first 'fishing for fun' regulations were implemented on the Bradley Fork and West Prong of the Little Pigeon River in the Great Smoky Mountain National Park in 1954 (Thompson 1958, Barnhart 1989), no kill trout fisheries have grown in popularity. Catch and release regulations function to "recycle" fish to help provide a quality fishing experience for many anglers. Schill et al. (1986) found that Yellowstone cutthroat trout *(Oncorhynchus clarki bouvieri)* in a heavily fished section of the Yellowstone River in Yellowstone National Park, were caught and released an average of 9.7 times during a 3.5 month season in 1981.

Following implementation of catch and release regulations for resident trout, fish size and abundance generally increase in lakes and streams (Vincent 1984) and have therefore been embraced by many fishery managers as well as angler groups (Anderson and Nehring 1984). The size and number of rainbow trout in the Yakima River did not increase following a change from regulations having a minimum size and bag limit to catch and release. Fishing regulations on the Yakima River changed in 1990 from a bag and size limit of two fish over fifteen inches with a season opening in June and closing in October, to a catch and release fishery open year around. Population estimates conducted from 1990 to 1994 suggest that rainbow trout abundance and size structure have remained relatively stable (Martin et al. 1994). We hypothesized that hook-scarred fish had lower foraging success due to physical trauma and stress than non hook-scarred fish, subsequently affecting fish size and abundance in the Yakima River.

Our primary objective was to compare the length to weight relationships between hook-scarred and non hook-scarred rainbow trout (0. *mykiss*) from 1990 to 1994 in five sections of the Yakima River. Our secondary objectives were to determine what percentage of the rainbow trout population in five survey sections of the Yakima River had hook-scars, and whether that percentage changed over time.

Study Area

Our study area included the 94.6 stream km of the Yakima River from Roza Dam (irrigation diversion dam) upstream to the confluence of the Cle Elum River (Bartrand et al. 1994). The study area is divided into five sampling sections (Figure 1). The Yakima River is dammed approximately 18.2 km above our study area at Easton Dam where water flow is regulated for agricultural irrigation. Mean daily flows in the Yakima River vary annually from about 14 to 142 m^3/s.

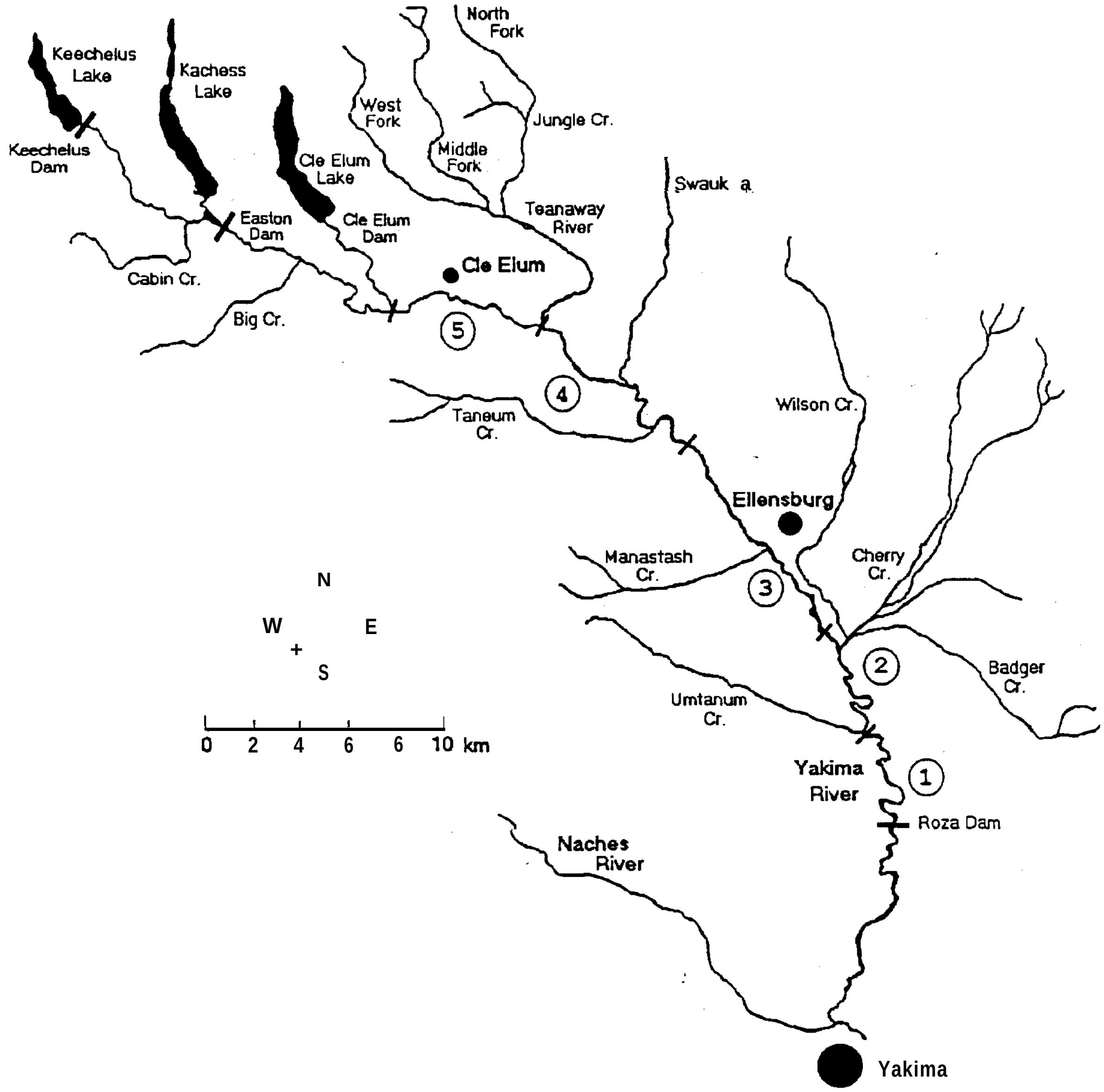

Figure 1. Map of the Yakima River drainage north of the City of Yakima, Washington. Study area includes sections 1 through 5 of the mainstem Yakima River above Roza Dam. Sections are separated by heavy lines on the figure and are labeled with circles.

Currently, regulations in this area of the Yakima River allow for year around trout fishing but limit anglers to catch and release fishing with artificial flies and lures having only single **barbless** hooks. Regulations were changed in 1990 from a more traditional season opening in June and closing at the end of October. Prior to 1990, anglers could fish using artificial flies and lures with any hook configuration and were restricted to a selective daily bag limit (two fish over 15 inches (381 mm)).

Study sections 1 and 2 are paralleled on one side by a highway (State Route 821) where the river flows through a scenic canyon. Wilson Creek, which drains much of the agricultural land in the Kittitas Valley, flows into the Yakima River near the upper end of section 2. An influx of turbid'water from Wilson Creek is especially pronounced during the irrigation season which increases the turbidity in the Yakima River below that point. Sections 1 and 2 have very few side channels and only a small amount of in-stream large woody debris (LWD). Section 3 has many side channels and appears to have a higher proportion of LWD than any other section. Section 4 has low habitat complexity with few side channels, and the lowest density of trout in the study area. Section 5, the highest elevation area, contains several side channels and some LWD. During field activities we observed that fishing pressure increased as elevation decreased, with the highest pressure occurring in sections 1 and 2. In addition, angling pressure appeared to increase in all sections from 1990 to 1994.

Methods

The proportion of rainbow trout with hook-scars was determined by examining fish collected during the process of conducting annual population estimates. Sampling was conducted to estimate population size in five sections of the **mainstem** Yakima River during the fall from 1990 through 1994 (Martin et. al. 1995). Sampling was conducted by drift boat electrofishing at night to maximize capture efficiencies (Leob 1957). Fish were anesthetized using Tricaine Methane Sulfonate **(MS-222),** measured to the nearest millimeter (fork length), and weighed to the nearest gram. Fish were examined for signs of hook-scars by carefully inspecting the mouth externally and internally. We defined a hook scar as: any blemish, disfigurement or other sign of trauma to any part of the mouth of a fish (e.g., torn membrane, missing or damaged mandible or maxillary) which may have been caused by angling.

Fish were separated into four age groups; 0+, **1+, 2+** and **>3** based on previously collected, back calculated length-at-age data to facilitate comparison of fork length distribution of **hook-**scarred and non hook-scarred fish (Table 1).

Table 1. Categories used for ageing fish based on back-calculated length at age data for each section of the Yakima River (Martin et al. 1994).

Section	Length at Age (mm)			
	0+	1+	2+	3
LCYN (1)	< 96	96-218	219-296	> 296
UCYN (2)	< 94	94-220	221-308	> 308
EBURG (3)	< 84	84-199	200-305	> 305
THORP (4)	< 79	79-179	180-262	> 262
CELUM (5)	< 77	77-176	177-248	> 248

Age 0+ fish were excluded from the data set due to small sample sizes in both hook-scarred and non hook-scarred categories. In the final analyses, the age 1+ category was limited to fish 160 mm or larger due to a lack of hook-scarred fish 159 mm or less. The age 3+ group consolidated age 3 through age 5 fish due to overlap in back calculated length at age data (Martin et. al. 1994).

Rainbow trout with injuries of unknown origin, such as lacerations or blindness, were excluded from the data set. Due to spatial and temporal variations in environmental conditions, analysis of length to weight relationships between hook-scarred and non hook-scarred fish were assessed with sections and years as factors. An analysis of covariance (ANCOVA) was performed using PCSAS (SAS Institute 1993) for each of the three age groups. We used a natural log (*ln*) transformation to normalize the data. We tested for differences in section, year, and hook-scar in the relationships of In weight to In fork length. In the ANCOVA we used *ln* of fork length as the covariate, *In* of weight as the dependent variable, and sections, years, and scar presence, as the treatments.

Results

No significant differences were detected ($P > 0.05$) in length weight relationship between hook-scarred and non hook-scarred fish regardless of age or section of river sampled (Table 2). Length **weight relationship was significantly different among** sections, $P = 0.0002$, 0.0001, 0.0001, for age 1+, 2+, and >3 respectively (Table 2), but not among years, $P = 0.1546$, 0.2108, 0.0965, for age 1+, 2+, and >3, respectively (Table 2).

Table 2. Summary of the Analysis of Covariance (ANCOVA) results for each age group of rainbow trout collected from the upper Yakima River 1990 through 1994. The ANCOVA tested for differences in the relationship of fork length (*ln*Len) to weight for each section of river (Sect), year of sample (Year), and presence of scar (Scar).

	Age 1+		Age 2+		Age >3	
Source of effect	df	P	df	P	df	P
Sect	4	0.0002	4	0.0001	4	0.0001
Year	4	0.1546	4	0.2108	4	0.0965
Scar	1	0.6758	1	0.7917	1	0.6649
*ln*Len	1	0.0001	1	0.0001	1	0.0001
*ln*Len x Sect	4	0.0005	4	0.0001	4	0.0001
*ln*Len x Year	4	0.1361	4	0.1823	4	0.0984
*ln*Len x Scar	1	0.7083	1	0.7757	1	0.6136
*ln*Len x Sect x Scar	4	0.6860	4	0.1018	4	0.2711
*ln*Len x Sect x Year	16	0.0800	16	0.0001	16	0.0001
*ln*Len x Year x Scar	4	0.5664	4	0.9162	4	0.4868
*ln*Len x Sect x Year x Scar	12	0.7886	16	0.1859	16	0.7853

Comparison of length to weight regression slopes for three age groups and five sections of river indicated that fish grew fastest in sections 1 and 2 (Figure 2) except for shorter fish collected in section 1.

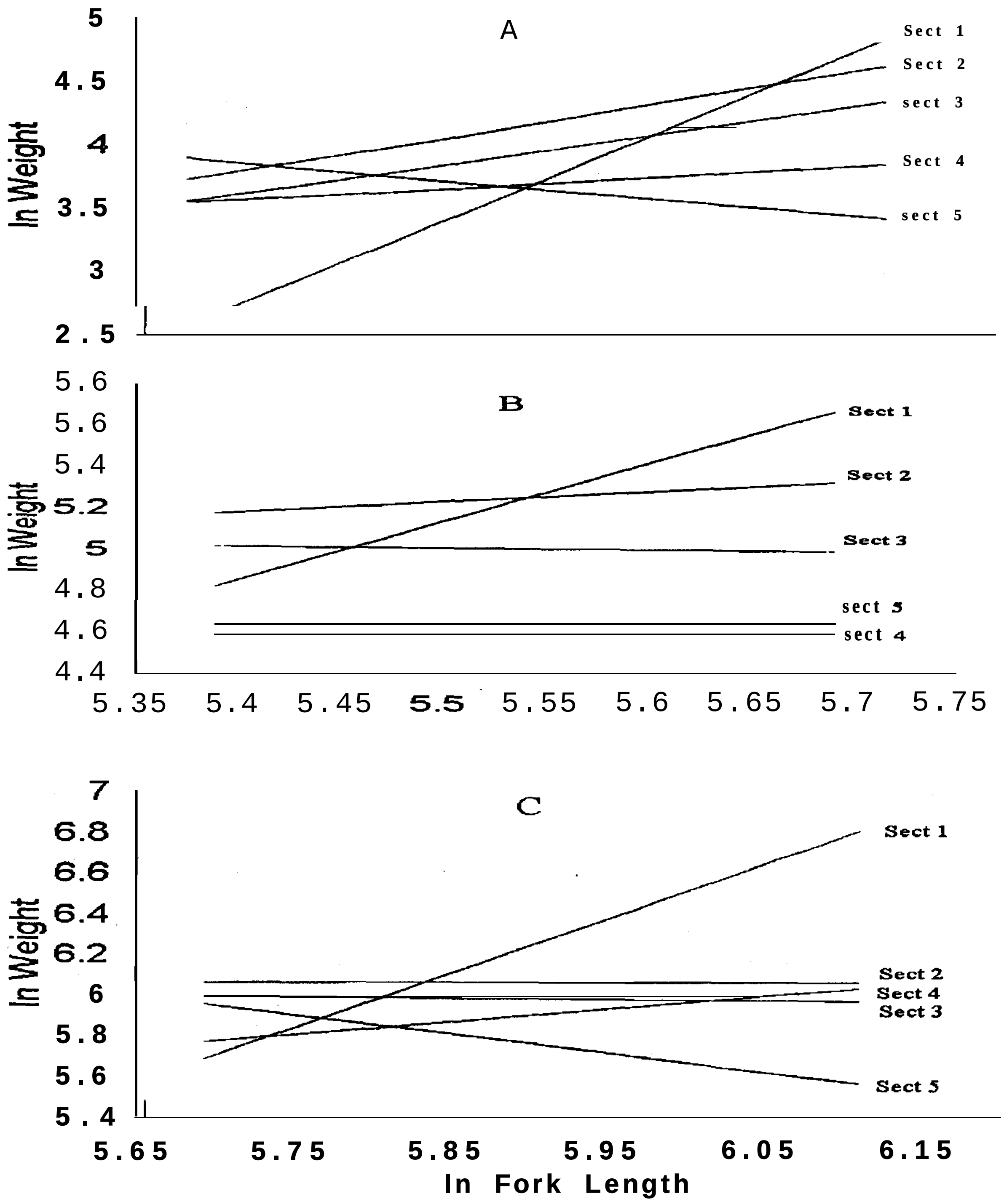

Figure 2. Regression analysis of ln fork length to ln weight for five sections of the Yakima River for A. age 1+, B. age 2+, C age >3.

The statistical interaction of In of fork length with section of river was also significant, P = 0.0005, 0.0001, 0.0001, for age 1+, 2+, and >3, respectively (Table 2). Analysis of interactions beyond this level are not statistically relevant due to significance of the interaction between In of fork length and section.

The percentage of hook-scarred fish varied between sections from a low of 7% in section 5 in 1994 to a high of 36% hook-scarred fish in section 1 in 1994 (Figure 3). Sections 1 and 2 consistently had the highest percentages of hook-scarred fish, whereas sections 4 and 5 generally had the lowest.

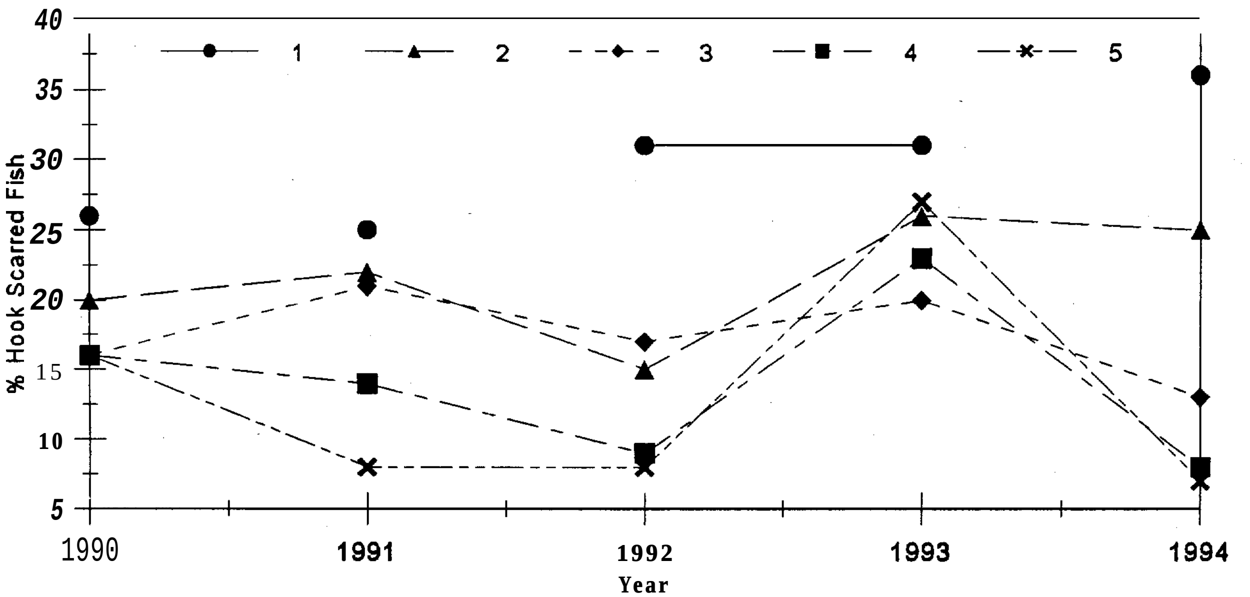

Figure 3. Percentage of hook-scarred rainbow trout collected from 1990 through 1994 in five sections of the upper Yakima River.

In all sections pooled, the proportion of fish with hook scars increased as fish length or age increased (Figure 4). Age 1+ rainbow

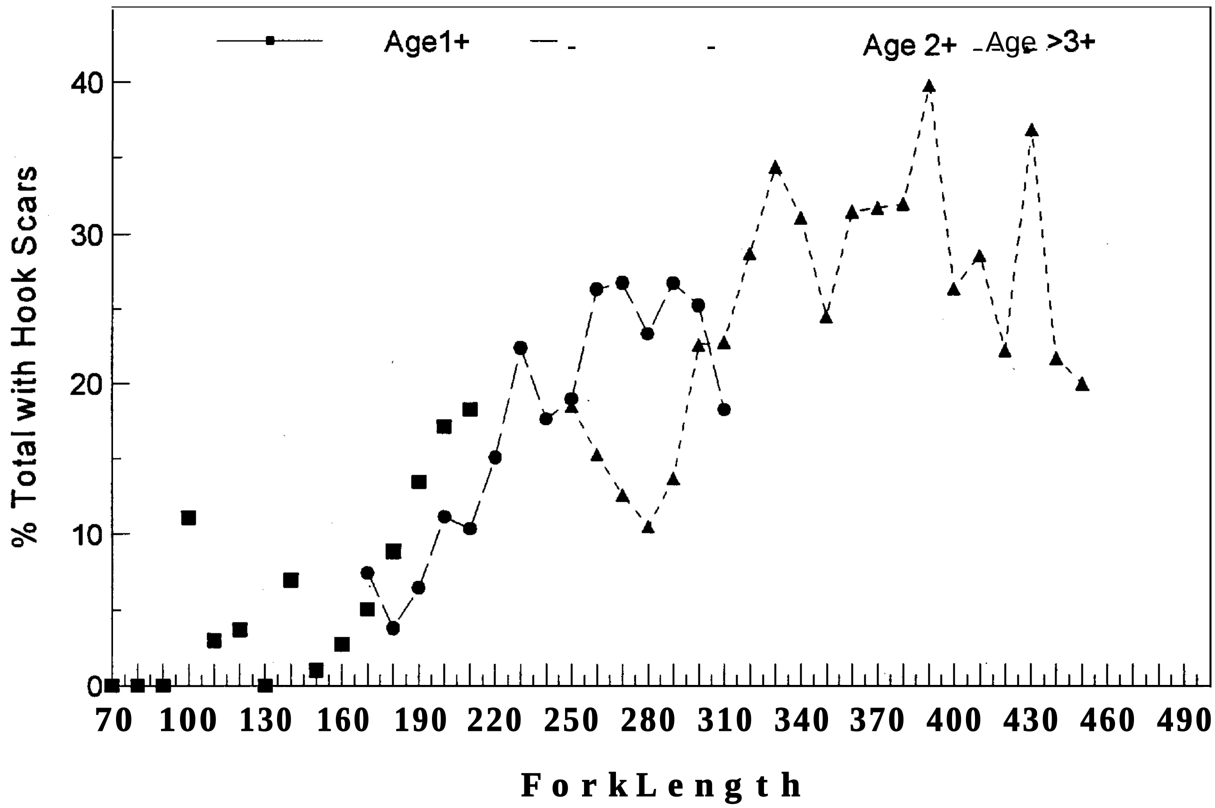

Figure 4. Percent frequency of age 1+, age 2+, and age >3 hook-scarred and non hook-scarred rainbow trout by length in all sections and years combined in the Yakima River.

trout in the 160 mm length group comprised 11.6% of all age 1+ fish, however only 2.8% of these fish had hook-scars. Contrastingly, age 1+ rainbow trout in the 210 mm category comprised 16.5% of all age 1+ fish, 18.3% of these fish had hook-scars. This trend occurred in all age groups sampled (Figure 4), suggesting that hook-scarred fish were larger than non hook-scarred fish within and among age classes.

Discussion

Contrary to our hypothesis, the length to weight relationship of rainbow trout in the Yakima River were not statistically different between rainbow trout with and without hook-scars. Consequently, past hooking injuries cannot explain the lack of increase in fish size and abundance following regulation change in the Yakima River. However, fish may be hooked and released without developing a detectable hook-scar, which could conceal differences in condition between hook-scarred and non hook-scarred fish. Importantly, fish condition did not decrease during the five years when angling pressure appeared to increase, suggesting that angling did not influence fish condition. Furthermore, our study only examined fish that survived a hooking event and did not attempt to assess short term differences in condition. Most mortality associated with hooking and playing occurs within 72 hours (Titus and Vanicek 1988, Nuhfer and Alexander 1963, Schill et al. 1986, Muoneke and Childress 1994 for review). Our results may not apply to other rivers with catch and release regulations, because of differences in environmental conditions among rivers.

We would expect that fish that are caught many times, or experience short growing seasons, to have the greatest difference between hook-scarred and non hook-scarred groups. Fish that are caught or injured repeatedly require time to recover and heal, resulting in a loss of feeding time. The average cutthroat trout inhabiting a heavily-fished section of the Yellow&one River may be caught as many as 9.7 times in a 3.5 month season (Shill et al. 1986). Acute stress, such as might be experienced during a hook and release event, may preclude feeding in some fish for up to three days (Pickering et al. 1986). In addition, fish inhabiting streams with short growing seasons experience a greater proportional loss of feeding time with each hooking event compared to fish inhabiting streams with longer growing seasons.

Although not assessed in the, present study; hook-scars may be a useful measure for monitoring differences in angling pressure within a river as well as the size at which fish recruit into a fishery. In most years we observed, the percentage of hook-scarred fish decreased as section elevation increased. Although no data are presented to support it, we speculate that this circumstance is representative of the angling pressure in each section. The proportions of hook-scarred fish increased with fish length. Very few fish less than 160 mm were scarred, suggesting that fish less than 160 mm were either not fully recruited into the fishery or were not likely to be scarred by a hooking event due to their small size.

Traditional angler surveys are generally labor intensive and costly. Contrastingly, costs of monitoring scar presence would be minimal especially if a population is already being sampled for other biological data. Conceptually, scar presence data would reflect the effects of angling pressure over an entire year or more as opposed to expanding intensive surveys collected during a narrow time frame. However, this comes at a cost to resolution at finer spatial and temporal scales. Scar presence monitoring could enhance knowledge when used in conjunction with traditional angler surveys.

Among the factors we examined, the section of river inhabited influenced fish condition the most. The highest densities and conditions of rainbow trout were in sections 1 and 2 which were below a major source of agricultural runoff which was high in nutrients (Leland 1995). Any effect that was caused by hook-scarring was masked by the effect of the section of river inhabited. In the Yakima River, the conditions of fish that survived a hooking event were not significantly different than non hook-scarred fish.

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