REF ID 88260

Montana Cooperative Fishery Research Unit

Comparative Use of Modified and Natural Habitats of the Upper Yellowstone River by Juvenile Salmonids

by

Alexander V. Zale and Douglas Rider

Montana Cooperative Fishery Research Unit, USGS
Department of Ecology, Montana State University–Bozeman
Bozeman, MT 59717

March 2003

Contents

Comparative Use of Modified and Natural Habitats of the Upper Yellowstone River by Juvenile Salmonids

Alexander V. Zale and Douglas Rider
Montana Cooperative Fishery Research Unit, USGS
Department of Ecology, Montana State University
Bozeman, MT 59717

Abstract: We compared juvenile salmonid use of stabilized main-channel banks (riprap, barbs, jetties) of the upper Yellowstone River to their use of natural, unaltered habitats by electrofishing in spring, summer, and fall, 2001 and 2002. Use of barbs and jetties was similar to that of natural outside bends, and use of riprap sections was higher than that of outside bends. Artificially-placed boulders and shoreline irregularities associated with the stabilized banks likely attracted juvenile salmonids. Bank stabilization did not *directly* decrease quality or quantity of juvenile salmonid habitat along the main channel of the upper Yellowstone River; indirect, geomorphically derived effects of bank stabilization on fish habitat were not examined. We also estimated abundances of juvenile salmonids in ephemeral lateral side channels during high discharge associated with spring runoff to determine if and to what extent juvenile salmonids used side channels. The average 50-m side-channel sample unit (250.8 m^2) contained about 6.3 juvenile trout (all species) and 15.2 juvenile salmonids (trout plus mountain whitefish). Because of low-water conditions during both years of the study, the side channels were inundated for only about 3 to 10 days in 2001 and 1 to 3 weeks in 2002. The rapidity with which these habitats were colonized during the brief periods they were available suggests that juvenile fish positively selected for these habitats. Habitat modifications that reduce the frequency and duration of inundation of side channels, or reduce side-channel formation rates, or directly preclude inundation or accessibility of side channels would likely decrease juvenile fish habitat and possibly recruitment.

Key words: riprap, barb, jetty, bank stabilization, side channel, trout, salmonid

Introduction

Bank stabilization, flow deflection, and flow confinement structures are common features of the upper Yellowstone River in Montana. The reach extending from Gardiner to Springdale includes 18.9 km of dikes and levees, 33.7 km of riprap, and 276 deflection structures (Chuck Dalby, Montana Department of Natural Resources and Conservation, personal communication). The goal of our study was to assess the extent to which changes in aquatic habitats caused by bank stabilization, flow deflection, and flow confinement structures affect juvenile salmonid habitat in the upper Yellowstone River. In main-channel riverine habitats, juvenile salmonids require and are largely restricted to shallow, low-velocity habitats associated with streambanks. Lateral side channels, backwaters, off-channel pools, and tributaries are important nursery habitats not associated with main channels. Our first objective was to compare seasonal juvenile fish use of altered main-channel habitat types to their use of natural, unaltered main-channel habitats to allow assessment of past and future effects of habitat modifications on the fishery resource of the Yellowstone River. Our second objective was to estimate abundances of juvenile salmonids in ephemeral lateral side channels during high discharge associated with spring runoff. We determined if and to what extent juvenile salmonids used side channels to allow estimation of how many fish are displaced when a side channel is disconnected or dewatered as a result of bank stabilization. Both objectives were designed to provide information for the concurrent fish habitat study conducted by Zachary H. Bowen, Ken D. Bovee, and Terry J. Waddle of the U.S. Geological Survey Fort Collins Science Center.

Bank stabilization structures include riprap revetments, flow deflection devices such as barbs, jetties, spur dikes, and fish groins, and flow confinement structures such as berms, levees, or dikes. Riprap revetments are bank-stabilization structures constructed with boulders, broken concrete or similar erosion-resistant materials (Sandheinrich and Atchison 1986). The materials can range in size from medium-sized cobble (12-15 cm in diameter) to round or angular boulders as large as 3 m in diameter; angular boulders are typically used along the Yellowstone River. Jetties, spur dikes, rock deflectors, and wingdams are all rock flow-deflection structures with rocks oriented perpendicular to the water flow or angled downstream. Barbs are rock structures oriented upstream with their height not exceeding the water surface at bankfull discharge (Buddy Drake, Drake and Associates, personal communication). Deflectors have been used widely for fish habitat restoration and bank stabilization and provide diverse fish habitats superior to continuous revetment or riprap (Elser 1968; Witten and Bulkley 1975; Li et al. 1984; Knight and Cooper 1991; Shields et al. 1995) because they create scour holes at their riverward tips, produce slow-water habitat immediately adjacent to the mainstream, and form a complex of depth-velocity-bed type combinations not found adjacent to continuous riprap (Beckett et al. 1983; Li et al. 1984; Baker et al. 1988).

In the spring, many western U.S. rivers and streams experience high discharge fed by snowmelt from high mountain tributaries. Juvenile fish can be flushed long distances downstream in river mainstems during periods of high discharge (Vanderford 1980; Ottaway and Clarke 1981; Ottaway and Forest 1983). Temporary or ephemeral side channels that flow during high discharge are believed to be important habitats for juvenile fish during high flows because they offer shallow, low-velocity refuge, largely not available in the main channel (Orsborn 1990). More permanent secondary channels and backwaters that flow over a wider range of discharges are likely even more critical to fish diversity and production, as they provide water velocities, depths, and substrates not present in the main river channels over longer time periods (Hjort et al. 1984). Backwaters, off-channel pools, side channels, and tributaries are important for young fish for both rearing and winter habitat (Ragland 1974; Bustard and Narver 1975; Ellis et al. 1979; Tschaplinski and Hartman 1983; Sedell et al. 1984; Hartman and Brown 1987; Mesick 1995). Loss of these habitats would be expected to adversely affect abundances of juvenile fish by limiting recruitment and increasing emigration to downstream sections of the river (Orsborn 1990). Side channels can be lost or dewatered by main channel incision resulting from bank stabilization, dewatered by berms or dikes, or prevented from forming by stabilization or modification of main channel banks (Vanderford 1980; Hjort et al. 1984; Dister et al. 1990). Dike or berm structures that block or severely restrict flow through secondary channels produce habitats in which the biotic communities are much different from areas that remain flowing (Baker et al. 1987). They restrict migration between the side and main channels, and can change habitat in the side channels to the degree that they are no longer good fish habitat (Baker et al. 1987).

A literature review on the effects of bank stabilization structures on fish and their habitat conducted at the beginning of this study is reproduced in Appendix 2 of this report.

Objectives

Few studies have examined the effects of bank stabilization on fish distribution and their associated habitats in more than one season or year or at more than a handful of sites. Because of the shortage of long-term, large-scale studies pertaining to juvenile use of particular bank habitats during seasonal changes, and because existing studies provide contradictory or inconsistent findings (Appendix 2), conclusive determinations about how shoreline modifications affect juvenile salmonids cannot be made. Our comparative-use study was designed to help address this deficiency, specifically for juvenile salmonids in the upper Yellowstone River. We also examined juvenile salmonid abundances in ephemeral side channels of the upper Yellowstone River during runoff to assess their importance in this system. Abundance estimates may allow estimation of how many fish are displaced when a side channel is cut-off or dewatered as a result of stabilization projects. Main-channel stream banks and lateral channels are the habitats directly affected by bank stabilization structures.

We focused on juvenile salmonids because juvenile abundances and survival rates typically regulate adult abundances and because this life stage requires and is largely restricted to shallow, low velocity habitats associated with main-channel stream banks and lateral channels (Peters et al. 1998; Bradford and Higgins 2001), in addition to off-channel backwaters, pools, and tributaries. Newly emerged salmonids occupy slow water at the edge of stream channels (Keenleyside 1962; Chapman 1966; Lister and Genoe 1970). Juvenile salmonids conceal in rocky substrates during the day (Keith et al. 1998; Dare et al. 2002) and in winter (Rimmer et al. 1983; Conner et al. 2002; Dare et al. 2002). Juvenile salmonids avoid velocities greater than 11 cm/sec and are typically found at depths less than 30 cm (Li et al. 1984). This has been noted especially along main channels of large rivers where virtually all age-0 trout were within a few meters of the edge of the water (Contor 1989; Schrader and Griswold 1992; Griffith and Smith 1993). Such behavior appears to be a combined response of selecting positions of low velocity and proximity to cover on lower gradient stream reaches. Because of their narrow and specific habitat requirements, juvenile fish assemblages are good indicators of habitat structure and the ecological integrity of large river systems (Schiemer et al. 1991).

Specific objectives of our study were to:

1. Compare juvenile salmonid use of altered bank habitats to use of natural, unaltered bank habitats on the upper Yellowstone River; and

2. Determine juvenile salmonid use of lateral, ephemeral side-channel habitats during periods of high run-off on the upper Yellowstone River.

Study Area

Our study area encompassed parts of the upper Yellowstone River from the Mallard's Rest Fishing Access in Paradise Valley 16 km south of Livingston to the Mayor's Landing Fishing Access on the east end of Livingston (Figure 1). The study area was divided into two primary reaches, designated Study Reaches 1 and 2, respectively. Study Reach 1 (segments 7 and 8 in the Upper Yellowstone River Physical Features Inventory Report; USDA Natural Resources Conservation Service and Montana Department of Environmental Quality 1998) extended from Mallard's Rest to about 450 meters upstream from the mouth of Nelson's Spring Creek (Figures 2 and 3), a distance of 10.7 river km (6.7 miles), but with a 2.4-km reach (1.5 miles) omitted from Pine Creek Bridge downstream. This reach was omitted to meet total river-length limitations of the concurrent fish habitat study conducted by the USGS Fort Collins Science Center. Study Reach 2 (segments 9 and 10 in the Upper Yellowstone River Physical Features Inventory Report) extended from Carter's Bridge Fishing Access to Mayor's Landing (Figure 4). This reach was about 8.1 river km long (5.0 miles). The total distance of the two reaches was about 16.4 river km (10.2 miles). The diverse array of bank habitats, their proximity to each other, and the river access points made these segments the preferred study reaches. Native salmonid species in the study area are Yellowstone cutthroat trout *(Oncorhynchus clarki bouvieri)* and mountain whitefish *(Prosopium williamsoni).* Nonnative salmonids are rainbow trout *(O. mykiss),* brown trout *(Salmo trutta)* and brook trout *(Salvelinus fontinalis).*

Methods

Comparative Use Study

Sample sites were 50-m long reaches of shoreline selected using a stratified-random sampling design (Brown and Austen 1996). All riverbanks within each reach were stratified according to shoreline type as either inside bends (point bars), outside bends, straight segments, riprap, jetties, or barbs. Shoreline types and stabilization structures were identified using aerial photos, maps, and on-site inspection. Each reach of continuous shoreline type (unaltered banks and riprap) was divided into numbered 50-m sites. Deflection structures (barbs and jetties) were numbered and partitioned into 50-m long sites with the deflection structure at the center of the site. Sample sites were selected randomly within each reach from the entire set of possible sites of each bank type in each reach. An exception to this was that some natural outside-bend habitats in Reach 1 were excluded because they were impossible to sample safely during spring. These sites had high water velocities and steep banks that could not be negotiated on foot. We sampled four such 50-m sites in summer to assess their value as juvenile fish habitat, but captured only one juvenile trout therein. We believe the exclusion of these sites did not affect the validity of our findings.

Eight sites of each of the 6 bank types were selected in Reach 1 (Figures 2 and 3). Six sites of each type were randomly chosen in Reach 2 (Figure 4). Sites were assigned a reach-specific number and located using UTM coordinates (Table 1). We used the same set of sites during each sampling season during both years of the study (2001 and 2002).

Sampling was conducted during three functional seasons (spring, summer, and fall) each year to assess seasonal habitat-use patterns. Habitat use may change over time as a function of fish size and changing physiological needs (Hunt 1969; Tschaplinski and Hartman 1983; Rimmer et al. 1984; Bisson et al. 1988). Sampling seasons were as follows:

> Spring: prior to runoff (April 1 to May 15);
> Summer: during summer low flow (July 1 to August 31); and
> Fall: as water temperatures declined and fish shifted to wintering habitats (October 1 to November 21).

We sampled all 48 sites in Reach 1 during all 6 seasons of the study. All 36 sites in Reach 2 were sampling in both spring and both summer seasons, but winter conditions prevented us from completing all sites there during both fall seasons (Table 1). Only 12 sites were sampled in fall 2001 (2 of each bank type) and 26 sites were sampled in fall 2002 (4 inside bends, straight sites, jetties, and barbs; 5 outside bends and riprap sites). Overall, we sampled 14 replicates of each of 6 bank types, in two river reaches, in six seasons over two years (470 samples).

Fish were sampled using an aluminum drift boat outfitted for electrofishing with a Coffelt VVP-15 electrofishing unit and a gasoline-powered generator. A hand-held mobile electrode, 2.5 m in length with a 10-cm diameter anode ring, was connected to the electrofishing unit with 33 m of electric cable. The aluminum driftboat functioned as the cathode. Current was smooth DC at 200 volts. This setup is considered the most efficient for capturing juvenile fish in large streams (Copp 1989). A team of three operators sampled fish, moving upstream from the lower end of each 50-m site to the upper end. One person operated the hand-held electrode, a second person netted stunned fish and deposited them in the boat's livewell, and a third person maneuvered the drift boat along the bank. Because juvenile salmonids inhabit only shallow water, we sampled only depths less than about 55 cm. Few juvenile salmonids were captured in trial runs in water deeper than 60 cm, except in scour holes immediately adjacent to the upstream sides of barbs and jetties, and among large boulders in riprap, all of which we sampled throughout the study. All fish captured at each site were temporarily anesthetized with clove oil and identified to species. Salmonids were enumerated and measured (mm total length). Almost all fish were immediately returned to the river alive; some small rainbow and cutthroat trout could not be definitively identified in the field and were preserved for conclusive identification in the laboratory.

We considered only juvenile salmonids in our analyses. These included fish from the 2000 year class in spring 2001, the 2001 year class in summer and fall 2001 and spring 2002, and the 2002 year class in summer and fall 2002. Length-frequency distributions were constructed for each species during each sampling season and year in each reach to establish maximum lengths that encompassed the appropriate year classes (DeVries and Frie 1996). Fish longer than these maximum lengths were excluded from further consideration. Although we had originally intended to consider all fishes in our analyses, we subsequently limited our investigation to salmonids because other species were too numerous to allow completion of our sampling design; our examination of non-salmonids was limited and cursory.

Fish abundances were expressed as the number of juveniles captured at each 50-m site during a single electrofishing pass. Significant differences among natural-logarithm transformed mean abundances of fish at different bank types were tested using analysis of variance (SAS version 8.2). Bank type, season, reach, and year were considered class variables. Tukey's multiple comparisons test was used to distinguish habitats in which abundances were significantly different. For all tests, significance was set at $\alpha = 0.05$. Primary comparisons of interest were between outside bends and the stabilized banks. Fish abundances tend to be highest at natural outside banks (White 1991) and bank stabilization structures are typically built on outside bends because lateral erosion is greatest there. Sampling of inside bends and straight sections was performed primarily to insure comprehensive coverage of all available bank types.

A potential problem with our approach is that it depends on equal catchability of fish inhabiting different habitats. For example, if fish found along a natural outside bend were more, or less, catchable by one-pass electrofishing than fish inhabiting riprap, then one-pass catches in these habitats would not be directly comparable as indicators of fish abundance. We therefore conducted 3 or 4-pass depletion sampling at a subset of sample sites (2 inside bends, 2 straight sites, 4 outside bends, 4 riprap sites, 4 barbs, and 5 jetties) in summer to calculate capture probabilities in each of the bank types. Capture probabilities were calculated using the maximum-likelihood generalized removal estimator (Otis et al. 1978) using the computer program CAPTURE (White et al. 1982) and compared among bank types using analysis of variance (SAS version 8.2).

Habitat parameters were recorded within the area sampled for fish at each sample site (less than 55 cm deep or within 1 m of the shoreline). These included water velocity, water depth, sample-area width, and substrate. Measurements were recorded at 1-m intervals along 6 equally-spaced transects 10 m apart extending perpendicularly out from shore at the continuous-shoreline sites (natural banks and riprap). At deflection structures, 7 transects were located 12.5 and 25 m upstream and downstream from the center of each structure, at the offshore tip of the structure, and at the 2 junctions of the structure with the shoreline. Substrates were classified according to a modified Wentworth particle-size scale as follows: large boulder >512 mm diameter; small boulder 256-512 mm; cobble 64-256 mm; pebble 4-64 mm; gravel 2-4 mm; fines <2 mm.

Side Channel Study

Ephemeral side channels in both reaches (Figures 2, 3, and 4; Table 2) were located using aerial photos, advice of local experts, and site visits. We defined ephemeral side channels as those that flowed during spring runoff and not during other seasons. Eleven side channel sites were sampled from 18 to 31 May in 2001. Fifteen sites were sampled from 2 June to 2 July in 2002, including 5 of the sites sampled in 2001. During both years, duration of runoff limited the number of sites we could sample. Side channels flowed for only 3 to 10 days in 2001 and 1 to 3 weeks in 2002.

Absolute abundances of juvenile salmonids were estimated in 50-m reaches of the side channels by 3 or 4-pass backpack electrofishing depletion sampling. Block nets were used to restrict fish movements within the sampled area. The electrofishing crew consisted of one person electrofishing and one or two persons netting the fish. Captured fish were measured and identified to species. Abundance estimates were calculated using the maximum-likelihood generalized removal estimator (Otis et al. 1978) using the computer program CAPTURE (White et al. 1982). Only fish judged to be juveniles based on the length-frequency analyses described in the previous section were included in our calculations. Densities of juvenile fish in side channels were calculated by dividing estimated abundances by sampled areas.

Results

Comparative Use Study

Most of the salmonids captured during the study were rainbow trout (N=2763, 62.0%), followed by brown trout (1189, 26.7%), mountain whitefish (334, 7.5%), Yellowstone cutthroat trout (166, 3.7%), and brook trout (1, <0.1%). Sizes of fish captured encompassed a broad range including fish over 500 mm TL (Figures 5-16), but most of the fish were juveniles as expected given our sampling protocol. Maximum lengths that encompassed the appropriate year classes of each species in each reach during each sampling season and year as judged by length-frequency analyses are indicated in Figures 5 through 16. In general, juvenile brown trout were larger than sympatric rainbow trout and mountain whitefish in any season and reach; Yellowstone cutthroat trout were smallest. These size differences corresponded to sequence of spawning and emergence; brown trout spawn in fall and emerge earlier than rainbow trout, which spawn in spring, and Yellowstone cutthroat trout spawn in early summer. Juvenile mountain whitefish were smaller than brown trout (both are fall spawners) because their eggs sizes are smaller. Among seasons, juvenile fish were largest in spring because they consisted of fish produced during the previous year. Fish were smallest in summer when they were only a few months post-hatch and larger in fall. Fish tended to be slightly larger downstream (Reach 2) than upstream (Reach 1). Excluding fish longer than the juvenile maxima indicated by the length-frequency analyses, we considered 2415 rainbow trout (66.7%), 932 brown trout (25.8%), 169 mountain whitefish (4.7%), 102 Yellowstone cutthroat trout (2.8%), and 1 brook trout (<0.1%) in subsequent analyses.

No significant difference among bank types was found among mean capture probabilities of juvenile fish collected at a subset of sites subjected to depletion sampling (P=0.5945; Figure 17). The overall mean capture probability was 0.743. In other words, the probability that any individual juvenile fish inhabiting one of our 50-m sample sites would be captured during a single electrofishing pass was about 74.3% and did not differ among bank types. Because mean capture probabilities were not significantly different among the six bank types, we were able to directly compare one-pass catches among the habitats as indicators of fish abundance therein. Numbers of each salmonid species captured at each 50-m sampling site during each sampling season are listed in Appendix 1.

Mean numbers of rainbow trout captured were significantly different among the six bank types ($P<0.0001$; Figure 18; Table 3). No significant interaction existed between bank type and reach, season, year, or combination thereof (all $P>0.05$). Mean abundance at inside bends (0.769) was lowest, followed by straight sections (3.359). Abundances at barbs (4.974), outside bends (5.684), and jetties (7.692) were not significantly different. Mean abundance at riprap sites was highest (8.304), but not significantly different from abundance at jetties. Abundances were significantly different among seasons ($P<0.0001$) and between reaches ($P<0.0001$) but not between years ($P=0.0804$).

A significant interaction existed between bank type and reach among mean abundances of brown trout captured at the six bank types ($P=0.0003$). In other words, the relationships among the abundances at the different bank types were different in Reaches 1 and 2. Specifically, abundances at outside bends and jetties were lower in Reach 2 than in Reach 1 relative to expected abundances at the other bank types (Figure 18; Table 4). We therefore treated abundances at the six bank types in each reach separately and repeated the analysis of variance. Bank type, season, and year were considered class variables. Mean numbers of brown trout captured were significantly different among the six bank types in the two reaches ($P<0.001$; Figure 18; Table 4). No significant interaction existed between bank type and season, year, or combination thereof (all $P>0.05$). Abundances were significantly different among seasons ($P<0.0001$) and between years ($P=0.0001$). In Reach 1, mean abundances at inside bends (0.354) and straight sections (1.229) were lowest. Mean abundances at barbs (1.896), outside bends (2.313), jetties (3.250), and riprap (3.625) were not significantly different. In Reach 2, mean abundances at inside bends (0.133) and outside bends (0.774) were lowest, but mean abundances at jetties (1.400) and straight sections (2.233) were not significantly higher than at outside bends (Figure 18). Mean abundance at barbs (2.333) was significantly higher than at outside bends, but was not significantly different from mean abundances at jetties and straight sections. Mean abundance was highest in riprap (3.774), but was not significantly different from abundances at barbs and straight sections.

Numbers of juvenile mountain whitefish (169), Yellowstone cutthroat trout (102), and brook trout (1) captured were insufficient to test for differences in abundances among bank types. We combined abundances of all four trout species to test for differences in abundances of the trout assemblage as a whole among bank types. Mean numbers of trout captured were significantly different among the six bank types ($P<0.0001$; Figure 18; Table 5). No significant interaction existed between bank type and reach, season, year, or combination thereof (all $P>0.05$). Mean abundance at inside bends (1.038) was lowest, followed by straight sections (5.103). Mean abundances at barbs (7.436) and outside bends (7.747) were not significantly different. Mean abundance at jetties (10.449) was not significantly different from mean abundances at barbs or riprap (12.203), but abundance at riprap was significantly higher than at barbs. Abundances were significantly different among seasons ($P<0.0001$) and between reaches ($P<0.0001$) but not between years ($P=0.5614$).

Inclusion of mountain whitefish in the analysis resulted in essentially the same conclusions for all salmonids in aggregate (Figure 18; Table 6). Mean numbers of all salmonids captured were significantly different among the six bank types ($P<0.0001$). No significant interaction existed between bank type and reach, season, year, or combination thereof (all $P>0.05$). Abundances were significantly different among seasons ($P=0.0004$) and between reaches ($P<0.0001$) but not between years ($P=0.7775$). Multiple comparisons testing revealed the same relationships among bank types as among rainbow trout abundances (Figure 18). Mean abundance at inside bends (1.538) was lowest, followed by straight sections (5.423). Abundances at barbs (7.923), outside bends (8.443), and jetties (10.590) were not significantly different. Mean abundance at riprap sites was highest (12.215), but not significantly different from abundance at jetties.

Habitat characteristics of the six bank types suggested some reasons for the differences and similarities in juvenile fish abundances we observed. Inside bends and straight sections tended to be wider and more open than the other bank types whereas riprap sites and jetties were the narrowest (Figure 19); widths of outside bends and barbs were intermediate. Depth distributions of inside bends and straight sections showed these habitats were uniformly shallow whereas riprap and jetties tended to have little shallow habitat relative to deep areas (Figure 20). A wide distribution of depths characterized outside bends and barbs. Slopes of the bank types reflected a combination of their depths and widths (Figure 21). Inside bends sloped gradually, whereas slopes along many riprap and jetty transects were steep. Straight sites, outside bends, and barbs had intermediate slopes, though some transects at barbs were relatively steep. Modal water velocities at all of the bank types were close to zero (Figure 22). High velocities were most common at outside bends and inside bends and to a lesser extent at riprap. Negative velocities (upstream flows) were evident in eddies formed by barbs, jetties, and inside bends. Most of the fish we captured at barb and jetty sites were found immediately upstream and adjacent to these structures in the eddies formed there. Perhaps the most obvious difference between the natural and stabilized sites was the invariable presence of large and small boulders at the latter (Figure 23). Substrates at the natural sites were primarily cobble. On a micro-habitat scale, regardless of bank type, presence of boulders whether natural or artificial, tended to be the best predictor of juvenile fish presence. Notable also was the prevalence of fines at barb and jetty sites (Figure 23), primarily in the silty depositional areas downstream from the deflection structures; fish were almost never found in these areas. The master's thesis currently being prepared by the junior author will examine the influences of site-specific habitat characteristics on juvenile salmonid abundances in greater detail.

The most common non-game species we encountered was the mottled sculpin (*Cottus bairdi*), which was collected in 442 of our 470 main-channel samples (94%). Mottled sculpin were found in all types of habitats but were most numerous in cobble substrates where water velocities were high. Longnose dace (*Rhinichthys cataractae*) were collected in 286 samples (61%), mostly in shallow, slow habitats and eddies; few were seen near boulders at stabilized sites. Longnose suckers (*Catostomus catostomus*) and mountain suckers (*C. platyrhynchus*) were both collected, with longnose more common. Sub-adult and adult suckers were found in deep, slow water near stabilization structures. Schools of juvenile suckers were typically found in slackwater near riprap and along sand beaches between barbs. Two juvenile common carp (*Cyprinus carpio*) and one brook stickleback (*Culaea inconstans*) were collected during summer 2001 at jetty site 36 near the Free River fishing access site.

Side Channel Study

Mean side channel widths ranged from 1.9 to 13.2 m and averaged 5.0 m wide. Areas of the 50-m long sample units ranged from 95 to 658 m^2 (mean 250.8 m^2). Flow durations were 3 to 10 days in 2001 and 1 to 3 weeks in 2002. Most of the juvenile fish captured in side channels were mountain whitefish (60.1%), followed by rainbow trout (30.2%), brown trout (8.4%), and Yellowstone cutthroat trout (1.3%). Estimated abundances of all trout species combined ranged from 0 to 10 fish per sample unit in 2001 and 0 to 14 fish in 2002 (Table 2). Estimated abundances of all salmonids (trout plus mountain whitefish) ranged from 1 to 39 fish per sample unit in 2001 and 3 to 39 fish in 2002 (Table 2). Densities were higher in 2002 than 2001. Mean densities of trout were 0.0124 fish/m^2 (SD $\pm$0.0117, range 0-0.0343 fish/m^2) in 2001 and 0.0346 fish/m^2 (SD $\pm$0.0362, range 0-0.1340 fish/m^2) in 2002. The mean trout density for both years combined was 0.0252 fish/m^2 (SD $\pm$0.0302, range 0-0.1340 fish/m^2). Mean densities of all salmonids were 0.0491 fish/m^2 (SD $\pm$0.0606, range 0.0056-0.2191 fish/m^2) in 2001 and 0.0691 fish/m^2 (SD $\pm$0.0971, range 0.0061-0.4021 fish/m^2) in 2002. The mean salmonid density for both years combined was 0.0606 fish/m^2 (SD $\pm$0.0828, range 0.0056-0.4021 fish/m^2). On average, each 50-m side-channel sample unit contained about 6.3 trout and 15.2 salmonids.

Juvenile salmonids used the side channels during runoff, in appreciable numbers in some instances, despite the short durations of inundation experienced in both years. On several occasions, we observed juvenile fish actively entering the lower ends of side channels as they filled.

Discussion

Comparative Use Study

Our first objective was to compare juvenile salmonid use of altered main-channel bank habitats to use of natural, unaltered bank habitats along the upper Yellowstone River. We successfully completed this component, which was the most comprehensive investigation of its type to date. Use was examined at 14 replicates of each of 6 bank types, in two river reaches, in six seasons over two years (470 samples). Our primary findings were that, in general, juvenile salmonid use of barbs and jetties was similar to that of natural outside bends, and that use of riprap sections was higher than that of natural outside bends. We can infer from these findings that bank stabilization does not *directly* decrease juvenile salmonid habitat along the main channel of the upper Yellowstone River and that therefore juvenile salmonid recruitment from main-channel habitats should not be affected by bank stabilization. Indirect, geomorphically derived effects of bank stabilization (e.g., incision, aggradation, changes in bank lengths) may affect juvenile salmonid habitat, but such effects were outside the scope of our study design.

These results are somewhat surprising in light of findings of previous studies in coldwater systems, most of which showed negative effects of bank stabilization on fish (Appendix 2), and the general belief that natural habitats are better than altered habitats for wild salmonids. The simplest explanation for this incongruity is that many (but not all) of the natural banks of the main channel of the segments of the upper Yellowstone River we sampled are at present relatively poor juvenile salmonid habitat. Many of these banks are relatively uniform and are characterized primarily by cobble substrates. They largely lack the complex, irregular form and roughness elements such as boulders, vegetation, and large woody debris (logs, root wads) that juvenile salmonids prefer for foraging sites, visual isolation from conspecifics, cover from predators (Bryant 1983; Platts 1991; Fausch 1993), and winter habitat (Heifetz et al. 1986; Hillman et al. 1987; Griffith and Smith 1993; Riehle and Griffith 1993; Quinn and Peterson 1996). Moreover, heterogeneous substrates provide low-velocity refuges for salmonid fry, thus decreasing the probability of downstream displacement during high discharges (Heggenes 1988; Moore and Gregory 1988; Meyer and Griffith 1997). An inference from these studies is that simplification of complex natural streambank by stabilization structures would lead to reduction of habitat diversity, which would be detrimental to juvenile salmonids. On the other hand, diversification of simple, homogeneous natural habitat by stabilization structures would be beneficial. Artificially-placed boulders and shoreline irregularities associated with stabilized banks of the Yellowstone River provide such structure and therefore attract juvenile salmonids. Most of the studies that inferred negative effects of bank stabilization were conducted in small, relatively pristine streams, which likely had less-uniform banks and more structural elements than the Yellowstone River. In those streams, bank stabilization may have simplified bank habitats and therefore reduced their value for fish. Most of the studies that inferred positive effects were conducted in small streams highly degraded by farming and grazing (Hunt 1988; Binns 1994; Avery 1995); stabilization there likely increased bank habitat complexity. Findings from the much larger Thompson River in British Columbia (mean annual discharge 775 m^3/s) and Skagit River in Washington (472 m^3/s) were similar to ours; large riprap supported more juvenile salmonids than small riprap or natural cobble-boulder banks (Lister et al. 1995; Beamer and Henderson 1998). The incremental effects of bank stabilization are likely site-specific and dependent on whether or not artificial structures increase or decrease habitat diversity, and more importantly, whether or not juvenile habitat is limiting.

Another line of supporting evidence for our contention that main-channel banks of the upper Yellowstone River are at present relatively poor or unimportant juvenile salmonid habitat is provided by corresponding data from other rivers. The overall mean number of juvenile salmonids we captured at 50-m main-channel sample sites along the Yellowstone River was 7.3 (Table 5). Corresponding juvenile abundances in the Box Canyon and Pinehaven-Riverside reaches of the Henry's Fork of the Snake River in Idaho were 80.6 and 5.3 rainbow trout, respectively (Mitro and Zale 2002), in the Barnosky and Woodson reaches of the Ruby River, Montana, 14.2 and 27.4 brown trout, respectively (Opitz 1999), and in Poindexter Slough, Montana, 63.0 brown trout (Opitz 1999). In general, abundances captured along the Yellowstone River were comparatively low. It seems likely therefore that main-channel bank habitats of the Yellowstone River are not especially important juvenile-rearing habitats; recruitment likely occurs from other habitats such as tributary streams, upstream reaches, the spring creeks, backwaters, or other off-channel habitats.

Our study had a number of limitations that could affect interpretation of our results. For example, the scarcity of large woody debris along the Yellowstone River may be natural, indicative of an already altered system, or a temporary anomaly caused by the 1996 and 1997 floods. If riparian trees were cleared historically or were prevented from recruiting to the river by bank stabilization, grazing, or other land management practices, then the abundances of fish we captured along natural banks may have been artificially low. Minor forest clearing has occurred, but probably not enough to make a difference; the lack of wetted large woody debris is likely related to channel geometry that exports such debris downstream during runoff (Mike Merigliano, University of Montana, personal communication) or causes it to be deposited above the waterlines we sampled (Chuck Dalby, personal communication). The floods of 1996 and 1997 likely contributed to these processes. A related limitation was that both years of our study were low-water years (USGS provisional data; http://mt.waterdata.usgs.gov/nwis/). This may have affected our findings, if for example, low water elevations prevented juvenile salmonids from accessing preferred natural habitats or large woody debris and restricted them to accessible stabilized banks instead. However, low water restricted access to some stabilized banks at times as well.

Perhaps the most important limitation of our study is that we do not know how important main-channel banks are to recruitment of salmonids in the Yellowstone River. They may be inconsequential if most juveniles in the system are produced in tributaries, spring creeks, side channels, or farther upstream. Conversely, main-channel banks could be producing most of the fish that later recruit to the fishable population. The salmonid fishery of the Yellowstone River is relatively unique in that it is not considered recruitment-limited (Joel Tohtz, Montana Department of Fish, Wildlife and Parks, personal communication), meaning that adult abundances do not track disturbances or weather conditions that typically affect juvenile survival and abundance; other bottlenecks apparently limit adult salmonid abundances in this river. Lack of a recruitment limitation is likely related to natural resilience of the system resulting from its present environmental quality and connectivity (Joel Tohtz, personal communication). In the long term, cumulative insults to the system may degrade this resiliency and elicit recruitment limitations.

Our study was also limited in that it addressed only juvenile fish. Adequate recruitment is a necessary factor in maintaining a healthy fish population, but it is only one component. Habitat and food for sub-adult and adult fish are also required, as are spawning sites. Our study did not address the effects of bank stabilization on these factors. If habitat for older fish is decreased by bank stabilization, or it decreases food availability, or limits spawning habitat, then the fact that bank stabilization does not limit main-channel habitat for juvenile salmonids may be irrelevant. Conditions for all life stages must be met to produce adequate numbers of catchable-sized adults. Our study only showed that bank stabilization does not diminish the value of main-channel banks as juvenile habitat. Furthermore, we ignored all non-salmonid species. Our findings should not be construed to mean that bank stabilization is "good for fish" across the board.

Adult salmonid abundance monitoring as conducted by Montana Fish, Wildlife and Parks is perhaps the most effective and comprehensive method for assessing fundamental effects of environmental perturbations on fish in the upper Yellowstone River system. Such monitoring may not allow inference of precisely what is causing a problem, but it can identify if a problem exists. Studies can then be designed to determine exactly where the problem lies. Trends in adult abundance will reflect significant effects of bank stabilization on the fishery. Of course, countermeasures may be difficult or functionally impossible by the time that declines in adult abundances are noted.

Side Channel Study

Side channels may be important natural nursery habitat for juvenile salmonids in the

Yellowstone River system, considering the relative paucity of boulders, large woody debris, and other cover and roughness elements along the main-channel banks of the Yellowstone River. Their role may be especially important during runoff when shallow, low-velocity habitat is negligible along the main channel and is present primarily in the side channels and overbank areas (Zachary Bowen, personal communication). The densities of fish we estimated in the side channels were not exceptionally remarkable, except that they were attained in short time periods. Because of low-water conditions during both years of the study, the side channels we sampled were inundated for only about 3 to 10 days in 2001 and 1 to 3 weeks in 2002. Often, our samples had to be made within a few days of the commencement of flow. Nevertheless, none of the side channels were completely barren of fish and some contained high densities, especially of mountain whitefish. Because flow durations were short, it is unlikely that the densities we estimated approximated the potential carrying capacity of the side channels. The rapidity with which these habitats were colonized during the brief periods they were available suggests that juvenile fish congregated in these habitats. If side channels were inundated for longer durations, more frequently, and over greater areas, then it seems likely that availability of juvenile fish habitat would be increased and therefore perhaps greater recruitment would be elicited. On the other hand, if main-channel bank stabilization causes main-channel incision and reduces the frequency and duration of inundation of side channels, or reduces side-channel formation rates, or directly precludes inundation or accessibility of side channels by dike or berm structures, then juvenile fish habitat and recruitment will likely be reduced. An understanding of the effect and extent of such geomorphological changes is needed to better comprehend the effects of bank stabilization on the fishery resources of the Yellowstone River. The concurrent geomorphology study being conducted by Montana DNRC is examining the type and abundance of side channels from Gardiner to Springdale and how and why those characteristics may have changed from 1948-49 to 1999 (Chuck Dalby, personal communication).

Additional Research Needs

Several additional investigations would provide a more comprehensive understanding of the effects of bank stabilization on aquatic biota of the upper Yellowstone River. First, additional sampling during years with higher discharges, both along main-channel banks and in side channels, would allow inference about the applicability of our findings under more normal conditions. Second, assessment of the effects of bank stabilization on non-game fishes, macroinvertebrates, and adult and sub-adult salmonids would provide a more holistic assessment of this issue. Third, a comprehensive assessment of recruitment dynamics of salmonids in the upper Yellowstone River system would provide managers with an understanding of which habitats (e.g., tributaries, spring creeks, backwaters, side channels, upstream reaches) actually produce the juvenile fish that later become catchable adults and therefore may require protection.

Management Implications

Because juvenile salmonid abundances along altered main-channel banks of the upper Yellowstone River were similar or greater than those along unaltered banks, juvenile salmonid recruitment from main-channel habitats should not be deleteriously affected by incremental increases in bank stabilization. Indirect or cumulative effects of bank stabilization, or both, may affect juvenile salmonid habitat.

Habitat modifications that reduce the frequency and duration of inundation of side channels, or reduce side-channel formation rates, or directly preclude inundation or accessibility of side channels would likely decrease juvenile fish habitat and possibly recruitment.

Acknowledgments

Funding for this study was provided by the U.S. Army Corps of Engineers, Omaha District, under the patient oversight and guidance of Michael C. Gilbert. The study was conducted in association with the Governor's Upper Yellowstone River Task Force; the Task Force's Members, Coordinator Liz Galli-Noble, and Technical Advisory Committee greatly facilitated conduct of the study. Billie Kerans graciously provided expert statistical advice. John Borkowski and Matthew G. Mitro assisted with study design. Zachary Bowen, Ken Bovee, Terry Waddle, and other staff at the USGS Fort Collins Science Center provided technical and moral support. Critical reviews of an earlier draft of this report were graciously provided by Chuck Dalby, Marcus Miller, Robert S. Nebel, Bradley Shepard, Allan Steinle, and Joel Tohtz. William Gould confirmed non-game species identifications. Field and technical assistance were provided by Ben Bailey, Tera Bandy, Mary Beatty, Carrie Brooke, Pete Brown, Adam J. Craig, Chris Curre, Pete Finch, David Hamalainen, Cody Hankinson, Ryan Harnish, Nathan Jones, Shane Keep, Ryan Limb, Halcyon Lukins, Thomas E. McMahon, Chris Owen, Sid Rider, Denton Saunders, Kevin Szafranski, and Mike Zambarano. We extend our heartfelt thanks to all who helped in this endeavor.

Literature Cited

Avery, E. L. 1995. Effects of streambank riprapping on physical features and brown trout standing stocks in Millville Creek. Wisconsin Department of Natural Resources Research Report 167.

Baker, J. A., C. H. Pennington, C. R. Bingham, L. E. Winfield. 1987. An ecological evaluation of five secondary channel habitats in the Lower Mississippi River. Report 7; Lower Mississippi River Environmental Program, U.S. Army Corps of Engineers, Mississippi River Commission, Vicksburg, Mississippi.

Beamer, E. M., and R. A. Henderson. 1998. Juvenile salmonid use of natural and hydromodified stream bank habitat in the mainstem Skagit River, northwest Washington. Skagit System Cooperative, LaConner, Washington.

Beckett, D. C., C. R. Bingham, and L. G. Sanders. 1983. Benthic macroinvertebrates of selected habitats of the lower Mississippi River. Journal of Freshwater Ecology 2:247-261.

Binns, N. A. 1994. Long-term responses of trout and macrohabitats to habitat management in a Wyoming headwater stream. North American Journal of Fisheries Management 14:87-98.

Bisson, P. A., K. Sullivan, and J. L. Nielson. 1988. Channel hydraulics, habitat use, and body form of juvenile coho salmon, steelhead, and cutthroat trout in streams. Transactions of the American Fisheries Society 117:262-273.

Bradford, M. J., and P. S. Higgins. 2001. Habitat-, season-, and size-specific variation in diel activity patterns of juvenile chinook salmon (*Oncorhynchus tshawytscha*) and steelhead trout (*Oncorhynchus mykiss*). Canadian Journal of Fisheries and Aquatic Sciences 58:365-374.

Brown, M. L., and D. J. Austen. 1996. Data management and statistical techniques. Pages 17-62 *in* B. R. Murphy and D. W. Willis, editors. Fisheries techniques, 2nd edition. American Fisheries Society, Bethesda, Maryland.

Bustard, D. R., and D. W. Narver. 1975. Aspects of the winter ecology of juvenile coho salmon *(Oncorhynchus kisutch)* and steelhead trout *(Salmo gairdneri)*. Journal of the Fisheries Research Board of Canada 32:667-680.

Bryant, M. D. 1983. The role and management of woody debris in west coast salmonid nursery streams. North American Journal of Fisheries Management 3:322-330.

Chapman, D. W. 1966. Food and space as regulators of salmonid populations in streams. American Naturalist 100:35-357.

Conner, W., H. L. Burge, R. Waitt, and T. C. Bjornn. 2002. Juvenile life history of wild chinook salmon in the Snake and Clearwater Rivers. North American Journal of Fisheries Management 22:703-712.

Contor, C. R. 1989. Diurnal and nocturnal winter habitat utilization by juvenile rainbow trout in the Henry's Fork of the Snake River, Idaho. M.S. thesis, Idaho State University, Pocatello, Idaho.

Copp, G. H. 1989. Electrofishing for fish larvae and 0+ juveniles: equipment modifications for increased efficiency with short fishes. Aquaculture and Fisheries Management 20:453-462.

Dare, M. R., W. A. Hubert, and K. G. Gerow. 2002. Changes in habitat availability and habitat use and movements by two trout species in response to declining discharge in a regulated river during winter. North American Journal of Fisheries Management 22:917-928.

DeVries, D. R., and R. V. Frie. 1996. Determination of age and growth. Pages 483-512 *in* B. R. Murphy and D. W. Willis, editors. Fisheries techniques, 2nd edition. American Fisheries Society, Bethesda, Maryland.

Dister, E., D. Gomer, P. Obrdilk, P. Peterman, and E. Schneider. 1990. Water management and ecological perspectives of the upper Rhine's floodplains. Regulated Rivers: Research and Management 5:1-15.

Ellis, J. M., G. B. Farabee, and J. B. Reynolds. 1979. Fish communities in three successional stages of side channels in the Upper Mississippi River. Transactions of the Missouri Academy Science 13:5-20.

Elser, A. A. 1968. Fish populations of a trout stream in relation to major habitat zones and channel alterations. Transactions of the American Fisheries Society 97:389-397.

Fausch, K. D. 1993. Experimental analysis of microhabitat selection by juvenile steelhead (*Oncorhynchus mykiss*) and coho salmon (*Oncorhynchus kisutch*) in a British Columbia stream. Canadian Journal of Fisheries Aquatic Sciences 50:1198-1207.

Griffith, J. S., and R. W. Smith. 1993. Use of winter concealment cover by juvenile cutthroat and brown trout in the South Fork of the Snake River, Idaho. North American Journal of Fisheries Management 13:823-830.

Hartman, G. F., and T. G. Brown. 1987. Use of small, temporary, floodplain tributaries by juvenile salmonids in a west coast rain-forest drainage basin, Carnation Creek, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 44:262-270.

Heggenes, J. 1988. Substrate preferences of brown trout (*Salmo trutta*) in artificial stream channels. Canadian Journal of Fisheries and Aquatic Sciences 45:1801-1806.

Heifetz, J., J. Murphy, and K. V. Koski. 1986. Effects of logging on winter habitat of juvenile salmonids on Alaskan streams. North American Journal of Fisheries Management 6:52-58.

Hillman, T. W., J. S. Griffith, and W. S. Platts. 1987. Summer and winter habitat selection by juvenile chinook salmon in a highly sedimented Idaho stream. Transactions of the American Fisheries Society 116:185-195.

Hjort, R. C., P. L. Hulett, L. D. Labolle, and H. W. Li. 1984. Fish and invertebrates of revetments and other habitats in the Willamette River, Oregon. Technical Report E-84-9, U.S. Army Engineer Waterway Experiment Station, Vicksburg, Mississippi.

Hunt, R. L. 1988. A compendium of 45 trout stream habitat development evaluations in Wisconsin during 1953-1985. Wisconsin Department of Natural Resources Technical Bulletin 162.

Hunt, R. L. 1969. Overwinter survival of wild fingerling brook trout in Lawrence Creek, Wisconsin. Journal of the Fisheries Research Board of Canada 26:1473-1483.

Keith, R. M., T. C. Bjornn, W. R. Meehan, N. J. Hetrick, and M. A. Brusven. 1998. Response of juvenile salmonids to riparian and instream cover modifications in small streams flowing through second-growth forests of southeast Alaska. Transactions of the American Fisheries Society 127:889-907.

Keenleyside, M. H. A. 1962. Skin-diving observations of Atlantic salmon and brook trout in the Miramichi River, New Brunswick. Journal of the Fisheries Research Board of Canada 19:625-634.

Knight, S. S., and C. M. Cooper. 1991. Effects of bank protection on stream fishes. Pages 13-34 to 13-39 *in* Proceeding of the Fifth Federal Interagency Sedimentation Conference. Federal Energy Regulatory Commission, Washington, D.C.

Li, H. W., C. B. Schreck, and R. A. Tubb. 1984. Comparison of habitats near spur dikes, continuous revetments, and natural banks for larval, juvenile, and adult fishes of the Willamette River. Water Resources Research Institute, Oregon State University, Corvallis.

Lister, D. B., and H. S. Genoe. 1970. Stream habitat utilization by cohabiting underyearlings of chinook *(Oncorhynchus tshawytscha)* and coho salmon *(Oncorhynchus kisutch)* in the Big Qualicum River, British Columbia. Journal of the Fisheries Research Board of Canada 27:1215-1224.

Lister, D. B., R. J. Benniston, R. Kellerhals, and M. Miles. 1995. Rock size affects juvenile salmonid use of streambank riprap. Pages 621-632 *in* C. R. Thorne, S. R. Abt, B. J. Barends, S. T. Maynord, and K. W. Pilarczyk, editors. River, coastal and shoreline protection: erosion control using riprap and armourstone. John Wiley and Sons, New York.

Mesick, C. F. 1995. Response of brown trout to streamflow, temperature, and habitat restoration in a degraded system. Rivers 5:57-95.

Meyer, K. A., and J. S. Griffith. 1997. Effects of cobble-boulder substrate configuration on winter residency of juvenile rainbow trout. North American Journal of Fisheries Management 17:77-84.

Mitro, M. G., and A. V. Zale. 2002. Seasonal survival, movement, and habitat use of age-0 rainbow trout in the Henrys Fork of the Snake River, Idaho. Transactions of the American Fisheries Society 131:271-286.

Moore, K. M., and S. V. Gregory. 1988. Response of young-of-the-year cutthroat trout to manipulations of habitat structure in a small stream. Transactions of the American Fisheries Society 117:162-170.

Opitz, S. T. 1999. Effects of whirling disease on recruitment of brown trout in the Ruby River and Poindexter Slough, Montana. Master's thesis. Montana State University, Bozeman.

Orsborn, J. F. 1990. Pilot study on the condition of fisheries environments in river basins on the Olympic Peninsula. Prepared for the USDA Forest Service, Olympic National Forest, Olympia, Washington.

Otis, D. L., K. P. Burnham, G. C. White, and D. R. Anderson. 1978. Statistical inference from capture data on closed animal populations. Wildlife Monographs 62.

Ottaway, E. M., and A. Clarke. 1981. A preliminary investigation into the vulnerability of young trout *(Salmo trutta)* and Atlantic salmon *(Salmo salar)* to downstream displacement by high water velocities. Journal of Fish Biology 19:135-145.

Ottaway, E. M., and D. R. Forest. 1983. The influence of water velocity on downstream movement of alevins and fry of brown trout, *Salmo trutta.* Journal of Fish Biology 23:221-227.

Peters, R., B. R. Missildine, and D. L. Low. 1998. Seasonal fish densities near riverbanks treated with various stabilization methods. First year report of the Flood Technical Assistance Project. U.S. Fish and Wildlife Service, North Pacific Coast Ecoregion, Western Washington Office, Aquatic Resources Division, Lacey, Washington.

Platts, W. S. 1991. Influences of forest and rangeland management on salmonid fishes and their habitat. American Fisheries Society Special Publication 19:387-423.

Quinn, T. P., and N. P. Peterson. 1996. The influence of habitat complexity and fish size on over-winter survival and growth of individually marked juvenile coho salmon (*Oncorhynchus kisutch*) in Big Beef Creek, Washington. Canadian Journal of Fisheries and Aquatic Sciences 53:1555-1564.

Ragland, D. V. 1974. Evaluation of three side channels and the main channel border of the middle Mississippi River as fish habitat. Contract Report Y-74-1, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Riehle, M. D., and J. S. Griffith. 1993. Changes in habitat use and feeding chronology of juvenile rainbow trout (*Oncorhynchus mykiss*) in fall and the onset of winter in Silver Creek, Idaho. Canadian Journal of Fisheries and Aquatic Sciences 50:2119-2128.

Rimmer, D. M., U. Paim, and R. L. Saunders. 1983. Autumnal habitat shift of juvenile Atlantic Salmon *(Salmo salar)* in a small river. Canadian Journal of Fisheries and Aquatic Sciences 40:671-680.

Rimmer, D. M., U. Paim, and R. L. Saunders. 1984. Changes in the selection of microhabitat by juvenile Atlantic salmon *(Salmo salar)* at the summer-autumn transition in a small river. Canadian Journal of Fisheries and Aquatic Sciences 41:469-475.

Sandheinrich, M. B., and G. J. Atchison. 1986. Environmental effects of dikes and revetments on large riverine systems. Technical Report E-86-5. U.S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Schiemer, F., T. Spindler, A. Wintersbeerger, A. Schneider, and A. Chovanec. 1991. Fish fry associations: important indicators for the ecological status of larger rivers. Pages 2497-2500 *in H.* Kausch and W. Lampert, editors. International Association for Theoretical and Applied Limnology Congress, Munich, 1989.

Schrader, W. C., and R. G. Griswold. 1992. Winter habitat availability and utilization by juvenile cutthroat trout, brown trout, and mountain whitefish in the South Fork, Snake River, Idaho. Final Report to U.S. Bureau of Reclamation, Idaho Department of Fish and Game, Boise, Idaho.

Sedell, J. R., J. E. Yuska, and R. W. Speaker. 1984. Habitats and salmonid distribution in pristine, sediment-rich valley systems: South Fork Hoh and Queets River, Olympic National Park. Pages 33-46 *in* W. R. Meehan, T. R. Merrel, and T. A. Hanley, editors, Fish and wildlife relationships in old-growth forests. American Institute of Fishery Research Biologists.

Shields, F. D., Jr., C. M. Cooper, and S. Testa. 1995. Towards greener riprap: environmental considerations from microscale to macroscale. Pages 557-574 *in* C. R. Thorne, S. R. Abt, B. J. Barends, S. T. Maynord, and K. W. Pilarczyk, editors. River, coastal and shoreline protection: erosion control using riprap and armourstone. John Wiley and Sons, New York.

Tschaplinski, P. J., and G. F. Hartman. 1983. Winter distribution of juvenile coho salmon *(Oncorhynchus kisutch)* before and after logging in Carnation Creek, British Columbia, and some implications for overwinter survival. Canadian Journal of Aquatic Sciences 40:452-461.

USDA Natural Resources Conservation Service Montana Department of Environmental Quality. 1998. Upper Yellowstone River physical features inventory report, Gardiner to Springdale. Prepared for the Upper Yellowstone River Task Force and Park Conservation District.

Vanderford, M. J. 1980. Fish and wildlife, GREAT I study of the upper Mississippi River, technical appendices, Vol. 5. Department of Nuclear Engineering Science, University of Florida, Gainesville.

White, G. C., D. R. Anderson, K. P. Burnham, and D. L. Otis. 1982. Capture-recapture and removal methods for sampling closed populations. Los Alamos National Laboratory, LA-8787-NERP, Los Alamos, New Mexico.

White, R. J. 1991. Resisted lateral scour in streams–its special importance to salmonid habitat and management. American Fisheries Society Symposium 10:200-203.

Witten, A. L., and R. V. Bulkley. 1975. A study of the effects of stream channelization and bank stabilization on warmwater sport fish in Iowa. Subproject No. 2. A study of the impact of selected bank stabilization structures on game fish and associated organisms. U.S. Fish and Wildlife Service, Office of Biological Services Report 76/12, Iowa State University, Ames, Iowa.

Table 1. Shoreline sample site locations and sampling dates, Yellowstone River, 2001 and 2002.

	UTM coordinates		2001			2002		
Site	E	N	Spring	Summer	Fall	Spring	Summer	Fall
				Reach 1				
Inside Bend								
1	530176	5037001	30 MAR	22 AUG	31 OCT	24 APR	23 JUL	22 OCT
11	532615	5038793	6 APR	23 JUL	9 OCT	24 APR	14 AUG	22 OCT
34	532740	5039628	18 APR	18 AUG	25 OCT	17 APR	31 JUL	22 OCT
44	532766	5039671	25 APR	18 AUG	25 OCT	17 APR	31 JUL	10 OCT
22	532944	5041945	10 APR	24 JUL	23 OCT	8 MAY	7 AUG	9 NOV
28	532996	5042040	17 APR	31 JUL	23 OCT	8 MAY	7 AUG	7 NOV
42	532411	5042985	25 APR	15 AUG	31 OCT	8 MAY	7 AUG	15 OCT
30	532943	5043183	17 APR	15 AUG	3 NOV	10 MAY	5 AUG	8 OCT
Straight								
3	530833	5037325	31 MAR	20 AUG	25 OCT	24 APR	23 JUL	17 OCT
43	530901	5037377	25 APR	20 AUG	25 OCT	24 APR	31 JUL	17 OCT
5	531094	5037437	5 APR	23 JUL	9 OCT	12 APR	17 JUL	10 OCT
14	531271	5034525	17 APR	23 JUL	26 OCT	24 APR	15 AUG	10 OCT
12	531882	5038047	7 APR	23 JUL	26 OCT	12 APR	15 JUL	17 OCT
21	532639	5042480	10 APR	31 JUL	23 OCT	8 MAY	7 AUG	7 NOV
23	532599	5042540	13 APR	15 AUG	26 OCT	10 MAY	14 AUG	7 NOV
20	532772	5043422	10 APR	15 AUG	7 OCT	20 APR	5 AUG	8 OCT
Outside Bend								
2	532093	5038447	31 MAR	23 JUL	16 OCT	5 APR	17 JUL	17 OCT
6	532778	5039610	15 APR	23 JUL	19 OCT	5 APR	15 AUG	17 OCT
17	532599	5038717	7 APR	18 AUG	26 OCT	12 APR	15 JUL	19 OCT
37	533001	5042203	24 APR	17 AUG	7 OCT	20 APR	7 AUG	9 NOV
25	533008	5042016	13 APR	24 JUL	23 OCT	20 APR	2 AUG	15 OCT
38	533015	5042015	24 APR	17 AUG	23 OCT	20 APR	2 AUG	15 OCT

	UTM coordinates		2001			2002		
Site	E	N	Spring	Summer	Fall	Spring	Summer	Fall
27	532408	5042960	17 APR	15 AUG	31 OCT	8 MAY	5 AUG	7 NOV
19	532776	5044202	10 APR	31 JUL	3 NOV	3 MAY	7 AUG	18 OCT
Riprap								
9	532094	5038109	6 APR	10 AUG	20 OCT	19 APR	31 JUL	19 OCT
7	532100	5038078	6 APR	10 AUG	20 OCT	19 APR	31 JUL	15 OCT
24	532985	5043135	13 APR	15 AUG	7 OCT	10 MAY	5 AUG	5 NOV
33	532985	5043280	13 APR	15 AUG	7 OCT	10 MAY	5 AUG	8 OCT
40	532706	5043980	24 APR	16 AUG	3 NOV	3 MAY	13 AUG	5 NOV
41	532221	5044718	24 APR	31 JUL	31 OCT	20 APR	14 AUG	8 OCT
31	532223	5044777	17 APR	21 AUG	31 OCT	9 MAY	14 AUG	5 NOV
29	532237	5044849	17 APR	21 AUG	31 OCT	9 MAY	14 AUG	9 NOV
Jetty								
35	531835	5037960	18 APR	10 AUG	20 OCT	19 APR	16 JUL	19 OCT
10	531972	5038071	6 APR	23 JUL	20 OCT	19 APR	15 AUG	19 OCT
8	531912	5038035	6 APR	10 AUG	20 OCT	19 APR	31 JUL	19 OCT
16	532493	5039451	7 APR	23 JUL	26 OCT	5 APR	23 JUL	17 OCT
46	532601	5041435	25 APR	17 AUG	18 OCT	9 MAY	13 AUG	15 OCT
26	532650	5041476	13 APR	17 AUG	23 OCT	9 MAY	13 AUG	15 OCT
39	532706	5044104	24 APR	16 AUG	23 OCT	3 MAY	13 AUG	5 NOV
48	532712	5044063	26 APR	16 AUG	3 NOV	3 MAY	13 AUG	5 NOV
Barb								
36	530337	5036959	18 APR	20 AUG	16 OCT	12 APR	17 JUL	10 OCT
13	530416	5036990	7 APR	20 AUG	16 OCT	12 APR	17 JUL	10 OCT
45	531655	5037892	25 APR	22 AUG	19 OCT	12 APR	16 JUL	22 OCT
4	531701	5037901	31 MAR	20 AUG	19 OCT	12 APR	16 JUL	22 OCT
15	531758	5037905	7 APR	22 AUG	19 OCT	5 APR	16 JUL	22 OCT
47	532701	5042553	26 APR	15 AUG	18 OCT	8 MAY	5 AUG	8 OCT
18	532798	5044250	10 APR	17 AUG	31 OCT	3 MAY	7 AUG	15 OCT

	UTM coordinates		2001			2002		
Site	E	N	Spring	Summer	Fall	Spring	Summer	Fall
32	532932	5043281	3 APR	21 AUG	7 OCT	3 MAY	31 JUL	9 NOV
				Reach 2				
Inside Bend								
14	533768	5049998	1 MAY	25 JUL	16 OCT	17 APR	25 JUL	12 NOV
27	533374	5053038	4 MAY	9 AUG	8 OCT	26 APR	24 JUL	25 OCT
16	533678	5052807	1 MAY	9 AUG	--	26 APR	24 JUL	--
25	533352	5053187	3 MAY	1 AUG	--	26 APR	24 JUL	19 NOV
8	534640	5055679	30 APR	30 JUL	--	23 APR	9 AUG	23 OCT
20	535896	5056684	3 MAY	26 JUL	--	11 APR	9 AUG	--
Straight								
28	533238	5051456	4 MAY	25 JUL	7 OCT	6 APR	25 JUL	25 OCT
23	533165	5051470	3 MAY	8 AUG	--	30 APR	1 AUG	--
11	533744	5053257	1 MAY	1 AUG	--	3 MAY	8 AUG	15 NOV
4	533507	5056045	23 APR	7 AUG	--	23 APR	25 JUL	24 OCT
9	534792	5055630	30 APR	2 AUG	--	25 APR	8 AUG	--
5	535975	5056731	23 APR	26 JUL	14 OCT	25 APR	8 AUG	9 NOV
Outside Bend								
22	533899	5049679	3 MAY	25 JUL	16 OCT	16 APR	25 JUL	12 NOV
30	533912	5049760	4 MAY	8 AUG	--	16 APR	25 JUL	--
13	533313	5050994	1 MAY	1 AUG	--	30 APR	1 AUG	8 OCT
21	533627	5050531	3 MAY	3 AUG	--	30 APR	1 AUG	19 NOV
29	533687	5050397	4 MAY	3 AUG	--	30 APR	1 AUG	11 NOV
35	533713	5052799	5 MAY	3 AUG	14 OCT	6 APR	8 AUG	12 NOV
Riprap								
24	533101	5052178	3 MAY	8 AUG	--	26 APR	24 JUL	21 NOV
31	533581	5053377	4 MAY	1 AUG	--	26 APR	24 JUL	25 OCT
7	534641	5055676	30 APR	2 AUG	--	23 APR	12 AUG	21 NOV
1	534863	5055150	23 APR	2 AUG	--	25 APR	9 AUG	--

	UTM coordinates		2001			2002		
Site	E	N	Spring	Summer	Fall	Spring	Summer	Fall
18	534863	5055718	3 MAY	26 JUL	7 OCT	25 APR	9 AUG	23 OCT
3	535702	5056191	23 APR	7 AUG	14 OCT	11 APR	9 AUG	23 OCT
Jetty								
12	533714	5054529	1 MAY	25 JUL	--	30 APR	8 AUG	25 OCT
26	533976	5054649	3 MAY	1 AUG	3 OCT	3 MAY	8 AUG	25 OCT
17	535352	5055940	3 MAY	2 AUG	--	11 APR	9 AUG	--
2	535370	5055955	23 APR	26 JUL	18 OCT	23 APR	9 AUG	23 OCT
36	533206	5050936	5 MAY	18 AUG	--	26 APR	25 JUL	--
32	533470	5053320	4 MAY	9 AUG	--	6 APR	24 JUL	19 NOV
Barb								
15	533747	5053257	1 MAY	8 AUG	14 OCT	3 MAY	25 JUL	12 NOV
19	535880	5056810	3 MAY	2 AUG	8 OCT	17 APR	12 AUG	9 NOV
10	535880	5056879	30 APR	7 AUG	--	17 APR	12 AUG	--
6	535882	5056940	30 APR	30 JUL	--	17 APR	12 AUG	--
34	535885	5057096	5 MAY	7 AUG	--	16 APR	12 AUG	9 NOV
33	535886	5057097	5 MAY	30 JUL	--	16 APR	12 AUG	9 NOV

Table 2. Locations, dates sampled, and areas of sampled side channels, and estimated numbers and densities of all juvenile trout and all juvenile salmonids (including mountain whitefish) therein, Yellowstone River, 2001 and 2002.

	UTM coordinates				Trout		Salmonids	
Site	E	N	Date	Area (m^2)	Number	Density	Number	Density
				2001				
A	533711	5037212	18 MAY	195	4	0.0205	12	0.0615
B	531441	5038101	18 MAY	178	0	0	1	0.0056
C	531444	5037902	18 MAY	178	1	0.0056	39	0.2191
D	536216	5057984	21 MAY	350	10	0.0286	21	0.0600
E	532853	5042954	31 MAY	175	3	0.0171	3	0.0171
F	532815	5042941	31 MAY	175	6	0.0343	6	0.0343
G	532850	5042956	31 MAY	144	2	0.0139	2	0.0139
H	532325	5038994	21 MAY	400	1	0.0025	7	0.0175
I	532343	5039084	21 MAY	300	1	0.0033	11	0.0367
J	533723	5037233	18 MAY	275	0	0	2	0.0073
K	534186	5053885	18 MAY	450	5	0.0111	30	0.0667
				2002				
A	533711	5037212	5 JUN	215	3	0.0140	6	0.0279
C	531444	5037902	1 JUL	178	3	0.0168	5	0.0281
D	536216	5057984	2 JUL	338	14	0.0414	18	0.0532
I	532343	5039084	14 JUN	658	0	0	4	0.0061
K	534186	5053885	3 JUN	180	0	0	16	0.0889
N	531903	5038356	1 JUL	215	2	0.0093	5	0.0232
O	533136	5050479	2 JUN	337	10	0.0297	11	0.0326
P	532487	5042733	5 JUN	95	3	0.0316	3	0.0316
Q	533470	5050414	13 JUN	97	13	0.1340	39	0.4021
R	532717	5041344	2 JUN	362	7	0.0193	8	0.0221
S	532486	5042739	4 JUN	320	0	0	3	0.0094
U	535861	5057343	4 JUN	108	8	0.0741	8	0.0741
V	534202	5055386	20 JUN	112	8	0.0714	13	0.1161
X	535870	5056717	4 JUN	120	6	0.0500	8	0.0667
Z	533311	5050527	2 JUL	365	10	0.0274	20	0.0548

Table 3. Numbers of rainbow trout captured by single-pass electrofishing at 50-m sample sites along specific bank types, Yellowstone River, 2001-2002.

Bank type	N	Mean	Standard deviation	Standard error	95% confidence interval	Range
Inside bend	78	0.769	1.528	0.173	0.425-1.114	0-10
Straight	78	3.359	5.730	0.649	2.067-4.651	0-30
Outside bend	79	5.684	8.727	0.982	3.729-7.638	0-48
Riprap	79	8.304	6.699	0.754	6.803-9.804	0-27
Barb	78	4.974	6.457	0.731	3.519-6.430	0-46
Jetty	78	7.692	11.835	1.340	5.024-10.361	0-90

Table 4. Numbers of brown trout captured by single-pass electrofishing at 50-m sample sites along specific bank types, Yellowstone River, 2001-2002.

Bank type	N	Mean	Standard deviation	Standard error	95% confidence interval	Range
			Reach 1			
Inside bend	48	0.354	0.729	0.105	0.142-0.566	0-3
Straight	48	1.229	2.055	0.297	0.632-1.826	0-8
Outside bend	48	2.313	2.528	0.365	1.579-3.046	0-11
Riprap	48	3.625	4.077	0.588	2.441-4.809	0-19
Barb	48	1.896	2.354	0.340	1.212-2.579	0-9
Jetty	48	3.250	2.678	0.386	2.472-4.028	0-11
			Reach 2			
Inside bend	30	0.133	0.346	0.063	0.004-0.262	0-1
Straight	30	2.233	3.461	0.632	0.941-3.526	0-16
Outside bend	31	0.774	1.230	0.221	0.323-1.226	0-4
Riprap	31	3.774	5.690	1.022	1.687-5.861	0-31
Barb	30	2.333	2.202	0.402	1.511-3.156	0-8
Jetty	30	1.400	1.958	0.358	0.669-2.131	0-7

Table 5. Combined numbers of rainbow, brown, Yellowstone cutthroat and brook trout captured by single-pass electrofishing at 50-m sample sites along specific bank types, Yellowstone River, 2001-2002.

Bank type	N	Mean	Standard deviation	Standard error	95% confidence interval	Range
Inside bend	78	1.038	1.717	0.194	0.651-1.425	0-10
Straight	78	5.103	7.545	0.854	3.401-6.804	0-37
Outside bend	79	7.747	10.068	1.133	5.492-10.002	0-52
Riprap	79	12.203	7.905	0.889	10.432-13.973	0-34
Barb	78	7.436	7.702	0.872	5.699-9.172	0-49
Jetty	78	10.449	12.338	1.397	7.667-13.230	0-93

Table 6. Combined numbers of all salmonids (all trout and mountain whitefish) captured by single-pass electrofishing at 50-m sample sites along specific bank types, Yellowstone River, 2001-2002.

Bank type	N	Mean	Standard deviation	Standard error	95% confidence interval	Range
Inside bend	78	1.538	2.542	0.288	0.965-2.111	0-13
Straight	78	5.423	7.657	0.867	3.697-7.150	0-37
Outside bend	79	8.443	9.946	1.119	6.215-10.671	0-52
Riprap	79	12.215	7.924	0.891	10.440-13.990	0-34
Barb	78	7.923	7.561	0.856	6.218-9.628	0-49
Jetty	78	10.590	12.370	1.401	7.801-13.379	0-94

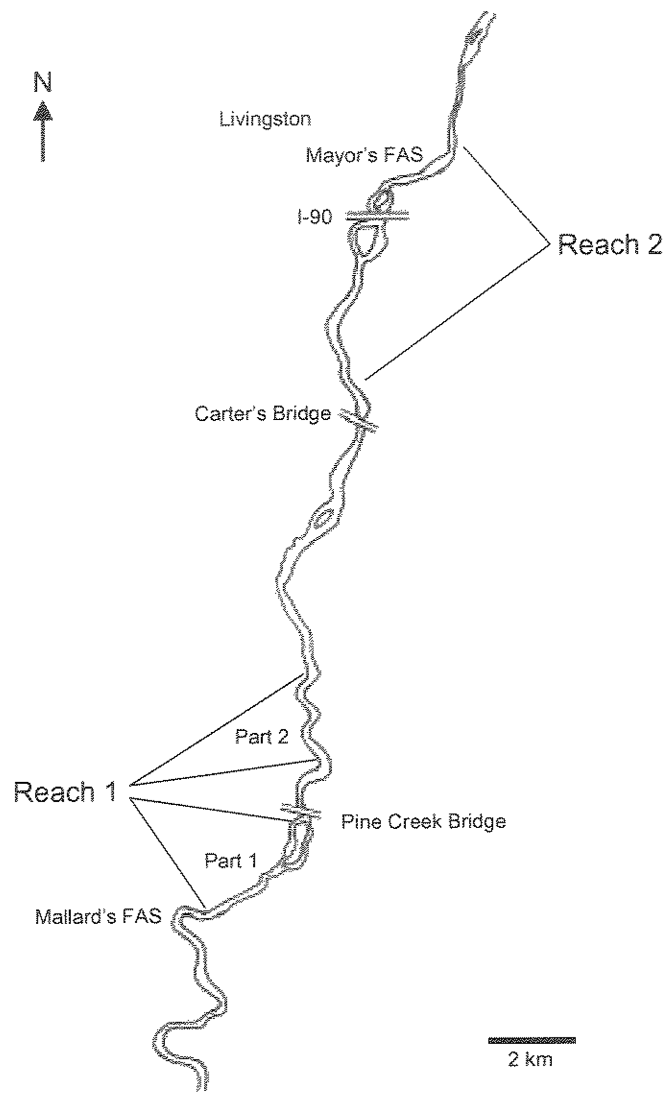

Figure 1. Study Reaches 1 and 2, upper Yellowstone River, 2001 and 2002.

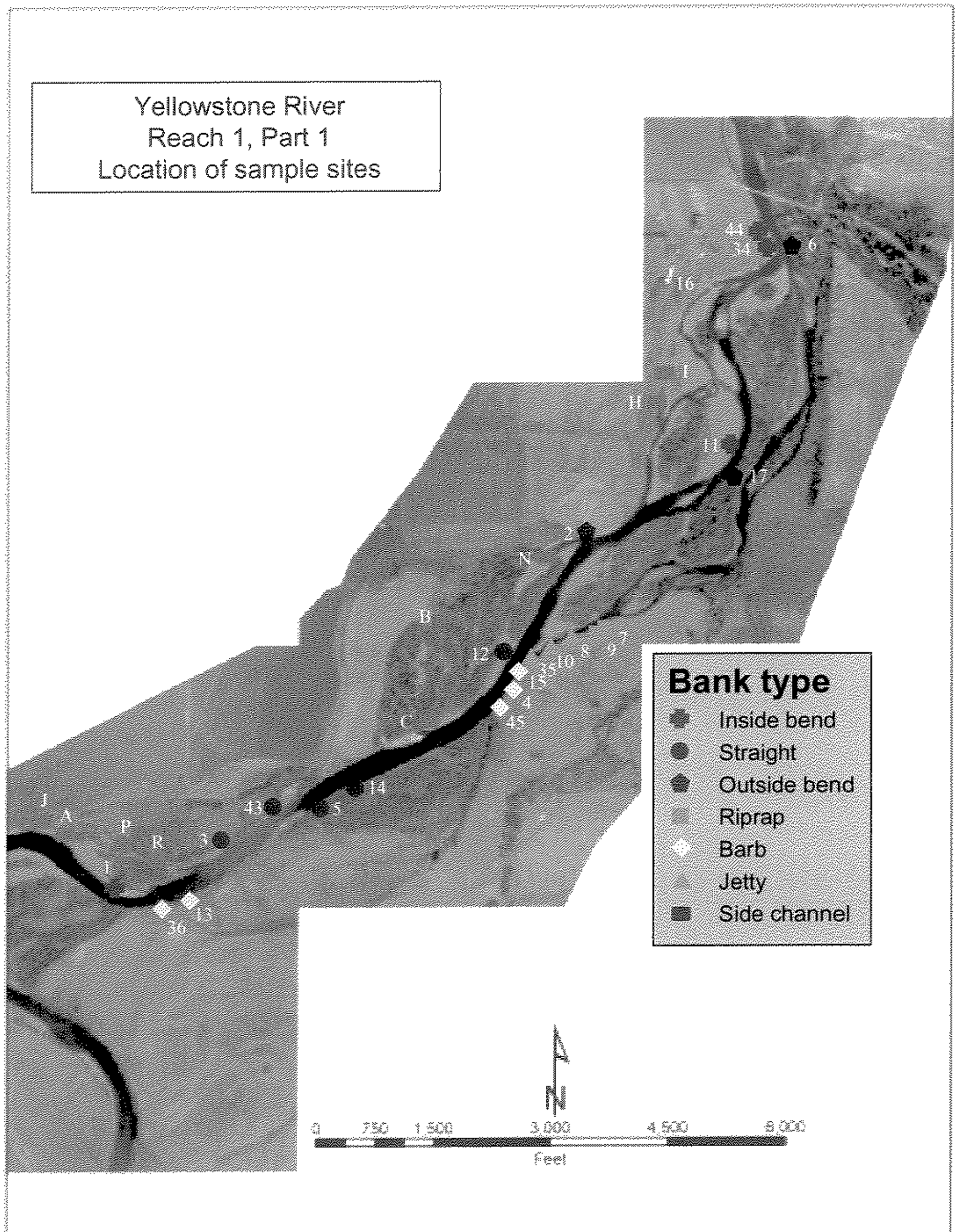

Figure 2. Approximate locations of Reach 1, Part 1 sample sites by bank type, Yellowstone River, 2001 and 2002. Site numbers and letters correspond to those listed in Tables 1 and 2, respectively.

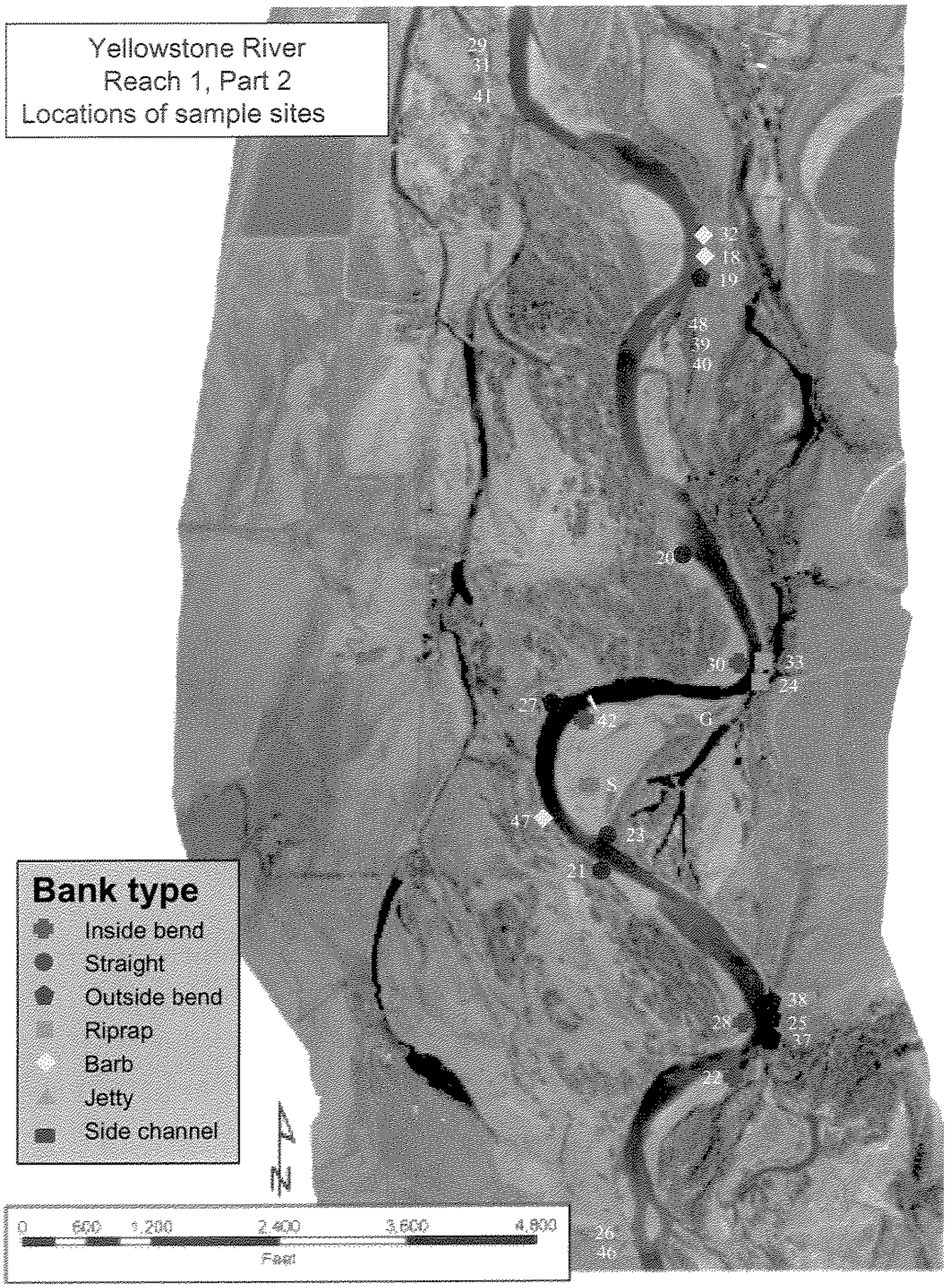

Figure 3. Approximate locations of Reach 1, Part 2 sample sites by bank type, Yellowstone River, 2001 and 2002. Site numbers and letters correspond to those listed in Tables 1 and 2, respectively.

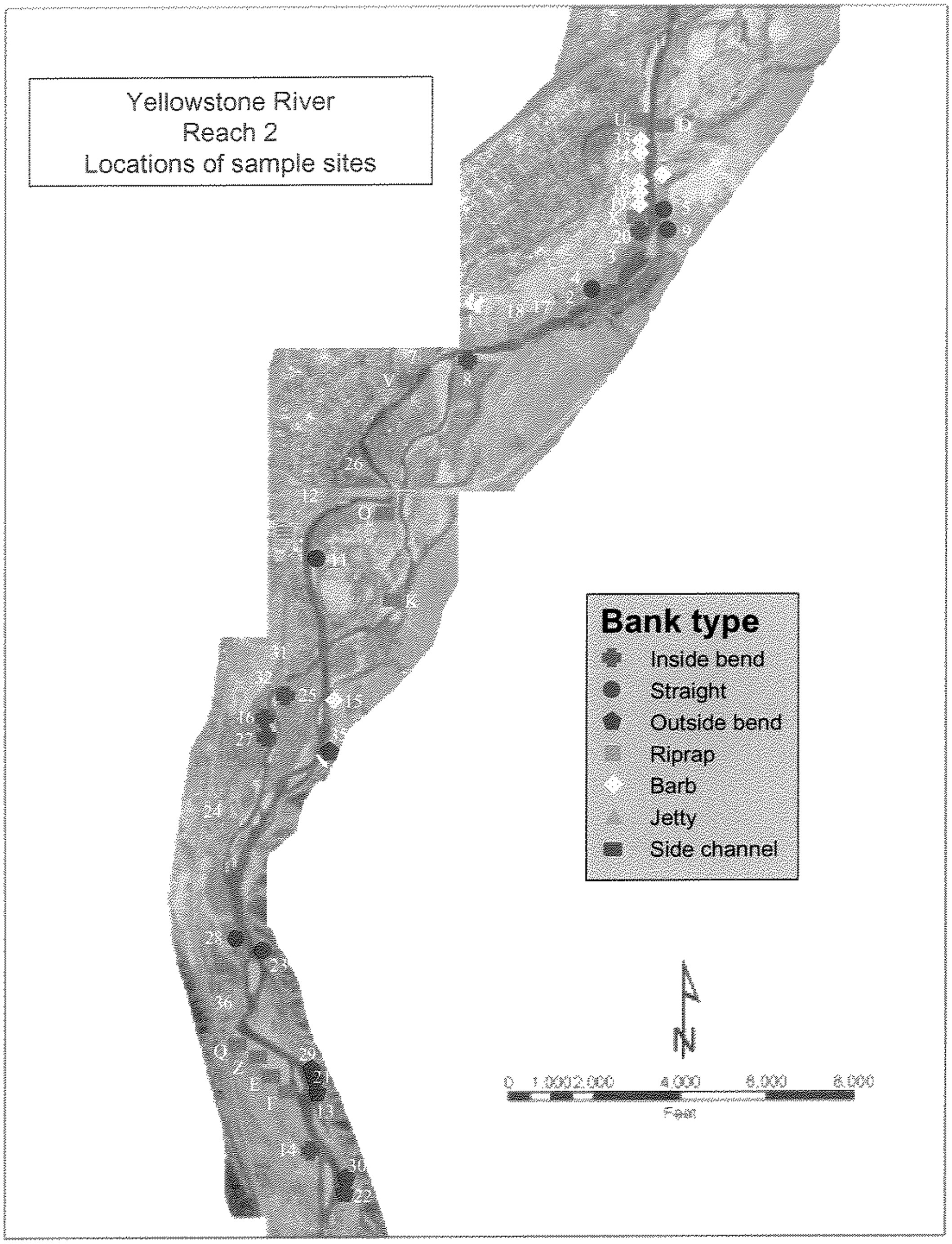

Figure 4. Approximate locations of Reach 2 sample sites by bank type, Yellowstone River, 2001 and 2002. Site numbers and letters correspond to those listed in Tables 1 and 2, respectively.

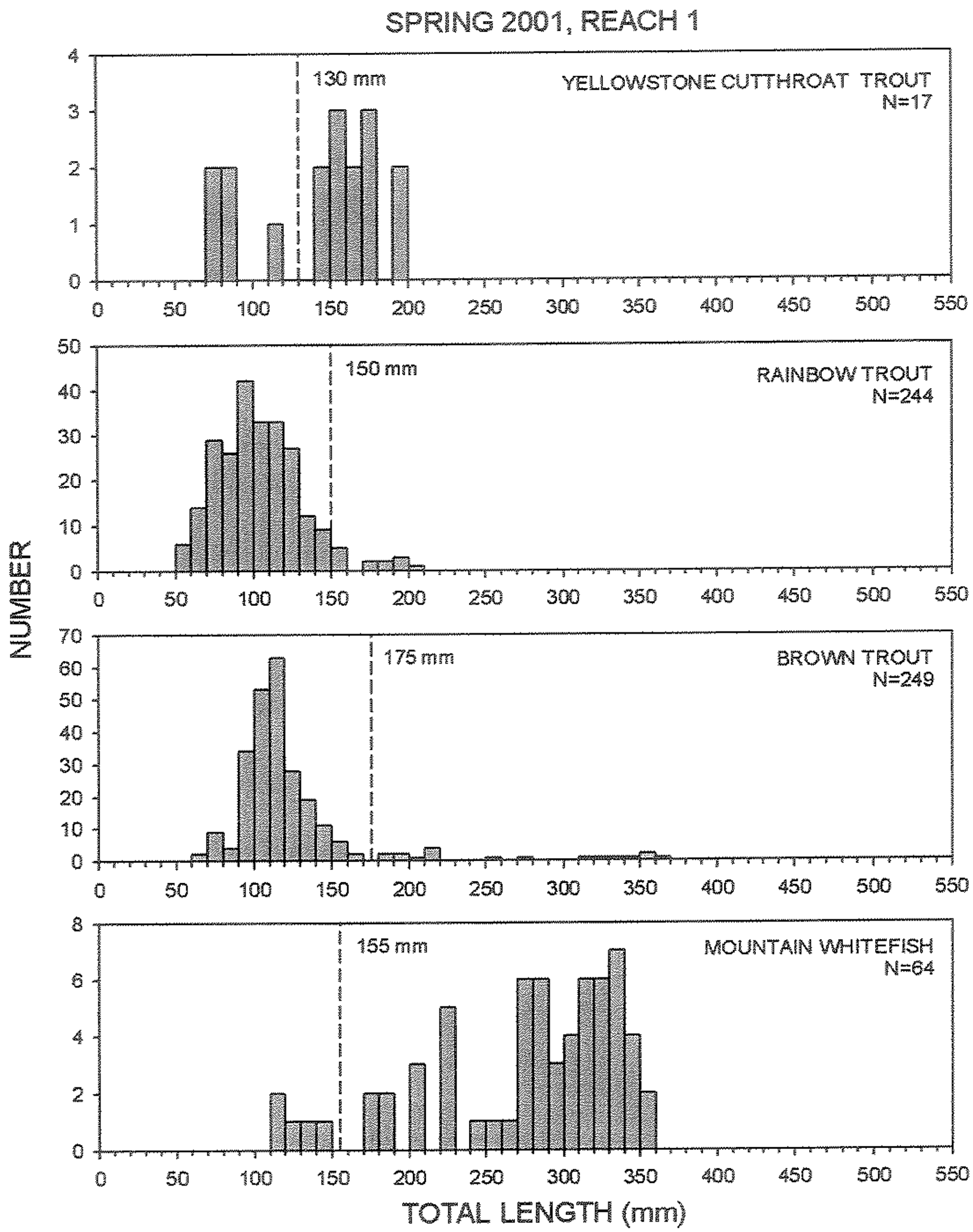

Figure 5. Length-frequency distributions, Spring 2001, Reach 1, Yellowstone River. Dashed vertical lines indicate maximum lengths of fish considered juveniles.

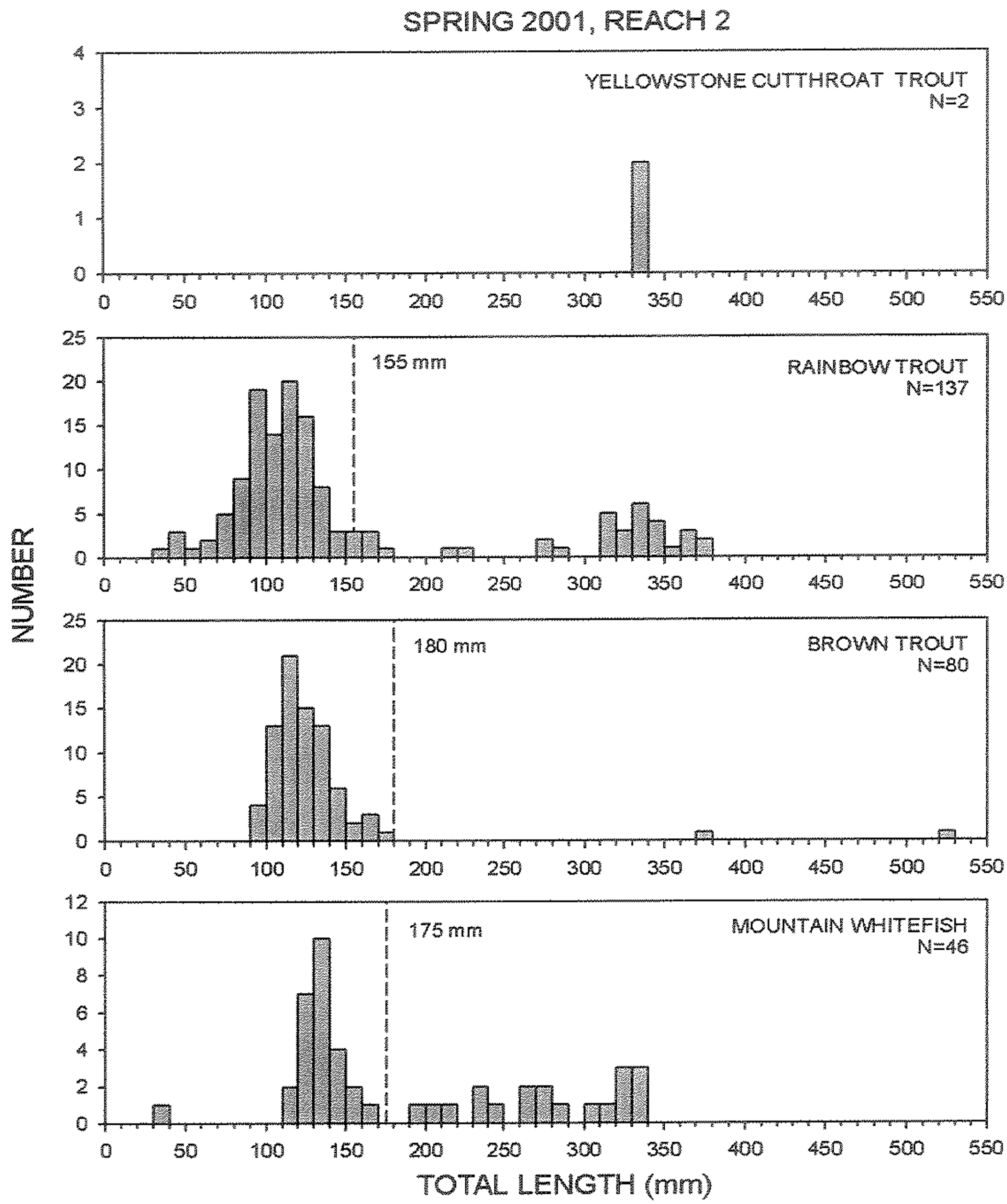

Figure 6. Length-frequency distributions, Spring 2001, Reach 2, Yellowstone River. Dashed vertical lines indicate maximum lengths of fish considered juveniles.

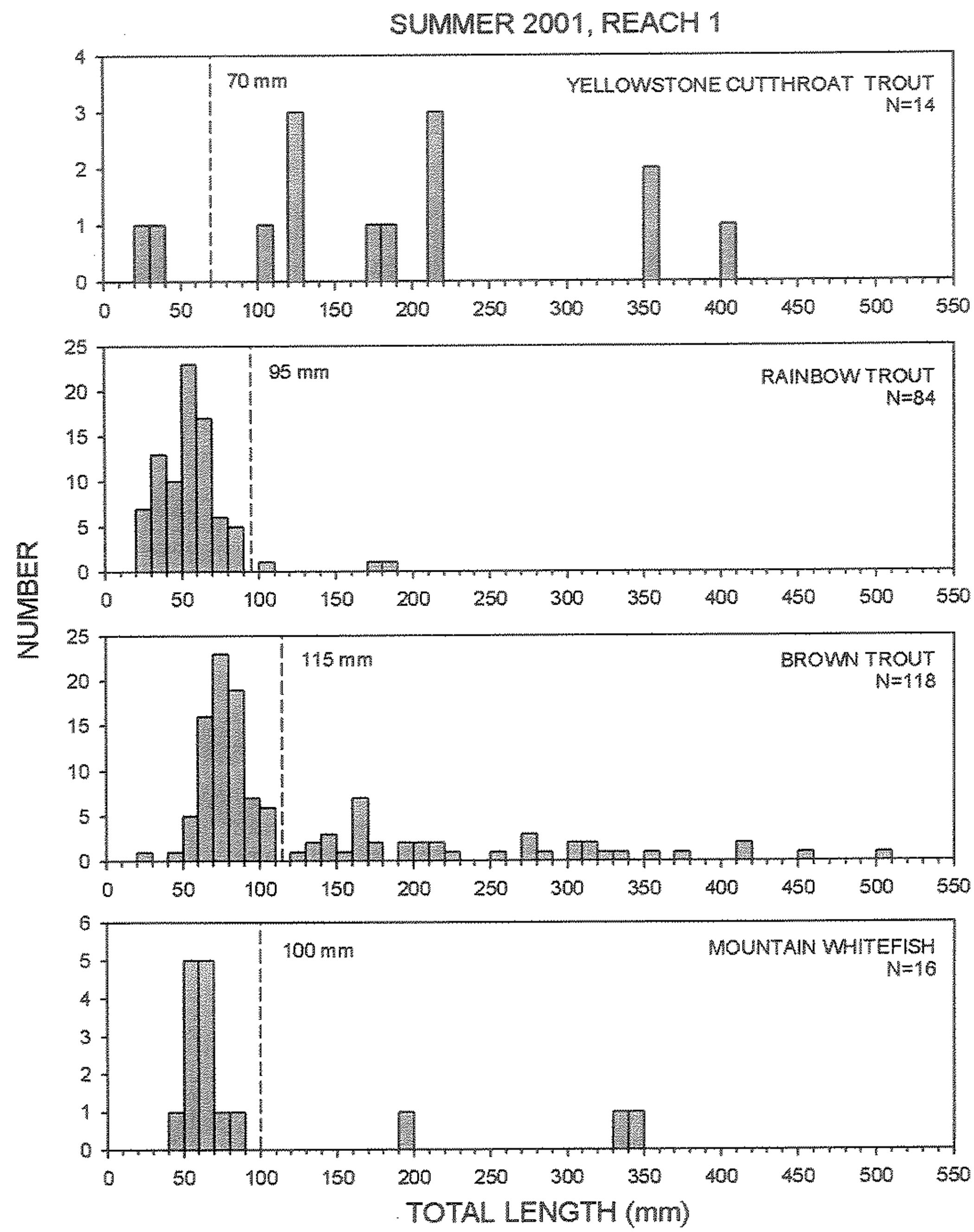

Figure 7. Length-frequency distributions, Summer 2001, Reach 1, Yellowstone River. Dashed vertical lines indicate maximum lengths of fish considered juveniles.

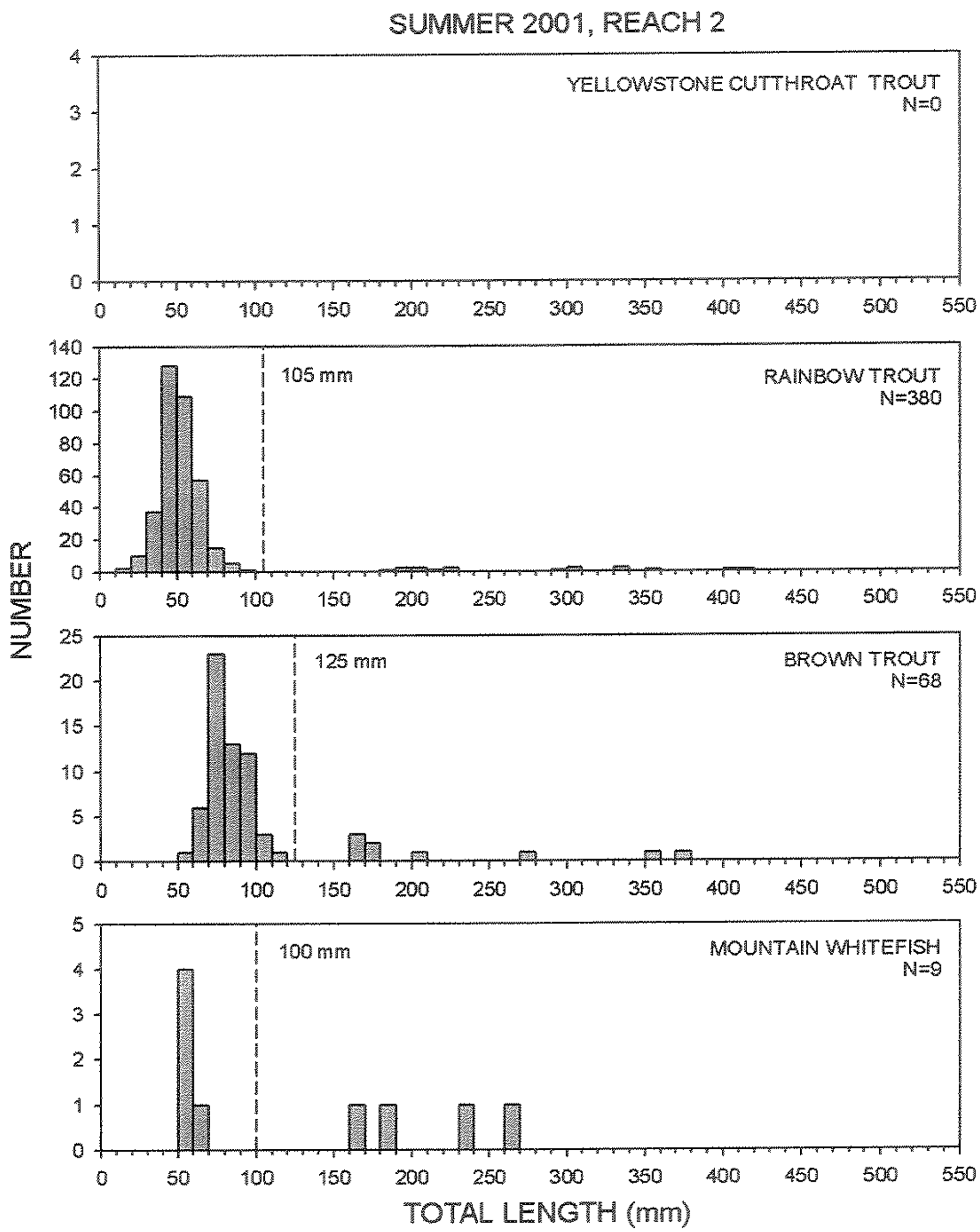

Figure 8. Length-frequency distributions, Summer 2001, Reach 2, Yellowstone River. Dashed vertical lines indicate maximum lengths of fish considered juveniles.

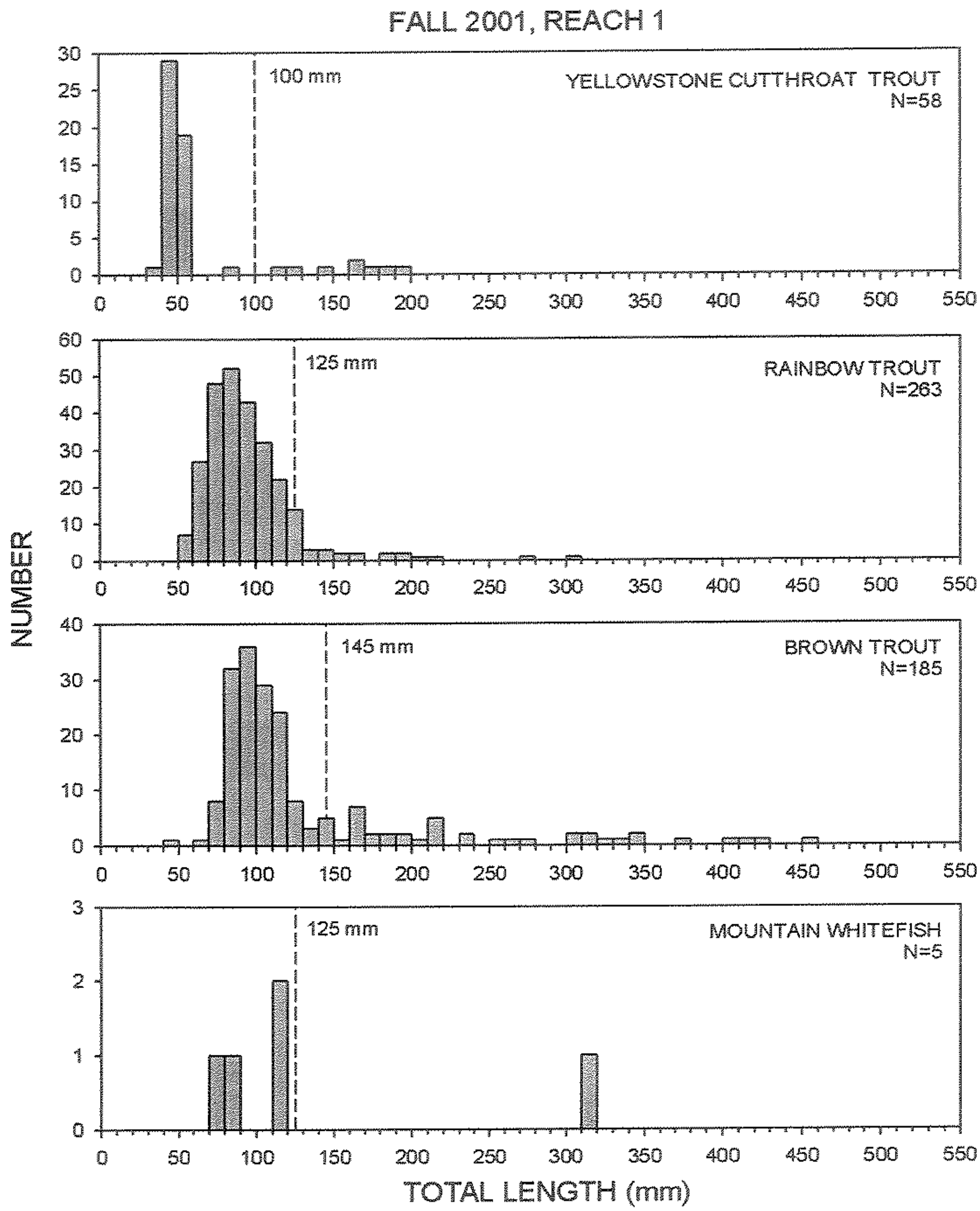

Figure 9. Length-frequency distributions, Fall 2001, Reach 1, Yellowstone River. Dashed vertical lines indicate maximum lengths of fish considered juveniles.

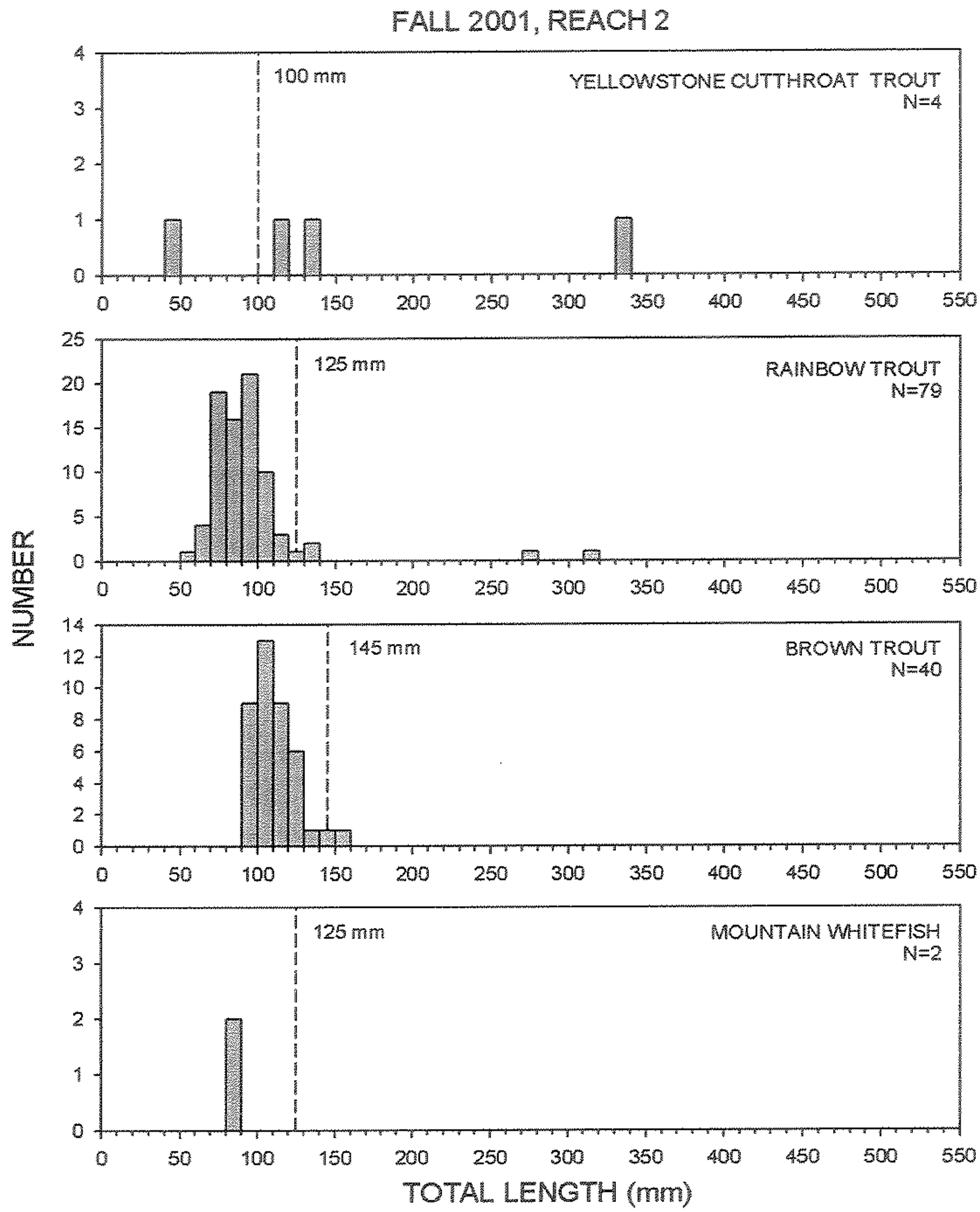

Figure 10. Length-frequency distributions, Fall 2001, Reach 2, Yellowstone River. Dashed vertical lines indicate maximum lengths of fish considered juveniles.

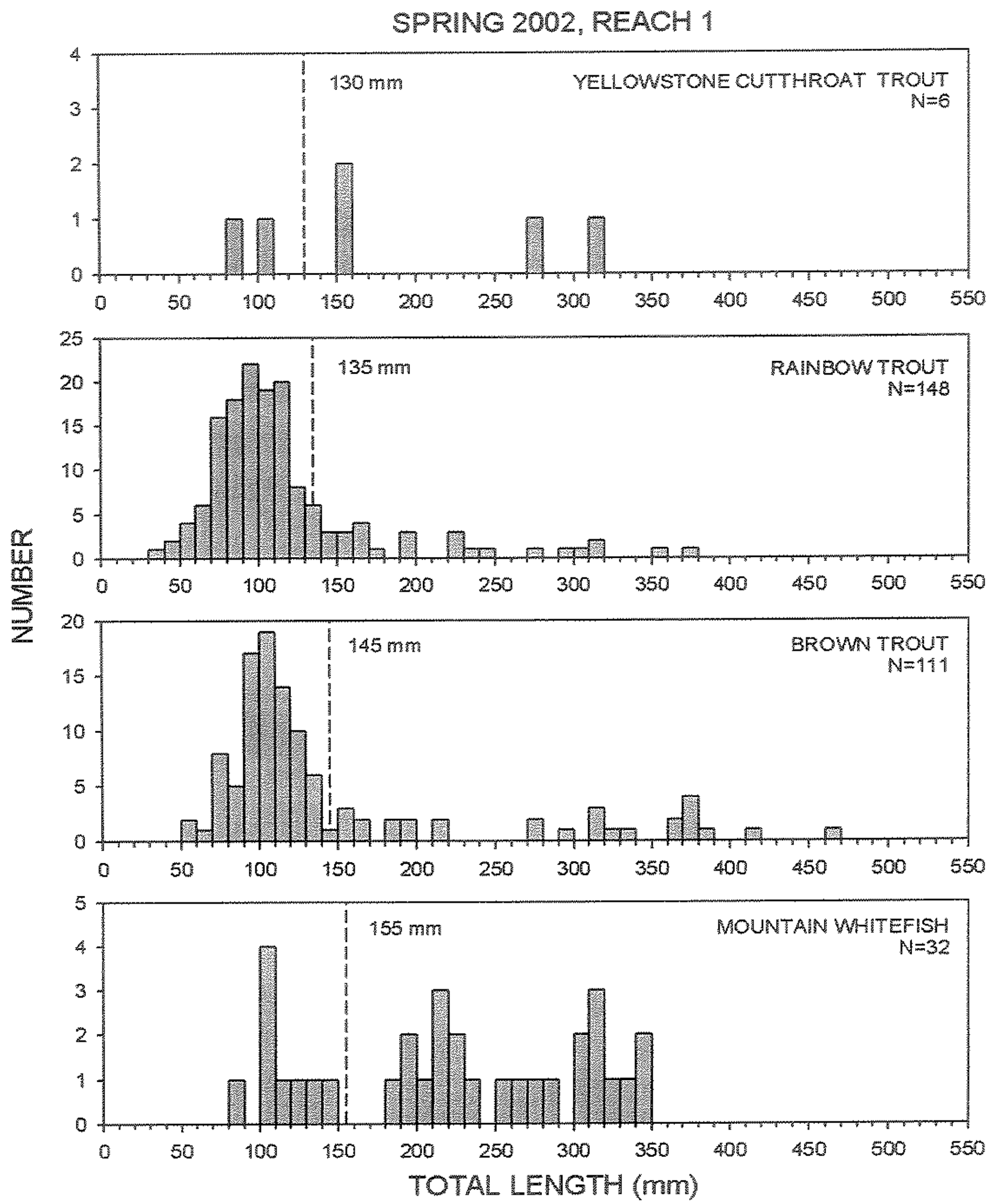

Figure 11. Length-frequency distributions, Spring 2002, Reach 1, Yellowstone River. Dashed vertical lines indicate maximum lengths of fish considered juveniles.

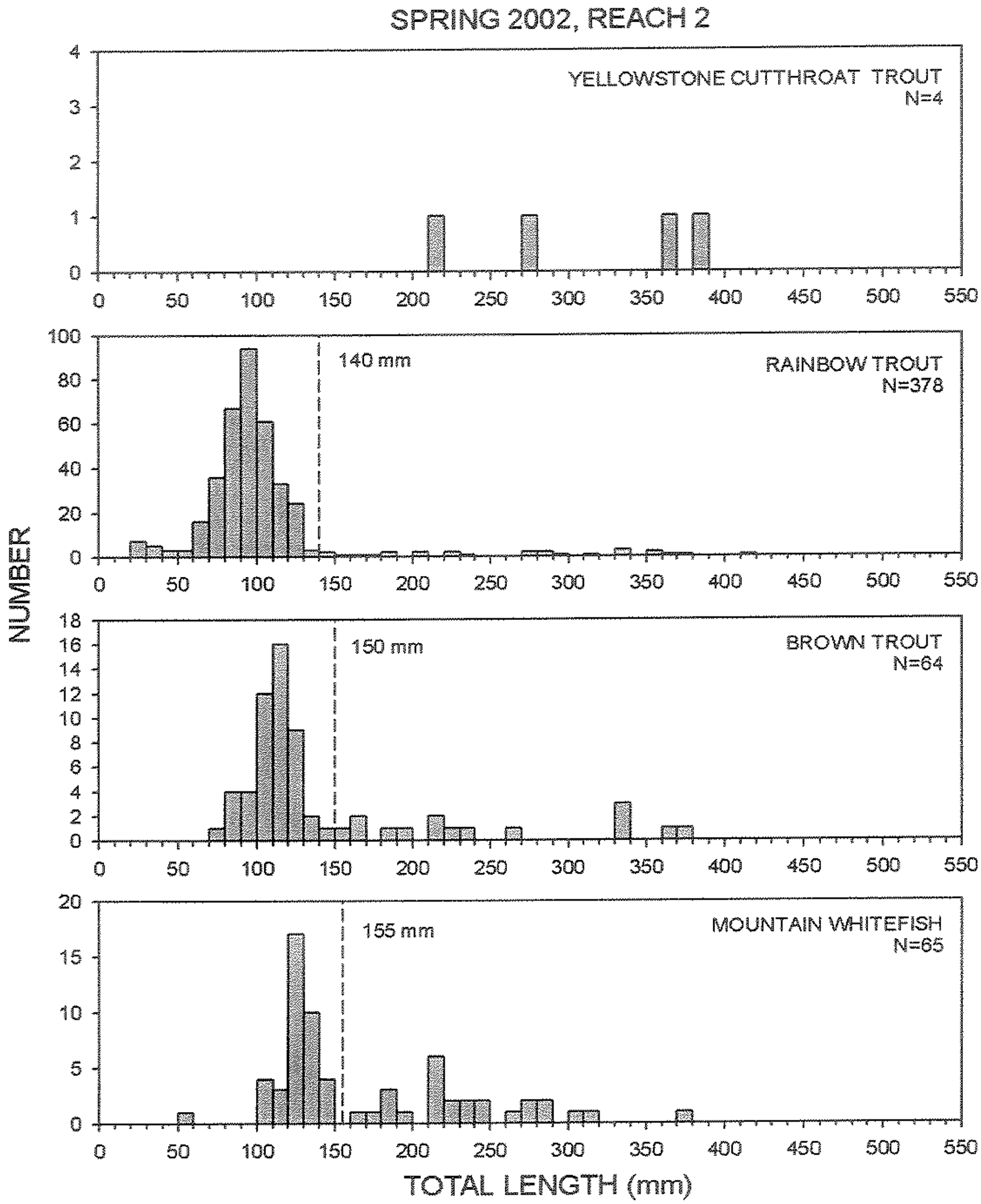

Figure 12. Length-frequency distributions, Spring 2002, Reach 2, Yellowstone River. Dashed vertical lines indicate maximum lengths of fish considered juveniles.

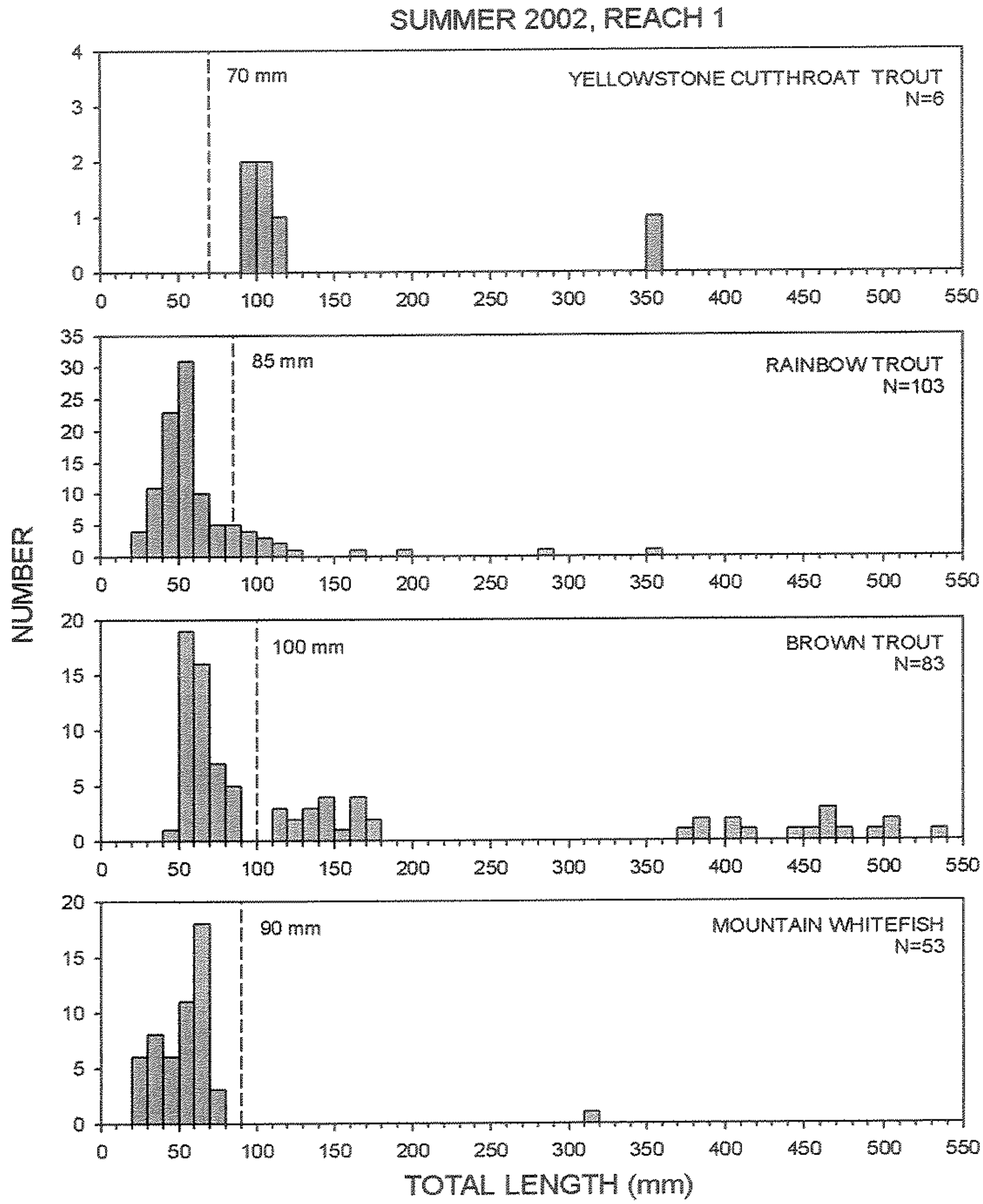

Figure 13. Length-frequency distributions, Summer 2002, Reach 1, Yellowstone River. Dashed vertical lines indicate maximum lengths of fish considered juveniles.

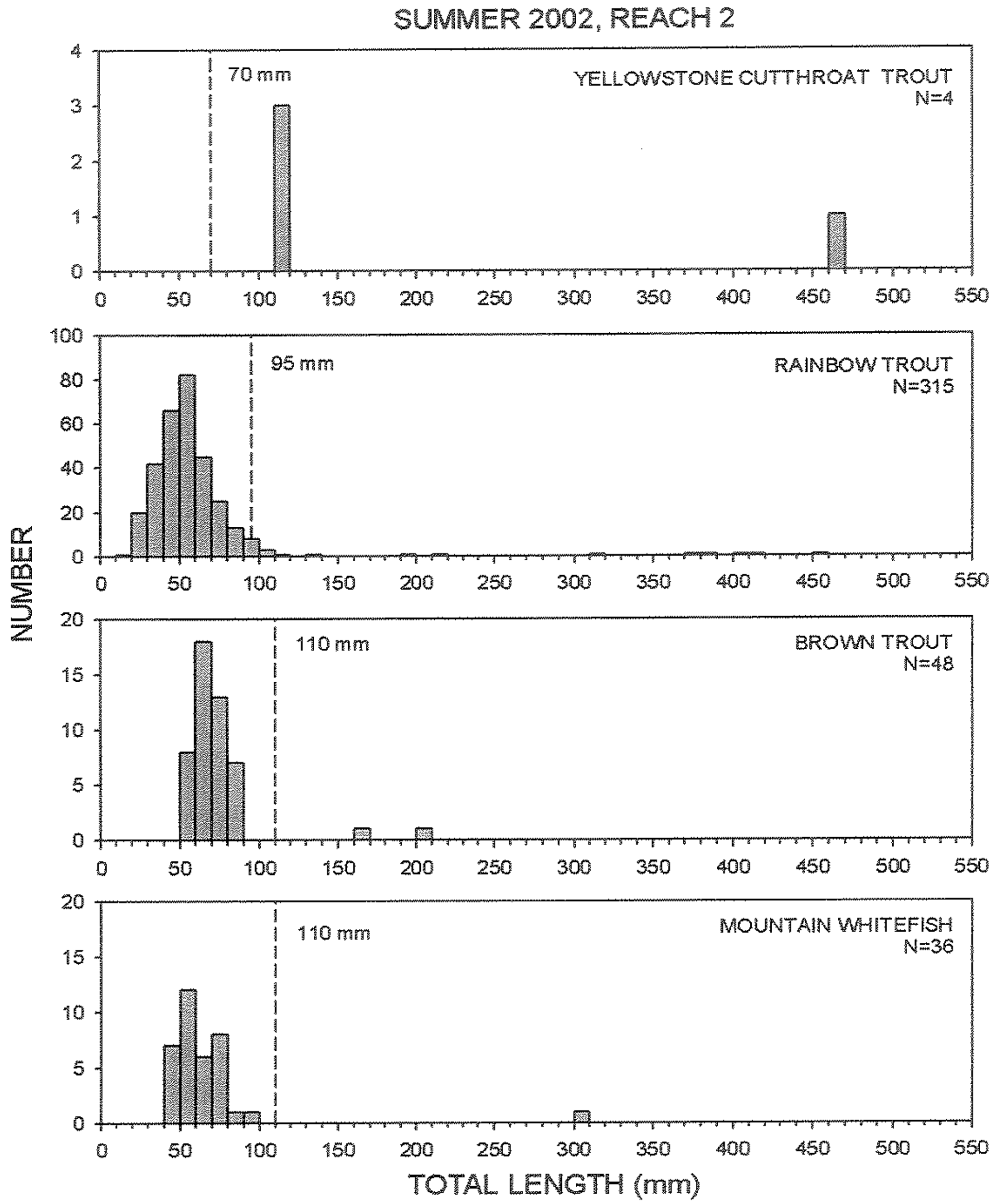

Figure 14. Length-frequency distributions, Summer 2002, Reach 2, Yellowstone River. Dashed vertical lines indicate maximum lengths of fish considered juveniles.

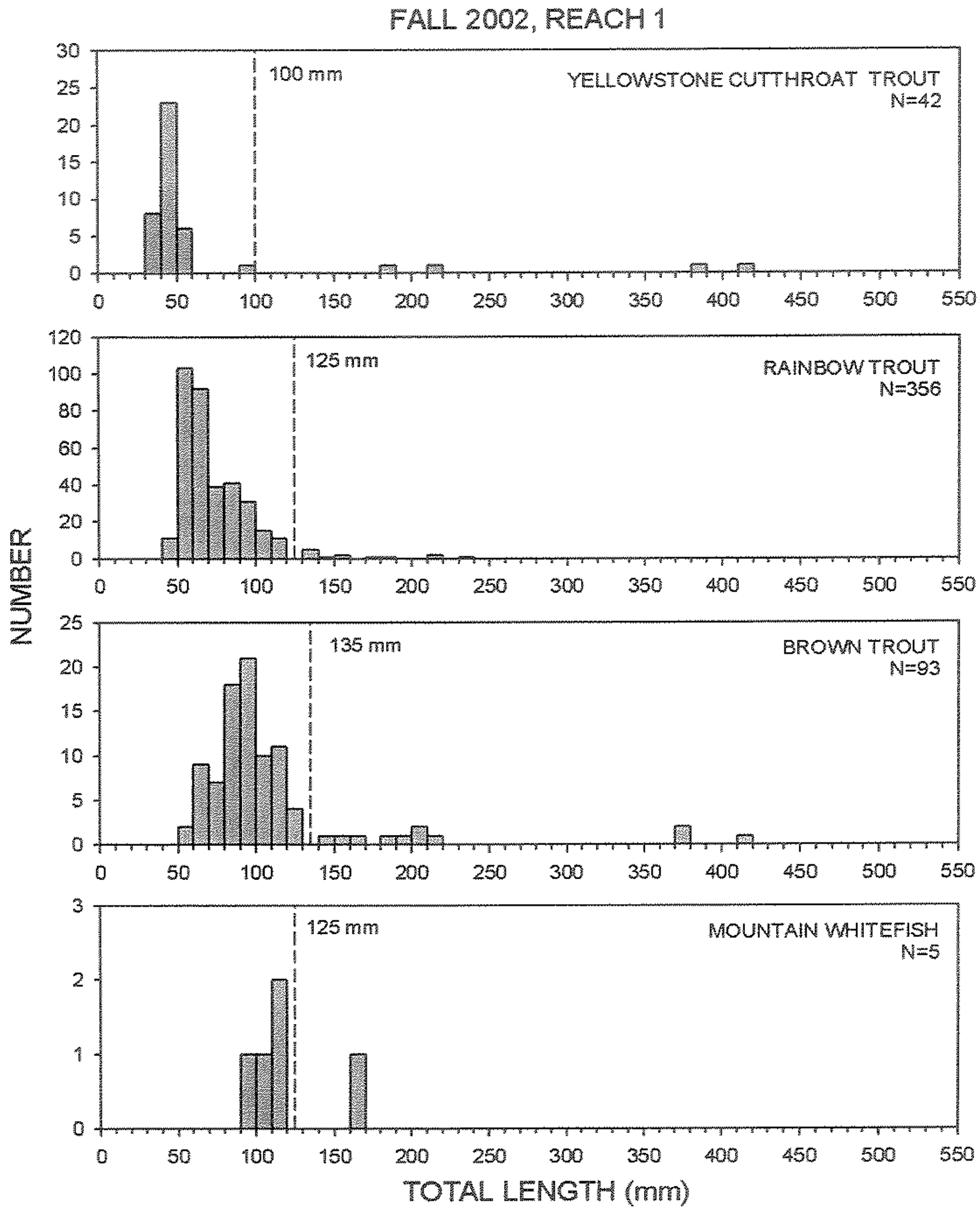

Figure 15. Length-frequency distributions, Fall 2002, Reach 1, Yellowstone River. Dashed vertical lines indicate maximum lengths of fish considered juveniles.

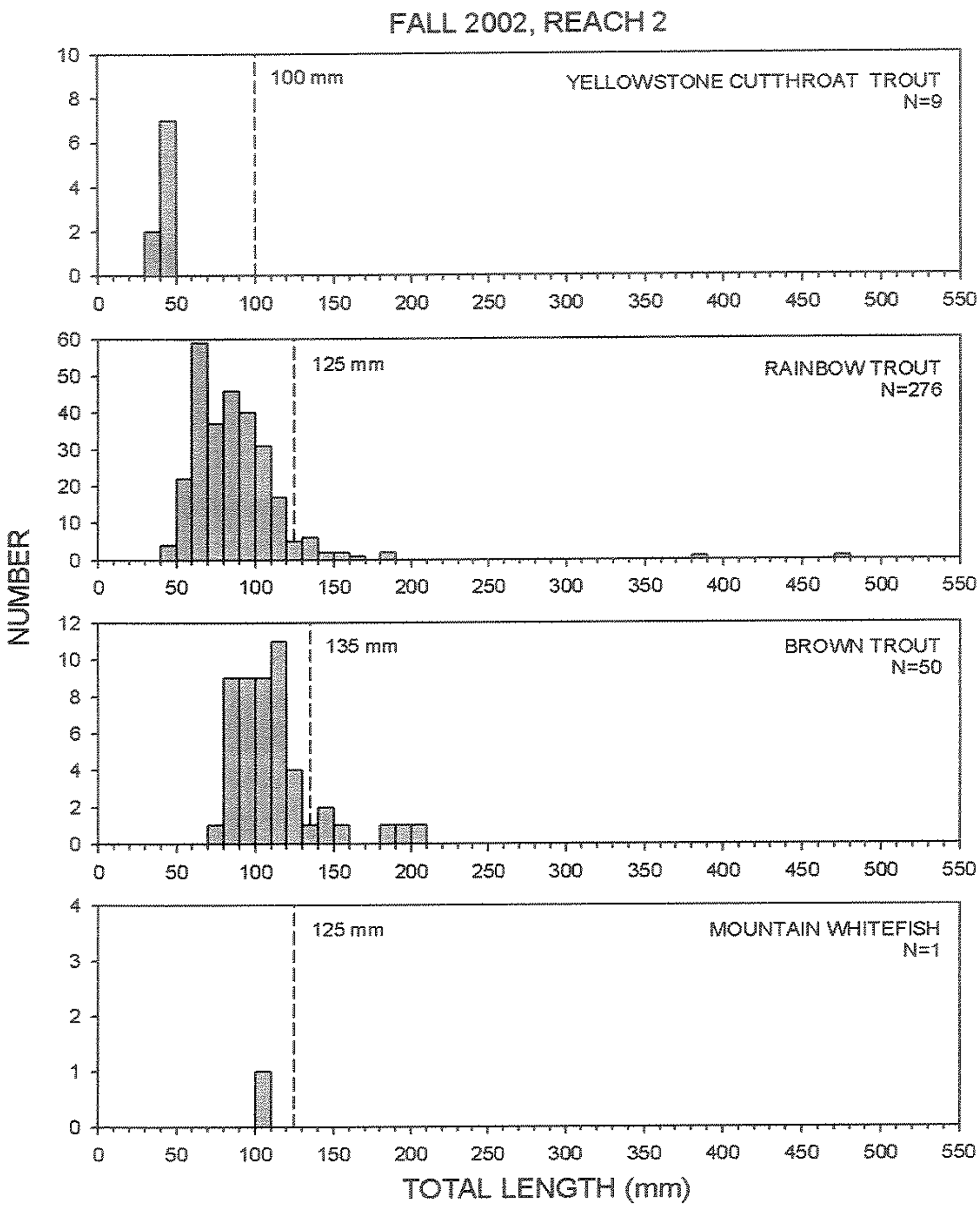

Figure 16. Length-frequency distributions, Fall 2002, Reach 2, Yellowstone River. Dashed vertical lines indicate maximum lengths of fish considered juveniles.

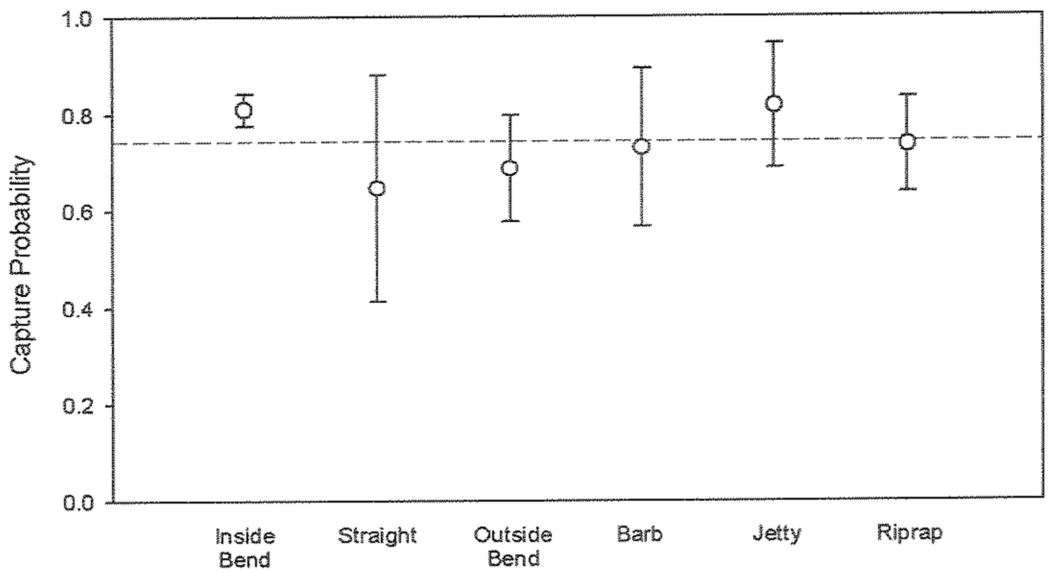

Figure 17. Mean capture probabilities of juvenile salmonids by bank type, Yellowstone River. Error bars represent ±1 SD. The dashed horizontal line indicates the overall mean of 0.743.

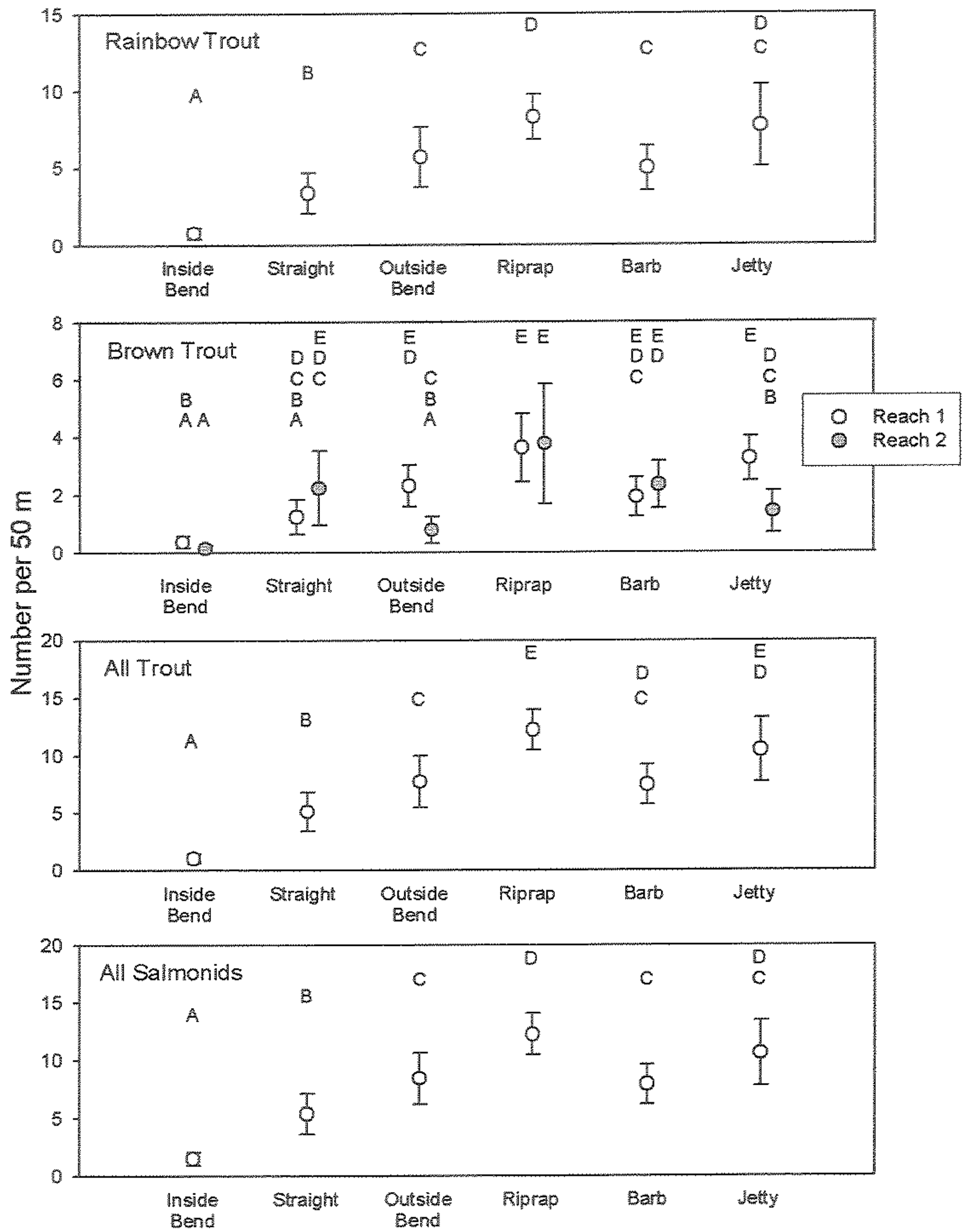

Figure 18. Mean numbers of juvenile salmonids captured by one-pass electrofishing by bank type, 2001 and 2002, Yellowstone River. Error bars represent 95% confidence intervals. Means with the same letter are not significantly different.

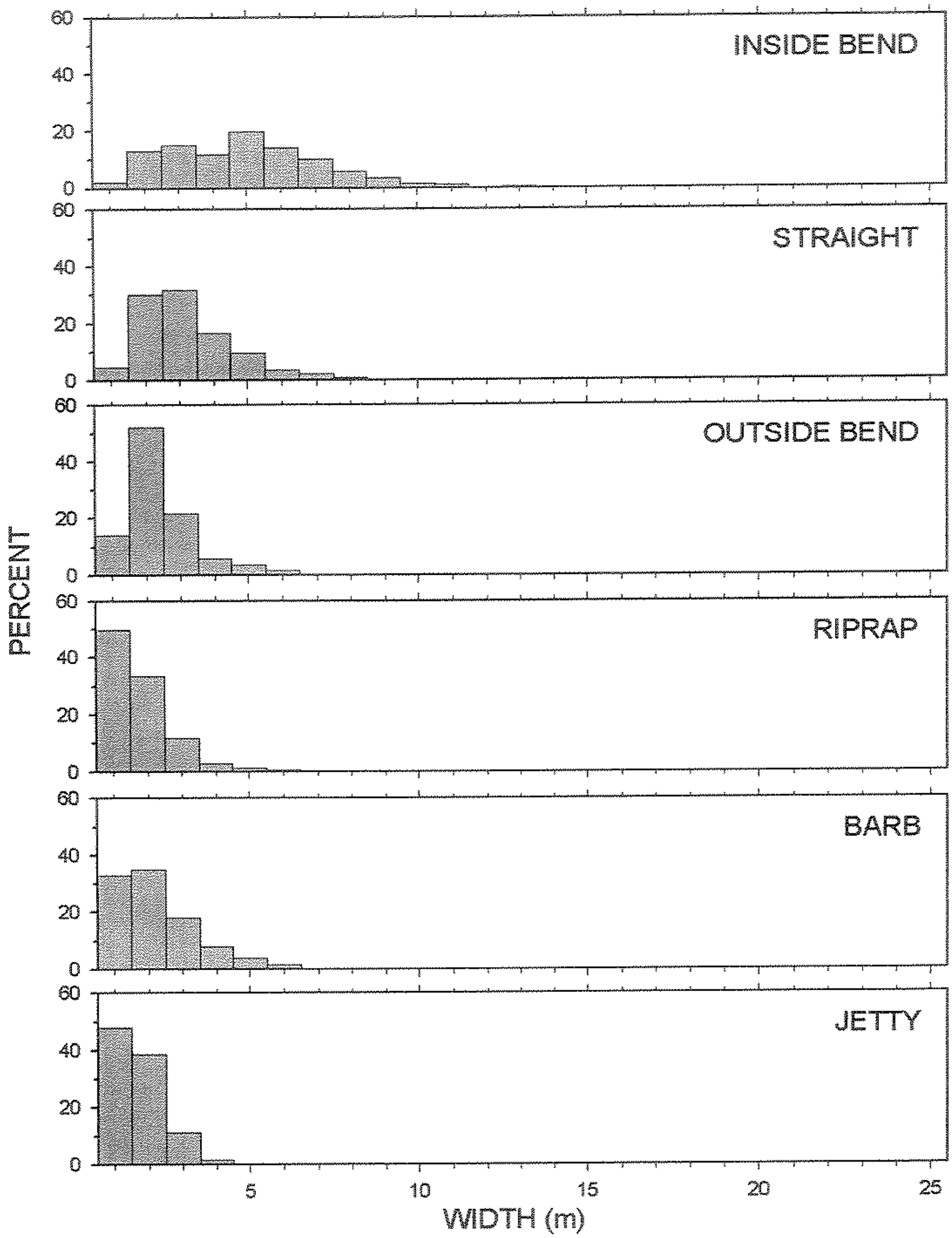

Figure 19. Frequency distributions of transect widths by bank type, Yellowstone River.

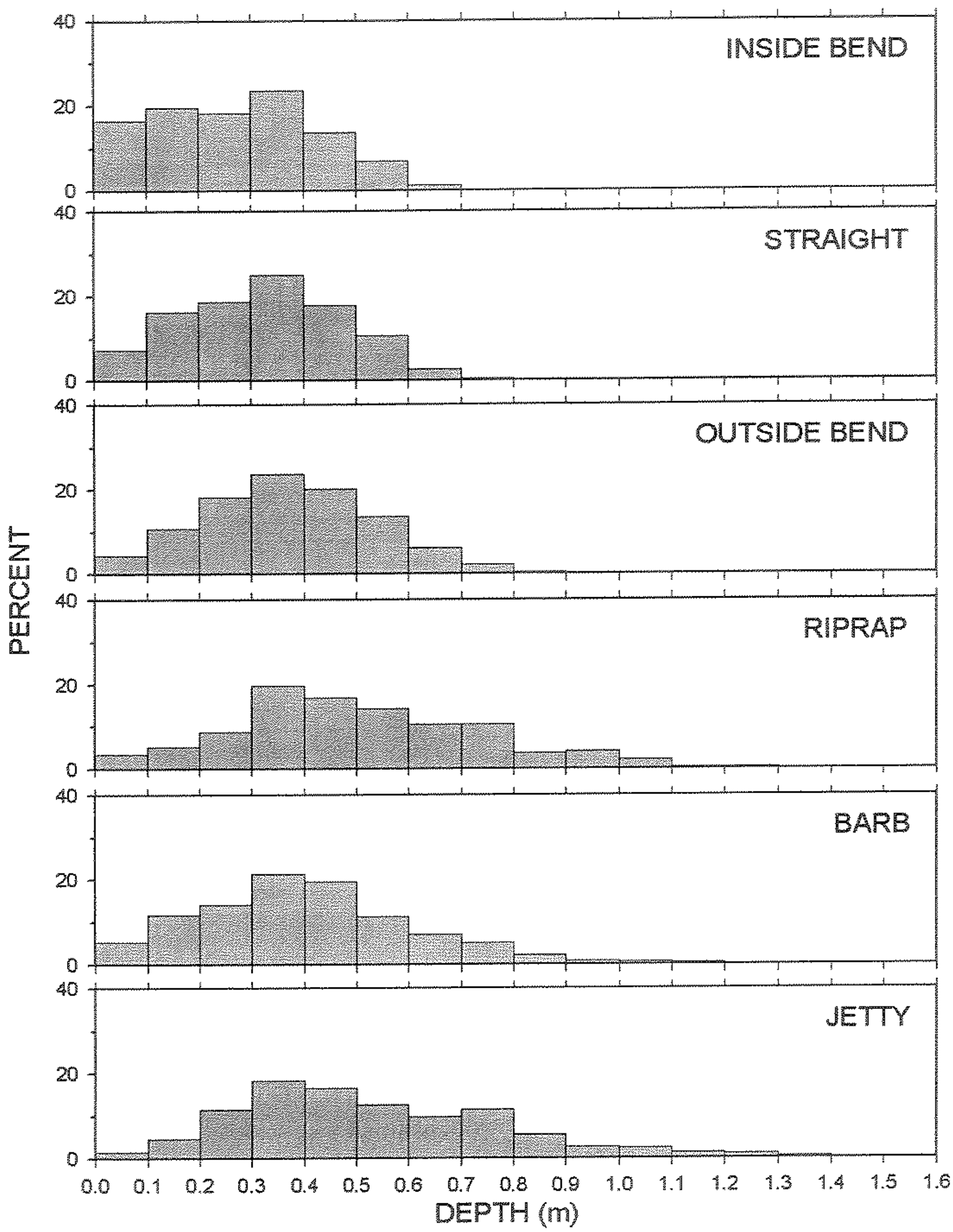

Figure 20. Frequency distributions of depths along transects by bank type, Yellowstone River.

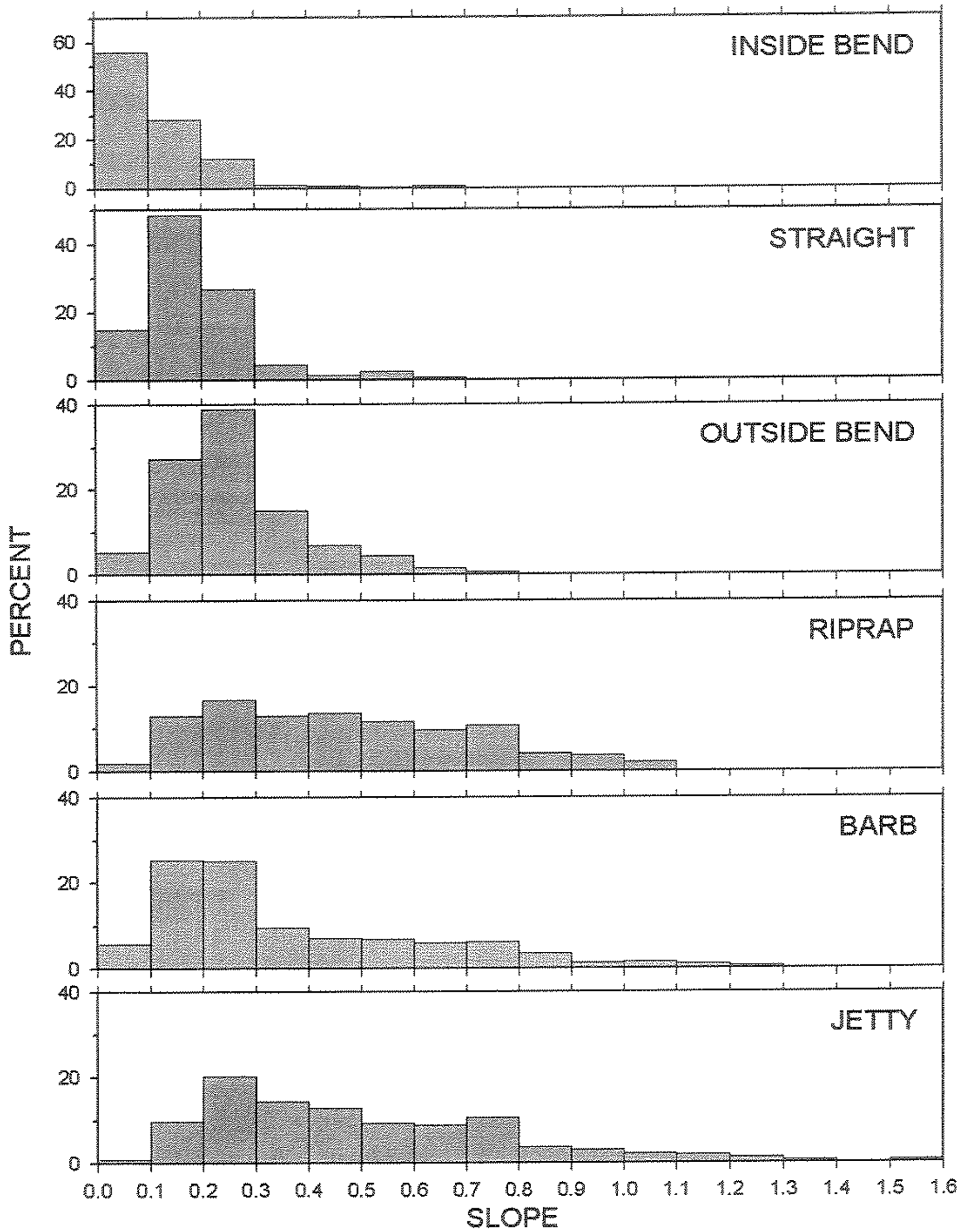

Figure 21. Frequency distributions of transect slopes by bank type, Yellowstone River.

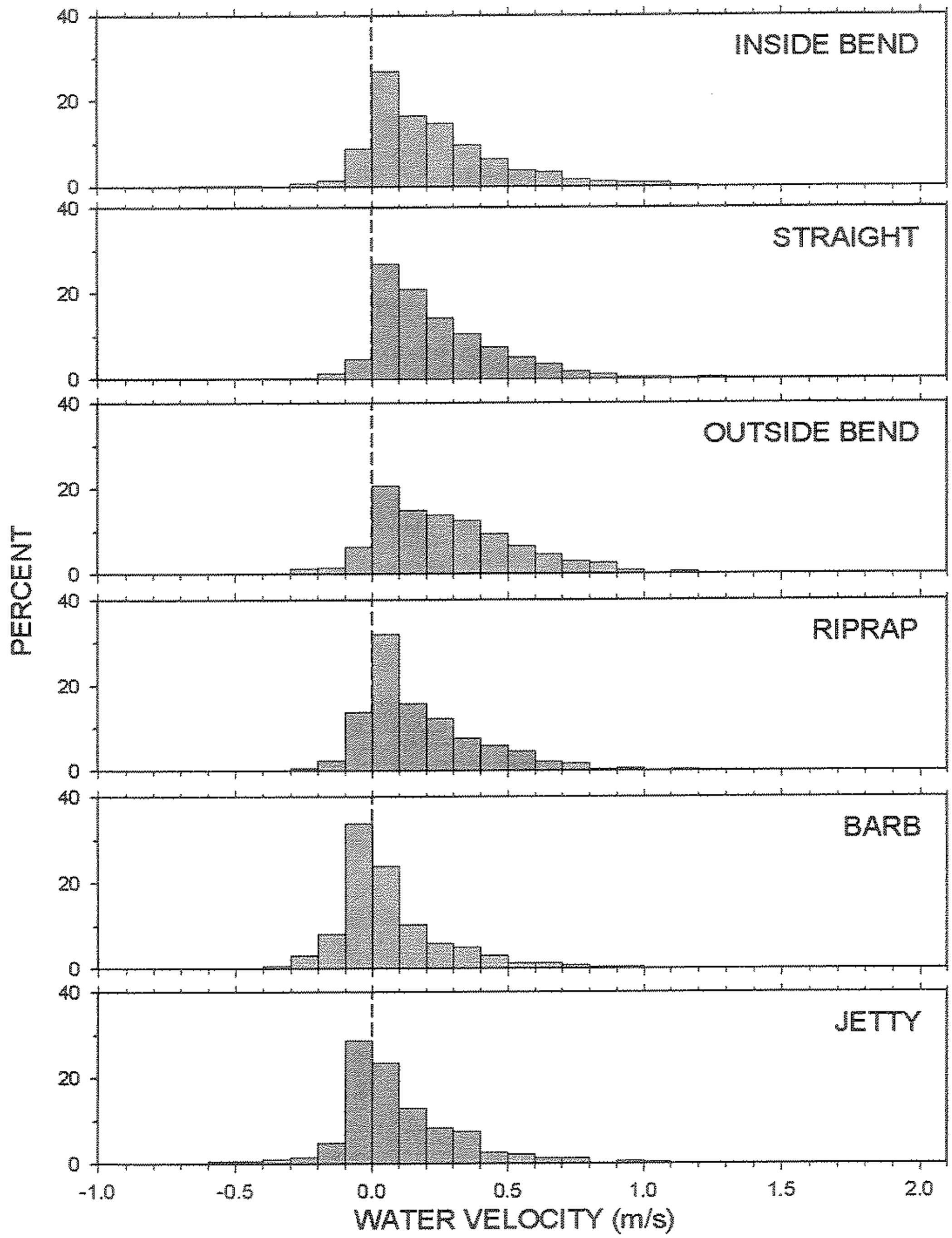

Figure 22. Frequency distributions of water velocities along transects by bank type, Yellowstone River.

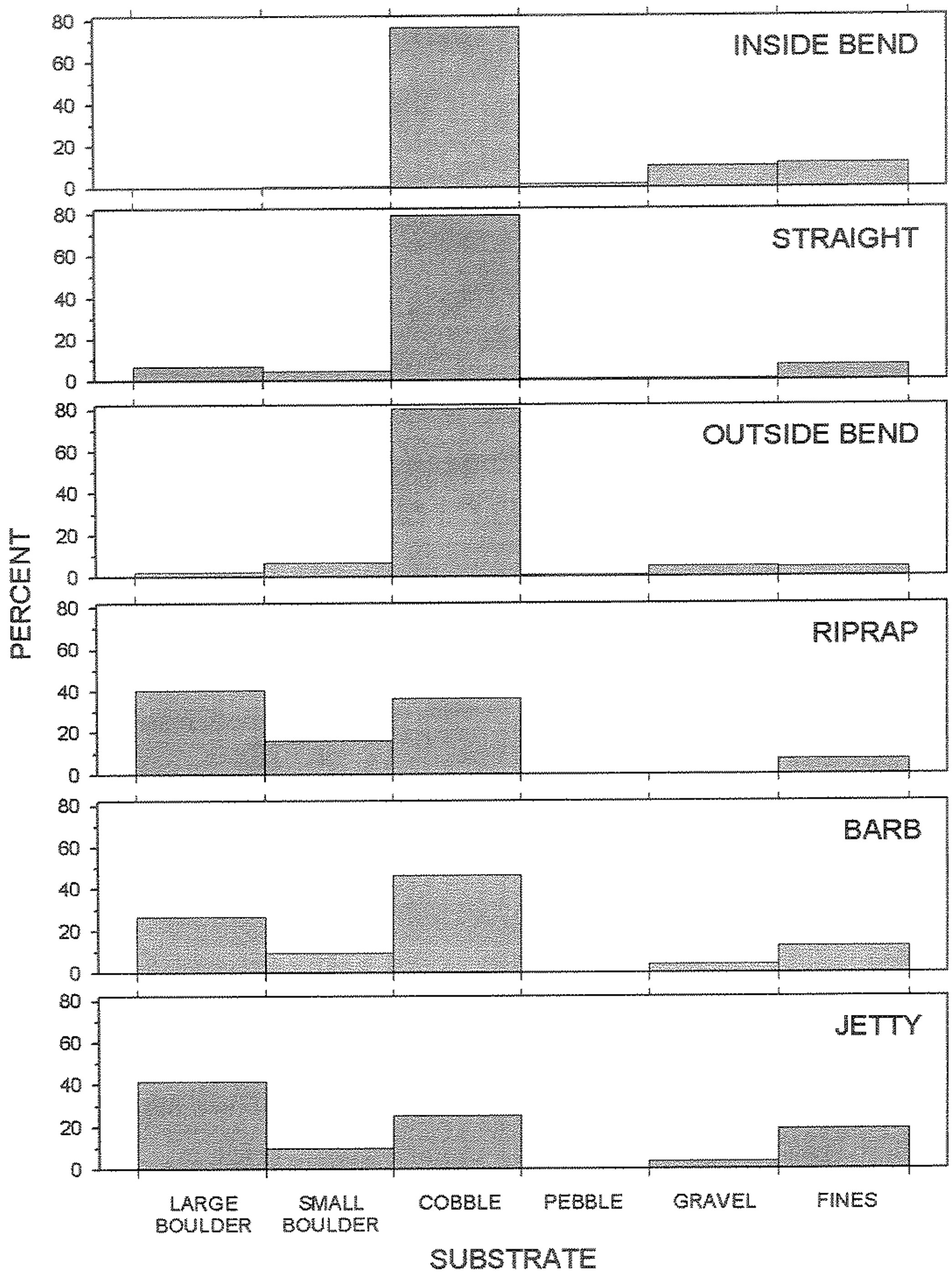

Figure 23. Frequency distributions of substrate sizes along transects by bank type, Yellowstone River.

Appendix 1

Individual Sample Records

Appendix 1. Numbers of each salmonid species captured by single-pass electrofishing at each 50-m sampling site during each sampling season. BA = barb, IB = inside bend, JT = jetty, OB = outside bend, RR = riprap, and ST = straight.

Year	Season	Reach	Bank type	Site number	Brown trout	Mountain whitefish	Rainbow trout	Yellowstone cutthroat trout	Brook trout
2001	spring	1	BA	4	6	0	1	1	0
2001	spring	1	BA	13	4	0	0	0	0
2001	spring	1	BA	15	2	0	7	0	0
2001	spring	1	BA	18	2	0	4	0	0
2001	spring	1	BA	32	4	0	7	0	0
2001	spring	1	BA	36	3	0	2	0	0
2001	spring	1	BA	45	9	0	6	0	0
2001	spring	1	BA	47	3	0	3	0	0
2001	spring	1	IB	1	2	0	1	0	0
2001	spring	1	IB	11	0	1	1	0	0
2001	spring	1	IB	22	0	0	0	0	0
2001	spring	1	IB	28	0	0	0	0	0
2001	spring	1	IB	30	0	0	0	0	0
2001	spring	1	IB	34	0	1	0	0	0
2001	spring	1	IB	42	2	2	2	0	0
2001	spring	1	IB	44	0	0	1	0	0
2001	spring	1	JT	8	5	0	4	0	0
2001	spring	1	JT	10	6	0	3	0	0
2001	spring	1	JT	16	7	0	8	0	0
2001	spring	1	JT	26	6	0	3	0	0
2001	spring	1	JT	35	8	0	6	0	0
2001	spring	1	JT	39	11	0	6	0	0
2001	spring	1	JT	46	5	0	3	0	0
2001	spring	1	JT	48	3	0	2	0	0
2001	spring	1	OB	2	5	0	7	0	0
2001	spring	1	OB	6	4	0	5	0	0
2001	spring	1	OB	17	0	0	0	1	0
2001	spring	1	OB	19	1	0	0	0	0
2001	spring	1	OB	25	11	0	12	0	0
2001	spring	1	OB	27	2	0	0	0	0
2001	spring	1	OB	37	3	0	3	0	0
2001	spring	1	OB	38	8	0	11	0	0
2001	spring	1	RR	7	7	0	4	0	0
2001	spring	1	RR	9	18	0	2	0	0
2001	spring	1	RR	24	4	0	15	0	0
2001	spring	1	RR	29	11	0	8	0	0
2001	spring	1	RR	31	6	0	5	0	0
2001	spring	1	RR	33	6	0	4	0	0
2001	spring	1	RR	40	19	0	11	0	0
2001	spring	1	RR	41	4	0	5	0	0
2001	spring	1	ST	3	0	0	0	0	0
2001	spring	1	ST	5	8	0	18	2	0
2001	spring	1	ST	12	1	0	0	0	0
2001	spring	1	ST	14	3	0	9	1	0

Year	Season	Reach	Bank type	Site number	Brown trout	Mountain whitefish	Rainbow trout	Yellowstone cutthroat trout	Brook trout
2001	spring	1	ST	20	2	0	2	0	0
2001	spring	1	ST	21	0	0	0	0	0
2001	spring	1	ST	23	0	0	1	0	0
2001	spring	1	ST	43	6	1	5	0	0
2001	summer	1	BA	4	3	0	2	0	0
2001	summer	1	BA	13	0	0	0	0	0
2001	summer	1	BA	15	2	0	1	0	0
2001	summer	1	BA	18	0	0	2	0	0
2001	summer	1	BA	32	0	0	4	0	0
2001	summer	1	BA	36	0	0	0	0	0
2001	summer	1	BA	45	0	0	0	0	0
2001	summer	1	BA	47	3	0	4	0	0
2001	summer	1	IB	1	1	0	0	0	0
2001	summer	1	IB	11	0	1	0	0	0
2001	summer	1	IB	22	0	0	0	0	0
2001	summer	1	IB	28	0	0	0	0	0
2001	summer	1	IB	30	0	0	0	0	0
2001	summer	1	IB	34	3	5	0	0	0
2001	summer	1	IB	42	1	0	0	0	0
2001	summer	1	IB	44	0	0	0	0	0
2001	summer	1	JT	8	1	0	4	0	0
2001	summer	1	JT	10	0	0	0	0	0
2001	summer	1	JT	16	5	0	0	0	0
2001	summer	1	JT	26	6	0	2	0	0
2001	summer	1	JT	35	5	0	0	0	0
2001	summer	1	JT	39	0	0	0	0	0
2001	summer	1	JT	46	2	0	7	0	0
2001	summer	1	JT	48	3	0	3	0	0
2001	summer	1	OB	2	1	1	0	0	0
2001	summer	1	OB	6	4	0	1	0	0
2001	summer	1	OB	17	0	0	0	0	0
2001	summer	1	OB	19	0	0	0	0	0
2001	summer	1	OB	25	2	0	0	0	0
2001	summer	1	OB	27	0	0	0	0	0
2001	summer	1	OB	37	4	0	5	0	0
2001	summer	1	OB	38	1	0	1	0	0
2001	summer	1	RR	7	5	0	0	0	0
2001	summer	1	RR	9	2	0	5	0	0
2001	summer	1	RR	24	0	0	0	0	0
2001	summer	1	RR	29	2	0	4	0	0
2001	summer	1	RR	31	1	0	3	0	0
2001	summer	1	RR	33	0	0	4	0	0
2001	summer	1	RR	40	3	0	8	0	0
2001	summer	1	RR	41	5	0	2	0	0
2001	summer	1	ST	3	0	0	1	0	0
2001	summer	1	ST	5	0	0	0	0	0
2001	summer	1	ST	12	0	3	0	0	0
2001	summer	1	ST	14	2	0	5	0	0
2001	summer	1	ST	20	0	0	0	0	0

Year	Season	Reach	Bank type	Site number	Brown trout	Mountain whitefish	Rainbow trout	Yellowstone cutthroat trout	Brook trout
2001	summer	1	ST	21	0	0	0	0	0
2001	summer	1	ST	23	0	0	0	0	0
2001	summer	1	ST	43	2	0	0	0	0
2001	fall	1	BA	4	4	0	6	4	0
2001	fall	1	BA	13	0	0	0	0	0
2001	fall	1	BA	15	6	0	4	4	0
2001	fall	1	BA	18	0	0	13	0	0
2001	fall	1	BA	32	0	0	4	0	0
2001	fall	1	BA	36	0	0	1	0	0
2001	fall	1	BA	45	5	0	6	4	0
2001	fall	1	BA	47	3	0	3	1	0
2001	fall	1	IB	1	2	0	2	0	0
2001	fall	1	IB	11	0	0	0	0	0
2001	fall	1	IB	22	0	0	0	0	0
2001	fall	1	IB	28	0	0	0	0	0
2001	fall	1	IB	30	0	0	0	0	0
2001	fall	1	IB	34	1	0	3	0	0
2001	fall	1	IB	42	1	0	0	0	0
2001	fall	1	IB	44	0	1	1	0	0
2001	fall	1	JT	8	2	0	8	1	0
2001	fall	1	JT	10	5	0	6	1	0
2001	fall	1	JT	16	9	0	5	3	0
2001	fall	1	JT	26	4	0	6	0	0
2001	fall	1	JT	35	6	0	2	0	0
2001	fall	1	JT	39	1	0	1	0	0
2001	fall	1	JT	46	6	0	15	3	0
2001	fall	1	JT	48	1	0	3	0	0
2001	fall	1	OB	2	6	0	10	3	0
2001	fall	1	OB	6	5	0	3	0	0
2001	fall	1	OB	17	0	2	0	0	0
2001	fall	1	OB	19	0	0	0	0	0
2001	fall	1	OB	25	0	0	7	2	0
2001	fall	1	OB	27	6	0	7	0	0
2001	fall	1	OB	37	1	0	18	1	0
2001	fall	1	OB	38	6	0	6	5	0
2001	fall	1	RR	7	6	0	2	0	0
2001	fall	1	RR	9	8	0	1	2	0
2001	fall	1	RR	24	5	1	15	3	0
2001	fall	1	RR	29	2	0	17	1	0
2001	fall	1	RR	31	0	0	10	2	0
2001	fall	1	RR	33	1	0	7	1	0
2001	fall	1	RR	40	2	0	17	3	0
2001	fall	1	RR	41	0	0	5	0	0
2001	fall	1	ST	3	0	0	0	0	0
2001	fall	1	ST	5	0	0	0	0	0
2001	fall	1	ST	12	7	0	0	0	0
2001	fall	1	ST	14	5	0	7	3	0
2001	fall	1	ST	20	6	0	2	0	0
2001	fall	1	ST	21	0	0	0	0	0

Year	Season	Reach	Bank type	Site number	Brown trout	Mountain whitefish	Rainbow trout	Yellowstone cutthroat trout	Brook trout
2001	fall	1	ST	23	1	0	2	0	0
2001	fall	1	ST	43	0	0	1	0	0
2002	spring	1	BA	4	4	0	2	0	0
2002	spring	1	BA	13	0	0	1	0	0
2002	spring	1	BA	15	0	0	3	0	0
2002	spring	1	BA	18	0	0	1	0	0
2002	spring	1	BA	32	1	0	0	0	0
2002	spring	1	BA	36	0	0	0	0	0
2002	spring	1	BA	45	1	0	4	0	0
2002	spring	1	BA	47	1	2	1	0	0
2002	spring	1	IB	1	1	0	2	0	0
2002	spring	1	IB	11	0	0	0	0	0
2002	spring	1	IB	22	0	0	0	0	0
2002	spring	1	IB	28	0	0	0	0	0
2002	spring	1	IB	30	0	0	0	0	0
2002	spring	1	IB	34	0	0	0	0	0
2002	spring	1	IB	42	0	0	1	0	0
2002	spring	1	IB	44	0	5	1	0	0
2002	spring	1	JT	8	6	0	5	1	0
2002	spring	1	JT	10	3	0	2	0	0
2002	spring	1	JT	16	5	0	0	0	0
2002	spring	1	JT	26	2	0	3	0	0
2002	spring	1	JT	35	4	0	2	0	0
2002	spring	1	JT	39	0	0	2	0	0
2002	spring	1	JT	46	2	0	4	0	0
2002	spring	1	JT	48	2	0	2	0	0
2002	spring	1	OB	2	6	0	5	0	0
2002	spring	1	OB	6	1	0	1	0	0
2002	spring	1	OB	17	0	2	0	0	0
2002	spring	1	OB	19	0	0	0	0	0
2002	spring	1	OB	25	2	0	3	0	0
2002	spring	1	OB	27	1	0	1	0	0
2002	spring	1	OB	37	4	0	11	0	0
2002	spring	1	OB	38	0	0	8	0	0
2002	spring	1	RR	7	4	0	1	0	0
2002	spring	1	RR	9	6	0	4	0	0
2002	spring	1	RR	24	2	0	6	0	0
2002	spring	1	RR	29	0	0	4	0	0
2002	spring	1	RR	31	1	0	5	0	0
2002	spring	1	RR	33	1	0	4	1	0
2002	spring	1	RR	40	4	0	3	0	0
2002	spring	1	RR	41	3	0	11	0	0
2002	spring	1	ST	3	1	0	2	0	0
2002	spring	1	ST	5	0	0	2	0	0
2002	spring	1	ST	12	1	0	0	0	0
2002	spring	1	ST	14	4	0	4	0	0
2002	spring	1	ST	20	0	0	0	0	0
2002	spring	1	ST	21	0	0	0	0	0
2002	spring	1	ST	23	0	0	0	0	0

Year	Season	Reach	Bank type	Site number	Brown trout	Mountain whitefish	Rainbow trout	Yellowstone cutthroat trout	Brook trout
2002	spring	1	ST	43	1	0	1	0	0
2002	summer	1	BA	4	1	1	4	0	0
2002	summer	1	BA	13	0	4	0	0	0
2002	summer	1	BA	15	1	0	0	0	0
2002	summer	1	BA	18	0	0	0	0	0
2002	summer	1	BA	32	1	0	1	0	0
2002	summer	1	BA	36	0	1	0	0	0
2002	summer	1	BA	45	7	0	5	0	0
2002	summer	1	BA	47	0	0	3	0	0
2002	summer	1	IB	1	2	1	0	0	0
2002	summer	1	IB	11	0	0	0	0	0
2002	summer	1	IB	22	0	0	0	0	0
2002	summer	1	IB	28	0	0	0	0	0
2002	summer	1	IB	30	0	0	0	0	0
2002	summer	1	IB	34	0	0	0	0	0
2002	summer	1	IB	42	0	0	0	0	0
2002	summer	1	IB	44	0	0	1	0	0
2002	summer	1	JT	8	0	0	1	0	0
2002	summer	1	JT	10	0	0	1	0	0
2002	summer	1	JT	16	4	0	1	0	0
2002	summer	1	JT	26	2	0	0	0	0
2002	summer	1	JT	35	1	7	0	0	0
2002	summer	1	JT	39	0	0	3	0	0
2002	summer	1	JT	46	2	0	9	0	0
2002	summer	1	JT	48	0	0	0	0	0
2002	summer	1	OB	2	0	0	2	0	0
2002	summer	1	OB	6	0	0	0	0	0
2002	summer	1	OB	17	4	10	2	0	0
2002	summer	1	OB	19	0	0	0	0	0
2002	summer	1	OB	25	0	0	2	0	0
2002	summer	1	OB	27	2	0	4	0	0
2002	summer	1	OB	37	3	0	8	0	0
2002	summer	1	OB	38	2	0	4	0	0
2002	summer	1	RR	7	5	0	2	0	0
2002	summer	1	RR	9	0	0	1	0	0
2002	summer	1	RR	24	0	0	0	0	0
2002	summer	1	RR	29	0	0	2	0	0
2002	summer	1	RR	31	0	0	0	0	0
2002	summer	1	RR	33	0	0	1	0	0
2002	summer	1	RR	40	2	0	2	0	0
2002	summer	1	RR	41	0	0	1	0	0
2002	summer	1	ST	3	2	12	2	0	0
2002	summer	1	ST	5	2	0	0	0	0
2002	summer	1	ST	12	0	0	0	0	0
2002	summer	1	ST	14	0	0	1	0	0
2002	summer	1	ST	20	0	0	1	0	0
2002	summer	1	ST	21	0	0	2	0	0
2002	summer	1	ST	23	0	0	2	0	0
2002	summer	1	ST	43	0	0	1	0	0

Year	Season	Reach	Bank type	Site number	Brown trout	Mountain whitefish	Rainbow trout	Yellowstone cutthroat trout	Brook trout
2001	summer	2	JT	17	2	0	13	0	0
2001	summer	2	JT	26	0	0	16	0	0
2001	summer	2	JT	32	0	0	16	0	0
2001	summer	2	JT	36	2	0	19	0	0
2001	summer	2	OB	13	0	0	0	0	0
2001	summer	2	OB	21	1	0	5	0	0
2001	summer	2	OB	22	0	0	2	0	0
2001	summer	2	OB	29	0	0	7	0	0
2001	summer	2	OB	30	0	0	5	0	0
2001	summer	2	OB	35	4	0	48	0	0
2001	summer	2	RR	1	1	0	27	0	0
2001	summer	2	RR	3	0	0	8	0	0
2001	summer	2	RR	7	2	0	10	0	0
2001	summer	2	RR	18	2	0	21	0	0
2001	summer	2	RR	24	12	0	9	0	0
2001	summer	2	RR	31	2	0	19	0	0
2001	summer	2	ST	4	0	0	5	0	0
2001	summer	2	ST	5	5	1	1	0	0
2001	summer	2	ST	9	2	0	1	0	0
2001	summer	2	ST	11	2	0	23	0	0
2001	summer	2	ST	23	0	0	2	0	0
2001	summer	2	ST	28	0	0	0	0	0
2001	fall	2	BA	15	2	0	7	0	0
2001	fall	2	BA	19	8	0	7	0	0
2001	fall	2	IB	14	1	1	1	0	0
2001	fall	2	IB	27	1	0	2	0	0
2001	fall	2	JT	2	3	0	9	0	0
2001	fall	2	JT	26	7	0	7	0	0
2001	fall	2	OB	22	1	0	5	1	0
2001	fall	2	OB	29	1	0	7	0	0
2001	fall	2	RR	3	3	0	16	0	0
2001	fall	2	RR	18	8	0	11	0	0
2001	fall	2	ST	5	4	1	3	0	0
2001	fall	2	ST	28	0	0	0	0	0
2002	spring	2	BA	6	0	0	8	0	0
2002	spring	2	BA	10	2	0	4	0	0
2002	spring	2	BA	15	2	0	3	0	0
2002	spring	2	BA	19	0	3	2	0	0
2002	spring	2	BA	33	2	5	2	0	0
2002	spring	2	BA	34	0	12	0	0	0
2002	spring	2	IB	8	0	0	0	0	0
2002	spring	2	IB	14	1	0	1	0	0
2002	spring	2	IB	16	0	0	2	0	0
2002	spring	2	IB	20	0	0	2	0	0
2002	spring	2	IB	25	0	0	0	0	0
2002	spring	2	IB	27	1	0	6	0	0
2002	spring	2	JT	2	0	0	2	0	0
2002	spring	2	JT	12	1	0	13	0	0
2002	spring	2	JT	17	1	0	6	0	0

Year	Season	Reach	Bank type	Site number	Brown trout	Mountain whitefish	Rainbow trout	Yellowstone cutthroat trout	Brook trout
2002	spring	2	JT	26	2	0	5	0	0
2002	spring	2	JT	32	3	1	90	0	0
2002	spring	2	JT	36	7	0	48	0	0
2002	spring	2	OB	13	0	1	0	0	0
2002	spring	2	OB	21	0	3	3	0	0
2002	spring	2	OB	22	1	5	1	0	0
2002	spring	2	OB	29	1	0	7	0	0
2002	spring	2	OB	30	0	9	2	0	0
2002	spring	2	OB	35	3	0	44	0	0
2002	spring	2	RR	1	0	0	14	0	0
2002	spring	2	RR	3	4	0	12	0	0
2002	spring	2	RR	7	2	0	11	0	0
2002	spring	2	RR	18	0	0	8	0	0
2002	spring	2	RR	24	3	0	14	0	0
2002	spring	2	RR	31	2	0	9	0	0
2002	spring	2	ST	4	7	0	7	0	0
2002	spring	2	ST	5	2	0	20	0	0
2002	spring	2	ST	9	0	0	1	0	0
2002	spring	2	ST	11	2	0	5	0	0
2002	spring	2	ST	23	0	0	0	0	0
2002	spring	2	ST	28	0	0	0	0	0
2002	summer	2	BA	6	3	0	10	0	0
2002	summer	2	BA	10	5	0	7	0	0
2002	summer	2	BA	15	3	1	9	0	0
2002	summer	2	BA	19	2	0	6	0	0
2002	summer	2	BA	33	2	0	11	0	0
2002	summer	2	BA	34	0	0	3	0	0
2002	summer	2	IB	8	0	0	1	0	0
2002	summer	2	IB	14	0	2	3	0	0
2002	summer	2	IB	16	0	3	10	0	0
2002	summer	2	IB	20	0	0	1	0	0
2002	summer	2	IB	25	0	0	0	0	0
2002	summer	2	IB	27	0	4	4	0	0
2002	summer	2	JT	2	0	0	2	0	0
2002	summer	2	JT	12	0	0	14	0	0
2002	summer	2	JT	17	1	0	15	0	0
2002	summer	2	JT	26	0	0	3	0	0
2002	summer	2	JT	32	0	0	21	0	0
2002	summer	2	JT	36	0	0	7	0	0
2002	summer	2	OB	13	0	10	2	0	0
2002	summer	2	OB	21	0	2	10	0	0
2002	summer	2	OB	22	0	0	2	0	0
2002	summer	2	OB	29	3	0	5	0	0
2002	summer	2	OB	30	0	4	1	0	0
2002	summer	2	OB	35	0	0	10	0	0
2002	summer	2	RR	1	2	0	17	0	0
2002	summer	2	RR	3	1	0	8	0	0
2002	summer	2	RR	7	2	0	10	0	0
2002	summer	2	RR	18	1	0	20	0	0

FEBRUARY 2001

MONTANA COOPERATIVE
FISHERY RESEARCH UNIT

EFFECTS OF BANK STABILIZATION STRUCTURES ON FISH AND THEIR HABITAT

*

A LITERATURE REVIEW

ADAM J. CRAIG

ALEXANDER V. ZALE

USGS, DEPARTMENT OF ECOLOGY
MONTANA STATE UNIVERSITY - BOZEMAN
BOZEMAN, MONTANA 59717

Executive Summary

This literature review is the first deliverable associated with a research project entitled "Comparative use of modified and natural habitats of the Upper Yellowstone River by juvenile salmonids" conducted by the Montana Cooperative Fishery Research Unit with funding provided by the U.S. Army Corps of Engineers, Omaha District, in association with the Governor's Upper Yellowstone River Task Force. The goal of the study is to assess the extent to which changes in aquatic habitats caused by bank stabilization, flow deflection, and flow confinement structures affect juvenile fish in the upper Yellowstone River. The field component of the study will involve comparing juvenile fish use of altered aquatic habitat types to their use of natural, unaltered habitats. This information will be used to estimate past and future effects of habitat modifications on the fishery resource of the Yellowstone River.

This review was conducted to summarize pertinent research and to guide the development of the sampling program. It summarizes and integrates previous studies addressing the effects of bank stabilization structures on river processes, invertebrates, and foremost, fish. We have organized this literature review based on the predominant concepts we found in the literature including hydrologic processes in rivers, importance of side channels and backwaters in providing a diversity of habitats, and the positive and negative effects of bank stabilization on rivers and their biota. Also included is a section addressing sampling techniques described in the literature that may be useful on the Yellowstone River. Finally, we have provided annotations of the most important references expressly dealing with the effects of bank stabilization on fish.

Previous studies examining the physical effects of banks stabilization structures on rivers showed that these structures reduce channel braiding and meandering, thereby reducing physical habitat diversity, which results in less diverse and productive fish assemblages. Because riprap provides many interstitial spaces and high amounts of surface area, aquatic invertebrates (i.e., fish food) flourish therein. Some studies showed higher diversities and abundances of fish along revetted banks than natural banks. These studies tended to take place in previously degraded habitats or warmwater ecosystems. Other studies showed decreases in abundances of fish along revetted banks compared to unaltered banks. These studies generally examined relatively pristine habitats or coldwater ecosystems inhabited by salmonids. Banks stabilized with deflection structures had higher densities and diversities of fish than revetted banks. Deflection structures created habitats with low water velocities directly adjacent to the mainstream and more heterogeneity of depth, velocity, and stream bed than revetted banks; this diversity of habitat characteristics was beneficial to fish. We found no studies which comprehensively addressed long-term effects of bank stabilization over large spatial scales. None of the fish sampling techniques used in previous studies addressing effects of bank stabilization structures appears to be perfectly suited to our needs on the Yellowstone River.

Effects of Bank Stabilization on Physical River Processes

Physical attributes (e.g., channel pattern and shape, pool-riffle spacing, sediment size distribution) of a river's channel result from complex interactions among supply of water and sediment to the river and localized hydraulic processes which govern sediment erosion, transport and deposition (Leopold et al. 1964; Dunne and Leopold 1978). Movement of water and sediment through the river's channels over time tend to create a relatively stable equilibrium fluvial geomorphology (form and structure) that efficiently transports supplied water and sediments (Leopold et al. 1964). In snowmelt driven rivers such as the Yellowstone, bank full stage discharge during spring runoff dictates this geomorphology (Williams 1978; Andrews 1980; Andrews and Nankervis 1995). Stable alluvial channels typically accommodate snowmelt runoff through an annual pattern of lateral (e.g., bank erosion and point bar deposition) and vertical (e.g., scour and fill) processes that maintain channel width and bed elevation as the channel migrates across the flood plain. This annual cycle is also responsible for maintaining a diverse mix of sediment types and sizes (Gordon et al. 1992), which is important because different species of aquatic organisms differ in their substrate preferences and requirements (Gordon et al. 1992). For example, chironomid midge larvae require mud into which they can burrow, whereas salmonids require a mix of gravel, sand, and cobble for optimum spawning substrate (Beschta and Platts 1986; Gordon et al. 1992). Thus, the distribution of sediment types and sizes along a stream can be a paramount factor affecting the persistence of fish and invertebrates (Gordon et al. 1992).

Bank stabilization and flow diversion structures alter a river's natural adjustment processes, thereby causing changes in channel morphology, hydraulic geometry (width, depth, slope, roughness), channel pattern, bank erodability, and supplies of sediment and large woody debris (Beschta and Platts 1986; Brookes 1988). Responses may include changes in rates of lateral channel migration, substrate size distributions, channel-bed elevation, pool-riffle spacing, and frequency of side channel and over-bank flows (Leopold et al. 1964; Gregory and Walling 1973; Schumm 1977; Simons and Senturk 1977; Steiger et al. 1998; Petts et al. 1989; Klingeman et al. 1999). Channel incision resulting from bank stabilization lowers stage at a given flow (Stern et al. 1980) and thereby reduces the frequency of inundation of side channels. Coupled with increased sediment deposition in side channels caused by decreased water velocities there, such incision can eventually cause side channels to become part of the flood plain and not the active channel.

For example, revetted banks (banks stabilized with riprap) on the lower Mississippi River shortened the river length 229 km, and levees reduced the floodplain by 90% (Baker et al. 1988a). Levees and dikes along the Vistula River, Poland, reduced the number of islands and braided reaches, decreased the channel width by 50%, and deepened the riverbed by 1.3 m (Backiel and Penczak 1989). The Piave River in Italy also became less braided and its channel width decreased after flow deflection structures were installed (Surian 1999). Bank revetments along the Rhine River caused the riverbed to deepen by up to 7 m and reduced the number of backwaters, braids, and side channels (Dister et al. 1990). Channelized and riprapped sections of Little Prickly Pear Creek north of Helena, Montana, were uniformly shallow and homogeneous, whereas unaltered sections varied in depth and alternated between pools and riffles (Elser 1968). Thus, bank stabilization structures not only alter the banks they are designed to protect, but by redirecting a river's energy, change the morphology and physical structure of a river. These changes, in turn, would be expected to change the quantity and quality of fish habitats.

Importance of Channel Migration, Side Channels, and Backwaters to Fish Habitat

Importance of Channel Migration

When a river is not allowed to move its channel laterally, unnatural regimens of sediment flow occur that lead to decreased amounts of important habitats where fish can find food, cover, or spawning substrates (White 1991; Schmetterling et al. in press). In particular, creation of pools and riffles typical of meandering streams may be limited (Montgomery and Buffington 1997). Channel migration provides a river with large woody debris (Murphy and Koski 1989), which is a critical habitat requirement in most trout streams. Input of large woody debris to a river stabilizes the channel, traps sediment and debris that modifies channel shape by redirecting currents, and provides shelter for fish (Gordon et al. 1992). Abundances and biomasses of trout in reaches of 13 Montana streams altered by channel relocation, riprapping, clearing, and diking to preclude natural meandering were only 29% and 11%, respectively, of those in unaltered reaches (Peters and Alvord 1964). Channel migration can also provide required spawning substrates. For example, erosive channel widening on the South Fork Kern River, California, resulted in significantly more spawning habitat and higher densities of redds and age-0 golden trout (*Oncorhynchus aguabonita*) than in stable narrow reaches (Knapp et al. 1998). Because bank stabilization structures restrain a river's natural lateral channel migration, they allow less large woody debris input, substrate deposition, and pool, riffle, and side-channel formation, and thereby lead to decreased habitat quality for fish. These changes in turn, would be expected to limit abundance and production of fish.

Sidechannels and Backwaters Provide Nutrients and Habitat Complexity

Biotic production in rivers is positively correlated with periodic inundation of their floodplains (Odum et al. 1979; Junk et al. 1989; Bayley 1991) as exchange of water, sediments, nutrients, and organisms between rivers and their backwaters on the floodplain is thereby achieved (Junk et al. 1989; Dister et al. 1990; Bayley 1991). Flooded lateral habitats are major production zones for plankton, which are released into the river as flood waters recede and are essential food for early life stages of fish (Schiemer and Spindler 1989; Schiemer et al. 1991). Flood control and channel stabilization projects may eliminate backwaters on floodplains or disconnect them from the river (Sandheinrich and Atchison 1986; Dister et al. 1990) thereby reducing productivity of the river. Bank stabilization projects may also impede establishment of riparian vegetation, especially cottonwoods, and thereby limit energy inputs, shade, and sediment and pollutant filtration (Robert Hazlewood, U.S. Fish and Wildlife Service, personal communication).

Backwaters, braids, and side channels provide water velocities, depths, and substrates not present in adjoining main river channels (Hjort et al. 1984) and thereby increase available habitat diversity to the benefit of fish and invertebrates. For example, fish species richness was greatest in backwaters of the Missouri River in South Dakota, Nebraska, and Iowa (Kallemeyn and Novotny 1977) and reaches in North Dakota with extensive backwaters had higher densities of invertebrates than revetted or wooded reaches (Burress et al. 1982). Banks along the Danube and Morava rivers that included littoral bays supported higher densities and diversities of juvenile fish than adjacent riprapped banks (Schiemer and Spindler 1989; Jurajda 1995). The fish assemblage in a backwater of the Willamette River, Oregon, was characterized by more trophic complexity and larger fish than the main river channel itself (Hjort et al. 1984).

Habitat diversity is especially important for salmonids because they require different water velocities (Cunjak and Power 1987; Greenberg et al. 1996; Petays et al. 1997), depths (Cunjak and Power 1987; Baltz et al. 1991; Greenberg et al. 1996; Petays et al. 1997), cover types (Heggenes 1988; Mesick 1988), and substrates (Greenberg et al. 1996) at different sizes and ages. Small trout tend to prefer shallow, low-velocity areas with small substrate sizes or vegetation, whereas large trout prefer deeper water with higher water velocities and larger substrate sizes and overhead cover. Growth and survival rates of juvenile trout are higher in side channels than the main channel, and side channels are a preferred spawning location for salmonids (Mesick 1995; Downing 2000). Preference of large trout for deep water may help avoid predation by terrestrial predators, whereas preference for shallow water by small trout may be an attempt to avoid competition and predation by large trout (Schlosser 1987).

Salmonids also exhibit seasonal shifts in habitat use, especially during winter when mortality of juveniles is highest and year-class strength is determined. Movement into slower, deeper water in winter and taking refuge in the substrate during daylight hours when water temperatures decrease below 10 °C is a general response for age-0 salmonids (Rimmer et al. 1983; Campbell and Neuner 1985; Baltz et al. 1987; Contor and Griffith 1995). Fish may move singly or in small groups into interstices in the substrate (Hartman 1963) anywhere from 15 to 30 cm deep beneath the substrate surface (Griffith and Smith 1993). Such concealment cover typically consists of large substrate sizes that provide appropriately sized interstices (Mitro 1999). Age-0 cutthroat (*O. clarki*) and brown trout (*Salmo trutta*) were absent from cobble substrate but present in boulder substrate during winter on the South Fork of the Snake River (Griffith and Smith 1993); the smallest substrate used was 20 cm in diameter. Size of substrate used by juvenile trout in winter may also depend on time of day. In artificial streams, small brown trout concealed themselves in coarse substrates less frequently in the evening than during the day (Heggenes et al. 1993). These studies show that presence of coarse substrate that provides interstitial cover is a critical requirement of juvenile trout in winter. Reducing addition of such substrates by precluding bank erosion would therefore be detrimental. However, bank stabilization structures that incorporate coarse substrates may benefit salmonids in winter if such substrates are rare along unaltered banks. Coarse substrates may also become more common as channels incise in response to constrainment of the channel and reduced addition of bank material caused by stabilization structures. Older age classes of wintering trout also shelter within the interstitial spaces of coarse substrates (Bjornn 1971; Hillman et al. 1987; Petays et al. 1997) or use backwaters with abundant overhead cover, low water velocities, and groundwater inflows (Cunjak and Power 1986).

A diversity of habitat types, as provided by backwaters, braids, and side channels in addition to a main channel, is therefore required to support all of the sizes and ages of a wild trout population. Loss of this diversity through elimination of habitats other than the main channel would be expected to negatively affect abundances of trout by limiting recruitment and increasing emigration to more diverse reaches elsewhere. For example, experimental increases in lateral backwaters and eddies on Mack Creek, Oregon, resulted in greater densities of age-0 cutthroat trout, whereas these fish were almost eliminated from stream sections where these lateral habitats were reduced (Moore and Gregory 1988). Similarly, increases in salmonid production resulted from the opening of ponds adjacent to the channel on Fish Creek, Oregon (F. E. Everest et al., U.S.F.S Pacific Northwest Research Station, unpublished data in Frissel and Nawa 1992).

Biological Effects of Riprap

Effects of Riprap on Invertebrates

Because riprap provides many interstitial spaces and high amounts of surface area, aquatic invertebrates flourish therein. Riprap in streams often becomes a location for sediment and debris deposition (Shields 1991), which enhances habitat for benthic invertebrates by providing additional food and cover (Burress et al. 1982; Mathis et al. 1982), except when the deposited sediments consist of sand (Sanders et al. 1986). Channelized reaches of the Missouri River in South Dakota had higher diversities, but lower densities, of invertebrates than natural reaches (Wolf et al. 1972). Invertebrate drift was greater along riprapped, channelized banks of the Missouri River in Iowa than along natural banks (Kallemeyn and Novotny 1977) and current-swept rocks in dikes and revetments supported more diversity and a higher density of macroinvertebrates than did natural stream substrates along the Missouri River in North Dakota (Burress et al. 1982). Similarly, higher total numbers of invertebrates were collected from revetted banks than natural banks along the Willamette River, Oregon (Hjort et al. 1984). On the other hand, artificial substrates placed in an unchannelized stretch with natural banks on the Missouri River near Vermillion, South Dakota, had 70% greater standing crops of invertebrates than at riprapped banks near Sioux City, Iowa (Nord and Schmulbach 1973). Abundant aquatic invertebrates in riprap may serve as a superior food source for fish, but no studies have been conducted that directly show that higher abundances of aquatic invertebrates in riprap benefit fish.

Positive Effects of Riprap on Fish

Positive or neutral effects on fish resulting from bank stabilization with riprap have been observed in warmwater systems, primarily the Mississippi River. Revetted banks along the lower Mississippi River in Mississippi supported the highest percentage, by weight, of fish species considered to have a sporting or commercial value compared to natural banks (Pennington et al. 1983a; Pennington et al. 1983b). Fish abundances (mostly freshwater drum *Aplodinotus grunniens*, flathead catfish *Pylodictis olivaris*, common carp *Cyprinus carpio*, and blue catfish *Ictalurus furcatus*) in the Mississippi River near Eudora, Arkansas, were similar along old revetments, new revetments, and natural banks, which suggested that fish inhabiting natural riverbanks recovered rapidly after bank perturbation caused by the placement of riprap (Pennington et al. 1985). Abundances and aggregate weights of all species combined were greater at revetted banks than natural banks of Pool 24 of the Mississippi River in Missouri, and fish diversities at both bank types were equal (Farabee 1986). Revetted banks of the Willamette River, Orgeon, supported higher densities of small warmwater fish than unaltered banks, which were inhabited by low densities of large fish (Hjort et al. 1984).

Positive or neutral effects of riprap have also been observed in coldwater systems. Abundance of 6 to 12 inch brown trout increased 35% and abundance of 12-inch and larger brown trout increased 86% after 0.7 miles of riprap were installed on Willow Creek, Wisconsin (Hunt 1988). Also in Wisconsin, Millville Creek was stabilized with riprap to mitigate effects of bank degradation caused by cattle grazing and row crop farming in the riparian zone (Avery 1995). Mean densities of brown trout increased from 65 fish/mile to 102 fish/mile after the bank stabilization (but the author considered this increase in density insufficient justification for the $26,800/mile cost of the riprap). Seven years after 2,150 feet of riprap and 111 habitat improvement devices (deflectors, plunges, overhangs, channel blocks, ramps, logs) were installed on Beaver Creek, Wyoming, to mitigate habitat degradation stemming from cattle grazing, abundances of brook trout (*Salvelinus fontinalis*) 6 inches and longer had increased 1,814% and abundances of brook trout less than 6 inches had increased 1,462% (Binns 1994). Abundances of yearling steelhead (*O. mykiss*) and cutthroat trout increased shortly after banks along large streams in central western Washington were riprapped (Knudsen and Dilley 1987). Fish species diversity (but not abundance) was greater along riprapped banks than along natural banks of the Sacramento River, California (Michny 1988). Large riprap (rock >30 cm in diameter) supported higher juvenile chinook salmon (*O. tshawytscha*) and steelhead trout densities than natural cobble-boulder banks on the Thompson River in British Columbia in both summer and winter (Lister et al. 1995). The overall densities of yearling and older salmonids in 15 western Washington rivers were unaffected or increased at riprapped banks (Peters et al. 1998) and sub-yearling rainbow trout (*O. mykiss*) in the Skagit River, Washington, were more abundant in riprap compared to the mean reach abundance (Beamer and Henderson 1998). Several of these studies are described more completely in the annotated bibliography of this document.

Negative Effects of Riprap on Fish

Few studies conducted in warmwater systems indicated negative effects of riprap on fish. Riprapped banks of the Willamette River, Oregon, were poor habitat for larvae of warmwater fish compared to natural banks (Li et al. 1984) and species diversity along revetted banks was lower than at the unaltered banks (Hjort et al. 1984). Riprap also provides habitat favoring introduced exotic species. Riprapped banks on the St. Clair River, Michigan, had higher densities of round gobies (*Neogobius melanostomus*), tubenose gobies (*Proterorhinus marmoratus*), and zebra mussels (*Dreissena polymorpha*) than natural sand and macrophyte-dominated substrate (Jude and DeBoe 1996).

Assessments of riprap in coldwater systems inhabited by salmonids tended to show deleterious effects. Most of these studies are covered in greater detail in the annotated bibliography. Brown and rainbow trout were significantly more abundant in unaltered sections than in channelized and riprapped sections of Little Prickly Pear Creek north of Helena, Montana, and non-salmonid fishes were almost completely absent in the altered reaches (Elser 1968). Biomasses of juvenile coho salmon (*O. kisutch*), juvenile steelhead, and cutthroat trout decreased shortly after long lengths of bank were stabilized along small streams in central western Washington (Knudsen and Dilley 1987); in larger streams, only slight reductions in numbers of juvenile coho salmon and young-of-the-year cutthroat trout occurred. Densities of rainbow trout in the Big Wood River, Idaho, were highest in areas with diverse channel features and in the presence of woody cover (17.4 trout/100 m^2), whereas riprapped banks held almost as few fish (2.1 trout/100 m^2) as habitats lacking any cover (1.2 trout/100 m^2) (Thurow 1988). Sub-yearling cutthroat trout, coho salmon, and chinook salmon densities were lower at riprapped banks than natural banks on the Skagit River (Beamer and Henderson 1998) and 15 other rivers in western Washington (Peters et al. 1998). Relative abundances of juvenile chinook salmon along riprapped banks of the Sacramento River, California, were 25% of those along natural banks (Michny 1987; U.S. Fish and Wildlife Service 1992).

Enhancements to Riprap that Benefit Fish

Various enhancements can be incorporated into riprapped banks to benefit fish including off-bankline revetments, larger rock size, fish groins, filling interstices with gravel, rearing benches, and indented revetments (Shields et al. 1995). Some of these show great promise, but have not been extensively deployed yet (Shields et al. 1995).

Placing boulders of 1.0 m to 1.5 m in diameter along the toe of the bank (intersection of bank and edge of the stream channel) on the Coldwater River, British Columbia, appeared to increase rearing densities of all salmonids except sub-yearling steelhead trout by providing cover from high water velocities (Lister et al. 1995). Traditional riprap revetments along the Sacramento River, California, enhanced with low ridges of riprap called "fish groins" running perpendicular to the channel from the toe of the bank to the top of the bank were used by juvenile salmonids more than unimproved riprap, but not as much as natural banks (U.S. Fish and Wildlife Service 1992). A gradually sloping (1V:5H) gravel bench parallel to the channel, called a rearing bench, placed at an elevation where it was inundated at moderate flows provided shallow habitat for juvenile salmonids by simulating hydraulic conditions associated with natural gravel bars (Michny 1987); fish abundances therein were intermediate between those at natural and unimproved riprapped banks. Incorporation of notches or gaps in revetted banks facilitates formation of littoral bays, which enhance fish abundances in altered reaches (Kallemeyn and Novotny 1977). Combined longitudinal and transverse dikes along the River Rhône in France created backwater impoundments inhabited by more abundant and diverse assemblages of juvenile fish than in the adjacent river (Poizat and Pont 1996; Nicolas and Pont 1997)

The size of material used in constructing riprap affects microhabitat selection by salmonids because substrate size is an important criterion determining habitat suitability as described previously (Bustard and Narver 1975; Rimmer et al. 1984; Greenberg et al. 1996). On the Skagit River, Washington, small rock (i.e., rubble from 64 to 256 mm in diameter) riprap adversely affected all species (coho salmon, chinook salmon, chum salmon *O. keta*, and rainbow trout) of fish compared to boulder riprap (Beamer and Henderson 1998). A Mississippi River bank riprapped with 60-cm diameter rock had a fish biomass catch-per-unit-effort rate more than twice as great as a similar bank riprapped with rock 30 to 60 cm in diameter (Farabee 1986). Banks of the Thompson and Coldwater Rivers, British Columbia, riprapped with material of mean diameter greater than 30 cm supported higher chinook salmon, coho salmon, and steelhead trout densities during summer and winter than banks riprapped with material of mean diameter less than 30 cm (Lister et al. 1995). Filling the interstices of riprap with gravel can also enhance habitat value of riprap for juvenile salmonids (Michny 1987; U.S. Fish and Wildlife Service 1992).

The type of material used in bank stabilization may also affect fish density. Revetted banks that incorporate woody vegetation provide more cover for fish and have a more natural appearance than rock riprap (Hunter 1991; McClure 1991; Shields 1991). Furthermore, revetted banks on the Sacramento River, California, that incorporated woody vegetation suffered less damage from high flow velocities than unvegetated banks of the same age and similar curvature (Shields 1991).

Speculation about Conflicting Findings of Riprap Studies

The studies described above addressing the effects of bank stabilization with riprap on river biota provide ambiguous results when considered in aggregate. Some case studies showed higher diversities and abundances of fish and invertebrates along riprapped banks than natural banks. Other studies indicated decreases in abundances and diversities of fish along riprapped banks compared to natural banks. In some studies, benefits were accrued by some species while others were deleteriously affected. In this section, we provide conjecture on why such disparate findings exist. However, it is important to note that this is, for the most part, mere speculation.

Some of the studies showing positive effects of riprap on fish (i.e., Binns 1994 and Avery 1995) were before-and-after studies conducted in streams suffering previous bank degradation from cattle grazing or other agriculture-related effects. Pre-existing conditions in these cases were already degraded and were no longer natural. Therefore, the ostensible positive effects of riprap in these cases may be viewed more realistically as partial mitigation of more severe past damage.

The beneficial effects of riprap in large, warmwater rivers such as the Mississippi may perhaps be viewed similarly. Historically, such rivers were congested with woody-debris snags, which because they were often the only hard substrates available (most substrates in these rivers consist of sand and gravel), were important sources of cover for fish and attachment sites for benthic invertebrates (Allan 1995). For example, in the Saltilla River, Georgia, woody debris represented only 4% of the total habitat surface available, but supported 60% of the total invertebrate biomass (Benke et al. 1985). Four of the 8 species of fish collected in this study obtained at least 60% of their prey biomass from woody debris, and all of the fish species used woody debris to some extent as cover (Benke et al. 1985). Removal of snags during the 19^{th} and 20^{th} centuries from large rivers to facilitate navigation (Funk and Robinson 1974) has severly diminished availability of hard substrates, leaving only shifting sand substrates. When riprap is introduced into these hard-substrate-limited systems, it is quickly colonized by invertebrates and used as cover by fish (Dardeau et al. 1995). Again, the pre-existing conditions used to compare riprapped banks to were already somewhat degraded and no longer provided a valid comparison. Studies conducted in coldwater systems tended to show negative effects of riprap on salmonids. In these systems, results may have differed from those in warmwater systems because hard substrate was likely not a limiting factor, considering that many freestone trout streams are characterized by a diverse range of substrate sizes, often including boulders. In addition, the absence of undercut banks along revetments may have been detrimental to salmonids.

Differing effects of riprap in different studies may also have been an artifact of when those studies were conducted and which life stages or species were focused on. Because microhabitat requirements change diurnally, seasonally, ontogenically, and as a function of prevailing weather and flow conditions, temporal and procedural differences in sampling protocols may have introducted confounding factors and led researchers to different conclusions.

Finally, it is important to recognize that riprap comes in various forms, sizes, and configurations, and can be made up of a variety of materials, which can influence its suitability as invertebrate and fish habitat. The physical descriptions of riprap in many of the studies we read were often incomplete or vague, thus making it difficult to recognize important distinctions (e.g., size of rock, incorporation of LWD) that may have helped reduce the uncertainty of our conclusions.

Current Deflection Structures

Current deflection structures are the primary alternative to riprap for stabilizing the longitudinal profile of rivers. In general, current deflection structures extend from a riverbank into the channel to redirect water flow away from the bank toward the middle of the channel (Peters et al. 1998; NRCS and DEQ 1998). The redirected flow is sometimes intended to maintain a navigation channel (Sandheinrich and Atchison 1986). Many variations of these structures exist and the nomenclature in the literature defining them is inconsistent. Spur dikes, wingdams, transverse dikes, and rock deflectors are all current deflection structures made with human-placed rock oriented downstream (NRCS and DEQ 1998; Joel Tohtz, Montana Fish, Wildlife and Parks, personal communication). Dike fields are a series of deflection structures and the associated pools between them (Pennington et al. 1983a). Barbs are structures made with human-placed rock oriented upstream (Buddy Drake, Drake and Associates, personal communication). Barbs can also be distinguished from other deflection structures because their height should not exceed the water surface at bankfull discharge (Buddy Drake, personal communication). Our literature search and consultations did not reveal any information explicitly describing the effects of barbs on fish.

Positive Effects of Current Deflection Structures on Fish

Deflectors are considered a superior bank stabilization type for fish compared to riprap (Li et al. 1984; Sandheinrich and Atchison 1986; Peters et al. 1998). Rock deflectors in the Wolf Creek Canyon section of Prickly Pear Creek, Montana, created physical stream characteristics comparable to those associated with natural banks (Elser 1968). Highest densities of larval fish in the Willamette River, Oregon, were found at a shallow, sloped beach habitat adjacent to deflection structures (Li et al. 1984). Fish densities along banks with deflector structures in Batupan Bogue Creek, Mississippi, were comparable to densities along natural banks, and were significantly greater than densities along riprapped banks (Knight and Cooper 1991); scour holes associated with the deflectors provided deepwater refuges for fish, including large individuals. During high flows, juvenile brown trout in the Rio Grande River, Colorado, moved to locations downstream of boulder bank deflectors, and age-0 trout were frequently observed in the low velocity areas there as well (Shuler et al. 1994). Fish densities in 15 western

Washington rivers were generally greater at deflector-stabilized banks than natural banks in winter (Peters et al. 1998).

The superior performance of deflectors as fish habitat compared to riprap is related to their creation of stable pools or scour holes (Witten and Bulkley 1975; Bulkley et al. 1976; Knight and Cooper 1991; Shields et al. 1993; Shields et al. 1995), lentic habitat connected with the main channel (Backiel and Penczak 1989), and provision of a complex of depth-velocity-bed type combinations, which are not typically found adjacent to riprap (Beckett et al. 1983; Li et al. 1984; Baker et al. 1988b). Deflectors, especially when in series (dike fields), provide more habitat heterogeneity than simple revetted banks and therefore support more diverse fish and macroinvertebrate assemblages and can also provide spawning and nursery areas for some species (Pennington et al. 1983a; Sandheinrich and Atchison 1986; Baker et al. 1987). Fish habitat value of channel reaches that lack a diversity of habitats, especially reaches of low hydraulic contrast with minimal pools, would likely be enhanced by the addition of deflectors. Larger and more numerous deflectors would be expected to provide more habitat (up to a point) in such reaches.

Negative Effects of Current Deflection Structures on Fish

Dike fields and other deflection structures can also deleteriously affect physical riverine processes and biota. Because dike fields and other deflection structures redirect flow into the thalweg of a river, riverbed degradation and dewatering of sidechannels and backwaters may result (Sandheinrich and Atchison 1986). Densities of cutthroat trout during spring were significantly less at deflector-stabilized banks than natural banks in western Washington rivers, perhaps because large woody debris incorporated into the structures was poorly placed (Peters et al. 1998). Deflection structures in the Willamette River, Oregon, provided better habitat for larval fishes than riprap, but not as good as at natural banks (Li et al. 1984).

Shortcomings of Bank Stabilization Studies

In the section on conflicting findings of riprap studies we identified confounding factors influencing conclusions drawn from specific studies. In addition to these factors, the scope of these studies tends to limit their applicability. Most studies are limited to certain seasons and are of short duration (<2 years), thereby limiting understanding of year-class, population, and fishery level effects because patterns of habitat use vary depending on life stage and species (Schiemer et al. 1991; Jurajda 1995; Lister et al. 1995). Effects, or lack thereof, observed during a given season of inquiry may be eclipsed by more pervasive effects during other seasons when sampling was not conducted. These studies also invariably examine only localized effects of bank stabilization on physical and biological properties of rivers. Therefore, macroscale (channel reaches at least ten or more channel widths long including a variety of habitat types) and long-term effects of bank stabilization have not been clearly addressed (Shields et al. 1995). Compensatory effects (e.g., shifts in habitat use that compensate for localized deleterious effects) have therefore not been examined. These factors render understanding of the cumulative effects of bank stabilization on the fish populations and fishery characteristics over entire river reaches incomplete.

Sampling Methods

Numerous sampling methods have been used for assessing fish abundances in altered and unaltered bank habitats. Our literature review and consultations identified a number of possible sampling alternatives that may determine relative, and possibly absolute, abundances of fish in different bank habitats along the Yellowstone River. These include grid-point or transect electrofishing via driftboat or jet boat, multi-pass electrofishing with a driftboat and shore-based backpack electrofisher in combination, snorkeling surveys, and underwater video imaging and photography. Because the upper Yellowstone River is a dynamic system with diverse habitats, some techniques may be more conducive to sampling certain areas than other techniques. Seasonal differences in performance may also exist. However, none of the techniques found in the literature has been tested and validated for the purpose of assessing juvenile salmonid abundances along different kinds of banks of a river the size and configuration of the Yellowstone.

Acknowledgments

Bradley B. Shepard, Joel Tohtz, and Bruce Rich of the Montana Department of Fish, Wildlife and Parks, Thomas E. McMahon, Montana State University, and Buddy Drake, Drake and Associates, contributed to this review by providing literature citations, local knowledge, and technical advice. Our work was aided greatly by a draft manuscript entitled "Effects of riprap bank stabilization on stream salmonids in the western US" by David A. Schmetterling, Christopher G. Clancy, and Troy M. Brandt, which will soon be published in *Fisheries* magazine. The manuscript introduced us to many of the citations and concepts found in our review. An earlier draft of this report was reviewed by Charles Dalby, Montana Department of Natural Resource Conservation, Robert Hazlewood, U.S. Fish and Wildlife Service, Bradley B. Shepard, Robert S. Nebel, U.S. Army Corps of Engineers, and Michael Merigliano, University of Montana; these reviews significantly enhanced this document.

Prominent Literature Annotations

The following annotations summarize the most important literature citations we found that expressly addressed bank stabilization structures and their effects on fish. Four of the annotations summarize studies conducted on warmwater systems and the other seven annotations describe work conducted on salmonids. They are listed in reverse chronological order within each group.

Warmwater

Farabee, G. B. 1986. Fish species associated with revetted and natural main channel border habitats in Pool 24 of the upper Mississippi River. North American Journal of Fisheries Management 6:504-508.

Two revetted and two natural banks within Pool 24 of the Mississippi River in Missouri were electrofished over a 3-year period to determine the fish species and species diversities associated with the two bank types. One of the revetted banks was stabilized with rock greater than and equal to 2 feet in diameter, whereas the other revetted bank was stabilized with rock less than 2 feet in diameter. Thirty-three species of fish were collected along the revetted and natural bank types alike, but gizzard shad (*Dorosoma cepedianum*) and common carp (*Cyprinus carpio*) dominated catches (65% combined). Seventy percent of all fish collected during the study were taken at the revetted banks, and 58 percent of those were collected from the bank stabilized with the larger rock. Catch-per-unit-effort rates and aggregate weights of fish collected were highest at the large-rock revetment, intermediate at the small-rock revetment, and lowest at the natural banks. The author concluded that need for bank stabilization measures and provision of fish habitat in the upper Mississippi River may be reconciled if large-diameter, loosely placed rocks ($\geq$2 feet in diameter) are used when revetments are constructed.

Pennington, C. H., J. A. Baker, and C. L. Bond. 1983. Fishes of selected aquatic habitats on the lower Mississippi River. Technical Report E-83-2, U. S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Fishes in a 60-mile reach of the lower Mississippi River near Vicksburg, Mississippi, were sampled to determine species diversity, abundance, and distribution in dike fields (series of transverse and vane dikes and the associated pools and bars), revetted banks, natural banks, and an abandoned river channel. Fish were sampled with gill nets, hoop nets, electrofishing, seines, and minnow traps. Dike fields harbored considerably more species than the other habitats and appeared to provide suitable habitats for many life history stages, from larvae to adults. Occurrence of age-0 fish of numerous species indicated the importance of dike fields as rearing areas. Revetted and natural banks supported similar fish species overall, but revetted banks supported the highest percentage, by weight, of fish with sporting or commercial value. Abundances and total weights of these species were lowest in the abandoned channel.

Despite the differences in number of species, catch-per-unit-effort (both in number and weight) was not greater in the dike fields than the other habitat types. The authors noted that comparisons among the four habitat types were accurate only assuming that the equipment used in each habitat type adequately sampled the fish occurring there. The authors opined that this assumption was not strictly met. In addition, differences in fish assemblages among the habitats were less distinct during high-water periods than low-water periods, probably because of decreased habitat segregation and increased fish movement. Certain times of the year precluded the use of some types of sampling equipment, which also may have contributed to the lack of distinctness among assemblages.

Pennington, C. H., J. A. Baker, and M. E. Potter. 1983. Fish populations along natural and revetted banks on the lower Mississippi River. North America Journal of Fisheries Management 3:204-211.

Fish populations along two natural and two revetted banks on the lower Mississippi River near Greenville, Mississippi, were sampled with baited hoop nets and electrofishing. Numbers of species collected in both habitats were similar, with 24 species collected along natural banks and 27 species collected along revetted banks. Six species were significantly more abundant along revetted banks, while four were more abundant at the natural banks. Species considered to have sport or commercial value were, in aggregate, more abundant by weight along revetted banks than natural banks. Fish abundances at the two natural banks were similar year-round, whereas abundances at the two revetted banks were more variable, suggesting movements to and from or between other habitats.

Hjort, R. C., P. L. Hulett, L. D. LaBolle, and H. W. Li. 1984. Fish and invertebrates of revetments and other habitats in the Willamette River, Oregon. Technical Report E-84-9, prepared by Oregon State University for the U. S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Physical and biological characteristics of revetted riverbanks, unaltered riverbanks, and secondary channels were compared on the Willamette River, Oregon, at low (221-238 m^3/sec) and moderate (283-425 m^3/sec) flows. Higher total numbers of invertebrates were collected from revetted banks than natural banks and the diversity of benthic invertebrates at revetted riverbanks was comparable to that of unaltered riverbanks. High densities of small fishes characterized fish assemblages at revetted riverbanks, but species diversity was lower than at the unaltered riverbanks. Low densities of large fish characterized the unaltered banks. Catches of fishes in the secondary channels were low in number of individuals and number of species compared to the other locations. Logs and overhanging vegetation may have precluded some areas within the secondary channels from effective electrofishing, thereby causing the low catches there. Although revetted banks supported higher densities of fish than unaltered banks, the authors cautioned that revetted banks may reduce total habitat area and diversity over time, but their study did not address such effects.

Coldwater

Beamer, E. M., and R. A. Henderson. 1998. Juvenile salmonid use of natural and hydromodified stream bank habitat in the mainstem Skagit River, northwest Washington. Miscellaneous Report, Skagit System Cooperative, LaConner, Washington.

Juvenile salmonid (chinook, coho, and chum salmon, and rainbow trout) use of paired natural and modified streambanks along 80 miles of the Skagit River, Washington, was compared using grid-point electrofishing. Wood cover along banks was the primary determinant of juvenile chinook and coho salmon abundances, and because natural banks had more and more-complex wood cover, these species were more abundant along natural banks than nearby riprapped banks. Similarly, juvenile chum salmon preferred banks with aquatic plants and cobble substrates, and because these cover types were more common along natural banks than riprapped banks, chum salmon abundances were greater along natural streambanks than those modified with riprap. Conversely, juvenile rainbow trout were more abundant in modified banks with boulder-sized riprap (≥256 mm in diameter) than along natural banks, but the reverse was true for banks modified with cobble-sized (64 mm to 256 mm diameter) riprap. Incorporation of wood and plant cover into riprap banks, and use of boulder-sized riprap, may therefore mitigate localized deleterious effects of bank modification. However, the authors cautioned that such measures may not mitigate the effects of reduced channel migration and avulsion rates caused by bank stabilization programs on habitat characteristics of long river reaches (but then again, their study did not specifically address such effects).

Peters, R., B. R. Missildine, and D. L. Low. 1998. Seasonal fish densities near riverbanks treated with various stabilization methods. First year report of the Flood Technical Assistance Project. U. S. Fish and Wildlife Service, North Pacific Coast Ecoregion, Western Washington Office, Aquatic Resources Division, Lacey, Washington.

Determination of which bank stabilization methods supported the greatest fish densities was attempted by conducting snorkel surveys at 2 to 8 sites in each of 15 rivers in western Washington. In general, sub-yearling cutthroat and steelhead trout, coho salmon, and chinook salmon were lower at riprap-stabilized banks than natural banks. In contrast, yearling and older trout densities were unaffected or increased at riprap-stabilized banks. Fish densities were generally greater at current deflector-stabilized banks than natural banks in winter. Large woody debris (LWD) incorporated into riprap did not increase fish densities. Large woody debris incorporated into current deflectors appeared to increase fish densities, but the effect was not statistically significant. The authors believed that the LWD was a negligible enhancement to bank stabilization structures because it was poorly placed, small in size, and lacked complexity of shape. They noted that conclusions from this study were based on small sample sizes and that more data may result in different conclusions. For example, total fish densities in the spring were on the average 20,000 fish/km fewer at current deflector-stabilized sites than control sites. However, the statistical conclusion for this test showed no significant difference, because of small sample sizes (i.e., too few study sites).

Avery, E. L. 1995. Effects of streambank riprapping on physical features and brown trout standing stocks in Millville Creek. Research Report 167, Wisconsin Department of Natural Resources.

Millville Creek, a small (7 to 10 cfs during summer) brown trout stream in southwestern Wisconsin was riprapped to counteract effects of bank degradation caused by dairy-cattle grazing and row-crop farming in the riparian zone. Prior to treatment, the stream was characterized by near absence of riparian woody vegetation, unstable streambanks, and extreme streambank erosion. Following riprapping, mean stream depth increased and density of brown trout increased significantly from 65 fish/mile to 102 fish/mile. Although the author did not consider the increase worth the cost ($26,000/mile), the study shows that riprapping can have beneficial effects for trout in severely degraded systems.

Lister, D. B., R. J. Beniston, R. Kellerhals, and M. Miles. 1995. Rock size affects juvenile salmonid use of streambank riprap. Pages 621-632*in* C. R. Thorne, S. R. Abt, F. B. J. Barends, S. T. Maynord, and K. W. Pilarczyk, editors. River, coastal and shoreline protection: erosion control using riprap and armourstone. John Wiley and Sons Ltd., New York.

Assessment of bank stabilization effects on fish was conducted on the Thompson and Coldwater rivers in British Columbia. Snorkel surveys on the Thompson River in summer and winter revealed that large riprap (rock >30 cm in diameter) supported higher chinook salmon and steelhead trout densities than small riprap (≤30 cm in diameter) or natural cobble-boulder banks. Chinook salmon, steelhead trout, and hatchery-reared coho salmon densities were greater in large riprap than small riprap on the Coldwater River in summer. Placing large (1.0 to 1.5 m diameter) boulders along the toe (intersection of bank with edge of the channel) of the bank on the Coldwater River appeared to increase rearing densities of all salmonids except sub-yearling steelhead trout. The authors concluded that size modifications to standard riprap rock could increase fish habitat value. They cautioned however, that no single design prescription would be appropriate for all rivers because the size of rock required to increase fish habitat value is dependent on the hydraulic and biological requisites of the particular river. Patterns of fish habitat use should also be known because requirements may vary from case to case, depending on species, life stage and other factors.

Knudsen, E. E., and S. J. Dilley. 1987. Effects of riprap bank reinforcement on juvenile salmonids in four western Washington streams. North American Journal of Fisheries Management 7:351-356.

Summer and fall juvenile salmonid abundances were estimated on four streams in central western Washington shortly before and after the banks were stabilized with riprap. Electrofishing and seining were used for capturing fish for mark-recapture analyses. Biomasses of juvenile coho salmon, juvenile steelhead, and cutthroat trout decreased after long lengths of bank were stabilized in the smaller streams. In larger streams, slight reductions in numbers of juvenile coho salmon and young-of-the-year cutthroat trout occurred, but numbers of yearling steelhead and cutthroat trout increased. The authors surmised that short-term negative effects of riprap construction were greater on smaller salmonids than larger salmonids in large streams, and that effects were more severe in smaller streams than large streams.

Michny, F. 1987. Sacramento River Chico Landing to Red Bluff Project 1986 juvenile salmon study. U. S. Fish and Wildlife Service, Division of Ecological Services, Sacramento, California.

This study evaluated juvenile salmon use of alternatives to standard riprap bank stabilization. Juvenile salmon were observed and counted (not netted) on the water surface after being shocked by an electrofishing boat. Salmon abundances were greatest at the natural banks and lowest at the standard rock revetments. Salmon abundances were intermediate at modified revetted banks, which were either covered with 1 to 4 inch river-run gravel or incorporated a 5:1 "fish rearing slope." The author concluded that rearing habitat values of standard riprap were substantially lower than natural banks, but that modifications to standard riprap reduced rearing habitat loss.

Elser, A. A. 1968. Fish populations of a trout stream in relation to major habitat zones and channel alterations. Transactions of the American Fisheries Society 97:389-397.

Physical stream characteristics and trout abundances were compared in altered and natural sections of Little Prickly Pear Creek north of Helena, Montana. Altered sections were channelized and riprapped in association with railroad and highway construction. Channelized sections were uniformly shallow and homogeneous, whereas unaltered sections varied in depth and alternated between pools and riffles. Brown and rainbow trout were significantly more abundant (by up to 78%) in the unaltered sections than in the altered sections, and non-salmonid fishes were almost completely absent in the altered reaches. Pairs of transverse rock deflectors installed as velocity checks to improve habitat quality in the highway segment resulted in physical stream conditions nearly comparable to unaltered sections, except for absence of vegetative cover. Fish abundances there remained depressed, but the author postulated that the situation would improve with time, given that the alterations were recent.

Literature Cited

Alabyan, A. M., and R. S. Chalov. 1998. Types of river channel patterns and their natural controls. Earth Surface Processes and Landforms 23:467-474.

Allan, J. D. 1995. Stream ecology: Structure and function of running waters. Chapman and Hall, London.

Andrews, E. D. 1980. Effective and bankfull discharges in the Yampa River Basin, Colorado and Wyoming. Journal of Hydrology 46:311-330.

Andrews, E. D., and J. M. Nankervis. 1995. Effective discharge and the design of channel maintenance flows for gravel-bed rivers. Pages 151-164*in* J. E. Costa, A. J. Miller, K. W. Potter, and P. R. Wilcock, editors. Natural and anthropogenic influences in fluvial geomorphology. Geophysical Monograph Series 89. American Geophysical Union.

Avery, E. L. 1995. Effects of streambank riprapping on physical features and brown trout standing stocks in Millville Creek. Wisconsin Department of Natural Resources Research Report 167.

Backiel, T., and T. Penczak. 1989. The fish and fisheries in the Vistula River and its tributary, the Pilica River. Pages 488-503 *in* D. P. Dodge, editor. Proceedings of the International Large River Symposium (LARS). Canadian Special Publication of Fisheries and Aquatic Sciences 106, Department of Fisheries and Oceans, Ottawa.

Baker, J. A., R. L. Kasul, L. E. Winfield, C. R. Bingham, C. H. Pennington, and R. E. Coleman. 1988a. An ecological investigation of revetted and natural bank habitats in the lower Mississippi River. Report 9, Lower Mississippi River Environmental Program, U.S. Army Corps of Engineers, Mississippi River Commission, Vicksburg, Mississippi.

Baker, J. A., R. L. Kasul, L. E. Winfield, C. R. Bingham, C. H. Pennington, and R. E. Coleman. 1988b. An ecological investigation of the Baleshed Landing–Ben Lomond and Ajax bar dike systems in the lower Mississippi River, river miles 481 to 494 AHP. Report 12, Lower Mississippi River Environmental Program, U.S. Army Corps of Engineers, Mississippi River Commission, Vicksburg, Mississippi.

Baker, J. A., C. H. Pennington, C. R. Bingham, and L. E. Winfield. 1987. An ecological review of five secondary channel habitats in the lower Mississippi River. Report 7, Lower Mississippi River Environmental Program, U.S. Army Corps of Engineers, Mississippi River Commission, Vicksburg, Mississippi.

Baltz, D. M., B. Vondracek, L. R. Brown, and P. B. Moyle. 1987. Influence of temperature on microhabitat selection choice by fishes in a California stream. Transactions of the American Fisheries Society 116:12-20.

Baltz, D. M., B. Vondracek, L. R. Brown, and P. B. Moyle. 1991. Seasonal changes in microhabitat selection by rainbow trout in a small stream. Transactions of the American Fisheries Society 120:166-176.

Bayley, P. B. 1991. The flood pulse advantage and the restoration of river-floodplain systems. Regulated Rivers: Research and Management 6:75-86.

Beamer, E. M., and R. A. Henderson. 1998. Juvenile salmonid use of natural and hydromodified stream bank habitat in the mainstem Skagit River, northwest Washington. Skagit System Cooperative, LaConner, Washington.

Beckett, D. C., C. R. Bingham, and L. G. Sanders. 1983. Benthic macroinvertebrates of selected habitats of the lower Mississippi River. Journal of Freshwater Ecology 2:247-261.

Benda, L. 1990. The influence of debris flows on channels and valley floors in the Oregon Coast Range, U.S.A. Earth Surfaces Processes and Landforms 15:457-466.

Benke, A. C., R. L. Henry, D. M. Gillespie, and R. J. Hunter. 1985. Importance of snag habitat for animal production in southeastern streams. Fisheries 10:8-13.

Beschta, R. L., and W. S. Platts. 1986. Morphological features of small streams: significance and function. Water Resources Bulletin 22:369-379.

Binns, N. A. 1994. Long-term responses of trout and macrohabitats to habitat management in a Wyoming headwater stream. North American Journal of Fisheries Management 14:87-98.

Bjornn, T. C. 1971. Trout and salmon movements in two Idaho streams as related to temperature, food, stream flow, cover, and population density. Transactions of the American Fisheries Society 100:423-438.

Brevard, J., C. Amoros, and G. Pautou. 1986. Impact of civil engineering works on the successions of communities in a fluvial system. Oikos 47:92-111.

Brookes, A. 1988. Channelized rivers–perspectives for environmental management. John Wiley Interscience, New York.

Bulkley, R. V., R. W. Bachman, K. D. Carlander, H. L. Fierstine, L. R. King, B. W. Menzel, A. L. Witten, and D. W. Zimmer. 1976. Warmwater stream alteration in Iowa. Extent, effects on habitat, fish, and fish food, and evaluation of stream improvement structures (summary report). U.S. Fish and Wildlife Service Report FWS/OBS-76/16.

Burress, R. M., D. A. Krieger, and C. H. Pennington. 1982. Aquatic biota of bank stabilization structures on the Missouri River, North Dakota. Technical Report No. E-82-6. U.S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Bustard, D. R., and D. W. Narver. 1975. Aspects of the winter ecology of juvenile coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). Journal of the Fisheries Research Board of Canada 32:667-680.

Campbell, R. F., and J. H. Neuner. 1985. Seasonal and diurnal shifts in habitat utilized by resident rainbow trout in western Washington Cascade Mountain streams. Pages 39-48 *in* F. W. Olsen, R. G. White, and R. H. Hamre, editors. Proceedings of the symposium on small hydropower and fisheries. Western Division and Bio-Engineering Section, American Fisheries Society, Bethesda, Maryland.

Contor, C. R., and J. S. Griffith. 1995. Nocturnal emergence of juvenile rainbow trout from winter concealment relative to light intensity. Hydrobiologia 299:179-183.

Cunjak, R. A., and G. Power. 1987. Cover use by stream-resident trout in winter: A field experiment. North American Journal of Fisheries Management 7:539-544.

Cunjak, R. A. and G. Power. 1986. Winter habitat utilization by stream resident brook trout (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*). Canadian Journal of Fisheries and Aquatic Sciences 43:1970-1981.

Dardeau, E. A., Jr., K. J. Killgore, Jr., and A. C. Miller. 1995. Using riprap to create or improve riverine habitat. Pages 609-620 *in* C. R. Thorne, S. R. Abt, B. J. Barends, S. T. Maynord, and K. W. Pilarczyk, editors. River, coastal and shoreline protection: erosion control using riprap and armourstone. John Wiley and Sons, New York.

Dister, E., D. Gomer, P. Obrdlik, P. Petermann, and E. Schneider. 1990. Water management and ecological perspectives of the upper Rhine's floodplains. Regulated Rivers: Research and Management 5:1-15.

Downing, D. C. 2000. Spawning and rearing ecology of Madison River rainbow trout in relation to whirling disease infection risk. Master's thesis. Montana State University, Bozeman.

Dunne, T., and L. B. Leopold. 1978. Water in environmental planning. W. H. Freeman, San Francisco.

Elser, A. A. 1968. Fish populations of a trout stream in relation to major habitat zones and channel alterations. Transactions of the American Fisheries Society

97:389-397.

Farabee, G. B. 1986. Fish species associated with revetted and natural main channel border habitats in Pool 24 of the upper Missouri River. North American Journal of Fisheries Management 6:504-508.

Frissel, C. A., and R. K. Nawa. 1992. Incidence and causes of physical failure of artificial habitat structures in streams of western Oregon and Washington. North American Journal of Fisheries Management 12:182-197.

Funk, J. L., and J. W. Robinson. 1974. Changes in the channel of the lower Missouri River and effects on fish and wildlife. Aquatic Series No. 11, Missouri Department of Conservation.

Gordon, N. D., T. A. McMahon, and B. L. Finlayson. 1992. Stream hydrology: an introduction for ecologists. John Wiley and Sons, New York.

Greenberg, L., P. Svendsen, and A. Harby. 1996. Availability of microhabitats and their use by brown trout (*Salmo trutta*) and grayling (*Thymallus thymallus*) in the River Vojman, Sweden. Regulated Rivers: Research and Management 12:287-303.

Gregory, K. J., and D. E. Walling. 1973. Drainage basin form and process–a geomorphological approach. Halsted Press. John Wiley and Sons, New York.

Griffith, J. S., and R. W. Smith. 1993. Use of winter concealment cover by juvenile cutthroat and brown trout in the South Fork of the Snake River, Idaho. North American Journal of Fisheries Management 13:823-830.

Hartman, G. F. 1963. Observations on behavior of juvenile brown trout in a stream aquarium during winter and spring. Journal of the Fisheries Research Board of Canada 23:769-787.

Heede, B. H. 1986. Designing for dynamic equilibrium in streams. Water Resources Bulletin 22:351-357.

Heede, B. H., and J. N. Rinne. 1990. Hydrodynamic and fluvial morphologic processes: implications for fisheries management and research. North American Journal of Fisheries Management 10:249-268.

Heggenes, J. 1988. Substrate preferences of brown trout (*Salmo trutta*) in artificial stream channels. Canadian Journal of Fisheries and Aquatic Sciences 45:1801-1806.

Heggenes, J., O. M. Krog, O. R. Lindas, J. G. Dokk, and T. Bremnes. 1993. Homeostatic behavioural responses in a changing environment: brown trout (*Salmo trutta*) become nocturnal during winter. Journal of Animal Ecology

62:295-308.

Hillman, T. W., J. S. Griffith, and W. S. Platts. 1987. Summer and winter habitat selection by juvenile chinook salmon in a highly sedimented Idaho stream. Transactions of the American Fisheries Society 116:185-195.

Hjort, R. C., P. L. Hulett, L. D. LaBolle, and H. W. Li. 1984. Fish and invertebrates of revetments and other habitats in the Willamette River, Oregon. Technical Report E-84-9, U.S. Army Engineer Waterway Experiment Station, Vicksburg, Mississippi.

Hunt, R. L. 1988. A compendium of 45 trout stream habitat development evaluations in Wisconsin during 1953-1985. Wisconsin Department of Natural Resources Technical Bulletin 162.

Hunter, C. J. 1991. Better trout habitat–a guide to stream restoration and management. Island Press, Washington, D.C.

Jude, D. J., and S. F. DeBoe. 1996. Possible impact of gobies and other introduced species on habitat restoration efforts. Canadian Journal of Fishery and Aquatic Sciences 53:136-141.

Junk, W. J., P. B. Bayley, and R. E. Sparks. 1989. The flood pulse concept in river-floodplain systems. Pages 110-117 *in* D. P. Dodge, editor. Proceedings of the International Large River Symposium (LARS). Canadian Special Publication of Fisheries and Aquatic Sciences 106, Department of Fisheries and Oceans, Ottawa.

Jurajda, P. 1995. Effect of channelization and regulation on fish recruitment in a flood plain river. Regulated Rivers: Research and Management 10:207-215.

Kallemeyn, L. W., and J. F. Novotny. 1977. Fish and food organisms in various habitats of the Missouri River in South Dakota, Nebraska, and Iowa. FWS/OBS-77.25, U.S. Fish and Wildlife Service, National Stream Alteration Team, Columbia, MO.

Klingeman, P. C., R. L. Beschta, P. D. Komar, and J. B. Bradley, editors. 1999. Gravel bed rivers in the environment. Water Resource Publications, Highlands Ranch, Colorado.

Knapp, R. A., V. T. Vredenberg, and K. R. Matthews. 1998. Effects of stream channel morphology on golden trout spawning habitat and recruitment. Ecological Applications 8:1104-1117.

Knight, S. S., and C. M. Cooper. 1991. Effects of bank protection on stream fishes. Pages 13-34–13-39 *in* Proceedings of the Fifth Federal Interagency Sedimentation Conference. Federal Energy Regulatory Commission, Washington, D.C.

Knighton, A. D., and G. C. Nanson. 1993. Anastomosis and the continuum of channel pattern. Earth Surface Processes and Landforms 18:613-625.

Knudsen, E., and S. J. Dilley. 1987. Effects of riprap bank reinforcement on juvenile salmonids in four western Washington streams. North American Journal of Fisheries Management 7:351-356.

Leopold, L. B., M. G. Wolman, and J. P. Miller. 1964. Fluvial processes in geomorphology. W. H. Freeman, San Francisco.

Li, H. W., C. B. Schreck, and R. A. Tubb. 1984. Comparison of habitats near spur dikes, continuous revetments, and natural banks for larval, juvenile, and adult fishes of the Willamette River. Water Resources Research Institute, Oregon State University, Corvallis.

Lisle, T. E. 1982. Effects of aggradation and degredation on riffle-pool morphology in natural gravel channels, northwestern California. Water Resources Research 18:1643-1651.

Lister, D. B., R. J. Benniston, R. Kellerhals, and M. Miles. 1995. Rock size affects juvenile salmonid use of streambank riprap. Pages 621-632 *in* C. R. Thorne, S. R. Abt, B. J. Barends, S. T. Maynord, and K. W. Pilarczyk, editors. River, coastal and shoreline protection: erosion control using riprap and armourstone. John Wiley and Sons, New York.

Mathis, D. B., C. R. Bingham, and L. G. Sanders. 1982. Assessment of implanted substrate samples for macroinvertebrates inhabiting stone dikes of the lower Mississippi River. Miscellaneous Paper E-82-1, U.S. Army Engineer Waterways Experiment Station, CE, Vicksburg, Mississippi.

McClure, W. V. 1991. Initial effects of streambank stabilization on a small trout stream. M.S. thesis, Montana State University, Bozeman.

Mesick, C. F. 1988. Effects of food and cover on numbers of Apache and brown trout establishing residency in artificial stream channels. Transactions of the American Fisheries Society 117:421-431.

Messick, C. F. 1995. Response of brown trout to streamflow, temperature, and habitat restoration in a degraded system. Rivers 5:75-95.

Michny, F. 1988. Concluding report, evaluation of palisade bank stabilization, Woodson Bridge, Sacramento River, California. U.S. Fish and Wildlife Service, Division of Ecological Services, Sacramento, California.

Michny, F. 1987. Sacramento River, Chico Landing to Red Bluff project, 1986 juvenile salmon study. U.S. Fish and Wildlife Service, Division of Ecological Services, Sacramento, California.

Mitro, M. G. 1999. Sampling and analysis techniques and their application for estimating recruitment of juvenile rainbow trout in the Henrys Fork of the Snake River, Idaho. Doctoral dissertation, Montana State University, Bozeman.

Montgomery, D. R., and J. M. Buffington. 1997. Channel-reach morphology in mountain drainage basins. Geological Society of America Bulletin 109:596-611.

Moore, K. M. S., and S. V. Gregory. 1988. Response of young-of-the-year cutthroat trout to manipulation of habitat structure in a small stream. Transactions of the American Fisheries Society 117:162-170.

Murphy, M. L., and K. V. Koski. 1989. Input and depletion of woody debris in Alaska streams and implications for streamside management. North American Journal of Fisheries Management 9:427-436.

NRCS (USDA Natural Resources Conservation Service) and DEQ (Montana Department of Environmental Quality). 1998. Upper Yellowstone River physical features inventory, Gardiner to Springdale. Report to Upper Yellowstone River Task Force and the Park Conservation District.

Nicolas, Y., and D. Pont. 1997. Hydrosedimentary classification of natural and engineered backwaters of a large river, the lower Rhône: possible applications for the maintenance of high fish biodiversity. Regulated Rivers: Research and Management 13:417-431.

Odum, E. P., J. T. Finn, and E. H. Franz. 1979. Perturbation theory and the subsidy-stress gradient. Bioscience 29:349-352.

Pennington, C. H., J. A. Baker, and C. L. Bond. 1983a. Fishes of selected aquatic habitats on the Mississippi River. Technical Report E-83-2, U.S. Army Engineer Waterways Experiment Station, CE, Vicksburg, Mississippi.

Pennington, C. H., J. A. Baker, and M. E. Potter. 1983b. Fish populations along natural and revetted banks on the lower Mississippi River. North American Journal of Fisheries Management 3:204-211.

Pennington, C. H., S. S. Knight, and M. P. Farrell. 1985. Responses of fishes to revetment placement. Arkansas Academy of Science Proceedings 39:95-97.

Petays, A. M., T. Muotka, A. Hussko, P. Tikkanen, and P. Kreivi. 1997. Seasonal changes in habitat use and preferences by juvenile brown trout, *Salmo trutta*, in a northern boreal river. Canadian Journal of Fisheries and Aquatic Sciences 54:520-530.

Peters, J. C., and W. Alvord. 1964. Man-made channel alterations in thirteen Montana streams and rivers. Transactions of the North American Wildlife and Natural Resources Conference 29:93-102.

Peters, R., B. R. Missildine, and D. L. Low. 1998. Seasonal fish densities near river banks stabilized with various stabilization methods. First Year Report of the Flood Technical Assistance Project. U.S. Fish and Wildlife Service, North Pacific Coast Ecoregion, Western Washington Office, Aquatic Resources Division, Lacey, Washington.

Petts, G. E., H. Moller, and A. L. Roux. 1989. Historical change of large alluvial rivers: western Europe. John Wiley and Sons, New York.

Poizat, G., and D. Pont. 1996. Multi-scale approach to species-habitat relationships: juvenile fish in a large river section. Freshwater Biology 36:611-622.

Rimmer, D. M., U. Paim, and R. L. Saunders. 1983. Autumnal shift of juvenile Atlantic salmon (*Salmo salar*) in a small river. Canadian Journal of Fisheries and Aquatic Sciences 40:671-680.

Rimmer, D. M., U. Paim, and R. L. Saunders. 1984. Changes in the selection of microhabitat by juvenile Atlantic salmon (*Salmo salar*) at the summer-autumn transition in a small river. Canadian Journal of Fisheries and Aquatic Sciences 41:469-475.

Sanders, L. G., C. R. Bingham, and D. C. Beckett. 1986. Macroinvertebrate gear evaluation. Miscellaneous Paper E-86-3, U.S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Sandheinrich, M. B., and G. J. Atchison. 1986. Environmental effects of dikes and revetments on large riverine systems. Technical Report E-86-5, U.S. Army Engineer Waterway Experiment Station, Vicksburg, Mississippi.

Schiemer, F., and T. Spindler. 1989. Endangered fish species of the Danube River in Austria. Regulated Rivers: Research and Management 4:397-407.

Schiemer, F., T. Spindler, A. Wintersberger, A. Schneider, and A. Chovanec. 1991. Fish fry associations: Important indicators for the ecological status of larger rivers. Pages 2497-2500 *in* H. Kausch and W. Lampert, editors. International Association for Theoretical and Applied Limnology Congress, Munich 1989.

Schlosser, I. J. 1987. The role of predation in age- and size-related habitat use by stream fishes. Ecology 68:651-659.

Schmetterling, D. A., C. G. Clancy, and T. M. Brandt. In Press. Effects of riprap bank stabilization on stream salmonids in the western United States. Fisheries.

Schumm, S. A. 1977. The fluvial system. John Wiley Interscience, New York.

Shields, D. F. 1991. Woody vegetation and riprap stability along the Sacramento River Mile 84.5–119. Water Resources Bulletin 27:527-536.

Shields, F. D., Jr., C. M. Cooper, and S. S. Knight. 1993. Initial habitat response to incised channel rehabilitation. Aquatic Conservation: Marine and Freshwater Ecosystems 3:93-103.

Shields, F. D., Jr., C. M. Cooper, and S. Testa. 1995. Towards greener riprap: environmental considerations from microscale to macroscale. Pages 557-574 *in* C. R. Thorne, S. R. Abt, B. J. Barends, S. T. Maynord, and K. W. Pilarczyk, editors. River, coastal and shoreline protection: erosion control using riprap and armourstone. John Wiley and Sons, New York.

Shuler, S. W., R. B. Nehring, and K. D. Fausch. 1994. Diel habitat selection by brown trout in the Rio Grande River, Colorado, after placement of boulder structures. North American Journal of Fisheries Management 14:99-111.

Simons, D. B., and F. Senturk. 1977. Sediment transport technology. Water Resources Publications, Ft. Collins, Colorado.

Steiger, J., M. James, and F. Gazelle. 1998. Channelization and consequences on floodplain system functioning on the Garonne River, SW France. Regulated Rivers: Research and Management 14:13-23.

Stern, D. H., M. S. Stern, and Missouri Institute of River Studies. 1980. Effects of bank stabilization on the physical and chemical characteristics of streams and small rivers: a synthesis. U.S. Fish and Wildlife Service Report FWS/OBS-80/11.

Surian, N. 1999. Channel changes due to river regulation: the case of the Piave Piver, Italy. Earth Surface Processes and Landforms 24:1135-1151.

Thurow, R. F. 1988. Effects of stream alterations on rainbow trout in the Big Wood River, Idaho. Pages 175-188 *in* S. Wolfe, editor. Proceedings of the 68th Conference of the Western Association of Fish and Wildlife Agencies, Albuquerque, New Mexico.

U.S. Fish and Wildlife Service. 1992. Juvenile salmon study, Butte Basin reach, Sacramento River bank protection project. Final Report to U.S. Army Corps of Engineers, Sacramento, California.

White, R. J. 1991. Resisted lateral scour in streams–its special importance to salmonid habitat and management. American Fisheries Society Symposium 10:200-203.

Williams, G. P. 1978. Bank-full discharge of rivers. Water Resources Research 14:1141-1154.

Witten, A. L., and R. V. Bulkley. 1975. A study of the effects of stream channelization and bank stabilization on warmwater sport fish in Iowa. Subproject No. 2: A study of the impact of selected bank stabilization structures on game fish and associated organisms. U.S. Fish and Wildlife Service Report FWS/OBS-76/12.

Wolf, J., J. McMahon, and S. M. Diggins. 1972. Comparison of benthic organisms in semi-natural and channelized portions of the Missouri River. Proceedings of the South Dakota Academy of Science 51:160-167.