Fish Community Structure Associated With Bank Stabilization in the Metals-Contaminated Lower Coeur d'Alene River, Idaho

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Catherine A. Gidley

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Major Professor: Dennis Scarnecchia, Ph.D

Disclaimer

This thesis is a report of graduate student studies. Although U.S. Fish and Wildlife Service assisted with study design and data collection, conclusions do not represent official Fish and Wildlife Service position.

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Major Professor Dennis L. Scarnecchia Date 3-10-08
Dennis L. Scarnecchia

Committee Members James A Hansen Date 2-27-08
Jim Hansen
James Congleton Date 3-11-08
James Congleton

Department Administrator Kerry Paul Reese Date 3-10-08
Kerry P. Reese

Discipline's College Dean Date 12 March 2008
Steven B. Daley-Laursen

Final Approval and Acceptance by the College of Graduate Studies

Date 5/14/08
Margrit von Braun

Abstract

Fish sampling and habitat assessment were conducted at 24 sites in the lower 54 km of the Coeur d'Alene River, Idaho in 2005 and 2006 to 1) characterize four shoreline habitat types-- failing banks (FB), riprap (RR), riprap with vegetation (RRV), and vegetation (V)--according to ranked and quantifiable habitat variables, 2) assess differences in relative fish abundance (catch-per-unit effort; CPUE), species diversity, and community composition associated with the four shoreline habitat types, two sections (upstream and downstream), and three seasons (summer, spring, fall), 3) assess the relationships between relative fish abundance and habitat variables, and 4) assess the relationship between relative fish abundance and a) depth of riprap structure and b) riprap rock diameter. The four habitat types differed significantly in habitat characteristics based on Rapid Bioassessment Protocols (RBP) scores ($F=5.73$, $P<0.001$). All four habitat types scored poor or marginal in substrate/available cover, pool variability, sediment deposition, and riparian zone width. Relative fish abundance as measured by CPUE at stabilized (RR and RRV) shorelines was consistently higher than at unstabilized (FB and V) shorelines among all seasons. Relative fish abundance was not significantly different between stabilized and unstabilized habitat types for gillnetting ($F=1.95$, $P=0.167$), but was significantly higher at stabilized than unstabilized habitats for electrofishing ($F=5.66$, $P=0.020$). Differences in species diversity were only evident between sections, not among habitat types or seasons. Fish community differences were apparent among habitat types as well as between sections and among seasons. Brown bullhead *Ameirus nebulosus*, northern pike *Esox lucius*, and pumpkinseed *Lepomis gibbosus* were captured in significantly higher numbers at stabilized than unstabilized

sites and longnose suckers *Catostomus catostomus* were captured in greater numbers at unstabilized than stabilized sites. Water temperature and spawn timing influenced fish abundance and composition in spring. Large diameter riprap supported a higher abundance of fish than did smaller diameter riprap. Overall, stabilized shorelines on the lower Coeur d'Alene River were not found to be adversely affecting overall fish relative abundance, diversity, and species composition under the existing conditions of a low percentage of banks stabilized with riprap (2.5%). Instead, stabilized shorelines provided beneficial habitat in a river system with low quality and diversity of available habitats. This result should not be predicted, however, to apply as increasingly high percentages of the river bank are stabilized, and loss of river function occurs.

Data from the above sampling were also analyzed to assess possible impact of piscivores (smallmouth bass *Micropterus dolomieui*, northern pikeminnow *Ptychocheilus oregonensis*, and northern pike) on native salmonids, especially Westslope cutthroat trout *Oncoryhynchus clarki lewisi*. The objectives were to 1) evaluate and quantify salmonid use of stabilized and unstabilized shoreline habitats by season and river section, 2) evaluate and quantify piscivorous species use of stabilized and unstabilized shoreline habitat by season and river section, and 3) determine if an overlap exists between salmonid and piscivore use of stabilized and unstabilized shoreline habitats by season and river section. In all, 81 salmonids were captured, or 2% of the total fish catch. Salmonid catch in the lower river was greatest during spring when water temperatures were low and juveniles were outmigrating to Lake Coeur d'Alene. No significant differences in salmonid catch were evident between stabilized and unstabilized habitats ($\chi2=0.064$, $P=0.800$), though juvenile westslope cutthroat trout showed some affinity for stabilized areas with 10 out of 12 individuals captured at these sites. In all, 670 piscivores were captured, or 19% of the total fish catch. Piscivore catch was significantly lower in spring, when salmonid numbers were highest, than in fall and summer, when salmonid numbers were lowest ($\chi 2$=17.465, P<0.001). Overall, piscivores were not captured in significantly different numbers at stabilized and unstabilized habitats ($\chi 2$=0.243, P=0.622); however, northern pike (N=22) were captured in significantly higher numbers at stabilized habitats. The overall effects of habitat type on salmonid and piscivore overlap were not clear. Based on the data available, season seems more important than habitat in affecting salmonid and piscivore impacts. Specific studies are outlined that need to be conducted for a clearer understanding of the relation between salmonids and potential predation from piscivores in the lower river.

In comparison of catches between the two gears (gillnetting and electrofishing), species composition ($\chi 2$=831.46, P<0.001) and length selectivity (t=37.86, P<0.001) were significantly different. Electrofishing captured a greater numbers of individuals (N=2,915) than gillnetting (N=596), but individuals were much smaller for electrofishing (mean length 96 mm) than for gillnetting (mean length 331 mm). Gillnets more readily captured longnose suckers and largescale suckers *Catostomus macrocheilus* (50% of total catch), whereas electrofishing captured larger numbers of yellow perch *Perca flavescens* and pumpkinseed (54% of total catch). The use of these two gears together provided a more representative sample of the fish community than either gear could have provided alone.

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I would especially like to thank U.S. Fish and Wildlife Service, Idaho Fish and Game, my advisor, and my committee members for all of their help and support. A special thanks goes out to Jim Hansen for being involved from start to finish. I would also like to thank all of the people who influenced the path of my fisheries career and my husband Lance for his patience and ability to keep me smiling.

Table of Contents

List of Figures

List of Tables

Background

The Coeur d'Alene River Basin in northern Idaho has been the site of mining activity since the discovery of gold in the basin in the 1880's (Stoll 1932). The area has long been noted as one of the leading silver-lead and zinc producing areas in the world, and has also yielded large amounts of cadmium, copper, antimony, and gold (Mink 1971). Mining operations in the basin have continued for over a century and today there are more than 200 sites affected by mining located within the basin (Maret and MacCoy 2002).

From the onset of mining activities until the late 1960s when settling ponds were installed, mining wastes were either left at the mine site or discharged into the river (Mink 1971). Most of the mine tailings initially settled in the upper South Fork of the river below the mining sites. By 1932, tailings were heavily deposited at Mission Flats more than 50 km downstream of the mines, resulting in a broader, shallower river channel (Ellis 1932). The Mission Flats area of the river, near the Cataldo Mission, was reported by steamboat and tugboat captains to be as deep as 10-15 m in the late 1800's. By 1932, depths were only 4-5 m (Ellis 1932) and have become even shallower by 2007.

Water quality in and downstream of the Coeur d'Alene basin has been impacted by the presence of metals from mine tailings entering the river system (Mink 1971, Funk et al. 1975, Maret and MacCoy 2002). The river, as well as Lake Coeur d'Alene and the Spokane River, which flows out of the lake (Figure 1), have high levels of metals in water and sediments including zinc, copper, cadmium, lead, and arsenic (Mink 1971, Wissmar 1972, Funk et al. 1975, Sprenke et al. 2000, Maret and MacCoy 2002). Mink (1971) raised concerns regarding the hydrograph of the river and its effects on transport of metals during a water quality assessment of the South Fork and lower Coeur d'Alene River. Mink (1971)

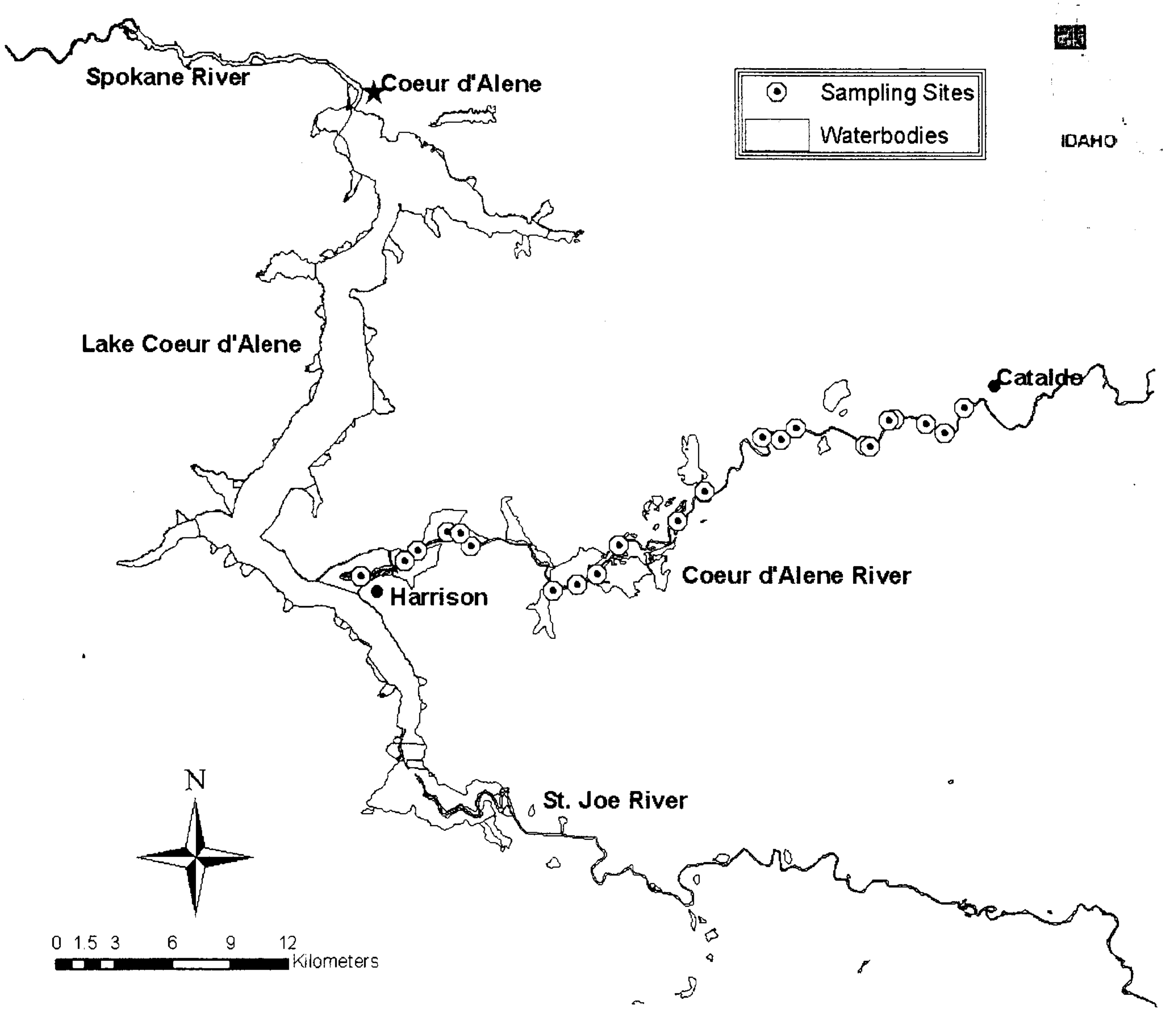


Figure 1. Map showing the Coeur d'Alene Basin, Idaho and the 24 sampling site locations and their division into upstream and downstream sections.

indicated that zinc and other elements entered the river during high spring flows, evidently from old tailings upstream. Frequent floods continue to re-suspend sediments previously deposited on riverbanks, impacting water quality in the Coeur d'Alene River as well as Lake Coeur d'Alene and the Spokane River.

Increased concentrations of metals in the lower Coeur d'Alene River have been associated with a depletion of aquatic life. An extensive biological survey was conducted on the Coeur d'Alene River in 1929 and 1930. Ellis (1932) found little aquatic life (benthic invertebrates, phytoplankton, and zooplankton). No live fish of any species was reported from below the confluence of the North and South Forks to the mouth of the river near the town of Harrison, Idaho, nor in the South Fork below the town of Wallace, located just downstream of the mining activity. Mink (1971) found zinc and cadmium concentrations above toxic limits for survival of fish. In the summer of 1973, rainbow trout *Oncorhychus mykiss* placed in live boxes on the South Fork all died within 48 hours (Bowler 1974). Sappington (1969) reported 96-h lethal limit levels of 0.09 mg/L for Zn in westslope cutthroat trout *Oncorhynchus clarki lewisi.* Zinc levels during the time of Sappington's study in water from the lower river were reported at 1.55-4.7 mg/L, well above the 96-h lethal limit (Funk et al.1975).

In contrast to the lower river and South Fork, areas upstream of the mining activity were reported to contain aquatic invertebrates from the families Trichoptera, Plecoptera, and Ephemeroptera (Ellis 1932, Hoiland 1992, Hoiland et al. 1994), which are typically indicators of healthy aquatic systems (Barbour et al. 1999). Furthermore, the chain lakes, 11 lakes connected to the lower river (Figure 2), were inhabited by several species of fish (Ellis

1932). More recent studies have shown an increase in fish and other aquatic life throughout the lower river and South Fork (Laumeyer 1976, Hoiland 1992, Hoiland et al.1994, Maret and MacCoy 2002), although concentrations of metals in the water and sediments throughout the basin remain high.

Remediation efforts are currently active throughout the basin. In 1983, the EPA designated a 34 square kilometer area on the South Fork, heavily polluted by mining wastes, as the Bunker Hill Superfund Site. The site was originally separated into two operable units: operable unit 1 (OU1) and operable unit (OU2). In 1998, the EPA extended its remediation efforts outside this 34 km^2 area with the creation of operable unit 3 (OU3), which includes the entire basin as well as Lake Coeur d'Alene and the Spokane River (NRC 2005). The Basin Environmental Improvement Project Commission (BEIPC) was created by the Idaho Legislature in 2001, as the authority responsible for implementing cleanup efforts in the Coeur d'Alene basin, pursuant to the Superfund Act. The BEIPC has authority under the EPA to implement the Record of Decision (ROD; EPA 2002, BEIPC 2004), finalized in 2002, in order to advance cleanup efforts in OU 3 of the Bunker Hill complex. Under the ROD, the Coeur d'Alene Lake Management Plan (1995) and a subsequent addendum (2003), bank stabilization was identified as one possible method to reduce accelerated bank erosion, and thereby reduce suspension of nutrients and metals, in the lower Coeur d'Alene River. In 2003, Congress allocated Clean Water Act funding to the BEIPC, which authorized studies to identify and test methods appropriate for stabilizing failing banks on the lower Coeur d'Alene River consistent with the EPA ROD and lake improvement goals. As of 2007, bank stabilization only constitutes about 2.8% of the total riverbank on the lower 54 km of river, and riprap totals 90% (or 2.5% of the total riverbank) of these stabilized shorelines (KSSWCD 2004). Interest has been expressed, however, in stabilizing much of the lower river and many homeowners are stabilizing shorelines on their properties to prevent erosion. On the lower Coeur d'Alene River permits must be obtained from the U.S. Army Corps of Engineers (under Section 404), as well as from the Idaho Department of Water Resources and Idaho Department of Lands (under the Lake Protection Act), depending on location (Ken Knoblock Idaho Department of Water Resources (IDWR), Personal Communication). No additional guidelines for bank stabilization exist at this time for the lower Coeur d'Alene River.

In any ongoing and proposed remediation of the lower Coeur d'Alene River with bank stabilization, it is important to evaluate the baseline fish community and how that fish community is and will be affected by present and planned bank stabilization structures. The U.S. Fish and Wildlife Service (USFWS) raised concerns regarding the possible effects of bank stabilization on salmonids, particularly westslope cutthroat trout and bull trout *Salvelinus confluentus*. The native salmonids use the lower river primarily as a migration corridor between upriver rearing areas and Lake Coeur d'Alene, with possible use of the chain lakes. The USFWS has expressed particular concern that extensive bank stabilization has the potential to increase the abundance and affect the behavior of piscivores, including northern pikeminnow *Ptychocheilus oregonensis*, smallmouth bass *Micropterus dolomieui*, and northern pike *Esox lucius*, species that have previously been reported to prey on juvenile salmonids (Brown and Moyle 1981, Zimmerman 1999, Fresh et al. 2003, Fritts and Pearsons 2006).

Fish community monitoring in areas that are presently stabilized should be implemented with consideration of sampling gear. Gear type often significantly impacts fish catch and composition (both species and size) (Hubert 1983), so the use of several gears can provide a more accurate portrayal of abundance and diversity of species (Weaver et al. 1993).

Although bank stabilization has been applied in aquatic systems world-wide for reasons including flood control, road construction, and erosion control, relatively few studies have been conducted relative to the effects of bank stabilization on fish communities, and no such studies have been conducted on the lower Coeur d'Alene River.

The primary goal of this study was to develop and implement a general sampling design to provide 1) baseline information on the fish community structure in the lower river and 2) information on the relation between the bank stabilization structures currently in place and the fish abundance and community composition immediately associated with them. Any information obtained under this general sampling design related to salmonid and piscivore interactions would be useful in addressing possible impacts of piscivores on native salmonids. This research will provide a better understanding of the effects of bank stabilization on fish community structure, assess methods for monitoring fisheries with regard to bank stabilization, and provide information for future implementation of bank stabilization on the lower Coeur d'Alene River.

This thesis is divided into three chapters. In Chapter One, I investigated if stabilized and unstabilized shoreline habitats differ in terms of habitat characteristics, fish abundance, fish species diversity, and fish community composition. I assessed if there were significant differences in fish abundance associated with different characteristics of riprap structures. In Chapter Two, I investigated fish community structure in greater detail by examining comparative usage of unstabilized and stabilized shoreline habitats by salmonids and the piscivorous species that may prey upon them. In Chapter Three, I assessed differences in catch, fish community composition and diversity, and length selectivity sampled by two gears, electrofishing and gillnetting.

Study Area

The Coeur d'Alene basin is located within 150 km of two large population centers, Coeur d'Alene, Idaho and Spokane, Washington, and provides an array of recreational pursuits, including fishing and boating. The basin encompasses 10,360 km^2, draining the west slope of the Bitterroot Range between Montana and Idaho (Funk et al. 1975). The lower river (54 km in length) lies downstream of the confluence of the North and South Forks of the Coeur d'Alene River, and drains into Lake Coeur d'Alene near the town of Harrison (Figure 1). The North Fork is relatively free of mining activity, in sharp contrast to the heavily-impacted South Fork. Whereas the North and South Forks are high gradient streams in narrow valleys, the lower river is characterized by fine substrates, low gradient, and a meandering channel in a broad valley. The lower river is connected to 11 shallow chain lakes by natural streams and dredged channels (Figure 2). These lakes are generally relatively shallow, less than 9 m (Sprenke et al. 2000), with much of their shorelines blanketed in emergent or submerged aquatic vegetation (Bowles 1985, Rieman 1987). The lakes provide habitat for many warm- and coolwater species. All 11 lakes lie within the river floodplain, providing wetland habitat for waterfowl and fish spawning (Sprenke et al. 2000). Land use along the lower river is predominantly agricultural, residential, and recreational.

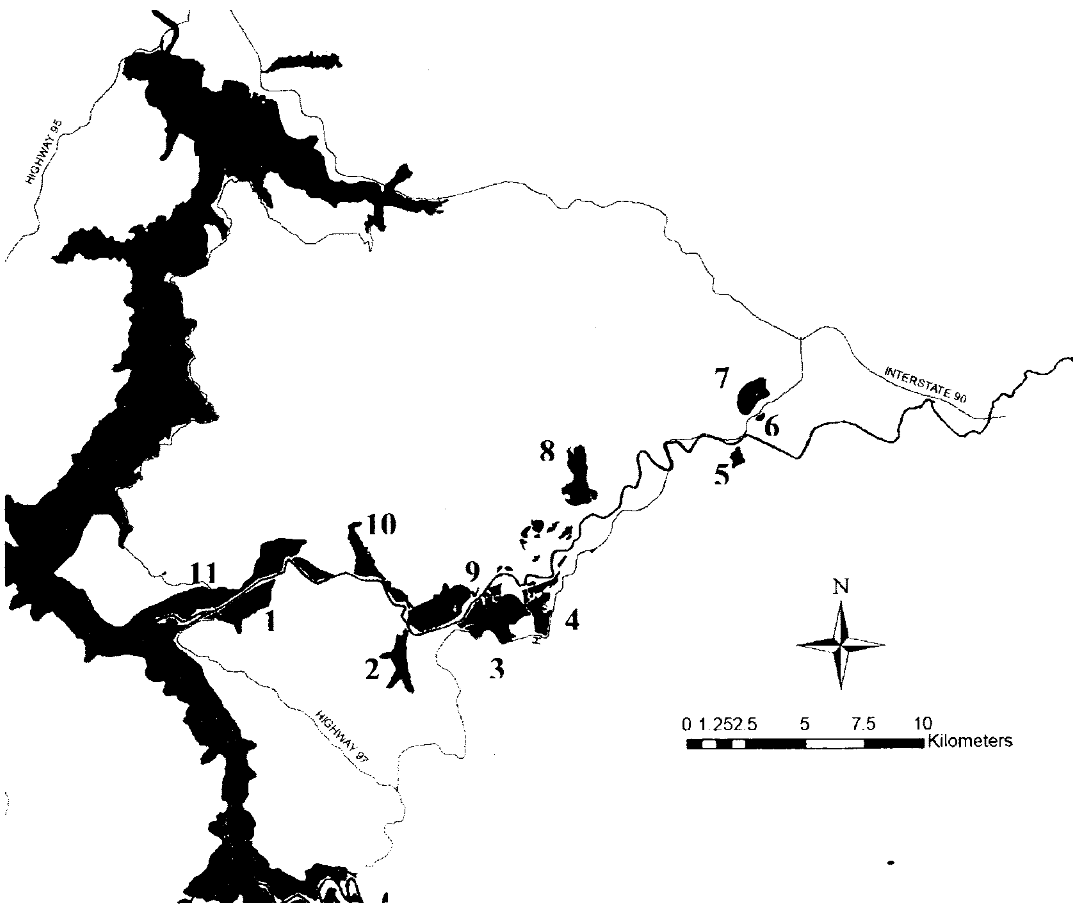


Figure 2. Map showing the Chain Lakes adjoining the lower Coeur d'Alene River, Idaho. The lakes include: (1) Anderson Lake (2) Black Lake (3) Cave Lake (4) Medicine Lake (5) Bull Run Lake (6) Porters Lake(7) Rose Lake (8) Killarney Lake (9) Blue Lake (10) Swan Lake (11)Thompson Lake.

The Union Pacific Railroad line, which formerly ran along much of the lower Coeur d'Alene River, has been converted into the Trail of the Coeur d'Alenes, a recreational trail spanning more than 118 km. The trail is managed by the Coeur d'Alene Tribe and Idaho Department of Parks and Recreation.

The lower river downstream of the Cataldo boat ramp (54 km) is considered slackwater created by Post Falls Dam. The dam, which is located on the Spokane River 14.5 km downstream of the Lake Coeur d'Alene outlet, regulates the lake level at 648.6 m (Avista 2005).

Few studies have been conducted on the lower Coeur d'Alene River to establish baseline fisheries information and fish community structure. In 2000, the USGS National Water Quality Assessment Program sought to evaluate fish assemblages, environmental variables, and associated mine densities throughout the basin. Eighteen fish species were collected from the South Fork of the Coeur d'Alene River representing the families Salmonidae, Cottidae, Cyprinidae, Catostomidae, Centrarchidae, and Ictaluridae (Maret and MacCoy 2002). Laumeyer (1976) reported a distribution of fish families, similar to that collected during USGS sampling efforts, throughout the river basin.

The basin as a whole contains a variety of coldwater, coolwater, and warmwater species, both native and non-native. Native species include westslope cutthroat trout, bull trout, northern pikeminnow, and mountain whitefish *Prosopium williamsoni*. Non-native species, which are present in both the river and lake systems, include rainbow trout, brook trout *Salvelinus fontinalis*, kokanee salmon *Oncorhynchus nerka*, and Chinook salmon *Oncorhynchus tshawytscha*. Warmwater species include largemouth bass, black crappie *Pomoxis nigromaculatus*, bluegill *Lepomis macrochirus*, brown bullhead, and tench *Tinca*

tinca, and coolwater species include yellow perch *Perca flavescens,* pumpkinseed *Lepomis gibbosus*, northern pikeminnow, smallmouth bass, and northern pike. The warmwater and coolwater species are present in Lake Coeur d'Alene, the Coeur d'Alene River, and the chain lakes. Coldwater species, including cutthroat trout, mountain whitefish, Chinook salmon, and kokanee salmon are present in the lower river during periods of low water temperatures as well as in Lake Coeur d'Alene.

Chapter One

Fish Community Structure Associated with Stabilized and Unstabilized Shoreline Habitats in the Coeur d'Alene River, Idaho

Abstract

Fish sampling and habitat assessment were conducted at 24 sites in the lower 54 km of the Coeur d'Alene River in 2005 and 2006 to 1) characterize four shoreline habitat types-- failing banks (FB), riprap (RR), riprap with vegetation (RRV), and vegetation (V)--according to ranked and quantifiable habitat variables, 2) assess differences in relative fish abundance (catch-per-unit effort; CPUE), species diversity, and community composition associated with the four shoreline habitat types, section (upstream and downstream), and season (summer, spring, fall), 3) assess the relationships between relative fish abundance and habitat variables, and 4) assess the relationship between relative fish abundance and a) depth of riprap structure and b) riprap rock diameter. The four habitat types differed significantly in habitat characteristics based on the Rapid Bioassessment Protocols (RBP) score (F=5.73, P<0.001). All four habitat types scored poor or marginal in substrate/available cover, pool variability, sediment deposition, and riparian zone width. Fish relative abundance as measured by CPUE at stabilized (riprap (RR) and riprap with vegetation (RRV)) shorelines was consistently higher than at unstabilized (failing bank (FB) and vegetation (V)) shorelines, among all seasons. Relative fish abundance was not significantly different between stabilized and unstabilized habitat types for gillnetting (F=1.95, P=0.167), but was significantly higher at stabilized than unstabilized habitats for electrofishing (F=5.66, P=0.020). Differences in species diversity were only evident between sections, not among habitat types or seasons. Fish community differences were apparent among habitat types as well as between sections and among seasons. Brown bullhead *Ameirus nebulosus*, northern pike *Esox lucius*, and pumpkinseed *Lepomis*

gibbosus were captured in significantly higher numbers at stabilized than unstabilized sites and longnose suckers *Catostomus catostomus* were captured in greater numbers at unstabilized than stabilized sites. Water temperature and spawn timing influenced fish abundance and composition in spring. Large diameter riprap supported a higher abundance of fish than did smaller diameter riprap. Overall, stabilized shorelines on the lower Coeur d'Alene River were not found to be adversely affecting overall fish relative abundance, diversity, and species composition under the existing conditions of a low percentage of the bank stabilized with riprap (2.5%). Instead, stabilized shorelines provided beneficial habitat in a river system with low quality and diversity of available habitats. This result indicates that modest amounts of bank additional stabilization would not be expected to harm the overall fish community. This result should not be predicted, however to apply as increasingly high percentages of the river are stabilized, and loss of river function occurs.

Introduction

Reinforcement and stabilization of the banks of rivers and streams has become widely used in large river systems as a means of reducing erosion. Excessive riverbank erosion can result from a wide array of factors, including extreme hydraulic conditions, highly erodible bed and bank material, high wave action caused by wind and boat traffic, vegetation removal, and recreational and other land uses (Simons 1995). Stabilization is often implemented in order to prevent or slow lateral channel erosion to protect land and improvements such as roads and bridges, by directing and defining the location of the channel (Simons 1995, Schmetterling et al. 2001).

Numerous stabilization approaches exist, including "soft" applications of natural materials such as willow plantings, large woody debris (LWD), trees, and rootwads (Shields et al. 1995, Shields et al. 2000, Schmetterling et al. 2001), and "hard" revetments such as stone spurs, dikes, concrete walls, and riprap (large angular rock) (Gore and Shields 1995, Schmetterling et al. 2001). Riprap is the most common type of bank stabilization structure. Riprap is relatively inexpensive and has proven effective in controlling site-specific erosion in numerous applications (Simons 1995, Schmetterling et al. 2001).

Although riverbank stabilization has been shown to improve water quality by reducing sedimentation, modifications of the natural habitat in a system also impact the hydrology and function of a river. Yorke (1978) identified ten physical-chemical factors that can be affected by bank stabilization, including suspended solids and dissolved substances. Bank stabilization also often results in channelization, an artificial straightening and confining of the channel. Water is forced to remain in the channel rather than be free to naturally scour banks, which often leads to a deepening of the channel (Yorke 1978). Stream modifications such as bank stabilization often create channels with uniform gradient lacking natural riffles and pools (Keller 1975), and subsequent loss of aquatic species diversity (Scarnecchia 1988).

Stabilizing riverbanks using riprap has been shown to have significant, often detrimental effects on fish habitat, especially in cases where stabilization covers extensive areas and most, or all, of the available habitat (e.g., the lower portion of the Missouri River; Hesse et al. 1989). Bank stabilization and channelization can eliminate or adversely impact habitat characteristics such as LWD, riffles and pools, undercut banks, and natural suspended sediment levels that are important to fish as well as other aquatic life (Leopold et al. 1964). LWD and undercut banks provide important habitat for both coldwater fish species such as salmon and trout (Bryant 1983) as well as warmwater fish species (Angermeier and Karr 1984). Habitat alterations can also favor invasion by non-native species. Moyle and Light (1996) reported that California streams with altered hydrologic regimes and reductions in habitat variability (as a result of dams) supported large numbers of exotic species and particular fish communities that would probably not have existed under more natural conditions.

Several studies have found that fish exhibit a significant preference for unstabilized versus stabilized areas of lakes and rivers, especially areas with riprap (Elser 1968, Knudsen and Dilley 1987, Garland et al. 2002). Conversely, bank stabilization has also been shown to benefit certain species or life stages in some instances (Chapman and Knudsen 1980, Trial et al. 2001, Zale and Rider 2003). On the Upper Yellowstone River, abundance of juvenile salmonids (rainbow trout, mountain whitefish, Yellowstone cutthroat trout *Oncorhynchus clarki bouvieri*, brown trout *Salmo trutta*, and brook trout *Salvelinus fontinalis*) was higher at riprap sections than at sections with natural banks as a result of poor habitat (lack of complexity and cover) at the natural sites (Zale and Rider 2003). The size of rock used in riprap structures is also important; large rocks have been shown to support higher numbers of juveniles than smaller rocks and cobbles (Lister et al. 1995).

In the Coeur d'Alene River basin, bank stabilization has been used by various private landowners and governmental entities in past decades to reduce erosion and thereby the amount of metals-contaminated soil entering the river. A riverbank stabilization inventory conducted in 2004 by the Kootenai-Shoshone Soil and Water Conservation District (KSSWCD) identified 24 bank stabilization projects on the lower river, which covered an estimated 2.8 % of the total riverbank (KSSWCD 2004). An estimated 90% (or 2.5% of the total riverbank) of stabilized shoreline consisted of riprap, while the remaining shorelines consisted of vegetation (6%), waddles (1%), and other stabilization types (3%). The projects vary greatly in design and scale. Remnants of historic (20-50 years ago) stabilizations to facilitate log transport are visible throughout the lower river (McClay 1940). These structures most often consisted of pilings with bulkheads that have disappeared by 2007. In addition, riprap was often applied along the railroad grade to prevent erosion. Many of these older riprapped areas now have shoreline vegetation, mainly red alder *Alnus rubra* and black cottonwood *Populus trichopcarpa*, whereas more recent riprap applications have little or no such vegetation. Stabilization projects implemented more recently have used both rock and biological approaches such as willow and cottonwood post plantings to stabilize failing banks.

Future plans call for more stabilization on the lower Coeur d'Alene River. Examining differences in the fish community between stabilized areas, both older riprap with vegetated shorelines (RRV) and newer riprap without vegetation (RR), and unstabilized areas, both unaltered relatively stable habitats with shoreline vegetation (V) as well as areas with failing banks (FB), will provide a means of selecting the most effective types of stabilization structures. In addition, it is important to understand seasonal changes as well as upstream and downstream differences in fish community usage of stabilized and unstabilized habitats. Understanding fish community differences between stabilized and unstabilized habitats will also aid in making recommendations for future stabilization projects.

The objectives of this study were to:

1. Characterize four shoreline habitat types-- failing banks (FB), riprap (RR), riprap with vegetation (RRV), and vegetation (V)--according to ranked and quantifiable habitat variables.

2. Assess differences in relative fish abundance (catch-per-unit effort; CPUE), species diversity, and community composition associated with the four shoreline habitat types, section (upstream and downstream), and season (summer, spring, fall).

3. Assess the relationships between relative fish abundance (CPUE) and habitat variables.

4. Assess the relationship between relative fish abundance (CPUE) and a) depth of riprap structure and b) riprap rock diameter.

Methods

The lower Coeur d'Alene River was divided into two sections: upstream from the Cataldo boat ramp downstream to the Highway 3 bridge, and downstream from the bridge to the Lake Coeur d'Alene inlet (Figure 1). The two river sections were not of equal lengths because the downstream section was delineated by its direct connection with several of the chain lakes. In both sections, four major shoreline habitat types were identified, two stabilized types (riprap (RR) and riprap with vegetation (RRV)) and two unstabilized types (vegetation (V) and failing banks (FB)). The V habitat types represented an unaltered, relatively stable habitat whereas the FB habitat type was a candidate for future stabilization. Stablized RR and RRV habitat types were established with respect to the amount of shoreline vegetation. RRV sites were generally older riprapped areas with established vegetation, whereas RR sites were newer areas with little or no vegetation. For each of the four habitat types, six sites were identified, three in the upstream section and three in the downstream section, for a total of 24 sites. All 24 sites consisted of 150 m of shoreline. In addition to these sites, 3 boat ramps with riprap were sampled. These boat ramps provide access to the river at Rose Lake, Anderson Lake, and Thompson Lake. All three boat ramp sites had riprap both upstream and downstream of the launches but were considered separately because they consisted of less than 150 m of shoreline. Sampling at all sites was conducted in three seasons, summer, spring, and fall. Summer sampling occurred during late July and early August of 2005, and spring and fall sampling were conducted during May and October of 2006. Fall sampling in the river occurred when Lake Coeur d'Alene water levels were lower, at winter pool (646.5 m), whereas summer and spring sampling occurred when lake water levels were higher, at summer pool (648.6 m). Both lake pool levels and river discharge influenced river elevation and therefore the degree to which bank structures were submerged.

Habitat Characteristics

A comprehensive habitat assessment, based on the EPA's Rapid Bioassessment Protocols (RBP), was conducted at each site (Barbour et al. 1999). RBP habitat characteristics included measures of substrate/available cover, pool substrate, pool variability, sediment deposition, channel flow, channel alteration, channel sinuosity, bank stability, vegetative protection, and riparian zone width. In addition to these RBP parameters, other habitat characteristics recorded were depths at 1.5 m and 3 m from the shoreline, river width, percent overhanging cover, dominant vegetation, percent submerged aquatic vegetation, LWD, aspect, bank slope, land use, and maximum depth. At RR and RRV sites additional habitat characteristics of riprap depth (m) and rock diameter (mm) were quantified. Prior to sampling each day, temperature, conductivity, and weather conditions were recorded. Flows in cubic meters per second (m^3/s) were retrieved from the USGS gauging station at Cataldo (Gauge number 12413500). Datasheets used to record habitat characteristics are attached in Appendix B.

Fish Sampling

Sampling was conducted at each of the 24 sites and 3 boat ramps using gillnetting and electrofishing. Gillnets are applicable in large rivers with little or no flow and effectively sample waters at depths greater than 3 m. Electrofishing has proven to be an effective sampling technique, applicable in various aquatic habitats, though usually limited to depths between 0.5 and 3.0 m (Reynolds 1983). Because of gear-specific selectivity associated with fish size, species, and sampling location, two gears were used to provide a more representative sample of the fish community than would have resulted from using either gear alone (Weaver et al. 1993). Goffaux et al. (2005) concluded that electrofishing alone was not sufficient for assessing fish assemblages in large river systems and that the addition of gillnetting will provide additional information on fish community structure.

Experimental (30m x 2m) monofilament gillnets consisted of four panels of varying mesh size (1.9 cm, 2.54 cm, 3.81 cm, 7.62 cm). The nets were set parallel to shore, forming a loose enclosure (Figure 3). Nets were set within one hour of sunset, left to sample overnight, and removed the following morning. Both the time set and time removed were recorded. Relative abundance (catch-per-unit-effort; CPUE) was calculated as fish caught per square meter per hour (fish/m^2/h) of sample time (Hubert 1983).

Electrofishing equipment consisted of a 6-m boat equipped with a Smith Root electrofishing unit. Pulse-DC current was used in order to minimize negative impacts to fish. Power output was maximized to effectively shock fish without causing harm and was adjusted based on water conductivity and temperature (Reynolds 1983) in the lower river. The 150-m length of shoreline was identified as adequate to assess species richness and percent abundance by ensuring that sufficient numbers of individuals were captured (Reynolds et al. 2003). CPUE was expressed as the number of fish caught per second of shock time (fish/s).

For both sampling gears all captured fish were identified to species, measured for total length and weight, and any abnormalities in body condition were noted. Fish community composition was estimated as a proportion of fish captured by habitat type, season, or section. Species diversity based on the Shannon Index (Peet 1975), was expressed as:

$$H' = -\sum_{i=1}^{S} p_i \ln p_i$$

where:

n_i is the number of individuals in each species or the abundance of each species,

p_i is the relative abundance of each species, calculated as the proportion n_i/N of individuals of a given species to the total number of individuals in the community,

S is the number of species, $\sum_{i=1}^{S} n_i$

and

N is the total number of all individuals.

Statistical Analysis

To test for differences in habitat characteristics among habitat types, I compared Rapid Bioassessment Protocols (RBP) scores and characteristics using analysis of variance (ANOVA). If significant overall differences in characteristics were found among habitat types, pair-wise comparisons were made using Tukey's test (Ott and Longnecker 2001, Higgins 2004).

To assess differences in relative fish abundance and species diversity, CPUE was evaluated by habitat type, season, section, and habitat type by section interaction, for each gear using ANOVA. Results are shown in ANOVA tables in Appendix C. To assess differences in fish community structure, the catches from both gears were combined. Catches were tested to detect differences by habitat type, season, and section using a non-parametric Kruskal-Wallis test (Higgins 2004). All pair-wise comparisons were made using Tukey's test (Ott and Longnecker 2001).

To assess the relationships between relative fish abundance (y) and habitat characteristics (x) among habitat types, a stepwise y on x linear regression was used. Akaike's Information Criterion was conducted to explain variance. Habitat variables section, percent aquatic vegetation, percent overhanging vegetation, 1.5-m and 3-m depths from shore, maximum mean depth, width, and bank slope were used in the analysis. To meet assumptions of normality and homogeneity of variance, all

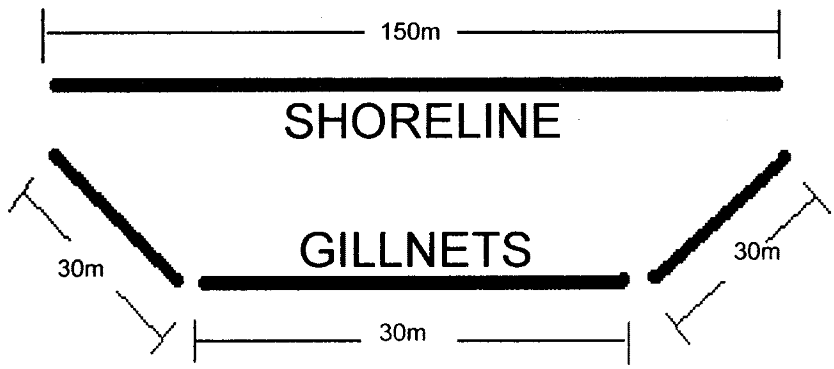


Figure 3. Schematic showing placement of experimental monofilament gillnets with respect to the shorelines for sites on the lower Coeur d'Alene River, Idaho.

CPUE data were square-root transformed. If this transformation did not normalize data, ranked ANOVA was utilized (Higgins 2004).

To assess the relationship between relative fish abundance (y) and riprap site characteristics (x; depth of riprap structure and riprap diameter) a y on x linear regression was used. All statistical testing was conducted using SAS (SAS Institute 2000). In all statistical tests, an alpha value of 0.10 was required for significance rather than the more typical 0.05 because of the high degree of variability in large river studies.

Results

Fish Catch and River Conditions

A total of 3,511 fish was captured, representing 17 species and 7 families (Appendix A). Gillnetting effort consisted of 1,270 hours of net time and resulted in the capture of 596 fish. Electrofishing effort consisted of 34 hours and resulted in the capture of 2,915 fish. In summer, 1,402 fish were captured, 300 by gillnetting and 1,107 by electrofishing. In spring, 703 fish were captured, 83 by gillnetting and 620 by electrofishing. In fall 1,407 fish were captured, 213 by gillnetting and 1,194 by electrofishing.

The seasons differed in average water temperature, discharge (Figure 4), the amount of aquatic vegetation present, and the extent to which riprap structures were submerged as a result of river discharge and lake elevation. In summer, average water temperature was highest (21.5°C), discharge averaged 134 m^3/s, aquatic vegetation was abundant throughout the study area, and riprap structures were submerged to the greatest extent. In spring, average water temperature was nearly 8°C lower than summer (13.9°C), average discharge was highest (1,120 m^3/s), aquatic vegetation was largely absent from shorelines, and stabilized sites were not fully submerged. In fall, average

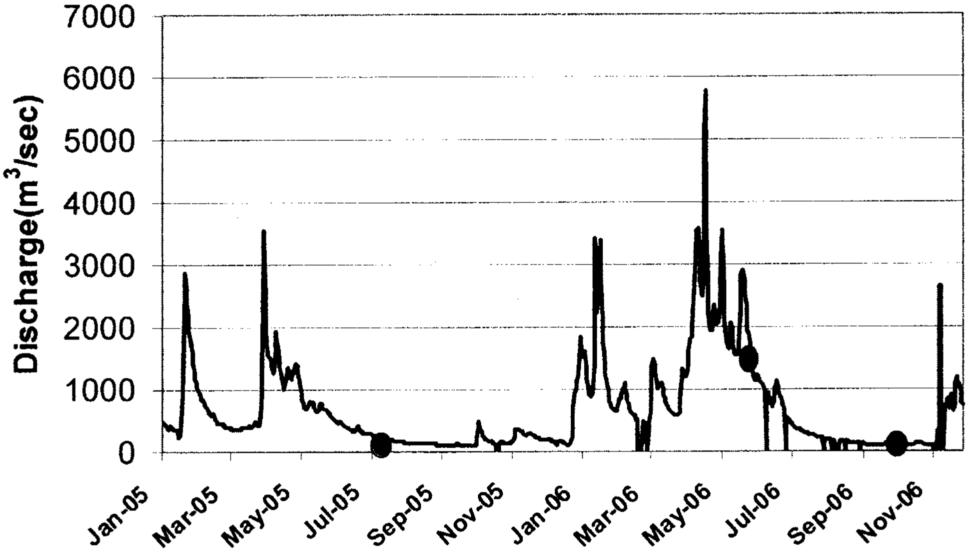


Figure 4. Average discharge in m^3/sec as recorded at the U.S. Geologic Survey Cataldo stream gauge (12413500) on the Coeur d'Alene River, Idaho.

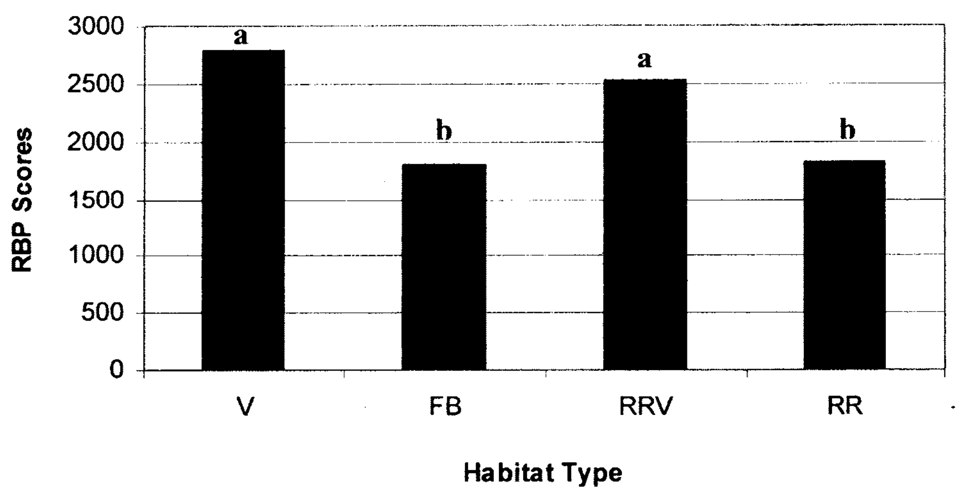


Figure 5. Scores compiled based on the EPA Rapid Bioassessment Protocols (RBP) at four habitats types on the lower Coeur d'Alene River. V=vegetated FB=failing bank RRV=riprap with vegetation and RR=riprap. Columns with the same letter were not significantly different (α=0.10).

temperature was similar to spring (13.4°C), average discharge was similar to summer (107 m^3/s), aquatic vegetation was largely absent from shorelines, and stabilized sites were not fully submerged.

Habitat Characteristics

The four habitat types, failing bank (FB), riprap (RR), riprap with vegetation (RRV), and vegetation (V), differed significantly in habitat characteristics based on the Rapid Bioassessment Protocols (RBP) scores ($F=5.73$, $P<0.001$). V sites had the highest score, followed by RRV, RR, and FB sites (Figure 5). In pair-wise comparisons of RBP scores, V and RRV sites did not differ significantly from each other but were significantly higher than RR and FB sites. The RR and FB sites did not differ significantly from each other. Overall, all four habitat types scored poor or marginal in substrate/available cover, pool variability, sediment deposition, and riparian zone width. Individual RBP variables that differed significantly among habitat types were substrate/available cover ($\chi2=13.935$, $P=0.003$), pool substrate ($\chi2=8.009$, $P=0.046$), channel alteration ($\chi2=8.049$, $P=0.045$), bank stability ($\chi2=16.032$, $P=0.001$), and vegetative protection ($\chi2=19.159$, $P<0.001$). For substrate/available cover, the V sites had the highest scores, followed by RRV, RR, and FB sites. Scores were significantly higher at V, RRV, and RR than for FB sites (Figure 6). Pool substrate quality was higher at RRV, RR, and V sites than at FB sites. For channel alteration, FB sites showed the least, followed by V, RRV, and RR. FB site scores were significantly higher than RR site scores. As with substrate/available cover, bank stability scores were significantly higher for V, RRV, and RR than for FB sites (Figure 6). Vegetative protection showed a similar pattern, though FB sites had marginally higher scores than RR sites. Vegetative protection was significantly higher at V sites than FB and RR sites, and RRV sites had higher scores than FB and RR sites (Figure 6).

The four habitat types were also significantly different with respect to the percentage of aquatic vegetation, overhanging vegetation, 1.5-m depth, 3-m depth, maximum depth, and bank slope

(Table 1). Both the percentages of aquatic and overhanging vegetation were highest at V sites, followed by RRV, RR, and FB sites. The 1.5-m and 3-m depth distances were highest at V sites, followed by FB, RRV, and RR sites. Maximum depth was greatest depth at FB sites, followed by RR and RRV, and V sites. Bank slope was highest at RR sites, followed by RRV, FB, and V sites. River width was not significantly different among habitat types.

Stabilized RR and RRV sites exhibited some significantly different habitat characteristics. RR sites had significantly less overhanging vegetation (4%) than RRV sites (68%; $\chi2$=8.486, *P*=0.004), significantly higher bank slopes ($\chi2$=3.103, *P*=0.078), and significantly greater average riprap depth at RR (1.7 m) than RRV (1.4 m) sites ($\chi2$=3.871, *P*=0.049; Table 1).

The two stabilized habitat types also had many similarities. The 1.5-m and 3-m depths, river width, percent aquatic vegetation, and maximum depth, were not significantly different between RR sites and RRV sites (Table 1). Average riprap diameter did not differ significantly between RR sites (852 mm) and RRV sites (660 mm; $\chi2$=1.339, *P*=0.247).

Habitat at the three boat ramp sites (Rose, Anderson, and Thompson Lakes) was similar to that of other stabilized (RR and RRV) sites. The riprap at the Thompson Lake boat ramp was entirely (100%) covered in vegetation, much like most RRV sites. In contrast, neither the Rose Lake nor Anderson Lake boat ramps had any vegetation (0%) much like most RR sites. The Anderson and Thompson Lake boat ramps consisted of relatively shallow riprap structures (0.5 m and 1.2 m) whereas the Rose Lake ramp was stabilized much deeper (2.5 m). In addition, the mean diameter of rock used at the Rose Lake ramp was larger (810.5 mm), than at either Anderson (553.8 mm) or Thompson (614.6 mm) lakes.

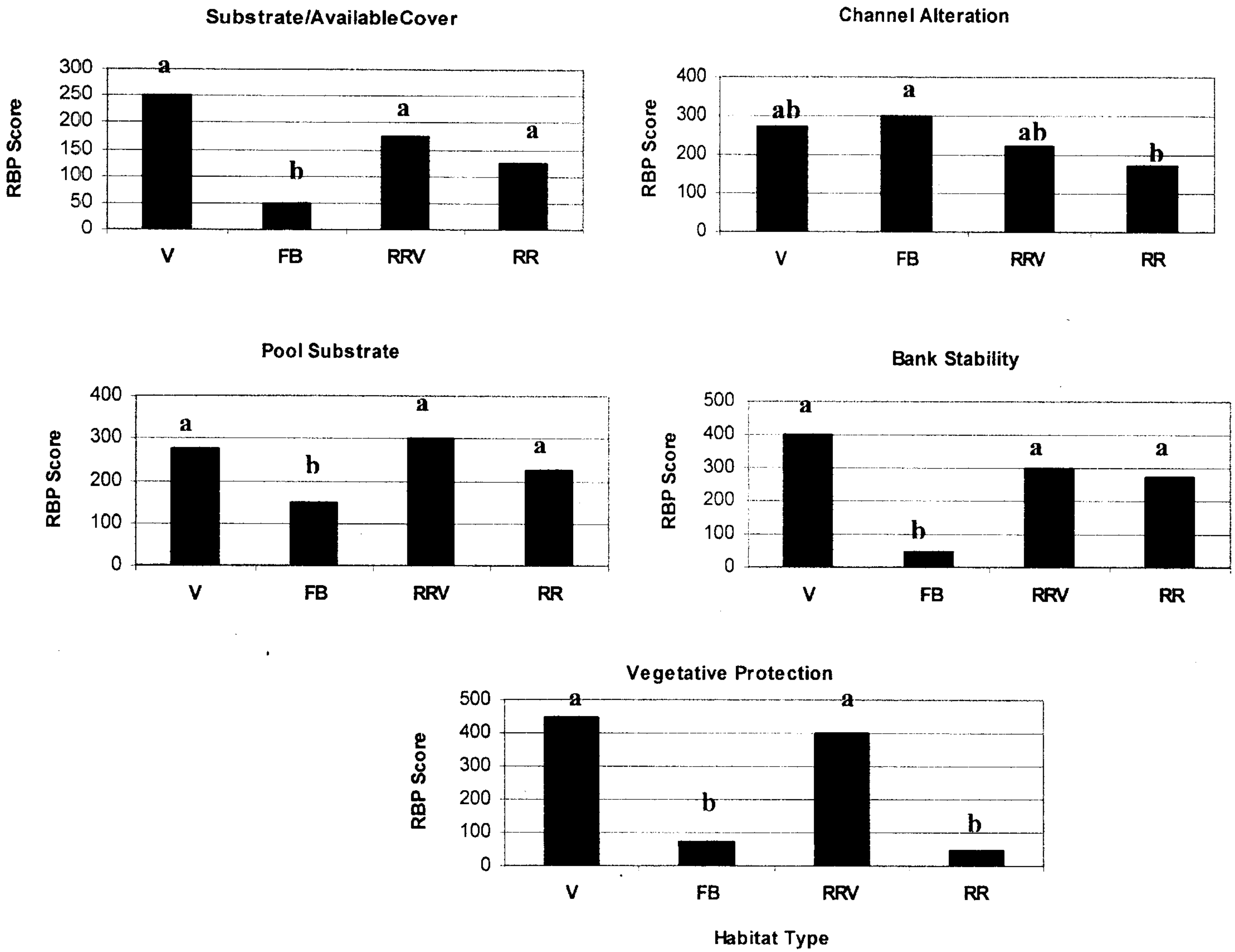


Figure 6. Pairwise comparisons of habitat variables significantly different among habitat types. Columns with different letters were significantly different (α=0.10).

Table 1. Kruskal-Wallis results for comparison of habitat variables by habitat types (FB, RR, RRV, V) and by stabilized habitat types (RR and RRV). Statistical results shown are chi-squared and *P*-values, significant at α=0.10.

	Variable	**X^2**	***P*-value**
All Habitat Types	Aquatic vegetation	8.229	*0.042*
	Overhanging vegetation	19.545	*<0.001*
	1.5-m depth	9.323	*0.025*
	3-m depth	7.923	*0.048*
	Max depth	10.813	*0.013*
	Width	3.378	0.337
	Slope	11.058	*0.011*
Stabilized Habitat Types	Aquatic vegetation	0.233	0.629
	Overhanging vegetation	8.486	*0.004*
	1.5-m depth	2.077	0.150
	3-m depth	2.573	0.109
	Max depth	2.077	0.150
	Width	0.103	0.749
	Slope	3.103	*0.078*
	Riprap diameter	1.339	0.247
	Riprap depth	3.871	*0.049*

Relative Fish Abundance, Species Diversity, and Community Composition by Habitat Type, Season and Section

Relative Fish Abundance – Relative fish abundance as measured by CPUE was not significantly different between unstabilized and stabilized habitat types for gillnetting (F=1.45, P=0.233), but was significantly higher at stabilized than unstabilized habitats for electrofishing (F=5.76, P=0.020; Appendix C). Both gillnetting and electrofishing CPUE were significantly different by habitat type, season, and section, but not by the habitat by season interaction (Figure 7, Figure 8). For gillnetting, CPUE was significantly higher at FB (0.0044 fish/m^2/hr) sites than at RR (0.0023 fish/m^2/hr), V (0.0021 fish/m^2/hr) and RRV (0.0026 fish/m^2/hr) sites (F=2.88, P=0.044). CPUE was significantly higher in summer than in both spring and fall and was significantly higher in fall than spring (F=23.97, P<0.001). CPUE was significantly higher upstream than downstream (F=4.62, P=0.036). For electrofishing, CPUE was significantly higher at RR (0.0270 fish/sec) and RRV (0.0223 fish/sec) sites than at FB (0.0128 fish/sec) sites, but was not significantly higher than V (0.0209 fish/sec) sites (F=4.62, P=0.006). CPUE was significantly higher in fall and summer than during spring (F=11.27, P<0.001) and higher downstream than upstream (F=27.14, P<0.001).

Summer CPUE among habitat types did not differ significantly for gillnetting (F=1.92, P=0.160) or electrofishing (F=1.42, P=0.268), but was significantly different between sections for both gears (Figure 9). For gillnetting, CPUE was higher upstream than downstream (F=4.64, P=0.044). In contrast, electrofishing CPUE was significantly higher downstream than upstream (F=11.66, P=0.003).

Spring CPUE was not significantly different among habitat types for gillnetting (F=0.060, P=0.980) or electrofishing (F=2.29, P=0.111) (Figure 10). For gillnetting, FB sites had high variability in catches. The highest CPUE for all sites in spring was recorded at FB2, and FB site

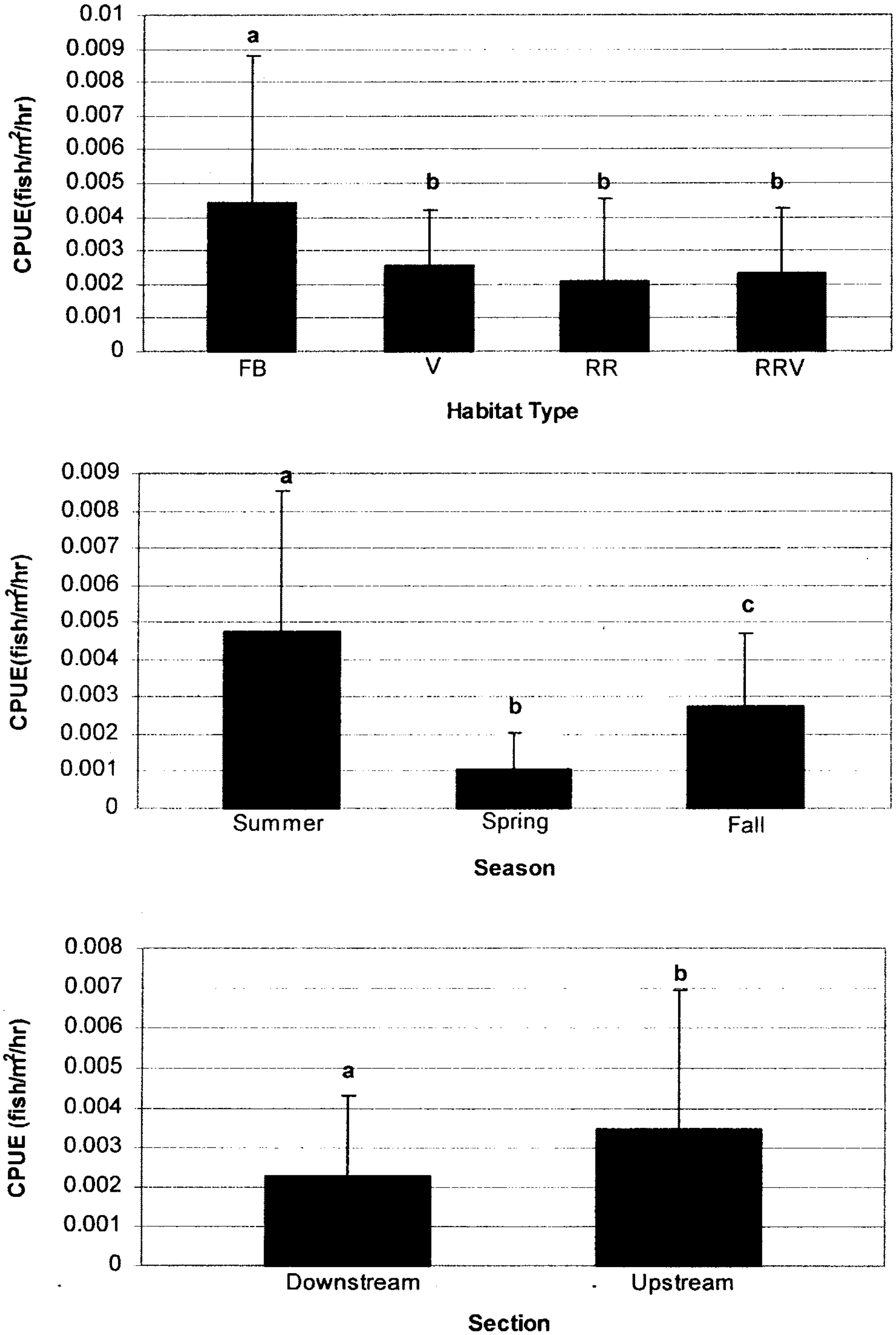
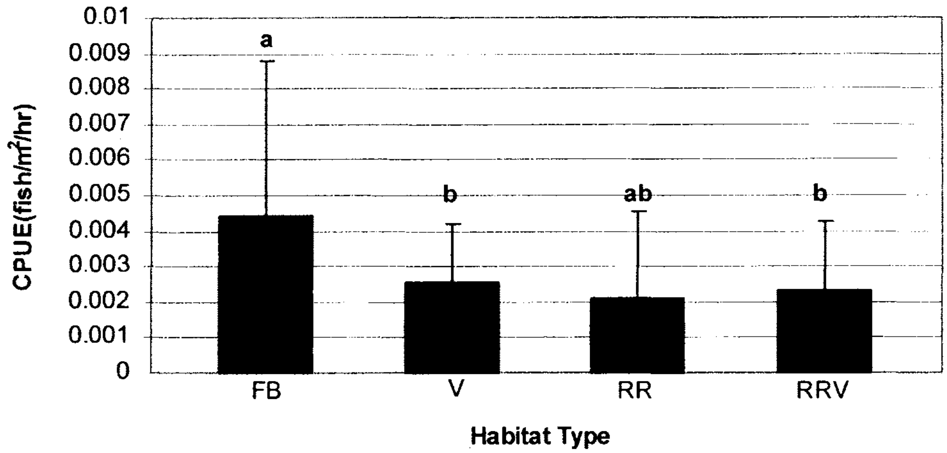
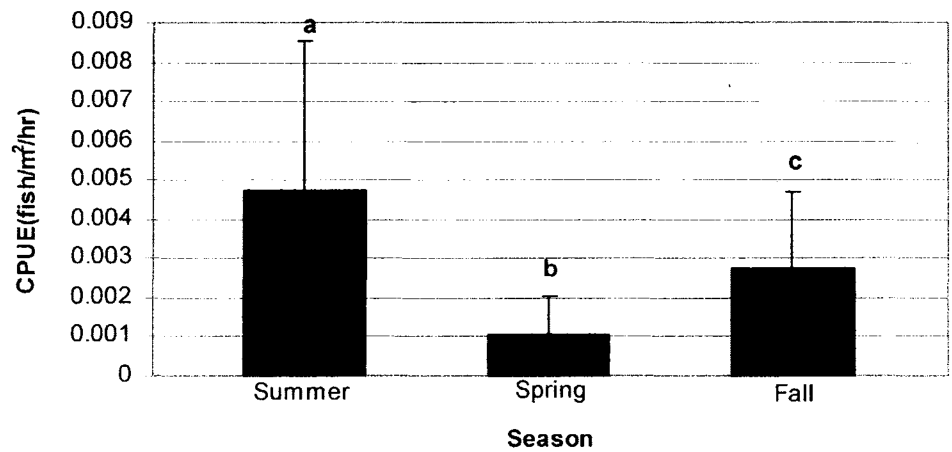


Figure 7. Overall gillnetting relative fish abundance (CPUE) by habitat type (FB, RR, RRV, V), season (summer, spring, fall), and section (downstream, upstream). Columns with the same letter are not significantly different (α=0.10). Error bars show standard deviation.





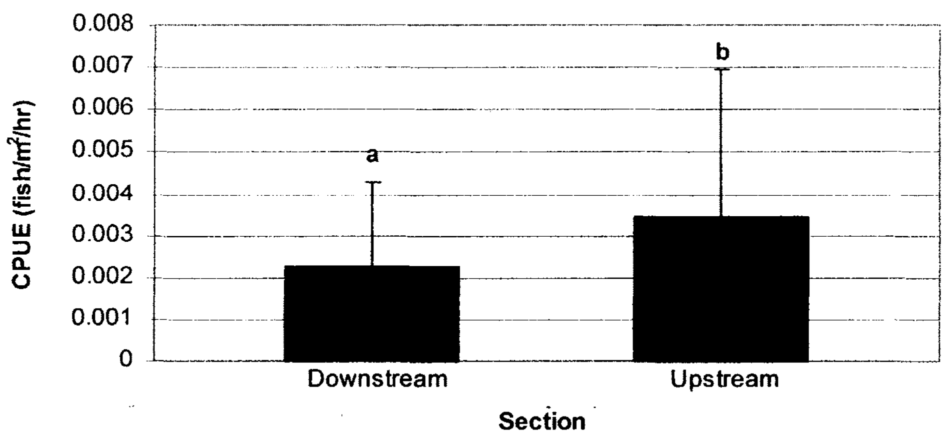


Figure 8. Overall electrofishing relative fish abundance (CPUE) by habitat type (FB, RR, RRV, V), season (summer, spring, fall), and section (downstream, upstream). Columns with the same letter are not significantly different (α=0.10). Error bars show standard deviation.

catches also included two zeros. CPUE was not significantly different between sections for gillnetting (F=0.140, *P*=0.714) or electrofishing (F=1.95, *P*=0.179; Figure 10).

Fall CPUE was significantly different among habitat types for gillnetting and between sections for electrofishing (Figure 11). For gillnetting, CPUE was significantly higher at FB sites than RR sites (F=3.29, *P*=0.043). CPUE was not significantly different between the upstream and downstream sections (F=1.78, *P*=0.198). For electrofishing, CPUE was not significantly different among habitat types (F=1.28, *P*=0.309), but was significantly higher upstream than downstream (F=15.11, *P*<0.001).

Relative abundance at the 3 boat ramp sites was higher than all shoreline habitat types for gillnetting and electrofishing (Figure 12). Rose Lake CPUE was higher than the other habitat types, overall, as well as for individual seasons. In spring and fall, gillnetting CPUE at Rose Lake was nearly twice that of the CPUE of the next highest habitat type.

<u>Species Diversity</u> – Differences in species diversity as indicated by the Shannon Index, were only significant between sections, not among habitat types or seasons. Overall diversity was H'=2.13 with H_{max}= 2.94, which would indicate maximum evenness among species. Species diversity for RR (H'=2.06), RRV (H'=2.05), FB (H'=2.03), and V (H'=2.02) sites were not significantly different (F=0.49, *P*=0.691). Diversity was higher at the boat ramps (average H'=2.13), overall, than any other shoreline habitat type and was significantly higher than RR, V, and FB sites (F=2.32, *P*=0.065). Diversity was similar for spring (H'=2.24) and fall (H'=2.01) and lower in summer (H'=1.80), but the values were not significantly different from each other (F=1.60, *P*=0.2143). Diversity upstream (H'=2.21) was significantly higher than downstream (H'=1.84) (F=4.58, *P*=0.036).

<u>Community Composition</u> – Fish community differences were apparent among habitat types as well as between sections and among seasons. The overall fish community was composed largely of

Percids and Centrarchids (71%; Figure 13). Yellow perch was the most common fish caught (34%), followed by pumpkinseed (14%), largemouth bass (11%), brown bullhead (10%), largescale sucker (7%), bluegill (7%), smallmouth bass (6%), and longnose sucker (5%). These eight species accounted for 94% of the total catch from all seasons and both gears. Salmonids captured in this study constituted 2% of the total catch and piscivores (largemouth bass, smallmouth bass, northern pikeminnow, and northern pike) constituted 19% of the total catch. Overall length of fish captured during this study averaged 133 mm (range, 26-1337 mm).

Catches of four species were significantly different between stabilized and unstabilized habitat types. Brown bullhead ($\chi 2$=6.150, P=0.013), northern pike ($\chi 2$=4.075, P=0.044), and pumpkinseed ($\chi 2$= 10.745, P=0.001) were captured in significantly higher numbers at stabilized than unstabilized sites and longnose suckers ($\chi 2$= 3.444, P=0.064) were captured in greater numbers at unstabilized than stabilized sites. Species including largescale sucker, northern pikeminnow, bullhead, and pumpkinseed were captured in significantly different numbers among habitat types (Figure 14, Table 2). Largescale suckers were captured in significantly greater numbers at FB sites than RR sites, but did not have higher numbers than RRV and V sites ($\chi 2$=6.683, P=0.083). Northern pikeminnow were also captured in significantly greater numbers at FB sites ($\chi 2$=8.337, P=0.040). FB site catch was significantly higher than RRV site catch, but was not significantly higher than RR and V sites. Bullhead and pumpkinseed were more common at stabilized RR and RRV sites than FB sites ($\chi 2$=9.676, P=0.022 and $\chi 2$=10.979, P=0.012). For both bullhead and pumpkinseed, catches were not significantly different between RR, RRV, and V sites; catches at V sites were not significantly different from catches at FB sites. Other species catches did not differ significantly among habitat types.

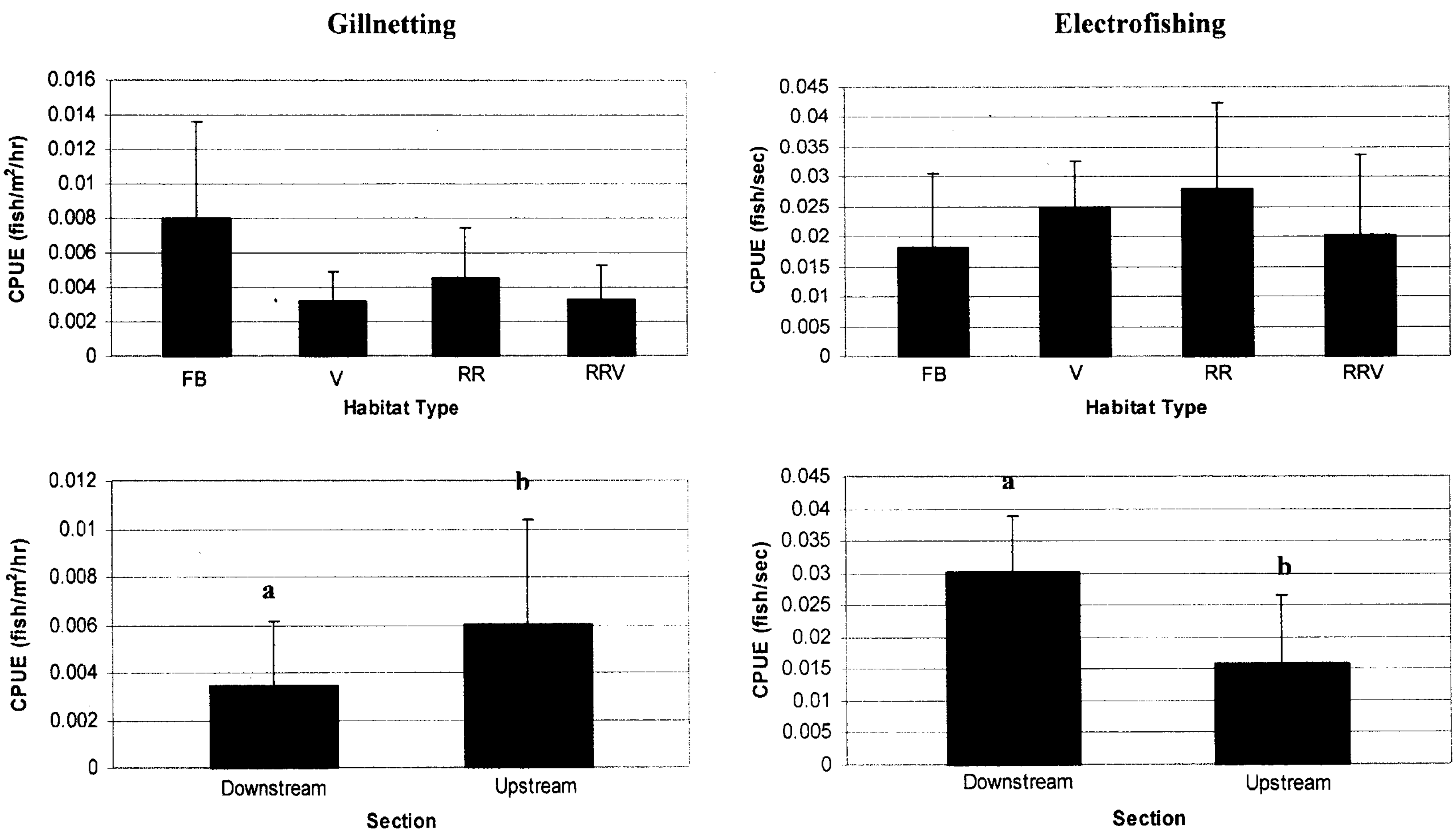


Figure 9. Summer fish relative abundance (CPUE) by habitat type (FB, RR, RRV, V) and section (1=downstream, 2=upstream) for both gears (gillnetting and electrofishing). No significant differences among habitat types were evident for either gear. Section 1 and 2 differed significantly for both gears. Error bars show standard deviation.

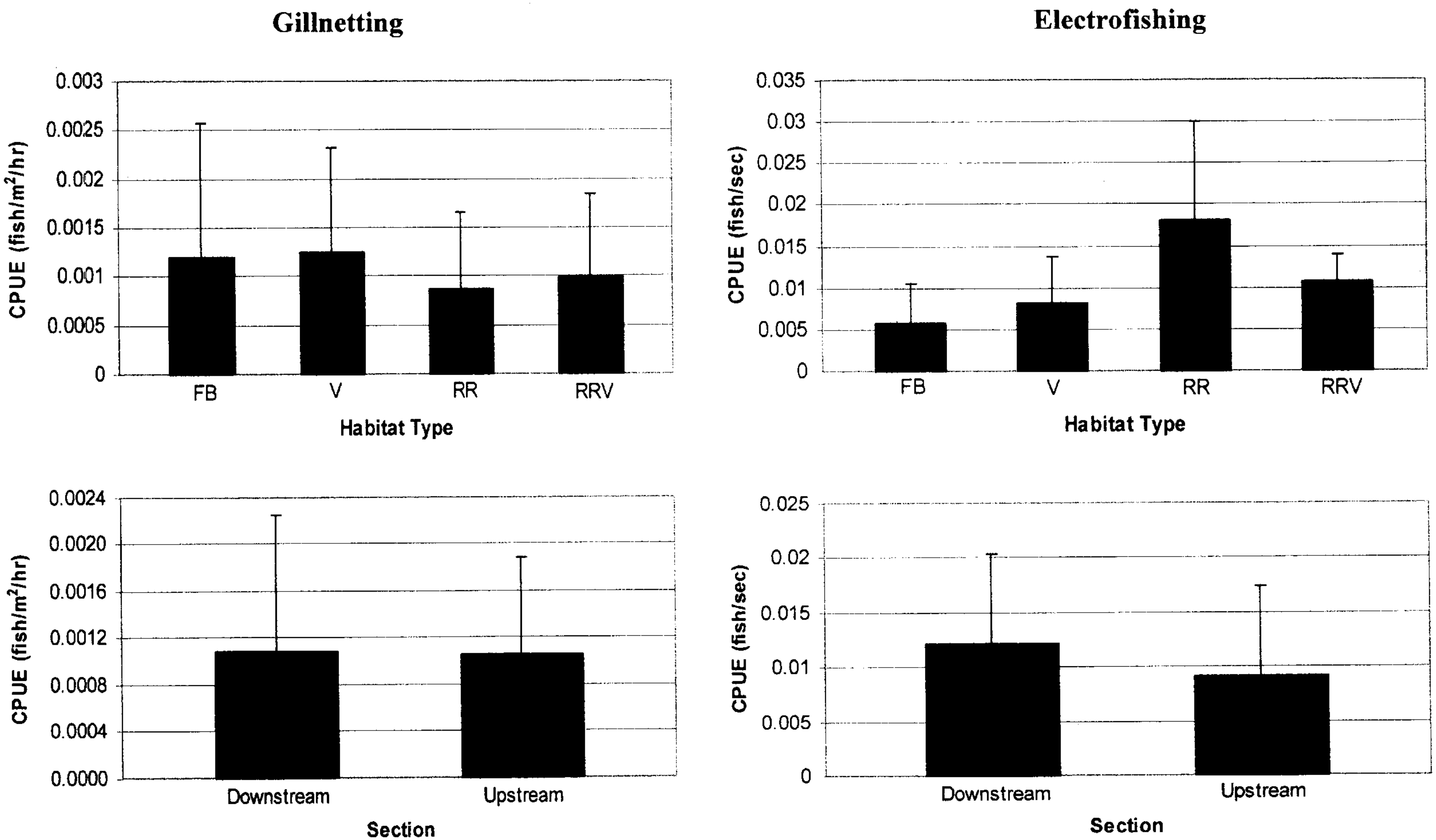


Figure 10. Spring sampling fish relative abundance (CPUE) by habitat type (FB, RR, RRV, V) and section (downstream, upstream) for both gears (gillnetting and electrofishing). No significant differences among habitat types or sections were evident for either gear. Error bars show standard deviation.

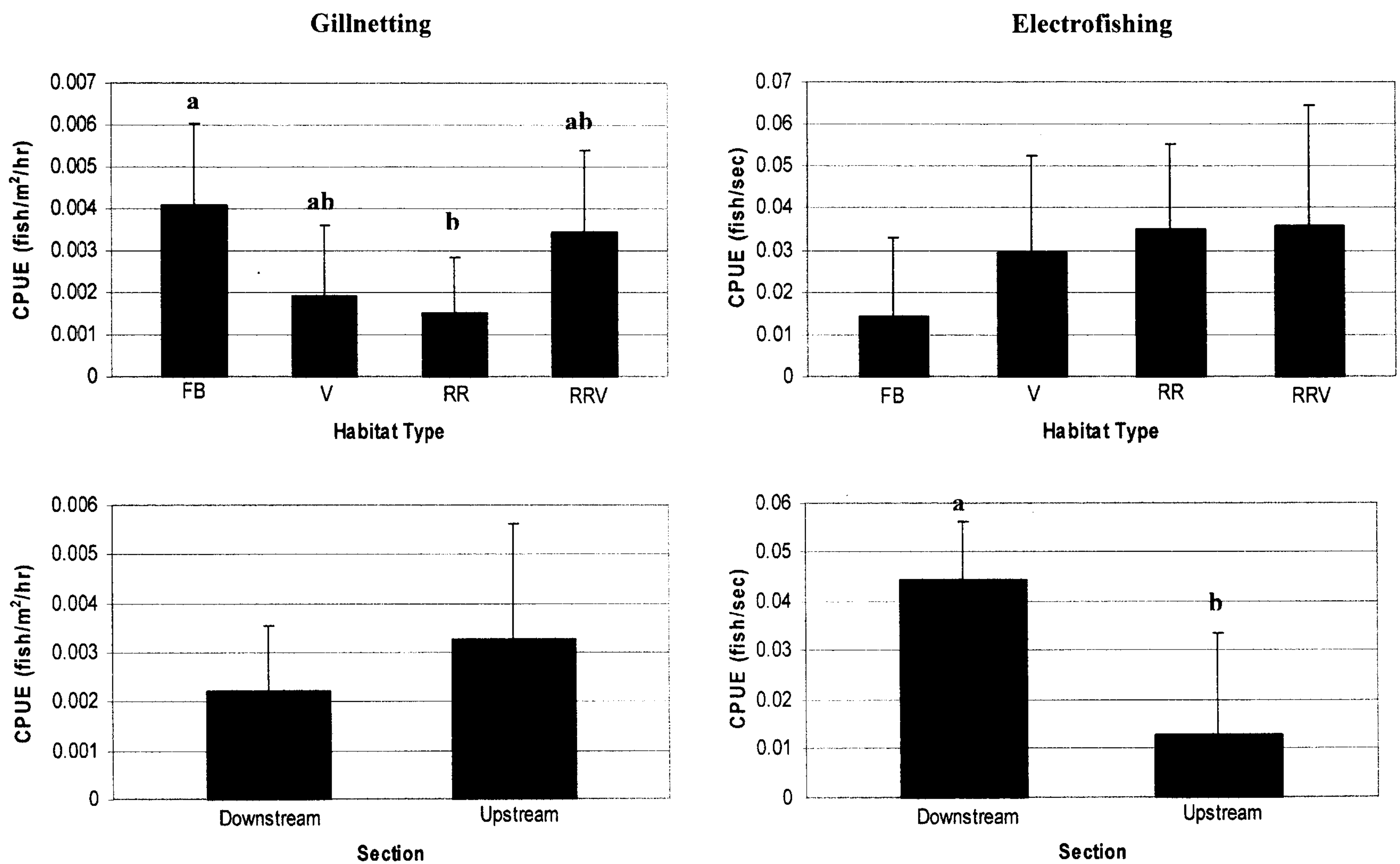
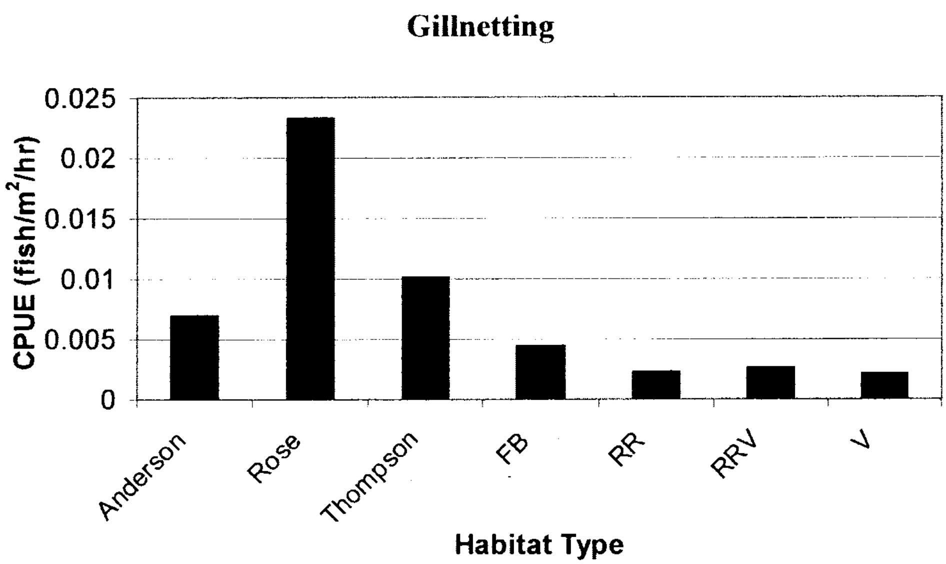


Figure 11. Fall sampling fish relative abundance (CPUE) by habitat type (FB, RR, RRV, V) and section (downstream, upstream) for both gears (gillnetting and electrofishing). Habitat types were significantly different for gillnetting and sections were significantly different for electrofishing. Columns with the same letter are not significantly different. Error bars show standard deviation.



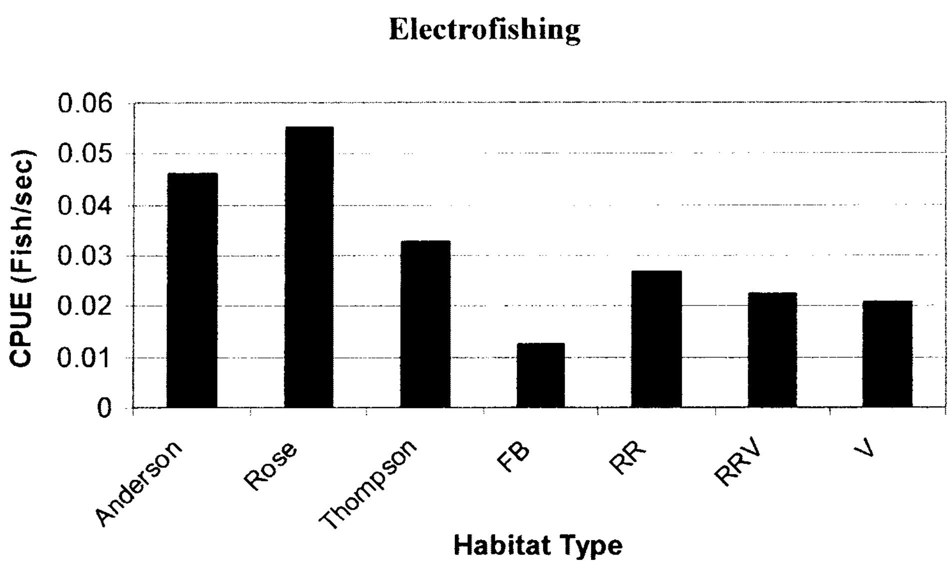


Figure 12. Relative fish abundance (CPUE) for each of the three boat ramps (Anderson, Rose, and Thompson) as compared to overall CPUE for stabilized (RR and RRV) and unstabilized (FB and V) habitat type.

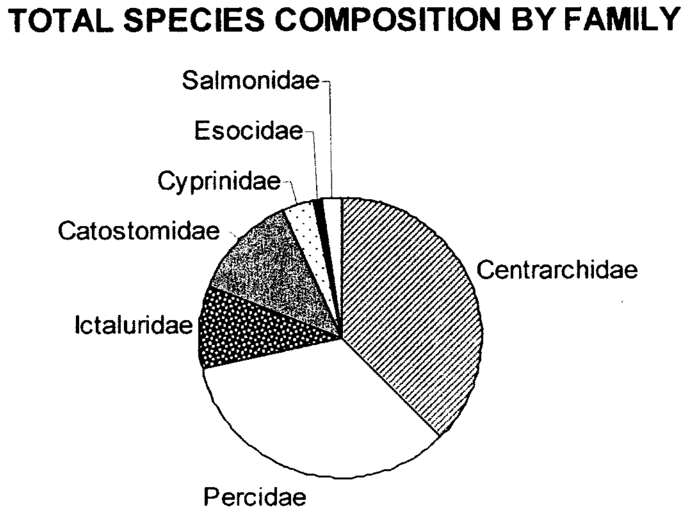


Figure 13. Total species composition by family as a percentage of total catch for both gears (gillnetting and electrofishing) on the lower Coeur d'Alene River, Idaho.

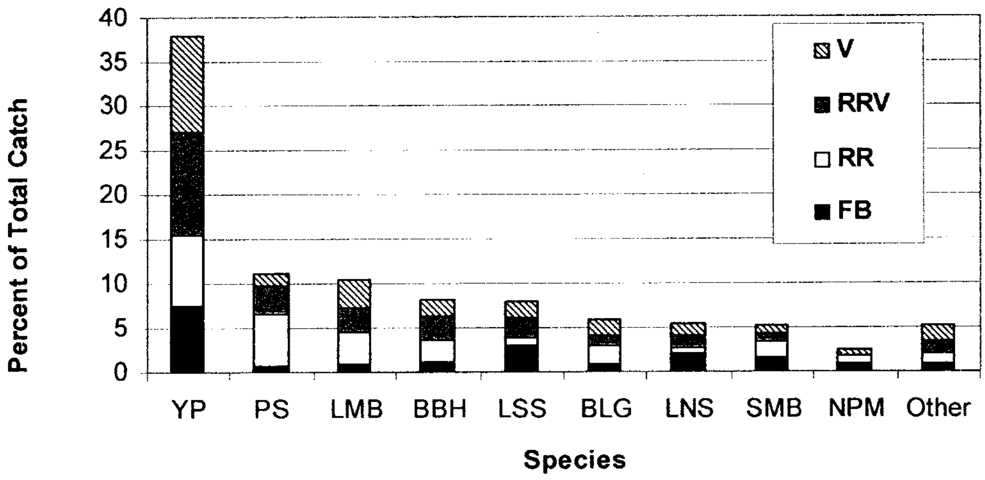


Figure 14. Percent of total catch for individual species by habitat type (FB, RR, RRV, V) on the lower Coeur d'Alene River, Idaho. YP=yellow perch, PS=pumpkinseed, LMB=largemouth bass, BBH=brown bullhead, LSS=largescale sucker, BLG=bluegill, LNS=longnose sucker, SMB=smallmouth bass, NPM=northern pikeminnow.

Table 2. Total number of individuals captured by species and habitat type (FB=failing banks, RR=riprap, RRV=riprap with vegetation, V=vegetation, BOAT=boatramp) in the lower Coeur d'Alene River, Idaho.

	Habitat Type				
Species	**FB**	**RR**	**RRV**	**V**	**BOAT**
Catostomus catostomus	60	16	39	41	25
Catostomus macrocheilus	81	26	69	52	23
Esox lucius	2	6	9	2	3
Ictalurus nebulosus	30	73	76	52	115
Lepomis gibbosus	21	165	90	42	166
Lepomis macrochirus	25	59	32	53	60
Micropterus dolomieui	47	51	24	25	47
Micropterus salmoides	26	101	81	91	60
Onchorhynchus nerka	5	9	6	5	2
Onchorhynchus tshawytscha	1	0	0	19	0
Perca flavescens	212	226	332	309	106
Pomoxis nigromaculatus	1	6	6	8	15
Prosopium williamsoni	1	2	2	3	0
Ptychocheilus oregonensis	27	22	3	17	14
Oncorhynchus clarki lewisi	6	5	10	2	2
Salmo gairdneri	0	0	1	0	1
Salvelinus fontinalis	0	0	0	1	0
Tinca tinca	10	6	7	10	9
Total	555	773	787	732	648

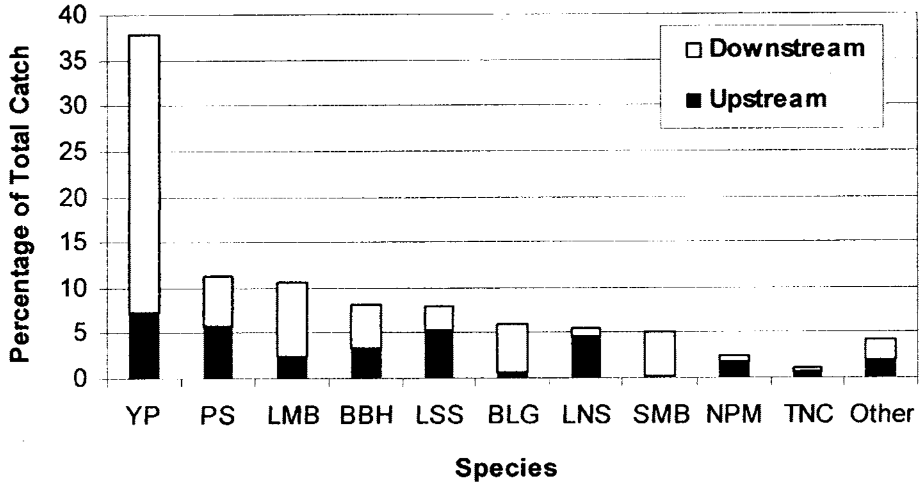


Figure 15. Percentage of total catch by species and section (1=downstream, 2=upstream) on the lower Coeur d'Alene River, Idaho. YP=yellow perch, PS=pumpkinseed, LMB=largemouth bass, BBH=brown bullhead, LSS=largescale sucker, BLG=bluegill, LNS=longnose sucker, SMB=smallmouth bass, NPM=northern pikeminnow, TNC=tench.

Table 3. Number of individuals captured by season (summer, spring, and fall) and by section (1=downstream, 2=upstream) in the lower Coeur d'Alene River, Idaho.

	Season			Section		
Species	**Fall**	**Summer**	**Spring**	**1**	**2**	**Total**
Catostomus catostomus	39	73	44	26	130	156
Catostomus macrocheilus	87	107	34	76	152	228
Esox lucius	14	5	0	10	9	19
Ictalurus nebulosus	63	102	66	137	94	231
Lepomis gibbosus	120	91	107	153	165	318
Lepomis macrochirus	146	9	14	149	20	169
Micropterus dolomieui	57	61	29	139	8	147
Micropterus salmoides	202	89	8	230	69	299
Onchorhynchus nerka	1	0	24	9	16	25
Onchorhynchus tshawytscha	0	1	19	20	0	20
Perca flavescens	374	593	112	874	205	1,079
Pomoxis nigromaculatus	9	10	2	10	11	21
Prosopium williamsoni	0	1	7	3	5	8
Ptychocheilus oregonensis	18	46	5	20	49	69
Salmo clarki lewisi	3	3	17	6	17	23
Salmo gairdneri	1	0	0	0	1	1
Salvelinus fontinalis	0	0	1	1	0	1
Tinca tinca	7	23	3	16	17	33
Total	1,141	1,214	492	1,879	968	2,847

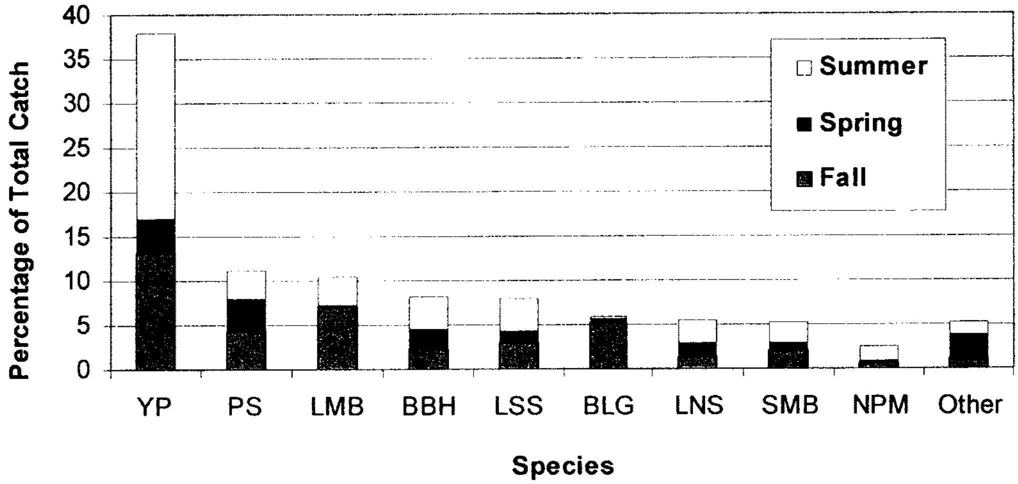


Figure 16. Individual species as a percentage of total catch given by season (summer, spring, fall) on the lower Coeur d'Alene River, Idaho. YP=yellow perch, PS=pumpkinseed, LMB=largemouth bass, BBH=brown bullhead, LSS=largescale sucker, BLG=bluegill, LNS=longnose sucker, SMB=smallmouth bass, NPM=northern pikeminnow.

Many species showed significant differences in catches between river sections (Figure 15, Table 3). Perch ($\chi2$=10.552, *P*<0.001), largemouth bass ($\chi2$=5.77, *P*=0.016), bullhead ($\chi2$=5.968, *P*=0.015), and smallmouth bass ($\chi2$=22.653, *P*<0.001) catches were significantly higher downstream than upstream. Largescale sucker ($\chi2$= 3.352, *P*=0.067), longnose sucker ($\chi2$=26.602, *P*<0.001), northern pikeminnow ($\chi2$=4.746, *P*=0.029), and westslope cutthroat trout ($\chi2$=7.553, *P*=0.006) catches were significantly higher upstream than downstream. Pumpkinseed, tench, and northern pike catches were nearly identical between the two sections and the remaining species catches were low and similar in numbers between sections (Figure 16, Table 3).

Catches for several individual species showed significant differences among seasons (Figure 16, Table 3). Catches were highest in summer for yellow perch ($\chi2$=10.614, *P*=0.005), brown bullhead ($\chi2$=7.35, *P*=0.025), tench ($\chi2$=14.180, *P*<0.001), smallmouth bass ($\chi2$=4.706, *P*=0.095), and northern pikeminnow ($\chi2$=16.225, *P*<0.001). Bullhead, tench, and northern pikeminnow were captured in greater numbers in summer than in either spring or fall. Yellow perch and smallmouth bass catches were higher in summer than spring but did not differ significantly from fall. Westslope cutthroat trout were captured in significantly higher numbers in spring than during either summer or fall ($\chi2$=8.990, *P*=0.011). In fall, significantly greater numbers of largemouth bass ($\chi2$=17.368, *P*<0.001), bluegill ($\chi2$=11.333, *P*=0.004), and northern pike ($\chi2$=8.378, *P*=0.015) were captured. Bluegill numbers in fall were significantly greater than during either summer or spring. Largemouth bass and northern pike numbers did not differ significantly between fall and summer, but were higher than in spring.

The fish community composition at boat ramp sites was similar to that for other stabilized (RR and RRV) sites (Table 2). Bullhead and pumpkinseed, which showed an affinity for riprap structures, were commonly captured at boat ramp sites as well. At the Rose Lake site, pumpkinseed was the most abundant species captured followed by bullhead, perch, and suckers, which together constituted 83% of the total catch at this site. Suckers were captured in relatively high numbers at the Rose Lake site, constituting just over 10% of the catch. At the Anderson Lake site, yellow perch was the most abundant species, followed by smallmouth bass, pumpkinseed, and bluegill which together constituted 75% of the catch at that site. At the Thompson Lake site largemouth bass were most abundant, followed by yellow perch, bluegill, and pumpkinseed. These species constituted 81% of the total catch at that site. Only two cutthroat trout were captured at these sites, one at Anderson Lake and one at Rose Lake.

Relative Fish Abundance and Habitat Variables

Significant differences in relative fish abundance (CPUE) were explained by habitat variables for both gears in summer and fall. Overall, differences in relative abundance (CPUE) among sites were best explained by the habitat variable river section (upstream vs. downstream) (Table 4). However, differences in CPUE for spring gillnetting and summer electrofishing were not significantly explained by section, and section was absent from the best model for this gear/season combination.

CPUE differences in summer were significantly explained by habitat variables. CPUE differences for gillnetting were best explained by a six variable model with variables section, slope, width, percent overhanging vegetation, 1.5-m depth, and 3-m depth (F=3.16, P=0.036, R^2=0.340). The variables section, width, overhanging vegetation, and 1.5-m depth were individually significant (Table 4). For electrofishing, a model consisting of the variables width, overhanging vegetation, aquatic vegetation, slope, and maximum depth best explained differences in CPUE (F=5.82, P=0.003, R^2=0.534). All variables in this model were significant. Aquatic vegetation was present at nearly all sites and through observation during electrofishing, most fishes were sampled from this vegetation in summer when such cover was abundant.

CPUE differences in spring were not significantly explained by habitat variables. For gillnetting, the best model, including variables width and 3-m depth, was not able to significantly explain differences in CPUE among habitat types (F=1.12, P=0.347, R^2= 0.011). For electrofishing, a two variable model with section and maximum depth best explained differences in CPUE, but was not significant (F=1.68, P=0.212, R^2=0.061). Neither of the variables in this model was significant (Table 4).

CPUE differences in fall were significantly explained by habitat variables. For gillnetting, a three variable model with section, width, and 1.5-m depth was significant in explaining CPUE (F=3.31, P=0.041, R^2=0.232) and variables were all individually significant (Table 4). Similarly, electrofishing CPUE was significantly explained by a model with variables section, overhanging vegetation, slope, 1.5-m depth, and maximum depth (F=15.93, P<0.001, R^2=0.781). Variables section, overhanging vegetation, and slope were significant (Table 4). Aquatic vegetation was not present in spring and was minimal in fall; therefore, it was not included in the regression model for these seasons.

Relative Fish Abundance, Depth of Riprap, and Riprap Rock Diameter

CPUE at stabilized sites was significantly explained by the habitat variables riprap depth and rock diameter during certain season/gear combinations (Figure 17, Figure 18). Gillnetting CPUE was positively correlated with riprap depth during all seasons (Figure 17), and showed the strongest correlation in summer (F=8.10, P=0.017, R^2=0.3922). Gillnetting CPUE showed a weakly positive correlation with riprap diameter in summer and spring and a weakly negative correlation in fall. Electrofishing CPUE showed negative correlations with riprap depth in summer and fall, and showed

Table 4. Results of a multiple regression using Akaike's Information Criterion (AIC), showing habitat variables that best explained differences in CPUE by season (summer, spring, and fall) and gear (gillnetting and electrofishing).

	Gillnetting			**Electrofishing**		
	Variable	**t**	***P*-value**	**Variable**	**t**	***P*-value**
Summer	Section	2.24	*0.040*	Width	-2.36	*0.032*
	Slope	-1.62	0.124	Overhanging vegetation	2.09	*0.053*
	Width	-2.54	*0.022*	Aquatic vegetation	3.62	*0.002*
	Overhanging vegetation	-2.68	*0.016*	Slope	-2.73	*0.015*
	1.5-m depth	2.52	*0.023*	Max depth	2.14	*0.048*
	3-m depth	-1.69	0.110			
Spring	Width	-1.35	*0.045*	Section	1.38	0.185
	3-m depth	0.89	0.383	Max depth	1.18	0.252
Fall	Section	1.87	*0.076*	Section	5.27	*<0.001*
	Width	-2.60	*0.017*	Overhanging vegetation	1.99	*0.064*
	1.5-m depth	2.27	*0.034*	Slope	1.99	*0.064*
				1.5-m depth	1.44	0.170
				Max depth	1.69	0.111

a weakly positive correlation in spring. CPUE was significantly correlated with riprap depth for both summer (F=5.61, P=0.039, R^2=0.295) and fall (F=13.65, P =0.004, R^2=0.535).

CPUE showed positive correlations with riprap diameter for all season/gear combinations except fall gillnetting (Figure 18). Gillnetting CPUE was not significantly correlated with riprap diameter during any seasons. Electrofishing CPUE showed a positive correlation with riprap diameter in all seasons, though only spring was significant (F=5.33, P=0.044, R^2=0.283).

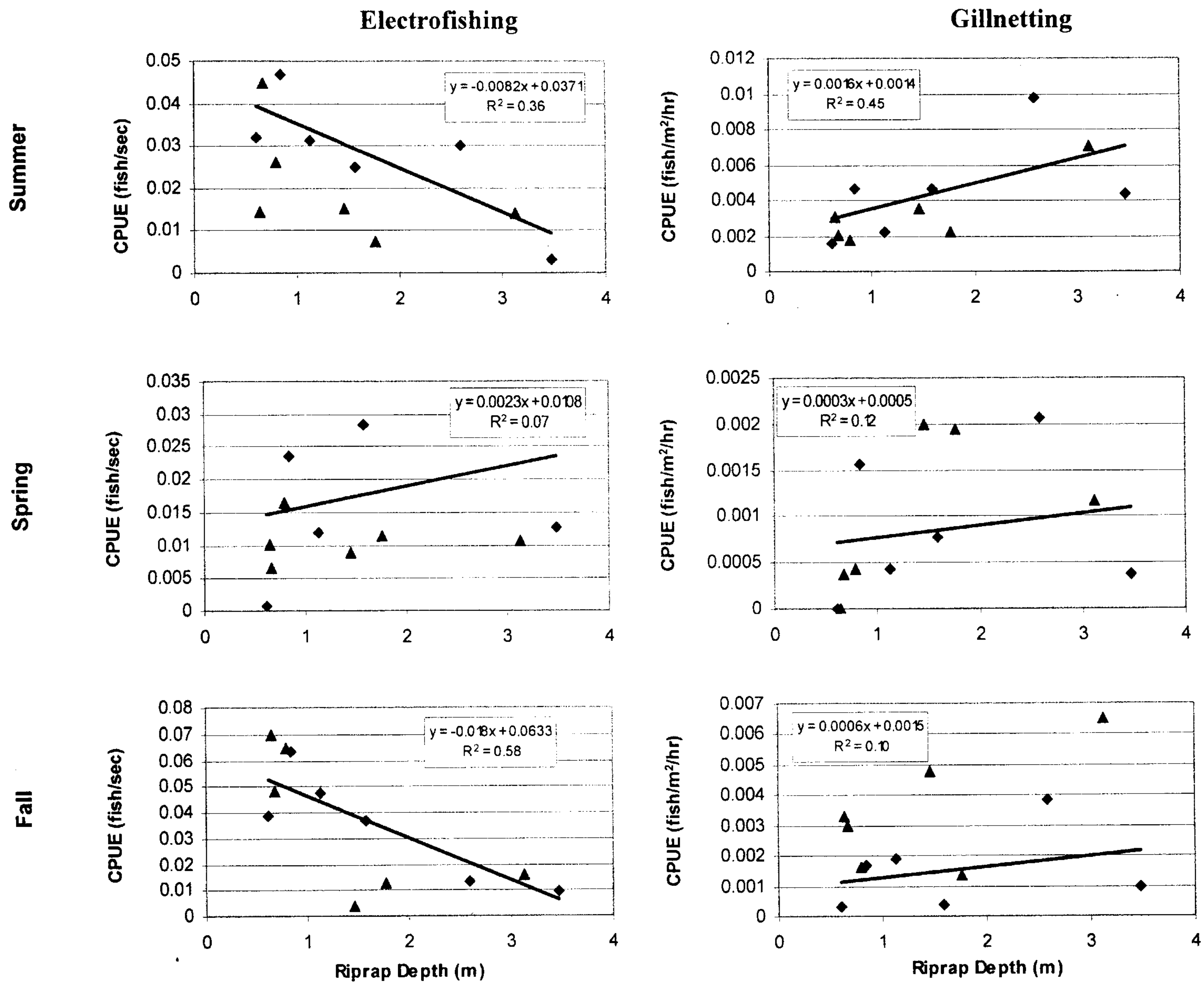


Figure 17. Linear regressions of CPUE (y-variable) with riprap depth (x-variable) for gear (gillnetting and electrofishing) and season (summer, spring, and fall). Slope equations and r^2 values are shown in the insets. ▲=RRV ♦=RR sites.

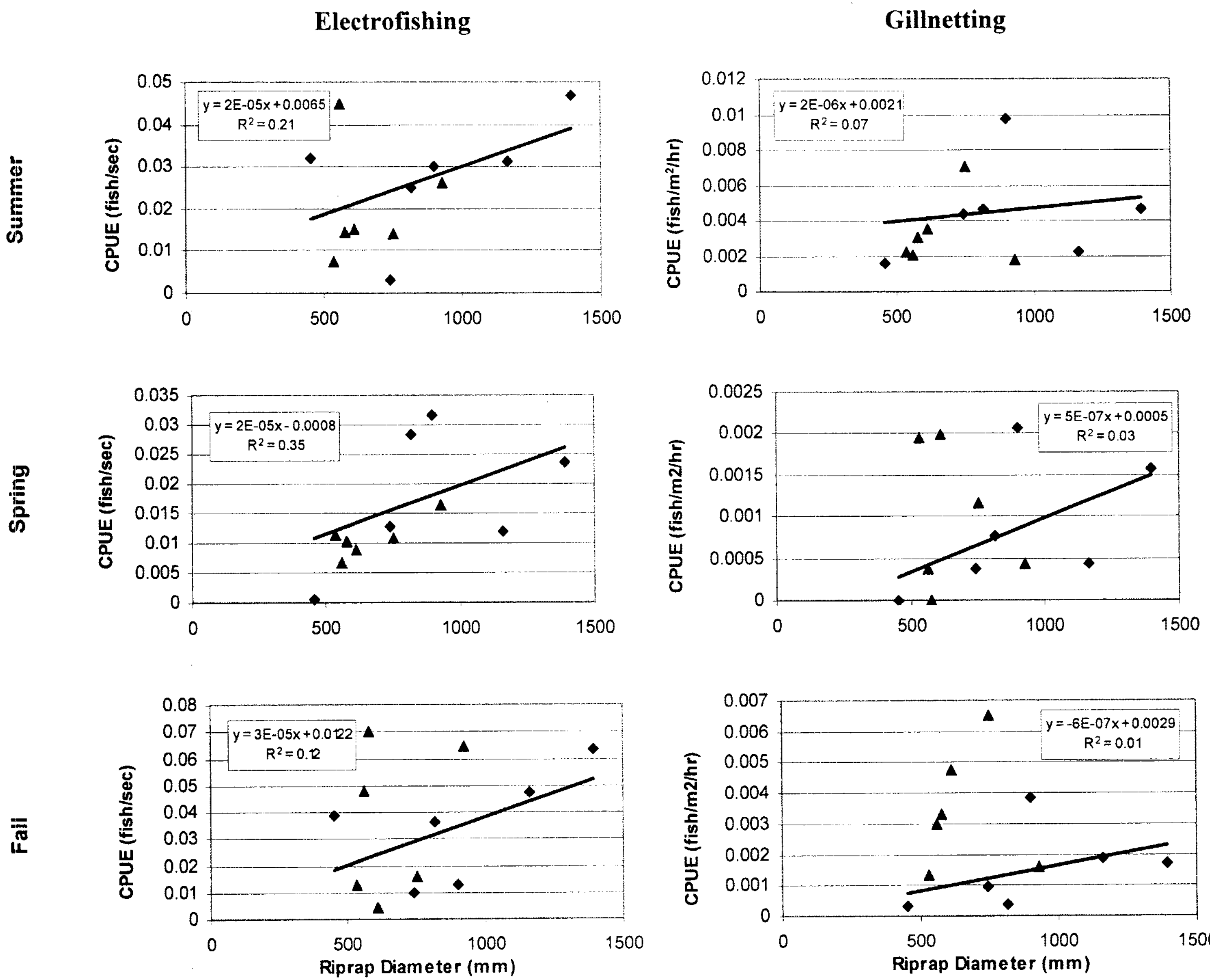


Figure 18. Linear regressions of CPUE (y-variable) with riprap diameter (x-variable) for gear (gillnetting and electrofi shing) and season (summer, spring, and fall). Slope equations and r^2 values are shown in the insets. ▲=RRV ◆=RR sites.

Discussion

The higher overall relative fish abundance (CPUE) at stabilized (RR and RRV) sites in comparison with unstabilized (V and FB) sites reported in this study is contrary to several other studies that have shown a decrease in fish abundance with bank stabilization (Chapman and Knudsen 1980, Knudsen and Dilley 1987, Schmetterling et al. 2001, Garland et al. 2002). Chapman and Knudsen (1980) identified decreased habitat due to channelization as the cause of decreased cutthroat trout and overall salmonid biomass. Similarly, Schmetterling et al. (2001) cited several studies in which salmonid densities decreased as a result of habitat loss through bank stabilization, particularly due to decreased amounts of LWD.

Other studies, however, have reported increases in fish abundance in areas with bank stabilization (Farabee 1986, Dardeau et al. 1995, Lister et al. 1995, Trial et al. 2001, Zale and Rider 2003). On the upper Mississippi River, catch at stabilized sites (70% of fish) was higher than natural sites and several species were captured in greater numbers at these stabilized sites (Farabee 1986). Steelhead *Oncorhynchus mykiss* in a British Columbia stream were more abundant along a riprapped shoreline than a shoreline with trees and shrubs (Swales and Levings 1989). Zale and Rider (2003) reported that densities of juvenile salmonids in rip-rapped sections of the Upper Yellowstone River were higher than in natural outside bends of the river.

The conflicting results of the above studies regarding the beneficial or adverse impacts of bank stabilization may result from at least two factors. First, in systems where suitable natural habitat is unfavorable or limiting, riprap and other bank stabilization structure may provide habitat complexity where little is otherwise present. The lack of measurable adverse impacts from bank stabilization in this study may thus have resulted from the poor-quality, low-diversity habitat in unstabilized areas. This hypothesis is supported by the low Rapid Bioassessment Protocols scores found in the lower river. In an earlier study in the basin, Maret and MacCoy (2002) reported that, "instream cover was found to be limited at all sites, with woody debris especially scarce" even upstream on the North and South Forks, where habitat is better than that found in the lower river. Low overall fish species diversity (as indicated by the Shannon Index) can also be interpreted as indicative of low diversity and quality of habitats. Studies have shown a correlation between species diversity and habitat diversity (Schlosser 1982).

Other studies elsewhere support our interpretation. Zale and Rider (2003) reported habitat for juvenile salmonids in the natural portions of the mainstem Upper Yellowstone River were relatively poor compared to many other locations, and under these conditions, additional structure provided by riprap and other stabilization materials may have proved beneficial to immediate, on-site habitat conditions. In western Washington, yearling cutthroat trout and steelhead standing stocks increased in newly riprapped sections in large streams. The increase in cutthroat was attributed to increased quantity of habitat (Knudsen and Dilley 1987).

Conversely, natural rivers and small streams where high-quality habitat such as large woody debris exists may suffer declines in habitat quality from actions associated with bank stabilization (Elser 1968, Angermeier and Karr 1984, Knudsen and Dilley 1987, Angradi et al. 2004, Craig and Zale 2003). In western Washington streams, the severity of habitat alterations was related to stream size: salmonid standing stocks decreased as a result of riprap addition in small streams and increased in large streams (Knudsen and Dilley 1987). Craig and Zale (2003) surmised that diversity and abundance of salmonids at stabilized banks, as compared with unstabilized banks, often increased in previously degraded habitats and decreased in pristine habitats. As summarized by Zale and Rider (2003), "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… habitat is limiting" (p. 13).

Secondly, the measurable response of a fish community to bank stabilization structures will depend not only on the habitat quality of the unstabilized river, but also on the overall extent of bank stabilization in the basin. In the lower Coeur d'Alene River, where only 2.5% of the river bank is currently stabilized with riprap (KSSWCD 2004), the main aspects of natural river function persist, including a natural hydrograph, connectivity with the floodplain, and exchange of nutrients and biota between main channel and off-channel areas (Bookstrom et al. 2004). Under these conditions, modest bank stabilization functions positively as structure and provides habitat complexity. However, as bank stabilization covers an increasing percentage of the riverbanks, it functions negatively as gains in local habitat quality are exceeded by losses in river function. This interpretation is supported by evidence from several studies. As applications of riprap become more dominant in a river channel, the outcomes are channels having more uniform gradients, fewer natural riffles and pools (Keller 1975, Schmetterling et al. 2001), less large woody debris (Chapman and Knudsen 1980, Angradi et al. 2004, Schmetterling et al. 2001), altered flow patterns (Pegg et al. 2003), reduced connectivity to the floodplain (Ward and Stanford 1995), and reduced aquatic species diversity and biomass (Elser 1968, Chapman and Knudsen 1980, Scarnecchia 1988). Fish communities are also negatively impacted. For example, major losses in habitat quality for native fishes have been reported on the lower Missouri River, where excessive bank stabilization has converted most of the river into a lined channel (Hesse et al. 1989, Morris et al. 1968, Hesse and Sheets 1993). The detrimental effects of riprap and other bank stabilization on river function and habitat thus become cumulative and detrimental when applied to large stretches of river (Jennings et al. 1999, Schmetterling et al. 2001).

The results of this study that stabilized banks were not associated with lower densities of the immediate fish community around the structures (and may even be associated with higher fish densities) thus should not be extrapolated greatly beyond existing river conditions. My study was designed only to compare the fish communities at stabilized and unstabilized sites in a largely unstabilized river. Indirect and cumulative effects of riprap were not considered in this study.

Seasonal differences in relative abundance among habitat types indicate that stabilized RR and RRV structures, including boat ramps, on the lower Coeur d'Alene River are providing local habitat improvements during all times of year. In Lake Conroe, Texas, riprap structures provided habitat that was constant year-round in comparison with seasonally variable vegetated sites (Trial et al. 2001). On the lower Coeur d'Alene River, vegetation was abundant in summer but largely absent in spring and fall whereas riprap provided at least some habitat in all seasons, having the highest relative abundance among habitat types for all seasons. Some of the riprapped areas in this study were not submerged at winter pool elevations, however, because they were only armored at the wave line and above. My data analysis did not account for this seasonal difference in the amount of riprap habitat available, but I estimated that four of the stabilized shoreline sites that I sampled had less than 0.3 m of submerged riprap in fall.

The domination of the species composition of the lower Coeur d'Alene River by non-native fish species (84% of the total catch), including bullhead, crappie, bluegill, Chinook salmon, kokanee, largemouth bass, smallmouth bass, northern pike, pumpkinseed, tench, and yellow perch, was not unexpected, because such species often flourish after habitat alterations. Non-native fishes have been shown to have detrimental effects on native fish and other aquatic organisms through predation, competition, and hybridization (Wydoski and Bennett 1981, Miller et al. 1989, Ross 1991, Schade and Bonar 2005). A recent study showed that, in the western U.S., one of every four individual fish is non-native (Schade and Bonar 2005), which is a conservative estimate for the lower Coeur d'Alene River, where the number of non-native species outnumber native species by nearly 4 to 1.

Differences in relative fish abundance and species composition, for both section (upstream, downstream) and season (summer, spring, and fall) observed in this study, can be largely attributed to responses of different species to water temperature and spawn timing. Most species captured in greater numbers upstream are classified as coolwater species, including largescale suckers and pikeminnow, and coldwater species such as longnose suckers and cutthroat trout (Simpson and Wallace 1978, Zaroban et al. 1999, Mebane et al. 2003). The greater abundance of cutthroat trout upstream was associated with the proximity of more suitable habitat, including increased water velocities and lower water temperatures in the North Fork (Dupont IDFG personal communication, Parametrix 2005). In addition to upstream/downstream temperature differences, seasonal differences in water temperature were also associated with changes in relative abundance and species composition. Coldwater species (cutthroat trout, kokanee, Chinook salmon, and mountain whitefish) were captured in greater numbers during spring when water temperatures were relatively low. Catches of warmwater and some coolwater species were lowest in spring probably because these species were seeking warmer temperatures in the adjoining chain lakes. Although this study did not focus on water temperatures, on May 31 the temperature in the Thompson Lake channel was 18°C whereas the temperature in the main river was only 10°C. Summer and fall were similar in terms of relative abundance and species composition as well as temperatures.

Spawn timing also contributed to differences in relative abundance and community composition among seasons and between sections. Most fish captured in greater numbers downstream were classified as warmwater species, including largemouth bass and brown bullhead, or as coolwater species, including smallmouth bass and yellow perch. The majority of these warmwater and coolwater fish species spawn in early to late spring, from the time ice breaks up until temperatures are warmer. Species such as smallmouth bass, largemouth bass, and northern pike are known to be present in greater numbers in the chain lakes than in the river in late spring as a result of their spawning activities (Mark Liter Idaho Department of Fish and Game (IDFG), Personal Communication). Northern pike migrate to flooded marshes and wetlands or shallow shorelines with vegetation shortly after ice-out (Casselman and Lewis 1996). In this study, northern pike were completely absent from the spring catch. Northern pikeminnow spawn from late May to early July over gravelly substrates in shallow water (Simpson and Wallace 1978) when temperatures increase from 9° to 12°C (Reid 1971, Beamesderfer 1983). Such habitats are largely absent from the lower river and these fish are likely seeking such habitats further upstream in the basin. Whereas warm- and many coolwater species catches were low in spring, coldwater salmonids were most abundant, as juvenile Chinook salmon, kokanee, and cutthroat trout were outmigrating to Lake Coeur d'Alene from spawning and rearing areas higher in the basin.

Differences in fish relative abundance (CPUE) were most definitively explained by the habitat variable section (i.e. upstream vs. downstream), consistent with other reports that have shown a gradation of change in community composition from upstream to downstream corresponding to changes in habitat (Sheldon 1968, Hynes 1970, Vannote et al. 1980, Schlosser 1982). In this study, the major difference between upstream and downstream sections was the proximity of the downstream section to the chain lakes and their associated wetland habitats. Such floodplains associated with large rivers provide a variety of habitats including backwaters, marshes, and lakes, and are typically warmer, highly productive, and valuable for fish species (Forbes 1925, Guillory 1979, Amoros 1991, Ross and Baker 1983).

Though the habitat variable section had the greatest power in explaining differences in CPUE, other habitat variables were also explanatory. Aquatic vegetation explained a significant amount of variability in CPUE for electrofishing (R^2=0.297, *P*= 0.003) and appeared to provide important habitat and cover in summer. Killgore et al. (1989) found that overall mean fish abundance was highest at sites with high submersed aquatic plant density in the Potomac River. The greater amount of aquatic vegetation at V sites than other habitat type sites may have been due mainly to lower average bank slope and lower maximum depth in these areas. Vegetation was completely absent in spring and had largely died back by fall. Macrophytes and aquatic vegetation provide habitat for fish in areas where other cover may not be available (Killgore et al. 1989). The amount of overhanging and shoreline vegetation on the lower river was highly dependent upon the riparian zone. In many areas, land uses such as agriculture and the old railroad bed (now the "Rails to Trails" path) limit the amount of vegetation. Most of the stabilized areas were implemented due to failure of the railroad bed or the trail and therefore have little or no vegetation. Shoreline vegetation has been shown to directly influence the density of LWD in rivers. Where shorelines were forested, LWD density was much higher than in areas where shorelines were open (agricultural or residential areas) or stabilized (Angradi et al. 2004). However, in the current study, few fish were electrofished out of areas that contained woody debris. The woody debris in the study reach was heavily silted, providing little cover.

At stabilized RR and RRV sites, higher fish relative abundance was correlated with greater riprap diameter, a finding that is corroborated by several other studies. On the Upper Mississippi River Farabee (1986), found that catch was highest at the station with the largest diameter rock (averaging >60 cm in diameter, and loosely placed) during 5 out of 6 months sampled between May and October. Similarly, juvenile Chinook salmon and steelhead densities were higher at sites with larger riprap (>30 cm diameter) in two southern British Columbia streams (Lister et al. 1995). The sites on the lower Coeur d'Alene River with the largest riprap diameters show the greatest affinity for providing habitat due to the availability of interstitial spaces among larger boulders. The Rose Lake boatramp had a much larger average riprap diameter than the other two ramps, and had a higher CPUE than any other site. This site seems to exemplify the correct combination of rock diameter, depth, and placement to maximize local fish abundance on the lower Coeur d'Alene River.

The relationship between CPUE and riprap depth was dependent on gear during this study, which may be explained by differences in length selectivity. Gillnets captured larger fish, which tend to inhabit deeper waters, and are likely to benefit from deeper riprap applications. In turn, the smaller average size of fish captured by electrofishing likely explains the negative correlation between CPUE and riprap depth in summer and fall. Juvenile salmonids captured in British Columbia streams similarly had higher densities in areas with large riprap and greater depths (Lister et al. 1995). In spring, both gears sampled larger fish than during other seasons, which explains the slightly positive relationship between CPUE and riprap depth.

Conclusions

- Relative abundance at stabilized banks was consistently higher than at unstabilized banks among all seasons on the lower Coeur d'Alene River.

- Stabilized banks did not show statistically significant losses in fish species diversity and community composition as compared with unstabilized banks.

- Bullhead, northern pike, and pumpkinseed were the only species captured in greater numbers at stabilized than unstabilized sites.

- Large diameter riprap supported a higher abundance of fish than smaller riprap.

- Greater riprap depth supported a higher abundance of fish species and lengths more readily captured by gillnets, whereas shallow riprap depth supported a higher abundance of species and lengths more readily captured by electrofishing.

• In spring, water temperature and spawn timing influenced fish abundance and composition. The abundance of warm-water species (e.g., largemouth bass) and cool-water species (e.g., smallmouth bass, northern pike, and northern pikeminnow) was lowest in the spring whereas coldwater species abundance (cutthroat trout, Chinook, kokanee, and mountain whitefish) was highest in the spring.

• Upstream and downstream sections were markedly different in terms of relative abundance; larger species (and individual fish) such as suckers and northern pikeminnow were captured in greater numbers upstream, whereas smaller fish and fish such as largemouth bass, smallmouth bass, bass, perch, and bullhead were captured in greater numbers downstream.

• Under existing conditions, with a low percentage (2.5%) of the total bank stabilized with riprap, these banks provided structure at that benefitted the overall fish community. The stabilization provided habitat in a river system with existing low-quality, low-diversity habitat.

• Application of large riprap structure at selected high-impact, individual sites along the lower Coeur d'Alene River can be expected to result in positive or neutral benefits to the overall fish abundance, community diversity, and species composition, as long as the main portion of the channel remains unstabilized. Based on the scope of this study, outcomes cannot be predicted as the number of riprap structures increases and the cumulative effects of loss of river function occur.

Chapter Two

Salmonid and Piscivore Dynamics Associated with Stabilized and Unstabilized Shoreline Habitats on the Coeur d'Alene River, Idaho

Abstract

In 2005 and 2006, fish samples from 24 sites on the lowermost 54 km of the Coeur d'Alene River were analyzed to assess possible impact of piscivores (mainly smallmouth bass *Micropterus dolomieui*, northern pikeminnow *Ptychocheilus oregonensis*, and northern pike *Esox lucius*) on native salmonids, especially westslope cutthroat trout *Oncoryhynchus clarki lewisi.* The objectives were to 1) evaluate and quantify salmonid use of stabilized and unstabilized shoreline habitats by season and river section, 2) evaluate and quantify piscivorous species use of stabilized and unstabilized shoreline habitat, by season and river section, and 3) determine if an overlap exists between salmonid and piscivores use of stabilized and unstabilized shoreline habitats by season and river section.

In all, 81 salmonids were captured, or 2% of the total fish catch. Salmonid catch in the lower river was greatest during spring when water temperatures were low and juveniles were outmigrating to Lake Coeur d'Alene. No significant differences in catch were evident between stabilized and unstabilized habitats ($\chi2=0.064$, $P=0.800$) though juvenile westslope cutthroat trout showed some affinity for stabilized areas with 10 out of 12 individuals captured at these sites.

In all, 670 piscivores were captured, or 19 % of the total fish catch. Piscivore catch was significantly lower in spring than fall and summer ($\chi2=17.465$, $P<0.001$). Overall, piscivores were not captured in significantly different numbers at stabilized and unstabilized habitats ($\chi2=0.243$, $P=0.622$); however, northern pike catch was significantly higher at stabilized than unstabilized habitats. The overall effects of habitat type on salmonid and piscivore overlap were not clear. Based on the data available, season seems more important than habitat in affecting salmonid and piscivore

impacts. The highest abundance of salmonids in spring, when predators are least abundant, may result in some benefit to migrating salmonids. Specific studies are outlined that must be conducted, however, for a clearer understanding of the relation between salmonids and potential predation from piscivores in the lower river.

Introduction

In the Coeur d'Alene basin, Idaho, both westslope cutthroat trout *Oncorhynchus clarki lewisi* and bull trout *Salvelinus confluentus* have declined greatly in abundance in the last century (Mauser et al. 1988, USFWS 1999, Joe Dupont, Idaho Department of Fish and Game (IDFG), Personal Communication). Bull trout were last observed in 1998 in the Coeur d'Alene River basin and are considered to be at high risk of extirpation in the Coeur d'Alene River (Cross and Everest 1995, Scott Deeds U.S. Fish and Wildlife Service (USFWS) Personal Communication). Cutthroat trout are found throughout the basin, primarily in the North Fork and its tributaries (Joe Dupont, IDFG, Personal Communication).

The lower Coeur d'Alene River has historically served as an important migratory corridor and over-wintering habitat for both westslope cutthroat and bull trout exhibiting adfluvial and fluvial resident life histories (Lewynsky 1986, Dunnigan 1997). As of 2007, resident westslope cutthroat trout populations are present only in the headwaters. In 2003, Dupont (unpublished), found that adfluvial adult cutthroat migrated upriver in April and May to spawn in June and subsequently returned to the lake. Similarly, in the St. Joe River, Idaho Averett and MacPhee (1971), reported that adult cutthroat trout began their upstream migrations in early April with spent adults being captured by anglers in the lower river during their migration back to the lake in early June through July. Juvenile adfluvial cutthroat trout spend an average of 2-4 years rearing in the river before migrating downstream to the lake (Dunnigan 1997). As temperatures begin to drop in the fall, cutthroat migrate to overwintering areas, particularly pools associated with large woody debris (LWD) (Brown and MacKay 1995, Jakober and McMahon 1998, Parametrix 2005).

Stabilizing riverbanks with riprap has been shown in many cases to have a detrimental effect on fish habitat (Schmetterling et al.2001, Gorman and Karr 1978), especially in places where stabilization degrades existing high-quality habitats (Elser 1968, Knudsen and Dilley 1987, Peters et al. 1998) commonly used by intolerant species such as most salmonids (Mebane et al. 2003, Zaroban et al. 1999). Bank stabilization and channelization can eliminate or adversely impact habitat such as undercut banks, LWD, and riffles and pools (Schmetterling et al. 2001). On the Upper Missouri River, areas with bank stabilization had much lower concentrations of LWD (7.2 pieces per km) than did unstabilized areas (27.3 pieces per km) (Angradi et al. 2004). Naturally occurring riffles and pools, often eliminated by riprap, are important to fish as well as other aquatic life (Leopold et al. 1964). In addition, LWD and undercut banks, eliminated by riprap, provide important habitat in coldwater systems inhabited by salmonid populations (Bryant 1983).

Salmonids have been shown to exhibit a significant preference for unstabilized over stabilized areas of lakes and rivers (Elser 1968, Knudsen and Dilley 1987, Garland et al. 2002). Garland et al. (2002) found that the probability of fish presence was greater in unstabilized habitats than in riprap habitats and substrate size was most important in determining presence of sub-yearling fall chinook salmon *Oncorhynchus tshawytscha.* Similarly, in Little Prickly Pear Creek, Montana where interstate highway construction led to a replacement of riparian vegetation with riprap, trout (brown trout *Salmo trutta*, rainbow trout *Onchorhynchus mykiss*, and brook trout *Salvelinus fontinalis*) were 78 percent more abundant in the unaltered than in the altered sections (Elser 1968).

Applications of riprap can also benefit certain species or life stages, especially under circumstances where existing natural habitat is poor. A study of the Coldwater River, British Columbia found that juvenile steelhead *Onchorhynchus mykiss* were predominant at riprapped areas during winter (Swales and Levings 1989). In four western Washington streams, salmonid biomass was higher in channelized reaches with significantly reduced overhead cover, sinuosity, wetted area, and woody cover than in control reaches (Chapman and Knudsen 1980). The size of rock used in riprap structures has also been shown to have an effect on salmonids. Large size riprap has been shown to support higher numbers of juvenile salmonids than smaller rocks and cobbles (Lister et al. 1995).

Piscivorous fish species can have pronounced influences on an aquatic ecosystem including reducing the abundance or size of a prey species, or altering the overall species composition (Moyle and Li 1979). In the Coeur d'Alene River basin largemouth bass *Micropterus salmoides*, smallmouth bass *Micropterus dolomieui*, northern pikeminnow *Ptychocheilus oregonensis*, and northern pike *Esox lucius* are the primary predators. Smallmouth bass, northern pikeminnow, and northern pike have been shown to be voracious predators on other fishes and particularly on salmonids (Brown and Moyle 1981, Zimmerman 1999, Fresh et al. 2003, Fritts and Pearsons 2006).

Largemouth bass are present in the chain lakes as well as the lower river and are largely associated with vegetated areas. Largemouth bass from the chain lakes are among the fastest growing in northern Idaho (Dillon 1991, Fredericks 2002). This species becomes piscivorous early in life. Spawning in Idaho occurs when temperatures reach 15-18°C in areas with sand, small gravel, or rooted vegetation (Simpson and Wallace 1978).

Smallmouth bass, a widespread piscivore introduced to rivers and lakes throughout Idaho, become piscivorous at an earlier age than northern pikeminnow and have been shown to prey on salmonids (Zimmerman 1999, Fritts and Pearsons 2006). Preferred habitats include cool stream waters with riffle areas and clean gravel or cobble substrates, as well as rock outcrops or ledges (Simpson and Wallace 1978). Spawning of smallmouth bass in Idaho occurs when temperatures reach 15-18°C in areas with sandy gravel or rock substrates (Simpson and Wallace 1978).

Northern pikeminnow, a piscivore native to the Coeur d'Alene River, has had serious negative impacts on juvenile salmonids throughout the western U.S. (Brown and Moyle 1981, Zimmerman 1999, Fresh et al. 2003). Major eradication efforts, including applications of a selective piscicide (Squoxin) and bounties, have targeted this species (Brown and Moyle 1981). Habitat preference varies with fish size, with young fish usually occurring in schools in shallow, low velocity areas with sand or silt substrates, whereas adults are found alone or in pairs in moderately deep, low velocity areas with gravel or cobble substrates (Beamesderfer 1983). Pikeminnow spawn from late May to early July over gravelly substrates in shallow water (Simpson and Wallace 1978). In the St. Joe River, fish began spawning when temperatures increased from 9° to 12°C, around the middle to end of June (Reid 1971, Beamesderfer 1983), migrating from Lake Coeur d'Alene to areas upstream. Some fish return to the lake after spawning while others remain in the river until late fall (Reid 1971).

Northern pike were illegally introduced into the Coeur d'Alene River in 1973, and represent the only significant pike fishery in the state of Idaho (Rich 1992). Northern pike prefer weedy habitats, often ambushing their prey from submerged vegetative cover (Simpson and Wallace 1978, Raat 1988, Casselman and Lewis 1996). Rich (1992) found that pike preferred vegetated habitats in shallow (<5 m) areas in the Coeur d'Alene River system in summer. Spawning typically occurs in spring shortly after ice melts when water reaches 8-12°C (Simpson and Wallace 1978, Casselman and Lewis 1996). Spawning grounds consist of shallow water with vegetation, often in flooded marshes and wetlands or along shorelines (Casselman and Lewis 1996).

Little published literature is available on the use of riprap habitats by largemouth bass, smallmouth bass, northern pikeminnow, or northern pike is available. Sammons and Bettoli (1999) reported that smallmouth bass abundance was highest over rubble (natural rock banks with steep shorelines) and riprap habitats, and lowest in non riprapped habitats in Normandy Reservoir, Tennessee. Smallmouth bass have been shown to have a preference for boulder substrates (Munther1970, Rankin 1986, Todd and Rabeni 1989). Although these studies were not focused in areas with riprap, large rock stabilizations could provide the same habitats.

Understanding how salmonids as well as their potential predators use stabilized and unstabilized shoreline habitats is important in protecting the native salmonids, particularly since salmonid habitats in the basin have been degraded or diminished. Under the general community-based sampling design developed to test the effects of bank stabilization structures (Chapter One), additional information was sought related to salmonid and piscivore interactions that would be potentially useful in addressing possible impacts of piscivores on native salmonids. The major objectives of this chapter were to:

1. Evaluate and quantify salmonid use of stabilized and unstabilized shoreline habitats by season and river section.
2. Evaluate and quantify piscivorous species use of stabilized and unstabilized shoreline habitat, by season and river section.
3. Determine if an overlap exists between salmonid and piscivores use of stabilized and unstabilized shoreline habitats by season and river section.

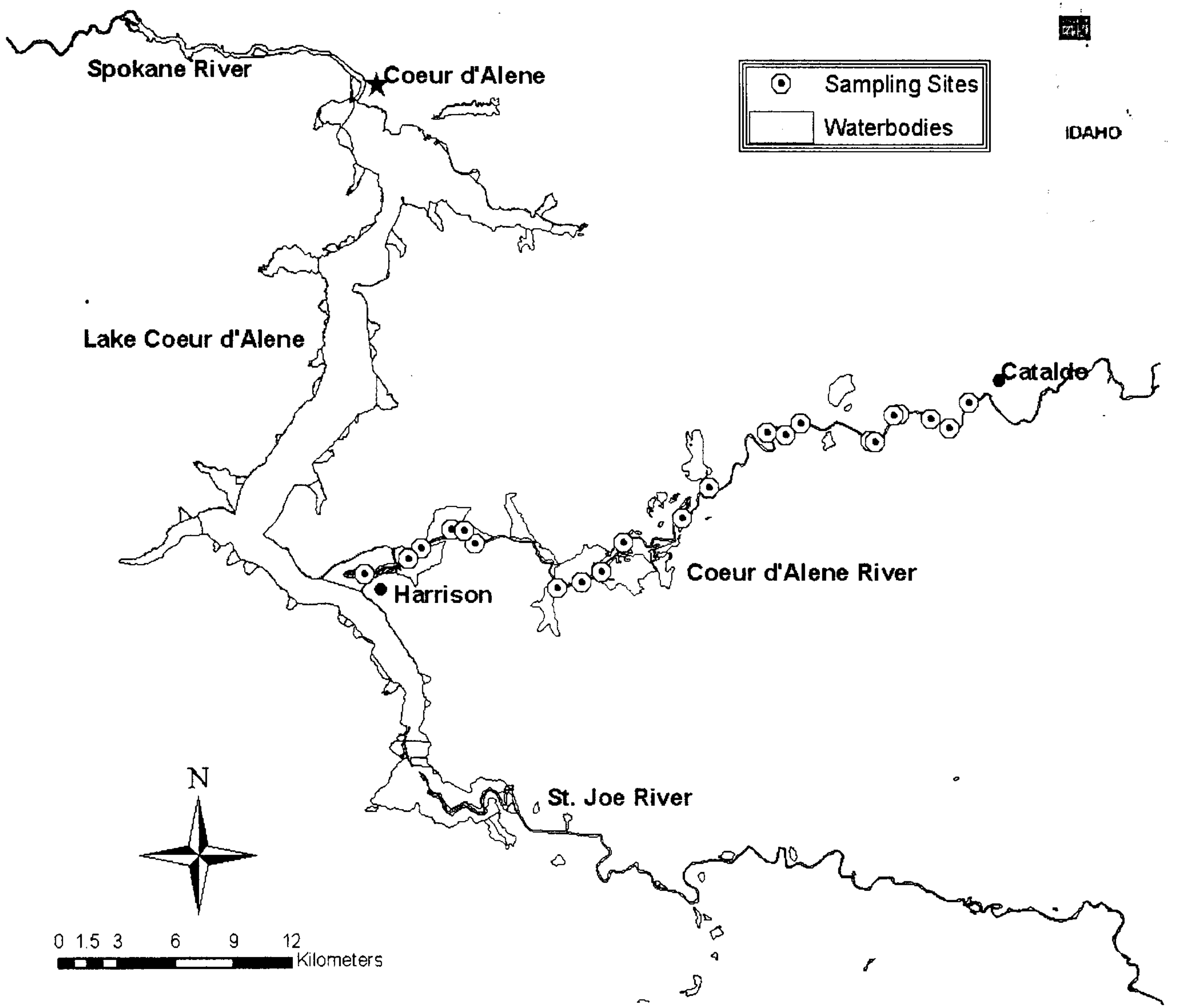


Figure 1. Map showing the Coeur d'Alene Basin, Idaho and the 24 sampling site locations and their division into upstream and downstream sections.

Methods

The lower Coeur d'Alene River was divided into two sections, the upstream section from the to the Lake Coeur d'Alene inlet (Figure 1). The two river sections were not of equal lengths because the downstream section was delineated by its direct connection with several of the chain lakes. In both sections, four major shoreline habitat types were identified, two stabilized types (riprap (RR) and riprap with vegetation (RRV)) and two unstabilized types (vegetation (V) and failing banks (FB)). The V habitat types represented an unaltered, relatively stable habitat whereas the FB habitat type was a candidate for future stabilization. For each of the four habitat types, 6 sites were identified, 3 in the upstream section and 3 in the downstream section, a total of 24 sites. All 24 sites consisted of 150 m of shoreline. In addition to these sites, 3 boat ramps with riprap were sampled. These boat ramps provide access to the river at Rose Lake, Anderson Lake, and Thompson Lake. All three boat ramp sites had riprap both upstream and downstream of the launch but were considered separately because they consisted of far less than 150 m of shoreline. Sampling at all sites was conducted during three seasons, summer, spring, and fall. Summer sampling occurred during late July and early August of 2005. Spring and fall sampling were conducted during May and October of 2006. Fall sampling in the river occurred when Lake Coeur d'Alene water levels were lower, at winter pool (646.5 m), whereas summer and spring sampling in the river occurred when lake water levels were higher, at summer pool (648.6 m). Both lake pool levels and river discharge influenced river elevation and therefore the degree to which bank structures were submerged.

Fish Sampling

Sampling was conducted at each of the 24 sites and 3 boat ramps using gillnetting and electrofishing. Gillnets are applicable in large rivers with little or no flow and effectively sample waters at depths greater than 3 m. Electrofishing has proven to be an effective sampling technique, applicable in various aquatic habitats, though usually limited to between 0.5 and 3.0 m in depth (Reynolds 1983). Because of gear-specific selectivity associated with fish size, species, and sampling location, two gears were used to provide a more representative sample of the fish community than would have resulted from using either gear alone (Weaver et al. 1993). Goffaux et al. (2005) concluded that electrofishing alone was not sufficient for assessing fish assemblages in large river systems and that gillnetting may provide additional information on fish community structure.

Experimental (30m x 2m) monofilament gillnets consisted of four panels of varying mesh size (1.9 cm, 2.54 cm, 3.81 cm, 7.62 cm). The nets were set parallel to shore, forming a loose enclosure (Figure 3). Nets were set within one hour of sunset, left to sample overnight, and removed the following morning. Both the time set and time removed were recorded. Relative abundance (catch-per-unit-effort; CPUE) was calculated as fish caught per meter squared per hour (fish/m^2/h) of sample time (Hubert 1983).

Electrofishing equipment consisted of a 6-m boat equipped with a Smith Root electrofishing unit. Pulse-DC current was used in order to minimize negative impacts to fish. Power output was maximized to effectively shock fish without causing harm and was adjusted based on water conductivity and temperature (Reynolds 1983) in the lower river. The 150-m length of shoreline was identified as adequate to assess species richness and percent abundance by ensuring that sufficient numbers of individuals were captured (Reynolds et al. 2003). CPUE was expressed as the number of fish caught per second of shock time (fish/s).

For both sampling gears all captured fish were identified to species, measured for total length and weight, and any abnormalities in body condition were noted. Fish community composition was estimated as a proportion of fish captured by habitat type, season, or section. For individual species,

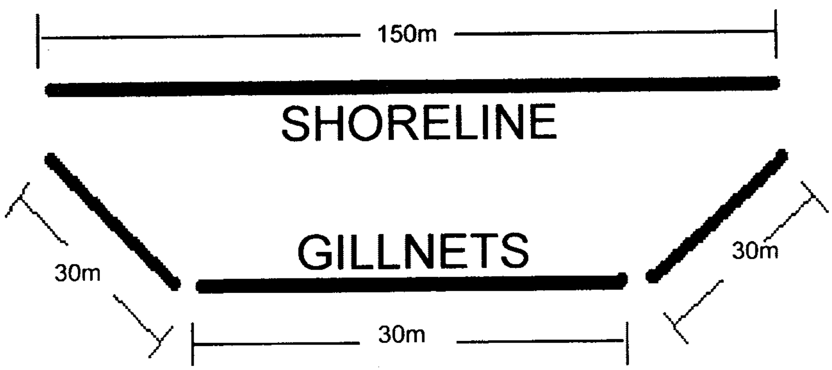


Figure 3. Schematic showing placement of experimental monofilament gillnets with respect to the shorelines for sites on the lower Coeur d'Alene River, Idaho.

adults and juveniles were distinguished by length. Fish greater than 300 mm in length were considered adults.

Statistical Analysis

In order to evaluate differences in shoreline habitat use by salmonids and piscivores, fish community composition was analyzed by using a non-parametric Kruskal-Wallis statistic (Higgins 2004). For piscivorous species, only adults (>300 mm) were included in the analysis. Composition was evaluated by habitat type, season, and section. In order to combine sampling gear data, raw numbers were used and therefore a non-parametric test was necessary, as the data was not normally distributed. All statistical testing was conducted using SAS (SAS Institute 2000) and an alpha value of 0.10 was required for significance rather than the more typical 0.05 because of the high degree of variability in large river studies.

Results

Catches during three sampling seasons on the lower Coeur d'Alene River included several salmonid as well as piscivorous species. In all, 82 salmonids were captured, or 2% of the total fish catch. Catches by species were westslope cutthroat trout (N=25), kokanee salmon (N=27), mountain whitefish (N=8), Chinook salmon (N=20), rainbow trout (1), and brook trout (1). No bull trout were captured.

Of the 82 salmonids caught, 15 were juveniles (82%). For cutthroat trout, 12 of the 25 fish caught were juveniles. Mountain whitefish *Prosopium williamsoni* catch included 3 adults and 5 juveniles. For kokanee salmon *Oncorhynchus nerka* and Chinook salmon, only juveniles were captured, averaging 57.2 mm and 75.7 mm in length, respectively. Similarly, the one brook trout and one rainbow trout captured were both juveniles.

In all, 670 piscivores were captured, or 19 % of the total fish catch. Largemouth bass were the most frequently captured species (N=359), followed by smallmouth bass (N=196), northern pikeminnow (N=83), and northern pike (N=22). Adult (>300 mm) piscivores (N=84) constituted 12.5% of the total piscivorous species catch. Few largemouth (N=4, 1%) and smallmouth (N=20, 12%) bass adults were captured. Average length for largemouth bass was 73 mm and for smallmouth bass was 140 mm. Largemouth bass adult numbers were too low to evaluate statistically.

Salmonid catch by habitat type, season, and section

Catch by habitat type – Overall, salmonid catch did not differ significantly between stabilized and unstabilized habitat types ($\chi 2$=0.064, P=0.800). Total salmonid catch was not significantly different among habitat types (χ^2=5.075, P=0.166; Figure 19), nor was westslope cutthroat trout catch ($\chi 2$=3.494, P=0.322; Figure 20). Just over a quarter of cutthroat individuals were captured at FB sites, the most undesirable in terms of habitat characteristics and Rapid Bioassessment Protocols (RBP) scores (from Chapter One). Ten of 12 juveniles were captured at stabilized (RR and RRV) sites. Catches of all other salmonid species were sufficiently low that catches were not able to be tested individually for differences among habitat types or seasons, or between sections.

Catch by season – Overall salmonid catch in spring was significantly higher than catches in both summer and fall (χ^2 = 6.574; P = 0.037). In spring, overall salmonid catches were not significantly different for habitats (χ^2=2.201, P=0.532) or sections (χ^2=1.280, P=0.258). Cutthroat trout was the only salmonid captured in all three seasons (Figure 21) and catch differed significantly among seasons ($\chi 2$ = 8.890; P = 0.011; Figure 20). Spring catch was significantly higher than catches in summer and fall; catches in summer and fall were not significantly different from each other. In spring, cutthroat trout catch (N=17) was not significantly different among habitats ($\chi 2$=5.577, P=0.134) or between sections ($\chi 2$=2.386, P=0.122).

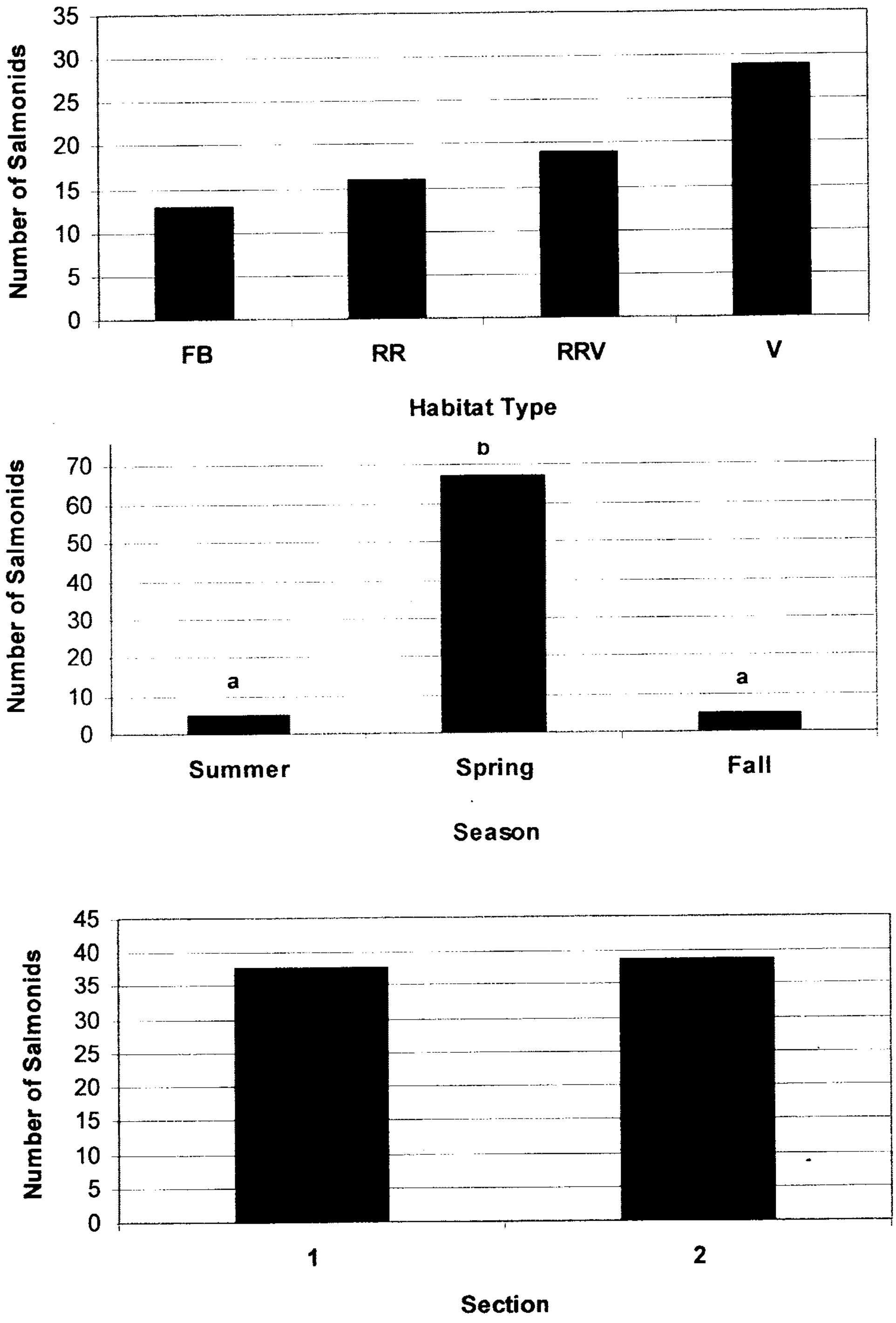


Figure 19. Distribution of salmonids among habitat types (FB, RR, RRV, V) and seasons (summer, spring, fall), and between sections (1=downstream, 2=upstream). Bars with the same letter are not significantly different.

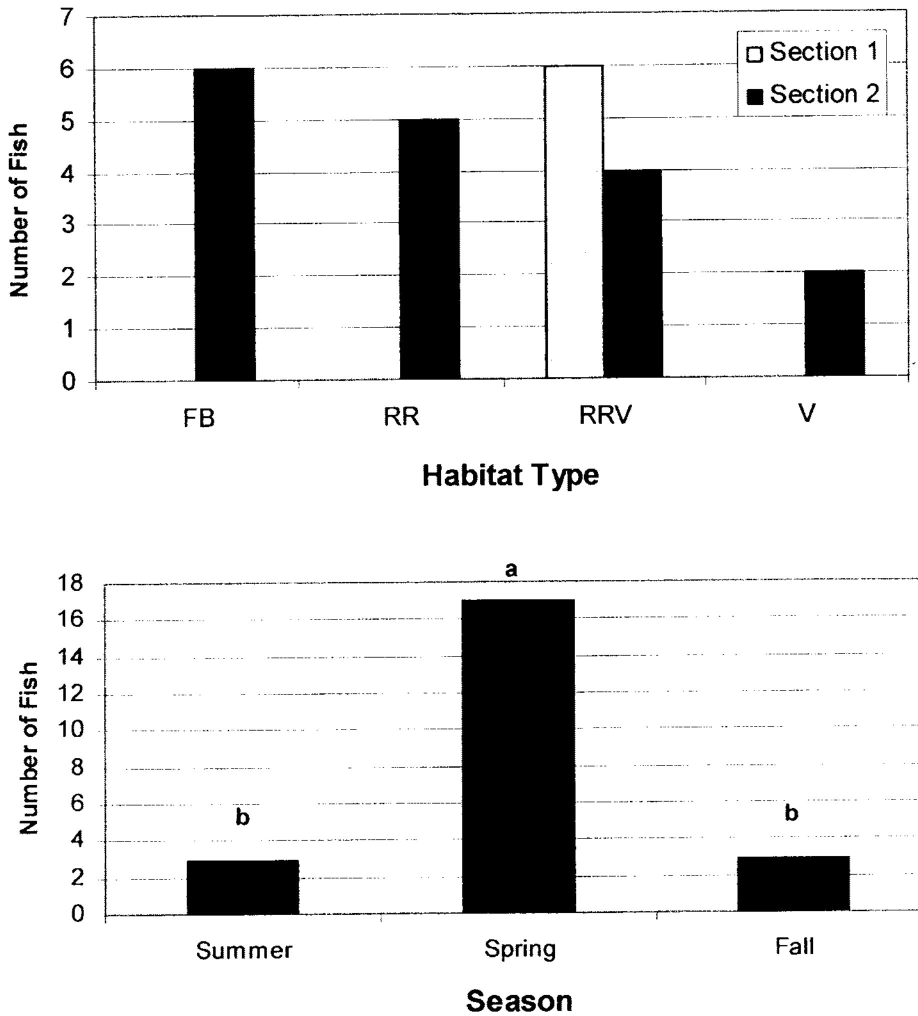


Figure 20. Distribution of westslope cutthroat trout among habitat types (FB, RR, RRV, V) and sections (1=downstream, 2=upstream), and among seasons (summer, spring, fall). Bars with the same letter are not significantly different.

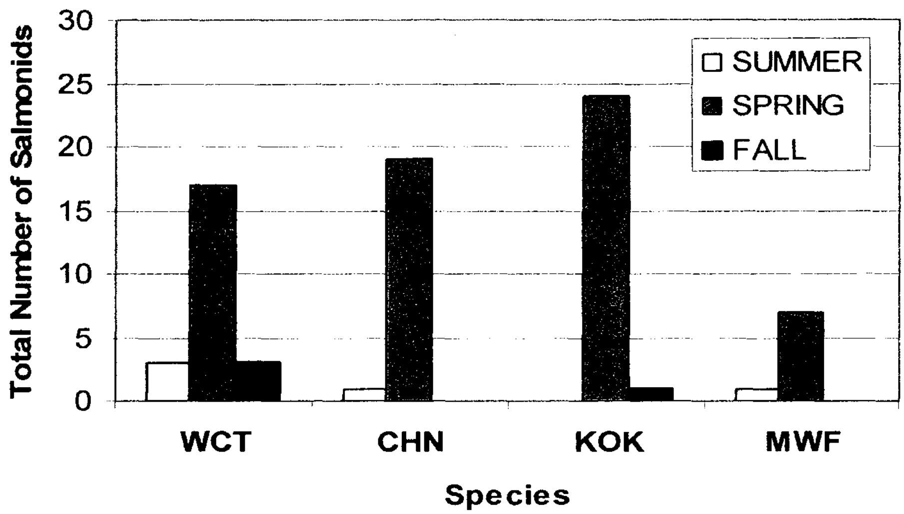


Figure 21. Total number of salmonid individuals captured during all sampling periods, delineated by season (summer, spring, fall).

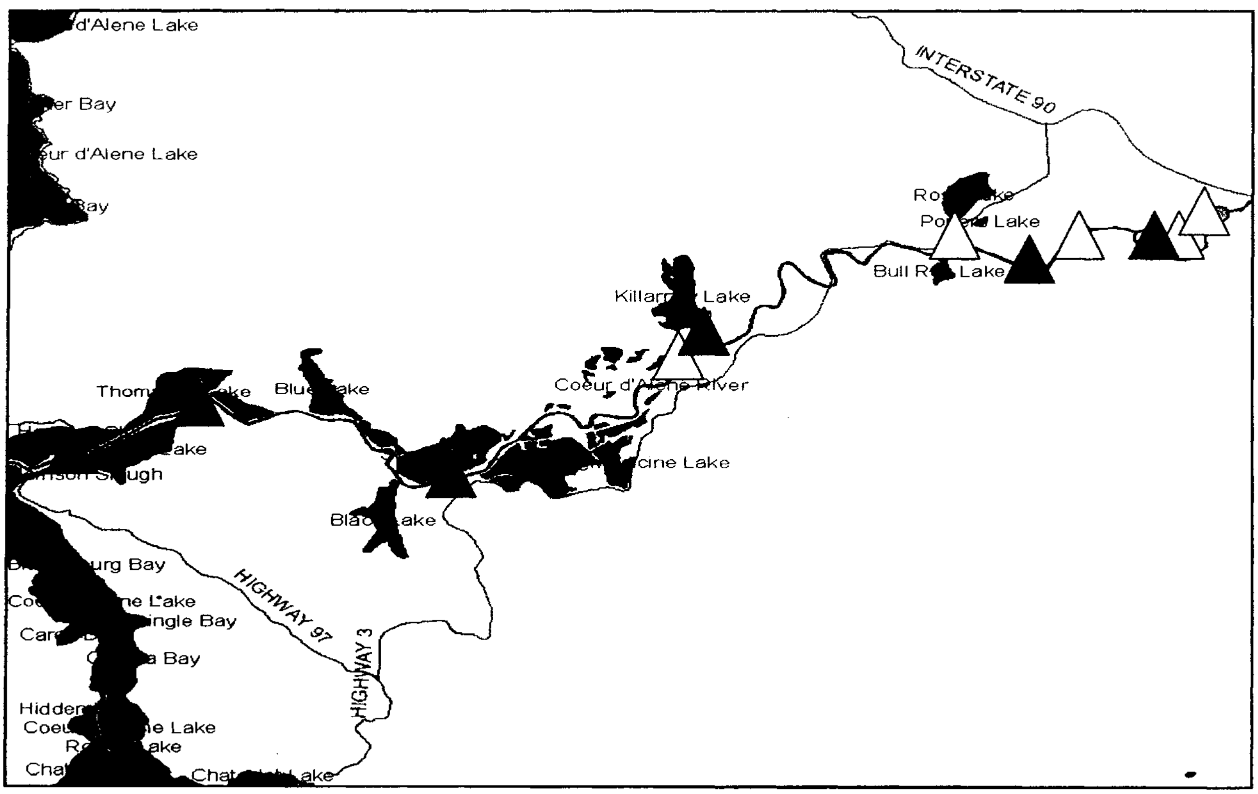


Figure 22. Westslope cutthroat locations during spring 2006. Black triangles indicate juvenile and white triangles indicate adult locations.

Catch by section– Salmonid catch was not significantly different upstream than downstream (χ^2=2.348, *P*=0.126) (Figure 19). Catch of cutthroat trout, however, was significantly greater upstream than downstream (Figure 21; χ^2=7.553, *P*=0.006). Catches of juvenile and adult cutthroat demonstrated partial spatial segregation (Figure 22). Juveniles were captured at sites throughout the entire study area, whereas adults were captured largely in the upstream section.

Piscivore catch by habitat type, season, and section

Catch by habitat type – Overall piscivore catch did not differ significantly between stabilized (RR and RRV) and unstabilized (FB and V) sites ($\chi2$=0.243, *P*=0.622), nor was catch significantly different among individual habitat types ($\chi2$=2.834, *P*=0.418; Figure 23). Smallmouth bass catches were not significantly different between stabilized and unstabilized sites ($\chi2$=0.078, *P*=0.780) nor among habitat types (Figure 24; $\chi2$=3.738, *P*=0.291). Northern pikeminnow were captured at all habitat types and during all seasons (Figure 24). Catches were not significantly different between stabilized and unstabilized sites ($\chi2$=2.513, *P*=0.113) but were significantly different among habitat types ($\chi2$=28.337, *P*=0.040), with FB sites having the highest catch (Figure 24). Catch at FB sites was significantly higher than at RRV sites. Catches of northern pike were significantly higher at stabilized than unstabilized habitat types ($\chi2$=4.075, *P*=0.044) but were not significantly different among individual habitat types (Figure 24; $\chi2$=4.864, *P*=0.182).

Catch by season – Piscivore catch was significantly different among seasons ($\chi2$=17.465, *P*<0.001); catch was significantly higher in summer and fall than in spring ($\chi2$=17.465, *P*<0.001; Figure 23). Smallmouth bass catch was significantly higher in summer than in spring ($\chi2$=5.888, *P*=0.053), though summer and fall catches were not significantly different. Similarly, pikeminnow catches were significantly higher in summer than in spring or fall ($\chi2$=16.225 *P*<0.001). Northern pike catch was significantly higher in fall and summer than spring (none caught) ($\chi2$=8.378, P=0.015).

Catch by section – Piscivore catch was significantly higher downstream than upstream ($\chi2$=5.588, P=0.018; Figure 23). Smallmouth bass catch was significantly higher downstream than upstream ($\chi2$ =6.195, P=0.013). Northern pikeminnow catch was significantly higher upstream than downstream ($\chi2$=4.746 P=0.029). Northern pike catch was not significantly different between the upstream and downstream sections ($\chi2$=0.066, P=0.797).

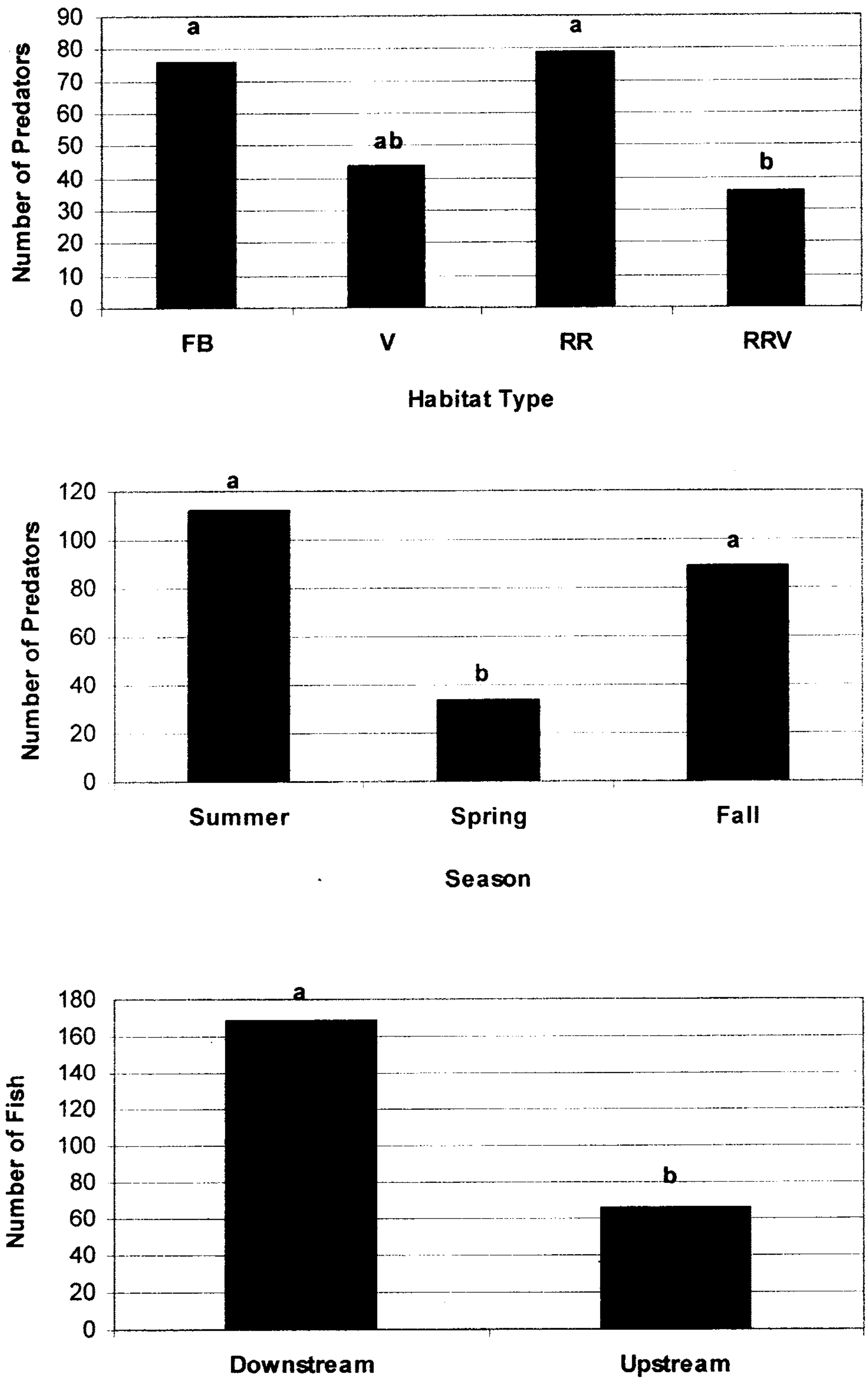


Figure 23. Distribution of piscivores among habitat types (FB, RR, RRV, V) and seasons (summer, spring, fall), and between sections (downstream, upstream). Bars with the same letter are not significantly different.

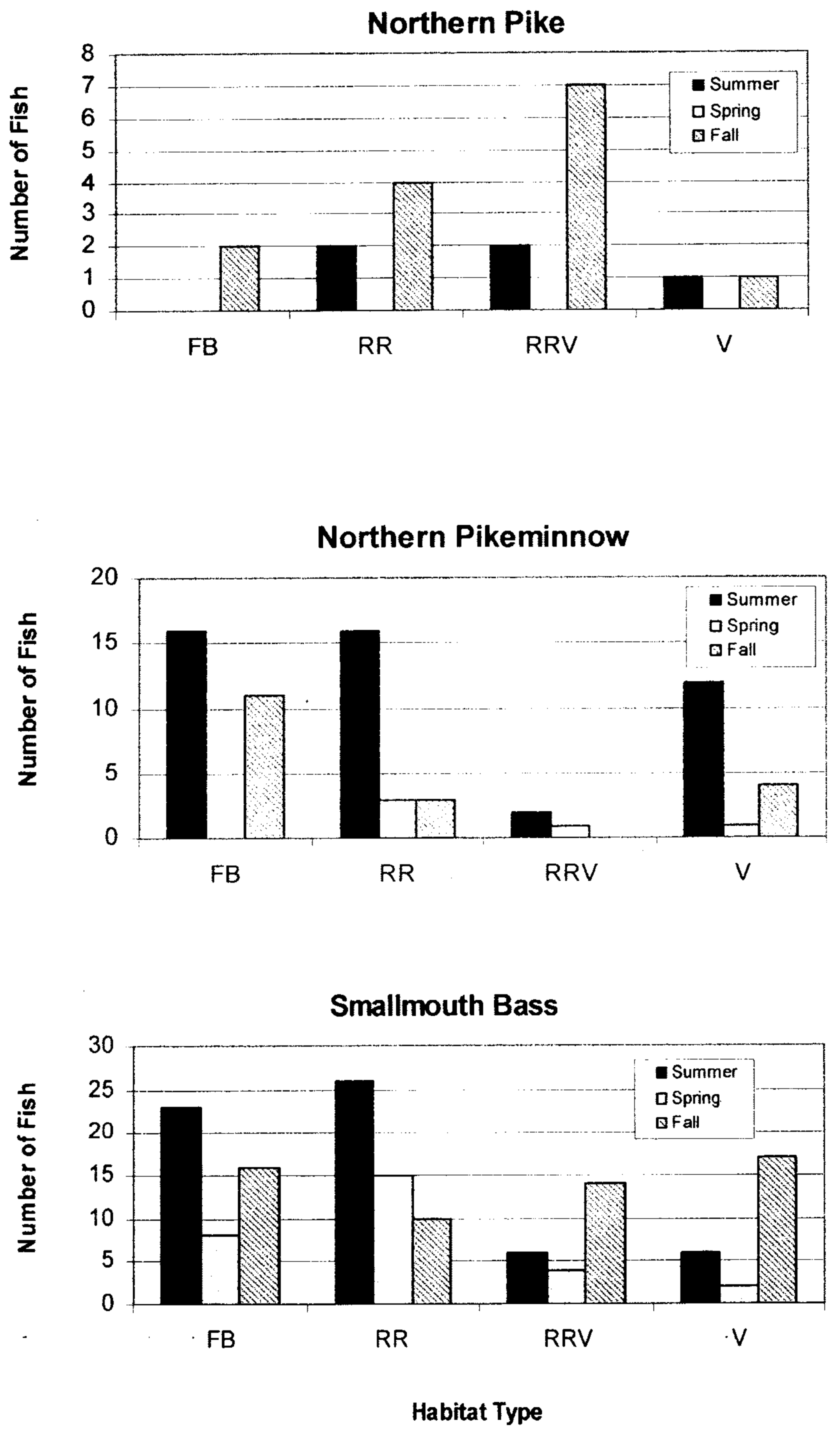


Figure 24. Piscivorous species captured on the lower Coeur d'Alene River at the four shoreline habitat types (FB, RR, RRV, V) by season (summer, spring, fall).

Discussion

Salmonid catch by habitat type, season, and section

The lack of significant differences in salmonid catch among habitat types is contrary to other studies which show avoidance of altered and degraded habitats (Behnke 1992, Binns and Remmick 1994, Harig and Fausch 2002). Juvenile cutthroat trout seemed to show some affinity for riprap sites with 10 of the 12 individuals captured being sampled from RR or RRV sites, whereas adults did not show such an apparent affinity for any shoreline habitat type. On the Upper Yellowstone River, juvenile salmonids were captured in higher numbers at stabilized habitats than natural habitats, which was attributed to the poor condition of natural habitats (Zale and Rider 2003). Larger sample sizes are needed to clarify the relationship between catch and habitat type.

The greater number of salmonids, and particularly cutthroat, captured in spring can be attributed to water temperatures and migration, as corroborated by other findings (USFWS 1999, Parametrix 2005, Dupont IDFG personal communication). As coldwater species, salmonids have lower temperature preferences than most other fishes in the river. Only in spring, late fall, and winter do temperatures drop into the range (7-16°C; USFWS 1999) suitable for coldwater fishes. Stream temperatures generally decrease after July, from a maximum of about 20°C to about 10°C in mid-October, remaining low until May (Parametrix 2005). Thermal conditions and typical spring migratory timing of juvenile salmonids is consistent with the higher catches at this time.

The lack of overall differences in overall salmonid catches between the upstream and downstream sections does not consider differences between adults and juveniles. More adults were captured upstream in proximity to more suitable habitat and spawning grounds. In particular, adult cutthroat trout were more abundant upstream associated with the proximity of more suitable habitat including increased water velocities and lower water temperature in the North Fork (Joe Dupont

IDFG personal communication, Parametrix 2005), which along with its tributaries provide spawning habitats. Adult cutthroat captured during this study likely represented the adfluvial life history and were moving through the inundated reach after spawning. In contrast, more juvenile salmonids were captured downstream, perhaps as a result of ongoing outmigration to Lake Coeur d'Alene. Juvenile cutthroat were captured largely in the downstream section in spring.

Piscivore catch by habitat type, season, and section

The use of habitats on the lower river by piscivorous species can be largely attributed to individual species habitat preferences. The lack of differences in catches of smallmouth bass among the four shoreline habitat types in this study is consistent with a study conducted on a warmwater stream in West Virginia in which both adult and juvenile smallmouth bass were found to be nearly ubiquitous in their distribution among 11 identified habitats (Lobb and Orth 1991). This species is abundant in Lake Coeur d'Alene and many fisherman target the fish in the bays and shallows. The proximity of the lower river to Lake Coeur d'Alene and chain lakes explains their abundance in the lower section. Surprisingly, smallmouth bass did not exhibit any sort of preference for rip-rapped sites. This species is typically associated with areas with rock outcrops that provide cover (Munther 1970, Simpson and Wallace 1978, Rankin 1986, Todd and Rabeni 1989).

The greater numbers of northern pikeminnow captured at FB sites was a result of the characteristics of these habitats, including substrate and depth, which pikeminnow are known to select for (Reid 1971, Simpson and Wallace 1978, Beamesderfer 1983). Bottom substrates at FB sites consisted largely of silt and sand and depths were moderate, favoring the species.

Higher numbers of northern pike at stabilized sites may be a result of prey availability. Northern pike typically inhabit shallow, littoral areas with aquatic vegetation (Cook and Bergersen 1988, Raat 1988, Casselman and Lewis 1996). In this study, stabilized sites had moderate amounts of aquatic vegetation and moderate maximum depths as compared with other sites. This seems to indicate that this species was not purely selecting for aquatic vegetation and may have been in these areas due to a higher abundance of prey. Nearly all northern pike were captured by gillnets which may also indicate that they are making movements between sunset and sunrise, a period not sampled with electrofishing. Activity levels of northern pike have been shown to be greatest during twilight hours (Cook and Bergersen 1988).

Differences in piscivore abundance for both season (summer, spring, and fall) and section (upstream, downstream) observed in this study can be largely attributed to water temperature and spawn timing. All piscivores in this study spawn in spring with rising water temperatures. In spring catches of smallmouth bass and northern pike were lowest. These species may have been seeking warmer temperatures, perhaps in the adjoining chain lakes. Although this study did not focus on water temperatures, on May 31 the temperature in the Thompson Lake channel was 18°C whereas the temperature in the main river was only 10°C. Smallmouth bass, largemouth bass, and northern pike are spawning in the chain lakes in late May and early June (Mark Liter Idaho Department of Fish and Game personal communication). Similarly, northern pike typically migrate to flooded marshes and wetlands, or shallow shorelines with vegetation shortly after ice-out (Casselman and Lewis 1996). Such habitats are common in the chain lakes, which may explain the absence of northern pike from spring catches. Northern pikeminnow spawn from late May to early July over gravelly substrates in shallow water (Simpson and Wallace 1978) when temperatures increase from 9° to 12°C (Reid 1971, Beamesderfer 1983). Such habitats are largely absent from the lower river. The higher catches of northern pikeminnow in the upstream section may thus be associated with the proximity of spawning habitat higher in the basin.

Potential for salmonid overlap with piscivores.

The low catches of salmonids in this study restrict the number of conclusions that can be drawn regarding piscivore impacts on salmonids. Low catch may be a result of several factors, including water temperature, water quality (contaminants; Goldstein et al. 1999), and poor-quality habitat (excessive sedimentation, lack of LWD, lack of overhanging vegetation, lack of pools; USFWS 1999). Cutthroat trout are not abundant in the lower river (Parametrix 2005), and mountain whitefish, Chinook salmon, and kokanee are all more numerous in other areas of the basin. Whitefish are relatively abundant in the North Fork of the Coeur d'Alene River, where cooler water temperatures and better water quality prevail (Maret and MacCoy 2002, Dupont Idaho Department of Fish and Game personal communication). Kokanee salmon occupy Lake Coeur d'Alene, spawning mostly along the lakeshore but to a lesser extent in the Coeur d'Alene River. Chinook salmon, first introduced in 1982 (Labolle 1986), spawn primarily in the North Fork and to a lesser extent the metals-contaminated South Fork (Woodward et al. 1997; Goldstein et al. 1999). All kokanee and Chinook individuals captured were juveniles, evidently moving downstream to Lake Coeur d'Alene. Future efforts at more clearly assessing piscivore impacts on salmonids should involve more directed, intensive sampling of salmonids and piscivores.

Although this study was not specifically designed to evaluate the impacts of piscivores on salmonids, and sample sizes of salmonids were low, some general results can be discerned. First, salmonid catches were highest in spring, consistent with what is known about their life histories and ecological requirements, especially temperature (Bjornn and Reiser 1991). Spring is when the juvenile salmonids would most likely be in the lower river, and is also the period when they would be most apt to be preyed upon there. During spring, piscivore numbers were the lowest among all seasons, however. Smallmouth bass and pikeminnow catches were low and no pike were captured.

Although the exact whereabouts of those species are not known, their most likely location was the chain lakes, where more optimal ecological conditions (including temperature) and spawning occur.

Second, habitat type may affect likelihood of predation. Too few juvenile trout were captured to detect any significant differences in catch by habitat type; however, 10 of 12 juvenile cutthroat trout were caught at stabilized (RR and RRV) sites. Northern pike were captured in higher numbers at stabilized than unstabilized sites and northern pikeminnow were captured in relatively high numbers at RR sites. The potential for overlap in salmonids and piscivores may exist at RR sites, the habitat type where overall piscivore catches were highest. In spring, however, when salmonids catches were highest, the low piscivore catches were not significantly different among habitat types. The overall effect of habitat type on salmonid and piscivore overlap is thus not clear. Based on the data available, season seems more important than habitat in affecting salmonid and piscivore impacts.

Finally, for the ecological interactions between salmonids and piscivores (including predation) to be adequately understood, more extensive quantitative sampling of salmonids and piscovires in the lower river is needed. In addition, several other studies need to be conducted that were outside the scope of this study. First, information is needed on the food habits of the piscivores in the lower Coeur d'Alene River, as well as in the chain lakes. In an earlier study on several chain lakes and Cougar Bay in Lake Coeur d'Alene, Rich (1992) reported that yellow perch were the most numerically abundant food item for pike during spring and fall. No cutthroat were found in stomachs of fish sampled during spring, however, cutthroat made up 15% of stomach contents during fall. A few large cutthroat (mean total length, 215 mm) were found in stomachs of pike captured in Killarney lake, and cutthroat were also present in pike captured from Lake Coeur d'Alene in Cougar Bay. Subsequently in 1996, yellow perch (76%) were the most abundant food item in northern pike stomachs and no cutthroat trout were found (Nelson 1995). More information is needed by species, season, and habitat type, season, section, and fish size in the lower river, chain lakes, and Lake Coeur d'Alene near the river mouth. Extra sampling should be focused in spring, during May and June, when the potential for overlap is greatest.

Information is also needed on the seasonal movements of piscivores in and out of the chain lakes, and the relation of those movements to the movements and upstream and downstream migrations of salmonids. Such studies should span several years to understand differential movements with different annual conditions in the river and lakes. Knowledge of seasonal movements of salmonids and predators and food habits will permit a better assessment of the potential for salmonid and piscivore interactions.

Conclusions

•Salmonid catch (N=81) in the lower river was greatest during spring when water temperatures were low and juveniles were outmigrating to Lake Coeur d'Alene.

•Salmonid catches were not statistically different among habitat types although 10 out of 12 jueveniles were captured at stabilized sites.

•Piscivore catch was lowest during spring. The exact whereabouts of the piscivores at this time were unknown, but many were probably in the chain lakes for more optimal ecological conditions (including temperature) and spawning.

•Overall, piscivores were not captured in significantly different numbers at stabilized and unstabilized habitats. Northern pike were captured in greater numbers at stabilized habitats but no individuals were captured in spring. Similarly, northern pikeminnow were abundant at RR sites but were captured in lower numbers during spring.

•The highest abundance of salmonids in spring, when predators are least abundant, may result in some benefit to migrating salmonids.

•Based on the data available, season seems more important than habitat in affecting salmonid and piscivore impacts. More information is needed by species, season, and habitat type, season, section, and fish size in the lower river, chain lakes, and the Coeur d'Alene Lake near the river mouth. Information is also needed on the seasonal movements of piscivores in and out of the chain lakes, and the relation of those movements to the movements and upstream and downstream migrations of salmonids. Extra sampling should be focused in spring, during May and June, when the potential for overlap is greatest.

Chapter Three
Fish species catch and compositional differences based on sampling gear selectivity

Abstract

Fish sampling was conducted at 24 sites in the lower 54 km of the Coeur d'Alene River in 2005 and 2006 to examine the differences in selectivity between two gear types, gillnetting and electrofishing, in terms of catch composition, length selectivity, and species diversity. Species composition and length selectivity were significantly different between the two gear types. Electrofishing captured a greater numbers of individuals (N=2,915) than gillnetting (N=596), but individuals were much smaller for electrofishing (mean total length, 96 mm) than for gillnetting (mean total length, 331 mm). Gillnets more readily captured longnose suckers *Catostomus catostomus* and largescale suckers *Catostomus macrocheilus* (50% of total catch), whereas electrofishing captured larger numbers of yellow perch *Perca flavescens* and pumpkinseed *Lepomis gibbosus* (54% of total catch). The combined use of these two gears resulted in a greater number of species captured, higher species diversity, and greater range of fish lengths captured than either gear alone could have provided.

Introduction

Sampling gears are selective for different species and sizes of fish. (Hubert 1983). The gear or gears that most representatively sample a fish population is dependent on many factors including the habitat to be sampled, the goals of the study, and aspects of the fish community itself such as species composition and size distribution.

Sampling gears can be classified as passive (gear is not actively moved, relying on entanglement or entrapment) or active (actively removing fish from the water; Hubert 1983). Gillnets (set at specific locations) are a passive gear most applicable in lakes and rivers with low water velocities. Gillnets are typically most effective when allowed to sample overnight to intercept moving fish (Weaver et al. 1993). Because gillnets have an inherent bias according to mesh size, the use of experimental gillnets with graded mesh sizes catch a more representative sample of the total fish community.

Electrofishing is a widely used gear applicable in lakes, rivers, and small streams. Efficiency is directly related to habitat characteristics (temperature, conductivity, substrate, and cover), fish species (habitat preference and size), and conditions (weather, time of day, season) (Hubert 1983). The time of day at which sampling occurs has been shown to impact efficiency. Sampling at night generally captures more species, larger individuals, and more fish than does sampling during the day (Witt and Campbell 1959, Hubert 1983, Paragamian 1989).

The use of several gears often provides a more representative portrayal of abundance and diversity of species than any one gear (Weaver et al. 1993, Goffaux et al. 2005). Multiple gears are often necessary on large rivers because of their size and habitat complexity (Weaver et al. 1993, Argent and Kimmel 2005, Goffaux et al. 2005). On the lower Coeur d'Alene River, I sought to determine differences in fish community abundance, diversity, and composition at stabilized and unstabilized areas. Two sampling gears, gillnetting and electrofishing, were used. The major objective of this portion of the study was to examine the differences in selectivity between the two gear types in terms of catch composition, length selectivity, and species diversity. I also make gear recommendations for future monitoring.

Methods

The lower Coeur d'Alene River was divided into two sections, the upstream section from the Cataldo boat ramp downstream to the Highway 3 bridge and the downstream section from the bridge to the Lake Coeur d'Alene inlet (Figure 1). The two river sections were not of equal lengths because the downstream section was delineated by its direct connection with several of the chain lakes. In both sections, four major shoreline habitat types were identified, two stabilized types (riprap (RR) and riprap with vegetation (RRV)) and two unstabilized types (vegetation (V) and failing banks (FB)). The V habitat types represented an unaltered, relatively stable habitat whereas the FB habitat type was a candidate for future stabilization. For each of the four habitat types, 6 sites were identified, 3 in the upstream section and 3 in the downstream section, for a total of 24 sites. All 24 sites consisted of 150 m of shoreline. In addition to these sites, 3 boat ramps with riprap were sampled. These boat ramps provide access to the river at Rose Lake, Anderson Lake, and Thompson Lake. All three boat ramp sites had riprap both upstream and downstream of the launch but were considered separately because they consisted of far less than 150 m of shoreline. Sampling at all sites was conducted during three seasons, summer, spring, and fall. Summer sampling occurred during late July and early August of 2005. Spring and fall sampling were conducted during May and October of 2006. Fall sampling in the river occurred when Lake Coeur d'Alene water levels were lower, at winter pool (646.5 m), whereas summer and spring sampling in the river occurred when lake water levels were higher, at summer pool (648.6 m). Both lake pool levels and river discharge influenced river elevation and therefore the degree to which bank structures were submerged.

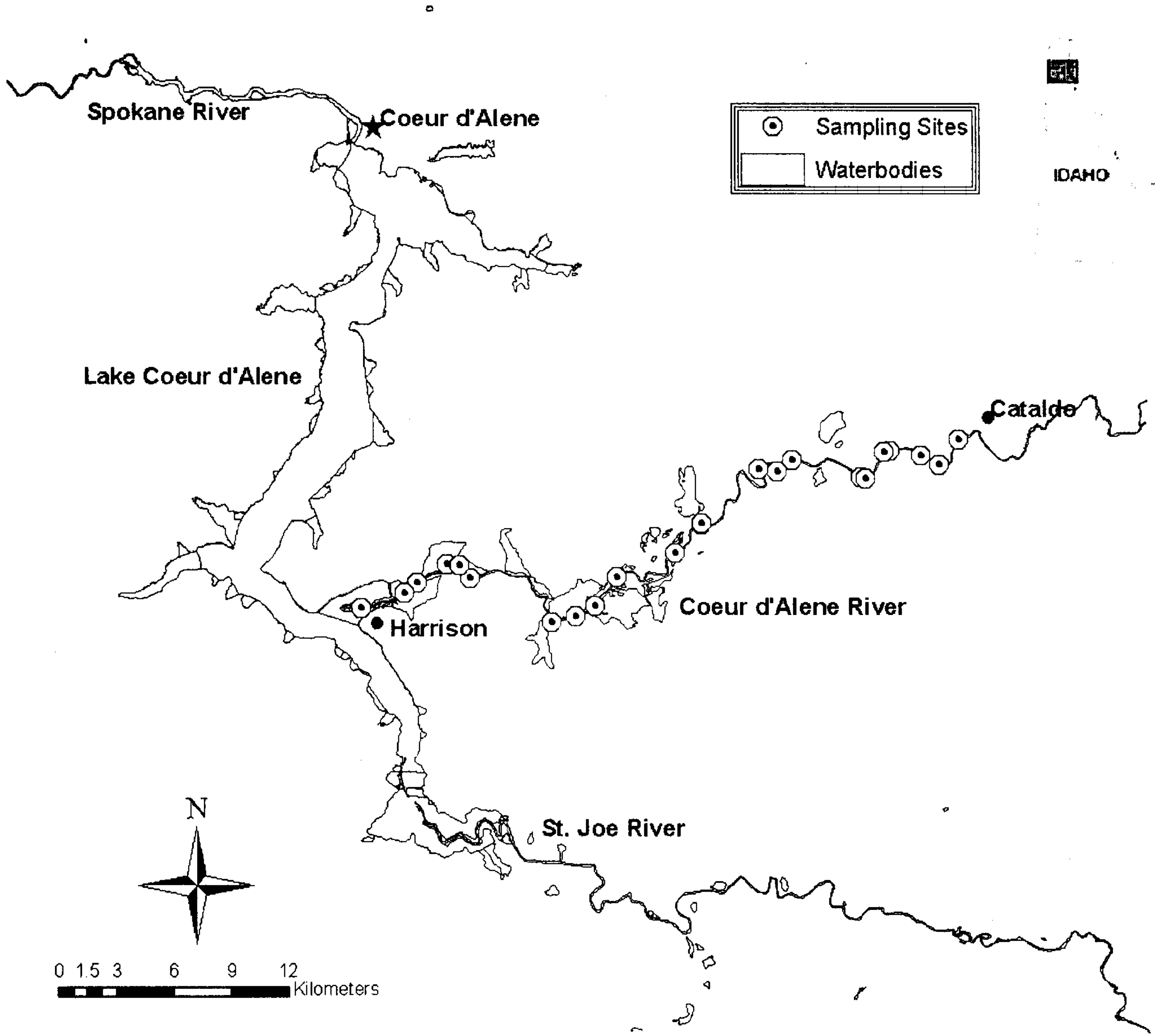


Figure 1. Map showing the Coeur d'Alene Basin, Idaho and the 24 sampling site locations and their division into upstream and downstream sections.

Fish Sampling

Gillnetting and electrofishing were conducted at each of the 24 sites and 3 boat ramps. Experimental (30m x 2m) monofilament gillnets consisted of four panels of varying mesh size (1.9 cm, 2.54 cm, 3.81 cm, 7.62 cm). The nets were set parallel to shore, forming a loose enclosure (Figure 3). Nets were set within one hour of sunset, left to sample overnight, and removed the following morning. Both the time set and time removed were recorded. Relative abundance (catch-per-unit-effort; CPUE) was calculated as fish caught per meter squared per hour (fish/m^2/h) of sample time (Hubert 1983).

Electrofishing equipment consisted of a 6-m boat equipped with a Smith Root electrofishing unit. Pulse-DC current was used in order to minimize negative impacts to fish. Power output was maximized to effectively shock fish without causing harm and was adjusted based on water conductivity and temperature (Reynolds 1983) in the lower river. The 150-m length of shoreline was identified as adequate to assess species richness and percent abundance by ensuring that sufficient numbers of individuals were captured (Reynolds et al. 2003). CPUE was expressed as the number of fish caught per second of shock time (fish/s).

For both sampling gears all captured fish were identified to species, measured for total length and weight, and any abnormalities in body condition were noted. Fish community composition was estimated as a proportion of fish captured by gear. Species diversity based on the Shannon Index (Peet 1975), was expressed as:

$$H' = -\sum_{i=1}^{S} p_i \ln p_i$$

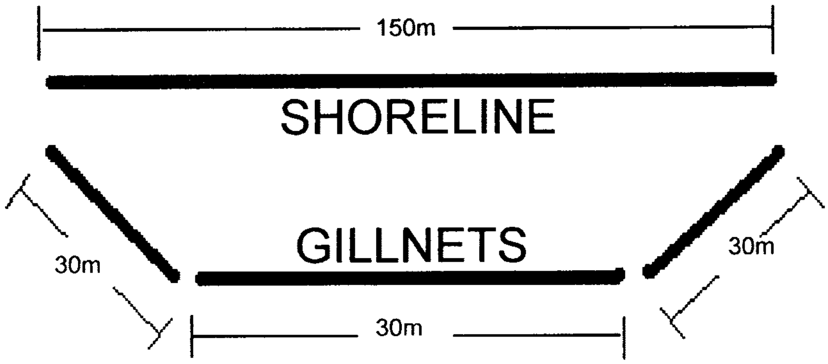


Figure 3. Schematic showing placement of experimental monofilament gillnets with respect to the shorelines for sites on the lower Coeur d'Alene River, Idaho.

where:

n_i is the number of individuals in each species or the abundance of each species,

S is the number of species, $\sum_{i=1}^{S} n_i$

N is the total number of all individuals,

and

p_i is the relative abundance of each species, calculated as the proportion n_i/N of individuals of a given species to the total number of individuals in the community.

Statistical Analysis

Differences in catch composition between sampling gears were evaluated by using the total catch for individual species and applying a Chi-square test of homogeneous proportions to evaluate differences between gears. Species composition differences between gears were further evaluated using a Kruskall-Wallis test. Differences in mean fish total length captured between the two gears were analyzed using a Satterthwaite t-test and Levene's test for equality of variance. All statistical testing was conducted using SAS (SAS Institute 2000).

Results

Electrofishing captured more fish (N=2,915) than gillnetting (N=596) and catch composition was significantly different between sampling gears (χ^2=831.46 P<0.001). The gillnetting catch consisted mostly of largescale *Catostomus macrocheilus* and longnose *Catostomus catosomus* suckers, accounting for just over half of the total catch (Table 5). Yellow perch *Perca flavescens* and northern pikeminnow *Ptychocheilus* *oregonensis* constituted 13% and 11%, respectively, to the catch and all other species together constituted less than 10% of the catch. In contrast, the electrofishing catch consisted mostly of yellow perch and pumpkinseed *Lepomis gibbosus*, accounting for 54% of the total catch. Largemouth bass *Micropterus salmoides* and brown bullhead *Ameirus nebulosus* (13 % and 11%, respectively) were also caught frequently, and all other species together constituted less than 10% of the catch.

Nearly all species were captured in significantly higher numbers with one gear or the other (Table 6, Figure 25). Only smallmouth bass *Micropterus dolomieui* and westslope cutthroat trout *Oncorhynchus lewisi clarki* catches did not differ significantly between gears. For gillnetting, catch of longnose suckers, largescale suckers, northern pike *Esox lucius*, northern pikeminnow, and tench *Tinca tinca* was significantly higher than for electrofishing. For electrofishing, the catch of brown bullhead, bluegill *Lepomis macrochirus*, largemouth bass, pumpkinseed, and yellow perch was significantly higher. The remaining species were captured in numbers too small to evaluate.

The mean length of fish captured differed significantly between the two gears (Table 7; $t=37.86$, $P<0.001$). Gillnetting captured larger fish with a mean total length of 331 mm, whereas electrofishing captured fish with a mean total length of 96 mm. In addition, results of the Levene's test for homogeneity of variance showed that the range of lengths captured during gillnetting was greater (range, 57-1337 mm) than for electrofishing (range, 26-526 mm; $F=163.95$, $P<0.001$)

Diversity (based on Shannon Index scores) was low for both gears. For the overall index, the Hmax=2.83, which would indicate complete evenness of species

Table 5. Proportion of individual species captured by gear (gillnetting and electrofishing) in the lower Coeur d'Alene River, Idaho.

Species	Gillnetting	Electrofishing
Ameirus nebulosus	4.9	10.9
Pomoxis nigromaculatus	0.7	1.1
Salvelinus fontinalis	0	0.03
Lepomis macrochirus	0	7.9
Onchorhynchus tshawytscha	0	0.7
Onchorhynchus nerka	0	0.9
Micropterus salmoides	0.3	12.3
Catostomus catostomus	19.1	2.3
Catostomus macrocheilus	34.1	1.7
Prosopium williamsoni	0.2	0.2
Esox lucius	3.2	0.1
Ptychocheilus oregonensis	11.1	0.6
Lepomis gibbosus	1.0	16.5
Salmo gairdneri	0.2	0
Micropterus dolomieui	5.4	5.6
Tinca tinca	5.1	0.4
Salmo clarki lewisi	2.2	0.4
Perca flavescens	12.7	38.3
	100.00	**100.00**

Table 6. Results of Kruskal-Wallis comparison of individual species catch between gears (gillnetting and electrofishing). Statistical results shown are chi-squared and *P*-values, significant at $\alpha=0.10$.

Species	$\chi 2$	*P*-value
Catostomus catostomus[1]	12.513	*<0.001*
Catostomus macrocheilus[1]	25.787	*<0.001*
Esox lucius[1]	6.917	*0.009*
Ictalurus nebulosus[2]	45.580	*<0.001*
Lepomis gibbosus[2]	68.989	*<0.001*
Lepomis macrochirus[2]	31.111	*<0.001*
Micropterus dolomieui[2]	0.779	0.377
Micropterus salmoides[2]	59.546	*<0.001*
Perca flavescens[2]	36.248	*<0.001*
Ptychocheilus oregonensis[1]	28.642	*<0.001*
Tinca tinca[1]	5.788	*0.016*

[1] Species catch greater for gillnetting
[2] Species catch greater for electrofishing

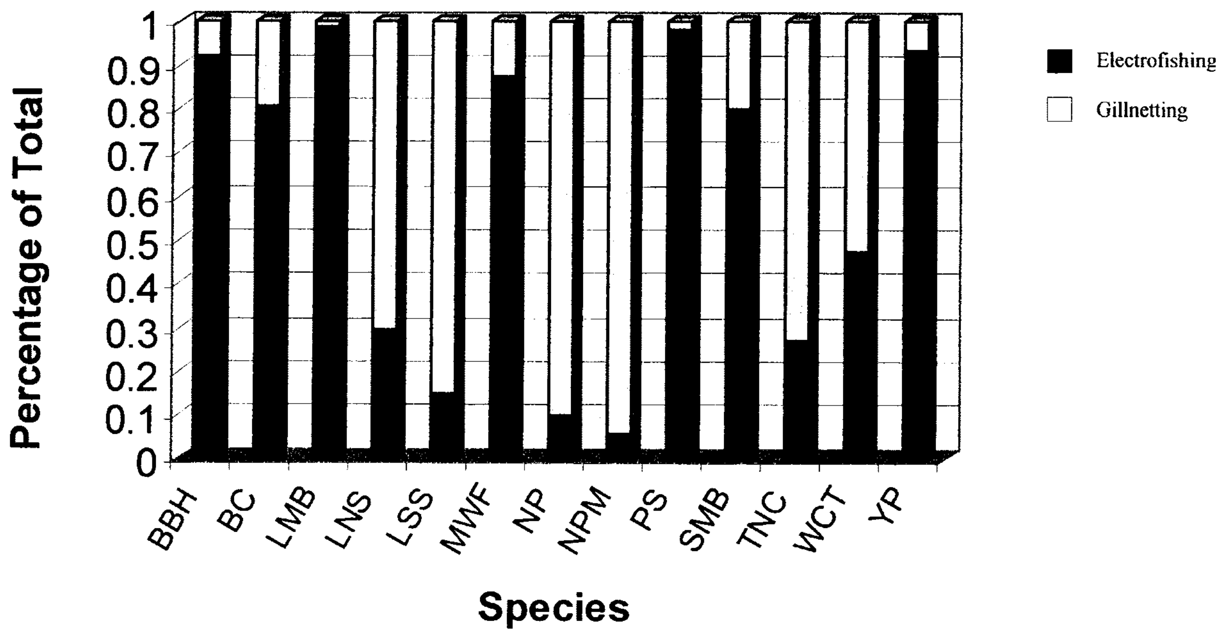


Figure 25. Individual species as a percentage of total catch by gear (gillnetting and electrofishing) on the lower Coeur d'Alene River, Idaho. BBH=brown bullhead, BC=black crappie, LMB=largemouth bass, LNS=longnose sucker, LSS=largescale sucker, MWF=mountain whitefish, NP=northern pike, NPM=northern pikeminnow, PS=pumpkinseed, SMB=smallmouth bass, TNC=tench, WCT=westslope cutthroat trout, YP=yellow perch.

numbers. Gillnetting captured 16 species throughout the three seasons of sampling (H'=1.98) whereas electrofishing captured 17 species (H'=1.91).

The combined use of two gears resulted in an increased number of species captured (N=18), a greater diversity of species captured (H'= 2.01), and a greater range of fish lengths captured (26-1337 mm) than either gear alone.

Discussion

The two gear types used in this study differed both in species composition captured and mean length of individuals captured, which is consistent with the findings of other studies (Weaver et al. 1993, Goffaux et al. 2005). The significant differences in species composition are directly related to differences in mean length of fish captured by the two gears: electrofishing sampled generally smaller individuals such as bluegill and pumpkinseed, while gillnetting captured larger individuals such as pikeminnow and suckers. The use of experimental gillnets with graded mesh sizes resulted in the capture of a large range of fish lengths.

The lack of difference in diversity between gears and low overall diversity resulted from an uneven distribution of individual species numbers for both gears. Disproportionately high numbers of longnose and largescale suckers resulted in low diversity for gillnetting and high numbers of yellow perch resulted in low diversity for electrofishing.

An important consideration for future monitoring pertains to the time of day. Though electrofishing during this study was conducted during the day, studies have found that night electrofishing is more effective during the night as fish tend to be more active at this time (Paragamian 1989). To offset this bias, gillnets were set overnight;

however, the effectiveness of night versus day electrofishing should be evaluated on the lower Coeur d'Alene River.

Conclusions

•Electrofishing (N=2,915) captured nearly five times as many fish as gillnetting (N=596).

•Gillnetting and electrofishing were significantly different in terms of both species composition and length selectivity. Gillnets more readily captured larger individuals including largescale and longnose suckers and northern pikeminnow, and electrofishing more readily captured smaller individuals including bluegill, pumpkinseed, and yellow perch. Gillnetting also captured a significantly greater range of fish mean lengths.

•Diversity of species captured did not differ between gear types.

• The combined use of these two gears resulted in a greater number of species captured, higher species diversity, and greater range of fish lengths captured than either gear alone could have provided.

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Appendix A.

Scientific Name	Common Name	Family	Abbreviation	Trophic Guild
Catostomus catostomus	Longnose Sucker	Catostomidae	LNS	
Catostomus macrocheilus	Largescale Sucker	Catostomidae	LSS	
Esox lucius	Northern Pike	Esocidae	NP	P
Ameirus nebulosus	Brown Bullhead	Ictaluridae	BBH	I
Lepomis gibbosus	Pumpkinseed	Centrarchidae	PS	I
Lepomis macrochirus	Bluegill	Centrarchidae	BLG	I
Micropterus dolomieui	Smallmouth Bass	Centrarchidae	SMB	I/P
Micropterus salmoides	Largemouth Bass	Centrarchidae	LMB	I/P
Oncorhynchus nerka	Kokanee	Salmonidae	KOK	
Oncorhynchus tshawytscha	Chinook salmon	Salmonidae	CHN	
Perca flavescens	Yellow Perch	Percidae	YP	I/P
Pomoxis nigromaculatus	Black Crappie	Centrarchidae	BC	I/P
Prosopium williamsoni	Mountain Whitefish	Salmonidae	MWF	
Ptychocheilus oregonensis	Northern Pikeminnow	Cyprinidae	NPM	
Oncoryhynchus clarki lewisi	Westslope Cutthroat Trout	Salmonidae	WCT	
Salmo gairdneri	Rainbow Trout	Salmonidae	RBT	
Salvelinus fontinalis	Brook Trout	Salmonidae	BKT	
Tinca tinca	Tench	Cyprinidae	TNC	

I=Insectivores P=Piscivores H=Herbivores O=Omnivores

Appendix B.

HABITAT ASSESSMENT LOWER COEUR D'ALENE RIVER

Site: Date: GPS Coordinates: River Side:

	DEPTHS		WIDTHS	% OVERHANG	DOMINANT	VEGETATION SPECIES			% SUB AO	LWD	RIPRAP	RIPRAP
	5 FEET	10 FEET		VEG	VEG	TREES	SHRUBS	GRASSES	VEG		SIZE	DEPTH
1												
2												
3												
4												
5												

ASPECT	BANK	HABITAT TYPES			Max	COMMENTS
	SLOPE	OPPOSITE	UPSTREAM	DOWNSTREAM	Depth	

HABITAT PARAMETERS COEUR D'ALENE RIVER

PARAMETER	CONDITION OPTIMAL	SUBOPTIMAL	MARGINAL	POOR
Substrate/ Available Cover	Greater than 50% of substrate favorable for epifaunal colonization and fish cover, mix of snags, submerged logs, undercut banks, cobble or other stable habitat and at stage to allow full colonization potential (i.e. logs/snags that are not new fall and not transient).	30-50% mix of stable habitat; well-suited for full colonization potential; adequate habitat for maintenance of populations; presence of additional substrate in the form of newfall, but not yet prepared for colonization.	10-30% mix of stable habitat; habitat availability less than desirable; substrate frequently disturbed or removed.	Less than 10% stable habitat; lack of habitat is obvious; substrate unstable or lacking.
Pool Substrate	Mixture of substrate materials, with gravel and firm sand prevalent; root mats and submerged vegetation common.	Mixture of soft sand, mud, or clay; mud may be dominant; some root mats and submerged vegetation present.	All mud or clay or sand bottom; little or no root mat; no submerged vegetation.	Hard-pan clay or bedrock; no root mat or vegetation.
Pool Variability	Even mix of large-shallow, large-deep, small-shallow, small-deep pools present.	Majority of pools large-deep; very few shallow.	Shallow pools much more prevalent than deep pools.	Majority of pools small-shallow or pools absent.
Sediment Deposition	Little or no enlargement of islands or point bars and less than 20% of the bottom affected by sediment deposition.	Some new increase in bar formation, mostly from gravel, sand or fine sediment; 20-50% of the bottom affected; slight deposition in pools.	new gravel, sand or fine sediment on old and new bars; 50-80% of the bottom affected; sediment deposits at obstructions, constrictions, and bends; moderate deposition of pools prevalent.	Heavy deposits of fine material, increased bar development; more than 80% of the bottom changing frequently; pools almost absent due to substantial sediment deposition.
Channel Flow	Water reaches base of both lower banks, and minimal amount of channel substrate is exposed.	Water fills >75% of the available channel; or <25% of channel substrate exposed.	Water fills 25-75% of the available channel, and/or riffle substrates are mostly exposed.	Very little water in channel and mostly present as standing pools.
Channel Alteration	Channelization or dredging absent or minimal; stream with normal pattern.	present, usually in areas of bridge abutments; evidence of past channelization, i.e., dredging, (greater than past 20 yrs) may be present, but recent channelization is not present.	Channelization may be extensive; embankments or shoring structures present on both banks; and 40-80% of stream reach channelized and disrupted.	Banks shored with gabion or cement; over 80% of the stream reach channelized and disrupted. Instream habitat greatly altered or removed entirely.
Channel Sinuosity	The bends in the stream increase the stream length 3 to 4 times longer than if it was in a straight line.	The bends in the stream increase the stream length 1 to 2 times longer than if it was in a straight line.	The bends in the stream increase the stream length 1 to 2 times longer than if it was in a straight line.	Channel straight; waterway has been channelized for a long distance.
Bank Stability	LEFT: RIGHT: Banks stable; evidence of erosion or bank failure absent or minimal; little potential for future problems. <5% of bank affected.	LEFT: RIGHT: Moderately stable; infrequent, small areas of erosion mostly healed over. 5-30% of bank in reach has areas of erosion.	LEFT: RIGHT: Moderately unstable; 30-60% of bank in reach has areas of erosion; high erosion potential during floods.	LEFT: RIGHT: Unstable; many eroded areas; "raw" areas frequent along straight sections and bends; obvious bank sloughing; 60-100% of bank has erosional scars.
Vegetative Protection	LEFT: RIGHT: More than 90% of the streambank surfaces and immediate riparian zone covered by native vegetation, including trees, understory	LEFT: RIGHT: 70-90% of the streambank surfaces covered by native vegetation, but one class of plants is not well-represented; disruption	LEFT: RIGHT: 50-70% of the streambank surfaces covered by vegetation; disruption obvious; patches of bare soil or closely cropped	LEFT: RIGHT: Less than 50% of the streambank surfaces covered by vegetation; disruption of streambank vegetation is very high; vegetation has been
Riparian Zone Width	LEFT: RIGHT: Width of riparian zone >18 meters; human activities (i.e., parking lots, roadbeds, clear-cuts, lawns, or crops) have not impacted zone.	LEFT: RIGHT: Width of riparian zone 12-18 meters; human activities have impacted zone only minimally.	LEFT: RIGHT: Width of riparian zone 6-12 meters; human activities have impacted zone a great deal.	LEFT: RIGHT: Width of riparian zone <6 meters; little or no riparian vegetation due to human activities.

Appendix C.

ANOVA Gillnetting CPUE by Habitat Type, Section, Season, and Habitat Type x Season

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	12	0.02493049	0.00207754	5.88	<.0001
Error	59	0.02083383	0.00035312		
Corrected Total	71	0.04576432			

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
habitattype	3	0.00304577	0.00101526	2.88	0.0436
section	1	0.00163108	0.00163108	4.62	0.0357
season	2	0.01693042	0.00846521	23.97	<.0001
habitattype*season	6	0.00332322	0.00055387	1.57	0.1724

ANOVA Electrofishing CPUE by Habitat Type, Section, Season, and Habitat Type x Season

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	12	0.14249421	0.01187452	5.58	<.0001
Error	59	0.12557762	0.00212843		
Corrected Total	71	0.26807183			

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
habitattype	3	0.02947910	0.00982637	4.62	0.0057
section	1	0.05776087	0.05776087	27.14	<.0001
season	2	0.04799072	0.02399536	11.27	<.0001
habitattype*season	6	0.00726352	0.00121059	0.57	0.7535

ANOVA Gillnetting CPUE by Stabilized/Unstabilized, Section, Season, and Stabilized/Unstabilized x Season

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	6	0.01930448	0.00321741	7.90	<.0001
Error	65	0.02645984	0.00040707		
Corrected Total	71	0.04576432			

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
stab/unstab	1	0.00059079	0.00059079	1.45	0.2327
section	1	0.00163108	0.00163108	4.01	0.0495
season	2	0.01693042	0.00846521	20.80	<.0001
stab/unstab*season	2	0.00015219	0.00007609	0.19	0.8299

ANOVA Electrofishing CPUE by Stabilized/Unstabilized, Section, Season, and Stabilized/Unstabilized x Season

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	6	0.07535458	0.01255910	4.24	0.0012
Error	65	0.19271725	0.00296488		
Corrected Total	71	0.26807183			

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
stab/unstab	1	0.01708441	0.01708441	5.76	0.0192
section	1	0.00632619	0.00632619	2.13	0.1489
season	2	0.04799072	0.02399536	8.09	0.0007
stab/unstab*season	2	0.00395326	0.00197663	0.67	0.5169

Ranked ANOVA Summer Gillnetting CPUE by Habitat Type and Section

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	4	407.166667	101.791667	2.60	0.0687
Error	19	742.833333	39.096491		
Corrected Total	23	1150.000000			

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
habitattype	3	225.6666667	75.2222222	1.92	0.1600
section	1	181.5000000	181.5000000	4.64	0.0442

Ranked ANOVA Summer Electrofishing CPUE by Habitat Type and Section

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	4	524.333333	131.083333	3.98	0.0164
Error	19	625.666667	32.929825		
Corrected Total	23	1150.000000			

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
habitattype	3	140.3333333	46.7777778	1.42	0.2678
section	1	384.0000000	384.0000000	11.66	0.0029

Ranked ANOVA Spring Gillnetting CPUE by Habitat Type and Section

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	4	18.500000	4.625000	0.08	0.9880
Error	19	1120.500000	58.973684		
Corrected Total	23	1139.000000			

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
habitattype	3	10.33333333	3.44444444	0.06	0.9809
section	1	8.16666667	8.16666667	0.14	0.7139

Ranked ANOVA Spring Electrofishing CPUE by Habitat Type and Section

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	4	364.333333	91.083333	2.20	0.1075
Error	19	785.666667	41.350877		
Corrected Total	23	1150.000000			

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
habitattype	3	283.6666667	94.5555556	2.29	0.1114
section	1	80.6666667	80.6666667	1.95	0.1786

Ranked ANOVA Fall Gillnetting CPUE by Habitat Type and Section

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	4	437.000000	109.250000	2.91	0.0491
Error	19	713.000000	37.526316		
Corrected Total	23	1150.000000			

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
habitattype	3	370.3333333	123.4444444	3.29	0.0431
section	1	66.6666667	66.6666667	1.78	0.1983

Ranked ANOVA Fall Electrofishing CPUE by Habitat Type and Section

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	4	0.00781447	0.00195362	8.31	0.0005
Error	19	0.00446837	0.00023518		
Corrected Total	23	0.01228284			

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
habitattype	3	0.00176835	0.00058945	2.51	0.0899
section	1	0.00604613	0.00604613	25.71	<.0001