

FISH COMMUNITY STRUCTURE ASSOCIATED WITH STABILIZED AND UNSTABILIZED SHORELINE HABITATS, COEUR d'ALENE RIVER, IDAHO, USA

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ABSTRACT

Stabilization of riverbanks and lake shorelines has become widely used to reduce erosion. Studies on effects of stabilization on fish species and communities have indicated highly variable effects from beneficial to detrimental. Bank stabilization implemented to reduce shoreline erosion of metals-contaminated sediments in the lower Coeur d'Alene River, Idaho, prompted concern of possible effects on the fish community. Fish sampling (electrofishing and gillnetting) and habitat assessment were conducted at 24 sites in the lower 54 km of river during 2005 and 2006 to assess differences in the fish community (relative abundance, species diversity and community composition) at stabilized and unstabilized shorelines. Within stabilized and unstabilized shorelines we evaluated seasonal differences as well as upstream and downstream differences. We captured 3511 fish from 17 species and 7 families; 83% of fish were captured by electrofishing. Fish relative abundance was significantly higher at stabilized than unstabilized sites for electrofishing. We also found positive correlations between relative abundance and diameter of rock at stabilized sites for both sampling gears. Three species (brown bullhead *Ameiurus nebulosus*, northern pike *Esox lucius* and pumpkinseed *Lepomis gibbosus*) were captured more readily at stabilized shoreline sites. Seasonally, the differences in relative abundance among habitat types indicate that stabilized structures are providing stable habitat year-round. 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 low fraction (2.5%) of bank stabilization. Based on these results and reviews of other studies, we suggest that two factors affecting the results of bank stabilization studies are (i) the habitat quality of the unstabilized river, and (ii) the percentage of the river that has been stabilized. Copyright © 2010 John Wiley & Sons, Ltd.

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INTRODUCTION

Riverbank stabilization is widely implemented to prevent or slow lateral channel erosion, to direct and define the location of the channel, and thereby protect land and improvements such as agricultural activity, roads and bridges (Simons, 1995; Schmetterling *et al.*, 2001). Numerous stabilization approaches exist. 'Hard' revetments include riprap (large angular rock; the most commonly used revetment), stone spurs, dikes and concrete walls (Gore and Shields, 1995; Simons, 1995; Schmetterling *et al.*, 2001). 'Soft' applications include natural materials such as willow plantings, large woody debris (LWD), trees and rootwads (Shields *et al.*, 1995, 2000).

At a large scale, riverbank stabilization and the resulting channelization (channel straightening) can alter the hydrology and function of a river (Hesse *et al.*, 1989; Ward and Stanford, 1989). Extensive riprap can result in

reduced connectivity and reduced movement of suspended sediment, nutrients and aquatic life between the main channel and other habitats such as side channels and backwaters. As stabilization becomes increasingly pervasive, a common result at a more local habitat scale is a channel with a more uniform gradient, loss of natural riffles and pools (Keller, 1975) and loss of natural instream structure such as rootwads and LWD (Leopold *et al.*, 1964; Bryant, 1983; Angermeier and Karr, 1984). Changes in chemical factors such as suspended solids and dissolved substances can also occur (Yorke, 1978).

Studies on the effects of bank stabilization on fish communities have reported a wide range of often conflicting results. The differences in results reflect different study objectives, different experimental designs, different sampling scales and different aquatic environments. Impaired river function and reduced habitat diversity can result in reduced diversity and abundance of fish and other aquatic species (Scarnecchia, 1988; Ward and Stanford, 1989). Several studies show that fish exhibit a significant preference for unstabilized over stabilized areas of lakes and rivers, especially areas with riprap (Elser, 1968; Knudsen and Dilley, 1987; Garland *et al.*, 2002). Conversely, bank stabilization can also benefit certain species or life stages in

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some instances (Chapman and Knudsen, 1980; Trial *et al.*, 2001; Zale and Rider, 2003). Other studies have reported minimal impacts of localized stream channelization and bank stabilization (King and Carlander, 1976). The size of rock used in riprap structures is also important; large rocks in riprap can support higher numbers of juvenile fish than smaller rocks and cobbles (Lister *et al.*, 1995).

In the Coeur d'Alene River basin, Idaho, USA, water quality has been negatively impacted by the presence of metals, including zinc, copper, cadmium, lead and arsenic, from mine tailings entering the river system (Mink, 1971; Funk *et al.*, 1975; Maret and MacCoy, 2002). Bank stabilization has been used to reduce erosion and thereby the amount of metals-contaminated soil entering the river. A 2004 inventory identified 24 bank stabilization projects on the lower river covering an estimated 2.8% of the total riverbank (KSSWCD, 2004). An estimated 90% of stabilized shoreline consisted of riprap, while the remaining stabilized shoreline (10%) consisted of vegetation (6%) and other stabilization types. Future plans call for more stabilization on the lower river.

In this paper we examine differences in the fish community structure between stabilized and unstabilized banks. We assess seasonal changes as well as upstream and

downstream differences in fish community usage in reaches with stabilized and unstabilized banks. The objectives of this study were to: (i) assess differences in relative fish abundance (catch-per-unit effort; CPUE), species diversity and community composition associated with stabilized banks (riprap (RR) and riprap with vegetation (RRV)) and unstabilized banks (failing banks (FB) and vegetation (V)) by section (upstream and downstream) and season (summer, spring, fall); (ii) assess the relationships between relative fish abundance (CPUE) and local scale habitat variables along stabilized and unstabilized banks and (iii) assess the relationship between relative fish abundance (CPUE) and (a) depth of riprap structure and (b) riprap rock diameter.

STUDY AREA

The Coeur d'Alene River basin (area, 10 360 km²) drains the west-slope of the Bitterroot Range between Montana and Idaho (Funk *et al.*, 1975). The lower river lies downstream of the confluence of the North and South Forks of the Coeur d'Alene River, and drains into Lake Coeur d'Alene (Figure 1). The North Fork is relatively free of mining activity, in sharp contrast to the heavily-impacted South

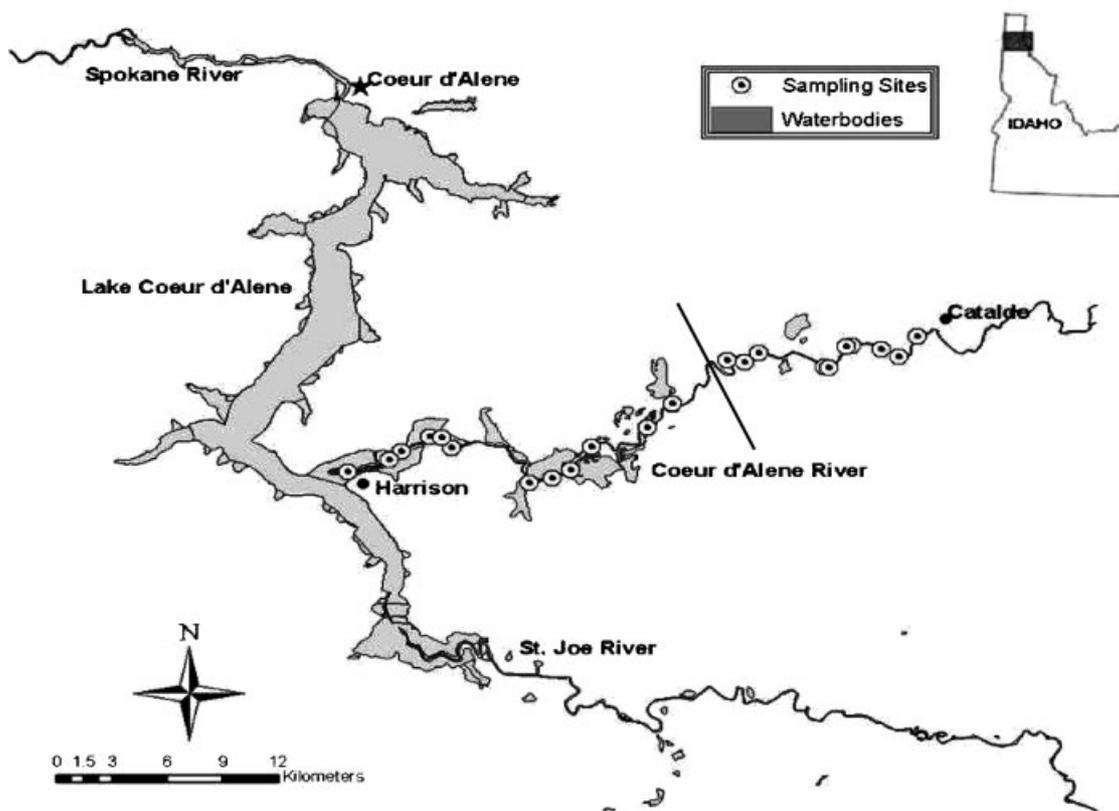


Figure 1. Coeur d'Alene Basin, Idaho and the 24 sampling site locations and their division into upstream and downstream sections.

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 (<9 m; Sprenke *et al.*, 2000), vegetated chain lakes by natural streams and dredged channels (Bowles, 1985; Rieman, 1987). The lower river downstream of the Cataldo boat ramp (a distance of 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 above sea level (Avista, 2005).

The basin contains a variety of coldwater, coolwater and warmwater species both native and non-native, representing the families Salmonidae, Cottidae, Cyprinidae, Catostomidae, Centrarchidae and Ictaluridae (Laumeyer, 1976; Maret and MacCoy, 2002). Native coldwater species include westslope cutthroat trout *Oncorhynchus clarki*, bull trout *Salvelinus confluentus*, northern pikeminnow *Ptychocheilus oregonensis* and mountain whitefish *Prosopium williamsoni*. Non-native coldwater species, which are present in both the river and lake systems, include rainbow trout *Oncorhynchus mykiss*, brook trout *Salvelinus fontinalis*, kokanee salmon *Oncorhynchus nerka* and Chinook salmon *Oncorhynchus tshawytscha*. Non-native coolwater species include yellow perch *Perca flavescens*, smallmouth bass *Micropterus dolomieu* and northern pike *Esox lucius*. Non-native warmwater species include largemouth bass *Micropterus salmoides*, black crappie *Pomoxis nigromaculatus*, bluegill *Lepomis macrochirus*, pumpkinseed *Lepomis gibbosus*, brown bullhead *Ameiurus nebulosus* and tench *Tinca tinca*. The coolwater and warmwater 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 found in the lower river during periods of low water temperatures.

METHODS

Sampling design

The lower Coeur d'Alene River was divided into two sections: (i) the upstream section from the Cataldo boat ramp downstream to the Highway 3 Bridge and (ii) the downstream section from the bridge to the Lake Coeur d'Alene inlet (Figure 1). 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)). We randomly selected six sites from each of the four habitat types, three in the upstream section and three in the

downstream section, or 24 total sites. Each site consisted of 150 m of shoreline.

Sampling at all sites was conducted during summer (July–August 2005), spring (May 2006) and fall (October 2006). Fall sampling in the river occurred when Lake Coeur d'Alene water levels were lower (646.5 m), whereas summer and spring sampling in the river occurred when lake water levels were higher (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

The two stabilized (RR, RRV) and two unstabilized (V, FB) shoreline habitat types were characterized according to a series of ranked and quantifiable habitat variables. 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 characteristics, other habitat characteristics recorded were depths at 1.5 and 3-m from the shoreline, river width, per cent overhanging cover, dominant vegetation, per cent 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 ($\text{m}^3 \text{s}^{-1}$) were retrieved from the USGS gauging station at Cataldo (Gauge number 12413500).

Fish sampling

Sampling was conducted at each of the 24 sites using gillnetting and electrofishing. 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).

Experimental (30 m × 2 m) monofilament gillnets consisted of four panels of varying mesh size (1.90, 2.54, 3.81 and 7.62 cm). The nets were deployed in sets of 3, with the middle net parallel to the shore and the upper and lower nets at about 30° to shore, forming a loose enclosure (Gidley, 2008). Nets were set within 1 h of sunset, left to sample overnight, and removed the following morning. Both the time set and time removed were recorded. Relative abundance (CPUE) was calculated as fish caught per square meter per hour ($\text{fish m}^{-2} \text{h}^{-1}$) 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 per cent abundance by ensuring that sufficient numbers of individuals were captured (Reynolds *et al.*, 2003). Sampling occurred within 10 m of the shoreline. CPUE was expressed as the number of fish caught per second of shock time (fish s^{-1}).

For both sampling gears all captured fish were identified to species and measured for total length (mm) and weight (g). We evaluated metrics including relative fish abundance, species diversity and community composition in relation to habitat type, season and section. 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, 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

To test for differences in habitat characteristics among habitat types, we compared 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. 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 (AIC) was conducted to explain variance. Habitat variables section, per cent aquatic

vegetation, per cent overhanging vegetation, 1.5 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 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 software (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 metrics

In all, 3511 fish were captured, representing 17 species and 7 families. Gillnetting consisted of 1270 h of set time and resulted in the capture of 596 fish. Electrofishing effort consisted of 34 h and resulted in the capture of 2915 fish. In summer, 1402 fish were captured, 300 by gillnetting and 1107 by electrofishing. In spring, 703 fish were captured, 83 by gillnetting and 620 by electrofishing. In fall 1407 fish were captured, 213 by gillnetting and 1194 by electrofishing.

Relative fish abundance. Relative fish abundance (CPUE) was not significantly different between stabilized and unstabilized sites for gillnetting ($F = 1.95$, $p = 0.167$), but was significantly higher at stabilized than unstabilized sites for electrofishing ($F = 5.66$, $p = 0.020$). Both gillnetting and electrofishing CPUE were significantly different by habitat type, season and section, but not by the habitat by season interaction (Figures 2–4).

For gillnetting, CPUE was significantly higher at FB ($0.0044 \text{ fish m}^{-2} \text{ h}^{-1}$) sites than at RR ($0.0023 \text{ fish m}^{-2} \text{ h}^{-1}$), V ($0.0021 \text{ fish m}^{-2} \text{ h}^{-1}$) and RRV ($0.0026 \text{ fish m}^{-2} \text{ h}^{-1}$) 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 ($0.0270 \text{ fish s}^{-1}$) and RRV ($0.0223 \text{ fish s}^{-1}$) sites than at FB ($0.0128 \text{ fish s}^{-1}$) sites, but was not significantly higher than V ($0.0209 \text{ fish s}^{-1}$) sites ($F = 5.30$, $p = 0.003$). CPUE was significantly higher in fall and summer than during spring ($F = 12.94$, $p < 0.001$) and higher downstream than upstream ($F = 31.14$, $p < 0.001$).

Summer CPUE among habitat types did not differ significantly for gillnetting ($F = 1.63$, $p = 0.215$) or electrofishing ($F = 1.11$, $p = 0.376$), but was significantly different

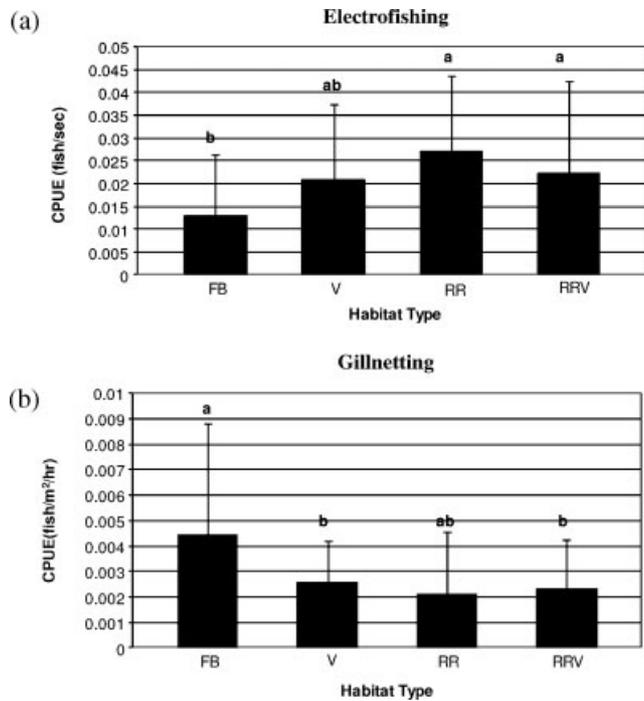


Figure 2. Overall electrofishing and gillnetting relative fish abundance (CPUE) by habitat type (failing bank (FB), vegetation (V), riprap (RR), riprap with vegetation (RRV)). Columns with the same letter are not significantly different ($\alpha = 0.10$). Error bars show standard deviation.

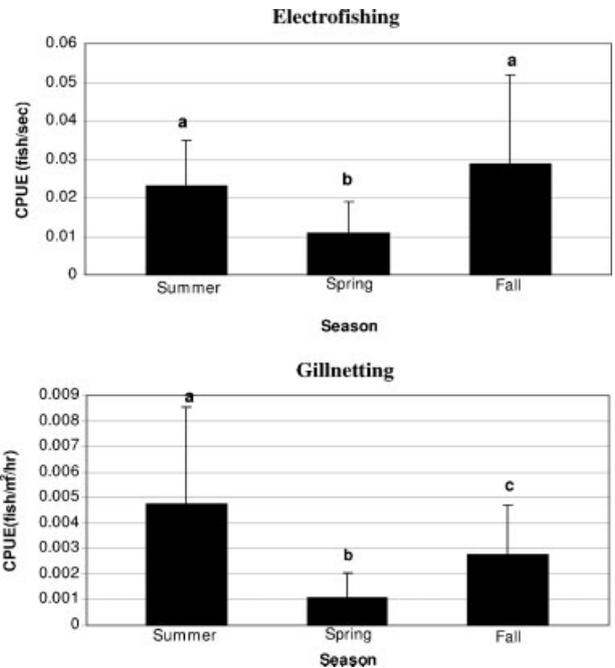


Figure 4. Overall electrofishing and gillnetting relative fish abundance (CPUE) by season. Columns with the same letter are not significantly different ($\alpha = 0.10$). Error bars show standard deviation.

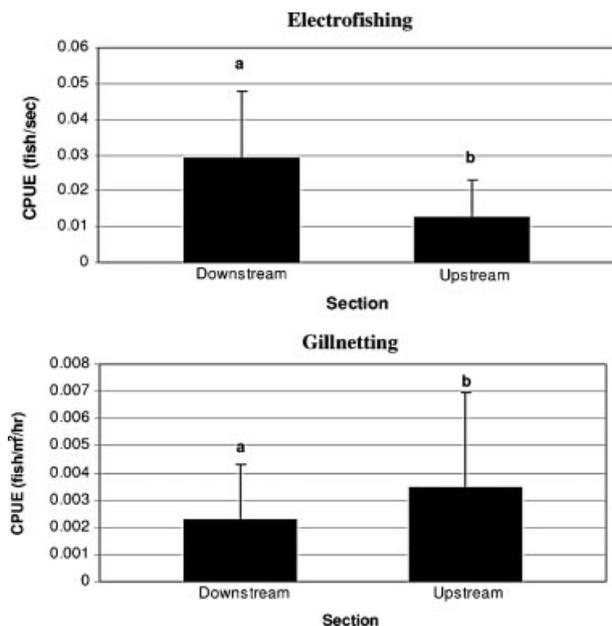


Figure 3. Overall electrofishing and gillnetting relative fish abundance (CPUE) by section. Columns with the same letter are not significantly different ($\alpha = 0.10$). Error bars show standard deviation.

between sections for both gears. For gillnetting, CPUE was higher upstream than downstream ($F = 3.02$, $p = 0.096$). In contrast, electrofishing CPUE was significantly higher downstream than upstream ($F = 11.02$, $p = 0.004$).

Spring CPUE was not significantly different among habitat types for gillnetting ($F = 0.060$, $p = 0.980$) or electrofishing ($F = 2.22$, $p = 0.125$). For gillnetting, FB sites had high variability in catches. The highest CPUE for all sites in spring was recorded at FB2, and FB site catches also included two zeros. CPUE was not significantly different between sections for gillnetting ($F = 0.160$, $p = 0.694$) or electrofishing ($F = 0.96$, $p = 0.341$).

Fall CPUE was significantly different among habitat types for gillnetting and between sections for electrofishing. For gillnetting, CPUE was significantly higher at FB sites than RR sites ($F = 3.26$, $p = 0.049$). CPUE was not significantly different between the upstream and downstream sections ($F = 1.69$, $p = 0.212$). For electrofishing, CPUE was not significantly different among habitat types ($F = 1.28$, $p = 0.309$) but was significantly higher upstream than downstream ($F = 15.11$, $p < 0.001$).

Species diversity. Differences in species diversity as indicated by the Shannon Index, were significant only 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 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 ($F = 1.60$, $p = 0.214$). Diversity upstream ($H' = 2.21$) was significantly higher than downstream ($H' = 1.84$) ($F = 4.58$, $p = 0.036$).

Community composition. 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%). Yellow perch was the most common fish caught (34%), followed by pumpkinseed (14%), largemouth bass (11%), brown bullhead (10%), largescale sucker *Catostomus macrocheilus* (7%), bluegill (7%), smallmouth bass (6%) and longnose sucker *Catostomus catostomus* (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 was 133 mm (range, 26–1337 mm TL).

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, brown bullhead and pumpkinseed were captured in significantly different numbers among habitat types (Figure 5, Table I). Largescale sucker were captured in significantly greater numbers at FB sites than RR

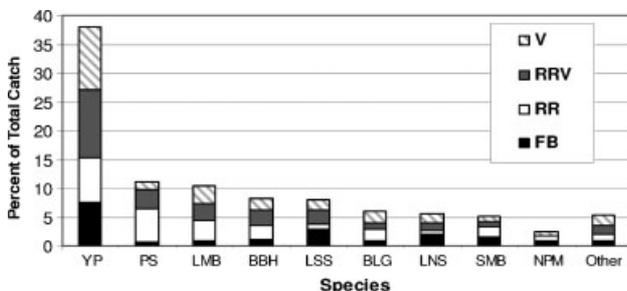


Figure 5. Per cent of total catch for individual species by habitat type (failing bank (FB), vegetation (V), riprap (RR), riprap with vegetation (RRV)) on the lower Coeur d'Alene River, Idaho. YP, yellow perch; PS, pumpkinseed; LMB, largemouth bass; BBH, brown bullhead; LSS, largescale sucker; BLG, bluegill; LNS, longnose sucker; SMB, smallmouth bass; NPM, northern pikeminnow.

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

| Species | Habitat type | | | |
|---------------------------|--------------|-----|-----|-----|
| | FB | RR | RRV | V |
| Longnose sucker | 60 | 16 | 39 | 41 |
| Largescale sucker | 81 | 26 | 69 | 52 |
| Northern pike | 2 | 6 | 9 | 2 |
| Brown bullhead | 30 | 73 | 76 | 52 |
| Pumpkinseed | 21 | 165 | 90 | 42 |
| Bluegill | 25 | 59 | 32 | 53 |
| Smallmouth bass | 47 | 51 | 24 | 25 |
| Largemouth bass | 26 | 101 | 81 | 91 |
| Kokanee | 5 | 9 | 6 | 5 |
| Chinook salmon | 1 | 0 | 0 | 19 |
| Yellow perch | 212 | 226 | 332 | 309 |
| Black crappie | 1 | 6 | 6 | 8 |
| Mountain whitefish | 1 | 2 | 2 | 3 |
| Northern pikeminnow | 27 | 22 | 3 | 17 |
| Westslope cutthroat trout | 6 | 5 | 10 | 2 |
| Rainbow trout | 0 | 0 | 1 | 0 |
| Brook trout | 0 | 0 | 0 | 1 |
| Tench | 10 | 6 | 7 | 10 |
| Total | 555 | 773 | 787 | 732 |

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.

Many species showed significant differences in catch between river sections (Figure 6). Perch ($\chi^2 = 10.552$, $p < 0.001$), largemouth bass ($\chi^2 = 5.77$, $p = 0.016$), bullhead ($\chi^2 = 5.968$, $p = 0.015$) and smallmouth bass ($\chi^2 = 22.653$, $p < 0.001$) catches were significantly higher downstream than upstream. Largescale sucker ($\chi^2 = 3.352$, $p = 0.067$), longnose sucker ($\chi^2 = 26.602$, $p < 0.001$), northern pikeminnow ($\chi^2 = 4.746$, $p = 0.029$) and westslope cutthroat trout ($\chi^2 = 7.553$, $p = 0.006$) 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.

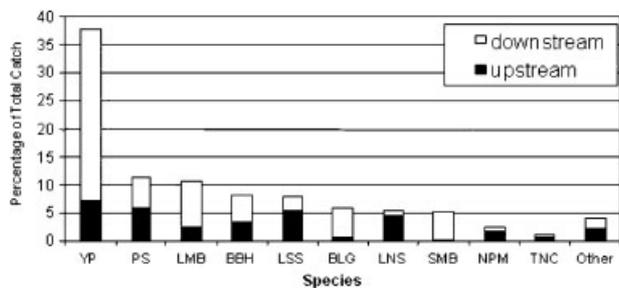


Figure 6. Percentage of total catch by species and section on the lower Coeur d'Alene River, Idaho. YP, yellow perch; PS, pumpkinseed; LMB, largemouth bass; BBH, brown bullhead; LSS, largescale sucker; BLG, bluegill; LNS, longnose sucker; SMB, smallmouth bass; NPM, northern pikeminnow.

Relative fish abundance and habitat variables

Shoreline habitat quality (RBP scores) in the lower Coeur d'Alene River was generally poor, even at unstabilized, vegetation sites. Overall, all four shoreline habitat types scored poor or marginal in terms of substrate/available cover, pool variability, sediment deposition and riparian zone width.

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 'section' (upstream vs. downstream) (Table II). However, differences in CPUE for spring gillnetting and summer electrofishing were not 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, per cent 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 II). For electrofishing, a model consisting of the variables width, overhanging vegetation, aquatic vegetation, slope and maximum depth 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, did not 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 II).

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 II). 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 II). Aquatic vegetation was not present in spring and was minimal in fall; therefore, it was not included in the regression model for these seasons.

Table II. 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). Italics show significant p -values

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

Relative fish abundance at stabilized sites

CPUE at stabilized sites was significantly explained by the habitat variables riprap depth and rock diameter during certain season/gear combinations (Figures 7 and 8). Gillnetting CPUE was positively correlated with riprap depth during all seasons (Figure 7), and showed the strongest correlation in summer ($F = 8.10$, $p = 0.017$, $R^2 = 0.3922$). Electrofishing CPUE showed negative correlations with riprap depth in summer and fall, and showed 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 8). Gillnetting CPUE was not significantly correlated with riprap diameter during any seasons. Electrofishing CPUE showed a positive correlation with riprap diameter in all

seasons, although only spring was significant ($F = 5.33$, $p = 0.044$, $R^2 = 0.283$).

DISCUSSION

The higher overall relative fish abundance (CPUE) for electrofishing at stabilized (RR and RRV) sites than 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.

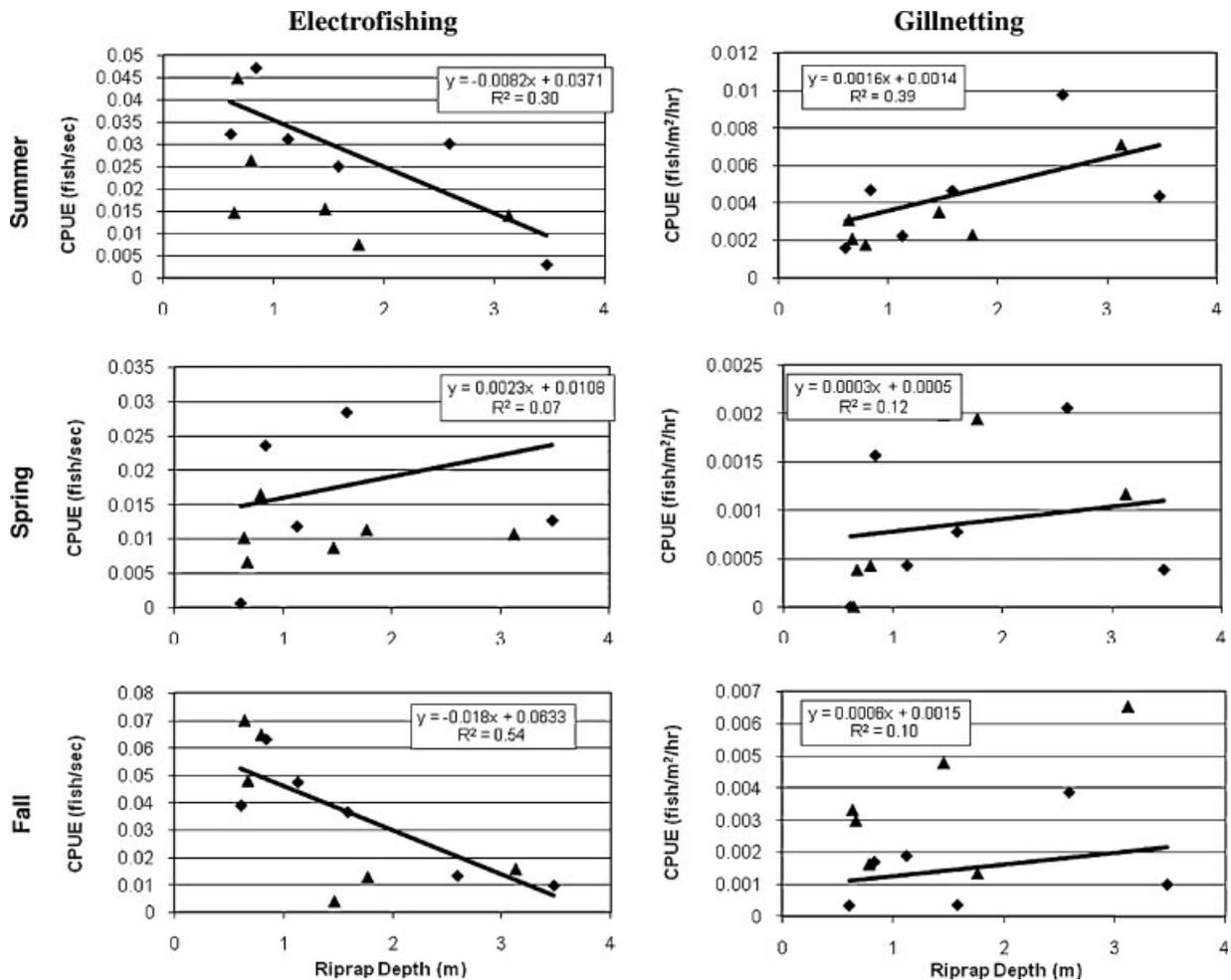


Figure 7. Linear regressions of CPUE (y-variable) with riprap depth (x-variable) for gear (gillnetting and electrofishing) and season (summer, spring and fall). Regression equations and r^2 values are shown in the insets. Triangles are riprap with vegetation (RRV) sites and diamonds are riprap sites (RR).

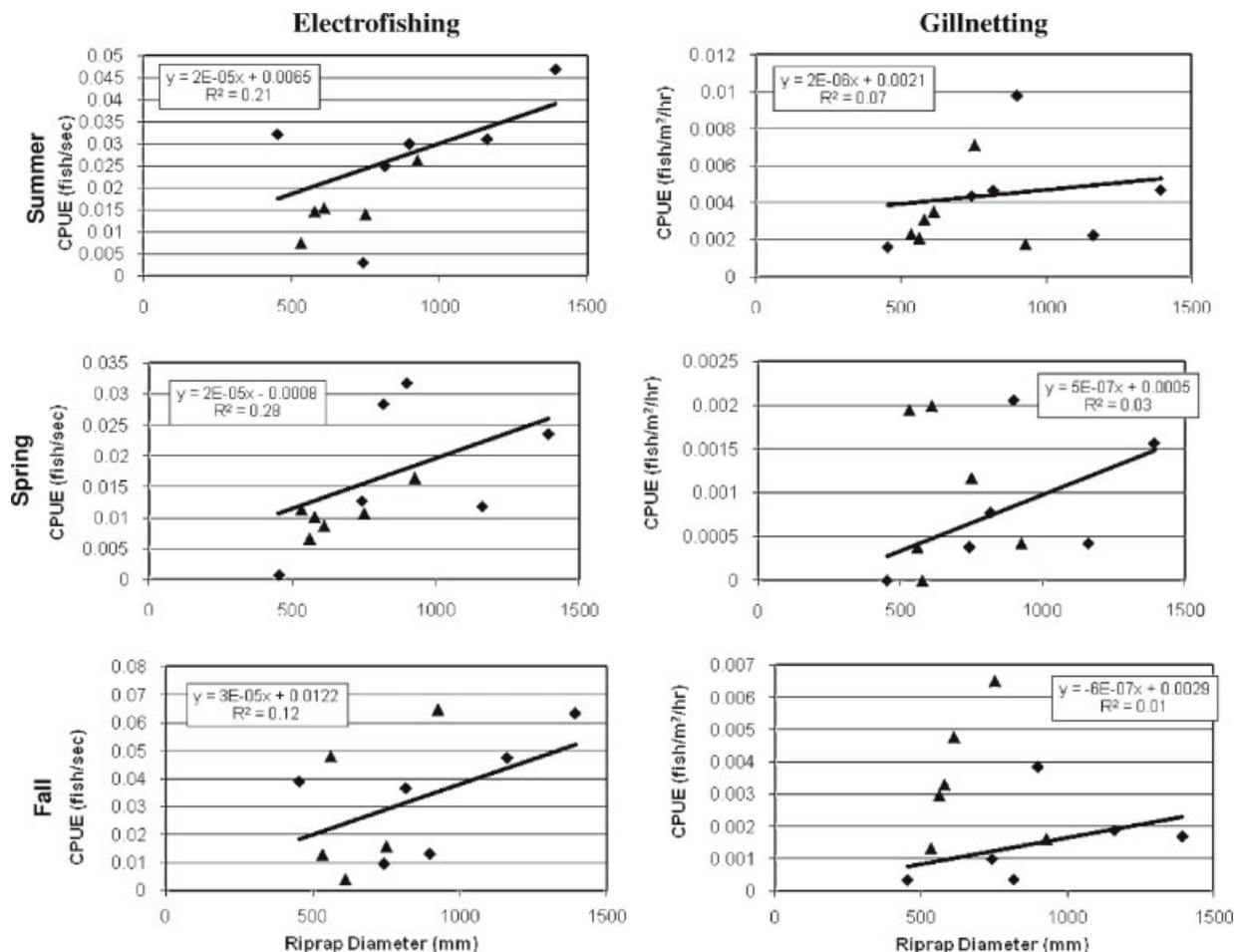


Figure 8. Linear regressions of CPUE (y-variable) with riprap diameter (x-variable) for gear (gillnetting and electrofishing) and season (summer, spring and fall). Regression equations and r^2 values are shown in the insets.

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; White *et al.*, 2009). 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 (*O. 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 riprapped 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 effects of bank stabilization on fish communities are a source of continuing confusion and have significant implications for habitat management in large rivers. Based on a review of the above studies, we suggest that the variable results are attributable to at least two factors: (i) the habitat quality of

the unstabilized river, and (ii) the percentage of the river that has been stabilized.

First, in systems where suitable natural habitat is scarce and limiting to production, 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 RBP 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. On the Upper Yellowstone River, abundance of juvenile salmonids (rainbow trout, mountain whitefish, Yellowstone cutthroat trout *Oncorhynchus clarki bowleri*, brown trout *Salmo trutta* and brook trout *S. 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). 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 *O. mykiss* standing stocks increased in newly riprapped sections in large streams. The increase of cutthroat was attributed to increased quantity of habitat (Knudsen and Dilley, 1987).

Conversely, natural rivers and small streams where high-quality habitat such as LWD exists may suffer declines in habitat quality from actions associated with bank stabilization (Elser, 1968; Angermeier and Karr, 1984; Knudsen and Dilley, 1987; Craig and Zale, 2003; Angradi *et al.*, 2004). In western Washington streams, the severity of habitat alterations coincided with stream size. Salmonid standing stocks decreased as a result of riprap in small streams and increased in large streams (Knudsen and Dilley, 1987). Craig and Zale (2003) surmised that diversity and abundance of fish at stabilized banks as compared with unstabilized banks, often increased in previously degraded habitats and decreased in pristine habitats inhabited by salmonids. 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 structure may depend not only on the habitat quality of the unstabilized river but also on the overall extent of the stabilization in the basin. In the lower Coeur d'Alene River, only 2.5% of the river bank had been stabilized as of 2004 (KSSWCD, 2004). Although the river has been altered from its original state as Post Falls Dam created slackwater habitat, some aspects of natural river function persist, including 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 may function positively as the additional structure provides habitat complexity without major loss of river function.

Although our study was not designed to test this hypothesis, this interpretation is supported by evidence from several other 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 LWD

(Chapman and Knudsen, 1980; Schmetterling *et al.*, 2001; Angradi *et al.*, 2004), 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 (Morris *et al.*, 1968; Hesse *et al.*, 1989; Hesse and Sheets, 1993). The detrimental effects of riprap and other bank stabilization on river function and habitat thus become cumulative and may be detrimental when applied to large stretches of river (Jennings *et al.*, 1999; Schmetterling *et al.*, 2001). In a recent study of fish communities and riprap on the Kansas River, White *et al.* (2009) employed a sampling design and sampling scale similar to this study, found similar results as in this study (overall, more fish at riprap sites) and reached similar conclusions about larger scale effects such as loss of channel connectivity with backwaters. They suggested that 'the impacts of riprap in riverine systems may be scale dependent'.

Seasonal differences in relative abundance among habitat types demonstrated in RR and RRV structures on the lower Coeur d'Alene River are providing local habitat benefits 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 only armored at the wave line and above and were not submerged at winter pool elevations. We were unable to account for this seasonal difference in the amount of riprap habitat available, but we estimated that four of the stabilized shoreline sites 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 brown bullhead, black crappie, bluegill, Chinook, kokanee, largemouth bass, smallmouth bass, northern pike, pumpkinseed, tench and yellow perch, was not unexpected because such species often flourish after habitat alterations (Moyle and Light, 1996). The habitat changes include an altered hydrograph, slackwater habitat (spring to fall) created by Post Falls Dam, in addition to the bank stabilization and other changes associated with the addition of mine tailings.

Differences in relative fish abundance and species composition, for both section (upstream and 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 sucker, and northern pikeminnow, and coldwater species longnose sucker and cutthroat trout (Simpson and Wallace, 1978; Zaroban *et al.*, 1999; Mebane *et al.*, 2003). By season, salmonids (cutthroat trout, kokanee, Chinook and mountain whitefish) were captured in greater numbers during spring because water temperatures are low during this season. During the same time, warmwater and some coolwater species catches were lowest and these species were seeking warmer temperatures, likely in the adjoining chain lakes. For example, 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 more 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 warmwater species (e.g. largemouth bass and brown bullhead) or coolwater species (e.g. smallmouth bass and yellow perch). Most of these species spawn in early to late spring. At spawning time in late spring, 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 (M. Liter, Idaho Department of Fish and Game, Coeur d'Alene, Idaho, Personal Communication). Northern pike migrate to flooded marshes and wetlands, or shallow shorelines with vegetation shortly after ice-out (Casselmann and Lewis, 1996). In this study northern pike were completely absent from 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. In contrast, coldwater salmonids were most abundant in spring. Chinook, kokanee and cutthroat trout juveniles were probably migrating into Lake Coeur d'Alene from spawning and rearing areas higher in the basin.

The result that the differences in fish relative abundance (CPUE) were most definitively explained by the habitat variable 'section' (i.e. upstream vs. downstream), is consistent with other reports that have shown a gradation of change in community composition from upstream to downstream according to changes in habitat (Sheldon, 1968; Hynes, 1970; Vannote *et al.*, 1980; Schlosser, 1982). On the lower Coeur d'Alene River the major difference between sections is the proximity to the chain lakes and their associated wetland habitats to the lower section. 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; Ross and Baker, 1983; Amoros, 1991).

Though the habitat variable 'section' had the greatest power in explaining differences in CPUE, other habitat variables were also explanatory. Aquatic vegetation, for example 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 maximum depth in these areas. Such vegetation was completely absent in spring and had largely died back by fall.

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 five out of 6 months sampled between May and October. Similarly, juvenile Chinook and steelhead densities were higher at sites with larger riprap (>30 cm diameter) in two southern British Columbia streams (Lister *et al.*, 1995).

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 low fraction (2.5%) of bank stabilization. The primary 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) should not be extrapolated greatly beyond existing river conditions. Our study was designed only to compare the fish communities at stabilized and unstabilized sites in a largely unstabilized lower river segment lacking good natural habitat. Indirect and cumulative effects of high percentage of riprapped bank were not evaluated in this study. For an adequate evaluation of effects of bank stabilization on the entire fish community of the lower Coeur d'Alene River, the site specific study conducted here must be complemented by a more comprehensive study of effect on river function.

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