EFFECTS OF HABITAT REHABILITATION ACTIVITIES ON FISH ASSEMBLAGES AND POPULATIONS IN THE KOOTENAI RIVER, IDAHO

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ABSTRACT

The Kootenai River supports some of the Pacific Northwest's most important cultural and recreational fishery resources. Lotic fishes have been substantially influenced by alteration to the course and function of large rivers throughout the world. Like many large river systems, the Kootenai River has been degraded by extensive water development and land use activities that have disrupted its function and integrity. As such, managers have attempted to rehabilitate degraded lotic habitats to support a variety of fish assemblages. The Kootenai River is one such large river where managers are improving habitat conditions for native fishes. I investigated spatial variation in fish assemblage structure and the influence of habitat on fish assemblages and species-specific occurrence and relative abundance within the context of an ongoing habitat rehabilitation project. In addition, I evaluated the sampling effort required to estimate various levels of species richness to guide future monitoring efforts of fishes in the Kootenai River. Fishes were collected throughout the braided reach of the Kootenai River during the summers of 2012 and 2013 to evaluate patterns in fish assemblage and population structure among rehabilitation and reference sites. In general, I found that fish assemblage structure varied among sites and that native fishes had high spatial association with one another. I identified several environmental variables associated with the occurrence and relative abundance of native species and evaluated the effect of those variables. My research also provided insight on sampling effort requirements to achieve target levels of species richness that will help fishery scientists be efficient during monitoring efforts. My results provide guidance and valuable information on the spatial occurrence of fishes in the Kootenai River and on habitat use patterns that can be used to design future rehabilitation projects to benefit native species.

In addition, my evaluation of sampling effort requirements will help managers to develop realistic sampling objectives and optimize protocols for evaluating the effects of habitat rehabilitation on fish assemblages in the Kootenai River.

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CHAPTER 1: GENERAL INTRODUCTION

Freshwater ecosystems are among the world's most endangered and degraded systems. Declines in biodiversity are much greater in aquatic ecosystems than in terrestrial ecosystems making them a particularly important focus for conservation (Schlosser 1991; Rinne et al. 2005). Large rivers are particularly imperiled due to extensive anthropogenic disturbances (e.g., dams, dikes, levees, diversions) associated with acquiring various services (e.g., energy production, flood control, water storage; Hunt 1992; Dynesius and Nilsson 1994; Galat and Zweimüller 2001; Nilsson et al. 2005; Dudgeon et al. 2006). Large rivers are greatly affected by what occurs in their catchments; thus, water development and land-use disturbances have the potential to precipitate a host of deleterious effects on ecosystem function (Benke 1990; Bayley 1995) that are often reflected in changes to fish assemblage structure and function (Paragamian 2002).

Freshwater ecosystems support nearly 43.0% of all described fish species worldwide, even though freshwater only accounts for 0.01% of the total water in the biosphere (Dudgeon et al. 2006; Jelks et al. 2008). Of the freshwater available, only 0.006% is present in rivers and streams at any point in time (Master 1990; Malmqvist and Rundle 2002). The disproportionately high level of biodiversity found in freshwater, coupled with freshwater's vulnerability to environmental change, makes large rivers important hotspots for biodiversity (Master 1990). In light of anthropogenic changes to many large rivers, conservation efforts have focused on improving ecosystem integrity in rivers and streams that have been affected by fragmentation and flow alteration (Gore and Shields 1995; Lake et al. 2007; Roni et al. 2008). Therefore, lotic systems have become the target of habitat rehabilitation efforts aimed at improving environmental conditions for target species (Lake et al. 2007).

Since disconnection of the Kootenai River from its historic floodplain, side-channel habitats, though scarce, have been of high importance to the fish assemblage. While much of the lower Kootenai River was once a prominent floodplain ecosystem, flood control initiatives left a simple channelized river and a uniform terrestrial floodplain throughout all but the “braided section” near Bonners Ferry, Idaho. The Kootenai River near Bonners Ferry is now a relic of a once dynamic and complex portion of the river composed of multiple side channels, alcoves, and shallow pool-riffle complexes. Being the only remaining river segment with habitat characteristics similar to the historic floodplain and bearing higher connectivity to the terrestrial environment, the braided section of the Kootenai River is thought to be of high importance to fishes. Side-channel habitats, in particular, are thought to provide important habitat diversity and refugia for fishes, and likely serve as the last remaining link to terrestrial habitats (Barko and Herzog 2003; KTOI 2009). The Kootenai Tribe of Idaho (KTOI), working in combination with multiple agency partners, is using a broad, ecosystem-based approach to restoring and maintaining habitat conditions in the lower Kootenai River. Impetus for the project emerged from declines in native fish species, particularly white sturgeon *Acipenser transmontanous*. A major focus of the project is to use habitat treatments to address factors limiting native fish populations, thereby sustaining economic and cultural value provided by the Kootenai River.

In light of recent efforts to improve habitat conditions for fishes in the Kootenai River, the purpose of this research was to assess spatial variation in fish assemblage and population structure in main- and side-channel areas throughout the Kootenai River. Along with identifying patterns of fish assemblage structure and specific habitat associations, I sought to provide guidelines for future sampling effort to estimate assemblage indices. An overarching objective of this research was to provide comprehensive baseline information on the spatial occurrence of fishes in the braided section of the Kootenai River for making meaningful comparisons during future fish assemblage evaluations, especially those conducted to evaluate future habitat rehabilitation efforts.

THESIS ORGANIZATION

This thesis contains four chapters. Chapter two evaluates patterns in fish assemblage structure and associations with environmental characteristics among main- and side-channel habitats throughout the braided section of the Kootenai River. Chapter three evaluates sample size requirements for estimating species richness for monitoring habitat rehabilitation activities in the Kootenai River. Chapter four synthesizes results from all chapters and outlines the general conclusions of this thesis.

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CHAPTER 2: PATTERNS OF FISH ASSEMBLAGE STRUCTURE AND HABITAT USE AMONG MAIN- AND SIDE-CHANNEL HABITATS IN THE LOWER KOOTENAI RIVER, IDAHO

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ABSTRACT

The Kootenai River has been extensively modified by land-use and water-development activities throughout the past century where habitat loss has resulted in the decline of native fishes. Fish assemblage structure was compared among seven sites within main- and side-channel habitats during the summers of 2012 and 2013 in the lower Kootenai River, Idaho. Fishes and habitat characteristics were sampled using standard methods at each site on a biweekly schedule to evaluate the association of fish assemblages and populations with environmental variables. Fish assemblage relationships were evaluated using information on occurrence and relative abundance. Species-specific habitat associations were modeled to identify variables that were important in explaining variability in occurrence and relative abundance. Side-channel habitats were characterized by a high proportion of fine substrate, low current velocities, shallow depths, and abundant wood relative to main-channel habitats. Species composition was dominated by largescale sucker *Catostomus macrocheilus* and mountain whitefish *Prosopium williamsoni* at all sites. Side-channel habitats tended to have higher species richness than main-channel habitats, and native fishes had high reach similarity. Newly-rehabilitated sites had high catch rates of non-native fishes. Habitat use was directly related to each species' ecology and important habitat variables associated with occurrence and relative abundance of a given species were different. Results of this study suggest that non-native fishes were associated with rehabilitated side-channel habitats at the downstream extent of our study area. In addition, biotic interactions between native and non-native fishes may reflect the lack of reach similarity observed in our study. Overall, this study provides information on large river fish assemblages and patterns of habitat use in a large river system.

INTRODUCTION

The inherent nature of running waters has long made them the focus of human settlement and exploitation. High concentration of human settlement around large rivers has resulted in various land- and water-use alterations (Benke 1990; Bayley 1995). Large rivers throughout the world have been modified by water development activities to serve human needs (e.g., transportation, irrigation, and power generation), which has resulted in the loss and degradation of habitat used by fishes (Dynesius and Nilsson 1994; Nilsson et al. 2005). In particular, dams (i.e., instream development) and their impoundments have been identified as one of the greatest threats to aquatic ecosystem health and biodiversity. Modification of natural flow regimes has resulted in the homogenization of aquatic habitat and a general loss of habitat complexity. Habitat loss has subsequently resulted in the decline of various riverine fish species in North America (e.g., white sturgeon *Acipenser transmontanous*, pallid sturgeon *Scaphirhynchus albus*, Chinook salmon *Oncyhynchus tshawytscha*) and changes in fish assemblage structure (Pflieger and Grace 1987; Paragamian 2002; Barko et al. 2004). Construction of levees and other shoreline development activities along rivers can have deleterious effects on the function of large rivers by restricting floodplain access. Levees restrict flow to the main channel and prevent water from moving laterally onto the terrestrial floodplain during high discharge events, thereby decreasing habitat complexity and heterogeneity (White et al. 2009; Schloesser et al. 2011). Instream and shoreline alteration in large rivers often shifts species composition toward an assemblage consisting of more generalist species and fewer fluvial specialists with complex life histories (Kinsolving and Bain 1993; Paragamian 2002; Pegg and McClelland 2004). For example, species that require flowing water to carry out all or portions of their life histories may be adversely affected by loss of habitat due to impoundments. In addition, species that require seasonal variability in discharge and off-channel habitats created by flooding often display poor growth and recruitment (Pringle et al. 2000; Galat and Zweimüller 2001; Giannico and Hinch 2003).

Loss of natural habitat and habitat homogenization have been implicated as factors contributing to the decreased abundance and growth of various fluvial species throughout the world (Rahel 2000). As such, efforts to mitigate for habitat loss and degradation have been a major focus of many natural resource agencies interested in supporting native fish populations. Habitat rehabilitation in fluvial systems is a common method for improving habitat complexity for fishes at small spatial scales. Engineered habitat structures (e.g., riprap banks, scour pools, woody debris additions) have been widely-used in large river systems throughout the midwestern United States to provide habitat for fishes and to simulate floodplain environments (Madejczk et al. 1998; White et al. 2009; Schloesser et al. 2011). Engineered (man-made) structures (e.g., wing dikes, artificial alcoves, coarse banks) create roughness barriers in channelized rivers which, in turn, provide low-velocity shallow water habitats (Barko et al. 2004). Oftentimes, the only low-velocity habitat available in highly modified large rivers exist near engineered structures. These structures are critical not only because they provide important physical habitat that meet the ecological needs of many fluvial species, but also because they create isolated areas along the river continuum where the hydrologic dynamics mimic pre-impoundment or pre-channelization conditions. Decreased velocity and dispersed flow associated with engineered structures facilitate sedimentation, nutrient exchange, and can moderate water temperature (Cushman 1985; Junk et al. 1989). Several studies have documented benefits to fluvial species in large rivers with habitat rehabilitation projects, thus leading to increased interest from natural resource agencies in using habitat rehabilitation to increase the abundance of important fish species (Gore and Shields 1995; Lake et al. 2007; Romanov et al. 2012).

The Kootenai River is a large river that originates in British Columbia, Canada and flows through the panhandle of northern Idaho. The portion of the Kootenai River in Idaho is characterized by a prominent floodplain that historically provided a substantial amount of flooded terrestrial habitat during high runoff events; however, many flood-related habitats (e.g., side channels, sloughs, oxbows) in the Kootenai River have been eliminated as a result of instream and shoreline development. The lower Kootenai has experienced considerable water development and flow modification over the last century to serve the needs of a growing human population (Knudsen 1993). Primary modifications of the Kootenai River include the construction of levees along the floodplain of the lower river and construction of Libby Dam, a large hydropower facility completed in 1974. Libby Dam has been implicated in changes to natural flow regimes (e.g., temperature, nutrients, and discharge) and shifts in fish assemblage structure. The decline of two important fish species in the lower Kootenai River has been directly attributed to water development and flow modification (Partridge 1983; Duke 1999). Specifically, declines in federally-endangered white sturgeon (Partridge 1983; Apperson 1990; Paragamian 2012) and imperiled burbot *Lota lota* (Paragamian and Hansen 2009) populations have warranted recent efforts to improve aquatic habitat in the Kootenai River. The rationale behind these habitat improvement projects is to enhance existing habitat and create side-channel habitat suitable to support various life history stages of native fishes in the Kootenai River. Habitat rehabilitation in the Kootenai River has been ongoing since 2011 and is largely focused on improving habitat in existing side-channel environments in the lower river. It has been hypothesized that deep, high velocity habitat is abundant and not limiting for most native fishes (KTOI 2009). As such, engineered structures that reduce current velocity, disperse flow, stabilize banks, enhance substrate heterogeneity, and provide cover have been identified as desirable features for supporting native fishes (Schloesser et al. 2009; KTOI 2009). It is thought that these habitat characteristics are limiting for native fishes in the lower Kootenai River. Information gleaned from studies in other systems has provided insight on habitat use for the native fishes found in the Kootenai River and provided preliminary guidance for rehabilitation activities. However, baseline information on fish assemblage structure and a general description of potential responses that may be elicited from habitat rehabilitation is needed.

Although abiotic environmental characteristics influence fish assemblage- and population-level structure, biotic interactions also function to structure fish assemblages at various spatial scales. Non-native fishes have direct (e.g., predation) and indirect (e.g., habitat alteration) effects on native fishes. For instance, Maiolie et al. (2002) reported declines in kokanee *Oncorhynchus nerka* in Lake Pend Oreille following the introduction of non-native lake trout *Salvelinus namaycush*. Indirect effects of non-native fishes in Iowa lakes have also been reported. Jackson et al. (2010) found that common carp *Cyprinus carpio* degraded water quality in Iowa lakes and created turbid conditions that negatively influenced native, sight-feeding piscivores (e.g., bluegill *Lepomis macrochirus* and black basses *Micropterus* spp.). In cases where habitat rehabilitation has been used to support native fish populations, issues associated with improvement of habitat for non-native fishes and their resulting colonization are often ignored. Oftentimes, habitat use patterns overlap among native and non-native fishes, and desired results from habitat rehabilitation may be confounded by interactions with non-native fishes. As such, natural resource managers must carefully consider the potential effects of non-native fishes and the potential overlap of habitat requirements with native fishes. Understanding the habitats relationships of native and non-native fishes in large rivers can help managers prioritize rehabilitation projects and provide insight on useful treatments for restoring native fish populations.

Little is known about the structure and habitat associations of fish assemblages in large western river systems and the relative importance of side-channel habitat. Rehabilitation projects on small streams are common and there is a large body of knowledge surrounding the techniques and expected responses from the biotic community (Regier et al. 1989; Kern 1992; Binns 2004). However, recent interest in restoring native fish assemblages and populations in large rivers has led to rehabilitation projects becoming increasingly common. Unlike small stream projects, rehabilitation in large rivers is generally focused on improving the quantity and quality of existing habitat rather than creating new habitat (Gore and Milner 1990; Gore and Shields 1995). Improved areas are intended to be attractive to native colonists and recovery is often defined in terms of the change in ecological structure or function and its associated value (Gore and Shields 1995). In light of this interest, a comprehensive description of fish assemblage structure and habitat characteristics structuring assemblages in large coldwater rivers is needed to guide management and rehabilitation plans. The objectives of this study were two-fold. First, we sought to describe fish assemblage structure throughout a portion of the Kootenai River (i.e., braided section) where intensive habitat rehabilitation has been undertaken and where additional rehabilitation projects are planned to occur. We also sought to evaluate habitat characteristics related to the occurrence and relative abundance of fishes in the braided section of the Kootenai River. Given these objectives, our research hypotheses were that (1) side-channel habitats would have higher occurrences of rare species and have higher relative abundances of all species compared to main-channel habitats, (2) native and non-native fish assemblages would display spatial overlap in terms of reach-specific occurrence, and (3) environmental characteristics associated with species-specific habitat use would vary among taxa.

METHODS

Study area

The Kootenai River is the second largest tributary to the Columbia River and has an international watershed encompassing portions of both Canada and the United States. The Kootenai River watershed is approximately 50,000 km^2 in area, mountainous, mostly forested, and greatly influenced by spring runoff from snowmelt (Knudson 1993). The Kootenai River originates in the mountains of British Columbia, Canada in Kootenay National Park and flows south into the United States where it is impounded by Libby Dam near Jennings, Montana forming Lake Koocanusa. Upon release from Libby Dam, the Kootenai River flows west into Idaho before returning to British Columbia, Canada where it forms the southern arm of Kootenay Lake.

The Idaho portion of the Kootenai River is delineated into three distinct river sections based on channel morphology: canyon, braided, and meander sections (Fossness and Williams 2009). The braided section (246-257 river kilometer) extends from the confluence of the Moyie River downstream to Bonners Ferry, Idaho (Figure 2.1). The braided section is characterized by a shallow (< 2 m) and wide channel type with a heterogeneous substrate composition and high flow variability (KTOI 2009; Smith 2013). The braided section has a braided channel type with a variety of side-channel habitats that are particularly prominent during high discharge events. The braided section is particularly unique because it still contains side-channel habitats that are characteristic of pre-impoundment conditions historically found throughout the lower Kootenai River. In addition, the braided section contains greater variability in habitat compared to the canyon and meander sections. Substrate composition and flow velocities are homogenous with low habitat variability throughout the canyon (velocity > 1 m/s; cobble and boulder substrate) and meander (velocity < 0.5 m/s; sand and silt substrate) sections. However, in the braided section, heterogeneity in velocity is higher and substrate composition is much more diverse (KTOI 2009; KTOI 2012; Smith 2013).

Sampling design and data collection

Sampling occurred at seven sites within the braided section of the Kootenai River during the summers of 2012 (July-September) and 2013 (May-September; Figure 1). Sites were stratified as occurring in either side-channel or main-channel (i.e., existing only in the main channel) environments. Because a focus of this study was to provide baseline data from sites that would undergo habitat rehabilitation in the future, we selected sites based on planned rehabilitation. Main- and side-channel environments were treated as separate strata for this study based on inherent differences in habitat characteristics. Three side-channel complexes and four main-channel meanders were designated as sampling sites (hereafter referred to as sites). Each side-channel site was then divided into individual 100-m long reaches (hereafter referred to as reaches) along the thalweg. Each main-channel site was divided into 100-m long reaches along both the inside and outside bend. A global positioning system (Garmin International, Inc., Olathe, Kansas) was used to georeference the upper and lower terminus of each reach and fluorescent flagging was placed along the bank to identify the spatial extent of each reach during sampling.

All reaches were sampled on a biweekly schedule from May through September to account for temporal variability in fish assemblage structure. Previous studies have shown that electrofishing is an effective sampling technique for riverine fishes and is commonly used to conduct fish surveys (Reynolds and Kolz 2012; Smith 2013). As such, sampling was conducted during the day using a pulsed-DC boat-mounted electrofisher composed of an Infinity model electrofishing box (Midwest Lake Electrofishing Systems, Inc., Polo, Missouri) and a 5,000-W generator (American Honda Motor Co., Torrance, California). Electrofishing power output was standardized to 3,000 W based on ambient water conductivity and temperature (Miranda 2009). Two netters collected immobilized fish from the bow of the boat during sampling. Dip nets consisted of a 2.8-m long handle and 6.3-mm bar measure knotless mesh. For side-channel sites, electrofishing effort began at the most upstream reach within each site. A single pass was allocated to both the right and left banks of each reach before proceeding to the next reach. Electrofishing began at the uppermost site in each main-channel bend and a single pass was conducted along the bank of each reach before proceeding to the next. Data collected from inside and outside bends were treated separately in subsequent analyses, but nested within the same site. Data from 100-m long reaches in side-channels were considered as replicates nested within their respective site. The catch from each reach was enumerated upon completion of electrofishing. Effort was recorded as the number of seconds of “pedal down” time. Upon completion of sampling in each reach, fishes were identified to species, measured (total length; mm), and released. All fish were released in a location away from subsequent sampling reaches to minimize the influence of immediate emigration back into the sampling area.

Habitat characteristics were measured to evaluate the influence of abiotic factors on fish assemblage structure, species occurrence, and relative abundance. Substrate composition, bank type, and woody debris were measured once during both 2012 and 2013. Depth and velocity were measured during each sampling event. The Kootenai River is highly regulated and lateral water movement and high discharges are infrequent; therefore, habitat characteristics for some variables were sampled on an annual basis due to their static nature. Substrate was evaluated at 25-m intervals along each reach within side-channel sites following Wilhelm et al. (2005). For main-channel sites, we measured from the center of the channel to the corresponding bank of each site. Substrate was then sampled along each transect (extending from the bank to the middle of the main channel) and transects were spaced at 25-m intervals. Substrate composition was estimated as the proportion belonging to one of five categories: silt-sand (< 0.0004–0.2 mm), gravel (0.2–64.0 mm), cobble (64.0–256.0 mm), boulder (> 256.0 mm), and bedrock (modified from Orth and Maughan 1982). All widths were measured using a laser rangefinder and each transect was divided into seven equidistant points where visual estimates of substrate composition were taken (Neebling and Quist 2011). The proportion of bank type was visually estimated as belonging to one of four categories: eroding, vegetated, silt-sand (≤ 0.2 mm), and cobble-boulder (i.e., riprap structure; ≥ 64.0 mm). Woody debris was measured as the total surface area of woody instream cover greater than 0.2 m in diameter and 0.5 m in length in each sampling reach. Estimates of mean water column velocity and mean depth were obtained from the River

Design Group (RDG; River Design Group, Inc., Whitefish, Montana). The RDG has compiled channel morphology and flow data from throughout the braided section of the Kootenai River to establish baseline information on habitat characteristics prior to rehabilitation. Kootenai River hydraulics were simulated using the flow and sediment transport with morphological evolution of channels (FaSTMECH) two-dimensional model (Nelson 1996). The model computes flow-field hydraulics on a curvilinear-fitted grid by solving the depth-averaged shallow-water equations using an eddy viscosity turbulence closure based on a dimensionless drag coefficient. The two-dimensional hydraulic model geometry was sampled at a nominal 3-m resolution from bathymetric data collected by the United States Geological Survey (USGS) in 2009 and merged into 2010 terrestrial light detection and ranging radar (LiDAR). Hydraulic roughness in the two-dimensional model was spatially varied as a function of depth using a nominal roughness height in the gravel to cobble substrate range. The two-dimensional hydraulic model was calibrated to a range of measured stage data from a detailed gage network from water years 2010 and 2011 (Czuba et al. 2011). Two-dimensional model results were post-processed in ArcGIS (Esri, Redlands, California) to develop mean depth, mean velocity, associated variances, and shear stress grids for flow conditions that occurred during fish sampling.

Fish assemblage structure

Nonmetric multidimensional scaling (NMDS) was used to describe spatial patterns in fish assemblage structure among sampling sites and coarse-scale associations between fish assemblages and habitat characteristics. Nonmetric multidimensional scaling is a robust ordination technique that is commonly used to investigate patterns in fish assemblage structure (Ruetz et al. 2007; Rowe et al. 2009). Assemblage structure was evaluated in two analyses: one using occurrence (i.e., presence-absence) and one using relative abundance (i.e., catch per unit effort [CPUE] = fish/hr) data. Differences in assemblage structure were evaluated using a permutational multivariate analysis of variance (PERMANOVA; Loisl et al. 2014). If a significant difference ($P \leq 0.05$) was detected among sites, then habitat vectors were fitted to the ordination with rotational vector fitting (Faith and Norris 1989). Only significant ($P \leq 0.05$) habitat variables were fit to the ordination with a permutation test using the Envfit function in the Vegan package in Program R (R Development Core Team 2009; Oksanen et al. 2011). A Bray-Curtis distance measure was used for all NMDS ordinations using the MetaMDS and Adonis functions in the Vegan package, Program R (R Development Core Team 2012; Oksanen et al. 2011; Oksanen 2013).

A cluster analysis was used to evaluate spatial similarity in fish assemblage structure using all 100-m long sampling reaches. Jaccard's index of similarity (Jongman et al. 1995) was calculated for all possible pairs of species occurrence in reaches sampled during both years. The matrix of similarity values for species was then clustered using the unweighted paired-group method (UPGMA; Cairns and Kaesler 1971; Jongman et al. 1995; Matthews 1998; Quist et al. 2005). The resulting dendrogram displayed clusters of species with similarity in terms of occurrence in sampling reaches. The cluster analysis was performed using PC-ORD (McCune ad Mefford 2006). This allowed us to describe spatial similarities in fish assemblages and identify groups of sympatric species.

Species-specific habitat use

Patterns of habitat use for individual species were evaluated to provide further insight on habitat associations with fish populations in the Kootenai River. Generalized linear mixed models (Bolker et al. 2008; Irwin et al. 2013) were used to evaluate environmental characteristics associated with the occurrence and relative abundance of selected fish species. We used mixed models to account for lack of true independence among sampling reaches and temporal covariance due to repeated sampling events (Hurlbert 1984). We treated environmental covariates as fixed effects, and reach and sampling date as random effects to induce spatial-temporal correlation using a random-intercept model (Irwin et al. 2013). With a random-intercept model, the random effect simply adjusts the intercept of the model based on the grouping factors (i.e., sampling date and reach). Thus, the model includes a nested random intercept for each reach and each reach has a unique intercept that is a random deviation away from the average. This was conducted because reaches within a site were expected to be more alike than reaches across sites. Models were fit using the glmmadmb function, glmmADMB package in Program R (R Development Core Team 2012; Bolker 2008; Bolker et al. 2008; Irwin et al. 2013). Only species detected in at least 10% of the reaches across all sampling events were retained for species-specific analysis. A binomial error distribution was used to model species occurrence. Error distributions for relative abundance models were selected by creating global models (i.e., model containing the most parameters) and then using Akaike's Information Criterion (AIC) to choose among models with different error distributions. We fit four global models for each species using a poisson distribution, negative binomial distribution, and zero-inflated versions of both. The negative binomial distribution had the lowest AIC value for all species; therefore, a negative binomial distribution was used for subsequent modeling of relative abundance.

Evaluation of mixed-effects models was performed following Irwin et al. (2013) to provide an indication of structural flaws in each model and to assess if there were any major violations of assumptions. Following this framework, we constructed plots using the proportion of observed and predicted counts to assess whether our models were over- or under-predicting observations. Anscombe residuals (Anscombe 1953) were plotted against predicted counts to assess heterogeneity of variance (Anscombe 1949; Hilbe 2008; Irwin et al. 2013). The Anscombe transformation is useful for diagnosing assumptions of mixed-effects models because the distribution of residuals is expected to be normal (Pierce and Schafer 1986). If we observed additional heteroscedasticity in the variance that was not already explained by the model, we considered the model to be a poor predictor of habitat use. Lastly, we used the root mean square error as a measure of model fit to assess the predictive ability of each regression model. Root mean square error is a useful model evaluation metric because it provided an average prediction error for the observed data in the same units as the response variable (i.e., counts of each species) that is easily interpreted.

Multicollinearity among habitat covariates was assessed prior to creating models. Pearson's correlation was used to evaluate the correlation among all possible pairs of habitat covariates. When two habitat covariates were significantly correlated (Pearson's $r \geq |0.70|$; $P \leq 0.05$), the variable with the most ecological relevance of a significantly-correlated pair was retained for further analyses. Mean depth and mean velocity were significantly correlated ($r = 0.83$; $P = 0.003$); therefore, we retained mean depth and used the coefficient of variation (CV) of velocity in the models because heterogeneity in velocity was thought to be important for fishes. Environmental covariates used to explain variation in occurrence and relative abundance are provided in Table 2.1.

Occurrence and relative abundance models consisted of seven to fifteen *a priori* candidate models that were developed for each species. We then used an information theoretic approach to select the most parsimonious models among each set of candidate models for each species (Burnham and Anderson 2002). The *a priori* candidate models consisted of environmental variables that were hypothesized to be of importance to the ecology of each species based on previous studies and general knowledge of large river fish ecology. For lesser-studied species, we generated candidate models by gleaning information from the literature on species with similar ecology. Candidate multiple-regression models were ranked using AIC (Burnham and Anderson 2002). Candidate models with a ΔAIC value ≤ 2 were considered to be nearly as parsimonious as one another and were retained for interpretation (Burnham and Anderson 2002). Coefficient estimates and 95% confidence intervals for the highest confidence models were then calculated to assess the precision of covariates and determine which covariates were most important in explaining variation in occurrence and relative abundance for each species.

RESULTS

We sampled seven total sites containing 113 individual 100-m long reaches during 18 events. A total of 8,338 individual fishes belonging to eight families was collected over the course of the summers of 2012 and 2013. A higher total number of species was sampled from side-channel sites than main-channel sites (Table 2.3). We sampled one federally-threatened species (bull trout; scientific names provided in Table 2.3) and one imperiled species (burbot) in both main and side-channel sites of the braided section. Species composition varied by site; however, mountain whitefish and largescale sucker dominated the catch. For example, the combined catch of largescale sucker and mountain whitefish was 88.9% of the total catch from Treatment Bend 2 and 85.2% of the total catch in the South Side Channel. The North Side Channel had higher catch rates of species that were rare in the other six sites. In addition, catch in the North Side Channel was dominated by largescale sucker (28.4%), northern pikeminnow (16.5%), peamouth (16.8%), and redside shiner (17.7%). The North Side Channel also displayed the lowest percent composition of redband trout (1.5%), a recreationally-important species in the Kootenai River.

Nonmetric multidimensional scaling ordinations provided several insights on fish assemblage structure among sampling sites. A two-dimensional solution was found for both the occurrence (stress = 4.7) and relative abundance (stress = 2.1) ordinations. The NMDS ordination fitted to species occurrence data indicated that largemouth bass, pumpkinseed, and yellow perch were most closely associated with the North Side Channel; whereas, burbot, brown trout, and westslope cutthroat trout were most closely associated with the South Side Channel (Figure 2.2). The NMDS ordination fit to relative abundance data displayed a similar pattern where the North Side Channel had higher relative abundance of largemouth bass, pumpkinseed, and yellow perch compared to the other sites (Figure 2.3). Northern pikeminnow and peamouth were also more abundant in the North Side Channel than the other six sites (Figure 2.3).

Several environmental variables were significantly correlated with the NMDS scores. Habitat variables that were significantly correlated with the NMDS ordination for species occurrence included the CV in velocity (r_s = 0.89; P = 0.004; Figure 2.2). Habitat variables that were significantly correlated with the NMDS ordination for species relative abundance included proportion of fine substrate (r_s = 0.32; P = 0.04), CV in velocity (r_s = 0.45; P = 0.02), and woody cover (r_s = 0.53; P = 0.02; Figure 2.3).

The cluster analysis provided insight that complimented the NMDS ordinations. A dendrogram was produced that displayed a distinct cluster of native species (Figure 2.4). This cluster was composed of kokanee, longnose sucker, largescale sucker, mountain whitefish, northern pikeminnow, peamouth chub, redband trout, and redside shiner. Non-native species failed to exhibit clear associations with native species and other non-native species.

Models explaining habitat use were developed for largescale sucker, mountain whitefish, northern pikeminnow, peamouth, redband trout, redside shiner, and torrent sculpin. For several rare species, habitat use models were not developed due to convergence issues. Several habitat use patterns emerged from regression models that varied among species and families (Table 2.4). Presence of mountain whitefish and redband trout was positively related to the amount of wood and proportion of coarse substrate (≥ 64.0 mm in diameter). Presence of peamouth and redside shiner was positively related to the proportion of fine substrate (≤ 2.0 mm in diameter) and proportion of vegetated bank, but negatively related to the coefficient of variation in current velocity. Reaches with higher mean depth had a higher probability of occurrence for northern pikeminnow, a pattern different than that observed for other native cyprinids.

Models predicting relative abundance of native species indicated that relationships with environmental characteristics differed from those associated with occurrence (Table 2.5). Proportion of cobble-boulder bank was positively related to abundance of torrent sculpin and the proportion of coarse substrate was positively related to abundance of largescale sucker, mountain whitefish, and redband trout. In contrast, the proportion of fine substrate was positively related to the relative abundance of native cyprinids (i.e., peamouth, redside shiner). Heterogeneity of velocity were also important variables related to the relative abundance of cyprinids. Specifically, the CV of velocity was negatively related to relative abundance of peamouth and redside shiner (Table 2.5).

Parameter estimates from the top models for each species were examined and compared to evaluate the precision and relative influence of environmental covariates. Model redundancy was evident for all species where multiple models were retained in the confidence set. For example, the top models predicting relative abundance of northern pikeminnow contained mean depth. Upon further examination, 95% confidence intervals around the parameter estimate for the additional covariate (CV of velocity) encompassed zero. Therefore, the estimate was imprecise and we interpreted the association with depth as more important in explaining variation in relative abundance of northern pikeminnow (Table 2.7). This pattern held true for all other species where a single environmental covariate was most important in explaining occurrence and relative abundance of each species (Table 2.6; Table 2.7). The only case where this pattern was not observed was for models predicting occurrence of redband trout and redside shiner. Evaluation of the top model for redband trout occurrence indicated that surface area of woody cover and proportion of coarse substrate were both important covariates (Table 2.6). Covariates predicting the occurrence of redside shiners included proportion of fine substrate, proportion of vegetated bank, and the CV in velocity (Table 2.6).

DISCUSSION

The Kootenai River is one of the most important and unique resources in the Pacific Northwest. The Kootenai River supports a variety of fish species of cultural and recreational value which have declined precipitously due to effects of water development and instream habitat alteration (Paragamian 2002). The construction of Libby Dam and the levee network along the floodplain represent fundamental discontinuities in processes related to the function of the river (Vannote et al. 1980; Junk et al. 1989). Consequently, interactions between the terrestrial environment and the main channel, as well as interactions between the upper and lower portions of the Kootenai River watershed have been disrupted. In regulated lotic systems like the Kootenai River, variability in habitat and discharge have been replaced by moderated flows and confined channels with relatively homogeneous habitat. The floodplain of the Kootenai River has been reduced to ~25% of its historical extent (KTOI 2009). Moreover, Libby Dam has moderated discharge to roughly 60% of historic discharge that occurred during spring runoff (Duke et al. 1999). Moderated flows and levees have simplified the waterway and reduced the development of seasonal backwater habitats and refugia.

Little information is available regarding the importance of side-channel habitats to riverine fish assemblages, particularly in the western United States. Side-channel habitat is limited throughout the Kootenai River and fishes have little access to flooded terrestrial areas. Previous studies have identified side-channel habitats as important for foraging, spawning, and as nursery habitat (Beckett and Pennington 1986; Baker et al. 1991; Gianicco and Hinch 2003). The importance of side-channel habitat to fish assemblages is related to the habitat diversity provided by flooding. Lower current velocities, high substrate diversity, depth diversity, and additional cover around inundated areas result from seasonal flooding and support fluvial fishes at a variety of life history stages (Barko and Herzog 2003; Giannico and Hinch 2003; Eder 2009; Crites et al. 2012). Side-channel habitats often serve as depositional environments (Whiteman et al. 2011) where low flow velocities allow suspended sediment to settle. Additionally, heterogeneity in current velocity can lead to a mix of substrate types that may support a higher diversity of species. Side-channel habitats contained a higher proportion of fine substrates, shallower depths, and lower current velocities than main-channel habitats. We also observed patterns of fish assemblage structure that followed recognizable patterns that appeared related to habitat characteristics among main- and side-channel environments.

Species richness was generally higher and rare species were more common in side-channel habitats compared to main-channel habitats in the Kootenai River. This observation is consistent with previous studies that have documented higher occurrence and relative abundances of rare species in side-channel habitats compared to main-channel habitats (Whiteman et al. 2011). Eder (2009) reported that flooded terrestrial areas and submersed vegetation found in side-channel habitats was important for age-0 blue suckers *Cycleptus elongates* in the Missouri River. Similarly, Fisher and Willis (2000) reported that the occurrence of rare species was higher in flooded habitats than in the main channel of the Missouri River. Our findings compliment these studies and highlight the importance of maintaining floodplain connectivity and providing habitat diversity for fluvial specialists. Similar findings have been reported in other large floodplain rivers where extensive habitat loss and homogenization has occurred. Whiteman et al. (2011) reported higher relative abundances of shoal chub *Macrhybopsis aestivalis*, sturgeon chub *M. gelida*, silver chub *M.*

storeriana, and river shiner *Notropis blennius* in side channel habitats compared to main-channel habits in the lower Missouri River. Side channels were characterized by high depth diversity, abundant fine substrate, and low current velocities. Our results are similar to previous studies in that the relative abundance of cyprinids (e.g., peamouth, northern pikeminnow) were primarily associated with low current velocities and fine substrate, habitat attributes characteristic of side-channel environments (Torgersen et al 2006).

Patterns of species-specific habitat use were directly related to each species' ecology. Habitat associations of salmonids are well-understood and our results support previous research regarding habitat use of redband trout and mountain whitefish. The presence of coarse substrates has been shown to influence both the occurrence and relative abundance of various salmonid species across western North America (Scott and Crossman 1973; Lanka et al. 1987; Rahel and Hubert 1991), including the Kootenai River (Smith 2013). Coarse substrates are often important for providing habitat for macroinvertebrates that serve as the primary prey resource for salmonids in lotic systems (Flecker and Allen 1984). Stevens and DuPont (2011) found that the occurrence of salmonids in side channel habitats was associated with water temperature in the North Fork Coeur d'Alene River, Idaho. Rosenfeld et al. (2008) found that coho salmon *Oncorhynchus kisutch* parr densities and growth rates were higher in side-channel habitats relative to main-channel habitats. Although we did not observe notable differences in occurrence or relative abundance of salmonids between main- and side-channel habitats, we did observe habitat associations similar to those identified in other studies (Lanka et al. 1987; Rahel and Hubert 1991). Our findings further support the notion that presence of coarse substrate is an important habitat component for salmonid populations. Furthermore, our results are also consistent with habitat use patterns identified by Smith (2013) who found that salmonids were most abundant in the canyon and braided sections of the Kootenai River, and less so in the meander where coarse substrate is rare.

Expansion of non-native fishes is a growing concern and active area of research (Rahel 2000; Quist and Hubert 2004; Benjamin and Baxter 2010). Our results indicate that non-native fishes (i.e., pumpkinseed, largemouth bass, yellow perch) were more common and abundant in the North Side Channel complex, which is the most extensively rehabilitated site in the braided section of the Kootenai River. We also found that native species shared few reaches in common with non-native species. Lack of reach similarity between native and non-native fishes may be a result of negative biotic interactions (Richter et al. 1997). Native fishes are often displaced by non-native fishes through indirect (e.g., competition for food and space resources; Thompson and Rahel 1996; Gido and Brown 1999) or direct (e.g., predation; Ruzycki et al. 2003) mechanisms. Consequently, growth, recruitment, and abundance of native fishes are often negatively influenced by resource competition and predation from non-native fishes. A negative association between native and non-native fishes may also be a reflection of disturbance and habitat characteristics.

Increased occurrence of non-native fishes has been previously associated with habitat alteration in riverine systems throughout the western United States that are similar to the Kootenai River (Hughes and Gammon 1987; Richards 1997). Habitat alteration has been shown to facilitate the range expansion and abundance of non-native fishes because they are often readily adapted to the habitat conditions that emerge from disturbance (Macdougall and Turkington 2005; Light and Marchetti 2007). We observed higher relative abundances of non-native fishes in newly disturbed areas around rehabilitation treatments compared to non-rehabilitation areas. Our results support previous research, but also raise questions associated with expansion and colonization of non-native fishes in the Kootenai River. Although we observed higher occurrence and relative abundances associated with restoration areas in our study, these patterns emerged in the most downstream site in our study, which is closest to the meander section. The meander section is deep, has low current velocities, and less habitat heterogeneity than the braided section. The fish assemblage in the meander section is largely dominated by cyprinids and non-native fishes (Smith 2013), suggesting the potential for upstream colonization of newly disturbed habitats by non-native fishes. In addition, we found that the habitat-use patterns of non-native fishes reported by Smith (2013) are similar to existing habitat conditions in restored side-channel habitats (i.e., North Side Channel). It is therefore likely that the habitat conditions present in the North Side Channel are conducive to the persistence of non-native fishes that have already established viable populations in the meander section.

Results of this study showed that habitat associations were variable among taxa and related to the ecology of each species, fish assemblages varied among main- and side-channel habitats, and the North Side Channel displayed higher relative abundance of non-native fishes compared to the other sites. Several potential mechanisms may be related to the upstream colonization of non-native fishes into the braided reach and facilitate their expansion. The first of these may be the proximity of source populations in the meander section. Fish assemblage surveys previously conducted on the Kootenai River have indicated that catch rates of non-native fishes are higher in the meander section. The second potential mechanism is that rehabilitated habitats are more easily colonized by non-native fishes because they are often more tolerant of disturbance than native fish. We hypothesize, however, that the colonization of treatment areas may be related to a combination of these two mechanisms, which would best explain the patterns we observed. The patterns we observed have important implications for future rehabilitation projects where non-native fishes are present. Projects using habitat improvement as a management strategy to benefit native species must account for the potential expansion and colonization of non-native species and consider the biotic interactions that may result from range range expansion of non-native species. Carefully considering the potential effects outlined here will help managers to identify areas where habitat rehabilitation can be undertaken most responsibly while limiting externalities caused by range expansion of non-native species.

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TABLES

Table 2.1. Description of habitat variables collected from seven sampling sites in the Kootenai River, Idaho during the summers (May –September) of 2012 and 2013.

Variable	Description
Woody cover	Mean surface area of instream woody debris (> 0.2 × 0.5 m; m^2)
$\text{Substrate}_{\text{Fine}}$	Proportion of substrate consisting of fine particles (≤ 2 mm diameter; %)
$\text{Substrate}_{\text{Coarse}}$	Proportion of substrate consisting of coarse particles (≥ 64 mm diameter; %)
$\text{Bank}_{\text{Co-Bo}}$	Proportion of bank consisting of cobble and boulder (riprap; %)
Bank_{Veg}	Proportion of bank consisting of vegetation (%)
Velocity	Mean water column velocity (m^2/sec)
CV_{Vel}	Coefficient of variation of water column velocity (%)
Depth	Mean water column depth (m)
CV_{Depth}	Coefficient of variation of water column depth (%)

Table 2.2. Mean and standard error (in parentheses) of habitat variables measured at seven sampling sites in the Kootenai River, Idaho during the summers (May-September) of 2012 and 2013. Sites include North Side Channel (NSC), South Side Channel (SSC), Upper Side Channel (USC), Control Bend 1 (CB 1), Control Bend 2 (CB 2), Treatment Bend 1 (TB 1), and Treatment Bend 2 (TB 2).

	Site						
Variable	NSC	SSC	USC	CB 1	CB 2	TB 1	TB 2
Wood	48.47 (3.36)	12.08 (0.87)	34.21 (2.82)	3.43 (0.49)	0.00 (0.00)	0.00 (0.00)	64.62 (6.14)
$Substrate_{Fine}$	81.97 (1.63)	0.00 (0.00)	18.41 (1.79)	1.36 (0.07)	0.00 (0.00)	15.00 (1.82)	12.53 (1.74)
$Substrate_{Coarse}$	18.03 (1.61)	100 (0.00)	81.59 (1.79)	98.64 (0.10)	100 (0.00)	30.00 (1.49)	37.48 (1.56)
$Bank_{Co-Bo}$	0.00 (0.00)	10.42 (1.47)	23.64 (1.72)	8.93 (1.09)	28.67 (2.62)	4.67 (1.25)	15.01 (1.04)
$Bank_{Veg}$	57.92 (1.50)	87.51 (1.85)	45.00 (2.04)	81.07 (1.31)	71.33 (2.62)	21.33 (2.82)	47.50 (3.01)
Velocity	0.31 (0.01)	0.80 (0.02)	0.68 (0.01)	1.31 (0.02)	1.04 (0.03)	1.05 (0.02)	1.08 (0.02)
CV_{Vel}	44.64 (1.18)	35.95 (0.70)	38.76 (0.80)	32.04 (0.47)	42.65 (1.33)	37.72 (0.87)	50.99 (1.37)
Depth	1.92 (0.06)	1.50 (0.04)	1.48 (0.04)	2.67 (0.07)	3.35 (0.18)	2.64 (0.08)	2.84 (0.12)
CV_{Depth}	43.10 (0.98)	36.64 (0.93)	43.84 (1.12)	40.22 (0.98)	53.91 (1.12)	40.72 (1.41)	53.33 (1.87)

Table 2.3. Species composition (%) for each sampling site represented as the proportion of total catch for fishes sampled from the Kootenai River, Idaho during the summers (May-September) of 2012 and 2013. Sites include North Side Channel (NSC; 22 reaches; n = 1,908), South Side Channel (SSC; 12 reaches; n = 1,509), Upper Side Channel (USC; 22 reaches; n = 2,290), Control Bend 1 (CB 1; 15 reaches; n = 665), Control Bend 2 (CB 2; 15 reaches; n = 938), Treatment Bend 1 (TB 1; 15 reaches; n = 621), and Treatment Bend 2 (TB 2; 12 reaches; n = 362).

		Site						
Common name	Scientific name	NSC	SSC	USC	CB 1	CB 2	TB 1	TB 2
Cyprinidae								
Longnose dace	*Rhinichthys cataractae*	0.05	0.27	0.13	0.00	1.53	0.00	0.00
Northern pikeminnow	*Ptychocheilus oregonensis*	16.46	2.12	2.93	2.11	2.44	4.19	1.38
Peamouth	*Mylocheilus caurinus*	16.51	2.52	2.31	1.20	0.71	5.64	0.83
Redside shiner	*Richardsonius balteatus*	17.77	4.64	14.41	1.20	23.80	1.93	2.21
Catostomidae								
Largescale sucker	*Catostomus macrocheilus*	28.41	36.05	37.42	62.92	31.64	37.68	47.52
Longnose sucker	*Catostomus catostomus*	2.46	0.46	2.10	0.45	0.71	4.35	1.38
Ictaluridae								
Brown bullhead	*Ameiurus nebulosus*	0.05	0.00	0.09	0.00	0.10	0.00	0.00
Salmonidae								
Brown trout	*Salmo trutta*	0.05	0.20	0.00	0.30	0.00	0.16	0.00
Brook trout	*Salvelinus fontinalis*	0.00	0.07	0.00	0.00	0.10	0.16	0.00
Bull trout	*Salvelinus confluentus*	0.00	0.13	0.04	0.15	0.00	0.16	0.00
Westslope cutthroat trout	*Oncorhynchus clarki lewisi*	0.52	0.20	0.13	0.15	0.20	0.32	0.00
Kokanee	*Oncorhynchus nerka*	0.52	0.33	0.48	0.60	0.31	1.45	0.83
Mountain whitefish	*Prosopium williamsoni*	9.02	49.17	36.20	25.66	34.69	41.38	41.44
Redband trout	*Oncorhynchus mykiss*	1.52	2.58	2.75	3.16	1.53	2.09	2.76

Table 2.3 continued

Gadidae								
Burbot	*Lota lota*	0.00	0.13	0.00	0.15	0.00	0.00	0.00
Cottidae								
Torrent sculpin	*Cottus rhotheus*	0.26	1.00	0.87	1.95	0.61	0.32	1.66
Centrarchidae								
Largemouth bass	*Micropterus salmoides*	0.42	0.07	0.00	0.00	0.00	0.00	0.00
Pumpkinseed	*Lepomis gibbosus*	5.36	0.07	0.04	0.00	0.00	0.00	0.00
Percidae								
Yellow perch	*Perca flavescens*	0.63	0.00	0.00	0.00	0.00	0.00	0.00

Table 2.4. Top multiple-regression models for occurrence of fishes sampled from the Kootenai River, Idaho (2012-2013). Akaike's Information Criterion (AIC) was used as an indication of model rank and the negative log-likelihood (-*Log*(*l*)) is included. Direction of relationship between covariates and occurrence is indicated (negative [-], positive [+]).

Species	Model	AIC	ΔAIC	-*Log*(*l*)
Largescale sucker	-Depth	1328.4	0.00	-659.22
	-Depth, +CV_{Vel}	1330.4	2.00	-659.22
Mountain whitefish	+$Substrate_{Coarse}$, +$Bank_{Veg}$, +CV_{Vel}, -Depth, -Wood	1213.8	0.00	-598.93
	+$Substrate_{Coarse}$, +$Bank_{Veg}$, +CV_{Vel}, -Depth	1215.5	1.17	-598.77
Northern pikeminnow	+Depth, -CV_{Vel}	703.6	0.00	-345.81
	+Depth, -CV_{Vel}, +$Bank_{Veg}$	704.9	1.30	-345.44
Peamouth	+$Substrate_{Fine}$, +$Bank_{Veg}$, -CV_{Vel}	554.0	0.00	-269.99
Redband trout	+Wood, +$Substrate_{Coarse}$, +$Bank_{Veg}$, +CV_{Vel}, -Depth	884.0	0.00	-432.10
Redside shiner	-CV_{Vel} +$Bank_{Veg}$, +Depth, +$Substrate_{Fine}$	452.3	0.00	-218.14
	-CV_{Vel}, +$Bank_{Veg}$	454.2	1.90	-221.08
Torrent sculpin	+$Bank_{Co\text{-}Bo}$, +CV_{Vel}	355.8	0.00	-171.89

Table 2.5. Top multiple-regression models for relative abundance of fishes sampled from the Kootenai River, Idaho (2012-2013). Akaike's Information Criterion (AIC) was used as an indication of model rank. The negative log-likelihood (-*Log*(*l*)), α (dispersion parameter), and root mean squared error (RMSE) are included. Direction of relationship between covariates (Table 1) and relative abundance is indicated (negative [-], positive [+]).

Species	+Model	AIC	ΔAIC	-*Log*(*l*)	α	RMSE
Largescale sucker	+$\text{Substrate}_{\text{Coarse}}$, -Depth	4036.90	0.00	-2011.46	1.70	2.39
	+$\text{Substrate}_{\text{Coarse}}$, -Depth, +$\text{CV}_{\text{Vel}}$, +$\text{Bank}_{\text{Veg}}$	4037.90	1.00	-2009.94	1.71	2.40
Mountain whitefish	-Wood, +$\text{Substrate}_{\text{Coarse}}$, +$\text{Bank}_{\text{Veg}}$, +$\text{CV}_{\text{Vel}}$, -Depth	4389.30	0.00	-2184.67	2.23	2.51
Northern pikeminnow	+Depth	1034.90	0.00	-511.46	0.35	1.50
	+Depth, +CV_{Vel}	1036.70	1.80	-511.36	0.36	1.50
Peamouth	+$\text{Substrate}_{\text{Fine}}$, +Wood	1014.40	0.00	-500.21	0.70	1.87
	+$\text{Substrate}_{\text{Fine}}$, -$\text{CV}_{\text{Vel}}$	1015.50	1.10	-500.73	0.69	1.88
Redband trout	+$\text{Substrate}_{\text{Coarse}}$, +$\text{CV}_{\text{Vel}}$	1064.60	0.00	-525.29	3.38	0.43
Redside shiner	+$\text{Substrate}_{\text{Fine}}$, -$\text{CV}_{\text{Vel}}$	849.50	0.00	-417.77	0.22	6.57
Torrent sculpin	+$\text{Bank}_{\text{Co-Bo}}$	472.50	0.00	-230.27	0.47	0.33
	+$\text{Bank}_{\text{Co-Bo}}$, +$\text{CV}_{\text{Vel}}$	474.50	2.00	-230.26	0.48	0.33

Table 2.6. Parameter estimates, standard error (SE), and lower and upper limits for 95% confidence intervals derived from the most parsimonious model developed to identify environmental variables predicting occurrence of species sampled from the Kootenai River, Idaho (2012-2013).

				95% Confidence Interval	
Species	Model parameters	Estimate	SE	Lower	Upper
Largescale sucker	Depth	-0.257	0.282	-0.434	-0.079
Mountain whitefish	Wood	0.002	0.003	-0.005	0.009
	$Substrate_{Coarse}$	0.026	0.004	0.019	0.034
	$Bank_{Veg}$	-0.006	0.005	-0.015	0.003
	CV_{Vel}	-0.005	0.007	-0.019	0.009
	Depth	-0.243	0.088	-0.416	-0.070
Northern pikeminnow	Depth	0.209	0.097	0.020	0.399
	CV_{Vel}	0.007	0.009	-0.010	0.025
Peamouth	$Substrate_{Fine}$	0.026	0.006	0.015	0.037
	$Bank_{Veg}$	-0.002	0.005	-0.013	0.008
	CV_{Vel}	0.689	0.569	-0.426	1.805
Redband trout	Wood	0.007	0.585	0.002	0.013
	$Substrate_{Coarse}$	0.010	0.004	0.003	0.018
	$Bank_{Veg}$	-0.009	0.004	-0.017	-0.001
	CV_{Vel}	0.003	0.007	-0.012	0.017

Table 2.6 continued

	Depth	-0.155	0.093	-0.337	0.027
Redside shiner	$\text{Substrate}_{\text{Fine}}$	0.025	0.009	0.007	0.042
	Bank_{Veg}	0.037	0.014	0.011	0.064
	CV_{Vel}	0.034	0.014	0.007	0.061
	Depth	0.010	0.178	-0.337	0.358
Torrent sculpin	$\text{Bank}_{\text{Co-Bo}}$	0.014	0.007	0.001	0.027
	CV_{Vel}	0.007	0.012	-0.017	0.031

Table 2.7. Parameter estimates, standard error (SE), and lower and upper limits for 95% confidence intervals derived from the most parsimonious model developed to identify environmental variables predicting relative abundance of species sampled from the Kootenai River, Idaho (2012-2013).

				95% Confidence Interval	
Species	Model parameters	Estimate	SE	Lower	Upper
Largescale sucker	$Substrate_{Coarse}$	0.007	0.002	0.003	0.011
	Depth	-0.178	0.051	-0.277	-0.078
Mountain whitefish	Wood	0.001	0.002	-0.003	0.004
	$Substrate_{Coarse}$	0.021	0.002	0.016	0.025
	$Bank_{Veg}$	-0.001	0.002	-0.005	0.004
	CV_{Vel}	-0.002	0.003	-0.008	0.004
	Depth	-0.164	0.047	-0.257	-0.071
Northern pikeminnow	Depth	0.246	0.105	0.040	0.452
Peamouth	Wood	-0.003	0.003	-0.010	0.003
	$Substrate_{Fine}$	0.020	0.004	0.013	0.027
Redband trout	$Substrate_{Coarse}$	0.010	0.003	0.004	0.016
	CV_{Vel}	-0.144	0.080	-0.301	0.012
Redside shiner	$Substrate_{Fine}$	0.001	0.013	-0.024	0.026
	CV_{Vel}	-2.912	1.024	-4.921	-0.904

Table 2.7 continued

Torrent sculpin	$Bank_{Co\text{-}Bo}$	0.015	0.007	0.001	0.029

FIGURES

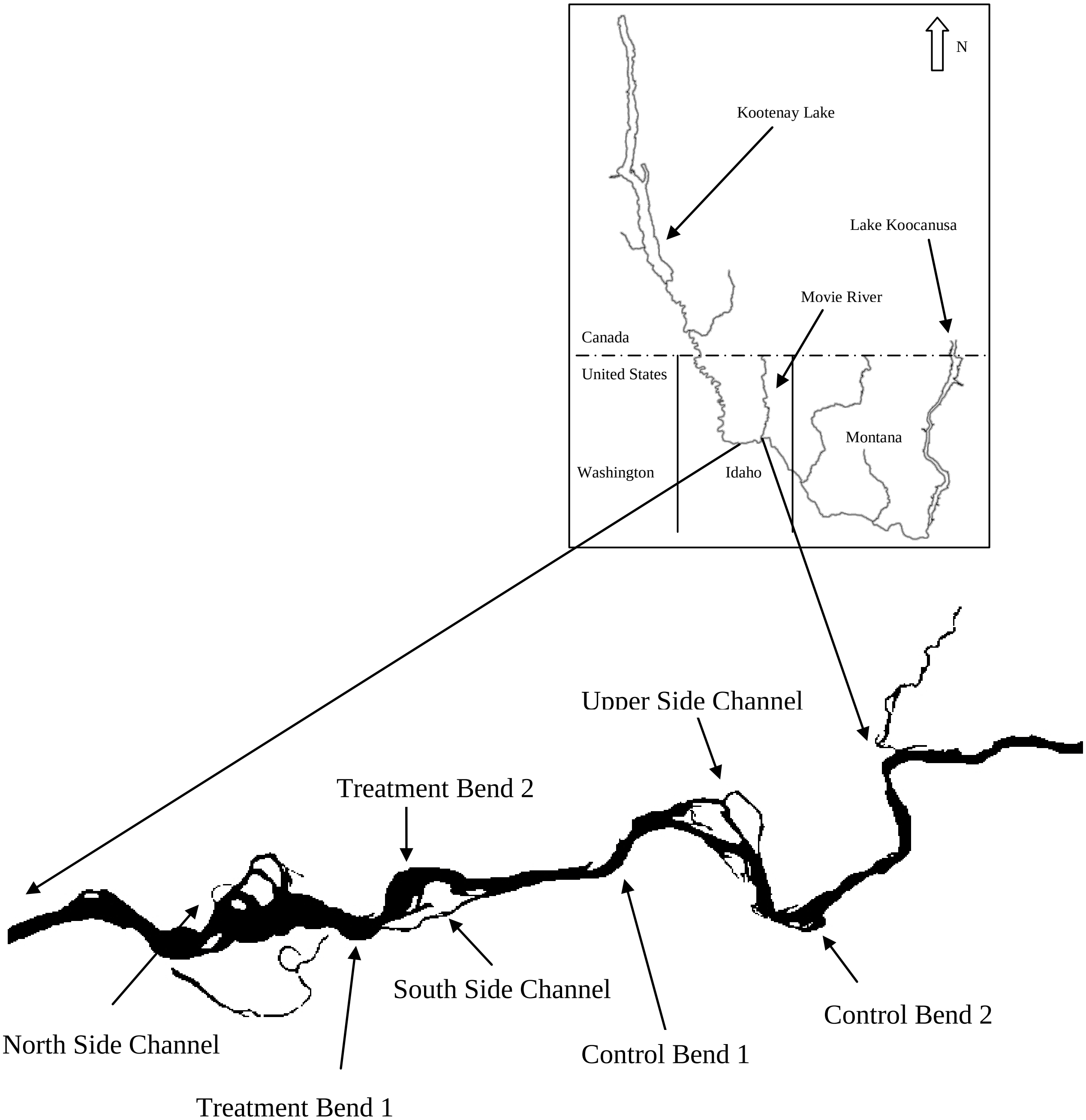
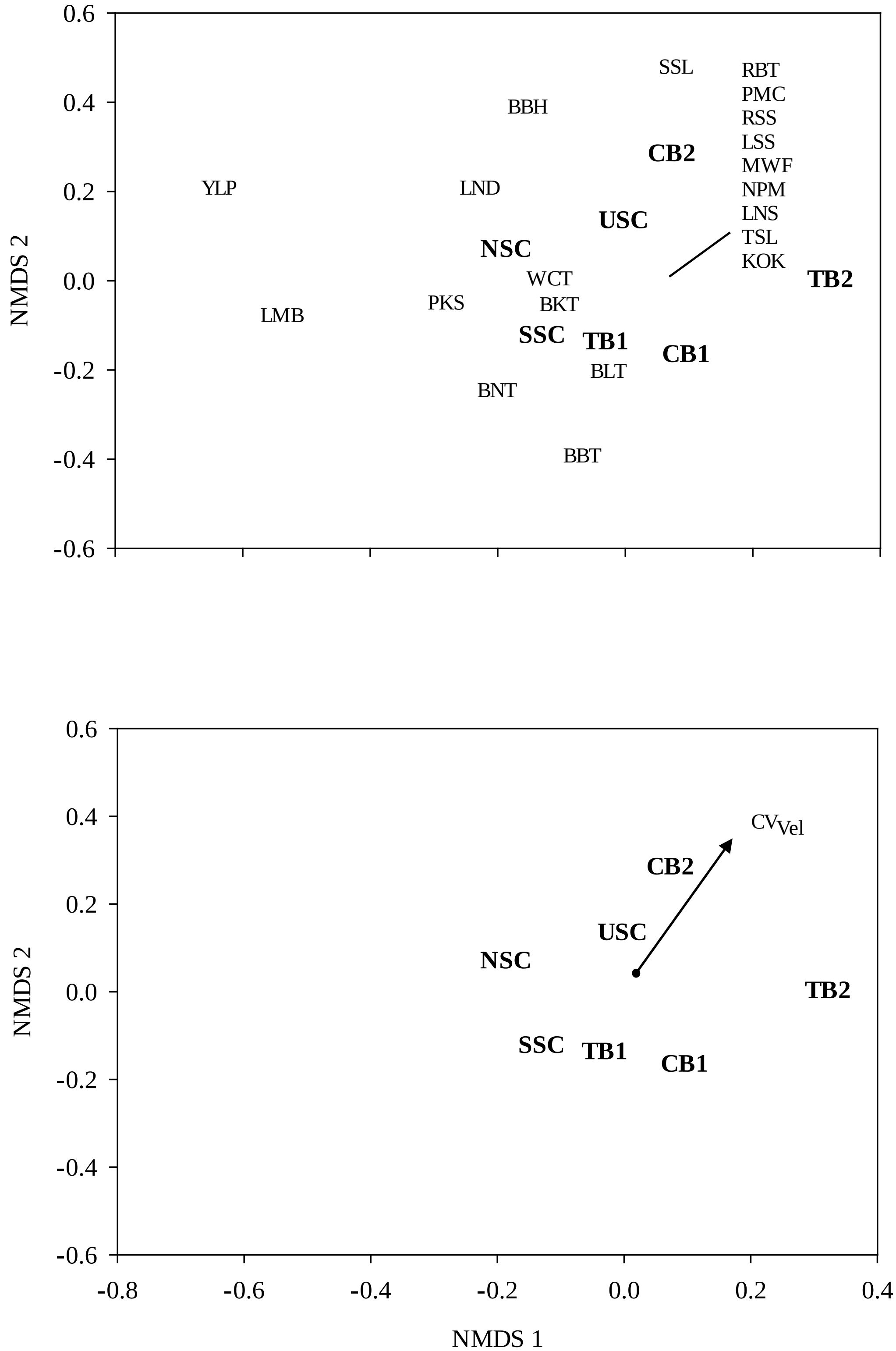


Figure 2.1



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SSL
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TSL
KOK
BBH
CB2
YLP
LND
USC
NSC
TB2
WCT
PKS
BKT
LMB
SSC
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CB1
BLT
BNT
BBT
NMDS 2
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CV$_{Vel}$
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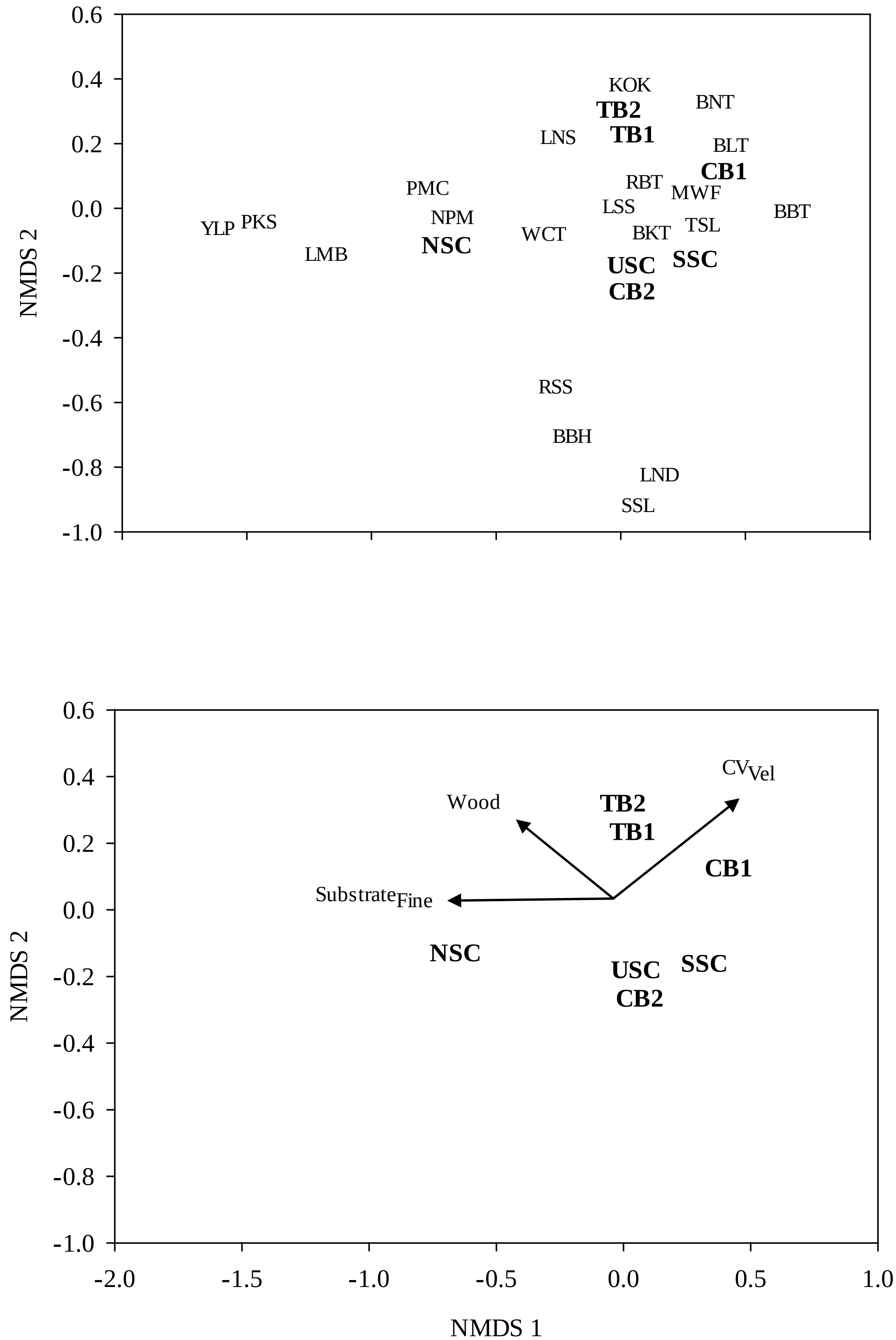
NMDS 1

Figure 2.2.



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NMDS 2
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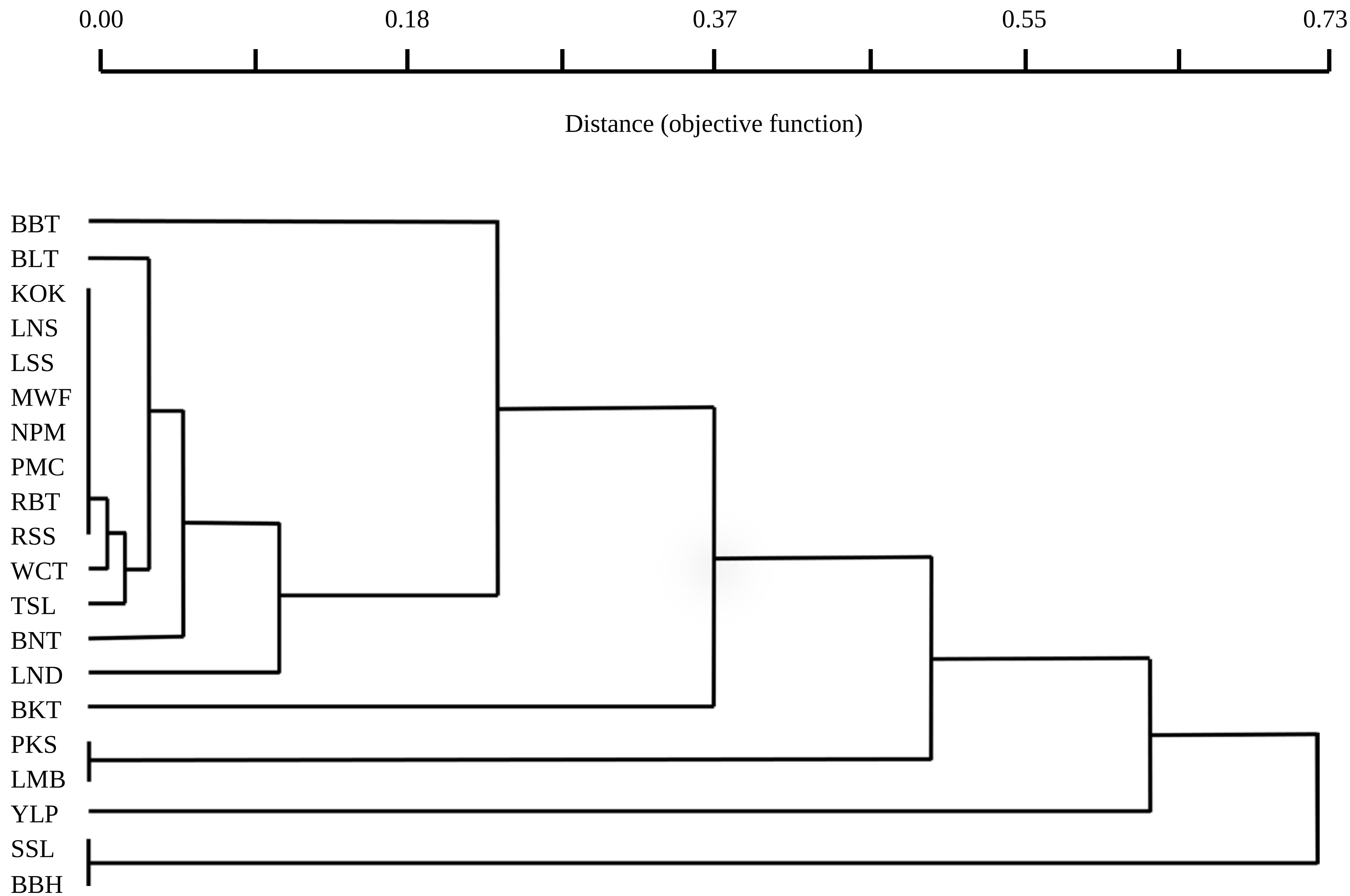
Figure 2.3.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LND
BKT
PKS
LMB
YLP
SSL
BBH

Figure 2.4.

Figure 2.1. Map of the Idaho portion of the Kootenai River. The braided section is located between the confluence of the Moyie River and the town of Bonners Ferry, Idaho.

Figure 2.2. Nonmetric multidimensional scaling (NMDS) ordination (stress = 4.7) of fish assembalge structure using occurrence data from seven sites in the Kootenai River, Idaho during the summers (May-September) of 2012 and 2013. Site and species scores are provided in the top panel and signifcant ($P \leq 0.05$) habitat vectors are shown in the lower plot. Sites are indicated by bold font and include North Side Channel (NSC), South Side Channel (SSC), Upper Side Channel (USC), Control Bend 1 (CB 1), Control Bend 2 (CB 2), Treatment Bend 1 (TB 1), and Treatment Bend 2 (TB 2). Taxa indicated in the top panel include burbot (BBT), brown bullhead (BBH), brook trout (BKT), bull trout (BLT), brown trout (BNT), kokanee (KOK), largemouth bass (LMB), longnose dace (LND), longsnose sucker (LNS), largescale sucker (LSS), mountain whitefish (MWF), northern pikeminnow (NPM), pumpkinseed (PKS), peamouth (PMC), redband trout (RBT), redside shiner (RSS), Cottidae (TSL, SSL), westslope cutthroat trout (WCT), and yellow perch (YLP). Descriptions of envrionmental variables fit to NMDS ordinations can be found in Table 1.

Figure 2.3. Nonmetric multidimensional scaling ordination (stress = 2.1) of fish assembalge structure using catch rate information from seven sites in the Kootenai River, Idaho during the summers (May-September) of 2012 and 2013. Site and species scores are provided in the top panel and signifcant ($P \leq 0.05$) habitat vectors are shown in the lower plot. Sites are indicated by bold font and include North Side Channel (NSC), South Side Channel (SSC), Upper Side Channel (USC), Control Bend 1 (CB 1), Control Bend 2 (CB 2), Treatment Bend 1 (TB 1), and Treatment Bend 2 (TB 2). Taxa indicated in the top panel include burbot (BBT), brown bullhead (BBH), brook trout (BKT), bull trout (BLT), brown trout (BNT), kokanee (KOK), largemouth bass (LMB), longnose dace (LND), longsnose sucker (LNS), largescale sucker (LSS), mountain whitefish (MWF), northern pikeminnow (NPM), pumpkinseed (PKS), peamouth (PMC), redband trout (RBT), redside shiner (RSS), Cottidae (TSL, SSL), westslope cutthroat trout (WCT), and yellow perch (YLP). Descriptions of envrionmental variables fit to NMDS ordinations can be found in Table 1.

Figure 2.4. Dendrogram illustrating reach similarity among all fish species sampled from the Kootenai River, Idaho during the summers (May-Spetember) of 2012 and 2013. Taxa included include burbot (BBT), brown bullhead (BBH), brook trout (BKT), bull trout (BLT), brown trout (BNT), kokanee (KOK), largemouth bass (LMB), longnose dace (LND), longsnose sucker (LNS), largescale sucker (LSS), mountain whitefish (MWF), northern pikeminnow (NPM), pumpkinseed (PKS), peamouth (PMC), redband trout (RBT), redside shiner (RSS), Cottidae (TSL, SSL), westslope cutthroat trout (WCT), and yellow perch (YLP).

CHAPTER 3: SAMPLING EFFORT REQUIREMENTS FOR ESTIMATING SPECIES RICHNESS IN THE KOOTENAI RIVER, IDAHO

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ABSTRACT

This study was conducted on the Kootenai River, Idaho to provide insight on sampling requirements to optimize future monitoring efforts associated with the response of fish assemblages to habitat rehabilitation. Our objective was to define the effort needed to have a 95% probability of sampling 50, 75, and 100% of the observed species richness and to evaluate the relative influence of depth, velocity, and instream woody cover on sample size requirements. Side-channel habitats required more sampling effort to achieve 75 and 100% of the total species richness. The sampling effort required to have a 95% probability of sampling 100% of the species richness was 1,100 m for main-channel sites and 1,400 m for side-channel sites. We hypothesized that the difference in sampling requirements between main- and side-channel habitats was largely due to differences in habitat characteristics and species richness between main- and side-channel habitats. In general, main-channel habitats had lower species richness than side-channel habitats. Habitat characteristics (depth, current velocity, and woody instream cover) were not related to sample size requirements. Our guidelines will provide insight on sampling requirements for future monitoring efforts in the Kootenai River and other large western river systems.

INTRODUCTION

Optimizing fish sampling efforts to save time and resources is of high interest to fisheries scientists, particularly those focused on fish assemblages in large rivers (Flotemersch et al. 2011). Fisheries scientists are commonly interested in how a management action might affect the occurrence of species. Achieving such objectives requires information on fish assemblage structure and species occurrence to monitor responses to management (Moerke and Lamberti 2003). Knowledge of how much effort is required to detect a given level of species richness can help reduce the amount of time spent sampling, while also providing the information needed to evaluate potential changes in species composition. Guidelines for sampling fish assemblages in large western rivers are lacking. However, guidelines are necessary to improve efficiency while maintaining meaningful and comparable datasets for monitoring and evaluating the influence of management activities on fish assemblages.

Considerable effort has focused on understanding sample size requirements for systems throughout North America. Sample size requirements for estimating species richness have been studied among a variety of lentic (Bailey et al. 2005; Quist et al. 2007) and lotic (Patton et al. 2000; Fischer and Paukert 2009; Paller 1995; Flotemersch et al. 2011) habitats in several geographic regions. Much of the previous research has focused on standing waters and small streams, but relatively little work has been conducted on large rivers, particularly in the western United States. Neebling and Quist (2011) investigated sample size requirements to detect various levels of species richness with different gear combinations in large rivers in Iowa. Similarly, Flotemersch and Blocksom (2005) evaluated the influence of sampling design and sample size on reach lengths needed to estimate assemblage metrics; however, these studies are limited to rivers of the midwestern United States.

Given the importance and interest in gaining information on large river fish assemblages, development of sampling guidelines is critical (Patton et al. 2000). Management actions, such as habitat rehabilitation, are occurring in many large rivers throughout the western United States and such projects are tasked with monitoring the effects of management actions (Moerke and Lamberti 2003; Schloesser et al. 2011; Romanov 2012). Monitoring in most systems focuses on determining if management actions elicit responses from fish assemblages. As such, statistically rigorous sampling designs are needed to provide information that fishery managers can confidently use to assess these actions. Knowledge of species occurrence allows managers to prioritize future projects and better understand the factors structuring fish assemblages.

The Kootenai River is a large floodplain river that originates in British Columbia, Canada and flows through the panhandle of northern Idaho. The portion of the Kootenai River in Idaho is characterized by a prominent floodplain that historically provided a substantial amount of flooded terrestrial habitat during high runoff events; however, many flood-related habitats (e.g., side channels, sloughs, oxbows) in the Kootenai River have been eliminated as a result of instream and shoreline development. The Kootenai River has been extensively altered by anthropogenic disturbance from water development and land use activities (Knudsen 1993). Native fishes in the Kootenai River, some of which provide important fisheries, have declined in abundance from historic levels, especially white sturgeon *Acipenser transmontanous* (Paragamian 2012), burbot *Lota lota* (Paragmian and Hansen 2010), and kokanee *Oncorhynchus nerka* (Apperson 1990). The completion of

Libby Dam and construction of levees have been implicated as causes for the decline of fish populations over the past several decades (Paragamian 2012). In addition to habitat loss from water development, Libby Dam has altered the historic flow, nutrient, and thermal regimes which have negatively influenced native fishes in the lower Kootenai River. While the headwaters of the Kootenai River are relatively intact, the lower portion of the Kootenai River ecosystem functions quite differently than it did historically (Knudsen 1993). The once large and prominent floodplain that extended throughout the lower Kootenai River has since been replaced by a deep, incised main channel with little floodplain connectivity. Managers of the Kootenai River have recently instituted a large-scale habitat rehabilitation project focused on improving environmental conditions for native fishes and mitigating for water development (KTOI 2009). Primary objectives of the project are to improve and(or) create side-channel habitat that is scarce along a developed floodplain, and to improve degraded habitat in the main channel (KTOI 2009; KTOI 2012). Habitat rehabilitation has been used successfully in many systems to enhance the abundance, growth, and reproduction of target fish species (Moerke and Lamberti 2003; Romanov et al. 2012). Habitat rehabilitation projects have become a popular means for restoring native fish assemblages and have been shown to bring about change in fish assemblage and population structure (Binns 2003; Romanov et al. 2012). However, designs for monitoring and evaluating the success of habitat rehabilitation are often ignored. Furthermore, sampling designs for monitoring changes in fish assemblages, particularly in large western river systems, is lacking in the primary literature. Evaluating the amount of sampling effort needed to detect different levels of assemblage and population indices can improve the efficiency of projects aimed at monitoring trends in fish assemblages and species occurrence

(Lyons 1992; Bayley and Dowling 1993; Flotemersch et al. 2011). Oftentimes, trends in occurrence of native and non-native fish can inform managers about the expansion or contraction of a species' distribution at small (e.g., river segment; Bayley et al.1993; Angermeier and Smoger 1995) and large scales (e.g., drainage basin; Jennings et al. 1995). Information on occurrence can also be used to identify patterns of colonization and extirpation in relation to management actions.

Managers of the Kootenai River are tasked with the responsibility of conserving native fishes and evaluating the influence of management actions on fish assemblages. Given the onset of a large-scale habitat rehabilitation project occurring in the lower Kootenai River, evaluating management actions is more important than ever. Large river systems usually possess habitat characteristics that are different than habitat in small streams, where most sample size investigations have previously been conducted. Other studies have shown that electrofishing is both size and species selective (Shoenebeck and Hansen 2005; Reynolds and Kolz 2012). Electrofishing selectivity is further complicated by the fact that catchability is often related to habitat characteristics (Rogers et al. 2003). Given these sampling issues, understanding the outcomes of sampling effort and the relative influence of habitat has ramifications for monitoring and developing effective sampling designs. As such, the purpose of this study was to determine the number of 100-m long sampling reaches required to estimate various levels of species richness. Additionally, we sought to provide insight on how sampling requirements are related to habitat characteristics in a large western river system.

METHODS

Site selection

The Idaho portion of the Kootenai River is delineated into three distinct sections: canyon, braided, and meander sections (Fossness and Williams 2009). The braided section (246-257 river kilometer) was the focus of our study and extends from the confluence of the Moyie River downstream to Bonners Ferry, Idaho (Figure 3.1). Sampling occurred at seven sites within the braided section of the Kootenai River during the summer (May-September) of 2013. Sites were designated as either side-channel or main-channel (i.e., existing only in the main channel). Because a major focus of this study was to collect information from sites that would undergo habitat rehabilitation in the future, we selected sites based on planned rehabilitation. Main- and side-channel habitats were treated as two separate strata for this study based on inherent differences in habitat characteristics. Three side-channel complexes and four main-channel meanders were designated as sampling sites (hereafter referred to as sites). Each side-channel site was then divided into individual 100-m long reaches (hereafter referred to as reaches) along the thalweg. Each main-channel site was divided into 100-m long reaches along both the inside and outside bend. Reaches in side-channel sites were treated as segments containing both banks while reaches in main-channel sites were considered as either the inside or outside bend. Reach lengths of 100 m allowed us to easily enumerate the total distance required during subsequent analyses. A handheld global positioning system was used to georeference the upper and lower terminus of each reach and florescent flagging was placed along the bank to identify the spatial extent of each reach during sampling.

Fish sampling

All reaches were sampled using pulsed-DC boat-mounted electrofishing on a biweekly schedule to account for temporal variability in fish assemblage structure. Previous studies have shown that electrofishing is an effective sampling technique for riverine fishes and is commonly used by natural resource agencies conduct fish surveys (Reynolds and Kolz 2012). As such, sampling was conducted during the day with a boat-mounted electrofisher composed of an Infinity model box (Midwest Lake Electrofishing Systems, Inc., Polo, Missouri) and a 5,000-W generator (American Honda Motor Co., Torrance, California). Electrofishing power output was standardized to 3,000 W based on ambient water conductivity and water temperature following Miranda (2009). Two netters were stationed at the bow of the boat to collect immobilized fish during sampling. For side-channel sites, electrofishing effort began at the most upstream reach within each site, and a single pass was allocated to both the right and left banks of each reach before proceeding to the next reach. Electrofishing began at the uppermost reach in each main-channel bend and a single pass was conducted along the bank of each reach before proceeding to the subsequent reach. Upon completion of sampling in each reach, fishes were identified to species, measured (total length; mm), and released. Data collected from reaches along the inside and outside bends of main-channel sites were treated independently in all analyses, but were nested within their respective sites. Similarly, data collected from reaches in side-channel sites were nested within their respective site. After processing, all fish were released in a location away from subsequent sampling reaches to minimize the influence of emigration back into the sampling area.

Habitat sampling

Environmental variables were measured to evaluate the influence of abiotic habitat characteristics on the effort required to attain various proportions of total species richness. Woody debris was measured as the total surface area (m^2) of woody instream cover in each sampling reach. Only woody debris particles greater than 0.20 m in diameter and 0.50 m in length were measured. Estimates of mean water column velocity and depth were obtained from the River Design Group (RDG; River Design Group, Inc., Whitefish, Montana). The RDG has compiled channel morphology and flow data from throughout the braided reach of the Kootenai River to monitor the response of habitat within rehabilitation areas. Water hydraulics were simulated using the flow and sediment transport with morphological evolution of channels hydraulic flow model (FaSTMECH; Nelson 1996). The model computes flow-field hydraulics on a curvilinear fitted grid by solving the depth-averaged shallow-water equations using an eddy viscosity turbulence closure based on a dimensionless drag coefficient. The two-dimensional hydraulic model geometry was sampled at a nominal 3-m resolution from bathymetric data collected by the U.S. Geological Survey (USGS) in 2009 and merged into 2010 the terrestrial light detection and ranging model radar (LiDAR). Hydraulic roughness in the two-dimensional model was spatially varied as a function of depth using a nominal roughness height in the gravel to cobble substrate range. The two-dimensional hydraulic model was calibrated to a range of measured stage data from a detailed gage network from water years 2010 and 2011 (Czuba et al. 2011). Two-dimensional model results were post-processed in ArcGIS (ArcGIS 2000, Esri, Redlands, California) to develop mean depth, mean velocity, associated variances, and shear stress grids for flow conditions that occurred during each fish sampling event.

Data analysis

Species richness was calculated for each site by month during the field season as the total number of species detected. Two sampling events were conducted during each month of the summer (May-September). The sampling event during each month with the highest standard deviation in species richness was used for subsequent analyses. This allowed for the most conservative estimate of required effort. The effort required to sample 50, 75, and 100% of the species captured in each site (species richness) with 95% confidence was estimated during each month of the field season using a re-sampling procedure. Monte Carlo simulations were used to resample random combinations of reaches within each site (Manly 1991). Simulations were performed to randomly sample (without replacement) combinations of reaches (representing sampling effort) within each site (Patton et al. 2000; Quist et al. 2007; Neebling and Quist 2011). One thousand iterations were performed for each sample size and the total number of species was recorded for each iteration. We assumed that at least two reaches (i.e., 200 m of thalweg [side channels] or shoreline [main channel] distance) could be sampled by fisheries scientists with ease; this was a reasonable assumption given the nature of sampling in large river systems. For each sample size, the number of times out of 1,000 replicates that a given species richness was attained was recorded. This provided raw probabilities for detecting various levels of species richness with each sample size. The mean minimum number of reaches required to sample a given level of species richness (i.e., 50, 75, or 100%) with a 95% probability. For instance, 12 reaches were sampled during May from the South Side Channel (SSC) and nine species were captured. For a sample size of two reaches, two reaches were repeatedly sampled 1,000 and the number of species from each iteration was recorded. This process was repeated up to the maximum sample size (*n* = 12). All simulations were performed using Program R (R Development Core Team 2012).

The number of stream reaches was converted to a measure of distance in meters (measured as the total thalweg [side-channel sites] or shoreline distance [main-channel sites]) required to sample given target levels of species richness (Bayley and Dowling 1993; Patton et al. 2000). Simple linear regression was used to evaluate the influence of environmental characteristics on the distance required to estimate species richness. The total distance required to sample 100% of the species present in a site with 95% confidence was converted to a proportion of the total site length and used as the response variable in regression analyses. We investigated woody cover, velocity, and coefficient of variation (CV) in depth as potential explanatory variables influencing the proportion of each site required to sample 100% of the species.

RESULTS

We sampled 112 reaches comprising seven sites during the summer of 2013, and each site was sampled 10 times. Side-channel sites varied in thalweg length from 1,200-2,200 m and contained 56 reaches, whereas main-channel sites varied in length from 1,200-1,500 m in shoreline length and contained 57 reaches (Table 3.1). Two side-channel sites were 2,200 m in thalweg length and one side-channel site was 1,200 m in thalweg length. Two main-channel sites were 1,500 m in total shoreline length, one was 1,400 m long, and another was 1,200 m long. Mean water velocity and depth were higher in main-channel sites than side-channel sites (Table 3.1). Mean area of woody cover was generally higher in sites that had undergone habitat rehabilitation (i.e, North Side Channel, Upper Side Channel, Treatment Bend 2).

We sampled 20 species representing eight families among the seven sites (Table 3.2) and sampled between 4-12 species at each reach across all months. Largescale sucker (scientific names provided in Table 3.2) and mountain whitefish were the most common species and the only species detected at every sampling reach throughout the summer; cumulatively, largescale sucker and mountain whitefish composed 66% of the total catch. Longnose sucker, northern pikeminnow, peamouth, redband trout, redside shiner, and Cottidae were detected at every site, but not every reach. Furthermore, although these species were detected at every reach, they were not among the most abundant, making up only 25% of the total catch by number. Estimates of effort required to achieve target levels of species richness varied by site and by season. Species richness was generally higher for side-channel sites than for main-channel sites (Table 3.3; Figure 3.2). In addition, species richness increased during the late summer (August and September) in side-channel sites, whereas main-channel sites had higher species richness during the early summer (May and June).

The amount of sampling effort required to sample various percentages of species richness varied by site and month. In general, sites with fewer reaches required less sampling effort to achieve all target levels of species richness during all months (Table 3.3). Clear patterns emerged among the number of species detected and the distance required to sample various levels of species richness. As would be expected, the number of reaches required to have a 95% probability of sampling 100% of the total number of species generally increased as more species were detected. This was true for most sampling sites, with the exception of the North Side Channel which displayed the opposite pattern (Table 3.3). For example, in the South Side Channel during May, nine species were sampled with electrofishing and required an average of 354 m to sample 50%, 698 m to sample 75%, and 1,019 m to sampled 100% of the total estimated species richness (Figure 3.3). For the South Side Channel during June, the required number of reaches required increased to 399 m for 50% of species richness, decreased to 618 m for 75%, and increased slightly to 1,100 m for 100%.

Regression modeling exercises showed little support for environmental covariates hypothesized to influence the proportion of site length required to sample 100% of the estimated species richness. The proportion of site length required to sample 100% of the species was inversely related to CV in depth (R^2 = 0.14, P = 0.03) and surface area of instream woody debris (R^2 = 0.14, P = 0.02); the relationship was negative for both variables. While both the CV in depth and area of woody debris were negatively related to distance required to sample all presumably present species, the relationship was not well-supported.

DISCUSSION

Information regarding species occurrence is of importance to fisheries biologists because it allows them to monitor temporal and spatial trends in the distribution of fishes and can provide insight on the response of fish assemblages to management actions or environmental change. In addition, information on the occurrence of non-native fishes has increased in importance as it can provide insight on colonization rates and range expansion of undesirable species. Obtaining accurate information on species occurrence and species richness requires sufficient sampling effort to ensure that all species are captured and that sampling is efficient.

Overall, the sites sampled in our study displayed variation in monthly species richness and habitat. The amount of sampling effort required to estimate various levels of species richness varied among main- and side-channel reaches, but patterns in effort required to obtain 50, 75, and 100% of the estimated species richness was similar for all months during the field season. Other studies focused on evaluating species richness have focused on small streams or standing waters in the eastern or midwestern United States. Consequently, many studies have developed guidelines using sampling gears that are not appropriate for large rivers (e.g., seining or backpack electrofishing). Neebling and Quist (2011) evaluated sample size requirements to estimate species richness using boat-mounted electrofishing in Iowa rivers. The authors reported that 2,500 m were required to have a 95% probability of sampling 100% of the species richness. We did not observe a similar pattern in our study. In fact, we found that the distance required to sample 100% of the total species richness varied from 425-1,818 m among all sites and months. More specifically, required effort for main-channel sites varied from 425-1,286 m and from 975-1,818 m for side-channel sites. The efficacy of using both proportional- and fixed-distance designs for estimating species richness in large rivers has previously been evaluated (Lyons 1992; Flotemersch and Blocksom 2005). Given the former design, our results tend to differ from Neebling and Quist (2011) where the proportion of site length required to estimate species richness in Iowa rivers was approximately 50% of the original site length. Our results suggest that 35-86% of the original site length is required to sample 100% of the species richness in main-channel sites, and 81-86% of the site length is required in side-channel sites. Therefore, in regard to proportion of the total original site length required to estimate species richness, our results tend to be higher than has been reported in other large river systems. The higher proportion of site length required in our study may reflect the presence of rare species that are patchily-distributed throughout the Kootenai River.

Western rivers are typically depauperate compared to systems in the midwestern and eastern United States. Moreover, fish assemblages in regulated rivers are generally dominated by relatively few, generalist species (Paragamian 2002). The Kootenai River exhibits a similar pattern where the fish assemblage is dominated by a few species. Smith (2013) found that fish assemblage structure and species composition varied throughout the entire lower Kootenai River, and that species richness was highest in the braided section. We found that the fish assemblage in the braided section was largely composed of a few species; however, most species tended to be rare and occurred only in localized areas with suitable habitat. We argue that the high amount of proportional sampling effort required to estimate species richness was an artifact of patchily-distributed rare species.

Previous studies have found that habitat is related to sampling effort requirements. Patton et al. (2000) reported that stream width and stream area were inversely related to the number of reaches required to sample 90% of the species in small southeastern Wyoming streams. Our results suggested that habitat was not associated with the sample size requirements. Although the Kootenai River is composed of relatively homogeneous habitat with little complexity, similar to streams sampled by Patton et al. (2000), we did not observe similar patterns. However, in general, habitat in regulated rivers tends to be highly homogeneous (Nilsson et al. 2005) while small streams are often unregulated and possess more habitat complexity (Angermeier and Smoger 1995).

Our results suggest that stratification of sampling effort based on habitat characteristics may not be warranted during future monitoring efforts. We found that the relationship between instream woody cover, velocity, and depth was statistically weak and that these variables had little association with sample size requirements. Additionally, our results suggest that sampling 1,800 m of thalweg corresponded to a 95% probability of sampling 100% of the observed species richness in side-channel sites. For main-channel sites, sampling 1,400 m of shoreline produces a 95% probability of sampling 100% of the observed species richness. In general, these samples sizes will provide the highest confidence estimates of species richness and avoid unnecessary sampling. In most cases, the most conservative level of sampling effort required for 100% of species richness was close to the total length of each site, particularly in side-channel sites which required more effort than main-channel sites. However, the guidelines provided here will allow mangers of the Kootenai River to conduct future sampling efforts with reasonable certainty of species richness estimates among main- and off-channel environments. This study is the first of its kind for large coldwater rivers, but future work should focus on similar analyses in other large rivers characteristic of the western United States.

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TABLES

Table 3.1. Channel type, total thalweg or shoreline length, mean velocity, mean depth, and mean surface area of woody cover in sampling sites in braided section of the Kootenai River, Idaho sampled during the summer (May-September) of 2013. Standard errors for mean habitat variables are provided in parentheses. Sites include NSC = North Side Channel, SSC = South Side Channel, USC = Upper Side Channel, CB 1 = Control Bend 1, CB 2 = Control Bend 2, TB 1 = Treatment Bend 1, TB 2 = Treatment Bend 2.

Site name	Length (m)	Velocity (m^2/sec)	Depth (m)	Woody cover (m^2)
		Side channel		
NSC	2200	0.31 (0.01)	1.92 (0.06)	48.47 (3.36)
SSC	1200	0.80 (0.02)	1.50 (0.04)	12.08 (0.87)
USC	2200	0.68 (0.01)	1.48 (0.04)	34.21 (2.82)
		Main channel		
CB 1	1500	1.31 (0.02)	2.67 (0.07)	3.43 (0.49)
CB 2	1500	1.04 (0.03)	3.35 (0.18)	0.00 (0.00)
TB 1	1500	1.05 (0.02)	2.64 (0.08)	0.00 (0.00)
TB 2	1200	1.08 (0.02)	2.84 (0.12)	64.62 (6.14)

Table 3.2. Fishes sampled from the braided section of the Kootenai River, Idaho during the summer (May-September) of 2013: NSC = North Side Channel, SSC = South Side Channel, USC = Upper Side Channel, CB 1 = Control Bend 1, CB 2 = Control Bend 2, TB 1 = Treatment Bend 1, TB 2 = Treatment Bend 2 (where X indicates that a species was present at a site). Species are listed alphabetically by common name and scientific name is provided.

		Site						
Common name	Scientific name	NSC	SSC	USC	CB 1	CB 2	TB 1	TB 2
Burbot	*Lota lota*		X		X			
Brown bullhead	*Ameiurus nebulosus*	X		X		X		
Brook trout	*Salvelinus fontinalis*		X			X		
Bull trout	*Salvelinus confluentus*		X				X	
Brown trout	*Salmo trutta*		X		X			
Kokanee	*Oncorhynchus nerka*	X	X	X		X	X	
Largemouth bass	*Micropterus salmoides*	X						
Longnose dace	*Rhinichthys cataractae*		X	X		X		
Longnose sucker	*Catostomus catostomus*	X	X	X	X	X	X	X
Largescale sucker	*Catostomus macrocheilus*	X	X	X	X	X	X	X
Mountain whitefish	*Prosopium williamsoni*	X	X	X	X	X	X	X
Northern pikeminnow	*Ptychocheilus oregonensis*	X	X	X	X	X	X	X
Pumpkinseed	*Lepomis gibbosus*	X						
Peamouth	*Mylocheilus caurinus*	X	X	X	X	X	X	X
Redband trout	*Oncorhynchus mykiss*	X	X	X	X	X	X	X
Redside shiner	*Richardsonius balteatus*	X	X	X	X	X	X	X
Cottus spp.	Cottidae	X	X	X	X	X	X	X
Westslope cutthroat trout	*Oncorhynchus clarki lewisi*	X	X	X			X	
Yellow perch	*Perca flavescens*	X						

Table 3.3. Site abbreviation, species richness (*S*), and amount of effort (m; [95% CI]) required to attain various percentages (i.e., 50, 75, and 100%) of species richness for each month during the summer of 2013: NSC = North Side Channel, SSC = South Side Channel, USC = Upper Side Channel, CB 1 = Control Bend 1, CB 2 = Control Bend 2, TB 1 = Treatment Bend 1, TB 2 = Treatment Bend 2. Measures of effort are provided as total distance electrofished.

		Percent of total species		
Site	*S*	50%	75%	100%
		May		
NSC	8	587 (248)	965 (283)	1,397 (213)
SSC	9	354 (131)	698 (222)	1,019 (184)
USC	7	366 (168)	841 (351)	1,703 (337)
CB 1	6	320 (119)	700 (243)	984 (256)
CB 2	5	407 (158)	489 (199)	763 (201)
TB 1	6	426 (195)	717 (262)	1,076 (268)
TB 2	4	355 (146)	506 (186)	799 (191)
		June		
NSC	7	328 (134)	731 (315)	1,532 (316)
SSC	10	399 (142)	618 (198)	1,100 (122)
USC	9	350 (149)	906 (354)	1,762 (365)
CB 1	5	410 (189)	676 (282)	1,041 (302)
CB 2	8	605 (199)	698 (200)	1,063 (151)
TB 1	8	330 (122)	575 (217)	1,077 (252)
TB 2	5	298 (107)	658 (197)	907 (181)
		July		
NSC	9	394 (152)	977 (389)	1,349 (302)
SSC	8	441 (161)	624 (212)	982 (135)

Table 3.3 continued

USC	10	565 (250)	903 (304)	1,810 (265)
CB 1	7	319 (124)	460 (158)	992 (195)
CB 2	9	444 (175)	720 (195)	1,081 (130)
TB 1	8	390 (152)	548 (188)	923 (232)
TB 2	4	296 (109)	439 (147)	651 (147)
		August		
NSC	12	511 (187)	679 (241)	1,235 (177)
SSC	8	326 (134)	702 (209)	975 (207)
USC	8	708 (350)	1,216 (458)	1,814 (347)
CB 1	4	461 (268)	682 (272)	986 (296)
CB 2	6	482 (219)	500 (113)	784 (252)
TB 1	4	285 (101)	462 (132)	674 (122)
TB 2	4	224 (43)	281 (81)	425 (89)
		September		
NSC	10	374 (127)	766 (207)	942 (221)
SSC	9	398 (136)	615 (262)	991 (195)
USC	8	651 (303)	1,065 (438)	1,818 (349)
CB 1	5	357 (146)	635 (228)	845 (281)
CB 2	9	321 (102)	603 (192)	1,079 (195)
TB 1	6	466 (209)	982 (249)	1,286 (207)
TB 2	4	262 (81)	386 (146)	620 (161)

FIGURES

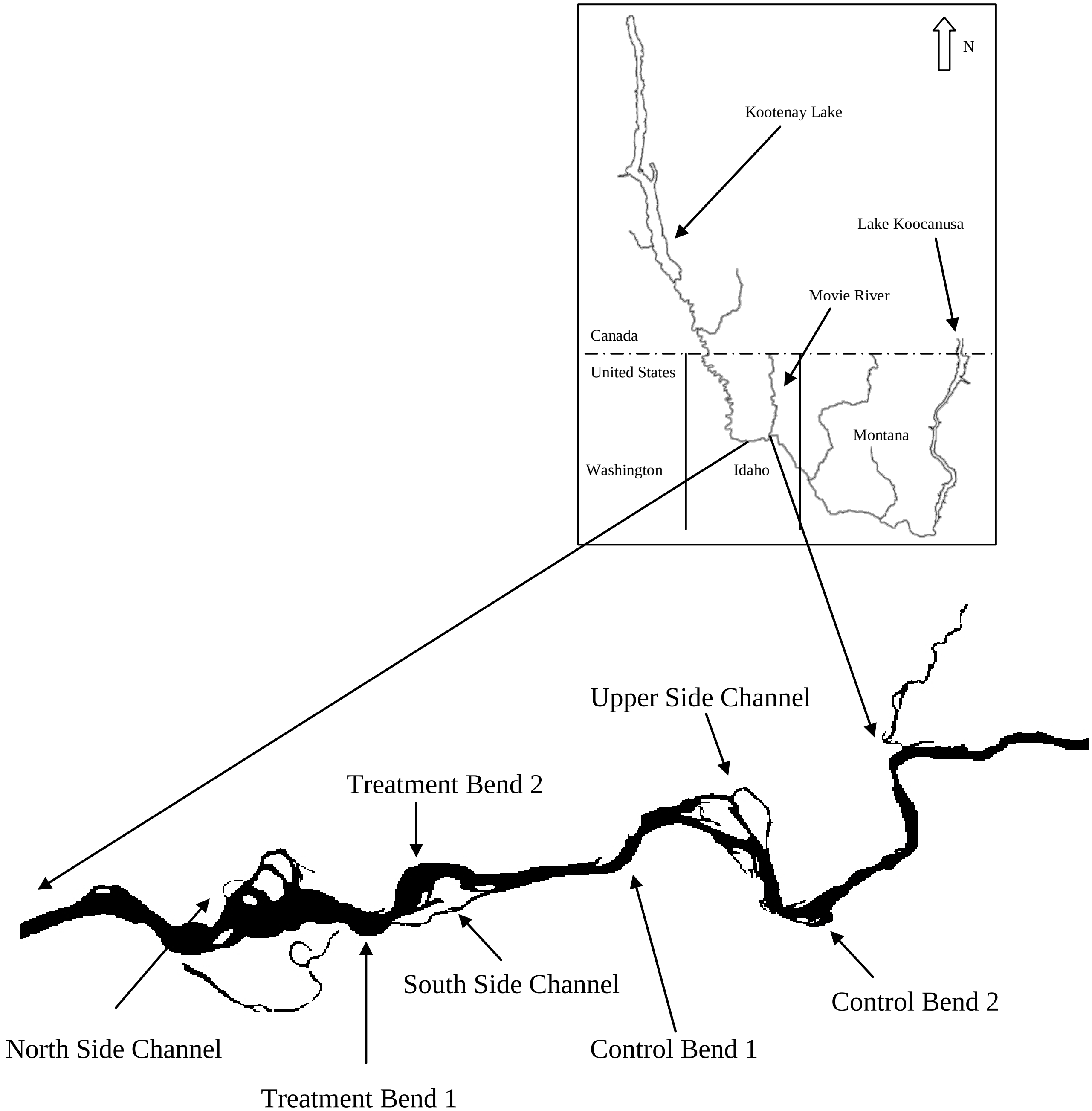
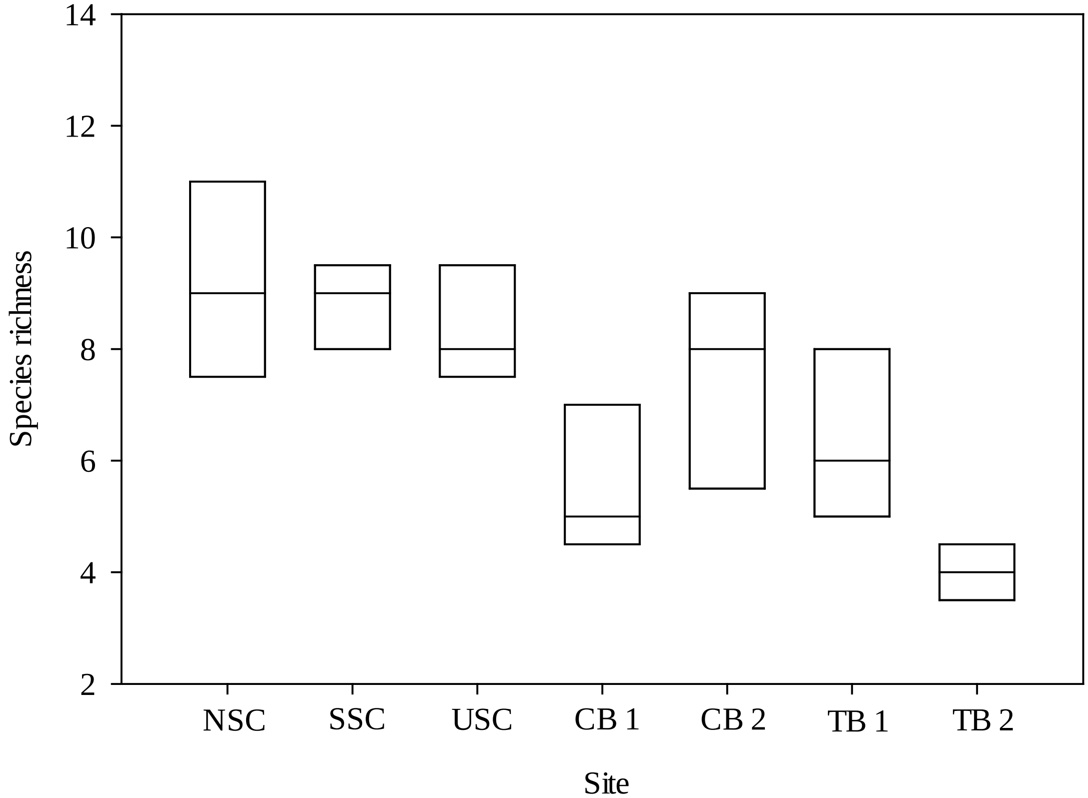


Figure 3.1



14
12
10
8
6
4
2
Species richness
NSC
SSC
USC
CB 1
CB 2
TB 1
TB 2
Site

Figure 3.2.

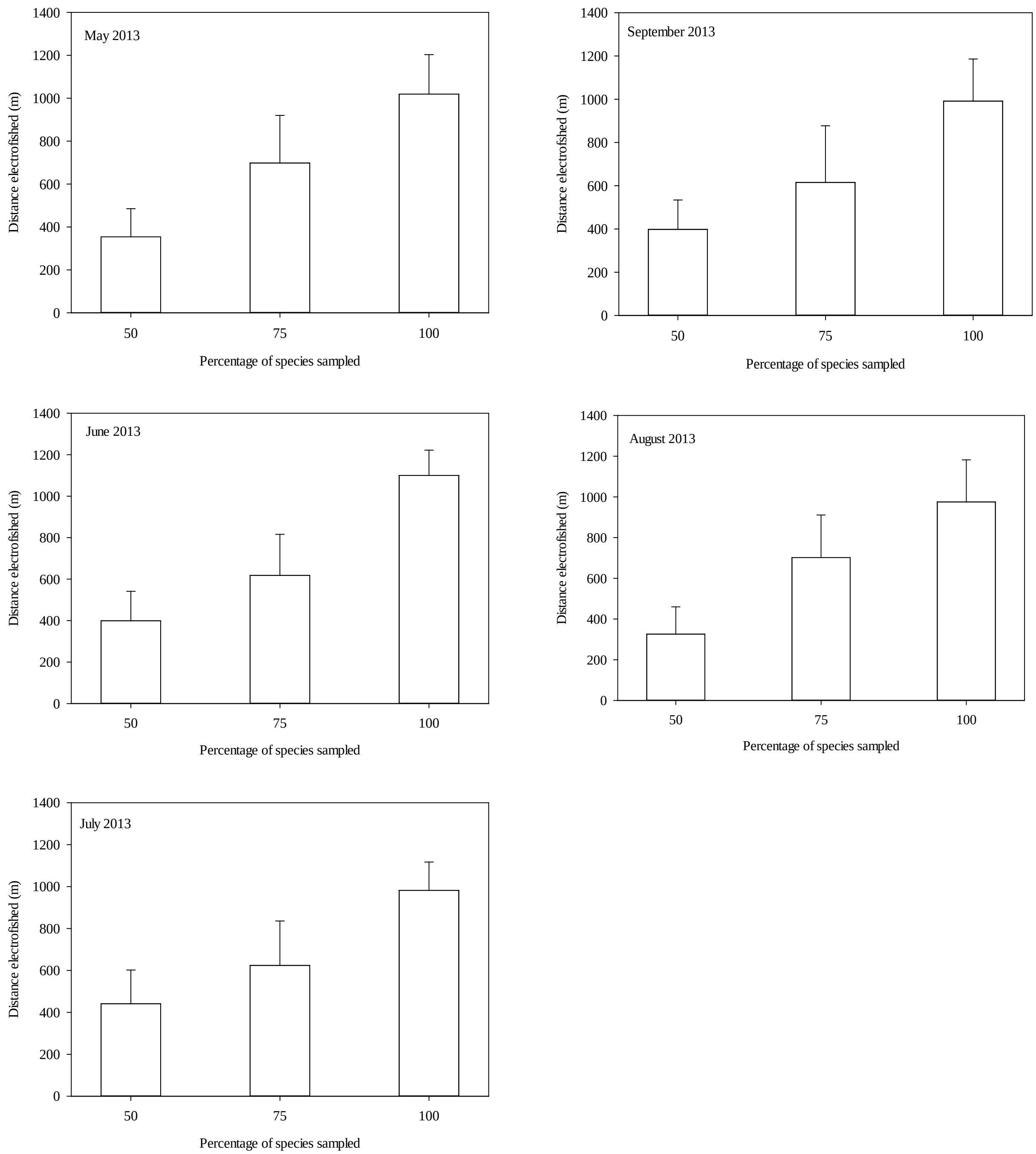

May 2013
Distance electrofished (m)
0
200
400
600
800
1000
1200
1400
50
75
100
Percentage of species sampled
September 2013
Distance electrofished (m)
0
200
400
600
800
1000
1200
1400
50
75
100
Percentage of species sampled
June 2013
Distance electrofished (m)
0
200
400
600
800
1000
1200
1400
50
75
100
Percentage of species sampled
August 2013
Distance electrofished (m)
0
200
400
600
800
1000
1200
1400
50
75
100
Percentage of species sampled
July 2013
Distance electrofished (m)
0
200
400
600
800
1000
1200
1400
50
75
100
Percentage of species sampled

Figure 3.3

Figure 3.1. Map of the Idaho portion of the Kootenai River. The braided section is located between the confluence of the Moyie River and the town of Bonners Ferry, Idaho.

Figure 3.2. Box plot of species richness for seven sampling sites in the braided section of the Kootenai River, Idaho (2013). The lines within each box represent the median species richness and the upper and lower portions of each box represent the 75^{th} and 25^{th} percentiles, respectively. Sampling sites include North Side Channel (NSC), South Side Channel (SSC), Upper Side Channel (USC), Control Bend 1 (CB 1), Control Bend 2 (CB 2), Treatment Bend 1 (TB 1), and Treatment Bend 2 (TB 2).

Figure 3.3. Graphical example of effort (distance) required to sample 25, 50, and 75% of species in the South Side Channel (SSC) of the Kootenai River where twelve, 100-m long reaches were sampled each month during May-September 2013.

CHAPTER 4: GENERAL CONCLUSIONS

Large rivers are among the most degraded and endangered ecosystems worldwide and they support a variety of important fishery resources. Connectivity between rivers and their floodplains has been limited by water development and has lead to a decrease in the availability and quality of habitat for native fishes. As such, habitat rehabilitation has been used in many cases as a management tool to mitigate for the adverse effects of development on fish and their habitat. Engineered side-channel habitats and instream habitat improvements have been used extensively to restore fish populations in large rivers throughout the midwestern United States; however, use of engineered structures to create fish habitat in large rivers of the western United States has only been used recently as a management tool. In light of recent efforts to restore fish habitat in large rivers, information used for fish assemblage and population evaluation and monitoring is more important than ever. Collection of baseline data and development of efficient sampling protocols is critical for ensuring that meaningful and comparable data are available to inform future management.

I sampled fishes from the braided section of the Kootenai River over the course of two years (2012 and 2013) and collected a total of 20 species representing 8 families. In general, my results indicated that fish assemblage structure varied among sites in terms of occurrence and relative abundance of species. Moreover, fish assemblages varied between main- and side-channel environments as well, where side-channel sites generally had higher species richness and contained more “rare” species. Habitat also varied between main- and side-channel habitats and I observed conditions in side-channel habitats that may be similar to that of historic floodplain environments. I found that differences in habitat characteristics among sites were influential in explaining differences in fish assemblage structure. I observed high reach similarity of native fishes was observed, while non-native fishes had low reach similarity with native fishes.

I identified patterns of habitats use for several native species and evaluated the importance of habitat variables for explaining variability in occurrence and relative abundance of many ecologically-valuable species. Habitat use information will provide managers of the Kootenai River with a basic understanding of the processes influencing native fish populations in a section of river where little previous information has been gathered. I established guidelines for sampling effort requirements for obtaining various levels of species richness for effective monitoring of rehabilitation treatments. I found fewer reaches are required to sample 100% of the species in main-channel sites compared to side-channel sites, and hypothesize that this may be due to the higher occurrence of “rare” species commonly sampled in side-channel sites. I did not observe patterns in effort requirements throughout the summer, but rather found that effort requirements were similar for all months during the field season. Habitat characteristics (i.e., woody cover, depth, and velocity) did not have much influence on the proportion of total site distance required to sample 100% of the species.

This study provides guidelines on sample size requirements to estimate species richness and occurrence of fishes among different channel types in the braided section of the Kootenai River. Designing effective sampling protocols depend on knowledge of how environmental characteristics are associated with effort requirements to attain desired levels of species richness. Information contained in this thesis will help managers of the Kootenai River be more efficient in developing a sampling framework that will assist in future monitoring efforts. In addition, this thesis provides a general description of fish assemblage structure among rehabilitation and reference areas throughout the braided section of the Kootenai River and identifies environmental variables associated with the occurrence and abundance of native fishes. Moreover, information on the influence of habitat variability on sampling efficiency can provide insight on stratification and allocation of sampling effort during future monitoring. This information will be valuable for managers of large rivers throughout the western United States and has contributed substantial knowledge to our understanding of regulated rivers and large river ecology.