Articles

Detection Probabilities of Electrofishing, Hoop Nets, and Benthic Trawls for Fishes in Two Western North American Rivers

Christopher D. Smith*, Michael C. Quist, Ryan S. Hardy

Christopher D. Smith
Idaho Cooperative Fish and Wildlife Research Unit, Department of Fish and Wildlife Sciences, University of Idaho, 875 Perimeter Dr. MS 1141, Moscow, Idaho 83844
Present address: Minnesota Department of Natural Resources, 23070 North Lakeshore Drive, Glenwood, Minnesota 86334

Michael C. Quist
U.S. Geological Survey, Idaho Cooperative Fish and Wildlife Research Unit, Department of Fish and Wildlife Sciences, University of Idaho, 875 Perimeter Dr. MS 1141, Moscow, Idaho 83844

Ryan S. Hardy
Idaho Department of Fish and Game, 2885 W. Kathleen Ave., Coeur d'Alene, Idaho 83814

Abstract

Research comparing different sampling techniques helps improve the efficiency and efficacy of sampling efforts. We compared the effectiveness of three sampling techniques (small-mesh hoop nets, benthic trawls, boat-mounted electrofishing) for 30 species in the Green (WY, USA) and Kootenai (ID, USA) rivers by estimating conditional detection probabilities (probability of detecting a species given its presence at a site). Electrofishing had the highest detection probabilities (generally greater than 0.60) for most species (88%), but hoop nets also had high detectability for several taxa (e.g., adult burbot *Lota lota*, juvenile northern pikeminnow *Ptychocheilus oregonensis*). Benthic trawls had low detection probabilities (<0.05) for most taxa (84%). Gear-specific effects were present for most species indicating large differences in gear effectiveness among techniques. In addition to gear effects, habitat characteristics also influenced detectability of fishes. Most species-specific habitat relationships were idiosyncratic and reflected the ecology of the species. Overall findings of our study indicate that boat-mounted electrofishing and hoop nets are the most effective techniques for sampling fish assemblages in large, coldwater rivers.

Keywords: fish; detectability; gear comparison; river

Received: February 6, 2015; Accepted: September 4, 2015; Published Online Early: September 2015; Published: December 2015

Citation: Smith CD, Quist MC, Hardy RS. 2015. Detection probabilities of electrofishing, hoop nets, and benthic trawls for fishes in two western North American rivers. *Journal of Fish and Wildlife Management* 6(2):371–391; e1944-687X. doi: 10.3996/022015-JFWM-011

* Corresponding author: Christopher.d.smith@state.mn.us

Introduction

Freshwater aquatic systems are threatened worldwide and have experienced high rates of biodiversity decline relative to terrestrial systems (Dudgeon et al. 2006; Strayer and Dudgeon 2010). Declines of freshwater fishes have been widely reported with approximately 700 fishes listed as imperiled (endangered, threatened, vulnerable) in North America (Jelks et al. 2008). Unfortunately, extinction rates are expected to increase in future decades (Ricciardi and Rasmussen 1999). Due to widespread declines in aquatic biodiversity, increased effort has been allocated by natural resource agencies to monitor fish assemblages (Noble et al. 2007). In particular, increased effort and multiple sampling techniques will likely be required to monitor large riverine systems as a result of high habitat complexity and diverse fish assemblage structure (Bonar et al. 2009a; Simpkins and Mistak 2010).

Effective management of fish populations and assemblages commonly relies on long-term monitoring programs to detect shifts in population and assemblage structure (Thomas 1996; Magurran et al. 2010). Unfortunately, few long-term monitoring programs account for imperfect detection probabilities (probability of detecting a species given its presence at a site [*p*]; Boulinier et al. 1998; Yoccoz et al. 2001; MacKenzie et al. 2006; Falke et al. 2012). Failure to account for differences in detectability when investigating temporal or spatial trends may confound true patterns and result in erroneous conclusions, particularly if detection probabilities are highly variable over long temporal or broad spatial scales (Boulinier et al. 1998; Nichols et al. 1998; MacKenzie et al. 2006). Understanding and accounting for imperfect detectability allows managers to obtain more accurate estimates of fish population trajectories and assemblage composition.

Large rivers generally have high habitat complexity and biodiversity, particularly in warmwater systems (Guy et al. 2009; Paukert and Galat 2010; Simpkins and Mistak 2010). As a result, several techniques are commonly used for single species and fish assemblage surveys in warmwater rivers (Guy et al. 2009). For instance, Guy et al. (2009) recommend the use of eight standardized gears including bag seines, large-mesh benthic trawls, small-mesh benthic trawls, boat-mounted electrofishing, drifted trammel net, and three types of hoop nets (small-, medium-, large-mesh) for the collection of fishes in warmwater river systems. The reason for multiple gears is that many techniques are specialized and only sample a single species or functional group effectively. However, other techniques (boat-mounted electrofishing) have been reported to sample the majority of fish assemblages in warmwater river systems (Neebling and Quist 2011).

Although a limited amount of research comparing different sampling gears has been conducted in warmwater systems, even less research has focused on sampling fish assemblages in large coldwater rivers (Pugh and Schramm 1998; Lapointe et al. 2006; Neebling and Quist 2011; Schloesser et al. 2012). Of the research conducted in large coldwater rivers, general gear recommendations include boat- or raft-mounted electrofishing, seines, gill nets, and snorkeling (Curry et al. 2009). Similar to warmwater systems, electrofishing surveys are common and assumed to sample the majority of species in coldwater systems (Hughes and Herlihy 2007; Maret et al. 2007). However, additional guidance regarding gear selectivity would improve sampling efficiency and management of coldwater river systems.

The goal of this study was to investigate differences in detectability among techniques and fishes in large rivers of western North America. Specific objectives were to use detection probabilities to compare the effectiveness of hoop nets, benthic trawls, and boat-mounted electrofishing for a variety of fish species, evaluate potential differences in detection probabilities of large-bodied fishes by length category, and to evaluate relationships between detectability and habitat characteristics.

Methods

Study area

This study was conducted in the Green and Kootenai rivers, two large river systems in the western United States. The Green River has a drainage basin of approximately 117,000 km^2 and a length of 1,175 km (Vanicek 1970) with its headwaters originating in the Wind River Mountain Range of Wyoming (Cooper et al. 1999). The Green River is the largest tributary of the Colorado River, flowing through Wyoming, Utah, and Colorado before its confluence with the Colorado River at Canyonlands National Park, Utah. Much of the Green River watershed is located in the Rocky Mountain and Wyoming Basin ecoregions, with primary land use consisting of agriculture (e.g., grazing, production of livestock forage), mining (e.g., coal, uranium, petroleum), recreation, and logging (Wyoming Game and Fish Department 2010). In addition to land use alterations, several impoundments have been constructed in the Green River and include Flaming Gorge Dam (construction completed in 1962), Fontenelle Dam (1964), Meeks Cabin Dam (1971), and Stateline Dam (1979; Wyoming Water Development Office 2001).

The Kootenai River has an international watershed of approximately 45,600 km^2, draining portions of both the United States and Canada (Knudson 1994). Headwaters of the Kootenai River are located in Kootenay National Park, British Columbia, Canada. From its headwaters, the Kootenai River flows south into the United States, first entering Montana and then Idaho, before returning to British Columbia and entering the Columbia River near Castlegar, British Columbia. Land use in the watershed includes logging, agriculture, mining, and some industrial facilities (e.g., fertilizer production, municipal waste facilities; Richards 1997). However, much of the anthropogenically-altered land uses occur in the lower reaches of the river, leaving headwaters relatively pristine. In addition to altered land use and levee construction, completion of Libby Dam at Libby, Montana arguably had the largest influence on the Kootenai River system. Libby Dam was completed in 1972 by the U.S. Army Corps of Engineers and impounds nearly 148 km of the Kootenai River, forming Lake Koocanusa (Whitfield and Woods 1984).

Reach selection and field sampling

Sampling in the Green River occurred at 28, 500-m long reaches during the summers (June - August) of 2012 and 2013. Forty-three reaches (500 m) were surveyed in the Kootenai River during the summers (June - August) of 2012. River reaches were selected using a systematic random sampling design. Additional reaches were added to the design using a simple random sampling design to increase sample size. Due to logistical constraints (i.e., low water levels, access limitations) some of the systematic reaches in the Green River were relocated to accessible river sections. Each reach was subdivided into two 250-m subsections (hereafter referred to as a sampling site). All reaches sampled in a season were resampled at least once and many (74%) were resampled three or four times in a single season. A series of four hoop nets, four benthic trawl hauls, and two electrofishing runs in one day for a single reach constituted a visit for the occupancy analyses. Occupancy status of a species and life history stage in each reach was assumed to be constant for a single season. In other words, if a species was sampled in a 250 m site, it was assumed to be available for all techniques in the entire 500 m reach during the period of closure. Although we were unable to formally examine this assumption, all resampling efforts occurred within six weeks of initial surveys and most reaches ($>$ 80%) were resampled within one week. In addition, on-going research and monitoring efforts in the Kootenai and Green rivers suggest that movement of the study species is minimal, particularly during the seasons when our study was conducted. Furthermore, occupancy models were constructed for only common species (i.e., species found in $\geq$ 10% of surveyed reaches) and the species-level occupancy status of a reach was unlikely to change for these ubiquitous species. Due to relatively homogenous habitat (e.g., substrate type) in the 500 m reaches, a short period of site closure, and minimal movement patterns, we believe the assumption of site closure on the species level is appropriate.

Once sampling reaches were identified, one of the 250-m sites (randomly selected) was sampled with small-mesh hoop nets and the remaining site was sampled with a modified-Missouri benthic trawl. Electrofishing sampled bank habitat along the entire reach. Such a survey design was necessary to distribute sampling effort. In total, we used four small-mesh hoop nets, four trawl hauls, and two 250-m electrofishing runs in each sample reach for every sampling event. Active gears (i.e., electrofishing, trawls) sampled reaches first; passive gears (i.e., hoop nets) were set following completion of active sampling and allowed to sample overnight.

Hoop nets are a passive sampling gear commonly used in rivers to capture a variety of fishes (Guy et al. 2009). We used small-mesh hoop nets in an attempt to capture small-bodied fishes. Hoop nets used in this study had an overall length of 3.7 m and were covered with 6.3-mm bar measure knotless delta style mesh. Nets had six, 60-cm diameter circular steel hoop frames to keep nets open while fishing. Two throats (10 cm minimum diameter) were located between the second and fourth, and fourth and sixth hoop frames. Four hoop nets were deployed following electrofishing and trawling. Nets were baited with fresh, dead local non-game fishes (e.g., *Catostomus* spp.) and fished for approximately 12 hrs (i.e., overnight).

The modified Missouri River benthic trawl is an effective gear for small-bodied and juvenile fishes (i.e., $<$ 300 mm) in riverine habitats (Herzog et al. 2005; Neebling and Quist 2011). The trawl was constructed with a large inner mesh in the body (0.10 cm diameter No. 7 multifilament nylon twine, 3.5-cm bar measure mesh) and a smaller outer mesh surrounding the body and composing the wings of the trawl (6.3-mm bar measure delta style, knotless mesh). Small mesh completely encompassing the larger inner mesh limited the loss of small-bodied fish through the trawl and minimized the chance of smaller fishes being damaged by large-bodied fish or debris collected by the trawl. A chain (5-mm link diameter) was attached to the 3.7-m footrope to maintain contact with the bottom at all times. Due to differences in depth, length of towlines (2.2 cm twisted nylon rope) varied by river system (Green River towline length = 20.0 m, Kootenai River towline length = 30.5 m) to ensure that trawls were fished along the bottom. Otter doors (61 cm × 30.5 cm) were attached to the trawl and prevented the collapse of the trawl while deployed. Trawls were deployed off the bow and pulled downstream. Nets were fished at a velocity slightly faster than the current. Each trawl was fished for 50 m or until snagged. If a trawl sampled 30 m or less, data from the initial haul was discarded and a new haul was conducted. A global positioning system receiver (Lowrance, Tulsa, OK) was used to measure distance sampled and measurement started once the trawl was fully deployed. Distance traveled with the trawl fully deployed was used as a measure of effort.

Electrofishing is a common and effective fish sampling method in both lentic and lotic systems (Guy et al. 2009; Miranda and Boxrucker 2009). Electrofishing gear consisted of an Infinity model electrofisher (Midwest Lake Electrofishing Systems, Inc., Polo, MO) and a 5,000 W Honda generator (American Honda Motor Co., Inc., Torrance, CA). Electrofishing power output was standardized to 2,750 – 3,250 W (Miranda 2009). Two netters were stationed near the bow of the boat to collect fish. Dip nets used to collect fish had 6.3-mm knotless mesh. Pulsed-DC electrofishing effort was conducted during the day. Effort was randomly allocated to one bank for the upstream sampling site and alternated to the other bank for the downstream site. Electrofishing was initiated in the uppermost portion of the reach and sampled fish with a single pass (approximately 250 m) in a downstream direction. Fish catch was enumerated at the completion of electrofishing for every sample site. Effort was recorded as the number of seconds with electricity being applied to the water.

Fishes were identified in the field. If fishes were unable to be accurately identified, voucher specimens were collected and identified in the laboratory. Following identification, all fish were measured (total length; mm), enumerated, and released at a location away from the subsequent sample sites (Table S1, *Supplemental Material*).

An evaluation of aquatic habitat was completed for every reach and was modeled after Neebling and Quist (2011), U.S. Environmental Protection Agency non-wadeable river protocols (Flotermersch et al. 2006), and Wilhelm et al. (2005). Some habitat variables were assumed to be constant in a season (e.g., bank type, substrate composition; hereafter referred to as site-scale variables), whereas other variables likely varied for each sampling event (e.g., discharge, temperature; hereafter referred to as sample-scale variables). Sample-scale habitat characteristics were collected at the completion of each sampling event (water temperature [°C], water clarity [m], conductivity [μS/cm]). Average daily discharge (m^3/s) was acquired from U.S. Geological Survey gaging stations. Due to high water clarity and low depths in some reaches, a binary index of maximum water clarity (a function of both turbidity and depth) was provided for each sampling event using a Secchi disk. Specifically, if the Secchi disk was visible on the substrate at the deepest point in the surveyed reach, a value of one was recorded. Otherwise, a value of zero was recorded.

In addition to sample-scale covariates, we collected site-scale habitat characteristics for each reach. Proportion of bank type was classified into one of four categories: eroding, vegetation, silt-sand ($\leq$ 0.2 mm), and cobble-boulder ($\geq$ 64.0 mm). Due to the upstream impoundment of both systems and relatively stable water levels, metrics of wetted width, depth (m), flow velocities (m/s), and substrate type were assumed to be constant for our limited period of site closure. Habitat variables (width, depth, flow velocity, substrate type) were measured using systematically assigned transects (six habitat transects for each sample reach). Wetted width measurements were estimated to the nearest 0.5 m with a laser rangefinder at every transect (Wilhelm et al. 2005). Seven equidistant points across the wetted width of each transect were measured for depth, velocity, and substrate composition (Neebling and Quist 2011). Water depth was recorded using a Lowrance depth finder (Lowrance, Tulsa, OK). In depths less than 1 m, mean water column velocities were measured at 60% of the depth using an electromagnetic flow meter (modified from Kaufmann et al. 1999; Flotermersch et al. 2001; Neebling and Quist 2011; Marsh-McBirney, Loveland, CO). If depth was greater than 1 m, a measure of water velocity was collected at 20% and 80% of the depth and averaged to estimate the mean water column velocity. Bottom velocity measurements were collected 8 cm from the bottom. Substrate composition was classified into one of five categories: organic matter-sand (<0.0004 – 0.2 mm), gravel (0.2 – 64.0 mm), cobble (64.0 – 256.0 mm), boulder (> 256.0 mm), or bedrock (modified from Orth and Maughan 1982).

Statistical analysis

Occupancy models were used to investigate gear-specific detection probabilities of fishes and relationships among detection probabilities and habitat characteristics. Occupancy models use information on detection and non-detection to estimate site occupancy (probability of sites being occupied; $\hat{\psi}$) and account for imperfect detectability (MacKenzie et al. 2002; MacKenzie et al. 2006). To account for imperfect detectability, sampling must be replicated (spatially or temporally) in a period of demographic closure. Occupancy models estimate the probability of observing each detection history (i.e., binary code indicating the detections or non-detections of a species at a site through repeated surveys) based a series of detection or non-detection information (MacKenzie et al. 2002; MacKenzie et al. 2006; Burdick et al. 2008). For example, given that a species is present, the probability of observing a detection history consisting of two sampling events takes the form of:

$$\Pr(h) = \hat{\psi}(\hat{p}_1)(1-\hat{p}_2)$$

where Pr(h) is the probability of the detection history, $\hat{\psi}$ is the probability of occupancy, and $\hat{p}$ indicates whether a species was detected or not (1-$\hat{p}$) during each sampling event (MacKenzie et al. 2006; Burdick et al. 2008; Williams 2010). Maximum likelihood is then used to estimate parameters given the observed data (MacKenzie et al. 2006; Burdick et al. 2008). Estimates of variability (standard error) were obtained using the delta method (MacKenzie et al. 2006). Due to an unknown occupancy status of sites without a species detection (a site could be truly unoccupied or occupied and not detected), detection probabilities where only estimated for sites with at least one detection.

Fish species were classified as small- or large-bodied fishes (median total length of adults for a species $\geq$ 200 mm) because differences in length may influence habitat use, gear selectivity, and detection probabilities (Argent and Kimmel 2005; Bonar et al 2009b; Schloesser et al. 2012; Haynes et al. 2013). Occupancy models for large-bodied fishes were separated by life history stage, specifically length at maturity. Although we were not explicitly interested in life history stage, categorizing fish as juveniles or adults provided an objective way to separate fishes into groups while using information on their ecology to investigate size- and life history stage-dependent relationships with detectability. Approximate lengths at maturity were obtained from the literature and used to classify large-bodied fish as juvenile or adult (Table S2, *Supplemental Material*).

Some species were not detected with all sampling techniques. Consequently, detection histories were used from only techniques that detected a particular species or life history stage at least once in a system. Due to a potential lack of independence among sampling

Table 1. Mean and standard error (in parentheses) of sample- and site-scale habitat variables collected from 71 reaches on the Kootenai and Green rivers during 2012 and 2013. Both site- [percentage of bank consisting of large substrate (Bk_{Bo}), percentage of bank consisting of vegetation (Bk_{Veg}), coefficient of variation in water column velocity (CV_{Vel}), average depth (Depth), mean discharge (Q), percentage of substrate consisting of cobble and boulder ($Sub_{Co\text{-}Bo}$), mean water column velocity (Vel_{Avg}), average substrate velocity (Vel_{Sub}), mean channel width (Width)] and sample-scale [water clarity (Clarity), a metric of water clarity and average depth ($Clarity_{Mx}$), mean temperature (Temp)] habitat variables were used to investigate relationships between occupancy and detectability of fishes with single-season occupancy models.

Variable	Description	Green River	Kootenai River
Bk_{Bo}	Percentage of bank consisting of large substrate (i.e., cobble, boulder; %)	27.7 (4.6)	15.9 (3.1)
Bk_{Veg}	Percentage of bank consisting of vegetation (%)	56.5 (5.0)	38.8 (4.6)
Clarity	Mean Secchi disk depth (m)	1.8 (<0.1)	2.8 (<0.1)
$Clarity_{Mx}$	Factor (Water clarity was greater than max depth in reach)	0.6 (<0.1)	<0.1 (<0.1)
CV_{Vel}	Mean coefficient of variation in water column velocity	0.4 (<0.1)	0.4 (<0.1)
Depth	Mean depth (m)	0.8 (<0.1)	6.5 (0.3)
Q	Mean discharge (m^3/s)	35.3 (2.1)	702.0 (16.4)
$Sub_{Co\text{-}Bo}$	Percentage of substrate consisting of cobble and boulder (%)	42.1 (6.4)	10.1 (3.1)
Temp	Mean temperature (°C)	13.1 (0.3)	14.6 (0.1)
Vel_{Avg}	Mean water column velocity (m/s)	0.8 (<0.1)	0.5 (0.1)
Vel_{Sub}	Mean substrate velocity (m/s)	0.6 (<0.1)	0.4 (<0.1)
Width	Mean channel width (m)	75.2 (2.7)	177.8 (6.1)

techniques, cumulative detection histories were constructed for each sampling technique and sampling event. For example, consider a situation where a species was collected in one electrofishing run and in two hoop nets, but was not detected with the trawl. A detection history for the species and sampling event would be (detected = 1, undetected = 0; electrofishing, hoop net, trawl) 1, 1, 0. An alternative approach would be to treat each unit of effort (i.e., one hoop net set, a trawl haul, electrofishing run) as an individual detection event. In other words, every daily survey would have a detection history consisting of ten events (i.e., two electrofishing runs, four hoop nets, four trawl hauls). However, cumulative detection histories were necessary given our study design, as the capture of an individual in a hoop net would prevent the capture of the same individual in other hoop nets for that event. Such scenarios could potentially violate the assumption that occupancy status and availability of a species for sampling would not change in a reach over the period of closure.

A single-species, single-season model was constructed for each species and life history stage. Use of other model types (e.g., multi-state, multi-species) were investigated, but due to a moderate sample size and interest in providing detailed species-specific detectability relationships we used a traditional single-species, single-season model. Single-season models were constructed in three stages for each species (small-bodied fishes) or life history stage (large-bodied fishes) detected in more than 10% of reaches (MacKenzie et al. 2005; modified from Schloesser et al. 2012). We first determined which habitat characteristics most influenced occupancy rates and held those covariates constant for remaining model stages (J. Nichols, Patuxent Wildlife Research Center, personal communication). This helped minimize additional variation not attributed to detectability in the occupancy models. The second stage compared models with constant versus gear-specific detection probabilities. The final stage used the top candidate model (i.e., lowest AIC_c value [Akaike's Information Criterion adjusted for small sample size]) from the second stage to investigate relationships between detectability and habitat covariates. Species- and life history state-specific detectability estimates were collected from the second model stage (i.e., models with only gear effects).

To reduce the risk of multicollinearity, Spearman's correlation coefficient was used to investigate potential relationships among habitat variables. In general, the most ecologically relevant or interpretable variable of a highly correlated pair ($|\hat{\rho}| > 0.70$) was retained for consideration in *a priori* candidate models (Sindt et al. 2012; Table 1). However, some variables deemed ecologically important were highly correlated, but were both included because they likely influenced detectability via different mechanisms. For example, average depth and the proportion of large substrate were highly correlated ($\hat{\rho}$ = -0.76), but could influence occupancy and detectability differently. A variable investigating spatial dependence (i.e., distance from dam) was included in initial models; however, covariates did not have support relative to other, more interpretable, environmental variables (e.g., depth, substrate composition). Temporal variables were also considered (i.e., Julian date), particularly for juvenile fishes, but had minimal support. A lack of model support warranted the omission of these variables from the final analysis. Gear-specific interactions were also of interest. Unfortunately, such relationships were unable to be investigated due to a moderate sample size and the risk of model overparamerization. As a result, general relationships with habitat characteristics and detectability were provided.

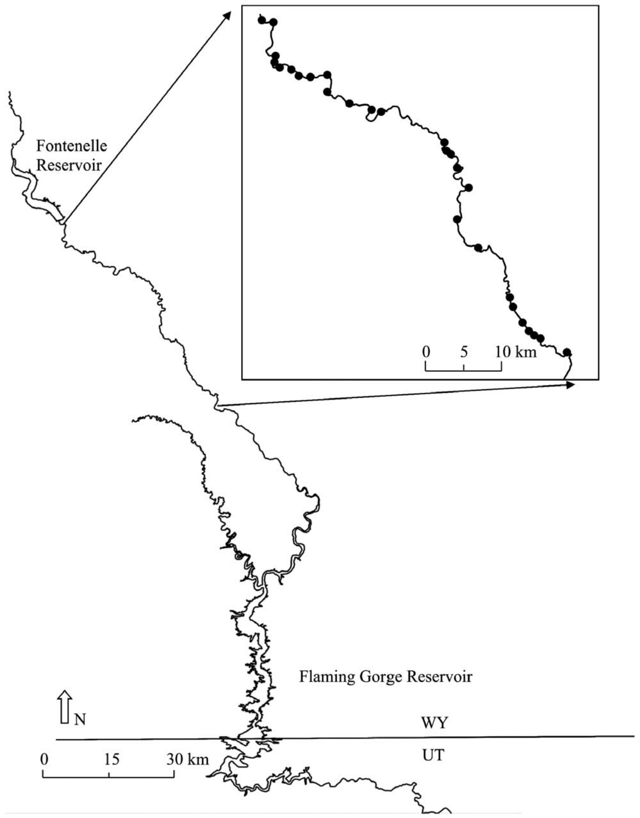

Figure 1. Reaches in the Green River (black circles) surveyed for fishes and aquatic habitat in 2012 and 2013.

Seven to 15 candidate models were constructed for each species based on *a priori* hypotheses to investigate relationships between detectability and habitat covariates (Burnham and Anderson 2002). Candidate models were ranked using AIC_c adjusted for small sample size (AIC_c; Burnham and Anderson 2002). A conservative estimate of effective sample size (i.e., total number of reaches) was used to adjust for small sample size. If overdispersion (estimated with the variance inflation factor; $\hat{c} > 1$) was present, quasi-AIC_c (QAIC_c) was used to rank candidate models (Burnham and Anderson 2002). Akaike model weights were calculated to document the conditional probability of each model in the candidate suite (Burnham and Anderson 2002). Additionally one additional parameter was added to the total number of parameters if $\hat{c}$ was greater than one (Burnham and Anderson 2002). Models were considered to have equal support if they were within two AIC_c or QAIC_c values (Burnham and Anderson 2002).

An overdispersion parameter and parametric bootstrap goodness-of-fit tests (1,000 iterations) were used to evaluate model fit (White et al. 2002; MacKenzie and Bailey 2004; MacKenzie et al. 2006). If $\hat{c}$ was greater than one, data were likely either overdispersed or there was a lack of model fit and estimated model variance was adjusted with $\hat{c}$ (Burnham and Anderson 2002; MacKenzie et al. 2006). Underdispersed models were not adjusted with $\hat{c}$. Occupancy models were fit using program PRESENCE (Hines 2006).

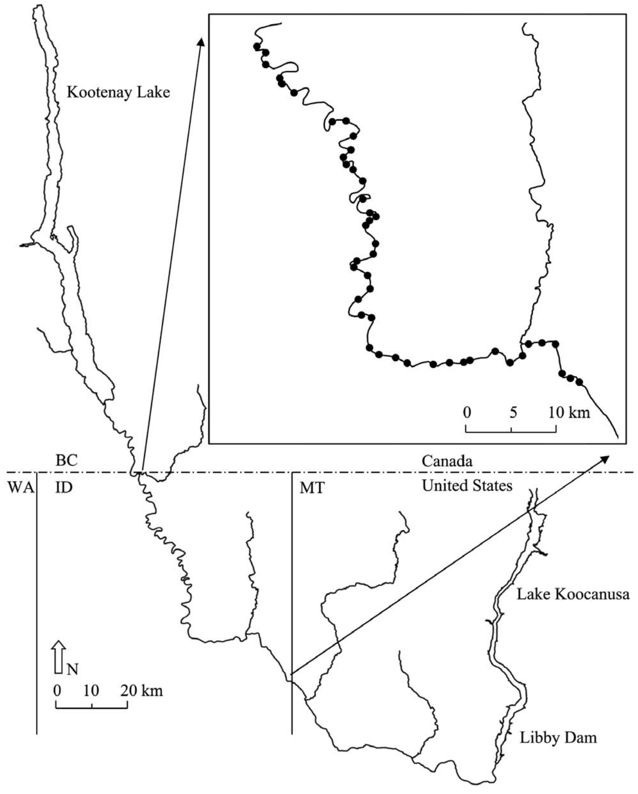

Figure 2. Reaches surveyed in the Kootenai River for fishes and aquatic habitat in 2012 and 2013.

Results

One-hundred and ninety-nine sampling events occurred at 28 reaches (total number of sampling events = 83) in the Green River (Figure 1) and 43 reaches (116) in Kootenai River (Figure 2) in the summers of 2012 and 2013. An average of 3.0 sampling events (minimum, maximum; 2, 5) occurred in river reaches in the Green River and 2.7 events (2, 3) in the Kootenai River. A total of 9,810 individuals consisting of 30 species and nine families was sampled (Table 2). Two federally listed species (species and gear by which it was sampled; bull trout *Salvelinus confluentus*, electrofishing; white sturgeon *Acipenser transmontanus*, electrofishing and trawl) were collected in the Kootenai River. Additionally, two species of high conservation concern in Idaho and Wyoming were collected: flannelmouth sucker *Catostomus latipinnis* (status and gear by which it was sampled; Species of Concern in Wyoming, electrofishing and trawl) and burbot *Lota lota* (Imperiled Species in Idaho, electrofishing and hoop nets).

Electrofishing collected four unique species (i.e., a species not collected with other techniques) in the Green River and five unique species in the Kootenai River. Hoop nets detected one unique species in the Green River (fathead minnow *Pimephales promelas*). Trawls failed to detect any unique species in either river system. Electrofishing collected the majority of individuals (percentage of total catch = 80%), followed by hoop nets (12%), and trawls (8%).

Gear-specific detectability in the Green River was greatest with electrofishing for adult white sucker *Catostomus commersonii* ($\hat{p}$ ± standard error; 0.70 ± 0.05) and common carp *Cyprinus carpio* (0.68 ± 0.13), and with hoop nets for adult burbot (0.61 ± 0.06). Similarly, detection probabilities in the Kootenai River were highest with electrofishing for adult largescale sucker *Catostomus macrocheilus* (0.87 ± 0.03) and peamouth *Mylocheilus caurinus* (0.80 ± 0.06; Table 3). Detection probabilities with hoop nets in the Kootenai River were highest for juvenile (0.86 ± 0.04) and adult (0.46 ± 0.05) northern pikeminnow *Ptychocheilus oregonensis*. Overall, detection probabilities among systems were greatest with electrofishing for 75% of small-bodied species, 92% of juvenile large-bodied fishes, and 88% of adult large-bodied fishes. Detectability of the trawl was low for most species (<0.10) and was highest for speckled dace *Rhinichthys osculus* (0.44 ± 0.06) in the Green River and juvenile mountain whitefish *Prosopium williamsoni* in the Green (0.24 ± 0.17) and Kootenai (0.38 ± 0.05) rivers.

Gears with the greatest species-specific detection probability were consistent across river systems for all shared species (i.e., taxa found in both river systems; mountain whitefish, rainbow trout *Oncorhynchus mykiss*, burbot). Electrofishing had the greatest detection probabilities for both juvenile ($\hat{p}$ ± standard error in the Green River, $\hat{p}$ ± standard error in the Kootenai River; 0.31 ± 0.19, 0.57 ± 0.05) and adult (0.48 ± 0.08, 0.68 ± 0.05) mountain whitefish and juvenile (0.42 ± 0.09, 0.43 ± 0.14) and adult (0.54 ± 0.07, 0.24 ± 0.09) rainbow trout (Table 3). Hoop nets had the greatest detectability of compared techniques for adult burbot in both the Green (0.61 ± 0.14) and Kootenai (0.12 ± 0.03) rivers.

Gear-specific effects on detectability were apparent for most species or species-life history stage combinations (Table 4). Occupancy models of most small-bodied (100%), juvenile large-bodied (60%), and adult large-bodied fishes (100%) had more support with gear-specific models than constant detectability models in the Green River. Similarly, gear-specific models had more support than constant detectability models for 80% of small-bodied, 100% of juvenile large-bodied, and 100% of adult large-bodied fishes in the Kootenai River.

Although detectability of many fishes was best predicted solely with gear covariates, environmental characteristics were also related to detectability of some fishes in the Green River. For instance, detectability of juvenile rainbow trout was positively related to water velocity, whereas detection probabilities of Utah chub *Gila atraria*, adult brown trout *Salmo trutta*, adult burbot, and mottled sculpin *Cottus bairdii* were negatively related to velocity (Table 4). Other variables, such as the coefficient of variation in velocity were related similarly (i.e., negatively) to the detectability of salmonids (e.g., adult brown trout, juvenile rainbow trout). Site-specific covariates were influential for more species than sample-scale covariates in the Green River.

Similar to the Green River, occupancy models for samples in the Kootenai River indicated variable effects of environmental characteristics on detectability. Average depth and the proportion of cobble or boulder substrate appeared in top models for many species (Table 4). Specifically, depth was negatively related to detectability of adult largescale sucker and several adult salmonids (mountain whitefish, rainbow trout). The proportion of cobble or boulder substrate was negatively related to detectability of many fishes (e.g., juvenile northern pikeminnow), but a positive relationship was apparent for adult mountain whitefish. Sample covariates (e.g., water clarity) were selected among top candidate models for samples in the Kootenai River for several fishes. Negative relationships between water clarity and detectability were observed for yellow perch *Perca flavescens*, juvenile longnose sucker *Catostomus catostomus*, adult longnose sucker, adult kokanee *Oncorhynchus nerka*, and adult rainbow trout; however, a positive relationship was observed for adult mountain whitefish. Goodness-fit-tests and estimates of overdispersion provided no evidence for a lack of fit of occupancy models for most (89%) species. However, models for several species were underdispersed (i.e., $\hat{c} < 0.5$). Underdispersion was likely the result of low detections and moderate amounts of data despite a large survey effort.

Discussion

Sampling techniques used in our study have been used to sample fish assemblages in warmwater and coldwater rivers of North America (Curry et al. 2009; Guy et al. 2009). Our comparison of sampling techniques using detection probabilities provides an important comparison of techniques and will guide gear selection in similar river systems. Moreover, while actual detection probability estimates may vary across systems due to biotic and abiotic factors (e.g., water temperature, species abundance), general species-specific trends in gear efficacy (i.e., most effective gear) likely hold true across similar coldwater rivers. Detectability estimates for several taxa had considerable levels of uncertainty (i.e., large standard error estimates), likely related to low numbers of detections. Despite the uncertainty associated with estimates of detectability for several taxa, general trends in gear effectiveness (i.e., most effective technique for taxa) are consistent with our field observations and provide insight on which gears perform best for different taxa. Lastly, it is worth noting that care should be taken when comparing detection probabilities in our study to others, as units of effort may not be identical.

Occupancy models can be an effective method for comparing sampling techniques. Historically, different sampling techniques have been compared by estimating catchability (proportion of the population sampled with one unit of effort); however, catchability requires unbiased estimates of abundance (Peterson and Paukert 2009). Abundance can be difficult to estimate with reasonable precision in riverine systems due to high connectivity and habitat complexity (Speas et al. 2004; Peterson and Paukert 2009). Comparisons may also be made using relative abundance estimates (e.g., catch-per-unit-effort), but due to differences in measures of effort, particularly between active and passive gears, results are difficult to

Table 2. Species composition of catch and total number of individuals sampled (in parentheses) from 71 reaches on the Kootenai and Green rivers during 2012 and 2013. Taxa are organized by river system, species, and life history stage. Large-bodied species (median total length of adults for a species ≥ 200 mm) were separated by life stage (i.e., juvenile, adult). Sampling techniques include electrofishing (EF), hoop nets (HN), and trawl (TR). Total number of sites with at least one detection (in brackets) and non-native species (†) status are indicated below.

Taxa and life history stage	Species Composition by Gear		
	EF	HN	TR
Green River			
Cyprinidae			
Common Carp *Cyprinus carpio*†			
Adult [7]	2 (19)	0 (0)	0 (0)
Speckled dace *Rhinichthys osculus* [24]	2 (25)	8 (15)	29 (76)
Utah Chub *Gila atraria*† [4]	<1 (5)	0 (0)	0 (0)
Catostomidae			
Flannelmouth Sucker *Catostomus latipinnis*			
Adult [7]	<1 (10)	0 (0)	<1 (1)
White Sucker *Catostomus commersonii*†			
Juvenile [19]	6 (62)	5 (10)	<1 (1)
Adult [26]	45 (484)	9 (17)	3 (9)
Salmonidae			
Brown Trout *Salmo trutta*†			
Juvenile [15]	3 (34)	2 (4)	5 (14)
Adult [15]	3 (35)	<1 (1)	1 (3)
Cutthroat Trout *Oncorhynchus clarkii*			
Juvenile [4]	<1 (3)	<1 (1)	<1 (1)
Adult [16]	3 (31)	1 (2)	<1 (1)
Mountain Whitefish *Prosopium williamsoni*			
Juvenile [16]	3 (35)	2 (4)	52 (139)
Adult [19]	15 (164)	0 (0)	1 (3)
Rainbow Trout *Oncorhynchus mykiss*†			
Juvenile [14]	4 (48)	4 (7)	<1 (2)
Adult [20]	7 (76)	3 (5)	2 (4)
Gadidae			
Burbot *Lota lota*†			
Juvenile [3]	<1 (2)	2 (3)	0 (0)
Adult [24]	2 (17)	64 (124)	0 (0)
Cottidae			
Mottled Sculpin *Cottus bairdii* [19]	3 (34)	<1 (1)	4 (11)
Kootenai River			
Cyprinidae			
Northern Pikeminnow *Ptychocheilus oregonensis*			
Juvenile [42]	14 (878)	33 (320)	2 (3)
Adult [33]	3 (166)	8 (76)	0 (0)
Peamouth *Mylocheilus caurinus*			
Juvenile [35]	24 (1,523)	<1 (8)	0 (0)
Adult [37]	18 (1,164)	0 (0)	0 (0)
Redside Shiner *Richardsonius balteatus* [41]	8 (493)	51 (480)	0 (0)
Catostomidae			
Largescale Sucker *Catostomus macrocheilus*			
Juvenile [28]	7 (461)	2 (17)	1 (2)
Adult [40]	10 (676)	0 (0)	0 (0)
Longnose Sucker *Catostomus catostomus*			
Juvenile [28]	<1 (27)	<1 (3)	0 (0)
Adult [19]	<1 (43)	<1 (1)	0 (0)

Table 2. Continued.

Taxa and life history stage	Species Composition by Gear EF	HN	TR
Ictaluridae			
Brown Bullhead *Ameiurus nebulosus*† [7]	<1 (3)	<1 (7)	0 (0)
Salmonidae			
Kokanee *Oncorhynchus nerka*			
Juvenile [22]	3 (203)	0 (0)	<1 (1)
Adult [12]	<1 (25)	0 (0)	0 (0)
Mountain Whitefish			
Juvenile [43]	3 (171)	<1 (3)	93 (177)
Adult [37]	5 (309)	<1 (1)	<1 (1)
Rainbow Trout			
Juvenile [13]	<1 (23)	0 (0)	0 (0)
Adult [21]	<1 (31)	<1 (2)	0 (0)
Gadidae			
Burbot			
Adult [11]	0 (0)	1 (14)	0 (0)
Cottidae			
Torrent Sculpin *Cottus rhotheus* [29]	2 (150)	1 (11)	3 (6)
Centrarchidae			
Pumpkinseed *Lepomis gibbosus*† [15]	2 (97)	<1 (4)	0 (0)
Percidae			
Yellow Perch *Perca flavescens*† [9]	<1 (23)	<1 (2)	0 (0)

interpret (Peterson and Paukert 2009). We argue that the use of occupancy models and detectability (comparisons are made among gears using probabilities) is advantageous over traditional comparison methods.

Previous research indicates electrofishing is an effective method for monitoring fish assemblages in large river systems (Pugh and Schramm 1998; Lapointe et al. 2006; Mercado-Silva and Escandon-Sandoval 2008; Neebling and Quist 2011). Mercado-Silva and Escandon-Sandoval (2008) reported greater estimates of species richness, diversity, and biomass with electrofishing compared to seines in the San Francisco River, Mexico. Results from our research are similar for western riverine systems where electrofishing not only collected the majority of individuals, but sampled most detected species and nine unique species between the two river systems. Due to high water clarity, nighttime electrofishing efforts may increase detection probabilities of some species, particularly larger-bodied fishes in rivers (Curry et al. 2009). For example, Sanders (1992) sampled substantially greater numbers of individuals, larger individuals, and species in nighttime versus daytime electrofishing samples in the Ohio and Muskingum rivers. However, due to safety concerns associated with nighttime surveys in coldwater systems (Curry et al. 2009), our study only used daytime electrofishing surveys. Despite occurring in daytime hours, the electrofishing technique detected the greatest number of species and had the highest detection probabilities for most species in the current study.

Hoop nets are a passive sampling technique that is commonly used to collect fishes (e.g., ictalurids, cyprinids) in both warmwater (Pugh and Schramm 1998; Lapointe et al. 2006; Stone 2010; Van Haverbeke et al. 2013) and coldwater lotic systems (Guy et al. 2009). Hoop nets collected four unique species in the Detroit River, Ontario and were an important supplement to the other gears (e.g., electrofishing, seines; Lapointe et al. 2006). Similarly, hoop nets detected one unique species in our research (fathead minnow) and had greater detection probabilities for several species (e.g., burbot, northern pikeminnow) in both river systems. Consequently, our study supports the observation that hoop nets and electrofishing are complementary in fish assemblage surveys of western rivers.

Modified Missouri benthic trawls are effective for sampling small-bodied benthic fishes in non-wadeable Midwestern rivers (Herzog et al. 2005; Neebling and Quist 2011). Neebling and Quist (2011) found that trawls collected the greatest number of individuals and the most species (73% of species observed) as part of a gear comparison study (boat-mounted electrofishing, benthic trawls, seines) in non-wadeable Iowa rivers. In our study, trawls had low detection probabilities for most species (<0.10) and did not sample any unique species. Water clarity varied by system and season, but was generally in excess of 2 m. Consequently, net avoidance due to high water clarity is a likely mechanism for the low efficacy observed in our study. Similar negative relationships between water clarity and catch have been reported for benthic trawls in large lentic systems (Duijse et al. 1992). Recent research has investigated the use of electrified benthic trawls to minimize net avoidance in warmwater systems with high water clarity (Freedman et al. 2009). For example, Freedman et al. (2009) determined an electrified

Table 3. Estimated detection probability ($\hat{p}$) of taxa collected from 71 reaches on the Kootenai and Green rivers during 2012 and 2013. Taxa are organized by river system, species, and life history stage. Only occupancy models for large-bodied species (median total length of adults for a species $\geq$ 200 mm) were separated by life stage (i.e., juvenile, adult). Sampling techniques include electrofishing (EF), hoop nets (HN), and trawl (TR). Estimates of detectability were collected from the second model stage (i.e., gear only). Standard error of detectability was adjusted by the variance inflation factor (in parentheses). A hyphen indicates that detection probabilities were unable to be calculated because no fish were sampled with the respective technique(s). Non-native species (†) are indicated below.

	Detection probability ($\hat{p}$)		
Taxa and life history stage	**EF**	**HN**	**TR**
	Green River		
Cyprinidae			
Common Carp *Cyprinus carpio*†			
Adult	0.68 (0.13)	-	-
Speckled dace *Rhinichthys osculus*	0.27 (0.05)	0.09 (0.03)	0.44 (0.06)
Utah Chub *Gila atraria*†	0.12 (0.12)	-	-
Catostomidae			
Flannelmouth Sucker *Catostomus latipinnis*			
Adult	0.24 (0.10)	-	0.03 (0.03)
White Sucker *Catostomus commersonii*†			
Juvenile	0.45 (0.08)	0.12 (0.04)	0.02 (0.02)
Adult	0.70 (0.05)	0.12 (0.04)	0.09 (0.03)
Salmonidae			
Brown Trout *Salmo trutta*†			
Juvenile	0.29 (0.09)	0.03 (0.03)	0.07 (0.04)
Adult	0.34 (0.08)	0.02 (0.01)	0.05 (0.03)
Cutthroat Trout *Oncorhynchus clarkii*			
Juvenile	0.11 (0.14)	0.04 (0.06)	0.04 (0.06)
Adult	0.30 (0.08)	0.03 (0.02)	0.01 (0.01)
Mountain Whitefish *Prosopium williamsoni*			
Juvenile	0.31 (0.19)	0.04 (0.07)	0.24 (0.17)
Adult	0.48 (0.08)	-	0.04 (0.02)
Rainbow Trout *Oncorhynchus mykiss*†			
Juvenile	0.42 (0.09)	0.08 (0.04)	0.04 (0.03)
Adult	0.54 (0.07)	0.05 (0.03)	0.06 (0.03)
Gadidae			
Burbot *Lota lota*†			
Juvenile	0.19 (0.14)	0.19 (0.14)	-
Adult	0.12 (0.09)	0.61 (0.14)	-
Cottidae			
Mottled Sculpin *Cottus bairdii*	0.27 (0.07)	0.01 (0.01)	0.10 (0.04)
	Kootenai River		
Cyprinidae			
Northern Pikeminnow *Ptychocheilus oregonensis*			
Juvenile	0.78 (0.05)	0.86 (0.04)	0.02 (0.01)
Adult	0.69 (0.05)	0.46 (0.05)	-
Peamouth *Mylocheilus caurinus*			
Juvenile	0.78 (0.18)	0.06 (0.10)	-
Adult	0.80 (0.06)	-	-
Redside Shiner *Richardsonius balteatus*	0.63 (0.06)	0.40 (0.05)	-
Catostomidae			
Largescale Sucker *Catostomus macrocheilus*			
Juvenile	0.57 (0.07)	0.10 (0.03)	0.02 (0.02)
Adult	0.87 (0.03)	-	-

Table 3. Continued.

	Detection probability ($\hat{p}$)		
Taxa and life history stage	**EF**	**HN**	**TR**
Longnose Sucker *Catostomus catostomus*			
Juvenile	0.36 (0.11)	0.07 (0.04)	-
Adult	0.39 (0.09)	0.01 (0.01)	-
Ictaluridae			
Brown Bullhead *Ameiurus nebulosus*†	0.05 (0.04)	0.14 (0.08)	-
Salmonidae			
Kokanee *Oncorhynchus nerka*			
Juvenile	0.44 (0.09)	-	0.01 (0.02)
Adult	0.29 (0.13)	-	-
Mountain Whitefish			
Juvenile	0.57 (0.05)	0.03 (0.01)	0.38 (0.05)
Adult	0.68 (0.05)	0.01 (0.01)	0.01 (0.01)
Rainbow Trout			
Juvenile	0.43 (0.14)	-	-
Adult	0.24 (0.09)	0.02 (0.02)	-
Gadidae			
Burbot			
Adult	-	0.12 (0.03)	-
Cottidae			
Torrent Sculpin *Cottus rhotheus*	0.55 (0.05)	0.11 (0.03)	0.02 (0.02)
Centrarchidae			
Pumpkinseed *Lepomis gibbosus*†	0.26 (0.12)	0.05 (0.03)	-
Percidae			
Yellow Perch *Perca flavescens*†	0.40 (0.14)	0.07 (0.05)	-

Missouri trawl captured significantly greater numbers of species and larger-bodied species than a traditional Missouri trawl in Pennsylvania rivers. Future research could explore the use of electrified benthic trawls in coldwater systems to improve detection probabilities of benthic, large-bodied fishes. Overall, benthic trawls used in our study appear to contribute little to fish assemblage surveys in western rivers with high water clarity.

Similarities in the effectiveness of sampling techniques were apparent among species shared between river systems. Detection probabilities were greatest with electrofishing for both juvenile and adult mountain whitefish and rainbow trout in both systems. Further, hoop nets had the greatest detectability for adult burbot in both systems. Species-specific trends in gear effectiveness are not surprising given that their ecology is probably similar among systems. For example, previous research supports our findings that burbot are effectively sampled in rivers with hoop nets given their affinity for seeking shelter in benthic habitats (Bernard et al. 1991; Klein et al. 2015). While our research compared sampling techniques across two coldwater rivers, our findings are likely applicable to similar systems across western North America. Moreover, due to a paucity of research on different sampling gears in coldwater river systems, our study provides important guidance to fisheries managers regarding the selection of appropriate gears.

The effect of gear type on detectability was apparent for most species. Gear-specific or constant detectability models had more support than models incorporating environmental covariates. Gear effects are not surprising, as gear biases have been previously reported for a variety of passive (Hubert et al. 2012) and active (Hayes et al. 2012; Reynolds and Koltz 2012) sampling techniques (Bonar et al. 2009a). Gear biases are likely related to species ecology and explain differences in species-specific detection probabilities among techniques. For example, detectability of adult white sucker in the Green River varied among techniques (gear, detection probability; electrofishing, 0.70; hoop nets, 0.12; trawls, 0.09). Our results are similar to Fischer (2012), who compared detection probabilities of seven sampling techniques for fish assemblages in reservoirs and natural lakes in the midwestern region of the United States. Fischer (2012) found species-specific detection probabilities varied among gears. For instance, detectability of black bullhead *Ameiurus melas* was low for most gears (<0.40), but was highest for fyke nets (~ 0.80). Behavioral responses to sampling techniques may also influence detection probabilities of some species. Use of dead fishes may have reduced the probability of detection of conspecifics in hoop nets. Previous studies have documented alarm responses and potentially avoidance when odorants are present (e.g., fathead minnow; Chivers and Smith 1994). Results of our research support

Table 4. Top candidate models ($2 \leq \Delta$ Akaike's Information Criterion [AIC$_c$] or quasi-Akaike's Information Criterion [QAIC$_c$]) and global model (italicized) for each candidate suite investigating the effect of sample- and site-scale habitat characteristics on detection probabilities ($\hat{p}$) for fishes collected the Kootenai and Green rivers in the summers of 2012 and 2013. Occupancy models for large-bodied species were separated by life history stage (i.e., juvenile, adult). Parameter estimates and standard errors (in brackets) are provided for all gear variables (electrofishing [EF], hoop nets [HN], trawl [TR]. Both site- [percentage of bank consisting of large substrate (Bk$_{Bo}$), percentage of bank consisting of vegetation (Bk$_{Veg}$), coefficient of variation in water column velocity (CV$_{Vel}$), average depth (Depth), mean discharge (Q), percentage of substrate consisting of cobble and boulder (Sub$_{Co\text{-}Bo}$), mean water column velocity (Vel$_{Avg}$), average substrate velocity (Vel$_{Sub}$), mean channel width (Width)] and sample-scale [water clarity (Clarity), a metric of water clarity and average depth (Clarity$_{Mx}$), mean temperature (Temp)] habitat variables were used to investigate relationships between occupancy and detectability of fishes. Parameter estimates of gear are gear-type specific. Species having greater support with a constant detectability model are indicated with the gear term. Akaike's Information Criterion (AIC$_c$ or QAIC$_c$) adjusted for small sample size was used to rank candidate models. Total number of parameters (*K*), Akaike model weight (w_i), and two times the log-likelihood (*-2Log(l)*) are included. A Pearson chi-square statistic and parametric bootstrap was used to evaluate goodness-of-fit (GOF). A *p*-value from the GOF test and variance inflation ($\hat{c}$) factor are provided. A *p*-value greater than 0.05 indicates there is no evidence of a lack of model fit. An asterisk next to a species name or life history stage indicates collection with only one technique in a system. Parameter estimates are on the logit scale.

Taxa and life history stage	Model name	AIC$_c$ or QAIC$_c$	Δ AIC$_c$ or QAIC$_c$	*K*	w_i	*-2Log(l)*	*p-value*	$\hat{c}$
	Green River							
Cyprinidae								
Common Carp *Cyprinus carpio*								
Adult*	$\hat{p}$(−4.4 [2.9]EF, −7.7 [4.4]Sub$_{Co-Bo}$)	45.88	0.00	4	0.49	36.14	0.68	0.55
	$\hat{p}$(−28.9 [8.5]EF, 17.9 [5.3]Depth, 0.8 [0.3]Temp, 15.1 [7.7]Vel$_{Avg}$)	50.94	5.06	6	0.08	34.94	0.45	0.94
Speckled dace *Rhinichthys osculus*								
	$\hat{p}$(−0.7 [0.7]EF, −1.9 [0.8]HN, 0.1 [0.7]TR, −2.7 [0.9]Depth, 2.9 [1.2]Vel$_{Sub}$)	255.22	0.00	6	0.32	239.22	0.61	0.23
Utah Chub *Gila atraria**								
	$\hat{p}$(2.3 [3.3]EF, −8.3 [6.1]Vel$_{Avg}$)	35.82	0.00	4	0.65	26.08	0.62	0.69
	$\hat{p}$(1.5 [4.7]EF, −8.8 [6.8]Vel$_{Avg}$, 2.1 [8.7]CV$_{Vel}$)	38.75	2.93	5	0.15	26.02	0.68	0.65
Catostomidae								
Flannelmouth Sucker *Catostomus latipinnis*								
Adult	$\hat{p}$(−1.2 [0.6]EF, −3.5 [1.1]TR)	59.88	0.00	4	0.41	51.88	0.23	0.79
	$\hat{p}$(−0.5 [1.4]EF, −2.8 [1.6]TR, −1.0 [1.9]Depth)	61.58	1.70	5	0.18	51.58	0.17	0.95
	$\hat{p}$(−1.0 [1.4]EF, −3.3 [1.7]TR, −0.5 [3.3]Vel$_{Sub}$)	61.86	1.98	5	0.15	51.86	0.19	0.76
	$\hat{p}$(−1.0 [2.0]EF, −3.4 [2.1]TR, −0.3 [4.0]CV$_{Vel}$)	61.87	1.99	5	0.15	51.87	0.20	0.92
	$\hat{p}$(−4.3 [4.2]EF, −6.7[4.4]TR, 3.4 [3.2]Clarity, −5.4 [6.4]CV$_{Vel}$)	62.72	2.84	6	0.10	50.72	0.37	0.51
White Sucker *Catostomus commersonii*								
Juvenile	$\hat{p}$(−0.2 [0.3]EF, −2.0 [0.4]HN, −4.1 [1.0]TR)	167.62	0.00	5	0.33	154.89	0.35	0.59
	$\hat{p}$(−0.7 [0.5]EF, −2.5 [0.6]HN, −4.6 [1.1]TR, 0.1 [0.1]Sub$_{Co-Bo}$)	168.87	1.25	6	0.18	152.87	0.39	0.23
	$\hat{p}$(−0.1 [1.0]EF, −1.9 [1.1]HN, −4.0 [1.4]TR, 0.4 [1.2]Depth, −0.7 [1.4]Vel$_{Avg}$)	174.25	6.63	7	0.01	154.65	0.35	0.19
Adult	$\hat{p}$(3.9 [1.4]EF, 1.0 [1.4]HN, 0.67[1.4]TR, −1.7 [0.8]Clarity)	215.33	0.00	5	0.53	202.60	0.44	0.15
	$\hat{p}$(5.8 [1.9]EF, 2.8 [1.8]HN, 2.5 [1.9]TR, −2.0 [0.8]Clarity, −0.2 [1.0]Depth, −1.7 [1.1]Vel$_{Avg}$)	218.96	2.74	7	0.09	199.36	0.35	0.31
Salmonidae								
Brown Trout *Salmo trutta*								
Juvenile	$\hat{p}$(−0.9 [0.3]EF, −3.4 [0.7]HN, −2.7 [0.5]TR)	57.14	0.00	5	0.39	130.42	0.03	2.38
	$\hat{p}$(−4.1 [2.8]EF, −6.6 [3.0]HN, −5.9 [2.9]TR, 3.3 [4.6]CV$_{Vel}$, −0.1 [0.1]Sub$_{Co-Bo}$, 2.7 [1.9]Vel$_{Avg}$)	68.29	11.15	9	0.00	127.71	0.03	3.17

Table 4. Continued.

Taxa and life history stage	**Model name**	**AIC$_c$ or QAIC$_c$**	**Δ AIC$_c$ or QAIC$_c$**	***K***	**w_i**	***-2Log(l)***	***p-value***	**$\hat{c}$**
Adult	$\hat{p}$(−0.7 [0.3]EF, −4.1 [1.0] HN, −2.9 [0.6]TR)	134.51	0.00	5	0.36	121.78	0.19	0.48
	$\hat{p}$(0.8 [1.0]EF, −2.6 [1.3]HN, −1.5 [1.1]TR, −2.3 [1.5]Vel$_{\text{Avg}}$)	135.35	0.84	6	0.24	119.35	0.41	0.13
	$\hat{p}$(1.53 [3.2]EF, −1.9 [3.4]HN,−0.8 [3.3]TR, −1.1 [3.8]CV$_{\text{Vel}}$, −0.1 [1.9]Depth, −2.7 [1.9]Vel$_{\text{Avg}}$)	142.83	8.32	8	0.01	119.25	0.40	0.24
Cutthroat Trout *Oncorhynchus clarkii*								
Juvenile	$\hat{p}$(−13.1 [11.1]Gear, 0.7 [0.7]Temp)	50.59	0.00	3	0.34	43.59	0.65	0.04
	$\hat{p}$(−2.8 [1.0]Gear)	52.42	1.83	2	0.14	47.94	0.28	0.55
	$\hat{p}$(−16.4 [5.4]Gear, 4.2 [4.6]Depth, −0.1 [0.1]Sub$_{\text{Co−Bo}}$, 0.8 [0.3]Temp)	55.22	4.63	5	0.03	42.49	0.72	0.01
Adult	$\hat{p}$(−0.8 [0.4]EF, −3.5 [0.7]HN, −4.2 [1.0]TR)	132.70	0.00	4	0.31	122.96	0.70	0.11
	$\hat{p}$(1.1 [2.9]EF, −1.7 [2.9]HN, −2.4 [3.0]TR, −0.3 [1.0]Clarity, −0.9 [3.3]CV$_{\text{Vel}}$, −1.0 [1.6]Depth)	142.09	9.39	7	0.01	122.49	0.66	0.06
Mountain Whitefish *Prosopium williamsoni*								
Juvenile	$\hat{p}$(−1.4 [0.2]Gear)	186.52	0.00	4	0.32	176.78	0.30	0.50
	$\hat{p}$(−5.9 [1.6]Gear, −0.2 [1.2]Depth, 0.1 [0.1]Sub$_{\text{Co−Bo}}$, 0.3 [0.1]Temp)	189.76	3.24	6	0.01	173.76	0.85	0.24
Adult	$\hat{p}$(−2.5 [1.0]EF, −5.7 [1.3]TR, 3.6 [3.1]*CV$_{\text{Vel}}$*)	139.09	0.00	5	0.30	126.36	0.75	0.22
	$\hat{p}$(−3.0 [1.1]EF, −6.3 [1.4]TR, 5.6 [2.5]CV$_{\text{Vel}}$, 0.1 [0.1]Sub$_{\text{Co−Bo}}$)	139.73	0.64	6	0.22	123.73	0.78	0.10
	$\hat{p}$(2.1 [1.1]EF, −1.1 [1.1]TR, −2.8 [1.3]Depth)	140.55	1.46	5	0.14	127.82	0.85	0.16
Rainbow Trout *Oncorhynchus mykiss*								
Juvenile	$\hat{p}$(−2.7 [1.0]EF, −4.9 [1.2]HN, −5.7[1.3]TR, 3.1 [1.3]Vel$_{\text{Avg}}$)	146.92	0.00	5	0.31	134.19	0.20	0.55
	$\hat{p}$(2.0 [1.1]EF, −0.2 [1.1]HN, −0.9 [1.2]TR, −6.1 [2.7]CV$_{\text{Vel}}$)	147.47	0.55	5	0.23	134.74	0.25	0.43
	$\hat{p}$(1.0 [2.9]EF, −1.2 [2.9]HN, −2.0 [2.9]TR, −4.5 [3.7]CV$_{\text{Vel}}$, −2.2 [2.3]Depth, 2.8 [1.8]Vel$_{\text{Avg}}$)	152.30	5.11	7	0.02	132.43	0.23	0.53
Adult	$\hat{p}$(0.2 [0.3]EF, −3.0 [0.6]HN, −2.7 [0.5]TR)	171.22	0.00	5	0.29	158.49	0.38	0.10
	$\hat{p}$(1.5 [1.2]EF, −1.7 [1.3]HN, −1.4 [1.2]TR, −0.1 [0.1]Temp)	173.09	1.87	6	0.11	157.09	0.36	0.29
Gadidae								
Burbot *Lota lota*	*$\hat{p}$(2.5 [2.5]EF, −0.8 [2.5]HN, −0.5 [2.5]TR, −0.5 [0.9]Clarity, −0.9 [3.3]CV$_{\text{Vel}}$, −1.3 [1.6]Depth)*	180.59	9.37	8	0.00	157.01	0.37	0.29
Juvenile	$\hat{p}$(−1.4 [0.7]Gear)	36.32	0.00	3	0.19	29.32	0.56	0.28
	$\hat{p}$(−9.4 [7.8]Gear, 0.6 [0.6]Temp)	37.12	0.80	4	0.13	27.38	0.28	0.22
	$\hat{p}$(−41.9 [4.8]Gear, 25.7 [8.7]CV$_{\text{Vel}}$, 7.7 [7.3]Depth, 1.7 [0.3]Temp)	41.19	4.87	6	0.01	25.19	0.12	0.87
Adult	$\hat{p}$(−2.0 [0.4]EF, 0.4 [0.3]HN)	40.57	0.00	3	0.32	168.02	0.03	5.32
	$\hat{p}$(−0.1 [0.7]EF, 2.4 [0.8]HN, −3.9 [1.4]Vel$_{\text{Sub}}$)	40.05	1.48	4	0.15	159.81	0.01	5.89
	$\hat{p}$(−1.4 [1.3]EF, 1.1 [1.4]HN, −4.8 [1.7]Vel$_{\text{Sub}}$, 0.1 [0.1]Temp)	45.15	4.58	6	0.03	158.86	0.02	5.45
Cottidae								
Mottled Sculpin *Cottus bairdii*								
	$\hat{p}$(1.2 [0.8]EF, −2.3 [1.2]HN, −0.1 [0.8]TR, −4.4 [1.6]Vel$_{\text{Sub}}$)	163.30	0.00	5	0.77	150.57	0.89	0.08
	$\hat{p}$(3.9 [2.3]EF,0.4 [2.4]HN, 2.6 [2.2]TR, −1.5[1.1]Clarity, 0.1 [0.1]Sub$_{\text{Co−Bo}}$, −6.0 [1.9]Vel$_{\text{Sub}}$)	166.41	2.74	7	0.19	146.44	0.83	0.12

Table 4. Continued.

Taxa and life history stage	Model name	AIC$_c$ or QAIC$_c$	Δ AIC$_c$ or QAIC$_c$	K	w_i	-2Log(l)	p-value	$\hat{c}$
Kootenai River								
Cyprinidae								
Northern Pikeminnow *Ptychocheilus oregonensis*								
Juvenile	$\hat{p}$(1.1 [0.8]EF, 1.8 [0.9]HN, −4.5 [1.1]TR, 0.1 [0.1]Depth, −3.8 [1.2]Sub$_{Co-Bo}$)	216.44	0.00	6	0.51	202.11	0.26	0.68
Adult	$\hat{p}$(2.6 [0.5]EF, 1.6 [0.4]HN, −5.4 [1.2]CV$_{Vel}$)	258.75	0.00	5	0.73	247.13	0.74	0.90
	$\hat{p}$(3.0 [1.1]EF, 1.9 [1.0]HN, −0.1 [0.2]Clarity, −5.5 [1.3]CV$_{Vel}$, −0.1 [0.1]Depth)	264.17	5.42	7	0.05	246.97	0.69	0.90
Peamouth *Mylocheilus caurinus*								
Juvenile	$\hat{p}$(1.3 [0.3]EF, −2.7 [0.4]HN)	18.32	0.00	4	0.32	156.94	0.01	16.67
	$\hat{p}$(1.2 [1.4]EF, −3.1 [1.5]HN, −0.6 [0.3]Clarity, 0.2 [0.1]Depth, 0.1 [0.1]Bk$_{Veg}$)	26.48	8.16	8	0.01	146.20	0.01	23.46
Adult*	$\hat{p}$(2.7 [0.8]EF, −5.4 [1.6]CV$_{Vel}$, 0.1 [0.1]Bk$_{Veg}$)	68.89	0.00	6	0.49	85.67	0.09	1.57
	$\hat{p}$(3.6 [0.7]EF, −5.8 [1.6]CV$_{Vel}$)	69.24	0.35	5	0.41	90.46	0.03	2.00
Redside Shiner *Richardsonius balteatus*								
	$\hat{p}$(0.5 [0.2]EF, −0.4 [0.2]HN)	233.64	0.00	3	0.32	303.19	0.02	1.40
	$\hat{p}$(0.9 [0.4]EF, −0.1 [0.4]HN, 0.1 [0.1]Bk$_{Bo}$, −0.8 [0.9]CV$_{Vel}$, −0.1 [0.1]Q)	241.16	7.52	7	0.01	302.35	0.04	1.32
Largescale Sucker *Catostomus macrocheilus*								
Juvenile	$\hat{p}$(−1.8 [0.4]EF, −4.3 [0.5]HN, −5.8 [1.2]TR, 0.1 [0.1]Q)	223.95	0.00	5	0.42	212.33	0.29	0.68
	$\hat{p}$(0.1 [0.9]EF, −2.5 [0.9]HN, −3.9 [1.2]TR, 0.1 [0.1]Clarity, −1.0 [1.6]CV$_{Vel}$, 0.1 [0.1]Depth)	230.81	6.85	7	0.03	213.60	0.41	0.60
Adult	$\hat{p}$(2.4 [1.1]EF, −0.2 [0.1]Depth, 0.1 [0.1]Bk$_{Veg}$)	91.91	0.00	5	0.63	80.29	0.69	0.75
	$\hat{p}$(1.3 [1.1]EF, 0.1 [0.1]Q, −0.3 [0.1]Depth, 0.1 [0.1]Bk$_{Veg}$)	93.82	1.91	6	0.24	79.49	0.69	0.78
Longnose Sucker *Catostomus catostomus*								
Juvenile	$\hat{p}$(0.7 [0.9]EF, −1.3 [1.0]HN, −0.5 [0.3]Clarity)	114.78	0.00	4	0.20	105.73	0.62	0.64
	$\hat{p}$(−0.6 [0.5]EF, −2.6 {0.6]HN)	115.20	0.42	3	0.16	108.58	0.61	0.59
	$\hat{p}$(−2.3 [0.6]EF, −4.3 [0.8]HN, 0.1 [0.1]Q)	115.97	1.19	4	0.11	106.92	0.54	0.67
	$\hat{p}$(3.7 [3.5]EF, 1.6 [3.4]HN, −0.3 [0.2]Temp)	116.25	1.47	4	0.10	107.20	0.58	0.71
	$\hat{p}$(−0.4 [0.5]EF, −2.4 [0.7]HN, −1.2 [1.4]Sub$_{Co-Bo}$)	116.78	2.00	4	0.08	107.73	0.62	0.67
	$\hat{p}$(−2.0 [1.3]EF, −4.0 [1.4]HN, −0.1 [0.2]Depth, 0.1 [0.1]Q, −0.1 [0.1]Bk$_{Veg}$)	121.00	6.22	6	0.01	106.67	0.52	0.76
Adult	$\hat{p}$(1.1] [1.0]EF, −2.8 [1.3]HN, −0.6 [0.3]Clarity)	113.60	0.00	4	0.25	127.47	0.29	1.12
	$\hat{p}$(−0.4 [0.4]EF, −4.3 [1.0]HN)	113.92	0.32	3	0.21	131.09	0.48	0.89
	$\hat{p}$(−0.1 [0.5]EF, −3.9 [1.1]HN, −0.1 [0.1]Bk$_{Veg}$)	115.23	1.63	4	0.11	129.51	0.33	1.02
	$\hat{p}$(−0.4 [0.4]EF, −4.2 [1.0]HN, −2.1 [1.7]Sub$_{Co-Bo}$)	115.34	1.74	4	0.10	129.65	0.43	0.94
	$\hat{p}$(−1.3 [0.8]EF, −5.2 [1.3]HN, 0.1 [0.1]Depth)	115.41	1.81	4	0.10	129.73	0.43	1.00
	$\hat{p}$(−0.8 [1.8]EF, −4.7 [2.1]HN, −0.6 [0.4]Clarity, 2.9 [2.3]CV$_{Vel}$, 0.1 [0.1]Depth)	117.77	4.17	7	0.03	128.50	0.26	1.25
Ictaluridae								
Brown Bullhead *Ameiurus nebulosus*								
	$\hat{p}$(−2.6 [0.6]Gear)	71.62	0.00	3	0.23	65.00	0.60	0.63
	$\hat{p}$(−3.0 [0.8]Gear, 3.0 [2.2]Sub$_{Co-Bo}$)	72.05	0.43	4	0.19	63.00	0.62	0.67

Table 4. Continued.

Taxa and life history stage	Model name	AIC$_c$ or QAIC$_c$	Δ AIC$_c$ or QAIC$_c$	*K*	w_i	*-2Log(l)*	*p-value*	$\hat{c}$
	$\hat{p}$(−1.6 [1.0]Gear, −0.15 [0.15]Depth)	73.09	1.47	4	0.11	64.04	0.40	0.92
	$\hat{p}$(−2.6 [0.7]Gear, 0.1 [0.1]Bk$_{Bo}$)	73.25	1.63	4	0.10	64.20	0.53	0.75
	$\hat{p}$(−2.6 [3.4]Gear, −2.3 [4.2]CV$_{Vel}$, 0.1 [0.3]Depth, 4.9 [3.7]Sub$_{Co-Bo}$)	76.82	5.20	6	0.02	62.49	0.81	0.53
Salmonidae								
Kokanee *Oncorhynchus nerka*								
Juvenile	$\hat{p}$(−0.3 [0.4]EF, −4.2 [1.0]HN)	114.61	0.00	4	0.39	128.73	0.26	1.10
	$\hat{p}$(0.1 [0.5]EF, −3.9 [1.1]HN, −0.1 [0.1]Bk$_{Veg}$)	116.51	1.90	5	0.15	127.72	0.26	1.20
	$\hat{p}$(−0.9 [1.5]EF, −4.9 [1.8]HN, 1.2 [1.8]CV$_{Vel}$,0.1 [0.2]Depth, −0.1 [0.1]Bk$_{Veg}$)	121.88	7.27	8	0.01	127.06	0.24	1.19
Adult*	*$\hat{p}$(9.4 [4.3]EF, −2.3 [1.1]Clarity, −10.7 [4.9]CV$_{Vel}$, −0.2 [0.2]Depth)*	81.56	0.00	7	0.22	73.37	0.30	1.14
	$\hat{p}$(−0.9 [0.6]EF)	81.65	0.09	3	0.21	82.76	0.56	0.88
	$\hat{p}$(2.9 [0.6]EF, 0.1 [0.1]Q)	83.18	1.62	4	0.10	81.58	0.61	0.83
	$\hat{p}$(0.5 [1.7]EF, −3.8 [4.4]CV$_{Vel}$)	83.31	1.75	4	0.09	81.72	0.55	0.88
	$\hat{p}$(0.9 [2.3]EF, −0.7 [0.9]Clarity)	83.53	1.97	4	0.08	81.97	0.57	0.88
Mountain Whitefish								
Juvenile	$\hat{p}$(0.3 [0.2]EF, −3.6 [0.60]HN, −0.5 [0.2]TR)	338.06	0.00	4	0.24	329.01	0.58	0.65
	$\hat{p}$(0.1 [0.6]EF, −3.9 [1.0]HN, −0.8 [0.6]TR, −0.1 [0.1]Depth, 0.1 [0.1]Q, −0.4 [0.9]Sub$_{Co-Bo}$)	344.82	6.76	7	0.01	327.62	0.54	0.66
Adult	$\hat{p}$(1.3 [1.2]EF, −5.1 [1.7]HN, −5.1 [1.7]TR, 0.5 [0.1]Clarity, −0.3 [0.1]Depth)	156.94	0.00	6	0.34	142.61	0.16	1.16
	$\hat{p}$(3.2 [0.7]EF, −2.7 [1.1]HN, −2.7 [1.1]TR, −0.4 [0.1]Depth)	158.05	1.11	5	0.19	146.43	0.20	1.08
	$\hat{p}$(2.4 [0.9]EF, −4.0 [1.5]HN, −4.0 [1.5]TR, −0.3 [0.1]Depth, 2.5 [1.8]Sub$_{Co-Bo}$)	158.74	1.80	6	0.14	144.41	0.12	1.77
	$\hat{p}$(1.5 [1.3]EF, −4.8 [1.7]HN,−4.8 [1.7]TR, −0.1 [0.1]Bk$_{Bo}$, 0.59 [0.3]Clarity, −0.4 [0.1]Depth)	159.56	2.62	7	0.10	142.14	0.16	0.64
Rainbow Trout								
Juvenile*	$\hat{p}$(−4.3 [0.5]EF, 0.1 [0.1]Q)	85.04	0.00	4	0.23	85.16	0.09	1.57
	$\hat{p}$(−0.3 [0.5]EF)	85.87	0.83	3	0.15	89.11	0.22	1.27
	$\hat{p}$(−4.7 [0.6]EF, 0.1 [0.1]Bk$_{Bo}$, 0.1 [0.1]Q)	86.92	1.88	5	0.09	84.20	0.08	1.69
	$\hat{p}$(−8.0 [1.7]EF, 0.1 [0.1]Bk$_{Bo}$ 0.3 [0.2]Depth, 0.1 [0.1]Q)	89.42	4.38	7	0.03	83.78	0.42	1.00
Adult	$\hat{p}$(1.8 [1.2]EF, −1.1 [1.3]HN, −0.6 [0.3]Clarity, −0.3 [0.1]Depth)	143.64	0.00	5	0.25	132.02	0.26	0.99
	$\hat{p}$(2.1 [1.2]EF, −0.8 [1.3]HN, −0.1 [0.1]Bk$_{Bo}$, −0.4 [0.3]Clarity, −0.3 [0.1]Depth)	144.05	0.41	6	0.20	129.72	0.31	0.83
Gadidae								
Burbot								
Adult*	$\hat{p}$(−2.0 [0.3]HN)	73.87	0.00	3	0.38	73.89	0.34	1.05
	$\hat{p}$(−2.7 [0.7]HN, 0.1 [0.1]Bk$_{Veg}$)	74.84	0.97	4	0.24	72.07	0.48	0.87
	$\hat{p}$(−0.9 [0.4]HN, −0.1 [0.1]Q)	75.81	1.94	4	0.15	73.18	0.29	1.13
	$\hat{p}$(−4.1 [10.3]HN, 0.1 [0.3]Depth, −0.6 [2.5]Sub$_{Co-Bo}$, 0.1 [0.6]Temp)	82.44	8.57	7	0.01	74.37	0.30	1.14

Table 4. Continued.

Taxa and life history stage	Model name	AIC$_c$ or QAIC$_c$	Δ AIC$_c$ or QAIC$_c$	*K*	w_i	*-2Log(l)*	*p-value*	$\hat{c}$
Cottidae								
Torrent Sculpin *Cottus rhotheus*								
	$\hat{p}$(−0.5 [0.4]EF, −2.8 [0.5]HN, −4.5 [0.8]TR, 1.8 [1.0]CV$_{Vel}$)	209.43	0.00	5	0.24	196.93	0.28	0.82
	$\hat{p}$(0.2 [0.2]EF, −2.1 [0.3]HN, −3.7 [0.7]TR)	209.86	0.43	4	0.20	200.26	0.28	0.71
	$\hat{p}$(0.3 [0.2]EF, −2.0 [0.3]HN, −3.6 [0.7]TR, −1.2 [0.9]Sub$_{Co-Bo}$)	210.65	1.22	5	0.13	198.15	0.32	0.72
	$\hat{p}$(−0.3 [0.5]EF, −2.6 [0.6]HN, −4.3 [0.8]TR, 0.1 [0.1]Depth)	210.95	1.52	5	0.11	198.45	0.30	0.78
	$\hat{p}$(−0.8 [1.0]EF, −3.1 [1.1]HN, −4.8 [1.3]TR, 0.1 [0.1]Bk$_{Bo}$, 0.1 [0.2]Clarity, 0.1 [0.1]Depth)	216.78	7.35	7	0.01	197.69	0.31	0.72
Centrarchidae								
Pumpkinseed *Lepomis gibbosus*								
	$\hat{p}$(−1.0 [0.5]EF, −3.0 [0.7]HN)	85.93	0.00	3	0.26	117.63	0.14	1.38
	$\hat{p}$(−0.7 [0.6]EF, −2.6 [0.7]HN, −0.1 [0.1]Bk$_{Veg}$)	87.54	1.61	4	0.12	116.16	0.12	1.41
	$\hat{p}$(−0.8 [0.6]EF, −2.5 [0.8]HN, −0.1 [0.1]Clarity)	87.62	1.69	4	0.11	116.27	0.14	1.45
	$\hat{p}$(−0.3 [1.9]EF, −2.3 [1.9]HN, −1.4 [2.4]CV$_{Vel}$, −0.1 [0.2]Depth, −0.9 [2.6]Sub$_{Co-Bo}$)	93.53	7.60	7	0.01	116.79	0.13	1.53
Percidae								
Yellow Perch *Perca flavescens*								
	$\hat{p}$(−0.4 [0.6]EF, −2.6 [0.8]HN)	90.41	0.00	3	0.25	83.79	0.36	0.88
	$\hat{p}$(1.2 [1.4]EF, −1.0 [1.4]HN, −0.7 [0.5]Clarity)	91.21	0.80	4	0.17	82.16	0.51	0.76
	$\hat{p}$(−1.8 [1.6]EF, −3.9 [1.7]HN, 0.2 [0.2]Depth)	92.05	1.64	4	0.11	83.00	0.43	0.89
	$\hat{p}$(−0.3 [0.6]EF, −2.6 [0.8]HN, −2.0 [2.7]Sub$_{Co-Bo}$)	92.33	1.92	4	0.10	83.28	0.40	0.86
	$\hat{p}$(−0.8 [0.9]EF, −3.1 [1.1]HN, 0.1 [0.1]Bk$_{Veg}$)	92.36	1.95	4	0.09	83.31	0.29	1.07
	$\hat{p}$(1.3 [0.7]EF, −0.9 [1.0]HN, −0.1 [0.1]Q)	92.38	1.97	4	0.09	83.67	0.39	0.86
	$\hat{p}$(0.5 [2.7]EF, −1.7 [2.8]HN, −0.6 [0.6]Clarity, −1.0 [3.8]CV$_{Vel}$, 0.1 [0.2]Depth)	95.92	5.51	6	0.01	81.59	0.52	0.79

previous work indicating species-specific detection probabilities vary greatly by gear and differences are likely related to gear biases and species ecology. Our findings underscore the importance of investigating gear-specific detection probabilities in an effort to improve gear selection and sampling efficiency for future fish assemblage surveys.

Although detectability of many fishes in the present study was best explained by gear, relationships with some environmental characteristics were apparent although their effects were highly variable. Most relationships with species-specific detection probabilities and habitat characteristics were idiosyncratic and reflected differences in the ecology of collected species. For instance, detectability of speckled dace in the Green River was negatively related to depth and likely reflects their affinity for benthic habitats (Rinne 1992). Overall, depth and the proportion of cobble or boulder substrate had widespread effects on the detectability of fishes. For example, depth effects were selected among the top candidate models for approximately half of the species in the Kootenai River samples. Depth has previously been reported to influence sampling efficiency (Falke et al. 2010; Reynolds and Kolz 2012). Dauwalter and Fisher (2007) reported negative relationships between capture probabilities of smallmouth bass *Micropterus dolomieu* and mean thalweg depth in Oklahoma streams. In addition to direct effects of habitat on detection probabilities, habitat may influence another component of detectability, abundance (Royle and Nichols 2003). Relationships of abundance and detection probabilities have been widely proposed in previous research (Schloesser et al. 2012; Haynes et al. 2013; Jensen and Vokoun 2013). Despite an inability to directly model abundance with detectability in our study, the inclusion of important habitat covariates in occupancy models likely indirectly account for some differences in detectability due to shifts in fish abundance (Haynes et al. 2013). Understanding and identifying habitat variables that may influence abundance is important in occupancy models, particularly when independent estimates of abundance are unavailable.

Size-related biases are common for most sampling techniques, particularly active gears, and increased total length generally increases susceptibility for most gears (Bonar et al. 2009a; Reynolds and Kolz 2012; Hubert et al. 2012). For instance, total length of smallmouth bass was positively related to the probability of electrofishing capture in small lotic systems of Oklahoma (Dauwalter and Fisher 2007). Although detectability of some life stages were related to habitat in a different manner, habitat relationships were similar for some species. For example, detectability of both juvenile and adult longnose sucker was related to water clarity in the Kootenai River. Water clarity has been reported to influence catch of fishes with our gears (Bonar et al. 2009a; Guy et al. 2009; Hubert et al. 2012; Reynolds and Kolz 2012). Stone (2010) reported a positive relationship with hoop net catch of native fishes in the Colorado River and water clarity. However, electrofishing efficiency may be negatively related to water clarity in systems with traditionally high water clarity (Dewey 1992; Reynolds and Kolz 2012).

Our research confirms differences in detectability among gears and fishes, and illustrates the importance of understanding species-specific detection probabilities. Overall, electrofishing and hoop nets were the most effective techniques in coldwater river systems. Trawls were less effective than hoop nets and electrofishing, and sampled a redundant portion of the fish assemblage. General trends in gear efficacy appear consistent for species and families across systems. However, differences in fish assemblage structure and habitat characteristics suggest that further investigations focusing on the influence of habitat on detectability are needed in other systems. An understanding of detection probabilities and the influence of environmental characteristics will allow managers to improve sampling designs and efficiency. Our study indicates that boat-mounted electrofishing techniques and hoop nets are an effective means of sampling fish assemblages in coldwater river systems and allows fisheries scientists to select appropriate sampling techniques for different management and conservation goals.

Supplemental Material

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Table S1. Worksheet that includes the river system, reach identification number, fish collection event, date, gear type, sampling effort (electrofishing, s; hoop nets, h; trawl, m), species and count collected from two river systems (Kootenai and Green rivers) in western North America from the summers of 2012 and 2013.

Found at DOI: http://dx.doi.org/10.3996/022015-JFWM-011.S1 (316 KB XLSX)

Table S2. Table shows approximate length at maturity of large-bodied fishes (≥200 mm).

Found at DOI: http://dx.doi.org/10.3996/022015-JFWM-011.S2 (16 KB DOCX)

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This Online Early paper will appear in its final typeset version in a future issue of the *Journal of Fish and Wildlife Management*. This article has been accepted for publication and undergone full peer review but has not been through the copyediting, typesetting, pagination and proofreading process, which may lead to differences between this version and the Version of Record. The

findings and conclusions in this article are those of the author(s) and do not necessarily represent the views of the U.S. Fish and Wildlife Service.

Acknowledgments

We thank C. Brown, D. Donnely, W. Field, K. Griffin, J. Hansen, E. Landers, N. Porter, J. Walrath, C. Watkins, S. Whitlock, and J. Yates for assistance with field research. We also thank C. Gidley, P. Rust, and T.J. Ross of Idaho Department of Fish and Game and R. Keith, T. Neebling, D. Rhea, A. Senecal, and H. Sexauer of Wyoming Game and Fish Department for assistance planning and implementing field work. Assistance with occupancy models was received from J. Nichols of the Patuxent Wildlife Research Center. Helpful comments on a previous version of the manuscript were provided by J. Falke and three anonymous reviewers. Funding for the project was provided by Idaho Department of Fish and Game. Additional support was provided by the U.S. Geological Survey, Idaho Cooperative Fish and Wildlife Research Unit. The Unit is jointly sponsored by the U.S. Geological Survey, University of Idaho, and Idaho Department of Fish and Game, and Wildlife Management Institute. This project was conducted under the University of Idaho Institutional Animal Care and Use Committee Protocol 2011-33.

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