

## Comparison of a prepositioned areal electrofishing device and fixed underwater videography for sampling riverine fishes

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**ABSTRACT.**—Prepositioned areal electrofishing devices (PAEDs) are used to evaluate microhabitat use by fishes because they minimize fright biases associated with traditional electrofishing techniques (e.g., boat electrofishing). Similarly, fixed underwater videography (FUV) is commonly used to minimize the effect of observers on fish behavior. The specific objectives of this research were to evaluate estimates of taxonomic occurrence and diversity between PAEDs and FUV and determine an appropriate time interval between positioning and electrifying of a PAED to reduce effects of PAED positioning on fish occurrence. Video cameras were positioned instream at 28 locations on the Kootenai River, Idaho, prior to PAED deployment such that the entire immobilization zone of the PAED was captured on camera. Following a 4-min acclimation period, cameras recorded fish behavior approximately 15 min prior to and 20 min following PAED deployment. Electrical current was applied to the PAEDs for 20 s immediately following the FUV procedure, and immobilized fishes were collected and processed. Video footage was subsampled in the laboratory, and fishes in the video were identified and enumerated in 5-s or 20-s intervals. Fixed underwater videography sampled more taxa than PAEDs at any given site. However, fishes sampled with FUV were difficult to identify, and most individuals were classified as “unidentifiable.” Consequently, direct comparisons between FUV and PAEDs are limited. Our results indicate that PAEDs should remain undisturbed for a minimum of 12 min before the equipment is electrified. Both PAEDs and FUV provide an estimate of taxonomic occurrence, but logistical and financial constraints along with project objectives must be considered when selecting between these 2 gear types. Results from this study provide information on the effectiveness of each gear type as it relates to the characterization of riverine fish assemblages at a small spatial scale.

**RESUMEN.**—Los dispositivos de electropesca para áreas preposicionadas (PAED, por sus siglas en inglés) se utilizan para evaluar el uso que los peces hacen de los microhábitats, dado que minimizan la tendencia de los peces a huir asociada a las técnicas tradicionales de electropesca (e.g., embarcaciones de electropesca). De manera similar, la videografía submarina fija (FUV, por sus siglas en inglés) se utiliza, comúnmente, para minimizar el impacto de los observadores sobre el comportamiento de los peces. Los objetivos específicos de esta investigación fueron comparar los estimados obtenidos de presencia taxonómica y diversidad de los PAED y la FUV, y determinar un intervalo de tiempo apropiado entre la colocación y el electrificado de los PAED, que reduzca el efecto de la instalación de los PAED en la presencia de peces. Las cámaras de video se colocaron en 28 sitios dentro del río Kootenai, Idaho, previo a la instalación de los PAED, de tal forma que la cámara pudiera captar toda la zona de inmovilización de los PAED. Luego de un período de aclimatación de cuatro minutos, las cámaras registraron el comportamiento de los peces aproximadamente 15 minutos antes y 20 minutos después de la implementación de los PAED. Se aplicó corriente eléctrica a los PAED durante 20 segundos e inmediatamente después el procedimiento de la FUV, y se recogieron y procesaron los peces inmovilizados. El archivo de video fue submuestreado en el laboratorio, donde los peces fueron identificados y enumerados en intervalos de 5 a 20 segundos. La videografía submarina fija muestreó mayor cantidad de taxa que los PAED en cualquier sitio. Sin embargo, los peces muestreados con FUV fueron difíciles de identificar, siendo la mayoría clasificados como “no identificables”. Por lo tanto, las comparaciones directas entre las FUV y los PAED son limitadas. Nuestros resultados indican que los PAED deben permanecer intactos durante un mínimo de 12 minutos antes de electrificar el equipo. Tanto los PAED como las FUV proporcionan un estimado de presencia taxonómica. Sin embargo, las limitaciones logísticas y financieras, junto con los objetivos del proyecto deben ser considerados al elegir entre estos dos tipos de dispositivos. Los resultados de este estudio proporcionan información sobre la eficacia de cada tipo de dispositivo en lo que respecta a la caracterización de conjuntos de peces ribereños en una escala espacial pequeña.

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Understanding the relationship between fishes and their environment has long been an important focus of fish scientists (Rosenfeld 2003, Fisher et al. 2012). Resource use and partitioning by lotic fishes regulate the structure of fish assemblages, particularly at small spatial scales (Gorman and Karr 1978, Ross 1986). Aquatic habitats are often considered within the context of a hierarchy, where upper-level processes occur over long temporal and large spatial scales (e.g., a watershed) and lower-level processes occur over shorter temporal and smaller spatial scales (e.g., channel units; Quist et al. 2005). Microhabitat data are an integral component of this hierarchy because they reflect specific habitat conditions (e.g., depths, current velocities) selected by stream fishes and can be used to understand habitat requirements and distributions of fishes.

Reductions in habitat quality and quantity have been identified as primary factors that contribute to declining freshwater fauna populations across North America (Ricciardi and Rasmussen 1999). Large rivers have been extensively modified by water development (e.g., dams, levees, diversions) to serve societal needs (e.g., power generation, flood control, navigational routes; Nilsson et al. 2005, Dudgeon et al. 2006). In an effort to mitigate habitat degradation, natural resource and conservation agencies have implemented over 37,000 habitat improvement projects on rivers and streams in the United States (Bernhardt et al. 2005). Habitat improvement programs lead to species recovery, particularly by enhancing instream habitat or by reconnecting isolated habitats (Roni et al. 2008). An integral component of the habitat improvement process is to establish a monitoring program that can effectively evaluate the response of the fish assemblage to improvements (Kondolf and Micheli 1995, Palmer et al. 2005). Unfortunately, many monitoring programs have focused on the physical responses to instream habitat improvements and have neglected to thoroughly address the biological responses (Roni et al. 2002). Arguably, assessing the biological response to habitat improvement is perhaps the best measure of effectiveness, and obtaining an accurate measure of the response elicited by fishes starts by selecting suitable sampling equipment.

Quantifying microhabitat use by fishes requires careful selection of an appropriate

sampling gear. A variety of gears have been used to sample riverine fishes (e.g., seines, gill nets, electrofishing, snorkeling; Bonar et al. 2009). Electrofishing is generally considered the most effective gear type for sampling cold-water fishes in rivers because it minimizes the mortality, injury, and size selectivity that are often associated with other gears (e.g., gill nets and hoop nets; Curry et al. 2009). However, active electrofishing techniques (i.e., boat or backpack electrofishing) are conducted by personnel who maneuver the electrodes through the water and the resulting disturbance can drive fish from their original location (“fright bias”; Bovee 1982). Because microhabitat use occurs at a small scale, precautionary measures must be taken to sample fishes from their naturally selected habitats and not from habitats used in response to sampling.

To reduce the influence of fright bias, prepositioned areal electrofishing devices (PAEDs) are often used to sample fishes when evaluating microhabitat use (Bain et al. 1985, Reynolds and Kolz 2012, Dauwalter et al. 2014). Prepositioned areal electrofishing devices use fixed electrodes that are deployed prior to the sampling event and remain undisturbed for a period of time. After an allotted amount of time has passed (commonly referred to as the PAED “set time”), current is applied and the fish are immobilized. Typically, the design of a PAED is relatively inconspicuous compared to other passive sampling gears (e.g., hoop nets and trap nets), thereby minimizing the effect of gear avoidance or attraction by fishes. Additionally, PAEDs sample a discrete area, allowing the investigator to quantify habitat characteristics that reflect those selected by fishes at a small scale. An important assumption when using PAEDs is that fish return to the sampling area following the initial disturbance associated with deploying the gear.

Direct observation techniques are commonly used to characterize fish assemblages in natural environments (Thurrow et al. 2012). Direct observation methods have been used to evaluate fish distribution and abundance (Hankin and Reeves 1988), population size structure (Griffith 1981), and habitat use (Fausch and White 1981, Bozek and Rahel 1991). Snorkeling is one of the most common and simplest ways to observe fish and requires minimal equipment. In general, snorkeling

can be effective in a variety of habitats, but environmental conditions and differences in ability, training, and experience among personnel can bias results (Thurow et al. 2012). Moreover, the presence of human observers during snorkeling events has been shown to elicit a flight response by fish (e.g., Peterson et al. 2005). Thus, remote methods of direct observation (e.g., cameras) have been developed to reduce the influence of human presence during direct observation procedures.

Similar to PAEDs, fixed underwater videography (FUV) can be used to sample a discrete area and serves to decrease flight bias. Underwater videography has been used to quantify fish density (Willis and Babcock 2000), abundance (Ellis and DeMartini 1995, Stoner et al. 2008), species richness (Ebner and Morgan 2013), and habitat use (Becker et al. 2010). However, FUV is most commonly applied to studies of fish behavior (e.g., Suzuki et al. 2003, Kane et al. 2004, Ebner et al. 2009). Most recently, underwater video has been used to sample fish in areas where other gears fail (e.g., dense aquatic vegetation; Wilson et al. 2015). Recent advancements in technology have provided consumers with inexpensive video cameras that are capable of capturing high-quality underwater video footage. Previous studies have incorporated FUV techniques with and without bait (e.g., Stobart et al. 2007, Hannah and Blume 2012), but many of those studies were conducted in marine environments or coral reef systems and used large, conspicuous video lander systems (e.g., Ellis and DeMartini 1995, Willis and Babcock 2000, Stobart et al. 2007). Using cameras outfitted with bait tends to provide samples with increased fish counts, but the increase in sample size comes as a result of attracting fishes to the gear (Watson et al. 2005, Harvey et al. 2007). Thus, the concept of sampling fishes from naturally selected habitats is negated.

In the current study, we evaluated the efficacy of a PAED and FUV for sampling fishes in riverine environments. In addition, we sought to determine an appropriate “set time” (i.e., elapsed time between electrode deployment and electrification) for sampling riverine fishes with a PAED. We empirically assessed the relative abundance of fishes prior to and following electrode deployment using FUV techniques. The deployment sequences for PAEDs and FUV are nearly identical;

both sampling techniques disturb the intended sampling area for a similar amount of time and require that the area remain undisturbed following deployment of the gear. We postulated that FUV may be used to sample riverine fishes with equal accuracy relative to PAED samples while minimizing bias due to gear or observers.

## METHODS

### Study Area

The Kootenai River (spelled Kootenay in Canada) has an international and interstate watershed that receives up to 3000 mm of precipitation annually and drains an area of approximately 49,987 km<sup>2</sup> (Woods 1982, Knudson 1994). The river originates in Kootenay National Park, British Columbia, Canada, at an elevation of 3618 m and flows for 775 km. From British Columbia, the river flows southward into Montana, USA, where it is impounded by Libby Dam, which forms Lake Koocanusa. From Libby Dam, the river flows northwest into the panhandle of Idaho, USA, then north into British Columbia where it enters Kootenay Lake before joining the Columbia River near Castlegar, British Columbia, at an elevation of 418 m. The Kootenai River is the second largest tributary to the Columbia River in terms of runoff volume and third largest in drainage area (Knudson 1994).

In Idaho, the Kootenai River is categorized into 3 distinct sections based on unique geomorphologies: canyon, braided, and meander (Smith et al. 2016). The canyon section has high current velocities, large substrate (e.g., cobble, boulder), a restricted floodplain, and is characterized by the occurrence of native salmonids (rainbow trout *Oncorhynchus mykiss* and mountain whitefish *Prosopium williamsoni*; Smith et al. 2016). The braided section is a transitional zone that has high rates of sediment deposition, a low gradient, a wide valley with prominent floodplain, a braided channel type, and is characterized by the occurrence of native salmonids, catostomids (largescale sucker *Catostomus macrocheilus*), and redbelt shiner *Richardsonius balteatus*. The meander section has low current velocities, a low gradient, a single, sinuous channel, and is characterized by the occurrence of cyprinids (northern pikeminnow *Ptychocheilus oregonensis* and peamouth *Mylocheilus caurinus*).

The braided section has the highest level of habitat complexity and dynamism compared to the canyon and meander sections (Smith et al. 2016). Consequently, the braided section has the highest species richness estimate relative to the canyon and meander sections. As such, the braided section has been the primary focus of a large-scale and long-term habitat rehabilitation program that aims to enhance existing habitat for the benefit of native fishes at all life history stages (KTOI 2009, Watkins et al. 2015). The objectives of the habitat rehabilitation program are numerous, but some of the primary projects include treatments designed to disperse flow, create floodplain habitat, increase substrate heterogeneity, and create complex in-water habitats by adding woody structures.

#### Field Sampling

Sampling occurred during daylight hours in wadeable areas of the braided section of the Kootenai River, Idaho, in August 2014. Twenty-eight sites were randomly selected to receive sampling effort using FUV and PAEDs. Microhabitat characteristics differed across sites; current velocities varied from 0.00 to 1.06 m/s, depths varied between 0.12 and 1.10 m, and substrate size varied from sand (>0.07 mm) to boulder (>257.00 mm). Electrodes were deployed in pairs consisting of a cathode and anode that collectively constituted a PAED. Each electrode was constructed with a 9.1-m length of insulated, tinned-copper wire that terminated in a plug (Midwest Lakes Electrofishing Systems; Polo, MO). The insulated wire articulated with a length of 4.8-mm-diameter stainless steel aircraft (SSA) cable that remained exposed and served as the conducting material for closure of the electrical circuit. The cathode was constructed using 6.1 m of SSA cable, and the anode used 3.4 m. A wire rope clip was used to secure a loop for the anode, producing a circular ring (area = 0.80 m<sup>2</sup>). We positioned the cathode approximately 1 m downstream from the anode perimeter to ensure consistent electrical fields among sampling locations. The electrodes were powered by applying pulsed direct current standardized to 500–800 W using a LR-24 backpack electrofisher (Smith-Root Inc., Vancouver, WA) retrofitted to accept the terminal plugs on the PAED. Given the physical properties of pulsed direct current, fishes were

immobilized beyond the confines of the anode. Pilot studies indicated that the immobilization zone of the PAED was approximately 4 m<sup>2</sup>.

Video footage was collected using a GoPro Hero3+<sup>®</sup> camera (GoPro, Inc., San Mateo, CA). The camera was housed in a black underwater casing mounted to a metal stake, and it collected footage at a rate of 30 frames per second. The camera was positioned in-stream such that the entire immobilization zone of the PAED was included in the field of view (area of approximately 4 m<sup>2</sup>). A sampling event occurred in 3 phases (Fig. 1). First, a video camera was positioned in-stream and collected 15–20 min of footage (T1, “baseline” footage). Second, a PAED was positioned approximately 0.50 m in front of the camera and was oriented perpendicular to the thalweg while video footage continued to be recorded. The area remained undisturbed for an additional 15–35 min following PAED deployment (T2, “set” footage). Third, the PAED was electrified for 20 s and a single netter entered the water and collected immobilized fishes (T3). Fishes were identified, enumerated, measured (total length), and returned to the water. Baseline and set times used in this study were motivated by the results of Bain et al. (1985). The authors found no significant correlation between PAED set time and number of fish captured ( $P = 0.43$ ; Bain et al. 1985) and noted that a set time of 10 min was adequate for their study. We chose a longer set time to (1) account for instances where 10 min may not have provided enough time for fishes to return and (2) strike a balance with video processing time. Electrical current was applied to the PAEDs for 20 s to standardize electrofishing effort across sites. This time period provided enough time for personnel to enter the water and collect immobilized individuals while minimizing undue stress to the fish.

#### Video Processing

Video footage was processed using VLC Media Player (VideoLan Team) and viewed in real time on a computer monitor. Video footage was systematically subsampled in 5-s increments to minimize the chance of missing the return of a taxon to the sample area. A single observer enumerated and identified fishes. For instances when high fish density made enumeration difficult, 2 observers reviewed the video in 20-s increments and determined

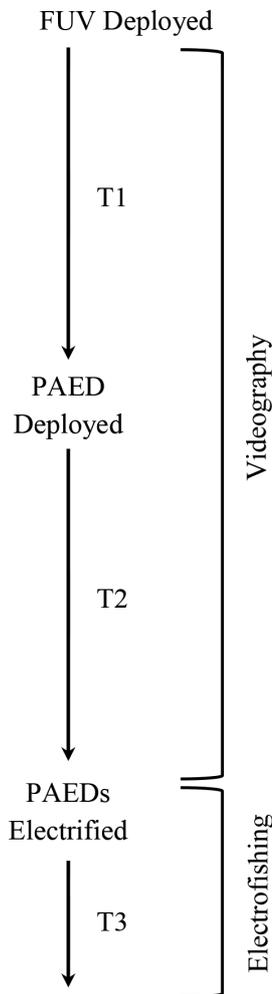


Fig. 1. Representation of the continuous field sampling procedure that occurred in 3 distinct phases. Time period one (T1) represents the elapsed time between fixed underwater videography (FUV) and prepositioned areal electrofishing device (PAED) deployment (i.e., “baseline” footage). Time period two (T2) represents the elapsed time between the PAED deployment and electrification (i.e., PAED “set” footage). Time period three (T3) represents the time during which electricity was applied to the PAED (20 s).

a final count by consensus. Fishes were identified to the lowest possible taxonomic level. If identification of a particular fish could not be determined, the individual was recorded as “unidentifiable.” Multiple viewings of footage were necessary in certain situations (e.g., high fish density, presence of small fishes, poor underwater visibility) to ensure accurate identification and counts.

## Data Analysis

Subsamples from the first 4 min of video footage were omitted from the analysis. This period was based on an evaluation of fish response to positioning the camera instream. Similar acclimation periods for FUV studies have been reported (Frezza et al. 2003, Harvey et al. 2007). Count data were summarized by gear type to estimate taxonomic occurrence and relative abundance. Fish counts for FUV data were summarized by estimating the frequency of occurrence divided by the number of subsamples from each video time period (i.e., T1 and T2). Fish count data obtained with PAEDs (i.e., T3) provided a measure of taxonomic occurrence that served as a baseline against which to compare FUV data. As such, care was taken to avoid frightening additional fish into the electrical field while also capturing every immobilized fish. Relative abundance estimates for FUV data were calculated using a modified MaxN index following Ellis and DeMartini (1995). Specifically, MaxN is the maximum number of individuals for any taxa present in the field of view at the same time. This approach reduces the chance of counting the same fish more than once (Becker et al. 2010, Ellender et al. 2012). Since our subsampling procedure enumerated fish at relatively short time intervals, we estimated MaxN for each minute of video footage by selecting the maximum MaxN among the subsamples in a minute of video. Maximum MaxN was averaged across all minutes and time periods (i.e., T1, T2, T3) at each site to calculate a mean MaxN (mMaxN) value for each identified taxon. Mean MaxN values were then averaged across all sites to estimate the mean mMaxN value for each identified taxon.

To evaluate the efficacy of the FUV approach, we compared occurrence and total relative abundance estimates from PAEDs with those from FUV over the integrated time period of T1, T2, and T3. A paired Wilcoxon signed-rank test was applied to the relative abundance data from T1 and T2 to evaluate the effect of the PAED on the fish assemblage. To determine an appropriate PAED set time, we measured the length of time until fishes returned to the sample area following PAED deployment. Taxon-specific return times were averaged across all sites, and a suggested PAED set time was determined by averaging those values.

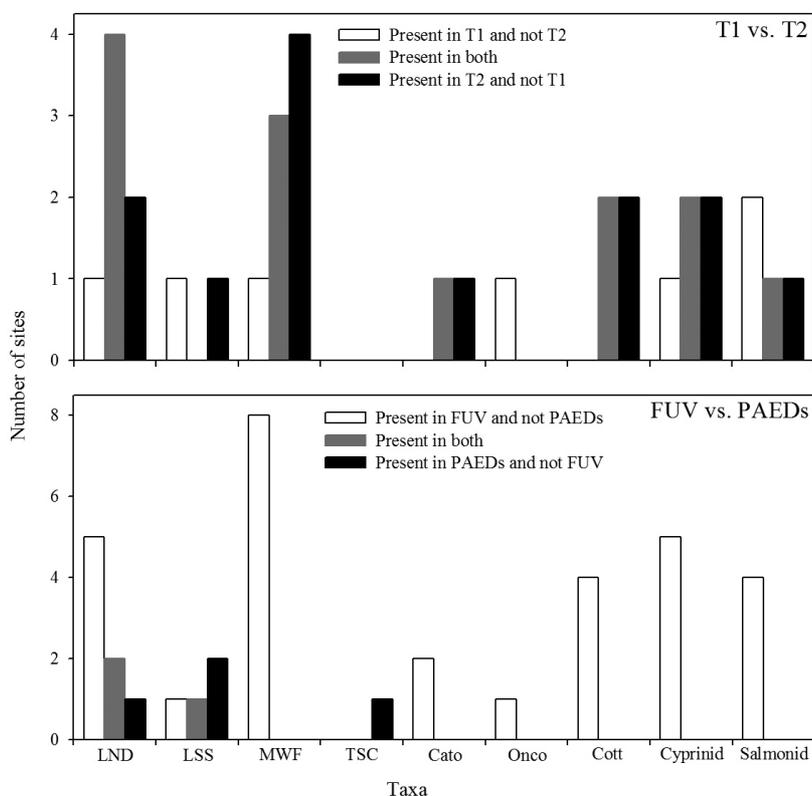


Fig. 2. Total number of sites where each taxon (LND = longnose dace, LSS = largescale sucker, MWF = mountain whitefish, TSC = torrent sculpin, Cato = *Catostomus* spp., Onco = *Oncorhynchus* spp., Cott = *Cottus* spp., Cyprinid = Cyprinidae, Salmonid = Salmonidae) sampled from the Kootenai River, Idaho, was detected using fixed underwater videography (FUV) and prepositioned areal electrofishing devices (PAEDs) across 15 sites. The top panel compares occurrence estimates from time period one (T1, elapsed time between FUV and PAED deployment) and time period two (T2, elapsed time between PAED deployment and electrification). The bottom panel compares occurrence estimates between FUV and PAEDs.

## RESULTS

Of the 28 sites sampled concurrently with PAEDs and FUV, 13 sites had unusable data due to malfunctioning PAEDs ( $n = 3$ ), malfunctioning video cameras ( $n = 3$ ), and visual obstructions (e.g., dense aquatic vegetation) in the video footage ( $n = 7$ ). Accordingly, 15 sites were used in the analysis. Nine fish were sampled from 7 sites with PAEDs. All fish sampled by PAEDs were identified to species and included largescale sucker, longnose dace *Rhinichthys cataractae*, and torrent sculpin *Cottus rhotheus*. In total, 1086 fish were sampled across all 15 sites with FUV using the modified MaxN index. Fishes sampled with FUV were difficult to identify with confidence and were generally classified to genus or higher taxonomic levels, or as “unidentifiable.”

Unidentified fishes comprised 53% of all FUV detections and 87% of the catch from the MaxN index. Eight of 9 identified taxa were sampled at least once with FUV, whereas PAEDs accounted for only 3 of 9 identified taxa across all sites (Fig. 2, lower panel).

Occurrence estimates for identified taxa were reasonably similar among video time periods (Fig. 2, upper panel) and variable among gear types (Fig. 2, lower panel). No salmonids were sampled by PAEDs; conversely, torrent sculpin were only sampled by PAEDs. Fixed underwater videography detected mountain whitefish at 8 sites. The species was undetected with PAEDs. In contrast, torrent sculpin, largescale sucker, and longnose dace were sampled at 4 different locations by PAEDs but were not identified using FUV at those sites (Fig. 2, lower panel).

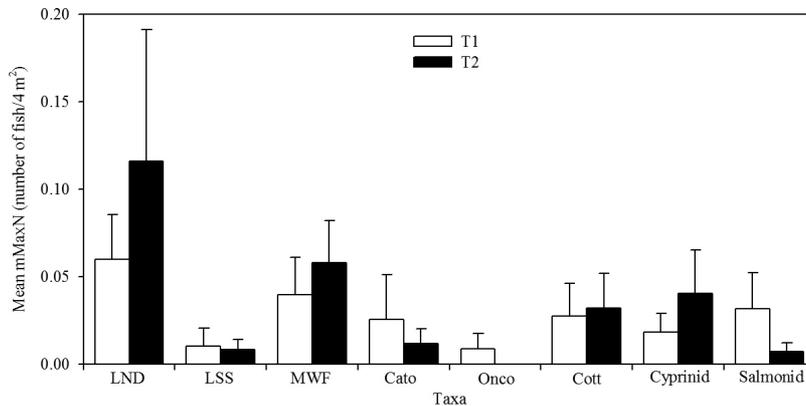


Fig. 3. Relative abundance (Mean mMaxN) of identified taxa (LND = longnose dace, LSS = largescale sucker, MWF = mountain whitefish, Cato = *Catostomus* spp., Onco = *Oncorhynchus* spp., Cott = *Cottus* spp., Cyprinid = Cyprinidae, Salmonid = Salmonidae) sampled using FUV from the Kootenai River, Idaho, across all sites ( $n = 15$ ) for T1 (“baseline” footage; i.e., no electrode present in frame) and T2 (“set” footage; i.e., electrode present in video frame). Relative abundance estimates were calculated using a modified MaxN index, where MaxN was the maximum number of individuals within each minute of video footage. Mean MaxN (mMaxN) was calculated by taking the mean of MaxN across all minutes of reviewed FUV footage at each site. Mean mMaxN was estimated by computing the mean of mMaxN across all sites where a taxon was observed. Error bars represent  $\pm 1$  standard error of the mean mMaxN.

*Oncorhynchus* spp. was the least observed taxon and was only observed during T1 at one site. Relative abundance estimates for identified fishes sampled with FUV were similar across T1 and T2 for most taxa except longnose dace, *Catostomus* spp., *Oncorhynchus* spp., and Cyprinidae (Fig. 3). A Wilcoxon signed-rank test indicated no significant difference in relative abundance between the fish assemblage during T1 and T2 ( $P = 0.74$ ).

The elapsed time for each identified taxon to return to the sampling area following deployment of the PAED varied across sites (Table 1). All identified taxa exhibited at least one instance of failing to return to the sampling area or appearing in T2 without having been identified in T1. Only 2 of the 15 sites sampled displayed identical fish assemblages between T1 and T2. One additional site had identical assemblages between T1 and T2, although a torrent sculpin was sampled by electrofishing that was undetected by FUV. However, it is worth mentioning that FUV detected a *Cottus* spp. at that site that was likely a torrent sculpin because that was the only sculpin species we observed. Five sites displayed instances where the assemblages identified in T1 returned with at least one additional taxonomic group in T2. Average time for taxa identified in T1 to return to the sample area was 12.24 min.

## DISCUSSION

The current study aims to evaluate the efficacy of 2 sampling gears for estimating the occurrence and relative abundance of riverine fishes. Across 15 sites, 9 taxa were collected from a relatively small sampling area. At any particular site, PAEDs and FUV sampled different taxa. All fishes collected with PAEDs were identified to species, whereas most fishes sampled by FUV were classified as “unidentifiable.” Factors such as fish size, fish position, elapsed time spent by fish in the video frame, underwater visibility (e.g., turbidity, current velocity), and proficiency of the reviewer contribute to the ability to classify fish to lower taxonomic levels. The relative imprecision of identifying fish with FUV did not allow us to make direct comparisons between FUV data and PAED data. For example, *Cottus* spp. was detected with FUV at one site where torrent sculpin was sampled with PAEDs. Cyprinidae were sampled with FUV at several sites where longnose dace were sampled with PAEDs. Although the unidentified cyprinids were likely longnose dace, FUV could not provide conclusive identification.

At any particular sampling location, the occurrence estimates generated from data gathered by PAEDs and FUV varied. One reason that differences in occurrence were

TABLE 1. Elapsed time (min) for each identified taxon (LND = longnose dace, LSS = largescale sucker, MWF = mountain whitefish, TSC = torrent sculpin, Cato = *Catostomus* spp., Onco = *Oncorhynchus* spp., Cott = *Cottus* spp., Cyprinid = Cyprinidae, Salmonid = Salmonidae) to return to the video frame after deployment of a PAED (T2). Data are arranged by site in the Kootenai River, Idaho. Footage from fixed underwater videography (FUV) was subsampled in 5-s or 20-s intervals following deployment of the electrofishing gear. The bottom 2 rows of the table represent means and standard errors (value in parentheses) calculated for each taxon where (1) all observations during T2 are considered (i.e., species additions were included in the mean calculation) and (2) only returning individuals are considered (i.e., values with asterisks were removed from the mean calculation).

Site	LND	LSS	MWF	TSC	Cato	Onco	Cott	Cyprinid	Salmonid
1	—	—	∞	—	—	∞	—	—	—
2	—	a	—	—	—	—	—	—	—
3	—	†∞	23.58*	—	29.42	—	—	—	—
4	10.00*	—	—	—	8.92*	—	—	10.25*	6.75
5	4.67†	—	—	—	—	—	—	∞	—
6	—	—	2.92*	—	—	—	—	—	—
7	11.50†*	—	13.25	—	—	—	26.33*	7.33*	—
8	—	—	2.42	—	—	—	—	—	9.83*
9	—	—	18.58	—	—	—	—	—	—
10	—	a	—	—	—	—	4.50	—	—
11	6.50	—	—	a	—	—	18.50	—	—
12	∞	—	—	—	—	—	—	—	∞
13	11.17	—	10.17*	—	—	—	—	—	—
14	21.25	22.17*	5.33*	—	—	—	—	0.25	∞
15	a	—	—	—	—	—	8.92*	6.67	—
ALL	10.85(2.35)	22.17	10.89(3.06)	—	19.17(10.25)	—	14.56(4.89)	6.13(2.11)	8.29(1.54)
RETURNING	10.89(3.71)	—	11.42(4.75)	—	29.42	—	11.50(7.00)	3.46(3.21)	6.75

†Sampled with PAED.

aNot observed during FUV processing but sampled with PAEDs.

\*Taxon was not identified in the sampling frame prior to deployment of PAED (T1).

∞Taxon was not identified in the sampling frame during T2.

observed may have been the amount of sampling time. The field sampling procedures required approximately 45 min to complete. The first 2 phases in the 3-phase sampling procedure sampled for approximately 40 min with FUV. Following FUV data collection at a site, PAEDs sampled the same area for only 20 s. Nine individual fish were sampled by PAEDs across 15 sites, whereas FUV sampled over 1000 fish at the same locations. We would expect more observations of fishes with a longer sampling duration. However, the increase in observations with FUV comes with a substantial increase in processing time. On average, the video review procedure required approximately 2 h for 40 min of video footage. In contrast, data collected by PAEDs required minimal processing after the field procedure. Both PAEDs and FUV provide a measure of taxonomic occurrence, but the occurrence estimates generated from FUV necessitate laborious video processing in exchange for less precise data. Taxonomic classification of FUV samples was necessary for comparative purposes because all fish sampled by PAEDs were easily identified to species. Previous studies using FUV have employed other clas-

sification systems, such as a guild approach (e.g., functional feeding group; Becker et al. 2010) or general body shape (Frezza et al. 2003). Selecting between PAEDs and FUVs is ultimately dependent on study objectives and the ability to resolve individual fishes to usable taxonomic classifications.

The likelihood of observing a fish by using PAEDs or FUV was low. The PAED set time used throughout this study likely had no effect on the number of fish sampled by PAEDs. Bain et al. (1985) were the first to describe the use of prepositioned electrofishing equipment. They noted that set times >10 min appeared adequate and speculated that “very long set times seemed unnecessary.” Results from our study indicated that a set time of at least 12 min reduced the effect of fright bias associated with deploying the gear. Our analysis of FUV footage indicated that fright bias was reduced 4 min after positioning the camera instream, further corroborating the hypothesis that set time had little influence on catch. The sparse observations of fish are likely due to discrepancies between movement dynamics of riverine fishes and the relatively small sampling area. Baxter (2002) found that movements of

mountain whitefish in the Pacific Northwest varied from 0.2 km to 190.0 km throughout the study, regardless of fish size. Similarly, large-scale sucker movements varied from 17.2 km to nearly 300.0 km in the same study.

Despite the limitations of FUV, the sampling technique may be appropriate for some research questions in freshwater systems provided that the ability to identify fish satisfies project objectives. Video techniques would be especially useful for applications involving rare or imperiled fishes, which generally require special permits that limit fish handling procedures. For instance, Chaudoin et al. (2015) found that underwater videography outperformed above-water videography and in-person surveys for monitoring spawning activity of Devils Hole pupfish *Cyprinodon diabolis*. Ellender et al. (2012) determined that underwater videography accurately sampled imperiled fishes of South African headwater streams by comparing relative abundance estimates with those generated by 3-pass electrofishing. Video techniques have also been used to validate or estimate catchability for other sampling gears. Grant et al. (2004) used underwater video techniques to estimate retention probability for walleye *Sander vitreus* in gill nets.

Small-scale processes like microhabitat use by fishes are difficult to capture, especially when sampling large river systems. Bain et al. (1985) suggested that PAEDs may be inefficient in fast, deep, turbid waters and that habitat specificity decreases as sample area and relative catch increase. Consequently, pilot studies may be warranted when using PAEDs to ensure that catch and habitat specificity meet study objectives. Similar limitations apply to the FUV methods used in this study, and visual obstructions (e.g., boulders, woody debris) further limit the ability to observe and identify fishes. For example, we could not use the data from one quarter of the sites sampled in this study due to visual obstruction. Many habitat rehabilitation programs incorporate the placement of large structures instream to enhance habitat complexity (Roni et al. 2002). Evaluating the response of fish to those habitat improvements using FUV may be ineffective given that cryptic fish species may be overlooked. Compared to electrofishing, underwater videography has known biases toward sampling simpler fish assemblages and

small-bodied fish (Frezza et al. 2003). Small-bodied fishes composed the majority of individuals sampled by FUV in this study, but FUV samples provided increased taxonomic richness and diversity compared to PAED samples. These results are confounded to some extent due to the video review procedure and taxonomic “nestedness,” but PAEDs sampled only one species at 6 sites where FUV sampled additional fishes belonging to different species, genera, and families.

Both PAEDs and FUV adequately reduce fright bias and provide fish occurrence and relative abundance estimates with regard to discrete microhabitats. Originally, we intended to process video footage by examining still frames in an attempt to decrease processing time. However, nearly all of the fishes sampled in this study were relatively small, and viewing video footage in real time allowed the reviewer to locate fishes by observing movements. If large-bodied fishes were sampled more frequently, the still-frame reviewing approach would substantially reduce processing time. The time spent reviewing and processing video footage may detect more taxa than would otherwise be sampled using PAEDs, but the time spent processing video footage could be allocated to sampling a broader spatial distribution with PAEDs. Therefore, we conclude that choosing between PAEDs and FUV for sampling riverine fishes requires a clear understanding of project objectives, and we recommend that careful consideration be given to the tradeoffs that exist between the gears.

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