FISHERY RESEARCH

KOOTENAI RIVER RESIDENT FISH MITIGATION: WHITE STURGEON, BURBOT, NATIVE SALMONID MONITORING AND EVALUATION

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CHAPTER 1: KOOTENAI STURGEON MONITORING AND EVALUATION

ABSTRACT

Kootenai River White Sturgeon *Acipenser transmontanus* were listed as endangered in 1994 primarily due to recruitment failure and overharvest. The population has been declining for the last 40 years, and natural reproduction has been inconsistent since 1974. Libby Dam, completed in 1972, drastically changed the Kootenai River ecosystem by disrupting the natural flow regime and altering seasonal and daily water temperatures. Idaho Department of Fish and Game (IDFG) is funded through Bonneville Power Administration (BPA) to monitor and evaluate the effects of mitigated flows from Libby Dam on all life stages on Kootenai Sturgeon and to provide recommendations for recovery to action agencies. The objective of these studies is to determine how current dam operations influence spawning behaviors and juvenile recruitment of Kootenai River White Sturgeon. Flows during the 2017 and 2018 spring spawning period were maintained near flood stage for almost three months due to a wet spring and large snowpack. Eighteen adult Kootenai Sturgeon were implanted with acoustic transmitters in the spring of 2017 and 2018. These fish, along with previously tagged individuals, allowed us to assess the full extent of spawning migrations. We estimated that over 50% of the acoustically tagged spawning group of Kootenai Sturgeon moved above Bonners Ferry in 2017 into previously developed habitat projects. Two substrate enhancement pilot projects were constructed in winter 2014 near Shorty's Island and Myrtle Creek as spawning and incubation habitat improvements. In 2017 and 2018, we continued new sampling techniques to evaluate adult Kootenai Sturgeon habitat use, spawning distribution, and larval hatching successes resulting from the aforementioned habitat enhancement projects. Results indicated that Kootenai Sturgeon were using and spawning on the new habitat, but successful, large-scale larval recruitment was not documented in 2017 or 2018. Hatchery produced juvenile Kootenai Sturgeon densities remained high, similar to numbers seen in 2016. Our evaluations indicated that the high densities are likely negatively influencing growth and survival of juvenile hatchery fish stocked into the system. The hatchery population was well distributed throughout the Kootenai River and Kootenay Lake, and many age classes were represented.

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INTRODUCTION

Since the 1970s the population of Kootenai River White Sturgeon (hereafter Kootenai Sturgeon) has been in decline. The primary drivers of the population decline have been a combination of overharvest and anthropogenic caused habitat degradation. Although harvest was eliminated in 1994, a lack of adequate spawning and rearing habitats has limited recruitment to almost nonexistent levels since the installation of Libby Dam. Libby Dam was constructed in 1972 and has been the largest factor contributing to the decline of suitable sturgeon spawning and rearing habitats. Historic Dam operations disrupted natural flow and temperature regimes, specifically, spring flows. Spring flows, which coincide with Kootenai Sturgeon spawning, are significantly smaller in magnitude and shorter in duration compared to historic discharge records. The dam-mediated reduced spring flows prevent a number of hydrologic processes from occurring including spawning gravel recruitment and floodplain inundation. In conjunction, these changes in the Kootenai River have created an environment unable to support reliable sturgeon recruitment.

The spawning period for Kootenai Sturgeon begins each spring when the Kootenai River experiences its highest flows. These high flows act as a cue for adult sturgeon residing in Kootenay Lake and in the lower parts of the Kootenai River to begin their upstream migration to the spawning grounds (Paragamian et al. 2002). Spawning and egg deposition tends to occur on the descending limb of the hydrograph when instream temperatures begin to increase to approximately 9°C (Hardy et al. 2016). Kootenai Sturgeon typically spawn once every four to six years, which means on any given year only ~20% of the adult population will make the spawning migration (Paragamian et al. 2005). Fertilized eggs are commonly observed in the river; however, wild-origin juvenile and larval sturgeon have rarely been observed since the construction of Libby Dam.

Although the exact mechanisms responsible for reduced levels of recruitment remain unclear, years of study by Idaho Department of Fish and Game (IDFG) suggest that mortality occurs between egg and larval stages (Paragamian et al. 2002). Over a decade of artificial substrate mat sampling has indicated that 9-20 spawning events occur annually, and these events are capable of producing viable embryos (Paragamian et al. 2002). Most post-Libby Dam spawning events have been documented in reaches where substrate conditions appear to be unsuitable for egg incubation and larval rearing (Paragamian et al. 2001). Only a handful of larvae (i.e., <10) and relatively few wild juveniles have been collected, despite years of intensive sampling. Our investigations suggest that egg and/or larval suffocation, predation, or other mortality factors associated with these early life stages contribute to persistent recruitment failure (Kock et al. 2006).

In response to the decline in Kootenai Sturgeon recruitment, the Kootenai Tribe of Idaho (KTOI) began a conservation aquaculture program in 1989. Each spring, broodstock (approximately six females and twelve males) are collected and spawned in the hatchery. The resulting offspring are reared and are stocked back into the river after six to nine months. In addition to maintaining the abundance of Kootenai Sturgeon in the basin, the KTOI aquaculture program and associated IDFG population monitoring have provided many insights into the timing and potential causes of recruitment failure. Hatchery-reared juveniles (as young as nine months of age at release) grow approximately 6.4 cm per year, and second year survival rates exceed 90% (Ireland et al. 2002). Growth and survival of hatchery juveniles released at a minimum of age-1 further suggest that mortality occurs at the egg, embryonic, or larval stage.

In recent years IDFG efforts have been focused on monitoring and evaluating Kootenai Sturgeon spawning and juvenile rearing. Through extended monitoring we hope to refine our understanding of the exact mechanism causing recruitment failure of the Kootenai River White Sturgeon. In this report we present the findings from our efforts during the 2017-2018 spawning and rearing season.

OBJECTIVE

The overarching objective of this project is to recover the Kootenai Sturgeon population to a self-sustaining level that can support sportfishing opportunity for the public. In support of this effort, we are tasked with (1) evaluating the response of all life stages of Kootenai Sturgeon to flow augmentation from Libby Dam provided by the U.S. Army Corps of Engineers (USACOE), and (2) evaluating other mitigation efforts to improve suitable spawning, rearing, and incubation habitat for Kootenai Sturgeon for successful wild recruitment.

STUDY SITE

The Kootenai River originates in Kootenay National Park, British Columbia (BC), Canada. The river flows south into Montana and turns northwest at Jennings, near the site of Libby Dam, at river kilometer (rkm) 352.4 (Figure 1.1). In this study, rkms increase as you move upstream, with rkm 18 at the mouth of the Duncan River at the north end of Kootenay Lake, BC. The Duncan River is a tributary on the north end of Kootenay Lake which is also available to adult and juvenile sturgeon. Kootenai Falls, 42 rkms downstream of Libby Dam, is thought to be a historically impassable barrier to Kootenai Sturgeon. As the river flows through the northeast corner of Idaho, there is a gradient transition at Bonners Ferry. Upstream from Bonners Ferry, the channel has an average gradient of 0.6 m/km, and the velocities are often higher than 0.8 m/s. Downstream from Bonners Ferry, the river slows to velocities typically less than 0.4 m/s (average gradient 0.02 m/km), and the channel deepens as the river meanders north through the Kootenai River Valley. The river returns to BC at rkm 170 and enters the South Arm of Kootenay Lake at rkm 120. Kootenay Lake empties through its West Arm and joins the Columbia River at Castlegar, BC. A natural barrier at Bonnington Falls (now a series of four dams) has isolated Kootenai Sturgeon from other populations in the Columbia River basin for approximately 10,000 years (Northcote 1973). The watershed of the Kootenai River encompasses a total area of 49,987 km^2 (Bonde and Bush 1975). Regulation of the Kootenai River following the construction of Libby Dam in 1974 changed the natural hydrograph and temperatures of the river (Partridge 1983). Spring flows were reduced to about one third of pre-dam levels, and flows during winter are three to four times higher than under the natural flow regime (Figure 1.2). However, starting in 1991, Libby Dam has been operated to provide increased spring discharge when water supplies are suitable (>630 m^3/s or 22,248 ft^3/s for 42 d at Bonners Ferry) to improve spawning conditions for Kootenai Sturgeon adults and rearing conditions for embryos and larvae. Post-dam water temperatures are on average cooler in summer and warmer in winter.

METHODS

Water Levels, Discharge, and River Temperature Manipulation

The exact shape, timing, and volume of flows during the year are detailed through System Operations Request (SOR) FWS # 2017-1018 that was submitted to the USACOE's regional multiagency/entity Technical Management Team (TMT). The intent of these SORs was to maintain higher, more stable summer discharges to the extent possible with the available water to meet Kootenai Sturgeon and Bull Trout *Salvelinus confluentus* ESA responsibilities (USFWS 2006) and to attempt to mimic a more natural river hydrograph (under VarQ regime). Another objective of the SORs is to provide spawning and incubation flows to meet attributes for water depth, water velocity, and water temperature in the Kootenai River as defined in the 2006 Biological Opinion RPA for Kootenai Sturgeon (USFWS 2006). An additional objective of this SOR is to improve conditions for spawning sturgeon to migrate upstream of Bonners Ferry into the braided reach (i.e., above rkm 246).

The 2017-2018 SOR was designed to meet these objectives by providing peak river stages/flows during the spring run-off period. This peak in flow is timed during high elevation run-off below Libby Dam and is intended to first provide sturgeon cues to begin upstream migration and staging. As river temperatures warm to 8-10°C, sturgeon migrate further upstream from their staging areas and spawn on the peak and descending limb of the spring hydrograph. Overall, the goal of flow management is to provide conditions that will enable sturgeon to migrate to, and spawn over, rocky substrates that exist upstream of Bonners Ferry. Beginning in 2013, IDFG, in collaboration with the USACOE, begin operating Libby Dam to manipulate spring flows to have two distinct peaks. It was hypothesized that the first peak would initiate staging movements of spawning adults and the second peak would initiate actual spawning. The exact timing and magnitude of these peaks differed between years and was subject to water availability forecasts and precipitation events. Although a two-peak approach was successfully implemented in 2013 and 2014, lower than average water supply forecasts prevented the two-peak approach in 2015 and 2016. The plan in 2017 was to implement the two-peak approach again and to collect more information on how different water management strategies influenced the spawning behavior of Kootenai River White Sturgeon. Results of this evaluation can be found in Chapter 2.

Adult Kootenai Sturgeon Sampling

Adult Kootenai Sturgeon were collected by angling and setlining (Paragamian et al. 1996) during two different periods in the spring and fall of 2017 and 2018. Additional adult sampling also occurred in the spring of 2018. Given the interval of interest for this report (May 1, 2017 – April 30, 2019), those data are also reported. The vast majority of adult sampling (by effort) occurs in the Idaho portion of the river. These areas are backwater habitats and have depths in excess of 20 m and low current velocities (<0.05 m/s) and incorporate spawning locations (near and above rkm 229). Much of the angling effort in the Idaho portion of the river coincided with KTOI broodstock collection for the conservation hatchery. Fall sampling in 2017 occurred throughout the lower river (rkm 207.5-308) and into Kootenay Lake in BC, at the Creston delta (rkm 118) and the Lardeau River Delta at the north end of the lake (rkm 18). During sampling we attempted to determine the gender and level of maturity of adult sturgeon following gonadal biopsy protocol of Conte (1988) and Van Eenennaam and Doroshov (1998). Adult Kootenai River Sturgeon are currently defined by a length of ≥ 115 cm fork length (fl) or ≥120 cm total length (tl; Paragamian et al. 1995). Male and female Kootenai Sturgeon expected to spawn each spring were tagged with Vemco model V16 sonic transmitters and released. In addition, in collaboration with KTOI, adult Kootenai Sturgeon that were expected to spawn (based on gonadal examination) but that were not used for in-river research purposes were transported to the KTOI Hatchery for production.

Adult Kootenai Sturgeon Telemetry

We continued to monitor daily and seasonal spawning movements of Kootenai Sturgeon throughout the Kootenai River/Kootenay Lake system using a passive telemetry array. Beginning in 2003 and continuing to the present, we have maintained an array of 89 Vemco model VR2 and VR2W sonic receivers located from rkm 18, at the Lardeau-delta near the mouth of the Duncan River in Kootenay Lake, BC, upstream to rkm 306, below Kootenai Falls (Figures 1.3 and 1.4). From this array we are able to analyze occupancy (i.e., presence/absence) as well as individual movements in different reaches throughout Kootenay Lake and the Kootenai River. A total of 121 implanted transmitters (*n* females = 107, *n* males = 14) were active in adults during the sampling period of May 1, 2017 – April 30, 2018. Additional adults were tagged from May 1, 2018 – October 23, 2019 and added to the number of active transmitters (*n* females = 11, *n* males = 3, *n* unknown = 3). Receivers were located in areas where fish pass through but do not typically hold for long periods to avoid redundant data collection. Most receiver locations were below river bends or along straight reaches that allowed for maximum signal reception but were reasonably free of drifting debris and at low risk of potential vandalism/theft. We tethered each receiver to an anchored float that was chained to the riverbank in order to keep the hydrophone off the substrate (Neufeld and Rust 2009). We downloaded the movement information from the receivers twice during the report period, once in late winter, prior to the spawning season, and again in the fall.

Substrate Enhancement Pilot Projects

Previous observations have shown that Kootenai Sturgeon spawn primarily between rkms 228 and 240.5 (Paragamian et al. 2002). The substrates of this reach are primarily comprised of sand, silt, and clay (Fosness 2013); substrate types often considered unsuitable for successful survival in the early life stages of White Sturgeon. Other White Sturgeon populations in the Columbia basin spawn specifically over some combination of rock and gravel, which provides adequate egg and larval aeration as well as suitable hiding spaces (Parsley et al. 1993). Because of the lack of substrates that are able to support spawning and early life-stages in the meander reach of the Kootenai River, the Kootenai River Recovery Implementation Plan (Kootenai Tribe of Idaho 2009, 2013) proposed “adding rock substrate in the current spawning areas and evaluating its role in providing suitable spawning and incubation conditions.”

In April 2010, under the authority provided by the Continuing Authorities Program, Section 1135, USACOE, in cooperation with the KTOI, initiated a feasibility study to “identify and implement cost-effective, self-sustaining ecosystem restoration actions to improve ecosystem function and habitat attributes for the early life stage survival of the ESA-listed Kootenai Sturgeon” (US Army Corps of Engineers 2012). The USACOE’s feasibility study recommended a substrate enhancement pilot project (SEPP) at two locations, Shorty’s Island South and Myrtle Creek (Figure 1.5 – 1.7). In 2013, KTOI continued the implementation of the SEPP at two sites in the meander reach. The objective of the SEPP was to test “the sustainability and effectiveness of placing rock substrate over existing clay surfaces in two sub-reaches of the river where wild Kootenai Sturgeon currently spawn” (Kootenai Tribe of Idaho 2013). Construction of the SEPPS was completed in winter 2014.

The fourth and final year of IDFG monitoring of the SEPP was 2017. We monitored the biological responses of Kootenai Sturgeon during three life stages to evaluate how the SEPP influenced: 1) habitat selection by spawning females, 2) occurrence of spawning on the projects, and 3) larval production on the site. Habitat selection was evaluated using a Vemco VR2W Positioning System (VPS) system. VPS is a low-cost, non-real-time underwater acoustic fine-scale positioning system, using the same equipment deployed in our passive telemetry array.

VPS arrays allowed us to track individuals in two dimensions during the spawning period. The location data obtained from the study allowed us to directly evaluate habitat selection behaviors of spawning Kootenai Sturgeon. A full write-up of the VPS habitat selection is provided in Usvyatsov (2020).

Spawning Occurrence

We deployed artificial substrate mats (McCabe and Beckman 1990) in the spring of 2017 and 2018 to evaluate whether Kootenai Sturgeon were spawning either on or off the SEPPs and specifically to determine if spawning females used the substrate additions at higher frequencies compared to non-substrate addition ("control") sites. Our sampling efforts were targeted on or near both the SEPP sites to evaluate their efficacy for spawning adults. We also sampled an area near Bonners Ferry (rkm 246) to document egg deposition near town. We sampled Shorty's Island and Myrtle Creek using a systematic design (Figure 1.7) with 21 mats at each site. Seven mats were deployed in three independent treatment locations including one on each habitat enhancement site (Strata 1) and two control locations. At each location the first control site (Strata 2) was approximately 500 m downstream of the substrate enhancement site in an area that has traditionally yielded eggs and had similar physical conditions to the treatment site prior to the installation of the SEPP. An additional control site (Strata 3) was on river left (West of Strata 2), 150 m downstream of the treatment site in an area where few eggs have been collected in the past. Total area sampled within the treatment reach and at the two control sites were identical. Designs were identical for both the Shorty's Island and Myrtle Creek SEPP sites (Figure 1.7). We retrieved and reset mats at least twice per week, and all eggs were stored in formalin and brought back to the laboratory for analysis. All eggs were staged by viewing at 120X magnification under a dissecting microscope to estimate spawn date following methods described by Beer (1981). More details on the substrate sampling methods are available in Rust and Wakkinen (2010).

We used a multi-season occupancy model, along with our egg mat data, to estimate the probability of egg deposition while accounting for detection probability. The goal of this analysis was to determine the influence the SEPP had on the probability of egg deposition. The developed model was largely an extension of a single season occupancy model (Royle and Kéry 2007). The state-space formulation of the model used in the analysis is presented below:

$$y_{ijt} \sim Bernoulli(z_{ijt} \cdot p)$$
$$z_{ijt} \sim Bernoulli(\psi_{ijt})$$
$$logit(\psi_{ijt}) = BX$$

where y_{ijt} = 1 if one or more eggs are detected on an eggmat and y_{ijt} = 0 otherwise. The parameter p is the probability of detecting one or more eggs on an eggmat given they are present, z_{ijt} is a binary state parameter that tracks the latent egg deposition process, ψ_{ijt} is the probability that one or more eggs are deposited on an eggmat. The last equation is a simple logistic regression on ψ_{ijt} that allowed us to evaluate how different environmental factors influenced the probability of egg deposition.

We included four environmental covariates in the logistic regression analysis on ψ_{ijt}. The first covariate and indicator was for eggmat location - Shorty's Island or Myrtle Creek. We also included indicator variables to represent the effect that each strata would have on the probability of egg deposition. We included temperature as a pair of indicator variables, one for if the temperature during the eggmat deployment was between 8-10°C and one for if the temperature was +10°C. Lastly, we evaluated the effect of the change in flow during eggmat deployment. The variable was calculated by taking the difference in flow during a seven-day window that overlapped with eggmat deployment. Thus, negative numbers represented a drop in flow and positive numbers represented an increase in flow.

The egg occupancy model was fit in a Bayesian framework using STAN in the package rstan() in the statistical program R (R Core Development Team 2018; Stan Development Team 2018). We sampled the posterior using three chains with 5,000 samples in each chain. 2,500 of each of those samples were allocated to the warmup sampling. Convergence was evaluated by checking traceplots for adequate mixing as well as through the Gelman-Rubin statistic. All posteriors were reported using 90% highest posterior density (HDP) intervals.

Larval Production

Hatching success was determined through extensive larval sampling around the Shorty's Island and Myrtle Creek SEPP sites and occurred concurrently with egg mat sampling. We continued the larval sampling design of Crossman and Hildebrand (2014) for the duration of the study. Two nets were paired on a metal frame with each net being 80 cm wide X 60 cm high with 1.6 mm net mesh. The cod end of each net was made of a 3 gallon polyurethane bucket with 1.6 mm mesh windows. While deployed, each frame was independently anchored to the substrate which allowed us to retrieve each set of nets without resetting the anchor. When the nets were retrieved the cod ends were replaced and the debris was examined for larval sturgeon. In 2017, we sampled two pairs of frames (i.e., four nets per site) above and below each SEPP and one pair of nets in the lower end of the straight reach near Bonners Ferry (near rkm 245.0). We attempted to begin sampling about 10 days after the first eggs were observed on the egg mats. Sampling efficiency and duration was a function of river conditions (e.g., debris and flow) and sampling effort and duration increased as the hydrograph receded and debris load reduced. Full 24 h sets began once drifting debris was at a low enough level to allow nets to fish the entire night period without debris fully saturating net holding capacity.

Juvenile Kootenai Sturgeon Sampling

Beginning in 1990 and continuing to the present, the KTOI and BC hatcheries have released over 286,000 juvenile Kootenai Sturgeon. There were two primary reasons for sampling juvenile Kootenai Sturgeon. The first was to evaluate the distribution, stock status, and densities of marked hatchery juveniles and the second was to document any natural recruitment as determined by capture of unmarked juveniles. Since the Kootenai Sturgeon population is transboundary, data collected in Canada was included. We used weighted multifilament gill net with 2.5, 5.1, and 7.6 cm stretch mesh to sample juvenile and young-of-the-year (age-0) sturgeon. We followed the sampling methodologies provided in Ross et al. (2015). Gill nets were set during the daytime and checked every hour to reduce mortality and all sturgeon were released alive. All fish that were sampled in gill nets were checked for passive integrated transponder (PIT) tags as well as scute removals. The markings allowed us to keep track of recaptures and differentiate between wild- and hatchery-origin individuals.

From 1992 to 2004, prior to release, each hatchery reared sturgeon received a PIT tag, and a pattern of scutes was removed either at the KTOI hatchery or at the Kootenay Trout and Sturgeon Hatchery located in Ft. Steele, BC (operated by the Freshwater Fishery Society of BC as the backup facility for the KTOI). Most (i.e., 92%) of the juvenile Kootenai Sturgeon released from 2005-2007 were not PIT tagged; however, scutes were removed from each fish prior to release. Most hatchery reared juvenile sturgeon released in the Kootenai River after 2007 were again PIT tagged and all had scutes removed. PIT tagging fish prior to release provided a unique individual identifier for each fish and allowed tracking of the size at release, rearing facility, release location, and time of release as well as subsequent individual performance (e.g., annual growth, etc.). Scute removal patterns only identify brood year and rearing location; however, due to the nature of such marks, there can be subjective errors with applying and recording scute patterns (e.g., miscounts or incomplete scute removals). Fork (FL) and total length (TL), weight, PIT tag numbers, fish condition, and scute removal patterns were recorded for each sampled sturgeon. Additionally, pectoral fin ray sections were removed from all wild juvenile Kootenai Sturgeon for age estimation. Each newly encountered wild sturgeon received a PIT tag and the second left scute was removed for future identification.

Juvenile Hatchery Kootenai Sturgeon Age-1 Survival and Total Abundance

For the analysis we used the large dataset of capture and recapture data from 1992-2018 for Kootenai White Sturgeon. We analyzed annual age-specific survival using the live recaptures (Cormack-Jolly-Seber; CJS) model in Program MARK (White and Burnham 1999). The parameters in the model were annual apparent survival (ϕ) and conditional capture probability (p). Additional specific estimations were performed following methods described by Dinsmore et al. (2014). We then constructed a deterministic stage-based population projection model to estimate the number of fish each year that were attributed to each release year. The estimate was constructed following methods described by Beamesderfer et al. (2013) and Dinsmore et al. 2015). We then summed estimated abundance across age classes to get annual estimates of the population. It is important to note that the model used estimates of apparent survival (apparent survival is the product of true survival and fidelity), which may be biased low if there was substantial permanent emigration from sampling sites. In addition, we present these estimates with an understanding that they are most informative of population processes when combined with other information such as annual survival, recruitment, and dispersal from stocking locations.

Juvenile Hatchery Kootenai Sturgeon Growth

Growth of hatchery produced Kootenai Sturgeon was estimated using a von Bertalanffy growth function (VBGF; Bertalanffy 1938). Attempts were made to estimate VBGF on annually recaptured fish through time. However, if recaptures of a particular age class were not adequate, length-at-capture that year class was estimated over time by pooling data across two or more sample years. In addition, we evaluated differences in growth for fish recaptured in Kootenay Lake versus the mainstem Kootenai River. Sample years were separated into ten-year increments: 1995-2004, 2005-2014, and 2015-2024 for evaluation through time. Although movement rates were unknown, we estimated that the majority of fish were located in their respective recapture strata for most of their growth. Fork length for this model was termed the “typical” version of the VBGF throughout the module. The typical VBGF was represented by:

$$l_t = L_\infty(l\text{-}e^{-K(t-t_0)}$$

Where L_∞ was the mean maximum length a fish could reach given an infinite lifespan ($t = \infty$), K was the growth coefficient, t was age, t_o was the “age” the fish length was zero, and L_t was the length at age t.

RESULTS

Water Levels, Discharge, and River Temperature

The 2017 sturgeon flow operation was marked by a long duration of high flows. In fact, the high flows did not allow for a traditional "peak" in hydrograph; instead, discharge remained around 10 kcfs from mid-March to the beginning of July (Figure 1.8). The high springtime flows were the result of a heavy snowpack from the winter and a very wet spring. High local inflows below the dam prevented the dam operations from shaping the hydrograph. Historically, before the construction of Libby Dam, stream flows would increase gradually beginning in March (Figure 1.2). Flows would increase until the beginning of June before gradually tapering off. In 2017 we saw a very abrupt increase in discharge beginning in March, followed by a sharp decrease in flows beginning in mid-June. The duration of high flows was almost 3.5 months, which was a much longer period of high flows compared to the prior four years. Unlike 2017, the flow peaks in the spring of 2018 and 2019 were shorter in duration and increased in mid-April and abruptly declined by mid-June (Figure 1.8).

Water temperatures in the spring of 2017-2019 were similar in the Straight Reach (at Bonners Ferry) during sturgeon flow augmentation, ranging from 4 to 18°C. Temperatures of 10°C at Bonners Ferry were observed in the middle of May in 2017 and 2018, but were colder at that same time period in 2019 (Figure 1.8). As with most years, the decrease in volume released from Libby Dam (during the receding limb of the hydrograph) coincided with the increase in river temperatures at the spawning locations near Bonners Ferry.

Adult Kootenai Sturgeon Sampling

Between April 18 and October 12, 2017, IDFG, British Columbia Ministry of Forests, Lands, Natural Resources and Rural Development (FLNRORD), and KTOI crews expended more than 1500 rod-hours to capture 170 adult Kootenai Sturgeon by angling (Table 1.1). The effort resulted in a mean total catch rate of 0.107 fish per rod-hour. The vast majority of the effort took place in the Idaho section of the Kootenai River (rkms >170). The effort primarily occurred during the spring season, which coincided with adult broodstock collection for KTOI. Of the adults collected, the majority (~87%) were recaptures and nine were wild fish that had not been previously captured. In 2018, we sampled between May 2 and October 23, where crews expended more than 1200 rod-hours to capture 111 adult Kootenai Sturgeon by angling (Table 1.2). The effort resulted in a mean total catch rate of 0.103 fish per rod-hour. The vast majority of the effort took place in the Idaho section of the Kootenai River (rkms >170). As with most years, the effort primarily occurred during the spring season, which coincided with adult broodstock collection for KTOI. Of the adults collected, the majority (~85%) were recaptures and five were wild fish that had not been previously captured.

Additionally, IDFG and FLNRORD sampled for adult sturgeon using set lines in the spring and fall of 2017 and 2018. A total of 38,495 hook-hours was expended in 5/1/2017-4/30/18, capturing 109 adults, which resulted in 0.003 fish per setline hour (Table 1.3). In 5/1/2018-10/23/2018, approximately 33,161 hook-hours captured 54 adults and resulted in 0.002 fish per setline hour (Table 1.4). Lastly, only 10 adults (five each year) were sampled during the gill netting, all of which were recaptures.

Adult Kootenai Sturgeon Telemetry

We monitored adult White Sturgeon spawning migration, movement extent, and behavior during the Libby Dam flow augmentation operations using Vemco acoustic transmitters. Seven adult Kootenai Sturgeon were tagged with Vemco sonic transmitters in fall 2017 and four were tagged in spring 2018. All were tagged with Vemco V-16TP-6X transmitters to coincide with the 2015 SEPP evaluations using VPS. In addition to providing detailed information within the VPS arrays, the VPS tags were also compatible with the existing VR2W array and individuals containing these transmitters were included in large-scale movement analyses. To be considered a spawning adult for this evaluation, a sonically tagged adult needed to have made an upstream migration to at least Shorty's island (≥rkm 230). In 2017 we detected 54 spawning adults in the Idaho portion of the Kootenai River. Of the detected spawning adults, 43 (80%) tagged adult sturgeon moved upstream as far as rkm 240 (i.e., below Deep Creek), and 37 (68%) of the migrating adults went upstream as far as rkm 244.5 (i.e., Ambush Rock). Additionally, 29 of the tagged migrating adult sturgeon went upstream of the Highway 95 Bridge in Bonners Ferry into the braided reach in 2017 (Figures 1.9 and 1.10). The furthest upstream we detected a spawning adult was at rkm 255, just below the confluence with the Moyie River. In 2018 we detected 54 spawning adults in the Idaho portion of the Kootenai River. Of these spawning adults, 43 (80%) tagged adult sturgeon moved upstream as far as rkm 240 (Below i.e., Deep Creek), and 32 (59%) of the migrating adults went upstream as far as rkm 244.5 (i.e., Ambush Rock). Additionally, almost half (25) of the tagged migrating adult sturgeon went upstream of the Highway 95 Bridge in Bonners Ferry into the braided reach in 2018 (Figure 1.11 and 1.12). The furthest upstream we detected a spawning adult was at rkm 255, just below the confluence with the Moyie River. Further evaluation and discussion can be found in Chapter 2.

Substrate Enhancement Pilot Projects

Sampling in 2017 provided the fourth and final year of SEPP monitoring. Below are summaries of our egg mat and larval sampling with regards to the SEPPs. A full documentation of VPS system that was implemented at the SEPP sites is provided in Usvyatsov (2020).

Spawning Occurrence

We deployed substrate mats in 2017 to evaluate the temporal extent of Kootenai Sturgeon spawning events in the Kootenai River. We sampled a total of 48,671.58 mat-hours between May 22 and July 10, 2017 and collected 256 eggs (Table 1.5). The highest catch and the highest catch rate came from the Myrtle Creek area (rkm 234.5, Table 1.6). Egg catch rates peaked just before June 1. The proportion of traps that eggs were detected on also peaked at this time. Egg collection on the SEPPs in 2017 were consistent with previous years where all the eggs were found in Strata 1 and 2 and no eggs were collected on the control sites (Strata 3).

The occupancy model showed that both strata and flow were the biggest factors influencing whether or not an egg mat was occupied with eggs. Detection probability, p, had a mean posterior value of 0.6468 and a fairly wide credible interval (Table 1.5). The wide credible interval is an indication of the sparse nature (zero inflation) of the data set. Site (Myrtle Creek vs Shorty's Island) was not a strong indicator of the probability of egg deposition. Egg mats that were placed in either strata 2 (on the SEPP) or strata 3 (downstream of SEPP) both had 90-fold increases in the probability that an egg would be deposited on them compared to mats placed in the control area (Strata 1; Table 1.7). In other words, it was almost completely improbable to have eggs deposited in Strata 1. However, these two effects were almost identical to one another, thus it seems to be equally likely that eggs would be deposited on strata 2 or strata 3. The temperature indicators also did not appear to influence the probability of egg deposition. The seven-day change in flow proved to be a strong indicator of the likelihood of egg deposition. Within the ranges observed, the greater the drop in the discharge, the more likely it was that eggs would be deposited on egg mats and, conversely, increases in flow were related to decreases in the same probability (Figure 1.13). The analysis indicated that a relatively rapid decline in flow resulted in a higher probability of egg deposition on collection mats.

Hatching Success

We sampled for larval sturgeon between June 27 and July 20 in 2017 and between June 11 and July 23 in 2018 for a total of 4508 and 2,816 hours of effort, respectively. Sampling effort was similar between the Shorty's Island and the Myrtle Creek sites. Effort was much lower at the sites near Bonners Ferry due to high debris loads in the water column. As a result, the Bonners Ferry site was sampled following a reduction in flow and subsequent debris loads. Only a single larval sturgeon was collected at Myrtle Creek in 2017, and one at Shorty's Island in 2018. Non-target larvae were collected at all three sites but were not quantified. Most of the non-target larval fish were Mountain Whitefish *Prosopium williamsoni*.

Juvenile Kootenai Sturgeon Sampling

In 2017, IDFG and FLNRORD sampled for juvenile Kootenai Sturgeon with gill nets between June 29 and September 27 in the Idaho and Canadian sections of the Kootenai River and Kootenay Lake. We sampled 25 sites between rkm 18.0 and 244.5 and collected 1,745 juvenile sturgeon with 510 hours of effort (Figure 1.1, Table 1.8). In 2018, IDFG and FLNRORD sampled for juvenile Kootenai Sturgeon with gill nets between July 16 and October 10 in Idaho and Canadian sections of the Kootenai River and Kootenay Lake. We sampled 25 sites between rkm 18.0 and 244.5 and collected 1,765 juvenile sturgeon with 543 hours of effort (Figure 1.1, Table 1.9). Although the highest catch rate was recorded in Kootenay Lake at the southern delta (rkm 121), the majority of the catch was from the Idaho portion of the river (>170.0 rkm), which coincided with the highest catch rates, as well (Tables 1.10-1.11). Juvenile sturgeon were well distributed throughout the river and lake. Ferry Island (rkm 207), Rock Creek (rkm 215), and Ambush Rock (rkm 244.5) had the highest catch rates in the river, but most areas throughout the river had catch rates that exceeded one fish per hour. All sizes of gill nets caught sturgeon at comparable rates; however, the 2-inch mesh had the highest catch rates in 2017 (Figure 1.14). Likewise in 2017, the 2-inch mesh was fished the most, representing 49% of the sets. The average fork and total length of the juvenile Kootenai Sturgeon captured in gill nets was 45.3 cm and 52.7 cm, respectively, and weight of averaged 0.86 kg in 2017. A similar evaluation breakdown was not performed on 2018 data.

Only ten wild juvenile Kootenai Sturgeon were captured in gill nets in 2017 and four in 2018. The TL of these individuals ranged from 26.0 to 75.5 cm, and weights ranged from 0.095 to 1.36 kg. All wild juveniles were aged by sectioning the pectoral fin ray and counting annuli. Figure 1.16 shows the year class assignments from a sample of the wild juvenile Kootenai Sturgeon collected between 1977 to present that could be aged. Figure 1.15 shows the number of wild juvenile Kootenai Sturgeon collected annually from 1977 to 2017.

Juvenile Hatchery Kootenai Sturgeon Age-1 Survival and Total Abundance

Similar to results reported by Dinsmore et al. (2015), age-1 survival was high during the early to mid-1990s through around 2000. The period of high survival was followed by a precipitous decline in survival from approximately 90% to an estimated 4% in a seven-year period (Figure

1.17). Since then, age-1 survival has averaged around 8-10%. Subsequent juvenile abundance estimates also reflected this leveling off of survival resulting in an abundance of around 14-16,000 individuals since 2005 (Figure 1.18)

Juvenile Hatchery Kootenai Sturgeon Growth

Growth curves using the VBGF were unable to be properly fit due to poor estimates of K between the river and lake. More accurate estimates will be attempted when data on transition rates between the lake and the river environments are collected. In addition, we were unable to obtain enough recaptures of individual fish; therefore, we combined fish length at capture by brood year and location of capture (lake or river). General descriptive plots revealed growth differences between fish reared on ambient river water and those reared on accelerated temperatures by the KTOI hatcheries. It is apparent that fish captured in the mainstem of the river and reared on ambient Kootenai River water have declined in length-at-capture over time, yet those captured in the lake did not exhibit a similar reduction (Figure 1.19). Conversely, those fish captured in the river and the lake that were reared on accelerated temperatures exhibited little difference in growth (Figure 1.20). In addition, evaluation of individual year class recaptures (e.g. 2009) suggested that those reared on ambient river temperatures have reduced in their growth over time (Figure 1.21).

DISCUSSION

Libby Dam Kootenai Sturgeon flow augmentation operations for 2017 and 2018 consisted of a single period of high flow, beginning at the start of March and continuing through April. The periods of high flow were then followed by flows that quickly descended to summer-time base flows starting in mid-June. Based on our sonic telemetry and egg mat collections, this operation of extended high flows improved upstream movement of adult sturgeon into better spawning habitat, and the rapid declining flows following after mid-June appears to optimize egg deposition. The flow management in 2017 was a departure from previous years due to the size and duration of the high flow events. Additionally, we saw an increase in the total number and proportion (~50%) of the tagged spawning group that migrated above Bonners Ferry (rkm 246). Flow operations in 2017 provided evidence that Kootenai Sturgeon may respond to extended duration high flows by migrating further upstream (see Chapter 2 for further analysis and discussion).

Sonic tagging efforts to monitor Kootenai Sturgeon movements have generally been focused on adults in the past; however, we believe it would be beneficial to incorporate more juvenile fish into those efforts. Relatively little is known about the movement of juvenile Kootenai Sturgeon especially transition between the lake and river environments. Stephenson and Evans (2018) summarized movements after three years of tracking juveniles that were tagged in lake sampling and there were limited movements into the river. Close to half of the lake tagged juveniles were detected in the river, but the mean time a juvenile spent in the river in a given year was 27.8 days (SE = 3.4; range 1-165 days), and all juveniles returned to the lake. Juvenile Kootenai Sturgeon sampled in Kootenay Lake tend to be larger than same-age fish sampled in the river; however, it remains unclear to what extent they use the lake habitat. Additionally, it would be beneficial to tag hatchery origin juveniles (>120 cm fl) that would potentially join the spawning population in the next 5-10 years. Currently, we have deployed tags (*n* >10) into juvenile fish that we expect to transition to spawning adults in the next decade. These fish will represent the first cohort of hatchery origin spawners and would likely provide insight on factors that influence the transition from juvenile to spawning adults.

Larval captures continued to be low in 2017 and 2018, with only one captured in tows in both years. Currently, we are unable to determine whether the low catch rates reflect extremely low larval production, low capture efficiency with current nets, or both. We have started working with U.S. Geological Survey to develop a particle drift model to aid in determining the efficacy of our larval nets. The drift model incorporates a 2-D hydraulic model of the Kootenai River and then simulates particles deterministically drifting downstream. Such a model can allow us to determine which locations on the river we would expect to see a high concentration of particles and better understand how larval Kootenai Sturgeon may drift downstream. Additionally, the model may be used to provide simulated estimates of our gear efficiency. If we understand how effective our gear is at capturing larval sturgeon, we may be able to back-calculate total larval production in the river. In the future the drift model will be used to inform our larval sampling design in order to maximize our sampling efficiency.

After four years of monitoring, the SEPPs have not produced clear evidence to suggest they have increased egg deposition rates or larval/juvenile production to date. If substantial production has occurred and larval sampling was unable to detect it, the next life stage (juveniles) should be detected in gillnet sampling. The results of our analysis (also see Usvyatsov 2020) suggest that the SEPPs did not deter Kootenai Sturgeon from spawning, as evidenced by the substantial numbers of adults frequenting these locations at the peak of spawning in addition to the collection of eggs to confirm spawning. However, the egg occupancy model which evaluated the relation of flow shape and egg deposition suggested that egg deposition had an equal probability of occurring on or off of the SEPP. The locations (Strata 1 and 2) where almost all eggs were observed at both sites are on a clay shelf that exists throughout the lower Kootenai River. Paragamian et al. (2009) suggested that Kootenai Sturgeon are selecting the area around the current SEPPs because it is the area of highest available velocities and depths over a range of flows. Although the SEPP did not impede spawning through changing velocities or depths, the results of our extensive larval sampling also suggested that the SEPPs have not resulted in an increase in larval production, despite successful egg deposition. It does not appear that substrate enhancement projects alone will increase larval hatching success for Kootenai Sturgeon. The SEPP monitoring process has, however, allowed us to further narrow the window of when the recruitment failure may be occurring. Future efforts should be focused on investigating whether eggs deposited in the lower Kootenai River are in fact hatching. Continuing to refine and update our understanding of the spawning and recruitment process for Kootenai Sturgeon will aid in developing new approaches to effectively address factors limiting recruitment.

We observed high gillnet catch rates of juvenile Kootenai Sturgeon in 2017 and 2018 in both the lake habitat and lower portion of the Kootenai River, which was similar to previous years. The 10-year trend of annual catch rates continues to increase throughout the river and lake and juvenile Kootenai Sturgeon are well distributed throughout the system. Although high catch rates continue to confirm the success of KTOI conservation aquaculture program, our evaluation of survival highlight the impacts of increased of juvenile hatchery fish in the system. The updated analysis of juvenile hatchery Kootenai Sturgeon show that current survival and growth rates of age-1 fish are substantially lower compared to those estimated more than 20 years ago. In addition to reduced survival, the levelling of population growth despite increased stocking, suggests that density may be presenting another bottleneck that could potentially affect hatchery and wild fish at similar levels. Although the stocking program is releasing numbers commensurate with recovery targets, there is evidence that suggests that continued stocking at these levels may cause density effects to intensify. Future investigations should focus on understanding how density may influence juvenile survival and growth rates.

RECOMMENDATIONS

1. Depending on annual water supply, provide augmented flows from Libby Dam during the spawning season (approximately April 1st-July 1st) to achieve ≥850 m^3/s at Bonners Ferry to encourage spawning Kootenai Sturgeon to move further upstream into more suitable spawning and egg incubation habitats. In addition, provide stable or increasing temperatures using the selective withdrawal gate system at Libby Dam as needed to assist and maintain spawning migration of Kootenai Sturgeon.

2. Develop criteria to determine optimal timing for quickly lowering discharge to initiate egg deposition. Ideally, this would be done when spawning adult Kootenai Sturgeon are as far upstream as possible.

3. Identify the limiting factor in larval production in lower river spawning sites. Particularly, we should focus on determining hatching success.

4. Develop a stochastic population model to evaluate the influence of potential density effects on population structure and dynamics.

5. Continue collecting fin rays from hatchery reared juvenile sturgeon to evaluate changes in growth over time using incremental growth analysis.

TABLES

Table 1.1. Angling sampling effort of adult and juvenile Kootenai Sturgeon during 2017. All sampling was done by Idaho Department of Fish and Game (ID), Kootenai Tribe of Idaho (KTOI) and British Columbia Ministry of Forests, Lands, Natural Resource Operations and Rural Development (BC). Untraceable recaptures refers to fish that were captured but their origin (wild vs hatchery) is unknown due to incomplete marking or tagging. Sampling done in the Montana portion of the Kootenai River is not included.

							Total Catch		Total CPUE		Recaptures		Untraceable Recaptures	
Season	Location	RKM Range	Crew	Start Date	End Date	Total Effort (Rod hrs)	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile
Fall	Lake	≤122	BC	9/11/2017	10/12/2017	60.134	8	6	0.133	0.100	5	6	2	0
			KTOI	9/11/2017	9/14/2017	96.39	11	8	0.114	0.083	9	7	1	0
	BC River	123-170	BC	9/20/2018	10/12/2017	3.898	1	0	0.257	0.000	1	0	0	0
	ID River	≥170	ID	9/20/2017	10/12/2017	26.815	19	7	0.709	0.261	15	6	2	1
Spring	Lake	≤122	BC	4/18/2017	4/19/2018	5.3	0	0	0.000	0.000	0	0	0	0
	BC River	123-170	BC	4/18/2017	4/18/2017	1.941	0	0	0.000	0.000	0	0	0	0
			KTOI	5/1/2017	4/30/2018	1388.955	131	11	0.094	0.008	115	11	11	0

Table 1.2. Angling sampling effort of adult and juvenile Kootenai Sturgeon during 2018. All sampling was done by Idaho Department of Fish and Game (ID), Kootenai Tribe of Idaho (KTOI) and British Columbia Ministry of British Columbia Ministry of Forests, Lands, Natural Resource Operations and Rural Development (BC). Untraceable recaptures refers to fish that were captured but their origin (wild vs hatchery) is unknown due to incomplete marking or tagging. Sampling done in the Montana portion of the Kootenai River is not included.

							Total Catch		Total CPUE		Recaptures		Untracable Recaptures	
Season	Location	RKM Range	Crew	State Date	End date	Total Effort (Rod hrs)	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile
Fall	Lake	≤122	BC	9/10/2018	9/27/2018	36	3	6	0.082	0.165	2	5	1	1
			KTOI	9/10/2018	9/12/2018	135	24	11	0.178	0.081	18	10	4	1
	BC River	123-170	BC	9/18/2018	9/26/2018	3	0	0	0.000	0.000	0	0	0	0
	ID River	≥170	ID	7/16/2018	10/23/2018	42	12	7	0.288	0.168	10	6	2	1
Spring	Lake	≤122	BC	5/2/2018	5/3/2018	7	0	0	0.000	0.000	0	0	0	0
			KTOI	5/2/2018	6/20/2018	1052	72	4	0.068	0.004	64	2	5	1

Table 1.3. Setline sampling effort of adult and juvenile Kootenai Sturgeon during 2017. All sampling was done by Idaho Department of Fish and Game (ID) and British Columbia Ministry of British Columbia Ministry of Forests, Lands, Natural Resource Operations and Rural Development (BC). Untraceable recaptures refers to fish that were captured but their origin (wild vs hatchery) is unknown due to incomplete marking or tagging.

							Total Catch		Total CPUE		Recaptures		Untraceable	
Season	Location	RKM Range	Crew	Start Date	End Date	Total Effort (Hook hrs)	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile
Fall	Lake	≤122	BC	9/18/2017	10/12/2017	988	12	11	0.0121	0.0111	8	11	1	0
	BC River	123-170	BC	9/19/2017	10/12/2017	1899	8	6	0.0042	0.0032	7	6	1	0
	ID River	≥170	ID	9/20/2017	10/26/2017	355	7	5	0.0197	0.0141	7	4	0	1
Spring	Lake	≤122	BC	4/18/2018	4/19/2018	396	1	1	0.0025	0.0025	1	1	0	0
	BC River	123-170	BC	4/18/2018	4/19/2018	577	2	1	0.0035	0.0017	2	1	0	0
	ID River	≥170	ID	5/1/2017	4/30/2018	34281	76	9	0.0022	0.0003	73	8	1	1

Table 1.4. Setline sampling effort of adult and juvenile Kootenai Sturgeon during 2018. All sampling was done by Idaho Department of Fish and Game (ID), British Columbia Ministry of British Columbia Ministry of Forests, Lands, Natural Resource Operations and Rural Development (BC), and Montana Fish, Wildlife, and Parks (MT). Untraceable recaptures refers to fish that were captured but their origin (wild vs hatchery) is unknown due to incomplete marking or tagging.

							Total Catch		Total CPUE		Recaptures		Recaptures	
Season	Location	RKM Range	Crew	State Date	End date	Total Effort (Hook hrs)	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile
Fall	Lake	≤122	BC	9/10/2018	10/11/2018	1,224	21	32	0.017	0.026	20	25	0	7
	BC River	123-170	BC	9/18/2018	10/11/2018	2,261	8	6	0.004	0.003	7	5	1	1
	ID River	≥170	ID	10/11/2018	10/23/2018	295	12	6	0.041	0.020	9	4	1	2
	MT River	≥170	MT	7/3/2018	10/23/2018	18,609	0	36	0.000	0.002	0	28	0	8
Spring	Lake	≤122	BC	5/2/2018	5/4/2018	352	6	3	0.017	0.009	5	3	1	0
	BC River	123-170	BC	5/3/2018	5/4/2018	759	1	2	0.001	0.003	1	2	0	0
	ID River	170-276	ID	5/1/2018	5/31/2018	3,189	6	0	0.002	0.000	6	0	0	0
	MT River	≥276	MT	5/15/2018	6/29/2018	6,472	0	22	0.000	0.003	0	21	0	1

Table 1.5. Total effort and catch of Kootenai River White Sturgeon eggs via artificial substrate mat sampling during the spring (May 22 – July 10) of 2018. Three sites were sampled continuously throughout the period.

Site Description	River km	Depth Range (ft)	Temperature Range (°C)	Total Mat Effort (Hours)	Total Eggs	CPUE
Shorty's Island	230.5	8 - 49	9.5 - 16	20729.88	95	0.0046
Myrtle Creek	234.5	12 - 73	9.4 - 16	17582.68	139	0.0079
Bonners Ferry	245	5 - 40	9.6 - 16	10359.02	22	0.0021

Table 1.6. Distribution of eggs near KTOI SEPPs from 2014 to 2017 by artificial substrate mats. Effort was similar across years. Strata 1 refers to mats that were on the SEPP, Strata 2 refers to mats that were just downstream of the SEPP, and Strata 3 refers to mats that were on a dissimilar substrate. Numbers represent the mean catch per unit effort (number of eggs/h) of mats at each location. Standard deviations are in parentheses. In each year, egg mats were deployed from the last week of May to the first week of July.

			CPUE			
Site Description	River Km	Strata	2014	2015	2016	2017
Shorty's Island	230.5	1	0.004 (0.0188)	0.0015 (0.013)	0.0058 (0.0256)	0.0154 (0.0878)
	230.5	2	0.0031 (0.0108)	0.0053 (0.0285)	0.0052 (0.0226)	0.0106 (0.0589)
	230.5	3	0 (0)	0 (0)	0 (0)	0 (0)
Myrtle Creek	234.5	1	0.0092 (0.0396)	0.0014 (0.0124)	0.0158 (0.0745)	0.0067 (0.026)
	234.5	2	0.0112 (0.0476)	0.0132 (0.0661)	0.0082 (0.0511)	0.0055 (0.0236)
	234.5	3	0 (0)	0 (0)	0 (0)	0.0032 (0.0148)

Table 1.7. Parameter estimates from the posterior distribution of the egg mat occupancy analysis. All presence parameters are reported in the logit scale. All intervals are a 90% highest density probability credible intervals. The continuous variable (β_{flow}) was estimated after standardizing the covariate.

Parameter	Description	Mean	Median	Credible Interval
p	Detection Probability	0.6258	0.6468	(0.239, 0.964)
β_{Int}	Presence : Intercept	-6.5361	-7.1081	(-9.997, -3.153)
β_{site}	Presence : Myrtle Cr. Site effect	-0.0390	-0.0302	(-1.335, 0.952)
$\beta_{Strata2}$	Presence : Strata 2 effect	4.5716	4.2920	(2.073, 7.018)
$\beta_{strata3}$	Presence : Strata 3 effect	5.0905	4.7921	(2.642, 7.93)
β_{temp8}	Presence : Temperapture effect between 8 and 10 °C	1.4069	1.7519	(-2.561, 5.728)
β_{temp10}	Presence : Temperapture effect greater than 10 °C	1.5626	2.0031	(-2.156, 5.925)
β_{flow}	Presence : Effect of the 7 day change in flow	-0.9968	-0.9581	(-1.59, -0.503)

Table 1.8. Gillnet sampling effort of adult and juvenile Kootenai Sturgeon during 2017. All sampling was done by Idaho Department of Fish and Game (ID) and British Columbia Ministry of British Columbia Ministry of Forests, Lands, Natural Resource Operations and Rural Development (BC). Untraceable recaptures refers to fish that were captured but their origin (wild vs hatchery) is unknown due to incomplete marking or tagging. Sampling done in Montana portion of the Kootenai River is not included.

							Catch		CPUE		Recaptures		Untraceable Recaptures	
Crew	Location	RKM Range	Start Date	End Date	Total Number of Sets	Total Effort (hr)	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile
BC	Lake	≤122	8/3/2017	9/12/2017	65	58.335	2	389	0.034	6.668	2	282	0	106
BC	BC River	123-170	7/17/2017	9/27/2017	104	123.933	1	217	0.008	1.751	1	154	0	61
ID	ID River	≥170	6/29/2017	9/27/2017	267	328.186	2	1139	0.006	3.471	1	975	1	163

Table 1.9. Gillnet sampling effort of adult and juvenile Kootenai Sturgeon during 2018. All sampling was done by Idaho Department of Fish and Game (ID) and British Columbia Ministry of British Columbia Ministry of Forests, Lands, Natural Resource Operations and Rural Development (BC). Untraceable recaptures refers to fish that were captured but their origin (wild vs hatchery) is unknown due to incomplete marking or tagging. Sampling done in Montana portion of the Kootenai River is not included.

							Total Catch		Total CPUE		Recaptures		Untracable Recaptures	
Crew	Location	RKM Range	State Date	End date	Total Number of Sets	Total Effort (hr)	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile
BC	Lake	≤122	7/24/2018	9/17/2018	98	93	2	347	0.021	3.719	2	247	0	100
BC	BC River	123-170	7/16/2018	9/26/2018	109	128	1	278	0.008	2.177	1	204	0	74
ID	ID River	≥170	7/30/2018	10/10/2018	242	322	2	1140	0.006	3.538	2	986	0	153

Table 1.10. Total catch and effort of Kootenai Sturgeon via gill net sampling by Idaho Department of Fish and Game and British Columbia Ministry of British Columbia Ministry of Forests, Lands, Natural Resource Operations and Rural Development summarized by river kilometer. Sampling occurred between June 29, 2017 and September 27, 2017. For reference, the US/Canada border is at rkm 170 and rkm <122 is in Kootenay Lake.

River Kilometer	Number of Sets	Hours of Effort	Adults Captured	Juveniles Captured	CPUE (fish/hour)
18.0	7	7.20	0	11	1.5276
25.0	1	1.95	0	6	3.0785
120.0	31	29.73	0	143	4.8095
121.0	26	19.45	2	229	11.8754
123.0	8	11.09	0	47	4.2400
130.0	16	20.27	1	60	3.0101
141.0	8	8.58	0	12	1.3986
145.0	16	15.96	0	42	2.6321
150.0	8	11.40	0	14	1.2279
157.0	8	9.49	0	7	0.7379
161.0	16	18.37	0	14	0.7622
165.0	16	18.68	0	18	0.9635
170.0	8	10.11	0	3	0.2967
174.0	24	28.65	2	35	1.2914
176.0	21	28.35	0	33	1.1642
190.0	25	30.60	0	85	2.7776
193.0	21	27.66	0	47	1.6993
205.0	19	23.23	0	165	7.1032
207.0	20	23.63	0	217	9.1840
207.5	19	23.60	0	97	4.1107
213.0	21	26.10	0	123	4.7125
215.0	16	23.10	0	152	6.5789
225.0	20	22.54	0	39	1.7300
234.5	9	13.02	0	20	1.5366
244.5	20	23.95	0	121	5.0532
253.5	4	4.99	0	0	0.0000
255.0	1	1.21	0	0	0.0000
256.0	6	7.18	0	1	0.1393
270.0	1	1.05	0	4	3.8059
279.6	2	3.59	0	0	0.0000
280.5	2	3.66	0	0	0.0000
306.5	8	12.09	0	0	0.0000

Table 1.11. Total catch and effort of Kootenai Sturgeon via gill net sampling by Idaho Department of Fish and Game and British Columbia Ministry of British Columbia Ministry of Forests, Lands, Natural Resource Operations and Rural Development summarized by river kilometer. Sampling occurred between July 16, 2018 and October 10, 2018. For reference, the US/Canada border is at rkm 170 and rkm <122 is in Kootenay Lake.

River Kilometer	Number of Sets	Hours of Effort	Adults Captured	Juveniles Captured	CPUE (fish/hr)
17	3	3.39	0	2	0.590
18	31	32.98	0	66	2.001
20	4	4.91	0	2	0.408
30	4	3.75	0	0	0.000
120	27	24.77	1	130	5.288
121	29	23.50	1	147	6.297
123	8	12.53	0	102	8.139
130	29	34.16	0	83	2.430
141.5	8	8.84	0	17	1.923
145	16	17.67	0	22	1.245
150	8	10.67	0	14	1.312
157	8	8.47	0	16	1.888
161	16	18.16	1	12	0.716
165	16	17.18	0	12	0.699
174	25	33.43	0	29	0.868
176	23	29.13	0	20	0.687
190	28	35.41	0	101	2.852
193	28	37.09	0	60	1.618
205	24	28.72	1	106	3.726
207	23	32.51	0	190	5.844
207.5	25	32.37	0	145	4.480
213	12	16.16	0	42	2.599
215	18	25.47	0	202	7.931
225	20	27.21	1	49	1.837
244.5	16	24.70	0	196	7.934

FIGURES

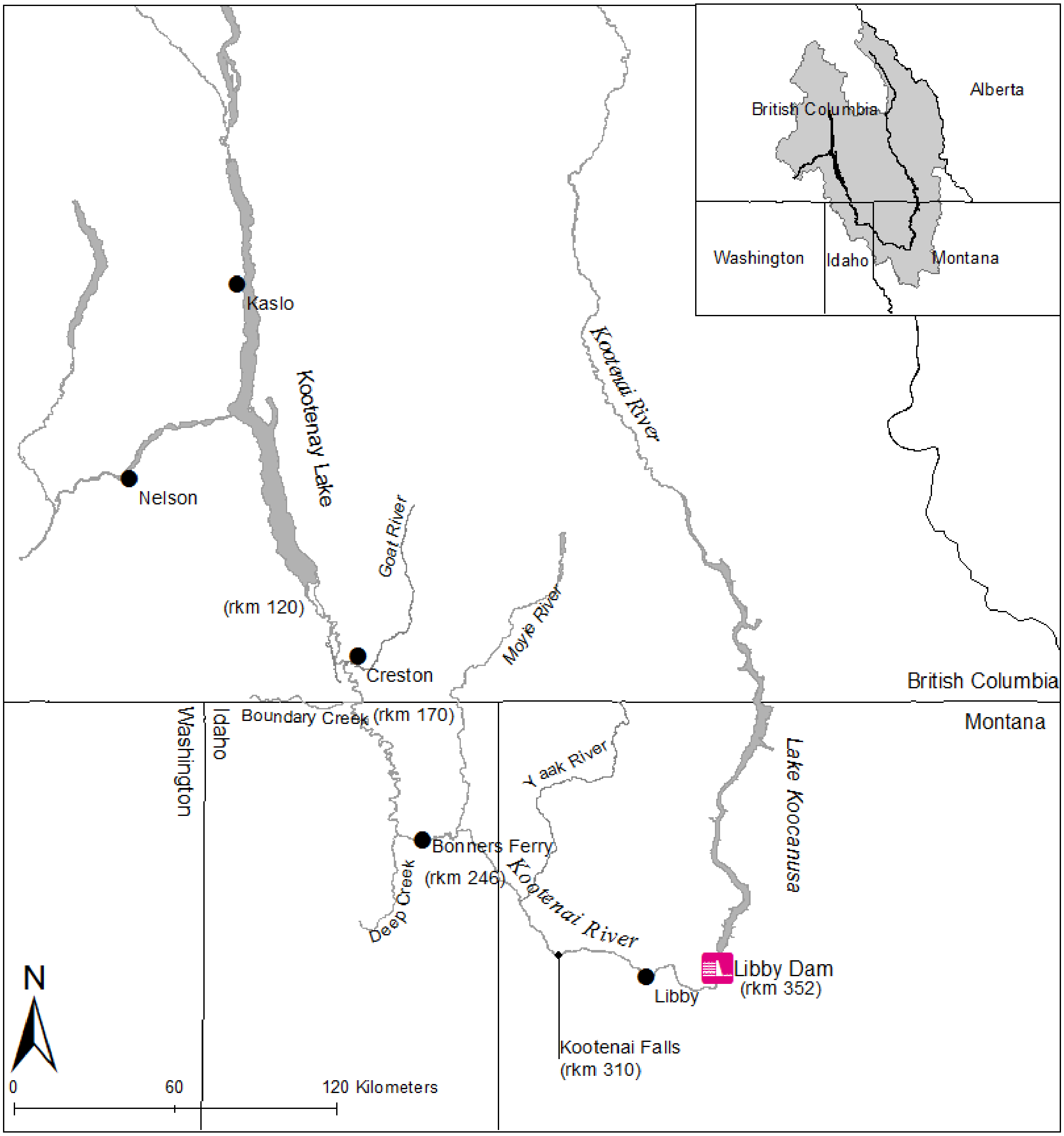

Figure 1.1. Location of the Kootenai River, Kootenay Lake, Lake Koocanusa, and major tributaries. River distances are from the northernmost reach of Kootenay Lake and are in river kilometer (rkm).

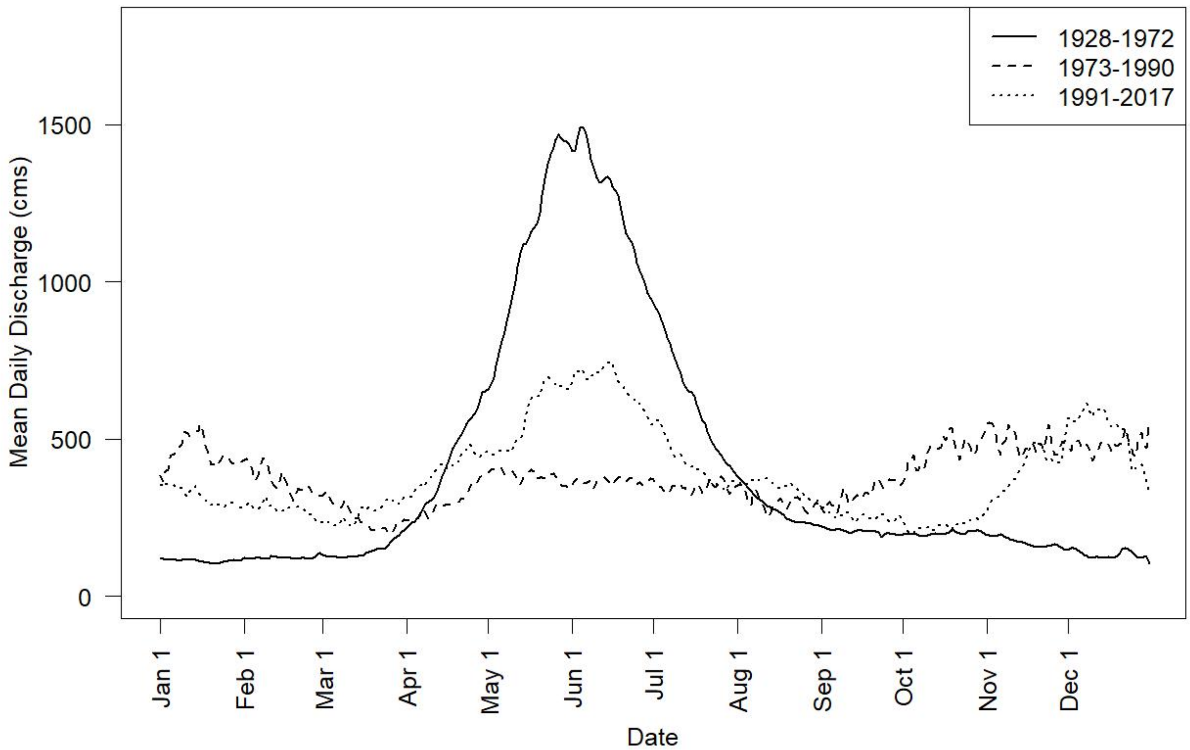

Figure 1.2. Mean daily flow patterns in the Kootenai River at Bonners Ferry, Idaho from 1928-1972 (pre-Libby Dam), 1973-1990 (post-Libby Dam) and 1991-2015 (post-Libby Dam with augmented springtime flows). Data obtained from USGS gauge located in Bonners Ferry, ID (12309500).

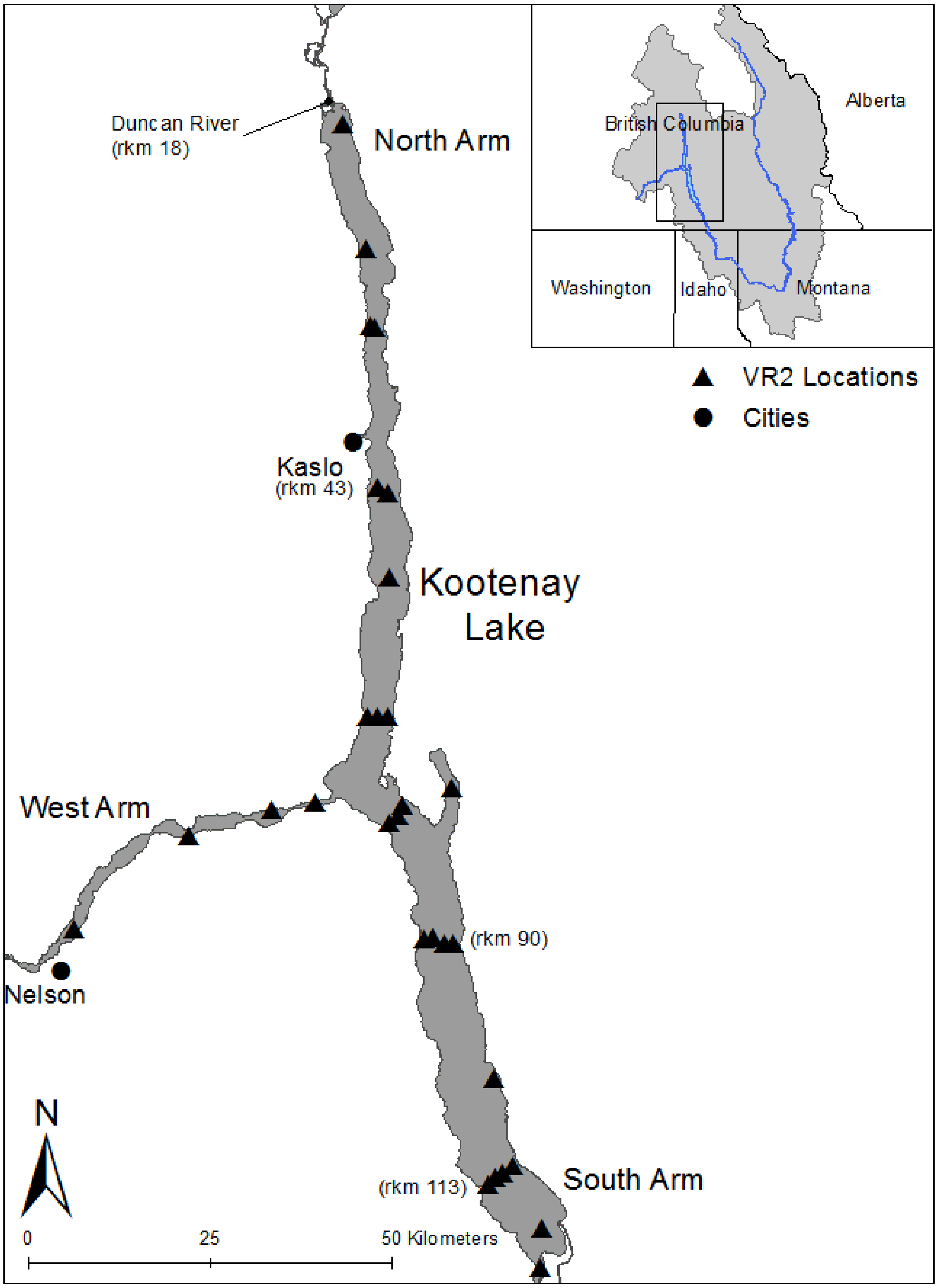

Figure 1.3. Location of Vemco VR2W remote sonic receivers in Kootenay Lake, BC, Canada.

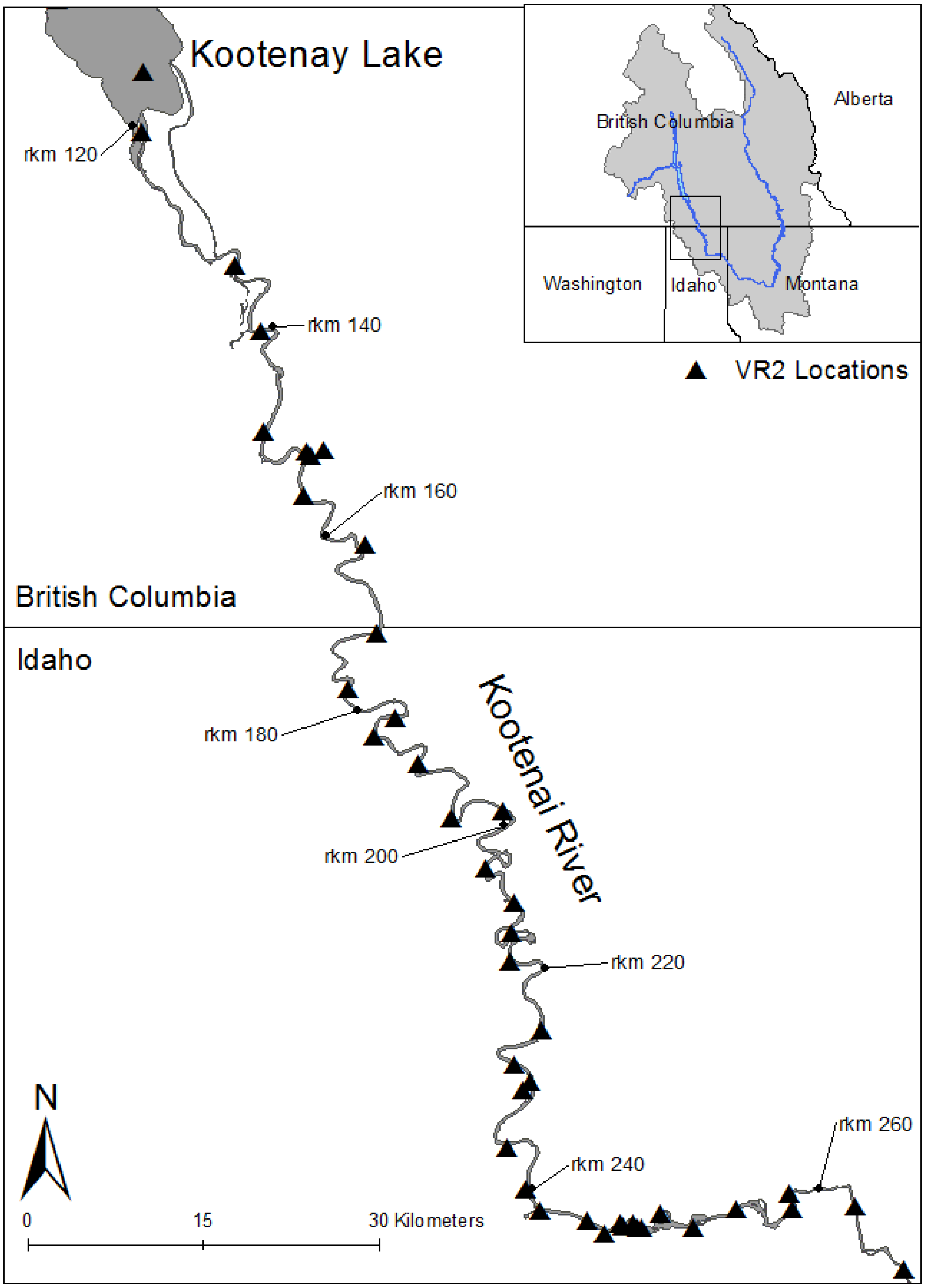

Figure 1.4. Location of Vemco VR2W remote sonic receivers in Kootenai River in BC, Canada, and Idaho.

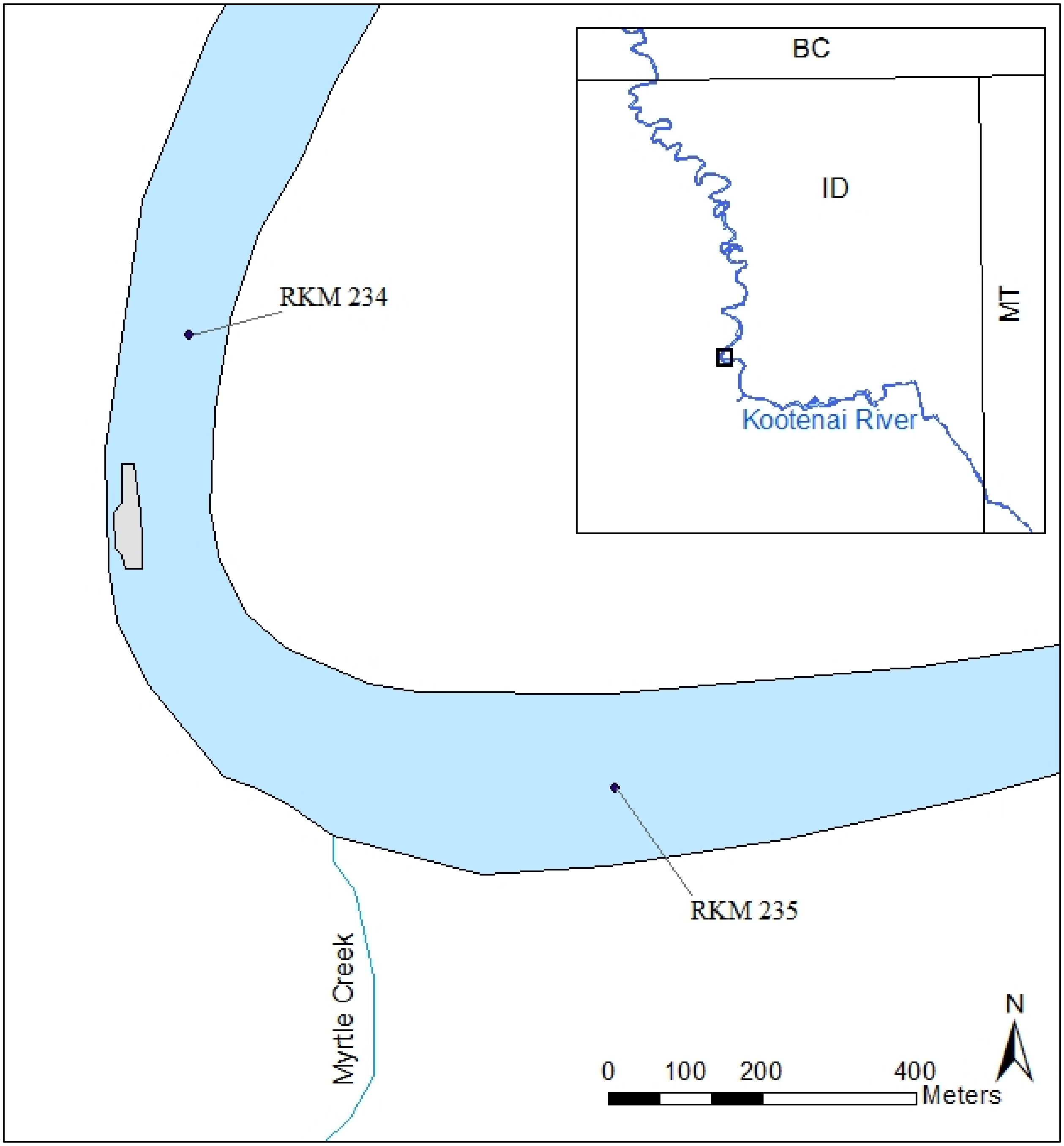

Figure 1.5. Location of Spawning Enhancement Pilot Projects (SEPP), at the Myrtle Creek Site (rkm 234.5). Grey area represents the location and approximate site of the SEPP constructed in 2014.

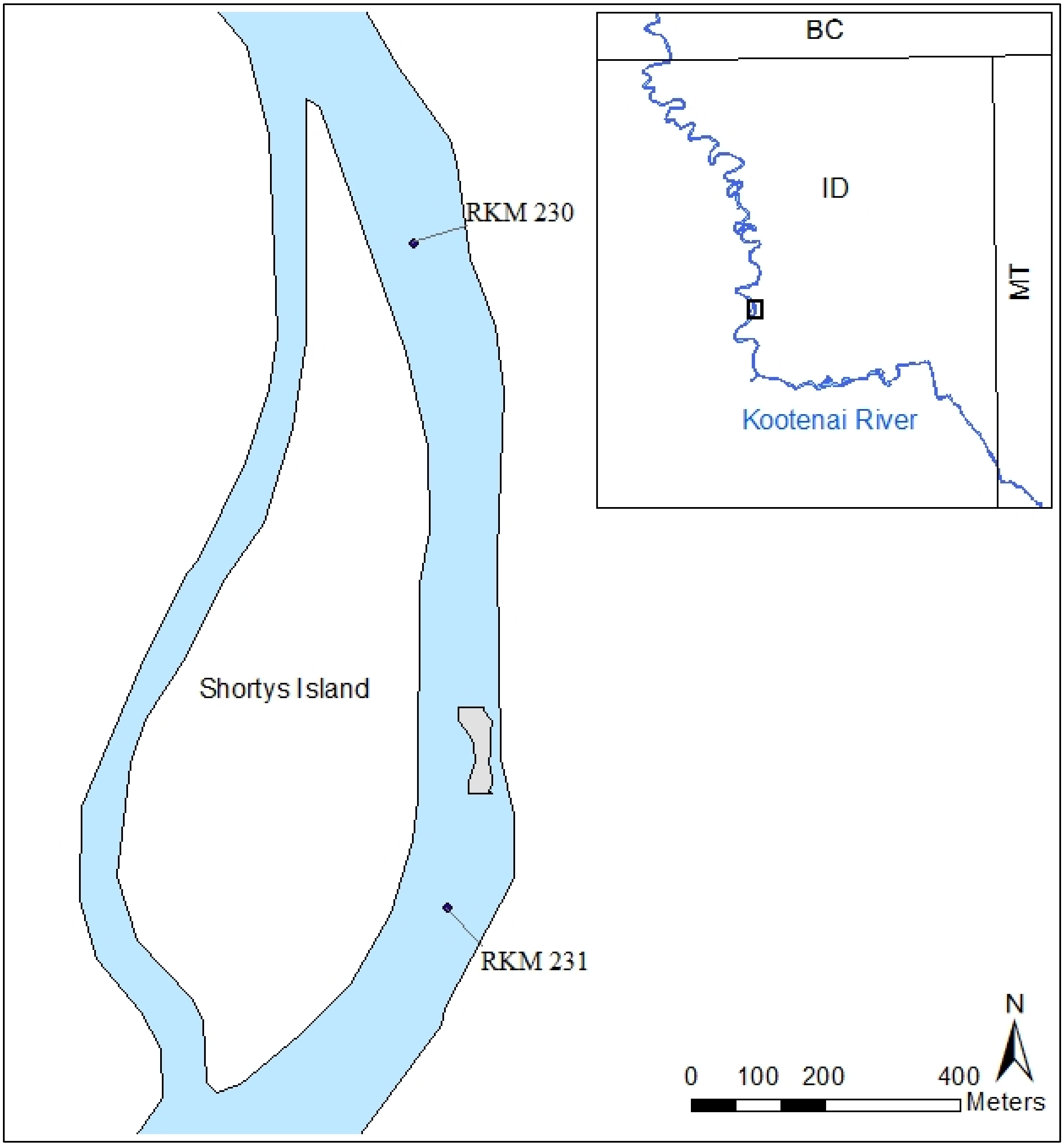

Figure 1.6. Location of Spawning Enhancement Pilot Projects (SEPP), at the Shorty's Island Site (rkm 230.5). Grey area represents the location and approximate site of the SEPP constructed in 2014.

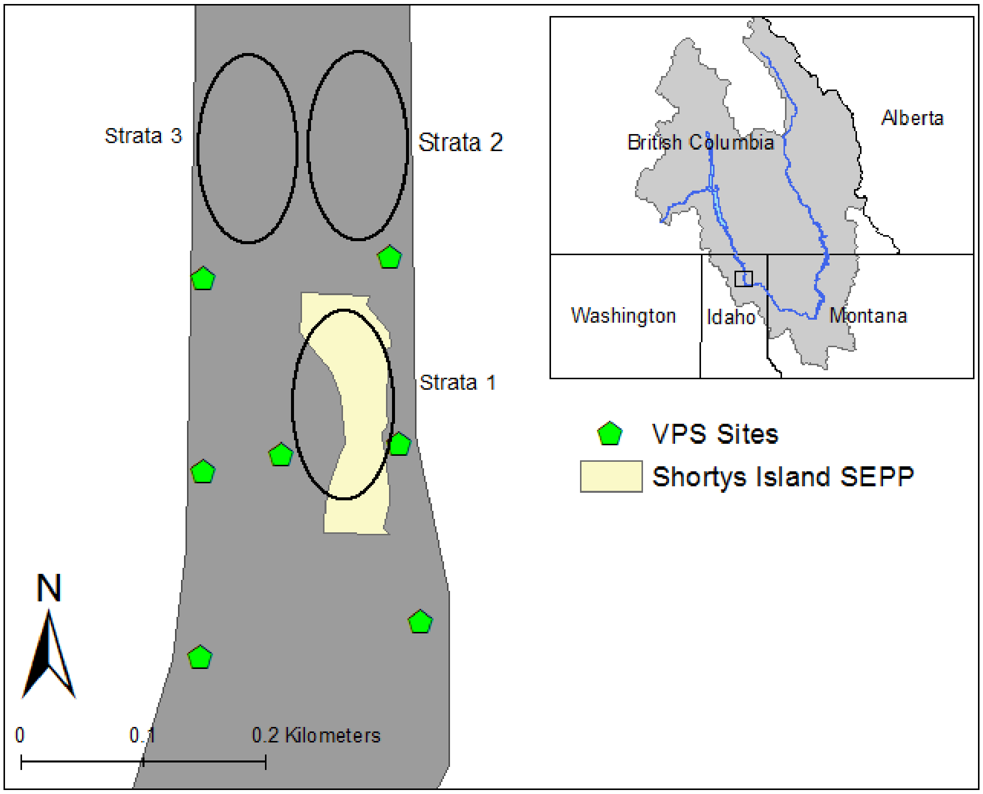

Figure 1.7. Substrate mat sampling design with reference to Spawning Enhancement Pilot Project (SEPP) area at Shorty's Island (rkm 231), Kootenai River, ID. The ovals (strata) depict the different areas where artificial substrate (egg mat) sampling occurred. Strata 1 and 2 were both areas with similar substrates and documented spawning occurrence prior to SEPP construction. Strata 3 was sand substrates dissimilar to that of 1 and 2. Green dots on periphery denote location and arrangement of the Vemco VR2Ws for the VPS study.

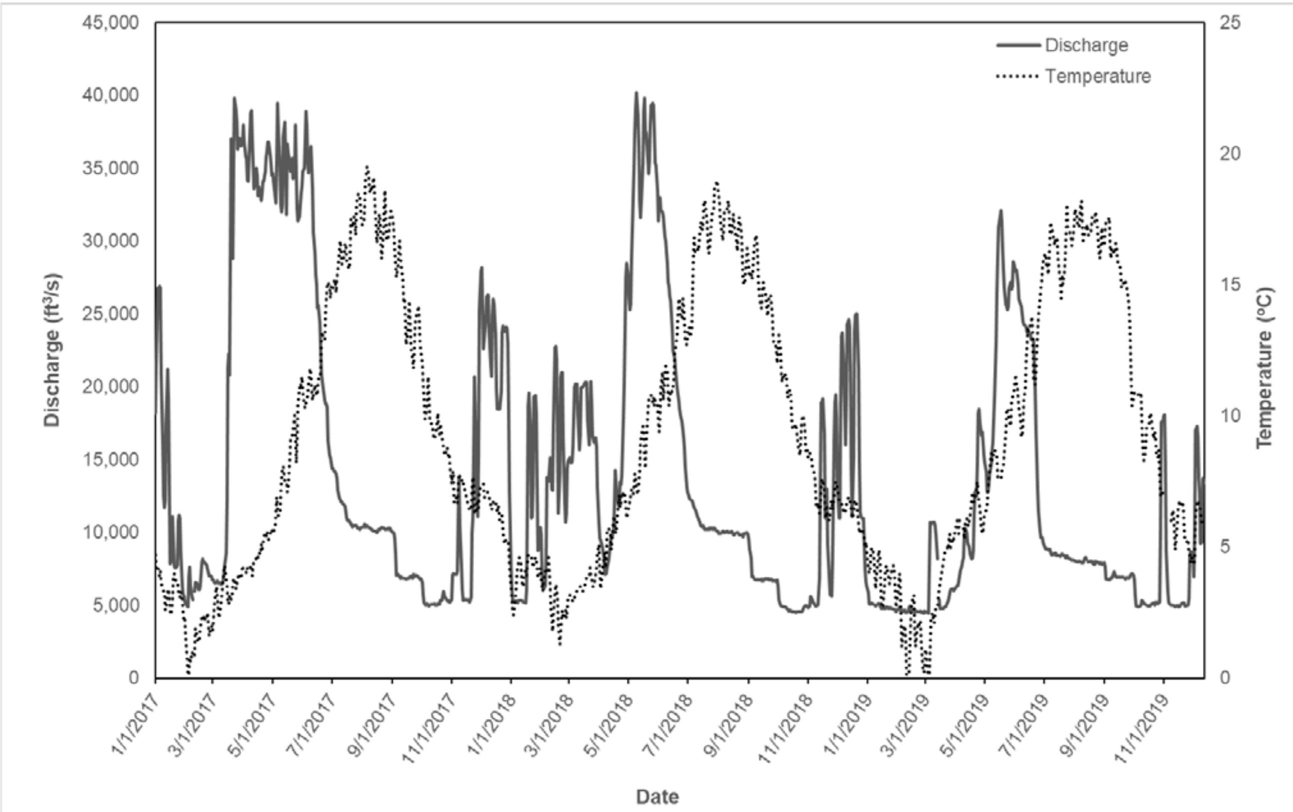

Figure 1.8. Mean daily discharge (ft^3/s) and temperature (°C) of the Kootenai River, ID for January 1, 2017 – December 11, 2019. Data retrieved from USGS gauge stations 12310100 and 12309500.

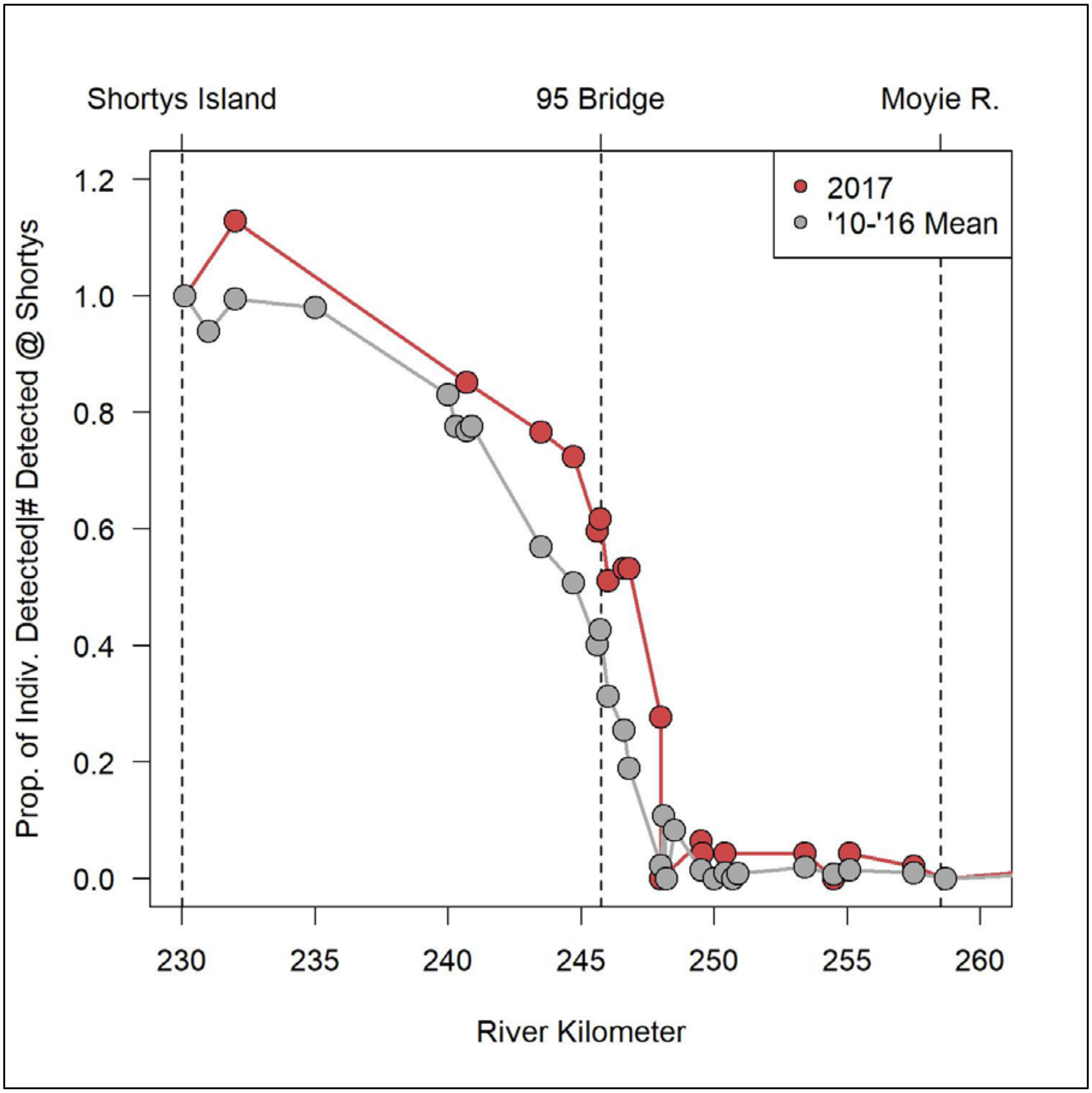

Figure 1.9. Proportion of individually unique detections of adult Kootenai Sturgeon tagged with acoustic transmitters by river kilometer.

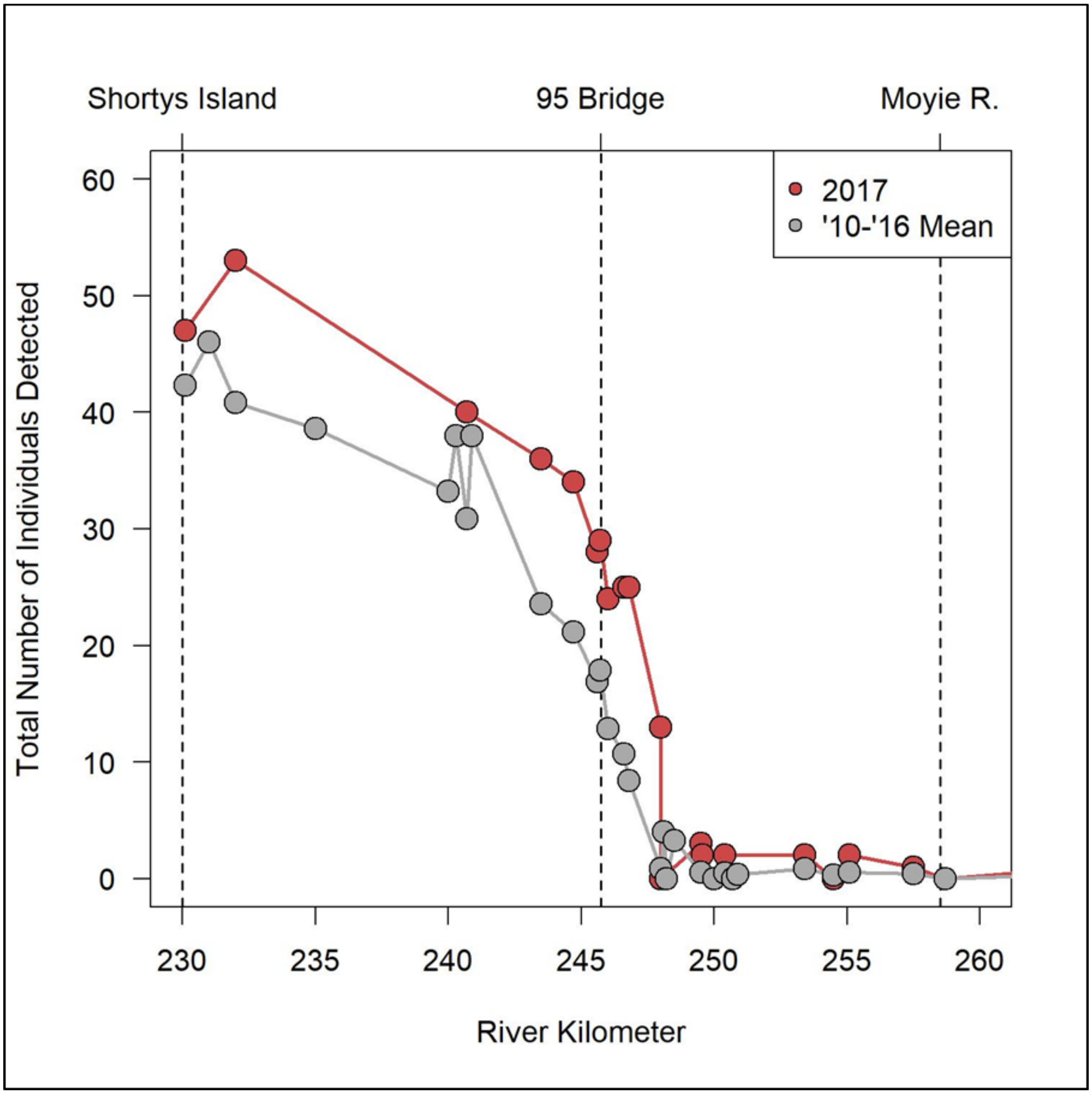

Figure 1.10. Number of individually unique detections of adult Kootenai Sturgeon tagged with acoustic transmitters by river kilometer.

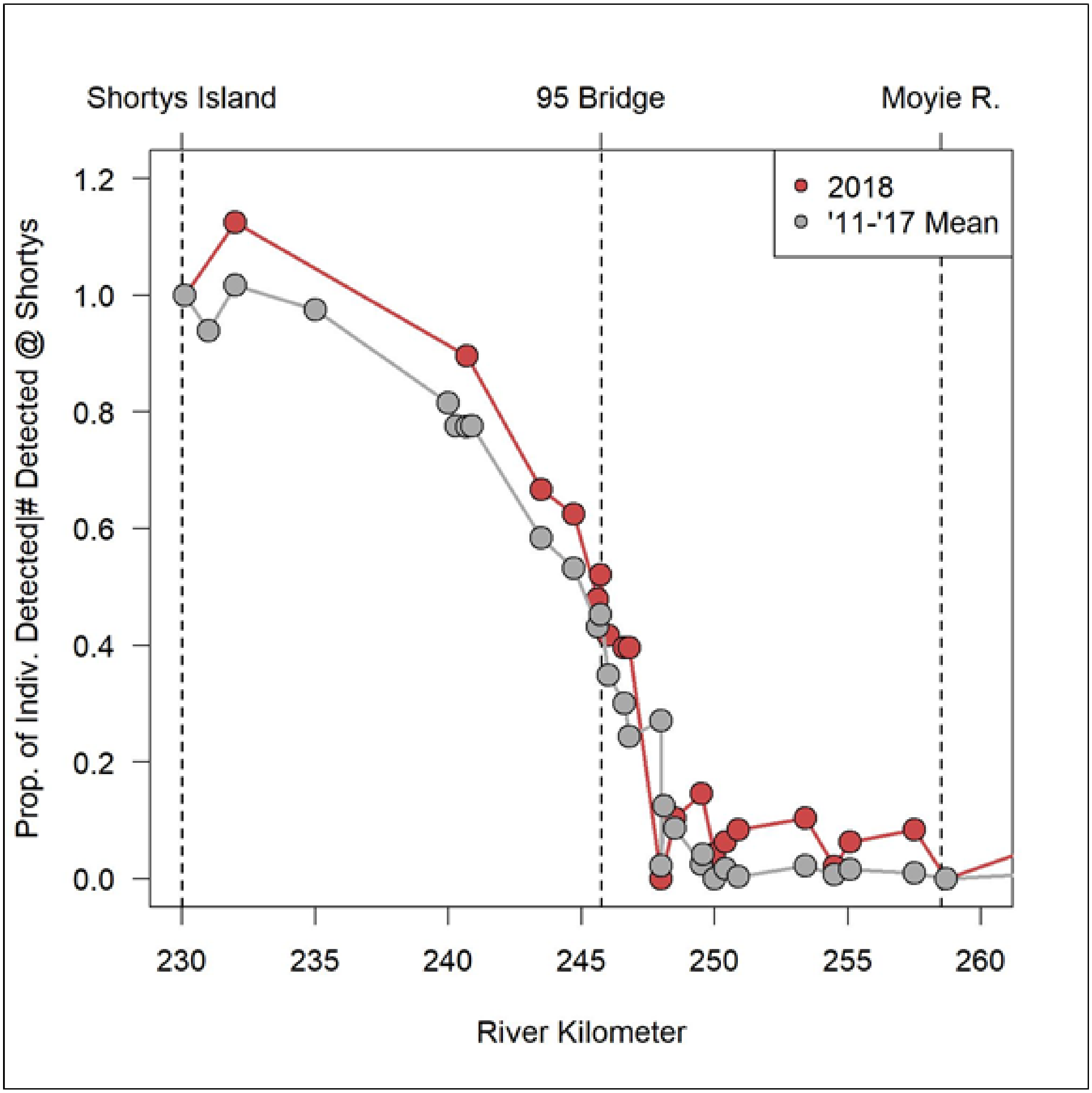

Figure 1.11. Proportion of individually unique detections of adult Kootenai Sturgeon tagged with acoustic transmitters by river kilometer.

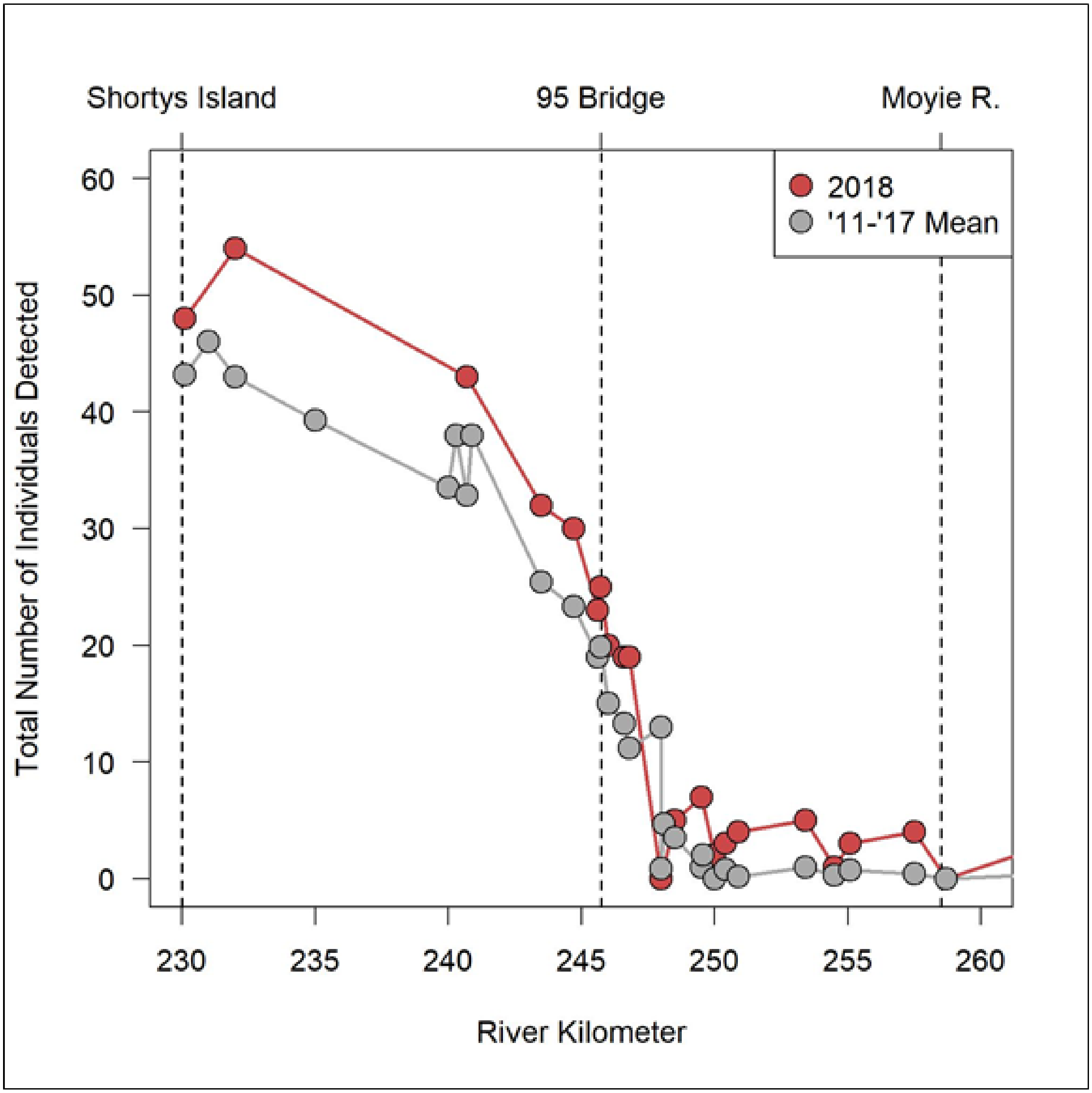

Figure 1.12. Proportion of individually unique detections of adult Kootenai Sturgeon tagged with acoustic transmitters by river kilometer.

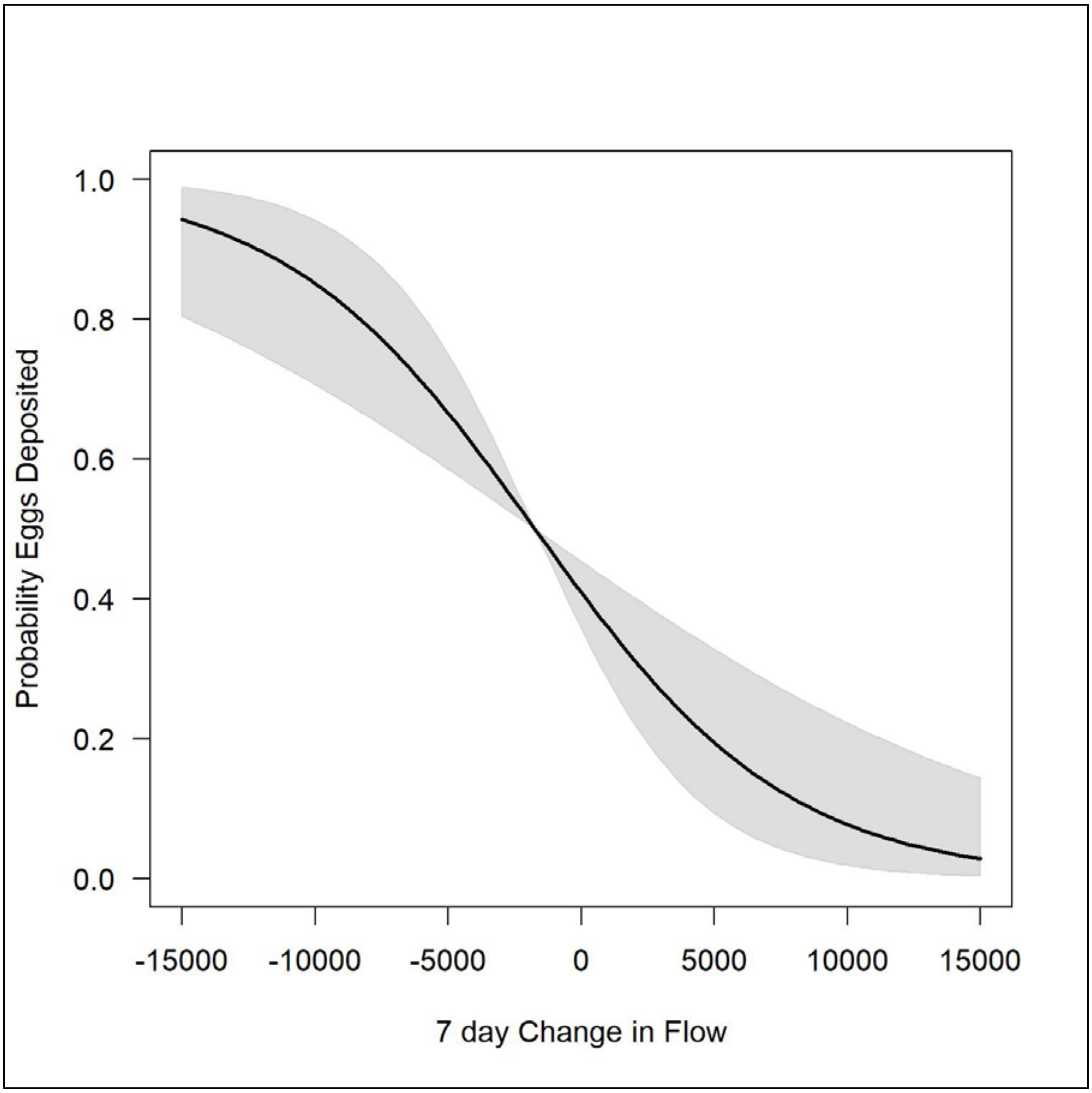

Figure 1.13. The predicted influence of the 7-day change in flow covariate on the probability that eggs are deposited (ψ) for Kootenai River White Sturgeon is shown above. This curve is predicted for an egg mat at Myrtle Creek on Strata 1. The line represents mean posterior values and the grey area represents the 90% HDP credible interval for the prediction.

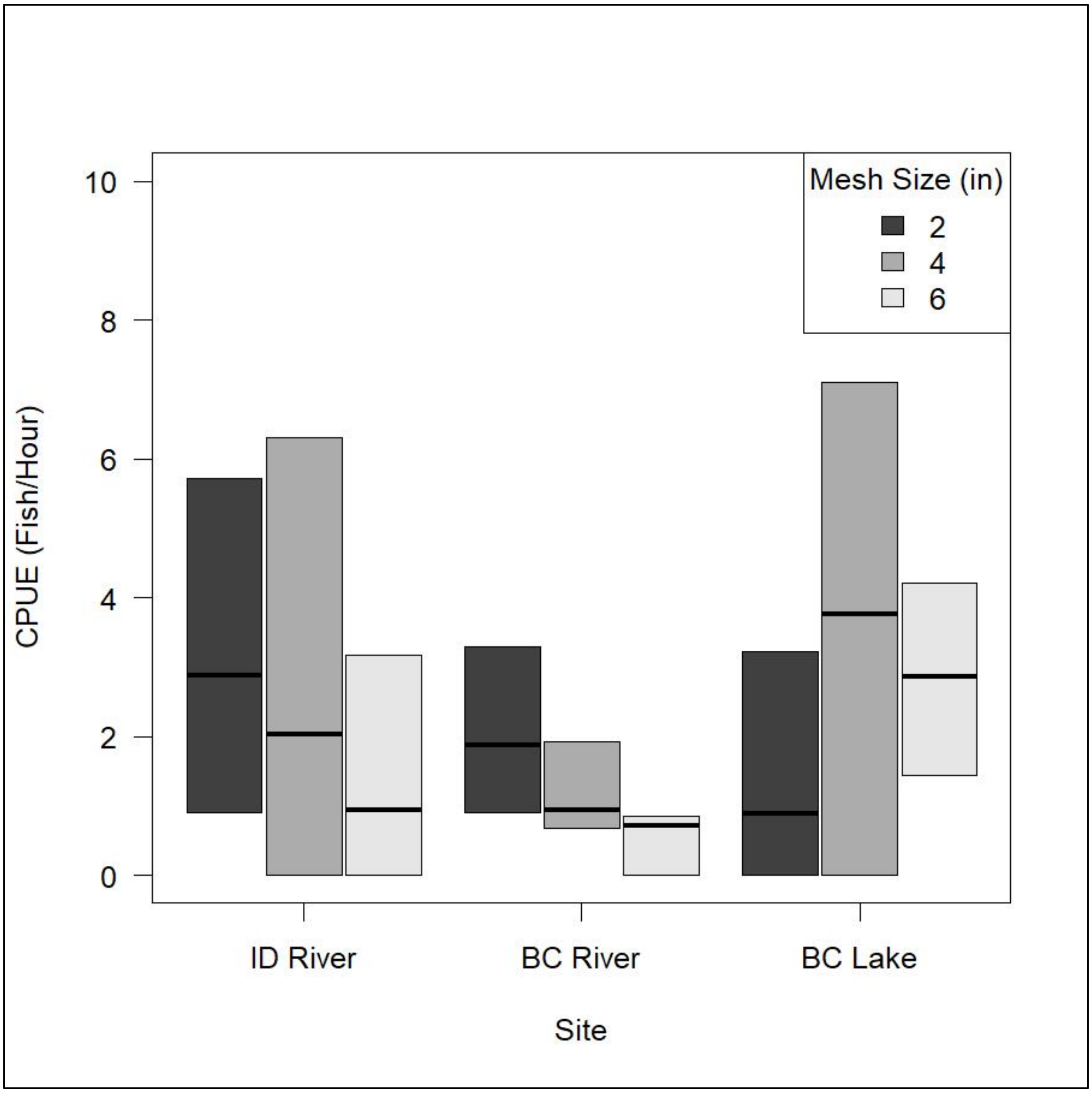

Figure 1.14. Box plots that show the relationship between CPUE (Fish/hour) and gill net mesh size in different parts of Kootenay Lake and Kootenai River for the 2017 juvenile sampling season. The boxes represent the interquartile range (25th - 75th percentile) and the line is the median.

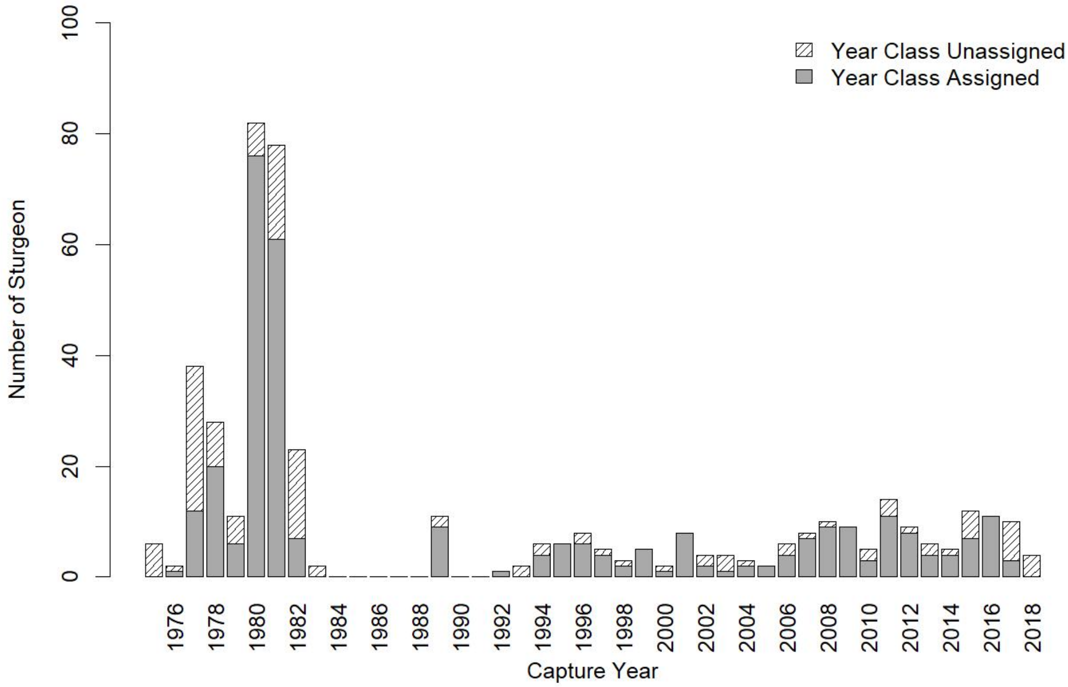

Figure 1.15. Number of first encounters (no recaptures) of wild juvenile Kootenai Sturgeon captured annually in the Kootenai River, ID 1977-2018. Age class was determined by sectioned pectoral fins, however not all fish were assigned a year class in every year.

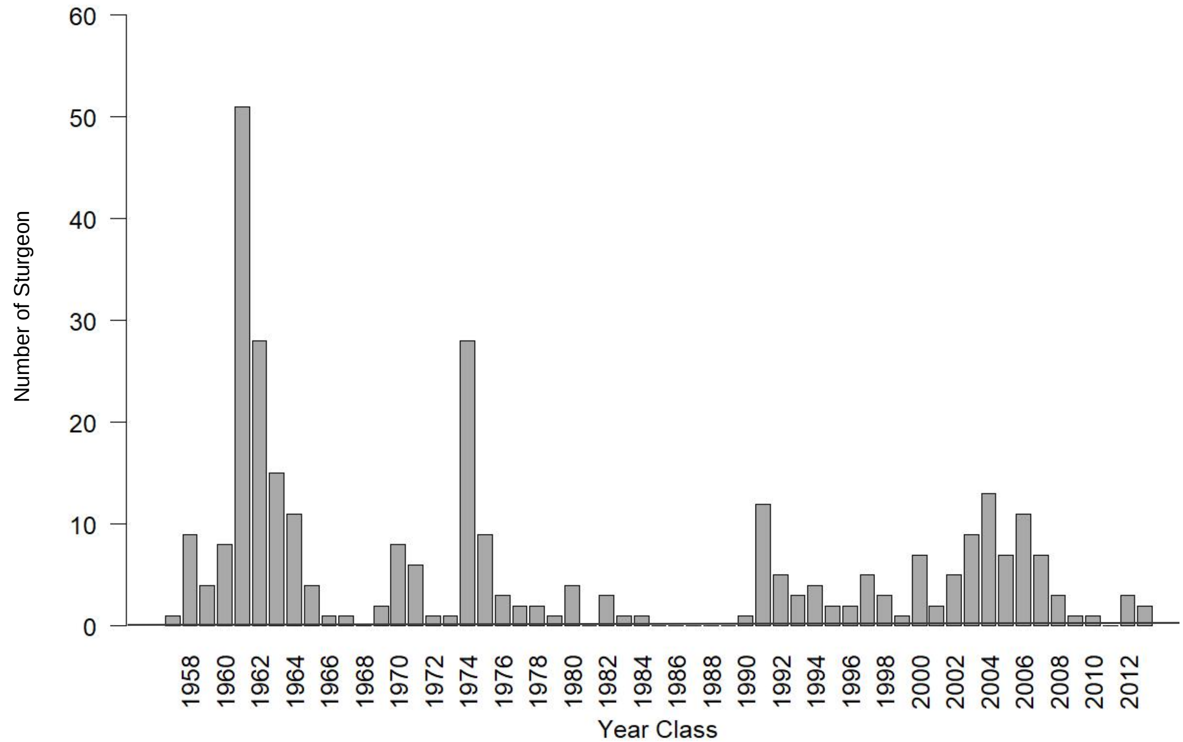

Figure 1.16. Number of wild juvenile Kootenai Sturgeon by age class captured in the Kootenai River, ID 1977-2018. Age class was determined by sectioned pectoral fins.

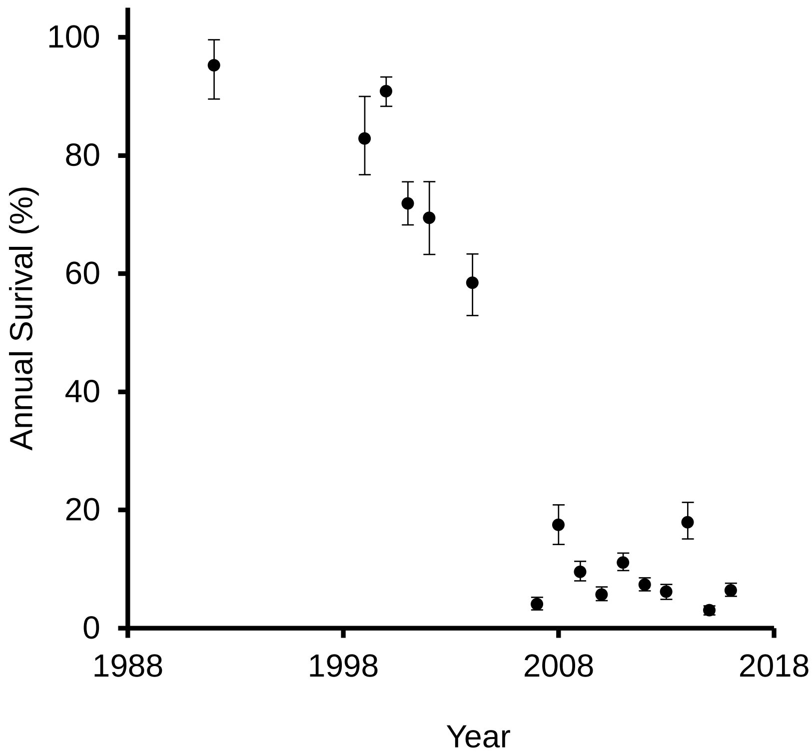

Figure 1.17. Estimated age-1 survival for juvenile hatchery Kootenai River White Sturgeon sampled in the Kootenai River and Kootenay Lake, BC from 1992-2016.

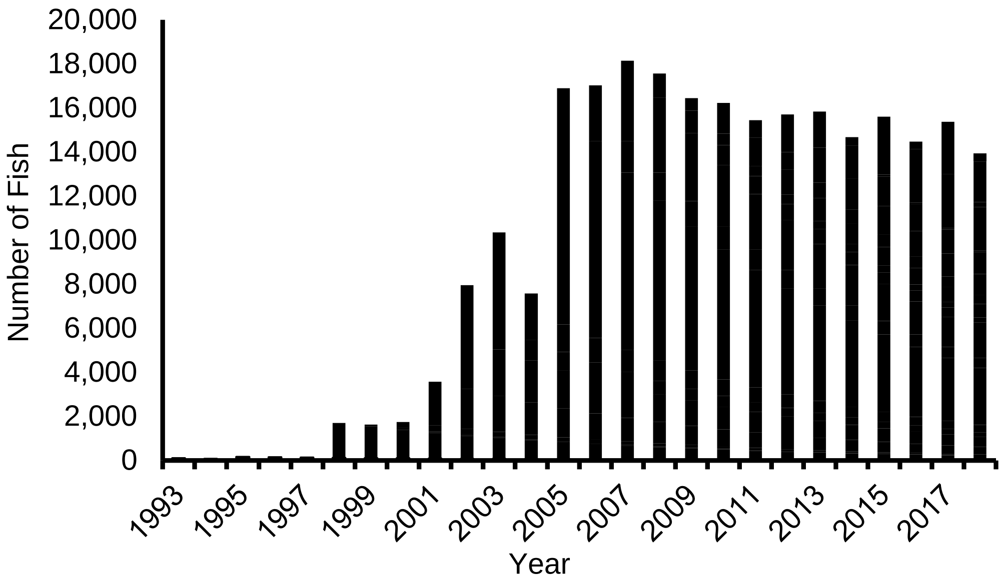

Figure 1.18. Estimated abundance of juvenile hatchery Kootenai River White Sturgeon in the Kootenai River Basin from 1993-2018.

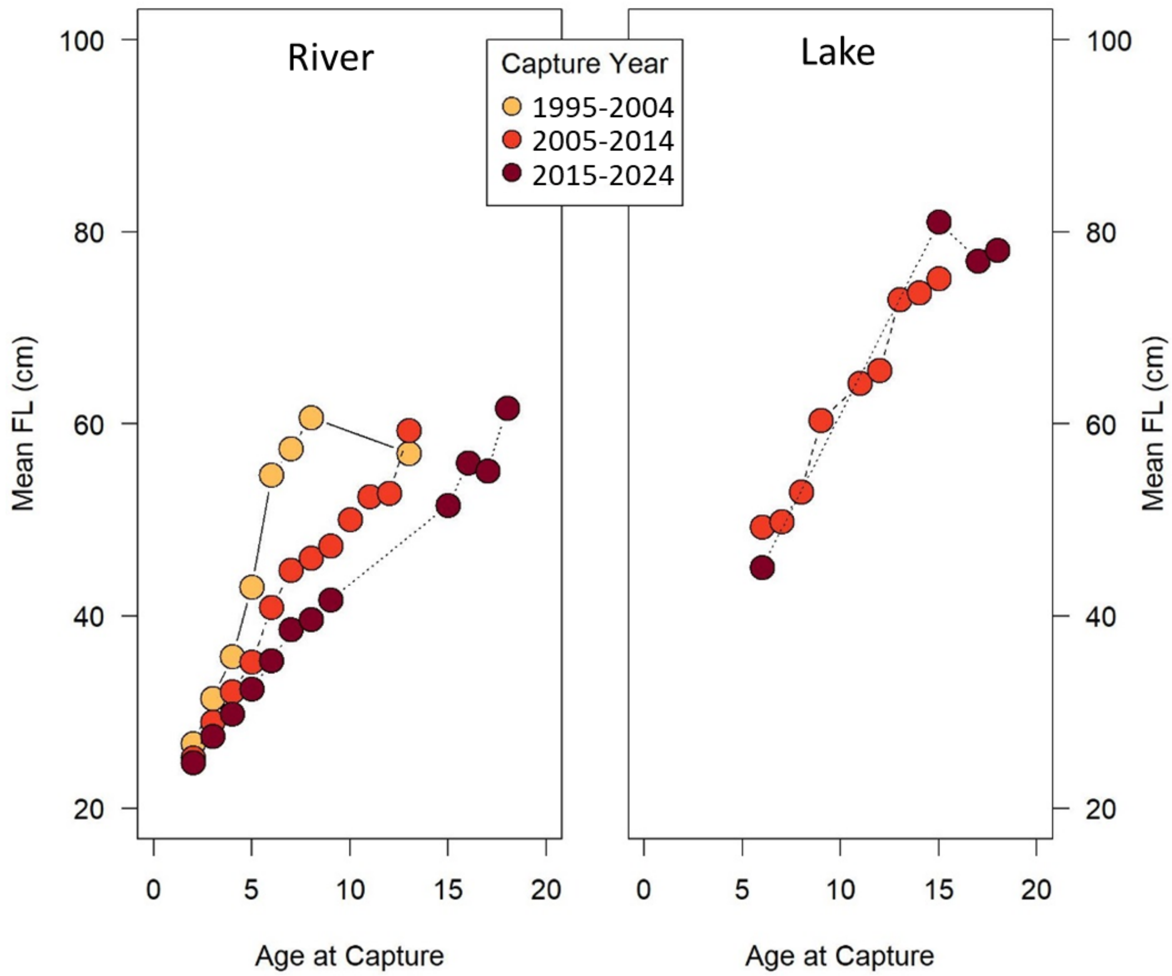

Figure 1.19. Fork length of hatchery Kootenai River White Sturgeon recaptured in the Kootenai River from 1995-2019. The figures represent fish reared on ambient river water and released as age-1 into the Kootenai River basin. Recapture information is represented as those captured in Kootenay Lake and those recaptured in the Kootenai River.

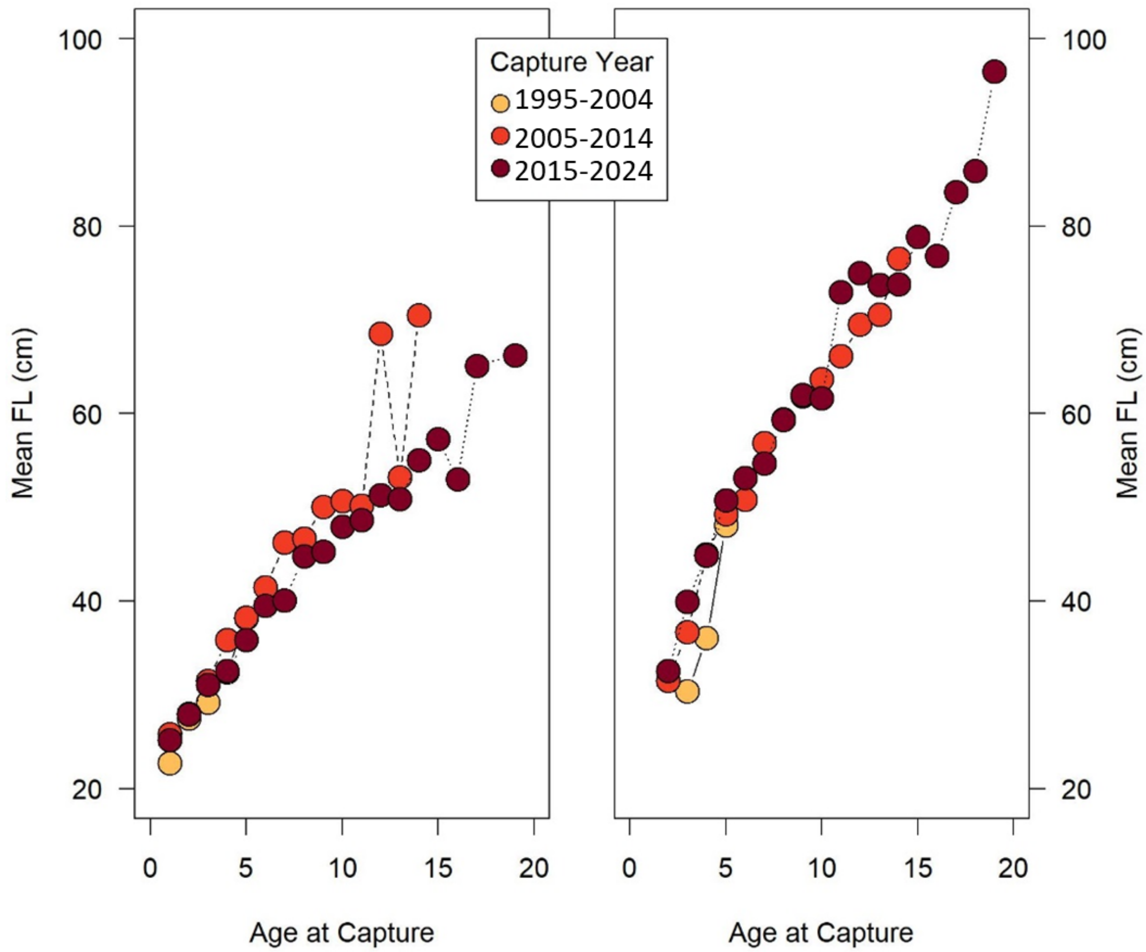

Figure 1.20. Fork length of hatchery Kootenai River White Sturgeon recaptured in the Kootenai River from 1995-2019. The figures represent fish reared on accelerated river water and released as age-1 into the Kootenai River basin. Recapture information is represented as those recaptured in the Kootenai River (left panel) and those recaptured in Kootenay Lake (right panel).

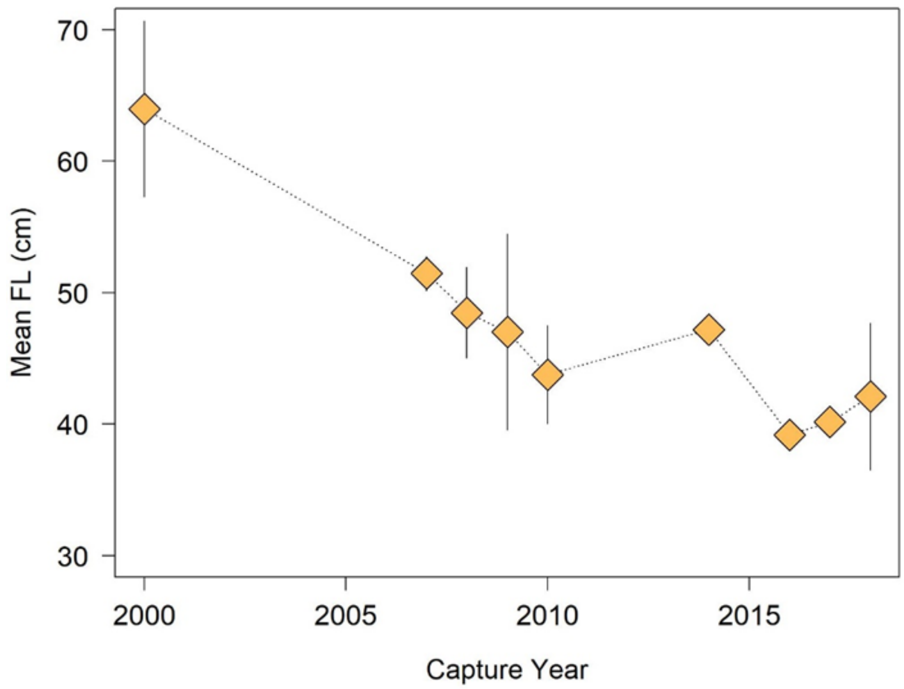

Figure 1.21. Fork length of age-9 hatchery Kootenai River White Sturgeon recaptured in the Kootenai River from 2000-2018. The figures represent fish reared on ambient river water and released as age-1 into the Kootenai River basin.

CHAPTER 2: EVALUATING THE UPSTREAM EXTENT OF SPAWNING MIGRATIONS OF KOOTENAI RIVER WHITE STURGEON

ABSTRACT

Dam construction is responsible for significant losses of suitable spawning and rearing habitats in a number of populations of White Sturgeon. Dam operations alter the natural hydrology of rivers and often degrade suitable spawning habitat. We evaluated whether environmental factors influenced spawning migration behavior of a population of Kootenai River White Sturgeon *Acipenser transmontanus*. We monitored the spawning migration of 214 adult spawning sturgeon from 2005 to 2017 using an array of passive acoustic receivers. Logistic regression and model selection indicated that the duration of high flow events and the presence of newly created, upstream habitat increased the proportion of spawners that migrated into high quality spawning habitat. The reported findings were used to make recommendations for future flow management of the Kootenai River that would benefit Kootenai River White Sturgeon spawning.

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INTRODUCTION

White Sturgeon abundance has declined across their historic range during the last 100 years (Pikitch et al. 2005; Hildebrand et al. 2016). White Sturgeon are currently distributed across Western North America, primarily in large river systems such as the Fraser, Columbia, Sacramento, and San Joaquin rivers. These systems historically provided complex, connected, cold water habitats that White Sturgeon depended on. However, anthropogenic activities such as agriculture, urbanization and dam construction have left such habitats fragmented and degraded (Duke et al. 1999; Hildebrand et al. 1999). Overexploitation has been another large contributing factor to the decline of White Sturgeon in several systems. Like other long-lived, late-maturing animals, sturgeon populations are especially vulnerable to over exploitation (Pikitch et al. 2005). For instance, female White Sturgeon are not sexually mature until they are nearly 25 years old, which means even small increases in mortality can have large impacts on juvenile recruitment rates.

Dam construction in particular is responsible for significant losses of suitable spawning and rearing habitats in a number of populations of White Sturgeon. For example, in the Columbia River basin, there are more than 20 dams, many of which have directly impacted connectivity and habitat (Parsley and Beckman 1994; Jager et al. 2001). Generally, dams disrupt natural temperature and flow regimes (Bunn and Arthington 2002; Allan and Castillo 2007), which provide important environmental cues for spawning White Sturgeon (Hildebrand et al. 2016). Gravel recruitment, and other alluvial processes, are known to decrease after the construction of a dam (Bunte 2004; Allan and Castillo 2007). White Sturgeon tend to spawn in areas with high flows and small to medium-sized spawning gravels to increase egg aeration and survival (Boucher et al. 2014). A lack of these habitat characteristics may lead to large increases in egg and larval mortality. For these reasons, dams represent one of the largest threats to White Sturgeon spawning and juvenile recruitment.

The Kootenai River is an example of a system that was drastically altered after the installation of a dam. The Kootenai River is an upper tributary of the Columbia River. It originates in British Columbia and flows south into Montana and Idaho before heading back north into British Columbia where it flows into Kootenay Lake. Libby Dam was completed on the Kootenai River in 1972 just above Kootenai Falls, MT. The primary purpose of the dam was to generate electricity and provide flood control for downstream Bonner's Ferry, ID. The result of the dam was a flattening out of the historic hydrograph, with much lower flows in the springtime and higher flows during the winter. In fact, peak springtime flows became roughly half of what they had been before the dam. In addition to the dam, there is an extensive dike system downstream to prevent flooding of the surrounding agricultural lands. The altered hydrograph has led to a number of physical changes downstream of the dam. The result has been a river that has become highly channelized and disconnected from its historic floodplain.

Since the installation of Libby Dam, the population of Kootenai River White Sturgeon (hereafter Kootenai Sturgeon) has declined. Although White Sturgeon are typically an anadromous species, the Kootenai Sturgeon population has been isolated from the Columbia River (via Bonnington Falls in British Columbia) for the last 10,000 years (Northcote 1973). Historic abundances are estimated to be in excess of 8,000 adults; however, current estimates are between 1,000 and 2,000 adults (Beamesderfer et al. 2014). Habitat degradation combined with overharvest were the primary drivers of the population decline (Paragamian and Wakkinen 2002a; Paragamian et al. 2005). By 1989, the fishery (even catch-and-release) was closed in British Columbia, Montana, and Idaho waters. The population decline caused the United States to list the population as endangered under the Endangered Species Act in 1994 (USFWS 1994) and for Canada to also list the population as endangered under the Species at Risk Act in 2006 (Fisheries and Oceans Canada 2014). Although spawning and eggs are routinely observed in the river, no significant wild recruitment has been observed since 1978 (Paragamian et al. 2002; Ross et al. 2015). Evidence suggests there is a bottleneck limiting juvenile recruitment; however, the exact mechanism limiting recruitment remains unknown.

The altered habitat and flow regime due to Libby Dam has caused a decline in the observed extent of spawning migrations of the White Sturgeon population. Adult White Sturgeon in the Kootenai River typically reside in Kootenay Lake and the lower portion of the Kootenai River outside of the spawning season, which takes place between April and June (Paragamian et al. 2002). During the spawning season these fish migrate upstream to spawn in the riverine habitat. Before the dam, spawning adult sturgeon could be found as far upstream as Kootenai Falls, MT (roughly 190 river kilometers from Kootenay Lake), which is a natural fish barrier (Paragamian et al. 2001). The river from Bonner's Ferry upstream to Kootenai Falls tends to have a higher gradient and more suitable spawning gravels for egg incubation. Currently, the extent of observed spawning has been limited to areas below Bonner's Ferry (<125 kilometers from Kootenay Lake). The lower portion of the river is characterized by a lower gradient and a substrate that is dominated by clay and sand (Fosness and Williams 2009; Barton et al. 2010). It remains unknown exactly what is causing spawning adults to shorten their spawning migrations and spawn over suboptimal substrates. Facilitating upstream migration (above rkm 245) could potentially allow Kootenai Sturgeon to spawn in more suitable habitats that would also provide more effective rearing habitats for newly hatched fish. Given the lack of any large-scale documented recruitment, any flow management plan that may increase upstream spawner movement and/or recruitment should be considered. However, evaluations of how flow shape and duration can impact Kootenai Sturgeon need to be conducted in a rigorous hypothesis testing framework rather than in a loose ad-hoc manner. Objectives, management alternatives, and response variables need to be fully defined in order for a comprehensive evaluation to be possible.

OBJECTIVES

In this evaluation, we examined which environmental factors influence spawning White Sturgeon to migrate from the slow-moving lower river to the higher gradient upper river during the spawning season (using Bonner's Ferry as a threshold cutoff between the lower and upper river). The ultimate objective of this evaluation was to provide proscriptive recommendations for future flow and habitat management actions in the Kootenai River that would maximize the likelihood of spawning adults moving above Bonner's Ferry. Facilitating further upstream spawning migration would increase the likelihood of sturgeon spawning over suitable substrate, thereby increasing egg-to-juvenile survival.

STUDY AREA

The Kootenai River watershed covers more than 50,000 km^2 in British Columbia, Montana, and Idaho. It originates in British Columbia in Kootenay National Park from which it flows south into Montana, then Idaho before heading back north where it empties into Kootenay Lake, British Columbia. The Kootenai River runs through a variety of habitats from isolated, high gradient alpine headwaters to agricultural, low gradient reaches of the lower river. Our study was limited to the lowest portions of the Kootenai River below Kootenai Falls, which is located just downstream of Libby, MT (Figure 2.1). Kootenai Falls is a natural fish barrier that prevents upstream movement of all species. We divided the study portion of the river into three reaches: the canyon reach, the braided reach, and the meander reach. The canyon reach is a high gradient portion of the river where the substrate is dominated by cobble and gravel. The braided reach is the transition zone where the river shifts from high to low gradient and includes several braided stream channel features. It is characterized by smaller gravels and more complex habitat relative to the canyon reach. Lastly, the meander reach is the lowest portion of the Kootenai before it enters Kootenay Lake, BC. The meander reach is characterized by a low gradient and meandering stream channel. This low-laying portion of the Kootenai River historically flooded with the spring freshet and created a large amount of floodplain habitat. Despite the substrate in the meander reach being dominated by sand and fine clay, this is where most of the current spawning activity occurs for Kootenai River White Sturgeon.

METHODS

Libby Dam Operations

Libby Dam forms Lake Koocanusa and was completed in 1972 just upstream of Libby, MT. The primary functions of Libby Dam are to provide flood control for downstream communities and hydropower generation. The installation of Libby Dam fundamentally changed the hydrodynamics of the lower Kootenai River. Dam operations have caused peak spring flows to be smaller in magnitude and shorter in duration. Additionally, springtime river temperatures were lower relative to pre-dam conditions due to dam releases. It has been hypothesized these changes in the flow and thermal regimes have interfered with sturgeon spawning migration and timing. Starting in 1994, the US Army Corps of Engineers (USACOE) working with IDFG, USFWS, and KTOI began a program to augment spring flows to enhance White Sturgeon spawning behaviors per a recommendation from the 2006 USFWS biological opinion report (USFWS 2006). One of the goals of the flow augmentation was to increase the proportion of spawners that migrated upstream above the meander reach into the suitable spawning habitats of the braided and canyon reaches. Each year the amount of water available for flow augmentation, known as the “sturgeon pulse,” was determined based on the inflow forecasts for Lake Koocanusa. The interagency working group then provided recommendations to USFWS and USACOE as to how to shape each springtime flow based on the available water forecast.

Adult Sampling and Tagging

Beginning in 2004 adult Kootenai Sturgeon were collected by angling and setlining (Paragamian et al. 1996) during the spring and fall in each year. The vast majority of adult sampling (by effort) occurs in the Idaho portion of the river during the spring sampling season (Ross et al. 2015). These areas are backwater habitats and have depths in excess of 20 m and low current velocities (<0.05 m/s) and incorporate spawning locations (near and above rkm 229). Fall sampling in 2017 occurred throughout the lower river (rkm 207.5-308) and into the Kootenay Lake in BC, at the Creston delta (rkm 118) and the Lardeau-delta at the north end of the lake (rkm 18). During sampling we attempted to sex and determine the stage of maturity of adult sturgeon following the gonadal biopsy protocol of Conte et al. (1988), and Van Eenennaam and Doroshov (1998). Fish for which sex could not be determined were excluded from this study to ensure we were monitoring sexually mature, potential spawners. Male and female Kootenai Sturgeon were tagged with Vemco V13 (five-year battery expectancy) and V16 (10-year battery expectancy) model sonic transmitters and released at their place of capture. The transmitters were programmed to transmit an individually coded signal randomly every 30 to 90 seconds. A total of 10–26 new tags were deployed each year. The majority of tags were implanted in females due to the difficulty in confidently sexing male sturgeon.

Acoustic Array

We monitored seasonal spawning movements of Kootenai Sturgeon throughout the Kootenai River/Kootenay Lake system using a passive telemetry array. Beginning in 2003, we deployed and maintained an array of 89 Vemco model VR2 and VR2W sonic receivers located from rkm 18, near the mouth of the Duncan River in Kootenay Lake, BC, upstream to rkm 306, below Kootenai Falls (Figure 2.1). From this array, we are able to analyze occupancy (presence/absence) as well as individual movements in different reaches throughout Kootenay Lake and the Kootenai River. Receiver locations were nearly constant through the study; however, some movement was inevitable due to changing river conditions as the study progressed. In order to avoid redundant data collection, receivers were located in areas where fish pass through but do not usually hold for long periods. Most sites were below river bends or along straight reaches to allow for maximum signal reception but were reasonably free of drifting debris and at low risk of potential vandalism/theft. We tethered each receiver to an anchored float that was chained to the riverbank in order to keep the hydrophone off the substrate (Neufeld and Rust 2009). Due to the care taken during the placement of the hydrophones, we assumed detection was constant among receivers within any given time period. We downloaded the movement information from the receivers twice a year, once in late winter, prior to the spawning season, and again in the fall.

Analysis

We used logistic regression to estimate the annual probability that an individual spawning White Sturgeon would migrate from the meander reach into the braided reach at least once during the springtime spawning season (March-August) in 2005-2017. Individual fish movement data were compiled from the passive acoustic array to determine the proportion of spawners that made the migration each year. For the analysis we assumed any tagged adult that was present at or above Shorty's Island (rkm 230.5) during the spawning season to be a spawner. During the rest of the year no adult fish were found in this portion of the river. We evaluated five environmental covariates for their effect on the proportion of the spawners that would migrate into the braided reach in a given spawn year *j*:

- $Days30k_j$ – The number of days where flow at Bonners Ferry (rkm 246) was greater than 30k cfs during the spawning season (April-July). This metric represented the duration of high flows during the spawning season (Figure 2.2).

- $Flowpeak_j$ – The number of distinct peaks in the hydrograph during the spawning season. In some years the sturgeon pulse was manipulated to test the hypothesis that multiple, distinct peaks may increase migratory behavior.

- $Templag_j$ – The number of days between when water temperature first hits 8°C and the peak in the hydrograph (Figure 2.3). The temperature 8°C is a threshold for spawning for Kootenai River White Sturgeon (Paragamian and Wakkinen 2002b). The covariate connects the timing of stream temperatures and flow. Values of the covariate could be negative which indicated that the hydrograph peak occurred before stream temperature rose to 8°C.

- $Habitatproject_j$ – A binomial indicator variable to specify whether or not habitat improvement projects in the braided reach were completed. The projects were part of an effort by the Kootenai Tribe of Idaho and Bonneville Power to restore lost habitat as mitigation for the effects of Libby Dam. One of the goals, among several others, of the braided reach projects was to provide holding habitat for spawning adult sturgeon to encourage upstream migration. The restoration projects included excavating deep pools, building in-channel islands, and creating side channel habitats (Kootenai Tribe of Idaho 2010, 2012). The habitat improvements occurred between rkms 248--250 and were completed beginning in 2015.

- *Density$_j$* – The total number of tagged spawners present in year *j*.

Each model covariate represented a different independent hypothesis of what might be influencing White Sturgeon spawning movements and the probability that White Sturgeon would migrate into the braided reach (Table 2.1). Kootenai River White Sturgeon do not spawn every year, so we also incorporated a year based random effect to explain any differences in spawning movements that existed among spawning cohorts.

Model fitting was in a Bayesian framework using STAN (Stan Development Team 2018) in the R package *brms* (Bürkner 2018, R Core Development Team 2018). A total of 4 chains, each with 10,000 iterations (5,000 burn-in iterations) was used to fit each model. We assumed completely uninform (i.e. "flat") priors during all analyses. Model convergence was checked by evaluating the posterior sample traceplots and by the Gelman-Rubin diagnostic statistic (Rhat). The mean and 90% highest posterior density (HDP) credible intervals were reported from joint posterior distribution for each estimated parameter (Box and Tiao 1992). All combinations of environmental covariates were examined (31 total models) and model fit was assessed using WAIC and leave-one-out cross-validation in the R package *loo* (Yao et al. 2017).

RESULTS

A total of 214 spawning White Sturgeon (47 males and 167 females) were detected in our acoustic array above rkm 230.5 during the spawning periods (May–July) in 2005–2017. Females in the study had a mean fork length of 184.8 cm and a mean weight of 56.9 kg. Males in the study had a mean fork length of 169.6 cm and a mean weight of 40 kg. Females made up the largest proportion, >50%, of tagged spawners within each year (Table 2.2). We detected an average of 88 spawners each year of the study with a maximum of 116 in 2017 and 2015 and a minimum of 17 in 2005. The annual proportion of total spawners that migrated into the braided reach (>rkm 246) varied between 0% and 46% with a mean of 28% (Figure 2.4).

The results of the model selection process provided evidence that the duration of high flow events and the presence of the habitation projects in the braided reach best described the annual proportion of spawners that migrated into the braided reach. The model selection process identified seven competitive models (ΔWAIC <2; Table 2.3). The most common covariates in the competitive models were the number of days of flow greater than 30k cfs (*Days30k$_j$*; 7 of 7 models) and the indicator variable for the completion of the habitat projects (*Habitatproject$_j$*; 5 of 7 models). The top model with the lowest WAIC score had two explanatory covariates: the number of days of flow above 30k cfs and the indicator variable for the completion of the braided reach habitat projects. The other competitive models included combinations of all five of the covariates.

The posterior estimates for the linear coefficient effect sizes for the top model were both positively related to the seasonal proportion of spawners that migrated into the braided reach (Figure 2.5, Table 2.4). The median value of the effect size for the 30k cfs flow covariate was 0.01345 with a 90% lower and upper HDP of 0.00718 and 0.0211, respectively. In other words, we would expect the proportion of spawners that migrate into the braided reach to increase by

2% for every 10 days of flows greater than 30k cfs. The posterior median of coefficient for the completion of the habitat projects was 0.648 with a 90% upper and lower HDP of 0.272 and 1.071, respectively, which would translate into an 11% increase in the proportion of spawners that would move into the braided reach in any given year with the completion of habitat projects.

DISCUSSION

We found that the number of days where flow was above 30k cfs was the best predictor of the probability of an individual sturgeon migrating into the braided reach. High springtime flows provide important cues for White Sturgeon to initiate migration and spawning (Paragamian and Wakkinen 2002a; Hildebrand et al. 2016). In our study we quantified the amount of high flow necessary to increase the probability of a White Sturgeon making a specific spawning migration (meander reach to braided reach). Based on our sonic telemetry, extended high flows improved upstream movement of adult sturgeon into better spawning habitat. Therefore, we recommend that Libby Dam operations should attempt to maximize the number of days with flow greater than 30k cfs during the spawning season (March-July). Additional evaluations of egg deposition (previous chapter) indicates that once adults are present on the spawning grounds, the more rapid the decline in flows appears to optimize spawning events. We acknowledge that the 30k cfs cutoff is somewhat arbitrary; however, it is merely meant to represent a benchmark for the duration of high flows. For instance, we would recommend a higher number of days with flows slightly below 30k cfs rather than a few days of flow above 30k cfs when there is a limited amount of water allocated for the sturgeon pulse. The preference should be given to the option that would maximize the duration of time of high flows rather than alternative options that would solely maximize the magnitude of flows. Lastly, we did not assess how flows greater than 30k cfs influence spawning movement probabilities due to their rarity. Flows greater than 40k cfs are typically avoided because they result in flooding in Bonners Ferry and throughout the meander reach.

In addition to flow, temperature can be an important cue during migration and spawning for adult White Sturgeon. In our analysis, there was little support for a temperature covariate during the model selection portion of the analysis. Our results suggested that temperature alone is not a major factor in initiating fine scale movements (e.g. moving from the meander reach to the braided reach). Despite this, temperature should remain a large consideration for future dam operations. In the Kootenai River, White Sturgeon tend to spawn when river temperatures are between 8 and 12°C (Paragamian and Wakkinen 2002a). There is also evidence that temperature acts as a cue for the staging behaviors prior to the spring spawning season in the Kootenai River. Although we recognize that temperature control out of Libby Dam can be extremely variable, future operations should continue to use selective withdrawal gates in the spring attempt to balance known temperature requirements with the flow recommendations outlined above.

Our modeling demonstrated that the completion of the habitat projects immediately upstream of Bonners Ferry resulted in an 11% increase in the probability of a spawner entering the braided reach. One of the goals of the habitat projects was to encourage White Sturgeon to spawn over the more suitable spawning habitat in the braided reach. Our data are unable to determine whether the fish that did move into the braided reach actively spawned there. The coarse resolution of the data collected from the VEMCO VR2W receivers is only able to identify broad movements and not specific, fine-scale, behaviors (e.g., spawning). Although IDFG does sample for spawning activity using artificial substrate mats, these efforts have generally been limited to known spawning locations in the meander reach. However, beginning in 2018 IDFG began sampling for egg deposition in the braided reach using artificial substrate mats to evaluate whether or not spawning was occurring in the braided reach.

The habitat alterations that have taken place in the Kootenai River since the installation of Libby Dam are numerous and influence every life stage of the White Sturgeon life cycle. It is unlikely that increasing the duration of high flows in the Kootenai River alone would cause recruitment to return to its pre-dam levels. Several survival bottlenecks exist even if spawning is successful. For instance, the loss of off channel rearing habitats and limiting nutrients have likely reduced the meander reach's ability to support larval and juvenile sturgeon. Encouraging further upstream migration of spawners needs to be part of a larger plan for the restoration of White Sturgeon in the Kootenai River.

Impounded and fragmented river systems are common for Columbia River basin White Sturgeon populations; however, only the Kootenai River population is listed as endangered. It is likely that there are several other factors limiting recruitment of Kootenai River White Sturgeon that may not be present in other Columbia River basin White Sturgeon populations. The changes in hydrology, stream habitat, nutrient levels, and the unique configuration between the Kootenai River and Kootenay Lake may all serve as pressures that limit population recovery. Through this study we have begun to understand what may influence more Kootenai River White Sturgeon to move into the braided reach; however, it remains poorly understood why they are spawning in the meander reach in the first place. It is likely a combination of the conditions unique to the Kootenai River that are driving this behavior. Future work should focus on addressing this knowledge gap.

TABLES

Table 2.1. Covariates used in logistic regression analysis to determine the proportion of Kootenai River White Sturgeon spawners that migrated into the braided reach each year.

Year	Total Number Spawners	Number Spawners Detected in Braided Reach	Days flow > 30k cfs *Days30k* $_j$	Lag (days) between 8C and Flow Peak *Templag* $_j$	Number of Flow Peaks *Flowpeak* $_j$	Habitat Project Completed *HabitatProject* $_j$	Adult Density *Density* $_j$
2005	16	0	2	19	3	0	16
2006	31	5	42	17	1	0	31
2007	28	7	33	13	2	0	28
2008	29	13	39	-11	2	0	29
2009	24	3	10	19	2	0	24
2010	36	9	16	31	1	0	36
2011	35	13	52	-6	1	0	35
2012	34	13	91	37	1	0	34
2013	43	12	50	-8	2	0	43
2014	45	13	65	-5	2	0	45
2015	49	14	6	17	1	1	49
2016	49	18	15	21	2	1	49
2017	54	25	73	-13	1	1	54

Table 2.2. Total acoustic tags deployed each year and the number of male and female spawners in each year.

Year	Total Number of Adult Tags Deployed	Number Detected Male Spawners	Number Detected Female Spawners
2005	17	4	12
2006	40	13	18
2007	66	11	17
2008	82	13	16
2009	89	12	12
2010	112	7	29
2011	106	8	27
2012	95	10	24
2013	89	7	36
2014	105	9	36
2015	116	9	40
2016	113	10	39
2017	116	7	47

Table 2.3. The top logistic models (ΔWAIC<2) and the covariates that were included in those models. There was the most support for models that included the $Days30_k$ and $HabitatProject_j$ covariates. All models also included a random intercept for year.

Potential Covariates						
$Days30k_j$	$Templag_j$	$Flowpeak_j$	$HabitatProject_j$	$Density_j$	*WAIC*	*ΔWAIC*
+	-	-	+	-	68.50313	0
+	-	-	-	+	68.98008	0.47695
+	+	-	+	-	69.59327	1.09014
+	-	-	+	+	69.87013	1.367
+	+	+	+	-	70.10271	1.59958
+	-	+	-	+	70.29439	1.79126
+	-	+	+	-	70.29958	1.79645

Table 2.4. Estimates from the posterior distribution of the top model, estimates are in logit terms.

Parameter	Descriptions	Posterior median	90% HPD	Rhat
$\beta_{Intercept}$	Intercept term	-1.618	(-2.052, -1.265)	1
$\beta_{Days30k}$	Linear coefficient for *Days30k* covariate	0.01345	(0.00718, 0.0211)	1
$\beta_{HabitatProject}$	Linear coefficient for *HabitatProject* covariate	0.6486	(0.272, 1.071)	1
σ_{year}	Standard deviation for *year* Random Effect	0.2507	(0.0512, 0.5855)	1

FIGURES

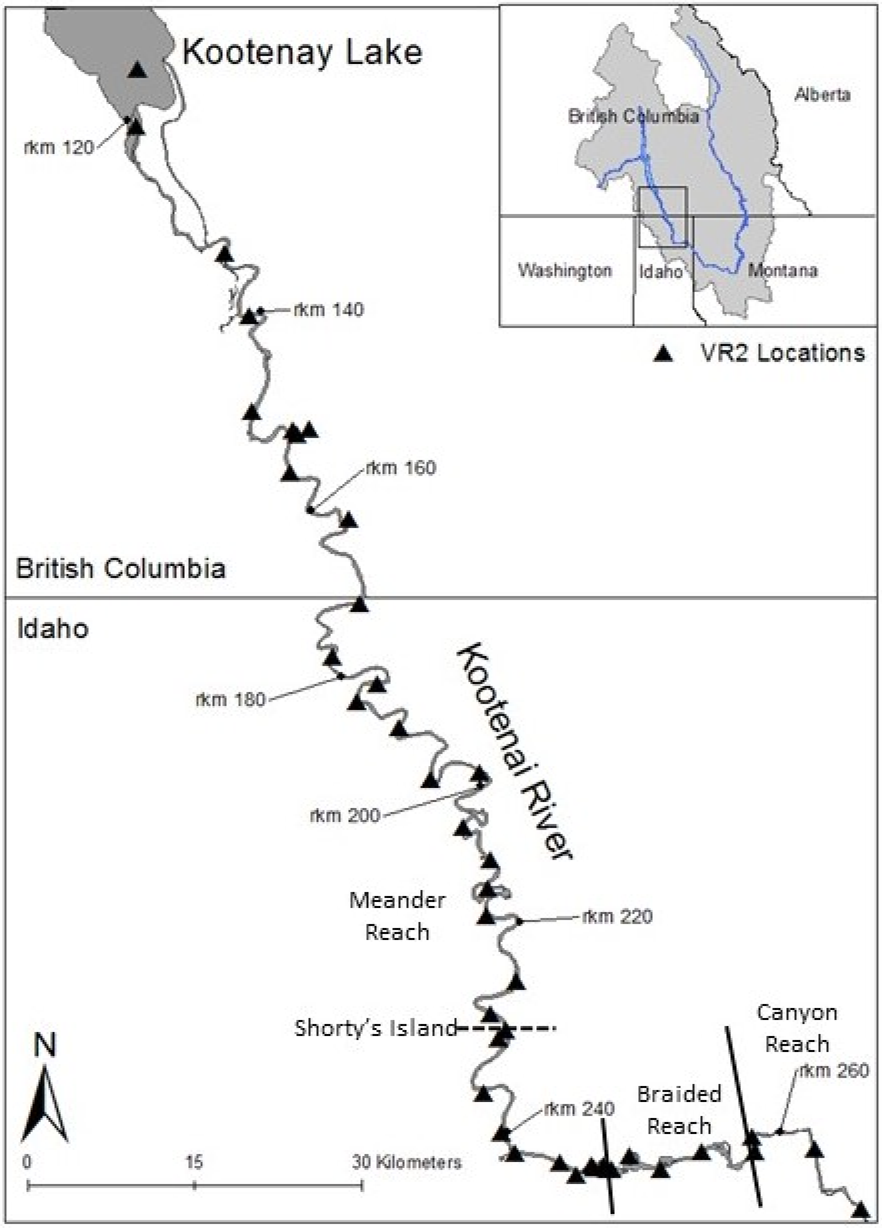

Figure 2.1. Map of Kootenai River basin and the lower Kootenai River. The triangles represent approximate locations of VEMCO VR2W acoustic receivers. The solid lines separate the river into the three primary reaches: the lower meander reach, the middle braided reach and the upper canyon reach. The dashed line indicates the location of Shorty's Island (rkm 230.5) which is the threshold for White Sturgeon spawning.

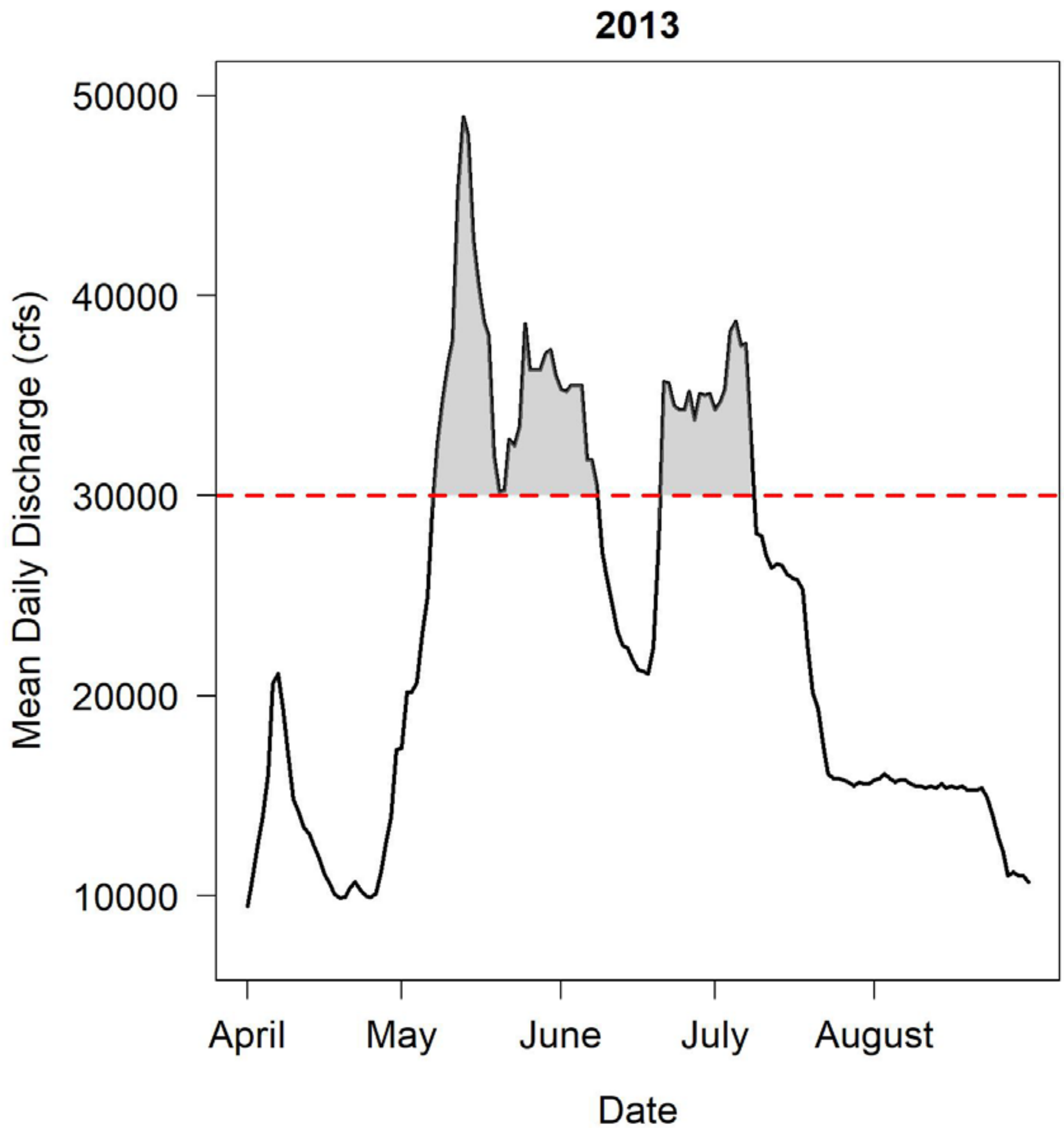

Figure 2.2. An example of how the *Days30k* metric is calculated on the hydrograph from 2013. The shaded area represents the period of time during the spawning season where flow was greater than 30k cfs at Bonners Ferry. This metric was calculated for each year of the study: 2005-2017.

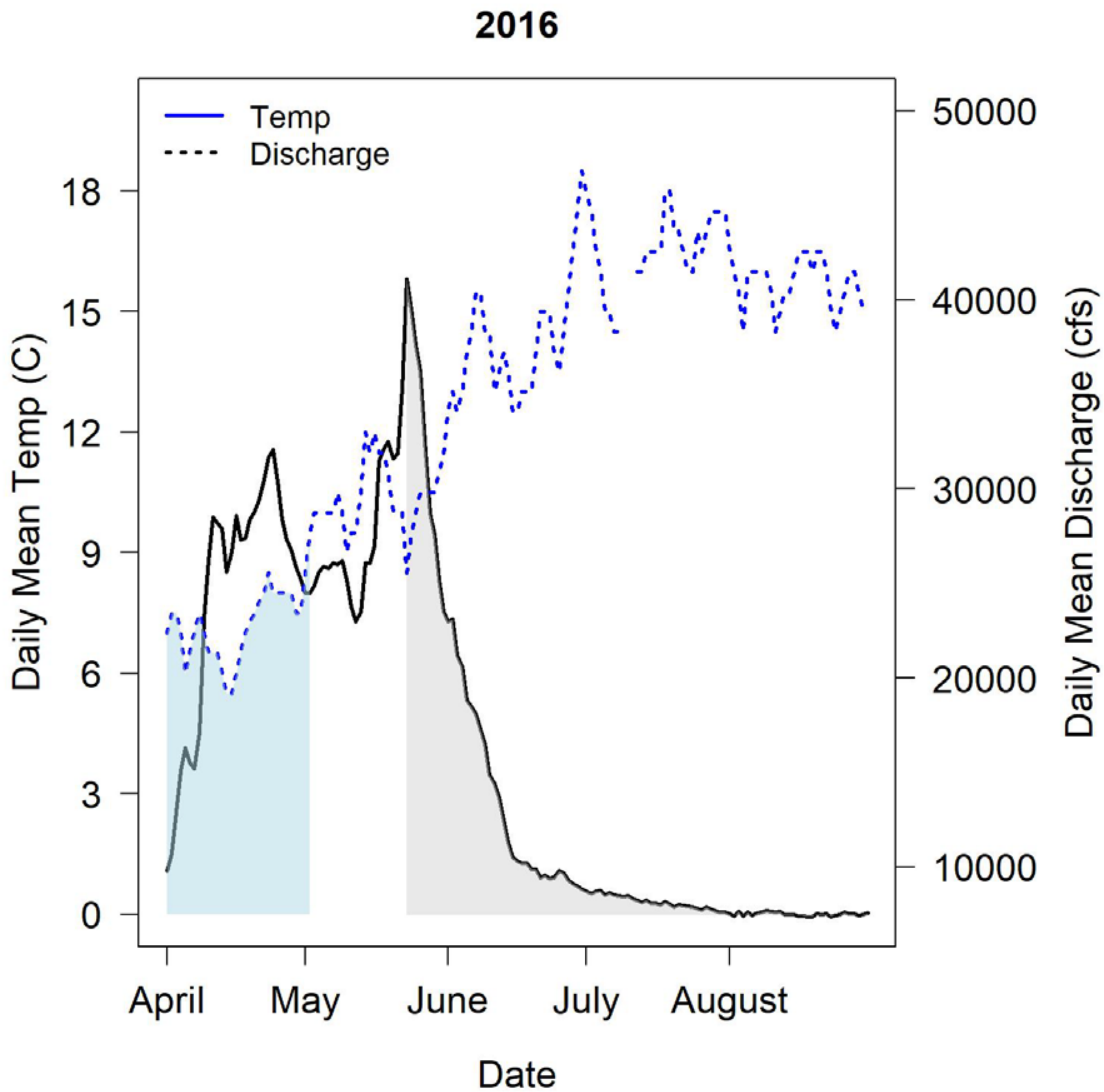

Figure 2.3. An example of how the *Templag*$_j$ covariate was calculated for 2016. The shaded areas represent period of time when the river temp at Bonners Ferry was <8°C or after the peak of the hydrograph. The white area is the lag (number of days) between those two events. This metric was calculated for each study year: 2005-2017.

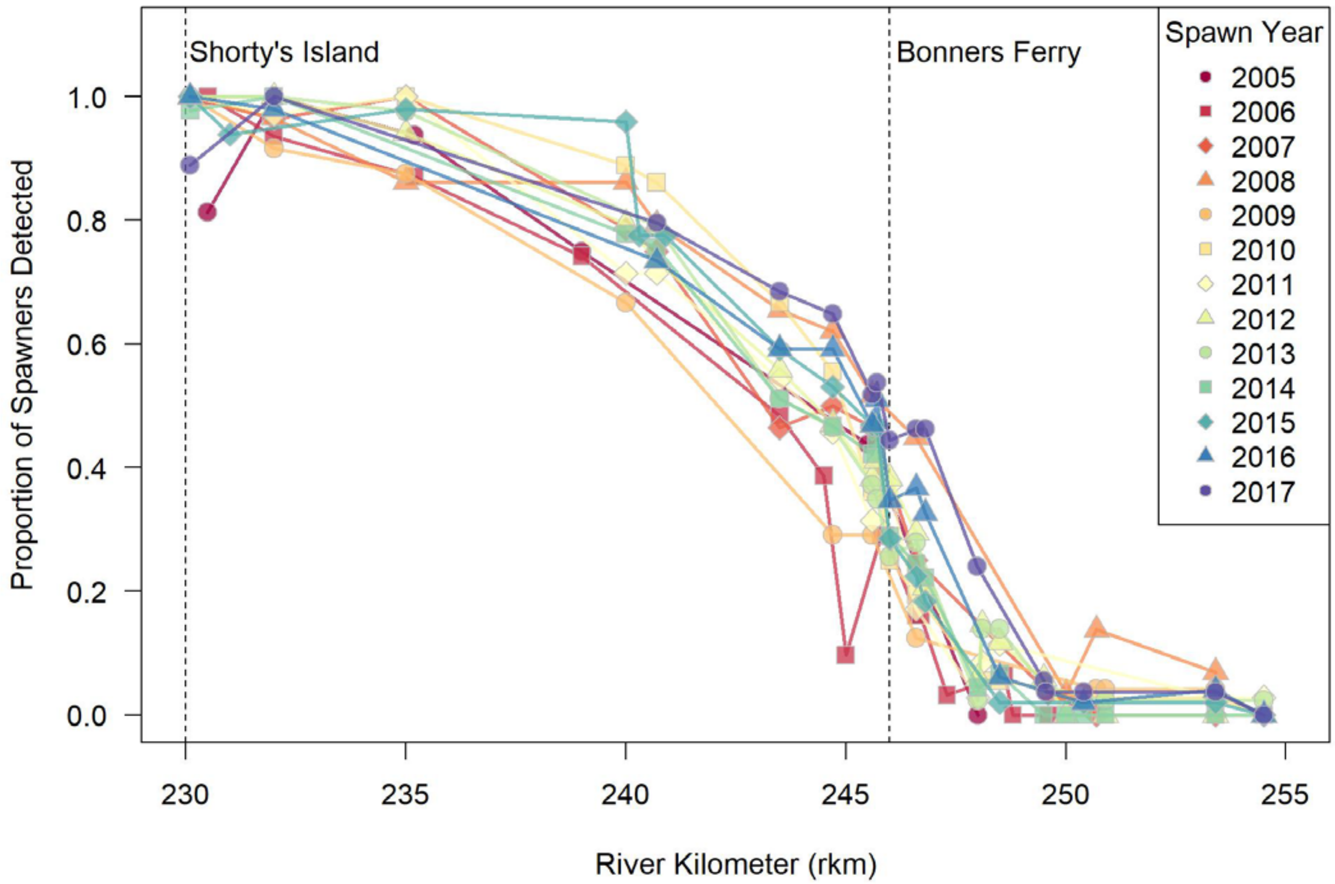

Figure 2.4. The proportion of Kootenai River White Sturgeon spawners detected at each of the VRW receivers between rkms 230 and 255 for each study year. The dashed lines represent Shorty's Island (rkm 230) and Bonners Ferry (rkm 246). All adult fish detected at or above Shorty's Island during the spawning season (April-July) are considered part of the spawning population for that spawn year. The figure demonstrates how relatively few spawners in any year move above Bonners Ferry into the Braided Reach.

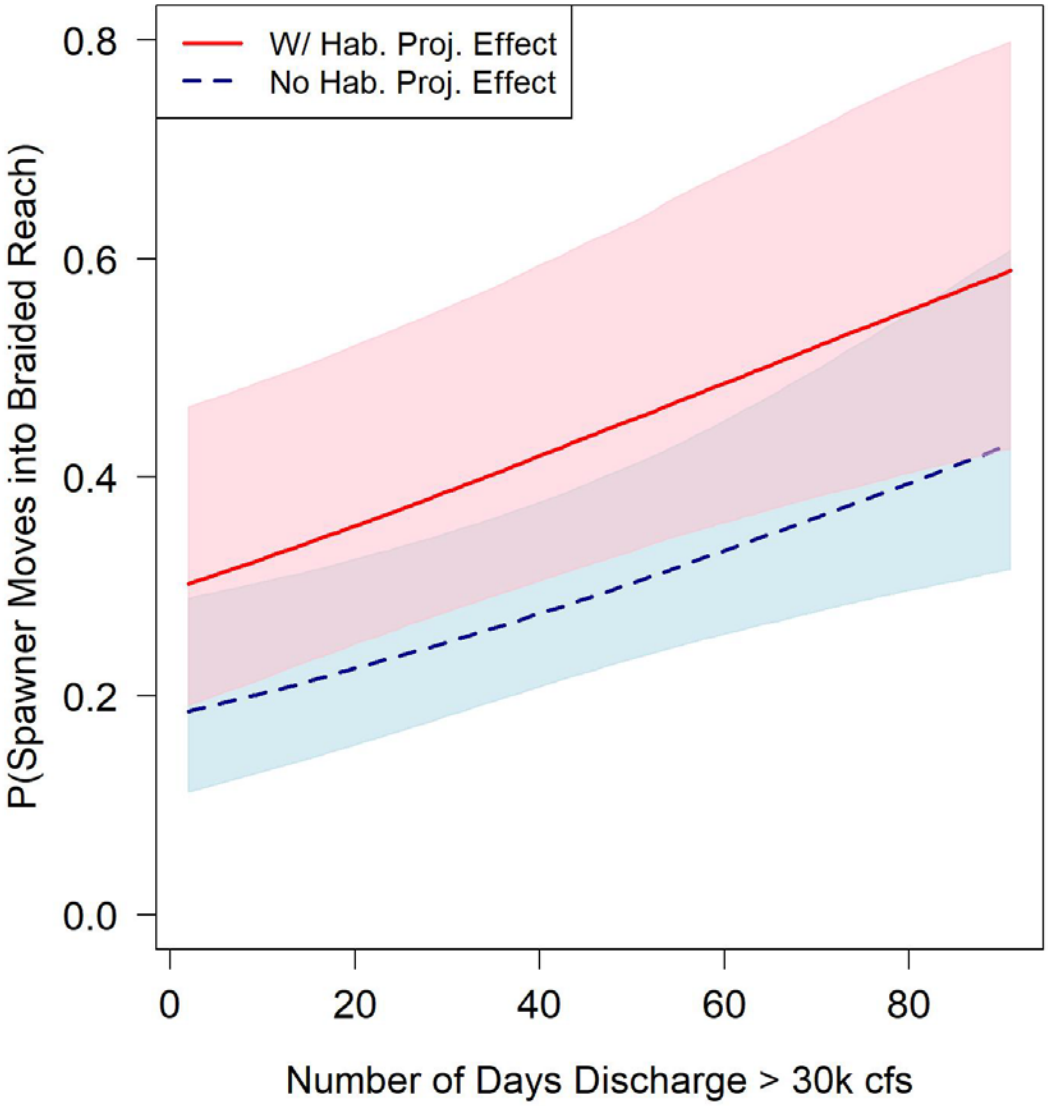

Figure 2.5. The back calculated influence of the two explanatory variables from the top logistic model. The slope of the lines represents the effect that the number of days when discharge is >30k cfs at Bonners Ferry has on the proportion of spawners that move into the braided reach in a given year. The difference between the lines demonstrates the influence of the habitat projects in the braided reach have had on that movement probability.

CHAPTER 3: WILD ADULT KOOTENAI RIVER WHITE STURGEON POPULATION UPDATE

ABSTRACT

The current analysis represents the most up-to-date estimates of Kootenai River White Sturgeon *Acipenser transmontanus* population abundance and apparent survival from 1990 through 2017. This analysis substantially improves the precision of the population parameter estimates and provides a more accurate assessment of current population conditions. Revised estimates of adult Kootenai Sturgeon abundance suggest that the remaining wild adult population is larger (≈ 1,744) and survival is higher (≈ 96%) than previously reported. However, population monitoring also indicates that natural recruitment is still insufficient to sustain the population and that high levels of mortality are occurring in early life stages. The current analysis provides new information and highlights the importance of frequent and robust population monitoring including sampling on a wider temporal scale that allows more unmarked adults to be encountered. The updated analysis highlights that the Kootenai Sturgeon population is larger than previously estimated and is not exhibiting the sudden increase in mortality previously reported. This updated information should provide managers a more reliable benchmark from which to gauge the effectiveness of long-term recovery strategies implemented in the system.

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INTRODUCTION

Since the 1970s, the population of wild-origin Kootenai River White Sturgeon (hereafter Kootenai Sturgeon) has been in decline. The primary drivers of the population decline have been a combination of overharvest and anthropogenic habitat and hydrologic changes. Although legal sport fishing was completely eliminated in 1994, the lack of adequate spawning and rearing habitats has limited recruitment to almost nonexistent levels since the installation of Libby Dam in 1972. As a result of lowered spring flows, the Kootenai River no longer inundates its historic floodplains and cleans spawning substrates required for larval incubation. The aforementioned changes in the Kootenai River have created an environment that results in poor recruitment of Kootenai Sturgeon and abundance of wild fish has continued to decline as a result.

The U.S. Fish and Wildlife Service (USFWS) finalized a recovery plan in 1999 to guide recovery strategies for Kootenai Sturgeon. The document provided recommendations for implementing actions that would reduce or eliminate threats to the species while promoting a self-sustaining population. In order to determine the success and effectiveness of these efforts, the Idaho Department of Fish and Game, along with B.C. Ministry, and Montana Fish, Wildlife, and Parks, intensively monitors the population to determine the success and efficacy of targeted efforts to mitigate limiting factors for the population. These efforts have also provided managers with an intensive and long-term assessment of population status for multiple life stages.

The current population of Kootenai Sturgeon is comprised of primarily larger and older individuals as a result of low natural recruitment for the past 50 years (Paragamian et al. 2005). A previous mark-recapture based estimate of abundance reported a steady decline from 6,000 adults in 1980 to 760 in 2000. The estimates were used to forecast that less than 300 adults would remain by 2011 (Paragamian et al. 2005). An additional abundance analysis conducted in 2013 incorporated an individual-based random effect in capture probability. The methodology was designed to account for the transition of spawning adults between Kootenay Lake in British Columbia and the upper Kootenai River where many of the recaptures were being sampled. Based on the analysis, abundance of White Sturgeon in 2011 was estimated to be approximately 990 individuals (Beamesderfer et al. 2014). In addition, mean annual apparent survival rates declined from 96% in 2000-2007 to 85% in 2007-2010.

This report provides an interim updated status assessment of wild adult Kootenai Sturgeon abundance and survival based on five years of additional mark-recapture data (2013-2017) collected in the annual monitoring program. A more comprehensive reporting of these data and the specific analyses involved will be available in the spring of 2020.

OBJECTIVE

Estimate population abundance and survival of adult Kootenai Sturgeon using recapture data from 1990-2017.

STUDY SITE

The Kootenai River originates in Kootenay National Park, British Columbia (BC), Canada. The river flows south into Montana and turns northwest at Jennings, near the site of Libby Dam, at river kilometer (rkm) 352.4 (Figure 3.1; for a more complete description of the study area see Chapter 1).

METHODS

Adult Kootenai Sturgeon Sampling

Adult Kootenai Sturgeon were collected by angling and set lining in the spring from March-May in 1995-2017 following the methods of Paragamian et al. (1996). During this time, the majority of sampling occurred in the “staging” areas between rkms 200 and 215. As the spring progressed, areas closer to documented spawning locations (i.e., near and above rkm 229) were sampled more frequently. Fall (September-October) sampling was incorporated into the annual monitoring beginning in 2012. The fall sampling primarily occurred throughout the lower river (rkms 207.5-308) and in Kootenay Lake at the Creston delta (rkm 118) and the Lardeau-delta (rkm 18) in British Columbia.

Adult Kootenai Sturgeon Population Modeling

To estimate annual apparent survival and population abundance, we used the same methodology as described in Beamesderfer et al. (2014) with additional steps to refine model selection and fit. Four individual open population Jolly-Seber models were evaluated in our study. Each candidate model represented a combination of constant versus time-varying annual apparent survival (ϕ) and an individual-based random effect (heterogeneous) versus homogeneous detection probability (p). All models were fit in a Bayesian framework using STAN (Stan Development Team 2018) in R (R Core Development Team 2018). A total of four chains, each with 10,000 iterations (5,000 burn-in iterations) was used to fit each model. We assumed completely uninform (i.e. “flat”) priors during all analyses. Model convergence was checked by evaluating the posterior sample traceplots and by the Gelman-Rubin diagnostic statistic (Rhat). The mean and 90% highest posterior density (HPD) credible intervals were reported from the joint posterior distribution for each estimated parameter (Box and Tiao 1992). Model fit was assessed using Watanabe-Akaike information criterion (WAIC) and leave-one-out cross-validation in the R package *loo* (Yao et al. 2017).

RESULTS

The inclusion of routine fall sampling since 2012 (Figure 3.2) appears to have provided a more robust assessment of the adult population, with relatively more encounters of both marked and unmarked fish. Analysis of updated adult capture data showed that the mean value of p_t increased from 1990-2017 while the mean value of ϕ_t remained relatively constant at 96.7% (Figure 3.3). A comparison of our results indicated that the current estimates of ϕ_t did not drop dramatically from 2007-2010 as reported by Beamesderfer et al. (2014). Similarly, our estimated mean abundance of wild adults in 2011 was higher (2,072) than reported by Beamesderfer et al. (2014; 990 adults; Figure 4). We estimated that mean adult abundance was 1,744 (CI 1,232, 2,182) individuals in 2017 (Figure 3.4).

DISCUSSION

The current analysis represents the most recent estimates of Kootenai Sturgeon population abundance and apparent survival. This report added an additional five years of data to the 22 years that were included in the previous analysis. Including fall sampling data substantially improved the population abundance and apparent survival estimates and provided a more accurate assessment of current population status.

Our estimate of Kootenai Sturgeon abundance suggests that the remaining adult population is larger than previously thought. The increase in these two parameters from those reported in 2014 were attributed to an increase in wild unmarked adults captured in fall sampling efforts in the Kootenai River and Kootenay Lake; lake sampling levels increased, and the site at rkm 18 was added in 2012 (Stephenson et al. 2014). In addition, incorporating additional annual samples into the open Jolly-Seber model allowed more individuals to be recaptured, whereas non-recaptured fish were presumed to be mortalities in the previous modeling effort.

Regardless of the observed increase in estimated adult abundance, natural recruitment still remains extremely low. Sampling efficiencies of hatchery juveniles were used to estimate that an average of 85 wild juvenile Kootenai Sturgeon are produced annually in the Kootenai River (Hardy et al. 2016). This estimate of natural recruitment is not sufficient to sustain the population and suggests that high levels of early life stage mortality are still occurring.

The current analysis provided new information that highlights the importance of population monitoring that adequately captures spatial and temporal variability in a highly mobile large river species. Specifically, it demonstrated the need to continue sampling at a wider temporal scale to allow more unmarked adults to be sampled. Fall sampling provided information about the population outside the spring spawning period when adult sampling occurred historically. Fall sampling of the lower Kootenai River and Kootenay Lake delta allowed us to mark and recapture individuals that were not sampled in the spring. Although it is uncertain whether these individuals contribute to recruitment, they are still part of the population at-large and are important to include when assessing the current status of the Kootenai Sturgeon population.

Although recruitment is still an ever-present issue with this population, the updated analysis indicated that the Kootenai Sturgeon population was larger than previously estimated and was not exhibiting the sudden increase in mortality as reported by Beamesderfer et al. (2014). This information should provide managers a more reliable benchmark to gauge effectiveness of long-term recovery strategies implemented in the system.

RECOMMENDATIONS

1. Estimate adult population abundance and survival at least every four years to assess Kootenai Sturgeon population status.

2. Continue fall sampling for adults in the Kootenai River and Kootenay Lake to improve estimates of Kootenai Sturgeon abundance and survival.

FIGURES

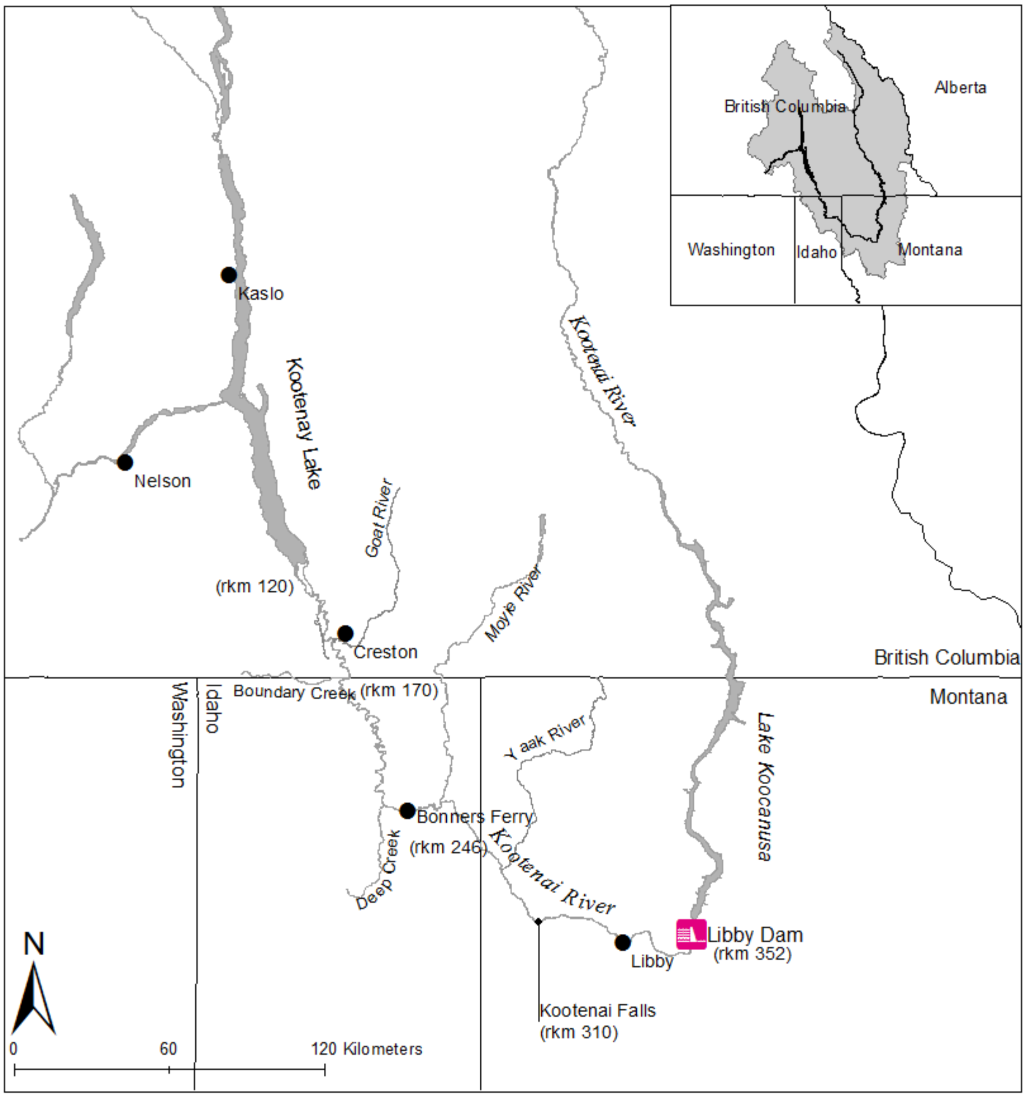

Figure 3.1. Location of the Kootenai River, Kootenay Lake, Lake Koocanusa, and major tributaries. River distances are from the northernmost reach of Kootenay Lake and are in river kilometer (rkm).

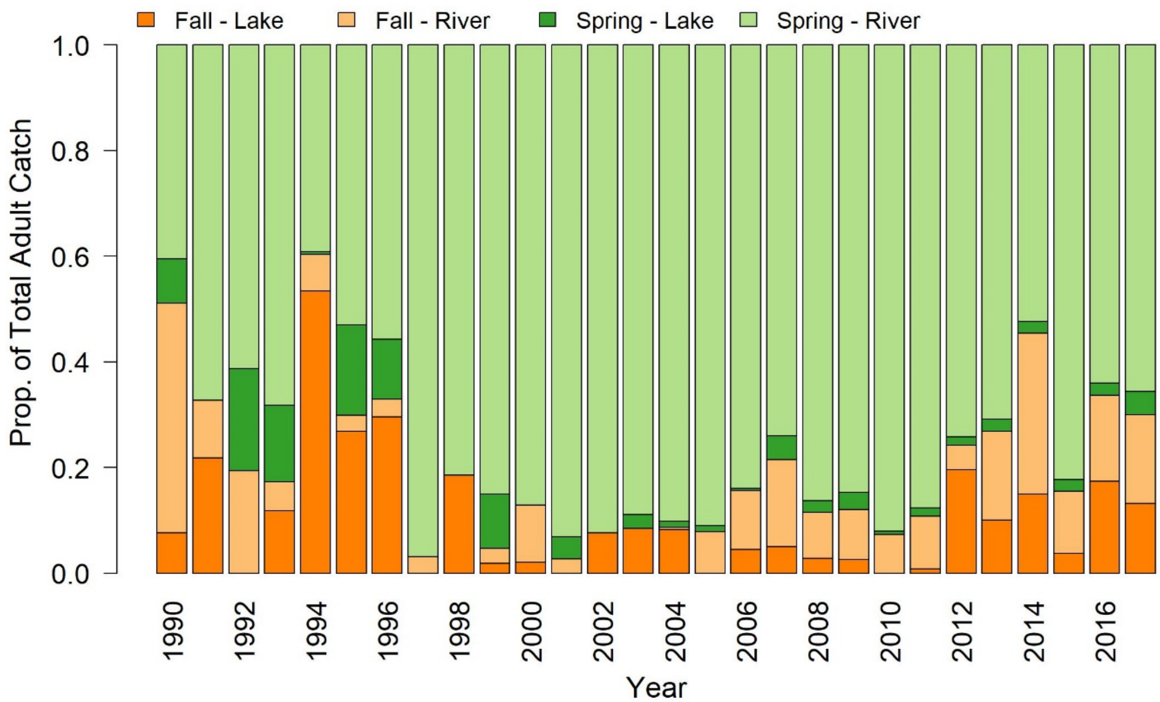

Figure 3.2. Proportion of Kootenai Sturgeon (wild adults) captured in the spring and fall in the Kootenai River and Kootenay Lake from 1990-2017.

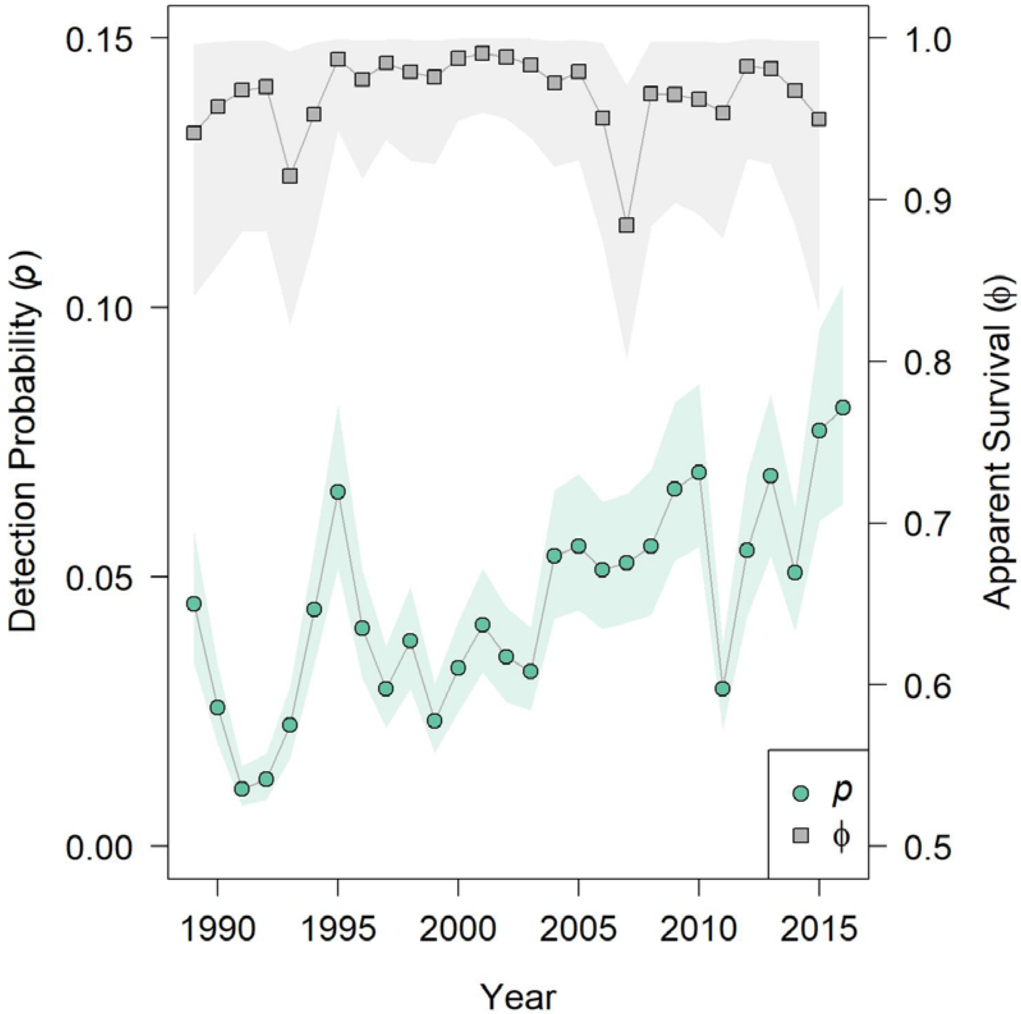

Figure 3.3. Parameter estimates for the model with constant annual survival and individual heterogeneity in detection probability. Values are the mean and 95% credible interval.

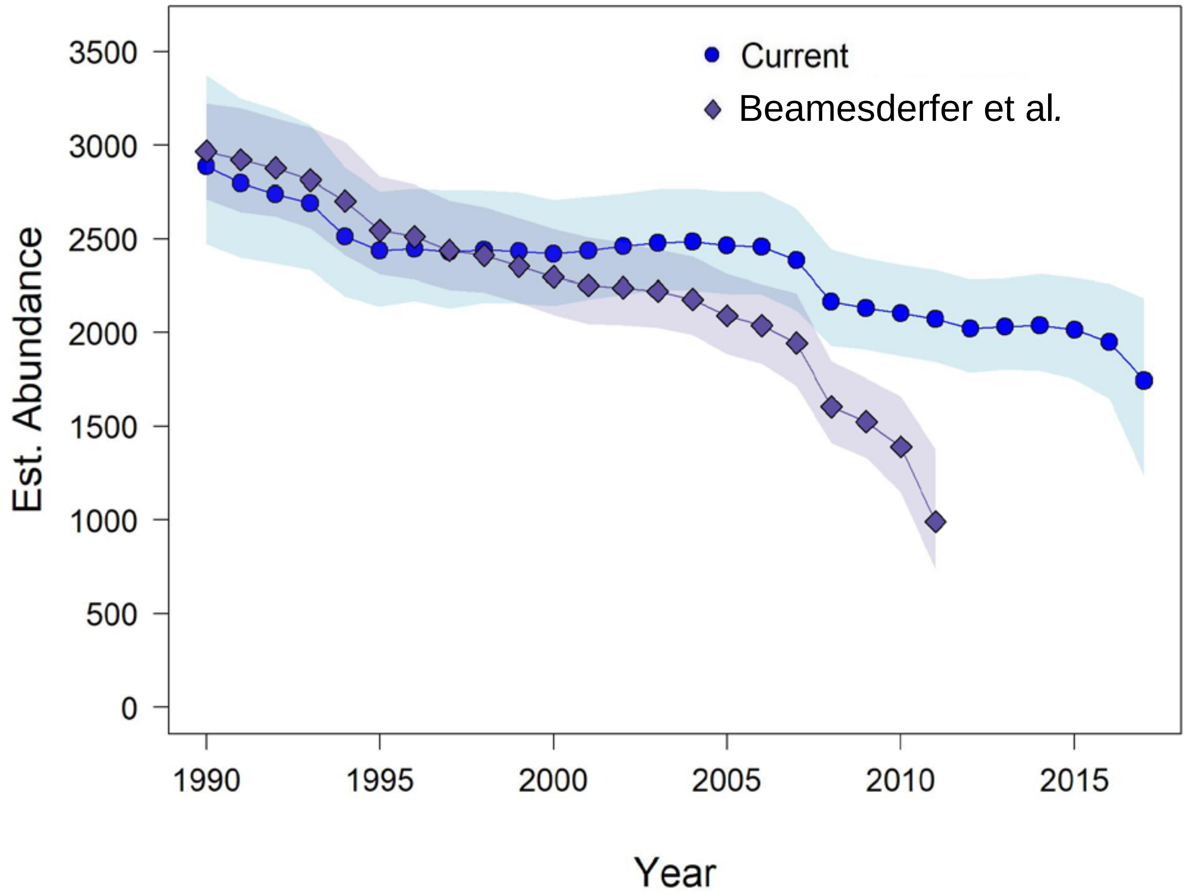

Figure 3.4. Comparison of current adult Kootenai Sturgeon population abundance and those reported by Beamesderfer et al. (2014a). The current estimate uses time-varying survival and individual heterogeneity in detection probability. Values are the mean and 95% credible interval.

CHAPTER 4: BURBOT MONITORING AND EVALUATION

ABSTRACT

Burbot (*Lota lota maculosa*) were once abundant in the Kootenai/ay River basin in Idaho, Montana, and British Columbia where they provided important commercial, recreational, and cultural fisheries throughout the basin. However, cumulative effects of purported over-exploitation and the completion of Libby Dam in Montana in 1972 resulted in the entire fishery collapsing and being closed to harvest by 1992. Until recent years, the population was considered functionally extirpated. Conservation aquaculture efforts by the University of Idaho Aquaculture Research Institute and the Kootenai Tribe of Idaho, in conjunction with largescale mitigation efforts and long-term research, monitoring, and evaluation, have revealed key insights into the current status of the species in the Kootenai River basin. Cumulatively, efforts from all project collaborators resulted in successfully opening a harvest fishery for Burbot in January 2019. Results from 2018 and 2019 indicated that (1) Burbot numbers exceeded restoration targets for adults in January 2019, (2) the Burbot population could sustain an annual fishing mortality of 10-15%, (3) the fishery was comprised of multiple year classes produced from hatchery efforts, (3) two life stages of larval-released hatchery Burbot survived in the river and in off-channel habitats, (4) hatchery-reared Burbot were showing signs of successful reproduction in the river for two consecutive years, and (5) survival of post-feeding, hatchery-reared Burbot in the mainstem river was 0.3%. Burbot continue to pioneer into tributary habitats during the spawning season, which could potentially provide a mechanism for continued natural recruitment if the temperature regime in the mainstem river is responsible for the recruitment bottleneck, as purported. All indicators, including initial evidence of natural recruitment, suggest that the population is increasing and has reached restoration targets due to the release of hatchery-reared Burbot. As such, managers and project collaborators are confident the population can continue to support 10-15% annual fishing mortality while also meeting objectives outlined in the 2005 Burbot Conservation Strategy.

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INTRODUCTION

Although Burbot *Lota lota maculosa* are widespread and abundant throughout much of their natural range (Evenson and Hansen 1991), many populations are in severe decline (Arndt and Hutchinson 2000; Paragamian et al. 2000). As a result, restoration efforts have been initiated to mitigate factors threatening populations with further decline or localized extirpation (Dillen et al. 2008; Worthington et al. 2009; Stapanian et al. 2010). A primary source of decline has been attributed to significant changes in habitat, often stemming from the construction of dams used for flood control or hydropower (i.e. the Kootenai River basin in Idaho, Montana, and British Columbia). Libby Dam, constructed in the early 1970s, has significantly increased discharge and water temperature during the winter spawning period for Burbot (Partridge 1983), which is thought to have negatively impacted recruitment (Hardy and Paragamian 2013). Additional impacts from the construction of the dam and diking within the Kootenai floodplain include decreased nutrient availability and loss of habitat from floodplain isolation (Hardy 2003). As a result, the Burbot fishery rapidly declined through the mid-1980s and ultimately culminated with a complete closure of the fishery in 1992. Concomitant to the collapse in the Idaho portion of the Kootenai River, a rapid decline of the Burbot fishery in Kootenay Lake and Kootenay River, British Columbia (BC) was observed, resulting in those fisheries closing in 1997 (Paragamian et al. 2000).

Due to the widespread cultural and recreational importance of Burbot in the Kootenai River basin prior to the collapse of the population, an International Burbot Conservation Strategy was developed by a community-wide working group to help restore the population (Paragamian et al. 2002; KVRI 2005; Ireland and Perry 2008) such that it was self-sustaining and could support a harvest fishery. The Strategy outlined rehabilitation measures, including changes to operations of Libby Dam and development of conservation aquaculture to supplement the wild stock during population rehabilitation. Because the Burbot population was speculated to be too small to recover on its own or provide gametes for a conservation aquaculture program, managers deemed it necessary to locate and use a donor stock to aid in restoration efforts. Of the many water bodies sampled, Burbot from Moyie Lake, BC (Figure 4.1) were selected as a suitable donor stock because they were found to be of a similar phylogenetic group as the Kootenai River population (Powell et al. 2008), abundant enough to provide sufficient gametes, and had spawning sites that provided easy access to spawners. Concurrent with studies to locate a broodstock source, intensive rearing techniques were successfully developed at the University of Idaho Aquaculture Research Institute (UIARI; Jensen et al. 2008). As a result of this success, the Kootenai Tribe of Idaho (KTOI), Idaho Department of Fish and Game (IDFG), and British Columbia Ministry of Forests, Lands, Natural Resource Operations, and Rural Development (FLNRORD) have stocked larval, juvenile, and adult Burbot into the Kootenai River basin and its tributaries since 2009 in an effort to aid natural production and test specific population-limiting factors.

GOAL

The long-term management goal of this study was to restore a naturally reproducing and harvestable Burbot population in the Kootenai basin.

OBJECTIVES

1. Broadly characterize the status of the Burbot population in the Kootenai River, Idaho.

2. Characterize spatiotemporal occurrence(s) of spawning in the Kootenai River, Idaho.

3. Provide broodstock from the Kootenai River for the Kootenai Tribe of Idaho hatchery at Twin Rivers, Idaho.

4. Evaluate the success of various aquaculture stocking strategies, from 2009-2019.

5. Identify and experimentally evaluate potential factors limiting recruitment of Burbot in the Kootenai River, Idaho.

6. Evaluate broad-scale use of Deep, Boundary, Smith, and Ball creeks by Burbot in the winter months.

7. Develop an age-based Leslie matrix model to (1) estimate current and project future abundance of Burbot in the river and (2) simulate potential effects of fishing mortality on the Burbot population.

8. Scope and implement a Burbot fishery for the Kootenai River beginning in January 2019.

9. Design and implement a creel survey to monitory the Burbot fishery in the Kootenai River.

STUDY AREA

The Kootenai River is the second largest tributary to the Columbia River, and its drainage is the third largest (approximately 49,987 km^2; Bonde and Bush 1975). The river originates in Kootenay National Park, BC and flows south into Montana, where Libby Dam impounds water into Canada and Montana to form Lake Koocanusa (Figure 4.1). The river flows west from Libby Dam, northwest into Idaho, then north into BC and Kootenay Lake. The river then drains out of the West Arm of Kootenay Lake, and it eventually joins the Columbia River near Castlegar, BC. Kootenay Lake has a surface area of 390 km^2 and is a fjord-like lake, running north-south in the trench formed between the Selkirk and Purcell mountains. Approximately 105 river kilometers (rkms) flow through the Idaho section of the Kootenai River basin.

During the study period reported herein (i.e., 2018 and 2019), hoopnet sampling for Burbot occurred at 28 sites between rkm 144.5 (Nick's Island, near Creston, BC) and rkm 276 (Leonia, at the Idaho and Montana border) (Figure 4.2). A passive integrated transponder (PIT) tag array was installed in Deep Creek in October 2012, approximately seven km upstream from its confluence with the Kootenai River (Figure 4.2), and it has been operated continuously since installation. Unfortunately, the array malfunctioned and was not operational during winter 2017/18; however, the array was fully functional and operating during winter 2018/19. Passive integrated transponder (PIT) tag arrays were also installed in Boundary, Smith, and Ball creeks in October 2017; all three arrays remained operational through May 2018.

METHODS

Burbot Hoopnet Sampling

Burbot Stocking

Following the success of intensive culture at the UIARI and the new KTOI Burbot and Sturgeon hatchery, a total of approximately 14,200,000 Burbot (ranging in age from larvae to age-2) were released into the Kootenai River, its tributaries, and its floodplain from 2009-April 2018 (Table 4.1). Approximately 64,000 of that total were juveniles (age-0 to age-2) tagged with PIT tags (full duplex [FDX; 2009-2013] and half duplex [HDX; 2014-2017]; BioMark Inc. 9 mm and Oregon RFID 12 mm, respectively) and released into tributaries to and the mainstem of the Kootenai River by KTOI, MFLNRORD and IDFG personnel (Table 4.1).

Approximately 315,000 planktivorous-feeding larvae were released at each of Ambush Rock and Porthill in spring 2015. Approximately 700,000 and 1,600,000 planktivorous feeding larvae were released at Nimz Ranch (KTOI-owned naturally inundated floodplain habitat) and Ferry Island in spring 2018, respectively. Lastly, 800,000 and 1,800,000 pre-feeding larvae were released at Ferry Island and Nimz Ranch in spring 2018, respectively. The aforementioned larval releases were in support of an in-basin experiment to test various recruitment limitation hypotheses using genetic analyses and release-specific survival estimates.

Survival Analyses

First-year survival of Burbot from the planktivorous-feeding larval releases in 2015 was estimated using a Bayesian version of the Cormack-Jolly Seber model, where detection probability and survival were allowed to vary among sampling events (years). The objective was only to estimate first-year (i.e., larval) survival. Details for estimating survival of other age-classes of Burbot can be found in Ross et al. 2015.

Mainstem Hoopnet Sampling

Adult Burbot were sampled at 28 locations using 56 baited hoopnets during winters 2017/18 and 2018/19 (eight Canadian and 20 U.S. sites; Figure 4.2) in order to measure relative changes in the population through catch-per-unit-of-effort (CPUE; number of Burbot/net-day) and other population metrics. Six historical index sites (i.e., since 1994) along with an additional 22 sites were sampled from December 1, 2017 to March 31, 2018 and December 1, 2018 to March 21, 2019 to collect information on CPUE, growth, year class survival, and spawning activity within the Kootenai River. From 1996-2009, each river site was sampled using hoopnets (2.00 m x 0.61 m) with 25.4 and 19.1 mm bar-mesh sizes. Beginning in 2010, two hoopnets of 19.1 and 6.4 mm bar-mesh sizes were paired at each site to evaluate gear selectivity; this has been the standard protocol since 2010. All nets were baited with frozen kokanee *Oncorhynchus nerka* and other fish species native to the Kootenai River basin and checked 2-3 times per week. All captured Burbot were counted, measured for TL (mm), weighed (g), and examined for previous tags. All untagged Burbot were injected with a unique half duplex PIT tag into the right anterior dorsal muscle for future analyses, including population estimates, growth, and survival by year class, and others. Tissue samples for genetic analysis were collected from the anterior portion of the dorsal fin of all tagged and untagged Burbot to determine origin (i.e., hatchery or wild), year-class, release location, age-at-release, and sex using Parental Based Tagging (PBT) analysis (methods described by Anderson and Garza 2005; Steele and Campbell 2011). In addition, all Burbot captured from January 2 - January 31 in both 2018 and 2019 were transported to the Kootenai

Tribe of Idaho hatchery at Twin Rivers where they were used as broodstock for year class production and various experimental lab and field studies.

Tributary Use by Burbot in the Kootenai Basin

On October 11, 2012, a dual-reader (i.e., FDX and HDX) Biolite BioMark Passover PIT antenna was installed in Deep Creek, Idaho, approximately seven rkms from the confluence with the Kootenai River (Figure 4.2). Details regarding operations of the array, Burbot PIT tagging protocol, and Burbot release numbers and locations (from 2012-2015) can be found in Ross et al. (2015). The antenna in Deep Creek malfunctioned prior to the beginning of the 2017/18 spawning season and remained non-operational for the entirety of the season. The array was serviced by BioMark in summer 2018 and remained operational for the entirety of the 2018/19 spawning season. Oregon RFID HDX PIT tag antennas were installed in Boundary, Smith, and Ball creeks in October 2017 approximately 0.25-0.5 rkms from the tributary's confluences with the Kootenai River; all antennas remained operational through May 2018. Methods outlined in Beard et al. (2017b) were used to install and operate all Oregon RFID HDX PIT tag antennas. The purpose of the antennas in Deep, Boundary, Smith, and Ball creeks during winters 2017/18 and 2018/19 was to document broad scale use of tributaries by Burbot during the spawning season and to characterize the composition and timing of any observed migrations of Burbot in tributaries that were being monitored.

Potential Effects of Water Temperature on the Egg Hatching Success, Larval Development, and Larval Survival of Burbot

The Idaho Department of Fish and Game began funding a PhD dissertation with the UIARI in fall 2015. The project has been completed, and data are currently being analyzed with the intent of producing multiple peer-reviewed manuscripts. The primary research objectives were threefold: (1) to assess historic and contemporary regimes of the lower Kootenai River in relation to Libby Dam operations and climate; (2) to conduct experiments that tested the effects of different temperatures on Burbot spawning, embryo development, and larvae energetics; and (3) to identify models of river flow management and climate that supported early life stages of Burbot. The overarching goal was to provide resource managers with information that determines if post-dam river regimes enable or prohibit natural recruitment. Results from the study will be available in 2019-2020 in the form of multiple peer-reviewed manuscripts and a dissertation. The first of several publications is complete and can be found in Ashton et al. (2019).

Briefly, methodology for the project included: (1) describing "typical" thermal regimes and temperature spikes for the lower Kootenai River from 1994-2016 and historic pre-dam temperature regimes from 1967-1972, (2) exposing adult Burbot to different temperature regimes before spawning to assess effects on spawn timing and early embryo development, and (3) evaluating the effect of varying temperature regimes on Burbot hatching success, embryo deformity, and larval survival.

Burbot Population Model: Population Projections and Harvest Simulations

We developed a stochastic, age-based population dynamics model for Burbot in the Kootenai River to (1) estimate and project the current and future abundance of Burbot in the river, and (2) better understand the potential effects of varying levels of fishing mortality on the Burbot population. An age-based Leslie matrix was used to model annual survival and reproduction of Burbot in the Kootenai River; methods used for the model generally followed the structure outlined in Klein et al. (2016). The transition matrix, $\boldsymbol{A}$, took the form:

$$
\boldsymbol{A} = \begin{bmatrix} f_0 & f_1 & \cdots & \cdots & f_{5+} \\ s_0 & 0 & \ldots & \ldots & \vdots \\ 0 & s_1 & & \ldots & \vdots \\ \vdots & \vdots & \ddots & \ldots & \vdots \\ 0 & 0 & \cdots & s_4 & s_{5+} \end{bmatrix}
$$

where f_i is the fecundity rate of age class *i* and s_i is the annual survival rate for age class *i*. At each time-step, age-specific abundances were calculated as:

$$
\boldsymbol{N}_{t+1} = \boldsymbol{A}\boldsymbol{N}_t - \boldsymbol{Q}_t + \boldsymbol{H}_t
$$

$$
\boldsymbol{Q}_t = F_t \cdot \left[\sum_{\boldsymbol{i=2}}^{\boldsymbol{6}} \boldsymbol{N}_{t,i}\right] \cdot \boldsymbol{Select}
$$

where $\boldsymbol{N}_t$ is a vector that represents the abundances of each age class *i* at time step ***t***. The vector $\boldsymbol{Q}_t$ represents the number of individuals in each age class that get harvested in time step ***t***. The value of $\boldsymbol{Q}_t$ is determined by multiplying the total number of individuals available for harvest by the additive fishing mortality rate F_t and the angling selectivity ***Select***. The ***Select*** is a simplex vector of equal length of ***N*** that describes the proportion of the total angling harvest that is applied to each age class *i*. Lastly, $\boldsymbol{H}_t$ is the age-specific number of hatchery origin individuals released at time step *t*.

Burbot are known to exhibit cannibalistic behaviors at high densities, so we included a density dependent estimate of s_0 which was calculated as:

$$
s_0 = s_{0max} \cdot \left[1 - exp\left(-K \big/ s_{0max} \cdot \textstyle\sum_{\boldsymbol{i=2}}^{\boldsymbol{6}} \boldsymbol{N}_{t,i}\right)\right]
$$

where s_{0max} is the maximum survival rate for individuals in age class 0, and $-K$ is a rate parameter that describes the abundances at which density dependence begins to manifest. Although we do not have empirical estimates of $-K$, this function allowed us to evaluate whether a density dependence effect on early life history would influence outcomes from various management strategies.

We developed the model to utilize our current population vital rate estimates for Kootenai River Burbot. Specifically, the model inputs included: age-specific survival rates, age-specific fecundity rates, current age-specific abundances (i.e., as of 2016), annual fishing mortality, angling size/age selectivity, and annual hatchery release numbers of age-0 Burbot. Parameter values, associated variances, and sources from which they were derived are outlined in Table 4.2. Age-specific survival rates and age-specific initial starting numbers were estimated by Ross et al. (2018). We should note that age-specific fecundity rates and wild egg survival rates are not applicable to the Kootenai River Burbot population since natural recruitment is negligible or functionally nonexistent; these parameters were assumed to be equal to zero for all model simulations. When the model was initially developed, survival rates had not yet been estimated for larval-released Burbot in the Kootenai River; however, 0.002 was agreed upon via expert opinion among all program co-managers. The value was used only for the purpose of back-calculating potential survivor numbers from larval releases in 2009-2017 (Ross et al. 2018). Hatchery age-0 release numbers were derived from previous release numbers and a future target release number of 125,000 age-0 juveniles/year as identified in the International Burbot

Conservation Strategy. The associated variance in annual age-0 release numbers was derived from variability in release numbers since 2011. Angling selectivity allowed us to simulate scenarios where fishing pressure was more weighted toward smaller younger fish, larger older fish, or equally between the two. Younger fish were defined as age-0 to age-2 (i.e., sexually immature), and older fish were defined as age-3+ (i.e., sexually mature).

Annual angling mortality was user defined based on the desired model simulation, and values ranged between 0.0 and 1.0. The density dependent function (DDF) was also user defined such that predictable declines in age-0 survival occurred differentially across a defined set of values for adult abundance (Figure 4.3). The maximum allowable age-0 survival was that which was derived from Ross et al. (2018); simulated declines in age-0 survival in response to predefined thresholds of adult abundance (i.e., 17,500 adults) all remained below the starting value of 0.10. Prior to running the model, one-way sensitivity analyses were run with and without the DDF in effect. One-way sensitivity analyses were run to quantify and better understand which model parameters were most influential in the simulated model results. The population model, DDF, and sensitivity analyses were all run in R (R Core Development Team 2018).

Four particular modeling scenarios were of interest. Scenario 1 simulated population trends in the absence of both the DDF and fishing mortality. Scenario 2 simulated population trends in the absence of the DDF but with the effect of varying levels of fishing mortality. The intention was to introduce systematic increases in fishing mortality to quantify the annual fishing mortality rate the population could withstand without allowing adult abundance to drop below the restoration target of 17,500. Scenario 3 simulated population trends with the DDF in effect but in the absence of fishing mortality. Finally, scenario 4 simulated population trends under the effects of both the DDF and varying levels of fishing mortality.

Burbot Fishery

During 2018, IDFG developed a proposal to provide a harvest fishery for Burbot based on the restoration actions and population monitoring data available at that time. Following a public input process and eventual IDFG Commission approval, the Burbot fishery opened on January 1, 2019 with a daily bag limit of six Burbot of any size.

To monitor angler participation, catch, and harvest in the newly-opened fishery, IDFG implemented a stratified roving creel survey beginning in January 2019. The primary objectives of the creel survey were to (1) ensure that annual fishing mortality rates did not exceed the 15% threshold identified in the population model agreed upon by all project collaborators, (2) better understand angler distribution and catch in the Kootenai River and how they might affect model predictions, and (3) maintain a consistent and positive IDFG presence on the river during the first year of the fishery opening.

The creel survey began on January 1, 2019 and ended on March 10, 2019. During the survey period, the reach of river from the US Highway 95 bridge to the Porthill boat launch (Figure 4.2) was surveyed, as well as sections of Deep, Boundary, and Smith creeks. We anticipated that most anglers would fish from the riverbank; therefore, creel survey strata were delineated such that creel clerks could adequately patrol one survey stratum per shift, targeting areas with public shoreline access. The delineated strata included (stratum 1) the US Highway 95 bridge to the Deep Creek boat launch, (stratum 2) the Deep Creek boat launch to the Copeland boat launch, (stratum 3) the Copeland boat launch to the Porthill boat launch, and (stratum 4) Deep, Boundary, and Smith creeks. Strata were weighted prior to designing the formal creel schedule in order to proportionally allocate more survey effort to areas expected to receive more angler effort. Stratum weighting was as follows: stratum 1 (50%), stratum 2 (20%), stratum 3 (20%), and stratum 4 (10%). Creel shifts were randomly selected (with weighting) on a 24-hour schedule and a seven-day week. Evening shifts were more heavily weighted than daytime shifts (70% and 30%, respectively), and weekend shifts were more heavily weighted than weekday shifts (70% and 30%, respectively). During a given shift, angler interviews were opportunistically conducted, and two randomly scheduled, instantaneous angler counts were conducted within the stratum being surveyed.

In addition to the creel survey, trail cameras (Reconyx Hyperfire 2) were placed at five different locations to help further expand angler effort estimates. Two cameras were placed near Ambush Rock, one near the Deep Creek boat launch, one near the Copeland boat launch, and one near the Porthill boat launch. Cameras were positioned such that all pictures would capture the primary parking and fishing areas at each location. All cameras were set to take one picture per hour, 24-hours per day (i.e., not instantaneous photos based on detected movement). Cameras began operating on January 1, 2019 and were removed on March 10, 2019. After the creel survey was complete, camera data were tallied to count the number of vehicles, bank anglers, and boat trailers.

Independent of the creel survey, IDFG researchers also used the IDFG Tag You're It! program to estimate angler exploitation rates. During December 2018, 179 non-reward Floy tags were deployed in Burbot captured in hoopnets. Tags were deployed from rkm 123.5-244.5 (i.e., in the US and in CA) and in fish from all age- and size-classes (i.e., 274-872 mm; range). All tags contained a unique identification number, as well as information (for anglers) on how to report tags, when encountered. An additional 15 tags were deployed in Burbot in December 2018 that were $100 USD reward tags. The purpose of the reward tags was to allow researchers to estimate angler tag reporting rates in order to adjust the number of non-reward tags reported and better estimate exploitation rates. See Cassinelli and Meyer (2018) for details on Tag You're It!, angler tag reporting rates, and subsequent exploitation estimates. The estimated total number of Burbot harvested was calculated by multiplying exploitation by estimated abundance of Burbot. Since it was unknown whether angler harvest would be equally distributed among age- and size-classes or skewed, total abundance of all age-classes of Burbot was considered catchable/harvestable.

Creel Survey Analyses

Total angling effort in angler hours on day *d* at site *m*, ($\hat{E}_{dm}$) was estimated as:

$$\hat{E}_{dm} = T_d \bar{I}_{dm},$$

where T_d was the total number of hours in the fishing day and $\bar{I}_{dm}$ was the mean of the angler counts conducted on day *d*. Catch rate in weekly stratum *k* at site m ($\hat{R}_k$) was estimated as:

$$\hat{R}_{km} = \frac{\sum_{i=1}^{j_{km}} c_i}{\sum_{i=1}^{j_{km}} h_i},$$

where j_{km} was the total number of anglers interviewed in the stratum. Catch in time stratum *k* was estimated as the product of stratum effort and catch rate:

$$\hat{C}_{km} = \hat{E}_{km} \hat{R}_{km}.$$

Total catch in stratum k was estimated using the Horvitz-Thompson estimator:

$$\hat{C}_k = \frac{\hat{C}_{km}}{\pi},$$

where π was the sampling probability for the *m*th site. Catch was summed among strata to estimate catch over the duration of the season.

Exploitation was calculated from creel survey data by dividing the estimated number of Burbot harvested by the estimated abundance of Burbot in the river. Since it was unknown whether angler harvest would be equally distributed among age- and size-classes or skewed, total abundance of all age-classes of Burbot was considered catchable/harvestable.

RESULTS

Burbot Hoopnet Sampling

Burbot Stocking

A total of 939 Burbot from the 2015 year class were captured during winters 2017/18 and 2018/19. Of these, approximately 27% assigned (via PBT) to fish released at Ambush Rock and Boundary Creek confluences with the mainstem Kootenai River as planktivorous feeding larvae. Similar to results highlighted in Ross et al. (2018), larval-released Burbot from the 2015 year class captured in 2017/18 were, on average, 16% larger by length (i.e., 451.8 ± 42.3 mm; Figure 4.4) and 40% larger by weight (i.e., 692.9 ± 205.1 g) than Burbot from the 2015 year class that were released as juvenile fingerlings (i.e., 377.7 ± 48.7 mm and 412.3 ± 109.2 g). Captures of Burbot in 2018/19 from these release groups further support this notion in that they were, on average, 25% larger by length (i.e., 530.1 ± 44.7 mm for larval versus 422.1 ± 42.6 mm for juvenile) and 115% larger by weight (i.e., 1060.1 ± 329.3 g for larval versus 492.5 ± 174.6 g for juvenile).

To-date, zero Burbot from the pre- or planktivorous-feeding larval releases at Ferry Island in spring 2018 have been recaptured in annual hoopnetting efforts. However, targeted sampling by KTOI in floodplain habitats on Nimz Ranch documented survival of Burbot from both the pre- and planktivorous-feeding larval groups that were released directly into Nimz Ranch. In August 2018, KTOI staff sampled the main Nimz Ranch pond with minnow traps and captured a total of 107 Burbot; 39 were from the planktivorous-feeding larval release group and 68 were from the pre-feeding larval release group. Consistent with growth variability previously observed between planktivorous-feeding larval released and juvenile-released Burbot, the pre-feeding Burbot were 20% larger by length (i.e., 103.8 ± 8.3 mm) than conspecifics from the planktivorous-feeding releases (i.e., 82.1 ± 6.4 mm; Figure 4.5).

Survival Analyses

A total of 632,590 planktivorous-feeding larval Burbot were released into the mainstem Kootenai River in spring 2015; approximately 302,000 were released at Ambush Rock (rkm 244.5) and 330,000 at the confluence of Boundary Creek with the Kootenai River (rkm 170). From winter 2016-17 to 2018-19, 303 (0.05% of the total released) Burbot from the 2015 planktivorous-feeding larval Burbot releases were recaptured during annual hoopnetting efforts. One was recaptured in 2016, 46 in 2017, 175 in 2018, and 81 in 2019. These data were used to generate encounter histories and subsequent preliminary survival estimates for planktivorous-feeding larval Burbot. It is important to note that all fish released in 2015 as planktivorous-feeding larvae were genetically tagged via PBT. As such, every individual released was used for the analysis. The estimate generated represents the only known estimates of larval Burbot survival, to-date. Planktivorous-feeding larval Burbot were estimated to survive at a rate 0.003 (0.001-0.010 [95% credible intervals]).

Mainstem Hoopnet Sampling

Eighteen river sites downstream from Bonners Ferry were sampled from December 1, 2017 to March 31, 2018, totaling 3,953 net-days and 1,181 captured Burbot. Catch-per-unit-of-effort across all sites (i.e., index and non-index) in the 2017/18 season (0.30 Burbot/net-d; Figure 4.6a) was only 9% lower than the record high observed in the 2014/15 season (0.33 Burbot/net-d) and greater than all other CPUE values observed since 1993. Similarly, CPUE at only the index sites during the 2017/18 season (0.291 Burbot/net-d; Figure 4.6b) was only 0.3% lower than CPUE observed in the record high 2014/15 season (0.292 Burbot/net-d) and greater than all other CPUE values since 1997. An additional ten sites upstream from Bonners Ferry were sampled from January 9, 2018-February 22, 2018, totaling 763 net-days, 35 captured Burbot, and a catch rate of 0.05 Burbot/net-d. Although the catch rate in the 2018 season (0.05 Burbot/net-d) was 20% greater than that observed in the 2016/17 season (0.04 Burbot/net-d), it was a biologically meaningless increase. Given the low catch rates and dangerous winter river conditions in the stretch of river upstream from Bonners Ferry, further sampling in this river section was discontinued beginning winter 2018/19.

Eighteen river sites downstream from Bonners Ferry were again sampled from December 1, 2018 to March 31, 2019, totaling 3,468 net-days and 612 captured Burbot. It is important to note that, similar to winter 2016/17, climate conditions and dam operations in winter 2018/19 caused the Kootenai River to freeze, which precluded all hoopnet sampling from February 7 – February 14, 2019 and again from February 21 – March 11, 2019. The dates during which the river was frozen coincided with historical peak spawn timing for Burbot, which resulted in substantially fewer numbers of Burbot being captured during winter 2018/19. Catch-per-unit-of-effort across all sites (i.e., index and non-index) in the 2018/19 season (0.18 Burbot/net-d; Figure 4.6a) was 53% lower than the record high observed in the 2014/15 season (0.33 Burbot/net-d) and lower than all other CPUE values observed since winter 2013/14 (0.13 Burbot/net-d). Similarly, CPUE at only the index sites during the 2018/19 season (0.17 Burbot/net-d; Figure 4.6b) was 59% lower than CPUE observed in the record high 2014/15 season (0.292 Burbot/net-d) and the second lowest CPUE on record since winter 2016/17 (0.13 Burbot/net-d). Catch rates are intended to serve as an index of Burbot abundance; however, winters 2016/17 and 2018/19 suggest that catch rates are highly dependent on climate conditions and dam operations and how they affect our ability to sample the river throughout the winter. Catch-per-unit-of-effort at the index sites has been consistently monitored from 1996-2016, and, therefore, is the most reliable metric for long-term trend interpretation. With continued releases of Burbot from the KTOI-owned hatchery, it is expected that CPUE will continue to increase in future years.

Since 2012, notable increases in catch rates from mid-February to mid-March have been observed primarily at Ambush Rock (rkm 244.5; Figures 4.7 and 4.8), Deep Creek Confluence (rkm 240), Myrtle Creek (rkm 234), and Porthill (rkm 170). Catch rates across all river sites in 2017/18 remained relatively constant (temporally) throughout the sampling period, with the exception of a substantial increase at one site in Canada (i.e., Corn Creek, rkm 150.2) and two sites in the United States (i.e., Myrtle Creek and Deep Creek confluence; Figure 4.7). Catch rates at Ambush Rock and Porthill did not increase over the sampling period like they have in previous years (Figure 4.8). Peaks in CPUE have consistently been documented between mid-February and mid-March since 2012; however, the peaks typically occurred near Ambush Rock (rkm 244.5), Deep Creek confluence (rkm 240.5), and Porthill (rkm 170). In recent years, the timing of the peaks has not changed; however, the location at which the peaks have occurred has varied. Historically, very little activity had been observed at Corn Creek (rkm 150.2). Some of the highest catch rates ever observed in the river (i.e., ≥4 Burbot/net-d) were documented at Corn Creek in winter 2017/18 (Figure 4.7). Similar comparisons were not made for winter 2018/19 since river sampling conditions were poor during the historic peak spawning period.

The age-structure of the Burbot population has been shifting (i.e., increasing in age and size) since the aquaculture program began in 2011 (Figure 4.9). The shift has been documented via fish that were PIT tagged at release and fish that were assigned to a year class via PBT. Catch composition during the 2016 season represented the most diverse age-structure observed to-date (Figure 4.9c), with representation from eight year classes from 2009-2016. It appears that the once strong 2011 year class of Burbot has begun to senesce and be replaced by younger age-classes; the shift has been observed since the 2013/14 season (Figure 4.9). Specifically, there is a significant amount of demographic momentum via the 2015 year class of Burbot (Figure 4.9c), and this became increasingly evident in the catch from both the 2017/18 and 2018/19 seasons. A total of 577 and 362 Burbot from the 2015 year class were captured during the 2017/18 and 2018/19 seasons, respectively, which is approximately 49% and 66% of the total catch for the two seasons, respectively. In addition, catch of Burbot from the 2015 year class in 2017/18 was 74% greater than the number captured in the 2016/17 season, and 17% greater in the 2018/19 season compared to the 2017/18 season.

Since the Burbot aquaculture program began, it has been clear that the primary direction of movement from stocking location has been upstream (Hardy et al. 2016). The trend has been evaluated using release information from PIT tags and PBT assignments. A similar trend was also observed during the 2017/18 and 2018/19 seasons; however, there was also a notable amount of movement downstream (Figure 4.10). Most of the downstream movement was attributable to larval Burbot that were released at Ambush Rock and the Boundary Creek confluence. It is likely that larval-released fish were carried downstream until they found refuge, which appears to have been somewhere around rkm 150.2 (Figure 4.10). A similar trend was observed with larval-released Burbot from the 2015 year class during the winter of 2016/17 (Ross et al. 2018). Perhaps of most significance, approximately 8% of the Burbot captured in the mainstem river in winter 2018/19 were originally released into Kootenay Lake as juveniles. Fish from this release group were captured at all sampling sites between rkm 150 and rkm 244.5 (Figure 4.10b). This marks the first documentation of such notable movements of Burbot from Kootenay Lake into the river.

Winter 2017/18 was the first season during which confirmed, naturally produced Burbot were captured in the Kootenai River; winter of 2018/19 marked the second documentation. Fifty-seven Burbot were captured in 2017/18 that were designated as unknown origin since they did not have a PIT tag and did not assign to the PBT baseline. Based on select genetic and biological criteria, 15 of the 57 unknown origin Burbot were confirmed to be from wild production. Further genetic evaluations indicated that 7/15 were of a genetic origin that mirrored the Remnant Kootenai River population (i.e., historical population), 2/15 were of Kootenay Lake genetics, 5/15 were of Moyie Lake genetics, and one remained unknown. It is important to note that this was the first documented reproduction of hatchery-origin Burbot in the river since the implementation of the Burbot restoration program. Similar results were found in winter 2018/19. Seventeen unknown origin Burbot were confirmed to be from wild production using similar genetic evaluations as in 2017/18. Of these, two remained of unknown origin, 4/17 from the Remnant Kootenai River population, and 11/17 from the Moyie Lake genetics. We anticipate these numbers to continue to grow in subsequent years as natural production continues to improve.

Tributary Use by Burbot in the Kootenai Basin

Between January 13, 2018 and March 22, 2018, 38 unique Burbot were detected crossing the Smith Creek PIT tag array (Figure 4.11). During roughly the same period, six individual Burbot were detected crossing the Boundary Creek PIT tag array and one crossing the Ball Creek array. Unfortunately, the Deep Creek array was not operational during winter 2017/18, so no data exist for Deep Creek. Similar to the use of Deep Creek by Burbot in previous years (Ross et al. 2018), Burbot in Smith, Boundary, and Ball creeks were only detected during the months of January, February, and March, with the largest spike in detections occurring between mid-February and mid-March. This information, in combination with previous years of data from Deep Creek, further suggest that Burbot are likely using tributary habitats to spawn. Interestingly, detected fish represented year classes from 2011-2015, and release locations from as far upstream as Ambush Rock, and as far downstream as the Goat River. Substantial use of Deep Creek by Burbot during the spawning season has been consistently documented since the winter of 2014/15. Documentation of Burbot using additional tributaries in the Kootenai Basin further supports the notion that Burbot are pioneering into diverse tributary habitats during the spawning season – a trait that bodes well for the potential of future gains in natural reproduction.

Although the PIT tag array in Deep Creek remained operational for the duration of the spawning season in winter 2018/19, no Burbot were detected crossing the array. River flows during winter 2018/19 were unusually low and ice cover was unusually prevalent and thick, so we speculate that access to tributaries for spawning Burbot was minimal or nonexistent during winter 2018/19, which would explain the lack of use documented in Deep Creek.

Potential Effects of Water Temperature on the Egg Hatching Success, Larval Development, and Larval Survival of Burbot

This study was completed in summer 2018. A PhD dissertation and multiple peer-reviewed manuscripts are currently underway and will be ready in 2019-2020. The first of several publications is complete and can be found in Ashton et al. (2019).

Burbot Population Model: Population Projections and Harvest Simulations

One-way sensitivity analyses with the DDF in effect and not in effect both indicated that the same model parameters were influential in understanding population trends of Burbot in the Kootenai River. Both analyses revealed that age-specific survival (i.e., age-0 through age-5+), fishing mortality, hatchery age-0 release number, initial number of age-1 fish, and hatchery larval survival most affected model results (Figure 4.12). Not surprisingly, when the DDF was in effect, it also ranked in the top ten parameters that most affected model results as it functionally replaced age-0 survival (Figure 4.12b).

When the DDF was not in effect and all other parameters were defined at the values in Table 4.2, model results revealed important insights about population growth and potential responses to fishing mortality. First, in the absence of both the DDF and fishing mortality (scenario 1), the population experienced rapid growth within the first ten years (i.e., beginning in 2016). The restoration target of 17,500 adults in the river was reached in less than five years (i.e., by 2019), and the adult population stabilized at approximately 50,000 individuals (Figure 4.13a). In the absence of the DDF but experiencing 15% annual fishing mortality (scenario 2), the population still experienced rapid growth within the first ten years; however, the addition of fishing mortality resulted in the abundance of adults stabilizing at approximately 16,000-18,000 individuals. The population target of 17,500 adults was still met by 2019, but the purported excess of fish under scenario 1 (Figure 4.13a) was allotted to fishing mortality. Under scenario 2, the total abundance (i.e., all age-classes) of Burbot stabilized at approximately 48,000 fish (Figure 4.13 a), and the 15% annual fishing mortality resulted in approximately 5,000-10,000 Burbot being harvested, annually (Figure 4.13b).

When the DDF was in effect at the threshold of 17,500 age-4+ adults in the river while all other parameter values were defined at the values in Table 4.2, model results predictably varied from those portrayed in scenarios 1 and 2. With the DDF in effect but in the absence of fishing mortality (scenario 3), the population experienced rapid initial growth very similar to that observed in scenario 1 – reaching the restoration target of 17,500 adults by 2019. However, unlike scenario 1, the population stabilized at approximately 30,000 adults (Figure 4.15a), rather than 50,000. With the introduction of 10% annual fishing mortality while experiencing effects of the DDF at the 17,500 adult threshold (scenario 4), population response(s) closely mirrored those from scenario 2. After reaching adult restoration targets by 2019, the population quickly stabilized at approximately 16,000-18,000 adults (i.e., within range of the restoration targets [Figure 4.15b]). The total abundance of Burbot (i.e., all age-classes) ultimately stabilized at approximately 33,000 individuals (Figure 4.16a), and the 10% annual fishing mortality translated to approximately 2,000-5,000 Burbot being harvested, annually (Figure 4.16b). In summary, all project collaborators agreed that the fishery was sustainable and that the appropriate threshold for annual exploitation was 15%. There was also agreement that the existing monitoring program is sufficient to detect density dependent effects should they occur as the population continues to increase. Any evidence of density dependence could lead to changes in stocking strategies, harvest management, or both.

One-way sensitivity analyses indicated that release numbers of age-0 Burbot greatly affected adult abundance (Figure 4.12). Figure 4.17 provides a visual representation of the relationship among adult abundance, age-0 release numbers, and annual fishing mortality with (Figure 4.17b) and without (Figure 4.17a) the DDF in effect. When the DDF was not in effect and an average of 125,000 age-0 juveniles were released from the hatchery, the population withstood approximately 15% annual fishing mortality, which was corroborated by results depicted in Figure 4.13b. When annual releases of age-0 juveniles averaged approximately 250,000/year, the population withstood an additional 5% annual fishing mortality (i.e., 20% total annual fishing mortality). Similarly, with the DDF in effect at the 17,500 threshold and annual releases averaging 125,000 age-0 juveniles/year, the population withstood approximately 10% annual fishing mortality; results that mirror those are depicted in Figure 4.15b. With an increase of annual release numbers to 250,000 age-0 juveniles, the population withstood an additional 5% annual fishing mortality (i.e., 15% total annual fishing mortality).

Burbot Fishery

After losing Burbot fishing opportunity nearly 30 years ago, resident and non-resident anglers enjoyed a successful first winter of fishing for Burbot. Creel data collected with the trail cameras were not fully analyzed by the time of this report. As such, reported results cameras are based only on the stratified creel survey. Creel survey results indicated approximately 3,200 hours of angler effort on the Kootenai River and its tributaries during winter 2019. The majority of the effort was recorded in stratum 1 (US Highway 95 bridge to the Deep Creek boat launch) relative to other strata. In addition, the vast majority of effort (i.e., 86%) was documented in the mainstem Kootenai River relative to its tributaries (i.e., 14% in Deep Creek). A total of 599 anglers were interviewed during winter 2019, 93% of whom were Idaho residents and 7% of whom were nonresidents. Estimated catch rates total catch for all fish species were low (Table 4.3); however, catch rates of Burbot (i.e., # Burbot/hour) were the highest among all species (i.e., 0.03

Burbot/hour; Table 4.3). Total estimated catch of White Sturgeon (i.e., estimated # captured) was the highest among all species, followed by Burbot (Table 4.3). Despite low catch rates, anglers reported above average satisfaction (6.5/10) with the Burbot fishery. During the course of the survey, 18 Burbot were observed in the creel. Of these, 89% were captured in stratum 1 and 11% were captured in stratum 2. Ninety-five percent of the harvested Burbot were from the 2015 year class, and 5% were from the 2013 year class. Sixty-one percent of the harvested Burbot were originally released as juveniles into Kootenay Lake. The remainder were released as juveniles at Porthill (17%), Deep Creek confluence (5.5%), and Ferry Island (5.5%) and as larvae at Ambush Rock and the Boundary Creek confluence (11%).

Annual exploitation estimates derived from the creel survey and the “Tag You’re It!” program varied substantially. According to the Burbot population model, there were an estimated 61,000 Burbot across all age-classes alive and in the Kootenai River basin during winter 2019, which equated to a calculated annual exploitation rate of less than 1%. Results from the “Tag You’re It!” exploitation study estimated annual exploitation at 6.7%, which equated to a calculated harvest number of approximately 4,100 Burbot. Of the 179 non-reward tags deployed in December 2018, five were reported; one of the fifteen deployed reward tags were reported. These data equated to an estimated non-reward tag reporting rate of 45%.

DISCUSSION

Although the Burbot population in the Kootenai River was considered functionally extirpated by the early 2000s, the population trend has largely reversed, and current understanding of factors potentially limiting natural recruitment has substantially grown. There was some concern over the decline in hoop net catch rates in winter 2016/17; however, as expected, catch rates in winter 2017/18 rebounded to the second highest on record since 1993. It is becoming increasingly clear that season-wide catch rates are largely driven by climate conditions and dam operations which affect our ability to conduct annual hoopnetting surveys. Although catch rates of Burbot have substantially increased relative to what they were in the mid-1990s, the Kootenai River population remains low in abundance relative to other Burbot populations. For comparison, catch rates of Burbot in Moyie Lake, B.C. were 0.5 to 2.2 fish/net-d (Prince 2007), in the Chena and Tanana rivers of Alaska were 0.9 and 1.2 fish/net-d, respectively (Evenson 1993), and in four Alaskan Lakes ranged from 0.5-3.0 fish/net-d (Parker et al. 1988). However, population projections for the Kootenai River suggest Burbot abundance will continue to increase rapidly through at least 2025, and we expect our sampling CPUE to improve, as well. Multiple lines of evidence since 2011 suggest that adult Burbot attempted to spawn in the Kootenai River and its tributaries on an annual basis. For example, increases in catch rates have been documented at Ambush Rock, Deep Creek, Myrtle Creek, Corn Creek, and Porthill during mid-February, and adult Burbot have been documented migrating into Deep, Boundary, Smith, and Ball creeks during mid-February. Although few in number, the confirmed documentation of five wild Burbot produced from sexually mature hatchery Burbot in 2018/19 and eleven in 2018/19 is a landmark for the restoration program. With adult numbers increasing, it is likely that natural reproduction will become more common in future years.

Age-structure of Burbot captured in winters 2017/18 and 2018/19 remained diverse, with representation from eight year classes and a wild component in both seasons. These seasons were not, however, quite as diverse as the catch during the winter 2016/17 season. The 2015 year class was the largest juvenile cohort ever released by KTOI; nearly 275,000 juveniles and 650,000 planktivorous feeding larvae were released. Fish from the 2015 year class fully recruited to hoopnets in 2017/18, and the 2015 year class clearly dominated catch for both the 2017/18 and 2018/19 seasons. The diversity in age and size class composition and the significant demographic momentum from the 2015 cohort are all positive signals that are indicative of a healthy and robust population. This finding has important implications for both population recovery and fishery implementation and performance. First, it is well established that diverse age structure is desirable and beneficial from a reproductive perspective. More specifically, studies have suggested that across many fish species, different age cohorts may spawn at different times and locations in a given system (Berkeley et al. 2004; Hixon et al. 2013), which may ensure that there is at least some reproductive success within a given year. In fact, Ashton et al. (2019) reported that younger (i.e., age-1) and older (i.e., age-7+) female Burbot spawned, on average, 4-12 days later than age-5 and age-6 females. Since harvest of Burbot in the Kootenai River has been closed since 1992, it is unknown whether fishing mortality will be skewed toward younger (smaller) fish, older (larger) fish, or not skewed at all. The diverse age and size structure of Burbot in the Kootenai River allows for increased population resiliency to harvest, regardless of the age or size selectivity in angler catch. Lastly, based on results from the creel survey, the demographic momentum of the 2015 year class provided increased opportunities for angler success once the fishery was opened (i.e., 95% of physically creeled Burbot were from the 2015 year class).

Recaptures of larval-released Burbot continued to reveal important insights that had a variety of implications for hatchery production and operations, as well as future research, monitoring, and evaluation. Much of the Burbot research, monitoring, and evaluation work in the Kootenai River has focused on identifying the cause of and life-stage at which recruitment failure has been occurring. Prior to recapturing larval-released Burbot in winter 2016/17, it was well established that juvenile (i.e., six-month-old fingerling) Burbot released at different locations in the Kootenai River survived to sexual maturity. With the recapture of larval-released Burbot in winters 2016/17, 2017/18, and 2018/19, the recruitment failure window was significantly narrowed to occurring sometime prior to feeding on zooplankton (i.e., egg incubation or early feeding). This finding has instigated collaborative research to further investigate specific early life stages in both lab and field settings. More specifically, over 2,500,000 pre-feeding larvae were released at Ferry Island and in a naturally inundated floodplain habitat (i.e., Nimz Ranch) in spring 2018. An additional 2,300,000 planktivorous feeding larvae were released in the same locations later in spring 2018. Thanks to the sampling efforts of KTOI staff, the recapture of both pre- and planktivorous-feeding larvae that were released into Nimz Ranch filled a significant knowledge gap in the Kootenai River Burbot recovery program: newly-hatched (i.e., pre-feeding) larval Burbot can successfully feed and survive in off-channel, floodplain habitats. Conspecifics from the same experimental releases in the mainstem Kootenai River have yet to be encountered. It is currently unknown exactly why pre-feeding larval Burbot can survive in an off-channel floodplain habitat and not the mainstem river, but it is hypothesized that both abiotic (i.e., water temperature) and biotic (i.e., prey densities) conditions vastly differ between off-channel and mainstem river habitats, which very likely affect survival probabilities of pre-feeding larval Burbot.

The apparent fitness advantage (i.e., based on weight and total length) of both pre-feeding versus planktivorous-feeding and planktivorous-feeding versus juvenile Burbot is striking and raises important considerations for hatchery operations and production. The apparent fitness advantage does not appear to be limited to the first year-at-large; rather, it appears to be sustained into the second and third years-at-large. Significant resources (e.g., time, money, and effort) are expended to raise Burbot from eggs to six-month-old juveniles in the hatchery. It may be worthwhile for the KTOI hatchery to consider stocking planktivorous-feeding larval Burbot in the future; the decision may not only save resources, but it appears it may also result in a more robust end product in the river. In order for operational changes to be implemented, KTOI needed a concrete survival estimate for Burbot released as planktivorous feeding larvae. Upon estimating planktivorous-feeding larval survival (i.e., 0.03%), initial conversations began about shifts in hatchery stocking strategies to incorporate planktivorous-feeding larval releases for the purpose of year class production rather than simply research purposes. Researchers and managers are hopeful the survival estimate can be further refined in the future to better inform implementation for the purposes of year class production. Currently, in order to reach adult objectives for the Burbot recovery program, the KTOI hatchery target for juvenile releases is 125,000. Juvenile Burbot are known to survive at approximately 10% for their first year-at-large (Ross et al. 2018), which results in 12,500 age-1 Burbot. Under this scenario and utilizing the new larval survival estimate, the KTOI hatchery could release 4,200,000 planktivorous-feeding larval Burbot and still achieve the age-1 objective while holding Burbot in the hatchery approximately 40% fewer days.

During winter 2016/17, over 60% of the larval released Burbot from the 2015 year class that were recaptured in hoopnets were found near the confluence of Deep Creek and Corn Creek (Ross et al. 2018). A similar trend was observed in the catch of winters 2017/18 and 2018/19. It is unknown what drove the observed trend, but it is possible that Deep Creek and Corn Creek provide viable prey sources for larval Burbot residing in the river. Interestingly, data from 2018/19 in particular indicated that larval Burbot released at both Ambush Rock and Boundary Creek widely dispersed throughout most stretches of the river (i.e., rkm 144.5-234.5), in both upstream and downstream directions. Additional effort should be placed on PBT-based larval evaluations in the Corn Creek and Deep Creek drainages, as these tributaries (and others) may provide spawning opportunities not affected by conditions in the mainstem Kootenai River.

Adult Burbot passed over the PIT tag array in Deep Creek during the spawning season for three consecutive winters (i.e., 2014-2016). The observed activity in Deep Creek prompted researchers to investigate whether or not Burbot were using other tributaries in the basin. The discovery of Burbot using Deep, Ball, Boundary, and Smith creeks has important implications for recruitment bottleneck(s) and future research, monitoring, and evaluation. The majority of the adult fish passing over PIT tag arrays in these tributaries were stocked at mainstem Kootenai River locations as far downstream as the Goat River (rkm 152) and as far upstream as the Moyie River (rkm 259). Furthermore, they represented nearly all year classes and ages-at-release, to-date. Detections of adult Burbot in the four studied tributaries indicate that hatchery Burbot have the ability to pioneer into novel habitats and tributaries, presumably in search of suitable spawning habitats. Some Burbot populations use tributary habitats for spawning (Arndt and Hutchinson 2000), and it appears the Kootenai River population may do the same. The thermal, nutrient, and hydrologic regimes of the Kootenai River are heavily altered, so spawning attempts by Burbot in tributaries may afford spawning conditions that are more conducive to successful recruitment than the mainstem Kootenai River. Furthermore, ice cover is a common denominator among thriving, naturally reproducing Burbot populations (McPhail and Paragamian 2000). Such ice cover was once common in the Kootenai River; however, ice rarely forms for extended periods of time since the completion of Libby Dam. Conversely, extensive ice cover forms most winters in tributaries to the Kootenai River, including Deep, Boundary, Ball, and Smith creeks and the Goat River, which could potentially bolster the chances of natural recruitment. Additional research (i.e., eDNA methods) should be conducted to evaluate the extent of tributary use in the Kootenai River basin.

In general, Burbot are an under-studied fish species (McPhail and Paragamian 2000), so it stands to reason that very little information exists on general population dynamics, particularly relative to fishing mortality. Analyses outlined in Ross et al. (2018) offered age-specific survival estimates from Burbot ages 0-5+; such estimates did not exist in the literature, and, thus, provided valuable baseline information for Burbot biology. In addition, the planktivorous-feeding larval survival estimate is the first and only estimate of Burbot survival at that life stage, to-date. Although the Burbot population model developed for the present study was similar to the one developed by Klein et al. (2016), the purpose of the analysis conducted by Klein at al. (2016) was to determine whether or not the Burbot population in the Green River, Wyoming could be effectively suppressed via harvest mortality. Conversely, the objectives of the population model in the study reported herein were to project future population trends and determine how fishing mortality and other factors interacted with population trajectories and targets. Results from various harvest scenarios suggested that the Kootenai River Burbot population could withstand approximately 10-15% annual fishing mortality, which equated to approximately 2,500-10,000 individual Burbot harvested annually. Based on model projections and expected fishing mortality rates, it was hypothesized that the Burbot population would continue to grow despite the onset of some level of fishing mortality. Results from the creel survey on the Kootenai River conducted during winter 2018/19 confirmed that current harvest mortality is substantially below the 10-15% threshold identified in the population model. Harvest monitoring, paired with ongoing population monitoring, will ensure that harvest rates in Idaho remain compatible with overall abundance goals.

Exploitation estimates from the creel survey and "Tag You're It!" program vastly differed (i.e., 0.13% and 6.7%, respectively). Although the specific mechanism for the difference is not entirely understood, there are several probable explanations. First, the exploitation estimate calculated from the creel survey assumed that all 61,000 Burbot in the Kootenai River population were residing in the Idaho section of the river and available for harvest; this assumption was likely violated. Hoopnetting efforts in the Canadian portion of the river from December 2018-March 2019 verified ample numbers of Burbot residing in Canadian waters, making them unavailable for harvest by anglers in Idaho. Future research should focus on estimating the number of Burbot residing in Idaho during the spawning season, which will further refine exploitations estimated and associated interpretations. Second, data from previous years indicated that once Burbot arrived at spawning locations, they generally stayed until the end of the spawning season. Many of our historic hoopnet sites are at known spawning locations with easy shoreline access for anglers, which resulted in many of the "Tag You're It!" tags being deployed in these same areas. It is possible that this resulted in specific (spawning) areas having high concentrations of exploitation tags available to anglers, which may have resulted in an over-estimate of exploitation. Regardless of the mechanism, both of the estimates were well below the 10-15% exploitation threshold identified in the population model, so there is currently no concern among managers of overharvest of Burbot in the Kootenai River in Idaho.

Catch estimates and catch rates from the creel survey offered seemingly conflicting values for Burbot and White Sturgeon. Catch rates for Burbot were the highest among all species (0.026 Burbot/hour); however, total catch of White Sturgeon was the highest among all species (226, [0-630, range]). Although these values seem to disagree with one another, they were an artifact of the analysis for the creel survey. Catch rates were not simply calculated by the season-wide total number of each species captured divided by season-wide numbers of hours fished. Rather, they were calculated each day, and then averaged over the course of the season. The elevated catch of White Sturgeon was attributable to a single day, during which a single angler reported catching 27 White Sturgeon. Naturally, the result was an elevated estimate of season-wide catch of White Sturgeon, but the season-wide catch rate was low. Conversely, anglers more consistently caught Burbot throughout the season, but there was not a single day during which an angler caught a substantial number of Burbot, which resulted in a lower catch estimate but a higher catch rate estimate than for White Sturgeon.

Projections from the population model suggested that the restoration target of 17,500 adults would be exceeded by 2019 and then continue to climb in subsequent years. That, combined with the first detections of wild produced fish (i.e., winters 2017/18 and 2018/19), are significant successes for the Burbot program. Furthermore, all lines of biological and statistical inference indicated that over-harvest in the Kootenai River was unlikely and ultimately not occurring. Year class production shifted from the UI-ARI hatchery to the KTOI hatchery beginning in 2015, and as a result of increased capacity at the new hatchery, annual year class size has grown substantially. The average release number of juveniles from 2015-2017 was approximately 169,000/year compared to approximately 32,000/year from 2011-2014. As a result, there is significant demographic momentum from the 2015-2017 year classes that will soon be entering into the spawning adult population. When Burbot from these year classes enter the adult population, there will likely be further adults in excess of the restoration targets that are available for harvest.

Future research should focus on evaluating whether density dependent effects manifest as the population continues to grow, and quantifying density effects on growth and survival. Additional research is also needed to further hone understanding of the specific life stage at which recruitment failure is occurring.

RECOMMENDATIONS

1. Work with IDFG regional fisheries management staff to intensively monitor the Burbot fishery in winter 2019/20.

2. Develop and implement a study to characterize growth of Burbot in the Kootenai River.

3. Evaluate natural production and hatchery contribution using PIT tags and PBT genetic marking. Consider specifically targeting different Kootenai River Habitat Restoration Program projects and their effect(s) on larval survival.

4. Continue sampling index locations to measure changes in abundance, survival, and size structure.

5. Develop and implement a study using eDNA to broadly characterize current use of tributaries by Burbot.

6. Refine existing survival estimates and initial population numbers to then update the Burbot population model.

7. When feasible, estimate age-0 and age-1 survival, through time, and then compare with trends in density to understand and quantify the effect, if any, of the DDF.

TABLES

Table 4.1. Total number of Burbot released from 2009-2018 into the Kootenai River and its tributaries. Fish were tagged with FDX PIT tags from 2009-2013 HDX PIT tags from 2014-2016, and via genetic tags from 2017 and on. Untagged fish from 2011-2018 will be able to have year class and gender assigned by genetic analysis, and untagged fish from 2015 and beyond will also be able to have release location and age-at-release assigned by genetic analysis. It is important to note that the number released in 2018 indicate only those fish released as planktivorous and pre-feeding larvae. Fish released as juvenile fingerlings in 2018 were not included in the total because they were released after the scope of this report.

Stock Year	Year Class	Tagged Releases	Untagged Releases	Total Release Number
2009	2006	7	-	7
	2007	23	-	23
	2008	1	-	1
	2009	-	178	178
2010	2007	5	-	5
	2008	18	-	18
	2009	551	4	555
	2010	-	1,576	1,576
2011	2009	6	26	32
	2010	30	90	120
	2011	16,289	53,975	70,264
2012	2010	82	-	82
	2011	656	-	656
	2012	3,392	268,305	271,697
2013	2011	71	-	71
	2012	600	1	601
	2013	10,011	450,872	460,883
2014	2010	16	-	16
	2012	16	-	16
	2013	218	-	218
	2014	3,473	-	3,473
2015	2014	30	-	30
	2015	9,946	895,205	905,151
2016	2016	14,618	123,618	138,236
2017	2017	3,813	7,368,453	7,368,453
2018	2018	-	4,996,711	4,996,711
Total		**63,872**	**14,159,014**	**14,222,886**

Table 4.2 Parameter names, values, variances, and sources in the age-based Leslie matrix population model. Under the source column, KR denotes Kootenai River, NA denotes not applicable (i.e., parameter was not used in any model simulations), and Est denotes expert elicitation. * denotes that full descriptions of parameter, value, variance, and source can be found in the methods.

Parameter	**Value**	**Variance**	**Source**
Age-0 survival (annual)	0.10	0.02	KR
Age-1 survival (annual)	0.95	0.04	KR
Age-2 survival (annual)	0.94	0.04	KR
Age-3 survival (annual)	0.92	0.05	KR
Age-4 survival (annual)	0.89	0.06	KR
Age-5+ survival (annual)	0.80	0.09	KR
Hatchery age-0 survival (DDF) (annual)*	--	--	Est
Hatchery larval survival (annual)	0.02	0.003	KR
Wild egg survival (annual)	0.06	0.2 of value	NA
Initial # age-0 (# fish)	11,906	0	KR
Initial # age-1 (# fish)	16,596	0	KR
Initial # age-2 (# fish)	44	0	KR
Initial # age-3 (# fish)	755	0	KR
Initial # age-4 (# fish)	967	0	KR
Initial # age-5 (# fish)	4,598	0	KR
Age-3 fecundity (# eggs/female)	0.000001	0.2 of value	NA
Age-4 fecundity (# eggs/female)	0.000001	0.2 of value	NA
Age-5+ fecundity (# eggs/female)	0.00001	0.2 of value	NA
Hatchery age-0 release # (# fish)	125,000	0.4 of value	KR
Hatchery larval release # (#fish)	0.000001	0.2 of value	NA
Fishing age target*	young, equal, or old	--	Est
Fishing mortality (annual)	0.00-1.0	0.2 of value	Est

Table 4.3 Estimated catch rates and catch from the creel survey conducted in winter 2019. Catch values denote means ± 95% credible intervals.

Fishery	Species	Catch rate (# fish/hour)	Catch (# fish)
Mainstem	Burbot	0.026	53 (8-98)
Mainstem	White Sturgeon	0.019	228 (0-630)
Mainstem	Rainbow Trout	0.008	4 (0-11)
Mainstem	Mountain Whitefish	0.007	13 (0-40)
Mainstem	Peamouth Chub	0.006	1 (0-1)
Mainstem	Northern Pikeminnow	0.004	12 (0-36)
Mainstem	Largescale Sucker	0.001	5 (5-5)

FIGURES

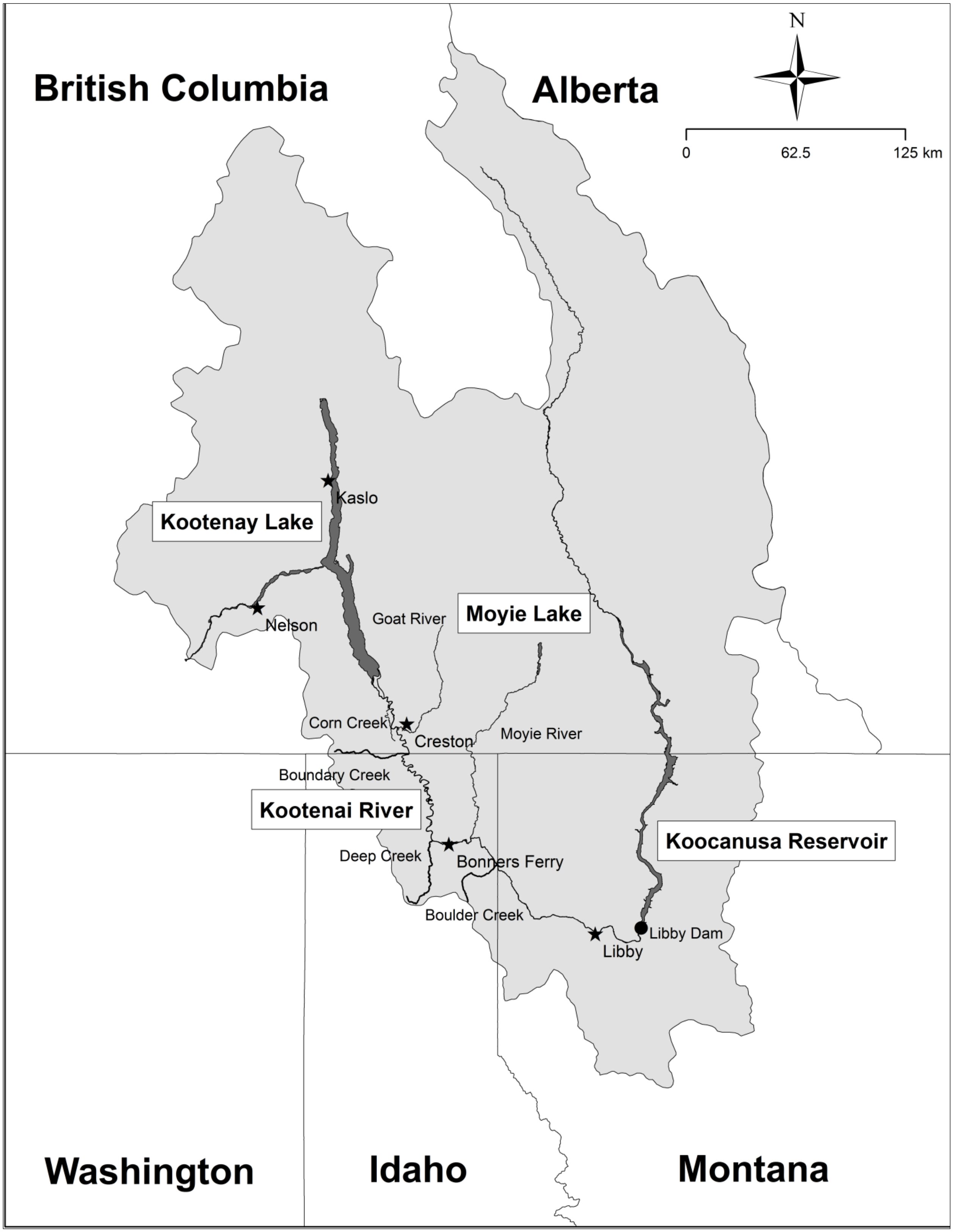

Figure 4.1. Map of the Kootenai Basin, including Kootenay Lake, the Kootenai River, Koocanusa Reservoir, Moyie Lake, and major tributaries to the Kootenai River in Idaho and British Columbia.

Figure 4.2. Map of all hoopnet locations sampled in the Kootenai River in Idaho and British Columbia during winter 2017/18.

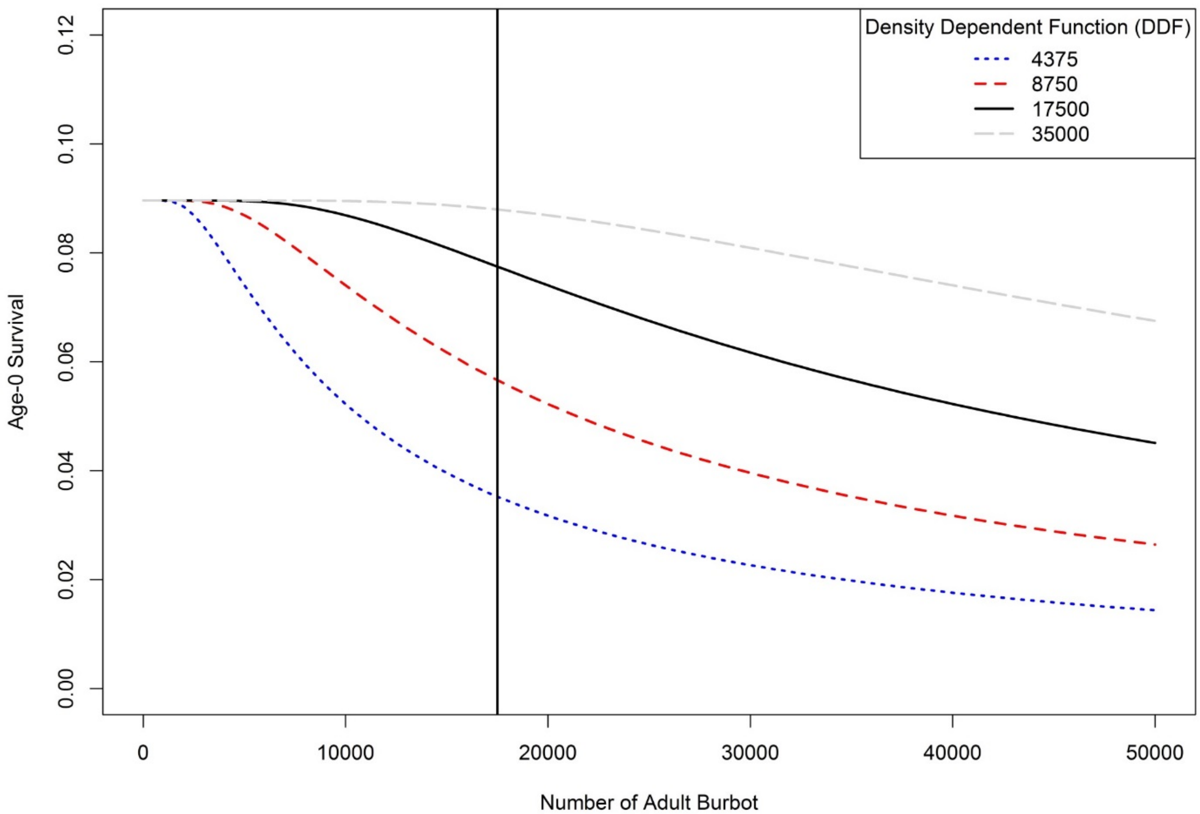

Figure 4.3. Estimated density dependent function used in the population model. The various lines represent the simulated effect that density dependence would begin enacting on survival of age-0 Burbot when the adult population reaches an abundance of 4,375 (blue), 8,750 (red), 17,500 (black), and 35,000 (gray).

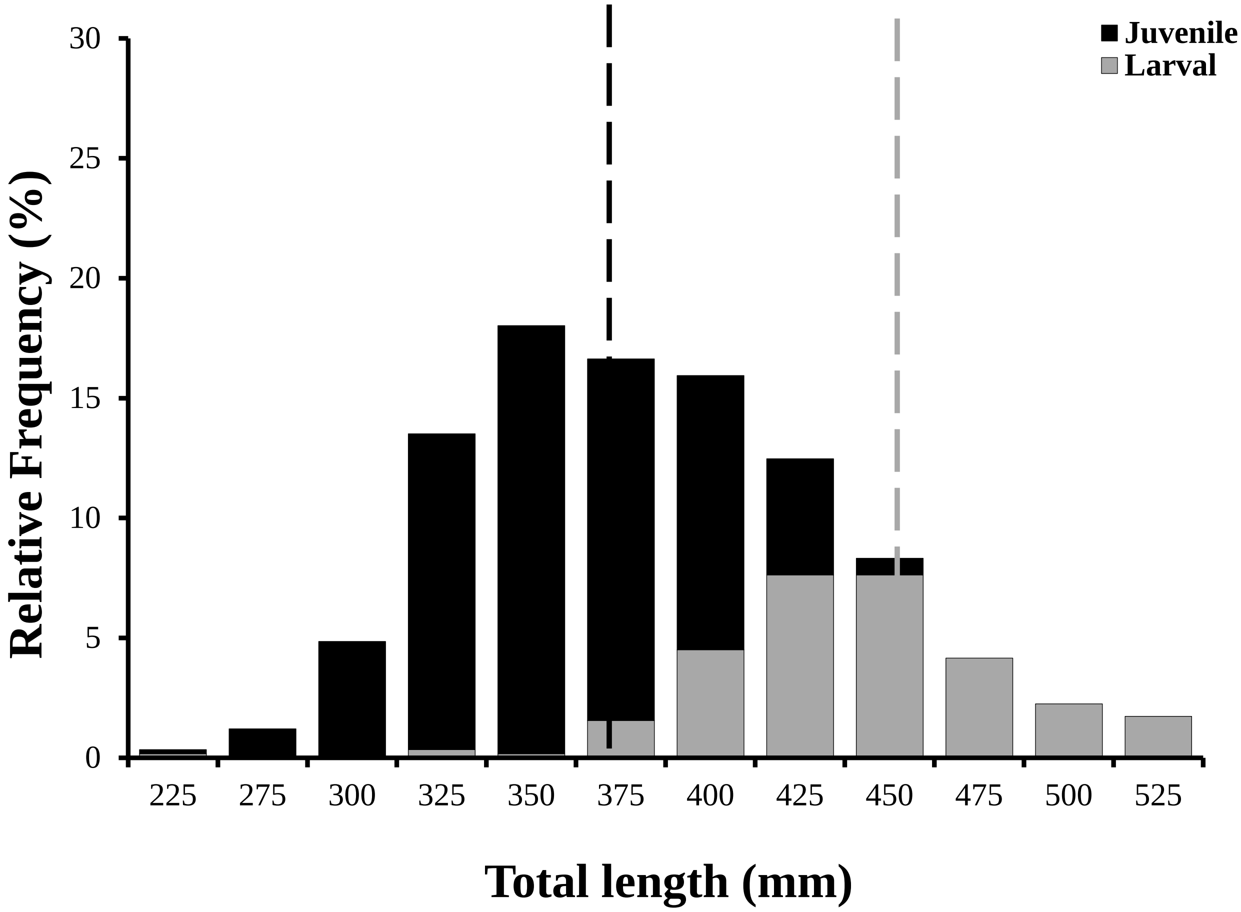

Figure 4.4. Length frequency of Burbot from the 2015 year class that were captured during 2018. Black bars denote fish that were released into the river as six-month-old juveniles (*n = 404* captures). Gray bars denote fish that were released into the river as planktivorous feeding larvae (*n = 173* captures). Dotted lines represent average lengths for each age-at-release group (by color).

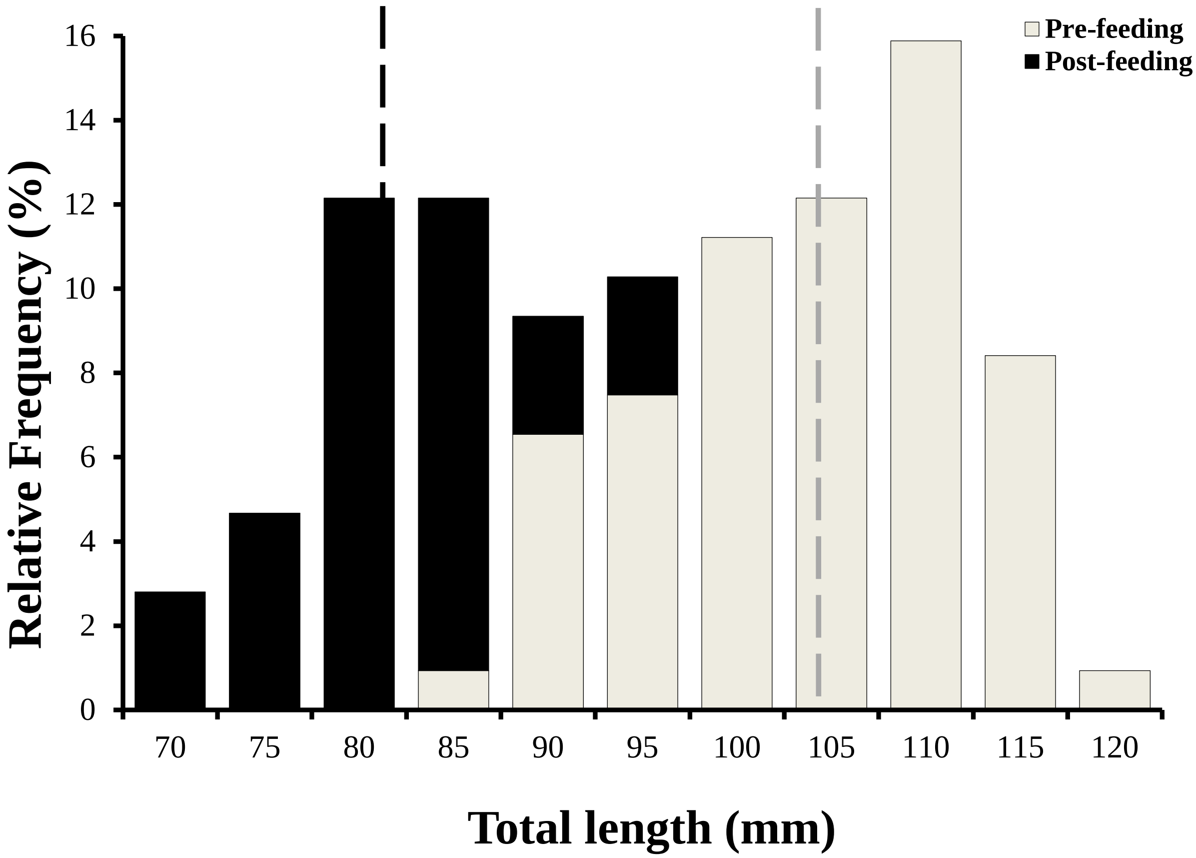

Figure 4.5. Length frequency of Burbot from the 2018 year class that were captured during August 2018 in Nimz Ranch via minnow traps. Black bars denote fish that were released into Nimz Ranch pods as planktivorous-feeding larvae (*n* = *39* captures). Gray bars denote fish that were released into Nimz Ranch as pre-feeding larvae (*n* = *68* captures). Dotted lines represent average lengths for each age-at-release group (by color).

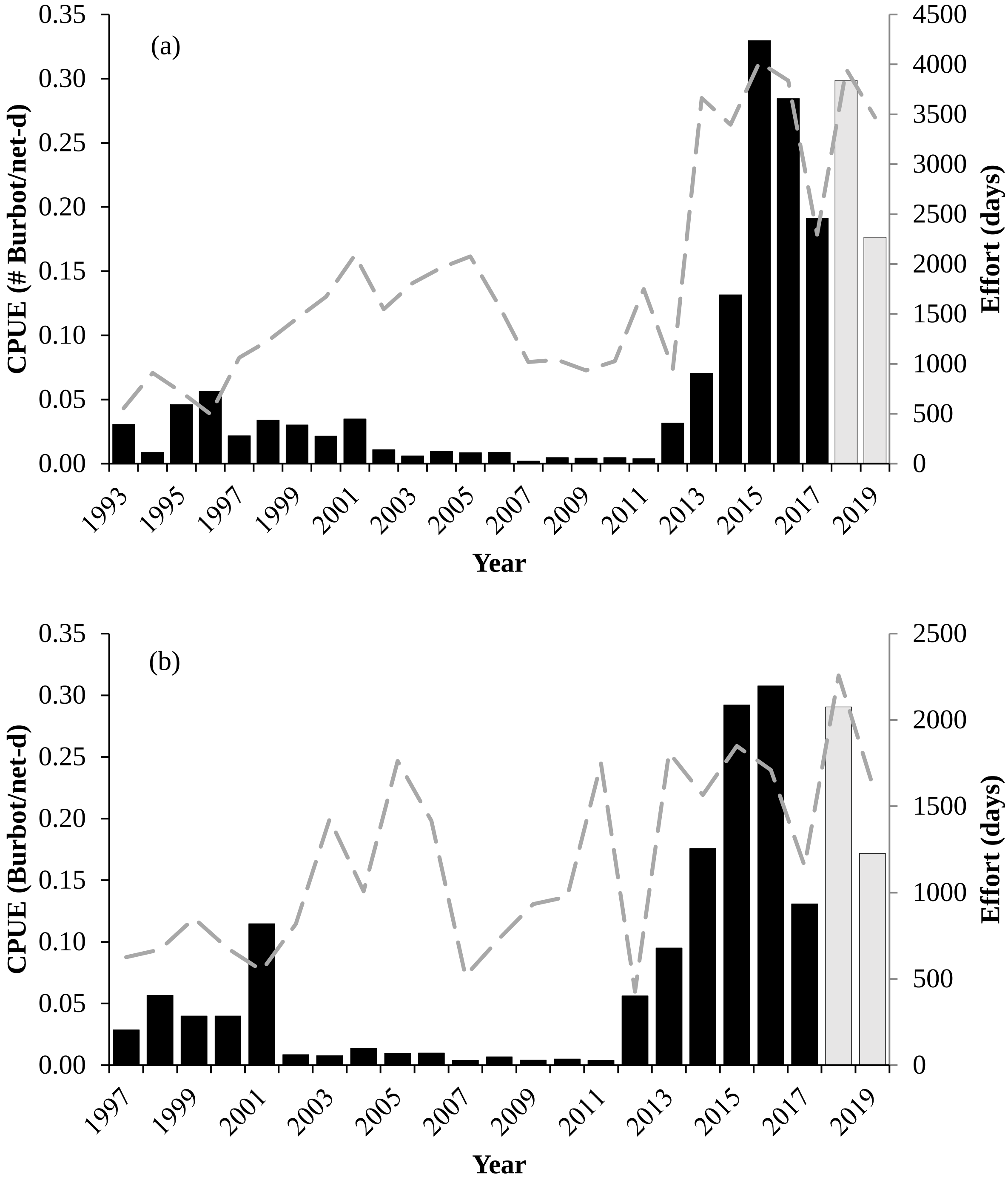

Figure 4.6. Catch-per-unit-of-effort (Burbot/net-day) of hoopnet sampling for (a) all sites and (b) index sites from 1992-2019. The gray dotted line represents hoopnetting effort (days). Data from sites upstream from Bonners Ferry are not included. Annual sampling started December 1 and ended March 31.

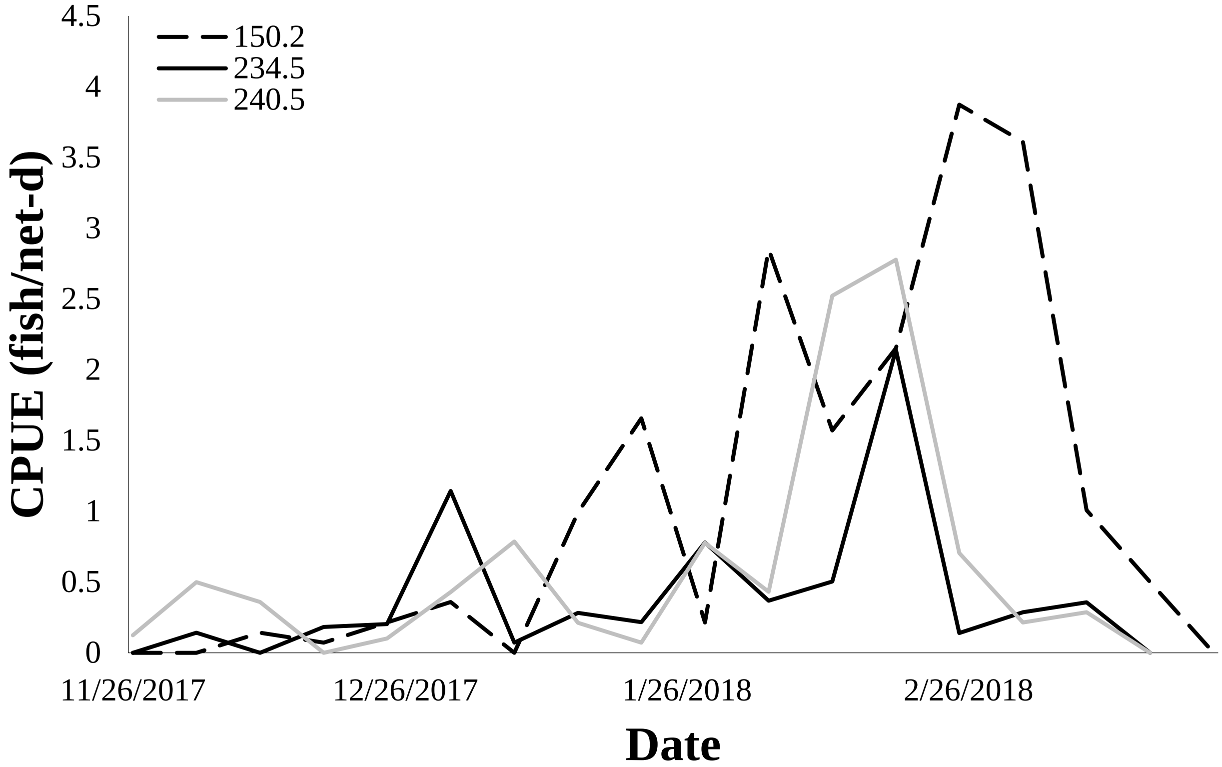

Figure 4.7. Spawn timing at Corn Creek (rkm 150.2), Myrtle Creek (rkm 234.5), and Deep Creek confluence (rkm 240.5) as gauged by CPUE (Burbot/net-day).

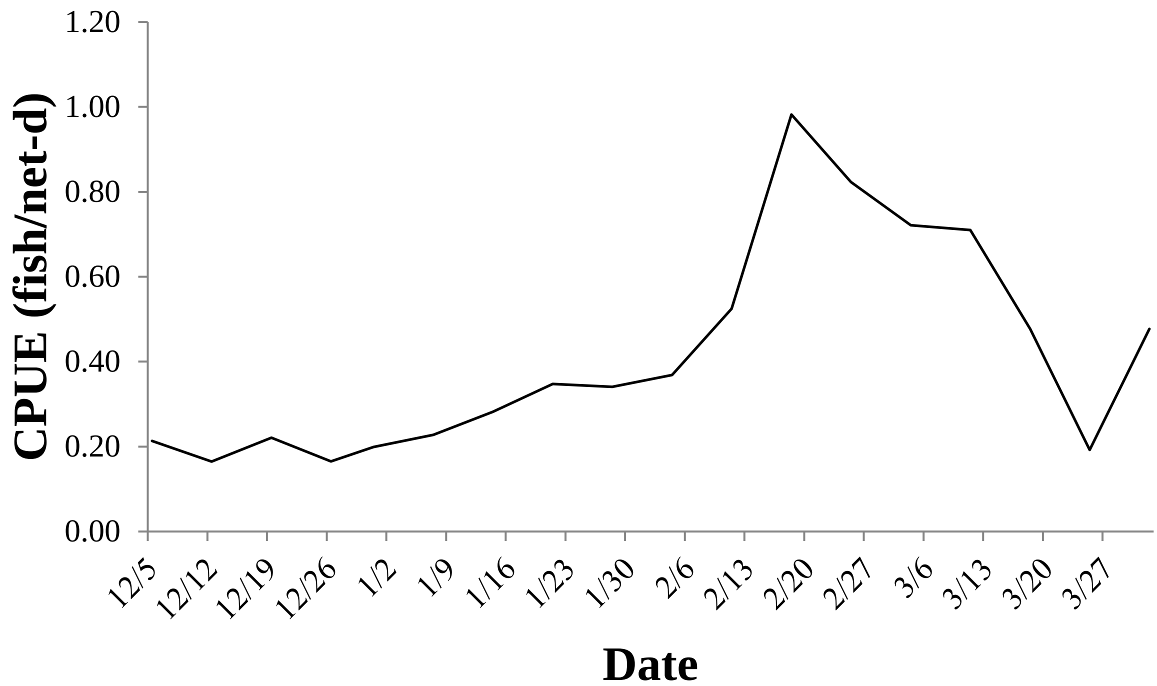

Figure 4.8. Mean spawn timing of Burbot captured at Ambush Rock (rkm 244.5; historical index location) as gauged by CPUE (Burbot/net-day). Data shown represent all years from 2012-2018.

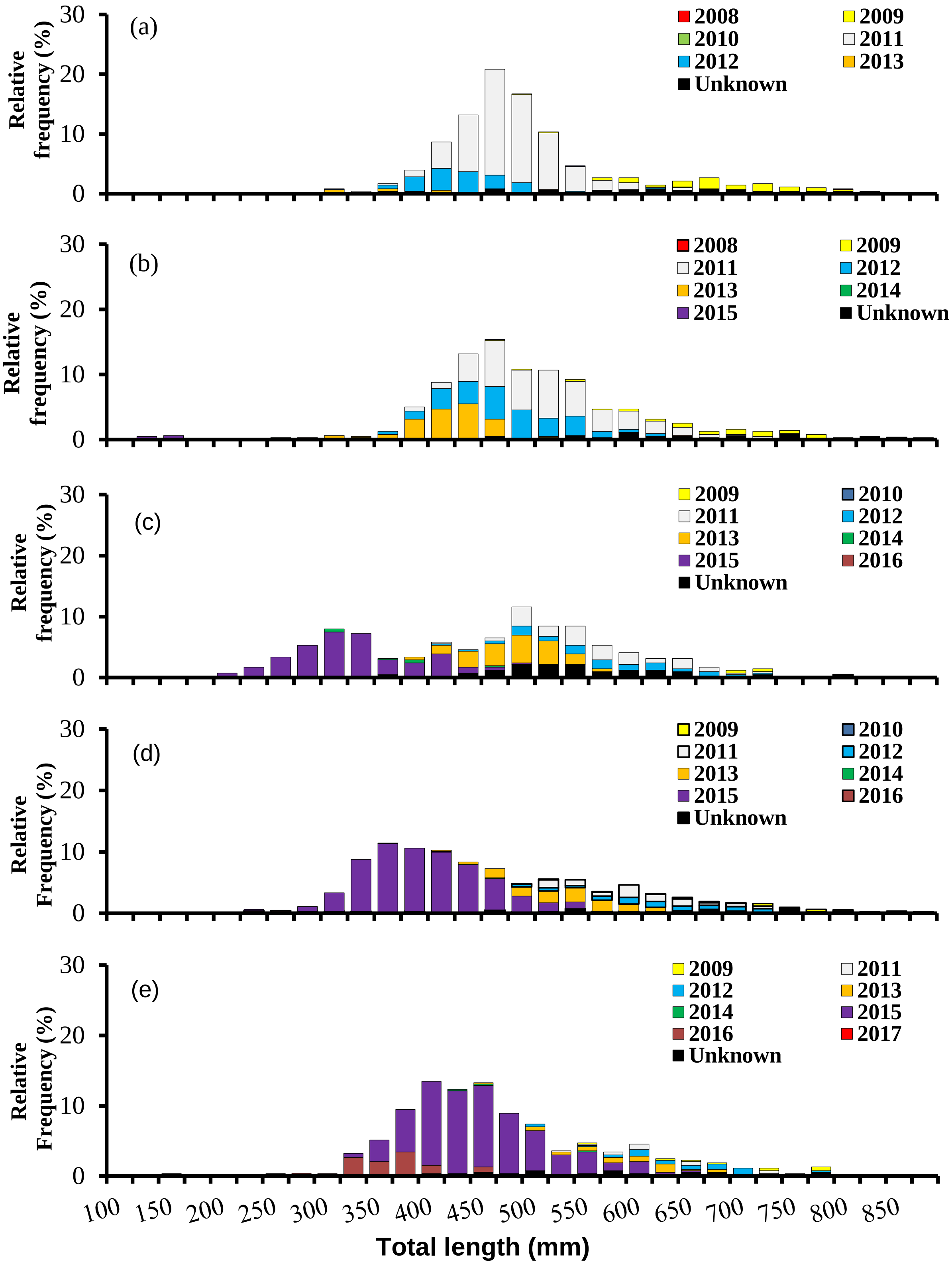

Figure 4.9. Length frequency and year class assignments from PIT-tagged and PBT-assigned Burbot captured in hoopnets in the Kootenai River from December 1 through March 31 during (a) 2015, (b) 2016, (c) 2017, (d) 2018, and (e) 2019.

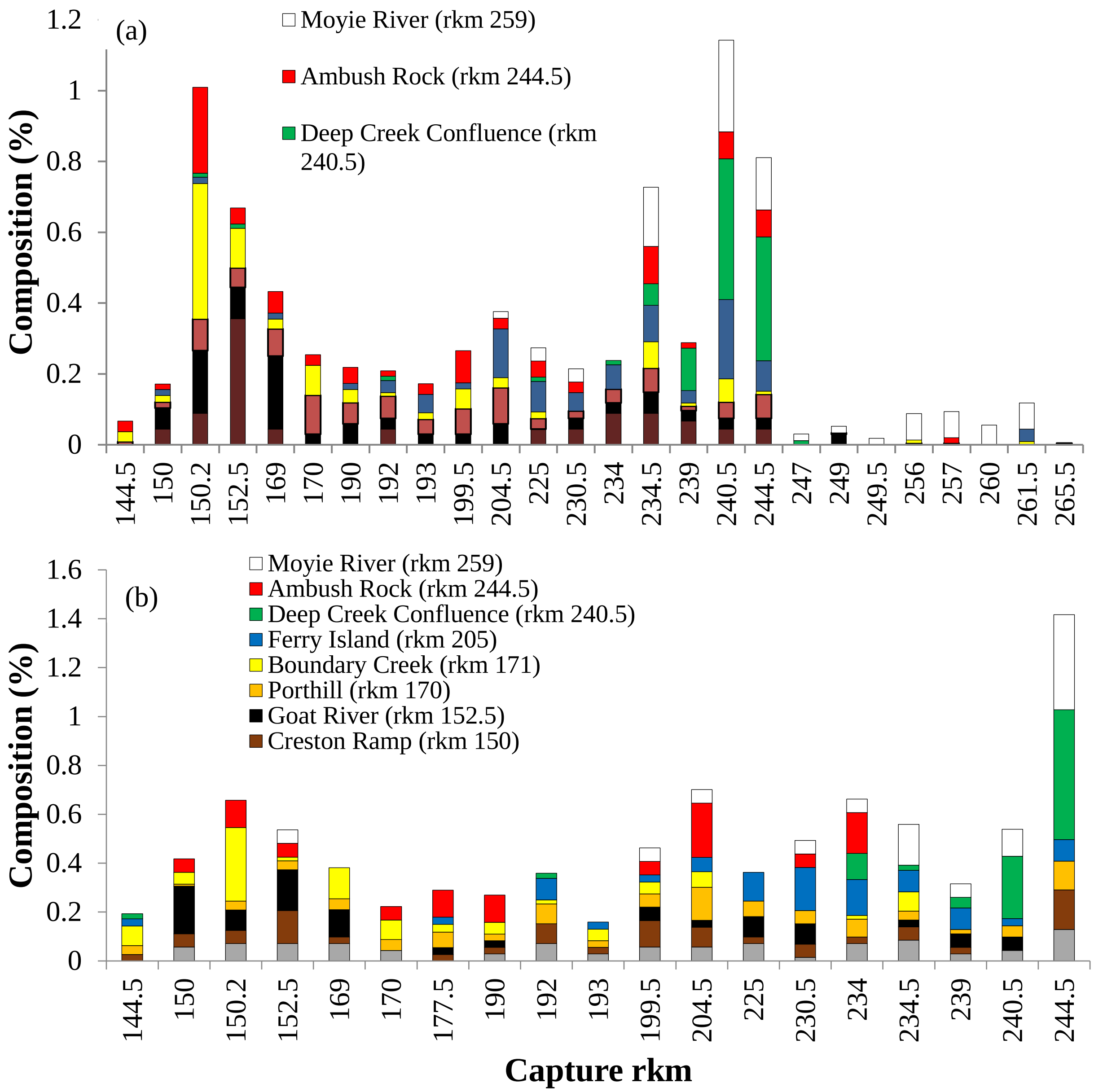

Figure 4.10. Proportion of Burbot recaptured in the (a) 2017/18 and (b) 2018/19 winter hoopnet sampling seasons relative to their original release location. Note that the rkms sampled differ between the two sampling seasons.

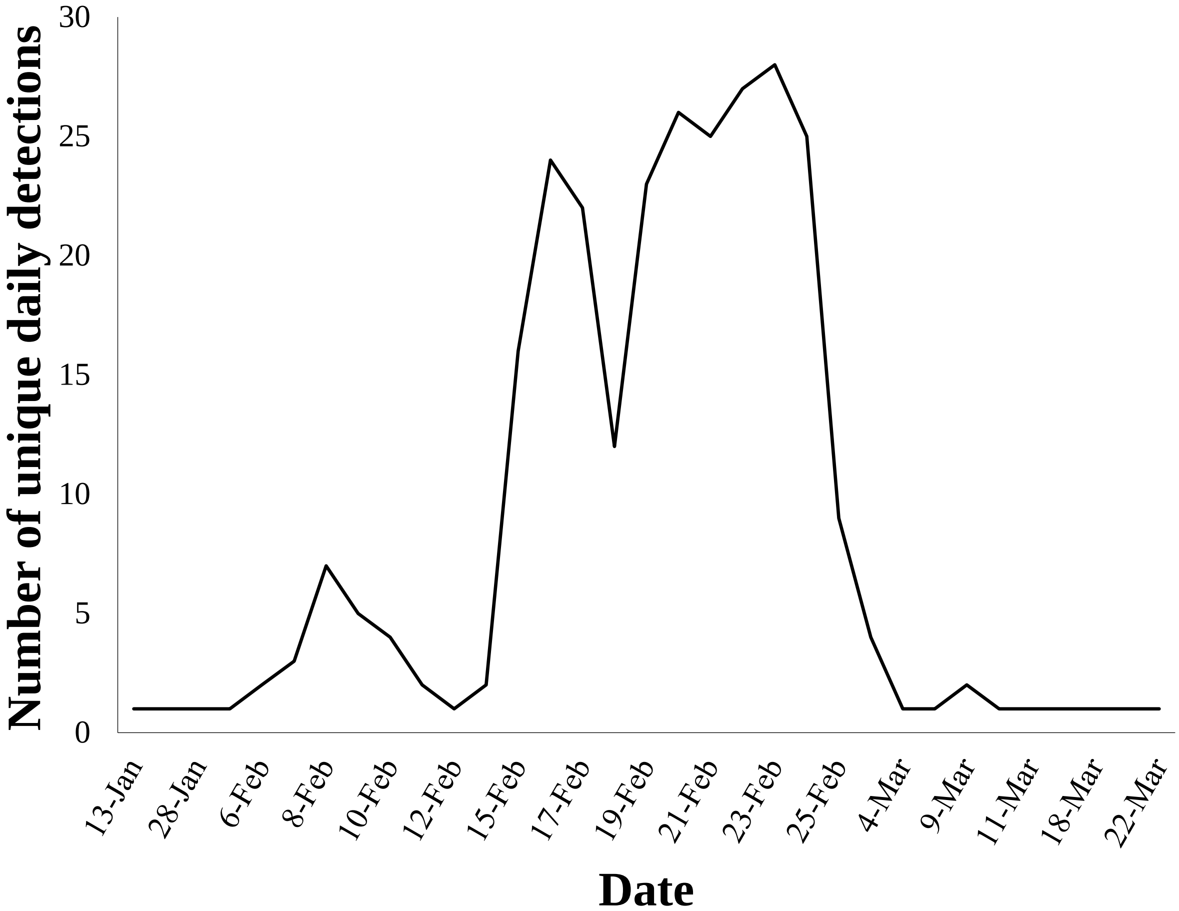

Figure 4.11. Number of unique daily detections at the Smith Creek PIT-tag array during the 2018 winter season.

(a)

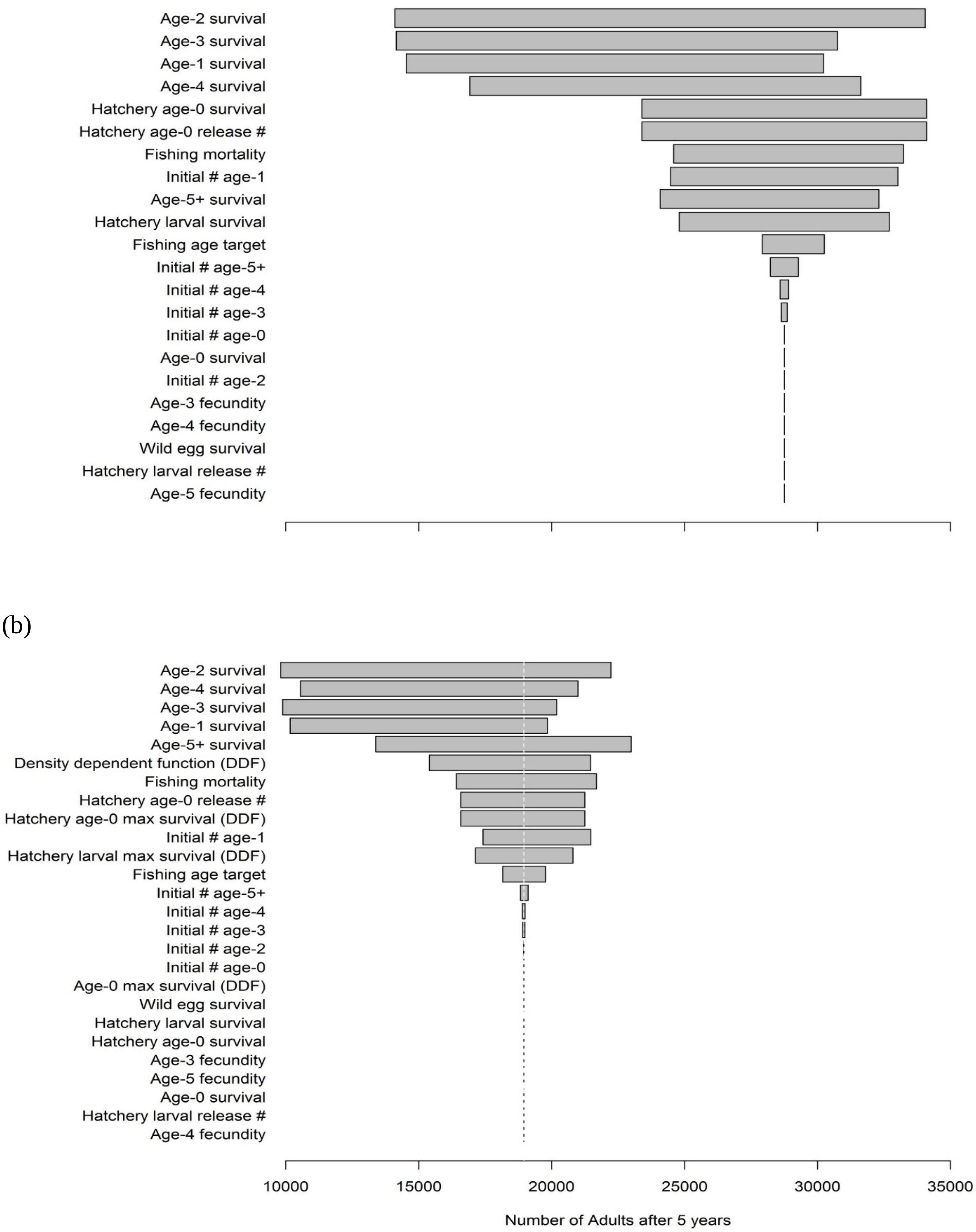

Figure 4.12. Results from one-way sensitivity analyses (a) without the DDF in effect and (b) with the DDF in effect at the 17,500 adult abundance threshold.

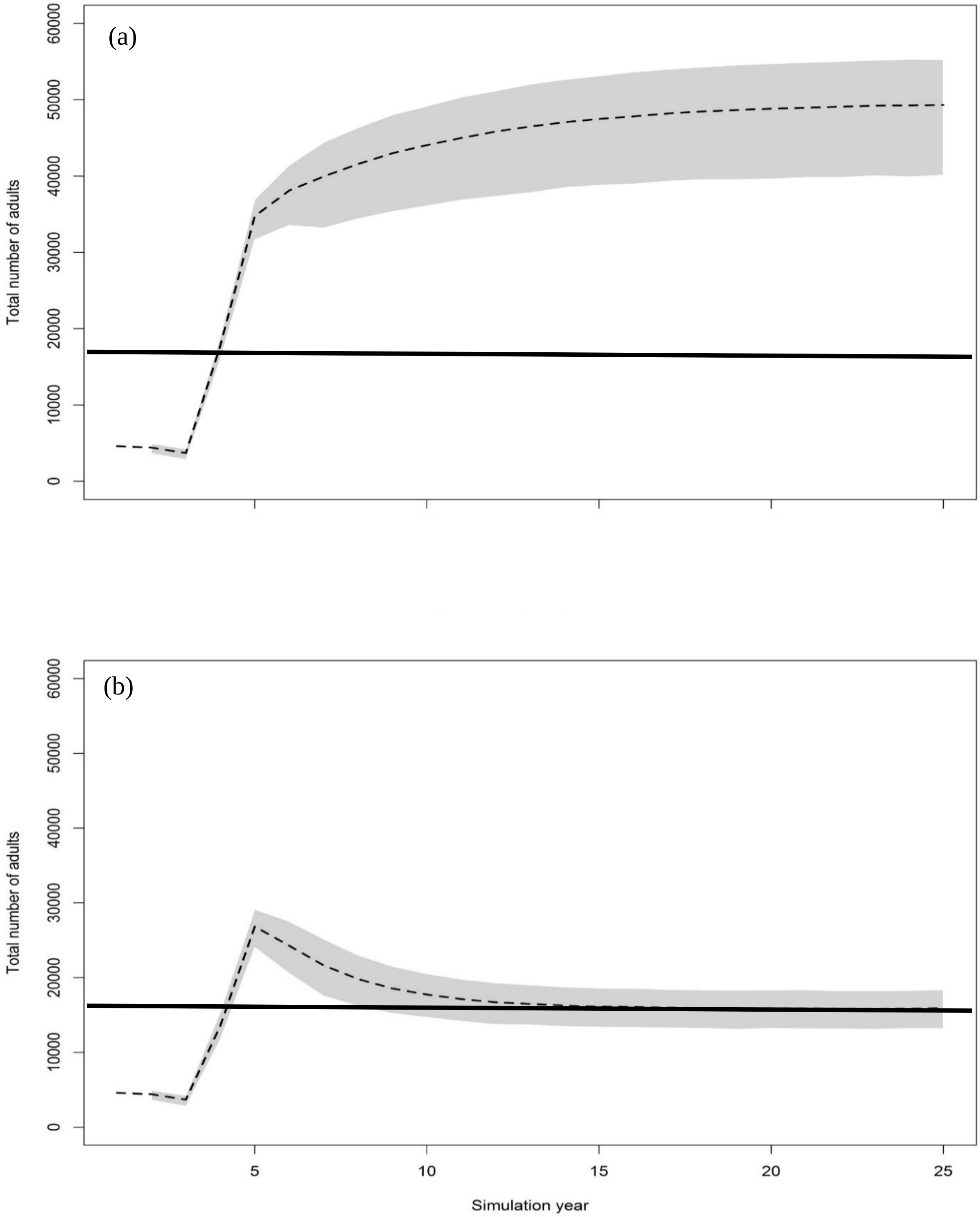

Figure 4.13. Model results when the DDF was not in effect and annual fishing mortality was (a) 0% (i.e., scenario 1) and (b) 15% (i.e., scenario 2). The solid black line represents the restoration target for adult Burbot of 17,500 age-4+ Burbot in the system.

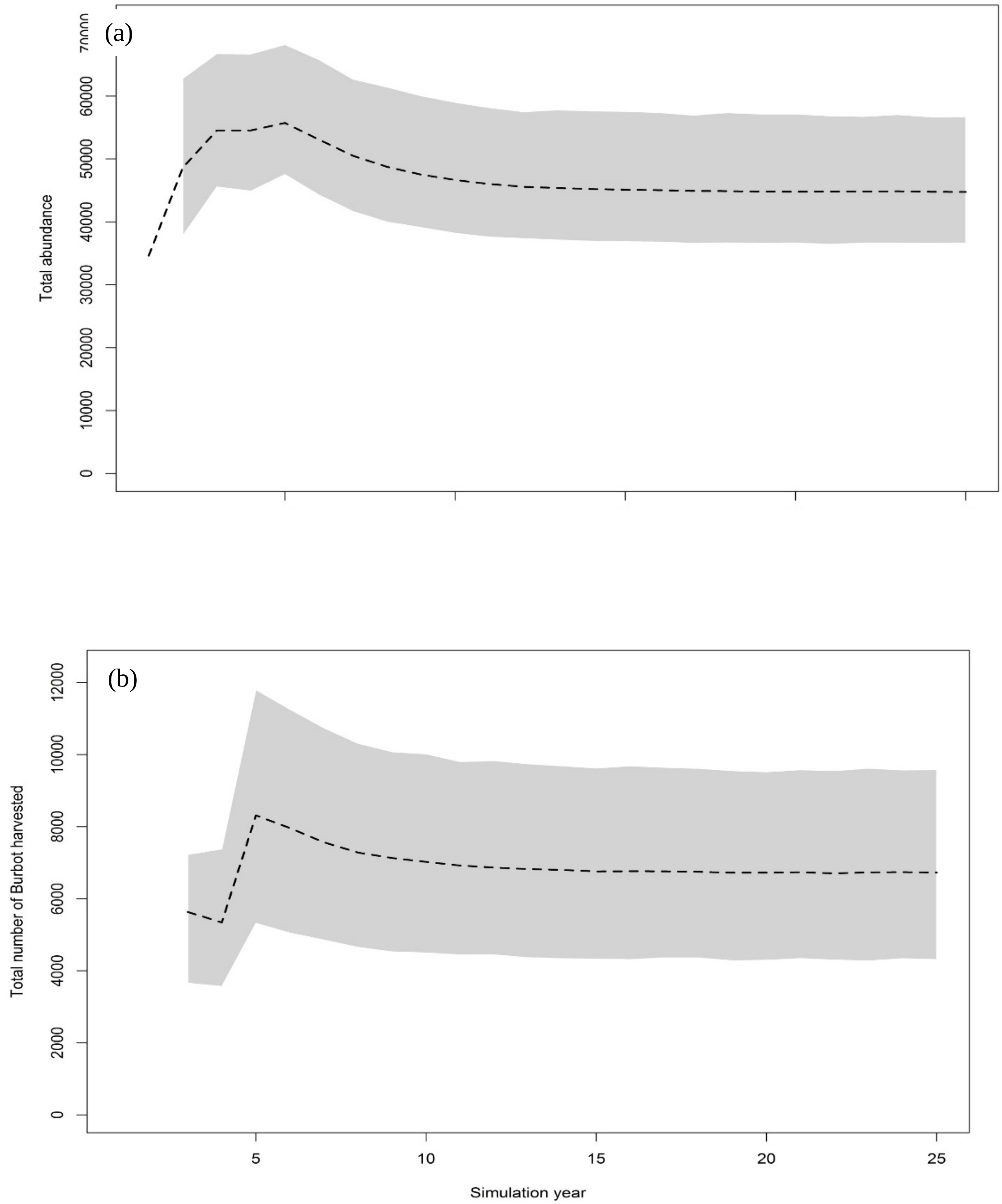

Figure 4.14. Model results when the DDF was not in effect and annual fishing mortality was 15% (i.e., scenario 2). Panel (a) represents total abundance (i.e., all age-classes) under the aforementioned model settings and panel (b) represents the total number of Burbot harvested under the same modeling scenario.

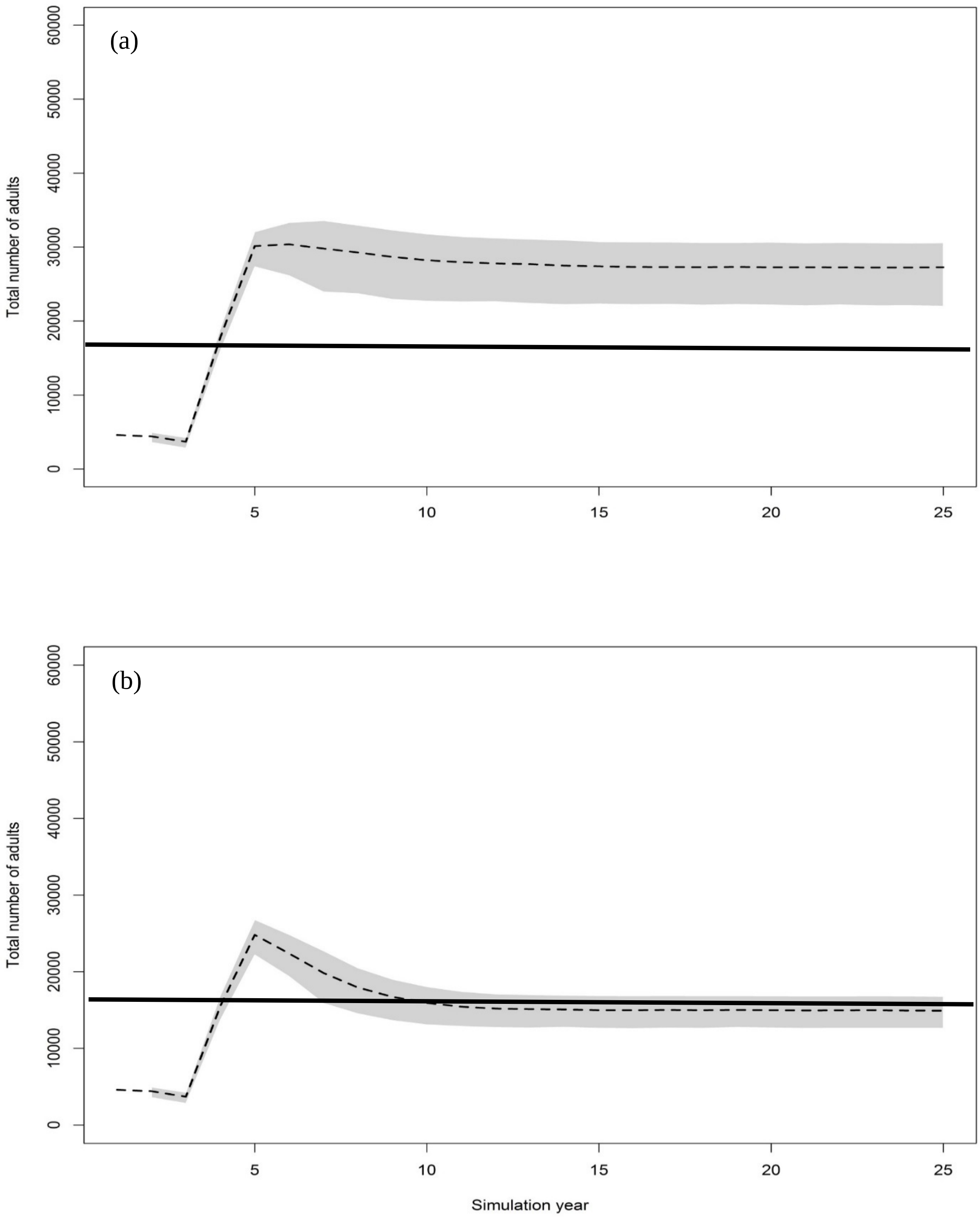

Figure 4.15. Model results when the DDF was in effect at the 17,500 adult abundance threshold and annual fishing mortality was (a) 0% (i.e., scenario 3) and (b) 10% (i.e., scenario 4). The solid black line represents the restoration target for adult Burbot of 17,500 age-4+ Burbot in the system.

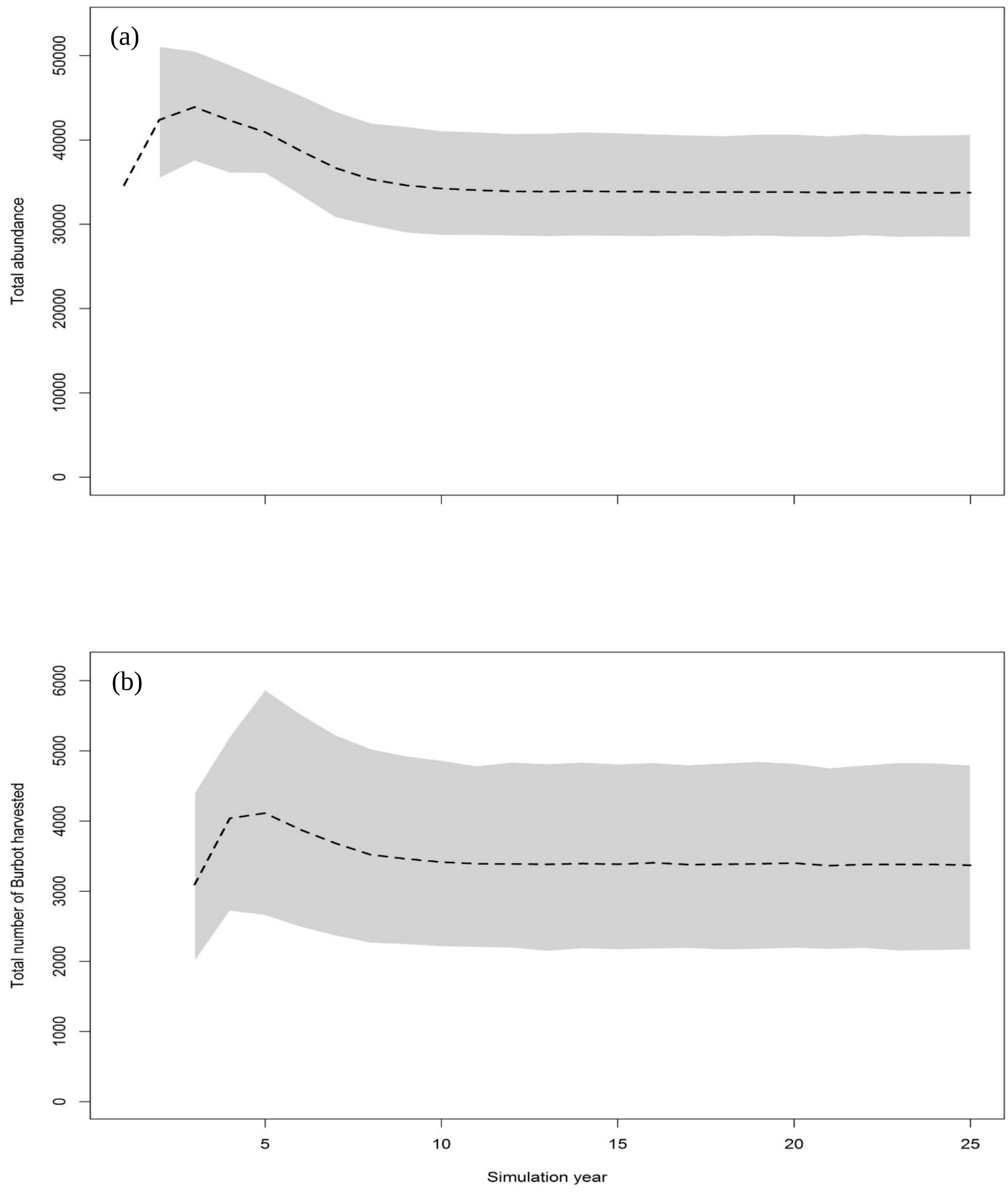

Figure 4.16. Model results when the DDF was in effect at the 17,500 adult abundance threshold and annual fishing mortality was 10% (i.e., scenario 4). Panel (a) represents total abundance (i.e., all age-classes) under the aforementioned model settings and panel (b) represents the total number of Burbot harvested under the same modeling scenario.

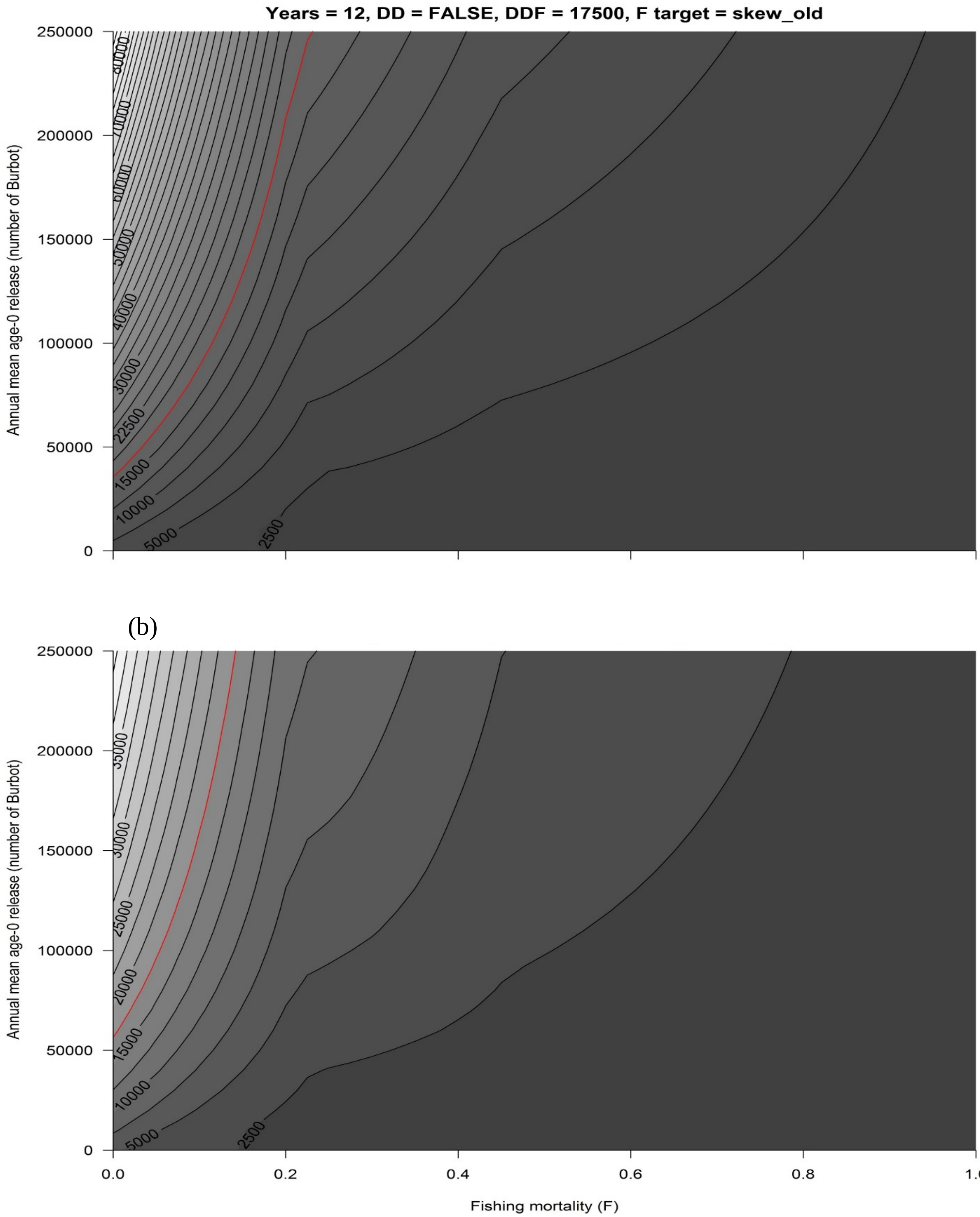

Figure 4.17. Contour plots representing the relationships between release numbers of age-0 Burbot (y-axis), annual fishing mortality (x-axis), and adult abundance (contour lines). Panel (a) represents relationships when the DDF was not in effect, and panel (b) represents relationships when the DDF was in effect at the 17,500 adult abundance threshold.

CHAPTER 5: NATIVE SALMONID MONITORING AND EVALUATION

ABSTRACT

Declines in fish stocks in the Kootenai River have long been attributed to the loss of nutrients (along with other factors) via bottom-up trophic cascades. A large-scale nutrient restoration program (using phosphate fertilizer) was implemented in the Idaho portion of the Kootenai River in 2005 to restore resident fisheries by increasing primary production. Annual electrofishing surveys were conducted at multiple sites in Idaho and Montana before and after nutrient addition in order to evaluate the responses of various fish species. The addition of liquid ammonium polyphosphate fertilizer to the Kootenai River at target concentrations (i.e., 3 µg/L) increased fish abundance and biomass over the 20 km stretch of river downstream of the treatment site. Increases were most notably documented in Largescale Suckers *Catostomus macrocheilus,* Mountain Whitefish *Prosopium williamsoni*, and Rainbow Trout *Oncorhynchus mykiss*, although nearly all fish species responded at some level*.* The Kootenai River is approximately 30 times larger in discharge than other rivers that have been experimentally fertilized, making it the largest river fertilization program to-date. In addition, our study provides compelling evidence that the mitigation of nutrient declines in rivers of this size can result in positive influences on the fish community where food is purportedly limiting growth, survival, and recruitment.

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CHAPTER STRUCTURE

Data for the nutrient restoration program from 2002-2017 have historically been broadly analyzed and summarized for annual reports for the Idaho Department of Fish and Game (IDFG; Ross et al. 2015). However, the most comprehensive and complete analyses, to-date, were completed and submitted for publication in 2019 (Hardy et al. *In Review*). The years from 2002-2017 were included in the analyses reported in Hardy et al. (*In Review*). As such, the manuscript is provided in lieu of the 2017 annual report, and project-related data from 2018-2019 are provided in this report in the form of updated figures and tables in the manuscript. All report-related information for introduction, objectives, study area, methodology, results, and discussion is consistent with work that was completed in 2018 and 2019 and can be found in Hardy et al. (*In Review*).

INTRODUCTION

Anthropogenic reduction of nutrient inputs into aquatic systems has long been recognized as a concern for aquatic communities (Ney 1996; Stockner et al. 2000). Cultural oligotrophication of rivers primarily occurs from nutrient abatement or the construction of dams for flood control or hydropower (Ney 1996). Dams trap sediment and nutrients, and each additional facility upstream compounds the negative effects to downstream fish populations through reduced primary production (Ney 1996). Unlike reductions in marine derived nutrients caused by declines or extirpation of anadromous salmon runs (Thomas et al. 2003), cultural oligotrophication requires perpetual mitigation.

Primary production forms the foundation of the food web in large streams (Minshall 1978), and in freshwater ecosystems it is often limited by the availability of nutrients, specifically nitrogen (N) and phosphorus (P) (Thomas et al. 2003). Fluctuations in the availability of N and P often influence autotrophic production (Grimm and Fisher 1986; Peterson et al. 1993) and, consequently, may affect the rate functions of various fish populations (Chapman 1966; Slaney and Northcote 1974; Dill et al. 1981). As such, novel mitigation programs have been developed to restore fisheries and ecosystems with a more holistic resource management model that addresses factors limiting growth, survival, and recruitment.

Mitigation for cultural oligotrophication and the associated declines in abundance, biomass, and biodiversity of aquatic communities has been successfully performed via stream or lake fertilization programs (Stockner and Ashley 2003). The goal of aquatic fertilization programs is often to restore fish production and recruitment via bottom-up mechanisms. The conceptual model is as follows: increased N and P levels result in increases in periphyton accrual rates, followed by increases lower trophic levels (e.g. insect biomass and abundance), which is then followed by positive responses in fish abundance, growth, survival, and recruitment (Ward and Slaney 1988; Johnston et al. 1990; Perrin and Richardson 1997; Mundie et al. 1991).

Nutrient supplementation programs have been initiated relatively recently across the northwestern United States to mitigate for nutrient losses from diminished salmon *Oncorhynchus spp.* stocks or from the construction of impoundments on free flowing rivers (Stockner and Ashley 2003). One such river that has experienced a substantial reduction in nutrient levels over the past 50 years is the Kootenai River, a large 7th order river that originates in southeastern British Columbia, Canada and flows through northwestern Montana and northern Idaho. Historically a nutrient-rich river, nutrient input into the system was dramatically reduced with the construction of Libby Dam in 1972. In addition, levy construction for flood control and agriculture isolated floodplain habitats that once contributed to significant seasonal nutrient inputs to the lower river (Northcote 1973; Woods 1982). After the construction of Libby Dam, studies found that the reservoir retained approximately 65% of P and 25% of N (Woods 1982), resulting in ultra-oligotrophic conditions downstream from the dam (Ashely et al. 1997; Schindler et al. 2011). Consequently, these changes resulted in substantial reductions in algal, macro-invertebrate, and fish production, particularly in the Idaho section of the Kootenai River (Paragamian 2002; Snyder and Minshall 2005).

The trapping of nutrients behind Libby Dam was implicated as the major cause for reduced densities of Rainbow Trout *Oncorhynchus mykiss gairdneri* in the Idaho portion of the Kootenai River, as well as many other native fish species. Although the Rainbow Trout fishery was regarded as the most important sport-fishery in the Idaho portion of the river (Paragamian 1995a; Walters 2003), densities averaged only 50 fish/km by the mid-1990s (Paragamian 1995a, b; Downs 2000; Walters and Downs 2001) compared to similar regional rivers which exhibited 3-4 fold greater densities (Bennett and Underwood 1988). Similar to Rainbow Trout, reductions in densities of other native fish species such as Mountain Whitefish *Prosopium williamsoni*, Largescale Suckers *Catostomus macrocheilus*, and Redside Shiners *Richardsonius balteatus* were documented (Paragamian 2002; Hardy 2008). In addition, other native fish species such as Kootenai River White Sturgeon *Acipenser transmontanus*, Bull Trout *Salvelinus confluentus*, and Burbot *Lota lota* have declined due to temperature, discharge, nutrient, and habitat losses in the river.

In an attempt to mitigate for the nutrient-related effects of Libby Dam on downstream biota and fish populations, a nutrient restoration program was implemented in 2005 by the Kootenai Tribe of Idaho (KTOI) and the Idaho Department of Fish and Game (IDFG) (Holderman and Hardy 2004; Minshall et al. 2014). This restoration effort marked a substantial cooperative venture by these two organizations to implement the largest stream fertilization program in the world to date. The study focused exclusively on fish community structure and its response to experimental nutrient additions in the Idaho section of the Kootenai River. Although stream fertilization studies have been performed in Canada (Peterson et al. 1993; Deegan et al. 1997; Larkin et al. 1999; Slavik et al. 2004), most of the studies occurred in small to mid-order streams. Comparatively few studies have experimentally supplemented nutrient regimes of large rivers that were historically void of marine derived nutrients, and even fewer involve oligotrophic rivers (Dodds 2006; Minshall et al. 2014). Thus, our study provides important information on restoring fish populations in large rivers that have experienced significant impacts from nutrient loss as a result of upstream impoundments. The specific objectives of our research were to evaluate the effects of nutrient addition on assemblage structure, density, and biomass of various fish populations among treatment zones.

STUDY AREA

The Kootenai River flows south from its headwaters in Kootenay National Park in southeastern B.C., Canada through northwestern Montana where it enters Lake Koocanusa, the reservoir formed by Libby Dam (Figure 5.1). The river then flows northwest into the panhandle of Idaho, then north into B.C. to form Kootenay Lake, and finally enters the Columbia River at Castlegar, B.C. The Kootenai River is the second largest of the Columbia River tributaries and third largest in drainage size (i.e., approximately 50,000 km^2; Bonde and Bush 1975). Historically, peak discharges of the Kootenai River near were >2,832 m^3/s, which is now greatly reduced by Libby Dam. The study area was comprised of approximately 106 km of the river that flows through the panhandle of Idaho, along with one control site near the Yaak River confluence in Montana (Figure 5.1).

The Montana and Idaho portions of the Kootenai River below Libby Dam can be separated into three distinct geomorphic habitat types. Directly below the dam the river flows through a narrow canyon segment characterized by steep canyon walls, high gradients, and cobble and boulder substrates. In this segment of the river, the channel has an average gradient of 0.6 m/km, and the velocities are often greater than 0.8 m/s. Downstream from the canyon segment, there is a braided transition segment that extends from the Moyie River to the town of Bonners Ferry (Figure 5.1). Downstream from the braided transition segment, velocities slow to less than 0.4 m/s, gradient reduces to 0.02 m/km, the channel deepens, and the river meanders through the Kootenai Valley (Snyder and Minshall 2005).

METHODS

Nutrient Addition

We added agricultural-grade ammonium polyphosphate ($[NH_4, P_2O_5]_n$; 10-34-0) and urea ammonium nitrate ($CO[NH_2]_2NH_4NO_3$; 32-0-0) liquid fertilizer seasonally from approximately June 1 - September 30 from 2005 through 2017 at a single location near the Idaho and Montana border (Figure 5.1). Nutrients were applied at sufficient rates to ensure that the epilimnetic dissolved inorganic nitrogen:total dissolved phosphorus (DIN:TDP) ratio remained greater than 10:1 on a weight: weight basis throughout the growing season (Ashley and Stockner 2003). Fertilizer was precisely applied to achieve a target of 3.0 µg/L of TDP and 30-50 µg/L of DIN. However, 32-0-0 was seldom added since DIN was typically above the desired target throughout the growing season. Fertilizer was supplied to the river via a gravity-flow system with the aid of low-flow pumps designed to dose at loading rates directly proportional to the daily flow rates of the Kootenai River at the application site. River flow was determined daily at an on-site US Geological Survey gaging station (12305000 at Leonia, ID) to aid in pump calibrations. Nutrient addition began in 2005 to reach 1.5 µg/L of TDP and increased to 3.0 µg/L from 2006-2017 to achieve the targeted treatment concentration.

Field Sampling

Sampling sites for this study were established to gather fisheries and lower trophic level data prior to and after the addition of nutrients. For the purposes of this study, we will only report on the fish responses to nutrient additions. Additional trophic response results were reported by Minshall et al. (2014) and Hoyle et al. (2014). Fish populations were annually surveyed at five sampling sites (Figure 5.1). The control site (KR10) was located in the Montana portion of the Kootenai River (i.e., upstream from the nutrient addition site), termed the “control zone.” Two sites were located within the “nutrient addition zone” of the river (i.e., sites KR9 and KR6), which was upstream from the town of Bonners Ferry, ID, but immediately downstream from the nutrient addition site. Site KR9 was located approximately ten river kilometers (rkms) downstream from the nutrient addition site, and site KR6 was located approximately 20 rkms downstream from the nutrient addition site. Two additional sites were located downstream from the town of Bonners Ferry, ID, and they were considered to be in the “downstream zone” of the river. Site KR4 was approximately 68 rkms downstream from the nutrient addition site, and site KR2 was approximately 157 rkms downstream from the nutrient addition site (Figure 5.1). The aforementioned “river zone” delineations remained consistent throughout the study and for all statistical analyses.

Boat electrofishing was conducted during August and September from 2002-2017 at five sampling sites. Sites were sampled using a jet boat equipped with a Coffelt VVP-15 electroshocker powered by a 5,000 watt Honda generator. Electrofishing settings were typically set to generate 6-8 amps at 175-200 volts. The sampling crew consisted of two netters and one boat driver. All fish, regardless of species and size, were netted in order to get a representative sample of the fish community at each site. In order to increase sampling replication, each site was divided into six equal subsections of 333 m with 150 m separating each to ensure that each subsection was independent of the next. The sampling design resulted in one kilometer of electrofishing occurring on both the left and right banks of the river for a total of two kilometers of sampling, per site. A single pass was made through each subsection, starting with downstream sections first to ensure that no fish drifted into areas that had not yet been sampled. After each subsection was sampled, the elapsed sampling time was recorded and fish that had been collected were taken to a workup station where they were identified to species, measured (total length [TL], mm), and weighed (g). Specific population indices that were indexed included relative species abundance as catch-per-unit-of-effort (CPUE) and abundance by weight as biomass-per-unit-of-effort (BPUE). Data collected from these sites were used to document temporal trends in the fish community and to evaluate the effectiveness of the nutrient addition program.

Statistical Analysis

The years from 2002-2005 were considered to be pretreatment and 2006-2017 were considered to be post-treatment, for all analyses. Program R (R Development Core Team 2018) and SAS 9.3 (SAS Institute, Cary NC) were used for all statistical tests.

Fish Assemblage

Fish assemblage relationships were evaluated following methods similar to those described by Kwak and Peterson (2007) using hierarchical clustering analysis and nonmetric multidimensional scaling (NMDS). Bray-Curtis dissimilarity values were calculated using presence-absence (i.e., species occurrence) data that included all sites, years, and fish species. In addition, Bray-Curtis dissimilarity values were calculated using CPUE data and BPUE data from all sites, years, and fish species. Data were pooled across sites within respective river zones and across years within respective pre- and post-treatment periods. The resulting dissimilarity matrices were used in (1) hierarchical clustering analysis (average-linkage) and (2) NMDS. Clustering analysis was done using only the species occurrence dissimilarity matrix; whereas, three separate NMDS analyses were run, each using different dissimilarity matrices. One NMDS analysis was run using the species occurrence dissimilarity matrix, one using the CPUE dissimilarity matrix, and one using the BPUE dissimilarity matrix. Differences in fish assemblage structure (i.e., by zone, period, and the interaction of zone*period) were evaluated using a permutational multivariate analysis of variance (PERMANOVA) for each of the species occurrence, CPUE, and BPUE dissimilarity matrices. Bray-Curtis dissimilarity matrices were calculated using the Vegdist function, hierarchical clustering analyses were done using the Hclust function, and NMDS and PERMANOVA analyses were done using the MetaMDS and Adonis functions, respectively, in the Vegan package, Program R.

Abundance and Biomass

We used generalized linear mixed models to evaluate the effects of nutrient addition on abundance (CPUE) and biomass (BPUE) of fish populations in the Kootenai River. The structure of these models closely followed the experimental design of the project. Each model used the same fixed and random effects. For the count and biomass data the models took the form:

[1] $\mu_{tskji} = \beta_0 + \beta_j X_{i,j} + \beta_k X_{i,k} + \beta_{j:k} X_{i,j:k} + \alpha_t + \gamma_s + effort_i$

[2] $y_{tskji} \sim NegBinom(exp[\mu_{tskji}], \delta)$

where β_0 is the intercept, β_j is the effect of the *j*th zone, β_k is the effect of the *k*th period and $\beta_{j:k}$ is the interaction between zone and period.

All models were fit using a Bayesian framework using the package *brms* in R. The posterior distribution was sampled using 4 chains, each with a total of 2000 samples. One half of the samples were used in the "burn-in" process, leaving a total of 4000 total posterior samples. Convergence was assessed by evaluating traceplots as well through the Gelman-Rubin convergence diagnostic. 90% highest density probability (HDP) credible intervals were reported for each marginal posterior distribution.

Population Estimates

Mark-recapture population estimates were periodically conducted within a three kilometer (km) section within the nutrient addition zone from 1980 until 2016 using boat electrofishing as described by Downs (2000). Although the population estimates were not originally designed to evaluate the effect of nutrient addition on fish populations, the data were a useful reference in monitoring abundance trends in combination with additional statistical modeling specifically designed for the nutrient addition study. In order to estimate abundance, Mountain Whitefish, Largescale Sucker, and Rainbow Trout were uniquely marked the second week in August and recaptured the following week to allow adequate mixing within the sample location. Population estimates were calculated using Chapman's modification of the Petersen Method (Ricker 1975; Krebs 1999):

$$N = \left[(M+1) * \frac{C+1}{R+1}\right] - 1$$

where *N* is abundance, *M* is the number of marked fish, *C* is the number of fish captured during the recapture sample, and *R* is the number of recapture marks encountered in the recapture sample.

The 95% confidence limits for the population estimates were calculated based on the Poisson distribution (Ricker 1975; Seber 1982).

RESULTS

A total of 25,375 fish from 21 different species were sampled from five sites in the Kootenai River during the years 2002-2017. Early in the design of the nutrient addition project, Rainbow Trout, Mountain Whitefish, and Largescale Sucker were identified as abundant and focal indicator species likely to most notably respond to nutrient addition efforts. Approximately 97% of the fish sampled were either Mountain Whitefish, Northern Pikeminnow *Ptychocheilus oregonensis,* Largescale Sucker, Redside Shiner, Peamouth Chub *Mylocheilus caurinus*, or Rainbow Trout. The remaining 3% of the catch represented 16 less abundant native and nonnative fish species (Table 5.1). The proportion of species in the catch remained relatively consistent across sampling years and within river zones. Six fish species dominated catch and biomass in the control and nutrient addition zones, including Mountain Whitefish, Largescale Sucker, Rainbow Trout, Peamouth Chub, Northern Pikeminnow, and Redside Shiner. The same species dominated catch and biomass in the downstream zone with the exception of Mountain Whitefish and Rainbow Trout.

The cluster analysis corroborated the hypothesis that species composition varied among sampling sites and subsequent river zones. More specifically, sampling sites within the control and nutrient addition zones of the river (i.e., KR10, KR9, and KR6) were most closely associated with one another and least associated with sites in the downstream zone (Figure 5.2); no distinct clusters of pre- and post-treatment periods were found either within or across sites (Figure 5.2). Results from the cluster analysis indicate that fish assemblage structure in the Kootenai River is largely driven by habitat (i.e., specific species prefer specific habitat types).

The NMDS ordinations complemented the cluster analysis and provided several additional insights. Similar to the cluster analysis, the NMDS ordination that was fit to the CPUE data indicated that sampling sites in the control and nutrient addition zones were more closely associated to one another than to sampling sites in the downstream zone (Figure 5.3). Furthermore, sites in the nutrient addition and control zones were most closely associated with Mountain Whitefish, Rainbow Trout, Westslope Cutthroat Trout, Brown Trout, and Brook Trout; whereas, sites in the downstream zone were most closely associated with Northern Pikeminnow, Redside Shiner, Smallmouth Bass, and Pumpkinseed (Figure 5.3). Perhaps most noteworthy, standard error ellipses for the pre- and post-treatment periods in the nutrient addition zone did not overlap, suggesting that the fish assemblage shifted from the pre- to post-treatment period in that zone. Conversely, standard error ellipses for the pre- and post-treatment periods in both the control and downstream zones displayed distinct overlap (Figure 5.3). The PERMANOVA analysis for CPUE corroborated the NMDS ordination plot, indicating that CPUE differed by zone (F = 89.5, P = 0.001), period (F = 7.0, P = 0.002), and the interaction between zone and period (F = 3.2, P = 0.01). The NMDS ordination that was fit to the BPUE data displayed a similar, but generally less supported, pattern. Species associations by river zone were similar to those observed in the CPUE ordination, except Largescale Sucker were closely associated with sites in the nutrient addition zone during the post-treatment period (Figure 5.4). Identical to the CPUE ordination, pre and post-treatment period ellipses in the nutrient addition zone did not display distinct overlap (Figure 5.4). The PERMANOVA analysis for BPUE generally supported the NMDS ordination plot: BPUE differed by zone (F = 57.1, P = 0.001) and period (F = 6.2, P = 0.003), but not the interaction between the two (F = 1.4, P = 0.22).

Although the multivariate analyses produced informative results, catch and biomass metrics were highly variable among sites and years (Figures 5.5 and 5.6), which complicated analysis results and subsequent interpretations. However, despite variability in the catch and biomass data, the generalized linear mixed models provided multiple useful insights on the fish assemblage in the Kootenai River. In general, CPUE and BPUE metrics were greater in the control and nutrient addition zones relative to the downstream zone and greater in the post- relative to the pretreatment period (Tables 5.2 and 5.3; Figures 5.5 and 5.6). Results from the count models indicated that all species cumulatively responded positively to the addition of nutrients. Total count was 85% greater in the nutrient addition than in the control zone, 35% greater in the post- than in the pretreatment period, and 24% greater in the nutrient addition zone in the post- than in the pretreatment period (Table 5.4). Although similar effects (i.e., positive) were observed for the three focal indicator species, not all effects were well-supported by the model (Table 5.4). Largescale Sucker count was 44% higher in the nutrient addition zone in the post- than in the pretreatment period, and Rainbow Trout and Mountain Whitefish count was 55% and 33% higher, respectively, in the post- than in the pretreatment period (Table 5.4). Interestingly, the increase from pre- to post-treatment periods was most notable for Rainbow Trout in the control zone (Table 5.2). Results from the biomass models were similar to those from the CPUE models; all species cumulatively responded positively to nutrient addition. Total biomass was 84% greater in the nutrient addition relative to the control zone, 20% greater in the post- than in the pretreatment period, and 33% greater in the nutrient addition zone in the post- than in the pretreatment period (Table 5.5). Largescale Sucker biomass was 185% greater in the nutrient addition relative to the control zone and 55% higher in the nutrient addition zone in the post- than in the pretreatment period (Table 5.5). Nearly identical to the count models for Rainbow Trout, the biomass models indicated 42% greater Rainbow Trout biomass in the post- relative to the pretreatment period (Table 5.5), and again, this was most pronounced in the control zone (Table 5.3). Although Mountain Whitefish biomass did respond positively to nutrient addition, none of the effects were well supported by the model (Table 5.5).

Population estimates at Hemlock Bar largely corroborated CPUE data from sites in the nutrient addition zone. Specifically, mean abundance estimates for all three focal indicator species increased from the pre- to post-treatment periods (Figure 5.7). Furthermore, temporal trends in CPUE tracked closely with temporal trends in abundance for each focal indicator species (Figures 5.5 and 5.7), suggesting that CPUE was a viable surrogate for abundance. Unfortunately, abundance estimates were not available for these species at locations in the control zone, precluding full evaluation of changes as a result of nutrient additions. Regardless, inferences gleaned from the abundance estimates spanning the pre- and post-treatment periods support the interpretations from the more formal analyses.

DISCUSSION

The addition of liquid ammonium polyphosphate fertilizer to the Kootenai River at a relatively low target concentration accomplished the intended effect of increasing fish abundance and biomass over the 20 km stretch of river downstream of the treatment site. Initial multivariate analyses indicated a slight shift in CPUE metrics within the nutrient addition and control zones of the river from the pre- to post-treatment periods. Further univariate analysis indicated that the greatest increase over the treatment period was in Largescale Suckers with a 44% increase in abundance and 55% increase in biomass in the nutrient addition zone. Similarly, a marked increase in these response variables was documented in Rainbow Trout and Mountain Whitefish, as well. The substantial increases in focal indicator species were also corroborated with trend population estimates performed at the same time of year. The mechanisms responsible for these increases are likely bottom-up effects on primary production that ultimately increased food resources for fish. Hoyle et al. (2014) reported that following the first five years of fertilization to the Kootenai River there were significant increases in chlorophyll accrual rates and densities of edible green algae and diatoms. Therefore, it is not surprising that Largescale Suckers, a species known to have diets comprised of nearly 90% periphyton (Dauble 1986), exhibited the most notable responses to nutrient addition efforts. Likewise, Minshall et al. (2014) reported a 69% increase in the total abundance of benthic macroinvertebrates and a 49% increase in their biomass. As such, Rainbow Trout and Mountain Whitefish, two species known to feed primarily on macroinvertebrates and their larvae, also exhibited notable increases after treatment. Similar studies of other western rivers have documented positive effects of fertilization to fish populations through the increase of trophic production. Peterson et al. (1993) reported an increase in young-of-the-year Arctic Grayling following four years of fertilization and attributed it to increases in epilithic algae and insects. A comparable study by Wilson et al. (2003) reported a four-fold increase in Rainbow Trout following four seasons of inorganic nutrient additions.

Largescale Suckers most notably responded to nutrient addition efforts, followed by Mountain Whitefish, and then Rainbow Trout. Watkins et al. (2017), evaluated the influence of nutrient addition efforts in the Kootenai River on the growth, survival, and recruitment of Largescale Suckers and Mountain Whitefish and found that incremental growth of the Largescale Sucker was positively correlated with the addition of nutrients. Our results clearly corroborate these findings, as we documented marked increases (i.e., 185%) in biomass of Largescale Suckers in response to nutrient addition efforts. Conversely, Watkins et al. (2017) also found that although the abundance of Mountain Whitefish nearly doubled since nutrient addition efforts began, incremental growth declined, suggesting a possible density dependent response in growth beginning around 2010. The results of our study also corroborate these findings, documenting a decline in biomass of Mountain Whitefish in the latter years that the Watkins et al. (2017) study spanned (i.e., 2009-2012). It is important to note that biomass of Mountain Whitefish has been consistently climbing since its lowest point in 2012; however, it is unknown whether or not incremental growth patterns have also shifted since 2012. The observed response of Mountain Whitefish is not entirely understood; however, such results are not unique to the Kootenai River. For example, reduced growth caused by intraspecific competition, indirectly caused by nutrient additions, were reported for Arctic Grayling *Thymallus arcticus* in the Kuparuk River, Alaska (Deegan et al. 1997). Furthermore, it is plausible that during the years of our population trend monitoring (i.e., 2002-2017), we captured the cyclic behavior of the Mountain Whitefish population in the Kootenai River. Population cycles in various freshwater and anadromous fish species are well documented (Townsend 1989; Levy and Wood 1992), and the cyclic behavior is often attributed to the effects of density-dependence on fecundity or survival of eggs or larvae subject to interaction with predators. Similar studies on the growth, survival, and recruitment of Rainbow Trout have not been done but would likely enhance the body of knowledge on salmonid responses to nutrient addition efforts in the Kootenai River.

All multivariate and univariate analyses indicated that abundance and biomass of Rainbow Trout increased to the same or greater degree from the pre- to post-treatment periods in the control zone compared to the nutrient addition zone. This result was initially surprising, but upon further investigation of spatiotemporal movements and behaviors of Rainbow Trout, was a logical and explainable result. We chose a study design with a control zone located approximately 10 rkms above the nutrient addition zone to maintain independence among sampling locations, yet remain close enough to be comparable in their habitat complexity and ambient productivity at lower trophic levels. For most species sampled within each trophic level, the distance from the control zone to the nutrient addition zone was sufficient to maintain this independence (Holderman et al. 2009; Hoyle 2012; Minshall 2014); however, our study indicated that nutrient treatments may have influenced Rainbow Trout abundance and biomass in both the control and nutrient addition zones. Although the mechanisms driving this response are not entirely understood, the reasons for the increase are likely linked to adult spawning activity. A telemetry study completed when nutrient additions began indicated that the majority of adult Rainbow Trout residing in the nutrient addition zone migrate upstream past the control zone and enter Montana tributaries to spawn (Walters et al. 2005). Such behavior likely results in variable out-migrant dispersal from these tributaries downstream to the nutrient addition zone. Rainbow Trout spawning migrations typically occur in the spring, which does not temporally coincide with sampling efforts for our study that occur in the fall; therefore, it is unlikely that movement of adult Rainbow Trout directly influenced abundance and biomass in the Control Zone. In fact, further evaluation indicated that the majority of the observed increases in Rainbow Trout in both the control and nutrient addition zones were attributable to juvenile Rainbow Trout, indicative of an increase in recruitment. Johnston et al. (1999) showed that after five years of adding inorganic nutrients to a montane lake in British Columbia, Canada, Rainbow Trout reproductive output, growth, and yield significantly increased. Therefore, although not directly quantified, it is possible that Rainbow Trout within the nutrient addition zone of the Kootenai River had greater reproductive potential (post-treatment), resulting in increased production from both Idaho and Montana tributaries. Assuming this is true, Rainbow Trout would out-migrate from spawning tributaries within or upstream from the control zone, and then exhibit variable dispersal. Widespread post-emergence dispersal patterns have been seen in other salmonid species. Bradford and Taylor (1997) showed that stream-type Chinook Salmon *Oncorhynchus tshawytscha* exhibited post-emergence dispersal patterns ranging up to 100 km downstream. It is possible that newly emerged and out-migrated Rainbow Trout spawned in Montana tributaries exhibited dispersal patterns that influenced both the control and nutrient addition zones of this study. Long-term population monitoring of Rainbow Trout conducted by Montana Fish Wildlife and Parks showed that the majority of dispersal occurs during the early juvenile years, followed by establishing a well-defined, localized home range as adults (Jim Dunnigan, Montana Fish Wildlife and Parks, personal communication). Such results highlight the need to fully understand recruitment trends of Rainbow Trout in the Kootenai River as well as the extent of migration behavior when designing a study to evaluate treatment effects on a particular fish population.

While it may be reasonable to expect to quickly (i.e., within 3-5 years) document effects of largescale mitigation activities at lower trophic levels, fish life histories and longevities complicate the timeframe during which results might be observed. As such, researchers and managers should carefully consider research objectives and hypotheses and subsequent timeframes for sampling and data analysis to ensure a more thorough and comprehensive interpretation of mitigation effects. For example, Watkins et al. (2017) indicated that Largescale Suckers in our study area were not fully recruited to boat electrofishing gear until approximately age-11. Regardless of the mechanism(s), it is clear that the benefits of performing a longer term evaluation allowed us to capture the effects of nutrient addition efforts on this particular species that might have otherwise been missed if sampling had been discontinued after 3-5 years. Similarly, the long-term sampling design of our study allowed us to capture the initial increases in numbers and subsequent declines in growth (i.e., density dependent response) of Mountain Whitefish that likely would have been missed if sampling duration had been shorter. Other stream fertilization projects reported similar results (Bowden et al. 1994) and suggested that short-term studies (even up to eight years) are poor predictors of the full ecological effects that nutrient addition efforts will eventually provide to a system (Slavik et al. 2004).

It is often difficult to predict the outcome of large-scale, manipulation-type experiments at all trophic levels, and it is not uncommon for unexpected or unforeseen outcomes to arise (Davis et al. 2010; Cross et al. 2011). As such, it is important for researchers and managers to consider possible changes in community structure to non-target species (changes which may be undesirable). Fortunately, our results showed that nutrient addition efforts to the Kootenai River did not enhance populations of non-native or invasive fish species that have been periodically sampled since nutrient addition efforts began. It stands to reason that a comprehensive evaluation of factors limiting non-target species should be completed to determine whether or not mitigation efforts to address native fish recruitment bottlenecks will inadvertently improve non-native fish species as well. The results of our study also provide some evidence that perhaps factors limiting most non-native species in the Kootenai River may be related to specific habitat preference or availability rather than food production.

The longitudinal effects of nutrient addition efforts to the Kootenai River on the abundance, biomass, and structure of the fish community appeared to decay by 20 to 45 rkms downstream from the addition site. We found no evidence of nutrients affecting fish communities in the downstream zone at sampling sites (i.e., KR4 and KR2) located approximately 45 and 100 rkms (respectively) downstream of where nutrients were added. This is generally consistent with trends reported for water quality, chlorophyll *a*, and benthic macro invertebrates (Holderman et al. 2009; Hoyle 2012; Minshall et al. 2014), except that the extent of the treatment effect is even less (i.e., 10-15 rkms) for lower trophic levels. Studies have shown that the effective distance of nutrients is directly related to water velocity as well as the ability of the trophic communities to recycle or "spiral" nutrients and release them back into the water column (Ashley and Stockner 2003; Mulholland 1996). The typical prescription of nutrient dosing locations is to have them coincide with uptake distances and remain consistent with a river's more natural food web processes. For example, the Keough River in British Columbia, Canada flows 30 km from the source to the ocean and possessed a spiraling distance of approximately 6 km (Ashley and Stockner 2003). Therefore, a slow release fertilizer was applied at five equidistant locations to facilitate sustained nutrient effects throughout the system (Ashley and Stockner 2003). Although additional research may be required to fully understand the nutrient spiraling processes in the Kootenai River basin, our results, along with the documented effects on other trophic levels, suggest that there may be a need to add additional dosing sites to extend the benefits of nutrients to fish communities and lower trophic levels in the lower parts of the basin.

Not unlike many large rivers in the northwestern United States, the Kootenai River is a heavily altered system due to the effects of impoundment and subsequent spatiotemporal alterations in flow, water temperature, sediment regime, and nutrient regime (Snyder and Minshall 2005). In addition, historic and current mining activity in the headwaters of the Kootenai River basin in British Columbia have also created additional anthropogenic disturbances, the most pervasive of which is altered nitrogen and selenium concentrations in the river and its impounded reservoir (Jim Dunnigan, MFWP, personal communication). As such, the Kootenai River has been the target of extensive mitigation efforts and supporting research, monitoring, and evaluation. The study results reported herein could not account for the many anthropogenic perturbations and subsequent mitigation and restoration activities, so it is possible some of the reported results are confounded by factors beyond the scope of this study. Although the benefits of nutrient addition and its positive influence on multiple trophic levels are well-understood (Perrin et al. 1987; Slaney and Ward 1993; Bowden et al. 1994), the Kootenai River is approximately 30 times larger in discharge than other rivers that have been experimentally fertilized (Minshall et al. 2014), making it the largest river fertilization program to-date. For this reason alone, our study is a defining milestone and provides compelling evidence that the mitigation of nutrient declines in rivers of this size can result in positive influences in the fish community where food is limiting growth, survival, and recruitment. The impacts of inorganic nutrient additions in the present study were somewhat complex rather than a simple proportional increase from one trophic level to the next. The fish community in the Kootenai River changed and is continuing to benefit as more long-lived species recruit to sampling gear. This observation emphasizes the need to approach these types of mitigation efforts with a long-term evaluation lens in order to better understand the full response of the food web through time.

Future research should focus on (1) further characterizing rate function responses (i.e., growth, recruitment, and survival) of focal fish species (e.g., Rainbow Trout) to nutrient addition efforts and (2) evaluating whether or not a second nutrient addition site would benefit the food web in the Kootenai River.

RECOMMENDATIONS

1. Conduct an age, growth, recruitment, and survival analysis for Rainbow Trout similar to the analysis conducted for Mountain Whitefish and Largescale Sucker in the Kootenai River by Watkins et al. (2017).

2. Evaluate the efficacy of a second nutrient addition site near the Moyie River confluence with Kootenai River in Idaho.

3. Continue annual addition of nutrients at the existing nutrient addition location.

4. Continue annual electrofishing sampling at existing sampling sites.

5. Continue conducting mark-recapture population estimate surveys at Hemlock Bar every two years.

TABLES

Table 5.1. Fish species captured during fall electrofishing surveys on the Kootenai River from 2002-2019. Species are listed in order of relative abundance (percentage). Data from 2018 and 2019 were not included in formal analyses and are not included in Hardy et al. (*In Review*); however, they are included here in support of the 2018 and 2019 IDFG annual reports.

Common name	**Scientific name**	**Percentage of catch**
Mountain Whitefish	*Prosopium williamsoni*	36.1
Northern Pikeminnow	*Ptychocheilus oregonensis*	19.8
Redside Shiner	*Richardsonius balteatus*	12.6
Peamouth Chub	*Mylocheilus caurinus*	12.3
Largescale Sucker	*Catostomus macrocheilus*	10.4
Rainbow Trout	*Oncorhynchus mykiss*	5.23
Yellow Perch	*Perca flavescens*	1.25
Longnose Sucker	*Catostomus catostomus*	1.10
Slimy Sculpin	*Cottus cognatus*	0.31
Westslope Cutthroat Trout	*Oncorhynchus clarki lewisi*	0.22
Longnose Dace	*Rhinichthys cataractae*	0.18
Kokanee Salmon	*Oncorhynchus nerka*	0.13
Pumpkinseed	*Lepomis gibbosus*	0.11
Burbot	*Lota lota*	0.10
Brown Trout	*Salmo trutta*	0.07
Brown Bullhead	*Ameiurus nebulosus*	0.05
Brook Trout	*Salvelinus fontinalis*	0.01
Smallmouth Bass	*Micropterus dolomieu*	0.01
Bluegill	*Lepomis macrochirus*	0.01
Black Crappie	*Pomoxis nigromaculatus*	0.01
Largemouth Bass	*Micropterus salmoides*	0.01

Table 5.2. Average catch-per-unit-of-effort +/- 95% confidence intervals for total, Mountain Whitefish (MWF), Largescale Suckers (LSS), and Rainbow Trout (RBT) in the Kootenai River. Mean values represent pooled averages by period and river zone. Data from 2018 and 2019 were not included in formal analyses and are not included in Hardy et al. (*In Review*); however, they are included here in support of the 2018 and 2019 IDFG annual reports.

	control zone		**nutrient addition zone**		**downstream zone**	
	pre	**post**	**pre**	**post**	**pre**	**post**
Total	3.44 ± 1.18	4.90 ± 0.71	4.80 ± 0.65	9.82 ± 0.95	4.38 ± 0.75	7.08 ± 2.15
MWF	2.25 ± 0.97	2.95 ± 0.66	3.68 ± 0.70	7.63 ± 1.02	0.18 ± 0.12	0.18 ± 0.09
LSS	0.46 ± 0.22	0.45 ± 0.10	0.53 ± 0.11	1.15 ± 0.28	0.47 ± 0.16	0.66 ± 0.17
RBT	0.37 ± 0.09	0.83 ± 0.12	0.29 ± 0.13	0.66 ± 0.12	0.05 ± 0.04	0.06 ± 0.02

Table 5.3. Mean +/- 95% confidence intervals for biomass-per-unit-of-effort for total, Mountain Whitefish (MWF), Largescale Suckers (LSS), and Rainbow Trout (RBT) in the Kootenai River. Mean values represent pooled averages by period and river zone. Data from 2018 and 2019 were not included in formal analyses and are not included in Hardy et al. (*In Review*); rather, they are included here in support of the 2018 and 2019 IDFG annual reports.

	control zone		**nutrient addition zone**		**downstream zone**	
	pre	**post**	**pre**	**post**	**pre**	**post**
Total	0.82 ± 0.34	0.90 ± 0.17	0.98 ± 0.18	1.92 ± 0.28	0.38 ± 0.06	0.44 ± 0.11
MWF	0.42 ± 0.17	0.43 ± 0.11	0.49 ± 0.14	0.73 ± 0.13	0.00 ± 0.00	0.00 ± 0.00
LSS	0.28 ± 0.15	0.26 ± 0.08	0.37 ± 0.10	0.98 ± 0.23	0.38 ± 0.06	0.29 ± 0.09
RBT	0.08 ± 0.02	0.14 ± 0.02	0.06 ± 0.03	0.13 ± 0.03	0.01 ± 0.00	0.01 ± 0.00

Table 5.4. Results from the generalized linear mixed models on the count data. Response variables include counts for total, Rainbow Trout, Mountain Whitefish, and Largescale Suckers. The table includes the posterior median estimate and 90% highest probability density (HPD) lower and upper credible intervals (CI) for each fixed effect and interaction in the model. Bolded, italicized text indicates parameters with strong support, and asterisks indicate parameters with substantial effect support, as gauged by the HPDs.

Parameter	Posterior median	HPD lower CI	HPD upper CI
Total count			
intercept	1.67	1.37	1.98
zonedownstream	-0.03	-0.34	0.27
zonenutrientaddition	***0.62***	***0.33***	***0.90***
periodpre	***-0.44***	***-0.87***	***-0.05***
zonedownstream:periodpre*	0.28	-0.06	0.60
zonenutrientaddition:periodpre*	-0.27	-0.60	0.05
Rainbow Trout count			
intercept	-0.20	-3.76	3.53
zonedownstream	-2.97	-7.23	1.58
zonenutrientaddition	-0.29	-4.92	4.22
periodpre	***-0.80***	***-1.27***	***-0.32***
zonedownstream:periodpre	***0.55***	***0.05***	***1.08***
zonenutrientaddition:periodpre	0.11	-0.31	0.51
Mountain Whitefish count			
intercept	1.22	-1.92	4.22
zonedownstream*	-3.17	-7.19	0.53
zonenutrientaddition	0.77	-2.88	4.73
periodpre*	-0.40	-0.95	0.07
zonedownstream:periodpre	0.45	-0.14	1.09
zonenutrientaddition:periodpre	-0.27	-0.88	0.31
Largescale Sucker count			
intercept	-0.79	-1.81	0.21
zonedownstream	0.27	-0.96	1.61
zonenutrientaddition	0.70	-0.55	1.89
periodpre	-0.02	-0.48	0.52
zonedownstream:periodpre	-0.25	-0.64	0.12
zonenutrientaddition:periodpre	***-0.57***	***-0.96***	***-0.18***

Table 5.5. Results from the generalized linear mixed models on the biomass data. Response variables include biomass for total, Rainbow Trout, Mountain Whitefish, and Largescale Suckers. The table includes the posterior median estimate and 90% highest probability density (HPD) lower and upper credible intervals (CI) for each fixed effect and interaction in the model. Bolded, italicized text indicates parameters with strong support, and asterisks indicate parameters with substantial effect support, as gauged by the HPDs.

Parameter	**Posterior median**	**HPD lower CI**	**HPD upper CI**
Total biomass			
intercept	6.90	5.49	7.97
zonedownstream	-0.80	-2.19	0.40
zonenutrientaddition*	0.61	-0.23	2.32
periodpre*	-0.22	-0.65	0.09
zonedownstream:periodpre	0.06	-0.35	0.46
zonenutrientaddition:periodpre	***-0.39***	***-0.74***	***-0.02***
Rainbow Trout biomass			
intercept	4.97	3.16	7.03
zonedownstream	***-2.25***	***-4.87***	***-0.20***
zonenutrientaddition	-0.11	-2.74	2.10
periodpre	***-0.55***	***-0.95***	***-0.12***
zonedownstream:periodpre	***0.59***	***0.08***	***1.07***
zonenutrientaddition:periodpre	0.06	-0.35	0.51
Mountain Whitefish biomass			
intercept	6.31	4.27	8.55
zonedownstream	***-4.74***	***-7.32***	***-1.91***
zonenutrientaddition	0.25	-2.40	2.84
periodpre	-0.17	-0.65	0.29
zonedownstream:periodpre	***0.62***	***0.04***	***1.22***
zonenutrientaddition:periodpre	-0.09	-0.60	0.46
Largescale Sucker biomass			
intercept	5.74	4.78	6.68
zonedownstream	0.07	-1.12	1.28
zonenutrientaddition*	1.05	-0.20	2.23
periodpre	-0.06	-0.56	0.39
zonedownstream:periodpre*	-0.35	-0.85	0.08
zonenutrientaddition:periodpre	***-0.80***	***-1.24***	***-0.35***

FIGURES

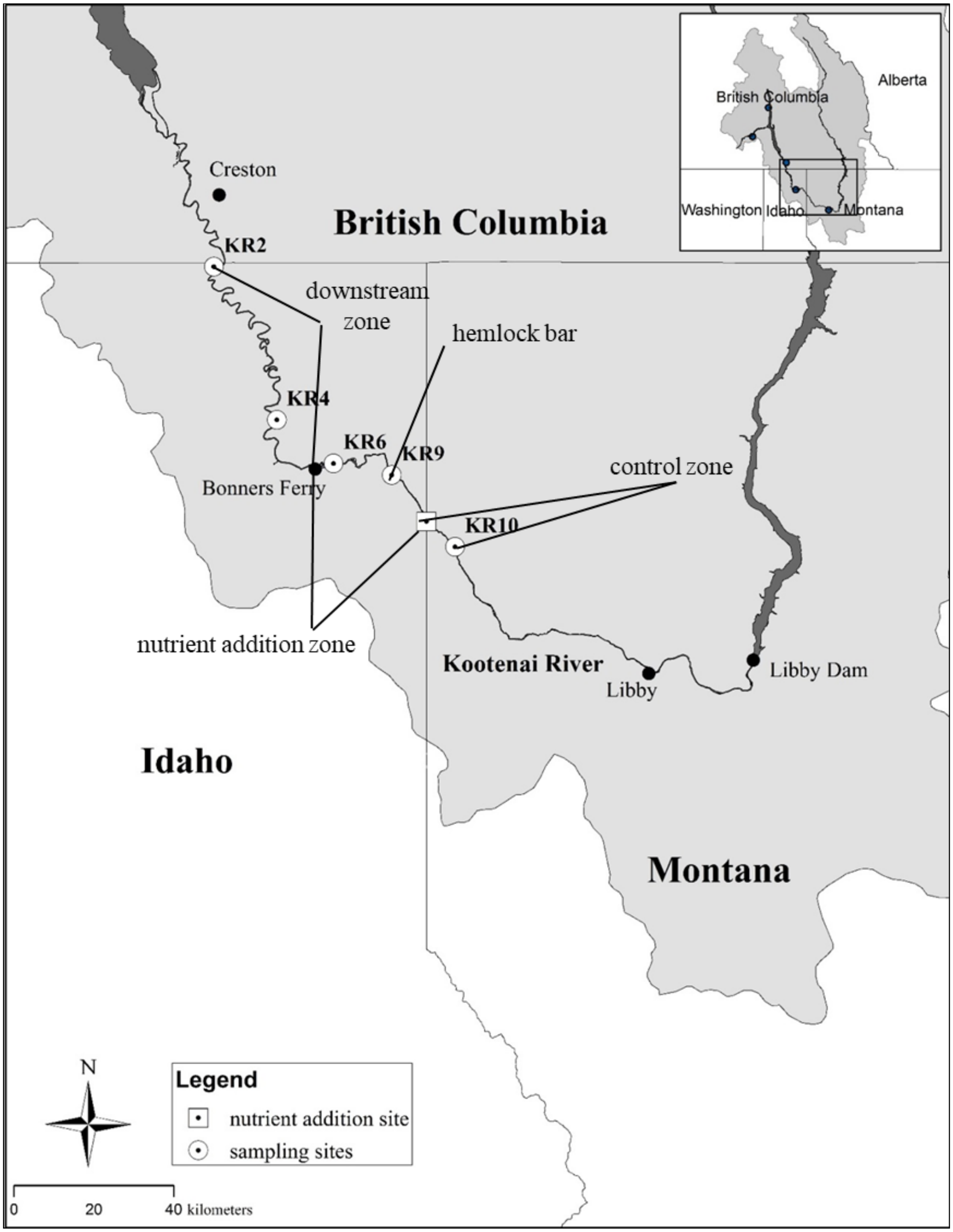

Figure 5.1. Map of the study area in the Kootenai River, Idaho. Shown are Libby Dam, the nutrient addition site, sampling sites, and river zones. The shaded area denotes the Kootenai River watershed.

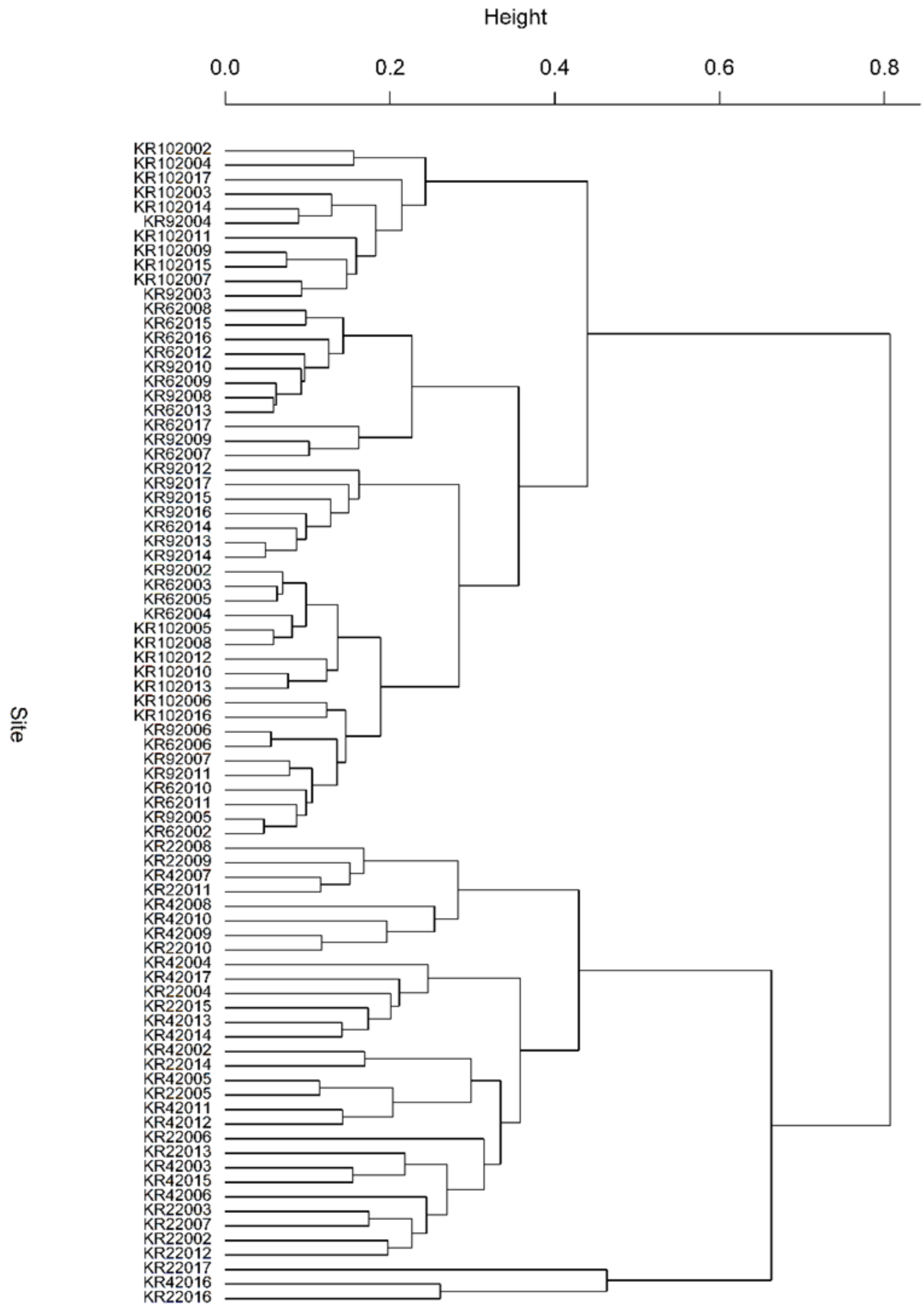

Figure 5.2. Hierarchical cluster analysis of the catch-per-unit-of-effort data. Values on the x-axis represent all site-year combinations for the duration of the study. Cluster “a” is composed of sites within the control and nutrient addition zones during both pre and post nutrient addition periods. Cluster “b” is composed of sites within the downstream zone during both pre and post nutrient addition periods.

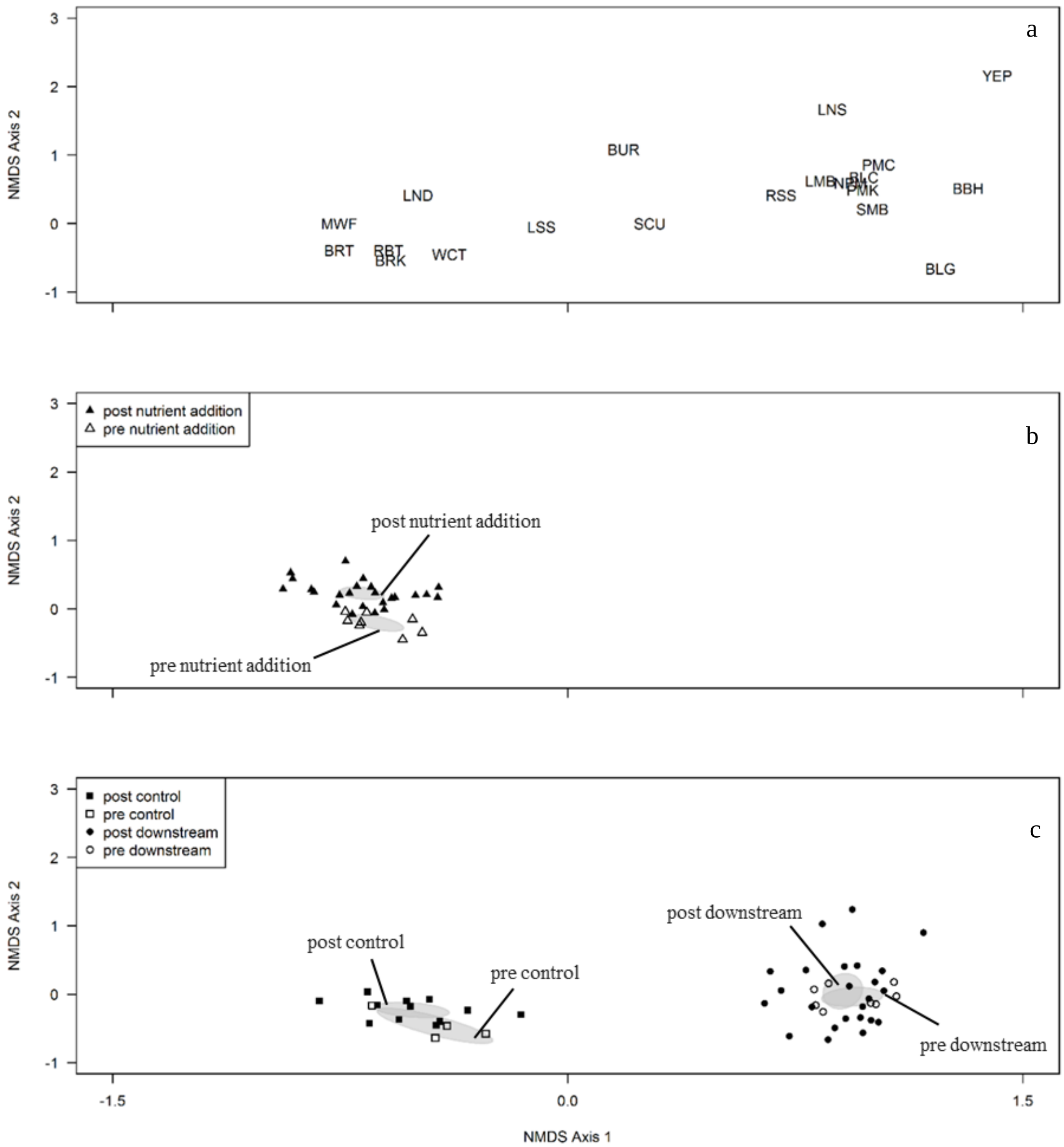

Figure 5.3. Nonmetric multidimensional scaling ordination (stress = 0.06) of the Kootenai River fish assemblage utilizing the catch-per-unit-of-effort data (final iteration of stress: 0.06). Panel “a” displays species scores in the ordination space, panel “b” displays site-year combinations in the nutrient addition zone in the ordination space, and panel “c” displays site-year combinations in the control and downstream zones in the ordination space. Taxa present in panel “a”, include: Brown Bullhead (BBH), Bluegill (BLG), Brook Trout (BKT), Brown Trout (BRT), Burbot (BUR), Black Crappie (BLC), Largemouth Bass (LMB), Longnose Dace (LND), Longnose Sucker (LNS), Largescale Sucker (LSS), Mountain Whitefish (MWF), Northern Pikeminnow (NPM), Peamouth Chub (PMC), Pumpkinseed (PMK), Rainbow Trout (RBT), Redside Shiner (RSS), Sculpin (SCU), Smallmouth Bass (SMB), Westslope Cutthroat Trout (WCT), and Yellow Perch (YEP). Shaded ellipses in panels “b” and “c” depict standard errors in the ordination space.

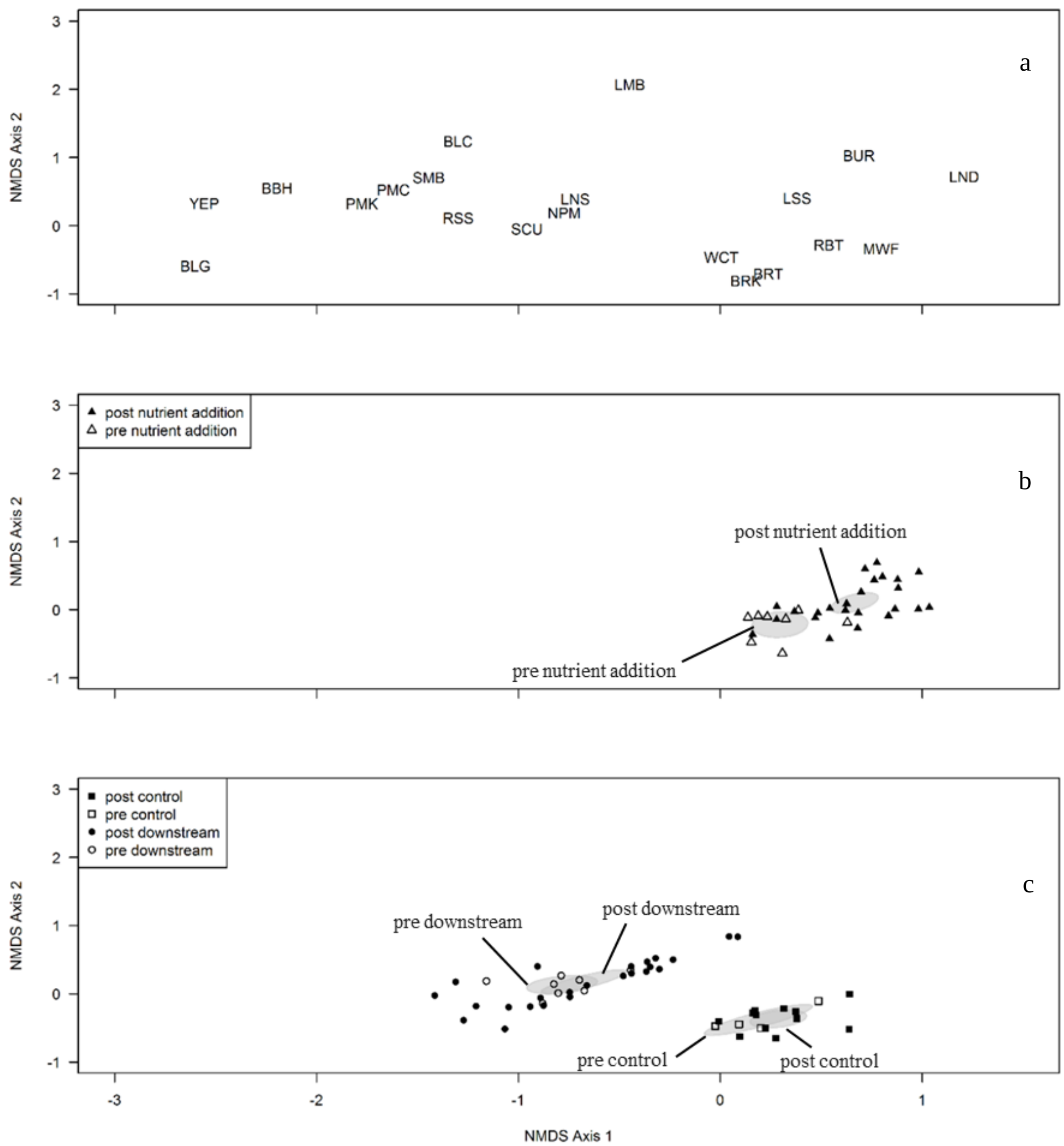

Figure 5.4. Nonmetric multidimensional scaling ordination (stress = 0.09) of the Kootenai River fish assemblage utilizing the biomass-per-unit-of-effort data (final iteration of stress: 0.09). Panel “a” displays species scores in the ordination space, panel “b” displays site-year combinations in the nutrient addition zone in the ordination space, and panel “c” displays site-year combinations in the control and downstream zones in the ordination space. Taxa present in panel “a”, include: Brown Bullhead (BBH), Bluegill (BLG), Brook Trout (BKT), Brown Trout (BRT), Burbot (BUR), Black Crappie (BLC), Largemouth Bass (LMB), Longnose Dace (LND), Longnose Sucker (LNS), Largescale Sucker (LSS), Mountain Whitefish (MWF), Northern Pikeminnow (NPM), Peamouth Chub (PMC), Pumpkinseed (PMK), Rainbow Trout (RBT), Redside Shiner (RSS), Sculpin (SCU), Smallmouth Bass (SMB), Westslope Cutthroat Trout (WCT), and Yellow Perch (YEP). Shaded ellipses in panels “b” and “c” depict standard errors in the ordination space.

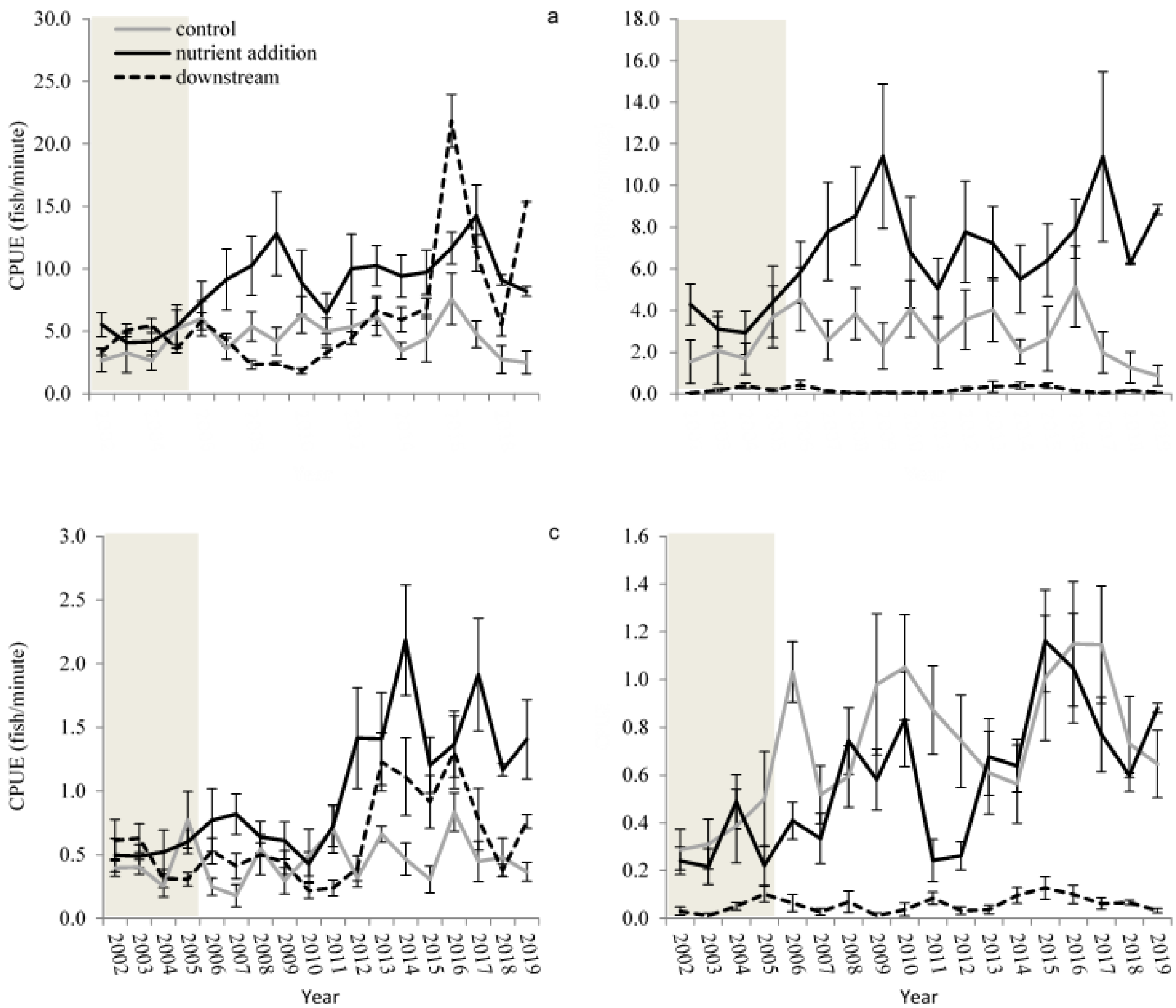

Figure 5.5. Mean catch-per-unit-of-effort (CPUE) in the control, nutrient addition, and downstream zones of the Kootenai River, Idaho. Years represented are from 2002-2005 (shaded gray; pre nutrient addition) and 2006-2019 (no shading; post nutrient addition). Error bars are +/- one standard error, calculated for each year and river zone combination. Shown are mean CPUE values for (a) total (i.e., all species, combined), (b) Mountain Whitefish, (c) Largescale Sucker, and (d) Rainbow Trout. Data from 2018 and 2019 were not included in formal analyses and are not included in Hardy et al. (*In Review*); however, they are shown here in support of the 2018 and 2019 IDFG annual reports.

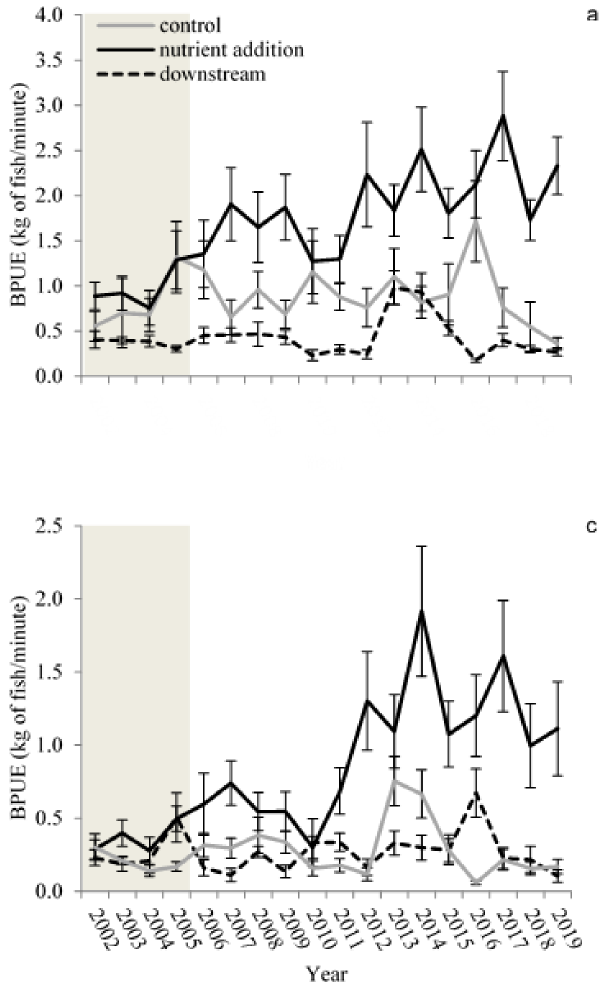

Figure 5.6. Mean biomass-per-unit-of-effort (BPUE) in the control, nutrient addition, and downstream zones of the Kootenai River, Idaho. Years represented are from 2002-2005 (pre nutrient addition) and 2006-2017 (post nutrient addition). Error bars are +/- one standard error, calculated for each year and river zone combination. Shown are mean BPUE values for (a) total (i.e., all species, combined), (b) Mountain Whitefish, (c) Largescale Sucker, and (d) Rainbow Trout. Data from 2018 and 2019 were not included in formal analyses and are not included in Hardy et al. (*In Review*); however, they are shown here in support of the 2018 and 2019 IDFG annual reports.

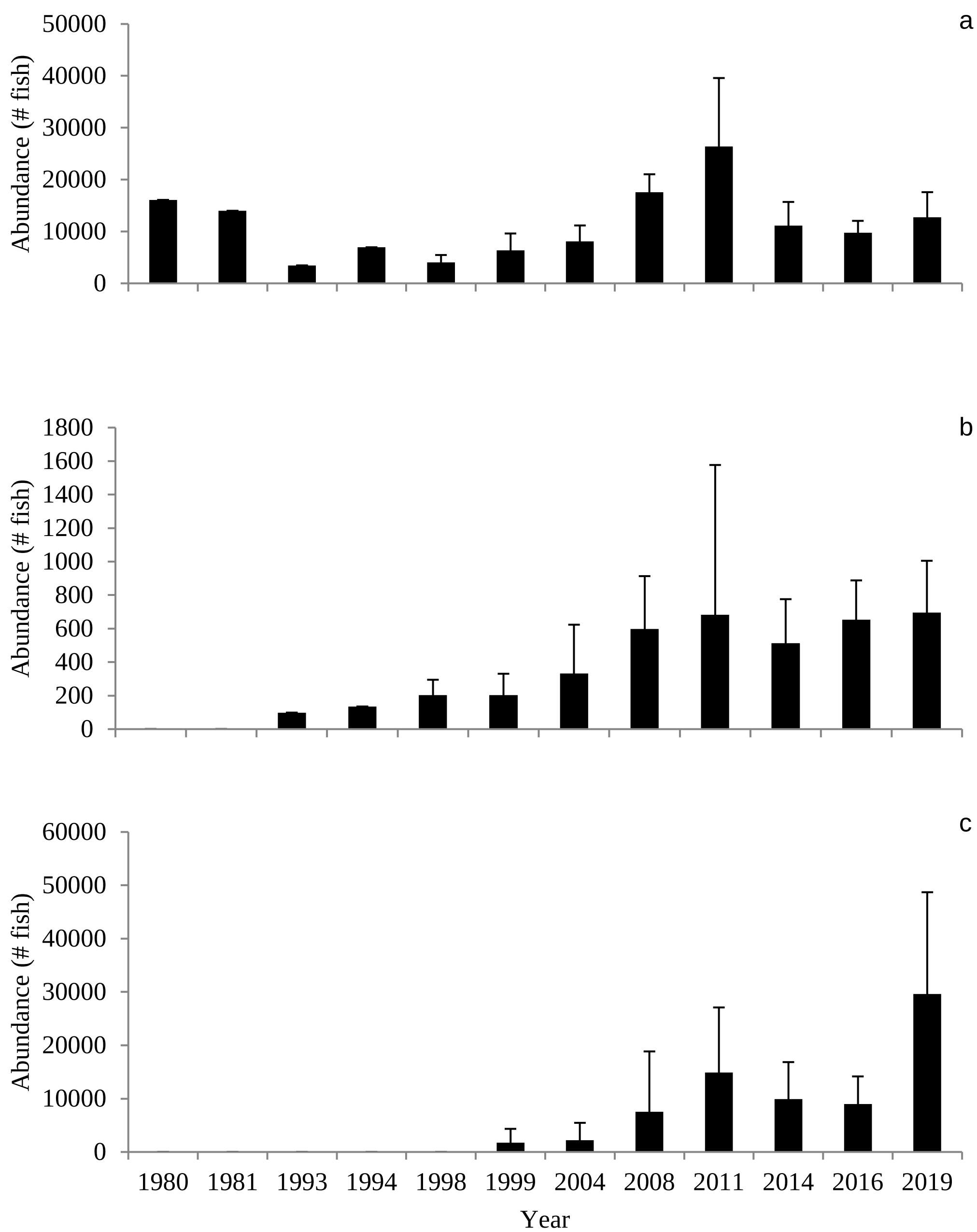

Figure 5.7. Abundance estimates for (a) Mountain Whitefish, (b) Rainbow Trout, and (c) Largescale Suckers in a three-kilometer reach of the Kootenai River located in the nutrient addition zone. Estimates represent the number of fish of each species within the river reach, and error bars represent 95% upper and lower confidence intervals. Data from 2019 were not included in formal analyses and are not included in Hardy et al. (*In Review*); however, they are shown here in support of the 2019 IDFG annual report.

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APPENDIX

APPENDIX 1: 2019 BIOP REPORT

As per BPA instructions, this Biological Opinion (BIOP) report is attached to the end of this document to fulfill in-season reporting requirements for the previous calendar year. This report serves as a link to the Libby BIOP, which can be found on BPA's Columbia Basin Fish and Wildlife website (www.cbfish.org).

SECTION 1: IMPLEMENTATION

Operation of Libby Dam for hydropower and flood control has significantly changed seasonal flows in the Kootenai River relative to historical flow regimes. Generally, both springtime peak flows and base flows are lower than what they were prior to the installation of Libby Dam. Additionally, during the last 100 years almost all of the lower portion of the river below Bonners Ferry, Idaho has been diked. These two factors, dam operations and dike construction, prevent the Kootenai River from inundating most of its historic floodplains which has caused the river to become incised and deeply channelized. The channelization of the Kootenai River has caused the degradation of spawning and rearing habitats that Kootenai River White Sturgeon *Acipenser transmontanus* rely on. Habitat degradation and flow alteration have created unfavorable ecological conditions for consistent, successful Kootenai River White Sturgeon recruitment. Since the installation of Libby Dam there has only been two years with high recruitment rates of juvenile White Sturgeon (1974 and 1991). Both these years experienced exceptional precipitation and river discharge.

The Kootenai River White Sturgeon was listed as an Endangered Species in September of 1994. The listing was consistent with the population abundance and genetic status. Genetic analysis of the Kootenai River White Sturgeon in 1991 indicated this population was genetically distinct from other populations of Sturgeon in the Columbia basin. At the request of the Kootenai River White Sturgeon Steering Committee (comprised of representatives from the agencies and tribes), the U.S. Army Corps of Engineers has provided mitigating, experimental flows for White Sturgeon spawning and rearing since 1991. The objective of this investigation is to determine flow and habitat conditions that will affect recovery of this population. This study is supported by, and adheres to conditions set by the Recovery Plan for the Kootenai River White Sturgeon.

This information is presented in accordance with the lettered or numbered items identified in the "Special Terms and Conditions" for the Subpermit (dated October 14, 1997). The USFWS has issued three opinions on Libby Dam (1995, 2000, and 2006). In accordance with these Biological Opinions, this project is listed as necessary and appropriate. The 2006 Libby BiOp specifically lists Reasonable and Prudent Alternatives (RPA) that our IDFG sponsored program is directly responsible for either for implementation or monitoring and evaluation of mitigation actions. A list and description of the RPA components and their associated actions in listed in http://www.cbfish.org/Project.mvc/Display/1988-065-00. Results from our 2019 investigations are listed below.

SECTION 2: RESULTS

Adult Movements

In 2019, as in other years, Idaho Department of Fish and Game (IDFG) personnel monitored spawning migrations of sonic-tagged Sturgeon in the Kootenai River downstream and upstream of Bonners Ferry (Figure 1). IDFG and BC deployed 89 acoustic receivers throughout Kootenay Lake and Kootenai River to detect the movements of tagged individuals. Forty-five sonic-tagged adult Sturgeon were detected above Shorty's Island (river kilometer [RKM]) 230; Table 1). Of these 45 Sturgeon, 19 were documented as far upstream as Ambush Rock (RKM 244.5), and 5 were detected in the braided reach above the Highway 95 Bridge (RKM 246). The proportion of spawning adults (adults that move up to ≥ rkm 230) in 2019 was lower compared to previous years (Figure 2). This reduction in movement is likely due to the reduction in duration and amount of flow from Libby Dam. During the last six years, the Kootenai Tribe of Idaho (KTOI) has implemented several habitat restoration projects above Bonners Ferry. One of the goals of these projects was to provide attractive holding habitat for spawning Kootenai River White Sturgeon. Evaluation of upstream movements in relation environmental variables showed that the duration of high flow as well as the addition of the habitat projects have had positive influences on adults moving upstream of Bonners Ferry into proper spawning habitat. The next few field seasons will be aimed at understanding the extent of spawning in these reaches and if it is resulting in increased recruitment.

Spawning and Early Life History Monitoring

Sturgeon spawning habitat quality is critical to successful egg deposition and hatching. Poor Sturgeon spawning habitat quality in the Kootenai River has been identified as a potential limiting factor responsible for lack of recruitment into the population for over 30 years. IDFG systematically monitors egg deposition location with artificial substrate egg mats in the Kootenai River and in 2019 reported that a total of 156 eggs were collected between May 23rd and July 11th (Table 2). Eggs were sampled from rkm 226–245. Two other sites were added (rkm 246 and 248) in 2919 above Ambush Rock, yet no eggs were sampled. Sampling effort was similar between the Myrtle Creek and Shorty's Island sites.

In addition to monitoring egg deposition, IDFG also tracks hatching success through larval sampling. Larval sampling is done through the use of passive drift nets that are anchored to the substrate. Low flows and turbidity allowed for overnight sampling to occur in 2019. Sampling was focused below Shorty's Island and Myrtle Creek spawning locations as well as in the straight reach. In 2019, despite >3,000 hours of total fishing effort between June 24th and July 17th, only a single larval Sturgeon was captured at Shorty's Island (Table 2).

Gillnetting by IDFG and British Columbia Ministry of Forest Land and Resource Operations (FLNRORD) personnel was conducted from Ambush Rock downstream to Kootenay Lake, including both the Kootenay River delta and the Lardeau River delta at the north end of the lake to determine density, distribution, and length-frequency and age distribution of hatchery reared and wild juvenile White Sturgeon in the system. Sampling in 2019 occurred between July 22nd and September 11th. We used gill nets with panels including 2.5, 5.1, and 7.6 cm bar mesh. Soak time for our gill net sampling ranged from 60 to 90 minutes to minimize risk of accidental mortality. All Sturgeon were measured, weighed, scanned for PIT tags, and released. If no PIT tag was found, a new tag was implanted in the individual. Combining IDFG and FLNRORD efforts, a total of 1,876 (566 in BC) juvenile Sturgeon were captured in gillnets in 2019 (Table 2). Five of the sampled juveniles were of wild origin (three captured in US and two in BC). All wild origin fish were aged by removing a portion of the pectoral fin ray. There were no mortalities.

Sampling in Idaho and Canada by IDFG or FLNRORD for adult Sturgeon commenced on March 18th and continued through October 23rd, 2019. Two gear types were used: rod and reel angling, and setlines with 14/0, and 16/0 circle hooks set with six to eight hooks per line. A total of 145 adult Sturgeon were captured with setlines and angling in 2019 (Table 2). Of these 145 captured, 119 were of wild origin and 26 were of hatchery origin. Hatchery origin fish are considered adults once they grow past 120cm TL. 1,864 hatchery reared juvenile Sturgeon were captured in 2019. Of the wild adult Sturgeon captured in Idaho and BC in 2019, approximately 95% were recaptures from previous years. Seven adult and nine juvenile Kootenai Sturgeon were tagged with special Vemco V16 VPS sonic transmitters in 2019 as part of a habitat selection study as well as overall influence of flow on adult movements to proper spawning habitats.

SECTION 3: FUTURE MANAGEMENT

Based on the U.S. Fish and Wildlife Service's (Service) February 2006 Biological Opinion (2006 BO) on operations of Libby Dam, and the May final April-August volume runoff forecast of 4.98 million acre-feet (MAF), we are within a Tier 2 operations year for Kootenai River White Sturgeon. The minimum recommended release volume for Sturgeon conservation in a Tier 2 year is 0.80 MAF, and we recommend the following procedures for discharge of at least this minimum volume from Libby Dam: The precise means that will be utilized to meet these objectives are largely dependent on real-time conditions and in-season management. It is not possible to develop a single definitive recommendation for a Sturgeon operation at this time due to the uncertainties in the forecast, and shape and volume of inflow. Given these uncertainties, the Service has developed the following guidelines for Sturgeon operations in 2019. Specific details on 2019 Libby Dam Sturgeon operations is available at:

http://pweb.crohms.org/tmt/sor/2019/0515_2019_FINAL_Libby_Sturgeon_SOR.pdf

Although we are still constrained by Libby Dam operations and flood control issues at Bonners Ferry, small-scale flow management actions are important for understanding how Sturgeon respond to different flow regimes and eventually may allow us to increase the proportion of the spawning population that migrate above Bonners Ferry. The Vemco telemetry array (which currently consists of 89 receivers throughout Kootenay Lake and Kootenai River) has been deployed in the Kootenai River for 16 years and has greatly improved our understanding of qualitative aspects of Sturgeon movements and behaviors. Our evaluation of extended high flows as well as the construction of habitat showed positive influences on adult movements above Bonners Ferry. The next step is to incorporate movement data with some specific physical habitat variables (e.g. amount of spawning habitat, number of deep pools, etc.) to further quantify the influence of these improvements to spawning.

In addition to spawning habitat use, we have moved forward on analysis of specific metrics from hatchery produced juveniles to aid in determining population demographics as it relates to stocking strategies. With the anticipated continued stocking of hatchery-reared Sturgeon, it is evident that these fish can fulfill a continued useful role for research. One of our key objectives is to refine recommendations for stocking rates and release strategies to meet abundance and diversity objectives while minimizing impacts to wild production. There is a need to evaluate changes in growth rates over time to determine what, if any, effects stocking density or other habitat improvements are having on growth. Growth analysis of hatchery juveniles has proven difficult due to transition between river (slow growth) and lake (fast growth) environments. Additional research to determine transition rate, affinity to stocking location and cohorts, as well as diet may allow us to determine the major factors influencing growth and survival in the Kootenai system.

Table 1. Extent of movement of tagged adult Kootenai River White Sturgeon since 2005. RKM 229 is at Shorty's Island. RKM 264 is upstream of the Moyie River. Blue shaded area represents the Straight Reach (RKM 240-246); green shaded area represents Braided Reach (RKM 246-257). Fish movement is depicted as numbers of fish observed at receivers located at a particular RKM. Blank cells indicate a receiver was not present at that RKM for a given year.

RKM	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
230.1			28	29	23		35	34	43	44	49	49	48	47	45
230.5	13	31													
231											46				
232	16	29	27	28	21		34	34	43	45		48	54	52	45
235			28	25	20	36	35	32	42		48				43
235.2	15	27													
239	12	23													
240			22	25	15	32	25	27		35	47				33
240.3											38				
240.7			21	23		31	25	25	34	34		36	43	41	38
240.9											38				
243.5		15	13	19		24	19	19	22	23	29	29	37	31	19
244.5		12	0												
244.7			14	18	6	20	16	16	20	21	26	29	35	27	16
245		3	0												
245.5	7	0													
245.6			13	15	6	13	11	13	16	19	22	23	28	21	10
245.7						15	13	14	15	20	23	25	29	23	11
245.8		9	0												
245.9		9	0												
246						9	13	13	11	12	14	17	24	18	5
246.6		5	7	13	3	7	6	10	12	11	11	18	25	18	5
246.7		5	0												
246.8								7	0	10	9	16	25	17	5
247.3		1	0												
247.99													13		
248	0	0			0	0	1	2	1	2	0	0	0	0	0
248.1						2	3	5	6						
248.2					0	0									
248.5						2	4	4	6	2	1	3		4	0
248.6		2	0												
248.8		0													
249.5						0	0	2	2	0	0	0	3	5	0
249.55													2		
249.6		0	1												
250				1	0	0	0	0	0	0	0	0	0	2	0
250.4									0	0	1	1	2	2	0
250.7			0	4	0	0									
250.9				0	0	1	0	0	1	0	0		0	2	0
253.4			0	2	0	1	1	0	1	0	1	2	2	4	0
254.5					0	0	1	0	1	0	0	0	0	0	0
255.1				0	0	1	1	0	1	0	1	0	2	2	
256	0	0	0												
256.1			0												
257.5				0	0	1	0	0	1	0	1	0	1	2	0
258.7								0	0	0	0	0	0	0	0
264			0	1	1	0	1	0	1	0	1	0	1	2	0
268.5	0	0	0	1	0	0	1	0	1	0	1	0	1	2	0
273.5				0	0	0	1	0	1	0	0	0	1	1	0
275.5	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
278.9								0	1	0	0	0	1	0	0
282				0	0	0	0	0	1	0	0	0	0	0	0
282.5			0	0											
300.4								0	0	0	0	0	0	0	0
305								0	0	2	1	2	1	1	0

Table 2. Summary of IDFG and FLNRORD Kootenay Lake and Kootenai River White Sturgeon sampling efforts in 2019 under US Fish and Wildlife Service Permit 702631.

Target	Sampling Dates	Adults	Juveniles	Larvae	Eggs	Mortality	Gear Type
Adult	3/18/2019 - 10/23/2019	145	57	-			Rod and Reel, Set Lines, (14/0 and 16/0 Hooks)
US Juvenile	7/22/2019 - 9/11/2019	1	1253	-			Gill Net (2", 4", and 6" stretch mesh)
B.C. Juvenile	7/15/2019 - 9/23/2019	2	566				
Egg	5/23/2019 - 7/11/2019	-			156 eggs on 24 mats	156	Artificial substrate egg mats
Larvae	6/24/2019 - 7/17/2019	-		1		0	Paired larval plankton nets
Totals		148	1876	1	0	156	

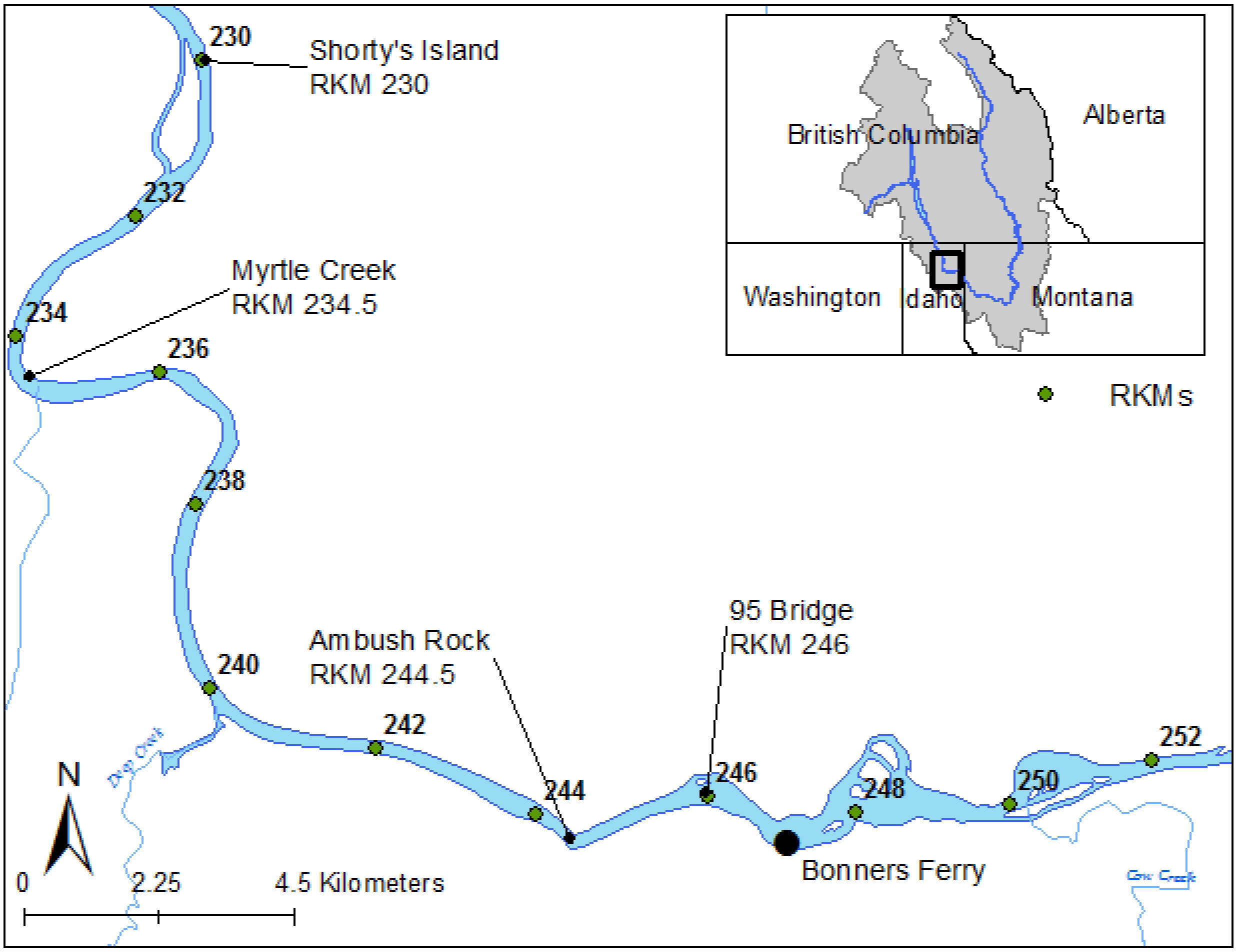

Figure 1. Kootenai River White Sturgeon spawning reach near Bonners Ferry, Idaho. Pictured above are the river kilometers (RKMs) where the majority White Sturgeon spawning occurs. RKM delineations begin at the north end of Kootenay Lake and increase as one moves up stream. USFWS Critical Habitat designation for Kootenai River White Sturgeon is RKM 230 (Shorty's Island) to RKM 158.5 (Moyie River confluence).

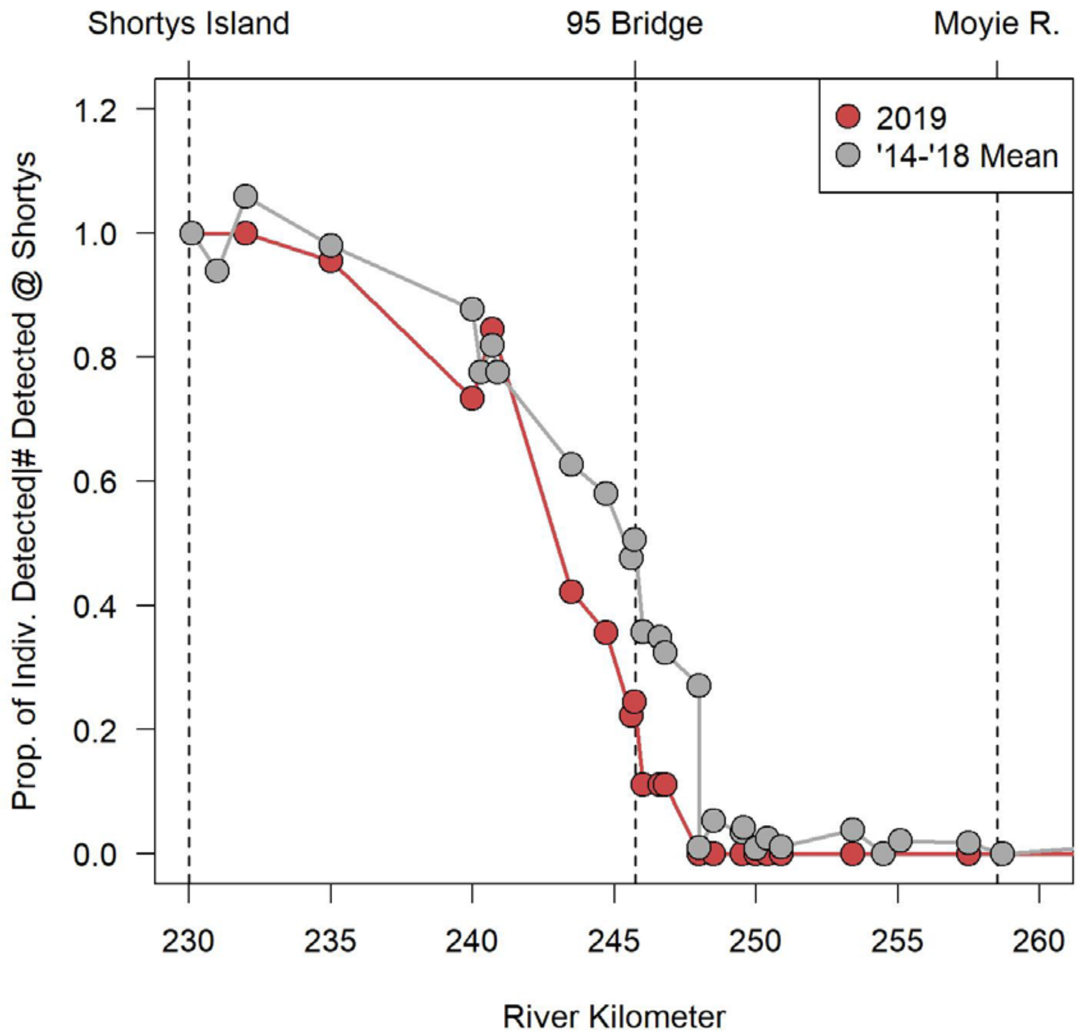

Figure 2. Proportion of individually unique detections of adult Kootenai Sturgeon tagged with acoustic transmitters by river kilometer.

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