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Protocol –

Native stream fish occupancy monitoring for Banff National Park

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1.0 INTRODUCTION

1.1 THE NEED FOR LONG-TERM STREAM FISHES MONITORING

Freshwater fishes are some of the most imperiled taxa worldwide as evident by the number of threatened and endangered species (Strayer and Dudgeon 2010). For example, the Alberta population of westslope cutthroat trout (WSCT; *Oncorhynchus clarkii lewisi*) has been assessed as "threatened" provincially and nationally (Fisheries and Oceans Canada 2013). This species has also been listed under the Species at Risk Act (Fisheries and Oceans Canada 2013). In Banff National Park (BNP), native WSCT populations have been reduced by almost 80% of their range. The remaining, genetically pure native populations (believed to be 10 in BNP) persist as severely fragmented, remnant headwater populations. Bull trout (BLTR; *Salvelinus confluentus*) are also listed as a Species of Special Concern under the Alberta Wildlife Act. The Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2014) has recommended BLTR to be listed listing as Threatened for the Nelson and Saskatchewan Rivers populations. Given this, Parks Canada Agency (PCA) has been a part of a multi-agency effort to recovery these species and ensures their persistence.

Threats to WSCT, BLTR, and inland fisheries in general, can be broadly classified as habitat degradation, invasive species, water pollution, nutrient loading, flow modification and fragmentation (Dudgeon et al. 2006). In addition, there are larger scale threats caused by warming shifts in precipitation and runoff (Richter et al. 1997; Dudgeon et al. 2006). Indeed, US fisheries scientists have modeled the predicted effects of climate change on cutthroat trout abundance and distributions (e.g., Williams et al. 2009; Al-Chokhachy et al. 2013; Wenger et al. 2011). Warmer water, changes in stream flow and increased frequency and intensity of other disturbances (e.g. flooding, non-native species invasion) are among the factors associated with climate change that have been shown to effect cutthroat trout populations. Studies from the U.S suggest that climate change could also reduce thermally suitable habitat for BLTR (Rieman et al. 2007; Jones et al. 2014). Stream and river temperatures are rising in the United States (Kaushal et al. 2010). Therefore, despite a lack of this knowledge for the Canadian Rockies, it is likely that stream temperatures have also been rising. Furthermore, brook trout (BKTR) exist in nearly all native fish habitat in BNP which leads to the displacement of WSCT and BLTR via competition and introgression (Rieman et al. 2006). Climate change is expected to facilitate these interaction (Rahel and Olden 2008) due to the relatively higher thermal tolerance of BKTR (McMahon et al. 2007). Given this, Parks Canada Agency (PCA) may need increased control efforts for non-native fishes under certain climate change scenarios.

Strategies to preserve freshwater biodiversity in the long term include reserves that protect key water bodies and species at risk. For the sake of BLTR, this includes high elevation, cold water bodies. In parallel, scientists must quantify and communicate a decline in freshwater resources to stakeholders and policy makers so they can understand the losses, future risks and make informed decisions (Cooke et al. 2013). The PCA

State of the Parks report is a tool that is used as a “report card” to stakeholders and the public, and as a document that highlights important issues that need to be addressed in the Banff National Park Management Plan. Another framework for communicating the loss of biodiversity is the Alberta Fisheries Sustainability Index, a provincial initiative which compares current population estimates to historic estimates as a means of reporting on the current status of recreational fisheries in Alberta. However, PCA’s past sampling methods for freshwater species such as WSCT and BLTR have been inadequate to satisfy the requirements of all of these conservation initiatives. This problem is not unique to Banff National Park (ASRD 2012; COSEWIC 2012). The reason for the lack of quantitative estimates for species declines in BNP is likely due to limited resources in the past, difficulty in accessing some remote locations and the overall difficulty of knowing how, when and where to sample stream fishes in a statistically-defensible manner.

1.2 OBJECTIVES

Much of the stream fish monitoring literature is dedicated to testing the assumptions related to quantitative abundance monitoring using mark-recapture. For example, in order to estimate trout abundance (e.g. density or biomass), one must know something about the capture efficiency of the species in question. Estimating unbiased capture efficiencies is time consuming, expensive and challenging. Therefore, previous work monitoring salmonids has revealed several limitations to monitoring abundance, including: (1) Low statistical power (Ham and Pearsons 2000; Maxell 1999), (2) errors in detecting abundance (Dunham et al. 2001; Peterson et al. 2004), (3) High population variability (Platts and Nelson 2004), (4) a weak connection between abundance and habitat (Fausch et al. 1988) and (5) the high cost of rigorous abundance estimates, which limit the number of sampling sites and the geographic scale of inference (Al-Chokhachy et al. 2005; USFWS 2008). Many of these problems are exacerbated for bull trout which often exist in low densities.

Monitoring approaches that focus on species distributions and temporal patterns of occurrence overcome many of the limitations associated with abundance monitoring and are being broadly adopted (MacKenzie et al. 2002, 2006). This approach involves estimation of the proportion of an area that is occupied by a target species (MacKenzie et al. 2006). Monitoring the distribution of a species requires less intense sampling at individual sites than measuring abundance. Therefore, distribution monitoring makes it possible to sample larger areas for many species and provides information at scales relevant to the species at risk communication tools mentioned above (i.e. State of the Parks Report; provincial FSI). Indeed, the benefits of distribution sampling are being realized by many researchers and managers as shown by the increasing rate of publication of occupancy models for stream fishes (e.g. Albanese et al. 2011; Wagner et al. 2013; Dextrase et al. 2014; McManamay et al. 2014; Midway et al. 2014; Rodka et al. 2015). A scientifically-defensible occupancy survey program in Banff National Park can be used to track the decline of native fish due to environmental change, but more importantly, the replacement (and/or displacement) of native species by non-native species.

However, there is still a large number of considerations for sampling design in order to estimate occupancy. Advice is available in the literature, but sampling programs that are appropriate for some regions are not necessarily appropriate for BNP. Many of the published occupancy studies are from the eastern U.S. which have different habitat and species diversity (Albanese et al. 2011; Wagner et al. 2013; Dextrase et al. 2014; McManamay et al. 2014; Midway et al. 2014). In addition, considerations for spatial sampling design, sample size, and detection probability are not always straight-forward to understand. Given these concerns, this report was designed as a literature review to inform an occupancy monitoring protocol in addition to the protocol itself. The report is structured in sections that are relevant to each component of an occupancy survey program (e.g. spatial sampling design, estimation of detection efficiency; habitat predictors of occupancy). While not exhaustive, each section is a summary of knowledge from the current literature as a way of informing the multiple choices made in this protocol. There is only a small amount of literature specifically related to BLTR and WSCT occupancy surveys. So, I reviewed abundance monitoring studies when the information was also relevant for occupancy surveys. Following each section, there is a bulleted list of the main key points that make of the actual protocol. I avoided using step by step instructions for the actual electrofishing surveys, as the general protocols for electrofishing is already known by fisheries managers. Instead, I focused on the main nuisances specifically associated with occupancy monitoring.

The report has the following objectives:

1. Review the literature relevant to stream fish occupancy monitoring programs.
2. Outline key considerations for a scientifically-defensible occupancy survey to quantify the distribution of native and nonnative species across Banff National Park.

2.0 SAMPLING DESIGN

2.1 SPATIAL SAMPLING DESIGN

There has been a lot of scientific interest in BLTR sampling in the US due to the monitoring requirements for recovery (See Isaak et al. 2009; USFWS 2008). Bull trout are at the southern extremity of their range in the U.S. Therefore, they exist in a series of patches; usually at the highest elevations in mountain landscapes and are often separated by unsuitably warm stream lengths. This restrictive distribution of bull trout make patch-based assessments appropriate. A bull trout patch is a network where temperatures are cold enough to support spawning and early juvenile rearing (Dunham et al. 2003). Given the distribution of bull trout, a number of bull trout sampling programs have been designed to reflect this patch network (Dunham and Rieman 1999;

Dunham et al. 2003) and most sampling guidance from the U.S. is also constructed this way (Isaak et al. 2009; USFWS 2008).

In contrast, all stream networks in Banff National Park are likely cold enough to support bull trout. Bull trout are broadly distributed across the park. Their occupancy at any specific site may be more related to the presence of nonnative fishes rather than being confined by temperature. However, a “patch design” still provides advantages in BNP because it provides groupings of sites that based on watershed delineations that can be randomly selected for sampling in the case that the entire watershed cannot be sampled. It also provides groupings that are logical on the watershed scale for determining individual, creek-based occupancy and detection estimates. Therefore, we sampled a randomly within patches and patches themselves would be chosen randomly in the case of really large watersheds.

We reviewed the stream fish monitoring literature to determine how different monitoring programs had structured sampling sites in the past. Unfortunately, the practical limitations of monitoring programs in the wilderness (i.e. moving personnel/gear in remote locations) combined with limited resources by most monitoring agencies, has resulted in non-random selection of sampling sites in most studies. We found that many monitoring agencies did not consider random sampling because they wanted sites that were accessible by road (e.g. Thurow et al. 2004). Unfortunately, this is insufficient, from a statistical point of view, for extrapolating trends across the region (Pollock et al. 2002). Indeed, the probability of occurrence for bull trout in Alberta is directly related to their proximity to roads (Alberta SRD 2012; Yau and Taylor 2013). Therefore, road-side sampling would bias the results, limiting the ability to predict trends across the whole park. Probabilistic designs are a better approach because the site selection is randomized. Since each site has an equal chance at being selected, statistically valid, unbiased estimates can be extrapolated to an entire region (i.e. watershed; Isaak 2009). However, purely random selection can result in spatial clustering of sites that may not represent the environmental gradients that occur across watersheds. To address this, we used the Generalized Random Tessellation Stratified design (GRTS; Stevens and Olsen 2004) which uses a randomized hierarchical grid that arrays sites throughout a stream network to achieve spatial representation.

Random sites should also be stratified by variables that are likely to be predictive of the outcome of interest (e.g. native species occupancies) to not bias the results towards a certain outcome. Furthermore, if a stratified random sample is used, then data can be analyzed for each stratum separately (MacKenzie et al. 2006). Given that we are interested in the relative proportion of native versus non-native species, it is reasonable to stratify sampling by the variable that is most likely to affect the presence of native versus non-native species. A number of studies have identified temperature as a driving factor in predicting the occurrence of native versus non-native trout species (Paul and Post 2001; Dunham et al. 2003; Dunham and Rieman 1999). For example, Paul and Post (2001) found that brook trout are generally found at lower elevations in the Canadian Rockies compared to native species as the result of a combination of stocking history and their preferential downstream movement. Furthermore, only temperature was strongly associated with the distribution of bull trout in the western U.S. (Dunham et al. 2003). Given this, random sites could be stratified by elevation as a proxy for temperature (Isaak et al. 2009). Similarly, bull trout are often excluded from relatively steep sections of stream as density declines as slope increases (Dunham and Rieman 1999; McCleary and Hassan 2008). We would argue that steepness of the streams is correlated with elevation (i.e., steeper streams occur at higher elevations). In the end, we felt that stream order (Strahler 1952) is a good proxy for stream size, gradient and elevation. Stream-order can be derived using GIS and is therefore appropriate for sites with no measured physical stream characteristics. Lower order streams tend to be smaller size, steeper, and at higher elevations. Higher order streams tend to be a larger size, lower-gradient, and at lower elevation. Therefore, we stratified patches by Strahler-stream order (1^{st}, 2^{nd}, and 3^{rd} order) at the 1:50,000 map scale. Higher stream orders (> 3^{rd} order) cannot be surveyed effectively by backpack electrofishing. Sites were allocated using spsurvey v.2.5 (Kincaid et al. 2015) in R (R Core Team 2013), and following the criteria below:

- *One watershed/year was chosen for a comprehensive occupancy survey. In three years, three entire watershed will be surveyed.*
- *All stream patches within each chosen watershed were assigned sites, although a random selection of patches could be chosen from very large watersheds.*
- *A probabilistic design was used: sites were randomly assigned while stratifying by stream order (1^{st}, 2^{nd}, and 3^{rd}; USFWS Bull Trout Recovery Team).*
- *Sites were allocated proportionally by the length of streams across stream orders to match the distribution of stream orders on the landscape.*
- *Stream segments with slopes > 15% were excluded as well as stream segments upstream of excluded segments.*
- *Streams with waterfall barriers were not excluded without prior field-based knowledge to confirm the presence of a barrier. However, by excluding streams > 15%, gradient barriers would have been indirectly excluded.*
- *One hundred sites were assigned from the first watershed sampled (Cascade) and additional watersheds were sampled according to its proportional area relative to the Cascade (i.e. proportional allocation; Peterson et al. 2002).*

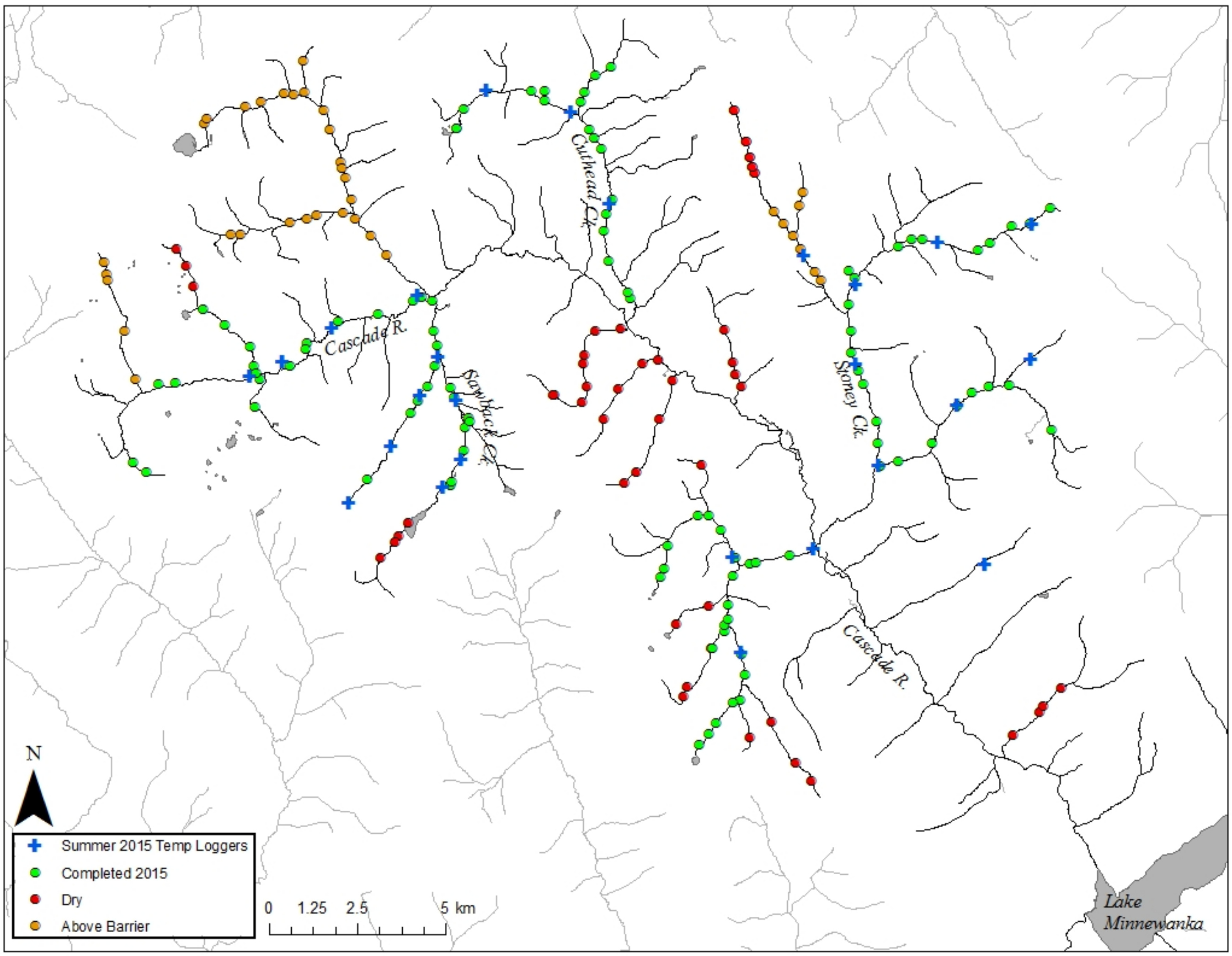

Figure 1. Map of randomly chosen sites, stratified by stream order, from the Cascade watershed (2015). Some sites were determined to be upstream of barriers. Some 1st order streams were dry when visited in 2015.

2.2 SAMPLING SITES

Backpack electrofishing is the most efficient method of sampling relatively small streams for occupancy. Some 3rd order streams can be deep in low-gradient habitat and electrofishing capture efficiency would be inconsistent with more traditional habitat (e.g. parts of the Upper Spray River near Palliser cabin).

- *Backpack electrofishing methods will be used to sample each site.*
- *Each site will be sampled by a three person crew (one electrofisher, one netter and one bucket person) using a single pass at each visit. Only one netter is to be used in order to maintain a consistent effort across all sites.*
- *Electrofishing settings will be adjusted according to water conditions and fish response. Adjustments will be made to illicit just enough of a forced swimming response by fishes that they can be netted with a dip-net. Settings will be determined early in the sampling window and then the same settings will be used throughout the watershed. Site-specific differences in electrofishing efficiency will be accounted for by testing for the effect of sampling-level covariates on detection probability.*
- *Fish will be identified to species, enumerated and measured.*

2.3 REPEATED SAMPLING AND FALSE ABSENCE RATE

Of concern with occupancy surveys is accounting for the imperfect detection of the species of interest (i.e. “false-absence rate”; MacKenzie et al. 2006). One of the premises of any occupancy protocol is that the probability of detecting the species in a survey can be estimated. Generally, this requires repeated surveys of at least a subset of sites (Mackenzie et al. 2006). Repeated surveys provide the necessary information about the chances of detecting a species so that occupancy of that species can be corrected for the fact that a non-detection doesn’t necessarily mean the species is absent. The repeated surveys may be represented by temporal replication at discrete time occasions, replication of different observers or spatial replication at separate locations (MacKenzie et al. 2006). Not all sites need to be surveyed more than once. Detection probabilities can be estimated from the data collected at sites where repeated surveys were conducted and those estimates can be applied to sites that were only survey once (Mackenzie et al. 2006).

There are critical assumptions for single-species, single-season occupancy models (from MacKenzie et al. 2006): (1) occupancy status at each site does not change over the survey season, sites are “closed” to changes in occupancy (2) the probability of occupancy is constant across sites, or differences in occupancy probability are modeled using covariates (3) the probability of detection is constant across all sites and surveys or is a function of site-survey covariates (4) detection of a species and detection histories at each location are independent. Much like closed population mark-recapture models, if these assumptions are not met, occupancy estimates may be biased and habitat parameters that predict both occupancy and/or detection may be incorrect (MacKenzie et al. 2006).

There is considerably less challenges to meeting these assumptions for occupancy monitoring compared to population monitoring with mark-recapture models. Although the first assumption is that sites are closed to changes in occupancy, sites used for occupancy surveys do not need to be closed using block nets. Block nets are very difficult and time consuming to install, so this aspect of occupancy monitoring is advantageous to Parks Canada and for landscape-scale surveys in general. As long as movement in and out of a site (within a season) are random, occupancy estimates will be unbiased. Therefore, it's important to complete all occupancy surveys in one "season". A season is defined as a time frame of sufficient length such that species are either always present or always absent (MacKenzie et al. 2006). The reason why block nets are not required is because the closure assumption is at the species-scale, not the individual scale. In other words, as long as the entire group of fishes inhabiting a site do not arrive or leave at the same time, the site is closed at the species scale. Considering that bull trout may occupy different habitats among season, it is important that all surveys be conducted in the same season (e.g. summer or fall, but not both). Although we will be focusing on juvenile trout, incidental capture of ripe adult bull trout will be important to validate that sites occupied by juvenile bull trout are actually the same site where spawning occurs. Furthermore, many creeks in Banff National Park are not able to be sampled efficiently (or safely) before the fall because water levels are too high. Therefore, we will conduct surveys in September and October when water levels allow more efficient sampling. This also corresponds to the spawning period for bull trout so the assumption that the sites are closed to occupancy is likely valid.

The second and third assumptions require that both the probability of detection and probability of occupancy are the same across all sites. This is never actually the case which is ok as long as there are site-level covariates that can explain the differences in and detection occupancy. For example, Rodtka et al. (2015) found that water conductivity was an important factor explaining changes in detection probability and included site-level values for water conductivity and corrected for this in their model. Temperature is also likely to affect occupancy of bull trout and cutthroat trout (see "Temperature" subsection below). As an another example, the presence of a headwater lake containing bull trout could potentially be a factor explaining stream site occupancy as lake residents are likely to disperse downstream. This emphasizes the importance of measuring site level habitat variables (see "Predictors of Detection Probability and Occupancy" below).

The fourth assumption, detection histories at a site are independent among repeated sampling, is also important and can be addressed by doing repeated surveys during a short enough period of time that one doesn't cross seasonal boundaries, but leaving enough time between sampling that the sampling itself doesn't affect detection probabilities. Fish that have been shocked after the initial pass may be less catchable as the result of a change in behaviour, resulting in lower detection efficiency from pass to pass (Rosenberger and

Dunham 2005; Mesa and Schreck 1989). The assumption of independence can be honored simply by allowing a sufficient recovery period between surveys (e.g. 24 h; Mesa and Schreck 1989; Peterson et al. 2004).

- *Temporal replication, rather than spatial replication, will be used to estimate detection probabilities.*
- *Replicate surveys will be conducted 2 or 3 times at the same site depending on the amount of time needed to complete replicates.*
- *Block nets will not be used to meet the closure assumption, rather all sites will be surveyed within a single season, at least 24 hrs apart.*
- *Site-level habitat data will be collected to use as covariates to adjust detection probabilities and explain differences in site and patch level occupancy.*

2.4 REACH LENGTH

We found relatively little guidance in regards to reach length for sampling from existing occupancy surveys, likely because occupancy surveys are only now being published and the majority of them are from the eastern U.S. where species diversity is much higher and the consideration for reach length must take into account the detection of many species that are rare (e.g. Albanese et al. 2011; Wagner et al. 2013; Dextrase et al. 2014; McManamay et al. 2014; Midway et al. 2014). Most authors in the larger pool of abundance monitoring work reported a specified reach length without reasoning as to why that length was chosen. For example, Kruse et al. (1998), Thurow et al. (2004), and Rosenberger and Dunham (2005) chose 100 m reaches while Jones and Stockwell (1995) and Reid et al. (2009) chose 50 m reaches. No reasoning was given for any of these choices. The most advice was found in Pearsons et al. (2004). These authors suggested that when trout abundance or size variables are of interest, sampling effort requirements may be based on predetermined sampling distances. However, when species richness is the primary variable of interest, lineal sampling distances should be based upon multiple of the mean wetted width (i.e. large streams will require longer sampling sections than small streams).

Considering there is not high species diversity in the Canadian Rockies, one designated stream length should be sampled (Mochnacz, pers. comm.). Sample sites for an ongoing bull trout occupancy monitoring protocol in the Prairie Creek watershed, Nahanni National Park, NWT, uses 100 m sites. These researchers have found that 100 m sites were adequate and have replicated the same protocol for arctic grayling (Poesch, pers. comm.). Rodtka et al. (2015) used 250 m sites to sample occupancy of bull trout in tributaries of the Clearwater watershed, downstream of Banff National Park. However, this protocol used “space for time” sampling rather than temporal replicates, therefore, a 250 m reach was needed to be divided up into replicate sections. Our choice of temporally replicated surveys is favorable compared to spatially replicated surveys (e.g., Kendall and White 2009). Given the advice from the Nahanni protocol; pilot data collected in Banff National Park in 2014; the need for relatively rapid surveys; and the history of using 100 m sites for past electrofishing surveys, we feel that 100 m sites are adequate.

- *A site is defined as a 100 m reach.*

3.0 PREDICTORS OF DETECTION PROBABILITY AND OCCUPANCY

3.1 CHANNEL MORPHOLOGY AND STRUCTURE

As with the other topics in this review, there is no specific advice from the literature that is exactly relevant to the Parks Canada's stream fishes occupancy monitoring protocol. I summarized the habitat variables and data collection methods from 7 stream fish population modeling protocols as these are more available than occupancy specific protocols and the relevance of habitat to population monitoring is just as important as occupancy monitoring (Thurow et al. 2004; Rosenberger and Dunham 2005; Kruse et al. 1998; Paul and Post 2001; Peterson et al. 2004; Peterson et al. 2002; Dunham et al. 2003). Most of these studies had measured similar variables (e.g. channel dimensions, substrate, large woody debris (LWD), cover, slope and water temperature). In all studies habitat variables were measured along transects perpendicular to the flow of water. Transect spacing ranged from 5 – 25 m within 100 m reaches (Thurow et al. 2004; Rosenberger and Dunham 2005; Kruse et al. 1998; Paul and Post 2001; Peterson et al. 2004; Peterson et al. 2002; Dunham et al. 2003). For example, Rosenberger and Dunham (2005) measured habitat at transects spaced every 5 m in a 100 m reach while performing mark-recapture of rainbow trout on small headwater tributaries of the Salmon River, Idaho. Kruse et al. (1998) used transects spaced at 25 m while electrofishing cutthroat, rainbow, brown and brook trout in mountain streams with sparse habitat in Northwestern Wyoming.

Mean and maximum depths were commonly measured at transects (Thurow et al. 2004; Rosenburger and Dunham 2005; Kruse et al. 1998; Peterson et al. 2004; Peterson et al. 2002; Dunham et al. 2003). Some researchers quantified instream habitat. For example, Thurow et al. (2004) estimated the proportion of pool habitat per reach by dividing pool length by total length and assuming all pools are channel width. Kruse et al. (1998) included pools as a type of instream cover such as LWD, undercut banks and overhanging terrestrial vegetation. Peterson et al. (2004) noted habitat type (e.g. pool, riffle etc.) at each transect at 20 m intervals (100 m reach) while evaluating multipass electrofishing for estimating the abundance of bull and cutthroat trout.

- *Conduct channel measurements along transects spaced 50 m apart (i.e. 3/reach).*

- *Wetted and bankfull widths are measured once/transect (3/reach).*
- *Estimate overhead cover using a spherical densiometer at each transect (3/reach).*
- *Depth and velocity is measured at ¼, ½ and ¾ along each transect (9/reach).*

- *Measure instream structure across the entire length of the 100 m reach.*
  - *Estimate % pool habitat by summing the measured lengths of each pool and assume all pools are channel width.*
  - *Estimate % undercut banks by summing the measured lengths of undercut banks that are at least 10 cm deep.*
  - *Count total number of individual LWD > 150 cm long and 10 cm thick.*
  - *Count total number of aggregate LWD*
  - *Estimate % willow cover by summing the measured lengths of willow that are immediately adjacent to water.*

3.2 SUBSTRATE

The literature review of 7 population monitoring protocols most relevant to native species in Banff National Park were used to examine methodology for estimating substrate composition in each sampling reach. Typically, these studies visually estimated substrate size classes along the same transects that channel morphology was estimated (e.g. Thurow et al. 2004; Kruse et al. 1998; Peterson et al. 2004). For example, Thurow et al. (2004) and Peterson et al. (2004) visually estimated the percent of substrate in four size classes (fines, gravel, cobble and rubble) along a one-metre band parallel to the transect. These percentages of different size classes at each transect were averaged across the entire survey unit. In other studies, it wasn't explicit that size classes were estimated visually or if sediment was actually measured with a ruler (e.g. Rosenberger and Dunham 2005; Paul and Post 1985). In one study (Dunham et al. 2003), only the percentage of fines was estimated along each transect.

Overall, it appears as though early studies visually estimated sediment size classes. Even though there are clear definitions for each class (i.e. Wentworth scale), a visual estimate of the percent composition may lead to biases between researchers conducting the habitat assessment. Furthermore, coarse habitat categorisations may reduce the chance of be able to use habitat to explain variation in native fish occupancy and/or native fish detection probability. Indeed, a recent occupancy survey in the Clearwater River, just outside of Banff National Park, Rodtka et al. (2015) did not find any effects of habitat variables on occupancy of juvenile bull trout, but only performed rapid qualitative assessments of some habitat conditions. They concluded that their habitat data may have been too coarse to detect effects that operated at a smaller scale. Therefore, we recommend using a quantitative measure of substrate to increase the predictive abilities of habitat models. The Wolman Pebble Count (Wolman 1954), as a quantitative alternative to visually estimated sizing that Rodtka (2015) completed.

- *Measure the intermediate axis of 20 randomly chosen rocks along a random walk (i.e. Wolman Pebble Count; Wolman 1954) starting at the beginning of each transect (60 rocks/reach).*

3.3 WATER PHYSIO-PARAMETERS

Water physio-parameters such as pH, conductivity, temperature and dissolved oxygen are commonly measured in electrofishing surveys as they provide can be correlated with nutrients such as inorganic nitrogen and total phosphorus (i.e. indicators of water quality degradation). More importantly, conductivity can actually affect detection efficiency as it affects an electric currents ability to pass through water. For example, Rodtka (*In press*) found that of all habitat variables measured, water conductivity had a predominant effect on bull trout detection probability; an 85 µS/cm increase resulted in a tenfold increase in detection probability.

- *Take water physio-parameters before sampling each site.*

3.4 TEMPERATURE

Stream and river temperatures are rising in the United States (Kaushal et al. 2010). Indeed, US fisheries scientists have modeled the predicted effects of climate change on cutthroat trout and BLTR abundance and distributions (e.g. Williams et al. 2009; Al-Chokhachy et al. 2013; Wenger et al. 2011; Rieman et al. 1997; Rieman et al. 2007; Jones et al. 2014). Warmer water, changes in stream flow and increased frequency and intensity of other disturbances (e.g. flooding, non-native species invasion) are among the factors associated with climate change that have been shown to effect native trout populations. Furthermore, climate change is expected to facilitate interactions with BKTR (Rahel and Olden 2008) due to their relatively higher thermal tolerance (McMahon et al. 2007).

The lack of continuous temperature data for minimally disturbed, free-flowing streams in the Canadian Rockies will impede our understanding of community shifts (native to nonnative) in the future. Given that this occupancy survey will be repeated every ~ 10 years (frequency of the State of the Parks reporting), this protocol includes the installation of temperature loggers. Hobo V2 loggers generally have a five-year life span. They will be installed at multiple random locations in each watershed before the occupancy survey is completed for each watershed. In the short-term, this thermal data will help our understanding of how native fish occupancy varies spatially among patches and watersheds. In the long term, this thermal data will help determine if changes in native fish occupancy are attributable to climate change (see "Long-term sampling design" below). Furthermore, thermal data will allow for park-wide temperature modeling (see Jones et al. 2014) resulting in a thermal map of the park that can be used for prioritizing native fish restoration sites and habitat that is most vulnerable to change.

Equipment needs, configuration, placement, installation techniques, data retrieval and data processing were reviewed from three main documents (Dunham et al. 2005; Isaak and Horan 2011; and EPA 2013). We mostly followed existing protocols but made changes in the installation of these loggers to better suit the steep, fast and dynamic rivers in Banff National Park.

- *Temperature loggers will be installed at random locations, stratified by stream order, using Banff National Park's stream temperature monitoring protocol (Carli and Taylor 2015).*

4.0 LONG-TERM SAMPLING DESIGN

To meet the recovery objectives for WSCT and management objectives for BLTR, long term monitoring is necessary. It is not possible to know if we are protecting and maintaining stable or increasing populations otherwise. Tracking their distribution, rather than abundance, is an increasingly popular alternative to estimating abundance of fishes, especially species at the watershed scale (e.g. Williams et al. 2009; Al-Chokhachy et al. 2013; Wenger et al. 2011; Rieman et al. 1997; Rieman et al. 2007; Jones et al. 2014; Rodtka et al. *In press*).

To meet the needs of native fish management and recovery plans we must repeat the entire park-wide sample size every 10 years. While schedule corresponds with the State of the Parks Report, it is also an appropriate schedule in that we don't expect the occupancy of native or non-native species to change on a shorter time scale. To disperse the effort and cost needed to manage a complete survey, only 3 watersheds will surveyed in the park (1 comprehensive survey in one watershed/year) every 10 years. Should funding be available, we could look at the feasibility of sampling every watershed (7 total) within a 10 years cycle. Knowledge of fish community composition is important across the entire park and the occupancy approach is a valid way to sample; however, funding for this EI Measure has only been guaranteed for 3 of 10 years.

5.0 ANALYSIS

Given that it is near-impossible (with a reasonable amount of resources) to estimate and track changes in the absolute abundance of fish over time, we are taking a distributional-approach to monitoring the presence/absence of native fish in a subset of watersheds across the park. A key part of this approach is the imperfect detection of species, especially rare species or species that are naturally found at low abundance (i.e. bull trout). The appropriate analytical techniques are well documented in books such as “Occupancy Estimation and Modeling” (Mackenzie et al. 2007). We will use separate single-species models to estimate the occupancy of native fish (WSCT and BLTR) at the patch and park-scale and compare it to the same occupancy estimates for nonnative species. Multi-species models that account for species interactions (MacKenzie et al. 2007) will also be considered. The difference in both of these occupancy estimates will be used to index the relative occupancy of native species to nonnative species. The program PRESENCE (Hines 2006) will be used initially, but more sophisticated models can be generated using hierarchical modeling (Royle and Dorazio 2008), following a Bayesian approach.

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