Arctic Grayling (*Thymallus arcticus*) Abundance and Trend in the Parsnip River Watershed, 1995-2020.

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EXECUTIVE SUMMARY

Over the 1995-2007 period, the Fish and Wildlife Compensation Program – Peace Region (FWCP) periodically monitored Arctic Grayling abundance and trend in the Parsnip River watershed using replicated snorkeling surveys, during the month of August, in two index reaches of the Table River and four index reaches of the Anzac River. In 2017, FWCP identified the ten-year hiatus in the monitoring program since 2007 as a high priority knowledge gap for Arctic Grayling. The most important component of our study, which was initiated in 2018, has been to address this information gap by resuming snorkeling surveys in these long-term index reaches. This report presents snorkeling survey results from August 2020, the third year of our proposed 5-year study. A second component of our study addresses another high-priority knowledge gap for FWCP: the lack of information delineating critical habitats and abundance in other sub-basins of the Parsnip River watershed. In 2020, we utilized single-pass reconnaissance snorkeling surveys to identify critical summer rearing habitats and estimate Arctic Grayling abundance in the Hominka River, Wichcika Creek, and Reynolds Creek. In addressing these information gaps, the study is aligned with FWCP's *Streams Action Plan* (2014) Action *1b-3*: "Undertake Arctic Grayling monitoring as per recommendations of the monitoring program and develop specific, prioritized recommendations for habitat-based actions which correspond to the monitoring results."

In 2020, snorkeling surveys were conducted over two periods August 18-22 and August 27-30, necessitated because of high water and poor underwater visibility prior to August 18 and over the August 22-26 period. During snorkeling surveys, two independent, three-person crews were utilized. Snorkeling counts were replicated by both crews in just one long-term index section of the Table River: 35-31 km. Visibility conditions precluded a second replicate in Table 26-22 km, and the four long-term index sections of the Anzac River could not be surveyed at all in 2020 because of unsuitable visibility throughout August.

A key area of focus for the 2020 Arctic Grayling snorkeling study was the exploratory application of models that analyze replicated count data to estimate detection probability and abundance. We successfully applied two types of models to replicated snorkeling count data from the Parsnip River watershed: 1) a conceptually-simple Binomial-Likelihood Model that contingent on detection probability p and site-specific abundance N_{it}; and 2) a Poisson-Mixture Model contingent on p, site-specific abundance N_{it}, and integration over a prior distribution (Poisson) on mean annual abundance N_t.

In both modeling approaches, the best model included stream width as a predictor of p (negative). For the Binomial-Likelihood model, the best model also included a second predictor of p, horizontal underwater visibility (positive). For the Binomial-Likelihood Model, modeled p ranged from 0.56-0.73 among the six long-term index sections at their average values of stream width and underwater visibility, while for the Poisson-Mixture model the range was 0.49-0.67.

Estimates of trend, derived using the maximum-likelihood estimates of abundance from each of the models, are in agreement and suggest a significant increase in Arctic Grayling abundance has occurred in the Parsnip River watershed over the 1995-2020 period.

Single-pass reconnaissance snorkeling surveys have now been completed in key potential Arctic Grayling streams of the upper Parsnip River watershed: the Hominka River and Wichcika Creek surveyed in 2020, and the Missinka River surveyed in 2019. The most productive summer rearing habitats for adult Arctic Grayling in the upper Parsnip River watershed are distributed from 36-29 km of the Missinka River, and from 48-32 km of the Hominka River. However, snorkeling counts and estimated abundance of Arctic Grayling in reconnaissance sections of these watersheds are much lower than in key zones of the Table and Anzac Rivers, which therefore should be the highest priorities for Arctic Grayling habitat conservation, restoration, and enhancement actions in the Parsnip River watershed.

For the remaining two years of this proposed 5-year study, we recommend: 1) continued monitoring of Arctic Grayling abundance in long-term index sites of the Anzac and Table rivers, using replicated snorkeling surveys, 2) continued collaboration with UNBC's FFEL lab on improved models for estimating detection probability and abundance, 3) a quantitative assessment of potential limiting factors affecting distribution and abundance, 4) application of single-pass, reconnaissance snorkeling surveys in the Reynolds Creek, Colbourne Creek, and Misinchinka River sub-basins, and 5) continued dialogue with McLeod Lake Indian Band to identify opportunities for information exchange, training, and employment. These steps will provide the key background information necessary to implement effective habitat conservation, restoration, and enhancement actions, as outlined in FWCP's Arctic Grayling Synthesis Report and Arctic Grayling Monitoring Framework documents.

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1.0 INTRODUCTION

Following construction of the W.A.C. Bennett dam in 1967 and the formation of Williston Reservoir, Arctic Grayling (*Thymallus arcticus*) populations were devastated in the flooded portion of the Parsnip River watershed. Impoundment resulted in the permanent loss of over 110 km of critical Arctic Grayling habitats in the Parsnip River mainstem, and the loss of local populations that depended on these habitats (Stamford et al. 2017; Pearce et al. 2019). The remnant distribution of Arctic Grayling in the Williston Reservoir watershed appears to be restricted to stream habitats within major tributary watersheds only, which are isolated demographically from each other by the reservoir which Arctic Grayling do not appear to migrate through (Clarke et al. 2005).

The Arctic Grayling is also a species for which land use-related habitat degradation has been linked to population declines (Armstrong 1986; Northcote 1993; Walker 2005; USFWS 2010; Cahill 2015), meaning that Williston populations that survived flooding may still be under threat from current industrial activities. Key mechanisms of habitat degradation are increases in sediment transport, stream flow variation, water temperature, and access for anglers on resource roads (de Bruyn and McCart 1974; Tack 1974; Birtwell et al. 1984; McLeay et al. 1987; Reynolds et al. 1989; Clark 1992; Deegan et al. 1999; Cowie and Blackman 2003; Hawkshaw et al. 2013; Hawkshaw and Shrimpton 2014). A major increase in industrial activity has occurred since 2019 in sub-basins of the Parsnip River system. This is mostly due to a salvage logging initiative related to an outbreak of the spruce beetle *Dendroctonus rufipennis*, which will involve incursions of forestry and related road building into currently-unroaded areas throughout the watershed. In the Anzac River watershed, cumulative effects of forestry, Coastal GasLink pipeline construction, and associated road building are especially concerning.

Given conservation concern for Williston populations as described above, along with the high value of the species for First Nations subsistence fishers and BC recreational anglers, the Arctic Grayling is a priority fish species for FWCP. Our study provides key indicators of conservation status for Arctic Grayling populations in the Parsnip River watershed upstream of the reservoir influence, and identifies critical habitats where conservation actions should be directed. The 2020 field study was the third of a proposed five years in the study plan, which has been funded by the Fish and Wildlife Compensation Program (FWCP) with contributions from the Freshwater Fisheries Society of British Columbia (FFSBC).

Abundance and population growth rate (trend) are the two most important indicators of conservation status and risk for vertebrate populations (Franklin 1980; Nunney and Campbell 1993; Caughley 1994; McElhany et al 2000; O'Grady et al. 2004). Extirpation risks posed by demographic stochasticity, inbreeding depression, and long-term genetic losses/genetic drift are magnified greatly at very small population sizes (Franklin 1980; Nunney and Campbell 1993). Caughley (1994) has suggested that population trend should be considered an even more important indicator of population viability. Unless the external factors driving negative population growth in the first place – often overharvest and habitat destruction in salmonid populations – can be identified and corrected, extirpation may be a likely outcome.

Over the 1995-2007 period, FWCP monitored Arctic Grayling abundance and trend in the Parsnip River watershed using August snorkeling surveys in two index sections of the Table River and four index sections of the Anzac River (Cowie and Blackman 2012). In 2017, the ten-year hiatus in the monitoring program since 2007 was identified as a high priority knowledge gap in FWCP's Arctic Grayling synthesis report (Stamford et al. 2017). The core component of our study has been to address this information gap by resuming snorkeling surveys in the Anzac River and Table River long-term index sections beginning in 2018 (Hagen et al. 2019). This report presents snorkeling survey results from August 2020, the third consecutive year of surveys in these reaches.

The accuracy and precision of snorkeling counts as indices of fish abundance are affected by snorkeling detection probability, i.e., the proportion of fish actually present that are seen and counted by observers. Results from published accounts suggest that snorkeling detection probability can vary substantially from system to system. Correlated factors have included species differences, underwater visibility, instream cover, stream size, and observer experience (Northcote and Wilkie 1963; Schill and Griffith 1984; Slaney and Martin 1987; Zubik and Fraley 1988; Young and Hayes 2001; Korman et al. 2002; Hagen and Baxter 2005). A common method of estimating detection probability in snorkeling surveys has been through mark-resight studies (Slaney and Martin 1987; Zubik and Fraley 1988; Young and Hayes 2001; Korman et al. 2002; Hagen and Baxter 2005). Mark-resight studies however may be difficult to implement and/or costly, resulting in inadequate replication of detection probability estimates (Royle 2004). Mark-resight estimates of snorkeling detection probability have been attempted previously in long-term index sites of the Parsnip River watershed and not considered reliable because of evidence for post-tagging movements out of index sites prior to the snorkeling surveys (Cowie and Blackman 2012 and references therein).

Alternative, no-mark methods for estimating detection probability and abundance have been implemented in circumstances where counts at a particular site are repeated over time. In the simplest approach, replicated counts are assumed to be from a Binomial (N, p) distribution, where N and p are abundance (i.e. number of trials) and detection probability, respectively. Values for N and p can then be found that maximize the binomial likelihood of the count data (Binomial-Likelihood Model: Olkin et al. 1981; Royle 2004). In 2019, during exploratory statistical analysis using this approach, we were able to derive plausible estimates of N and p from replicated Arctic Grayling snorkeling count data over the 1995-2019 period (Hagen and Gantner 2020). By incorporating site-level estimates of N into our subsequent analysis of trend, we reduced the potentially confounding effects of variable detection probability. In 2020, we improve upon the 2019 analysis by including physical site attributes as predictors of p in the maximum likelihood estimation.

Royle (2004) described a class of models (*N*-Mixture models) in which the likelihoods of *N* and *p* across sites, given the count data, are integrated over a prior distribution assumed for *N* (e.g. Poisson) resulting in improved estimates of *p* under certain circumstances. In this report, we advance our exploratory statistical analysis of Arctic Grayling abundance in the Parsnip River watershed by applying a Poisson-Mixture Model to the 1995-2020 snorkeling count data, and by comparing the resulting estimates of *N* and *p* to those derived from the Binomial-Likelihood Model.

Arctic Grayling are known or suspected to be present in other sub-basins of the Parsnip River watershed in addition to the Anzac and Table rivers (Hagen et al. 2015). The lack of monitoring data indicating Arctic Grayling abundance and critical habitat locations for these populations comprises a second, high-priority information gap identified in the Arctic Grayling synthesis document (Stamford et al. 2017). By ‘critical habitats,’ we mean those that are necessary for the species to persist and thrive, and which limit or have the potential to limit the number of Arctic Grayling surviving to adulthood in the population (Rosenfeld and Hatfield 2006; Richardson et al. 2010). For conservation actions to be effective in maintaining a population, they must target limiting factors operating within critical habitat for that population (Hagen and Stamford 2017). In 2020, an additional component of our study was to utilize single-pass snorkeling surveys to sub-sample the accessible lengths of the Hominka River and Wichcika Creek, which have been identified as potentially-important Arctic Grayling streams of the upper Parsnip River watershed (Stamford et al. 2017). These surveys allowed us to provide estimates of critical summer rearing habitats for Arctic Grayling and other species, and assess the relative importance of these habitats within the Parsnip River watershed as a whole.

2.0 GOALS AND OBJECTIVES

The FWCP is partnership between BC Hydro, the Province of BC, Fisheries and Oceans Canada, First Nations and public stakeholders. In the Peace Region, FWCP’s aim is to conserve and enhance fish and wildlife impacted by the construction of the W.A.C. Bennett and Peace Canyon dams on the Peace River, and the subsequent creation of the Williston and Dinosaur Reservoirs.

Our study has been designed specifically to address two high-priority recommendations of FWCP’s *Arctic Grayling Monitoring Framework for the Williston Reservoir Watershed* (Hagen and Stamford 2017), using the methodology of snorkeling surveys in the Parsnip River watershed. The study therefore is aligned with *Streams Action Plan* priority action *1b-3* (FWCP 2014):

Action *1b-3*: Undertake Arctic Grayling monitoring as per recommendations of the monitoring program and develop specific, prioritized recommendations for habitat-based actions which correspond to the monitoring results.

The study had the following specific objectives:

1. Conduct replicated snorkeling counts of Arctic Grayling and other species in long-term index sites located in the Anzac and Table rivers, using a snorkeling survey methodology consistent with past surveys.

2. Estimate detection probability and utilize the estimates to improve the analysis of trend for Arctic Grayling in the Parsnip River watershed.

3. Acquire counts of Arctic Grayling and other species in the Homika River and Wichcika Creek using a single-pass snorkeling survey methodology, to assess relative abundance and identify critical summer rearing habitats.

3.0 STUDY AREA

The Parsnip River watershed lies within the traditional territory of the McLeod Lake Indian Band, and the Anzac River and Table River watersheds and their natural resources are of critical community interest (Hagen et al. 2015). The mouths of the Anzac River, Table River, Hominka River, and Wichcika Creek are located approximately 40 km, 60 km, 70 km, and 80 km southeast of the village of McLeod Lake, respectively (Figure 1). These rivers also enjoy high popularity amongst the recreational angling community in northcentral BC.

Historically, the Parsnip River flowed roughly 280 km along the Rocky Mountain Trench from Arctic Lake to its confluence with the Finlay River, where the two rivers joined to form the Peace River. Construction of the 183 m high W.A.C. Bennett Dam, which was completed in 1967, resulted in the formation of Williston Reservoir, which reached full pool in 1972 (Hirst 1991) and flooded the lower ~110 km of the Parsnip River.

The post-impoundment Parsnip River system is a 6th order stream that has a watershed area of 5,600 km^2 (Table 1). Major sub-basins of the Parsnip (Misinchinka, Colbourne, Reynolds, Anzac, Table, Hominka, Missinka, Upper Parsnip), range from 290 km^2 to 1,000 km^2 and drain mountainous terrain in the Hart Ranges of the Rocky Mountains, lying to the east of the trench. In contrast, smaller sub-basins on the west side of the Parsnip (95 km^2 to 182 km^2) drain lower elevation areas of the Nechako Plateau (Figure 1; Table 1).

Streamflow is snowmelt driven, with peak discharge occurring, on average, in late-May to early-June in the Parsnip River watershed (Water Survey of Canada Station 07EE007 *Parsnip River above Misinchika River*). Much of the watershed drains higher elevation, mountainous areas. Consequently, sediment load is relatively high among sub-basins, as evidenced by turbid water flows in spring, wide channels relative to stream size, and extensive bar development (Bruce and Starr 1985). Substantial glacial influence occurs only within the Upper Parsnip sub-basin (Figure 1). Consequently, in most years water clarity is excellent throughout watershed sub-basins throughout much of the year, and by late summer the Parsnip mainstem itself becomes relatively clean in areas downstream of the Missinka River (Anonymous 1978).

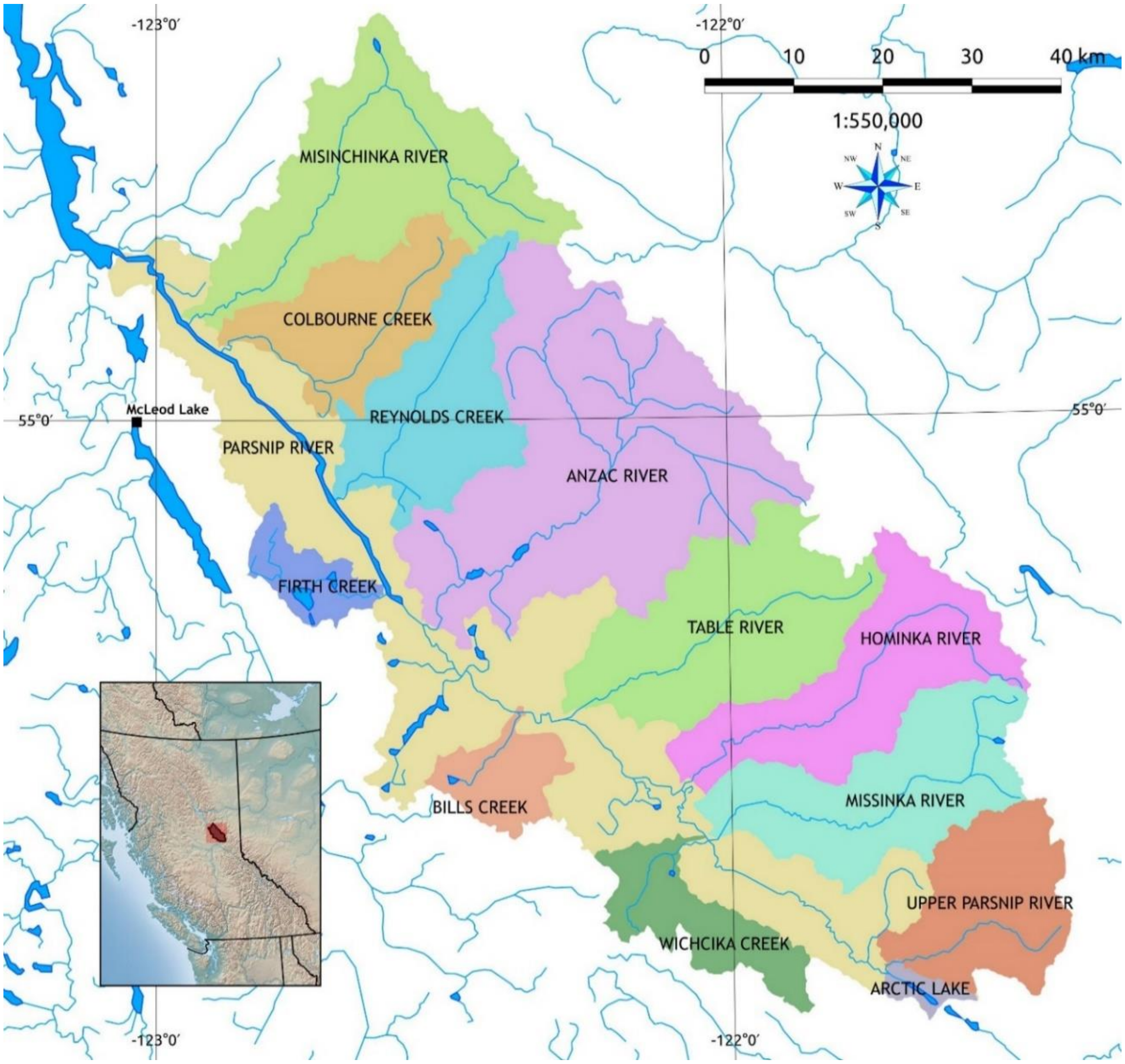

Figure 1. Sub-basins of the Parsnip River watershed (Parsnip mainstem, Misinchinka, Colbourne, Reynolds, Firth, Anzac, Bill's, Table, Hominka, Missinka, Wichcika, Arctic Lake, Upper Parsnip) potentially utilized by Arctic Grayling.

Table 1. Biophysical characteristics of sub-basins potentially utilized by Arctic Grayling within the Parsnip River watershed (adapted from Hagen et al. 2015).

Watershed	Sub-basin	Watershed area (km^2)	Stream order	Fish species present*
Parsnip	Parsnip total	5,612	6	GR, EB, BT, BB, KO, LKC, LT, LW, CSU, LNC, LSU, MW, NSC, PCC, CAS, PW, RB, RSC, CCG, WSU
Parsnip	Misinchinka River	595	4	GR, BT, BB, LSU, MW, RB, CCG
Parsnip	Colbourne Creek	289	4	GR, BT, CSU, LSU, MW, RB, CCG
Parsnip	Reynolds Creek	366	5	GR, BT, BB, LKC, CSU, LNC, LSU, MW, RB, RSC, CCG
Parsnip	Firth Creek	95	3	GR, BB, LKC, LW, LNC, LSU, MW, RB, CCG
Parsnip	Anzac River	1,044	5	GR, BT, BB, LKC, LT, LW, LSU, MW, PCC, CAS, RB, RSC, CCG
Parsnip	Tacheeda Lakes	95	4	BT, KO, LT, LW, LNC, LSU, MW, NSC, PCC, CAS, PW, RB, RSC, WSU
Parsnip	Bill's Creek	122	5	GR, BB, MW, RB, CCG
Parsnip	Table River	504	5	GR, BT, BB, LW, CSU, LSU, MW, NSC, RB, CCG, WSU
Parsnip	Hominka River	433	5	GR, BT, BB, LSU, MW, PCC, RB, CCG, WSU
Parsnip	Missinka River	434	5	GR, BT, BB, LKC, CSU, LNC, LSU, MW, NSC, RB, RSC, CCG
Parsnip	Wichcika Creek	182	5	GR, BT, BB, MW, RT, CCG
Parsnip	Arctic Lake	31	-	GR, BT, KO, LT, LW, LSU, MW, NSC, RB, RSC, WSU
Parsnip	Upper Parsnip	303	-	GR, BT, BB, KO, LT, LW, CSU, LSU, MW, NSC, RB, RSC, CCG, WSU

*From records in databases linked to the BC Geographic Warehouse, accessed January 2015

4.0 METHODS

4.1 Survey Conditions

Water Survey of Canada (WSC) Station 07EE007 *Parsnip River above Misinchinka River* is located on the Parsnip River near its mouth. It is the only WSC flow monitoring station for the Parsnip River watershed. This WSC station provided real time stream discharge data which was utilized to assess the potential safety and feasibility of snorkeling surveys in August 2020.

Two physical habitat attributes potentially affecting snorkeling detection probability were also monitored within surveyed stream sections. These were: 1) underwater visibility and 2) wetted stream width. We measured underwater visibility in snorkeling survey sections in two ways: 1) horizontal underwater Secchi disk visibility (Figure 2), and 2) horizontal underwater distance at which the species identity of a 30 cm Arctic Grayling model could no longer be discerned. We estimated wetted stream width using a laser range finder. Visibility was measured only once per reach, at the beginning of the survey, while wetted stream width was measured at 10 locations in each reach.

Figure 2. Estimating horizontal underwater visibility in the Anzac River, August 2020.

To facilitate analysis of Arctic Grayling snorkeling detection probability across the 1995-2020 time series, physical site data from the 1995-2007 period were compiled, following a search of available data (spreadsheets and reports) on file with FWCP, in a separate data report (Cowie 2021). The following variables were of key interest for ongoing population monitoring and the primary focus of this data compilation: 1) survey date, 2) site identification (typically river km), 3) number of replicate swims, 4) number of swimmers per replicate, 5) discharge on the survey date (from WSC Station 07EE007), 6) horizontal underwater visibility, and 7) wetted stream width.

4.2 Snorkeling Methods

In 2020 we conducted snorkeling surveys in only two of the six long-term index reaches of the Parsnip River watershed. These were the 35-31 km and 26-22 km sections of the Table River (Figure 3) on August 19 and August 20, respectively. Long-term index sites of the Anzac River watershed (Figure 4) were not surveyed in 2020 due to very low underwater visibility, which was assessed on multiple occasions over the August 15-29 period. Visibility assessments were made by members of the University of Northern BC’s Freshwater Fish Ecology Laboratory (FFEL), who were also working in the Parsnip River watershed, and by our crew during road- and helicopter-based reconnaissance.

Surveys of 8 reconnaissance sections of the Hominka River (Figure 5), 5 sections of Wichcika Creek (Figure 6), and 1 section of Reynolds Creek (Figure 7) were conducted over the August 18-August 30 period.

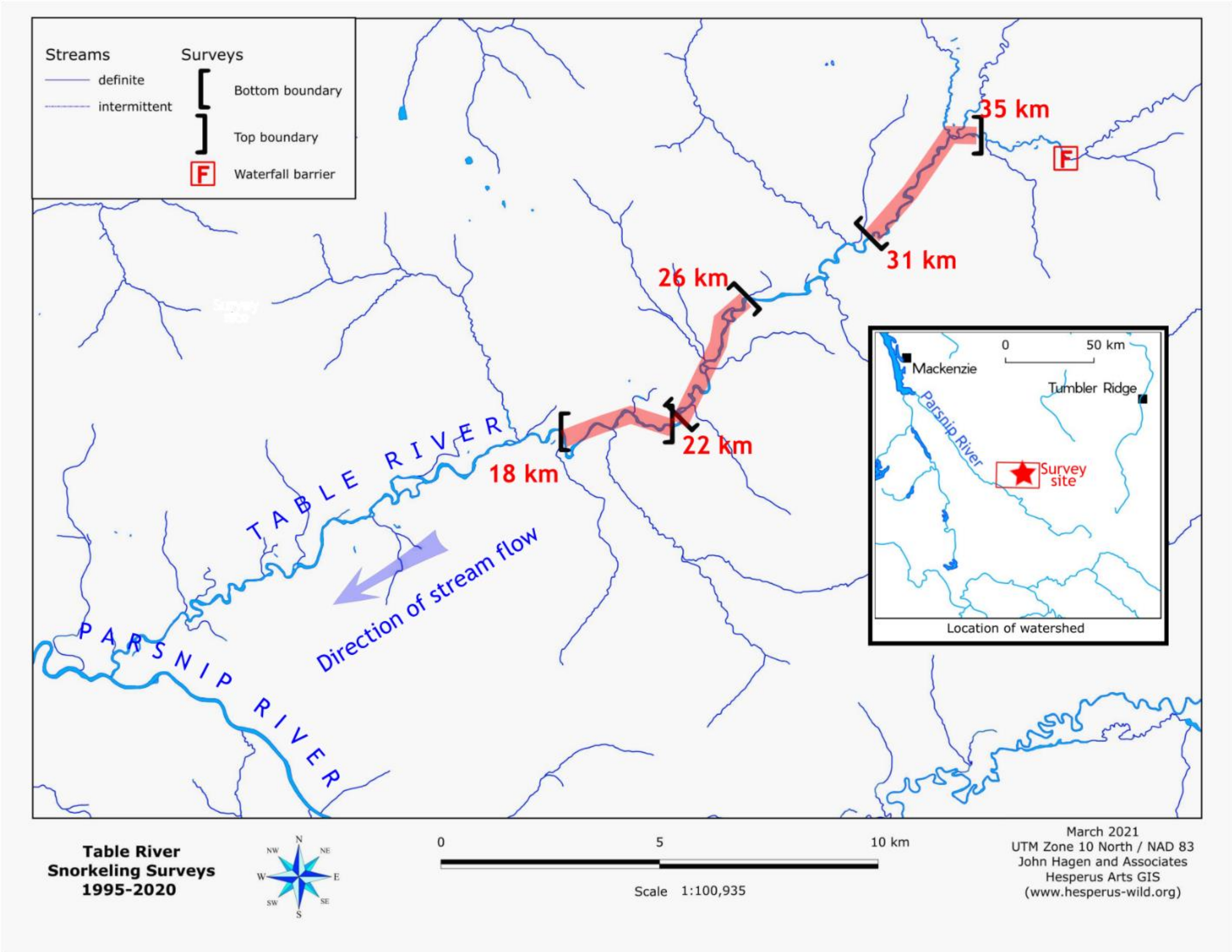

Figure 3. Index sections of the Table River utilized for snorkeling surveys to monitor Arctic Grayling abundance, 1995-2020. Site 22-18 km was utilized up to 2007, but was replaced by site 26-22 km in 2018 due to the presence of a clay slump at 22 km compromising underwater visibility in the lower site.

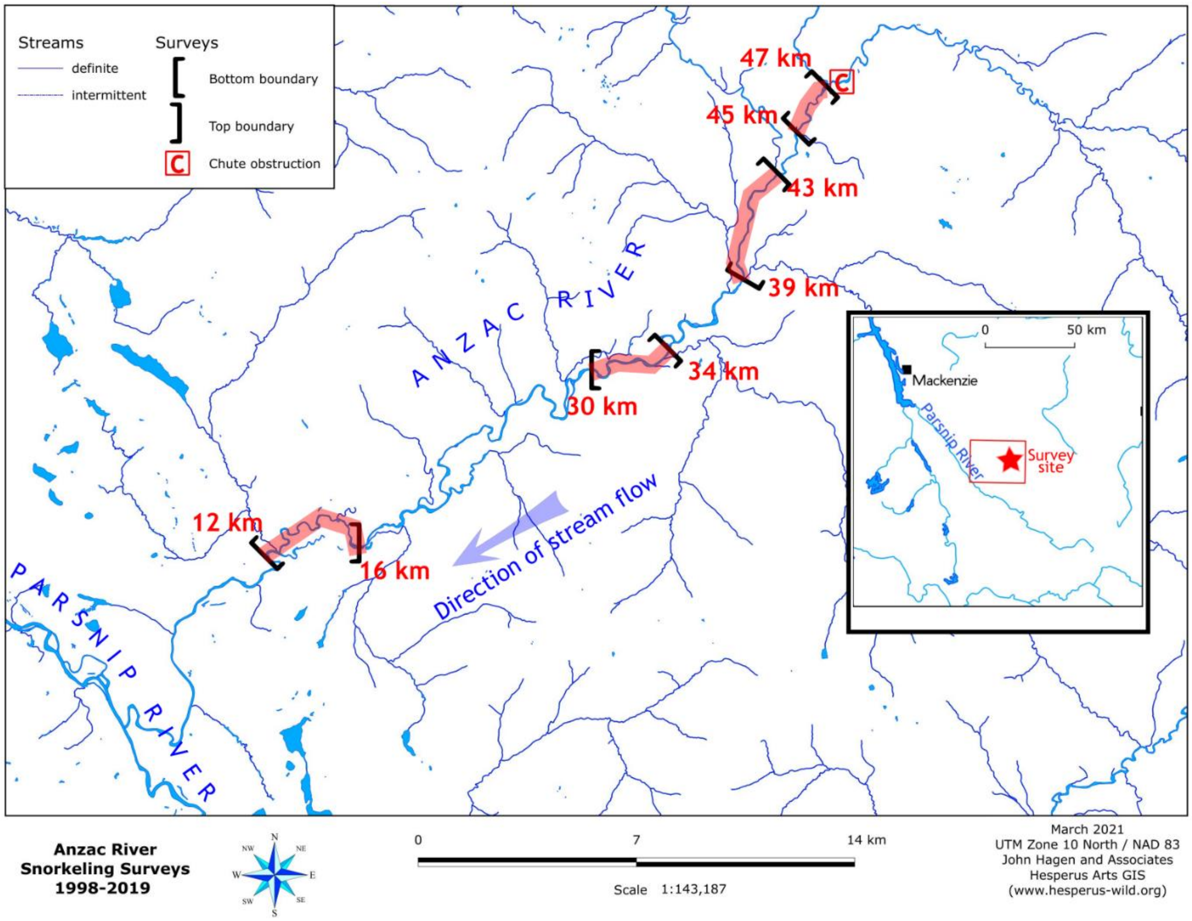

Figure 4. Index sections of the Anzac River utilized for snorkeling surveys to monitor Arctic Grayling abundance, 1995-2019. Surveys were not possible in summer 2020 because of unsuitable levels of underwater visibility.

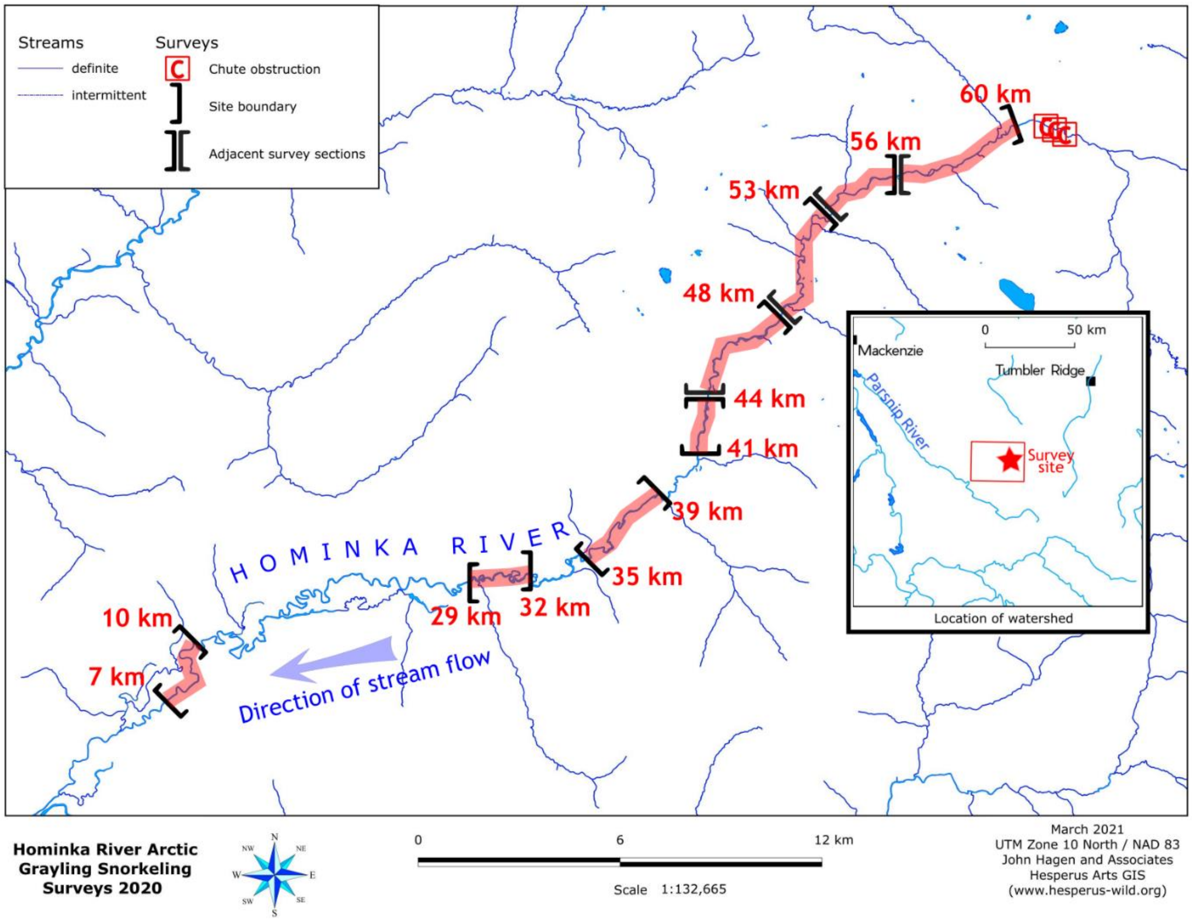

Figure 5. Reconnaissance sections of the Hominka River utilized for snorkeling surveys to monitor Arctic Grayling abundance, August 2020.

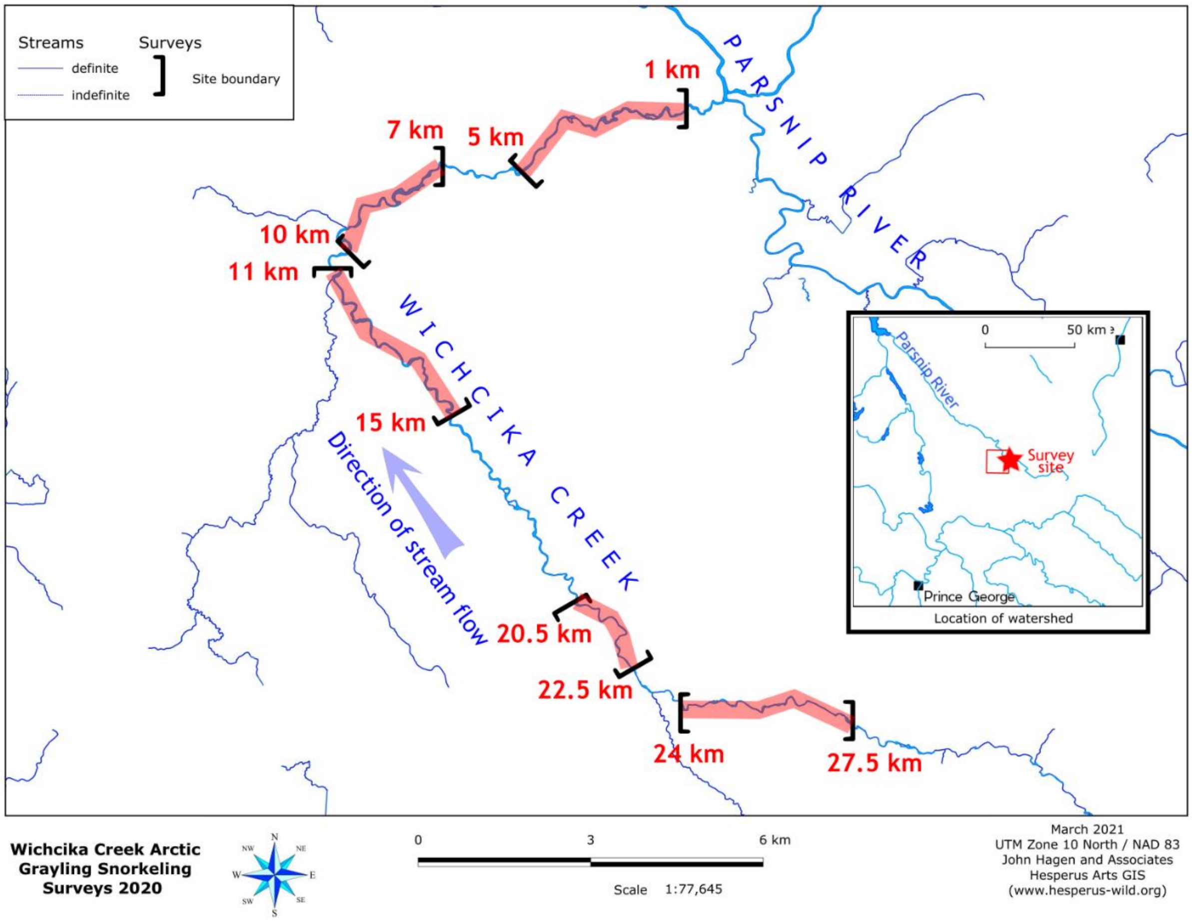

Figure 6. Reconnaissance sections of Wichcika Creek utilized for snorkeling surveys to monitor Arctic Grayling abundance, August 2020.

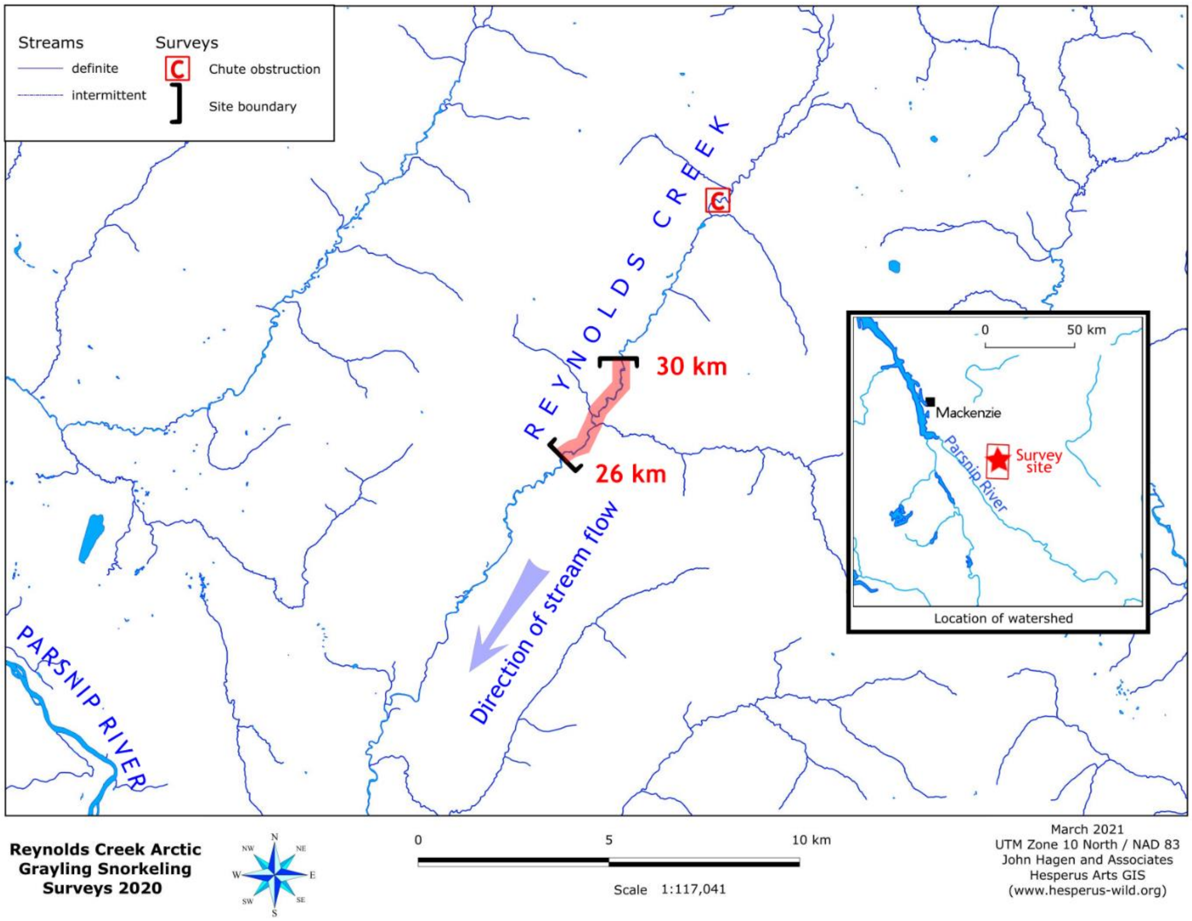

Figure 7. Reconnaissance section in Reynolds Creek utilized for a snorkeling survey to monitor Arctic Grayling abundance, August 2020.

Replicate snorkeling surveys in the long-term index sections of the Table River were conducted by two independent, three-person crews. A minimum one-hour delay was enforced between surveys to allow the site to recover from disturbance. Single-pass reconnaissance sections in the Hominka River, Wichcika Creek, and Hominka River were surveyed by just one crew, but were otherwise surveyed by the same method. The exception was the 32-29 km reach of the Hominka River. Because of poor Secchi disk visibility < 3m at that location, this reach was surveyed by a crew of four swimmers.

Consistent with methods utilized up to 2007 (Cowie and Blackman 2012), snorkeling counts were made by two observers in drysuits (Figures 8, 9), organized in lanes of width determined by horizontal underwater visibility and estimated habitat suitability for Arctic Grayling (see Blackman 2001 for subadult/adult Arctic Grayling habitat use). All observers had experience in at least one other snorkeling study and had received in-the-water training with the study protocol prior to the survey period, and both crew leaders had 20+ years of snorkeling experience. During snorkeling surveys, typically observers surveyed adjacent lanes on either side of the thalweg, and scanned the water ahead of them and to the right or left depending on which side of the stream they were responsible for. Observers attempted to count only fish that were in their lane. If fish moved in reaction to observers, frequent communication ensured that double counting did not occur. In areas where the usable width of the stream was greater than the width of two lanes surveyed in this manner, one or both of the observers would extend their lane width and look both ways. Observed fish (Figure 10) were classified to species, and tallied in one of five size categories: 0-20 cm, 20-30 cm, 30-40 cm, 40-50 cm, and 50+ cm.

The third member of each crew was a safety boater with appropriate swiftwater rescue training and equipment. The safety boater paddled behind the line of snorkelers in an inflatable kayak that could navigate the range of stream features encountered and that could be stowed deflated in the basket of the helicopter (Figure 9).

At the start of each survey, size estimation was practiced under water using Arctic Grayling models (laminated, trimmed photographs).

Figure 8. Arctic Grayling holding water in the Hominka River watershed, August 2020.

Figure 9. Snorkeling team in the Wichcika Creek watershed, August 2020.

Figure 10. Table River Arctic Grayling.

4.3 Analyses

Models of detection probability and abundance.

Binomial-Likelihood Model. As one approach in our 2020 analysis of snorkeling detection probability and abundance, we utilized a binomial-likelihood model (Olkin et al. 1981; Royle 2004) similar to our exploratory statistical analysis in 2019 (Hagen and Gantner 2020). We assumed for all years 1995-2019 that the population of Arctic Grayling in each index reach was closed with respect to movement, mortality, etc. between the start and finish of all replicate surveys. We further assumed that replicated counts n_{ir} were binomially-distributed random variables from the distribution

$$n_{ir} \sim \text{Binomial}\ (N_{it}, p)$$

where i is the site, r the replicate (among R replicated surveys), N_{it} the population size at index site i and year t and p is the detection probability. The likelihood statement for data from a site is detailed in Royle (2004), and represented in simplified form here as:

$$L(N_{it}, p|\{n_{i1}, \dots, n_{iR}\}) = \prod_{r=1}^{R} Binomial(n_{ir}; N_{it}, p)$$

(1)

Joint likelihood across all sites of interest is given by the product of the site-specific likelihoods (Royle 2004). In 2020, we estimated the N_{it} and p by searching for parameter values that maximized the joint likelihood across all the site/year possibilities for the 1995-2020 period, (e.g. a site with replicated count data in four separate years would yield four site/year-specific likelihoods conditional on the four N_{it} and one p).

Poisson-Mixture Model. As our second approach in 2020, we also estimated snorkeling detection probability and abundance utilizing the Poisson-Mixture Model of Royle (2004), one of a class of N-mixture models which treat the N_{it} as independent random variables distributed according to a specified prior distribution for N. Prior parameters for the Poisson distribution are the N_{it} and λ_t (λ_t = both mean and variance for abundance N_t), which are estimated after integrating Equation (1) above over the prior distribution for N_t (Royle 2004). In simplified form, the integrated likelihood from Royle (2004) for a given year t is:

$$L(p, \lambda_t|\{n_{ir}\}) = \prod_{i=1}^{I} \left\{ \sum_{N_{it}=\max n_{ir}}^{\infty} \left(\prod_{r=1}^{R} Binomial(n_{ir}; N_{it}, p) \right) * Poisson(N_{it}; \lambda_t) \right\}$$

(2)

We computed the joint likelihood across all years of interest as the product of the year-specific likelihoods. In searching for values of λ_t and p that maximized the joint likelihood across all years for the 1995-2020 period, λ_t was year-specific but p was not.

Covariate effects on detection probability. A question of key interest to us in 2020 was whether predictions of abundance and detection probability p could be improved through the use of logistic regression models for p. To do this, we identified a series of candidate models representing different hypotheses about the effects of physical site attributes on p, then compared these models using an information-theoretic approach (Burnham and Anderson 2002). Model selection was conducted for both classes of models described above: i) the Binomial Likelihood Model and ii) the Poisson-Mixture Model (Royle 2004).

Our a priori expectation was that two factors in particular would affect snorkeling detection probability: 1) the stream wetted width in index reaches, which is related to the cross-sectional area to be searched by snorkelers, and 2) horizontal underwater visibility. During some years over the 1995-2007 period, snorkeling crews were comprised of 3 swimmers rather than 2. We therefore defined a third variable, lane width, as the stream wetted width divided by the number of swimmers. Width and visibility estimates were acquired from the data compilation exercise described in Section *4.1*. Unfortunately, stream width data were relatively sparse among years. For years without stream width estimates, we entered average values for the wetted width and lane width variables.

Model selection was performed using replicated count data from all years 1995-2007 (snorkeling surveys were replicated at least twice). We used the Akaike information criterion corrected for small sample size (AIC_c) for the comparisons among models. We computed the strength of evidence for each candidate model being the best in the set by computing the likelihood of each model given the data $L(g_i|x)$, then normalizing these likelihoods as a set of Akaike weights w_i (Burnham and Anderson 2002).

We assessed the trend in Arctic Grayling abundance over time for the Parsnip River watershed using model estimates of abundance in preference to raw count data, to account for the effects of variable detection probability. Binomial-Likelihood Model estimates of N_{it} (see preceding paragraphs) were analyzed within a linear mixed effects analysis, performed using the Stata statistical analysis program (StataCorp, 2009) and the 'xtmixed' function (Rabe-Hesketh and Skrondal 2008). The N_{it} were entered into the model as a fixed effect, along with observation year. As random effects, we had intercepts for sites nested within streams. Poisson-Mixture Model estimates of annual mean abundance N_t were analyzed using simple linear regression on years.

5.0 RESULTS

5.1 Survey Conditions

August 2020 was wet, with several significant rain events recorded on the Water Survey of Canada's gauge in the lower Parsnip River. A visual inspection of graphs of accumulated precipitation and discharge at the station suggests that rainfall preceded flow increases by roughly 24 hours (Figure 11). Significant precipitation events on August 15 and August 21-22 resulted in peaks of discharge on August 16 and August 23. As a result of this wet weather, discharge in the Parsnip River watershed remained well above long-term average levels for the entire August 15-30 period over which the study had been scheduled (Figure 11).

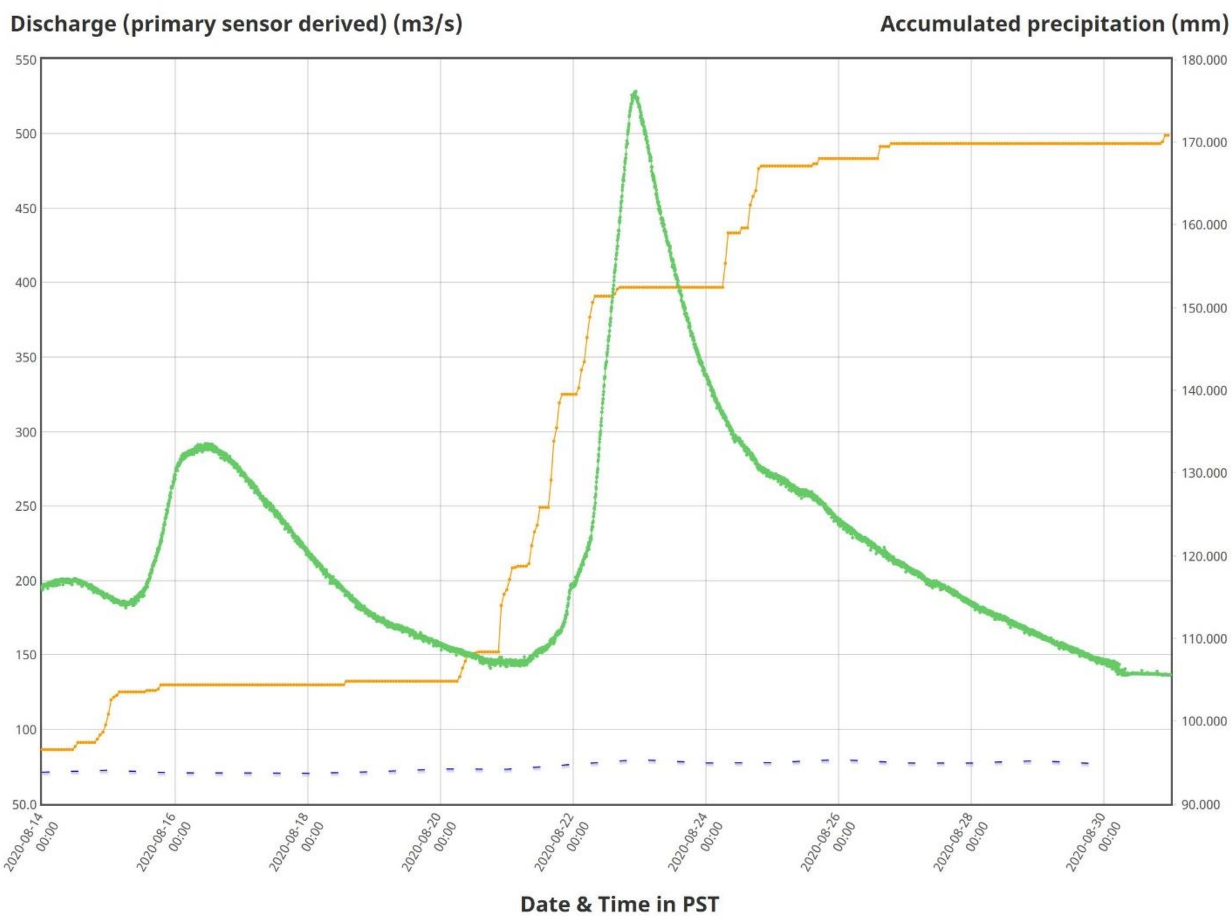

Figure 11. 2020 Discharge (green line), long-term average discharge (dashed line), and 2020 accumulated precipitation (orange line) estimated for the lower Parsnip River (WSC gauge 07EE007), August 14-30, 2020.

To accommodate the rainy weather forecast and high flows, the start date of the 2020 study was delayed from August 15 to August 18. Surveys were suspended over the August 23-26 period, corresponding to a major spike in discharge (Figure 11, Figure 12), then resumed August 27-30.

Poor visibility in long-term index sites of the Anzac and Table rivers in 2020 necessitated several schedule changes and ultimately a modification of the study scope. Fieldwork was originally scheduled for the August 15-21 period for two crews of three. To accommodate high flows (Figure 11) and poor visibility conditions (Table 2, Figure 12), snorkeling fieldwork was first delayed and then conducted over two periods: 1) August 18-22, and 2) August 27-30.

Table 2. Visibility assessments made during August, 2020 to identify whether snorkeling surveys could proceed.

Date	Stream	Location	Fish Visibility (m)	Secchi Visibility (m)	Comment
15-Aug	Table	22 km	~0.5	na	From Joe Bottoms (FFEL) via inReach
15-Aug	Anzac	na	<1	na	From Joe Bottoms (FFEL) via inReach
18-Aug	Table	22 km	2.8	4	Helicopter-based visibiity reconnaissance
18-Aug	Anzac	34 km	<2	<2	Obviously out, visually estimated from the air
20-Aug	Table	26 km	3.8	4.9	Secchi declined to 3.8 m prior to second replicate
20-Aug	Anzac	34 km	2.4	2.7	Helicopter-based visibiity reconnaissance
25-Aug	Table	22 km	<1	na	Road-based visibility reconnaissance
25-Aug	Anzac	35 km	<1	na	Road-based visibility reconnaissance
25-Aug	Reynolds	6 km	<1	na	Road-based visibility reconnaissance
25-Aug	Wichcika	11 km	~2	na	Road-based visibility reconnaissance
28-Aug	Table	34 km	2.1	2.8	Helicopter-based visibiity reconnaissance
29-Aug	Anzac	16 km	1.9	2.4	Helicopter-based visibiity reconnaissance
29-Aug	Anzac	34 km	2.2	3.1	Helicopter-based visibiity reconnaissance
29-Aug	Anzac	39 km	2.2	2.7	Helicopter-based visibiity reconnaissance
29-Aug	Anzac	47 km	2.4	3.2	Helicopter-based visibiity reconnaissance
29-Aug	Anzac	54.5 km	4	5.5	Helicopter-based visibiity reconnaissance
29-Aug	Reynolds	7.5 km	2.4	2.7	River out below a slump at 26 km
29-Aug	Reynolds	30 km	2.7	3.9	Survey from 30-26 km completed

Figure 12. Anzac River at lower bridge (stream km 6) on August 22.

Although replicated surveys of the long-term index sections in the Anzac and Table rivers were the first priority for fieldwork, only the Table 35-31 km reach (surveyed August 19) received a replicated survey due to unsuitable visibility on all or part of all other field days. The first replicate of the Table 26-22 km reach (surveyed August 20; Table 2) was conducted under suitable visibility conditions, but visibility became unworkable during the second replicate (just 2 hours later) which could not be completed.

It appears unlikely that any of the Anzac River index sites had suitable visibility for snorkeling surveys at any time during July and August 2020. It also appears likely that industrial development in the watershed negatively influenced water clarity over this period. The final visibility assessment of the Anzac River took place on August 29 (Table 2), at which time suitable visibility for snorkeling surveys was only identified upstream of the index reaches and areas of road and pipeline construction (54.5 km; Table 2). Photographic evidence of sediment inputs originating from road construction (e.g. Figures 12, 13) was forwarded to Natural Resource Officer service at FLNRORD Prince George, and resulted in a field inspection (Zsolt Sary, FLNRORD Ecosystems Section, Prince George, pers. comm. September 2020).

Figure 13. Slumping of newly-constructed forestry road into Anzac River floodplain at stream km 48 (10 U 553116 6093016).

Secchi disk visibility in 2019 ranged from 5.1 m to 4.9 m in two long-term index sections of the Table River, from 2.9 m to 15 m in 8 Hominka River reconnaissance sections, from 4.0 m to 5.2 m in 5 Wichcika Creek reconnaissance sections, and was 3.9 m in the Reynolds Creek reconnaissance section. Visibility for discernment of Arctic Grayling models was 3.8 m and 4.7 m in the 26-22 km and 35-31 km index sites of the Table River, within the ranges of fish visibility previously recorded (Table 3).

Additional physical site data from the 1995-2007 period (see Section *4.1*), along with an overview summary of FWCP study findings over the period, have been compiled and are presented in a separate data report (Cowie 2021).

Table River site 26-22 km was a replacement for the previous long-term index site 22-18 km in the Table River, due to a major clay slump into the river at 22 km, for the third consecutive year. The extensive nature of the slump indicates that visibility in the formerly-surveyed reach may be compromised for years to come.

Table 3. Horizontal underwater visibility in index sites of the Parsnip River watershed, August 2020.

	Table River		Hominka River								Wichcika Creek					Reynolds Creek
Site	35-31	26-22	60-56	56-53	53-48	48-44	44-41	39-35	32-29*	10-7	27.5-24	22.5-20.5	15-11	10-7	5-1	30-26
Date	19-Aug	20-Aug	21-Aug	18-Aug	18-Aug	21-Aug	21-Aug	21-Aug	22-Aug	30-Aug	30-Aug	28-Aug	28-Aug	27-Aug	27-Aug	29-Aug
Secchi disk visibility (m)	5.1	4.9	14	12	12	15	11	5.5	2.9*	6.5	4	4.7	5.2	4.2	4.8	3.9
Fish model visibility (m)	4.7	3.8	8.0	10	10	na	na	4	2.6*	4.5	3.5	3.6	4.6	3.9	4.6	2.7
1995-2019 range (fish)	3.5-7	3.0-7.7														

* Surveyed by a crew of four swimmers

5.2 Detection Probability and Abundance

Over the 1995-2020 period, a total of 37 surveys have taken place in the six long-term index sections of the Anzac and Table rivers in which at least two replicate counts were made (Table 4). These data were analyzed within both the Binomial Likelihood Model and Poisson-Mixture Model frameworks. Both modeling frameworks estimate abundance conditional on snorkeling detection probability *p*, (Section *4.3*) a parameter of key interest to us given the logistical challenges in estimating *p* by other means (e.g. mark-resight studies). In our exploratory statistical analysis in 2020, we were interested in three variables potentially influencing *p* and subsequent inferences about population status: 1) site wetted width *WIDTH*, 2) horizontal underwater visibility at which Arctic Grayling could be discriminated *FISH_VIS*, and 3) and lane width *LANE* (stream width divided by the number of swimmers) (Table 4).

Binomial-Likelihood Model. For the Binomial-Likelihood Model, we evaluated a candidate set of 6 logistic regression models for *p* using AIC_c (Table 5). The best model contained both *WIDTH* and *FISH_VIS* as predictor variables with the expected signs (negative for *WIDTH*, positive for *FISH_VIS*), and resulted in a significant improvement in Binomial-Likelihood Model likelihood relative to the constant-*p* model (Chi-square *P* <0.001). The likelihood of this model being the best, as indicated by the ratios of its Akaike weight w_i to those of other candidate models, was 0.779 (Table 5). The logistic model was:

$$p = \frac{1}{1 + \exp\{-(1.707 - 0.0590 * WIDTH + 0.0329 * FISH\_VIS)\}}$$

(3)

Table 4. Replicated snorkeling counts of Arctic Grayling >20 cm in index sites of the Anzac and Table rivers, 1995-2020.

Site	Year	Replicate Counts of Arctic Grayling >20 cm R1	R2	R3	R4	*FISH_VIS*	*WIDTH*	No. of swimmers	*LANE*	Predicted N_{it}
Table 22-18	1998	54	79			5.5	20.2	3	6.7	104
	2000	39	30	40	38	4	21.0	2	10.5	62
	2001	35	48			3	22.5	2	11.3	65
	2003	75	62	72		4.5	21.0	2	10.5	115
	2007	39	57	42		4	22.5	3	7.5	75
Table 26-22	2018	75				7.7	22.5	2	11.3	115
	2019	116				4	22.5	2	11.3	186
	2020	67				3.8	27.8	2	13.9	122
Table 35-31	1995	107	115			5	14.0	2	7.0	145
	1998	137	135			5.5	12.0	2	6.0	164
	2000	101	111	136	145	7	15.0	2	7.5	167
	2001	80	102			3.5	15.0	2	7.5	124
	2003	139	138	134		3.5	12.0	2	6.0	171
	2005	96	104	94		3.6	15.0	2	7.5	122
	2007	124	103	112		3.7	15.0	2	7.5	142
	2018	191	230	209		5.7	15.0	2	7.5	264
	2029	188				3.5	14.0	2	7.0	257
	2020	139	164			4.7	22.0	2	11.0	187
Anzac 16-12	1998	13	3			6.5	30.8	3	10.3	19
	2001	6	15			4	27.9	2	14.0	23
	2003	18	30	22		4.5	25.0	3	8.3	53
	2005	26	31			4.5	27.9	3	9.3	59
	2007	44	50			3.8	27.9	3	9.3	94
	2018	22				7.7	27.9	2	14.0	38
	2019	40				4.4	27.9	2	14.0	73
Anzac 34-30	1998	116	96			9.5	27.5	3	9.2	191
	2001	48	55			3	27.3	2	13.6	95
	2003	54	68	41		3.5	27.0	2	13.5	107
	2005	98	56	82		4	27.3	3	9.1	154
	2007	34	83	67		3.7	27.3	3	9.1	124
	2018	138	93	111		6	27.3	2	13.6	238
	2019	82	67	68		5	27.3	2	13.6	139
Anzac 43-39	1998	167	114	127		4.5	27.3	2	13.7	224
	2001	73	96			4.5	27.2	2	13.6	133
	2003	144	181	172		4.5	27.0	2	13.5	269
	2005	99	83			4.3	27.2	3	9.1	148
	2018	173	187	185		4.6	27.2	2	13.6	294
	2019	140	149	167		5.1	27.2	2	13.6	246
Anzac 47-45	1998	157	171			4.5	15.0	2	7.5	272
	2000	69	67			3	15.0	2	7.5	115
	2001	15	25			3	15.0	2	7.5	39
	2003	62	80	92		3.5	15.0	2	7.5	127
	2018	194				5.5	15.0	2	7.5	265
	2019	110	85	77		4.4	15.0	2	7.5	150

The model containing only *WIDTH* as a predictor of p also had support (w_i = 0.221), while the likelihoods for all other models being the best were far less (w_i <<0.001). The *WIDTH* logistic model was:

$$p = \frac{1}{1 + \exp\{-(1.849 - 0.0584 * WIDTH)\}}$$

(4)

Along with p, abundance N_{it} (population size N at site i in year t) is a parameter of the Binomial-Likelihood model. The N_{it} for the 37 replicated sites 1995-2020 are presented in Table 4 along with count data and physical site attributes. In Table 4, the N_{it} for the 7 sites without replicated counts were predicted using the *WIDTH* + *FISH_VIS* logistic regression model for p.

Table 5. Comparison among predictors of detection probability p estimated within a binomial-probability model framework (see text) from replicated count data in the Parsnip River watershed. Symbols K, Log (L), AIC_c, Δi, L (g_i|x), and w_i, denote 1) the number of estimable parameters, 2) model log-likelihoods, 3) the Akaike information criterion values adjusted for small sample size, 4) the difference in AIC_c values between each model and the model with the lowest AIC_c score, 5) the likelihood that the candidate model is the best among the set, and 6) Akaike weights, respectively.

Model	*K*	Log($\mathcal{L}$)	AIC_c	Δ_i	$\mathcal{L}(g_i \mid x)$	w_i
Constant-only	2	-788.87	1582.10	375.80	2.49347E-82	1.94E-82
WIDTH	3	-601.05	1208.83	2.52	0.283195985	0.221
FISH_VIS	3	-788.75	1584.23	377.92	8.61697E-83	6.72E-83
LANE	3	-695.32	1397.37	191.06	3.24918E-42	2.53E-42
WIDTH+FISH_VIS	4	-598.528	1206.31	0.00	1	0.779
LANE+FISH_VIS	4	-695.074	1399.40	193.09	1.17618E-42	9.17E-43
$MinAIC_C$			1206.31			
n	37					

Poisson-Mixture Model. For the Poisson-Mixture Model, we evaluated the same candidate set of 6 logistic regression models for p using AIC_c (Table 6). The best model contained only *WIDTH*, and resulted in a significant improvement in Poisson-Mixture Model likelihood relative to the constant-p model (Chi-square P <0.001). The likelihood of this model being the best, as indicated by the ratios of its Akaike weight w_i to those of other candidate models, was 0.767 (Table 6). The logistic model was:

$$p = \frac{1}{1 + \exp\{-(1585 - 0.0584 * WIDTH)\}}$$

(5)

The model containing *WIDTH* and *FISH_VIS* as predictor variables also had support (w_i = 0.233), while the likelihoods for all other models being the best were far less (w_i <<0.001). The *WIDTH* + *FISH_VIS* logistic model was:

$$p = \frac{1}{1 + \exp\{-(1.528 - 0.0585 * WIDTH + 0.0116 * FISH\_VIS)\}}$$

(6)

Table 6. Comparison among predictors of detection probability *p* estimated within a Poisson-mixture model framework (see text) from replicated count data in the Parsnip River watershed. Symbols K, Log (L), AIC_c, Δi, L (g_i|x), and w_i, denote 1) the number of estimable parameters, 2) model log-likelihoods, 3) the Akaike information criterion values adjusted for small sample size, 4) the difference in AIC_c values between each model and the model with the lowest AIC_c score, 5) the likelihood that the candidate model is the best among the set, and 6) Akaike weights, respectively.

Model		*K*	Log($\mathcal{L}$)	AIC_c	Δ_i	$\mathcal{L}(g_i \vert x)$	w_i
Constant-only		2	-908.30	1820.96	91.16	1.6039E-20	1.23E-20
WIDTH		3	-861.54	1729.80	0.00	1	0.767
FISH_VIS		3	-908.29	1823.31	93.51	4.95547E-21	3.80E-21
LANE		3	-904.841	1816.41	86.61	1.56192E-19	1.20E-19
WIDTH+FISH_VIS		4	-861.471	1732.19	2.39	0.303044017	0.233
LANE+FISH_VIS		4	-904.811	1818.87	89.07	4.56179E-20	3.50E-20
	$MinAIC_C$			1729.80			
	n	37					

The Poisson-Mixture Model likelihoods are conditional on the Poisson distribution parameter λ, along with *p*. Estimates of λ (Table 7) are also the expected means and variances of abundance (in sites), but require additional interpretation and/or adjustments before they can be used in estimates of total abundance or trend. There are two reasons for this. First, 5 of the 6 long-term index sites are the same length (4 km), but the site Anzac 47-45 km is half this length meaning that λ cannot be considered an unbiased estimate of density. Second, the Poisson-Mixture Model utilized in this exploratory analysis does not account for missing data from sites, which is a relatively frequent occurrence in the 1995-2020 time series for the long-term index sections.

Table 7. Maximum-likelihood estimates of the Poisson distribution parameter λ, the unadjusted mean and variance of expected abundance (see text) in long-term index sites of the Parsnip River watershed, 1995-2020.

Year	1995	1998	2000	2001	2003	2005	2007	2018	2019	2020
λ	163	179	118	88	149	138	121	326	192	265

Modeled detection probability can be compared for Binomial-Likelihood and Poisson-Mixture models by applying the respective best model for p to mean wetted with and visibility data for each long-term index site (Table 8). Detection probability estimates are somewhat higher for the Binomial Likelihood Model but both show a consistent pattern of highest detection probability in the smaller reaches at the top of the Arctic Grayling distribution in each system. Mark-resight data to validate the detection probability estimates exist for just a single site in one year. In 2019, telemetry data from fixed receivers in the upper Table River suggest that 15 acoustic-tagged Arctic Grayling were present between 35-31 km on the date of the snorkeling survey (Joe Bottoms, FFEL, pers. comm. February 2020). On that date (August 15), 12 tagged fish were observed equating to a detection probability estimate of 0.80 (80% confidence interval: 0.61-0.92). This suggests that the relatively high model estimates for this site of 73% and 67%, from the Binomial-Likelihood and Poisson-Mixture models, respectively (Table 8), are plausible.

Table 8. Predictions of mean detection probability p at long-term index sections of the Parsnip River watershed, computed at average values of wetted width (WIDTH) and underwater visibility at which Arctic Grayling can be discerned (FISH_VIS).

	Table River		Anzac River			
Site	35-31	22-18	47-45	43-39	34-30	16-12
WIDTH	14.9	22.5	15.0	27.2	27.3	27.9
FISH_VIS	4.6	4.6	4.0	4.6	5.0	5.1
Binomial-Likelihood Model estimated p	0.73	0.63	0.72	0.56	0.57	0.56
Poisson-Mixture Model estimated p	0.67	0.57	0.67	0.50	0.50	0.49

5.3 Population Trend

Average counts of Arctic Grayling >20 cm in long-term index sections over the 1995-2020 period are depicted in Figure 14. In the Table River, counts over the 2018-2020 period are higher than counts up to 2007, notwithstanding the change from 22-18 km to 26-22 km for the lower site location. In the Anzac River, counts up to 2019 also suggest an increasing trend, although variability is high among years. Variability is especially high for the furthest Anzac River site upstream (Anzac 47-45; Figure 14), located above a 2-km section of rapids that may limit upstream movements in some years.

To account for the unwanted effects of variable detection probability, we utilized model estimates of abundance (see previous section) in our estimates of trend in preference to raw count data. The site- and year-specific population estimates N_{it} from the Binomial-Likelihood Model (Table 4), analyzed using a linear mixed-effects model in which N_{it} and *YEAR* were utilized as fixed effects and *STREAM* and *SITE* as nested random effects, indicated a significant increase in the abundance of Arctic Grayling >20 cm in the Parsnip River watershed (P <0.001) over the 1995-2020 period. A benefit of the linear mixed-effects model framework was that it could account for missing site data which occurs in most years.

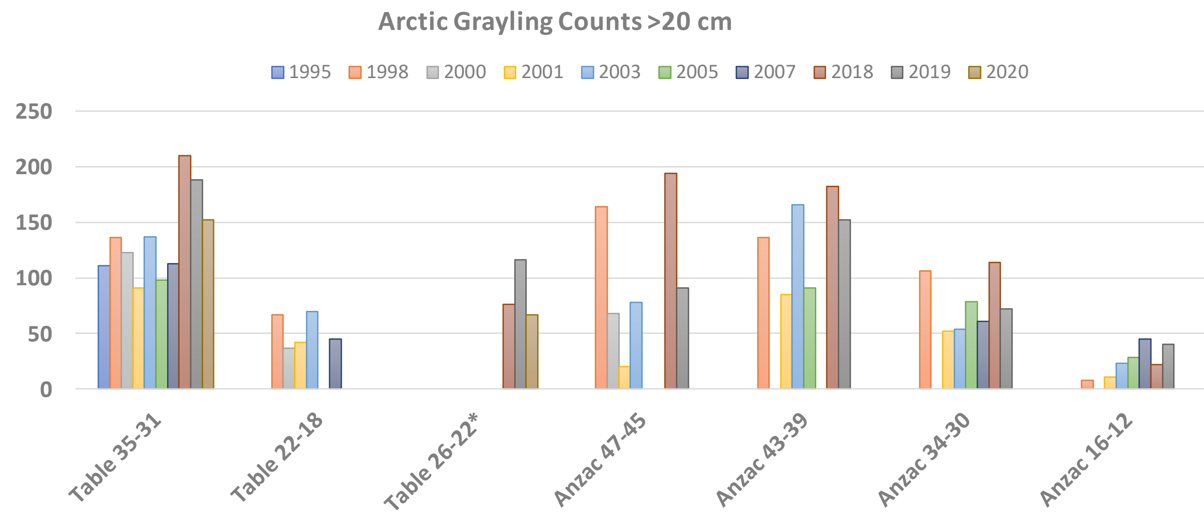

*Figure 14. Snorkeling counts of Arctic Grayling >20 cm in index sites of the Anzac River and Table River watersheds 1995-2020. Values are averages of replicate counts. *Beginning in 2018, Table River section 26-22 was substituted for 22-18 which was affected by a major clay slump at 22 km.*

As described in Section *5.2*, Poisson-Mixture Model estimates of λ require further work before they can be considered reliable estimates of abundance. Nonetheless, model estimates over the 1995-2020 period (Table 7) also indicate a significant increasing linear trend when analyzed using simple linear regression (t = 2.829, P = 0.022).

5.4 Critical Summer Habitats in the Hominka River, Wichcika Creek, and Reynolds Creek

The use of the Hominka River, Wichcika Creek, and Reynolds Creek by adult and subadult Arctic Grayling >20 cm has not been previously assessed. Their potential is known only from unpublished fry (young-of-year)-oriented sampling conducted by FWCP in 2005 (Hagen et al. 2015; Stamford et al. 2017), which revealed low densities of fry in each of these systems (e.g. a single Arctic Grayling fry captured in the Hominka River). In 2020, single-pass snorkeling surveys in reconnaissance sections of these streams (Figures 5-7) identified that all were utilized by populations of adult Arctic Grayling in summer (Table 9). Estimated 2020 abundance N_{it} for each reconnaissance section, computed using equation (3) and measurements of wetted width and underwater visibility, are also presented in Table 9 to facilitate an assessment of the relative importance of these sections relative to long-term index sections in the Table and Anzac rivers (Table 4).

Hominka River

In the Hominka River, several chute obstructions between 62-60 km (Figure 5) have previously been assessed, and together they are highly likely to be a barrier to upstream migration of Arctic

Grayling migration (Hagen et al. 2015). In 2020, on August 18, 21, 22, and 30, we conducted single-pass snorkeling surveys in 8 sections below this barrier totaling 29 km of stream habitat (Figure 5). Counts of Arctic Grayling and estimated abundance in 3-5 km sections of the Hominka River (Table 9) are lower than in comparable sections of the Table and Anzac rivers (Table 4), but reveal a substantial population distributed over at least 20 km (from 48 km to 29 km; Table 9).

In the Hominka River below the chute obstructions, three distinct reaches were evident in 2020 based on stream gradient and snorkeling observations. The Hominka River's meandery, low-gradient lower reach extends from the mouth to approximately 32 km. This reach was not expected a priori to be important for rearing Arctic Grayling in summer, and was sub-sampled in two sections (32-29 km and 10-7 km; Figure 5) which indeed indicated low abundance (Table 9). The middle reach extending from 32 km to 48 km (sections 39-35, 44-41, and 48-44 km; Figure 5) was of higher gradient with a riffle-pool channel morphology, and was sampled in a nearly-continuous manner. This middle reach was the core of the 2020 Arctic Grayling distribution in the Hominka River (Table 9). The upper reach extending from the chutes to 48 km (sections 60-56 km, 56-53 km, and 53-48 km; Figure 5) is relatively steep, and Arctic Grayling were present only at very low densities in the lowest reconnaissance section of this reach (53-48 km).

Snorkeling observations in the middle and upper reaches of the Hominka River watershed also corroborated previous aerial redd count data identifying a substantial population of large-bodied, migratory Bull Trout spawners (Table 9). Rainbow Trout were extremely rare in the Hominka River, while Mountain Whitefish were abundant (Table 9).

Wichcika Creek

Distinct reaches of Wichcika Creek were not evident during our reconnaissance survey, either in terms of channel morphology/gradient or in our snorkeling count data. With the exception of the stream mouth area, this relatively small stream (Figure 9) has a consistent, moderate gradient and riffle-pool channel morphology. Relatively high channel confinement is a consistent feature along the length of Wichcika Creek. We documented use of the system by adult Arctic Grayling up to 27.5 km, (Figure 6) during reconnaissance snorkeling surveys on August 27, 28, and 30. Counts and estimated levels of abundance (Table 9) were extremely low, however, relative to index sections in the Table and Anzac rivers (Table 4), suggesting Wichcika Creek provides somewhat marginal habitat for the species.

During our survey of Wichcika Creek, the upstream limit of migration for Arctic Grayling was not identified, although the stream becomes very small at 37 km likely precluding habitat use by Arctic Grayling adults. To help refine the distribution of the species in summer 2020, we collected water samples at the top boundary of our highest site, at 27.5 km, and upstream of the site at 35 km for eDNA analysis as part of FWCP project no. PEA-F21-F-3198 *Williston Grayling Distribution: Peace, Parsnip, Dinosaur*. The assay for Arctic Grayling eDNA was negative at both locations (Stamford 2021 in prep.), suggesting that the top of Arctic Grayling habitat use in August 2020 was likely 27.5 km.

For the first time, Wichcika Creek has been identified as potential critical habitat for large-bodied, migratory Bull Trout, based on our observation of a single, 70 cm+ male in the 27.5-24 km section (Figure 6; Table 9). In contrast to observations in snorkeling survey sections in the Table, Anzac, and Hominka rivers, Rainbow Trout were more abundant than Arctic Grayling in reconnaissance sections of Wichcika Creek (Table 9), with some fish observed up to 40 cm or more. Similar to other portions of the Parsnip River watershed, Mountain Whitefish were numerically dominant (Table 9).

Reynolds Creek

Use of Reynolds Creek by adult Arctic Grayling was also confirmed during an opportunistic snorkeling survey of a single reconnaissance section (30-26 km; Figure 7) on August 29 (Table 9). This date corresponded to the final visibility assessment of the Anzac River (Table 2). Reynolds Creek surveys were not proposed for 2020, but were attempted when it became clear that the Anzac River surveys would not be possible. The surveyed 30-26 km section was located in the upper reach of Reynolds Creek between the impassable chute at approximately 36 km (Figure 7) and a clay slump at 26 km. All of Reynolds Creek downstream of this slump was unsurveyable on August 29. The 30-26 km section should be re-surveyed in 2021, when Reynolds Creek and adjacent Colbourne Creek are the proposed subjects of reconnaissance surveys, hopefully under better conditions.

Table 9. Snorkeling counts of Arctic Grayling, Bull Trout, Rainbow Trout, and Mountain Whitefish >20 cm in reconnaissance sections of the Hominka River, Wichcika Creek, and Reynolds Creek, August 2020.

Stream	Section	Arctic Grayling	Bull Trout	Rainbow Trout	Mountain Whitefish	*WIDTH*	*FISH_VIS*	Arctic Grayling N_{it}
Hominka River	60-56	0	14	0	162	15.0	8	0
	56-53	0	10	0	150	19.4	10.2	0
	53-48	2	26	0	220	24.7	10.2	3
	48-44	42	30	0	381	16.3	11.6	56
	44-41	23	2	0	144	20.8	8.5	34
	39-35	29	5	1	343	25.2	4	49
	32-29	4	3	0	28	37.0	2.6	10
	10-7	1	1	0	41	26.2	4.5	2
Wichcika Creek	27.5-24	1	0	1	16	13.875	3.5	1
	22.5-20.5	2	3	8	62	15	3.6	3
	15-11	5	0	17	65	17.7	4.6	7
	10-7	0	0	4	9	16.3	3.9	0
	5-1	1	2	14	4	18.4	4.6	1
Reynolds Creek	30-26	6	0	1	121	16.6	2.7	9

5.5 Other Species

In our study, Arctic Grayling were the first priority for snorkeling observations and also our focus for analyses of abundance and trend. However, Bull Trout, Rainbow Trout, and Mountain Whitefish were also counted simultaneously in long-term index sections of the Anzac and Table rivers (Figures 15-17; Table 10).

Bull Trout counts in index sites are highly variable among years potentially indicating an effect of stream conditions on pre-spawning migration and staging behaviour (Figure 15, Table 10). For example, counts of Bull Trout in 2018 were above long-term averages at most index sites, but this may be an artefact of record low water conditions reducing the suitability of spawning tributaries for staging prior to spawning (Hagen et al. 2019). A more reliable methodology for monitoring Bull Trout abundance in the Parsnip River watershed is through counts of gravel nests, or 'redds' following the completion of spawning (Hagen et al. 2015).

Snorkeling counts of Rainbow Trout (Figure 16, Table 10) are also highly variable among years. The time series of snorkeling count data indicates that Rainbow Trout are rarely abundant at index sites (Figure 16). Low Rainbow Trout counts are of interest because of potential interspecific competition among Rainbow Trout, Arctic Grayling, and Bull Trout, with Rainbow Trout expected to become increasingly more prevalent as systems warm (Parkinson and Haas 1996; Parkinson et al. 2012; Hawkshaw et al. 2013; Hawkshaw and Shrimpton 2014). Although we did not conduct an analysis of trend for Rainbow Trout abundance, visual inspections of the time series of count data do not indicate obvious increases at this point in time.

Counts of Mountain Whitefish (Figure 17, Table 10) are especially variable. Mountain Whitefish are far too numerous to count reliably and were assigned the lowest priority during our snorkeling surveys. Therefore, Mountain Whitefish counts should be considered of low precision and accuracy relative to the other three species. This prioritization was obviously in place during previous surveys also: Mountain Whitefish counts for 2005 are missing altogether. Irrespective, visual inspections of the time series of counts at index sites do not indicate obvious cause for conservation concern (i.e. low abundance, declining trend) for this species.

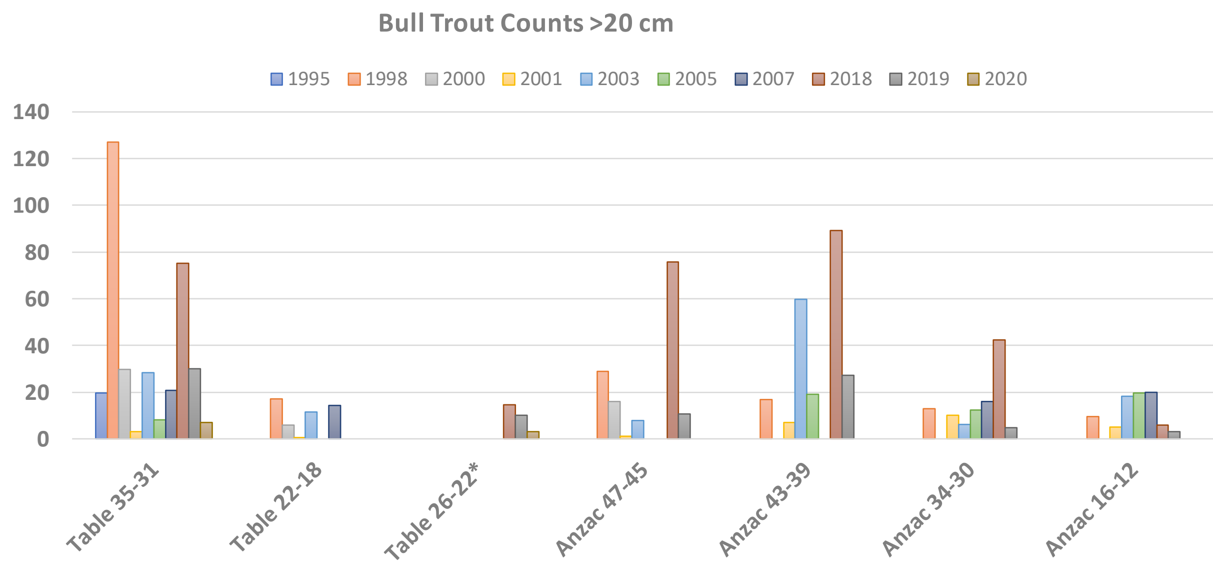

*Figure 15. Counts of Bull Trout >20 cm in sections of the Anzac and Table River watersheds, 1995-2020. *Table 26-22 is a replacement for Table 22-18 beginning in 2018.*

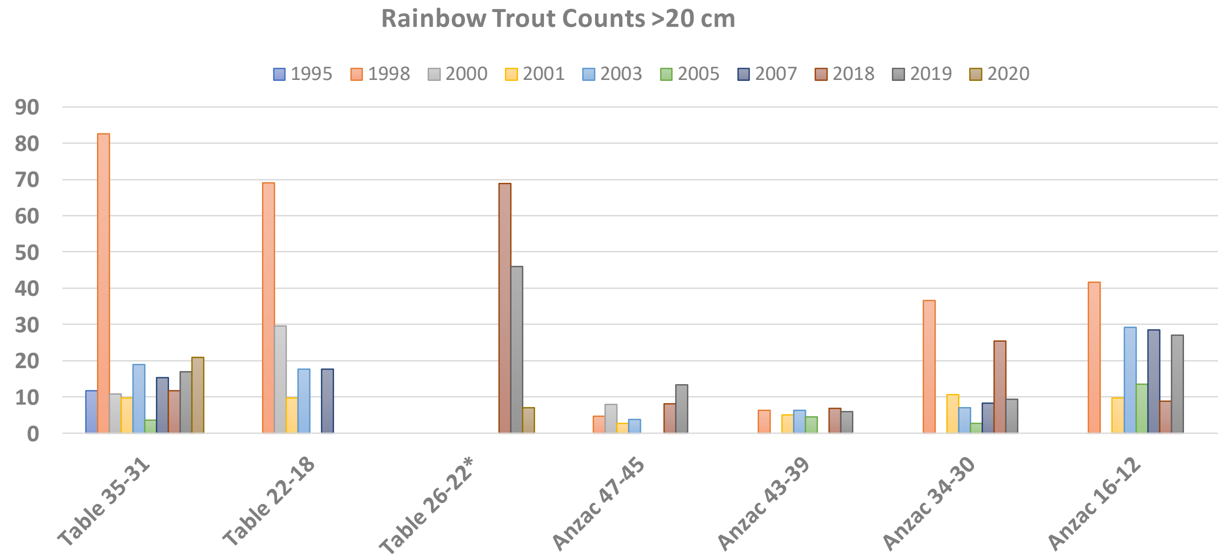

*Figure 16. Counts of Rainbow Trout >20 cm in sections of the Anzac and Table River watersheds, 1995-2020. *Table 26-22 is a replacement for Table 22-18 beginning in 2018.*

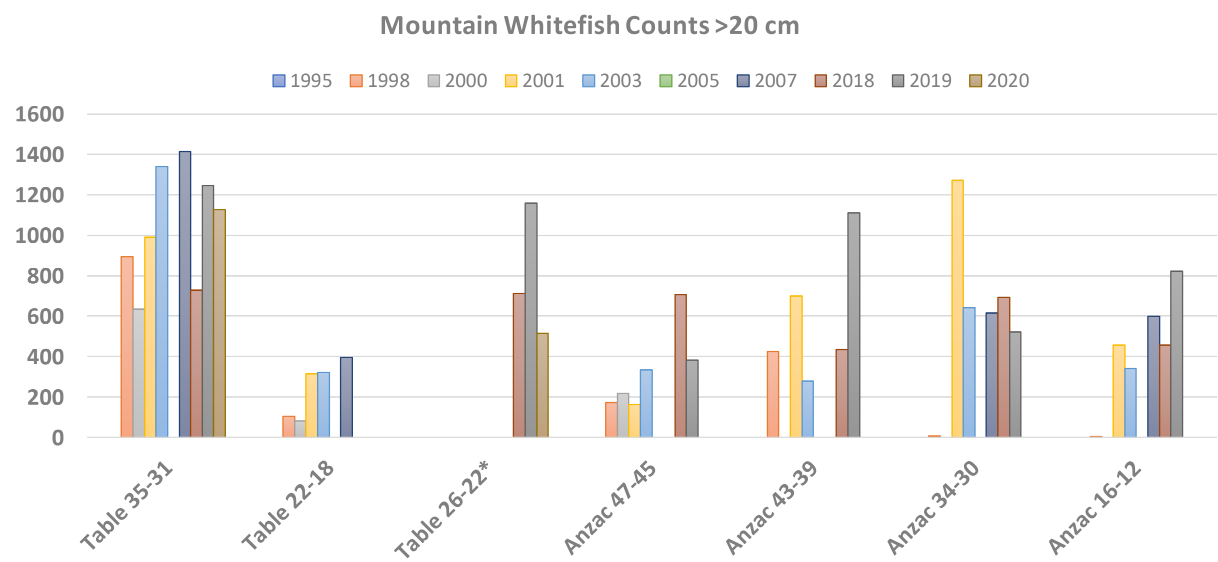

*Figure 17. Counts of Mountain Whitefish >20 cm in sections of the Anzac and Table River watersheds, 1995-2020. *Table 26-22 is a replacement for Table 22-18 beginning in 2018.*

Table 10. Average counts of Arctic Grayling, Bull Trout, Rainbow Trout, and Mountain Whitefish in long-term index sections of the Table and Anzac rivers, 1995-2020.

		Table River Sites			Anzac River Sites			
Year	Species	Table 35-31	Table 22-18	Table 26-22*	Anzac 47-45	Anzac 43-39	Anzac 34-30	Anzac 16-12
1995	GR	111						
	BT	20						
	RB	12						
	MW							
1998	GR	136	67		164	136	106	8
	BT	127	17		29	17	13	10
	RB	83	69		5	6	37	42
	MW	894	105		170	426	8	1
2000	GR	123	37		68			
	BT	30	6		16			
	RB	11	30		8			
	MW	636	82		217			
2001	GR	91	42		20	85	52	11
	BT	3	1		1	7	10	5
	RB	10	10		3	5	11	10
	MW	991	315		161	700	1272	458
2003	GR	137	70		78	166	54	23
	BT	28	12		8	60	6	18
	RB	19	18		4	6	7	29
	MW	1341	320		333	277	641	340
2005	GR	98				91	79	29
	BT	8				19	12	20
	RB	4				5	3	14
	MW							
2007	GR	113	45				61	45
	BT	21	14				16	20
	RB	15	18				8	29
	MW	1415	394				616	600
2018	GR	210		76	194	182	114	22
	BT	75		14	76	89	42	6
	RB	12		69	8	7	25	9
	MW	730		711	705	433	692	458
2019	GR	188		116	91	152	72	40
	BT	30		10	11	27	5	3
	RB	17		46	13	6	9	27
	MW	1246		1160	383	1111	522	821
2020	GR	152		67				
	BT	7		3				
	RB	21		7				
	MW	1128		516				

*replacement for Table River section 22-18 beginning in 2018.

6.0 DISCUSSION

6.1 Accounting for Detection Probability in Estimates of Abundance and Trend

Analytical approaches

In 2020, we learned relatively little about Arctic Grayling abundance in long-term index sections of the Table and Anzac Rivers, given that 4 of 6 sections could not be surveyed due to poor underwater visibility. In this third year of a proposed 5-year monitoring program, we have nonetheless advanced our understanding of abundance and trend by exploring new analytical approaches.

A key area of focus for the 2020 Arctic Grayling snorkeling study was the exploratory application of models that analyze replicated count data to estimate detection probability and abundance (Olkin et al. 1981; Royle 2004). The approach has two potential advantages over more traditional mark-resight studies in its application to snorkeling count data in the Parsnip River watershed. First, reliable information from the mark-resight approach is limited to a single index section (Table River 35-31) in 2019 (Section *5.2*). Previous attempts at mark-resight validation in the Parsnip River watershed were abandoned and the results considered biased based on evidence of movement of marked fish out of index sections before they could be surveyed (Cowie and Blackman 2012). Second, snorkeling surveys in long-term index sections of the Table and Anzac rivers have replicated at least twice in all but a small number of cases (37 of 44 over the 1995-2007 period; Table 4), meaning there is a rich data set available for analysis using the no-mark modeling approach.

In this report, we demonstrate that it is feasible to apply two types of models to replicated snorkeling count data from the Parsnip River watershed: 1) a conceptually-simple Binomial-Likelihood Model that conditional on detection probability p and site-specific abundance N_{it}; and 2) a Poisson-Mixture Model conditional on p, site-specific abundance N_{it}, and integration over a prior distribution (Poisson) on mean annual abundance N_t (Royle 2004). Furthermore, modeling results appear plausible: maximum-likelihood estimates of p (p = 0.73, 0.67 for the Binomial-Likelihood and Poisson-Mixture models, respectively) are in decent agreement with the one existing mark-resight estimate of p available from Table River 35-31 (0.80; 80% confidence interval: 0.61-0.92). Estimates of trend, derived using the maximum-likelihood estimates of abundance from each of the models, are in agreement and suggest a significant increase in Arctic Grayling abundance has occurred in the Parsnip River watershed over the 1995-2020 period.

In the manner we applied them, however, these models have limitations, suggesting further developments in the analytical approach are needed in the proposed final two years of this project. The Binomial-Likelihood Model is known to suffer from unstable estimates particularly when count data are sparse or p low (Olkin et al. 1981; Royle 2004). This is not the case in long-term index sections of the Parsnip River watershed, but these scenarios are likely when adapting the model to new watersheds with limited prior replication data.

The Poisson-Mixture Model results address the need to integrate site data and generate abundance estimates at the scale of the whole study area (the Poisson distribution parameter λ is the mean and variance of expected density among index sections). This strength is also a limitation, given that we are not just interested in mean abundance, but also in how abundance changes in relation to potential limiting factors (e.g. water temperature, landscape characteristics, time – Section *6.2*). Including these potential limiting factors as covariates in the Poisson-Mixture Model (e.g. Royle 2004) is a step that still needs to occur prior to the study completion in 2022.

Given that Binomial-Likelihood Model performed adequately within the relatively rich dataset we had to work with, its results may be a good benchmark for refining *N*-Mixture Models (Royle 2004) in future. For example, Poisson-Mixture Model estimates of *p* were consistently lower (Table 8), potentially indicating that other distributions on *N* should also be evaluated (e.g. Joseph et al. 2009). Our partner in advancing the analytical approach will be the Freshwater Fish Ecology Laboratory (FFEL) at UNBC (Dr. Eduardo Martins, pers. comm. March 2021), funded by a separate funding proposal submitted to the Freshwater Fisheries Society of British Columbia in November, 2021.

Arctic Grayling detection probability

A key result of our analysis was that estimated detection probability *p* was relatively high for Arctic Grayling, despite relatively low levels of underwater visibility (Table 8). Species differences in behaviour may mediate the effect of underwater visibility on *p*. Rainbow Trout in particular appear to react to the line of divers and can move out of the range of detection at low levels of underwater visibility, resulting in a stronger relationship between visibility and *p* (Northcote and Wilkie 1963; Korman et al. 2002; Hagen and Baxter 2005). In contrast, consistently high levels of detection probability have been recorded for Westslope Cutthroat Trout across a range of underwater visibility levels (e.g. p = 74% at 3 m visibility for Slaney and Martin 1987; p = 79%, 81% at 12.9 m, 12.2 m, respectively for Hagen and Baxter 2005). We have observed that Arctic Grayling appear to be highly similar to Westslope Cutthroat in their behaviour, often holding station and continuing to feed immediately in front of the masks of the approaching divers.

It is not surprising that stream width was the most important predictor of *p* in our analysis. This is a logical potential relationship given that two-person crews are utilized in all index reaches regardless of stream size. In the Thompson River watershed, for example, cross-sectional area of sites has been found to be the most important physical habitat variable affecting snorkeling detection probability of two-person crews for juvenile steelhead (Hagen et al. 2010). It was more surprising that stream width was a much better predictor of *p* than lane width, although this finding corroborates the assertion by Cowie and Blackman (2012) that the addition of a third observer to snorkeling crews did not make a difference to the counts.

Development of a generalized model for Arctic Grayling snorkeling *p*, in which the wealth of data from the Parsnip long-term index sections are augmented by replicated count data from other systems, may be a desirable step for future. Such data may strengthen relationships with predictor variables such as underwater visibility by providing increased levels of contrast. For example, replicated count data in the Ingenika River watershed, validated by a mark-resight study, are now available as an outcome of FWCP Project PEA-F20-F-2963 *Ingenika Arctic Grayling Monitoring* (Hagen and Stamford 2021 in prep.). Horizontal underwater distances at which Arctic Grayling can be discerned are typically 6-9 m in the extremely-clear Ingenika River (Strohm et al. 2020).

Population trend

The increasing trend of Arctic Grayling in index reaches of the Parsnip River watershed over the 1995-2020 period is obviously encouraging, and has important implications for British Columbians. Most importantly, it appears that human use of Parsnip Arctic Grayling in catch-and-release sport fisheries and First Nations subsistence fisheries has been sustainable over this period. The introduction of the catch-and-release regulation in 1996 is a plausible potential factor behind the population increase. Physical habitat conditions also appear to have remained productive for Table River and Anzac River Arctic Grayling populations. The Table River in particular was subjected to intensive forestry prior to the 1995-2007 monitoring period, and the recovery of critical habitat over time is a second plausible factor behind the positive trend.

However, as discussed in the following section, rapid increases in industrial activity and angler access in the Parsnip River watershed, associated with forestry and pipeline initiatives, are major potential threats to Arctic Grayling productivity. To assess the potential effects of these threats, future analyses of trend may need to shift focus and evaluate changes in Arctic Grayling abundance following these new watershed developments, rather than keeping to the current focus on the monotonic trend over the entire 1995-2020 time period.

6.2 Critical Habitats, Conservation Actions, and Limiting Factors

At the time of the 2017 FWCP Arctic Grayling information synthesis (Stamford et al. 2017), there was a great deal of uncertainty about the distribution and abundance of adult and subadult grayling in the Parsnip River watershed outside of the Table and Anzac sub-basins. Stamford et al. (2017) speculated that there may be a second hub of Arctic Grayling abundance in the upper Parsnip River watershed, in addition to a known, major population centered around the Anzac and Table rivers. With the completion of reconnaissance snorkeling surveys in the Hominka River and Wichcika Creek in 2020, following an assessment of the Missinka River in 2019 (Hagen and Gantner 2020), we can now address this speculation.

Arctic Grayling are present and widely distributed in the upper Parsnip River watershed, utilizing at least 34 km of the Missinka River (Hagen and Gantner 2020), 48 km of the Hominka River, and 27.5 km of Wichcika Creek (Table 9). Wichcika Creek appears to be somewhat marginal for Arctic Grayling, with very low densities numerically-dominated by Rainbow Trout. The most productive summer rearing habitats for adult Arctic Grayling in the upper Parsnip River watershed are distributed from 36-29 km of the Missinka River (Hagen and Gantner 2020), and from 48-32 km of the Hominka River (Table 9). However, snorkeling counts and estimated abundance of Arctic Grayling in reconnaissance sections of these watersheds (Table 9) are much lower than in the Table and Anzac Rivers (Table 4). In the Anzac River, high Arctic Grayling abundance extends over a 30-km zone between a chute obstruction at 47 km to 16 km (Hagen et al. 2019). The distribution of high Arctic Grayling abundance has not been defined with reconnaissance surveys in the Table River, but it extends over a minimum 20-km zone from the waterfall migration barrier at 37 km to 18 km (bottom of long-term index section 22-18 km; Figure 14). These zones of the Anzac and Table Rivers should therefore be treated as the highest priorities for habitat conservation and enhancement actions. The key summer rearing zones of the Missinka and Hominka rivers nonetheless warrant special habitat management to maintain their productivity. Currently, *Fisheries Sensitive Watershed* designations have been applied to all four of these watersheds by the Government of British Columbia, with special land use objectives in place designed to limit impacts to water quality (Sandra Sulyma, FLNRORD Ecosystems Section, pers. comm. April 2020). However, should evidence of habitat degradation in the Anzac River watershed in 2020 (Section *5.1*) prove to be a longer-term problem, it would indicate that additional habitat conservation, restoration, and enhancement measures are warranted.

As mentioned previously (Section *1.0*), The Arctic Grayling is a species for which land use-related habitat degradation has been linked to population declines (Armstrong 1986; Northcote 1993; Walker 2005; USFWS 2010; Cahill 2015), and a major increase in industrial activity has occurred since 2019 in sub-basins of the Parsnip River system. This is mostly due to a salvage logging initiative related to an outbreak of the spruce beetle, which will involve incursions of forestry and related road building into currently-unroaded areas throughout the watershed. In the Anzac River sub-basin, which contains the core of adult Arctic Grayling summer rearing habitat in the Parsnip River watershed, the potential cumulative effects of intensive forestry, Coastal GasLink pipeline construction, and associated road building are especially concerning. The heavy silt load in the Anzac River in 2020 appears to be at least partially related to this industrial activity (Section *5.1*).

To discern the cumulative effects of land use, climate change, and other potential limiting factors on Arctic Grayling populations, fish and fish habitat monitoring in the Parsnip River watershed may require additional effort, some of which is likely to be beyond the scope of this study. Our efforts to increase the accuracy and precision of abundance estimates in long-term index sections (Section *6.1*), and to establish baseline estimates of abundance and critical summer rearing habitats in other sub-basins, are key to assessing land use-related habitat degradation as a potential limiting factor. There are, however, additional information needs for a thorough assessment of limiting factors. These include:

1. indices of watershed hydrology (Tack 1974; Clark 1992): e.g. sediment monitoring, peak flow magnitude and timing, summer base flow levels,
2. water temperature (Ballard and Shrimpton 2009; Hawkshaw and Shrimpton 2014; Hawkshaw et al. 2013),
3. abundance of competitors/predators (Clark 1992; Buzby and Deegan 2004),
4. stream gradient and other fluvial geomorphology variables (Blackman 2004; Lamothe and Magee 2004),
5. stream nutrients (Wilson et al. 2008; Deegan and Peterson 1992), and
6. distance from overwintering and spawning locations (Blackman 2002).

A quantitative assessment of some or all of these potential limiting factors will be completed prior to the completion of this proposed 5-year study in the 2022-23 funding cycle. By mutual agreement, this assessment will involve a collaboration with FLNRORD specialists and UNBC's FFEL. With improved knowledge of critical habitats and limiting factors, FWCP and its partners will be better able to identify and prioritize potential conservation, restoration, and enhancement actions.

7.0 RECOMMENDATIONS

Our multi-year study addresses two high-priority knowledge gaps identified in FWCP's Arctic Grayling Synthesis Report (Stamford et al. 2017): 1) the lack of monitoring data from long-term index sections of the Parsnip River watershed since 2007, and 2) the lack of monitoring data delineating critical habitats outside of the Table and Anzac rivers.

With respect to these two information deficiencies, we consider both study components to be key to understanding patterns of distribution and abundance in the Parsnip core area. Population abundance and trend are the most important indicators of the viability of remnant populations of Williston Arctic Grayling following flooding, and of limiting factors affecting population productivity. Data from the long-term index sections provide critical evidence of population trend for statistical analysis. Data from both the long-term index sections and the reconnaissance sections provide evidence of total population size and the distribution of critical habitats at the scale of whole sub-basins. Estimates of trend, population size, and critical habitat locations are all important for identifying limiting factors.

To improve the ability of the snorkeling program to detect changes in abundance and identify limiting factors in the proposed 4^{th} and 5^{th} years of this 5-year study, we have the following recommendations:

1. Continue monitoring of Arctic Grayling abundance in long-term index sites of the Anzac and Table rivers, using replicated snorkeling surveys.

2. Continue collaboration with UNBC's FFEL on improved models for estimating detection probability and abundance. Key areas for consultation are: i) the value of replication in reconnaissance sections vs. long-term index sections, ii) alternative prior distributions on *N* for the Poisson-Mixture Model, and iii) covariates of *N* (i.e., potential limiting factors) to include in the Poisson-Mixture Model.

3. Conduct a quantitative assessment of potential limiting factors affecting distribution and abundance of subadult/adult Arctic Grayling in the Parsnip River watershed, and identify implications for habitat conservation, restoration, and enhancement actions (including specific recommendations where possible).

4. For 2021 and 2022, conduct single-pass reconnaissance surveys in 4-km index sections spaced along the accessible length of Reynolds Creek, Colbourne Creek, and the Misinchinka River. For Colbourne Creek and the Misinchinka River, there are no records of adult Arctic Grayling summer habitat use since the 1970s (Hagen et al. 2015; Stamford et al. 2017). The mouths of these streams are closer to the reservoir's influence, meaning that the likelihood of extirpation is greater (Hawkshaw 2013; Hawkshaw and Shrimpton 2014; Stamford et al. 2017). The discovery of remnant populations in the Colbourne Creek and Misinchinka River would be highly significant for this very reason. Importantly, the completion of reconnaissance surveys among major sub-basins of the Parsnip River watershed will allow the best basis for assessments of limiting factors, critical habitat locations, and total subadult/adult abundance.

5. Maintain dialogue with McLeod Lake Indian Band to identify opportunities for information exchange, training, and employment.

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9.0 REFERENCES

Anonymous. 1978. Report on environmental studies for the McGregor Diversion Project. Report prepared by Reid, Crowther, and Partners, Ltd. for BC Hydro and Power Authority.

Armstrong, R.H. 1986. A review of the Arctic grayling studies in Alaska, 1952-1982. Biol. Paper Univ. Alaska No. 23:3-17.

Ballard. S and J.M. Shrimpton. 2009. Summary Report of Arctic Grayling Management and Conservation 2009. A synopsis of the information available on Arctic grayling in the Omineca region of northern British Columbia and identification of additional information needs. Peace/Williston Fish and Wildlife Compensation Program Report No. 337. 66 pp plus appendices.

Birtwell, I.K., Hartman, G.F., Anderson, B., McLeay, D.J., and Malick, J.G. 1984. A brief investigation of Arctic grayling (*Thymallus arcticus*) and aquatic invertebrates in the Minto Creek drainage, Mayo, Yukon Territory: an area subjected to placer mining. Can. Tech. Rep. Fish. Aquat. Sci. 1287: xi + 57 p.

Blackman B.G. 2004. 1997 Arctic grayling habitat use in the Table and Anzac rivers with ancillary information on other associated species. Peace/Williston Fish and Wildlife Compensation Program Report No. 296. 34 pp plus appendices.

Blackman, B.G. 2002. Radio telemetry studies of Arctic grayling migrations to overwinter, spawning and summer feeding areas in the Parsnip River watershed. Peace/Williston Fish and Wildlife Compensation Program, Prince George, B.C.

Blackman, B.G. 2001. An Information Summary on Arctic Grayling from the Williston Reservoir Watershed with Data Gap Analysis and Recommendations. PWFWCP Report No. 258.

Bruce, P.G., and P.J. Starr. 1985. Fisheries resources and fisheries potential of Williston Reservoir and its tributary streams, volume II: fisheries resources potential of Williston Lake tributaries - a preliminary overview. BC Government Fisheries Technical Circular No. 69.

Burnham, K.P., and D.R. Anderson. 2002. Model selection and inference: an information-theoretic approach. Springer-Verlag, New York.

Buzby, K.M. and L.A. Deegan. 2004. Long-term survival of adult Arctic grayling (*Thymallus arcticus*) in the Kuparuk River, Alaska. Can. J. Fish. Aquat. Sci. 61: 1954-1964.

Cahill, C.C. 2015. Status of the Arctic Grayling in 2015: update 2015 Report prepared for Alberta Environment and Parks and Alberta Conservation Association. Alberta Wildlife Status Report no. 57 (Update 2015).

Caughley, G. 1994. Directions in conservation biology. Journal of Animal Ecology 63: 215-244.

Clark, R.A. 1992. Influence of stream flows and stock size on recruitment of Arctic grayling (*Thymallus arcticus*) in the Chena River, Alaska. Can. J. Fish. Aquat. Sci. 49:1027-1034.

Clarke, A.D., K. Telmer and J.M. Shrimpton. 2005. Population structure and habitat use by Arctic grayling (*Thymallus arcticus*) in tributaries of the Williston Reservoir using natural elemental signatures. Peace/Williston Fish and Wildlife Compensation Program Report No. 300. 62p. + 1 Appendix.

Cowie, D.M. 2021. Data Compilation and Summary of Reports Related to Arctic Grayling Studies in the Table, Anzac and Parsnip Rivers, 1995-2007. Data report prepared for the Fish and Wildlife Compensation Program – Peace Region and BC Ministry of Forests, Lands, Natural Resource Operations, and Rural Development, Prince George, BC.

Cowie D.M. and B.G. Blackman. 2012. An Overview and Summary of Methodologies of Arctic grayling (*Thymallus arcticus*) Projects Conducted in the Parsnip, Table, and Anzac Rivers from 1995 to 2007. Fish and Wildlife Compensation Program – Peace Region Report No. 349. 24 pp. plus appendices.

Cowie D.M. and B.G. Blackman. 2003. 2001-2002 Arctic grayling (*Thymallus arcticus*) Fry Surveys in the Omineca and Osilinka Rivers. Peace/ Williston Fish and Wildlife Compensation Program Report No. 276. 18 pp plus appendices.

de Bruyn, M., and P. McCart. 1974. Life history of the grayling (*Thymallus arcticus*) in Beaufort Sea drainages in the Yukon Territory. *In* P.J. McCart [ed.] Arctic Gas Biological Report Series, Vol. 15, Chapter 2, 41p.

Deegan, L.A., H.E. Golden, C.J. Harvey, and B. J. Peterson. 1999. Influence of Environmental Variability on the Growth of Age-0 and Adult Arctic Grayling. Trans, Am. Fish. Soc. 128(6): 1163-1175.

Deegan, L.A., and B.J. Peterson. 1992. Whole-river fertilization stimulates fish production in an Arctic tundra river. Can. J. Fish. Aquat. Sci. 49: 1890-1 901.

Franklin, I.A. 1980. Evolutionary changes in small populations. Pages 135–150 *in* M. Soule and B. A. Wilcox, editors. Conservation biology: an evolutionary perspective. Sinauer Associates, Sunderland, Massachusetts.

FWCP. 2014. Fish and Wildlife Compensation Program – Peace Basin Streams Action Plan. BC Hydro, Fort St. John, BC.

Hagen, J. and J.S. Baxter. 2005. Accuracy of diver counts of fluvial rainbow trout relative to horizontal underwater visibility. North American Journal of Fisheries Management 25:1367-1377.

Hagen, J., A.S. Decker, J. Korman, and R. Bison. 2010. Effectiveness of night snorkeling for estimating steelhead parr abundance in a large river basin. North American Journal of Fisheries Management 30:1303-1314.

Hagen, J., and N. Gantner. 2020. Abundance and Trend of Arctic Grayling (*Thymallus arcticus*) in Index Sites of the Parsnip River Watershed, 1995-2019. Report prepared for the Fish and Wildlife Compensation Program – Peace Region, Prince George, BC. FWCP project no. PEA-F20-F-2959.

Hagen, J., R. Pillipow, and N. Gantner. 2019. Trend in Abundance of Arctic Grayling (*Thymallus arcticus*) in Index Sites of the Parsnip River Watershed, 1995-2018. Report prepared for the Fish and Wildlife Compensation Program – Peace Region, Prince George, BC. FWCP project no. PEA-F19-F-2625.

Hagen, J., and M. Stamford. 2017. FWCP Arctic Grayling Monitoring Framework for the Williston Reservoir Watershed. Prepared for Fish and Wildlife Compensation Program, Peace Region. http://fwcp.ca/app/uploads/2017/07/FWCP-Arctic-Grayling-Monitoring-Framework_Final.pdf

Hagen, J., and M. Stamford. 2021 in prep. Ingenika Arctic Grayling Monitoring 2004-2020. Report prepared for the Fish and Wildlife Compensation Program – Peace Region, Prince George, BC. FWCP project no. PEA-F20-F-2963

Hagen, J., Williamson, S., Stamford, M.D., and R. Pillipow. 2015. Critical Habitats for Bull Trout and Arctic Grayling within the Parsnip River and Pack River watersheds. Report prepared for: McLeod lake Indian Band, McLeod Lake Indian Band, Sekani Drive, McLeod Lake, BC.

Hawkshaw, S.C.F. and J.M. Shrimpton. 2014. Temperature preference and distribution of juvenile Arctic grayling (*Thymallus arcticus*) in the Williston Watershed, British Columbia, Canada. Fish and Wildlife Compensation Program – Peace Region Report No. 366. 40 pp plus appendices.

Hawkshaw, S.C.F., Gillingham, M.C. and J.M. Shrimpton. 2013. Habitat characteristics affecting occurrence of a fluvial species in a watershed altered by a large reservoir. Ecology of Freshwater Fish, 12pp.

Hirst, S.M. 1991. Impacts of the operations of existing hydroelectric developments on fishery resources in British Columbia, Volume II: inland fisheries. 91. Canada. Department of Fisheries and Oceans. Canadian Manuscript Report of Fisheries and Aquatic Sciences.

Joseph, L.N., C. Elkin, T.G. Martin, and H.P. Possingham. 2009. Modeling abundance using *N*-mixture models: the importance of considering ecological mechanisms. Ecological Applications 19: 631-642.

Korman, J., R.N.M. Ahrens, P.S. Higgins, and C.J. Walters. 2002. Effects of observer efficiency, arrival timing, and survey life on estimates of escapement for steelhead trout (*Oncorhynchus mykiss*) derived from repeat mark-recapture experiments. Canadian Journal of Fisheries and Aquatic Sciences 59:1116-1131.

Lamothe, P. and J. Magee. 2004. Reintroducing fluvial Arctic grayling (*Thymallus arcticus*): A Summary Report Based on the efforts to reintroduce Arctic grayling to the upper Ruby River, MT. Montana Fish, Wildlife & Parks.

McElhany, P., M.H. Ruckelhaus, M.J. Ford, T.C. Wainwright, and E.P. Bjorkstedt. 2000. Viable salmonid populations and the recovery of evolutionarily significant units. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-42.

McLeay B. J., Birtwell I. K., Hartman G. F., and G. L. Ennis. 1987. Responses of Arctic grayling (*Thymallus arcticus*) to acute and prolonged exposure to Yukon placer mining sediment. Can. 1. Fish. Aquat. Sci. 44: 658-673.

Northcote, T.G. 1993. A review of management and enhancement options for the Arctic grayling (*Thymallus arcticus*) with special reference to the Williston Reservoir watershed in British Columbia. Province of British Columbia, Ministry of Environment Lands and Parks, Fisheries Branch. Fisheries Management Report No. 101: 69pp.

Northcote, T.G., and D.W. Wilkie. 1963. Underwater census of stream fish populations. Transactions of the American Fisheries Society 92:146-151.

Nunney, L., and K.A. Campbell. 1993. Assessing minimum viable population size: demography meets population genetics. Trends in Ecology and Evolution 8: 234-239.

O’Grady, J.J., D.H. Reed, B.W. Brook, and R. Frankham. 2004. What are the best correlates of predicted extinction risk? Biological Conservation 118: 513–520

Olkin, I., A.J. Petkau, and J.V. Zidek. 1981. A comparison of N estimators for the binomial distribution. Journal of the American Statistical Association 76: 637-642.

Parkinson, E., and G. Haas. 1996. The role of macrohabitat variables and temperature in defining the range of bull trout. Province of British Columbia Fisheries Project Report 51.

Parkinson, E.A., E. Lea, and M.A. Nelitz. 2012. A framework for designating “Temperature Sensitive Streams” to protect fish habitat, Part 2: Identifying temperature thresholds associated with fish community changes in British Columbia, Canada. Province of BC, Ministry of Environment, Ecosystem Protection & Sustainability Branch, Fisheries Technical Report No. RD111.

Pearce, T., Lieske, S., Morgan, J., A. Solonas. 2019. First Nations Information Gathering on Kokanee, Bull Trout and Arctic Grayling. Project Number: PEA-F19-F-2866. Prepared for Fish and Wildlife Compensation Program and Tse’Khene Nation (McLeod Lake Indian Band).

Rabe-Hesketh, S., and A. Skrondal. 2008. Multilevel and Longitudinal Modelling Using Stata. 2nd Ed. Stata Press, College Station, TX.

Reynolds, J.B., Simmons, R.C., and A.R. Burkholder. 1989. Effects of placer mining discharge on health and food of Arctic grayling. Journal of the American Water Resources Association 25(3): 625–635.

Richardson, J.S., E.B. Taylor, D. Schluter, M. Pearson, and T. Hatfield. 2010. Do riparian zones qualify as critical habitat for endangered freshwater fishes? Canadian Journal of Fisheries and Aquatic Sciences 67: 1197-1204.

Rosenfeld, J.S, and T. Hatfield. 2006. Information needs for assessing critical habitat of freshwater fish. Canadian Journal of Fisheries and Aquatic Sciences 63 (3): 683-698.

Royle, J.A. 2004. *N*-Mixture models for estimating population size from spatially replicated counts. Biometrics 60: 108-115.

Schill, D.J. and J.S. Griffith. 1984. Use of underwater observations to estimate cutthroat trout abundance in the Yellowstone River. North American Journal of Fisheries Management 4:479-487.

Slaney, P.A. and A.D. Martin. 1987. Accuracy of underwater census of trout populations in a large stream in British Columbia. North Am. J. Fish. Manage. 7:117-122.

Stamford, M.D., Hagen, J. and S. Williamson. 2017. FWCP Arctic Grayling Synthesis Report: Limiting Factors, Enhancement Potential, Conservation Status, and Critical Habitats for Arctic Grayling in the Williston Reservoir Watershed, and Information Gaps Limiting Potential Conservation and Enhancement Actions. Prepared for Fish and Wildlife Compensation Program, Peace Region.
http://fwcp.ca/app/uploads/2017/07/FWCP_Grayling_Synthesis_Final.pdf

StataCorp. 2009. Stata Longitudinal-data/Panel-data Reference Manual, Release 11. Stata Press, College Station, TX.

Tack, S.L. 1974. Distribution, abundance, and natural history of Arctic grayling in the Tanana River drainage. Alaska Department of Fish and Game, Juneau, Alaska. Federal Aid in Fish Restoration, Annual Performance Report Project F-9-6.

United States Fish and Wildlife Service (USFWS). 2010. Endangered and Threatened Wildlife and Plants; Revised 12-Month Finding to List the Upper Missouri River Distinct Population Segment of Arctic Grayling as Endangered or Threatened. Federal Register / Vol. 75, No. 173 / Wednesday, September 8, 2010 / Proposed Rules.

Walker, J. 2005. Status of the Arctic grayling in Alberta. Prepared for Alberta Sustainable Resource Development, Fish and Wildlife Division, and Alberta Conservation Association. Wildlife Status Report No. 57.

Wilson, G.A., K. I. Ashley, P.A. Slaney and R.W. Land. 2008. Williston Reservoir River Fisheries Restoration: The Mesilinka River Fertilization Experiment, 1992-99. Peace/Williston Fish and Wildlife Compensation Program Report NO. 319. 95 pp + appendices.

Young, R.G., and J.W. Hayes. 2001. Assessing the accuracy of drift-dive estimates of brown trout (*Salmo trutta*) abundance in two New Zealand Rivers: a mark-resighting study. New Zealand Journal of Marine and Freshwater Research 35:269-275.

Zubik, R.J., and J.J. Fraley. 1988. Comparison of snorkel and mark-recapture estimates for trout populations in large streams. North American Journal of Fisheries Management 8:58-62.