Site C Clean Energy Project

Synthesis Review of the Site C Fish Stranding Monitoring Program (Mon-12)

Construction Years 3 to 6 (2017 to 2020)

Colby Whelan, MSc, RPBio
Ecofish Research Ltd.

Noel Swain, MSc, RPBio
Ecofish Research Ltd.

Matthew Sparling, MSc
Ecofish Research Ltd.

Steve Nicholl, BSc, RPBio
Ecofish Research Ltd.

Adam Lewis, MSc, RPBio
Ecofish Research Ltd.

March 2023

Site C Mon-12

Fish Stranding Monitoring Program
Synthesis Review

Prepared for:

BC Hydro and Power Authority
333 Dunsmuir Street
Vancouver, BC, V6B 5R3

March 30, 2023

Prepared by:

Ecofish Research Ltd.

Photographs and illustrations copyright © 2023

Published by Ecofish Research Ltd., 600 Comox Rd., Courtenay, B.C., V9N 3P6

For inquiries contact: Technical Lead documentcontrol@ecofishresearch.com 250-334-3042

Citation:

Whelan, C., N. Swain, M. Sparling, S. Nicholl. A. Lewis. 2023. Site C Mon-12 Fish Stranding Monitoring Program Synthesis Review. Consultant's report prepared for BC Hydro by Ecofish Research Ltd., March 30, 2023.

Certification: *Certified - stamped version on file.*

Senior Reviewer:

Adam Lewis, M.Sc., R.P.Bio. No. 494

Fisheries Scientist/Principal

Technical Lead:

Adam Lewis, M.Sc., R.P.Bio. No. 494

Fisheries Scientist/Principal

Disclaimer:

This report was prepared by Ecofish Research Ltd. for the account of BC Hydro. The material in it reflects the best judgement of Ecofish Research Ltd. in light of the information available to it at the time of preparation. Any use which a third party makes of this report, or any reliance on or decisions to be made based on it, is the responsibility of such third parties. Ecofish Research Ltd. accepts no responsibility for damages, if any, suffered by any third party as a result of decisions made or actions, based on this report. This numbered report is a controlled document. Any reproductions of this report are uncontrolled and may not be the most recent revision.

EXECUTIVE SUMMARY

Fish stranding monitoring is required at BC Hydro's Site C Clean Energy Project (the Project) to quantify fish stranding on the Peace River, as outlined in the monitoring plan for the Site C Fish Stranding Monitoring Program (Mon-12; BC Hydro 2015b). Ecofish Research Ltd. (Ecofish) was retained by BC Hydro to provide technical oversight of field data collection conducted by Ecora Engineering & Resource Group Ltd. (Ecora) to ensure that data were being collected appropriately to address the primary Mon-12 fisheries management questions and hypotheses.

This report provides a synthesis review of the Mon-12 program background, management questions and hypotheses, study area, field and supporting methods, and data collected from Construction Years 3 to 6 (2017 to 2020). This report is preceded by annual data reports by Ecora submitted to BC Hydro following each year of baseline monitoring, which provide detailed field methods and raw data summaries specific to individual years of monitoring (Ecora 2018, 2019, 2020). The first year of monitoring in 2016 (i.e., Construction Year 2; Ecora 2017) was not included in this synthesis review due to differing methodology and scope.

This report describes and summarizes the analysis of the data collected by Ecora and Ecofish and provides:

1) Estimates of fish stranding and isolation rates (referred to as the magnitude of fish stranding and isolation by BC Hydro 2015b) within the Mon-12 study reaches of the Peace River for each trip in each year, based on fish stranding and isolation observations made during interstitial and pool sampling conducted within the reaches during each trip. Each trip was planned to monitor the effects of specific ramping events during, and directly after associated discharge reductions originating at the Peace Canyon Dam.
2) Analyses of how observed fish stranding and isolation within interstitial habitat and pools varied among study reaches, single- and multi-thread channel habitat, high and low-risk stranding habitat, and with shoreline slope and river hydrology.

The spatial sampling strategy of Mon-12 was modified in Construction Year 3 (2017) to follow the hierarchy of Reach > Channel Type > Mesohabitat > Microhabitat, as described in Nicholl and Lewis (2016) for modelled channel segments within each reach (i.e., future diversion headpond, Reach 1, Reach 2, and Reach 3).

Fish stranding and isolation rates within dewatered habitats were quantified through interstitial sampling, and the isolation rates of fish in pools were quantified through electrofishing sampling in pools. For each searched ramping event, the rates of fish stranding and isolation in interstitial habitats were calculated as the number of fish detected through interstitial searches per shoreline length searched at each site. For each searched ramping event, the rates of fish isolation within pools were calculated as the number of fish detected per area of electrofished isolated pools within each site.

The extent to which fish stranding and isolation varied among reaches, between single- and multi-thread channels, and low and high stranding risk mesohabitat, as well as over varying magnitudes and rates of discharge reduction, wetted histories of dewatered habitat, and shoreline slopes were evaluated through non-parametric tests. In line with Irvine *et al.* (2015), the relative importance, magnitude, and direction of effects of predictor variables on the probability of fish stranding and isolation were assessed through generalized linear mixed effects models, model selection via Akaike's information criterion corrected for small sample size (AICc), and model averaging.

Across all years and searched ramping events, the mean combined stranding and isolation rate was 5.62 (standard error (SE) 2.43) fish/100 m in the future diversion headpond, 1.18 (SE 0.4) fish/100 m in Reach 1, 0.38 (SE 0.21) fish/100 m in Reach 2, and 9.7 (SE 4.45) fish/100 m in Reach 3. Differences among reaches are likely largely due to variability in the timing of sampling in each reach relative to that of the associated ramping events and the differing magnitudes of ramping events associated with sampling events in each year (Ecora 2018, 2019, 2020). For example, Reach 3 was also only sampled in 2017, when the highest magnitude ramping events were searched, whereas the other reaches were also searched in 2018, 2019, and/or 2020 when, on average, searched ramping events were smaller. Differences among reaches also likely reflect the longitudinal variability in the distribution and abundance of large and small-bodied fish within the Peace River (Golder and Gazey 2018, Mainstream 2009 2010, 2012). Slimy Sculpins, followed by suckers, Longnose Dace, and Lake Chub made up the majority of isolated and stranded fish during all sampling events, and, with the exception of Slimy Sculpin, the vast majority (i.e., ~90%) of these fish were young-of-year or juveniles.

Overall, results indicated that the rates of stranding and isolation differed among study reaches and between single- and multi-thread channels in interstitial habitat but not pools, and that these variables were not significant predictors of the probability of stranding and/or isolation in either interstitial habitat or pools. Further, the probability of interstitial stranding and isolation increased with wetted history length and the magnitude of flow change measured at the nearest WSC hydrometric station, and mostly occurred in high-risk mesohabitat, where as the probability of fish isolation in pools increased with wetted history length, the magnitude of flow ramping rates, and to a lesser extent, flow changes and stage change rates measured at the nearest WSC hydrometric station. Top ranked mixed effects models explained up to 43% of the variance in the probability of stranding and isolation in interstitial habitat, but only up to 15 – 18% in pools.

These analyses provide estimates of fish stranding and isolation rates by reach per event and identify the relative effects of predictor variables explaining variance in these rates and the probability of ramping and isolation. Rates of fish stranding and isolation provided herein will be used as benchmarks to evaluate the influence of river diversion and Project operations on fish stranding and isolation, with due consideration of predictor variables and the frequency and characteristics of ramping events. Estimates of the magnitude of fish stranding and isolation provided herein apply only to those ramping events that were monitored during the study and are not intended to be the sole benchmarks for the assessment of changes in fish stranding and isolation.

Together with Ecora's annual data summary reports (Ecora 2017, 2018, 2019, 2020), this synthesis review satisfies the baseline monitoring requirements for Mon-12 up to Construction Year 6. Field data collection is on track to address the primary fisheries management questions and hypotheses with the next synthesis review scheduled for Construction Year 9 (2023).

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF MAPS

1. INTRODUCTION

BC Hydro developed the Site C Fisheries and Aquatic Habitat Monitoring and Follow-up Program (FAHMFP; BC Hydro 2015a) in accordance with Provincial Environmental Assessment Certificate Condition No. 7 and Federal Decision Statement Condition Nos. 8.4.3 and 8.4.4 for the Site C Clean Energy Project (the Project). The Site C Fish Stranding Monitoring Program (Mon-12), included as Appendix M of the FAHMFP, aims to quantify fish stranding and isolation along the Peace River during baseline conditions compared that under construction and operations phases of the Project in order to address the primary fisheries management questions and hypotheses (BC Hydro 2015b; Section 1.2). Monitoring is focused on a study reach of the Peace River which extends from the estimated upstream extent of the future diversion headpond upstream of the Project to the Many Islands area in Alberta approximately 139 km downstream. The study reach is broadly divided into two sections: the future diversion headpond upstream of the Project (18 km) and the Peace River downstream to the Many Islands area in Alberta (122 km). The downstream section of the Peace River is further divided into three reaches (Reaches 1 to 3) with breaks at the Pine River and Alces River confluences (Map 1).

Ecofish Research Ltd. (Ecofish) was retained by BC Hydro to provide technical oversight of field data collection conducted by Ecora Engineering & Resource Group Ltd. (Ecora) to ensure that data were being collected appropriately to address the primary fisheries management questions and hypotheses, and to conduct data analyses as described in the monitoring plan (BC Hydro 2015b). Management questions and hypotheses are also summarized in Section 1.2. This report is preceded by annual data reports by Ecora following each year of baseline monitoring, which provide detailed field methods and raw data summaries specific to individual years of monitoring (Ecora 2018, 2019, 2020). Data collected by Ecora in Construction Years 2 to 5 were augmented by additional monitoring conducted by Ecofish in Construction Year 6 (2020). Data from the first year of monitoring in 2016 (i.e., Construction Year 2; Ecora 2017) were not included in this synthesis review due to differing methodology and scope; however, 2016 data collection allowed the development of an effective sampling strategy for subsequent years (Nicholl and Lewis 2016).

This report provides a synthesis review of the Mon-12 program background, management questions and hypotheses, study area, field and supporting methods, and data collected from Construction Years 3 to 6 (2017 to 2020). The analyses described herein provide estimates of the rate of interstitial fish stranding and isolation within the study reaches of the Peace River for each monitoring trip (i.e., periods of 2-3 consecutive days when stranding searches were conducted following a single or two staged ramping events) in each year and an investigation of how observed fish stranding and isolation differ among reaches, channel type (single- or multi-thread), mesohabitat (low or high stranding risk), microhabitat (shoreline slope), and with river hydrology metrics characterizing flow reduction events (i.e., ramping events), together referred to as predictor variables herein.

1.1. Background

Fish stranding and isolation generally occurs when fish become separated from their primary waterbody and may result in injury or mortality (Lewis *et al.* 2013). Changes in river flow and water level may dewater fish habitat, which may result in stranding fish or isolating them from their primary waterbody (Lewis *et al.* 2013). Fish are considered stranded when they are found dead out of water or are at imminent risk of death from the dewatering of wetted habitats, including within the interstitial spaces of coarse substrates (Golder 2014a). Stranding may occur through rapid declines in stage that a fish is unable to avoid, through their reluctance to leave the cover of coarse substrates, or through the gradual dewatering of isolated depressions or channels as described below (Nicholl and Lewis 2016).

Isolation occurs when fish become trapped in wetted areas of habitat that have become disconnected from a main waterbody. An isolated fish may not be in imminent risk of mortality but may be at an elevated risk from predation, deteriorating water conditions (i.e., increased water temperature or freezing, or reduced dissolved oxygen) and may become stranded if stage continues to decrease through subsurface outflow (Nicholl and Lewis 2016). The relative risk to isolated fish usually depends on physical characteristics of an isolated pool (i.e., size, depth, substrates, and presence of cover), weather (which can affect evaporation, temperature, and dissolved oxygen), and the length of time before the pool becomes reconnected to a main waterbody (Lewis *et al.* 2013). While all fish may be at risk, depending on the magnitude of flow reductions, young-of-year (YOY), juvenile, and small-bodied fish are often at a higher risk of stranding or isolation due to their typical association with shallow, near-shore habitats and reduced swimming capacity (i.e., Triton 2009, Lewis *et al.* 2013).

Fish stranding and isolation may occur during natural water level fluctuations but may be exacerbated by hydroelectric activities that increase the relative frequency, rate, and magnitude of stage and flow reductions (Nagrodski *et al.* 2012; Irvine *et al.* 2015). The magnitude of stranding and isolation is typically closely related to the magnitude and rate of flow reductions (hereafter referred to as ramping events; Irvine *et al.* 2009). The risk of fish stranding and isolation is also influenced by a number of other factors including the duration of time habitat is wetted prior to a ramping event (i.e., wetted history), the rate at which a reduction occurs (i.e., ramping rate), and the physical characteristics of habitat dewatered by an event, including shoreline slope, substrates composition and cover, and the presence of depressions or other areas that may collect water during stage reductions (Golder and Poisson 2010a, 2010b).

During the five years of baseline monitoring synthesized herein, flow regimes within the study area of the Peace River have been largely influenced by operation of the Peace Canyon Dam (PCN) approximately 85 km upstream of the Project near Hudson's Hope, BC. Thus, monitoring has been focused on the ramping events initiated at PCN, the effects of which travel downstream throughout the study area.

1.2. Management Questions and Hypotheses

The primary objective of Mon-12 is to collect data that address four primary fisheries management questions (BC Hydro 2015b):

Q1. What is the magnitude of fish stranding in the diversion headpond relative to baseline conditions?

Q2. Which species and life stages of fish are most affected by stranding in the diversion headpond relative to baseline conditions?

Q3. During Project operation, what is the magnitude of fish stranding by species and life stage in the Peace River downstream of the Project relative to baseline conditions?

Q4. Do mitigation strategies (i.e., fish salvage and habitat enhancement) reduce fish stranding rates relative to baseline conditions?

The management questions will be addressed by testing the following hypotheses:

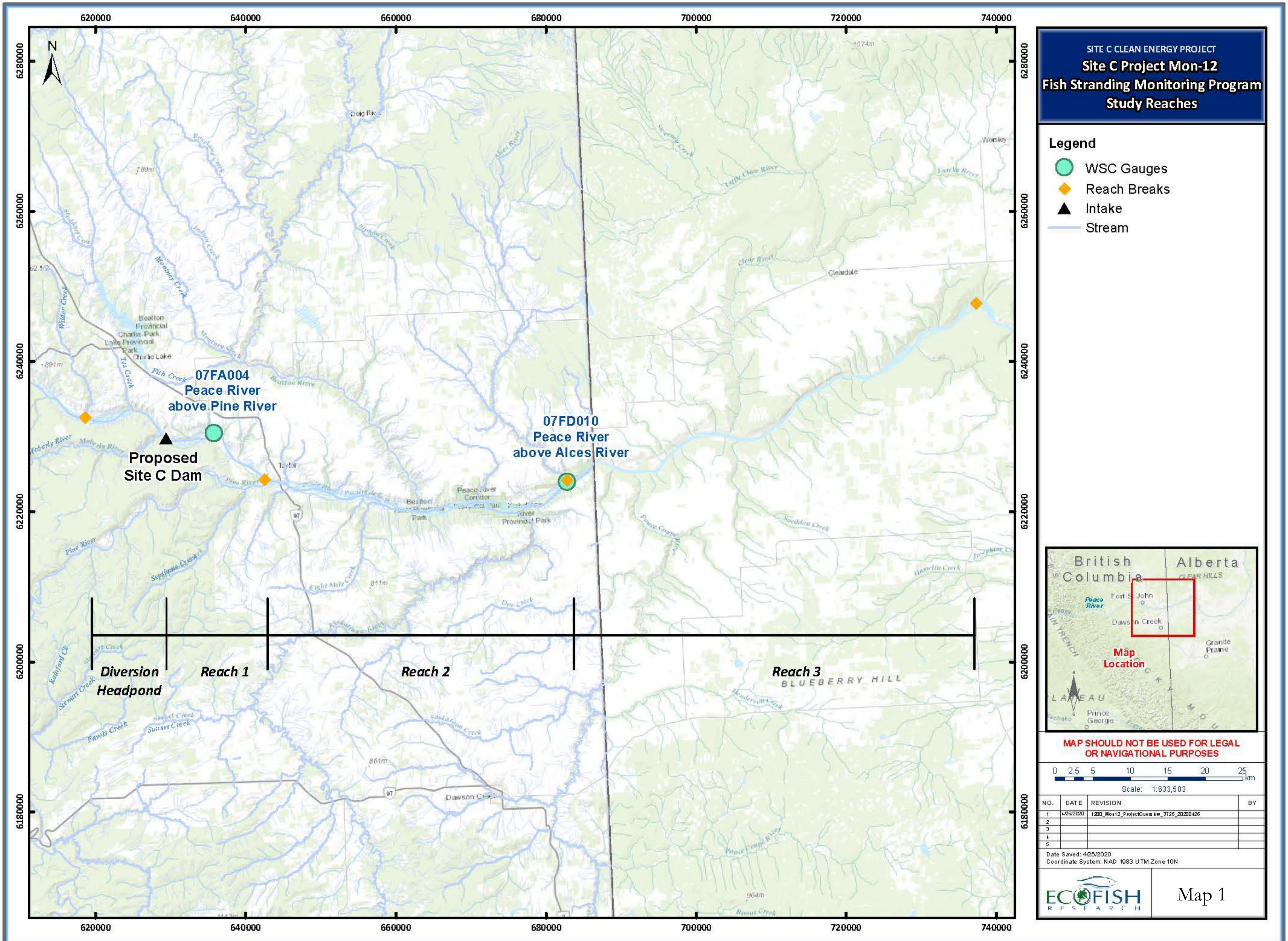
H1. During Project construction, fish stranding in the diversion headpond increases relative to baseline conditions.

H2. During Project operation, fish stranding in the Peace River between the Project and the Pine River confluence increases relative to baseline conditions.

H3. During Project operation, fish stranding in the Peace River between the Pine River confluence and the Many Islands area in Alberta is similar to baseline conditions.

H4. Proposed mitigation measures in the diversion headpond during the river diversion phase of Project construction and side channel enhancement and contouring in the Peace River downstream of the Project during operations are effective in reducing fish stranding rates.

This synthesis review summarizes relevant baseline data that will be used to address these management questions. Rates of fish stranding and isolation provided herein will be used as benchmarks to evaluate the influence of river diversion and Project operations on fish stranding and isolation, with due consideration of predictor variables and the frequency of ramping events.



Path: M:\Projects-Active\1200_SiteC\MXD\Overview\1200_Mon12_ProjectOverview_3726_20200426.mxd

2. METHODS

Methods for monitoring targeted ramping events under the Mon-12 program were developed following those used by Golder Associates Ltd. (Golder) to monitor other rivers influenced by ramping from BC Hydro facilities (e.g., the Columbia River (Golder 2011, 2014a) and Duncan River (Golder 2014b)) as well as provincial standards for ramping compliance monitoring of run-of-river hydroelectric facilities (Lewis *et al.* 2011). Detailed descriptions of the study area, delineation of Peace River shorelines, characterization of ramping events, site selection, field monitoring methods, and data analyses are presented in the following sections.

2.1. Study Area

The Mon-12 study area is comprised of approximately 139 km of the Peace River, from the Wilder Creek confluence, downstream to the Many Islands area in Alberta (Map 2). As defined by the Mon-12 monitoring plan (BC Hydro 2015b), this study area is split into two general sections:

1) The Site C future diversion headpond, extending approximately 18 km, from the Wilder Creek confluence downstream to the Project dam site; and
2) The Peace River downstream of the Project, extending approximately 121 km, from the Project dam site downstream to the Many Islands area in Alberta, which is further divided into three reaches:
   - Reach 1 – from the Project dam site downstream to the Pine River confluence (16 km);
   - Reach 2 – from the Pine River confluence downstream to the Alces River confluence (42 km); and
   - Reach 3 – from the Alces River confluence, downstream to the Many Islands area (63 km).

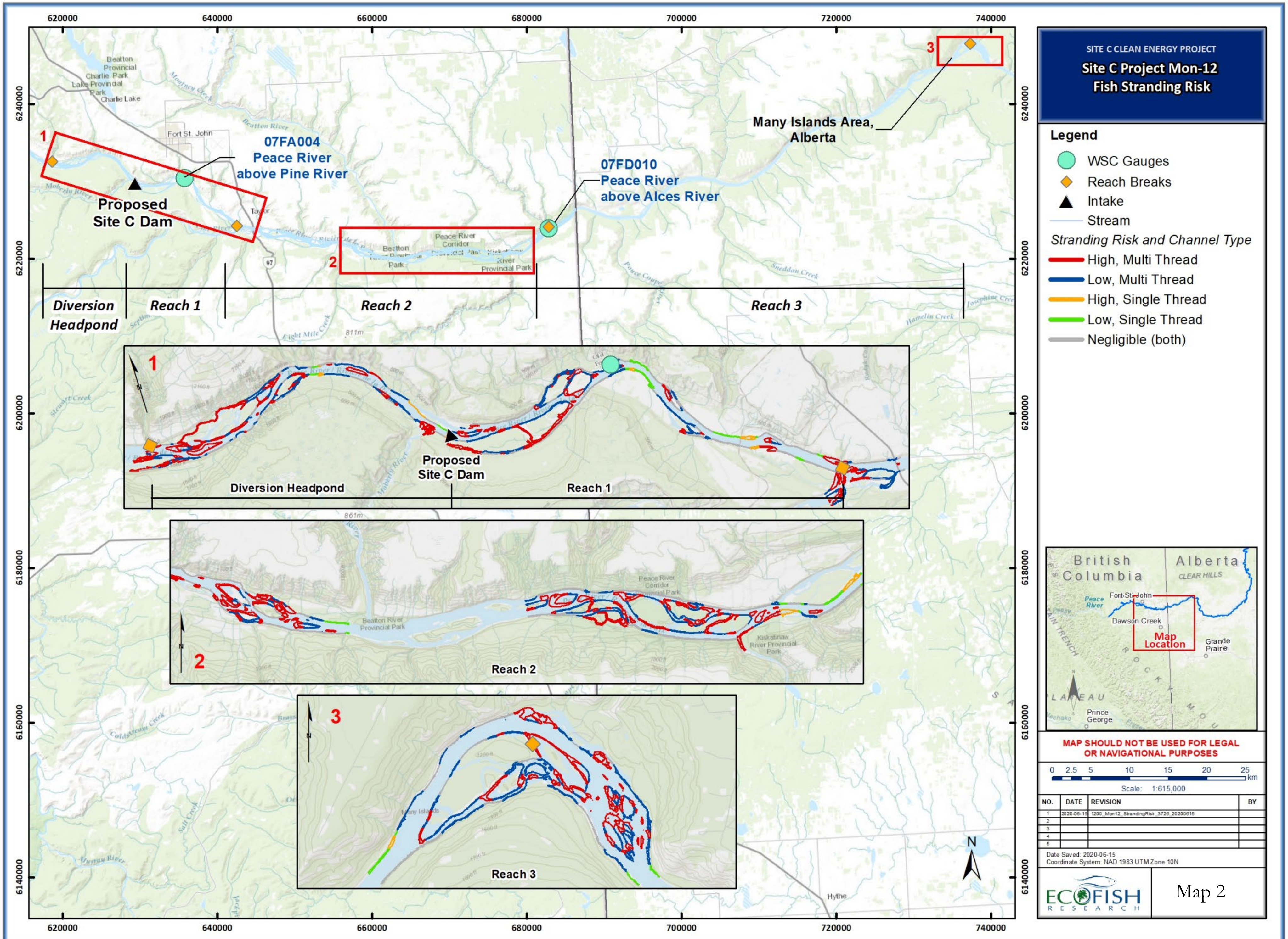
Baseline monitoring was concentrated within sections of these reaches where the majority of stranding habitat was expected to be present that were delineated by channel type and mesohabitat through spatial surveys and modelling, as described below in Section 2.2. These sections included all of the future diversion headpond and Reach 1 in 2017. In subsequent baseline years (2018 – 2020) these sections excluded the lower, approximately 3 km of the future diversion headpond and upper, approximately 5 km, of Reach 1 where active construction to recontour headpond shoreline, build dam infrastructure, and enhance Reach 1 side channels limited access and negated use of spatial data derived prior to instream Project construction (Ecora 2019). Similarly Reach 2 included two delineated areas, an upper, approximately 18 km section from Taylor Bridge, downstream to near the Beatton River confluence, and a lower, approximately 13 km section, from 5 km downstream of the Beatton River confluence, to approximately 4 km downstream of Raspberry Island in the Peace River Corridor Provincial Park. The monitored area in Reach 3 encompassed a delineated area roughly 8 km in length around the Many Islands area, at the downstream end of the reach.

2.2. Channel Delineation by Mesohabitat Type

The spatial sampling strategy of the Mon-12 study was modified in Construction Year 3 (2017) and was adopted in Construction Years 4 (2018), 5 (2019), and 6 (2020) to follow the hierarchy of Reach > Channel Type > Mesohabitat > Microhabitat, as described in Nicholl and Lewis (2016). Existing spatial data were used to delineate shorelines which were categorized to the mesohabitat level based on desktop review of available data. First, the study area was delineated into the four reaches as defined in Section 2.1.

Second, under the rationale that stranding risk is elevated in multi-thread channels due to increased habitat complexity, each reach was delineated into single and multi-thread channel segments based on the side channel inventory and mapping conducted by Mainstream (2013) for the future diversion headpond, and Northwest Hydraulic Consultants (NHC) (NHC 2012, 2013) for the downstream reaches. Where Mainstream or NHC had identified one or more side channels, all shorelines of the mainstem of the Peace River and associated side channels were delineated as multi-thread channel segments between the channel forks and confluences. Between these segments where no side channels had been identified, the Peace River was considered a single-thread channel segment.

Finally, discrete sections of shoreline were further delineated into mesohabitat types corresponding to stranding risk categories (high-risk, low-risk, and negligible risk) based on a review of spatial slope data derived from a digital elevation model (DEM), and river shorelines delineated from a River2D (Steffler and Blackburn 2002) model provided by BC Hydro and Fish Habitat Assessment Procedure (FHAP) data collected by Mainstream (2013). The River2D model provided minimum and maximum wetted shoreline margins within modelled sections of the future diversion headpond and the three downstream reaches. Where River2D data were not available, minimum and maximum wetted shoreline margins were delineated based on FHAP polygons. The DEM was generated using Blue Kenue™ software (NRC 2017) from which a slope layer with a 1 m grid cell size was created. The slope layer was classified into three stranding risk categories based on % gradient: high-risk (≤ 5%), low-risk (6 – 20%), and negligible risk (>20%) consistent with previous studies of fish stranding (e.g., Bell *et al.* 2008, Golder 2017), the slopes of sites established by Ecora, and associated stranding observations in the first year of monitoring in 2016 (Ecora 2017) and overlain with the River2D model and FHAP derived minimum and maximum wetted shoreline layers. These spatial data were then reviewed along with orthophotos by a fisheries biologist experienced in fish stranding studies to delineate all shorelines within the study reaches where model data were available. Shorelines were delineated into ≥ 100 m long mesohabitat sections composed of similar habitat units characterized as high, low, or negligible stranding risk based on the dominant slope categories as defined above (Map 2). The stranding risk classification at individual sites were then confirmed with clinometer measurements of shoreline slope and assessments of substrate and habitat structure in the field (Ecora 2020).


SITE C CLEAN ENERGY PROJECT
Site C Project Mon-12
Fish Stranding Risk
Legend
WSC Gauges
Reach Breaks
Intake
Stream
Stranding Risk and Channel Type
High, Multi Thread
Low, Multi Thread
High, Single Thread
Low, Single Thread
Negligible (both)
07FA004
Peace River
above Pine River
07FD010
Peace River
above Alces River
Many Islands Area,
Alberta
Proposed
Site C Dam
Diversion
Headpond
Reach 1
Reach 2
Reach 3
Diversion Headpond
Reach 1
Reach 2
Reach 3
British
Columbia
Alberta
Map
Location
MAP SHOULD NOT BE USED FOR LEGAL
OR NAVIGATIONAL PURPOSES
Scale: 1:615,000
NO. DATE REVISION BY
1 2020-06-15 1200_Mon12_StrandingRisk_3726_20200615
Date Saved: 2020-06-15
Coordinate System: NAD 1983 UTM Zone 10N
ECOFISH RESEARCH
Map 2

2.3. Hydrology

Searched ramping events resulted from changes in flow releases from PCN that caused down-ramping events in the study reaches downstream. Five-minute continuous discharge (m^3/s) and stage (m) data associated with searched ramping events were obtained from the Water Survey of Canada (WSC) hydrometric stations at the Peace River above Pine River (07FA004; PAP), representing conditions in the future diversion headpond and Reach 1, and at the Peace River above Alces River (07FD010; PAA), representing conditions in Reaches 2 and 3. WSC station data were retrieved approximately three to five months after the completion of annual Mon-12 monitoring in each year (i.e., January to March, 2018, 2019, and 2020) from the Environment Canada historical hydrometric data online search portal at (https://wateroffice.ec.gc.ca/search/historical_e.html). As noted within the search portal, these data are provisional estimates, and neither the WSC nor Ecofish conducted extensive quality assurance or data cleaning of these datasets prior to characterizing searched ramping events.

The start of searched ramping events was defined as the time of the maximum stage preceding the first stage decline following the beginning of flow reductions as measured at PCN. The end time was defined as the minimum stage as measured at each WSC station during an event. Hydrology metrics calculated from these data as measured at each WSC station for each searched event included total flow change (m^3/s), derived from subtracting the minimum flow from the maximum flow for a given event, flow ramping rate (m^3/s per hr) and stage change rate (cm/hr), calculated as the maximum change in flow and stage in one hour over the course of an event, respectively, and wetted history (days). These hydrology metrics were provided as a general characterization of each monitored event as site-specific hydrometric data collection at interstitial and pool sampling sites is not currently considered under the Mon-12 scope. Flow ramping rate and stage change rate were calculated by:

1) Calculating the maximum flow or stage observed over the past hour for each data point i as:

$$hmax(t_i) = \max\big(h(t_{i-k}), \dots, h(t_{i-1})\big)$$

where h is flow or stage, k is the number of data points recorded per hour, and t is time, and

2) Calculate the maximum flow or stage decrease over the past hour relative to time t_i, $\Delta hmax(t_i)$, as:

$$\Delta hmax(t_i) = h(t_i) - hmax(t_i)$$

Wetted history was calculated for every data point of a ramping event as the time period in days since stage was last less than or equal to the stage measured at a gauge at a given 15-minute interval. The 90^{th} percentiles of these wetted histories (i.e., the duration that 90% of the habitat dewatered by the event had been wetted) over the course of individual ramping events were used to characterize the overall wetted history of dewatered habitat for a given event in the plots and models described below.

2.4. Field Data

Fish stranding monitoring following targeted ramping events was conducted by Ecora in Construction Years 3 to 5 (2017 to 2019) and by Ecofish in Construction Year 6 (2020). Monitoring trips were scheduled in coordination with BC Hydro Operations Planning Engineers at PCN to coincide with forecasted ramping events during the summer and early fall (typically July to October) when flows were higher and more variable, and when fish were more likely to be present in shallows most effected by ramping (e.g., Mainstream 2009, 2010, 2012, Ecora 2018). Events of varying magnitudes and wetted histories were targeted to be representative of the range of magnitudes observed during Project construction. In 2017 and 2020, individual targeted ramping events were monitored over two subsequent days, beginning upstream in the future diversion headpond, and following the ramping event as it travelled downstream. In 2018 and 2019, monitoring trips were typically scheduled to coincide with staged events, whereby two down-ramping events would occur overnight, 19 to 31 hours apart, with each searched within the targeted reaches on the following day. Ecora conducted five monitoring trips in each year for a total of ten days of stranding monitoring per year and corresponding to five, nine, and 10 ramping events in 2017, 2018, and 2019, respectively, while Ecofish conducted monitoring following three ramping events, for a total of seven days of monitoring in 2020 (three days of monitoring were conducted after the first searched ramping event in 2020).

2.4.1. Site Selection

Targeted high-risk monitoring sites were initially selected during field reconnaissance in 2016 based on habitat characteristics known to increase the risk of fish stranding and/or isolation (Ecora 2017). Specifically, Ecora focused monitoring effort on habitats where shoreline gradients were < 4%, characterized by large relative areas of potentially dewatered substrate (i.e., > 500 m²), prevalent cover (i.e., large relative substrates such as cobble and boulder, low substrate embeddedness, and/or woody debris), and natural stream habitats most likely to strand or isolate fish as described by Lewis *et al.* (2011):

- Where the river cross-section has a relatively flat slope with large substrate that could strand fish, or finer substrate with depressions that could trap fish;
- Cobble and gravel bars, with roughness characteristics that create refuges that juvenile fish are known to prefer and may be reluctant to leave during a ramp down event; and
- Side channels or shallow pools along stream margins that are known to be preferred by rearing juvenile fish.

In subsequent baseline years, sites where stranding had been detected previously were repeatedly sampled, and additional targeted sites were established based on the above criteria augmented with linear mapping of shorelines as single- or multi-thread channel, and high, low, or negligible risk (Section 2.2) to ensure that areas of high stranding risk habitat representative of the overall shoreline characteristics of each study reach were monitored. Ecora characterized targeted sites as large polygons of shoreline composed of similar habitat. These polygons were repeatedly searched following multiple events when possible and were augmented with newly established sites when river stage and discharge made conditions at existing sites inappropriate for conducting searches. To determine whether targeted sites were representative of overall habitat and fish stranding within the reaches, 11% of searches were conducted at waypoints randomly selected through GIS mapping tools within each of the stratifications described in Section 2.2 except for negligible-risk mesohabitats, which were deemed unsuitable to monitor due to a lack of any appreciable stranding habitat (as confirmed by field observations). A list of these random waypoints was compiled and ordered using a random number generator in R (R Development Core Team 2020) and visited sequentially over the course of baseline monitoring. Typically, randomly selected sites were only searched once, with new randomly selected sites generated each year. However, in some cases where randomly selected sites were determined to be representative of high-risk stranding habitat, they were added to the list of targeted sites for a given stratification, and revisited following subsequent ramping events when conditions were appropriate based on the professional judgement of the monitoring crews. Stranding searches (as described in Section 2.4.2 below) were conducted over subsections of appropriate stranding habitat within targeted sites deemed to be appropriate based on an assessment of river and site conditions at the time of searches. Therefore, similar sections of habitat were typically searched in each site under similar conditions but varied over time due to differing river stage and discharge conditions among searched ramping events. Targeted sites searched following a given ramping event were selected based on whether appropriate stranding habitat had been dewatered within a site as determined by flow and stage conditions of, and following an event, and verification of conditions at specific sites in the field.

2.4.2. Field Monitoring

Table 1 presents a summary of sampling efforts from 2017 to 2020. Field methods included both interstitial and pool sampling as described below and were consistently employed by Ecora and Ecofish over the course of baseline monitoring, with year-specific variations and details described in annual Mon-12 monitoring reports (Ecora 2018, 2019, 2020).

2.4.2.1. Interstitial Sampling

Interstitial sampling was conducted within dewatered habitat at each selected monitoring site determined to be appropriate for searches following a specific ramping event (i.e., an interstitial sampling event). In 2017, the interstitial sampling methods initially involved searching 1 m² quadrats placed at regular intervals along 100 m transects within a portion of each site following methods similar to those used to monitor ramping events on the Duncan River (Golder 2014b). This method was intended to reduce searcher bias and increase accuracy of searches, however, it resulted in low detection success during the first two trips due to the non-random occurrence of fish stranding and isolation as described by Ecora (2018) and further discussed in Section 2.5. Consequently, interstitial sampling methods were revised for the remainder of 2017 and subsequent years of baseline monitoring (2018 to 2020) to adopt a combination of broad-based (visual overview) searches and hot-spot searches (targeted excavation of substrate) as described below.

Broad-based searches were conducted along a length of transect over a portion of an overall site where dewatered stranding habitat was present under conditions at the time of sampling. A transect length of 100 m was targeted for broad-based searches, although lengths varied from 15 to 450 m in length (median = 100 m) depending on the relative dimensions of dewatered stranding habitat at a given site. The width of broad-based searches varied depending on the width of dewatered stranding habitat at a given site, ranging from 1 to 100 m (median = 10 m).

During each broad-based search, crews searched the transect in an upstream direction covering the shoreline from the wetted edge up to the estimated extent that the substrate was wetted prior initial stage declines associated with the ramping event being searched. During the broad-based search, areas of highest stranding risk were identified for hotspot searches. The length, width, number of searchers, effort per searcher (minutes), start and end time, and weather conditions were recorded, and representative photographs and waypoints of the upstream and downstream extent were taken for each broad-based search. If newly establishing a site or conducting searches in a new section of stranding habitat within an established site, % substrate composition, cover (vegetation or other), and shoreline slope were also recorded. Where shoreline slope was not recorded in the field, values were extracted from the DEM described in Section 2.2 at the location of the site waypoint on a map. Presence of bird activity or other scavenging within sites were recorded, as scavenging and predation may result in the removal of isolated or stranded fish within a site prior to their detection during searches.

Once a broad-based search was complete, five hotspot transects were selected to characterize the highest risk stranding habitat within the broad-based area based on characteristics described by Lewis *et al.* (2011) (e.g., shallow depressions, small pools of residual water, and/or areas with abundant coarse substrate or other cover) and professional judgement. At each hotspot transect, measuring tapes were used to delineate the dimensions of the area to be searched. An area of 20 m² was targeted for each hotspot transect to sample a combined area of approximately 100 m² at each site. Within each hotspot transect, crews worked close to the ground (i.e., on hands and knees), and overturned all large substrate and other cover to search for fish. The length, width, number of searchers, search effort per searcher (minutes), and representative photographs were recorded for each hotspot search. All fish that were observed or captured during interstitial sampling were processed and recorded as described in Section 2.4.2.3.

2.4.2.2. Pool Sampling

Pool sampling was conducted by two to three person crews using backpack electrofisher units (Smith-Root LR-24) within pools isolated from the Peace River mainstem by the searched ramping events where present in selected monitoring sites using the following procedure.

- Upon arrival at each site, reconnaissance of the area was conducted to determine the presence and suitability of isolated pools within the site. To be suitable for sampling, pools needed to be ≥ 1 m², have a maximum depth of ≥ 5 cm, and be disconnected from the mainstem (i.e., isolated), with no evidence of consistent surface or subsurface flow.

- Up to three pools were sampled per site. Where more than three pools were present at a given site, the three pools with the highest likelihood of containing fish were selected based on habitat suitability, size, substrate composition, and cover.
- Sampled pools were first searched visually to verify fish presence and then sampled through 2 to 3 electrofishing passes to determine fish abundance and salvage isolated fish where possible. Electrofishing voltage, frequency, and duty cycle settings were set using the LR-24 quick setup based on water conditions, and manually adjusted as necessary to optimize capture success.
- For each sampled pool, the wetted length, width, maximum depth, and where possible, estimated maximum pre-event length, width, and depth (referred to as bankfull measurements) were recorded along with water temperature, visibility, substrate composition, presence of cover, electrofishing effort (seconds), electrofisher settings, and representative photographs, and a waypoint of pool locations were recorded using an iPad or handheld GPS. In 2018 through 2020, all additional suitable pools to those sampled within a site were enumerated, visually inspected for fish presence, and estimated wetted and bankfull length, width, and maximum depth were recorded.

Pools were selected for sampling each time a site was visited based on conditions and suitability of individual pools present at the time. Accordingly, over the course of baseline monitoring, some pools were repeatedly sampled whereas others were only sampled once. This is in contrast to pool sampling conducted on the Duncan River (i.e., Golder 2018), where pools at each site were initially demarcated and a new subset of which were sampled to determine fish presence during each subsequent site visit.

2.4.2.3. Fish Sampling

All fish observed or captured during interstitial or pool sampling were recorded, as well as those observed incidentally outside of specifically surveyed areas. All live fish were placed in buckets filled with river water until processing and released to the mainstem or connected side channel habitat adjacent to where they were originally captured, once they had recovered. Each fish was identified to species, except when poor relative condition (i.e., desiccation or decay), or when a fish was briefly observed but not captured, and species could not be verified. In these cases, general species group (e.g., sucker, sculpin, cyprinid) was recorded if possible. The fork length of each fish (or total length for sculpins) was recorded to the nearest millimeter using a measuring board or fish viewer (or estimated for fish that were not captured), and the relative life stage YOY, juvenile, or adult) was determined based on general length-at-age keys derived from reference material (McPhail 2007, McPhail and Carveth 1993, Mainstream 2011; Table 2). Fish were classified as stranded if they were completely out of the water at the time of observation, and isolated if they were immersed in water. Fish condition (live or dead) and the cause of mortality (i.e., natural, ramping event induced, or from sampling/processing) were recorded. Representative photographs of fish at each site were taken, and in 2020, voucher specimens of mortalities were retained for verification of uncertain species identification in the field.

Table 1. Summary of all fish stranding searches conducted from 2017 to 2020. Interstitial searches tallies the discrete sites where interstitial searches (broad-based and/or hotspots) were conducted, whereas pool searches are tallied by discrete sites and individual pools sampled on a given date.

Year	Ramping Event	Sampling Day	Date	Interstitial Searches	Pool Sampling		Total Sampling Events
					Sites	# of Pools	
2017	1	Day 1	29-Jul	7	1	1	8
		Day 2	30-Jul	5	5	11	10
	2	Day 3	12-Aug	10	10	16	20
		Day 4	13-Aug	7	7	14	14
	3	Day 5	26-Aug	9	9	20	18
		Day 6	27-Aug	5	6	15	11
	4	Day 7	9-Sep	9	7	18	16
		Day 8	10-Sep	8	4	10	12
	5	Day 9	23-Sep	10	9	23	19
		Day 10	24-Sep	7	5	12	12
			Total	**77**	**63**	**140**	**140**
2018	1	Day 1	11-Aug	10	6	24	16
		Day 2	12-Aug	10	10	35	20
	2	Day 3	18-Aug	7	3	15	10
		Day 4	19-Aug	6	4	10	10
	3	Day 5	8-Sep	13	—[1]	—[1]	13
		Day 6	9-Sep	10	7	18	17
	4	Day 7	15-Sep	15	7	30	22
		Day 8	16-Sep	15	2	4	17
	5	Day 9	2-Oct	17	9	38	26
		Day 10	3-Oct	18	9	31	27
			Total	**121**	**57**	**205**	**178**
2019	1	Day 1	27-Jul	12	7	23	19
		Day 2	28-Jul	11	7	20	18
	2	Day 3	10-Aug	12	5	20	17
		Day 4	11-Aug	14	6	23	20
	3	Day 5	5-Sep	12	10	40	22
		Day 6	7-Sep	11	5	22	16
	4	Day 7	19-Sep	11	5	27	16
		Day 8	20-Sep	8	5	26	13
	5	Day 9	19-Oct	11	6	34	17
		Day 10	20-Oct	8	1	9	9
			Total	**110**	**57**	**244**	**167**
2020	1	Day 1	19-Aug	9	1	3	10
		Day 2	20-Aug	9	5	5	14
		Day 3	21-Aug	6	3	8	9
	2	Day 4	3-Sep	11	3	17	14
		Day 5	4-Sep	5	2	12	7
	3	Day 6	29-Sep	8	—[1]	—[1]	8
		Day 7	30-Sep	8	—[1]	—[1]	8
			Total	**56**	**14**	**45**	**70**

[1] Unable to conduct electrofishing due to heavy rain on September 8, 2018, and no pools were appropriate for sampling at sites visited on September 29 or 30th, 2020.

Table 2. List of fish species that were captured or observed during baseline Mon-12 monitoring including common and scientific names, and general length-at-age ranges for YOY, juvenile, and adults.

Group	Species		Min. Length-at-Age (mm)		
	Common Name	Scientific Name	YOY[1]	Juvenile	Adult
Sport fish	Arctic Grayling	*Thymallus arcticus*	< 130	130	300
	Burbot	*Lota lota*	< 80	80	400
	Kokanee	*Oncorhynchus nerka*	< 90	90	200
	Mountain Whitefish	*Prosopium williamsoni*	< 100	100	200
	Northern Pike	*Esox Lucius*	< 130	130	351
	Rainbow Trout	*Oncorhynchus mykiss*	< 150	150	250
	Yellow Perch	*Perca flavescens*	< 55	55	120
	Walleye	*Sander vitreus*	< 110	110	301
Suckers	Largescale Sucker	*Catostomus Macrocheilus*	< 50	50	300
	Longnose Sucker	*Catostomus Catostomus*	< 50	50	300
	White Sucker	*Catostomus commersonii*	< 50	50	300
Minnows	Flathead Chub	*Platygobio gracilis*	< 90	90	180
	Lake Chub	*Couesius plumbeus*	< 30	30	81
	Longnose Dace	*Rhinichthys cataractae*	< 30	30	61
	Northern Pikeminnow	*Ptychocheilus oregonensis*	< 60	60	180
	Redside Shiner	*Richardsonius balteatus*	< 50	50	65
	Trout-Perch	*Percopsis omiscomaycus*	< 30	30	80
Sculpins	Prickly Sculpin	*Cottus asper*	< 40	40	61
	Slimy Sculpin	*Cottus cognatus*	< 40	40	61
	Sculpin *spp.*	*Cottus spp.*	< 40	40	61

[1] YOY = young-of-year.

References: McPhail 2007, McPhail and Carveth 1993, Mainstream 2011.

2.5. Quantifying Fish Stranding and Isolation - Interstitial Sampling

Fish stranding and isolation rates were quantified through interstitial sampling (Table 1), as described in Section 2.4 and depicted in Map 2:

- 77 interstitial sampling events were conducted in 2017 between July 29 and September 24;
- 121 interstitial sampling events were conducted in 2018 between August 11 and October 3;
- 110 interstitial sampling events were conducted in 2019 between July 27 and October 20; and
- 56 interstitial sampling events were conducted in 2020 between August 19 and September 30.

2.5.1. Rates of Fish Stranding and Isolation and Predictors

The rates of fish stranding and isolation from interstitial sampling events were calculated as a linear density for each site through dividing the combined number of stranded or isolated fish observed during both broad-based and hotspot searches at a given site by the length of the broad-based transect searched. Combining broad-based and hotspot searches was justified given that stranding and isolation is not evenly distributed throughout a site, but rather concentrated in smaller areas of the highest-risk habitat (i.e., Lewis *et al.* 2013); this supports an assumption of interstitial sampling that any obvious fish stranding would be noted during broad-based searches, and that the majority of the highest risk habitat within the site would be searched thoroughly during hotspot searches, and thus most stranding and isolation within dewatered habitat would be detected. Because observed fish densities were often very low, these linear density estimates (i.e., fish/m) were multiplied by 100 to be expressed as fish/100 m in order to present more tangible numbers in figures and summary tables. While stranding and isolation rates can be reported in terms of area (i.e., fish/m^2), linear rates of stranding and isolation (i.e., fish/100 m) were used, as accurate estimates of dewatered area were not available at all sites following all ramping events. For the non-parametric tests and multi-variate analysis we calculated the "combined rate" (the rate of fish stranding and isolation combined) to quantify the combined effect and to minimize the number of zeros in the dataset.

As described in Section 2.4.2.1, observed stranding and isolation rates were considerably lower during interstitial searches following the first two targeted ramping events in 2017 compared to those observed during searches using revised broad-based and hotspot search methodology following the remaining three targeted ramping events in 2017 (Ecora 2018). Only two stranded and zero isolated fish were observed over the course of 29 interstitial sampling events following the first two ramping events. In contrast, stranded and/or isolated fish were observed during 19 of the 48 sampling events conducted following the remaining three searched events, with an average of 8.6 fish observed per interstitial sampling event (standard error (SE) 2.0) and between 30 and 47 fish observed following a given event, despite similar hydrology and sampling conditions to those recorded for and following the first two events. Based on field observations, these higher numbers of observed fish following the last three events in 2017 (i.e., a 96 – 100% increase from those observed following the first two events) were attributed to an increased success rate in detecting stranded and isolated fish using the adjusted search methods (Ecora 2018). Accordingly, these adjusted interstitial search methods were applied in 2018, 2019, and 2020. Given that the methods used following the first two events in 2017 differed from those used in all subsequent interstitial searches and appeared to result in comparatively lower success in detecting stranded and isolated fish, interstitial data associated with the first two searched events in 2017 were excluded from synthesis analyses described herein.

Stranding and isolation rates were plotted as a function of predictor variables (i.e., reach, channel type, stranding risk of mesohabitat, shoreline slope, and hydrology metrics) to investigate possible relationships as per BC Hydro (2015b) and Nicholl and Lewis (2016). Reaches included the future diversion headpond, Reach 1, Reach 2, and Reach 3 (BC Hydro 2015b, Map 1); channel type included single- and multi-thread channels (Nicholl and Lewis 2016); and stranding risk of mesohabitat was characterized as high-risk (targeted), high-risk (randomly selected), or low-risk. As discussed in Section 2.4.1, targeted sites were typically searched following multiple ramping events and selected based on the presence and quality of stranding habitat present under the conditions at the time of sampling whereas randomly selected sites were typically only searched once, but were reclassified to targeted high-risk and returned to multiple times where appropriate to augment targeted sites in some cases. Further, no sampling events were conducted along shorelines categorized as being negligible stranding risk mesohabitat. Randomly selected sites that were confirmed to exhibit high-risk stranding habitat were combined with targeted high-risk sites and those determined to be low-risk were combined with targeted low-risk sites for data summaries and analyses herein. Predictor variables also included shoreline slope (Section 2.2) and several hydrology metrics (total flow change, flow ramping rate, stage change rate, and 90th percentile of wetted histories for a given ramping event (Section 2.3)) as measured at the nearest WSC hydrometric station.

Kruskal-Wallis rank sum tests (e.g., Hollander and Wolfe 1973) run through the "stats" package in R were used to test whether fish stranding and/or isolation rates differed among reaches, channel types, and mesohabitats (based on mean ranks). Kendall rank correlation tests (e.g., Hipel and McLeod 1994) run through the "Kendall" package in R (McLeod 2011) were used to test whether there was evidence of relationships between stranding and/or isolation rates and continuous predictors (i.e., hydrology metrics and shoreline slope). Non-parametric tests were used in place of parametric tests (e.g., analysis of variance, t-tests, and linear regression) because the dataset was highly non-normal (skewness $\geq$ 12.1 and kurtosis $\geq$ 171.2), meaning that assumptions of homoscedasticity of residuals were not met (as determined from residual plots of initially run parametric tests), even with log-transformed data (e.g., Kloke and McKean 2014). All tests used a significance level of $\alpha = 0.05$. However, p-values from all non-parametric tests for a given response variable were compared to corrected α values based on a modified Bonferroni Trimmed Simes Test (e.g., Simes 1986) to correct for multiple tests of significance on the same dataset and thus avoid the risk of a type 1 error. Effect size of differences in groups of categorical predictors (e.g., reach, stranding risk, and channel type) was calculated using Epsilon-Squared which is a measure of the relative degree to which one group has data with higher ranks than another group with large differences being considered those > 0.26 (e.g., Cohen 1988). Effect size of continuous predictors were characterized as the slope coefficient from linear regression because Sen's Slope (typically used as a measure of effect size for Kendall rank correlation tests; Sen 1968) is not representative of effect size in continuous datasets where > 50% of response values are zero (i.e., the case for these datasets). Where p-values were below corrected α values, trend lines (± 95% CI) derived from linear regression were added to plots for continuous predictors. All analyses were completed in R (R Development Core Team 2020).

2.6. Quantifying Fish Isolation - Pool Sampling

Densities of isolated fish were quantified through pool sampling (Table 1), as described in Section 2.4:

- 140 pools were sampled over 63 pool sampling events conducted between July 29 and September 24, 2017;
- 205 pools were sampled over 57 pool sampling events conducted between August 11 and October 3, 2018;
- 244 pools were sampled over 57 pool sampling events conducted between July 27 and October 20, 2019; and
- 45 pools were sampled over 14 pool sampling events conducted between August 19 and September 30, 2020.

2.6.1. Fish Isolation Relationships

The weighted average density of fish in sampled isolated pools within each site was calculated as the total number of isolated fish caught through electrofishing within all sampled pools in that site on a given date divided by the combined area of the sampled pools. As for linear densities derived from interstitial sampling, areal densities of isolated fish in sampled pools were multiplied by 100 and expressed as number fish/100 m^2 in order to present more tangible density estimates in figures and summary tables.

Pool sampling methodology was consistent across Construction Years 3 to 6; therefore, all pool data were included in the summaries and analyses herein, including data collected following the first two targeted ramping events in 2017. Consistent with interstitial sampling (Section 2.5), relationships between densities of isolated fish in pools and predictor variables (as described in Section 2.5.1) were plotted to investigate possible correlations. As for interstitial data, Kruskal-Wallis rank sum tests were used to test whether densities of isolated fish in pools differed among reaches, channel types or low- and high-risk mesohabitat with effect size characterized by Epsilon-Squared. Likewise, Kendall rank correlation tests were used to test for relationships between isolated fish densities and continuous predictors with effect size characterized by slope coefficients from linear regression. All tests were considered significant at $\alpha = 0.05$ but compared to corrected α values based on a modified Bonferroni Trimmed Simes Test, with significant trends depicted through trend lines (± 95% confidence intervals (CI)) derived from linear regression added to corresponding plots. Again, all analyses were conducted in R.

2.7. Variables Affecting Probability of Fish Stranding and Isolation

2.7.1. Interstitial Stranding and Isolation

Effects of predictor variables on the combined rates of interstitial fish isolation and stranding were further investigated following a multivariate approach similar to that used by Irvine *et al.* (2015). This approach improves upon non-parametric tests described in Sections 2.5 and 2.6 by allowing the effects of variables to be assessed together and by accounting for variability among monitoring trips and years of sampling.

We used generalized linear mixed effects models with a binomial distribution and logistic link function (e.g., Collett 2003) and model selection via Akaike information criterion corrected for small sample size (AICc) to determine the relative effects of predictor variables (fixed effects) on the probability of a stranding/isolation event within interstitial habitat at a typical stranding site. This model type was chosen to account for both the fixed and random effects, and to account for the error distribution of the dataset. The logistic regression response was whether or not an event occurred, which was defined as any sampling event where ≥ 1 fish was observed stranded and/or isolated within interstitial habitat at a given site. A logistic response was chosen because the dataset contained many zeros (i.e., no fish observed), and was non-normally distributed, limiting the effectiveness of modeling the magnitude of stranding and isolation events, and because a similar model structure was used in a related previous study (Irvine *et al.* 2015). While Irvine *et al.* (2015) used varying definitions of stranding events (e.g., 1, 50, 200, or 1,000 fish) to investigate how fixed effects influenced the magnitude of stranding, we could not follow this methodology due to the exceptionally high number of zeros and low counts of observed fish in our dataset (e.g., zero fish were observed in 78% and < 10 fish were observed in 91% of interstitial sampling events). Fixed effects included predictors investigated through plots and non-parametric tests in Section 2.5.1 and pertaining to the specific management hypotheses of Mon-12 (Section 1.2), including study reach, channel type, stranding risk, magnitude and rate of flow change, stage change rate, and wetted history. One predictor, shoreline slope, was excluded from models because data were only available for 50% of sampling events and because this variable did not appear to influence fish isolation or stranding based on plots and non-parametric tests (Section 2.5.1).

Multi-collinearity among fixed and random effects were investigated through correlation coefficients and variance inflation factors (VIF) to inform appropriate model structure (e.g., Zuur *et al.* 2008). Fixed effects with VIF scores >5.0 were considered further. Ideally models would include a random effects structure of monitoring trip nested within year to account for pseudoreplication at both levels, however the dataset could not support such a complex random effects structure. The two levels were therefore accounted for by including a random effect of individual trips across all years of baseline monitoring. Models were fit using the package "lme4" in R (Bates *et al.* 2015).

We evaluated the relative support for these hypotheses using an all-model combinations approach (n = 48), restricting candidate models to include up to three fixed effects, and excluding models containing any combination of flow ramping rate (VIF = 4.8), flow change (VIF = 7.0), and stage change rate (VIF = 6.0) because of the high level of collinearity among these three fixed effects

(Pearson correlation coefficients = 0.85 – 0.89). In order to compare among all parameters and interpret the fixed effects, we standardized continuous predictor variables by subtracting global means from each value (centering) and dividing by 2 standard deviations (scaling; Gelman 2008).

Model performance and uncertainty were assessed using Akaike Information Criteria corrected for small sample size (AICc) which ranks models based on the principle of parsimony (Anderson 2008). The lower the AICc score for a given model, the better the trade-off between complexity and optimal fit for that model. Model fits were assessed through review of residual plots. We used the "MuMIn" package in R (Barton 2012) to compete models based on ΔAICc values and AICc weights (w_i). Because there was often little distinction among high-ranking models based on these values, we also calculated multi-model averaged parameter estimates and relative variable importance (the sum of AICc weights from all models containing the variable of interest) for fixed effects from models that made up 95% of the cumulative model weights (Grueber *et al.* 2011). These averaged values were then used to assess the relative effect size (i.e., whether a predictor had a positive or negative effect on the response that was consistent among models and how large that effect was compared to other predictors) and importance of fixed effects on the probability of a fish isolation and/or stranding event. We also calculated marginal and conditional pseudo-R^2 as measures of goodness-of-fit and explanatory power of top models ($\Delta AICc \leq 3$; derived from regressions of the observed data versus fitted values, e.g., Piñeiro *et al.* 2008, Nakagawa *et al.* 2017). All statistical analyses were conducted in R (R Development Core Team 2020).

2.7.2. Fish Isolation within Pools

The same model structures, model selection, and model averaging methods described in Section 2.7.1 were also used to evaluate the relative effect size and importance of fixed effects on the probability of isolation events within pools at a typical stranding site with 22 candidate models competed based on ΔAICc and AICc weights. The exception was that stranding risk was excluded from models because less than 4% of pool sampling events were conducted in low-risk mesohabitat.

3. RESULTS

3.1. Channel Delineation by Reach and Mesohabitat

Within the modelled areas of the study reach (i.e., the sections of the Peace River where River2D, FHAP, and DEM spatial data were available) a total of 618,118 m of shoreline was delineated, of which 7% was single-thread and 93% was multi-thread channel (Table 3). Of the single-thread channel shoreline, mesohabitat stranding risk was classified as 18% high-risk, 41% low-risk, and 41% negligible risk. Of the multi-thread channel shoreline, 42% was classified as high-risk, 36% as low-risk, and 22% as negligible risk.

Table 3. Summary of the total estimated shoreline length by reach, channel type and mesohabitat (stranding risk category) within modelled areas of the study reach as per delineation via visual assessment of slope spatial data and orthophotos described in Section 2.2.

Reach	Channel Type	Stranding Risk Category	Modelled Shoreline Length (m)[1]	Percent of Channel Type in Reach
Future Diversion Headpond	Single	High	999	34%
		Low	1,260	43%
		Negligible	680	23%
	Multi	High	82,195	58%
		Low	43,548	31%
		Negligible	17,027	12%
Total			**145,708**	
Reach 1	Single	High	2,790	13%
		Low	8,698	41%
		Negligible	9,852	46%
	Multi	High	36,011	29%
		Low	48,517	40%
		Negligible	38,163	31%
Total			**144,033**	
Reach 2	Single	High	3,952	25%
		Low	5,742	36%
		Negligible	6,107	39%
	Multi	High	94,741	42%
		Low	72,426	32%
		Negligible	56,296	25%
Total			**239,264**	
Reach 3	Single	High	562	10%
		Low	2,835	52%
		Negligible	2,025	37%
	Multi	High	30,308	36%
		Low	41,731	50%
		Negligible	11,652	14%
Total			**89,114**	
Grand Total			***618,118***	

[1] Based on pre-channel contouring shoreline lengths in the lower diversion headpond and upper Reach 1.

3.2. Hydrology During Searched Events

Flows measured at the Peace River above Pine River (PAP) and Peace River above Alces River (PAA) WSC hydrometric stations in each monitoring year are presented in Figure 1 and Figure 2, respectively, and corresponding hydrology metrics derived from flow and stage data recorded at the two stations for each searched ramping event in each year of monitoring are summarized in Table 4 and Table 5. On average, flow change, ramping rate, stage change rate, and wetted history were greater for searched events in 2017 and 2018 than in 2019 and 2020. Flow change averaged approximately -974 m^3/s (i.e., flow reduction) in 2017 for all searched events as measured at both hydrometric stations and averaged -1,008 m^3/s for all searched events in 2018 as measured at the PAP hydrometric station(only the diversion headpond and Reach 1 were monitored in 2018). In 2019, flow changes averaged -453 m^3/s for all searched events as measured at both hydrometric stations. Similarly, ramping rates averaged -162 $m^3/s/hr$ and -153 $m^3/s/hr$ for all searched events in both 2017 and 2018, respectively, compared to averages of approximately -82 $m^3/s/hr$ and -116 $m^3/s/hr$ for searched events in 2019 and 2020, respectively. Stage change rates of searched events in 2017 and 2018 averaged -23 cm/hr and -25 cm/hr, respectively, compared to an average of -14 cm/hr for those searched in 2019 and -16 cm/hr for those searched in 2020. Searched ramping events in 2019 were also characterized by shorter wetted histories relative to previous years while the longest wetted histories among all searched events were recorded for the three events searched in 2020. 90th percentiles of wetted histories over individual searched ramping events ranged from 2.93 days up to 29.23 days in 2017 and from 0.83 days to 21.67 days in 2018, compared to between 0.64 and 15.14 days in 2019 and between 28.75 and 62.44 days in 2020.

Figure 1. Peace River flow as measured at the Peace River above Pine River WSC hydrometric station (07FA004) in each year of monitoring. Vertical green lines represent monitoring trips following targeted ramping events.

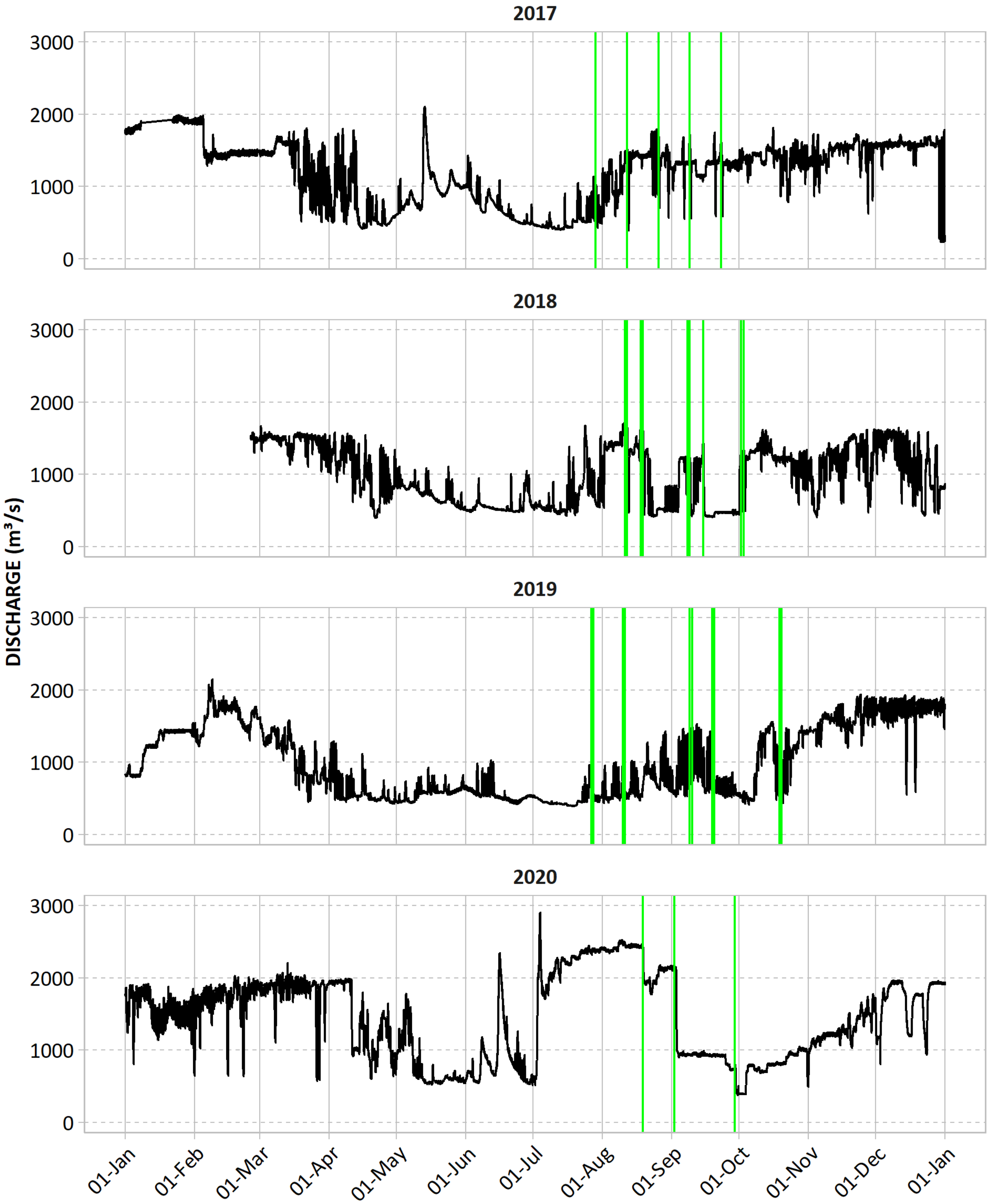


Figure 2. **Peace River flows as measured at the Peace River above Alces River WSC hydrometric station (07FD010) in each year of monitoring. Vertical green lines represent monitoring trips following targeted ramping events. Note no monitoring was conducted within Reaches 2 or 3 in 2018 or 2020).**

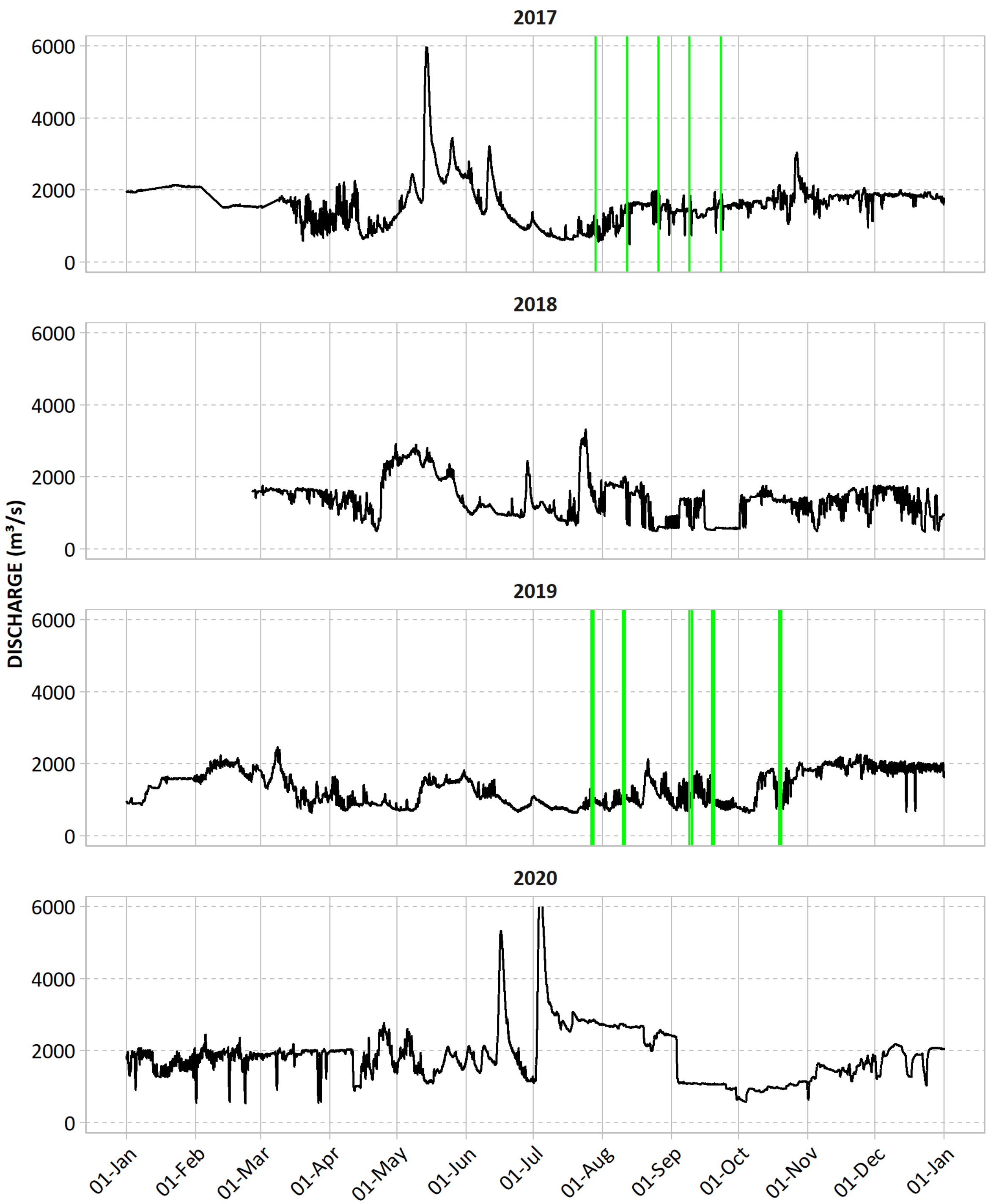


Table 4. Summary of hydrometric data from WSC hydrometric stations for all targeted ramping events in 2017 and 2018.

Year	Hydrometric Station	Trip	Reduction Start		Reduction End		Stage (cm)				Flow (m^3/s)				Wetted History		
			Date	Time (PST)	Date	Time (PST)	Start	End	Total Change	Ramping Rate (cm/hr)	Start	End	Total Change	Ramping Rate ($m^3/s/hr$)	10th %tile	Median	90th %tile
2017	Peace Above Pine River	1	29-Jul-17	05:35	29-Jul-17	20:00	129	13	-116	-21	1,015	423	-591	-105	0.07	0.81	11.21
		2	12-Aug-17	03:30	12-Aug-17	17:35	201	4	-196	-26	1,509	389	-1,119	-169	1.19	4.71	29.23
		3	26-Aug-17	03:55	26-Aug-17	11:55	229	74	-155	-27	1,733	700	-1,033	-197	0.39	0.70	13.62
		4	9-Sep-17	02:20	9-Sep-17	12:55	227	45	-182	-28	1,717	560	-1,156	-186	0.18	2.63	2.88
		5	23-Sep-17	03:05	23-Sep-17	14:55	212	51	-162	-26	1,600	585	-1,015	-168	0.22	2.75	3.01
	Peace Above Alces River	1	29-Jul-17	11:25	30-Jul-17	03:50	273	161	-113	-16	1,162	560	-602	-80	0.03	0.76	4.61
		2	12-Aug-17	05:25	13-Aug-17	00:20	342	143	-199	-22	1,646	487	-1,159	-136	0.09	2.59	18.49
		3	26-Aug-17	06:50	26-Aug-17	18:30	373	236	-137	-23	1,899	933	-966	-149	0.10	0.56	13.55
		4	9-Sep-17	08:20	9-Sep-17	20:10	365	199	-166	-24	1,836	738	-1,098	-295	0.17	0.87	2.88
		5	23-Sep-17	07:25	23-Sep-17	21:30	372	229	-143	-20	1,891	894	-998	-139	0.32	1.98	2.93
2018	Peace Above Pine River	1	10-Aug-18	22:20	11-Aug-18	17:50	227	18	-209	-27	1,720	476	-1,244	-170	0.04	9.60	21.67
			12-Aug-18	3:55	12-Aug-18	18:50	217	17	-200	-27	1,640	472	-1,168	-180	0.26	0.59	0.88
		2	18-Aug-18	1:20	18-Aug-18	17:10	218	12	-206	-28	1,640	455	-1,185	-180	0.14	5.48	5.77
			19-Aug-18	3:10	19-Aug-18	18:00	214	9	-205	-27	1,610	441	-1,169	-170	0.18	0.55	0.87
		3	8-Sep-18	2:20	8-Sep-18	15:05	164	18	-146	-22	1,240	476	-764	-122	0.03	0.74	4.70
			9-Sep-18	3:20	9-Sep-18	20:30	163	3	-160	-22	1,230	419	-811	-127	0.07	0.67	15.42
		4	15-Sep-18	1:15	15-Sep-18	23:10	192	4	-188	-24	1,440	422	-1,018	-149	0.05	2.67	5.68
		5	2-Oct-18	1:15	2-Oct-18	17:55	177	19	-158	-24	1,330	480	-850	-143	0.07	1.15	1.50
			3-Oct-18	3:05	3-Oct-18	18:10	179	22	-157	-23	1,350	489	-861	-135	0.15	0.52	0.83

Table 5. Summary of hydrometric data from WSC hydrometric stations for all targeted ramping events in 2019 and 2020.

Year	Hydrometric Station	Trip	Reduction Start		Reduction End		Stage (cm)				Flow (m^3/s)				Wetted History		
			Date	Time (PST)	Date	Time (PST)	Start	End	Total Change	Ramping Rate (cm/hr)	Start	End	Total Change	Ramping Rate ($m^3/s/hr$)	10th %tile	Median	90th %tile
2019	Peace Above Pine River	1	27-Jul-19	00:20	27-Jul-19	21:05	123	28	-95	-15	973	484	-489	-76	0.13	0.75	2.62
			28-Jul-19	05:45	28-Jul-19	19:00	44	23	-21	-4	556	463	-93	-19	0.01	0.37	3.54
		2	10-Aug-19	04:05	10-Aug-19	18:55	123	36	-87	-13	976	519	-457	-73	0.24	0.71	1.03
			11-Aug-19	05:00	11-Aug-19	19:40	53	30	-24	-6	597	490	-107	-27	0.02	0.52	2.84
		3	9-Sep-19	05:55	9-Sep-19	13:55	141	74	-67	-15	1,090	699	-391	-88	0.05	0.31	0.64
			10-Sep-19	01:20	10-Sep-19	14:50	199	83	-116	-18	1,500	746	-754	-120	0.03	0.58	0.90
		4	19-Sep-19	02:05	19-Sep-19	15:50	189	58	-132	-23	1,420	619	-801	-135	0.02	0.63	0.90
			20-Sep-19	03:25	20-Sep-19	19:10	84	47	-37	-11	752	569	-183	-54	0.01	0.45	3.02
		5	19-Oct-19	04:25	19-Oct-19	15:55	172	20	-152	-33	1,300	452	-848	-192	0.05	0.59	1.92
			20-Oct-19	05:10	20-Oct-19	14:15	120	16	-104	-28	957	435	-522	-146	0.07	0.64	15.14
	Peace Above Alces River	1	27-Jul-19	07:05	28-Jul-19	04:35	339	250	-89	-13	1,580	1,010	-570	-90	0.02	0.67	1.67
			28-Jul-19	11:25	29-Jul-19	02:05	259	230	-29	-7	1,060	903	-157	-40	0.01	0.17	2.75
		2	10-Aug-19	10:30	11-Aug-19	02:30	333	265	-68	-11	1,540	1,090	-450	-70	0.03	0.62	0.98
			11-Aug-19	09:30	12-Aug-19	03:25	275	249	-26	-7	1,150	1,000	-150	-30	0.01	0.23	2.26
		3	9-Sep-19	13:00	9-Sep-19	20:15	283	232	-50	-11	1,200	913	-287	-61	0.02	0.31	0.56
			10-Sep-19	08:35	10-Sep-19	21:00	351	257	-94	-13	1,670	1,050	-620	-90	0.14	0.61	0.87
		4	19-Sep-19	07:10	19-Sep-19	23:15	356	236	-120	-17	1,710	933	-777	-120	0.02	0.61	0.96
			20-Sep-19	11:20	21-Sep-19	01:20	249	214	-35	-6	1,000	822	-178	-32	0.02	0.70	4.22
		5	19-Oct-19	10:55	19-Oct-19	23:00	334	194	-140	-18	1,540	728	-812	-110	0.13	0.67	2.00
			20-Oct-19	11:55	20-Oct-19	21:20	275	196	-79	-14	1,150	737	-413	-72	0.09	0.48	0.82
2020	Peace Above Pine River	1	19-Aug-20	0:55	20-Aug-20	8:20	307	248	-59	-9	2,440	1,900	-540	-90	0.00	0.01	37.12
		2	2-Sep-20	18:35	4-Sep-20	0:20	277	111	-166	-24	2,150	904	-1,246	-190	0.00	0.01	62.44
		3	29-Sep-20	6:40	30-Sep-20	7:20	92	2	-91	-14	796	379	-417	-68	0.00	0.01	28.75

3.3. Fish Observations by Species and Age Class

Total fish observations and summaries of isolation, stranding, and combined rates by year, trip, and reach during interstitial and pool surveys are presented in Table 6 and Table 7, respectively. Total fish observations during interstitial sampling were highest in 2020 and lowest in 2019, and fish captures in pools were highest in 2017 and lowest in 2019. Fish observations were generally higher during pool sampling than during interstitial sampling, except in 2020 when the total number of interstitial fish observations was nine times higher than the total number of fish caught during pool electrofishing. Total fish observations across all years of monitoring and interstitial and pool sampling combined are presented by species and age class in Table 8. In order of relative proportions, Slimy Sculpins (*Cottus cognatus*), followed by suckers (including both Longnose Sucker (*Catostomus catostomus*) and White Sucker (*C. commersonii*)), Longnose Dace (*Rhinichthys cataractae*), and Lake Chub (*Couesius plumbeus*) made up most of the total isolated and stranded fish during all trips in all years. Similarly, with the exception of Slimy Sculpins, the vast majority (i.e., ~90%) of total isolated and stranded fish of these species were young-of-year (YOY) or juveniles, with these life stages also making up the vast majority of total estimates for other species as well.

Table 6. Summary of total counts and isolation, stranding, and combined rates of fish by year, trip, and reach during baseline interstitial surveys.

Year	Trip	Reach	Total Isolated Fish Observed	Isolation Rate (fish/100 m)				Total Stranded Fish Observed	Stranding Rate (fish/100 m)				Total Fish Observed	Combined Rate (fish/100 m)[1]			
				Mean	Min.	Max.	SE		Mean	Min.	Max.	SE		Mean	Min.	Max.	SE
2017	3	1	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a
		2	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a
		3	26	37.3	17.1	57.6	20.3	1	2.0	n/a	n/a	n/a	27	25.5	n/a	n/a	n/a
		DH	0	n/a	n/a	n/a	n/a	6	4.6	4.6	4.6	0.0	6	4.6	4.6	4.6	0.0
	4	1	5	15.2	15.2	15.2	0.0	0	n/a	n/a	n/a	n/a	5	15.2	n/a	n/a	n/a
		2	1	1.0	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	1	1.0	n/a	n/a	n/a
		3	0	n/a	n/a	n/a	n/a	23	65.7	65.7	65.7	0.0	23	65.7	65.7	65.7	0.0
		DH	0	n/a	n/a	n/a	n/a	1	1.1	n/a	n/a	n/a	1	1.1	n/a	n/a	n/a
	5	1	0	n/a	n/a	n/a	n/a	1	0.8	n/a	n/a	n/a	1	0.8	n/a	n/a	n/a
		2	4	2.2	2.2	2.2	0.0	1	1.1	n/a	n/a	n/a	5	1.6	n/a	n/a	n/a
		3	16	32.7	32.7	32.7	0.0	13	4.8	1.1	10.0	1.9	29	12.9	1.1	36.7	8.2
		DH	6	3.3	1.0	5.6	2.3	6	6.1	6.1	6.1	0.0	12	6.3	5.6	7.1	0.8
2018	1	1	7	7.3	0.9	20.0	6.3	0	n/a	n/a	n/a	n/a	7	7.3	n/a	n/a	n/a
		DH	35	7.8	1.3	21.3	3.8	0	n/a	n/a	n/a	n/a	35	7.8	n/a	n/a	n/a
	2	1	9	3.2	0.9	7.6	2.2	0	n/a	n/a	n/a	n/a	9	3.2	n/a	n/a	n/a
		DH	6	4.3	4.1	4.5	0.2	0	n/a	n/a	n/a	n/a	6	4.3	n/a	n/a	n/a
	3	1	0	n/a	n/a	n/a	n/a	2	1.9	1.9	1.9	0.0	2	1.9	1.9	1.9	0.0
		DH	0	n/a	n/a	n/a	n/a	1	0.8	n/a	n/a	n/a	1	0.8	n/a	n/a	n/a
	4	1	0	n/a	n/a	n/a	n/a	23	9.6	0.8	27.3	8.8	23	9.6	0.8	27.3	8.8
		DH	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a
	5	1	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a
		DH	5	2.2	0.7	3.2	0.8	1	3.2	n/a	n/a	n/a	6	3.2	n/a	n/a	n/a
2019	1	1	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a
		2	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a
		DH	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a
	2	1	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a
		2	2	0.7	0.7	0.7	0.0	0	n/a	n/a	n/a	n/a	2	0.7	n/a	n/a	n/a
		DH	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a
	3	1	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a
		2	2	1.7	1.7	1.7	0.0	0	n/a	n/a	n/a	n/a	2	1.7	n/a	n/a	n/a
		DH	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a
	4	1	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a
		2	2	1.6	1.6	1.6	0.0	3	1.3	1.0	1.6	0.3	5	2.1	1.0	3.2	1.1
		DH	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a
	5	1	0	n/a	n/a	n/a	n/a	1	0.4	n/a	n/a	n/a	1	0.4	n/a	n/a	n/a
		2	0	n/a	n/a	n/a	n/a	12	12.0	12.0	12.0	0.0	12	12.0	12.0	12.0	0.0
		DH	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a	0	n/a	n/a	n/a	n/a
2020	1	1	0	n/a	n/a	n/a	n/a	62	7.6	1.0	18.6	2.5	62	7.6	1.0	18.6	2.5
		DH	0	n/a	n/a	n/a	n/a	18	3.9	1.1	10.0	1.6	18	3.9	1.1	10.0	1.6
	2	DH	195	18.8	0.7	135.0	13.1	535	36.5	2.0	193.0	15.0	730	44.7	1.6	328.0	22.4
	3	DH	23	10.8	2.0	27.5	8.3	50	9.9	2.0	31.0	5.6	73	16.4	2.0	40.0	7.9

[1] Combined stranded and isolated fish.

Table 7. Summary of total observed counts and isolation rates of fish by year, trip, and reach during baseline pool surveys.

Year	Trip	Reach	Total Isolated Fish Observed	Isolation Rate (fish/100 m)			
				Mean	Min.	Max.	SE
2017	1	3	18	5.6	1.2	10.0	4.4
		DH	0	n/a	n/a	n/a	n/a
	2	1	3	0.7	0.6	0.7	0.0
		2	14	1.6	0.3	3.1	0.8
		3	18	1.4	0.3	4.1	0.9
		DH	1	0.2	n/a	n/a	n/a
	3	1	1	1.9	n/a	n/a	n/a
		2	0	n/a	n/a	n/a	n/a
		3	5	1.8	0.7	3.1	0.5
		DH	0	n/a	n/a	n/a	n/a
	4	1	27	5.2	3.4	7.0	1.8
		2	2	6.6	6.6	6.6	0.0
		3	29	8.4	1.3	20.9	6.2
		DH	11	6.1	0.7	11.5	5.4
	5	1	11	6.2	6.2	6.2	0.0
		2	2	4.3	4.3	4.3	0.0
		3	1	0.6	n/a	n/a	n/a
		DH	7	3.5	1.8	5.6	1.0
2018	1	1	27	5.6	1.3	8.9	2.3
		DH	40	2.3	0.3	7.7	0.8
	2	1	36	3.4	2.6	4.1	0.7
		DH	11	4.0	0.4	9.3	2.7
	3	1	0	n/a	n/a	n/a	n/a
		DH	5	2.1	0.2	3.9	1.8
	4	1	0	n/a	n/a	n/a	n/a
		DH	1	1.7	n/a	n/a	n/a
	5	1	0	n/a	n/a	n/a	n/a
		DH	2	1.5	1.3	1.8	0.3
2019	1	1	0	n/a	n/a	n/a	n/a
		2	1	0.3	n/a	n/a	n/a
		DH	4	1.1	0.8	1.3	0.2
	2	1	0	n/a	n/a	n/a	n/a
		2	3	2.8	0.8	6.3	1.7
		DH	0	n/a	n/a	n/a	n/a
	3	1	4	1.1	0.2	2.2	0.4
		2	2	7.4	7.4	7.4	0.0
		DH	1	1.4	n/a	n/a	n/a
	4	1	6	2.1	1.0	3.2	1.1
		2	30	6.1	0.4	14.7	3.1
		DH	0	n/a	n/a	n/a	n/a
	5	1	4	0.7	0.5	0.9	0.2
		2	0	n/a	n/a	n/a	n/a
		DH	2	0.5	0.3	0.6	0.1
2020	1	1	40	1.4	0.0	5.3	1.3
		DH	6	2.4	2.4	2.4	0.0
	2	DH	52	2.5	0.2	6.0	1.0

Table 8. Total fish observations by species and age class tallied across all years and surveys.

Group	Species	YOY[1]	Juvenile	Adult	Total[2]
Sport Fish	Arctic Grayling	1	1	0	2
	Burbot	3	2	0	5
	Kokanee	0	1	0	1
	Mountain Whitefish	31	3	0	34
	Northern Pike	8	13	1	22
	Rainbow Trout	5	0	0	5
	Walleye	1	0	0	1
	Yellow Perch	0	4	0	4
Suckers	Largescale Sucker	3	9	0	12
	Longnose Sucker	26	46	1	73
	White Sucker	1	2	0	3
	Sucker *spp*.	242	34	0	276
Minnows	Flathead Chub	0	1	0	1
	Lake Chub	11	27	7	45
	Longnose Dace	101	74	10	185
	Northern Pikeminnow	0	2	0	2
	Redside Shiner	5	1	5	11
	Trout-Perch	0	1	0	1
	Dace *spp*.	9	0	0	9
Sculpins	Prickly Sculpin	29	16	4	49
	Slimy Sculpin	93	92	97	282
	Sculpin *spp*.	249	30	0	279
Other	Unknown	196	1	0	197
Grand Total		***1,014***	***360***	***125***	***1,499***

[1] YOY = Young-of-year.

[2] Excludes 24 fish that could not be assigned to an age class because their lengths were not recorded.

3.4. Quantifying Fish Stranding and Isolation - Interstitial Sampling

3.4.1. Fish Stranding and Isolation Relationships

3.4.1.1. Effect of Reach

Across all years and searched ramping events, the mean combined stranding and isolation rate was 5.62 (standard error (SE) 2.43) fish/100 m in the future diversion headpond, 1.18 (SE 0.4) fish/100 m in Reach 1, 0.38 (SE 0.21) fish/100 m in Reach 2, and 9.7 (SE 4.45) fish/100 m in Reach 3 (Table 9). Differences among reaches are likely due in part, to variability in the timing of sampling in each reach relative to that of the associated ramping event and the differing magnitudes of searched ramping events in each year (Ecora 2018, 2019, 2020; Section 3.2). For example, the higher combined mean rates in Reach 3 are likely the result of several factors: 1) exceptionally high fish observations along a single high-risk bar over multiple trips in 2017, and 2) more search effort expended in Reach 3 relative to other reaches in 2017 (the only year that this reach was searched), when the magnitude of ramping events were higher on average compared to other years. The lower mean rates of fish isolation and stranding in Reaches 1 and 2 may also in part be due to a high proportion of searches within these reaches being conducted in 2019 during lower flows and following smaller ramping events (Ecora 2020).

Considering all years of monitoring, there is evidence of differences in combined fish isolation and stranding rates among reaches (e.g., difference in mean ranks; Kruskal-Wallis H Test (K-W): Chi-squared (Chi-sq.) = 12.78; degrees freedom (df) = 3; $p = 0.01$ at a Simes-Corrected α (S-cor. α) = 0.03, although statistical differences were small with Epsilon-Squared (E-Sq.) = 0.04; Figure 3, Table 12). These results were consistent when considering stranding and isolation rates separately as well.

Figure 3. Linear densities of stranded and isolated fish by reach (future diversion headpond (DH), and Reaches 1, 2, and 3) observed during interstitial sampling events in all years including sites where no fish were captured (NFC). Data points have been jittered for presentation.

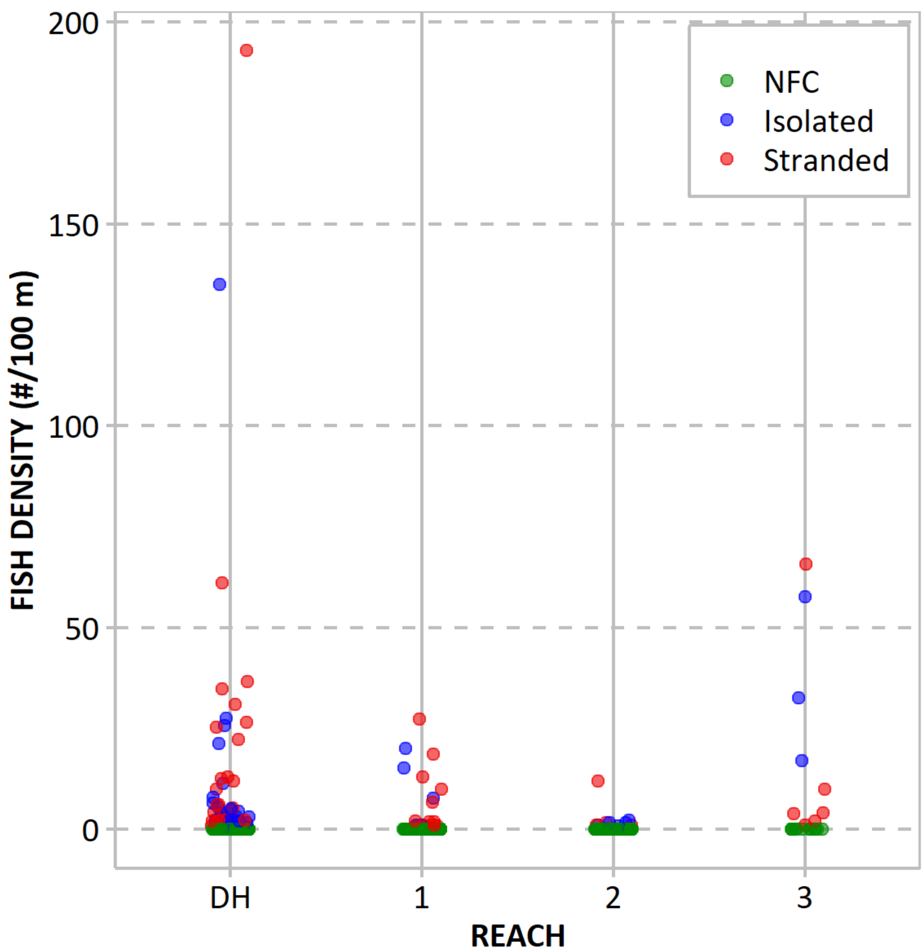


Table 9. Summary of fish stranding and isolation rates by reach observed during interstitial sampling events in each year.

Year	Reach	Isolation Rate (fish/100 m)					Stranding Rate (fish/100 m)					Total Observed (fish/100 m)[2]				
		Total[1]	Mean	Min.	Max.	SE	Total[1]	Mean	Min.	Max.	SE	Total[1]	Mean	Min.	Max.	SE
2017	DH	6	0.66	0.00	5.56	0.55	13	1.19	0.00	6.12	0.71	19	1.84	0.00	7.14	0.88
	1	5	2.16	0.00	15.15	2.16	1	0.11	n/a	n/a	n/a	6	2.28	0.00	15.15	2.15
	2	5	0.29	0.00	2.16	0.21	1	0.10	n/a	n/a	n/a	6	0.39	0.00	2.16	0.22
	3	42	5.37	0.00	57.58	3.28	37	4.34	0.00	65.71	3.28	79	9.70	0.00	65.71	4.45
2018	DH	46	0.92	0.00	21.25	0.42	2	0.07	0.00	3.23	0.06	48	0.98	0.00	21.25	0.43
	1	16	0.51	0.00	20.00	0.34	25	0.50	0.00	27.27	0.44	41	1.00	0.00	27.27	0.55
2019	DH	0	0.00	n/a	n/a	n/a	0	0.00	n/a	n/a	n/a	0	0.00	n/a	n/a	n/a
	1	0	0.00	n/a	n/a	n/a	1	0.01	n/a	n/a	n/a	1	0.01	n/a	n/a	n/a
	2	6	0.08	0.00	1.71	0.05	15	0.30	0.00	12.00	0.25	21	0.38	0.00	12.00	0.25
2020	DH	218	4.70	0.00	135.00	2.95	603	10.79	0.00	193.00	4.38	821	15.49	0.00	328.00	7.17
	1	0	0.00	n/a	n/a	n/a	62	5.31	0.00	18.62	2.07	62	5.31	0.00	18.62	2.07
All	DH	270	1.97	0.00	135.00	0.99	618	3.66	0.00	193.00	1.49	888	5.62	0.00	328.00	2.42
	1	21	0.42	0.00	20.00	0.23	89	0.76	0.00	27.27	0.33	110	1.18	0.00	27.27	0.40
	2	11	0.12	0.00	2.16	0.06	16	0.26	0.00	12.00	0.20	27	0.38	0.00	12.00	0.21
	3	42	5.37	0.00	57.58	3.28	37	4.34	0.00	65.71	3.28	79	9.70	0.00	65.71	4.45

[1] Total fish observed over entire area of searched habitat.

[2] Combined stranded and isolated fish.

3.4.1.2. Effect of Channel Type

There were differences in the distribution of stranding and isolation rates among multi-thread and single-thread channel habitats (mean rates of 4.12 (SE 1.29) fish/100 m compared to 0.07 (SE 0.03) fish/100 m, respectively, Figure 4, Table 10). Although these results were supported by statistical tests (Table 12; K-W: Chi-sq. = 8.65, df = 1; p = 0.003 at S-cor. α = 0.02), statistical differences were small (E-Sq. = 0.03). This was consistent for stranded rates but not isolation rates.

Figure 4. Linear densities of stranded and isolated fish by channel type observed during interstitial sampling events including sites where no fish were captured (NFC). Data points have been jittered for presentation.

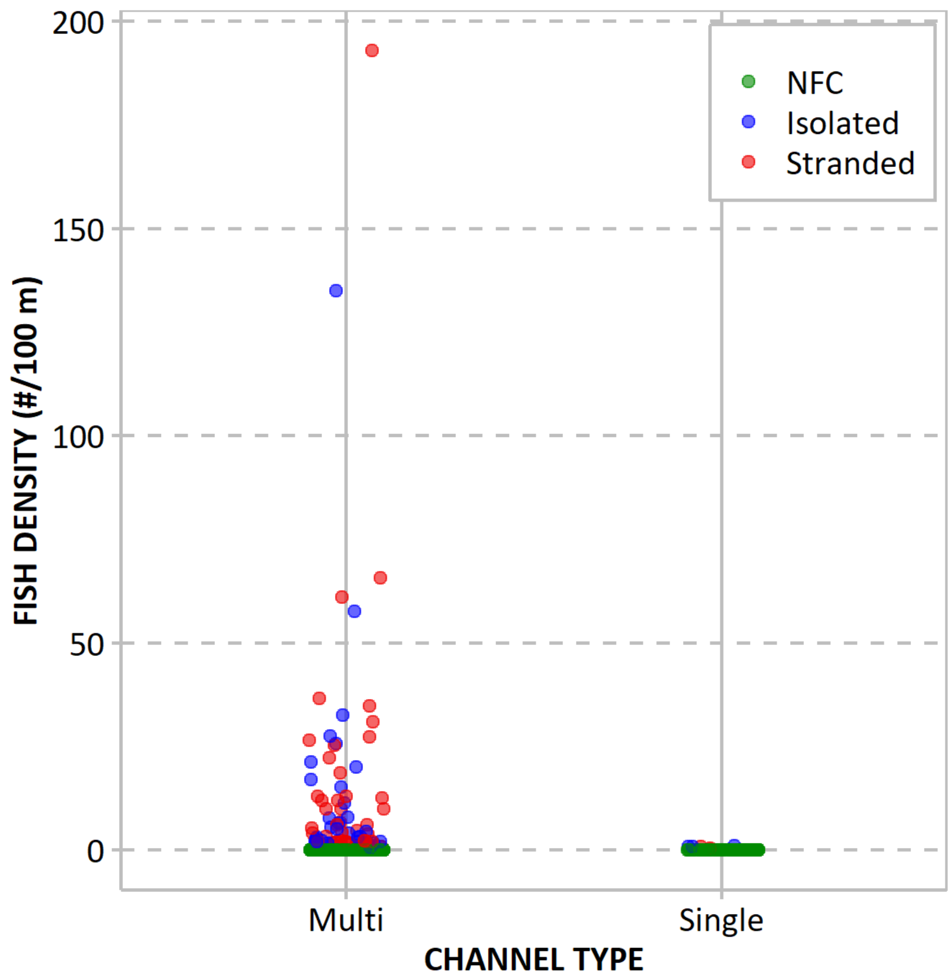


Table 10. Summary of fish stranding and isolation rates by channel type observed during interstitial sampling events in each year.

Year	Channel Type	Isolation Rate (fish/100 m)					Stranding Rate (fish/100 m)					Total Observed (fish/100 m)[2]				
		Total[1]	Mean	Min.	Max.	SE	Total[1]	Mean	Min.	Max.	SE	Total[1]	Mean	Min.	Max.	SE
2017	Multi	58	3.15	0.00	57.58	1.62	51	2.37	0.00	65.71	1.58	109	5.52	0.00	65.71	2.22
	Single	0	0.00	n/a	n/a	n/a	1	0.13	n/a	n/a	n/a	1	0.13	n/a	n/a	n/a
2018	Multi	60	0.86	0.00	21.25	0.33	27	0.36	0.00	27.27	0.28	87	1.22	0.00	27.27	0.43
	Single	2	0.08	0.00	1.03	0.06	0	0.00	n/a	n/a	n/a	2	0.08	0.00	1.03	0.06
2019	Multi	4	0.04	0.00	1.71	0.03	15	0.18	0.00	12.00	0.15	19	0.22	0.00	12.00	0.15
	Single	2	0.03	0.00	0.75	0.03	1	0.02	n/a	n/a	n/a	3	0.05	0.00	0.75	0.03
2020	Multi	218	3.87	0.00	135.00	2.44	665	9.83	0.00	193.00	3.63	883	13.70	0.00	328.00	5.93
	Single	0	0.00	n/a	n/a	n/a	0	0.00	n/a	n/a	n/a	0	0.00	n/a	n/a	n/a
All	Multi	340	1.58	0.00	135.00	0.57	758	2.54	0.00	193.00	0.81	1,098	4.12	0.00	328.00	1.29
	Single	4	0.05	0.00	1.03	0.03	2	0.02	0.00	0.79	0.02	6	0.07	0.00	1.03	0.03

[1] Total fish observed over entire area of searched habitat.

[2] Combined stranded and isolated fish.

3.4.1.3. Effect of Stranding Risk Category

Across all years of interstitial sampling, there was evidence of a difference in the distribution of stranding and isolation rates between high and low stranding risk mesohabitats, with only one stranded and one isolated fish ever observed in low-risk mesohabitat compared to an average of 4.11 (SE 1.29) fish/100 m in high-risk mesohabitat (Figure 5, Table 11, Table 12, K-W: Chi-sq. = 14.04; df = 1; p = 0.0002 at S-cor. α = 0.01; E-Sq. = 0.04). These results were consistent for both stranding rates and isolation rates.

Figure 5. Linear densities of stranded and isolated fish by mesohabitat stranding risk observed during interstitial sampling events in all years, including sites where no fish were captured (NFC). Data points have been jittered for presentation.

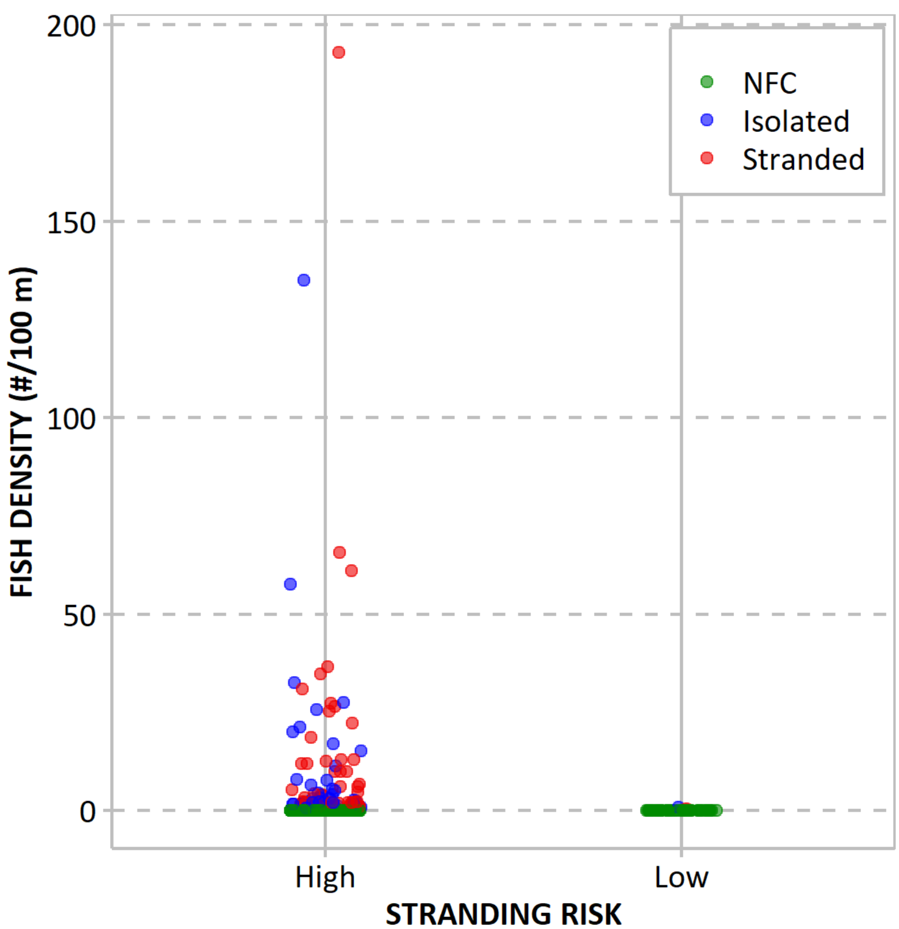


Table 11. Summary of fish stranding and isolation rates by mesohabitat stranding risk observed during interstitial sampling events in each year.

Year	Stranding Risk	Isolation Rate (fish/100 m)					Stranding Rate (fish/100 m)					Total Observed (fish/100 m)[2]				
		Total[1]	Mean	Min.	Max.	SE	Total[1]	Mean	Min.	Max.	SE	Total[1]	Mean	Min.	Max.	SE
2017	High	58	3.57	0.00	57.58	1.83	52	2.72	0.00	65.71	1.78	110	6.29	0.00	65.71	2.49
	Low	0	0.00	n/a	n/a	n/a	0	0.00	n/a	n/a	n/a	0	0.00	n/a	n/a	n/a
2018	High	61	0.87	0.00	21.25	0.33	27	0.36	0.00	27.27	0.28	88	1.23	0.00	27.27	0.43
	Low	1	0.04	n/a	n/a	n/a	0	0.00	n/a	n/a	n/a	1	0.04	n/a	n/a	n/a
2019	High	6	0.05	0.00	1.71	0.03	15	0.16	0.00	12.00	0.14	21	0.21	0.00	12.00	0.14
	Low	0	0.00	n/a	n/a	n/a	1	0.02	n/a	n/a	n/a	1	0.02	n/a	n/a	n/a
2020	High	218	3.87	0.00	135.00	2.44	665	9.83	0.00	193.00	3.63	883	13.70	0.00	328.00	5.93
	Low	0	0.00	n/a	n/a	n/a	0	0.00	n/a	n/a	n/a	0	0.00	n/a	n/a	n/a
All	High	343	1.58	0.00	135.00	0.57	759	2.54	0.00	193.00	0.81	1102	4.11	0.00	328.00	1.29
	Low	1	0.02	n/a	n/a	n/a	1	0.01	n/a	n/a	n/a	2	0.02	0.00	0.86	0.02

[1] Total fish observed over entire area of searched habitat.

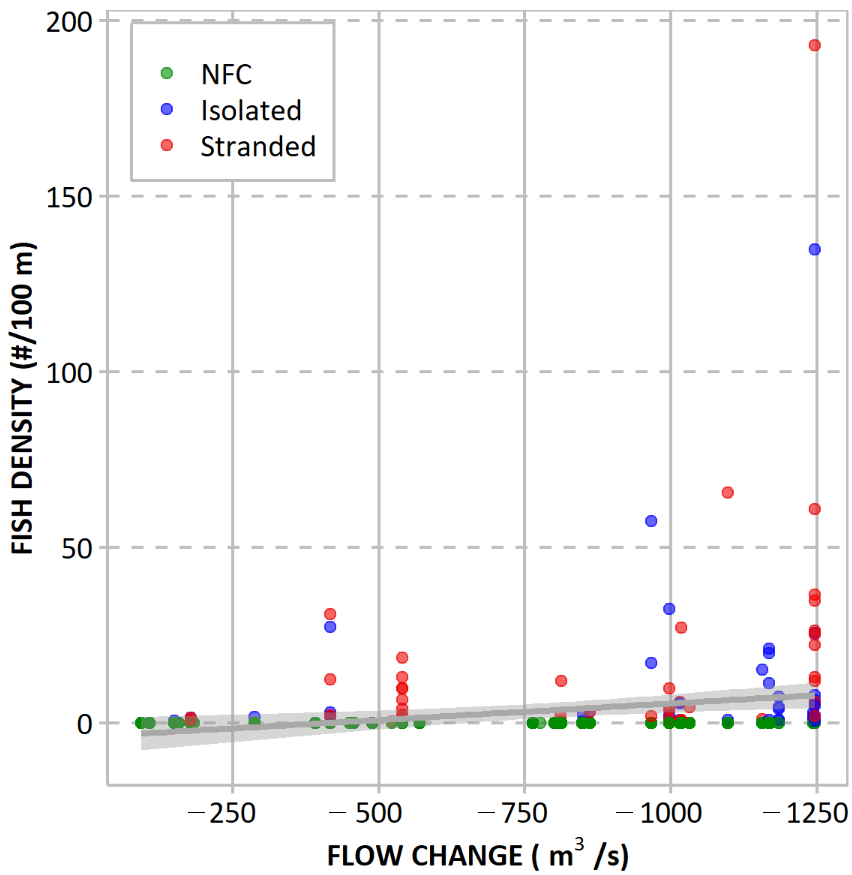
[2] Combined stranded and isolated fish.

3.4.1.4. Effects of Changes in Hydrology and Shoreline Slope

Flow Change

There is evidence that frequency and magnitude of combined stranding and isolation rates increased with flow change measured at the nearest WSC hydrometric station, increasing from a mean and maximum rate of 1.32 and 40.00 fish/100 m, respectively, to a mean and maximum rate of 4.65 and 328.00 fish/100 m, respectively, at reductions greater than 750 m^3/s (Figure 6, Table 12); Kendall Rank Correlation (K-R): Tau = 0.22; $p < 0.0001$ at S-cor. $\alpha = 0.008$; slope = 0.01 for combined rates). This relationship was also evident for isolation but not for stranding.

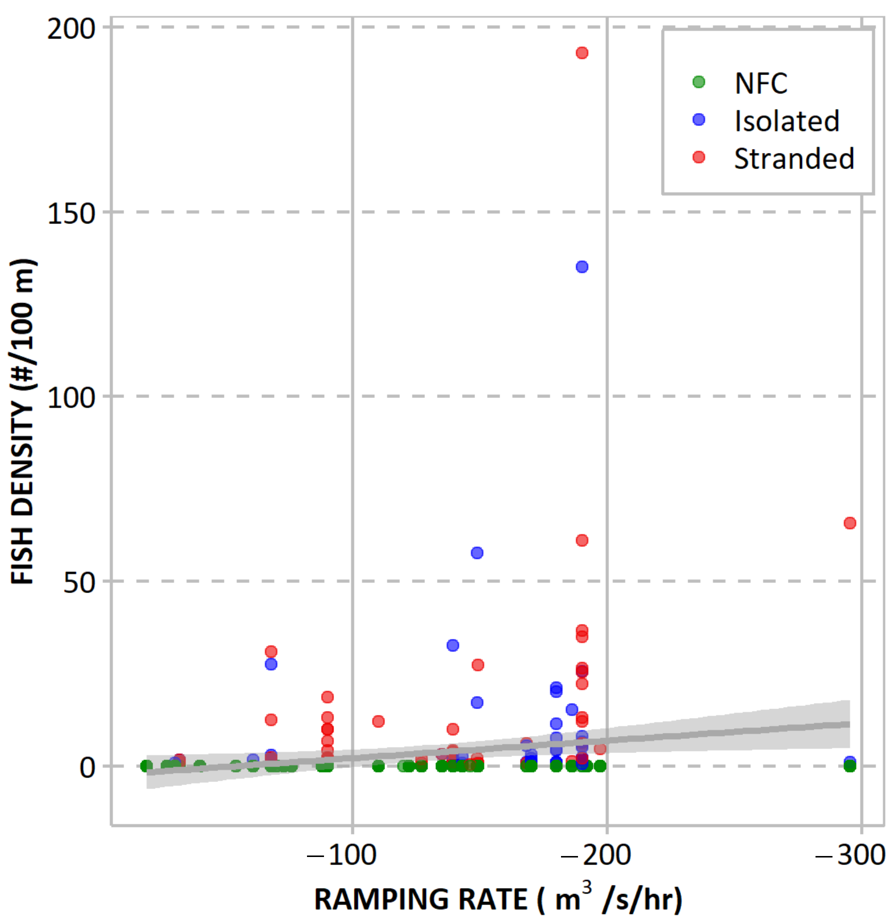
Figure 6. Linear densities of stranded and isolated fish by flow change (as measured at the nearest WSC hydrometric station) observed during interstitial sampling events in all years including sites where no fish were captured (NFC).



Ramping Rate

Frequency and magnitude of stranding and isolation rates also increased with ramping rate measured at the nearest WSC hydrometric station, increasing from a mean and maximum rate of 1.34 and 40.00 fish/100 m, respectively, to a mean and maximum rate of 4.61 and 328.00 fish/100 m, respectively at rates above 100 $m^3/s/hr$ (Figure 7, Table 12; K-R: Tau = 0.18; $p < 0.0001$ at S-cor. $\alpha = 0.01$; slope = 0.05) for combined rates). As with flow change, this relationship was also evident for isolation rates, but not stranding rates.

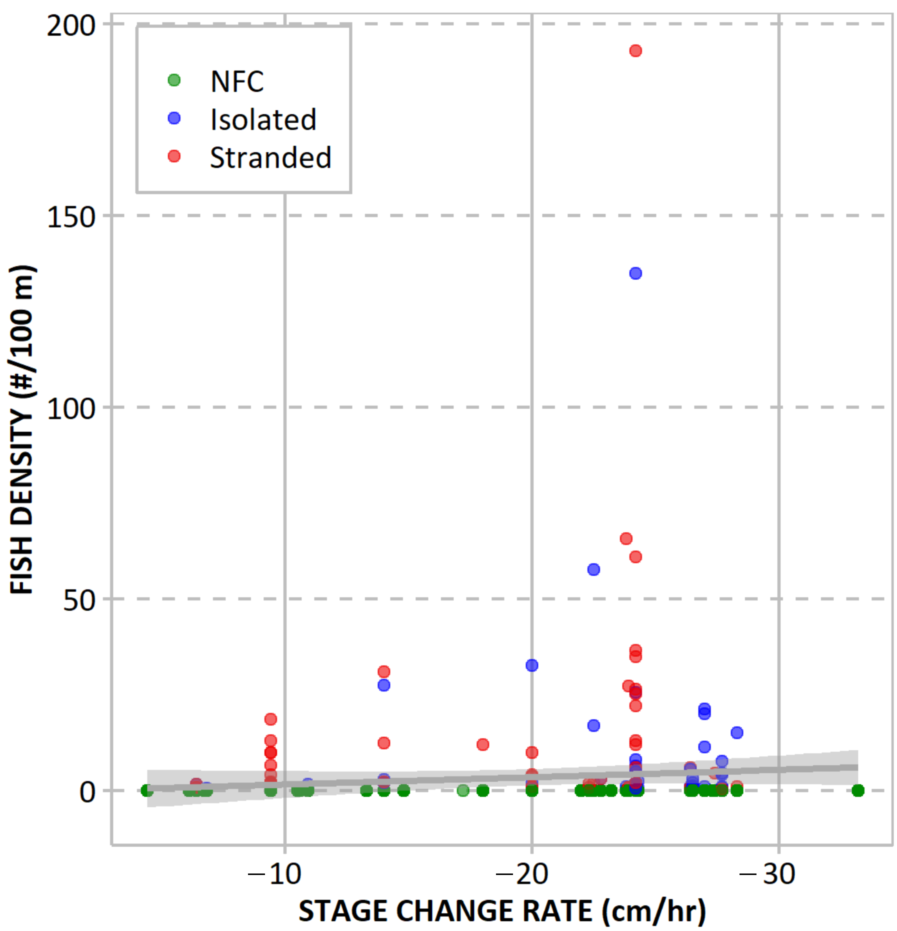
Figure 7. Linear densities of stranded and isolated fish by flow ramping rate (as measured at the nearest WSC hydrometric station) observed during interstitial sampling events in all years including sites where no fish were captured (NFC).



Stage Change Rate

Frequency and magnitude of stranding and isolation rates also increased with stage change rate measured at the nearest WSC hydrometric station, increasing from a mean and maximum rate of 1.34 and 40.00 fish/100 m, respectively, to a mean and maximum rate of 4.75 and 328.00 fish/100 m, respectively at rates ≥ 20 cm/hr (Figure 8, Table 12; K-R: Tau = 0.09; $p = 0.03$ at S-cor. $\alpha = 0.05$; slope = 0.19 for combined rates). As with flow change and ramping rate, this relationship was also evident for isolation rates, but not stranding rates. Note that site-specific stage change rates were not measured. The PAP hydrometric station is at rkm 112 and the PAA hydrometric station is at rkm 164, while individual monitoring sites ranged from rkm 34 to rkm 231; stage change rates at individual monitoring sites may therefore differ due to attenuation, inflow and differences in channel morphology. Validation of the relationship between stranding or isolation rates and stage change rate would require additional site-specific hydrometric data collection which is not currently considered under the Mon-12 scope.

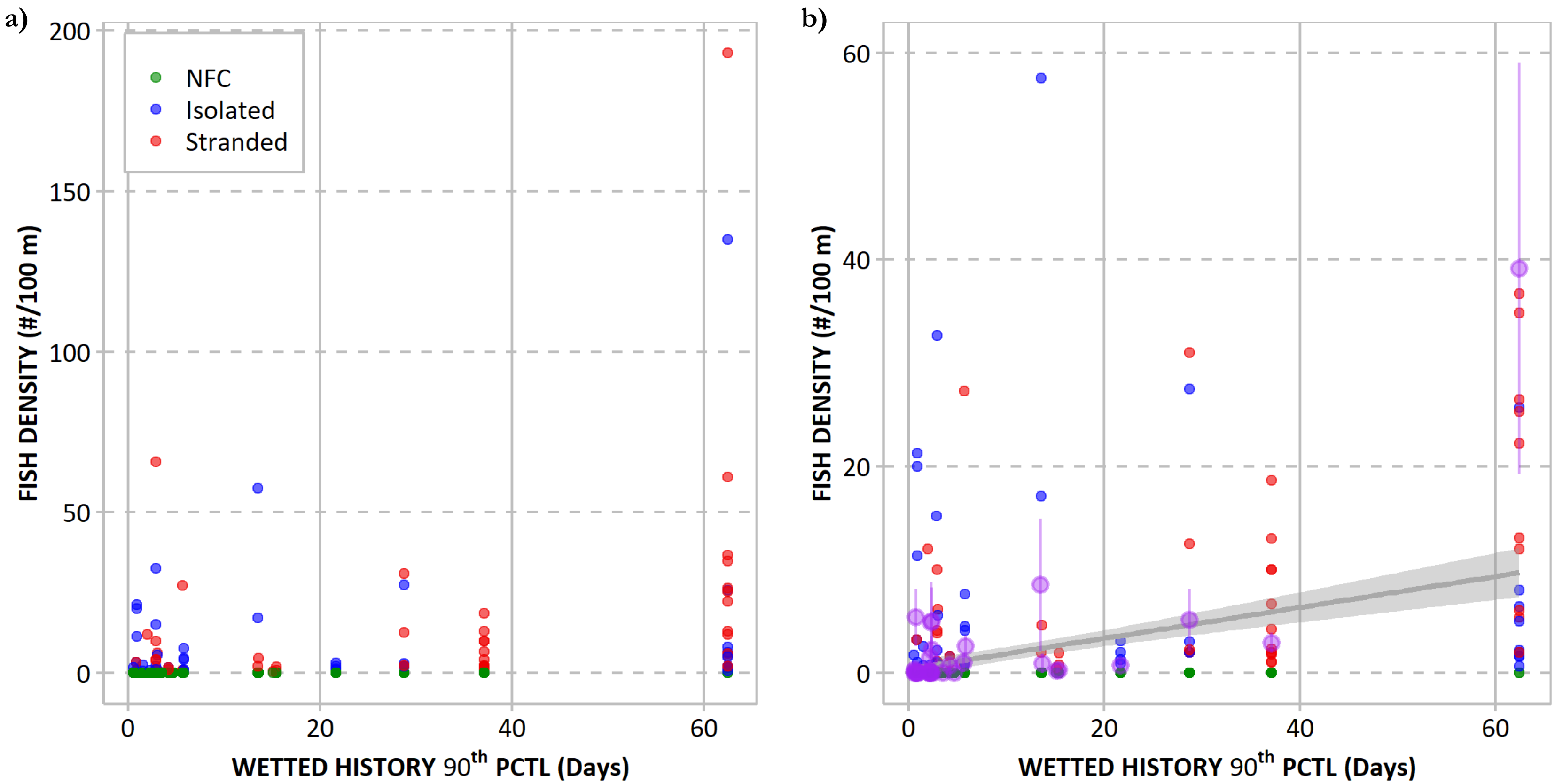
Figure 8. Linear densities of stranded and isolated fish by stage change rate (as measured at the nearest WSC hydrometric station) observed during interstitial sampling events in all years including sites where no fish were captured (NFC).



Wetted History

As expected, there was evidence that stranding and isolation rates also increased with longer wetted histories measured at the nearest WSC hydrometric station (Figure 9, Table 12; K-R: Tau = 0.30; $p < 0.0001$ at S-cor. A = 0.007, slope = 0.40), although the relationship is heavily driven by high stranding and isolation rates recorded following ramping events with long wetted histories (90th percentile wetted history of 62.4 days as recorded at the Peace River above Pine River hydrometric gauge on September 2, 2020). This relationship was consistent when considering stranding and isolation rates separately. This relationship is not readily apparent due to the wide range and density of stranding rates following short wetted histories (e.g., < 10 days; Figure 9, panel A). Therefore, the data are also presented for the lower range of combined stranding and isolation rates (0 to 60 fish/100 m) with the mean combined rates (± SE) and trend line (± 95% CI) derived from a linear regression to better illustrate the overall relationship tested above (Figure 9, panel B).

Figure 9. Linear densities of stranded and isolated fish by wetted history observed during interstitial sampling events in all years including sites where no fish were captured (NFC) for the entire range of data (A) and up to 60 fish/100 m with mean combined rates (± SE; purple) and trend line from linear regression (± 95% CI) (B).



Shoreline Slope

The shoreline slope characterizing individual sites ranged from 0% to 37.8%. Combined rates of fish stranding and isolation decreased with shoreline slope (Figure 10, Table 12; K-R: Tau = 0.14; p = 0.03; Slope = 0.09). This relationship was consistent for stranding rates but not for isolation rates. The mean combined fish stranding and isolation rates below and above the high-risk threshold of 5% were essentially the same at 5.6 and 5.7 fish/100 m, respectively, whereas the maximum combined rate was considerably higher below the 5% threshold at 328.0 fish/100 m compared to only 65.7 fish/100 m for sites with shoreline slopes > 5%. No stranding or isolation was observed at shoreline slopes > 16.9%; with one exception that had high measurement error[1]. Considering measurement uncertainty, while some stranding and isolation of fish was observed at site-level shoreline slopes above the high-risk threshold of 5%, these slopes, extracted from spatial data, were typically higher than those of the high-risk sections of habitat actually being searched within the sites as corroborated by field observations and photographs.

[1] At one site a combined rate of 32.7 fish/100 m was recorded at a high-risk stranding site with a recorded shoreline slope of 37.8%. However, shoreline slope was not recorded for this site in the field but rather extracted from spatial data at a single waypoint recorded for the site. Photographs taken during searches showed a lower slope within the habitat searched. In addition, the high isolation and stranding rates within this site were likely due to the long wetted history (62.4 days), large total flow change (-1,246 m^3/s) and high ramping rate (-190 $m^3/s/hr$) – this event was among the largest of all events searched during baseline monitoring.

Figure 10. **Linear densities of stranded and isolated fish by overall site shoreline slope observed during interstitial sampling events in all years including sites where no fish were captured (NFC).**

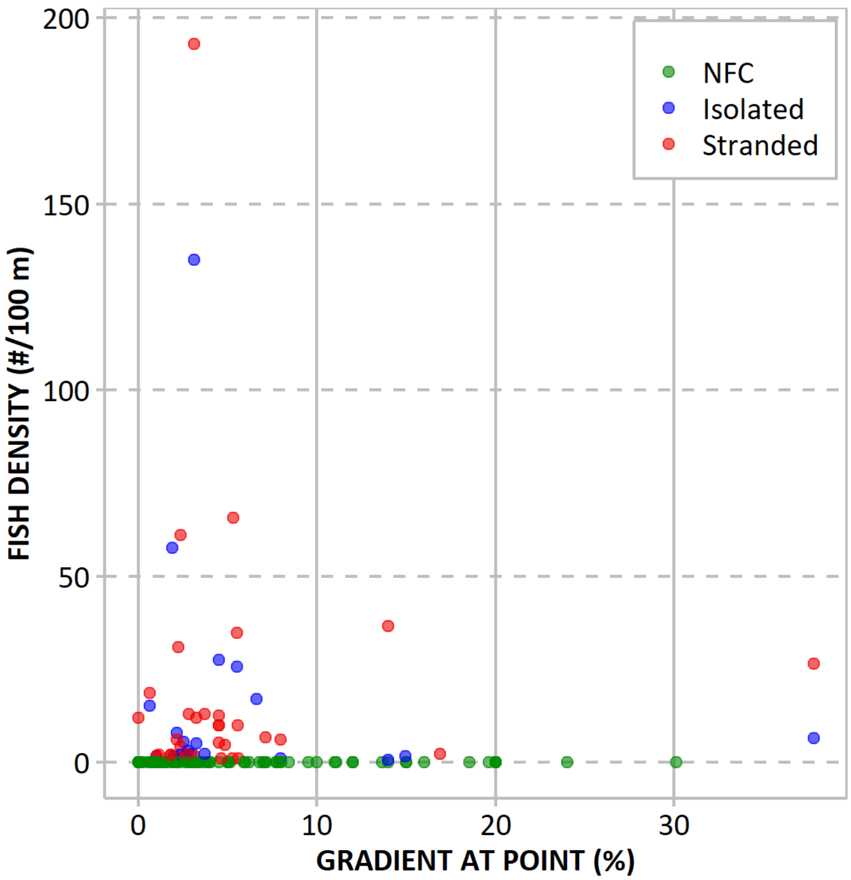


Table 12. Summary of all non-parametric tests for fish isolation, stranding, and combined rates observed during interstitial sampling events.

Condition	Non-Parametric Test[1]	Explanatory Variable	P-Value	Simes Modified α[2]	Effect Size[3]
Isolated	Kruskal-Wallis	Reach	0.03	0.03	0.03
		Channel Type	0.08	0.05	0.01
		Stranding Risk	0.01	0.02	0.02
	Kendall	Flow Change (m^3/s)	< 0.0001	0.007	0.00
		Ramping Rate ($m^3/s \times hr^{-1}$)	< 0.0001	0.008	0.02
		Stage Change Rate (cm/hr)	< 0.0001	0.01	0.10
		Wetted History (Days)	0.001	0.01	0.11
		Slope at Point (%)	0.11	n/a	-0.03
Stranded	Kruskal-Wallis	Reach	0.01	0.013	0.03
		Channel Type	0.01	0.01	0.02
		Stranding Risk	0.002	0.01	0.03
	Kendall	Flow Change (m^3/s)	0.03	0.02	0.006
		Ramping Rate ($m^3/s \times hr^{-1}$)	0.11	0.03	0.03
		Stage Change Rate (cm/hr)	0.49	0.05	0.09
		Wetted History (Days)	< 0.0001	0.007	0.29
		Slope at Point (%)	0.02	n/a	0.12
Combined	Kruskal-Wallis	Reach	0.01	0.03	0.04
		Channel Type	0.003	0.02	0.03
		Stranding Risk	0.0002	0.01	0.04
	Kendall	Flow Change (m^3/s)	< 0.0001	0.008	0.01
		Ramping Rate ($m^3/s \times hr^{-1}$)	< 0.0001	0.01	0.05
		Stage Change Rate (cm/hr)	0.03	0.05	0.19
		Wetted History (Days)	< 0.0001	0.007	0.40
		Slope at Point (%)	0.03	n/a	0.09

[1] Kruskal-Wallis U Test used for categorical variables; Kendall Rank Correlation used for

[2] n/a = not applicable; slope at point tests were run on less than half of the dataset so were not included in Simes modifications.

[3] Effect size = Epsilon-Squared for Kruskal-Wallis U Test (Cohen 1988) and slope coefficient from linear regression for Kendall Rank Correlation.

3.5. Quantifying Fish Isolation – Pool Sampling

3.5.1. Fish Isolation Relationships

3.5.1.1. Effect of Reach

Across all four years of monitoring, average densities of fish in pools were 1.31 (SE 0.28) fish/100 m^2 in the diversion headpond, 1.05 (SE 0.28) fish/100 m^2 in Reach 1, and 1.34 (SE 0.45) fish/100 m^2 in Reach 2 (Figure 11, Table 13). Reach 3 was only sampled in 2017 with an average of 1.84 (SE 0.84) fish/100 m^2 isolated in sampled pools. Considering all years of monitoring, there was no statistical evidence that the distribution of isolated fish densities in pools differed among reaches (Table 16; K-W: Chi-sq. = 3.13; p = 0.37 at S-cor. A = 0.025; E-Sq. = 0.02).

Figure 11. Areal densities of isolated fish by reach (future diversion headpond (DH), and Reaches 1, 2, and 3) observed during pool sampling events in all years including sites where no fish were captured (NFC). Data points have been jittered for presentation.

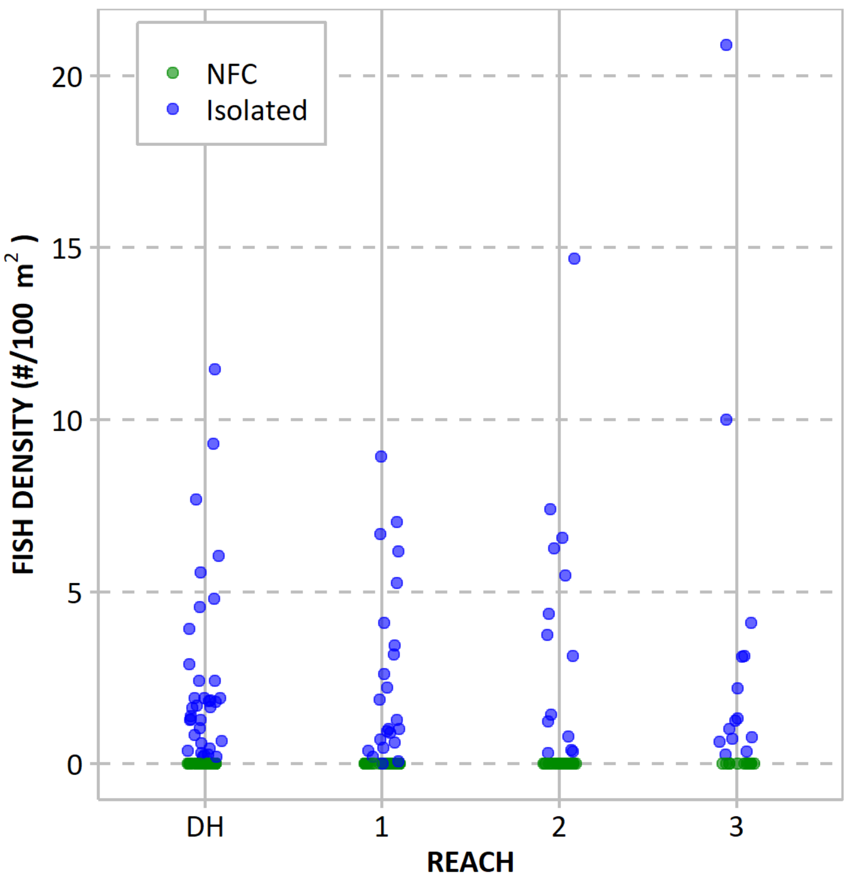


Table 13. Summary of isolated fish densities by reach observed during pool sampling events in each year.

Year	Reach	Isolation Rate (fish/100 m²)				
		Total[1]	Mean	Min.	Max.	SE
2017	DH	19	2.03	0.00	11.48	0.94
	1	42	2.47	0.00	7.02	0.99
	2	18	1.04	0.00	6.56	0.52
	3	71	1.84	0.00	20.90	0.84
2018	DH	59	1.30	0.00	9.30	0.39
	1	63	0.91	0.00	8.92	0.44
2019	DH	7	0.34	0.00	1.39	0.14
	1	14	0.58	0.00	3.16	0.22
	2	36	1.50	0.00	14.69	0.65
2020	DH	58	1.66	0.00	6.04	0.66
	1	40	1.14	0.00	5.26	1.03
All	DH	143	1.31	0.00	11.48	0.28
	1	159	1.05	0.00	8.92	0.28
	2	54	1.34	0.00	14.69	0.45
	3	71	1.84	0.00	20.90	0.84

[1] Total fish observed over entire area of searched habitat.

3.5.1.2. Effect of Channel Type

Overall, isolated fish densities within pools in single- and multi-thread channel habitats did not differ markedly (mean density of 1.51 (SE 0.61) and 1.29 (SE 0.21) fish/100 m², respectively, Figure 12, Table 14) and there was no statistical evidence that the distribution of fish densities in isolated pools differed between the two channel types (Table 16; K-W: Chi-sq. = 0.01; p = 0.96 at S-cor. A = 0.05; E-Sq. = 0.0002). The majority of pool sampling was conducted in multi-thread channels (90% of all pool sampling events across all years). The greater effort devoted to sampling multi-thread habitats reflects the tendency of pools to form along mid-stream or side channel bars in multi-thread channels, rather than along single-thread channels. Ecora (2018, 2019, 2020) observed less high-risk mesohabitat within single-thread compared to multi-thread habitat.

Figure 12. Densities of isolated fish by channel type observed during pool sampling events including sites where no fish were captured (NFC). Data points have been jittered for presentation.

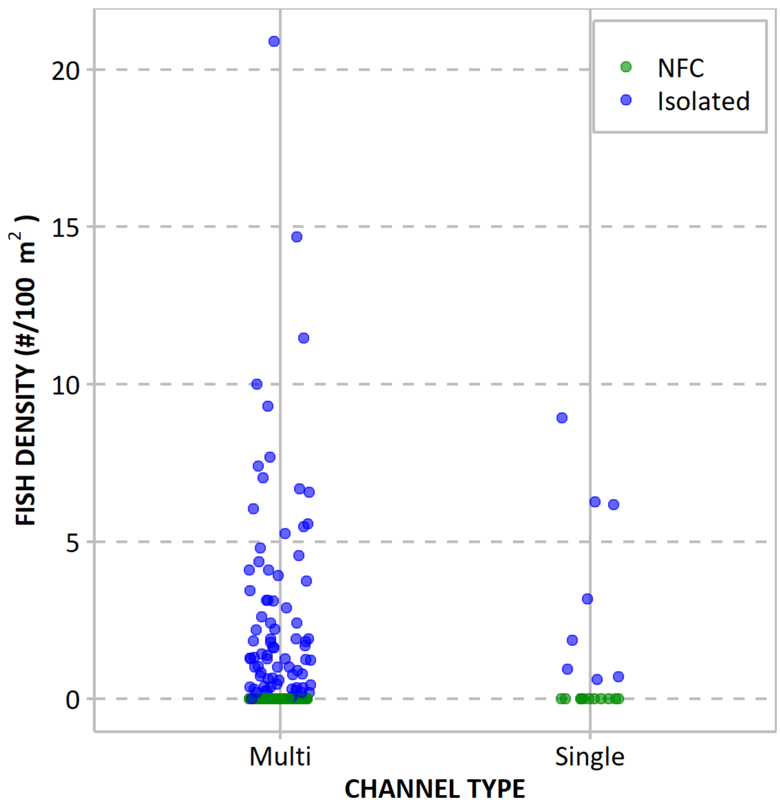


Table 14. Summary of isolated fish densities by channel type observed during pool sampling events in each year.

Year	Channel Type	Isolation Rate (fish/100 m²)				
		Total[1]	Mean	Min.	Max.	SE
2017	Multi	135	1.76	0.00	20.90	0.47
	Single	15	1.87	0.00	6.18	1.12
2018	Multi	100	1.13	0.00	9.30	0.30
	Single	22	1.12	0.00	8.92	1.12
2019	Multi	50	0.87	0.00	14.69	0.34
	Single	7	1.73	0.00	6.25	1.03
2020	Multi	98	1.48	0.00	6.04	0.54
	Single	0	0.00	n/a	n/a	n/a
All	Multi	383	1.29	0.00	20.90	0.21
	Single	44	1.51	0.00	8.92	0.61

[1] Total fish observed over entire area of searched habitat.

3.5.1.3. Effect of Stranding Risk

Isolated fish were only observed in pools in high-risk mesohabitat (mean density of 1.36 (SE 0.21) fish/100 m²; Figure 13, Table 15). Pools only formed in a single low-risk area within the study reaches in all years, accordingly all low-risk mesohabitat sampling was conducted in this location. Isolated pools form more frequently in high-risk mesohabitats, supporting focused sampling in these areas. No statistical tests of differences between high and low-risk mesohabitat were conducted because < 4% of pool sampling events were in low-risk mesohabitat.

Figure 13. Densities of isolated fish by mesohabitat stranding risk observed during pool sampling events in all years including sites where no fish were captured (NFC). Data points have been jittered for presentation.

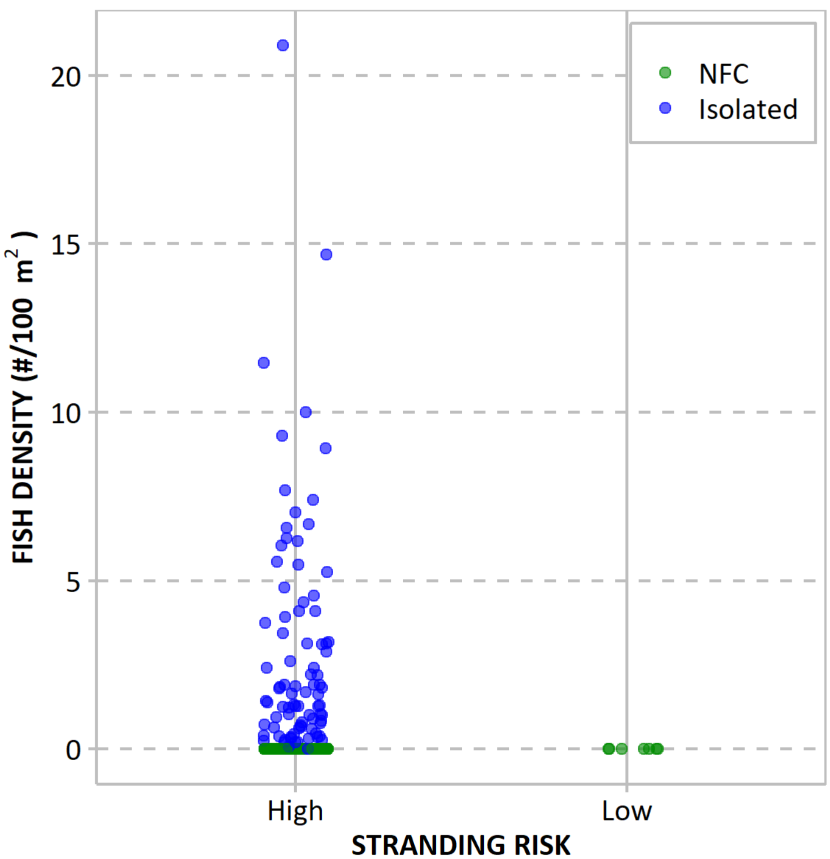


Table 15. Summary of isolated fish densities by mesohabitat stranding risk observed during pool sampling events in each year.

Year	Stranding Risk	Isolation Rate (fish/100 m²)				
		Total[1]	Mean	Min.	Max.	SE
2017	High	150	1.80	0.00	20.90	0.45
	Low	0	0.00	n/a	n/a	n/a
2018	High	122	1.26	0.00	9.30	0.32
	Low	0	0.00	n/a	n/a	n/a
2019	High	57	0.96	0.00	14.69	0.32
	Low	0	0.00	n/a	n/a	n/a
2020	High	98	1.48	0.00	6.04	0.54
	Low	0	0.00	n/a	n/a	n/a
All	High	427	1.36	0.00	20.90	0.21
	Low	0	0.00	n/a	n/a	n/a

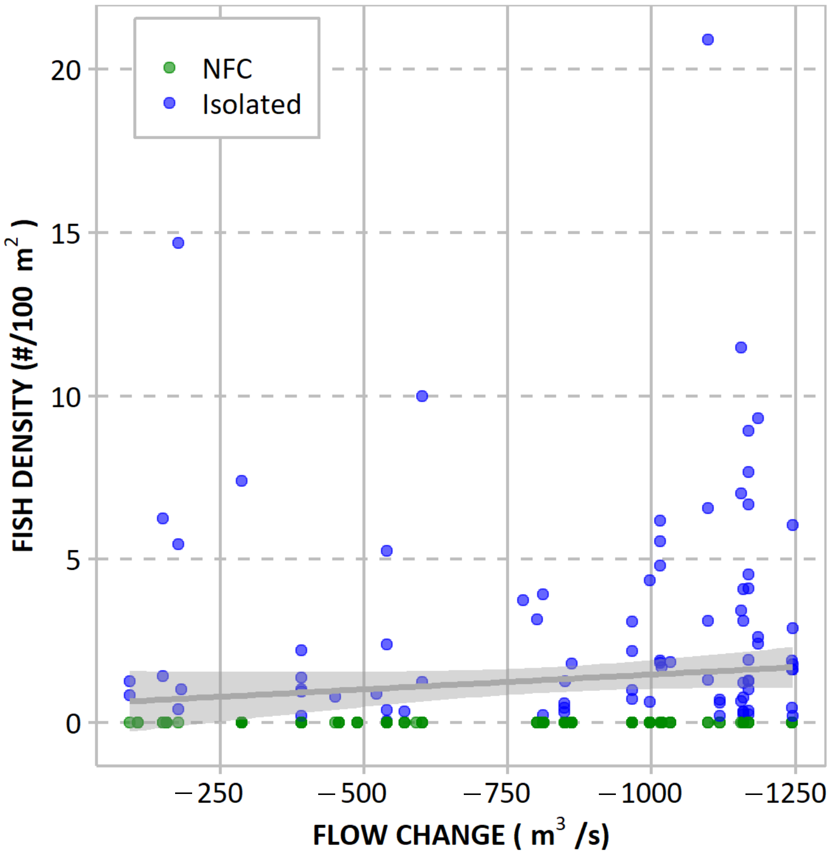
[1] Total fish observed over entire area of searched habitat.

3.5.1.4. Effect of Changes in Hydrology and Shoreline Slope

Flow Change

There was a positive relationship between the magnitude of isolated fish densities within pools and flow change measured at the nearest WSC hydrometric station (Figure 14, Table 16; K-R: Tau = 0.18, p = 0.0008, S-cor. A = 0.01; slope = 0.002).

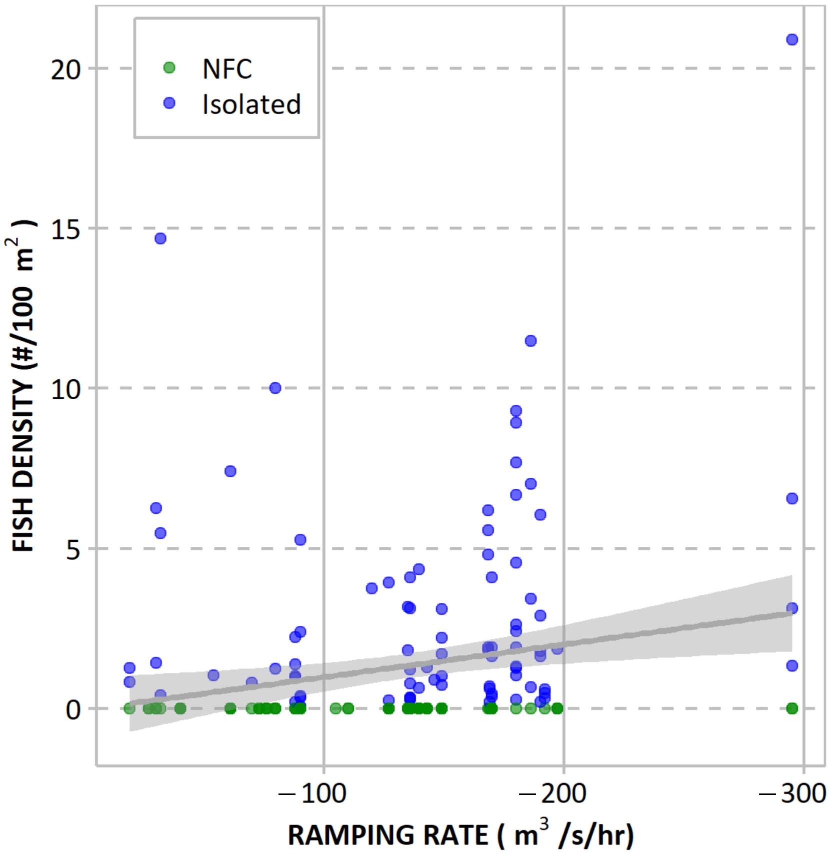
Figure 14. **Densities of isolated fish by flow change (as measured at the nearest WSC hydrometric station) observed during pool sampling events in all years including sites where no fish were captured (NFC).**



Ramping Rate

There was a positive relationship between the magnitude of isolated fish densities within pools and flow ramping rates measured at the nearest WSC hydrometric station (Figure 15, Table 16; K-W: Tau = 0.19, p = 0.0003, S-cor. α = 0.008; slope = 0.01).

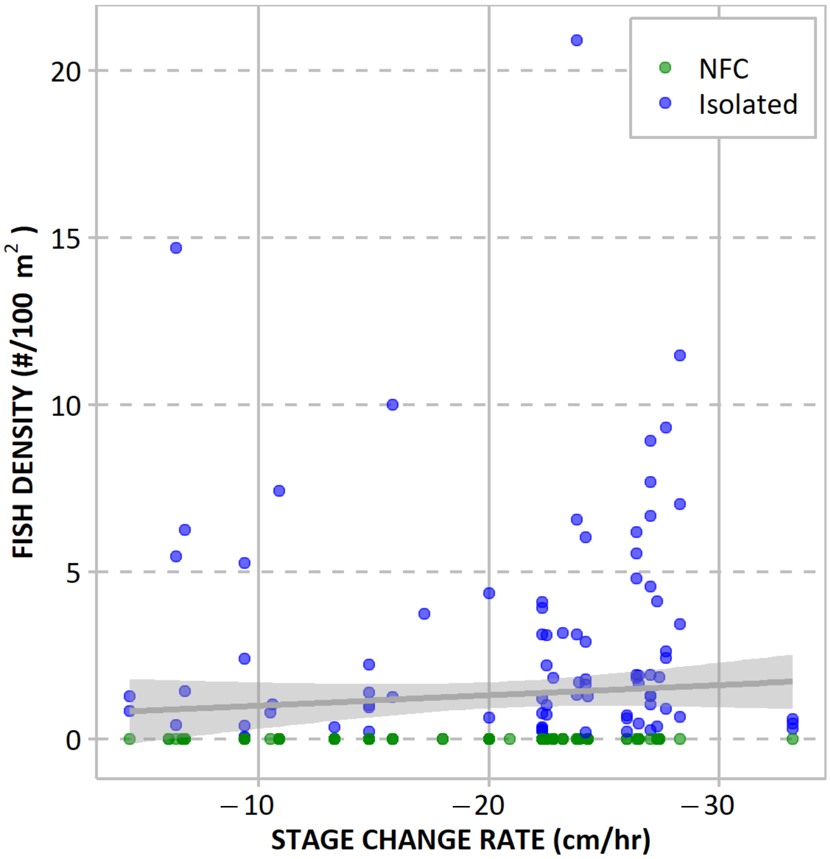
Figure 15. Densities of isolated fish by flow ramping rate (as measured at the nearest WSC hydrometric station) observed during pool sampling events in all years including sites where no fish were captured (NFC).



Stage Change Rate

There was a positive relationship between the magnitude of isolated fish densities within pools and stage change rates measured at the nearest WSC hydrometric station (Figure 16, Table 16; K-W: Tau = 0.16, p = 0.004, S-cor. α = 0.01; slope = 0.1) despite the low densities of isolated fish observed following an event with the largest stage change rate (33.2 cm/hr). Note that site-specific stage change rates were not measured. The PAP hydrometric station is at rkm 112 and the PAA hydrometric station is at rkm 164, while individual monitoring sites ranged from rkm 34 to rkm 231; stage change rates at individual monitoring sites may therefore differ due to attenuation, inflow and differences in channel morphology. Validation of the relationship between isolation rates and stage change rate would require additional site-specific hydrometric data collection which is not currently considered under the Mon-12 scope.

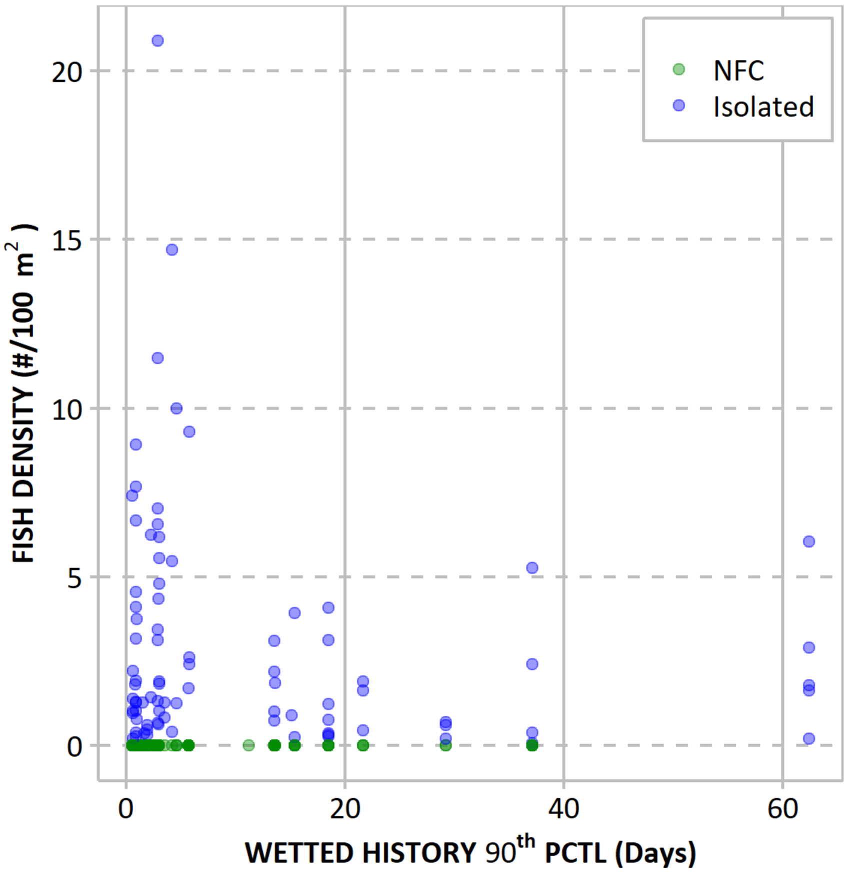
Figure 16. **Densities of isolated fish by stage change rate (as measured at the nearest WSC hydrometric station) observed during pool sampling events in all years including sites where no fish were captured (NFC).**



Wetted History

There was little evidence of a relationship between the magnitude of isolated fish densities in pools and wetted history measured at the nearest WSC hydrometric station, Figure 17, Table 16; K-R: Tau = 0.06, p = 0.27, S-cor. α = 0.02; slope = -0.01).

Figure 17. Densities of isolated fish by wetted history (as measured at the nearest WSC hydrometric station) observed during pool sampling events in all years including sites where no fish were captured (NFC).



Shoreline Slope

The shoreline slope at individual sites where pools were sampled ranged from 0 to 37.8%. The mean and maximum densities of isolated fish below the high-risk threshold of ≤ 5% were 0.99 and 11.48 fish/100 m², respectively, more than 50% lower than at sites with shoreline slopes > 5% (2.93 and 20.90 fish/100 m, respectively). This apparent positive relationship between pool isolation rate and shoreline slope was driven by a clear outlier where densities over 20.90 fish/100 m² were observed at a site with an overall shoreline slope of 15.61%, combined with the fact that the most pool sampling events (84%) occurred at sites with shoreline slopes ≤ 5%, which included many sites where no fish were captured. Expectedly, there was no statistical evidence for a relationship (Figure 18, Table 16; K-R: Tau = 0.09, p = 0.22; slope = 0.08), which likely reflects a lack of fine scale shoreline slope data for localized areas of high-risk mesohabitat within overall sites where pool sampling was conducted.

Figure 18. Densities of isolated fish by overall site shoreline slope observed during pool sampling events in all years including sites where no fish were captured (NFC).

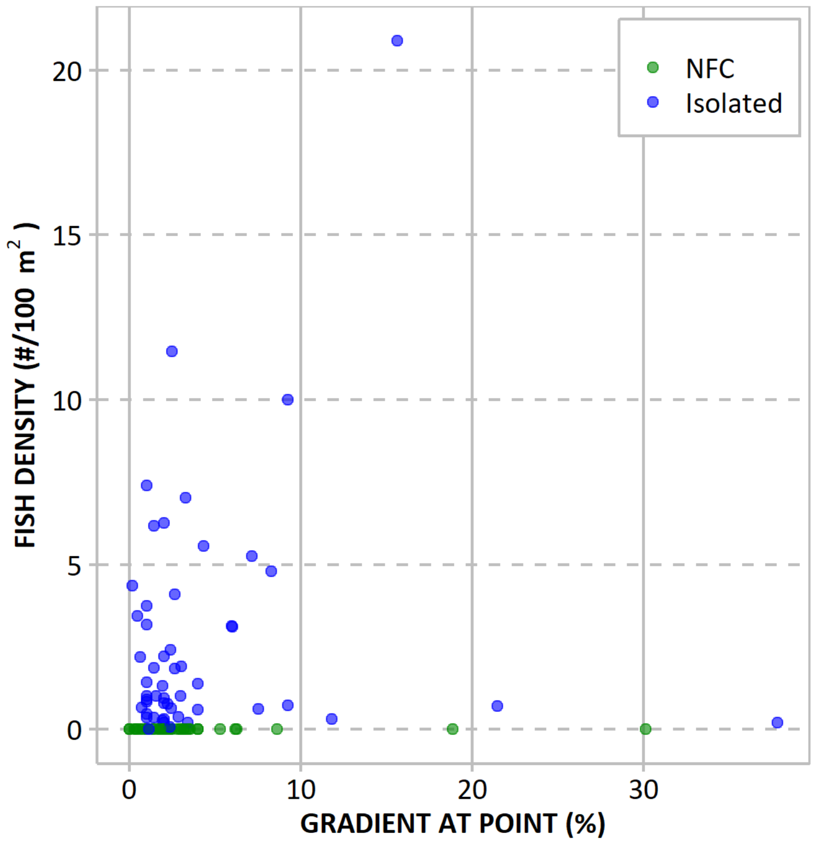


Table 16. Summary of all non-parametric tests for isolated fish observed during pool sampling events in all years.

Non-Parametric Test[1]	Explanatory Variable	Test Statistic[2]	P-Value	Simes Modified α[2]	Effect Size[3]
Kruskal-Wallis	Reach	3.13	0.37	0.03	0.02
	Channel Type	0.00	0.96	0.05	0.00
Kendall	Flow Change (m^3/s)	0.18	0.0008	0.01	0.001
	Ramping Rate ($m^3/s \times hr^{-1}$)	0.19	0.0003	0.008	0.01
	Stage Change Rate (cm/hr)	0.16	0.004	0.01	0.03
	Wetted History (Days)	0.06	0.27	0.02	-0.01
	Slope at Point (%)	0.09	0.22	n/a	0.08

[1] Kruskal-Wallis U Test used for categorical variables; Kendall Rank Correlation used for continuous variables.

[2] n/a = not applicable; slope at point tests were run on less than half of the dataset so were not included in Simes modifications.

[3] Effect size = Epsilon-Squared for Kruskal-Wallis U Test (Cohen 1988) and slope coefficient from linear regression for Kendall Rank Correlation.

3.6. Probability of Fish Stranding and Isolation

3.6.1. Probability of Stranding and Isolation Within Interstitial Habitat

The probability of stranding and isolation was consistently higher within high-risk mesohabitat in all models, and in line with results from Irvine *et al.* (2015), also increased with both wetted history and the overall magnitude of flow changes (Table 17 and Figure 19). These results were also generally in line with those from non-parametric tests presented in Section 3.4.1. These three fixed effects were the most important in explaining variation in the probability of stranding and isolation as they were the only three included in the top-ranking model (Δ AICc ≤ 3), had the highest relative variable importance, and had consistently positive relative effects across all models based on their averaged coefficients and 95% confidence intervals (CI). Stranding risk of mesohabitat had the largest relative effect among all explanatory variables (i.e., 17% and 39% greater than that of wetted history and flow change, respectively), but also had the widest variability in relative effect among models. In contrast, although the relative effect of wetted history was more modest, once monitoring trip was accounted for, it explained the most variation in combined stranding and isolation rates of all fixed effects (20%), almost two times greater than that explained by stranding risk (13%), and five time greater than that explained by flow change (4%). Similar to results from Irvine *et. al.* (2015), there was less support for an effect of ramping rate or stage change rate on the probability of stranding and isolation, as these variables had: low relative variable importance (0.08); inconsistent effects (95% CI of model averaged coefficients crossed zero); and were not included in the top-ranking interstitial model. Further, despite their high collinearity with flow change, the relative effect sizes of ramping rate and stage change rate were approximately 50% lower than that of flow change. Also, there was little support for the probability of stranding and/or isolation rates differing among reaches or channel types, with both variables absent from the top ranked model or the 95% confidence set used to derive averaged coefficients. The top candidate model explained less than half of the variation in the probability of combined interstitial fish stranding and isolation, having marginal and conditional pseudo-R^2 values (i.e., measures of goodness-of-fit) of 0.39 and 0.43, respectively. The timing of interstitial sampling appears to be important with the null model including only the random effect for trip (i.e., the unique sets of monitoring days following each single or set of ramping events across all years of baseline monitoring) explaining almost as much variation in interstitial stranding and isolation (i.e., conditional pseudo-R^2 of 0.34) as those models that included fixed effects.

Table 17. Results for model selection using AICc showing top generalized linear mixed effects models (Δ AICc ≤ 3) of the probability of fish isolation and/or stranding within interstitial habitat.

Model[1]	Parameters[2]	Log Likelihood	R^2_M[3]	R^2_C[4]	AICc	Δ AICc	Weight
P(Stranding or Isolation) ~ Flow Change + Stranding Risk + Wetted History + (1|Trip)	5	-137.5	0.39	0.43	285.2	0.00	0.70

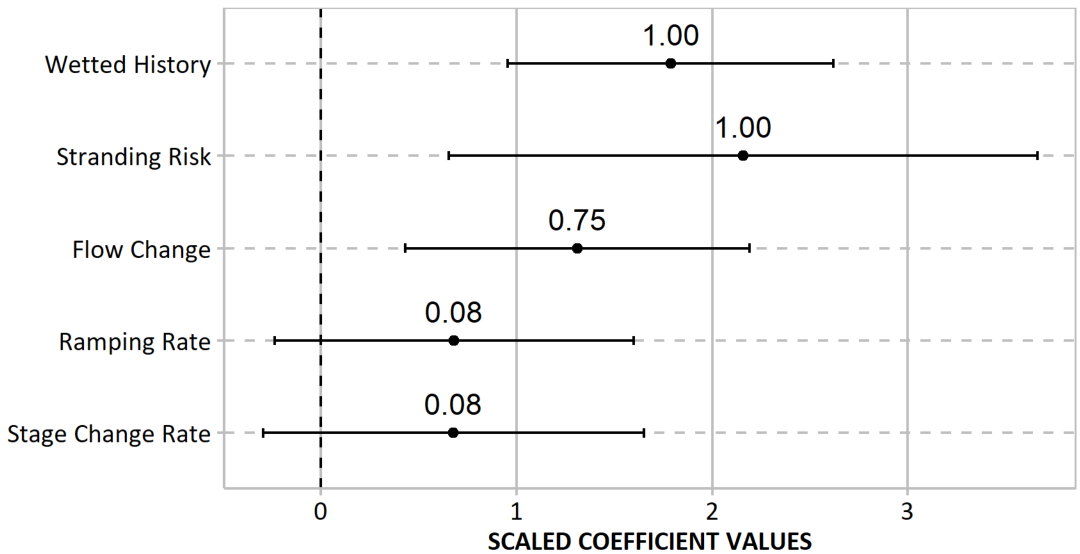
[1] P (Stranding or Isolation) = Probability of fish stranding and/or isolation (>1 fish observed) during interstitial surveys.

[2] Parameters = number of model parameters.

[3] Marginal R^2; represents the variance explained by the fixed effects (Nakagawa *et al.* 2017).

[4] Conditional R^2; interpreted as the variance explained by an entire model, including both fixed and random effects (Nakagawa *et al.* 2017).

Figure 19. Scaled fixed effects coefficient estimates with 95% unconditional CI from averaged generalized linear mixed effects models of the probability of a fish isolation and/or stranding event in interstitial habitat. Fixed effects are ordered by their relative variable importance (indicated above points) to the averaged model on a scale of 0 to 1. Values that fall to the right of the zero line indicate a positive effect and those that fall to the left indicate a negative effect. 95% CI that cross zero indicate that the influence of the fixed effect is inconsistent, varying from positive to negative among candidate models.



3.6.2. Probability of Stranding and Isolation Within Pools

Results from generalized linear mixed effects models of the probability of fish isolation within pools were less clear and differed somewhat from results for interstitial modelling (Table 18 and Figure 20). Candidate models explained little of the variation in the probability of fish isolation within pools with marginal and conditional pseudo-$R^2 \leq 0.08$ and ≤ 0.18, respectively for all top models, and less support for any one model with 11 top-ranking models (Δ AICc ≤ 3) of similarly low explanatory power and weight. Further, the relative influence of individual fixed effects varied considerably among candidate models, with wide 95% CI around model averaged coefficients, all of which crossed zero. Nevertheless, in line with interstitial model and non-parametric tests results, the probability of isolation within pools generally increased with wetted history and ramping rate, and to a lesser extent stage change rate and flow change. In line with results from other studies (e.g., Bradford 1997, Irvine *et al.* 2009, Irvine *et al.* 2015, Golder 2014a), wetted history was the strongest predictor among these fixed effects having among the most positive relative effect size, being included in the two top-most ranking models (and six of the 11 top models) and having the highest relative variable importance (0.61). In contrast to the interstitial models and results from other studies (e.g., Bradford 1997, Irvine *et al.* 2015), ramping rate was the next most important predictor variable in explaining variability in the probability of isolation in pools, being included in the highest-ranking model and four of the 11 top models, having second highest relative variable importance (0.36), and a relative effect size similar to that of wetted history. In contrast, despite their high collinearity with ramping rate (> 0.89), there was less support for a consistent effect of stage change rate or flow change with these two variables being included in only 3 and 2 lower ranking models, respectively, and having 36% to 53% lower relative variable importance and 16% to 18% lower relative effect size compared to those for ramping rate. The probability of fish isolation in pools was not different between reaches (absent in all top models) or channel types (included only in only the three lowest ranking top models). Accordingly, both reach and channel type had comparatively low relative variable importance (0.25 and 0.04, respectively) and negligible relative effect sizes close to zero. It should be noted that the results of lower ranking and relative effect of stage change rate and flow change should not be weighed too heavily as models with combinations of these variables and ramping rate were excluded from model selection and averaging to avoid issues arising from their high collinearity, and therefore, the three variables were penalized relative to other explanatory variables like wetted history and channel type. Regardless, all assessed explanatory variables were poor predictors of isolation within pools with the random effects explaining as much or more variation in the probability of isolation than all fixed effects combined. Fish isolation in pools is influenced by the timing of sampling: the null model including only the random effect of trip was a top-ranking model with higher conditional pseudo-R^2 (0.18) than any model containing fixed effects.

Table 18. Results for model selection using AICc showing top generalized linear mixed effects models (Δ AICc ≤ 3) of the probability of fish isolation within pools.

Model[1]	Parameters[2]	Log Likelihood	R^2_M[3]	R^2_C[4]	AICc	Δ AICc	Weight
P(Isolation) ~ Ramping Rate + Wetted History + (1|Trip)	4	-119.5	0.08	0.16	247.2	0.00	0.15
P(Isolation) ~ Stage Change Rate + Wetted History + (1|Trip)	4	-119.8	0.07	0.17	247.8	0.68	0.11
P(Isolation) ~ Ramping Rate + (1|Trip)	3	-121.0	0.04	0.15	248.1	0.89	0.09
P(Isolation) ~ Wetted History + (1|Trip)	3	-121.0	0.04	0.17	248.2	1.02	0.09
P(Isolation) ~ Flow Change + Wetted History + (1|Trip)	4	-120.3	0.07	0.16	248.8	1.61	0.07
P(Isolation) ~ (1|Trip)	2	-122.5	0.00	0.18	249.1	1.93	0.06
P(Isolation) ~ Stage Change Rate + (1|Trip)	3	-121.5	0.03	0.17	249.2	2.01	0.05
P(Isolation) ~ Flow Change + (1|Trip)	3	-121.5	0.03	0.15	249.2	2.02	0.05
P(Isolation) ~ Channel Type + Ramping Rate + Wetted History + (1|Trip)	5	-119.5	0.08	0.16	249.3	2.10	0.05
P(Isolation) ~ Channel Type + Stage Change Rate + Wetted History + (1|Trip)	5	-119.8	0.07	0.17	249.9	2.78	0.04
P(Isolation) ~ Channel Type + Ramping Rate + (1|Trip)	4	-121.0	0.04	0.15	250.1	2.97	0.03

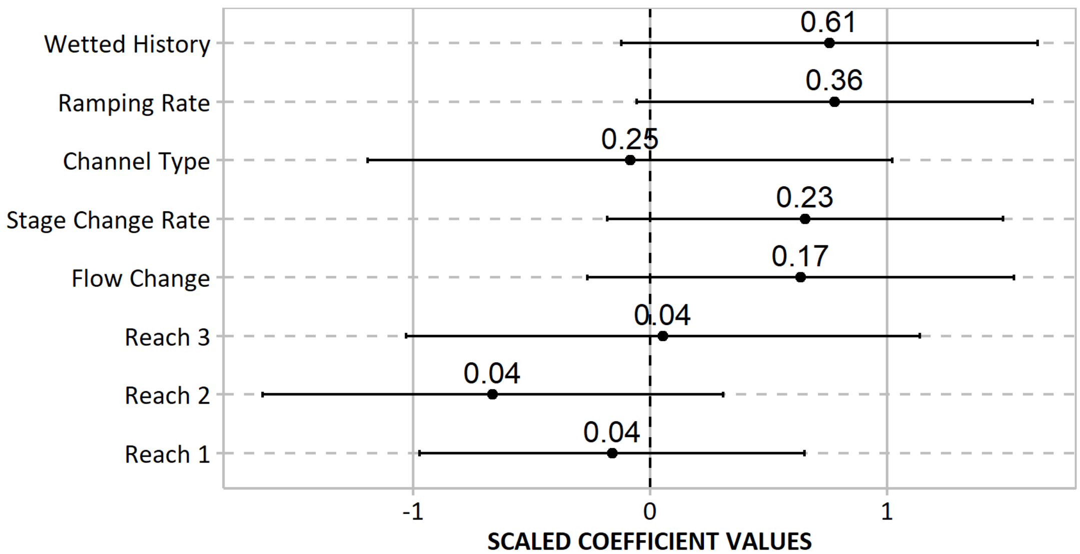
[1] P (Isolation) = Probability of fish isolation (>1 fish observed) during pool surveys.

[2] Parameters = number of model parameters.

[3] Marginal R^2; represents the variance explained by the fixed effects (Nakagawa *et al.* 2017).

[4] Conditional R^2; interpreted as the variance explained by an entire model, including both fixed and random effects (Nakagawa et al. 2017).

Figure 20. Scaled fixed effects coefficient estimates with 95% unconditional CI from averaged generalized linear mixed effects models of the probability of a fish isolation within pools. Fixed effects are ordered by their relative variable importance (indicated above points) to the averaged model on a scale of 0 to 1. Values that fall to the right of the zero line indicate a positive effect and those that fall to the left indicate a negative effect. 95% CI that cross zero indicate that the influence of the fixed effect is inconsistent, varying from positive to negative among candidate models.



4. DISCUSSION

Fish stranding was monitored in the Peace River between the future diversion headpond and the Many Islands area in Alberta from Construction Year 3 to 6 (2017 to 2020). Initiated in 2016, Mon-12 aims to quantify and compare the magnitude of fish stranding along a 139 km study reach of the Peace River during the construction and operation of the Project. Key program objectives as outlined in Section 1.2 include assessing the magnitude of fish stranding relative to baseline conditions in the future diversion headpond and the downstream Peace River (management questions Q1 and Q3, respectively), evaluating which species and life stages of fish are most affected by stranding within the future diversion headpond (Q2), and determining whether mitigation strategies are effective at reducing fish stranding (Q4) (BC Hydro 2015b). Sampling was conducted in accordance with the monitoring plan (BC Hydro 2015b), and baseline data collection for Mon-12 is now complete. Data collected to date will contribute to addressing the management questions by providing the baseline data required to test the management hypotheses, through comparisons with monitoring data collected during river diversion and Project operations. Specifically, fish species- and life stage-specific baseline data were collected, and fish stranding and isolation rates were calculated in the future diversion headpond and downstream reaches of the Peace River under baseline conditions (i.e., Q1, Q3, and Q4). These data will be compared to construction monitoring results, to determine which species and life stages are most sensitive to stranding and isolation relative to baseline conditions within the future diversion headpond (Q2). The relationships between fish stranding and isolation and predictor variables were defined, which may help to more precisely quantify the effects of river diversion and the creation of the future diversion headpond during construction, and later dam operations, on fish stranding and isolation both upstream and downstream of the Project.

The results indicated that the probability and magnitude of interstitial stranding and isolation increased with wetted history length and the magnitude of flow change measured at the nearest WSC hydrometric station and occurred mostly in high-risk mesohabitat. Similarly, the probability of fish isolation in pools increased with wetted history length, but also with the magnitude of flow ramping rates, and to a lesser extent, flow changes and stage change rates measured at the nearest WSC hydrometric station. The rates of stranding and isolation differed among study reaches and between single - and multi-thread channels in interstitial habitat but not in pools, however these variables were not significant predictors of the probability of stranding and/or isolation in either interstitial habitat or pools. These results are generally consistent with experimental studies (e.g., Bradford 1997) and similar fish stranding and isolation monitoring studies conducted by BC Hydro and others in the Duncan, Columbia, and Kootenay Rivers (e.g., Golder 2014a, 2014b, Irvine *et al.* 2009, Irvine *et al.* 2015) and other systems (Nagrodski *et al.* 2012).

A multivariate approach was used to evaluate how fish stranding and isolation differed among reaches, channel type, stranding risk of habitat, and ramping event characteristics. Top ranking models explained a modest amount of the variation in the probability of stranding and isolation within interstitial habitats, and little of the variation in the probability of fish isolation within pools but demonstrated the relative importance of predictor variables. As data become available following additional years of monitoring, these models can be refined through incorporating more comprehensive fixed and random effects structures, to better explain the factors driving stranding and isolation within the study reaches.

Sculpins, followed by suckers, Longnose Dace, and Lake Chub, made up the majority of total isolated and stranded fish across all reaches and years, with the majority of these fish being YOY or juveniles. In the future diversion headpond, 41% of the fish were isolated and 59% were stranded. Most were Slimy Sculpin (59%, of which 83% were adults) and YOY/juvenile suckers (21%), with the remaining 19% comprised of YOY/juvenile Dace, (mostly Longnose Dace), Lake Chub, Mountain Whitefish, Northern Pike, Rainbow Trout, Redside Shiners, Burbot, Northern Pikeminnow, an individual Arctic Grayling, Flathead Chub, and Kokanee, as well as three adult Redside Shiners and a single adult Longnose Sucker. The species composition reflects the relative abundance in the future diversion headpond area and shows that juvenile and small-bodied fish are at a higher risk of stranding than adults and larger-bodied species in the study reaches, consistent with the literature on stranding (Nagrodski *et al.* 2012).

The rate of fish stranding and isolation derived herein can be used as initial benchmarks to compare those observed during individual ramping events during river diversion and Project operations. However, the overall rates of fish stranding and isolation over a given period largely depends on the wetted history prior to, and frequency and magnitude of ramping events, which in turn are largely determined by operation of PCN, rather than by river diversion or Project operations. Therefore, to properly quantify Project effects, comparisons of fish stranding and isolation rates between the baseline period and either river diversion or Project operation periods must account for between period variance in flow releases from PCN.

The construction phase of Q4 will be addressed during river diversion, which commenced in the late fall of 2020 and is expected to continue into 2023 through implementation of monitoring and mitigation measures under the Mon-12 program. Fish stranding mitigations downstream of the Project, including channel enhancement and recontouring, will be assessed during operations due to commence in the fall of 2023.

5. CLOSURE

The Site C Fish Stranding Monitoring Program (Mon-12; BC Hydro 2015b) specifies the baseline monitoring frequency and reporting required for fish stranding monitoring. Together with Ecora's annual data summary reports (Ecora 2018, 2019, 2020), this synthesis review (the first) satisfies the requirements for monitoring program reporting to date. Field data collection is on track to address the primary fisheries management questions and hypotheses with the next synthesis review scheduled for Construction Year 9 (2023).

6. GLOSSARY

Term	Definition
AICc	Akaike Information Criteria corrected for small sample size.
Bankfull (length, width, depth, area)	Refers to estimated maximum pre-event length, width, and depth of isolated pool.
Broad-based Search	Visual overview of a transect of varying length and width within a section of a monitoring site to assess a large area of habitat for fish stranding and isolation.
Chi-sq.	Chi-squared.
CI	Confidence Interval.
Combined Rate	Combined density of stranded and isolated fish.
DEM	Digital elevation model.
df	Degrees freedom.
DH	Future diversion headpond.
DHFSMP	Diversion Headpond Fish Stranding Management Plan.
E-Sq.	Epsilon-Squared.
FAHMFP	Site C Fisheries and Aquatic Habitat Monitoring and Follow-up Program.
FHAP	Fisheries Habitat Assessment Procedure.
Fixed effects	Predictor variables included in generalized linear mixed effects models of stranding and/or isolation.
Future Diversion Headpond	Estimated area that will be inundated above the Project dam site during river diversion commensing in Construction Year 6 (2020) which extends approximately 18 km, from the Wilder Creek confluence downstream to the Project dam site.
Hotspot Search	Targeted excavation of substrate within defined area of highest risk habitat delineated with measuring tape.
Interstitial Searches	Searches for stranded or isolated fish in dewatered areas of substrate (see Broad-based and Hotspot Searches).
Isolation	Isolation occurs when fish become trapped in wetted areas of habitat that have become disconnected from a main waterbody.
K-R	Kendall Rank Correlation.
K-W	Kruskal-Wallis H Test.
monitoring sites	Large polygons of shoreline composed of similar high or low stranding risk habitat in which smaller areas of habitat are searched via interstitial and/or pool sampling, which may vary depending on conditions at the time (i.e., river stage and flow).
Monitoring trip	Period of 2-3 consecutive days when stranding searches were conducted following a single or two staged ramping events.
PCN	Peace Canyon Generating Station.
Pool Searches	$\geq$ 1 m², have a maximum depth of $\geq$ 5 cm, and be disconnected from the mainstem (i.e., isolated), with no evidence of consistent surface or subsurface flow; conducted in up to 3 pools per site using electrofishing.
Predictor variables	Study reach (future diversion headpond, Reach 1, 2, or 3), channel type (single- or multi-thread), stranding risk of mesohabitat (high or low), bankslope gradient, and hydrology metrics (flow change, flow ramping rate, stage change rate, wetted history).
Ramping Event	Flow reduction originating at the Peace Canyon Dam.
Random effects	Grouping variables included in generalized linear mixed effects models of stranding and/or isolation to account for pseudo-replication and improve model fit but which are not related to specific hypotheses being tested.
Random Site	New monitoring sites based on randomly selected waypoints along each stratification of mapped shoreline in each reach (high or low stranding risk).
Rate of stranding or Isolation	Linear or areal density of fish (per 100 m or 100 m2, respectively) observed and/or captured in a unit length or area during interstitial or pool searches.
Reach 1	Length of the Study Area of the Peace River from the Project dam site downstream to the Pine River confluence (16 km).
Reach 2	Length of the Study Area of the Peace River from the Pine River confluence downstream to the Alces River confluence (42 km).
Reach 3	Length of the Study Area of the Peace River from the Alces River confluence, downstream to the Many Islands area (63 km).
River2D	Two-Dimensional Depth Averaged Model of River Hydrodynamics and Fish Habitat.
Sampling Event	Defined as a set of interstitial and/or pool-based searches at an individual monitoring site following a particular targeted flow reduction event.
S-cor. α	Simes-Corrected α.
SE	Standard Error.
Stranding	Fish are considered stranded when they are found dead or are at imminent risk of death from the dewatering of wetted habitats, including within the interstitial spaces of coarse substrates.
Stranding Searches	See Interstitial searches and Pool Searches.
Study Area	Approximately 139 km of the Peace River, from the Wilder Creek confluence, downstream to the Many Islands area in Alberta.
Targeted Site	Monitoring site characterized by high stranding risk habitat based on shoreline gradients < 4% and characteristics described in Section 2.4.1.
Trip	Same as monitoring trip.
VIF	Variance Inflation Factor.
Wetted History	Duration of time habitat is wetted prior to a ramping event; characterized as 90th percentile of wetted histories for a given ramping event.
WSC	Water Survey Canada.
YOY	Young-of-year.

REFERENCES

Anderson, D.R. 2008. Model based inference in the life sciences: A primer on evidence. Springer Science, New York, N.Y.

Barton, K. 2012. R Package 'MuMIn': Multi-model inference.

Bates D., M. Mächler, B. Bolker, S. Walker. 2015. Fitting Linear Mixed-Effects Models Using lme4. Journal of Statistical Software, 67:1-48.

BC Hydro. 2015a. Fisheries and Aquatic Habitat Monitoring and Follow-up Program. Report prepared by BC Hydro for the Site C Clean Energy Project. 38 p. + Appendices.

BC Hydro. 2015b. Site C Mon-12 – Site C Fish Stranding Monitoring Program. Report prepared by BC Hydro for the Site C Clean Energy Project. 11 p.

Bell, E., S. Kramer, D. Zajanc, J. Aspittle. 2008. Salmonid fry stranding mortality associated with daily water level fluctuations in Trail Bridge Reservoir, Oregon. North American Journal of Fisheries Management 28: 1515–1528. doi: 10.1577/M07-026.1.

Bradford, M.J. 1997. An experimental study of stranding of juvenile salmonids on gravel bars and in sidechannels during rapid flow decreases. Regulated Rivers: Research & Management, 13:395-401.

Cohen, J. 1988. Statistical Power Analysis for the Behavioral Sciences, 2nd Edition. Routledge.

Collett, D. 2003. Modelling Binary Data. Chapmand & Hall/CRC: Boca Raton, Florida.

Ecora (Ecora Engineering and Resource Group Ltd.). 2017. Site C Fish Stranding Monitoring Program (Mon-12). Draft Report. Report prepared for BC Hydro and Power Authority, January 13, 2017.

Ecora (Ecora Engineering and Resource Group Ltd.). 2018. Site C Fish Stranding Monitoring Program (Mon-12) Year 2 (2017). Report prepared for BC Hydro and Power Authority, February 5, 2018.

Ecora (Ecora Engineering and Resource Group Ltd.). 2019. Site C Fish Stranding Monitoring Program (Mon-12) 2018 Data Report. Report prepared for BC Hydro and Power Authority, January 9, 2019.

Ecora (Ecora Engineering and Resource Group Ltd.). 2020. Site C Fish Stranding Monitoring Program (Mon-12) 2019 Data Report. Report prepared for BC Hydro and Power Authority, January 24, 2020.

Gelman, A. 2008. Scaling regression inputs by dividing by two standard deviations. Stat. Med. 23: 2865 - 2873.

Golder Associates Ltd. and Poisson Consulting Ltd. 2010a. Columbia and Kootenay River Fish Stranding Protocol Review: Literature Review and Fish Stranding Database Analysis. 34 p + appendices.

Golder Associates Ltd. and Poisson Consulting Ltd. 2010b. Duncan Dam Project Water Use Plan. Lower Duncan River. Implementation Year 2. Reference: DDMMON#15. Lower Duncan River Stranding Protocol Development and Finalization. Study Period: 2010. 47 p.

Golder Associates Ltd. 2011. Canadian Lower Columbia River: Fish Stranding Risk Assessment and Response Strategy, Report prepared for BC Hydro, Columbia Power Corporation, Fortis BC, Columbia operations Fish Advisory Committee (COFAC) and Canal Plant Agreement Operating Committee, Golder Report No. 09-1480-0055F: 31 p. + 4 appendices.

Golder (Golder Associates Ltd.). 2014a. Lower Columbia River [CLBMON #42(A)] and Kootenay River Fish Stranding Assessments: Annual summary (April 2013 to April 2014). Report prepared for BC Hydro, Columbia Power Corporation, and FortisBC, Castlegar, BC. Golder Report No. 10-1492-0042 and 10-1492-0100.

Golder (Golder Associates Ltd.). 2014b. DDMMON-16 Lower Duncan River fish stranding impact monitoring: Year 6 report (April 2013 to April 2014). Report prepared for BC Hydro, Castlegar, BC. Golder Report No. 12-1492-0117F.

Golder (Golder Associates Ltd.). 2017. DDMMON-16 Lower Duncan River fish stranding impact monitoring: Year 8 report (April 2015 to April 2016). Report prepared for BC Hydro, Castlegar, BC. Golder Report No. 1535517D: 69 p. + 3 app.

Golder (Golder Associates Ltd.) 2018. DDMMON-16 Lower Duncan River fish stranding impact monitoring: Year 9 Report (April 2016 to April 2017). Report prepared for BC Hydro, Castlegar, BC. Golder Report No. 1535517F: 54 p. + 3 app.

Golder (Golder Associates Ltd.) and W.J. Gazey Research. 2018. Peace River Large Fish Indexing Survey - 2017 Investigations. Report prepared for BC Hydro, Vancouver, British Columbia. Golder Report No. 1670320: 118 pages + 8 appendices.

Grueber, C. E., S. Nakagawa, R.J. Laws, and I.G. Jamieson. 2011. Multimodel inference in ecology and evolution: Challenges and solutions. J. Evol. Biol. 24:699-711.

Hipel, K.W. and A.I. McLeod. 1994. Time Series Modelling of Water Resources and Environmental Systems. Elsevier Science, New York, N.Y.

Hollander, M. and D.A. Wolfe. 1973. Nonparametric Statistical Methods. John Wiley & Sons, New York, N.Y.

Irvine, R.L., T. Oussoren, J.S. Baxter, and D.C. Schmidt. 2009. The effects of flow reduction rates on fish stranding in British Columbia, Canada. River Research and Applications 25:405-415.

Irvine, R.L., J.L. Thorley, R. Westcott, D. Schmidt, and D. DeRosa. 2015. Why do fish strand? An analysis of ten years of flow reduction monitoring data from the Columbia and Kootenay Rivers, Canada. River Research and Applications 31:1242–1250.

Kloke, J. and J. McKean. 2014. Nonparametric Statistical Methods Using R. Chapman & Hall, New York, N.Y.

Lewis, A., C. Zyla., and P. Gibeau. 2011. Flow Ramping Guidelines for Hydroelectric Projects: Developing, Testing, and Compliance Monitoring, Draft V1. Consultant's report prepared by Ecofish Research Ltd for Clean Energy BC, Department of Fisheries and Oceans Canada, and the BC Ministry of Environment.

Lewis, A., A. Harwood, C. Zyla, K. Ganshorn, and T. Hatfield. 2013. Long term Aquatic Monitoring Protocols for New and Upgraded Hydroelectric Projects. DFO Can. Sci. Advis. Sec. Res. Doc. 2012/166. ix + 88 p.

Mainstream (Mainstream Aquatics Ltd.). 2009. Site C fisheries studies – Juvenile fish use and habitat inventory of Peace River tributaries in summer 2008. Prepared for BC Hydro. Report No. 08008CF: 78 p. + Appendices.

Mainstream (Mainstream Aquatics Ltd.). 2010. Site C fisheries studies – Halfway River and Moberly River fall mountain whitefish migration and spawning study 2009. Prepared for BC Hydro. Report No. 09008CF: 31 p. + appendices.

Mainstream (Mainstream Aquatics Ltd.). 2011. Site C fisheries studies – 2010 Peace River Fish Inventory. Prepared for B.C. Hydro Site C Project, Corporate Affairs Report No. 10005F: 102 p. + plates and appendices.

Mainstream (Mainstream Aquatics Ltd.). 2012. Site C Clean Energy Project – Fish and fish habitat technical data report. Prepared for BC Hydro. Report No. 12002F: 269 p.

Mainstream (Mainstream Aquatics Ltd.). 2013. Site C Fisheries Studies – Peace River Habitat Assessment. Mainstream Aquatics Ltd., M. Miles and Associates Ltd, Integrated Mapping Technologies Inc.

McLeod, A.I., 2011. R Package 'Kendall': Kendall rank correlation and Mann-Kendall trend test.

McPhail, J.D. 2007. The Freshwater Fishes of British Columbia. University of Alberta Press.

McPhail, J.D. and R. Carveth. 1993. Field Key to the Freshwater Fishes of British Columbia. Resources Inventory Committee (Canada).

Nagrodski, A., Raby, G. D., Hasler, C. T., Taylor, M. K., & Cooke, S. J. 2012. Fish stranding in freshwater systems: sources, consequences, and mitigation. *Journal of Environmental Management 103*:133-141.

Nakagawa, S., P.C.D. Johnson, and H. Schielzeth. 2017. The coefficient of determination R^2 and intra-class correlation coefficient from generalized linear mixed-effects models revisited and expanded. J. R. Soc. Interface. 14:20170213.

NHC (Northwest Hydraulic Consultants). 2012. Peace River Hydraulic Habitat Study (Contract Q9-9105). Prepared for BC Hydro. Report No. 09005D: 65 p.

NHC (Northwest Hydraulic Consultants). 2013. BC Hydro Site C Clean Energy Project: Fish Habitat Mitigation - Peace River Channel Restoration Opportunities Downstream of Site C. Prepared for BC Hydro, March 2014.

Nicholl, S. and A. Lewis. 2016. Site C Fish Stranding Monitoring Methods. Consultant's memorandum prepared for BC Hydro by Ecofish Research Ltd. October 27, 2016.

NRC (National Research Council Canada). 2017. Blue Kenue™: software tool for hydraulic modellers. Available online at: https://nrc.canada.ca/en/research-development/products-services/software-applications/blue-kenuetm-software-tool-hydraulic-modellers Accessed on June 7, 2021.

Piñeiro, G., S. Perelman, J.P. Guerschman, and J.M. Paruelo. 2008. How to evaluate models: observed vs. predicted or predicted vs. observed? Ecol. Model. 216:316-322.

R Development Core Team. 2020. R: a language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing.

Sen, P.K. (1968), Estimates of the regression coefficient based on Kendall's tau, Journal of the American Statistical Association 63, 1379–1389.

Simes, R.J. 1986. An Improved Bonferroni Procedure for Multiple Tests of Significance. Biometrika. 73:751-754.

Steffler, P. and J. Blackburn. 2002. River2D, Two-Dimensional Depth Averaged Model of River Hydrodynamics and Fish Habitat, Introduction to Depth Averaged Modeling and User's Manual.

Triton (Triton Environmental Consultants Ltd.). 2009. Middle Columbia River Juvenile Fish Stranding Assessment (Year 2009) Reference: CLBMON#53 Columbia River Water Use Plan Monitoring Program: Middle Columbia River Juvenile Fish Stranding Assessment Study Period: 2009-2010. 44 p. + appendices.

Zuur. A.F., E.N. Ieno, N.J. Walker, A.A. Saveliev, and G.M. Smith. 2008. Mixed Effects Models and Extensions in Ecology with R. Springer Science, New York, N.Y.