Duncan Dam Project Water Use Plan

Adaptive Stranding Protocol Development Program

Implementation Year 13

Reference: DDMMON-16

Lower Duncan River: Fish Stranding Impact Monitoring: Year 13 Interpretive Report

Study Period: Years 1 to 13 – February 2008 to December 2020

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23 December 2021

REPORT

DDMMON-16: Lower Duncan River

Lower Duncan River Fish Stranding Impact Monitoring: Years 1 to 13 (February 2008 to December 2020) Synthesis Report

Submitted to:

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Cover Photo: Upstream view of Site S3.5-4.0R, 11 April 2020.

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Executive Summary

Although natural flow fluctuations from unregulated tributaries are known to cause fish stranding, fish stranding in the lower Duncan River (LDR) can be exacerbated by Duncan Dam (DDM) operations that influence the frequency and magnitude of flow fluctuations. The Lower Duncan River Fish Stranding Impact Monitoring Program (DDMMON-16), initiated under BC Hydro Water License Requirements (WLR), assesses Rainbow Trout (*Oncorhynchus mykiss*) and Mountain Whitefish (*Prosopium williamsoni*) population level impacts associated with dam operations.

The results from this 13 year (2008-2020) WLR monitoring program will help inform flow management decisions that may impact fish stranding in the LDR. Based on the findings of this program, the flow reduction measures implemented under the Water Use Plan (WUP) are effective at reducing fish stranding. When possible, flow reductions at DDM follow recommendations made by the Lower Duncan River Stranding Protocol Development and Finalization Program (DDMMON-15). DDM operations could increase the risk of stranding in certain seasons (Spring and Fall) and during flow reductions of high magnitude and larger/faster changes to river stage. Based on data collected up to October 2020, documented stranding rates of juvenile Mountain Whitefish are very low and are not believed to result in population level effects. The total stranding estimates for juvenile Rainbow Trout ranged between 1.7% and 10.1% of the estimated total population in all study years, with upper confidence/credible limits under 9% in most years.

This program also conducted proportional stranding analysis to allow for comparisons with the probability of stranding analysis of the DDMMON-1 Lower Duncan River Ramping Rate Study. This allowed to further address Management Question 1 of that program:

What is the relationship between fish stranding and:

- *Rate of river stage/total stage change* – the findings of the DDMMON-16 Program corroborate the conclusion of the DDMMON-1 Program (larger ramping rates lead to increased risk of stranding). Predicted proportional stranding also indicates larger total reduction magnitudes lead to increased risk of stranding.
- *Time of day (day/night)* – no evidence was found to refute the findings of the DDMMON-1 Program (risk to strand juvenile and small bodied fish was higher at night).
- *Substrate* – the findings of the DDMMON-16 Program corroborate the conclusion of the DDMMON-1 Program (link between larger substrates and higher stranding rates).
- *Habitat configuration (channel bed gradient and topography)* – the findings of the DDMMON-16 program indicate there is a relationship, although not statistically significant, between channel gradient and interstitial stranding rates. There were no statistical differences between stranding rates among habitat configurations (side channel, mainstem bar).
- *Cover* – no evidence was found to refute the findings of the DDMMON-1 Program (increased cover availability is linked to increased stranding).
- *Species* – differences in stranding rates between the target species of DDMMON-16 violate the assumption that DDM operations pose an equal risk to stranding for all species, as previously stated in the DDMMON-1 Program.

- *Time of year (spring, fall, winter)* – the findings of DDMMON-16 indicate that fall stranding rates for Rainbow Trout are significantly higher than in the winter/spring season.
- *Habitat stability (wetted history)* – a relationship between wetted history and fish stranding was not found. Therefore, the DDMMON-16 Program did not find evidence to support the DDMMON-1 finding of a non-significant trend that increased wetted history was linked to increased stranding risk.

This report presents the results from Years 1 to 13 of the DDMMON-16 program, and the status of management questions for DDMMON-16 is provided in Table EI.

Table EI: DDMMON-16 Year 13: Status of Management Questions and Objectives.

Objective	DDMMON-16 Management Question	DDMMON-16 Specific Hypothesis	DDMMON-16 Year 13 (February 2008 – December 2020) Status Summary
1) To determine the effectiveness of the operation measures implemented as part of the ASPD program	1) How effective are the operating measures implemented as part of the Adaptive Stranding Protocol Development (ASPD) program?	N/A	- Based on the current state of knowledge, the flow reduction measures implemented under the WUP are effective at reducing fish stranding. - When feasible, flow reductions at DDM should follow recommendations made by the Lower Duncan River Stranding Protocol Development and Finalization Program (DDMMON-15). - There was no relationship between wetted history and fish stranding. This program did not find evidence that would warrant addressing this issue in the Adaptive Stranding Protocol Development Program (ASPD).
2) To determine the levels of impact to resident fish populations associated with fish stranding events on the lower Duncan River.	2) What are the levels of impact to resident fish populations associated with fish stranding events on the lower Duncan River?	*Ho_1: Fish stranding observed at index sites along the lower Duncan River floodplain is representative of overall stranding.*	- Index sites were not originally selected to be representative of the entire LDR but were selected to focus on sites believed to have the highest frequency of stranding based on the spatial extent of dewatered area and suitability of habitat. - Index sites tend to be of lower gradient and wider than random sites, therefore more area dewaters at these sites. - There was no significant site effect on the formation of pools (density) and pool stranding rates. - The low number of interstitial stranding datapoints precluded the examination of the effect of site on interstitial stranding. - Based on the current state of knowledge, Hypothesis H01 is accepted.

Objective	DDMMON-16 Management Question	DDMMON-16 Specific Hypothesis	DDMMON-16 Year 13 (February 2008 – December 2020) Status Summary
2) To determine the levels of impact to resident fish populations associated with fish stranding events on the lower Duncan River.	2) What are the levels of impact to resident fish populations associated with fish stranding events on the lower Duncan River?	*Ho_2: Fish populations in the lower Duncan River are not significantly impacted by fish stranding events.*	- Estimates for the number of juvenile Rainbow Trout stranded in pools and interstitially were relatively low, with high precision. - There was a seasonal effect on Rainbow Trout stranding, with stranding rates approximately seven times higher in the fall than in the winter/spring season. This appears to be due to lower juvenile fish densities in the system in the winter/spring versus the fall and a decreased risk of stranding in that period. - Based on the current dataset, hypothesis H_{02} is rejected for Rainbow Trout. Annual total percent stranding estimates for three study years (2014, 2015, and 2018) were approximately 10%, which may significantly impact the Lower Duncan River Rainbow Trout population. - Mountain Whitefish encounters were minimal in all study years. This consistently low level of stranding was not considered significant and will likely not result in a population level effect. Based on the current dataset, study hypothesis H_{02} is accepted for Mountain Whitefish. - Within the current dataset, there were no relationships between the number of fish stranded in pools and slope of substrate. - There was a relationship, albeit not statistically significant, between slope and the number of interstitially stranded fish (majority of fish strandings were on substrates with gradients ≤9%).

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Table of Contents

TABLES

FIGURES

APPENDICES

1.0 INTRODUCTION

1.1 Background

The lower Duncan River (LDR) originates from Duncan Dam (DDM) and flows for approximately 11 km before entering the north end of Kootenay Lake. Below DDM, the river flows through a man-made channel for 1 km to the confluence with the Lardeau River. Downstream from the confluence, the LDR is composed of a series of single and braided channel sections with continually changing morphology that includes debris jams, bars, and islands. Although natural flow fluctuations from unregulated rivers are known to cause fish stranding, fish stranding in the LDR can be exacerbated from DDM operations (Golder 2002) by influencing the frequency and magnitude of flow fluctuations. Formal assessments of fish stranding impacts related to changes in operations at DDM began in the fall of 2002. In 2004, BC Hydro developed a fish stranding assessment protocol that included communication protocols, recommended flow reduction rates, and fish stranding assessment methodologies (BC Hydro 2004). An assessment of fish stranding impacts on the LDR related to DDM operations from November 2002 to March 2006 was previously completed (Golder 2006). In 2008, an annual summary of DDM related stranding events was completed (Golder 2008).

The current program, the Lower Duncan River Fish Stranding Impact Monitoring Program (DDMMON-16), was initiated under the BC Hydro Water License Requirements (WLR) Program. One of the main objectives of the WLR Program is to evaluate the effectiveness of the operating regime defined in the Water Use Plan (WUP) and to identify opportunities to improve dam operations to maximize fish abundance and diversity in the Duncan River Watershed in consideration of other values. This involves assessing the influence of flow reductions on migrating, resident and/or rearing fish populations in the LDR. As a result of several uncertainties in WUP assumptions, the Adaptive Stranding Protocol Development Program (ASPD) was developed to address the impacts of flow reductions on fish. This adaptive management program was implemented over the WUP review period based on the results from a collective group of monitoring studies. The DDM water license and ASPD requires a minimum average daily flow from DDM of 3 m^3 s^{-1} (160 ft^3 s^{-1}) and has seasonal targets for discharge, based on Columbia River Treaty discharge requirements. The water license also requires that a minimum flow of 73 m^3 s^{-1} (2578 ft^3 s^{-1}) be maintained in the LDR at the Lardeau River Water Survey of Canada (WSC) gauging station (DRL). In addition, the maximum hourly flow reduction (ramping rate) allowed under the WUP is 28 m^3 s^{-1} (989 ft^3/s), and the maximum daily flow change allowed is 113 m^3 s^{-1} (3,991 ft^3 s^{-1}). All LDR water license discharge requirements are subject to available inflows into Duncan Reservoir and are dependent on tributary inflows.

One component of the broader ASPD is DDMMON-16, which in conjunction with other assessment tools being developed during the WUP review period, assesses Rainbow Trout (*Oncorhynchus mykiss*) and Mountain Whitefish (*Prosopium williamsoni*) population level impacts associated with dam operations. The information generated by these assessments will ultimately form the rationale for the implementation of a final operating protocol for DDM discharge releases that minimizes impacts on fish. The DDMMON-16 program conducted in Years 1 to 13 (2008 – 2020) built on historic methodologies, expanded the program's datasets, updated the boundaries of identified sites where stranding occurs, and analyzed pre- and post-WUP DDM operations and how they relate to fish stranding. This monitoring program was also created to develop and refine LDR stranding estimates that can be used to determine population level impacts. To accomplish this objective, extrapolation of fish stranding rates for the entire length of the river using information from BC Hydro's LDR Hydraulic Model (DDMMON-3) and other interrelated studies (Lower Duncan River Ramping Rate Monitoring [DDMMON-1], Lower Duncan River Habitat Use Monitoring [DDMMON-2], Lower Duncan River Kokanee Spawning Monitoring [DDMMON-4], and Lower Duncan River Stranding Protocol Review [DDMMON-15]) was conducted.

These extrapolated stranding rates are then compared to fish abundance estimates obtained as part of this and other study programs.

1.2 Objectives, Management Questions, and Hypotheses

As stated in the Lower Duncan River Water Use Plan Terms of Reference (BC Hydro 2008), the overall management question to be addressed within the ASPD program is as follows:

What are the best operating strategies at Duncan Dam to reduce fish stranding in the lower Duncan River?

The specific management questions associated with DDMMON-16 are as follows:

1. *How effective are the operating measures implemented as part of the ASPD program?*
2. *What are the levels of impact to resident fish populations associated with fish stranding events on the lower Duncan River?*

To address the specific management questions associated with this monitoring program, the primary objectives of DDMMON-16 are as follows:

1) To determine the effectiveness of the operating measures implemented as part of the ASPD program.
2) To determine the levels of impact to resident fish populations associated with fish stranding events on the lower Duncan River.

These objectives directly reflect the uncertainties facing the DDM WUP Consultative Committee when making decisions regarding BC Hydro operations on the LDR. It is anticipated that by addressing these objectives, an understanding of fish stranding impacts and the potential for making operating/monitoring improvements at DDM can be applied in the future. The Terms of Reference did not state specific hypotheses to address Objective 1. Therefore, Objective 1 was addressed by assessing DDM operations in relation to stranding variables (Golder and Poisson 2012) within direct management control.

To address Objective 2, the TOR stated two hypotheses that DDMMON-16 must test, which are related to the assumptions to be used in the monitoring program. The specific hypotheses that are addressed in this report as part of the second objective are as follows:

Ho_1: Fish stranding observed at index sites along the lower Duncan River floodplain is representative of overall stranding.

Ho_2: Fish populations in the lower Duncan River are not significantly impacted by fish stranding events.

The investigations during Year 1 (2008–2009) and Year 2 (2009–2010) of DDMMON-16 addressed Objective 1, "*the effectiveness of operating measures*", and Hypothesis Ho_1*, "fish stranding at index sites is representative of overall stranding*" (Golder 2009a, 2010). Sampling efforts focused on monitoring and calibrating fish stranding impacts associated with DDM flow reduction within the LDR from the Duncan/Lardeau confluence downstream to Kootenay Lake under different temporal variations and variable ramping rates. Sampling and analysis methodologies were instituted in Year 4 to further refine our understanding of Hypothesis Ho_1.

Objective 2, "*to empirically assess the influence of stranding events on resident and/or rearing fish population levels in the LDR*", was the focus of Year 3 (2010–2011), Year 4 (2011–2012), Year 5 (2012–2013), Year 6 (2013–2014), Year 7 (2014–2015), Year 8 (2015–2016), Year 9 (2016–2017), Year 10 (2017–2018),

Year 11 (2018–2019), Year 12 (2019–2020) and the present study year (Year 13: April to December 2020) of DDMMON-16 (Golder 2011, 2014, 2015, 2017a, 2017b, 2018; Golder and Poisson 2012, 2019a, 2019b and 2020).

1.3 Report Scope

As outlined in the Terms of Reference (BC Hydro 2008), the species of interest for this program are Rainbow Trout and Mountain Whitefish. This report provides information on abundance estimation and fish stranding observed for these species over all assessed flow reductions in Years 1 to 13 of DDMMON-16 (2008 to 2020). This report also presents detailed statistical analysis in relation to the multi-year program objectives and incorporates several aspects of the DDMMON-3 TELEMAC-2D hydraulic model, including the Digital Elevation Model (DEM; NHC 2013). Lastly, the analysis from the current program was compared to the conclusions of the DDMMON-1 to further address the outstanding management questions from that program.

2.0 METHODS

The following description of sampling methodologies and data analysis milestones during the DDMMON-16 Program are provided as summaries in this synthesis report. In-depth descriptions of sampling and data analysis methodologies employed in each Study Year can be found in the corresponding annual reports (Golder 2009a, 2010, 2011, 2014, 2015, 2017a, 2017b, 2018; Golder and Poisson 2012, 2019a, 2019b and 2020).

2.1 Study Area

The geographic scope of the study area for DDMMON-16 included the 11 km of mainstem LDR from DDM to the mouth of Kootenay Lake (Figure 1). This study area (collectively known as the LDR) includes the Duncan-Lardeau rivers confluence, as well as the Meadow, Hamill and Cooper creek mouths. For the purpose of all WLR studies, the mainstem Duncan River was divided into five reaches:

1) Reach 1 (River Km [RKm] 0.0 - at DDM spill gates to RKm 0.8)
2) Reach 2 (RKm 0.8 to RKm 2.6)
3) Reach 3 (RKm 2.6 to RKm 5.7)
4) Reach 4 (RKm 5.7 to RKm 6.7)
5) Reach 5 (RKm 6.7 to RKm 11.0 – at the mouth to Kootenay Lake)

For the purpose of the DDMMON-16 Program, 49 potential fish stranding sites were identified based on previous studies (AMEC 2004; Golder 2006, 2008, 2009a, 2010, 2011, 2014, 2015, 2017a, 2017b, 2018; Golder and Poisson 2012, 2019a, 2019b and 2020). These stranding sites included 11 index stranding assessment sites and 38 random stranding assessment sites (Appendix A, Figures A1 to A7). Sites were named based on the type of habitat they encompass (mainstem or side channel), approximate distance downstream from Duncan Dam (Rkm), and which downstream bank they were located on (right downstream bank, left downstream bank. For example, index site M0.8R is located on mainstem Duncan River habitat, is approximately 0.8 RKm downstream of Duncan Dam and is located along the right downstream bank.

Original identified stranding site boundaries were established using orthophotography taken in 2008 and 2009 (provided by BC Hydro). As LDR channel movement frequently occurs, sites may change over time due to erosion or sediment deposition. Therefore, the site boundaries were continually adjusted over the course of this program to reflect changes observed in the field. Updated orthophotography (from 2012 and 2018; provided by BC Hydro) also informed refinements to stranding site boundaries.

Based on the findings of the DDMMON-16 Program, the following 11 sites were removed from the list of 49 potential stranding sites identified during this program.

- SLARD0.3R
- M0.6-1.7L
- M1.1-1.7R
- S2.7L
- S4.0R
- S4.1R
- M6.0R
- M7.2-7.8R
- S7.7R
- M8.4-9.1R
- S11.5R

The risk to strand fish at these sites during DDM operations was negligible due to factors such as high gradient (over 20%), very low amounts of dewatered area during flow reductions, and the attenuation of flow reductions because of backwater from Kootenay Lake water levels. Habitats situated outside of the identified sites typically have steep banks with fine substrates. Habitats with these characteristics have very low stranding risk. Consequently, additional major fish strandings at locations outside of the potential fish stranding sites used in this study, are unlikely to occur.

2.2 Study Design and Rationale

Golder has conducted fish stranding assessments on the LDR between 2002 and 2020. A wide variety of fish capture/observation techniques were utilized to ensure the study design during each sample year met BC Hydro's objectives. Recommendations made during annual data analysis and reporting included changes to the study design to address gaps in the dataset identified during data analysis (Golder 2011, Golder 2014, 2015, Golder 2017a, 2017b, Golder 2018, Golder and Poisson 2012, 2019, 2019b and 2020).

The datasets from the current program were analyzed and compared to the conclusions of the DDMMON-1 Program to further address the outstanding management questions from that program. DDMMON-1 was designed to estimate the probability of stranding while the current program was designed to estimate the number of fish stranded. To allow for comparisons between the DDMMON-1 and DDMMON-16 Programs, the model used to estimate the number of fish stranded was updated to use information on site-level fish abundance to estimate the probability/proportion of stranding from the number of fish stranded. This approach allowed updated estimates of the effect of the following factors on the proportion of fish stranding:

- Day of the Year;
- Total LDR stage change (total flow reduction magnitude);
- Initial LDR Discharge before flow reductions (measured at the DRL);
- Rate of river stage change (ramping rate); and
- Habitat stability (wetted history).

As part of the DDMMON-15 program, a workshop was held 14 January 2016 which was attended by Lower Duncan River WUP study leads, BC Hydro personnel, and Ministry of Forests, Lands and Natural Resource Operations representatives. One of the topics discussed during the workshop was shifting the DDMMON-16 program from its initial goal of examining the impact of fish stranding on target fish species populations to a program focused on long term monitoring and salvage operations. In Study Years 11 to 13 (2018 – 2020), modifications were made to sampling methodologies and the Lower Duncan River Stranding Database (Section 2.9) to facilitate this shift in focus.

2.3 Sampling Chronology

To accommodate annual deliverable deadlines and for the purposes of data analysis, each study year except for Year 13, was commenced on 15 April and concluded on 14 April of the following year. As the DDMMON-16 Program ended at the cessation of the 2020 calendar year, Study Year 13 was set from 15 April 2020 to 31 December 2020.

Each assessed reduction from DDM was assigned a reduction event number (RE; see Section 2.5) and Table 1 outlines all assessment activities conducted during the DDMMON-16 Program. An in-depth summary of the chronology of sampling and project milestones in all study years is provided in Appendix A, Tables A1 to A26.

Table 1: Sampling activities for all years of the DDMMON-16 Lower Duncan River Fish Stranding Impact Monitoring Program.

DDMMON-16 Study Year	Fish Stranding Assessments	Calibration Surveys	Fall Abundance Snorkel Surveys
1 (2008-2009)	Yes	No	No
2 (2009-2010)	Yes	Yes	No
3 (2010-2011)	Yes	No	No
4 (2011-2012)	Yes	No	No
5 (2012-2013)	Yes	No	No
6 (2013-2014)	Yes	No	Yes
7 (2014-2015)	Yes	No	Yes
8 (2015-2016)	Yes	No	Yes
9 (2016-2017)	Yes	No	Yes
10 (2017-2018)	Yes	No	No
11 (2018-2019)	Yes	No	No
12 (2019-2020)	Yes	No	No
13 (2020)	Yes	No	No

LEGEND

- WSC GAUGE STATION
- WATERCOURSE
- ROAD
- CONTOUR (30 M INTERVAL)
- WETLAND

1 0 1

SCALE 1:50,000 KILOMETRE

REFERENCE

1) GROUND SURFACE: TDUNC3 CONVERTED INTO RASTER, PROVIDED BY CLIENT.
2) CONTOUR, ROAD, WATERCOURSE, DAM, FOOTBRIDGE, TRAIL, WETLAND, WATERBODY OBTAINED FROM CANVEC © DEPARTMENT OF NATURAL RESOURCES CANADA. ALL RIGHTS RESERVED.
3) IMAGERY: DIGITALGLOBE, DATED 2011-05-11.

DATUM: NAD83 PROJECTION: UTM ZONE 11

PROJECT

BC HYDRO
DDMMON-16 LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING PROGRAM

TITLE

OVERVIEW OF STUDY AREA

GOLDER

PROJECT		1535517	FILE No.	
DESIGN	BH	14 JUN. 2018	SCALE AS SHOWN	REV. 0
GIS	JG	27 MAY 2019	**FIGURE: 1**	
CHECK	BH	27 MAY 2019		
REVIEW	SR	27 MAY 2019		

Document Path: N:\Bur-Graphics\Projects\2012\1492\12-1492-0117\GIS\Mapping\MXD\Fish\2019\Fig1_1214920117_OVERVIEW_REV0.mxd

2.4 Physical Parameters – All Study Years

2.4.1 Water Temperature

Water temperatures for the LDR were obtained downstream of the Lardeau River Water Survey of Canada gauging station (DRL) which is located downstream of the Duncan-Lardeau confluence at RKm 2.1. The DRL station uses a Lakewood™ Universal temperature probe (accuracy ±0.5°C).

Spot measurements of water temperature were also obtained at all stranding assessment sites at the time of sampling using a handheld alcohol thermometer (accuracy ±1.0ºC).

2.4.2 River Discharge

The DRL gauging station was selected as the compliance monitoring station for LDR discharge reductions for the entire Duncan River study area. All DDM releases and discharge data for the LDR were obtained from BC Hydro.

2.5 Fish Stranding Assessment – All Study Years

A formalized fish stranding assessment methodology was developed for the Duncan River in 2004, entitled "Strategy for Managing Fish Stranding Impacts in the Lower Duncan River Associated with Flow Reductions at Duncan Dam" (BC Hydro 2004). This protocol provided the standard methodology for conducting fish stranding assessments on the Duncan River prior to the present study. The protocol was updated in 2012 and 2014 (Golder 2012, Golder and Poisson 2014) and addressed up to date sampling methodologies, protocols related to fish stranding, and DDM operations. Based on the updated protocol, when DDM flow reduction is planned, BC Hydro will contact the organization responsible for conducting stranding assessments. The planned flow reduction is assigned an RE and a list of criteria is followed to determine if a stranding assessment is required (Golder 2012, Golder and Poisson 2014).

Because of the remote location of the LDR and limited development, access to the study area was by boat and foot. Boat launches are situated at the confluence of the Duncan and Lardeau rivers (BC Hydro private launch), at Argenta near the mouth of the river into Kootenay Lake, and at Lardeau on Kootenay Lake approximately 3.5 km downstream of the mouth of the LDR. Since late 2007, debris jams have formed in Reach 3 between RKm 4.1 and 4.7, periodically preventing continuous boat access along the river. Channel movement frequently occurs at the river's mouth to Kootenay Lake and access to the LDR from Kootenay Lake is difficult at lower LDR discharges and lake elevations.

2.5.1 Stranding Site Selection

In Study Years 1 to 3 (2008–2011), fish stranding assessments focused effort on index sites, as these sites had a larger amount of dewatered area during flow reductions and were also believed to strand higher numbers of fish. Due to this focused methodology, limited assessments of random sites were conducted and in-depth statistical analysis of stranding rates at both index and random sites was not possible.

To allow for comparisons of stranding rates between index and random sites, effort expended for random sites from Study Years 4 to 7 (2011 – 2015) was increased. This was accomplished by creating two strata (index and random) and then randomly selecting sites from each stratum to sample. The number of sites in each stratum selected for sampling was proportionate to the area dewatered in each stratum, based on the estimated area dewatered from each individual DDM flow reduction. The dewatered area at each site was calculated using site-specific area regression that was completed during Year 3 (2010 – 2011: Section 2.8.3: Golder 2011).

With the analysis of the Year 7 (2014 – 2015) dataset, Ho_1 (*Fish stranding observed at index sites along the lower Duncan River floodplain is representative of overall stranding*) was accepted. Therefore, during Study Years 8 to 13 (2015 – 2020), the dichotomous classification of sites into index and random was removed and all identified sites were grouped into the same strata. Sites for assessment were then randomly selected from this single group prior to each assessment.

2.5.2 Stranding Assessment Methodology

2.5.2.1 Isolated Pools

In all study years, all isolated pools within each sampled stranding sites (that formed during each of the assessed DDM flow reductions) were enumerated and the area (m^2) of each pool was estimated and recorded. Pool sampling was the primary focus of Study Years 1 to 3 (2008 – 2011), therefore all isolated pools within each assessed site were sampled using single pass electrofishing, dip nets and/or visual inspection. In addition, to determine the observer (capture) efficiency during stranding assessments, multi-pass electrofishing was conducted at a subset of randomly selected pools. The effort in the first pass was consistent with previous year's effort to allow comparison with previous year's data. The effort for each subsequent pass was as consistent as possible with the first pass. The fish salvaged and effort for each pass was recorded separately.

This resulted in relatively precise pool stranding estimates for Rainbow Trout during Years 3 (2010–2011) and 4 (2011 – 2012) data analysis (Golder 2011, Golder and Poisson 2012). Therefore, sampling effort was refocused in Year 4 (2011 – 2012) to assess interstitial stranding in more detail. In Years 4 to 12 (2011 – 2020), sampling effort for isolated pools was reduced to randomly sampling 50% of the pools at each assessed site, up to a maximum of three. In Year 13 (2020), isolated pool sample effort was further reduced to randomly sampling one pool at each assessed site. In all study years, sampling also occurred in isolated pools that were not selected at random, as field crews observed fish in these stranding mechanisms as they traversed sites. The fish salvaged and effort expended in non-randomly selected pools were recorded separately.

As observer efficiency can differ with the amount of cover present in each pool, the complexity of all isolated pools was classified into one of the following two categories:

- Zero to Low complexity (0% – 10% total cover)
- Moderate to High complexity (>10% total cover)

Pools with 0% – 10% cover were classified as Zero to Low complexity if surface area was 5 m^2 or less. Zero to Low Complexity pools are generally smaller in size so that fish could be captured readily by backpack electrofishing. Moderate to High Complexity pools are likely to have larger surface areas, larger substrate that could provide cover to fish including larger cobble and gravel or boulder, and some portions of the pool that are not visible because of woody debris or other cover types.

For all pools, associated cover types (and percentages within the pool) were recorded based on the following categories:

- Large woody debris (woody debris with diameter of >10 cm)
- Small woody debris (woody debris with diameter of <10 cm)
- Aquatic vegetation
- Submerged Terrestrial Vegetation
- Overhanging vegetation
- Organic debris (leaves, bark etc.)
- Cut bank
- Shallow pool
- Deep pool
- Other (metal, garbage, etc.)

The life history data for fish found stranded in isolated pools were recorded (Section 2.5.2.4). To maintain consistency in all study years, if time allowed, the dominant and subdominant substrate in all pools were recorded using a Modified Wentworth Scale (Table 2).

Table 2: Modified Wentworth Substrate Sizes and Codes.

Substrate Category	Code	Size (mm)
Bedrock/Silt	1	-
Sand	2	< 2
Fine Gravel	3	2 – 8
Medium Gravel	4	9 – 17
Large Gravel	5	18 – 32
Very Large Gravel	6	33 – 64
Small Cobble	7	65 – 128
Medium Cobble	8	129 – 192
Large Cobble	9	193 – 256
Boulder	10	> 256

2.5.2.2 Dried Pools

The working field definition of a dried pool is a low point, which when disconnected from the mainstem would create a wetted pool but was drained at the time of assessment. After the Study Year 4 (2011–2012) data analysis, it was recommended that dried pools be classified as a third stranding mechanism to further refine the fish stranding dataset. It was determined that there is a possibility that fish trapped in an isolated pool which subsequently drains could be classified as interstitially stranded during assessments. This new mechanism category removed the possibility of misidentifying the mechanism that stranded observed fish and allowed for more accurate estimates of fish stranding in the LDR.

The life history data for fish found stranded in dried pools were recorded (Section 2.5.2.4). Unlike isolated pools, the habitat parameters described in Section 2.5.2.1 were not recorded for dried pools as the areal extent of the pools at time of isolation from the mainstem river could not be accurately determined.

2.5.2.3 Interstitial Sampling

In Study Years 1 to 3 (2008–2011), dewatered habitat at each site was assessed by conducting two randomly placed grids (each grid has area of 0.5 m^2). The larger substrate (cobbles and gravels) and all cover were removed from each grid so that only fines remained, and the stranded fish enumerated. During Year 3 (2010–2011) data analysis, estimates of both interstitial stranding per unit area (m^2) and total interstitial stranding in the LDR showed high uncertainty (Golder 2011). To reduce this uncertainty and obtain a more complete representation of fish stranding in the LDR, interstitial sampling effort from Year 4 (2011–2012) onward was increased.

To further reduce uncertainty related to interstitial stranding estimates, transect sampling was implemented in Year 7 (2014–2015). Transect sampling allowed for an increase in the amount of dewatered habitat assessed at each site without increasing the amount of time crews spent conducting interstitial sampling surveys. A maximum of 10 randomly selected transects were surveyed within dewatered interstitial habitats with gradients and substrates having the potential to strand fish. A measuring tape was laid on the substrate from the wetted edge to the top of the dewatered area, and the length was recorded. The substrate near the tape was then visually assessed (0.5 m on either side of the tape along its entire length) with all fish stranded recorded. Although transect sampling did increase the amount of area surveyed, encounters of interstitially stranded fish remained very low.

In Year 11 (2018–2019), updated methodologies were implemented to further increase the area of dewatered habitat sampled, to increase the encounters of interstitially stranded fish. To assess interstitial stranding at each surveyed site, field crews censused areas of randomly selected dewatered habitat with consistent habitat characteristics (i.e., substrate size and slope) within a site by counting all stranded fish encountered. Consistent effort (approximately twenty minutes) was expended at each site to ensure an adequate number of sites along the entire LDR were sampled during each assessment. The main objective of this approach was to increase the amount of interstitially sampled habitat per site to reduce the uncertainty of previously estimated interstitial stranding rates. The total area within these censused areas were recorded.

In all study years, fish were opportunistically found stranded on exposed substrate outside of randomly selected areas (i.e., in dried pools or encountered non-randomly on dewatered substrate) while field crews traversed sites. The fish salvaged and effort expended in non-randomly selected interstitial areas were recorded separately. To maintain consistency in all study years, the dominant substrate in each grid, transect and/or censused area was recorded using a Modified Wentworth Scale (Table 2).

2.5.2.4 Fish Life History Data

For each fish captured during pool and interstitial sampling, the following life history data were recorded:

- Species
- Total or Fork Length (depending on species) in mm
- Condition (alive or dead)
- Salvaged (Yes/No)
- Habitat association (if possible)

Observed fish that were not captured and remained in the pool or interstices after sampling was completed were also documented. If the number of captured fish from a pool or interstices was high and time did not allow for the measuring of all fish, an estimate of the number of fish by species captured in the pool or interstices was recorded and individuals from a subsample (30 to 50) of each species from the salvaged fish were measured for length.

2.6 Calibration Assessment – Study Year 2

To address Management Question 2 (*What are the levels of impact to resident fish populations associated with fish stranding event on the Lower Duncan River?*) and Alternate Study Hypothesis H_{01} (*Fish stranding observed at index sites along the Lower Duncan River floodplain are representative of overall stranding*), Golder proposed that Annual Calibration Assessments be conducted in conjunction with regular fish stranding index monitoring following Duncan Dam flow reductions. To extrapolate fish stranding estimates observed at index sites to the entire area of the lower Duncan River, the calibration stranding assessments were designed to determine if the stranding rates at index sites were representative of previously unsampled habitat. In an addendum to the proposed study plan dated 19 January 2009 and in discussions with the BC Hydro Contract Authority at the time (Trevor Oussoren) on 18 August 2009, it was determined that since the Calibration Assessment methodology and proposed effort included in the original study proposal (Golder 2009b) differed from the methodology and effort of Index Stranding assessments, the results obtained may not be comparable. Therefore, it was decided to alter the proposed Calibration Assessment study plan.

The original study plan for DDMMON-16 (Golder 2009b) described the methodology of the Calibration Assessments as follows:

"*In Year 2, two calibration assessment crews will be used in addition to the index monitoring crew during a September flow reduction event. This will allow crews to visit up to 30 sites, although the actual number of sites visited will depend on the effort required to access and properly assess each site. Calibration assessment crews will visit sites and conduct a thorough assessment of dewatered habitat (2 m^2) immediately adjacent to the river at the midpoint of the site. Where the horizontal width of the drawdown zone is sufficient, the calibration area will extend parallel to the river for 1 m and perpendicular to the river channel for 2 m. If the horizontal width of the drawdown is less than 2 m, the entire width of the drawdown zone will be sampled and the site will extend as necessary parallel to the river channel to achieve the 2 m^2 sample area. If the calibration site lies within a portion of a pool, the entire pool will be sampled and a revised estimate of area calculated. To develop an understanding of whether fish stranding observed at index sites is representative of overall fish stranding, calibration sites will* *be established every 500 m downstream from Duncan Dam on each of the left and right banks of the mainstem and associated side channels, assessing 2 m^2 areas of dewatered habitat until reaching the mouth of the Duncan River. The locations of these sites will be determined from maps previously provided by BC Hydro.*"

The revised Calibration Assessment study plan had the following two assumptions:

1. The term "Overall Stranding" in Management Question 2 refers to all sites that have been previously identified to have potential risks of fish stranding (AMEC 2004); and,
2. All areas (i.e., riverbank habitat) not identified as having potential fish stranding risks are assumed to have low rates of stranding that do not affect fish population levels.

The Calibration Assessment methodology was revised to include the following:

"*Golder proposes to undertake a single calibration assessment in Year 2 during the fall. The focus of monitoring will continue to be on assessment of fish stranding and not fish salvage. The revised methodology will focus on stranding assessments with consistent effort and methods, analysed by type of habitat dewatered.*

The delineation of appropriate habitat types that differentiate fish stranding risk will be based on general habitat categories (i.e., side channel, mainstem bar, and riverbank). Air photo mosaics (AMEC 2004) and information from the HEC model (if available) will be used to define habitat types and rank priorities for sampling in areas that are likely to be dewatered during the flow reduction event. Index sites will be assigned a habitat description based on the majority of habitat present. If more than one habitat type is present, the index site size will be modified accordingly, and a description of modifications created so that the information can be used to compare against historic stranding assessments. Calibration sites of similar size to the index sites will be defined to be representative of the variety of habitat types for the length of the lower Duncan River. A single crew will assess the regular index sites using the historic methodology and revised to include 3 pass depletion estimates for larger pools where all fish cannot be visually observed. The two additional calibration assessment crews will each be assigned to fish stranding assessments at the calibration sites."

As Assumption 2 above states that riverbank habitat poses a low risk of fish stranding due to substrate slope and composition and is assumed to not have a population level effect, riverbank sites were excluded during site selection for the calibration assessment. This decision was made because the amount of effort needed to demonstrate a statistically reliable estimate for this habitat type would not be within the existing budget and would detract from improved precision in the habitats that may pose a significant risk of stranding. The hydraulic model (DDMMON-3), information on fish species distribution (DDMMON-2) and aerial photos provided by BC Hydro (AMEC 2004) were used to identify potential calibration assessment sites containing side channel and mainstem/bar habitat types. In total, 18 potential mainstem bar habitat sites and 15 side channel sites were identified. To select the sites to be included in the survey during the calibration assessment, sites containing mainstem bar habitat and side channel habitat were separated into two different strata. Eight sites within the side channel stratum and seven sites within the mainstem bar stratum were then randomly selected. The random selection was completed systematically (i.e., sampling every *n*th site with *n* determined by the number of samples divided by the total number of sites in each stratum) to ensure complete spatial coverage in the study area.

Two main factors were considered in deciding when to conduct the calibration assessment. The fall transition to Kokanee protection flows usually consists of three flow reductions from DDM. The magnitude of each of the three reductions was considered, as well as access to the proposed calibration sites below log jams present on the Duncan River at River Kilometers (RKm) 4.0 and 4.5. It was anticipated that the initial flow reduction (25 September 2009) would not dewater sufficient amounts of habitat for calibration assessments, and the calibration assessment sites below RKm 4.0 sites would not be accessible by boat during the third and final flow reduction (1 October 2009). Therefore, the second flow reduction (28 September 2009) was chosen for the calibration assessment, as this reduction was predicted to dewater sufficient amounts of habitat for calibration assessment, while still allowing boat access to all sample sites.

To obtain data during the Calibration Assessment that was comparable to data collected at index sites, calibration assessment field crews followed the same methodology used during fish stranding assessments at index sites (Section 2.5).

2.7 Fall Target Species Abundance Assessment – Study Years 6 to 9

2.7.1 Abundance Site Selection

Based on the DDMMON-2 results of fish habitat use (Thorley et. al. 2011 and 2012), the TELEMAC2D hydraulic model developed as part of the DDMMON-3 program was used to divide the shorelines of the LDR mainstem and side channels into the following 4 strata:

- Shallow (≤ 0.4 m) and slack (≤ 0.02 m/s)
- Shallow (≤ 0.4 m) and flowing (> 0.02 m/s to 0.5 m/s)
- Deep (> 0.4 m to 1.5 m) and slack (≤ 0.02 m/s)
- Deep (> 0.4 m to 1.5 m) and flowing (> 0.02 m/s to 0.5 m/s)

Sites were randomly selected using linear Generalized Random Tessellation Stratification (GRTS) along the thalweg using the statistical environment R, v. 3.1.0 (R Development Core Team 2015) using the package *spsurvey* (Kincaid and Olsen 2013). Sites were not stratified by main and side channel, since previous reports found no significant differences in abundance among the two types of habitat (Thorley et al. 2011). A total of 15 main and 30 oversample points were selected for each stratum.

Prior to nighttime snorkel sampling, the crew surveyed the GRTS-selected sampling sites in the day by boat to determine if the site was suitable for sampling. The sites selected for sampling were marked using flagging tape at their upstream and downstream boundaries. Field conditions were not always as predicted by the TELEMAC 2D model, rendering some pre-selected sites unusable. If the crew assessed both main and oversample GRTS points and still fell short of the expected seven sites per stratum, sites were added to the sampling scheme based on proximity to GRTS site, site-measured depth and professional judgement of current velocity. Once the crew finished sampling sites allocated for each stratum, they proceeded to sampling additional sites, chosen in the field. This was performed since: 1) most sampled sites fell short of the expected sampling length, and hence total covered shoreline length was deemed inadequate; 2) the budget allowed additional sampling; and 3) an increase in sampling site numbers would improve fish abundance estimates.

2.7.2 Snorkel Surveys

Snorkel surveys were conducted in Study Years 6 to 9 (2013 to 2017) to estimate the abundance of juvenile (<250 mm fork length) Mountain Whitefish and Rainbow Trout. To ensure sufficient darkness, snorkelling assessments of abundance commenced at least 30 minutes after sunset. Typically, two snorkelers surveyed each site; while at narrow sites one snorkeler conducted the sampling, depending on site conditions. Sites were surveyed by snorkelers to a maximum depth of 1.5 m, as Thorley et al. (2012) reported that the vast majority of Mountain Whitefish and Rainbow Trout juveniles (fry and parr) were found in depths <1.5 m. In the shallows (15 cm depth or less), fish were observed by carefully walking and using a spotlight. For each site, field crews recorded the following information: date, time of beginning and end of sampling of each site, GPS location of the upstream and downstream boundaries of each site, and the number and life stage of the observed target species.

2.8 Stranding and Abundance Estimation Modeling

2.8.1 Calibration Stranding Assessment – Year 2

To extrapolate fish stranding estimates observed at index sites to the entire area of the lower Duncan River, the calibration stranding assessments were designed to determine if the stranding rates at index sites were representative of previously unsampled habitat. The calibration assessment data exhibited excess clumping of zero counts which resulted in the violation of assumptions of normality for linear regression, as only 4 of 14 of the calibration sites had positive fish stranding results (> 1 fish) compared to 70 of 106 sampling events at index sites. To account for this during analysis, two different models were used to compare the data.

To account for the large number of zeros in the data, calibration versus index sites were evaluated by the occurrence of stranding at each site using a contingency table analysis. A General Linear Model (GLM) was used to evaluate the statistical relationship of stranding densities and stranding numbers to variables including macro habitat type, ramping rates, sampling gear type, initial wetted area of the site, resultant wetted area of the site after flow reduction, number of pools formed at the site during dewatering (transformed to normalized as Log Number Pools=Log (Number Pools+1), and the temporal variables YEAR and MONTH when the stranding event occurred. Data used in the analysis were examined for normality using the nonparametric Kolmogorov-Smirnoff (KS) one sample test and data were Log transformed to achieve normality for the analysis. The GLM model was also run using the classification of calibration versus index site as a categorical variable by excluding all sites that had zero counts. The GLM model run was used to determine both the probability of stranding occurring at sites randomly selected outside of the index areas and the magnitude when stranding does occur.

2.8.2 Statistical Analysis

In-depth statistical analysis for this program was conducted by Poisson Consulting (Thorley J.L. 2021) and the methodology used is included in this Section.

Model parameters were estimated using Bayesian methods (McElreath 2016). The Bayesian estimates were produced using JAGS (Plummer 2003). Unless indicated otherwise, the Bayesian analyses used weakly informative normal or half-normal prior distributions (Gelman, Simpson, and Betancourt 2017). The posterior distributions were estimated from 1500 Markov Chain Monte Carlo (MCMC) samples thinned from the second halves of three chains (Kery and Schaub 2011, 38–40). Model convergence was confirmed by ensuring that the potential scale reduction factor ($\hat{R} \leq 1.05$) (Kery and Schaub 2011, 40) and the effective sample size ($ESS \geq 150$) (Brooks et al. 2011) for each of the monitored parameters (Kery and Schaub 2011, 61).

The parameters are summarized in terms of the point *estimate*, *lower* and *upper* 95% credible limits (CLs) and the surprisal *s-value* (Greenland 2019). The estimate is the median (50th percentile) of the MCMC samples while the 95% CLs are the 2.5th and 97.5th percentiles. The s-value can be considered a test of directionality. More specifically, it indicates how surprising (in bits) it would be to discover that the true value of the parameter is in the opposite direction to the estimate. An s-value of 4.3 bits, which is equivalent to a p-value (Kery and Schaub 2011; Greenland and Poole 2013) of 0.05, indicates that the surprise would be equivalent to throwing 4.3 heads in a row. The condition that non-essential explanatory variables have s-values ≥ 4.3 bits provides a useful model selection heuristic (Kery and Schaub 2011).

Where computationally practical, the sensitivity of the parameters to the choice of prior distributions was evaluated by increasing the standard deviations of all normal, half-normal and log-normal priors by an order of magnitude and then using $\hat{R}$ to test whether the samples were drawn from the same posterior distribution (Thorley and Andrusak 2017). The results are displayed graphically by plotting the modeled relationships between particular variables and the corresponding response(s) with the remaining variables held constant. In general, continuous and discrete fixed variables are held constant at their mean and first level values, respectively, while random variables are held constant at their typical values (expected values of the underlying hyperdistributions) (Kery and Schaub 2011, 77 – 82). When informative, the influence of particular variables is expressed in terms of the *effect size* (i.e., percent or n-fold change in the response variable) with 95% credible intervals (CIs, Bradford, Korman, and Higgins 2005).

The analyses were implemented using R version 4.0.4 (R Core Team 2020) and the *mbr* family of packages. The complete model specification used is provided in Appendix B.

2.8.3 Dewatered Area

In 2009, the outputs of several RIVER-2D hydraulic model runs were provided by Northwest Hydraulic Consultants (NHC) as part of the DDMMON-3 WUP program. During Study Year 2 (2009–2010: Golder 2010), the results of the hydraulic model were incorporated into the stranding analysis. In reviewing these initial model results, some inaccuracies in the model predictions (wetted areas increased when flows decreased) were discovered when conducting the fish stranding analysis. This original modeling effort relied on 25 $m^3 s^{-1}$ increments from DRL discharges ranging from 25 $m^3 s^{-1}$ to 325 $m^3 s^{-1}$ and produced 13 model runs. The increments between model runs led to data gaps and rounding errors when calculating dewatered area as resulting from flow reductions from DDM. These rounding errors were substantial when the flow decreases were small. A model output data quality rating was developed to determine how accurate the dewatered area estimates from these initial model runs matched the flow reduction in the DDMMON-16 dataset. The rating was increased as the difference between the flow reduction initial discharge and the resultant DRL discharge increased. Also, the closer these discharges matched the provided model outputs, the higher the rating assigned to the reduction. Lastly, the rating was positively weighted when the proportion of the initial DRL discharge, for each RE, decreased relative to the maximum annual DRL discharge (i.e., more weight was given to the lower water levels in the LDR).

This rating led to the classification of 39% of the flow reductions within the dataset as a “Poor” fit to the initial model outputs. This included flow reductions outside the range of the model outputs or reductions that were too small to be compared with minimum 25 $m^3 s^{-1}$ incremental output of the model. Of the flow reductions in the dataset, 29% were classified as a good fit to the model outputs.

To fill in data gaps and to obtain a more accurate estimate of the dewatered area during DDM flow reductions, an additional set of seven model runs were provided by NHC. The outputs of this second set of model runs were selected based on requirements of several ongoing WUP programs on the lower Duncan River. Upon initial inspection of the second set of model outputs, the dewatered area shown in the new model outputs were not comparable to the model runs provided previously. This was because the new outputs were processed through the RIVER-2D model with different calibration parameters than the previous model runs and the model was not run to a steady state because of excessive run times required, which led to differing outputs (personal communication, NHC, March 2011).

To solve this incompatibility between model runs, a procedure involving regression analysis (which assumed model error) of the data produced from site specific model predictions was developed in Year 3 (2010–2011). This new procedure required four additional model runs at DRL discharges higher than the original outputs provided. This additional data from the high-water model runs allowed the regression analysis to predict more accurate estimates of dewatered area during DDM flow reductions at higher river stages. A regression model was prepared that would interpolate between the estimated wetted area values for each of the study sites. This procedure, which smoothed the data from the RIVER-2D model, was used because the error within the hydraulic model estimates of wetted area at a particular flow would confound linear interpolation.

The areas for each of the study sites were generated with ArcMap vs. 10.7. The stranding sites were stored as a polygon feature class and stored in a project geodatabase. The wetted area in each of the sites was calculated in ArcMap with the input used being the River 2D model output for each of the flows simulated in the runs. These data were compiled and a Lowess regression model was applied to the data set of wetted area versus flow for each of the simulations, using the time series module from the statistical program SYSTAT version 13.0. For most of the simulations, a tension setting of 0.15 was used. After examining the graphic results for each site where the interpolated values were matched against the River 2D model output, tension values were adjusted if the Lowess regression model did not predict increases between the area estimates provided by the 2D River model at each site when flows used in the simulation increased. This step was repeated until the smallest tension setting was obtained that provided logical results (increases in flow resulting in increases in wetted area). The sites where tensions other than the default, 0.15, were used include S7.7R: Tension = 0.50; M2.5L: Tension = 0.18; and S4.0-S4.2R: Tension = 0.25. The Lowess regression model predicted areas at 1 $m^3 s^{-1}$ increments across the range of simulated flows. Projections outside of the flow range of the River 2D model were not conducted as the polynomial functions used by the Lowess algorithm do not provide accurate predictions outside of the interpolative range of the input data.

In 2018, updates to the Lower Duncan River 2D hydraulic model were completed as part of the DDMMON-3 program. Additional runs from the updated 2D hydraulic model (16 runs ranging from 25 $m^3 s^{-1}$ to 400 $m^3 s^{-1}$, in 25 $m^3 s^{-1}$ increments) were included in the Study Year 13 (2020) data analysis. The updated 2018 stranding sites were intersected with the updated 2D hydraulic model (16 runs). This resulted in a summary of wetted area for each stranding site for a given run. The updated hydraulic model estimates did not confound linear interpolation, therefore linear regressions were used to interpolate between the estimated wetted area values for each of the study sites.

To compare pre- and post-WUP operations, Years 1 to 13 (2008–2020) DDM and DRL flow data were compiled the discharge dataset. The change in site dewatered area was then estimated using a hierarchical Bayesian regression model (Golder and Poisson 2012). For the purposes of the historical comparison, discharge reduction events were defined as a decline in the hourly discharge caused by DDM operations as measured at the WSC gauge at DRL. The difference in discharge when a reduction event occurred was then multiplied by the slopes estimated for the high and low slope habitats and summed together to obtain a total riverine area exposed for each reduction. These total areas were summed over the entire year to estimate the total area exposed by year.

2.8.4 Slope Analysis

In Years 4 to 7 (2011–2015), to facilitate data analysis and reduce uncertainty related to interstitial stranding rates of the target species, dewatered habitat was categorized as low slope (0-4% gradient) or high slope (>4%). The categories of low (0-4%) and high slope (>4%) used in the analyses were based on values in the literature (Bauersfeld 1978; Flodmark 2004). Whole river estimates of dewatered habitats were estimated using the following method:

1) Slopes of the dewatered area for the whole river were extracted from the 2012 NHC digital elevation model (DEM) for three reductions (225 m^3 s^{-1} to 200 m^3 s^{-1}, 200 m^3 s^{-1} to 125 m^3 s^{-1}, 125 m^3 s^{-1} to 75 m^3 s^{-1}),
2) The differences in the estimated dewatered area for each slope category were plotted and a linear regression model fitted to the differences for each slope type,
3) The regression equation's slope was extracted and used as a multiplier in order to determine the areas of high and low slope habitat exposed by each operational reduction, and
4) The low and high slope areas were summed to obtain the total area in the LDR exposed for each reduction.

To further reduce interstitial sampling uncertainty and facilitate data analysis, this dichotomous slope classification was expanded to 5 categories (i.e., 0-5%, 6-10%, 11-15%, 16-20%, >20%) in Year 8 (2015–2016). Further refinements to slope classification and analysis were implemented in Study Year 10 (2017–2018).

To estimate the slope of the active streambed at different discharges a GIS water inundation model of the river was created using the DDMMON 3 Digital Elevation Model and a three-dimensional plane. The plane was inclined and distorted to the gradient of the river. Field observations were used to improve the real-world accuracy of the plane. A total of 10 discharges between the highest and lowest encountered during previous stranding assessments were selected for input into the GIS model. Discharges were correlated to elevation data using a DRL stage curve provided by BC Hydro. Inputting the 10 elevations into the inundation model allowed for estimation of the area of streambed within a series of percent slope categories (i.e., 0-2%, 2-4%, 4-6%, 6-8%, >8%) that were inside of the wetted area at each inputted discharge rate. This data was used during the extrapolation of pool and interstitial stranding rates over the entire study area. To expand on the slope analysis conducted in Study Year 10 (2017–2018), an additional 4 discharge levels (for a total of 14 discharge levels) were inputted into the GIS model in Year 11 (2018–2019). In Year 13 (2020), the GIS inundation model was revised with the inclusion of a new version of the DEM, which was updated in 2018.

2.8.5 Target Species Stranding Estimation

Bayesian Models were used to estimate pool and interstitial abundance, numbers of fish stranded in isolated pools, and numbers of fish stranded interstitially. The analyses detailed in the next sections were implemented as in Section 2.8.2.

2.8.5.1 Pool Stranding

The number of fish stranding in each randomly sampled and unsampled pool was estimated using a multi-pass removal model (Wyatt 2002). Key assumptions of the final model included the following:

- The expected abundance varies by season.
- The expected capture efficiency remained constant.
- The abundance is gamma-Poisson distributed.
- The number of fish removed on each pass is binomially distributed.

The predicted abundances for all sampled and unsampled pools were summed by site and flow reduction event. The model failed to fit to Mountain Whitefish, therefore total stranding estimates (Section 2.8.5.4) for this target species were not calculated. The model code is provided in Appendix B.

2.8.5.2 Interstitial Stranding

The density of fish stranding in the interstitial area was calculated by species and by slope categories (0-2%, >2-4%, >4-6%, >6-8%, >8%) by dividing the total number of fish randomly observed stranded by the total area surveyed. The density of fish stranding in the interstitial area was also calculated by species and season by dividing the total number of fish by the total area surveyed. Due to the extreme nature of variation in the data, which could not be reliably modeled using a log-normal, zero-inflated or over dispersed Poisson distribution, Cis were not estimated.

2.8.5.3 Non-Random Stranding

The number of fish observed stranded in dewatered pools or non-randomly selected interstitial areas (areas where fish were opportunistically found stranded on exposed substrate outside of randomly selected areas) was summed by site and flow reduction event.

2.8.5.4 Total Stranding

The estimates of the total number of fish (interstitial, non-random and pools) stranding at each surveyed site during each flow reduction event were analyzed using a bias-corrected over-dispersed Poisson model.

Key assumptions of the final model include:

- The expected fish stranding areal density varies by season.
- The expected fish stranding areal density varies randomly by site and reduction.
- The actual number of fish stranding is gamma-Poisson distributed.

The model was then used to estimate the total number of fish stranding at the sites not surveyed, as well as the relative importance of individual sites. The percent stranding of the spring abundance of age-1 Rainbow Trout was then estimated using the total stranding estimates for each site during each flow reduction event.

Key assumptions of the percent stranding estimates included the following:

- The observer efficiency during the fall abundance surveys was 15% (Andrusak and Thorley 2018).
- The spring abundance surveys were conducted on 15 March.
- The fall abundance surveys were conducted on 20 September.
- Since abundance surveys were not conducted in the 2014 spawn year, spring abundance was assumed to be the same as the 2017 spawn year spring abundance.
- The total interstitial stranding for each reduction was the sum of the expected densities multiplied by the area for each slope category.
- The overwintering mortality from 1 September to 1 April was 70% (Decker and Hagen 2009).
- The total pool and interstitial stranding for each reduction as well as the fall and spring abundance were adjusted for the expected mortality assuming a constant mortality rate between 1 September and 1 April.
- The percent stranding was the total adjusted stranding divided by the adjusted spring abundance plus the total adjusted stranding.

The model code is provided in Appendix B.

2.8.6 Proportional Stranding

The proportion of the Rainbow Trout population stranded during each flow reduction event was calculated by dividing the estimated adjusted stranding by the estimated adjusted Spring abundance for a given spawn year. The ramping rate was the standardized average discharge change per hour at Duncan Dam while the wetted history was the standardized log number of days up to a maximum of 50 since the discharge at DRL was last at half of the flow reduction.

Key assumptions of the percent stranding estimates included the following:

- The proportion of the population stranded varies by date, flow reduction magnitude, the starting discharge level at the DRL, abundance of the cohort and wetted history.
- The residual log-odds proportion of the population stranding is normally distributed.

Preliminary analysis indicated that second-order polynomials were not clearly directional predictors.

2.8.7 Target Species Abundance Estimation

In Study Year 3 (2010–2011), estimates for overall juvenile Rainbow Trout and Mountain Whitefish population abundance were obtained from the juvenile habitat use component of the DDMMON-2 study (Thorley et al. 2011). The DDMMON-2 population estimates for Rainbow Trout and Mountain Whitefish were based on snorkel survey data obtained from September 15-21, 2010 (Thorley et al. 2011). Population abundance was only estimated in Fall 2010 for Mountain Whitefish so all discussion of population effects in these study years was in relation to this point estimate in time.

In Study Years 6 to 9 (2013 to 2016), separate abundance estimates were conducted for Mountain Whitefish and Rainbow Trout juveniles (fork length <250 mm). Hierarchical Bayesian Models (HBMs) were used to estimate total abundance. In the Bayesian implementation of the model, fish abundance was assumed to be Poisson distributed, with a mean expected density drawn from a log-normal distribution. Observed fish counts were assumed to be binomially distributed, with estimated fish abundance as the number of trials and observer efficiency as probability of success. Fish density was modeled using fixed effects of depth (shallow/deep) and year (2013, 2014, 2015, and 2016) and a random effect of site, to allow density to vary by site. The significance of model parameters was determined based on whether the parameters' 95% CRI overlapped zero. Since the first level of each factor effect (depth and year) was set to zero, if a parameter's 95% CRI overlapped zero, it suggested that there was no significant difference between that parameter and the first level of the factor.

The estimated stratum fish density (fish/m^2) and the total area of each depth/flow stratum, derived from the DDMMON-3 RIVER-2D hydraulic model, were used to estimate the total abundance of fish in each stratum. Summing of estimates across all sampled strata yielded the total abundance of fish within the LDR (expressed as mean and 95% CRI). Observer efficiency, derived from previous work on Rainbow Trout and Mountain Whitefish in the LDR (Thorley et al. 2011 and 2012), was used to estimate total fish abundance at each site from the number of observed fish. The complete list of variables and parameters, as well as model specification used are described in detail in the Year 9 (2015-2016) annual report (Golder 2018).

For Study Years 3 to 13 (2010–2020), the spring age-1 Rainbow Trout abundance estimates used were provided by Greg Andrusak of the Ministry of Environment (Andrusak and Thorley 2019). Updated observer efficiencies of 15% (Andrusak and Thorley 2018) and overwintering mortality from 1 September to 1 April of 70% (Decker and Hagen 2009) were used to re-estimate previously reported fall abundances.

The data were prepared for analysis using R version 4.0.3 (R Core Team 2020). The complete model specification used is provided in Appendix B.

2.9 Duncan Stranding Database and Data Management

The MS-Access database (referred to as the LDR stranding database) was created in Study Year 2 (2009–2010) and populated with all available stranding data collected during Study Years 1 to 13. Presently, 106 individual stranding assessments are in the database. Results from 14 assessments prior to 15 September 2006 were not included in the stranding estimation dataset as sampling methodology was not consistent with more recent assessments.

Protocols for information management for data collected during this program have been created by DDMMON-15: Lower Duncan River Protocol Development and Finalization and are presented in the revised document: “Adaptive Stranding Protocol for Managing Fish Impacts in the Lower Duncan River Associated with Flow Reductions at Duncan Dam” (Golder 2012, Golder and Poisson 2014). Currently an updated version of this document is in preparation. To meet the goals of the DDMMON-15 workshop in 2016, the Lower Duncan River Fish Stranding Database was modified at the onset of Year 10. The database was altered to a risk/status at water elevation-based classification for all identified sites, similar to the BC Hydro Lower Columbia River Fish Stranding Database utilized by the Lower Columbia River Fish Stranding Program (CLBMON-42; Golder 2019). This was to allow for more informed stranding responses in future years.

3.0 RESULTS

3.1 Differences between Pre-WUP and Post-WUP Operations

Based on DDM flow data provided by BC Hydro, the DDMMON-3 RIVER 2D model outputs, and subsequent analysis, the mean of annual overall areas exposed during pre-WUP operations was 17.0 km^2, in comparison to 12.2 km^2 during the post-WUP operational regime (Figure 2). The area exposed was less variable from year to year in the post-WUP operational regime over the years assessed and was lower in general, especially between 2013 and 2017, as well as in 2019. The maximum annual exposed area (20.2 km^2) was observed in 2006, during pre-WUP operations. The minimum exposed area (9.8 km^2) was observed in 2019 during post-WUP operations. Exposed area per reduction was on average higher in the pre-WUP period than in the post-WUP period (0.43 and 0.30 km^2, respectively; Figure 3). The difference was statistically significant (t-test; P=0.002). Annually, mean exposed areas in reported reductions ranged from 0.2 km^2 (2015 stranding year) to 0.6 km^2 (2005 stranding year).

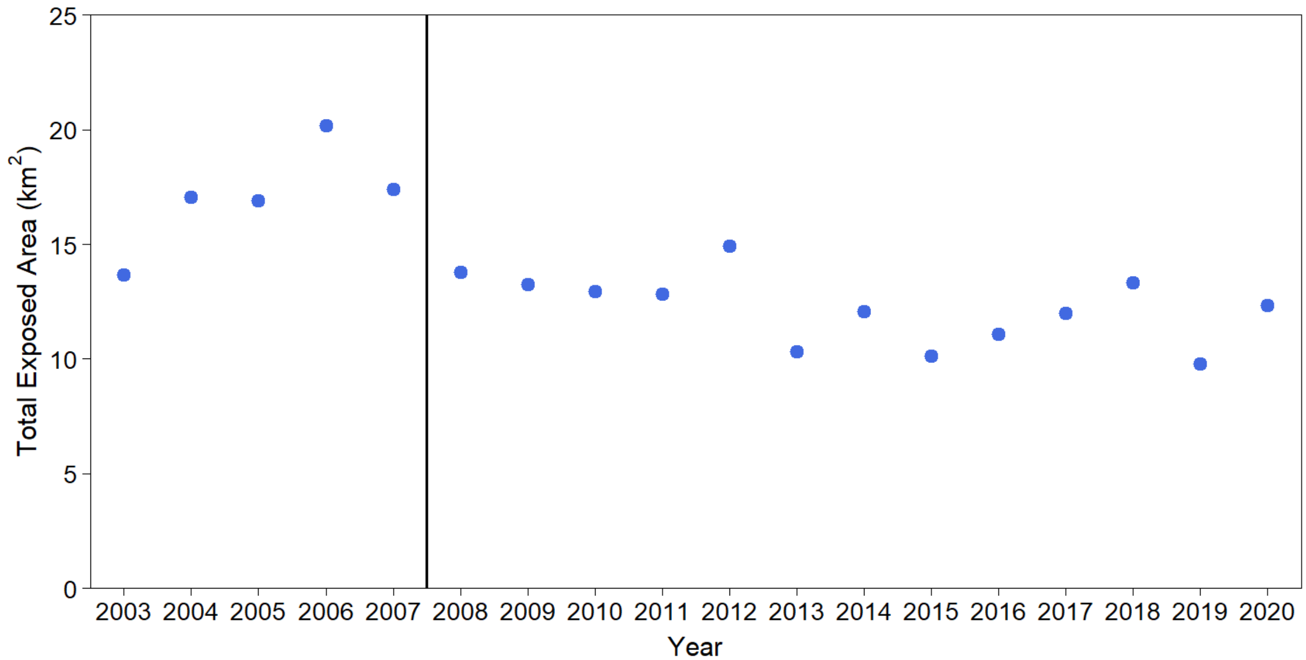


Figure 2: Total area exposed by all annual reductions in the LDR by year of operations, calculated using DRL discharge. The vertical line denotes the beginning on WUP flows in 2008. Note that label on Y-axis denotes study year, not calendar year.

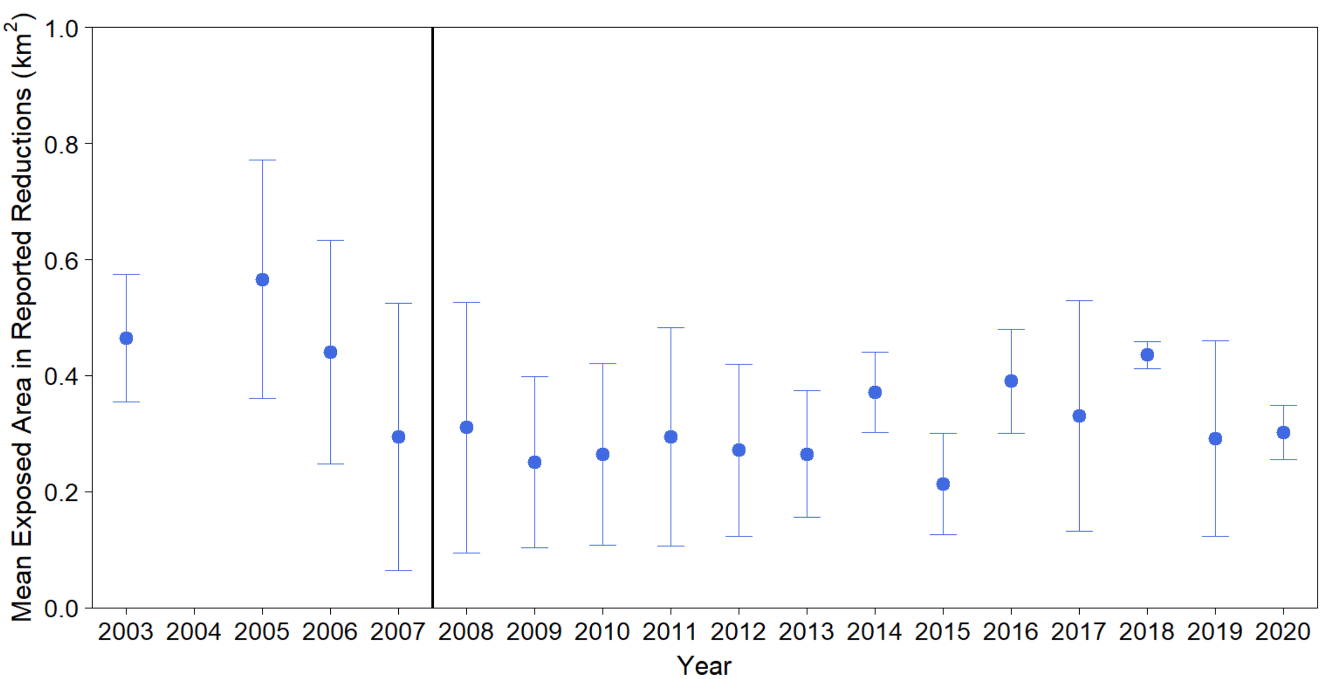


Figure 3: Mean area exposed by all annual reductions in the LDR by year of operations, calculated using DRL discharge. Bars represent 1 standard deviation. The vertical line denotes the beginning on WUP flows in 2008. Label on Y-axis denotes study year, not calendar year.

Interannual variability in mean discharge, as assessed at the gauge at DRL, was higher overall in the pre-WUP period, with the greatest reduction in discharge variation seen in the October to December period in the post -WUP period. Generally, under the post-WUP operational regime (implemented in 2008), there was almost no interannual deviation during the October to January period (Golder 2017b). However, in 2015, the DRL discharge was increased to approximately 250 m^3/s (8829 ft^3/s: Golder 2017b), resulting in high interannual variability during the October-January period (Figure 4). Decreased discharge variability post-WUP was also recorded in March, where discharge trend changed from gradual increase pre-WUP to a stable flow or a slight gradual decrease post--WUP.

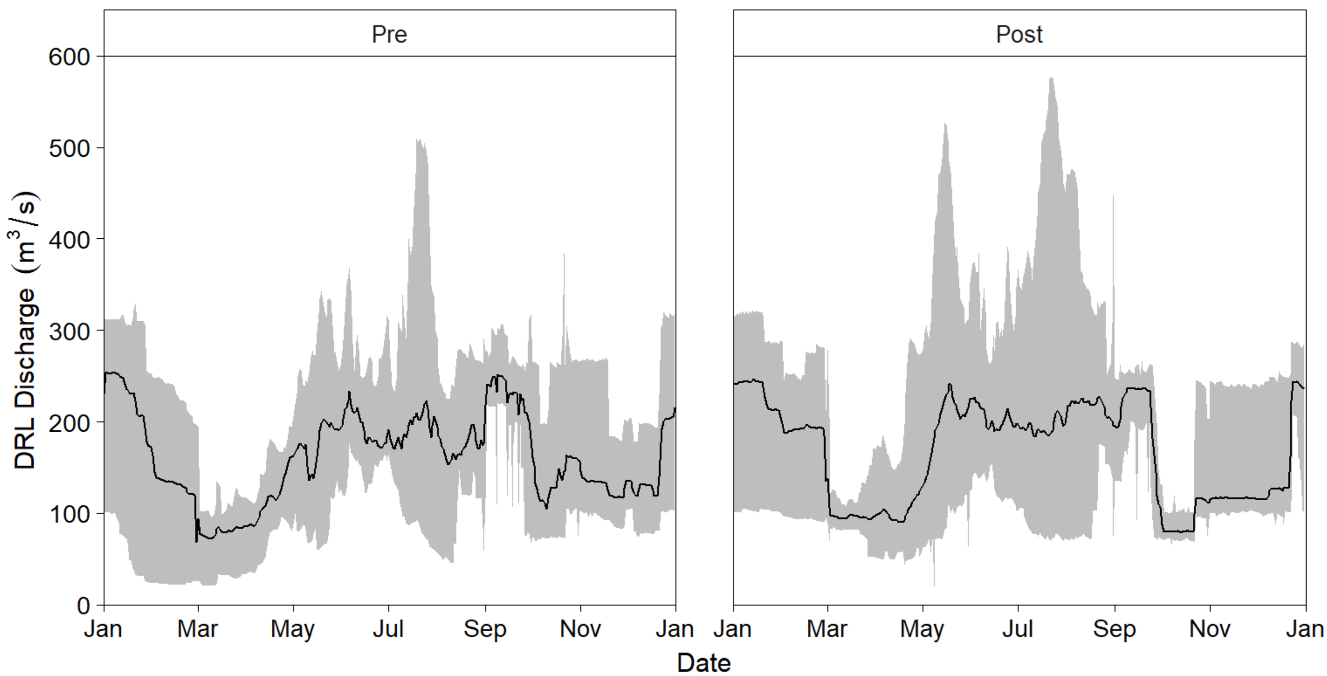


Figure 4: Minimum, maximum (grey ribbon) and mean (black line) discharge as measured at the WSC DRL gauge in the LDR during pre-WUP operations (2002–2007) and post-WUP operational implementation (2008–2020).

Although the magnitude of pre-WUP flow reductions from DDM exhibited narrower ranges within each year in comparison to some post-WUP operation years, the mean and median magnitudes during pre-WUP conditions were higher in most years (Figure 5). Substantial differences in the reduction magnitude between pre- and post-WUP- operations were not identified in early post-WUP years. However, between 2013 and 2016, as well as in 2018 and 2020, reductions had narrow ranges and were generally smaller than pre-WUP operations.

In three of the four years examined during pre-WUP operations, ramping rate ($\Delta m^3\ s^{-1}\ h^{-1}$) exhibited substantial variations and range (Figure 5). The remaining year in the pre-WUP period was similar to operations during post-WUP-. Overall, post-WUP ramping rates were similar between years, with maximum ramping rates ranging between 24 $m^3/s/h$ (in 2019 - 2020 and in 2020) and 38 $m^3/s/h$ (in 2009 - 2010).

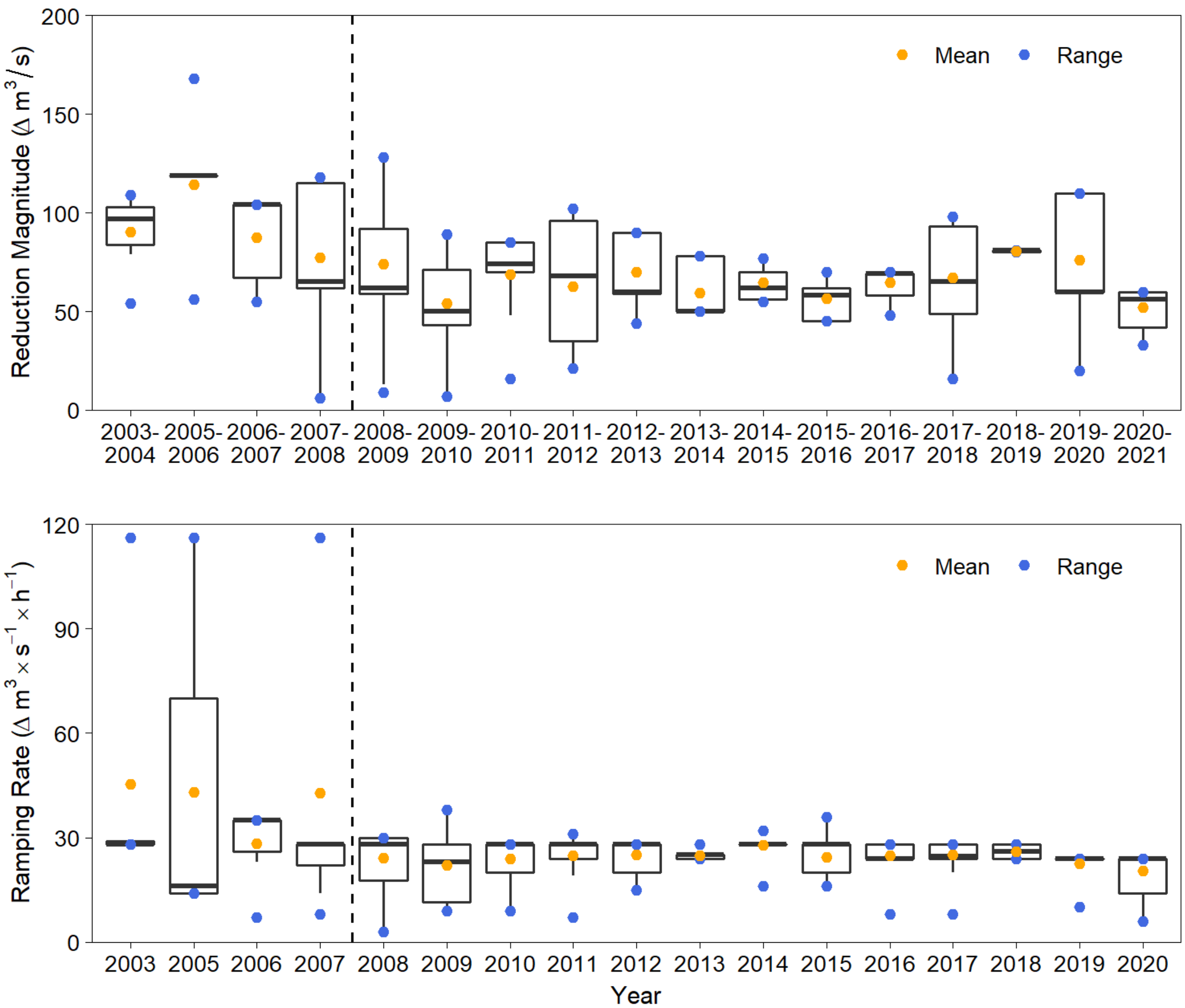


Figure 5: Boxplots of reduction magnitude ($\Delta m^3/s$; top panel) and ramping rates ($\Delta m^3\ s^{-1}\ h^{-1}$; bottom panel) by year. Each box represents the 25th and 75th quantiles (bottom and top lines, respectively), and the median (middle bold line); whiskers extend to 1.5 times the interquartile distance. Yearly mean, minimum, and maximum values are shown as individual points.

3.2 Fish Stranding Assessment Results (2006 to Present)

Fish stranding assessment results have been presented from 2006 to 2020 during a period of consistent and comparable assessment methodology. Results from assessments prior to 15 September 2006 were excluded from the dataset because the data were inconsistently collected. All fish encountered during the assessments were split into sportfish and non-sportfish categories for analysis (Table 3).

Table 3: Scientific names, species codes, and conservation status of fish encountered during fish stranding assessments on the lower Duncan River, September 2006 to December 2020.

Category	Species	Scientific Name	Species Code[a]	Species At Risk Act (SARA) Status	Provincial Conservation Status[b]
Sportfish	Rainbow Trout	*Oncorhynchus mykiss*	RB	Not SARA Listed	Not Listed
	Bull Trout	*Salvelinus confluentus*	BT	Not SARA Listed	Blue Listed
	Mountain Whitefish	*Prosopium williamsoni*	MW	Not SARA Listed	Yellow Listed
	Pygmy Whitefish	*Prosopium coulteri*	PW	Not SARA Listed	Yellow Listed
	Kokanee	*Oncorhynchus nerka*	KO	Not SARA Listed	Not Listed
	Burbot	*Lota lota*	BB	Not SARA Listed	Red Listed
Non-sportfish	Longnose Dace	*Rhinichthys cataractae*	LNC	Not SARA Listed	Yellow Listed
	Dace spp.	*Rhinichthys species*	DC	Not Applicable	Not Applicable
	Slimy Sculpin	*Cottus cognatus*	CCG	Not SARA Listed	Yellow Listed
	Torrent Sculpin	*Cottus rhotheus*	CRH	Not SARA Listed	Yellow Listed
	Prickly Sculpin	*Cottus asper*	CAS	Not SARA Listed	Yellow Listed
	Sculpin spp.	*Cottus species*	CC	Not Applicable	Not Applicable
	Sucker spp.	*Catostomus species*	SU	Not Applicable	Not Applicable
	Redside Shiner	*Richardsonius balteatus*	RSC	Not SARA Listed	Yellow Listed
	Northern Pikeminnow	*Ptychocheilus*	NSC	Not SARA Listed	Yellow Listed
	Peamouth	*Mylocheilus caurinus*	PCC	Not SARA Listed	Yellow Listed

[a] As defined by the BC *Ministry of Environment*.
[b] As defined by the British Columbia Conservation Data Center: https://www2.gov.bc.ca/gov/content/environment/plants-animals-ecosystems/conservation-data-centre.

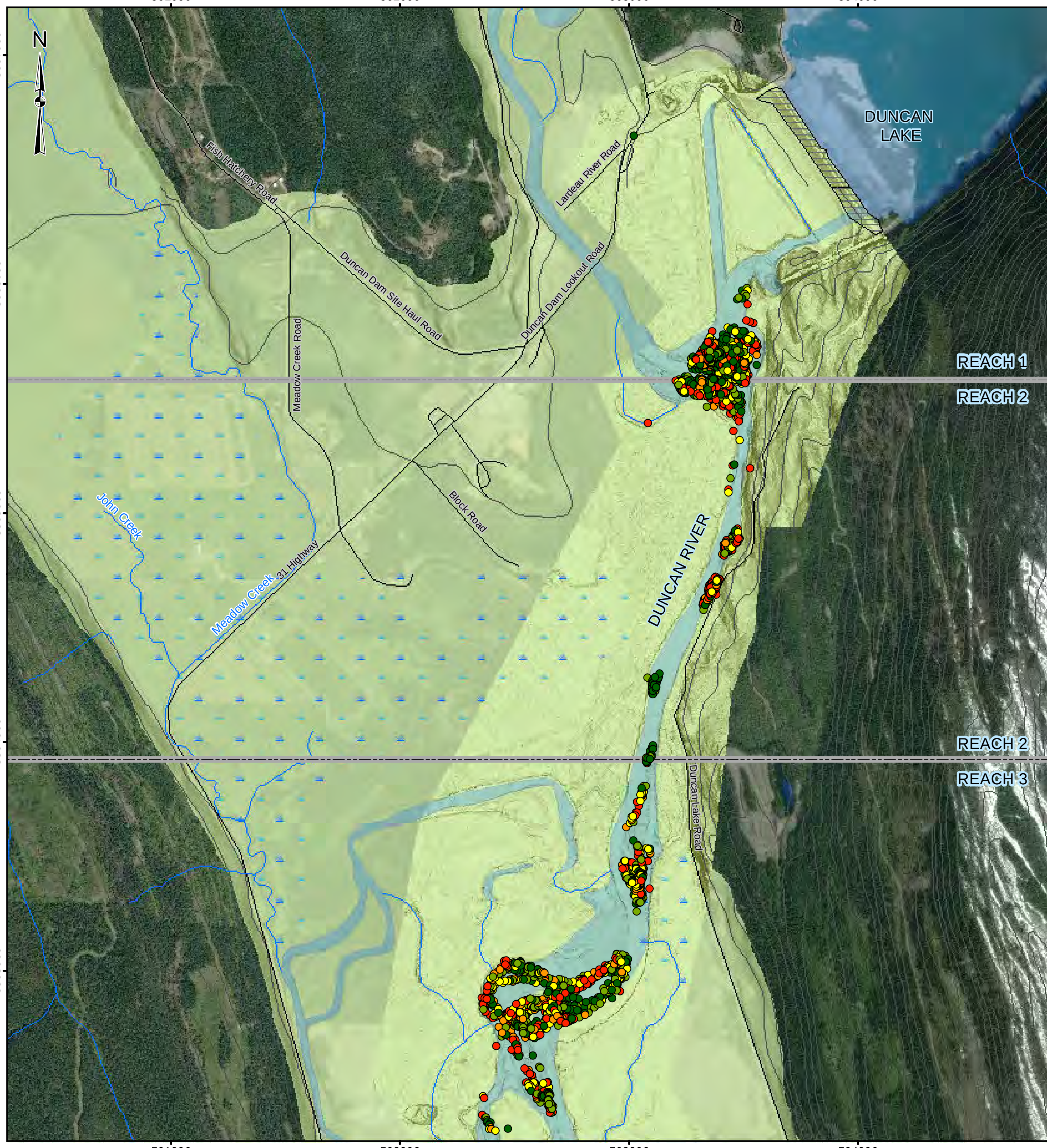
Within the dataset, the number of reduction events assessed for fish stranding per study year ranged from two (pre WUP 2006–2007) to eight (Study Year 1 2008–2009 and Study Year 10 2017–2018: Table 4). As discussed above, the focus of sampling shifted from index sites to random sites in Study Year 4 (2011–2012), which accounted for a larger proportion of random sites sampled in Study Years 5 to 13 (2012–2013 to 2020). In Study Year 10 (2017–2018) sampling effort in isolated pools was reduced to allow for more intensive interstitial sampling. The locations of all sampled stranding mechanisms within the dataset are presented in Figure 6 and Figure 7.

Over the course of the DDMMON-16 program (2008 – 2020), annual total stranding of all fish documented ranged from 246 fish in Study Year 8 to 2,268 fish in Study Year 13 (Table 5 and Figure 8). The single highest annual total stranding occurred in 2007 - 2008 (n = 2,409), the year prior to the onset of this WUP program. In the combined dataset, Kokanee (all life stages combined; n = 5,374) were the most numerous sportfish encountered, and had the highest total annual encounters (pre WUP 2007 - 2008; n = 1,690). Rainbow Trout were the second most numerous sportfish encountered (all years and life stages combined; n = 4,745). A total of 946 Mountain Whitefish were found stranded between 2006 and 2021, as were 77 juvenile Bull Trout (all life stages; Table 5 and Figure 8). Small numbers of Burbot and Pygmy Whitefish were encountered between 2006 and 2020 (n = 4 and n = 3, respectively; Table 5 and Figure 8). Except for Kokanee, encounters of stranded adult sportfish between 2006 and 2020 were extremely infrequent (Table 5 and Figure 8).

The most common non sportfish were Sculpin and Dace (all species combined) species in all years combined, with 2,281 and 1,993 of each species stranded, respectively (Table 5).

Table 4: All stranding assessment sampling effort during Duncan Dam flow reductions by study year.

DDMMON-16 Study Year	Number Assessed			Number of Stranding Mechanisms Sampled			
	Reductions	Index Sites	Random Sites	Pools	Interstitial Grids	Interstitial Transects	Censused Interstitial Areas
2006-2007	2	16	0	144	15	0	0
2007-2008	7	56	0	346	40	0	0
1 (2008-2009)	8	42	0	233	34	0	0
2 (2009-2010)	6	33	14	221	40	0	0
3 (2010-2011)	7	50	22	346	96	0	0
4 (2011-2012)	7	30	20	133	411	0	0
5 (2012-2013)	7	20	18	86	331	0	0
6 (2013-2014)	5	13	16	60	325	0	0
7 (2014-2015)	6	21	18	64	124	101	0
8 (2015-2016)	5	14	19	106	0	135	0
9 (2016-2017)	6	15	20	210	0	145	0
10 (2017-2018)	8	20	29	76	0	236	0
11 (2018-2019)	3	14	6	23	0	0	40
12 (2019-2020)	6	18	19	60	0	0	66
13 (2020)	6	22	15	31	0	0	36



LEGEND

STANDING MECHANISMS 2006-2020

SLOPE (%)

- 0-2
- 2-4
- 4-6
- 6-8
- >8

SLOPE (%)

High : 237

Low : 0

500 0 500

SCALE 1:25,000 METRES

REFERENCE

1) GROUND SURFACE: (A) 2012 TDUNC3 CONVERTED INTO RASTER, (B) DDM_RIVER_2018_DTM_ARC_ASCII_GRID_30CM, PROVIDED BY CLIENT.
2) CONTOUR, ROAD, WATERCOURSE, DAM, FOOTBRIDGE, TRAIL, WETLAND, WATERBODY OBTAINED FROM CANVEC © DEPARTMENT OF NATURAL RESOURCES CANADA. ALL RIGHTS RESERVED.
3) IMAGERY: DIGITALGLOBE, DATED 2011-05-11.

DATUM: NAD83 PROJECTION: UTM ZONE 11

PROJECT

BC HYDRO
DDMMON-16 LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING PROGRAM

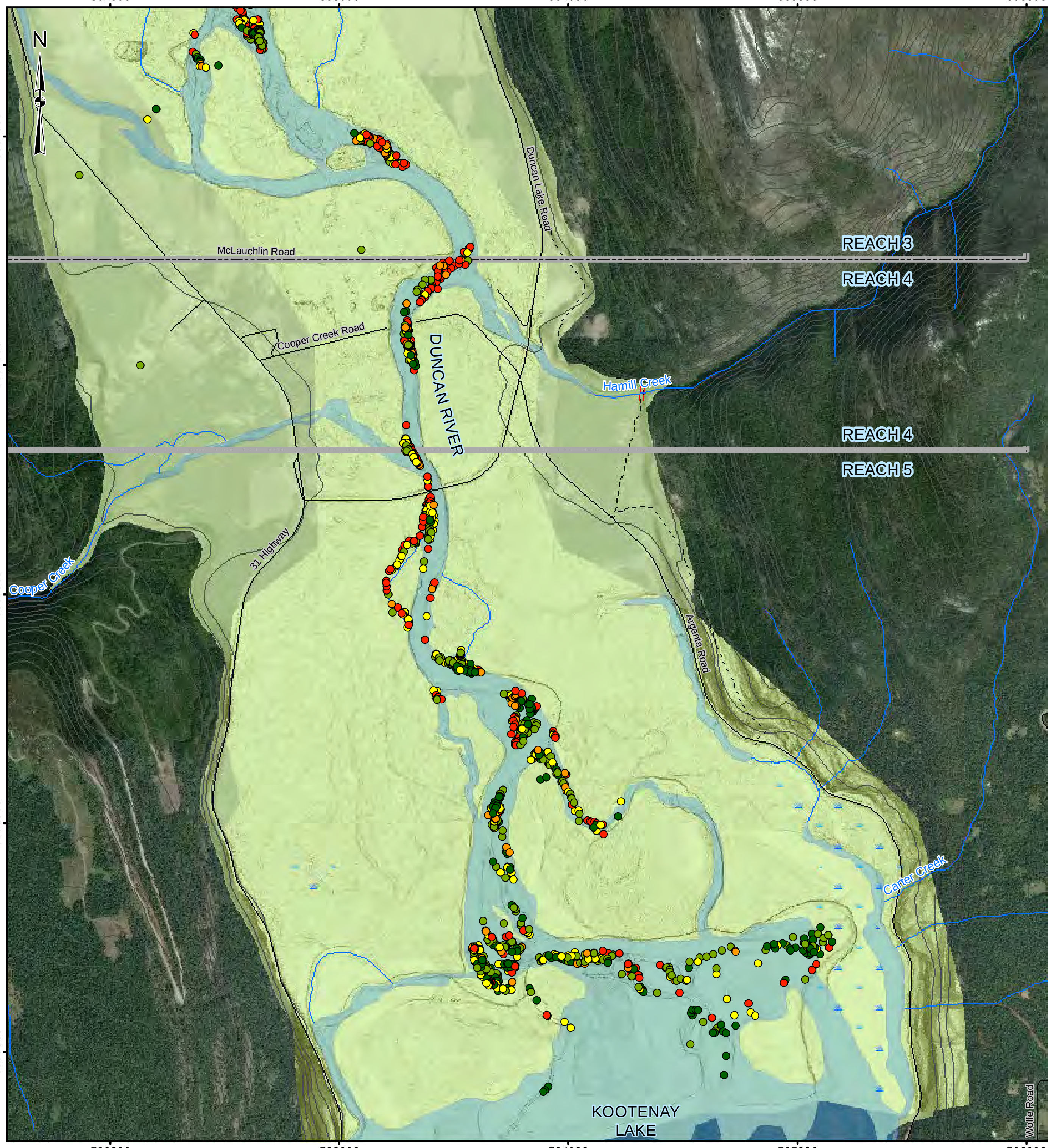
TITLE

LOCATION AND SLOPE (%) OF SAMPLED STRANDING MECHANISMS (SEPT. 2006 - APR. 2021) REACHES 1 TO 3

GOLDER

PROJECT		1810759	FILE No.	
DESIGN	BH	20 NOV 2020	SCALE AS SHOWN	REV. 0
GIS	JG/GS	28 JAN 2021	**FIGURE: 6**	
CHECK	BH	28 JAN 2021		
REVIEW	SR	24 MAR2021		

Document Path: \\golder.gds\gal\burnaby\CAD-GIS\Client\BC_Hydro\Duncan_Reservoir\99_PROJECTS\1810759\2021\Fig2_1214920117_Stranding Mechanism and slope_year13_REVA.mxd



LEGEND

STANDING MECHANISMS 2006-2020

SLOPE (%)

- 0-2
- 2-4
- 4-6
- 6-8
- >8

SLOPE (%)

High : 237

Low : 0

500 0 500

SCALE 1:25,000 METRES

REFERENCE

1) GROUND SURFACE: (A) 2012 TDUNC3 CONVERTED INTO RASTER, (B) DDM_RIVER_2018_DTM_ARC_ASCII_GRID_30CM, PROVIDED BY CLIENT.
2) CONTOUR, ROAD, WATERCOURSE, DAM, FOOTBRIDGE, TRAIL, WETLAND, WATERBODY OBTAINED FROM CANVEC © DEPARTMENT OF NATURAL RESOURCES CANADA. ALL RIGHTS RESERVED.
3) IMAGERY: DIGITALGLOBE, DATED 2011-05-11.

DATUM: NAD83 PROJECTION: UTM ZONE 11

PROJECT

BC HYDRO
DDMMON-16 LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING PROGRAM

TITLE

LOCATION AND SLOPE (%) OF SAMPLED STRANDING MECHANISMS (SEPT. 2006 - APR. 2021) REACHES 3 TO 5

GOLDER

PROJECT		18107549	FILE No.	
DESIGN	BH	28 MAY 2019	SCALE AS SHOWN	REV. 0
GIS	JG/GS	28 JAN 2021	**FIGURE: 7**	
CHECK	BH	28 JAN 2019		
REVIEW	SR	24 MAR 2021		

Document Path: \\golder.gds\gal\burnaby\CAD-GIS\Client\BC_Hydro\Duncan_Reservoir\99_PROJECTS\18107559\2021\Fig3_1214920117_Standing Mechanism and slope_year13_REVA.mxd

Table 5: Total number and relative percent composition in parentheses of fish species captured or observed during all stranding assessments conducted on the lower Duncan River from September 2006 to December 2020.

Species and Life Stage		N Fish (% of total within each year)														
		2006-2007	2007-2008	2008-2009	2009-2010	2010-2011	2011-2012	2012-2013	2013-2014	2014-2015	2015-2016	2016-2017	2017-2018	2018-2019	2019-2020	2020-2021
Sportfish																
Rainbow Trout	**Adult**	0	0	0	1 (0.1)	0	0	0	1 (0.2)	0	0	2 (0.1)	0	0	1 (0.1)	0
	Juvenile	130 (37.1)	278 (11.5)	530 (33.2)	113 (12.3)	343 (25.2)	452 (24.2)	332 (37.1)	241 (40.2)	737 (58.4)	52 (21.1)	164 (8.6)	122 (31.1)	362 (53)	233 (15.1)	890 (39.2)
Bull Trout	**Adult**	0	0	0	4 (0.4)	0	0	0	0	0	0	0	0	0	0	0
	Juvenile	2 (0.6)	0	11 (0.7)	1 (0.1)	6 (0.4)	2 (0.1)	3 (0.3)	2 (0.3)	16 (1.3)	1 (0.4)	4 (0.2)	1 (0.3)	0	1 (0.1)	23 (1.0)
Mountain Whitefish	**Adult**	0	1 (0)	0	0	0	0	0	0	0	0	0	0	0	0	0
	Juvenile	1 (0.3)	157 (6.5)	70 (4.4)	4 (0.4)	45 (3.3)	225 (12.1)	6 (0.7)	49 (8.2)	3 (0.2)	8 (3.3)	7 (0.4)	31 (7.9)	4 (0.6)	246 (15.9)	90 (4.0)
Pygmy Whitefish	**Adult**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Juvenile	0	0	0	1 (0.1)	2 (0.1)	0	0	0	0	0	0	0	0	0	0
Kokanee	**Adult**	0	97 (4)	572 (35.8)	112 (12.2)	42 (3.1)	55 (3)	111 (12.4)	0	0	0	0	0	0	0	257 (11.3)
	Juvenile	0	5 (0.2)	2 (0.1)	68 (7.4)	0	3 (0.2)	0	0	15 (1.2)	0	96 (5)	11 (2.8)	0	0	0
	YOY	0	1690 (70.2)	83 (5.2)	41 (4.5)	83 (6.1)	858 (46)	257 (28.7)	0	7 (0.6)	12 (4.9)	63 (3.3)	2 (0.5)	0	283 (18.3)	649 (28.6)
Burbot	**Adult**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Juvenile	0	0	1 (0.1)	0	0	1 (0.1)	1 (0.1)	0	0	0	0	0	1 (0.1)	0	0
Non-sportfish																
Longnose Dace		117 (33.4)	15 (0.6)	103 (6.5)	273 (29.7)	551 (40.5)	30 (1.6)	32 (3.6)	227 (37.8)	143 (11.3)	73 (29.7)	117 (6.1)	53 (13.5)	95 (13.9)	19 (1.2)	145 (6.4)
Dace spp.		0	0	0	12 (1.3)	1 (0.1)	0	0	0	0	0	1 (0.1)	0	0	0	0
Slimy Sculpin		0	13 (0.5)	11 (0.7)	62 (6.8)	39 (2.9)	6 (0.3)	0	1 (0.2)	12 (1)	11 (4.5)	101 (5.3)	40 (10.2)	13 (1.9)	56 (3.6)	42 (1.9)
Torrent Sculpin		0	1 (0)	1 (0.1)	0	0	3 (0.2)	0	0	0	0	4 (0.2)	1 (0.3)	1 (0.1)	0	0
Prickly Sculpin		0	0	0	0	2 (0.1)	0	0	0	2 (0.2)	0	1 (0.1)	2 (0.5)	2 (0.3)	1 (0.1)	0
Sculpin spp.		23 (6.6)	16 (0.7)	65 (4.1)	34 (3.7)	165 (12.1)	99 (5.3)	130 (14.5)	46 (7.7)	189 (15)	23 (9.3)	14 (0.7)	77 (19.6)	191 (28)	654 (42.4)	127 (5.6)
Sucker spp.		2 (0.6)	4 (0.2)	26 (1.6)	166 (18.1)	54 (4)	9 (0.5)	16 (1.8)	32 (5.3)	42 (3.3)	8 (3.3)	25 (1.3)	20 (5.1)	8 (1.2)	17 (1.1)	19 (0.8)
Redside Shiner		0	112 (4.6)	8 (0.5)	15 (1.6)	0	0	7 (0.8)	0	3 (0.2)	18 (7.3)	3 (0.2)	20 (5.1)	6 (0.9)	17 (1.1)	12 (0.5)
Northern Pikeminnow		0	0	2 (0.1)	0	15 (1.1)	7 (0.4)	1 (0.1)	1 (0.2)	0	8 (3.3)	1 (0.1)	1 (0.3)	0	0	0
Lake Chub		0	0	0	1 (0.1)	0	0	0	0	0	0	0	0	0	0	0
Peamouth		0	0	6 (0.4)	6 (0.7)	0	0	0	0	0	0	2 (0.1)	4 (1)	0	12 (0.8)	1 (<0.1)
Unidentified		75 (21.4)	20 (0.8)	105 (6.6)	4 (0.4)	13 (1)	114 (6.1)	0	0	92 (7.3)	31 (12.6)	1310 (68.4)	7 (1.8)	0	4 (0.3)	13 (0.6)
All Species Total		350	2409	1596	918	1361	1864	896	600	1261	246	1915	392	683	1544	2,268

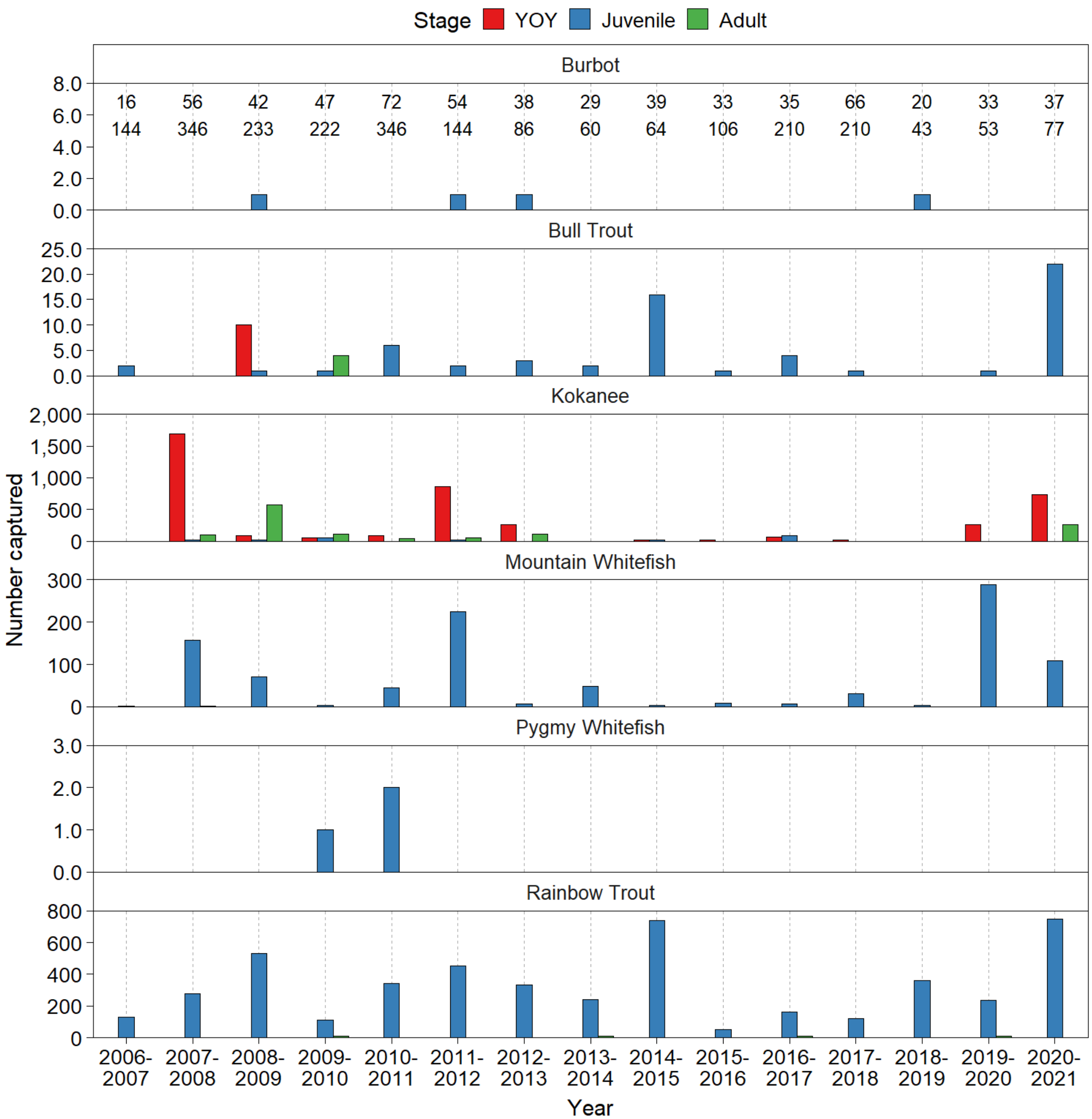


Figure 8: Abundances of sportfish species, separated by life stage, observed in stranding assessments between 2006 and 2020. Note the different y-axis scales among panels. On the uppermost panel, the numbers of sampled sites and pools are provided in the first and second lines, respectively.

In the combined dataset (2006 – 2020), a total of the 1,443 juvenile Rainbow Trout were measured for fork length during assessments. Fork lengths ranged from 21 mm to 200 mm FL (median = 42 mm FL). The largest proportion of the measured catch was within the 50 – 60 mm FL size interval (Figure 9). The substantial mode between 30 – 80 mm FL size-range represented the age-0 cohort. A total of 169 juvenile Mountain Whitefish were measured for fork length during stranding assessments (all years combined). Lengths ranged from 17 mm to 100 mm FL (median = 76 mm FL). For all years combined, there were two modes: one in the 20-40 mm FL size-range and another in the 60-100 mm FL size-range. These two modes represented young of the year Mountain Whitefish and older individuals in the age-0 cohort, respectively.

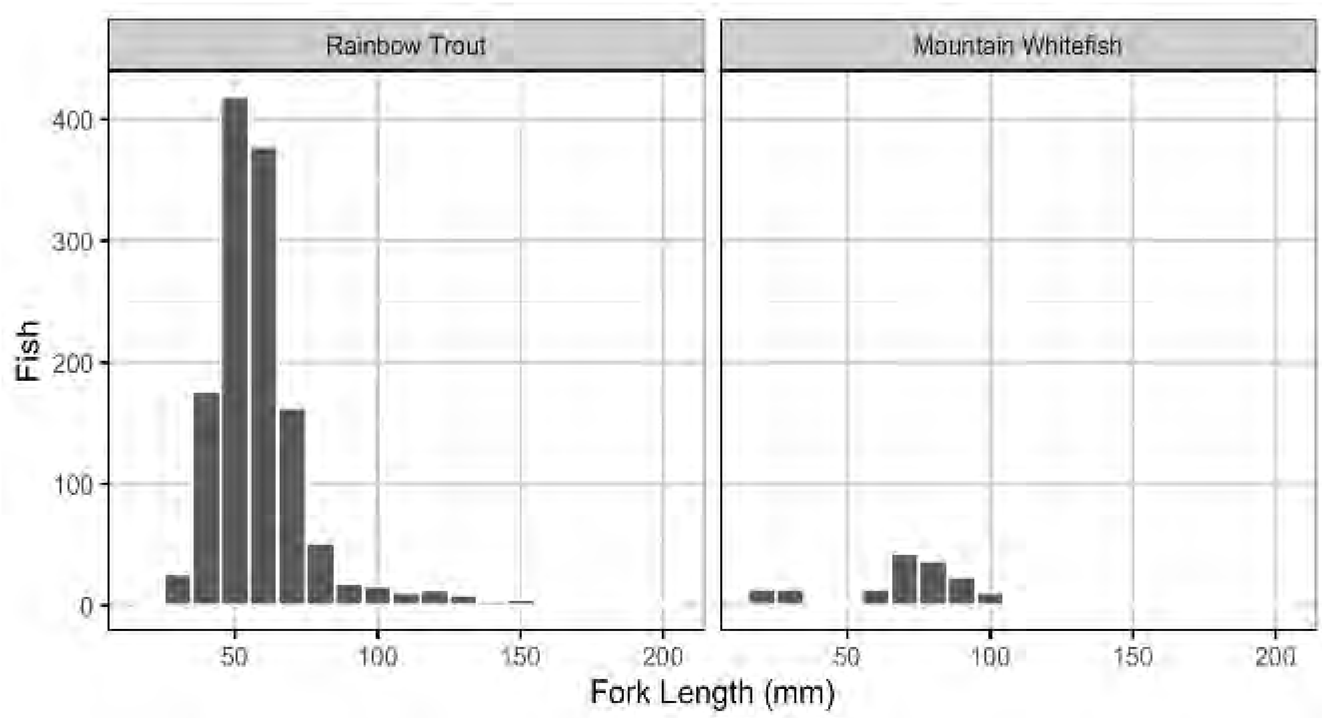


Figure 9: Length frequency distribution of stranded juvenile target species (Rainbow Trout and Mountain Whitefish) observed between 2006 to 2021.

3.3 Fish Abundance Assessment

Fall target species abundance assessment snorkel sites from study Years 6 - 9 are presented in Figure 10 to Figure 13. Total number of sites sampled per year ranged from 34 in Study Year 6 (2013–2014) to 57 sites during the Study Year 9 (2016–2017) snorkeling fish abundance assessments, with the highest counts occurring in Study Year 7 (2014–2015: Table 6). The variability in total numbers of the target species observed in each stratum was high within and between years, and in all years Mountain Whitefish were the most numerous target species observed. The lowest mean counts of Mountain Whitefish (5.1 fish/site) were recorded in shallow/slack sites in Study Year 6 (2013–2014), whereas deep/slack sites in Study Year 7 (2014–2015: 75.4 fish/site) had the highest mean counts (Table 6). The lowest mean counts of Rainbow Trout (2.1 fish/site) were recorded in deep, slack sites in Study Year 9 (2016–2017), whereas shallow/fast sites in Study Year 7 (2014–2015: 53.1 fish/site) had the highest Rainbow Trout mean counts, with 11.6 fish/site at slack sites, and 4.1 fish/site at fast sites (Table 6).

Table 6: Summary of fish counts across depth and flow strata, as recorded from Study Year 6 to 9 (September 2013 to September 2016) snorkel surveys.

Study Year	Stratum	Number of Sites (*n*)	Mountain Whitefish			Rainbow Trout		
			n	Mean	SD	*n*	Mean	SD
Year 6 (2013–2014)	Shallow/Slack	13	260	20.0	39.5	101	7.8	10.1
	Shallow/Fast	10	225	22.5	36.2	104	10.4	15.9
	Deep/Fast	11	144	13.1	14.1	36	3.3	3.2
Year 6 (2013–2014) Total		**34**	**629**			**241**		
Year 7 (2014–2015)	Shallow/Slack	9	161	17.9	28.9	134	14.9	13.5
	Shallow/Fast	16	281	17.6	23.9	849	53.1	45.3
	Deep/Slack	8	603	75.4	104.8	21	2.6	2.4
	Deep/Fast	16	325	20.3	18.6	128	8.0	9.2
Year 7 (2015–2015) Total		**49**	**1,370**			**1,132**		
Year 8 (2015–2016)	Shallow/Slack	11	56	5.1	10.2	129	11.7	28.3
	Shallow/Fast	16	265	16.6	34.5	179	11.2	12.6
	Deep/Slack	11	176	17.6	19.6	52	5.2	6.1
	Deep/Fast	8	185	23.1	22.8	49	6.1	6.0
Year 8 (2015–2016) Total		**46**	**682**			**409**		
Year 9 (2016–2017)	Shallow/Slack	14	125	8.9	12.8	57	4.1	5.0
	Shallow/Fast	17	154	9.1	11.4	198	11.6	17.4
	Deep/Slack	11	133	12.1	18.7	26	2.4	5.9
	Deep/Fast	15	213	14.2	9.9	32	2.1	2.4
Year 9 (2016–2017) Total		**57**	**625**			**313**		

LEGEND

SNORKEL SITE LOCATION (2013)
- START LOCATION
- STOP LOCATION
- MAIN CHANNEL
- SIDE CHANNEL
- REACH BREAK
- CONTOUR (30 m INTERVAL)
- WATERCOURSE
- FOOTBRIDGE
- TRAIL (OTHER)
- ROAD SEGMENT
- WETLAND
- DAM
- WATERBODY

1 0 1

SCALE 1:50,000 KILOMETRES

REFERENCE

1) IMAGERY: DIGITALGLOBE DATED 2011-05-11.
2) CONTOUR, ROAD, WATERCOURSE, DAM, FOOTBRIDGE, TRAIL, WETLAND, WATERBODY OBTAINED FROM CANVEC © DEPARTMENT OF NATURAL RESOURCES CANADA. ALL RIGHTS RESERVED.

DATUM: NAD83 PROJECTION: UTM ZONE 11

PROJECT

BC HYDRO
DDMMON-16 LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING PROGRAM

TITLE

SNORKEL SITE LOCATIONS (SEPT. 2013)

GOLDER

PROJECT		12-1492-0117	FILE No.	
DESIGN	BH	14 Apr 2014	SCALE AS SHOWN	REV. 0
GIS	KLN	14 Jun 2018	FIGURE: 10	
CHECK	BH	05 Dec 2018		
REVIEW	SR	05 Dec 2018		

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LEGEND

SNORKEL SITE LOCATION (2014)

- START LOCATION
- STOP LOCATION
- MAIN CHANNEL
- SIDE CHANNEL
- REACH BREAK
- CONTOUR (30 m INTERVAL)
- WATERCOURSE
- FOOTBRIDGE
- TRAIL (OTHER)
- ROAD SEGMENT
- WETLAND
- DAM
- WATERBODY

1 0 1

SCALE 1:50,000 KILOMETRES

REFERENCE

1) IMAGERY: DIGITALGLOBE, DATED 2011-05-11.
2) CONTOUR, ROAD, WATERCOURSE, DAM, FOOTBRIDGE, TRAIL, WETLAND, WATERBODY OBTAINED FROM CANVEC © DEPARTMENT OF NATURAL RESOURCES CANADA. ALL RIGHTS RESERVED.

DATUM: NAD83 PROJECTION: UTM ZONE 11

PROJECT

BC HYDRO
DDMMON-16 LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING PROGRAM

TITLE

SNORKEL SITE LOCATIONS (SEPT. 2014)

GOLDER

PROJECT		12-1492-0117	FILE No.
DESIGN	BH	14 APR 2014	SCALE AS SHOWN / REV. 0
GIS	JG	14 JUN 2018	**FIGURE: 11**
CHECK	BH	05 DEC 2018	
REVIEW	SR	05 DEC 2018	

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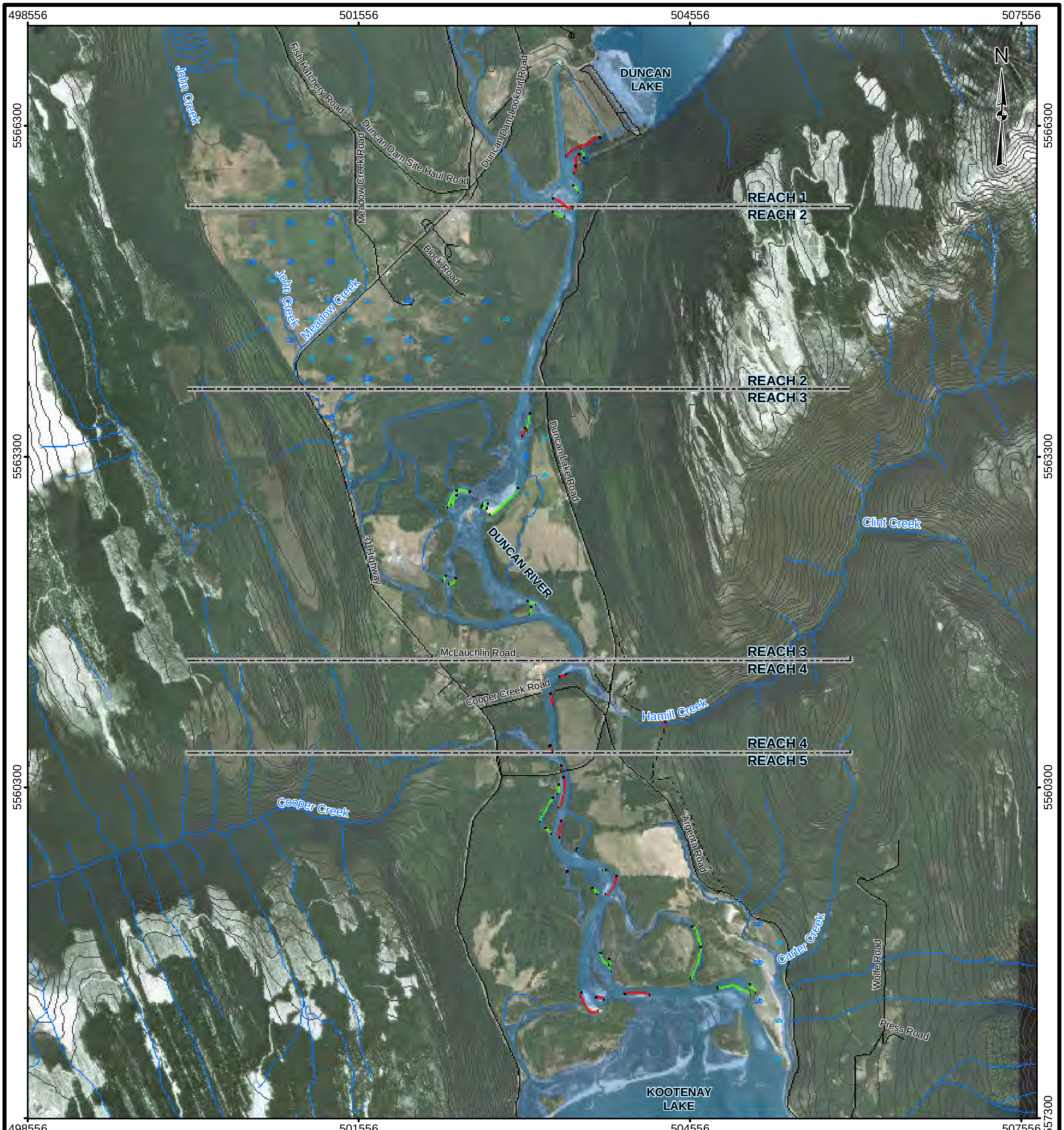

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LEGEND

SNORKEL SITE LOCATION (2015)

- START LOCATION
- STOP LOCATION
- SIDE CHANNEL
- MAIN CHANNEL
- REACH BREAK
- WATERCOURSE
- FOOTBRIDGE
- TRAIL (OTHER)
- ROAD SEGMENT
- CONTOUR (30 m INTERVAL)
- WETLAND
- DAM
- WATERBODY

1 0 1

SCALE 1:50,000 KILOMETRES

REFERENCE

1) BACKDROP IMAGERY: DIGITALGLOBE DATED 2011-05-11.
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DATUM: NAD83 PROJECTION: UTM ZONE 11

PROJECT

BC HYDRO
DDMMON-16 LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING PROGRAM

TITLE

SNORKEL SITE LOCATIONS (SEPT. 2015)

GOLDER

PROJECT		1535517	FILE No.
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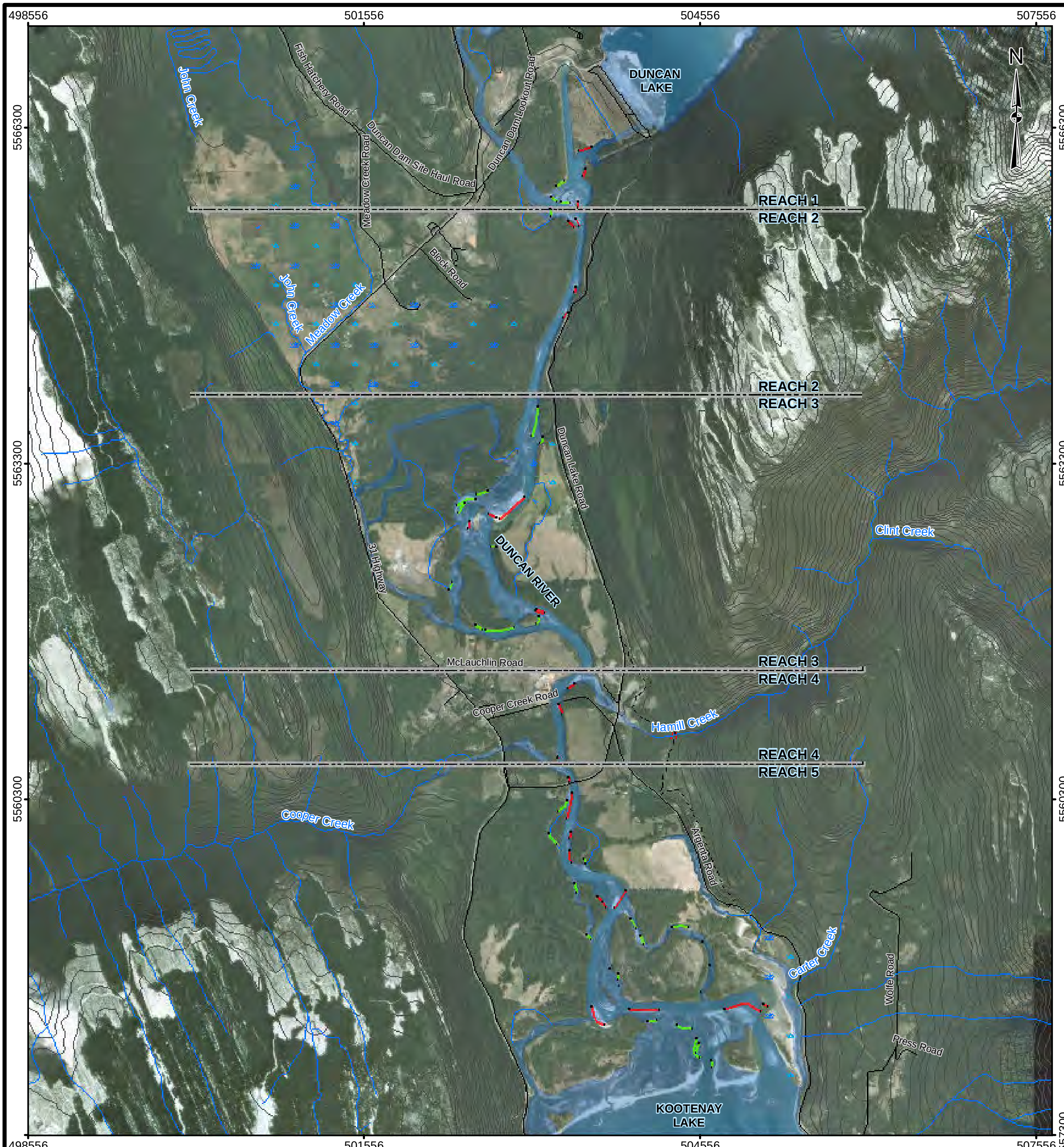
GIS	JG	14 JUN 2018	
CHECK	BH	05 Dec 2018	**FIGURE: 12**
REVIEW	SR	05 Dec 2018	



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LEGEND

SNORKEL SITE LOCATION (2016)

- START LOCATION
- STOP LOCATION
- SIDE CHANNEL
- MAIN CHANNEL
- REACH BREAK
- WATERCOURSE
- FOOTBRIDGE
- TRAIL (OTHER)
- ROAD SEGMENT
- CONTOUR (30 m INTERVAL)
- WETLAND
- DAM
- WATERBODY

1 0 1

SCALE 1:50,000 KILOMETRES

REFERENCE

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DATUM: NAD83 PROJECTION: UTM ZONE 11

PROJECT

BC HYDRO
DDMMON-16 LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING PROGRAM

TITLE

SNORKEL SITE LOCATIONS (SEPT. 2016)

GOLDER

PROJECT		1535517	FILE No.	
DESIGN	BH	14 JUN 2018	SCALE AS SHOWN	REV. 0
GIS	JG	14 JUN 2018	**FIGURE: 13**	
CHECK	BH	05 DEC 2018		
REVIEW	SR	05 DEC 2018		

Fish densities recorded in 2016 were generally lower than those from 2013 and 2014, but comparable to densities recorded in 2015 (Figure 14). Mountain Whitefish densities were high in deep, fast habitats, similar in both shallow and deep slack habitats, and lowest in shallow fast habitats. Rainbow Trout densities were lowest at deep, fast areas, and highest in shallow, slack areas.

The variability of fish density within strata was high throughout the years (Figure 14). In 2016, Mountain Whitefish zero densities accounted for 12% to 29% of the cases, calculated across strata. Non-zero densities ranged from 0.001 fish/m² to 0.09 fish/m². Rainbow Trout zero densities accounted for 14% to 64% or the cases, calculated across strata. Non-zero densities ranged from 0.0004 fish/m² to 0.06 fish/m².

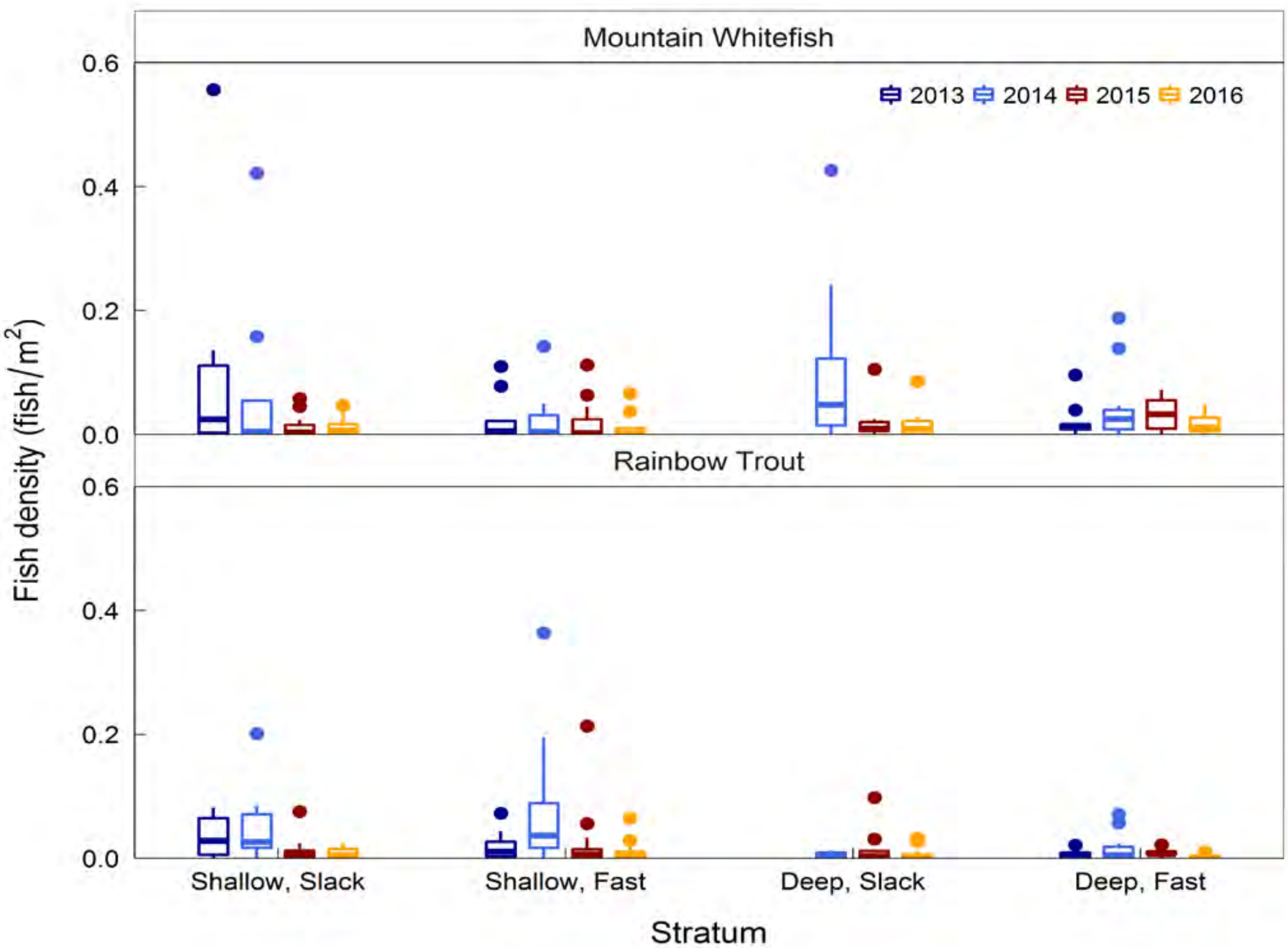

Figure 14: Boxplots of density (fish/m²) across species, depth, and flow strata for 2013-2016 data from fall snorkel surveys. Each box represents the 25th and 75th quantiles (bottom and top lines, respectively), and the median (middle bold line); whiskers extend to 1.5 times the interquartile distance; outliers are shown as individual points.

Overall, spring Rainbow Trout abundance estimates have fluctuated substantially since 2010. Abundance decreased between 2010 and 2015, and from 2017 to 2018 (Table 7 and Figure 15). Abundance increased in 2016, 2017, and 2019. The fall age-0 Rainbow Trout abundance estimates were similar to the spring age-1 Rainbow Trout abundance estimates in 2015 and were lower in 2013 and 2016 (Table 7 and Figure 15).

The fall total abundance estimates for Rainbow Trout ranged from 4,362 in 2016 to 48,981 in 2010 (Table 7 and Figure 15: Golder 2018). Overall, abundance decreased from 2010 to 2013. Abundance increased for the fall of 2014, followed by decreases in the two subsequent years. Estimated Mountain Whitefish fall abundance ranged from 21,691 in 2016 to 49,496 in 2013. Generally, Mountain Whitefish fall abundance increased between 2010 and 2013, and then remained relatively stable between 2013 and 2014. This was followed by a substantial decrease in 2015, and relative stability between the 2015 and 2016 fall abundance (Table 7 and Figure 15; Golder 2018).

Table 7: Total annual abundance estimates of Mountain Whitefish and Rainbow Trout. Abundances are mean Bayesian estimates, with lower and upper 95% credibility intervals in parentheses; numbers are rounded to nearest fish.

Study year	Abundance Estimation Using Spring Snorkel Surveys [a]		Abundance Estimate Using Fall Snorkel Surveys	
	Rainbow Trout	Mountain Whitefish	Rainbow Trout [b]	Mountain Whitefish [b]
Year 3 (2010-2011)	49,246 (32,220 – 77441)	-	48,981 (30,828 – 73,594)	38,625 (28,269 – 52,641)
Year 4 (2011-2012)	46,022 (30,965 – 61,155)	-	-	-
Year 5 (2012-2013)	29,376 (20,223 – 43,056)	-	-	-
Year 6 (2013-2014)	20,750 (14,183 – 30,918)	-	12,225 (6,105 – 22,595)	49,496 (24,852 – 97,746)
Year 7 (2014-2015)	-	-	24,216 (14,464 – 39,757)	46,023 (25,711 – 78,616)
Year 8 (2015-2016)	7,606 (5,074 – 11,721)	-	8,627 (4,844 – 14,992)	21,691 (11,721 – 37,924)
Year 9 (2016-2017)	14,978 (10,258 – 22,047)	-	4,362 (2,627 – 7,178)	22,251 (13,203 – 36,150)
Year 10 (2017-2018)	26,619 (18,010 – 40,092)	-	-	-
Year 11 (2018-2019)	8,308 (5,598 – 12,957)	-	-	-
Year 12 (2019-2020)	20,565 (13,801 – 32,293)	-	-	-

[a] Obtained from Andrusak and Thorley 2019.

[b] 2010 estimates obtained from Thorley et al. 2011.

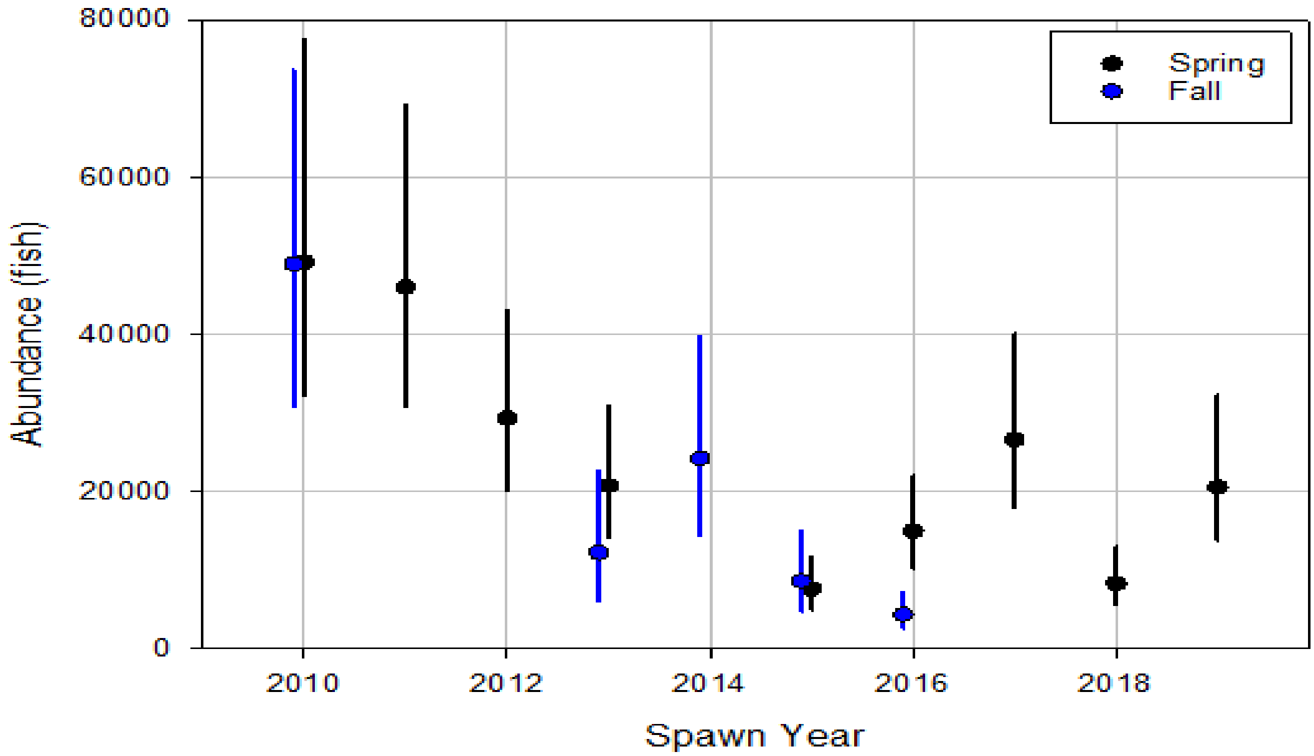

Figure 15: Estimated abundance of Rainbow Trout by spawn year and season in the lower Duncan River (with 95% CIs).

3.4 Stranding Estimation

The presentation of data regarding stranding assessment results includes both target species. As the impacts of flow regulation are not considered significant to juvenile Mountain Whitefish and will likely not result in population level effects (Golder 2018, Golder and Poisson 2019a, 2019b and 2020), pool and interstitial stranding estimation in the following sections refer only to Rainbow Trout.

3.4.1 Analysis of Slope

The elevations and slope categories selected for GIS modelling, as well as the estimated wetted area for each category are presented in Table 8 and Figure 16. Habitat with less than 0 – 2% slope were the most abundant in all examined DRL discharges, followed by habitats with slopes between 2 – 4% and >8%.

Table 8: Estimated wetted area (m^2) by slope in the lower Duncan River, based on DRL discharge.

Discharge at DRL (m^3 s^{-1})	Slope Category (%)					
	<2	2-4	4-6	6-8	>8	Total
68.0	306,250	274,138	146,194	82,319	155,731	964,631
73.0	316,294	280,594	149,625	845,19	160,238	991,269
110.8	373,588	324,356	172,313	981,38	188,556	1,156,950
148.6	408,225	342,263	182,119	104,300	202,800	1,239,706
186.4	494,313	380,063	203,156	115,956	239,725	1,433,213
224.2	536,744	399,731	213,956	121,963	262,450	1,534,844
262.0	577,731	414,488	222,438	127,869	283,700	1,626,225
299.8	614,038	426,781	229,994	133,269	303,600	1,707,681
337.6	656,163	439,194	237,981	138,488	322,488	1,794,313
375.4	697,388	452,850	245,938	143,906	340,850	1,880,931
390.2	713,631	458,494	249,138	145,763	347,669	1,914,694
428.0	747,175	473,900	256,944	150,463	363,619	1,992,100
465.8	784,906	488,938	264,138	154,663	377,888	2,070,531
488.0	809,075	497,150	267,881	156,738	385,369	2,116,213

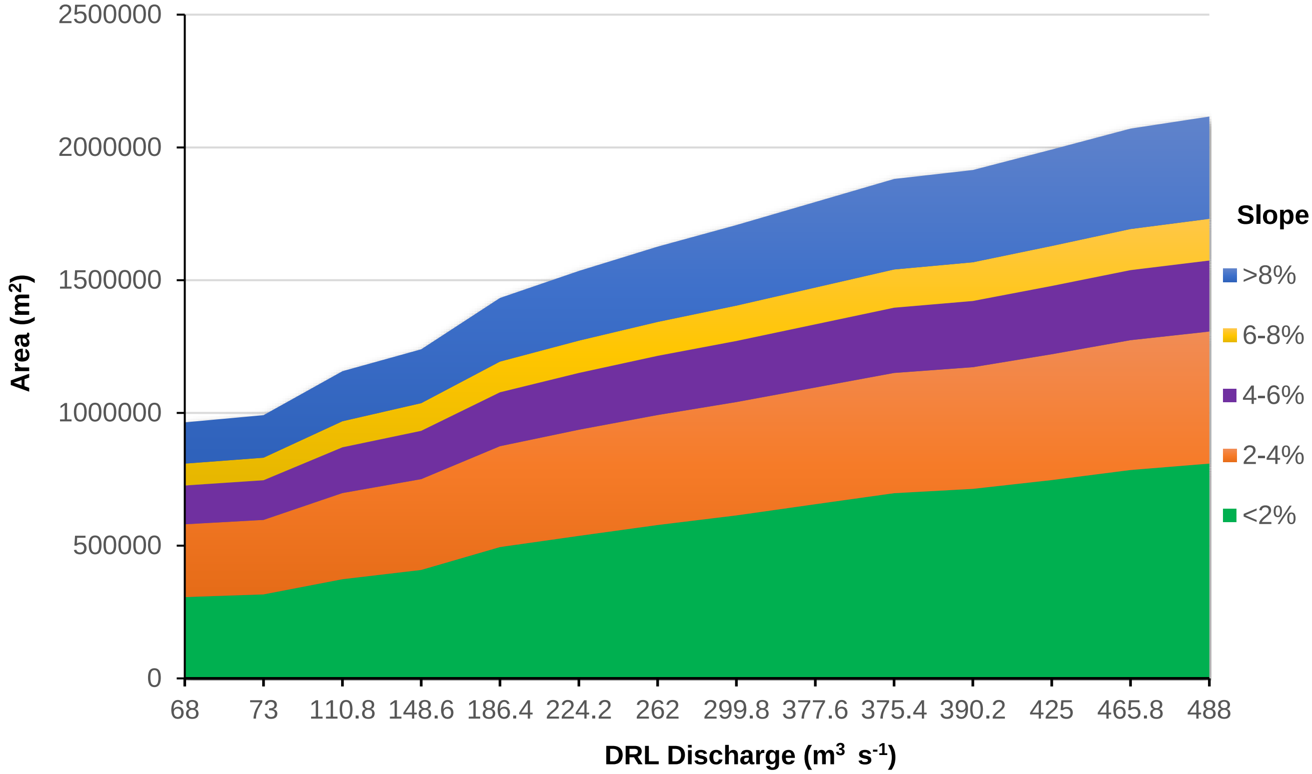

Figure 16: Cumulative wetted area in the Lower Duncan River by slope and DRL discharge.

The slope of each stranding mechanism throughout 11 years of stranding assessments (Study Years 3 to 13: 2010 – 2020) was calculated using the elevation models for the study area. Slopes ranged from 0% to 59%, however all values above 20% (a total of 22 cases) were deemed artifacts of the elevation model and were removed from analysis.

Pool density (calculated by summing assessed pools during stranding assessments) is the count of the number of isolated pools per slope designation (%) that formed following flow reduction events and their formation relates to risk of stranding. Generally, pool density was slightly higher at lower slope values (0% to 5%); however, the relationship was variable and weak (Figure 17). While pool densities in random sites exhibited slightly higher variation in comparison to index sites in some years (i.e., 2010, 2016, 2017), recorded pool densities at random sites were low, often lower than those recorded at index sites (Figure 17). The number of fish stranded per pool was similar throughout the different slopes (Figure 18). Most of the documented interstitially stranded fish (89%) were found on exposed areas with low slopes (≤9%; Figure 19). Estimated Interstitial stranding density of Rainbow Trout increased as slopes increased between 0% and 6% (Figure 20). As slope increased above 6%, the risk of interstitial stranding was found to decrease. All Mountain Whitefish encountered interstitially stranded were on substrates with gradients between 2 and 4% (Figure 20).

A relationship between slope and randomly sampled pool or interstitial stranding rates was not found in the combined dataset; therefore, slope was removed as a predictor of stranding during the current estimation modelling.

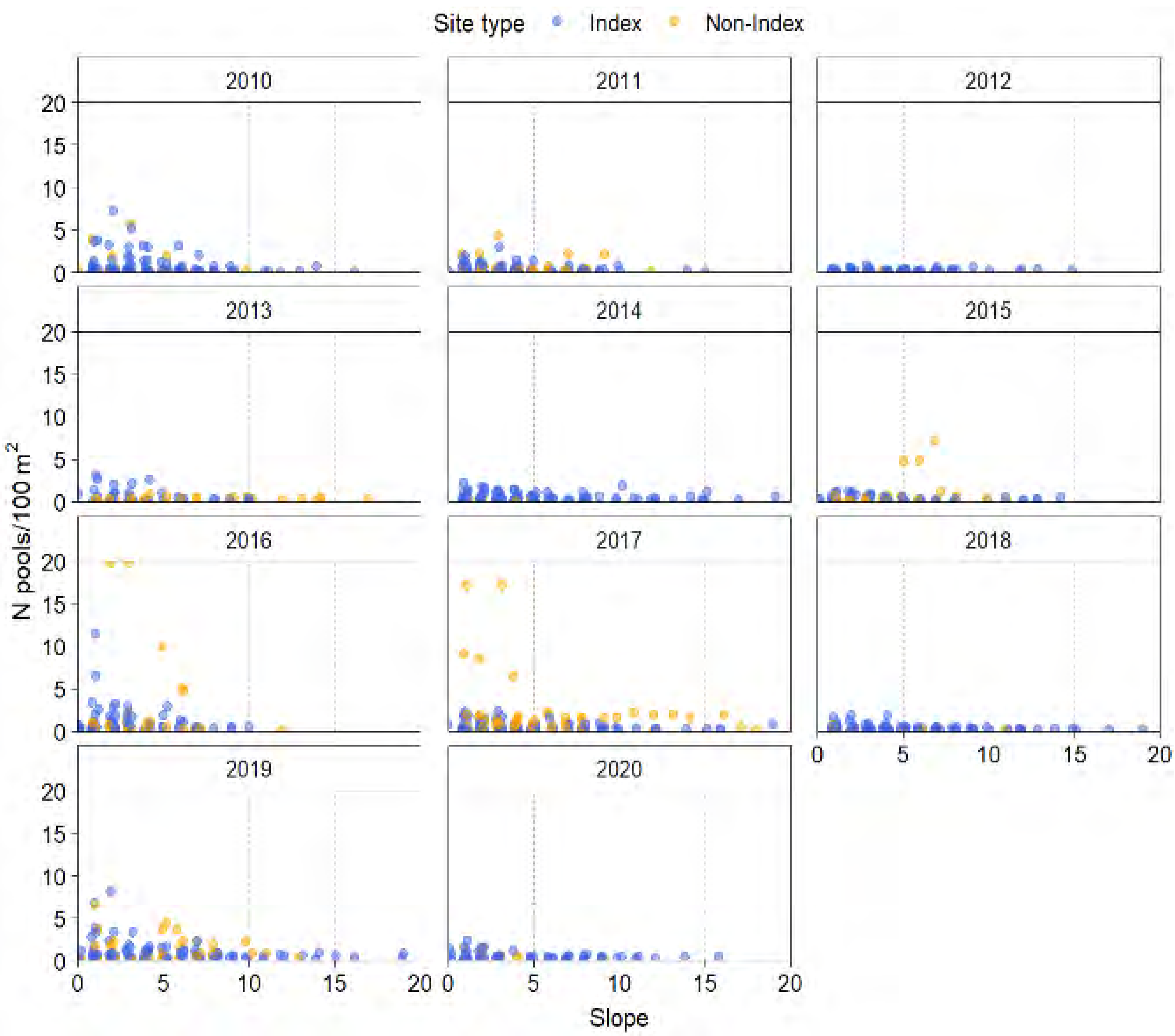

Figure 17: Density of pools recorded per reduction versus slope as a continuous variable, 2010-2020.

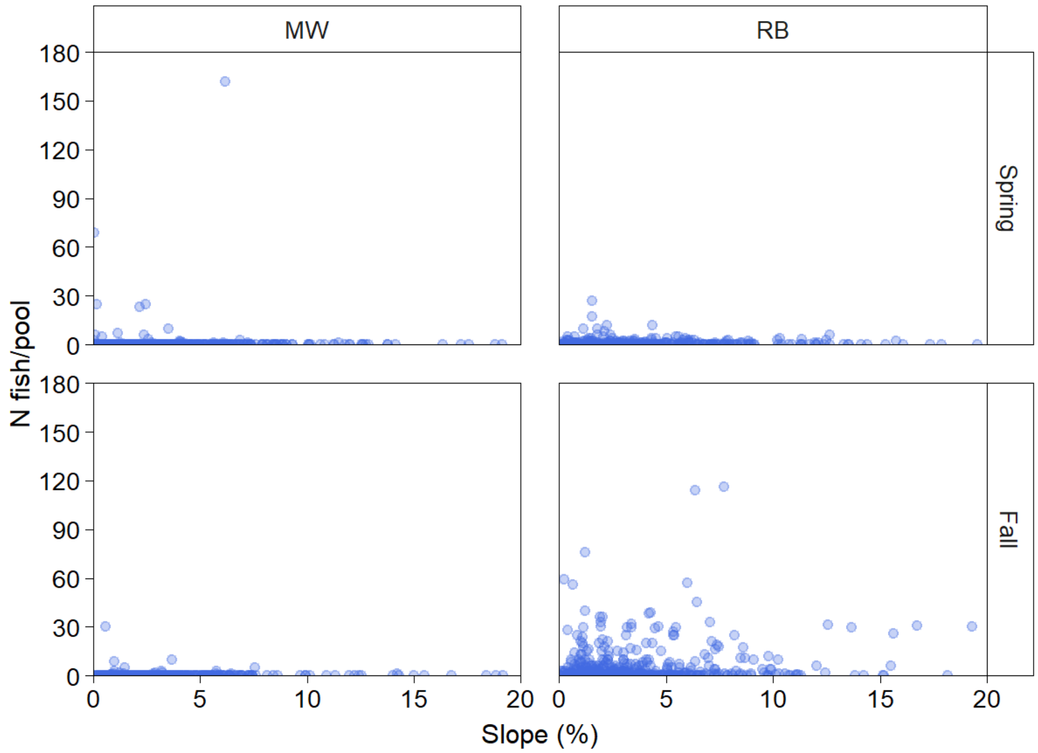

Figure 18: Number of collected fish per pool by slope, 2010–2020.

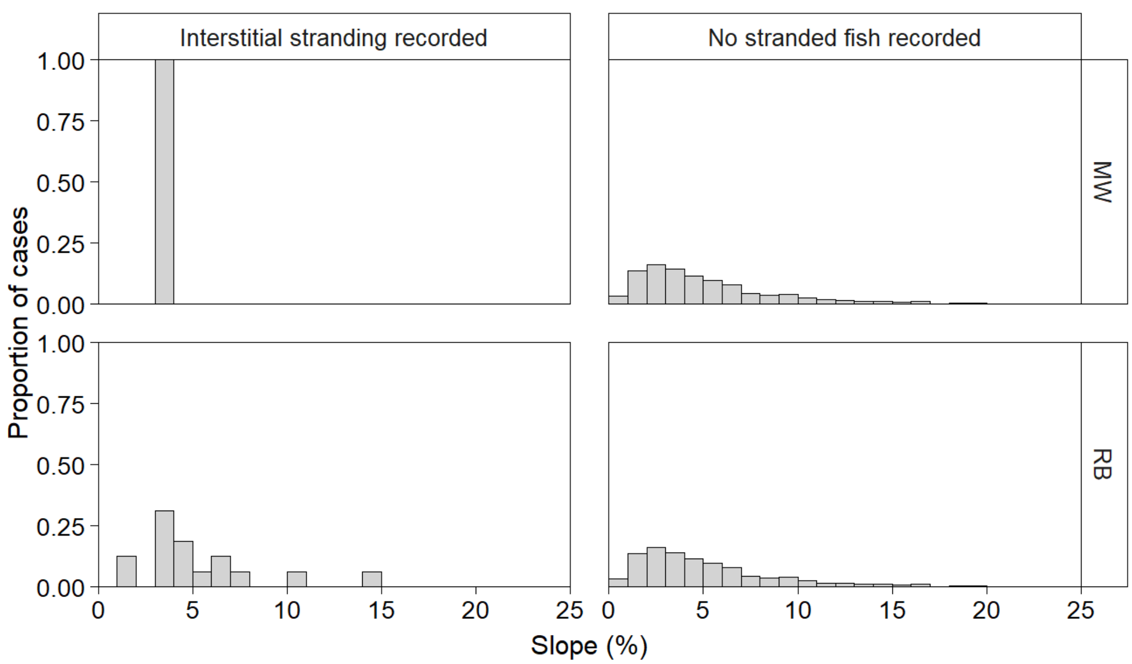

Figure 19: Histogram of 2011–2020 interstitially stranded Mountain Whitefish and Rainbow Trout in the lower Duncan River, plotted by species and slope (%).

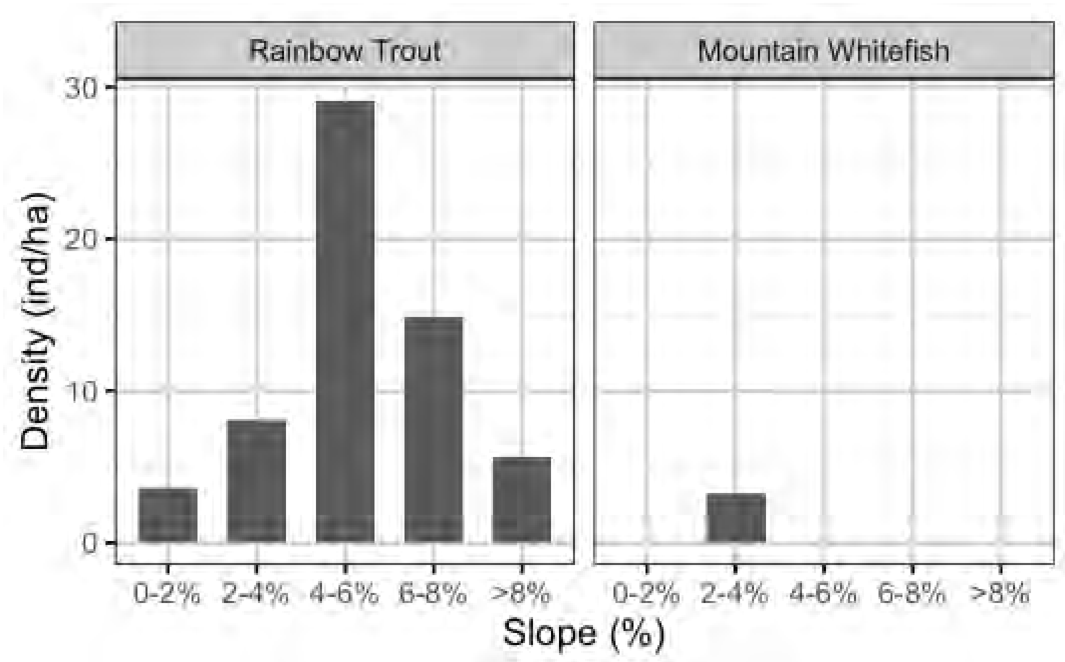

Figure 20: Interstitial stranding density by slope category and species.

3.4.2 Pool Stranding

Mountain Whitefish pool stranding density was low across all substrate sizes (Figure 21). Rainbow Trout densities were similar across different substrate sizes, except for small cobble, where stranded fish densities were lowest. Mean Rainbow Trout densities were highest in pools with substrate of very large gravel (2.9 fish/m²), large gravel (0.8 fish/m²), and sand (0.7 fish/m²; Figure 21). A relationship between substrate size and pool stranding rates was not found in the combined dataset.

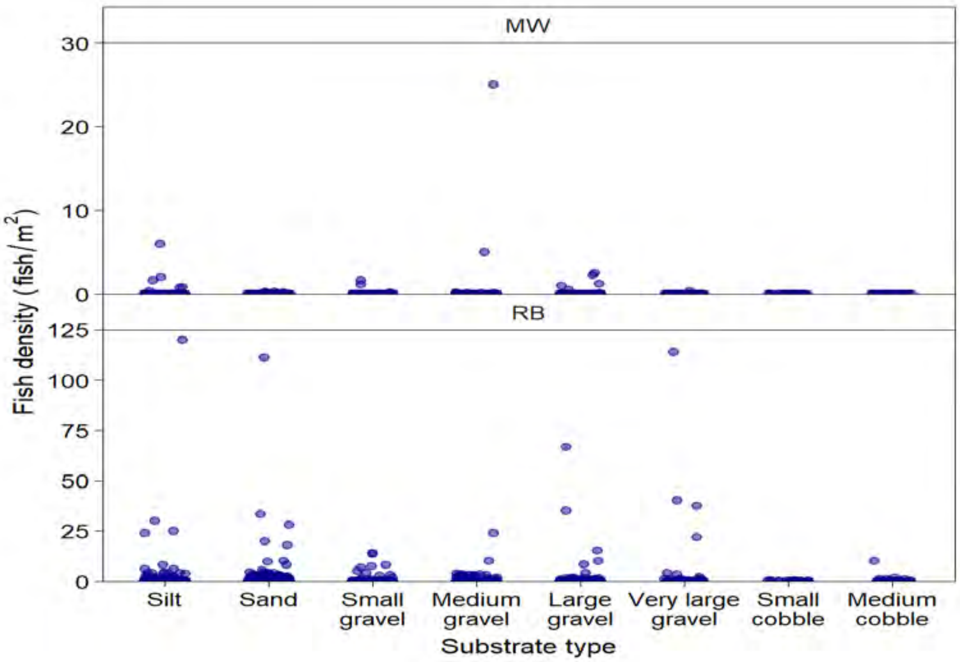

Figure 21: Scatter plot of pool-stranded fish density (fish/m²) versus dominant pool substrate size, 2006–2020, plotted by target species.

The effect of season on pool stranding of Rainbow Trout juveniles was significant, with the mean fall pool abundance estimates approximately seven times higher than those for winter/spring (Figure 22). The mean number of Rainbow Trout juveniles per pool in the fall (July to December) was estimated at 5.20 (CRI of 3.84 – 7.32), while spring season (January to June) juveniles per pool was estimated to be 0.76 fish/pool (CRI of 0.57 – 1.08; Figure 22).

Based on the seasonal effect on abundance of Rainbow Trout juveniles in pools and the predicted pool abundance, it was then possible to estimate the number of fish stranded in pools for individual reduction events (Figure 23). Generally, spring Rainbow Trout estimates of pool stranding were lower than fall estimates. Fall pool estimates were highest at index sites (i.e., M0.8R, Lard0.3R, S3.5-4.0R, S4.0-4.2R, S4.1L, S6.9R, and S9.2L; Figure 22 and Figure 23).

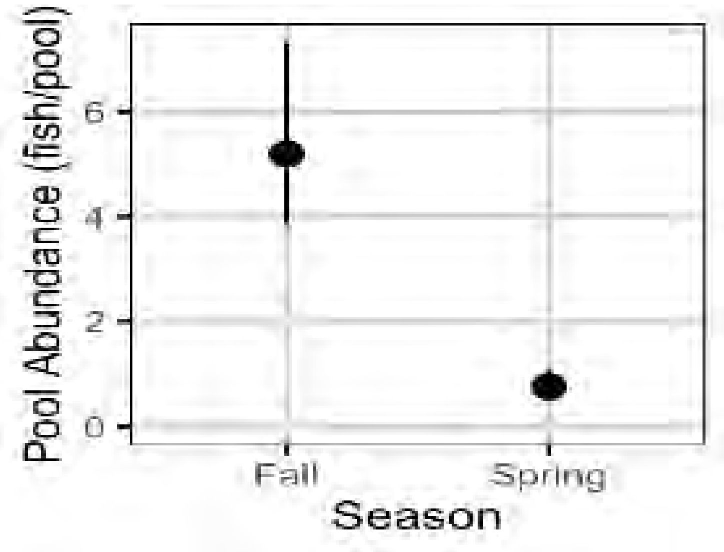

Figure 22: The predicted abundance of Rainbow Trout in pools during a typical reduction event by season in the lower Duncan River. Error bars are 95% credibility intervals.

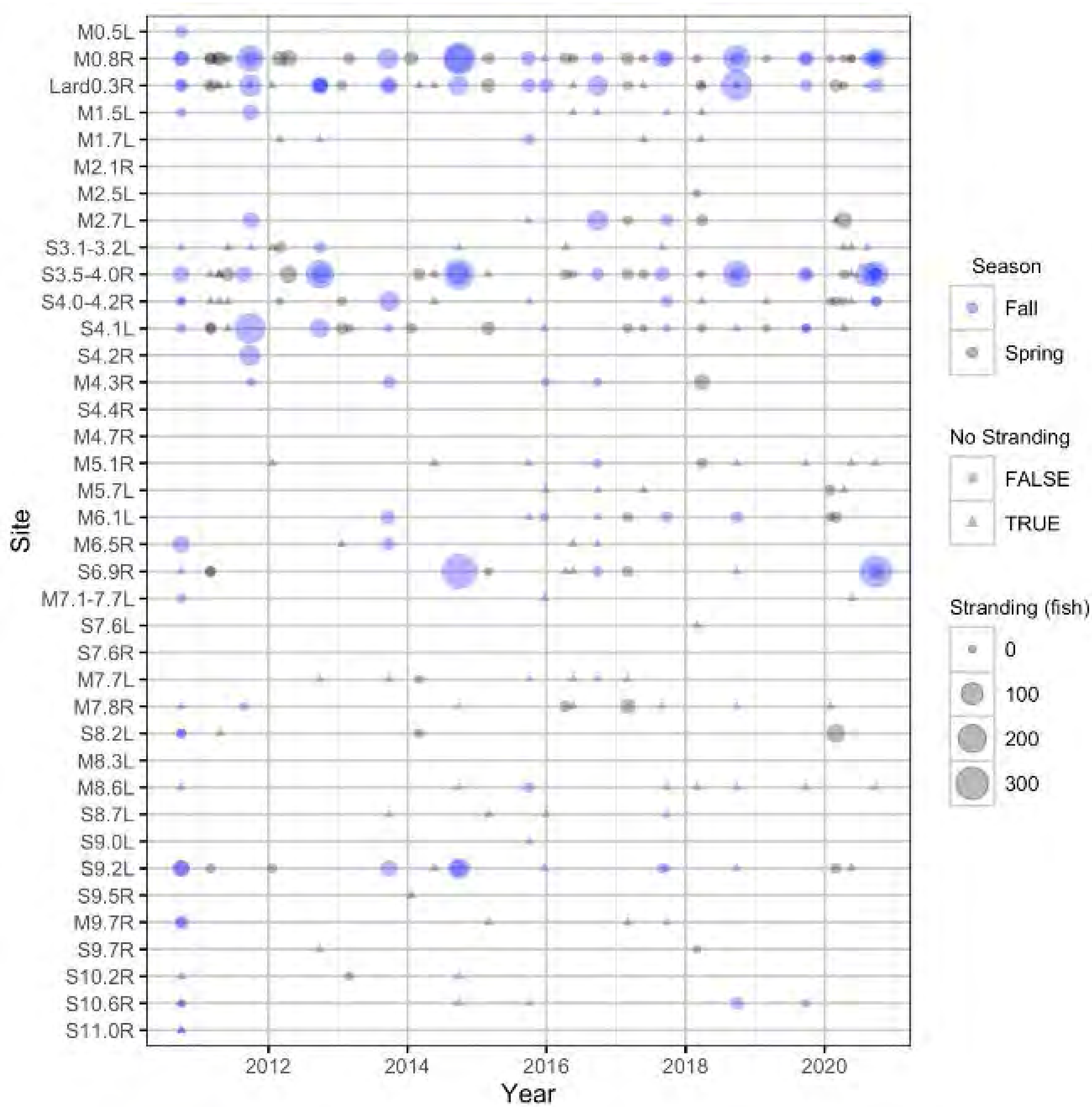

Figure 23: The estimated isolated pool stranding of Rainbow Trout by site, stranding assessment date and season.

3.4.3 Interstitial Stranding

Between Year 4 (2011–2012) and Year 13 (2020), 26 Rainbow Trout and 1 Mountain Whitefish were found to be interstitially stranded on dewatered substrates ranging in size from silt to small cobble (Figure 24). Interstitial sample methodology was standardized using transect sampling in Year 6; between Year 6 and Year 10, only one interstitially stranded Rainbow Trout was observed (in Year 6; Golder 2015). In Year 11, when interstitial census methodology was implemented, seven interstitially stranded Rainbow Trout were documented (Golder and Poisson 2019b). In Year 12, two Rainbow Trout were recorded as interstitially stranded, and in Year 13, one interstitially stranded Rainbow Trout was recorded. A relationship between substrate size and randomly sampled interstitial stranding rates was not found in the combined dataset.

The effect of season on interstitial stranding density of Rainbow Trout was significant, with the mean fall stranding estimates approximately 9 times higher than those for winter/spring (Figure 25). The calculated of Rainbow Trout juveniles per hectare of interstitial habitat for the spring season (January to June) was 1.81 fish/hectare, in comparison to 15.61 fish/hectare in the fall (July to December). Since the single interstitially stranded Mountain Whitefish was encountered in the spring season, only spring season density estimates (1.81 fish/hectare) could be calculated (Figure 25).

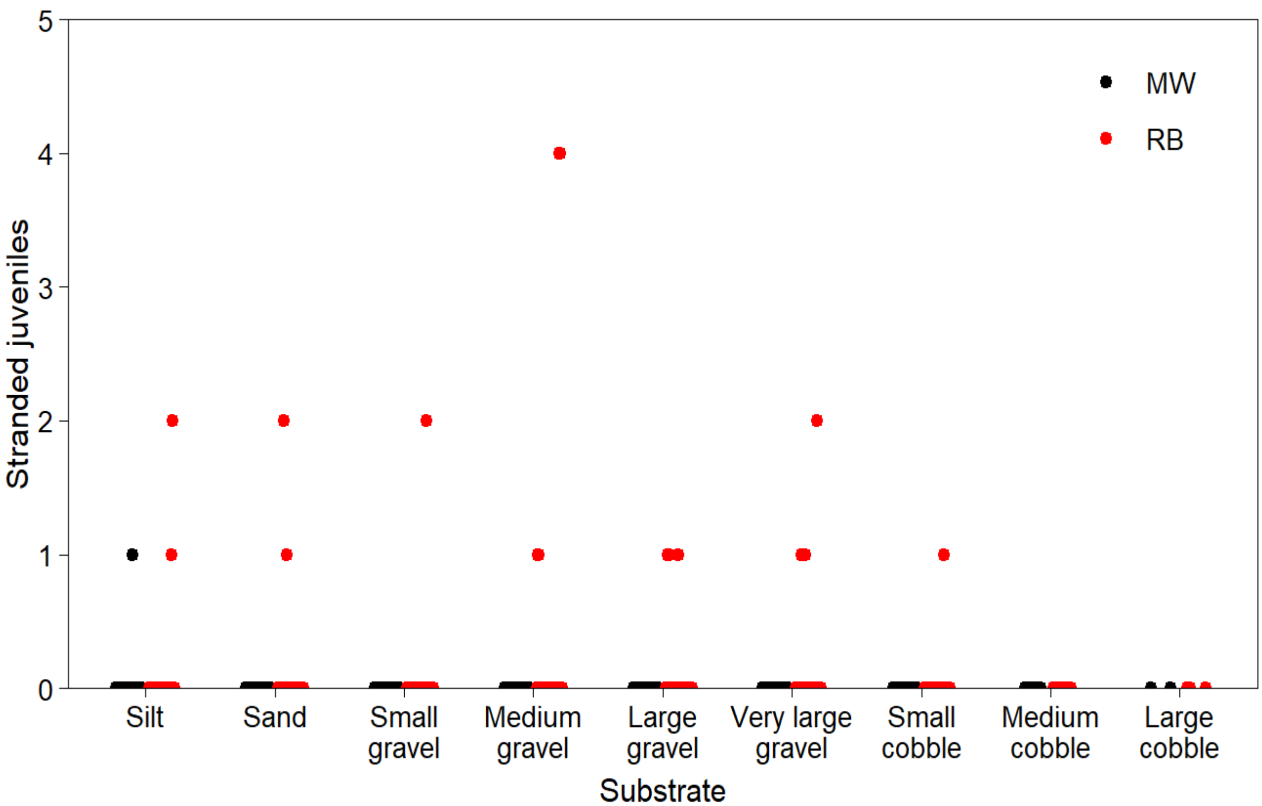

Figure 24: Counts of 2011–2020 interstitially stranded Mountain Whitefish and Rainbow Trout in the lower Duncan River, plotted by substrate size.

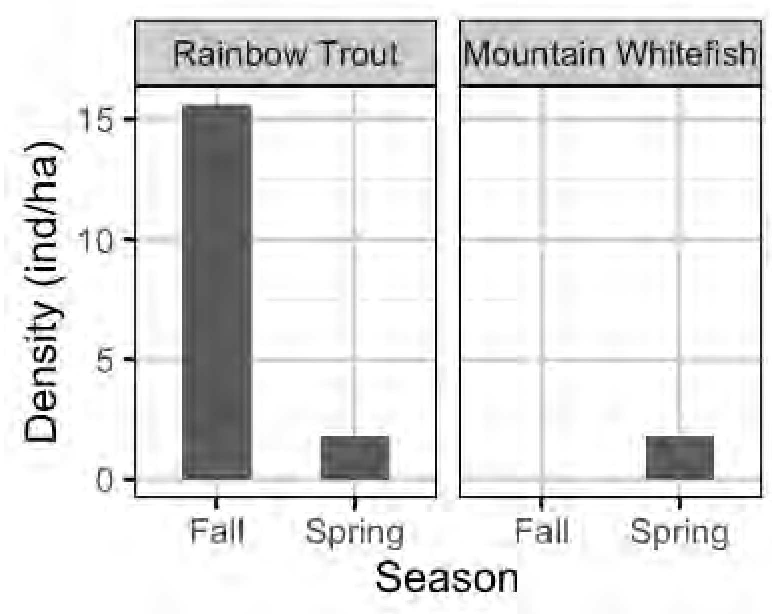

Figure 25: Interstitial stranding density by season and species.

Rainbow Trout interstitial stranding differed between seasons (Figure 26). In all study years, fall interstitial stranding were higher and more variable in comparison to the spring season. Conversely, interstitial stranding at sites remained relatively consistent between study years. There were no significant differences between interstitial stranding at index and random sites (Figure 26).

Interstitial stranding for Mountain Whitefish in the spring was relatively consistent between study years, and calculated values were extremely low (Figure 27). There were no significant differences between interstitial stranding at index and random sites. As interstitially stranded Mountain Whitefish were not encountered in fall assessments over the course of the program, stranding could not be calculated for that season (Figure 27).

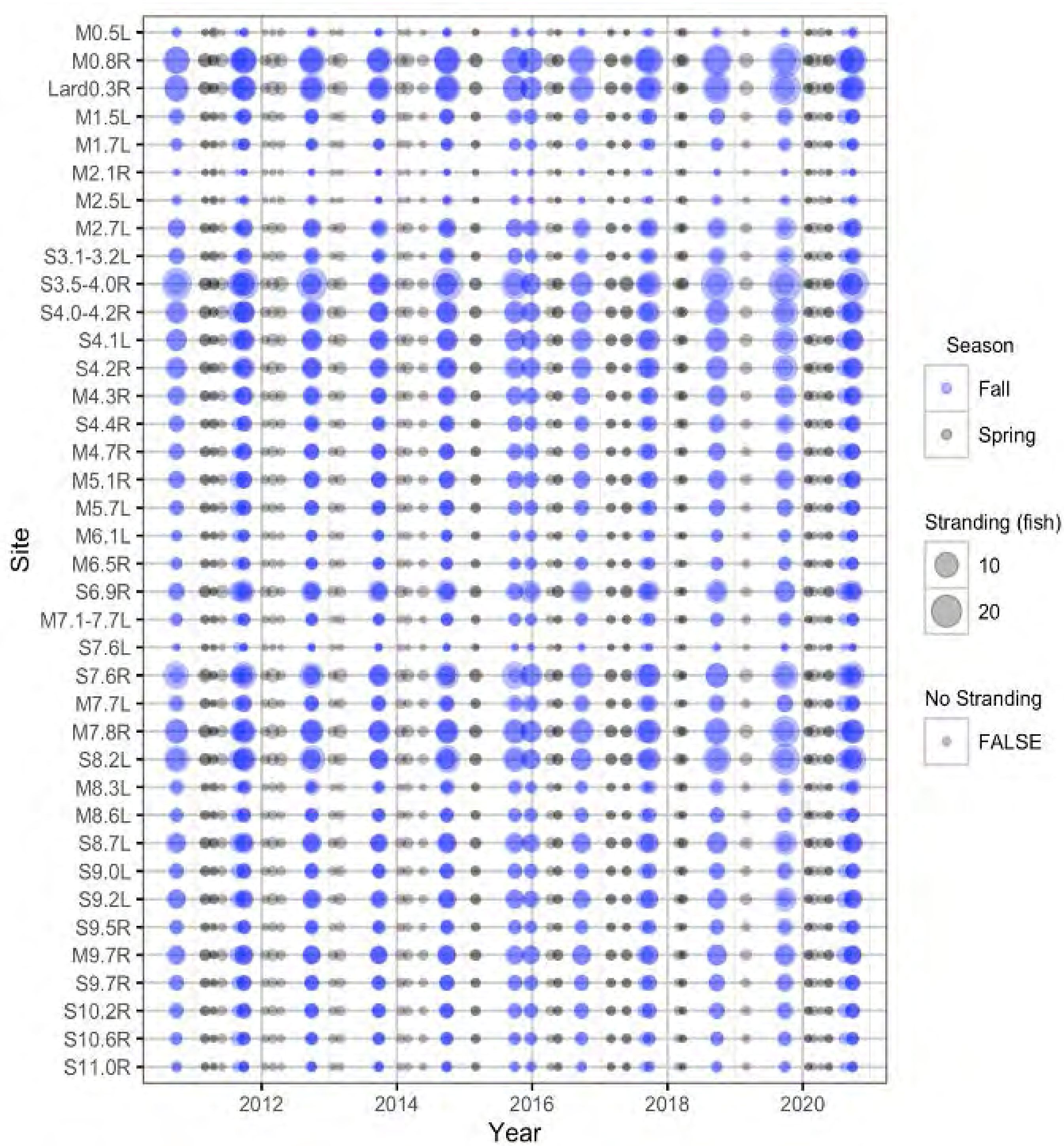

Figure 26: Rainbow Trout interstitial stranding by site, stranding assessment date and season.

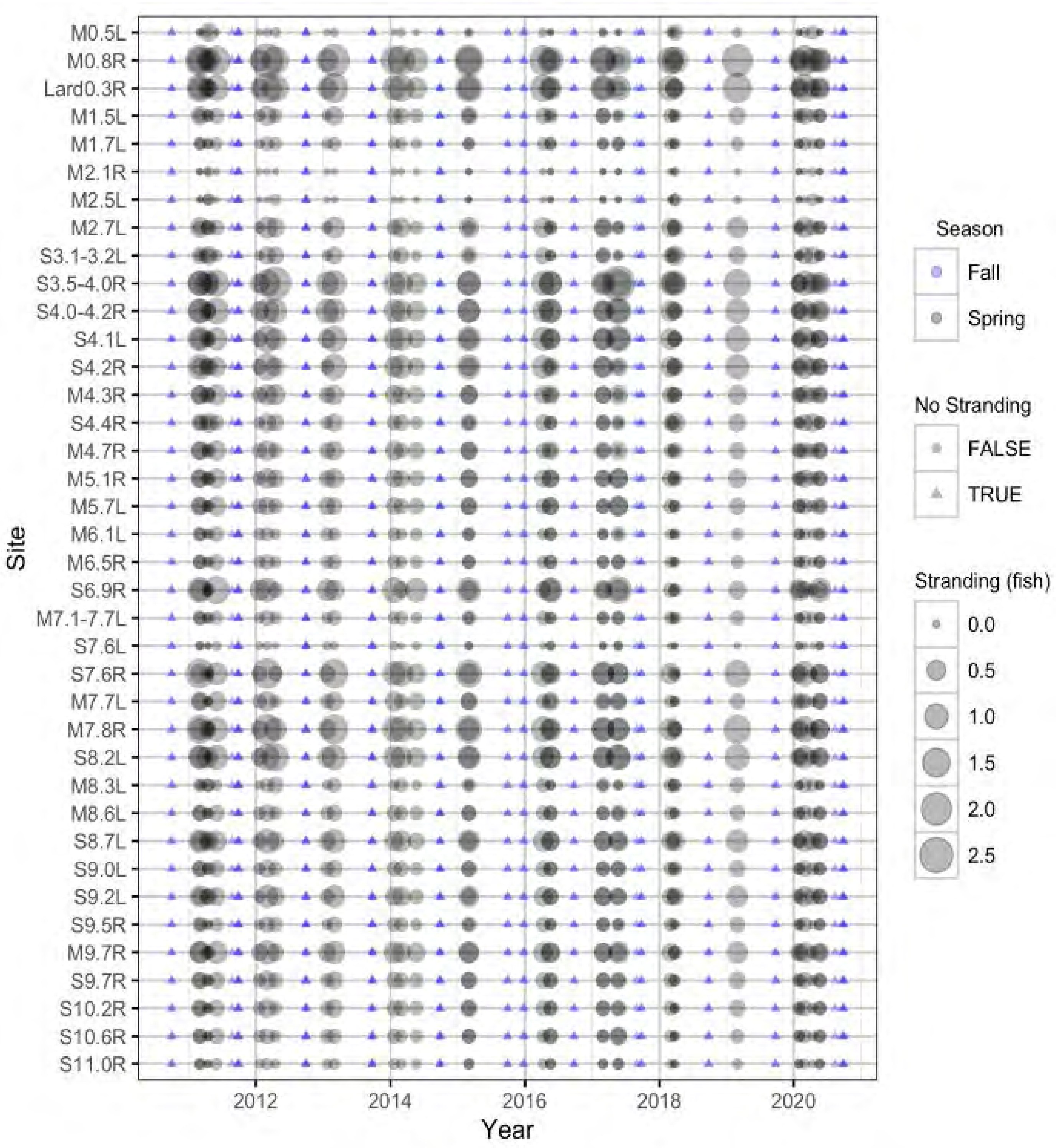

Figure 27: Mountain Whitefish interstitial stranding by site, stranding assessment date and season.

3.4.4 Non-Randomly Sampled Stranding

Observations of stranded target species in stranding mechanisms that were not randomly sampled (i.e., in dried or non-randomly sampled pools, encountered non-randomly on dewatered substrate) were higher in the fall season (Figure 28 and Figure 29). Approximately 96% of non-randomly sampled target species were encountered at index sites (i.e., M0.8R, Lard0.3R, S3.5-4.0R, S4.0-4.2R, S4.1L, S6.9R, and S9.2L).

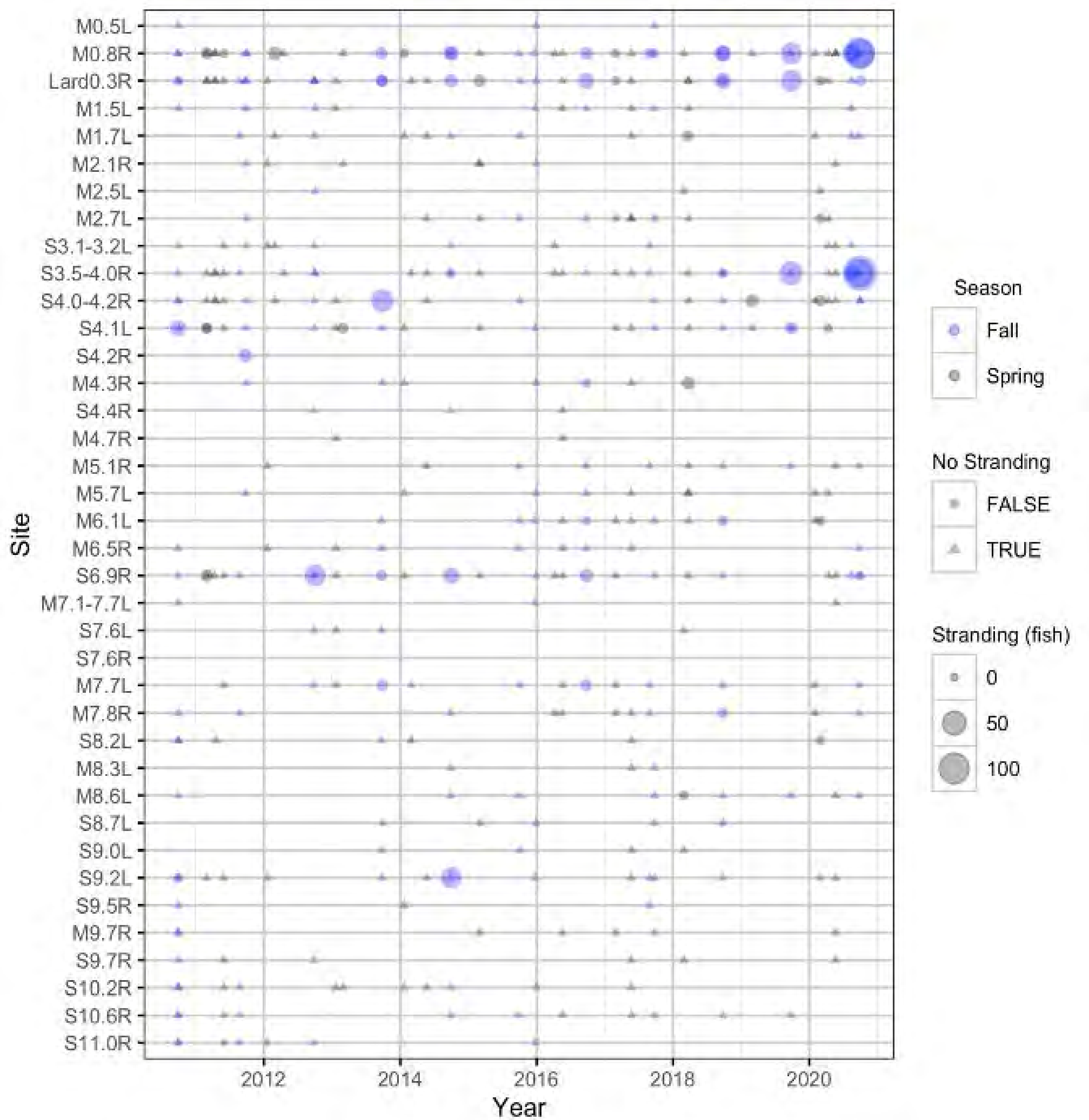

Figure 28: Observed non-random stranding of Rainbow Trout by site, stranding assessment date and season.

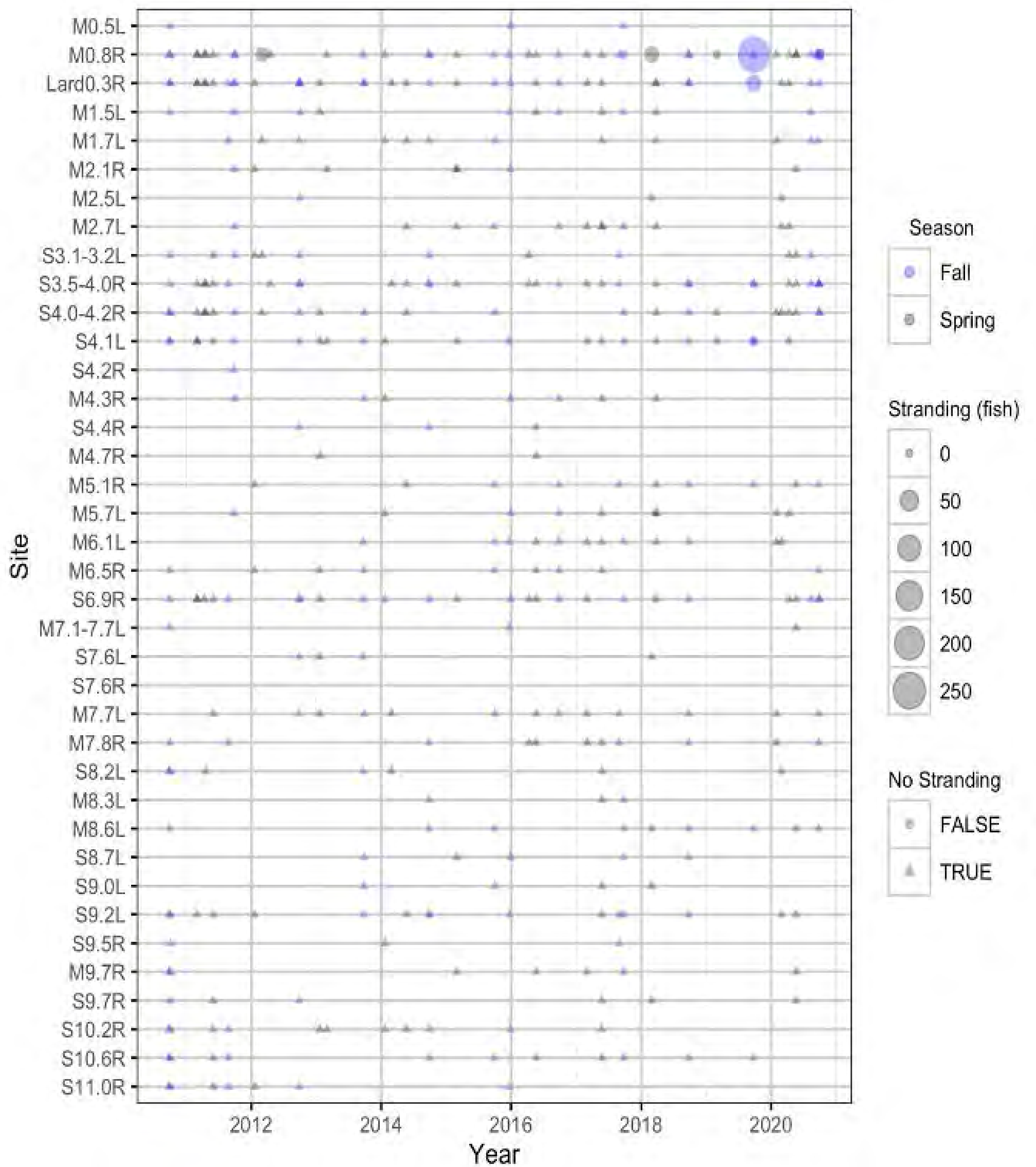

Figure 29: The observed non-random stranding of Mountain Whitefish by site, stranding assessment date, and season.

3.4.5 Total Stranding Estimates

3.4.5.1 Index vs Random Sites

In Study Year 2 (2009 – 2010), there were more stranded fish encountered at calibration sites (*n* = 134) than at index sites (*n* = 50). Initial analysis of the density of stranded fish indicated that the stranding rates associated with calibration sites were not distinguishable from index sites (Section 2.6; Golder 2010). However, after detailed examination of the data, the high frequency of zero counts suggested assumptions of normality were violated using the least squares model previously described, despite using logs to normalize the data.

Additional analysis was conducted using the frequency of the sites with zero counts compared with those where at least a single stranded fish was observed, with comparison between the index sites and the calibration sites. Only 4 of 14 of the calibration sites had stranding, which differed significantly from the 70 of 106 index sites where stranding was observed (Table 9; Chi Sq.= 7.344, df=1, p=0.007).

Table 9: Summary of stranding presence/absence between index and calibration sites with the comparative percentages reflecting the occurrence of stranding at individual sites. Counts are provided in parenthesis.

	Index	Calibration	Total	Total Number of Sites
No Stranding	33.9%(36)	71.4%(10)	38.3%	46
Stranding occurred	66.0%(70)	28.6%(4)	61.7%	74
Total Number of Sites	106	14		120

Because of the significance of these results, the GLM model was run excluding zero count data from the analysis (Table 10). As a result, the number of pools in a stranding area was no longer significant but Calibration Sites were significantly different from Index sites. The least square means (LSM) were substantially different (Index = 1.570, SE=0.17, n=70; Calibration=2.87, SE=0.570, *n*=4), reflecting a fourfold difference in actual density (counts/unit area) because of the LSM reflecting the logarithm of the densities.

The higher levels of fish stranding at the calibration sites were in contrast with the small proportion of calibration sites that had stranding (28.6%), in comparison to the larger proportion of the index sites that had stranding (66.0%) (Table 9). Therefore, the difference in fish stranding levels and the proportion of sites where stranding was observed (less frequent stranding but higher densities) indicate inconsistent values with expected results and required more complex modelling.

Table 10: ANOVA of the effects of hypothesized factors on fish stranding densities expressed as Log((Fish number+1)/dewatered area).

ANOVA excluding zero counts	Sum-of-Squares	df	Mean-Square	F-ratio	p
Month	29.659	5	5.932	6.671	0.000
Initial Wetted Area	8.376	1	8.376	9.421	0.003
Final Wetted Area	7.097	1	7.097	7.982	0.006
Calibration	4.679	1	4.679	5.262	0.025
Year	8.281	4	2.070	2.328	0.066
Error	54.238	61	0.889		

In Study Years 7 to 13 (2015 – 2020), there were no statistically significant effect of index and random site type on pool formation, and subsequently on pool stranding rates (Figure 17). The low numbers of fish in the dataset that were found interstitially stranded precluded the examination of the effect of index/random site effect on interstitial stranding rates.

In the combined data set, total Rainbow Trout stranding in the fall was generally highest and more variable at index sites, specifically at sites upstream of Rkm 7.0 (Figure 30). Variability in spring total stranding estimates was substantially lower for both index and random sites. When the proportion of the total Rainbow Trout stranding in the dataset was examined by site, although index sites typically strand higher proportions of fish, there were no significant differences in stranding rates between site types (Figure 31).

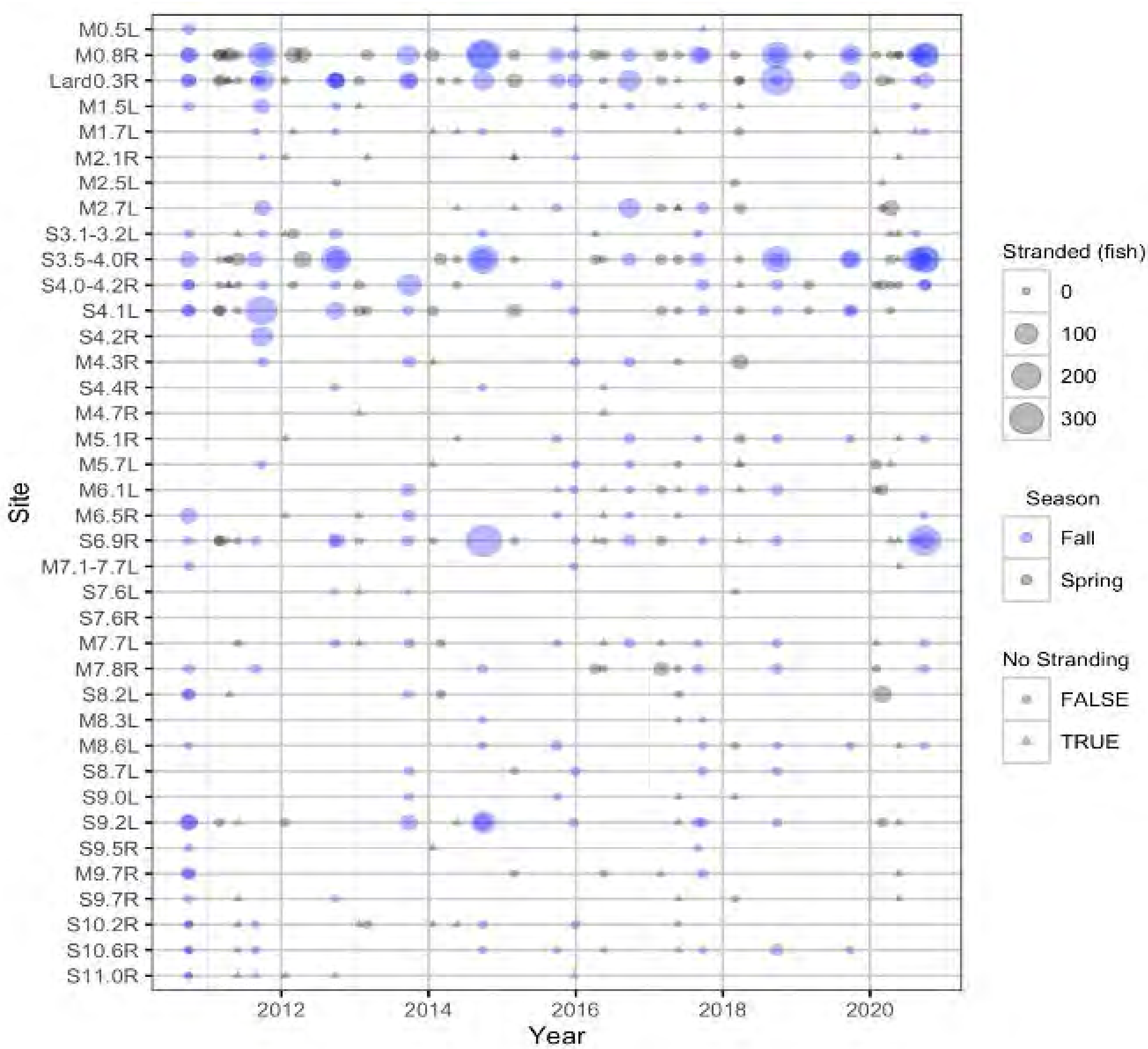

Figure 30: Total Rainbow Trout stranding (combined pool, interstitial and non-random) by site, stranding assessment date, and season.

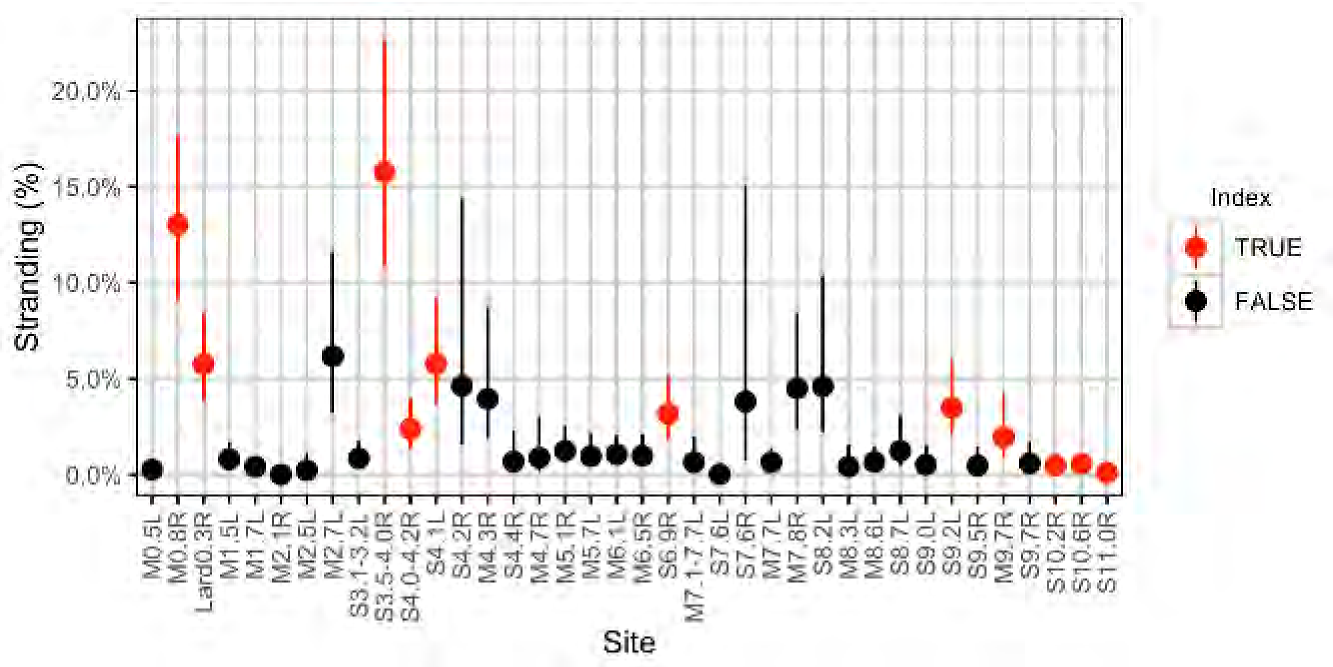

Figure 31: Relative importance (proportion of total Rainbow Trout stranded) by site. True indicates Index sites and False indicates non-index sites. Error bars are 95% credibility intervals.

3.4.5.2 Overall Rainbow Trout Total Stranding

To obtain total stranding estimates of Rainbow Trout juveniles, Rainbow Trout abundance and stranding estimates were adjusted to account for survival rate by date (Decker and Hagen 2009: Figure 32). Estimated total stranding for each individual flow reduction event in the combined dataset was highly variable (Figure 33). In most study years, fall flow reduction events were estimated to strand high numbers of Rainbow Trout juveniles in comparison to spring estimates. When comparing the estimated proportion of the spawn year cohort that was stranded by each flow reduction event, the differences between the spring and fall estimates was reduced (Figure 34). Fall flow reduction events typically stranded higher proportions of the spawn year cohort, although there is substantially more overlap between proportional estimates when compared to the numerical estimates by flow reduction event (Figure 34).

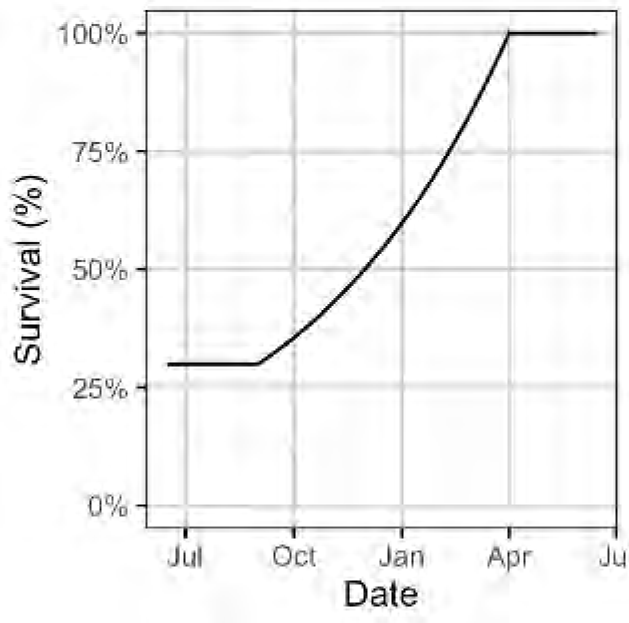

Figure 32: The estimated survival rate of age-0 Rainbow Trout used to adjust stranding and abundance estimation by date. Calculated from Decker and Hagen 2009.

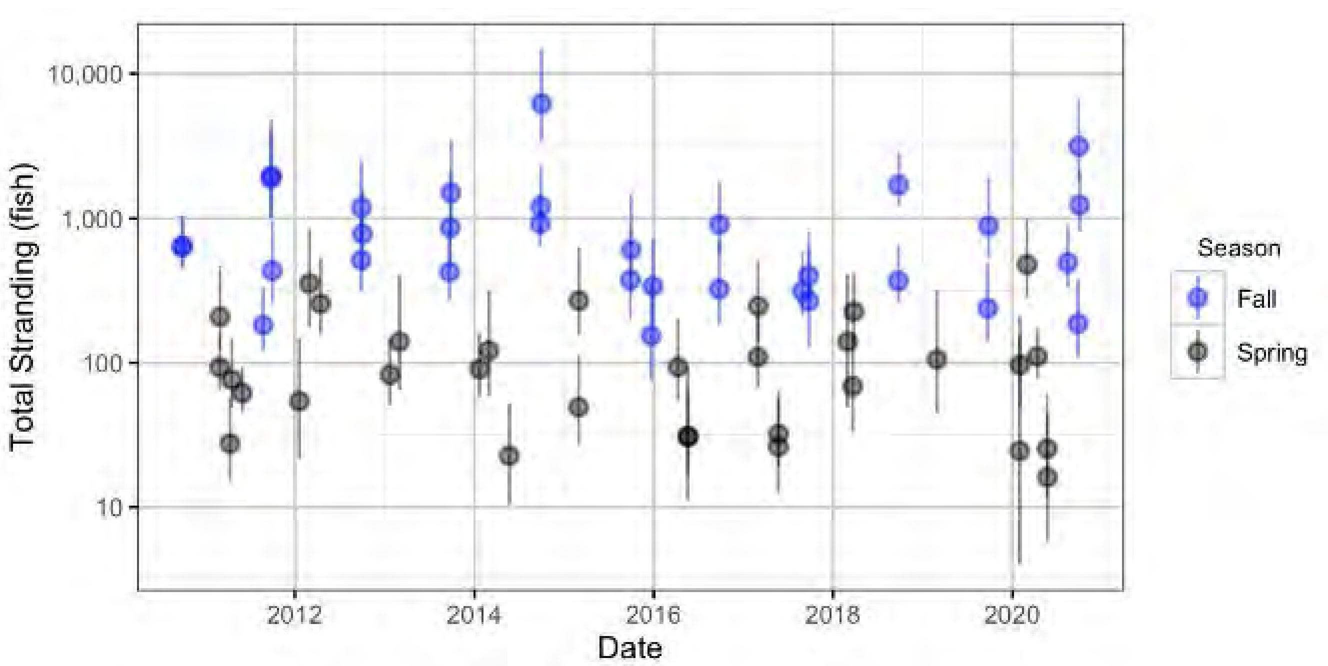

Figure 33: Estimated total Rainbow Trout stranding for each assessed flow reduction date. Error bars are 95% credibility intervals.

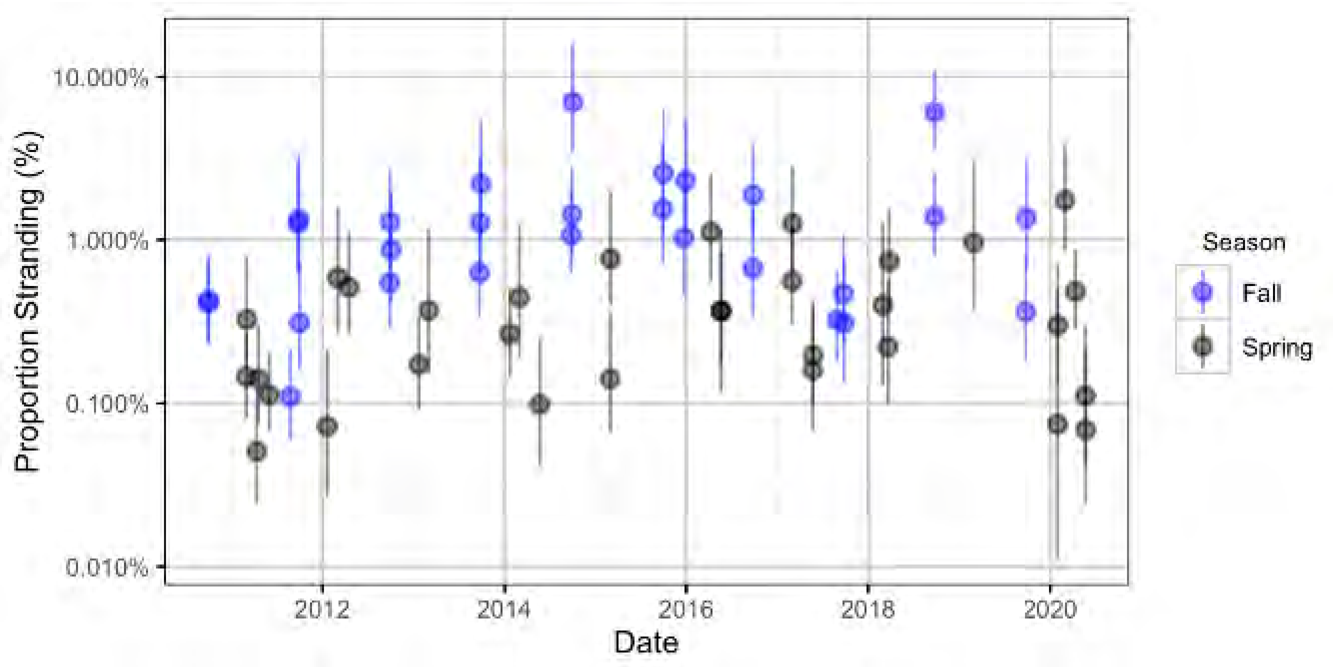

Figure 34: Proportion of Rainbow Trout spawn year cohort stranding by flow reduction date and season. Error bars are 95% credibility intervals.

For Rainbow Trout, mean annual total percent stranding estimation (interstitial, pool, and non-random combined) of the age-1 spring population ranged from 1.7% (95% CRI of 1.0 – 2.7%) in 2010 to 10.1% (95% CRI of 5.8 – 19.1%) in 2014 (Figure 35). The total percent stranding estimate for 2020 could not be obtained as abundance estimates for that spawn year were not available. During the 2010 and 2019 spawn years, 60% (6 of 10) of the mean annual Rainbow Trout percent stranding estimates were below 5% of the estimated age-1 spring abundance (Figure 35). With the growth of the dataset from additional study years, as well as the inclusion of non-random stranding data, current estimates for total percent stranded were higher and more precise than estimates obtained in previous study years (Golder and Poisson 2019a, 2019b, and 2020).

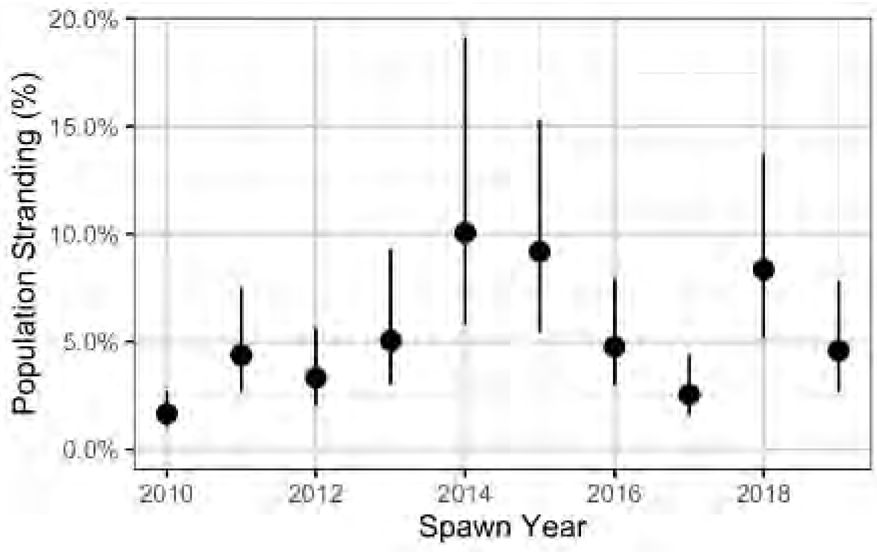

Figure 35: Estimates of total percent stranded Rainbow Trout by spawn year in the lower Duncan River. Error bars are 95% credibility intervals.

3.4.6 Proportional Stranding

The proportional stranding model compares the estimated proportion of Rainbow Trout juveniles in the system that are stranded as a function of several standardized environmental and operational variables. The standardized variables include:

- Date of the year (start date set at June 15, end date set at 14 June of the next calendar year, mean date of the year 2 January) (Figure 36).
- Flow reduction magnitude (mean of 60 m^3 s^{-1}) (Figure 37).
- DRL initial discharge level (mean of 197 m^3 s^{-1}) (Figure 38).
- Flow ramping rate (mean of 24 m^3 $s^{-1}hr^{-1}$) (Figure 39).
- Log wetted history (set to a maximum of 50 days) (Figure 40).

The predicted proportion of the total population of Rainbow Trout juveniles that were stranded decreased as a function of increasing date and overall discharge in the LDR (measured at the DRL; Figure 36 and Figure 38). Conversely, the predicted proportion of juveniles stranded increased as a function of increasing flow reduction magnitude and ramping rate (Figure 37 and Figure 39). There was no relationship between proportion of stranding and wetted history (Figure 40).

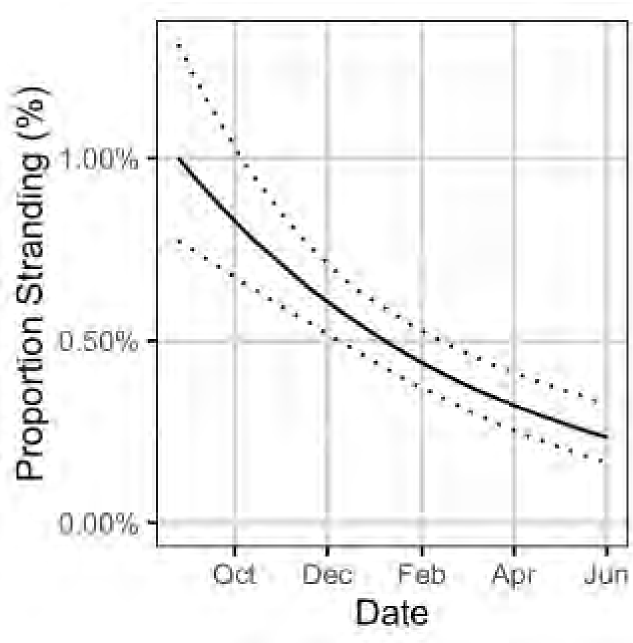

Figure 36: The predicted proportion (with 95% CIs) of the Rainbow Trout juvenile population stranded, by date, for a typical daily flow reduction of 60 m^3 s^{-1} at a ramping rate of 24 m^3 s^{-1} hr^{-1} from a typical initial discharge level of 197 m^3 s^{-1} .

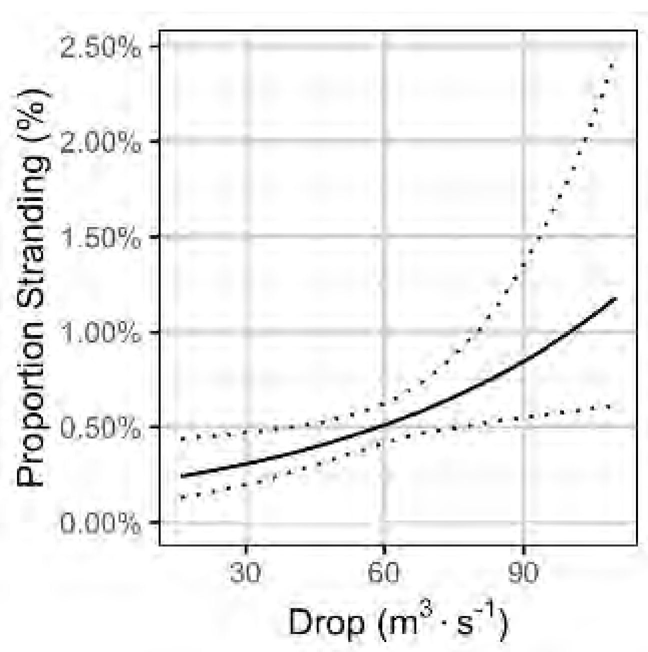

Figure 37: The predicted proportion (with 95% CIs) of the Rainbow Trout juvenile population stranded with a variable flow reduction magnitude (Drop) at a ramping rate of 24 m^3 s^{-1} hr^{-1} from a typical initial discharge level of 197 m^3 s^{-1} .

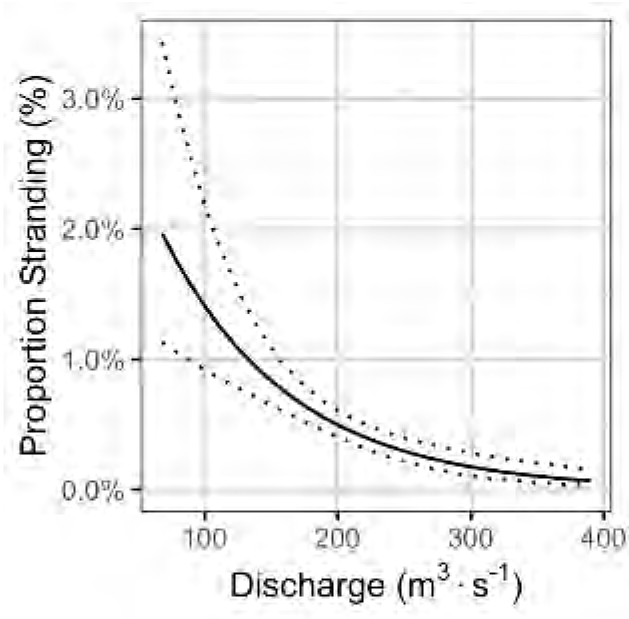

Figure 38: The predicted proportion (with 95% CIs).of the Rainbow Trout juvenile population stranded at a ramping rate of 24 m^3 s^{-1} hr^{-1} for a typical daily flow reduction of 60 m^3 s^{-1} by initial discharge level.

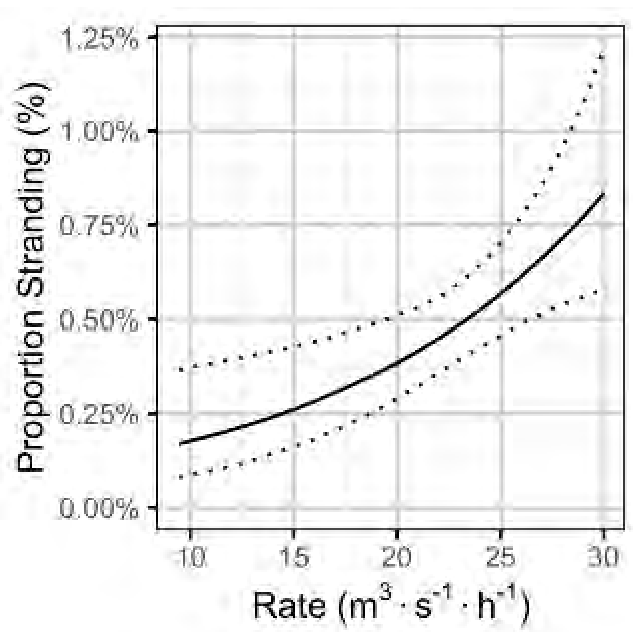

Figure 39: The predicted proportion (with 95% Cis) of the Rainbow Trout juvenile population stranded by ramping rate for a typical daily flow reduction of 60 m^3 s^{-1} from a typical initial discharge level of 197 m^3 s^{-1}.

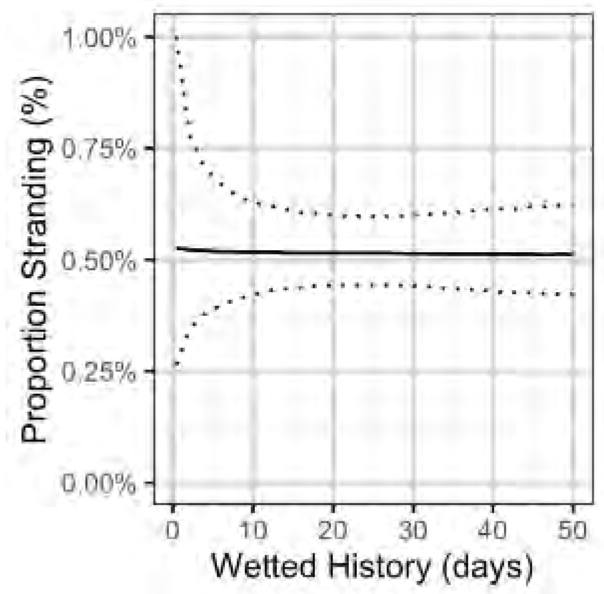

Figure 40: The predicted proportion (with 95% CIs) of the Rainbow Trout juvenile population stranded by wetted history at a ramping rate of 24 $m^3 s^{-1} hr^{-1}$ for a typical drop of 60 m^3 s^{-1} from a typical initial discharge level of 197 m^3 s^{-1}.

4.0 DISCUSSION

4.1 Current Duncan Dam Operations in Relation to Fish Stranding

The state of knowledge regarding the environmental and operational variables of interest that impact fish stranding was reviewed in detail as part of the Lower Duncan River Ramping Program (DDMMON-1; Irvine and Schmidt 2009; Poisson and Golder 2010). The DDMMON-1 program developed an impact hypothesis diagram to summarize the variables that may affect fish stranding in the Lower Duncan River (Figure 41; Poisson and Golder 2010) and to conceptually link the variables and their effects. This diagram includes a streamlined representation of the relationship between variables of interest and how they contribute to fish stranding, although interactions and/or autocorrelations are not shown in this diagram as they are too numerous to present.

As stated by the DDMMON-1 program, the multiplication of probability of fish stranding by fish density predicts the number of fish stranded (Figure 41; Poisson and Golder 2010). If a fish becomes stranded and succumbs, the fish becomes part of the stranding mortality component (sum of interstitial and pool stranding mortality) of the total mortality rate associated with the population.

4.1.1 Variables Affecting Fish Stranding

There are several environmental and operational variables of interest that can affect fish stranding (Figure 41). The DDMMON-16 Program included statistical analyses on Proportional Stranding to compare with the conclusions of the DDMMON-1 Program and to further address the following DDMMON-1 Management Question:

"*What is the relationship between fish stranding and*

- *Rate of river stage/total stage change,*
- *Time of day (day/night),*
- *Substrate,*
- *Habitat configuration (channel bed gradient and topography),*
- *Cover,*
- *Species,*
- *Time of year (spring, fall, winter); and,*
- *Habitat stability (wetted history)?"*

Within this suite of variables, those that are currently addressed by operational strategies to potentially reduce fish stranding are rate of river stage/total stage change and time of day (Golder 2011, Golder and Poisson 2012, Golder and Poisson 2014). The operational variable related to stranding that is currently not specifically addressed by the ASPD is wetted history (Golder and Poisson 2014).

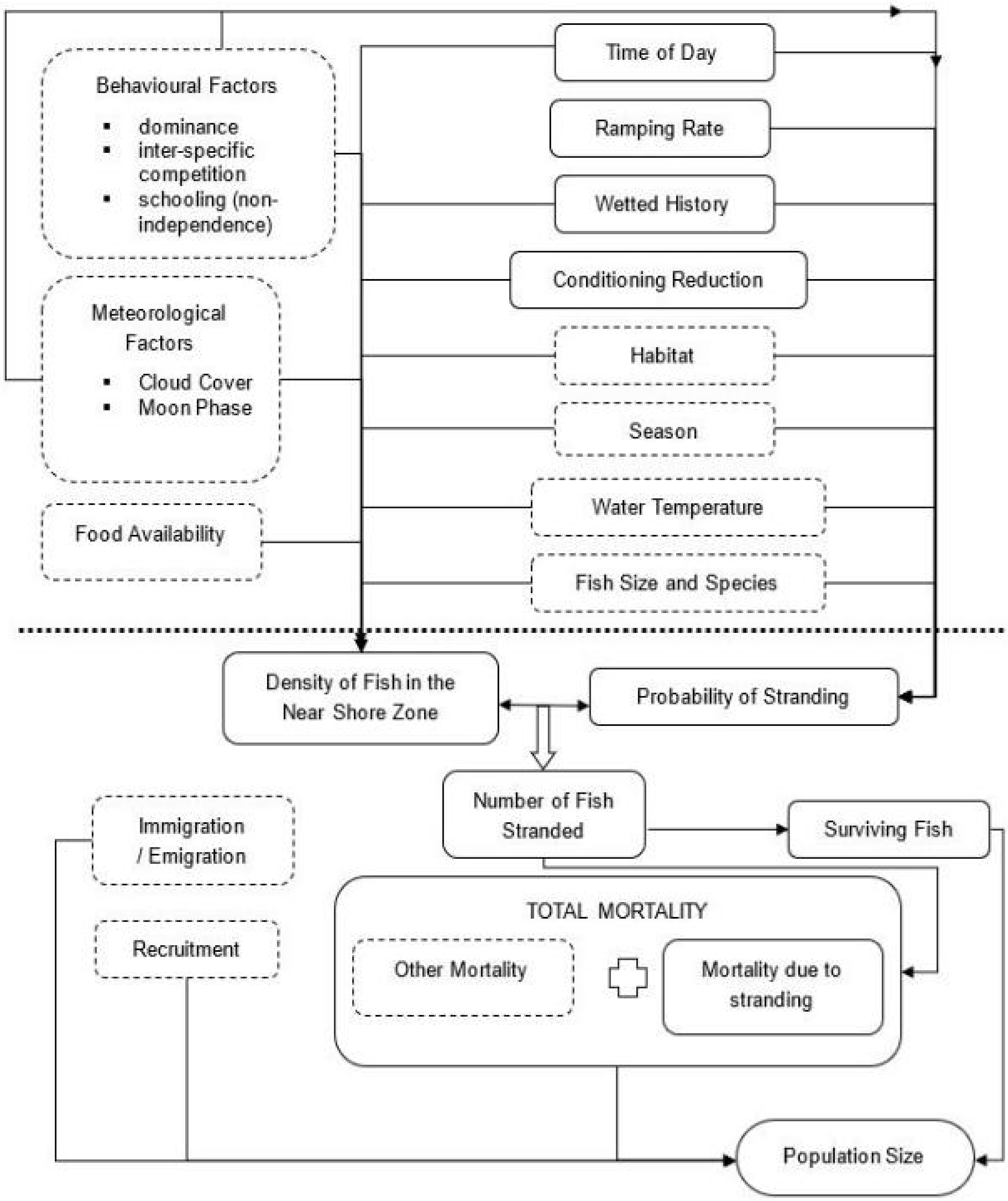

Figure 41: Impact hypothesis diagram for juvenile fish stranding on the lower Duncan River. Variables contributing to juvenile fish mortality are located above the dotted line, while items below the dotted line are processes feeding into the population size. Variables enclosed in boxes with dashed lines are not within direct operational control and those in solid boxes are within operational control (Golder and Poisson 2014).

4.1.1.1 Rate of River Stage/Total Stage Change

Initial analyses from DDMMON-1 experimental variations in ramping rate on the LDR suggested that although ramping rate never emerged as statistically significant in relation to fish stranding, the trend has consistently been that more fish strand at higher ramping rates (Poisson and Golder 2010). The analysis suggests that the slower the ramping rate, the less probability of stranding fish. DDMMON-1 recommended that flow reductions occur at a rate of 10 cm/hr or less to minimize the risk of fish stranding. It was also recommended in DDMMON-15 that where practical, each reduction should be completed in multiple, smaller operational increments rather than in one large drop (Golder and Poisson 2014).

The analysis of the stranding survey data from Year 2 (2009 – 2010) of DDMMON-16 also did not identify ramping rates as significantly contributing to the density of fish observed stranded in dewatered habitat (Golder 2010). It was documented that a single hourly flow reduction from DDM of 28 $m^3 s^{-1}$ resulted in a stage reduction rate of approximately 18 cm/hr at the Duncan/Lardeau confluence and 8 cm/hr at DRL (Golder 2010). This indicates that the stage reduction rate during a RE is attenuated as it moves downstream. This finding was supported by DDMMON-3, which also found that stage change during flow reductions diminishes as distance from DDM increases (NHC 2010). Since 82% of the index sites (9 of 11) and 89% of the non-index sites (24 of 27) are downstream of the DRL (Appendix A, Figures A1 to A7), the maximum hourly flow reduction allowed under the WUP (28 m^3 s^{-1}) will result in stage reduction rates of less than 10 cm/hr at the majority of all identified stranding sites. As post WUP operations at DDM have consistently been completed in multiple smaller increments, ramping rates were not analyzed again during the DDMMON-16 program until Year 13 (2020). Proportional stranding modelling conducted on the combined DDMMON-16 dataset indicated that the predicted proportion of the Rainbow Trout juvenile population stranded increases as ramping rates increase.

The total stage change for DDM operations was not experimentally tested during any phase of the DDMMON-1 program (Poisson and Golder 2010), and therefore was not analyzed as part of that program. Total stage change was examined in Year 13 (2020) of the DDMMON-16 Program, and proportional stranding modelling conducted on the combined dataset indicated that the predicted proportion of the Rainbow Trout juvenile population stranded increases as total reduction magnitude (Figure 37).

The stage of the river (lesser flows concentrating fish and having increased amounts of potential stranding habitat dewatered) also contributes to the observed stranding densities. River stage was not an experimental treatment in any phase of the DDMMON-1 program (Poisson and Golder 2010), and therefore was not analyzed as part of that program. Analysis conducted in Year 2 (2009 – 2010) of the DDMMON-16 Program examined the influence of dewatered habitat (initial wetted area and final wetted area) for each site using the initial discharge and the end discharge as variables that were related to the number of fish stranded per unit area dewatered (Golder 2010). Both the initial and final wetted areas had statistically significant relationships to fish stranded per area, with lower flows contributing to higher stranding densities, but were relatively minor contributors to explaining overall variability in the density of fish stranding (number of fish stranded per unit area dewatered). In Year 13 (2020) of the DDMMON-16, proportional stranding modelling on the combined dataset indicated that the predicted proportion of the Rainbow Trout juvenile population stranded increases as the LDR discharge events decreases prior to flow reduction (Figure 38).

The findings of the DDMMON-16 Program corroborate the conclusion of the DDMMON-1 Program that larger ramping rates lead to increased risk of stranding (Poisson and Golder 2010).

4.1.1.2 Time of Day

The data analysis conducted during the DDMMON-1 program showed a trend of increased fish stranding at night (Poisson and Golder 2010). This trend was not statistically significant due to the high variability of stranding rates during both day and night experiments. Flow Reduction and Fish Stranding Assessments conducted on the Lower Columbia and Kootenay Rivers near Castlegar BC also did not find a statistically significant effect of time of day on fish stranding probability (Golder 2007). The DDMMON-1 finding was also both in alignment and contradictory to studies conducted on other systems (Golder and Poisson 2014).

Diel variations in stranding rates could not be analyzed since all fish stranding assessments were conducted following DDM flow reductions that occurred daytime hours (between 05:00 and 18:00). However, previous DDMMON-1 experimental observations indicated that large numbers of Mountain Whitefish were observed stranded during rapid nighttime reductions in flow (Poisson and Golder 2010). Currently, all flow reductions under present DDM operations occur during the daytime period, which follows the recommendations of the DDMMON-1 and DDMMON-15 programs and allow for fish salvage responses immediately after the reductions.

4.1.1.3 Substrate

During the DDMMON-1 program, substrate was not assessed as a primary variable during analysis (Poisson and Golder 2010). The gap analysis completed as part of the DDMMON-1 program did however find a relationship between larger substrates and higher stranding rates during the review of applicable literature (Irvine and Schmidt 2009).

Although the DDMMON-16 program did not find significant differences between substrate size within pools and the density of stranded fish, stranded Rainbow Trout densities were highest in pools with very large cobbles and large gravels. There also was not any relationship between interstitially stranded fish counts and substrate size. Therefore, there is no evidence to refute the conclusion of the DDMMON-1 Gap Analysis that higher stranding rates are linked with larger substrates (Irvine and Schmidt 2009).

4.1.1.4 Habitat Configuration (Channel Bed Gradient and Topography)

The DDMMON-1 Program did not analyze stranding rates in response to channel gradient, channel topography or other variables that could be considered part of habitat configuration (Poisson and Golder 2010).
Habitat configuration data were collected during all phases of the study, but because these variables were never the primary variables of interest, and sites were not selected to represent a range of habitat configuration values, stranding rates in response to those variables were not analyzed.

As part of the data analysis for DDMMON-16, fish stranding was examined in relation to channel gradient (slope) and habitat configuration (mainstem habitat vs side channel). The DDMMON-16 Program did not examine channel topography, which was a focus of the DDMMON-3 Program. Based on early data analyses in the DDMMON-16 Program, considerably higher amounts of low slope habitats (≤ 8%) were dewatered during flow reductions from DDM in comparison to high slope habitats (≥8%), and the dewatered low slope habitats had substantially more interstitially stranded fish following flow reductions than high slope habitats (Golder and Poisson 2012). Analyses of the current combined dataset suggest that slope did not influence the formation of isolated pools within the study area. In most study years, pool density was highest at lower slope values (0% to 5%); however, the relationship was variable and weak. This indicated that slope was not a significant factor influencing pool formation and subsequently pool stranding rates. Based on this finding, the effect of slope was replaced in the pool stranding analysis with the effect of season as a predictor of stranding risk.

There appears to be a relationship between slope and interstitial stranding. Most fish found interstitially stranded (89%) in all study years were on habitats with gradients of 9% or less although this is not statistically significant. This corroborates the conclusion of the DDMMON-1 Gap Analysis that steeper gradients lead to lower interstitial stranding rates.

In Year 2 (2009 – 2010) of the DDMMON-16 Program, the relationship between habitat configuration and stranding rates was also examined (Golder 2010). Macro habitat types (side channels, mainstem bar), which were thought to be substantial contributors to observed stranding variability, were also not statistically significant in explaining observed fish stranding variability. This was supported by the finding of DDMMON-2 where preferences for mainstem, braid or side channel habitats were not shown by any of the species or life stages analyzed (Thorley et al. 2011).

4.1.1.5 Cover

During the DDMMON-1 Program, cover was assessed as an ancillary variable in past ramping rate experiments (Phase I to IV) and was again recorded in Phase V (Poisson and Golder 2010). The DDMMON-1 experimental data did not yield any significant trend between cover and stranding, although the Gap Analysis component of DDMMON-1 identified a strong link between increased cover and higher stranding rates in the reviewed literature (Irvine and Schmidt 2009).

In Study Year 3 (2010 – 2011) of the DDMMON-16 program, pools were grouped by complexity based on the size of each pool and total cover within each pool, and therefore cover types were not analyzed separately. During the analysis there was insufficient data available to test pool complexity. In the combined DDMMON-16 dataset, cover type was not assessed as multiple types present in individual pools would confound results, and cover is not within operation control. As habitat is continually changing in the LDR, new sites with variable habitat conditions are continually forming and old sites are frequently changing. For future salvage responses in the LDR, it is recommended the sites with high levels of cover that could potentially influence stranding rates be surveyed at a higher priority.

4.1.1.6 Species

Due to experimental methodology, species were not separated out in the analysis of the DDMMON-1 Program and equal risk was assumed for all species (Poisson and Golder 2010).

The species of interest for the DDMMON-16 Program were Rainbow Trout and Mountain Whitefish. Therefore, only stranding rates for these species were analyzed (Section 4.2). The differences in stranding rates between the target species (i.e., Rainbow Trout having the highest stranding risk) of the DDMMON-16 violates the equal risk assumption.

4.1.1.7 Time of Year (Spring, Fall, Winter)

Phase I to Phase V DDMMON-1 experiments were only conducted in the fall season and the program recommended that seasonal stranding rates be examined as part of the DDMMON-16 program.

In the combined DDMMON-16 dataset, seasonal effect on pool stranding numbers were found to be significant for Rainbow Trout, with mean fall stranding estimates significantly higher than those for winter/spring. This may be due to lower juvenile fish densities in the system in the winter/spring versus the fall or to a decreased risk of stranding in that period. This finding was corroborated by the predicted proportion of Rainbow Trout juveniles stranded in relation to the day of the year. As the date moved from the fall season into winter and spring, the predicted proportion of the population stranded decreased. This predicted decrease is likely related to juvenile mortality rates and the age-0 cohort aging over the winter season, which leads to a decrease in rt abundance available to strand over time. Higher stranding rates in the fall also lead to higher predicted proportional stranding. In Study Year 9 (2016 – 2017), there was no significant seasonal effect on Mountain Whitefish stranding, although fall stranding was estimated to be slightly lower than spring (Golder 2018). There is also an increased risk of stranding to emerging kokanee young-of-the-year during the spring. This risk is mitigated by the implementation of kokanee protection flows in the fall season which reduce flows in the lower Duncan River to limit adult kokanee spawning activities and stranding in side-channels (Golder 2010).

4.1.1.8 Habitat Stability (Wetted History)

During the data analysis of the DDMMON-1 Program, wetted history was tested as a secondary variable in the four of the five experimental phases. There was a trend of increased stranding risk with increased wetted history during the fall (Poisson and Golder 2010). This finding was not statistically significant, and DDMMON-1 recommended further analysis of DDMMON-16 stranding assessment data to confirm or deny this trend.

The DDMMON-16 program conducted proportional analysis on the combined dataset to examine fish stranding in relation to wetted history, and to allow comparisons with the findings of the DDMMON-1 Program. The range of wetted history examined by the DDMMON-16 analysis (0 to 50 days) was substantially larger that the range examined for DDMMON-1 (typically 5 to 10 days). There was no relationship between wetted history and the predicted proportion of fish stranded.

Therefore, the DDMMON-16 program did not find evidence to institute changes to operational strategies aimed at increasing habitat stability.

4.1.2 Pre- and Post-WUP Operating Regimes

Management Question 1) (*How effective are the operating measures implemented as part of the ASPD program?)* was addressed by examining the differences between the pre- and post-WUP flow regimes. Under the water license, two large reductions in DDM discharge occur on an annual basis. In the post-WUP regime, flow reductions occur in late September to early October for Kokanee protection by restricting access to spawning areas that pose high risks to stranding eggs and larvae. Also, in the post-WUP period, flow reductions in late winter were altered to support of Columbia River Mountain Whitefish management objectives (which are currently under review). The purpose of the late winter flow reductions is also to manage Duncan Reservoir flood control targets as defined under the Columbia River Treaty. In addition, there are several smaller reductions that occur throughout the year to effectively manage water resources and power generation at other facilities.

Total and mean area dewatered during all annual flow reductions were used to determine differences in pre- and post--WUP operations, as the area exposed relates directly to the hydraulic and stranding analysis models. Post--WUP flows have resulted in the dewatering an average of approximately 0.13 km^2 per reduction compared to pre--WUP operations.

Interannual variability in overall discharge and total reduction magnitude have also been reduced under post-WUP operations. As recommended by the DDMMON-1 and DDMMON -15 Programs (Poisson and Golder 2010, Golder 2012, Golder and Poisson 2014), DDM operations are required under the current water license to reduce flows at a maximum ramping rate of 28 m^3 s^{-1} (989 ft^3/s) per hour, and a maximum daily reduction limit of 113 m^3 s^{-1} (3,991 ft^3 s^{-1}). To ensure a stage change of 10 cm/hr or less at the majority of identified stranding sites, hourly flow reductions at DDM were split into 4 even reductions every 15 minutes when possible. Data trends identified in the DDMMON-1 and DDMMON-15 programs indicated that this slow rate of change during down ramping is believed to reduce the risk of fish stranding, which is also supported by studies conducted in Norway (Halleraker et al. 2003). Halleraker et al. (2003) recommended similar ramping rates to reduce stranding rates of salmonids, particularly after an extended period of stable flows. This operating strategy has resulted in consistently stable ramping rates during post-WUP operations in the LDR.

Based on the current state of knowledge, the flow reduction measures implemented under the ASPD are effective at reducing fish stranding and have resulted in less habitat being dewatered in the post-WUP operations. As the sampling programs assessing fish stranding levels have had different methodologies and various objectives, it is not possible to provide comparable fish stranding estimates between pre-WUP and post-WUP periods. Therefore, only assessments on the amount and rate of habitat dewatering can be made in determining the effectiveness of the ASPD measures.

4.2 Fish Stranding Summary

Management Question 2) (*What are the levels of impact to resident fish populations associated with fish stranding events on the lower Duncan River?)* was addressed. The species of interest were Rainbow Trout and Mountain Whitefish. During the 92 stranding assessments included in the combined dataset (from September 2006 to September 2021), a total of 4,754 Rainbow Trout and 946 Mountain Whitefish were encountered.

Determining how estimates of juvenile mortality due to stranding affect an overall fish population is difficult (Golder 2011). Several factors adversely affect fish populations including escapement, exploitation, predation, outmigration, food availability, availability of suitable spawning and rearing habitats, winter mortality, as well as inter- and intra-specific competition. Whether stranding events kill juvenile fish that would have died because of these factors or kill fish which would otherwise have survived these factors is unknown (Golder and Poisson 2012). As stated by Golder and Poisson (2014), determining whether stranding mortality has a population level effect (since compensatory mechanisms such as increased growth or survival may be a result of stranding mortality) is difficult. Higgins and Bradford (1996) stated that determining the density dependent mechanisms acting on a specific population would be difficult to ascertain with enough certainty to allow population projections. Also, there is high variability in fish responses to flow regulation that is difficult to attribute to factors of interest (Berland et al. 2004, Saltveit et al. 2001, Irvine 2009). Fish stranding rates are highly variable but can be explained partially by the differences in year and season, the extent and magnitude of the flow reduction event (the river stage at the start and end of the stranding event), coupled with variability in the occurrence of pools at each site.

4.2.1 Index and Random Stranding Sites

The first specific hypothesis (H_{01}) from Management Question 2 states: *Fish stranding observed at index sites along the lower Duncan River floodplain is representative of overall stranding*. Originally, the index sites were not selected to be representative of the entire LDR, but to focus salvage efforts on sites believed to have the highest amounts of stranding based on the spatial area dewatered and suitable habitat. Index sites tended to be of lower gradient than random sites (Golder and Poisson 2012, 2019a, 2019b, and 2020; Golder 2017a, 2017b, 2018).

In Study Year 2 (2009 – 2010), there were only 4 out of 14 calibration sites where stranding was observed. Those four sites had a fourfold increase in abundance per unit of dewatered area in comparison to index sites (Golder 2010). Because of the lack of fish species abundance data at these sites (as well as in the index sites), it was unknown if the level of stranding was in relation to abundance or if stranding resulted because of some unmeasured characteristics that affected the susceptibility of fish to stranding at these sites. Consequently, expansion of the stranding data to generate accurate estimates of total fish stranded by species was unable to be conducted at that time and such numbers would not be interpretable into biological impacts.

Interestingly the number of pools per unit area of exposed habitat in Year 6 did not vary between index and non-index sites nor did the number of fish per pools (Golder 2015). This indicated that stranding rates (stranding per lineal km of river) do not differ substantially between index and random sites. In Years 8 to 12 (Golder 2017b, 2018; Golder and Poisson 2019a 2019b, and 2020), as well as during analysis of the combined dataset in Year 13 (2020), there was no significant statistical effect of index and non-index site on pool density, and subsequently pool stranding rates.

The low number of fish in the dataset that were found interstitially stranded precluded the examination of the effect of site type on interstitial stranding. Based on these analyses, index sites do not exhibit a significant bias toward higher stranding rates and therefore, hypothesis H_{01} is accepted.

4.2.2 Rainbow Trout Juvenile Population

The second specific hypothesis (H_{02}) from Management Question 2 states*: Fish populations in the LDR are not significantly impacted by fish stranding events*. Estimates from the combined dataset for the number of Rainbow Trout juveniles stranded in pools are relatively precise. Previous analysis showed that residual wetted areas of pools was not a predictive variable for stranding probability (Poisson 2011, Golder and Poisson 2012).

In the combined dataset, seasonal effects on pool stranding numbers were found to be significant for Rainbow Trout, with mean fall stranding estimates significantly higher than those for winter/spring. This appears to be due to lower juvenile fish densities in the system in the winter/spring versus the fall and a decreased risk of stranding in that period.

Significant differences were not found between the density of stranded fish and substrate size within isolated pools, as well as slope on the formation of pools. Discharge in the LDR was found to influence pool formation and subsequently pool stranding, as the density of pools increased as DRL discharge decreased.

Very few interstitially stranded fish were observed in years when interstitial sample methodologies were standardized and random sampling was initiated. This resulted in high uncertainty related to interstitial stranding estimation in previous study years. Refinements to sampling methodologies has decreased the uncertainty related to interstitial stranding rates of Rainbow Trout and Mountain Whitefish juveniles in the combined dataset. This allowed for the determination of the effect of these rates on population levels (Section 5.0).

Spring abundance of juvenile Rainbow Trout decreased substantially from Study Year 3 (2010 – 2011) to Study Year 8 (2015 – 2016), and in Study Year 11 (2018 – 2019). Andrusak and Thorley (2019) reported that the decline in Study Year 11 (2018 – 2019) abundance was a result of changes in total spawner returns and fluctuations in egg deposition related to variation in size at maturity. This variation in size at maturity was associated with food limitations related to collapse of Kokanee. Conversely, spring abundance increased from Study Year 8 to 10 (2015 – 2017), and in Study Year 12 (2019 – 2020).

Fall abundance of juvenile Rainbow Trout decreased between Study Year 3 (2010 – 2011) and Study Year 6 (2013 – 2014), increased in Study Year 7 (2014 – 2015), followed by sharp decreases in Study Years 8 and 9 (2015 – 2017).

The similarities between spring and fall Rainbow Trout juvenile abundance estimates in 2010 and 2015, and the higher abundance estimates for spring versus fall in 2013 and 2016 were surprising, given that Decker and Hagen (2009) estimated the overwintering mortality to be approximately 71%. In Study Year 10 (2017 – 2018; Golder and Poisson 2019a), it was speculated that this discrepancy may be because the assumed observer efficiency estimates for the fall abundance estimates were too high (based on observer efficiencies reported in Andrusak 2017). Including updated observer efficiencies (Andrusak and Thorley 2018) in Study Year 11 (2018 – 2019) did not correct this discrepancy (Golder and Poisson 2019b). As reported in Study Year 10 (2017 – 2018; Golder and Poisson 2019a), if the decreasing juvenile Rainbow Trout populations documented by the Study Year 9 (2016 – 2017) fall abundance surveys is factual, it may be linked to a decline in Lardeau River Gerrard Rainbow escapement into the Duncan River (Andrusak and Andrusak 2015). These finding should be interpreted with caution as the models used in the individual programs were different.

As fall abundance surveys have not been conducted since Study Year 9 (2016 – 2017), and 2020 spring estimates were not available, estimated Rainbow Trout juvenile abundance for the combined dataset was calculated based on spring surveys conducted by Andrusak and Thorley (2019). Total mean annual estimates in the combined dataset (2006 to 2020) for the number of Rainbow Trout juveniles stranded were highly variable, but consistently lower than 5 % in most Study Years. Annual Rainbow Trout Stranding estimates ranged from 1.7% (95% CRI of 1.0 – 2.7%) of the Rainbow Trout age-1 spring population in 2010 to 10.1% (95% CRI of 5.8 – 19.1%) in 2014. These annual estimates were higher than reported in previous study years (Golder and Poisson 2019a, 2019b and 2020), which is the result of the inclusion of non-randomly sampled Rainbow Trout into the combined dataset and season replacing slope as the main predictor of stranding. Although higher, the difference was not statistically significant as confidence intervals overlapped between the current estimates and those in previous years indicates.

Based on the mean annual total percent stranding estimates of approximately 10% of the juvenile population for 2014, 2015, and 2018, significant impacts to the Lower Duncan River Rainbow Trout population may have occurred for these spawn years. Therefore, with the current state of knowledge study hypothesis H_{02} cannot be accepted for Rainbow Trout. The findings of Andrusak and Thorley (2019), and estimated increases in age-1 spring abundance in 2016, 2017 and 2019 after the three highest years of estimated total stranding provide evidence on the limited severity of the potential impacts.

4.2.3 Mountain Whitefish Juvenile Population

The second specific hypothesis (H_{02}) from Management Question 2 states*: Fish populations in the LDR are not significantly impacted by fish stranding events*. There were no spring abundance estimates for Mountain Whitefish in the DDMMON-16 combined dataset. The fall total abundance estimates for Mountain Whitefish obtained using abundance modelling (Table 7) increased between Study Years 3 and 6 (2010 – 2013), decreased from Years 6 to 8 (2014 – 2016), while stabilizing in Year 9 (2016 – 2017). Mountain Whitefish encounters have been consistently low in all study years. Fall Mountain Whitefish abundance estimates in Study Years 6 to 9 (2013 – 2017) were substantially higher than Rainbow Trout estimates, while encounters of stranded fish were substantially lower. The consistently low level of stranding was not considered ecologically significant and will likely not result in a population level effect on juvenile Mountain Whitefish. Based on the current state of knowledge, hypothesis H_{02} is not rejected for Mountain Whitefish. Therefore, it can be concluded that fish stranding as a result of DDM operations does not considerably affect juvenile Mountain Whitefish populations.

5.0 SUMMARY

The key findings for the Year 13 of the Lower Duncan River Fish Stranding Impact Monitoring Program (DDMMON-16) are as follows:

Outstanding DDMMON-1 Management Question 1) *(What is the relationship between fish stranding and):*

- *Rate of river stage/total stage change* – the findings of the DDMMON-16 Program corroborate the conclusion of the DDMMON-1 Program (larger ramping rates lead to increased risk of stranding; Section 4.1.1.1). Predicted proportional stranding also indicates larger total reduction magnitudes lead to increased risk of stranding.
- *Time of day (day/night)* – the findings of the DDMMON-1 Program (higher risk to strand juvenile and small bodied fish at night; Section 4.1.1.2) could not be tested
- *Substrate* – the findings of the DDMMON-16 Program corroborate the conclusion of the DDMMON-1 Program (link between larger substrates and higher stranding rates; Section 4.1.1.3).
- *Habitat configuration (channel bed gradient and topography)* – the findings of the DDMMON-16 program indicate there is a relationship, although not statistically significant, between channel gradient and interstitial stranding rates (Section 4.1.1.4). There were no statistically significant differences between stranding rates among habitat configuration (side-channel, mainstem bar).
- *Cover* – no evidence was found to refute the findings of the DDMMON-1 Program (increased cover availability is linked to increased stranding; Section 4.1.1.5).
- *Species* – there were differences in stranding rates between the target species of the DDMMON-16, which violates the DDMMON-1 equal risk assumption (Section 4.1.1.6).

  - *Time of year (spring, fall, winter)* – the findings of DDMMON-16 indicate that fall stranding rates for Rainbow Trout are significantly higher than in the winter/spring season (Section 4.1.1.7).
  - *Habitat stability (wetted history)* – there was no relationship between wetted history and fish stranding. Therefore, the DDMMON-16 Program did not find evidence to support the DDMMON-1 finding of a non-significant trend that increased wetted history is linked to increased stranding risk (4.1.1.8).

- **DDMMON-16 Management Question 1**) (*How effective are the operating measures implemented as part of the ASPD program?):*

  - Based on the current state of knowledge, the flow reduction measures implemented under the ASPD are effective at reducing fish stranding by reducing the amount and rate at which habitat becomes dewatered during DDM operations (Section 4.1.2).

- **DDMMON-16 Management Question 2**) (*What are the levels of impact to resident fish populations associated with fish stranding events on the lower Duncan River?):*

  - The seasonal effect on pool stranding was statistically significant for Rainbow Trout, with mean fall stranding estimates significantly higher than those for winter/spring (Section 4.1.1.7 and Section 4.2.2). A significant effect on pool stranding for Mountain Whitefish was not found (Section 4.1.1.7 and Section 4.2.3).

  - Interstitial stranding encounters were very low in all study years (Section 4.2.2 and Section 4.2.3).

  - Slope has an effect on interstitially stranded fish counts, although this effect is not statistically significant (Section 4.2.2).

  - There was no statistically significant relationship between pool density and slope (Section 4.2.2).

- DDMMON-16 Study Hypothesis H_{01}: *(Fish stranding observed at index sites along the lower Duncan River floodplain is representative of overall stranding)*:

  - There was no significant effect of site type on pool formation and pool stranding rates (Section 4.2.1).

  - The low number of fish in the dataset that were found interstitially stranded precluded the examination of the effect of index/random site on interstitial stranding (Section 4.2.1).

  - The study hypothesis H_{01} is accepted.

- DDMMON-16 Study Hypothesis H_{02}*: (Fish populations in the LDR are not significantly impacted by fish stranding events)*:

  - The study hypothesis H_{02} for Rainbow Trout cannot be accepted (Section 4.2.2).

  - The study hypothesis H_{02} Mountain Whitefish is accepted (Section 4.2.3).

In summary, this monitoring program provides an understanding of fish stranding in relation to DDM operations and its results may help management reduce the severity of fish stranding in the LDR. Based on the current state of knowledge, DDMMON-16 Objective 1) (*To assess the effectiveness of the operating measures implemented as part of the ASPD program)* was addressed with the determination that the flow reduction measures implemented under the WUP are effective at reducing fish stranding. Whenever feasible, flow reductions at DDM should follow recommendations made by the Adaptive Stranding Protocol and the various studies conducted on the LDR. With the growth of the dataset, the uncertainty of the mean estimated total stranding of Rainbow Trout for all study years was reduced. DDMMON-16 Objective 2) (*To empirically assess the influence of stranding events on resident and/or rearing fish population levels in the Lower Duncan River*) was also addressed. This program determined that stranding events may have led to significant impacts to the Lower Duncan River Rainbow Trout population for the 2014, 2015, and 2018 spawn years. The consistently low level of stranding of juvenile Mountain Whitefish was not considered ecologically significant and will likely not result in a population level effect.

After Study Year 10 (2017 – 2018), one of the main focusses of this program was to reduce the uncertainty related to interstitial stranding estimation. For Study Years 11 to 13 (2018 – 2020), several methods to reduce interstitial uncertainty were proposed including modifications to the interstitial sampling methodology and data analysis for stranding estimation, as well as substrate mapping. The modifications to interstitial sampling data analysis methodologies have proven effective at greatly reducing stranding estimation uncertainty.

6.0 RECOMMENDATIONS

Recommendations from the current year (Year 13) of the Lower Duncan River Fish Stranding Impact Monitoring Program (DDMMON-16) are as follows:

- For future stranding responses on the LDR, as many stranding sites (identified in the DDMMON-16 Program) as possible should be sampled during each stranding response in the following order, based on classification in the corresponding Lower Duncan River Stranding Database query:
    a. Major Effect – High Priority;
    b. No Data/Reconnaissance Survey – Moderate Priority; and,
    c. Minor Effect – Low Priority.
- Continue to sample selected sites in order from upstream to downstream. This approach ensures that the field crew does not move ahead of the receding water levels.
- To ensure that data is collected in a manner that will strengthen the Lower Duncan River Stranding Database, field crews should record habitat data (i.e., area of pools, cover types present, pool complexity, dominant substrate, GPS waypoint, etc.) for all stranding mechanisms sampled.

These recommendations are designed to shift the focus of future stranding responses to fish salvage as a mitigation to potential impacts related to DDM operations. The focus of stranding responses moving forward should also be on strengthening the Lower Duncan River Stranding Database (i.e., by filling in gaps in the database to more effectively inform future stranding responses), as well as maintaining as much sampling methodology consistency as possible to facilitate comparisons with historical data.

7.0 CLOSURE

We trust that this report meets your current requirements. If you have any further questions, please do not hesitate to contact the undersigned.

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BH/SR/cmc

https://golderassociates.sharepoint.com/sites/31732g/deliverables/issued to client-for wp/18107549-006-r-rev0/18107549-006-r-rev0-ddmmon-16 2020 year 13 23 dec_21 .docx

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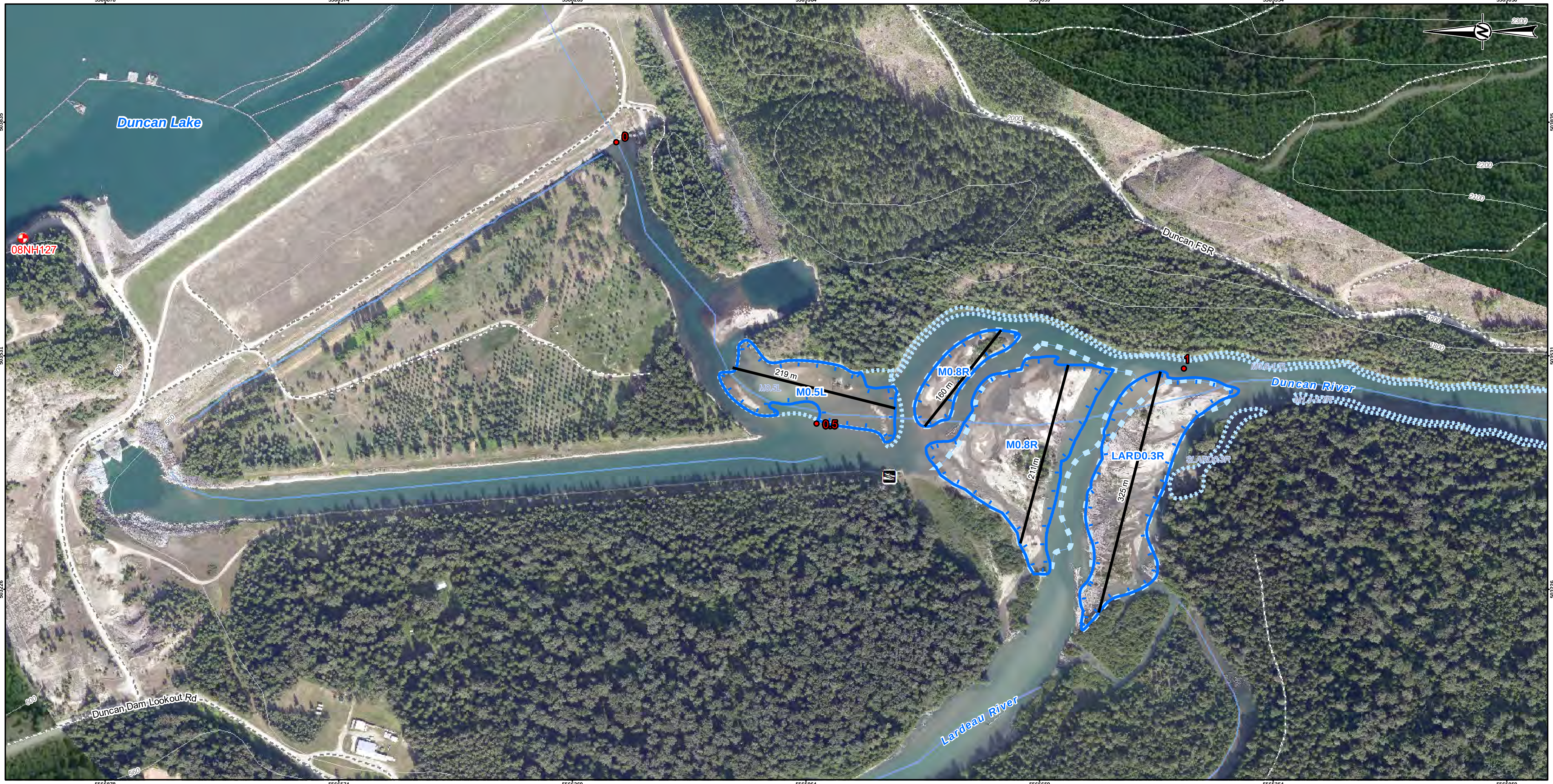
APPENDIX A

Project Maps and Sampling Chronology



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- INDEX

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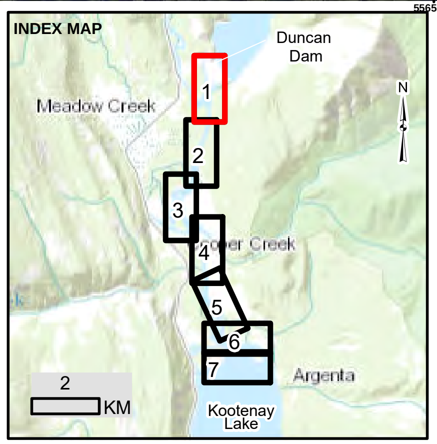
- RIVER KILOMETRE MARKER
- LOCAL ROAD
- CONTOUR (ft)
- WATERCOURSE

INDEX MAP



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4. COORDINATE SYSTEM: NAD 1983 UTM ZONE 11N

CLIENT
BC HYDRO

PROJECT
LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING (DDMMON-16)

CONSULTANT

GOLDER

YYYY-MM-DD	2021-01-25
DESIGNED	BH
PREPARED	JG
REVIEWED	SR
APPROVED	BH

TITLE
LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES

PROJECT NO.	CONTROL	REV.	FIGURE
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IF THIS MEASUREMENT DOES NOT MATCH WHAT IS SHOWN, THE SHEET SIZE HAS BEEN MODIFIED FROM: ANSI B

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- STRANDING SITE (2018)

FORMER STRANDING SITE (2012)

- NON-INDEX
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- LOCAL ROAD
- CONTOUR (ft)
- WATERCOURSE

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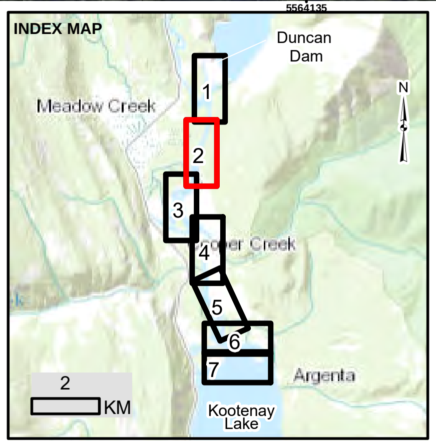

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CLIENT
BC HYDRO

PROJECT
LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING (DDMMON-16)

CONSULTANT
GOLDER

YYYY-MM-DD	2021-01-25
DESIGNED	BH
PREPARED	JG
REVIEWED	SR
APPROVED	BH

TITLE
LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES

PROJECT NO.	CONTROL	REV.	FIGURE
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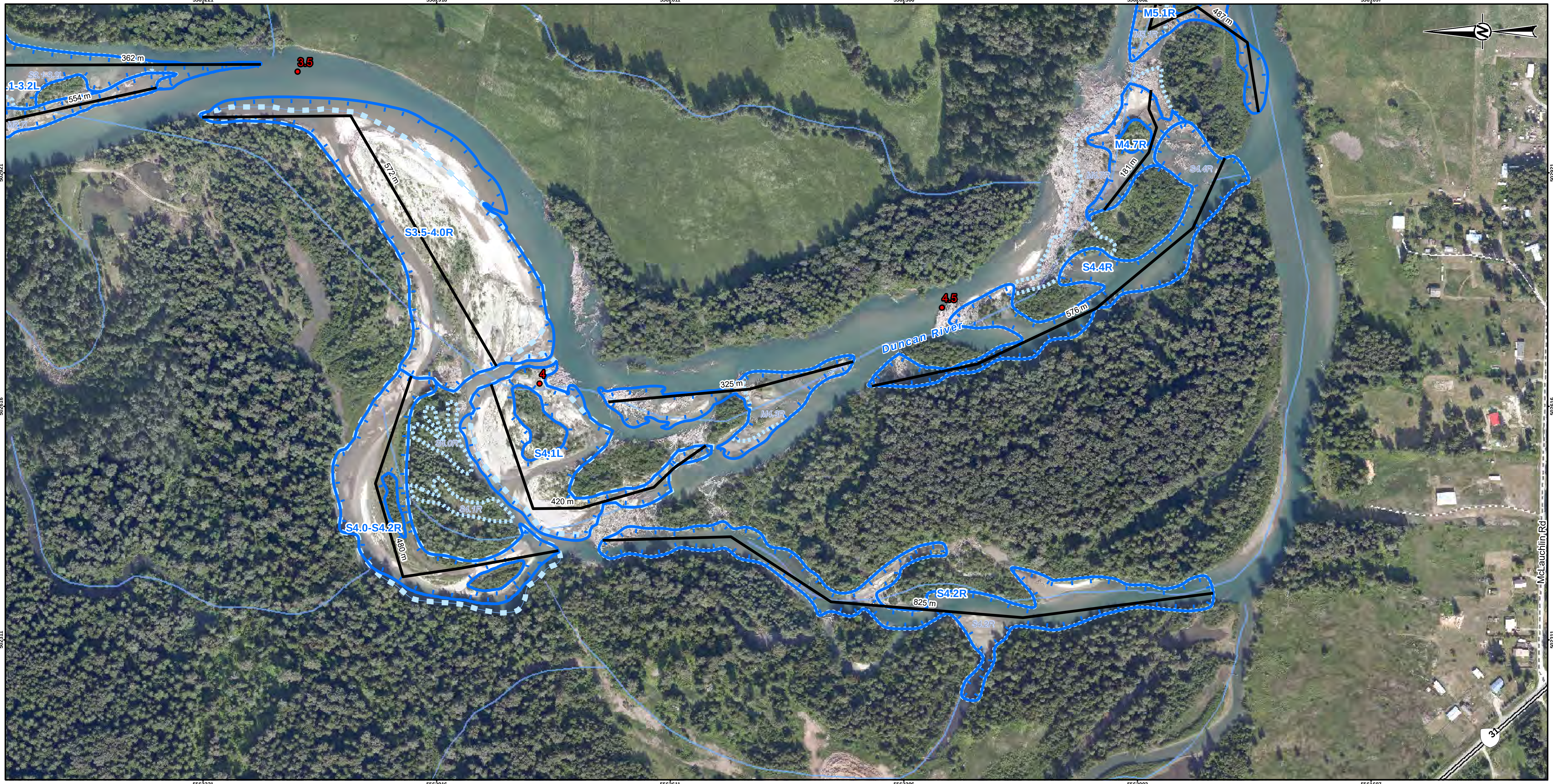

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- LOCAL ROAD
- WATERCOURSE

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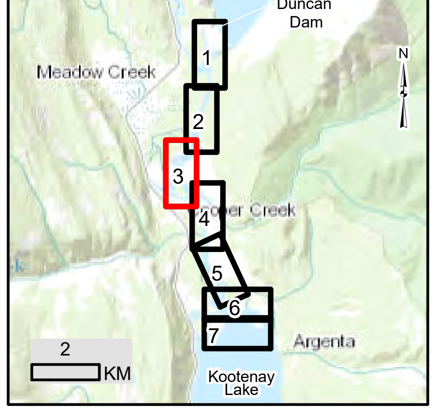

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4. COORDINATE SYSTEM: NAD 1983 UTM ZONE 11N

CLIENT
BC HYDRO

PROJECT
LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING (DDMMON-16)

CONSULTANT
GOLDER

YYYY-MM-DD	2021-01-25
DESIGNED	BH
PREPARED	JG
REVIEWED	SR
APPROVED	BH

TITLE
LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES

PROJECT NO.	CONTROL	REV.	FIGURE
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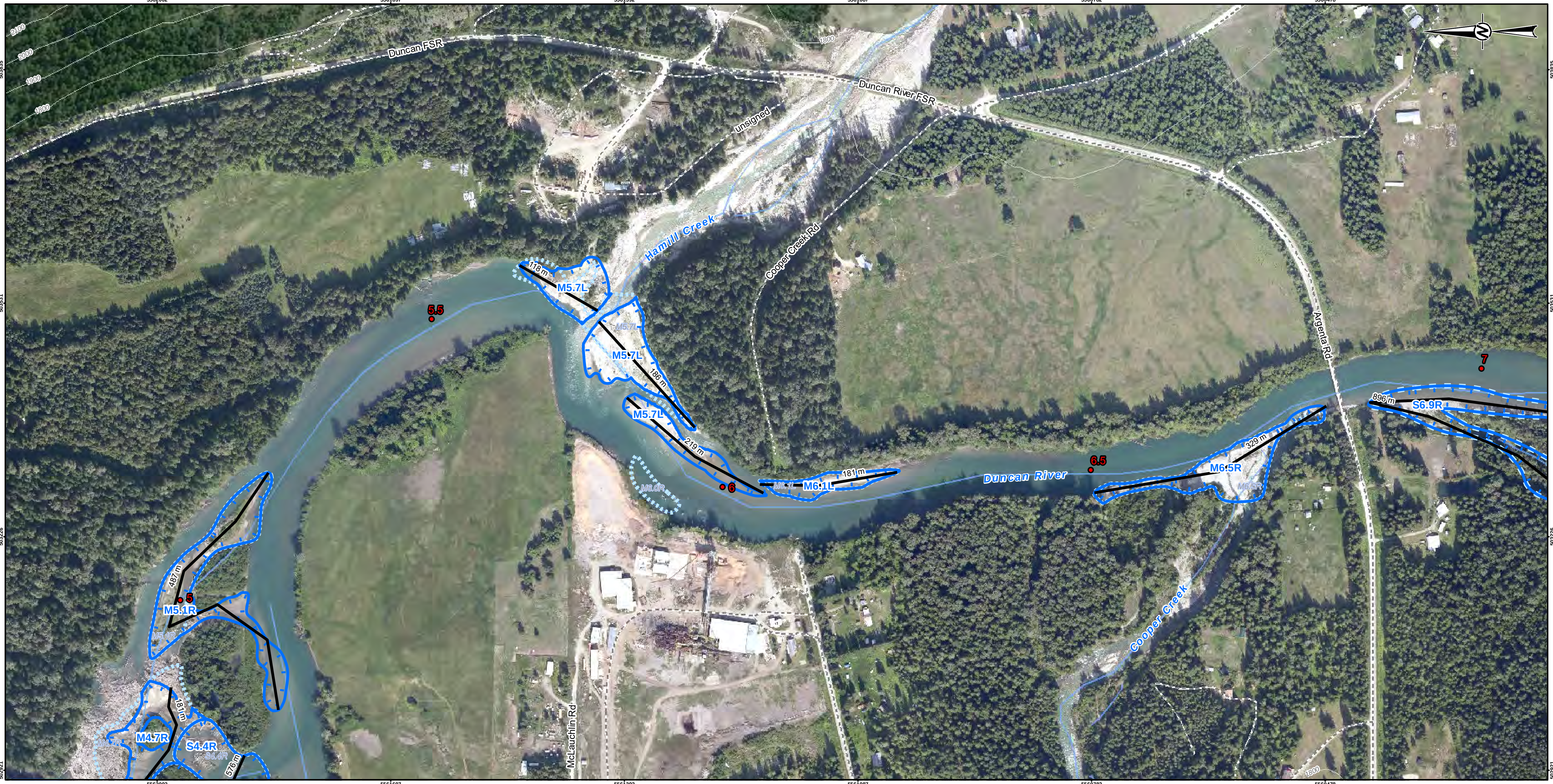

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- STRANDING SITE (2018)

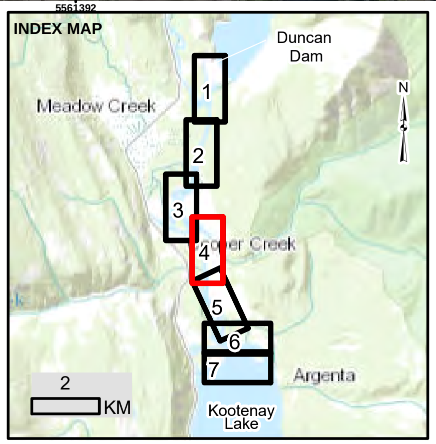
FORMER STRANDING SITE (2012)

- NON-INDEX
- INDEX

BASEMAP FEATURE

- WSC GAUGE
- RIVER KILOMETRE MARKER
- LOCAL ROAD
- CONTOUR (ft)
- WATERCOURSE

INDEX MAP



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3. CONTAINS INFORMATION LICENCED UNDER THE OPEN GOVERNMENT LICENCE – BRITISH COLUMBIA.
4. COORDINATE SYSTEM: NAD 1983 UTM ZONE 11N

CLIENT	BC HYDRO	PROJECT	LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING (DDMMON-16)
CONSULTANT	GOLDER	TITLE	LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES
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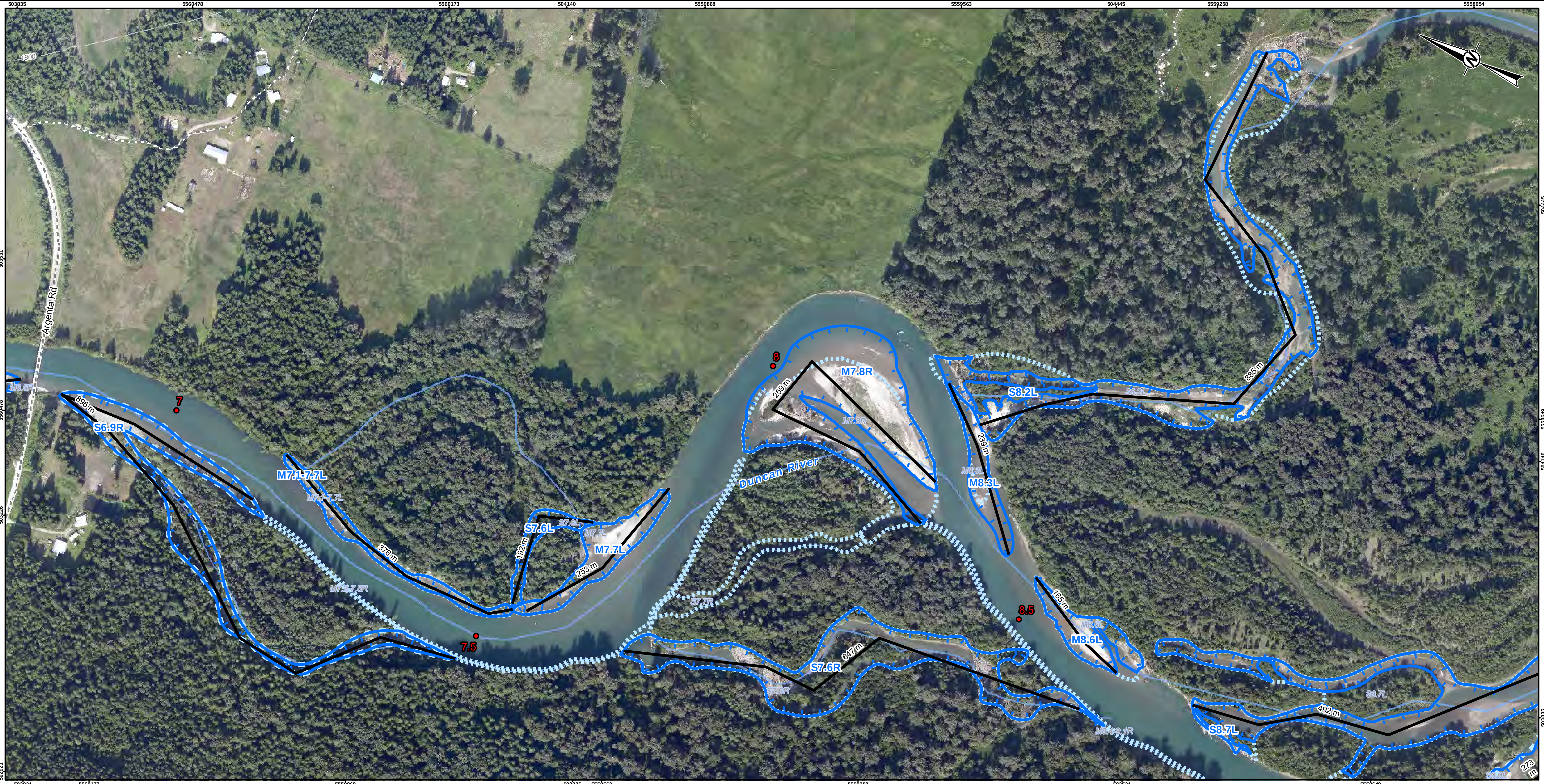
APPROVED	BH		



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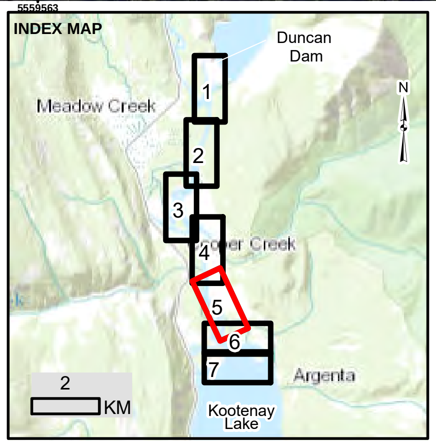
- STRANDING SITE (2018)

FORMER STRANDING SITE (2012)

- NON-INDEX
- INDEX

BASEMAP FEATURE

- WSC GAUGE
- RIVER KILOMETRE MARKER
- LOCAL ROAD
- CONTOUR (ft)
- WATERCOURSE



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CLIENT
BC HYDRO

PROJECT
LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING (DDMMON-16)

CONSULTANT

GOLDER

YYYY-MM-DD	2021-01-25
DESIGNED	BH
PREPARED	JG
REVIEWED	SR
APPROVED	BH

TITLE
LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES

PROJECT NO.	CONTROL	REV.	FIGURE
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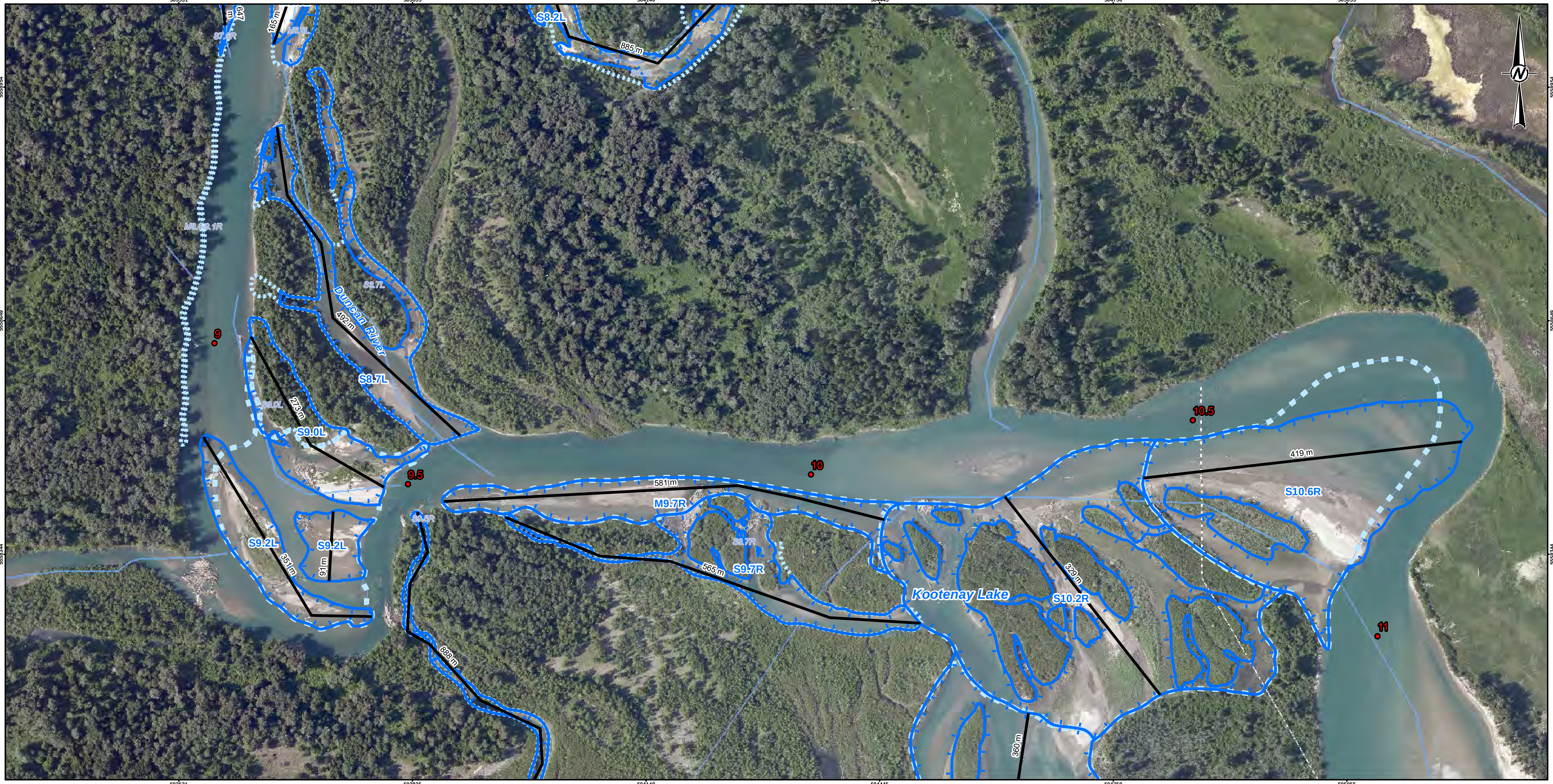

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STRANDING SITE (2018)

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BASEMAP FEATURE

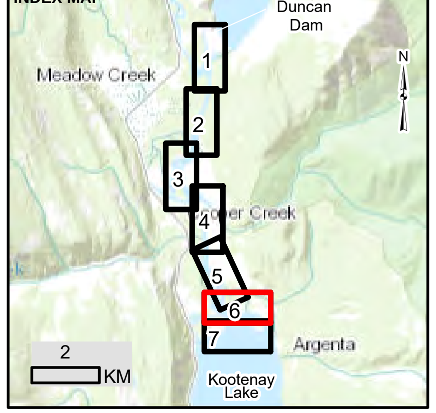
WSC GAUGE

RIVER KILOMETRE MARKER

LOCAL ROAD

WATERCOURSE

INDEX MAP



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CLIENT
BC HYDRO

PROJECT
LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING (DDMMON-16)

CONSULTANT

GOLDER

YYYY-MM-DD	2021-01-25
DESIGNED	BH
PREPARED	JG
REVIEWED	SR
APPROVED	BH

TITLE
LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES

PROJECT NO.	CONTROL	REV.	FIGURE
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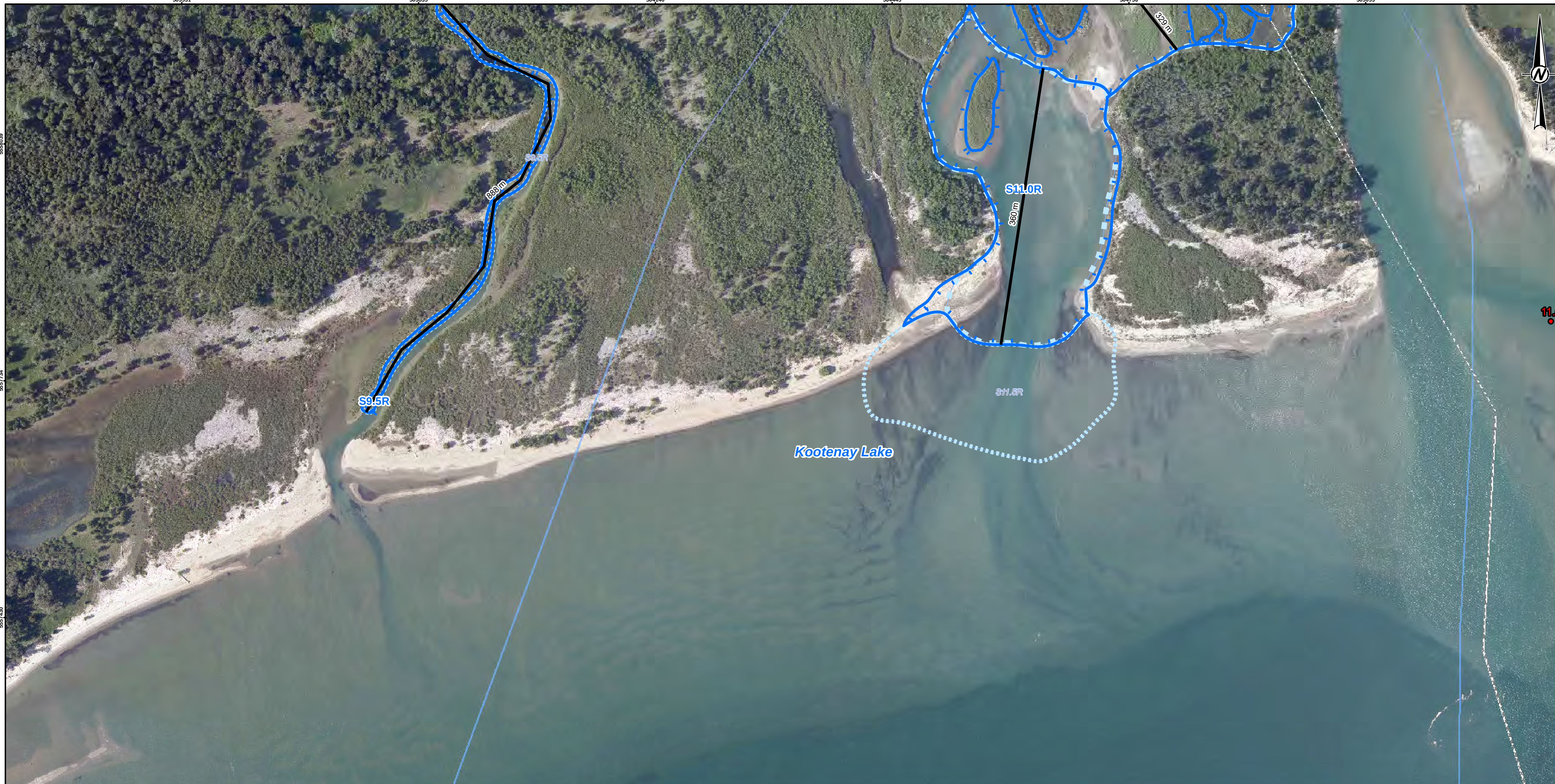

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- WSC GAUGE
- RIVER KILOMETRE MARKER
- LOCAL ROAD
- WATERCOURSE

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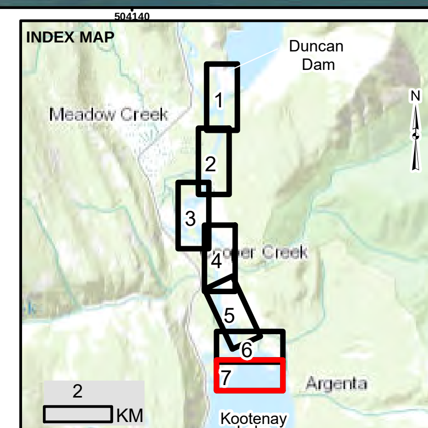

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CLIENT
BC HYDRO

PROJECT
LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING (DDMMON-16)

CONSULTANT

GOLDER

YYYY-MM-DD	2021-01-25
DESIGNED	BH
PREPARED	JG
REVIEWED	SR
APPROVED	BH

TITLE
LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES

PROJECT NO.	CONTROL	REV.	FIGURE
1810759	-	0	7

Table A1: Chronology of sampling activities for the 2008 - 2009 Lower Duncan River Fish Stranding Impact Monitoring, Year 1 Program.

Date(s)	Sampling Activities	Reduction Event Number	Number of Snorkel Sites Surveyed	Number of Index Sites Stranding Assessed	Number of Non-Index Stranding Sites Assessed
11 April 2008	Stranding Assessments	RE2008-02	-	5	-
15 April 2008	Stranding Assessments	RE2008-03	-	5	-
28 April 2008	Stranding Assessments	RE2008-04	-	6	-
22 July 2008	Stranding Assessments	RE2008-05	-	6	-
26 August 2008	Stranding Assessments	RE2008-06	-	6	-
25 September 2008	Stranding Assessments	RE2008-07	-	6	-
28 September 2008	Stranding Assessments	RE2008-08	-	5	-
01 October 2008	Stranding Assessments	RE2008-09	-	6	-
28 February 2009	Stranding Assessments	RE2009-01	-	2	-

Table A2: Summary of DDM flow reduction events in Study Year 1, from 11 April 2008 to 28 February 2009, for events when fish stranding assessments were conducted.

Date	Reduction Event (RE)	DDM Discharge m^3/s (ft^3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
11 April 2008	RE 2008-02	65 (2295)	57 (2013)	8 (283)	Single reduction of 8 m^3/s (283 ft^3/s)	Onset of reservoir storage
15 April 2008	RE 2008-03	40 (1412)	34 (1200)	6 (212)	Single reduction of 6 m^3/s (212 ft^3/s)	Delayed freshet, low reservoir
28 April 2008	RE 2008-04	49 (1730)	37 (1306)	12 (424)	Single reduction of 12 m^3/s (424 ft^3/s)	Delayed freshet, low reservoir
22 July 2008	RE 2008-05	31 (1095)	3 (106)	28 (989)	Two flow reductions of 14 m^3/s (494 ft^3/s)	High inflows, reservoir/flood management
26 August 2008	RE 2008-06	275 (9711)	178 (6286)	96 (3390)	Four flow reductions ranging from 15 to 28 m^3/s (530 to 989 ft^3/s)	High inflows, reservoir/flood management
25 September 2008	RE 2008-07	210 (7416)	150 (5297)	60 (2119)	Two flow reductions of 30 m^3/s (1059 ft^3/s)	Kokanee protection flows
28 September 2008	RE 2008-08	150 (5297)	90 (3178)	60 (2119)	Two flow reductions of 30 m^3/s (1059 ft^3/s)	Kokanee protection flows
01 October 2008	RE 2008-09	97 (3425)	40 (1412)	57 (2013)	Two flow reductions of 28 and 29 m^3/s (989 to 1024 ft^3/s)	Kokanee protection flows
28 February 2009	RE 2009-01	194 (6851)	76 (2684)	118 (4167)	Four flow reductions ranging from 28 to 30 m^3/s (989 to 1059 ft^3/s)	Begin of reservoir storage

[a] The flow decreases reflect the net total decrease in flows over specific intervals at DDM. Actual ramping rate (rate of stage or discharge decrease per unit time) at each of the stranding sites may be significantly higher over a shorter time interval or possibly attenuated to a lower rate at the downstream locations where stranding was observed.

Table A3: Chronology of sampling activities for the 2009 - 2010 Lower Duncan River Fish Stranding Impact Monitoring, Year 2 Program

Date(s)	Sampling Activities	Reduction Event Number	Number of Snorkel Sites Surveyed	Number of Index Sites Stranding Assessed	Number of Non-Index Stranding Sites Assessed
25 April 2009	Stranding Assessments	RE2009-02	-	6	-
25 September 2009	Stranding Assessments	RE2009-03	-	6	-
28 September 2009	Stranding and Calibration Assessments	RE2009-04	-	7	13
01 October 2009	Stranding Assessments	RE2009-05	-	5	-
22 January 2010	Stranding Assessments	RE2010-01	-	5	-
01 March 2010	Stranding Assessments	RE2010-02	-	5	-

Table A4: Summary of DDM flow reduction events in Study Year 2, from 25 April 2009 to 1 March 2010, for events when fish stranding assessments were conducted.

Date	Reduction Event (RE)	DDM Discharge m^3/s (ft^3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
25 April 2019	RE 2009-02	83 (2931)	76 (2684)	7 (247)	Down 9 m^3/s (318 ft^3/s) at 10:00.	Delayed freshet, low reservoir
25 September 2009	RE 2009-03	212 (7487)	184 (6498)	28 (989)	Down 28 m^3/s (989 ft^3/s) at 08:00.	Kokanee protection flows
28 September 2009	RE 2009-04	180 (6357)	128 (4520)	75 (2649)	Down 38 m^3/s (1342 ft^3/s) at 07:00, down 37 m^3/s (1306 ft^3/s) at 08:00.	Kokanee protection flows
01 October 2009	RE 2009-05	126 (4450)	76 (2684)	70 (2472)	Down 35 m^3/s (1236 ft^3/s) at 07:00, down 35 m^3/s (1236 ft^3/s) at 08:00.	Kokanee protection flows
22 January 2010	RE 2010-01	250 (8829)	207 (7310)	42 (1483)	Down 28 m^3/s (989 ft^3/s) at 07:00, down 14 m^3/s (494 ft^3/s) at 08:00.	Begin of reservoir storage
1 March 2010	RE 2010-02	190 (6710)	101 (3567)	89 (3143)	Down 28 m^3/s (989 ft^3/s) at 06:00, down 28 m^3/s (989 ft^3/s) at 07:00, down 22 m^3/s (777 ft^3/s) at 08:00.	Begin of reservoir storage

[a] The ramping rates reflect the net total decrease in flows over an hourly interval at DRL. Actual ramping rates at particular stranding sites may be significantly higher over a shorter time interval or possibly attenuated to a lower rate at the downstream locations where stranding was observed.

Table A5: Chronology of sampling activities for the 2010 - 2011 Lower Duncan River Fish Stranding Impact Monitoring, Year 3 Program

Date(s)	Sampling Activities	Reduction Event Number	Number of Snorkel Sites Surveyed	Number of Index Sites Stranding Assessed	Number of Non-Index Stranding Sites Assessed
27 August 2010	Stranding Assessments	RE2010-03	-	7	1
25 September 2010	Stranding Assessments	RE2010-04	-	7	3
28 September 2010	Stranding Assessments	RE2010-05	-	11	3
01 October 2010	Stranding Assessments	RE2010-06	-	10	13
01 March 2011	Stranding Assessments	RE2011-01	-	7	-
02 March 2011	Stranding Assessments	RE2011-02	-	4	-
12 April 2011	Stranding Assessments	RE2011-03	-	5	-

Table A6: Summary of DDM flow reduction events in Study Year 3, from 27 August 2010 to 12 April 2011, for events when fish stranding assessments were conducted.

Date	Reduction Event (RE)	DDM Discharge m^3/s (ft^3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
27 August 2010	RE 2010-03	143 (5050)	119 (4202)	24 (848)	Down 24 m^3/s (848 ft^3/s) at 08:00.	DDM Spillway Gate Testing
25 September 2010	RE 2010-04	144 (5085)	123 (4344)	21 (742)	Down 21 m^3/s (742 ft^3/s) at 08:00.	Kokanee protection flows
27 September 2010	RE 2010-05	123 (4344)	54 (1907)	69 (2437)	Down 20 m^3/s (706 ft^3/s) at 07:00, down 20 m^3/s (706 ft^3/s) at 08:00, down 20 m^3/s(706 ft^3/s) at 9:00 and down 9 m^3/s (318 ft^3/s) at 11:00.	Kokanee protection flows
01 October 2010	RE 2010-06	126 (4450)	76 (2684)	50 (1766)	Down 35 m^3/s (1236 ft^3/s) at 07:00, down 35 m^3/s (1236 ft^3/s) at 08:00.	Kokanee protection flows
01 March 2011	RE 2011-01	241 (8511)	168 (5933)	73 (2578)	Down 7 m^3/s (247 ft^3/s) every 15 minutes, from 6:15 until 8:30	Begin of reservoir storage
02 March 2011	RE 2011-02	168 (5933)	93 (3284)	75 (2649)	Down 7 m^3/s (247 ft^3/s) every 15 minutes, from 6:15 until 8:30	Reservoir storage
12 April 2011	RE 2011-03	72 (2543)	52 (1836)	20 (706)	Down 10 m^3/s (353 ft^3/s) at 9:00 and 9:30.	Discharge reduced to compensate for low inflows

[a] The flow decreases reflect the net total decrease in flows over specific intervals at DDM. Actual ramping rates (rate of stage or discharge decrease per unit time) at particular stranding sites may be significantly higher over a shorter time interval or possibly attenuated to a lower rate at the downstream locations where stranding was observed.

Table A7: Chronology of sampling activities for the 2011 - 2012 Lower Duncan River Fish Stranding Impact Monitoring, Year 4 Program.

Date(s)	Sampling Activities	Reduction Event Number	Number of Snorkel Sites Surveyed	Number of Index Sites Stranding Assessed	Number of Non-Index Stranding Sites Assessed
19 April 2011	Stranding Assessments	RE2011-04	-	4	1
01 June 2011	Stranding Assessments – start of random selection process for sample sites	RE2011-05	-	10	4
25 August 2011	Stranding Assessments	RE2011-06	-	7	3
25 September 2011	Stranding Assessments	RE2011-07	-	2	3
28 September 2011	Stranding Assessments	RE2011-08	-	2	2
01 October 2011	Stranding Assessments	RE2011-09	-	2	3
20 January 2012	Stranding Assessments	RE2012-01	-	3	4
01 March 2012	Stranding Assessments	RE2012-02	-	3	2

Table A8: Summary of DDM flow reduction events in Study Year 4, from 19 April 2011 to 20 January 2012, for those events when fish stranding assessments were conducted.

Date	Reduction Event (RE)	DDM Discharge m^3/s (ft^3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
19 April 2011	RE 2011-04	46 (1624)	18 (636)	28 (989)	Down 7 m^3/s (247 ft^3/s) at 14:00 on 18-Apr, down 7 m^3/s (247 ft^3/s) at 08:00, 08:15 and 08:30 on 19-Apr.	Discharge reduced to compensate for low inflows.
01 June 2011	RE 2011-05	82 (2896)	0 (0)	82 (2896)	Down 28 m^3/s (989 ft^3/s) at 07:00 and 08:00, down 26 m^3/s (918 ft^3/s) at 09:00.	To meet recreation water level targets in Duncan Reservoir.
25 August 2011	RE 2011-06	217 (7663)	161 (5686)	56 (1978)	Down 28 m^3/s (989ft^3/s) at 08:00 and 09:00.	To meet late summer flow targets in the lower Duncan River.
25 September 2011	RE 2011-07	190 (6710)	130 (4591)	60 (2119)	Down 30 m^3/s (1059 ft^3/s) at 07:30 and 08:30.	Onset of Kokanee protection flows.
28 September 2011	RE 2011-08	130 (4591)	70 (2472)	60 (2119)	Down 30 m^3/s (1059 ft^3/s) at 07:30 and 08:30.	Kokanee protection flows.
01 October 2011	RE 2011-09	70 (2472)	40 (1412)	30 (1059)	Down 15 m^3/s (530 ft^3/s) at 07:30 and 08:30.	Final transition to Kokanee protection flows.
20 January 2012	RE 2012-01	202 (7134)	164 (5792)	38 (1342)	Down 19 m^3/s (671 ft^3/s) at 07:00 and 08:00.	Discharge reduced to meet reservoir targets.
01 March 2012	RE2012-02	182 (6427)	79 (2790)	103 (3637)	Down 31 m^3/s (1095 ft^3/s) at 06:00, down 24 m^3/s (848 ft^3/s) at 07:00, 08:00 and 09:00.	Discharge reduced to meet reservoir targets.

[a] The flow decreases reflect the net total decrease in flows over specific intervals at DDM. Actual ramping rates (rate of stage or discharge decrease per unit time) at particular stranding sites may be significantly higher over a shorter time interval or possibly attenuated to a lower rate at the downstream locations where stranding was observed.

Table A9: Chronology of sampling activities for the 2012 - 2013 Lower Duncan River Fish Stranding Impact Monitoring, Year 5 Program

Date(s)	Sampling Activities	Reduction Event Number	Number of Snorkel Sites Surveyed	Number of Index Sites Stranding Assessed	Number of Non-Index Stranding Sites Assessed
15 April 2012	Stranding Assessments	RE2012-03	-	2	0
01 June 2012	Stranding Assessments	RE2012-04	-	Assessment cancelled by BC Hydro prior to reduction date	
26 September 2012	Stranding Assessments	RE2012-05	-	4	5
27 September 2012	Stranding Assessments	RE2012-06	-	3	2
01 October 2012	Stranding Assessments	RE2012-07	-	3	3
21 January 2013	Stranding Assessments	RE2013-01	-	5	6
01 March 2013	Stranding Assessments	RE2013-02	-	3	2

Table A10: Summary of DDM flow reduction events in Study Year 5, from 1 April 2012 to 31 March 2013, for events when fish stranding assessments were conducted.

Date	Reduction Event (RE)	DDM Discharge m^3/s (ft^3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
15 April 2012	RE 2012-03	90 (3178)	46 (1624)	44 (1554)	Down 20 m^3/s (706 ft^3/s) at 14:00, and down 14 m^3/s (494 ft^3/s) at 15:00.	Discharge reduced to compensate for low inflows.
1 June 2012	RE 2012-04	Assessment cancelled.				
26 September 2012	RE 2012-05	196 (6922)	140 (4944)	56 (1978)	Down 20 m^3/s (706 ft^3/s) at 06:30, 07:30, and 08:30.	Onset of Kokanee protection flows.
27 September 2012	RE 2012-06	140 (4944)	80 (2825)	60 (2119)	Down 20 m^3/s (706 ft^3/s) at 06:30, 07:30, and 08:30.	Kokanee protection flows.
1 October 2012	RE 2012-07	80 (2825)	41 (1448)	39 (1377)	Down 20 m^3/s (706 ft^3/s) at 05:30 and 06:30.	Final transition to Kokanee protection flows.
21 January 2013	RE 2013-01	244 (8617)	185 (6533)	59 (2084)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 06:00 to 07:45.	Discharge reduced to meet reservoir targets.
1 March 2013	RE 2013-02	170 (6003)	80 (2825)	90 (3178)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 06:00 to 08:45, and a final drop of 5 m^3/s (177 ft^3/s) at 09:00.	Discharge reduced to meet flow target at DRL.

[a] The flow decreases reflect the net total decrease in flows over specific intervals at DDM. Actual ramping rates (rate of stage or discharge decrease per unit time) at particular stranding sites may be significantly higher over a shorter time interval or possibly attenuated to a lower rate at the downstream locations where stranding was observed.

Table A11: Chronology of sampling activities for the 2013 - 2014 Lower Duncan River Fish Stranding Impact Monitoring, Year 6 Program

Date(s)	Sampling Activities	Reduction Event Number	Number of Snorkel Sites Surveyed	Number of Index Sites Stranding Assessed	Number of Non-Index Stranding Sites Assessed
14 and 15 September 2013	Abundance Estimation	-	Study Area Reconnaissance and Site Selection		
16 September 2013	Abundance Estimation	-	5	-	-
17 September 2013	Abundance Estimation	-	7	-	-
18 September 2013	Abundance Estimation	-	10	-	-
19 September 2013	Abundance Estimation	-	12	-	-
21 September 2013	Stranding Assessments	RE2013-03	-	3	5
24 September 2013	Stranding Assessments	RE2013-04	-	2	2
27 September 2013	Stranding Assessments	RE2013-05	-	2	4
21 January 2014	Stranding Assessments	RE2014-01	-	4	4
01 March 2014	Stranding Assessments	RE2014-02	-	2	2

Table A12: Summary of DDM flow reduction events in Study Year 6, from 21 September 2013 to 1 March 2014, for events when fish stranding assessments were conducted.

Date	Reduction Event	DDM Discharge m3/s (ft3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
21 September 2013	RE 2013-03	212(7487)	162 (5721)	50 (1766)	Down 6.25 m^3/s (221 ft^3/s) every 15 minutes from 07:00 to 08:45.	Onset of Kokanee protection flows.
24 September 2013	RE 2013-04	170 (6003)	120 (4238)	50 (1766)	Down 6.25 m^3/s (221 ft^3/s) every 15 minutes from 07:00 to 08:45.	Kokanee protection flows.
27 September 2013	RE 2013-05	120 (4238)	70 (2472)	50 (1766)	Down 6.25 m^3/(221 ft^3/s) every 15 minutes from 07:00 to 08:45.	Final transition to Kokanee protection flows.
21 January 2014	RE 2014-01	220 (7769)	142 (5015)	78 (2755)	Down 6 m^3/s 6.25 m^3/s (212 ft^3/s) every 15 minutes from 06:00 to 08:45.	Discharge reduced to meet reservoir targets.
1 March 2014	RE 2014-02	142 (5015)	85 (3002)	57 (2013)	Down 6 m^3/s 6.25 m^3/s (212 ft^3/s) every 15 minutes from 06:00 to 08:00.	Discharge reduced to meet flow target at DRL.

[a] The flow decreases reflect the net total decrease in flows over specific intervals at DDM. Actual ramping rates (rate of stage or discharge decrease per unit time) at particular stranding sites may be significantly higher over a shorter time interval or possibly attenuated to a lower rate at the downstream locations where stranding was observed.

Table A13: Chronology of sampling activities for the 2014 - 2015 Lower Duncan River Fish Stranding Impact Monitoring, Year 7 Program

Date(s)	Sampling Activities	Reduction Event Number	Number of Snorkel Sites Surveyed	Number of Index Sites Stranding Assessed	Number of Non-Index Stranding Sites Assessed
22 May 2014	Stranding Assessments	RE2014-03	-	5	4
18 and 19 September 2014	Abundance Estimation	-	Study Area Reconnaissance and Site Selection		
20 September 2014	Abundance Estimation	-	14	-	-
21 September 2014	Abundance Estimation	-	16	-	-
22 September 2014	Abundance Estimation	-	10	-	-
23 September 2014	Abundance Estimation	-	9	-	-
25 September 2014	Stranding Assessments	RE2014-04	-	3	5
28 September 2014	Stranding Assessments	RE2014-05	-	4	3
01 October 2014	Stranding Assessments	RE2014-06	-	3	0
01 March 2015	Stranding Assessments	RE2015-01	-	3	4
02 March 2015	Stranding Assessments	RE2015-02	-	2	3

Table A14: Summary of DDM flow reduction events in Study Year 7, from 22 May 2014 to 2 March 2015, for events when fish stranding assessments were conducted.

Date	Reduction Event	DDM Discharge (m^3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
22 May 2014	RE2014-03	60 (2119)	3 (106)	57 (2013)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 06:00 to 07:45.	Discharge reduced to meet flow target at DRL.
25 September 2014	RE2014-04	208 (7345)	142 (5015)	66 (2331)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 06:00 to 08:00, down 3 m^3/s (106 ft^3/s) at 08:15.	Onset of Kokanee protection flows.
28 September 2014	RE2014-05	142 (5015)	80 (2825)	62 (2190)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 06:00 to 07:45.	Kokanee protection flows.
01 October 2014	RE2014-06	92 (3249)	46 (1624)	46 (1624)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 06:00 to 07:15, down 4 m^3/s (141 ft^3/s) at 07:30.	Final transition to Kokanee protection flows.
1 March 2015	RE2015-01	224 (7910)	154 (5438)	70 (2472)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 06:00 to 08:15.	Discharge reduced to meet flow target at DRL.
2 March 2015	RE2015-02	154 (5438)	77 (2719)	77 (2719)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 06:00 to 08:30.	Discharge reduced to meet flow target at DRL.

[a] The flow decreases reflect the net total decrease in flows over specific intervals at DDM. Actual ramping rates (rate of stage or discharge decrease per unit time) at particular stranding sites may be significantly higher over a shorter time interval or possibly attenuated to a lower rate at the downstream locations where stranding was observed.

Table A15: Chronology of sampling activities for the 2015 - 2016 Lower Duncan River Fish Stranding Impact Monitoring, Year 8 Program

Date(s)	Sampling Activities	Reduction Event Number	Number of Snorkel Sites Surveyed	Number of Index Sites Stranding Assessed	Number of Non-Index Stranding Sites Assessed
21 and 22 September 2015	Abundance Estimation	-	Study Area Reconnaissance and Site Selection		
23 September 2015	Abundance Estimation	-	12	-	-
24 September 2015	Abundance Estimation	-	12	-	-
25 September 2015	Abundance Estimation	-	13	-	-
26 September 2015	Abundance Estimation	-	9	-	-
28 September 2015	Stranding Assessments	RE2015-03	-	2	4
01 October 2015	Stranding Assessments	RE2015-04	-	2	5
22 December 2015	Stranding Assessments	RE2015-05	-	4	3
29 December 2015	Stranding Assessments	RE2015-06	-	3	5
09 April 2016	Stranding Assessments	RE2016-01	-	3	2

Table A16: Summary of DDM flow reduction events in Study Year 8, from 28 September 2015 to 9 April 2016, for those events when fish stranding assessments were conducted.

Date	Reduction Event	DDM Discharge (m^3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
28 September 2015	RE2015-03	133 (4697)	78 (2755)	55 (1942)	Down 5 m^3/s (177 ft^3/s) every 15 minutes from 08:00 to 10:30.	Onset of Kokanee protection flows.
1 October 2015	RE2015-04	78 (2755)	33 (1165)	45 (1589)	Down 5 m^3/s (177 ft^3/s) every 15 minutes from 06:00 to 08:00.	Kokanee protection flows.
22 December 2015	RE2015-05	202 (7134)	140 (4944)	62 (2190)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 06:00 to 07:45, down 6 m^3/s (221 ft^3/s) at 08:00.	Discharge reduced to meet flow target at DRL
29 December 2015	RE2015-06	140 (4944)	80 (2825)	60 (2119)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 06:00 to 07:45, down 4 m^3/s (141 ft^3/s) at 08:00.	Discharge reduced to meet flow target at DRL.
9 April 2016	RE2016-01	128 (4520)	70 (2472)	58 (2048)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 06:00 to 07:30, down 9 m^3/s (318 ft^3/s) at 07:45.	Discharge reduced to meet flow target at DRL.

[a] The flow decreases reflect the net total decrease in flows over specific intervals at DDM. Actual ramping rates (rate of stage or discharge decrease per unit time) at particular stranding sites may be significantly higher over a shorter time interval or possibly attenuated to a lower rate at the downstream locations where stranding was observed.

Table A17: Chronology of sampling activities for the 2016 - 2017 Lower Duncan River Fish Stranding Impact Monitoring, Year 9 Program

Date(s)	Sampling Activities	Reduction Event Number	Number of Snorkel Sites	Number of Index Sites Stranding	Number of Non-Index Stranding
19 May 2016	Stranding Assessments	RE2016-02	-	1	5
20 May 2016	Stranding Assessments	RE2016-03	-	5	2
18 September 2016	Abundance Estimation	-	Study Area Reconnaissance and Site Selection		
19 September 2016	Abundance Estimation	-	10	-	-
20 September 2016	Abundance Estimation	-	15	-	-
21 September 2016	Abundance Estimation	-	13	-	-
22 September 2016	Abundance Estimation	-	10	-	-
23 September 2016	Abundance Estimation	-	9	-	-
24 September 2016	Stranding Assessments	RE2016-04	-	2	4
25 September 2016	Stranding Assessments	RE2016-05	-	2	4
01 March 2017	Stranding Assessments	RE2017-01	-	4	1
02 March 2017	Stranding Assessments	RE2017-02	-	1	4

Table A18: Summary of DDM flow reduction events in Study Year 9, from 19 May 2016 to 02 March 2017, for events when fish stranding assessments were conducted.

Date	Reduction Event	DDM Discharge m^3/s (ft^3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
19 May 2016	RE2016-02	142 (5015)	72 (2543)	70 (2472)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 08:00 to 10:15.	Discharge reduced to meet flow target at DRL.
20 May 2016	RE2015-03	72 (2543)	3 (106)	69 (2437)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 06:00 to 08:00, down 6 m^3/s (212 ft^3/s) at 08:15.	Discharge reduced to meet flow target at DRL.
24 September 2016	RE2015-04	212 (7487)	144 (5085)	68 (2401)	Down 6 m^3/s (212 ft^3/s) every 15 minutes from 06:00 to 08:30, down 2 m^3/s (71 ft^3/s) at 08:45.	Onset of Kokanee protection flows.
25 September 2016	RE2015-05	144 (5085)	75 (2649)	69 (2437)	Down 6 m^3/s (212 ft^3/s) every 15 minutes from 06:00 to 08:30, down 3 m^3/s (106 ft^3/s) at 08:45.	Kokanee protection flows.
01 March 2017	RE2017-01	128 (4520)	128 (4520)	54 (1907)	Down 6 m^3/s (212 ft^3/s) every 15 minutes from 06:00 to 08:00.	Discharge reduced to meet flow target at DRL.
02 March 2017	RE2017-02	128 (4520)	80 (2825)	48 (1695)	Down 6 m^3/s (212 ft^3/s) every 15 minutes from 06:00 to 07:45.	Discharge reduced to meet flow target at DRL.

[a] The flow decreases reflect the net total decrease in flows over specific intervals at DDM. Actual ramping rates (rate of stage or discharge decrease per unit time) at particular stranding sites may be significantly higher over a shorter time interval or possibly attenuated to a lower rate at the downstream locations where stranding was observed.

Table A19: Chronology of sampling activities for the 2017 - 2018 Lower Duncan River Fish Stranding Impact Monitoring, Year 10 Program.

Date(s)	Sampling Activities	Reduction Event Number	Number of Stranding Sites Assessed
23 May 2017	Stranding Assessments	RE2017-03	7
24 May 2017	Stranding Assessments	RE2017-04	3
30 August 2017	Stranding Assessments	RE2017-05	7
24 September 2017	Stranding Assessments	RE2017-06	6
25 September 2017	Stranding Assessments	RE2017-07	4
01 March 2018	Stranding Assessments	RE2018-01	7
22 March 2018	Stranding Assessments	RE2018-02	5
27 March 2018	Stranding Assessments	RE2018-03	10

Table A20: Summary of DDM flow reduction events in Study Year 10, from 23 May 2017 to 26 September 2018, for events when fish stranding assessments were conducted.

Date	Reduction Event	DDM Discharge m^3/s (ft^3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
23 May 2017	RE2017-03	202 (7134)	104 (3673)	98 (3461)	Down 7 m^3/s (247 ft^3/s) in 15 minute intervals	Discharge reduced to meet flow target at DRL.
24 May 2017	RE2017-04	104 (3673)	11 (388)	93 (3284)	Down 7 m^3/s (247 ft^3/s) in 15 minute intervals, down 2 m^3/s (71 ft^3/s) for last reduction	Discharge reduced to meet flow target at DRL.
30 August 2017	RE2017-06	149 (5262)	100 (3531)	49 (1730)	Down 6 m^3/s (212 ft^3/s) in 15 minute intervals	Discharge reduced to meet flow target at DRL.
24 September 2017	RE2017-07	195 (6886)	130 (4591)	65 (2295)	Down 6 m^3/s (212 ft^3/s) in 15 minute intervals, down 5 m^3/s (177 ft^3/s) for last reduction	Onset of Kokanee protection flows.
25 September 2017	RE2017-08	130 (4591)	70 (2472)	60 (2119)	Down 6 m^3/s (212 ft^3/s) in 15 minute intervals	Kokanee protection flows.
01 March 2018	RE2018-01	133 (4697)	91 (3214)	42 (1483)	Down 6 m^3/s (212 ft^3/s) in 15 minute intervals	Discharge reduced to meet flow target at DRL.
22 March 2018	RE2018-02	81 (2860)	65 (2295)	16 (565)	Down 4 m^3/s (141 ft^3/s) in 15 minute intervals	Discharge reduced to meet flow target at DRL.
27 March 2018	RE2018-03	29 (1024)	3 (106)	26 (918)	Down 7 m^3/s (247 ft^3/s) in 15 minute intervals, down 3 m^3/s (106 ft^3/s) for last reduction	Discharge reduced to meet flow target at DRL.

[a] The flow decreases reflect the net total decrease in flows over specific intervals at DDM. Actual ramping rate (rate of stage or discharge decrease per unit time) at each of the stranding sites may be significantly higher over a shorter time interval or possibly attenuated to a lower rate at the downstream locations where stranding was observed.

Table A21: Chronology of sampling activities for the 2018 - 2019 Lower Duncan River Fish Stranding Impact Monitoring, Year 11 Program.

Date(s)	Sampling Activities	Reduction Event Number	Number of Stranding Sites Assessed
25 September 2018	Stranding Assessments	RE2018-04	7
26 September 2018	Stranding Assessments	RE2018-05	4
01 March 2019	Stranding Assessments	RE 2019-01	3

Table A22: Summary of DDM flow reduction events in Study Year 11, from April 2018 to March 2019, for events when fish stranding assessments were conducted.

Date	Reduction Event	DDM Discharge m^3/s (ft^3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
25 September 2018	RE2018-04	192 (6780)	107 (3779)	85 (3001)	Down 7 m^3/s (247 ft^3/s) in 15 minute intervals	Onset of Kokanee protection flows
26 September 2018	RE2018-05	107 (3779)	28 (989)	79 (2790)	Down 6.5 m^3/s (230 ft^3/s) in 15 minute intervals	Kokanee protection flows
01 March 2019	RE2019-01	164 (5792)	80 (2825)	84 (2966)	Down 6.0 m^3/s (212 ft^3/s) in 15 minute intervals	Discharge reduced to meet flow target at DRL

[a] The flow decreases reflect the net total decrease in flows over specific intervals at DDM. Actual ramping rate (rate of stage or discharge decrease per unit time) at each of the stranding sites may be significantly higher over a shorter time interval or possibly attenuated to a lower rate at the downstream locations where stranding was observed.

Table A23: Chronology of sampling activities for the 2019 - 2020 Lower Duncan River Fish Stranding Impact Monitoring, Year 12 Program.

Date(s)	Sampling Activities	Reduction Event Number	Number of Stranding Sites Assessed
24 September 2019	Stranding Assessments	RE2019-02	6
27 September 2019	Stranding Assessments	RE2019-03	4
16 to 18 October 2019	UAV Photogrammetry	-	4
31 January 2020	Stranding Assessments	RE2020-02	7
29 February 2020	Stranding Assessments	RE2020-03	7
11 April 2020	Stranding Assessments	RE2020-04	9

Table A24: Summary of DDM flow reduction events in Study Year 12, from April 2019 to March 2020, for events when fish stranding assessments were conducted.

Date	Reduction Event	DDM Discharge m^3/s (ft^3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
24 September 2019	RE2019-02	195 (6886)	135 (4768)	60 (2119)	Down 6 m^3/s (212 ft^3/s) in 15 minute intervals	Onset of Kokanee protection flows
27 September 2019	RE2019-03	125 (4414)	17 (600)	108 (3814)	Down 6.0 m^3/s (212 ft^3/s) in 15 minute intervals	Kokanee protection flows
31 January 2020	RE2020-02	215 (7593)	167 (5898)	48 (1695)	Down 12.0 m^3/s (424 ft^3/s) in 30 minute intervals	Discharge reduced to meet flow target at DRL
29 February 2020	RE2020-03	145 (5121)	85 (3002)	60 (2119)	Down 12.0 m^3/s (424 ft^3/s) in 30 minute intervals	Discharge reduced to meet flow target at DRL
11 April 2020	RE2020-04	76 (2684)	56 (1978)	20 (706)	Down 5.0 m^3/s (177 ft^3/s) in 30 minute intervals	Discharge reduced to meet flow target at DRL

[a] The flow decreases reflect the net total decrease in flows over specific intervals at DDM. Actual ramping rate (rate of stage or discharge decrease per unit time) at each of the stranding sites may be significantly higher over a shorter time interval or possibly attenuated to a lower rate at the downstream locations where stranding was observed.

Table A25: Chronology of sampling activities for the 2020 Lower Duncan River Fish Stranding Impact Monitoring, Year 13 Program.

Date(s)	Sampling Activities	Reduction Event Number	Number of Stranding Sites Assessed
21 May 2020	Stranding Assessments	RE2020-05	8
22 May 2020	Stranding Assessments	RE2020-06	5
13 August 2020	Stranding Assessments	RE2020-07	7
24 September 2020	Stranding Assessments	RE2020-08	8
27 September 2020	Stranding Assessments	RE2020-09	4
30 September 2020	Stranding Assessments	RE2020-10	5

Table A26: Summary of DDM flow reduction events in study Year 13, from April 2020 to December 2020, for events when fish stranding assessments were conducted.

Date	Reduction Event (RE)	DDM Discharge m³/s (ft³/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
21 May 2020	RE2020-05	92 (3249)	36 (1271)	56 (1978)	Down 7.0 m^3/s (247 ft^3/s) in 30 minute intervals	Discharge reduced to meet flow target at DRL
22 May 2020	RE2020-06	36 (1271)	3 (106)	33 (1165)	Down 5.0 m^3/s (177 ft^3/s) in 30 minute intervals	Discharge reduced to meet flow target at DRL
13 August 2020	RE2020-07	156 (5509)	114 (4026)	42 (1483)	Down 6.0 m^3/s (212 ft^3/s) in 15 minute intervals	Maintenance on Low Level Output Gates
24 September 2020	RE2020-08	194 (6851)	134 (4732)	60 (2119)	Down 6.0 m^3/s (212 ft^3/s) in 15 minute intervals	Onset of Kokanee protection flows
27 September 2020	RE2020-09	134 (4732)	74 (2613)	60 (2119)	Down 6.0 m^3/s (212 ft^3/s) in 15 minute intervals	Continuation to Kokanee protection flows
30 September 2020	RE2020-10	74 (2613)	20 (706)	54 (1907)	Down 6.0 m^3/s (212 ft^3/s) in 15 minute intervals	Kokanee protection flows

[a] The flow decreases reflect the net total decrease in flows over specific intervals at DDM. Actual ramping rate (rate of stage or discharge decrease per unit time) at each of the stranding sites may be significantly higher over a shorter time interval or possibly attenuated to a lower rate at the downstream locations where stranding was observed.

https://golderassociates.sharepoint.com/sites/31732g/deliverables/working documents/year 13 report/draft report/appendices/app a - maps and sampling chronology/table a1 to a26.docx

APPENDIX B

Modelling Specifications and Code (Thorley 2021)

Model Templates

Pool Stranding

```
.model {
 bAbundance ~ dnorm(0, 2^-2)
 bEfficiency ~ dnorm(0, 2^-2)

 bSeasonAbundance[1] <- 0
 for(i in 2:nSeason){
   bSeasonAbundance[i] ~ dnorm(0, 2^-2)
 }

 sDispersion ~ dnorm(0, 2^-2) T(0,)
 for(i in 1:nObs){
   log(eAbundance[i]) <- bAbundance + bSeasonAbundance[Season[i]]
   eDispersion[i] ~ dgamma(sDispersion^-2, sDispersion^-2)
   bAbundancePool[i] ~ dpois(eAbundance[i] * eDispersion[i])

   eAbundancePass[i, 1] <- bAbundancePool[i]

   logit(eEfficiency[i]) <- bEfficiency
   for(pass in 1:nPass) {
     Pass[i, pass] ~ dbin(eEfficiency[i], eAbundancePass[i, pass])
     eAbundancePass[i, pass+1] <- eAbundancePass[i, pass] - Pass[i, pass]
   }
  }
```

Total Stranding

```
.model{
 b0 ~ dnorm(-4, 2^-2)
 bSeason[1] <- 0
 for(i in 2:nSeason) {
   bSeason[i] ~ dnorm(0, 2^-2)
 }
```

```
 sReduction ~ dnorm(0, 2^-2) T(0,)
 for(i in 1:nReduction) {
   bReduction[i] ~ dnorm(-sReduction^2 / 2, sReduction^-2)
 }
 sSite ~ dnorm(0, 2^-2) T(0,)
 for(i in 1:nSite) {
   bSite[i] ~ dnorm(-sSite^2 / 2, sSite^-2)
 }
 sDispersion ~ dnorm(0, 2^-2) T(0,)
 for(i in 1:nObs) {
   log(eStranded[i]) <- b0 + bSeason[Season[i]] + bReduction[Reduction[i]] + bSite[Site[i]] + log(SiteArea[i])
   eDispersion[i] ~ dgamma(sDispersion^-2, sDispersion^-2)
   Stranded[i] ~ dpois(eStranded[i] * eDispersion[i])
 }
```

Proportion Stranding

```
model{

 b0 ~ dnorm(-5, 2^-2)

 bRate ~ dnorm(0, 2^-2)

 bDischarge ~ dnorm(0, 2^-2)

 bDrop ~ dnorm(0, 2^-2)

 bDayte ~ dnorm(0, 2^-2)

 bWetted ~ dnorm(0, 2^-2)



 sProportion ~ dnorm(0, 2^-2) T(0,)

 for(i in 1:nObs) {

   logit(eProportion[i]) <- b0 + bRate * Rate[i] + bDischarge * Discharge[i] + bDrop * Drop[i] + bDayte * Dayte[i] + bWetted *
Wetted[i]

   LogitProportion[i] ~ dnorm(logit(eProportion[i]), sProportion^-2)
```

RESULTS

Tables

Interstitial Stranding

Table 1. The number and density (ind/ha) of interstitial fish salvaged by area (ha), species and season.

Species	Season	Area	Number	Density
RB	Fall	1.473395	23	15.610206
RB	Spring	1.105990	2	1.808335
MW	Fall	1.473395	0	0.000000
MW	Spring	1.105990	2	1.808335

Pool Stranding

Table 2. Parameter descriptions.

Parameter	Description
bAbundance	Intercept for log(eAbundance)
bEfficiency	Intercept for logit(eEfficiency)
bSeasonAbundance[i]	Effect of i^{th} Season on bAbundance
eAbundance[i]	Expected Abundance of fish at i^{th} pool
eAbundance[i]	Expected abundance of fish at the i^{th} pool prior to the first pass (without overdispersion)
eAbundancePass[i,j]	The expected abundance of fish at the i^{th} pool prior to the j^{th} pass
eEfficiency[i]	The expected capture efficiency at the i^{th} pool on the j^{th} pass
PoolArea[i]	The area of the i^{th} pool (m2)
sDispersion	SD of overdispersion

Rainbow Trout

Table 3. Model coefficients.

term	estimate	lower	upper	svalue
bAbundance	1.6479146	1.3462221	1.9902881	10.551708
bEfficiency	-0.3307284	-0.7475961	0.0518345	3.518285
bSeasonAbundance[2]	-1.9184369	-2.2469430	-1.6055437	10.551708
sDispersion	3.6850186	3.4547588	3.9475334	10.551708

Table 4. Model convergence.

n	K	nchains	niters	nthin	ess	rhat	converged
2288	4	3	500	100	326	1.007	TRUE

Table 5. Model sensitivity.

n	K	nchains	niters	rhat_1	rhat_2	rhat_all	converged
2288	4	3	500	1.007	1.023	1.023	TRUE

Total Stranding

Table 6. Parameter descriptions.

Parameter	Description
b0	Intercept for log(eStranded)
bReduction[i]	Effect of i^{th} Reduction on b0
bSeason[i]	Effect of i^{th} Season on b0
bSite[i]	Effect of i^{th} Site on b0
eStranded[i]	Expected number of fish stranding at site
sDispersion	Overdispersion term
sReduction	SD of bReduction
sSite	SD of bSite

Parameter	Description
Stranded[i]	Number of fish stranding at site

Rainbow Trout

Table 7. Model coefficients.

term	estimate	lower	upper	svalue
b0	-4.4879688	-4.9678245	-3.870525	10.55171
bSeason[2]	-1.9089393	-2.4078217	-1.406939	10.55171
sDispersion	0.9530048	0.8706192	1.046629	10.55171
sReduction	0.8821631	0.6867074	1.132523	10.55171
sSite	0.7470441	0.4951118	1.125205	10.55171

Table 8. Model convergence.

n	K	nchains	niters	nthin	ess	rhat	converged
411	5	3	500	500	504	1.004	TRUE

Table 9. Model sensitivity.

n	K	nchains	niters	rhat_1	rhat_2	rhat_all	converged
411	5	3	500	1.004	1.006	1.003	TRUE

Proportion Stranding

Table 10. Parameter descriptions.

Parameter	Description
b0	Intercept for logit(eProportion)
bDayte	Effect of Dayte on b0
bDischarge	Effect of Discharge on b0
bDrop	Effect of Drop on b0
bRate	Effect of Rate on b0

Parameter	Description
bWetted	Effect of Wetted on b0
Dayte[i]	Standardized day of the year since June 15th
Discharge[i]	Standardized initial discharge
Drop[i]	Standardized discharge drop
eProportion[i]	Expected proportion of population stranding during reduction
LogitProportion[i]	Log-odds expected proportion of population stranding during reduction
Rate[i]	Standardized ramping rate
sProportion	SD of residual variation in LogitProportion
Wetted[i]	Standardized log wetted history

Rainbow Trout

Table 11. Model coefficients.

term	estimate	lower	upper	svalue
b0	-5.2631048	-5.4607255	-5.0798108	10.5517083
bDayte	-0.4880427	-0.6959930	-0.2720032	10.5517083
bDischarge	-0.6893847	-0.9461802	-0.4182357	10.5517083
bDrop	0.3500488	0.0817496	0.6471318	7.3817833
bRate	0.3915074	0.1250384	0.6289175	7.0922766
bWetted	-0.0258325	-0.2656204	0.2174611	0.2513557
sProportion	0.7533055	0.6267453	0.9228583	10.5517083

Table 12. Model convergence.

n	K	nchains	niters	nthin	ess	rhat	converged
59	7	3	500	10	1287	1.004	TRUE

Table 13. Model sensitivity.

n	K	nchains	niters	rhat_1	rhat_2	rhat_all	converged
59	7	3	500	1.004	1.002	1.002	TRUE

APPENDIX C

Photographic Plates

Plate 1 Electrofishing isolated pool at site LARD0.3R, 25 September 2010.

Plate 2 Visually inspecting isolated pool at site M0.8R, 2 March 2011.

Plate 3 Large woody debris cover in isolated pool, 1 October 2010.

Plate 4 Small woody debris cover in isolated pool, 25 September 2010.

Plate 5 Interstitial survey at M0.8R, 2 March 2011.

Plate 6 Juvenile Mountain Whitefish on measuring board, site S4.0-4.2R, 12 April 2011.

Plate 7 Zero to Low complexity pool at site M08.R, 1 October 2011.

Plate 8 Moderate to High complexity pool at site M0.8R, 1 October 2011.

Plate 9 Overhanging vegetation cover at site SLARD0.3R, 25 August 2011.

Plate 10 Submerged terrestrial vegetation cover in isolated pool at site S4.1R, 25 August 2011.

Plate 11 Zero to Low complexity pool at site S3.5-4.0R, 28 September 2014. Note: red circle identifies school of stranded Rainbow Trout and Sculin Species.

Plate 12 Medium to High complexity pool at site S3.5-4.0R, 28 September 2014.

Plate 13 Assessing cover types in a pool located at site S10.6R, 28 September 2014.

Plate 14 Garbage encountered at site S9.2L, 22 May 2014.

Plate 15 Mainstem habitat at site M7.7L, 28 September 2014.

Plate 16 Side channel habitat at site S4.0-4.2R, 22 May 2014.

Plate 17 **Beaver Dam at Site S9.0L, 1 March 2018.**

Plate 18 **Downstream view of Site S3.5-4.0R, 24 May 2018.**

Plate 19 – Juvenile Rainbow Trout salvaged from isolated pool, 29 February 2020.

Plate 20 – Bull Trout Salvaged from pool, 29 February 2020.