Duncan Dam Project Water Use Plan

Adaptive Stranding Protocol Development Program

Implementation Year 11

Reference: DDMMON-16

Lower Duncan River: Fish Stranding Impact Monitoring: Year 11

Study Period: April 2018 to September 2019

Golder Associates Ltd.

Castlegar, BC

December, 6, 2019

REPORT

DDMMON-16: Lower Duncan River

Lower Duncan River Fish Stranding Impact Monitoring: Year 11 Report (April 2018 to April 2019)

Submitted to:

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Cover Photo: Upstream view of Site M0.8R, 01 March 2018.

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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 current survey, initiated under BC Hydro Water License Requirements (WLR), includes the continuation of the Lower Duncan River Fish Stranding Impact Monitoring Program (DDMMON-16).

The results from this monitoring program will help inform flow management decisions that may impact fish stranding in the LDR. Based on the current state of knowledge, 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). Based on collected data and the life history of species present in the LDR, DDM operations can increase the risk of stranding in certain seasons (Spring and Fall) and during periods of longer wetted histories. Based on data collected up to April 2019, documented stranding rates of juvenile Mountain Whitefish (*Prosopium williamsoni*) are very low and are not believed to result in population level effects. The total stranding rates for juvenile Rainbow Trout (*Oncorhynchus mykiss*) are estimated to be under 3% in most years, but possibly as high as 12%.

This report presents the results from Years 1 to 11 of the DDMMON-16 program, and the current status of management questions for DDMMON-16 is provided in Table EI.

Table EI: DDMMON-16 Year 11: Status of Management Questions and Objectives.

DDMMON-16 Management Question	DDMMON-16 Specific Hypothesis	DDMMON-16 Year 11 (April 2018 – December 2018) Status Summary
1) How effective are the operating measures implemented as part of the Adaptive Stranding Protocol Development (ASPD) program?	N/A	- <u>Based on the current state of knowledge, the flow reduction measures implemented under the WUP are effective at reducing fish stranding.</u> - When feasible, flow reductions at DDM should follow recommendations made by the Lower Duncan River Stranding Protocol Development and Finalization Program (DDMMON-15). - The relationship between wetted history and fish stranding is a currently outstanding issue in the Adaptive Stranding Protocol Development Program (ASPD).

DDMMON-16 Management Question	**DDMMON-16 Specific Hypothesis**	**DDMMON-16 Year 11 (April 2018 – December 2018) Status Summary**
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 the habitat. - Index sites tend to be of lower gradient and wider than the non-index sites, therefore more area dewaters at these sites. - In the current year (2018-2019), a significant site effect on the formation of pools (density) and pool stranding rates was not found. - The low number of interstitial stranding datapoints precluded the examination of the effect of site on interstitial stranding. - The stranding rates at both index and random sites should continue to be analyzed as separate strata as the dataset increases in size to allow for continued comparison with historical data. - <u>Based on the current state of knowledge, Hypothesis H_{01} cannot be rejected at this time</u>.
	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. - A seasonal effect on Rainbow Trout stranding rates was identified, with stranding rates approximately seven times higher in the fall in comparison to the winter season. Whether or not this relationship was due to lower densities in the system in the spring versus the fall or to a decreased risk of stranding could not be determined. - Mountain Whitefish encounters have been 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. - Within the current dataset, relationships between the number of pool stranded fish and slope of substrate were not found. - A relationship between slope and the number of interstitially stranded fish was found, although it was not statistically significant. - <u>Based on the current dataset, study hypothesis H_{02} is not rejected for Rainbow Trout or Mountain Whitefish.</u>

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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).

One of the main objectives of the Duncan Dam Water License Requirements (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. The DDM water license requires a minimum average daily flow from DDM of 3 m^3/s (160 ft^3/s) 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 (2578 ft^3/s) be maintained in the LDR at the Lardeau River Water Survey of Canada (WSC) gauging station (DRL). In addition, the maximum hourly flow reduction allowed under the WUP is 28 m^3/s (989 ft^3/s), and the maximum daily flow change allowed is 113 m^3/s (3991 ft^3/s). All LDR water license discharge requirements are subject to available inflows into Duncan Reservoir and are dependent on tributary inflows.

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 will be implemented over the WUP review period based on the results from a collective group of monitoring studies. One component of the broader program is the Lower Duncan River Fish Stranding Impact Monitoring Program (DDMMON-16). In conjunction with other assessment tools being developed during the monitoring period, DDMMON-16 assesses Rainbow Trout (*Oncorhynchus mykiss*) and Mountain Whitefish (*Prosopium williamsoni*) population level impacts associated with dam operations during the review period. 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 fish stranding impact monitoring program conducted in Year 11 (April 2018 – December 2018) builds on the historic methodology, expands the program's datasets, updates the boundaries of identified sites where stranding occurs, and analyzes 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 Report Scope

The state of knowledge regarding the environmental and operational variables of interest that impact fish stranding was reviewed in detail in the Gap Analysis for Lower Duncan River Ramping Program (DDMMON-1; Irvine and Schmidt 2009; Golder 2009a). The multiplication of probability of fish stranding by fish density predicts the number of fish stranded. If a fish becomes stranded, it can either survive or it can succumb; in the latter instance, the fish becomes a stranding mortality component of the total mortality rate associated with the population. Total mortality is the sum of interstitial and pool stranding mortality. The level of mortality associated with the population, as well as the recruitment rate and the level of immigration or emigration all combine to determine population size. Whether stranding mortality has a population level effect (since compensatory mechanisms such as increased growth or survival may be a result of the fish lost through stranding mortality) has yet to be determined. This determination would require knowledge about the density dependent mechanisms acting on a specific population and, as pointed out in Higgins and Bradford (1996), this is difficult to ascertain with enough certainty to allow population projections.

Previous research in the field of fish responses to hydro-peaking have demonstrated that there is substantial variability in the responses and that it is difficult to attribute the variability to single or even multiple factors (e.g., Berland et al. 2004; Saltveit et al. 2001; Irvine and Schmidt 2009). This uncertainty should be considered when interpreting the results of this program.

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 Year 11 of DDMMON-16 (15 April 2018 to 14 April 2019). 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).

1.3 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 and outside of 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.

Years 1 (2008–2009) and 2 (2009–2010) of DDMMON-16 worked toward addressing Objective 1) the effectiveness of operating measures, and addressing Hypothesis Ho_1, fish stranding at index sites is representative of overall stranding (Golder 2009b, 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) and the present study year (Year 11: April 2018 to April 2019) of DDMMON-16. Recommendations to refine study methodology and to better address both objectives and hypotheses in future years of DDMMON-16 have been developed (Section 6.0).

1.4 Study Design and Rationale

Golder conducted fish stranding assessments on the LDR between 2002 and 2018. 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 were made in Years 3 to 10 (2010–2018) and implemented in the present study year. These recommendations included changes to the study design to address gaps in the dataset identified during data analysis (Golder 2011, Golder and Poisson 2012, Golder 2014–2015, Golder 2017a, 2017b, Golder 2018, Golder and Poisson 2019 in prep.).

As part of the DDMMON-15 program, a workshop was held on 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 at the workshop was shifting the DDMMON-16 program from its current goal of examining the impact of fish stranding on target fish species populations to a program focused on long term monitoring and salvage operations. This shift led to substantial changes to DDMMON-16's study design between Years 8 to 10.

1.4.1 Stranding Site Selection

Prior to Year 4, 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 non-index sites were conducted and in-depth statistical analysis of stranding rates at both index and non-index sites was not possible. In turn, estimates of stranding rates may have been upwardly biased. To allow for comparisons of stranding rates between index and non-index sites, effort expended for non-index sites from Year 4 onward was increased.

As discussed in the 2016 DDMMON-15 workshop, in order to move towards a long-term monitoring program, changes were made to the site selection process. With the analysis of the Year 7 dataset, Ho_1 (*Fish stranding observed at index sites along the lower Duncan River floodplain is representative of overall stranding*) was not rejected. Therefore, for the current study year, the dichotomous classification of sites into index and non-index 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. Further information on site selection details is provided in Section 2.6.1.

1.4.2 Pool Sampling

As pool sampling was the primary focus of previous study years, relatively precise pool stranding estimates for Rainbow Trout were obtained in Years 3 and 4 (Golder 2011, Golder and Poisson 2012). Therefore, sampling effort was refocused in Year 4 to assess interstitial stranding in more detail.

After the Year 4 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 will allow for more accurate estimates of fish stranding in the LDR in the future.

1.4.3 Interstitial Sampling

During Year 3, 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 since Year 4 (2011–2012) was increased.

To further reduce uncertainty related to interstitial stranding estimates, transect sampling was implemented in Year 7. 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. Although transect sampling did increase the amount of area surveyed, encounters of interstitially stranded fish remained very low.

During the current study year, updated methodologies were implemented to further increase the area of dewatered habitat sampled, as well as attempt to increase the encounters of interstitially stranded fish (see Section 2.6.2.3).

1.4.4 Substrate Mapping

The Year 11 Study plan included a substrate mapping component that involved high definition aerial photographs of the LDR that would be collected by a drone when flows at the DRL were at the target minimum of 73 m^3/s. The aerial imagery will have sufficient resolution to geospatially document substrate size within each identified stranding site. This study component was scheduled for mid to late October 2018 after the Kokanee Protection Flow Target had been reached; however, flows in the LDR increased before the survey could be conducted. This component of the study will be conducted after the Kokanee Protection Flow Target has been reached in Year 12.

1.4.5 Abundance Estimates

To obtain abundance estimations for Rainbow Trout that could be compared to total stranding estimates, spring age-1 Rainbow Trout abundance estimates from the Gerrard Rainbow Trout Stock Productivity study (Andrusak and Thorley 2019) were used.

1.4.6 Lower Duncan River Fish Stranding Database

To meet the goals of the DDMMON-15 workshop, 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 will allow for more informed fish salvages during future years of this program.

1.4.7 Data Analysis

The modelling used in Year 10 (Golder and Poisson 2019 in prep.) of this program was updated to incorporate the current year's dataset and to further refine the slope classification when analyzing as a variable related to stranding rates. Updated observer efficiencies (Andrusak and Thorley 2018) were also used in the analysis.

2.0 METHODS

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 sections:

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 this study, 50 potential fish stranding sites were identified based on previous studies (AMEC 2004; Golder 2006, 2008, 2009b, 2010, 2011, 2014; Golder and Poisson 2012). These stranding sites included 11 index stranding assessment sites and 39 non-index stranding assessment sites (Appendix A, Figures 1 to 7). Habitats situated outside of the identified sites typically had steep banks with fine substrates. Habitats with these characteristics have very low stranding risk. Consequently, additional major fish stranding at locations outside of the 50 potential fish stranding sites used in this study, is unlikely to occur.

2.2 Study Period

Stranding assessment activities in Year 11 were conducted on September 25 and 26, 2018, as well as on 1 March 2019 during planned flow reductions at DDM. Each assessed reduction from DDM was assigned a reduction event number (RE; see Section 2.6) and Table 1 outlines all assessment activities during Year 11. An in-depth summary of the chronology of sampling and project milestones in all study years is provided in Appendix A, Tables A1 to A10.

Table 1: Sampling activities for the April 2018 to March 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

Figure 1: Lower Duncan River Fish Stranding Impact Monitoring Program: Overview of Study Area.

2.3 Physical Parameters

2.3.1 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)

2.3.2 River Discharge

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

2.4 Bayesian Analysis

Model parameters were estimated using Bayesian methods. The Bayesian estimates were produced using JAGS (Plummer 2015). For additional information on Bayesian modelling in the BUGS language, of which JAGS uses a dialect, the reader is referred to Kery and Schaub (2011).

Unless indicated otherwise, the Bayesian analyses used uninformative normal or half-normal prior distributions (Kery and Schaub 2011, 36). The posterior distributions were estimated from 1500 Markov Chain Monte Carlo (MCMC) samples thinned from the second halves of 3 chains (Kery and Schaub 2011, 38–40).
Model convergence was confirmed by ensuring that $\hat{R} \leq 1.1$ (Kery and Schaub 2011, 40) and ESS ≥ 150 for each of the monitored parameters (Kery and Schaub 2011, 61), where $\hat{R}$ is the potential scale reduction factor and ESS is the effective sample size.

The parameters are summarized in terms of the point *estimate*, standard deviation (*sd*), the *z-score*, *lower* and *upper* 95% confidence/credible limits (CLs) and the *p-value* (Kery and Schaub 2011, 37, 42). The estimate is the median (50th percentile) of the MCMC samples, the z-score is $\mathrm{sd}/\mathrm{mean}$ and the 95% CLs are the 2.5th and 97.5th percentiles. A p-value of 0.05 indicates that the lower or upper 95% CL is 0.

The results are displayed graphically by plotting the modeled relationships between variables and the 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 variables are expressed in terms of the *effect size* (i.e., percent change in the response variable), with 95% confidence/credible intervals (Cis; Bradford, Korman, and Higgins 2005).

The analyses were implemented using R version 3.6.0 (R Core Team 2015) and the jmbr package (Thorley 2018). The complete model specification used is provided in Appendix B.

2.5 Fish Abundance Assessment

2.5.1 Data Analysis

The spring age-1 Rainbow Trout abundance estimates used in the current analysis were provided by Greg Andrusak of the Ministry of Environment (Andrusak and Thorley 2019). Fall abundance estimates were obtained during previous study years (Years 6 to 9; Golder 2018). Observer efficiency used during fall abundance estimation in these years was derived from earlier work on Rainbow Trout and Mountain Whitefish in the LDR (Thorley et al. 2011, 2012). In the current year, updated observer efficiencies (15%; Andrusak and Thorley 2018) were used to re-estimate previously reported fall abundances.

The data were prepared for analysis using R version 3.6.0 (R Core Team 2018). The complete model specification used is provided in Appendix B.

2.6 Fish Stranding Assessment

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 (Golder 2012) 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 a RE and a list of criteria is followed to determine if a stranding assessment is required (Golder 2012).

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, 3.5 km downstream of the mouth of the LDR on Kootenay Lake. Since late 2007, debris jams have formed in Reach 3 between RKm 4.1 and 4.7, preventing continuous boat access along the river. During the current survey, a log jam in the mainstem LDR at RKm 4.7 prevented boat navigation at all available discharge levels. However, the downstream portions of the river were still accessible through a side channel located at RKm 4.5 that flowed into Meadow Creek near its outlet into the LDR. Channel movement frequently occurs at the river's mouth to Kootenay Lake and access to the LDR from Kootenay Lake is difficult at lower DRL discharges and lake elevations.

2.6.1 Year 11 Stranding Site Selection

Prior to each fish stranding assessment, 10 sites were randomly selected from all identified stranding sites. During early study years, this was accomplished by creating two strata (index and non-index) 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 as a result of the assessed DDM flow reduction.

During Years 8 to 11, stranding sites were not split into two strata. The 10 sites selected prior to each assessment were randomly selected from all 50 identified sites. The dewatered area at each site was calculated using site-specific area regression that was completed during Year 3 (Golder 2011).

2.6.2 Year 11 Sampling

2.6.2.1 Isolated Pools

Isolated pools within individual stranding sites (that formed as a result of the DDM flow reduction) were enumerated and the area (m^2) of each pool was estimated and recorded. The field crews then randomly sampled 50% of the pools at each assessed site, up to a maximum of three pools, using single pass electrofishing, dip nets and/or visual inspection. As observer efficiency can differ with the amount of cover present in each pool, the complexity of each sampled pool 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 at 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 each pool, 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)

To be consistent with past fish stranding assessments, if time allowed the dominant and subdominant substrate in each pool were recorded using a Modified Wentworth Scale.

2.6.2.2 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. The life history data for fish found stranded in dried pools were recorded (Section 2.6.2.4). Unlike isolated pools, the habitat parameters described in Section 2.6.2.1 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.

2.6.2.3 Interstitial Sampling

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. Consistent effort (i.e., a maximum of approximately twenty minutes) was conducted 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 in order to obtain sufficient numbers of data points to reduce the uncertainty of previously estimated interstitial stranding rates. The total area and dominant substrate within these areas was recorded.

If the above method was not possible due to the conditions at the site, a maximum of 10 transects were conducted 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).

To be consistent with past fish stranding assessments, the dominant substrate in each area and/or transect was recorded using a Modified Wentworth Scale.

2.6.2.4 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.

2.6.3 Data Analysis

2.6.3.1 Dewatered Area

To compare pre- and post-WUP operations, Year 11 DDM and DRL flow data were added to the discharge dataset. The calculations conducted in Year 4 (Golder and Poisson 2012) were then repeated with the updated dataset. 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.

2.6.3.2 Slope Analysis

To expand on the slope analysis conducted in Year 10 (Golder and Poisson 2019 in prep.), an additional 4 discharge levels (for a total of 14 discharge levels) were input into the GIS model. Discharges were correlated to elevation data using a DRL stage curve provided by BC Hydro. Inputting the 14 elevations into the inundation model allowed the estimation of the area of streambed to be calculated 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. These data were used during the extrapolation of pool and interstitial stranding rates over the entire study area.

2.6.3.3 Stranding

Hierarchical Bayesian Models (HBMs) were used to estimate pool presence, 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 2.5.1.

2.6.3.4 Pool Stranding

To obtain estimates for total fish stranded in pools, the number of pools in the exposed area and the number of fish per pool had to be estimated for each reduction. The number of pools at individual sites was estimated using an over-dispersed Poisson model (Kery and Schaub 2011, pp. 386–388).

Key assumptions of the final model included the following:

- The areal pool density varies by the initial discharge level as a second order polynomial.
- The areal pool density varies randomly by site and reduction.
- The number of pools is described by a gamma-Poisson distribution.

To estimate the total number of pools that form throughout the study area, mean expected pool counts were multiplied by total exposed area for each stranding event. The model code is provided in Appendix B.

The number of fish stranding in a 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 and pool area.
- The expected abundance varies randomly by study-year and reduction event.
- The abundance is gamma-Poisson distributed.
- The number of fish removed on each pass is binomially distributed.

Preliminary analyses indicated that site was not supported as a predictor. Season was defined as “spring” for January-July months and as “fall” for August-October. Reductions do not typically occur in November and December; therefore, these months were excluded from analyses.

The model code is provided in Appendix B.

2.6.3.5 Interstitial Stranding

The density of fish stranding in the interstitial area was estimated using a Generalized Linear Model (Kery and Schaub 2011). The number of fish and areas were summed by slope categories (0-2%, >2-4%, >4-6%, >6-8%, >8%).

Key assumptions of the final model included the following:

- The expected density varies by slope.
- The density is log-normally distributed.

The model code is provided in Appendix B.

2.6.3.6 Total Stranding

The percent stranding of the spring abundance of age-1 Rainbow Trout was estimated using the pool density, pool stranding and interstitial stranding models.

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 pool stranding for each reduction was the expected pool density multiplied by the expected pool stranding rate (for an average size pool) multiplied by the total area dewatered.
- 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.

2.7 Duncan Stranding Database and Data Management

The MS-Access database (referred to as the LDR stranding database) created in Year 2 (2009–2010) was populated with all available stranding data collected during Year 11. Presently, 94 individual stranding assessments are in the database. Results from 14 assessments prior to 15 September 2006 were not included in the 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).

3.0 RESULTS

3.1 Duncan Dam Discharge Reductions and Ramping Rates

Hourly discharge at DRL during the study period ranged from 48.3 m^3/s (1705.4 ft^3/s) on 21 April 2018 to 527.2 m^3/s (18619.0 ft^3/s) on 16 May 2018. Hourly discharge from DDM ranged from 2.0 m^3/s (69.3 ft^3/s) on 6 June 2017 to 280.4 m^3/s (9901.7 ft^3/s) on 12 May 2018 (Figure 2).

Lowest DDM flows typically occur during the spring/summer as Duncan Reservoir is recharged. During this period, there are temporary pulses of flow releases to meet Bull Trout (*Salvelinus confluentus)* migration requirements of daily average discharge. While DDM discharge is at its lowest during reservoir recharge, Lardeau River discharge is typically high, which satisfies flow requirements for the protection of fish and the maintenance of available habitat.

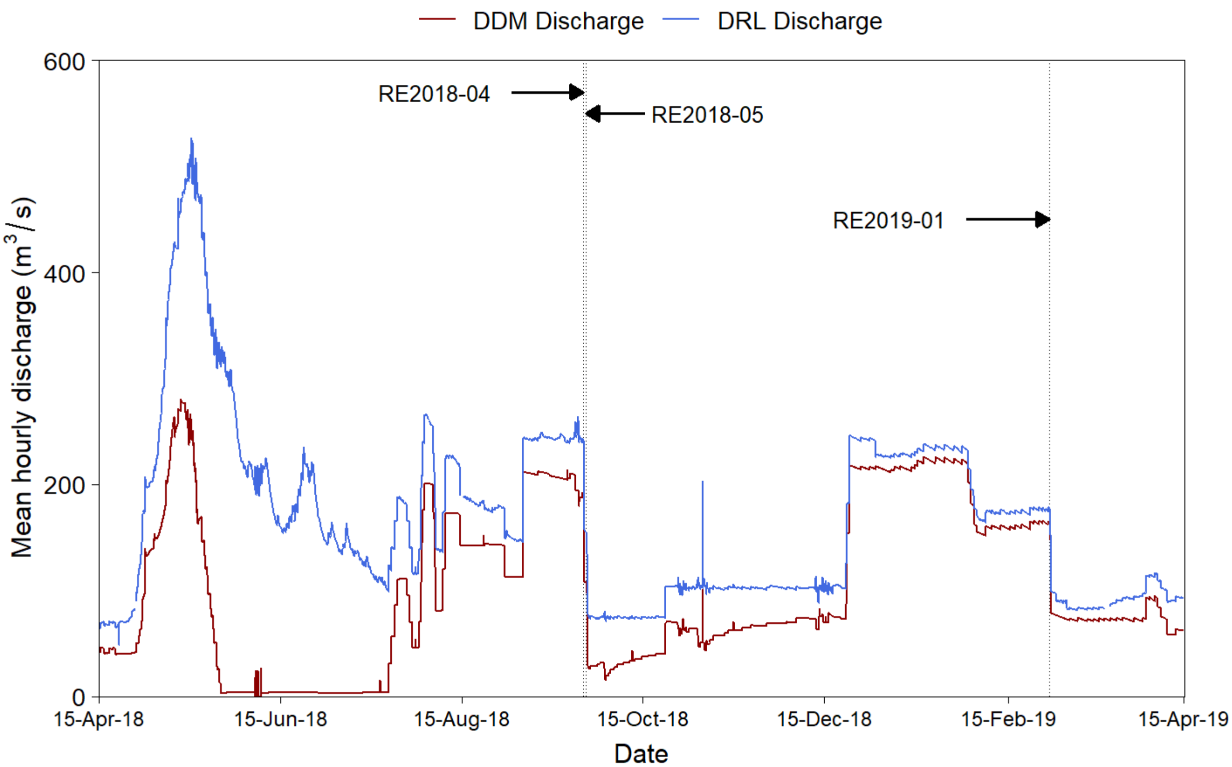

Figure 2: Hourly discharge at the Duncan Dam (DDM, red line) and at the lower Duncan River below the Lardeau River (DRL, blue line) from 15 April 2018 to 14 April 2019. Vertical dotted lines represent the timing of fish stranding assessments.

During the present study year, three reduction events occurred at DDM (Figure 2 and Table 2). During these reduction events, DDM decreased discharge between a high of 85 m^3/s (3001 ft^3/s) on 25 September 2018, and a low of 79 m^3/s (2790 ft^3/s) on 26 September 2018 (Table 2). These decreases represent the discharge reductions at DDM, rather than flow changes at particular downstream fish stranding sites.

Table 2: Summary of DDM flow reduction events, 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 Sep 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 Sep 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.

3.2 Fish Stranding Assessment Results (2006 to Present)

Fish stranding assessment results have been presented from 2006 to present 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. Stranding assessments were conducted following three flow reductions during study Year 11 (2018-2019). All fish encountered during the assessments were split into sportfish and non-sportfish categories for analysis (Table 3).

Table 3: Scientific names and species codes of fish encountered during fish stranding assessments on the lower Duncan River, September 2006 to March 2019.

Category	Species	Scientific Name	Species Code[a]
Sportfish	Rainbow Trout	*Oncorhynchus mykiss*	RB
	Bull Trout	*Salvelinus confluentus*	BT
	Mountain Whitefish	*Prosopium williamsoni*	MW
	Pygmy Whitefish	*Prosopium coulteri*	PW
	Kokanee	*Oncorhynchus nerka*	KO
	Burbot	*Lota lota*	BB
Non-sportfish	Longnose Dace	*Rhinichthys cataractae*	LNC
	Dace spp.	*Rhinicthys species*	DC
	Slimy Sculpin	*Cottus cognatus*	CCG
	Torrent Sculpin	*Cottus rhotheus*	CRH
	Prickly Sculpin	*Cottus asper*	CAS
	Sculpin spp.	*Cottus species*	CC
	Sucker spp.	*Catostomus species*	SU
	Redside Shiner	*Richardsonius balteatus*	RSC
	Northern Pikeminnow	*Ptychocheilus oregonensis*	NSC
	Peamouth	*Mylocheilus caurinus*	PCC

[a] As defined by the BC *Ministry of Environment*.

Within the dataset, the number of reduction events assessed for fish stranding per study year ranged from two (2006–2007) to eight (2008–2009 and 2017–2018). As discussed above, the focus of sampling shifted from index sites to non-index sites in Year 4 (2011–2012), which accounted for a larger proportion of non-index sites sampled in Years 5 to 11 (2012–2013 to 2018–2019). The number of pools sampled in the present year was also reduced to allow for more intensive interstitial sampling effort. During the current study year, 23 pools and 40 interstitial areas were surveyed (Table 4). The locations of all sampled stranding mechanisms within the dataset are presented in (Figure 3 and Figure 4).

Table 4: Sampling effort during reductions included in the present analysis by study year.

DDMMON-16 Study Year	Number Assessed			Number Sampled			
	Reductions	Index Sites	Non-Index 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

During Year 11, a total of 683 fish were observed, representing 10 species, of which three were sportfish and seven were non-sportfish species (Table 5). This total is the fifth lowest documented since 2006 (the median of the combined 2006–2018 dataset is 918 fish). Juvenile Rainbow Trout (*n* = 362) were the most abundant sportfish observed (53% of the total catch). During previous years, Rainbow Trout juveniles accounted for 8.6% to 58.4% of the total catch. Four juvenile Mountain Whitefish were documented stranded in Year 11, while a single Burbot juvenile was recorded as stranded. Both species accounted for 0.6% and 0.1% of the total catch, respectively (Table 5; Figure 5). The most common non-sportfish identified to species were Longnose Dace, Slimy Sculpin, and Redside Shiner, accounting for 13.9%, 1.9%, and 0.9% of the total number of encountered fish, respectively.

Figure 3: Locations and slope (%) of sampled stranding mechanisms (September 2006 to March 2019) Reaches 1 to 3.

Figure 4: Locations and slope (%) of sampled stranding mechanisms (September 2006 to March 2019) Reaches 4 and 5.

Table 5: Total number and relative composition of fish species captured or observed during all stranding assessments conducted on the lower Duncan River from September 2006 to April 2019.

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
Sportfish														
Rainbow Trout	**Adult**	0	0	0	1 (0.1)	0	0	0	1 (0.2)	0	0	2 (0.1)	0	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)
Bull Trout	**Adult**	0	0	0	4 (0.4)	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
Mountain Whitefish	**Adult**	0	1 (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)
Pygmy Whitefish	**Adult**	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
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
	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
	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
Burbot	**Adult**	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)
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)
Dace spp.		0	0	0	12 (1.3)	1 (0.1)	0	0	0	0	0	1 (0.1)	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)
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)
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)
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)
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)
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)
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
Lake Chub		0	0	0	1 (0.1)	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
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
All Species Total		350	2409	1596	918	1361	1864	896	600	1261	246	1915	392	683

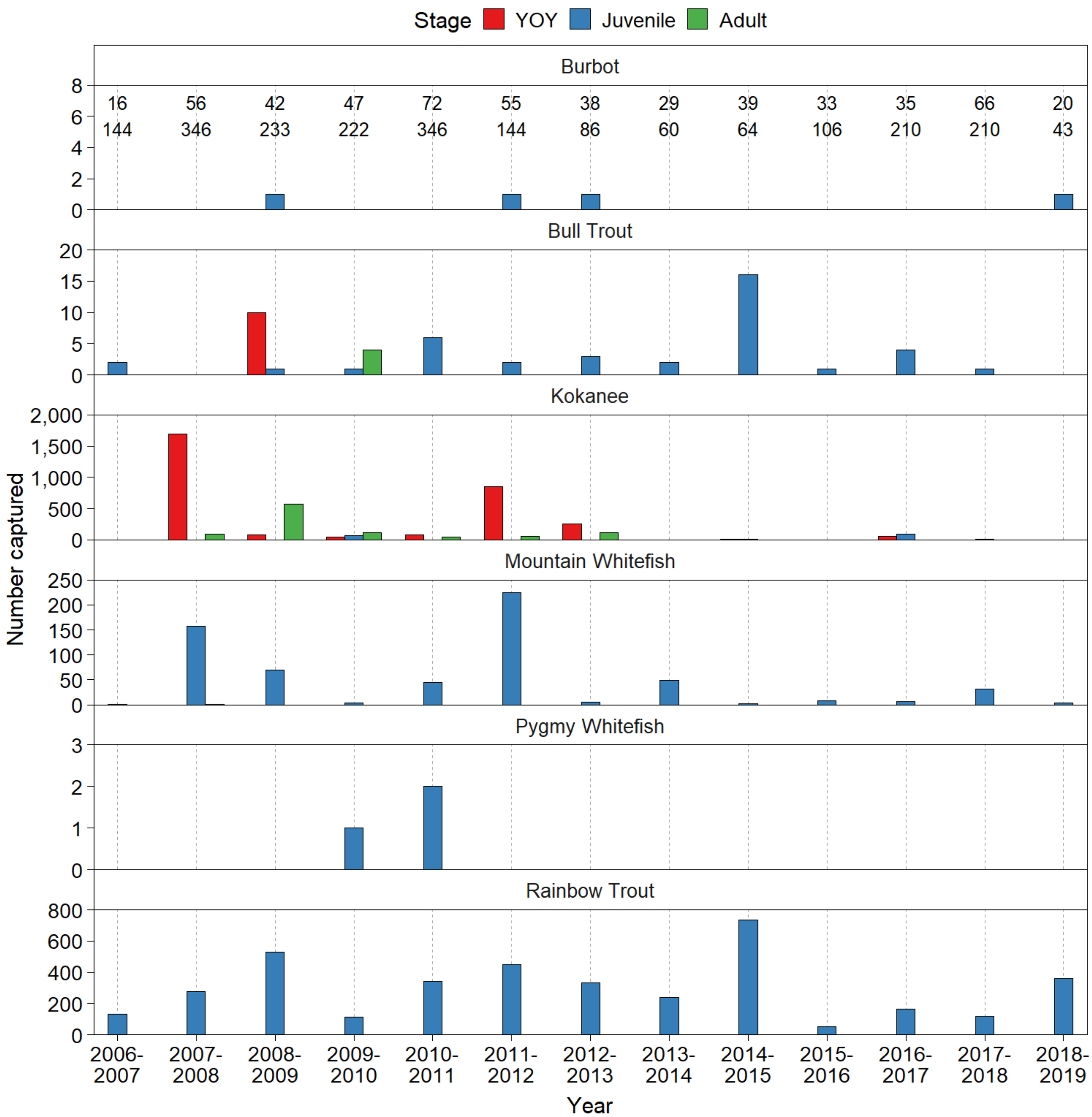

Figure 5: Abundances of sportfish species, separated by life stage, observed in stranding assessments between 2006 and 2019. Note the different y-axis scales among panels. On the uppermost panel, the numbers of sampled sites and pools are provided in the first and second lines, respectively.

3.3 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 km², in comparison to 12.4 km² during the post-WUP operational regime (Figure 6). The area exposed was less variable from year to year in the post-WUP operational regime over the years assessed and is lower in general, especially between 2013 and 2017. The maximum annual exposed area (20.5 km²) was observed in 2006, during pre-WUP operations. The minimum exposed area (10.2 km²) was observed in 2015 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 0.31 km^2, respectively; Figure 7). The difference was statistically significant (1-way ANOVA; P=0.004). Annually, mean exposed areas in reported reductions ranged from 0.2 km^2 (2015 stranding year) to 0.6 km^2 (2005 stranding years).

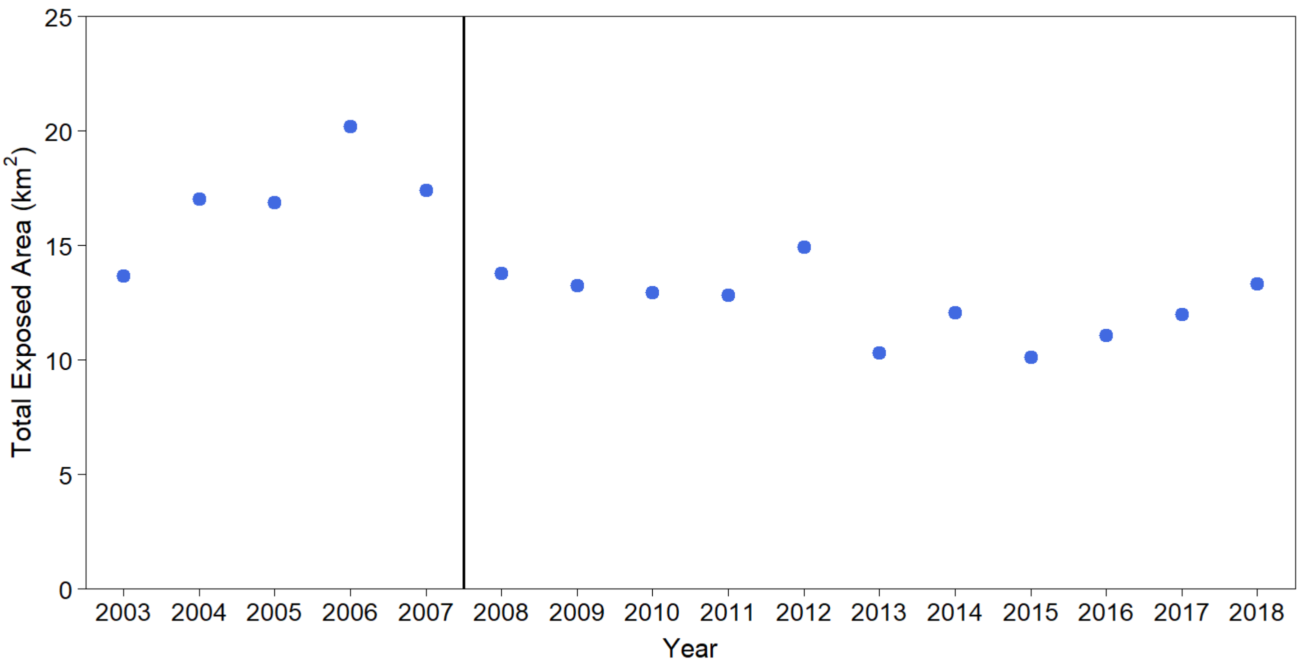

Figure 6: Total area exposed by all annual reductions in the LDR by year of operations, calculated using DRL discharge. The vertical line denotes the beginning on WUP flows in 2008. Note that label on Y-axis denotes study year not calendar year.

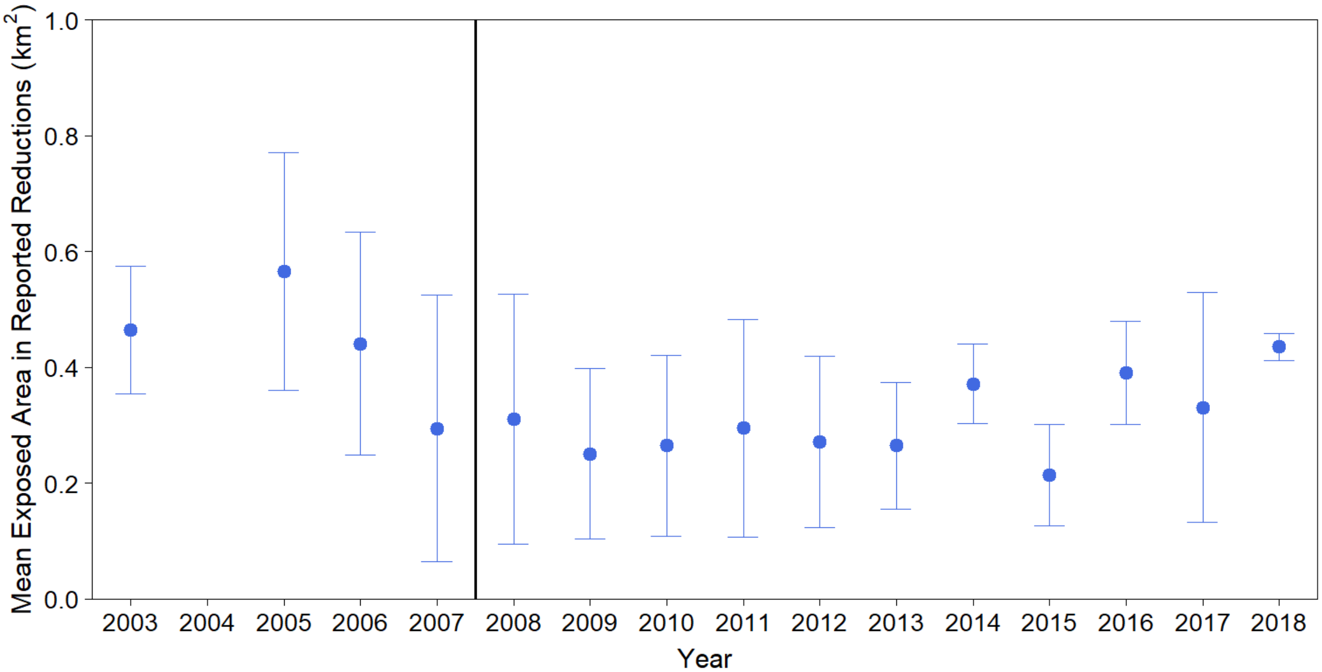

Figure 7: Mean area exposed by all annual reductions in the LDR by year of operations, calculated using DRL discharge. Bars represent 1 standard deviation. The vertical line denotes the beginning on WUP flows in 2008. Label on Y-axis denotes study year, not calendar year.

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 January period in all post-WUP years except for 2015. 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 f^3/s: Golder 2017b), resulting in high interannual variability during the October-January period (Figure 8). 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.

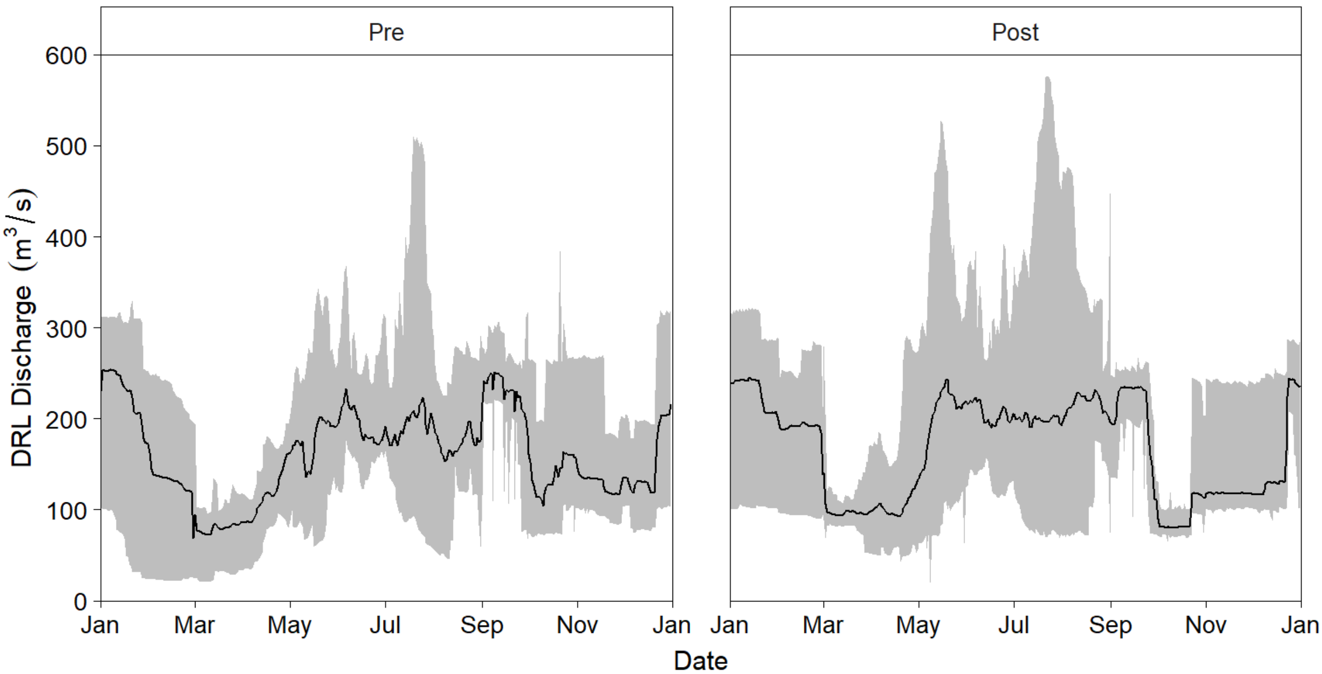

Figure 8: Minimum, maximum (grey ribbon) and mean (black line) discharge as measured at the WSC DRL gauge in the LDR during pre-WUP operations (2002–2007) and post-WUP operational implementation (2008–2019).

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 9). 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, 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 9). The remaining year in the pre-WUP period was similar to operations during post-WUP. Overall, post-WUP ramping rates were similar between years.

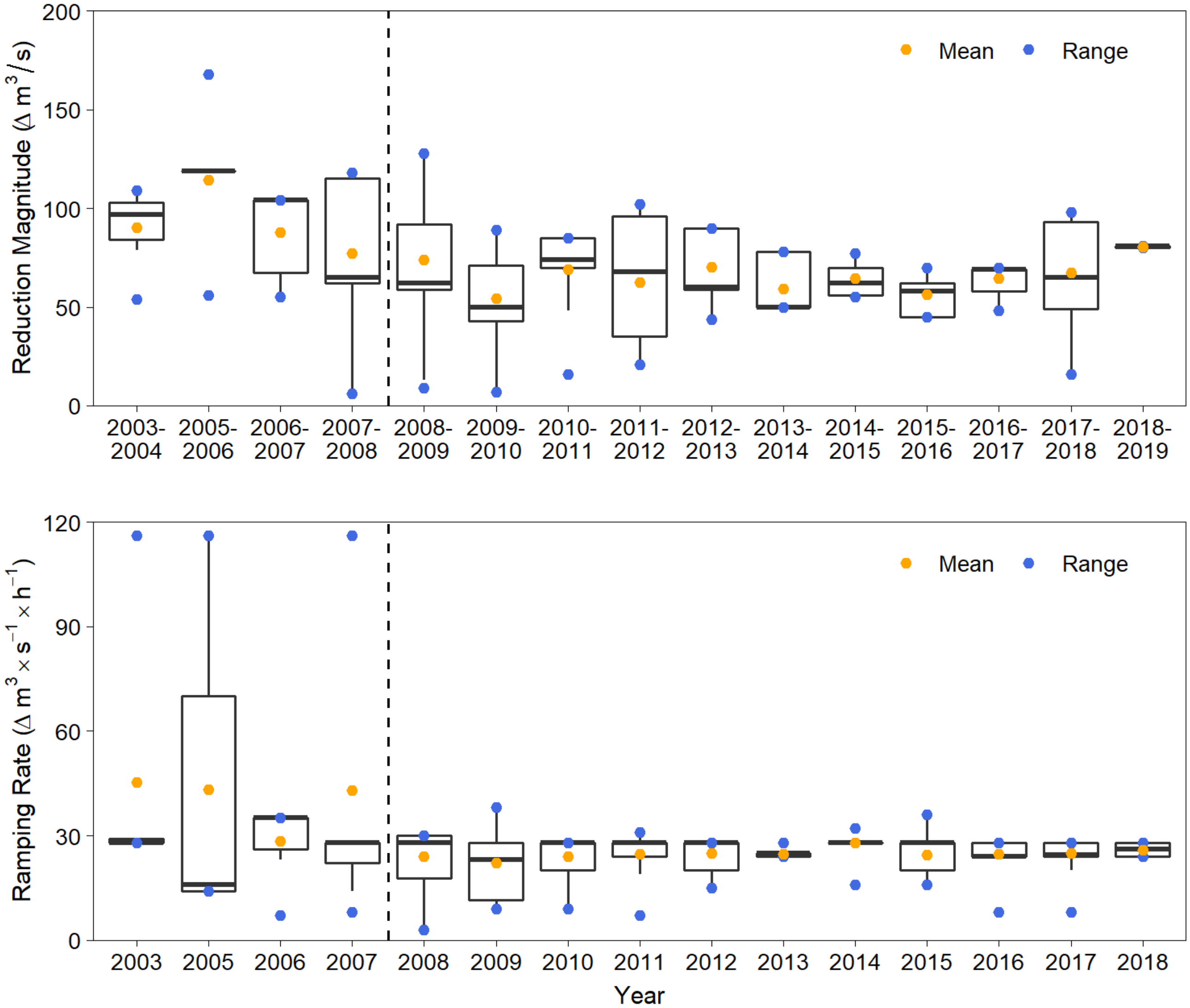

Figure 9: Boxplots of reduction magnitude ($\Delta m^3/s$; top panel) and ramping rates ($\Delta m^3\ s^{-1}\ h^{-1}$; bottom panel) by year. Each box represents the 25th and 75th quantiles (bottom and top lines, respectively), and the median (middle bold line); whiskers extend to 1.5 times the interquartile distance. Yearly mean, minimum, and maximum values are shown as individual points.

3.4 Fish Abundance Assessment

The fall total abundance estimates for Rainbow Trout ranged from 4,362 in 2016 to 24,216 in 2014 (Table 6 and Figure 10: Golder 2018). Overall, fall estimates decreased annually since the 2014 peak of estimated abundance. With the updated abundance estimation model, the estimated Mountain Whitefish population are substantially higher than previously reported (Golder and Poisson 2019 in prep.). Mountain Whitefish fall abundance in 2016 was similar to the 2015 estimates. Generally, Mountain Whitefish fall abundance decreased from approximately 50,000 in 2013 and 2014 to approximately 20,000 in 2015 and 2016 (Table 6 and Figure 10; Golder 2018).

Year 11 spring age-1 total abundance for Rainbow Trout was estimated at approximately 7,674, the lowest since 2013 (Table 6 and Figure 10). Overall, spring estimates decrease between 2013 and 2015, increased annually in 2016 and 2017, followed by a sharp decrease in 2018 (Table 6 and Figure 10).

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 6 and Figure 10).

Table 6: 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 Estimate Using Fall Snorkel Surveys		Abundance Estimation Using Spring Snorkel Surveys	
	Rainbow Trout	Mountain Whitefish	Rainbow Trout	Mountain Whitefish
Year 6 (2013)	12,225 (6,105 – 22,595)	49,496 (24,852 – 97,746)	21,099 (14,699 – 30,823)	-
Year 7 (2014)	24,216 (14,464 – 39,757)	46,023 (25,711 – 78,616)	-	-
Year 8 (2015)	8,627 (4,844 – 14,992)	21,691 (11,721 – 37,924)	8,333 (5,649 – 12,428)	-
Year 9 (2016)	4,362 (2,627 – 7,178)	22,251 (13,203 – 36,150)	15,362 (10,705 – 22,487)	-
Year 10 (2017)	-	-	26,382 (17,888 – 38,730)	-
Year 11 (2018)	-	-	7,674 (5,024 – 11,276)	-

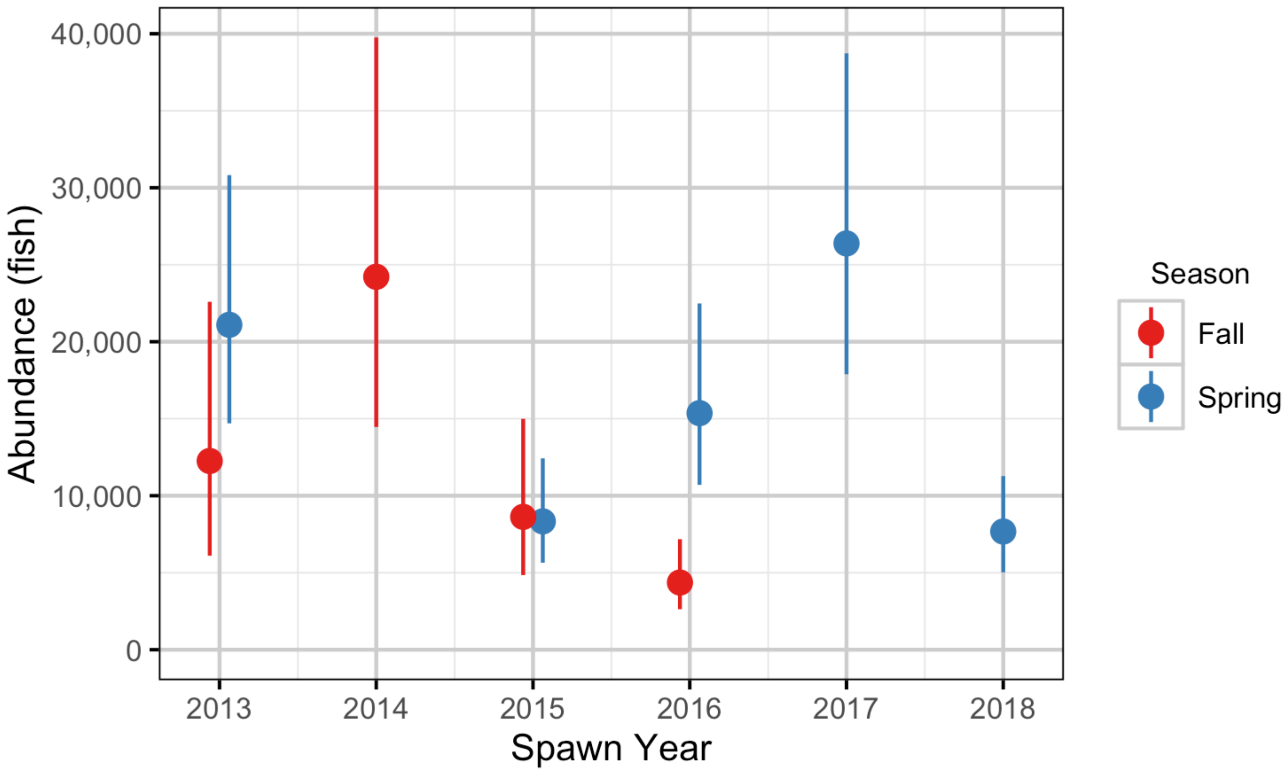

Figure 10: Estimated abundance of target species by spawn year and season in the lower Duncan River (with 95% CIs).

3.5 Fish Stranding Assessment

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 and 2019), pool and interstitial stranding estimation in the following sections refer only to Rainbow Trout.

3.5.1 Presence of Pools

The slope of each stranding mechanism sampled throughout seven years of stranding assessments (Years 3 to 11: 2010–2019) was calculated using the elevation models for the area. Slopes ranged from 0% to 60%, however all values above 20% (a total of 7 cases) were deemed artifacts of the elevation model and were removed from analysis. Generally, pool density was slightly higher at lower slope values (0% to 5%); however, the relationship was variable and weak (Figure 11). While pool densities in random sites exhibited slightly higher variation in comparison to index sites in some years (i.e., 2010, 2016, 2017), the majority of recorded pool densities were low, often lower than those recorded at index sites (Figure 11).

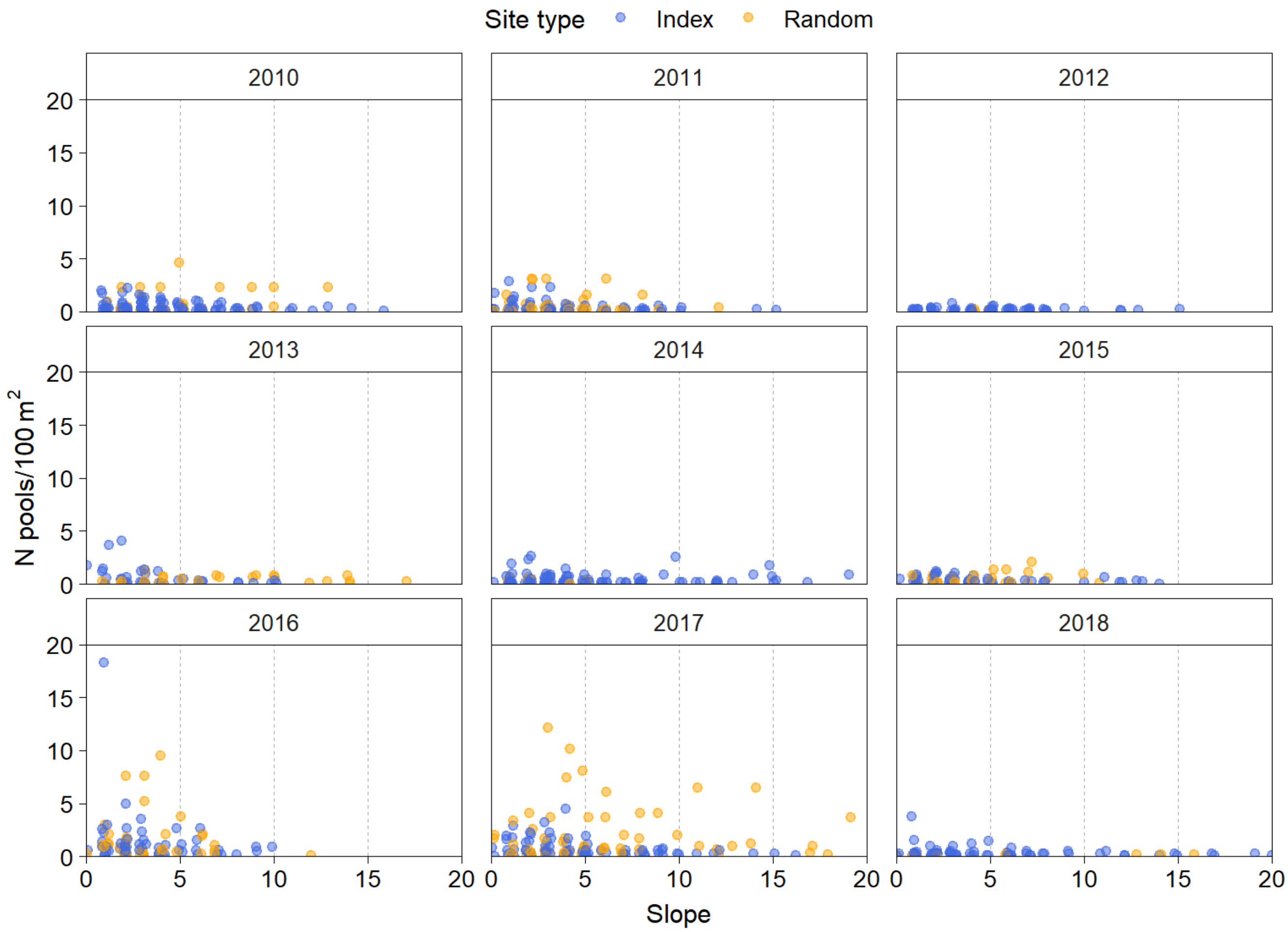

Figure 11: Density of pools recorded per reduction versus habitat slope as a continuous variable, 2010-2018.

The density of pools at typical site for a typical reduction and the number of pools per assessed flow reduction were estimated to allow the number of fish stranded per reduction (Section 3.5.2) to be calculated. Estimated pool density increases as DRL discharges decrease (Figure 12). During the late summer/early fall period (August to October) and the winter period (December to March), when flow reductions typically occur to meet operational targets, the mean number of pools that formed during stranding surveys between 2010 and 2017 was generally similar. In Year 11, the reduction-level estimates of pools were more variable between seasons, but not statistically different (Figure 13).

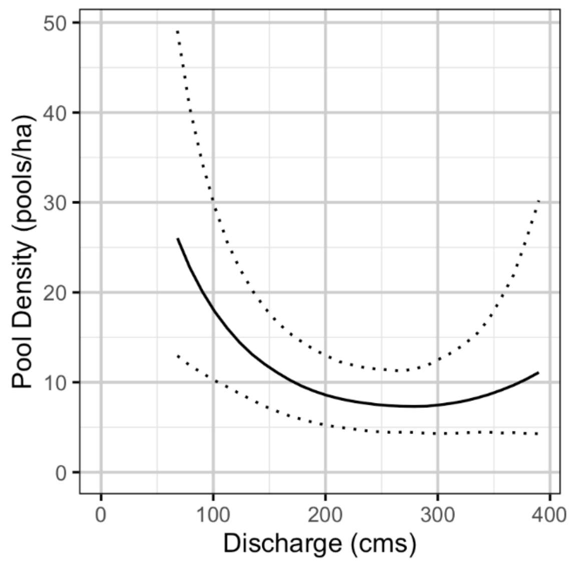

Figure 12: The estimated pool density at a typical site during a typical reduction by initial discharge.

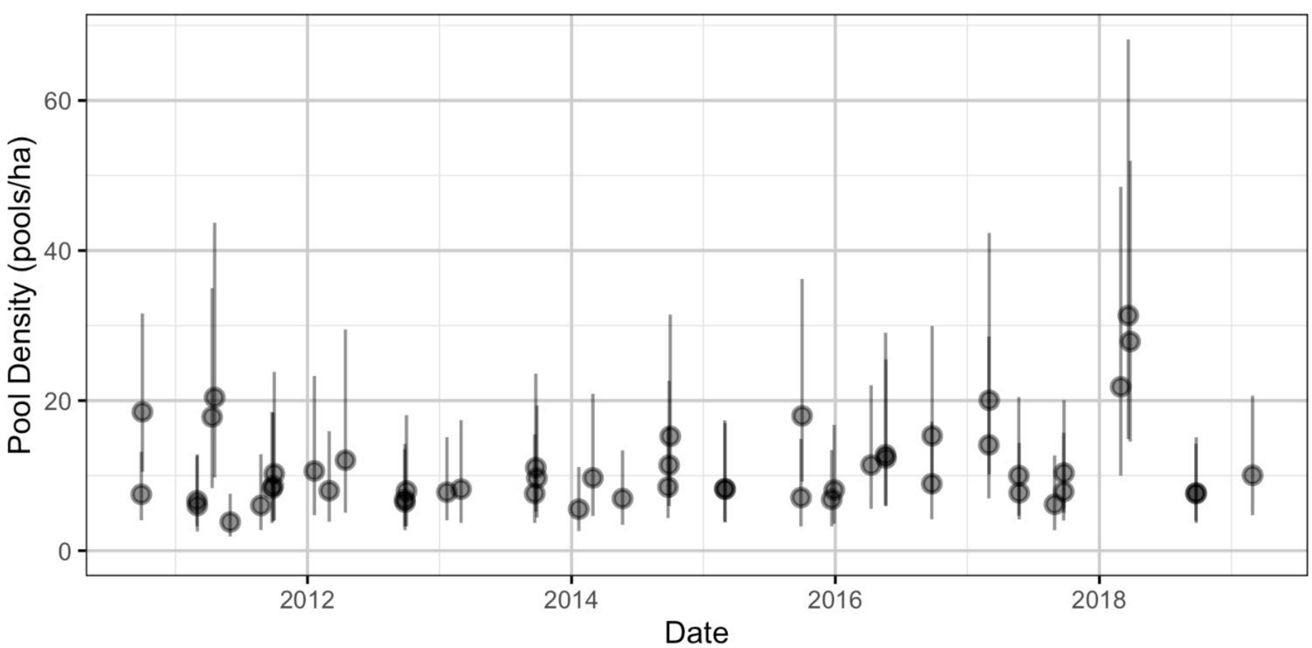

Figure 13: Estimates of pool densities by reduction event and date in the lower Duncan River. Error bars are 95% credibility intervals.

3.5.2 Pool Stranding

The number of fish stranded per pool was similar throughout the different slopes (Figure 14). This indicated that slope did not affect stranding of fish in pools.

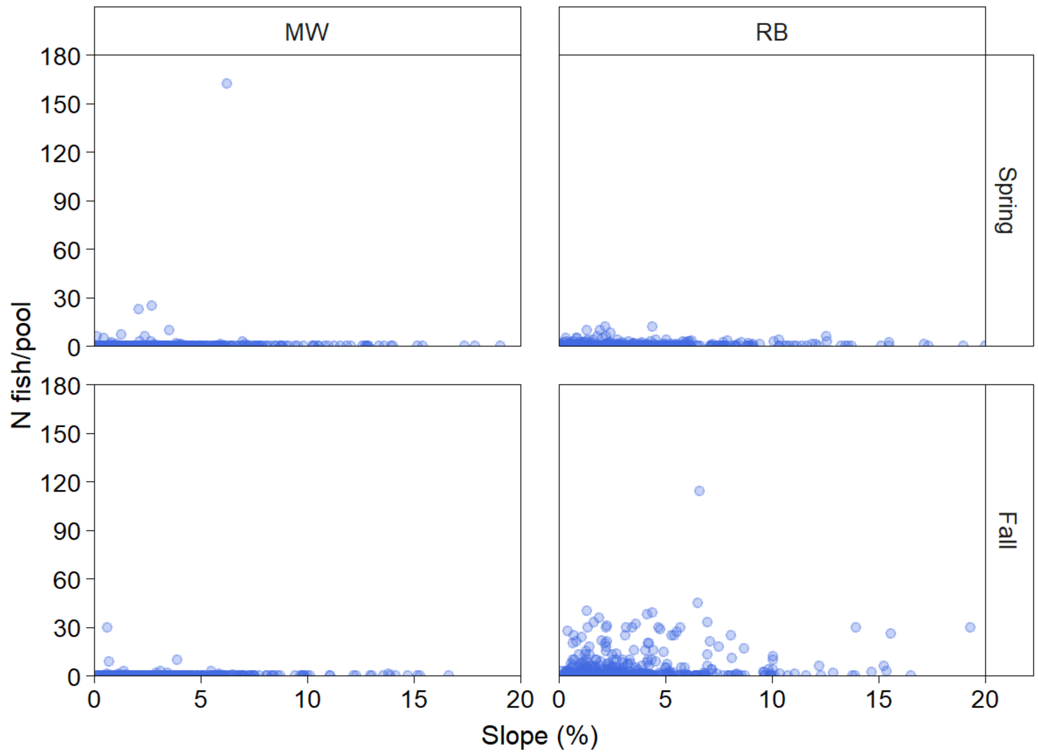

Figure 14: Number of collected fish per pool, plotted by slope and colour-coded by season, 2010–2018.

The density of pool-stranded fish differed by dominant substrate size and by species (Figure 15). Mountain Whitefish pool stranding density was low, except for pools with silt and small to large gravel. For Rainbow Trout, pool-stranded fish densities were similar across different substrate sizes. Mean Rainbow Trout densities were highest in pools with substrate ranging in size between silt and very large gravel (Figure 15).

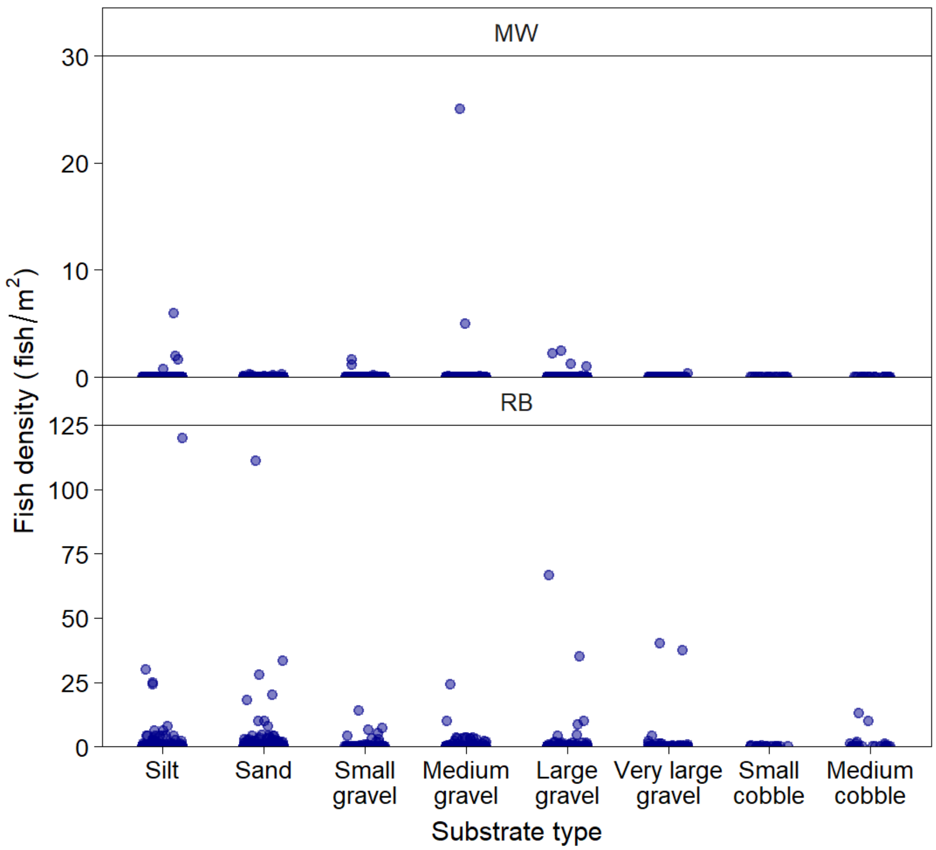

Figure 15: Scatter plot of pool-stranded fish density (fish/m²) versus dominant pool substrate size, 2006–2018, plotted by species.

The effect of season on pool stranding of Rainbow Trout was found to be significant ($p < 0.001$), with the median fall stranding estimates approximately eight times higher than those for winter/spring (Figure 16). The median number of Rainbow Trout juveniles per pool for the spring season (January to June) was estimated to be 0.42 fish/pool (CRI of 0.17 – 1.06) (Figure 16). In contrast, the median number of Rainbow Trout juveniles stranded per pool in the fall (July to December) was estimated at 3.43 (CRI of 1.55 – 8.49).

Based on the presence of pools and number of fish per pool estimates, it was then possible to estimate the number of fish stranded in pools for individual reduction events (Figure 17). Generally, spring Rainbow Trout estimates of pool stranding were lower than fall estimates. Fall pool estimates were highest between 2011 and 2014 (Figure 17).

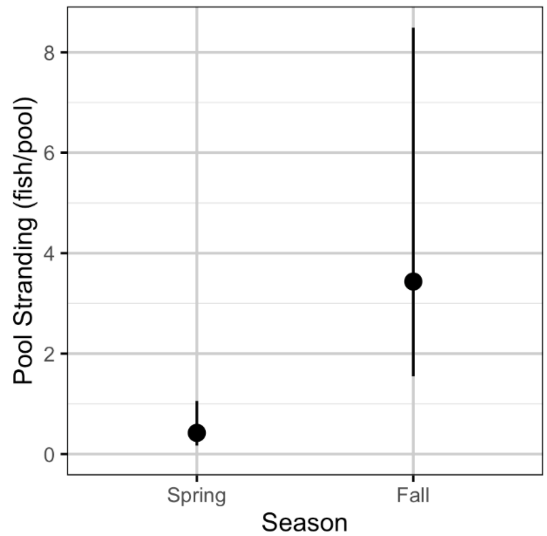

Figure 16: The expected pool stranding in an average pool during a typical reduction event by season in the lower Duncan River. Error bars are 95% credibility intervals.

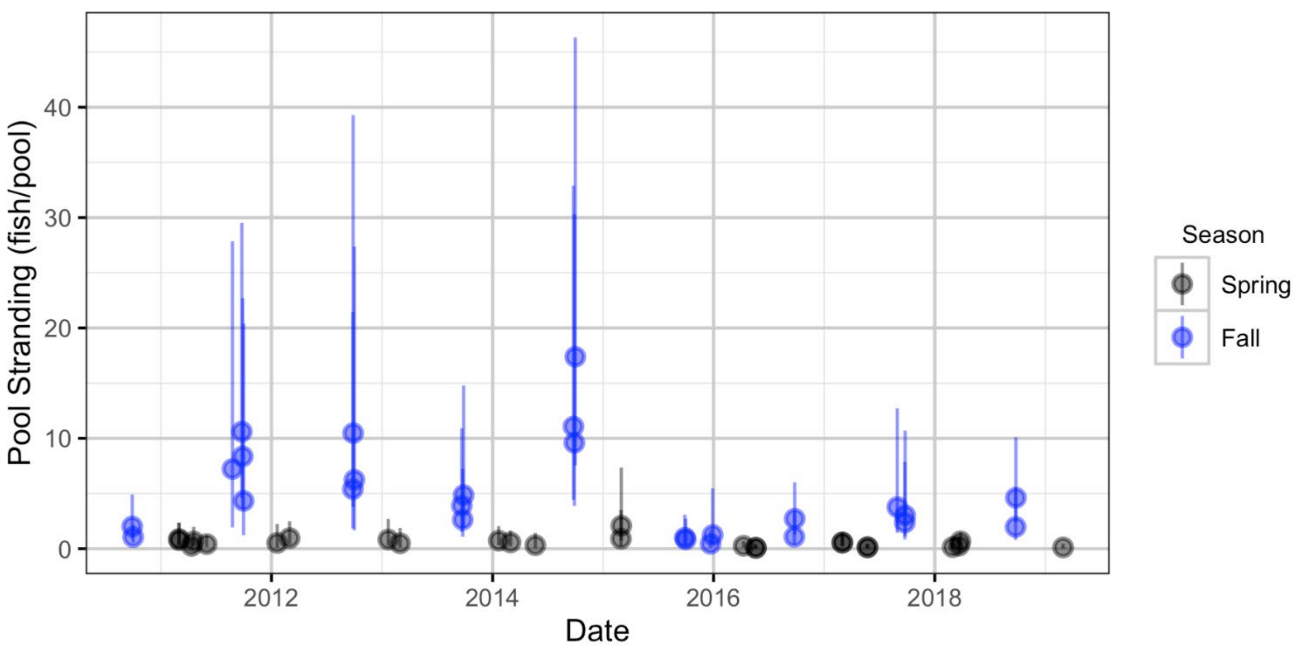

Figure 17: Estimates of Rainbow Trout per pool by date and season in the lower Duncan River. Error bars are 95% credibility intervals.

3.5.3 Interstitial Stranding

Between Year 4 (2011-2012) and Year 11 (2018-2019), 31 Rainbow Trout and 2 Mountain Whitefish were found to be interstitially stranded on substrates ranging in size from silt to large gravel (Figure 18). 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, seven Rainbow Trout were recorded as interstitially stranded. All documented interstitially stranded fish were found on exposed areas with low slopes (≤7%; Figure 19). As slope increases, the risk of interstitial stranding was found to decrease (Figure 20).

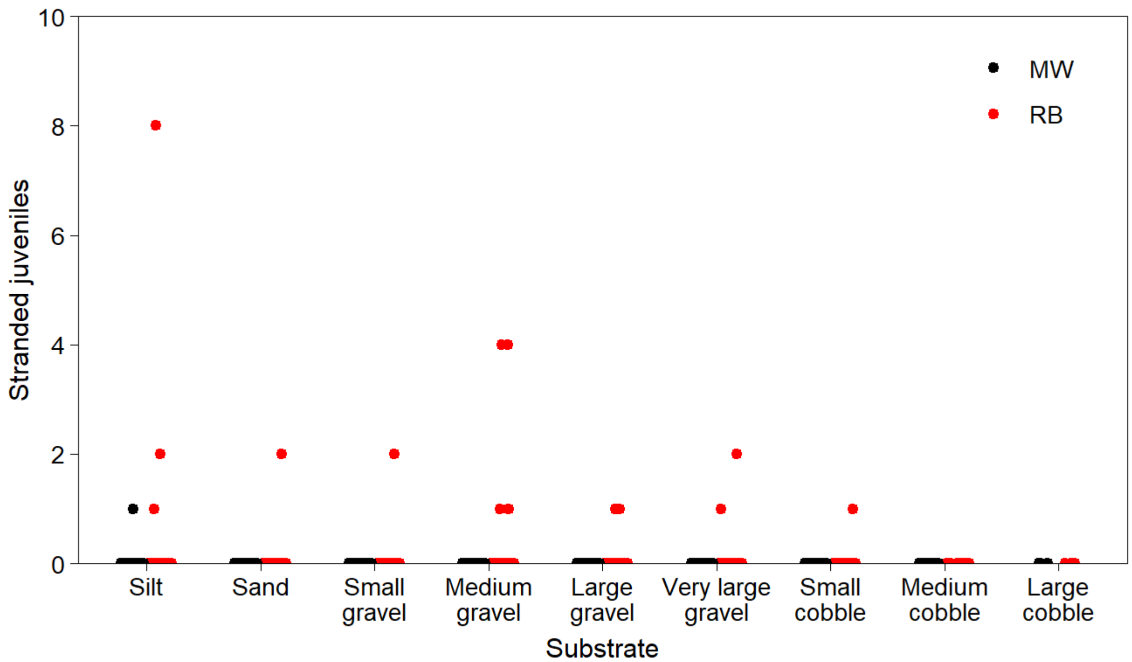

Figure 18: Counts of 2011–2018 interstitially stranded Mountain Whitefish and Rainbow Trout in the lower Duncan River, plotted by substrate size.

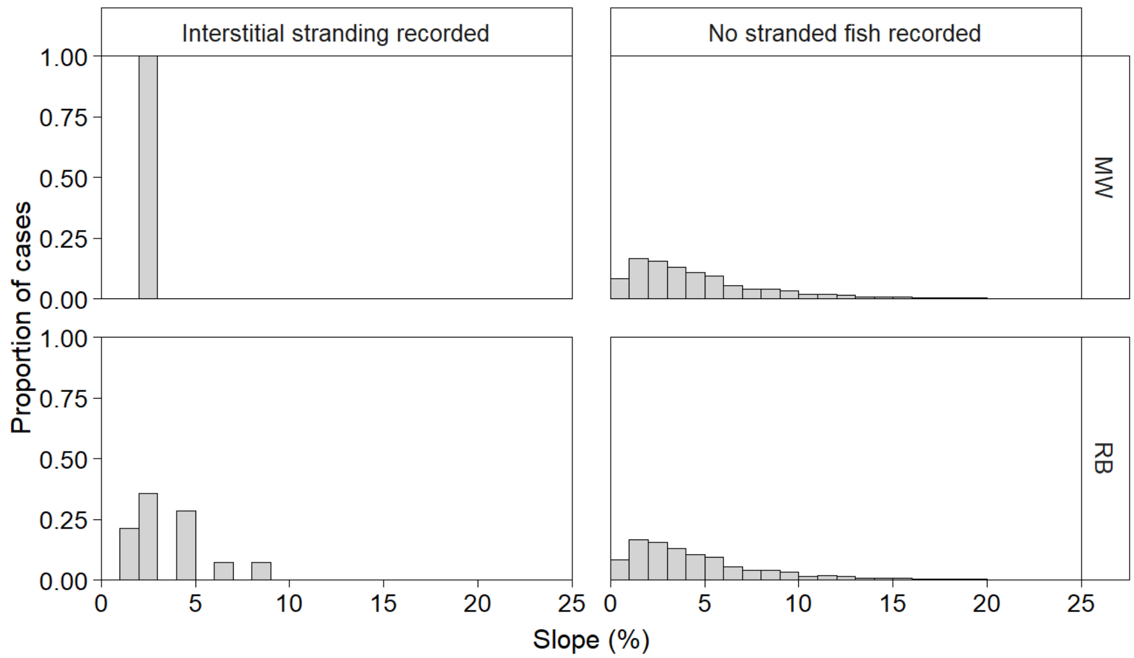

Figure 19: Histogram of 2011–2018 interstitially stranded Mountain Whitefish and Rainbow Trout in the lower Duncan River, plotted by species and slope (%).

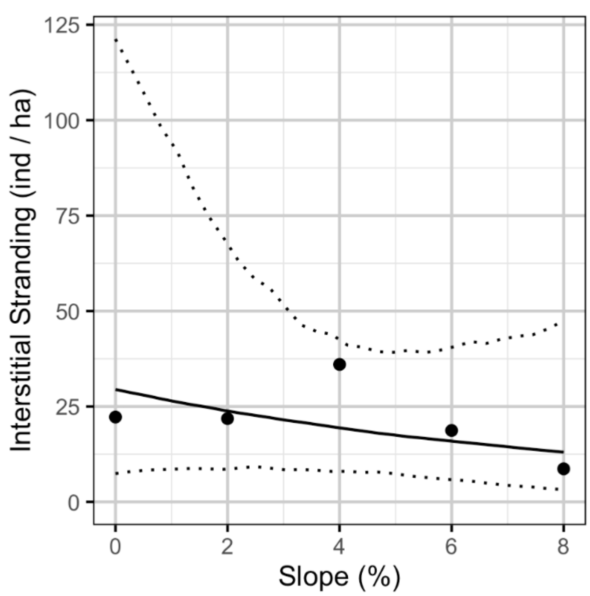

Figure 20: The estimated interstitial stranding density for Rainbow Trout in the lower Duncan River by slope.

3.5.4 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 7 and Figure 21. Habitat with greater than 8% slope were the most abundant in all examined DRL discharges, as well as areas with slopes between 0 – 2% and 6 – 8%.

Table 7: Estimated wetted area (m^2) by slope in the lower Duncan River, based on DRL discharge.

Slope Category (%)	Discharge at DRL (m^3/s)													
	68.0	73.0	110.8	148.6	186.4	224.2	262.0	299.8	337.6	375.4	390.2	428.0	465.8	488.0
0-2	185,775	197,075	238,975	260,050	327,975	383,325	443,850	522,600	595,500	650,100	664,700	767,425	850,300	890,525
2-4	200,400	224,025	262,325	279,700	334,675	369,525	401,700	433,875	466,575	498,675	509,375	535,600	568,775	588,875
4-6	134,625	143,150	167,775	179,650	215,875	238,100	256,800	274,875	291,625	305,775	311,150	335,525	351,475	361,250
6-8	97,275	102,350	120,100	127,675	148,775	163,500	174,925	185,400	195,550	204,975	208,575	227,925	237,925	243,250
>8	251,275	258,625	301,975	324,325	376,575	410,275	441,900	466,225	488,575	509,200	515,700	571,675	594,375	606,550
Total	869,350	925,225	1,091,150	1,171,400	1,403,875	1,564,725	1,719,175	1,882,975	2,037,825	2,168,725	2,209,500	2,438,150	2,602,850	2,690,450

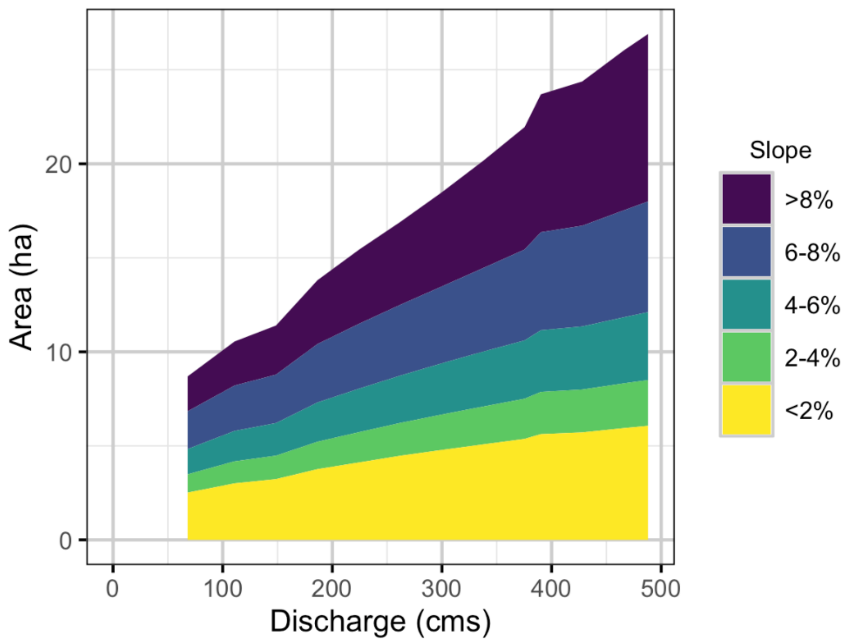

Figure 21: The calculated wetted area in the Lower Duncan River by slope and DRL discharge.

3.5.5 Total Stranding Estimates

Total spring Rainbow Trout pool stranding estimates were consistently low and invariable across study years examined (Figure 22). In most of the study years examined, total fall pool stranding estimates were higher and substantially more variable. When the seasons were combined in each study year, mean total pool stranding estimates ranged between approximately 0.2% (2010) and 1.9% (2014) of the projected spring age-1 Rainbow Trout population (Figure 23). Except for the 2014 study year (1.9%), mean annual pool stranding was estimated at less than 1.0% of the total spring Rainbow Trout population in the LDR.

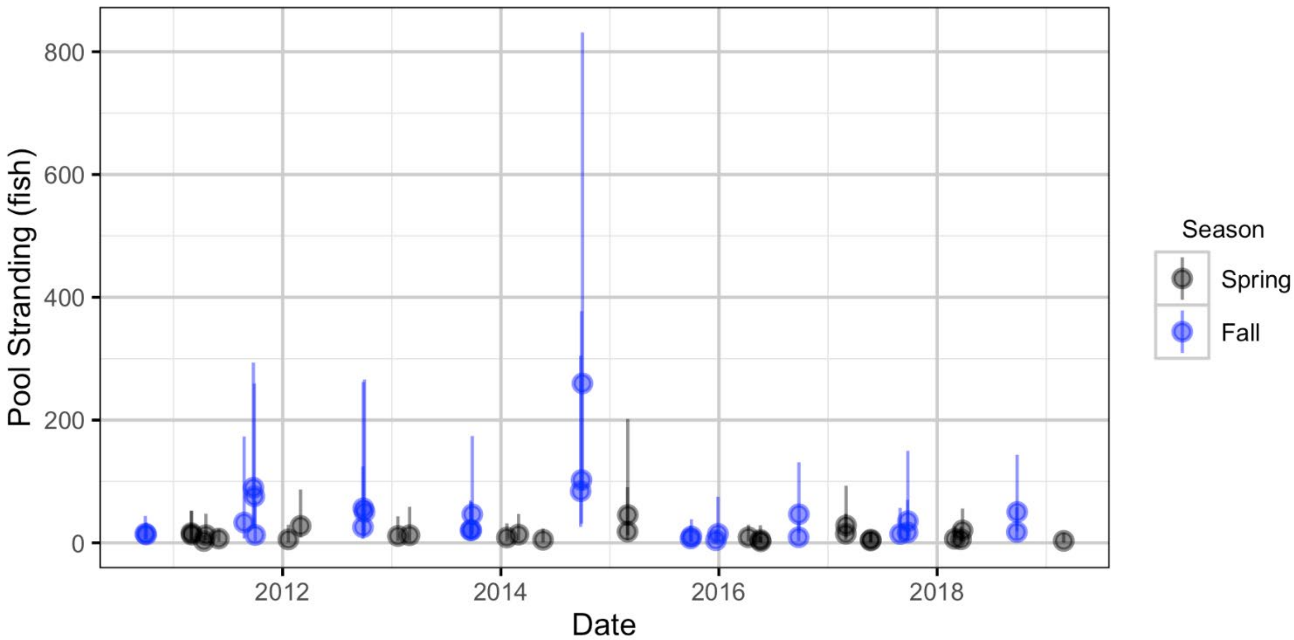

Figure 22: Estimates of total pool-stranded Rainbow Trout by date and season in the lower Duncan River. Error bars are 95% credibility intervals.

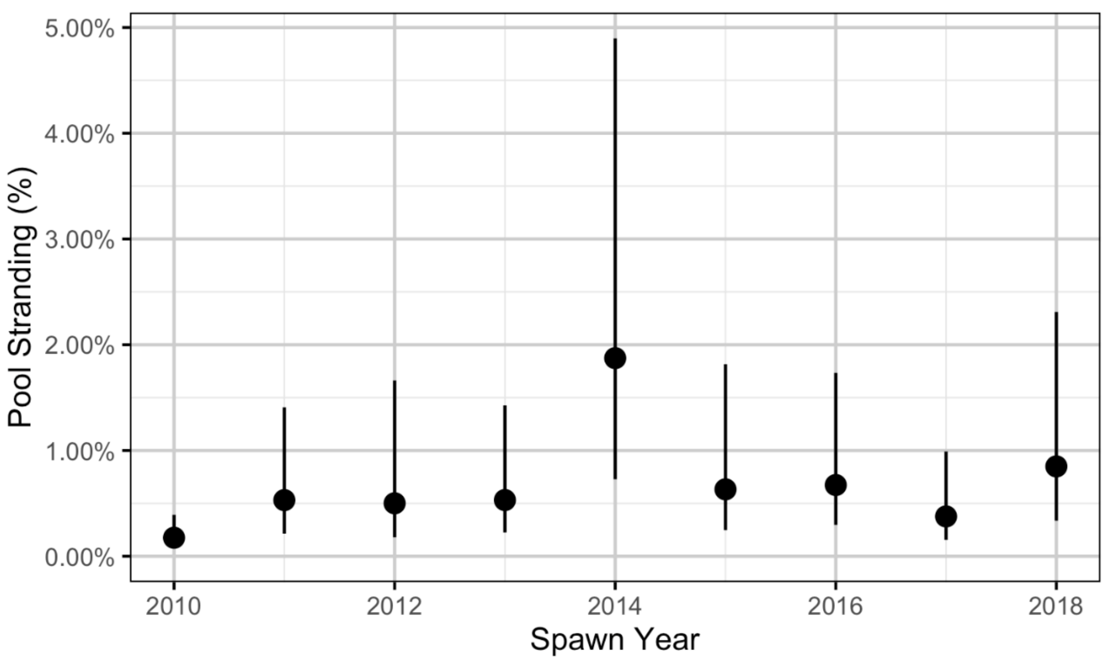

Figure 23: Estimated total pool stranding of Rainbow Trout in the lower Duncan River as a percent of spring abundance by spawn year. Error bars are 95% credibility intervals.

Total Rainbow Trout interstitial stranding estimates were highly variable between seasons and study years (Figure 24). In most study years, spring interstitial stranding estimates were typically higher and more variable in comparison to the fall season. When the seasons were combined in each study year, total mean interstitial stranding estimates ranged between approximately 0.4% (2011) and 3.5% (2015) of the projected spring age-1 Rainbow Trout population (Figure 25). Except for the 2015 study year (3.5%), mean annual interstitial stranding was estimated at less than 3.0% of the total spring Rainbow Trout population in the LDR.

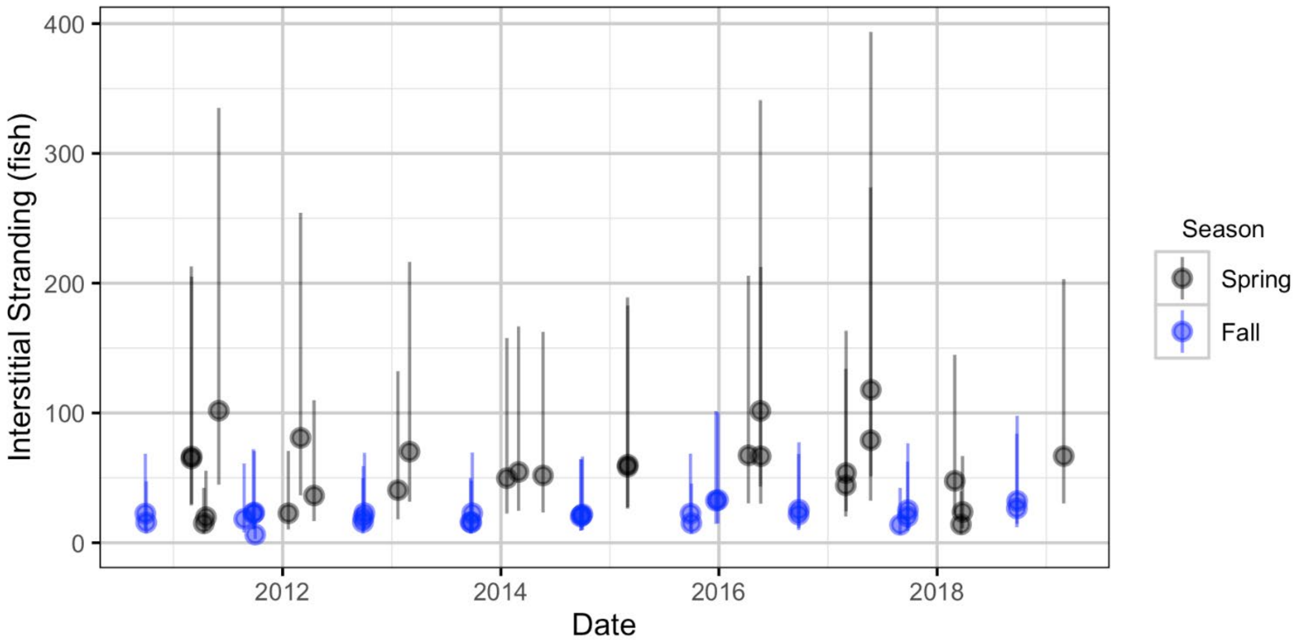

Figure 24: Estimates of total interstitial-stranded Rainbow Trout by date and season in the lower Duncan River. Error bars are 95% credibility intervals.

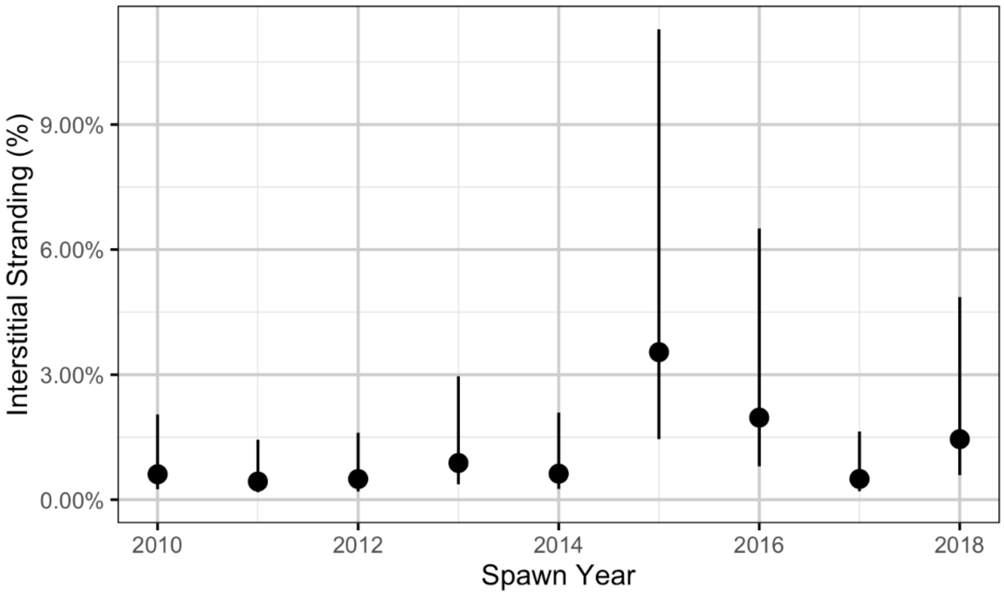

Figure 25: Estimated total interstitial stranding of Rainbow Trout in the lower Duncan River as a percent of spring abundance by spawn year. Error bars are 95% credibility intervals.

For Rainbow Trout, total stranding (interstitial and pool combined) for the current study year was estimated at 2.3% of the Rainbow Trout age-1 spring population (95% CRI of 1.2 – 5.9%; Figure 26). Total percent stranding remained relatively consistent from 2010 to 2012 and increased each year from 2013 to 2015. In 2016 and 2017, total percent stranding estimates decreased. The highest estimate was recorded in 2015, with a mean estimate of 4.2% of the Rainbow Trout spring age-1 population (95% CRI of 2.0–11.8%; Figure 26).

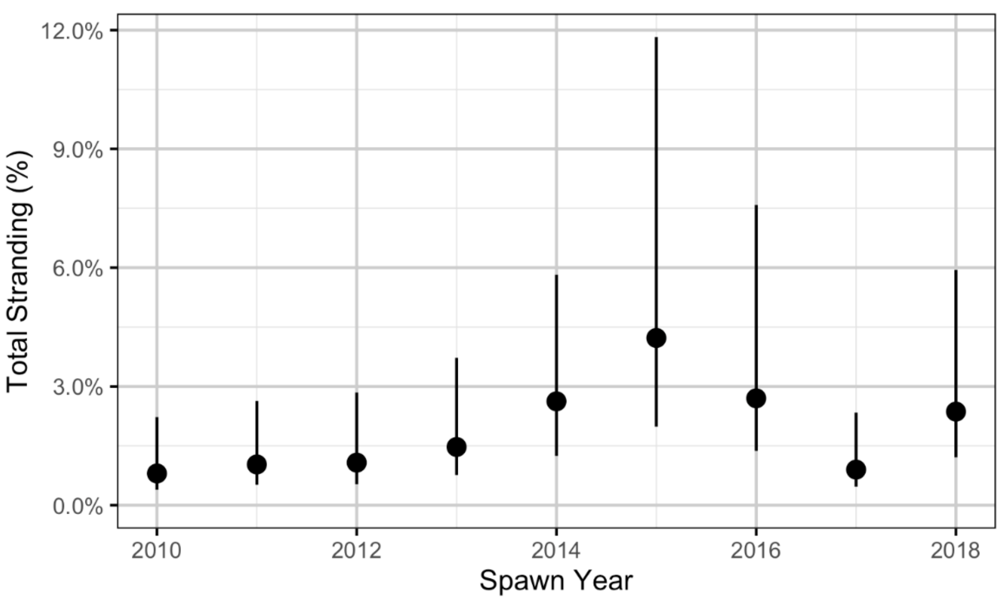

Figure 26: Estimates of total percent stranded Rainbow Trout by date and season in the lower Duncan River. Error bars are 95% credibility intervals.

4.0 DISCUSSION

4.1 Current Duncan Dam Operations in Relation to Fish Stranding

4.1.1 Variables Affecting Fish Stranding

There are several environmental and operational variables of interest that could affect fish stranding. Within that suite of variables, those that are currently addressed by operational strategies to potentially reduce fish stranding rates are ramping rate (discussed below in Section 4.1.2) and time of day (Golder 2011, Golder and Poisson 2012). The operational variable related to stranding that is currently not specifically addressed by the ASPD is wetted history (Poisson and Golder 2010). This variable was analysed and discussed in-detail as part of DDMMON-1 (Poisson and Golder 2010) and in Years 4 and 5 of this program (Golder and Poisson 2012, Golder 2014).

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 for support of Columbia River Mountain Whitefish management objectives (which are currently under review and may change). 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. The examination of the amount of exposed habitat per year due to LDR discharge reductions indicated that post-WUP flows have resulted in the dewatering of less area compared to pre-WUP operations. Interannual variability in discharge has also been reduced under post-WUP operations. During post-WUP operations, variability of total reduction magnitudes and ramping rates have also been reduced. As recommended by the DDMMON-1 and DDMMON -15 Programs (Poisson and Golder 2010, Golder 2012), DDM operations are required under the current water license to reduce flows at a ramping rate that ensures a stage change of 10 cm/hr or less at the majority of identified stranding sites when possible. Data trends identified in those 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 requirement has resulted in consistently similar ramping rates during post-WUP operations in the LDR.

<u>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.</u> Operations at DDM have been adjusted to reduce fish stranding rates and lower the amount of habitat dewatered under the post-WUP operating regime. As the sampling programs assessing fish stranding levels through time have had different methodologies and varying study foci through the years, it is not possible

to provide comparable fish stranding estimates from the 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 for this study program are Rainbow Trout and Mountain Whitefish. During the Year 11 assessments, 10 different species were encountered (three sportfish and seven non-sportfish species), but Rainbow Trout was the only species of interest with substantial numbers of stranded individuals.

4.2.1 Pool and Interstitial Stranding Rates

Current estimates for the number of Rainbow Trout juveniles stranded in pools were relatively precise and relatively low. The effect of estimated pool stranding rates on the juvenile Rainbow Trout population in the LDR is discussed below in Section 4.3. Previous analysis showed that residual wetted areas of pools was not a predictive variable (Poisson 2011, Golder and Poisson 2012). In the current 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. Significant differences were not found between substrate size within isolated pools and the density of pool stranded fish, 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 increases as DRL discharge decreases.

Over the study years when interstitial sample methodologies were standardized, very few interstitially stranded fish have been observed. In comparison to previous study years (Golder 2018), the uncertainty related to interstitial stranding rates of Rainbow Trout juveniles in the current study program has decreased substantially, which allows for the determination of the effect of these rates on population levels (Section 4.3). A relationship between interstitially stranded fish counts and substrate size was not found. This relationship should continue to be evaluated in future years as more data are collected.

4.2.2 Slope of Dewatered Area

The categories of low (0-4%) and high slope (>4%) used in the analyses during previous study years were based on values in the literature (e.g., Bauersfeld 1978; Flodmark 2004). Based on the previous data analyses, considerably higher amounts of low slope habitats were dewatered during flow reductions from DDM, and the dewatered low slope habitats had substantially more fish interstitially stranded following flow reductions than high slope habitats (Golder and Poisson 2012). The findings of the current study year support these conclusions.

Analyses on the current dataset suggested that slope did not influence the formation of isolated pools within the study area. As such, the effect of slope was not included in the pool stranding analysis. Pool density was slightly higher at lower slope values; however, the relationship was variable and weak. This indicated that slope was not a significant factor influencing pool stranding with the current dataset. This finding could be due to high variability, low DEM resolution and low data volume, and the effect of slope should be evaluated yearly as more data are collected.

Based on the analysis of the current dataset, a relationship between slope and interstitial stranding exists. However, statistically significant relationships between interstitially stranded fish counts and slope were not found. Fish found interstitially stranded in all study years analyzed were on slopes of 7% or less. As slope increases between 0 and 8%, the projected risk of interstitial stranding was found to decrease.

4.2.3 Index and Non-Index Stranding Sites

The first specific hypothesis to address 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. Based on the findings of previous study years (Golder and Poisson 2012, 2019 in prep.; Golder 2017a, 2017b, 2018), index sites tended to be of lower gradient than non-index sites. Interestingly, in Year 6 the number of pools per unit area of exposed habitat did not vary between index and non-index sites nor did the number of fish per pools (Golder 2015). This suggests that other than being lower gradient and therefore exposing more area, stranding rates (stranding per lineal km of river) do not differ substantially between index and non-index sites. The belief was that overall, index sites strand more fish because more area dewaters at these sites during flow reductions.

In Years 8 to 10 (Golder 2017b, 2018; Golder and Poisson 2019 in prep.), as well as in the current study year, there was no significant statistical effect of index and random 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 index/random site 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 not rejected.
In Year 12, stranding rates at both index and random sites should continue to be analyzed as the dataset increases in size.

4.3 Effect of Stranding on Fish Populations in the Lower Duncan River

The second specific hypothesis (H_{02}) to address Management Question 2 states: *Fish populations in the LDR are not significantly impacted by fish stranding events*. 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, predation, outmigration, food availability, availability of suitable 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).

4.3.1 Rainbow Trout Juvenile Population

As fall abundance surveys were not conducted during the current study year, estimated Rainbow Trout juvenile abundance was calculated based on spring surveys conducted by Andrusak and Thorley (2019). Previously estimated fall abundance for juvenile Rainbow Trout increased from 2013 to 2014, followed by sharp decreases in 2015 and 2016. Conversely, the spring surveys estimated an increase in the juvenile Rainbow Trout population from 2015 to 2017. The similarities between spring and fall Rainbow Trout juvenile abundance estimates in 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 Year 10 (Golder and Poisson 2019 in prep.), 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) did not correct this discrepancy. As reported in Year 10 (Golder and Poisson 2019 in prep.), if the decreasing juvenile Rainbow Trout populations documented by the previous 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.

Estimated spring abundance for juvenile Rainbow Trout in 2018 decreased sharply from the previous year and were the lowest since 2013. Andrusak and Thorley (2019) report that this decline is a result of large changes in total spawner returns and fluctuations in egg deposition related to variation in size at maturity associated with food limitations related to collapse of Kokanee. Total mean annual estimates for the number of Rainbow Trout juveniles stranded were consistently low, ranging from 0.8% (95% CRI of 0.4% to 2.2%) of the Rainbow Trout age-1 spring population in 2010 to 4.2% (95% CRI of 2.0% to 11.8%) in 2015. Based on these low estimates and the findings of Andrusak and Thorley (2019), there is no evidence to suggest a correlation between the decline in Rainbow Trout juvenile spring abundance in 2018 and DDM operations. Therefore, with the current state of knowledge hypothesis H_{02} is not rejected for Rainbow Trout. It can be concluded that fish stranding as a result of DDM operations does not have a significant impact on juvenile Rainbow Trout populations.

4.3.2 Mountain Whitefish Juvenile Population

Currently, spring abundance estimates for Mountain Whitefish are not available. The fall total abundance estimates for Mountain Whitefish obtained using abundance modelling decreased from Years 6 to 8, while stabilizing in Year 9. In the current year, only four stranded Mountain Whitefish were documented, and encounters have been low in all study years. This 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 have a significant impact on juvenile Mountain Whitefish populations. However, previous experimental stranding investigations indicated that large numbers of Mountain Whitefish could be stranded during rapid nighttime reductions in flow (Poisson and Golder 2010). Consequently, these conclusions assume that operations in the future will be within the range and the diel timing that occurred during this program.

5.0 SUMMARY

The key findings for the Year 11 of the Lower Duncan River Fish Stranding Impact Monitoring Program (DDMMON-16) are as follows:

- **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 4.1.1).
- **Management Question 2**) (*What are the levels of impact to resident fish populations associated with fish stranding events on the lower Duncan River?):*
  - As reported in Year 7 to 10 results (Golder 2017a, 2017b, 2018, 2019), seasonal effect on pool stranding in Year 11 was found to be statistically significant (Section 4.2.1)
  - As in previous study years, interstitial stranding encounters continue to be very low (Section 4.2.1)
  - Slope has an effect on interstitially stranded fish counts, although this effect is not statistically significant (Section 4.2.2)
  - Statistically significant relationships between pool density and slope in the current dataset were not found (Section 4.2.2)
- Study Hypothesis H_{01}: *(Fish stranding observed at index sites along the lower Duncan River floodplain is representative of overall stranding)*:
  - Site type was found to not have a significant effect on pool formation and pool stranding rates (Section 4.2.3)
  - 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 4.2.3)
- Study Hypothesis H_{02}*: (Fish populations in the LDR are not significantly impacted by fish stranding events)*:
  - With the analysis of the current dataset, the study hypothesis H_{02} for Rainbow Trout and Mountain Whitefish is not rejected (Section 4.3.1 and Section 4.3.2)

In summary, this monitoring program provides an understanding of fish stranding in relation to DDM operations and helps management reduce the severity of fish stranding in the LDR. Based on the current state of knowledge, 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 refinements to the modelling methodology and the growth of the dataset, the mean estimated total stranding of Rainbow Trout and estimation uncertainty for all study years was reduced. With continued enhancement to sampling and modelling methodology, and another year of data collection to increase the size of the dataset, the precision related to stranding estimation is expected to continue to increase. To better understand stranding related to the species of interest in the LDR, recommendations for methodology refinements are presented below in Section 6.0.

6.0 RECOMMENDATIONS

Recommendations from the current year (Year 11) of the Lower Duncan River Fish Stranding Impact Monitoring Program (DDMMON-16) are as follows:

1) Continue following the current pool sampling methodology used in Year 11 stranding assessments. This will continue to strengthen the existing dataset and allow for continued accurate estimates of fish stranding in the LDR.

2) Conduct mapping of the substrate in the LDR in Year 12. Possible substrate mapping methods include aerial drone high definition photography of the LDR when flows are at the target minimum of 73 m^3/s. The aerial imagery should have sufficient resolution to geospatially document substrate size.

3) In Year 12 (2019-2020) of this program, develop and present a protocol to address the outstanding management questions of the DDMMON-1 program. The outstanding management questions will be addressed in Year 13 and will include fish stranding as it relates to:

   a. Rate of river stage/total stage change,

   b. Cover, and

   c. Habitat stability (wetted history).

These recommendations are designed to build on the current dataset. The focus of study going forward should be on model refinements for stranding estimation and sampling consistency so comparisons with historical data can be maintained.

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.

Golder Associates Ltd.

Brad Hildebrand
Project Manager, Fisheries Biologist

Shawn Redden, RPBio
Project Director, Associate, Senior Scientist

BH/SR/cmc

https://golderassociates.sharepoint.com/sites/31732g/deliverables/working documents/year 11 report/final report/text/18107549-001-r-rev0-ddmmon-16 2017-2018 year 11 06dec_19.docx

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APPENDIX A

Project Maps and Sampling Chronology

APPENDIX B

Modelling Specifications and Code

Model Templates

Pool Density

```
.model {

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

bDischargeDensity ~ dnorm(0, 5^-2)

bDischargeDensity2 ~ dnorm(0, 5^-2)

sSiteDensity ~ dnorm(0, 2^-2) T(0,) for(j in 1:nSite){ bSiteDensity[j] ~ dnorm(0, sSiteDensity^-2) }

sReductionDensity ~ dnorm(0, 2^-2) T(0,) for(j in 1:nSite){ bReductionDensity[j] ~ dnorm(0, sReductio
nDensity^-2) }

sDispersion ~ dnorm(0, 2^-2) T(0,) for(i in 1:nObs){ log(eDensity[i]) <- bDensity + bDischargeDensity
* Discharge[i] + bDischargeDensity2 * Discharge[i]^2 + bSiteDensity[Site[i]] + bReductionDensity[Redu
ction[i]] eDispersion[i] ~ dgamma(sDispersion^-2, sDispersion^-2) NumPoolsPresent[i] ~ dpois(eDensity
[i] * eDispersion[i] * SiteArea[i]) } ..
```

Pool Stranding

```
.model {

bAbundance ~ dnorm(0, 5^-2)

bEfficiency ~ dnorm(0, 2^-2)

bAreaAbundance ~ dnorm(0, 5^-2)

bSeasonAbundance[1] <- 0 for(i in 2:nSeason){ bSeasonAbundance[i] ~ dnorm(0, 5^-2) }

sStudyYearAbundance ~ dnorm(0, 2^-2) T(0,) for(i in 1:nReduction){ bStudyYearAbundance[i] ~ dnorm(0,
sStudyYearAbundance^-2) }

sReductionAbundance ~ dnorm(0, 2^-2) T(0,) for(i in 1:nReduction){ bReductionAbundance[i] ~ dnorm(0,
sReductionAbundance^-2) }

sDispersion ~ dnorm(0, 2^-2) T(0,) for(i in 1:length(Reduction)){ log(eAbundance[i]) <- bAbundance +
bSeasonAbundance[Season[i]] + bAreaAbundance * log(Area[i]) + bStudyYearAbundance[StudyYear[i]] + bRe
ductionAbundance[Reduction[i]] eDispersion[i] ~ dgamma(sDispersion^-2, sDispersion^-2)

eAbundancePass1[i] ~ dpois(eAbundance[i] * eDispersion[i])

eAbundancePass[i, 1] <- eAbundancePass1[i]

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

Interstitial Stranding

```
.model {

bDensity ~ dnorm(0, 5^-2)

bSlopeDensity ~ dnorm(0, 5^-2)
```

```
sDensity ~ dnorm(0, 2^-2) T(0,) for(i in 1:length(Density)) { log(eDensity[i]) <- bDensity + bSlopeDe
nsity * Slope[i] Density[i] ~ dlnorm(log(eDensity[i]), sDensity^-2) } ..
```

RESULTS

Tables

Pool Density

Table 1. Parameter descriptions.

Parameter	Description
bDensity	Intercept for log(eDensity)
bDischargeDensity	Effect of Discharge on bDensity
bDischargeDensity2	Effect of Discharge^2 on bDensity
bReductionDensity[i]	Effect of i^{th} Reduction on bDensity
bSiteDensity[i]	Effect of i^{th} Site on bDensity
Discharge[i]	Initial discharge prior to i^{th} site visit
NumberPoolPresent[i]	Number of pools observed at the i^{th} site visit
sDispersion	SD of Overdispersion
SiteArea[i]	Area of the site exposed on the i^{th} site visit
sReductionDensity	SD of bReductionDensity
sSiteDensity	SD of bSiteDensity

Table 2. Model coefficients.

term	estimate	sd	zscore	lower	upper	pvalue
bDensity	2.1141465	0.2288052	9.195352	1.6411105	2.5474586	0.0007
bDischargeDensity	-0.2864772	0.0902326	-3.175150	-0.4643769	-0.1016464	0.0013

term	estimate	sd	zscore	lower	upper	pvalue
bDischargeDensity2	0.1491748	0.0669072	2.217452	0.0172563	0.2790029	0.0293
sDispersion	0.7536586	0.0601703	12.564966	0.6421933	0.8749424	0.0007
sReductionDensity	0.4382246	0.0995041	4.380769	0.2413295	0.6293942	0.0007
sSiteDensity	1.1710600	0.1987827	6.000444	0.8461868	1.6350677	0.0007

Table 3. Model summary.

n	K	nchains	niters	nthin	ess	rhat	converged
357	6	3	500	100	531	1.004	TRUE

Pool Stranding

Table 4. Parameter descriptions.

Parameter	Description
bIntercept	Intercept for log(eAbundance)
bReduction[i]	Effect of i^{th} ReductionEventID on bIntercept
bSeason[i]	Effect of i^{th} SeasonNum on bIntercept
eN[i]	Expected number of fish at i^{th} visit
eNPass[i,j]	Expected number of fish captured on j^{th} pass at i^{th} visit
eOverDispersion[i]	Expected overdispersion on i^{th} visit
p[i]	Capture efficiency for i^{th} SamplingGearNum
Pass[i,j]	Number of fish captured on j^{th} pass at i^{th} visit
sOverDispersion	SD of eOverDispersion
sReduction	SD of effect of bReduction

Rainbow Trout

Table 5. Model coefficients.

term	estimate	sd	zscore	lower	upper	pvalue
bAbundance	-0.7266535	0.4518690	-1.644794	-1.7176018	0.0715319	0.0720
bAreaAbundance	0.2596421	0.0479242	5.407446	0.1652264	0.3576636	0.0007
bEfficiency	-0.3487118	0.3127797	-1.170030	-0.9944946	0.1982566	0.2293
bSeasonAbundance2	1.9330797	0.2934127	6.633586	1.4241825	2.5421501	0.0007
sDispersion	2.3921145	0.1017908	23.499206	2.1996764	2.5987335	0.0007
sReductionAbundance	0.6978427	0.1568334	4.493323	0.4339524	1.0372336	0.0007
sStudyYearAbundance	0.8317994	0.3436766	2.579041	0.3766584	1.6995442	0.0007

Table 6. Model summary.

n	K	nchains	niters	nthin	ess	rhat	converged
1327	7	3	500	200	130	1.051	FALSE

Interstitial Stranding

Table 7. Parameter descriptions.

Parameter	Description
bDensity	Intercept for log(eDensity)
bSlopeDensity	Effect of Slope on bDensity
Density[i]	Density for i^{th} Slope (fish/ha)
eDensity[i]	Expected Density for i^{th} Slope

Parameter	Description
sDensity[i]	SD of residual variation in log(Density)
Slope[i]	Gradient for i^{th} slope (%)

Rainbow Trout

Table 8. Model coefficients.

term	estimate	sd	zscore	lower	upper	pvalue
bDensity	3.0356039	0.3652091	8.299318	2.2597486	3.8045491	0.0007
bSlopeDensity	-0.4302168	0.3839575	-1.159841	-1.2659609	0.3102709	0.1587
sDensity	0.5681133	0.4185178	1.639484	0.2616137	1.8395182	0.0007

Table 9. Model summary.

n	K	nchains	niters	nthin	ess	rhat	converged
5	3	3	500	10	1143	1.001	TRUE

APPENDIX C

Photographic Plates

GOLDER

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