Duncan Dam Project Water Use Plan

Adaptive Stranding Protocol Development Program

Implementation Year 10

Reference: DDMON-16

Lower Duncan River: Fish Stranding Impact Monitoring: Year 10

Study Period: April 2017 to September 2018

Golder Associates Ltd.
Castlegar, BC

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REPORT

DDMMON-16: Lower Duncan River

Lower Duncan River Fish Stranding Impact Monitoring: Year 10 Report (April 2017 to September 2018)

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Cover Photo: Upstream view of Site M2.7L, 23 May 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 program, initiated under the BC Hydro Water License Requirements (WLR) Program, includes the continuation of the DDMMON-16 Lower Duncan River Fish Stranding Impact Monitoring Program.

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 DDMMON-15 Lower Duncan River Stranding Protocol Development and Finalization Program. Based on collected data and the life history of species present in the system, DDM operations can increase the risk of stranding in certain seasons and during periods of longer wetted histories. Based on the data collected up to April 2018, 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 5% in most years, but possibly as high as 13%.

This report presents the results from Years 1 to 10 of the DDMMON-16 program, and the current status of management questions for DDMMON-16 is provided in Table EI below.

Table EI: DDMMON-16 Year 10: Status of Management Questions and Objectives.

DDMMON-16 Management Question	DDMMON-16 Specific Hypothesis	DDMMON-16 Year 10 (2017-2018) Status Summary
1) How effective are the operating measures implemented as part of the 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 DDMMON-15 Lower Duncan River Stranding Protocol Development and Finalization Program. - The relationship between wetted history and 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 10 (2017-2018) 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 amounts of stranding based on the amount of dewatered area and suitable 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, a significant site effect on the formation of pools (density) and pool stranding rates was not found. - The low interstitial stranding datapoints precluded the examination of the effect of site on interstitial stranding. - <u>Based on the current state of knowledge, hypothesis H01 cannot be rejected at this time but based on the initial study design, this hypothesis may be rejected in the future</u>. - The stranding rates at both index and random sites should continue to be analyzed as separate strata as the data set grows to allow for continued comparison with historical data.
	Ho_2: Fish populations in the lower Duncan River are not significantly impacted by fish stranding events.	- Estimates for the number of Rainbow Trout juveniles stranded in pools and interstitially were relatively low with high precision. - A seasonal effect on Rainbow Trout stranding was identified, with stranding rates approximately eight times higher in the fall in comparison to the winter season. At this point it cannot be determined whether this was due to lower densities in the system in the spring vs. the fall or to a decreased risk of stranding. - 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 pool stranded fish and slope of substrate were not found. - A relationship between slope and interstitially stranded fish was found, although it was not statistically significant. - <u>Based on the current dataset, study hypothesis H02 is not rejected for Rainbow Trout and 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 its 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 includes 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 for BC Hydro (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). Although ordered in the water licence, 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 DDMMON-16: Lower Duncan River Fish Stranding Impact Monitoring Program (FSIMP). In conjunction with other assessment tools being developed during the monitoring period, the FSIMP 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 10 (2017 - 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 the LDR Hydraulic Model (DDMMON-3) and other interrelated studies (DDMMON-1 – Lower Duncan River Ramping Rate Monitoring, DDMMON-2 – Lower Duncan River Habitat Use

Monitoring, DDMMON-4 - Lower Duncan River Kokanee Spawning Monitoring, and DDMMON-15 – Lower Duncan River Stranding Protocol Review) 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 DDMMON-1 – Gap Analysis for Lower Duncan River Ramping Program (Irvine and Schmidt 2009 and 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. The following document provides information on abundance estimation and fish stranding observed for these species, over all assessed flow reductions in Year 10 of this Program (15 April 2017 to 14 April 2018). In addition, because of the timing of this report two additional assessed flow reductions that occurred in Year 11 (25 and 26 September 2018) were available for inclusion in the current dataset. The inclusion of all available data strengthened the analysis conducted resulting in greatly reduced uncertainty for interstitial stranding estimation provided in this report. 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).

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:

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

The specific management questions associated with this monitoring program are:

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 the FSIMP are:

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 future. The Terms of Reference did not state specific hypotheses to address primary 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 the second primary objective, the TOR stated two hypotheses that the FSIMP 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:

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 the FSIMP worked toward addressing primary objective 1) the effectiveness of operating measures, and addressing Hypothesis Ho_1, fish stranding at index sites is representative of overall stranding (Golder 2009b and 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.

The second objective, 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) and the present study year (Year 10: 2017 – 2018) of the FSIMP. Recommendations to refine study methodology and to better address both objectives and hypotheses in future years of the FSIMP have been developed (Section 6.0).

1.4 Study Design and Rationale

Since 2002, Golder has conducted fish stranding assessments on the LDR. A wide variety of fish capture/observation techniques have been utilized to ensure the study design in each sample year met BC Hydro's objectives. Recommendations made in Years 3 to 9 (2010 – 2017) on changes to study design to address gaps in the data set identified during the data analysis (Golder 2011, Golder and Poisson 2012, Golder 2014, Golder 2015, Golder 2017a and 2017b, Golder 2018) were implemented in the present study year.

As part of the DDMMON-15 program, a workshop was held on 14 January 2016, which included the 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 the DDMMON-16 program in the previous three years (Years 8 to 10) of implementation.

1.4.1 Stranding Site Selection

Prior to study Year 4, fish stranding assessments focused on index sites, as these sites have the largest dewatered areas during flow reductions, and are also believed to strand the highest numbers of fish. Due to this focused methodology, limited assessments of non-index sites were conducted and therefore in-depth statistical analysis of stranding rates at both index and non-index sites were unable to be conducted. 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 on was increased.

As discussed in the DDMMON-15 workshop, in order to move towards a long-term monitoring program, changes were made to the site selection process in the current study year. With the analysis of the Year 7 data set, Ho_1: (*Fish stranding observed at index sites along the lower Duncan River floodplain is representative of overall stranding*) was not rejected. Therefore, in 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 that focused on pools in the previous study has been refocused since 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 data set. 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 future estimates of fish stranding in the LDR.

1.4.3 Interstitial Sampling

During data analysis in Year 3, estimates of both interstitial stranding per unit area (m^2) and total interstitial stranding, 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 time crews spent conducting interstitial sampling (See Section 2.6.2.3).

1.4.4 Abundance Estimates

Between 2013 and 2016 fall fish abundance assessments that were consistent with the DDMMON-2 – Lower Duncan River Habitat Use Monitoring (Thorley et al. 2011 and 2012) were conducted as part of DDMMON-16. After discussions with the BC Hydro contract authority and technical advisors (Darin Nishi, James Baxter and Alf Leake, pers comm.), the fall fish abundance sampling component was discontinued in 2017, which increased component budgets to assess additional reduction events and analyses. To obtain abundance estimations for the target species that could be compared to total stranding estimates, the spring age-1 Rainbow Trout abundance, which were estimated as part of the Gerrard Rainbow Trout Stock Productivity study (Andrusak 2017), were used.

1.4.5 Lower Duncan River Fish Stranding Database

The first step to shifting the DDMMON-16 program scope to meet the goals of the DDMMON-15 workshop was to modify the Lower Duncan River Fish Stranding Database. 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 CLBMON-42 Lower Columbia River Fish Stranding Program. This will allow for more informed fish salvages in the future years of this program.

1.4.6 Data Analysis

The modelling used in Year 9 (Golder 2018) of this program was updated to incorporate the current year's data set, to further refine the slope classification when analyzing as a variable related to stranding rates, and to analyze substrate size as a variable related to interstitial stranding.

2.0 METHODS

2.1 Study Area

The geographic scope of the study area for the FSIMP was 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 and Golder 2006, 2008, 2009b, 2010, 2011, 2014, 2015, 2016, 2017a, 2017b, and 2018; Golder and Poisson 2012). These stranding sites included 11 index stranding assessment sites and 39 non-index sites (Appendix A, Figures 1 to 7). The remaining habitats outside of the identified sites usually have steep banks with extreme gradients. Habitats with these characteristics have very low stranding risk. Consequently, additional major fish stranding locations outside of the 50 potential fish stranding sites used in this study, are unlikely to occur.

2.2 Study Period

In Year 3 (2010 – 2011), the study period for all study years was set between April 15 of that year, and continued until the following 14 April. Stranding assessment activities in the present study year are within the same study period, with specific stranding assessments conducted between 23 May 2017 and 27 March 2018, during planned flow reductions at DDM. Two additional stranding assessments from Year 11 (25 and 26 September 2018) were able to be included in the dataset. Each assessed reduction from DDM was assigned a reduction event number (RE; see Section 2.6) and Figure 2 outlines all assessment activities included in this report.
An in-depth summary of the chronology of sampling and project milestones in all study years is provided in Appendix A, Tables A1 to A9.

Table 1: Sampling activities for the 2017-2018 Lower Duncan River Fish Stranding Impact Monitoring, Year 10 and Year 11 Program.

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

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 below the Lardeau River Water Survey of Canada gauging station (DRL) 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 an alcohol handheld 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

Hierarchical Bayesian Models (HBMs) were used to estimate total spring abundance of age-1 Rainbow Trout. In the Bayesian implementation of the model, fish abundance was assumed to be Poisson-distributed, with a mean expected density drawn from a log-normal distribution. Observed fish counts were assumed to be binomially distributed, with estimated fish abundance as the number of trials and observer efficiency as probability of success. The significance of model parameters was determined based on whether the parameters' 95% CRI overlapped zero. Since the first level of each factor effect (depth and year) was set to zero, if a parameter's 95% CRI overlapped zero, it suggested that there was no significant difference between that parameter and the first level of the factor.

Observer efficiency, derived from previous work on Rainbow Trout and Mountain Whitefish in the LDR (Thorley et al. 2012), was used to estimate total fish abundance at each site from the number of observed fish. 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 an 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 river must occur by boat or on foot. Boat launches exist 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. At the time this document was created, a log jam in the mainstem LDR at RKm 4.7 could not be navigated at any discharge level. However, the downstream portions of the river can be accessed at all river elevations by boat through a side channel located at RKm 4.5 and flows into Meadow Creek near its outlet into the LDR. As the river nears the mouth to Kootenay Lake, the channel meanders on a yearly basis, and access to the LDR from Kootenay Lake is difficult at lower DRL discharges and lake elevations.

2.6.1 Stranding Site Selection

Prior to each fish stranding assessment, 10 sites were randomly selected from all identified stranding sites. In previous 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.

In study Years 8 to 10, 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 the site specific area regression that was completed in Year 3 (Golder 2011).

2.6.2 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 up to 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 from the following list:

- 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 studies (fish stranding assessments and ramping experiments), 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 field crews were unable to accurately determine the areal extent of the pools at time of isolation from the mainstem river.

2.6.2.3 Interstitial Sampling

In Year 10, to assess interstitial stranding at each surveyed site, randomized transect sampling was conducted when conditions on site would allow it. A maximum of 5 transects were conducted at each site. A measuring tape was laid on the substrate from the wetted edge to the top of the dewatered area, and the length recorded. The substrate near the tape was then visually assessed (0.5 m on either side of the tape along its entire length).

For the two assessments included in this report from Year 11, randomly selected dewatered habitat with consistent habitat characteristics (i.e., substrate size and slope) within a site was censused by field crews. Consistent effort (i.e., a maximum of approximately twenty minutes) was expended at each site to ensure an adequate number of sites along the entire lower Duncan River were sampled during each assessment. The main objective of this approach is 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 there was not sufficient dewatered area, or the substrate was too large to effectively conduct these methods, dewatered habitat at each site was assessed by conducting a minimum of twenty randomly placed interstitial grids (0.5 m^2). The substrate and all cover were removed from each grid and the stranded fish enumerated. To be consistent with past studies (fish stranding assessments and ramping experiments), the dominant substrate in each transect and/or grid were 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
- Length (mm; total or fork length measured was dependant on if species examined had a forked caudal fin)
- 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 - 50) of each species from the salvaged fish were measured for length, and the species recorded.

2.6.3 Data Analysis

2.6.3.1 Dewatered Area

To compare pre- and post-WUP operations, the Year 9 DDM and DRL flow data were added to the discharge data set. The calculations conducted in Year 4 (Golder and Poisson 2012) were then repeated with the updated data set. 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 habitat 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 estimate the slope of the active streambed at different discharges; a GIS water inundation model of the river was created using the DDMMON 3 Digital Elevation Model and a three-dimensional plane. The plane was inclined and distorted to the gradient of the river. Field observations were used to improve the real-world accuracy of the plane. A total of 10 discharges between the highest and lowest encountered during previous stranding assessments were selected for input into the GIS model.

Discharges were correlated to elevation data using a DRL stage curve provided by BC Hydro. Inputting the 10 elevations into the inundation model allowed for estimation of the area of streambed within a series of percent slope categories (i.e., 0-2%, 2-4%, 4-6%, 6-8%, >8%) that were inside of the wetted area at each inputted discharge rate. This data was used during the extrapolation of pool and interstitial stranding rates over the entire study area.

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 include:

- 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 include:

- 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.

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 include:

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

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 study Year 10. Presently, 91 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 m³/s (1705.7 ft^3/s) on 21 April 2018 to 527.2 m³/s (18617.9 ft^3/s) on 16 May 2018. Hourly discharge from DDM ranged from 0.8 m³/s (28.3 ft^3/s) on 5 July 2017, to 280.4 m³/s (9902.2 ft^3/s) on 16 May 2018 (Figure 2).

Lowest DDM flows typically occur during the spring/summer recharge of Duncan Reservoir. 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, the Lardeau River discharge is typically high, which satisfies flow requirements for the protection of fish and the maintenance of available habitat.

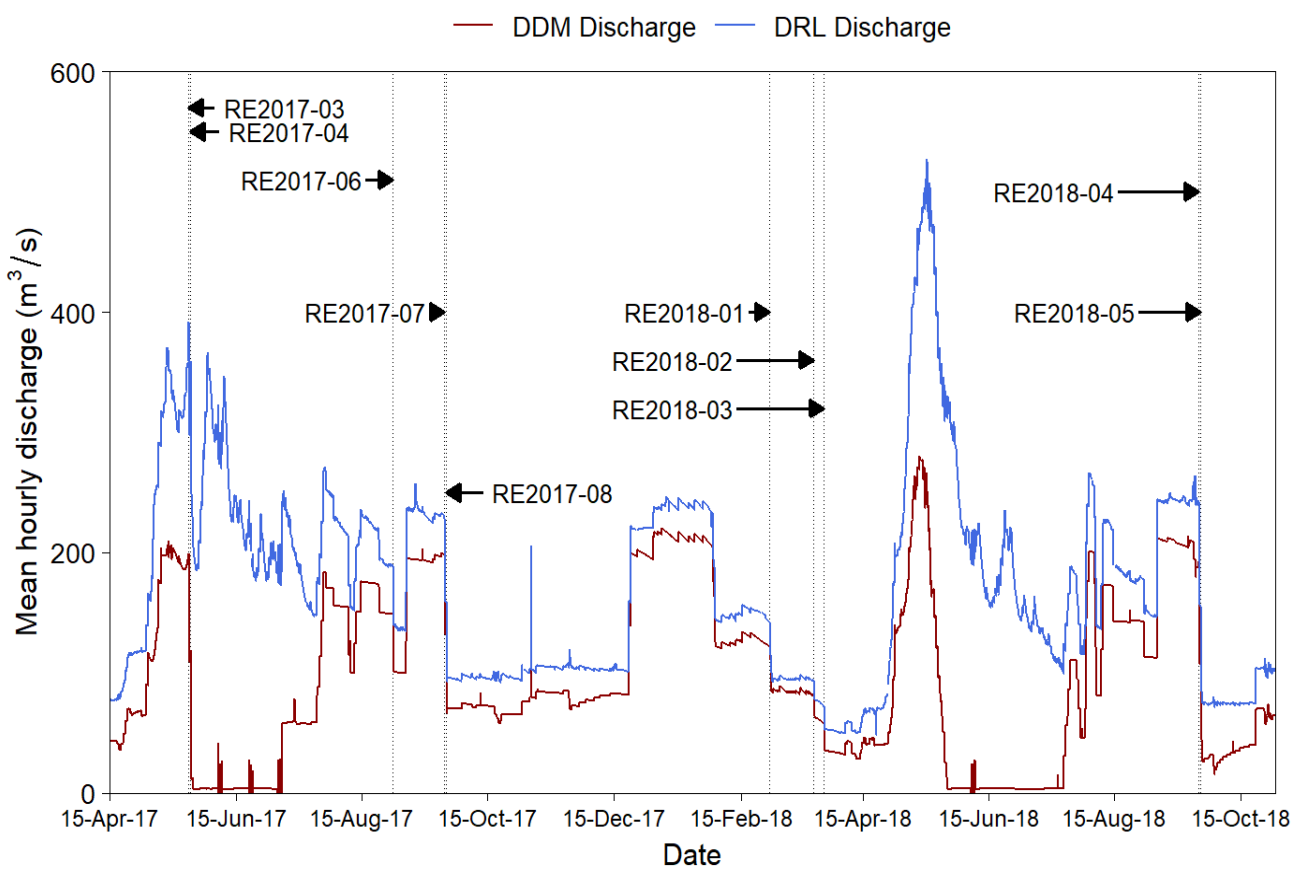

Figure 2: Hourly discharge at the Duncan Dam (DDM, red line) and at the lower Duncan River below the Lardeau River (DRL, blue line) from 15 April 2017 to 01 November 2018. Vertical dotted lines represent the timing of fish stranding assessments.

During the present study years, ten reduction events occurred at DDM (Figure 2 and Table 2). During the reduction events, DDM daily decreases of discharge ranged between 16 and 98 m^3/s (565 and 3461 ft^3/s; 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 23 May 2017 to 26 September 2018, for events when fish stranding assessments were conducted.

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

[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. Therefore, results from assessments prior to 15 September 2006 have been excluded from the dataset. Stranding assessments were conducted following ten flow reductions during the present study years. All fish encountered during the assessments have been split into sportfish and non-sportfish categories for analysis. The scientific names of all species in these categories are presented in Table 3.

Table 3: Scientific names of fish species encountered during fish stranding assessments on the lower Duncan River, September 2006 to April 2018.

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 Chub	*Mylocheilus caurinus*	PCC

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

Within the dataset analyzed, 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 the study years 5 to 10 (2012 – 2013 to 2017 – 2018). The number of pools sampled in Years 10 and 11 was also reduced to allow for more intensive interstitial sampling effort. This resulted in the sampling of 76 pools and the most number of interstitial transects (n = 236) assessed during Year 10. For the reductions included from Year 11, 19 pools and 34 censused interstitial areas were assessed (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 of each study year included in the present analysis.

DDMMON-16	Number Assessed	Number Sampled

Study Year	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)	2	11	6	19	0	0	34

Figure 3: Locations and slope (%) of sampled stranding mechanisms (September 2006 to September 2018) Reaches 1 to 3.

Figure 4: Locations and slope (%) of sampled stranding mechanisms (September 2006 to September 2018) Reaches 4 and 5.

In Year 10, a total of 392 fish were observed, representing 14 species, of which five were sportfish and nine were non-sportfish species. In addition, seven unidentified fish were recorded (Table 5). This total is the third lowest documented since 2006, higher only than the 2006-2007 and 2015-2016 sampling years. Juvenile Rainbow Trout encounters (*n* = 122) were the most abundant sportfish observed (31.1% of the total catch). In previous years, Rainbow Trout juveniles accounted for 8.6% to 58.4% of the total fish counts. Mountain Whitefish juveniles (*n* = 31) were the next most abundant sportfish, accounting for 7.9% of the total number of fish encountered. Kokanee young-of-the-year and juveniles accounted for 3.3% (*n* = 13) of the catch when combined (Table 5; Figure 5). The most common non-sportfish taxa identified to species were Slimy Sculpin and Longnose Dace, accounting for 19.6% and 13.5% of the total number of observed fish, respectively.

For the two reductions included from Year 11, a total of 657 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* = 356) were the most abundant sportfish observed (54.2% of the total catch). In previous years, Rainbow Trout juveniles accounted for 8.6% to 58.4% of the total fish counts. The other two sportfish species, Mountain Whitefish and Burbot, both had a single juvenile fish recorded as stranded, accounting for a total of 0.4% (both species combined (Table 5; Figure 5). The most common non-sportfish taxa identified to species were Longnose Dace, Slimy Sculpin, and Redside Shiner, accounting for 14.5%, 0.9%, and 0.9% of the total number of observed fish, respectively.

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 September 2018.

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)	356 (54.2)
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)	1 (0.2)
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
	Y-O-Y	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 (0.2)
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 (14.5)
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)	6 (0.9)
Torrent Sculpin		0	1 (0)	1 (0.1)	0	0	3 (0.2)	0	0	0	0	4 (0.2)	1 (0.3)	1 (0.2)
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)	181 (27.5)
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 Chub		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**	**657**

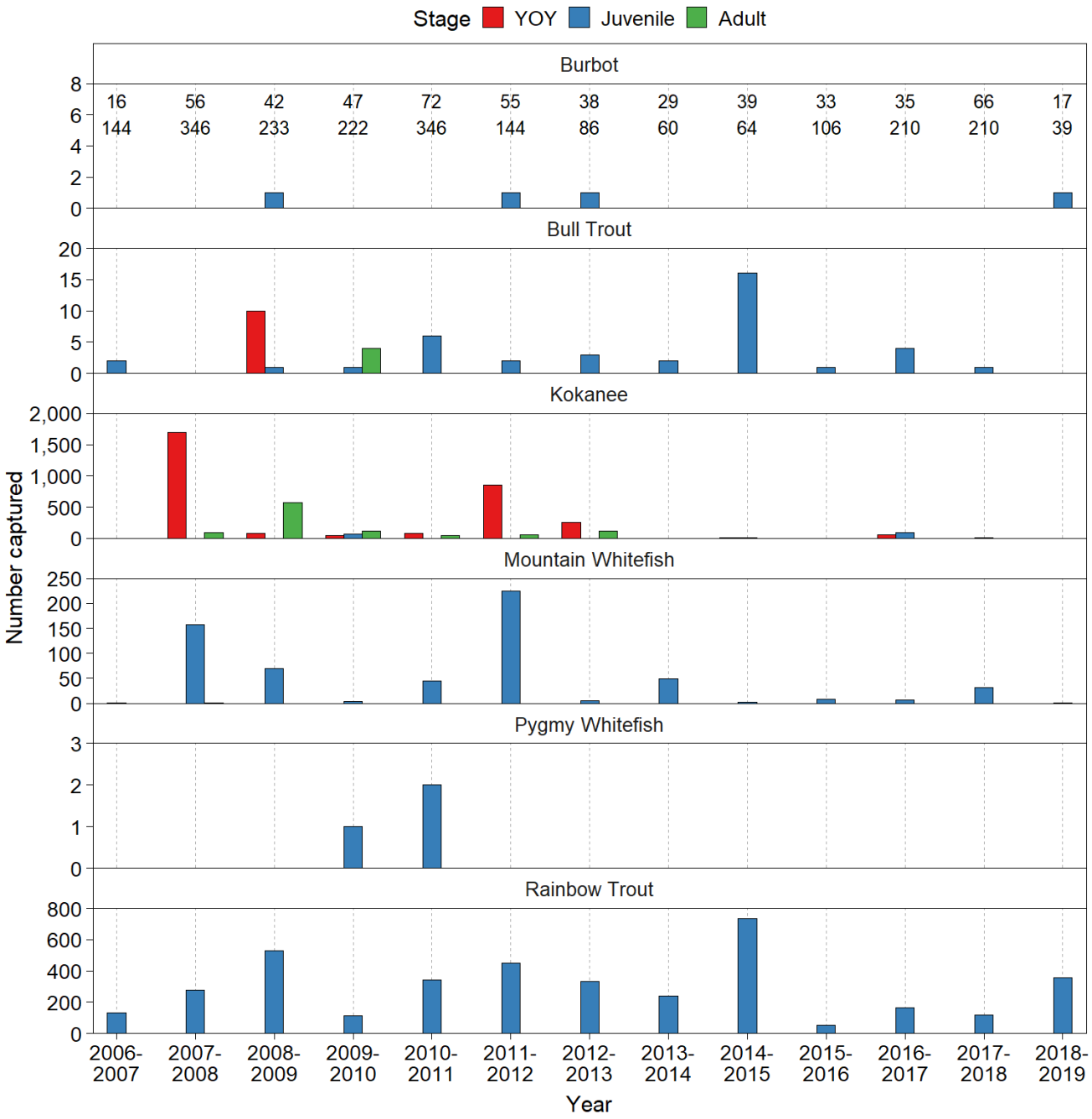

Figure 5: Abundances of sportfish species, separated by life stage, observed in stranding assessments between 2006 and 2019. Note the different y-axis scales among panels. On the uppermost panel, the numbers of sampled sites and pools are provided in the first and second lines, respectively.

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 0.30 km^2, respectively; Figure 7). The difference was statistically significant (1-way ANOVA; P=0.003). Annually, mean exposed areas in reported reductions ranged from 0.2 km^2 (2015 stranding year) to 0.6 km^2 (2005 stranding years).

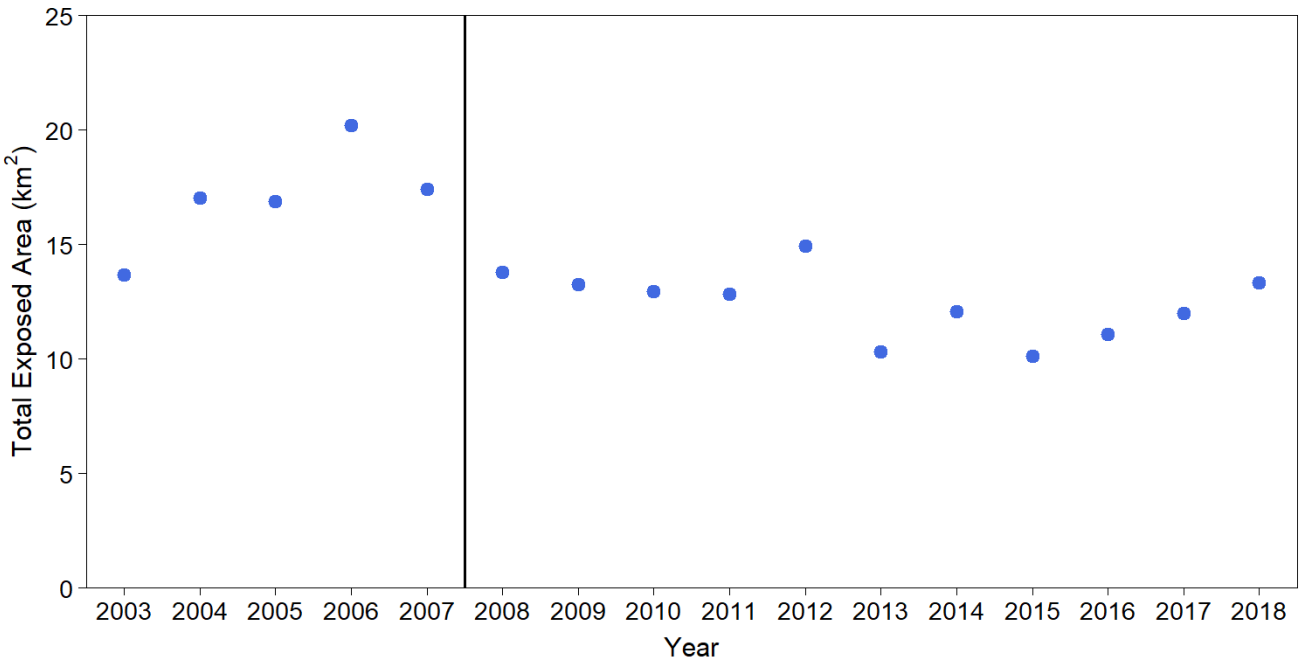

Figure 6: Total area exposed by all annual reductions in the LDR by year of operations, calculated using DRL discharge. The vertical line denotes the beginning on WUP flows in 2008. Note that label on Y axis denotes study year not calendar year.

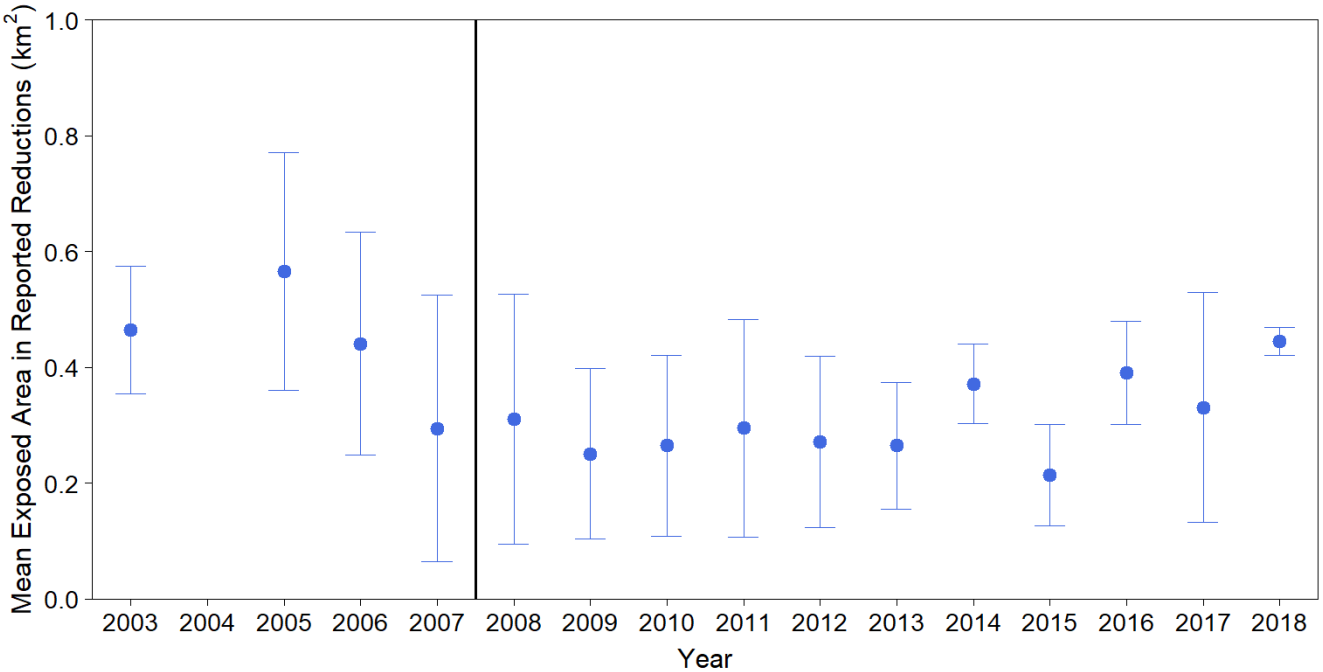

Figure 7: Mean area exposed by all annual reductions in the LDR by year of operations, calculated using DRL discharge. Bars represent 1 standard deviation. The vertical line denotes the beginning on WUP flows in 2008. Label on y axis denotes study year, not calendar year.

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.

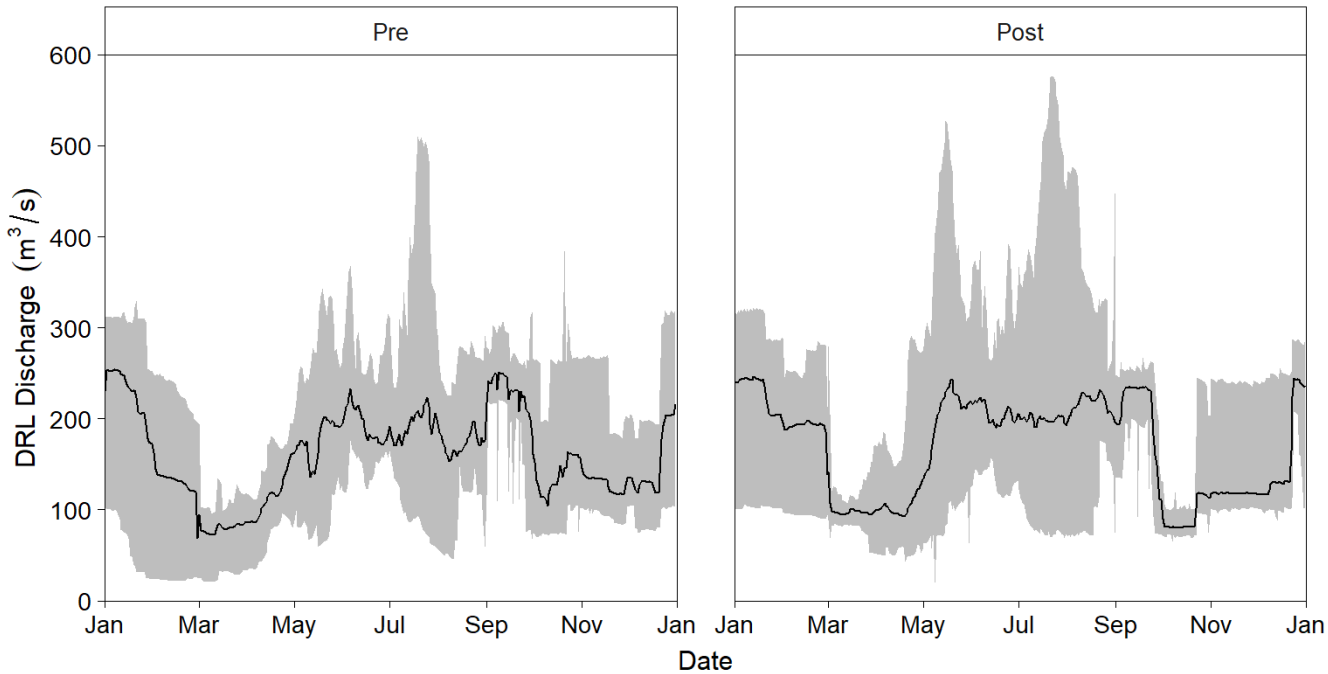

Figure 8: Minimum, maximum (grey ribbon) and mean (black line) discharge as measured at the WSC DRL gauge in the LDR by month during pre-WUP operations (2002 - 2007) and post-WUP operational implementation (2008 - 2018).

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.

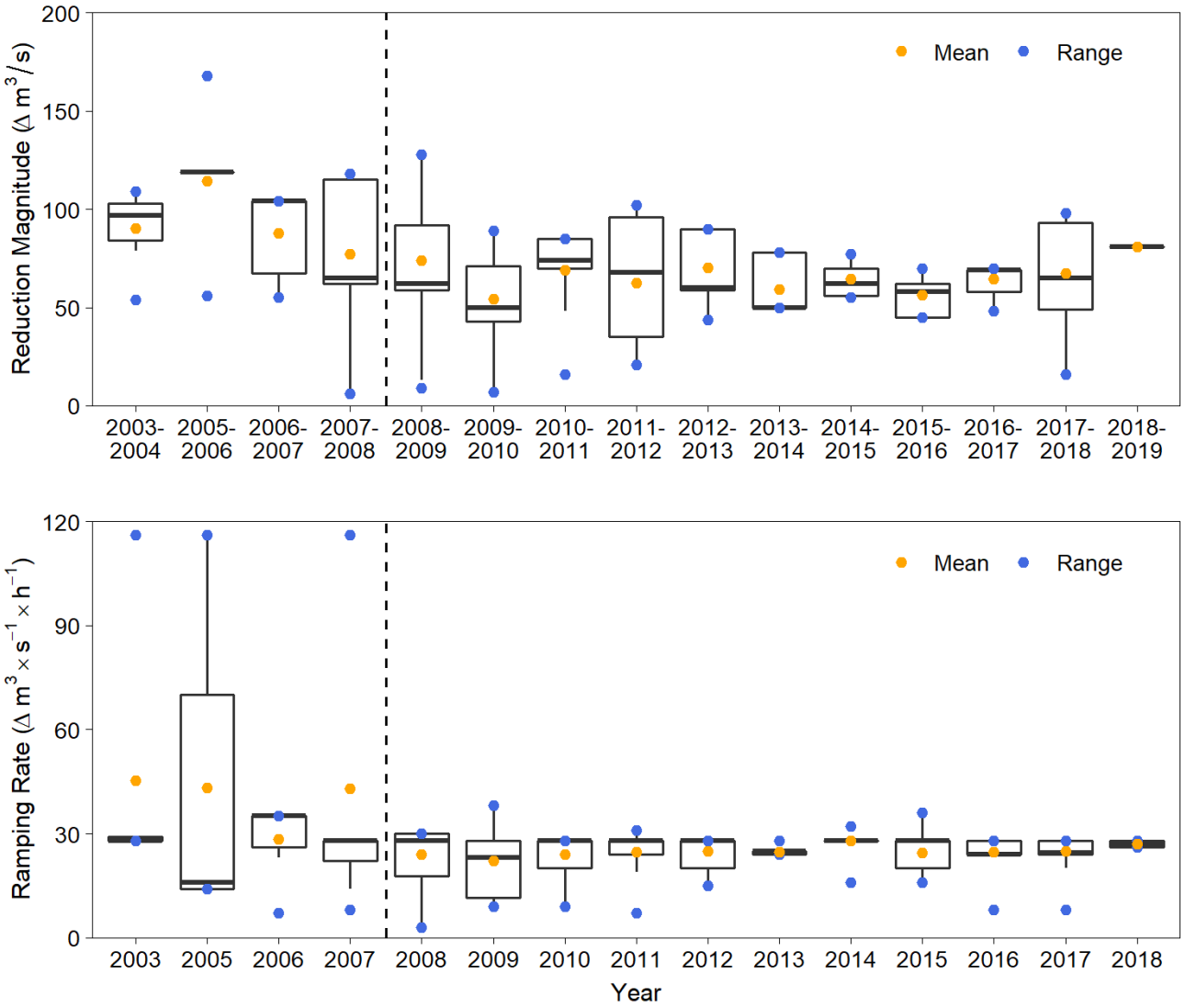

Figure 9: Boxplots of reduction magnitude (Δm³/s; top panel) and ramping rates (Δm³ s^{-1} h^{-1}; bottom panel) by year. Each box represents the 25th and 75th quantiles (bottom and top lines, respectively), and the median (middle bold line); whiskers extend to 1.5 times the interquartile distance. Yearly 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 5,287 in 2016 to 29,351 in 2014 (Table 6 and Figure 10: Golder 2018). Overall, fall estimates decreased annually since the 2014 peak of estimated abundance. Mountain Whitefish fall abundance in 2016 was similar to the 2015 estimates. Generally, Mountain Whitefish fall abundance decreased from approximately 20,000 in Year 6 and Year 7 to approximately 9,000 in Year 8 and Year 9 (Table 6 and Figure 10: Golder 2018).

Year 10 spring age-1 total abundance for Rainbow Trout was estimated at approximately 29,000, the highest since 2013 (Table 6 and Figure 10). Overall, spring estimates decrease between 2013 and 2015, and then increased annually in 2016 and 2017 (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 2013 and 2015, and were lower in 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	
	RB	MW	RB	MW
Year 6 (2013)	14,854 (7,400 – 27,386)	20,038 (10,061 – 39,369)	23,200 (16,200 – 33,900)	-
Year 7 (2014)	29,351 (17,531 – 48,187)	18,632 (10,409 – 31,827)	-	-
Year 8 (2015)	10,456 (5,871 – 18,171)	8,781 (4,745 – 15,535)	9,200 (6,200 – 13,700)	-
Year 9 (2016)	5,287 (3,184 – 8,700)	9,008 (5,345 – 14,635)	16,900 (11,800 – 24,800)	-
Year 10 (2017)	-	-	29,000 (19,700 – 42,600)	-

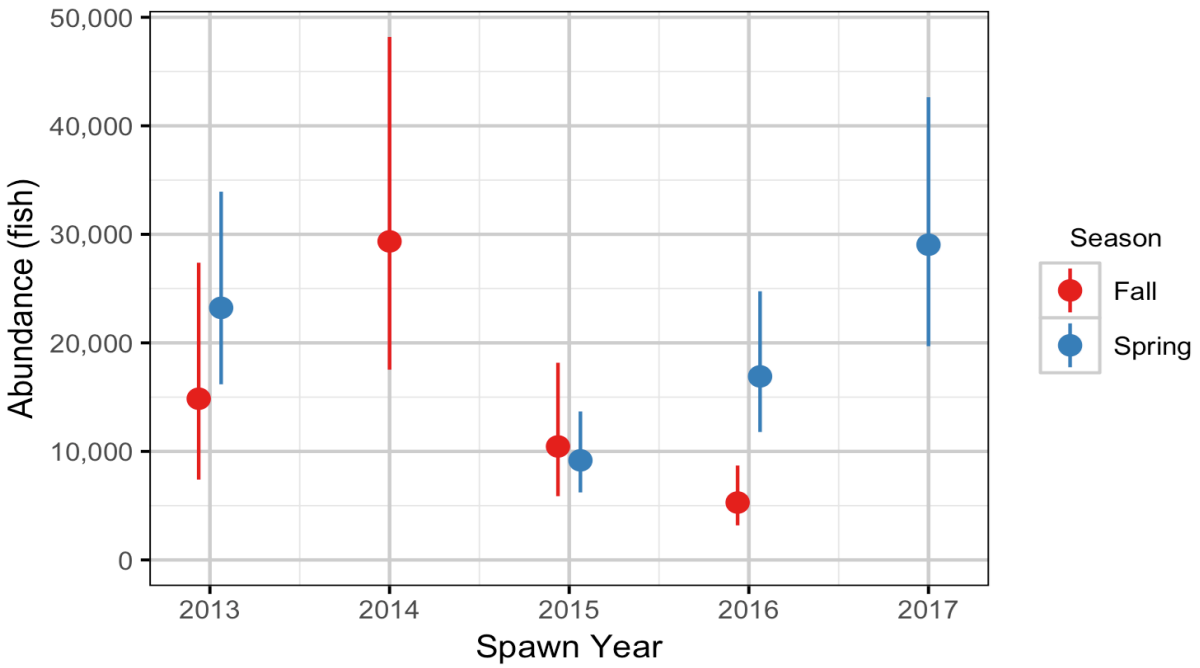

Figure 10: Estimated abundance of target species by spawn year and season in the Lower Duncan River (with 95% CIs).

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 on juvenile Mountain Whitefish and will likely not result in population level effects (Golder 2018), 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: 2010-2018) was calculated using the elevation models for the area. Slopes ranged from 0% to 60%, however all values above 20% (a total of 6 cases) were deemed artefacts of the elevation model and were removed from analysis. Generally, pool density was slightly higher at lower slope values (0-5%); 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 (e.g., 2010, 2016, 2017), the majority of recorded pool densities were low, often lower than those recorded at index sites (Figure 11).

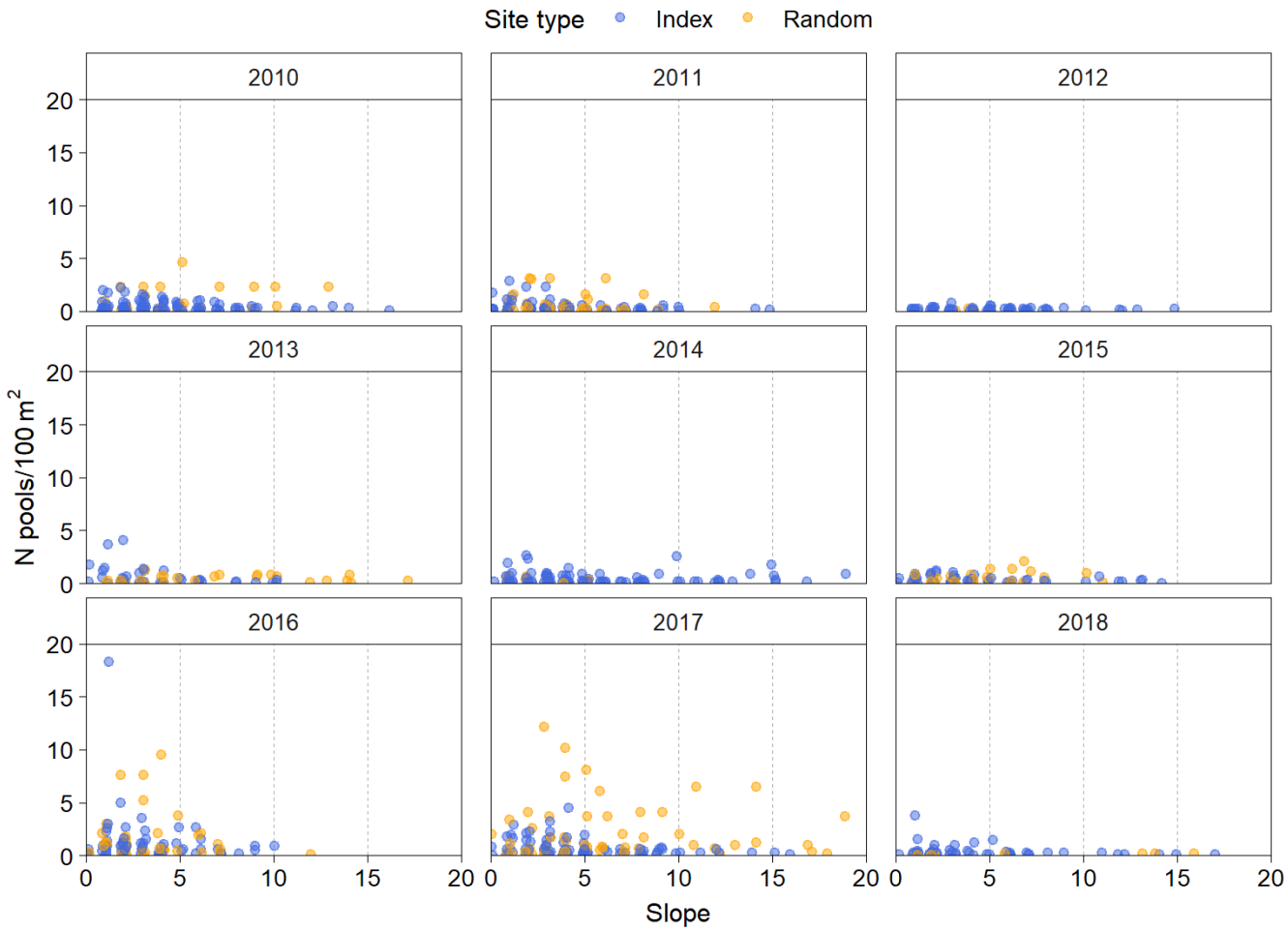

Figure 11: Density of pools recorded per reduction vs. habitat slope as a continuous variable, 2010-2018.

The number of pools per assessed flow reduction was estimated to allow the number of fish stranded per reduction (Section 3.5.2) to be calculated. 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 the stranding surveys between 2010 and 2017 was generally similar.
In 2018, the reduction-level estimates of pools were more variable between seasons, but not statistically different (Figure 12).

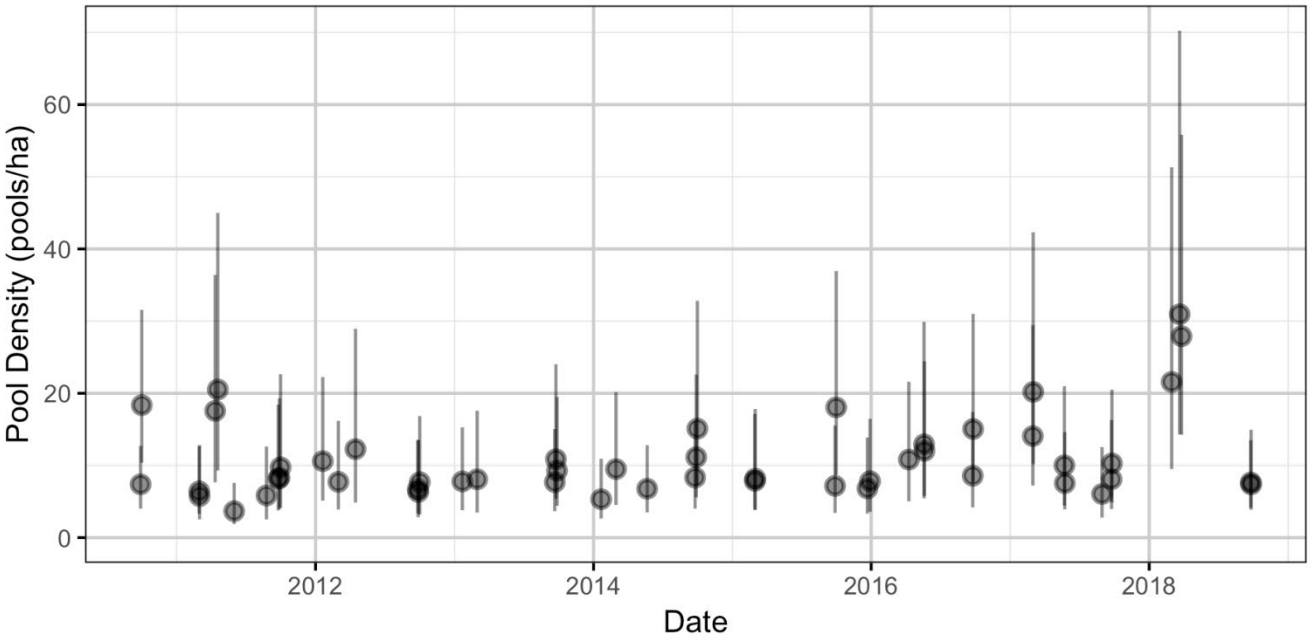

Figure 12: Estimates of pool densities by reduction event and date in the Lower Duncan River. Error bars are 95% credibility intervals.

3.5.2 Pool Stranding

The number of fish stranded per pool was similar throughout the different slopes (Figure 13). This indicated that slope did not affect stranding of fish in pools.

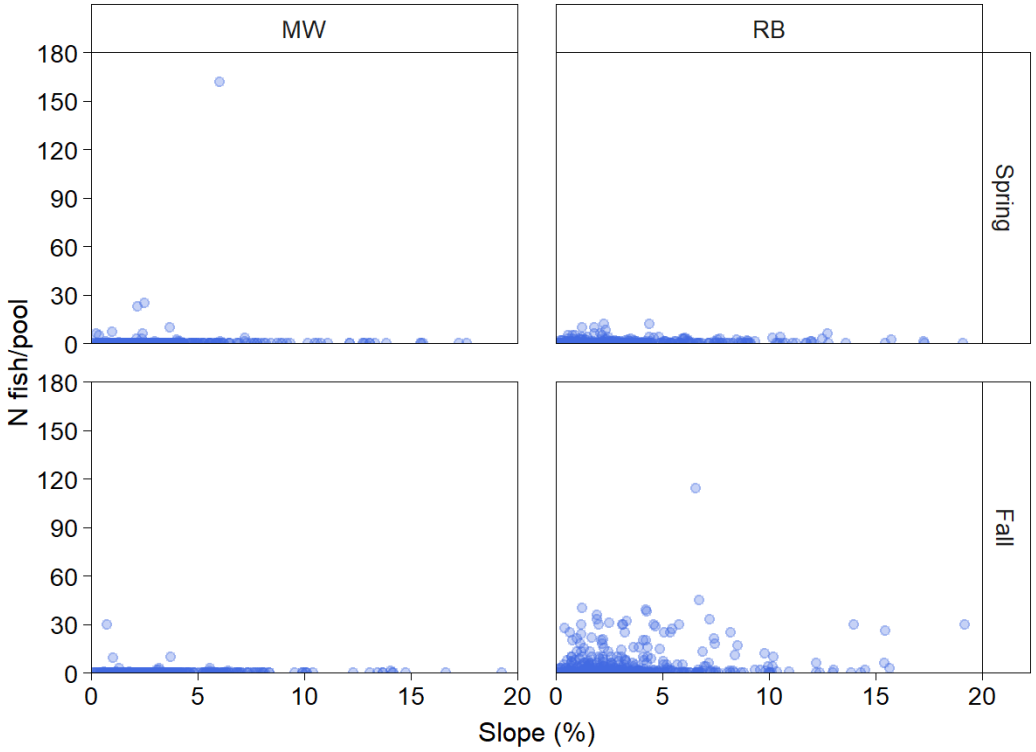

Figure 13: Number of collected fish per pool, plotted by slope and colour-coded by season, 2010 – 2018.

The density of pool-stranded fish differed by dominant substrate size and by species (Figure 14). 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 between sand and very large gravel (Figure 14).

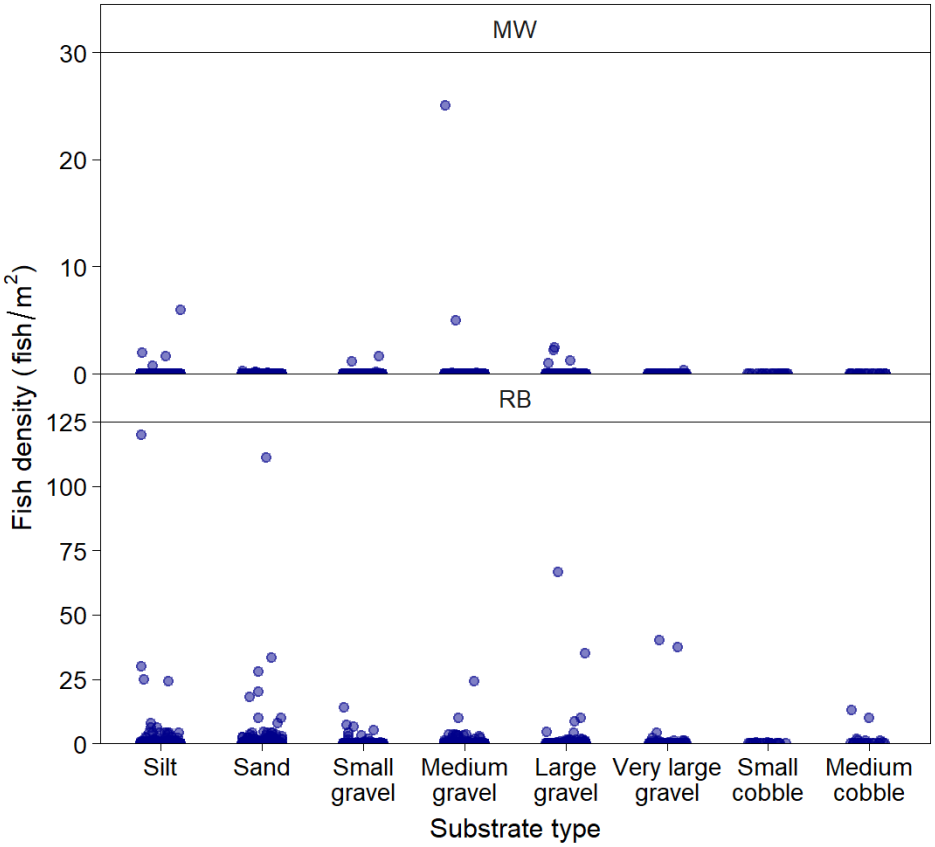

Figure 14: Scatter plot of pool-stranded fish density (fish/m²) vs. dominant pool substrate size, 2006-2018, plotted by species.

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 seven times higher than those for winter/spring (Figure 15). The median number of Rainbow Trout fry per pool for the spring season (January to June) was estimated to be 0.48 (CRI of 0.17 – 1.07) fish/pool (Figure 15). In contrast, the median number of Rainbow Trout juveniles stranded per pool in the fall (July to December) was estimated at 3.39 (CRI of 1.30 – 8.09).

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 16). Generally, spring Rainbow Trout estimates of pool stranding were lower than fall estimates. Fall pool estimates were highest between 2011 and 2014 (Figure 16).

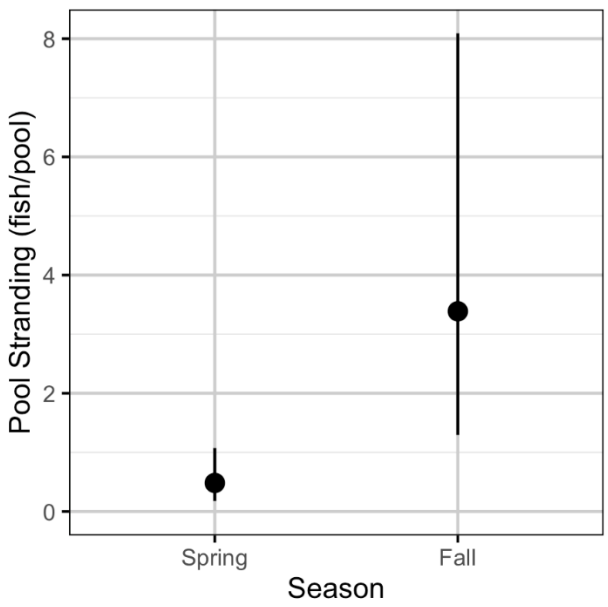

Figure 15: The expected pool stranding in an average pool in a typical reduction event by season in the Lower Duncan River. Error bars are 95% credibility intervals.

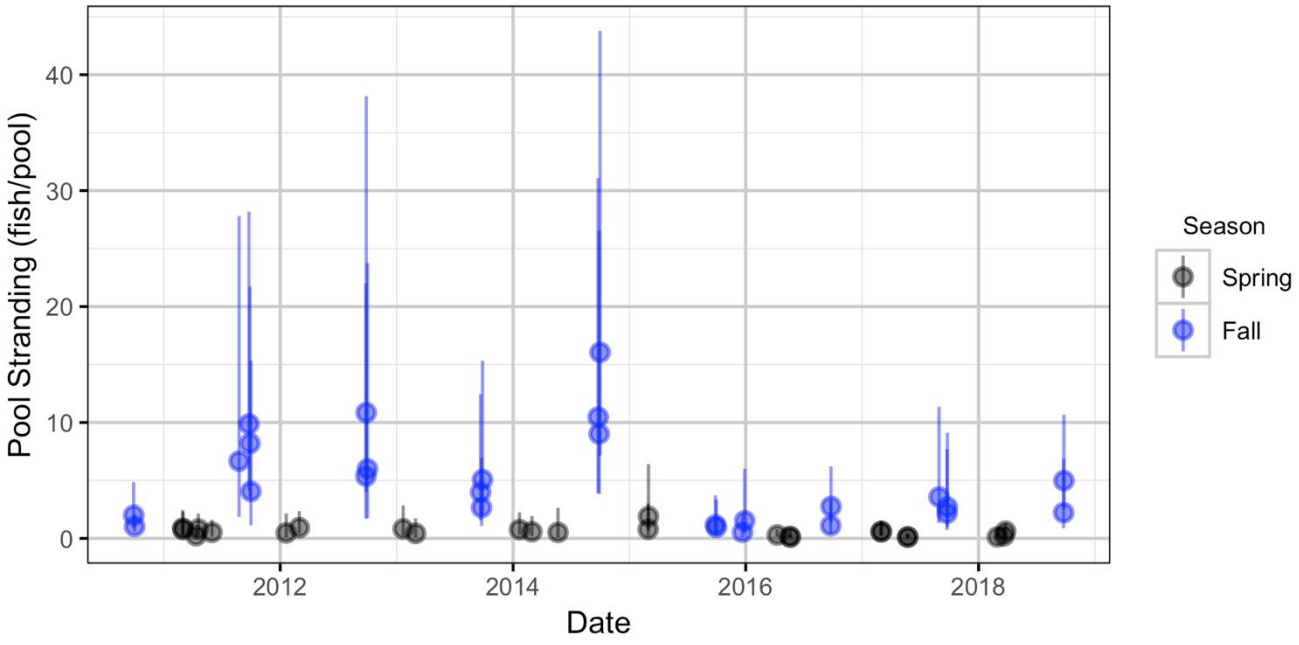

Figure 16: Estimates of Rainbow Trout per pool by date and season in the Lower Duncan River. Error bars are 95% credibility intervals.

3.5.3 Interstitial Stranding

Between 2011-2012 and 2018), 31 Rainbow Trout and 2 Mountain Whitefish were found to be interstitially stranded on substrates ranging from silt to large gravel in composition (Figure 17). Over the previous five study years when interstitial sample methodology was standardized with transect sampling, only one interstitially stranded Rainbow Trout was observed (in Year 6; Golder 2015). In Fall 2018, seven Rainbow Trout were recorded as interstitially stranded. All documented interstitially stranded fish were found on exposed areas with low slopes (≤7%; Figure 18). As slope increases, the risk of interstitial stranding was found to decrease (Figure 19).

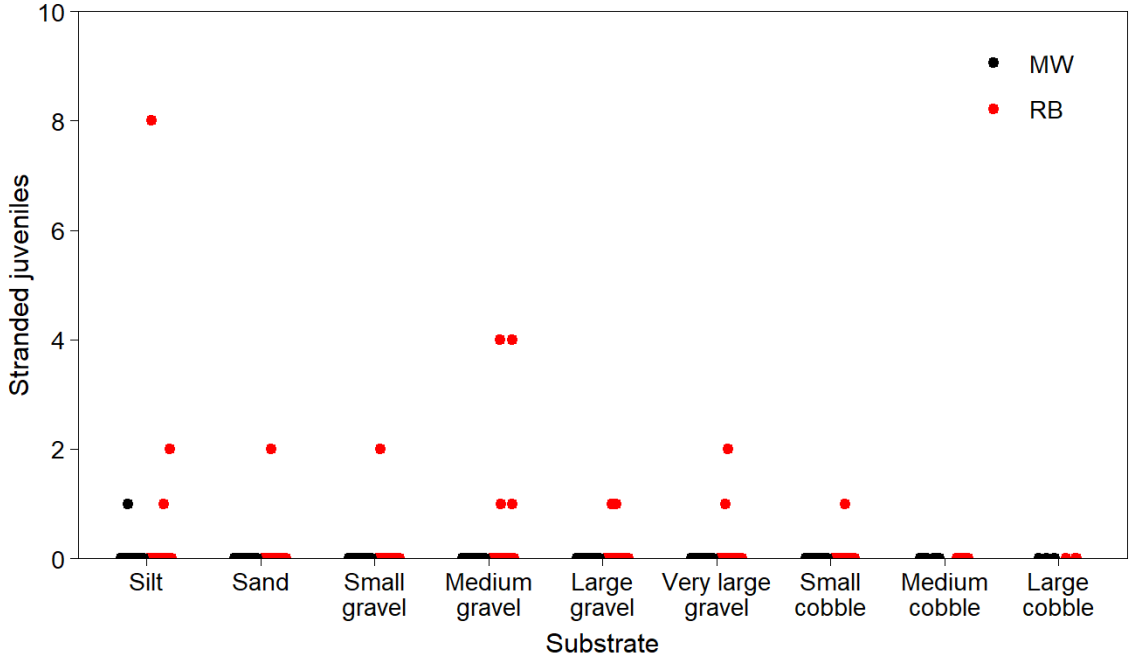

Figure 17: Counts of 2011-2018 interstitially stranded Mountain Whitefish and Rainbow Trout in the Lower Duncan River, plotted by substrate size.

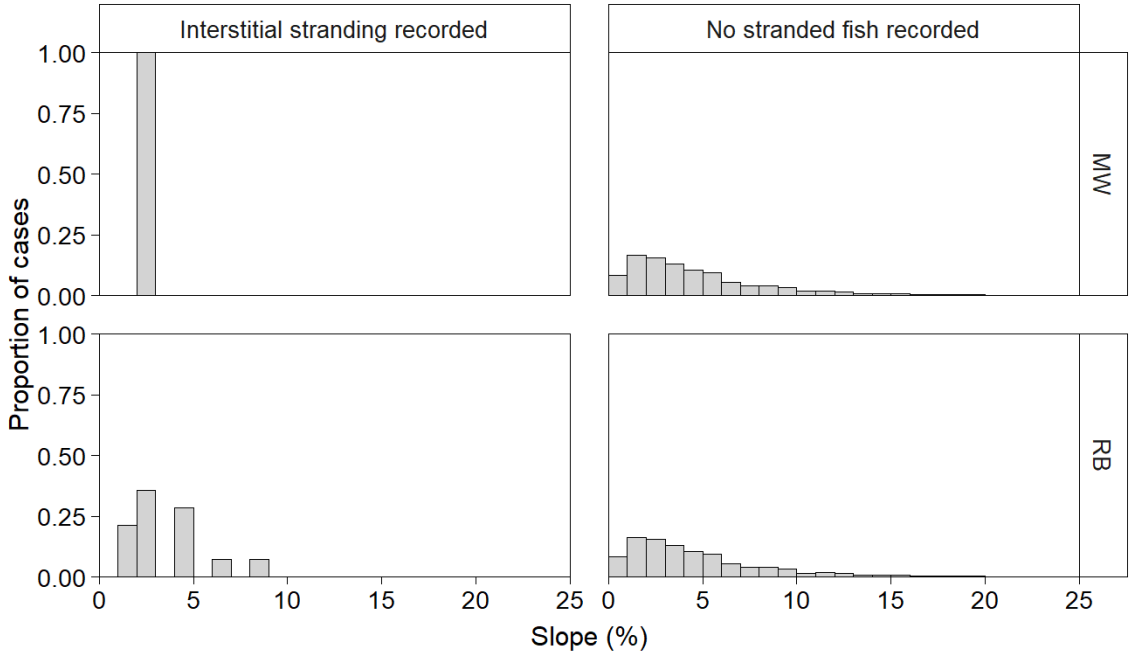

Figure 18: Histogram of 2011-2017 interstitially stranded Mountain Whitefish and Rainbow Trout in the Lower Duncan River, plotted by species and slope (%).

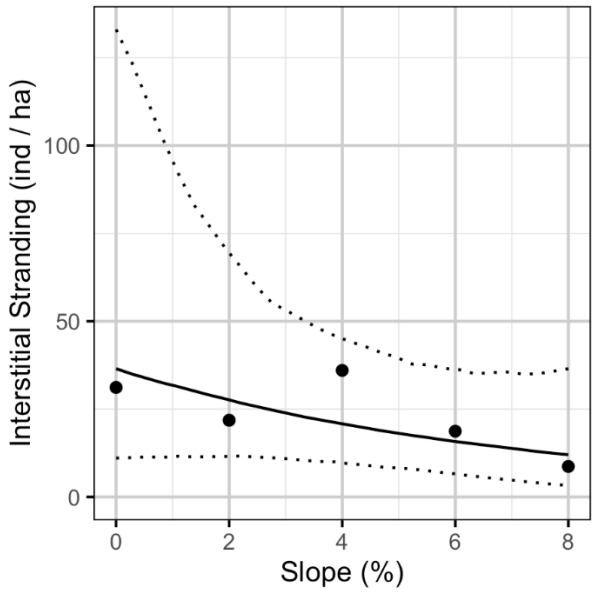

Figure 19: The estimated interstitial stranding density for Rainbow Trout in the Lower Duncan River by slope.

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 20. Substrate with 0 – 2% slope were the most abundant in all examined DRL discharges, followed by substrates with slopes between 6 – 8% and greater that 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)									
	73.0	110.8	148.6	186.4	224.2	262.0	299.8	337.6	375.4	390.2
0-2	197,075	238,975	260,050	327,975	383,325	443,850	522,600	595,500	650,100	664,700
2-4	224,025	262,325	279,700	334,675	369,525	401,700	433,875	466,575	498,675	509,375
4-6	143,150	167,775	179,650	215,875	238,100	256,800	274,875	291,625	305,775	311,150
6-8	102,350	120,100	127,675	148,775	163,500	174,925	185,400	195,550	204,975	208,575
>8	258,625	301,975	324,325	376,575	410,275	441,900	466,225	488,575	509,200	515,700
Total	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

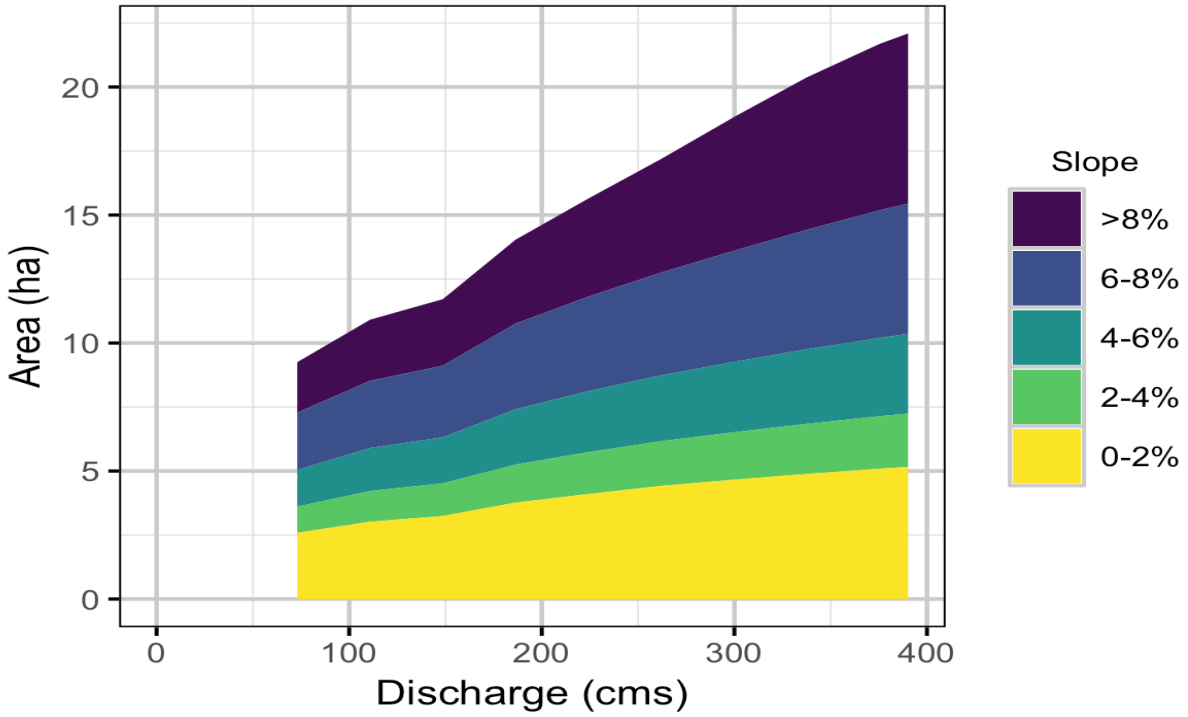

Figure 20: The calculated wetted area in the Lower Duncan River by slope and DRL discharge.

3.5.5 Total Stranding Estimates

Total spring Rainbow Trout pool stranding estimates were consistently low and invariable in each study year examined (Figure 21). In most of the study years examined, total fall pool stranding estimates were substantially higher and more variable. When the seasons were combined in each study year, mean total pool stranding estimates ranged between approximately 0.2% (2010) and 4.9% (2014) of the projected spring age-1 Rainbow Trout population (Figure 22). Except for two of the study years examined (2014 and 2018), mean annual pool stranding was estimated at less than 1.5% of the total spring Rainbow Trout population in the Lower Duncan River.

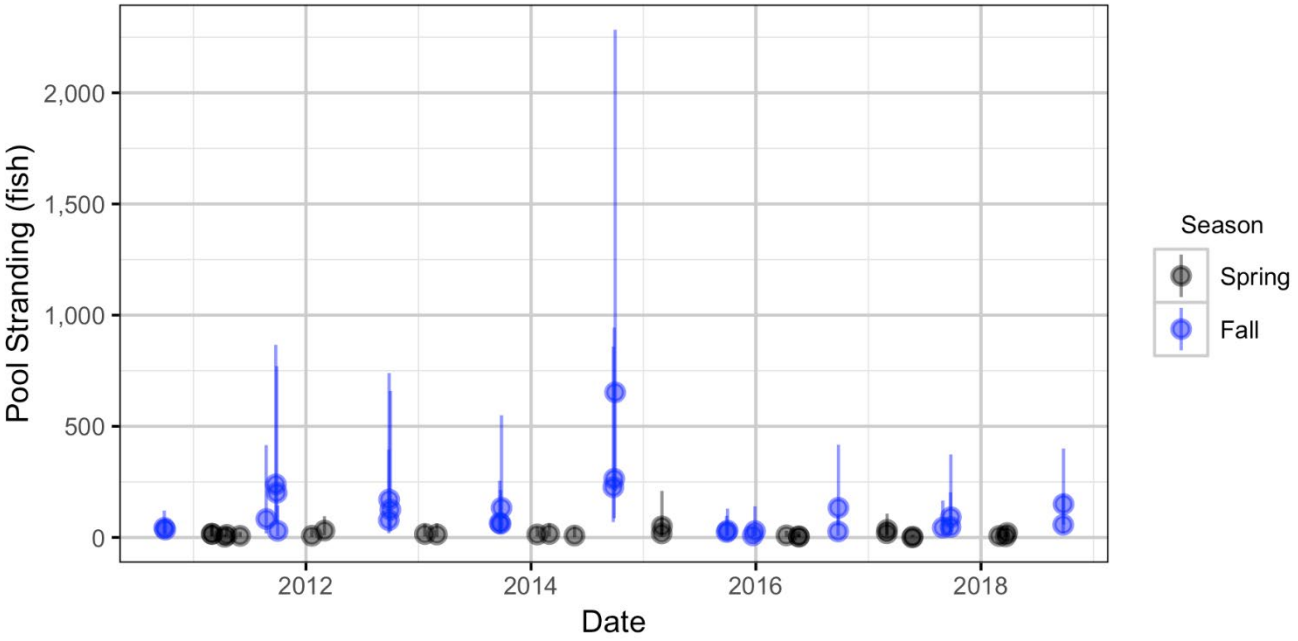

Figure 21: Estimates of total pool-stranded Rainbow Trout by date and season in the Lower Duncan River. Error bars are 95% credibility intervals.

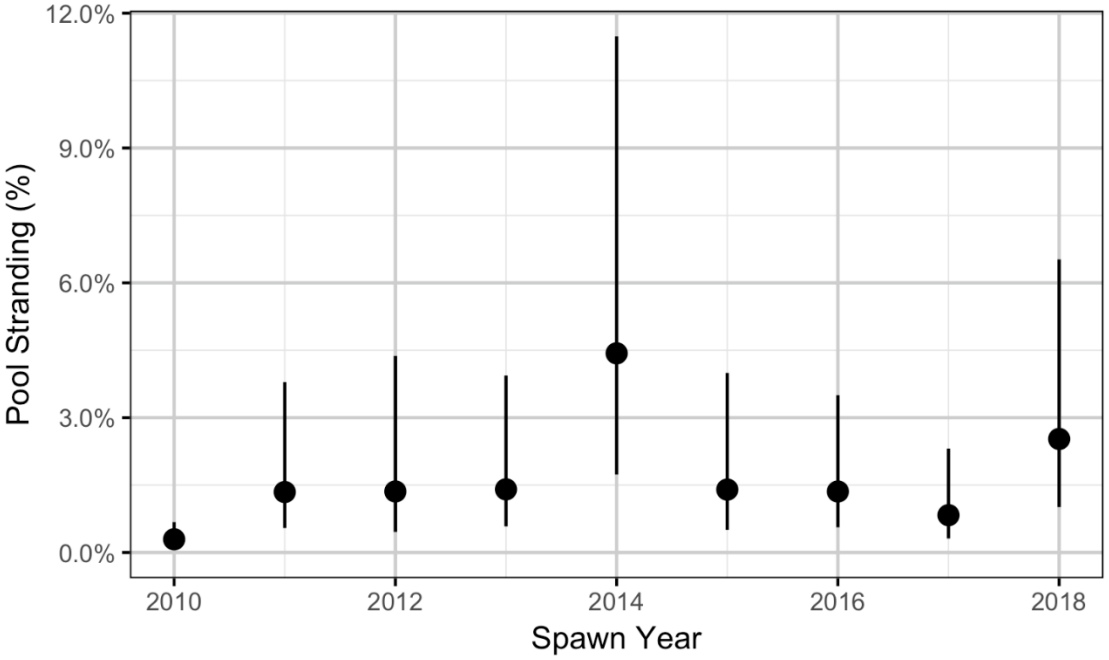

Figure 22: Estimated total pool stranding of Rainbow Trout in the Lower Duncan River as a percent of spring abundance by spawn year. Error bars are 95% credibility intervals.

Total Rainbow Trout interstitial stranding estimates were highly variable between seasons and study years (Figure 23). When the seasons were combined in each study year, total mean interstitial stranding estimates ranged between approximately 0.8% (2010) and 5.0% (2015) of the projected spring age-1 Rainbow Trout population (Figure 24). Except for the 2015 study year, mean annual interstitial stranding was estimated at less than 2.5% of the total spring Rainbow Trout population in the Lower Duncan River.

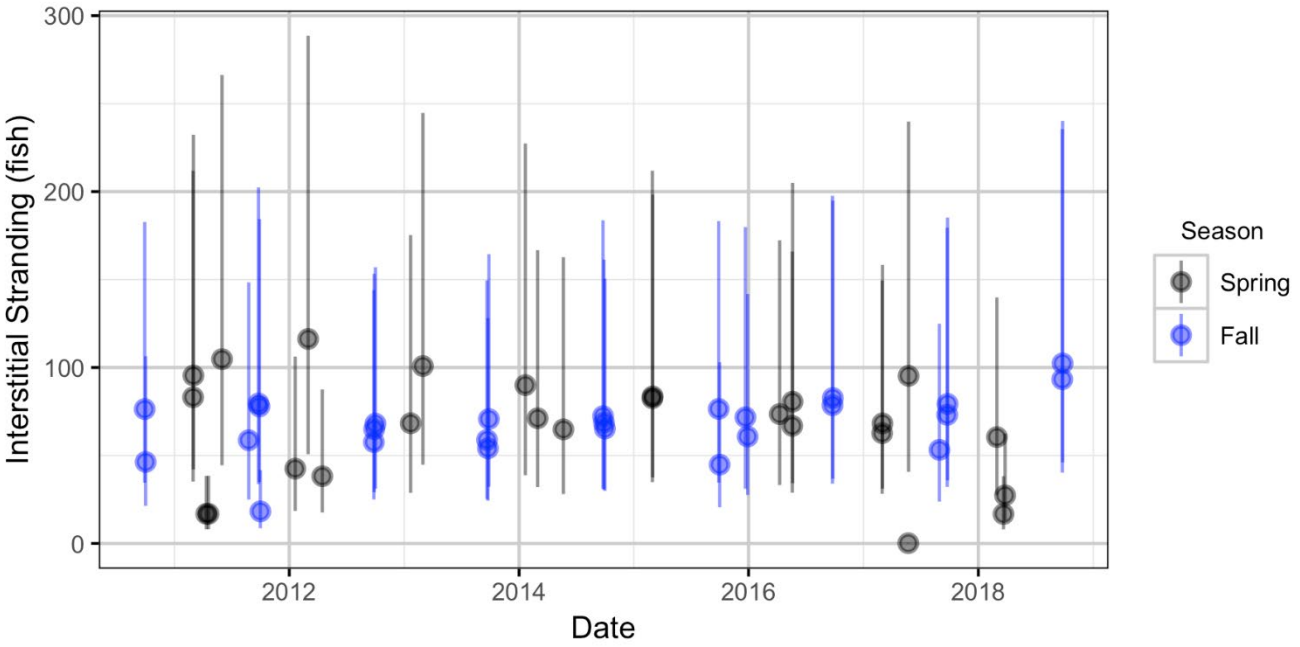

Figure 23: Estimates of total interstitial-stranded Rainbow Trout by date and season in the Lower Duncan River. Error bars are 95% credibility intervals.

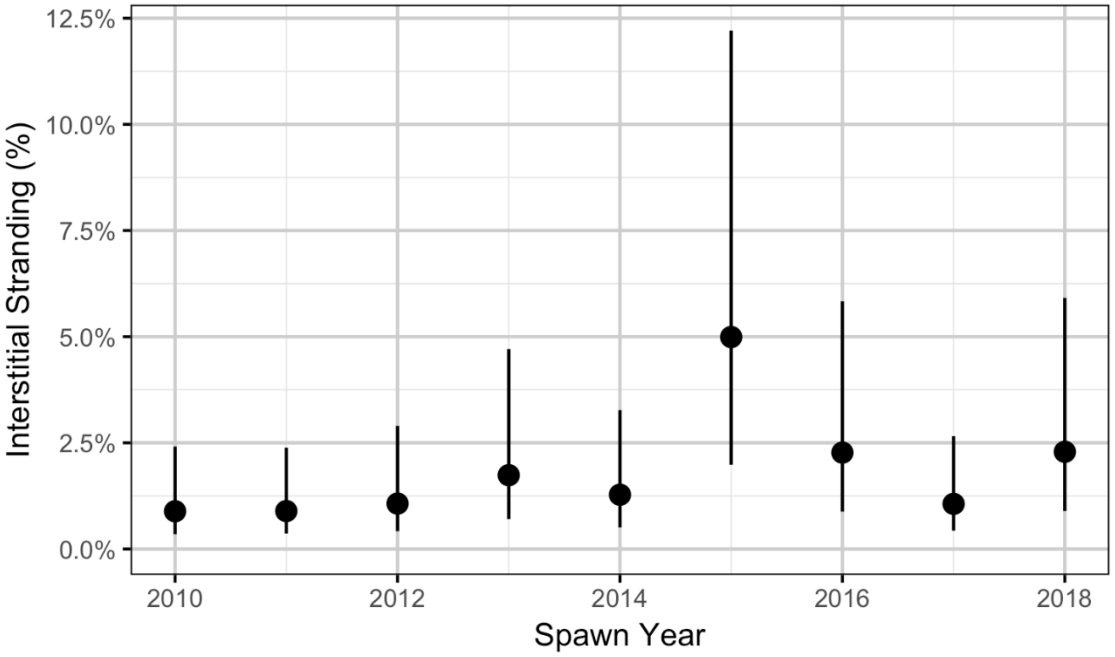

Figure 24: Estimated total interstitial stranding of Rainbow Trout in the Lower Duncan River as a percent of spring abundance by spawn year. Error bars are 95% credibility intervals.

For Rainbow Trout, total stranding (interstitial and pool combined) for the current study year was estimated at 4.5% of the Rainbow Trout age-1 spring population (95% CRI of 2.3 – 9.6%; Figure 25). Total percent stranding remained relatively consistent from 2010 to 2012 and increased in each year from 2013 to 2015. In 2016 and 2017, total percent stranding was estimated to have decreased. The highest stranding was estimated to have occurred in 2015, with a mean estimate of 6.2% of the Rainbow Trout spring age-1 population (95% CRI of 2.9 – 13.5%; Figure 25).

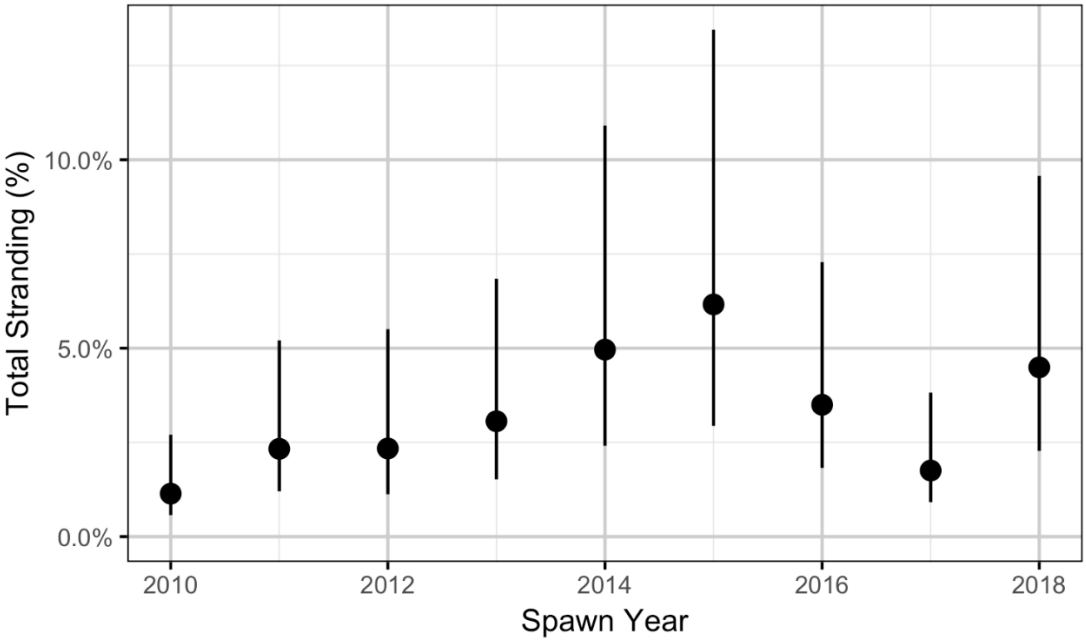

Figure 25: Estimates of total percent stranded Rainbow Trout by date and season in the Lower Duncan River. Error bars are 95% credibility intervals.

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 and Years 4 and 5 of this program (Poisson and Golder 2010, Golder and Poisson 2012, and 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 area 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 (Section 4.1.2 above). Interannual variability in discharge has also been reduced under post-WUP operations. During post-WUP operations, variability of total reduction magnitudes and ramping rates has also been reduced. As recommended by the DDMMON-1 and -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 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 amounts of habitat dewatered under the post-WUP operating regime. As the sampling programs assessing the 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 as to 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. 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 Lower Duncan River is discussed below in Section 4.3. Previous analysis showed that residual wetted area of pool was not a predictive variable (Poisson 2011, Golder and Poisson 2012). In the current dataset, seasonal effect on pool stranding numbers was 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 vs. 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.

Over the study years when interstitial sample methodology was 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 modelling analysis 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, and this relationship should continue to be evaluated in future years as more data are collected. Slope of dewatered substrate was found to influence interstitial stranding rates, but this effect was not found to be statistically significant.

4.2.2 Slope of Dewatered Area

The categories of low (0-4%) and high slope (>4%) used in the analysis of previous study years were based on values in the literature from previous stranding work (Bauersfeld 1978; Flodmark 2004). Based on previous data analysis, considerably higher amounts low slope habitat was 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.

As the results from the current dataset suggest that slope did not influence the formation of isolated pools within the study area, the effect of slope was not included in the 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 data set. This finding could be due to high variability 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 amount of dewatered area and suitable habitat. Based on the findings of previous study years (Golder and Poisson 2012, Golder 2017a, 2017b and 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 suggested 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 Study Years 8 and 9 (Golder 2017b and 2018), 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 11, stranding rates at both index and random sites should continue to be analyzed as the data set grows.

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 in the current study year, estimated Rainbow Trout juvenile abundance was calculated based on spring surveys conducted by Andrusak (2017). 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 in 2015 and 2016, as well as in the current study year. 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%.

This discrepancy may be because the assumed observer efficiency estimates for the fall abundance estimates were too high (Andrusak 2017). Although unlikely, 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 that has been identified (Andrusak and Andrusak 2015). These finding should be interpreted with caution as the models used in the individual programs were different.

Total mean annual estimates for the number of Rainbow Trout juveniles stranded were low, ranging from 1.1% (95% CRI of 0.5 – 2.7%) of the Rainbow Trout age-1 spring population in 2010 to 6.2% (95% CRI of 2.9 – 13.5%) in 2015. These estimates, combined with the increase in the juvenile Rainbow Trout population documented during spring surveys (Andrusak 2017) suggest positive levels of Rainbow Trout recruitment in the Lower Duncan River. Based on the current state of knowledge, hypothesis H_{02} is not rejected for Rainbow Trout. Therefore, 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 2017 and 2018, only 32 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 10 of the DDMMON-16 Lower Duncan River Fish Stranding Impact Monitoring Program 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 9 results (Golder 2017a, 2017b and 2018), seasonal effect on pool stranding in the current program 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)
  - Site type was found to have a significant effect on Rainbow Trout interstitial stranding rates in Year 9 but was not examined in the current year (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 data set, 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 to 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 data set, the uncertainty associated with interstitial stranding of Rainbow Trout was greatly reduced. With continued enhancement to sampling and modelling methodology, and another year of data collection to increase the data set, the uncertainty related to stranding estimation is expected to continue to decrease. 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 10) of the DDMMON-16 Lower Duncan River Fish Stranding Impact Monitoring Program are as follows:

1) Continue following the current pool sampling methodology used in Year 10 stranding assessments. This will continue to strengthen the existing dataset and allow for continued accurate estimates of fish stranding in the LDR.

2) To further reduce uncertainty related to stranding estimates, stratify the data and system by gradient and substrate. More specifically, prior to the onset of sampling in Year 11, re-analyse the large existing database to determine the gradients and substrates for which the stranding density is negligible. This will allow focussed sampling in habitats with the higher potential to strand fish and will allow more accurate and precise stranding estimates. To increase the amount of interstitially sampled habitat per site to obtain sufficient numbers of data points, field crews can assess areas of randomly selected dewatered habitat within a site that the prior stratification analysis indicated would have a reasonable probability to strand fish. Consistent effort should be conducted at each site to ensure an adequate number of sites are sampled during each assessment along the entire lower Duncan River.

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

4) Examine the feasibility of using updated observer efficiencies during abundance estimation analysis.

These recommendations are designed to build on the current data set. 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.

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https://golderassociates.sharepoint.com/sites/31732g/deliverables/issued to client-for wp/1535517-003-r-rev0/1535517-003-r-rev0-ddmmon-16 2017-2018 year 10 04dec_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) eAbundancePass
1[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], eAbundancePass[i, pass]) eAbundancePass[i, pass+1] <- eAbundance
Pass[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 ith `Reduction` on `bDensity`
`bSiteDensity[i]`	Effect of ith `Site` on `bDensity`
`Discharge[i]`	Initial discharge prior to i^{th} site visit
`NumberPoolPresent[i]`	Number of pools observed at the ith site visit
`sDispersion`	SD of Overdispersion
`SiteArea[i]`	Area of the site exposed on the ith site visit
`sReductionDensity`	SD of `bReductionDensity`
`sSiteDensity`	SD of `bSiteDensity`

Table 2. Model coefficients.

term	estimate	sd	zscore	lower	upper	pvalue
bDensity	2.1042882	0.2298701	9.159431	1.6581608	2.5627919	0.0007
bDischargeDensity	-0.2884065	0.0920692	-3.139629	-0.4717674	-0.1069057	0.0027

term	estimate	sd	zscore	lower	upper	pvalue
bDischargeDensity2	0.1527953	0.0688023	2.196370	0.0171922	0.2809240	0.0307
sDispersion	0.7495704	0.0632473	11.917109	0.6324556	0.8847371	0.0007
sReductionDensity	0.4333230	0.0976319	4.494835	0.2455812	0.6317236	0.0007
sSiteDensity	1.1792637	0.1976243	6.057918	0.8491480	1.6376088	0.0007

Table 3. Model summary.

n	K	nchains	niters	nthin	ess	rhat	converged
357	6	3	500	100	608	1.005	TRUE

Pool Stranding

Table 4. Parameter descriptions.

Parameter	Description
`bIntercept`	Intercept for `log(eAbundance)`
`bReduction[i]`	Effect of ith `ReductionEventID` on `bIntercept`
`bSeason[i]`	Effect of ith `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	-0.8163520	0.3868465	-2.148423	-1.6644923	-0.1410826	0.0200
bAreaAbundance	0.2525414	0.0509472	5.015622	0.1606509	0.3589610	0.0007
bEfficiency	-0.3474591	0.3195088	-1.157792	-1.0592020	0.1872382	0.2453
bSeasonAbundance[2]	2.0759012	0.3260158	6.386753	1.4187644	2.7536448	0.0007
sDispersion	2.3944029	0.1015109	23.601795	2.2038409	2.6011438	0.0007
sReductionAbundance	0.7820712	0.1713452	4.638347	0.5074494	1.1835713	0.0007
sStudyYearAbundance	0.7001339	0.3096772	2.406406	0.2665290	1.4774921	0.0007

Table 6. Model summary.

n	K	nchains	niters	nthin	ess	rhat	converged
1327	7	3	500	200	201	1.017	TRUE

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	3.0330979	0.3687691	8.201183	2.1860183	3.6870293	0.0007
bSlopeDensity	-0.4187632	0.3949521	-1.045161	-1.1824371	0.3986025	0.2067
sDensity	0.5701973	0.4539565	1.536747	0.2539611	1.8093181	0.0007

Table 9. Model summary.

n	K	nchains	niters	nthin	ess	rhat	converged
5	3	3	500	10	939	1.001	TRUE

APPENDIX C

Photographic Plates

GOLDER

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