Duncan Dam Project Water Use Plan

Lower Duncan River Fish Stranding Impact Monitoring

Implementation Year 9

Reference: DDMMON-16

Study Period: (April 2016 to April 2017)

Submitted to:
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1535517-002-R-Rev0

December 5, 2019

REPORT

DDMMON-16: Lower Duncan River

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Cover Photo: Upstream view of Site M0.8R with Duncan Dam in the background, 20 May 2016.

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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 2017, documented stranding rates of juvenile Mountain Whitefish (*Prosopium williamsoni*) are very low and are not believed to result in population level effects, while the current interstitial stranding estimates for juvenile Rainbow Trout (*Oncorhynchus mykiss*) are too uncertain to inform a confident total stranding estimate.

This report presents the results from Years 1 to 9 of the DDMMON-16 program, and the current status of management questions for DDMMON-16 is provided in Table EI below. Because of the high degree of variation in stranding rates, the uncertainty of the interstitial stranding estimates, and the many variables that could potentially contribute to stranding, these results should be interpreted with caution.

Table EI: DDMMON-16 Year 9: Status of Management Questions and Objectives.

DDMMON-16 Management Question	DDMMON-16 Specific Hypothesis	DDMMON-16 Year 9 (2016-2017) 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. - How wetted history is related to 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 9 (2016-2017) Status Summary
2) What are the levels of impact to resident fish populations associated with fish stranding events on the lower Duncan River?	*Ho₁: 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 effect of dewatering on the formation of pools (density) and interstitial stranding indicating a difference between index and random sites was not found. Since the lack of significance was marginal, the difference between the two types of sites may become significantly different as the data set grows. Therefore, based on these analyses, hypothesis H01 cannot be rejected at this time but based on the initial study design, this hypothesis will likely be rejected in the future. - 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.
	Ho₂: 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 were relatively low with high precision. However, the estimated numbers of interstitially stranded fish in the lower Duncan River were high with low precision. - The very low numbers of interstitially stranded fish encountered during assessments precludes in-depth modelling and estimation. - A seasonal effect on Rainbow Trout stranding was identified, with stranding rates approximately six 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. - A seasonal effect on Mountain Whitefish stranding was not identified, but fall stranding estimates were approximately half those estimated for the spring. - 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. Therefore, study hypothesis H02 is not rejected for Mountain Whitefish. - Within the current dataset relationships between pool and interstitially stranded fish and slope of substrate were not found. - Based on the current dataset, study hypothesis H02 cannot be reasonably rejected for Rainbow Trout.

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1.0 INTRODUCTION

1.1 Background

The lower Duncan River (LDR) originates from Duncan Dam (DDM), and runs for approximately 11 km before flowing into 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 9 (2016 - 2017) builds on the historic methodology, expands the program's data sets, 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 actually 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 9 of this Program (15 April 2016 to 14 April 2017). 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) and the present study year (Year 9: 2016 – 2017) 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 6 (2010 – 2011, 2011 – 2012, 2012 - 2013, and 2013 – 2014, respectively) 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, and Golder 2015) 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 will lead to substantial changes to the DDMMON-16 program in its final three years (Years 8 to 10) of implementation.

1.4.1 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 area 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

Field sampling during abundance assessments was conducted as consistently as possible with previous fish abundance assessments performed as part of the DDMMON-2 – Lower Duncan River Habitat Use Monitoring (Thorley et al. 2012). However, a few methodology changes were made in study Years 6 and 7 to ensure sampling robustness while addressing logistic difficulties. The goal during Year 9 was to sample similar numbers of sites, as well as similar length of river in comparison to Years 7 and 8.

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 is to modify the Lower Duncan River Fish Stranding Database. At the onset of Year 10, the database will be 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 salvaged in the future years of this program.

1.4.6 Data Analysis

The modelling used in Year 8 (Golder 2017) of this program was updated to incorporate the current year's data set, to remove the dichotomous slope classification when analyzing as a variable related to stranding rates, and to analyze substrate size as a variable related to interstitial stranding. To increase the precision of the estimates provided by this program, specific outputs from the updated TELEMAC 2D hydraulic model created by the DDMMON-3 program may be beneficial for this study. If deemed feasible, additional model runs in Year 10 of that program would provide updated wetted areas at stranding locations at various flow elevations, which would update the basis for extrapolation of stranding rates defined in this study.

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; these were termed Reach 1 (River Km [RKm] 0.0 - at DDM spill gates to RKm 0.8), Reach 2 (RKm 0.8 to RKm 2.6), Reach 3 (RKm 2.6 to RKm 5.7), Reach 4 (RKm 5.7 to RKm 6.7), and 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, and 2014; Golder and Poisson 2012). These stranding sites included 11 index stranding assessment sites and 39 non-index sites (Appendix A, Figures 1 to 7). Nearly all of the remaining habitats outside of the identified sites consist of steep banks with extreme gradient that would be considered to have very low stranding risk. Consequently, additional major fish stranding locations outside of the 50 potential fish stranding sites 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 were conducted from 19 May 2016 to 02 March 2017, during planned flow reductions at DDM. Each assessed reduction from DDM was assigned a reduction event number (RE; see Section 2.6) and Figure 2 outlines all assessment activities during Year 9. Abundance Estimation Sampling in the present study year was conducted between 18 and 23 September 2016. An in-depth summary of the chronology of sampling and project milestones in all study years is provided in Appendix A, Tables A1 to A8.

Table 1: Sampling activities for the 2016-2017 Lower Duncan River Fish Stranding Impact Monitoring, Year 9 Program.

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

LEGEND

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

1 0 1

SCALE 1:50,000 KILOMETRE

REFERENCE

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

DATUM: NAD83 PROJECTION: UTM ZONE 11

PROJECT

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

TITLE

OVERVIEW OF STUDY AREA

GOLDER

PROJECT		1535517	FILE No.	
DESIGN	BH	14 JUN. 2018	SCALE AS SHOWN	REV. 0
GIS	JG	14 JUN. 2018	**FIGURE: 1**	
CHECK	BH	05 DEC. 2018		
REVIEW	SR	05 DEC. 2018		

Document Path: \\golder.gds\gal\burnaby\CAD-GIS\Bur-Graphics\Projects\2012\1492\12-1492-0117\GIS\Mapping\MXD\Fish\2018\Fig1_1214920117_overview_20181205.mxd

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, as it provides information on the magnitude of flow reductions along the majority of the river channel. All DDM releases and discharge data for the LDR were obtained from BC Hydro.

2.4 Bayesian Analysis

The analysis was implemented using the statistical environment R, v. 3.5.0 (R 2018), interfaced with JAGS v. 3.3.0 (Plummer 2013) through the jmbr package (Thorley 2018). JAGS distributions and functions are defined in Table 2.

Table 2: JAGS distributions and functions used in the Bayesian models.

Distribution/function	Description
dbin(p, n)	Binomial distribution with n trials and p probability of success
dnorm(μ, τ)	Normal distribution with a mean μ and 1/variance τ
dunif(a, b)	Uniform distribution with a minimum of a and a maximum of b
dpois(λ)	Poisson distribution with a mean λ
log(x)	Natural logarithm function
logit(x)	Logit function

Unless indicated otherwise, the models used prior distributions that were vague in the sense that they did not affect the posterior distributions (Kéry and Schaub 2011, p. 36). The posterior distributions were estimated from 1,500 Markov Chain Monte Carlo (MCMC) samples thinned from the second halves of three chains (Kéry and Schaub 2011, pp. 38–40). Model convergence was confirmed by ensuring that Rhat (Kéry and Schaub 2011, p. 40) was less than 1.1 for each of the parameters in the model and the effective sample size is ≥150 for each of the monitored parameters (Kéry and Schaub 2011, p. 61). Model adequacy was confirmed by examination of residual plots.

The posterior distributions of the fixed (Kery and Schaub 2011, p. 75) parameters were summarized in terms of a point estimate (median), lower and upper 95% credible limits (2.5th and 97.5th percentiles), the standard deviation (SD), percent relative error (half the 95% credible interval as a percent of the point estimate), and significance (Kéry and Schaub 2011, p. 37, p. 42). Where applicable, model adequacy was confirmed by examination of residual plots.

The results were displayed graphically by plotting the modeled relationships between particular variables and the response with 95% credible intervals (CRIs) with the remaining variables held constant. In general, predictions were estimated when continuous and discrete fixed variables were 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 hyper distributions) (Kéry and Schaub 2011, pp. 77–82). When informative, effects were expressed in terms of *effect size* (i.e., percent change in the response variable) with 95% confidence/credible intervals (CIs, Bradford et al. 2005).

2.5 Fish Abundance Assessment

2.5.1 Fish Abundance Site Selection

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

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

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

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

2.5.2 Snorkel Surveys

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

2.5.3 Data Analysis

Separate abundance estimates were conducted for Mountain Whitefish and Rainbow Trout juveniles (fork length <250 mm). Hierarchical Bayesian Models (HBMs) were used to estimate total abundance.

In the Bayesian implementation of the model, fish abundance was assumed to be Poisson-distributed, with a mean expected density drawn from a log-normal distribution (Table 3). Observed fish counts were assumed to be binomially distributed, with estimated fish abundance as the number of trials and observer efficiency as probability of success. Fish density was modeled using fixed effects of depth (shallow/deep) and year (2013, 2014, 2015, and 2016) and a random effect of site, to allow density to vary by site. The significance of model parameters was determined based on whether the parameters' 95% CRI overlapped zero. Since the first level of each factor effect (depth and year) was set to zero, if a parameter's 95% CRI overlapped zero, it suggested that there was no significant difference between that parameter and the first level of the factor.

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 shown in Table 4 and Table 5, and model code is provided in Appendix B.

Table 3: Variables and parameters in the Bayesian analysis of fish density and abundance.

Variable/parameter	Description
sSite	Standard deviation of the effect of site on expected fish density
bIntercept	Expected log fish density at a typical shallow, slack site in 2013
bEfficiencyVisit [k]	Random effect of observer efficiency at the *k*-th data point
eEfficiency[k]	Observer efficiency at the *k*-th data point
nSiteNum	Number of sampled sites
bSite[st]	The random effect of the *st*-th site on fish density
nYearNum	Number of sampling years
bYear[yr]	The effect of the *yr*-th year on fish density
nDepthNum	Number of depth strata
bDepth[i]	The effect of the *i*-th depth stratum on fish density
Area[k]	The site area of the *k*-th data point
eDensity[k]	Expected fish density at the *k*-th site
eAbundance[k]	Predicted fish abundance at the *k*-th site
SiteNum[k]	Numeric representation of site name of the *k*-th data point
YearNum[k]	Numeric representation of sampling year of the *k*-th data point
Visit[k]	Numeric representation of site visit of the *k*-th data point
DepthNum[k]	Numeric representation of the depth stratum of the *k*-th data point
Nfish[k]	The observed number of fish at the *k*-th data point

Table 4: Prior probability distributions in the Bayesian analysis of Rainbow Trout and Mountain Whitefish density and abundance.

Variable/parameter	Description
sSite	dunif(0, 5)
bSite[k]	dnorm(0, $sSite^{-2}$)
bDepth[i]	dnorm(0, 5^{-2})
bYear[i]	dnorm(0, 5^{-2})
bEfficiency	-0.53
bEfficiencyVisit[k]	dnorm(0, 0.68^{-2})

Table 5: Dependencies between variables and parameters in the Bayesian analysis of Rainbow Trout and Mountain Whitefish density and abundance.

Variable/parameter	Dependency
logit(eEfficiency[k])	bEfficiency + bEfficiencyVisit[Visit[k]]
log(eDensity[k])	bIntercept + bYear[YearNum[k]] + bDepth[DepthNum[k]] + bSite[SiteNum[k]]
eAbundance[k]	dpois(eDensity[k] * Area[k])
Nfish[k]	dbin(eEfficiency[k], eAbundance[k])

The estimated stratum fish density (fish/m²) and the total area of each depth/flow stratum (Table 6), derived from the DDMMON-3 RIVER-2D hydraulic model, were used to estimate the total abundance of fish in each stratum. Summing of estimates across all sampled strata yielded the total abundance of fish within the LDR (expressed as mean and 95% CRI).

Table 6: Areas (m²) of the different depth/flow strata, derived from the DDMMON-3 RIVER-2D hydraulic model.

Stratum	Area (m²)	
	2013	**2014-2016**
Shallow/Slack	66,217.5	73,276.6
Shallow/Fast	337,857.1	357,565.1
Deep/Slack	---	11,092.3
Deep/Fast	145,784.8	164,858.4

2.6 Fish Stranding Assessment

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

Because of the remote location of the LDR and limited development, access to the 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 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 remains in question at lower DRL discharges and lake elevations.

2.6.1 Year 9 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 and 9, 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 all sites was calculated using the site area regressions that were completed in Year 3 (Golder 2011).

2.6.2 Year 9 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. In addition, to determine the observer (capture) efficiency during stranding assessments, multi-pass electrofishing (two passes) was conducted at a subset of randomly selected pools. The effort for each subsequent pass was as consistent as possible with the first pass. The fish salvaged and effort for each pass were recorded separately. 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

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).

If there was not sufficient dewatered area, or the substrate was too large to effectively conduct transect sampling, 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 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 in to estimate the total area exposed by year.

2.6.3.2 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 2.5.3.

2.6.3.3 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 (separated by species; Rainbow Trout and Mountain Whitefish) had to be estimated for each reduction. The number of pools at individual sites was estimated using a zero-inflated model with a Bernoulli - Poisson distribution of pool counts (Kery and Schaub 2011, pp. 386–388). The model defined the number of pools present at a site to be Poisson-distributed, with a mean expected value determined by drop of discharge (difference between initial and resulting discharge, m^3/s), site area, and site-specific effect (Table 7).

To estimate the total number of pools that form throughout the study area, mean expected pool counts were multiplied by total exposed area using GIS-derived low-slope (0-4%) and high-slope (>4%) dewatered areas for each stranding event. The complete model specification used is shown in Table 8 and Table 9, and model code is provided in Appendix B. The model was run separately for Rainbow Trout and for Mountain Whitefish.

Table 7: Variables and parameters in the Bayesian analysis of pool density.

Variable/parameter	Description
bIntercept	Log pool density under zero flow drop, and typical site
pIntercept	Probability of pool formation (for zero-inflated model)
bDrop	Effect of discharge drop on log pool density
sSite	Standard deviation of the effect of site on expected number of pools
bSiteName[j]	The random effect of the *j*-th site on pool numbers
nSiteName	Number of unique sites visited
eP[i]	Binary estimate of whether the *i*-th case had formed pools
eDensityPool[i]	Expected density of pools at the *i*-th case
SiteArea[i]	The dewatered area at the *i*-th case
NumPoolsPresent[i]	Observed number of pools at the *i*-th case

Table 8: Prior probability distributions in the Bayesian analysis of pool density.

Variable/parameter	Description
sSite	dunif(0, 5)
pIntercept	dunif(0, 1)
bDrop	dnorm(0, 5^{-2})
bIntercept	dnorm(0, 5^{-2})
bSiteName[j]	dnorm(0, sSite^{-2})

Table 9: Dependencies between variables and parameters in the Bayesian analysis of pool density.

Variable/parameter	Dependency
NumPoolsPresent[i]	dpois(eDensityPool[i] * eP[i] * SiteArea[i])
log(eDensityPool[i])	bIntercept + bDrop*Drop[i] + bSiteName[SiteName[i]]
eP[i]	dbern(pIntercept)

The number of fish captured in multi-pass electrofishing was used to estimate fish catchability. Catchability was assumed to be 100% when using either visual or dip-net sampling. Single-pass pool abundance was estimated using the number of fish captured during the pass and the catchability estimated in multi-pass sampling.
Fish abundance was assumed to vary with season (fixed effect) and reduction (random effect). Season was

defined as "spring" for January-July months and as "fall" for August-October. Once season and reduction parameters were estimated (see Table 7 for full list of parameters), they were used to estimate total number of fish per pool at each reduction.

The Bayesian model for abundance of pool-stranded fish defined the number of fish at a pool to be Poisson-distributed, with an overdispersion, and a mean expected value determined by season and a reduction-specific random effect (Table 7). The number of fish removed in each sampling was modeled to be binomially distributed. To estimate total pool stranding, estimated pool abundance was multiplied by the number of estimated fish/pool. The complete model specification used is shown in Table 10 to Table 12, and model code is provided in Appendix B.

Table 10: Variables and parameters in the Bayesian analysis of pool stranding.

Variable/parameter	Description
bIntercept	Log fish abundance in the spring, under typical reduction
bSeason[i]	The effect of the *i*-th season on pool-stranded fish abundance, where i = 1 when season is winter/spring, and i = 2 when season is fall
p[j]	Catchability using the *j*-th sampling gear, where j = 1 for visual and dip-net, and j = 2 for electrofishing
r	Extra-Poisson variation (overdispersion) in fish counts per pool
eU[k]	Effect of extra-Poisson variation on fish counts at the *k*-th pool
sReduction	Standard deviation of the effect of reduction on expected fish counts per pool
bReduction[r]	The random effect of the *r*-th reduction on expected fish counts per pool
eAbundance[k]	Expected fish counts in the *k*-th pool
SeasonNum[k]	Season during which the *k*-th pool was sampled
ReductionEventID[k]	Reduction during which the *k*-th pool was sampled
eN[k]	Estimated fish counts at the *k*-th pool
eNPass[k, p]	Estimated number of fish present at the *k*-th pool prior to the *p*-th pass
Pass[k, p]	Sampled number of fish at the *k*-th pool prior to the *p*-th pass
SamplingGearNum[g]	Sampling gear used at the *k*-th pool, where g = 1 stands for visual or dip-net, and g = 2 stands for electrofishing

Table 11: Prior probability distributions in the Bayesian analysis of pool stranding.

Variable/parameter	Description
sSite	dunif(0, 5)
r	dunif(0, 5)
bIntercept	dnorm(0, 5^{-2})
bReduction[j]	dnorm(0, $sReduction^{-2}$)
eU[i]	dgamma($1/r^2$, $1/r^2$)
p[2]	dunif(0, 1)
bSeason[2]	dnorm(0, 5^{-2})

Table 12: Dependencies between variables and parameters in the Bayesian analysis of pool stranding.

Variable/parameter	Dependency
log(eAbundance[i])	bIntercept + bSeason[SeasonNum[i]] + bReduction[ReductionEventID[i]]
eN[i]	dpois(eAbundance[i]*eU[i])
eNPass[i, 1]	eN[i]
Pass[i, pass]	dbin(p[SamplingGearNum[i]], eNPass[i, pass])
eNPass[i, pass+1]	eNPass[i, pass] - Pass[i, pass]

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 9. Presently, 83 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 75.5 m³/s (2666.3 ft³/s) on 01 August 2016 to 659.1 m³/s (23275.9 ft³/s) on 19 May 2016. Hourly discharge from DDM ranged from 1.2 m³/s (42.4 ft³/s) on several days between early June and late July 2016, to 232.7 m³/s on 14 September 2016 (Figure 2). However, discharge data for March 2017 were missing, and the pattern of discharge for this period of time is not known.

Lowest DDM flows typically occur during the spring/summer recharge of Duncan Reservoir. During this period there are temporary pulses of flow 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.

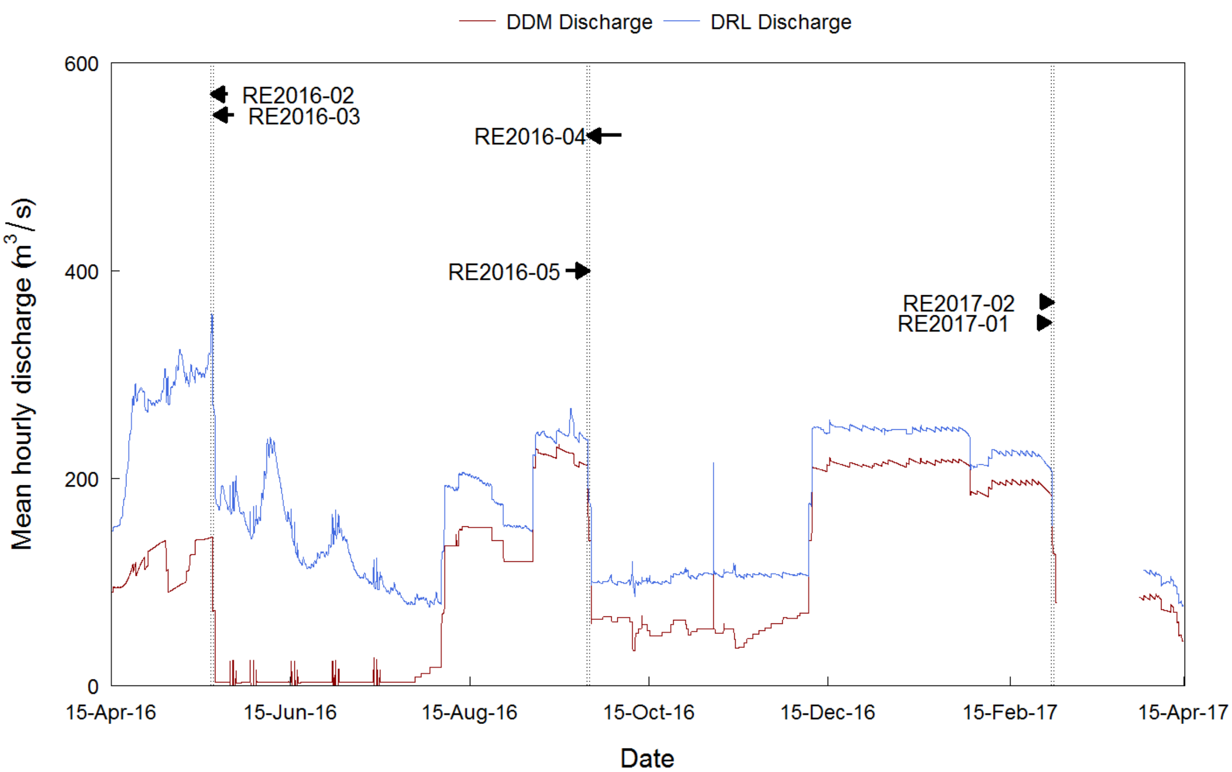


Figure 2: Hourly discharge at the Duncan Dam (DDM, red line) and at the lower Duncan River below the Lardeau River (DRL, blue line) from 15 April 2016 to 15 April 2017. Vertical dotted lines represent the timing of fish stranding assessments.

During the present study, six reduction events occurred at DDM (Figure 2 and Table 13). During the reduction events, DDM daily decreases of discharge ranged between 48 and 70 m³/s (1695 and 2472 ft³/s; Table 13). These decreases represent the discharge reductions at DDM, rather than flow changes at particular downstream fish stranding sites.

Table 13: Summary of DDM flow reduction events, from 19 May 2016 to 02 March 2017, for those events when fish stranding assessments were conducted.

Date	Reduction Event	DDM Discharge m^3/s (ft^3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
19 May 2016	RE2016-02	142 (5015)	72 (2543)	70 (2472)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 08:00 to 10:15.	Discharge reduced to meet flow target at DRL.
20 May 2016	RE2015-03	72 (2543)	3 (106)	69 (2437)	Down 7 m^3/s (247 ft^3/s) every 15 minutes from 06:00 to 08:00, down 6 m^3/s (212 ft^3/s) at 08:15.	Discharge reduced to meet flow target at DRL.
24 Sep 2016	RE2015-04	212 (7487)	144 (5085)	68 (2401)	Down 6 m^3/s (212 ft^3/s) every 15 minutes from 06:00 to 08:30, down 2 m^3/s (71 ft^3/s) at 08:45.	Onset of Kokanee protection flows.
25 Sep 2016	RE2015-05	144 (5085)	75 (2649)	69 (2437)	Down 6 m^3/s (212 ft^3/s) every 15 minutes from 06:00 to 08:30, down 3 m^3/s (106 ft^3/s) at 08:45.	Kokanee protection flows.
01 Mar 2017	RE2017-01	182 (6427)	128 (4520)	54 (1907)	Down 6 m^3/s (212 ft^3/s) every 15 minutes from 06:00 to 08:00.	Discharge reduced to meet flow target at DRL.
02 Mar 2017	RE2017-02	128 (4520)	80 (2825)	48 (1695)	Down 6 m^3/s (212 ft^3/s) every 15 minutes from 06:00 to 07:45.	Discharge reduced to meet flow target at DRL.

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

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 assessment methodology. Therefore, results from assessments prior to 15 September 2006 have been excluded from the dataset. Stranding assessments were conducted following six flow reductions during study Year 9 (2016-2017). 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 14.

Table 14: Scientific names of fish species encountered during fish stranding assessments on the lower Duncan River, September 2006 to March 2015.

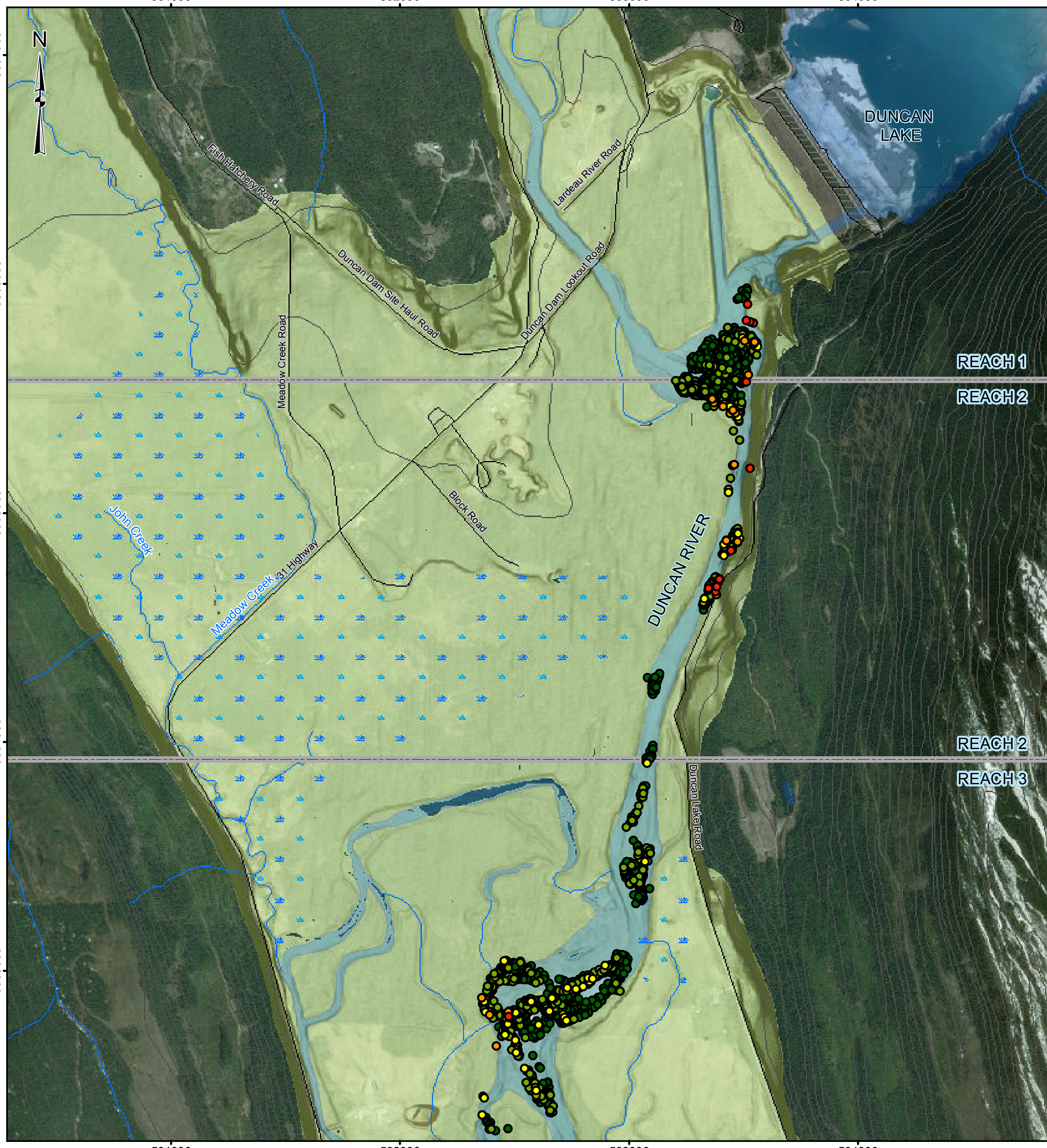
Category	Species	Scientific Name	Species Code[a]
Sportfish	Rainbow Trout	*Oncorhynchus mykiss*	RB
	Bull Trout	*Salvelinus confluentus*	BT
	Mountain Whitefish	*Prosopium williamsoni*	MW
	Pygmy Whitefish	*Prosopium coulteri*	PW
	Kokanee	*Oncorhynchus nerka*	KO
	Burbot	*Lota lota*	BB
Non-sportfish	Longnose 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). 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 9 (2012 – 2013 to 2016 – 2017). The number of pools sampled in the present year was also reduced to allow for more intensive interstitial sampling effort. This resulted in the sampling of 280 pools and the most number of interstitial transects (n = 145) assessed to date (Table 15). The locations of all sampled stranding mechanisms within the dataset are presented in Figure 3 and Figure 4.

Table 15: Sampling effort during reductions of each study year included in the present analysis.

DDMMON-16 Study Year	Number Assessed			Number Sampled		
	Reductions	Index Sites	Non-Index Sites	Pools	Interstitial Grids	Interstitial Transects
2006-2007	2	16	0	144	15	0
2007-2008	7	56	0	346	40	0
1 (2008-2009)	8	42	0	233	34	0
2 (2009-2010)	6	33	14	221	40	0
3 (2010-2011)	7	50	22	346	96	0
4 (2011-2012)	7	30	20	133	411	0
5 (2012-2013)	7	20	18	86	331	0
6 (2013-2014)	5	13	16	60	325	0
7 (2014-2015)	6	21	18	64	124	101
8 (2015-2016)	5	14	19	106	0	135
9 (2016-2017)	6	15	20	210	0	145



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LEGEND

STRANDING SAMPLE POINT (%)
- 0 - 5
- 5 - 10
- 10 - 15
- 15 - 20
- > 20

REACH BREAK

SLOPE (%)
HIGH : 67
LOW : 0

500 0 500
SCALE 1:25,000 METRES

REFERENCE

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

DATUM: NAD83 PROJECTION: UTM ZONE 11

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

TITLE
LOCATION AND SLOPE (%) OF SAMPLED STRANDING MECHANISMS (SEPT. 2006 - APR. 2017) REACHES 1 TO 3

GOLDER

PROJECT		1535517	FILE No.	
DESIGN	BH	14 JUN. 2018	SCALE AS SHOWN	REV. 0
GIS	JG	14 JUN. 2018	**FIGURE: 3**	
CHECK	BH	05 DEC. 2018		
REVIEW	SR	05 DEC. 2018		

LEGEND

STRANDING SAMPLE POINT (%)

- 0 - 5
- 5 - 10
- 10 - 15
- 15 - 20
- > 20

REACH BREAK

SLOPE (%)

HIGH : 67

LOW : 0

500 0 500

SCALE 1:25,000 METRES

REFERENCE

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

DATUM: NAD83 PROJECTION: UTM ZONE 11

PROJECT

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

TITLE

LOCATION AND SLOPE (%) OF SAMPLED STRANDING MECHANISMS (SEPT. 2006 - APR. 2017) REACHES 3 TO 5

GOLDER

PROJECT		1535517	FILE No.
DESIGN	BH	14 JUN. 2018	SCALE AS SHOWN REV. 0
GIS	JG	14 JUN. 2018	
CHECK	BH	05 DEC. 2018	**FIGURE: 4**
REVIEW	SR	05 DEC. 2018	

Document Path: \\golder.gds\gal\burnaby\CAD-GIS\Bur-Graphics\Projects\2012\1492\12-1492-0117\GIS\Mapping\MXD\Fish\2018\Fig3_4_1214920117_Stranding Mechanism and slope_2015Data_20181205.mxd

In Year 9, a total of 1,822 fish were observed, representing 14 species, of which four were sportfish and 10 were non-sportfish species; in addition, multiple unidentified fish were recorded (Table 16). This total is the third highest documented since 2006, lower only than the 2007-2008 and 2011-2012 sampling years. Juvenile Rainbow Trout encounters (*n* = 164) were the most abundant sportfish observed (9.0% of the total catch). In previous years, Rainbow Trout juveniles accounted for 11.5% to 58% of the total fish counts. However, in Year 9, many larval fish were not identified to species. This resulted in unidentified larval fish accounting for 68% of the total fish counts – more than three times the second highest proportion of unidentified fish recorded since 2006. The majority of these larvae were encountered during an assessment conducted on 20 May 2016.

Kokanee young-of-the-year were the next most abundant sportfish, accounting for 3.5% of the total number of fish encountered. Mountain Whitefish accounted for 0.4% (*n* = 7) of the catch (Table 16; Figure 5). The most common non-sportfish taxa identified to species were Longnose Dace and Slimy Sculpin, accounting for 6.4% and 5.5% of the total number of observed fish, respectively.

Table 16: Total number and relative composition of fish species captured or observed during all stranding assessments conducted on the lower Duncan River from September 2006 to April 2017.

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
Sportfish												
Rainbow Trout	**Adult**	0 (0)	0 (0)	0 (0)	1 (0.1)	0 (0)	0 (0)	0 (0)	1 (0.2)	0 (0)	0 (0)	2 (0.1)
	Juvenile	130 (37.1)	278 (11.6)	530 (33.2)	113 (12.1)	343 (25.2)	452 (24.2)	332 (37.1)	241 (40.2)	737 (58.4)	52 (20.4)	164 (9)
Bull Trout	**Adult**	0 (0)	0 (0)	0 (0)	4 (0.4)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
	Juvenile	2 (0.6)	0 (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)
Mountain Whitefish	**Adult**	0 (0)	1 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
	Juvenile	1 (0.3)	157 (6.5)	70 (4.4)	4 (0.4)	45 (3.3)	225 (12.1)	6 (0.7)	49 (8.2)	3 (0.2)	8 (3.1)	7 (0.4)
Pygmy Whitefish	**Adult**	0 (0)	0 (0)	0 (0)	1 (0.1)	2 (0.1)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
	Juvenile	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Kokanee	**Adult**	0 (0)	97 (4)	572 (35.8)	112 (12.1)	42 (3.1)	55 (3)	111 (12.4)	0 (0)	0 (0)	0 (0)	0 (0)
	Juvenile	0 (0)	2 (0.1)	2 (0.1)	15 (1.6)	0 (0)	1 (0.1)	0 (0)	0 (0)	0 (0)	10 (3.9)	3 (0.2)
	Y-O-Y	0 (0)	1690 (70.2)	85 (5.3)	109 (11.7)	83 (6.1)	861 (46.2)	257 (28.7)	0 (0)	22 (1.8)	12 (4.7)	63 (3.5)
Burbot	**Adult**	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
	Juvenile	0 (0)	0 (0)	1 (0.1)	0 (0)	0 (0)	1 (0.1)	1 (0.1)	0 (0)	0 (0)	0 (0)	0 (0)
Non-sportfish												
Longnose Dace		117 (33.4)	15 (0.6)	103 (6.4)	273 (29.2)	551 (40.5)	30 (1.6)	32 (3.6)	227 (37.8)	143 (11.4)	73 (28.6)	117 (6.4)
Dace spp.		0 (0)	0 (0)	0 (0)	12 (1.3)	1 (0.1)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	1 (0.1)
Slimy Sculpin		0 (0)	13 (0.5)	11 (0.7)	62 (6.6)	39 (2.9)	6 (0.3)	0 (0)	1 (0.2)	12 (1)	11 (4.3)	101 (5.5)
Torrent Sculpin		0 (0)	1 (0)	1 (0.1)	0 (0)	0 (0)	3 (0.2)	0 (0)	0 (0)	0 (0)	0 (0)	4 (0.2)
Prickly Sculpin		0 (0)	0 (0)	0 (0)	0 (0)	2 (0.1)	0 (0)	0 (0)	0 (0)	2 (0.2)	0 (0)	1 (0.1)
Sculpin spp.		23 (6.6)	16 (0.7)	65 (4.1)	34 (3.6)	165 (12.1)	99 (5.3)	130 (14.5)	46 (7.7)	189 (15)	23 (9.0)	14 (0.8)
Sucker spp.		2 (0.6)	4 (0.2)	26 (1.6)	166 (17.8)	54 (4)	9 (0.5)	16 (1.8)	32 (5.3)	42 (3.3)	8 (3.1)	25 (1.4)
Redside Shiner		0 (0)	112 (4.6)	8 (0.5)	15 (1.6)	0 (0)	0 (0)	7 (0.8)	0 (0)	3 (0.2)	18 (7.1)	3 (0.2)
Northern Pikeminnow		0 (0)	0 (0)	2 (0.1)	0 (0)	15 (1.1)	7 (0.4)	1 (0.1)	1 (0.2)	0 (0)	8 (3.1)	1 (0.1)
Lake Chub		0 (0)	0 (0)	0 (0)	1 (0.1)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Peamouth Chub		0 (0)	0 (0)	6 (0.4)	6 (0.7)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	2 (0.1)
Unidentified		75 (21.4)	20 (0.8)	105 (6.6)	4 (0.4)	13 (1)	114 (6.1)	0 (0)	0 (0)	92 (7.3)	31 (12.2)	1,310 (71.9)
All Species Total		**350**	**2,406**	**1,598**	**933**	**1,361**	**1,865**	**896**	**600**	**1,261**	**255**	**1,822**

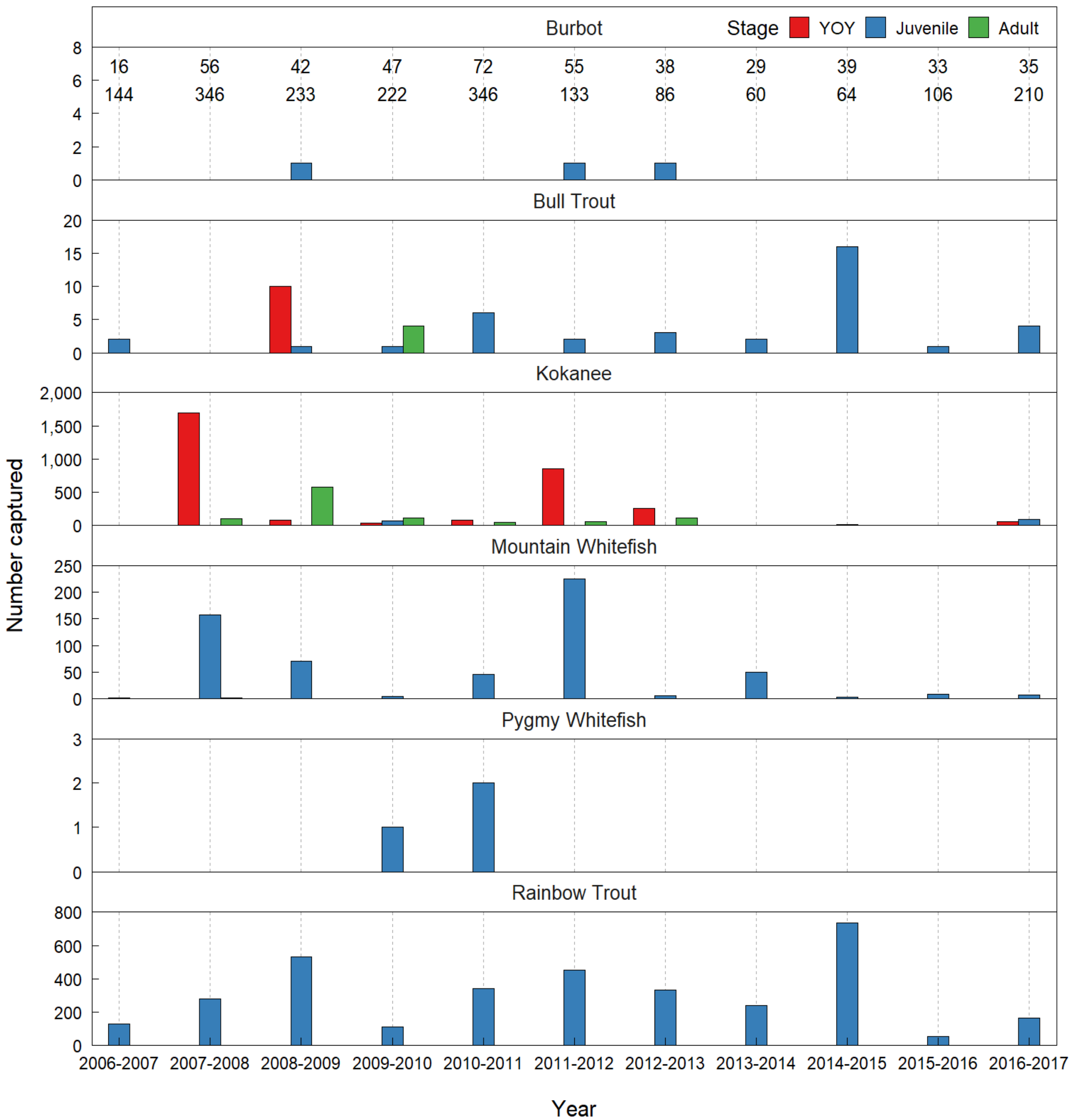


Figure 5: Abundances of sportfish species, separated by life stage, observed in stranding assessments between 2006 and 2017. Note the different y-axis scales among panels. On the uppermost panel, the numbers of sampled sites and pools are provided in the first and second lines, respectively.

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 overall mean area exposed during pre-WUP operations was 17.4 km², in comparison to 12.5 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 in general, lower, especially between 2013 and 2016. 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- period than in the post- period (0.45 and 0.29 km², respectively). The difference was statistically significant (1-way ANOVA; $P<0.001$). Annually, mean exposed areas ranged from 0.07 km² (2014 stranding year) to 0.22 km² (2007 and 2008 stranding years; Figure 7).

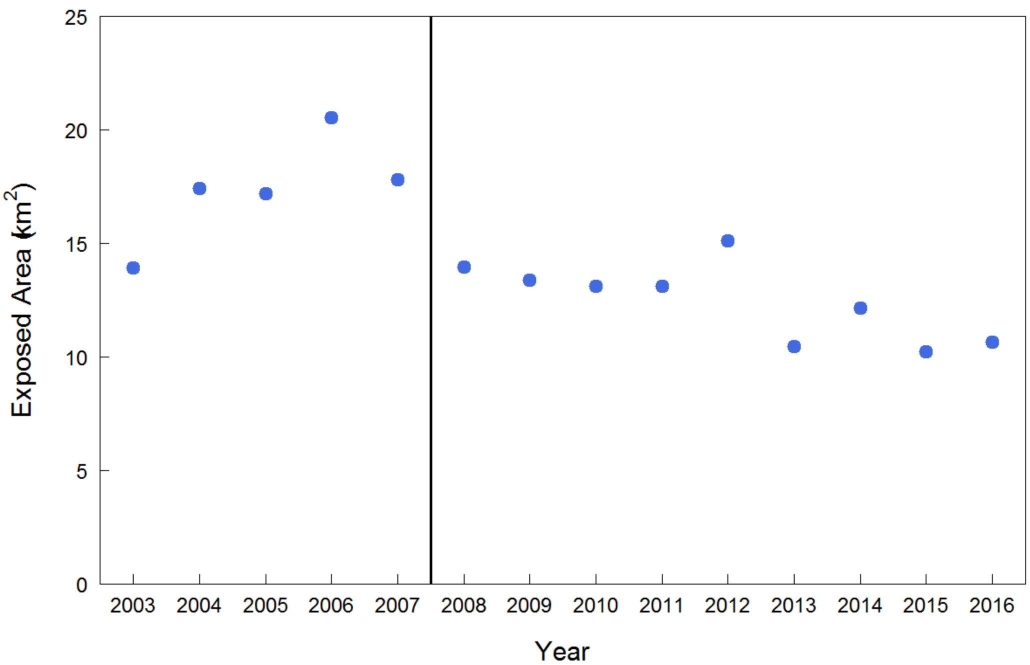


Figure 6: Total area exposed by all annual reductions in the LDR by year of operations, calculated using DRL discharge. The vertical line denotes the beginning on WUP flows in 2008. Note that label on Y axis denotes study year not calendar year.

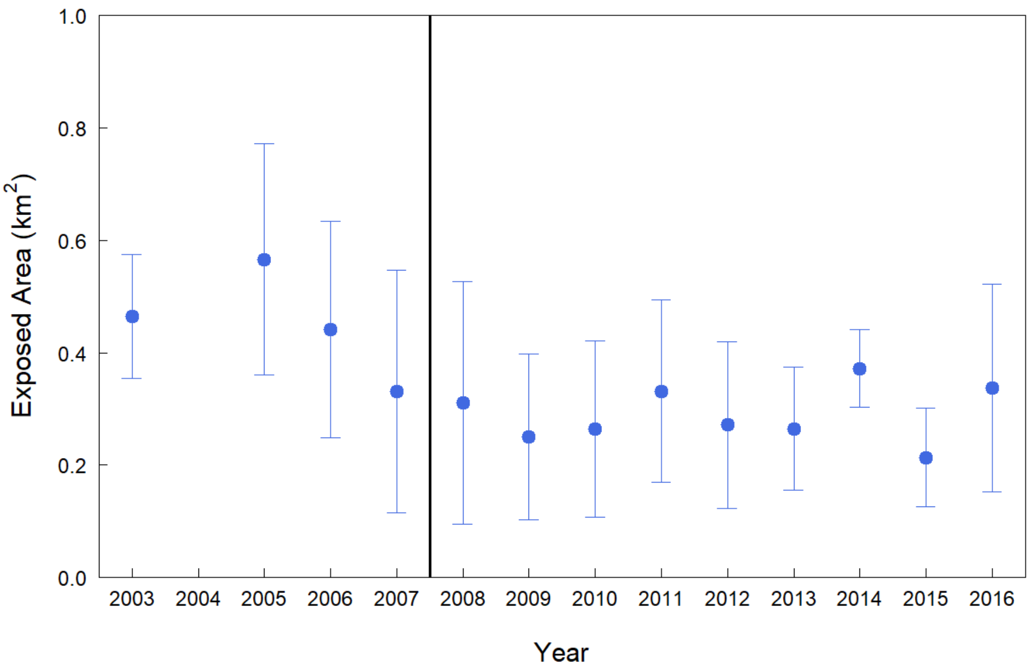


Figure 7: Mean area exposed by all annual reductions in the LDR by year of operations, calculated using DRL discharge. Bars represent 1 standard deviation. The vertical line denotes the beginning on WUP flows in 2008. Label on y axis denotes study year, not calendar year.

Interannual variability in mean discharge as assessed at the gauge at DRL, overall was higher 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 2017). However, in 2015, the DRL discharge was increased to approximately 250 m^3/s (8829 f^3/s: Golder 2017), resulting in high interannual variability during the October-January period (Figure 8). Decreased discharge variability post-WUP was recorded in March, where discharge trend changed from gradual increase pre-WUP to a stable flow or a slight gradual decrease post-WUP.

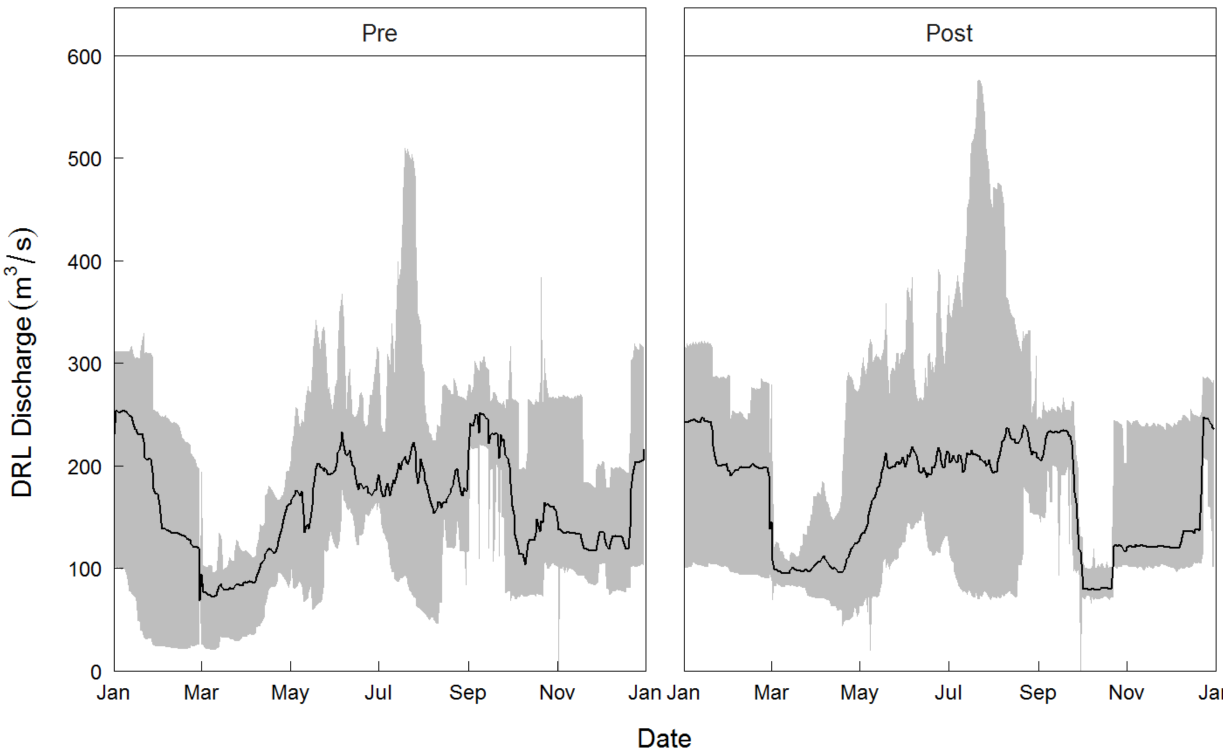


Figure 8: Minimum, maximum (grey ribbon) and mean (black line) discharge as measured at the WSC DRL gauge in the LDR by month during pre-WUP operations (2002 - 2007) and post-WUP operational implementation (2008 - 2016).

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, during 2013-2016, 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 in all years examined.

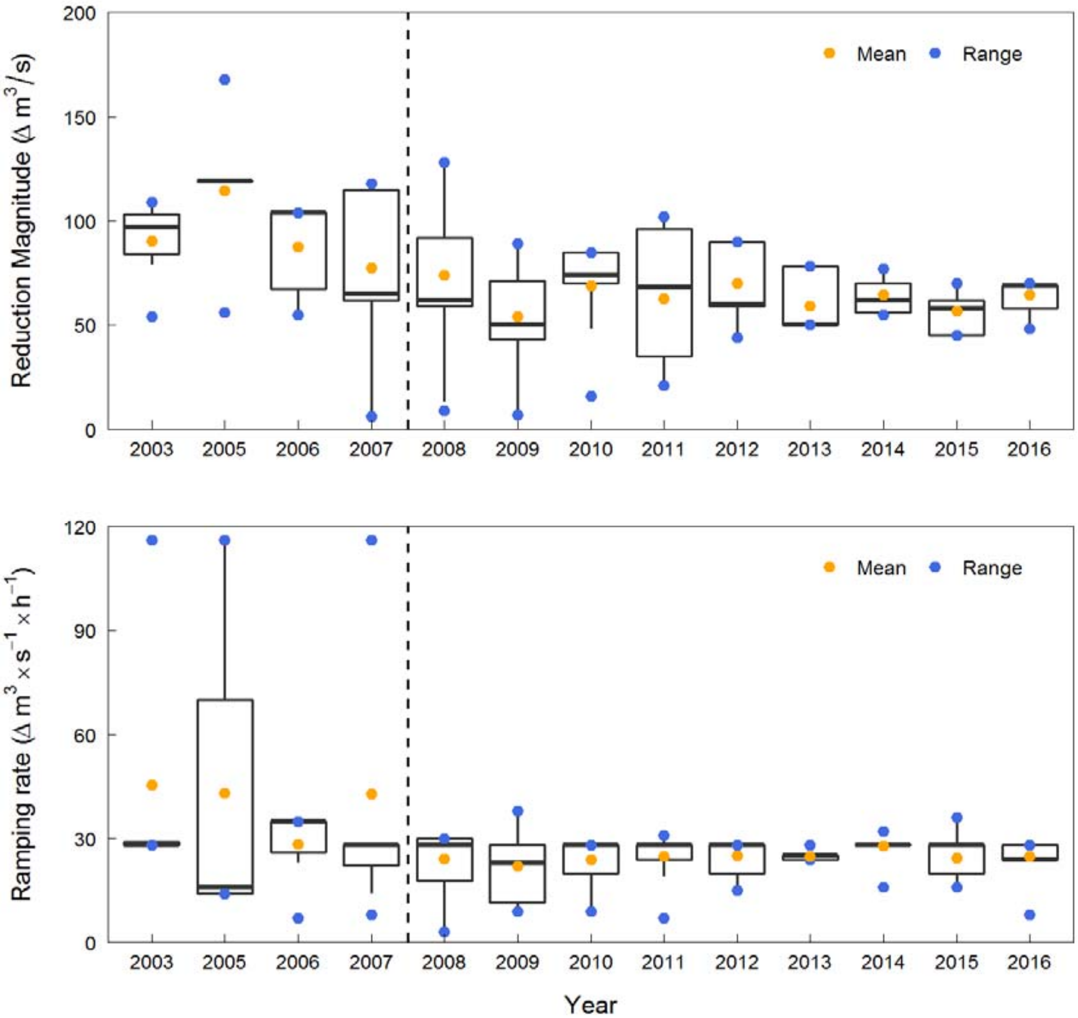
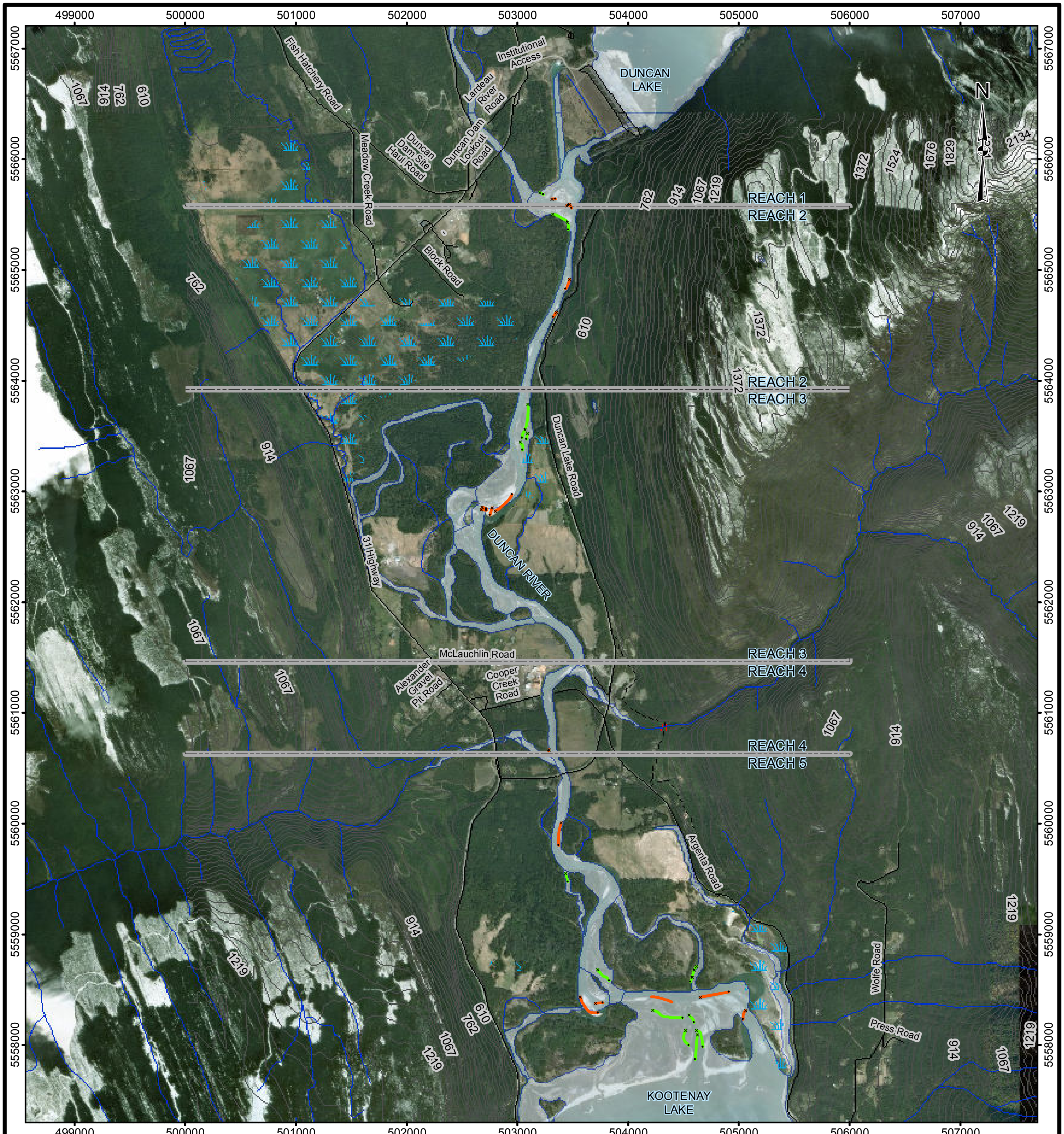


Figure 9: Boxplots of reduction magnitude (Δm³/s; top panel) and ramping rates (Δm³ s^{-1} h^{-1}; bottom panel) by year. Each box represents the 25th and 75th quantiles (bottom and top lines, respectively), and the median (middle bold line); whiskers extend to 1.5 times the interquartile distance. Yearly mean, minimum, and maximum values are shown as individual points.

3.4 Fish Abundance Assessment

Fish abundance assessment snorkel sites from study Years 6 - 9 are presented in Figures 10-1 to 10-4. A total of 57 sites and 6,807 m of shoreline were surveyed during the Year 9 (2016) snorkeling fish abundance assessment, with a total of 938 fish counted across all sites and strata (Table 17). The lowest mean counts of Mountain Whitefish (8.9 fish/site) were recorded in shallow, slack sites, whereas deep sites had high mean counts of Mountain Whitefish – 14.2 fish/site in fast sites, and 12.1 fish/site in slack sites. The lowest mean counts of Rainbow Trout (2.1 fish/site) were recorded in deep, slack sites, whereas shallow sites had higher Rainbow Trout mean counts, with 11.6 fish/site at slack sites, and 4.1 fish/site at fast sites.



LEGEND

SNORKEL SITE LOCATION (2013)

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

SCALE 1:50,000 KILOMETRES

REFERENCE

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

DATUM: NAD83 PROJECTION: UTM ZONE 11

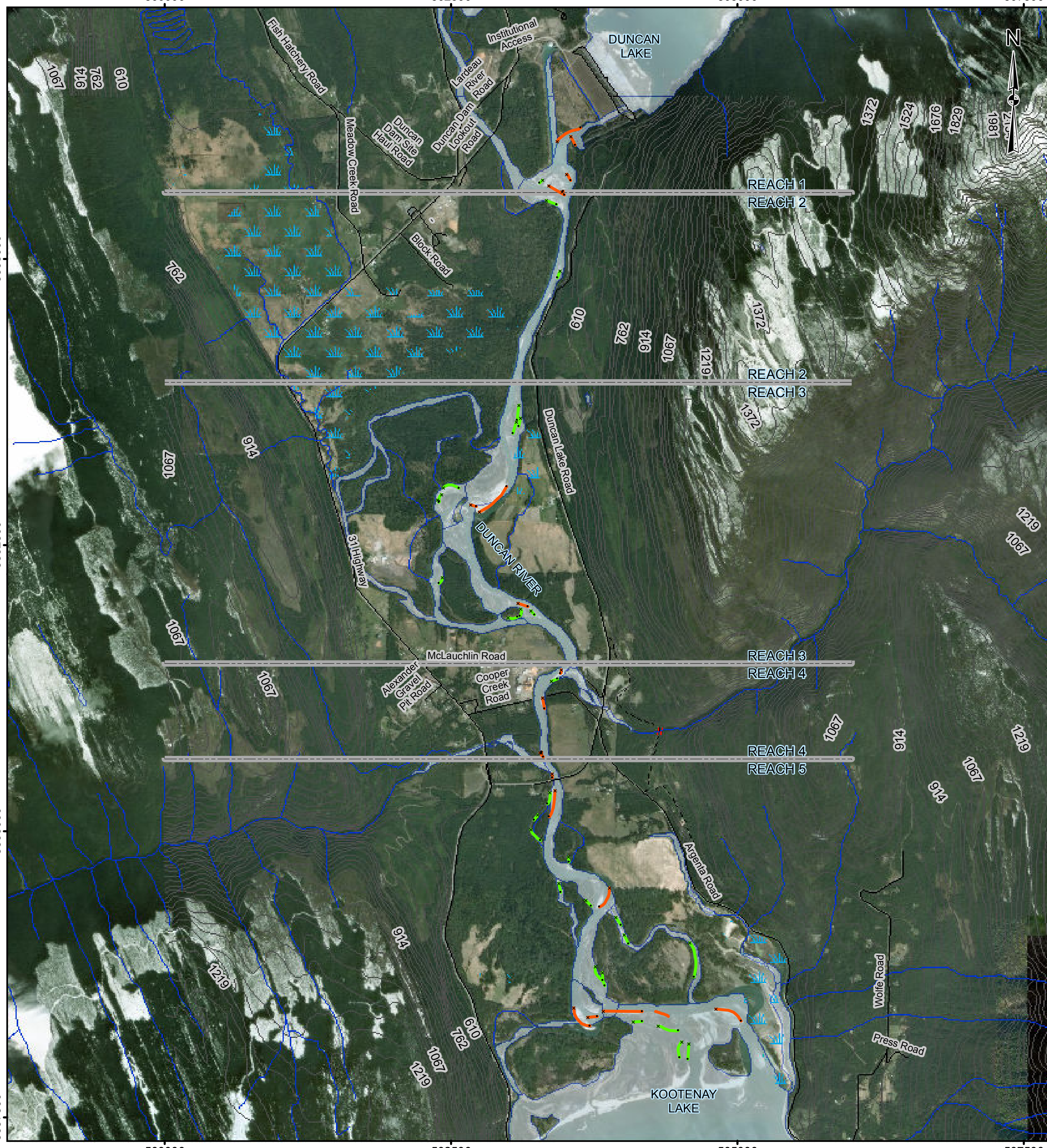
PROJECT
BC HYDRO
DDMMON-16 LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING PROGRAM

TITLE
SNORKEL SITE LOCATIONS (SEPT. 2013)

GOLDER

PROJECT		12-1492-0117	FILE No.	
DESIGN	BH	14 Apr 2014	SCALE AS SHOWN	REV. 0
GIS	KLN	14 Jun 2018	**FIGURE: 10.1**	
CHECK	BH	05 Dec 2018		
REVIEW	SR	05 Dec 2018		

Document Path: \\golder.gds\gal\burnaby\CAD-GIS\Bur-Graphics\Projects\2012\1492\12-1492-0117\GIS\Mapping\MXD\Fish\2018\Fig10-1_1214920117_Snorkel site locations_pre2014Data20181205.mxd



LEGEND

SNORKEL SITE LOCATION (2014)

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

1 0 1

SCALE 1:50,000 KILOMETRES

REFERENCE

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

DATUM: NAD83 PROJECTION: UTM ZONE 11

PROJECT

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

TITLE

SNORKEL SITE LOCATIONS (SEPT. 2014)

GOLDER

PROJECT		12-1492-0117	FILE No.	
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GIS	JG	14 JUN 2018	FIGURE: 10.2	
CHECK	BH	05 DEC 2018		
REVIEW	SR	05 DEC 2018		

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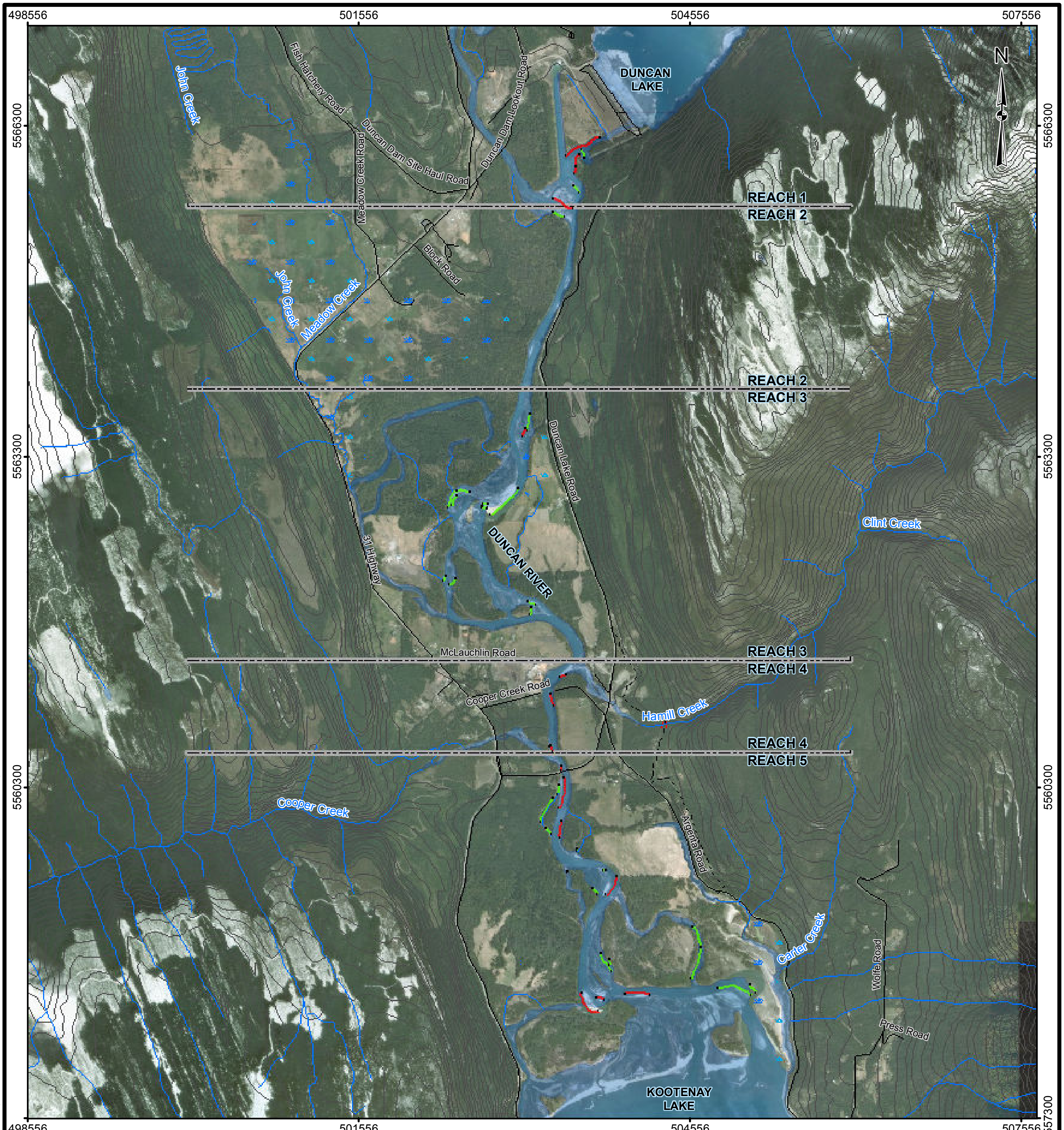

Document Path: \\golder.gds\gal\burnaby\CAD-GIS\Bur-Graphics\Projects\2012\1492\12-1492-0117\GIS\Mapping\MXD\Fish\2018\Fig10-3_1214920117_Snorkel site locations_pre2015Data_20181205.mxd

LEGEND

SNORKEL SITE LOCATION (2015)

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

1 0 1

SCALE 1:50,000 KILOMETRES

REFERENCE

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

PROJECT

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

TITLE

SNORKEL SITE LOCATIONS (SEPT. 2015)

GOLDER

PROJECT		1535517	FILE No.	
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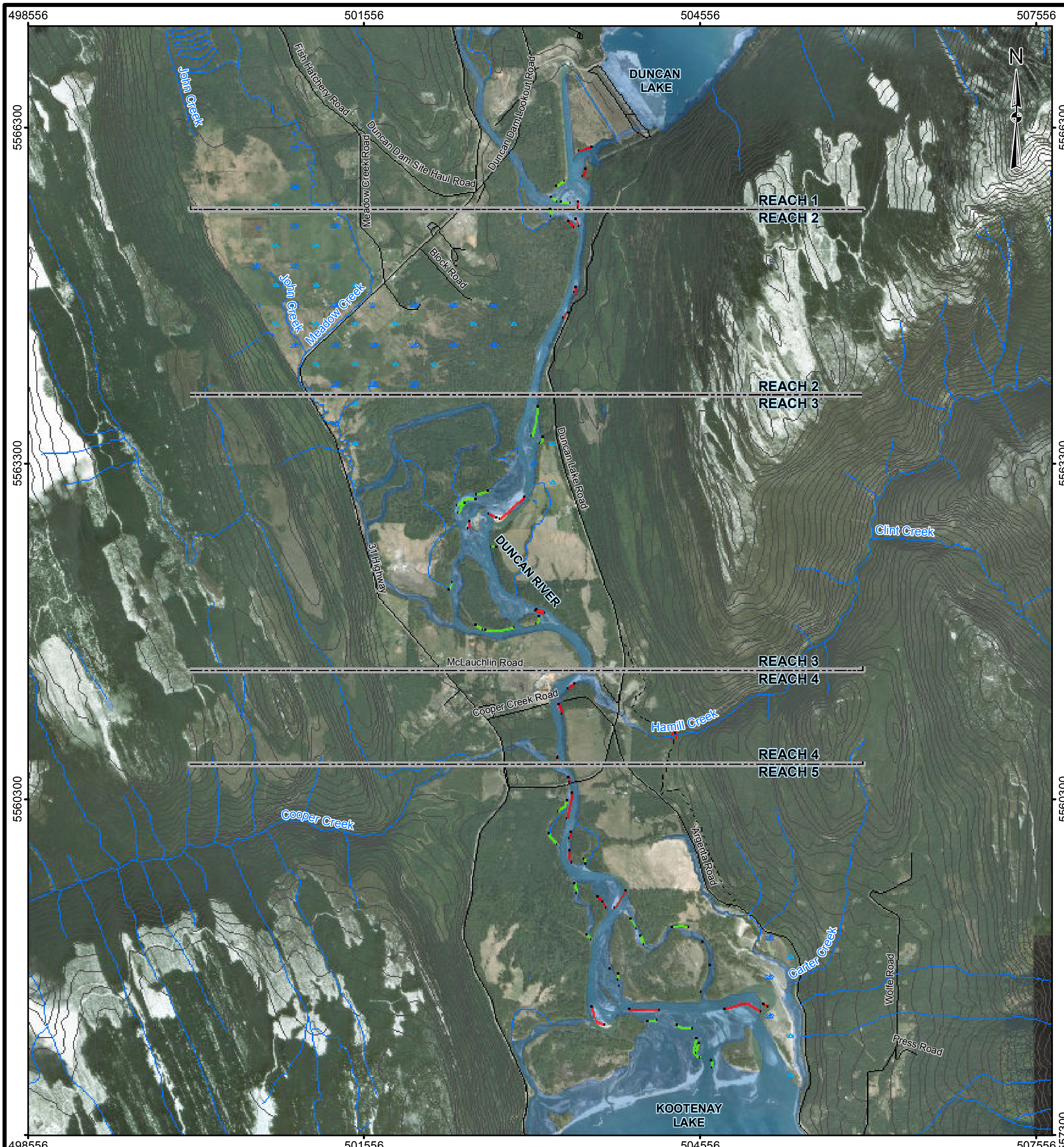
GIS	JG	14 JUN 2018	FIGURE: 10.3	
CHECK	BH	05 Dec 2018		
REVIEW	SR	05 Dec 2018		



LEGEND

SNORKEL SITE LOCATION (2016)

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

1 0 1

SCALE 1:50,000 KILOMETRES

REFERENCE

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

DATUM: NAD83 PROJECTION: UTM ZONE 11

PROJECT

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

TITLE

SNORKEL SITE LOCATIONS (SEPT. 2016)

GOLDER

PROJECT		1535517	FILE No.
DESIGN	BH	14 JUN 2018	SCALE AS SHOWN — REV. 0
GIS	JG	14 JUN 2018	FIGURE: 10.4
CHECK	BH	05 DEC 2018	
REVIEW	SR	05 DEC 2018	

Table 17: Summary of fish counts across depth and flow strata, as recorded from Year 9 (September 2016) snorkeling surveys.

Stratum	Depth	Flow	NSites	Mountain Whitefish			Rainbow Trout		
				N	Mean	SD	N	Mean	SD
1	Shallow	Slack	14	125	8.9	12.8	57	4.1	5.0
2	Shallow	Fast	17	154	9.1	11.4	198	11.6	17.4
3	Deep	Slack	11	133	12.1	18.7	26	2.4	5.9
4	Deep	Fast	15	213	14.2	9.9	32	2.1	2.4
Total				**625**			**313**		

Fish densities recorded in 2016 were generally lower than those from 2013 and 2014, but comparable to densities recorded in 2015 (Figure 11). Mountain Whitefish densities were high in deep, fast habitats, similar in both shallow and deep slack habitats, and lowest in shallow fast habitats. Rainbow Trout densities were lowest at deep, fast areas, and highest in shallow, slack areas.

The variability of fish density within strata was high throughout the years (Figure 11). In 2016, Mountain Whitefish zero densities accounted for 12% to 29% of the cases, calculated across strata. Non-zero densities ranged from 0.001 fish/m² to 0.09 fish/m². Rainbow Trout zero densities accounted for 14% to 64% or the cases, calculated across strata. Non-zero densities ranged from 0.0004 fish/m² to 0.06 fish/m².

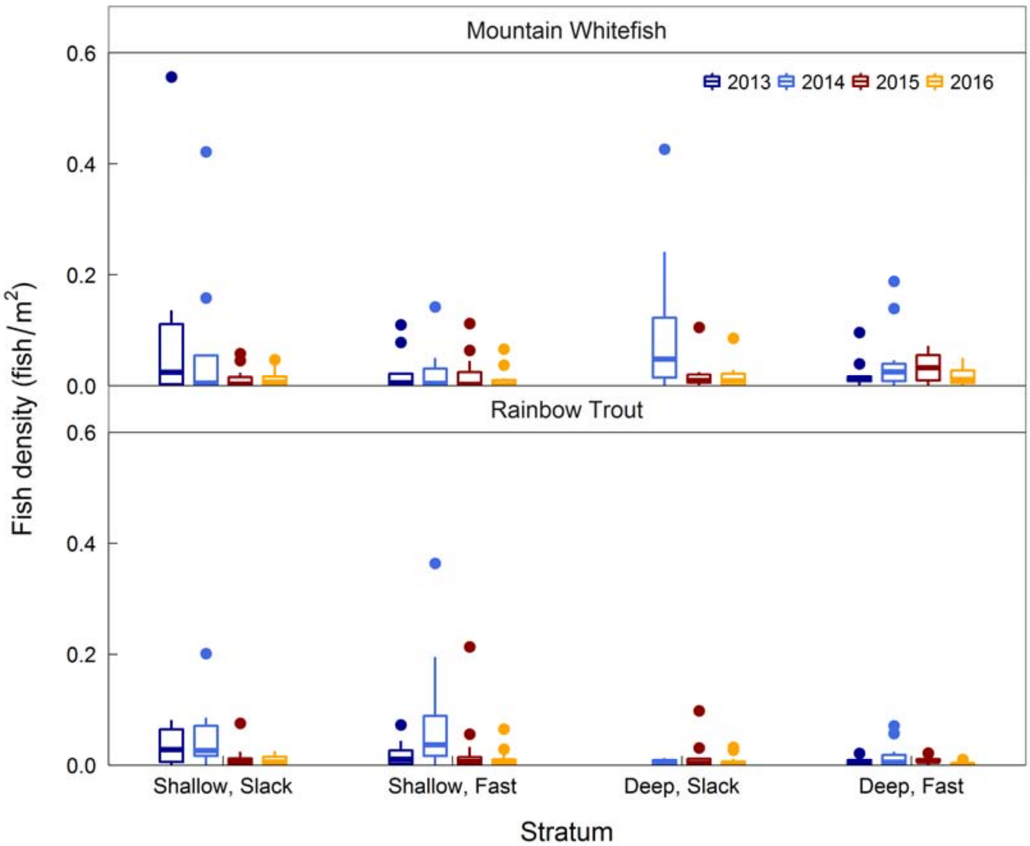


Figure 11: Boxplots of density (fish/m²) across species, depth, and flow strata for 2013-2016 data. Each box represents the 25th and 75th quantiles (bottom and top lines, respectively), and the median (middle bold line); whiskers extend to 1.5 times the interquartile distance; outliers are shown as individual points.

For both Mountain Whitefish and Rainbow Trout, abundance was modeled as a function of depth, year, and a random site effect. The effect of depth differed between Rainbow Trout and Mountain Whitefish – there was a significant reduction in Rainbow Trout log density at deep sites (estimated coefficient of -1.21, 95% CRI of -1.74 and -0.71). On the other hand, there was not a significant difference in Mountain Whitefish log densities between shallow and deep sites (estimated coefficient of 0.88, 95% CRI of 0.30 and 1.50).

Abundance estimates for Rainbow Trout were considerably higher in 2014 than in other sampling years (Figure 12). Rainbow Trout abundance in the deep strata was estimated to be low, ranging from 581 fish in 2016 to 3,163 fish in 2014. Abundance estimates for Mountain Whitefish were similar between 2013 and 2014, and between 2015 and 2016, however a reduction in estimated abundances was recorded between 2014 and 2015. The highest mean abundances for the species were estimated in the deep stratum, ranging from 4,392 fish in 2015 to 9,651 fish in 2013.

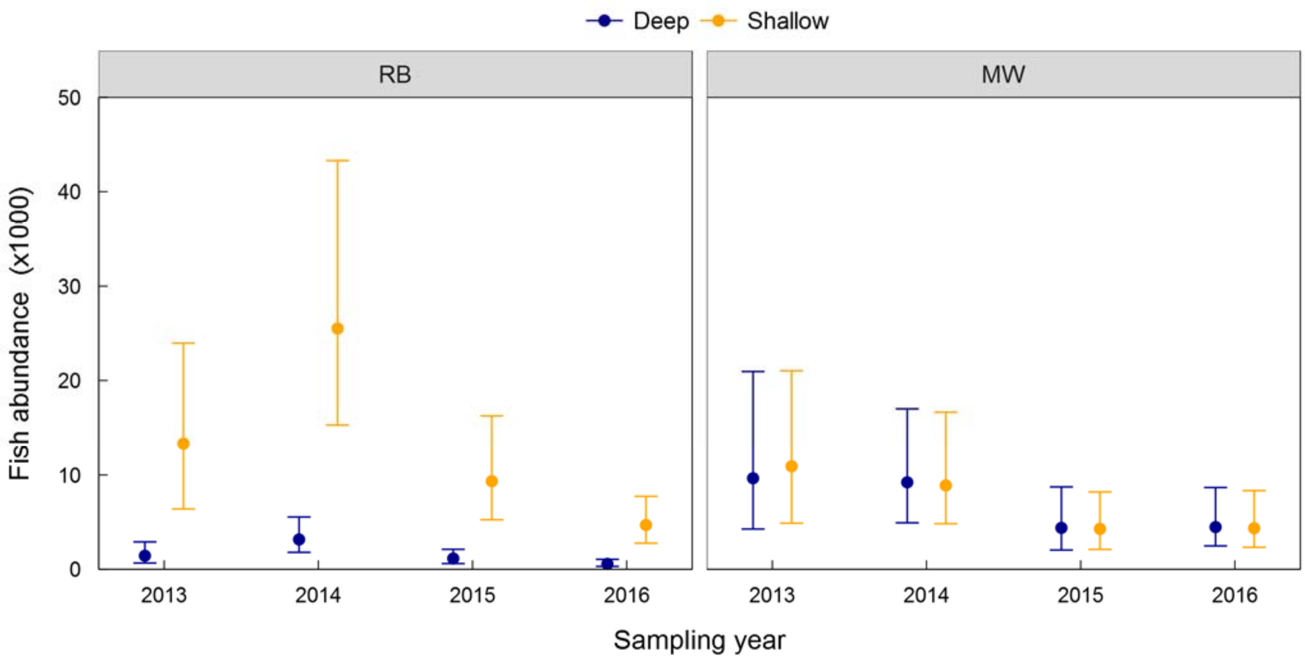


Figure 12: Median Rainbow Trout and Mountain Whitefish abundance (density x stratum area) and their respective 95% credibility intervals, plotted by sampling year and depth strata.

Total abundance estimates for Rainbow Trout ranged from 5,287 in 2016 to 29,351 in 2014 (Table 18). Overall, estimates decreased annually since the 2014 peak of estimated abundance. Mountain Whitefish abundance in 2016 was similar to the 2015 estimates. Generally, Mountain Whitefish abundance decreased from approximately 20,000 in Year 6 and Year 7 to approximately 9,000 in Year 8 and Year 9. These findings are similar to the results reported in Year 8 report (Golder 2017).

Table 18: 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	Species	
	RB	MW
Year 6 (Fall 2013)	14,854 (7,400 – 27,386)	20,038 (10,061 – 39,369)
Year 7 (Fall 2014)	29,351 (17,531 – 48,187)	18,632 (10,409 – 31,827)
Year 8 (Fall 2015)	10,456 (5,871 – 18,171)	8,781 (4,745 – 15,535)
Year 9 (Fall 2016)	5,287 (3,184 – 8,700)	9,008 (5,345 – 14,635)

3.5 Fish Stranding Assessment

Pool stranding estimates in the following sections refer to both Rainbow Trout and Mountain Whitefish populations.

3.5.1 Presence of Pools

The slope of each sample taken throughout seven years of stranding assessments (Years 3 to 9: 2010-2016) was calculated using the elevation models for the area. Slopes ranged from 0% to 60%, however all values above 20% (a total of 4 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 13). While pool densities in random sites exhibited slightly higher variation in comparison to index sites in some years (e.g., 2010, 2016), the majority of recorded pool densities were fairly low, often lower than those recorded at index sites (Figure 13).

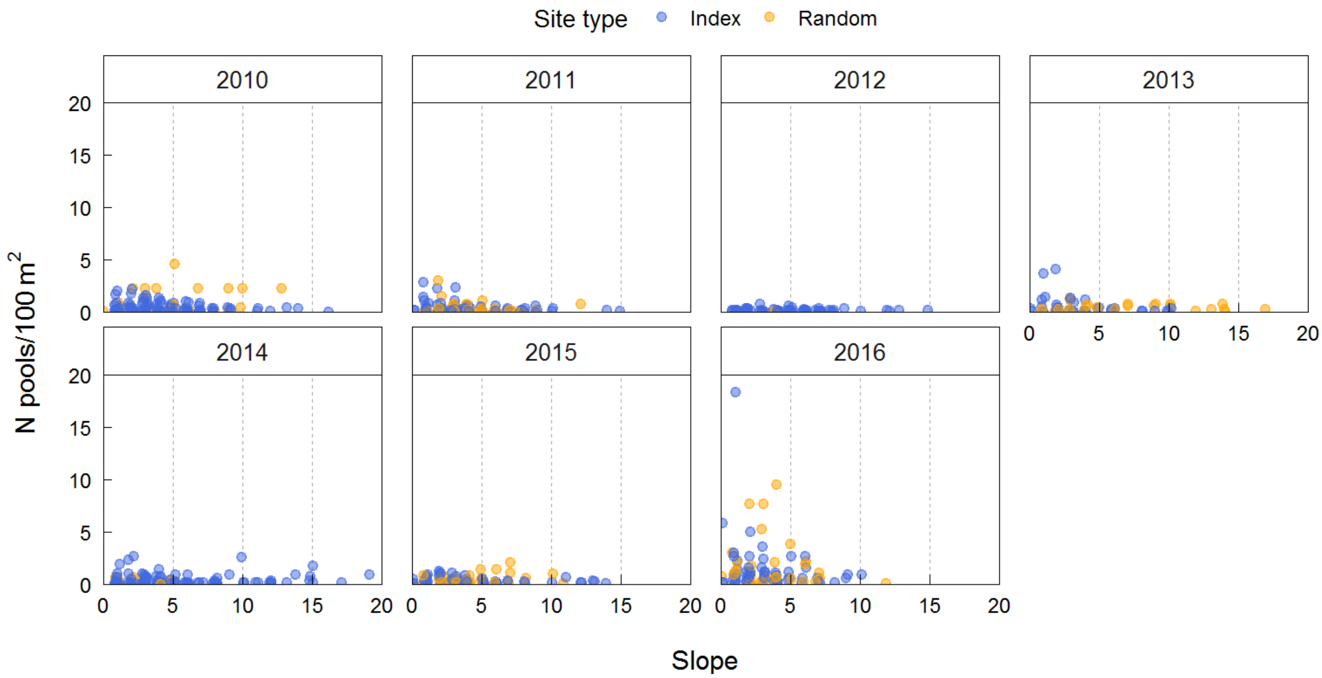


Figure 13: Density of pools recorded per reduction vs. habitat slope as a continuous variable, 2010-2016

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 operation targets, the mean number of pools that formed during the stranding surveys between Year 5 and Year 9 was generally similar, ranging between 256 and 344 pools. In comparison, in Year 3 and Year 4, the reduction-level estimates of pools ranged between 120 and 350 pools (Figure 14).

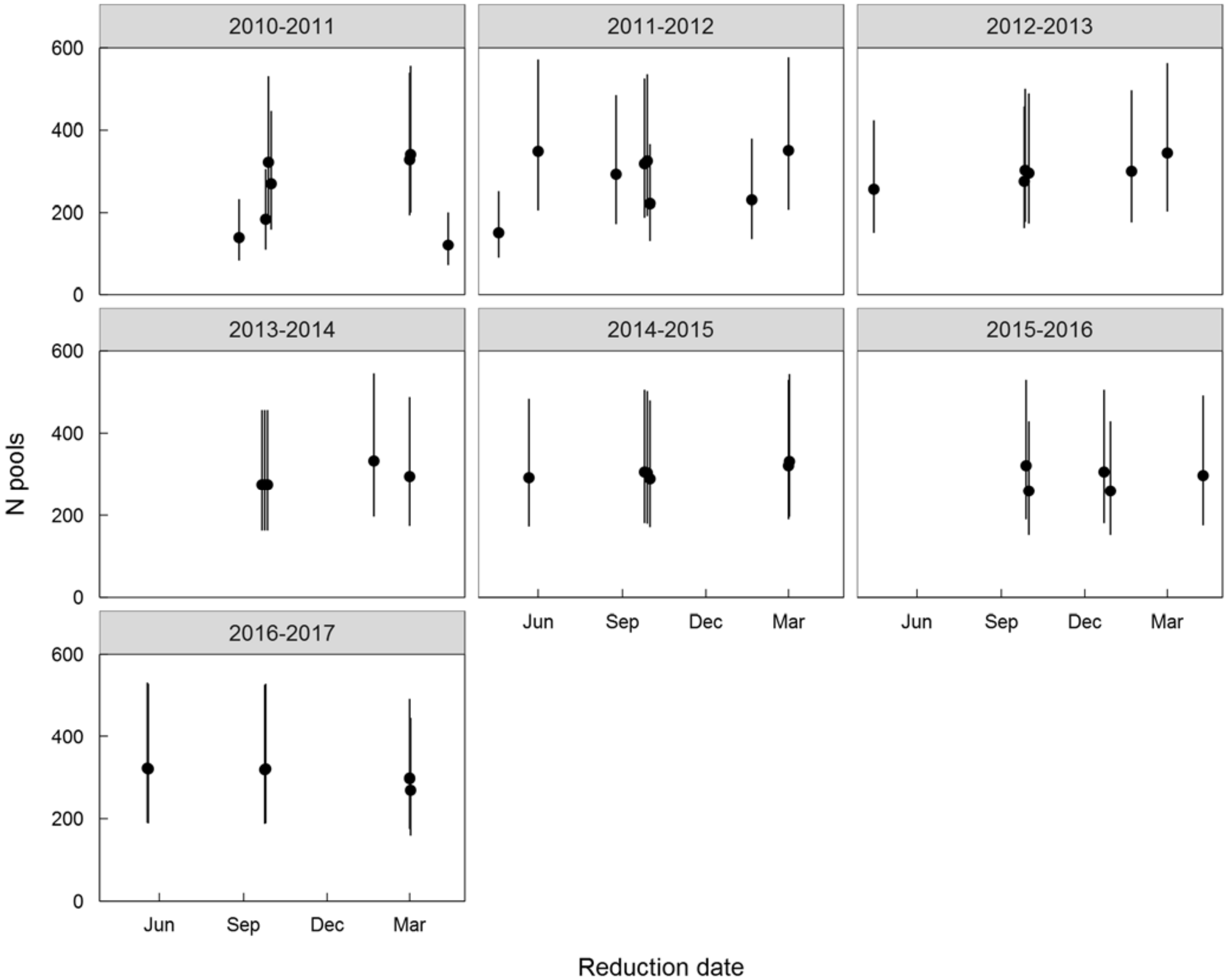


Figure 14: Mean estimates of pool numbers formed during the 2010-2016 stranding events, plotted by reduction and stranding year. Error bars are 95% credibility intervals.

3.5.2 Pool Stranding

For the purposes of the statistical analyses, the efficiency of visual counts or dip netting, which were primarily conducted in pools with low complexity, was assumed to be 100%. Catchability using backpack electrofishing was estimated to be 0.588 for Rainbow Trout (median value; 95% credibility interval of 0.528-0.641) and 0.441 for Mountain Whitefish (median value; 95% credibility interval of 0.315-0.564).

The variability in the number of fish stranded per pool was similar throughout the different slopes (Figure 15). This indicated that slope did not affect stranding of fish in pools. A large difference in pool stranding of Rainbow Trout was observed with season, where pool stranding was substantially higher in the fall reductions.

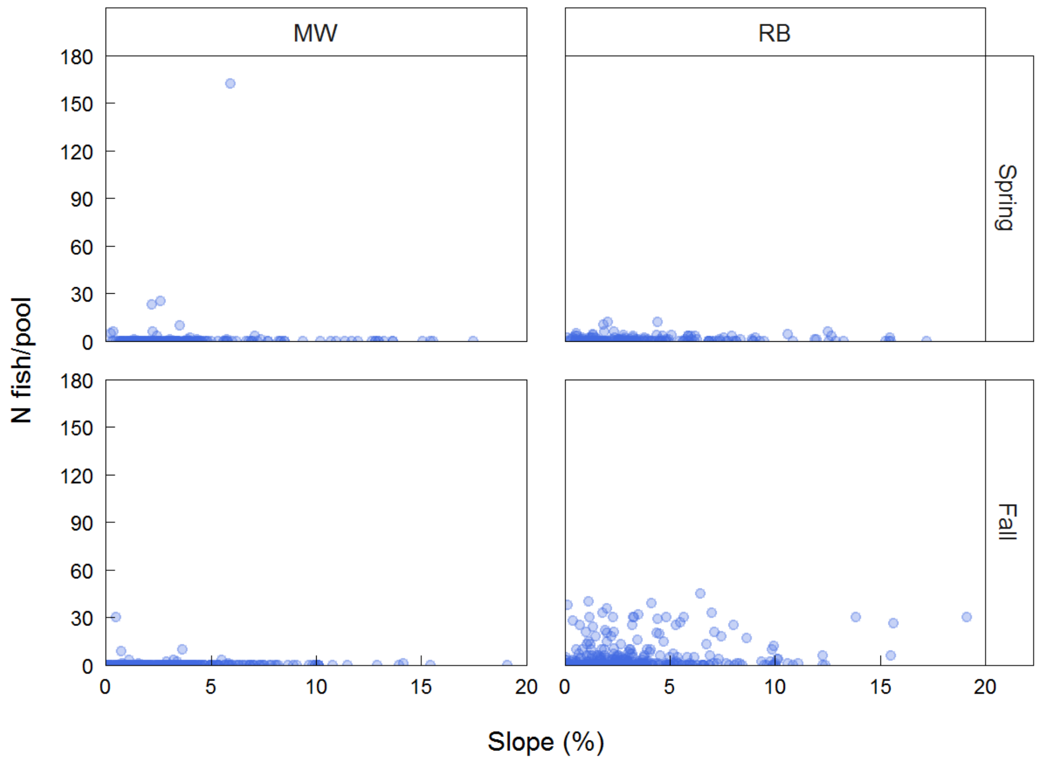


Figure 15: Number of collected fish per pool, plotted by slope and colour-coded by season, 2010 – 2016.

The density of pool-stranded fish differed by dominant substrate size and by species (Figure 16). Generally, 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 overall similar across different substrate sizes, apart from pools with substrate of large gravel or larger, where few pool-stranded Rainbow Trout were recorded (Figure 16).

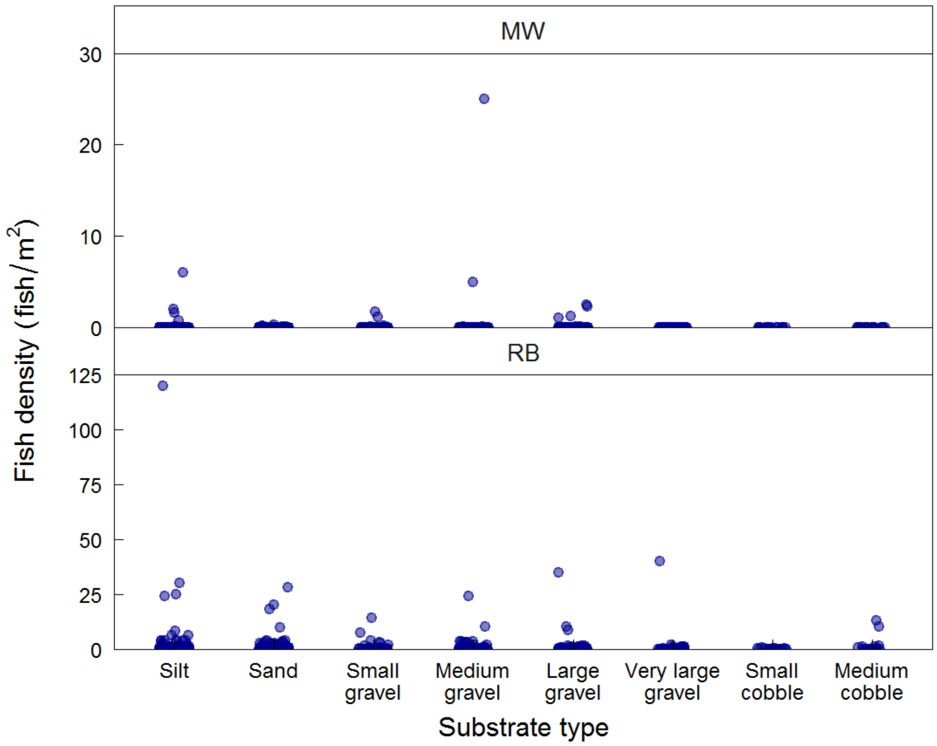


Figure 16: Scatter plot of pool-stranded fish density (fish/m²) vs. dominant pool substrate size, 2006-2016, plotted by species.

The median number of Rainbow Trout fry per pool for the spring season (January to June) was estimated to be 0.620 (0.328 – 1.133) fish/pool (Figure 17). In contrast, the median number of Rainbow Trout juveniles stranded per pool in the fall (July to December) was estimated at 3.962 (2.237 – 7.278). The effect of season on pool stranding of Rainbow Trout was found to be significant ($p < 0.001$), with median fall stranding estimates six times higher than those for winter/spring. For Mountain Whitefish, no strong effect of season was found ($p=0.5$), although fall stranding was estimated to be lower (median of 0.028 fish/pool, in comparison with 0.052 fish/pool in the spring).

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 18). Generally, Mountain Whitefish estimates of pool stranding were lower than those of Rainbow Trout. The spike of Mountain Whitefish presence in spring 2012 resulted from a single pool with 162 Mountain Whitefish (Figure 15) sampled in March 2012. The high estimate of pool stranding in spring 2011 (during 2010-2011 stranding year) was due to finding 36 Mountain Whitefish stranded in pools on a single reduction event in early April 2011.

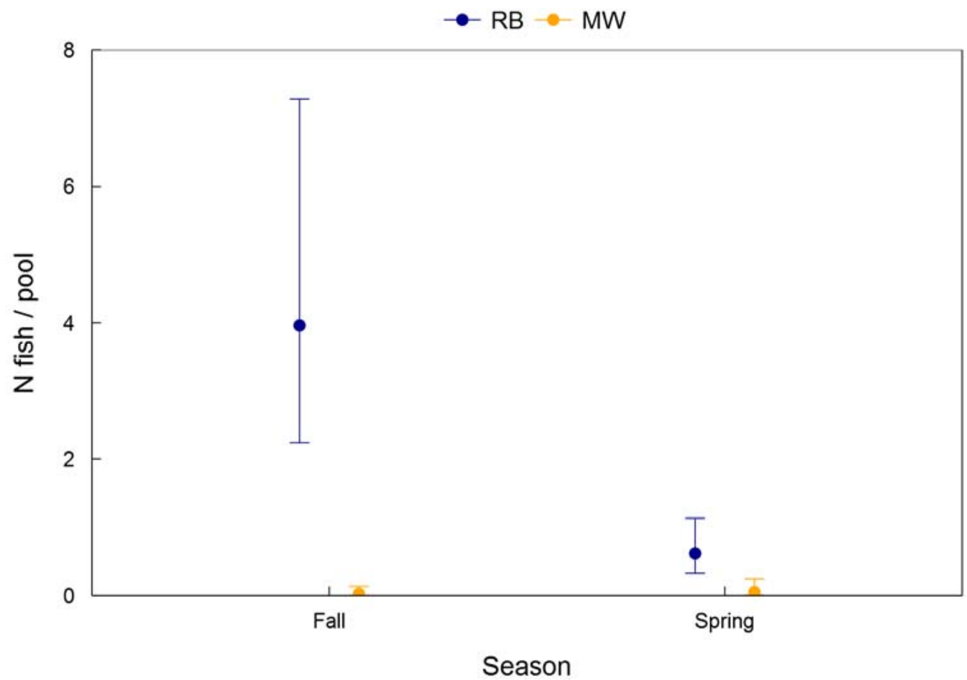


Figure 17: Median estimates of counts of Rainbow Trout and Mountain Whitefish per pool, by season, 2010 - 2016; 95% credibility intervals are provided.

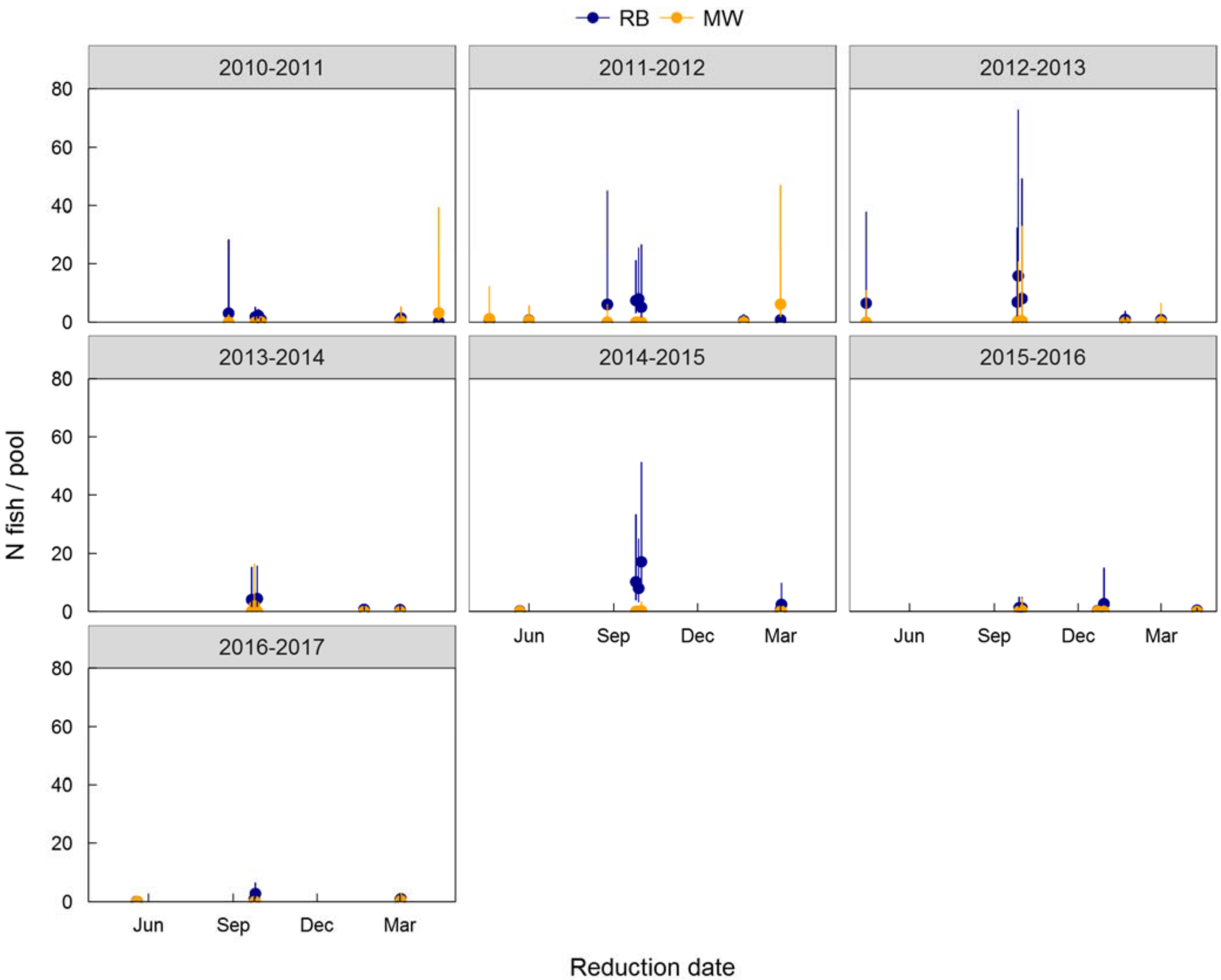


Figure 18: Median estimates of counts of pool-stranded Rainbow Trout and Mountain Whitefish during the 2010-2016 stranding events, plotted by reduction and stranding year. Error bars are 95% credibility intervals.

3.5.3 Interstitial Stranding

Between Year 4 (2011-2012) and Year 9 (2016-2017) of this program, 24 Rainbow Trout and 2 Mountain Whitefish were found to be interstitially stranded (Figure 19). Over the last four study years (Years 6 to 9) when interstitial sample methodology was standardized with transect sampling, only one interstitially stranded Rainbow Trout was observed (in Year 6; Golder 2015). All documented interstitially stranded fish were found on exposed areas with low slopes (≤7%) (Figure 20). The data scarcity related to interstitial stranding precluded in-depth modelling and estimation.

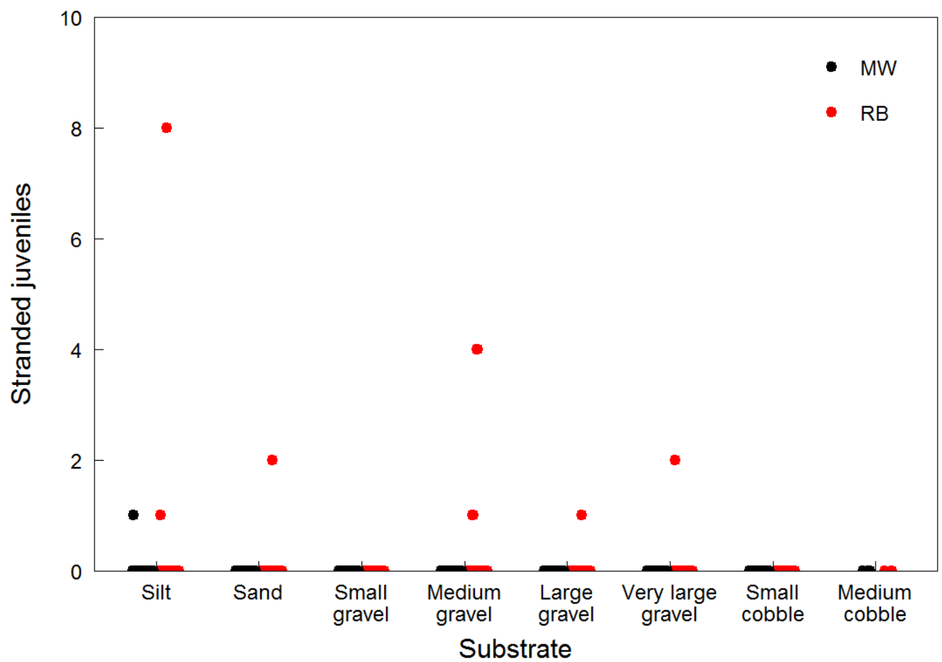


Figure 19: Counts of 2011-2016 interstitially stranded Mountain Whitefish and Rainbow Trout, plotted by substrate size.

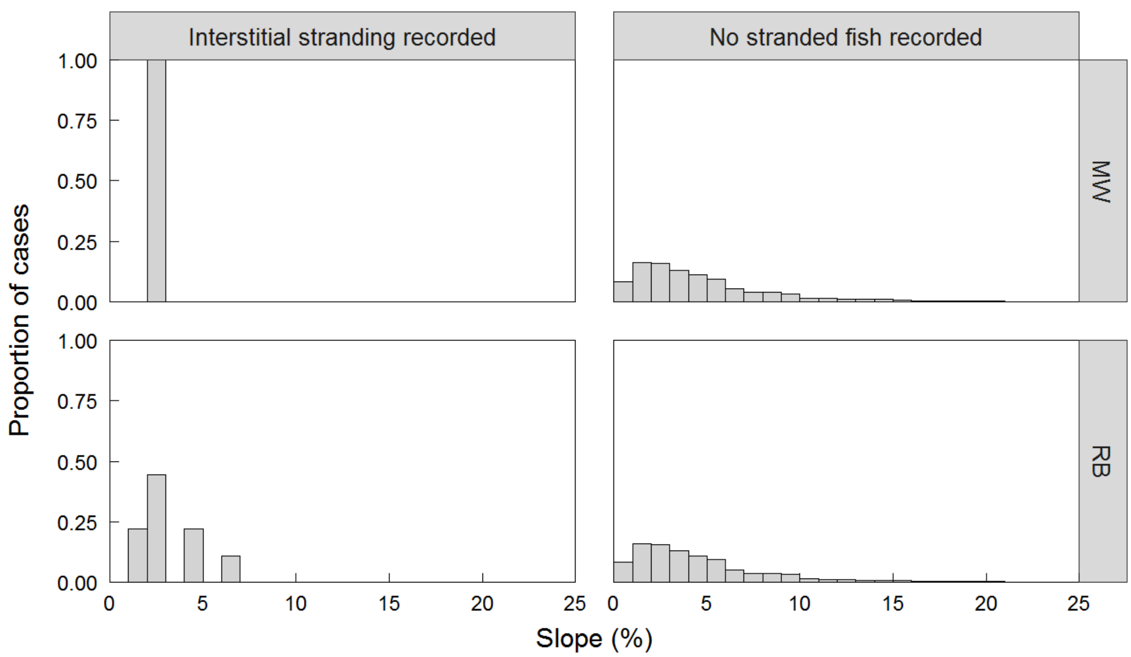


Figure 20: Histogram of 2011-2016 interstitially stranded Mountain Whitefish and Rainbow Trout, plotted by species and slope (%).

Total dewatered area is calculated using relationships for low-slope (≤4%) and high-slope (>4%) areas, and no relationship has been developed for these areas as a function of index/non-index sites.

3.5.4 Total Stranding Estimates

For Rainbow Trout, total pool stranding for the current year was estimated at 1833 (95% CRI of 939 - 3924 fish; Figure 21). Total annual pool stranding increased between 2010 and 2012, and generally decreased throughout 2013-2015. The highest stranding was estimated to have occurred in 2012-2013, with a mean estimate of 14,062 fish (95% CRI of 5,865-41,152 fish. For Mountain Whitefish, total annual pool stranding was generally low, with higher estimated stranding (and very wide 95% CRIs) estimated in 2011-2012 and 2012-2013 (Figure 21). Current total pool stranding was estimated at 158 (95% CRI of 35 - 1209). The highest stranding was estimated to have occurred in 2011-2012, with a median estimate of 3,257 fish (95% CRI of 991-19,717 fish). In all other stranding years, Mountain Whitefish pool stranding was estimated to range between 158 fish (Year 9) and 876 fish (Year 5).

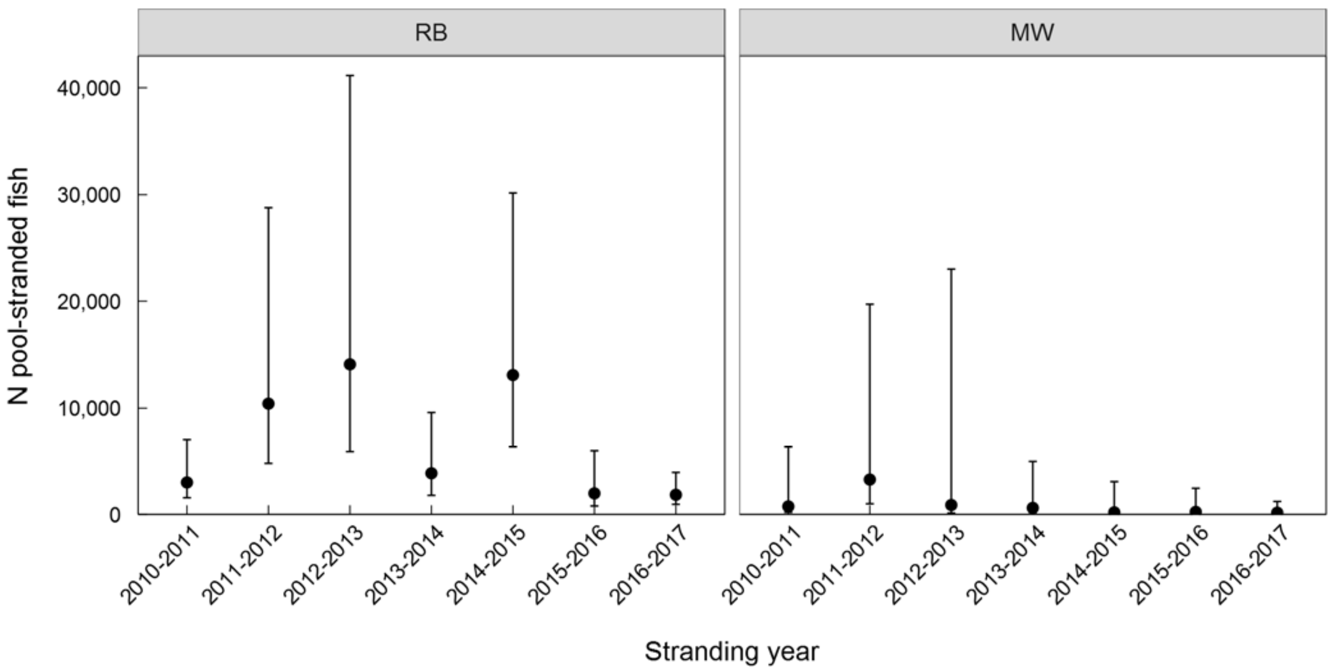


Figure 21: Median estimates of total pool-stranded Rainbow Trout and Mountain Whitefish during the 2010-2016 stranding years, plotted by stranding year. Error bars are 95% credibility intervals.

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 are 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 strand 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 dewater 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. During the Year 9 assessments, 14 different species were encountered (four sportfish and ten non-sportfish species), but Rainbow Trout was the only species of interest with substantial numbers of stranded individuals.

4.2.1 Pool and Interstitial Stranding Rates

Current estimates for the number of Rainbow Trout and Mountain Whitefish juveniles stranded in pools were relatively precise and relatively low. 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. Although seasonal effect on pool stranding for Mountain Whitefish was not statistically significant, fall stranding rates were lower than those estimated for winter/spring. Significant differences between substrate size within isolated pools and the density of pool stranded fish were not found.

Over these study years when interstitial sample methodology was standardized, very few interstitially stranded fish have been observed, and a relationship between stranded fish counts and substrate size was not observed. This relationship should continue to be re-evaluated in future years as more data are collected. While interstitial stranding is likely a contributing factor to overall fish stranding, the substantially higher numbers of stranded fish documented in pools and relatively precise estimates associated with pool stranding strongly indicates that the current pool stranding estimates are more representative of overall fish stranding in the Duncan River as a result of Duncan Dam operations.

4.2.2 Slope of Dewatered Area

The categories of low and high slope were based on values in the literature from previous stranding work (Bauersfeld 1978; Flodmark 2004). Based on the 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).

As the results from the current dataset suggest that slope did not have an effect on 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 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 re-evaluated yearly as more data are collected.

Based on the analysis of the current dataset, a relationship between slope and interstitial stranding may exist. Conversely, statistically significant relationships between interstitially stranded fish counts and slope in the current dataset were not found in Years 6 to 8 (Golder 2015, 2017a and 2017b). However, until a full estimation of exposed area by slope is available (for extrapolation to total interstitial stranding), the variable cannot be included in the models. A full examination of exposed area by slope is recommended in Year 10 (Section 5.0), and that the variable be included in the model during the analysis of the data set for that year.

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 and 2017b), 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 Year 8 (Golder 2017b) and the current study year, there was no significant statistical effect of index and random site on pool density, and subsequently pool stranding rates. The effect of index/random site on interstitial stranding was found to be significant only for Rainbow Trout. However, only two Mountain Whitefish have been found interstitially stranded; therefore, interstitial stranding estimates for the species are likely not reliable. Based on these analyses, index sites do not exhibit a significant bias toward higher stranding rates and therefore, hypothesis H_{01} cannot be rejected. In Year 10 stranding rates at both index and random sites should continue to be analyzed as the data set grows.

4.2.4 Rainbow Trout

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*. Over the four years analyzed, estimated Rainbow Trout abundance increased from 2013 to 2014, followed by a sharp decrease in 2015. This decrease could be linked to a decline in Lardeau River Gerrard Rainbow escapement into the Duncan River that has been identified (Andrusak and Andrusak 2015). Estimated juvenile Rainbow Trout abundance continued to decline between 2015 and current study year (2016). Abundance estimates for juveniles were substantially lower than those obtained in by the DDMMON-2 program in the fall of 2010 (Thorley et al. 2012). This suggests that the juvenile Rainbow Trout population in the LDR may have declined since 2010. This finding should be interpreted with caution as the models used in the individual programs were different.

Estimates for the number of Rainbow Trout juveniles stranded in pools obtained for this program were relatively precise and low. The very low numbers of interstitially stranded Rainbow Trout encountered during sampling invalidates in-depth modelling to calculate interstitial stranding estimates when determining DDM operations on

Rainbow Trout populations. Based on the relatively low pool stranding estimates in the current dataset, hypothesis H_{02} cannot be reasonably rejected. Therefore, it must be concluded that fish stranding as a result of DDM operations does not have a significant impact on juvenile Rainbow Trout populations.

4.2.5 Mountain Whitefish

The total abundance estimates for Mountain Whitefish obtained using the updated abundance model decreased from Years 6 to 8, while the current estimates for the current year were very similar to the estimates in Year 8. Similar to Rainbow Trout, current abundance estimates for Mountain Whitefish were substantially lower than those obtained in by the DDMMON-2 program in the fall of 2010 (Thorley et al. 2012). This suggests that the juvenile Mountain Whitefish population in the LDR may have declined since 2010. As the modelling used for the 2010 and current estimates were different, it is uncertain if this identified decline in juvenile Mountain Whitefish population is accurate. As documented in the DDMMON-2 program (Thorley et al. 2011), significant differences in juvenile Mountain Whitefish abundance within sidechannel and mainstem habitat were not identified in Year 6, and therefore abundance differences in these habitats were not examined in Years 7 and 8, or the present study year.

Similar to study Years 6 to 8 (Golder 2015, 2017a and 2017b), a seasonal effect on Mountain Whitefish stranding was not observed. In the current year, only 7 stranded Mountain Whitefish were documented, and encounters have been minimal 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. However, previous experimental stranding investigations indicated that large numbers of mountain whitefish could be stranded during rapid night time reductions in flow (Poisson and Golder 2010). Consequently, these conclusions are based on the assumption 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 9 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?):*
  - Similar to Year 7 and 8 results (Golder 2017a and 2017b), seasonal effect on pool stranding 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)

- Relationship between interstitially stranded fish counts and slope in the current dataset is unknown (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 (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 cannot be reasonably rejected (Section 4.2.4)
  - The continued stranding of low numbers of Mountain Whitefish will likely not result in a population level effect (Section 4.2.5)

With the refinements to the modelling methodology and the growth of the data set, the data scarcity associated with interstitial stranding of the target species remained unchanged. In order to reduce these uncertainties within the capabilities of the dataset, it is recommended that an estimation of exposed area by slope be incorporated into the fish stranding estimation (Section 6.0). With the analysis of slope and another year of data collection to increase the data set, the uncertainty related to pool stranding estimation is expected to decrease.

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 survive these factors is unknown (Golder and Poisson 2012).

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 possible, flow reductions at DDM should follow recommendations made by the Adaptive Stranding Protocol and the various studies conducted on the LDR. To better understand stranding related to the species of interest in the LDR, the interstitial stranding estimates for these species needs further refinement.
The refinements and other recommendations discussed in Section 6.0 will work towards resolving the lack of confidence around these estimates.

6.0 RECOMMENDATIONS

Recommendations from the current year (Year 9) of the DDMMON-16 Lower Duncan River Fish Stranding Impact Monitoring Program are as follows:

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

2) Expand the Year 10 data analysis to include a full estimation of exposed area by slope. This can be accomplished by using the existing Digital Elevation Model from the DDMMON 3 program to determine the slope of all potential areas that pose a risk to stranding fish within this program's study area. With this estimation, areas with high slope that do not pose a risk of stranding fish can be excluded from the analysis. Probabilities of pool formation and interstitial stranding could then be applied to the areas that remain to further refine estimation methodology; and

3) Remove the following 11 sites from the list of 50 potential stranding sites identified during this program. The risk to strand fish at these sites during DDM operations is negligible due to factors such as high gradient (over 20%), very low amounts of dewatered area during flow reductions, the attenuation of flow reductions as a result of Kootenay Lake levels, and inaccessibility for field crews.

   - M0.6-1.7L
   - M1.1-1.7R
   - S4.0R
   - S4.1R
   - M6.0R
   - M7.1-7.7L
   - M7.2-7.8R
   - S7.7R
   - M8.8-9.1R
   - S9.5R
   - S11.5R

These recommendations are designed to build on the current data set. The focus of study going forward should be on resolving the lack of confidence in the interstitially stranded fish estimates throughout the system, as well as ensuring that the abundance estimates obtained are as accurate as possible. As for Year 10 stranding assessments, sampling methods should remain such that comparisons with historical data can be maintained.

7.0 CLOSURE

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

Golder Associates Ltd.

Brad Hildebrand
Project Manager, Fisheries Biologist

Shawn Redden
Project Director, Associate, Senior Scientist

BH/SR/cmc

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APPENDIX A

Project Maps and Sampling Chronology

DRAFT

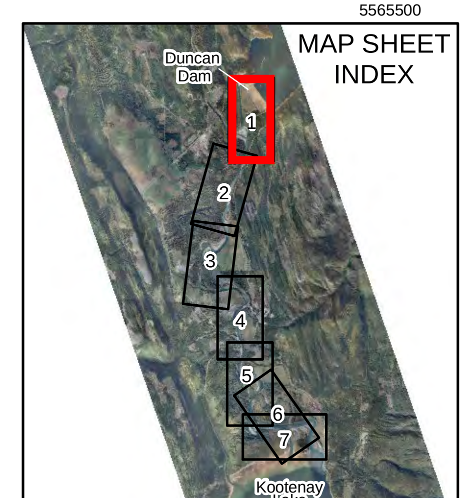
LEGEND

- RIVER KILOMETRE
- WSC GAUGE
- BOAT LAUNCH
- STRANDING SITE - INDEX
- STRANDING SITE - NON-INDEX

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



0 100 200 300 M

SCALE 1:5,000

PROJECT: DDMMON-16
LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING

TITLE: **LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES**

Golder Associates

PROJECT No. 12-1492-0117			SCALE AS SHOWN	REV. 0
DESIGN	BH	9 Apr. 2013	FIGURE 1	
GIS	CD	9 Apr. 2013		
CHECK	BH	31 Jan. 2014		
REVIEW	GB	31 Jan. 2014		

N:\Active\GIS\2012\12-1492-0117\Mapping\MXD\Fish\12_1492_0117_Stranding_Assessment_Sites.mxd

DRAFT

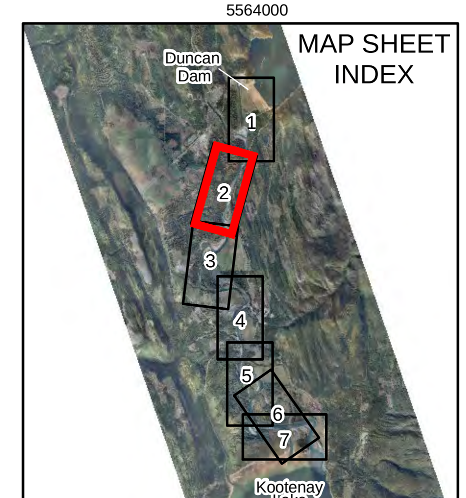
LEGEND

- RIVER KILOMETRE
- WSC GAUGE
- BOAT LAUNCH
- STRANDING SITE - INDEX
- STRANDING SITE - NON-INDEX

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



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SCALE 1:5,000

PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

PROJECT No. 12-1492-0117			SCALE AS SHOWN	REV. 0
DESIGN	BH	9 Apr. 2013	FIGURE 2	
GIS	CD	9 Apr. 2013		
CHECK	BH	31 Jan. 2014		
REVIEW	GB	31 Jan. 2014		

N:\Active\GIS\2012\12-1492-0117\Mapping\MXD\Fish\12_1492_0117_Stranding_Assessment_Sites.mxd

DRAFT

LEGEND

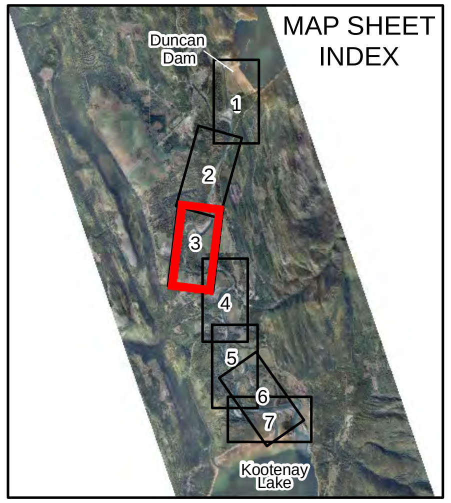
- RIVER KILOMETRE
- WSC GAUGE
- BOAT LAUNCH
- STRANDING SITE - INDEX
- STRANDING SITE - NON-INDEX

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83

MAP SHEET INDEX



0 100 200 300 M

SCALE 1:5,000

PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

PROJECT No. 12-1492-0117			SCALE AS SHOWN	REV. 0
DESIGN	BH	9 Apr. 2013	FIGURE 3	
GIS	CD	9 Apr. 2013		
CHECK	BH	31 Jan. 2014		
REVIEW	GB	31 Jan. 2014		

N:\Active\GIS\2012\12-1492-0117\Mapping\MXD\Fish\12_1492_0117_Stranding_Assessment_Sites.mxd

DRAFT

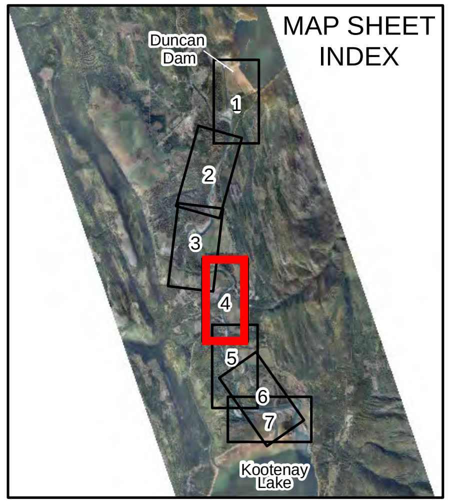
LEGEND

- RIVER KILOMETRE
- WSC GAUGE
- BOAT LAUNCH
- STRANDING SITE - INDEX
- STRANDING SITE - NON-INDEX

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



0 100 200 300 M

SCALE 1:5,000

PROJECT
DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE
LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

PROJECT No. 12-1492-0117			SCALE AS SHOWN	REV. 0
DESIGN	BH	9 Apr. 2013	FIGURE 4	
GIS	CD	9 Apr. 2013		
CHECK	BH	31 Jan. 2014		
REVIEW	GB	31 Jan. 2014		

N:\Active\GIS\2012\12-1492-0117\Mapping\MXD\Fish\12_1492_0117_Stranding_Assessment_Sites.mxd

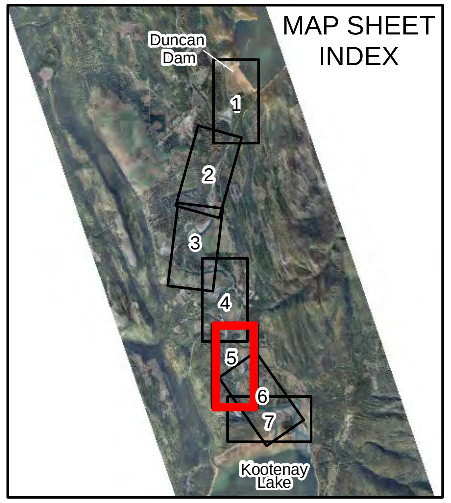
LEGEND

- RIVER KILOMETRE
- WSC GAUGE
- BOAT LAUNCH
- STRANDING SITE - INDEX
- STRANDING SITE - NON-INDEX

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



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SCALE 1:5,000

PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

**LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES**

Golder Associates

PROJECT No. 12-1492-0117			SCALE AS SHOWN	REV. 0
DESIGN	BH	9 Apr. 2013	FIGURE 5	
GIS	CD	9 Apr. 2013		
CHECK	BH	31 Jan. 2014		
REVIEW	GB	31 Jan. 2014		

N:\Active\GIS\2012\12-1492-0117\Mapping\MXD\Fish\12_1492_0117_Stranding_Assessment_Sites.mxd

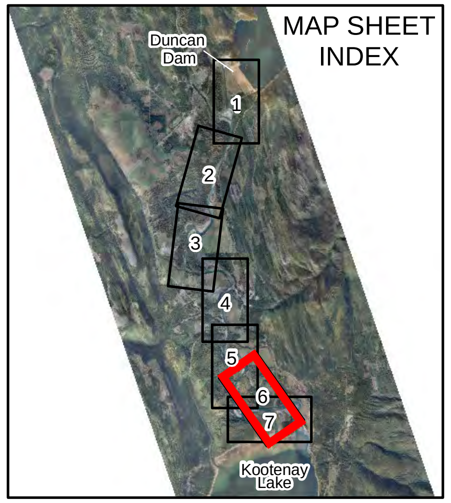
LEGEND

- RIVER KILOMETRE
- WSC GAUGE
- BOAT LAUNCH
- STRANDING SITE - INDEX
- STRANDING SITE - NON-INDEX

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



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SCALE 1:5,000

PROJECT
DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE
LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

PROJECT No. 12-1492-0117			SCALE AS SHOWN	REV. 0
DESIGN	BH	9 Apr. 2013	FIGURE 6	
GIS	CD	9 Apr. 2013		
CHECK	BH	31 Jan. 2014		
REVIEW	GB	31 Jan. 2014		

N:\Active\GIS\2012\12-1492-0117\Mapping\MXD\Fish\12_1492_0117_Stranding_Assessment_Sites.mxd

DRAFT

LEGEND

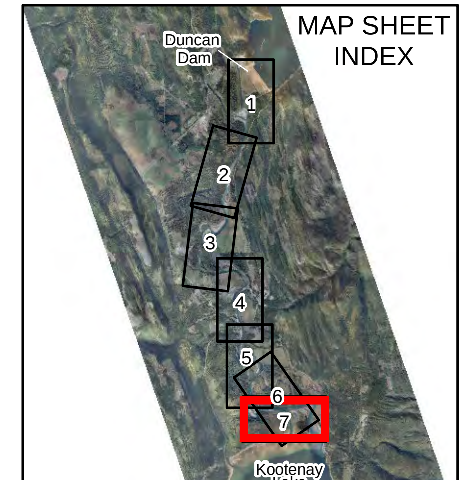
- RIVER KILOMETRE
- WSC GAUGE
- BOAT LAUNCH
- STRANDING SITE - INDEX
- STRANDING SITE - NON-INDEX

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83

MAP SHEET INDEX



0 100 200 300 M

SCALE 1:5,000

PROJECT	DDMMON-16 LOWER DUNCAN RIVER FISH STRANDING IMPACT MONITORING
TITLE	**LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES**

Golder Associates

PROJECT No. 12-1492-0117			SCALE AS SHOWN	REV. 0
DESIGN	BH	9 Apr. 2013	FIGURE 7	
GIS	CD	9 Apr. 2013		
CHECK	BH	31 Jan. 2014		
REVIEW	GB	31 Jan. 2014		

N:\Active\GIS\2012\12-1492-0117\Mapping\MXD\Fish\12_1492_0117_Stranding_Assessment_Sites.mxd

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

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

Table A2: Chronology of sampling activities for the 2009 - 2010 Lower Duncan River Fish Stranding Impact Monitoring, Year 2 Program

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

Table A3: Chronology of sampling activities for the 2010 - 2011 Lower Duncan River Fish Stranding Impact Monitoring, Year 3 Program

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

Table A4: Chronology of sampling activities for the 2011 - 2012 Lower Duncan River Fish Stranding Impact Monitoring, Year 4 Program.

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

Table A5: Chronology of sampling activities for the 2012 - 2013 Lower Duncan River Fish Stranding Impact Monitoring, Year 5 Program

Date(s)	Sampling Activities	Reduction Event Number	Number of Snorkel Sites Surveyed	Number of Index Sites Stranding Assessed	Number of Non-Index Stranding Sites Assessed
15 April 2012	Stranding Assessments	RE2012-03	-	2	0
01 June 2012	Stranding Assessments	RE2012-04	-	Assessment was planned, but cancelled by BC Hydro prior to reduction date	
26 September 2102	Stranding Assessments	RE2012-05	-	4	5
27 September 2012	Stranding Assessments	RE2012-06	-	3	2
01 October 2012	Stranding Assessments	RE2012-07	-	3	3
21 January 2013	Stranding Assessments	RE2013-01	-	5	6
01 March 2013	Stranding Assessments	RE2013-02	-	3	2

Table A6: Chronology of sampling activities for the 2013 - 2014 Lower Duncan River Fish Stranding Impact Monitoring, Year 6 Program

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

Table A7: Chronology of sampling activities for the 2014 - 2015 Lower Duncan River Fish Stranding Impact Monitoring, Year 7 Program

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

Table A8: Chronology of sampling activities for the 2015 - 2016 Lower Duncan River Fish Stranding Impact Monitoring, Year 8 Program

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

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APPENDIX B

Bayesian Models - Code

JAGS code for Bayesian analysis of abundance (both Rainbow Trout and Mountain Whitefish)

```
description <- c(
"`bIntercept`" = "Intercept for `log(eDensity)`",
"`bYear`" = "Effect of `YearNum` on `bIntercept`",
"`bDepth`" = "Effect of `DepthNum` on `bIntercept`",
"`bFlow`" = "Effect of `FlowNum` on `bIntercept`",
"`sSite`" = "SD of `bSite`",
"`bSite[i]`" = "Effect of `i`^th^ `SiteNum` on `bIntercept`",
"`bEfficiency`" = "Intercept for            logit(eEfficiency)",
"`bEfficiencyVisit[i]`" = "Effect of `i`^th^ visit on `bEfficiency`",
"`Area[i]`" = "Area surveyed on `i`^th^ visit",
"`Nfish[i]`" = "Number of fish observed on `i`^th^ visit"
)

model1 <- jags_model("model {
  bIntercept ~ dnorm(-5, 5^-2)

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

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

        sSite ~ dunif(0, 5)
        for(i in 1:nSiteNum){
                bSite[i] ~ dnorm(0, sSite^-2)
        }

  bEfficiency <- -0.53

  for(i in 1:nVisit) {
                bEfficiencyVisit[i] ~ dnorm(0, 0.68^-2)
  }
        for(k in 1:length(Nfish)){
                log(eDensity[k]) <- bIntercept + bYear[YearNum[k]] + bDepth[DepthNum[k]] +
bSite[SiteNum[k]]
    eAbundance[k] ~ dpois(eDensity[k] * Area[k])
                logit(eEfficiency[k]) <- bEfficiency + bEfficiencyVisit[Visit[k]]
    Nfish[k] ~ dbin(eEfficiency[k], eAbundance[k])
```

```
        }
}",
derived_code  =  "data{
        for(i in 1:length(YearNum)){
                log(eDensity[i]) <- bIntercept + bYear[YearNum[i]] + bDepth[DepthNum[i]] +
bSite[SiteNum[i]]
                eAbundance[i] <- eDensity[i] * Area[i]

                logit(eEfficiency[i]) <- bEfficiency + bEfficiencyVisit[Visit[i]]
                prediction[i] <- eEfficiency[i] * eAbundance[i]
                residual[i] <- Nfish[i] - prediction[i]
  }
}",
gen_inits = function(data) {
  inits <- list()
  inits$eAbundance <- data$Nfish + 1
  inits
},
random_effects = list(bSite = "SiteNum", bEfficiencyVisit = "Visit"))


models <- jaggernaut::combine(model1)
```

JAGS code for Bayesian analysis of interstitial stranding code (both Rainbow Trout and Mountain Whitefish)

```
description = c(
"`bIntercept`" = "Intercept for `log(eDensity)`",
"`pIntercept`" = "Intercept for `logit(eProb)`",
"`pIndex[2]`" = "Effect of index site on `pIntercept`",
"`eDensity[i]`" = "Expected density of fish at the `i`^th^ grid if stranding occurs",
"`eProb[i]`" = "Probability of one or more fish stranding at the `i`^th^ grid",
"`eP[i]`" = "Whether or not one or more fish stranded at the `i`^th^ grid",
"`Number[i]`" = "Number of fish observed at the `i`^th^ grid",
"`Area[i]`" = "Area of the `i`^th^ grid")

model1 <- jags_model("model {
	pIntercept ~ dnorm(0, 5^-2)
	bIntercept ~ dnorm(0, 5^-2)

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

	for(i in 1:length(Number)) {
		log(eDensity[i]) <-  bIntercept
		logit(eProb[i]) <- pIntercept + pIndex[Index[i]]
		eP[i] ~ dbern(eProb[i])
		Number[i] ~ dpois(eDensity[i]*eP[i]*Area[i])
  }
}",
derived_code  =  "data{
	for(i in 1:length(Number)){
		logit(eProb[i]) <- pIntercept + pIndex[Index[i]]
		log(ePrediction[i]) <- bIntercept
		prediction[i] <- ePrediction[i]*eProb[i]
		residual[i] <- Number[i] - prediction[i]
						}
}",
gen_inits = function (data) {
	inits <- list()
	inits$pIntercept <- 0.5
	inits
},
random_effects = list(bSiteName = "SiteName"))
```

```
model2 <- jags_model("model {
        pIntercept ~ dnorm(0, 5^-2)
        bIntercept ~ dnorm(0, 5^-2)

        for(i in 1:length(Number)) {
                log(eDensity[i]) <-  bIntercept
                logit(eProb[i]) <- pIntercept
                eP[i] ~ dbern(eProb[i])
                Number[i] ~ dpois(eDensity[i]*eP[i]*Area[i])
  }
}",
derived_code  =  "data{
        for(i in 1:length(Number)){
                logit(eProb[i]) <- pIntercept
                log(ePrediction[i]) <- bIntercept
                prediction[i] <- ePrediction[i]*eProb[i]*Area[i]
                residual[i] <- Number[i] - prediction[i]
  }
}",
gen_inits = function (data) {
        inits <- list()
        inits$pIntercept <- 0.5
        inits
},
random_effects = list(bSiteName = "SiteName"))

models <- jaggernaut::combine(model1, model2)
```

JAGS code for Bayesian analysis of pool density and pool stranding code; single model for both Rainbow Trout and Mountain Whitefish

```
description = c(
"`bIntercept`" = "Intercept for `log(eDensityPool)`",
"`pIntercept`" = "Intercept for `logit(eP)`",
"`bDrop`" = "Effect of `Drop` on `bIntercept`",
"`sSite`" = "SD of effect of `SiteName` on `bIntercept`",
"`bSiteName[j]`" = "Effect of `i`^th^ `SiteName` on `bIntercept`",
"`eP[i]`" = "Probability of one or more pools forming at `i`^th^ site visit",
"`eDensityPool[i]`" = "Expected number of pools at `i`^th^ site visit if pools formed",
"`NumberPoolPresent[i]`" = "Number of fish observed at the `i`^th^ site visit",
"`SiteArea[i]`" = "Area of the `i`^th^ site")

model1 <- jags_model("model {
	bIntercept ~ dnorm(-5, 5^-2)
	pIntercept ~ dunif(0, 1)

	bDrop ~ dnorm(0, 5^-2)

	sSite ~ dunif(0, 5)
	for(j in 1:nSiteName){
		bSiteName[j] ~ dnorm(0, sSite^-2)
	}

	for(i in 1:length(NumPoolsPresent)){
		eP[i] ~ dbern(pIntercept)
		log(eDensityPool[i]) <- bIntercept + bDrop*Drop[i] + bSiteName[SiteName[i]]
		NumPoolsPresent[i] ~ dpois(eDensityPool[i] * eP[i] * SiteArea[i])
	}
}",
derived_code  =  "data{
	for(i in 1:length(NumPoolsPresent)){
		log(ePrediction[i]) <- bIntercept + bDrop*Drop[i] + bSiteName[SiteName[i]]
		prediction[i] <- ePrediction[i] * SiteArea[i] * pIntercept
		residual[i] <- NumPoolsPresent[i] - prediction[i]
  }
}",
gen_inits = function (data) {
	inits <- list()
	inits$eP <- data$Presence
	inits$bIntercept <- rlnorm(1)
	inits$bDrop <- rlnorm(1)
	inits
```

```
},
random_effects = list(bSiteName = "SiteName"))

models <- jaggernaut::combine(model1)

description = c(
"`bIntercept`" = "Intercept for `log(eAbundance)`",
"`p`" = "Capture efficiency for different `SamplingGearNum`",
"`sReduction`" = "SD of effect of `ReductionEventID` on `bIntercept`",
"`bReduction[i]`" = "Effect of `i`^th^ `ReductionEventID` on `bIntercept`",
"`r`" = "SD of overdispersion",
"`bSeason[i]`" = "Effect of `i`^th^ `SeasonNum` on `bIntercept`",
"`Pass[i,j]`" = "Number of fish captured on `j`^th^ pass at `i`^th^ visit")

model1 <- jags_model("model {

        bIntercept ~ dnorm(0, 5^-2)
        p[1] <- 1
        p[2] ~ dunif(0, 1)
        sReduction ~ dunif(0, 5)
        r ~ dunif(0, 5)

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

        for(i in 1:nReductionEventID){
                bReduction[i] ~ dnorm(0, sReduction^-2)
                                        }

        for(i in 1:length(ReductionEventID)){
                eU[i] ~ dgamma(1/r^2, 1/r^2)
                log(eAbundance[i]) <- bIntercept + bSeason[SeasonNum[i]] +
bReduction[ReductionEventID[i]]
                eN[i] ~ dpois(eAbundance[i]*eU[i])
                eNPass[i, 1] <- eN[i]

                for(pass in 1:nPass){
                        Pass[i, pass] ~ dbin(p[SamplingGearNum[i]], eNPass[i, pass])
                        eNPass[i, pass+1] <- eNPass[i, pass] - Pass[i, pass]
                                                } #pass
                                } # i
}",
```

```
derived_code  =  "data{
	for(i in 1:length(ReductionEventID)){
		log(ePrediction[i]) <- bIntercept + bSeason[SeasonNum[i]] +
bReduction[ReductionEventID[i]]
		# n fish in first pass
		prediction[i] <- ePrediction[i] * p[SamplingGearNum[i]]
		residual[i] <- (Pass[i, 1] - prediction[i])/((prediction[i])^0.5)
						}
}",
modify_data = function (data) {
  data$Pass <- as.matrix(data_frame(data$Pass1, data$Pass2, data$Pass3))
  data[c("Pass1", "Pass2", "Pass3")] <- NULL
  data$nPass <- 3
  data
},

gen_inits = function (data) {
	inits <- list()
	inits$eN <- apply(data$Pass, 1, sum, na.rm = TRUE) + 1
	inits
},
random_effects = list(bReduction = "ReductionEventID"))

models <- jaggernaut::combine(model1)
```

APPENDIX C

Photographic Plates

Plate 1 Sampling a dried pool at site M0.8R, 24 September 2016

Plate 2 Unidentified larvae in a isolated pool, 20 May 2016

Plate 3 Backpack electrofishing at site S6.9R, 01 March 2017

Plate 4 Bull trout salvaged from isolated pool at site M6.1L, 02 March 2017

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