Duncan Dam Project Water Use Plan

Adaptive Stranding Protocol Development Program

Implementation Year 8

Reference: DDMMON-16

Lower Duncan River: Fish Stranding Impact Monitoring: Year 8

Data Report

Study Period: April 2015 to April 2016

Golder Associates Ltd.
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April 20, 2017

DDMMON-16: LOWER DUNCAN RIVER

Lower Duncan River Fish Stranding Impact Monitoring: Year 8 Report (April 2015 to April 2016)

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REPORT

Golder Associates

Cover Photo: Sampling an isolated pool at site S10.6R, September 28, 2014.

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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 2016, 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 8 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 are somewhat sensitive to interpretation.

Table EI: DDMMON-16 Year 8: Status of Management Questions and Objectives.

DDMMON-16 Management Question	DDMMON-16 Specific Hypothesis	DDMMON-16 Year 8 (2015-2016) 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 8 (2015-2016) Status Summary
2) What are the levels of impact to resident fish populations associated with fish stranding events on the lower Duncan River?	*Ho_1: Fish stranding observed at index sites along the lower Duncan River floodplain is representative of overall stranding.*	- Index sites were not originally selected to be representative of the entire LDR, but were selected to focus on sites believed to have the highest amounts of stranding based on the amount of dewatered area and suitable habitat. - Index sites tend to be of lower gradient and wider than the non-index sites, therefore more area dewaters at these sites. - In the current year, a significant 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.
	Ho_2: Fish populations in the lower Duncan River are not significantly impacted by fish stranding events.	- Estimates for the number of Rainbow Trout juveniles stranded in pools were relatively low with high precision. However, the estimated numbers of interstitially stranded fish in the lower Duncan River were high with low precision. - Significant progress has been made on reducing the uncertainty related to interstitial stranding estimates. - 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. - Mountain Whitefish encounters have been minimal in all study years. This consistently low level of stranding was not considered significant and will likely not result in a population level effect. - Within the current dataset relationships between pool and interstitially stranded fish and slope of substrate were not found. - Until the uncertainty related to interstitial stranding estimation is reduced, at this time we cannot accept or reject hypothesis *Ho_2* for Rainbow Trout. We can accept the premise for Mountain Whitefish because of the lack of evidence to reject this hypothesis.

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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 the confluence of the Lardeau River. Downstream from the confluence, the LDR is comprised 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 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 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, and the maximum daily flow change allowed is 113 m^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 8 (2015 - 2016) 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 8 of this Program (April 15, 2015 to April 14, 2016). This report also presents detailed statistical analysis in relation to the multi-year program objectives, and incorporates several aspects of the updated 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) and the present study year (Year 8: 2015 – 2016) 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 5.0).

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 January 14, 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, increased sampling effort from Year 4 on assessed non-index sites.

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 2.0.

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 was 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 8 was to sample similar numbers of sites, as well as similar length of river in comparison to Year 7.

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 9, 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 monitoring operations in the future years of this program.

1.4.6 Data Analysis

The modelling used in Year 7 (Golder 2017) of this program was updated to incorporate the current years' data set, to remove the dichotomous slope of substrate 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).

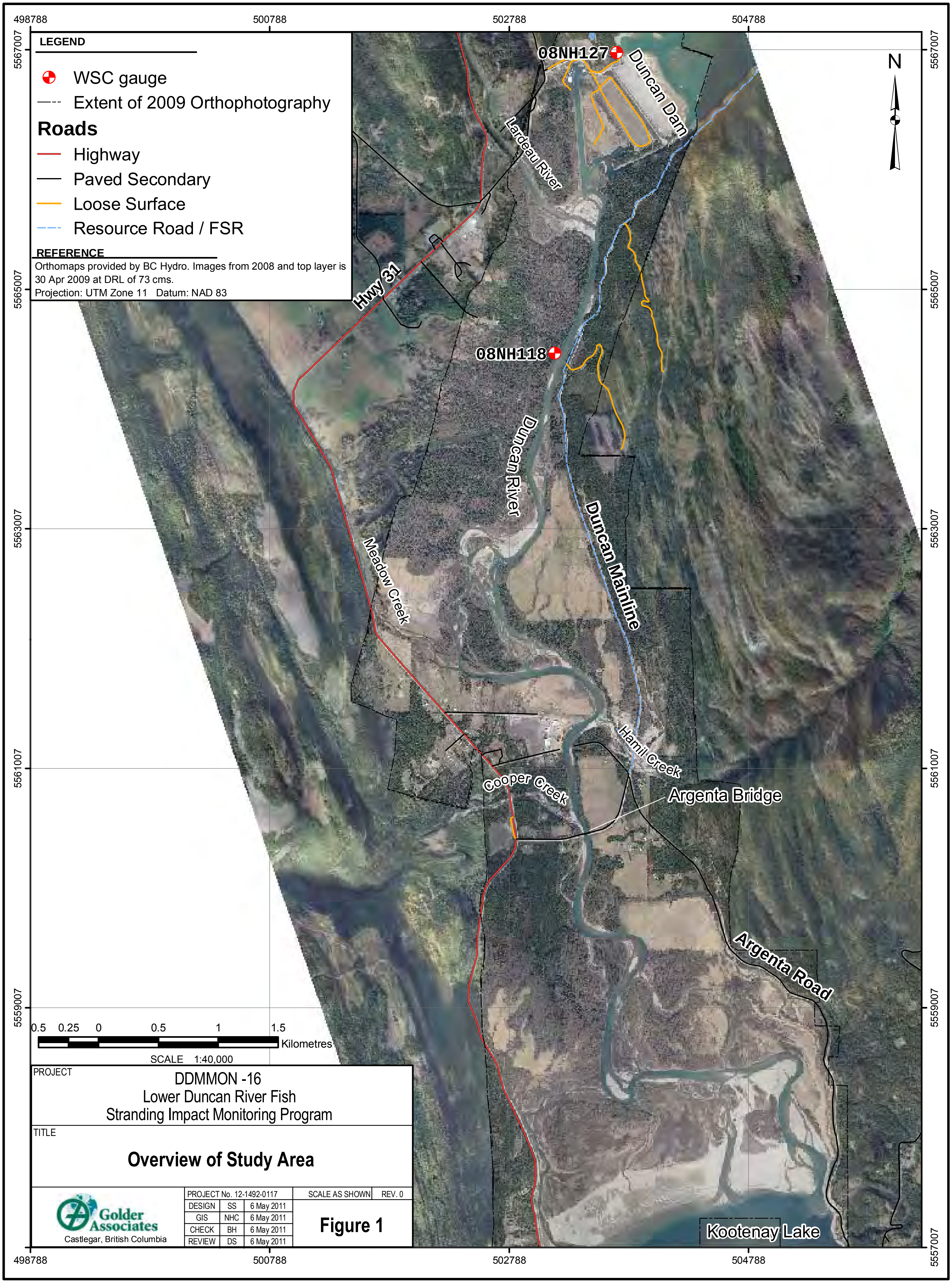
For the purpose of this study, 50 potential fish stranding sites were identified based on previous studies (AMEC 2004 and Golder 2006, 2008, 2009b, 2010, 2011, and 2014; Golder and Poisson 2012). These stranding sites included 11 index stranding assessment sites and 39 non-index sites (Appendix A, Figures 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 April 14. Stranding assessment activities in the present study year were conducted from September 28, 2015 to April 9, 2016, during planned flow reductions at DDM. Each assessed reduction from DDM was assigned a reduction event number (RE; see Section 2.6) and Table 1 outlines all assessment activities during Year 8. 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: 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
September 21 and 22, 2015	Abundance Estimation	-	Study Area Reconnaissance and Site Selection		
September 23, 2015	Abundance Estimation	-	12	-	-
September 24, 2015	Abundance Estimation	-	12	-	-
September 25, 2015	Abundance Estimation	-	13	-	-
September 26, 2015	Abundance Estimation	-	9	-	-
September 28, 2015	Stranding Assessments	RE2015-03	-	2	4
October 1, 2015	Stranding Assessments	RE2015-04	-	2	5
December 22, 2015	Stranding Assessments	RE2015-05	-	4	3
December 29, 2015	Stranding Assessments	RE2015-06	-	3	5
April 9, 2016	Stranding Assessments	RE2016-01	-	3	2



LEGEND
WSC gauge
Extent of 2009 Orthophotography
Roads
Highway
Paved Secondary
Loose Surface
Resource Road / FSR
REFERENCE
Orthomaps provided by BC Hydro. Images from 2008 and top layer is 30 Apr 2009 at DRL of 73 cms.
Projection: UTM Zone 11 Datum: NAD 83
498788
500788
502788
504788
5567007
5565007
5563007
5561007
5559007
5557007
N
08NH127
Duncan Dam
Lardeau River
Hwy 31
08NH118
Duncan River
Duncan Mainline
Meadow Creek
Hamill Creek
Cooper Creek
Argenta Bridge
Argenta Road
Kootenay Lake
0.5 0.25 0 0.5 1 1.5
Kilometres
SCALE 1:40,000
PROJECT
DDMMON -16
Lower Duncan River Fish
Stranding Impact Monitoring Program
TITLE
Overview of Study Area
Golder Associates
Castlegar, British Columbia
PROJECT No. 12-1492-0117
SCALE AS SHOWN
REV. 0
DESIGN SS 6 May 2011
GIS NHC 6 May 2011
CHECK BH 6 May 2011
REVIEW DS 6 May 2011
Figure 1

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 Lakewood™ Universal temperature probes (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. 3.2.3 (R Development Core Team 2015), interfaced with JAGS v. 3.3.0 (Plummer 2013) through the jaggernaut v. 2.3.3 package (Thorley 2014). 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 (Kery and Schaub 2011, p. 36). The posterior distributions were estimated from a minimum of 1,000 Markov Chain Monte Carlo (MCMC) samples thinned from the second halves of three chains (Kery and Schaub 2011, pp. 38–40). Model convergence was confirmed by ensuring that Rhat (Kery and Schaub 2011, p. 40) was less than 1.1 for each of the parameters in the model (Kery 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 (mean), 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 (Kery and Schaub 2011, p. 37, p. 42).

Variable selection was achieved by dropping insignificant fixed variables (Kery and Schaub 2011, p. 37, p. 42) and uninformative random variables (Kery and Schaub 2011, pp. 77–82). A fixed variable was considered to be insignificant if its significance was ≥0.05 while a random variable was considered to be uninformative if its percent relative error was ≥80%. The Deviance Information Criterion (DIC) was not used because it is of questionable validity when applied to hierarchical models (Kery and Schaub 2011, p. 469).

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) (Kery and Schaub 2011, pp. 77–82).

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 2014) 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 by adding all fixed effects (flow [slack/fast], depth [shallow/deep], and year [2013, 2014, 2015]) and random effects (site effect), and using stepwise elimination if an effect is found insignificant or non-informative. If two effects were found significant, their interaction was added and tested. A random site effect was used in all models, 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 (flow, 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 site
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
nFlowNum	Number of flow strata
bFlow[i]	The effect of the *i*-th flow stratum on fish density
Area[k]	The area of the *k*-th site
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
FlowNum[k]	Numeric representation of the flow 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})
bFlow[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]] + bFlow[FlowNum[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 - 2013	Area - 2014	Area - 2015
Shallow/Slack	66,217.5	73,276.6	73,276.6
Shallow/Fast	337,857.1	357,565.1	357,565.1
Deep/Slack	---	11,092.3	11,092.3
Deep/Fast	145,784.8	164,858.4	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 8 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 Year 8, stranding sites were not split into two strata. The 10 sites selected prior to each assessment were randomly selected from all 51 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 8 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:

1) Zero to Low complexity (0% – 10% total cover; Appendix C, Plate 1)

2) Moderate to High complexity (>10% total cover; Appendix C, Plate 2)

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; Appendix C, Plate 3) 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.; Appendix C, Plate 4)

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 Pool

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 (1 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 and subdominant 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 8 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 in order to obtain a total riverine area exposed for each reduction. These total areas were summed over the entire year in order 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 2.4.

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 Poisson model (Kery and Schaub 2011, pp. 386–388). The number of pools was described by a Bernoulli-Poisson distribution to account for zero inflation. 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 counts in the spring, under typical reduction
bSeason[i]	The effect of the *i*-th season on pool-stranded fish numbers, 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 = 1 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
mu[i]	bIndex_site[Index_site[i]] + bSiteName[SiteName[i]] + bArea*SiteArea[i] (where bIndex_site was used)(where no bIndex_site was used)
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]

2.6.3.4 Interstitial Stranding

In the Bayesian model of interstitial stranding, the number of fish stranded in each sample (grid or transect) was defined as Poisson-distributed with zero inflation, with a mean expected value determined by fish abundance and the probability of stranding at each sampled location (Table 10). In addition, a separate model was constructed, where the mean expected value of probability of interstitial stranding at a site was also influenced by whether the site was identified as index or random site. The significance of the index/random variable was determined based on whether the 95% CRIs of index site coefficient overlapped those of random site coefficient. The complete model specification used is shown in Table 13 to Table 15, and model code is provided in Appendix B. The model was run separately for Rainbow Trout and for Mountain Whitefish.

For the overall estimates of stranding, we used the original model, which did not contain the effect of whether the site was index or random. To incorporate the index/random information into the final estimates of interstitial stranding, the breakdown of total dewatered area by index/random, and slope (low/high) within each discharge scenario would be required. To estimate total interstitial stranding, mean expected fish numbers were multiplied by mean expected probability of interstitial stranding, and by total exposed area using GIS-derived low-slope (0-4%) and high-slope (>4%) dewatered areas at each stranding event.

Table 13: Variables and parameters in the Bayesian analysis of interstitial stranding.

Variable/parameter	Description
bIntercept	Expected fish density in each examined grid or transect
pIntercept	Logistic stranding probability
bIndex[i]	The effect of whether the *i*-th sampling area is in an index or random site on probability of stranding
Index[i]	Whether the *i*-th site is index or random
Area[i]	The area of the *i*-th sampling area (whether grid or transect)
log(eDensity[i])	Expected fish density in the *i*-th sampling area
logit(eProb[i])	Stranding probability in the *i*-th sampling area
eP[i]	Whether a fish was stranded in the *i*-th sampling area
Number[i]	Number of fish observed in the *i*-th sampling area

Table 14: Prior probability distributions in the Bayesian analysis of interstitial stranding.

Variable/parameter	Description
pIntercept	dnorm(0, 5^{-2})
bIntercept	dnorm(0, 5^{-2})
pIndex[2]	dnorm(0, 5^{-2})

Table 15: Dependencies between variables and parameters in the Bayesian analysis of interstitial stranding.

Variable/parameter	Dependency
pIndex[1]	0
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])

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 8. Presently, 77 individual stranding assessments are in the database. Results from 14 assessments prior to September 15, 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 70.1 m^3/s on July 22, 2015 to 403.2 m^3/s on November 1, 2015. Hourly discharge from DDM ranged from 0 m^3/s on several days between early June and early July 2015, to 335.2 m^3/s on November 1, 2015 (Figure 2). 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.

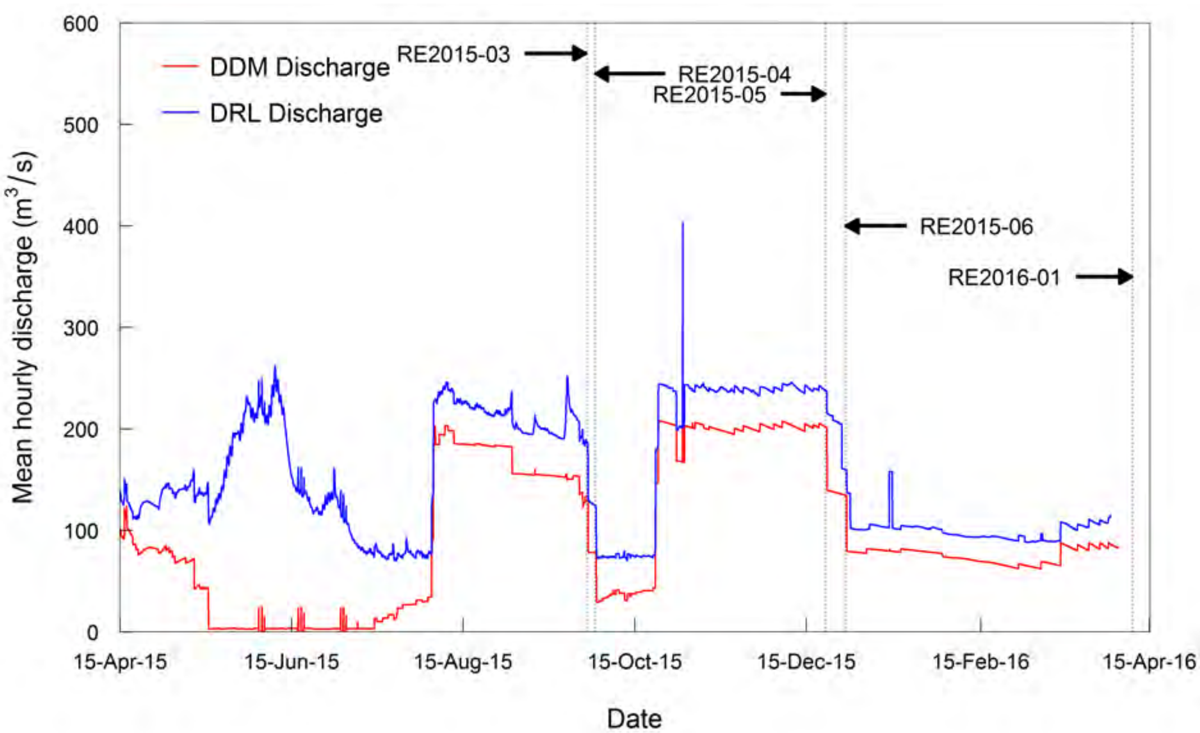


Figure 2: Hourly discharge at the Duncan Dam (DDM, red line) and at the lower Duncan River below the Lardeau River (DRL, blue line) from April 15, 2015 to April 15, 2016. Vertical dotted lines represent the timing of fish stranding assessments.

During the present study, five reduction events occurred at DDM (Figure 2 and Table 16). During the reduction events, DDM decreases of discharge ranged between 45 and 62 m^3/s (Table 16). These decreases represent the discharge reductions at DDM, rather than flow changes at particular downstream fish stranding sites.

Table 16: Summary of DDM flow reduction events, from Sept 28, 2015 to April 9, 2016, for those events when fish stranding assessments were conducted.

Date	Reduction Event	DDM Discharge (m^3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
Sep 28, 2015	RE2015-03	133	78	55	Down 5 m^3/s every 15 minutes from 08:00 to 10:30.	Onset of Kokanee protection flows.
Oct 01, 2015	RE2015-04	78	33	45	Down 5 m^3/s every 15 minutes from 06:00 to 08:00.	Kokanee protection flows.
Dec 22, 2015	RE2015-05	202	140	62	Down 7 m^3/s every 15 minutes from 06:00 to 07:45, down 6 m^3/s at 08:00.	Discharge reduced to meet flow target at DRL
Dec 29, 2015	RE2015-06	140	80	60	Down 7 m^3/s every 15 minutes from 06:00 to 07:45, down 4 m^3/s at 08:00.	Discharge reduced to meet flow target at DRL.
Apr 09, 2016	RE2016-01	128	70	58	Down 7 m^3/s every 15 minutes from 06:00 to 07:30, down 9 m^3/s at 07:45.	Discharge reduced to meet flow target at DRL.

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

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 September 15, 2006 have been excluded from the dataset. Stranding assessments were conducted following five flow reductions during study Year 8 (2015-2016). 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 17.

Table 17: Scientific names of fish species encountered during fish stranding assessments on the lower Duncan River, September 2006 to March 2015.

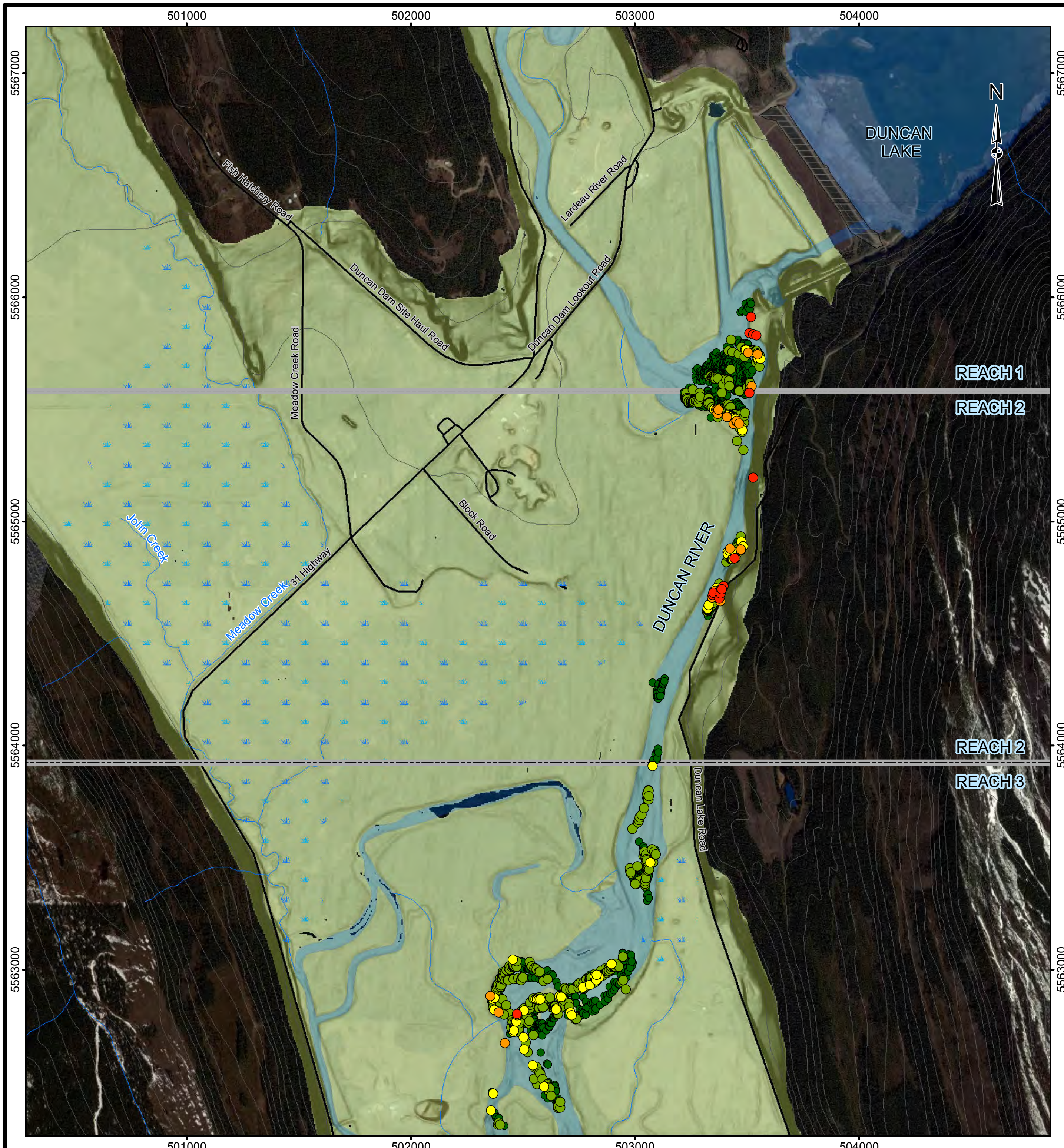
Category	Species	Scientific Name	Species Code[a]
Sportfish	Rainbow Trout	*Oncorhynchus mykiss*	*RB*
	Bull Trout	*Salvelinus confluentus*	*BT*
	Mountain Whitefish	*Prosopium williamsoni*	*MW*
	Pygmy Whitefish	*Prosopium coulteri*	*PW*
	Kokanee	*Oncorhynchus nerka*	*KO*
	Burbot	*Lota lota*	*BB*
Non-sportfish	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; Table 18). 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 8 (2012 – 2013 to 2015 – 2016). The number of pools sampled in the present year was also reduced to allow for more intensive interstitial sampling effort. This resulted in lower numbers of pools sampled (*n* = 148), and the most number of interstitial transects (*n* = 135) assessed to date (Table 18). The locations of all sampled stranding mechanisms within the dataset are presented in Figure 3 and Figure 4.

Table 18: Sampling effort during reductions of each study year that were included in the present analysis.

DDMMON-16 Study Year	Number of Reductions Assessed	Number of Assessments at Index Sites	Number of Assessments at Non-Index	Total Number of Pools Sampled	Total Number of Interstitial Grids Conducted	Total Number of Interstitial Transects Conducted
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	29	21	92	411	0
5 (2012-2013)	7	20	18	78	331	0
6 (2013-2014)	5	13	16	56	325	0
7 (2014-2015)	6	21	18	98	124	101
8 (2015-2016)	5	14	19	148	0	135



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

PROJECT

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

TITLE

LOCATION AND SLOPE (%) OF SAMPLED STRANDING MECHANISMS (SEPT. 2006 - APR. 2016) REACHES 1 TO 3

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) BACKDROP IMAGERY: IKONOS CANADA . USED UNDER LICENSE, ALL RIGHTS RESERVED. IMAGERY DATE: 20080502.
DATUM: NAD83 PROJECTION: UTM ZONE 11

Golder Associates

PROJECT		1535517	FILE No.	
DESIGN	BH	13 Mar. 2017	SCALE AS SHOWN	REV. 0
GIS	JG	13 Mar. 2017		
CHECK	BH	20 Apr. 2017	**FIGURE: 3**	
REVIEW	DA	20 Apr. 2017		

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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

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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. 2016) REACHES 3 TO 5

Golder Associates

PROJECT		1535517	FILE No.	
DESIGN	BH	13 Mar. 2017	SCALE AS SHOWN	REV. 0
GIS	JG	13 Mar. 2017		
CHECK	BH	20 Apr. 2017	**FIGURE: 4**	
REVIEW	DA	20 Apr. 2017		

Document Path: \\golder.gds\gal\burnaby\CAD-GIS\Bur-Graphics\Projects\2012\1492\12-1492-0117\GIS\Mapping\MXD\Fish\2017\Fig3_4_1214920117_Stranding Mechanism and slope_2015Data.mxd

In Year 8, a total of 246 fish were observed, representing 12 species [five sportfish and seven non-sportfish species (Table 19)]. This total is the lowest documented in all study years. In comparison to all study years, juvenile Rainbow Trout (n = 52) encounters were the lowest in Year 8, however they still were the most abundant sportfish observed (21.1% of the total catch). Kokanee young-of-the-year were the next most abundant sportfish, accounting for 4.9% of the total number of fish encountered. Mountain Whitefish accounted for 3.3% (n = 8) of the catch (Table 19, Figure 5). The most common non-sportfish taxa identified to species were Longnose Dace and Sculpin spp., accounting for 29.7% and 9.3% of the total number of observed fish, respectively.

Table 19: 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 2016.

Species and Life Stage		N Fish (% of total within each year)									
		2006-2006	2007-2008	2008-2009	2009-2010	2010-2011	2011-2012	2012-2013	2013-2014	2014-2015	2015-2016
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)
	Juvenile	130 (37.1)	278 (11.5)	530 (33.2)	113 (12.3)	343 (25.2)	452 (24.2)	332 (37.1)	241 (40.2)	737 (58.4)	52 (21.1)
Bull Trout	**Adult**	0 (0)	0 (0)	0 (0)	4 (0.4)	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)
Mountain Whitefish	**Adult**	0 (0)	1 (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 (3.3)
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)
	Juvenile	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 (12.2)	42 (3.1)	55 (3)	111 (12.4)	0 (0)	0 (0)	0 (0)
	Y-O-Y	0 (0)	1690 (70.4)	85 (5.3)	109 (11.9)	83 (6.1)	861 (46.2)	257 (28.7)	0 (0)	22 (1.8)	12 (4.9)
	Eggs	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	1 (0.4)
Burbot	**Adult**	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)
Non-sportfish											
Longnose Dace		117 (33.4)	15 (0.6)	103 (6.5)	273 (29.7)	551 (40.5)	30 (1.6)	32 (3.6)	227 (37.8)	143 (11.4)	73 (29.7)
Dace spp.		0 (0)	0 (0)	0 (0)	12 (1.3)	1 (0.1)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Slimy Sculpin		0 (0)	13 (0.5)	11 (0.7)	62 (6.8)	39 (2.9)	6 (0.3)	0 (0)	1 (0.2)	12 (1)	11 (4.5)
Torrent Sculpin		0 (0)	1 (0)	1 (0.1)	0 (0)	0 (0)	3 (0.2)	0 (0)	0 (0)	0 (0)	0 (0)
Prickly Sculpin		0 (0)	0 (0)	0 (0)	0 (0)	2 (0.1)	0 (0)	0 (0)	0 (0)	2 (0.2)	0 (0)
Sculpin spp.		23 (6.6)	16 (0.7)	65 (4.1)	34 (3.7)	165 (12.1)	99 (5.3)	130 (14.5)	46 (7.7)	189 (15)	23 (9.3)
Sucker spp.		2 (0.6)	4 (0.2)	26 (1.6)	166 (18.1)	54 (4)	9 (0.5)	16 (1.8)	32 (5.3)	42 (3.3)	8 (3.3)
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 (7.3)
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 (3.3)
Lake Chub		0 (0)	0 (0)	0 (0)	1 (0.1)	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)
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 (12.6)
All Species Total		**350**	**2,409**	**1,596**	**918**	**1,361**	**1,864**	**896**	**600**	**1,261**	**246**

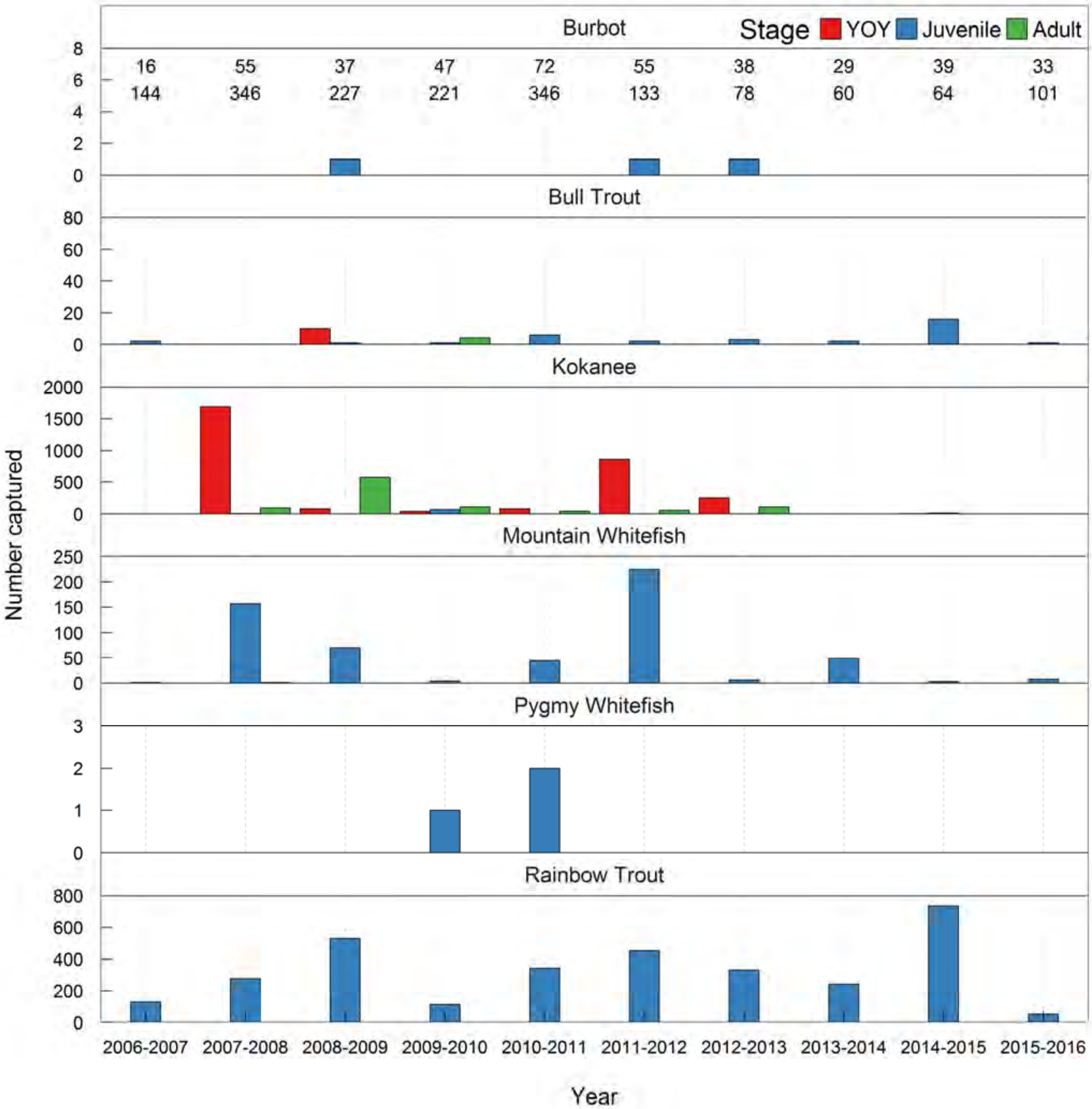


Figure 5: Abundances of sportfish species, separated by life stage, observed in stranding assessments between 2006 and 2016. Note the different y-axis scales among panels. On the uppermost panel, the numbers of sampled sites and pools are provided in the first and second lines, respectively. The Kokanee egg stranded in 2015-2016 sampling is not shown.

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 17.8 km², in comparison to 13.5 km² during the post-WUP operational regime (Figure 6). The area exposed is less variable from year to year in the post-WUP operational regime over the years assessed and is in general, lower (Figure 6). The maximum annual exposed area (21.5 km²) was observed in 2006, during pre-WUP operations. The minimum exposed area (11.3 km²) was observed in 2013 during post-WUP operations (Figure 6).

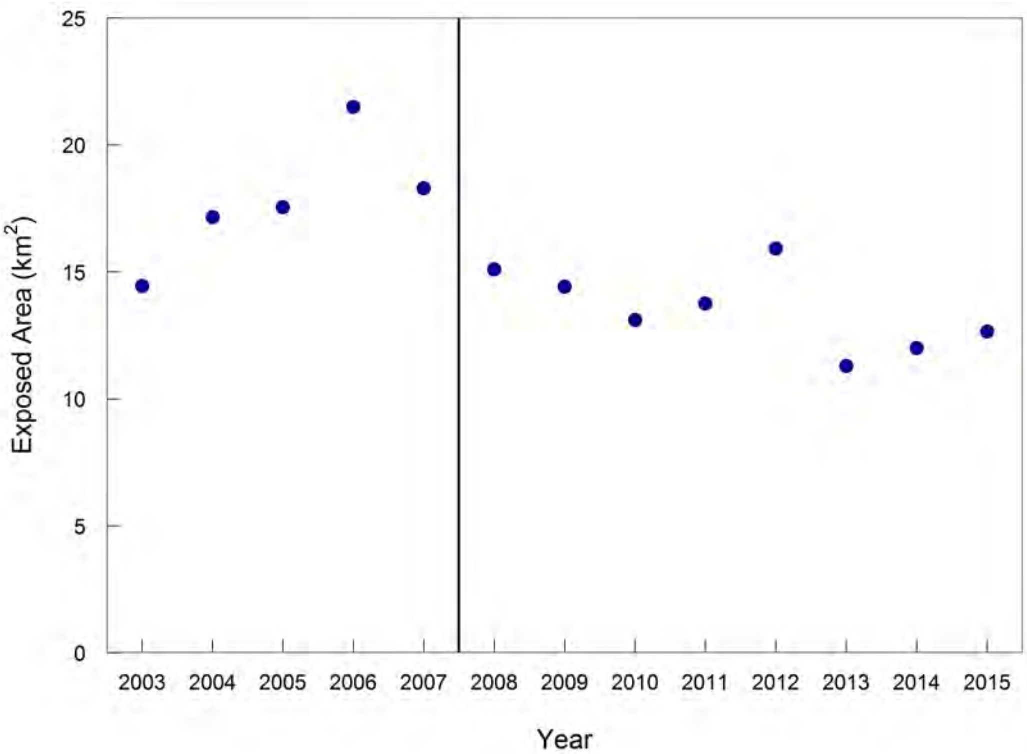


Figure 6: Total area exposed by all annual reductions in the LDR by year of operations, calculated using DRL discharge. The vertical line denotes the beginning on WUP flows in 2008.

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. Prior to 2015, 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 m³/s (Figure 2), resulting in high interannual variability during the October-January period (Figure 7). Decreased discharge variability post-WUP was recorded between March and May, where discharge trend changed from gradual increase pre-WUP to a stable flow or a slight gradual decrease post-WUP (Figure 7).

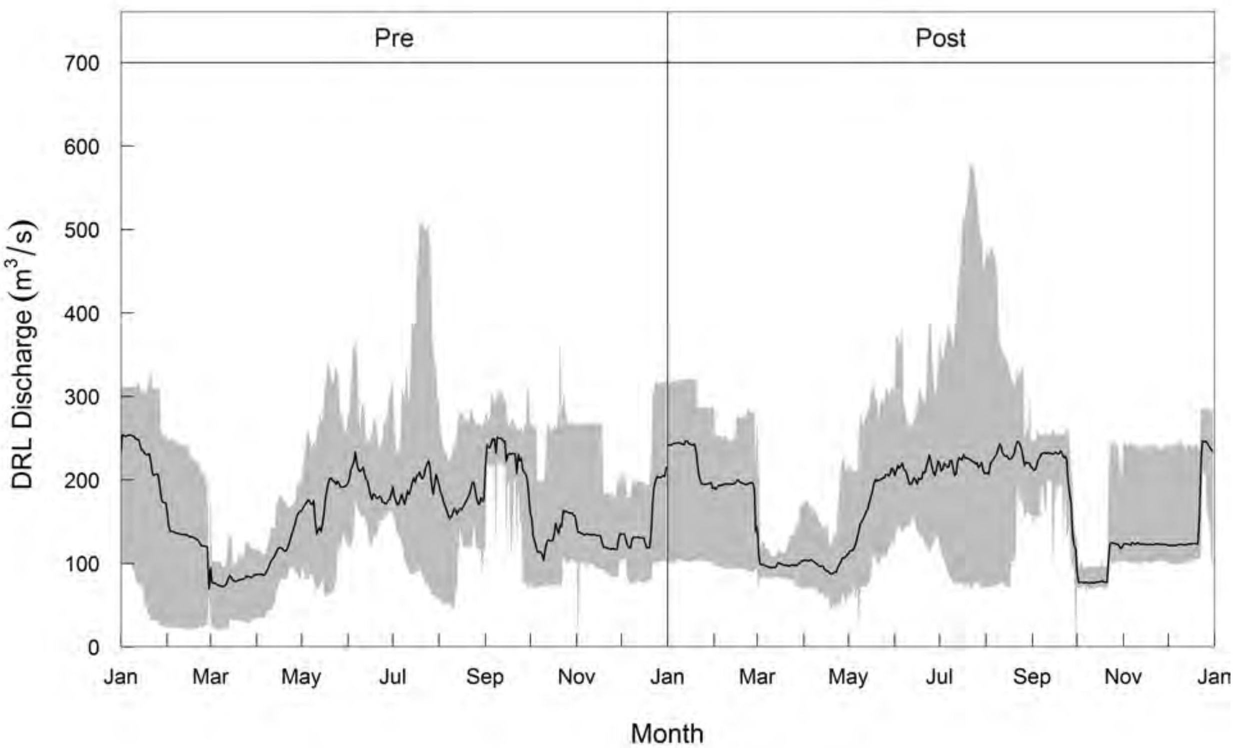


Figure 7: Minimum, maximum (grey ribbon) and mean (black line) discharge as measured at the WSC DRL gauge in the LDR by month during pre-WUP operations (2002 - 2007) and post-WUP operational implementation (2008 - 2015).

Although the total 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 total magnitude during pre-WUP conditions was higher in most years (Figure 8). Substantial differences in the total reduction magnitude between pre- and post-WUP operations were not identified in early post-WUP years. However, during 2013-2015, reductions had narrow ranges and were generally smaller than pre-WUP operations.

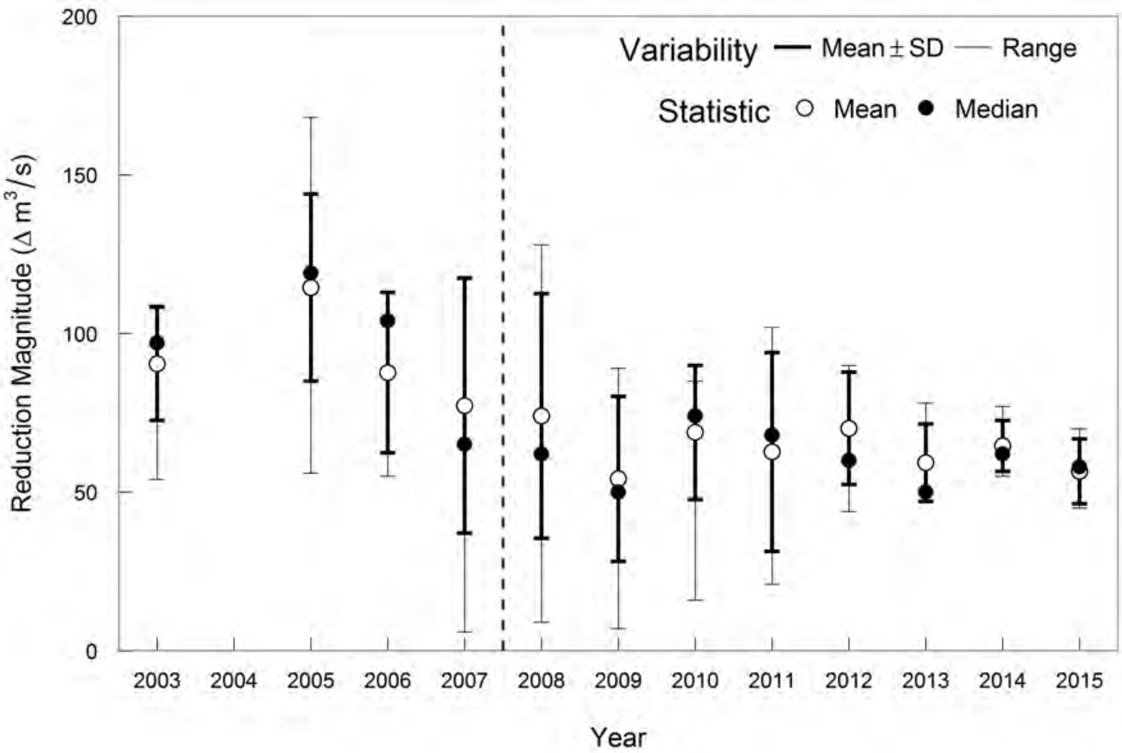


Figure 8: Reduction magnitude (Δm³/s) by year, depicting annual range (min, max), mean, and median, as well as mean ± SD.

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.

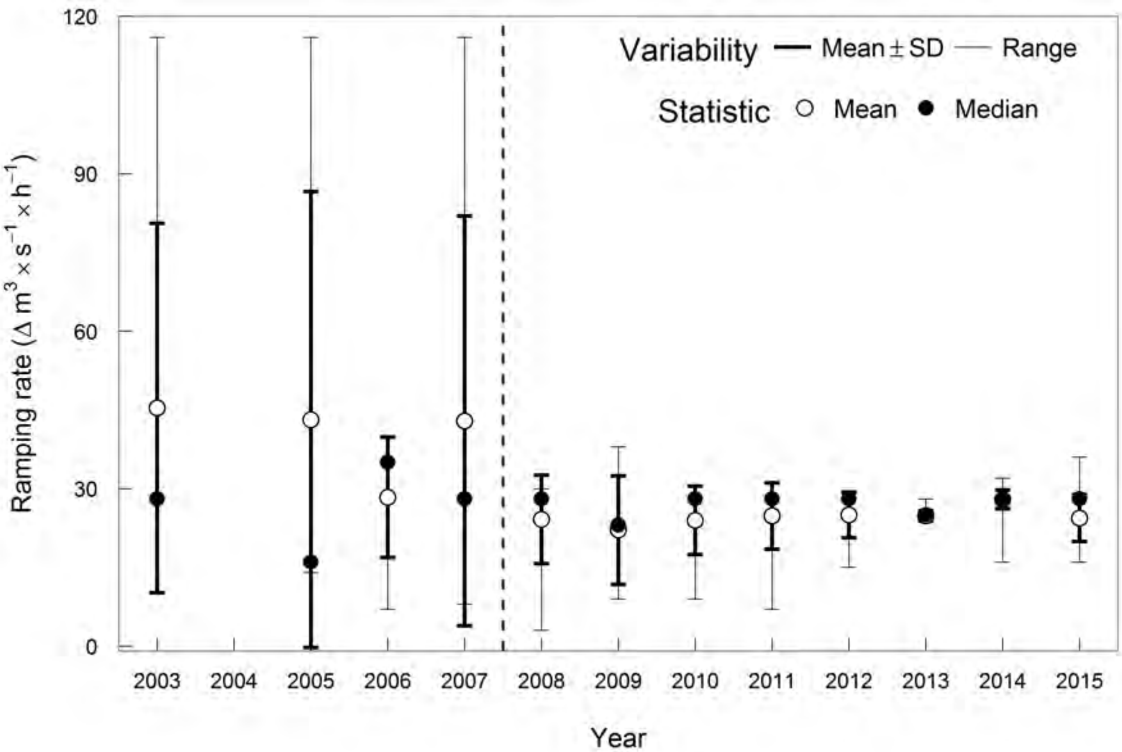
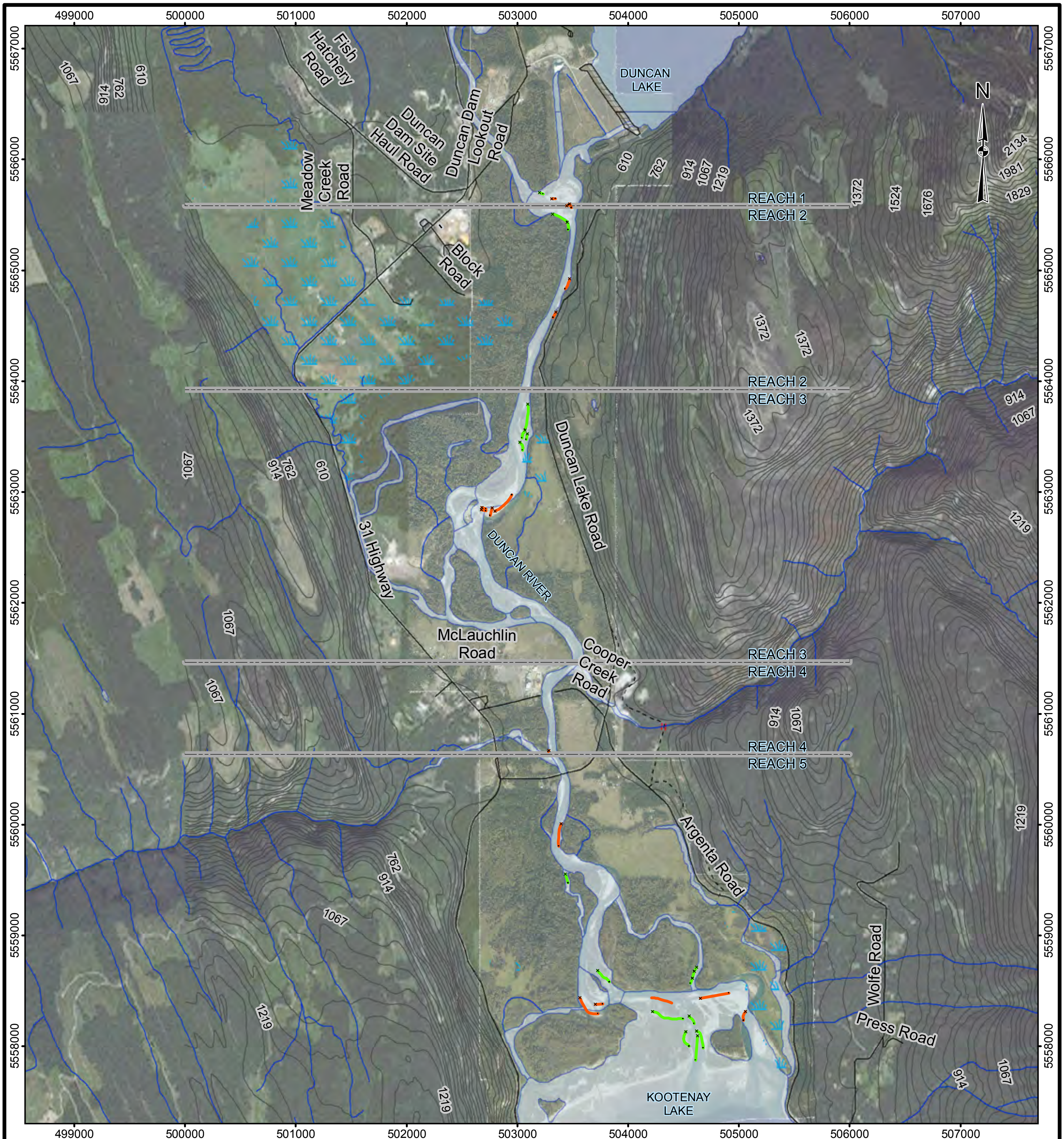


Figure 9: Ramping rate ($\Delta m^3 s^{-1} h^{-1}$) by year, depicting annual range (min, max), mean, and median, as well as mean ± SD.

3.4 Fish Abundance Assessment

Fish abundance assessment snorkel sites from study Year 6 (2013) and study Year 7 (2014) are presented in Figure 10 and Figure 11, respectively. A total of 46 sites and 5,622 m of shoreline were surveyed during the Year 8 (2015) snorkeling fish abundance assessment, with a total of 1,091 fish counted across all sites and strata (Figure 12 and Table 20). The lowest mean counts of Mountain Whitefish (5.1 fish/site) were recorded in shallow, slack sites, whereas deep sites had high mean counts of Mountain Whitefish – 23.1 fish/site in fast sites, and 17.6 fish/site in slack sites (Table 20). The lowest mean counts of Rainbow Trout (5.2 fish/site) were recorded in deep, slack sites, whereas shallow sites had high Rainbow Trout mean counts, with 11.7 fish/site at slack sites, and 11.2 fish/site at fast sites.



LEGEND

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

1 0 1 Kilometers

SCALE 1:50,000

REFERENCE

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DATUM: NAD83 PROJECTION: UTM ZONE 11

PROJECT

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

TITLE

SNORKEL SITE LOCATIONS (SEPT. 2013)

Golder Associates

PROJECT		12-1492-0117	FILE No.	
DESIGN	BH	14 Apr. 2014	SCALE AS SHOWN	REV. 0
GIS	KLN	19 May 2015		
CHECK	BH	20 Apr. 2017	**FIGURE: 10**	
REVIEW	DA	20 Apr. 2017		

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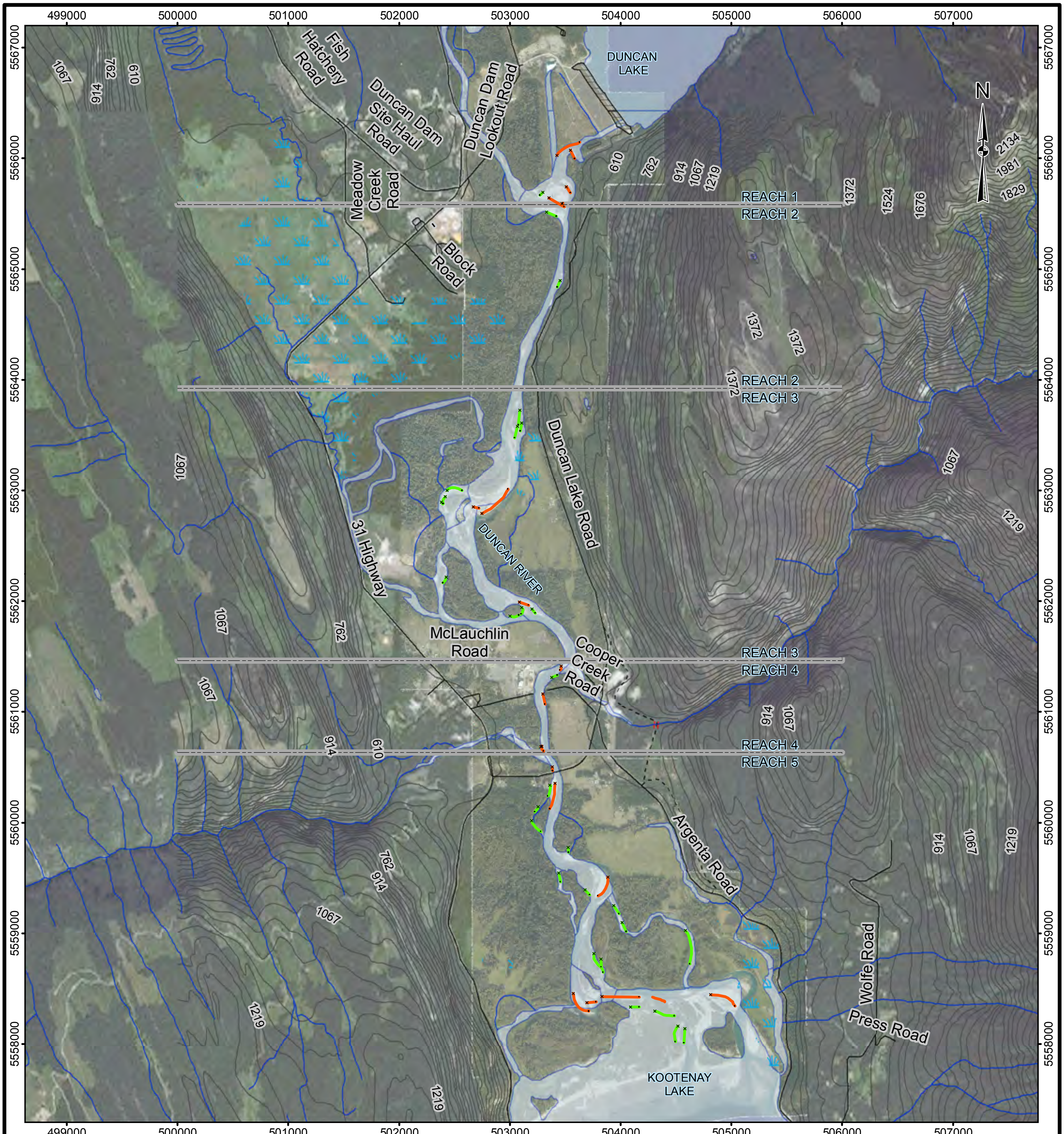

LEGEND

SNORKEL SITE LOCATION (2014)

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

1 0 1 Kilometers

SCALE 1:50,000

REFERENCE

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DATUM: NAD83 PROJECTION: UTM ZONE 11

PROJECT

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

TITLE

SNORKEL SITE LOCATIONS (SEPT. 2014)

Golder Associates

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CHECK	BH	20 Apr. 2017		
REVIEW	DA	20 Apr. 2017		

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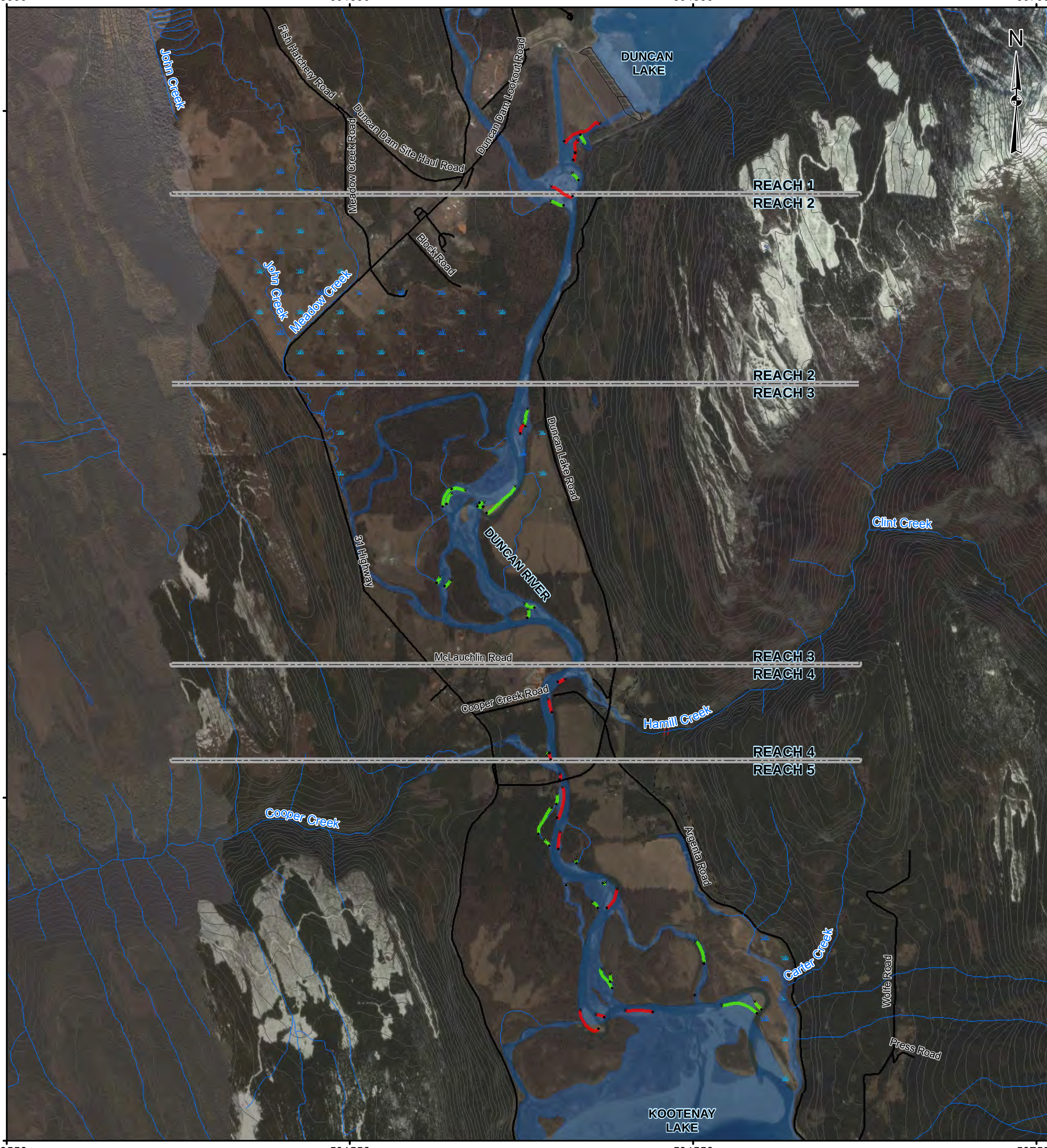

LEGEND

SNORKEL SITE LOCATION (2015)

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

1 0 1

SCALE 1:50,000 KILOMETRES

REFERENCE

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DATUM: NAD83 PROJECTION: UTM ZONE 11

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

TITLE
SNORKEL SITE LOCATIONS (SEPT. 2015)

Golder Associates

PROJECT		1535517	FILE No.	
DESIGN	BH	13 Mar. 2017	SCALE AS SHOWN	REV. 0
GIS	JG	13 Mar. 2017	FIGURE: 12	
CHECK	BH	20 Apr. 2017		
REVIEW	DA	20 Apr. 2017		

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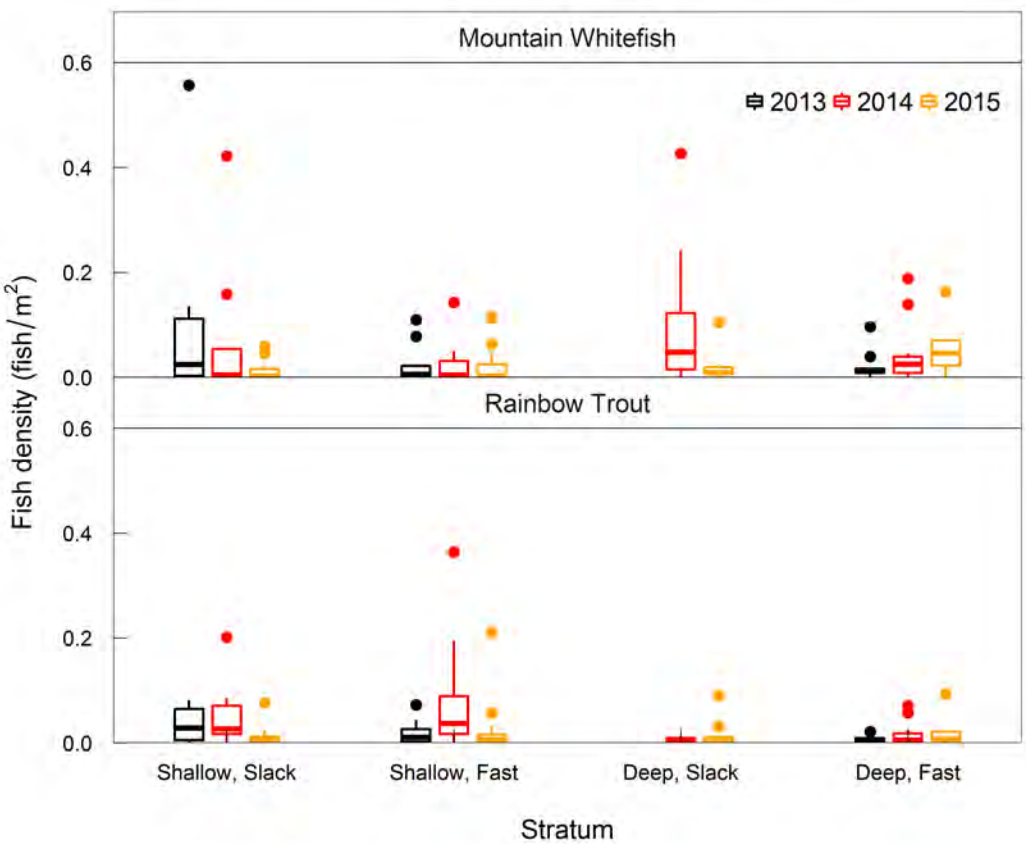

Figure 13: Boxplots of density (fish/m²) across species, depth, and flow strata for 2013-2015 data. Each box represents the 25th and 75th quantiles (bottom and top lines, respectively), and the median (middle bold line); whiskers extend to 1.5 times the interquartile distance; outliers are shown as individual points.

Table 20: Summary of fish counts across depth and flow strata, as recorded from Year 8 (September 2015) snorkeling surveys.

Stratum	Depth	Flow	NSite	Mountain Whitefish			Rainbow Trout		
				N	Mean	SD	N	Mean	SD
1	Shallow	Slack	11	56	5.1	10.2	129	11.7	28.3
2	Shallow	Fast	16	265	16.6	34.5	179	11.2	12.6
3	Deep	Slack	11	176	17.6	19.6	52	5.2	6.1
4	Deep	Fast	8	185	23.1	22.8	49	6.1	6.0
Total				**682**			**409**		

The variability of fish density within strata is clearly seen in Figure 13. Mountain Whitefish zero densities ranged from 13% to 45% of the cases, calculated across strata. Non-zero densities ranged from 0.001 fish/m² to 0.16 fish/m². In the Shallow/Slack stratum, Year 8 densities were lower than either Year 6 or Year 7 data; however, the densities were comparable in the Shallow/Fast stratum, and generally higher than previous years in the Deep/Fast stratum. In Year 8, sites with zero densities of observed Rainbow Trout ranged from 6% to

45%, calculated across strata. Non-zero densities ranged from 0.001 fish/m² to 0.21 fish/m² (Figure 13). Densities at shallow sites were generally lower in 2015 in comparison to 2013/2014 data; at deep sites, densities were comparable to previous years.

For both Rainbow Trout and Mountain Whitefish analyses, flow was not a significant variable, since the 95% CRI values of the "Fast" stratum coefficients overlapped zero. Once the variable was removed, the models where fish density was a function of intercept, depth, year, and a random site effect were chosen as final models for both species. 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 -1.151, 95% CRI of -1.796 and -0.503). Conversely, there was a significant increase in Mountain Whitefish log densities at deep sites (estimated coefficient of 1.102, 95% CRI of 0.369 and 1.915).

Abundance estimates for Rainbow Trout in the shallow strata were considerably higher in 2014 than in 2013 and 2015 (Figure 14). Rainbow Trout abundance in the deep strata was estimated to be low, ranging from 1,320 fish in 2015 to 3,374 fish in 2014. Abundance estimates for Mountain Whitefish decreased throughout the sampling program across both depth strata (Figure 15). The highest mean abundances for the species were estimated in the deep stratum, ranging from 5,417 fish in 2015 to 10,670 fish in 2013.

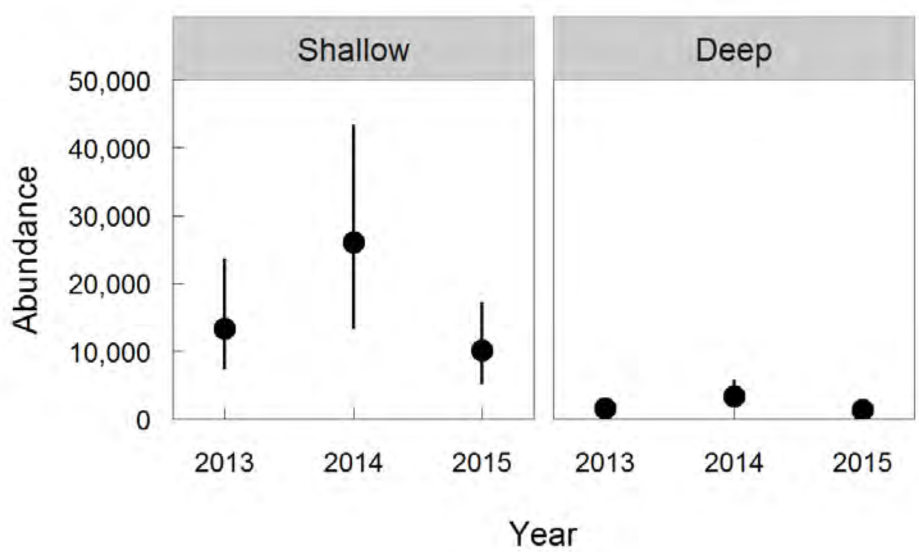

Figure 14: Mean Rainbow Trout abundance (density x stratum area) and their respective 95% credibility intervals, plotted by depth strata.

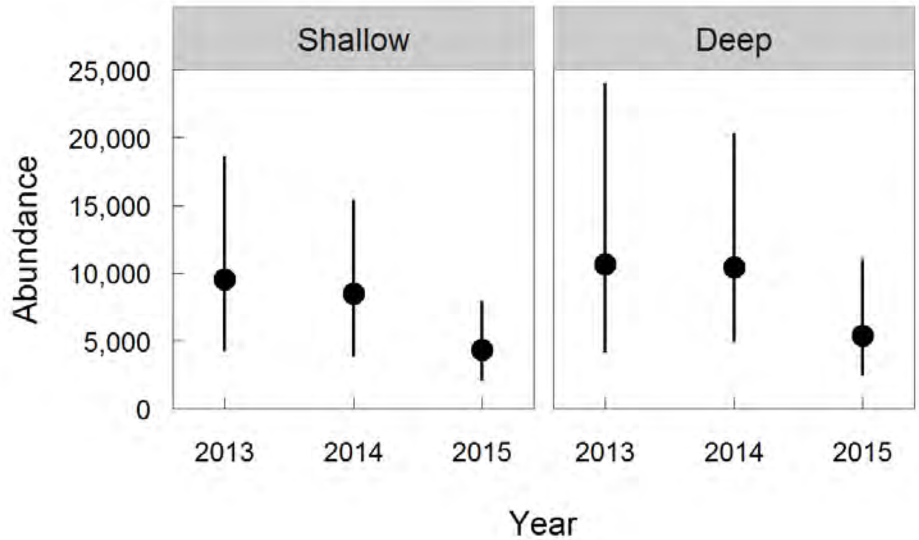

Figure 15: Mean Mountain Whitefish abundance (density x stratum area) and their respective 95% credibility intervals, plotted by depth strata.

Total abundance estimates for Mountain Whitefish decreased across the three sampling years, while Rainbow Trout had an abundance peak in 2014, followed by a reduction in abundance estimates in 2015 (Table 21).

Table 21: 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.

Species	Year 6 (Fall 2013)	Year 7 (Fall 2014)	Year 8 (Fall 2015)
MW	20,210 (9,280 – 39,300)	18,940 (10,210 – 33,780)	9,760 (4,910 – 17,280)
RB	14,910 (8,050 – 26,000)	29,410 (15,440 – 48,030)	11,420 (5,940 – 19,280)

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 six years of stranding assessments (Years 3 to 8: 2010-2015) 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 16).

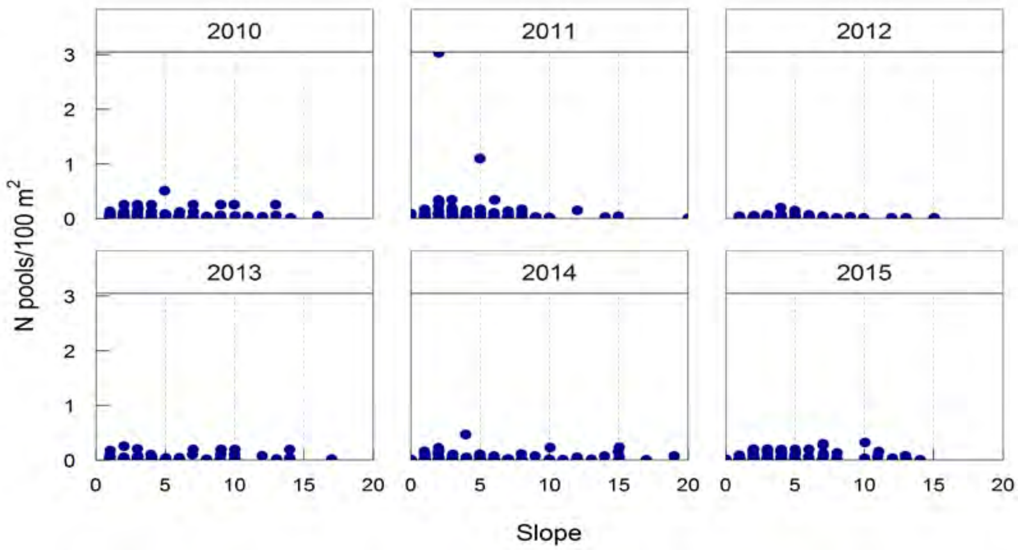

Figure 16: Density of pools recorded per reduction vs. habitat slope as a continuous variable, 2010-2015.

While pool densities in random sites exhibited slightly higher variation in comparison to index sites in some years, the majority of recorded pool densities were fairly low, often lower than those recorded at index sites (Figure 17).

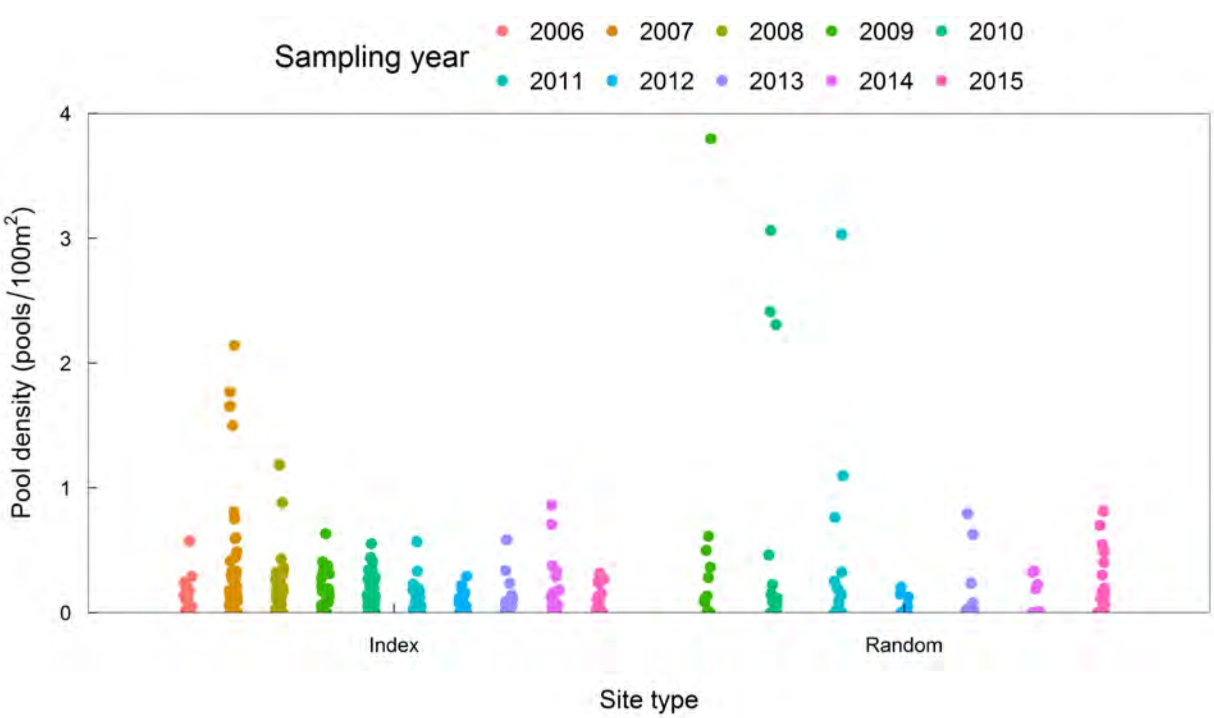

Figure 17: Pool densities, recorded during 2006-2015 stranding years, plotted against site type.

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 (Aug-Oct) and the winter period (Dec-Mar) when flow reductions typically occur to meet operation targets, the mean number of pools that formed during the stranding surveys was similar in 2012-2016 and more variable in 2010-2012 (Figure 18).

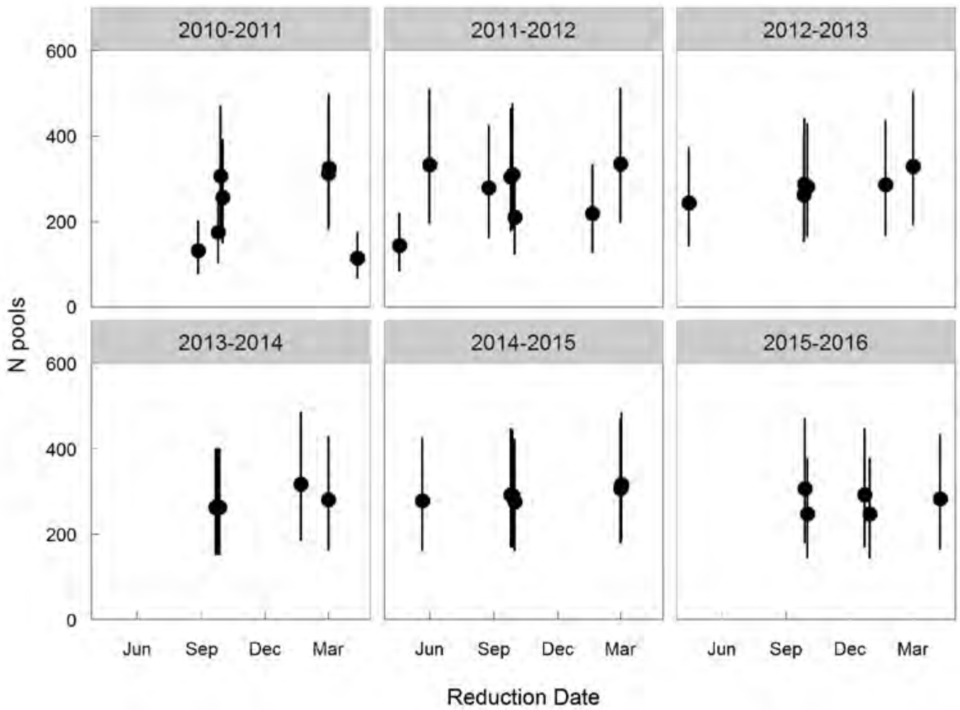

Figure 18: Mean estimates of pool numbers formed during the 2010-2016 stranding events, plotted by reduction and stranding year. Error bars are 95% credibility intervals.

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 0.575 for Rainbow Trout (mean value; 95% credibility interval of 0.514-0.632) and 0.424 for Mountain Whitefish (mean value; 95% credibility interval of 0.293-0.541).

The variability in the number of fish stranded per pool was similar between low (0-4%) and high slope (>4%) habitat (Figure 19). This indicated that slope was not a factor influencing pool stranding. A large difference in pool stranding of Rainbow Trout was observed with season, where pool stranding was substantially higher in the fall reductions (Figure 20).

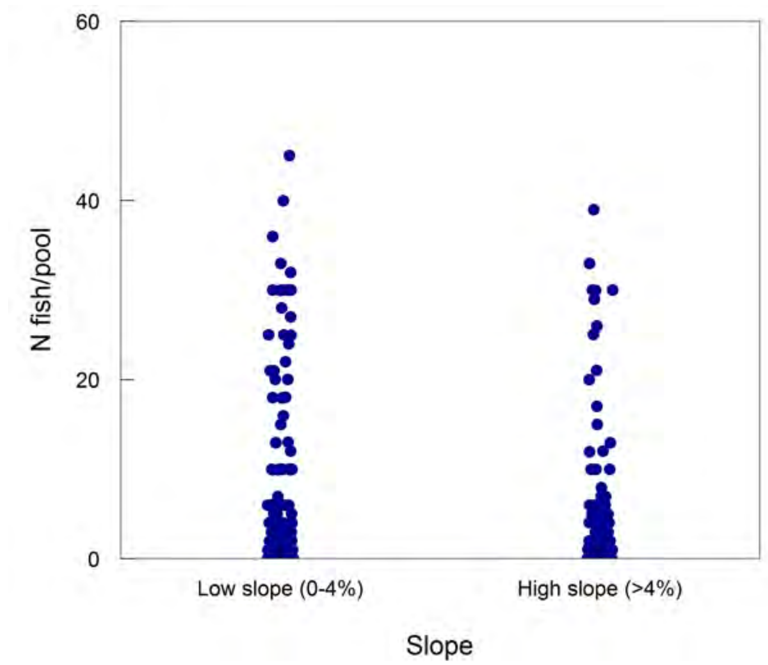

Figure 19: Number of collected fish per pool, plotted by low-slope and high-slope habitat, 2010 - 2016.

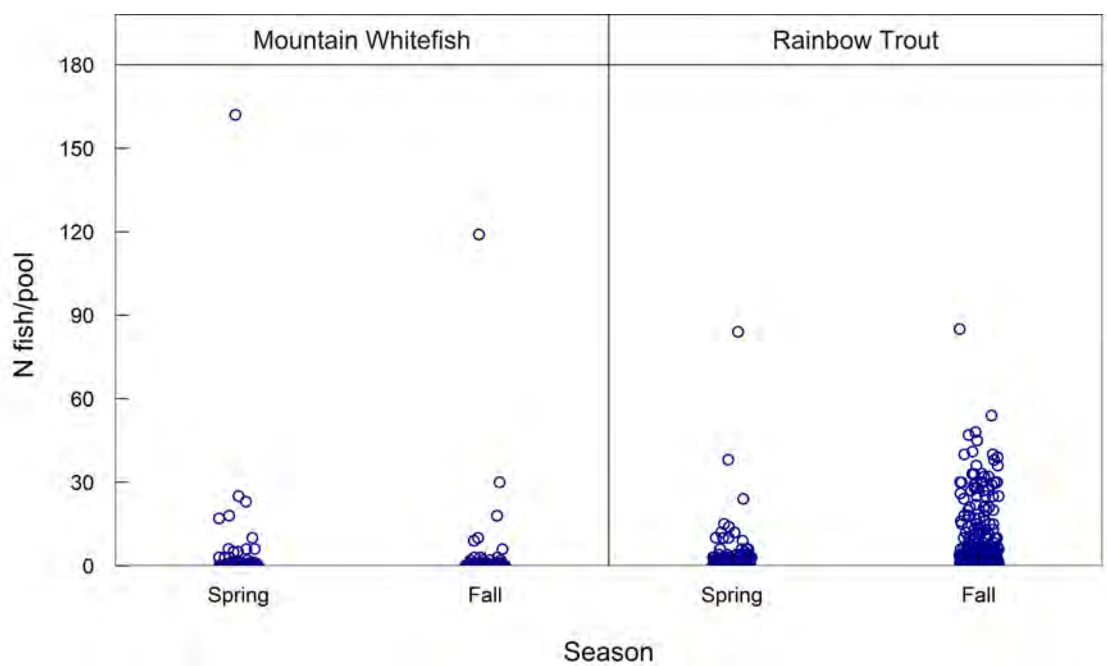

Figure 20: Counts of fish/pool, recorded during 2006-2016 stranding surveys, and plotted by season and species.

The density of pool-stranded fish differed by dominant substrate size and by species (Figure 21). Generally, Mountain Whitefish pool stranding density was low, except for pools with silt, and small to large gravel. However, the relationship may be due to 1) relatively low numbers of pool-stranded Mountain Whitefish (i.e.,sparse data), 2) unexamined potential relationships between substrate type and pool densities.

For Rainbow Trout, pool-stranded fish densities were overall similar across different substrate sizes, apart from large gravel, very large gravel, and small cobble, where few pool-stranded Rainbow Trout were recorded (Figure 21).

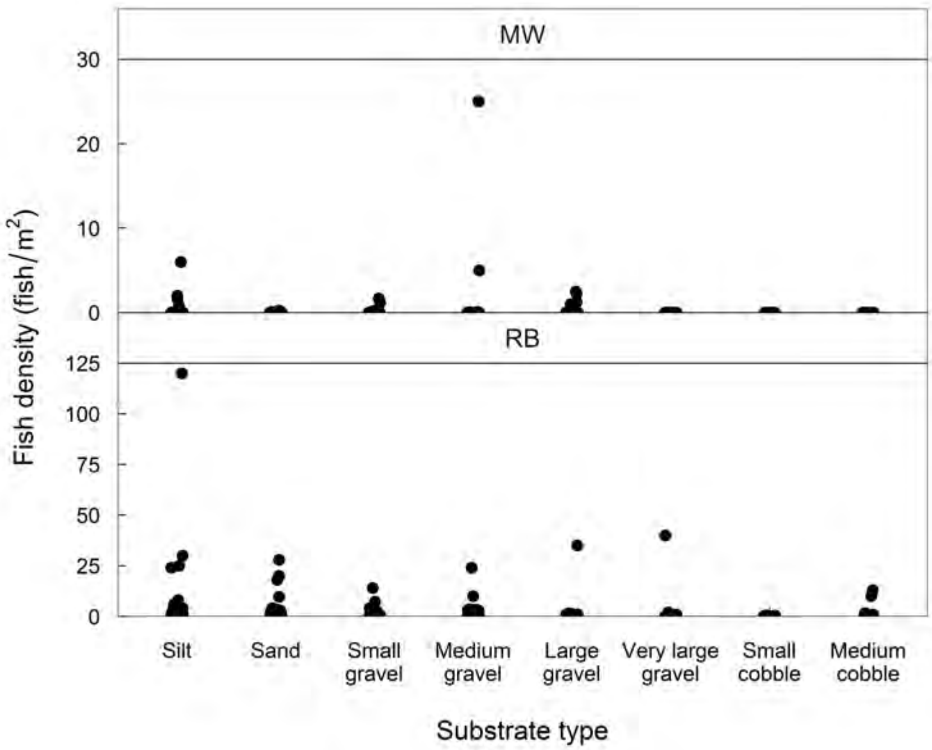

Figure 21: Scatter plot of pool-stranded fish density (fish/m²) vs. dominant pool substrate size, 2006-2016, plotted by species.

The mean number of Rainbow Trout juveniles per pool for the spring season (January to June) was estimated to be 0.772 (0.389 – 1.508) fish/pool (Table 22). In contrast, the mean number of Rainbow Trout juveniles stranded per pool in the fall (July to December) was estimated at 4.446 (0.965 – 22.065). The season effect on pool stranding of Rainbow Trout was found to be significant ($p < 0.05$), with mean fall stranding estimates approximately six times higher than those for winter/spring. For Mountain Whitefish, no strong effect of season was found, with fall stranding estimated to be lower, but more variable, than spring stranding (Table 22).

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 22, Figure 23). For Rainbow Trout, means estimates of pool stranded individuals were higher and more variable in the fall season in most years. In Year 8, Rainbow Trout per pool were highest and most variable in the winter season (Figure 22). With the exception of 2010-2011 and 2011-2012 for Mountain Whitefish, the resultant pool stranding estimates indicated lower levels of stranding in the months of January and April in comparison to the fall season. The spike of Mountain Whitefish presence in spring 2011-2012 resulted from a single pool with 162 Mountain Whitefish (Figure 23) 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.

Table 22: Mean estimates of counts of Rainbow Trout and Mountain Whitefish per pool, by season, 2010 - 2016; 95% credibility intervals are provided.

Species	Season	Mean	95% credibility interval	
			Lower limit	Upper limit
Rainbow Trout	Spring	0.772	0.389	1.508
	Fall	4.446	0.965	22.065
Mountain Whitefish	Spring	0.074	0.012	0.413
	Fall	0.036	0.001	1.623

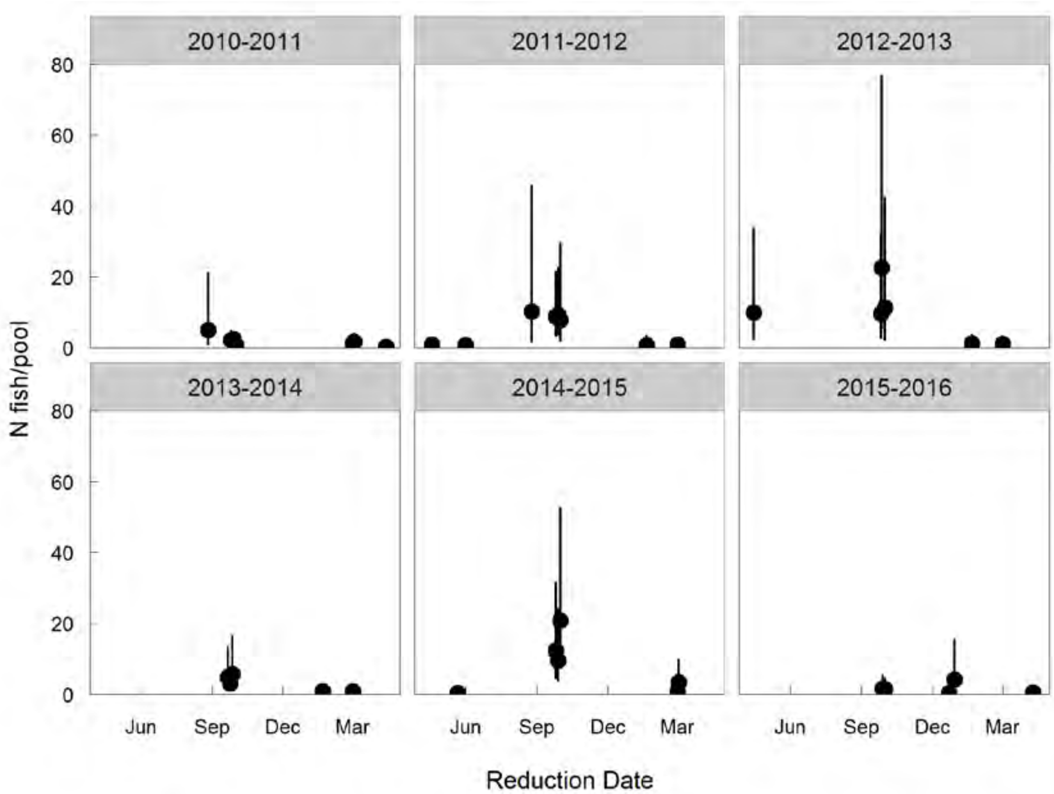

Figure 22: Mean estimates of counts of pool-stranded Rainbow Trout during the 2010-2016 stranding events, plotted by reduction and stranding year. Error bars are 95% credibility intervals.

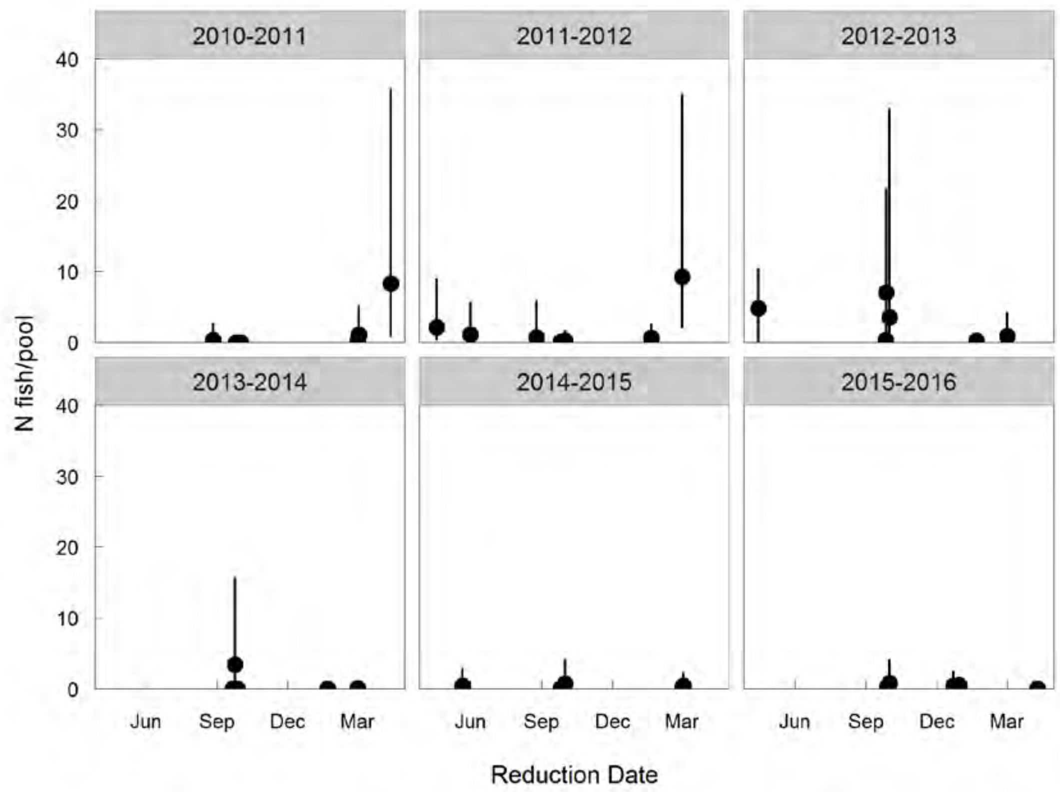

Figure 23: Mean estimates of counts of pool-stranded Mountain Whitefish during the 2010-2016 stranding events, plotted by reduction and stranding year. Error bars are 95% credibility intervals.

3.5.3 Interstitial Stranding

Between Year 4 (2011 - 2012) to Year 8 (2015 - 2016) of this program, 24 Rainbow Trout and 2 Mountain Whitefish were found to be interstitially stranded (Figure 24). Over the last three study years (Years 6 to 8) when interstitial sample methodology was standardized with transect sampling, only one interstitially stranded Rainbow Trout was observed (Figure 24). A relationship between stranded fish counts and slope was not observed in the data (Figure 24 and Figure 25); therefore, the variable was not included in the model.

A relationship between interstitial stranding and ramping rates was suspected (Figure 25); however, the Bayesian analysis indicated that the variable was not significant, and it was removed from the model (data not shown). As more data are accumulated, the relationship may become significant. The effect should therefore be re-evaluated in future work.

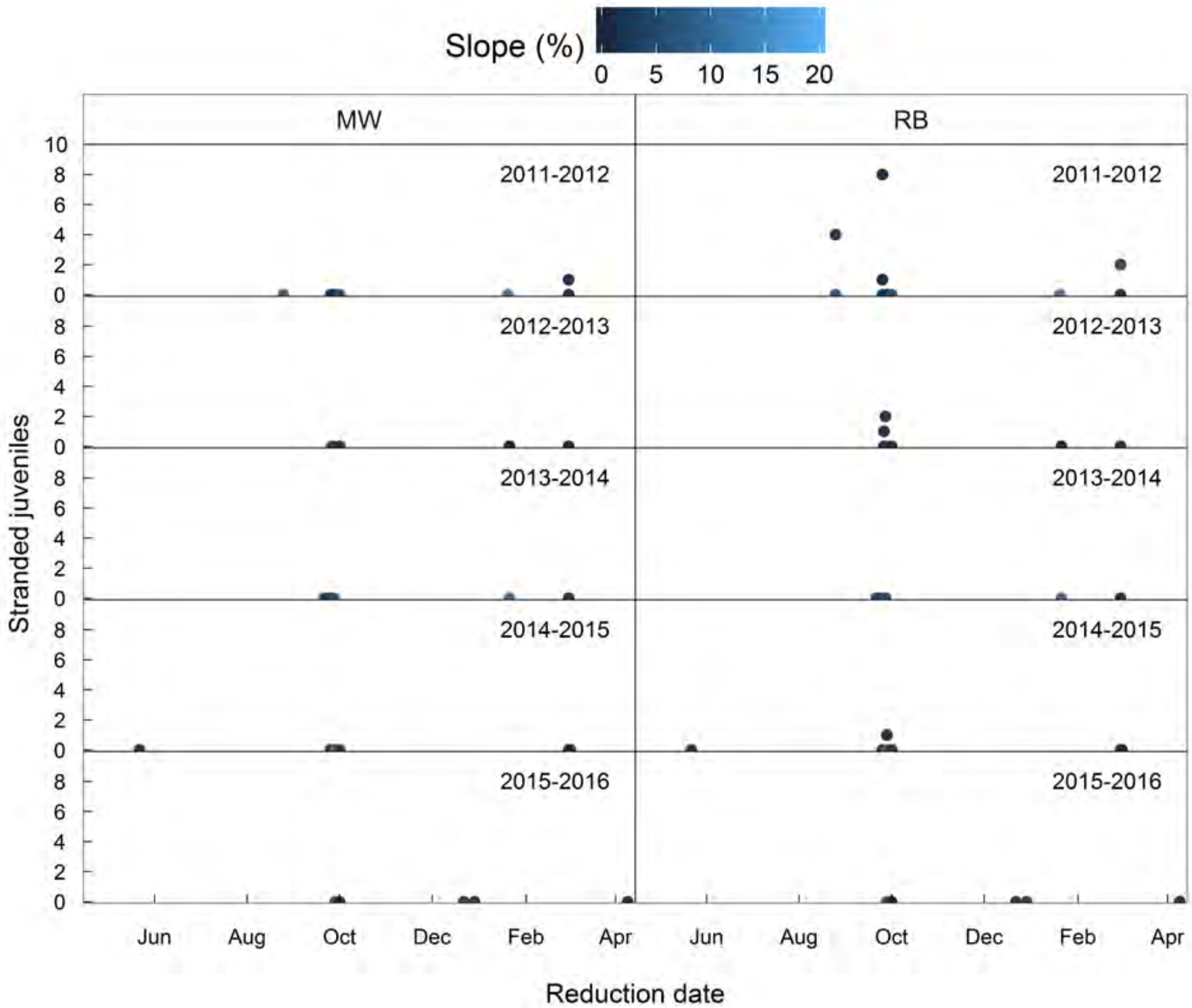

Figure 24: Counts of 2011-2016 interstitially stranded Mountain Whitefish and Rainbow Trout, plotted by study year, reduction date, and colour-coded by continuous slope (%).

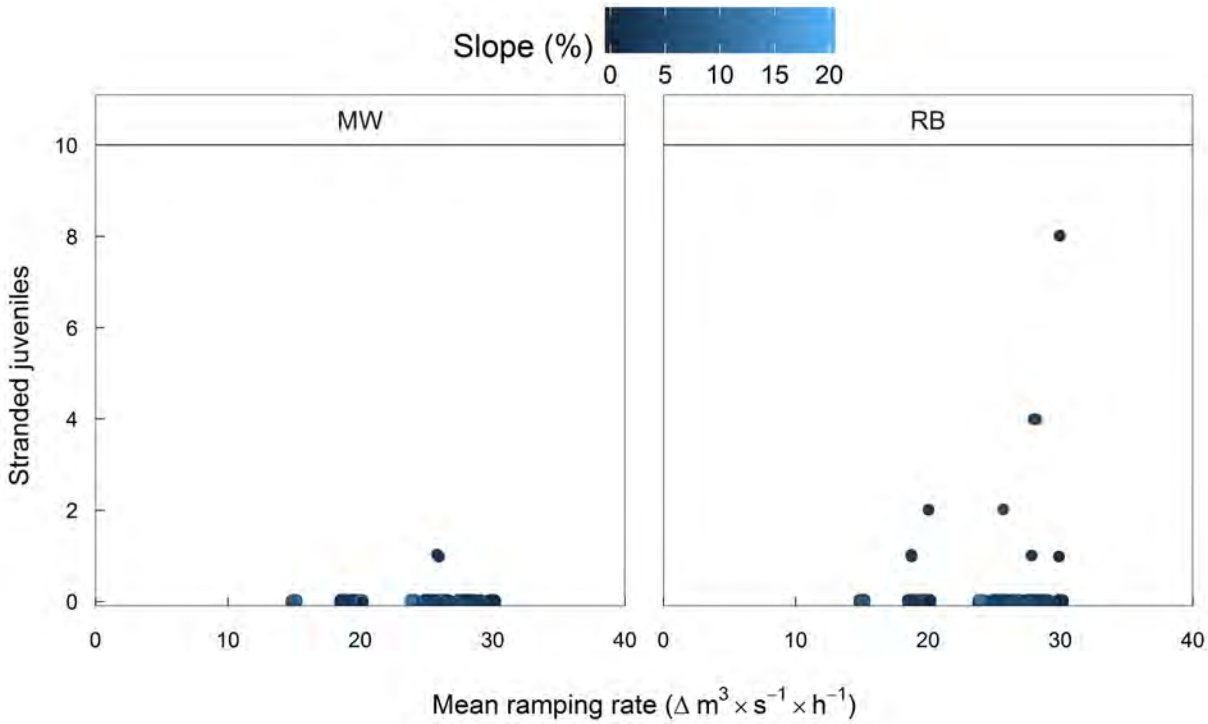

Figure 25: Counts of 2011-2016 interstitially stranded Mountain Whitefish and Rainbow Trout vs. mean ramping rate, plotted by species and colour-coded by slope (%).

The effect of index/random site on interstitial stranding of Rainbow Trout was found to be statistically significant, with a significant increase in interstitial stranding of fish at random sites (probability of stranding at an index site estimated as 0.016 [95% CRI of 0.006-0.036], and at a random site estimated as 0.049 [95% CRI of 0.007 0.275]). However, total dewatered area is calculated using relationships for low-slope (0-4%) and high-slope (>4%) areas, and no relationship has been developed for these areas as a function of index/random sites. Therefore, the final model used to estimate overall interstitial stranding contained no variable to describe whether a site was random or index. For Mountain Whitefish, the effect of index/random site on interstitial stranding was not found to be significant.

Summed by reduction, mean Rainbow Trout interstitial stranding estimates ranged from 1,969 fish (April 19, 2011) to 9,565 fish (March 01, 2012; Figure 26). The 95% credibility values ranged between 3,096 fish (April 19, 2011) to 15,036 fish (March 01, 2012; Figure 26). Summed by reduction, mean Mountain Whitefish interstitial stranding estimates ranged from 216 fish (April 19, 2011) to 1,048 fish (March 01, 2012; Figure 27). The 95% credibility values ranged between 884 fish (April 19, 2011) to 4,294 fish (March 01, 2012; Figure 27).

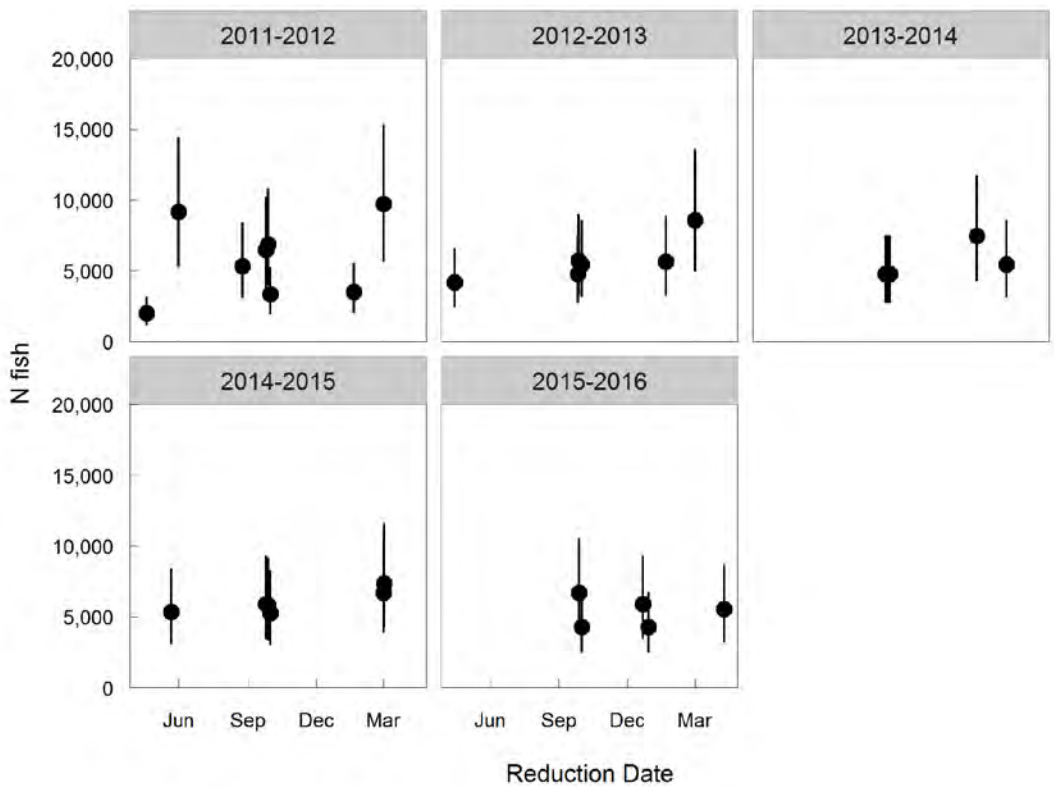

Figure 26: Mean estimates of Rainbow Trout juveniles interstitially stranded during the 2011-2016 stranding events, plotted by study year and reduction date. Error bars are 95% credibility intervals.

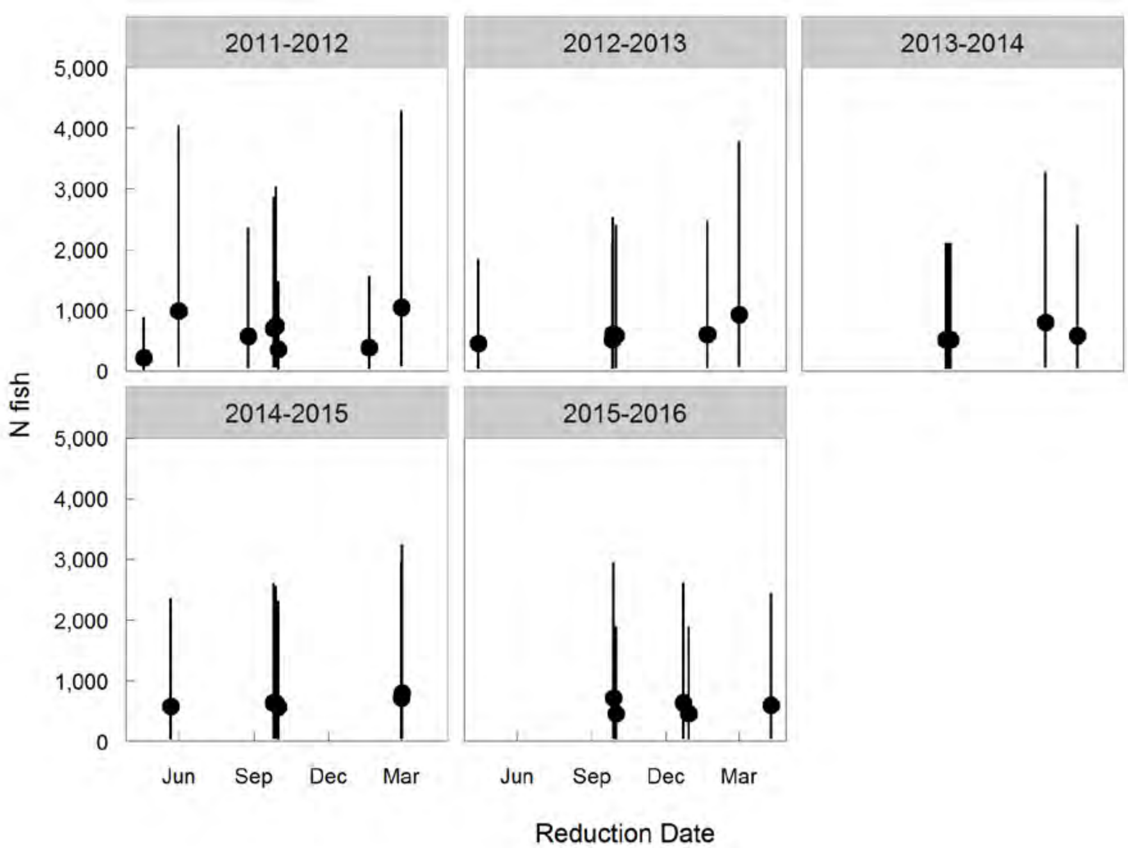

Figure 27: Mean estimates of Mountain Whitefish juveniles and fry interstitially stranded during the 2011-2016 stranding events, plotted by study year and reduction date. Error bars are 95% credibility intervals.

3.5.4 Total Stranding Estimates

For Rainbow Trout, total annual pool stranding increased between 2010 and 2012, and generally decreased throughout 2013-2016 (Figure 28). The highest stranding was estimated to have occurred in 2012-2013, with a mean estimate of 15,420 fish, and 95% CRI of 5,600-39,230 fish. For Mountain Whitefish, total annual pool stranding increased between 2010 and 2012, and was generally low throughout 2013-2016 (Figure 29). The highest stranding was estimated to have occurred in 2012-2013, with a mean estimate of 5,180 fish, and 95% CRI of 110-23,220 fish.

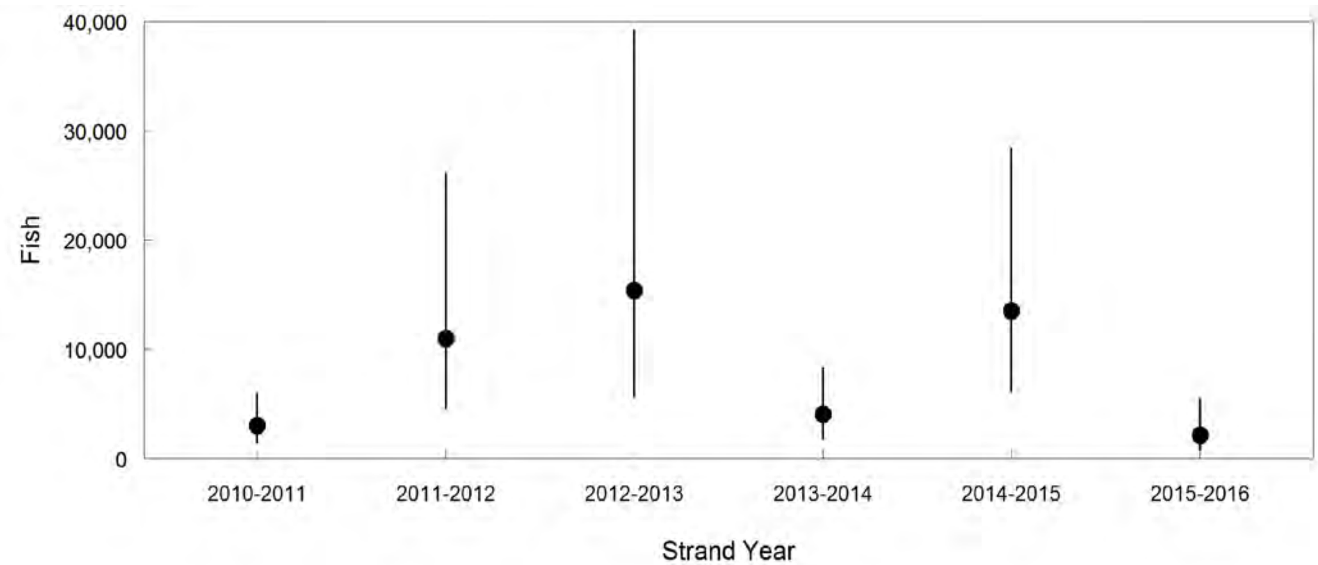

Figure 28: Mean estimates of total pool-stranded Rainbow Trout during the 2010-2016 stranding years, plotted by stranding year. Error bars are 95% credibility intervals.

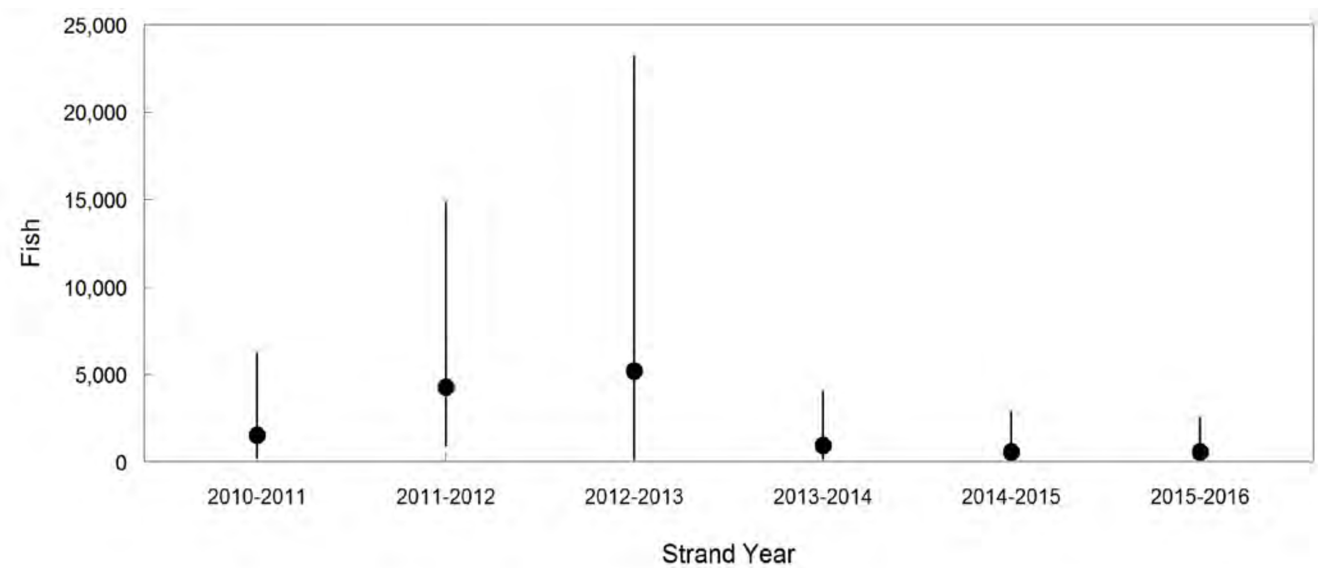

Figure 29: Mean estimates of total pool-stranded Mountain Whitefish during the 2010-2016 stranding years, plotted by stranding year. Error bars are 95% credibility intervals.

Total yearly estimates of Rainbow Trout and Mountain Whitefish interstitial stranding are summarized in Figure 30 and Figure 31, respectively. Estimated interstitial stranding of both species decreased from 2011-2012 and remained relatively stable between 2012-2013 and 2015-2016 stranding years, although uncertainty remained extremely high. Estimated Mountain Whitefish interstitial stranding, with mean yearly estimates of ≤5,000 fish, was considerably lower than that of Rainbow Trout, where mean yearly estimates ranged between ~26,000 and ~46,000 fish.

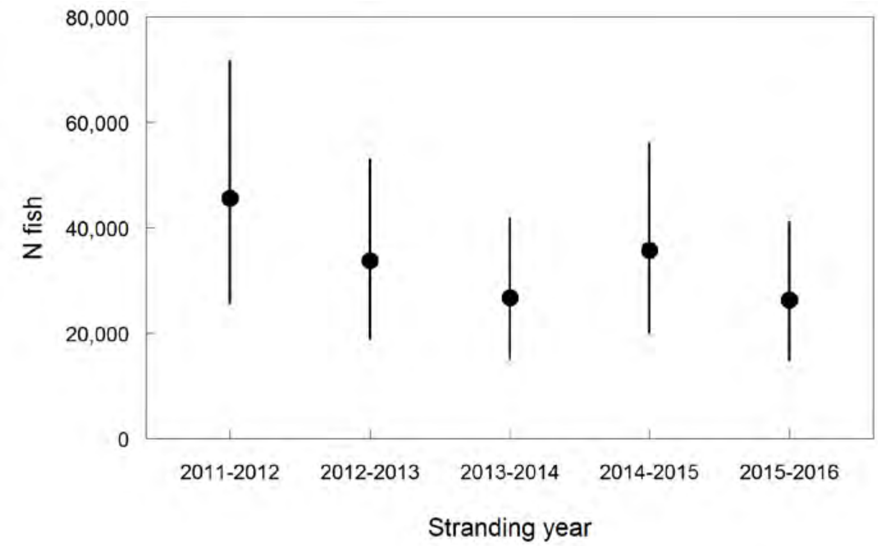

Figure 30: Mean estimates of total interstitially stranded Rainbow Trout during the 2011-2016 stranding years, plotted by stranding year. Error bars are 95% credibility intervals.

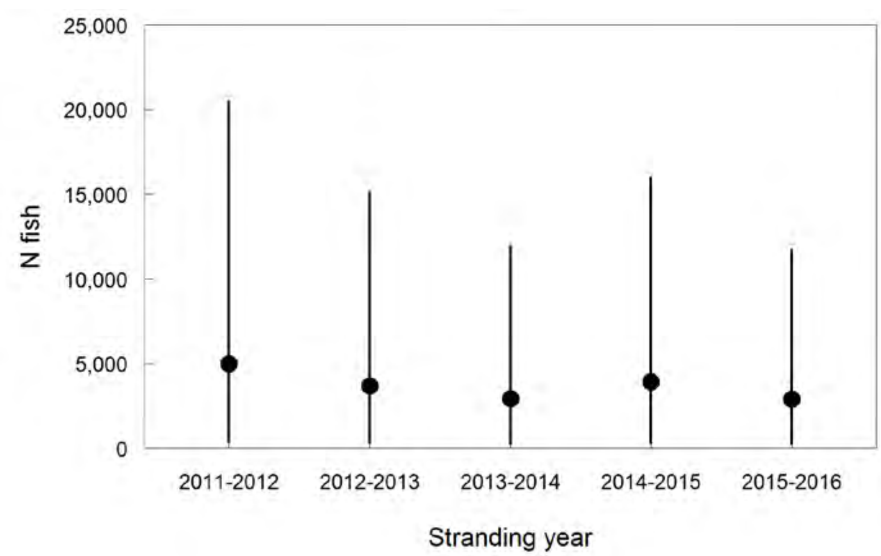

Figure 31: Mean estimates of total interstitially stranded Mountain Whitefish during the 2011-2016 stranding years, plotted by reduction and stranding year. Error bars are 95% credibility intervals.

The estimated percentage stranding out of estimated abundance values for each of 2013-2016 stranding years illustrated the exceedingly high uncertainty associated with both estimates for both species (Figure 32, Figure 33). For Rainbow Trout, in all three years of sampling for abundance, both mean and lower 95% CRL stranding estimates were >100% of population estimates (Figure 32).

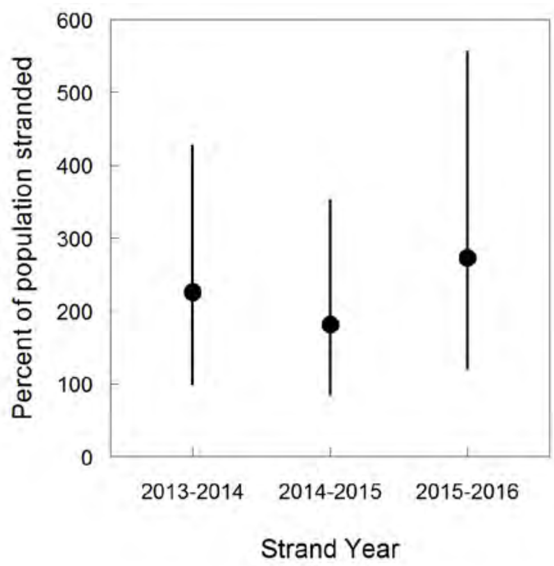

Figure 32: Mean estimates of percent stranding out of total estimated abundance of Rainbow Trout during the 2013-2016 stranding years, plotted by stranding year. Error bars are 95% credibility intervals.

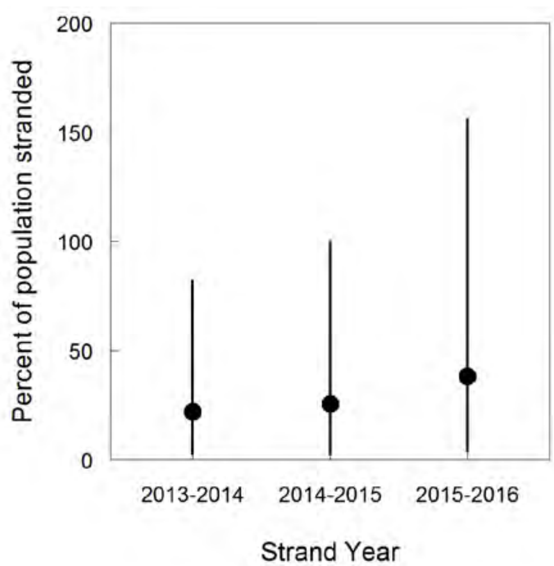

Figure 33: Mean estimates of percent stranding out of total estimated abundance of Mountain Whitefish during the 2013-2016 stranding years, plotted by stranding year. Error bars are 95% credibility intervals.

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). Operational variables related to stranding that are currently not addressed by the ASPD are wetted history and season (Poisson and Golder 2010). These variables were 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. The purpose of the late winter flow reductions is 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 area dewatered during all annual flow reductions was 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 3.3 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, especially 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.</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 has had different methodologies and varying study foci through the years, it is not possible to provide comparable fish stranding estimates from the pre-WUP and post-WUP periods. Therefore, only assessments on the amount and rate of habitat dewatering can be made in regards 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 8 assessments, 12 different species were encountered (five sportfish and seven non-sportfish species), but Rainbow Trout was the only species of interest with substantial numbers of stranded individuals.

4.2.1 Pool and Interstitial Stranding Rates

Current estimates for the number of Rainbow Trout 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 approximately six times 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.

Similar to previous study years (Golder 2014, 2015 and 2017), the current year's interstitial stranding estimates were high and uncertain. 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 slope was not observed. A relationship between interstitially stranded fish counts and ramping rate in the current program was also not found. These relationship should continue to be re-evaluated in the upcoming study 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 strongly indicates that the current interstitial estimates are upwardly biased and uncertain. The probable reason for the upward bias is that the modelled abundance for interstitial stranding assumes a Poisson distribution, and data scarcity in regards to interstitially stranded fish can lead to relatively high and uncertain estimated stranding as extensive amounts of habitat are dewatered. The high means and uncertainty related to interstitial stranding estimates indicate that additional data and further modeling refinement are necessary before a robust estimate of stranding loss as a percentage of fish abundance can be derived. With the exception of Year 7 (2014 - 2015), the estimated mean total interstitial stranding and uncertainty for both species of interest have generally decreased between Year 4 (2011 - 2012) and Year 8 (2015 - 2016) of the program. Given this trend, it is anticipated that with the current methodology utilized by the program, robust interstitial stranding estimates will not be obtained within the timeframe of this program.

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). Conversely, statistically significant relationships between interstitially stranded fish counts and slope in the current dataset were not found in Years 6 and 7 (Golder 2015 and 2017). Similarly, a relationship between stranded fish counts and slope was not observed in the data in the current year.

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. Also, in Year 6 and 7 a relationship between slope and the number of fish stranded in isolated pools was not identified (Golder 2015 and 2017). The dichotomous high (>4%)/low (0-4%) classification of slope habitat was believed to be too vague to determine the effects of slope on both pool and interstitial stranding. Therefore, the slope categories were reclassified when analysing pool formation and interstitial stranding.

A relationship between interstitially stranded fish counts and the new slope classifications was not observed in the data. 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.

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 dewatered area and suitable habitat. Based on the findings of previous study years (Golder and Poisson 2012, Golder 2017), 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 the current 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 future study years 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 three years analyzed, Rainbow Trout abundance increased from 2013 to 2014, followed by a sharp decrease in 2015. This decrease could be linked to a decline in Gerrard Rainbow escapement into the Duncan River that has been identified (Andrusak and Andrusak 2015). Current abundance estimates for Rainbow Trout 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 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.

The sum of the estimated interstitial and pool stranded Rainbow Trout from 2013 to 2015 had both mean and lower 95% CRL stranding estimates that were greater than 100% of population estimates. These findings differ from those reported in Year 6 and Year 7 reports (Golder 2015; Golder 2017), due to changes in the model specification and increased data set. Estimates for the number of Rainbow Trout juveniles stranded in pools obtained for this program were relatively precise and low, while the interstitial estimates continued to have high bias and uncertainty. Considering the very low numbers of interstitially stranded Rainbow Trout encountered during sampling, the high bias and uncertainty related to the interstitial modelling invalidates combining the pool and interstitial stranding estimates when determining DDM operations on Rainbow Trout populations.

Based on the overestimated interstitial stranding estimates, 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 Rainbow Trout populations. The further refinement of interstitial stranding rates may reduce the uncertainty of this finding. To address hypothesis H_{02} more confidently, it is critical that the uncertainties associated with the interstitial stranding estimates continue to be refined.

4.2.5 Mountain Whitefish

The total abundance estimates for Mountain Whitefish obtained using the updated abundance model decreased from Years 6 to 8. These findings differ from those reported in Year 6 and Year 7 reports (Golder 2015; Golder 2017), due to changes in the model specification and increased data set. 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 Mountain Whitefish population in the LDR may have declined since 2010. The confidence intervals associated with these estimates overlap, which indicated that the differences in these estimated values are not statistically significant. As the modelling used for the 2010 and current estimates were different, it is uncertain if this identified decline in Mountain Whitefish population is factual. As documented in the DDMMON-2 program (Thorley et al. 2011), significant differences in 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 Year 7 or the present study year.

Similar to study Years 6 and 7 (Golder 2015 and 2017), a seasonal effect on Mountain Whitefish stranding was not observed. In the current year, only 8 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 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.

4.3 Summary

The key findings for the Year 8 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 dewaters during DDM operations (Section 4.1.2)
- **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 results (Golder 2017), seasonal effect on pool stranding was found to be statistically significant (Section 4.2.1)
  - As in previous study years, interstitial stranding estimates continue to be upwardly biased and uncertain (Section 4.2.1)
  - Statistically significant relationships between interstitially stranded fish counts and slope in the current dataset were not found (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)

Progress has been made to reduce the uncertainties associated with interstitial stranding estimates of the target species. As the dataset continues to grow each year, the uncertainty related to this estimate is expected to decrease. Unfortunately, it is anticipated that within the timeframe of this program, robust interstitial stranding estimates will not be obtained.

Determining how estimates of 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 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 5.0 will work towards resolving the lack of confidence around these estimates.

5.0 RECOMMENDATIONS

Recommendations from the current year (Year 8) of the DDMMON-16 Lower Duncan River Fish Stranding Impact Monitoring Program are as follows:

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

2) Explore the feasibility of conducting several model runs with the updated TELEMAC 2D model from the DDMMON-3 program to refine the current wetted area of the Duncan River at varying DRL discharges. If completed, the dataset be updated, dewatered areas at each site can be recalculated using the most up to date model outputs, and the most up to date and representative data will be available to conduct the stranding analyses in Years 9 and 10.

3) Examine alternative modelling methodology to reduce the high bias and uncertainty related to interstitial stranding estimates.

These recommendations will focus sampling effort and 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 future fish stranding assessments, sampling methods should remain such that comparisons with historical data can be maintained.

6.0 REFERENCES

AMEC Earth and Environmental 2004. Duncan Dam Stranding Assessment Protocol. Prepared for BC Hydro, Castlegar, BC.

Andrusak, G.F. and H. Andrusak. 2015. Gerrard Rainbow Trout Growth and Condition with Kokanee Prey at Low Densities. Report prepared for Fish and Wildlife Compensation Program – Columbia Basin (Nelson, BC) by Redfish Consulting Ltd. (Nelson, BC). FWCP Report No. F-F15-15. 40 pp. + 8 app..

BC Hydro. 2004. Strategy for managing fish stranding impacts in the lower Duncan River associated with flow reductions at Duncan Dam. 22 pp + 6 app.

BC Hydro. 2008. Lower Duncan River Water Use Plan. Lower Duncan River Fish Stranding Impact Monitoring – DDMMON -16 Terms of Reference, December 2008.

Bauersfeld, K. 1978. Stranding of juvenile salmon by flow reductions at Mayfield Dam on the Cowlitz River, 1976. Washington Department of Fisheries.

Berland G, Nickelsen T, Heggenes J, Okland F, Thorstad EB and J Halleraker. 2004. Movements of wild Atlantic salmon parr in relation to peaking flows below a hydropower station. River Research and Applications. 20:957-966.

Flodmark LEW. 2004. Hydropeaking—a potential threat or just a nuisance? Experiments with daily discharge fluctuations and their effects on juvenile salmonids. Ph.D. Thesis, Faculty of Mathematics and Natural Sciences, University of Oslo.

Golder Associates Ltd. 2002. Fish and Aquatic Habitat Resources in the Lower Duncan River – 1998-1999 Investigations. Prepared for BC Hydro, Castlegar, BC. Golder Report No. 741D.

Golder Associates Ltd. 2006. Duncan River Fish Stranding Summary (November 2002 to March 2006). Prepared for BC Hydro, Castlegar, BC. Golder Report No. 07-1480-0038.

Golder Associates Ltd. 2008. Duncan River Fish Stranding Summary (April 2006 to January 2008). Prepared for BC Hydro, Castlegar, BC. Golder Report No. 05-1480-0051.

Golder Associates Ltd. 2009a. Gap Analysis for Lower Duncan River Ramping Program, DDMMON -1. Memo Report Prepared for: BC Hydro Columbia Generation Area, 601 18th Street, Castlegar, BC, V1N 2N1. Golder Report No. 07-1480-0062.

Golder Associates Ltd. 2009b. DDMMON -16 Lower Duncan River fish stranding impact monitoring: Year 1 Summary report (February 2008 to April 2009). Report prepared for BC Hydro, Castlegar, B.C. Golder Report No. 09-1480-0007F: 13 p. + 1 app.

Golder Associates Ltd. 2010. DDMMON -16 Lower Duncan River fish stranding impact monitoring: Year 2 summary report (April 2009 to April 2010). Report prepared for BC Hydro, Castlegar, BC. Golder Report No. 09-1480-0007F: 32 p. + 4 app.

Golder Associates Ltd. 2011. DDMMON-16 Lower Duncan River fish stranding impact monitoring: Year 3 summary report (April 2010 to April 2011). Report prepared for BC Hydro, Castlegar, BC. Golder Report No. 10-1492-0110F: 27 p. + 3 app.

Golder Associates Ltd. 2012. Adaptive Stranding Protocol for Managing Fish Impacts in the Lower Duncan River Associated with Flow Reductions at Duncan Dam. Document prepared for BC Hydro, Castlegar, BC. Golder Document No. 09-1492-5010F: 32 p. + 6 app.

Golder Associates Ltd. 2014. DDMMON-16 Lower Duncan River fish stranding impact monitoring: Year 5 data report (April 2012 to April 2013). Report prepared for BC Hydro, Castlegar, BC. Golder Report No. 12-1492-0117F: 25 p. + 3 app.

Golder Associates Ltd. 2015. DDMMON-16 Lower Duncan River fish stranding impact monitoring: Year 6 report (April 2013 to April 2014). Report prepared for BC Hydro, Castlegar, BC. Golder Report No. 12-1492-0117F: 49 p. + 3 app.

Golder Associates Ltd. 2017. DDMMON-16 Lower Duncan River Fish Stranding Impact Monitoring: Year 7 Report (April 2014 to April 2015). Report prepared for BC Hydro, Castlegar, BC. Golder Report No. 12-1492-0117: 59 p. + 3 app.

Golder Associates Ltd. and Poisson Consulting Ltd. 2012. DDMMON-16 Lower Duncan River fish stranding impact monitoring: Year 4 summary report (April 2011 to January 2012). Report prepared for BC Hydro, Castlegar, BC. Golder Report No. 10-1492-0110D: 39 p. + 2 app.

Halleraker, J.H., Saltveit, S.J., Harby, A., Arnekleiv, J.V., Fjedstad, H.P., and B. Kohler. 2003. Factors influencing stranding of wild juvenile brown trout (*Salmo trutta*) during rapid and frequent flow decreases in an artificial stream. River Research and Applications 19: 589-603.

Higgins, P. S. and Bradford, M.J. 1996. Evaluation of a large-scale fish salvage to reduce the impacts of controlled flow reduction in a regulated river. *North American Journal of Fisheries Management* **16**: 666-673.

Irvine, R. and Schmidt, D. 2009. Gap Analysis for Lower Duncan River Ramping Program DDMMON -1. *Memo Report Prepared for: BC Hydro Columbia Generation Area*. Castlegar.

Kery M. and Schaub, M. 2011. Bayesian Population Analysis using WinBUGS: A hierarchical perspective. Academic Press, NY.

Kincaid, T.M. and Olsen, A.R. 2013. spsurvey: Spatial Survey Design and Analysis. R package version 2.6. URL: http://www.epa.gov/nheerl/arm/

Plummer, M. 2013. rjags: Bayesian graphical models using MCMC. R package version 3-10. http://CRAN.R-project.org/package=rjags

Poisson Consulting Ltd. 2011. DDMMON-16: Lower Duncan River Fish Stranding Impact Monitoring. Memo Report Prepared for Golder Associates Ltd, Castlegar, BC.

Poisson Consulting Ltd. and Golder Associates Ltd. 2010. DDMMON-1 - Lower Duncan River ramping rate monitoring: Phase V investigations. Report prepared for BC Hydro, Castlegar, BC. Golder Report No. 09-1492-5008F: 41 p. + 6 app.

R Development Core Team. 2015. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org/.

Saltveit, S.J., Halleraker, J.H., Arnekleiv, J.V., and A Harby. 2001. Field experiments on stranding in juvenile Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) during rapid flow decreases caused by hydro peaking. Regulated Rivers: Research and Management 17: 609-622.

Thorley, J., L. Porto, J. Baxter and J. Hagen. 2011. Year 2 Data Report DDMMON-2: Lower Duncan River Habitat Use Monitoring. *Duncan Dam Project Water Use Plan - Lower Duncan River Habitat Use Monitoring*. BC Hydro. Castlegar, BC. Poisson Consulting Ltd., AMEC Earth & Environmental and J. Hagen and Associates.

Thorley, J.L., R.L. Irvine, J.T.A. Baxter, L. Porto, C. Lawrence. 2012. Lower Duncan River Habitat Use (DDMMON-2). Year 3 Final Report. Report Prepared for: BC Hydro, Castlegar. Prepared by: AMEC Environment & Infrastructure Ltd., Poisson Consulting Ltd., and Mountain Water Research. 86 pp + 2 app.

Thorley, J.L. 2014. jaggernaut: An R package to facilitate Bayesian analyses using JAGS (Just Another Gibbs Sampler). https://github.com/joethorley/jaggernaut.

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.

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Fisheries Biologist, Project Manager

Dr. Dana Schmidt
Senior Fisheries Biologist/Limnologist

Darryl Arsenault
Associate, Senior Reviewer

BH/DS/DJA/cmc

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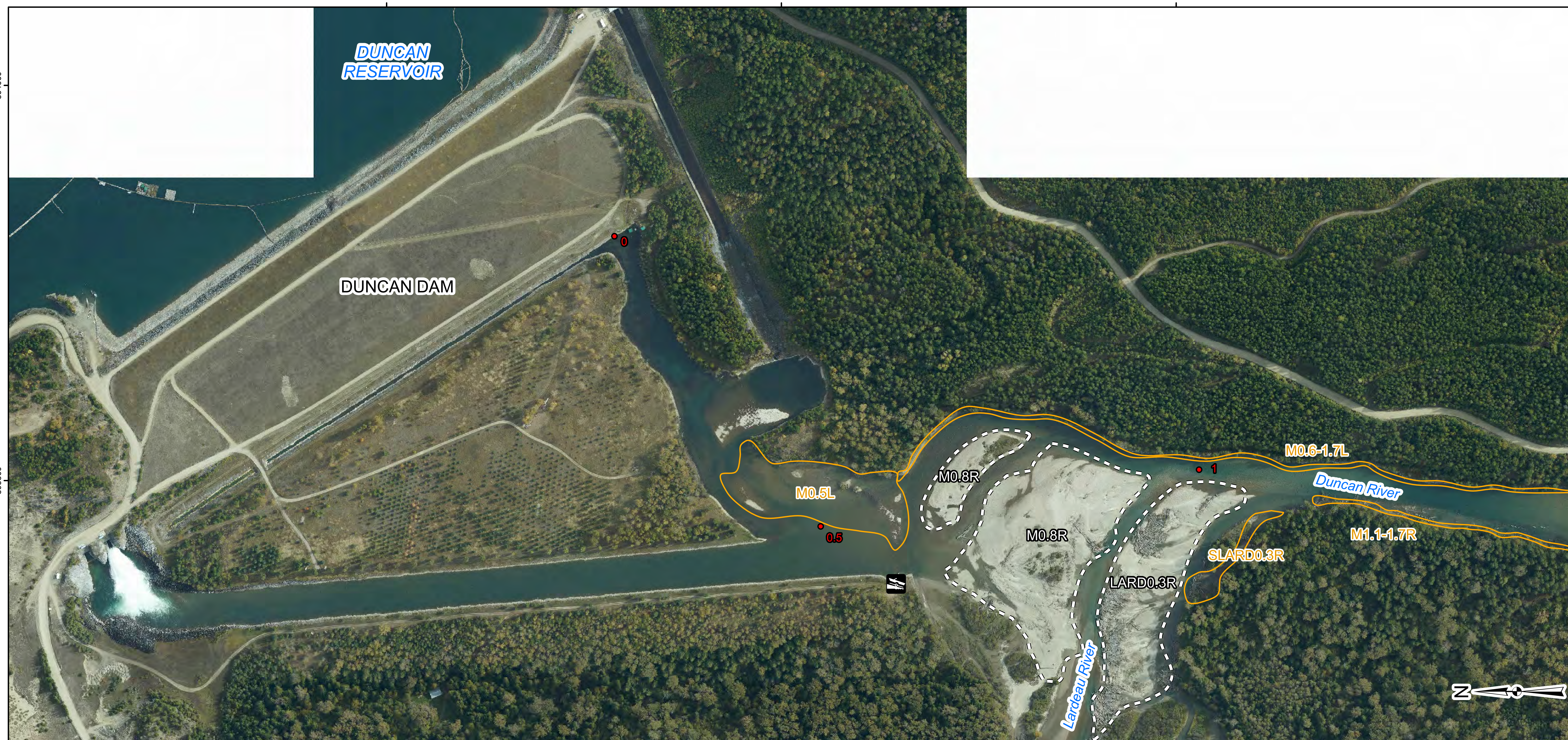
APPENDIX A

Sampling Chronology and Summary of Identified Stranding Sites



LEGEND

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

REFERENCE

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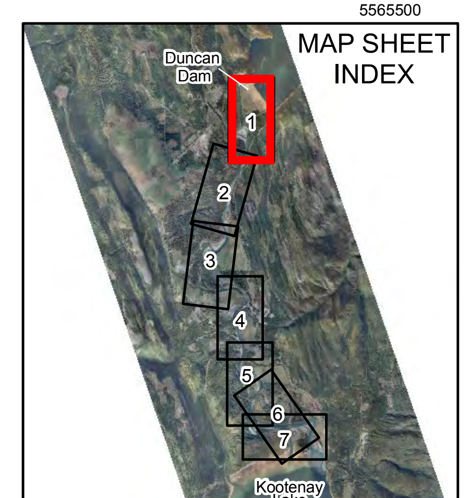

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TITLE	**LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES**

Golder Associates

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

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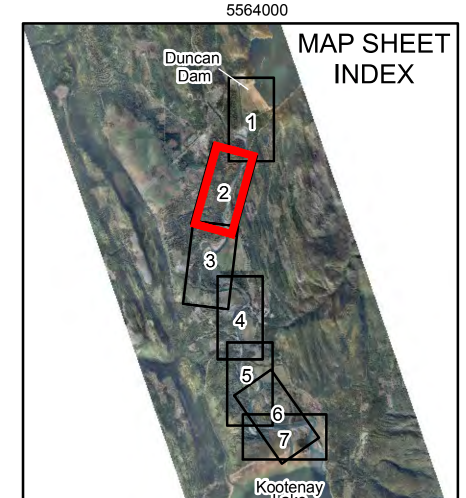

LEGEND

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

TITLE
LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

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REVIEW	GB	31 Jan. 2014		

LEGEND

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

REFERENCE

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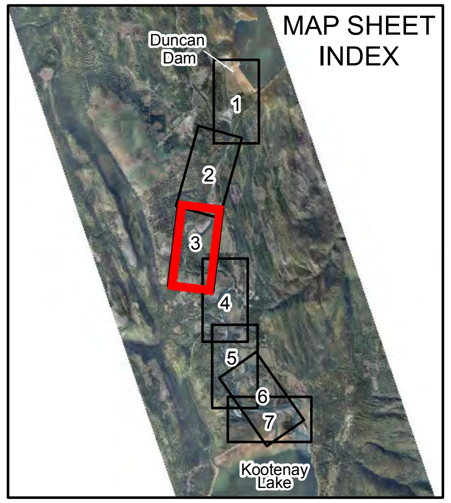

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PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

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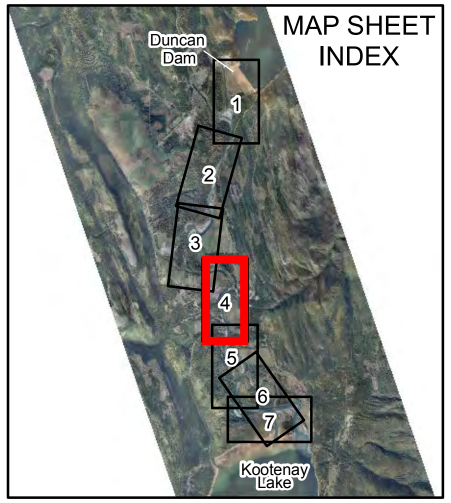

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LEGEND

- 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

PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

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PROJECT No. 12-1492-0117			SCALE AS SHOWN	REV. 0
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LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

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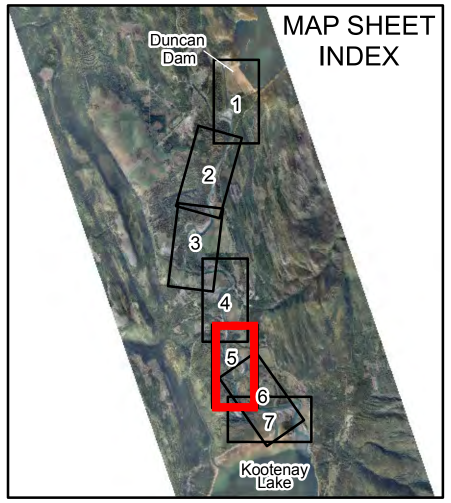

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PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

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

Golder Associates

FIGURE 5

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LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

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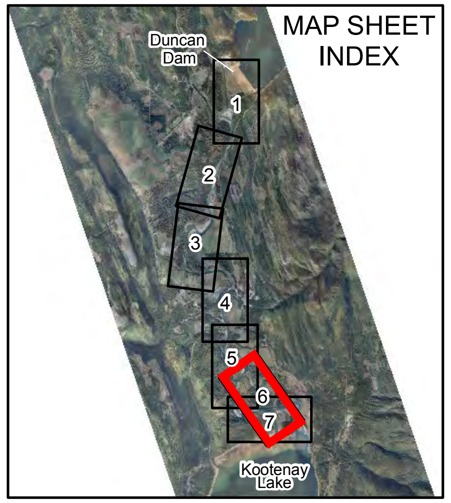

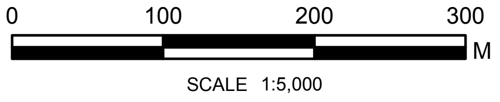

PROJECT
DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE
LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

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LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

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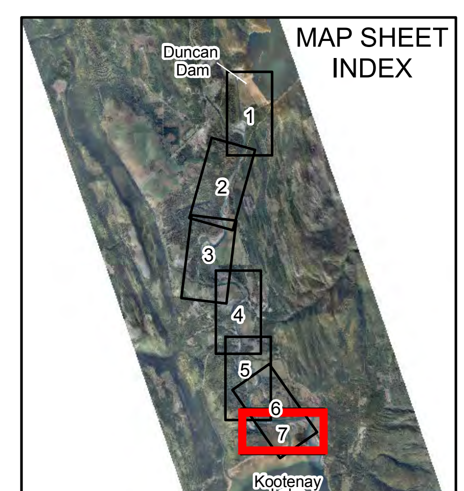

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PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

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CHECK	BH	31 Jan. 2014		
REVIEW	GB	31 Jan. 2014		

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
April 11, 2008	Stranding Assessments	RE2008-02	-	5	-
April 15, 2008	Stranding Assessments	RE2008-03	-	5	-
April 28, 2008	Stranding Assessments	RE2008-04	-	6	-
July 22, 2008	Stranding Assessments	RE2008-05	-	6	-
August 26, 2008	Stranding Assessments	RE2008-06	-	6	-
September 25, 2008	Stranding Assessments	RE2008-07	-	6	-
September 28, 2008	Stranding Assessments	RE2008-08	-	5	-
October 1, 2008	Stranding Assessments	RE2008-09	-	6	-
February 28, 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
April 25, 2009	Stranding Assessments	RE2009-02	-	6	-
September 25, 2009	Stranding Assessments	RE2009-03	-	6	-
September 28, 2009	Stranding and Calibration Assessments	RE2009-04	-	7	13
October 1, 2009	Stranding Assessments	RE2009-05	-	5	-
January 22, 2010	Stranding Assessments	RE2010-01	-	5	-
March 1, 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
August 27, 2010	Stranding Assessments	RE2010-03	-	7	1
September 25, 2010	Stranding Assessments	RE2010-04	-	7	3
September 28, 2010	Stranding Assessments	RE2010-05	-	11	3
October 1, 2010	Stranding Assessments	RE2010-06	-	10	13
March 1, 2011	Stranding Assessments	RE2011-01	-	7	-
March 2, 2011	Stranding Assessments	RE2011-02	-	4	-
April 12, 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
April 19, 2011	Stranding Assessments	RE2011-04	-	5	0
June 1, 2011	Stranding Assessments – start of random selection process for sample sites	RE2011-05	-	12	2
August 25, 2011	Stranding Assessments	RE2011-06	-	6	4
September 25, 2011	Stranding Assessments	RE2011-07	-	1	4
September 28, 2011	Stranding Assessments	RE2011-08	-	2	2
October 1, 2011	Stranding Assessments	RE2011-09	-	2	3
January 20, 2012	Stranding Assessments	RE2012-01	-	3	4

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
April 15, 2012	Stranding Assessments	RE2012-03	-	2	0
June 1, 2012	Stranding Assessments	RE2012-04	-	Assessment was planned, but cancelled by BC Hydro prior to reduction date	
September 26, 2102	Stranding Assessments	RE2012-05	-	5	4
September 27, 2012	Stranding Assessments	RE2012-06	-	3	2
October 1, 2012	Stranding Assessments	RE2012-07	-	3	3
January 21, 2013	Stranding Assessments	RE2013-01	-	6	5
March 1, 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
September 14 and 15, 2013	Abundance Estimation	-	Study Area Reconnaissance and Site Selection		
September 16, 2013	Abundance Estimation	-	5	-	-
September 17, 2013	Abundance Estimation	-	7	-	-
September 18, 2013	Abundance Estimation	-	10	-	-
September 19, 2013	Abundance Estimation	-	12	-	-
September 21, 2013	Stranding Assessments	RE2013-03	-	3	4
September 24, 2013	Stranding Assessments	RE2013-04	-	2	2
September 27, 2013	Stranding Assessments	RE2013-05	-	2	4
January 21, 2014	Stranding Assessments	RE2014-01	-	4	4
March 1, 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
May 22, 2014	Stranding Assessments	RE2014-03	-	5	4
September 18 and 19, 2014	Abundance Estimation	-	Study Area Reconnaissance and Site Selection		
September 20, 2014	Abundance Estimation	-	14	-	-
September 21, 2014	Abundance Estimation	-	16	-	-
September 22, 2014	Abundance Estimation	-	10	-	-
September 23, 2014	Abundance Estimation	-	9	-	-
September 25, 2014	Stranding Assessments	RE2014-04	-	3	5
September 28, 2014	Stranding Assessments	RE2014-05	-	4	3
October 1, 2014	Stranding Assessments	RE2014-06	-	3	0

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

Plate 2 Medium to High complexity pool at site S3.5-4.0R, September 28, 2014.

Plate 3   Mainstem habitat at site M7.7L on September 28, 2014.

Plate 4   Side channel habitat at site S4.0-4.2R on May 22, 2014.

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