Duncan Dam Project Water Use Plan

Adaptive Stranding Protocol Development Program

Implementation Year 7

Reference: DDMMON-16

Lower Duncan River: Fish Stranding Impact Monitoring: Year 7 Data Report

Study Period: April 2014 to April 2015

Golder Associates Ltd.
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DDMMON-16: LOWER DUNCAN RIVER

Lower Duncan River Fish Stranding Impact Monitoring: Year 7 Report (April 2014 to April 2015)

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Cover Photo: Downstream view of Site S8.2R with Kokanee spawners, September 19, 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 on 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 2015, 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 7 of the DDMMON-16 program, and the current status of management questions for DDMMON-16 is provided in the table below. Because of the high degree of variation in stranding rates, the uncertainty of the interstitial stranding estimates, and the many variables that could potentially contribute to stranding, these results should be treated as preliminary as they are somewhat sensitive to assumptions.

Table EI: DDMMON-16 Year 7: Status of Management Questions and Objectives.

DDMMON-16 Management Question	DDMMON-16 Specific Hypothesis	DDMMON-16 Year 7 (2014-2015) 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 7 (2014-2015) 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) 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. <u>Therefore, based on these analyses, hypothesis *Ho_1* cannot be rejected at this time but based on the initial study design, this hypothesis will likely be rejected in the future</u>. - Future study years should consider that stranding observed at both index sites and random sites as representative of overall stranding. 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 3 to 4 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 were not found. - <u>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.</u>

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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 Duncan River 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 Duncan River can be exacerbated by 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 Duncan River 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 at the Duncan River below the Lardeau River Water Survey of Canada (WSC) discharge monitoring 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. 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: the Lower Duncan River Fish Stranding Impact Monitoring Program (FSIMP). In conjunction with other assessment tools being developed during the monitoring period, the FSIMP assesses 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 this year (Year 7) 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 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 all other mortality mechanisms and 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 2009). This uncertainty should be considered when interpreting the results of this program.

As outlines in the Terms of Reference (BC Hydro 2008) the species of interest for this program are Rainbow Trout (*Oncorhynchus mykiss*) and Mountain Whitefish (*Prosopium williamsoni*). The following document provides information on abundance estimation and fish stranding observed for these species, over all assessed flow reductions from the timing of the last report on April 22, 2014 (Golder 2015) to April 14, 2015 (Year 7). This report also presents detailed statistical analysis in relation to the multi-year program objectives, and also 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) and the present study year (Year 7: 2014 – 2015) 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.

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. 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.

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 to improve sampling coverage while addressing logistic difficulties. The goal during Year 7 sampling was to increase the number of sites sampled in comparison to Year 6, which is anticipated to reduce uncertainty associated with abundance estimation. To sample a longer length of river in comparison to previous studies and years (5700 m, 3600 m, and 4000 m in 2010, 2012 and 2013, respectively), 60 sites were selected, of varying length. In Year 6 abundance estimation sampling, it was determined that the majority of sites were too narrow for the three snorkelers that conducted the sampling. Therefore in Year 7, the snorkelers were reduced from three to two, which reduced effort expended at each site sampled and allowed a higher number of sites to be sampled within the project budget.

1.4.5 Data Analysis

The modelling used in Year 6 (Golder 2015) of this program was updated to incorporate the current years' data set, and to analyze season as a variable related to stranding rates. 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 8 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 mouths of Meadow, Hamill and Cooper creeks. For the purpose of all WLR studies, the mainstem Duncan River was divided into five sections; these were termed Reach 1 (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).

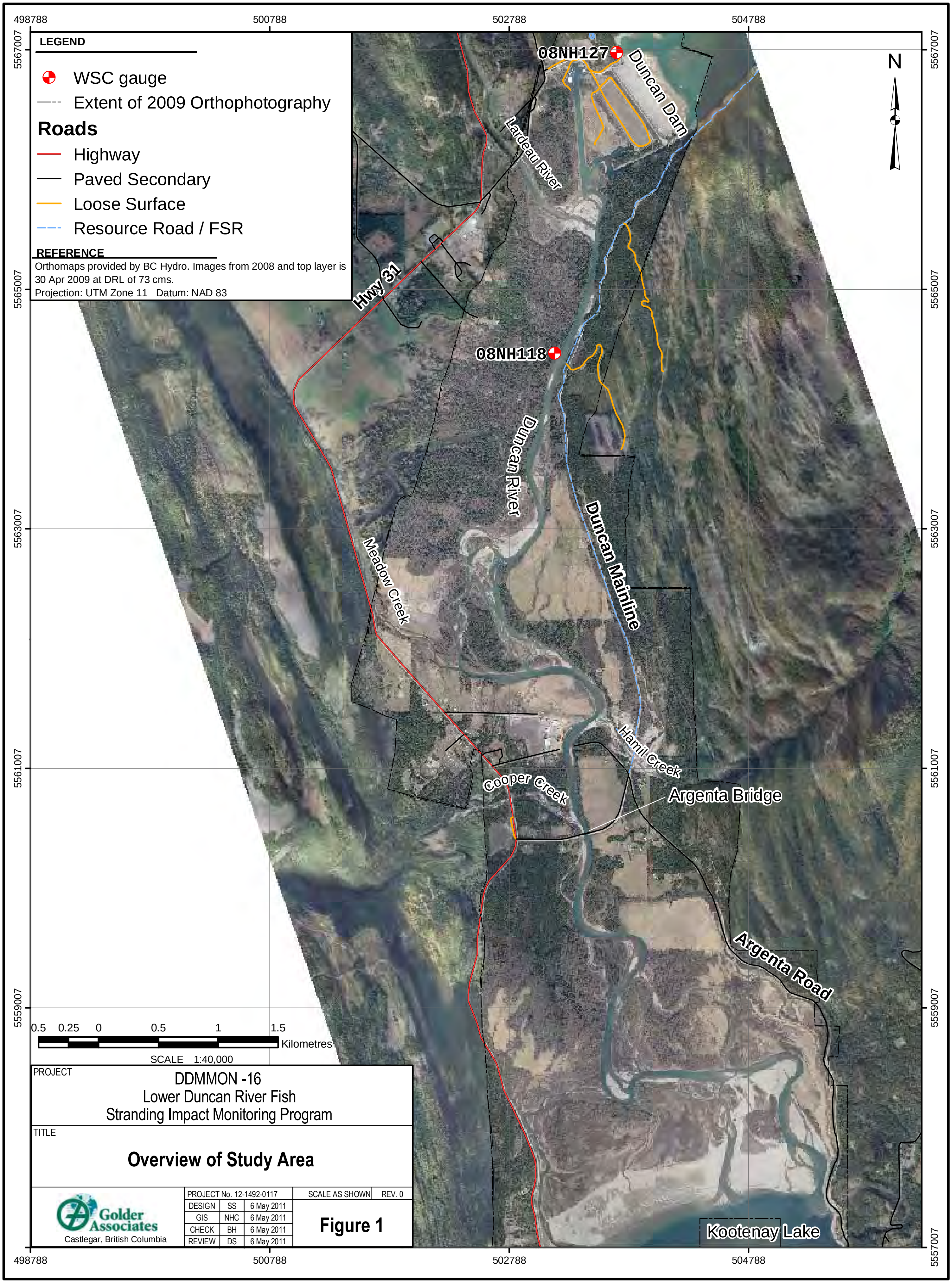
For the purpose of this study, 50 potential fish stranding sites were identified based on previous studies (AMEC 2004; Golder 2006, 2008, 2009b, 2010, 2011, 2014; Golder and Poisson 2012). These stranding sites included 11 index stranding assessment sites and 39 non-index sites (Appendix A, Figures 1 to 7). The remaining habitats outside of the identified sites consist of steep banks with extreme gradient that would not be considered to strand fish.

2.2 Study Period

In Year 3 (2010 – 2011), the study period was set between April 15, 2010, and continued until the following April 14, 2011. Stranding assessment activities in the present study year were conducted from May 22, 2014 to October 1, 2014, during planned flow reductions at DDM. Each assessed reduction from DDM was assigned a reduction event number (RE; see Section 2.5) and Table 1 outlines all assessment activities during Year 7. An in-depth summary of the chronology of sampling and project milestones in all study years is provided in Appendix A, Tables A1 to A7.

Table 1: 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



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 from the Duncan River below Lardeau River Water Survey of Canada gauging station (DRL) located downstream of the Duncan-Lardeau confluence at River Km (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 Fish Abundance Assessment

2.4.1 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); and,
- 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.

2.4.2 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 shallower depths. 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.

2.4.3 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. The analysis was implemented using the statistical environment R, v. 3.1.0 (R Development Core Team 2014), interfaced with JAGS v. 3.3.0 (Plummer 2014) through the rjags package (Plummer 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
dgamma(κ, θ)	Gamma distribution with a shape parameter κ and a rate parameter θ
log(x)	Natural logarithm function
logit(x)	Logit function

In the Bayesian implementation of the model, fish counts were assumed to be Poisson-distributed with extra-Poisson variability, with a mean expected density drawn from a log-normal distribution (Table 3). Density was modeled using several models, as a function of: 1) intercept only; 2) depth (deep/shallow); 3) flow (fast/slack); 4) their interaction (stratum; shallow/fast, shallow/slack, deep/fast, and deep/slack); and 5) year (2013/2014), and was allowed to randomly vary by site in all models. Deviance Information Criterion (DIC) values could not be computed for all models in this report, since JAGS cannot compute DIC values for models with negative binomial or truncated distributions. Therefore, to maintain consistency, all models were compared using the significance of parameters, which was determined based on whether the parameters' 95% CRI region overlapped each other (i.e., whether strata were significantly different from each other) and whether the 95% CRI region overlapped zero (i.e., whether strata effects were significantly different from zero). 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 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 lower 95% credibility levels, median, and upper 95% credibility levels across all three sampled strata yielded the total abundance of fish within the LDR. The prior distributions for all parameters were vague or uninformative to avoid biasing estimates (Table 4). 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
LogitEfficiency[k]	Expected log odds of observer efficiency at the *k*-th site
p[k]	Observer efficiency at the *k*-th site
NSite	Number of sampled sites
bSite[k]	The random effect of the *k*-th site on fish density
bYear[yr]	The effect of the *yr*-th year on fish density
bStratum[i]	The effect of the *i*-th stratum on fish density
bDepth[i]	The effect of the *i*-th depth stratum on fish density
bFlow[i]	The effect of the *i*-th flow stratum on fish density
Area[k]	The area of the *k*-th site
mu [k]	Expected fish density at the *k*-th site
SiteNum[k]	Numeric representation of site name of the *k*-th site
StratumNum[k]	Numeric representation of stratum of the *k*-th site
DepthNum[k]	Numeric representation of the depth stratum of the *k*-th site
FlowNum[k]	Numeric representation of the flow stratum of the *k*-th site
N[k]	Number of fish at the *k*-th site
Count[k]	The total observed number of fish at the *k*-th site
NStrata13	The number of depth/flow strata in 2013
NStrata14	The number of depth/flow strata in 2014
mutotal13 [i]	Expected fish density at the *i*-th stratum in 2013
mutotal14 [i]	Expected fish density at the *i*-th stratum in 2014
CountTotal13[i]	Total fish abundance at the *i*-th stratum in 2013
CountTotal14[i]	Total fish abundance at the *i*-th stratum in 2014
P_out	Average observer efficiency
AreaStratum[i]	Area of the *i*-th stratum
Censor[k]	A variable to truncate abundances to be above observed counts at the *k*-th site

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

Variable/parameter	Description
sSite	dgamma(0.01, 0.01)
bSite[k]	dnorm(0, sSite^-2)
bStratum[i]	dnorm(0, 0.01)
bDepth[i]	dnorm(0, 0.01)
bFlow[i]	dnorm(0, 0.01)
bYear[i]	dnorm(0, 0.01)
bEfficiency	-0.53
sEfficiency	0.68
LogitEfficiency	dnorm(bEfficiency, sEfficiency^-2)

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

Variable/parameter	Dependency
logit(p[k])	LogitEfficiency
logit(p_out)	bEfficiency
log(mu[k])	bIntercept + bSite*SiteNum[k] (model 1)
log(mu[k])	bIntercept + bSite*SiteNum[k] + bDepth*DepthNum[k] (model 2)
log(mu[k])	bIntercept + bSite*SiteNum[k] + bFlow*FlowNum[k] (model 3)
log(mu[k])	bIntercept + bSite*SiteNum[k] + bStratum*StratumNum[k] (model 4)
log(mu[k])	bIntercept + bSite*SiteNum[k] + bYear*YearNum[k] (model 5)
Count[k]	dpois(mu[k] * area[k] * p[k])
log(mutotal13[i]); log(mutotal14[i])	bIntercept (model 1)
log(mutotal13[i]); log(mutotal14[i])	bIntercept + bDepth*DepthNum[k] (model 2)
log(mutotal13[i]); log(mutotal14[i])	bIntercept + bFlow*FlowNum[k] (model 3)
log(mutotal13[i]); log(mutotal14[i])	bIntercept + bStratum*StratumNum[k] (model 4)
log(mutotal13[i]); log(mutotal14[i])	bIntercept + bYear*YearNum[i] (model 5)
CountTotal13[i]	dpois(mutotal13[i]*AreaStratum13[i])
CountTotal14[i]	dpois(mutotal14[i]*AreaStratum14[i])

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

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

Median values of density and abundance and 95% credibility intervals were calculated in R. The Monte Carlo (MC) error for each parameter estimate was recorded. The MC error quantifies the variability in the estimates that is due the sampling error in the simulation-based solution for Bayesian analysis. We chose simulations run lengths so that MC error was <5% of the posterior standard deviation for a parameter (Kery 2010). The posterior distributions, which were estimated using Gibbs sampling (Kery 2010), were derived from 2,700 Markov Chain Monte Carlo (MCMC) simulations, and thinned from 90,000 runs of three MCMC chains of 10^5 iterations in length. Model convergence was confirmed by ensuring that R-hat (the Gelman-Rubin Brooks potential scale reduction factor) was less than 1.1 for each of parameters in the model (Kery 2010).

2.5 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 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 just 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 higher 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.

In 2010, DDMMON-15 reviewed all LDR aquatic study reports and provided recommendations on the data collection methodology used during fish stranding assessments. This led to the modification of the assessment methodology at the onset of Years 3 and 4 to improve the accuracy of fish stranding estimates, and to increase the amount of long-term data available for stranding impact analysis on the lower Duncan River (Golder 2011, Golder and Poisson 2012).

2.5.1 Year 7 Stranding Site Selection

Prior to each fish stranding assessment, 10 sites were randomly selected from all identified stranding sites. 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. The dewatered area at all sites was calculated using the site area regressions that were completed in Year 3 (Golder 2011).

2.5.2 Year 7 Sampling

2.5.2.1 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); and

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; and
- 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.

2.5.2.2 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 2.5.2.4). Unlike isolated pools, the habitat parameters described in Section 2.5.2.1 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.

2.5.2.3 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 is 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.

2.5.2.4 Fish Life History Data

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

- Species;
- Length (mm);
- Condition (alive or dead);
- Salvaged (yes/no); and
- 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.

2.5.3 Data Analysis

2.5.3.1 Dewatered Area

To compare pre- and post-WUP operations, the Year 7 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.

2.5.3.2 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 using the statistical environment R, v. 3.1.0 (R Development Core Team 2014), interfaced with JAGS v. 3.3.0 (Plummer 2014) through the rjags package (Plummer 2014).

For each parameter of interest, median values and their upper and lower 95% credibility intervals were calculated in R. We chose simulations run lengths so that MC error was <5% of the posterior standard deviation for a parameter (Kery 2010).

2.5.3.3 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 model defined the number of pools present at a site to be Poisson-distributed, with a mean expected value determined by dewatered site area and site-specific effect (Table 7). In addition, a separate model was constructed, where the mean expected value of pool numbers at a site was also influenced by whether the site was index or randomly assigned. 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. If the variable was determined to be not significant (i.e., 95% CRIs overlapped), the original model was used for further analysis without an index/random effect. 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. In the separate model that included the effect of index/random site on pool density, the total dewatered area of all index sites was estimated and multiplied by index-site pool density. All other dewatered area was considered random, and was multiplied by the random-site pool density.

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. The season variable was used to estimate the pool stranding of Rainbow Trout. A relationship between season and Mountain Whitefish pool stranding was not found, therefore Mountain Whitefish pool stranding was estimated using an intercept and a random reduction variable. 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 a mean expected value determined by season and a reduction-specific random effect (Table 7). To estimate total pool stranding, estimated pool abundance (Section 2.5.3.3) was multiplied by the number of estimated fish/pool.

The prior distributions for all parameters were vague or uninformative to avoid biasing estimates (Table 8). The complete model specification used is shown in Table 8 and Table 9, and model code is provided in Appendix B. The posterior distributions, which were estimated using Gibbs sampling (Kery 2010), were derived from 12,000 Markov Chain Monte Carlo (MCMC) simulations, and thinned from 40,000 runs of three MCMC chains of 5 x10^4 iterations in length. Model convergence was confirmed by ensuring that R-hat (the Gelman-Rubin Brooks potential scale reduction factor) was less than 1.1 for each of parameters in the model (Kery 2010).

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

Variable/parameter	Description
sSite	Standard deviation of the effect of site on expected number of pools
r	Extra-Poisson overdispersion of number of pools
bArea	The effect of dewatered area on pool numbers
SiteArea[i]	The dewatered area at the *i*-th case
bIntercept	Log pool density
bIndex_site[i]	The effect of whether the *i*-th site was index or random on pool density
bSiteName[j]	The random effect of the *j*-th site on pool numbers
nObs_pool	Number of pool count data points
u[i]	Effect of extra-Poisson overdispersion on number of pools at the *i*-th case
mu_pool[i]	Expected number of pools at the *i*-th case
NumPoolsPresent[i]	Observed number of pools at the *i*-th case
bIntercept_fish	Log fish counts (for MW only)
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 (for RB only)
muEff[j]; muEff.MW[j]	Logistic catchability using the *j*-th sampling gear, where j = 1 for visual and dip-net, and j = 2 for electrofishing; separate values for RB and MW

Variable/parameter	Description
p[j]; p_MW[j]	Catchability using the *j*-th sampling gear, where j = 1 for visual and dip-net, and j = 2 for electrofishing; separate values for RB and MW
sReduction, sReduction.MW	Standard deviation of the effect of reduction on expected fish counts per pool; separate values for RB and MW
bReduction[r]; bReduction.MW[r]	The random effect of the *r*-th reduction on expected fish counts per pool; separate values for RB and MW
NObs; Nobs.MW	Number of pool stranding data points; separate values for RB and MW
mu[k]; mu.MW[k]	Expected species-specific fish counts in the *k*-th pool
SeasonNum[k]	Season during which the *k*-th pool was sampled (for RB only)
ReductionNum[k]	Reduction during which the *k*-th pool was sampled
censor[k]; censor.MW[k]	A variable used to truncate fish counts for the *k*-th pool, so that estimated fish numbers are not lower than those observed; separate values for RB and MW
N[k]; N.MW[k]	Estimated fish counts at the *k*-th pool; separate values for RB and MW
MinFish[k]; MinFish.MW[k]	Observed number of fish at the *k*-th pool; separate values for RB and MW
NPass[k, p]; NPass.MW[k, p]	Estimated number of fish present at the *k*-th pool prior to the *p*-th pass; separate values for RB and MW
Pass[k, p]; Pass.MW[k, p]	Sampled number of fish at the *k*-th pool prior to the *p*-th pass; separate values for RB and MW
SamplingGearNum[g]; SamplingGearNum.MW[g]	Sampling gear used at the *k*-th pool, where g = 1 stands for visual or dip-net, and g = 1 stands for electrofishing; separate values for RB and MW
nReductions	Number of reductions
muReduction[r]; muReduction.MW[r]	Expected number of fish stranded per pool at the *r*-th reduction; separate values for RB and MW
High[r]	Exposed high-slope (>4%) area at the *r*-th reduction (where no bIndex_site was used)
Low[r]	Exposed low-slope (0-4%) area at the *r*-th reduction (where no bIndex_site was used)
High_index[r], High_random[r]	Exposed high-slope (>4%) area at the *r*-th reduction, separated by index and random areas
Low _index[r], Low _random[r]	Exposed low-slope (0-4%) area at the *r*-th reduction, separated by index and random areas
HighSlope	Slope of linear model between high-slope exposed area and DRL discharge
LowSlope	Slope of linear model between low-slope exposed area and DRL discharge
Total_pools[r]	Estimated number of pools formed at the *r*-th reduction (where no bIndex_site was used)
Total_pools_index[r]; Total_pools_random[r]	Estimated number of pools formed at the *r*-th reduction, separated by index and random areas
Total[r]; Total.MW[r]	Estimated number of fish stranded in pools at the *r*-th reduction (where no bIndex_site was used; separate values for RB and MW)
Total_index[r]; Total.MW_index[r]; Total_random[r]; Total.MAW_random[r]	Estimated number of fish stranded in pools at the *r*-th reduction, separated index and random areas; separate values for RB and MW

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

Variable/parameter	Description
sSite	dgamma(0.1,0.1)
r	dgamma(0.1,0.1)
bArea	dnorm(0, 1)
bIntercept	dnorm(0, 0.01)
bIndex_site	dnorm(0, 0.01)
bSiteName[j]	dnorm(0, sSite$^{\wedge -2}$)
u[i]	dgamma(r, r)
bSeason[i]	dnorm(0, 0.01)
bIntercept_fish	dnorm(0, 0.01)
muEff[2]; muEff.MW[2]	dnorm(0, 0.01)
sReduction; sReduction.MW	dunif(0, 5)

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

Variable/parameter	Dependency
mu[i]	bIntercept + bSiteName[SiteName[i]] + bArea*SiteArea[i] (where no bIndex_site was used)
mu[i]	bIndex_site[Index_site[i]] + bSiteName[SiteName[i]] + bArea*SiteArea[i] (where bIndex_site was used)(where no bIndex_site was used)
NumPoolsPresent[i];	dpois(mu[i]*u[i])
muEff[1]; muEff.MW[1]	10
bReduction[r]; bReduction.MW[r]	dnorm(0, sReduction^-2); dnorm(0, sReduction.MW^-2)
log(mu[k]); log(mu.MW[k])	bSeason[SeasonNum[k]] + bReduction[ReductionNum[k]]; bIntercept_fish + bReduction.MW[ReductionNum.MW[k]]
censor[k]; censor.MW[k]	dinterval(N[k], MinFish[k]); dinterval(N.MW[k], MinFish.MW [k])
N[k]; N.MW[k]	dpois(mu[k]); dpois(mu.MW[k])
NPass[k, 1]; NPass.MW[k, 1]	N[k]; N.MW[k]
Pass[k, p]; Pass.MW[k, p]	dbin(p[SamplingGearNum[k]], NPass[k, p]); dbin(p.MW[SamplingGearNum.MW[k]], NPass.MW[k, p])
NPass[i, pass+1]; NPass.MW [i, pass+1]	NPass[k, p] - Pass[k, p]; NPass.MW [k, p] - Pass.MW [k, p]
log(muReduction[r]); log(muReduction[r])	bSeason[ReductionSeasonNum[r]] + bReduction[Reduction_ReductionNum[r]]; bIntercept_fish + bReduction.MW [Reduction_ReductionNum[r]]
High[r]	bIntercept + bArea*(HighSlope*Drop[r])
Low[r]	bIntercept + bArea*(LowSlope*Drop[r])

Variable/parameter	Dependency
High_index[r]; High_random[r]	bIntercept + bArea*(IndexArea[r]/1000); bIntercept + bArea*((HighSlope*Drop[i]/1000) - IndexArea[i]/1000)
Low_index[r]; Low_random[r]	bIntercept + bArea*(IndexArea[r]/1000) ; bIntercept + bArea*((LowSlope*Drop[i]/1000) - IndexArea[i]/1000)
Total_pools[r]	High[r] + Low[r]
Total_pools_index[r]; Total_pools_random[r]	High_index[r] + Low_index[r]; High_random[r] + Low_random[r]
Total[r]; Total.MW[r]	muReduction[r]*Total_pools[r]; muReduction.MW[r]*Total_pools[r]
Total_index[r]; Total_index.MW[r]	muReduction[r]*Total_pools_index[r]; muReduction.MW[r]*Total_pools_index [r]
Total_random[r]; Total_random.MW[r]	muReduction[r]*Total_pools_random [r]; muReduction.MW[r]*Total_pools_random [r]

2.5.3.4 Interstitial Stranding

In the Bayesian model of interstitial stranding, the number of fish stranded in each sampled location was defined as Poisson-distributed, with a mean expected value determined by fish abundance and the probability of stranding at each sampled location (Table 10). To estimate total interstitial stranding, mean expected fish numbers were multiplied by total exposed area using GIS-derived low-slope (0-4%) and high-slope (>4%) dewatered areas at each stranding event. The extrapolation of the total interstitial stranding estimate also accounted for the total area of pools forming in the exposed area. The lower, median, and upper estimates of pool presence (Section 2.5.3.3) were multiplied by average pool size (19.6 m^2) calculated using data collected since September 2006.

The effects of ramping rate and index/random site on interstitial stranding were examined using separate models, where both variables influenced the probability of stranding. If the variable was determined to be not significant (i.e., 95% CRIs overlapped), the original model, with no index/random effect was used for further analysis.

Rainbow Trout and Mountain Whitefish interstitial stranding were analyzed separately. The prior distributions for all parameters were vague or uninformative to avoid biasing estimates (Table 11). The posterior distributions, which were estimated using Gibbs sampling (Kery 2010), were derived from 4,500 Markov Chain Monte Carlo (MCMC) simulations and thinned from 15,000 runs of three MCMC chains of $2*10^4$ iterations in length. Model convergence was confirmed by ensuring that R-hat (the Gelman-Rubin Brooks potential scale reduction factor) was less than 1.1 for each of parameters in the model (Kery 2010). The complete model specification used is shown in Table 11 and Table 12, and model code is provided in Appendix B.

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

Variable/parameter	Description
bIntercept	Expected number of fish in the *i*-th 0.5 m^2 grid
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
pRamp[i]	The effect of ramping rate in the *i*-th case on probability of stranding
IndexNum[i]	Whether the *i*-th site is index or random
Area[i]	The area of the *i*-th sampling area (whether grid or transect)
ReductRamp[r]	Ramping rate of the *r*-th reduction
nObs	Number of data points
mu.d[i]	Expected fish density in the *i*-th sampling area
mu.p[i]	Stranding probability in the *i*-th sampling area
p[i]	Whether a fish was stranded in the *i*-th sampling area
Fish[i]	Number of fish observed in the *i*-th sampling area

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

Variable/parameter	Description
pIntercept	dnorm(0, 0.001)
bIntercept	dnorm(0, 0.001)
bIndex[i]	dnorm(0, 0.001)
pRamp	dnorm(0, 0.001)

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

Variable/parameter	Dependency
log(mu.d[i])	bIntercept
logit(mu.p[i])	pIntercept (in model with no index/random or ramping effects)
logit(mu.p[i])	pIntercept + pRamp*ReductRamp[r] (in model with ramping effect)
logit(mu.p[i])	bIndex[1] for index sites or bIndex[2] for random sites (in model with index/random effect)
p[i]	dbern(mu.p[i])
Fish[i]	dpois(mu.d[i]*p[i]*Area[i])

2.6 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 the present study year. Presently, 72 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 73.6 m^3/s on October 3 and 15, 2014 to 314.75 m^3/s on May 18, 2014. Hourly discharge from DDM ranged from 0 m^3/s on several days between early June and early July 2014, to 255.934 m^3/s on January 31, 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.

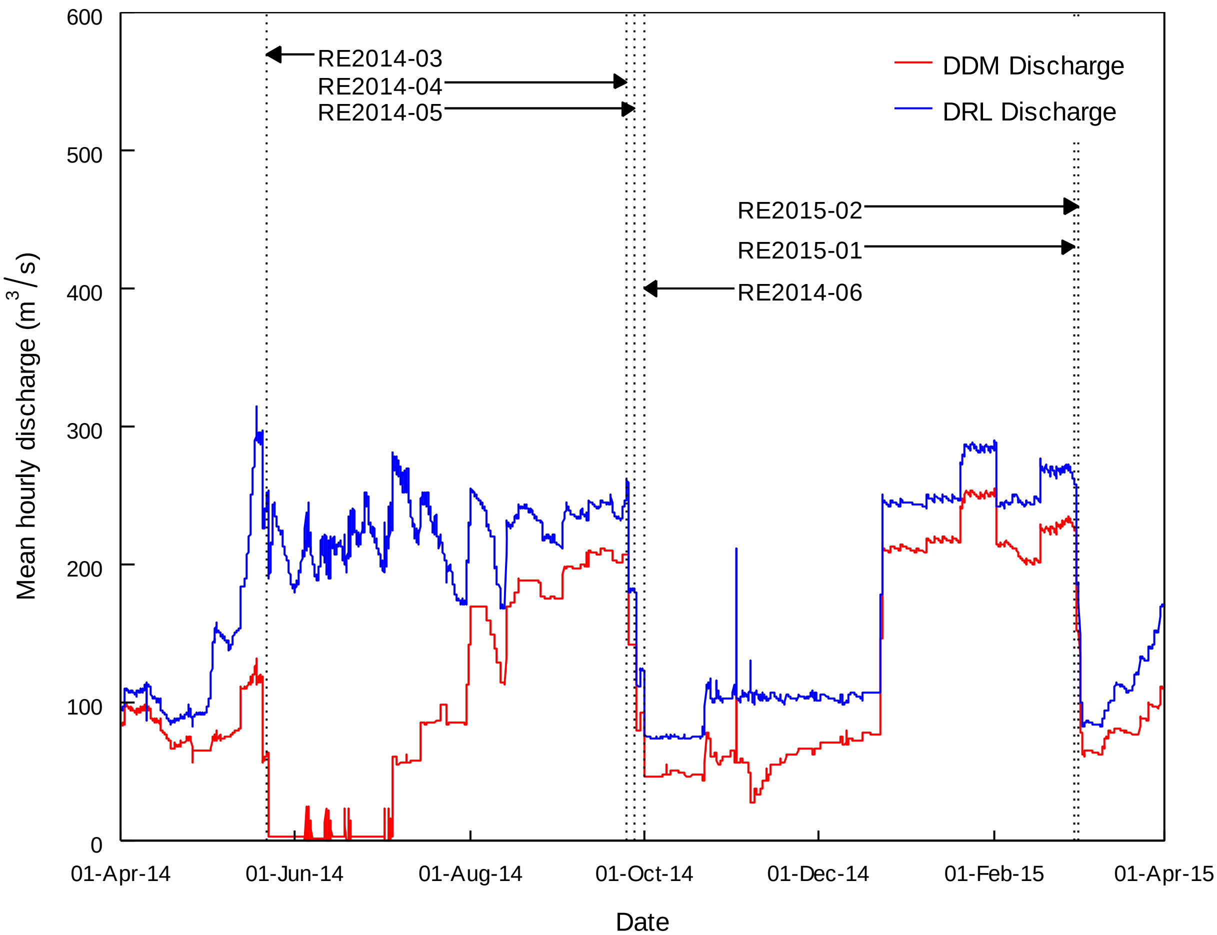


Figure 2: Hourly discharge at the Duncan Dam (DDM, red line) and at the Duncan River below the Lardeau River (DRL, blue line) from April 1, 2014 to April 1, 2015. Vertical dotted lines represent the timing of fish stranding assessments.

During the present study, six reduction events occurred at DDM (Figure 2 and Table 13). During the reduction events, DDM decreases of discharge ranged between 46 and 77 m^3/s (Table 13). These decreases represent the discharge reductions at DDM, rather than flow changes at particular downstream fish stranding sites.

Table 13: Summary of DDM flow reduction events, from May 22, 2014 to March 2, 2015, 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		
May 22, 2014	RE2014-03	60	3	57	Down 7 m^3/s every 15 minutes from 06:00 to 07:45.	Discharge reduced to meet flow target at DRL.
Sep 25, 2014	RE2014-04	208	142	66	Down 7 m^3/s every 15 minutes from 06:00 to 08:00, down 3 m^3/s at 08:15.	Onset of Kokanee protection flows.
Sep 28, 2014	RE2014-05	142	80	62	Down 7 m^3/s every 15 minutes from 06:00 to 07:45.	Kokanee protection flows.
Oct 01, 2014	RE2014-06	92	46	46	Down 7 m^3/s every 15 minutes from 06:00 to 07:15, down 4 m^3/s at 07:30.	Final transition to Kokanee protection flows.
Mar 01, 2015	RE2015-01	224	154	70	Down 7 m^3/s every 15 minutes from 06:00 to 08:15.	Discharge reduced to meet flow target at DRL.
Mar 02, 2015	RE2015-02	154	77	77	Down 7 m^3/s every 15 minutes from 06:00 to 08:30.	Discharge reduced to meet flow target at DRL.

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

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 the present study. All fish encountered during the assessments have been split into sportfish and non-sportfish categories for analysis. The scientific names of all species in these categories are presented in Table 14.

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

Category	Species	Scientific Name	Species Code[a]
Sportfish	Rainbow Trout	*Oncorhynchus mykiss*	*RB*
	Bull Trout	*Salvelinus confluentus*	*BT*
	Mountain Whitefish	*Prosopium williamsoni*	*MW*
	Pygmy Whitefish	*Prosopium coulteri*	*PW*
	Kokanee	*Oncorhynchus nerka*	*KO*
	Burbot	*Lota lota*	*BB*
Non-sportfish	Longnose Dace	*Rhinichthys cataractae*	*LNC*
	Dace spp.	*Rhinichthys 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 15). 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 7 (2012 – 2013 to 2014 – 2015). The number of pools sampled in the present year was also reduced to allow for more intensive interstitial sampling effort. This resulted in the fourth lowest number of pools sampled to date (n = 98), and the fourth most number of interstitial grids (n = 124) assessed in a single study year (Table 15). In addition to interstitial grids, 101 interstitial transects were sampled, increasing the overall interstitial sampling effort. The locations of all sampled stranding mechanisms within the dataset are presented in Figure 3 and Figure 4.

Table 15: Sampling effort during reductions of each study year 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
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

LEGEND

STRANDING SAMPLE POINT (SLOPE)

- × 0-4%
- • >4%
- REACH BREAK
- WATERCOURSE
- ROAD SEGMENT
- CONTOUR (30 m INTERVAL)
- WETLAND
- DAM
- WATERBODY

SLOPE (%)

HIGH : 67

LOW : 0

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 - MAR. 2015) REACHES 1 TO 3

Golder Associates

PROJECT		12-1492-0117	FILE No.	
DESIGN	BH	14 Apr. 2014	SCALE AS SHOWN	REV. 0
GIS	KLN	19 May 2015	**FIGURE: 3**	
CHECK	BH	5 Jan. 2017		
REVIEW	DS	5 Jan. 2017		

Document Path: Y:\burnaby\CAD-GIS\Bur-Graphics\Projects\2012\1492\12-1492-0117\GIS\Mapping\MXD\Fish\Fig3_4_1214920117_Stranding Mechanism and slope_2014Data.mxd

LEGEND

STRANDING SAMPLE POINT (SLOPE)

- × 0-4%
- • >4%
- REACH BREAK
- WATERCOURSE
- FOOTBRIDGE
- TRAIL (OTHER)
- ROAD SEGMENT
- CONTOUR (30 m INTERVAL)
- WETLAND
- WATERBODY

SLOPE (%)

HIGH : 67

LOW : 0

500 0 500 Meters

SCALE 1:25,000

REFERENCE

1) GROUND SURFACE: TDUNC3 CONVERTED INTO RASTER, PROVIDED BY CLIENT.
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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 - MAR. 2015) REACHES 3 TO 5

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	5 Jan. 2017	**FIGURE: 4**
REVIEW	DS	5 Jan. 2017	

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In the present year of study (2014 - 2015), a total of 1169 fish were observed, representing 10 species [four sportfish and six non-sportfish species (Table 16)]. In comparison to all study years, juvenile Rainbow Trout (*n* = 737) encounters were the highest in the present year, and were the most abundant sportfish observed (63.0% of the total catch). Kokanee young-of-the-year were the next abundant sportfish, accounting for 1.9% of the total number of fish encountered (Table 16, Figure 5). The most common non-sportfish taxa were Sculpin spp. and Longnose Dace, accounting for 16.2 and 12.2% of the total number of observed fish, respectively.

Table 16: Total number and relative composition of fish species captured or observed during all stranding assessments conducted on the lower Duncan River from September 2006 to April 2015.

Species and Life Stage			N Fish (% of Total) 2006-07	N Fish (% of Total) 2007-08	N Fish (% of Total) 2008-09	N Fish (% of Total) 2009-10	N Fish (% of Total) 2010-11	N Fish (% of Total) 2011-12	N Fish (% of Total) 2012-13	N Fish (% of Total) 2013-14	N Fish (% of Total) 2014-15
Sport Fish	Rainbow Trout	Adult	0 (0)	0 (0)	0 (0)	1 (0.1)	0 (0)	0 (0)	0 (0)	1 (0.2)	0 (0)
		Juvenile	130 (37.1)	278 (11.5)	530 (33.2)	113 (12.3)	343 (25.2)	419 (24.0)	332 (37.1)	241 (40.2)	737 (63.0)
	Bull Trout	Adult	0 (0)	0 (0)	0 (0)	4 (0.4)	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 (1.4)
	Mountain Whitefish	Adult	0 (0)	1 (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)	181 (10.4)	6 (0.7)	49 (8.2)	3 (0.3)
	Pygmy Whitefish	Adult	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
		Juvenile	0 (0)	0 (0)	0 (0)	1 (0.1)	3 (0.1)	0 (0)	0 (0)	0 (0)	0 (0)
	Kokanee	Adult	0 (0)	97 (4.0)	572 (35.8)	112 (12.2)	42 (3.1)	55 (3.2)	111 (12.4)	0 (0)	0 (0)
		Y-O-Y	0 (0)	1695 (70.4)	85 (5.3)	109 (11.9)	83 (6.1)	844 (48.3)	257 (28.7)	0 (0)	22 (1.9)
	Burbot	Adult	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)
Non-sport fish	Longnose Dace		117 (33.4)	15 (0.6)	103 (6.5)	273 (29.7)	551 (40.5)	30 (1.9)	32 (3.6)	227 (37.8)	143 (12.2)
	Dace spp.		0 (0)	0 (0)	0 (0)	12 (1.3)	1 (0.1)	0 (0)	0 (0)	0 (0)	0 (0)
	Slimy Sculpin		0 (0)	13 (0.5)	11 (0.7)	62 (6.8)	39 (2.9)	3 (0.2)	0 (0)	1 (0.2)	12 (1.0)
	Torrent Sculpin		0 (0)	1 (0)	1 (0.1)	0 (0)	0 (0)	3 (0.2)	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)
	Sculpin spp.		23 (6.6)	16 (0.7)	65 (4.1)	62 (6.8)	165 (12.1)	80 (4.5)	130 (14.5)	46 (7.7)	189 (16.2)
	Sucker spp.		2 (0.6)	4 (0.2)	26 (1.6)	166 (18.1)	54 (4.0)	9 (0.5)	16 (1.8)	32 (5.3)	42 (3.5)
	Redside Shiner		0 (0)	112 (4.6)	8 (0.5)	15 (1.6)	0 (0)	0 (0)	7 (0.8)	0 (0)	3 (0.3)
	Northern Pikeminnow		0 (0)	0 (0)	2 (0.1)	0 (0)	15 (1.1)	4 (0.2)	1 (0.1)	1 (0.2)	0 (0)
	Peamouth Chub		0 (0)	0 (0)	6 (0.4)	6 (0.7)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
	Unidentified		75 (21.4)	20 (0.8)	105 (6.6)	4 (0.4)	13 (1.0)	114 (6.4)	0 (0)	0 (0)	0 (0)
All Species Total			**350**	**2409**	**1596**	**918**	**1361**	**1745**	**896**	**600**	**1169**

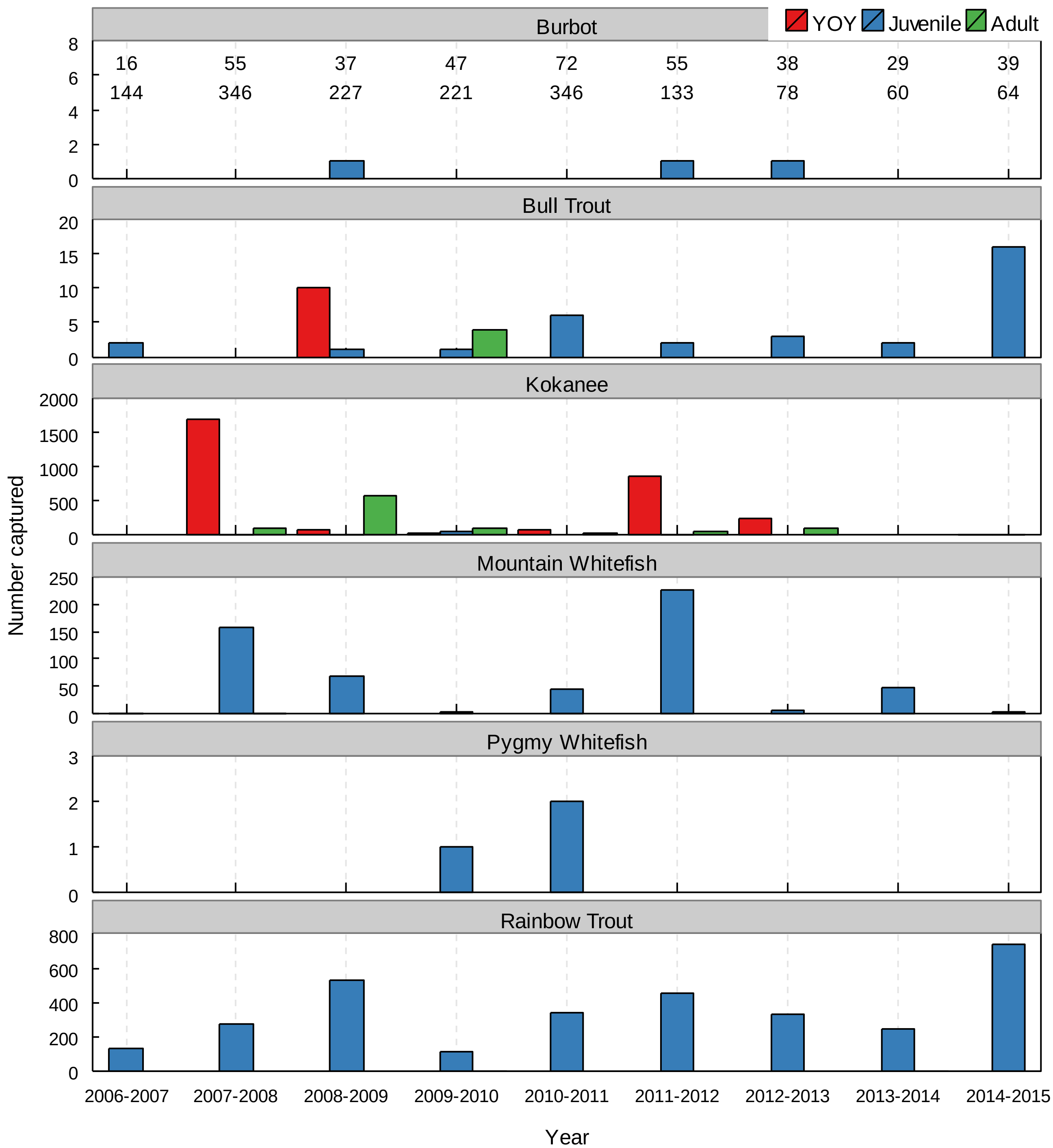


Figure 5: Abundances of sportfish species, separated by life stage, observed in stranding assessments between 2006 and 2015. Note the different y-axis scales among panels. On the uppermost panel, the numbers of sampled sites and pools are provided in the first and second lines, respectively.

3.3 Differences between Pre-WUP and Post-WUP Operations

Based on DDM flow data provided by BC Hydro, the DDMMON-3 RIVER 2D model outputs, and subsequent analysis, the overall mean area exposed during pre-WUP operations was 17.8 km², in comparison to 13.7 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).

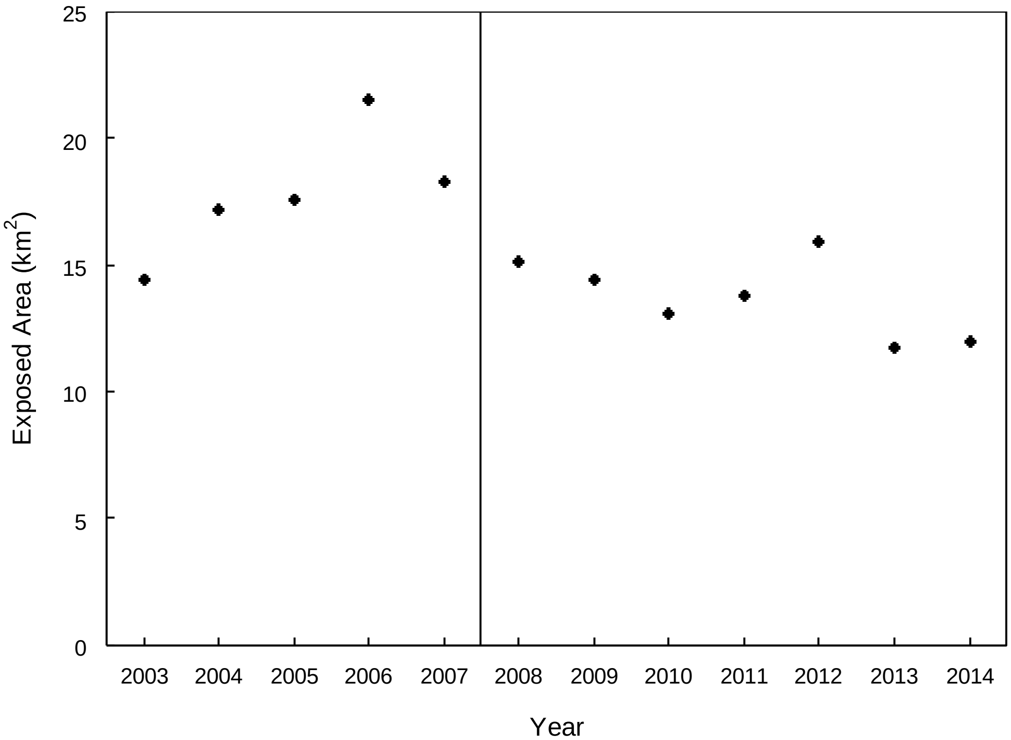


Figure 6: Total area exposed by all annual reductions in the LDR by year of operations, calculated using DRL discharge (black) or DDM only (red). The vertical line denotes the beginning on WUP flows in 2008.

Interannual variability in mean discharge as assessed at the gauge at DRL overall is higher in the pre-WUP period, with the greatest reduction in discharge variation seen in the October to January period. Under the current operational regime (i.e., 2008 to present), there is almost no interannual deviation during the October to January period (Figure 7). Decreased discharge variability post-WUP is also seen between January and March. An additional change in discharge patterns is seen between March and May, where discharge trend changed from gradual increase pre-WUP to a gradual decrease post-WUP (Figure 7).

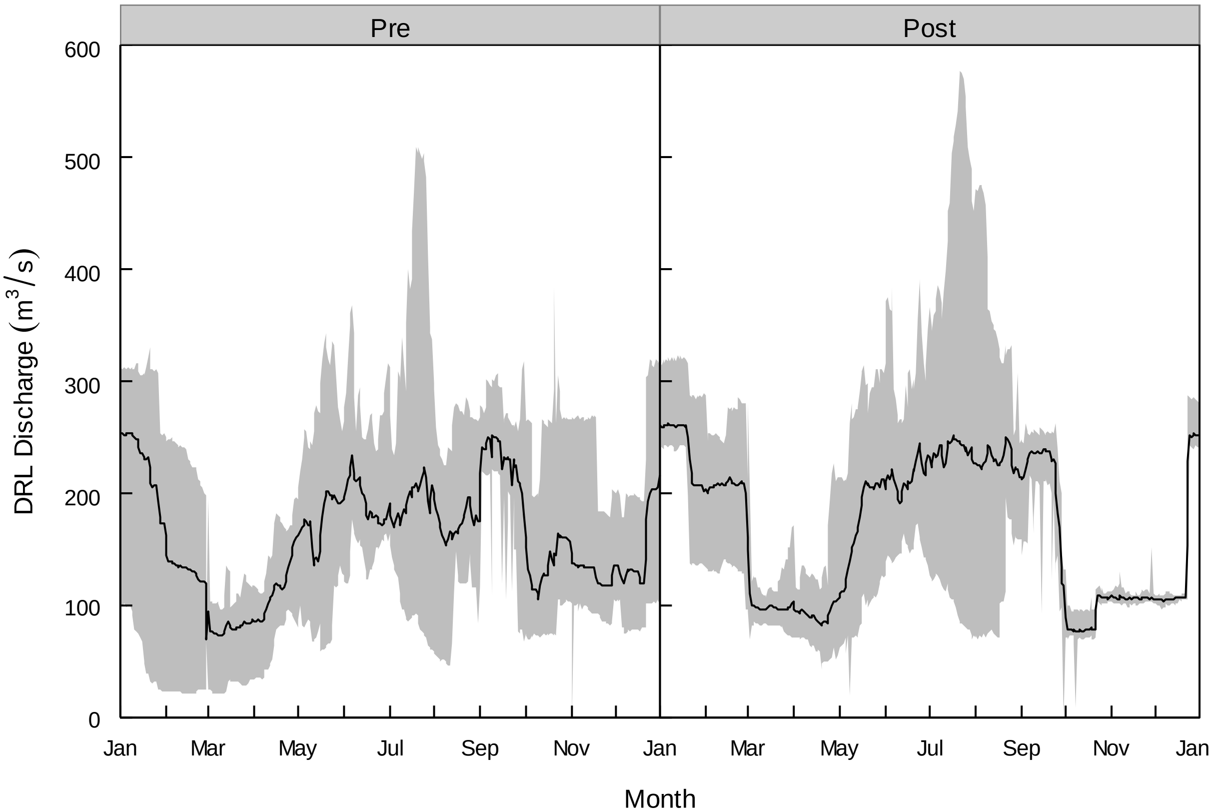


Figure 7: Minimum, maximum (grey ribbon) and mean (black line) discharge as measured at the WSC DRL gauge in the LDR by month during pre-WUP operations (2002 - 2007) and post-WUP operational implementation (2008 - 2015).

Although the total magnitude of pre-WUP flow reductions from DDM exhibited smaller ranges within each year in comparison to post-WUP operations, 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.

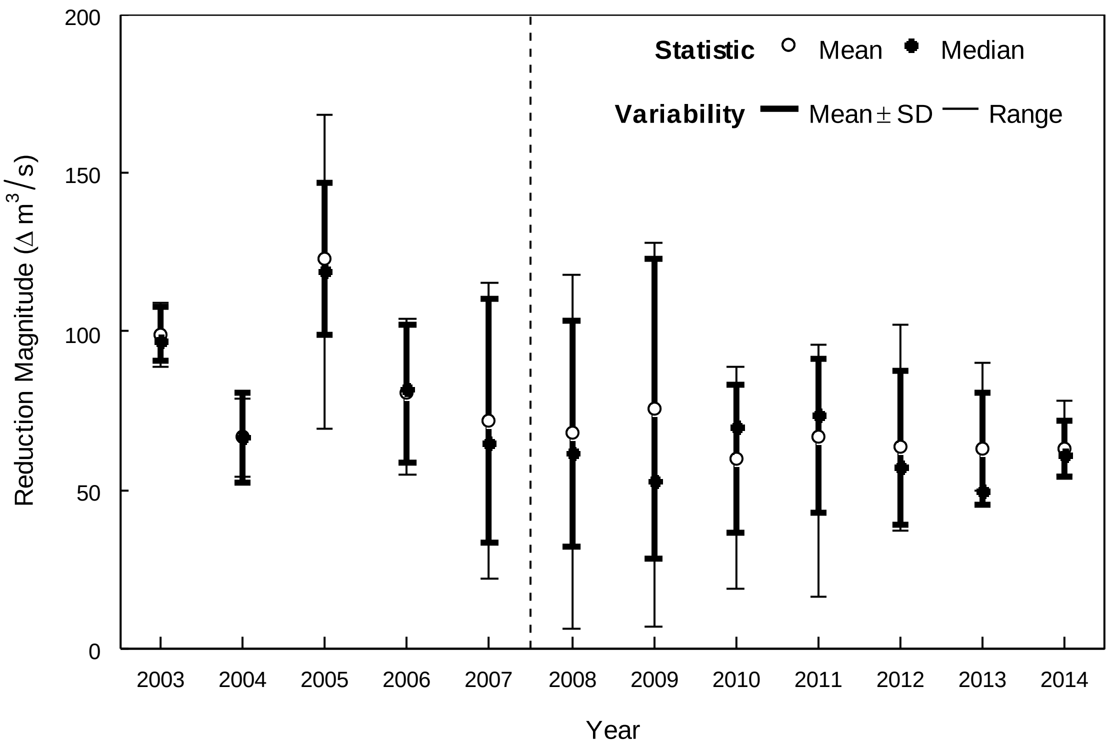


Figure 8: Reduction magnitude (Δm³/s) by year, depicting annual range (min, max), mean, and median, as well as mean ± SD.

In three of the five years examined during pre-WUP operations, ramping rate (Δm³ s^{-1} h^{-1}) exhibited substantial variations and range (Figure 9). The remaining years in the pre-WUP period were similar to operations during post-WUP. Overall, post-WUP ramping rates were similar in all years examined.

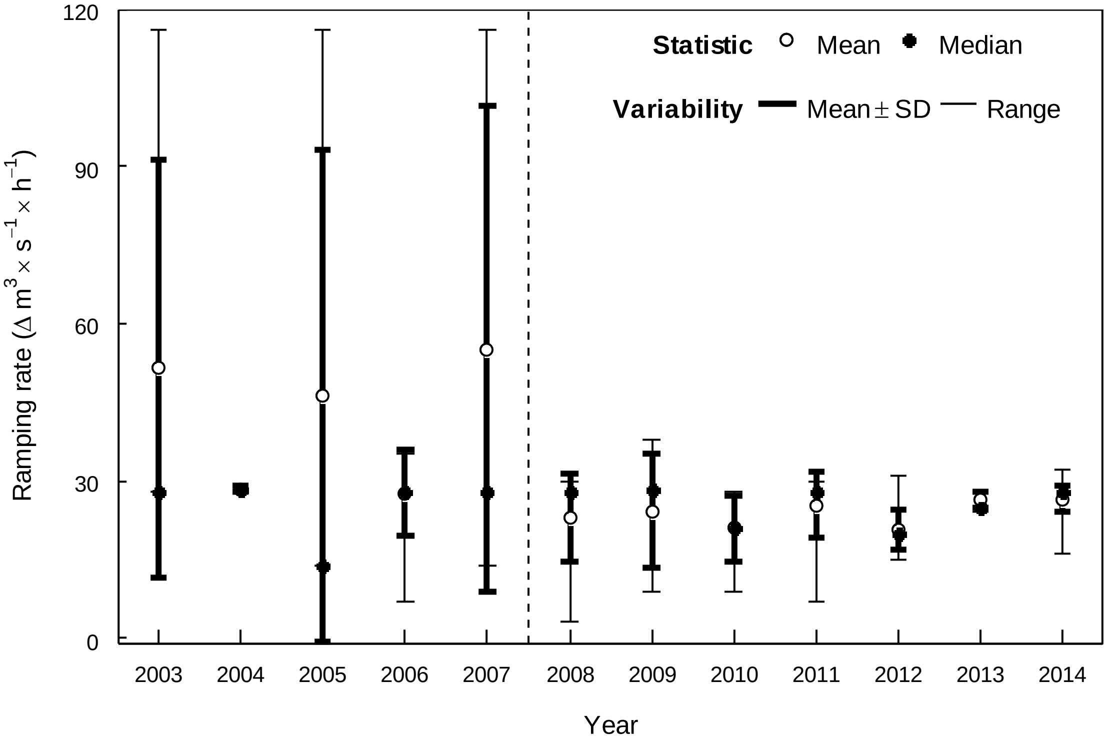
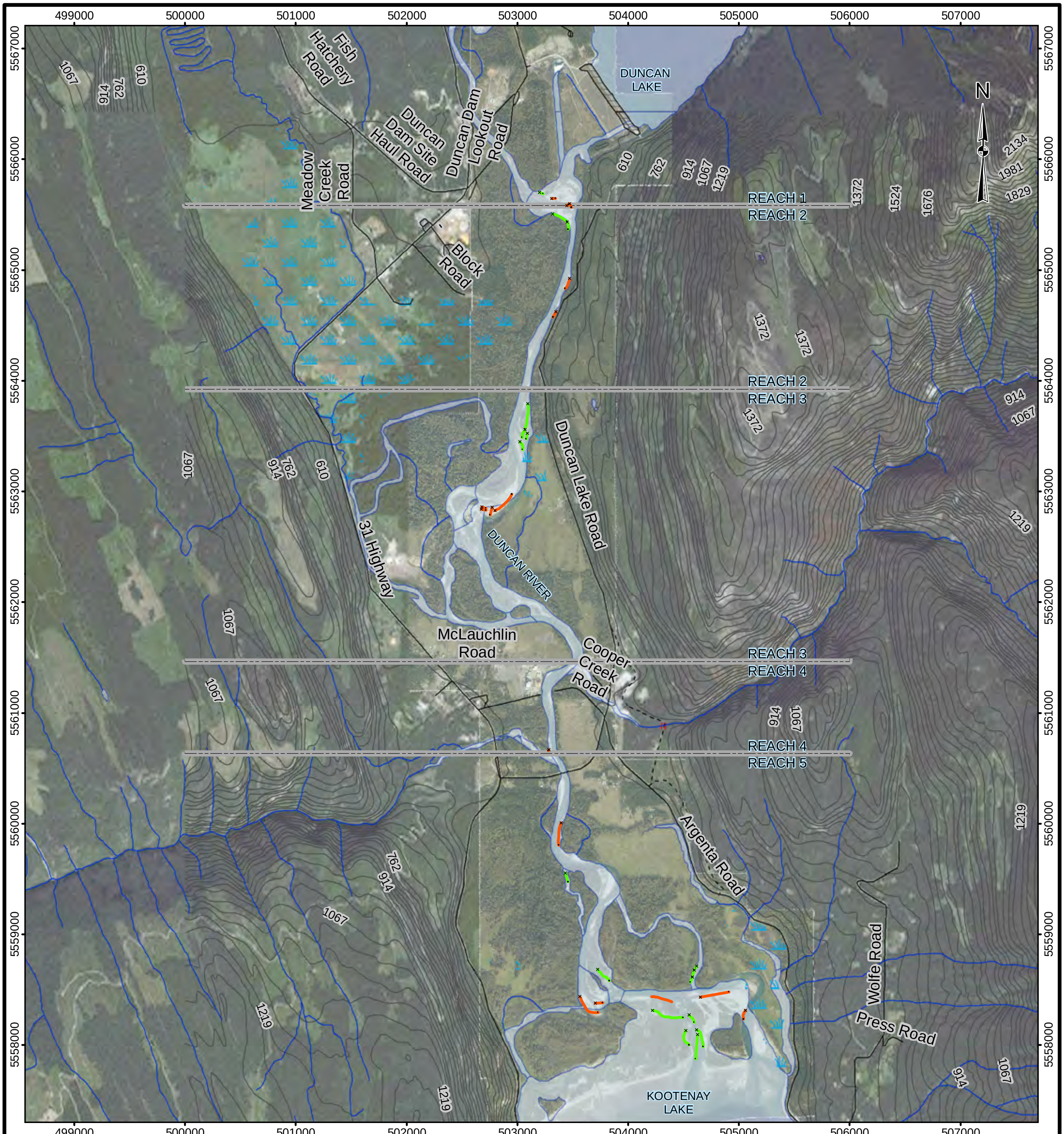


Figure 9: Ramping rate (Δm³ s⁻¹ h⁻¹) by year, depicting annual range (min, max), mean, and median, as well as mean ± SD.

3.4 Fish Abundance Assessment

A total of 49 sites and 11,526 m of shoreline were surveyed during the 2014 snorkeling fish abundance assessment (Figure 10), with a total of 2,502 fish counted across all sites and strata (Figure 11 and Table 12). Maximum fish density for Mountain Whitefish was recorded at a deep/slack site, with 0.43 fish/m², whereas maximum Rainbow Trout density was recorded in a shallow/fast site, with 0.36 fish/m² (Figure 12). The lowest mean counts of Mountain Whitefish (17.6 fish/site) were recorded in shallow, slack sites, whereas highest mean counts (75.4 fish/site) were observed in deep, slack sites. This relatively high mean was largely driven by the results at site 35, where 318 Mountain Whitefish were observed. The lowest mean counts of Rainbow Trout (2.6 fish/site) were recorded in deep, slack sites, whereas highest mean counts (53.1 fish/site) were observed in shallow, fast sites.



LEGEND

SNORKEL SITE LOCATION (2013)

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

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	FIGURE: 10
CHECK	BH	5 Jan. 2017	
REVIEW	DS	5 Jan. 2017	

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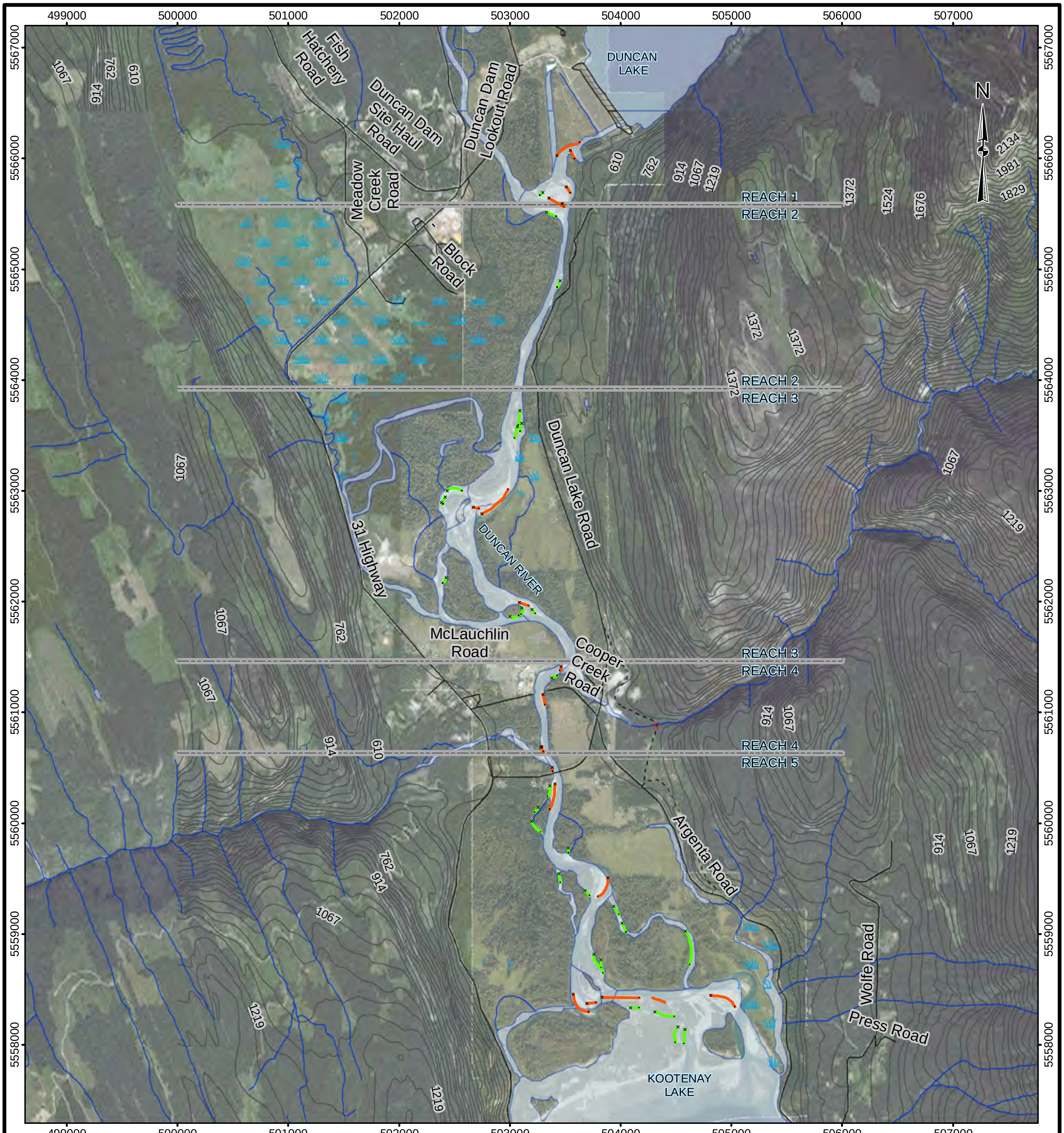

LEGEND

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

PROJECT		12-1492-0117	FILE No.	
DESIGN	BH	14 Apr. 2014	SCALE AS SHOWN	REV. 0
GIS	KLN	19 May 2015	FIGURE: 11	
CHECK	BH	5 Jan. 2017		
REVIEW	BH	5 Jan. 2017		

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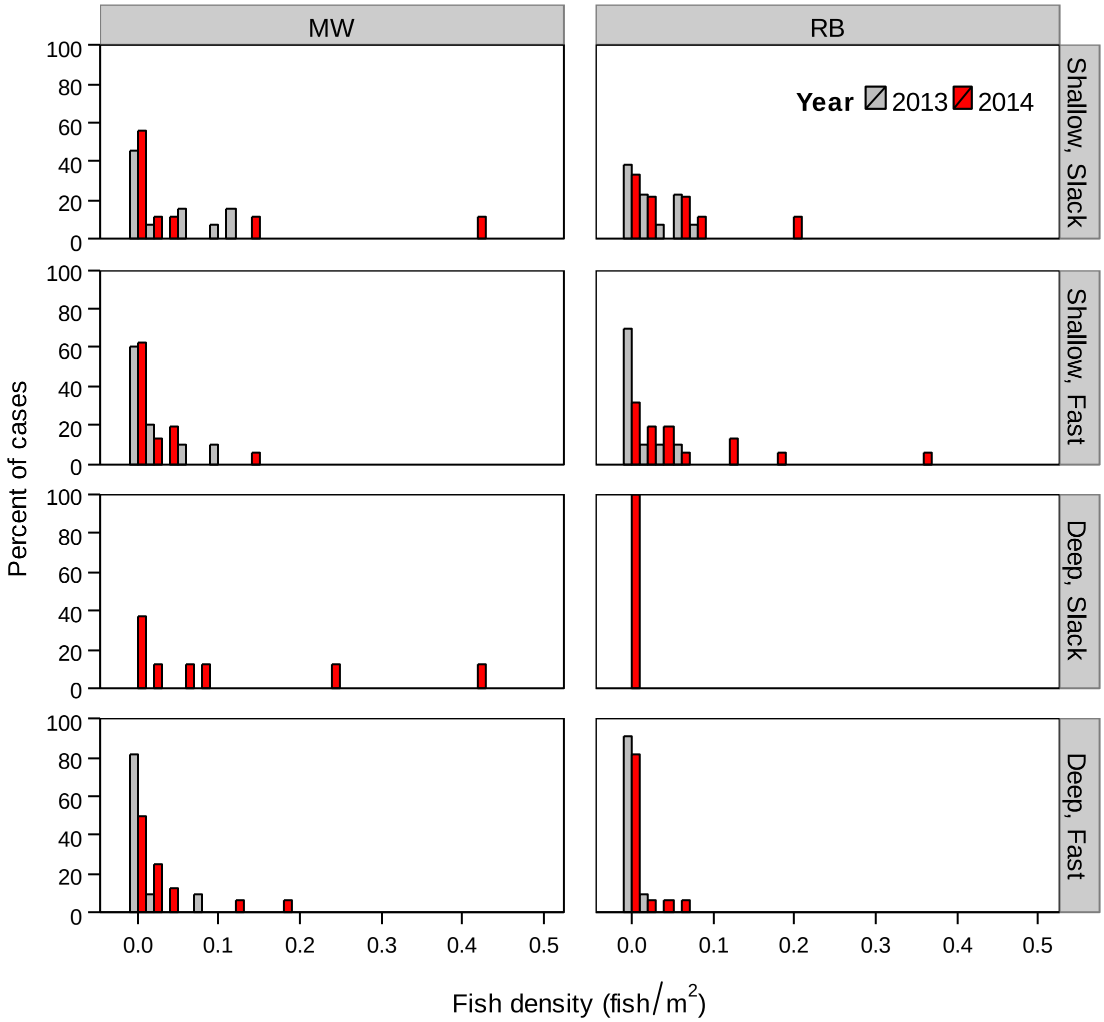

Figure 12: Fish density (fish/m²) across species, depth, and flow strata for 2013 and 2014 data.

Table 17: Summary of fish counts across depth and flow strata, as recorded from Year 7 (September 2014) snorkeling surveys.

Stratum	Depth	Flow	NSite	Mountain Whitefish			Rainbow Trout		
				N	Mean	SD	N	Mean	SD
1	Shallow	Slack	9	161	17.9	28.9	134	14.9	13.5
2	Shallow	Fast	16	281	17.6	23.9	849	53.1	45.3
3	Deep	Slack	8	603	75.4	104.8	21	2.6	2.4
4	Deep	Fast	16	325	20.3	18.6	128	8.0	9.2
Total			**49**	**1,370**			**1,132**		

The variability of fish density within strata is depicted in Figure 12. Mountain Whitefish zero densities ranged from approximately 40% to 80% of the cases, depending upon the strata. Densities of non-zero data ranged from 0.02 – 0.4 fish/m². In comparison to Year 6 results (2013), the percentage of these low densities was slightly higher in 2014 in both shallow strata. For the Deep/Fast stratum, all Mountain whitefish densities in Year 6 (2013) were below 0.1, in comparison to 60% in Year 7 (2014). The highest density of Mountain Whitefish, 0.43 fish/m², was recorded in a Deep/Slack site in Year 7.

In Year 7, sites with zero densities of observed Rainbow Trout ranged from 31% to 81% in all strata, except for Deep/Slack where fish were not observed (Figure 12). Densities were consistently lower across all strata in 2013 in comparison to 2014. The highest density of Rainbow Trout, 0.36 fish/m², was recorded in a Shallow/Fast site in Year 7.

For Rainbow Trout analysis, flow was not a significant variable, since the 95% CRI values of the coefficients for the two strata overlapped (95% CRI values of -4.208 to -3.200 and -4.066 to -2.683 for the slack and fast strata, respectively). In the model incorporating the four strata as a variable, the first two strata (shallow/fast and shallow/slack) were significantly different from the other strata (deep/fast and deep/slack). However, there was no significant difference within the depth strata (i.e., the 95% CRI values of both shallow strata overlapped, and so did those of both deep strata). Year was not a significant variable (95% CRI values of -4.506 to -3.204 for 2013 and -3.937 to -2.879 for 2014). The depth model was the only one with a significant effect, with shallow sites (median of -2.799, 95% CRI values of -3.252 to -2.357) being significantly different from deep sites (median of -4.520, 95% CRI values of -5.125 to -3.989).

Abundance estimates for Rainbow Trout were slightly higher in 2014 than in 2013, across all three strata that were sampled in both years (Figure 13). This reflected the slight increase in areas of strata 1, 2, and 4 between 2013 and 2014 (Table 6). The Deep/Slack stratum, sampled only in 2014, was estimated to contain a median of 1,788 Rainbow Trout (95% CRI of 970 to 3,049).

For Mountain Whitefish analysis, flow was not a significant variable, since the 95% CRI values of the coefficients for the two strata overlapped (95% CRI values of -4.217 to -3.156 and -3.513 to -2.026 for the slack and fast strata, respectively). In the model incorporating the four strata as a variable, there was no significant difference among the first, second, or fourth strata (shallow/slack, shallow/fast, and deep/fast; 95% CRI values of -3.945 to -2.154, -4.979 to -3.362, and -3.988 to -2.539). The third stratum (deep/slack; 95% CRI values of -3.694 to -1.007) was marginally significantly different from the second stratum (shallow/fast). Year was not a significant variable (95% CRI values of -4.004 to -4.048 for 2013 and -4.048 to -2.870 for 2014). Depth was also not significant, with shallow sites (median of -3.649, 95% CRI values of -4.272 to -3.084) overlapping the deep sites (median of -3.003, 95% CRI values of -3.657 to -2.388). Therefore, the model chosen for interpretation was the intercept-only model, with site-specific random effects.

Similar to Rainbow Trout, abundance estimates for Mountain Whitefish were slightly higher in 2014 than in 2013, across all three strata that were sampled in both years (Figure 13), reflecting the slight increase in areas of strata 1, 2, and 4 between 2013 and 2014 (Table 6). The Deep/Slack stratum, sampled only in 2014, was estimated to contain a median of 5,646 Mountain Whitefish (95% CRI of 3,500 to 8,785 fish).

Abundance estimates for both Mountain Whitefish and Rainbow Trout were the highest and most variable for Shallow/Fast habitats, and the lowest and least variable for the Shallow/Slack habitats reflecting the area distribution among the four strata (Table 6).

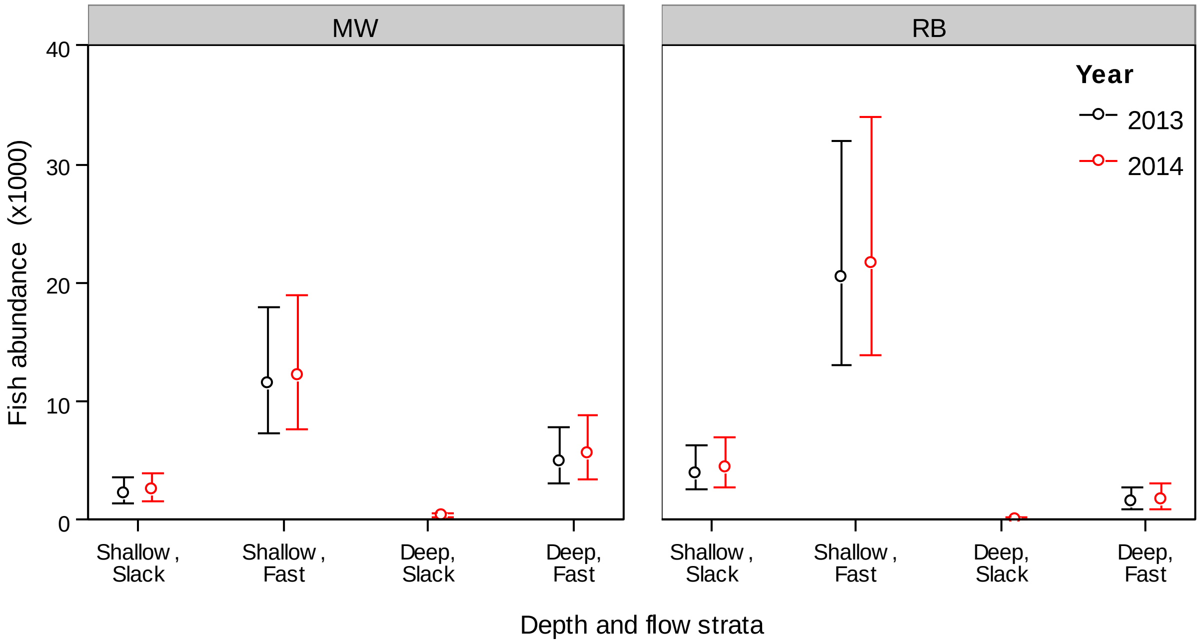

Figure 13: Median Mountain Whitefish and Rainbow Trout abundance (density x stratum area), plotted by depth and flow strata and their respective 95% credibility intervals.

The total abundance estimates were similar across the two sampling years (Table 18). However, the 2013 estimates differed from those reported in Year 6 report (Golder 2015), due to changes in the model specification and increased data set.

Table 18: Comparison of fall 2013 and 2014 abundance estimates of Mountain Whitefish and Rainbow Trout. Abundances are median Bayesian estimates, with lower and upper 95% credibility intervals in parentheses; numbers are rounded to nearest fish.

Species	Year 6 Report (Fall 2013: Golder 2014)	Current Abundance Estimation	
		Fall 2013	Fall 2014
MW	36,936 (23,315 – 52,325)	18,840 (11,807 – 29,275)	20,782 (12,915 – 32,326)
RB	16,330 (9,985 – 22,874)	26,148 (16,477 – 40,940)	28,083 (17,662 – 44,104)

3.5 Fish Stranding Assessment

Pool stranding estimates in the following sections refer to Mountain Whitefish and Rainbow Trout populations only. Low encounters in all years within the dataset precluded estimates for the other species of interest.

3.5.1 Presence of Pools

The variability of pools numbers documented in each reduction was similar between low and high slope habitat in the Year 6 assessment (Golder 2015). However, in the current study, low-slope sites had somewhat elevated pool densities (Figure 14). Due to the high variability and low data volume, the effect of slope was not included in the current analysis. However, future studies may be necessary to explore the effect of slope on pool stranding.

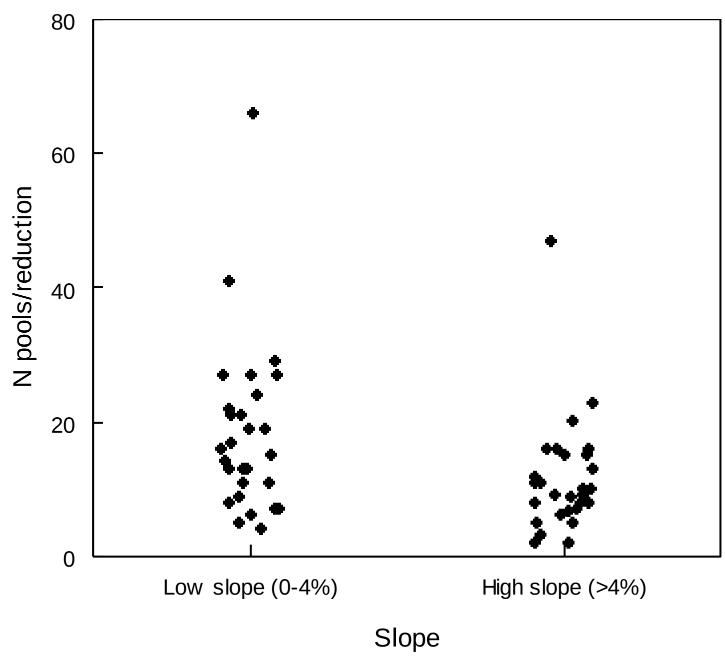

Figure 14: Number of pools recorded per reduction in low-slope and high-slope habitat, 2010 - 2014.

A significant statistical effect of stranding site type on pool density was not found, as the estimated 95% credibility intervals overlapped (index coefficient estimated as 0.891 [95% CRI of 0.244-1.544] and random site coefficient estimated as -0.158 [95% CRI of -0.678-0.292]). Therefore, the variable was removed from the model, and the data were analysed without the effect of index and random site differentiation. While pool densities in random sites exhibited higher variation in comparison to index sites, the majority of recorded pool densities were fairly low, often lower than those recorded at index sites (Figure 15). However, the current dataset is likely not large enough to detect a significant effect. With continued future sampling, the relationship should be re-evaluated.

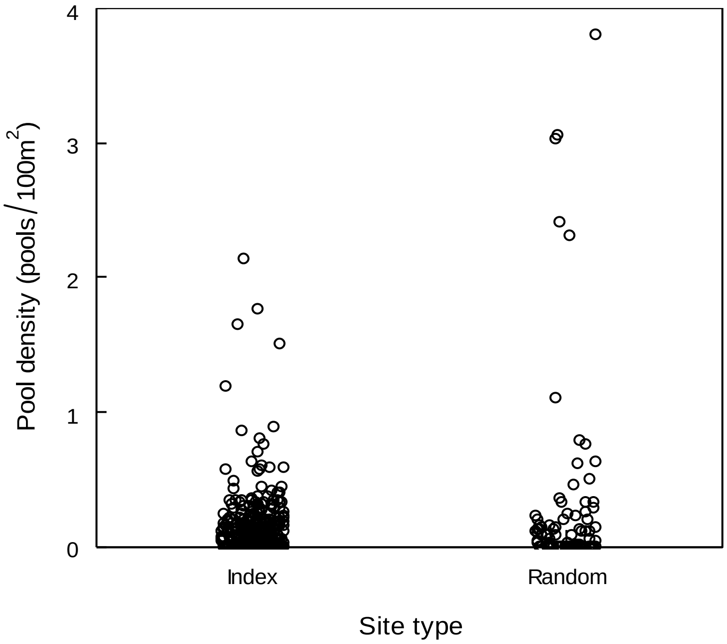

Figure 15: Pool densities, recorded during 2006-2014 stranding years, plotted against site type.

The number of pools per assessed flow reduction was estimated to allow the number of fish 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 median number of pools that formed during the stranding surveys was similar in 2012-2015 and slightly higher in 2010-2012 (Figure 16).

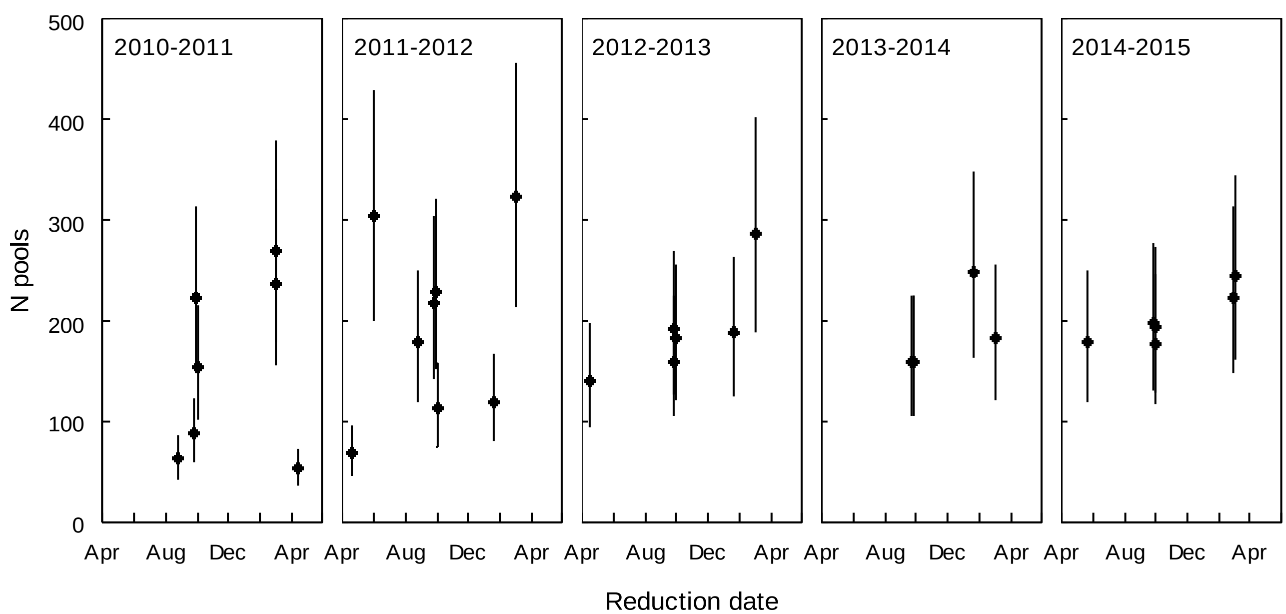

Figure 16: Median estimates of pool numbers formed during the 2010-2014 stranding events, plotted by reduction and stranding year. Error bars are 95% credibility intervals.

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 0.470 for Rainbow Trout (median value; 95% credibility interval of 0.448-0.492) and 0.145 for Mountain Whitefish (median value; 95% credibility interval of 0.127-0.164).

The variability in the number of fish stranded per pool was similar between low and high slope habitat (Figure 17). 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 18). On the other hand, a seasonal effect was not observed for Mountain Whitefish, and therefore seasonal effect on pool stranding was included only for Rainbow Trout abundance estimation.

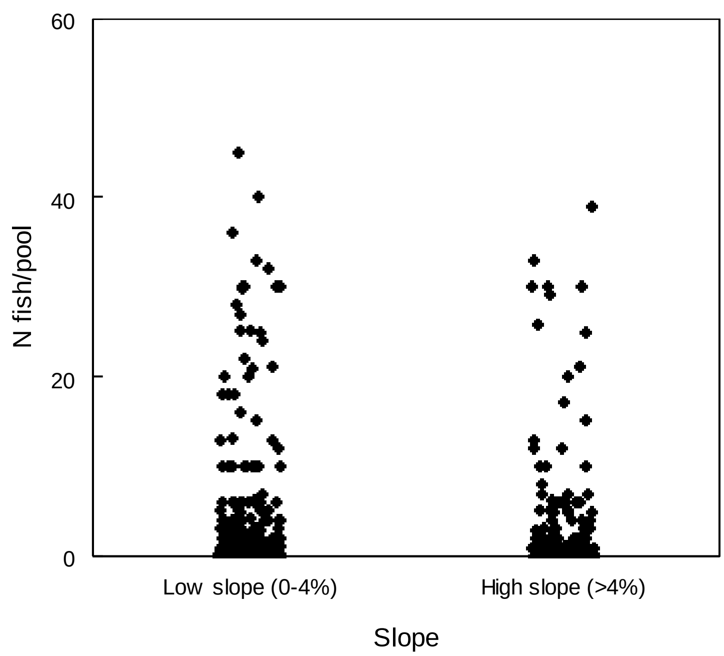

Figure 17: Number of collected fish per pool, plotted by low-slope and high-slope habitat, 2010 - 2014.

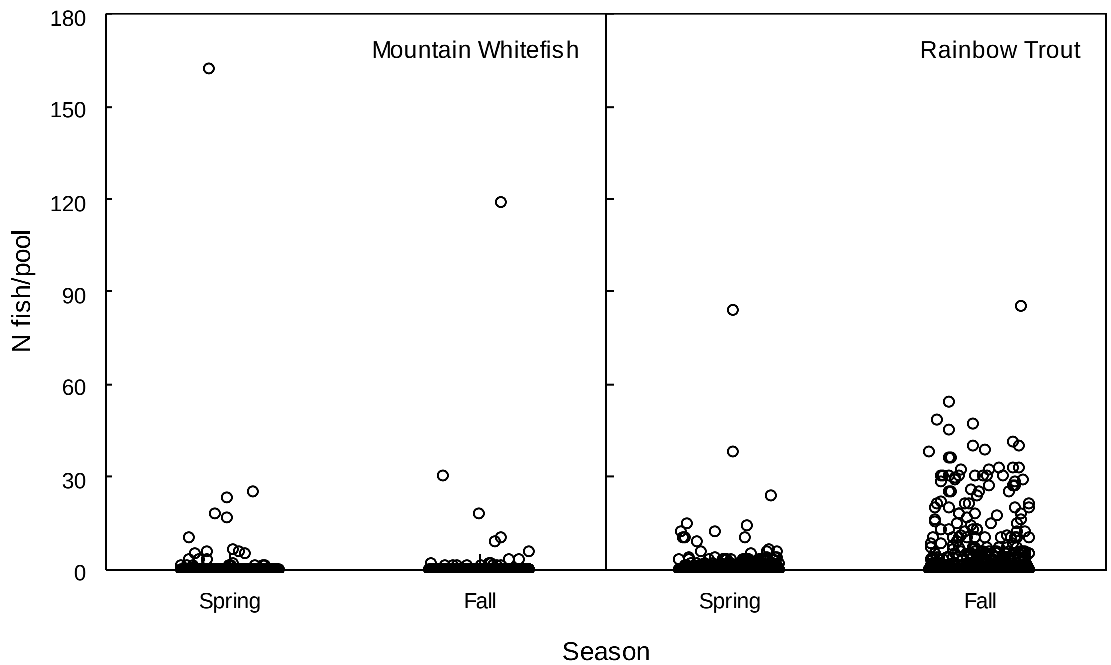

Figure 18: Counts of fish/pool, recorded during 2006-2014 stranding surveys, and plotted by season and species.

The median number of Rainbow Trout fry per pool for the spring season (January – June) was estimated to be 2.06 (1.38 – 3.02) fish/pool (Figure 19). In contrast, the median number of Rainbow Trout juveniles stranded per pool in the fall (July to December) was estimated at 8.22 (5.73 – 11.83; Figure 19). The season effect on stranding numbers was found to be significant ($p < 0.05$), with median fall stranding estimates four times higher than those for winter/spring.

Based on the presence of pools and number of fish per pool estimates, it was then possible to estimate the number of fish stranded in pools for individual reduction events (Figure 20). With the exception of 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 20) sampled in March 2012. The two highest Rainbow Trout stranding events were estimated for September 2012 (median stranding of 4,567 fish; 95% CRI of 2,949 and 6,679) and October 2014 (median estimate of 3,418 fish; 95% CRI of 2,230 and 4,902).

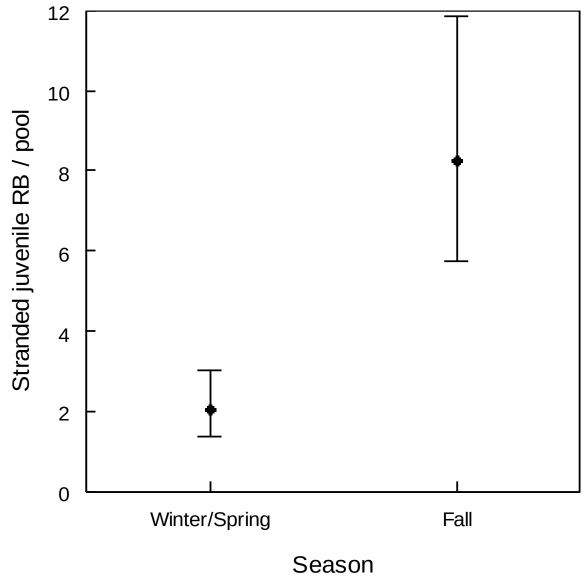

Figure 19: Median estimates of stranded Rainbow Trout per pool, plotted by season, 2010 - 2014. Error bars are 95% credibility intervals.

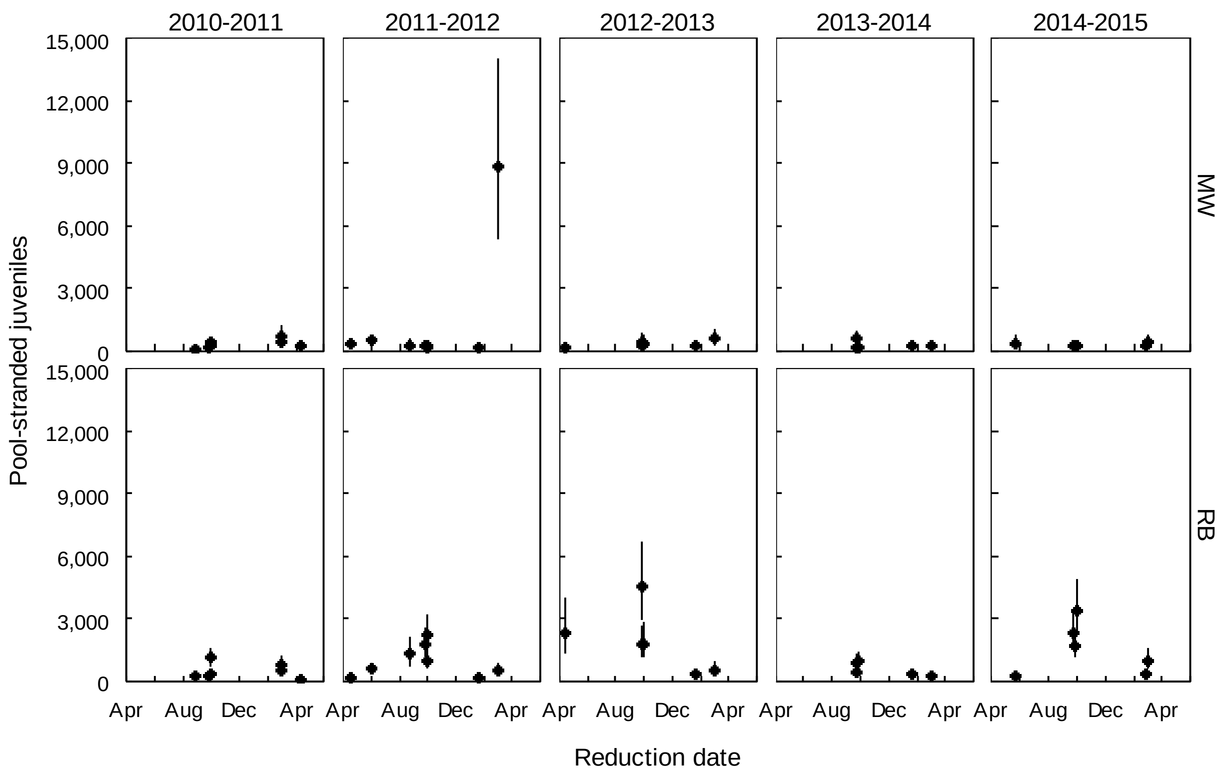

Figure 20: Median estimates of pool-stranded Mountain Whitefish and Rainbow Trout during the 2010-2014 stranding events, plotted by species, reduction, and stranding year. Error bars are 95% credibility intervals.

3.5.3 Interstitial Stranding

Over the last three study years when interstitial sample methodology was standardized, very few interstitially stranded fish have been observed (Figure 21). In total, 24 Rainbow Trout and 2 Mountain Whitefish were found to be interstitially stranded. A relationship between stranded fish counts and slope was not observed in the data (Figure 21 and Figure 22); therefore the variable was not included in the model.

A relationship between interstitial stranding and ramping rates was suspected (Figure 22). The model that included ramping rate as an effect on the probability of stranding indicated no significant relationship between the two variables (estimated coefficient of 0.025 [median value; 95% CRI between -0.122 and 0.210]). The variable was therefore removed from the final model. However, as more data are accumulated, the relationship may become significant. The effect should therefore be re-evaluated in future work.

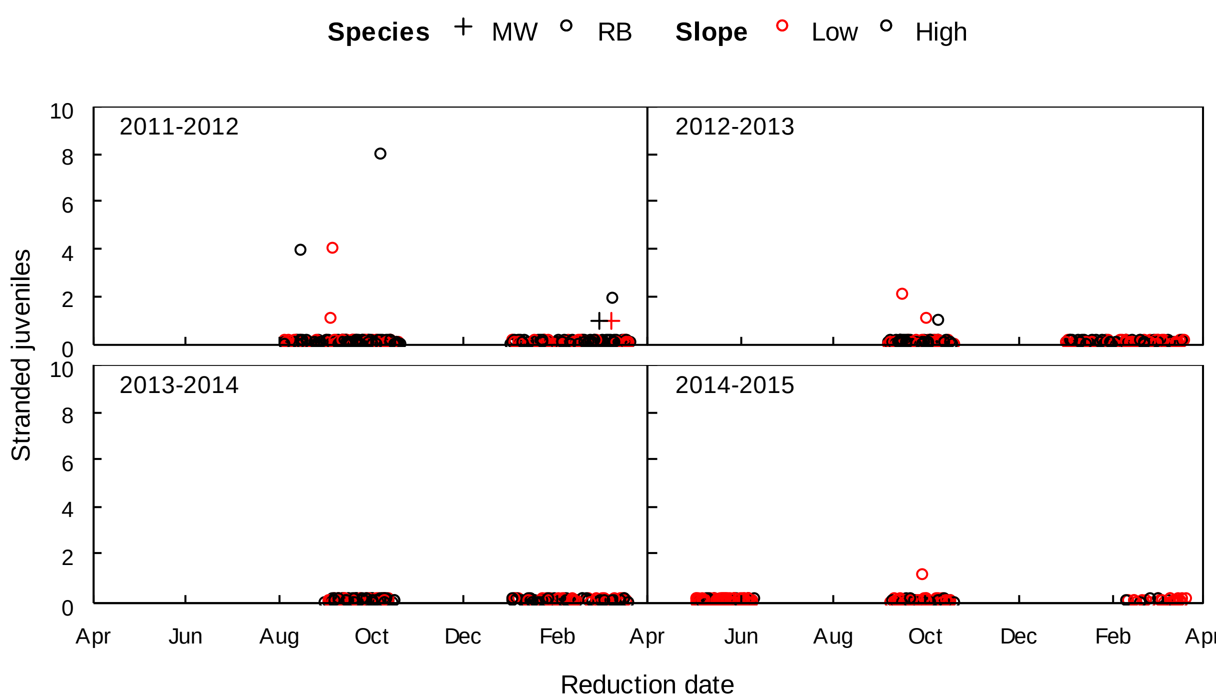

Figure 21: Counts of 2011-2014 interstitially stranded Mountain Whitefish and Rainbow Trout, plotted by study year, reduction date, and slope (where low slope is 0-4%, and high slope is >4%).

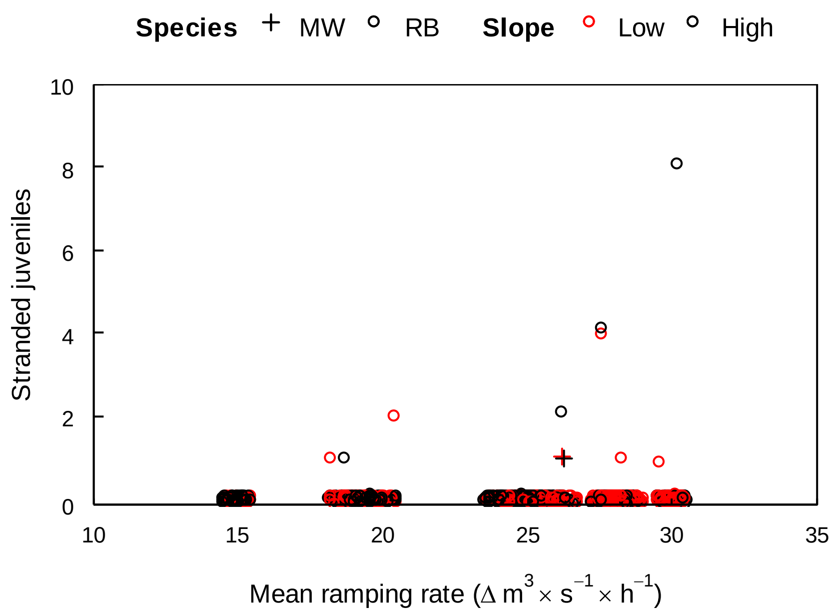

Figure 22: Counts of interstitially stranded Mountain Whitefish and Rainbow Trout vs. mean ramping rate, plotted by species and slope (where low slope is 0-4%, and high slope is >4%), 2011 - 2014.

The effect of index/random site on interstitial stranding was not found to be significant, as the 95% CRIs of the two coefficients overlapped (index site coefficient was estimated to be -5.346, with 95% CRI between -6.520 and -4.480, and random site coefficient was estimated to be -4.826, with 95% CRI between –5.884 and -4.027).

Summed by reduction, median interstitial stranding estimates ranged from 1,263 fish (October 01, 2011) to 3,682 fish (March 01, 2012; Figure 23). Upper 95% credibility values ranged between 2,646 (October 01, 2011) to 7,712 fish (March 01, 2012; Figure 23). When using grid data only, estimates were substantially higher, with median values ranging between 4,317 fish and 12,584 fish, and upper 95% credibility limits ranging between 9,834 and 25,293 fish (data not shown).

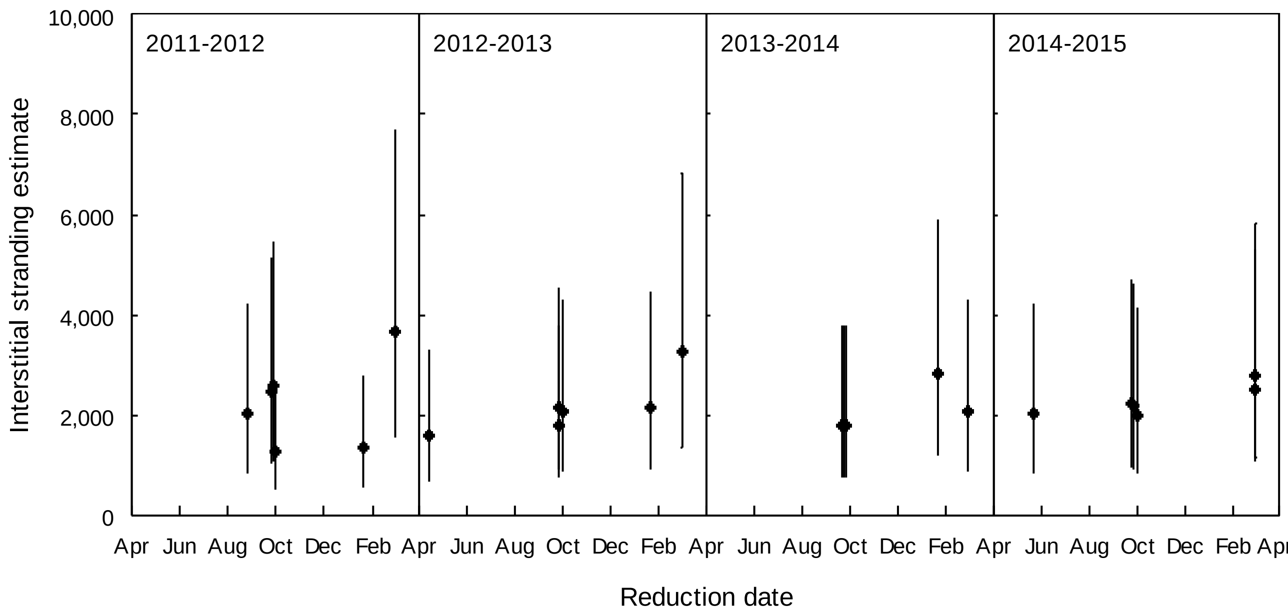

Figure 23: Median estimates of Rainbow Trout juveniles and fry interstitially stranded during the 2011-2014 stranding events, plotted by study year and reduction date. Error bars are 95% credibility intervals.

Similar to the Rainbow Trout results, Mountain Whitefish interstitial stranding rate was not significantly affected by ramping (coefficient estimate of 0.098 [median value; 95% CRI between -0.212 and 0.494]) or index/random sites – index site coefficient was estimated as -6.112 (95% CRI between -8.075 and -4.948) and random site coefficient estimated as -26.354 (95% CRI between -74.045 and -7.694). The extremely low estimates of random-site coefficient were due to zero Mountain Whitefish recorded at random sites. Both parameters were therefore removed from the final model. These results were based on modeling of extremely low fish counts (only two Mountain Whitefish were recorded as interstitially stranded during 2011-2014 sampling years). The models are not considered robust, and should be interpreted with care. With additional data, the results will likely change.

As with Rainbow Trout, Mountain Whitefish interstitial stranding estimates were considerably lower than those reported in previous years. Estimated median values ranged between 353 fish on October 01, 2011, to 1,030 fish (March 01, 2012). Upper 95% credibility values ranged from 2,218 fish on October 01, 2011, to 6,464 fish on March 01, 2012; Figure 24).

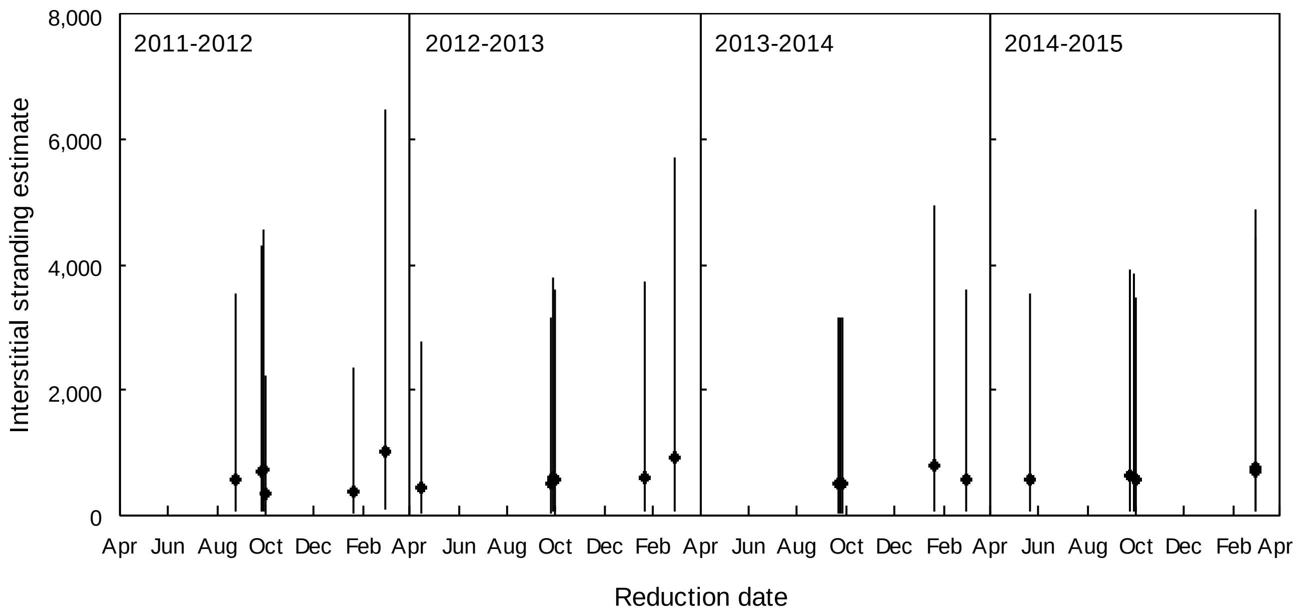

Figure 24: Median estimates of Mountain Whitefish juveniles and fry interstitially stranded during the 2011-2014 stranding events, plotted by study year and reduction date. Error bars are 95% credibility intervals.

The total yearly interstitial stranding values estimated for Rainbow Trout in the current study were 3-4 times lower than those reported previously (Table 19). This was a result of a combination of increased number of data points and the use of transect sampling. During the 2011-2014 study years, median estimates of total annual interstitial stranding of Rainbow Trout ranged between 10,287 and 13,753 fish, with 95% credibility intervals ranging from 4,312 (lower limit, 2013) to 28,804 fish (upper limit, 2014; Table 19). Median estimates of total annual interstitial stranding of Mountain Whitefish using the updated model decreased approximately 2 - fold in comparison to previous values (Year 6 model estimates; Golder 2015). Current median estimates were 2,878 (Year 6) and 3,847 fish (current year), with 95% credibility intervals of 228 (lower limit, Year 6) to 24,144 (upper limit, current year).

Table 19: Total annual interstitial stranding estimates, detailing median and 95% credibility interval (CRI) values for Rainbow Trout and Mountain Whitefish in each of 2011-2014 study years.

Study Year	Rainbow Trout		Mountain Whitefish	
	Median	95% CRI	Median	95% CRI
2011-2012 (obtained during Year 4: Golder and Poisson 2012)	71,261	19,418 – 197,418	Estimate not obtained	Estimate not obtained
2011-2012 (Year 4, using Year 6 modelling)	50,671	20,290 – 117,468	8,067	898 – 40,142
2012-2013 (Year 5, using Year 6 modelling)	49,287	19,736 – 114,261	7,846	873 – 39,046
2013-2014 (Year 6, using Year 6 modelling)	39,029	15,628 – 90,480	6,213	691 – 30,920
2011-2012 (Year 4, using current modelling)	13,356	5,599 – 27,973	3,736	295 – 23,447
2012-2013 (Year 5, using current modelling)	12,991	5,446 – 27,209	3,634	287 – 22,807
2013-2014 (Year 6, using current modelling)	10,287	4,312 – 21,546	2,878	228 – 18,060
2014-2015 (Year 7, using current modelling)	13,753	5,765 – 28,804	3,847	304 – 24,144

3.5.4 Total Stranding Estimates

Overall estimates of Rainbow Trout and Mountain Whitefish abundance and stranding by stranding mechanism for the current study year are summarized in Table 20 and Table 21, respectively. The sum of pool and interstitial stranding of Rainbow Trout resulted in a high loss value in relation to the 2014 abundance estimate. The high uncertainty related to interstitial stranding estimates suggests 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. The issue was less prominent with Mountain Whitefish. However, stranding estimates for the species were calculated based on a restricted dataset (e.g., only two interstitially stranded Mountain Whitefish were collected during 2011-2014 surveys), which decreased the robustness of estimates.

Table 20: Year 7 pool and interstitial stranding estimates for Rainbow Trout juveniles by assessed flow reduction, compared to current abundance estimation.

Reduction Event Number	Pool Stranding Estimates			Interstitial Stranding Estimates			Fall 2014 Abundance Estimate		
	Lower	Median	Upper	Lower	Median	Upper	Lower	Median	Upper
RE2014-03	113	249	491	847	2,021	4,234	17,662	28,083	44,104
RE2014-04	1,489	2,294	3,316	938	2,238	4,687			
RE2014-05	1,124	1,720	2,467	923	2,202	4,612			
RE2014-06	2,230	3,418	4,902	832	1,985	4,158			
Fall 2014	4,956	7,680	11,176	3,541	8,447	17,691			
RE2015-01	187	346	594	1,059	2,527	5,292			
RE2015-02	589	983	1,580	1,165	2,780	5,822			
Spring 2015	776	1,329	2,174	2,224	5,306	11,114			
Year 7 Total	5,733	9,009	13,350	5,765	13,753	28,805			

Table 21: Year 7 pool and interstitial stranding estimates for Mountain Whitefish juveniles by assessed flow reduction, compared to current abundance estimation.

Reduction Event Number	Pool Stranding Estimates			Interstitial Stranding Estimates			Fall 2014 Abundance Estimate		
	Lower	Median	Upper	Lower	Median	Upper	Lower	Median	Upper
RE2014-03	164	379	809	113	249	491	12,915	20,782	32,326
RE2014-04	128	235	399	1,489	2,294	3,316			
RE2014-05	126	223	371	1,124	1,720	2,467			
RE2014-06	134	235	388	2,230	3,418	4,902			
Fall 2014	552	1,071	1,968	4,956	7,680	11,176			
RE2015-01	145	284	518	187	346	594			
RE2015-02	191	402	771	589	983	1,580			
Spring 2015	336	686	1,289	776	1,329	2,174			
Year 7 Total	888	1,757	3,257	5,733	9,009	13,350			

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 and 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 (Figure 6; Section 3.3 above). Interannual variability in discharge has also been reduced under post-WUP operations. Conversely, significant differences in total reduction magnitude and mean ramping rate between pre- and post-WUP operations were not identified, although pre-WUP ramping rates exhibited much higher variability.

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.

Based on the current state of knowledge, the flow reduction measures implemented under the ASPD are effective at reducing fish stranding. 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 7 assessments, ten different species were encountered (four sportfish and six 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

Estimates for the number of Rainbow Trout 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 ($p < 0.05$), with median fall stranding estimates approximately four 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. This change in density could be a result of the juveniles exhibiting different habitat preferences in overwintering habitats (i.e., deeper, higher velocity habitats that pose less of a stranding risk).

The current year's interstitial stranding estimates were considerably lower in comparison to estimates obtained in Years 5 and 6 of this program (Golder 2014 and 2015). This is due to both increase in number of data points and the introduction of transect measurements of interstitial stranding. Conversely, pool stranding estimates in the current year were significantly higher than estimates in Year 6 (Golder 2015). Wetted area, reduction magnitude, and ramping rates were very similar during Years 6 and 7 of the study, as were the abundance estimates for Rainbow Trout. This indicates that the higher total stranding estimates in Year 7 were not related to changes in DDM operations (flow reduction procedures) or overall fish densities.

Although the estimated numbers of interstitially stranded Rainbow Trout and Mountain Whitefish in the LDR are relatively high and the estimates are still uncertain, they are much more precise than the estimates obtained in previous years (Golder 2011, Golder and Poisson 2012, Golder 2015). Median estimates of total annual interstitially stranded Mountain Whitefish decreased approximately 2-fold in comparison to previous values, while interstitial stranding estimates for Rainbow Trout were 3 – 4 times lower than previously reported (Golder 2015). While interstitial stranding is likely to be biologically relevant, 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.

Similar to the findings in Year 6 (Golder 2015), a relationship between interstitially stranded fish counts and ramping rate ($p > 0.05$) in the current program was not found. This relationship should continue to be re-evaluated in the upcoming study years as more data are collected.

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 (at the 0.05 significance level) between interstitially stranded fish counts and slope in the current dataset were not found in Year 6 (Golder 2015). Similarly, a relationship between stranded fish counts and slope was not observed in the data in the current year.

In the current year, low-slope sites had somewhat elevated pool densities. Also, the variability in the number of fish stranded per pool was similar between low and high slope habitat. This indicated that slope was not a factor influencing pool stranding. Due to the high variability and low data volume, the effect of slope on stranding was not significant. This relationship should be re-evaluated as more data are collected.

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 a relationship between slope and the number of fish stranded in isolated pools was not identified (Golder 2015). The dichotomous high/low classification of slope habitat may be too vague to determine the effects of slope on both pool and interstitial stranding. Reclassifying the slope categories may assist in ascertaining its effect on fish stranding. This was proposed in Year 6 but was not conducted in the present study year due to budget limitations.

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 2015), 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, a significant statistical effect of index and random site on pool density was not found. The raw densities of pools sampled during 2006 – 2015 stranding surveys suggest that while random sites had several cases of high pool density (>2.5 pools/100 m^2), the majority of recorded pool densities were often lower than those recorded at index sites, consistent with coefficient estimates being higher for index sites than random sites. Since the lack of significance was marginal, as more data are accumulated, the difference between the two types of sites may become significantly different at the 95% level. Based on these analyses, index sites do not exhibit a significant bias toward higher stranding rates and therefore, hypothesis H_{01} cannot be rejected. Future study years should consider this when selecting sites for assessment, and 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*. The estimated fall 2010 population of Rainbow Trout juveniles within the LDR as modeled from DDMMON-2 data was 48,981 (95% credibility intervals range from 30,828 – 73,594; Thorley et al. 2012). The findings in Year 6 indicated a potential decline in the Rainbow Trout population from 2010 to 2013 (Golder 2015). With updated modelling protocols and an additional year of data, the updated abundance estimates for Years 6 and 7 were very similar. The confidence intervals of the estimates for these two sample years were also similar, indicating comparable levels of uncertainty. In comparison to the abundance estimates obtained in 2010, the current estimates were also substantially lower. This finding should be interpreted with caution as the models in the individual studies were different.

The sum of the estimated fall 2014 interstitial and pool stranded Rainbow Trout in the LDR had a median value of 22,762 and minimum and maximum 95% credibility intervals of 11,498 and 42,155 respectively. Estimates for the number of Rainbow Trout juveniles stranded in pools obtained for this program were relatively precise and low, while the uncertainty related to interstitial estimates continued to be high. In comparison to pool stranding in Year 6 (median: 2,832, minimum and maximum 95% credibility intervals of 1,777 and 42,155 respectively; Golder 2015), pool stranding estimates in the current year were significantly higher (median: 9,009, minimum and maximum 95% credibility intervals of 5,733 and 13.350, respectively). Based on the likely overestimated interstitial stranding estimates, combined with the precise pool estimates from the present dataset, hypothesis H_{02} cannot be reasonably rejected. Therefore, we must conclude 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 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. An ongoing management program (run jointly by the Fish and Wildlife Compensation Program and the Habitat Conservation Trust Fund) provides spring Rainbow Trout abundance estimates in both the Lardeau and lower Duncan rivers. The snorkel surveys from that program that are conducted to obtain abundance estimates occur in March of every calendar year (Andrusak 2010, 2013a and 2013b). As the timing of the abundance estimates provided occurs after the winter/spring fish stranding assessments, it is not possible to utilize those estimates in determining if spring stranding rates impact population levels of Rainbow Trout. The yearly results of this program should be examined to refine the abundance estimation methodology of this study and to monitor the comparability of the spring abundance estimates it provides.

4.2.5 Mountain Whitefish

The total abundance estimates for Mountain Whitefish obtained using the updated abundance model were similar between Years 6 and 7. This is as expected, since a significant year effect was not found for fish densities. Current abundance estimates for Mountain Whitefish were substantially lower than those obtained in 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 the present study year.

In both Year 6 (Golder 2015) and the current study year, a seasonal effect on Mountain Whitefish stranding was not observed. In the current year, only 3 stranded Mountain Whitefish were documented, and encounters have been minimal in all study years. <u>This consistently low level of stranding was not considered ecologically significant and will likely not result in a population level effect on Mountain Whitefish.</u> 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 7 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?):*
  - 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); although the current year's estimates were considerably lower in comparison to estimates obtained in Years 5 and 6 of this program (Golder 2014 and 2015)
  - Statistically significant relationships between interstitially stranded fish counts and slope in the current dataset were not found (Section 4.2.2)

- 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)

Substantial 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 will continue to decrease.

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 reducing the uncertainly around these estimates.

5.0 RECOMMENDATIONS

Recommendations for the final year (Year 7) 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) Adjust the site selection protocol to a random selection of all identified stranding sites. As a statistically significant difference between pool densities at index and non-index sites was not identified, there is no need to split the sites into two separate strata during the site selection process. Pool densities between the strata should continue to be examined, as annual analysis of the growing data set may reverse this finding.

3) 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 Year 8 to 10.

4) As the data analysis in the current year did not find a significant difference between fish stranding related to the dichotomous high/low slope definition, refine the slope bin classification. This will assist in determining which slopes pose the highest risk to strand fish and further of understanding the potential impacts of DDM operations on fish stranding.

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 the refinement of 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.

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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, PhD, RPBio
Senior Fisheries Biologist, Limnologist, Associate, Project Director

BH/DS/cmc

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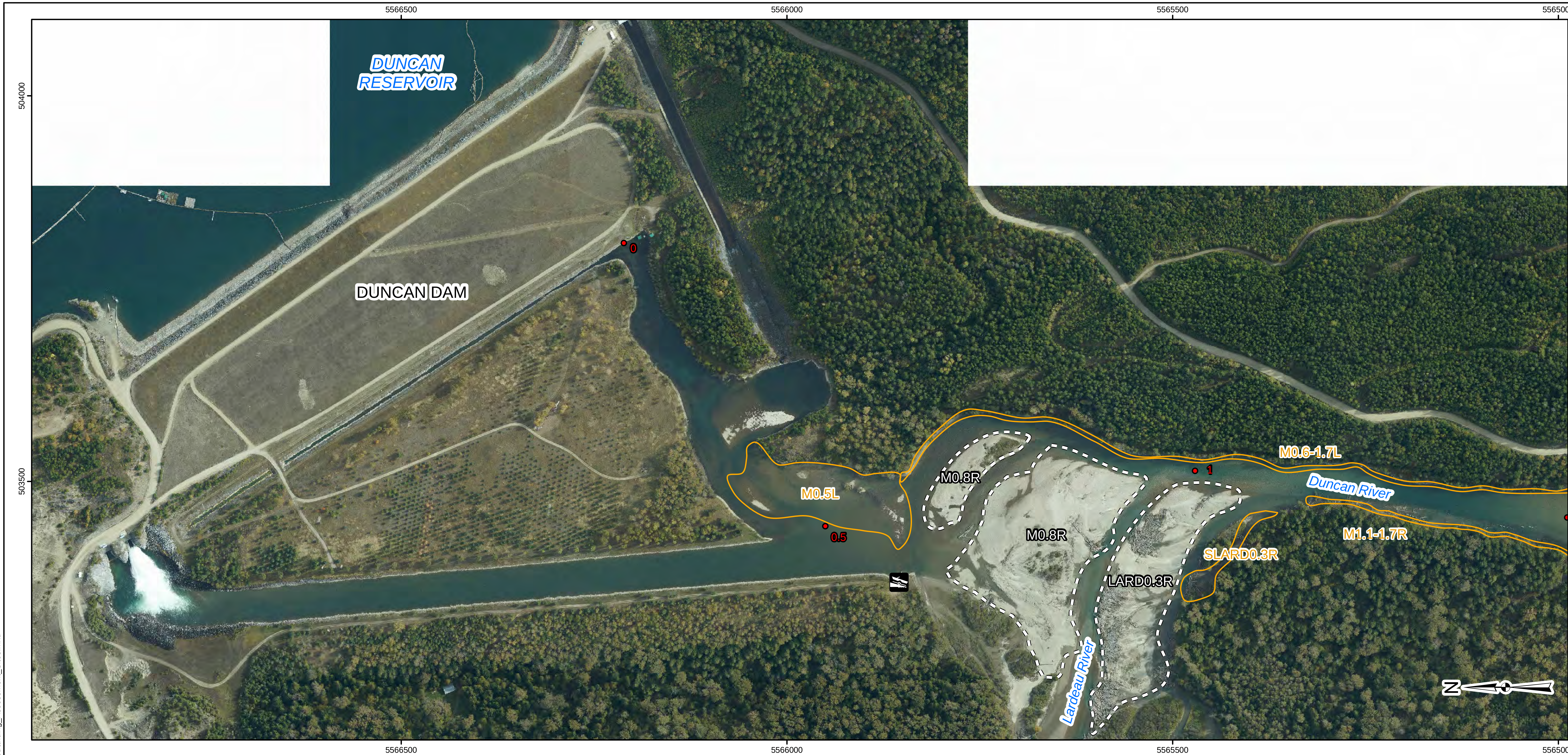
APPENDIX A

Sampling Chronology and Summary of Identified Stranding Sites



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

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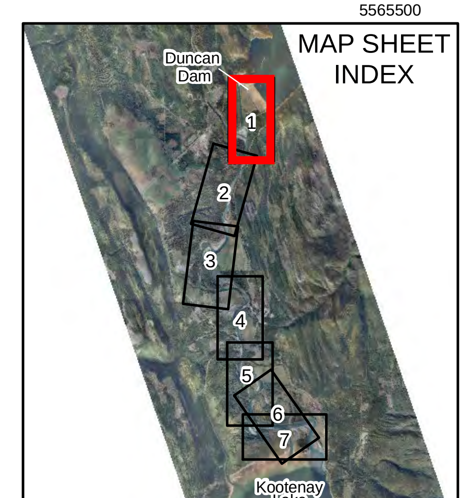

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PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

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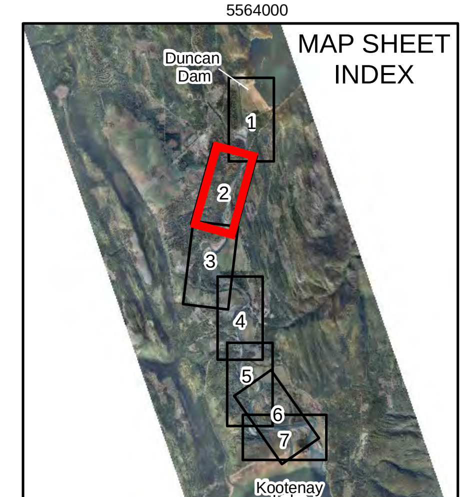

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PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

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STRANDING ASSESSMENT SITES

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

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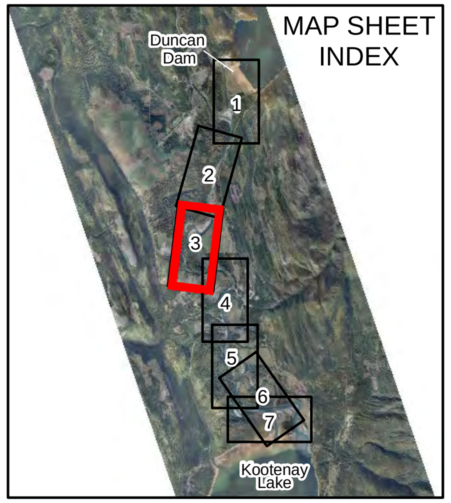

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PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

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- WSC GAUGE
- BOAT LAUNCH
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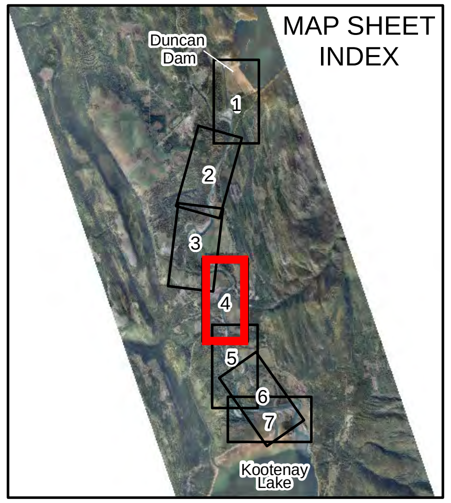

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PROJECT
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LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

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LEGEND

- RIVER KILOMETRE
- WSC GAUGE
- BOAT LAUNCH
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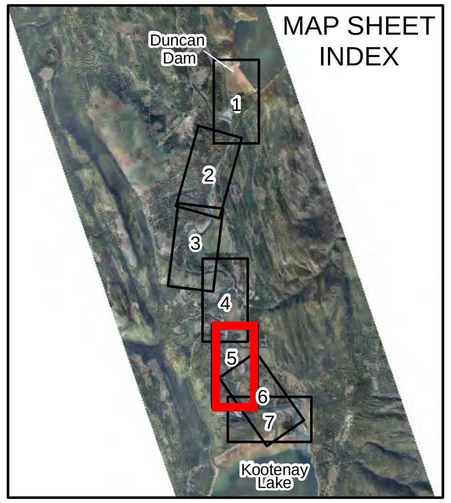

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STRANDING IMPACT MONITORING

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STRANDING ASSESSMENT SITES

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LEGEND

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

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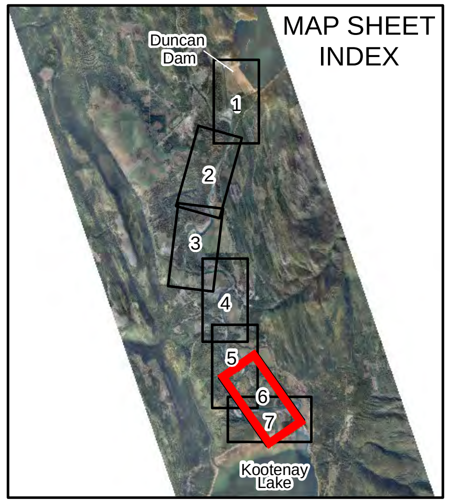

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LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

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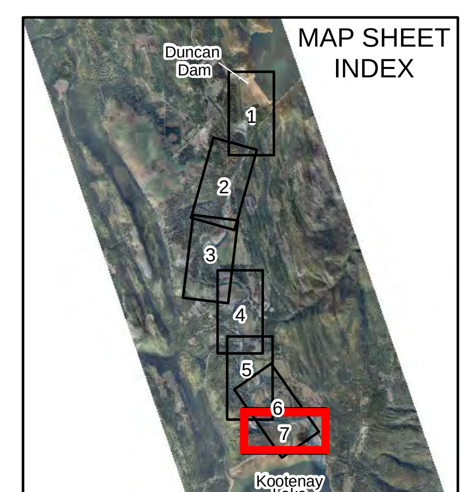

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LEGEND

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- BOAT LAUNCH
- STRANDING SITE - INDEX
- STRANDING SITE - NON-INDEX

REFERENCE

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TITLE	**LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES**

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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

APPENDIX B

Bayesian Models - Code

JAGS code for Bayesian analysis of Rainbow Trout density and abundance

```
model {
	sSite ~ dunif(0, 5)

	for(i in 1:4){
		bDepthYear[i] ~ dnorm(0, 0.01)
		  }

	for(k in 1:NSite){   # model the number of fish in each sampling site; random variable
		bSite[k] ~ dnorm(0, sSite^-2)
		LogitEfficiency[k] ~ dnorm(bEfficiency, sEfficiency^-2)
		logit(p[k]) <- LogitEfficiency[k]
		#logit(p[k]) <- bEfficiency

		log(mu[k]) <- bSite[SiteNum[k]] + bDepthYear[DepthYearNum[k]]

		Count[k] ~ dpois(p[k] * mu[k] * area[k])
		Uncounted[k] ~ dpois((1 - p[k]) * mu[k] * area[k])
		N[k] <- Count[k] + Uncounted[k]
			}

	for(i in 1:NStrata13){
		log(mutotal13[i]) <- bDepthYear[AreaStratum13DepthYearNum[i]]
		CountTotal13[i] ~ dpois(mutotal13[i]*AreaStratum13[i])
			}

	for(i in 1:NStrata14){
		log(mutotal14[i]) <- bDepthYear[AreaStratum14DepthYearNum[i]]
		CountTotal14[i] ~ dpois(mutotal14[i]*AreaStratum14[i])
			}
	}
```

JAGS code for Bayesian analysis of Mountain Whitefish density and abundance

```
model {
	sSite ~ dunif(0, 5)
	bIntercept ~ dnorm(0, 0.01)

	for(k in 1:NSite){   # model the number of fish in each sampling site; random variable
		bSite[k] ~ dnorm(0, sSite^-2)
		LogitEfficiency[k] ~ dnorm(bEfficiency, sEfficiency^-2)
		logit(p[k]) <- LogitEfficiency[k]
		#logit(p[k]) <- bEfficiency

		log(mu[k]) <- bIntercept + bSite[SiteNum[k]]
		Count[k] ~ dpois(p[k] * mu[k] * area[k])
		Uncounted[k] ~ dpois((1 - p[k]) * mu[k] * area[k])
		N[k] <- Count[k] + Uncounted[k]
			}

	logit(p_out) <- bEfficiency

	for(i in 1:NStrata13){
		log(mutotal13[i]) <- bIntercept
		CountTotal13[i] ~ dpois(mutotal13[i]*AreaStratum13[i])
			}

	for(i in 1:NStrata14){
		log(mutotal14[i]) <- bIntercept
		CountTotal14[i] ~ dpois(mutotal14[i]*AreaStratum14[i])
			}
	}
```

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

```
model{
        pIntercept ~ dnorm(0, 0.001)
        bIntercept ~ dnorm(0, 0.001)

        for(i in 1:nObs){
                log(mu.d[i]) <-  bIntercept # density per m2
                logit(mu.p[i]) <- pIntercept
                p[i] ~ dbern(mu.p[i])
                Fish[i] ~ dpois(mu.d[i]*p[i]*Area[i]) ## Number of fish counted in each area
                        }
        }
```

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

```
model {
## stranding model:
#RB
        for(i in 1:2){
                bSeason[i] ~ dnorm(0, 0.01)
                    }

        muEff[1] <- 10
        muEff[2] ~ dnorm(0, 0.01)
        logit(p[1]) <- muEff[1]
        logit(p[2]) <- muEff[2]
        sReduction ~  dunif(0, 5) #dgamma(0.1, 0.1)

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

        for(i in 1:NObs){
                log(mu[i]) <- bSeason[SeasonNum[i]] + bReduction[ReductionNum[i]]
                censor[i] ~ dinterval(N[i], MinFish[i])
                N[i] ~ dpois(mu[i])
                NPass[i, 1] <- N[i]

                for(pass in 1:3){ #MaxPass[i]
                        Pass[i, pass] ~ dbin(p[SamplingGearNum[i]], NPass[i, pass])
                        NPass[i, pass+1] <- NPass[i, pass] - Pass[i, pass]
                                        } #pass
                        } # i

#MW

        bIntercept_fish ~ dnorm(0, 0.01)
        muEff.MW[1] <- 10
        muEff.MW[2] ~ dnorm(0, 0.01)
        logit(p.MW[1]) <- muEff.MW[1]
        logit(p.MW[2]) <- muEff.MW[2]
        sReduction.MW ~  dunif(0, 5)

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

```
	for(i in 1:NObs.MW){
		log(mu.MW[i]) <- bIntercept_fish + bReduction.MW[ReductionNum.MW[i]]
		censor.MW[i] ~ dinterval(N.MW[i], MinFish.MW[i])
		N.MW[i] ~ dpois(mu.MW[i])
		NPass.MW[i, 1] <- N.MW[i]

		for(pass in 1:3){ #MaxPass[i]
			Pass.MW[i, pass] ~ dbin(p.MW[SamplingGearNum.MW[i]],
				NPass.MW[i, pass])
			NPass.MW[i, pass+1] <- NPass.MW[i, pass] - Pass.MW[i, pass]
					} #pass
			  } # i

## pool model:
	sSite ~ dgamma(0.1, 0.1)
	r ~ dgamma(0.1,0.1)
	bArea ~ dnorm(0, 0.01)
	bIntercept ~ dnorm(0, 0.01)

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

	for(i in 1:nObs_pool){
		u[i] ~ dgamma(r,r)
		log_mu_pool[i] <- bIntercept + bSiteName[SiteName[i]] + bArea*SiteArea[i]
		log(mu_pool[i]) <- log(log_mu_pool[i])
		NumPoolsPresent[i] ~ dpois(mu_pool[i]*u[i])
			}

## DERIVED
	for(i in 1:nReductions){
		log(muReduction[i]) <- bSeason[ReductionSeasonNum[i]] +
			 bReduction[Reduction_ReductionNum[i]]
		log(muReduction.MW[i]) <- bIntercept_fish +
			bReduction.MW[Reduction_ReductionNum[i]]

		High[i] <- bIntercept + bArea*(HighSlope*Drop[i]/1000)
		Low[i] <- bIntercept + bArea*(LowSlope*Drop[i]/1000)
		Total_pools[i] <- High[i] + Low[i]
		Total[i] <- muReduction[i]*Total_pools[i]
		Total.MW[i] <- muReduction.MW[i]*Total_pools[i]
				}
	} #model
```

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

Plate 4 Garbage encountered at site S9.2L on May 22, 2014.

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