Duncan Dam Project Water Use Plan

Adaptive Stranding Protocol Development Program

Implementation Year 6

Reference: DDMMON-16

Lower Duncan River: Fish Stranding Impact Monitoring: Year 6 Data Report

Study Period: April 2013 to April 2014

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July 3, 2015

July 3, 2015

DDMMON-16: LOWER DUNCAN RIVER

Lower Duncan River Fish Stranding Impact Monitoring: Year 6 Report (April 2013 to April 2014)

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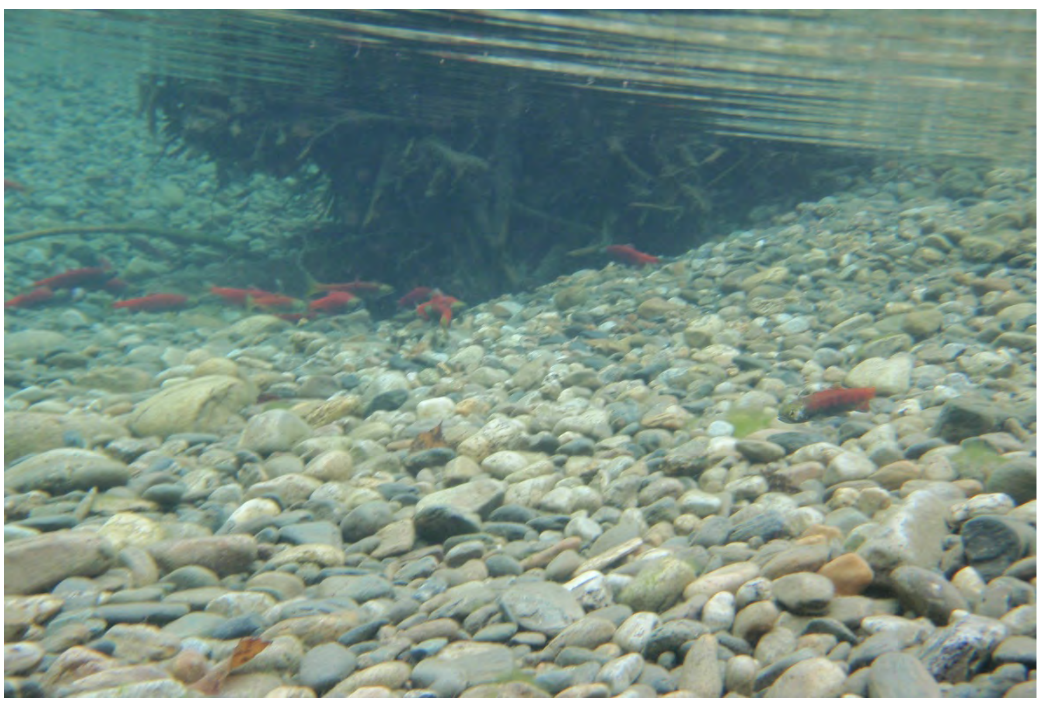

Report **Number:** 12-1492-0117

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REPORT

Cover Photo: Underwater view of Kokanee spawners stranded in a pool isolated from the mainstem Duncan River at Site S3.5-4.0R, October 1, 2011.

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

Keywords: Duncan River, Lardeau River, Duncan Dam, Water License Requirements, Duncan River Water Use Plan, fish stranding, juvenile population estimate, flow reduction, flow ramping, stranding mechanism.

Executive Summary

Although natural flow fluctuations from unregulated tributaries are known to cause fish stranding in the lower Duncan River (LDR), it can be exacerbated by Duncan Dam (DDM) operations influencing the frequency and magnitude of flow fluctuations. The Duncan Dam Water Use Plan process identified fish stranding as a high priority for resolution, and recommended a series of studies and activities to support the improvement of DDM operations to reduce stranding. The BC Hydro Water License Requirements (WLR) Program includes the continuation of the DDMMON-16 Lower Duncan River Fish Stranding Impact Monitoring Program to inform the effectiveness of fish stranding mitigation measures.

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 Plane (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 increase the risk of stranding in certain seasons and during periods of longer wetted histories. Based on results up to April 2014 documented low stranding rates of juvenile Mountain Whitefish (*Prosopium williamsoni*) are not believed to result in population level effects, while the current interstitial stranding estimation for juvenile Rainbow Trout (*Oncorhynchus mykiss*) is too uncertain to inform a confident total stranding estimate.

This report presents the results from Year 6 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, high uncertainty of abundance and interstitial stranding estimates, and the many variables that could potentially contribute to stranding, these results should be treated with caution as they are somewhat sensitive to assumptions.

Table EI: DDMMON-16 Year 6: Status of Management Questions and Objectives.

DDMMON-16 Management Question	DDMMON-16 Specific Hypothesis	DDMMON-16 Year 6 (2013-2014) Status Summary
1) How effective are the operating measures implemented as part of the ASPD program?	N/A	- Based on the current state of knowledge, the flow reduction measures implemented under the WUP are effective at reducing fish stranding. - When possible, flow reductions at DDM should follow recommendations made by the DDMMON-15 Lower Duncan River Stranding Protocol Development and Finalization Program. - Variables related to stranding that are currently outstanding in the Adaptive Stranding Protocol Development Program (ASPD) are wetted history and season.

DDMMON-16 Management Question	DDMMON-16 Specific Hypothesis	DDMMON-16 Year 6 (2013-2014) 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 to focus on sites believed to have the highest amounts of stranding based on amount 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 Year 4 analysis, 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. Stranding rates per lineal distance do not differ between index and non-index sites, but differ due to greater dewatered area within index sites. Therefore, the greater area dewatered in index sites strands higher numbers of fish in comparison to non-index sites. Index sites appear to provide an estimate that is biased high. Therefore, hypothesis Ho_1 is rejected. - This will be re-examined in the Year 7 in-depth interpretive report with all available project data to determine if complete dataset supports the rejection of hypothesis Ho_1. The Year 7 summary report will also examine site selection for future years of this program and will make recommendations on selecting representative sites during mitigation activities.
	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 very low precision. There was a seasonal component to pool stranding, with higher stranding in fall, but at this point it cannot be determined whether this was due to less fish in the system in the spring vs. the fall or to a decreased risk of stranding. - The most recent fall season abundance estimates for the Rainbow Trout juveniles in the LDR suggest a decline in the population from 2010 to present. - 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. - Contrasting the findings of previous study years, within the current dataset relationships between pool and interstitially stranded fish and slope were not found. - Until the interstitial stranding estimation methodology is improved, we cannot reasonably reject this hypothesis

Acknowledgements

Special thanks are extended to Margo Dennis (BC Hydro, Burnaby) for support, advice, and assistance. The following **BC HYDRO** personnel are also gratefully acknowledged for their contributions of information and assistance during this monitoring program.

Alf Leake, Burnaby
James Baxter, Castlegar
Len Wiens, Duncan Dam
Kris Wiens, Duncan Dam
Ron Greenlaw, Duncan Dam

The following employees of **GOLDER ASSOCIATES LTD.** contributed to the collection of data and the preparation of this report.

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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) that influence 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 will optimize the stranding assessment procedure and measure 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 reductions that minimizes impacts on fish.

The fish stranding impact monitoring program conducted this year (Year 6) builds on previous assessment methodologies, expands the program's data sets, updates the boundaries of identified sites where stranding occurs, and analyzes pre-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.

The following document provides information on fish stranding observed over all assessed flow reductions from the timing of the last report on January 20, 2013 (Golder 2014) to April 14, 2014. This report presents detailed statistical analysis in relation to the multi-year program objectives, and also incorporates several aspects of the updated DDMMON-3 hydraulic model, including the Digital Elevation Model (DEM) and runs of the model to select sample locations.

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.

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), and the present study year (Year 6; 2013 - 2014) 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 5 (2010 – 2011, 2011 – 2012, and 2012 - 2013, 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) were implemented in the present study year.

1.4.1 Site Selection

In previous study years, 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 during the present study 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 in Year 5 and the present study 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 will remove 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 in Year 4 and Year 5, as well as in the present study 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 ensure sampling robustness while addressing logistic difficulties. To sample a similar length of river as in previous reports (5,700 and 3,600 m in fall 2010 and spring 2012 surveys, respectively), 20 sites were selected, approximately 250 m each. These longer sites, in comparison to the 2010 survey, allowed a reduction in the number of sites sampled and therefore reducing travel time and associated logistics, while still maintaining random and robust coverage of the Duncan system.

1.4.5 Data Analysis

At the time this draft document was prepared, communications were underway with the DDMMON-3 study team to determine the usability of the updated TELEMAC 2D hydraulic model outputs for this program. If deemed feasible, updated wetted areas at stranding locations predicted by the model at various flow elevations may be used to provide the basis for extrapolation of stranding rates defined in this study.

2.0 METHODS

2.1 Study Area

The geographic scope of the study area for the FSIMP was the 11 km of mainstem LDR from DDM to the mouth of Kootenay Lake (Figure 1). This study area (collectively known as the LDR) includes the Duncan-Lardeau rivers confluence, as well as the Meadow, Hamill and Cooper Creek mouths. For the purpose of all WLR studies, the mainstem Duncan River was divided into five sections; these were termed Reach 1 (RKm 0.0 – at DDM spill gates to RKm 0.8), Reach 2 (RKm 0.8 to RKm 2.6), Reach 3 (RKm 2.6 to RKm 5.7), Reach 4 (RKm 5.7 to RKm 6.7), and Reach 5 (RKm 6.7 to RKm 11.0 – at the mouth to Kootenay Lake).

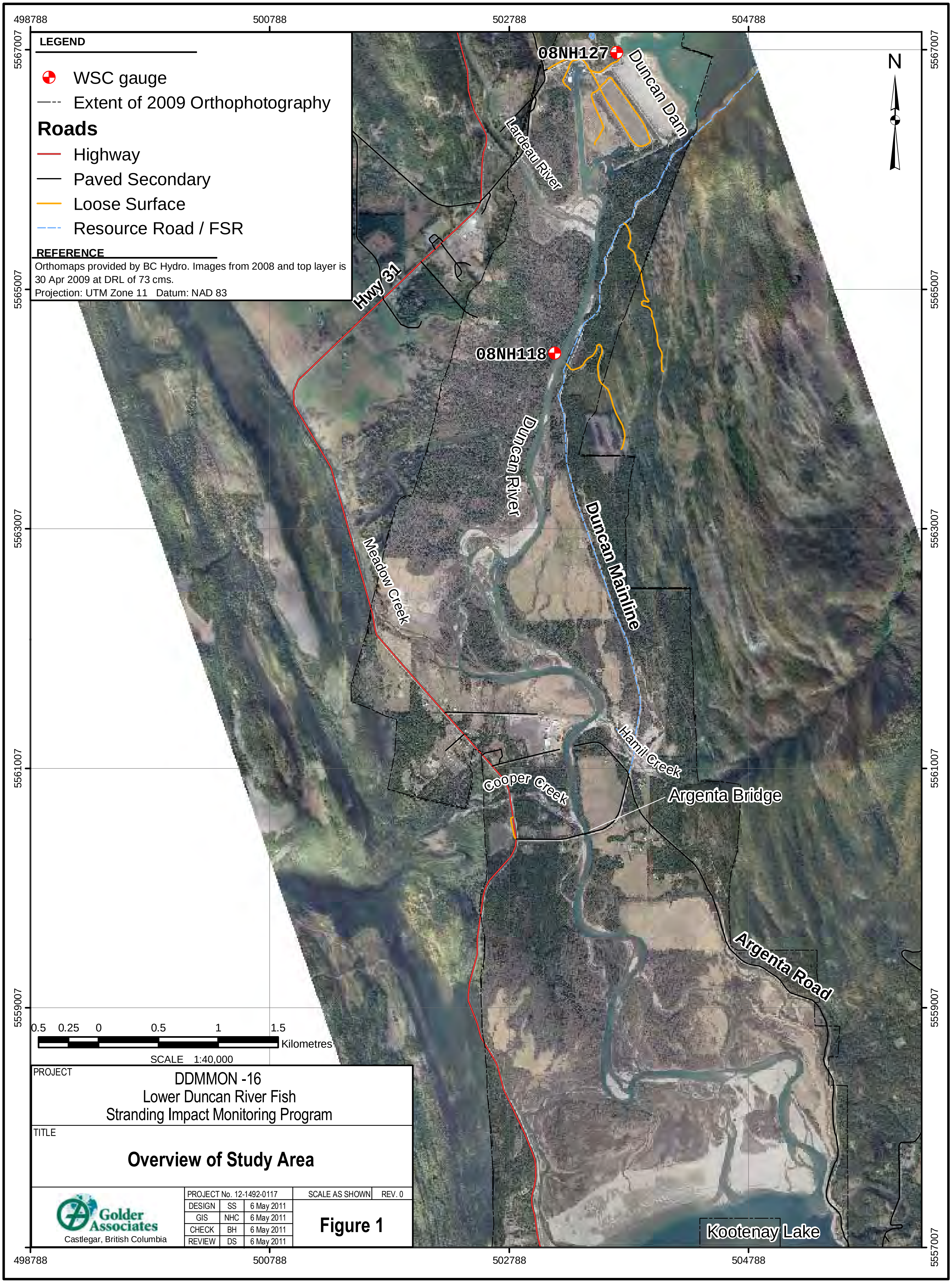
For the purpose of this study, 50 potential fish stranding sites were identified based on previous studies (AMEC 2004 and Golder 2006, 2008, 2009b, 2010, 2011, and 2014; Golder and Poisson 2012). These stranding sites included 11 index stranding assessment sites and 39 non-index sites (Appendix B, Figures 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 for each year was set between April 15 of that year, and continued until the following April 14. Stranding assessment activities in the present study year were conducted from September 21, 2013 to March 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 6.

Table 1: 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	2013-03	-	3	4
September 24, 2013	Stranding Assessments	2013-04	-	2	2
September 27, 2013	Stranding Assessments	2013-05	-	2	4
January 21, 2014	Stranding Assessments	2014-01	-	4	4
March 1, 2014	Stranding Assessments	2014-02	-	2	2



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 Power Records.

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 RIVER2D 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. 3.0.2 (R Development Core Team, 2013) using the package spsurvey (Kincaid and Olsen 2013). Sites were not be 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 seven main and 14 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. The sites that were to be sampled were marked using flagging tape at their upstream and downstream boundaries. Field conditions were not always as predicted by the 2D River Model, rendering some pre-selected site 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 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 three snorkelers surveyed each site; while at narrow sites two or even 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. Since very little area was classified as stratum 3 (Deep/Slack water), the stratum was removed from the analysis. The analysis was implemented using the statistical environment R, v. 3.0.2 (R Development Core Team, 2013), 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
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). In the full model, density was stratified by depth/flow, i.e., modeled to vary across depth/flow strata, as well as by river reach (Figure 1), and was allowed to randomly vary by site. Once the full model was constructed, it was examined for parameter significance (at the 0.05 level of significance) and convergence. It was then reduced to the most parsimonious model, which contained only a fixed effect of depth and site-specific random effects. 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/m^2) and the total area of each depth/flow stratum, 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 A.

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
p	Observer efficiency
NSite	Number of sampled sites
bSite[k]	The random effect of the *k*-th site on fish density
bArea	The effect of area on observed fish abundance
Area[k]	The area of the *k*-th site
mu [k]	Expected fish abundance at the *k*-th site
SiteNum[k]	Numeric representation of site name of the *k*-th site
N[k]	Number of fish at the *k*-th site
min_pop[k]	The minimum known number of fish at the *k*-th site
Count[k]	The observed number of fish at the *k*-th site
NStrata	The number of depth/flow strata
Mutotal[i]	Expected fish abundance at the *i*-th stratum
AreaStratum[i]	Area of the *i*-th stratum

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

Variable/parameter	Description
sSite	dgamma(0.1, 0.1)
bSite[k]	dnorm(0, sSite^-2)
bArea	dnorm(0, 0.01)

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

Variable/parameter	Dependency
mu[k]	bSite[SiteNum[k]] + bArea*Area[k]
N[k]	dpois(mu[k]*area[k]) T(min_pop[k],)
Count[k]	dbin(p, round(N[k]))
mutotal[i]	bArea*AreaStratum[i]

Median values of density and abundance and 95% credibility intervals were calculated in R. 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 5,000 runs of three MCMC chains of 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).

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 (Appendix C, Plate 1), 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 lead 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 6 Stranding Site Selection

Utilizing the methodology developed in Year 4 (Golder and Poisson 2012), 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 6 Sampling

2.5.2.1 Isolated Pools

Following the methodology used in Year 4, once sampling commenced, 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); and,

2) Moderate to High complexity (>10% total cover).

Pools with 0% – 10% cover were classified at Zero to Low complexity if surface area was 5 m^2 or less. Zero to Low Complexity pools are generally smaller in size so that fish could be captured readily by backpack electrofishing. Moderate to High Complexity pools are likely to have: larger surface areas, larger substrate that could provide cover to fish including larger cobble and gravel or boulder, and some portions of the pool that are not visible because of woody debris or other cover types.

For each pool, associated cover types (and percentages within the pool) were recorded from the following list:

- Large woody debris (woody debris with diameter of >10 cm);
- Small woody debris (woody debris with diameter of <10 cm);
- Aquatic vegetation;
- Submerged Terrestrial Vegetation (Appendix C, Plate 2);
- Overhanging vegetation (Appendix C, Plate 3);
- Organic debris (leaves, bark etc.);
- Cut bank;
- Shallow pool;
- Deep pool; and,
- Other (metal, garbage, etc.).

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

2.5.2.2 Dried Pool

As a result of recommendations made in Year 4, a third class of stranding mechanism, dried pools, was incorporated into the project methodology. 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 was 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 pool at time of isolation from the mainstem river.

2.5.2.3 Interstitial Sampling

Dewatered habitat at each site was assessed by conducting a maximum of 20 randomly placed grids (each grid has area of 0.5 m^2). The substrate and all cover were removed from each grid, and the stranded fish enumerated (Appendix C, Plate 4). 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 6 DDM and DRL flow data were added to the discharge data set. The modelling conducted in Year 4 (Golder and Poisson 2012) was 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. 3.0.2 (R Development Core Team 2013), 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. The posterior distributions, which were estimated using Gibbs sampling (Kery 2010), were derived from 9,500 Markov Chain Monte Carlo (MCMC) simulations, and thinned from 95,000 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).

2.5.3.3 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 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 6). To estimate the total number of pools that form throughout the study area, mean expected pool counts were multiplied by total exposed area using GIS-derived low-slope (0-4%) and high-slope (>4%) dewatered areas for each stranding event.

The number of fish captured in multi-pass electrofishing was used to estimate fish catchability. Catchability was assumed to be 100% when using either visual or dip-net sampling. Single-pass pool abundance was estimated using the number of fish captured during the pass and the catchability estimated in multi-pass sampling. Fish abundance was assumed to vary with season (fixed effect) and reduction (random effect). Season was defined as “spring” for January-July months and as “fall” for August-October. Once season and reduction parameters were estimated (see Table 6 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 6). 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 7). The complete model specification used is shown in Table 7 and Table 8, and model code is provided in Appendix A.

Table 6: Variables and parameters in the Bayesian analysis of pool density.

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
Area[i]	The dewatered area at the *i*-th case
bIntercept	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
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
muEff[j]	Logistic catchability using the *j*-th sampling gear, where j = 1 for visual and dip-net, and j = 2 for electrofishing
P[j]	Catchability using the *j*-th sampling gear, where j = 1 for visual and dip-net, and j = 2 for electrofishing
sReduction	Standard deviation of the effect of reduction on expected fish counts per pool
bReduction[r]	The random effect of the *r*-th reduction on expected fish counts per pool
NObs	Number of pool stranding data points
Mu[k]	Expected fish counts in the *k*-th pool
SeasonNum[k]	Season during which the *k*-th pool was sampled
ReductionNum[k]	Reduction during which the *k*-th pool was sampled
censor[k]	A variable used to truncate fish counts for the *k*-th pool, so that estimated fish numbers are not lower than those observed
N[k]	Estimated fish counts at the *k*-th pool
MinFish[k]	Observed number of fish at the *k*-th pool
NPass[k, p]	Estimated number of fish present at the *k*-th pool prior to the *p*-th pass
Pass[k, p]	Sampled number of fish at the *k*-th pool prior to the *p*-th pass
SamplingGearNum[g]	Sampling gear used at the *k*-th pool, where g = 1 stands for visual or dip-net, and g = 1 stands for electrofishing
nReductions	Number of reductions
muReduction[r]	Expected number of fish stranded per pool at the *r*-th reduction
High[r]	Exposed high-slope (>4%) area at the *r*-th reduction
Low[r]	Exposed low-slope (0-4%) area at the *r*-th reduction
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
Total[r]	Estimated number of fish stranded in pools at the *r*-th reduction

Table 7: Prior probability distributions in the Bayesian analysis of pool density.

Variable/parameter	Description
sSite	dgamma(0.1,0.1)
r	dgamma(0.1,0.1)
bArea	dnorm(0, 0.01)
bIntercept	dnorm(0, 0.01)
bSiteName[j]	dnorm(0, sSite^$^{-2}$)
u[i]	dgamma(r, r)
bSeason[i]	dnorm(0, 0.25)
muEff[2]	dnorm(0, 0.01)
sReduction	dunif(0, 5)

Table 8: Dependencies between variables and parameters in the Bayesian analysis of pool density.

Variable/parameter	Dependency
mu[i]	bIntercept + bSiteName[SiteName[i]] + bArea*SiteArea[i]
NumPoolsPresent[i]	dpois(mu[i]*u[i])
muEff[1]	10
bReduction[r]	dnorm(0, sReduction^-2)
log(mu[k])	bSeason[SeasonNum[k]] + bReduction[ReductionNum[k]]
censor[k]	dinterval(N[k], MinFish[k])
N[k]	dpois(mu[k])
NPass[k, 1]	N[k]
Pass[k, p]	dbin(p[SamplingGearNum[k]], NPass[k, p])
NPass[i, pass+1]	NPass[k, p] - Pass[k, p]
log(muReduction[r])	bSeason[ReductionSeasonNum[r]] +
High[r]	bIntercept + bArea*(HighSlope*Drop[r])
Low[r]	bIntercept + bArea*(LowSlope*Drop[r])
Total_pools[r]	High[r] + Low[r]
Total[r]	muReduction[r]*Total_pools[r]

2.5.3.4 Interstitial Stranding

In the Bayesian model of interstitial stranding, the number of fish stranded in each sampled 0.5 m^2 grid was defined as Poisson-distributed, with a mean expected value determined by fish abundance and the probability of stranding at each grid (Table 9). 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 (20.5 m^2) calculated using data collected since September 2006. A separate analysis was conducted for Rainbow Trout and Mountain Whitefish. The prior distributions for all parameters were vague or uninformative to avoid biasing estimates

(Table 10). The complete model specification used is shown in Table 10 and Table 11, and model code is provided in Appendix A.

Table 9: 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
nObs	Number of data points
Mu.d[i]	Expected number of fish in the *i*-th 0.5 m^2 grid
Mu.p[i]	Stranding probability in the *i*-th 0.5 m^2 grid
P[i]	Whether a fish was stranded in the *i*-th 0.5 m^2 grid
Fish[i]	Number of fish observed in the *i*-th 0.5 m^2 grid

Table 10: Prior probability distributions in the Bayesian analysis of interstitial stranding.

Variable/parameter	Description
pIntercept	dnorm(0, 5^-2)
bIntercept	dnorm(0, 5^-2)

Table 11: Dependencies between variables and parameters in the Bayesian analysis of interstitial stranding.

Variable/parameter	Dependency
mu.d[i]	bIntercept
logit(mu.p[i])	pIntercept
P[i]	dbern(mu.p[i])
Fish[i]	dpois(mu.d[i]*p[i])

2.6 Duncan Stranding Database and Data Management

The MS Access database (referred to as the LDR stranding database) created in Year 2 was populated with all available stranding data collected during the present study year. The database underwent several refinements during the analysis to facilitate data entry and queries. Presently, 66 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 86.4 m^3/s on April 25, 2013 to 386.6 m^3/s on July 7, 2013. Hourly discharge from DDM ranged from 0 m^3/s on several days between early June and early July 2013, to 200.4 m^3/s on August 14, 2013 (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.

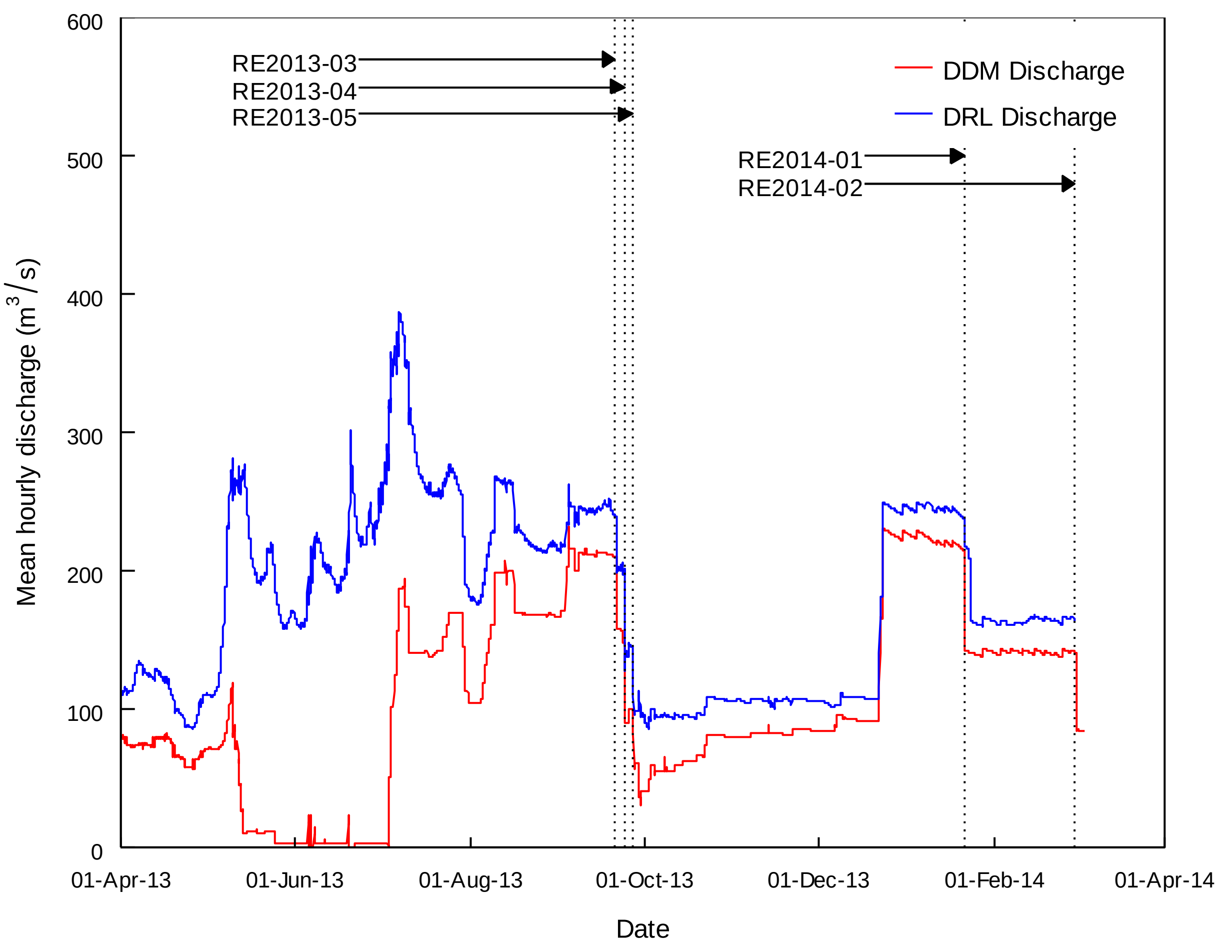


Figure 2: Hourly discharge at the Duncan Dam (DDM, red line) and at the Duncan River below the Lardeau River (DRL, blue line) from April 1, 2013 to April 1, 2014. Vertical dotted lines represent the timing of fish stranding assessments.

During the present study, five reduction events occurred at DDM where stranding could be assessed (Figure 2 and Table 12). During the reduction events, DDM decreases of discharge ranged between 50 and 78 m^3/s (Table 12). These decreases represent the discharge reductions at DDM, rather than flow changes at particular downstream fish stranding sites.

Table 12: Summary of DDM flow reduction events, from Sep 21, 2013 to March 1, 2014, for those events when fish stranding assessments were conducted.

Date	Reduction Event	DDM Discharge (m^3/s)			Ramping Description[a]	Flow Reduction Rationale
		Initial	Resulting	Reduction		
Sep 21, 2013	RE 2013-03	212	162	50	Down 6.25 m^3/s every 15 minutes from 07:00 to 08:45.	Onset of Kokanee protection flows.
Sep 24, 2013	RE 2013-04	170	120	50	Down 6.25 m^3/s every 15 minutes from 07:00 to 08:45.	Kokanee protection flows.
Sep 27, 2013	RE 2013-05	120	70	50	Down 6.25 m^3/s every 15 minutes from 07:00 to 08:45.	Final transition to Kokanee protection flows.
Jan 21, 2014	RE 2014-01	220	142	78	Down 6 m^3/s every 15 minutes from 06:00 to 08:45.	Discharge reduced to meet reservoir targets.
Mar 01, 2014	RE 2014-02	142	85	57	Down 6 m^3/s every 15 minutes from 06:00 to 08:00.	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 13.

Table 13: Scientific names of fish species encountered during fish stranding assessments on the lower Duncan River, September 2006 to March 2014.

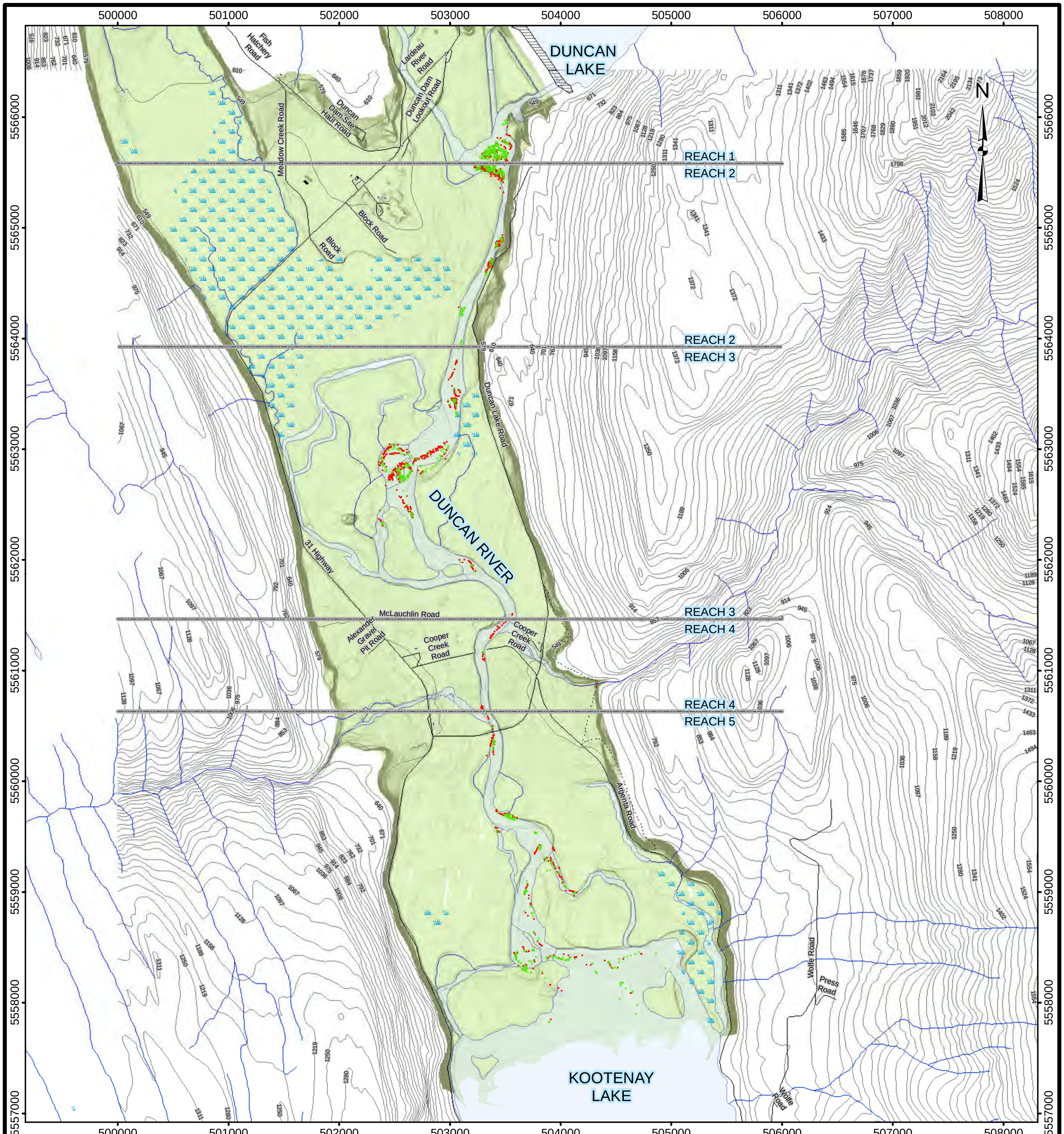
Category	Species	Scientific Name	Species Code[a]
Sportfish	Rainbow Trout	*Oncorhynchus mykiss*	*RB*
	Bull Trout	*Salvelinus confluentus*	*BT*
	Mountain Whitefish	*Prosopium williamsoni*	*MW*
	Pygmy Whitefish	*Prosopium coulteri*	*PW*
	Kokanee	*Oncorhynchus nerka*	*KO*
	Burbot	*Lota lota*	*BB*
Non-sportfish	Longnose Dace	*Rhinichthys cataractae*	*LNC*
	Dace spp.	*Cottus 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*
	Lake Chub	*Couesius plumbeus*	*LKC*

[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 14). 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 sites sampled in the present study year. The number of pools sampled in the present year was also reduced to allow for more intensive interstitial sampling effort. This resulted in the lowest number of pools sampled to date (n = 56), and the third most number of interstitial grids (n = 325) assessed in a single study year (Table 14). The locations of all sampled stranding mechanisms within the dataset is presented in Figure 3.

Table 14: Sampling effort during reductions of each study year that were included in the present analysis.

Study Year	Number of Reductions Assessed	Number of Assessments at Index Sites	Number of Assessments at Non-Index Sites	Total Number of Pools Sampled	Total Number of Interstitial Grids Conducted
2006-2007	2	16	0	144	15
2007-2008	7	56	0	346	40
2008-2009	8	42	0	233	34
2009-2010	6	33	14	221	40
2010-2011	7	50	22	346	96
2011-2012	7	29	21	92	411
2012-2013	7	20	18	78	331
2013-2014	5	13	16	56	325



LEGEND

REACH BREAK

STRANDING SAMPLE POINT

- > 4%
- 0-4%

SLOPE (%)

HIGH : 67

LOW : 0

WETLAND

WATERCOURSE

CONTOUR (30M INTERVAL)

WATERBODY

1,000 0 1,000

Metres

SCALE 1:50,000

REFERENCE

GROUND SURFACE: TDUNC3 CONVERTED INTO RASTER, PROVIDED BY CLIENT.
CONTOUR, ROAD, WATERCOURSE, WETLAND, WATERBODY: CANVEC.CA
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. 2014)

Golder Associates

PROJECT		12-1492-0117	FILE No.	
DESIGN	BH	14 Apr. 2014	SCALE AS SHOWN	REV. 0
GIS	JG	14 Apr. 2014	**FIGURE: 3**	
CHECK	BH	3 Jul. 2015		
REVIEW	DS	3 Jul. 2015		

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In the present year of study (2013 - 2014), a total of 600 fish were observed, representing 8 species [three sportfish and five non-sportfish species (Table 15)]. Juvenile Rainbow Trout were the most abundant sportfish, followed by juvenile Mountain Whitefish, accounting for 40.2 and 8.2% of the total number of fish, respectively (Table 15, Figure 4). The most common non-sportfish taxa were Longnose Dace and Sculpin spp., accounting for 37.8 and 7.7% of the total number of observed fish, respectively.

Table 15: 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 2014.

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
Sport Fish	Rainbow Trout	Adult	0 (0)	0 (0)	0 (0)	1 (0.1)	0 (0)	0 (0)	0 (0)	1 (0.2)
		Juvenile	130 (37.1)	278 (11.5)	530 (33.2)	113 (12.3)	343 (25.2)	419 (26.0)	332 (37.1)	241 (40.2)
	Bull Trout	Adult	0 (0)	0 (0)	0 (0)	4 (0.4)	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)
	Mountain Whitefish	Adult	0 (0)	1 (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)	43 (2.7)	6 (0.7)	49 (8.2)
	Pygmy Whitefish	Adult	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)
	Kokanee	Adult	0 (0)	97 (4.0)	572 (35.8)	112 (12.2)	42 (3.1)	55 (3.4)	111 (12.4)	0 (0)
		Y-O-Y	0 (0)	1695 (70.4)	85 (5.3)	109 (11.9)	83 (6.1)	844 (52.5)	257 (28.7)	0 (0)
	Burbot	Adult	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)
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)
	Dace spp.		0 (0)	0 (0)	0 (0)	12 (1.3)	1 (0.1)	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)
	Torrent Sculpin		0 (0)	1 (0)	1 (0.1)	0 (0)	0 (0)	3 (0.2)	0 (0)	0 (0)
	Prickly Sculpin		0 (0)	0 (0)	0 (0)	0 (0)	2 (0.1)	0 (0)	0 (0)	0 (0)
	Sculpin spp.		23 (6.6)	16 (0.7)	65 (4.1)	62 (6.8)	165 (12.1)	80 (5.0)	130 (14.5)	46 (7.7)
	Sucker spp.		2 (0.6)	4 (0.2)	26 (1.6)	166 (18.1)	54 (4.0)	9 (0.6)	16 (1.8)	32 (5.3)
	Redside Shiner		0 (0)	112 (4.6)	8 (0.5)	15 (1.6)	0 (0)	0 (0)	7 (0.8)	0 (0)
	Northern Pikeminnow		0 (0)	0 (0)	2 (0.1)	0 (0)	15 (1.1)	4 (0.2)	1 (0.1)	1 (0.2)
	Peamouth Chub		0 (0)	0 (0)	6 (0.4)	6 (0.7)	0 (0)	0 (0)	0 (0)	0 (0)
	Lake Chub		0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
	Unidentified		75 (21.4)	20 (0.8)	105 (6.6)	4 (0.4)	13 (1.0)	114 (7.1)	0 (0)	0 (0)
All Species Total			**350**	**2409**	**1596**	**918**	**1361**	**1607**	**896**	**600**

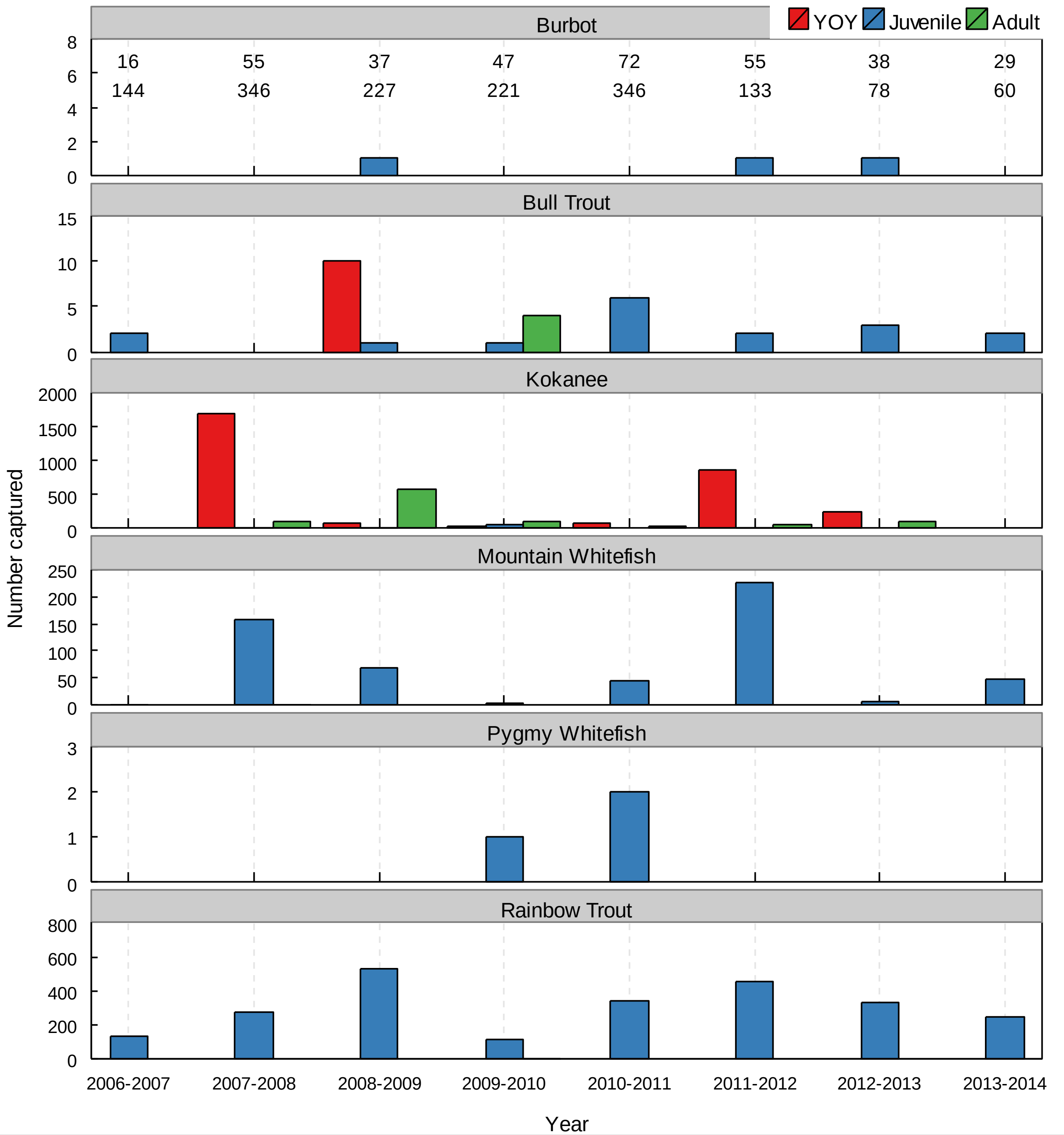


Figure 4: Abundances of sportfish species, separated by life stage, observed in stranding assessments between 2006 and 2014. Note the different y-axis scales among panels. On the uppermost panel, the numbers of sampled sites and pools are provided in the first and second lines, respectively.

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 13.9 km² during the post-WUP operational regime (Figure 5). 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 5). 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 5).

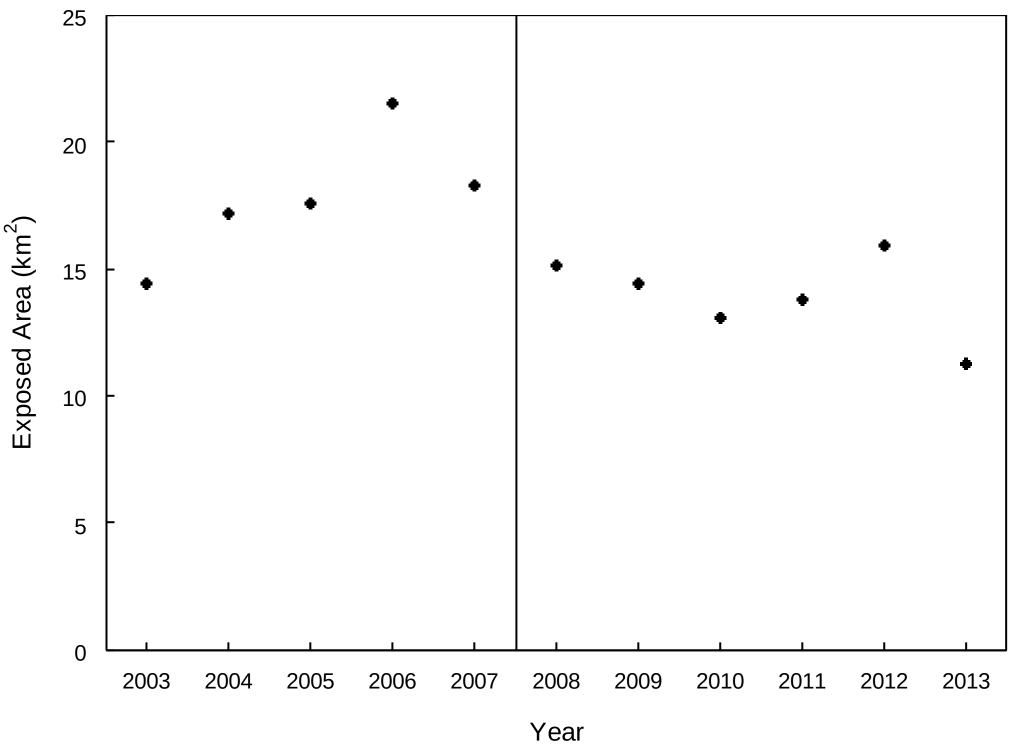


Figure 5: Total area exposed by all annual reductions in the LDR by year of operations, calculated using DRL discharge (black) or DDM only (red). The vertical line denotes the beginning on WUP flows in 2008.

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 6). 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 decrease pre-WUP to a gradual increase post-WUP (Figure 6).

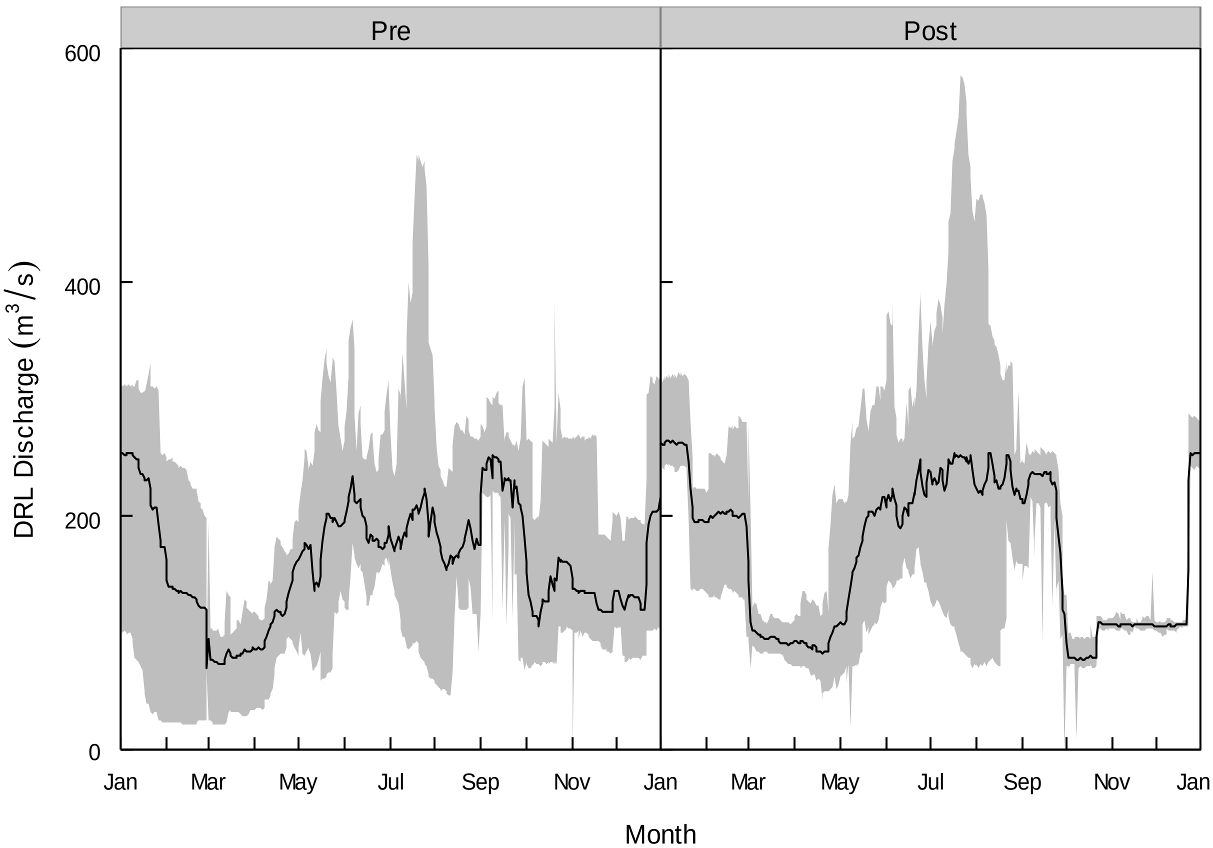


Figure 6: Minimum, maximum (grey ribbon) and mean (black line) discharge as measured at the WSC DRL gauge in the LDR by month during pre-WUP operations (2002 - 2007) and post-WUP operational implementation (2008 - 2014).

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 7). Substantial differences in the total reduction magnitude between pre- and post-WUP operations were not identified.

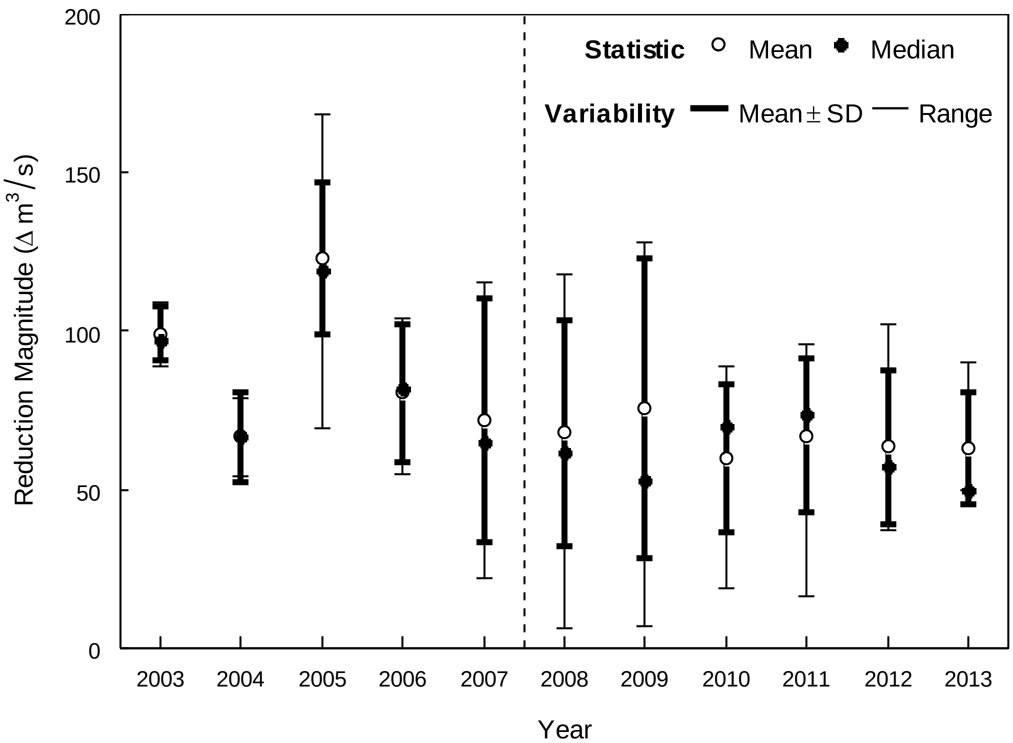


Figure 7: Reduction magnitude (Δm³/s) by year, depicting annual range (min, max), mean, and median, as well as mean ± SD.

In three of the five years examined during pre-WUP operations, ramping rate (Δm³ s^{-1} h^{-1}) exhibited substantial variations and range (Figure 8). 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.

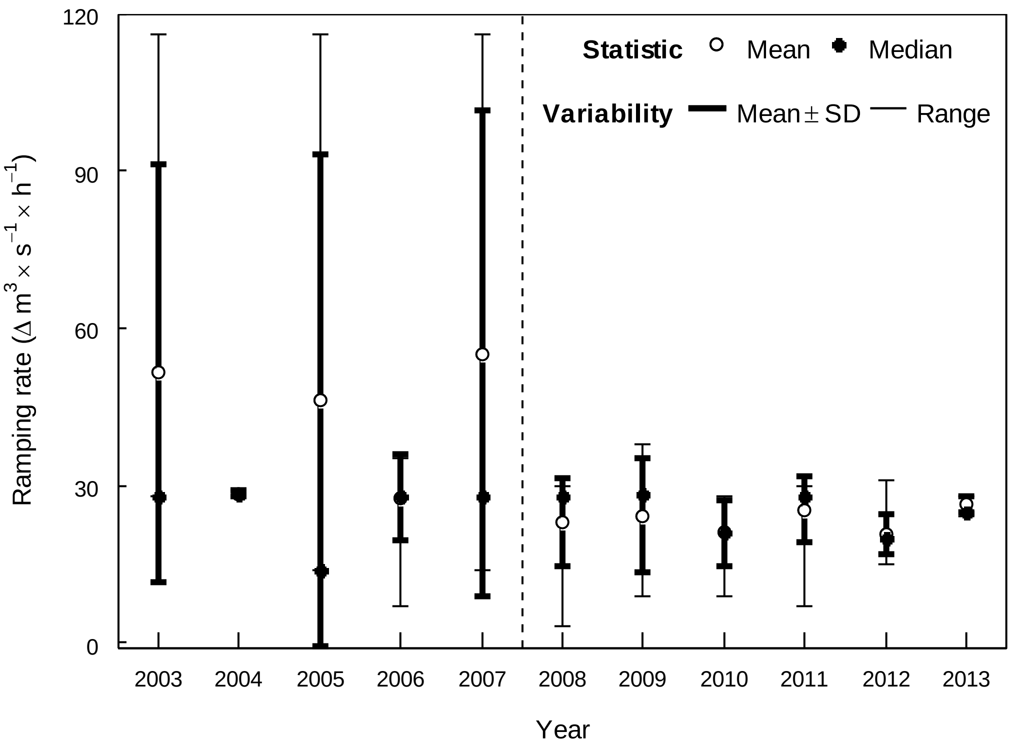
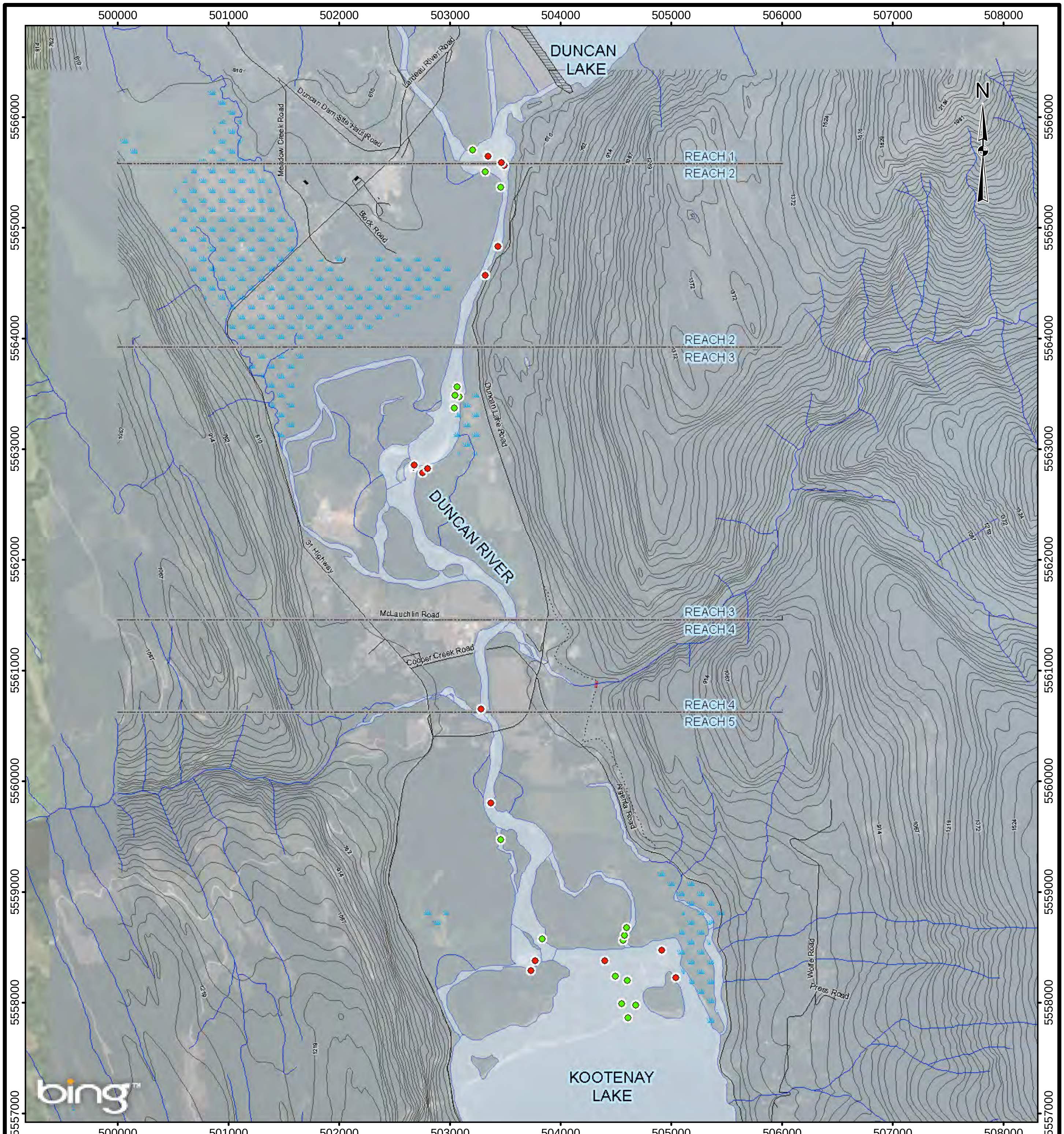


Figure 8: Ramping rate (Δm³ s^{-1} h^{-1}) by year, depicting annual range (min, max), mean, and median, as well as mean ± SD.

3.4 Fish Abundance Assessment

A total of 34 sites were surveyed during the 2013 snorkeling fish abundance assessment (Figure 9), with a total of 870 fish counted across all sites and strata (Table 16). The lowest numbers of encounters of both Mountain Whitefish and Rainbow Trout occurred in the Deep/Fast stratum, which also exhibited the lowest mean encounters and standard deviations. Although the total number of Mountain Whitefish documented was highest in Shallow/Slack habitat, the higher variability in encounters within this habitat stratum resulted in slightly lower encounters in comparison to Shallow/Fast habitat. For Rainbow Trout, encounters in both shallow-habitat strata were similar, while mean encounters and standard deviation were slightly lower in the Shallow/Slack strata (Table 16).



LEGEND

SNORKEL SITE LOCATION

- MAIN CHANNEL
- SIDE CHANNEL
- REACH BREAK
- WATERCOURSE
- CONTOUR (30 m INTERVAL)
- WETLAND
- WATERBODY

1,000 0 1,000 Metres

SCALE 1:50,000

REFERENCE

1) IMAGERY: DATED 2012, PROVIDED BY CLIENT.
2) SERVICE LAYER CREDITS: . USED UNDER LICENSE, ALL RIGHTS RESERVED. IMAGERY DATE: 20080502
3) CONTOUR, ROAD, WATERCOURSE, WETLAND, WATERBODY: CANVEC.CA

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	14 May 2015		
CHECK	BH	3 Jul. 2015	FIGURE: 9	
REVIEW	DS	3 Jul. 2015		

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Table 16: Summary of fish counts across depth and flow strata, as recorded from 2013 snorkeling surveys.

Depth	Flow	NSite	Mountain Whitefish			Rainbow Trout		
			N	Mean	SD	N	Mean	SD
Deep	Fast	11	144	13.1	14.1	36	3.3	3.2
Shallow	Fast	10	225	22.5	36.2	104	10.4	15.9
Shallow	Slack	13	260	20	39.5	101	7.8	10.1
Total		**34**	**629**			**241**		

In the present study, variability between habitat strata was relatively high, as was variability within strata (Figure 10). Variability in Mountain Whitefish and Rainbow Trout density was highest in Shallow/Slack sample sites of the mainstem. Within sidechannel sites, the highest variability in Mountain Whitefish and Rainbow Trout density was in Deep/Fast and Shallow/Fast habitat, respectively. Data collected during DDMMON-2 in 2012 exhibited much higher variability in both mainstem and side channel sites (Thorley et al. 2012). Although the total area sampled was similar between studies, higher numbers of shorter sites were sampled in that program, which contributed to the variability of the dataset (Figure 10).

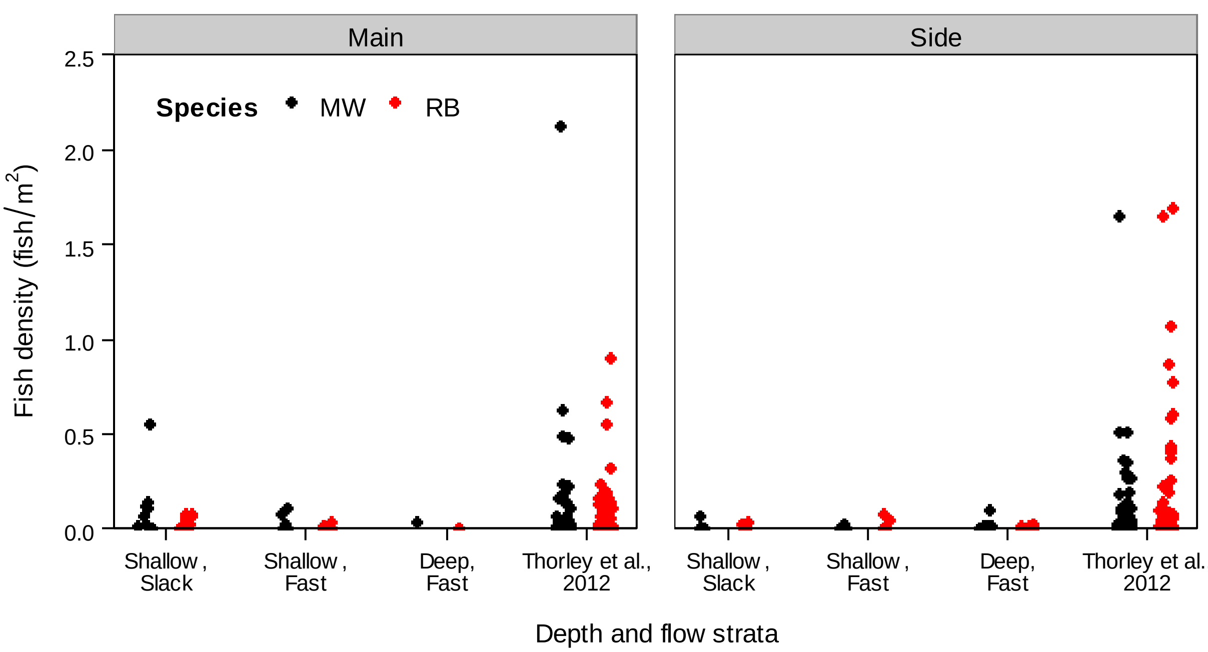

Figure 10: Fish density (fish/m²) across species, depth, flow, and channel type (main vs. side channel) strata for 2012 and 2013 data. Note: the strata of the 2012 data was grouped into one data series for each channel type.

Abundance estimates for Mountain Whitefish abundance in all strata were consistently higher and more variable than Rainbow Trout abundance estimates (Figure 11). Also, median 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. For both Mountain Whitefish and Rainbow Trout, non-overlapping confidence intervals indicated that there were statistically significant differences between abundance estimates between Shallow/Fast habitats and all other strata (Figure 11). The reason for this difference is related to the amount of area in each habitat strata. The habitat coefficients themselves (depth, flow) were found to be not significant during the modelling.

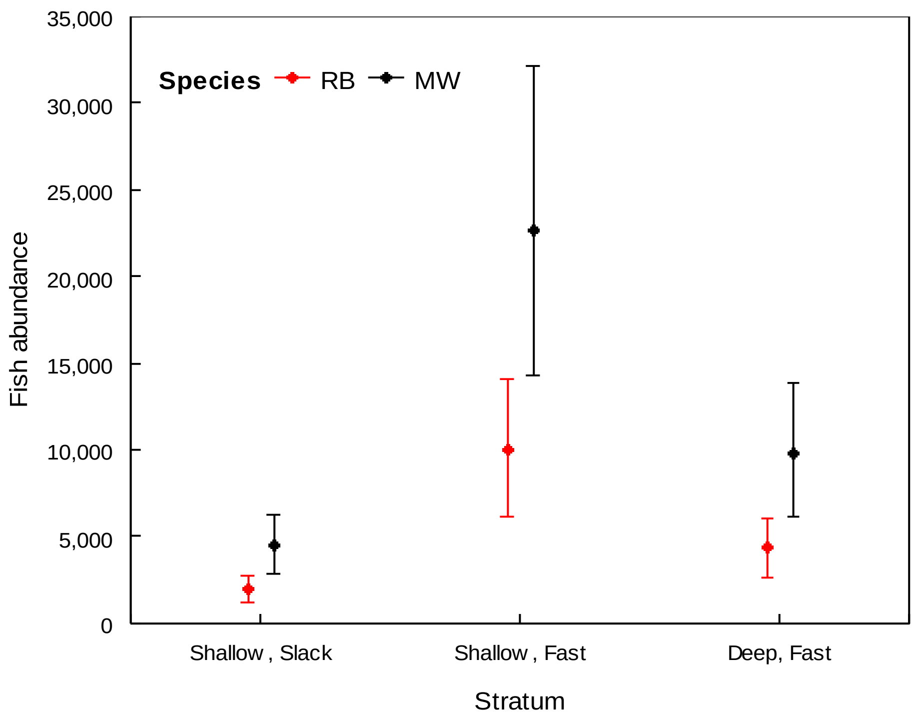

Figure 11: Median Mountain Whitefish and Rainbow Trout abundance, plotted by depth and flow strata and their respective 95% credibility intervals.

The total fall abundance estimate for Mountain Whitefish in the present study was similar to the estimate obtained for fall 2010, while the confidence interval associated with the current estimate was slightly wider (Table 17). Although the confidence intervals were narrower, the fall 2013 Rainbow Trout abundance estimate was less than half of the fall 2010 estimate (Table 17).

Table 17: Comparison of fall 2010, spring 2012, and fall 2013 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	Life stage	Fall 2010	Fall 2013
Mountain Whitefish	Fry	38,021 (28,079 – 49,030)	Combined into one estimate for juvenile life stage
	Parr	604 (190 - 3611)	
	Total	**38,625 (28,269 – 52,641)**	**36,936 (23,315 – 52,325)**
Rainbow Trout	Fry	43,489 (28,760 – 63,506)	Combined into one estimate for juvenile life stage
	Parr	5,492 (2,069 – 10,088)	
	Total	**48,981 (30,828 – 73,594)**	**16,330 (9,985 – 22,874)**

3.5 Fish Stranding Assessment

Pool stranding estimates in the following sections refer to 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 Figure 12. Therefore, the effect of slope on the formation of pools was not included in the final pool stranding model.

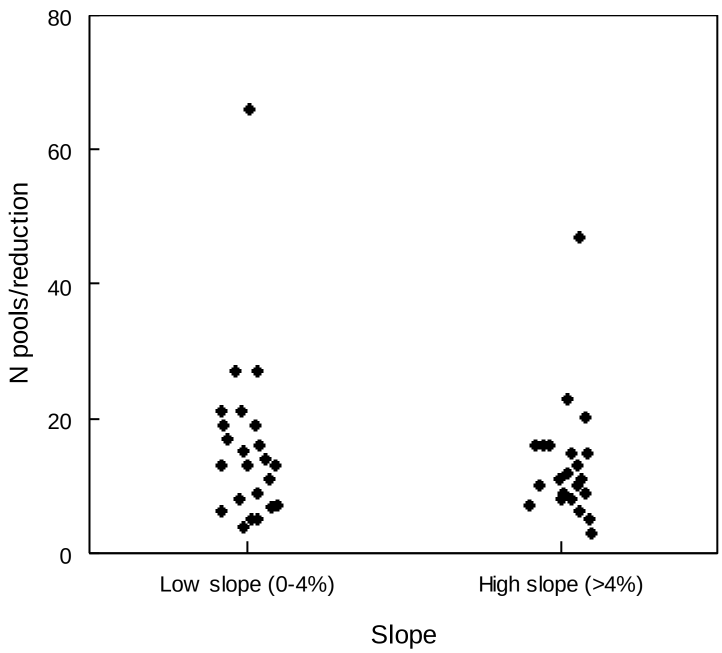

Figure 12: Number of pools recorded per reduction in low-slope and high-slope habitat, 2010 - 2014.

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 was similar in all years examined (Figure 13). In the two study years where spring assessments were conducted (2011-2012 and 2012-2013), non-overlapping credibility intervals indicated that the number of pools that formed between years differed significantly (Figure 13).

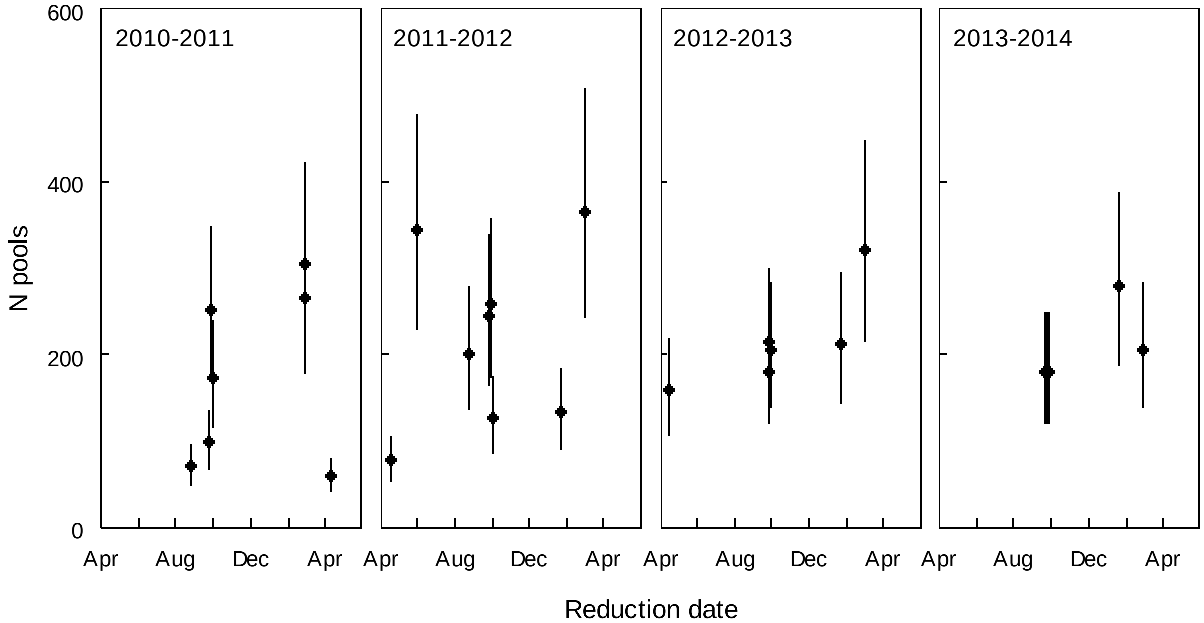

Figure 13: Median estimates of pool numbers formed during the 2010-2014 stranding events, plotted by reduction and stranding year. Error bars are 95% credibility intervals.

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 0.477 (median value; 95% credibility interval of 0.455-0.500).

The variability in the number of fish stranded per pool was similar between low and high slope habitat Figure 14. This indicated that slope was not a factor influencing pool stranding.

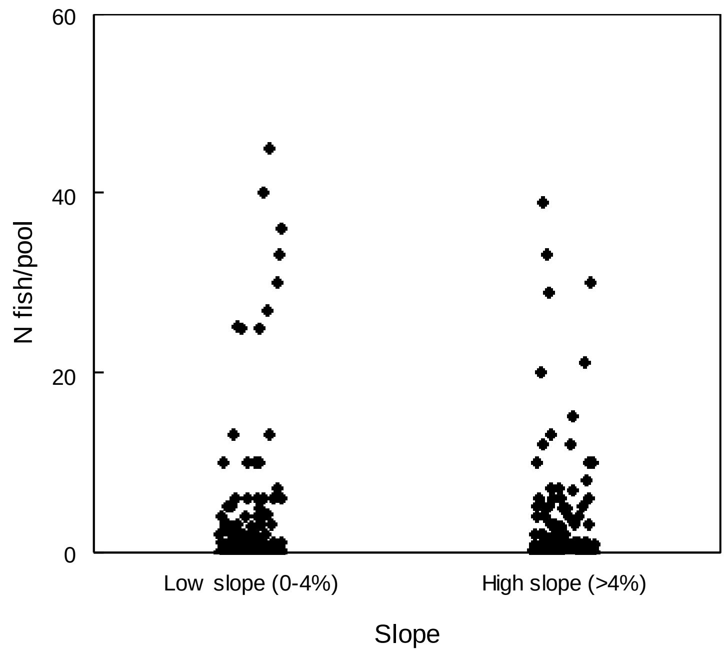

Figure 14: Number of collected fish per pool, plotted by low-slope and high-slope habitat, 2010 - 2014.

The median number of fry per pool for the spring season (January – June) was estimated to be 1.96 (1.33 - 2.87) fish/pool (Figure 15). In contrast, the median number of Rainbow Trout juveniles stranded per pool in the fall (July to December) was estimated at 6.61 (4.55 – 9.54) (Figure 15). The season effect on stranding numbers was found to be significant ($p < 0.05$), with median fall stranding estimates over three 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 16). With the exception of 2010-2011, the resultant pool stranding estimates indicated lower levels of stranding in the months of January and April in comparison to the fall season. In the 2012-2013 study year, one assessment in September had significantly higher estimates of pool stranded juvenile Rainbow Trout than all other assessments examined. Similar to the estimates on the formation of pools, non-overlapping credibility intervals indicated that the estimates of pool stranded Rainbow Trout for the two years when spring assessments were conducted (2011-2012 and 2012-2013) differed significantly (Figure 16).

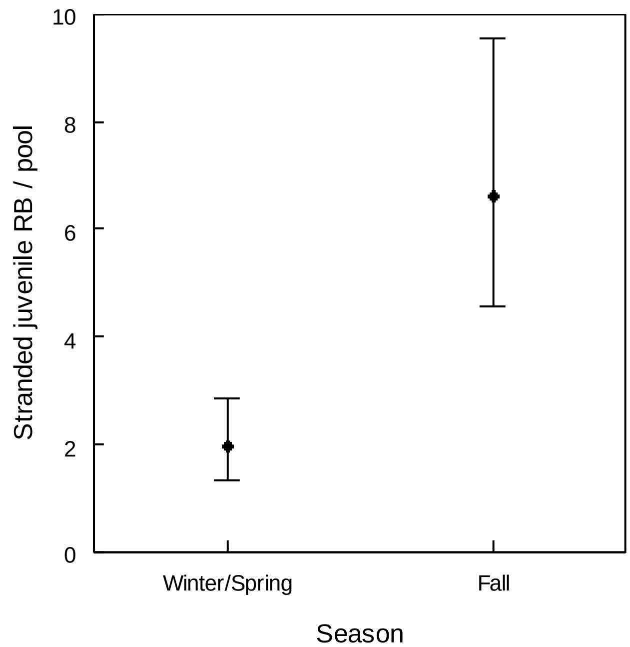

Figure 15: Median estimates of stranded Rainbow Trout per pool, plotted by season, 2010 - 2014. Error bars are 95% credibility intervals.

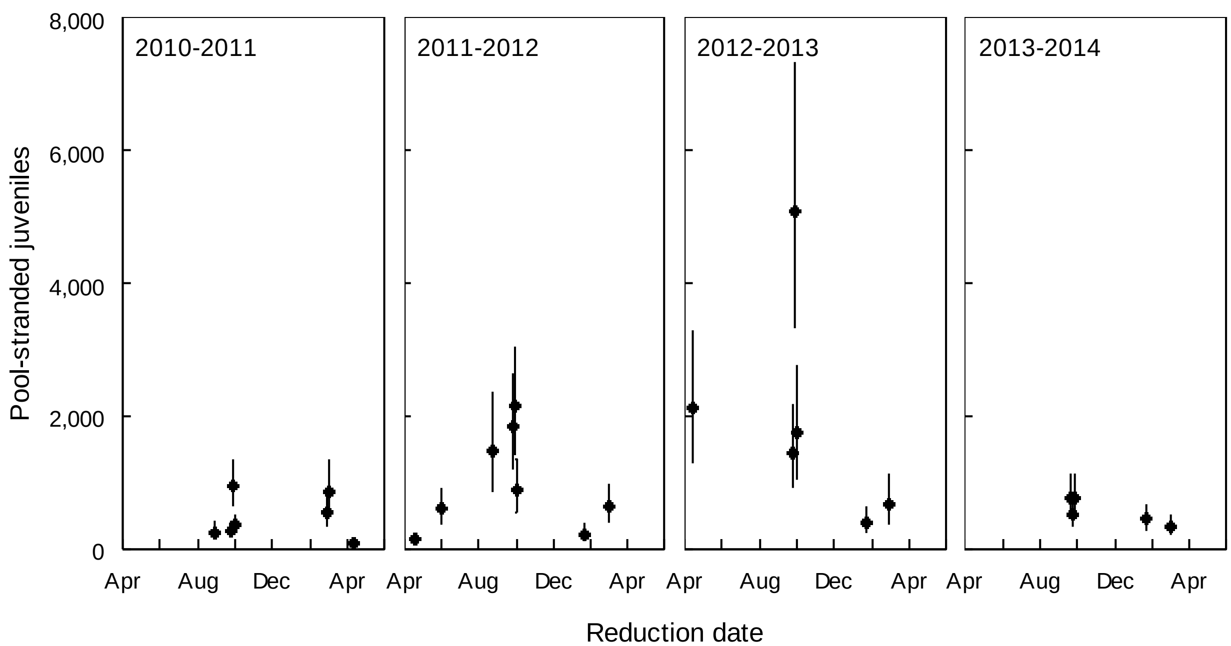

Figure 16: Median estimates of pool-stranded Rainbow Trout during the 2010-2014 stranding events, plotted by reduction and stranding year. Error bars are 95% credibility intervals.

3.5.3 Interstitial Stranding

Over the last three study years when the interstitial sampling methodology was standardized, very few interstitially stranded fish have been observed (Figure 17). In total, 14 Rainbow Trout and 2 Mountain Whitefish have been found to be interstitially stranded. Also, interstitially stranded fish were not found in the present study year (Figure 17). Relationships between stranded fish counts and slope or fish counts and ramping rate (p > 0.05 for both) were not found (Figure 17, Figure 18).

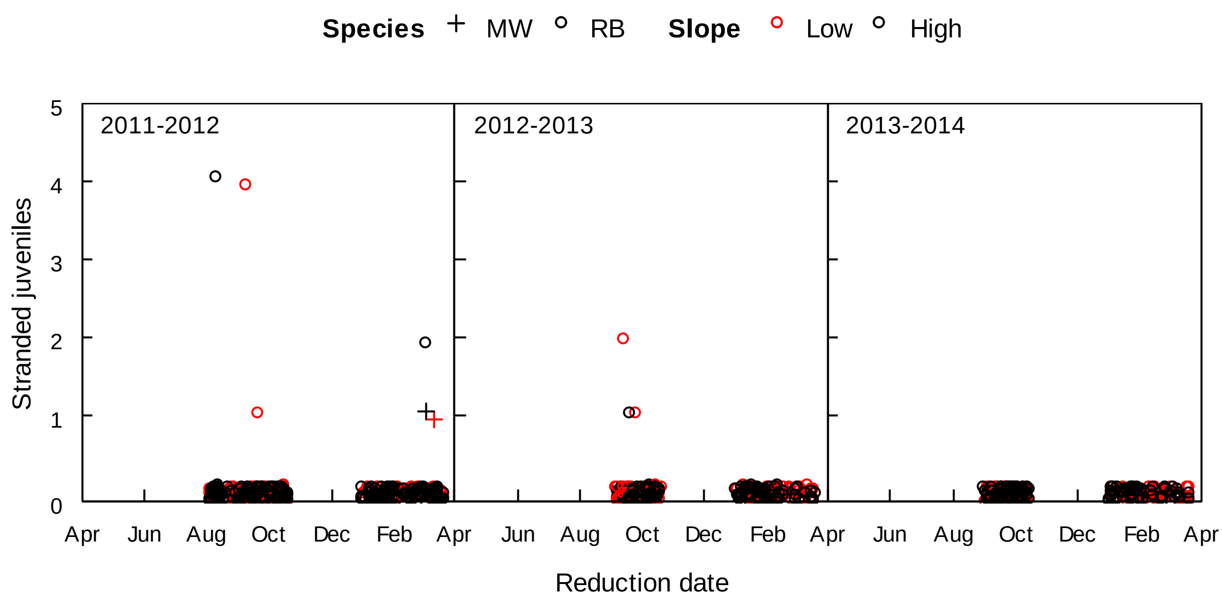

Figure 17: Counts of interstitially stranded Mountain Whitefish and Rainbow Trout, plotted by study year, reduction date, and slope (where low slope is 0-4%, and high slope is >4%).

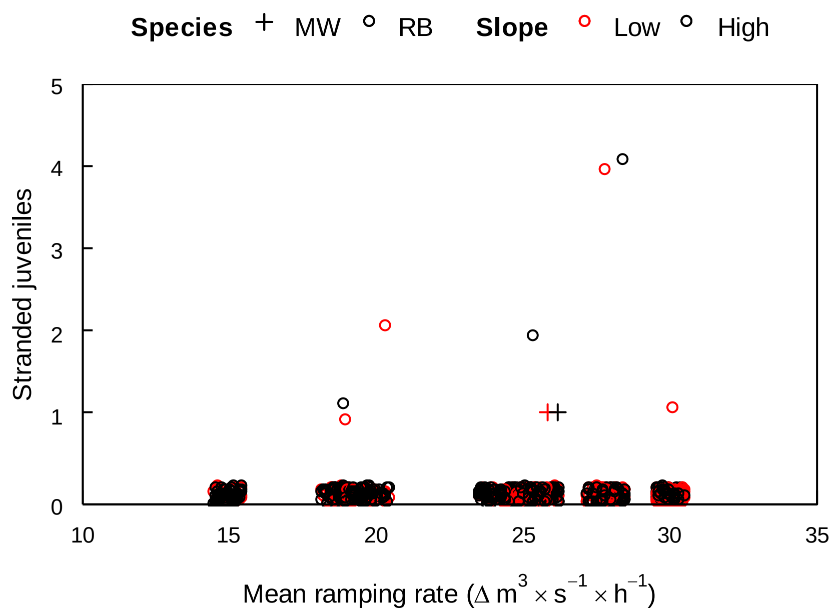

Figure 18: Counts of interstitially stranded Mountain Whitefish and Rainbow Trout vs. mean ramping rate, plotted by species and slope (where low slope is 0-4%, and high slope is >4%), 2011 - 2014.

Summed by reduction, median interstitial stranding estimates ranged from 1,309 fish (October 01, 2011) to 13,969 fish (March 01, 2012; Figure 19). The high variability in predictions, due to data scarcity, resulted in upper 95% credibility intervals ranging between 11,109 fish (September 26, 2012) to 32,386 fish (March 01, 2012; Figure 19).

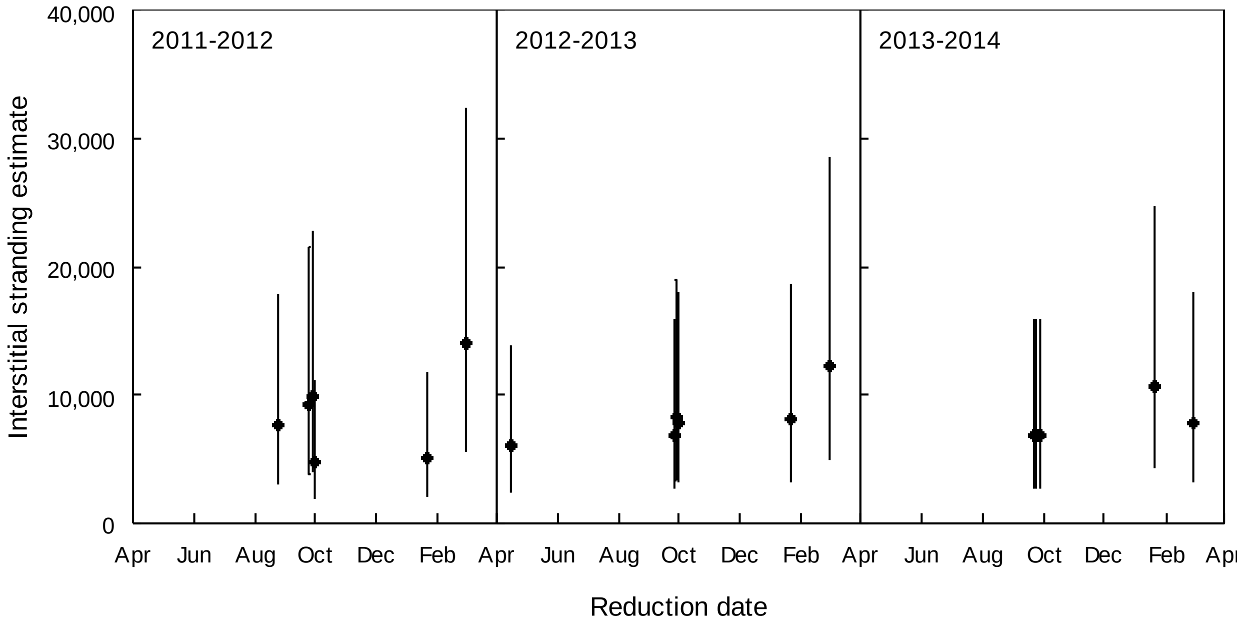

Figure 19: Median estimates of Rainbow Trout juveniles and fry interstitially stranded during the 2011-2013 stranding events, plotted by study year and reduction date. Error bars are 95% credibility intervals.

Median estimates of MW interstitial stranding by reduction event ranged from 762 to 2,223 fish (October 01, 2011 and Mar 01, 2012, respectively; Figure 20). While median estimates were very low (due to the low number of Mountain Whitefish collected during sampling), 95% credibility intervals were wide, reflecting the high uncertainty associated with interstitial stranding prediction. Upper 95% credibility intervals ranged from 3,796 fish in October 2011 to 11,067 fish in March 2012 (Figure 20).

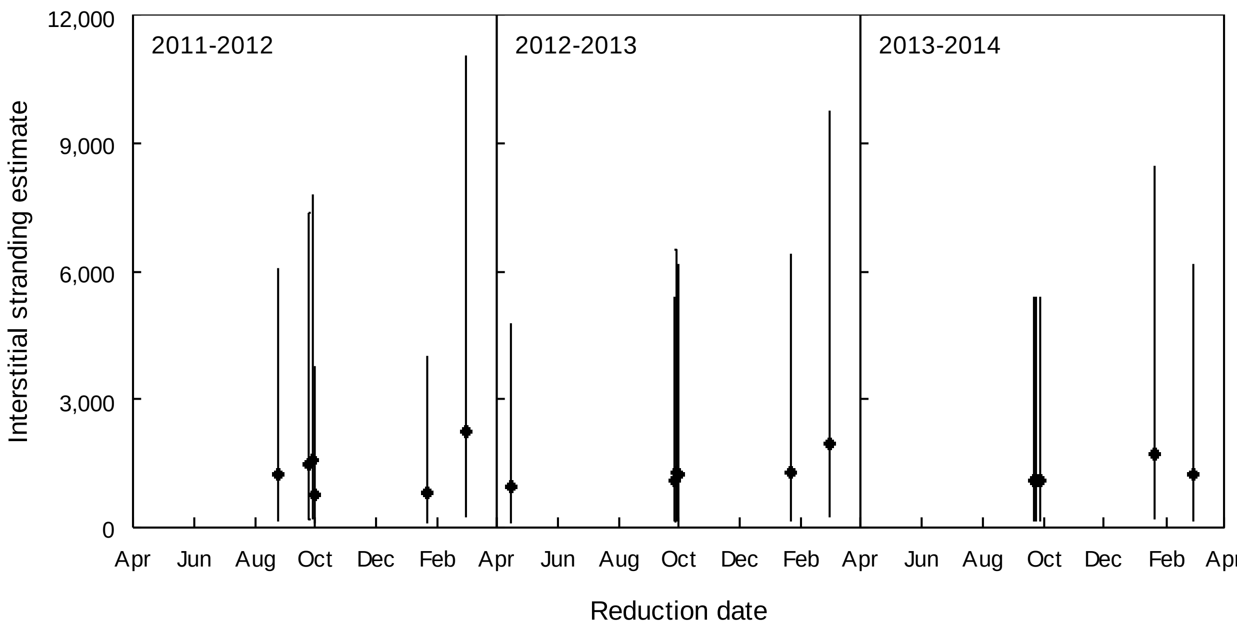

Figure 20: Median estimates of Mountain Whitefish juveniles and fry interstitially stranded during the 2011-2013 stranding events, plotted by study year and reduction date. Error bars are 95% credibility intervals.

During the 2011-2013 study years, median estimates of total annual interstitial stranding of Rainbow Trout ranged between 39,029 and 50,671 fish, with 95% credibility intervals ranging from 15,628 (lower limit, 2013) to 117,468 (upper limit, 2011; Table 18). Median estimates of total annual stranding of Mountain Whitefish ranged between 6,213 and 8,067 fish, with 95% credibility intervals ranging from 691 (lower limit, 2013) to 40,142 (upper limit, 2011). When comparing the previous interstitial stranding estimates for Rainbow Trout and Mountain Whitefish obtained in fall of Year 4 with those obtained in fall using the current dataset, the previous estimate for Year 4 was higher, and had wider confidence intervals (Table 18).

Table 18: Total annual interstitial stranding estimates, detailing median and 95% credibility interval (CRI) values for Rainbow Trout and Mountain Whitefish in each of 2011-2013 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 current modelling)	50,671	20,290 – 117,468	8,067	898 – 40,142
2012-2013 (Year 5, using current modelling)	49,287	19,736 – 114,261	7,846	873 – 39,046
2013-2014 (Year 6, using current modelling)	39,029	15,628 – 90,480	6,213	691 – 30,920

3.5.4 Total Stranding Estimates

Overall estimates of Rainbow Trout abundance and stranding by stranding mechanism for the current study year are summarized in Table 19. As pool stranding estimates could not be obtained for Mountain Whitefish, comparisons of stranding estimates in relation to abundance estimation could not be presented. Interstitial stranding estimates had wide confidence intervals, which indicate a high level of uncertainty related to the estimates (Table 19).

Table 19: Year 6 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 2013 Abundance Estimate		
	Lower	Median	Upper	Lower	Median	Upper	Lower	Median	Upper
RE2013-03	494	776	1,146	2,742	6,847	15,873	9985	16330	22874
RE2013-04	331	518	769	2,742	6,847	15,873			
RE2013-05	484	763	1,129	2,742	6,847	15,873			
Fall 2013	**1,309**	**2,057**	**3,044**	**8,226**	**20,541**	**47,619**			
RE2014-01	271	445	685	4,277	10,682	24,765			
RE2014-01	197	330	525	3,126	7,806	18,096			
Spring 2014	**468**	**775**	**1,210**	**7,403**	**18,488**	**42,861**			
Year 6 Total	**1,777**	**2,832**	**4,254**	**15,629**	**39,029**	**90,480**			

4.0 DISCUSSION

4.1 Current Duncan Dam Operations in Relation to Fish Stranding

4.1.1 Variables Affecting Fish Stranding

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

4.1.2 Pre- and Post-WUP Operating Regimes

Management Question 1) (*How effective are the operating measures implemented as part of the ASPD program?)* was addressed by examining the differences between the pre- and post- WUP flow regimes. Under the water license, two large reductions in DDM discharge occur on an annual basis. In the post-WUP regime, flow reductions occur in late September to early October for Kokanee protection by restricting access to spawning areas that pose high risks to strand eggs and larvae. Also in the post WUP period, flow reductions in late winter were altered for support of Columbia River Mountain Whitefish management objectives. The purpose of the late winter flow reductions is to manage Duncan Reservoir flood control targets as defined under the Columbia River Treaty. In addition there are several smaller reductions that occur throughout the year to effectively manage water resources and power generation at other facilities.

Total area dewatered during all annual flow reductions was used to determine differences in pre- and post-WUP operations, because the area exposed relates directly to the hydraulic and stranding analysis models. The examination of the amount of area of exposed habitat per year due to LDR discharge reductions indicated that post-WUP flows have resulted in the dewatering of less area compared to pre-WUP operations (Section 3.3 above). Interannual variability in discharge has also been reduced under post-WUP operations. 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. This is also supported by studies conducted in Norway (Halleraker et al. 2003), that 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.

A relationship between interstitially stranded fish counts and ramping rate ($p > 0.05$) in the current program was not found. While the data showed no trends for this relationship, this may also be due to data scarcity. This relationship should be re-evaluated as more data are collected.

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, based on only observations of stranded fish. Therefore, assessments of fish stranding can be inferred based on the amount and rate of habitat dewatering to assess 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 in the current study year. The species of interest for this study program are Rainbow Trout and Mountain Whitefish. During the Year 6 assessments, eight different species were encountered (three sportfish and five 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 over three 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.

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 more precise than the estimates obtained in previous years (Golder 2011, Golder and Poisson 2012). Random sampling of interstitial habitat began in August 2011, and is still a relatively new part of the program. While interstitial stranding is likely to be biologically important, 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.

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). However, those estimates may have been overestimates since they were based on any dewatered zone of the river being categorized as stranding habitat, while subsequent field assessments have excluded multiple areas based on the extreme gradient they contain. In addition, the estimates of area dewatered were only conducted from three outputs of the early version of the original DDMMON-3 hydraulic model. Conversely, statistically significant relationships between interstitially stranded fish counts and slope in the current dataset were not found ($p > 0.05$). While the data showed no trends for the relationship, this may be due to data scarcity. This relationship should be re-evaluated as more data are collected.

The results from the current dataset suggest that slope did not have an effect on the formation of isolated pools within the study area. Also, a relationship between slope and the number of fish stranded in isolated pools was not identified. 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.

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), index sites tended to be of lower gradient than non-index sites. Interestingly, 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. 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. Overall, index sites strand more fish because more area dewaters at these sites during flow reductions. Based on previous analyses, index sites had a bias toward higher stranding rates and therefore, hypothesis H_{01} was rejected.

This hypothesis was not specifically examined in the present study year. It will be re-examined in the Year 7 in-depth interpretive report with all available project data to determine if the complete dataset supports the rejection of hypothesis H_{01}.

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 fall 2013 abundance estimate for juveniles obtained in this program was 16,330 (95% credibility intervals range from 9,985 – 22,874). These findings should be interpreted with caution as the densities documented in the present study were substantially lower than in previous studies and can be related to external factors, such as abundance of the spawning population.

Further effort and analysis are required in Year 7 of this program to confirm the validity of the current abundance estimates.

The sum of the estimated fall 2013 interstitial and pool stranded Rainbow Trout in the LDR had a median value of 22,598 and minimum and maximum 95% credibility intervals of 9,535 and 50,663 respectively. Estimates for the number of Rainbow Trout juveniles stranded in pools were relatively precise and low when compared to population abundance and interstitial estimates, and attributed to only 9.1% of the total stranding estimated. Based on the likely overestimated interstitial stranding estimates, combined with the precise pool estimates from the present dataset, hypothesis H_{02} is rejected. With the rejection of hypothesis $H_{02,}$ we must therefore conclude that fish stranding as a result of DDM operations has a significant impact on Rainbow Trout populations. With the uncertainties in the current dataset, it is not possible to determine the level of impact. The further refinement of interstitial stranding rates may reverse this finding.

To address hypothesis H_{02} more confidently, it is critical that the uncertainties associated with the abundance and 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 continue to 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

Abundance estimates for Mountain Whitefish for the present were similar to those obtained in by the DDMMON-2 program in the fall of 2010 (Thorley et al. 2012). This suggests that the Mountain Whitefish population in the LDR has remained relatively stable since 2010. The confidence intervals associated with the current estimate were also similar to the previous estimate, which indicated a comparable level of uncertainty related to both. 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 the present study year.

Over the course of the study year, only 49 stranded Mountain Whitefish were documented, all of which were observed in the fall season during one assessment (RE2013-04) at index stranding site LARD0.3R (Appendix B, Figure 1). Mountain Whitefish encounters have been minimal in all study years. This consistently low level of stranding was not considered ecologically significant and will likely not result in a population level effect on Mountain Whitefish. However, previous experimental stranding investigations indicated that large numbers of mountain whitefish could be stranded during rapid night time reductions in flow (Poisson and Golder 2010). Consequently, these conclusions are based on the assumption that operations in the future will be within the range and the diel timing that occurred during the 2013-2014 investigations.

4.3 Summary

The key findings for the Year 6 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);
  - 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); and,
  - 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 various studies conducted on the LDR. To better understand stranding related to the species of interest in the LDR, the abundance and 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 stranding in the LDR.

2) Continue to build on the stranding site selection program to ensure that sites assessed are representative of the river.

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, we can update our dataset, calculate the dewatered area at each of our sites, and have the most up to date and representative data to conduct the stranding analyses in Year 7.

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.

5) Monitor the findings of the ongoing Rainbow Trout management program in the Lardeau and Duncan Rivers to determine if the decline identified from the fall 2013 abundance estimates can be tracked through the spring of 2014. If the findings of that program do not support the current estimation of declining abundance, explore the feasibility of altering abundance estimation methodology (i.e., increasing sample effort during the fall snorkel surveys) in order to refine the abundance estimates of the species of interest.

These recommendations will focus sampling effort and are designed to build on the current data set. The focus of study Year 7 should be on the refinement of interstitially stranded fish estimates throughout the system, as well as ensuring that the abundance estimates obtained in study Year 7 are as accurate as possible. The Year 7 summary report will also address site selection moving forward into future study years, to ensure that representative sites are selected during mitigation activities. 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.

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APPENDIX A

Bayesian Models - Code

JAGS code for Bayesian analysis of fish density and abundance

```
model {
	sSite ~ dgamma(0.01, 0.01)
	logit(p) <- bEfficiency
	bArea ~ dnorm(0, 0.01)

	for(k in 1:NSite){
		bSite[k] ~ dnorm(0, sSite^-2)
			}

	for(k in 1:NSite){
		mu[k] <- bSite[SiteNum[k]] + bArea*area[k]
		censor[k] ~ dinterval(N[k], min_pop[k])
		N[k] ~ dpois(mu[k])
		Count[k] ~ dbin(p, round(N[k]))
			}

	for(i in 1:NStrata){
		mutotal[i] <- bArea*AreaStratum[i]
			}
	}
```

JAGS code for Bayesian analysis of interstitial stranding code

```
model{
	pIntercept ~ dnorm(0,5^-2)
	bIntercept ~ dnorm(0,5^-2)

	for(i in 1:nObs){
		mu.d[i] <- bIntercept
		logit(mu.p[i]) <- pIntercept
		p[i] ~ dbern(mu.p[i])
		Fish[i] ~ dpois(mu.d[i]*p[i])
			}
	}
```

JAGS code for Bayesian analysis of pool stranding code

```
model {
## stranding model:
	bSeason[1] ~ dnorm(0, 0.25)
	bSeason[2] ~ dnorm(0, 0.25)
	muEff[1] <- 10
	muEff[2] ~ dnorm(0, 0.01)
	logit(p[1]) <- muEff[1]
	logit(p[2]) <- muEff[2]
	sReduction ~ dunif(0, 5)

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

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

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

## 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)
		mu_pool[i] <- bIntercept + bSiteName[SiteName[i]] + bArea*SiteArea[i]
		NumPoolsPresent[i] ~ dpois(mu_pool[i]*u[i])
			}
```

```
## Derived
	for(r in 1:nReductions){
		log(muReduction[r]) <- bSeason[ReductionSeasonNum[r]] +
			bReduction[Reduction_ReductionNum[r]]
		High[r] <- bIntercept + bArea*(HighSlope*Drop[r])
		Low[r] <- bIntercept + bArea*(LowSlope*Drop[r])
		Total_pools[r] <- High[r] + Low[r]
		Total[r] <- muReduction[r]*Total_pools[r]
				}

	muSeason[1] <- exp(bSeason[1])
	muSeason[2] <- exp(bSeason[2])

	} #model
```

APPENDIX B

Summary of Identified Stranding Sites

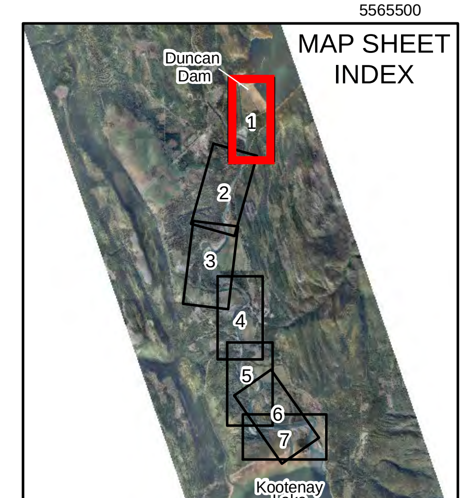
LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



0 100 200 300 M

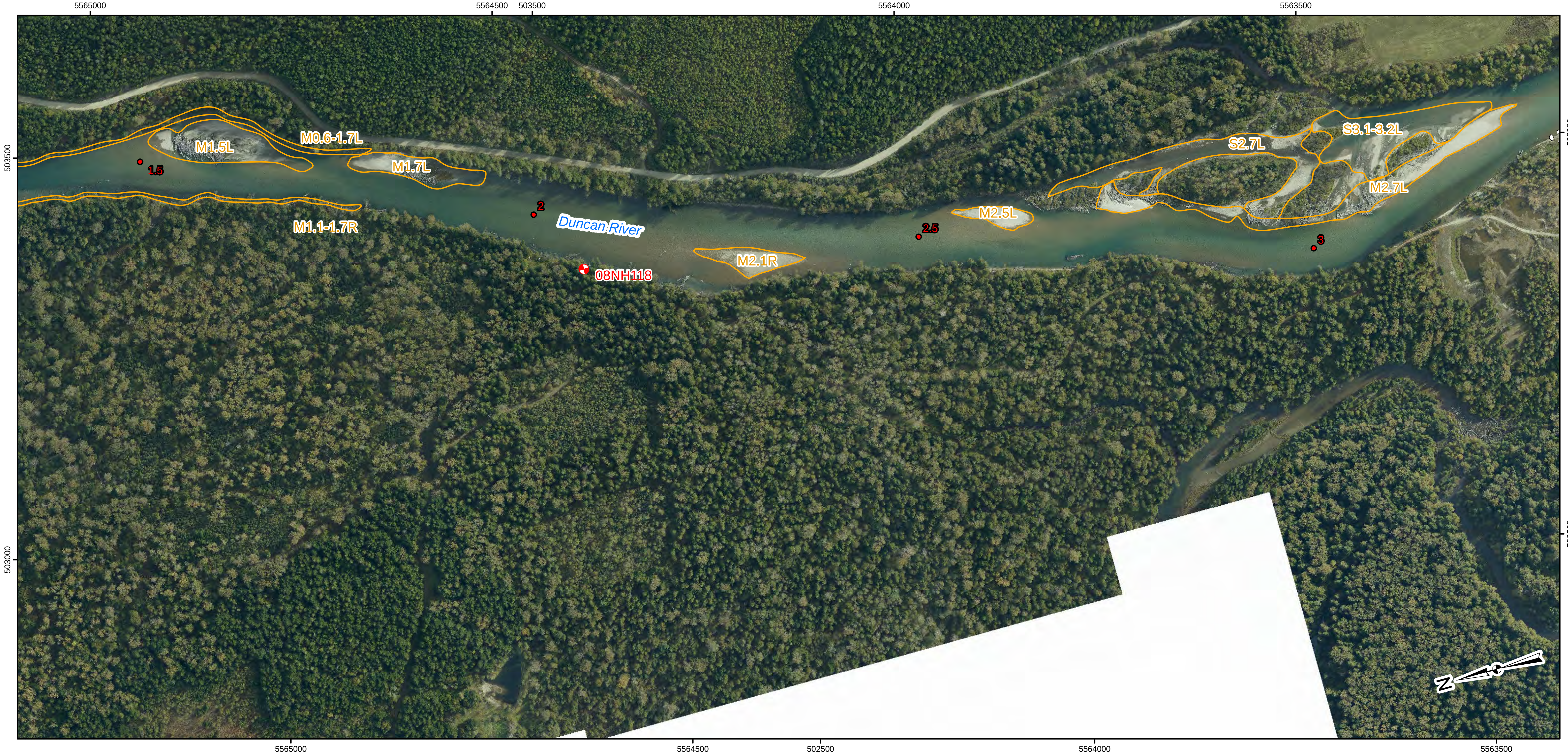
SCALE 1:5,000

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

Golder Associates

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

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



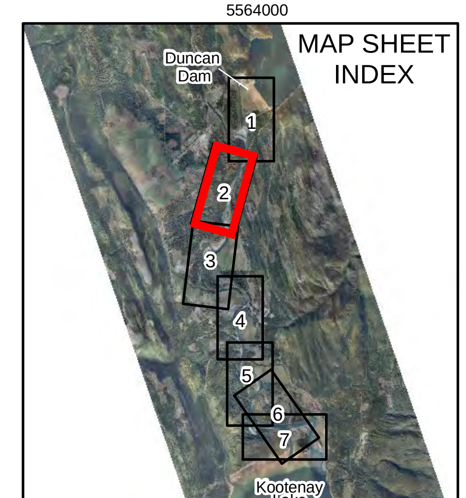
LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



0 100 200 300 M

SCALE 1:5,000

PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

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

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

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

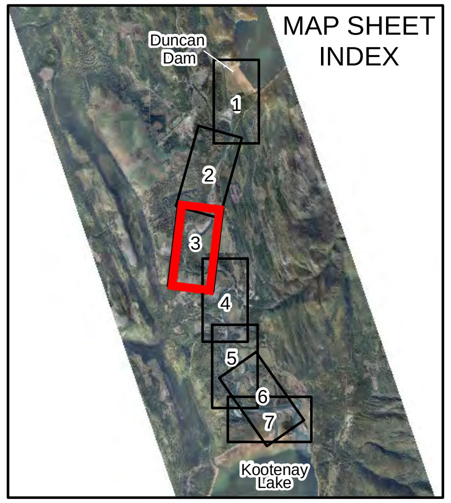
LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



0 100 200 300 M

SCALE 1:5,000

PROJECT

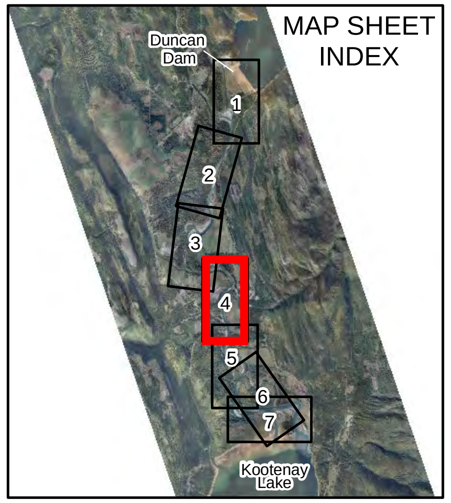
DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

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



LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83

0 100 200 300 M

SCALE 1:5,000

PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES

Golder Associates

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

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

LEGEND

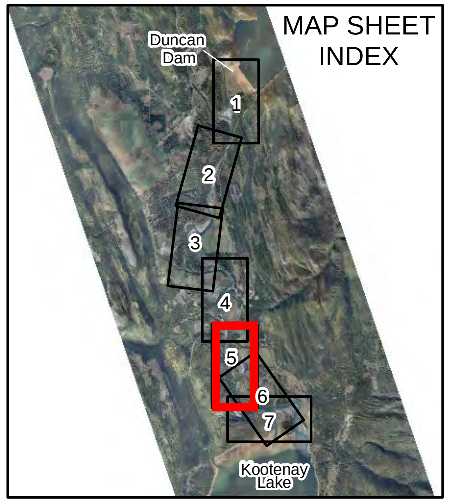
RIVER KILOMETRE

WSC GAUGE

BOAT LAUNCH

STRANDING SITE - INDEX

STRANDING SITE - NON-INDEX



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SCALE 1:5,000

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83

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

Golder Associates

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

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

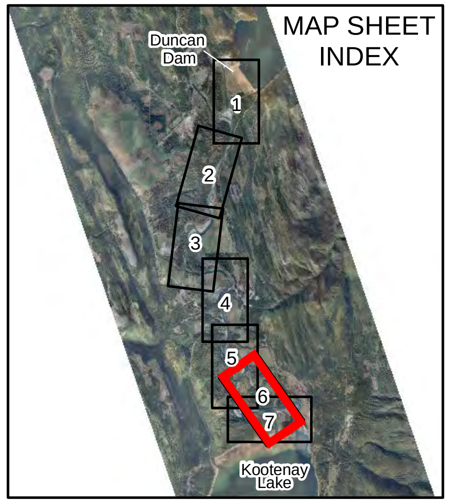
LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



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

Golder Associates

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

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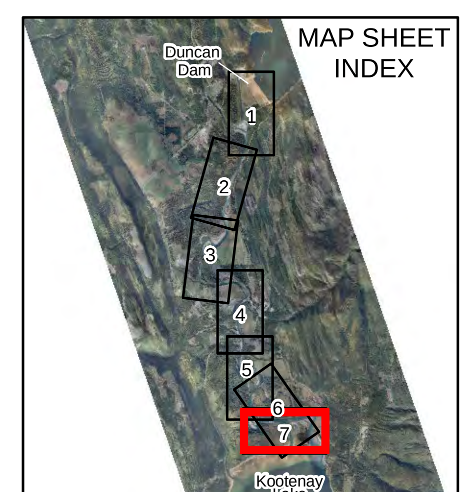
LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



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SCALE 1:5,000

PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES

Golder Associates

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

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APPENDIX C

Photographic Plates

Plate 1 Log Jam at Rkm 4.5 preventing access to downstream portions of the Lower Duncan River, September 26, 2012.

Plate 2 Submerged terrestrial vegeation cover in an isolated pool, site S4.1R on August 25, 2011.

Plate 3 Overhanging vegetation cover along sidechannel at site SLard0.3R on August 25, 2011.

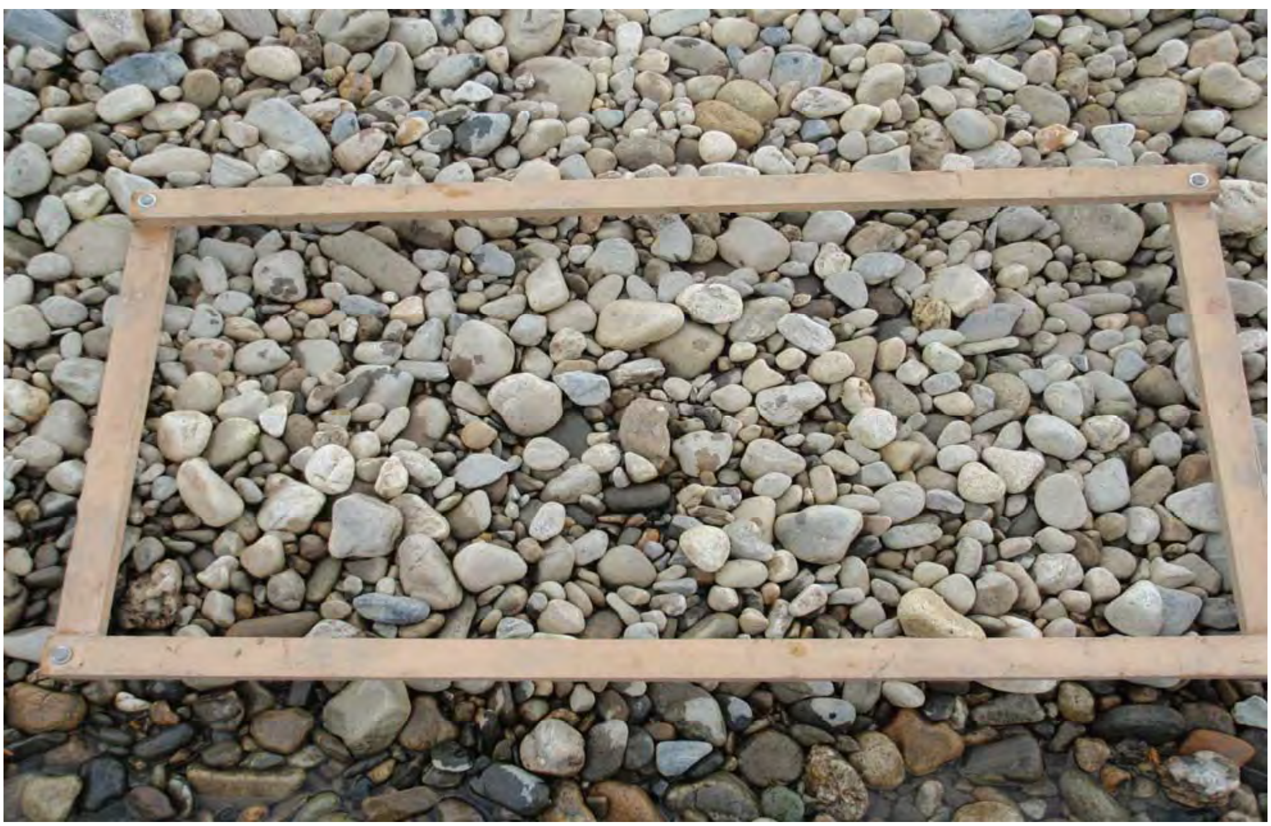

Plate 4 Interstitial grid prior to sampling, site S3.5-4.0R on October 01, 2011.

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