Duncan Dam Project Water Use Plan

Adaptive Stranding Protocol Development Program

Implementation Year 4

Reference: DDMMON-16

Lower Duncan River: Fish Stranding Impact Monitoring: Year 4 Summary Report

Study Period: April 2011 to January 2012

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July 11, 2012

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DDMMON-16: LOWER DUNCAN RIVER

Lower Duncan River Fish Stranding Impact Monitoring: Year 4 Summary Report (April 2011 to January 2012)

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REPORT

Report Number: 10-1492-0110

Distribution:

6 Hard Copies to BC Hydro - Castlegar
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Cover Photo: Icicles on Argenta Bridge Pilings after flow reduction, January 20, 2012.

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

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, fish stranding in the lower Duncan River (LDR) can be exacerbated by Duncan Dam (DDM) operations that influence the frequency and magnitude of flow fluctuations. The current program, initiated under the BC Hydro Water License Requirements (WLR) Program, includes the continuation of the DDMMON-16 Lower Duncan River Fish Stranding Impact Monitoring Program.

The results from this monitoring program will help inform flow management decisions that may impact on fish stranding in the LDR. Based on the current state of knowledge, the flow reduction measures implemented under the 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. At this time, documented low stranding rates of juvenile Mountain Whitefish (*Prosopium williamsoni*) are not believed to result in population level effects, while the current abundance and stranding estimates show a possible population level effect of stranding on juvenile Rainbow Trout (*Oncorhynchus mykiss*).

The current status of management questions for DDMMON-16 is presented in the table below. Because of the high degree of variation in stranding rates, high uncertainty of interstitial stranding estimates, and the many variables that could potentially contribute to stranding, these results should be treated as preliminary as they are somewhat sensitive to assumptions.

Table I: DDMMON-16 Year 4: Status of Management Questions and Objectives.

DDMMON-16 Management Question	DDMMON-16 Specific Hypothesis	DDMMON-16 Year 4 (2011-2012) 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 follow recommendations made by the DDMMON-15 Lower Duncan River Stranding Protocol Development and Finalization Program. The current WUP protocol reduces stranding rates by requiring daytime reductions at rates that result in slow stage changes rates (< 10 cm/hr) at the majority of identified stranding sites. - Variables related to stranding that are not currently addressed in the Adaptive Stranding Protocol Development Program (ASPD) are wetted history and season.

DDMMON-16 Management Question	DDMMON-16 Specific Hypothesis	DDMMON-16 Year 4 (2011-2012) 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. - 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.
	Ho_2: Fish populations in the lower Duncan River are not significantly impacted by fish stranding events.	- The estimate for the number of rainbow trout fry stranded in pools was relatively precise and relatively low. - The estimated numbers of interstitially stranded fish in the lower Duncan River were high. While interstitial stranding is likely to be biologically important, the current estimates may be upwardly biased and are uncertain. - There was a seasonal component to pool stranding, 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 abundance estimates for the rainbow trout fry in the LDR are uncertain. - With the current abundance and stranding estimates for Rainbow Trout, hypothesis Ho_2 is rejected. Therefore, Rainbow Trout fry populations are significantly impacted by fish stranding events. - Several factors affect fish populations including: predation, out migration, food availability, availability of suitable rearing habitats, winter mortality, as well as inter- and intra-species competition. Whether stranding events kill the fish that would succumb to these factors, or kill fish which would survive these factors is unknown. - 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.

Acknowledgements

Special thanks are extended to Trevor Oussoren (BC Hydro, Castlegar) 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
Guy Martel, Burnaby
Len Wiens, Duncan Dam
Kris Wiens, Duncan Dam
Ron Greenlaw, Duncan Dam
Linda Brownell, Power Records

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

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Table of Contents

TABLES

FIGURES

APPENDICES

1.0 INTRODUCTION

1.1 Background

The lower Duncan River originates from Duncan Dam (DDM), and runs for approximately 11 km before flowing into the north end of Kootenay Lake. Below DDM, the river flows through a man-made channel for 1 km to the confluence of the Lardeau River. Downstream from the confluence, the Duncan River is comprised of a series of single and braided channel sections with continually changing morphology that includes: debris jams, bars, and islands. Although natural flow fluctuations from unregulated rivers are known to cause fish stranding, fish stranding in the Duncan River can be exacerbated by DDM operations (Golder 2002) by influencing the frequency and magnitude of flow fluctuations. Formal assessments of fish stranding impacts related to changes in operations at DDM began in the fall of 2002. In 2004, BC Hydro developed a fish stranding assessment protocol that includes communication protocols, recommended flow reduction rates, and fish stranding assessment methodologies (BC Hydro 2004). An assessment of fish stranding impacts on the Duncan River related to DDM operations from November 2002 to March 2006 was previously completed (Golder 2006). In 2008, an annual summary of DDM related stranding events was completed for BC Hydro (Golder 2008).

One of the main objectives of the Duncan Dam Water License Requirements (WLR) Program is to evaluate the effectiveness of the operating regime defined in the Water Use Plan (WUP) and to identify opportunities to improve dam operations to maximize fish abundance and diversity in the Duncan River Watershed in consideration of other values. This involves assessing the influence of flow reductions on migrating, resident and/or rearing fish populations in the lower Duncan River. The Duncan Dam water license requires a minimum average daily flow from Duncan Dam 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 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 lower Duncan River 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 assess population level impacts associated with dam operations during the review period. The information generated by these assessments will ultimately form the rationale for the implementation of a final operating protocol for DDM discharge releases that minimizes impacts on fish.

The current program, initiated under the BC Hydro WLR Program, includes the continuation of the Lower Duncan River Fish Stranding Impact Monitoring Program. The fish stranding impact monitoring program conducted this year (Year 4) builds on the historic methodology, estimates total stranding, includes a more intensive analysis of the relevant data set, and analyzes pre-WUP DDM operations and how they relate to fish stranding. The monitoring program was 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 was to be accomplished by 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). The information obtained during the first three years of data collection and synthesis activities, combined with the information that will be obtained from other research activities internal and external to the Duncan Water Use Plan (WUP) monitoring programs, is expected to have a significant influence on the design of this stranding impact monitoring program into the future.

The following document provides information on fish stranding observed over all flow reductions from the timing of the last report on April 16, 2011 (Golder 2010) to January 20, 2012. The document includes new analyses that build on previously collected data and analyses, and includes reviews of the results of the analysis and field observations in relation to the multi-year program objectives.

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 variables that may affect fish stranding in the lower Duncan River were summarized in an impact hypothesis diagram (Figure 1), which conceptually links the variables and their effects. This diagram is a very simplified view of the interplay between the factors of interest and the outcomes. Interactions or autocorrelations are not shown in this diagram because they are so numerous, and would cause the diagram to become unreadable. The two processes that together entirely define the numbers of fish stranded are the probability of stranding and the density of fish in the near shore zone (fish available to strand).

The multiplication of probability of fish stranding by fish density predicts the number of fish stranded (Figure 1). 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. The impact hypothesis diagram shows total mortality, which 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. The impact hypothesis diagram has been constructed based on the best available information from the literature on hydro-peaking operations and fish stranding, but it is likely not comprehensive.

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 presented impact hypothesis diagram.

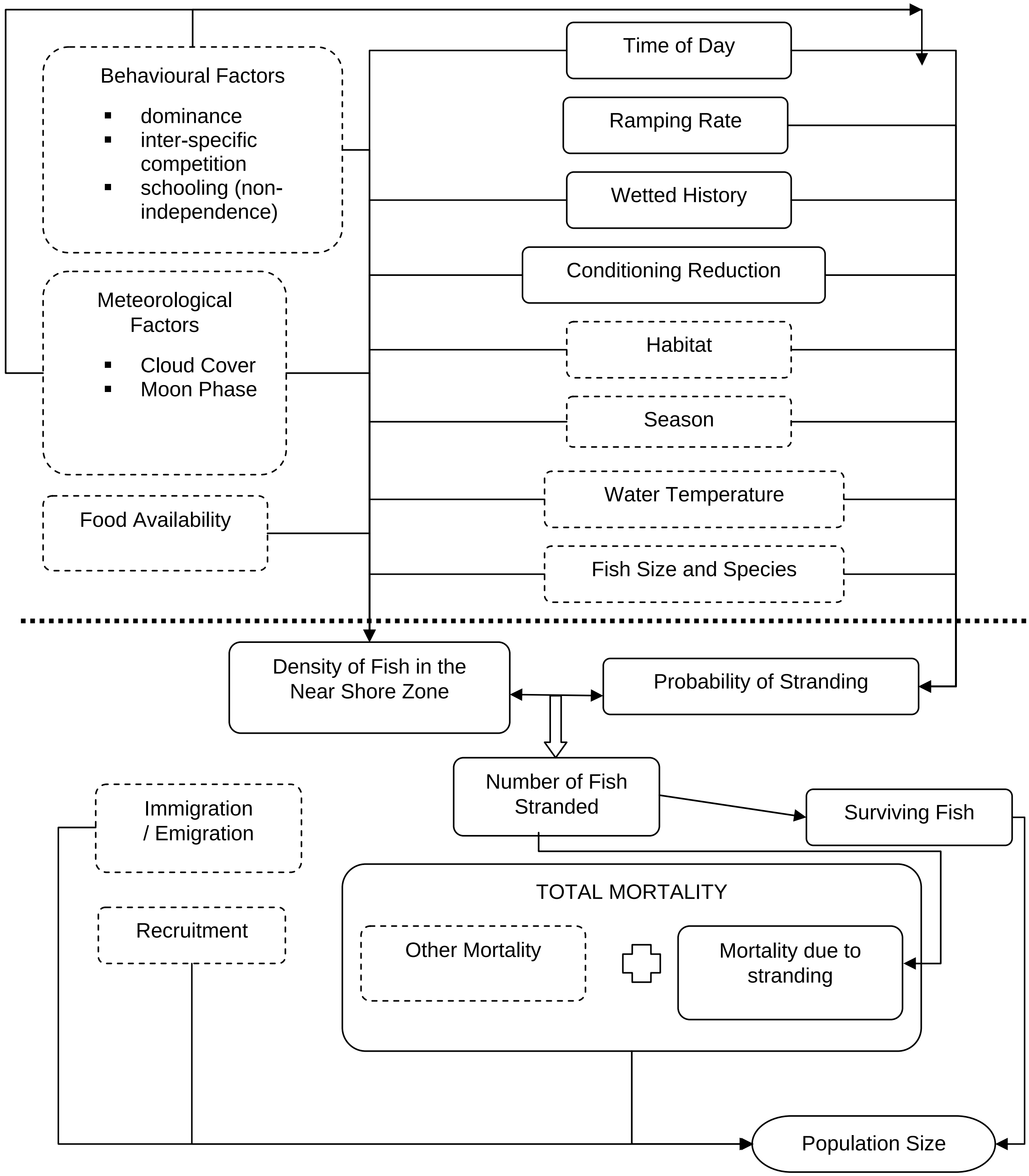


Figure 1: Impact hypothesis diagram for juvenile fish stranding on the lower Duncan River. Variables contributing to juvenile fish mortality are located above the dotted line, while items below the dotted line are processes feeding into the population size. Variables enclosed in boxes with dashed lines are not within direct management control and those in solid boxes are within management control.

1.2 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 lower Duncan River. It is anticipated that by addressing these objectives, an understanding of fish stranding impacts and the potential for making operating/monitoring improvements at Duncan Dam can be applied in future. The TOR did not state specific hypotheses to address primary objective 1. Therefore, objective 1 was addressed by assessing DDM operations in relation to stranding variables (Figure 1) 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 2nd 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 Duncan Dam flow reduction within the lower Duncan River 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 lower Duncan River, was the focus of Year 3 (2010 – 2011) and the present study (Year 4; 2011 - 2012) 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).

1.3 Study Design and Rationale

Since 2002, Golder has conducted fish stranding assessments on the lower Duncan River. 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. Several recommendations made in Year 3 on changes to study design to address gaps in the data set identified during the data analysis (Golder 2011) were implemented in the present study year.

1.3.1 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.

1.3.2 Pool Sampling

As pool sampling was the primary focus of previous study years, relatively precise pool stranding estimates for Rainbow Trout were obtained in Year 3 (Golder 2011). Therefore, sampling effort that focused on pools in the previous study was refocused in the present study to assess interstitial stranding in more detail.

1.3.3 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 the present study was increased.

2.0 METHODS

2.1 Study Area

The geographic scope of the study area for the Lower Duncan River Fish Stranding Impact Monitoring Program was the 11 km of mainstem lower Duncan River from DDM to the mouth of Kootenay Lake (Figure 2). This study area (collectively known as the lower Duncan River) includes the Duncan-Lardeau rivers confluence, as well as the Meadow, Hamill and Cooper Creek mouths. 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, and 2011). These stranding sites include 11 index stranding assessment sites and 39 non-index sites (Appendix A, Figures 1 to 7). The remaining habitat outside of the identified sites consists of banks with extreme gradient and is not considered to strand fish.

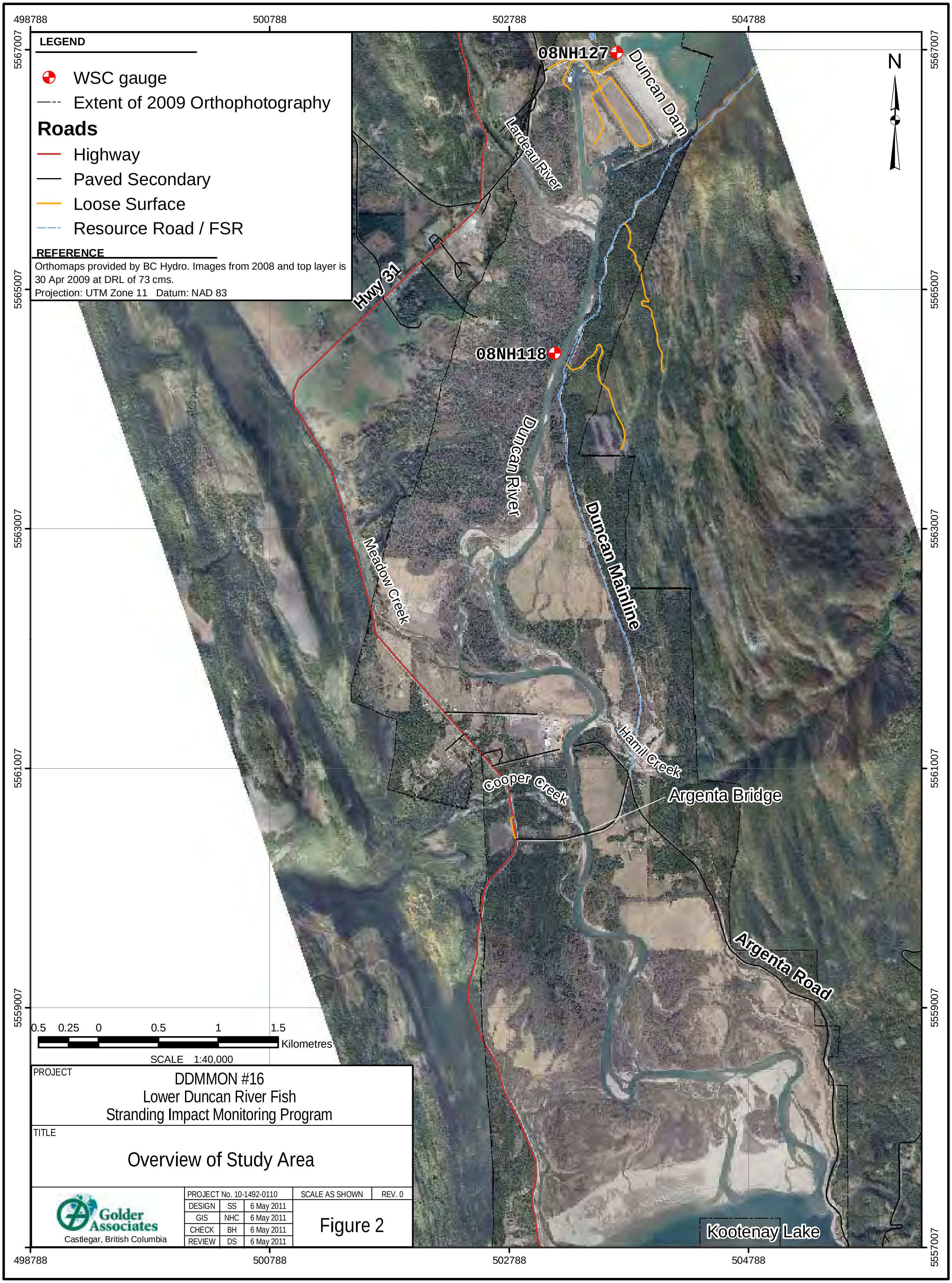
For the purpose of all WLR studies, the mainstem Duncan River was divided into five sections; these were termed Reach 1 (Km 0.0 – at DDM spill gates- to Km 0.8), Reach 2 (Km 0.8 to Km 2.6), Reach 3 (Km 2.6 to Km 5.7), Reach 4 (Km 5.7 to Km 6.7), and Reach 5 (Km 6.7 to Km 11.0 – at the mouth to Kootenay Lake).

2.2 Study Period

Stranding assessment activities were conducted from April 19, 2011 to January 20, 2012. For the two stranding assessments conducted prior to August 25, 2011 (RE 2011-04 and 05; Table 1), the methodology utilized in Year 3 was followed, assessing a portion of the index stranding sites. From RE 2011-06 onward, sampling methodology and site selection was modified (see Section 2.4) to address data gaps identified in Year 3. Each assessed reduction from DDM was assigned a reduction event number (RE) and Table 1 outlines all assessment activities during Year 4. In Year 3, the study period for each year was set between April 15 of that year, and continued until the following April 14, but for the purposes of the present data analysis, one assessment was not included in the analysis (March 1, 2012; RE 2012-02) in order to meet the reporting deadline.

Table 1: Chronology of sampling activities for the 2011 - 2012 Lower Duncan River Fish Stranding Impact Monitoring, Year 3 Program.

Date(s)	Reduction Event Number	Stranding Assessment Sampling Activities	Number of Index Sites Assessed	Number of Non-Index Sites Assessed
April 19, 2011	RE 2011-04	Index Stranding Assessment	5	0
June 1, 2011	RE 2011-05	Index and Non-Index Stranding Assessments	12	2
August 25, 2011	RE 2011-06	Index and Non-Index Stranding Assessments	6	4
September 25, 2011	RE 2011-07	Index and Non-Index Stranding Assessments	1	4
September 28, 2011	RE 2011-08	Index and Non-Index Stranding Assessments	2	2
October 1, 2011	RE 2011-09	Index and Non-Index Stranding Assessments	2	3
January 20, 2012	RE 2012-01	Index and Non-Index Stranding Assessments	3	4



498788
500788
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5557007
LEGEND
WSC gauge
Extent of 2009 Orthophotography
Roads
Highway
Paved Secondary
Loose Surface
Resource Road / FSR
REFERENCE
Orthomaps provided by BC Hydro. Images from 2008 and top layer is 30 Apr 2009 at DRL of 73 cms.
Projection: UTM Zone 11 Datum: NAD 83
N
08NH127
Duncan Dam
Lardeau River
Hwy 31
08NH118
Duncan River
Duncan Mainline
Meadow Creek
Hamil Creek
Cooper Creek
Argenta Bridge
Argenta Road
Kootenay Lake
0.5 0.25 0 0.5 1 1.5
Kilometres
SCALE 1:40,000
PROJECT
DDMMON #16
Lower Duncan River Fish
Stranding Impact Monitoring Program
TITLE
Overview of Study Area
Golder Associates
Castlegar, British Columbia
PROJECT No. 10-1492-0110
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 2

2.3 Physical Parameters

2.3.1 Water Temperature

Water temperatures for the lower Duncan River 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 lower Duncan River discharge, as it provides information on the magnitude of flow reductions along the majority of the river channel. All Duncan Dam releases and discharge data for the lower Duncan River were obtained from BC Hydro Power Records.

2.4 Fish Stranding Assessment Methodology

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. An update of the protocol is currently in preparation and will address up to date sampling methodologies and DDM operations.

According to the 2004 protocol, observations at each accessible pre-determined index site (AMEC 2004) are made regarding the presence or absence of pools that are isolated or dewatered by flow reductions, and the species and life stage of fish stranded. If presence/absence of fish in isolated pools cannot be determined due to the amount of cover, single pass backpack electrofishing is conducted to capture any stranded fish. Whenever possible, stranded fish are captured using dip nets or backpack electrofishing. The captured fish are identified, measured, enumerated, and then released into the mainstem. Since the primary goal is to assess fish stranding at all sites, fish salvage is a secondary objective. During stranding assessments, crews also look for evidence of interstitial fish stranding caused by DDM operations. In cases where the water in isolated pools has drained out prior to sampling, the effects of pool stranding cannot be differentiated from interstitial stranding; in these cases, all stranded fish are grouped into the pool stranding category. For the assessments included in the Year 3 analysis (September 15, 2006 – present), attempts have been made to conduct backpack electrofishing at all sites where fish are, or may be, present.

Because of the remote location of the lower Duncan River 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 lower Duncan River on Kootenay Lake. Since late 2007, debris jams have formed just between river kms (RKm) 4.1 and 4.5, preventing continuous boat access along the river. At the time this document was created, the downstream portions of the river can be accessed at higher river elevations by boat through a side channel located at RKm 4.2 and flows into Meadow Creek near its outlet into the lower Duncan River. As the river nears the mouth to Kootenay Lake, the channel meanders on a yearly basis, and access to the lower Duncan River from Kootenay Lake remains in question at lower DRL discharges.

In 2010, DDMMON-15 reviewed all lower Duncan River aquatic study reports and provided recommendations on the data collection methodology used during fish stranding assessments. This lead to the modification of assessment methodology at the onset of Year 3 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. In Year 4, assessment methodology was modified further to address data gaps identified in Year 3 (Sections 2.4.1 to 2.4.4). These modifications were completed prior to the stranding assessment conducted on August 25, 2011, and were implemented during all subsequent assessments.

2.4.1 Year 4 Stranding Site Selection

Prior to each fish stranding assessment, 10 sites were randomly selected from all identified stranding sites. This was accomplished by creating two strata (index and non-index) and then randomly selecting sites from each stratum to sample. The number of sites in each stratum selected for sampling was proportionate to the area dewatered in each stratum as a result of the assessed DDM flow reduction. The dewatered area at all sites was calculated using the site area regressions that were completed in Year 3 (Golder 2011).

2.4.2 Year 4 Pool Sampling

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; Appendix B, Plate 1); and,

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

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

For each pool, associated cover types (and percentages within the pool) 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,
- Overhanging vegetation (Appendix B Plate 3),
- Submerged Terrestrial Vegetation (Appendix B, Plate 4),
- 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.

2.4.3 Year 4 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 B, Plate 5). 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.

2.4.4 Year 4 Fish Life History Data

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

- Species,
- Length (mm; Appendix B, Plate 6),
- Condition (alive or dead),
- Salvaged (Yes/No); and,
- Habitat association (if possible).

Observed fish that were not captured and remained in the pool after sampling was completed were also documented. If the number of captured fish from a pool 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 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 Data Analysis

2.5.1 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. Modifications to the database's query output included adding fish stranding and dewatered area summary tables at the beginning of the output. Presently, 53 individual stranding assessments are into 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 in prep.)

2.5.2 Available Data for Analysis

The fish stranding data were extracted from the LDR stranding database and discharge data for the lower Duncan River were extracted from the LDR Access database developed for DDM WLR projects. Hydraulic model outputs from DDMMON-3 were provided by Northwest Hydraulic Consultants (NHC) from which Golder determined the area dewatered at each river discharge level for all surveyed sites using a regression modeling approach (Golder 2011). Whole river estimates of the dewatered area of low slope (0-4% gradient) and high slope (>4%) habitats were estimated using the following method:

1) a GIS specialist at Golder extracted the slope of the dewatered area for the whole river from the NHC digital elevation model (DEM) for three reductions (225cms to 200cms, 200cms to 125 cms, 125cms to 75cms),

2) the differences in the estimated dewatered area for each slope category were plotted and a linear regression model fitted to the differences for each slope type,

3) the regression equation's slope was extracted and used as a multiplier in order to determine the areas of high and low slope habitat exposed by each operational reduction,

4) the low and high slope areas were summed to obtain the total area in the LDR exposed for each reduction.

The estimates for overall juvenile Rainbow Trout population abundance were obtained from the juvenile habitat use component of the DDMMON-2 study (Thorley et al. 2011). The DDMMON-2 population estimates for Rainbow Trout and mountain whitefish were based on snorkel survey data obtained from September 15-21, 2010 (Thorley et al. 2011). Population abundance was only estimated in Fall 2010 so all discussion of population effects are in relation to this point estimate in time.

The assumptions, limitations and structure of each type of data are described in detail below and in the WLR study reports for DDMMON-2 and DDMMON-3. It was assumed that the data provided had been checked for accuracy, data entry or modeling errors. Data were also plotted and assessed during the analytic process to provide an additional level of quality assurance and control.

Within the LDR stranding database at the time of this analysis, there were 53 reduction events for which data were recorded from 2002-2012. The data set for analysis was limited to the 2010-2012 assessments since certain parameters of interest such as the slope of the habitat at a sample site were only collected in reductions monitored over the most recent years. The interstitial habitat sampling data were restricted to the data collected after August 25, 2011 since prior to this date, the sampling grids were not placed randomly. Thus, only five out of 53 assessments were used for interstitial analysis. The final reduction included in the data set used in this analysis was on January 20, 2012.

In the monitored stranding events, several species were observed and enumerated. However, for DDMMON-16, only Rainbow Trout and mountain whitefish were species of focus and for the analysis, there were only sufficient data to analyze the stranding rates of Rainbow Trout given the current modeling approach. The viability of analyzing mountain whitefish data should be reassessed in future years of the WLR study. Based on the length frequency plots and on previous work done on Rainbow Trout in the Lardeau River and LDR (e.g., Thorley et al. 2011) Rainbow Trout with a fork length between 90 and 155 mm were considered to be parr. Approximately 5% of the measured, stranded Rainbow Trout were considered parr by this definition. There were insufficient data on this life stage to carry out further analysis, but investigation of the effects of operational and environmental variables on parr should be reassessed in future years of the program. Rainbow Trout that were not measured or Rainbow Trout with a fork length less than 90 mm were considered to be fry in order to maximize the available data.

2.5.3 Data Analyses

To assess the first management question regarding whether the operational measures of the ASPD are effective, an historical assessment of the amount of area exposed by discharge reduction events was conducted as part of this analysis. 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.

The null management hypothesis two states that fish stranding events do not significantly impact fish populations in the lower Duncan River. In order to test this hypothesis, the total number of Rainbow Trout fry stranded from the 2010 cohort was estimated for the LDR and the resultant value compared to the snorkel-based riverine population estimate.

The stranding assessments focused on two types of habitat in which fish can strand in the LDR; pool habitat (P) and interstitial habitat (IS). The analyses are therefore structured so that the levels of interstitial stranding are estimated separately from the levels of pool stranding since different assumptions apply to each type of habitat.

Rainbow Trout fry interstitial and pool stranding data were analyzed using hierarchical Bayesian models (HBMs). The HBMs were implemented using the software package R 2.14.1 (R Development Core Team 2012) which interfaced with the Bayesian program JAGS (Just Another Gibbs Sampler) 3.2.0 (Plummer 2003) using the rjags and runjags libraries. JAGS distributions and functions are defined in Table 2. In general, the models assumed low information (Ntzoufras 2009) normal or gamma prior distributions. The posterior distributions, which were estimated using Gibbs sampling (Ntzoufras 2009), were derived from 1,500 Markov Chain Monte Carlo (MCMC) simulations thinned from the second halves of three MCMC chains of between 10^4 and 10^5 iterations in length depending on the analysis. 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 the primary parameters in the model (Gelman & Rubin 1992; Brooks & Gelman 1998; Gelman et al. 2004). The statistical significance of particular primary and derived parameters was assessed through the use of two-sided Bayesian p-values (Bochkina and Richardson 2007; Lin et al. 2009). Model adequacy was evaluated through the use of two-sided Bayesian p-values, the Deviance Information Criterion (Ntzoufras 2009).

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

Distribution/Function	Definition	Description
$\mathrm{dbern}(p)$	$p^x(1-p)^{1-x}$	Bernoulli distribution
$\mathrm{dbin}(p,n)$	$n!\,p^x(1-p)^{n-x}$	Binomial distribution
$\mathrm{dnorm}(\mu,\tau)$	$\sqrt{\tau/(2\pi)}\,\exp(-\tau(x-\mu)^2/2)$	Normal distribution
$\mathrm{dpois}(\lambda)$	$\exp\,(-\lambda)\lambda^x/x!$	Poisson distribution
$\log(x)$	$\log(x)$	Natural logarithm function
$\mathrm{logit}(x)$	$\log\,(x/(1-x))$	Logit function

2.5.3.1 Interstitial Stranding

The interstitial fish stranding density (fry/m^2) was estimated using a zero-inflated log-offset Poisson Bayesian model (Kery 2010). Key assumptions of the model included:

- The probability that a given grid is unsusceptible to fish stranding is greater than that expected under the Poisson distribution;
- The probability that a grid is unsusceptible varies with slope;
- The expected number of fish in a susceptible grid is the product of the expected density and the area of the grid; and ,
- The number of fish in a susceptible grid is described by the Poisson distribution

Variables that were included in preliminary versions of the model that were not found to be significant predictors of the probability of suitability with the current data included season and whether or not a site was an index site after gradient was accounted for. There were adequate data for comparing index and non-index sites, but season was not broadly represented through time, so it may emerge as a good predictor variable once additional data are collected. A parameter was not significant if the possibility that it had no effect could not be excluded.

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

Variable/Parameter	Description
$\boldsymbol{\beta}_0^\delta$	Log fry density
$\boldsymbol{\beta}_0^\pi$	Logistic grid suitability
$\boldsymbol{\beta}_g^\pi$	Effect of gth gradient on the logistic suitability
$\boldsymbol{\mu}_i^\eta$	Expected fry at ith grid
$\boldsymbol{\mu}_g^\pi$	Expected suitability at gth gradient
$\boldsymbol{\rho}_{i,g}$	Suitability of ith grid at gth gradient
$\mathbf{Area}_i$	Area of ith grid
$\mathbf{Fish}_{i,g}$	Fry in ith grid at gth gradient

Table 4: Prior probability distributions in the Bayesian analysis of interstitial fish stranding density.

Variable/Parameter	Prior Distribution
$\boldsymbol{\beta}_0^\delta$	$\mathbf{dnorm}(\mathbf{0}, \mathbf{5}^{-2})$
$\boldsymbol{\beta}_0^\pi$	$\mathbf{dnorm}(\mathbf{0}, \mathbf{5}^{-2})$
$\boldsymbol{\beta}_g^\pi$	$\mathbf{dnorm}(\mathbf{0}, \mathbf{5}^{-2})$

Table 5: Dependencies between variables and parameters in the Bayesian analysis of interstitial stranding density.

Variable/Parameter	Dependency
$\mathbf{log}(\boldsymbol{\mu}_i^\eta)$	$\boldsymbol{\beta}_0^\delta + \mathbf{log}\,(\mathbf{Area}_i\ )$
$\mathbf{logit}(\boldsymbol{\mu}_g^\pi)$	$\boldsymbol{\beta}_0^\pi + \boldsymbol{\beta}_g^\pi$
$\boldsymbol{\rho}_{i,g}$	$\mathbf{dbern}(\boldsymbol{\mu}_g^\pi)$
$\mathbf{Fish}_{i,g}$	$\mathbf{dpois}(\boldsymbol{\mu}_i^\eta * \boldsymbol{\rho}_{i,g})$

2.5.3.2 Pool Stranding

The number of fish stranding per pool was estimated using an over dispersed Poisson removal HBM (Ntzoufras 2009). Key assumptions of the model included:

- The expected number of fish per pool varies with season;
- The number of fish per pool is described by the Poisson-gamma mixture distribution, which is equivalent to the negative binomial distribution (Ntzoufras 2009);
- The electrofishing capture efficiency does not vary among pools or passes; and,
- The capture efficiency when dip netting or visually sampling a pool is 100%

Variables that were included in preliminary versions of the model that were not found to be significant predictors of the expected number of fish included "pool wetted area" (Year 3 analysis) and whether or not a pool was in an index site.

Table 6: Variables and parameters in the hierarchical Bayesian analysis of fish pool stranding.

Variable/Parameter	Description
$\mathbf{r_0}$	Extra-Poisson variation (overdispersion) in fry abundance
$\boldsymbol{\beta_0^{\eta}}$	Log fry abundance
$\boldsymbol{\beta_t^{\eta}}$	Effect of t^{th} season on log fry abundance
$\boldsymbol{\beta_m^{\rho}}$	Logistic capture efficiency of m^{th} method
$\boldsymbol{\epsilon_i}$	Effect of extra-Poisson variation on fry abundance at the i^{th} pool
$\boldsymbol{\mu_t^{\eta}}$	Expected pool abundance in t^{th} season
$\boldsymbol{\mu_m^{\rho}}$	Expected capture efficiency of m^{th} method
$\mathbf{N_{i,t}}$	Number of fry in i^{th} pool in t^{th} season
$\mathbf{n_{j,i,t}}$	Number of fry in i^{th} pool in t^{th} season prior to the j^{th} pass
$\mathbf{Fish_{m,j,i,t}}$	Number of fish caught in i^{th} pool in t^{th} season during j^{th} pass by m^{th} method

Table 7: Prior probability distributions in the hierarchical Bayesian analysis of fish pool stranding.

Variable/Parameter	Prior Distribution
$\mathbf{r_0}$	$\mathbf{dgamma(0.1, 0.1)}$
$\boldsymbol{\beta_0^{\eta}}$	$\mathbf{dnorm(0, 2^{-2})}$
$\boldsymbol{\beta_t^{\eta}}$	$\mathbf{dnorm(0, 2^{-2})}$
$\boldsymbol{\beta_m^{\rho}}$	$\mathbf{dnorm(0, 5^{-2})}$
$\boldsymbol{\epsilon_i}$	$\mathbf{dgamma(r_0\ , r_0\ )}$

Table 8: Dependencies between variables and parameters in the hierarchical Bayesian analysis of fish pool stranding.

Variable/Parameter	Dependency
$\log(\mu_t^{\eta})$	$\beta_0^{\eta} + \beta_t^{\eta}$
$\mathrm{logit}(\mu_m^{\rho})$	β_m^{ρ}
$N_{i,t}$	$\mathrm{dpois}(\mu_t^{\eta} * \epsilon_i\ )$
$n_{1,i,t}$	$N_{i,t}$
$n_{j+1,i,t}$	$N_{j,l,t}$-$Fish_{j,l,t}$
$Fish_{m,j,i,t}$	$\mathrm{dbin}(\mu_m^{\rho}, n_{j,i,t})$

2.5.3.3 Pool Density

The number of pools per unit area of exposed substrate was estimated using a log-offset over dispersed Poisson model (Ntzoufras 2009; Kery 2010). Key assumptions of the model included:

- The expected number of pools at a site is the product of the expected pool density drawn from a normal distribution and the area of the site; and,
- The number of pools at a site is described by a Poisson-gamma mixture model (Ntzoufras 2009)

Variables that were included in preliminary versions of the model that were not found to be significant predictors of the expected pool density included whether or not a site was an index site.

Table 9: Variables and parameters in the hierarchical Bayesian analysis of pool density.

Variable/Parameter	Description
σ_S	Standard deviation of the effect of site on log pool density
r_0	Extra-Poisson variation (overdispersion) in number of pools
β_0^{δ}	Log pool density
β_s^{δ}	Effect of s^{th} site on log pool density
ϵ_i	Effect of extra-Poisson variation on pool abundance at the i^{th} site survey
$\mu_{i,s}^{\eta}$	Expected pool abundance at i^{th} site survey at s^{th} site
$Area_i$	Area at i^{th} site survey
$Pool_{i,s}$	Number of pools at i^{th} site survey at s^{th} site

Table 10: Prior probability distributions in the hierarchical Bayesian analysis of pool density.

Variable/Parameter	Prior Distribution
σ_S^{δ}	$\mathrm{dgamma}(0.1, 0.1)$
r_0	$\mathrm{dgamma}(0.1, 0.1)$
β_0^{δ}	$\mathrm{dnorm}(0, 5^{-2})$
β_s^{δ}	$\mathrm{dnorm}(0, \sigma_S{}^{-2})$
ϵ_i	$\mathrm{dgamma}(r_0\ , r_0\ )$

Table 11: Dependencies between variables and parameters in the hierarchical Bayesian analysis of pool density.

Variable/Parameter	Dependency
$\log(\mu_{i,s}^{\eta})$	$\beta_0^{\delta} + \log(\text{Area}_i\ ) + \beta_s^{\delta}$
$\text{Pool}_{i,s}$	$\text{dpois}(\mu_{i,s}^{\eta} * \epsilon_i\ )$

2.5.3.4 Change in Site Area

The change in site dewatered area was estimated using a hierarchical Bayesian regression model. Key assumptions of the model included:

- The expected change in site dewatered area varies linearly with reduction in discharge and length of the site; and,
- The expected change in site dewatered area also varies by site and whether a site was an index or non-index site.

Table 12: Variables and parameters in the hierarchical Bayesian analysis of dewatered area.

Variable/Parameter	Description
σ_0	Standard deviation of the variation in area
σ_s	Standard deviation of the effect of site on change in area
β_0	Change in area
β_r	Effect of r^{th} site type (index versus non-index) on change in area
β_s	Effect of the s^{th} site on change in area
$\mu_{i,s,r}$	Expected change in area at the i^{th} reduction at the s^{th} site of the r^{th} type
$\Delta\text{Discharge}_i$	Change in discharge at the i^{th} reduction
$\Delta\text{Area}_{i,s,r}$	Change in area of the s^{th} site of the r^{th} type at the i^{th} reduction
$\text{Length}_{i,s,r}$	Length of the s^{th} site of the r^{th} type at the i^{th} reduction

Table 13: Prior probability distributions in the hierarchical Bayesian analysis of pool density.

Variable/Parameter	Prior Distribution
σ_0	$\text{dgamma}(0.1, 0.1)$
σ_s	$\text{dgamma}(0.1, 0.1)$
β_0	$\text{dnorm}(0, 1^{-2})$
β_r	$\text{dnorm}(0, 1^{-2})$
β_s	$\text{dnorm}(0, \sigma_S{}^{-2})$

Table 14: Dependencies between variables and parameters in the hierarchical Bayesian analysis of pool density.

Variable/Parameter	Dependency
$\mu_{i,s,r}$	$\Delta Discharge_{i} * (\beta_0 + \beta_r + \beta_s) * Length_{i,s,r}$
$\Delta Area_{i,s,r}$	$dnorm(\mu_{i,s,r}, \sigma_0)$

3.0 RESULTS

3.1 Duncan Dam Discharge Reductions and Ramping Rates

Hourly discharge at DRL during the study period ranged from 43.1 m^3/s on April 19, 2011 to 475.9 m^3/s on August 4 and 5, 2011. Hourly discharge from DDM ranged from 0 m^3/s on several days between early June and mid-July 2011, to 306.7 m^3/s on August 4, 2011 (Figure 3). The period of the year when DDM flows are not typically reduced occurs each spring/summer during recharge of Duncan Reservoir, at a time when dam discharge is typically at minimum flows. During this period there are temporary pulses of flow to meet daily average discharge requirements during bull trout migration through Duncan Dam. Also during this period, the Lardeau River discharge is typically high, which satisfies flow requirements for the protection of fish.

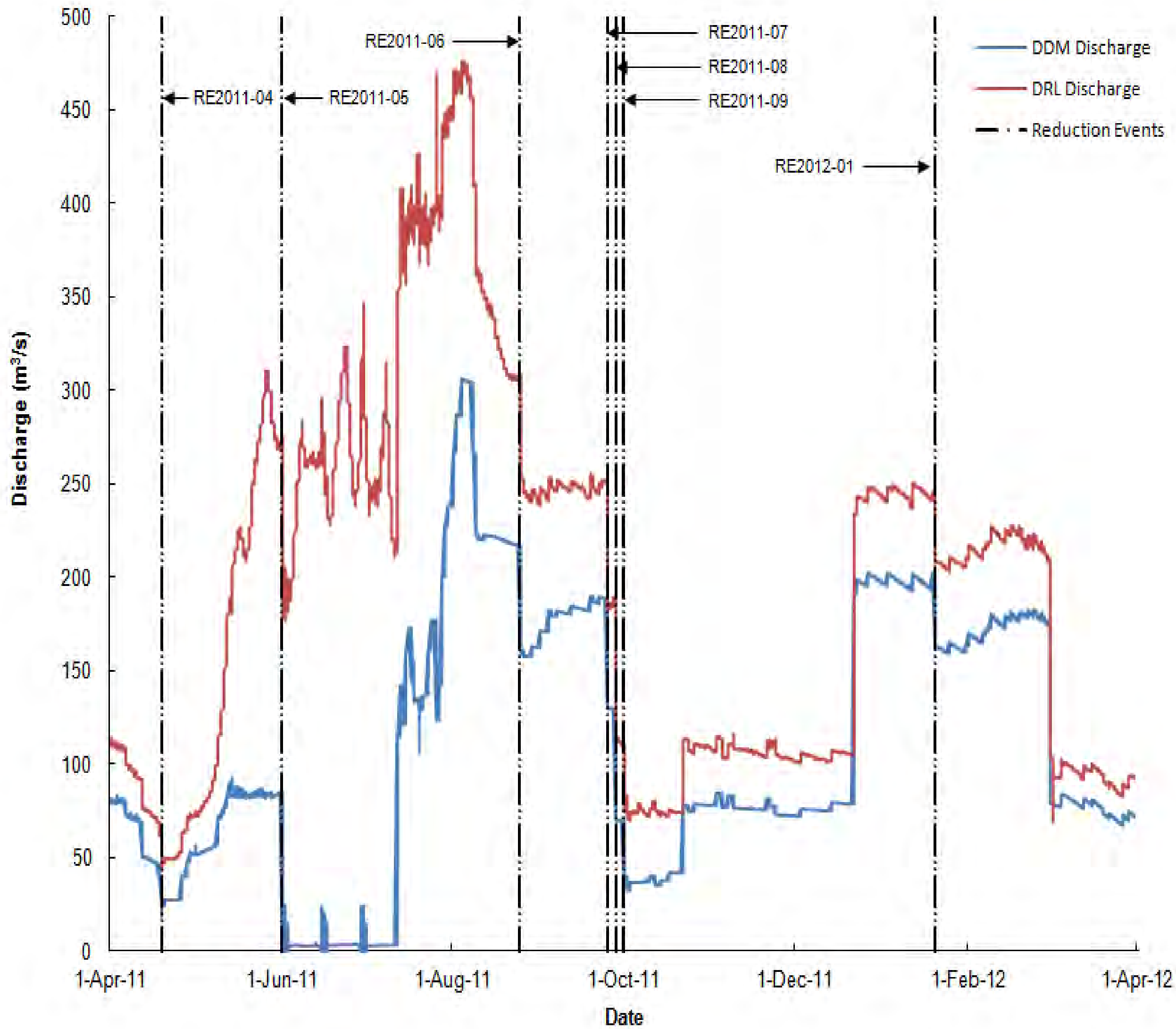

Figure 3: Hourly Discharge from Duncan Dam (DDM, blue line) and at the Duncan River below the Lardeau River (DRL, red line) from April 1, 2011 to April 1, 2012. Vertical dashed lines represent the timing of fish stranding assessments.

During the present study, seven REs occurred at Duncan Dam. The REs used a variety of flow ramping strategies, dependent upon the water level present at the onset of the REs; detailed descriptions of the flow reduction events are presented in Figure 3 and Table 15.

During REs that were assessed, DDM operations resulted in decreases of discharge that ranged from 28 to 82 m^3/s (Table 15). These decreases represent the discharge reductions at DDM and therefore do not represent actual rate of river stage changes or flow changes at particular downstream fish stranding sites. The values represent the net change in discharge at DDM at intervals ranging from 15 minutes to one hour.

Table 15: Summary of DDM flow reduction events, from April 15, 2011 to January 20, 2012, for those events when fish stranding assessments were conducted.

Date	Reduction Event (RE)	Initial Discharge	Resulting Discharge	Magnitude of Reduction	Ramping Description[a]	Flow Reduction Rationale
19-Apr-11	RE 2011-04	46 m^3/s	18 m^3/s	28 m^3/s	Down 7 m^3/s at 14:00 on 18-Apr, down 7 m^3/s at 08:00, 08:15 and 08:30 on 19-Apr.	Discharge reduced to compensate for low inflows.
01-Jun-11	RE 2011-05	82 m^3/s	0 m^3/s	82 m^3/s	Down 28 m^3/s at 07:00 and 08:00, down 26 m^3/s at 09:00.	To meet recreation water level targets in Duncan Reservoir.
25-Aug-11	RE 2011-06	217 m^3/s	161 m^3/s	56 m^3/s	Down 28 m^3/s at 08:00 and 09:00.	To meet late summer flow targets in the lower Duncan River.
25-Sep-11	RE 2011-07	190 m^3/s	130 m^3/s	60 m^3/s	Down 30 m^3/s at 07:30 and 08:30.	Onset of Kokanee protection flows.
28-Sep-11	RE 2011-08	130 m^3/s	70 m^3/s	60 m^3/s	Down 30 m^3/s at 07:30 and 08:30.	Kokanee protection flows.
01-Oct-11	RE 2011-09	70 m^3/s	40 m^3/s	30 m^3/s	Down 15 m^3/s at 07:30 and 08:30.	Final transition to Kokanee protection flows.
20-Jan-12	RE 2012-01	202 m^3/s	164 m^3/s	38 m^3/s	Down 19 m^3/s at 07:00 and 08:00.	Discharge reduced to meet reservoir targets.

[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 seven 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 16.

Table 16: Scientific names of fish species encountered during fish stranding assessments on the lower Duncan River, September 2006 to January 2012.

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 17). As discussed above, the focus of sampling shifted from index sites to non-index sites, 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 minimum number of pools (n = 92), and the maximum number of interstitial grids (n = 411) assessed in a single study year (Table 17).

Table 17: Sampling effort during reductions of each study year that were included in the present analysis.

Study Year	Number of Reductions Assessed	Number of Index Sites Assessed	Number of Non-Index Sites Assessed	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

During the stranding assessments conducted in the present study (2011/2012) and included in the data analysis, 1607 stranded fish were observed, representing eleven species (five sportfish and six non-sportfish species: Table 18). Kokanee young-of-the-year (YOY) were the most abundant sportfish encountered (52.5%), followed by Rainbow Trout (RB) juveniles at 26.0%. The majority of Kokanee YOY (*n* = 715 of 844) were observed during the sampling of RE2011-04.

The most abundant non-sportfish taxa recorded during the present study were unidentified Sculpin (5.0%). The second most abundant non-sportfish species observed were Longnose Dace (1.9%: Table 18).

Table 18: 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 2011.

Species and Life Stage			Total Number Recorded (Percent Composition) 2006/07	Total Number Recorded (Percent Composition) 2007/08	Total Number Recorded (Percent Composition) 2008/09	Total Number Recorded (Percent Composition) 2009/10	Total Number Recorded (Percent Composition) 2010/11	Total Number Recorded (Percent Composition) 2011/12
Sport Fish	Rainbow trout	Adult	0 (0)	0 (0)	0 (0)	1 (0.1)	0 (0)	0 (0)
		Juvenile	130 (37.1)	278 (11.5)	530 (33.2)	113 (12.3)	343 (25.2)	419 (26.0)
	Bull trout	Adult	0 (0)	0 (0)	0 (0)	4 (0.4)	0 (0)	0 (0)
		Juvenile	2 (0.6)	0 (0)	11 (0.7)	1 (0.1)	6 (0.4)	2 (0.1)
	Mountain whitefish	Adult	0 (0)	1 (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)
	Pygmy whitefish	Adult	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)
	Kokanee	Adult	0 (0)	97 (4.0)	572 (35.8)	112 (12.2)	42 (3.1)	55 (3.4)
		Y-O-Y	0 (0)	1695 (70.4)	85 (5.3)	109 (11.9)	83 (6.1)	844 (52.5)
	Burbot	Adult	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)
Non-sport fish	Longnose dace		117 (33.4)	15 (0.6)	103 (6.5)	273 (29.7)	551 (40.5)	30 (1.9)
	Dace spp.		0 (0)	0 (0)	0 (0)	12 (1.3)	1 (0.1)	0 (0)
	Slimy sculpin		0 (0)	13 (0.5)	11 (0.7)	62 (6.8)	39 (2.9)	3 (0.2)
	Torrent sculpin		0 (0)	1 (0)	1 (0.1)	0 (0)	0 (0)	3 (0.2)
	Prickly sculpin		0 (0)	0 (0)	0 (0)	0 (0)	2 (0.1)	0 (0)
	Sculpin spp.		23 (6.6)	16 (0.7)	65 (4.1)	62 (6.8)	165 (12.1)	80 (5.0)
	Sucker spp.		2 (0.6)	4 (0.2)	26 (1.6)	166 (18.1)	54 (4.0)	9 (0.6)
	Redside shiner		0 (0)	112 (4.6)	8 (0.5)	15 (1.6)	0 (0)	0 (0)
	Northern pikeminnow		0 (0)	0 (0)	2 (0.1)	0 (0)	15 (1.1)	4 (0.2)
	Peamouth chub		0 (0)	0 (0)	6 (0.4)	6 (0.7)	0 (0)	0 (0)
	Lake chub		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)
All Species Total			350	2409	1596	918	1361	1607

Note: individual study years include all stranding assessments over a one year period, commencing on April 15 and ending on April 14 of the following calendar year, with the exception of the present study year, when RE 2012-02 (March 1, 2012) was not included in the data analysis in order to meet reporting deadlines.

3.3 Stranding Site Slope

The linear regression between the discharge at the DRL WSC gauge and the amount of high slope (>4%) area had an R^2 value of 0.99 (Figure 4). In order to predict the amount of area in the river exposed by a particular drawdown, the discharge reduction is simply multiplied by the slope coefficient of 3863.6.

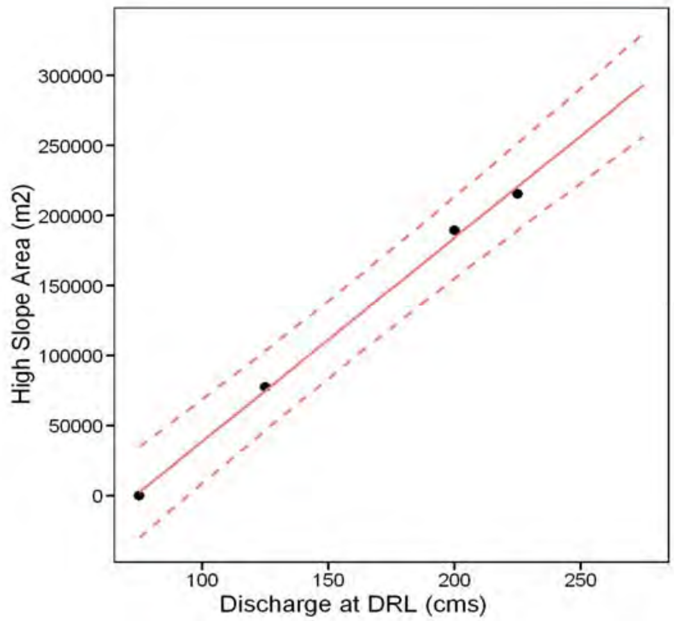

Figure 4: The total high slope (>4%) area in the LDR as predicted by discharge at the WSC DRL gauge. The red line shows the model predictions; dashed lines are the 95% prediction intervals.

The linear regression between the discharge at the DRL gauge and the amount of low slope (0-4%) area had an R^2 value of 0.99 (Figure 5). To predict the amount of area in the river exposed by a drawdown the difference in discharge was multiplied by 1453.8.

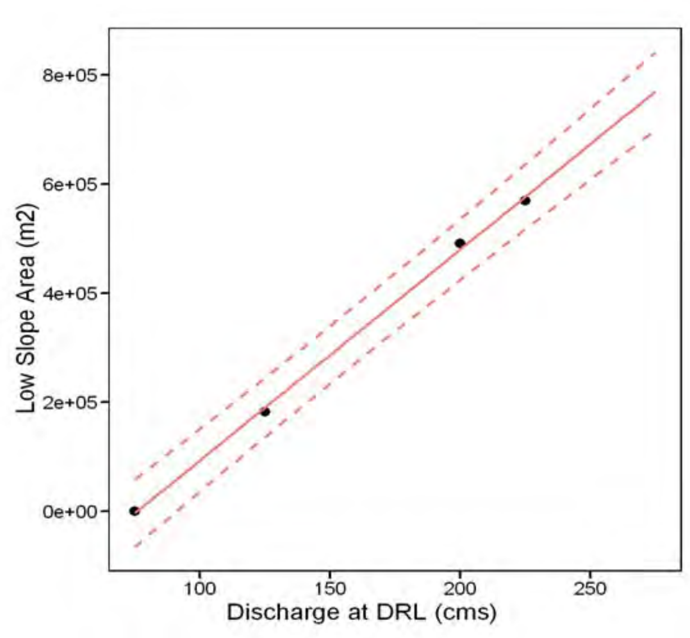

Figure 5: The total low slope (0-4%) area in the LDR as predicted by discharge at the WSC DRL gauge. The red line shows the model predictions; dashed lines are the 95% prediction intervals.

3.4 Fish Stranding During Assessments

The following stranding estimates provided in Section 3.4 refer to Rainbow Trout populations only as the limited dataset precluded estimates for the other species of interest.

3.4.1 Interstitial Stranding

As discussed in Section 2.6.2, the interstitial stranding analysis was for rainbow trout fry only, as there was insufficient data to analyze parr. The estimated level of interstitial stranding was higher in low slope habitat (0-4% gradient) than in high slope areas of the LDR (>4% gradient). A median density of 0.041 fish/m^2 was estimated for the low slope areas and 0.0008 fish/m^2 in the high slope areas (Figure 6). The zero values for the median and minimum estimates of interstitial stranding in the high slope habitat do not rule out the possibility that some interstitial stranding occurs in this type of habitat (Figure 6).

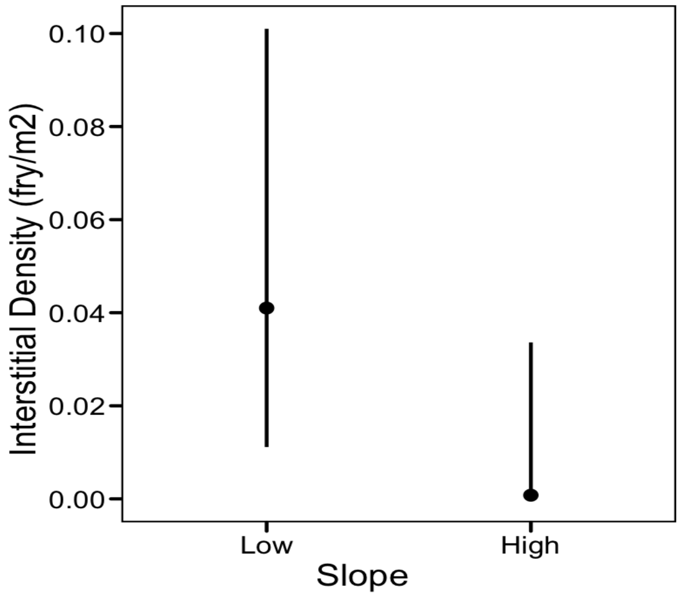

Figure 6: Median density of interstitially stranded Rainbow Trout fry with 95% credibility intervals.

Based on the estimated interstitial densities and the estimated areas of low and high slope habitat, the low slope habitat strands substantially more fry than the high slope habitat (Figure 7). Due to their lower magnitude and resultant lower areas of habitat dewatered, the reduction events in January, April, May and at the end of August stranded the least fish of all events over the 2010-2011 period (Figure 7). The March, June, September and October reduction events were estimated to strand the most fry (Figure 7). When summed over all habitats by year, the median estimate of the number of Rainbow Trout fry from the 2010 cohort to have been stranded interstitially in the 11 km length of the LDR was 71,261 with 95% credibility limits of 19,418 and 197,997.

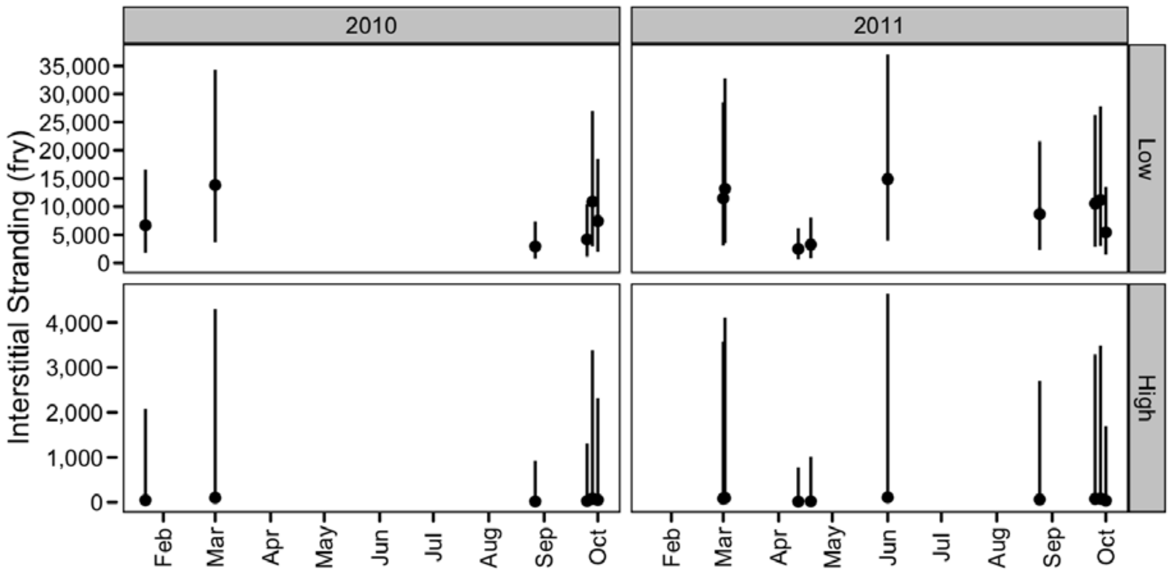

Figure 7: Median estimated number of Rainbow Trout fry interstitially stranded in relation to season and classified by year and by slope category where high is >4% and low is between 0-4% over all area in the LDR. The bars are 95% credibility intervals.

3.4.2 Pool Stranding

The pool density (number of pools per unit area) was estimated to allow the number of fish per pool (see below) to be expanded by area. There was no significant difference in the pool density between index and non-index sites (Figure 8). Nevertheless, the dewatered width of the index sites per m^3/s decrease in discharge was significantly greater than the non-index sites, which indicated that the index sites were generally of lower slope than the non-index sites (Figure 9).

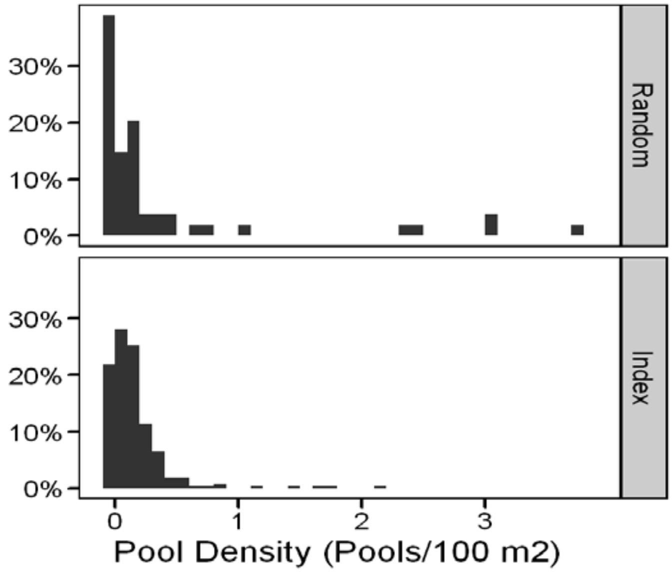

Figure 8: Percentage of sites by stratum type (non-index or index) vs. pool density per 100 m^2 for the LDR.

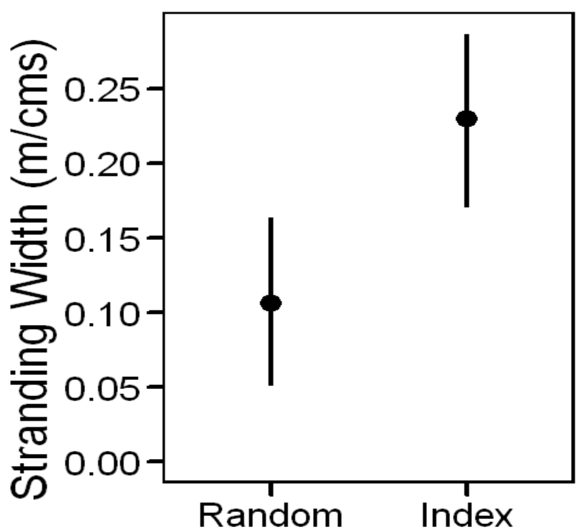

Figure 9: Stranding width by site type (non-index or index) for the LDR.

For the purposes of the analyses the efficiency of visual counts or dip netting, which were primarily conducted in pools with low complexity, was assumed to be 100%. The efficiency of the backpack removal electrofishing was estimated to have a median value of 46.4% with 95% credibility limits of 37.4% and

55.4%. The median number of fry per pool for the spring season (January – June) was estimated to be 0.80 (0.50 - 0.82) fish/pool (Figure 10). In contrast, the median number of Rainbow Trout fry stranded per pool in the fall (July to December) was estimated at 2.70 (1.98 - 3.88) (Figure 10). Based on the number of fish per pool and pools per area it was then possible to estimate the number of fish stranding in pools for individual reduction events (Figure 11). The resultant pool stranding estimates indicate very low levels of stranding in the months of January and April, moderate levels of stranding in March and June and the highest levels of stranding in August, September and October (Figure 11).

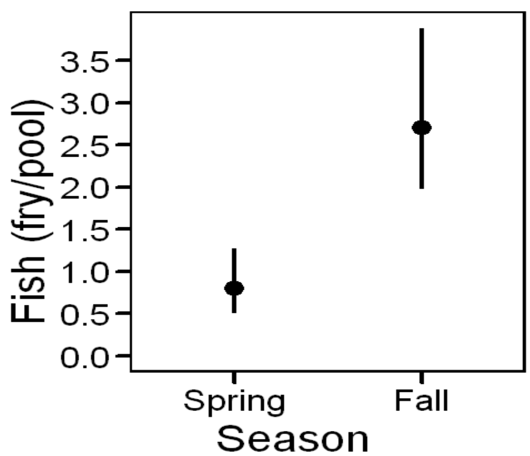

Figure 10: Median pool stranding levels for Rainbow Trout fry by season with 95% credibility intervals.

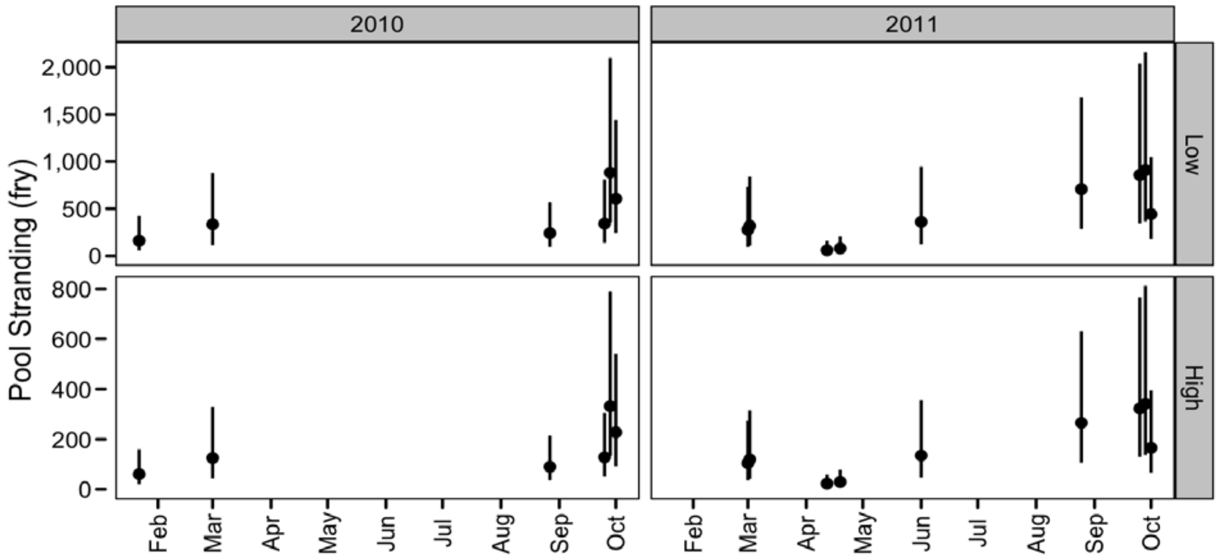

Figure 11: Estimated number of Rainbow Trout fry pool stranded in relation to season and classified by year and by slope category where high is >4% and low is between 0-4% over all area in the LDR. Median numbers of fish stranded and 95% credibility intervals are plotted.

3.4.3 Slope Effects on Overall Stranding Estimates

Overall estimates of Rainbow Trout stranding by habitat type and slope category are summarized in Figure 12. High slope habitat has a much lower stranding risk overall, particularly for interstitial stranding. Figure 12 also shows that for low gradient slopes, around 10 to 20 times as many fry were found stranded in interstitial habitats than in pools.

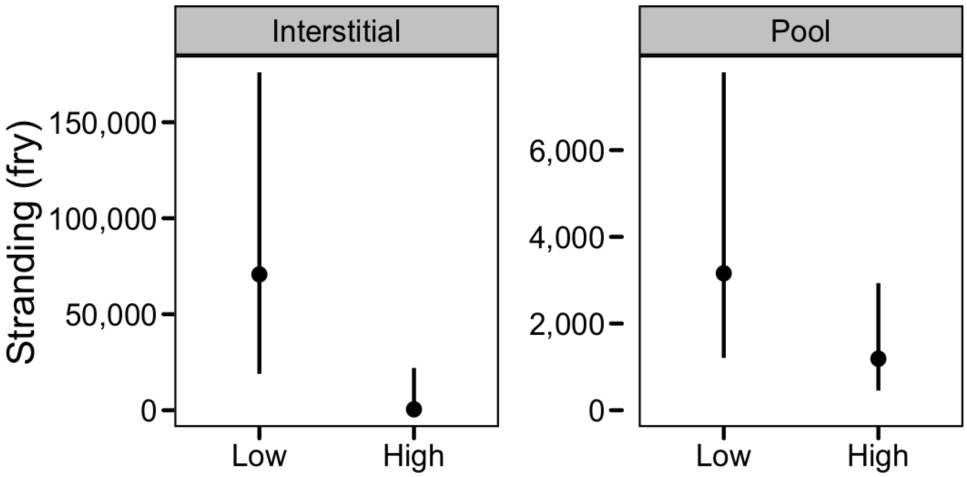

Figure 12: Estimated number of Rainbow Trout fry from the 2010 cohort stranded by interstitial and pool habitat. Median estimates and 95% credibility estimates are plotted by slope category where high is >4% and low is between 0-4% over all estimated stranding area in the LDR.

3.5 Differences between Pre-WUP and Post-WUP Operations

The amount of area exposed during all reductions occurring in a year was at its highest value in 2006 during pre-WUP operations (prior to 2008) with a value of 20.91 km^2. Area exposed was at its lowest value in 2010 during post-WUP operations, with a value of 12.5 km^2 (Figure 13). The overall mean area exposed during pre-WUP operations was 17.03 km^2/yr, in comparison to 13.52 km^2/yr during the post-WUP operational regime (Figure 13). 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 14).

Interannual variability in the discharge as assessed at the gauge at DRL overall is higher in the pre-WUP period, with the most radical difference seen in the October to January period. The current operational regime (i.e., 2008 to present) shows almost no deviation between years from the mean discharge levels during this period (Figure 14). During the late winter period (January to March) relatively stable mean discharge and less variation in discharge during post WUP operations is also evident. The other area that shows a marked difference is that the mean discharge trend in the spring months (March to May) in the post WUP operations has reversed directions from a gradual increase in the pre-WUP period to a gradual decrease (Figure 14).

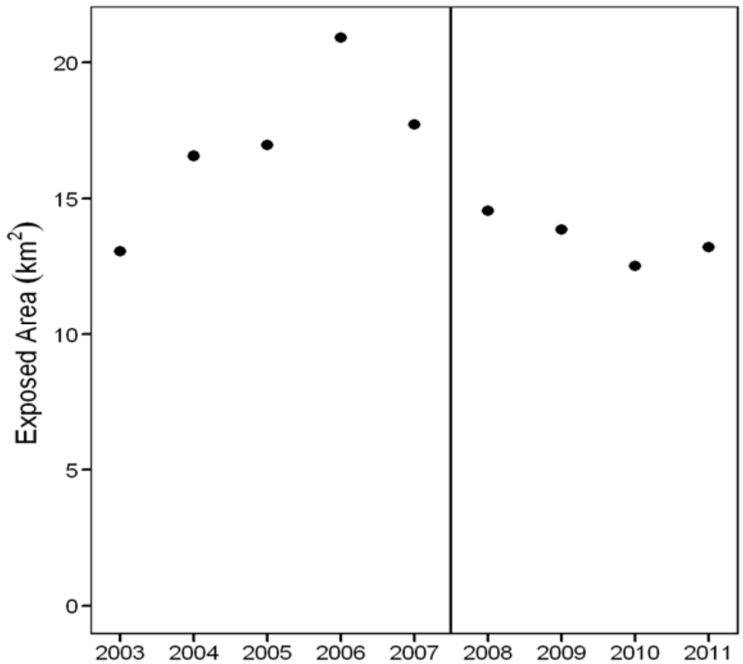

Figure 13: Total area exposed by all annual reductions in the LDR by year of operations. The vertical line denotes the beginning on WUP flows in 2008.

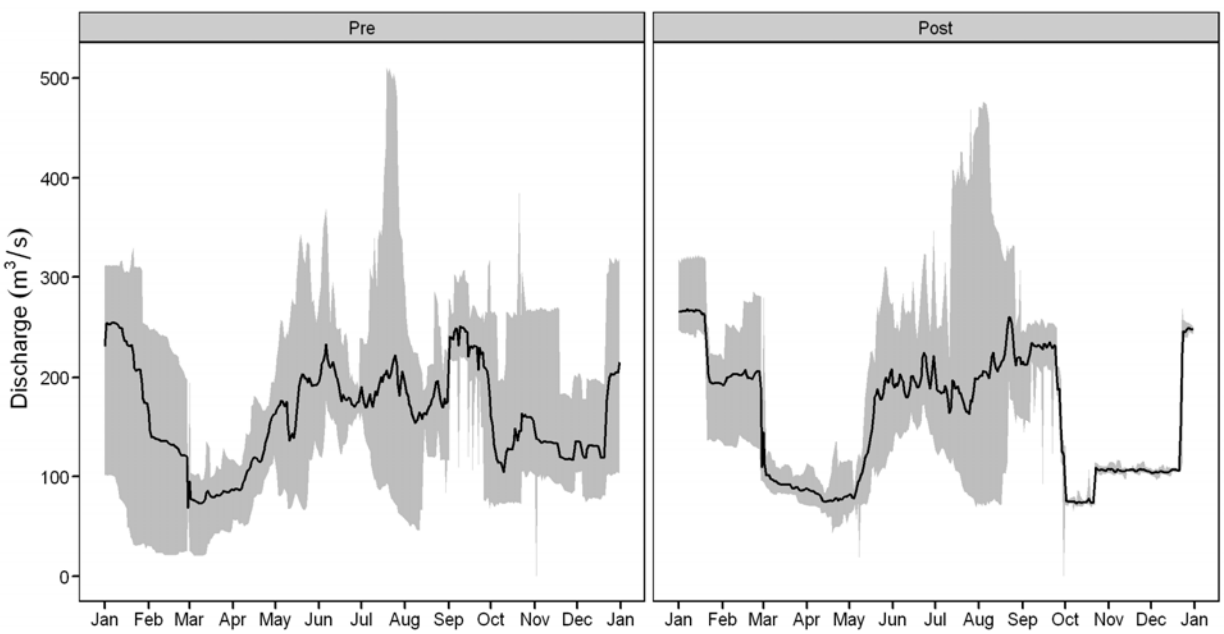

Figure 14: Minimum, maximum (grey ribbon) and mean (black line) discharge as measured at the WSC DRL gauge in the LDR by month during pre-WUP operations (2002-2007) and post-WUP operational implementation (2008-2011).

4.0 DISCUSSION

4.1 Current Duncan Dam Operations in Relation to Fish Stranding

4.1.1 Variables Affecting Fish Stranding

Management Question 1) (*How effective are the operating measures implemented as part of the ASPD program?)* was addressed by examining how current operations affect variables of interest relating to fish stranding, as well as the differences between the pre- and post WUP regimes. Under the water license, two large flow reductions occur on an annual basis, in late September to early October for Kokanee protection and in late winter 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.

As shown in Figure 1, 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 and time of day (Golder 2011). These variables were analysed and discussed as part of DDMMON-1 – Lower Duncan River Ramping Rate Monitoring (Poisson and Golder 2010). Based on the recommendations of that study, the ramping rate of flow reductions was reduced to ensure that a stage change 10 cm/hr or less occurs at the majority of identified stranding sites. This ramping rate is believed to minimize the risk of fish stranding and is supported by studies conducted in Norway (Halleraker et al. 2003), which recommended the same ramping rates to reduce stranding rates of salmonids, especially after an extended period of stable flows.

DDMMON-1 showed a trend of increased fish stranding at night (Poisson and Golder 2010). For the LDR, there is evidence to suggest night time flow reductions in the autumn period lead to more stranding of the species of interest than daytime reductions. This is consistent with use and activity patterns for juvenile salmonids observed in past work (AMEC 2003) and the current juvenile habitat use study program (DDMMON-2) (Thorley et al. 2011). All flow reductions under present DDM operations occur in the daytime period, which follows the recommendations of DDMMON -1 & -15 and allows for fish stranding assessments immediately after the reductions.

Operational variables related to stranding that are not currently addressed in the ASPD are wetted history and season. Although not statistically significant, a trend for increased stranding risk with longer wetted history during the fall season was identified by DDMMON-1 (Poisson and Golder 2010). Further study and analysis is required to confirm or deny this trend throughout the year. Time of year was a major variable in fish stranding with the highest observed rates of stranding occurring late summer/early fall, and late winter, which coincide with the two large annual flow reductions that typically occur under the current DDM operating regime.

4.1.2 Pre- and Post-WUP Operating Regimes

The assessment of the amount of area of exposed habitat per year due to LDR discharge reductions suggests that post-WUP flows have resulted in the dewatering of less habitat compared to pre-WUP operations. Interannual variability in discharge has also been reduced under post-WUP operations. However, the analysis of the amount of habitat dewatered does not take into account the magnitude or rate of the discharge reductions that may affect the stranding risk. Therefore, the estimates should be considered preliminary. Area was selected rather than the number of reductions per year because the area exposed relates directly to the models used to assess the stranding levels for pool and interstitial stranding. Using the scaling factor of the area exposed per year for back casting gives the opportunity to look back in time at possible stranding levels in years when the sampling methodologies were not comparable. The sampling program assessing the fish stranding levels through time has had different methodologies and varying study foci through the years so it is not possible to analyze fish stranding data from the pre-WUP and post-WUP periods. Therefore indirect assessments can be made on the effectiveness of the post-WUP operations, which according to the current preliminary estimates should have reduced the amount of Rainbow Trout fry stranding in the LDR.

4.2 Fish Stranding Summary

Management Question 2) (*What are the levels of impact to resident fish populations associated with fish stranding events on the lower Duncan River?)* was addressed. The species of interest for this study program are Rainbow Trout and Mountain Whitefish. Within the analyzed stranding assessment data set, seventeen different species were encountered (six sportfish and eleven non-sportfish species), but only a few species had significant numbers of stranded individuals (Table 18). Although these species were not a focus of the population estimation or the stranding surveys, this suggested that mortalities related to stranding of species other than juvenile Rainbow Trout, juvenile Kokanee and Longnose Dace may not result in a population level effect. To obtain estimates of stranding rates for the species of interest and determine population level impacts, several factors related to stranding were explored.

4.2.1 Interstitial and Pool Stranding Rates

Although the estimated numbers of interstitially stranded fish in the lower Duncan River for low slope habitats are relatively high and the estimates are still uncertain, they are more precise than the estimates obtained in Year 3 (Golder 2011). Random sampling of interstitial habitat began in August 2011, and is a new part of the program. While interstitial stranding is likely to be biologically important, the current estimates may be upwardly biased for the following reasons:

1) under the current classification scheme some “dewatered pools” may have been considered to be interstitial habitat;

2) although randomly selected, the sample of grids may nonetheless be biased by slope (dewatered area will be greatest in low gradient habitats, which could result in an large areas of habitat that are unsampled) and proximity to the shoreline.

One of the main classifications of fish stranding habitat that has persisted since the commencement of stranding and ramping studies in the Columbia Basin is the difference in stranding rates between interstitial and pool habitats. This distinction has a basis in the differences in the sampling methodologies and search efficiencies associated with each habitat type, but increasingly this division may not be serving the analysis optimally. The current variability in the estimated rates of interstitially stranded fish may simply be the natural variability associated with stranding, but it may be driven by study methodologies (i.e. some of the sampling grids that may have inadvertently contained pools that become dewatered).

Reductions on the LDR occur throughout the year, and given that estimates are the product of the density and amount of area exposed, the estimated number of Rainbow Trout fry stranded in interstitial habitat varies with the size of the reduction. The interstitial data collected to date are insufficient to robustly test the effect of season, species life history, or the magnitude of the reduction on interstitial stranding densities, which are likely to be important factors. For example preliminary data exploration suggests that a large magnitude reduction (>50 m^3/s) creates a higher density of stranded fish than a small magnitude reduction

(<50 m^3/s). As more data are collected, seasonal effects and the magnitude of the reduction should also be incorporated into the modeling effort.

Previous analysis showed that residual wetted area of pool was not a predictive variable (Poisson 2011). The densities of pool stranded and interstitially stranded Rainbow Trout fry were estimated for the two categories of habitat slope and were then multiplied by the area available to strand for each habitat type in order to derive whole river estimates.

4.2.2 Slope of Dewatered Area

The habitat categories and amounts of high and low slope habitat that were calculated in 2010 from the DDMMON-3 hydraulic modeling data were an excellent starting point for expanding the local estimates of stranding to the entirety of the LDR. Based on the current data analysis, considerably more low slope habitat is dewatered during flow reductions from DDM, and the dewatered low slope habitats have substantially more fish interstitially stranded following flow reductions than high slope habitats. However, these estimates may be overestimates since the estimates of area are based on any dewatered zone of the river being categorized as stranding habitat and field assessments demonstrate that this is not likely the case. In addition, the estimates of area dewatered were only conducted from three outputs of the early version of the DDMMON-3 hydraulic model. The estimated area would likely be more accurate if the model was run at more discharge levels (ideally all discharge levels encountered prior to and after the reduction events through time) and the relationship between index and non-index site areas plotted.

The categories of high and low slope were based on values in the literature from previous stranding work (Bauersfeld 1978; Flodmark 2004). However, like all categorical data, there are associated issues of forcing a continuous variable to be a categorical one. If the relationship between stranding risk and slope is not linear, and if the distribution of sampling locations or grids is not completely random with respect to the range of slopes within a category, the relative proportions of sampling on certain slopes will be higher which could bias the estimates.

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*. The first stage of addressing this hypothesis is to assess the characteristics of the two site strata; index and non-index. When assessing the width of the dewatered area at sampled sites, the index sites showed greater variability and overall greater width than the non-index sites with an equivalent change in discharge. This reveals that index sites tend 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 suggests that other than being lower gradient and therefore exposing more area, stranding rates do not differ substantially between index and non-index (stranding per lineal km of river). Overall, index sites strand more fish because more area dewaters at these sites during flow reductions.

The complexity of the pools was not consistently recorded over all pools over the time frame of the data set so the analysis of the pool stranding rates did not assess the differences in pool complexity between index and non-index sites. With consistent data collection in future study years, these differences could be examined. The 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. Based on these analyses however, index sites appear to be biased toward higher stranding rates and the data indicates they strand more fish. Therefore, hypothesis H_{01} is rejected.

4.2.4 Impacts of Stranding on Species of Interest

The second specific hypothesis to address Management Question 2 states*: Fish populations in the LDR are not significantly impacted by fish stranding events*. Only the dataset on stranded Rainbow Trout fry had sufficient data to support the current type of modeling and analysis.

4.2.4.1 Rainbow Trout

Estimates for the number of Rainbow Trout fry stranded in pools were relatively precise and low when compared to population abundance estimates of LDR fry and interstitial estimates. The median estimated value over all pool area in the LDR within a year was 4,347 fry stranded with a 95% credibility interval ranging from 1,663 to 10,725. This median value amounts to about 8% of the fall Rainbow Trout fry abundance estimated for 2010 (Thorley et al. 2011). There was a seasonal component to pool stranding, 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 estimated 2010 population of Rainbow Trout fry within the LDR as modeled from DDMMON-2 data was 54 000 (95% credibility intervals range from 39 726-75 249) (Thorley et al. 2011). Fall estimates of rainbow fry in the Lardeau River ceased in 2007, but the mean abundance of fry in 2007 was 53, 871 fish (Thorley et al. 2011). Summing the fish from the Lardeau River with the 2010 fry estimates from the LDR would give a mean estimated population of Rainbow Trout fry in both systems of ~108, 000 fry. The sum of the estimated interstitial and pool stranded fish for the 2010 fry year had a median value of 75,607 and minimum and maximum 95% credibility intervals of 20,627 and 208,722 respectively. The pool stranding is a small proportion of this total stranding estimate with only 5.7% of the total stranding estimated attributable to the pool stranding mechanism. Based on the likely overestimated interstitial stranding estimates, combined with the precise pool estimates from the present dataset, hypothesis H_{02} is rejected. 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 interstitial stranding estimate be refined. The current median estimate of interstitial stranding is 71,261 (95% credibility intervals 18,694-197,997). Furthermore, the abundance estimates for Rainbow Trout fry in the LDR might be under or overestimates. The uncertainty is such that the abundance of the juvenile Rainbow Trout could be either halved or doubled (Thorley et al. 2011).

Determining how these estimates of mortality due to stranding affect the population is difficult (Golder 2011). Several factors affect fish populations including: predation, outmigration, food availability, availability of suitable rearing habitats, winter mortality, as well as inter- and intra-specific competition. As discussed in the Year 3 report, whether stranding events kill fish that would have died because of these factors, or kill fish which would survive these factors is unknown (Golder 2011).

4.2.4.2 Mountain Whitefish

Over the course of the study year, only 43 stranded Mountain Whitefish were documented, all of which were observed in the spring season (RE2011-04 and 05). Similarly, 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. 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 2011-2012 investigations.

4.3 Summary

In summary, this monitoring program provides a better 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 follow recommendations made by the various studies conducted on the LDR. Based on collected data and the life history of species present in the system, risk of stranding increases in certain seasons and with longer wetted histories under the present operation regime of DDM. To obtain more accurate total stranding estimates for species of interest in the LDR, the estimates of interstitial stranding need further refinement. These refinements and other recommendations discussed in Section 5.0 will work towards reducing the uncertainly around stranding estimates.

5.0 RECOMMENDATIONS

Recommendations for next year of the DDMMON-16 Lower Duncan River Fish Stranding Impact Monitoring Program were developed by Golder and Poisson and are as follows:

1) Develop and implement a schedule to conduct annual stock assessments of target fish species. The stock assessment methodology should be refined to allow more accurate and up to date abundance estimates of the impacts of stranding on these species.

2) Explore the feasibility of obtaining spring fry abundance estimates to further refine annual stock assessments. As in Year 3 (Golder 2011), the Rainbow Trout surveys conducted by DDMMON-2 may be expanded to obtain abundance estimates for this program.

3) Continue following Year 4 methodology in future stranding assessments, with the exception of the classification of stranding mechanisms. It is recommended that dried pools be classified as a third stranding mechanism. A possible working field definition of a dried pool would be a low point which when disconnected from the mainstem would create a wetted pool but was drained at the time of assessment. This will strengthen the existing dataset and allow more accurate estimates of fish stranding in the Lower Duncan River.

4) Calculate the wetted history resulting from all flow reductions within the current dataset. This may allow for a continuation of the analyses conducted in DDMMON-1 on the relationship between wetted history and fish stranding.

5) Conduct additional work on slopes within each identified stranding site. This can be achieved with additional DDMMON-3 model runs at varying discharge levels. Also, the LIDAR data and the most accurate Digital Elevation Model possible from DDMMON-3 would need to be made available to GIS personnel of the DDMMON-16 study team. The slopes of each sample site can be calculated and modeled as a continuous variable to better understand the relationship between stranding risk and slope.

These recommendations will focus sampling effort and are designed to build on the current data set. The focus of future study years should be on the refinement of interstitially stranded fish estimates throughout the system, as well as improving the determination of the slope of sampled sites. Sampling methods should remain such that comparisons with historical data can be maintained.

6.0 REFERENCES

AMEC 2003. Diel fish use of aquatic habitats in the Lower Duncan River, BC. Data Report. October 2002. Prepared for BC Hydro and Power Authority, Castlegar, BC. 31 p. + 3 app.

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

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

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

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

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

Bochkina, N. and S. Richardson. (2007). Tail posterior probability for inference in pairwise and multiclass gene expression data. Biometrics. 63:1117-1125.

Brooks, S. and A. Gelman (1998). Alternative methods for monitoring convergence of iterative simulations. *Journal of Computational and Graphical Statistics* **7**: 434-455.

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

Gelman, A., J. B. Carlin, H. S. Stern and D. B. Rubin (2004). *Bayesian Data Analysis*. Second Edition. Boca Raton, Florida, CRC Press.

Gelman, A. and D. Rubin (1992). Inference from iterative simulation using multiple sequences. *Statistical Science* **7**: 457-511.

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

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

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

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

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

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

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

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

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

Irvine, R. L. a. D. C. S. (2009). Gap Analysis for Lower Duncan River Ramping Program DDMMON -1 *Memo Report Prepared for: BC Hydro Columbia Generation Area*. Castlegar.

Kery, M. (2010). *Introduction to WinBUGS for Ecologists*. China, Academic Press.

Ntzoufras, I. (2009). *Bayesian Modeling Using WinBUGS*. Hoboken, New Jersey, John Wiley & Sons, Inc.

Plummer, M. (2003). JAGS: A Program for Analysis of Bayesian Graphical Models Using Gibbs Sampling, Proceedings of the 3rd International Workshop on Distributed Statistical Computing (DSC 2003), March 20–22, Vienna, Austria. ISSN 1609-395X.

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

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

R Development Core Team (2011). R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

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

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

7.0 CLOSURE

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

GOLDER ASSOCIATES LTD.

ORIGINAL SIGNED ORIGINAL SIGNED ORIGINAL SIGNED

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APPENDIX A

Summary of Identified Stranding Sites

LEGEND

- Reach Break
- River Kilometre Downstream of Duncan Dam
- Paved Road
- Highway
- Boat Launch
- WSC Gauge

Stranding Site

- Index
- Non-Index

REFERENCE

- Orthophotography from 2008 and 30 Apr 2009, water level discharge 73 cms at DRL.
- Streams from BC Corporate Watershed Base.
- Thalweg and Side Channels digitized from orthophoto.
- Roads from Geobase National Road Network, with additions by NHC
- Projection: UTM Zone 11N Datum: NAD 83

100 50 0 100 200 300 Metres

SCALE 1:5,000 METRES

PROJECT

DDMMON #16
Lower Duncan River Fish
Stranding Impact Monitoring

TITLE

Lower Duncan River
Stranding Assessment Sites

Golder Associates
Castlegar, British Columbia

PROJECT No. 10-1492-0110			SCALE AS SHOWN	REV. 0
DESIGN	SS	6 May 2011	Figure A1	
GIS	SS	6 May 2011		
CHECK	BH	6 May 2011		
REVIEW	DS	6 May 2011		

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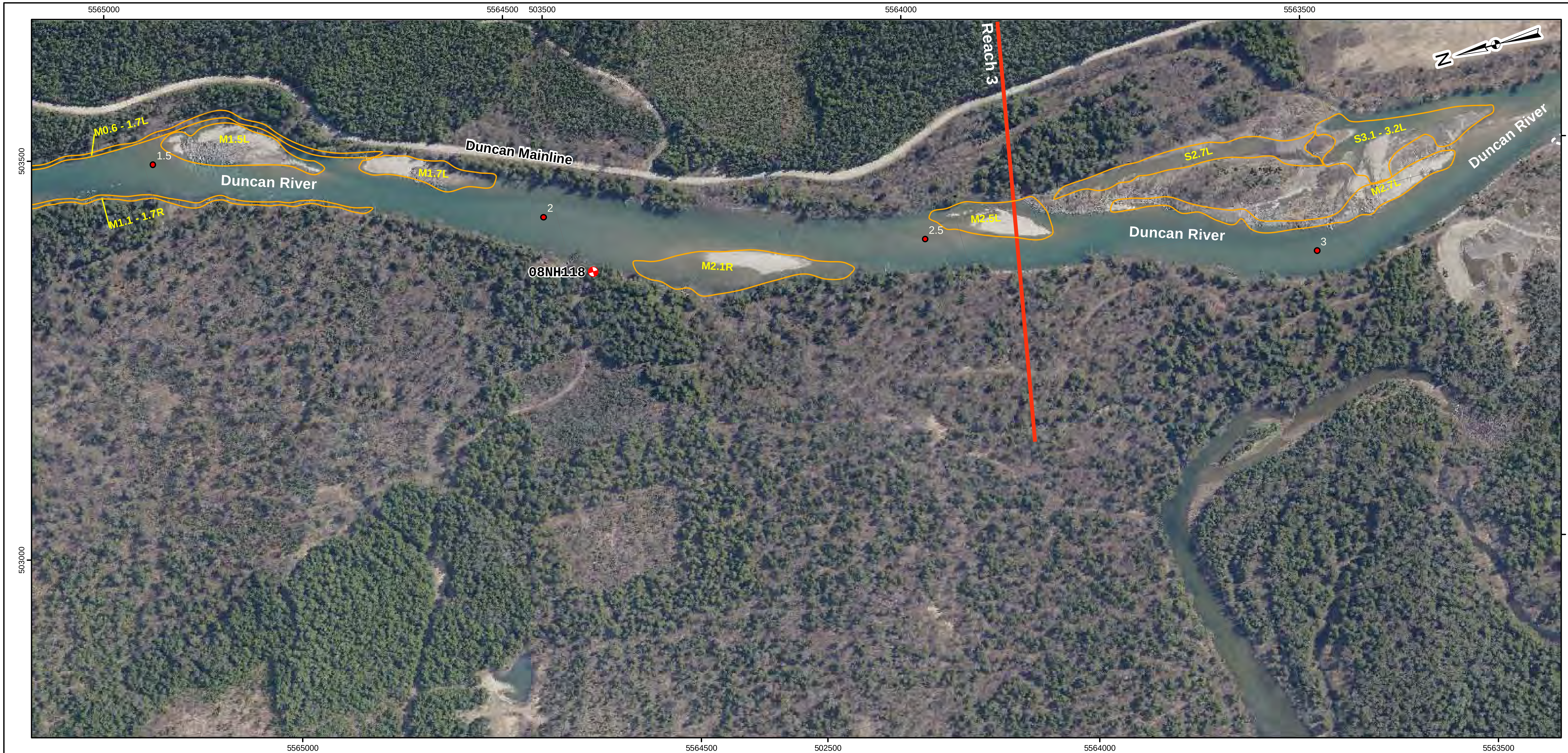

LEGEND

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REFERENCE

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- Streams from BC Corporate Watershed Base.
- Thalweg and Side Channels digitized from orthophoto.
- Roads from Geobase National Road Network, with additions by NHC
- Projection: UTM Zone 11N Datum: NAD 83

100 50 0 100 200 300 Metres

SCALE 1:5,000 METRES

PROJECT

DDMMON #16
Lower Duncan River Fish
Stranding Impact Monitoring

TITLE

Lower Duncan River
Stranding Assessment Sites

Golder Associates
Castlegar, British Columbia

PROJECT No. 10-1492-0110			SCALE AS SHOWN	REV. 0
DESIGN	SS	6 May 2011	Figure A2	
GIS	SS	6 May 2011		
CHECK	BH	6 May 2011		
REVIEW	DS	6 May 2011		

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LEGEND

- Reach Break
- River Kilometre Downstream of Duncan Dam
- Paved Road
- Highway
- Boat Launch
- WSC Gauge

Stranding Site

- Index
- Non-Index

REFERENCE

- Orthophotography from 2008 and 30 Apr 2009, water level discharge 73 cms at DRL.
- Streams from BC Corporate Watershed Base.
- Thalweg and Side Channels digitized from orthophoto.
- Roads from Geobase National Road Network, with additions by NHC
- Projection: UTM Zone 11N Datum: NAD 83

100 50 0 100 200 300 Metres

SCALE 1:5,000 METRES

PROJECT: DDMMON #16 Lower Duncan River Fish Stranding Impact Monitoring

TITLE: Lower Duncan River Stranding Assessment Sites

Golder Associates
Castlegar, British Columbia

PROJECT No. 10-1492-0110			SCALE AS SHOWN	REV. 0
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GIS	SS	6 May 2011		
CHECK	BH	6 May 2011		
REVIEW	DS	6 May 2011		

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LEGEND

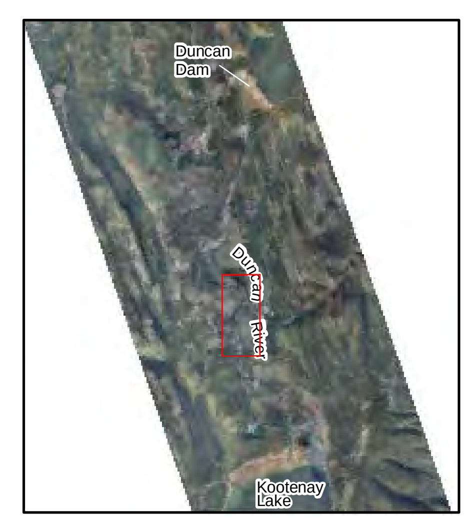
- Reach Break
- River Kilometre Downstream of Duncan Dam
- Paved Road
- Highway
- Boat Launch
- WSC Gauge

Stranding Site

- Index
- Non-Index

REFERENCE

- Orthophotography from 2008 and 30 Apr 2009, water level discharge 73 cms at DRL.
- Streams from BC Corporate Watershed Base.
- Thalweg and Side Channels digitized from orthophoto.
- Roads from Geobase National Road Network, with additions by NHC
- Projection: UTM Zone 11N Datum: NAD 83



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SCALE 1:5,000 METRES

PROJECT
DDMMON #16
Lower Duncan River Fish
Stranding Impact Monitoring

TITLE
Lower Duncan River
Stranding Assessment Sites

Golder Associates
Castlegar, British Columbia

PROJECT No. 10-1492-0110			SCALE AS SHOWN	REV. 0
DESIGN	SS	6 May 2011	Figure A4	
GIS	SS	6 May 2011		
CHECK	BH	6 May 2011		
REVIEW	DS	6 May 2011		

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LEGEND

- Reach Break
- River Kilometre Downstream of Duncan Dam
- Paved Road
- Highway
- Boat Launch
- WSC Gauge

Stranding Site

- Index
- Non-Index

REFERENCE

- Orthophotography from 2008 and 30 Apr 2009, water level discharge 73 cms at DRL.
- Streams from BC Corporate Watershed Base.
- Thalweg and Side Channels digitized from orthophoto.
- Roads from Geobase National Road Network, with additions by NHC
- Projection: UTM Zone 11N Datum: NAD 83

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100 50 0 100 200 300 Metres

SCALE 1:5,000 METRES

PROJECT	
	DDMMON #16 Lower Duncan River Fish Stranding Impact Monitoring
TITLE	Lower Duncan River Stranding Assessment Sites

Golder Associates
Castlegar, British Columbia

PROJECT No. 10-1492-0110			SCALE AS SHOWN	REV. 0
DESIGN	SS	6 May 2011	Figure A5	
GIS	SS	6 May 2011		
CHECK	BH	6 May 2011		
REVIEW	DS	6 May 2011		

LEGEND

- Reach Break
- River Kilometre Downstream of Duncan Dam
- Paved Road
- Highway
- Boat Launch
- WSC Gauge

Stranding Site

- Index
- Non-Index

REFERENCE

- Orthophotography from 2008 and 30 Apr 2009, water level discharge 73 cms at DRL.
- Streams from BC Corporate Watershed Base.
- Thalweg and Side Channels digitized from orthophoto.
- Roads from Geobase National Road Network, with additions by NHC
- Projection: UTM Zone 11N Datum: NAD 83

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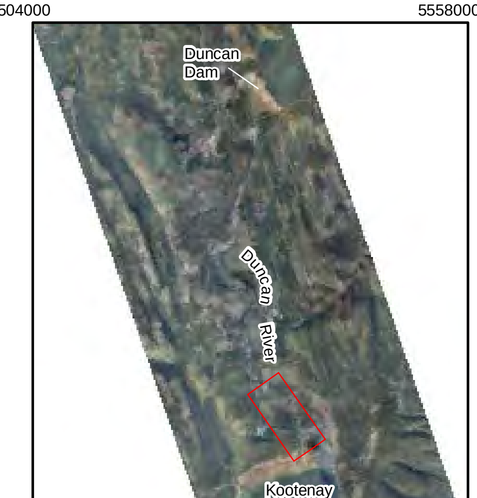

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SCALE 1:5,000 METRES

PROJECT	DDMMON #16 Lower Duncan River Fish Stranding Impact Monitoring
TITLE	Lower Duncan River Stranding Assessment Sites

Golder Associates
Castlegar, British Columbia

PROJECT No. 10-1492-0110			SCALE AS SHOWN	REV. 0
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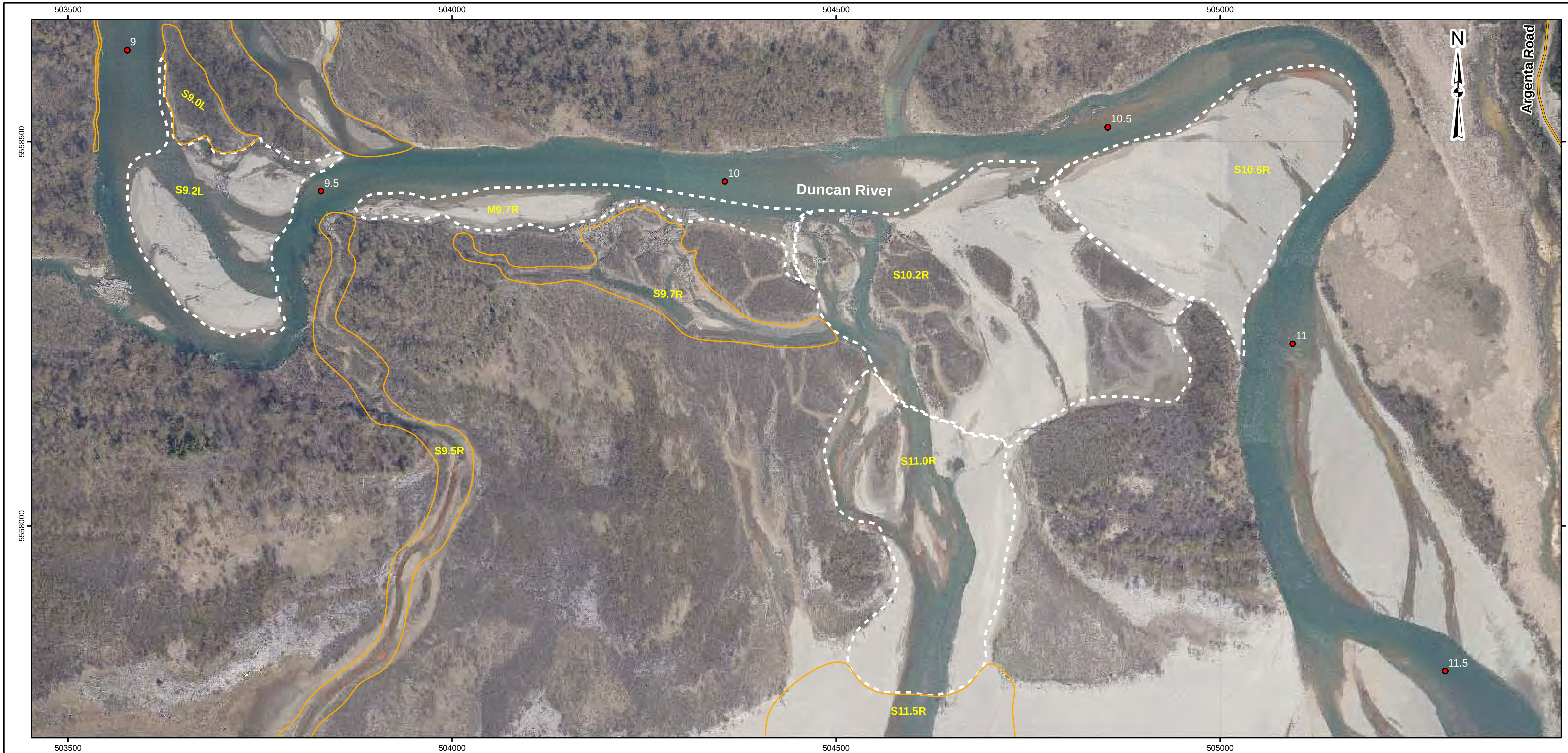
GIS	SS	6 May 2011		
CHECK	BH	6 May 2011		
REVIEW	DS	6 May 2011		



LEGEND

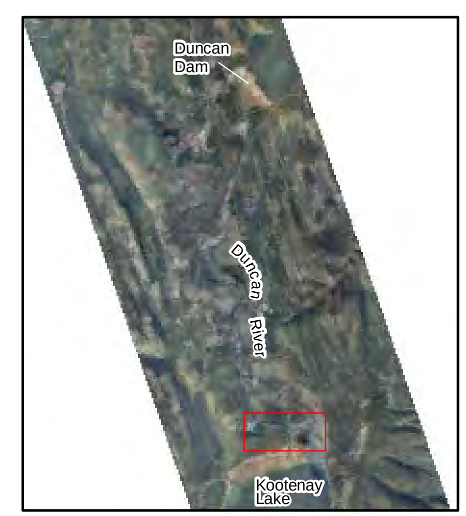
- Reach Break
- River Kilometre Downstream of Duncan Dam
- Paved Road
- Highway
- Boat Launch
- WSC Gauge

Stranding Site

- Index
- Non-Index

REFERENCE

- Orthophotography from 2008 and 30 Apr 2009, water level discharge 73 cms at DRL.
- Streams from BC Corporate Watershed Base.
- Thalweg and Side Channels digitized from orthophoto.
- Roads from Geobase National Road Network, with additions by NHC
- Projection: UTM Zone 11N Datum: NAD 83



100 50 0 100 200 300 Metres

SCALE 1:5,000 METRES

PROJECT
DDMMON #16
Lower Duncan River Fish
Stranding Impact Monitoring

TITLE
Lower Duncan River
Stranding Assessment Sites

Golder Associates
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PROJECT No. 10-1492-0110			SCALE AS SHOWN	REV. 0
DESIGN	SS	6 May 2011	Figure A7	
GIS	SS	6 May 2011		
CHECK	BH	6 May 2011		
REVIEW	DS	6 May 2011		

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APPENDIX B

Photographic Plates


Plate 1   Zero to Low complexity pool at site M08.R, October 1, 2011.


Plate 2   Moderate to High complexity pool at site M0.8R, October 1, 2011.

Plate 3 Overhanging vegetation cover at site SLARD0.3R on August 25, 2011.

Plate 4 Submerged terrestrial vegetation cover in isolated pool, site S4.1R on August 25, 2011.

Plate 5   Interstitial survey at M0.8R on October 1, 2011.

Plate 6   Juvenile Rainbow Trout on measuring board, site M2.7L on October 1, 2011.

At Golder Associates we strive to be the most respected global group of companies specializing in ground engineering and environmental services. Employee owned since our formation in 1960, we have created a unique culture with pride in ownership, resulting in long-term organizational stability. Golder professionals take the time to build an understanding of client needs and of the specific environments in which they operate. We continue to expand our technical capabilities and have experienced steady growth with employees now operating from offices located throughout Africa, Asia, Australasia, Europe, North America and South America.

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