Duncan Dam Project Water Use Plan

Lower Duncan River

Reference: DDMMON-15

Lower Duncan River Stranding Protocol Development and Finalization

Study Period: Year 4/5 (2012 – 2013)

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DDMMON-15

Lower Duncan River Stranding Protocol Development and Finalization Year 5 - 2012-2013

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Cover Photo: View of earth fill portion of Duncan Dam from BC Hydro launch, September 26, 2012.

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Glossary of Acronyms

ASPD	Adaptive Stranding Protocol Development
ATU	Accumulated Thermal Unit
AUC	Area Under the Curve
BT	Bull Trout
CC	Consultative Committee
DAL	Duncan River Above Lardeau
DCL	Duncan Dam Discharge Channel
DDM	Duncan Dam
DDMMON	Duncan Dam Monitoring Program
DRL	Duncan River Below Lardeau Gauging Station
ERH	Effective Rearing Habitat
FWCP	Fish and Wildlife Compensation Program
GBT	Gas Bubble Trauma
GOO	General Operating Order
HCTF	Habitat Conservation Trust Fund
HUC	Habitat Use Curve
HSI	Habitat Suitability Index
IHNV	Infectious Haematopoietic Necrosis Virus
Kcfs	Thousands of Cubic Feet Per Second
KO	Kokanee
LCR	Lower Columbia River
LDR	Lower Duncan River
LKR	Lower Kootenay River
LLOG	Low Level Operating Gate
MCSC	Meadow Creek Spawning Channel
MW	Mountain Whitefish
NTU	Nephelometric Turbidity Unit
PED	Potential Egg Deposition
PM	Performance Measures
RKm	River Kilometre
RB	Rainbow Trout
SPOG	Spillway Operating Gates
TGP	Total Gas Pressure
WLR	Water License Resource
WSC	Water Survey of Canada
WUA	Wetted Usable Area
WUP	Water Use Planning

Executive Summary

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. The Adaptive Stranding Protocol Development (ASPD) Program was developed to assess the results from a defined group of WUP monitoring studies, as well as conduct a review of relevant literature to make more informed decisions related solely to the stranding impacts of Duncan Dam (DDM) operations.

The scope of this report is to review the completed and ongoing WLR studies on the lower Duncan River (LDR) relevant to fish stranding issues, and recommend relevant conclusions for integration into the existing Adaptive Stranding Protocol (ASP). The ASP is scheduled to be updated regularly throughout the WLR review period, and finalized at the end of the review for long-term implementation.

Under the current Water Use Plan, 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 and Columbia River Treaty objectives. In addition, there are several smaller reductions that occur throughout the year to manage water resources and power generation at other facilities.

The fall DDM flow reductions to reach Kokanee protection flows pose the greatest stranding risk to juvenile Rainbow Trout and other small bodied species or juvenile stage fish in the LDR. The seasonal effect on stranding numbers was found to be significant ($p < 0.05$), with median fall stranding estimates over three times higher than those for winter/spring. Prior to the Water Use Plan, the stranding risks of recently emerged and juvenile Mountain Whitefish and Kokanee in the spring prior to the onset of freshet were considered high. Also, observations of stranding led to BC Hydro commitments to improve its stranding salvage and mitigation protocols. More work is required to inform the stranding protocol on both the updated understanding of risk and potential mitigation of larval and juvenile Mountain Whitefish and Kokanee stranding.

Juvenile Rainbow Trout utilize the LDR year round. Abundance estimates indicated that first winter Rainbow Trout survival is approximately 25%. Juvenile Mountain Whitefish numbers in the LDR are high during the fall and appear to decline precipitously (90-99%) during the winter. The possibility that the low Mountain Whitefish counts are due to extremely low observer efficiencies cannot currently be excluded.

Based on the information reviewed in this document and the fact that an update to the stranding protocol was recently completed (Golder 2013), there are no recommended specific updates to the existing Adaptive Stranding protocol at this time. Major outstanding data gaps identified by this program include:

1) There is still high uncertainty related to the abundance and stranding estimates of the species and life stages of interest. Data to refine these estimates continues to be collected, and as the dataset grows each year, the uncertainty related to this estimate will likely continue to decrease.

2) Mountain Whitefish emergence and juvenile rearing timing and requirements should be reviewed against typical operations in the late winter period to determine if there are critical operations that may contribute to stranding and any flexibility in the operations to address stranding risk. Current knowledge suggests that the increase to peak winter flows in late December may redistribute or strand the broadcast-spawned eggs and increase mortality in the egg to fry stage.

3) The Rainbow Trout spawning and incubation habitat use defined by recent studies has not been evaluated against DDM flows to determine if there are operations that can mitigate stranding of redds or the backwatering resulting from low Dam flows that is presumed to be one reason for high egg mortality and redd stranding immediately below the dam. Practices of moving or wetting exposed redds in the DDM tailrace have not been reviewed by this report to and are not completed annually.

4) Revisions to the Kokanee protection flows in the WUP were implemented in fall 2013 and were not part of this review. Future reports for the DDMMON-4 monitoring program will include an assessment of the effectiveness of those revisions and provide recommendations that are consistent with ongoing agency reviews. Egg to fry survival for Kokanee averaged 23% in the LDR (0-50% range) and evaluation of operations to mitigate dewatering and egg stranding should be assessed.

The predictive TELEMAC 2D hydraulic modeling tool created in the DDMMON-3 program has not been interpreted against stranding observations collected from DDMMON-16.

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Shelley Wenaas (Burnaby)

James Baxter (Castlegar)

Alf Leake (Burnaby)

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Table of Contents

TABLES

FIGURES

APPENDICES

1.0 INTRODUCTION

1.1 Background

Fish habitat dewatering on the lower Duncan River (LDR) occurs as a result of both natural flow variations (e.g., from the Lardeau River) and Duncan Dam operations (DDM; completed in 1967). On an annual basis there are approximately six flow reductions from DDM outside the freshet period (May – August), that typically range from 28 m^3/s (1 kcfs) to a maximum of 113 m^3/s (4 kcfs), which is the daily reduction limit allowed under the Columbia River Treaty. During the freshet period, the Lardeau River and other tributaries are the primary water contributors to the LDR during the reservoir refill period, and these tributaries can cause diel variation in LDR discharge and can result in still, backwatered habitat between DDM and the confluence. Annual hydrographs from 2008 to 2013 for the Duncan/Lardeau system since the implementation of Duncan Dam Project Water Use Plan (BC Hydro 2007), are presented in Figure 1. Water level information for the same period for Duncan Reservoir and Kootenay Lake are presented in Figure 2 and Figure 3, respectively. These plots show the annual operations at DDM, as well as the annual elevations in the Duncan Reservoir and Kootenay Lake which strongly influence those operations.

In addition to altering available fish habitat, flow reductions from DDM increase the potential for fish stranding in the LDR. Stranding risk varies with fish species and the life stages present in nearshore habitats that are dewatered during flow reductions. Stranding of fish and fish eggs is a common effect of water level variations below flow regulating facilities. Due to their habitat use tendencies and swimming ability, adult and sub-adult fish pose a low risk for stranding (Golder 2008b). An exception are adult Kokanee, which are susceptible to stranding in dewatered sidechannels during the fall flow reductions to reach Kokanee egg protection flow levels. These fall reductions also pose a stranding risk to the early life history stages (eggs, larvae, fry, and young-of-the-year) of other fish species present in the system.

The risk of fish stranding and potential mortality due to DDM operations depends on several environmental and operational factors reviewed in Irvine (2009) and in an updated review in section 6.0 of this document. The operational factors include: water level reduction (the difference between maximum and minimum elevation within the day); the duration of water level reductions; the speed of dewatering (ramping rate); and the wetted history (e.g., habitats infrequently submerged for shorter periods of time are less likely to be populated by fish). Channel morphology also plays a key role in assessing the risk of stranding, as cobble substrate, shallow depressions, and low gradients all increase fish stranding risk through both increased isolated pool formation and interstitial stranding.

Fish stranding resulting from dam discharge changes was raised as a significant issue in 2002 by provincial and federal fisheries agencies, the public, and through the Water Use Plan (WUP) consultation process (BC Hydro 2005). As a result, a key objective of the DDM WUP process was to maximize fish abundance and diversity in the Duncan River Watershed and specifically, reduce fish stranding risk and maximize habitat suitability and productivity in the LDR. Through the initial stages of the WUP process, several DDM flow management operations were modified, or proposed for modification, towards reaching these end objectives based on known or suspected life history timing, distribution, and habitat utilization. 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 management program will be implemented over the WUP review period based on the results from a collective group of monitoring studies (Section 5.0). The Adaptive Stranding Protocol for Managing Fish Impacts in the LDR Associated with Flow Reductions at

DDM ("the Protocol") (Westcott et al. 2013) will incorporate relevant findings from these studies to adapt operations, salvage and assessment activities towards improving the understanding and mitigation of stranding impacts. Such changes will require regulatory and BC Hydro approval before being finalized within the Protocol.

1.2 Program Scope

As stated in the Terms Of Reference (TOR; BC Hydro 2009), the end objective of this monitoring program is to finalize a flow reduction protocol, including stranding response procedures (e.g., fish salvaging), flow reduction procedures at DDM, internal and external correspondence procedures, stranding assessment methodology, and reporting requirements. This program integrates the findings of a selected group of WUP studies focused on stranding, as well as findings from independent literature sources, to aid in the reduction of fish stranding in the LDR through refinements to DDM operations. Only the diel timing, magnitude of reductions and rate of changes to DDM flow releases are to be considered in the protocol (ramping rates): revisions to flow targets and prescriptions will be reviewed as part of any future water planning process. This will be accomplished through annual review and reporting, the refinement and implementation of the ASPD, and the finalization of the Protocol.

1.2.1 Report Scope

This annual summary document tracks information related to management questions associated with the ASPD up to December 2013 and provides recommendations consistent with both the approach of the ASPD and the results of the studies. It has been developed as a framework to track ASPD objectives, hypotheses and management questions, and document progress toward meeting those objectives on an annual basis. Annual revisions or amendments will be made to this document as required to ensure it is consistent with the best available information. This report also addresses requirements for 2013 and the long-term approach to Protocol finalization in 2018. Protocol recommendations are vetted through BC Hydro to ensure the operating recommendations are appropriate and consistent with the Duncan Dam Water Use Plan before discussing with regulatory agencies. Interagency discussions also take place at the Columbia Operations Fisheries Advisory Committee (COFAC) annual meetings. Final changes to the Protocol and updated references to the Protocol integrated with its operating orders for the DDM are managed by BC Hydro.

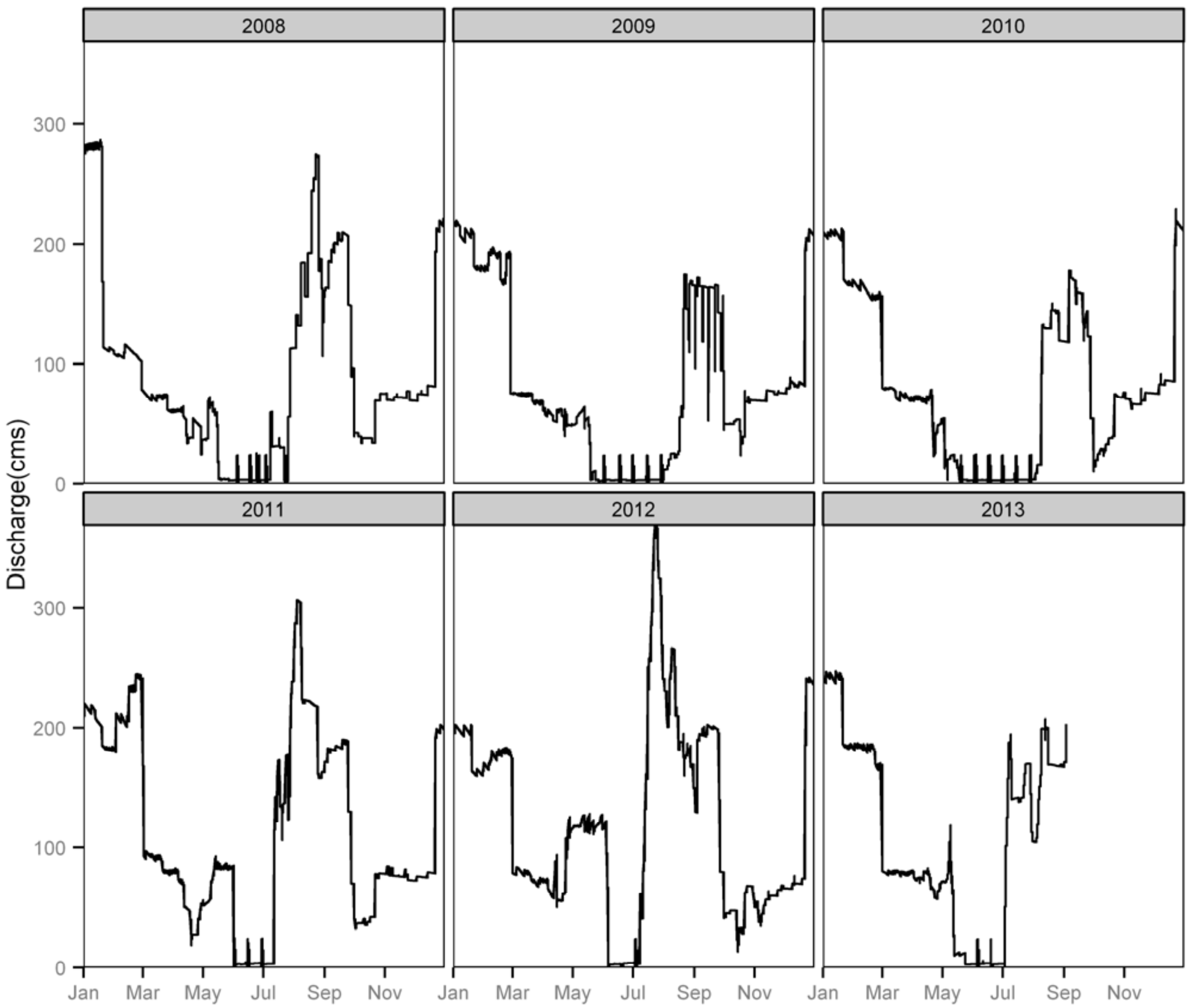

Figure 1: 2008-2013 Hourly discharge (m^3/s) from Duncan Dam.

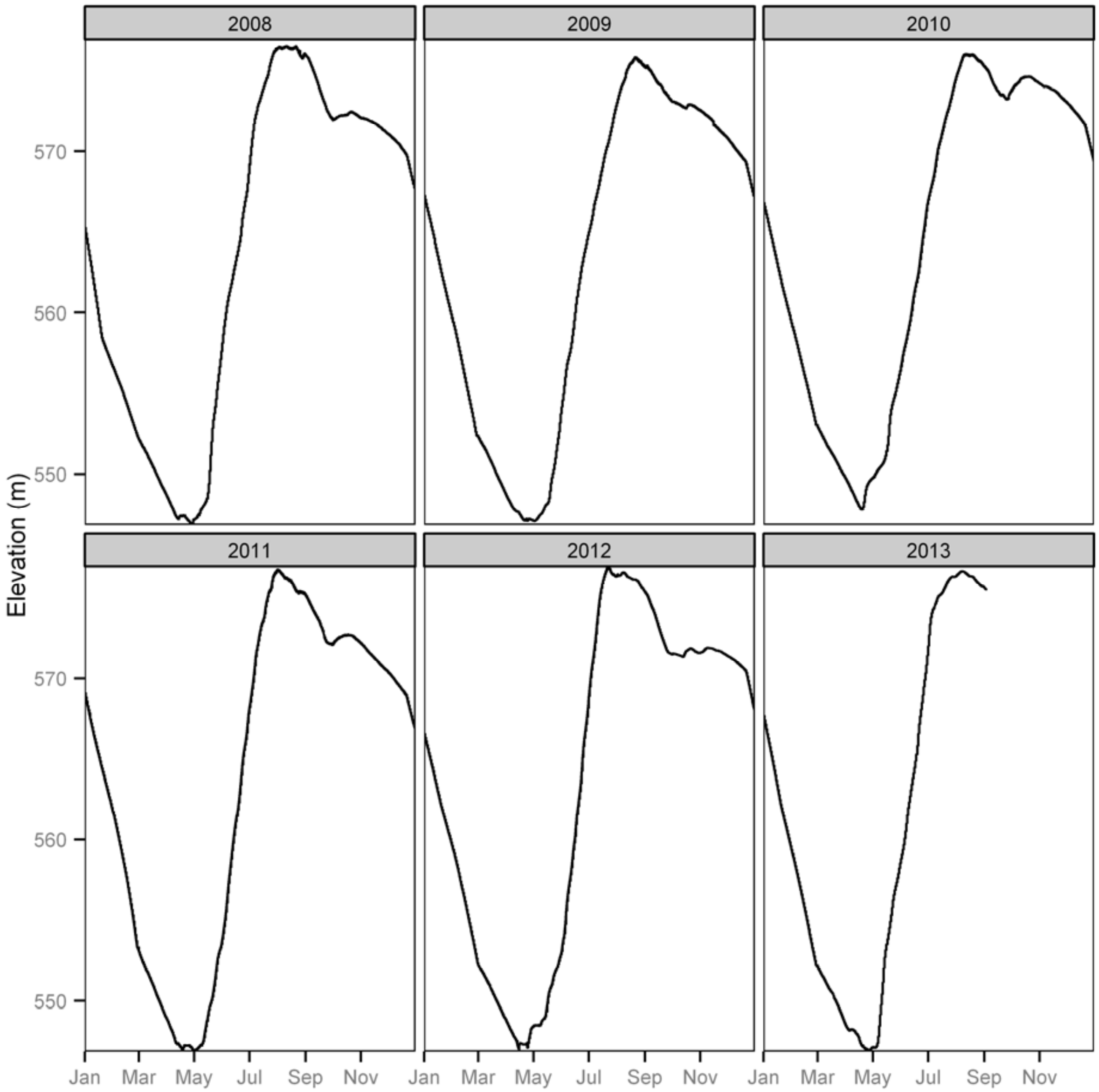

Figure 2: 2008-2013 Daily water elevations (m) for Duncan Reservoir in the forebay.

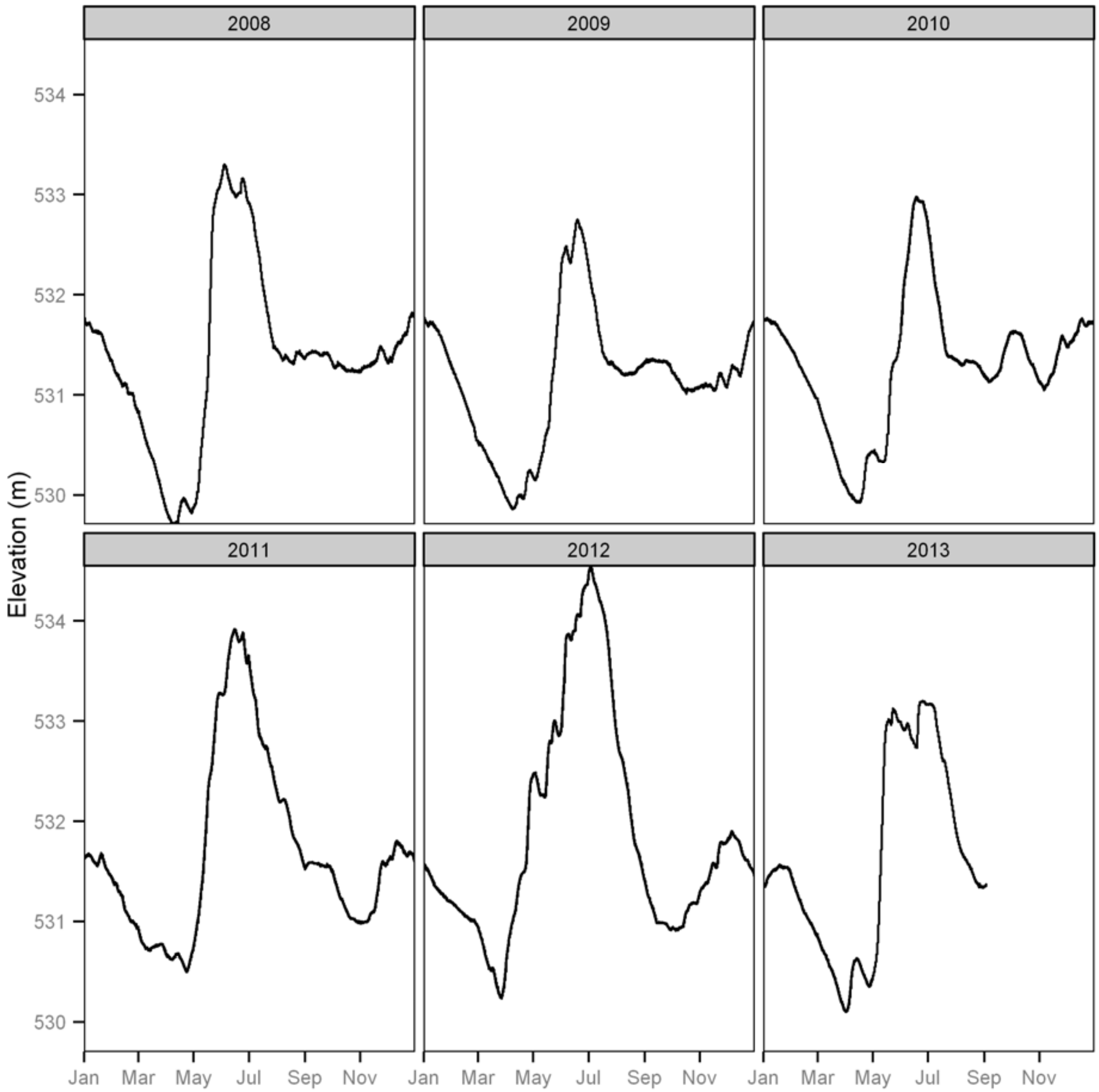

Figure 3: 2008-2013 Daily water elevations (m) for Kootenay Lake at Queen's Bay.

2.0 BACKGROUND

2.1 Historic Studies on the Lower Duncan River

A number of studies have been done on the LDR to understand fish habitat utilization, assess the risk of fish stranding and ultimately reduce the incidence of fish stranding which results from operational changes from DDM. These programs are referenced in the appropriate section(s) of this document; historic information that is relevant to the fish stranding in the LDR includes:

- The development of a Fish Stranding Corrective Action Plan (Higgins 2002);
- The completion of flow ramping assessments during the fall reduction period from 2004 to 2007 (Golder 2005, 2006a, and 2007); and the flow ramping assessment completed in the fall of 2009 (Irvine and Hildebrand 2010);
- Fish stranding assessments of flow reduction events between 2006 and present (Golder 2006b, 2008a, 2009, Hildebrand 2010, Hildebrand 2011b, 2014 in prep, Thorley et al. 2012 and Hildebrand and Irvine 2012). Data was available prior to 2006 but was not utilized in analyses due to differences in sampling methodology;
- An aerial assessment of the LDR to video tape potential stranding habitat (Castlegar BC Hydro Office);
- Seasonal assessments of fish habitat utilization to determine fish habitat presence by habitat type (AMEC 2002, 2003a, 2003b, 2003c, and 2003d);
- Installation of a Data Collection Platform at the Water Survey of Canada Gauge Station (at Km 2.05 on the LDR, downstream of the confluence of the Lardeau River) for real time monitoring of downstream flows and water temperature;
- A fluvial geomorphological assessment of the LDR (Mike Miles and Associates 2002); and,
- Completion of an information gap analysis and power analysis for the Lower Duncan River Ramping Rate Monitoring Program – DDMMON-1 (Irvine 2009).

In 2007, the Order from the Comptroller of Water Rights to implement the Duncan Water Use Plan was received and flow management targets were officially implemented (Table 1) and the ordered monitoring requirements were initiated.

Table 1: Maximum and Minimum Flows: Duncan River downstream of Lardeau River Confluence (DRL)

Date	Flow Targets		Rationale
	Minimum (m^3/s)	Maximum (m^3/s)	
August 1 to August 24	73	400	Control Flooding on LDR
August 25 to September 24	73	250	Flow reductions to protect Kokanee initiated
September 25 to 27	73	190	Flow reductions to protect Kokanee initiated
September 28 to 30	73	130	Flow reductions to protect Kokanee initiated
October 1 to 21	73	76	Kokanee spawning protection flow
October 22 to December 21	73	110	Whitefish spawning protection flow
December 22 to April 9	73	250[1]	Maintain riparian productivity through control of inundation
April 10 to May 15	73	120[2]	Minimize fish stranding prior to and during early freshet

[1] Opportunity to go above 250 to 300 m^3/s for the period providing the Comptroller of Water Rights is notified in a timely manner. This opportunity has been exercised each winter for the past five years with maximum flows of 329, 317, 310, 322 and 255 m^3/s in January of each year from 2005-2009 inclusive.

[2] Original intent to minimize flow increase during periods when newly emerged fry can get stranded. Current strategy is to maintain stable or increasing discharge at LDR.

2.2 On-going and Recent WUP Studies on the Lower Duncan River

Based in part on the knowledge gained from some of the LDR studies outlined above, as well as ongoing flow reduction management of the lower Columbia River system to minimize fish stranding, BC Hydro developed and implemented an interim strategy for managing flow reductions on the LDR entitled “Strategy for Managing Fish Stranding Impacts in the LDR Associated with Flow Reductions at Duncan Dam” (BC Hydro 2004) (revised in 2013 and hereafter referred to as ‘The Protocol’, Westcott et al. 2013). The Protocol outlines communication processes for flow reductions (both internal and external), a specific flow reduction strategy (e.g., ramping rate and timing), and environmental monitoring/response/assessment methods related to planned flow changes. The planned flow responses include, but are not limited to, procedures for conducting fish stranding assessments and communications requirements.

During the DDM WUP, it was determined that the Protocol would require review in order to incorporate recommendations on stranding emerging from WUP monitoring studies on the LDR. DDMMON #15 – the Duncan River Stranding Protocol Development and Finalization Program (hereafter called the Program) was developed and implemented to meet this need.

The study team’s approach to the Program includes the following assumptions based on extensive experience with fish stranding assessments, flow reduction studies and fish habitat assessments:

- Fish stranding is a function of numerous variables including flow reduction rate, frequency, magnitude, timing, channel morphology, and the presence of fish in a particular habitat (Irvine 2009).
- Fish stranding occurs in pool habitat, (river bed depressions which remain wetted after a flow reduction but are isolated from the normal watercourse), drained pool habitat (isolated pools that have drained before field crews arrived onsite), and interstitial habitat (dewatered substrate).
- There have been a number of changes to the operations of DDM over the past 40 years. Although some information on historical improvements made to operations is important to provide context for the protocol development, the focus of the Program will be looking forward.
- Some fish stranding events can be stochastic in nature and therefore difficult to predict based on historical stranding survey information (Golder and Poisson 2010).

During the period following the Year 1 workshop (November 2009 – June 2010), connections were made between the various study teams and there were a number of meetings held to review progress on the Lower Duncan River Hydraulic Model (DDMMON-3) and to further define the required outputs to assist the other study teams in meeting their respective study objectives.

Interim Protocol Reviews were completed in Years 1 and 2 of the Program, consisting of ASPD study result summaries, relevant operational recommendations and other potential Protocol revisions. (See Section 5.0). In Year 5, the Protocol was revised and as such represents findings from those studies implemented up until 2013.

3.0 PROGRAM OBJECTIVES AND SCOPE

This program (DDMMON-15) was initiated to provide annual updates to the Protocol based on the most recent information collected. To facilitate the Protocol development, a variety of information sources are coordinated and reviewed with respect to fish stranding management. As stated in Section 1.2 above, this report tracks progress of ASPD studies toward meeting their objectives on an annual basis. Revisions or amendments will be made to this document as required to ensure it is consistent with the best available information.

In study years 1, 2 and 5, (2009, 2010, and 2013) protocol sections were reviewed and revised where warranted, which included: stranding response procedures (e.g., fish stranding assessment); flow reduction procedures; internal and external correspondence procedures; stranding assessment methodology; and, reporting requirements (BC Hydro 2009). Year 5 (2013) marks the end of the WUP review period. Duncan Dam operational changes considered for fish stranding mitigation were limited by BC Hydro to the timing and rate of discharge change from DDM. Revisions to flow management targets and prescriptions will be reviewed as part of any future water planning process and as such any potential changes to flow timing and volumes that may alter stranding risk were not discussed in this document.

In addition to the tasks outlined below, it is the responsibility of the Program team to identify shortcomings of any ASPD study related to fish stranding and communicate with BC Hydro and with respective study leads at the earliest possible point in the review period to ensure study objectives are met. As stated in Section 1.2 above, this report addresses requirements for 2013 and the long-term approach to Protocol finalization at the end of the DDM WUP in 2019 (Year 11).

3.1 Implementation Plan

The Lower Duncan River Stranding Protocol Development was initially updated in 2010 (Westcott and Irvine 2010) and was subsequently sent to all study leads annually to ensure that all key tasks are identified and the related linkages with operations and other studies are understood. The protocol listed stranding related components from each of the relevant studies, responsibilities and timing, a review of approaches/methods, and ways of working toward a cost-effective, coordinated approach to bringing the appropriate information together. The Plan and Action items listed in the protocol have since been updated as new information became available, and currently includes:

- An update of the Literature Review and Data Gap Analysis that was completed under DDMMON-1 Ramping Experiments in 2008 with information from Duncan Water License Requirements Monitoring Program through 2013 and any new literature relevant to the fish stranding issue.
- The finalization of the Duncan Adaptive Stranding Protocol Development Report, which summarizes new learnings from four years of monitoring programs (Westcott et al. 2013).
- Participation in annual communication with agencies to review any changes to the Protocol or potential study changes.

- Completion of annual updates to Protocol (as applicable) assuming a full review in 2019, as knowledge is gained through DDMMON-3, and DDMMON-16 (and from the concurrent Columbia River flow ramping and fish stranding monitoring studies and international experience). Any information that will ultimately aid in reaching the end objective of reducing the number of fish stranded on the LDR as a result of dam operations will be identified and incorporated into the Protocol refinement and implementation.
- Maintenance of ASPD schedule and relevant Action Items List.
- In the final year of the WUP review period (2019), the study team will submit a finalized Protocol based on data and analyses of monitoring studies DDMMON-1, -2, -3, -4, -16 and DDMWORKS-4 for approval by BC Hydro and regulatory agencies. The final Protocol will be implemented and monitored according to the agreements struck during future water planning initiatives for DDM.

Based on the previous Stranding Protocol Development refinements (2010, 2012) and communications between BC Hydro and the DDMMON-15 study teams, it was determined that field visits and annual aquatics workshops were not required as all study teams were familiar with the LDR. If required, a workshop can be organized during future study years to inform study leads of the most recent stranding related information for the LDR.

4.0 LOWER DUNCAN RIVER ADAPTIVE STRANDING PROTOCOL DEVELOPMENT DATA COLLECTION PROGRAMS

The following section summarizes current and ongoing monitoring and data collection programs related to the LDR ASPD (Figure 4). Individual project reports can be obtained from BC Hydro by visiting the WUP webpage. Relevant information collection outside of the LDR drainage has also been summarized below and current contacts for further information are provided in Table 2.

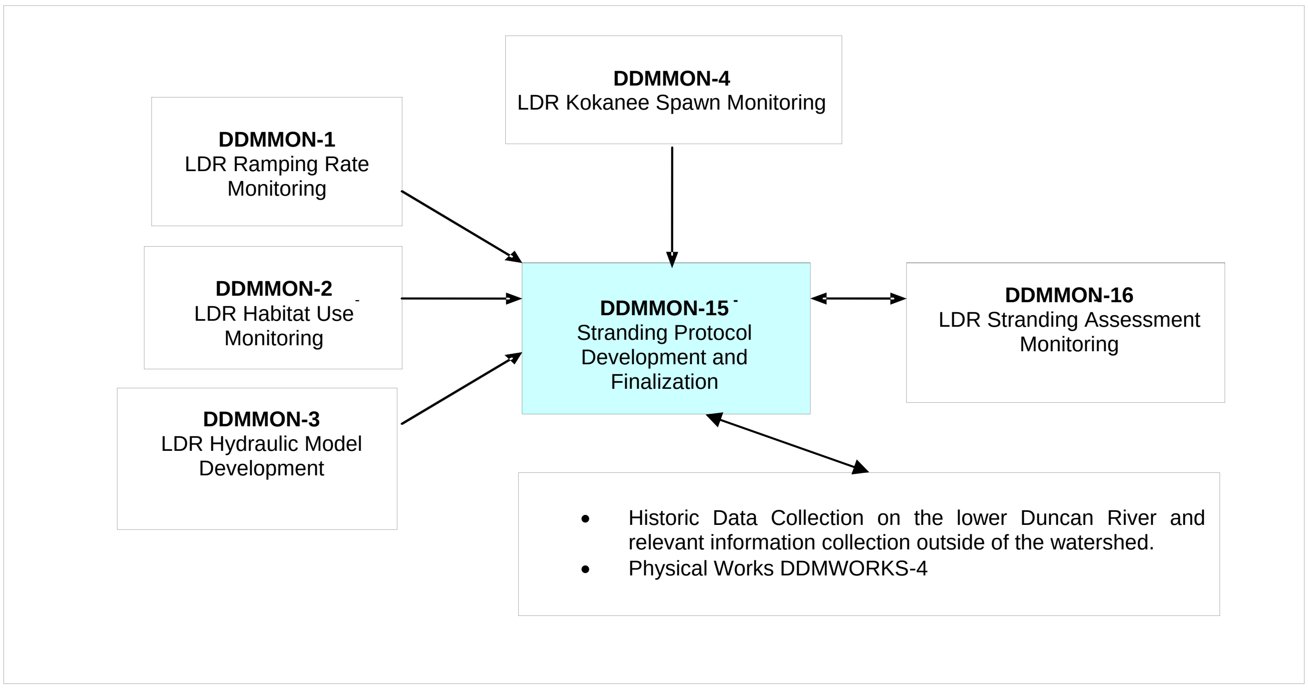

Figure 4: Lower Duncan River Adaptive Stranding Protocol Development and Finalization – Duncan WLR ASPD Study Linkages.

Table 2: Duncan ASPD Program Studies and Study Contacts 2008-2013.

STUDY	STUDY CONTACT	BC HYDRO CONTACTS
DDMMON-1	Golder/Poisson (Brad Hildebrand/Robyn Irvine)	WLR Watershed Lead -
DDMMON-2	AMEC/Poisson/Mountain Water Research (Louise Porto/Joe Thorley/Robyn Irvine/Jeremy Baxter)	Phil Bradshaw
DDMMON-3	Northwest Hydraulics Consultants (Barry Chillibeck)	
DDMMON-4	AMEC/Poisson/Mountain Water Research (Louise Porto/Joe Thorley/Robyn Irvine/Jeremy Baxter)	WLR Implementer – James Baxter
DDMMON-15	Golder/Poisson (Brad Hildebrand/Robyn Irvine)	
DDMMON-16	Golder (Brad Hildebrand)	SME – Alf Leake
CLBMON-42a	Golder (Demitria Burgoon)	WLR Implementer – James Baxter
CLBMON-42b	Golder/Poisson (Brad Hildebrand/Robyn Irvine)	SME – Guy Martel

4.1 Duncan Dam Water License Requirements Monitoring Programs

- **DDMMON-1 LDR Ramping Rate Monitoring** - A two-year monitoring program (2008 - 2009) to assess the impacts associated with the timing, magnitude, and rate of flow changes at DDM on LDR fish species life histories of interest. The program is intended to help define the best management strategy for reducing flows at DDM. In Year 1 (2008), a gap analysis was completed to assess the parameters that may influence fish stranding risk on the LDR. The fifth year of an experimental study program was completed in 2009 to further test the influence of environmental and operational variables on stranding risk in order to inform flow management procedures in the protocol.

- **DDMMON-2 LDR Habitat Use Monitoring** - A four-year monitoring program (2009 - 2012) to document habitat use and relative abundance of juvenile Rainbow Trout and Mountain Whitefish, the timing and use of Rainbow Trout and Mountain Whitefish spawning, and to assess Burbot (*Lota lota*) migration barriers in the LDR. This program assisted in defining seasonal stranding risk for vulnerable juvenile salmonids by delineating the habitat types with high abundance during the fall rampdown to Kokanee Protection Flows. Also, knowledge on the timing and locations for adult Mountain Whitefish and Rainbow Trout spawning was refined.

- **DDMMON-3 LDR Hydraulic Model** - A multi-year channel survey, hydraulic, and habitat modeling study of the LDR (2009, 2010, 2013, and 2018). This program assessed channel change and operational impacts on fisheries habitats for consideration in future flow planning processes and to help define the ramping rate protocol. This RIVER 2D model was intended to assist in assessing risk of fish stranding at different river stages and to inform habitat suitability curves for species of interest. In 2012 to 2013, the model underwent an update to reduce uncertainty and to address changes in channel morphology.

- **DDMMON-4 LDR Kokanee Monitoring** – This program consisted of annual Kokanee spawning surveys and mapping in LDR, Meadow Creek, and Lardeau River to assess the relative importance of Kokanee spawning in each system, as well as to determine the impacts of WUP operations on Kokanee spawning in the LDR (2008 – 2018). Adult Kokanee have stranded in side channels previously and there are potential implications of fall/winter flow changes on egg dewatering. This study was designed to assist in defining stranding risk to adult Kokanee and incubating eggs.

- **DDMMON-16 LDR Stranding Impact Monitoring** - This ongoing ten-year monitoring program (2008 - 2017) provides annual indices of the effectiveness of measures taken in the Adaptive Stranding Protocol Development Programme, including fish stranding rates and, in some study years, stock abundance. Through random sampling, this program will help to define the risk of fish stranding in the various habitat types found in the LDR.

- **DDMWORKS-4** - In the latter seven years of the WLR implementation period (2012 – 2018), BC Hydro is required to develop an action plan to minimize the risk of dewatering Kokanee redds and stranding adult Kokanee in the LDR downstream of the dam. This plan is to include an assessment of using physical works (e.g., recontouring) to minimize Kokanee adult and egg stranding. Since this program has not yet produced results, it will not be reviewed.

With the exception of DDMMON-4 and 16, the LDR WLR programs above are on defined timelines, subject to some modification as study results are reviewed. Based on the timing and results for several DDMMON programs, the DDMMON-15 program schedule for Years 2 to 5 was revised. Table 3 provides a 10 year study schedule to be consistent with the timeline for the LDR ASP Program.

Table 3: Duncan River Adaptive Stranding Protocol Development 10 Year Data Collection Plan.

Monitoring Study	Description	Year 1 2009	Year 2 2010	Year 3 2011	Year 4 2012	Year 5 2013	Year 6 2014	Year 7 2015	Year 8 2016	Year 9 2017	Year 10 2018
DDMMON-1	Flow Ramping Experiments	X	X								
DDMMON-2	Habitat Use Monitoring		X	X	X	X					
DDMMON-3	Hydraulic Modelling		X	X		X					X
DDMMON-4	Kokanee Monitoring	X	X	X	X	X	X	X	X	X	X
DDMMON-15[a]	Protocol Development and Finalization Original Schedule		X, Y, Z	X, Y	X, Y, Z	X, Y, Z	X	X	X	X	X
DDMMON-15[a]	Protocol Development and Finalization Revised Schedule		X, Y, Z	X	Z	X, Z	X	X	X	X	X
DDMMON-16	Stranding Assessments	X	X	X	X	X	X	X	X	X	X
DDMWORKS-4	Kokanee Adult Stranding Action Plan					X	X	X	X	X	X
CLBMON-42a	LCR Stranding Assessments	X	X	X	X	X	X	X	X	X	X
CLBMON-42b	LCR Protocol Finalization			X							

[a] X = annual report, Y = annual workshop, Z = ASPD Refinement.

4.2 Other Relevant Studies

In addition to the DDM WLR studies, information from other studies in the basin and around the world on fish stranding will be incorporated into the ASPD program to allow comparison and increase the rigour with which conclusions are made. Studies in the Columbia and Kootenay River watersheds of relevance to the ASPD include the following:

- **CLBMON-42a Lower Columbia River Fish Stranding Assessment and Ramping Protocol** - A 13-year program (2006 – 2018) to monitor planned and opportunistic flow reductions from HLK and BRD and define their impacts on fish stranding in the lower Columbia River. Operational procedures to mitigate flow reduction impacts were also examined in the early years of the program.

- **CLBMON-42b Columbia and Kootenay River Stranding Protocol Review** - A one year program (2010) to review and combine the current stranding protocols (Columbia and Kootenay) based on the knowledge gained to date (i.e., ramping studies, stranding assessment data, literature review). The combined protocol ensured that mitigation strategies for flow reductions in the system are based on the most up to date data and literature, and outlines future monitoring efforts to confirm the effectiveness of the current mitigation strategies. Data gaps that need to be addressed through further study were also identified.

5.0 LOWER DUNCAN RIVER ASPD MANAGEMENT QUESTIONS, HYPOTHESES, AND CURRENT STATUS

The appropriate fish stranding **hypotheses** and **management questions** associated with the individual monitoring studies (DDMMON-1, -2, -3, -4 and -16), as excerpted from the various BC Hydro WUP Terms of Reference documents, are outlined below in bold text. The following sections build on the information from the DDMMON-15 Years 1 and 2 reports and include study results from 2010-2013 as they relate to the ASPD. Each section defines the hypotheses and management questions related to fish stranding, summarizes the relevant ASPD outcomes to date, reviews the operations, questions and PMs associated with each management question, and identifies data gaps based on information gleaned from the relevant reports.

Primary Program Management Question

The Lower Duncan River ASPD Development and Finalization Program was developed by BC Hydro to address the following management question:

> ***What are the best operating strategies at Duncan Dam to reduce the number of fish stranded on the lower Duncan River?***

Each of the following DDMMON studies include management questions and hypotheses designed to help answer the above management question.

5.1 DDMMON-1: LDR Ramping Rate Monitoring

This WLR study was a 2-year monitoring program (2008 – 2009) to assess the impacts associated with the timing, magnitude and rate of operational changes at DDM on LDR fish species of interest. This program assessed stranding with mesocosm experiments in one year of study, which built on four years of experiments conducted previously during pre-WUP operations. This program also completed a detailed literature review and gap analysis on the issues of fish stranding, which is updated in this document (Section 6.0) to include the state of knowledge up to February 2013. Details on experimental findings can be found in Poisson Consulting Ltd. and Golder Associates Ltd. (2010).

Table 4 summarizes the DDMMON-1 implications of ramping experiments on fish stranding risk.

Table 4: DDMMON-1 - Implications of Ramping Experiments on Fish Stranding Risk.

Operational Application	Potential Benefits of Operational Application	Biological Opinion	Remaining Uncertainties and Research Recommendation	Recommendations for Protocol Revisions
Under the Columbia River Treaty, discharge changes are limited to a maximum of 113 m^3/s /day.	Provides a maximum daily stage reduction to minimize habitat dewatered and associated stranding.	From analyses done on the Columbia/Kootenay, the magnitude of the reduction is not as critical to stranding risk as the river stage at which habitat is dewatered, and the day of the year.	Do reductions of larger magnitude have a greater effect than those of smaller magnitude? Partially addressed by DDMMON-16 and DDMMON-3 program teams.	No changes recommended associated with daily maximum flow change.
Flow ramping of 28 m^3/s per hour from DDM provides stage reductions of approximately 15 cm/hr.	Reduced ramping rates may allow fish opportunities to escape receding water levels.	Preliminary testing showed that several small reductions (4/hr) were associated with lower stranding risk than one larger reduction (1/hr) to achieve the same magnitude of change. The down ramping rate for Kokanee spawning habitat and Rainbow fry rearing habitat were estimated in DDMMON-3 to range between 0 and 14cm/h.	None	Maintain operations that make flow reductions as slow as possible (i.e., 7 m^3/s per 15 minutes). Use DDMMON-3 gauges to refine estimates of rate of stage change and iterate operations to keep changes below 10cm/hr wherever possible.
Conduct flow reductions during daylight hours.	Reduces incidence of fish stranding based on trends observed from ramping experiments.	Juvenile whitefish susceptible to increased risk of fish stranding at night.	Current findings based on limited data. Other research on diel stranding by system and species is variable.	Conduct flow reductions from DDM during daylight hours until the data trends can be examined further.
Reducing water levels through areas of high cover may increase fish stranding.	Mitigation activities may reduce stranding.	More cover is generally accepted in the literature to be correlated with increased stranding risk as well as higher fish productivity.	Variability in influence of fish cover on stranding in this system and for species of interest.	Ensure cover is measured as variable in fish stranding assessment for future evaluation.

5.1.1 Project Outcomes

The analysis of the time of day variable (day vs. night) was not statistically significant and the response variable showed high variability both during daytime and at night, although stranding rates trended higher during night time ramping experiments (Poisson and Golder 2010). The results from the literature were variable with some species and systems demonstrating higher risk as night and others during the day (Section 6.3.1).

Longer periods of wetted history showed a non-significant but consistent trend with increased stranding risk. The analyses of the CLBMON-42a Columbia and Kootenay stranding database indicated increasing risk of stranding with increased periods of wetted history (Irvine 2010). This information is consistent with the data from DDMMON-1, although the range of wetted history values over which stranding risk could be assessed for the LDR was limited. A conditioning reduction approach was not considered to be a viable option for minimizing juvenile fish mortality prior to large operational drops after observing low survival rates for Mountain Whitefish on the LDR soon after fish were stranded (Poisson and Golder 2010). Therefore the main way in which wetted history and stranding risk can be addressed is to minimize the stage change that occurs with each reduction particularly after periods of sustained inundation.

The two hypotheses (H_{02}a and b) relating rate of river stage change to channel flow and channel morphology respectively have not been specifically addressed. The recommendation was made in the DDMMON-1 report to merge outcomes from DDMMON-1, -3 and -16 in upcoming study years of the WUP timeframe in order to assess the relationship between the outstanding questions around these variables and stranding risk. The successful assessment of these variables will require a blending of modeling and field based approaches from DDMMON-3 and -16 and may require further refinement or model runs of the ouput model from DDMMON-3.

Although ramping rate (rate of stage change) was not included in any of the top ranked statistical models and has never emerged as statistically significant due to high variance in the data and/or lack of a genuine trend, the stranding rates have consistently been higher with faster ramping rates in the LDR over the range tested (Poisson and Golder 2010). This pattern of higher stranding at faster ramping rates has been observed in the peer reviewed literature summarized in Section 6.3.2, but generally over a larger range of rates (with faster rates). Depending on river location and river stage, stage change can vary significantly with changes in channel morphology (NHC 2010). The magnitude of river stage change was not found to be directly related to fish stranding risk when tested in the Columbia and Kootenay systems, rather fish stranding risk was increased with low sloping habitats and the presence of cover. However, the magnitude of change may interact with the river stage so that at certain stages larger flow reductions may lead to higher rates of stranding. Furthermore, stage change at each site will have its own relationship with the amount of area dewatered (e.g., during equal stage changes a narrow-channelized river section will have less dewatered area than a wide braided section).

The preliminary answer to the hypothesis relating cover to stranding risk is based on the information from the literature - the consistent relationship is for increased fish stranding with increased cover (Section 6.3.5). From the experiments conducted on the LDR, there was a neutral or slightly positive relationship between increased cover and stranding risk, which was not statistically significant (Poisson and Golder 2010). The opportunities to mitigate cover availability to minimize stranding are negligible since removing cover could be contrary to the overall goal of increasing fish productivity.

The effects of substrate and habitat configuration on stranding risk were not explicitly assessed by DDMMON-1. These could potentially be assessed by looking at outcomes from DDMMON-2, -3 and -16. The dominant substrate has been mapped throughout the system as part of DDMMON-3 and this information could be coupled with the DDMMON-16 stranding assessment dataset to determine if stranding risk is higher in particular habitat/substrate types and information from DDMMON-2 would inform whether those habitats are high use by target fish species. This would require substantial random sampling effort throughout the LDR as part of DDMMON-16 to obtain a data set robust enough to test the effects of substrate and habitat configuration. Similarly, habitat configuration (gradient and topography) could be assessed as an explanatory variable of stranding risk once the LCR stranding database has sufficient data. Habitat changes to mitigate fish stranding have not been studied in detail for their benefits in the LDR. There may be opportunities to address the management question if high risk stranding sites are identified that are also good candidates for re-contouring, but such mitigation solutions may not be long lasting due to the dynamic nature of the system.The preliminary answer to Hypothesis H_{04} is that stranding rates will be higher on low slope habitats with more indentations or pools due to the topography (Irvine 2009).

Annual DDM flow reductions are limited in number with major flow reductions occurring in the fall and winter seasons of each year. Analysis between fish stranding risk and time of year is examined in the DDMMON-16 program (Section 5.5) but is limited by the number of events per year and their distribution throughout the seasons. Reductions of differing sizes and throughout the year as well as sites throughout the study reach should be sampled in order to accurately assess the thresholds for stranding risk on the LDR.

The stranding risk for adult and sub-adult fish during flow reductions from DDM is considered very low during most of the year due to the habitats they typically occupy. The Kokanee migration in September poses the greatest risk to adults as spawners are known to strand in sidechannels as discharge recedes to Kokanee protection flows (Poisson and Golder 2010). The earlier life stages (including eggs, larvae, fry, and young-of-the-year fish) are at the greatest risk for stranding in all seasons (Poisson and Golder 2010).

The methodologies used in DDMMON-1 did not allow the determination of differential stranding risk between fish species. This may be examined further as part of the DDMMON-16 program (Section 5.5). BC Hydro has identified an increased risk of stranding newly emerged Mountain Whitefish and Kokanee in the spring prior to the onset of freshet (Alf Leake, Pers. Comm).

5.1.2 Remaining Uncertainties and ASPD Focused Recommendations

It was recommended that further study in the LDR focus on obtaining data on fish stranding risk from modified fish stranding assessments (DDMMON-16) rather than on additional ramping experiments. The experiments have been very useful in determining trends related to stranding variables that are difficult to test with stranding assessments (e.g., time of day), as well as for modelling stranding risk with accurate corrections for capture efficiency. However, they do not obtain certain information that is vital for the next steps in understanding the factors leading to fish stranding. In order to obtain adequate data, the stranding assessment program may need to be strategic about what reductions they staff so that the range of magnitudes of reduction, as well as seasonal variability is captured by the program.

1) Flow reductions from DDM should only occur during daylight hours to minimize the risk of stranding juvenile fish and allow for fish stranding assessments. If nighttime reductions are necessary (i.e., an emergency situation) stranding assessment crews should be mobilized to commence sampling at first light in order to gather additional data on nighttime reductions.

2) Conduct flow reductions from DDM at the slowest ramping rate that is operationally feasible with a maximum achieved flow ramping rate of less than 10cm/hr throughout the length of the LDR. Make flow reductions with a series of multiple smaller increments rather than one large flow reduction. The DDM flow reduction rate currently set at a 28 m^3/s/hr maximum (slow rate) to maintain stage reductions of approximately 10 cm/hr (BC Hydro, GOO 2008). Even slower ramping rates that produce stage changes of less than 10 cm/hr are recommended (Poisson and Golder 2010) to allow fish to escape to deeper water habitats and allow monitoring crews to assess fish stranding (as required). If further analysis of the ramping rate variable is desired, the ramping rate should be varied following a preplanned study design during each of the reductions within a year to provide variation in the parameter to be tested.

3) Additional analysis is recommended to address the hypotheses (H_{02}a and b) relating stage change to channel flow and morphology. The Year 5 DDMMON-3 report discussed the addition of 4 hydrological sensors into the watershed and proposed looking at the effect of different channel morphology in Year 10 when they have more data to compare.

4) It is unlikely there will be further attempts to experimentally assess the effects of cover on the risk of stranding, though it will be recorded as a variable in stranding assessments for DDMMON-16 and may be analysed as part of that study program. Available cover is quite dynamic in the LDR so to model it would be difficult without constant updating and surveying. The influence of cover on stranding risk has been assessed in other studies and the literature on this topic is overviewed in the gap analysis component of DDMMON-1 (Irvine 2009). ASPD recommendations include identifying and monitoring sites with high levels of cover and to salvage fish where appropriate. Currently this is incorporated into the DDMMON-16 program.

5) Fish stranding rates were only assessed in autumn by this study program. Operational reductions on the LDR occur at particular times in a year as a result of system operational constraints, so stranding was only assessed when reductions were likely to occur. The monitoring program of DDMMON-16 continues to assess stranding after major reductions that occur throughout the seasons, thus examining the seasons in which operations that have associated stranding risk occur. It will take several years of data collection to be able to determine the relationship of season to risk due to the low number of reduction events per year. The DDMMON-16 program is scheduled to occur over 10 years which may provide an adequate dataset to address this question.

6) The risk to particular species could not be assessed by the September ramping experiment study program due to methodological constraints which was biased towards sampling predominantly Rainbow Trout. The stranding risk related to other species may also be determined through analysis of the database emerging from DDMMON-16. Until additional information is obtained, the assumption was made in the DDMMON-1 report and should continue to be held that all species are equally at risk of stranding. Emergence and residence timing for newly emerged larvae were uncertain for the LDR and given the fall focus of DDMMON-2, it is unlikely that this data gap will be addressed empirically under the current WLR studies.

5.2 DDMMON-2: LDR Habitat Use Monitoring

The main objective of the DDMMON-2 program was to collect information on the life history and habitat use for the two target species of Rainbow Trout and Mountain Whitefish and to complete a literature review on potential velocity obstacles to Burbot passage through the LDR. These species may be impacted by water level fluctuations resulting from daily and seasonal operations of DDM. In Year 1 of the program (2009), Burbot passage was assessed and information was collected on juvenile Mountain Whitefish and Rainbow Trout and on adult Rainbow Trout spawners. In Year 2 (2010), the adult Rainbow Trout spawning program and juvenile Rainbow Trout and Mountain Whitefish program was completed. In Years 2 and 3, adult Mountain Whitefish spawning was assessed and the adult Rainbow Trout spawning program continued, though no longer as a WLR study. Commencing in 2011, a study funded jointly by the HCTF and the FWCP continued to assess the juvenile Rainbow Trout in the LDR and the Lardeau River and findings from that study are also included in this section. Both studies on juveniles were carried out using the same methods employed in DDMMON-2 (Andrusak 2013) Associated references for DDMMON-2 and the ongoing adult Rainbow Trout spawning study include (Porto et al. 2009, Irvine and Porto 2010, Thorley et al. 2010, 2011 p. 2, 2012, Thorley and Baxter 2011).

Table 5 summarizes the findings from DDMMON-2 in relation to operational implications of habitat use on fish stranding risk.

Table 5: DDMMON-2 - Operational Implications of Habitat Use on Fish Stranding Risk.

Operational Application	Potential Benefits of Operational Application	Biological Opinion	Uncertainties	Recommendation for Protocol Revision.
Maintain stable or increasing discharge at DRL during the spring period appropriate to minimize stranding risk of newly emerged Kokanee and Mountain Whitefish.	Reduction of fish stranding risk due to DDM operations in spring.	Smoothing flows reduces the risk of fish stranding. Using the flows at DRL rather than Duncan Above Lardeau (DAL) puts larval and fry stages above the confluence at risk of stranding or dewatering if the Lardeau River fluctuates due to cold weather events and if DDM flows are not sufficient to keep the confluence area watered.	Larval Mountain Whitefish eggs hatch between February and May and may be affected by the warmer winter water drawn from the reservoir and by sudden increases in flow that may incite emergence. It is estimated that Kokanee emergence begins near the end of February and continues into early April. The potential effects of this operation on Rainbow Trout in the tailout of DDM are unknown.	When dropping from high winter flows, manage the magnitude of reductuions to minimize egg stranding.
Temporary changes in DDM discharge [e.g. 3 to 0 m^3/s from LLOG#2 and 0 to 24 to 0 from LLOG#1] to facilitate BT transfers.	The current operation eliminates the risk of attracting additional Bull Trout to LLOG#1.	If this specific operation was not in place Bull Trout could enter LLOG#1 and injury or mortality could occur. There may be an operation and physical modification that may address the Bull Trout concerns as well as downstream concerns (e.g. Rainbow Trout) and the existing operational constraints.	There are potential Rainbow Trout redd or fish stranding impacts with the current operation. Rainbow Trout redds were shown to be at risk of dewatering, incurring high temperatures or sediment from backwatering from the Lardeau when insufficient DDM flows were provided.	Decrease the magnitude of reductions within a 24 hour period to reduce egg and redd dewatering.
Operational strategies to maintain water over eggs deposited by the key species of Kokanee, Rainbow Trout and Mountain Whitefish.	Increase in spawning and hatch success for target species.	As Rainbow Trout deposit their eggs almost exclusively in the tailout area of DDM, there is a high risk of loss due to stranding or insufficient flow. The high flows normal to operation in Dec and January may put the Mountain Whitefish eggs at risk of resuspension and movement from spawned locations and damage.	Rainbow Trout incubation success is currently unknown based on DDM operations and a study has been proposed for spring 2014 to address this. Mountain Whitefish use habitat from 0-2 m in depth (peak use at 0.9m), with 67% of observed spawners on cobble and 32% on large gravel, and water with 1-1.48m/s velocity (peak use at 0.68m/s) for spawning. Kokanee will be discussed in the section on DDMMON#4.	When dropping from high winter flows, manage the magnitude of reductuions to minimize egg stranding.
Current spawning protection flows for Kokanee and Mountain Whitefish limit water discharge from reservoir and require higher discharges during September and Dec/Jan.	Current operational strategies are designed to reduce Kokanee spawning in areas that will dewater.	The higher discharges in Dec/Jan may have negative effects on spawning Mountain Whitefish and incubating eggs. DDMMON-3 shows increased wetted usable area (WUA) for Mountain Whitefish fry and incubation with increasing flows. This has not been empirically assessed.	The uncertainties associated with current operational targets on fish include lack of knowledge on abundance of Mountain Whitefish and recruitment levels of the LDR population.	There are no operational recommendations within the scope of this document.

5.2.1 Project Outcomes

5.2.1.1 Rainbow Trout Spawning

Rainbow Trout spawn timing, fry emergence timing and habitat preference curves were estimated for the LDR as part of DDMMON-2 and the subsequent study program. Rainbow Trout redds have been documented in the 'tailout" area of the river located between the end of the armoured DDM discharge channel and the confluence with the Lardeau River annually since 2004 (Hagen et al. 2010, Thorley and Baxter 2011, Thorley et al. 2012, Irvine et al. 2013). In the tailout area, Rainbow Trout spawn from mid-March until late-May. Redds have been found in depths as low as 30 cm, though the maximum depth is difficult to determine due to visibility issues. In spring 2010, evidence of spawning was also found in the side channel at 4.1R. This was the only instance when spawning was documented outside of the tailout area, despite suitable spawning habitat being documented in 17 discrete locations in the LDR (Thorley et al. 2012).

Rainbow Trout spawned in habitat in the LDR that ranged from 025 - 0.75 m in depth and with velocities of 0.3 - 0.8 m/s, though deeper spawning may occur and not be visible due to turbidity. Spawners seemed to be selecting the tailout for reasons other than depth or velocity as other suitable sites are not used. In the 9 years of study on this spawning area, the number of redds recorded in the tailout area ranged between 26 and 160. Most of the fish detected on the spawning bed below DDM were later detected on the spawning bed at Gerrard, which suggested that one single genetic population is present in the system (Thorley et al 2012). There was a correlation (R^2 = 0.75) between the date of commencement of spawning and the date at which the water temperature first exceeded 5°C. In post-WUP years, spawning typically occurred 1 - 2 weeks later in comparison to the pre-WUP years (Thorley et al. 2012).

5.2.1.2 Mountain Whitefish Spawning

The DDMMON-2 program focused on obtaining spawn timing, and habitat use curves for adult spawning fish, and also provided an approximation of egg emergence timing. Mountain Whitefish spawn timing was estimated to occur between October 15 and December 21, with 97.5% of the spawning completed by December 14th. Peak spawning occurs in the third week of November based on Gonado-Somatic Index data (Thorley et al. 2012). This is a slightly broader window than stated by the consultative process, which estimated the spawning period to occur from October 21 - December 21.

During the DDMMON-2 program, an index site was surveyed with night spotlighting throughout the spawning season, and at the peak of spawning 340 adult fish were observed within the 1 km long study area. The number of spawners encountered during the peak was lower than anticipated given the size of the spawning aggregations on the Columbia River (Hildebrand 2011a, Thorley et al. 2012). The habitat use curves estimated that the MW spawn in 0 - 2 m of water with peak spawning at 0.9 m. Spawning velocities were estimated between 0.1 - 1.48 m/s with peak at 0.68 m/s. The majority of spawning occurred over cobble (67%) and large gravel (32%) with 20% of the spawning proximal to large woody debris (Thorley et al. 2012). These depth, velocity and substrate use data may not indicate preference but rather what is available to the MW at the time of spawning; for example the estimated velocity is higher than predicted as optimal using data from other systems (Lewis and Healey 2009). Incubation of mountain whitefish eggs was estimated to range from mid-October to early May which narrowed the emergence window in the spring from the WUP predicted incubation period which was from October 21 to May 31.

5.2.1.3 Juvenile Rainbow Trout and Mountain Whitefish

The habitat use for the juvenile Rainbow Trout and Mountain Whitefish in the LDR was also assessed by DDMMON-2 and in 2012 was assessed by the ongoing HCTF and FWCP program run by Redfish Consulting Ltd. The WUP process assumed that juvenile Mountain Whitefish and Rainbow Trout use the LDR from April to October. Conversely, this study showed that juvenile Rainbow Trout rear in the LDR for 2 - 3 years, while Mountain Whitefish may also use the habitat year round. Very low numbers of Mountain Whitefish observed in winter and spring (AMEC 2005, Thorley et al. 2012) may mean they either suffer extremely high overwinter mortality or move out of the surveyed habitat into other locations in the winter months.

Rainbow Trout parr in the Lardeau River were implanted with acoustic tags in a separate project and were shown to out-migrate during freshet in May and June (Andrusak 2010); it remains unknown what percentage of younger fish out-migrate from the LDR or the Lardeau River to Kootenay Lake. The LDR has higher spring densities of Gerrard Rainbow Trout fry than the Lardeau River and provides important rearing habitat. Densities for RB fry were 1.25, 0.80 and 0.94 fish/m in 2011, 2012 and 2013 respectively (Andrusak 2013). However, the LDR has lower or equivalent densities of Rainbow Trout parr showing either differential mortality or emigration between the two systems with the LDR exhibiting the higher losses. This mortality consists of natural and operational causes, including a stranding related component.

Habitat use curves were developed for juvenile Mountain Whitefish and Rainbow Trout fry and parr (Table 6). Low velocity water (<0.5 m/s) is the most preferred habitat for both species and life stages though RB were typically observed in slower water than Mountain Whitefish and fry were in slower water than parr. Mountain Whitefish parr were documented to use the highest velocities of the two species. This may allow the determination of what areas might be usable habitat for each of these species and life stages in the LDR; the use of the Rainbow Trout habitat preferences as a proxy for Mountain Whitefish may be preferable as they are more conservative in modeling flows that maximize fish productivity and diversity - the stated objectives of these programs.

Rainbow Trout abundance in Fall 2010 for age-0 fish was estimated to range from 29,000 - 64,000 and 17,000 in spring 2012, with an apparent mortality rate of 75%. This is in the same range as the 65 – 77% mortality rates estimated previously for Lardeau River and LDR (Decker and Hagen 2009). Rainbow Trout parr abundance was estimated at 5500 in fall 2010 and 4200 in spring 2012. Mountain whitefish abundance was estimated for age-0 fish in fall 2010 to be 28,000 and in spring 2012 was estimated at only 126 fish for an apparent mortality rate of 99%. Mountain Whitefish parr estimates were not calculated due to high uncertainty.

Table 6: DDMMON-2 Peak Habitat Use Criteria for Rainbow Trout and Mountain Whitefish Juvenile Life Stages.

Species	Life Stage	Depth (m)	Velocity (m/s)
Rainbow Trout	Fry	0.19	0-0.5
Rainbow Trout	Parr	0.69	0-0.5
Mountain Whitefish	Fry	0.37	0-0.5
Mountain Whitefish	Parr	0.56	0-0.55

Sidechannels and mainstem sites were surveyed and there were no systematic differences in densities with the exception that during high winter flows, the numbers of Rainbow Trout parr within sidechannels increased.

Therefore, flow changes in the winter season may pose a stranding risk in sidechannel habitats. There was a relatively uniform distribution of juvenile fish lineal density with a few exceptional sites that had either very low or very high density (Thorley et al 2012). The three winter surveys that were completed in sidechannel 2.7R in December, January and March showed increased numbers of Rainbow parr over the winter and decreased numbers of Rainbow Trout and Mountain Whitefish fry, with Mountain Whitefish showing the most significant decline. These trends imply Rainbow parr movement between habitats, and either movement or overwinter mortality or some combination for the fry.

In order to define rainbow trout Effective Rearing Habitat (ERH), the DDM WUP Consultative Committee (CC) assumed that juvenile rainbow trout: 1) rear in the LDR mainstem and side channels between April 1 and October 31; 2) only utilize habitats that have been wetted for at least 10 days; and, 3) experience all available (wetted) habitat as equally suitable. Then when the resultant definition of ERH was factored into the various trade-offs and constraints of the WUP process the CC selected a year round minimum discharge at DRL of 73 m^3/s.

Key findings of this and other programs include the presence of significant numbers of juvenile rainbow trout in the LDR year-round; the almost complete absence of juvenile mountain whitefish in late winter in surveyed habitats; the almost exclusive use of slower, shallower, marginal habitat; and the absence of a significant difference in the lineal densities of juveniles in mainstem versus side channel habitats (except perhaps during high discharge). This allows future focused work on minimizing the operational effects targeted to species and habitats of high use. Finally, although not tested, the assumption that habitats must be wetted for at least 10 days in order to be utilized is consistent with a study on Brown Trout (*Salmo trutta*) in the Selwyn River, New Zealand (Davey and Kelly 2007), although recolonization of a side channel that has been completely dewatered might take longer.

5.2.1.4 Burbot

No implications of dam operations on stranding of adult or juvenile Burbot were identified in the study report (AMEC et al 2010). Burbot passage was assessed in the context of hydraulic conditions predicted by the initial DDMMON-3 model during the suspected migration/spawning period. It was found that the conditions in the LDR during this period should allow passage of Burbot based on ADCP transects the output from the River2D model and literature reviewed on critical swimming speeds of the species.

5.2.2 Remaining Uncertainties and ASPD Focused Recommendations

DDMMON-2 focused on obtaining information on timing and habitat use of the adult Rainbow Trout, adult Mountain Whitefish and juvenile stages of both species within the LDR. Remaining uncertainties related to stranding risk and recommendations for operational or scientific consideration are summarized by species and life stage.

5.2.2.1 Adult and Egg Stage Rainbow Trout

The main uncertainties regarding the impact of operational ramping rates, magnitude of reductions and time of day of flow changes on stranding risk of adult and egg stage Rainbow Trout in the LDR are: 1) whether the magnitude of stage change in spring operations can be modified during incubation to minimize the risks of egg mortality due to stranding, lack of oxygen, siltation or high temperatures to allow population recruitment from the LDR tailout site, and 2) can the stranding of juvenile Rainbow Trout be further reduced with modifications to the ramping rate, time of day and magnitude of reductions from DDM (which is discussed in the DDMMON-16 section).

The current spring target to meet a minimum flow requirement of 73 m^3/s at the DRL Water Survey of Canada (WSC) gauge can be met through combined flow from DDM and the Lardeau River or by strictly flow from the Lardeau River. The magnitude of this change in DDM flow coincides with the Rainbow Trout egg incubation period and the result may not provide sufficient flow, oxygenation and depth for redds and can result and has resulted in egg and redd stranding. This data gap was partially addressed in February 2011 by the installation of the Duncan Above Lardeau (DAL) gauging platform to collect stage, discharge and water temperature during incubation. Further combined modeling and empirical efforts from DDMMON-3 and -2 results may improve the hydraulic model to allow it to assess water depths at the fine scale required to predict Rainbow Trout redd dewatering (Thorley et al. 2012). A river stage of 1.5m is sufficient to keep the majority of redds inundated can likely be obtained by having flows of 100 m^3s from the dam or 200 m^3s from the Lardeau River in spring (Thorley et al. 2012) though further parameterization of the DDMMON-3 model to confirm or alter this projection would be helpful.

It is unknown whether the magnitude and time of day of DDM flow changes in the spring and summer to facilitate Bull Trout passage through the dam have an impact on dewatering Rainbow Trout redds and stranding the eggs or newly emergent fry. This data gap needs to be addressed to confirm that operations during the BT transfer period do not pose a risk to redd or fry stranding.

5.2.2.2 Adult and Egg Stage Mountain Whitefish

The main uncertainties with respect to stranding for adult and egg stage Mountain Whitefish are: 1) whether early spawning locations are vulnerable to dewatering and/or egg stranding due to the magnitude of change in the fall operations; and, 2) whether emergent fry and late stage incubating eggs could be dewatered by the size of the drop to spring level flows (April 10).

Mountain Whitefish were found to spawn in water ranging from 0 – 2 m deep, with peak spawning at 0.88 m deep, which means the spawning locations may be vulnerable to exposure due to water level reductions of particular magnitude. Data gaps around this stranding risk include identifying all spawning locations within the LDR and getting a more precise estimate of water depths and determining if eggs are being stranded by the fall operational drops. This is unlikely to be a major source of stranding given the Kokanee protection flows drop the water levels prior to the estimated main spawning window (Thorley et al. 2012).

The emergence timing was modelled with spawning event and water temperature data to occur from early December until early May with peak emergence in the February – March period (Thorley et al. 2012) which means the reduction to spring flow levels may strand eggs or emergent fry depending upon its magnitude.

Currently, the information on the number of ATUs required for emergence of Mountain Whitefish population in the LDR is limited to that described in the literature over a range of 321 - 495 degree days (Brown 1952, Rajagopal 1979, Stalnaker and Gresswell 1974, Ford et al. 1995, Brinkman and Vieira 2009, Golder 2014b). The drop to minimum flows prior to the end of the emergence period could strand deposited eggs. Opportunities to keep eggs inundated should be explored within operational constraints. Data gaps for this potential stranding risk include actual ATUs for MW in the LDR which could be resolved with fry seine netting and / or incubation experiments and depth of egg deposition which could be assessed with experiments using water hardened eggs and dispersing and tracking them in known spawning areas.

5.2.2.3 Juvenile Rainbow Trout and Mountain Whitefish

The main uncertainties with respect to stranding for the juvenile stages of Rainbow Trout and Mountain Whitefish include: 1) overwintering habitat needs, particularly where side channel habitat may be used thus increasing the risk of stranding depending upon the magnitude of reductions and the ramping rates used to achieve the drops; and, 2) the stranding mortality rates of the juveniles of each species and how that relates to the population sizes maintained in the LDR.

A data gap related to these uncertainties is the considerable uncertainty around abundance and movements of the early life stages of Rainbow Trout and Mountain Whitefish, particularly with regard to what habitats they are selecting that may lead to increased stranding risk for their first winter. It is also unknown how many days after rewetting a site that juvenile fish repopulate the area; this knowledge would allow a better understanding of the effect of dewatering on habitat suitability and use and on how to time operational reductions to minimize stranding and maximize fish productivity. Understanding how the stranding mortality may relate to population size of these two species has been reasonably well addressed for Rainbow Trout, but not for Mountain Whitefish. Data were collected on size and abundance of the juvenile Mountain Whitefish in 2012 and 2013, but in-depth analysis has not yet occurred as Rainbow Trout were the focus species of the HCTF and FWCP program (Andrusak 2013). It is highly recommended that these data be analysed and the abundance of the juvenile Mountain Whitefish be assessed inter-annually to determine if this population is in difficulty or if it is doing well in the LDR so that stranding risk studies can be narrowed to relevant species. In order to refine the estimates of stranding mortality rates and determine if there are population level effects, it is recommended to conduct more extensive mark-resighting experiments to reduce the uncertainty surrounding the observer efficiency estimates for both Rainbow Trout and Mountain Whitefish. It may be possible to implement this through the ongoing DDMMON-16 program.

5.3 DDMMON-3: LDR Hydraulic Model Development

The objectives of this multi-year study (2009, 2010, 2012, and 2017) are to conduct a channel survey, complete hydraulic and habitat modelling of the LDR to assess channel changes and operational impacts on fisheries habitats for consideration in future flow planning processes, and to help define ramping rate protocols. The main hypotheses ask the study program to compare 2-Dimensional models to the original HEC-RAS 1D model conducted for the WUP planning process, to determine if habitat use predictions from the model accurately reflect those observed empirically through habitat use studies, to assess the floodplain morphology and its effect on the flow-habitat relationships predicted by the model, and to determine if the modelled stranding risk estimates are a reasonable surrogate for stranding surveys. The program is implemented over the 10 year WUP review period to support the ASDP and improve Performance Measure accuracy. The model is viewed as a tool with which to develop options for managing dam operations to reduce potential impacts to fish and fish habitat. Relevant reports from this program are: (Lewis and Healey 2009, NHC 2010, Gellis and Chilibeck 2013).

Table 7 summarizes the findings from DDMMON-3 in relation to operational implications of model development on fish stranding risk.

Table 7: DDMMON-3 – Operational Implications of Model Development on Fish Stranding Risk.

Operational Application	Potential Benefits of Operational Application	Biological Opinion	Uncertainties	Recommended Revision for Protocol
The flow model could improve the ability to predict fish stranding habitats and stranding events.	Over time, the coupling of the model and the stranding assessment protocol could lead to refined stranding risk categories for areas of the river.	Recommend long-term database of stranding events for eventual critical analysis of factors linked to increased stranding risk, as well as efforts to link model runs and testing with stranding assessments.	Model needs to be rerun at actual flow levels before and after a drop and its performance assessed empirically. It was recommended in the 2010 DDMMON-3 report that the scale of the model should be restricted down to index sites or a smaller scale to attempt to improve results.	Assign risk categories to known stranding sites in the river for specific flow changes.
The habitat-flow relationships for fish species of interest may be incorporated into DDM WUP PMs for predicting the implications of operational targets.	May allow comparison of different potential operational strategies and an optimization of operations for fisheries resources.	Operations to reach discharge targets in the LDR may have impacts on the early life stages of the species of interest.	The modelled values for WUA need to be assessed in context of the findings of biologically focused WUP programs.	Evaluating habitat availability at each flow will provide opportunity to determine stranding risk periods/flow thresholds based on use.
What are the implications of Bull Trout transfer operations (zero discharge followed by 24 m^3/s increase)?	The current operation eliminates the risk of attracting additional Bull Trout to LLOG#1 which would likely result in either fish injury or fish mortality.	River stage changes, particularly in Reach 1 may be detrimental to fisheries resources using the area.	The implication of the operations to Rainbow Trout redds and fish stranding still needs to be further evaluated. This topic was not revisited in the 2013 report.	None at this time.

5.3.1 Project Outcomes

The Year 5 DDMMON-3 program updated the DEM (Digital Elevation Model) and hydraulic modeling platform from a RIVER 2D model to a TELEMAC 2D model with transient modeling capabilities to simulate flow change impacts. This was designed to increase model accuracy related to the temporal aspect of flow changes during down ramping events.

The DDMMON-3 Year 2 report (NHC 2010) concluded that there were significant differences between the original HEC-RAS 1D model developed for the Water Use Plan performance measures and the RIVER 2D model output. The differences were attributed to limitations in the 1D model and available data, as well as morphological changes in the LDR, which occurred between the origination of the 1D model in 2003 and the RIVER 2D model in 2010.

The test of whether the habitat use predicted in the model was accurate when compared to DDMMON-2 results was reported as inconclusive in the 2013 report (Gellis and Chilibeck 2013) and model predictions did not match empirical data in the DDMMON-2 report (Thorley et al. 2012). Incorporation of DDMMON-2 and -16 results during the Year 10 monitoring program for DDMMON–3 is required to determine the model's utility in addressing questions of habitat use relevant to stranding.

Habitat use curves (HUC) from Ecofish (2009) and Thorley et al. (2012) were utilized to estimate habitat suitability indices (HSI) and weighted usable area (WUA) for Rainbow Trout fry and spawning Kokanee. For Rainbow Trout, there was an inverse relationship between discharge and WUA with values that were lower than predicted with the earlier RIVER 2D version of the model. This was due to the tighter habitat use curves for depth and velocity estimated for DDMMON-2. Rainbow Trout fry use extremely shallow water, and the hydraulic model was not able to resolve differences at such a fine scale and the habitat use curve didn't include a zero probability at depth 0 due to the way HSI curves are derived.

When considering depth and velocity, the WUA for Kokanee spawners increased as flows increased with a maximum value at 275 m^3/s. When substrate was considered the maximum WUA value occurred at approximately 250 m^3/s due to gravel inundations. This result should be field verified with the ongoing DDMMON-4 to provide iterative refinement for the hydraulic model.

Assessment of the impact of channel morphology changes on flow-habitat relationships was deferred until Year 10 (2017) of the program. Flow-habitat relationships from the TELEMAC 2D model differed from the RIVER 2D model, but the changes may not be solely attributed to morphological change. Other contributing factors may include updated depth and velocity preference curves for rainbow trout fry rearing and improved model capability. Regardless of the changes in channel morphology, fish stranding is defined by the probability of stranding and by natural fish density in an area (Section 6.2, Figure 5). Habitat use studies help discern patterns of fish use by species and life stage within the LDR. Stranding predictions then allow the determination of stranding risk for fish in those areas and the amount of area dewatered. In Year 2, the RIVER 2D model did not provide predictions related to stranding as hoped. As a result, it was recommended to refine the model of stranding to smaller scale areas (i.e., DDMMON-16 index stranding sites) and in relation to additional habitat variables such as slope, sediment size or fish use (NHC 2010). These recommendations were not completed for the Year 5 report and should be planned for Year 10. The stranding estimate run with the TELEMAC 2D model in 2013 was overlaid on the DDMMON-16 index stranding site S4.2R for the three reductions that occurred between September 25 - October 2, 2012. The conclusions were that the predicted changes stayed within the boundary of the stranding site (Gellis and Chilibeck 2013). In Year 5, comparisons were not made between DDMMON-3 model results and fish stranding distributions obtained in the DDMMON-16 program. Further coordination between both programs prior to and during Year 10 is required to achieve this.

Ramping rates were calculated for the fall flow reductions carried out between September 25 and October 2, 2012 for model predicted Kokanee spawning habitat and Rainbow Trout fry habitat. The maximum ramping rate for Kokanee habitat was 30 cm/h with the normal rate generally less than 19 cm/h. Both rates were higher that the ramping rate of 10 cm/h that is recommended in the updated ASPD to reduce fish stranding (Westcott et al. 2013). The maximum ramping rate for Rainbow Trout fry habitat was 24 cm/h with the values generally between 5 and 9 cm/h.

5.3.2 Remaining Uncertainties and ASPD Focused Recommendations

Further model runs are required for the TELEMAC 2D model to aid in predicting dewatered area within identified stranding sites resulting from operational reductions or to aid in predicting fish habitat wetted usable area (WUA) within the LDR from which to determine stranding potential. Also, it is uncertain whether the WUA predicted by the hydraulic model are representative of the information gathered under other biological programs (i.e., the WUA for spawning Kokanee was not compared to the data from DDMMON-4) . The WUAs reported in 2013 do not take into account the refined spawn timing windows derived in DDMMON-2 and -4 nor do they account for the habitat use by several fish species life stages.

Based on the results of other related DDMMON programs, the following recommendations for Year 10 of the DDMMON-3 program are as follows:

1) It is recommended that the WUA predicted by the model take into account the biologically relevant information from other WUPs. Weighted Usable Area (WUA) calculations should be groundtruthed with observations by life stage and species.

2) Performance measure re-analysis in year 10 of the DDMMON-3 needs to include the updated WUA predictions from recommendation #1 and applied to updated life history timing provided from DDMMON-2 and DDMMON-4. It would be helpful to provide performance measures from DDMMON#3 in flow units rather than percentiles so they can be compared to BC Hydro order tables and known timing of changes.

3) It was recommended that while the six hydrologic gauges are in place, the stage rate change of differing flow reduction strategies be examined. For example, determining if splitting hourly flow reductions into four equal reductions that occur every 15 minutes reduces the stage rate change. This can be completed during large flow reductions such as the fall reductions to reach Kokanee protection flows.

4) Currently there is no analysis of the TELEMAC 2D model in relation to predicting dewatered area, and the current model outputs for stranding predictions need to be calibrated based on DDMMON-16 fish stranding results.

5.4 DDMMON-4: Lower Duncan River Kokanee Spawn Monitoring

Duncan Dam Water Use Plan operations during LDR Kokanee spawning required that significant magnitude flow reductions occur mid-way through the spawning period. The WUP consultative committee recommended studies to assess the potential impact of these operations be undertaken that would include the assessment of potential stranding on the LDR population, and the effectiveness of potential mitigation options where deemed necessary. Annual Kokanee spawning surveys and mapping in LDR, Meadow Creek and Lardeau River were designed to determine the impacts of WUP operations on Kokanee spawning in LDR. This study was developed to collect information on Kokanee spawning habitats and use, timing of spawning and fry emergence. Also, this study was developed to investigate the operational, environmental and physical cues for spawning onset, as well as the biological sampling of kokanee morphology in the LDR, Meadow Creek and the Lardeau River to determine if different sub-populations exist. The program is ongoing for enumeration, but the 2012 report reviewed here represents a synthesis of the four year Kokanee WUP program (2008 to 2011). This study assessed how Kokanee may be affected by water level fluctuations with respect to spawning habitat availability, as well as the success of egg incubation and fry emergence. Associated references for DDMMON-4 include (Porto and Lawrence 2010, 2010 p. -4, Porto et al. 2012 p. -4).

Table 8 summarizes the DDMMON-4 operational implications of Kokanee spawning strategies on stranding risk.

Table 8: DDMMON-4 - Operational Implications of Kokanee Spawning on Fish Stranding Risk.

Operational Application	Potential Benefits of Operational Application	Biological Opinion	Remaining Uncertainty and Research Recommendation	Recommendation for Protocol Revision
The effect of the maximum and minimum flow targets on reproductive success of Kokanee in the Duncan system.	Fall/Winter operations may increase the spawning success of Kokanee for the benefit of the Duncan/Lardeau population.	Spawn and emergence timing have been more accurately estimated with peak spawning between September 27 and October 13 and hatching occurring in late December and early January with ponding in February and March. These timing estimates will continue to be refined over the WUP review period. The fish spawning in the LDR, Lardeau River and Meadow Creek are all one genetic population and the LDR population contributes 0.14% of the total egg production for the population. The warmth of the water in the LDR causes emergence ~ 3 months earlier than in the other two systems.	While the Infectious Haematopoietic Necrosis Virus (IHNV) is present at the Meadow Creek spawning channel, it is recommended that all recommendations in Porto et al. 2012 be taken to provide as much production as possible.	Minimize the magnitude of reductions during the spawning and incubation period to prevent dewaering of eggs and redds..

5.4.1 Project Outcomes

Spawning Kokanee predominantly use the upper 7 km of the LDR and all sidechannels that are wetted at the time of spawning with the exception of the DDM discharge channel at river km (RKm) 0.6. There have been spawning Kokanee observed downstream of RKm 7 at RKm 8L, 8.3L and 10.2L when spawner numbers were high. Sidechannel numbers of spawning fish peaked prior to mainstem numbers in 2008 and 2009, but the opposite occurred in 2010 with the mainstem numbers peaking earlier. The upper LDR appears to be much more heavily used by Kokanee spawners than the bottom half of the river and habitats downstream river km 7 experience very low use. Impacts related to operations are more pronounced in the upper river as this portion of the river experiences higher ramping rates and higher amounts dewatered habitat during flow reductions. These impacts are dampened in the downstream portions of the LDR due to distance from DDM and the mitigative effect of tributary inflows.

The peak spawn timing and spawner abundance were analysed with Area Under the Curve Bayesian analysis (AUC) conducted on helicopter survey data of the spawning Kokanee. Peak counts in the LDR were observed in the last week of September or the first week of October, similar to findings in the system since 2002. Spawners were present in the system beginning in the first week of September and were usually gone by the last week of October. Kokanee are present in the LDR and subject to adult fish stranding between early September and October 15 during migration. The AUC found the peak for Kokanee spawning to lie between September 27 and October 13. The expansion factor to covert the peak count to the total spawner numbers for the LDR is 1.2

(range 0.8 - 2.7) and the LDR accounts for 3% of the total Kokanee escapement numbers for the combined Lardeau River, Meadow Creek and the LDR system. Between 2008 to 2011, LDR Kokanee spawners were estimated at 35,600, 16,900, 22,200 and 96,700, respectively.

Spawning success was measured using a modified Potential Egg Deposition (PED) formula and was assessed pre reduction and post reduction to provide a measure of spawning success [(PEDafter / PEDbefore)*100)]. The PED had the assumptions that: the sex ratio of all spawners was 1:1, dewatered areas kill 100% of the eggs within redds, that the number of females immediately before a reduction is representative and that female Kokanee make 1 redd and spawn once in the area where they are observed. The population impact of dewatering was then calculated using the PED and spawning success measures in addition to incorporating the survival estimates for each life stage with 23% survival assumed for the egg to fry and 5% survival for the fry to adult transition. The egg to fry survival was assessed with incubation capsule experiments conducted *in situ* in the LDR and ranged from 0 - 80% with the mean value of 23%. The Meadow Creek Spawning Channel (MCSC) has a mean egg to fry survival of 35% (range 6 - 64%). Overall spawning success in the mainstem was estimated at 100% over all years for which it was calculated (2009 - 2011) and spawning success in sidechannels was estimated at 67% in 2009, 94% in 2010 and 86% in 2011.

Potential differences in morphology between the Kokanee spawning in Meadow Creek, Lardeau River and LDR were assessed with a Bayesian linear model. Statistically significant morphological differences explained by the location in which an adult Kokanee was sampled were found in body size, egg retention and age at reproduction. However, DNA analyses of the three spawning stocks found no difference between them and found fish from all three systems to be from one genetic population (Lemay and Russello 2011) so stranding of LDR fish specifically is not of a particular management concern for a distinct genetic population.

Egg losses due to dewatering were estimated for each year with the highest losses occurring in 2011. Areas prone to dewatering were identified and the population impact of dewatering for the population in its entirety (including Meadow Creek, Lardeau River and LDR) was estimated to be 0.14%, which is equivalent to approximately 2000 fish (range from 0.01 - 0.24% or 1053 - 9194 fish). In 2008, sidechannel 6.9R dewatered after the Kokanee had spawned in the habitat and 96% of the eggs succumbed to mortality. Changes made to operations in 2009 and 2010 kept the sidechannel from dewatering to allow for monitoring, but normal operations typically dewater it on October 1st.

Over the four years of study, water temperatures during the spawning period averaged 10 – 13°C. The highest water temperatures were encountered in 2009 with values of 18 - 19°C in late August. Water temperatures typically declined to 8 – 9°C by the end of October. In the sidechannels, water temperatures were significantly higher in comparison to the mainstem LDR, while gravel and mean column temperatures were not significantly different from one another. Fry emergence was calculated based on known ATUs from Meadow Creek with eyed stage at 400 ATUs and hatch at 700 ATUs. The incubation studies conducted on the LDR found that the ATUs to eyed stage (413 ATUs) and alevins (692 ATUs) were within the range from Meadow Creek. Hatching in the LDR was in December and January with ponding in February and March, depending on the water temperatures experienced during incubation. Higher water temperatures in the LDR result in the emergence of fry occurring 3 months prior to emergence in the other systems making up the Kokanee population. In hatcheries, early emergent fry are weaker with larger yolk sacs in relation to body size.

5.4.2 Remaining Uncertainties and ASPD Focused Recommendations

Data gaps for the spawning Kokanee and the eggs and emergent fry in the LDR pertaining to stranding risk include: 1) is there a potential population level impact of dewatering the eggs spawned in the LDR due to the magnitude of reductions during incubation, 2) utilization of habitat by fry may be affected by the ramping rate and magnitude of reductions. The outstanding data gaps and recommended approaches to address them are summarized below.

1) Within Porto et al. 2012, it is very clearly stated that the impacts of dewatering redds on the overall population in the LDR are still uncertain. Given the fact the LDR's contribution to the overall population is relatively small; it seems that it may not be the best use of resources to invest in trying to save 0.14% of the production. If the Kokanee population in Kootenay Lake is at carrying capacity with the current fry production out of combined system, then the production from the LDR is likely of low importance and operational management can focus efforts elsewhere and consider the rate of loss within the LDR due to dewatering and operational effects as 'acceptable'. In the most recent publicly available reports on the Kootenay Lake fertilization program (Schindler et al. 2011, 2013) it is stated that "the Kokanee fish size and fecundity observed in the 1990s and 2000s is expected to decline and stabilize close to the long term average as the abundance of Kokanee reaches lake carrying capacity" and that this was expected to occur in the next few years. The report also notes that the fertilization has increased the carrying capacity from 540,000 spawners in the pre-fertilization era to ~727,000 spawners in the current regime of ongoing fertilization (Schindler et al. 2013). A request for more recent study year's reports was sent to FWCP to determine whether the Kokanee are at the newly predicted carrying capacity yet, but the reports for the study years 2011 to 2013 are currently not finalized.

2) The salmonid virus IHNV that infected the MCSC in 2013 confounded the assessment of ways to minimize operational impacts on the LDR Kokanee population. The transmission rates of IHNV are greater in areas with higher fish density as the mucus coating of fish is an infective vector, and nipping and territorial behaviour occur more frequently in situations of higher density (Foott et al. 2006). The mortality rates in MCSC will likely be higher than in the Lardeau River or LDR due to the density levels so the production in these rivers may become proportionally more important until the spawning channel becomes disease free. It was therefore recommended that the ongoing population monitoring focus on the proportion of healthy out-migrating fry from each system. The importance of the lost production due to dewatering can then be refined as the IHNV virus situation progresses and resolves. All of the recommendations described below can then be taken into consideration to minimize the losses in the LDR. If the situation returns to what was the status quo prior to infection, it is recommended that the above flow recommendation be followed and no further investitures occur.

3) The spawn timing estimates show that spawn onset ranges from September 27th to October 13th. The early end of that range is before the Kokanee protection flows are initiated so it is recommended that the transition to Kokanee protection flows be concluded on September 27th rather than October 1st to prevent spawning of adults as well as redds and eggs. In Porto et al. 2012 it is recommended that further monitoring of egg losses and fry production occur if this recommendation is implemented to see if there is any improvement in spawning success.

4) Sidechannel 6.9R was dewatered in 2008 with loss of 96% of the production of the approximately 1500 spawners that used that sidechannel. Operations were modified in 2009 and 2010 to prevent this, but normal operations can dewater sidechannels before fry emergence. The modeled flow data and habitat use curves derived in DDDMMON-3 can be used to determine the lost spawning production due to operations in a particular year in order to track the percentage loss per annum. The PED calculations from this study program could be compared with the use of the TELEMAC 2D model to see how the predictions compare between the two methods. These calculations should be completed annually with the ongoing monitoring program.

5) Analysis was not completed on operational variables to determine why the mainstem spawner numbers peaked before the sidechannel numbers in 2010. If 2010 was an atypical year operationally, it may inform some of the program's uncertainties. It is recommended that this analysis be conducted as part of the ongoing monitoring program.

6) Habitat use data was collected opportunistically in 2009, but not with the objective of developing system specific HUC for Kokanee in the LDR. With the updated TELEMAC 2D model from DDMMON-3, this should be revisited so that usable area for spawning kokanee and fry rearing can be calculated for the LDR and stranding mitigated by managing flows to keep spawning areas wetted.

5.5 DDMMON-16: LDR Fish Stranding Impact Monitoring

This 10 year monitoring program (2008 – 2017) assesses the effectiveness of operational measures on the levels and location of stranding in the LDR. It also looks at the level of impact of stranding may be on the fish populations for species of interest while developing an assessment protocol that optimizes the ability to estimate impacts and salvage stranded fish. References for this program include: Hildebrand 2009, 2010, 2011b, Hildebrand and Irvine 2012, Golder 2014a and 2014 in prep. Sampling protocols and data forms for this program are provided on Appendix A.

Table 9 summarizes the status of the DDMMON-16 Management Questions and Objectives.

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

DDMMON-16 Management Question	DDMMON-16 Specific Hypothesis	DDMMON-16 Year 6 (2013-2014) Status Summary
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. 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.
What are the levels of impact to resident fish populations associated with fish stranding events on the lower Duncan River?	*Ho_1: Fish stranding observed at index sites along the lower Duncan River floodplain is representative of overall stranding.*	Index sites were not originally selected to be representative of the entire LDR, but to focus on sites believed to have the highest amounts of stranding based on amount dewatered area and suitable habitat. Index sites tend to be of lower gradient and wider than the non-index sites, therefore more area dewaters at these sites. In the Year 4 analysis, the number of pools per unit area of exposed habitat did not vary between index and non-index sites nor did the number of fish per pools. Stranding rates per lineal distance do not differ between index and non-index sites, but differ due to greater dewatered area within index sites. Therefore, the greater area dewatered in index sites strands higher numbers of fish in comparison to non-index sites. Index sites appear to provide an estimate that is biased high. Therefore, hypothesis Ho_1 is rejected. This will be re-examined in the Year 7 in-depth interpretive report with all available project data to determine if complete dataset supports the rejection of hypothesis Ho_1.
	Ho_2: Fish populations in the lower Duncan River are not significantly impacted by fish stranding events.	Estimates for the number of Rainbow Trout juveniles stranded in pools were relatively low and precise. While interstitial stranding is likely to be biologically important, the current estimates were upwardly biased and are uncertain. There was a seasonal component to pool stranding, with higher stranding in fall, but at this point it cannot be determined whether this was due to less fish in the system in the spring vs. the fall or to a decreased risk of stranding. 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. Similar to previous study years, with the most recent abundance and stranding estimates for Rainbow Trout, hypothesis Ho_2 was rejected. Therefore, based on the current project dataset 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.

5.5.1 Project Outcomes

Based on the current state of knowledge, the flow reduction measures implemented under the ASPD are effective at reducing fish stranding. Comparable fish stranding estimates from the pre-WUP and post-WUP periods were not obtainable due to changing sampling methods. Therefore, only assessments on the amount and rate of habitat dewatering can be made in regards to the effectiveness of the ASPD measures.

Total area dewatered during all annual flow reductions was used to determine differences in pre- and post-WUP operations, because the area exposed relates directly to the hydraulic and stranding analysis models. The examination of the amount of area of exposed habitat per year due to LDR discharge reductions indicated that post-WUP flows have resulted in the dewatering of less area compared to pre-WUP operations. Interannual variability in discharge has also been reduced under post-WUP operations. Conversely, significant differences in total reduction magnitude and mean ramping rate between pre- and post-WUP operations were not identified, although pre-WUP ramping rates exhibited much higher variability.

As recommended by the DDMMON-1 and -15 Programs (Poisson and Golder 2010, Golder 2012), the current ASP stipulates that Duncan Dam releases be reduced at a maximum hourly ramping rate of 28 m^3/s. This is meant to ensure a stage change of 10 cm/hr or less at the majority of identified stranding sites in the LDR. Data trends identified in those programs indicated that this slow rate of change during down ramping is believed to reduce the risk of fish stranding, which is also supported by studies conducted in Norway (Halleraker et al. 2003). Halleraker et al. (2003) recommended similar ramping rates to reduce stranding rates of salmonids, especially after an extended period of stable flows. This operating requirement has resulted in consistently similar ramping rates during post-WUP operations in the LDR.

Estimates for the number of rainbow trout juveniles stranded in pools were relatively precise and relatively low. Previous analyses showed that the wetted area of pool was not a predictive variable for rainbow juvenile stranding (Thorley et al 2011, Hildebrand and Irvine 2012). In the current dataset, seasonal effect on pool stranding of juvenile Rainbow Trout was found to be significant, with median fall stranding estimates over three times higher than those for winter/spring. This may be due to lower juvenile fish densities in the system in the winter/spring vs. the fall or to a decreased risk of stranding in that period. Juvenile Mountain Whitefish encounters were too low to identify seasonal effects on stranding.

Although the estimated numbers of interstitially stranded Rainbow Trout and Mountain Whitefish in the LDR are high and the estimates are still uncertain, they are more precise than the estimates obtained in previous years (Hildebrand 2011a, Golder and Poisson 2012). Although interstitial stranding is likely to be biologically important, the substantially higher numbers of stranded fish documented in pools strongly indicates that the current interstitial estimates are upwardly biased and uncertain. The probable reason for the upward bias is that the modelled abundance for interstitial stranding assumes a Poisson distribution, and data scarcity in regards to interstitially stranded fish can lead to relatively high and uncertain estimated stranding as extensive amounts of habitat are dewatered. A relationship between interstitially stranded fish counts and ramping rate was not found. While the data showed no trends for this relationship, this may also be due to data scarcity.

In the early years of this program, habitat with potential stranding risk were stratified by gradient into low slope (≤4% gradient) and high slope (>4% gradient) areas. Based on the in-depth analysis conducted in Year 4 of this program, considerably higher amounts of low slope habitat was dewatered during flow reductions from DDM, and the dewatered low slope habitats had substantially more fish interstitially stranded following flow reductions than high slope habitats (Hildebrand and Irvine 2012). However, those estimates may have been overestimates since they were based on any dewatered zone of the river being categorized as stranding habitat, while subsequent field assessments have excluded multiple areas based on the extreme gradient they contain. In addition, the estimates of area dewatered were only conducted from three outputs of the early version of the original DDMMON-3 hydraulic model. Statistically significant relationships between interstitially stranded fish counts and slope in the current dataset were not found. While the data showed no trends for the relationship, this may be due to data scarcity. This relationship should be re-evaluated as more data are collected. The results from the current dataset also suggest that slope did not have an effect on the formation of isolated pools within the study area. Also, a relationship between slope and the number of fish stranded in isolated pools was not identified.

Based on the findings of previous study years (Hildebrand and Irvine 2012), index sites tended to be of lower gradient than non-index sites. Interestingly, the number of pools per unit area of exposed habitat did not vary between index and non-index sites nor did the number of fish per pools. This suggested that other than being lower gradient and therefore exposing more area, stranding rates (stranding per lineal km of river) do not differ substantially between index and non-index sites. Overall, index sites strand more fish because more area dewaters at these sites during flow reductions. Therefore, based on the Year 4 analyses, index sites had a bias toward higher stranding rates.

The estimated fall 2010 population of Rainbow Trout juveniles within the LDR as modeled from DDMMON-2 data was 48,981 (95% credibility intervals range from 30,828 – 73,594; Thorley et al. 2012). The fall 2013 abundance estimate for juveniles obtained in this program was 16,330 (95% credibility intervals range from 9,985 – 22,874). These findings should be interpreted with caution as the densities documented in the present study were substantially lower than in previous DDMMON-2 study years.

The sum of the estimated fall 2013 interstitial and pool stranded Rainbow Trout in the LDR had a median value of 22,598 and minimum and maximum 95% credibility intervals of 9,535 and 50,663 respectively. Estimates for the number of Rainbow Trout juveniles stranded in pools attributed to only 9.1% of the total stranding estimated. Based on the likely overestimated interstitial stranding estimates, combined with the precise pool estimates from the present dataset, it therefore must be concluded that fish stranding as a result of DDM operations has a significant impact on Rainbow Trout populations. With the uncertainties in the current dataset, it is not possible to determine the level of impact. The further refinement of interstitial stranding rates may reverse this finding. An ongoing management program (run jointly by the Fish and Wildlife Compensation Program and the Habitat Conservation Trust Fund) provides spring Rainbow Trout abundance estimates in both the Lardeau and lower Duncan rivers. The snorkel surveys from that program that are conducted to obtain abundance estimates occur in March of every calendar year (Andrusak 2010, 2013a and 2013b). As the timing of the abundance estimates provided occurs after the winter/spring fish stranding assessments, it is not possible to utilize those estimates in determining if spring stranding rates impact population levels of Rainbow Trout. The yearly results of that program should continue to be examined to refine the abundance estimation methodology of this study and to monitor the comparability of the spring abundance estimates it provides.

Abundance estimates and associated confidence intervals for Mountain Whitefish for 2013 were similar to those obtained in by the DDMMON-2 program in the fall of 2010 (Thorley et al. 2012). This suggests that the Mountain Whitefish population in the LDR has remained relatively stable since 2010.

Mountain Whitefish encounters have been minimal in all study years. This consistently low level of stranding was not considered ecologically significant and will likely not result in a population level effect on Mountain Whitefish. To avoid large numbers of mountain whitefish being stranded during rapid night time reductions in flow (Poisson and Golder 2010), recent ASPD updates have recommended that daytime changes occur at the dam.

5.5.2 Remaining Uncertainties and ASPD Focused Recommendations

The following data gaps and uncertainties related to this program have been identified:

1) Interstitial stranding estimates obtained in the program have high uncertainty and are likely over-estimated, and substantial progress has been made to reduce the uncertainties associated with these estimates. As the dataset continues to grow each year, the uncertainty related to this estimate will continue to decrease.
2) Issues were raised in the Year 6 DDMMON-16 report about the accuracy of the initial RIVER 2D model created as part of the DDMMON-3 program for modeling conditions at habitats that pose a fish stranding risk. It may reduce the uncertainty about the area of exposed habitat to conduct several model runs with the updated TELEMAC 2D model from the DDMMON-3 program at varying DRL discharges.
3) There is a documented seasonal difference in stranding rates, with the highest rates observed during the fall season. This highlights opportunities to prioritize scrutiny of fall operations to maximize salvage efficiency.
4) The current FWCP Redfish Consulting Ltd. study program on the LDR and Lardeau River (Andrusak 2013) is estimating the population abundance for Rainbow Trout fry each spring. These values need to be compared to the stranding estimates each year in addition to the fall snorkel survey data from the WLR study to better understand the number of fish at risk of stranding in the system by season, and to determine if the decline in the Rainbow Trout population identified from the fall 2013 abundance estimates can be tracked through the spring of 2014. If the findings of that program do not support the current estimation of declining abundance, explore the feasibility of altering abundance estimation methodology (i.e., increasing sample effort during the fall snorkel surveys) in order to refine the abundance estimates of the species of interest.

5.6 2013 LDR Adaptive Stranding Protocol Document

Based on the review of the above WLR studies on the LDR in the context of what has been learned from the parallel process in developing the Columbia River stranding protocol, a list of recommended revisions to the Stranding Protocol is herein documented for use in the next cycle of revision of the protocol document. The protocol document version referred to in the below review is (Westcott et al. 2013).

Currently, BC Hydro assesses the impacts of the majority of planned flow reductions except when winter access is limited (e.g., ice and snow cover) or when a limited fish stranding risk (e.g., small water level reduction [<10 cm stage change or 14.16 m^3/s reduction from DDM]) reduce the benefits of a fish stranding assessment. If feasible, it is recommended that stranding assessments include smaller magnitude reductions in order to determine if the assessment of limited risk is indeed accurate. This may result in the refinement of the minimum reduction below which no further assessments are required. Assessments for approximately 10 small reductions are needed to assess stranding risk.

Within the discharge reduction implementation section it states that "It is assumed dam operation mimics natural inflow conditions on a daily basis when matching inflows or maintaining target water levels in the LDR and as a result, monitoring of fish stranding will not occur in the LDR". As the established minimum target water levels are not solely based upon the natural hydrograph prior to the construction of DDM, this assumption is based on little data. Examining the target flows for DDM in relation to natural hydrograph of Lardeau to determine if and when they diverge substantially, then ensuring that data on fish stranding for the drops that occur in those periods are collected would address this data gap.

Although the ASP document is not up for review/revision this year, the contact information appendix requires regular updating as BC Hydro staffing changes. The field sampling section generally needs updating as well with current information on accommodations and reference to the correct and most current stranding methodology.

The main difficulty with the LDR ASP document is that it follows a parallel structure with the Columbia when in fact, the amount and type of data collected, as well as the number of sites visited between the two systems, are very different. There are 15 years of data for the Columbia program since the advent of the standardized data collection in 1999, compared to over seven years of standardized data for the Duncan Program (beginning in 2006). Because of the index site paradigm that was imposed after the initial risk assessment of stranding (AMEC Earth & Environmental Ltd. 2004), non-index sites have rarely been assessed. In Year 4 of the DDMMON-16 program, a new methodology was developed to allow for comparisons of stranding rates between index and non-index sites. 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.

6.0 LITERATURE REVIEW

6.1 Background and Outline

From 2002 to present, fish stranding assessments have been conducted on the LDR. From 2004 to 2007, ramping experiments were conducted each autumn on the LDR in order to assess the effects of a set of environmental variables and operational strategies on the stranding rates of juvenile fish. WLR studies in the LDR have been ongoing since 2008 and each study program has had a component assessing the effect of flow operations on the species or habitat of interest in relation to the issue of fish stranding. All of the ongoing studies occur under the WUP operations regime for the LDR, which is defined in Table 10.

Table 10: Maximum flows for Duncan River downstream at the DRL (BC Hydro 2005). The minimum flow is 73 m^3/s year round.

Date	Maximum Flow (m^3/s)
August 1 to August 24	400
August 25 to September 24	250
September 25 to September 27	190
September 28 to September 30	130
October 1 to October 21	76
October 22 to December 21	110
December 22 to April 9	250
April 10 to May 15	120
May 16 to July 31	400

In 2008, a gap analysis and power analysis were requested by BC Hydro to assess the current state of knowledge about factors influencing fish stranding, and to provide recommendations that would inform future study years on the LDR. The document produced for that gap analysis included an extensive literature review completed in the context of the overall management question to be addressed within the ASPD program - “What are the best operating strategies at Duncan Dam to reduce fish stranding in the lower Duncan River?” (Irvine 2009). This document is an update of the 2009 literature review in order to incorporate recent knowledge about fish stranding.

This literature review component of DDMMON-15 therefore includes:

- An impact hypothesis diagram (Figure 5) summarizing the main mechanisms (both tested and untested) that may contribute to fish stranding; and,
- A brief discussion of the state of knowledge about each of the factors identified in the impact hypothesis diagram and a summary of the state of knowledge from the peer reviewed literature on fish stranding.

6.2 Introduction

Juvenile fish tend to reside in near shore waters where warmer water and food, as well as refugia from velocity, predation and other fish is abundant (Vehanen et al. 2000). Unfortunately, this puts them at risk of being stranded in isolated pools or in the interstices of dewatered substrate by sudden changes in water levels. These changes can result from natural flow decreases such as flood abatement (Beck and Associates 1989), or human imposed water level changes such as hydro-electric dam operations (Saltveit 2001, Bragg et al. 2005, Irvine et al. 2009). Dam operations can increase the frequency and magnitude of flow changes in order to meet obligations for storage, or to produce power at peak times (Berland 2004) with hydropeaking (rapid changes in the water levels in response to power demands). These water fluctuations can affect fish mortality rates (Annear et al. 2002), behaviour (Scruton et al. 2008), stress levels (Flodmark et al. 2002), growth rates (Harvey et al. 2006, Korman and Campana 2009), habitat use (Vehanen et al. 2000), and movement (Irvine 1986, Heggenes 1988, Krimmer et al. 2011).

Several operational and environmental factors may affect levels of fish stranding. These include altered water velocity due to flow changes (Irvine 1986), time of day and season (Irvine 1986, Vehanen et al. 2000, Robertson et al. 2004), the rate of the reduction (Bradford et al. 1995), the size and species of fish present (Irvine 1986, Heggenes 1988), the amount of time since the last reduction (the wetted history) (Irvine et al. 2009), and the number of reductions in a row (Irvine 1986). Fish behaviour, meteorological factors (Girard et al. 2003), and species specific life histories (Saltveit 2001, Flodmark et al. 2002, Huusko et al. 2007) can also affect the stranding risk.

Technical contributors to the planning of the DDMMON-15 study program outlined two management questions in order to address the overall Adaptive Stranding Protocol Development management question. Firstly, '***What is the relationship between stranding rate and the following factors?':***

- **Rate of river stage/total stage change**
- **Time of Day (day/night)**
- **Substrate**
- **Habitat configuration**
- **Cover**
- **Species**
- **Time of year (spring, fall, winter)**
- **Habitat stability/wetted history**

The second question posed by the technical contributors was '***What opportunities are available to mitigate stranding rate through operations or habitat change?***'

Furthering the understanding of how the environmental and operational factors listed in the first management question affect fish downstream of dams could enable the answering of the second question and the development of improved management regimes to minimize fish stranding in the LDR. Inferences drawn from the analysis of stranding monitoring data on the lower Columbia River and lower Kootenay River are also discussed in relation to the relevant factors. This review also allowed the assessment of additional factors that can affect fish stranding risk. Additional factors that were identified and considered with respect to the stranding rate assessment included: season, water temperature, moon phase, cloud cover, inter-species competition, habitat, fish size, conditioning reductions, food availability and velocity.

These factors are summarized in an impact hypothesis diagram (Figure 5), which conceptually links the factors and their effects. Impact hypothesis diagrams are a means to summarize the processes through which anthropogenic activities affect ecosystems and to causally relate the effects of dam operations or environmental variables on an environmental indicator or suite of indicators. The diagram presented below summarizes the factors that may affect the number of fish stranded on the LDR. Factors in solid outlined boxes are those within management control, while those in dashed line boxes are not. Directional arrows indicate causal linkages. Constructs above the dotted line are factors that may affect the processes leading to population size while boxes below the dotted line are the processes contributing to the population size. This diagram is a very simplified view of the factors of interest; the numerous potential interactions or autocorrelations are not shown. 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. There are factors that are not included in the diagram or discussion that could affect the number of fish stranded.

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 by density is what results in the number of fish stranded (Figure 5). The effect of each factor on the processes defining the number of fish stranded is discussed below in Section 6.3, as well as the state of knowledge about each factor's effects and variability.

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 stranding mortality and all other mortality mechanisms. 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 initiate as a result of the fish lost through stranding mortality) has yet to be determined for the LDR. This determination would require knowledge about the density dependent mechanisms acting on a specific population and this is difficult to ascertain with enough certainty to allow population projections (Higgins and Bradford 1996).

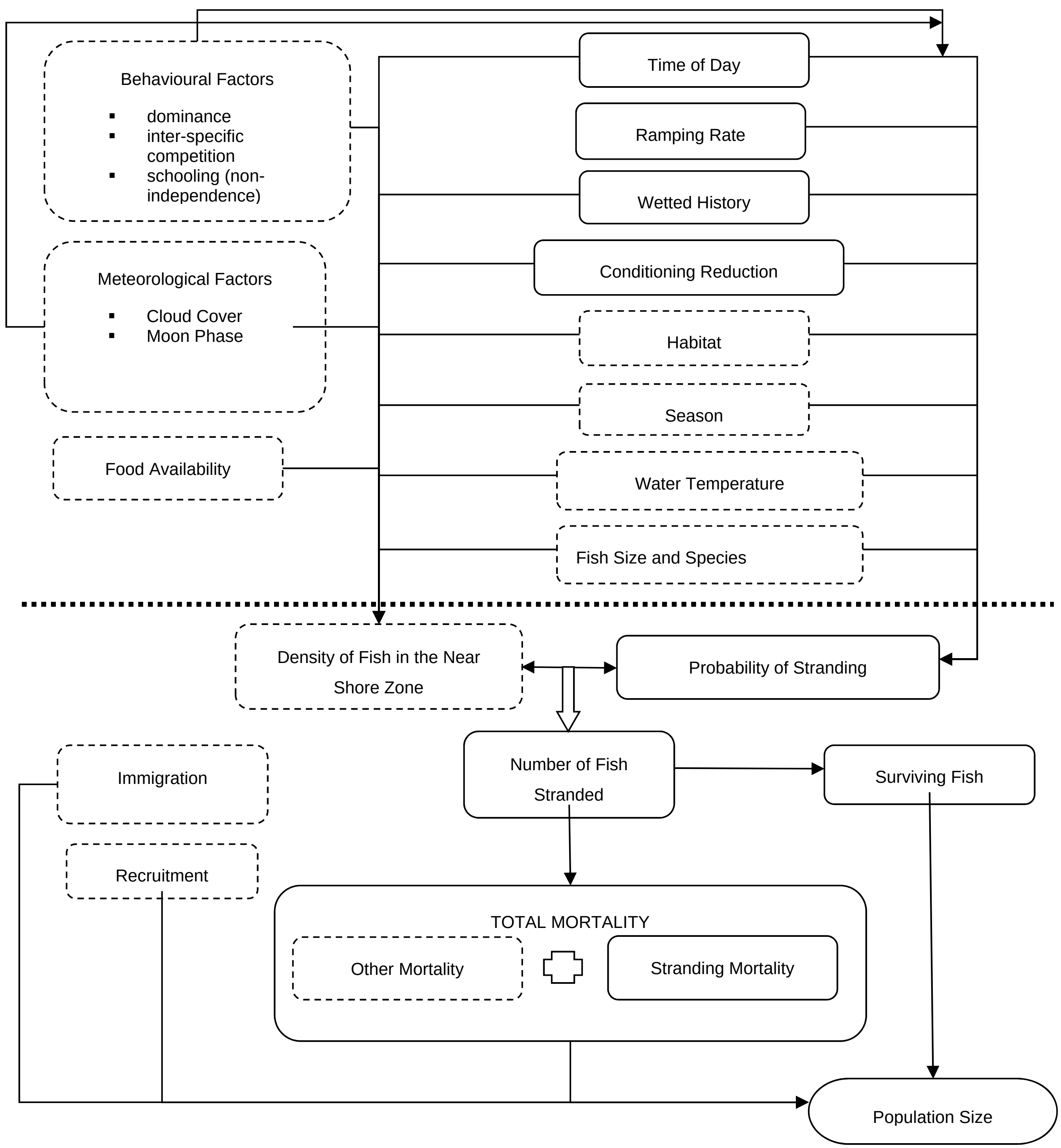

Figure 5: Impact hypothesis diagram for juvenile fish stranding on the lower Duncan River. Factors contributing to juvenile fish mortality are located above the dotted line, while items below the dotted line are processes feeding into the population size. Factors enclosed in boxes with dashed lines are not within direct management control and those in solid boxes are at least somewhat within management control.

6.3 Discussion of Factors from Impact Hypothesis Diagram

Many of the factors that potentially affect fish stranding interact in various ways or have significant correlations, and although the factors are divided into sections for ease of discussion, there is overlap.

As fish stranding is discussed below, certain terminology will be used. Pool stranding, also known in the literature as 'entrapment', denotes fish that have become stranded in potholes or divots that form in as the near shore area dewaters. The other mechanism for stranding normally defined in the literature is interstitial stranding. Interstitial stranding occurs when fish become stranded amongst cobble and pebble habitat as the water along the riverbank recedes. This type of stranding is also referred to as 'beaching' or 'gravel-bar stranding' in the literature. The 'varial zone' represents the area of the river that is continually dried and rewetted within operational norms. This is also called the 'nearshore zone' or the 'shallow margins'. When results of the ramping experiments conducted on the lower Columbia, Kootenay and Duncan rivers are discussed, only unmarked fish are used in the analysis and discussion. This was due to a documented difference in the behaviours of the marked and unmarked fish.

6.3.1 Time of Day

Time of day can be considered at a very fine temporal scale, but the simplest delineation is as a binomial variable with the values of night or day. This is how the LDR ramping experiments assessed this variable and how it is addressed in much of the literature on juvenile fish movement and behaviour. This approach to assessing time of day is limited in its subtlety since it is effectively testing the difference between behaviour in either dark or lighted conditions. This factor can either affect the probability of stranding or the density of fish in near shore areas.

In the four years of experimentation on the LDR, 15 night time and 16 daytime experimental net pens were completed successfully (Irvine and Hildebrand 2010). Although not statistically significant, there was a trend in the dataset for more fish to strand at night. This result is in alignment with other work (Hamilton and Buell 1976), as well as contradictory to some studies where night time drawdowns were associated with lower stranding rates (e.g., Bradford et al. 1995, Saltveit 2001). The variability in the Duncan River data is high, making it difficult to say with any certainty that the difference in stranding rates is robust. When a generalized linear model relating fish density on site with the time of day was fitted, the variable of time of day was not significant ($p = 0.08$), but fish density was slightly lower at night than during the day. This could mean that even with a higher probability of stranding at night, there could be less fish stranded due to lower density. This trend is also opposite to the trend in diurnal fish density seen in the Columbia River and in much of the literature. When data from 1999 - 2009 stranding monitoring surveys on the Columbia and Kootenay Rivers was examined, the worst time for stranding was late afternoon (Irvine 2010). Time of day was only significant in this binomial analysis at the most conservative definition of a stranding event, where 1 fish stranding was classified as an event. The current operational protocol on the LDR is to carry out daytime reductions if possible. To assess the impact of time of day further, stranding surveys done in the early morning after a night time reduction could be conducted and analyzed to ensure that the temporal risk of stranding is not being increased by this directive, given the weight of evidence in the literature.

Research on the movements and behaviour of juvenile salmonids with respect to time of day revealed a number of possible mechanisms that could explain the varied results. A study on brown trout, for example, found that the salmonids were more numerous in the varial zone at night than during the day in the winter season (Heggenes et al. 1993). The proposed cause for this behaviour was an optimization strategy of the brown trout. They sought shelter during the day to lower the risk of being trapped in bottom ice and foraged at night in the near shore areas with lower flow to minimize the energy expenditure and risk of predation while winter feeding (Heggenes et al. 1993, Linnansaari et al. 2008) Increased activity at night by salmonid species was noted in summer as well and it was postulated that this behaviour evolved to avoid avian predators (Gries 1997). In a study on Coho salmon (*Oncorhynchus kisutch*) and Rainbow Trout in stream channels in British Columbia, it was found that more fish stranded in the daytime because they were exhibiting hiding behaviour in interstitial spaces (Bradford et al. 1995). It is clear that time of day likely interacts strongly with season, with particularly large shifts in behaviour occurring in winter when juvenile fish are dealing with adverse conditions such as anchor and frazil ice, lowered metabolism, and reduced swimming ability (Heggenes et al. 1993, Vehanen et al. 2000, Huusko et al. 2007). Time of day likely interacts with fish size or life stage. This was the case in a study on Atlantic salmon (*Salmo salar*) when the parr were mainly active at night, while young-of-the-year were equally active during day and at night (Imre and Boisclair 2005). The effect of time of day on the downstream drift of fry has also been noted for several species, with Atlantic salmon mostly drifting at night, except when high flows forced daytime drift (Saltveit et al. 1995). Increase in downstream densities of juvenile Atlantic salmon was not seen even with the additional drift due to increased flows (Saltveit et al. 1995). Lowered Atlantic salmon parr densities have been observed in the reach immediately downstream of a hydroelectric dam; flow variables explained up to 42% of the variation in juvenile salmon densities (Ugedal et al. 2008).

6.3.2 Ramping Rate

Of the two processes driving the number of fish stranded – density of fish in the near shore zone and the probability of fish stranding – the ramping rate of flow reductions likely only affects the latter. Ramping rate is one of the factors that is in direct operational control and has been shown in other studies to have a statistically significant effect on the stranding rate of fish (e.g., Bradford et al. 1995). Downstream distance from the dam determines, in part, the ramping rate experienced by the fish as hydraulic processes attenuate flow reductions.

Ramping rate was assessed in every year of the DDMMON-1 program during ramping experiments on the LDR, either directly or indirectly through normal operational variation. The ‘fast’ rates tested on the Duncan system ranged from 19 to 36 cm/hour. The ‘slow’ category of ramping rate ranged from 4.9 to 9.8 cm/hour. Although a tendency for decreased stranding with slower ramping rates was consistently seen in the results from data from the LDR and lower Columbia River system, it has never been statistically significant in a classical sense, or in an information-theoretic sense (Burnham and Anderson 2002). In the analysis of 10 years of monitoring data from the lower Columbia River and lower Kootenay Rivers, ramping rate was never a significant predictor of the occurrence of a stranding event, neither was distance downstream from the dams (Irvine 2010).

There are certain study limitations on testing ramping rate effectively in the LDR, such as the operational constraint on DDM (4 kcfs/day) that sets an upper limit on rates changes that can be tested. Part of the reason that ramping rate on the LDR has never emerged as a statistically significant factor in stranding rates may be because the ‘fast’ ramping rate tested in this system is still slower than most ramping rates tested and reported in the literature. Another factor that affects the investigation of this variable in any large river system is that the sites on which stranding can be experimentally assessed are longitudinally distant from one another, and by definition experience different ramping rates and timing of the decrease in water stage. Related to the hydrological variability at a site scale is the limitation that is encountered at the pool level. It is difficult to know exactly what the fish are experiencing and almost impossible to know what ramping rate each individual fish is experiencing when the water drops.

Other studies have generally demonstrated a decrease in stranding rates during slower flow reductions, particularly with ramping rates less than 10 cm/hour (Saltveit et al. 2001, Hessevik 2002, Halleraker et al. 2003). When comparing stranding between slow (6 cm/hr) and fast (30 or 60 cm/hr) ramping rates, significantly less stranding were observed (Bradford et al. 1995). Critical down ramping rates at which ‘acceptable’ levels of stranding have been achieved, have been defined by a number of studies. However, other factors that influence stranding rates interact with this ‘critical rate’ so that any recommendations developed are only locally applicable and system specific in nature (Flodmark 2004). Flow ramping as the main fish stranding mitigation option may not be enough to significantly reduce the number of fish stranded, as was the result in a study on the Bridge River, BC (Higgins and Bradford 1996). In that study, flows were ramped down very slowly at a rate of 2 - 2.5 cm/hour and a large number of fish still became stranded. The hypothesis in the Bridge River example was that the majority of fish strandings occurred in pools that isolated from the mainstem river, which were likely less influenced by ramping rates than interstitial habitats (Higgins and Bradford 1996).

It was uncertain what mechanisms were responsible for the decreased probability of stranding with slower ramping rates reported in these studies. For example, fish may have had more time to find their way out of the varial zone habitat, or possible directional or behavioural cues that trigger an exit strategy may have more time in which to invoke a response during slow reductions. Fish using substrate concealment did not move until their backs and tails were exposed to air in flume experiments (Bradford et al. 1995).

6.3.3 Wetted History

Wetted history may influence stranding risk by affecting the behaviour of fish as well as their habitat use. Juvenile fish may be attached to a near shore site for various reasons, including: the increased predation risk associated with migration (Biro et al. 2003), the competitive disadvantage of leaving established territory (Ward et al. 2006) or increased food availability and/or cover in the varial zone when it has been wetted for a longer period (Cushman 1985, Heggenes et al. 1993). With longer periods of wetted history, juvenile fish may not have ‘learned’ to leave the varial zone during a flow reduction, potentially leading to an increase in the probability of stranding. A limitation on assessing this factor in the experimental studies is the fact that the stage data is only recorded at a relatively coarse level in the LDR, lower Columbia River and lower Kootenay River systems. In the LDR, sampling sites at which the stranding may occur may be experiencing a different wetted history than that recorded at the stage stations of the Water Survey of Canada stations at DRL or Birchbank as a result of their topography, river channel morphology, location in relation to the water’s edge, and cover variables.

Given the various demands for flow changes in regulated systems, it is often logistically difficult to test the ecological effects of a wide range of wetted histories. This is particularly problematic on the LDR where the experimental window is only about a month long (early September until early October), and the maximum value for wetted history is constrained by operations. The general trend observed over the four years of ramping studies conducted in the lower Columbia River and lower Kootenay River has been that there is a higher probability of stranding when the near shore varial zone has been inundated for a longer period of time, although this trend is not statistically significant. In the analysis of 10 years of stranding monitoring on the lower Columbia River and lower Kootenay River, the period of time the near shore area had been inundated was statistically significant at the 1, 50 and 200 fish definitions of a stranding event. When wetted history (less than 10 days verses 10 or more days) was assessed as a categorical variable of all surveyed flow reductions on the lower Columbia River and lower Kootenay River, there was significantly more stranding in the latter category (Irvine 2010).

The trend for increased stranding after longer periods with stable flows was seen in other hydropeaking research, though it has not often been a focus of that research. In flume experiments conducted in Norway on juvenile Brown Trout, increased stranding was observed after longer inundation periods (Halleraker et al. 2003).

6.3.4 Conditioning Reduction

The concept of the conditioning reduction was based on qualitative and quantitative observations in the lower Columbia River and lower Kootenay River that the number of fish in the near shore area, as well as the number fish stranded, both decreased through time during the first phase of the flow ramping assessments. The pattern of reduced stranding after multiple reductions has also been seen in mesocosm flume experiments (Irvine 1986, Halleraker et al. 2003). These results should be interpreted with caution, as the flume experiments tested the response the same fish during all reductions. Alternatively, during experiments in an open river system, there may be some fish some fish that are new to each conditioning reduction after having moved into the varial zone. The proportion of the population that experiences a second, third, or subsequent drawdown as a novel disturbance is very difficult to determine in an ecosystem level experiment. The occurrence of reduced stranding associated with subsequent drawdown events has not been a universal result. In the flume experiments of Hessevik (2002), individually tagged fish in an experimental stream channel did not demonstrate a shift in their learned escape response to dewatering after multiple experiments.

The frequent drying of the varial zone may produce a number of effects that could reduce fish stranding. One effect could be a reduction or elimination of cover and food in the form of algae and benthic invertebrates that would decrease fish densities in the varial zone (Cushman 1985). Another effect could be a learned, behavioural response of the fish so that over time, following a number of disturbances, fish develop escape behaviour.

This factor was not tested explicitly on the LDR system, though in the final year of the ramping experiments, the water levels were dropped rapidly and raised again for the fast ramping rate experiments, which constituted a conditioning reduction (Irvine and Hildebrand 2010). Significant mortality of Mountain Whitefish on gravel bars adjacent to the study sites was noted within the short time the substrate was dewatered between reductions (Irvine and Hildebrand 2010). In the literature, the time from dewatering until death is variable depending on species and life stage, but can be less than 30 minutes (Hessevik 2002). Based on these observations, as well as the limited efficacy of the approach to significantly reduce stranding rates in the lower Columbia River and lower Kootenay River, the examination of conditioning reductions as a management tool for mitigating fish stranding in the LDR was abandoned (Irvine et al. 2009, Irvine 2010).

Higgins and Bradford (1996) discussed an alternative idea to condition habitat that is dominated by pools (potholes) that entrap fish. The idea is drawn from data from work on the Skagit and Sauk rivers (Beck and Associates 1989). The pothole component of the stranding study indicated that the depth and velocity of water over the pothole where the pool would form was important in determining trapping risk for fish. In this study, the potholes with higher overflows were associated with lower numbers of fish stranding (Beck and Associates 1989). This factor was not an explicit part of the study design on the Skagit, but was assessed as part of the analysis.

6.3.5 Habitat

Habitat factors such as cover, substrate, and bank slope could affect the density of fish in the varial zone and/or the probability of stranding. Throughout the ramping experiments on the LDR, total cover was visually estimated and showed either a weak positive or neutral correlation with probability of stranding (Irvine and Hildebrand 2010). In the analysis of the LCR and LKR monitoring data, the river stage was a highly significant predictor of the risk of a stranding event and whether a high risk stranding site had been physically re-contoured to reduce available stranding habitat was also highly significant (Irvine 2010). The same experimental sites were used repeatedly in DDMMON-1 to minimize variability, so the testing of the effect of cover was compromised by clustered data and qualitative estimation methods. More recent analysis of the stranding surveys conducted as part of DDMMON-16 also incorporated bank slope into habitat stratification for study design and analysis (Section 5.5.1; Golder 2014 in prep.).

The results from the literature show a strong effect of cover, substrate and slope on the risk of fish stranding. A range of studies on fish stranding from the 1970s to 1990s determined that the stranding of fish was influenced by the habitat variables of riverbed profile, bank slope, and bottom substrate (Flodmark et al. 2004 and references therein). River bank profiles with low slopes often contain shallow, warm water that frequently has macrophytes and other cover that increase rearing suitability. Juvenile fish seek cover to avoid predation and to minimize energy expenditure (Heggenes et al. 1993, Vehanen et al. 2000). Consequently, these habitat factors that passively provide shelter will likely increase fish density that strengthens territoriality, which results in an increased stranding risk. Riverbank profiles with slopes greater than 4% have been associated with reduced stranding frequency (Bauersfeld 1978). Flume experiments done on 2% and 6% slopes with Coho and Rainbow Trout found that there was significantly less stranding on the 6% slope than the 2% slope (Bradford et al. 1995). River bank profiles with low slope may not provide sufficient directional current cues to trigger fish egress as water levels recede, thus potentially increasing the probability of stranding. In studying flow changes of magnitude 225 m^3/s, the average horizontal shift of water in the Colorado River was 6.5 m and 2.2 m in low and high angle shorelines, respectively (Korman and Campana 2009). The distance the juvenile fish have to travel to reach the new waterline is also simply greater in shallow gradient habitats which may also increase the rate of stranding.

Larger substrate size also has been associated with increased stranding. Salmonid parr develop holding positions in coarse substrate and may be unwilling to relinquish such preferred positions even when the water level is dropping (Monk 1989). It was noted in one study that Brown Trout greater than 67 mm in length were able to maintain position in rivers where there was natural substrate that provided low velocity refuges (Heggenes 1988). Substrate size also has an effect on evaluations of experimental success, since finding fish in larger substrates can be very difficult. This has been noted in the LDR, LCR and LKR ramping studies, and is supported by results of several studies on hydropeaking (e.g., Saltveit et al. 2001). In the BC Hydro flow ramping studies, difficulties with finding fish in coarse substrate required development of a correction factor to account for the fish that were never found (Irvine et al. 2009).

Another major habitat factor that determines stranding rates is the elevation of the river and the habitat available. In the Hanford Reach of the Columbia River, lower flow elevations presented the highest risk for stranding due to low angle river banks and high proportions of shallow pools that were isolated with down ramping (Hoffarth 2004). The addition of pools may influence stranding rates depending on the interactions with ramping rate and species. In flume experiments conducted on Coho and Rainbow Trout, when pools with cover were added to the habitat, fish escapement was higher at slower ramping rates (Bradford et al. 1995). In the lower Columbia River and lower Kootenay River systems, the number of pools increased with the lower river stage (Irvine 2010). An analogous survey and analysis has not yet been completed on the LDR. The availability of suitable habitat also affects the levels of stress the fish experience when flows are reduced; if sufficient preferred habitat still exists after the drop, the stress indicators showed no effect of flow reductions (Krimmer et al. 2011).

6.3.6 Season

Season could affect the number of fish stranded either by influencing fish density in the near shore area or the probability of stranding. Season is highly correlated with temperature, behavioural variables and fish size or life stage, all of which have their own effects on stranding. Operational and logistical constraints allowed for only autumn (early September to early October) ramping experiments on the LDR. Therefore, experimental data on the effect of season on stranding risk in this system is not available from DDMMON-1. However, the stranding surveys of DDMMON-16 occur throughout the year, and analysis showed a lower risk of stranding in the spring than in the fall (Hildebrand and Irvine 2012). Whether this was due to less fish in the system in spring or to a decreased risk of stranding in that season is currently uncertain (Hildebrand and Irvine 2012). In the LCR and LKR systems, the highest risk of stranding occurred in the mid-June-late July period, and was a highly significant predictor of the risk of stranding events occurring (Irvine 2010). A strong effect of season was also noted on the LCR and LKR ramping experiments. At the experimental sites examined, there was very little pool stranding in winter and very low interstitial stranding in the summer (Oussoren and Irvine 2007).

Different species show different susceptibilities to stranding by season (Beck and Associates 1989, Vehanen et al. 2000). Vehanen et al. (2000) found that season had an effect on fish displaced by high flows, as well as schooling behaviour. Competition for space in winter was found to be a key aggressive behaviour that occurs between 2 - 6°C (Cunjak et al. 1998).

Fish behaviour can vary dramatically by season, and has been linked to different requirements of various species and life stages throughout the year. For stream and river dwelling salmonids, the winter season can be a survival bottleneck due to adverse conditions (Heggenes et al. 1993). During winter, hydroelectric dam operations can impact fish survival by altering temperature regimes. This may result in warmer winter water temperatures, removal of protective ice cover, the creation of air space between the ice and the water surface that is utilized by predators, and necessitate the movement from established winter territories associated with large substrate or interstices when flows change (Cunjak et al. 1998, Huusko et al. 2007, Linnansaari et al. 2008). In some systems, winter has been the season associated with a far higher incidence of stranding as compared to summer and autumn (Saltveit 2001). Winter may require a juvenile fish to conserve energy and avoid ice entrapment (Linnansaari et al. 2008), while in summer juvenile fish behaviour is dominated by predator avoidance and rapid growth (Heggenes et al. 1993). The level of activity from dusk to dawn during winter can be high for juvenile salmonids, which has been surmised to link to lower incidence of stranding at night (Heggenes et al. 1993).

6.3.7 Water Temperature

Water temperature is highly correlated to season, and could either affect the density of fish in the near shore or their probability of stranding. The warmer water temperature of the near shore habitats provides improved growth during juvenile life stages (Korman and Campana 2009) and is one of the main factors that make the varial zone appealing to juvenile fish (Vehanen et al. 2000).

Water temperature was measured throughout the LDR flow ramping assessments, but has not been incorporated into the analyses to date. Complex river channels can have more variation in water temperature laterally across braids and sidechannels than is observed along the entire longitude of the river (Arscott et al. 2001). The LDR has a great deal of hydrological complexity in several sections, particularly the lower sections; therefore assessment of water temperature would have to occur on relevant spatial scales. One limitation in assessing this factor is understanding and measuring temperature at a scale that operates on fish behaviour. The current DDMMON-16 study program has mainly measured temperature as an ancillary variable at relatively coarse temporal and spatial scales.

In general, fish stranding has been found to be greater at lower temperatures because the fish are less active and are frequently hiding in the substrate (Heggenes et al. 1993; Bradford 1997; Saltveit et al. 2001). Studies using Brown Trout in flumes have shown a significant effect of water temperature on stranding, with higher stranding rates at 7°C than at 11°C (Halleraker et al. 2003). A study in side channels found more juvenile Chinook (*Oncorhynchus tshawytscha*) and Coho salmon were stranded at temperatures of 6°C than at 12°C (Bradford 1997). Previous analysis of the LCR stranding protocol suggested the opposite trend where fish stranding was greater at warmer temperatures (T. Oussoren, BC Hydro, pers. comm.). In addition to potentially being a determinant of stranding rate, temperature of both air and water can also alter the mortality rate associated with stranding. Direct mortality of juvenile Chinook salmon on the Hanford Reach occurred most often when air temperatures were greater than 24°C (Hoffarth 2004).

Water temperature is a final modulator of spawn timing in salmonid species (Wang et al. 2010). It has been shown to be associated with late spawning in Atlantic salmon where higher water temperatures as a result of regulation persisted into the spawning season (Saltveit et al. 1995). This can affect the incubation and emergence timing for species, which in turn can affect emergent fry with respect to temperatures, flow and food availability.

6.3.8 Fish Size and Species

Species and size can affect either the probability of stranding or the density of fish in the varial zone. These factors were documented for all fish that either escaped or became stranded during the early DDMMON-1 ramping experiments. However, a change in methodology implemented in order to obtain more natural behaviour from the fish led to the inability to collect fish species and size data (Golder 2007, 2008).

Fish size is related to behaviour and habitat needs, and has interactions with these factors as well as others. On the Hanford Reach of the Columbia River, juvenile Chinook salmon showed decreased susceptibility to interstitial stranding or entrapment (pool stranding) when they reached or exceeded a fork length of 50 mm. Only 0.7% of the stranded fish collected from 1999 to 2003 in this area were greater than 60 mm (Hoffarth 2004). This was attributed to differences in rearing habitat selection by different sized fish. In a flume experiment, age 0+ Brown Trout were twice as likely to become stranded as older and larger fry. The fish in the 0+ age class were also much more likely become stranded when they were in a mixed age group that included larger fry (Halleraker et al. 2003). This illustrates that size has competitive effects that may indirectly or directly affect stranding rates. Heggenes (1988) found that yearling fish were quite evenly distributed over the stream landscape, while larger juvenile brown trout concentrated in the deepest and slowest moving pools of the river. Active choice of microhabitats by young-of-year fish only occurs after they attain a certain size and swimming capacity (Pavlov et al. 2008).

Species can also lead to differential stranding rates. In a series of Scottish rivers where both Brown Trout and Atlantic salmon were found, the trout utilized pools more than the salmon at normal flows. However, these species were forced to share the pools when flow reductions occurred (Stradmeyer et al. 2008). The more aggressive Brown Trout established dominant feeding behaviours in the remaining pools and forced the salmon into hiding behaviours. In a stranding study on the Mississippi River, the variable of species explained more of the variation in the data than ramping rate (Adams 1999). The explanation for the high rate of stranding for certain species over others was linked to behavioural factors. Species that usually occupied backwater areas of the river passively drifted with the current as the water levels receded and had a higher probability of escape than fish that showed positive rheotaxis (Adams 1999).

Increases in flow and velocity are strongly correlated to increases in downstream drift of newly emergent larvae and more mature fry. In downstream sections of large rivers upstream of the sea or large lakes, this can result in a measurable loss of fry (Irvine 1986, Saltveit et al. 1995). This is not necessarily just an issue associated with regulated systems since flow increases in spring occur naturally, but it may be exacerbated depending on the system and flow regime. Although the fry drifting from further upstream sections of the river may not be lost, in some cases an increased density in fry seen in downstream sections is not documented (Saltveit et al. 1995). In some studied systems, there is a decrease in density of the resident fish species in the areas most affected by the sudden changes in discharge associated with flow regulation, immediately below the dam (Ugedal et al. 2008).

6.3.9 Behavioural Factors

Behavioural factors could affect either the density of fish in the varial zone and/or the probability of stranding. Analyses of the LDR ramping experimental data showed greater variability than expected in the dataset, which could indicate that there may be non-independent behaviours (such as schooling) in the data, or that the analysis may not have captured a major source of the variation (Collett 2003). Inter-specific competition and behavioural dominance are two factors that may affect the variation in the data. Limitations of the experimental program for assessing this variable included: batch marking of fish did not allow the observation of particular individual's behaviour, short intervals of observation may have been too coarse in a rapid reduction context to accurately assess behaviour in relation to ramping rate, and the experimental design was never focused on behaviour in particular. It is not possible for the DDMMON-16 surveys to document behavioural factors relating to stranding.

In observations of individually marked fish during the ramping experiments, it was found that individuals differed significantly in their tendency to move in response to flow fluctuations. This may be due to dominance status of individual fish and an associated reluctance to leave prime territories (Krimmer et al. 2011). Studies on the effects of competition between Atlantic salmon and Brown Trout in pools undergoing dewatering demonstrated clear dominance behaviour of the Brown Trout with substantial effects on the fish escaping from the pool (Stradmeyer et al. 2008). The non-dominant individuals in a pool were forced into a more stationary and cryptic behaviour pattern and because dewatering increased local population densities, dominance status became even more important (Stradmeyer et al. 2008). This may have direct implications for the DDMMON-16 stranding surveys conducted on the LDR as the natural fish density has often emerged as a strong explanatory variable and different behaviours are likely elicited at different density levels.

Behavioural factors have a strong seasonal component with aggressive interactions reduced in winter (Heggenes et al. 1993), perhaps due to energy conservation or the necessity of sharing limited habitat. Observation of the behavioural changes that occur with flow fluctuations demonstrated that the response is short lived and that fish behaviour returns to a baseline state quickly after the fluctuations cease (Krimmer et al. 2011).

6.3.10 Meteorological Factors: Moon Phase and Cloud Cover

Meteorological factors such as moon phase and cloud cover could affect either the density in the near shore zone or the probability of stranding. These factors have not been measured or analysed for the LDR ramping studies to date and are generally not recorded in the stranding surveys occurring as part of DDMMON-16. These factors are not under direct operational control; however, there is the possibility that if a factor like moon phase was shown to be a significant explanatory variable of the probability of stranding risk, operational changes could incorporate such a meteorological variable into the planning process at some periods in the year.

The results from studies that have examined the effect moon phase are inconclusive as to whether there is an effect on the fish density in the near shore area. A study on juvenile Atlantic salmon parr found no statistically significant effect of moon phase on the number of parr observed (Imre and Boisclair 2005). These authors initially thought that Atlantic salmon parr would exhibit more cryptic behaviour on nights with more ambient light from the moon, but this was not the result found. They hypothesized that the feeding efficiency gains from the increased ambient light associated with the full moon enticed the juvenile fish into greater activity in the near shore.

Cloud cover has been inversely related to the density of juvenile salmonids in the near shore area during the daytime (Girard et al. 2003). This was suspected to be a predator avoidance strategy, as reduced glare on the water surface during period of extensive cloud cover would for easier location of prey (Girard et al. 2003).

6.3.11 Food Availability

Food availability would most likely influence the density of fish in the near shore zone. Thus far, food availability has not been assessed on any of the flow ramping experiments or stranding surveys conducted for BC Hydro on either the LDR or lower Columbia River and lower Kootenay River projects. Food availability is likely strongly correlated to wetted history given patterns of colonization of benthos in areas with longer inundation periods (Cushman 1985, Hoffarth 2004).

Benthic organisms have been reported to show reduced diversity, density and mean individual weight in systems where flows vary rapidly (Cushman 1985). Resilient macro invertebrate families such as Chironomidae (midges) and certain Trichoptera (caddis flies) tend to dominate the invertebrate community when water levels fluctuate widely and frequently as a result of dam operations (Stark 2001). One of the artificial substrates used in the experiments of Stark (2001) that was continually inundated held four times more invertebrate biomass than a substrate that had been dewatered for anywhere from 1 - 24 hours. Smaller and less mobile early instars of invertebrates suffer increased mortality as a result of dewatering, while larger, more mature invertebrates can crawl or drift to safety with changes in flow (Stark 2001). More recent studies employing stable isotope technology have shown that food web structure is shortened and simplified by high ramping rates and high levels of disturbance resulting from flow fluctuations (Marty 2008, Marty et al. 2008). In addition, fish in a stable flow environment not only have more food available, but also exhibit higher food intake and growth than those in fluctuating environments. Flodmark et al. (2004) found that food intake was decreased by 50 - 55% by fish in fluctuating flow environments.

6.4 Literature Review Conclusion

Much of the research on the effects of flow reductions from hydro-electric dams on fish stranding has been completed while focusing on ecosystems at the medium level (e.g., lower Duncan River (as a part the entire Duncan drainage) [Irvine 1986, Linnansaari et al. 2008]). Therefore, large scale, long term monitoring of fish stranding like the data collected through stranding surveys are significant for understanding patterns of stranding responses in large river ecosystems (Irvine and Hildebrand 2010, Irvine 2010). The research in the literature, as well as the BC Hydro WLR studies, have demonstrated that there is substantial variability in stranding rates and that it is difficult to attribute the variability to single or even multiple factors (e.g., Saltveit 2001, Berland 2004, Irvine et al. 2009). However, the patterns observed are beginning to indicate certain conditions that can increase the risk of stranding. In the LDR studies to date, stranding rates trend higher at night, during faster ramping rates, after longer periods of wetted history, with lower fish densities, when fewer drops per hour are conducted to attain the reduction, and with increased cover availability. However, none of these variables were statistically significant predictors of stranding risk (Irvine and Hildebrand 2010). The analysis of the 1999 - 2009 stranding survey data from the lower Columbia River and lower Kootenay River showed that the worst case scenario for a stranding event was a large magnitude reduction carried out in the afternoon in midsummer, at low water levels when the near shore had been inundated for a long period (Irvine 2010).

The research on the effects of flow reductions on stress levels have shown that within 24 hours after a reduction event fish cortisol levels were back in normal ranges (Arnekleiv et al. 2004). Reduced growth as a result of flow fluctuations has been shown (Flodmark et al. 2002, Korman and Campana 2009), but the research is showing that the indirect effects are less severe than originally thought (Scruton et al. 2008). The main detectable effect of fish stranding is the direct effect of mortality through stranding. In some systems, the modification of operational flow regimes to reduce stranding risk has shown positive population level effects. In long term studies on the Skagit river, the density of Pink (*Oncorhynchus gorbuscha*) , Chum (*Oncorhynchus keta*) and Chinook salmon all increased in the decades following the alteration of the flow regime to moderate the operational effects on the fish fauna (Connor and Pflug 2004). The operations were designed to minimize redd dewatering, reduce the number of down ramping events annually, and were not conducted when diel risk was determined to be highest. The difference between the spawning and incubation flows was also minimized to reduce the area of the channel potentially subject to dewatering (Connor and Pflug 2004). The increases in density were greatest closest to the hydroelectric projects (Connor and Pflug 2004), which mirrors the results of other work done on the longitudinal effects of hydroelectric projects on fish densities (Ugedal et al. 2008). The LDR, lower Columbia River and lower Kootenay River do not yet have the 30+ year datasets that were in place to allow the detection of the population level changes in the Skagit, but the data are accruing and adaptively altering the flow regime to improve stranding rates is ongoing.

7.0 DISCUSSION

The species of interest that are discussed below are Rainbow Trout, Mountain Whitefish and Kokanee. The DDMMON-16 program is currently designed to provide both fall abundance and stranding estimates for Rainbow Trout and Mountain Whitefish. Based on the results of the DDMMON-4 and DDMMON-16, during specific DDM operations Kokanee eggs and emerging larvae may also be at risk of stranding (Sections 5.4 and 5.5).

7.1 Habitat

7.1.1 Habitat Use and Suitability

7.1.1.1 Rainbow Trout Spawning

As stated by the DDMMON-2 program, Rainbow Trout spawning occurs almost exclusively in the tailout area of DDM (Appendix B, Figures 3.7 and 3.8; Thorley et al. 2012). HUCs developed for Rainbow Trout spawning in the tailout area indicated that freshly encountered redds were at depths between 0.25 and 0.75 m and velocities between 0.3 and 0.8 m/s.

As Rainbow Trout spawning occurs after the winter DDM flow reductions, the tailout area only experiences flow fluctuations as DDM adjusts discharge to reach and maintain seasonal targets. These operations pose a high risk of stranding redds at the shallow end of the preferred depth scale. In past study years, redds became dewatered as DDM discharge was reduced to below 20 m^3/s for short durations (Thorley et al. 2012). The level of egg mortality related to these dewaterings was unknown.

In the spring of 2011, the DDMMON-2 program documented that when discharge from DDM was reduced to approximately 27 m^3/s, redds in deeper water were revealed (Thorley et al. 2012). This tentatively suggested that when the discharge in the Lardeau River is low (e.g., around 25 m^3/s), DDM discharge levels of approximately 27 m^3/s resulted in increased amounts of suitable spawning habitat relative to higher dam discharge levels. Therefore, it may be possible to increase spawning activity, as well as reduce egg stranding in the tailout area by holding discharge levels at low levels during the pre-freshet period (Thorley et al. 2012).

7.1.1.2 Mountain Whitefish Spawning

The HUCs developed by the DDMMON-2 program estimated that Mountain Whitefish spawn in 0 - 2 m of water with peak spawning at 0.9m. Spawning velocities were estimated between 0.1 - 1.48m/s with peak at 0.68 m/s. The majority of spawning occurred over cobble (67%) and large gravel (32%) with 20% of the spawning proximal to large woody debris (Thorley et al. 2012). These depth, velocity and substrate use data may not indicate preference but rather what is available to Mountain Whitefish at the time of spawning. As these conditions occur throughout the entire LDR, the current spatial extent of Mountain Whitefish spawning is unknown. Conditions in the LDR restricted spawning related assessment to the upstream 4 kms of river (Thorley et al. 2012).

Under the current DDM flow regime, discharge increases substantially in late December, and it was thought that the increases could negatively affect late Mountain Whitefish spawners (Thorley et al. 2011). As the majority of the spawning is complete by mid-December, the increase in flow in the LDR at the end of December most likely affects eggs rather than spawners. Fluctuations in hydraulic characteristics at egg incubation habitats may result in the re-suspension of deposited eggs and subsequent downstream drift, which would increase the risks of predation and mechanical damage (Golder 2014, in prep).

Dewatering of eggs is more likely later in the incubation period when DDM flows drop to the annual minimum (Thorley et al. 2012). Limiting the extent of flow reductions following peak spawning in November/December, either by ensuring the flows are lower during spawning (as is the current case), or reducing deviation from spawning flows over the incubation period, should be considered in future WUP flow reviews. Current winter flow reductions can result in up to an 80 cm reduction of river stage at the DRL. Any eggs that are still incubating in shallow portions of the river when these reductions occur may be dewatered unless they drifted to deeper parts of the river after release. Dewatering of significant numbers of Mountain Whitefish eggs is considered unlikely as flow levels are typically similar during the spawning and incubation periods, but could occur if eggs are redistributed into shallower habitat during the higher winter flows (Thorley et al. 2012).

7.1.1.3 Kokanee Spawning

Kokanee spawn in the upper 9.6 km of the LDR below DDM, mainly from Km 0.8 to 7.1 (Appendix B, Kokanee Spawning Figures 1 to 6) in the mainstem and in all side channels that remain wetted during the spawning season (1.1R; 2.7L; 3.5R; 4.1R; 4.4R; 6.9R; 7.6R; 8.2L; and, 8.8L: AMEC 2013). Observations suggest that early spawning begins in side channel habitats, but by the peak of spawning kokanee are in both side channel and mainstem habitats in approximately equal proportions. Also, it is likely that kokanee spawn within the upper river because these areas have more suitable spawning gravels compared to lower regions of the river, which are largely comprised of fines due to the influence of Kootenay Lake (NHC 2010).

The dewatering of redds resulting from the fall DDM flow reductions to reach Kokanee protection flows has been documented in several study years (AMEC 2012 and 2013). These baseline operations have resulted in annual kokanee egg dewatering of less than 1% of the total overall PED for the Duncan system. Examining egg dewatering strictly in the LDR, then approximately 4-16% of the PED each year has potentially been dewatered during these operations, all within side channel habitats. Dewatering of mainstem redds has been zero or minimal in comparison to dewatering in side channels of the LDR in previous years (AMEC 2012). Operations during kokanee development through to emergence stay over the 75 m^3/s minimum flow target. Therefore, egg depositional areas should have adequate flow coverage for development and hatching kokanee in the LDR should not be stranded during this period (AMEC 2012). Operational strategies to minimize sidechannel stranding have been reviewed and need further assessment before recommendations can be made to further reduce kokanee spawning impacts.

7.1.1.4 Juvenile Rearing

Preliminary data analysis conducted by the DDMMON-2 program indicated that, in general, juvenile Rainbow Trout and Mountain Whitefish do not exhibit a significant preference for main or side channel habitat (Thorley et al. 2012). The depth HUCs indicated that rainbow trout fry used shallower water than mountain whitefish fry, which in turn preferred shallower water than the parr of either species. The velocity HUCs suggested that juveniles of these species also almost exclusively utilize water slower than 0.5 m/s.

Utilizing the updated TELEMAC2D models, the DDMMON-3 program determined that the HSI for Rainbow Trout fry exhibited an inverse relationship between discharge and WUA, with values that were lower than predicted with the earlier RIVER 2D version of the model. This was due to the tighter habitat use curves for depth and velocity estimated for DDMMON-2 (NHC 2013). When considering depth and velocity, the WUA for Kokanee spawners increased as flows increased with a maximum value at 275 m^3/s. When substrate was included in the WUA calculation, the maximum WUA value occurred at approximately 250 m^3/s due to the inundation of smaller substrates (NHC 2013).

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 and Columbia River Treaty objectives. In addition there are several smaller reductions that occur throughout the year to effectively manage water resources and power generation at other facilities (Golder 2014 in prep.). The fall DDM flow reductions to reach Kokanee protection flows pose the greatest stranding risk to juvenile Rainbow Trout (Golder 2014, in prep.). The median number of juveniles per pool for the spring season (January – June) was estimated to be 1.96 (credibility interval of 1.33 - 2.87) fish/pool. In contrast, the median number of Rainbow Trout juveniles stranded per pool in the fall (July to December) was estimated at 6.61 (credibility interval of 4.55 – 9.54). The season effect on stranding numbers was found to be significant ($p < 0.05$), with median fall stranding estimates over three times higher than those for winter/spring. This is reinforced by the findings of the DDMMON-2 program, which documented lower abundances of juveniles during the winter season (Thorley et al. 2012).

BC Hydro has identified an increased risk of stranding to newly emerged Mountain Whitefish and Kokanee in the spring prior to the onset of freshet (Thorley et al. 2012). As the DDMMON programs related to these species primarily focused on spawning related issues, little is known about the larval life stage of these species in the LDR.

7.1.2 Slope

The categories of low slope (≤4% gradient) and high slope (>4% gradient) used in the DDMMON-16 program to classify potential stranding habitat were based on values in the literature from previous stranding work (Bauersfeld 1978; Flodmark 2004). Contrary to the finding of previous study years (DDMMON-16 Year 4; Hildebrand and Irvine 2012), the results from the current dataset (Year 6) did not show a significant relationship between slope and interstitial stranding rates, and suggested that slope did not have an effect on the formation of isolated pools within the study area. Also, a relationship between slope and the number of fish stranded in isolated pools was not identified (Golder 2014 in prep.). While the data showed no trends for the relationship, this may be due to data scarcity, and should be re-evaluated as more data are collected. The dichotomous high/low classification of slope habitat may be too coarse to determine the effects of slope on both pool and interstitial stranding. Reclassifying the slope categories may assist in ascertaining its effect on fish stranding, and will be examined in Year 7 of the DDMMON-16 program.

7.1.3 Index vs Non-Index Sites

For the DDMMON-16 program, 50 potential fish stranding sites along the LDR were identified based on previous studies (Hildebrand and Irvine 2012). These stranding sites included 11 index stranding assessment sites and 39 non-index sites (Appendix B, Stranding Assessment Site Figures 1 to 7). The remaining habitats outside of the identified sites consist of steep banks with extreme gradient that would not be considered to strand fish. Originally, the index sites were not selected to be representative of the entire LDR, but to focus salvage efforts on sites believed to have the highest amounts of stranding based on amount dewatered area and suitable habitat. Based on the findings of previous study years (Hildebrand and Irvine 2012), index sites tended to be of lower gradient than non-index sites, and therefore strand more fish because more area dewaters at these sites during flow reductions. Therefore, index sites had a bias toward higher stranding rates and are not representative of stranding over the entire LDR. The hypothesis related to index sites was not specifically examined in Year 6 of the DDMMON-16 program. It will be re-examined in the Year 7 in-depth interpretive report with all available project data to determine if the complete dataset supports the rejection of hypothesis.

7.2 Recruitment Bio-Standards

7.2.1 Egg Survival Estimates

7.2.1.1 Rainbow Trout

A preliminary assessment of egg survival in the tailout area was conducted in 2011 (Thorley and Baxter 2011). Survival of the eggs during the assessment was 0%, as the eggs were covered in fine sediment when the incubators were examined. Another assessment was planned for the 2012 field season. However, exceptionally high water levels prevented the 2012 assessment from taking place (Thorley et al. 2012).

During typical operations, discharge from DDM is reduced to the minimum mean daily operating requirement of 3 m^3/s at the onset of freshet in the Lardeau River. This reduced discharge decreases water velocity in the tailout area, which reduces dissolved oxygen, increases sediment deposition and induces episodes of high water temperatures (Thorley et al. 2012). These factors can lead to an increase in egg mortality rates. The results of the preliminary assessment of egg survival suggest that embryo mortality in the tailout can be extremely high although this conclusion should be interpreted with caution (Thorley and Baxter 2011).

7.2.1.2 Mountain Whitefish

Predicted egg mortality levels of Mountain Whitefish in the LDR could be high in the early portion of the spawning period (ranging from 50% to 100% egg mortality; Thorley et al. 2012). Mortality rates most likely decrease during the estimated peak spawning period when water temperatures are ideal for optimal egg survival. Additionally, the advent of winter high flows in late December (which typically occurs after peak spawning) may cause eggs to re-enter the water column after initial deposition, and increase rates of mortality (Thorley et al 2012).

Coregonids, including mountain whitefish, show cumulative increases in egg mortality and embryo defects with increases in water temperature outside of a narrow range (Rajagopal 1979, Brinkman and Vieira 2009). Mountain Whitefish may be especially vulnerable to water temperature effects on egg viability, fertilization, and successful incubation since they are broadcast spawners and the eggs are not buffered by groundwater temperatures as much as a species that constructs redds (Thorley et al 2012).

7.2.1.3 Kokanee

Mean egg-to-fry survival from buried incubation capsules was 23% (2009 spawning season) and ranged between 0% and 50% in both mainstem and side channel sites (AMEC 2012). In comparison, kokanee egg-to-fry survival at the MCSC ranged from 6% to 64% from 1968 to 2011 (Ministry of Environment unpublished). Survival estimates in the spawning channel are based on fry production from the study year divided by channel egg deposition from the previous year, which may not be directly comparable to the present study results. The recovery of the kokanee population in this system would require an average egg-to-fry survival rate of 20% (AMEC 2012).

7.2.2 Juvenile/Overwintering Survival Estimates

Juvenile Rainbow Trout utilize the LDR year round. Abundance estimates indicate that first winter Rainbow Trout survival is approximately 25% (Thorley et al 2012). Juvenile Mountain Whitefish numbers in the LDR are high during the fall and appear to decline precipitously (90-99%) during the winter. The possibility that the low Mountain Whitefish counts are due to extremely low observer efficiencies cannot currently be excluded (Thorley et al 2012).

In order to reliably predict changes in fish productivity from changes in operations, the periods during the populations' life-cycle when habitat is limiting must also be identified as well as other flow related factors such as temperature, food availability and channel morphology (Annear et al. 2004). Also, determining how estimates of mortality due to stranding affect an overall fish population is difficult (Golder 2014 in prep.). Several factors adversely affect fish populations, including: predation, outmigration, food availability, availability of suitable rearing habitats, winter mortality, as well as inter- and intra-specific competition. Whether stranding events kill fish that would have died because of these factors, or kill fish which would survive these factors is unknown (Hildebrand and Irvine 2012).

Survival estimates for Kokanee juveniles were not obtained as part of the DDMMON programs.

7.3 Stranding Stock Abundance Estimation

As the DDMMON-2 program has been concluded, stranding stock estimates for some of the species of interest (Rainbow Trout and Mountain Whitefish) will solely be provided by the DDMMON-16 program. The key features and assumptions of the Abundance Estimation study component are as follows:

1) Based on the current limitations of the DDMMON-16 budget, only fall juvenile abundance estimates will be provided in future study years. This will provide the most comparable data to the period of the year when the highest stranding rates are observed, being the fall drawdown to reach Kokanee Protection Flows.

2) The current methodology for the abundance estimates builds on the DDMMON-2 program, and has the following assumptions:

    a. There is no significant difference in the abundance of the target species between mainstem and side channel habitats,
    b. Observer efficiency, derived from previous work on Rainbow Trout and Mountain Whitefish in the LDR (Thorley et al. 2012), will continue to be used to estimate total fish abundance at each site from the number of observed fish.

An ongoing management program (run jointly by the Fish and Wildlife Compensation Program and the Habitat Conservation Trust Fund) provides spring Rainbow Trout abundance estimates in both the Lardeau and lower Duncan rivers. The snorkel surveys from that program that are conducted to obtain abundance estimates occur in March of every calendar year (Andrusak 2010, 2013a and 2013b). As the timing of the abundance estimates provided occurs after the winter/spring fish stranding assessments, it is not possible to utilize those estimates in determining if spring stranding rates impact population levels of Rainbow Trout. The yearly results of this program should continue to be examined to refine the abundance estimation methodology of the DDMMON-16 program and to monitor the comparability of the spring abundance estimates it provides.

The current abundance estimation of 16,330 (credibility interval of 9,985 – 22,874) should be interpreted with caution as the densities documented in the DDMMON-16 study in 2013 were substantially lower than in previous studies (Golder 2014 in prep.). Further effort and analysis are required in Year 7 of the program to confirm the validity of the current abundance estimates. Also, to address the DDMMON-16 study hypotheses more confidently, it is critical that the uncertainties associated with the abundance and interstitial stranding estimates continue to be refined. With the uncertainties in the current dataset, it is not possible to determine the level of impact fish stranding has on the population of the species of interest. The further refinement of abundance estimation and interstitial stranding rates may reverse this finding. Recommendations have been presented in the most recent (Year 6) DDMMON-16 report to reduce these uncertainties.

7.4 Fish Stranding Management Tool

The LDR Fish Stranding Database and Management Tool is a data management and planning tool that archives historic flow reduction assessment data and the extent of pool and interstitial fish stranding within the LDR to help anticipate potential impacts of proposed flow changes. The data from each stranding survey are entered into a MS Access database. The completed fish salvage field data sheets are entered into the database quarterly. BC Hydro will maintain (or have access to) completed field data sheets and the most current version of the database. The planning component of the tool maintains information on spawning/incubation timing, as well as information from the hydraulic model to assist in estimating the amount of habitat dewatered and when side channels become dewatered. All information will be kept current as new information becomes available.

To conduct a query of the LDR Fish Stranding Database and Management Tool to find relevant historic stranding assessment results for a planned flow reduction, these steps should be followed:

1) Complete the Query Parameters form that appears upon opening the database by entering the following data into its corresponding box:

   a. The date of the proposed reduction;
   b. The current discharge at the DRL (m^3/s);
   c. The resulting discharge at the DRL after the flow reduction (m^3/s); and,
   d. The current LDR water temperature. The water temperature can be found on the BC Hydro Regional Hydromet Data website at .

2) Press the “Generate a Stranding Report” button at the bottom of the form.

3) When the query output is generated, it can be saved as a PDF and then distributed.

Several summary tables appear on the first page of the query output that provide an overview of the habitat dewatered, fish life history periodicity at the time of the proposed flow reduction, side channel connection to the mainstem DRL, and previous fish stranding observations during similar flow reductions. After the summary tables, all relevant fish stranding data is presented for each of the 50 identified stranding sties.

7.5 ASPD Refinement and Overall Recommendations

Based on the current state of knowledge, the flow reduction measures implemented under the ASPD are reducing fish stranding. The ASPD document was recently updated in January 2013 (Westcott et al 2013). During that update process, the findings presented in this document were examined and incorporated, although the update was largely driven by the results from the DDMMON-1 and DDMMON-16 programs. Therefore, changes to the ASPD document (in regards to the time of day, magnitude and the ramping rate of flow reductions) are not required at this time. It is recommended that the contact information in contact information appendix of the ASPD document be updated on a regular basis to follow BC Hydro staffing changes.

Based on the data and findings presented above, overall recommendations for future LDR WUP study years are as follows:

1) Conduct a workshop to identify, discuss and examine the feasibility of alternative flow scenarios to increase overall fish productivity and reduce stranding related mortality for the species and life history stages of interest. This workshop should include all of the study leads for the DDMMON programs, as well as BC Hydro Dam operators, WLR Managers and Specialists.

2) Institute a study, independent of the DDMMON programs that will conduct in-depth statistical analysis of relevant data across the programs. BC Hydro is currently conducting a similar program with Mountain Whitefish on the Lower Columbia River. Some of the data gaps identified by this program could be resolved if the data from these WUP programs was analyzed together.

3) There is nothing written in the ASPD about season, wetted history or about managing larger magnitude reductions more conservatively than small magnitude (< 50 m^3/s) drops. As more data accrues, it is recommended that the risk by season be assessed and operations be examined for the possibility to minimize stranding risk seasonally. Increased wetted history has been shown to be linked to increased stranding risk (Irvine 2010). To address this, an analysis of flow stability could be conducted for the LDR. The ASPD could then include differing protocols for operators at DDM to follow, including a conservative approach to be used whenever possible after long periods of stable flows or during any other flow reductions that pose a high risk of fish stranding.

4) It is currently assumed in the ASPD that species respond similarly to stranding reduction approaches, but the majority of the data and analyses have been focused on Rainbow Trout. It is recommended that the existing juvenile Mountain Whitefish data (Andrusak 2013a) be analysed to better understand if there are population level impacts on this species so that the ASPD is tailored to optimize stranding reduction for all species of interest.

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9.0 CLOSURE

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POISSON CONSULTING LTD.

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Robyn Irvine
Statistical Ecologist

ORIGINAL SIGNED

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Associate, Sr. Fisheries Scientist/Limnologist

BH/RI/DS/cmc

n:\active_2009\1492 biosciences\09-1492-5010 duncan river protocol review\reports\year 5 report jan 2013\final report\files for submission\sept 2015 final\0914925010-001-r-rev2-ddmmon-15_9sep2015.docx

APPENDIX A

DDMMON-16: Lower Duncan River Fish Stranding Impact Monitoring Sampling Protocol and Data Forms

Prior to Field work:

1. Upon notification of a required fish stranding assessment response to a flow reduction at Duncan Dam from BC Hydro, create a new reduction event folder in: N:\Active_2010\1492 Biology\12-1492-0117 Lower Duncan River Fish Stranding Impact Monitoring Year 5\Reductions. Event numbers should be sequential.
2. All relevant correspondence for the reduction should be placed in reduction event folder.
3. Create fish stranding query from the Lower Duncan River Fish stranding Database using discharge information provided by BC Hydro. Determine current river temperature at: http://www.bchydro.com/energy-in-bc/our_system/transmission_reservoir_data/hydrometric_data/columbia.html. Distribute query to BC Hydro contract authority.
4. Organize flow reduction schedule, communication protocol and accommodations with contract authority: James Baxter: 250-365-4593.

Field Sampling:

Field sampling during an index fish stranding assessment is to be conducted according to the Duncan Fish Stranding Protocol (2011) and in a manner consistent with previous fish stranding assessments, with the following changes to methodology to ensure consistency with the WLR study requirements:

1) If feasible, travel to Meadow Creek or Kaslo the day before the scheduled reduction and stay overnight in staff house (arrangements to be made with Len Wiens 250-366-4257) or at the Kaslo Motel (Front Desk 250-353-2431). It is our experience that with the field crew staying overnight in Meadow Creek or Kaslo, reductions can be initiated earlier the following day, which allows field crews more time to conduct assessments.

2) In total, 10 previously identified stranding sites will be selected at random for assessment (see Appendix A, Site Maps and Sample Forms, for site maps). This will be accomplished prior to the field work in the random site selection spreadsheet 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 will be proportionate to the area dewatered in each stratum as a result of the assessed DDM flow reduction. Based on our experience with stranding sampling and subsequent data analysis, sampling 10 sites provides a good balance between field effort and statistical power.

3) Field crew will be onsite at the BC Hydro boat launch downstream of DDM and ready to start field work as the last flow reduction is made. Crews will then sample selected sites in order from upstream to down. It is Golder's experience that this approach ensures that the field crew does not move ahead of the receding water levels.

4) Once sampling commences, isolated pools as a result of the DDM flow reduction will be enumerated and their surface area estimated as they are identified. The field crews will then single pass electrofish 50% of the pools at each site, up to a maximum of three (it is Golder's experience that this level of replication and sampling effort is achievable and statistically robust). The pools to be sampled will be selected at random. Field crew are also equipped with small numbered flags to allow identification of the pools if crews have to come back to them.

5) Each field crew will have a GPS and record waypoints of every pool and interstitial grid sampled.

6) For each pool electrofished, associated cover types (and percentages within the pool) from the following list in the pool will be recorded on the Stranding Habitat and Fish Record data sheet:

   i. Small woody debris (woody debris with diameter of <10 cm);
   ii. Large woody debris (woody debris with diameter of >10cm);
   iii. Aquatic vegetation;
   iv. Submerged terrestrial vegetation;
   v. Organic debris (leaves, bark etc.);
   vi. Overhanging vegetation;
   vii. Cut bank;
   viii. Shallow pool;
   ix. Deep pool; and,
   x. Other (metal, garbage, etc.).

7) During sampling, if possible the habitat association of each fish will also be recorded on the fish record data sheet.

8) To determine the observer (capture) efficiency distribution during stranding assessments, multi-pass electrofishing will occur at a subset of electrofished pools, selected at random. As observer efficiency will likely differ with the amount of cover present in each pool, the pools will be divided into two categories:

   xi. Zero to Low Complexity; and,
   xii. Moderate to High Complexity.

   Zero to Low Complexity pools have 0-10% of the total area of the pool occupied by cover, with sand or small gravel substrate that would not be large enough to hide juvenile fish. 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 have >10% total cover and 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 due to woody debris or other cover types.

   Each field crew will conduct double pass removal in one Zero to Low Complexity pool and two Moderate to High Complexity pools per assessment. The effort for each subsequent pass will be as consistent as possible with the first pass. The fish salvaged and effort for each pass will be recorded separately.

9) Dewatered habitat at each site will be assessed by conducting a minimum of twenty randomly placed interstitial grids ($0.5m^2$) if logistically feasible within dewatered zone and allotted time commitments for this task. The substrate and all cover will be removed from each grid and the stranded fish enumerated. When selecting a location for each grid, the field crew will use a random number table to determine the location of each of the grids within the dewatered zone.

10) Pools that have completely dewatered as a result of the flow reduction will be assessed visually for stranded fish and recorded separately from interstitial and wetted pools. It is our experience that the field crews will be unable to accurately determine the size of the pool and what habitat it encompassed when it isolated from the mainstem and dewatered; therefore the area, cover types, pool complexity and substrate of the dried pools will not be recorded.

11) Field crews will record the length of each fish enumerated (in wetted or dried pools or on dewatered substrate). Based on our previous experience, fish numbers may be high and time may not allow measuring all fish at a site. In such situations, a subsample of all salvaged fish would be measured (estimate number of fish by species in pool, length measurements will be taken of at least 30 but no more than 50 of each species).

12) To be consistent with past studies (fish stranding assessments and ramping experiments), if time allows, the dominant and subdominant substrate in each stranding habitat type (interstitial and dried and wetted pools) will be recorded using the Modified Wentworth Scale.

13) Field crews will ensure that all relevant sections of data sheets are completed (see Appendix A, Site Maps and Sample Forms, for sample data sheets).

Post Sampling:

1. Once the crew returns to the office, all relevant equipment with data should be downloaded (i.e. camera, GPS) and put in the corresponding reduction folder.
2. The crew leader will visit the BC Hydro Hydromet website and save data for DCN, DRL, QBY and DBC stations as text files in reduction folder. http://www.bchydro.com/energy-in-bc/our_system/transmission_reservoir_data/hydrometric_data/columbia.html
3. All data sheets should be placed in the 10-1492-0110 unentered data folder in top drawer of right fireproof filing cabinet.

Equipment List

The following equipment should be prepared for field work:

- Truck with proper hitch
- G3 boat and appropriate safety gear
- Ice auger (if winter survey)
- Aquaview (if winter survey)
- Electroshocker
- 2 electroshocker batteries (fully charged)
- 2 or 3 (if available) pairs of linesmen gloves
- 2 interstitial grids (0.5m^2)
- Beach seine
- Long handled net
- 2 dip nets
- 1 bucket
- Fish sample kit
- Level 1 First Aid kit
- Bear kit
- Clipboard with Duncan River Fish Stranding Survey Form and Duncan Stranding Habitat and Fish Record datasheets on waterproof paper, scientific fish collection permit, HASP, BC Hydro South Interior Radio System Info sheet, WPP Local Component for Duncan Dam Info sheet, pencils, Fish ID key, Modified Wentworth Substrate Key, Duncan Stranding Protocol (2004), 10-1492-0110 Specific Work Instructions (this document)
- Fish measuring board
- Satellite phone
- VHF Radio with BC Hydro frequencies (Provided by BC Hydro)
- Laser Rangefinder

- Digital Camera
- GPS (WAAS Enabled)
- Thermometer
- Laminated Maps for identification of fish stranding sites (Duncan River Orthophotos)

Personal Gear

- Lifejacket
- Hat
- Polarized sunglasses
- Rain gear
- Waders
- Wading belt
- Dry bag
- Personal 1st Aid kit
- Snowshoes (if winter survey)

Random Number Table

05 38 04 41 45 20 08 06 00 18 15 37 08 32 37 34 45 48 12 33 34 43 02 48 26 41 09 28 47 42 31 11 20 21 19 24 09 02 09 39 01 00 16 20 22 14 39 03 46 31 13 15 35 12 17 31 41 10 23 11 48 24 46 45 21 03 20 07 11 36 11 22 16 34 31 02 24 48 40 36 48 13 28 49 37 46 18 13 42 44 25 16 21 29 19 50 08 08 06 11

12-1492-0117 Duncan River Fish Stranding Survey Form

Crew:			Follow-up Required (If so, why)?
Site Name:			
Index or Non-Index Site:			Future flow reduction problems (next 0.5m decrease)?
UTM Zone:	UTM Easting:	UTM Northing:	
Date:		Estimated Verticle Drop (m)[1]:	Ramping Description:
Time:		Previous Discharge (kcfs):	
Weather:		Resulting Discharge (kcfs):	**Comments:**
Air Temperature:		Flow Ramping? (yes or no):	
Mainstem Water Temperature:			

Isolated Pool Stranding

No. New Pools Present:	Number of pools connected:
No. Pools Sampled:	
Sampling Gear Used:	

Interstitial Egg & Fish Stranding

Substrate checked? Yes / No if not, why?	Size of area sampled (m^2):
Recon survey? Yes / No OR Detailed survey with separate datasheet? Yes / No	
Substrate Type (circle major types that apply): Sand / Gravel / Cobble / Boulder	

Photodocumentation

Camera Type (e.g., 35 mm, digital)

Image #	Orientation	Comments

1 The estimated vertical drop from the drawdown zone of the previous water elevation to the current water elevation.

Site Sketch (Reference the Duncan River Mainstem with arrow indicating direction of flow)	Area of site

12-1492-0117 Duncan Stranding Habitat and Fish Record

Date:_______________ Crew:___________________ Weather:______________________________________

Site #	Side Channel or Mainstem	Pool or Interstitial ID (i.e. P1 or I1)	Time at Stranding Mechanism	Area (m^2)	Complexity (Zero to Low or Moderate to High)	Substrate (sizes and dominance)	Cover Types (LWD, SWD, OV, CB, DP, SP, INT, N/A) and Percentage				Species	Length (mm)	Salvaged	Cover Association	Number of Fish Remaining in Pool	Comments (Is fish marked? Which pass, settings, effort and time on each pass)

GOLDER ASSOCIATES LTD.
SNORKEL SURVEY FORM

RIVER:_______________ SITE:______________ PROJECT:___________________ CREW:_____/____/____

SNORKELLING DATE______/_____/_______ TIME: ________ AIR TEMP(C):________ WATER TEMP(C):________

WEATHER:____________________________ DAYTIME:_______ NIGHTTIME ________

VISIBILITY (M)___________ DISTANCE(M)____________ TOTAL SURVEY TIME (MIN)______________

Fish Species	Life Stage	Number of Fish	Depth (m)	Substrate Type	Cover

Duncan River Abundance Snorkeling 12-1492-0117

Date ____________________ Time ____________________ Crew ____________________

Stratum (1-4) ____________

1 = shallow, slow
2 = shallow, fast
3 = deep, slow
4 = deep, fast

Site Name ______________________________

Site parameters

	Waypoint	Longitude (or Easting)	Latitude (or Northing)	Depth
Upstream				
Downstream				

Site Length (m) ____________________

Visibility (m) ____________________

Width sampled (m) ________________

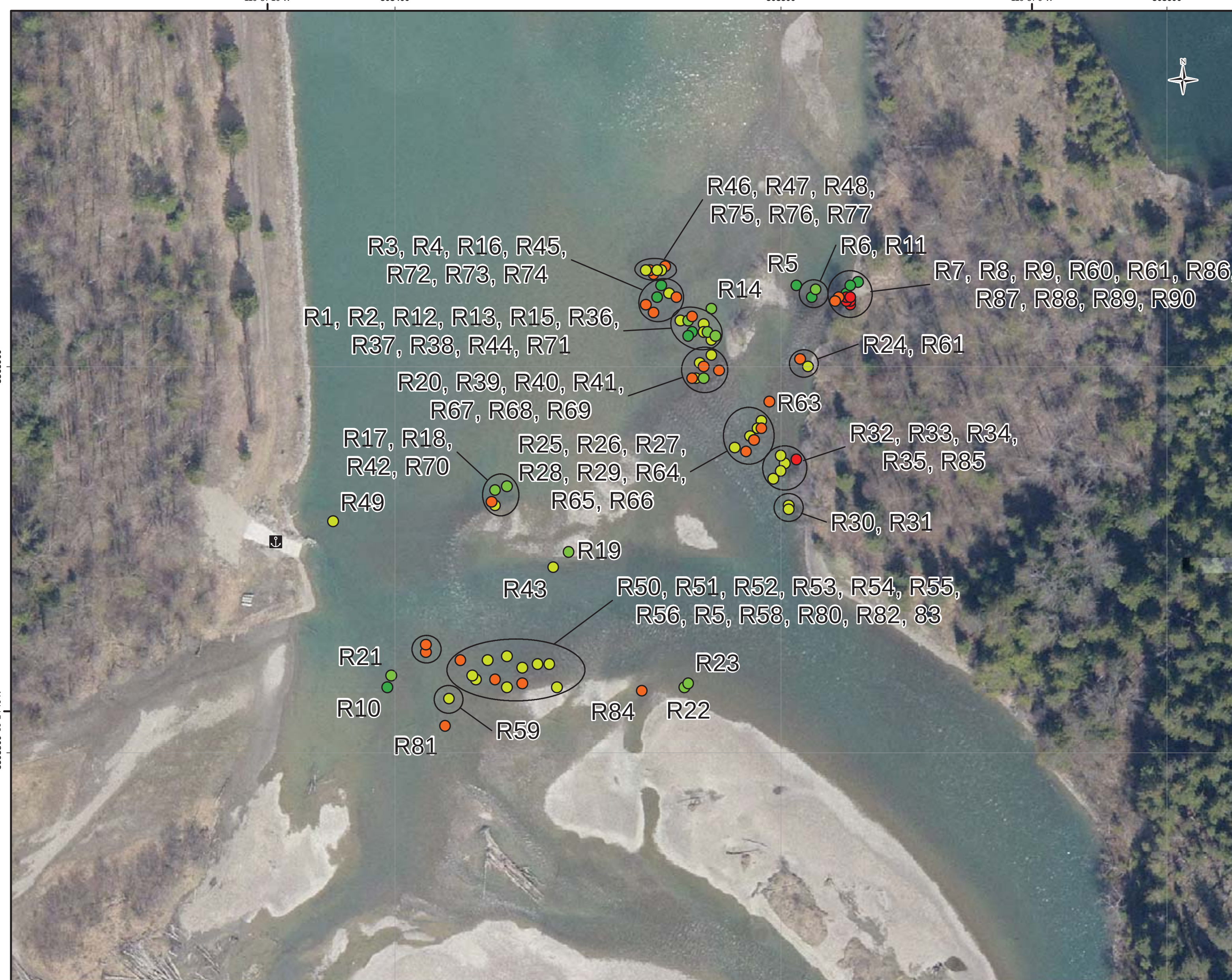
Fish counts

Snorkeller	MW	RB

Comments

APPENDIX B

Lower Duncan River Habitat Maps



Legend

Rainbow Trout Redd and Enumeration Date

- 13/04/2011
- 18/04/2011
- 26/04/2011
- 02/05/2011
- 01/06/2011

Boat Launch

Scale:1:1,000

20 10 0 20 40

Metres

Reference

Ortho Date: April 30, 2009
Discharge at DRL: 73 - 74cms
Rainbow Trout Redd, Piezometer,
Temperature Logger: Mountain Water Research

Disclaimer

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CLIENT:
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PROJECT:
DDMMON-2 Lower Duncan River Habitat Use Monitoring

Positions and Timing (Represented by Colour) of Rainbow Trout Redds in the Tailout Area, 2011

JOB No:	COMPILATION DATE:	Figure 3-7
VE51873.2011.337	2012/06/14	
GIS FILE: redd_location_2011_v2.mxd	REVISION DATE: 2012/09/05	
PDF FILE: redd_location_2011_v2.pdf	PRINTING DATE: 2012/09/05	

PROJECTION:	DATUM:	ANALYST:	QA/QC:	REQ:
UTM Zone 11	NAD83	KS	JB	JB

amec

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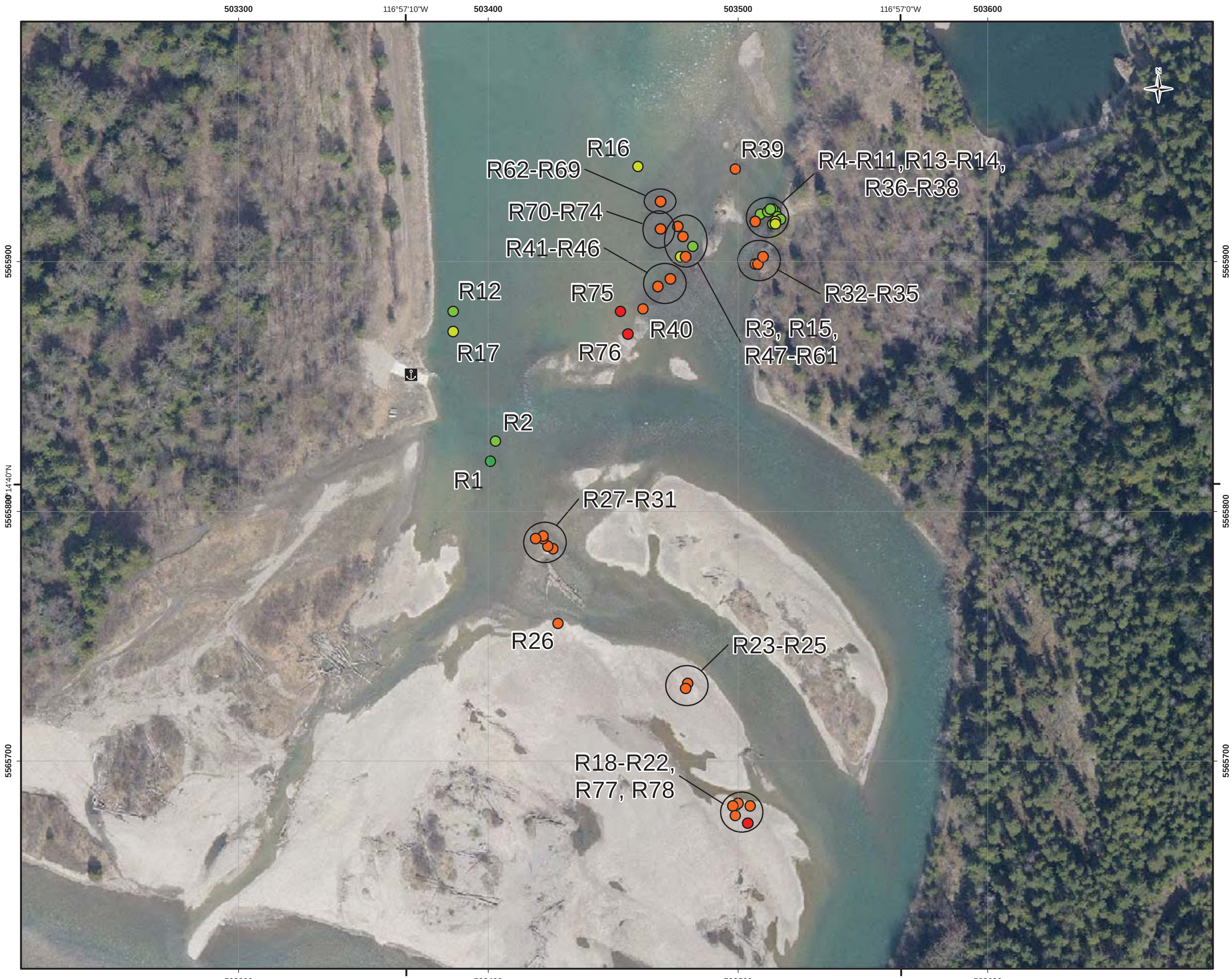

Legend

Rainbow Trout Redd and Enumeration Date

- 4/3/2012
- 4/10/2012
- 4/18/2012
- 5/8/2012
- 5/23/2012

Boat Launch

Scale:1:1,500

20 10 0 20 40

Metres

Reference

Ortho Date: April 30, 2009
Discharge at DRL: 73 - 74cms
Rainbow Trout Redd, Piezometer,
Temperature Logger: Mountain Water Research

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PROJECT:
DDMMON-2 Lower Duncan River Habitat Use Monitoring

Positions and Timing (Represented by Colour) of Rainbow Trout Redds in the Tailout Area, 2012

JOB No: VE51873.2011.337	COMPILATION DATE: 2012/06/14	Figure 3-8
GIS FILE: redd_location_2012_v2.mxd	REVISION DATE: 2012/09/05	
PDF FILE: redd_location_2012_v2.pdf	PRINTING DATE: 2012/09/05	

PROJECTION:	DATUM:	ANALYST:	QA/QC:	REQ:
UTM Zone 11	NAD83	KS	JB	JB

amec

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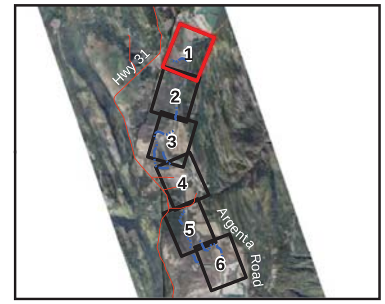
Legend

- River km (AMEC, 2009)
- Named Sidechannel
- Kokanee Spawning Use
- Kokanee Spawning Use - Area Lost

Note:
Mapped areas not to scale



Scale:1:5,000

100 50 0 100 200

Meters

Reference
Ortho Date: April 30, 2009
Discharge at DRL: 73 - 74cms
Kokanee Spawning: AMEC (Oct.2009)

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PROJECT:
DDMMON-4 Lower Duncan River Kokanee Spawning Monitoring

2012 Kokanee Spawning Areas and Areas Dewatered in the LDR

JOB No:		COMPILATION DATE:	
VE51905		February, 2013	
GIS FILE:		REVISION DATE:	
kokanee_spawning_2012_combined.mxd		February, 2013	
PDF FILE:		PRINTING DATE:	
kokanee_spawning_2012_combined.pdf		2012/01/02	

PROJECTION:	DATUM:	ANALYST:	QA/QC:	REQ:
UTM Zone 11	NAD83	AA	MY	CL

amec

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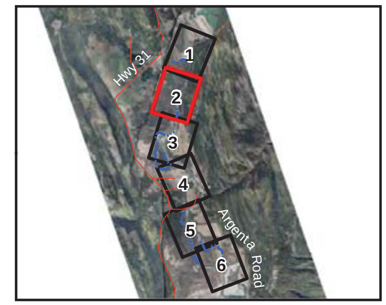
Legend

- River km (AMEC, 2009)
- Named Sidechannel
- Kokanee Spawning Use
- Kokanee Spawning Use - Area Lost

Note:
Mapped areas not to scale



Scale:1:5,000

100 50 0 100 200

Meters

Reference
Ortho Date: April 30, 2009
Discharge at DRL: 73 - 74cms
Kokanee Spawning: AMEC (Oct.2009)

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BC Hydro

PROJECT:
DDMMON-4 Lower Duncan River Kokanee Spawning Monitoring

2012 Kokanee Spawning Areas and Areas Dewatered in the LDR

JOB No: VE51905	COMPILATION DATE: February, 2013
GIS FILE: kokanee_spawning_2012_combined.mxd	REVISION DATE: February, 2013
PDF FILE: kokanee_spawning_2012_combined.pdf	PRINTING DATE: 2012/01/02

PROJECTION:	DATUM:	ANALYST:	QA/QC:	REQ:
UTM Zone 11	NAD83	AA	MY	CL

amec

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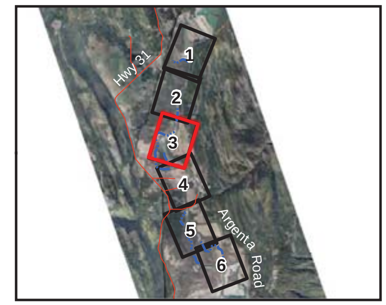
Legend

- River km (AMEC, 2009)
- Named Sidechannel
- Kokanee Spawning Use
- Kokanee Spawning Use - Area Lost

Note:
Mapped areas not to scale



Scale:1:5,000

100 50 0 100 200
Meters

Reference
Ortho Date: April 30, 2009
Discharge at DRL: 73 - 74cms
Kokanee Spawning: AMEC (Oct.2009)

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PROJECT: DDMMON-4 Lower Duncan River Kokanee Spawning Monitoring

2012 Kokanee Spawning Areas and Areas Dewatered in the LDR

JOB No: VE51905	COMPILATION DATE: February, 2013
GIS FILE: kokanee_spawning_2012_combined.mxd	REVISION DATE: February, 2013
PDF FILE: kokanee_spawning_2012_combined.pdf	PRINTING DATE: 2012/01/02

PROJECTION:	DATUM:	ANALYST:	QA/QC:	REQ:
UTM Zone 11	NAD83	AA	MY	CL

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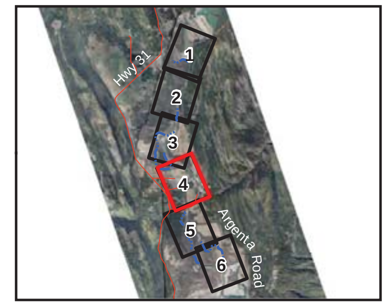
Legend

- River km (AMEC, 2009)
- Named Sidechannel
- Kokanee Spawning Use
- Kokanee Spawning Use - Area Lost

Note:
Mapped areas not to scale



Scale:1:5,000

100 50 0 100 200

Meters

Reference
Ortho Date: April 30, 2009
Discharge at DRL: 73 - 74cms
Kokanee Spawning: AMEC (Oct.2009)

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PROJECT:
DDMMON-4 Lower Duncan River Kokanee Spawning Monitoring

2012 Kokanee Spawning Areas and Areas Dewatered in the LDR

JOB No:			COMPILATION DATE:	
VE51905			February, 2013	
GIS FILE: kokanee_spawning_2012_combined.mxd			REVISION DATE: February, 2013	
PDF FILE: kokanee_spawning_2012_combined.pdf			PRINTING DATE: 2012/01/02	
PROJECTION: UTM Zone 11	DATUM: NAD83	ANALYST: AA	QA/QC: MY	REQ: CL

amec

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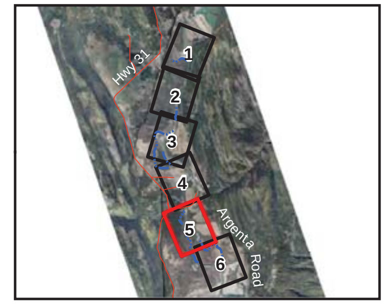
Legend

- River km (AMEC, 2009)
- Named Sidechannel
- Kokanee Spawning Use
- Kokanee Spawning Use - Area Lost

Note:
Mapped areas not to scale



Scale:1:5,000

100 50 0 100 200

Meters

Reference
Ortho Date: April 30, 2009
Discharge at DRL: 73 - 74cms
Kokanee Spawning: AMEC (Oct.2009)

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BC Hydro

PROJECT:
DDMMON-4 Lower Duncan River Kokanee Spawning Monitoring

2012 Kokanee Spawning Areas and Areas Dewatered in the LDR

JOB No: VE51905			COMPILATION DATE: February, 2013	
GIS FILE: kokanee_spawning_2012_combined.mxd			REVISION DATE: February, 2013	
PDF FILE: kokanee_spawning_2012_combined.pdf			PRINTING DATE: 2012/01/02	
PROJECTION: UTM Zone 11	DATUM: NAD83	ANALYST: AA	QA/QC: MY	REQ: CL

amec

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Legend

- River km (AMEC, 2009)
- Named Sidechannel
- Kokanee Spawning Use
- Kokanee Spawning Use - Area Lost

Note:
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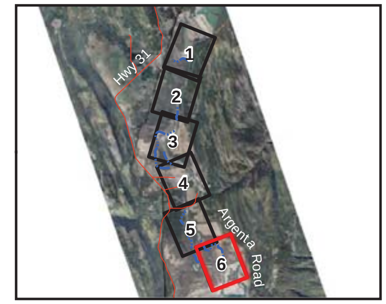


Scale:1:5,000

100 50 0 100 200

Meters

Reference
Ortho Date: April 30, 2009
Discharge at DRL: 73 - 74cms
Kokanee Spawning: AMEC (Oct.2009)

Disclaimer

CLIENT: BC Hydro

PROJECT: DDMMON-4 Lower Duncan River Kokanee Spawning Monitoring

2012 Kokanee Spawning Areas and Areas Dewatered in the LDR

JOB No: VE51905	COMPILATION DATE: February, 2013
GIS FILE: kokanee_spawning_2012_combined.mxd	REVISION DATE: February, 2013
PDF FILE: kokanee_spawning_2012_combined.pdf	PRINTING DATE: 2012/01/02

PROJECTION:	DATUM:	ANALYST:	QA/QC:	REQ:
UTM Zone 11	NAD83	AA	MY	CL

amec

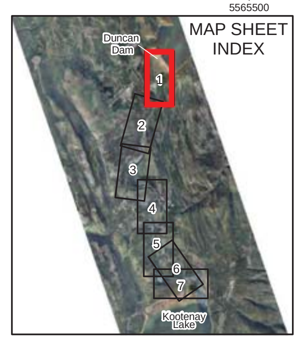
LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



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PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES

Golder Associates

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

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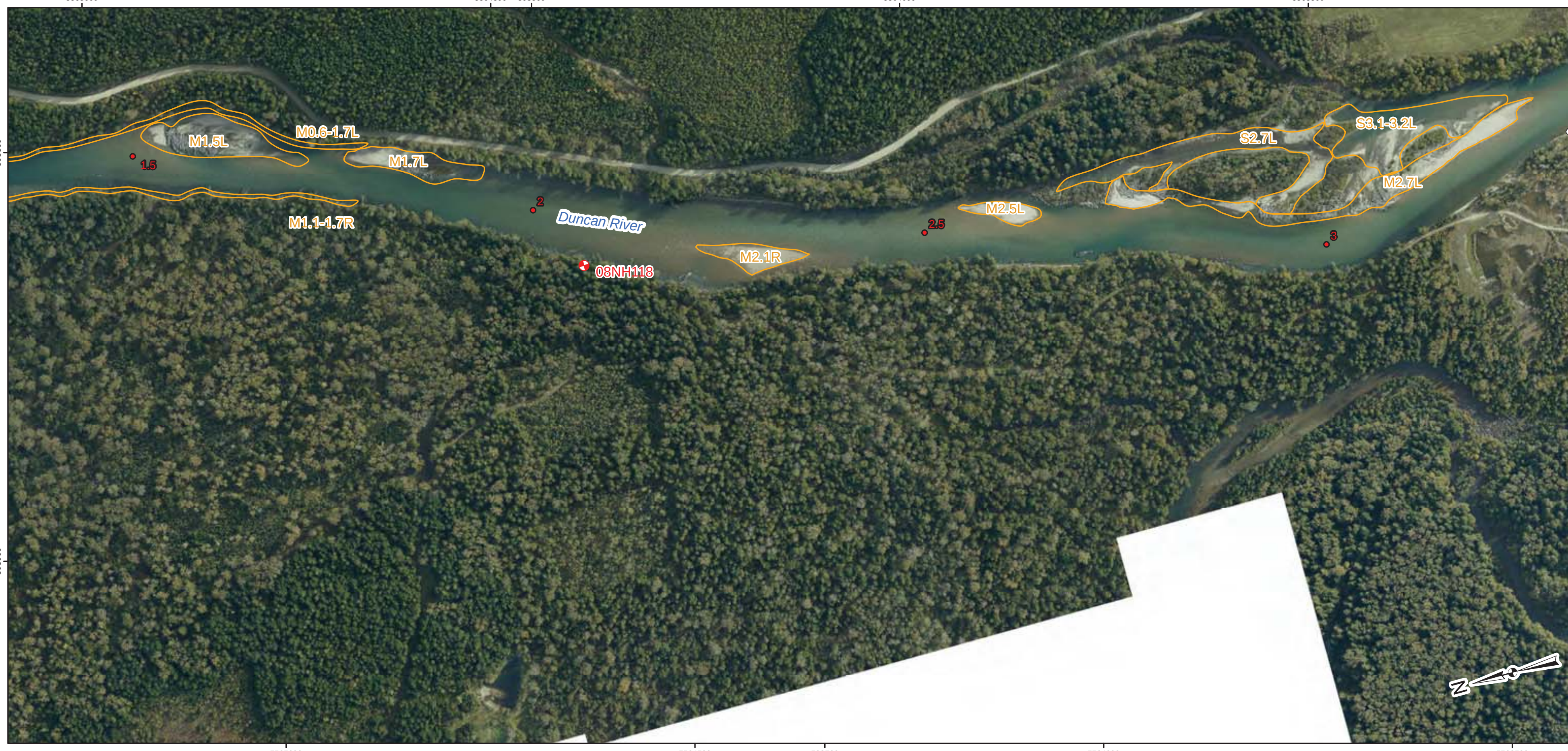

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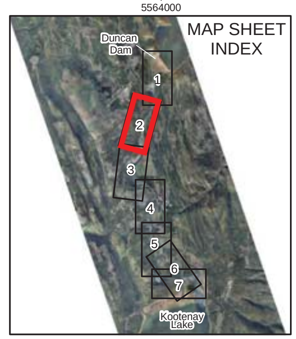
LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



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SCALE 1:5,000

PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

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

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503000 502500 5563000 5562500 5562000 5561500

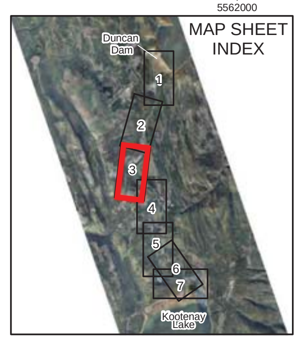
LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



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PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

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

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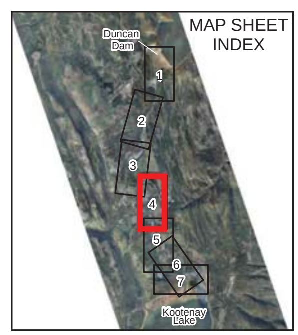

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LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83

PROJECT
DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE
LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

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

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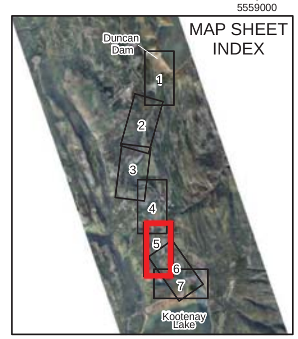
LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



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PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER STRANDING ASSESSMENT SITES

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

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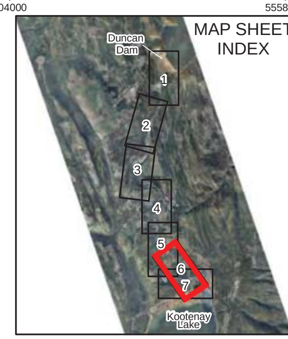
LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



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PROJECT

DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE

LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

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

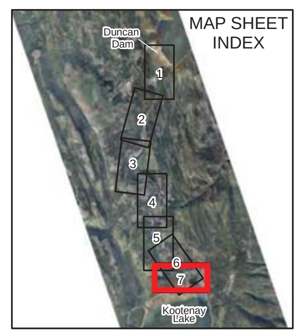
LEGEND

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

REFERENCE

1. IMAGERY FROM BC HYDRO, 2012

PROJECTION: UTM ZONE 11N DATUM: NAD 83



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PROJECT
DDMMON-16
LOWER DUNCAN RIVER FISH
STRANDING IMPACT MONITORING

TITLE
LOWER DUNCAN RIVER
STRANDING ASSESSMENT SITES

Golder Associates

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

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