

**DOWNSTREAM ENVIRONMENTAL IMPACTS OF RESERVOIR
HIGH OUTFLOWS – WITH A FOCUS ON FISHERIES**

by

Ali Naghibi

B.Sc., Sharif University of Technology, 1998

M.Sc., Sharif University of Technology, 2000

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Abstract

Extreme floods pose a significant risk to communities and environments in river systems throughout the world. In many cases, sensitivity of this issue is heightened for regulated rivers where downstream impacts of reservoirs are directly affected by operational decisions. Therefore, many North American jurisdictions require asset owners to assess downstream effects. Loss of life and economic impacts have been widely addressed in the literature. Nevertheless, immediate and long-term environmental impacts of such extreme events have not been holistically addressed. This work develops a framework for quantitatively estimating immediate and long-term fisheries impacts of extreme floods. The framework may also be generalized to other environmental systems. Several models are developed to support it. The immediate effects of extreme events are assessed with three models. These include: a probabilistic individual-based model that employs the results of a transient hydrodynamic model to estimate fish loss during extreme floods; a sampling simulation model that utilizes the results of a transient morphodynamic model and derives a probabilistic relationship between egg loss and flood intensity; and a habitat change estimation model that evaluates the available habitat difference before and after extreme events, given the results of hydrodynamic and morphodynamic models. A fish population recovery model is also developed and employed to estimate long-term impacts of extreme events, given the results of the immediate impact estimation models. An approach for estimating a number of risk-based performance measures that characterize the impacts and recovery from extreme events is also developed. These performance measures include existing formulations for vulnerability, engineering resilience, and ecological resilience, as well as a new measure which is introduced in this work, as vulnerability divided by engineering resilience. This new performance measure is designed to characterize both short- and long-term performance of the environmental system. Planning, design, and real-time operation of reservoirs, participatory water use planning, and licensing and relicensing decisions for proposed and existing water resource projects are cases in which such estimates may be useful. Applicability of this framework is demonstrated for the case study of the Lower Campbell River in British Columbia, Canada.

Preface

This work is conducted as part of a project in a research group led by Dr. Barbara Lence at the Department of Civil Engineering of the University of British Columbia. Several professors and graduate students collaborate in this project to estimate downstream life safety, economic, and environmental impacts of reservoir high outflows. My Ph.D. research is the environmental core of the project. A part of this research work involved collaborating and guiding Ms. Joanna Glawdel, an M.A.Sc. Student, in developing hydrodynamic and morphodynamic models of the Lower Campbell River, and in identifying an empirical relationship between flow conditions and egg loss rates for this system. Results of the hydrodynamic and morphodynamic models are used as input data in Chapter 6. However, instead of employing her developed empirical estimates of egg loss rate, I developed and applied a direct sampling method to estimate egg loss, in Chapters 4 and 6, respectively.

I developed and implemented several MATLAB codes for different models of the proposed framework in this research. These include: the pre-processor (*preprocessor.m*), main (*main.m*), and post-processor (*postprocessor.m*) program modules of the immediate fish survival estimation model; and the fish population recovery model (*salmon.m*). These codes are maintained at the Department of Civil Engineering of the University of British Columbia¹, and are available upon request². A user's guide for these codes is provided in Appendix A of this thesis.

¹ <http://www.civil.ubc.ca/people/faculty/faculty-lence.php>

² lence@civil.ubc.ca

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List of Abbreviations

HSC	:	Habitat Suitability Criteria
HSI	:	Habitat Suitability Index
IFIM	:	Instream Flow Incremental Methodology
LP	:	Linear Programming
MCDA	:	Multi-Criteria Decision Analysis
NFS	:	Numerical Fish Surrogate
NLP	:	Non-Linear Programming
SV	:	State Variable
V/R	:	Vulnerability ÷ Engineering Resilience
WUA	:	Weighted Usable Area

List of Notations

Δpop_{max}	:	maximum difference between the two time series of fish population, with and without the extreme event, in Figure 1-2
Δt_h	:	time step of hydrodynamic and morphodynamic models (sec)
Δt_{h-out}	:	time step of hydrodynamic and morphodynamic output files (sec)
Δt_f	:	time step of immediate fish survival estimation model (sec)
Δt_l	:	time step of long-term fish population dynamics model (sec)
$D_{deposition}$:	final depth of deposition (i.e., fill) during the flood (m)
D_{redds}	:	depth of redds (m)
D_{scour}	:	maximum depth of scour during flood (m)
HSI_t^i	:	habitat suitability index of i^{th} instream cell at time t
SV_t	:	value of state variable at time t
SV_{t+1}	:	value of state variable at time $t+\Delta t_l$
d_t^i	:	water depth of i^{th} instream cell at time t (m)
$iloss_t$:	immediate loss of state variable due to the extreme event between time t and time $t+\Delta t_l$
in_t	:	input from preceding state variable between time t and time $t+\Delta t_l$
n_e	:	number of spawning cells
n_f	:	number of fish in sample
$nloss_t$:	normal loss of state variable between t and $t+\Delta t_l$
n_{samp}	:	number of samplings in simulation
n_{se}	:	number of spawning cells in which eggs survive
n_{sf}	:	number of survived fish in sample
out_t	:	output to proceeding state variable between time t and time $t+\Delta t_l$
qx_t^i	:	x component of specific discharge for i^{th} instream cell at time t ($m^3/s/m$)
qy_t^i	:	y component of specific discharge for i^{th} instream cell at time t ($m^3/s/m$)
v_t^i	:	flow velocity magnitude for i^{th} instream cell at time t
vx_t^i	:	x component of flow velocity for i^{th} instream cell at time t (m/s)
vy_t^i	:	y component of flow velocity for i^{th} instream cell at time t (m/s)

CC	:	smolts carrying capacity of aquatic system
$E[x]$:	expected value of x
fry	:	number of emergent fry
i	:	instream cell counter
pop_1	:	population at point 1 in Figure 1-2
pop_2	:	population at point 2 in Figure 1-2
pop_3	:	population at point 3 in Figure 1-2
S	:	smolt survival rate at very low density
T_{ee}	:	duration of extreme event (sec)
v_{max}	:	maximum velocity in river reach during flood (m/s)
v_{sust}	:	sustained swimming speed of target species (m/s)
ΔL_{max}	:	maximum displacement of fish at each time step (m)
ΔL_{swim}	:	distance that fish swim at each time step (m)
n	:	probability parameter in binomial distribution function
p	:	number of samples in binomial distribution function
t	:	time (sec)
t_1	:	beginning time of extreme event (sec)
t_2	:	ending time of extreme event (sec) or (yrs)
t_3	:	time at which system reaches an equilibrium (yrs)

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Dedication

This work is dedicated to my beloved children, *Ava* and *Nima*, who are the reason why my heart beats.

Chapter 1: Introduction

1.1 Synopsis

The exposure of aquatic species to different hazards, the complexity of models that estimate the effects of such hazards, and the temporal and spatial scale with which the impact of such hazards are estimated contribute to uncertainty in the assessment of environmental performance of water resource systems. Although such uncertainty may affect the reliability and validity of water resource decisions, these effects have not adequately been accounted for in water resource decision-making approaches. In existing participatory water use planning processes, for example, uncertain estimates for different attributes, e.g., environmental impacts and economic benefits, are used to investigate trade-offs among potential water use plan alternatives, and to rank such alternatives based on these trade-offs. This work develops an approach for classifying the long- and short-term environmental impacts of water resource systems due to high controlled or uncontrolled flows – hereafter referred to as extreme events. From the long-term perspective, the probability of environmental system recovery to its pre-event status, or to a new equilibrium, is estimated. This task is undertaken through probabilistic analyses of an aggregated fish population model. From the short-term perspective, a suite of models to estimate the immediate impacts of extreme events is developed. These models include: immediate fish and egg loss, and immediate habitat change models. The applications of the developed research may be in planning, design, and real-time operation of reservoirs, in multi-stakeholder decision-making processes, in classifying dams based on their downstream consequences, and in negotiations between asset owners and the government. The Lower Campbell River (downstream of the John Hart Dam, British Columbia) is used as a case study to demonstrate the methods of analysis developed in this research.

1.2 Objective

The short- and long-term fisheries impacts of extreme events are identified and represented by estimates of risk-based performance measures. These may facilitate decision-making under uncertainty in water resource problems dealing with multiple attributes, including

environmental attributes. The research focus is on estimating impacts on fisheries downstream of dams, but it may be generalized for other environmental attributes.

1.3 Scope of the Research

A conceptual influence diagram for optimal operation of a reservoir is illustrated in Figure 1-1 where the optimal dam release is affected by a number of considerations. These include:

- Maintain integrity of the dam;
- Satisfy downstream water demand, e.g., water supply and irrigation;
- Maximize pre-defined benefits of the reservoir, e.g., hydropower energy production; and
- Minimize downstream impacts.

Federal Canadian and U.S. guidelines categorize the downstream impacts in terms of life safety, economic, and environmental effects (CDA 2007; FEMA 2004). While there is no force of regulation in these guidelines, the British Columbia Dam Safety Regulation mandates all dam owners to classify their assets based on these three aspects (BC Dam Safety Regulation 2000).

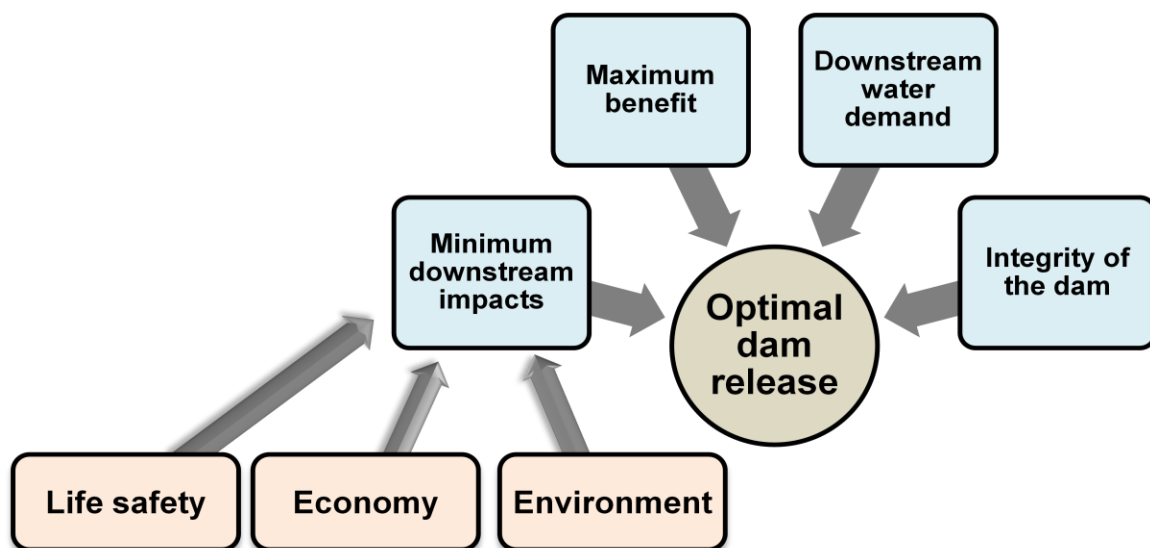


Figure 1-1. A sample influence diagram for reservoir operation

Quantitative life safety impacts of reservoir extreme outflows have been studied in the literature (Johnstone and Lence 2009; Jonkman et al. 2002). These studies include both empirical (McClelland and Bowles 2002) and mechanistic (Johnstone et al. 2005; Lind et al. 2004) life safety estimation approaches. The BC Hydro Life Safety Model (BC Hydro 2005) is a tool that has been developed to mechanistically estimate such impacts. Economic effects of extreme events have also been investigated (see, e.g., Hartford and Baecher 2004). The environmental impacts of extreme events, however, have not been quantitatively estimated (Lence et al. 2009). This dissertation focuses on developing a framework for estimating such impacts.

In estimating the environmental quality, i.e., environmental performance, of river systems under reservoir extreme outflow conditions, the downstream environmental attributes of concern must be identified. A number of candidates exist for such attributes. These include:

- fish population,
- fish habitat,
- geomorphology,
- water quality,
- riparian vegetation,
- plankton,
- invertebrates, and
- terrestrial wildlife

Among these environmental attributes, fish population may be considered the most holistic because it has direct relationships with all other attributes in the list. Fish habitat is the most commonly used attribute; however, for the reasons explained in Section 3.1, it may not fully describe the impacts of extreme events. Therefore, fish population is selected as the representative environmental attribute in this dissertation.

The focus of this dissertation is further narrowed to estimating the impacts on anadromous fish, and many of the models developed are specific to such species. However, the developed

framework is a modular one, and may easily be adopted for other species by applying modifications to the modules that are species-specific (e.g., life cycle of the fish).

Under normal flow conditions, fish population in a riverine system may be considered to be in ecological equilibrium. That is, although the population is not constant over time, the factors that tend to increase and decrease the population are in long-term balance. This results in a stationary population where the average and standard deviation of the population is constant over time even though the population is not constant. Extreme events, e.g., a flood with a 50-year return period, may change the fish population by scouring eggs, altering fish habitat, and washing adult fish away from their spawning habitat. Impacts of such extreme events are conceptualized in Figure 1-2.

Figure 1-2 shows a special case of a stationary population where the population is steady-state (i.e., does not change over time). A sample case of a general stationary population, where the population is not constant, is shown in Figure 1-3. This research does not rely on the assumption of steady-state conditions for developing and applying the models. However, Figure 1-2 (steady-state conditions) is frequently referenced instead of Figure 1-3 (i.e., stationary conditions) for visual simplicity and emphasis on the concepts of immediate and long-term impacts. However, development and application of the models in the thesis are based on stationary conditions.

Before point 1 in Figure 1-2, the river is regulated under normal operations, and the fish population has a stationary condition with natural variations. At point 1, an extreme event occurs and may result in some immediate impacts including fish mortality, egg loss, loss of other aquatic and terrestrial organisms, and changes in food and geomorphology. After point 2, where immediate effects may be considered complete, the system undergoes a transient recovery process until it reaches a new stationary condition at point 3, which may or may not be the same as the pre-event condition. To the author's knowledge, there is no published model to assess immediate environmental effects of single extreme events.

Using Figure 1-2, three risk-based performance measures may summarize short- and long-term impacts of the extreme event on the environmental systems. These are:

- Vulnerability, i.e., expected maximum population loss due to the extreme event,
- Engineering Resilience, i.e., expected speed of the system recovery, where recovery speed is the inverse value of recovery time, and
- Ecological Resilience, i.e., whether the system returns to its pre-event equilibrium. An advanced definition of ecological resilience, i.e., the degree to which the system returns to its pre-event equilibrium, is identified and employed in this work.

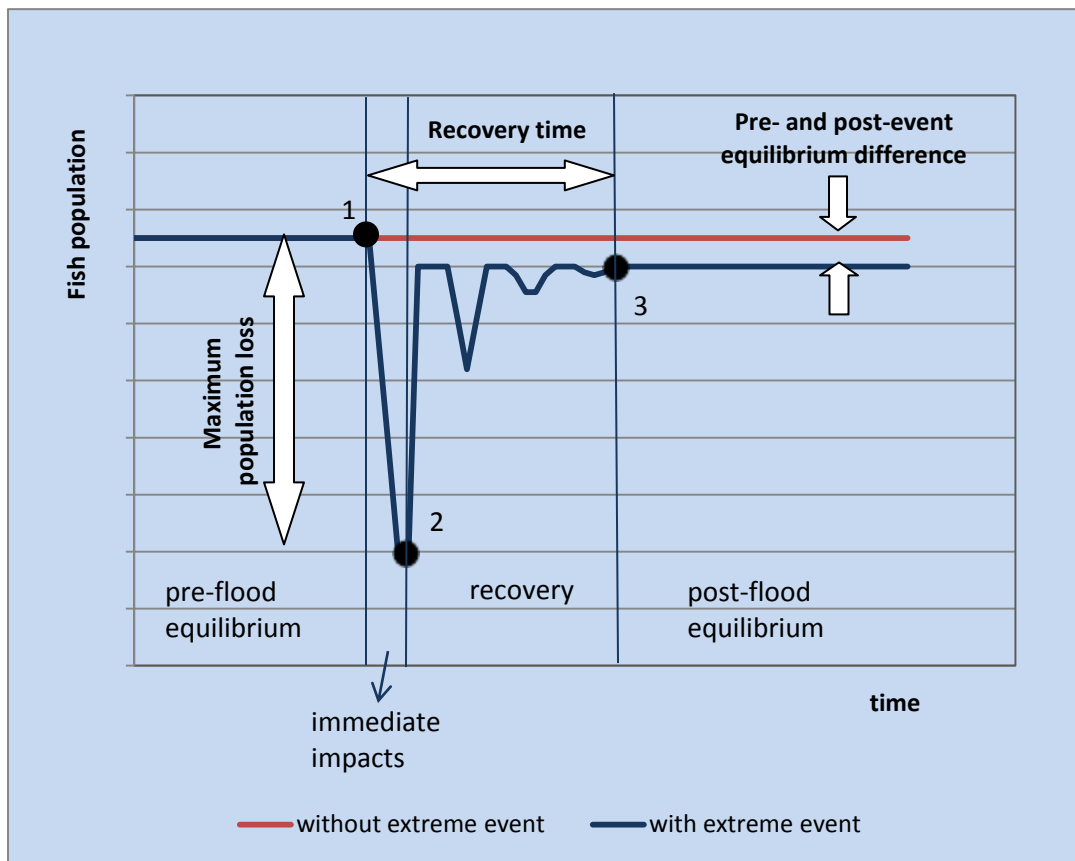


Figure 1-2. Hypothesized fish population dynamics for river systems before and after extreme events

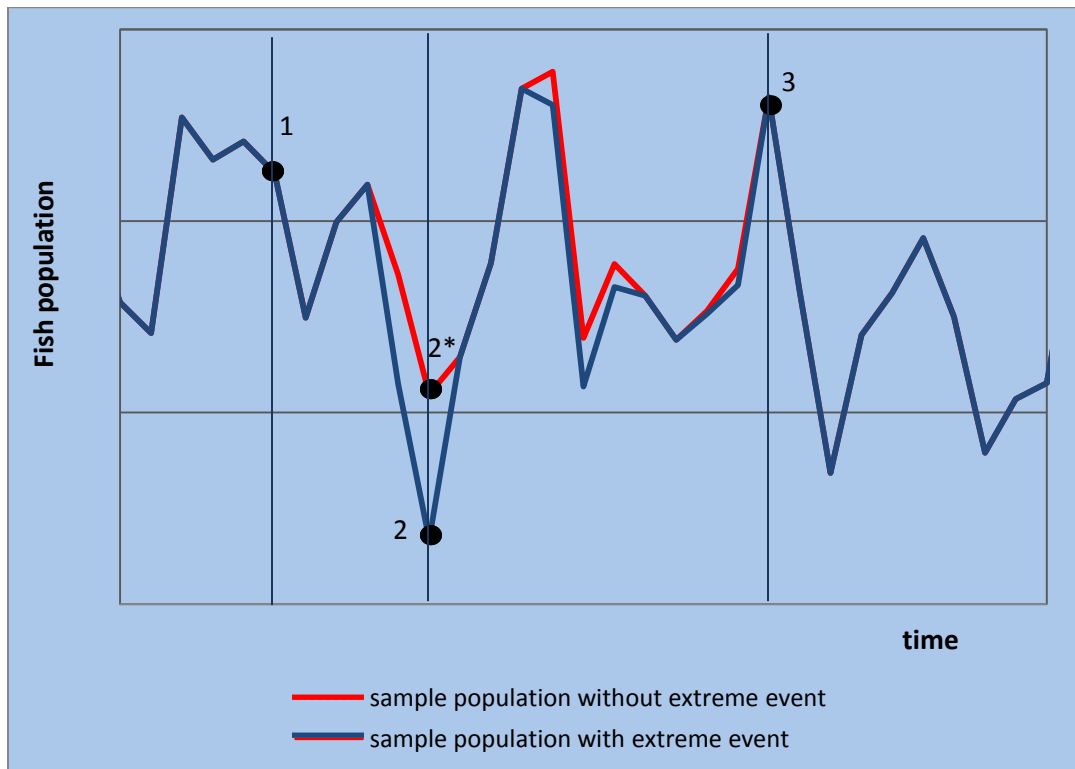


Figure 1-3. Sample actual fish population dynamics for river systems before and after extreme events

Background literature for environmental impact estimation of extreme events is reviewed in Chapter 2. In Chapter 3, a framework, composed of both short- and long-term models, is proposed to estimate immediate and long-term impacts of extreme events, as conceptualized in Figure 1-2. The short- and long-term models of this framework are introduced in Chapters 4 and 5, respectively. Since the proposed framework and developed models are designed to be general, they may be applied to a wide range of cases with different hydrological and geomorphological conditions, and to different anadromous fish species. Risk-based performance measures which are estimated based on the results of the models may be employed in decision-making processes as an environmental performance value. In order to demonstrate the application of the framework, these models are then applied to a case study of the Lower Campbell River in Chapter 6. Conclusions are provided in Chapter 7.

Chapter 2: Literature Review

The background literature which may be drawn upon in developing a framework for estimating environmental impacts of reservoir extreme outflows is investigated in this chapter. In Section 2.1, different approaches that are available for estimating environmental quality based on the instream flow are presented. Environmental considerations in reservoir operation and planning are discussed in Section 2.2. Literature related to impacts of extreme events on fisheries in particular, and to risk-based performance measures, are reviewed in Sections 2.3 and 2.4, respectively.

It should be noted that this chapter focuses on investigating the relevance and applicability of reviewed methodologies to the objective of this research rather than determining their values for the purposes for which they are developed.

2.1 Instream Flow and Environmental Quality

There are a range of approaches for estimating the environmental quality of river systems under different instream flows. These approaches may be categorized based on the metrics they employ to assess the environmental quality. It should be noted that this categorization is flexible, and a combination of different methods may also be used to estimate environmental quality.

2.1.1 Hydrological Methods

These methods, such as the Tennant (Montana) Method, for example, are applied to provide preliminary estimations of the minimum environmental flows required to maintain the environmental quality of an aquatic system. This method is an empirical approach based on measurements of width, average depth, and average velocity in eleven streams in Montana, Wyoming, and Nebraska. Tennant (1976) observes that the quality of instream habitat is critically low for flows less than 10% of the average annual flow, and therefore, suggests that the 10% of the average annual flow be the minimum flow required to sustain the environment. The Tennant method has widely been applied to cases located in regions far

from the watersheds on which it is based, either with modifications (Orth and Maughan 1981), or as is (see, e.g., Tharme 2003, for applications).

2.1.2 Hydraulic Rating Methods

Hydraulic rating methods may be used to estimate hydraulic conditions, as surrogates of environmental quality, and relate them to flow rate. The most commonly used hydraulic rating approach is to consider the wetted perimeter as the representative hydraulic condition (Tharme 1996). By implementing a hydraulic model, or by applying empirical analyses, the relationship between flow rate and wetted perimeter is identified. The breakpoint of the wetted perimeter vs. flow rate curve, below which the wetted perimeter decreases dramatically, is usually selected as the minimum environmental flow (Tharme 2003). Most hydraulic rating methods were developed in the 1960s and 1970s (Stalnaker and Arnette 1976), and have been replaced by more sophisticated physical habitat models (Tharme 2003).

2.1.3 Physical Habitat Models

Habitat may be investigated at three different spatial scales, the microhabitat, mesohabitat, and macrohabitat scales. Bovee et al. (1998) describe these spatial units as follows:

- microhabitat is a localized area of the river with relatively homogenous depth, velocity, substrate, and cover. This area is usually on the order of one to several square metres;
- mesohabitat may contain many microhabitats, but may be typified by a common slope, channel shape and structure. Pools and riffles are typical mesohabitats. The length of a mesohabitat is about the same order of magnitude as the width of the stream; and
- macrohabitat, which includes the population of target species, the scale of which varies from the length of a river reach to that of a drainage basin.

Physical habitat models estimate the environmental quality by integrating a few hydraulic parameters of the flow with other riverine conditions such as sediment composition, which are important to fish species. The Fish and Wildlife Service developed a commonly used physical habitat estimation model, i.e., PHABSIM, that estimates the suitability of

microhabitat units in a river reach (Milhous et al. 1989; Waddle 2001). The Habitat Suitability Indices (HSI) of these units are evaluated using available Habitat Suitability Criteria (HSC) for target species. Identifying HSC for target species is a complicated process. One way is to survey the locations throughout the reach which are occupied by target species. Here, depth, velocity, substrate, and cover at all of these locations are evaluated, and a frequency distribution analysis is implemented to find the likelihood of a microhabitat to be occupied by fish (Waddle 2001). The fitted distributions, which are functions of depth, velocity, substrate, and cover, are referred to as HSC.

Given the HSC, the HSI of all microhabitat units may be calculated for a specific flow rate and channel geomorphology by executing a hydraulic model. PHABSIM estimates the Weighted Usable Area (WUA) for this flow rate by integrating calculated suitability values of microhabitat units throughout the reach. The PHABSIM model results are very sensitive to the HSC, and thus using HSC that are developed for another location may not be acceptable unless such usage is justified in the PHABSIM analysis report (Waddle 2001).

There are many applications of habitat-based models in the literature. BC Hydro (2004), for example, uses a two-dimensional hydrodynamic model, i.e., River2D, to estimate the HSI for effective spawning habitat given the HSC for target species. This model adopts the same approach as PHABSIM except for employing River2D as its hydraulic engine. In cases where habitat areas lie in rapidly varying flows, two-dimensional models are more accurate than one-dimensional models in estimating river performance in terms of WUA (Lacey and Millar 2004; Waddle et al. 2000).

Wide use of physical habitat modeling in instream flow analyses has also been accompanied with criticism. Biological integrity and fish use of habitat space depends on many abiotic and biotic factors. Sale et al. (1982) show that among the domains of water quality, energy sources, habitat structure, and flow regimes, PHABSIM considers the last two. Likewise, Schwartz and Herricks (2008) explain that PHABSIM reliance on hydraulic point measurements (i.e., the microhabitat scale) may not accurately model the use of habitat, and show that fish use of habitat space may be better viewed at the mesohabitat scale. Railsback

et al. (2003) question the assumptions of habitat modeling approaches in general. These include assuming that: habitat with high animal densities represent habitat with high quality; and animal populations respond positively to the availability of habitat which are similar to those that have been highly selected. Railsback (1999) specifically criticizes PHABSIM for emphasizing the hydraulics of a river reach rather than the habitat; and further for relying on hydraulic models that are at a different scale than the area for which HSC are estimated.

2.1.4 Ecohydrological Approaches

Recently, a paradigm has been established that relates the environmental quality of regulated rivers to their natural flow regime (Poff et al. 1997). In this approach, ecohydrological indicators (i.e., hydrological indicators that satisfy ecological requirements of the species of interest) are employed to assess the environmental performance of river systems. Suen et al. (2009), for example, suggest a number of indicators such as the “number of high flow events within each dry season” and “mean duration of low flow events.” If the value of these two indicators in the regulated river is close to those of the unregulated regime, spawning conditions are considered satisfactory.

2.1.5 Expert Opinion Approach

There are cases where no mathematical model is used to assess the environmental quality. Here, where no historical record for river flow is available, the environmental performance of an operation scenario is subjectively judged by decision-makers by observing the response of river conditions (e.g., velocity and depth) or by conducting a fish sampling survey in the low flow period of the year (King et al. 2003). Railsback (2000) uses the term “qualitative observation method” to describe a similar method, where stakeholders view the river at different flows given releases from an existing dam. The habitat is observed or measured at specific spots of the river, and judgment is made regarding the quality of habitat for the species of interest under the given conditions.

2.1.6 Fish Population Dynamics Models

Fish population dynamics models estimate time series of population changes for different life stages of fish considering the inter-relationships among these life stages in the fish life cycle.

The life cycle of anadromous fish and different approaches for modeling it are summarized herein.

2.1.6.1 Anadromous Fish Life Cycle

The conceptual life cycle of anadromous fish, i.e., salmon, is illustrated in Figure 2-1. Adult anadromous fish migrate from the ocean to the river until they reach the proper spawning location. The number of adult fish that return back to the river each year to spawn is referred to as escapement. A number of adult fish, which are referred to as strays, may spawn at streams other than their original ones. In the Campbell River, for example, the percentage contribution of the spawners that originate from the Quinsam River, a tributary of the Campbell River, is as high as 40% of the total spawning population (Bennet et al. 2010). Eggs are laid in nests dug in the gravel bed of the river. These nests are referred to as redds and may contain up to a few thousand eggs. The depth of redds depends on the species and the geographical location. Hatched eggs, i.e., alevins with yolk sacs, may stay in the gravel for a period of time, depending on their species, and emerge out of gravel as fry. The fry may either immediately migrate downstream or stay in the freshwater for a period of up to two years. These are referred to as ocean type and stream type salmon, respectively (Healey 1991; Higgs et al. 1995; Levings et al. 1986). Whether salmon are ocean or stream type, varies within a species as well as among different species. Stream type salmon fry which are more than one year old are called parr. When stream type or ocean type salmon are ready for downstream migration, they go through biological adaptation to be able to live in the salt water (i.e., smoltification). The fish grow and stay in the ocean for one to five years. Again, the period over which salmon stay in the ocean varies within a species as well as among different species, and those who survive normal loss (e.g., predation and starvation) and fishing exploitation will migrate back to the streams to complete their life cycle by spawning.

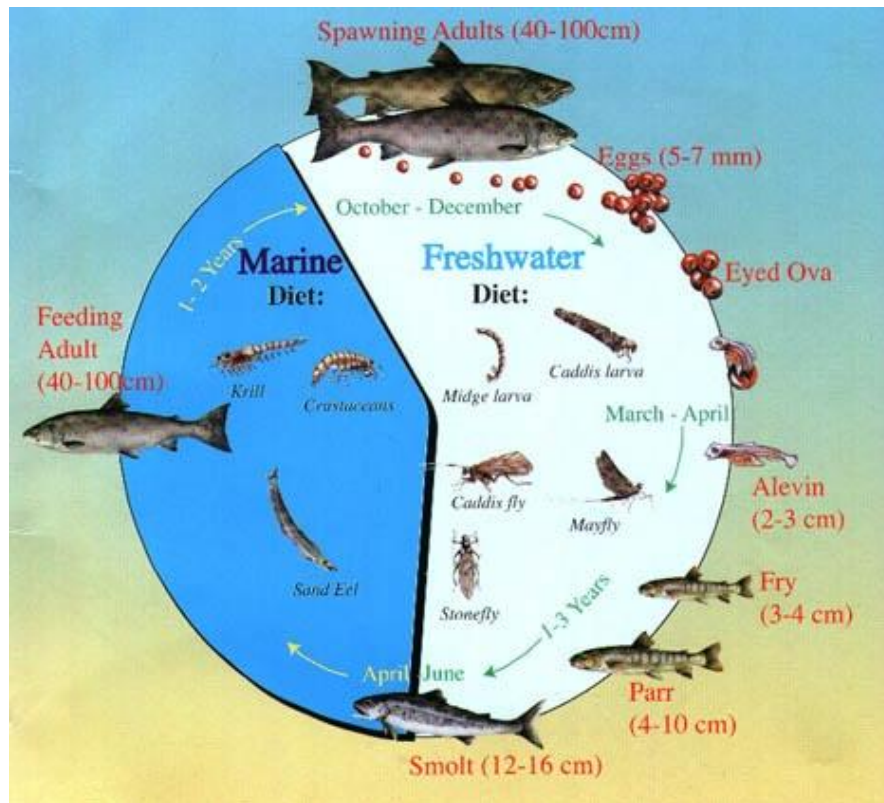


Figure 2-1. Conceptual life cycle of salmon (from EatonvilleNews 2011)

At each life stage throughout this life cycle, fish have specific needs (e.g., habitat and food) and may experience different threats (e.g., predation and disease). Fisheries and Oceans Canada (2009) summarizes these needs and threats in Table 2-1.

There are two major approaches in modeling the dynamics of environmental systems, e.g., fish population dynamics. These include: aggregated population-based and individual-based approaches.

Population-based models aggregate a group of individuals with common properties (e.g., eggs) in one population group, i.e., state variable, and estimate the dynamics of the system given presumed relationships among the state variables. Since the equations and behavioral assumptions are made on a population scale, and individual variations are ignored, these models are referred to as top-down approaches.

In individual-based models, on the other hand, the macro-scale behavior of the system is not addressed through equations between the state variables; rather emerges as a result of the behavior or processes that are followed by individuals within the system. Therefore, these models are called bottom-up approaches.

Population-based and individual-based modeling applications have been undertaken, and their advantages have been identified for different modeling contexts. The appropriateness of and choice of adopting either approach, or even a combination of both, is case-specific (Borshchev and Filippov 2004; Scholl 2001). While population-based models are computationally efficient due to the fact that they aggregate the individual components of a system, individual-based models are able to model effects of individual behavior, as well as individual variability (Grimm 1999). Therefore, the choice of population- or individual-based model is need-specific.

Table 2-1. Salmon life cycle needs and threats (from Fisheries and Oceans Canada)

Life cycle stage	Needs		Threats	
	Habitat	Food	Predators	Other
Egg <ul style="list-style-type: none"> • Head and body formation begins • Organ formation begins • Eyes become visible 	<ul style="list-style-type: none"> • Oxygenated water • Temperature from 5° to 9°C • Silt-free gravel bed • Steady water flow • Stream cover 	<ul style="list-style-type: none"> • Yolk of egg 	<ul style="list-style-type: none"> • Trout • Sucker • Squawfish • Whitefish • Kingfisher • Gull • Merganser • Mink • Otter 	<ul style="list-style-type: none"> • Gravel movement • Drastic change in water temperature • Drastic change in water level • Siltation • Fine sediment • Disease • Pollution
Alevin <ul style="list-style-type: none"> • Embryo breaks through egg membrane • Oxygen absorbed through gills • Lives in gravel spaces 	<ul style="list-style-type: none"> • Oxygenated water • Temperature from 5° to 14°C • Silt-free gravel bed • Steady water flow • Stream cover 	<ul style="list-style-type: none"> • Yolk sac 	<ul style="list-style-type: none"> • Trout • Sucker • Squawfish • Whitefish • Kingfisher • Gull • Merganser • Mink • Otter 	<ul style="list-style-type: none"> • Gravel movement • Drastic change in water temperature • Drastic change in water level • Siltation • Fine sediment • Disease • Pollution
Fry <ul style="list-style-type: none"> • Inflates swim bladder • Catches food • Exhibits darting reflex • Avoids light • Guards territory • Imprints home scent 	<ul style="list-style-type: none"> • Stream cover • Oxygenated water • Temperature from 5 to 14°C • Even water level and flow 	<ul style="list-style-type: none"> • Larval and adult terrestrial and aquatic insects, (e.g. mayfly, caddisfly, true flies) • Rotting fish carcasses • Fish eggs 	<ul style="list-style-type: none"> • Trout • Sucker • Squawfish • Whitefish • Kingfisher • Gull • Merganser • Mink • Otter 	<ul style="list-style-type: none"> • Gravel movement • Drastic change in water temperature • Drastic change in water level • Siltation • Fine sediment • Disease • Pollution • Blockage of migration route
Smolt <ul style="list-style-type: none"> • Migrates to estuary • Adapts to salt water • Develops scales and silver colour develop • Increases size 	<ul style="list-style-type: none"> • Unpolluted water in river and estuary • Estuary vegetation for shelter 	<ul style="list-style-type: none"> • Zooplankton (copepods amphipods, euphausiids) • Insects, (e.g. beetles, ants, grasshoppers, caterpillars) • Worms • Sandfleas • Shrimp 	<ul style="list-style-type: none"> • Mackerel • Grayling • Trout • Char • Loon • Heron • Tern • Kingfisher • Hake • Pollack • Dogfish • Older salmon 	<ul style="list-style-type: none"> • Filling or dredging of estuary • Pollution of estuary • Diversion of river water
Ocean Phase Salmon <ul style="list-style-type: none"> • Migrates into ocean • Increases size • Stocks intermingle, then return to home river 	<ul style="list-style-type: none"> • Ocean water 	<ul style="list-style-type: none"> • Zooplankton, (e.g. amphipods, copepods, euphausiids) • Larval crustaceans, (e.g. crab shrimp) • Small fish, (e.g. herring, squid, mackerel) 	<ul style="list-style-type: none"> • Tuna • Cod • Pollock • Hake • Lamprey • Gull • Heron • Cormorant • Seals • Whales • People 	<ul style="list-style-type: none"> • "Lost" nets • Ocean pollution • Ocean temperature change • Fishing
Spawner <ul style="list-style-type: none"> • Eggs, milt develop • Secondary sexual characteristics develop (colour, shape, teeth) • Scales absorbed • Eating stops • Organs degenerate 	<ul style="list-style-type: none"> • Migration route free from obstructions • Oxygenated water • Cool clean water • Silt-free gravel 	<ul style="list-style-type: none"> • None 	<ul style="list-style-type: none"> • Eagles • Bears • Otters • Minks • People 	<ul style="list-style-type: none"> • Very high or low water levels • Warm river temperatures • Obstructions (dams, slides, log jams, etc.) • Diseases • Pollution

2.1.6.2 Individual-based Modeling

Observing that PHABSIM relies on point hydraulic measurements for fish use of habitat, and that it lacks direct relationship between fish habitat and fish population, Railsback (2000) introduces the inSTREAM model as a free access environmental flow assessment model. inSTREAM is an individual-based model which simulates the fish population dynamics with hourly or daily time steps under different scenarios of flow, turbidity, and temperature time series. At each time step each individual fish may undertake any of four sets of actions: spawn, select a habitat, feed and grow, and survive or die (Railsback et al. 2009).

- Spawning happens when a female adult is present in a suitable habitat, and is biologically ready to spawn.
- Habitat selection, i.e., fish movement, is undertaken assuming that the fish is aware of its surrounding environment. Fish select cells that provide maximum fitness to them (i.e., cells with food and minimal mortality risk).
- Feeding and growth occur based on available food at the selected habitat and on the biological conditions of the fish.
- Survival is estimated by considering important mortality sources: high temperature, high velocity, stranding, poor condition, and predation. Survival of eggs is affected by dewatering, scouring and deposition, and high and low temperature.

Given these mechanisms, the model simulates the movement and survival of individual fish for time series of flow, turbidity, and temperature, and records the total population of fish at each time step. Although inSTREAM is originally developed for trout species, it may be applied for the river life stages of salmon species. Due to the structure of the model, it is most applicable to river life stages and at flows lower than bankfull levels. Also, the complexities of the model make it computationally intensive.

2.1.6.3 Population-based Modeling

Statistical and empirical data are usually available for most of the processes that fish go through in their life cycle (e.g., average egg survival rate for target species) at the population-scale. Therefore, population-based models that aggregate groups of fish and common properties (e.g., eggs in a river reach) may be employed to estimate fish population dynamics. SALMOD is a population-based model, developed by the U.S. Geological Survey (Bartholow et al. 2001), that simulates river life stages of salmon. Cohorts of fish at the same life stage (e.g., spawning adults, eggs, fry, and smolts) at each mesohabitat in the river reach are aggregated as state variables. Bartholow (1996) shows that results are not sensitive to spatial scale, and thus, a macrohabitat scale may be employed. The model employs a weekly time step and traces spawning, growth, and mortality rates of cohorts. Mortality rates are assumed to include three parts: base, water temperature-related, and habitat-related mortality. While base mortality is a deterministic empirical value for each species, water temperature-related and habitat-related mortality rates vary with the time series of temperature and flow at time steps of seven days. Using a physical habitat simulation model, e.g., PHABSIM, flow time series are converted to WUA time series, which are used to identify carrying capacity of different life stages. The model does not simulate the ocean life stages, and Bartholow (2001) recommends the analysis of only one or two years of simulation with weekly time steps. Therefore, utilizing the model for the entire life cycle, and for a large number of simulations that involve very long periods (e.g., decades) may not be practical.

Ford (1999) develops a System Dynamics model to estimate the impact of fishing policies on the population of chinook salmon in the Snake River, a tributary of the Columbia River, and shows how conveniently the complexities of that system may be modeled by the System Dynamics approach. The model utilizes monthly time steps, and aggregates all individual fish at the same life stage in the river reach to state variables (e.g., spawning fish, eggs, fry, and smolts). Moreover, he does not consider flow dependant carrying capacities and mortality rates. Rather, he employs constant carrying capacities and mortality rates for all life stages except for the smolts mortality rate, which is density-dependant. Therefore, computational time for each simulation is very short, and thus, several simulations are conveniently implemented to compare all possible operational policies, and to conduct

sensitivity analysis. Jessup (1998) also demonstrates the applicability of the System Dynamics approach for large-scale complex systems, by utilizing such an approach, with state variables aggregated on a watershed scale, to estimate the effect of urbanization on fish habitat and population.

2.2 Environmental Considerations in Reservoir Operation and Planning

In the 1960s, protecting non-market values of the rivers became a national concern in the United States, and the National Environmental Policy Act was passed in 1969. This act requires projects to prepare Environmental Impact Statements, which involves evaluating economic, social, and environmental consequences (Flug 1997). Since then, different approaches to estimate the relationship between environmental quality and instream flow (i.e., Section 2.1) have been employed to assess environmental impacts of reservoir operation.

Despite considerable similarities and differences, all existing approaches for determining required environmental flows for rivers may be categorized in two major groups: objective-based analyses and scenario-based analyses (Acreman 2005).

2.2.1 Objective-based Approaches

In objective-based approaches, a specific environmental (i.e., instream) flow is set, and all withdrawals from the river and releases from the reservoirs are regulated in order to meet this environmental flow. Hydrologic or hydraulic methods to measure the environmental quality, as explained in Sections 2.1.1 and 2.1.2, may be applied to set a minimum environmental flow (Tharme 2003). In the South African Building Block (King et al. 2000) and Australian Holistic Approaches (Gippel 2001), for example, ecologically acceptable flows are set through the use of expert opinion. The main problem with these approaches is that in many cases in practice there is no specific threshold beyond which the system actually fails. Fish abundance, for example, may have a linear relationship with the flow rate in the river as explained by Acreman (2005). In this case, there is no threshold for the instream flow which could identify an ecological failure. Also, setting predefined values for environmental flows is not suitable when analyzing trade-offs among multiple attributes, such as environmental

and power generating attributes. In such cases, the degree to which attribute criteria are met should be the measure of success rather than whether or not pre-set values are achieved.

2.2.2 Scenario-based Approaches

In the scenario-based approaches on the other hand, a flow regime is selected among candidate flows based on a comparison of the resulting environmental quality, economical conditions, and other attributes of the various alternatives.

2.2.2.1 Instream Flow Incremental Methodology (IFIM)

IFIM is a decision-making framework that integrates hydrologic, hydraulic, and habitat models (i.e., PHABSIM) to compare the WUA of flow regime alternatives with specified preferred habitat conditions for target species (Bovee et al. 1998; Stalnaker et al. 1995). Calculated WUA for alternative flow regimes are described both in terms of time series and of duration curves. Bovee et al. (1998) demonstrate different phases of IFIM, and how PHABSIM is incorporated in this process. PHABSIM output, i.e., WUA, is the major variable used as input to IFIM. Although other factors, e.g., stream productivity and fish mortality, may influence the fish population, Stalnaker et al. (1995) claim that in most cases, habitat is the most directly quantifiable, and thus preferred factor.

2.2.2.2 Systems Analysis Approaches

Sale et al. (1982) introduce a systems analysis approach to achieve a reservoir operation policy with an optimal instream flow. Their objective function maximizes the minimum downstream habitat value over all life stages of fish and all periods of planning. They employ both Linear Programming (LP) and Non Linear Programming (NLP) to find the optimal alternative for the instream flow. Cardwell et al. (1996) improve the work of Sale et al. (1982) by utilizing a multi-objective optimization model with human (i.e., water supply) and fish (i.e., available habitat) needs, and by introducing habitat metrics that account for ecological issues (e.g., focusing on juvenile fish outmigration). Their objective function includes both maximizing available habitat, and minimizing water supply shortage. Suen et al. (2009) also incorporate a multi-objective approach, and consider both human and

ecosystem objectives. The human objective includes maximization of how well water supply, agriculture, and power needs are met. The fish objective maximizes how close the ecohydrological indicators are to target indicators. They use ecohydrological indicators, as introduced in Section 2.1, instead of habitat metrics, and use fuzzy set concepts in their multi-objective approach.

In the planning stage of water resource projects, where prioritization and relative importance of environmental and other objectives are not agreed upon, participatory processes, e.g., Multi-Criteria Decision Analysis (MCDA), may be incorporated. The appropriateness of MCDA approaches for addressing environmental decision-making problems is discussed by Balasubramaniam and Voulvouis (2005). Others, e.g., Loomis (2000), suggest assigning market values for all non-market attributes (e.g., environmental attributes) in order to be able to make a sound trade-off analysis among all attributes. Due to the barriers and inflexibilities of converting all measures into monetary values, McDaniels (1996) introduces a non-monetary index, i.e., an index of environmental impact, and uses Multi-Attribute Utility Theory developed by Keeney and Raiffa (1976) to undertake a preliminary assessment of electrical utilities alternatives. General steps followed in a MCDA for water resource problems are explained by Hyde et al (2004) as follows:

- Identify final decision-makers, decision-making process participants, and stakeholders;
- Select decision-making criteria
- Define alternatives, i.e., possible water resource scenarios;
- Choose appropriate MCDA techniques;
- Determine the criteria weights;
- Assess the performance values, of each criterion for all alternatives;
- Transform the performance values into commensurable units, if possible;
- Apply the MCDA;
- Perform sensitivity analyses; and
- Make the final decision.

2.3 Environmental Impacts of Extreme Events

The contribution of high flows in creating heterogeneous habitat in rivers under natural flow regimes is widely recognized (Bockelmann et al. 2004; Marmulla 2001; Suen et al. 2009). However, adverse impacts of extreme events on fisheries have not been modeled quantitatively, and data to verify such models are limited to opportunistic samples taken before and after extreme events (Bell et al. 2001; Bischoff and Wolter 2001; Lojkasek et al. 2005; Nislow 2002; Pires et al. 2008). Beck (1996) emphasizes the need for transient short-term models when transient perturbations around an equilibrium are experienced. A high flow release from a reservoir is an example of transient perturbations in an aquatic ecosystem.

Schwartz and Herricks (2005) investigate mesohabitat units that are used by fish as refuge during flood events. They observe fish use of habitat during a bankfull and a one-half bankfull flow. These observations, even though conducted for bankfull and lower floods, may be used in verification of numerical simulation models of fish swimming behavior under flooding conditions.

Hydropeaking (i.e., hourly changes in hydropower release) impacts on the downstream environment have been studied (Bratrich et al. 2004; Cushman 1985; Garcia et al. 2011). Although the rate of change in river discharge is high, the maximum discharge is not an extreme event. For example, Garcia et al. (2011) show that in the case of the Bibio Watershed, the annual mean, maximum hydropeaking, and bankfull flows are 466, 650, and 3880 cms, respectively.

Fish mortality under extreme flow conditions has been studied in the analyses of migration of fish through turbines, spillways, and fish passages. Larinier (2001) reports five to 90% mortality among juvenile salmon passing through Francis turbines, and five to 20% mortality for Kaplan turbines. Sudden change in the pressure, including cavitation, extreme shear stress, i.e., strain rate, and grinding are the main causes of mortality in turbines (Cada et al. 2007). Strain rate in rivers, even during the extreme events, may not reach the level of

495 sec⁻¹ which is introduced by Neitzela et al. (2004) as a threshold beyond which juvenile fish are injured. Grinding and cavitation are specific to turbines, not rivers. Likewise, Larinier (2001) reports injury when fish in a flow with a velocity of 16 m/s pass through a spillway and may collide with solid surfaces.

Numerical Fish Surrogate (NFS) is an agent-based model, developed by U.S. Army Corps of Engineers, that integrates hydraulic models with concepts of cognitive ecology to estimate fish swimming behavior in reservoirs, passages, and rivers. Details of the model, and some applications are found in Goodwin et al. (2007). It is primarily developed for projects with a limited spatial scale and under operational flows (e.g., fish passage), rather than extreme events at the scale of river reach. However, some of the ecohydraulic concepts that are adopted in NFS and in inSTREAM (Railsback et al. 2009), which was introduced in Section 2.1.6.2, are used herein as a basis on which the modeled estimates of impacts of extreme events on fish are developed. These are: fish environmental awareness and behavioral response, as explained herein.

2.3.1 Fish Environmental Awareness

Goodwin et al. (2007) state that fish perceive flow strength and direction, acceleration, and spatial velocity gradients. Also, juvenile salmon are sensitive to pressure. This allows fish to create a hydrodynamic image of their surrounding, and to identify structures in their vicinity. Furthermore, as fish evolved in free-flowing rivers, they learn to evaluate information on the environment beyond their sensory range given the hydraulic patterns in their near-field environment (Goodwin et al. 2007). Likewise, Railsback et al. (2009) consider similar perception ability for fish in estimating food availability and mortality risk in their near environment.

2.3.2 Fish Behavioral Response

Fish respond to their environmental information with behaviors, i.e., movements, which minimize their migration time, bio-energetic cost, and exposure to predators (Goodwin et al. 2007). Such decisions also consider habitat preferences, e.g., food availability (Goodwin et al. 2001; Railsback et al. 2009). Given the environmental information, and hypothesized fish

response, the resultant fish movement is the total of their passive (i.e., due to flow) and volitional (i.e., based on behavioral response) movements (Nestler et al. 2005).

Tiffan et al. (2008) investigated behavioral responses of spawning chum salmon below the Bonneville Dam on the Columbia River, USA, to increased flow velocities using acoustic telemetry and dual-frequency identification sonar. Fish movement towards lower velocities was observed under unsuitably increased velocity conditions (in this case, above 0.8 m/s).

2.4 Risk-based Performance Measures

Ecological risk indicators have been introduced to measure ecological stability. Holling (1973) defines ecological resilience as the ability of an ecological system to absorb disturbance without moving from one equilibrium status to another. Subsequently, several other indicators (e.g., vulnerability, persistence, elasticity, and constancy) have been introduced to describe ecological stability for different ecological systems (Adger 2000; Gallopin 2006; Grimm and Wissel 1997; Gunderson 2000; Pinault and Berthier 2007). Several authors introduce different or even opposite meanings for these indicators, and thus Grimm and Wissel (1997) refer to the ecological stability terminology as Babel. They provide a collection of different terms which have been used to address ecological stability in the literature. This is summarized in Table 2-2 where the first column shows conventional terms, the second column provides a list of invented terms, and the third column show terms that are modified versions of the first column. The numbers in the parentheses denote the number of definitions available for each expression. Grimm and Wissel (1997) also suggest a checklist to follow before developing new indicators or before using the existing ones in order to avoid confusion. This checklist is also provided in Table 2-3. The first column shows the six features of ecological systems, the second column lists questions to be asked about these features, and the third column provides example answers to these questions.

Table 2-2. A list of stability terms to be found in the literature (from Grimm and Wissel 1997)

Conventional terms	Invented terms	Modified terms
Stability (25)	Attractor block	Adjustment [stability]
Persistence (15)	Amplitude (4)	Anthropogenic stability
Constancy (5)	Cyclicity	Biomass stability
Domain of attraction (2)	Damping	c-Stability
Ecological stability (6)	Dynamic boundedness	Connective stability
Elasticity (8)	Dynamic fragility (2)	Cyclical stability
Resilience (17)	Dynamic robustness (3)	D-stability
Resistance (9)	Ecological lability	Essential stability
	Ecosystem health	Functional stability
	Existence	Global stability
	Hysteresis (2)	k-stability
	Inertia (4)	Lagrange stability
	Malleability (2)	Local stability
	Maturity	Mathematical stability
	Mutual invasibility	Multi-stability
	Permanence	Natural stability
	Persistence at fixed densities	Neutral stability
	Persistence in the wide sense	o-stability
	Recurrence	Perceived stability
	Regulation	Practical stability
	Repellor	Qualitative stability
	Resiliency (2)	Relative stability
	Responsiveness	r-stability
	Semi-stable attractor	Resistance stability (2)
	Strictly persistent	Species deletion stability
	Strongly persistent	Structural stability (2)
	Vulnerability (2)	t-stability
	Weakly persistent	Temporal stability
		Terminal stability
		Total stability
		Trajectory stability
		Ultra-stability

Table 2-3. An ecological checklist for employing ecological stability statements (from Grimm and Wissel 1997)

Features of the ecological situation	Checklist question for this feature	Example answers
(1) Level of description	On what level of description is the stability properly examined?	Individual, population, community, ecosystem, landscape, ...
(2) Variable of interest	Which ecological variable of interest is being considered?	Biomass, population size, age structure, nutrient cycling rate, spatial pattern, ...
(3) Reference state or reference dynamic, respectively	What is the reference state or dynamic of the variable of interest without external influence?	Equilibrium, trend, cycles, high or low spatial or temporal variability, ...
(4) Disturbance	What does the disturbance look like?	Disturbance of the state variable or of a system parameter, lasting disturbance or short-term effect, intensity of the disturbance, frequency of the disturbance, ...
(5) Spatial scale	To which spatial scale does the stability statement refer?	Size of the researched area, ability of the researched species to spread, typical lengths in the spatial heterogeneity of the research area, ...
(6) Temporal scale	To which temporal scale does the stability statement refer?	Time horizon of the statement, longevity of the examined organisms, temporal structure in the environmental heterogeneity, ...

Hashimoto et al. (1982) introduce risk-based performance indicators in the water resources engineering context. These indicators are reliability (the probability of a system success), vulnerability (the expected magnitude of a system failure), and resilience (the expected value of lengths of time to recovery). These are statistical indicators which may be derived through simulation processes which use long-term time series of system variables (e.g., river flow rate at a specific point). Maier et al. (2001) develop a method to calculate reliability,

vulnerability, and resilience using the First-Order Reliability Method which may be computationally more efficient than simulation methods.

Holling (1996) refers to engineering resilience as “environmental resilience.” He claims that engineering or environmental resilience, unlike ecological resilience, only allows a system to be at a constant or static point (i.e., the equilibrium), and any change in the state variables of the system is assumed to be a failure. The author disagrees with Holling, because the failure boundary that is used in estimating engineering resilience is not equivalent to the equilibrium or non-disturbed point. Instead, it is a multiple-dimension threshold of all state variables, beyond which the system may be considered to be failed. In other words, the main difference between ecological and engineering resilience is not in viewing the environmental systems as static or constant vs. dynamic or changing. The difference stems from the fact that ecologists are interested in whether a system recovers to its equilibrium status after the external disturbance is removed; while engineers are interested in the recovery rate. Engineering risk indicators in water resource systems are usually calculated for recoverable systems, i.e., if the system fails, it will eventually recover. Thus, the important boundary in the random variable space is between success and failure. From the ecological point of view however, if the system goes beyond the defined boundary, it will move to a new equilibrium and will not return to the primary equilibrium even after removing the disturbance. This difference in points of view may be because of the time-scale difference between ecological and engineering studies. While engineering studies often deal with short-term events, e.g., an extreme flood which lasts a couple of days, ecological studies are interested in long-term changes, e.g., extinction of a specific species which may occur over a century. Engineering risk indicators may be quantified using statistical and probabilistic models where input data (e.g., the time-series of random variables) are available, while ecological performance indicators are often estimated qualitatively or subjectively. See Gunderson et al. (1995) for examples of ecological indicators.

Chapter 3: Research Needs and Proposed Framework

Given the scope and objective of the dissertation, explained in Chapter 1, and the existing background literature which was reviewed in Chapter 2, a modeling framework is proposed in this Chapter. This framework includes several models to estimate immediate and long-term impacts of extreme events on fish population as shown in Figure 1-2.

3.1 Research Needs

The limited quantitative analysis of the immediate effects of extreme events is partly due to lack of sufficient data to develop and validate a quantitative biological model, and partly due to the fact that ecological studies mainly deal with much longer time scales than those of extreme events. In other words, ecological studies mainly focus on long-term steady-state scales in which short-term snapshots of highly transient events may not be captured.

The inSTREAM model, as explained in Section 2.1.6, may best be used to estimate fish population dynamics under normal operating conditions (i.e., before point 1 and after point 3 in Figure 1-2). For example, the time scale and fish mortality mechanisms that inSTREAM employs for normal operations do not apply to the highly transient conditions between point 1 and point 2. Furthermore, due to computational limitations, it is not practical to apply this model between point 2 and point 3 if the fish population at point 2 is not known with certainty. The source code of NFS, reviewed in Section 2.1.6, is not accessible. To the author's knowledge, it has not been applied to simulate fish fate during extreme events at a river reach scale.

As explained in Sections 2.1.3 and 2.2.2.1, although habitat models do not directly consider ecological parameters, in many cases, their outputs (e.g., time series of WUA generated by PHABSIM) may be used as a surrogate for fish population. In the case of an extreme event however, this is not possible, because immediate fish and egg loss, which cumulatively affect the fish population dynamics, are not considered in simulated time series of WUA.

3.2 Proposed Framework

A framework for addressing these research needs is conceptualized by Naghibi et al. (2011), and is illustrated in Figure 3-1. This framework is used to estimate the immediate downstream impacts of extreme events (i.e., between point 1 and point 2 in Figure 1-2), and the recovery trace of fish population to a steady-state condition (i.e., between point 2 and point 3 in Figure 1-2). This framework includes several models for impact estimation, shown in Figure 3-1. These models are nominally categorized as short-term, including hydrotechnical and immediate impact, and long-term impact simulation models. Water quality and non-fish life loss models are not developed in this research, yet may be included in future work, and thus are shown in the framework for completeness.

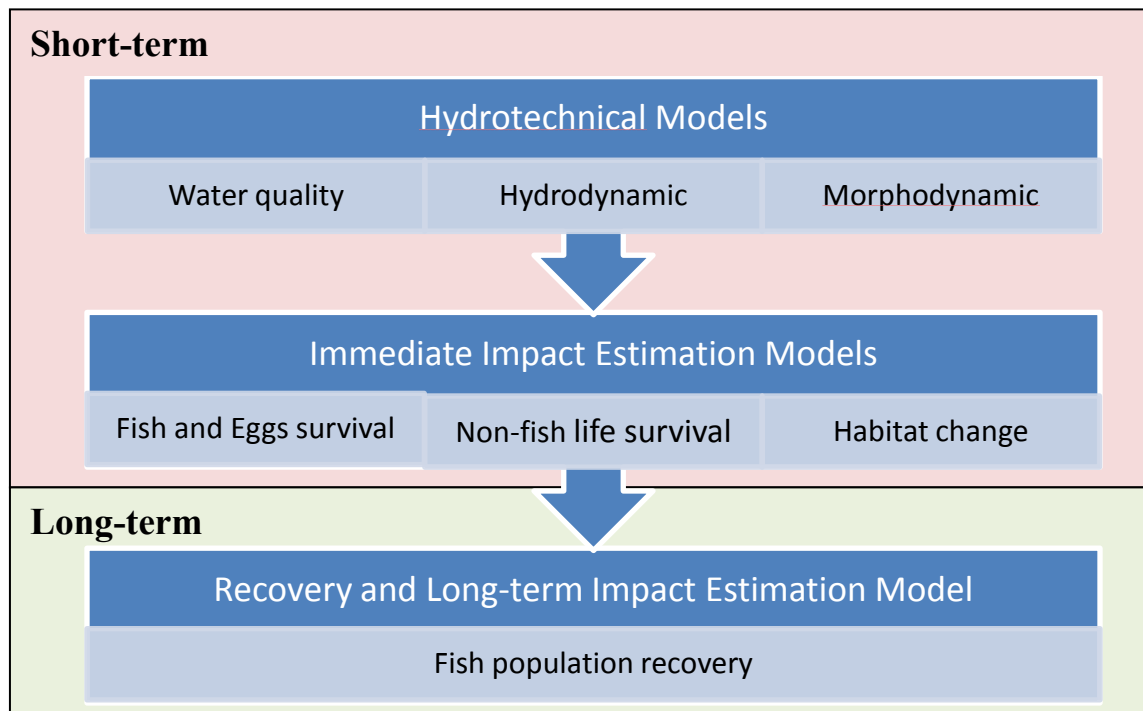


Figure 3-1. Structure of framework for immediate and long-term impact estimation

Although proposed approaches may be generalized for other water resource systems and other environmental attributes, the focus of this thesis is on impacts on anadromous salmon species in rivers downstream of regulated dams.

3.3 Modeling Considerations

Models shown in Figure 3-1 may have different temporal and spatial scales, and may require different modeling approaches. Integration of the models must acknowledge and account for such differences. Sample temporal and spatial scales of modeling units for different models for the Lower Campbell River system are shown in Table 3-1.

Table 3-1. Sample temporal and spatial scales for Lower Campbell River

Models		Temporal scale		Spatial scale	
		Time step	Number of time steps	Spatial units	Number of spatial units
Hydrotechnical*		< 1 sec	$\approx 10^7$	25 m ²	10 ⁴
Immediate fish survival		10 sec	10 ⁴ -10 ⁵	25 m ²	10 ⁴
Fish population recovery	(in river)	5-30 days	10 ³ -10 ⁴	River reach (1.7-km long)	1 ^{**}
	(in ocean)	1 month - 1 year	10 ²	The ocean	1 ^{**}

*: Immediate egg survival and habitat change estimation models use the output of hydrotechnical models, and do not have independent temporal and spatial scales.

**: Fish population recovery model is zero dimensional

3.3.1 Temporal Scale

Hydrotechnical models generally employ variable time steps, i.e., Δt_h , which are less than one second. The time horizon of these models should be on the order of the duration of the extreme event, e.g., a few hours to a few days.

For the immediate fish survival estimation model, the time step, i.e., Δt_f , is a function of the spatial scale and fish swimming capacity (see Sections 4.1 and 4.2 for details). The time horizon of this model is the same as those of the hydrotechnical models.

The immediate egg survival and habitat change estimation models incorporate the outputs of the hydrotechnical models, and do not have independent temporal and spatial scales.

The fish population recovery model utilizes different time steps for different downstream water bodies. Modeling the life stages of fish in the river requires a time step of a few days to one month, while the life stages within the ocean may be modeled with a monthly or yearly time step. The time horizon of the long-term model is on the order of decades.

3.3.2 Spatial Scale

The spatial scale of the models, e.g., the size of the simulation cells, of the hydrotechnical models, should be selected based on numerical stability considerations for these models, and on swimming capacity of fish as explained in Section 4.1. These considerations may result in cell sizes of 1-100 m². Here, the trade-off between computational burden and accurate biological response modeling should be considered in spatial scale selection.

Immediate fish survival, egg survival, and habitat change models should have the same spatial scale as the hydrodynamic and morphodynamic models.

The fish population recovery model considers the entire population at each life stage as a single state variable. These state variables may be located in the river or in the ocean.

3.3.3 Population-based vs. Individual-based Modeling

As mentioned in Section 2.1.6, the choice among population- and individual-based modeling frameworks is case-specific. Immediate impact estimation models in Figure 3-1 must represent a high degree of individual variation. That is, fish survival during the extreme event, for example, is dependent on the location of the fish in the river, and on the hydrodynamics at that location. Therefore, an individual-based model should be developed to simulate the behavior of single fish, and the total number of surviving fish may be determined by integrating the results of all individuals.

For the fish population recovery model, however, all fish at the same life stage may be aggregated and described by system level state variables. The relationships between these variables are also available either through observed statistics (e.g., birth, growth, and death rates) or empirical equations such as the smolt survival function shown in Equation 5-2. This

model is to be used for comparing fish population dynamics with and without an immediate loss of fish and eggs, rather than to provide precise point prediction of the population. Thus, the population-based model may be applied in this case where a large number of fish go through several processes (e.g., migration, growth, and death) over a long period of time (e.g., decades).

3.3.4 PHABSIM-like Method

Despite limitations of habitat modeling in simulating cumulative effects of immediate fish and egg loss on fish population dynamics, a PHABSIM-like method is employed in some models of the proposed framework. For example, probabilistic initial location of fish in the immediate fish survival estimation model is based on the HSI of simulation cells in the river reach. Also, selection of a destination cell by an individual fish at each time step in the immediate fish survival estimation model is based on the suitability of cells. Likewise, a PHABSIM-like approach is incorporated in estimating immediate habitat change due to the extreme event. The PHABSIM-like approach is useful in these cases as it requires only the habitat variables that are provided by the hydrotechnical models.

3.4 Major Modeling Assumptions

The structures of the developed models are described in Chapters 4 and 5. Major assumptions of the models are summarized herein.

3.4.1 Immediate Fish Survival Estimation

Given the literature on cognitive ecology (see Section 2.3), it is assumed that a fish is aware of the hydrodynamic conditions of its neighborhood. Also, as a result of such awareness, a fish will move in the direction with minimum bio-energetic cost. A fish actually undergoes several three-dimensional movements in a given time period, which are chosen to minimize their expended energy. However, in this work, this displacement is approximated with an average two-dimensional movement in the direction of the maximum suitability gradient.

Also, it is assumed that if an adult fish experiences fatigue and is washed away, it will not have sufficient energy to swim back and complete the spawning process. Therefore, the adult

fish may be considered lost. Such a mechanism is assumed to be valid for anadromous species.

3.4.2 Immediate Egg Survival Estimation

It is assumed that eggs are lost only if the depth of scour or deposition is greater than the depth of redds. That is, eggs are considered to survive an extreme event even if all but a thin layer of sediment above them is scoured.

3.4.3 Fish Population Recovery

This model is based on the assumption of stationarity in hydrology, geomorphology, and ecology before point 1 and after point 2, as is depicted in Figure 1-2. In cases of non-stationary hydrology or geomorphology, for example, under a dam breach in which the geomorphology continues to change throughout the fish recovery process, the simplified modeling approach which assumes stationary fish life-cycle loss rates will not be valid. Likewise, if yearly escapement does not follow a stationary pattern, , for instance, if ocean conditions or fishing pressure change survival rates, and hence escapement rates, the closed loop of the fish life cycle, as described in Chapter 5, will not be balanced.

Chapter 4: Short-term Models

In this chapter, the short-term models of environmental impacts are discussed. For hydrotechnical models, existing models are selected. A major contribution of this research includes developing an immediate fish survival estimation model by integrating hydrodynamic and biological considerations in an individual-based model. A second model is developed to estimate immediate egg survival given the results of the hydrotechnical models. Finally, an approach is proposed to estimate the change in fish habitat due to extreme events given the results of hydrotechnical models.

4.1 Hydrotechnical Models

Hydrodynamic and morphodynamic models provide depth, velocity, and sediment scour/fill. Utilizing one-dimensional hydrodynamic models is not recommended because depth, velocity, and sediment scour/fill may vary by orders of magnitude across the width of a river, and the immediate impact estimation models are dependent on these hydrodynamic parameters. Also, employing three-dimensional models, even though more accurate, may not be practical in cases of long river reaches (e.g., a few hundred meters) which are under extreme flooding for long periods of time (e.g., over a few days). Therefore, two-dimensional hydrodynamic and morphodynamic models, e.g., River2D (Steffler and Blackburn 2002) and River2DM (Vasquez et al. 2007), are used in this work to estimate the transient response of the river reach to a given flood hydrograph. While River2D models steady-state and transient flows in fixed-bed channels, and simulates time series of hydraulic parameters (e.g., depth and velocity) at each simulation cell, River2DM also models the scour and deposition of bed sediment throughout the simulation cells.

In the immediate impact estimation models, changes are simulated at the scale of simulation cells, the size of which are chosen based on the fish spawning territory for the species in question. Thus, the hydrodynamic and morphodynamic response models estimate the time series of velocity vector, depth, and sediment scour/fill at each cell of the river reach during the period between t_1 and t_2 in Figure 1-2. The spatial scale is selected based on considering

both numerical stability of the hydrotechnical models and fish movement at each time step. This may be estimated with Equation 4-1.

$$\begin{aligned}\Delta L_{swim} &= \Delta t_f \times v_{sust} \\ \Delta L_{max} &= \Delta t_f \times v_{max}\end{aligned}$$

Equation 4-1

where:

- Δt_f : time step of immediate fish survival estimation model (sec)
- v_{sust} : sustained swimming speed of target species as explained in Section 4.2.1 (m/s)
- v_{max} : maximum flow speed in the river reach during the flood (m/s)
- ΔL_{swim} : distance that fish swim at each time step (m)
- ΔL_{max} : maximum displacement of fish at each time step (m)

ΔL_{swim} must be greater than the size of the simulation cells so that a fish can move to a neighboring cell during a given time step if the conditions at that cell are better than those of its current cell. Also, ΔL_{max} must be small enough (e.g., a few times the size of the cells) so that at each time step only a manageable number of neighboring cells are investigated. This will be explained in detail in Section 4.2.1.

The simulation time step for the hydrotechnical models (i.e., Δt_h) varies during the simulation from a small fraction of a second to a few seconds. However, due to computer memory limitations, the output files of these models are recorded with a longer time step (i.e., Δt_{h-out}). For example, the output files of the hydrodynamic model for one given hydrograph that is used for the case study of the Campbell River in Chapter 6 have a time step of 30 minutes.

4.2 Immediate Impact Estimation Models

The immediate fish survival, egg survival, and habitat change estimation models, which are developed and applied in this research, are described in Sections 4.2.1, 4.2.2, and 4.2.3.

4.2.1 Immediate Fish Survival Estimation Model

An individual-based model is developed by integrating the output of a hydrodynamic model (i.e., time series of depth and velocity at all simulation cells) with the hypothesized behavior of fish under extreme events. The hypothesized fish behavior, as explained in Section 4.2.1.2, is closely related to the fish swimming capacity. Existing knowledge of fish swimming capacity primarily relies on studies undertaken by Beamish (1978) who conducted and integrated several experiments and studies about swimming capacity of different fish species. He defines the swimming capacities as: *sustained swimming speed* at which fish can swim without fatigue; *prolonged swimming speed* at which fish can swim for up to 200 minutes and then experience fatigue; and *burst swimming speed* at which fish can only swim for up to 20 seconds.

Since there are a large number of cells within a reach, all potential initial locations of fish before the extreme event occurs are identified first. Given the swimming capacity of the fish species, the location of each fish at each time step is then traced as the fish moves spatially throughout the reach, in response to the depth and velocity time series produced by the hydrodynamic model. At each time step, the fish in a cell compares the HSI of its current cell with those of all neighboring cells, shown in Figure 4-1, and moves to the cell with the highest HSI. In the case of an extreme event where velocities are high, it is logical to assume that velocity and depth suitability are the most significant bases on which a fish decides to choose its habitat. In such circumstances, other parameters, e.g., substrate and predation, are likely not as important in habitat selection. If all adjacent cells have zero values of HSI, the fish moves to the one with the lowest velocity. If movement to the selected cell is not possible because the water speed is greater than fish swimming capacity, the fish will be washed with the flow during that time step. A lookup table is then generated with survival or loss values for each initial location of fish. Finally, the survival rate of the fish population is estimated through a sampling process that assigns the initial location of n_f fish to n_f of cells, and by using the generated lookup tables, evaluates the number of surviving and lost fish. This sampling process is explained in Section 4.2.1.3.

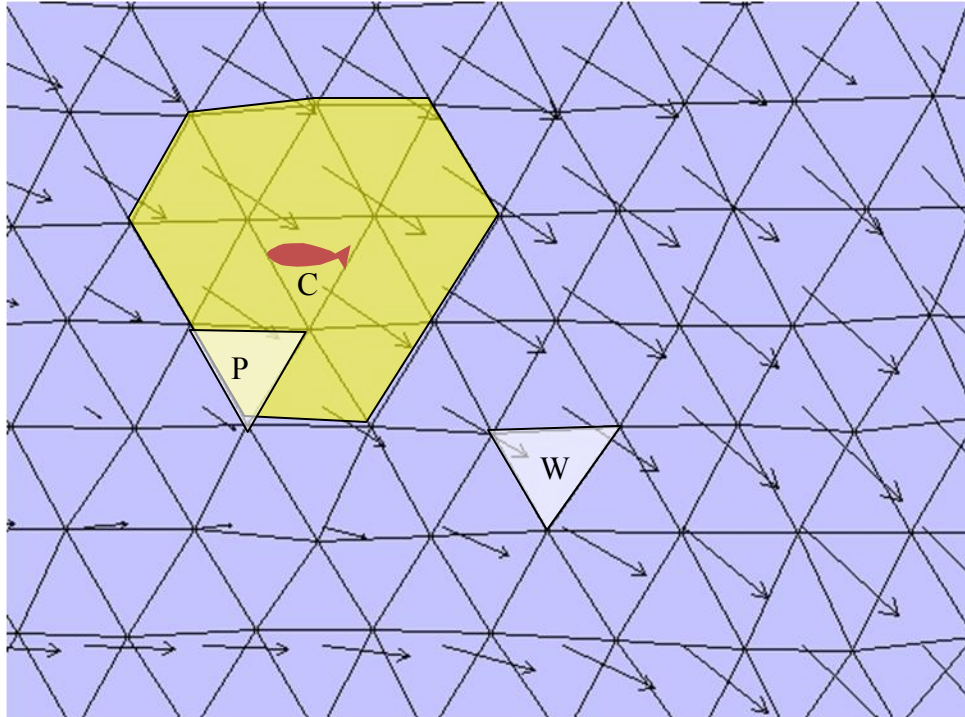


Figure 4-1. Immediate fish survival simulation cells, including current cell (C), preferred neighboring cell (P), and washed cell (W)

A modular model structure is designed in order to support a flexible (i.e., easily adaptable) and computationally efficient framework. As shown in Figure 4-2, a pre-processor, a main program, and a post-processor are developed in the immediate fish survival estimation model. The relationship among these modules and the hydrodynamic model is demonstrated in Figure 4-2, and explained as follows.

4.2.1.1 Pre-Processor

The time series of output from the hydrodynamic model include the matrices of depth (i.e., d_t^i) and specific discharge in both x and y directions (i.e., qx_t^i and qy_t^i) for all i cells in the river reach and time t , with a time step of Δt_{h-out} , and the x and y coordinates of all cells. In the pre-processor, these data are imported from the hydrodynamic model output files. Other inputs of the preprocessor include the time-independent swimming capacity and HSC for the target species. As an illustration, sample spawning HSC for chinook salmon are shown in Figure 6-7.

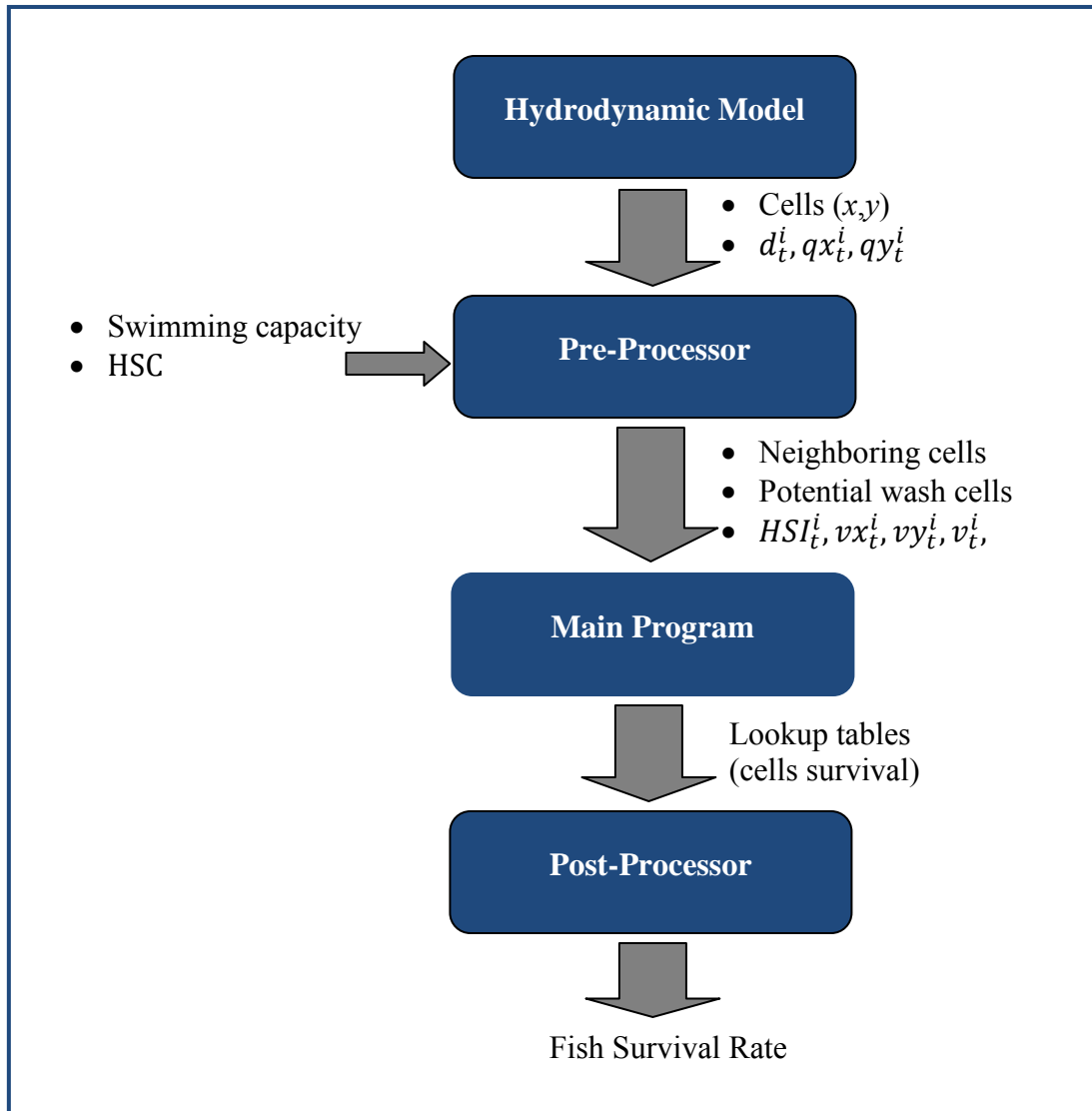


Figure 4-2. Immediate fish survival estimation model

The first task of the pre-processor module is to create a matrix in which the distance between any two cells in the river reach is calculated, given their x and y coordinates. This matrix, along with Equation 4-1, is used to find two sets of data. These are:

- Neighboring cells: all potential destination cells to which a fish may swim, for each cells in the river reach at each time step, i.e., Δt_f , and
- Potential Washed Cells: all potential destination cells to which a fish may be washed by high flows, for each cell in the river reach at each time step, i.e., Δt_f .

In the case study of the Campbell River, discussed in Chapter 6, different cells in the river reach have a few neighboring cells and up to 150 potential washed cells.

The next step is to create matrices of vx_t^i and vy_t^i using Equation 4-2. Since a cell is considered to be dry if only subsurface flow exists (i.e., non-positive d_t^i), flow velocities for these cells are assigned zero values.

$$vx_t^i = \begin{cases} \frac{qx_t^i}{d_t^i} & \text{if } d_t^i > 0 \\ 0 & \text{if } d_t^i \leq 0 \end{cases}, \quad vy_t^i = \begin{cases} \frac{qy_t^i}{d_t^i} & \text{if } d_t^i > 0 \\ 0 & \text{if } d_t^i \leq 0 \end{cases}$$

Equation 4-2

Also, a matrix of the velocity magnitude at all cells and time steps (i.e., v_t^i), is created based on Equation 4-3.

$$v_t^i = \sqrt{(vx_t^i)^2 + (vy_t^i)^2}$$

Equation 4-3

Given the depth and velocity magnitude matrices (i.e., d_t^i and v_t^i), and the habitat suitability criteria, a matrix of HSI is developed for all cells with a time step of Δt_{h-out} (i.e., HSI_t^i). Finally, all cells with positive pre-flood HSI are identified as potential initial locations for fish.

4.2.1.2 Main Program

Given the results of the pre-processing module, the main program traces the position of individual fish, which are initially located at all potential initial locations as identified in the preprocessor. Fish movement is simulated in time steps of Δt_f during the extreme event, that is, the time between point 1 and 2 in Figure 1-2, which is denoted as T_{ee} . This simulation process is summarized in Figure 4-3.

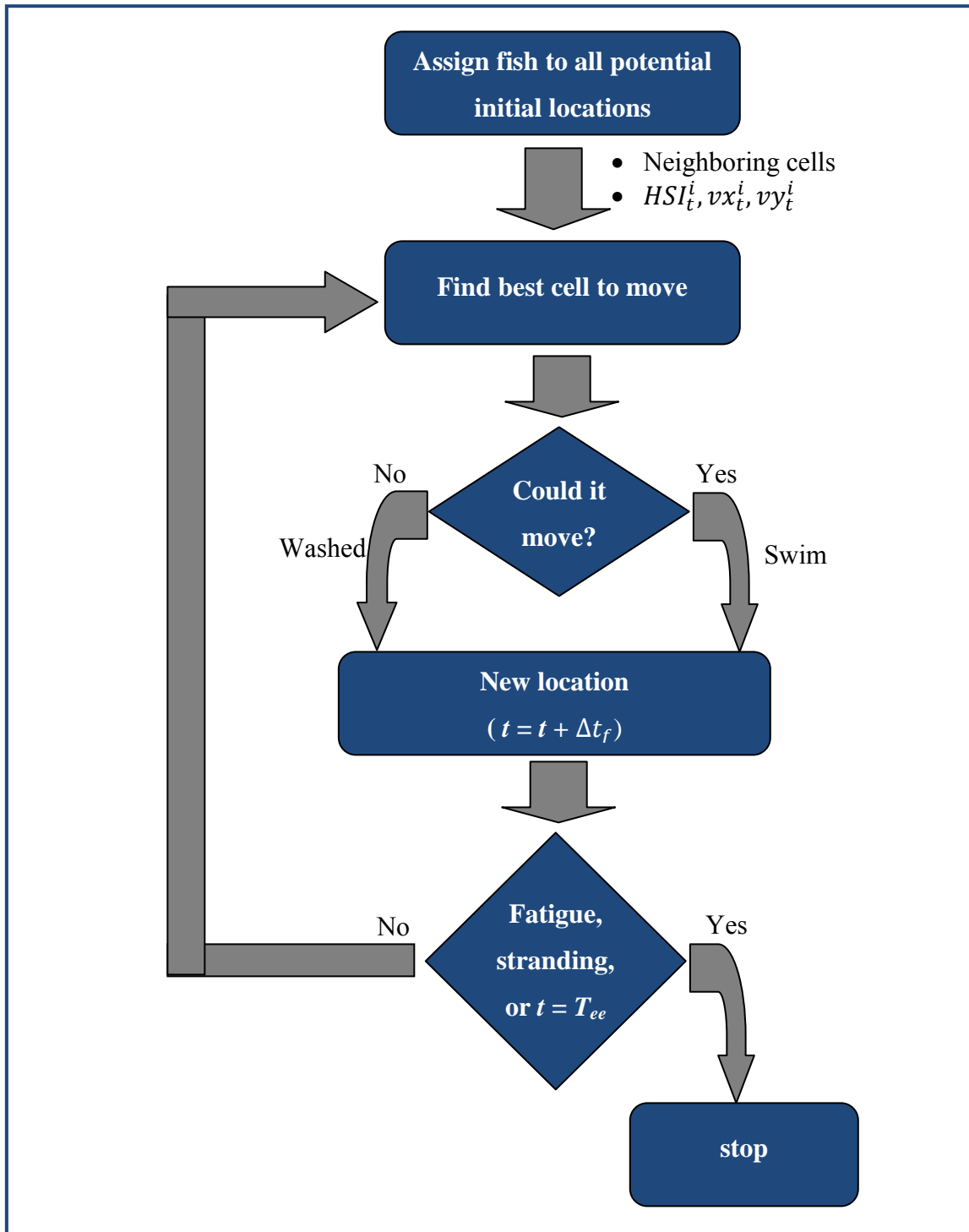


Figure 4-3. Main program module in the immediate fish survival estimation model

For each fish and at each time step, d_t^i , vx_t^i , vy_t^i , v_t^i , and HSI_t^i are calculated for all neighboring cells of its current location. This is accomplished by interpolating the data in the matrices of d_t^i , vx_t^i , vy_t^i , v_t^i , and HSI_t^i that are generated by the pre-processor. The interpolation is required, because as mentioned in Section 4.1, hydrodynamic output files are recorded with longer time steps than those of the immediate fish survival estimation model. The HSI_t^i of the current location is compared with those of all neighboring cells. The cell with the highest HSI_t^i is the preferred destination to which the fish moves in this time step. However, in the case of extreme events, it is expected that, due to very high flow velocities at many locations and times, both the current and neighboring cells have a zero-valued HSI_t^i . In such cases, the cell with the lowest v_t^i is the preferred destination to which the fish moves in this time step, i.e., in this case, the fish looks for refuge.

The fish will try swimming to the preferred destination at its sustained swimming speed. If this is not possible because of a high opposing flow velocity, the fish will try prolonged, and then burst swimming speeds. If the opposing flow velocity is greater than these speeds, the fish is considered to be washed during this time step with a velocity of wash relative to its swimming speed. In any case, (i.e., either if the fish can successfully swim to its preferred cell, or if it is washed during the time step) the new location of the fish is recorded. Throughout the simulation, the total sustained, prolonged, and sustained swimming time of each fish is stored and tracked. This is undertaken as a check to whether the fish experiences fatigue during the extreme event or not. If the fish experiences fatigue, it needs a few hours before it recovers energy to swim (Beamish 1978). By that time the fish is washed away for kilometers from its original location. In the case of an adult spawning fish, at this point, it is exhausted and may not be able to swim back to its spawning habitat (Ewart and Anderson 2010), and in case of juvenile fish, it may also be considered dead because it is out of its rearing habitat.

Also, at each time step, each individual fish is checked for stranding, i.e., whether it is trapping in dried cells with no connection to the rest of the river reach. This normally happens during the recession period of the extreme event.

Since the swimming capacity of target species are not known with certainty, the main program may be implemented for a number of possible fish swimming capacities (n_{sc}). Therefore, the output of the main program for a given flood may be a lookup table with $n_{sc}+2$ columns. The first column includes the identification number of all cells with initial positive HSIs, the second column provides the pre-flood (i.e., initial) values of HSI. Each of the next n_{sc} columns show whether a fish with a given swimming capacity will survive the extreme event given this initial cell location

4.2.1.3 Post-Processor

The initial location of fish in the river reach before the extreme event, and the swimming capacity of target species are not known with certainty. Therefore a simulation approach is employed in the post-processor module that includes a large number of samples to account for such uncertainties.

For each sample, the locations of n_f adult fish are assigned to n_f cells with positive pre-event HSIs, using a discrete probability distribution in which the probability of a cell containing a fish is equal to its HSI. Also, swimming capacities are sampled from a finite discrete probability distribution (e.g., a binomial or a discrete uniform distribution) with the same possible swimming capacities as considered in the main program. Survival or loss of each n_f fish in the sample is examined based on its corresponding lookup table. The survival rate of the sample is calculated by

$$\text{Sample Fish Survival Rate} = \frac{n_{sf}}{n_f}$$

Equation 4-4

where:

- n_f : number of fish in sample
- n_{sf} : number of survived fish in sample

By drawing a large number of samples, and recording their sample fish survival rate, the expected fish survival rate may be calculated, and the histogram or probability distribution of the survival rate may be determined.

4.2.2 Immediate Egg Survival Estimation Model

During an extreme event, each simulation cell of the river reach may experience sediment scour, fill, or both. This process may result in loss of eggs as shown in Figure 4-4. If the depth of scour (D_{scour}) is greater than the depth of the redd (D_{redds}), the redd will be scoured, and the eggs may be considered lost (Glawdel et al. 2011; Lapointe et al. 2000). Also, Glawdel et al. (2011) and May et al. (2009) assume that if the depth of deposition ($D_{deposition}$) is greater than the depth of the redd (D_{redds}), the redd will be buried at a depth of more than twice of its original depth. In this case, eggs may also be considered lost either due to suffocation, or because of their inability to swim out of the gravel bed. This is summarized in Equation 4-5.

$$if (D_{scour} \geq D_{redds}) or (D_{deposition} \geq D_{redds}) \rightarrow Eggs are lost$$

Equation 4-5

where:

D_{scour}	:	maximum depth of scour during a flood (m)
$D_{deposition}$:	final depth of deposition (i.e., fill) during a flood (m)
D_{redds}	:	depth of redds (m)

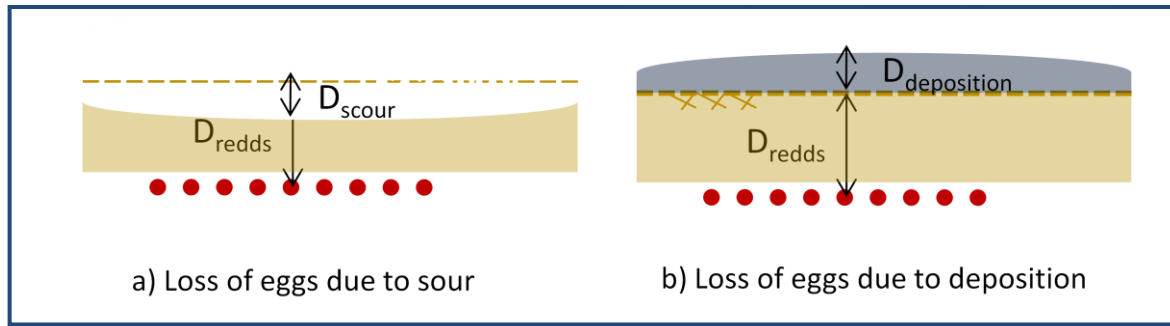


Figure 4-4. Egg loss due to scour or deposition (from Glawdel et al. 2011)

The process of immediate egg survival estimation is demonstrated in Figure 4-5. For each flood hydrograph, transient hydrodynamic and morphodynamic models (e.g., River2D and River2DM) are executed for the extent of the reach. As an output of the morphodynamic model, the amounts of scour and deposition at all i potential spawning cells during a flood are determined (i.e., D_{scour}^i and $D_{deposition}^i$). Then Equation 4-5 is applied to determine if eggs at each spawning cell are lost assuming a value for D_{redds} . D_{redds} could be either a single deterministic value, or a large number of random numbers generated by a sampling tool.

If D_{redds} is selected by a random number generator (i.e., a sampling approach), for each sample, Equation 4-2 may be evaluated to determine whether eggs in every instream spawning cell will survive an extreme event. The egg survival rate for all spawning cells of the sample may then be calculated by Equation 4-6.

$$\text{Sample Eggs Survival Rate} = \frac{n_{se}}{n_e}$$

Equation 4-6

where:

- n_e : number of spawning cells
- n_{se} : number of spawning cells in which eggs survive

By conducting a large number of samplings, and recording their sample egg survival rate, the expected egg survival rate may be calculated, and the histogram or probability distribution of survival rate may be reported. Example output of the immediate egg survival estimation model may be seen in Figure 6-17.

If a deterministic value of D_{redds} is considered, only one sample will be generated, in which Equation 4-6 determines the expected egg survival rate.

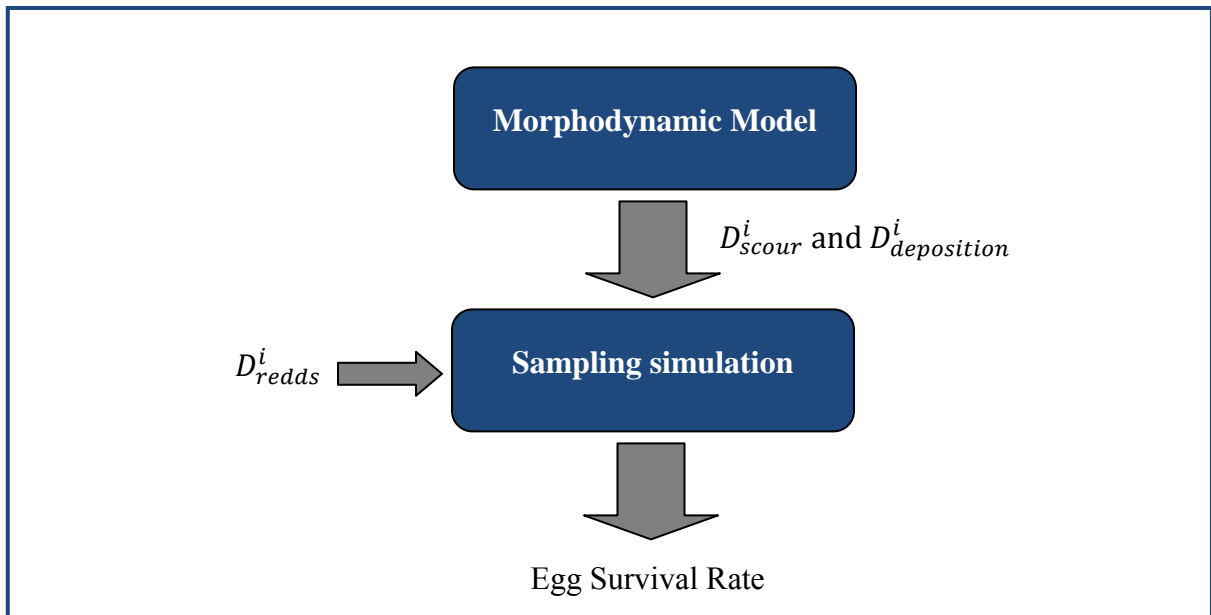


Figure 4-5. Immediate egg survival estimation model

4.2.3 Immediate Habitat Change Estimation Model

During an extreme event, i.e., between point 1 and point 2 in Figure 1-2, the geomorphology of a river reach may be modified. Such a modification may result in a change in the total available spawning and rearing habitat in the river reach. Despite the criticism against the validity of spawning and rearing HSIs (see Section 2.1.3), they are the only available

measure with which to compare the pre- and post- event habitat conditions. Figure 4-6 demonstrates the proposed process for conducting such a comparison. First, a hydrodynamic model is executed throughout the river reach given the base flow of the river (i.e., the flow before the extreme event), and the geomorphology of the river before the extreme event. Using spawning and rearing HSIs for the species of interest, the WUA for spawning and rearing habitat in the river reach may be calculated in the same manner as in the physical habitat estimation models, e.g., PHABSIM.

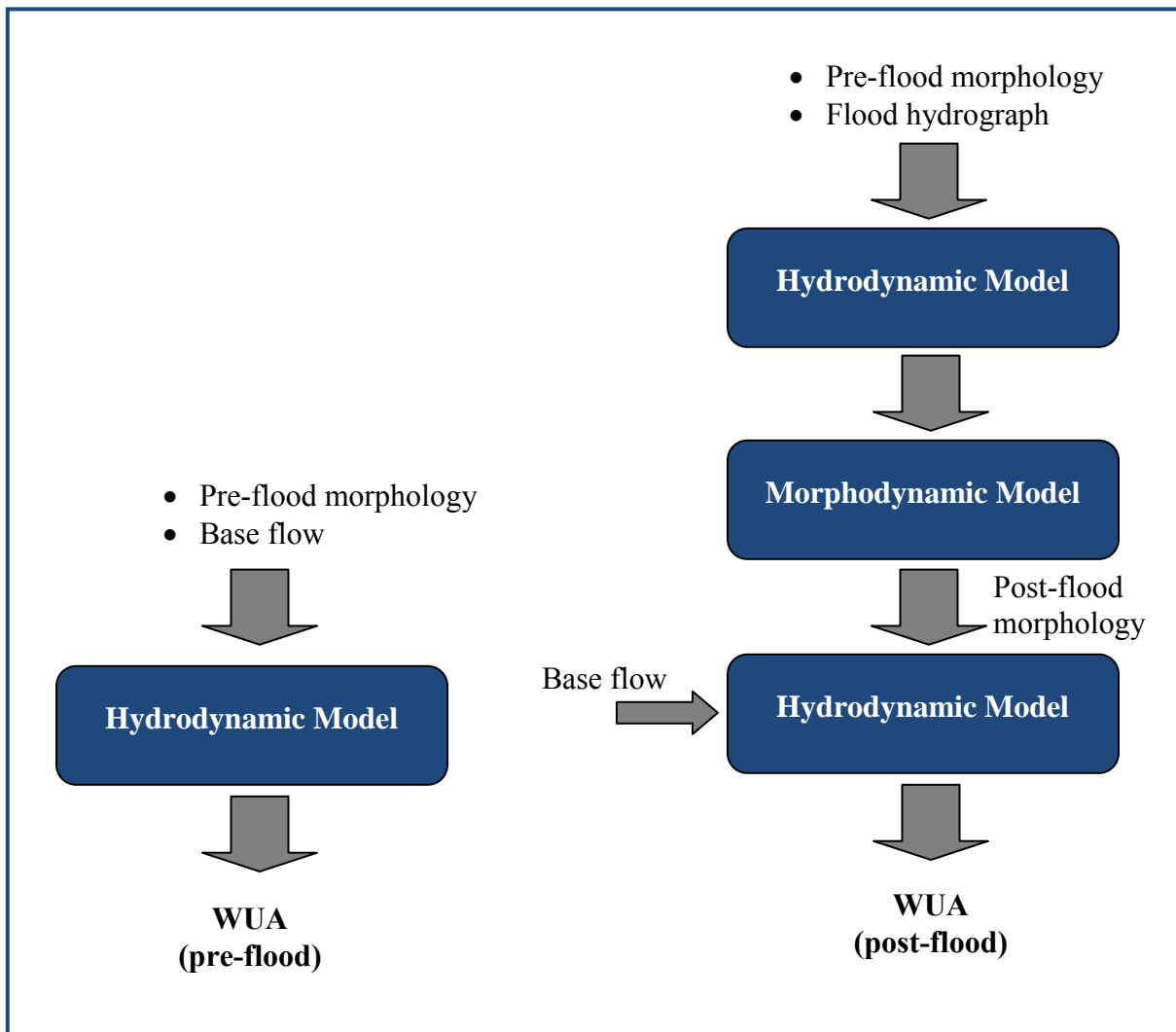


Figure 4-6. Immediate habitat change estimation

For determining the post-event WUA, transient hydrodynamic and morphodynamic models are executed in the river reach. The output of the morphodynamic model includes the post-

event geomorphology of the reach. By implementing a hydrodynamic model for the base flow of the river, given the post-event geomorphology, the post-event WUA for spawning and rearing habitat in the river reach may be calculated.

By comparing pre- and post-event WUA for the spawning and rearing habitat, one may infer whether the spawning or rearing capacity of the river reach is affected by the extreme event.

Chapter 5: Fish Population Recovery Model and Risk-based Performance Measures

Given the immediate changes due to extreme events (i.e., the output of models developed in Chapter 3), the fish population recovery model developed herein estimates the impacts of such immediate changes on the long-term population dynamics. Results are reported as risk-based performance measures for the fish (i.e., escapement) population including the expected value of fish loss, the expected time before the system recovers to equilibrium, and whether or not the fish population returns to its pre-event equilibrium.

Individual-based inSTREAM and population-based SALMOD models, as explained in Section 2.1.6, although very powerful in meeting their objectives, may not be incorporated here. The main reasons are:

- Both models are most applicable to river life stages, and for within-year simulations of salmonids (e.g., smolts downstream migration);
- In order to provide accurate population predictions, both models are computationally intensive. In this research, to estimate the risk-based performance measures, a large number of simulations (e.g., 10,000 simulations) for several extreme events occurring at different dates are required for a long time horizon (e.g., 100 years). Therefore, employing these models for this research is computationally impractical;
- The objective of this research is to compare population dynamics scenarios with and without a sudden loss, rather than accurately predict the population dynamics.

Therefore, a population-based model which considers both river and ocean life stages is developed herein to estimate fish population recovery.

5.1 Fish Population Recovery Model

The fish population recovery model aggregates the number of fish at the same life stage as a state variable (SV). The SVs considered in this model, which are aggregated at the river and ocean scale as shown in Figure 5-1, are: *spawning fish*, *eggs* including *alevins*, *fry*, *smolts*, and *fish in ocean*. The interrelationship among these variables is demonstrated in Figure 5-2, and may be formulated as a continuity equation (Equation 5-1).

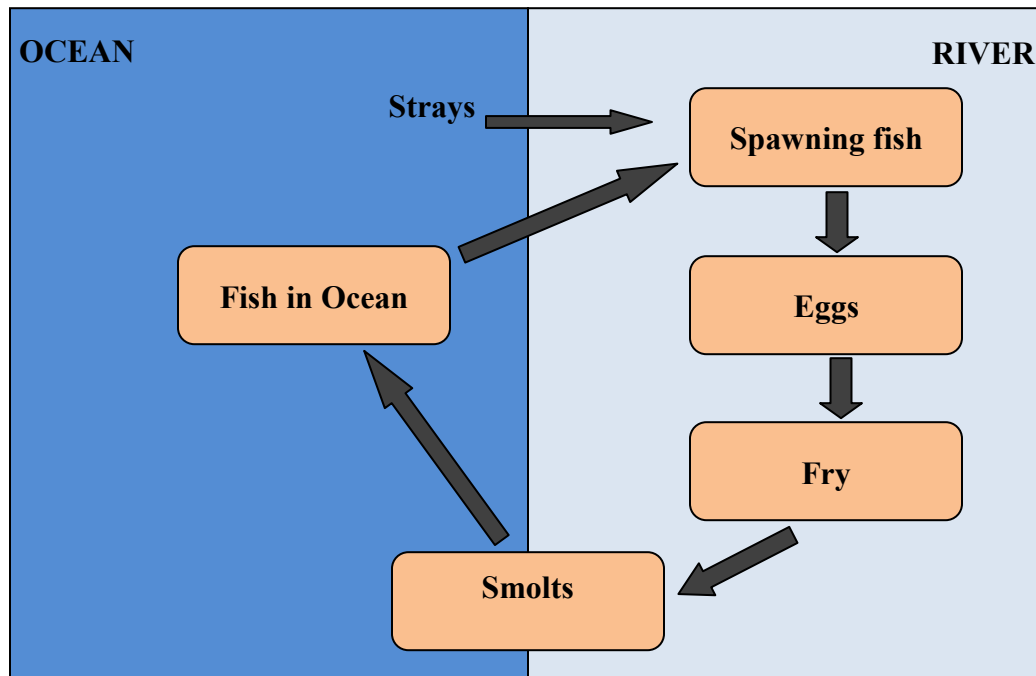


Figure 5-1. SVs in fish population recovery model

$$SV_{t+1} = SV_t + in_t - out_t - nloss_t - iloss_t$$

Equation 5-1

where:

SV_{t+1}	:	value of state variable at time $t+\Delta t_l$
SV_t	:	value of state variable at time t
in_t	:	input from preceding state variable between time t and time $t+\Delta t_l$
out_t	:	output to proceeding state variable between time t and time $t+\Delta t_l$
$nloss_t$:	normal loss of state variable between time t and time $t+\Delta t_l$
$iloss_t$:	immediate loss of state variable due to the extreme event between time t and time $t+\Delta t_l$

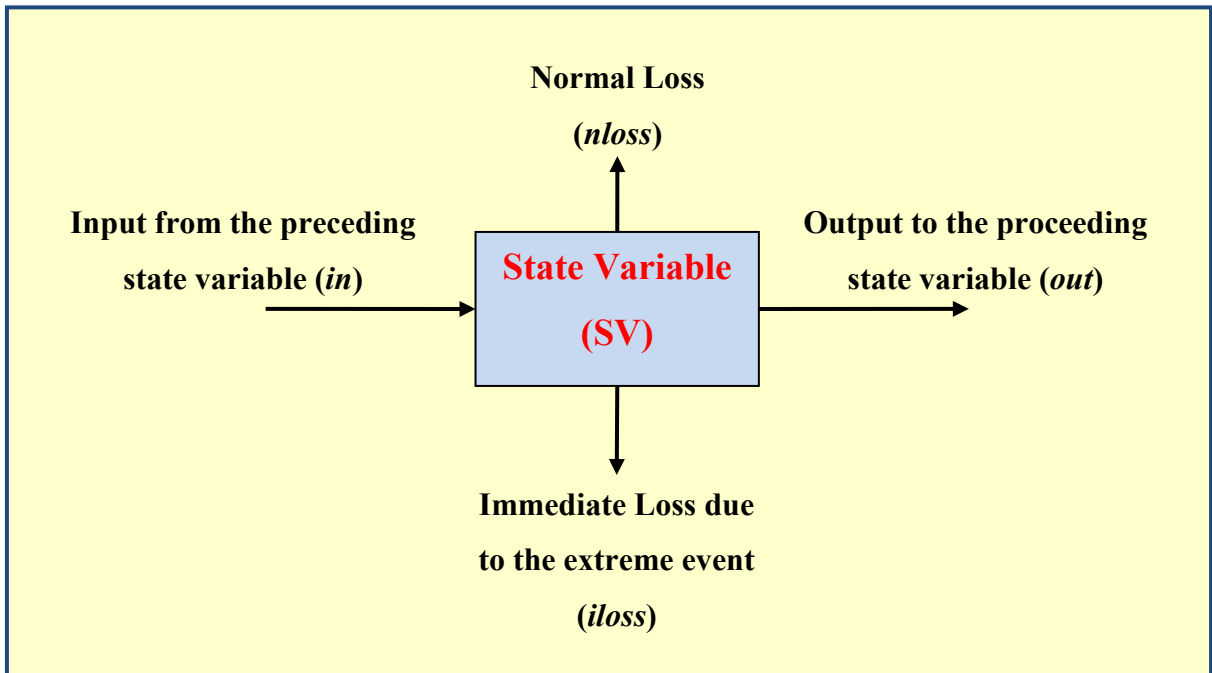


Figure 5-2. Relationship between SVs

The value of an SV (e.g., number of eggs) at time $t+\Delta t_l$ is equal to its value at time t , plus the accumulation of stock from the preceding SV between t and $t+\Delta t_l$ (i.e., in_t), minus any reductions of from the SV between t and $t+\Delta t_l$. These reductions may include normal loss of

the SV (i.e., $nloss_t$), immediate loss of the SV due to the extreme event (i.e., $iloss_t$), and the output of the SV to its proceeding SV (i.e., out_t). The values of in_t and out_t represent the growth of fish from one SV to another (e.g., eggs being hatched). Also, the values of the SVs are controlled with maximum values that represent carrying capacity. For example, the maximum number of eggs in the river reach is controlled by the total available spawning habitat. If, due to an extreme event, the available spawning habitat is reduced dramatically, the fish population will decrease even if a large number of adult fish are present to spawn in the river.

Similar to Ford (1999), constant rates for normal loss are assumed for all life stages except for smolts. The value of $nloss$ at the smolt life stage is considered to be density dependent, and based on Equation 5-2 (Ford 1999).

$$surviving\ smolt = \frac{fry}{\frac{fry}{CC} + \frac{1}{S}}$$

Equation 5-2

where:

- fry : number of emergent fry
- CC : smolts carrying capacity of the aquatic system
- S : smolt survival rate at very low density

Examples of the parameters employed in the model are provided for the case study in Chapter 6.

Thus, the normal loss rate for smolts, i.e., $nloss(smolt)$, may be derived by Equation 5-3.

$$nloss(smolt) = 1 - \frac{surviving\ smolt}{fry}$$

Equation 5-3

5.1.1 Time Scale Considerations

For simplicity, it is assumed that the extreme event has immediate impacts only on the river SVs (i.e., $iloss$ is not considered for ocean SVs). This assumption is justifiable, because high flow velocities due to extreme events dissipate as a river enters the ocean. Therefore, the simulation time step (i.e., Δt_l) for river SVs must be short enough (e.g., a few days) to capture immediate loss. Thus, the duration of the extreme event (i.e., T_{ee}) is selected as the time step of these SVs. Ocean SVs may have longer time steps (e.g., year-long time steps), because statistical data for their normal loss rates are generally available on a yearly, or even longer, basis. Selected simulation time steps for different SVs are summarized in Table 5-1.

Table 5-1. Simulation time steps for SVs, and the period in which they have positive values

SV		Simulation time step (Δt_l)	Positive-valued period
Spawning fish		T_{ee}	Spawning period
Eggs		T_{ee}	Incubation period
Fry	Stream type	T_{ee}	Until grown to smolts
	Ocean type	Month	Until grown to smolts
Smolts	Stream type	T_{ee}	Until grown to one-year olds
	Ocean type	Month	Until grow to one-year olds
Fish in ocean		Year	Year-round

Also, it should be noted that not all SVs are positive valued at all times of the year. Spawning fish are present in the river reach only during the spawning period. During this period, a short simulation time step (e.g., a few days) is utilized for this SV so that the immediate impacts of the extreme event are captured. For all other times of the year, zero values are assigned to this SV, and no simulation is required for it. The periods in which different SVs have positive values are shown in Table 5-1.

Eggs are also present during the incubation period, i.e., from the beginning of the spawning period until all fry emerge. Again, a short simulation time step is used in this period, and for the remainder of the year, zero values are assigned to this SV.

Fry and smolts that are reared in the river (i.e., stream type) and those which are reared in the ocean (i.e., ocean type) are modeled differently. Juvenile fish in the river are vulnerable to extreme floods, and must be simulated based on the same temporal scale as spawning fish and eggs. Ocean type juveniles, however, may not significantly be affected by extreme events, and may be simulated based on a longer time step. For example, a monthly time step may be employed to interact with other SVs with both shorter (e.g., for eggs in spawning beds) and longer (e.g. for fish in ocean) time steps.

Fish are present year-round in the ocean. The population of this SV may be subdivided further to SVs of fish at different ages (e.g., one-year, two-year, and three-year olds). If this approach is selected, then one may simulate adult returning fish of different ages (e.g., four- and five-year olds) migrating back to the river to spawn. This approach is employed in this research.

Depending on available data, strays may also be accounted for as a constant or variable input for spawning fish. Usually, stray data are available in the form of estimated percentage of yearly escapement. However, this would imply that the origin river system of the strays experiences the same extreme event and immediate impact as the modeled river reach.

Therefore, the fish population recovery model utilizes historical absolute number of strays. The model could easily be modified so that stray data are incorporated in the form of percentage of escapement.

Movement (i.e., growth) from one life stage, or SV, to another is assumed to follow the available empirical data (see, e.g., Figure 6-5), but the values of SVs are updated only at the end of each time step. For example, empirical data for a specific species may suggest that smolts leave the estuary for the ocean from the beginning of May to the end of June, and that their monthly rate of movement is uniform. In this case, if smolts are simulated monthly, half

of the smolts population is assumed to leave (be subtracted from) this SV at the end of May, and the other half will leave at the end of June. However, if the one-year-old fish in the ocean are simulated yearly, the corresponding SV for this life stage is updated only at the end of the year.

5.1.2 Data Uncertainty

Uncertainty in the initial location of fish, in fish swimming capacity, and in D_{redds} , results in uncertain immediate fish and egg survival. Also, the value of the carrying capacity is not always constant; rather, it varies with the WUA time series, which is a function of the flow time series. Likewise, the statistical data for $nloss$ are usually uncertain. The fish population recovery model may utilize uncertain data for immediate fish and egg loss (i.e., the distribution of $iloss$) and for ecological parameters, including $nloss$ and carrying capacity. The model may be applied using three approaches for addressing such uncertainty: These are nominally referred to as *deterministic*, *pseudo-probabilistic*, and *probabilistic* approaches in this work, as described herein.

- *Deterministic*: In this approach, the mean value of the immediate fish and egg loss rates (i.e., output of the immediate impact estimation models) are used for $iloss$. Ecological parameters are also estimated as deterministic values based on available empirical data. The output of the fish population recovery model for a given date and flood intensity is a single population time series similar to Figure 1-2.
- *Pseudo-probabilistic*: Here, a large number of $iloss$ samples are imported from the output of immediate fish and egg loss estimation models. However, ecological parameters are considered deterministic similarly to the deterministic approach. For a given flood date and intensity, the fish population recovery model may be applied for a large number of samples (n_{samp}), and n_{samp} population time series are then generated.
- *Probabilistic*: In this approach, both $iloss$ and ecological parameters are considered uncertain. A large number of $iloss$ samples are imported from the output of the immediate fish and egg survival estimation models. Moreover, in each simulation year, ecological parameters are sampled from presumed probability distributions based on available empirical data for such parameters. Similar to the pseudo-

probabilistic approach, fish population recovery modeling in this approach simulates n_{smp} population time series for each flood date and intensity.

Sample simulated population time series using these three approaches are provided in Figures 6-21, 6-22, and 6-23.

5.2 Risk-based Performance Measures Estimation

Vulnerability, engineering resilience, and ecological resilience as explained in Section 1.3, not only capture both short- and long-term performance of environmental systems under extreme event conditions, but also address both engineering and ecological aspects of system performance as explained in Section 2.4.

In order to estimate the risk-based performance measures, the population time series of target species at a specific life stage is simulated over a period of decades. For this purpose, for example, the yearly fish escapements, i.e., the number of adult fish that return back to the river each year to spawn, is simulated using the fish population recovery model introduced in Section 5.1. These simulations are conducted for different scenarios of flood intensity and date of occurrence, with all three approaches for addressing data uncertainty as explained in Section 5.1.2.

In the deterministic approach, risk-based performance measures may be directly calculated from the generated population time series and describe the impacts of an extreme event as shown in Figure 1-2.

If the pseudo-probabilistic or probabilistic approach is applied, the risk-based performance measures are the mean value of these measures for the n_{smp} generated samples of population time series.

In the probabilistic approach, inter-year variations in fish population may be so significant that points 2 and 3 in Figure 1-2 are not visually identifiable, and in order to determine them, one would need to undertake the following procedure. Here, the estimated time series of the

population without the extreme event would need to be compared with every simulation of fish population with the extreme event. An example of this comparison is demonstrated with Figure 5-3. Again, the extreme event occurs at point 1. Point 2, the location of maximum impact of the extreme event, is located at the time where the maximum difference between the two time series occurs. The population that occurs at this same time, but in the time series without the extreme event, is denoted as point 2*. If the system is ecologically resilient, and thus returns to pre-event conditions, point 3 may then be located at the point after which the two time series match.

If the deterministic or pseudo-probabilistic approach is employed, the population without the extreme event represents a constant time series of non-varying data as shown as a horizontal line in Figure 1-2. That is, the fish population at points 1 and 2* would be equal for these cases.

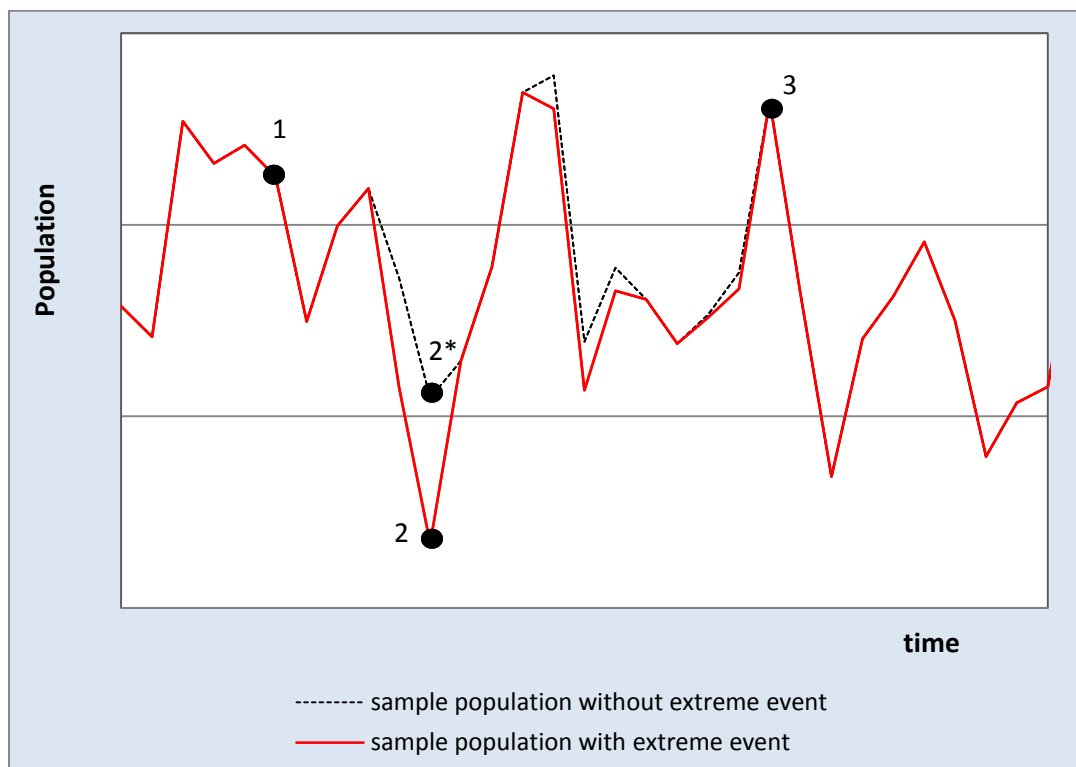


Figure 5-3. Conceptual population time series with and without extreme event when natural inter-year variations are significant under the probabilistic approach

5.2.1 Vulnerability

Vulnerability is the expected value of the ratio of maximum population loss to average pre-event population. Maximum population loss is the maximum difference between the two time series of fish population, with and without the extreme event. That is, the difference between the population at points 2 and 1 (in Figure 1-2) under the deterministic and pseudo-probabilistic approaches, or between the population at point 2 and 2* (in Figure 5-3) under the probabilistic approach. This is formulized in Equation 5-4.

$$Vulnerability = E \left[\frac{\Delta pop_{max}}{pop_1} \right]$$

Equation 5-4

where:

$E[x]$: expected value of x

Δpop_{max} : maximum difference between the two time series of fish population, with and without the extreme event. See Figure 1-2 and Figure 5-3, for the case of the deterministic or pseudo-probabilistic, and probabilistic approaches, respectively.

pop_1 : population at point 1 in Figure 1-2

Therefore, vulnerability is a dimensionless value which ranges from zero to one, and is desired to be minimized.

5.2.2 Engineering Resilience

Engineering resilience is the expected value of the recovery speed. That is the inverse value of the recovery time, i.e., the time difference between points 1 and 3 in Figure 1-2. This is formulized in Equation 5-5.

$$Engineering\ Resilience = E \left[\frac{1}{t_3 - t_1} \right]$$

Equation 5-5

where:

- $E[x]$: expected value of x
 t_1 : end time of extreme event in Figure 1-2 (yrs)
 t_3 : the time at which system reaches an equilibrium in Figure 1-2 (yrs)

Thus, engineering resilience has a unit of year⁻¹. Since the time series of population is recorded yearly, if a population loss occurs, the minimum and maximum recovery times are one year and infinity, respectively. Therefore, engineering resilience may vary from zero to one, and is desired to be maximized.

5.2.3 Ecological Resilience

Two approaches may be employed for estimating ecological resilience. In the first approach, ecological resilience is a logical measure, which is one if the system eventually recovers to its pre-event equilibrium, and zero if it does not.

$$Ecological\ Resilience = \begin{cases} 1 & \text{if } E[pop_3] = E[pop_1] \\ 0 & \text{if } E[pop_3] \neq E[pop_1] \end{cases}$$

Equation 5-6

where:

- $E[x]$: expected value of x
 pop_1 : population at point 1 in Figure 1-2
 pop_3 : population at point 3 in Figure 1-2

A second definition is proposed here which also shows the degree to which the pre- and post-event equilibriums are close. In this approach, the ecological resilience is the expected value of ratio of population at point 3 to population at point 1 in Figure 1-2. That is:

$$Ecological\ Resilience = E \left[\frac{pop_3}{pop_1} \right]$$

Equation 5-7

Therefore, ecological resilience is a dimensionless value which ranges from zero to one, and is desired to be maximized.

5.2.4 Vulnerability ÷ Engineering Resilience (V/R)

In order to employ a holistic indicator, which addresses both short- and long-term impacts of the extreme event, a new risk-based performance measure is proposed herein. This performance measure may integrate both vulnerability and engineering resilience, i.e., vulnerability divided by engineering resilience. It is referred to as V/R hereafter, and is calculated by Equation 5-8. By examination of Equation 5-8 and Figure 1-2, it may be observed that V/R includes both short- (i.e., maximum short-term loss) and long-term (i.e., time to recovery) impacts on the system, especially for systems which are ecologically resilient.

$$V/R = \frac{\text{Vulnerability}}{\text{Engineering Resilience}}$$

Equation 5-8

The performance measure of V/R has a unit of years which may range from zero to infinity. It is preferred to be minimized.

If the probabilistic approach for fish population recovery modeling is employed, and the system is not ecologically resilient, point 3 may not be identifiable (see Figure 5-3), because the two time series will not match. In such cases Equations 5-5 to 5-8 may not be applied. Here, a visual check may be used to determine if the system is ecologically resilient by observing whether the two time series match. Likewise, in such cases, engineering resilience may not be estimated, because point 3 may not be identified. As a result, V/R may not be evaluated as well, because it is a function of engineering resilience.

It should be noted that the V/R performance measure may be adapted for different decision-making criteria. For example, if long-term population reduction is considered more important than immediate population loss, V/R may be replaced by a V/R^2 performance measures.

Alternatively, if the maximum value of population loss in a single year is the major concern of decision-makers, a performance measure of V^2/R may be adopted.

5.3 Choice of Modeling Approach

All three modeling approaches (i.e., deterministic, pseudo-probabilistic, and probabilistic) may be used to approximate expected values of risk-based performance measures. However, the pseudo-probabilistic and probabilistic approaches provide richer data compared with the deterministic approach, because they also describe the uncertainty in these measures.

The choice of modeling approach is dependent on the information needs and on nature of the system. For example, if performance measures are to be incorporated in a decision-making process that utilizes deterministic performance values (e.g., a conventional MCDA), the deterministic approach may be employed. However, when performance measures are to be incorporated in a probabilistic decision analysis (e.g., a reliability-based systems analysis), a probability distribution function of performance measures is required rather than just their expected values. In the latter case, either the pseudo-probabilistic or the probabilistic approach must be employed, because these approaches produce a large number of simulated time series of population, and hence, a probability distribution function for risk-based performance measures. The probabilistic approach addresses more sources of uncertainty than the pseudo-probabilistic approach. However, application of the probabilistic approach is limited to systems which are ecologically resilient to potential extreme events. Therefore, selection between these two approaches varies for different systems.

Since time-efficient modeling techniques have been used in developing the fish population recovery model (e.g., incorporating different time scales), even the probabilistic approach is not computationally intensive. Therefore, time-efficiency would not be a factor in selection among the deterministic, pseudo-probabilistic, and probabilistic approaches.

Chapter 6: Case Study: the Lower Campbell River

In order to demonstrate the application of the proposed framework, it is used to investigate a case study of the Lower Campbell River. As mentioned in the preface of the dissertation, development of the hydrotechnical models for this case study (i.e., River2D and River2DM modeling) was conducted as part of an M.A.Sc. thesis which was guided by the author to support the needs of this dissertation.

The 1.7-km-long study reach of interest is on the Lower Campbell River, between the John Hart Power Generation Station, and the confluence with the Quinsam River. Location of the study reach in the Campbell River Basin, and a larger scale map focusing on the study reach, are shown in Figure 6-1 and Figure 6-2, respectively. Five species of Pacific salmon (*Oncorhynchus* spp.) are found in the Campbell/Quinsam system. These are pink (*O. gorbuscha*), chinook (*O. tshawytscha*), chum (*O. keta*), coho (*O. kisutch*), and sockeye (*O. nerka*). There are also steelhead trout (*O. mykiss*) and cutthroat trout (*O. clarki*) (Nagtegaal et al. 2000). Large specimens of chinook salmon, 14 kilograms or larger, produced in the Campbell River are referred to as Tyee. Chinook salmon is selected as the target species for this research due to its abundance in the study reach, availability of data related to its life cycle, and its environmental and economic value compared with the other species.

6.1 Available Data

6.1.1 Hydrology

The Campbell River headwaters are in the Vancouver Island Mountain Ranges. The watershed area is 1460 km², and the river generally flows to the north, and then to the east, until it drains into Discovery Passage at the Municipality of Campbell River (Burt 2004). The average flow rate in the study reach is 98.5 cms.

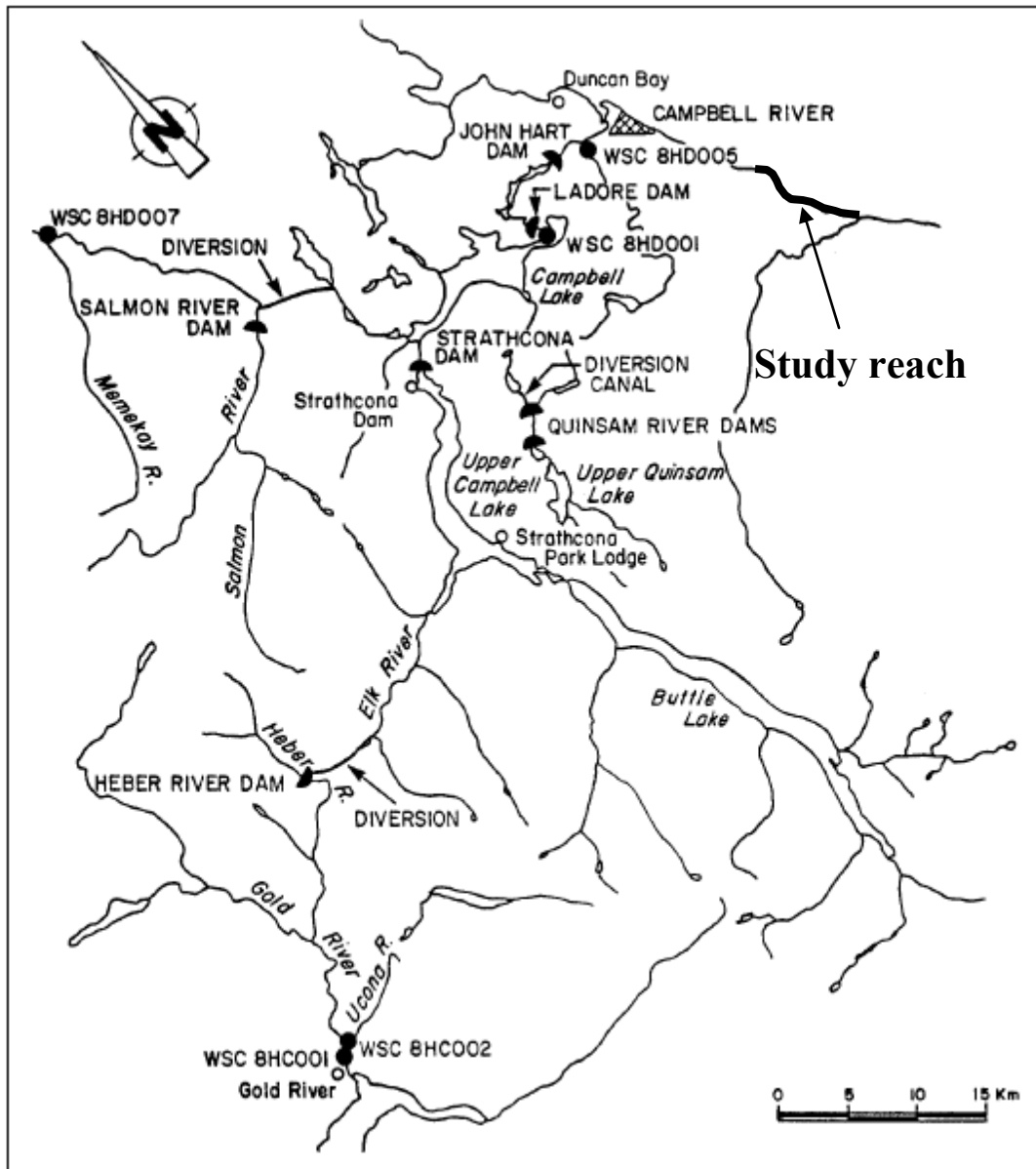


Figure 6-1. Location of study reach and hydro projects on Campbell River (from BCRB 2000)

Three major hydropower facilities have been built on the Campbell River system. The most downstream dam is the John Hart Dam (built in 1947). The Ladore Dam (built in 1957) is in the middle, and Strathcona Dam (built in 1958) is the most upstream one. Location of these three major hydropower projects which operate in series is shown in Figure 6-1. Development of these projects has not affected the migration path of anadromous fish. A 30-metre-high natural barrier downstream of the John Hart Dam, the Elk Falls, blocks upstream

migration of anadromous fish. However, significant change in flow and sediment regime has resulted in a significant fish population decline from 2500-6000 in the 1940s, to an average of 800 in the recent years (Bennet et al. 2010; Nagtegaal et al. 2000).

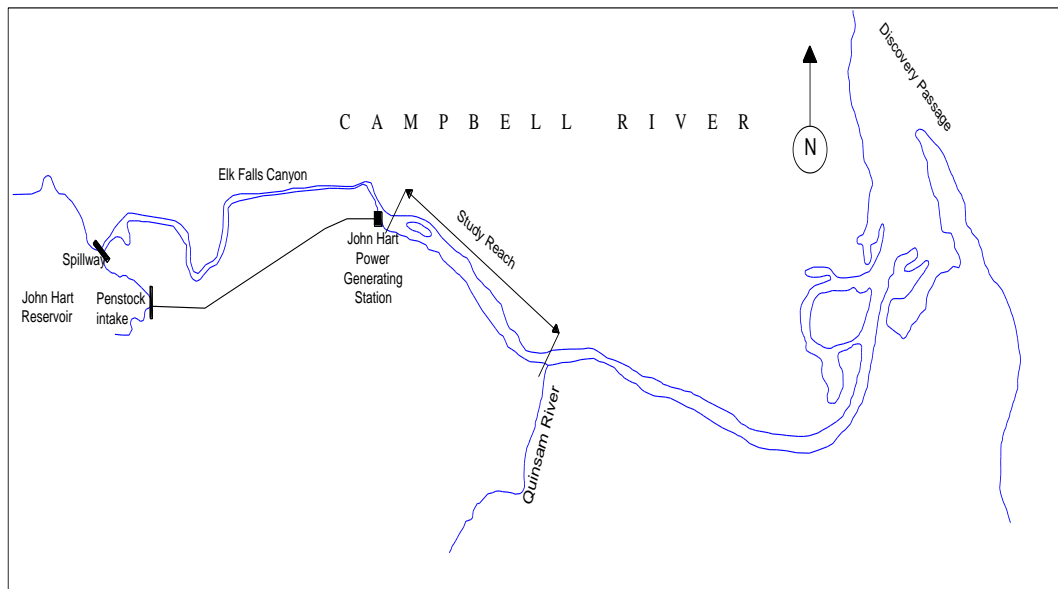


Figure 6-2. Study reach on Campbell River

Given the hydrological data of the Campbell River watershed, and the operating rules of the Strathcona Dam, Klohn Leonoff (1989) estimates the peak flood downstream of the John Hart Dam with different return periods as summarized in Table 6-1. Floods normally occur during October to March (Klohn Leonoff 1989). The hydrographs of these floods based on the data reported in Klohn Leonoff (1989), are shown in Figure 6-3. These studies are now being revised (Yusuf 2011) and floods with specific return periods are expected to have lower peak flows. However, this research utilizes the flood information in terms of the hydrograph peak flow values, rather than relying on any probabilistic analysis of the return periods. That is, the results are dependent on absolute flow intensity, not on return periods.

Table 6-1. Peak flow rates of Campbell River (cms)

Return Period	Peak Flow Rate (m ³ /s)
2-Yr	220
5-Yr	450
10-Yr	1073
50-Yr	1127
200-Yr	1240

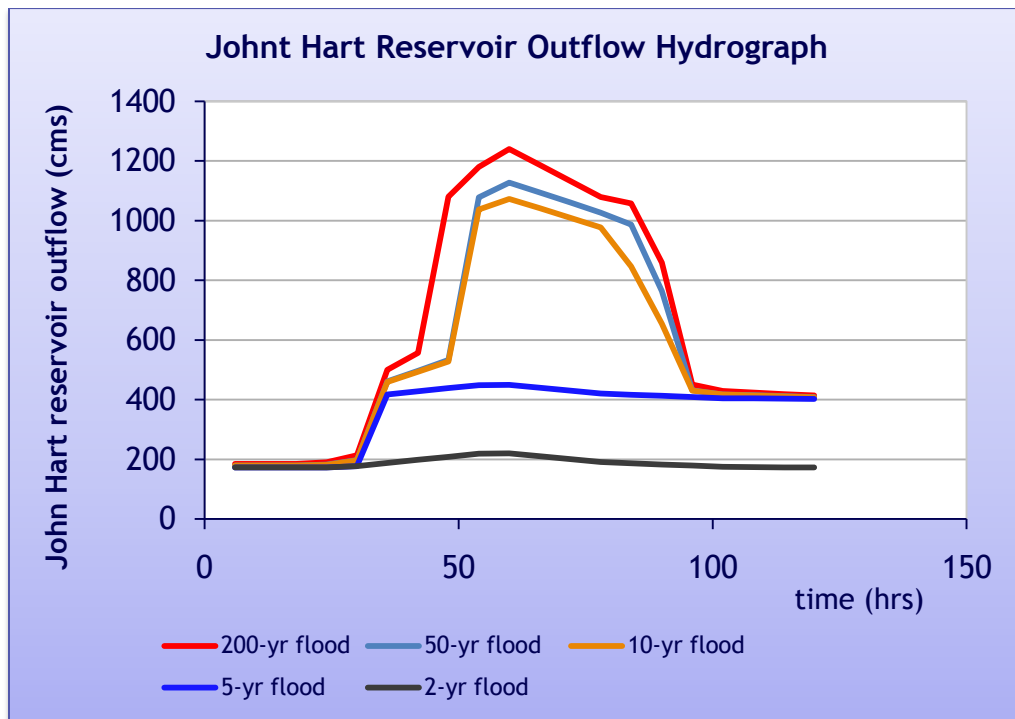


Figure 6-3. Hydrograph of John Hart Dam outflows

6.1.2 Ecology

The life history of Campbell River chinook salmon in normal years is illustrated in Figure 6-4. Also life history parameters, which are used for this case study, and are explained herein, are summarized in Table 6-2.

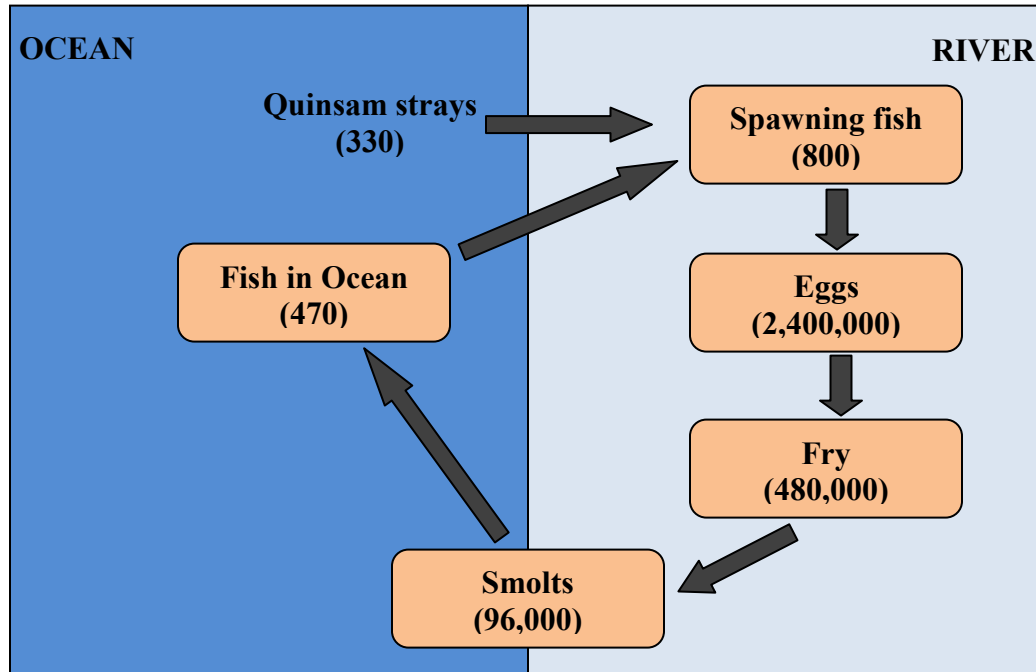


Figure 6-4. Life cycle of Campbell River chinook salmon

Table 6-2. Expected values and ranges of assumed ecological parameters

Parameter	Expected value	Range
Fecundity	6000	5750 - 6250
Egg-fry survival	20%	15% - 25%
Egg-smolt survival	4%	3.5% - 4.5%
Ocean survival	0.5%	0.2% - 0.8%
Quinsam stray	330	320 - 340

In a normal year, about 800 adult chinook salmon migrate to the study reach on the Campbell River (Ewart and Anderson 2010). About 41% of this population (i.e., 330 spawners) originate from the Quinsam River hatchery (Bennet et al. 2010; Burt 2004). The upstream migration starts in September; however, the spawning period is from the beginning of October until mid-November (Burt 2003). About 80% of the spawning occurs in the last two weeks of October, and 10% of it occurs at either ends of the spawning period (Ewart 2010). Given this distribution, and a female to male ratio of one (Bennet et al. 2010), the number of spawning pairs in the study reach are illustrated in Figure 6-5.

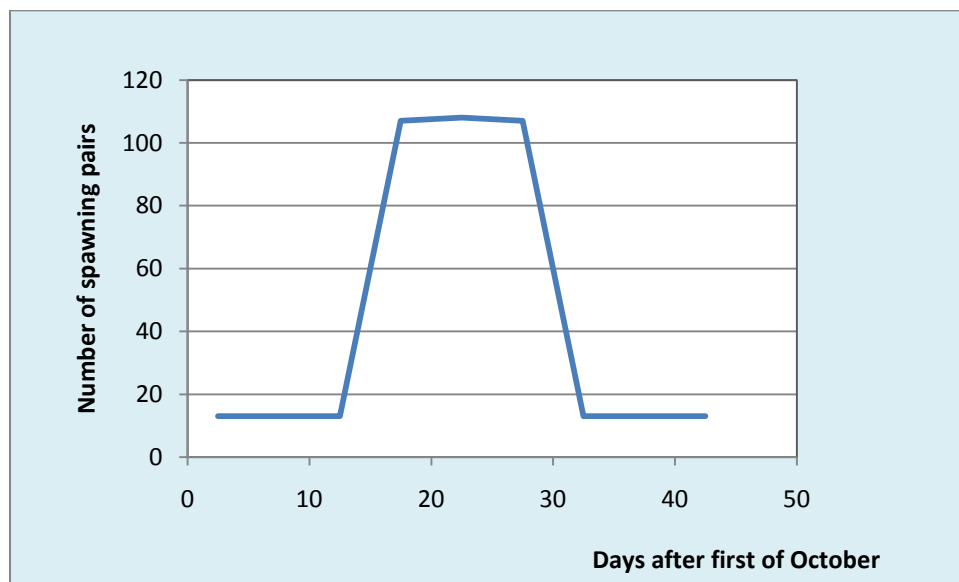


Figure 6-5. Distribution of number of spawning pairs in study reach

The average fecundity rate (i.e., eggs per female) is 6000 (Ewart and Anderson 2010; Nagtegaal et al. 2000). The D_{redds} is not known with certainty. Evenson (2001) conducted a survey of the depth of chinook redds in the Trinity River, California. The data from this survey show a geometric mean and standard deviation of 30.0 cm and 8.4 cm, respectively. As shown in Figure 6-6, there is a strong agreement between the surveyed data for the Trinity River and the log-normal distribution. Since such a survey has not been undertaken for the

Campbell River, data from the Trinity River are adopted here. That is, a log-normal distribution is employed in the random number generator of the immediate egg survival estimation model.

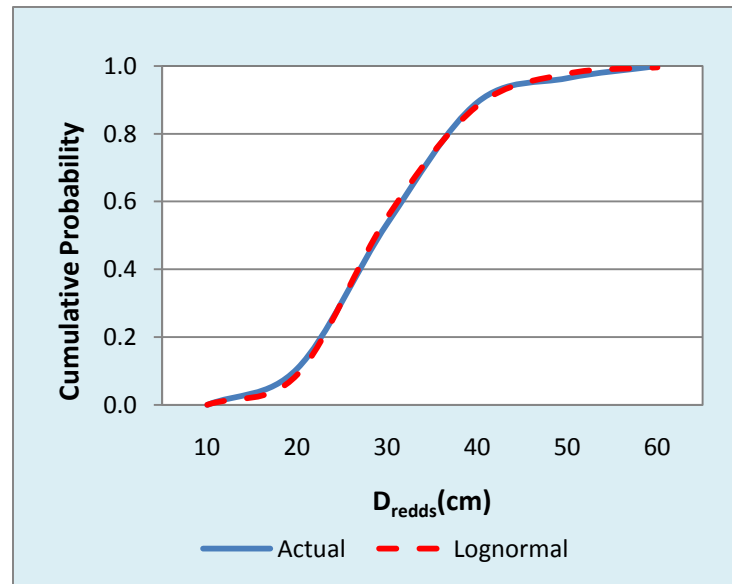


Figure 6-6. Cumulative distribution for depth of redds in Trinity River (from Evenson 2001)

In the Campbell River, eggs hatch and stay in the gravel bed during January and February, and start emerging as free swimming fry from April until June. Campbell River chinook is an ocean type, and most of the fry migrate downstream to be reared as smolts either in the estuary, or in the ocean. Most of the smolts leave the estuary to the ocean in the period of May to July (Burt 2003). Most of the fish return to the river as four- or five-year-old adults. The number of female three-year old or younger returning fish is insignificant. This number is less than 10% for male fish. In this research, precocious returns are ignored for simplicity. It is approximated from Bennet et al. (2010) that the ratio of four- to five-year-old returning adults is one.

Statistical survival rate data show significant variation and uncertainty. Healey (1991) analyzes previous studies of the egg-fry survival rate, and states that the results are not

conclusive. However, he suggests an upper bound of 30% for the egg-fry survival of chinook salmon. Korman et al. (1997) assume a range of 15 to 40% for the egg-fry survival rate. Nagtegaal et al. (2000) use a range of 8 to 16% for the chinook salmon egg-fry survival rate in the Campbell/Quinsam system. Bradford (1995) conducts a statistical analysis of egg-smolt survival rates, data for which are more available than egg-fry survival rates for chinook salmon, and suggests an average of 6.4 and 8.6%, for the egg-smolt survival rate of stream type and ocean type chinook salmon, respectively. Ewart (2010) suggests an average of 4% for the egg-smolt survival rate for the Campbell River chinook salmon.

In this research, an egg-fry survival rate of 20% is considered. The egg-smolt survival is calculated from Equation 5-2 and Equation 5-3. Carrying capacity (i.e., *CC*) is set equal to 160,000 based on Korman et al. (1997) and the smolt survival rate at a very low density (i.e., *S*) is assumed to be equal to two (Ford 1999). The average ocean survival rate is assumed to be 0.5% (Bennet et al. 2010; Ewart and Anderson 2010; Nagtegaal et al. 2000).

Given these assumptions for the survival rates, in a normal year, 800 spawning fish deposit 2,400,000 eggs, and 480,000 fry emerge from these eggs and grow as smolts. About 96,000 smolts survive and start growing in the ocean. That is, there is a 4% egg-smolt survival rate which is in agreement with observations of Ewart and Anderson (2010). Among these 96,000 fish, only 470 fish survive the natural and fishing hazards in their four to five years in the ocean, and migrate back to the Campbell River, along with 330 strays originating from the Quinsam River.

Given the uncertainty in the statistical ecological parameters, ranges of these parameters are considered in Table 6-2 based on available data for the river and in the literature. The range of fecundity is drawn from the available data in Nagtegaal et al. (2000). For the egg-fry survival rate, the selection of the range is based on Healey (1991), Korman et al. (1997), Bradford (1995), and Nagtegaal et al. (2000). The ranges of egg-smolt survival rate and ocean survival rate are selected based on personal communication with Ewart and Anderson (2010), and the range of the Quinsam River strays is based on available data from Bennet et al. (2010).

The average length of spawning chinook salmon in the Campbell River is 790 mm (Bennet et al. 2010). Fish swimming capacity is not known with certainty. The ranges of sustained, prolonged, and burst velocity for chinook salmon, as a function of fish length, are provided in Table 6-3 (Beamish 1978).

Table 6-3. Swimming capacity of chinook salmon (length/sec)

Parameter	Range
Sustained swimming speed	0.8 - 1.8
Prolonged swimming speed	4 - 7
Burst swimming speed	7 – 10.5

Spawning and rearing HSC for the Campbell River chinook salmon are shown in Figure 6-7 and Figure 6-9, respectively (Leake 2004). Since the only parameters which are considered in these HSCs are depth and velocity, they may be represented as two-dimensional surfaces. Such a contoured format for spawning and rearing HSC are shown in Figure 6-8 and Figure 6-10, respectively.

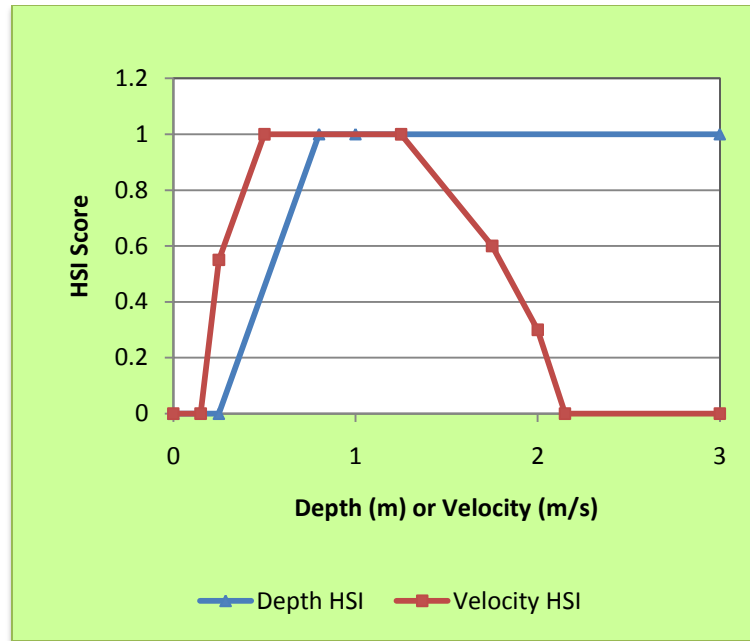


Figure 6-7. Spawning HSC for Campbell River chinook salmon (from Leake 2004)

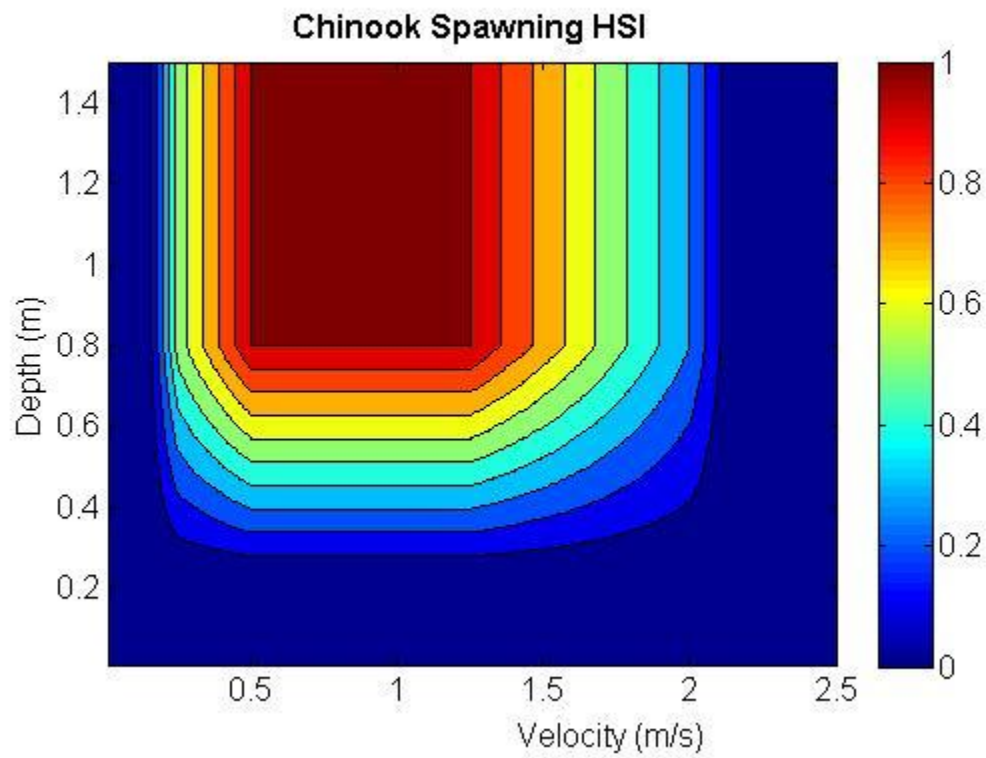


Figure 6-8. Contours of spawning HSC for Campbell River Chinook salmon

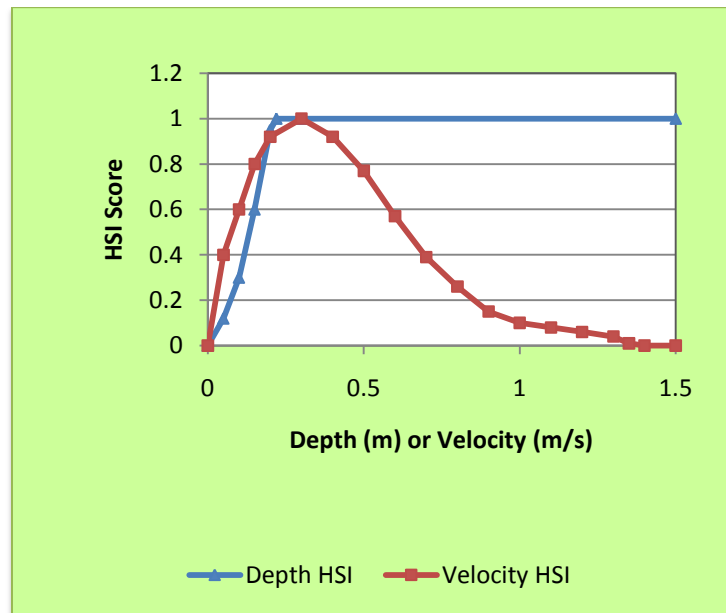


Figure 6-9. Rearing HSC for juvenile Campbell River chinook salmon (from Leake 2004)

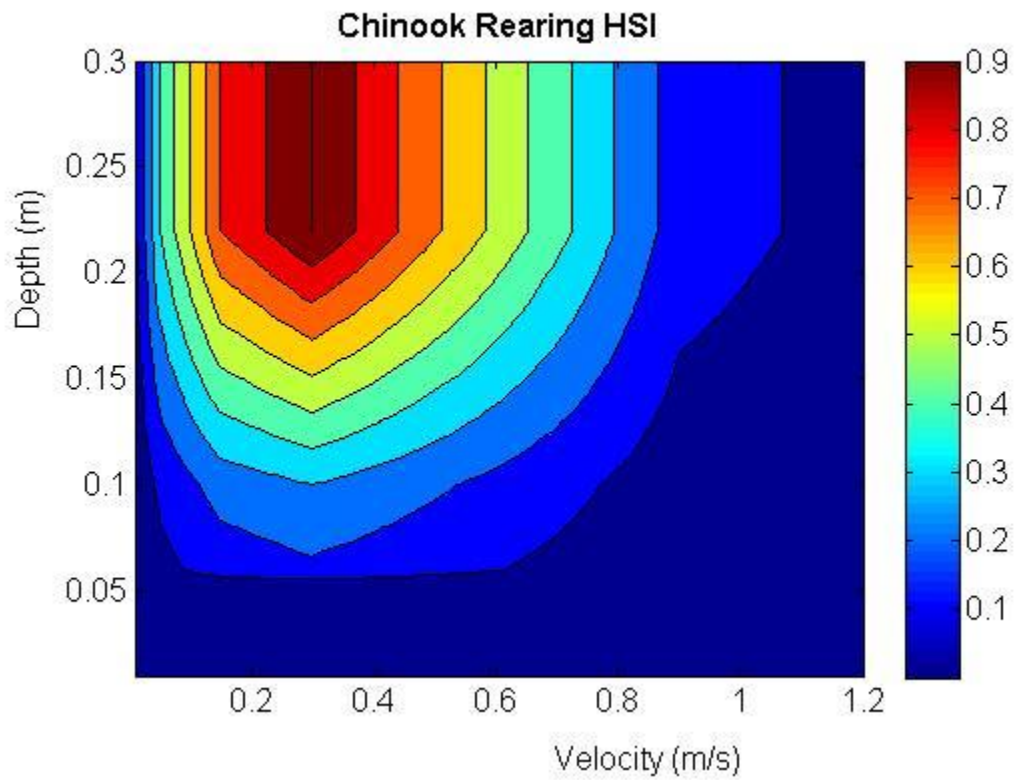


Figure 6-10. Contours of rearing HSC for juvenile Campbell River Chinook salmon

6.2 Applying Short-term Models: Hydrotechnical and Immediate Impacts

6.2.1 Hydrodynamic and Morphodynamic Models

As mentioned in the preface of this dissertation, hydrodynamic and morphodynamic modeling development and implementation for this case study were conducted as part of an M.A.Sc. thesis project which was guided by the author.

The transient hydrodynamic response of the study reach to the hydrographs of the extreme events was simulated using the River2D model. For the study reach, the maximum size of triangular simulation cells is 25 m², which is the same size as a chinook salmon spawning territory, and is considered to be a reasonable size in two dimensional transient hydrodynamic modeling for adequately representing changes in both depth and velocity. This size is also in accordance with Equation 4-1, and results in 9648 triangular simulation cells throughout the study reach. The hydrographs of the extreme events and a downstream rating curve (from Water Survey of Canada) were considered as the upstream and downstream boundary conditions of the model, respectively. The simulated water surface elevations at Water Survey of Canada gauge 08DH003 within the study reach were validated against two observed high floods of 323 and 561 cms. Simulated and observed water surface elevations are one percent different in both cases. A sample visual output of the model, i.e., a velocity snapshot for the 1073 cms flood, is shown in Figure 6-11.

As explained in Section 4.2.1.1, for any given flood, matrices that include depth (i.e., d_t^i), and specific discharge in both x and y directions (i.e., qx_t^i and qy_t^i) for all cells in the river reach are generated by the hydrodynamic model with a time step of Δt_{h-out} . Also, x and y coordinates of all cells are recorded in output files. These output files are used in the pre-processor and main program of the immediate fish survival estimation model.

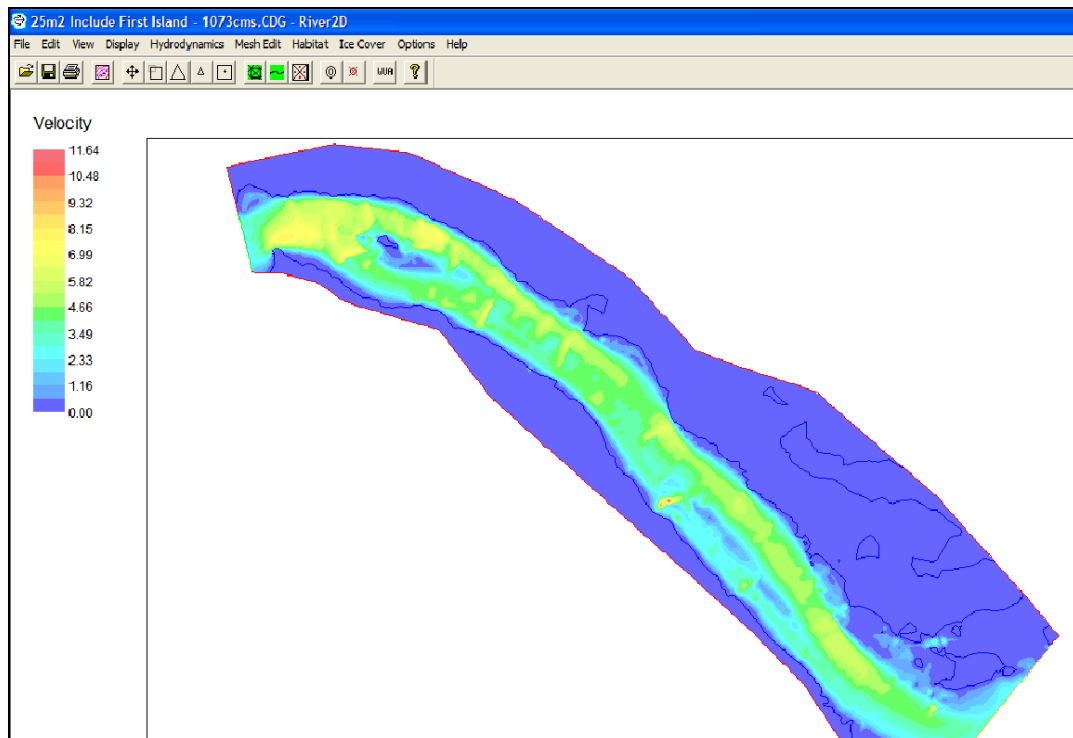


Figure 6-11. Sample visual output of River2D model: snapshot of velocity contours of 1073 cms flood (m/s)

Also, the morphodynamic response of the system (i.e., the sediment scour and deposition depths) for all extreme events was estimated using a River2DM model with the same geometric conditions as the River2D model. The scour and deposition depth output files are used in the immediate egg survival estimation, and immediate habitat change estimation models. The results of these models are presented in Sections 6.2.3 and 6.2.4, respectively.

It should be noted that the morphodynamic model, in this case, incorporates uniform substrate material throughout the reach, i.e., uniform grain size and spatial distribution is considered. This limitation is due to availability of data and computational time rather than a conceptual limitation.

6.2.2 Immediate Fish Survival Model

Given the swimming capacity range in Table 6-3, and the average length of 790 mm, scenarios for five evenly distributed discrete swimming capacities are identified and examined in this work. These are scenarios of: very low, low, medium, high, and very high swimming capacities which are summarized in Table 6-4. A correlation value of one is considered among sustained, prolonged, and burst swimming capacity possibilities. That is, if sustained swimming speed of a fish is very low, for example, its prolonged and burst swimming speeds are very low, too. Since all sustained, prolonged, and burst swimming capacities are correlated with fish length (i.e., strength), this assumption seems acceptable. Also, it limits the swimming capacities to five scenarios, which saves computational time. It should be noted however, that the model is not limited to such an assumption, and may be implemented with a large number of swimming speed capacities if computational time is not an issue.

Table 6-4. Swimming capacity scenarios for adult Campbell River chinook salmon (m/s)

Swimming Capacity	Very low	Low	Medium	High	Very high
Sustained	0.6	0.8	1.0	1.2	1.4
Prolonged	3.1	3.7	4.3	4.9	5.5
Burst	5.5	6.2	6.9	7.6	8.2

Given the output files of the hydrodynamic model and the spawning HSC, the pre-processor and main program of the immediate fish survival estimation model are implemented for 25 cases. These are combinations of the five extreme events in Figure 6-3, and the five swimming capacity scenarios in Table 6-4. By running the pre-processor, it is identified that 1518 cells, out of the 9648 simulation cells, have positive initial HSIs. These are the potential initial location cells that are simulated in the main program. Each of the 25 runs of the main program takes a few hours on a computer with four parallel processing cores (i.e., Intel core-i7). The results are five lookup tables of fish survival for five extreme floods, for all potential initial cells. Each lookup table includes the results of five fish swimming capacities. A sample lookup table, i.e., results of the 450 cms flood, is provided in Table 6-5.

Table 6-5. Sample lookup table for 450 cms flood

Cell No.	Pre-flood	Swimming capacity				
	HSI	Very low	Low	Medium	High	Very high
2513	0.02	0	0	0	0	1
2514	0.03	0	0	0	0	1
2515	0.04	0	0	0	0	1
2516	0.06	0	0	0	0	1
2517	0.12	0	0	0	0	1
2518	0.01	0	0	0	0	1
2524	1.00	0	1	1	1	1
2525	0.68	0	1	1	1	1
2526	0.44	0	0	0	1	1
2527	0.59	0	0	0	1	1
2528	0.56	0	0	0	1	1
2533	0.22	0	0	0	1	1
2534	0.90	0	0	0	1	1
2551	0.38	0	0	0	0	0
2552	0.01	0	0	0	0	0
2563	0.02	0	0	1	1	1
2564	0.04	0	0	1	1	1
2565	0.04	0	0	1	1	1
2566	0.05	0	0	0	1	1
2567	0.14	0	0	0	1	1
2575	0.04	0	1	1	1	1
2576	0.94	0	1	1	1	1
2577	0.66	0	1	1	1	1
2578	0.27	0	0	0	1	1
2579	0.85	0	0	0	1	1
2580	0.42	0	0	0	1	1
2585	0.72	0	0	0	1	1
2586	0.72	0	0	0	1	1
2601	0.04	0	0	0	0	0
2602	0.45	0	0	0	0	0
2603	0.71	0	0	0	0	0
2614	0.01	0	1	1	1	1
2615	0.03	0	1	1	1	1
2616	0.04	0	1	1	1	1
2617	0.05	0	1	1	1	1
2618	0.11	0	1	1	1	1
2619	0.08	0	1	1	1	1
2626	0.17	0	1	1	1	1
2627	0.15	0	1	1	1	1
2629	0.02	0	0	0	1	1
2630	0.86	0	0	0	1	1
2636	0.65	0	0	0	1	1
2637	0.39	0	0	0	0	1
2651	0.02	0	0	0	0	0
2652	0.05	0	0	0	0	0
2653	0.20	0	0	0	0	0
2665	0.02	0	1	1	1	1
2666	0.03	0	1	1	1	1
2667	0.04	0	1	1	1	1
2668	0.08	0	1	1	1	1
2669	0.17	0	1	1	1	1
2670	0.03	0	1	1	1	1
2677	0.04	0	1	1	1	1
2678	0.50	0	1	1	1	1
2680	0.20	0	0	0	1	1
2681	0.23	0	0	0	1	1
2687	1.00	0	0	0	1	1
2688	0.48	0	0	0	0	1
2716	0.02	0	1	1	1	1
2717	0.03	0	1	1	1	1
...*
...
...
9556	0.34	1	...1	1	1	1
9566	0.82	1	1	1	1	1

*: Complete table includes 1518 cells

Given these lookup tables, the post-processor module is implemented with three different approaches for sampling from the five swimming capacity possibilities as explained above. These are: using deterministic swimming capacity (i.e., medium swimming capacity), sampling from a binomial distribution, and sampling from a discrete uniform distribution. These sampling approaches are shown in Figure 6-12.

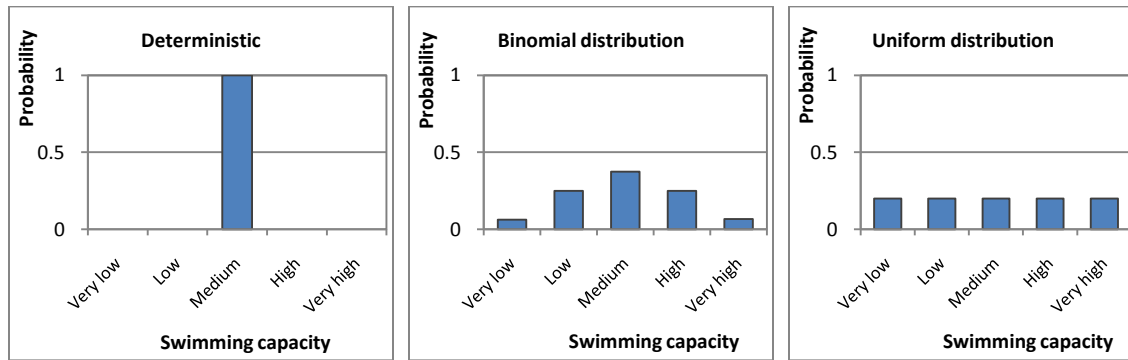


Figure 6-12. Deterministic value, binomial distribution, and discrete uniform distribution of swimming capacities

In the first approach, swimming capacities have been assumed as deterministic parameters. That is, only the medium swimming capacity is employed. By conducting 10,000 samplings for the initial location of fish in the river reach, the immediate fish survival rate for the five extreme events are evaluated for all samples, and the results are shown as histograms in Figure 6-13.

The second approach includes a binomial distribution with $p=0.5$ (i.e., symmetric) and $n=4$ (i.e., five possible outcomes) for the five swimming capacity scenarios. Post-processing in this case includes 10,000 samplings for both the initial location of the fish and the swimming capacity. In the absence of identified correlation between these two random variables, they are considered to be independent. However, this is not a conceptual limitation to the model. If such a correlation is identified, sampling may be conducted through multi-variate random distributions. Immediate fish survival rates for the five extreme events using this approach are shown as histograms in Figure 6-14.

The third approach is similar to the second one, except a discrete uniform distribution for the five swimming capacities is used. The results are shown in Figure 6-15.

Generated distributions of immediate fish survival rates, shown in Figures 6-13 to 6-15, may be used as input probabilistic data in the pseudo-probabilistic and probabilistic approaches for fish population recovery modeling as described in Section 5.1.2.

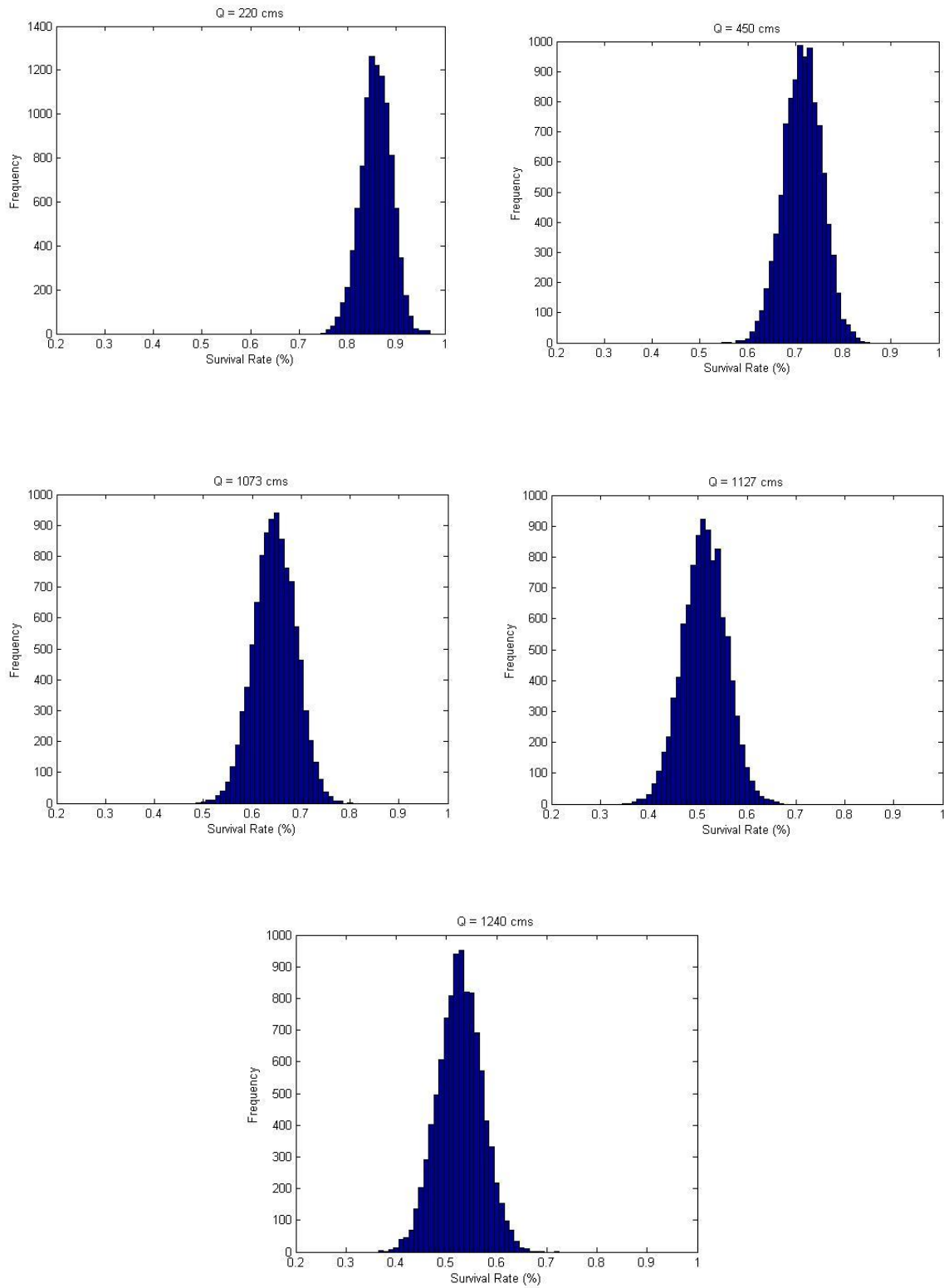


Figure 6-13. Survival rate histograms for deterministic swimming capacity

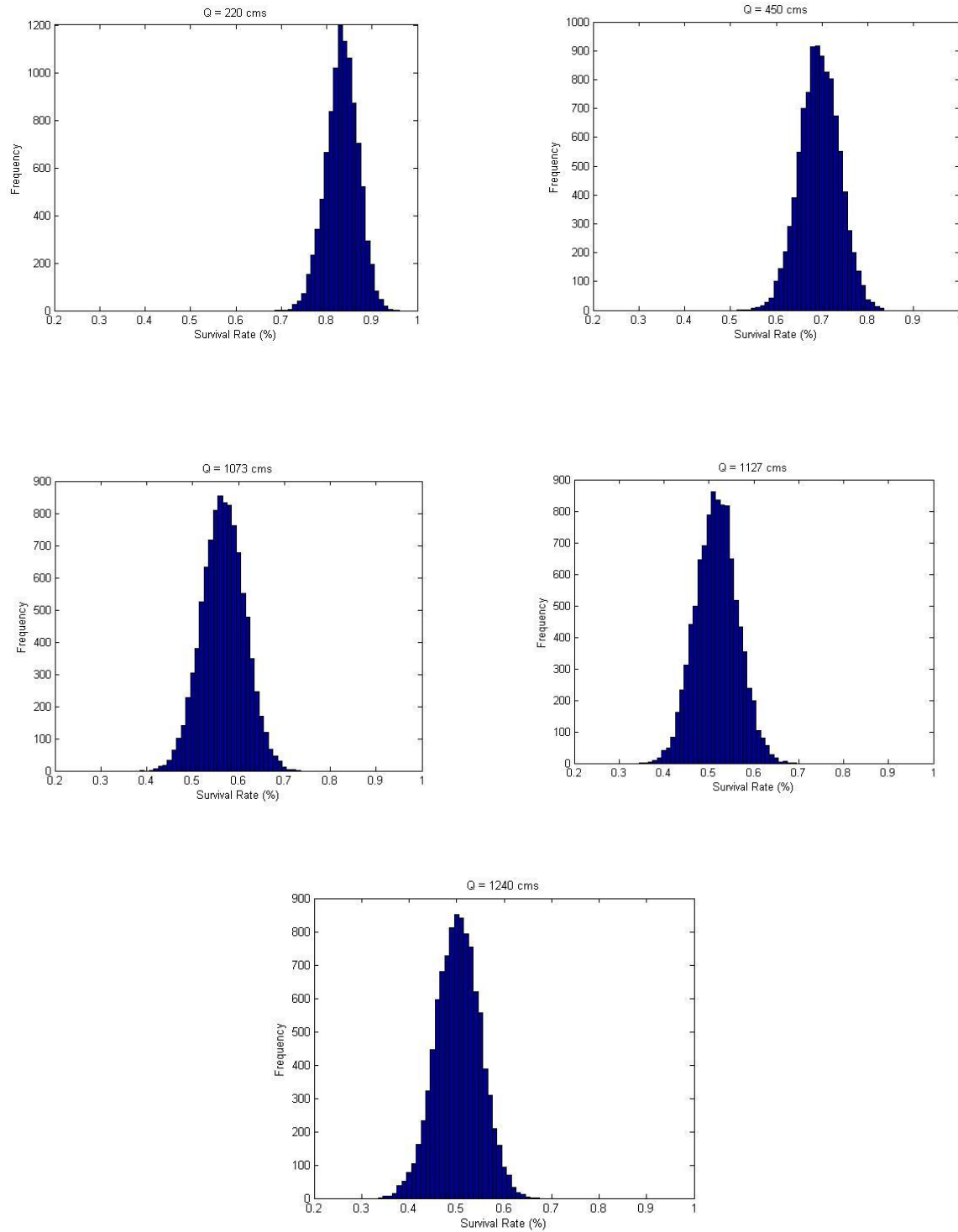


Figure 6-14. Survival rate histograms based on Binomial Distribution for swimming capacity

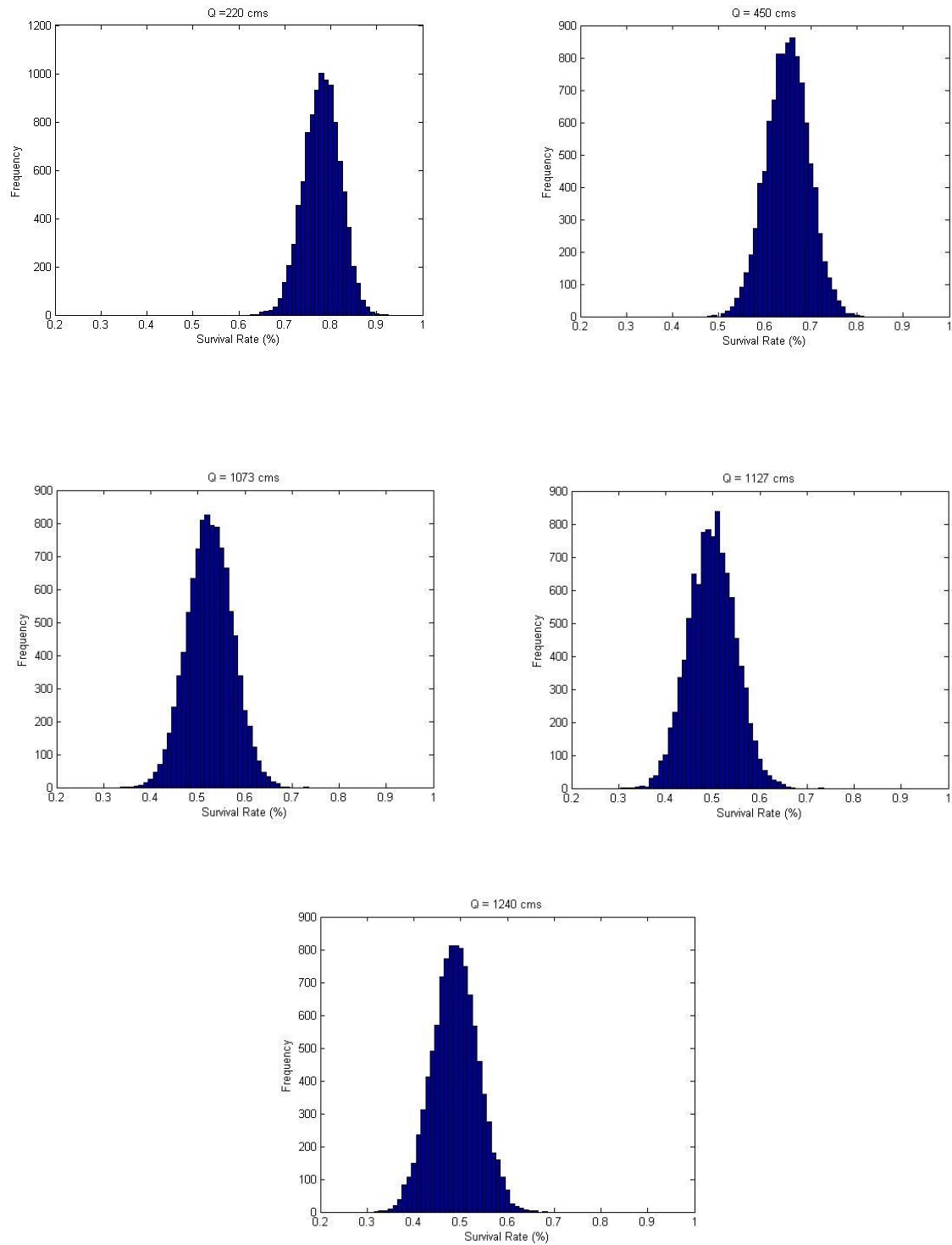


Figure 6-15. Survival rate histograms based on Discrete Uniform Distribution for swimming capacity

In order to compare Figures 6-12, 6-13, and 6-14 numerically, the mean and standard deviation of the survival rates for different flood levels and sampling approaches are summarized in Table 6-6. As one may expect, the results of the discrete uniform sampling and deterministic approaches show the highest and the lowest standard deviations, respectively. This is because of the difference in the spread of their input swimming capacity possibilities, as shown in Figure 6-12.

Also, the estimated immediate fish survival rates based on deterministic values and discrete uniform sampling are higher and lower than those based on binomial distribution sampling, respectively. This is probably due to the contribution of low and very low swimming capacities allowed in discrete uniform and binomial distribution sampling.

If the deterministic value approach is selected (i.e., based on the medium swimming capacity), the variability of swimming capacity is not considered. If the discrete uniform distribution is adopted, there is a risk of relying too heavily on swimming capacities at both extremes of the distribution, which are less reliable than the average value. Therefore, the binomial distribution which relies mainly on the average values, yet considers deviation, is preferred.

Table 6-6. Mean (and standard deviation) of immediate fish survival rates

Peak flow (cms)	Deterministic	Binomial distribution	Discrete uniform distribution
220	0.858 (0.031)	0.834 (0.035)	0.781 (0.039)
450	0.715 (0.040)	0.694 (0.043)	0.649 (0.046)
1073	0.646 (0.042)	0.567 (0.046)	0.526 (0.048)
1127	0.523 (0.044)	0.517 (0.047)	0.498 (0.049)
1240	0.527 (0.044)	0.503 (0.046)	0.488 (0.048)

Expected values of immediate fish survival rates for different flood levels with binomial sampling are shown in Figure 6-16. Here, the survival rate is not severely sensitive to the flood intensity. For example, the expected survival rates of 450 cms and 1073 cms are only 17% different from one another relative to the rate of the 450 cms flood. Even in the case of a 1240 cms flood, about 50% of fish survive and manage to spawn. This pattern is dramatically different for egg survival results as discussed in Section 6.2.3.

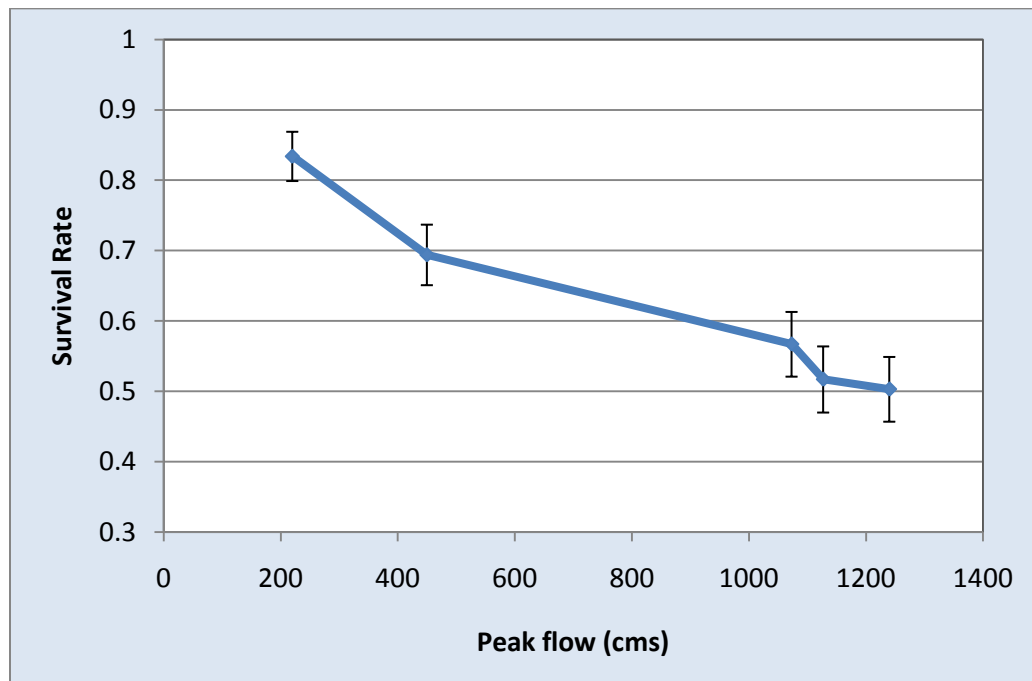


Figure 6-16. Expected immediate fish survival rate for different peak flows - standard deviations shown as error bars

It should also be mentioned that samples of 100 fish are assigned to potential cells in the post-processor module of the immediate fish survival model to estimate the immediate fish survival rate. Sensitivity of the survival rate to this number (i.e., n_f) is investigated. Since the number of fish in the study reach varies between 26 and 216 (Figure 6-5), the post-processor is implemented for three n_f values of 26, 100, and 216. The estimated survival rates for these n_f values are 71, 70, and 69%, respectively. These estimated rates are not significantly different, and thus, estimating survival rates based on 100 fish is considered acceptable.

6.2.3 Immediate Egg Survival Model

The results of the morphodynamic model include the maximum scour during the extreme event, and the final deposition depth at the end of the extreme event at 213 spawning cells. Sample results for the 450 cms flood are provided in Table 6-7. Given these results for extreme events with different intensities, a log-normal random number generator is utilized, and the immediate egg survival rates for 100,000 samples are estimated for the five extreme events following the procedure explained in Section 4.2.2. Histograms of the immediate egg survival rate for the 220 cms and 450 cms floods are shown in Figure 6-17 and Figure 6-18, respectively. Histograms for the other three extreme events (i.e., 1073, 1127, and 1240 cms) are not necessary, because for these extreme events all samples show 100% egg loss.

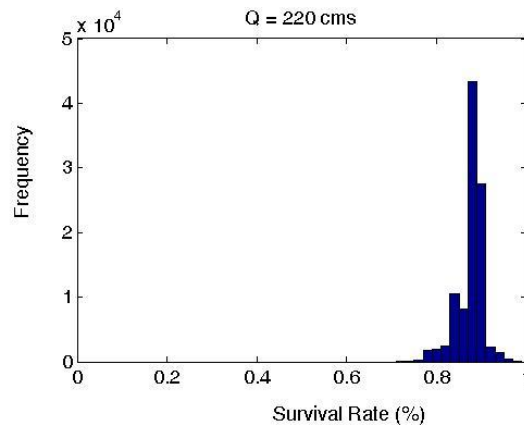


Figure 6-17. Immediate egg survival rate histogram for 220 cms flood

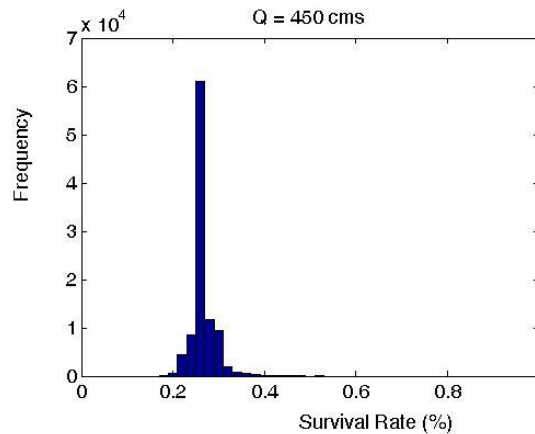


Figure 6-18. Immediate egg survival rate histogram for 450 cms flood

Table 6-7. Sample scour and deposition results from morphodynamic model for 450 cms flood

Cell No.	Scour (m)	Deposition (m)
5846		0.46
5847		2.05
5848		1.56
5849		1.89
5850		2.18
5853		0.53
5885		0.18
5886		1.53
5887		2.31
5888		2.08
5892		1.58
5923	0.96	
5924	0.70	
5925		0.63
5926		1.76
5927		1.88
5932		1.73
5964	0.84	0.06
5965	0.96	
5966	0.94	0.17
5967	0.48	
5968		1.14
5969		2.25
5970		1.92
5974		1.67
6005	1.15	
6006	1.12	
6007	1.28	
6008	0.05	
6009		1.87
6010		2.50
6011		2.14
6015		1.40
6016		0.92
6046	1.27	
6047	1.67	
6048	0.86	
6049		1.10
6050		1.95
6051		2.43
6056		1.86
6057	0.03	
6086	1.47	
6087	1.03	
6088	1.72	
5846		0.46
...*
...
...
8027		0.91
8028		1.47

*: Complete table includes 213 cells

The expected immediate egg survival rates for all of the extreme events are illustrated in Figure 6-19. The survival rate is severely sensitive to the flood intensity. For example, the expected survival rates of the 220 cms and 450 cms floods are about 70% different from each other relative to the rate of the 220 cms flood. For cases of a 1073 cms or higher flood, there is 100% egg loss.

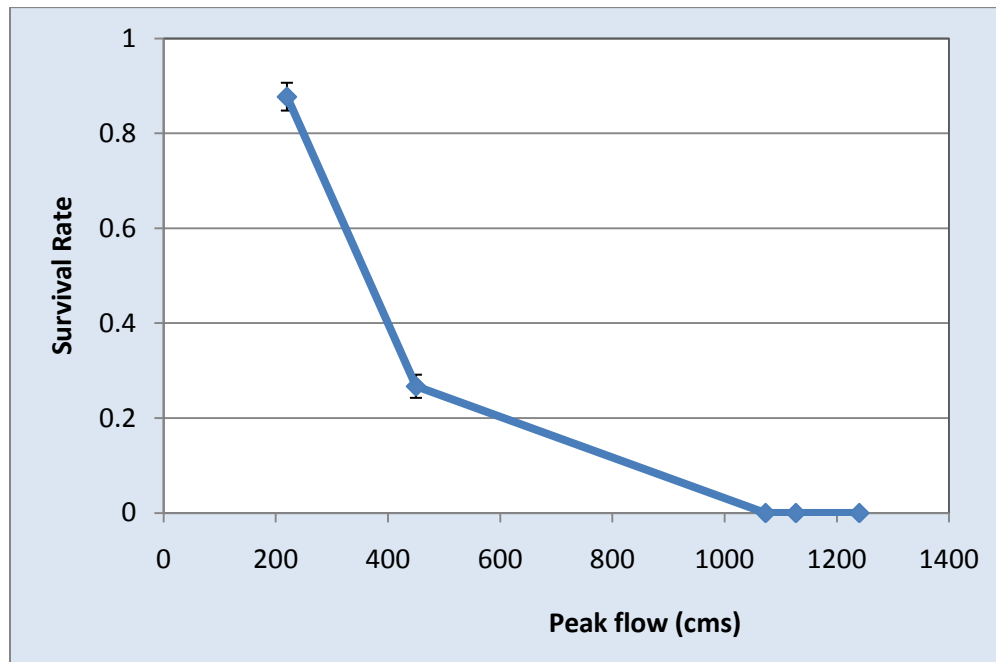


Figure 6-19. Expected immediate egg survival rate for different peak flows - standard deviations shown as error bars

6.2.4 Immediate Habitat Change Model

Given the pre- and post-flood geomorphology of the study reach (i.e., after the 220 cms and 450 cms floods), WUA for the spawning habitat are evaluated with the approach explained in Section 4.2.3. The results are provided in Figure 6-20.

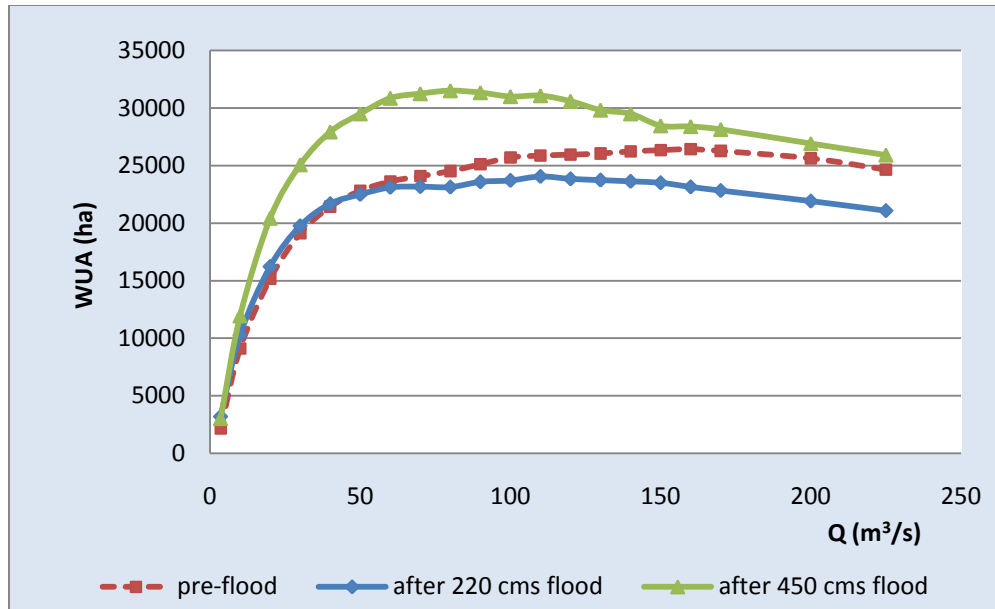


Figure 6-20. WUA for original geomorphology, after 220 cms and 450 cms floods

Habitat change, in terms of WUA, due to the extreme event is modeled and used as the carrying capacity for different life stages (e.g., eggs) in the population recovery model. However, the habitat change estimation model may not be used to provide such information in the Campbell River case study. Due to limited data availability in the Campbell River, habitat suitability is measured as a function of flow depth and velocity only. Although this is sufficient for the purpose of estimating immediate fish survival, it lacks some ecologically important factors of habitat suitability, e.g., substrate composition and cover. Therefore, estimated changes in WUA, as shown in Figure 6-20, may not be used to estimate the change in the carrying capacity for the river reach.

6.3 Fish Population Recovery Model and Risk-based Performance Measures

Given the results of the immediate fish and egg survival estimation models, and the ecological data introduced in Section 6.1.2, the fish population recovery model may be implemented as follows.

6.3.1 Fish Population Recovery Model

The fish population recovery model is run for 100 years or until the escapement reaches an equilibrium. It is assumed that the extreme event occurs within the 20th year of simulation. Since the number of fish and eggs present in the study reach is variable during the spawning period (i.e., the beginning of October to mid-November), the model is implemented for different occurrence dates within this period. Ten occurrence dates are selected with a time interval of five days, which is equivalent to the duration of extreme events. For the rest of the incubation period (i.e., until the end of February), there are no fish present in the river, and no more eggs are laid. Therefore, the model results are not dependant on the flood dates. Thus, 50 modeling runs are required for the combinations of five extreme events and ten flood dates. The model is implemented with the deterministic, pseudo-probabilistic, and probabilistic approaches introduced in Section 5.1.2.

If the deterministic approach is considered, each modeling run requires only one simulation. Sample results for a 450 cms flood that occurs in the fourth week of October are shown in Figure 6-21. In this case, a reduction of 102 fish is observed in the escapement four years after the event. Reduced escapement is observed for four generations. Given a life cycle of four to five years, fish population recovery takes on the order of 25 years.

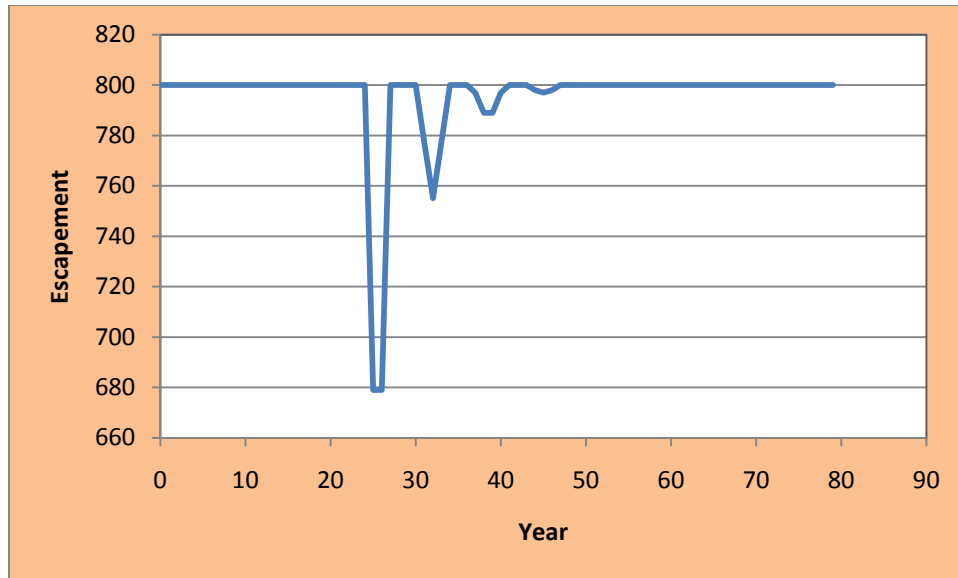


Figure 6-21. Fish population recovery model output with deterministic approach: deterministic immediate loss and ecological parameters for 450 cms flood in fourth week of October

If the pseudo-probabilistic approach is implemented, each of the 50 modeling runs employs 10,000 simulations. Such simulations involve sampling of random variables for the immediate fish survival rate (i.e., Figure 6-14), and for the immediate egg survival rate in the case of 220 cms and 450 cms floods (i.e., Figure 6-17 and Figure 6-18). Data in Figure 6-22 show sample results, including the simulated fish population time series for two example simulations, and for the average of all 10,000 simulations for a 450 cms flood that occurs in the fourth week of October. Although each simulation results in different reduction in escapement, the average time series of the simulations results in the same pattern as the modeling run with deterministic immediate fish and egg survival rates.

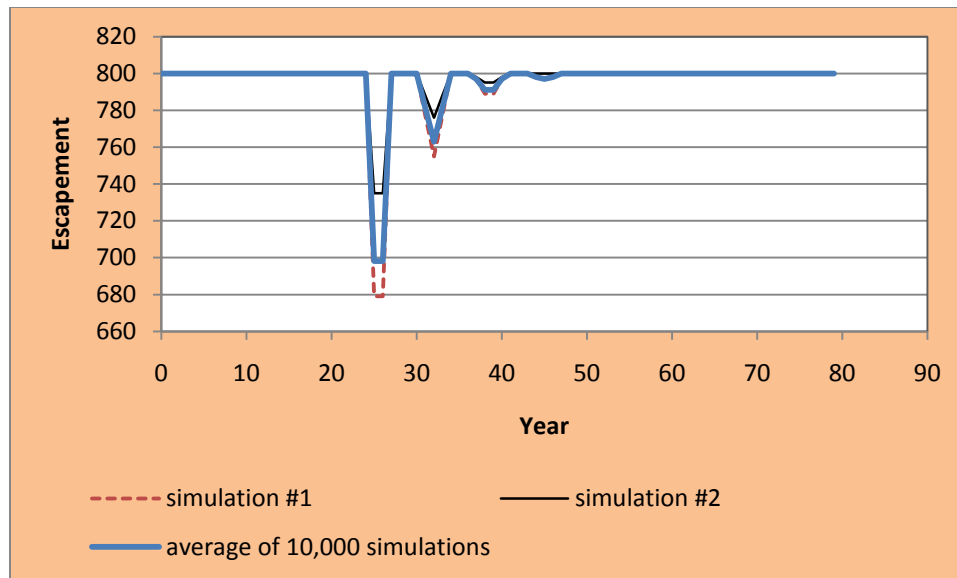


Figure 6-22. Fish population recovery model output with pseudo-probabilistic approach: probabilistic immediate loss and deterministic ecological parameters for 450 cms flood in fourth week of October

Similar to the pseudo-probabilistic approach, the probabilistic approach also involves employing 10,000 simulations for each of the 50 modeling runs. The fish population recovery model is implemented in the same way as in the pseudo-probabilistic approach. However, here, each of the immediate survival rates and the ecological parameters are sampled from uniform distributions with ranges provided in Table 6-2. Random variables are considered independent of each other, and 10,000 sampling simulations are conducted. In case of the probabilistic approach, as explained in Section 5.2, the time series of population both with and without the extreme event must be simulated in order to identify risk-based performance measures. Sample results, including the simulated fish population time series with and without the extreme event for one simulation, and for the average of all 10,000 simulations for a 450 cms flood that occurs in the fourth week of October, are shown in Figure 6-23.

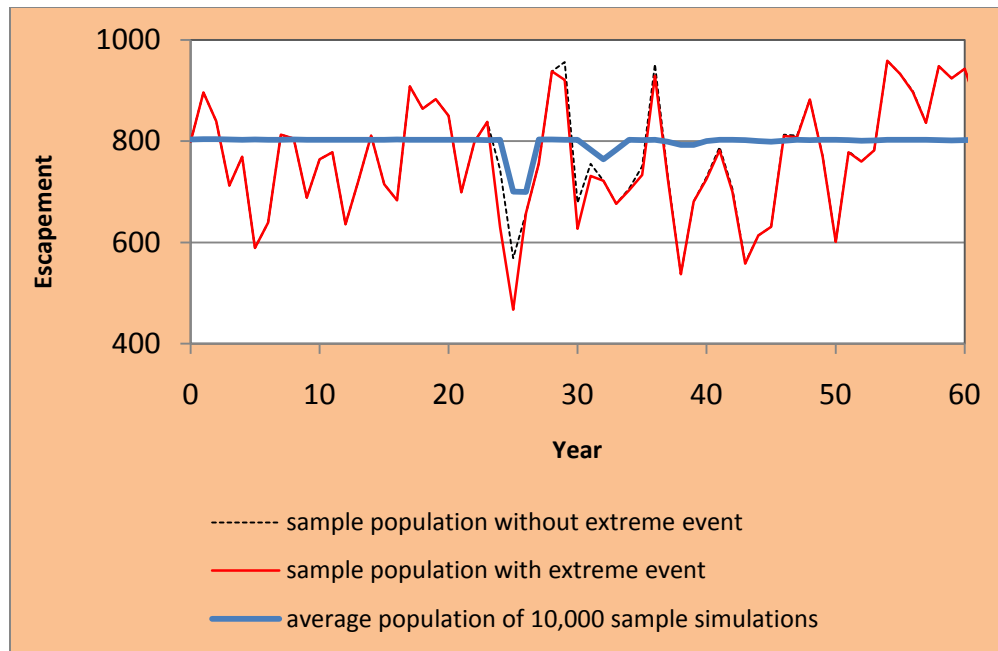


Figure 6-23. Fish population recovery model output with probabilistic approach: probabilistic ecological and immediate loss parameters for 450 cms flood in fourth week of October

The large inter-year variation of escapements, shown in Figure 6-23, is expected in real cases. For example, adult escapement estimates for the Lower Campbell River based on observed data, which are summarized in Table 6-8, also show significant inter-year variation (Bennet et al. 2010).

**Table 6-8. Adult escapement estimates to Lower Campbell River
(Bennet et al. 2010)**

Year	Escapement estimates
1999	1205
2000	791
2001	931
2002	1256
2003	568
2004	1199
2005	517
2006	1422
2007	650

Results of the probabilistic approach show that the extreme event increases the inter-year population variability. Such changes in population variability are summarized in Table 6-9. The coefficient of variation in fish population over ten years after the extreme events for each of 10,000 simulations is calculated. The average of 10,000 calculated coefficients for each flood is provided in Table 6-9. These results exhibit increased population variations for scenarios with extreme events compared with normal population variations (i.e., without the extreme event).

Table 6-9. Coefficient of variation of fish population over ten years after extreme event

Without extreme event (normal variation)	With extreme event		
	220 cms	450 cms	1073 cms or higher
15.5%	15.6%	16.4%	17.6%

6.3.2 Estimation of Risk-based Performance Measures

Risk-based performance measures of vulnerability, engineering and ecological resilience, and V/R for different flood intensities and dates of event are estimated and discussed herein. The deterministic, pseudo-probabilistic, and probabilistic approaches for fish population recovery modeling are investigated. In the deterministic approach, only one simulation is conducted to estimate the risk-based performance measures.

In all three approaches, the system is ecologically resilient, i.e., recovers to an equilibrium which is equal to that of the same system without an extreme event. The expected values of vulnerability and engineering resilience for the three approaches are identical. This is probably due to the fact that the probability distributions of the immediate fish and egg survival rates are almost symmetric and very narrow for this case study (see, e.g., Figure 6-14 and Figure 6-18).

Figure 6-24 integrates the vulnerability results of all extreme event intensity and date combinations in a two dimensional surface. Here, contours for expected values of vulnerability are demonstrated. Using the probabilistic modeling approach, uncertainty in these values may be expressed and is shown for three sample points as error bars which represent standard deviation of vulnerability. In general, for events occurring at a given time, uncertainty in the estimated vulnerability increases at higher flows. The variations in estimated vulnerability are more comprehensively addressed and shown in Figures 6-27 and 6.28. If risk-based performance measures are estimated to be employed in a probabilistic decision-making process, distribution of estimated performance measures (e.g., Figure 6-28) are required in addition to estimated expected values of such measures (e.g., Figure 6-24).

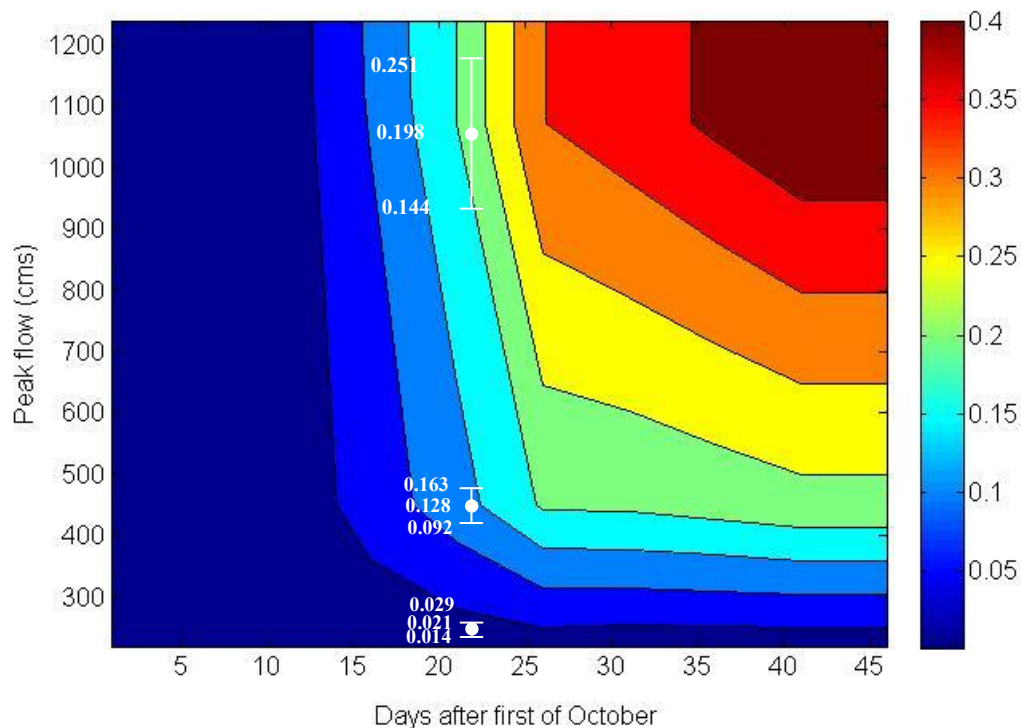


Figure 6-24. Expected values of vulnerability for different combinations of flood intensity and date. Mean, mean plus standard deviation, and mean minus standard deviation of sample points are shown.

Similar results for engineering resilience and V/R are shown in Figure 6-25 and Figure 6-26, respectively. However, since standard deviations of estimated engineering resilience are not significant, sample error bars are not demonstrated for expected values of engineering resilience in Figure 6-25.

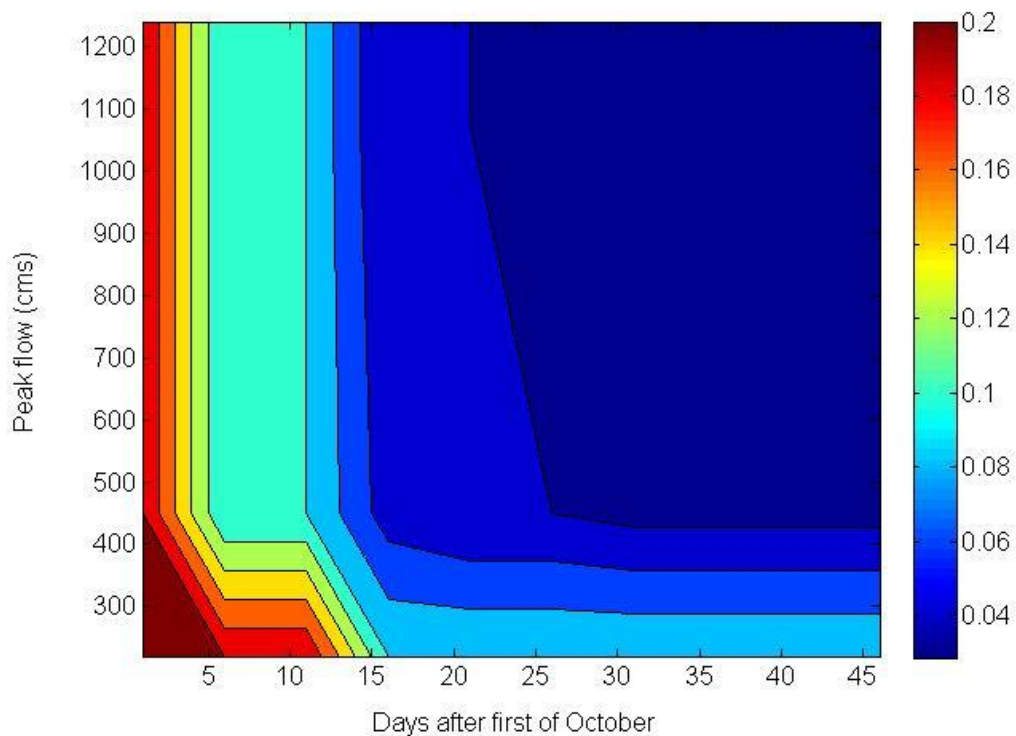


Figure 6-25. Expected values of engineering resilience for different combinations of flood intensity and date (year⁻¹)

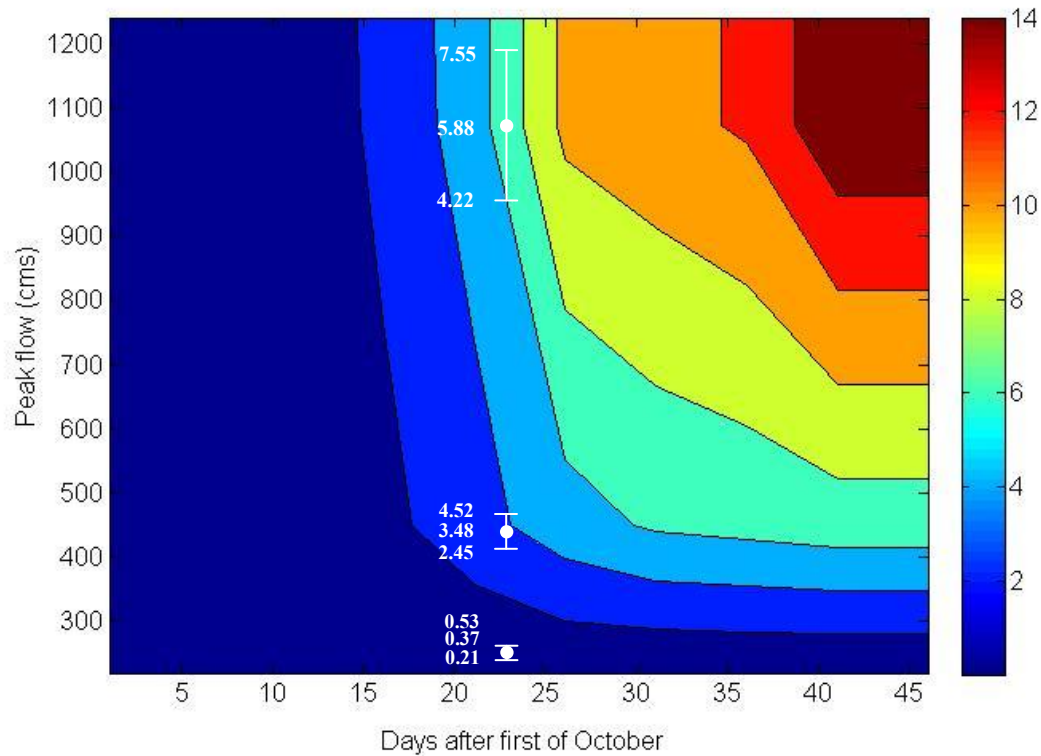


Figure 6-26. Expected values of V/R for different combinations of flood intensity and date (year). Mean, mean plus standard deviation, and mean minus standard deviation of sample points are shown.

Results of risk-based performance measures in Figures 6-24, 6-25, and 6-26 verify the dependency of these measures on both flood intensity and date of occurrence. V/R results, for example (Figure 6-26), are very sensitive to the extreme event occurrence date if it occurs between mid-October and mid-November. The contours are parallel and horizontal after mid-November until the end of February when fry emerge from the gravel. The time-independent dates (mid-November to end of February) are not shown in the Figures in order to maintain the high resolution of the Figures.

Likewise, Figures 6-24, 6-25, and 6-26 show that risk-based performance measures are constant for extreme floods of 1073 cms or higher, where the egg loss rate is 100%. However, the performance measures are very sensitive to the flood intensity for floods of less than 1073 cms during the incubation period (i.e., October-February) especially in and after the peak spawning season (November-February).

The comprehensiveness of these results may be compared with those of conventional approaches. In the Campbell River Water Use Plan studies (BC Hydro 2004), for example, a deterministic threshold of flow is assumed as the basis for comparing operation strategies with respect to impacts of extreme floods on fisheries. A threshold of 340 cms was suggested based on the empirical bed load transport equations. Operation strategies with flows beyond this threshold are considered to damage spawning habitat (i.e., a 0 or 1 logic). The results shown in Figure 6-26 indicate that 340 cms is a reasonable threshold estimate beyond which the fish population is negatively affected by the flow. However, it cannot be assumed that beyond this threshold, all redds are lost. Rather, comprehensive results of this framework, as shown in Figure 6-26, may be employed to estimate incremental impacts of floods with higher intensity.

Although deterministic, pseudo-probabilistic, and probabilistic approaches provide similar expected values for risk-based performance measures, the pseudo-probabilistic and probabilistic approaches further provide the histogram or probability distribution of these measures based on the results of the sampling simulations. This extra information, which is not available when using the deterministic approach, may be employed for evaluating the confidence level of estimates of the risk-based performance measures.

In the pseudo-probabilistic approach, engineering resilience results do not show any variation. However, some variation is observed in vulnerability results. These are provided as histograms for estimated vulnerability values given different floods in the fourth week of October in Figure 6-27.

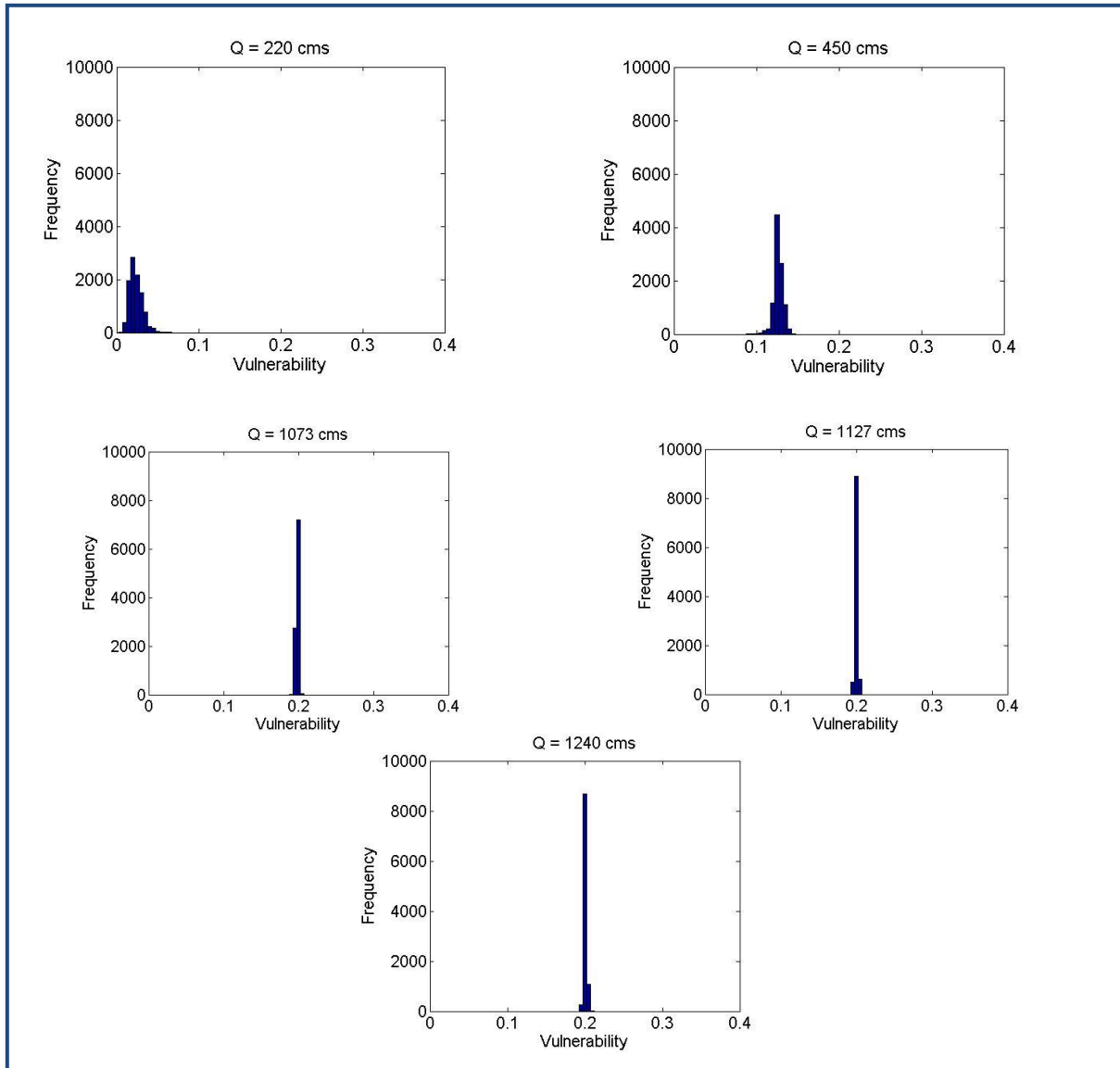


Figure 6-27. Histograms for estimated vulnerability with pseudo-probabilistic approach to data uncertainty for floods occurring in fourth week of October

If the probabilistic approach is employed, both vulnerability and engineering resilience results show variation. These are provided as histograms for estimated vulnerability and engineering resilience values given different floods in the fourth week of October in Figure 6-28 and Figure 6-29. Here, the risk-based performance measures exhibit much wider distributions in the probabilistic approach than in the pseudo-probabilistic approach. This is

because more uncertain parameters are considered in the probabilistic method than in the pseudo-probabilistic method.

It should be noted that accurate point prediction of the population is not the main objective of the fish population recovery model. Rather, it provides a tool to compare the performance measures of the system without and with extreme events (with different flow intensities and dates of occurrence). A more advanced model, which can reliably predict the population dynamics, yet may not necessarily account for impacts of extreme events, is not preferred.

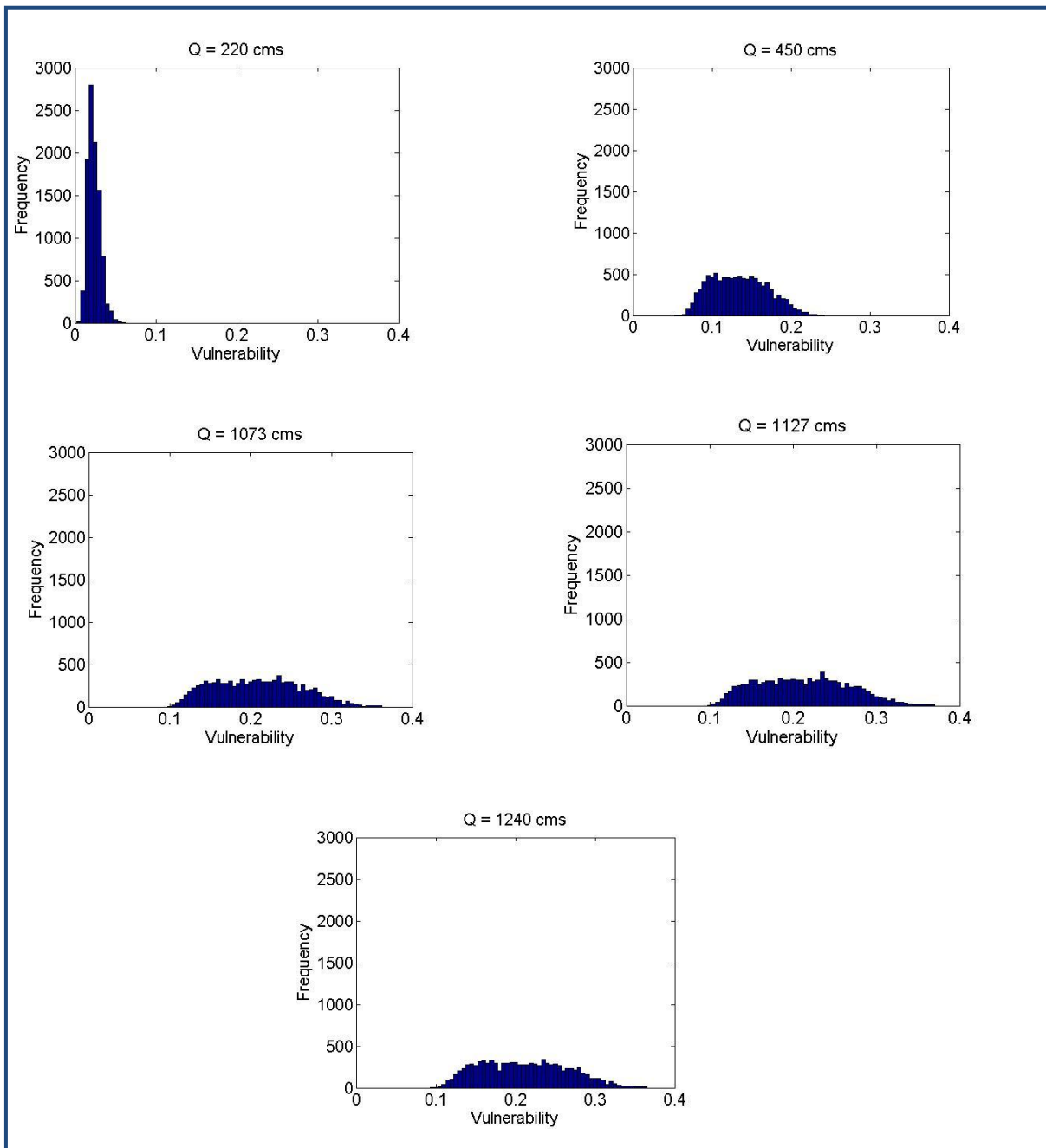


Figure 6-28. Histograms for estimated vulnerability with probabilistic approach to data uncertainty for floods occurring in fourth week of October

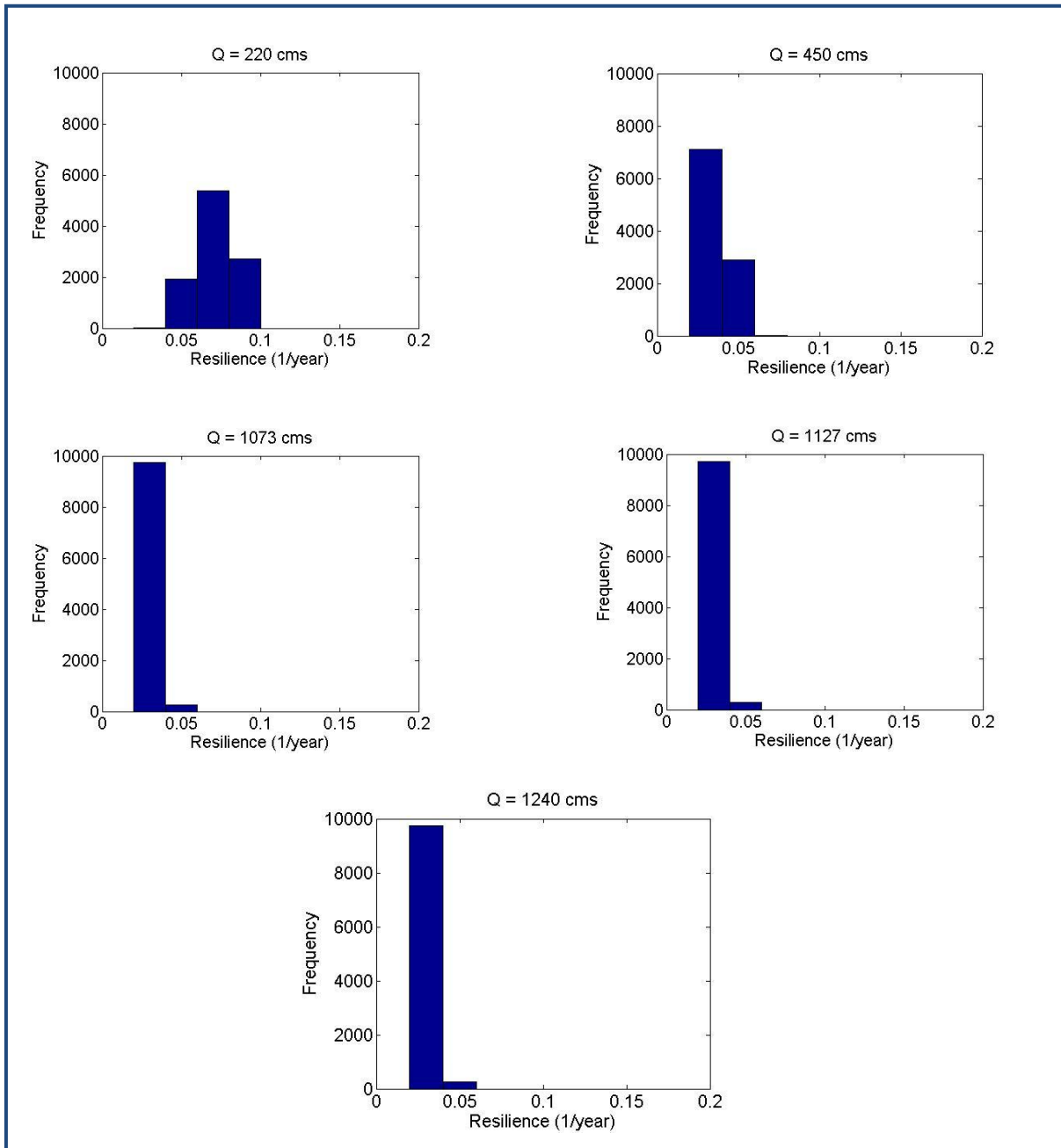


Figure 6-29. Histograms for estimated resilience with probabilistic approach to data uncertainty for floods occurring in fourth week of October

6.4 Further Considerations

Additional considerations include analysis of the relative importance of egg and fish loss for estimates of risk-based performance measures, impacts of multiple extreme events, uncertainty in depth of alevins, and sensitivity of risk-based performance measures to uncertainty in ecological parameters.

6.4.1 Relative Importance of Egg and Fish Loss

Both immediate fish and egg loss may contribute to the estimated risk-based performance measures. In the case study of the Campbell River, the immediate egg loss has a much higher impact on estimated risk-based performance measures than the immediate fish loss. Two reasons may be driving this difference. These are:

- Immediate egg survival rates are much less than immediate fish survival rates for all floods with peak flows of 450 cms or higher (Figure 6-16 and Figure 6-19); and
- When an extreme event happens, a maximum of 108 adult pairs may be present in the reach (Figure 6-5), and the flood does not affect the pairs that spawn before and after it occurs. However, all eggs that are deposited before the extreme event, and have survived the normal loss, are subject to impact by the flood.

Vulnerability, engineering resilience, and V/R of the case study as a result of immediate fish loss, immediate egg loss, and both are shown in Figures 6-30, 6-31, and 6-32, respectively, for different extreme events occurring in the fourth week of October.

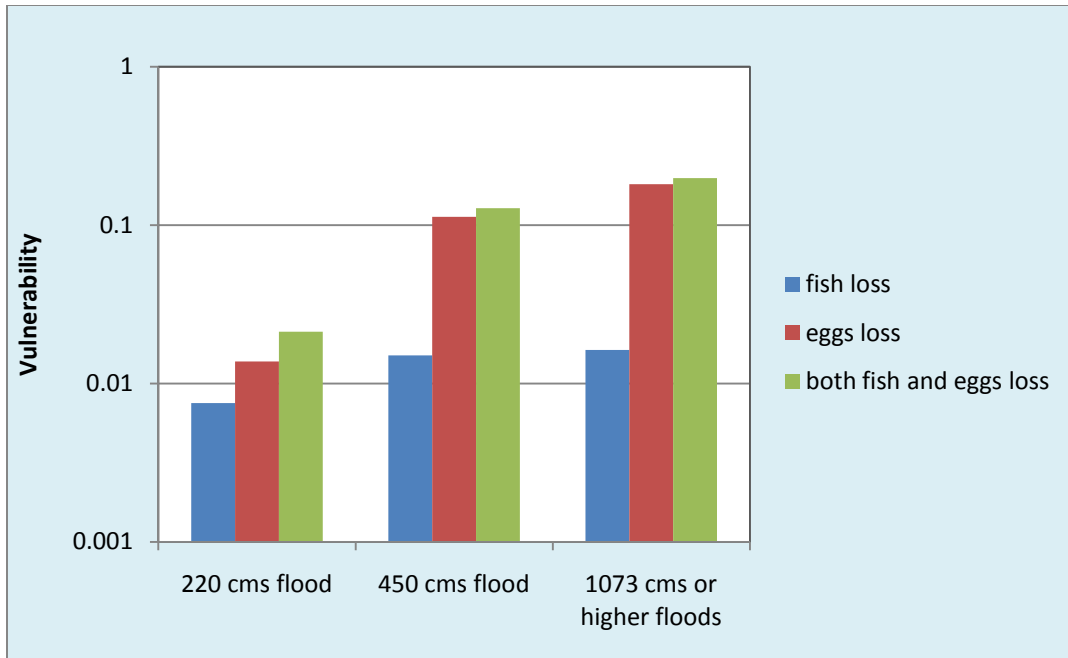


Figure 6-30. Vulnerability due to fish loss, egg loss, and both fish and egg loss for different floods occurring in fourth week of October

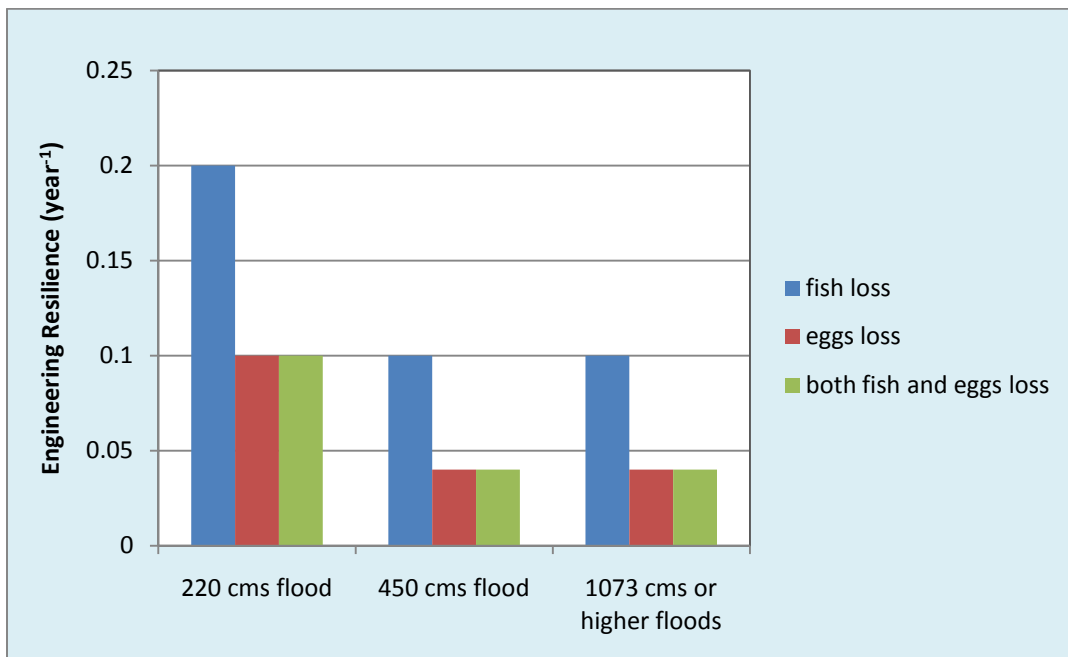


Figure 6-31. Engineering resilience due to fish loss, egg loss, and both fish and egg loss for different floods occurring in fourth week of October

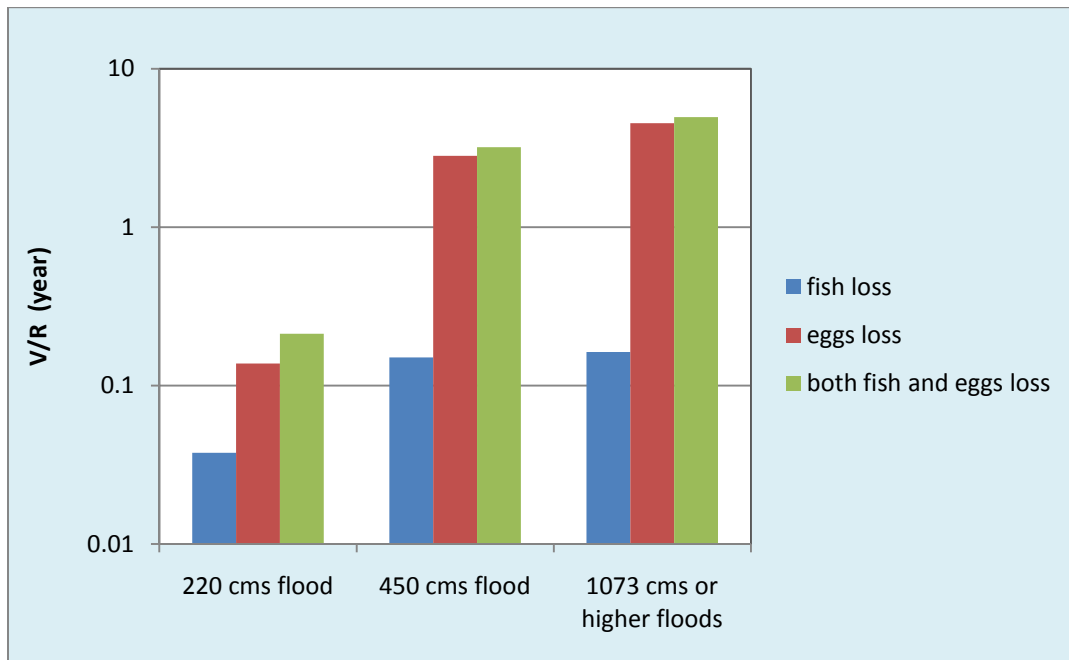


Figure 6-32. V/R due to fish loss, egg loss, and both fish and egg loss for different floods occurring in fourth week of October

Although these results suggest that for a given flood, performance measures are much more affected by egg loss than fish loss, they may not be generalized to other cases. Fish loss may be more important in systems with stream type salmon that may be washed out of their rearing habitat by flow velocities as low as 1-2 m/s. Also, in studying resident species, fish loss is an important factor because all of the fish stock is present in the reach, and subject to loss by the extreme event. Nevertheless, river reaches with resident fish may also be re-populated from the tributaries or by fish returning to their home reach after being washed downstream in the extreme event.

6.4.2 Impacts of Multiple Extreme Events

The developed framework may be utilized to estimate the impacts of multiple extreme events both within a year and across different years. Such a feature may be used in water resource planning projects where the occurrence probability of multiple floods is also of interest and must be estimated.

Two sample scenarios of two floods in series, both of which have a peak flow of 450 cms, are investigated, and the results are illustrated in Figure 6-33. Here, two floods occurring in two consecutive years increases vulnerability by 100% while two floods in the same year increases vulnerability by only 60% more than the single event.

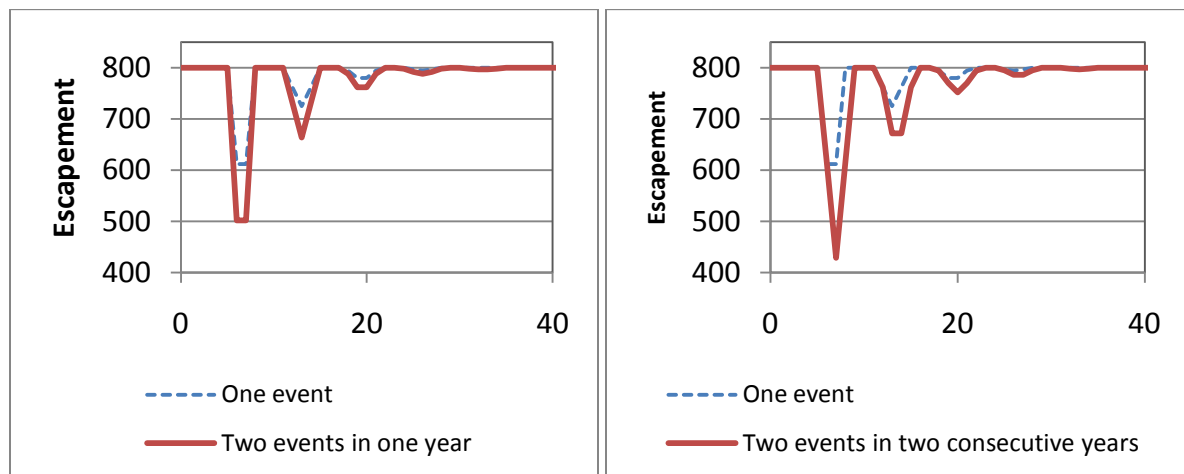


Figure 6-33. Impacts of multiple floods with peak flow of 450 cm

6.4.3 Uncertainty in Depth of Alevins

When eggs are hatched, alevins stay in the gravel bed but may move to a shallower depth. Risk-based performance measures are estimated for different floods occurring after the first of January when eggs start hatching. Different depths of alevins, from one to one-half of D_{redds} , are considered, and the results are shown in Figure 6-34. The results show that the uncertainty in depth of alevins does not make a significant difference in the estimated vulnerability. In estimating engineering resilience, such variations do not result in any change.

The insensitivity of these risk-based performance measures to the uncertainty in the depth of alevins in the case of 1073 cms and higher floods is obvious, because the mortality rate for the presumed D_{redds} , and for any shallower depth is 100%. Also, in the case of the 220 cms and 450 cms floods, most of the spawning cells experience either zero-valued scour/fill, or

scour/fill of 30 cm and higher. Therefore, changing the depth of alevins to shallower depths throughout the simulation does not affect the estimated survival at these cells.

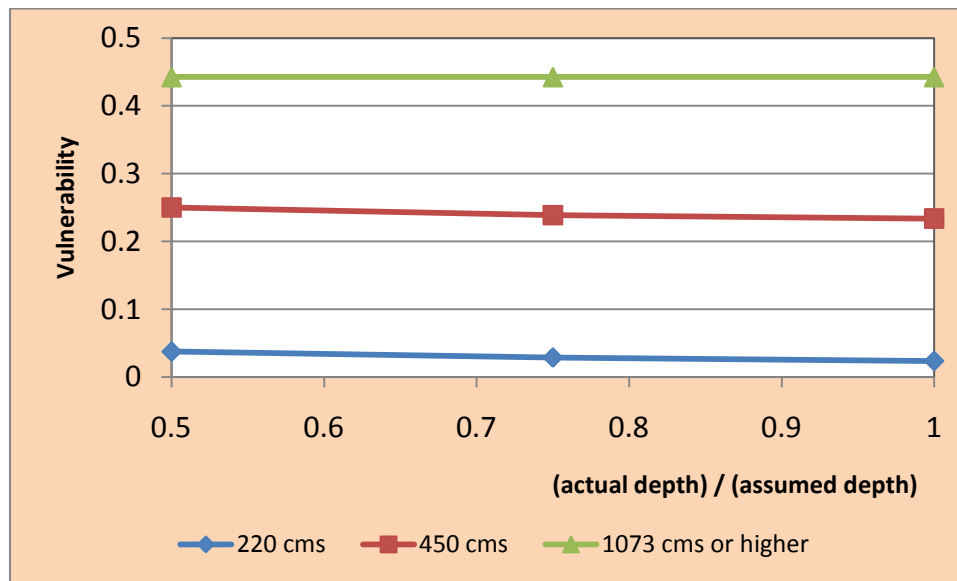


Figure 6-34. Sensitivity of vulnerability to actual depth of hatched alevins

6.4.4 Sensitivity to Uncertainty in Ecological Parameters

The probabilistic approach for fish population recovery modeling addresses the uncertainty in ecological parameters by evaluating distributions of risk-based performance measures. A sensitivity analysis (i.e., varying one parameter at a time) is performed herein to assess the affect of uncertainty in each of the ecological parameters on the estimated risk-based performance measures. Sample sensitivity analyses are performed for 10% variation of different ecological parameters for different floods occurring in the fourth week of October. Engineering and ecological resilience do not show sensitivity to any of these variations, though vulnerability does for some of them. The maximum change in estimated vulnerability for a ten percent variation (i.e., plus or minus ten percent) of the model parameters are reported in Table 6-10. These results show that estimated vulnerability is more sensitive to variation in fecundity and egg-fry survival rate than variation in egg-smolt survival rate and ocean survival rate. Likewise, they suggest that estimated vulnerability is more sensitive in floods with lower intensity. This is due to the fact that the results are provided in the form of

percentage change, and the absolute value of vulnerability is very low in the case of lower floods.

Table 6-10. Sensitivity of vulnerability to 10% variation of parameters for floods occurring in the fourth week of October (% of change in estimated vulnerability)

Parameters	Peak Flow		
	220 cms	450 cms	1073 cms or higher
Fecundity	12.2	7.0	5.3
Egg-fry survival rate	12.2	7.0	5.3
Egg-smolt survival rate	7.5	4.5	3.3
Ocean survival rate	7.5	4.5	3.3

It should be mentioned that the reported range of variation of these parameters are considerably different from each other (Table 6-2). Variation of estimated vulnerability within the reported range of ecological parameters in the Campbell River system (see Table 6-2), are demonstrated in Figure 6-35. Within these reported ranges, estimated vulnerability does not exhibit significant change due to variation in fecundity, though it changes with variations in other parameters.

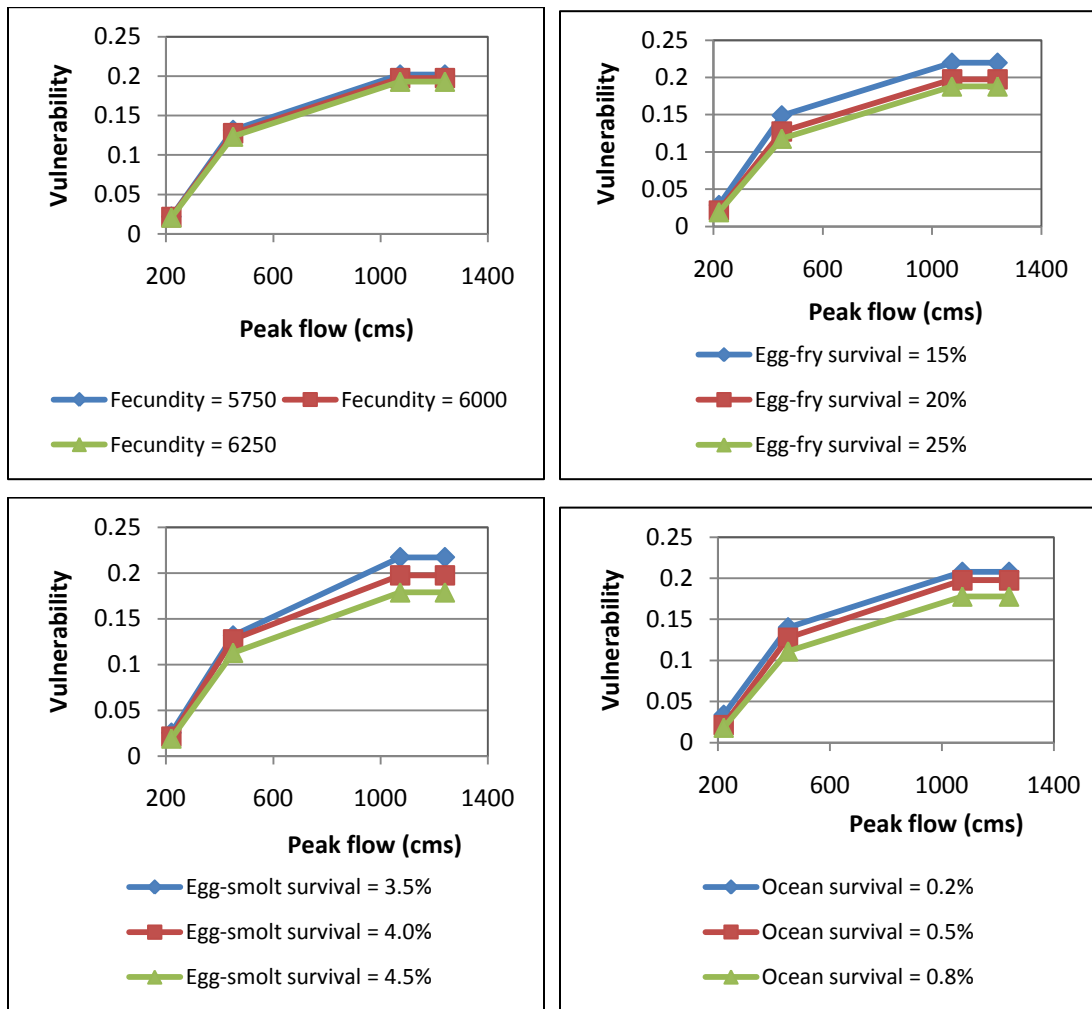


Figure 6-35. Variation of estimated vulnerability within the reported ranges of ecological parameters for floods occurring in the fourth week of October

Chapter 7: Conclusions

This research addresses the need for quantitative models to estimate immediate and long-term impacts of reservoir extreme outflows on downstream environments. This is achieved by developing several short- and long-term models that work together, yet at different temporal and spatial scales. The combination of numerical hydrotechnical models, an individual-based immediate fish survival estimation model, and an aggregated population dynamics estimation model are designed to interact consistently. In order to demonstrate the application of the general framework developed herein, it is applied to the case study of the Campbell River, for which all proposed risk-based performance measures are estimated.

The framework is intended to provide information for water resource decision-making processes in regulated rivers, such as real-time reservoir operation, participatory water use planning, and licensing and relicensing decisions for water resource projects. Such information are in the form of risk-based performance measures of the environment. The framework may be applied to any case study with anadromous species exhibiting stationary conditions before point 1 and after point 2 in Figure 1-2. Required parameters and input/output files of the models are described in the Appendix. The user needs to have access to hydrodynamic and morphodynamic models for the river system that they are analyzing, and to have data about some ecological parameters of the target species. These include swimming capacity, habitat suitability criteria, depth of redds, and life cycle parameters (e.g., egg-fry survival rate).

The objectives of modeling complex systems to support decision-making processes are generally to gain an understanding of the system and to predict the system response to changes in inputs and background conditions. When modeling such systems, it is necessary to start with reasonable simplifying assumptions and approaches in order to develop a practically applicable representation of the system. Once this simplified model is developed, more complex assumptions and processes may be advanced within the model. Examples of simplifications in the proposed framework include: considering a two-dimensional

swimming path; considering mortality mechanisms that may only be valid for anadromous species; and, for the time being, setting aside the impacts of water quality.

7.1 Contribution

This dissertation provides significant contributions to knowledge and practice in the field of water resource planning and management.

7.1.1 Developed Framework

The developed framework is based on hypotheses regarding the time trace of impacts of extreme events on downstream fish population dynamics; these are represented in Figure 1-2. The structure of the developed framework is considered a major step toward advancing reservoir operation and planning that includes holistic estimation of downstream environmental impacts. It provides the decision-makers with comprehensive information regarding short- and long-term impacts of both single and multiple high flow releases on the environment. The impact estimates may account for both intensity and timing of extreme events, and include incremental impacts of such events relative to normal loss under normal operation conditions. In existing methods of estimating environmental impacts of extreme events, the short- and long-term impacts are not distinguished, and impacts are usually estimated qualitatively.

7.1.2 Immediate Fish Survival Estimation Model

This model addresses the need for estimating fish response to extreme floods at a river reach scale. The individual-based model for immediate fish survival is a significant contribution to ecohydraulic modeling. It integrates the hydrodynamic variables and ecological behavior of individual fish under extreme event conditions. Since the model has a modular structure, it may easily be adapted for different species given their swimming capacity and habitat preference or HSC.

7.1.3 Risk-based Performance Measure V/R

The risk-based performance measure of V/R , which accounts for both short- and long-term impacts, is introduced and implemented through a case study. This provides decision-makers

with a quantitative measure that holistically estimates environmental performance of the system and advances existing qualitative performance measures. The concept of V/R may also be applied to other environmental attributes where both short- and long-term impacts are important, e.g., turbidity in lakes due to major landslides.

7.1.4 Immediate Egg Survival, Habitat Change, and Fish Population Recovery Models

Other sets of models are also developed for this framework. These include the: long-term fish population recovery, immediate egg survival estimation, and immediate habitat change estimation models. Although these models are developed by implementing minor modifications to existing models, the modifications are made so that all the models work consistently in terms of the temporal and spatial scales, and in terms of the modeling approach. For example, although developing a System Dynamics model for salmon population is not new, incorporating varying time steps (i.e., from five days to one year) in the model to build the capacity of working with short-term hydrotechnical models is a significant contribution. Also, evaluating the relative importance of immediate fish and egg loss, and habitat change, may guide decision-makers in searching for effective mitigation and restoration strategies.

7.2 Addressing Uncertainty

Models are purposeful misrepresentations of reality and are generally associated with uncertainty. Some models have the facility to address and provide output related to the uncertainty. Different sources of uncertainty have been considered in developing and implementing components of the proposed framework in this work. These are:

- In the immediate fish survival estimation model, uncertain fish swimming capacity and the initial location of fish are addressed with sampling procedures. In demonstrating this model for the Lower Campbell River system, it was shown that the estimated immediate survival rates are not significantly sensitive to the number of fish used to evaluate such estimates.

- The immediate egg survival estimation model accounts for uncertainty in the depth of redds. The model may be used to account for shallower depths of alevins than those of eggs. In the case of the Lower Campbell River, however, the results are not sensitive to such differences.
- In the fish population recovery model, depending on the choice of modeling approach, uncertain immediate fish and egg loss, and uncertain ecological parameters may be addressed.

If pseudo-probabilistic or probabilistic approaches of the fish population recovery model are employed, estimated risk-based performance measures may be reported not only in the form of expected values, but also in the form of uncertainty in such estimated measures. Also, sensitivity of estimated risk-based performance measures to uncertainty in presumed ecological parameters may be assessed.

Although the probabilistic approach appears to be more holistic than the pseudo-probabilistic and deterministic ones in addressing and reporting uncertainty, the choice of modeling approach is case specific. For example, if the system is not ecologically resilient, the probabilistic model may not be employed to estimate engineering resilience and V/R .

7.3 Applications

Given the ability of the framework to quantitatively address both short- and long-term impacts of extreme events, two major applications may be considered. These are:

7.3.1 Integration with Other Models of Downstream Impact Estimation

The Canadian and U.S. national guidelines (CDA 2007; FEMA 2004) and the British Columbia provincial regulation (BC Dam Safety Regulation 2000) call for estimating downstream impacts of dams in terms of life safety, economy, and environment. In order to conduct a comprehensive estimation, similar modeling approaches for estimating all of these three attributes may be attractive. Existing individual-based models to estimate life safety and economic impacts of extreme events may be integrated with this framework to implement such estimation.

For example, the BC Hydro Life Safety Model (BC Hydro 2005), which is individual-based, is a candidate for such integration. The BC Hydro Life Safety Model estimates human life safety and safety of objects (e.g., buildings and cars) by integrating the flood wave with the initial state of people, buildings, cars, and roads, and simulating the response of these objects to the wave. This is analogous to the immediate impact estimation models developed in this dissertation, which integrate the flood wave (i.e., hydrotechnical models output); the initial state of fish, eggs, and habitat; and the survival mechanisms for fish and eggs, to estimate fish and egg loss and habitat change.

7.3.2 Reservoir Planning and Operation

The developed framework may be applied in both planning and operation of reservoirs. At the macro-planning level, where multiple water resource projects are studied, MCDA (see Section 2.2.2.2) may be applied to compare the alternative reservoir designs and operating strategies. Existing MCDA approaches employ estimates of available habitat as environmental performance measures (see, e.g., Waddle et al. 1999). Despite their contribution as long-term environmental performance measures, habitat-based models may not estimate the impact of extreme events on the environment. Conventional approaches may only use logical (i.e., 0 or 1) or qualitative (i.e., high, medium, low) performance values that describe impacts of floods on redds and fish. If the developed framework is incorporated in the MCDA, quantitative estimates of risk-based performance measures (e.g., V/R) may be employed as performance values.

Likewise, systems analysis approaches to develop reservoir operation rules (see, e.g., Cardwell et al. 1996), may be improved by incorporating objective functions that minimize both short- and long-term impacts of the floods. Minimization of V/R may be considered in the objective function, along with other objective function terms, e.g., life safety impacts. This may involve investigation of several ramping scenarios under flooding conditions, which result in reservoir outflows with different hydrograph shapes, and therefore, in different impacts on downstream species.

Also, in reservoir design and planning stages, probability of occurrence of floods with different intensities may be integrated with their associated risk-based performance measures to select, for example, design controlled outflows.

Furthermore, since the results of risk-based performance measures may be estimated in the form of probability distributions, these results may be employed in probabilistic systems analysis approaches, e.g., reliability-based optimization of reservoir operation.

Finally, it should be emphasized that integrating environmental and life safety considerations in a comprehensive impact estimation model, and in systems analysis approaches for reservoir operation and planning, does not necessarily suggest a trade-off between these considerations. Rather, a general model should include components for addressing each of these considerations for different extreme events. These events may range from just over bankfull flow which may have environmental and economical loss and no life safety impacts, to a dam breach where life safety issues are dominant. A general model must have the capacity to address all possible outcomes.

7.4 Limitations

Two major assumptions of the proposed framework and models limit their generality. These are: applicability of adult fish mortality mechanism to anadromous species, and assumption of stationary conditions before point 1 and after point 2 in Figure 1-2. The framework would need to be modified before being applied to cases with resident fish species, or to non-stationary conditions, e.g., a dam breach case which may result in non-stationary geomorphology for several years.

This research provides a framework for estimating negative impacts of extreme events on the environments by estimating immediate fish and egg loss due to such events. However, positive impacts of floods, e.g., increased habitat diversity, are not properly addressed here. This is a major issue which must be considered in studying operational and mitigation strategies.

Habitat change estimation due to extreme events is proposed as a component of the environmental impact estimation framework (i.e., Figure 3-1) to provide estimates of carrying capacity values in the fish population recovery model. These data are ideally a set of post-event carrying capacity (e.g., maximum available spawning habitat) values. If post-event carrying capacities are significantly less than those of the pre-event conditions, the population may not recover to its pre-event equilibrium, and thus the system may not be ecologically resilient. In case of the Campbell River, the habitat change estimation model does not provide such information. Two factors cause this limitation. These are:

- In the Campbell River, available HSC are functions of depth and velocity only. Although this may be sufficient for the purpose of estimating immediate fish survival, it lacks ecologically important factors of habitat suitability, e.g., substrate composition and cover. Habitat measurements which consider more ecological factors, e.g., the habitat capacity metrics introduced by Cardwell et al. (1996), may be used to estimate the WUA as is shown in Figure 4-6. However, collecting data for such an approach in the study reach was outside the scope of this research.
- The habitat ecological suitability is better captured in mesohabitat scale estimations as suggested by Schwartz and Herricks (2008). This may be addressed in a future research project along with the approach suggested above, i.e., employing ecologically correlated habitat suitability metrics.

Even if the habitat estimation model is improved to reflect the aforementioned considerations, it may not fully address the relationship between available habitat and fish population. For the purposes of the models developed herein, however, it presents a reasonable starting point for estimating carrying capacity.

Conclusive recommendations for dam operation strategies may only be available if both negative and positive impacts of extreme events are considered, and other models in the proposed framework, which are not addressed in this dissertation (i.e., water quality issues, and impacts on non-fish life), are accounted for. A water quality model may be a critical component of the framework, e.g., in case of a tailings dam breach. Non-fish organisms

affect fish population through the food web, and depending on case-specific situations, may be crucial, e.g., in the case of rare or endangered species.

As mentioned in Chapter 2, data to verify impacts of extreme events on fisheries are limited to opportunistic samples taken before and after extreme events. For the Campbell River no sampling of fish has been conducted before and after floods. Published data for the cases with opportunistic sampling are generally sparse, or available in spatial scales which are not suitable for verifying an immediate impact estimation model.

Furthermore, it is assumed that swimming capacity ranges provided in Table 6-3 are valid for spawning adults. Nevertheless, these ranges may represent swimming capacities of adult chinook at different life stages, including adult chinook in the ocean. In this case, to account for the spawning condition of fish, lower swimming speeds may be most appropriate and should therefore be applied to better describe swimming capacity.

Moreover, it is assumed that emergent fry of ocean type chinook are not affected by floods, and will be moved to the estuary or the ocean where their rearing habitat is. Also, the effect of fish mortality due to collision with solid surfaces (including debris) has not been considered. Validation of these assumptions requires further research.

7.5 Future Research

Given the results and limitations of this research, several areas of future research are proposed. These are:

- Improving the developed models:
 - Incorporate non-uniform substrate material in the morphodynamic model: in the current model, uniform grain size and spatial distribution is considered. Not only may this assumption affect the results of sediment scour and fill, but it may also eliminate the possibility of utilizing ecological parameters (e.g., substrate material indices) in the HSC as explained in Section 7.4.

- Investigate fish mortality due to collision with solid surface and debris, and examine local velocity shelters that fish may use during extreme events. Such investigations may require three dimensional hydrodynamic modeling.
- Characterize fry mortality mechanisms and fish mortality mechanisms in the estuary during extreme events.
- Improve the immediate habitat change estimation model, and investigate the effect of immediate habitat change on risk-based performance measures.
- Modify both immediate and long-term estimation models to adapt for resident species, and for non-fish life.
- Investigate trade-offs between simplicity and accuracy by exploring simpler models (e.g., employing fewer parameters) and more complex models (e.g., considering three-dimensional fish movement).
- Develop new models to complete the Framework:
 - Develop both short- and long-term water quality models to estimate immediate and long-term changes in water quality constituents (e.g., turbidity) which are ecologically important. Both immediate and long-term fish and egg mortality mechanisms due to these changes should also be added to existing mechanisms.
 - Develop or adapt models to estimate both immediate and long-term impacts on riparian vegetation, invertebrates, and plankton.
- Verification:
 - Apply the framework in case studies with opportunistic pre- and post- event fish population data.
 - Examine validity of stationary conditions in different natural and regulated river systems.
- Reservoir operation and planning application:
 - Consider minimizing developed risk-based performance measures with systems analysis approaches to reservoir operation.
 - Employ developed risk-based performance measures as performance values in MCDA approaches to water resource systems planning.

- Explore the sensitivity of risk-based performance measures to different reservoir ramping rules (i.e., different shapes of hydrograph).
- Mitigation strategies:
 - Develop both operational and physical mitigation strategies not only by estimating the adverse environmental impacts but also by considering the positive long-term effects of high flows on fish habitat.

References

- Acreman, M. (2005). "Linking science and decision-making: Features and experience from environmental river flow setting." *Environmental Modelling and Software*, 20(2), 99-109.
- Adger, W. N. (2000). "Social and ecological resilience: are they related?" *Progress in Human Geography*, 24(3), 347-364.
- Balasubramaniam, A., and Voulvoulis, N. (2005). "The appropriateness of multicriteria analysis in environmental decision-making problems." *Environmental Technology*, 26(9), 951-962.
- Bartholow, J. (1996). "Sensitivity of Salmon Population Model to Alternative Formulations and Initial Conditions." *Ecological Modelling*, 88, 215-226.
- Bartholow, J., Heasley, J., Laake, J., Sandelin, J., Coughlan, B., and Moos, A. (2001). "SALMOD: a Population Model for Salmonids." U.S. Geological Survey, Fort Collins, Colorado.
- BC Dam Safety Regulation. (2000). "Water Act- British Columbia Dam Safety Regulation." B.C. Reg. 44/2000, Queen's Printer, Victoria, British Columbia, Canada
- BC Hydro. (2004). "Campbell River Water Use Plan." On behalf of The Consultative Committee for the Campbell River Water Use Plan, BC Hydro, British Columbia.
- BC Hydro. (2005). "BC Hydro Life Safety Model Formal Model Description." *E299 Wpr-2182*, BC Hydro.
- BCRP. (2000). "Bridge -Coastal fish and wildlife Restoration Program strategic plan, volume 2, chapter 2: Campbell River." http://www.bchydro.com/bcrp/about/docs/ch02_final.pdf.
- Beamish, F. W. H. (1978). "Swimming capacity." Fish physiology, W. S. Hoar and D. J. Randall, eds., Academic Press, New York, 101-187.
- Beck, M. B. (1996). "Transient Pollution Events: Acute Risks to the Aquatic Environment." *Water Science and Technology*, 33(2), 1-15.
- Bell, E., Duffy, W. G., and Roelofs, T. D. (2001). "Fidelity and Survival of Juvenile Coho Salmon in Response to Flood." *Transactions of the American Fisheries Society*, 130, 450-458.
- Bennet, J., Lewis, D., Van Will, P., and Tranholm, M. (2010). "Abundance, Age, Size, Sex, and Coded-Wire Tag Recoveries for Chinook Salmon Escapements of the Campbell and Quinsam Rivers, 1999-2007." Fisheries and Oceans Canada, South Coast Stock Assessment, Science Branch, Pacific Region.
- Bischoff, A., and Wolter, C. (2001). "The flood of the century on the River Oder: effects on the 0+ fish community and implications for floodplain restoration." *Regulated Rivers: Research & Management*, 17(2), 171-190.
- Bockelmann, B. N., Fenrich, E. K., Lina, B., and Falconer, R. A. (2004). "Development of an ecohydraulics model for stream and river restoration." *Ecological Engineering*, 22, 227-235.
- Borshchev, A., and Filippov, A. (2004). "From System Dynamics and Discrete Event to Practical Agent Based Modeling: Reasons, Techniques, Tools." *The 22nd International Conference of the System Dynamics Society*, Oxford, England.

- Bovee, K. D., Lamb, B. L., Bartholow, J. M., Stalnaker, C. B., Taylor, J., and Henriksen, J. (1998). "Stream Habitat Analysis Using the Instream Flow Incremental Methodology." *USGS/BRD-1998-0004*. viii, U.S. Geological Survey, Biological Resources Division.
- Bradford, M. J. (1995). "Comparative review of Pacific salmon survival rates." *Canadian Journal of Fisheries and Aquatic Sciences*, 52, 1327-1338.
- Bratrich, C., Truffer, B., Jorde, K., Markard, J., Meier, W., Peter, A., Schneider, M., and Wehrli, B. (2004). "Green Hydropower: a New Assessment Procedure for River Management." *River Research and Applications*, 20, 865-882.
- Burt, D. W. (2003). "Fisheries and Aquatic Resources of the Quinsam River System- A Review of Existing Information." BC Hydro, Nanaimo, BC.
- Burt, D. W. (2004). "Restoration of the Lower Campbell River, A Review of Projects to 2003 and a Plan for Future Works." Fisheries and Oceans Canada, Nanaimo, BC.
- Cada, G. F., Garrison, L. A., and Fisher, R. K. (2007). "Determining the effect of shear stress on fish mortality during turbine passage." *Hydro Review*, 26(7), 52-59.
- Cardwell, H., Jager, H. I., and Sale, M. J. (1996). "Designing Instream Flows to Satisfy Fish and Human Water Needs." *Journal of Water Resources Planning and Management*, 122(5), 356-363.
- CDA. (2007). "Dame Safety Guidelines." C. D. Association, ed., 82.
- Cushman, R. M. (1985). "Review of Ecological Effects of Rapidly Varying Flows Downstream from Hydroelectric Facilities " *North American Journal of Fisheries Management*, 5, 330-339.
- EatonvilleNews. (2011). "EatonvilleNews.net, Retrieved April 15, 2011." <http://www.eatonvillenews.net/thewaterfront.html>.
- Evenson, D. F. (2001). "Egg Pocket Depth and Particle Size Composition within Chinook Salmon Redds in the Trinity River, California," Humboldt State University, Arcata, CA.
- Ewart, D. (2010). "Personal Communication." Campbell River, BC.
- Ewart, D., and Anderson, S. (2010). "Personal communication." DFO, Campbell River, BC.
- FEMA. (2004). "Federal Guidelines for Dam Safety: Hazard Potential Classification System for Dams." U. S. D. O. H. S. F. E. M. AGENCY, ed., 21.
- Fisheries and Oceans Canada. (2009). "Salmonids in the Classroom - Intermediate: a Teachers' Resource for Studying the Biology, Habitat and Stewardship of Pacific Salmon." Community Involvement Division, Fisheries and Oceans Canada, Vancouver, BC.
- Flug, M. (1997). "Ecosystem Resource Considerations in Reservoir Management." *Journal of Contemporary Water Research and Education*, 108, 43-56.
- Ford, A. (1999). *Modeling the Environment - An Introduction to System Dynamics Modeling of Environmental Systems*, Island Press.
- Gallopin, G. C. (2006). "Linkages between vulnerability, resilience, and adaptive capacity." *Global Environmental Change*, 16, 293-303.
- Garcia, A., Jorde, K., Habita, E., Caaman, D., and Parra, O. (2011). "Downstream Effects of Dam Operations: Changes in Habitat Quality for Native Fish Species." *River Research and Applications*, 27, 312-327.
- Gippel, C. J. (2001). "Australia's environmental flow initiative: Filling some knowledge gaps and exposing others." *Water Science and Technology*, 43(9), 73-88.

- Glawdel, J., Kwan, S., Naghibi, A., Millar, R., and Lence, B. J. (2011). "Using River2D Morphology (R2DM) to Predict Salmon Redd Survival During High Flow Events from Hydroelectric Dam Operations." *World Environmental and Water Resources Congress*, Palm Springs, CA.
- Goodwin, R. A., Nestler, J. M., Anderson, J. J., and Weber, L. J. (2007). "A new tool to forecast fish movement and passage." *Hydro Review*, 26(4)(4), 58-71.
- Goodwin, R. A., Nestler, J. M., Loucks, D. P., and Chapman, R. S. (2001). "Simulating Mobile Populations in Aquatic Ecosystems." *Journal of Water Resources Planning and Management*, 127(6), 386-392.
- Grimm, V. (1999). "Ten years of individual-based modelling in ecology: what have we learned and what could we learn in the future?" *Ecological Modelling*, 115, 129-148.
- Grimm, V., and Wissel, C. (1997). "Babel, or the ecological stability discussions: an inventory and analysis of terminology and a guide for avoiding confusion." *OECOLOGIA*, 109, 323-334.
- Gunderson, L. H. (2000). "Ecological resilience - In theory and application." *Annual Review of Ecology and Systematics*, 31, 425-439.
- Gunderson, L. H., Holling, C. S., and Light, S. S. (1995). "Barriers and bridges to the renewal of ecosystems and institutions ", Columbia University Press, New York, 593.
- Hartford, D. D., and Baecher, G. B. (2004). *Risk and Uncertainty in Dam Safety*, Thomas Telford, London.
- Hashimoto, T., Stedinger, J. R., and Loucks, D. P. (1982). "Reliability, Resiliency, and Vulnerability Criteria for Water Resource System Performance Evaluation." *Water Resources Research*, 18(1), 14-20.
- Healey, M. C. (1991). "Life history of chinook salmon (*Oncorhynchus tshawytscha*)." Pacific Salmon Life Histories, C. Groot and L. Margolis, eds., UBC Press, Vancouver, 311-391.
- Higgs, D. A., Macdonald, J. S., Levings, C. D., and Dosanjh, B. S. (1995). "Nutrition and feeding habits in relation to life history stage." Physiological Ecology of Pacific salmon, C. Groot, L. Margolis, and W. C. Clarke, eds., University of British Columbia Press, Vancouver, BC, 159-315.
- Holling, C. S. (1973). "resilience and stability of ecological systems." *Annual Review of Ecology and Systematics*, 4, 1-23.
- Holling, C. S. (1996). "Engineering Resilience versus Ecological Resilience." Engineering within Ecological Constraints, P. C. Schulze, ed., National Academy Press, Washington, DC, 213.
- Jessup, B. K. (1998). "A strategy for simulating brown trout population dynamics and habitat quality in an urbanizing watershed." *Ecological Modelling*, 112, 151-167.
- Johnstone, W. M., and Lence, B. J. (2009). "Assessing The Value Of Mitigation Strategies In Reducing The Impacts Of Rapid-Onset, Catastrophic Floods." *Journal of Flood Risk Management*, 2, 209-221.
- Johnstone, W. M., Sakamoto, D., Assaf, H., and Bourban, S. (2005). "Architecture, Modelling Framework and Validation of BC Hydro's Virtual Reality Life Safety Model." International Symposium on Stochastic Hydraulics (ISSH 2005), Nijmegen, Netherlands.

- Jonkman, S. N., vanGelder, P. H. A. J. M., and Vrijling, J. K. (2002). "Loss of life models for sea and river floods." *Flood Defence 2002*, W. a. e. al., ed., Science Press, New York, 196-206.
- Keeney, R. L., and Raiffa, H. (1976). *Decisions with Multiple Objectives: Preferences and Value Trade-Offs* John Wiley & Sons Inc., U.S.A.
- King, J., Brown, C., and Sabet, H. (2003). "A scenario-based holistic approach to environmental flow assessment for rivers." *River Research and Applications*, 19(5-6), 619-639.
- King, J. M., Tharme, R. E., and Villiers, M. (2000). "Environmental Flow Assessment for Rivers: Manual for the Building Block Methodology." Freshwater research unit, University of Cape Town.
- Klohn Leonoff. (1989). "Floodplain Mapping Program- Campbell and Quinsam Rivers." British Columbia Ministry of Environment.
- Korman, J., Bravender, B., and Levings, C. D. (1997). "Utilization of the Campbell River Estuary by Juvenile Chinook Salmon (*Oncorhynchus tshawytscha*) in 1994." Fisheries and Oceans Canada, Nanaimo, BC.
- Lacey, R. W. J., and Millar, R. (2004). "Reach Scale Hydraulic Assessment of Instream Salmonid Habitat Restoration." *Journal of the American Water Resources Association*, 40(6), 1631-1644.
- Lapointe, M., Eaton, B., Driscoll, S., and Latulippe, C. (2000). "Modelling the probability of salmonid egg pocket scour due to floods." *Canadian Journal of Fisheries and Aquatic Sciences*, 57, 1120-1130.
- Larinier, M. (2001). "Environmental Issues, Dams and Fish Migration." Dams, Fish and Fisheries: Opportunities, Challenges and Conflict Resolution, G. Marmulla, ed., FAO Fisheries Technical Paper, Rome.
- Leake, A. (2004). "Campbell River Flow-habitat Modeling with River 2D." BC Hydro.
- Lence, B., Johnstone, W., and Naghibi, A. (2009). "Impacts of Hydropower-induced Flows: Estimation and Mitigation of Human and Environmental Risks." Canadian Dam Association Annual Conference, Whistler, BC, Canada, 15-28.
- Levings, C. D., McAllister, C. D., and Chang, B. D. (1986). "Differential use of the Campbell River estuary, British Columbia, by wild and hatchery-reared juvenile chinook salmon (*Oncorhynchus tshawytscha*)." *Canadian journal of Fisheries and Aquatic Sciences*, 43, 1386-1397.
- Lind, N., Hartford, D., and Assaf, H. (2004). "Hydrodynamic Models of Human Stability in a Flood." *JAWRA Journal of the American Water Resources Association*, 40, 89-96.
- Lojkasek, B., Lusk, S., Halacka, K., Luskova, V., and Drozd, P. (2005). "The impact of the extreme floods in July 1997 on the ichthyocenosis of the Oder Catchment area (Czech Republic)." *Hydrobiologia*, 548(1), 11-22.
- Loomis, J. B. (2000). "Environmental valuation techniques in water resource decision making." *Journal of Water Resources Planning and Management*, 126(6), 339-343.
- Maier, H. R., Lence, B. J., Tolson, B. A., and Foschi, R. O. (2001). "First-Order Reliability Method for Estimating Reliability, Vulnerability, and Resilience." *Water Resources Research*, 37(3), 779-790.
- Marmulla, G. e. (2001). "Dams, fish and fisheries. Opportunities, challenges and conflict resolution." FAO Fisheries Technical Paper, Rome.

- May, C. L., Pryor, B., Lisle, T. E., and Lang, M. (2009). "Coupling hydrodynamic modeling and empirical measures of bed mobility to predict the risk of scour and fill of salmon redds in a large regulated river." *Water Resources Research*, 45, 1-22.
- McClelland, D. M., and Bowles, D. S. (2002). "Estimating Life Safety Loss for Dam Safety Risk Assessment - a Review and New Approach." Institute for Water Resources, U.S. Army Corps of Engineers, Logan, Utah.
- McDaniels, T. L. (1996). "A multiattribute index for evaluating environmental impacts of electric utilities." *Journal of Environmental Management*, 46(1), 57-66.
- Milhous, R. T., Updike, M. A., and Schneider, D. M. (1989). "Physical habitat simulation system reference manual: Version II " 89(16), U.S. Fish and Wildlife Service Biological, Washington, DC.
- Naghibi, A., Lence, B., Glawdel, J., and Millar, R. (2011). "A Framework for Estimating Downstream Environmental Impacts of Reservoir Extreme Outflows." *World Environmental and Water Resources Congress*, Palm Springs, CA.
- Nagtegaal, D. A., Riddell, B., Lehmann, S., Ewart, D., and Adkins, B. (2000). "Assessment of Campbell/Quinsam Chinook Salmon ", C. S. A. Secretariat, ed.
- Neitzela, D. A., Daublea, D. D., Cada, G. F., Richmonda, M. C., Guenscha, G. R., Muellera, R. P., Abernethya, C. S., and Amidana, B. (2004). "Survival Estimates for Juvenile Fish Subjected to a Laboratory-Generated Shear Environment." *Transactions of the American Fisheries Society* 133, Issue 2(2), 447-454.
- Nestler, J. M., Goodwin, R. A., and Loucks, D. P. (2005). "Coupling of Engineering and Biological Models for Ecosystem Analysis." *Journal of Water Resources Planning and Management*, 131(2), 101-109.
- Nislow, K. H. (2002). "Within-Basin Variation in the Short-Term Effects of a Major Flood on Stream Fishes and Invertebrates." *Journal of Freshwater Ecology*, 17(2), 305-318.
- Orth, D. J., and Maughan, O. E. (1981). "Evaluation of the "Montana method" for recommending instream flows in Oklahoma streams." *Proceedings of the Oklahoma Academy of Science*, 61, 62-66.
- Pinault, J.-L., and Berthier, F. (2007). "A methodological approach to characterize the resilience of aquatic ecosystems with application to Lake Annecy, France." *Water Resources Research*, 43(1), W01418.
- Pires, A. M., Magalhães, M. F., Costa, L. M. D., Alves, M. J., and Coelho, M. M. (2008). "Effects of an extreme flash flood on the native fish assemblages across a Mediterranean catchment." *Fisheries Management and Ecology*, 15(1), 49-58.
- Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegard, K. L., Richter, B., Sparks, R., and Stromberg, J. (1997). "The natural flow regime: a new paradigm for riverine conservation and restoration." *BioScience*, 47, 769-784.
- Railsback, S. (1999). "Reducing Uncertainties in Instream Flow Studies." *Fisheries*, 24(4), 24-26.
- Railsback, S., Stauffer, H. B., and Harvey, B. (2003). "What can habitat preference models tell us? Tests using a virtual trout population." *Ecological Applications*, 13(6), 1580-1594.
- Railsback, S. F. (2000). "Instream Flow Assessment Methods: Guidance for evaluating Instream Flow Needs in Hydropower Licensing." Electric Power Research Institute (EPRI), CA.

- Railsback, S. F., Harvey, S. K., Jakson, S. K., and Lamberson, R. H. (2009). "inSTREAM: The individual-based Stream Trout Research and Environmental Assessment Model." US Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA.
- Sale, M. J., Brill, E. D., and Herricks, E. E. (1982). "An approach to optimizing reservoir operation for downstream aquatic resources." *Water Resources Research*, 18(4), 705-712.
- Scholl, H. (2001). "Agent Based and System Dynamics Modeling: A Call for Cross Study and Joint Research." *34th Annual Hawaii International Conference on System Sciences (HICSS-34)*, Maui, Hawaii 3003.
- Schwartz, J. S., and Herricks, E. E. (2005). "Fish use of stage-specific fluvial habitats as refuge patches during a flood in a low-gradient Illinois stream." *Canadian Journal of Fisheries and Aquatic Sciences*, 62, 1540-1552.
- Schwartz, J. S., and Herricks, E. E. (2008). "Fish use of ecohydraulic-based mesohabitat units in a low-gradient Illinois stream: implications for stream restoration." *Aquatic Conservation: Marine and Freshwater Ecosystems*, 18, 852-866.
- Stalnaker, C. B., and Arnette, J. L. (1976). "Methodologies for the determination of stream resource flow requirements: An Assessment." Office of Biological Services, F.W.S.
- Stalnaker, C. B., Lamb, B. L., Henriksen, J., Bovee, K., and Bartholow, J. (1995). "The Instream Flow Incremental Methodology: A Primer for IFIM." Biological Report 29, Washington, DC: U.S. Geological Survey.
- Steffler, P. M., and Blackburn, J. (2002). "River2D: Two-dimensional depth-averaged model of river hydrodynamics and fish habitats." University of Alberta, Edmonton, Alberta.
- Suen, J.-P., Eheart, J. W., Herricks, E. E., and Chang, F.-J. (2009). "Evaluating the Potential Impact of Reservoir Operation on Fish Communities." *Journal of Water Resources Planning and Management*, 135(6), 475-483.
- Tennant, D. L. (1976). "Instream Flow Regimens for Fish, Wildlife, Recreation and Related Environmental Resources " *Fisheries*, 1(4), 6-10.
- Tharme, R. E. (1996). "Review of International Methodologies for the Quantification of the Instream Flow Requirements of Rivers." the South African Department of Water Affairs and Forestry: Pretoria. Freshwater Research Unit: University of Cape Town, South Africa.
- Tharme, R. E. (2003). "A global perspective on environmental flow assessment: Emerging trends in the development and application of environmental flow methodologies for rivers." *River Research and Applications*, 19, 397-441.
- Tiffan, K. F., Haskell, C. A., and Kock, T. J. (2008). "Quantifying the behavioral response of spawning chum salmon to elevated discharges from Bonneville Dam, Columbia river." U.S. Department of Energy, Bonneville Power Administration, Environment, Fish and Wildlife Department, Portland, OR.
- Vasquez, J. A., Millar, R. G., and Steffler, P. M. (2007). "Two-dimensional finite element river morphology model." *Canadian Journal of Civil Engineering*, CSCE, 34(6), 752-760.
- Waddle, T. J. (2001). "PHABSIM for Windows User's Manual and Exercises " *01-340*, U.S. Geological Survey Open-File
- Waddle, T. J., Bowen, Z. H., and Bovee, K. D. (1999). "Including Long-Term Biological Index Performance in a Multi-Criteria Decision Support System." Preparing for the

21st century: proceedings of the 26th annual Water Resources Planning and Management Conference, American Society of Civil Engineers, Tempe, AZ 1-10.

Waddle, T. J., Steffler, P., Ghanem, A., Katopodis, C., and Locke., A. (2000). "Comparison of One and Two-dimensional Open Channel Flow Models for a Small Habitat Stream" *Rivers*, 7(3), 205-220.

Yusuf, F. (2011). "Personal Communication." BCHydro, Burnaby, BC.

Appendices

Appendix A User's Guide

This appendix provides instructions for implementing the computer programs developed for the immediate fish survival estimation and the fish population recovery models. These are general models that may be applied to river systems that meet the assumptions described in Section 3.4 in the thesis (e.g., assuming anadromous species under stationary conditions). The immediate egg survival rates which are estimated in this thesis are calculated using a spreadsheet approach which compares sampled depth of redds with actual depths of scour and fill as explained in Section 4.2.2.

There are some data and tool requirements for implementation of the models. These are:

- MATLAB codes for the pre-processor (*preprocessor.m*), main program (*main.m*), and post-processor (*postprocessor.m*) modules of the immediate fish survival estimation model; and the fish population recovery model (*salmon.m*). These codes are maintained at the Department of Civil Engineering of the University of British Columbia³, and are available upon request⁴.
- MATLAB software to execute the programs- basic MATLAB skills are also required.
- Multiple-core processors are highly recommended to increase the computational speed in the main program module of the immediate fish survival estimation model.
- Results of a transient two-dimensional hydrodynamic model for the river system of interest (e.g., River2D) under different flood hydrographs. These results should be in the form of time series of depth and specific discharge in both longitudinal, x , and lateral, y , directions for all simulation cells of the river system. These should also include x and y coordinates of all simulation cells.
- Swimming capacity, Habitat Suitability Criteria (HSC), and life cycle parameters (e.g., normal egg-fry loss rate) for the target species.

The user needs to execute the pre-processor, main program, post-processor, and fish population recovery programs in sequence. The pre-processor and main program must be

³ <http://www.civil.ubc.ca/people/faculty/faculty-lence.php>

⁴ lence@civil.ubc.ca

executed for a number of different scenarios of swimming capacity for any given flood (i.e., for outputs of hydrodynamic modeling of a given flood). In this version of the code, five scenarios of swimming capacity are considered. This can be adapted for the user's needs. The output files of these five implementations of the main program should then be integrated into an input file for the post-processor module, which produces the histogram of immediate fish survival rate for a presumed flood. Such results may be used by the fish population recovery model to estimate the environmental risk-based performance measures for this particular flood, and a given date of occurrence. This may be repeated for any arbitrary date of flood by re-executing the fish population recovery model for those dates of occurrence. However, in order to estimate the risk-based performance measures under another flood, the entire modeling process must be repeated. This includes implementing all modules of the immediate fish survival estimation and fish population recovery models. An overview of input and outputs, and parameters of each of these programs is provided herein.

For some parameters of each code, example values are suggested in square brackets to be used as generic values in the absent of specific data for the system being analyzed. When such suggested values are not provided in square brackets, specific data for the system are required to be collected by the user.

A.1 Pre-processor of Immediate Fish Survival Estimation Model ('preprocessor.m')

Given the outputs of a two-dimensional hydrodynamic model (e.g., River2D) of the river system (i.e., coordinates of simulation nodes and time series of depth, and specific discharge in both x and y directions for all simulation nodes), and HSC and swimming capacity data specified by the user, the pre-processor provides information for the main program module of the immediate fish survival estimation model. This information includes time series of depth, x and y components of velocity, magnitudes of velocities, and Habitat Suitability Indices (*HSI*) for all simulation cells, and also the identified neighboring cells and potential washed away cells (i.e., all potential cells to which a fish may be washed from its current location) for each simulation cell.

Input files of 'preprocessor.m'

Input files of the pre-processor are comma separated files which may be made from output of a hydrodynamic model. These input files include:

- '*nodes.csv*': This file has n rows for n simulation nodes in the hydrodynamic model, and three columns. The first column contains the node numbers, and the second and third columns show x and y coordinates of the nodes, respectively.
- '*depth_matrix.csv*': This file has $n+1$ rows where n is the number of simulation nodes in the hydrodynamic model. Number of columns is equal to the number of time steps at which the hydrodynamic model output values are recorded. The first row of the matrix contains times (sec) at which depth values are reported by the hydrodynamic model.
- '*qx_matrix.csv*': This file has the same format as '*depth_matrix.csv*' listing x components of specific discharge instead of depth.
- '*qy_matrix.csv*': This file has the same format as '*depth_matrix.csv*' listing y components of specific discharge instead of depth.

Parameters in 'preprocessor.m'

In this version of the code, ramp and trapezoidal shapes are used for depth and velocity suitability criteria (components of the HSC), respectively (Figures A-1 and A-2). However, more complex shapes may be adapted.

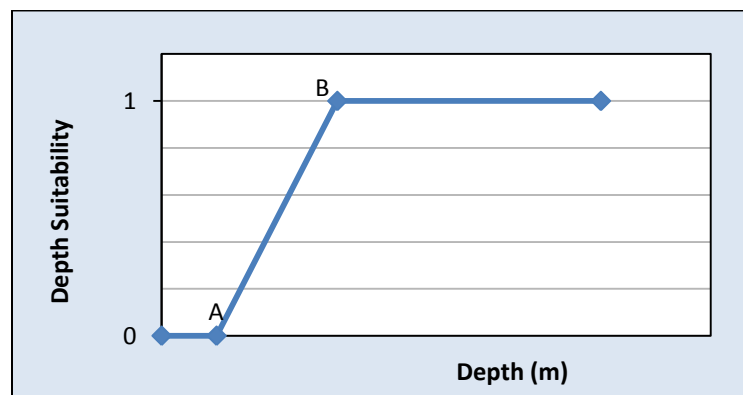


Figure A-1. Typical ramp shape for depth suitability criteria. Depths of points A and B for target species should be defined by user.

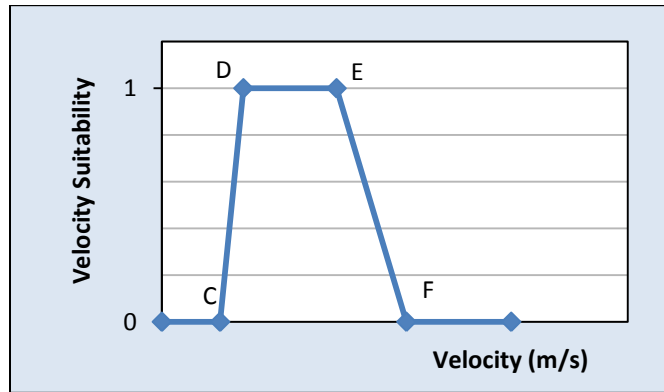


Figure A-2. Typical trapezoidal shape for velocity suitability criteria. Velocities of points C, D, E, and F for target species should be defined by user.

Swimming Capacity

- ‘*sust_vel*’: sustained swimming speed of target species (m/s).
- ‘*prol_vel*’: prolonged swimming speed of target species (m/s).
- ‘*burs_vel*’: burst swimming speed of target species (m/s).

Habitat Suitability Criteria

- ‘*depth_low*’: flow depth below which depth suitability is zero (m), i.e., depth of point *A* in Figure A-1.
- ‘*depth_high*’: flow depth above which depth suitability is one (m), i.e., depth of point *B* in Figure A-1.
- ‘*vel_low*’: flow velocity below which velocity suitability is zero (m/s), i.e., velocity of point *C* in Figure A-2.
- ‘*vel_mid_low*’: velocity of point *D* in Figure A-2 (m/s).
- ‘*vel_mid_high*’: velocity of point *E* in Figure A-2 (m/s).
- ‘*vel_high*’: flow velocity above which velocity suitability is zero (m/s), i.e., velocity of point *F* in Figure A-2.

Time Increments and Geometry

These parameters show the format (i.e., temporal and spatial scale) of inputs and outputs.

- ‘*timestep*’: time step of fish movement simulation (sec)
- ‘*input_dt*’: time step of the input files (sec), e.g., in ‘*depth_matrix.csv*’.
- ‘*initial_time*’: initial time in the input files (sec), e.g., in ‘*depth_matrix.csv*’.
- ‘*r2d_times*’: number of columns in the input files, e.g., in ‘*depth_matrix.csv*’.
- ‘*max_timestep*’: maximum time step in the input files, e.g., in ‘*depth_matrix.csv*’.
- ‘*max_node*’: number of simulation cells

Control

These parameters are used for computational stability and efficiency.

- ‘*depth_zero*’: minimum positive depth for velocity calculation (m). In order to avoid numerical instability, cells with less depth than this value are considered dry [0.01 m].
- ‘*max_flood_speed*’: maximum flood velocity (m/s). This is used to calculate the maximum distance that a fish may be washed during one time step.
- ‘*max_neighbours*’: maximum number of neighbors to be considered when comparing *HSI* of neighboring cells [50].
- ‘*max_wash_neighbours*’: maximum number of wash neighbors to be considered [500].

Output files

The pre-processor module does not create output files, and the results (i.e., time series of both x and y components of velocity, magnitudes of velocities, depth, and *HSI*, and the identified neighboring cells and potential washed away cells for each simulation cell) are only stored in the computer RAM. Therefore, the main program module must be executed before the computer RAM is cleared.

A.2 Main Program of Immediate Fish Survival Estimation Model ('*main.m*')

The main program of the immediate fish survival estimation model identifies whether an individual fish (with a given swimming capacity) which is initially located at any of the potential initial locations in the river reach can survive a given flood. The location of an individual fish is traced during the flood, and fatigue and stranding conditions are monitored.

Input files

No input file is required for this module. However, it must be executed after the pre-processor module before the computer RAM is cleared.

Parameters in '*main.m*'

- '*max_prol_time*': maximum prolonged swimming time (sec) [12000 sec].
- '*max_burst_time*': maximum burst swimming time (sec) [10 sec].
- '*min_depth*': minimum depth (m) for stranding calculation [0.05 m].
- '*max_timestep*': number of simulation time steps
- '*ds_cells*': number of the cells located at the most downstream end of the river reach. If a fish moves to these cells, it is assumed to be washed out of the reach. For example, the first 200 cells may be selected.
- '*fish_no*': number of cells with positive initial *HSI*.

Output files

Output files of the main program is a comma separated file '*results_matrix.csv*' with nine columns and *fish_no* rows where *fish_no* is the number of cells with positive *HSI* (this could be found by examining the initial values of the *HSI* time series). The nine columns include:

- initial location of fish
- final location of fish
- total displacement during the flood
- number of time steps over which the fish was washed
- survival indicator (1: if survives; 0: if lost)

- fatigue indicator (1: if lost due to fatigue; 0: otherwise)
- stranding indicator (1: if lost due to stranding; 0: otherwise)
- washed away indicator (1: if washed out of the reach; 0: otherwise)
- total survival time

A.3 Post-processor of Immediate Fish Survival Estimation Model (*'postprocessor.m'*)

The post-processor module of the immediate fish survival estimation model integrates the results of the pre-processor and the main program for five different swimming capacity scenarios with a sampling process and produces the distribution of the immediate fish survival rate for a given flood.

Input files

Before running the post-processor, the pre-processor and the main program must be executed for several swimming capacities. In the current version of the model, five swimming capacity scenarios of very low, low, medium, high, and very high are specified. Given the output files of these five scenarios (i.e., *'results_matrix.csv'*) the input file for the post-processor module (i.e., *'cell_suit_survival5.csv'*) is then created by the user. This file is a lookup table containing seven columns and *fish_no* rows where *fish_no* is number of cells with positive *HSI*. The first column shows the initial location of fish (i.e., in cells with positive initial *HSI*). The second column represents the pre-flood *HSI* of the initial location. The other five columns are survival indicators of these cells (1: if survives; 0: if lost) taken from the fifth column of the output files of the main program for five scenarios of very low, low, medium, high, and very high swimming capacities.

Parameters in *'postprocessor.m'*

- *'max_cells'*: number of cells with initial positive *HSI*
- *'sample_size'*: number of fish in each sample [100]
- *'sample_number'*: total number of samples [10,000]

Output files

The output file of the post-processor module (i.e., '*survival_rate_fish.csv*') is a vector of simulated survival rates for all *sample_number* of samples (e.g., a vector of 10,000 simulated survival rates). The histogram of this vector may represent the distribution of the immediate fish survival rate.

A.4 Fish Population Recovery Model ('*salmon.m*')

This code may implement all three approaches for fish population recovery modeling (i.e., the deterministic, pseudo-probabilistic, and probabilistic approaches), and estimates the risk-based performance measures, given a flood intensity and date of occurrence. Such estimates may be in the form of expected values (in the case of the deterministic approach) or of distributions (in the case of the pseudo-probabilistic and probabilistic approaches).

Input files

The input files of this model include a large number of simulated immediate fish and egg survival rates:

- '*survival_rate_fish.csv*': This is the output file of the post-processor module of the immediate fish survival estimation model, which is a vector of simulated immediate fish survival rates for a given flood.
- '*survival_rate_eggs.csv*': This is a vector of simulated immediate egg survival rates which could be calculated by comparing sampled depth of redds with actual depths of scour and fill as explained in Section 4.2.2 of this thesis. This vector has the same as '*survival_rate_fish.csv*'.

In the case of the deterministic approach for estimating risk-based performance measures, '*survival_rate_fish.csv*' and '*survival_rate_eggs.csv*' vectors will conflate to scalar numbers (i.e., expected values of immediate fish and egg survival rates, respectively).

Parameters in 'salmon.m'

For parameters with maximum and minimum values, if identical values are selected for these two parameters, the parameter will be deterministic; otherwise its value will be sampled from a uniform distribution between the minimum and maximum.

- *'life_cycle'*: length of the life cycle (years), e.g., 5 years.
- *'male_female'*: ratio of female to total adults [0.50]
- *'initial_adults'*: initial number of adults at the beginning of the simulation. An arbitrary number may be assumed [1000].
- *'fecundity_min'* and *'fecundity_max'*: minimum and maximum values of fecundity.
- *'normal_egg_fry_loss_min'* and *'normal_egg_fry_loss_max'*: minimum and maximum normal egg-fry loss rates.
- *'cc_min'* and *'cc_max'*: minimum and maximum smolts carrying capacities.
- *'S'*: smolts survival rate if their density is very low, e.g., 0.5.
- *'ocean_loss_min'* and *'ocean_yr_loss_max'*: minimum and maximum values of normal ocean loss rates.
- *'fishing_loss_min'* and *'fishing_loss_max'*: minimum and maximum values of fishing rate.
- *'stray_origin_adults_min'* and *'stray_origin_adults_max'*: minimum and maximum numbers of returning adult fish which originated in another river.
- *'max_sample'*: number of simulation samples [10,000]
- *'max_year'*: maximum years of simulation for each sample. This is control parameter to identify ecological resilience [100 years].
- *'tolerance'*: minimum significant difference between scenarios with and without the extreme event. This is a control parameter to identify ecological resilience [0.002].
- *'event_year'*: extreme event year (years after the beginning of simulation) [30 years or more]. This is specified to ensure stationarity of population prior to the extreme event.
- *'event_month'*: the month in which the extreme event occurs, e.g., 10 for October

- ‘*five_day_frac*’: fraction of the monthly population of spawning fish which are present in the river during a five-day window around the flood occurrence date. For example, if the flood occurs during the first five days of October; 20 fish spawn in the river reach during these five days; and 400 fish spawn during the month of October; this parameter should be 0.05, i.e., 20/400. If the extreme event occurs in a month when no spawning fish is present, this parameter should be zero.
- ‘*spawn_month(j)*’: fraction of total yearly spawning that occurs during the *j*th month of year
- ‘*egg_out_month(j)*’: fraction of eggs that hatch in during the *j*th month of year
- ‘*fry_out_month(j)*’: fraction of fry that grow to smolt during the *j*th month of year
- ‘*smolts_out_month(j)*’ : fraction of smolts that migrate to the ocean during the *j*th month of year

Output files

The output file of the fish population recovery model (i.e., ‘*risk.csv*’) contains series of simulated sample risk-based performance measures (e.g., vulnerability) which may also be represented in the form of distributions of such risk-based performance measures. In the case of the deterministic approach, such series reduce to single estimates for performance measures.