POPULATION RESPONSES OF COHO AND CHINOOK SALMON TO SEDIMENTATION ASSOCIATED WITH FOREST ROADS IN A COASTAL WATERSHED OF THE LOWER FRASER RIVER

by

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Abstract

In British Columbia, one of the main negative impacts on salmonid habitat is the production of fine sediments generated by forest roads or other human activities. Given this concern, this study's main objective was to develop a quantitative framework for estimating effects of extreme suspended-sediment events caused by forest road construction and use on populations of chinook (*Oncorhynchus tshawytscha*) and coho salmon (*Oncorhynchus kisutch*) in a medium-sized coastal watershed of the lower Fraser River. The framework incorporates existing knowledge of sediment production by forest roads to make a quantitative link between traffic levels and physiological responses of salmonids. The results suggest that extreme sedimentation events generated by heavy traffic levels negatively affect populations of chinook and coho salmon. Population numbers declined proportionally to the elevated levels of suspended sediments concentrations (SSC) following a non-linear trend in which Chinook salmon are more vulnerable to the deleterious effects of SSC than coho salmon.

Key Words: Population dynamics of salmonids, mixed-effects model, effects of suspended sediments on fish, stochastic life history model, Lower Fraser River salmonid mortality.

Dedication

This work is dedicated to Aidan, Maja, Gloria, Cielo, Lilia, and the rest of my family.

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Introduction

Evaluating the benefits society obtains from ecosystems provides a framework to achieve effective resource use and measure trade-offs among management scenarios. In multiple-use watersheds, one ecosystem service often occurs at the expense of another, just as consumption of resources by a sector of society may come at the expense of consumption by other sectors. This dilemma is characteristic of watersheds of South Western British Columbia where forest resource extraction often affects the terrestrial and aquatic habitat quality, and preserving valuable habitat has a negative impact on the amount of timber that can be extracted. Assessing trade-offs among uses offers a simple and viable way of quantifying the benefits society obtains from ecosystems while providing alternatives for conservation. In this study, I use three research sections to explore the connections between the sedimentation produced by logging roads and the salmonid population response to sediments in a coastal watershed of the Lower Fraser River, the Chilliwack River.

Chapter 1 describes a mixed-effects model that extrapolates suspended sediments from discharge data. Chapter 2 provides a model to quantify the daily suspended sediment loads generated by forest roads under different conditions of construction and use. In addition, this model permits isolating background (or natural) sedimentation from that produced by forest roads, utilizing hydrological time series data. Chapter 3 employs a population dynamics model to estimate the effects of extreme suspended-sediment events on populations of Chinook and coho salmon.

The ultimate objective of this study is to develop a quantitative framework for estimating the trade-offs between forestry activities, specifically logging road construction and use, and the Chinook and Coho fishery catches in a generic medium-size coastal watershed of the lower Fraser. I address uncertainty in the parameter estimates by evaluating the populations' response under different forest road management scenarios using a Monte Carlo simulation approach. The forest road management scenarios represent different forest road densities and use intensities. By using these scenarios I intended to capture the effect of road construction and use independent of other factors that may affect population abundances such as hatchery production, migration, and ocean-related conditions.

Chapter 1:
Estimating Suspended Sediment Concentrations in Areas with Limited Hydrological Data using a Mixed-effects Model

1.1 Introduction

Suspended sediment dynamics are important components of many physical, chemical, and biological processes in rivers and streams. Although there are positive factors associated sediment transport and deposition for aquatic organisms, many studies have linked increased suspended sediments over background levels with negative consequences for aquatic biota (Kerr, 1995). Most attempts to conserve and protect aquatic ecosystems from the deleterious effects of suspended sediments have focused their attention on the need to develop sediment control strategies for stream restoration and watershed management (e.g., Reiter et al., 2009). With respect to river ecology, it is important to characterise the movement of fine sediment within the fluvial system and identify the natural and human forces that drive this movement. For example, forest harvesting activities can lead to an increase in sediment supply and hence change the sediment texture of the streambed (Nistor, 1996; Nistor and Church, 2005). In particular, the accelerated production and mobilization of fine material can increase its proportion in the channel and impact the quality of habitat for fish and other aquatic species. Suspended sediments also reduce drinking water quality, which might pose health problems and increase the cost of water supply to urban communities (Reiter, 1998; Salant and Hassan, 2008).

Several studies of suspended sediment dynamics have shown that the bulk of the sediment is transported by single-flood events, usually of short duration and high magnitude, and that the relationship between the suspended sediment load and water discharge is highly variable (e.g., Walling and Webb, 1987; Salant et al., 2008). This variability is usually displayed by greater inter-event and intra-event differences, more noticeable for small and medium size streams than for large lowland rivers (Lenzi and Marchi, 2000). Different patterns of hysteresis in the relationship between suspended sediment and water discharge are related to the types and locations of active sediment sources (Lenzi and Marchi, 2000). In British Columbia and the Pacific North West of USA three main sources of fine sediment can be observed in forested streams: (1) episodic release due to mass movement events that reach the stream channels, (2) bank erosion during and between floods, and (3) release of sediment stored behind log jams. Very little sediment is contributed from erosion of the landscape surface by overland flow, and from within the bed (Nistor, 1996). Although fine sediments have been the focus of many studies, the current knowledge of sediment sources transfer and storage is inadequate to address the wide set of environmental effects caused by suspended sediments in forested landscapes.

The lack of an integrated physical understanding of the relations between water discharge and fine sediments is especially prevalent in the literature. Predictions of fine sediment transport rate via hydraulically-based functional relations are often more than an order of magnitude different than observed rates. This discrepancy has been explained by the major role that fine sediment supply plays on sediment dynamics in rivers and streams. Predicting suspended sediment by using channel hydraulics places an upper limit on how much fine material can be carried, although there is rarely enough sediment to meet a river's carrying capability. Sediment rating curves are the most common method used to estimate the suspended sediment load in streams with limited sediment concentration data (e.g. Walling, 1974; Walling, 1977; Church and Gilbert, 1975). However, the performance of sediment rating curves is often low because suspended sediment transport is a function of sediment supply more than of water discharge. Any variation in the sediment supply of a watershed results in shifts in the relationship between water discharge and suspended sediments, thus introducing variability (frequently in the form of hysteresis), which is not often captured by the rating curve.

The presence of a hysteresis effect leads to a non-unique relation between suspended sediment concentrations and water discharge. There are a number of ways to deal with the problem of hysteresis effects such as by the development of separate rating curves for rising and falling limbs of the hydrograph (e.g., Walling, 1974; Walling and Webb, 1987) or the inclusion of a sediment supply parameter in the model (cf. Asselman, 1999). Researchers have developed various modifications that include grouping the suspended sediment data according to seasonal or hydrological patterns, developing watershed specific correction factors, or using non-linear regression models (Bunte and MacDonald, 1999; Horowitz, 2003). These attempts to deal with sediment hysteresis have been limited in success because they fail to address the problem of the sediment supply and thus the changing relationship between water discharge and suspended sediment concentration (*SSC*). Furthermore, poor data quality and lack of continuous measurements inhibit progress toward development of reliable suspended sediment models for better management of stream ecosystems. This fact demonstrates the need for suspended sediment transport models that explicitly include hysteresis effects in a way that hydrologists can make use of the often limited hydrological data. Nevertheless, rating curves remain the main practical method for estimating suspended loads in streams.

An alternative way to approach sediment hysteresis effects is to model the relation between suspended sediment concentration (*SSC*) and discharge (*Q*) using a mixed-effects linear model. A mixed-effects model incorporates a mixture of fixed and random effects. Fixed effects are associated with the variability between groups of an entire population, where the modeler decides on the grouping (Pinheiro and Bates, 2000). For example, a hydrologist can determine the grouping in relation to sections of a hydrological cycle that share similar characteristics. Random effects associated with the variability within groups (Pinheiro and Bates, 2000). In this case, random effects are associated with the variability between years on the same month (e.g., January 1970, January 1971…January 1976), which may be related to hysteresis patterns, climatic influences, and changes in the sediment sources. The mixed-effects model approach allows using subsetable time that accounts for inter-year seasonality and hysteresis, especially in watersheds with clearly defined discharge regimes.

The purpose of this study was to present a novel methodology to estimate suspended sediment concentrations (*SSC*) for rivers and streams with limited hydrological data. To do this, I employed a mixed-effects model, which allows accounting for seasonality and variability introduced by hysteresis in the short term and heterogeneity in the long term (e.g. more variation during the winter and early summer than during the late summer and fall). To test whether or not the mixed-effects model performs better than the rating curve, represented by a generalized least squares regression (*gls*), I compared the estimates of *SSC* using discharge (*Q*) as the predictor variable resulting from applying both methods.

1.2 Methods

1.2.1 Errors in the Relationship *Q-SSC*

Prior to fitting a mixed-effects model, it is important to understand that estimation of sediment loads from temporal records of suspended sediment concentrations (*SSC*) and discharge (*Q*) are subject to errors that arise from field and laboratory methods and data processing. Suspended sediment rating curves have the form of power function:

$$SSC = aQ^b \tag{1}$$

where *SSC* is the suspended sediment concentration, *Q* is the water discharge, and *a* and *b* are regression constants. If the discharge data are reasonably accurate, we can expect measurement errors of the order of 5-10% (e.g., Smith et al., 2003 a, b). Errors in the suspended sediment estimates are related to different aspects of the methods of data collection, quality and frequency of sediment sampling and laboratory processing of samples (Webb et al., 1997; Phillips et al., 1999; Smith et al., 2003a, b). Mweempwa (1993) analyzed the sampling techniques employed by the Water Survey of Canada (WSC) and concluded that it presents reliable estimates contained within ±10% accuracy for the Chilliwack River. Specific hydrometric stations can also provide an idea of the quality of data they produce. For example, the standard error of the mean annual load for station Chilliwack River at Vedder Crossing (station number 08MH001, discussed below) ranged between 16.2% and 43.0% from 1965 to 1976 (Environment Canada, 1992). Other sources or errors may arise from the data format. When dealing with time series data, it is important to anticipate the likelihood of encountering temporal correlations in the structure (and mixed-effects models assume that data are independent and normally distributed). Data can also be irregularly spaced, leaving "blanks" that can bias parameter estimation in the model.

Prior to fitting a mixed-effects model, it is advisable that the researcher performs an exploratory data analysis, chooses the grouping in the data set, and decides on the potential structure of random effects (e.g. varying intercept, varying slope, or both). After fitting the model, it is essential to evaluate the residuals and conduct diagnostics for homogeneity, normality, and independence (see addressing autocorrelation structure below), as well as incorporate variance and covariance structures to reassess the fitted model.

1.2.2 Model Formulation

There are several fundamental assumptions for the application of linear mixed-effects models. Crawley (2007) emphasizes the normality and independence in observations between groups and within groups, and that the covariance matrix needs to be independent of the group. However, in time series, temporal correlation is expected, thus violating the assumption of independence. For this reason it is necessary to evaluate the correlation structures within groups containing data (e.g. a week of continuous *SSC* observations). In addition, the error variance depends on the temporal scales, which can change depending on the way the researcher divides the time series (Crawley, 2007). Once the autocorrelation structures have been addressed (discussed in more detail below), we can proceed with the linear mixed-effects model formulation. When evaluating temporal *Q*-*SSC* relationships, the experimental units are groups within which the observations of *SSC* and *Q* are made, which are specific to each situation. Thus, the researcher needs to define the grouping structure depending on the patterns present in the data (e.g., type of hydrological cycle) and considering factors such as data resolution, and the research needs. Following is a description of the model formulation employed using the data from the case study.

In the case study, there are two levels of variation in the data: groups (same month in different years) and observations nested within groups (individual measurement of *SSC* in and *Q* in each month measured in mg L^{-1} and m^3 seg^{-1}, respectively). In this particular study we employed 12 mixed-effects models, one for each month across all years in the data set. The reason for formulating 12 mixed-effects models was to account for seasonal variability from month to month while acknowledging within-month variability from year to year. In each model corresponding to a month (e.g., mixed-effects model for June), observations between years (for that month) are independent, but observations within year (in that month) are correlated (Pinheiro and Bates, 2000). In this manner, years represent the fixed effects along with an intercept and an SSC-discharge slope (Figure 1.1). Random slopes and intercepts were associated with the variability among the same month across years (i.e. June 1969 versus June 1970), often characterized by hysteresis patterns, changes in the sediment sources, and climatic factors (see Figure 1.2 and Figure 1.3). The software used to fit the mixed-effects models was the *lme* library in R, version 2.10.2 (Pinheiro et al., 2010).

Although there are several ways to formulate a linear mixed-effects model, we used different equations for the two levels of nested data (Singer, 1996; Snijders and Bosker, 1999; Lai and Helser, 2004; Zuur et al., 2009). The first level describes the variability in SSC as a function of discharge (*Q*) within each month, and the second level describes the variability in the *Q*-*SSC* relationship among years in the same month (e.g. variability among the Januaries of the data set). The first level is represented by:

$$y_{ij} = \beta_{0j} + \beta_{1j} x_{ij} + \varepsilon_{ij}, \qquad \varepsilon_{ij} \sim N(0, \sigma^2) \qquad (2)$$

where $y_{ij} = \log_e(SSC+1)$, $x_{ij} = \log_e(Q+1)$ for an observation i in year j, and β_{0j} and β_{1j} are the intercept and slope respectively. The +1 is used to avoid conflicts in the logarithmic transformation when zeros were present in the data. The data transformation is necessary and commonly used when addressing *Q-SSC* to ensure normality. The random error ε_{ij} represents the within-month variance across groups and it is assumed to be independent and identically normally distributed with mean equal to zero and common variance σ^2. The second level is represented by:

$$\beta_{0j} = \beta_0 + b_{0j} \quad , \qquad \beta_{1j} = \beta_1 + b_{1j} \tag{3}$$

$$b_j = \begin{bmatrix} b_{0j} \\ b_{1j} \end{bmatrix} \sim N\left(\begin{bmatrix} 0 \\ 0 \end{bmatrix}, \xi \right) \qquad \xi = \begin{bmatrix} \sigma_0^2 & \sigma_{01} \\ \sigma_{01} & \sigma_1^2 \end{bmatrix}$$

where the intercept and slope β_{0j} and β_{1j} are assumed to be multivariate normally distributed with mean (β_0, β_1) and variance-covariance matrix ξ representing the between-year variance and covariance for the vector of random effects b_j, which describes the variation of the *Q-SSC* relationship in each month among years. Substitution of (3) into (2) leads to the final linear mixed-effects model:

$$y_{ij} = \beta_0 + b_{0j} + \beta_1 x_{ij} + b_{1j} x_{ij} + \varepsilon_{ij}, \qquad \varepsilon_{ij} \sim N(0, \sigma^2) \tag{4}$$

In (4) the normally distributed random variables b_{0j} and b_{1j} represent the groups effects and incorporate the variability among the same month in different years. The fixed-effects coefficients β_0 and β_1 in (4) frequently are referred to as the population averages (Lai and Helser, 2004; Zuur et al., 2009). These coefficients were the ones used to extrapolate *SSC* using discharge as a predictor for the periods when only discharge was available. The residual error ε_{ij} (difference between fitted and observed values) and the standard deviation of random effects for the sub-population averages, b_{0j} and b_{1j}, were not used in our extrapolation procedure, but these could be further explored to apply a stochastic extrapolation. To compare the results from the mixed-effects model, we also estimated 12 generalized least squares regressions (*gls*) fitted by restricted maximum likelihood representing monthly sediment-rating curves, and then compared the performance of the rating curves and the linear mixed-effects models at representing the trends in the data.

The parameters of the variance-covariance matrix ξ in the mixed-effects model can be estimated using either maximum likelihood (ML) or restricted maximum likelihood (REML) (Harville, 1977; Lai and Kimura, 2002). We employed REML as the parameter estimation method because biases can be present with ML that result from lost degrees of freedom (Corbeil and Searle, 1976).

1.2.3 Addressing Autocorrelation Structure

In time series data it is common to have repeated measures within groups, where it is expected to see a strong temporal correlation between an observation at time t and an observation at time $t+1$. This correlation contravenes one of the central assumptions of mixed-effects models: within-group errors are required to be independent (Crawley 2007). While autocorrelation does not bias the model coefficient estimates, it tends to underestimate their standard error (*SE*) when the autocorrelations are present at low positive lags (as in Figure 1.4). When using the mixed-effects model population averages (mean intercept and mean slope) as the only coefficients taken for the extrapolation procedure, the effects of autocorrelation can be ignored. However, when employing the coefficients *SE* (e.g. when using Monte Carlo simulations to perform extrapolation), it is necessary to calculate the empirical autocorrelation structure of the residuals from the model using a standard autocorrelation function (ACF), (e.g. Figure 1.4).

1.2.4 Testing Mixed-Effects Models

To test whether a rating curve (*gls*) or a more sophisticated technique such as a mixed-effects model should be used to represent the relationship *Q-SSC*, several tests and measures of goodness of fit can be performed. For example, the likelihood ratio test can be used to compare the fit of nested (or hierachical) models. In this case the test will return a small p-value if the mixed-effects model is a good choice (Crawley, 2007). An alternative way to decide on what model to use is to compare the model scores of the Akaike information criterion (AIC) or the Bayesian information criterion (BIC). Finally, when assessing the validity of a mixed-effects model, standard graphical methods include plotting standardized residuals against fitted values to observe patterns in the residuals and violations to normality.

1.3 Case Study

The model was designed and tested using discharge and *SSC* data from the Chilliwack River. The main reasons for choosing this watershed include: (1) the length of record collected for 10.5 years in the form of daily averages of *SSC* and discharge, (2) the wide range of sediment sources within the basin, (3) the temporal variability present in the system with very active floods in the fall/winter and spring, and (4) habitat concerns related to the increased sedimentation due to human industrial activities.

The Chilliwack River is a tributary of the Fraser River, and it is located in the Skagit Range of the Cascade Mountains, British Columbia (Figure 1.5). The watershed covers approximately 1,230 km2 and it is composed primarily of rocks of the Chilliwack Terrane, consisting of marine sedimentary, volcanic, and metamorphic rocks. Portions of the watershed are underlain by Tertiary plutonic rocks of the Chilliwack Batholith (Monger 1970, 1989). Surficial materials common within the Chilliwack Valley include basal and alblation till and colluvial deposits as well as glaciofluvial and glaciolacustrine terraces. Cordilleran ice only briefly overtops the Chilliwack Valley and its erosive action is not significant (Waitt and Thorson 1983). Maximum relief is approximately 2300 meters, and there are extensive ridge systems above treeline, as well as some remnant pocket glaciers. The main biogeoclimatic zones within the Chilliwack Valley are Coastal Western Hemlock and Mountain Hemlock. Snow accumulations of up to 5 m occur from 1500 m of elevation and yearly rainfall averages 2500 mm in the valley bottom (Brayshaw, 1997). Due to the seasonal weather pattern, the Chilliwack River experiences two runoff peaks: the first occurring in the fall and winter associated with significant rainfall events and the second occurring in the spring due to snowmelt (Martin and Church, 1995). Although significant flow events occur in the spring, the fall and winter floods present the highest flows, which are responsible for the mobilization of the majority of sediment (Church et al., 1989). Four periods of monthly flow regime have been identified: (1) spring snowmelt, (2) summer low flow, (3) autumn rise due to rainstorms, and (4) winter low due to decrease in precipitation (McLean, 1980). Analysis of the data demonstrated that suspended sediment transport is highly episodic, with most of it occurring during large events reflecting the hydrological regime of the river with well defined seasonal patterns (Figure 1.6). During the 1960 to 2008 period, there were a number of significant high flow events along the Chilliwack River. The estimated 5-year peak flow events for the winter and spring were 430 and 300 m^3s^{-1}, respectively. The largest winter rainfall event had an estimated discharge of 776 m^3s^{-1} while the largest snowmelt event was 280 m^3 s^{-1} (Martin and Church, 1995). McLean (1980) identified three major sources of sediment supply into the river: (1) mass movement such as landslides and debris flow, (2) bank erosion, and (3) tributary creeks. Although mass movements are not frequent occurring events, when they do occur they can be an important source of sediment supply into the river (McLean, 1980). Church et al., (1989) reported that the December 1975 flood event had a greater sediment yield than any other entire year for the period between 1965 and 1975, the base period of *SSC* data for this study.

1.3.1 Data

The Water Survey of Canada provided daily averages of suspended sediment concentrations (SSC) (mgL-1) from 1965 to 1975 (Figure 1.7), and discharge (Q) (Ls-1) from 1965 to 2006, taken at the Vedder River crossing, station number 08MH001 (WSC, 2008). No recent continuous data on SSC were available. Daily averages are unlikely to capture within-event variability but given that most major events in the river last for several days (Church et al., 1989), especially in the case of the spring snowmelt. Working with daily averages provides a good opportunity to observe hysteresis patterns and the seasonal variability of the suspended sediment (e.g., Figure 1.2 and Figure 1.3). In addition, with the intention of defining the groups to formulate the mixed-effect model, I was interested in the within month and inter-year variability rather than the variability among single events. For the purpose of assessing the effects of sedimentation on aquatic organisms, daily averages provide a practical time frame (cf. Newcombe and Jensen, 1996).

Information on suspended sediment texture in the watershed is limited. Based on limited data, the grain size analysis shows that most of the suspended sediment falls within the silt and clay texture (Scott et al., 1993; EBA, 2001). The Chilliwack River channel is relatively steep and its flow velocity is relatively high and very turbulent. Therefore, most of the suspended load is likely to be washed out of the river system and little might settle within the channel. However, magnitude and duration of a flood are important factors in determining the amount of sediment mobilized. Inspection of the flood hydrographs showed that several large events occurred during the study period.
Inter-year variability is an important feature in the system as it depends on larger climatic influences such as the pacific decadal oscillation (PDO) or El Niño (Lofgren et al., 2010). Because of the large number of years with data, it is not feasible to present all results from the inter-year variability. Instead, we present selected results that typify the trends observed. This inter-year variability affects both the intercept and slope of the *Q-SSC* relationship.

1.4 Results

1.4.1 Mixed-Effects Model Coefficients

The coefficients produced by the linear mixed-effects model are shown in Table 1.1. The log-intercept (β_0) and log-slope (β_1) for each month varied considerably from month to month, as expected given the seasonal patterns observed in the data. The standard deviation of random log-intercepts (b_0) and random log-slopes (b_1) indicates the variability in the distribution of monthly intercepts and slopes with respect to the estimated parameter values β_0 and β_1 and reflects the inter-annual variability in the *SSC* and discharge patterns. The residual error (ε) is an additive value of the difference between the observed data points and the predicted values in $\log_e$ space from the mixed-effects linear model within each month of the year across all years. To assess for violations to homogeneity, I plotted the standardized residuals for all models. The standardized residuals concentrate around zero and no aggregation patterns in the plot quadrants were observed (e.g., Figure 1.8).

1.4.2 Extrapolating SSC using Discharge as Predictor

I produced a time series of SSC from 1976 to 2006 using the population averages from the mixed-effects model (intercept and slope) for each month across all years. For example, the coefficients extracted from the mixed- effects model for January across all years with complete *SSC* and *Q* data set were used to extrapolate SSC in the Januaries where only Q data were present. In addition, I compared the outcome from an alternative extrapolation procedure using the generalized least squares regressions *(gls)* representing the monthly sediment-rating curves. A year randomly taken from the simulated time series generated by the mixed-effects model and the rating curve is illustrated in Figure 1.9. Overall both extrapolation procedures seemed to capture the general seasonal variability. However the mixed-effects model does much better at representing the seasonal trend in the summer low-flow period, the month of August in particular. A closer look to the data reveals that the mixed-effects model presents less bias in representing the *Q-SSC* relationship in this month as it is less affected by episodic low flows with high *SSC* episodic events (Figure 1.10). The extrapolated time series reveals that large peaks tend to be slightly under-represented because, comparatively, very large events are only a minor portion of the total number of events given the relationship *Q-SSC* and therefore very unlikely to be predicted by the extrapolation from *Q* using the coefficients from the mixed-effects model (Figure 1.11). The negative intercept values β_0 produced a few negative *SSC* when extrapolated using very low discharge values, which were approximated to zero given the impossibility of obtaining negative *SSC* measurements.

1.4.3 Observed vs. Predicted Values

I compared the goodness of fit for both the mixed-effects model and the rating curve by using scores derived form Akaike information criterion (AIC). When applying AIC to evaluate the rating curve (*gls*) and the mixed-effects model for each month, the AIC for the mixed-effects model present lower scores (better goodness of fit) for all months (Table 1.2). When plotting the observed data from 1965 to 1975 against the fitted values for the same period (Figure 1.12), the plot corresponding to the mixed-effects model shows less total spread around the 1:1 line with a pseudo-R-squared = 0.528. The plot corresponding to the rating curve presents many desegregated values many of which are negative estimates, corresponding mainly to the low-flow summer months with a pseudo-R-squared = 0.276.

1.5 Discussion

I presented a novel methodology to estimate the coefficients (intercept and slope) of the linear relationship between *Q* and *SSC* that takes within group variability characterized by hysteresis and inter group variability characterized by climatic regimes into account. The fixed and random effects address this variability explicitly, thus composing the mixed-effects model. In my case study, the fixed effects represent the inter-year variability. The random effects reflect the variability occurring within months due to availability of the sediment sources.

The results from using the coefficients from both the mixed-effects model and the rating curve suggest that the mixed-effects model reflect more accurate patterns of extrapolated *SSC* from discharge data, especially for the low-flow summer months where there is less clear indication of the *Q-SSC* relationship. The *Q-SSC* relationship tends to be strong in the winter months while it can vary for the rest of the year, particularly during the spring and the late summer months. The mixed-effects model's better performance was achieved by including the mixture of fixed and random effects represented by the inter-annual and intra-month variability respectively, a characteristic ignored in traditional extrapolation techniques such as the sediment rating curve. Ignoring the sources of variability in the system resulted in anomalous values for the coefficients derived from months with odd relationships between discharge and *SSC* for the rating curve. However, the predictive power of both the mixed-effects model and the rating curve increases at higher values of discharge and SSC as observed for the spring and winter months.

In the watershed's hydrological cycle, two high *SSC* seasons are evident: in the fall associated with rainstorms and the spring due to the snowmelt. Although there is visible variability form year to year, the average monthly suspended sediment concentrations for the period 1965-1975 reveals defined seasonal trends (see Figure 1.6). For most years, the highest monthly mean suspended sediment concentrations were obtained for the snowmelt events in the spring (Figure 1.7). This outcome reflects the relative importance of the flood duration on the daily and monthly suspended sediment concentration. In the basin, the snowmelt events are of much longer duration than the rainfall counterpart. Of particular interest is the December 1975, which experienced the highest sediment yield in record (see Church et al., 1989). Despite the large magnitude of sedimentation events on this month, the sediment concentration remained within the range of results obtained for the rest of the records. This could be due to the high magnitude but short duration of the flood.

I assumed that the *SSC* dynamics in the river in the period represented by the data are still valid for the present and future, although changes in the sediment sources over time might have led to shifts in the rating curves or mixed-effects model coefficients. Even though I explicitly consider uncertainty, it is important to understand that errors or bias in the results may arise from using older data sets and the best outcome is achieved by using up-to-date information. Mixed-effects models assume a hierarchical structure in the data and even if all the groups of the population are sampled (as in this case) it is still necessary to consider the covariance structure of the entire population (Aitkin & Longford, 1986; Lai & Helser, 2004).

When direct *SSC* data are lacking, it is important to obtain accurate estimates of suspended sediments derived from discharge records. In this respect, better estimates of *SSC* can help managers and scientists determine the impact of suspended sediments on aquatic species. For example, in coastal British Columbia, the aquatic species contained in many coastal watersheds are susceptible to elevated *SSC* levels. Survival of incubating salmonids has been negatively related to the proportion of fine sediment deposited in the spawning gravels where they affect eggs and alevin survival by reducing intra- gravel water flow and by physically trapping emerging alevins (e.g., Chapman 1988; MacDonald et al., 1991). Behavioural and physiological effects of SSC have been observed for fish at different life stages (Newcombe and Jensen, 1996). Elevated suspended sediments likely increase turbidity, which impairs foraging effectiveness in many fresh water species interfering with feeding processes (Berg and Northcote, 1985; Gregory and Northcote, 1993). In addition, growth impediments related to increase sedimentation could have significant effects at the population level (Suttle et. al., 2004).

In conclusion, this study shows that mixed-effects models can help predict more accurate values of *SSC* in the absence of data using discharge as predictor. These models do so by taking the inter-group and intra-group variability into account. In my case study, the predictive power of both the mixed-effects model and the rating curve increases at higher values of discharge and *SSC*, as their relationship becomes stronger. For the low-flow summer months, the mixed-effects model proved to be a more valuable alternative than the sediment-rating curve, an important feature when estimating suspended sediment concentrations in areas with limited hydrological data. This advantage can equip managers and scientist with better knowledge for protecting aquatic species from the deleterious effects of elevated suspended sediment concentrations.

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1.7 References

Aitkin M, Longford N. 1986. Statistical modeling in school effectiveness studies. *Journal of the Royal Statistical Society* **A 149** (1):1-43.

Asselman NEM. 1999. Suspended sediment dynamics in a large drainage basin: the River Rhine. *Hydrol. Process* **13**: 1437-1450.

Berg L, Northcote TG. 1985. Changes in territorial, gill-flaring, and feeding behaviour in juvenile coho salmon, Oncorhynchus kisutch, following short term pulses of suspended sediment. *Canadian J. Fisheries Aquatic Sciences* **42**: 1410-1417.

Brayshaw D. 1997. Factors affecting post-logging debris flow initiation in steep forested gullies of the southwestern Canadian Cordillera, Fraser Valley Region. *M.Sc. Thesis, University of British Columbia, Department of Geography.* British Columbia, Canada.

Bunte K, MacDonald L. 1999. Scale considerations and the detectability of sedimentary cumulative watershed effects. *National Council for Air and Stream Improvements (U.S.).*NCASI Technical Bulletin No. 776.

Chapman DW. 1988. Critical review of variables used to define effects of fines in redds of large salmonids. *Transactions of the American Fisheries Society*. **117(1):** 1-21.

Church M, Gilbert R. 1975. Proglacial fluvial and lacustrine environments, glaciofuvial and glaciolacustrine sedimentation. In *Soc. Econ. Paleontol. Minerol. (Spec. Publ.)* Jopling A.V. and McDonald B.C. (eds) **23**: 40-100.

Church M, Kellerhals R, Day TJ. 1989. Regional clastic sediment yield in British Columbia. *Canadian Journal of Earth Sciences* **26**: 31–45.

Corbeil R, Searle S. 1976. Restricted maximum likelihood (REML) estimation of variance components in the mixed model. *Technometrics* **18 (1):**31-38.

Crawley M. 2007. The R book. *John Wiley and Sons, Ltd.* Chichester,UK. 629 p

EBA Engineering Consultants Ltd. 2001. Chilliwack River Watershed restoration plan - final report. Available at: www.chilliwackwatershedstrategy.ca/Bibliography/DOCS/ Access on 8 June 2008

Environment Canada. 1992. Sediment data British Columbia 1990, Water Resources Branch. *Water Survey of Canada.* Ottawa, Canada. 53 pp.

Gregory RS, Northcote TG. 1993. Surface, planktonic and benthic foraging by juvenile chinook salmon (*Oncorhynchus tshawytscha*) in turbid laboratory conditions. *Can. J. Fish. Aquat. Sci*. **50**: 233-240

Harville D. 1977. Maximum likelihood approaches to variance component estimation and related problems. *Journal of the American Statistical Association* **72**:320-338.

Horowitz A. 2003. An evaluation of sediment rating curves for estimating suspended sediment concentrations for subsequent flux calculations. *Hydrological Processes*. **17:** 3387-3409.

Kerr SJ. 1995. Silt, turbidity and suspended sediments in the aquatic environment: an annotated bibliography and literature review. *Ontario Ministry of Natural Resources.* Southern Region Science & Technology Transfer Unit Technical Report TR-008. 277 pp.

Lai H, Helser T. 2004. Linear mixed-effects models for weight-length relationships. *Fisheries Research* **70 (2-3)**: 377-387.

Lai H, Kimura D. 2002. Analyzing survey experiments having spatial variability with an application to sea scallop fishing experiment. *Fisheries Research.* **56 (3)**: 239-259.

Lenzi MA, Marchi L. 2000. Suspended sediment load during floods in a small stream of the Dolomites (Northeastern Italy). *Catena* **39**: 267–282.

Lofgren R, Samora B, Baccus B, Christoe B. 2010. Climate monitoring protocol for the North Coast and Cascades Network. Natural Resource Report NPS/NCCN/NRR—2010/240 Volume 1. *National Park Service,* Fort Collins, Colorado.

MacDonald LH, Smart AW, Wissmar RC. 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. EPA/910/9-91-001. Seattle: *U.S. Environmental Protection Agency*, Region 10. 166 p.

Martin Y, Church M. 1995. Bed-material transport estimated from channel surveys: Vedder River, British Columbia. *Earth Surface Processes and Landforms* **20**: 347–361.

McLean DM. 1980. Flood control and sediment transport study of the Vedder River, M.A.Sc. thesis, *University of British Columbia, Department of Civil Engineering*, 185 pp.

Monger JWH. 1970. Hope map-area, west half, British Columbia. *Geological Survey of Canada*, Paper 69-47.

Monger JWH. 1989. Geology, hope, British Columbia. *Geological Survey of Canada*, Map 41-1989, sheet 1.

Mweempwa H. 1993. Character and controls of suspended-sediment concentration and discharge effectiveness, Fraser River, British Columbia, Canada. *Phd. Thesis Geography Department. Simon Fraser University*. Burnaby, British Columbia.

Newcombe CP, Jensen JO. 1996. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. *North American Journal of Fisheries Management* **16**:693-727.

Nistor CJ, Church M. 2005. Suspended sediment transport regime in a debris-flow gully on Vancouver Island, British Columbia. *Hydrological Processes* **19**: 861- 885.

Nistor C. 1996. Temporal patterns in the normal-regime fine-sediment cascade in Russel Creek basin, Vancouver Island. *The University of British Columbia* 236 pp.

Phillips JM, Webb BW, Walling DE, Leeks GJL. 1999. Estimating the suspended sediment loads of rivers in the LOIS study area using infrequent samples. *Hydrobiol.* **13**:1035–1050.

Pinheiro J, Bates D. 2000. Mixed effects models in S and S plus. Springer-Verlag, New York. ISBN 0-387-98957-5.x

Pinheiro J, Bates D, DebRoy S, Sarkar D. 2010. nlme: Linear and Nonlinear Mixed Effects Models (R Package Version 3.1-92). *Institute for Statistics and Mathematics*, Vienna University of Economics and Business, Vienna. Available at: http://cran.r-project.org/web/packages/nlme.

Reiter M, Heffner JT, Beech S, Turner T, Bilby RE. 2009. Temporal and spatial turbidity patterns over 30 years in a managed forest of Western Washington. *Journal of the American Water Resources Association (JAWRA)* **45(3)**:793-808. DOI: 10.1111/ j.1752-1688.2009.00323.x

Salant NL, Hassan MA, Alonso CV. 2008. Suspended sediment dynamics at high and low storm flows in two small watersheds. *Hydrological Processes* **22**: 1573–1587. DOI: 10.1002/hyp.6743.

Salant NL, Hassan MA. 2008. Fine particles in streams: physical, ecological, and human connections. In *Coastal Watershed Management*, Fares A. and A. I. El-Kadi (eds). WIT Press Royal: Southampton, U.K, 432 pp.

Scott KJ, Kistritz RU, Galay VJ. 1993 Chilliwack River water quality and fish enhancement opportunities study. Scott Resource Services, British Columbia Habitat Conservation Fund. Available at:http://www.chilliwackwatershedstrategy.ca/Bibliography/CRWS%20Reference%20Database.htm. Accessed on 8 September, 2009

Singer J. 1996. Using SAS PROC MIXED to fit multilevel models, hierarchical models, and Individual growth models. *Journal of Educational and Behavioral Statistics* **23 (4)**: 323-355.

Smith BPG, Naden PS, Leeks GJL, Wass PD. 2003a. Characterising the fine sediment budget of the River Swale, Yorkshire, UK during the 1994 to 1995 winter season. *Hydrobiologia* **494**: 135-143.

Smith BPG, Naden PS, Leeks GJL, Wass PD. 2003b.The influence of storm events on fine sediment transport, erosion and deposition within the River Swale, Yorkshire, UK. *Science of the Total Environment*. **314/316**: 451-474.

Snijders TAB, Bosker RJ. 1999. Multilevel analysis: An introduction to basic and advanced multilevel modeling. *Sage Publishing.* London. 266 pp.

Suttle KB, Power ME, Levine JM, McNeely C. 2004. How fine sediment in riverbeds impairs growth and survival of juvenile salmonids. *Ecol. Appl.* **14**:969-974.

Waitt RB, Thorson RM. 1983. The Cordilleran ice sheet in Washington, Idaho, and Montana. *In*: Porter, S.C.and Wright, H.E., Jr. (eds), *Late-Quaternary environments of the United States*. University of Minnesota Press **1**: 53–70.

Walling DE, Webb BW. 1987. Material transport by the world's rivers: evolving perspectives. In: *Water for the Future:Hydrology in Perspective.* J.C. Rodda and N.C. Matalas (eds). Proc. Rome Symp., April 1987, IAHS **164**:313-329

Walling DE. 1974. Suspended sediment and solute yields from a small catchment prior to urbanisation. In *Fluvial processes in instrumented watersheds.* Gregory K.J. and Walling D.E. (eds). Inst. Br. Geog. (Spec. Publ.) **6:** 169-192.

Walling DE. 1977. Limitations of the rating curve technique for estimating suspended sediment loads, with particular reference to British rivers. In: *Erosion and solid matter transport in inland waters (Proceedings of the Paris symposium, July 1977).* IAHS Publ. **122**: 34-48.

Water Survey of Canada (WSC). 2008. HYDAT. Available at: www.wsc.ec.gc.ca. Accessed on 15 October 2008.

Webb BW, Phillips JM, Walling DE, Littlewood IG, Watts CD, Leeks GJL. 1997. Load estimation methodologies for British rivers and their relevance to the LOIS RACS(R) programme. *Sci Total Environ* **194**: 379–389.

Zuur AF, Leno EN, Walker N, Saveliev AA, Smith GM. 2009. Mixed effects models and extensions in ecology with R. *Springer*. New York.

1.8 Figures and Tables

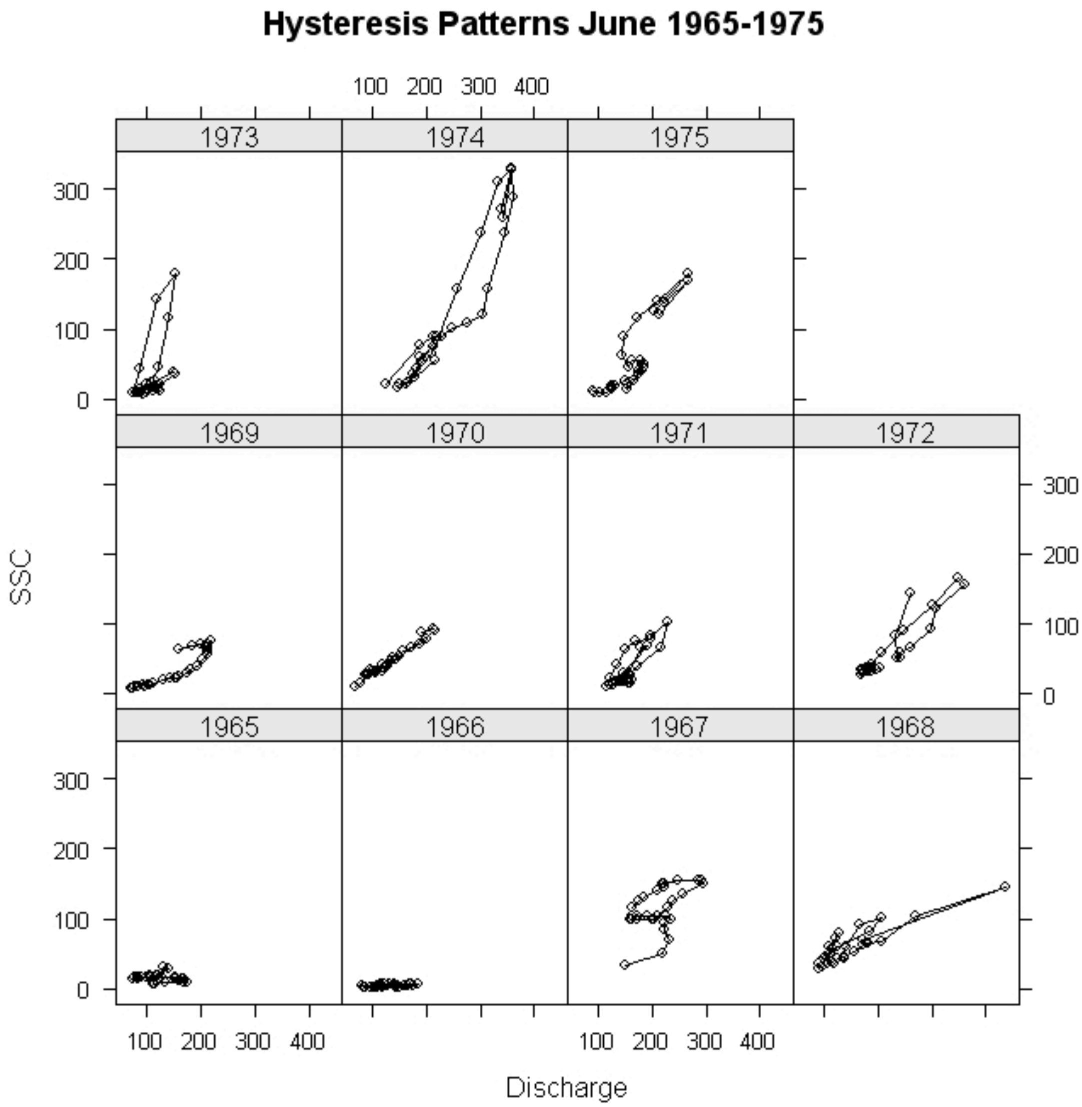

Figure 1.1 Different hysteresis patterns for the month of June across all years in the data set.

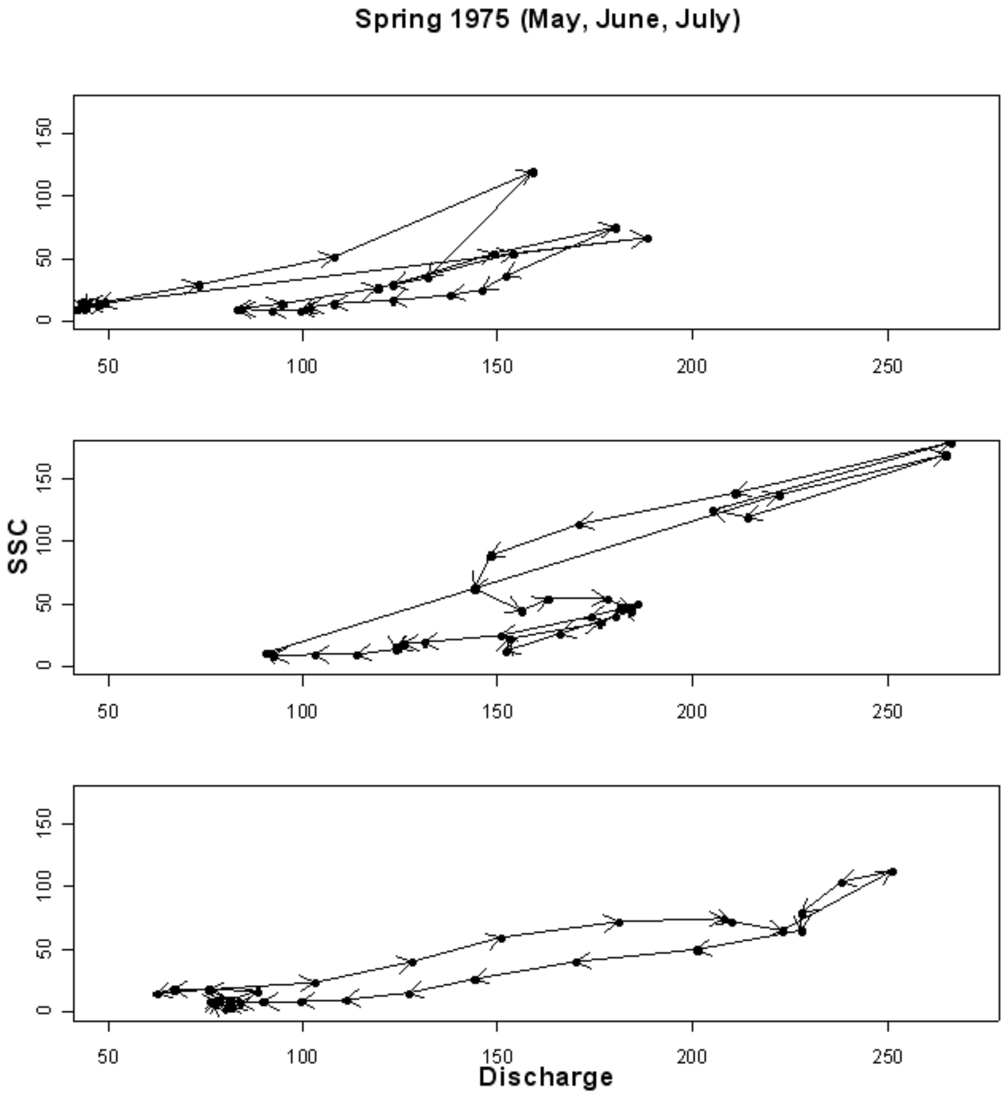

Figure 1.2 Example of hysteresis patterns observed in the Spring of 1975.

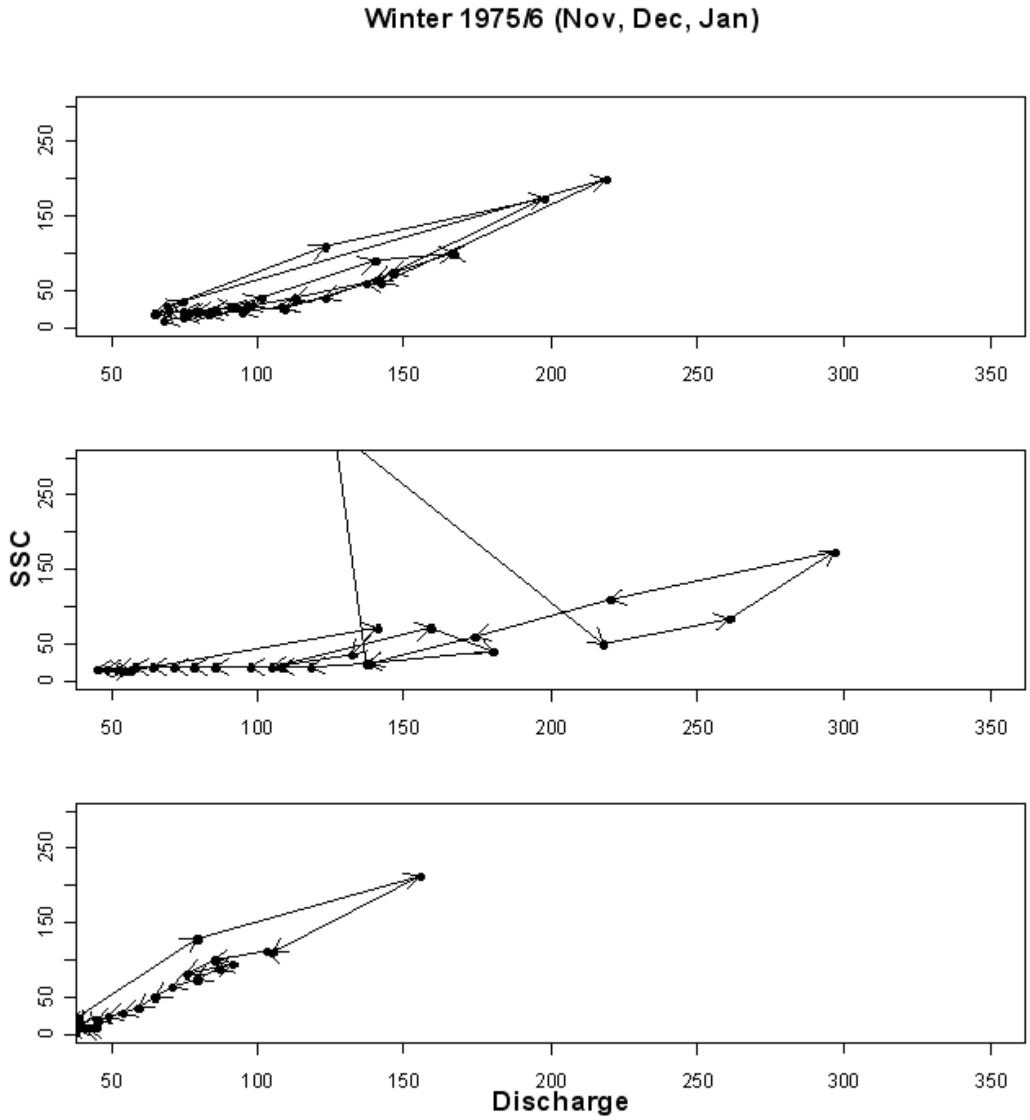

Figure 1.3 Example of hysteresis patterns observed in the Winter 1975/1976.

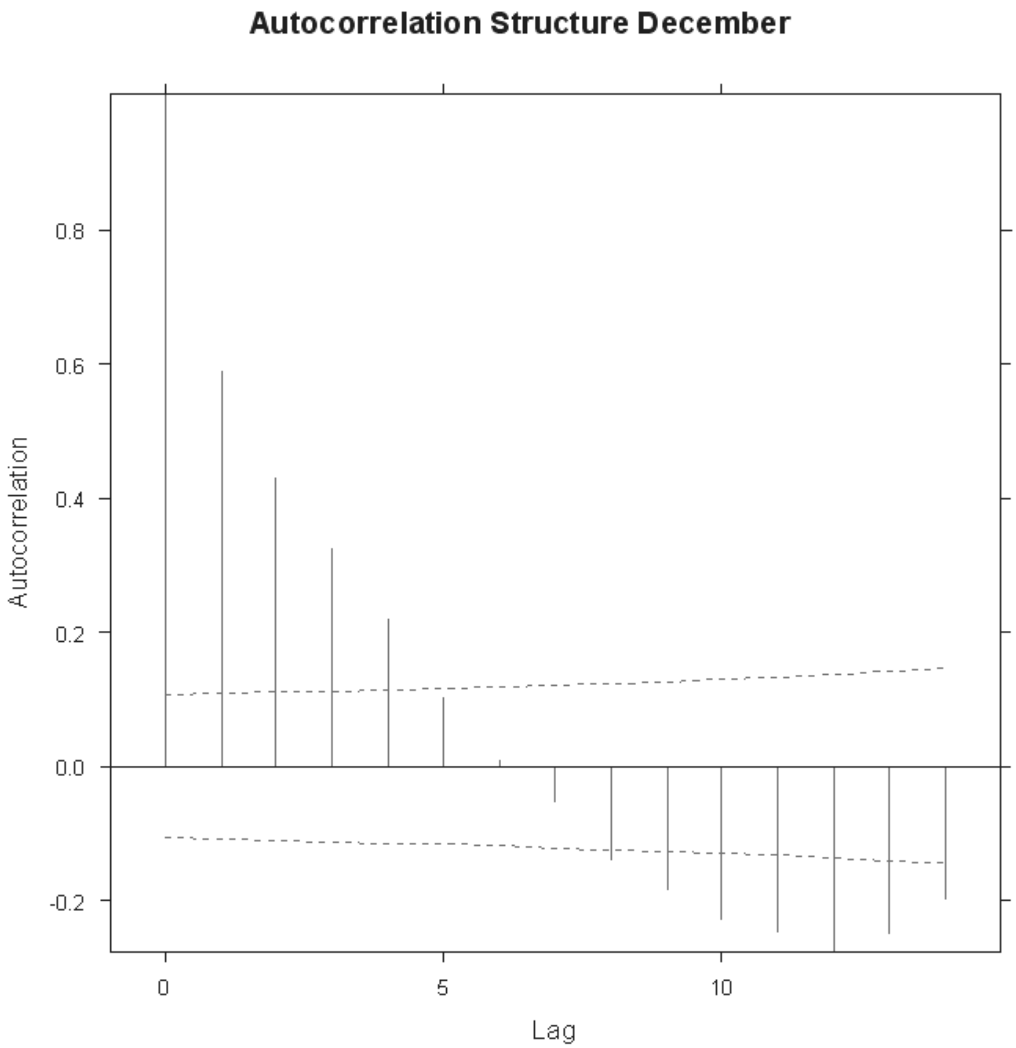

Figure 1.4 **Autocorrelation plot representing the temporal correlations among all observations for the month of December across years. In this example there is highly significant autocorrelation at lags 1 to 3 and marginally significant autocorrelation at lag 4. While autocorrelation does not bias the model coefficient estimates, the standard error (SE) of the coefficients tend to be underestimated at low positive lags.**

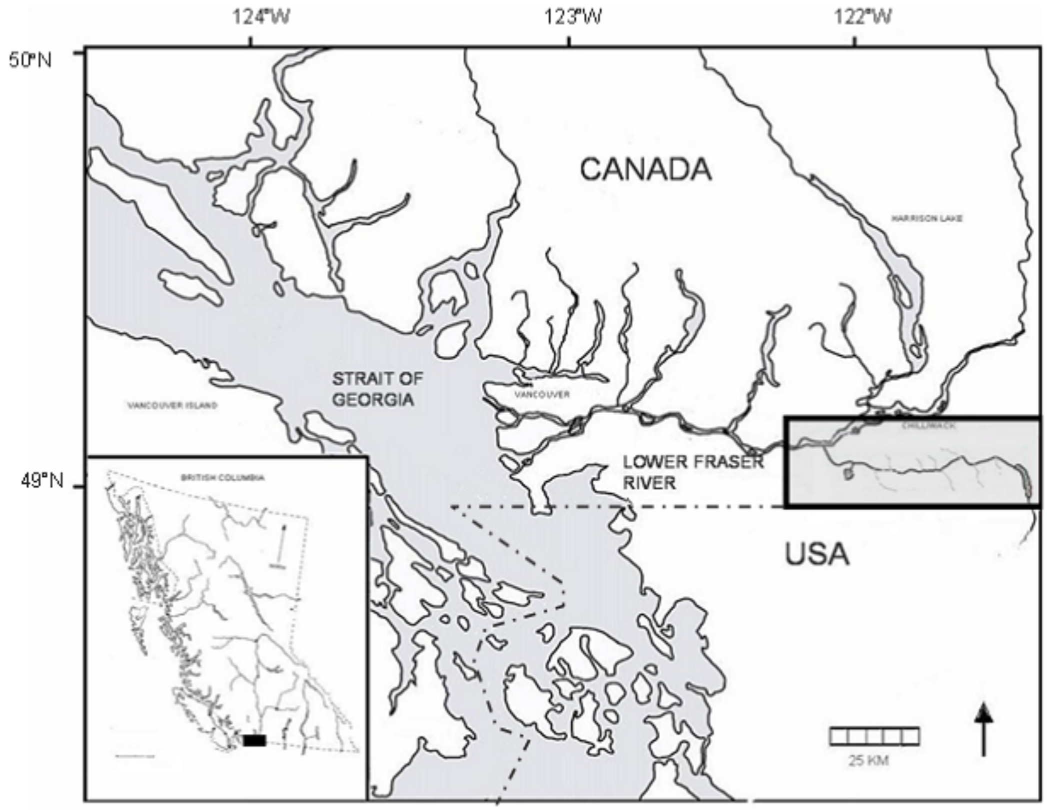

Figure 1.5 The Chilliwack River, a tributary for the Lower Fraser River.

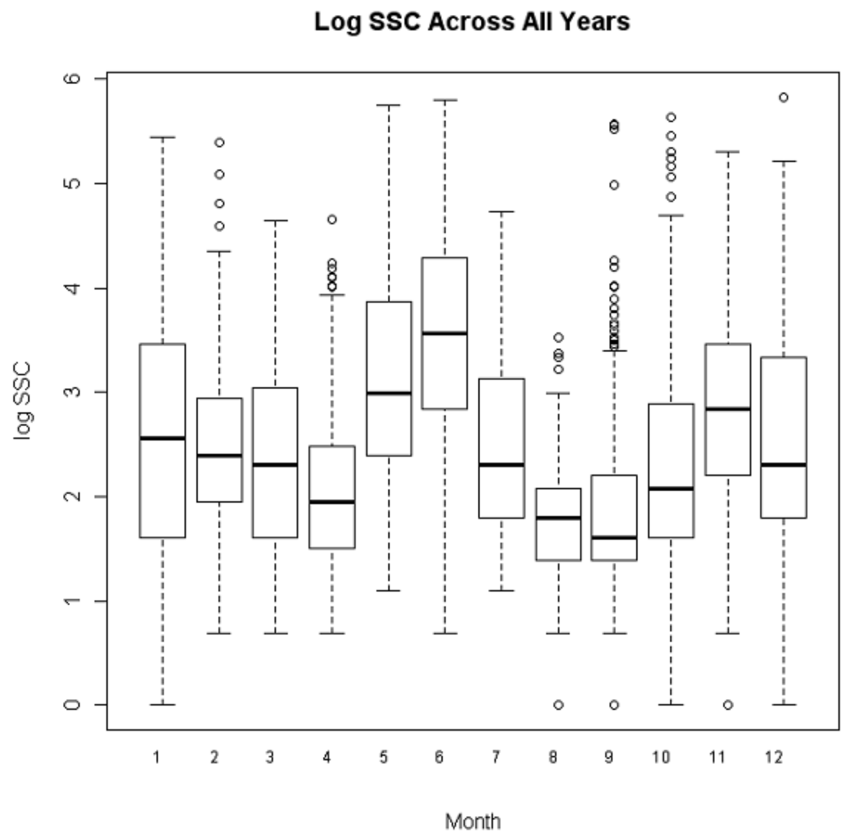

Figure 1.6 Monthly distributions of log SSC for all years in the data set showing a cyclical pattern. The highest values of SSC occur during spring to early summer and winter months. In addition, considerable less within-month variability is observed for the late summer and fall months exhibiting marked heterogeneity.

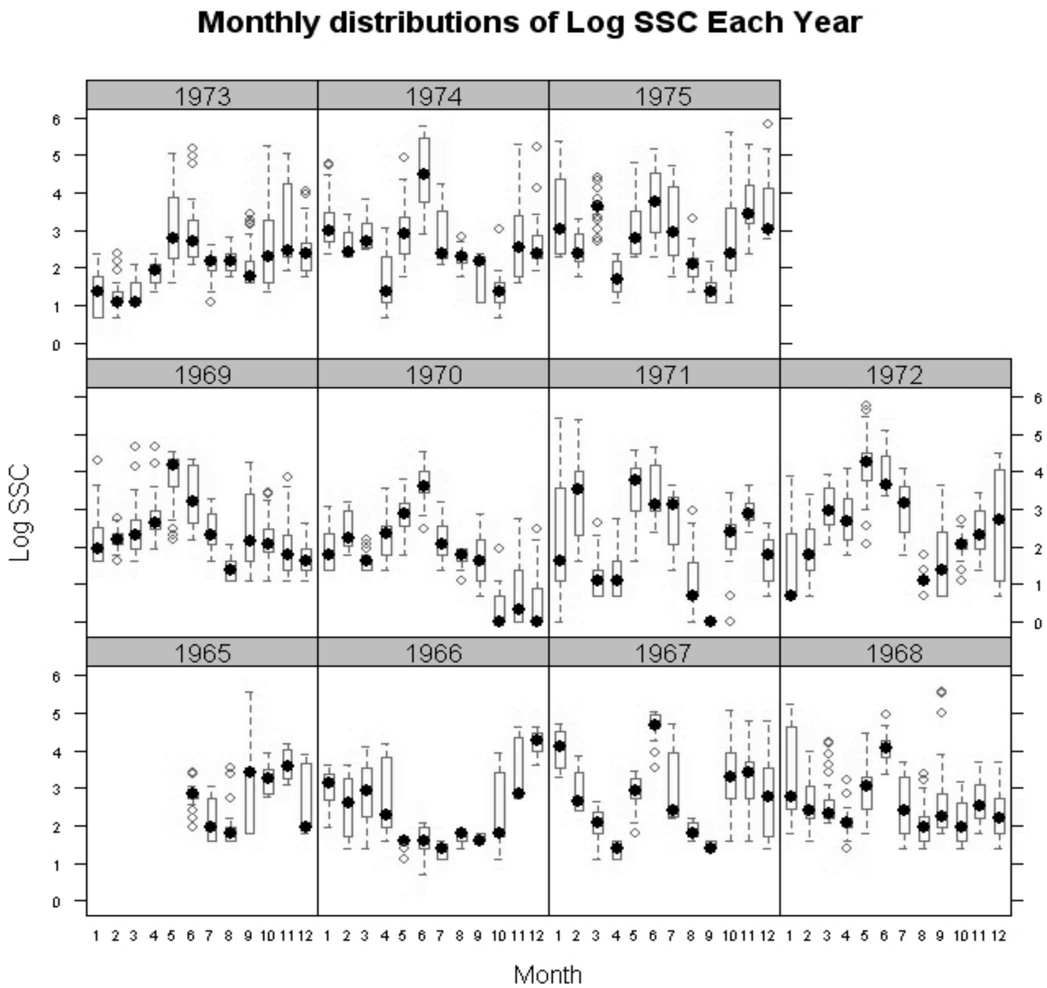

Figure 1.7 **Monthly distributions of log SSC in each year of the data set. Discharge and SSC present fairly defined cyclical seasonal patterns with the highest values of SSC during spring and early summer. Seasonal patterns depend upon winter snow accumulations, which affect the spring and summer discharge and vary from year to year.**

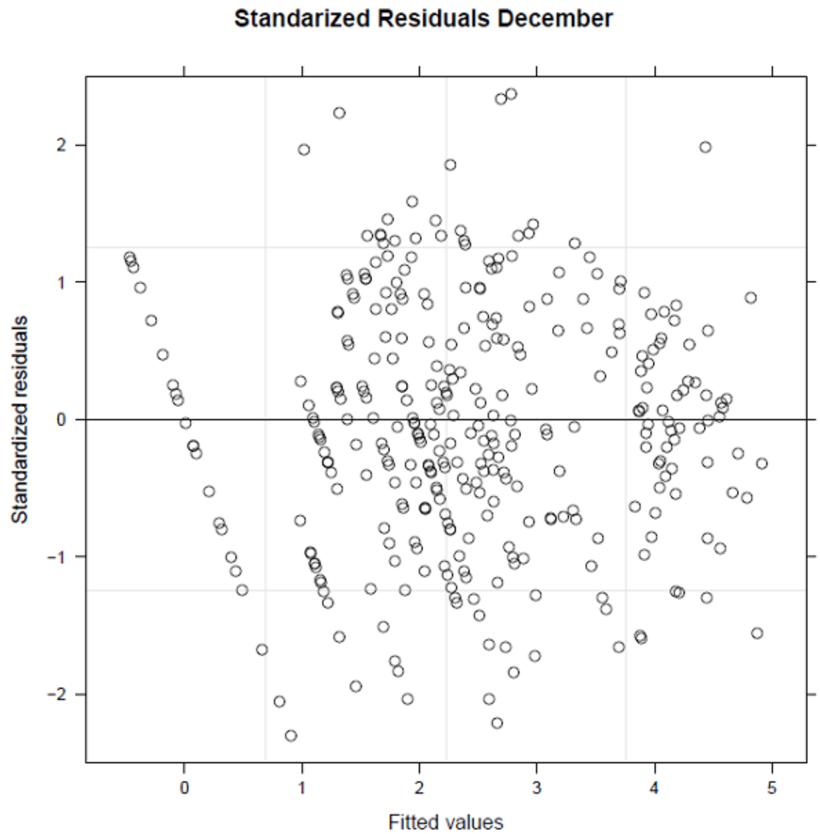

Figure 1.8 Plot of standardized residuals against fitted values for the mixed-effects model representing the month of December.

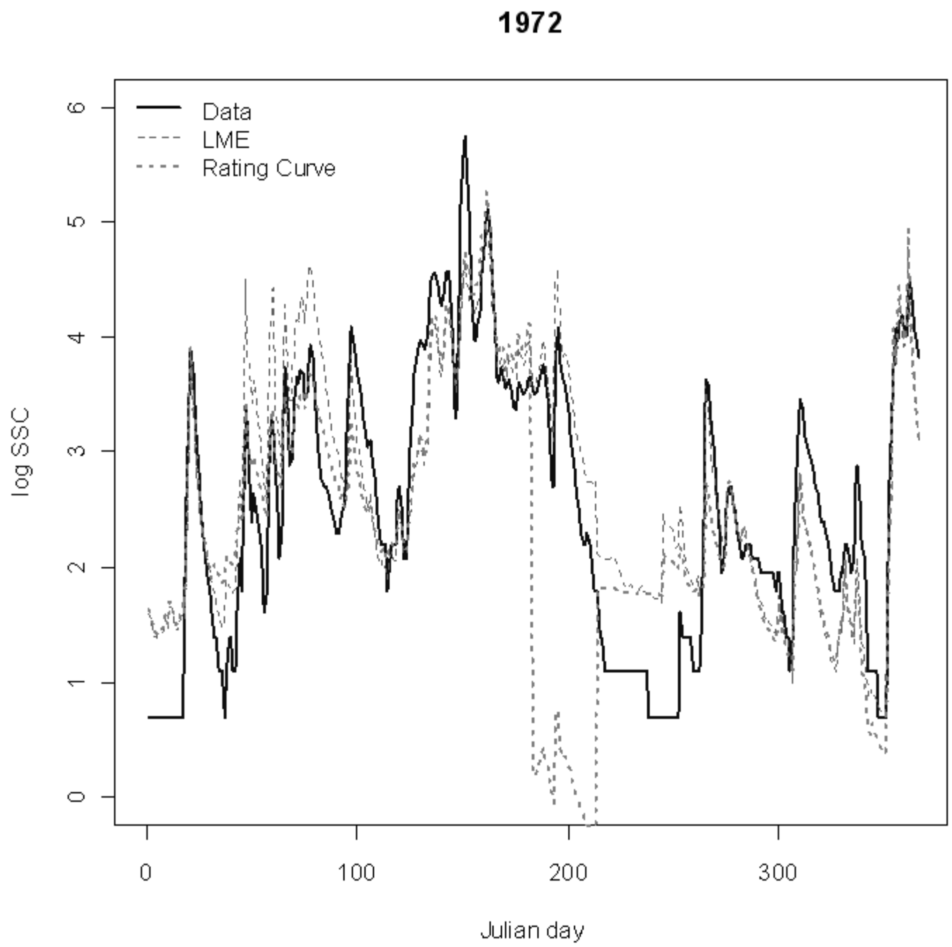

Figure 1.9 Comparison between the original time series of 1972, extrapolated values using the mixed-effects model coefficients, and the monthly Rating Curve estimates. The year was randomly selected within the data set. Notice the much better performance of extrapolation obtained from the mixed-effects model to represent SSC, particularly during the month of August compared to the performance of the Rating Curve.

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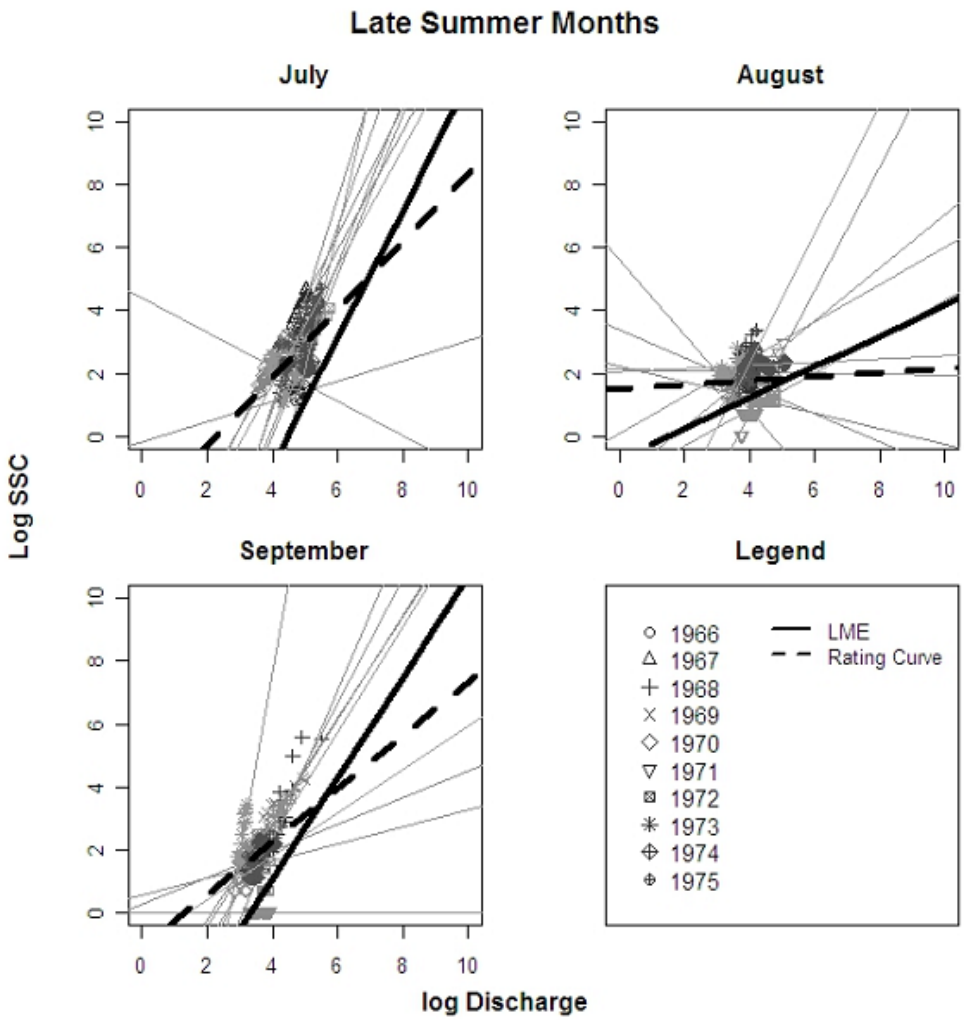

Figure 1.10 Late summer months across all years in the data set. These months are characterized by low flows with episodic high levels of SSC. The change in the dynamics of the relationship discharge-SSC tends to bias the estimates of the rating curve (dashed line) obtaining non-significant relationships (e.g. August). The mixed-effects model (Solid line) performs much better at capturing the trends in the data even in the presence of years with odd discharge-SSC relationships.

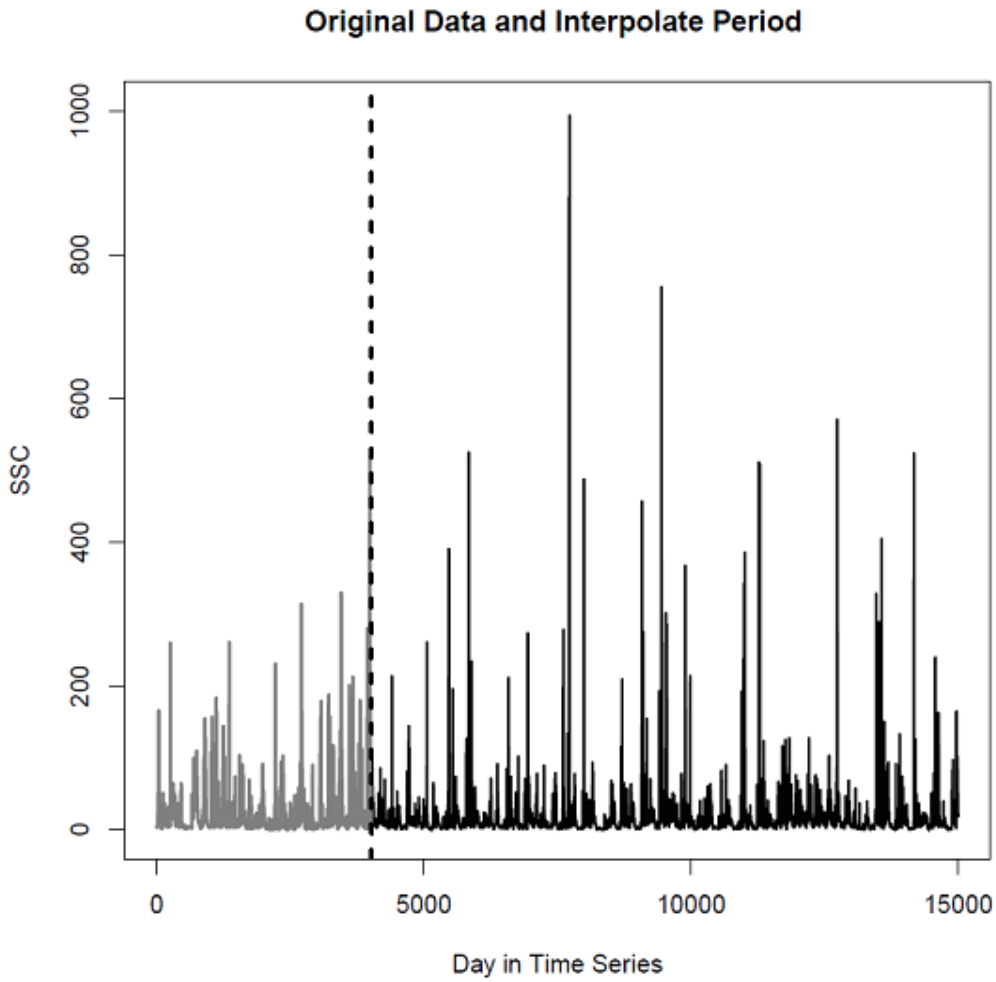

Figure 1.11 Chilliwack River raw data set of SSC (Julian day 1 to 4015) and extrapolated time series using the mixed-effects model and discharge as predictor (Julian day 4016 to 15330). Both periods are divided by the dashed line.

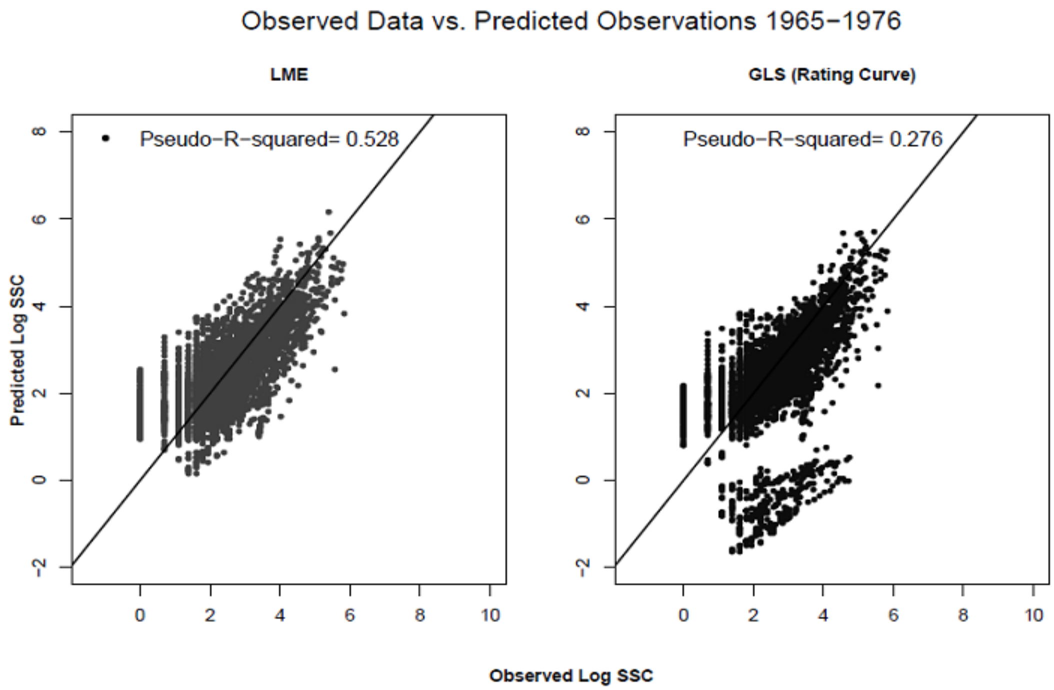

Figure 1.12 **Observed vs. predicted log SSC values for 1965 to 1976. The mixed-effects model captures better the trend in the observed values (pseudo-R-squared= 0.528) compared to the estimates of the rating curve (pseudo-R-squared= 0.276). The low-flow summer months with odd relationships discharge – SSC produced many negative estimates of SSC when using the rating curve, which makes results difficult to interpret. The predictive power of both the mixed-effects model and the rating curve increases at higher values of discharge and SSC.**

Table 1.1 **Mixed-effects model results for the Chilliwack River watershed. The table shows the log-coefficients from the mixed-effects model with their respective standard deviation (SD).**

Month	Log-Intercept β_0	Std error β_0	Log-slope β_1	Std error β_1	*SD* of random log-intercepts b_0	*SD* of random log-slopes b_1	Residual
Jan	-3.7	0.80	1.6	0.19	2.39	0.57	0.40
Feb	-5.61	1.40	2.14	0.42	4.25	1.29	0.29
Mar	-2.96	0.79	1.41	0.18	2.25	0.51	0.34
Apr	-6.22	1.54	2.09	0.35	4.64	1.04	0.37
May	-3.4	0.96	1.37	0.22	2.90	0.67	0.39
Jun	-5.68	1.55	1.81	0.31	4.97	1.01	0.34
Jul	-7.06	1.71	2.03	0.36	5.54	1.18	0.31
Aug	-0,25	1.07	0.49	0.27	3.41	0.85	0.28
Sep	-3.67	0.97	1.59	0.33	2.84	0.99	0.44
Oct	-3.24	0.63	1.48	0.14	1.91	0.41	0.48
Nov	-4.66	0.66	1.85	0.14	2.00	0.41	0.37
Dec	-4.25	0.79	1.67	0.15	2.51	0.48	0.39

Table 1.2 **Comparison of AIC scores for the mixed-effects model and the rating curve (*gls*) for all the months in the data set.**

Month	Linear Mixed-Effects Model	Rating Curve (*gls*)
January	392	766
February	216	502
March	300	760
April	326	587
May	375	669
June	326	740
July	289	705
August	215	581
September	480	906
October	532	767
November	355	696
December	392	684

Chapter 2: Modeling Suspended Sediment Concentrations due to Changes in Forest Road Density and Use in a Coastal Watershed of British Columbia

2.1 Introduction

Forest roads are an important component of timber extraction, but they also provide access for mining, fishing, and other commercial and recreational activities. However, the presence of roads in a watershed has well-known negative effects on terrestrial and aquatic environments. Forest roads also directly and indirectly produce more sediment than any other forestry activity (Beschta 1978; Reid and Dunne 1984; Meehan 1991; Hassan et al. 2005; Hudson 2006). The main impacts of forest roads include not only the production of sediments that eventually end up in streams, but also changes in basin geomorphology and hydrology, and changes in riparian and stream habitat characteristics (Naiman et al. 1992; Gucinski et al. 2001). Ecological effects of roads are of increasing concern for scientists, decision makers and society in general as the number of road-free wilderness areas decreases. Gucinski et al. (2001) provide a comprehensive review of the direct and indirect effects of forest roads on watershed processes.

Hydrological processes provide an integrating function in watersheds connecting geomorphic, climatic, and ecological elements (Northcote and Hartman 2004). Forest roads change the hydrology of watersheds by intercepting rainfall directly on their surfaces and banks (Jones and Grant 1996) and modifying the runoff corridors by which water flows to the stream channel (Moore and Wondzell 2005). Road surfaces along with their associated forest clearcuts increase snow storage in the watershed, altering the timing and rate of snowmelt and reducing forest evapotranspiration; thus changing the watershed's natural hydrological processes (Jones 2000). Roads can also increase surface water flows and intercept sub-surface groundwater that normally moves down the hill-slope; this accelerated movement of water contributes to increased rates of extreme flow events (Wemple et al. 1996; Gucinski et al. 2001).

Erosion from road surfaces, cut slopes, banks, and ditches represents a major source of chronic sediment input to streams (Beschta 1978; Reid and Dunne 1984; Meehan 1991; Naiman et al. 1992). Road surface erosion is particularly sensitive to use and traffic levels. In a watershed with roads and high traffic levels, the sedimentation produced by forest roads can be equal to that produced by landslides (Wald 1975; Reid and Dunne 1984). Inevitably, a large portion of the sediments produced by forest roads ends up in rivers and streams, and causes the degradation of aquatic habitats and direct behavioural and physiological effects on fish.

The amount of sediment delivery to rivers and streams after road construction and use has been an extensive area of research in the Pacific Northwest during the past 50 years, and an abundance of literature is available (see reviews by Kerr, 1995 and Gucinski et al., 2001). Some of the difficulties of quantifying road-related sediment generation are associated with watershed specific conditions such as terrain, soils, seasonal precipitation patterns, long time-scales, geomorphology, but also the quality of data (Ashmore et al. 2000). Even though road-generated sedimentation has been the focus of many studies, few studies have addressed simulation as a technique to quantify the sediment yields in response to changing road conditions such as density and use.

Applying hydrologic and sedimentation models to support forest management and decision-making has been the objective of recent research initiatives (e.g., Alila and Beckers 2001; Borga et al. 2004). In practice, these models are difficult to implement or they are only used to understand physical processes. The accuracy of these models is not tested and their precision depends on the input of high-quality data and extensive monitoring programs that rarely exist (Pike et al. 2007). Regardless of the limitations of hydrological and sedimentation models, they are helpful tools in assisting managers to evaluate and make decisions about the potential effects of forestry activities on forest hydrology and sedimentation. The Water Erosion Prediction model (WEPP) and its extended prototype, the Cross Drain Spacing and Sediment Yield Program (X-DRAIN), is a process-based model that estimates the amount of sediment generated by forest roads (Annual Sediment Yield- ASY) based on terrain, road characteristics, and traffic levels (Elliot et al. 1999a,b). The Road Sediment Delivery Model (SEDMODL2) model has been used to calculate the ASY using a data-driven spatial framework that includes topography, and terrain characteristics associated with sediment production, delivery, and retention (Dubé and McCalmon 2004; Welsh 2008). Although these models are very useful in assessing sediment production from forest roads, they require extensive information (often not available in many watersheds) and their outputs do not include potential changes in watershed hydrographs for given road densities and use, a desirable feature when assessing the effects of sediment pulses in aquatic systems. Given the complexity of quantifying the responses of suspended sediment concentrations (*SSC*) and discharge (*Q*) to changing road conditions, a data-driven simulation approach, that includes both theoretical considerations of sediment production by roads and the watershed's hydrological data, could be a useful tool for understanding the impact of road-related factors on the watershed's hydrology and sediment regime.

The main purpose of this study was to provide a model to quantify the daily suspended sediment concentrations (*SSC*) generated by forest roads under different conditions of use and density in a generic medium-sized coastal watershed of British Columbia. In addition, this method permited isolating background (or natural) sedimentation from that produced by forest roads, utilizing hydrological time series data. The method was tested using a scenario of road building at different usage levels associated with the current management strategy for habitat of the Northern Spotted Owl in the Chilliwack Forest District in British Columbia (Sutherland et al. 2007). This method is an attempt to overcome field data limitations using a mathematical approach and presents a starting point for experimental studies.

2.2 Methods

The model estimates the annual fine sediment yield (AFSY) under the current road status and use conditions in the watershed (baseline scenario). It does so based on the literature of sediment production and the physiographic parameters in the watershed. Subsequently, the model uses the total sediment yield, which is derived directly from daily hydrological data, to discriminate between natural and road-generated *SSC*. Lastly, the model simulates changes in the a time series of road-generated daily *SSC* for given forest management scenarios by introducing potential changes in peak discharge and volume of suspended sediments transported, specific to each scenario. The model is intended for application on medium-sized watersheds in the Pacific Northwest (between ~ 50 km^2 and 2000 km^2). I used the Chilliwack River, British Columbia, as a prototype watershed because of its history of forest road construction and use, and the availability of required hydrological and *SSC* data (See case study).

2.2.1 Estimating the Annual Fine Sediment Yield (AFSY) from Forest Roads

In the first component of the methodology, I estimated the AFSY, the amount of fine sediments (<62 μm) that ends up in the river in a year exclusively from road-related factors, working under the assumption that the physiographic parameters in the watershed remain constant (i.e., slope of the road, slope of the terrain, and number of streams). The variable factor is road use. To describe variations in road use I employed the definitions of Reid (1981): Light, use only by light vehicles; Moderate, use by fewer than four logging trucks per day; and Heavy, use by more than four logging trucks per day. To estimate the AFSY, it is necessary to obtain the slope and lengths of road segments for the entire study area. Estimation of the AFSY follows the linear relationship:

$$AFSY_i = l_i u_i (LR_i)(DR_i) \tag{1}$$

where $AFSY_i$ is the annual fine sediment yield (tons.km^{-1}yr^{-1}) for a road network (i). The road network (i) was estimated as the length (l_i) of the roads (measured in kilometers); u_i is the sediment yield in tonnes yr^{-1}km^{-1}, based on the road network's use and slope; LR_i is the loss ratio (Appendix 1, Table 1a), and DR_i is the delivery ratio (Appendix 1,Table 1b). LR_i reflects diversions of sediment-laden water by obstructions in the path of flow, and it depends on the slope of the roads. The DR_i corresponds to the percentage of sediments that can potentially end up in the stream for a given terrain slope, number of gullies, and ephemeral streams in the watershed (Appendix 1,Table 1b). The total suspended sediment yield generated exclusively by roads for the entire time series is estimated by multiplying $AFSY_i$ times the number of years in the time series (y), assuming that road use does not change during this period.

2.2.2 Separating Natural from Road-Related Sedimentation

A second component of the methodology incorporates hydrological data from the watershed, which allows estimating the total load of suspended sediments (L_d) (tonnes) for a day (d). Hence, it was necessary to obtain the average daily values of suspended sediments concentrations (SSC_d) (mg L^{-1}) and discharge Q_d in litres per day (L day^{-1}) from which L_d is determined:

$$L_d = SSC_d Q_d 10^{-9} \tag{2}$$

Thus, the total sediment yield (T) for the entire time series is defined as the sum of individual values of L_d expressed in tonnes.

It is important to note that estimation of sediment yields from daily SSC and discharge (Q) records is subject to biases and errors. Errors arise because suspended sediment samplers (point-integrated, depth-integrated or simple bottle grab samples) only collect instantaneous samples of SSC that can vary substantially through time due to turbulence in the water column. Topping et al. (2007) examined the error associated with depth-integrated suspended sediment sampling and found errors of 8% in suspended silt-clay concentration, 22% in suspended sand concentration, and 12% in the median grain-size. Others have reported errors of similar magnitude (e.g., Smith et al. 2003a, b: 5-10%; Wang et al., 2007: 16%). Biases may arise due to the sampling frequency and timing. Most samples are collected during high flows and replicate samples are rarely obtained. When using up to 12 samples, error ranges between 14 and 73% are reported (Webb et al. 1997; Phillips et al. 1999; Smith et al. 2003a, b). Biases may also arise from the rating curves. Using a power function based on a nonlinear least squares method and logarithmic data transformation, Asselman (2000) calculated that sediment loads were underestimated by 1% and 12%, respectively. Mweempwa (1993) analyzed the sampling techniques employed by the Water survey of Canada (WSC) concluding that these present reliability estimates contained within ±10% accuracy for the Chilliwack River. These errors and biases are quite substantial. It is beyond our present purpose to undertake a detailed error propagation exercise. Nevertheless, it is important to recognize them in the context of the model presented.

The next step in the simulation is to separate the natural daily suspended sediment loads from those generated by roads. To do so, I used the total sediment yield, which implicitly includes both natural and road-generated sediment yields. In this context "natural" includes all sediment generated by factors other than roads, which may incorporate other human activities. Although road-generated sedimentation is initially high immediately after road construction, it is moderated over time by natural revegetation (Bescheta, 1978) and remains active over long periods of time (Hagans et al. 1986). I assume that the natural and road-generated sediments represent constant proportions and ignore the initial high sediment pulse from road construction, concentrating instead on the long term chronic sediment inputs from road construction and use:

$$r_d = \frac{L_d \sum_{i=1}^{y} AFSY_i}{T} \tag{3}$$

$$n_d = L_d - r_d \tag{4}$$

where r_d are the *SSC* generated by roads; n_d is the natural daily suspended sediment loads; T is the total sediment yield, which incorporates both natural (n) and road generated-sediment yields (r); and y is the number of years in the data set (42).

2.2.3 Simulating the Scenario-Specific Time Series

So far I have described a method to estimate daily natural and road-related sediment yields (Eqs. 3 and 4) only under the current (baseline) conditions. To simulate changes in the frequency and magnitude of road-related *SSC*, it is necessary to consider changes in the density of roads and their level of use relative to baseline conditions. The following equations address these changes:

$$f_d = e^{\rho \ln(r_d + 1) - 1} \tag{5}$$

where f_d represents the daily values in the time series of *SSC* (tonnes day^{-1}) for a day (d) after changes in the frequency of extreme events only, while maintaining a constant volume of sediments; ρ modifies the frequency of extreme events, and it is assumed to be proportional to the density of roads (Jones and Grant 1996; Jones 2000).

$$g_d = \frac{\phi R f_d}{\sum_{d=1} f_d} \tag{6}$$

where g_d are the simulated road-related daily values of suspended sediments (tonnes day-^{1}), with ϕ affecting the magnitude of all events in the time series as a function of the level of road use. The total suspended sediment yield generated by roads (R) for the entire time series is estimated by multiplying $AFSY_i$ times the number of years in the time series (y), assuming that road use does not change during this period.

Calculation of these parameters is explained later in the case study. By adding g_d and n_d, we obtain the final simulated values (λ_d)(tonnes day^{-1}), which are then back-transformed into concentration units (mg L^{-1}).

$$\lambda_d = g_d + n_d \tag{7}$$

The sum of all λ_d values for the length of the time series represents the total sediment yield (λ_*) specific to combinations of ρ and ϕ– which include changes in both the frequency and magnitude of extreme events. The final output of the model is a time series composed of discrete average daily values of concentration (SSC_d)(mg L^{-1}). Parameter estimation is specific to the watershed of study (See case study for details).

The model was adapted from a sediment budget study conducted by Metro Vancouver (GVRD 1999), which used the results from Reid (1981) to estimate the annual fine sediment yield (AFSY) from forest roads in their three managed watersheds (Seymour, Capilano, and Coquitlam). Appendix 1, Table 1a, indicates the values of AFSY. For this study, the predicted average AFSY includes only SSC and excludes all other kinds of sediment, such as coarse-grained bedload materials in the form of sands, pebbles, cobbles and boulders (GVRD 1999; Dave Dunkley pers. Comm. Apr 29, 2009). The AFSY values estimated for the Metro Vancouver model were used in this study because their watersheds share many geomorphic and hydrological characteristics with the Chilliwack River (Appendix 1,Table 2). When applying this method to watersheds of very different terrain and slope characteristics, adjustments of parameters such as delivery radio, terrain slope, and road slope found in Reid (1981) are necessary, as I did with the Chilliwack River (Appendix 1,Table 3).

Field testing the model would require observations at the watershed scale and over the time scales at which the model operates (years, decades). As such, a field test is beyond our present purposes. Instead, I undertake a case study to demonstrate how the model could be applied. I anticipate that modelling prospective changes in the sediment yield due to changes in road construction and use will help develop a strategy for collecting the long term observations that are required to explicitly test the model.

2.3 Case Study

2.3.1 Area

The Chilliwack River is a tributary of the Lower Fraser River, and it is located in the Skagit Range of the Cascade Mountains, British Columbia (Figure 2.1). The watershed covers approximately 1,230 km^2 and ranges from 13 m to 2283 m ASL in altitude (Barlow et al. 2003). Most precipitation falls from October to April. Snow accumulations of up to 5 m occur at and above 1500 m elevation and yearly rainfall averages 2500 mm in the valley bottom (Brayshaw 1997). The Chilliwack River basin consists of a number of sub-basins, some of which provide an important contribution to fish habitat (See Table 2.1). A portion of the watershed lies upstream of the Chilliwack Lake, which is an effective sediment trap. Natural and road sedimentation generated from the sub-basins contained in this portion are not detectable below the lake outlet and therefore not included in this analysis. The watershed has been subject to logging, road construction, and road use in the past several decades. A large proportion of the roads are used for recreation and resource access. An assessment of the forest road conditions was completed in 1995 by SNC-Lavalin (contracted by the Ministry of Forest and Range, British Columbia). The contractors performed an inventory of forest roads in the watershed and evaluated the sediment-related risk for fish habitat. Intensive forestry activities in the watershed have resulted in the alteration of the flow regime and the generation of forestry-related landslides that remain a concern, particularly for their increasing fine sediment input into the main channel (DFO 1995; Boyle et al. 1997).

2.3.2 Hydrologic and Sediment Data

The Water Survey of Canada (WSC) undertook measurement at Vedder River crossing (station number 08MH001) of *SSC* (mgL^{-1}) from 1965 to 1976, and discharge (Ls^{-1}) from 1965 to 2006. Raw sediment data provided by the WSC is a mix of direct sampling and rating-curve extrapolations and no recent continuous data on *SSC* were available. It is important to understand that errors or bias in the results may arise from using older data sets and the best outcome is achieved by using up-to-date information. I assume that the *SSC* dynamics in the river in the period represented by the data are still valid for the present and future, although changes in the sediment sources over time might lead to shifts in the relationship *SSC*-discharge. To account for variability in this relationship, I employed a linear mixed-effects model (Chapter 1). The mixed-effects model is used to extrapolate SSC using discharge as a proxy while compensating for seasonality and hysteresis effects caused by changes in the sediment sources. The final years of data used ranged from 1965 to 2006 (42 years). The Chilliwack River is gravel-bedded throughout the reach of interest. Silt and clay-sized sediment < 62 μm are carried as washload and there are no significant fine sediment storage elements in the channel (Scott et al. 1993; EBA 2001).

2.3.3 **Forest Road Data**

Road data for the watershed were extracted from EBA (2001). EBA estimated there were 541.2 km of permanent forest roads in the entire watershed by 2001, after removing from service the 153 km of roads that were designated as high-priority for deactivation (inventoried by SNC-Lavalin 1995). GIS data were also available from Chilliwack Forest District, but we chose to use data from EBA (2001) because some licensee and spur roads were mapped as temporary roads in the GIS data. Although these roads were not intended for permanent use, they remain active due to funding and other constraints (Allan Johnsrude, BC Ministry of Forest and Range, pers. Comm., April 12 2010).

2.3.4 Forest Road Use Scenarios

I employed simulated data based on the current road-building strategy (SOMP-curr defined in Table 2.2) for the preservation of habitat of the Northern Spotted Owl in the Chilliwack Forest District of British Columbia (Sutherland et al., 2007). The simulated data was forecasted for a 100-year period using the Spatially Explicit Landscape Event Simulator (SELES) (Fall and Fall, 2001). The SELES output is given in kilometres of roads built per decade during 100 years, and it is specific for each land unit of the Chilliwack Forest District including the Chilliwack River watershed (Figure 2.2). I assumed that all simulated roads added to the production of sediments in the watershed, independently of their location. The current road-building strategy was associated with three hypothetical use levels: Light, Moderate, and Heavy, as defined above, thus comprising three scenarios. Two additional scenarios represent the current number of roads in the watershed (as of 2006) at moderate use (baseline) and a scenario representing natural sedimentation only, where roads are absent (natural). The latter is the represented by the time series of daily values n_d.

2.3.5 Calculation of Parameters Values

Peak SSC (ρ)

A broad range of hydrological research during the past 20 years describes changes in peak flows (and associated peak SSC) after forestry operations in British Columbia and the Pacific Northwest region of the USA (e.g., Jones and Grant 1996; Jones 2000; Scherer 2001, Lewis et al. 2001; Whitaker et al. 2002). Peak flow responses to forestry activities are highly variable and depend on many factors such as basin size, aspect, elevation, soils, and soil moisture storage (Jones and Grant 1996). Road-related variables, such as the overall road density or building on unstable soils, contribute a high probability (50-75%) of observing changes in peak flows and impacts on the suspended sediment yield (Carver 2001; McFarlane 2001). In Eq. 5, parameter ρ represents changes in the frequency of peak *SSC*. However, lack of data and the pervasive uncertainty contained in the parameter values for ρ make its estimation challenging. To overcome this shortcoming, I conducted a sensitivity analysis on the number of events exceeding the threshold of 25 mg L^{-1} for various combinations of ρ and ϕ in the model (Figure 2.3). Values higher than this threshold were assumed to be extreme events. The value of 25 mg L^{-1} was identified from the literature as the minimum value of SSC at which fish show some degree of physiological response to sediments (EIFAC, 1964; DFO/MELP, 1992). The value of 25 mg L^{-1} is also used as the permissive increase in suspended solids over background levels for projects affecting aquatic species habitat in British Columbia (Ministry of Environment of British Columbia, 2009). The number of events >25 mg L^{-1} is sensitive to ρ values between 0 and 2 for any combination of ϕ values. The ρ =1 and ϕ = 1 represent the baseline scenario. I assumed that ρ = 2 corresponds to a 100% decrease in road density, ρ = 0 corresponds to a 100% increase in road density, and intermediate changes in road density were interpolated linearly. Although this approach ignores some of the factors that affect peak flows, it is the best available approximation of ρ values, given the complexity and uncertainty present in the system. However, biases may arise from it.

Volume (ϕ)

The volume parameter (ϕ), which affects the total volume of suspended sediments moved during the hydrograph over the entire time series (Eq. 6), is a multiplier derived from the proportional difference between the total suspended sediment yield generated by roads $R_{k=1}$ (baseline scenario) and R_k of any other scenario. Here I assume that the simulated time series for a scenario (k) is a representative sample of the simulated 100 years of road building from the SELES model. This assumption implies that the road-related sediment generation processes that place today will remain unchanged for 100 years.

The SELES model output only specifies the amount of new roads built in each decade, not the total amount of roads present in that decade. All temporary forest roads have a predetermined service time, although given the lack of accurate information and the different road classifications (primary, secondary, and tertiary access, licensee, and spur roads) modeling individual road dynamics in this specific situation was not feasible. For this reason I did not replicate temporary forest roads individually but rather calculated the number of roads at equilibrium, which results from constant rates of construction and deactivation. Thus, a constantly exploited watershed will theoretically reach its constant road age distribution at some point during the 100-year simulation period regardless of the availability of timber.

Following this rationale, I employed discount theory in the form of annual equivalent cost (AEC), the cost of utilizing a long-lived asset (Bierman and Smidt, 1984). AEC works under the assumptions that (1) road-building cost is directly proportional to the length of the road network in the watershed, (2) timber is the asset, (3) and the road construction cost is one of the costs that society incurs to access the timber. Thus, the number of roads at equilibrium was estimated considering the discount rate and the depreciation for utility based on the forest road construction data extracted from SELES assuming constant sediment generation over time. The discount rate employed was 4% as a typical rate in forestry resource economics (See Appendix 2 for details). The total length of the road network at equilibrium (Ψ) in the watershed is then the sum of the number of permanent roads in 2006 (541.2 km) plus the roads at equilibrium. In my calculations, Ψ replaces the number of roads of length l when estimating $AFSY_{i,k}$ such that

$$AFSY_{i,k} = \psi_{i,k} u_i (LR_i)(DR_i) \tag{8}$$

Then, the volume parameter is calculated as:

$$\varphi = \frac{R_{i,k}}{R_{i,k=1}} \tag{9}$$

where $R_{i,k}$ is the sum of $AFSY_{i,k}$ across all years in the data set. Parameters values for ρ and ϕ are shown in Table 2.3.

2.4 Results

2.4.1 Sediment Yield with Respect to Use and Density

The results from this modeling exercise suggest that the road use level is the main factor affecting sediment production- rather than the density of roads alone (Figure 2.4). These results support previous road sediment generation studies (Beschta, 1978; Reid & Dunne, 1984; Wemple et al., 1996) in that heavier road use generates considerably more sediment than lighter road use. Road-related sediments are also dependent on the density of roads, although the sediment yield that occurs during low road use levels varies very little, regardless of the density (Figure 2.4). As road use levels increase, the number of extreme events increases too and the density contribution becomes more evident. When the density of roads is kept constant, there is a marked difference in the number of extreme SSC values (e.g., observations > 25 mgL^{-1}) that arise from different road use levels, showing an almost eight-fold difference between light and heavy road use levels for the scenario representing the current management plan (Figure 2.5 and Figure 2.6).

2.4.2 Influence of ϕ on Magnitude and Frequency

Parameter ϕ, which is a multiplier governed by the level of road use (heavy, moderate, and light), affects both the magnitude and the frequency of extreme sedimentation events in the watershed by modifying the volume of sediment that enters the stream (Figure 2.7). Parameter ϕ depends on the assumption that sediment generation remains constant regardless of the road age. Deactivated roads produce very little sediment because these are not subject to traffic (Reid, 1981). Based on these results, a watershed with a high density of deactivated roads is similar to a watershed with no roads from the perspective of sediment generation (i.e., without accounting for human activities other than road construction and use). That is not to say that the ecological integrity in a light-use watershed with roads would be similar to a watershed without roads- because ecological impacts of roads include many other factors in addition to sediment production (Gucinski et al., 2001).

2.4.3 Influence of ρ on the Magnitude and Frequency of Events.

Parameter ρ, which influences how the peak SSC in the time series will change for given road density conditions, impacts both the magnitude and the frequency of extreme sedimentation events in the watershed, having greater influence on the frequency (Figure 2.8). Smaller ρ values tend to increase the frequency of extreme sedimentation events that are greater than 25 mg L^{-1} , even though the total volume of sediment remains constant (Eq. 5). A watershed with a high density of roads will tend to have a higher frequency of sedimentation events exceeding a threshold than a watershed with a lower density of roads. Greater road density provides more low permeability surfaces intercepting precipitation and channeling surface and sub-surface water to stream channels, thereby increasing the peak discharge in the system, independently of the level of use of the roads. In practice, large changes in the historical maximum peak discharge values would require the removal of very large amounts of vegetative cover in addition to adding new forest roads (Jones and Grant, 1996; Jones, 2000). The simulation results suggest that ρ

has a smaller effect than ϕ for the simulated road building conditions in the study watershed where road density increases by ~21.5% with respect to the baseline scenario (Table 2.3).

2.5 Discussion

Changes in the use of the road network have well-understood sediment production effects, and many studies have addressed the subject (e.g., Beschta 1978; Reid and Dunne 1984; Meehan 1991; Hudson 2006). The results suggest that use and density of forest roads change the sediment dynamics and the hydrology of watersheds. These changes likely impact aquatic species habitat. In the case study, results indicate that reducing levels of suspended sediment yield caused by forest roads seems to be more effectively accomplished by limiting the use of available roads, rather than by controlling the road density alone. This finding does not suggests that road density is an unimportant factor when assessing impact on aquatic species because higher densities facilitate easier access and therefore more intense use (Trombulak and Frissell 2000).

Forest roads interact with many ecological and physical processes in watersheds beyond sediment production (Gucinski et al. 2001). For example, Bradford and Irvine (2000) found that increasing road density (up to 2 km/km^{-2}) was related to declines in the recruitment rate of coho salmon in the Thomson River, British Columbia, independent of other variables observed. Because changes in the peak discharge and peak suspended sediments are difficult to detect when there is not a very large change in the density of forest roads (Harr et al. 1975; Jones and Grant 1996; Jones 2000), it is important to keep in mind that ϕ has a larger contribution than ρ on the simulation results.

Although this study represents the sediment yield in the watershed of study by using a mathematical approach, factors such as road construction techniques and sediment control measures that minimize sediment output to rivers and streams are not accounted for. These sediment retention/deviation techniques can, to an extent, decrease the road-related sediment input to streams (Reiter et al., 2009). My results should be interpreted as a way to represent trends in sediment yield due to changes in forest road density and use rather than an accurate estimate of the system sediment yield. Nonetheless, this approach presents some advantages over conventional field based methods. For example, modeling provides the opportunity to study the hydrologic and sedimentation response of road use on time scales that go far beyond the costly field observations (Pike et al. 2007). Additional applications of this simulation method include modeling potential changes in the hydrology of watersheds produced by climate change; some authors suggest that climate change will increase the frequency of rainfall events that will, in turn, have a direct influence on the frequency of peak discharges (e.g., Groisman and Easterling 1994; Rodenhuis et al. 2007). It is not clear how this will impact basin sediment yield.

Employing a time series approach provides insights of the system behavior, making it possible to develop extrapolation models (Powers and Van Cleve 1991). A major assumption of my approach is that the number of forest roads will eventually reach an equilibrium, thus addressing the long-term chronic sediment production. For this reason, the results do not accurately represent the forest road dynamics or consider natural and other human sediment-generating changes in the watershed. However, this estimate allows for an approximation of the system response to changes in the use of the road network. Perhaps the method's biggest limitation is the difficulty of validating the model through fieldwork, since this will require a considerable amount of funding and effort.

2.6 Conclusions

Combining existing methods for estimating road sediment delivery with watershed-specific suspended sediment data provides a feasible framework to simulate effects of road use and density on a time series of SSC, especially in the absence of field observations. A comparison between the effects of road density and the level of road use suggests that the level of road use is the main factor in the generation of fine sediment that could enter a body of water, thus affecting admissible levels of SSC. However, higher road densities will generally provide greater access for recreational and resource extraction, thus increasing the probability of extensive use. Simply focusing on an individual sediment source (i.e., forest roads) may not provide the information required to manage watersheds sustainably. Nonetheless, the methodology presented here could provide insights into forestry impacts on watershed hydrological processes- and assist resource managers with planning effective conservation of natural resources in multiple-use watersheds.

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2.8 References

Alila, Y., and J. Beckers. 2001. Using numerical modeling to address hydrological forest Management issues in British Columbia. Hydrological Processes 15: 3371-3387.

Ashmore, P. F. M., Conly , D. deBoer, Y. Martin, E. Petticrew, and A Roy. 2000. Recent (1995–1998) Canadian research on contemporary processes of river erosion and sedimentation, and rivers mechanics. Hydrol. Processes 14: 1687–1706.

Asselman, N.E.M. 2000. Fitting and interpretation of sediment rating curves. Journal of Hydrology 234: 228-248.

B.A. Blackwell and Associates Ltd. 1999. The Greater Vancouver water district - an ecological inventory approach to fire hazard assessment and treatment for protection of water quality and other secondary values. Proceedings from the Joint Fire Science Conference and Workshop.

Barlow, J., Y. Martin, and S.E. Franklin. 2003. Detecting translational landslide scars using segmentation of Landsat ETM+ and DEM data in the northern Cascade Mountains, British Columbia. Canadian Journal of Remote Sensing 29: 510-517.

Beschta, R. 1978. Long-term patterns of sediment production following road construction and logging in the Oregon coast range. Water Resources Research 14 (6): 1011-1016.

Bierman, H., and S. Smidt. 1984. The Capital Budgeting Decision: Economic Analysis of Investment Projects. 6th ed. Macmillan, New York.

Borga, M., F. Tonelli, and J. Selleroni. 2004. A physically based model of the effects of forest roads on slope stability. Water Resources Research. 40: (12): W12202.

Boyle, C.A., L. Lavkulich, H. Schreier, E. Kiss. 1997. Changes in land cover and subsequent sffects on Lower Fraser basin ecosystems from 1827 to 1990. Environmental Management 21(2): 185–196.

Bradford, M. J. and J. R. Irvine. 2000. Land use, fishing, climate change and the decline of Thompson River, British Columbia, coho salmon. Canadian Journal of Fisheries and Aquatic Sciences 57:13-16.

Brardinoni, F., M.A. Hassan, H.O. Slaymaker. 2003. Complex mass wasting response of drainage basins to forest management in coastal British Columbia. Geomorphology. 49(1-2): 109-124.

Brayshaw, D. 1997. Factors affecting post-logging debris flow initiation in steep forested gullies of the southwestern Canadian Cordillera, Fraser Valley Region. M.Sc. Thesis, University of British Columbia, Department of Geography. British Columbia, Canada.

Department of Fisheries an Oceans Canada (DFO) 1995. Hydrology and water use for salmon streams in the Chilliwack/Lower Fraser habitat management area, British Columbia. Canadian Manuscript Report of Fisheries and Aquatic Sciences 2288.

Department of Fisheries and Oceans, Habitat Management Division; B.C.Ministry of Environment, Lands and Parks, Integrated Management Branch (DFO/MELP). 1992. Land Development Guidelines for the Protection of Aquatic Habitat. Fisheries and Oceans Canada, Pacific and Yukon Region, Vancouver; BC, and Government of British Columbia, Victoria, BC.

Dubé, K. and M. McCalmon. 2004. Technical documentation for SEDMODL, Version 2.0 road erosion/delivery model. Research Triangle Park, NC: National Council for Air and Stream Improvement, Inc. Available at http://www.ncasi.org/support/downloads. Acceded on January 23, 2010.

EBA Engineering Consultants Ltd. 2001. Chilliwack River Watershed restoration plan - final report. Available at: www.chilliwackwatershedstrategy.ca/Bibliography/DOCS/ Access on 8 June 2008

Elliot W.J., D.E. Hall, D.L. Scheele. 1999b. WEPP interface for predicting forest road runoff, erosion and sediment delivery. US Department of Agriculture, Forest Service, Rocky Mountain Research Station, San Dimas Technology and Development Center.

Elliot, W.J., D.E. Hall, S.R. Graves, D.L. Scheele. 1999a. The X-DRAIN cross drain spacing and sediment yield program Version 2.00. U.S.Department of Agriculture, Forest Service, Rocky Mountain Research Station, San Dimas Technology and Development Center.

European Inland Fisheries Advisory Commission (EIFAC). 1964. Water quality criteria for European freshwater fish. Report on finely divided solids and inland fisheries. European Inland Fisheries Advisory Commission. Food and Agriculture Organization of the United Nations. Rome.

Fall, A., and J. Fall. 2001. A domain-specific language for models of landscape dynamics. Ecological Modeling 141(1-3): 1-18.

Fraser Valley Regional District. 2005. The Chilliwack River Watershed: a backgrounder. Report prepared for the Chilliwack River Watershed Strategy .

Gomi, T., D. Moore, and M.A. Hassan. 2005. Suspended sediment dynamics in small forest streams of the Pacific Northwest. Journal of the American Water Resources Association (JAWRA) 41(4):877-898.

Greater Vancouver Regional District (GVRD). 1999. CD Annex to GVRD Analysis Report Watershed Management Plan #5.

Groisman, P. Ya, and D. R. Easterling. 1994. Variability and trends of precipitation and snowfall over the United States and Canada. J. of Climate 7:184-205.

Gucinski, H., M.J Furniss, R.R., Ziemer, and M.H. Brookes. 2001. Forest roads: a synthesis of scientific information. Gen. Tech. Rep. PNWGTR- 509. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 103 pp.

Hagans, D.K., Weaver W.E., and M.A. Madej. 1986. Long term on-site and off-site effects of logging and erosion in the redwood creek basin, northern california. National Council of the Paper Industry for Air and Stream improvement. Technical Bulletin 490: 38-66.

Harr, R. D., W. C. Harper, J. T. Krygier, and F. S. Hsieh, 1975. Changes in storm hydrographs after road building and clear-cutting in the Oregon Coast Range, Water Resour. Res., 11, 436–444,

Hassan, M. A., M.Church, T. Lisle, F. Brardinoni, L. Benda, and G. Grant. 2005. Sediment transport and channel morphology of small, forested streams. Journal of the American Water Resources Association (JAWRA) 41(4):853-876.

Hudson, R. 2006. Storm-based sediment budgets in a partially harvested watershed in coastal British Columbia. Forest Research Technical Report 9, British Columbia Ministry of Forests and Range.

Jones, J. 2000. Hydrological processess and peak discharge response to forest removal, regrowth, and roads in 10 small experimental basins, western cascades, Oregon. Water Resources Research , 36 (9): 2621-2642.

Jones, J., and G. Grant. 1996. Peak flow responses to clear-cutting and roads in small and large basins, western cascades, Oregon. Water Resources Research, 32 (4): 959-974.

Kerr, S.J. 1995. Silt, turbidity and suspended sediments in the aquatic environment: an annotated bibliography and literature review. Ontario Ministry of Natural Resources, Southern Region Science & Technology Transfer Unit Technical Report TR-008. 277 pp.

Lewis, J., S. Mori, E. Keppeler, and R. Ziemer. 2001. Impacts of logging on storm peak flows, flow volumes and suspended sediment loads in Caspar Creek, California. In: Land use and watersheds: human influence on hydrology and geomorphology in urban and forest areas. American Geophysical Union.

McFarlane, B.E. 2001. Retrospective Analysis of the effects of harvesting on peak flow in South western British Columbia. In Toews, D.A.A. and S. Chatwin (editors). Watershed assessment in the southern interior of British Columbia: workshop proceedings, March 9–10, 2001, Penticton, British Columbia, Canada. B.C. Ministry of Forests, Research Branch, Victoria, B.C. Working Paper 57.

Meehan, W.R. (ed). 1991. Influences of forest and rangeland management on salmonid fishes and their habitats. Spec. Publ. 19. Bethesda, MD. American Fisheries Society. 751 p.

Ministry of Environment of British Columbia. 2009. Guidelines for measuring turbidity and suspended sediment concentrations. Available at: www.env.gov.bc.ca/wat/wq/BCguidelines/samp_strat/sampstrat.html Accessed 10 January 2009.

Moore, D., and S. Wondzell. 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: a review. Journal of the American Water Resources Association. 41 (4): 763-784.

Mweempwa, H. 1993. Character and controls of suspended-sediment concentration and discharge effectiveness, Fraser River, British Columbia, Canada. Phd. Thesis Geography Department. Simon Fraser University.

Naiman, R. J., T.J. Beechie, L.E. Benda, P.A. Bisson, L.H. MacDonald, and M.D O'Connor. 1992. Fundamental elements of ecological healthy watersheds in the Pacific Northwest coastal ecoregion. In Naiman, R. J. (ed.), Watershed management balancing sustainability and environmental change. Springer, pp 127–189.

Northcote, T.G. and G.F. Hartman, (eds). 2004. Fishes and forestry: worldwide watershed interactions and management. Blackwell Science, Oxford, UK. p 46.

Phillips, J.M., B.W. Webb, D.E. Walling and G.J.L. Leeks. 1999. Estimating the suspended sediment loads of rivers in the LOIS study area using infrequent samples. Hydrobiol. 13:1035–1050.

Pike R.G., T. Redding, D. Wilford, R.D. Moore, G. Ice, M. Reiter, and Toews D.A. 2007. Detecting and predicting changes in watersheds. In Compendium of forest hydrology and geomorphology in British Columbia R.G. Pike et al. (eds). B.C. Ministry of Forests and Range Research Branch, Victoria, B.C. and FORREX Forest Research Extension Partnership, Kamloops, B.C. Land Management Handbook (TBD).

Powers, R. and K., Van Cleve. 1991. Long-term ecological research in temperate and boreal forest ecosystems. Agronomy Journal. 83: 11-24.

Reiter M, J.T., Heffner, S. Beech, T Turner, and R.E. Bilby. 2009. Temporal and spatial turbidity patterns over 30 years in a managed forest of Western Washington. Journal of the American Water Resources Association (JAWRA) 45(3):793-808.

Ried, L. 1981. Sediment production from gravel-surfaced forest roads, Clearwater basin, Washington. University of Washington. Seattle, USA: Fisheries Research Institute Publication.

Ried, L. and T. Dunne. 1984. Sediment production from forest road surfaces. Water Resources Research 20 (11): 1753-1761.

Rodenhuis, D., K.E. Bennett, A. Werner, T.Q. Murdock, and D. Bronaugh. 2007. Hydro-climatology and future climate impacts in British Columbia. Pacific Climate Impacts Consortium, Univ. Victoria, Victoria, B.C.

Scherer, B. 2001. Effects of changes in forest cover on stream flow: A literature review. In Toews, D.A.A. and S. Chatwin (editors). Watershed assessment in the southern interior of British Columbia: workshop proceedings, March 9–10, 2001, Penticton, British Columbia, Canada. B.C. Ministry of Forests, Research Branch, Victoria, B.C. Working Paper 57.

Scott, K.J., R.U. Kistritz and V.J. Galay 1993. Chilliwack River water quality and fish enhancement opportunities study. Scott Resource Services, British Columbia Habitat Conservation Fund. Available at:http://www.chilliwackwatershedstrategy.ca/Bibliography/CRWS%20Reference%20Database.htm. Accessed on 8 September, 2009

Smith, B.P.G., P.S. Naden, G.J.L. Leeks, and P.D. Wass. 2003a. Characterising the fine sediment budget of the River Swale, Yorkshire, UK during the 1994 to 1995 winter season. Hydrobiologia 494: 135-143.

Smith, B.P.G., P.S. Naden, G.J.L. Leeks, and P.D. Wass. 2003b. The influence of storm events on fine sediment transport, erosion and deposition within the River Swale, Yorkshire, UK. Science of the Total Environment, 314/316; 451-474.

Sutherland, G.D., D.T. O’Brien, S.A. Fall, F.L. Waterhouse, A.S., Harestad, and J.B. Buchanan (eds). 2007. A framework to support landscape analyses of habitat supply and effects on populations of forest-dwelling species: a case study based on the Northern spotted owl. Ministry of Forests and Range Forest Science Program. Technical Report 038.

Topping, D. J., S.A., Wright, T. S., Melis, and D.M. Rubin. 2007. High-resolution measurement of suspended sediment concentrations and grain size in the Colorado River in Grand Canyon using a multi-frequency acoustic system. In Proceedings of the 10th International Symposium on River Sedimentation, Moscow, Russia, 1-4 August, 330-339.

Trombulak, S.C. and C.A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology 14:1, 18-30.

Wald, A. 1975. Impact of truck traffic and road maintenance on suspended sediment yield for 14 standard forest roads. M.Sc. Thesis, University of Washington, Seattle, USA.

Water Survey of Canada. 2008. HYDAT. Available at: www.wsc.ec.gc.ca. Accessed on 15 October 2008.

Webb, B.W., J.M. Phillips, D.E. Walling, I.G. Littlewood, C.D. Watts and G.J.L. Leeks. 1997. Load estimation methodologies for British rivers and their relevance to the LOIS RACS(R) programme. Sci Total Environ 194: 379–389.

Welsh, M.J. 2008. Sediment production and delivery from forest roads and off-highway vehicle trails in the Upper South Platte River watershed, Colorado. Unpublished MS Thesis. Fort Collins, CO: Colorado State University. p 237.

Wemple, B.C., J.A. Jones, and G.E. Grant. 1996. Channel network extension by logging roads in two basins, western Cascades, Oregon. Water Resources Research 32:1195-1207.

Whitaker, A., Y. Alila, J. Beckers, and D.A.A. Toews. 2002. Evaluating peak flow sensitivity to clearcutting in different elevation bands of a snowmelt-dominated mountainous catchment. Water Resources Research 38: doe: 10.1029/2001514.

2.9 Figures and Tables

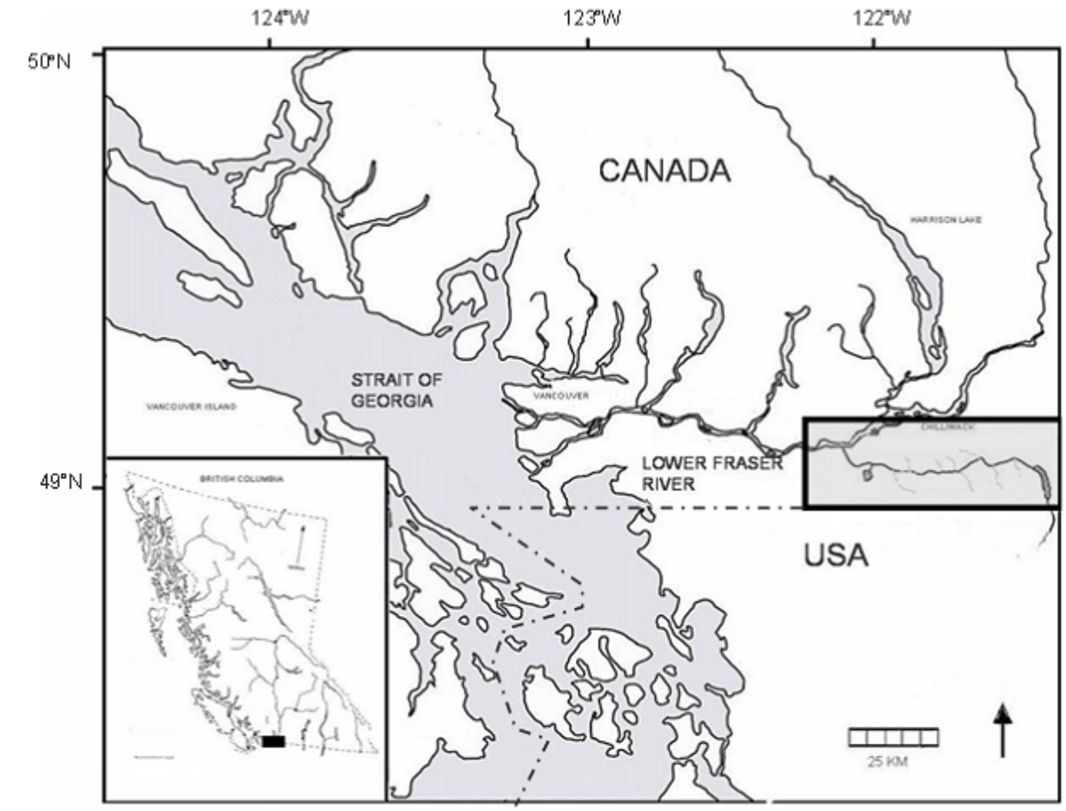

Figure 2.1. The Chilliwack River, a tributary for the Lower Fraser River.

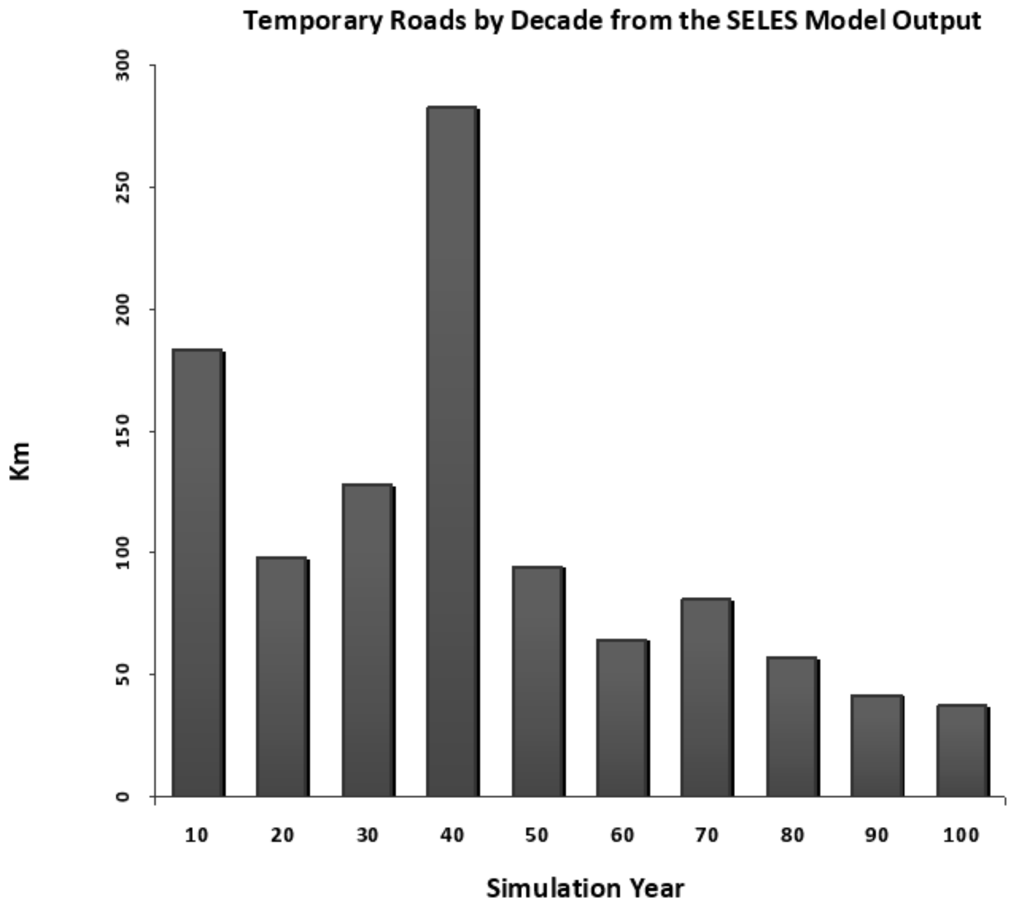

Figure 2.2. Length (km) of temporary forest roads forecasted for a 100-year period in the Chilliwack watershed under the current forest management plan (Sutherland et al, 2007) derived from the Spatially Explicit Landscape Event Simulator (SELES) (Fall and Fall, 2001).

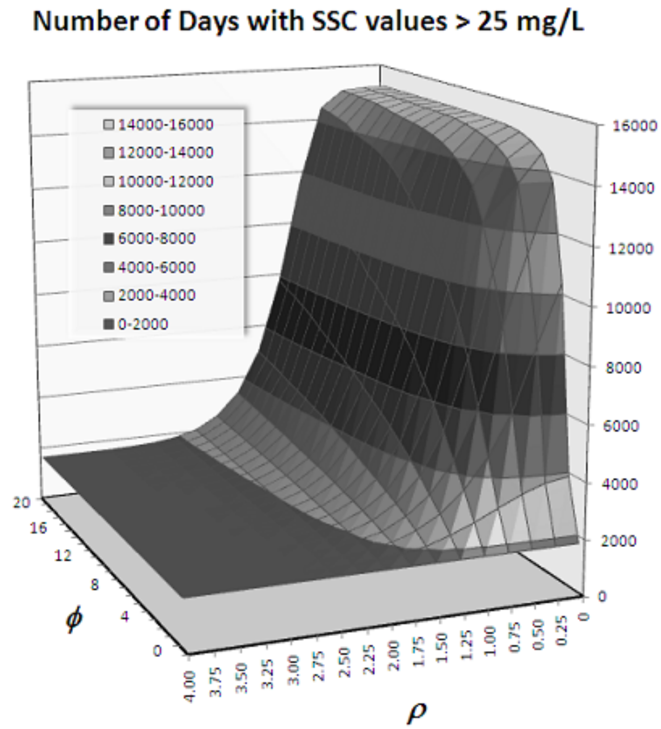

Figure 2.3. **Number of observations exceeding the threshold of 25 mg L^{-1} for various combinations of ρ and ϕ relative to the baseline (ρ=1 and ϕ= 1). The number of events >25 mg L^{-1} is sensitive to ρ values between 0 and 2 for any combination of ϕ values, increasing at higher values of ϕ.**

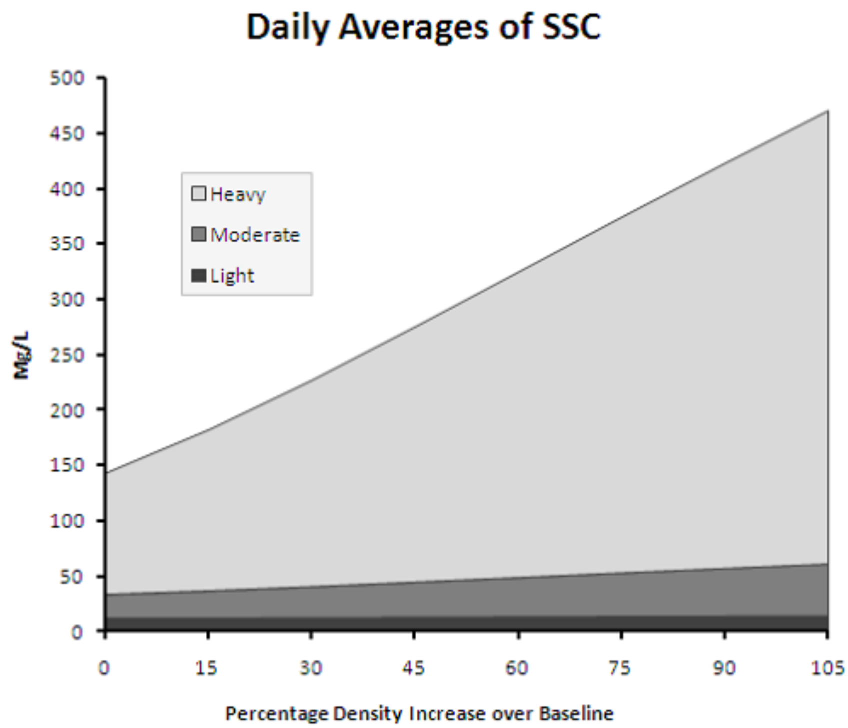

Figure 2.4. Daily averages of SSC under different conditions of road use for increasing road density over the baseline (simulated period).

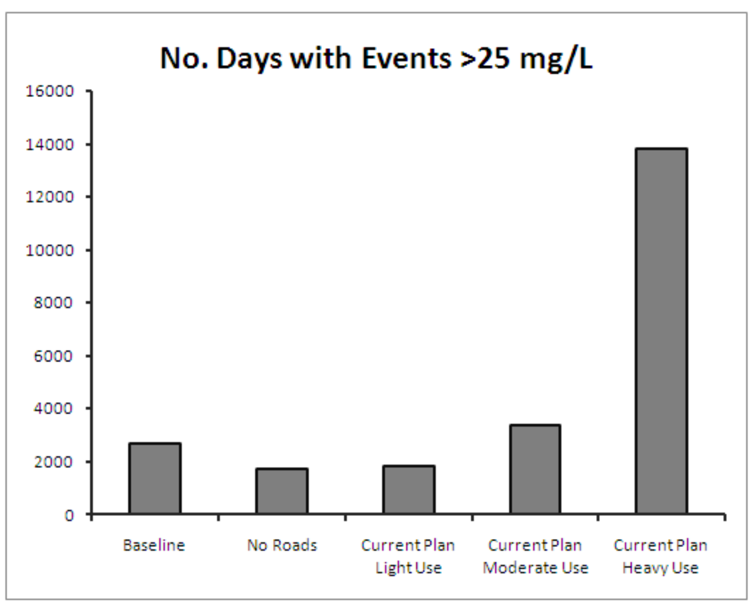

Figure 2.5. Number of observations exceeding 25mg L^{-1} per scenario.

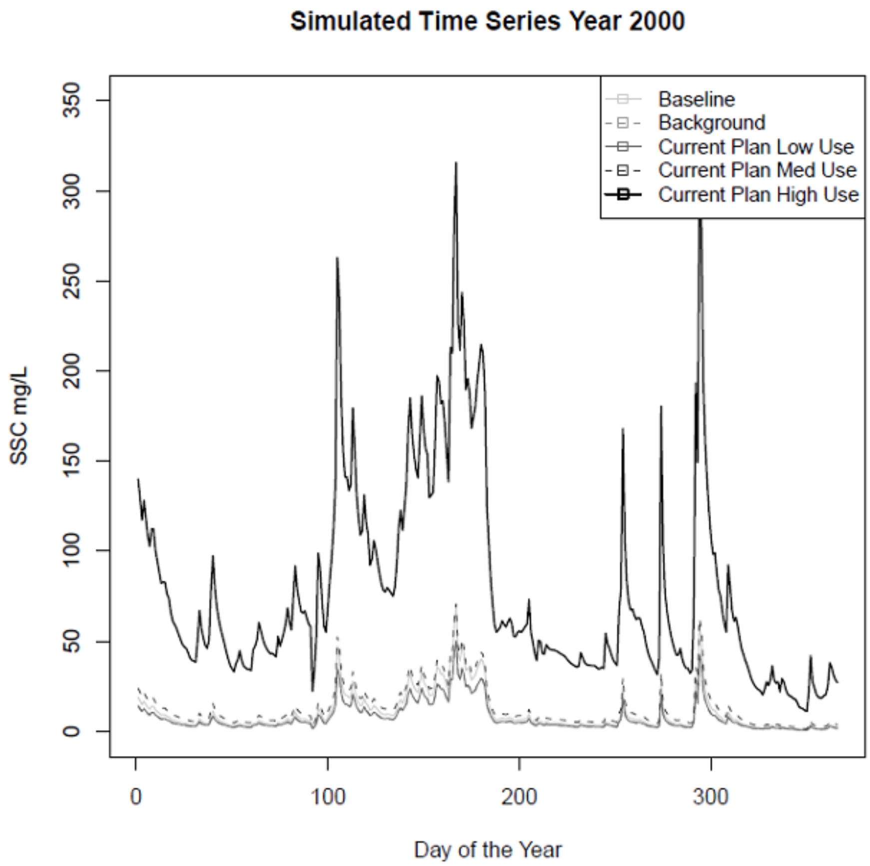

Figure 2.6. Simulated changes in SSC for the year 2000.

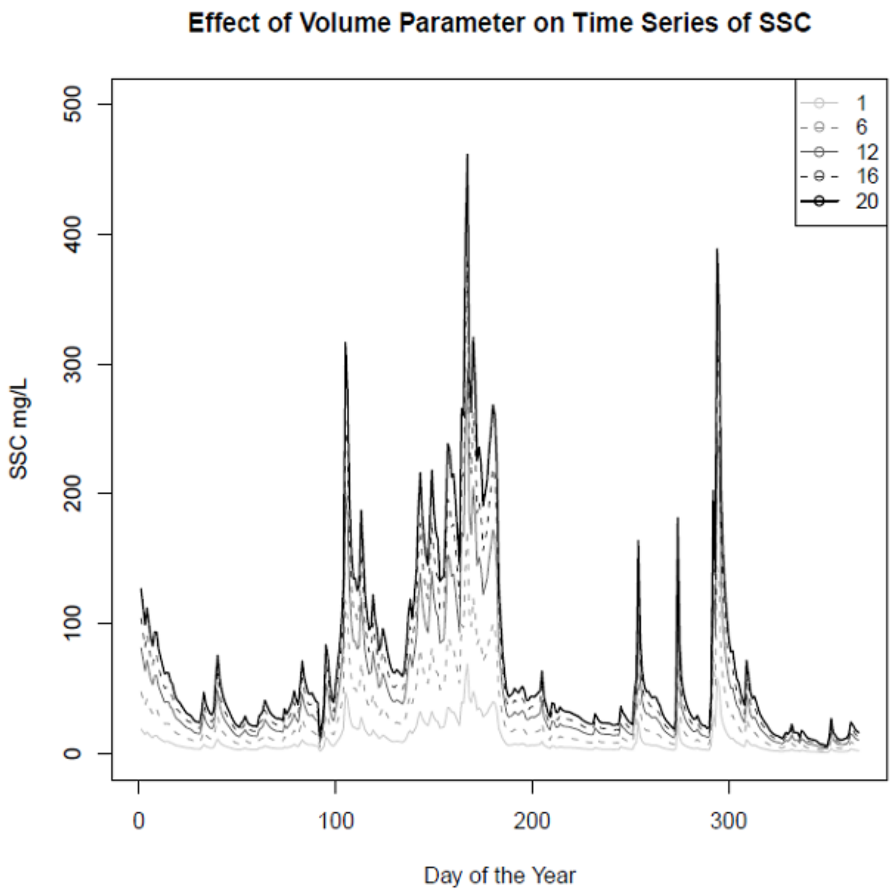

Figure 2.7. SSC time series modified by ϕ parameter values 6,12,16, and 20 with respect to the baseline (keeping ρ =1 to observe the impact of varying ϕ values).

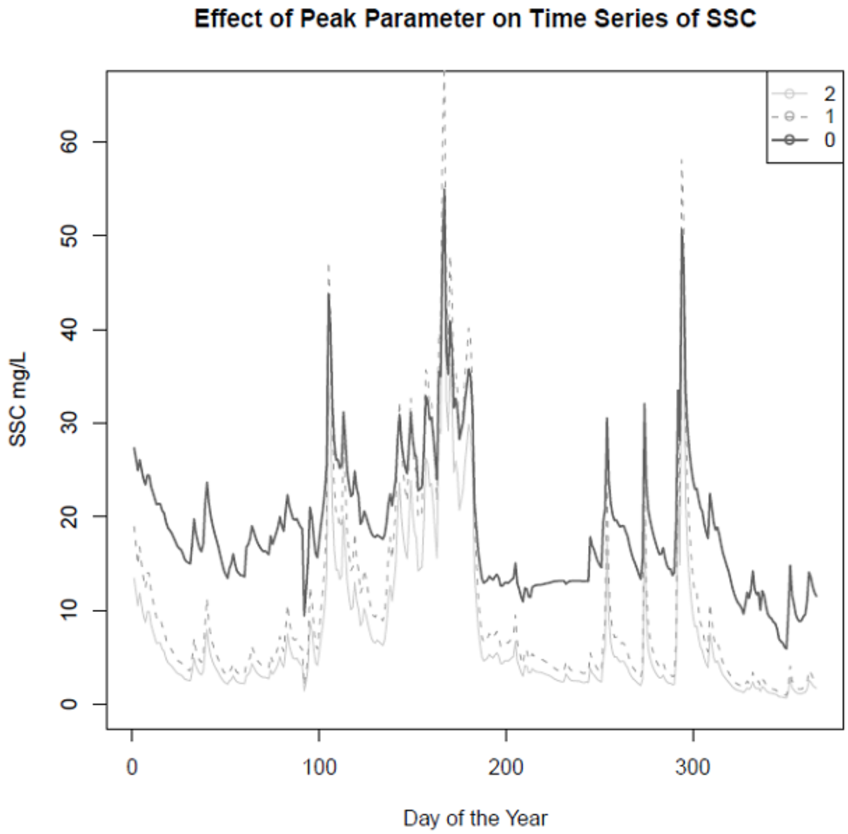

Figure 2.8. **Time series modified by ρ parameter values 2, 1, and 0 with respect to the baseline while keeping ϕ =1 (as in the baseline scenario) to observe the impact of varying ρ values.**

Table 2.1. Sub-basins of the Chilliwack River

Sub-Basin	Area (ha)	Area Km²	Permanent Roads in 1995 (km)	Sub-Basin % of total area	Influence on Aquatic habitat
Foley/Airplane Creek	7922	79	61	6.0	Significant
Centre Creek	4813	48	49	3.7	Significant
Chipmunk Creek	4644	46	95	3.5	Significant
Slesse Creek	16600	166	85	12.6	Significant
Nesakwatch Creek	4908	49	33	3.7	Significant
Borden Creek	4016	40	49	3.1	Significant
Tamihi Creek	5294	53	76	4.0	Significant
Little Tamihi Creek	4200*	42	62	3.2	Significant
Liumchen Creek	3994	40	28	3.0	Significant
Main channel and other	64163	642	136	48.9	Significant
Pierce Creek	4100*	41	22	3.1	Significant
Depot Creek	2821	28	NA	2.1	Minimal
Paleface Creek	3825	38	NA	2.9	Minimal
Total Watershed	131300	1313	**695**		

*approximate values

Table 2.2. **Forest road construction and use for selected scenarios in the Chilliwack Forest District. Sutherland et al. (2007) described different forest management scenarios based on the 1997 Spotted Owl Management Plan guidelines. The scenario SOMP-curr closely relates to current management practices within the Chilliwack Forest District in relation to the protection of Spotted Owl habiat (current plan).**

Forest Management Scenario	Description	Use level	Annual Fine Sediment Yield (tonnes year $^{-1}$)	Length at Constant Age Distribution (Km)
Baseline	The current (as of 2006) total length of the road network in the Chilliwack river watershed (permanent and temporary roads).	Moderate	7274	541.2
No-Roads (Natural)	Assumes that all roads in the watershed have been removed. However other sources of sediments still contribute to the potential changes in SSC.	NA	NA	NA
Current Plan (SOMP-curr)	Based on the current timber harvesting and road construction plan for the Chilliwack Forest District for 100 years. Implying that a minimum of 67% of Old Growth stands are reserved within long-term activity centres.	Light	806	662.7
		Moderate	8906	
		Heavy	106025	

Table 2.3. **Estimated parameter values of ϕ and ρ, Annual Fine Sediment Yield (AFSY), and changes in density with respect to the baseline for all forest management scenarios.**

Road Use	Forest Manag. Plan	Lengh of Road Network *(Km)*	AFSY tonnes. year $^{-1}$	ϕ	Density of Roads *km/km²*	% Increase in Density with respect to baseline	ρ
Moderate	Baseline	541.2	7274	1.00	0.41	0.00	1.00
None	No Roads	0.0	0	NA	0.00	-100.00	NA
Light	Current Plan	662.7	806	0.1	0.50	22.4	0.8
Moderate	Current Plan	662.7	8906	1.2	0.50	22.4	0.8
Heavy	Current Plan	662.7	106025	14.6	0.50	22.4	0.8

2.10 Appendix 1

Table 1a. **AFSY parameter values (ton/Km/yr) adapted from Metro Vancouver (GVRD 1999) to estimate the annual fine sediment yield (AFSY) from forest roads in their three managed watersheds (Seymour, Capilano, and Coquitlam).**

Road Slope	Light Use	Moderate	Heavy	Loss Ratio
0 to 2.5%	0.2	2.1	25	0.7
2.5 to 7.5%	1.9	21	250	0.8
7.5 to 12.5%	5.1	57	675	0.9
>12.5%	10.5	105	1,250	1.0

Table 1b. **Delivery Ratios Parameter values adapted from Metro Vancouver (GVRD 1999) based on the results of Reid (1981).**

Slope of the Terrain	No major or ephemeral streams	One ephemeral stream but no major streams	Major stream or more than one ephemeral stream
0 to 27%	0.05	0.3	0.5
28% to 49%	0.3	0.5	0.8
>50%	0.5	0.8	1

Table 2. Comparison of the Characteristics of the Capilano and Chilliwack River (CRW) Watersheds

Watershed Characteristic	Capilano Watershed	CRW
Climate	- Maritime	- Maritime and Continental (4)
Average Annual Precipitation (mm)	- Lower Elevation = 3,159 mm - Higher Elevation = 4,500 mm (1)	- Precipitation can be as high as 215.5 mm per month in the winter months and as low as 53.9 millimeters in the summer months (4)
Geology	- Bedrock consists primarily of intrusive igneous rocks(Roddick 1965) (1) - Gentle and moderate slopes, especially at mid to low elevations, are mantled by glacial till (2)	- Bedrock consists primarily of metamorphic,sedimentary volcanic, and sedimentary (sandstone) rock (4) - Valley bottom consists primarily of glacial, fluvial, and glacio-fluvial deposits (4)
Elevation (m)	- Pacific Ranges of the Coast Mountains (2) - 155 m to 1725 m (1)	- Skagit Range of the Cascade Mountains (5) - 660 m to 1200 m (6)
Biogeoclimatic Ecosystem Classification	- Coastal Western Hemlock Zone (CWH) and the Mountain Hemlock Zone (MH), with a very small area of the Alpine Tundra Zone (AT) (3)	- Coastal Western Hemlock zone and the Mountain Hemlock zone.(4)
(1) Greater Vancouver Regional District, 1999 (2) Brardinoni et al., 2003 (3) B.A. Blackwell and Associates Ltd., 1999 (4) Fraser Valley Regional District, 2005 (5) Brayshaw, 1997 (6) EBA Engineering Consultants Ltd., 2001		

Table 3. Baseline scenario for the Chilliwack River watershed (CRW)

Slope of the roads	Slope of the terrain	Streams	Road use	AFSY (tons/year)
7.5 to 12.5 degrees*	28% to 49%	One major stream or more than one ephemeral stream	Moderate (1 to 4 logging truck passes per day)	28,500

*Parameter value is an estimate based on best available information.

2.11 Appendix 2

Procedure to estimate the number of roads at equilibrium using the Equivalent Annual Cost (EAC).

The Equivalent Annual Cost (EAC) methodology was employed to account for road construction and deactivation, and thus provides a more realistic evaluation of the cumulative length of the road network likely to be present at any given time over the 100-year simulation period (roads at equilibrium). To estimate the EAC, I employed the following equation (Bierman and Smidt, 1984):

$$EAC = \frac{NPV}{1-(1+r)^{t}}$$

where NPV is the net present value of temporary roads built over 100 years. NPV can be defined as the present monetary value of an investment that can yield benefits into the future (e.g. a planted forest) accounting for depreciation and minus the initial investment. This information was extracted from the SELES model. The discount rate (r) acts as the deactivation rate (i.e. rate at which roads are no longer used in forestry operations). AEC works under the assumptions that road-building cost is directly proportional to the length of the road network in the watershed, timber is the asset, and the road construction cost is one of the costs that society incurs to access the timber.

Chapter 3:
Population Responses of Chinook and Coho Salmon to Suspended Sediments in a Coastal Watershed of British Columbia

3.1 Introduction

No one knows with certainty what is causing the declines of many populations of Pacific salmon. In the freshwater part of their life history, many factors have been identified as responsible for declines in population numbers, including habitat damage, pollution, high water temperatures, changes in hydrology, and increased sediment levels due to human activities (NRC, 1996; Reice et al., 1990). When considering only increased sediment loadings, forestry practices and other resource extraction activities- such as mining, increase the sediment concentrations above the background (natural) levels (Meehan, 1991; Naiman et. al., 1992; Gucinski et al., 2001). In British Columbia and the Pacific Northwest of USA, sediments and their effects on aquatic species have been intensively studied, particularly the response of salmonids to increasing suspended sediment concentrations (*SSC*) (Kerr, 1995; Trombulak and Frissell, 2000). However, to my knowledge, no study has yet addressed suspended sediment impacts at the population level over time and looked at the overall effects on salmonid productivity.

Increases of suspended and deposited sediments above background levels can adversely impact fish throughout their life history. The proportion of fine sediment deposited in the spawning gravels has been negatively related to the survival of incubating salmonids from eggs to emergent fry. Deposited sediments affect redds and alevin survival by reducing intergravel water flow and inhibiting alevin movements (e.g., Megahan et al., 1978; Cederholm and Reid, 1987; Chapman 1988; MacDonald et al., 1991). Increased sedimentation can also smother populations of certain benthic organism and lower algal production, both of which directly affect the diets of juvenile salmonids (Alexander and

Hansen, 1986; Everest et. al., 1987). In addition, sediment deposits in the stream can decrease habitat availability and may cause loss of cover, thus increasing predation risks (Bjornn et al., 1977; Chapman and McLeod, 1987).

Increases in SSC affect juvenile fish by reducing growth and feeding rates, promoting gill abrasion, and impairing foraging efficiency (Newcombe and Jensen, 1996). Growth delays related to SSC could have important effects at the population level (Suttle et. al., 2004). Growth impeded by SSC has been documented for coho salmon (Smith and Sykora, 1976; Sigler et al., 1984) and Chinook salmon (Birtwell and Korstrom, 2002). Suspended sediments cause cloudiness in the water, impairing foraging efficiency in salmonids and other fish and interfering with feeding processes (Noggle, 1978; Berg and Northcote, 1985; Gregory and Northcote, 1993). Adult salmonids show more resilience in the presence of elevated SSC than juveniles; however, very high SSC and long exposure reduces the success of fertilization and causes behavioural changes (Whitman et al.,1982; Galbraith et al., 2006).

Few studies have developed a comprehensive framework that defines functional and physiological responses of salmonids to sediments. Some of the reasons include the complexity of factors involved in the response of salmonids to sediments, population-specific behaviours, and the difficulty of data collection. However, Newcombe and Jensen (1996) designed a concentration–duration (dose) response model as a predictive management tool for assessing the environmental effects of suspended sediments on anadromous and freshwater fish at different life stages. Their study is a meta-analysis of 264 data sets, resulting in what the authors call the severity of ill effect (SEV), a ranked scale with empirically-determined scores, calculated by accounting for concentration and duration of exposure in six fish categories and at all life stages. In their model, the SEV scores include a series of behavioural, sublethal, and lethal effects.

Although there is a good qualitative understanding of how elevated levels of sediments affect fish and other aquatic biota, there is a lack of adequate quantitative tools that allow managers to make inferences regarding sediment impacts on fish habitat and responses at the population level (Moore and Wondzell, 2005). In recent years in British Columbia and the American Pacific Northwest, several models have been used in an attempt to represent forest-fish dynamics with the intention of creating management tools. Alexander et al. (1998) developed the Fish Forestry Interaction Program Management Model (FFIP-MM), a first prototype to link upslope processes, sediment and large organic debris transport with habitat conditions for small watersheds (<40 km^2). Heinzelman (2002) developed an integrated modeling framework to assess trade-offs associated with forestry management involving habitat conditions for coho salmon. Harvey and Railsback (2009) employed a spatially explicit, individual-based model to estimate the cumulative effects of forestry activities on cutthroat trout habitat. These models have focused primarily on assessing the impacts of forestry activities on fish habitat (e.g., channel scour). Although useful as research tools, none of the models are currently used in decision making. Some of the challenges of modeling natural processes such as forest-fish interactions include representing the complex and multiple interactions occurring in watersheds, lag effects, up-scaling, and insufficient data (Northcote and Hartman, 2004).

Rather than focusing on geomorphology and fluvial processes, my approach aims to estimate direct physiological responses of fish populations to sedimentation events using a life history model. Recent models of Pacific salmon life histories include specific life-stage density-dependent mechanisms (e.g., Sharma et al., 2005), while others combine a set of environmental variables and their effects on life stages (e.g., Scheuerell and Hilborn, 2009) in addition to applying techniques such as Monte Carlo simulations which account for uncertainty in parameter estimation and environmental variables.

This study's main purpose is to develop a quantitative framework for estimating the effects of extreme suspended-sediment events on populations of Chinook and coho salmon in a medium-sized coastal watershed of the Lower Fraser River. I used the Chilliwack River as a sample watershed (case study), and existing information on the effects of exposure to SSC on salmonids, particularly the study by Newcombe and Jensen (1996). In addition, I used a Monte Carlo simulation approach to account for uncertainty in the parameter estimates and data limitations by evaluating the variability in the populations' response. This methodology suits better my research needs because it allows evaluating the direct physiological response to suspended sediments at the population level. Other simulation techniques, such as the ones described above, would not work in this situation because they focus mainly on habitat changes regardless of the population specific effects.

3.2 Methods

The methodology I used to estimate population responses of Chinook (*Oncorhynchus tshawytscha*) and coho (*Oncorhynchus kisutch*) salmon involved a stochastic life history population dynamics model- coupled with time series data of daily SSC and a dose response model (Newcombe and Jensen, 1996). The intention behind including stochasticity in the model was to estimate the variability in the populations' responses to extreme sedimentation events (>25 mg L^{-1}) during the simulation period. Sedimentation events were derived from an existing set of simulated time series of daily SSC from several forest-road management scenarios (Chapter 2). In addition, I applied a hypothetical concentration linear multiplier to the baseline time series of SSC . The purpose of this multiplier was to progressively increase the level of SSC exposure to compare the trend and the differences in the response of each population, regardless of the suspended sediments that could be produced by the forest-road management scenarios. The model presented here is an alternative to extensive field work to investigate the effects of extreme sedimentation events at the population level, given that direct observation of sedimentation effects of salmonid populations involves many logistic challenges and confounding factors.

3.2.1 Data

The primary data required to run the model are daily values of SSC (mg L^{-1}) collected for a period of time sufficient to represent seasonal SSC patterns in the watershed of interest. In this study, SSC data were extracted from Chapter 2, where I estimated the annual fine sediment yield (AFSY) (tons $year^{-1}$) and provided a set of five simulated time series of daily SSC data (scenarios). Each scenario represents the sedimentation caused by different conditions of forest road use and building density in the Chilliwack River Watershed (Figure 3.1).

Marine and freshwater natural mortalities for both Chinook and coho were extracted from Bradford (1995) who estimated the mortality values based on natural populations. Stage-specific production data were not available for either coho or Chinook in the Chilliwack River watershed. However, production parameters were estimated using alternative methods explained in detail below (Section 3.2.3). Historical escapement data for the Chilliwack River were provided by the Canadian Department of Fisheries and Oceans NuSEDS V2.0 Regional Adult Salmon Escapement Database (DFO, 2010a,b). Harvest rates for coho salmon were obtained from the *Southern Coho Management Plan* (PSC, 2009a), and Chinook historical harvest rates were extracted from the Pacific Salmon Commission (PSC, 2009b) directly for Chilliwack River Fall Chinook (indicator stock for the lower Fraser River run).

Hatchery production is not included in the analysis for either Chinook or coho because hatchery smolts are not subject to the impact of SSC in fresh water; therefore, inclusion of hatchery-reared fish can bias the abundance estimates, given the large proportion of these fish that join the naturally-produced smolts (Gardener et al., 2004). For example, in the late 1990s the estimates of juvenile coho abundance in the Strait of Georgia indicated an average of 70% of all fish were of hatchery origin, although this proportion has decreased in recent years (Beamish et al., 2008). Because hatchery-reared fish are not included, the final model outputs will not reflect the current population dynamics of the system. Instead, the model should be understood as a tool to estimate the theoretical effects of extreme sedimentation events on natural populations only, unaffected by extreme ocean conditions and other environmental factors.

3.2.2 Life History

I developed a stochastic stage-structured model for ocean-type fall Chinook and coho salmon in which life stanza, i.e., the periods of development between life stages (Baker, 2009), coincide with seasons (Figure 3.2 and Figure 3.3). The reason for choosing this approach was to synchronize the highly seasonal variations in SSC with the vulnerable life histories of Chinook and coho because for many southern British Columbia salmon stocks, smoltification, migration, and spawning are cyclical life history traits related to seasonal photoperiodicity, water temperatures, and flow conditions (Shapovalov and Taft, 1954; Neave, 1964; Burgner, 1980). Information on timing of the life history stanzas for the Chilliwack River Chinook and coho were obtained from expert opinion (Ron Valer, Chilliwack River fish hatchery, pers. comm. June 22, 2009) and complemented with other information sources (Bradford et al., 2000; Sandercock, 1991, Groot and Margolis, 1991). The model runs in R, version 2.10.0 (R Development Core Team, 2010).

Life history models offer simple frameworks for assessing the contribution of environmental variables or habitat quality to population abundance (Hilborn, 2009). How well the model does so, however, depend on the species, the scale, and the modeling technique (Pyper et al., 2005; Peterman et al., 2009). The model does not incorporate all natural and anthropogenic factors that may influence the life cycle of Chinook and coho salmon, such as ocean conditions, extreme natural freshwater and marine mortality, climatic regimes, hatchery production, and other factors that may present confounding effects (Healey, 1991). In order to detect effects that are exclusively attributable to extreme sedimentation events, selected natural and human factors in the model affecting the marine portion of their life cycle were incorporated as random variables with values obtained mainly from the literature. These variables included freshwater and marine survival, log-normal stock-recruitment error, annual jacking proportions, and harvest rates. The inclusion of environmental variables allows us to isolate factors that affect population processes at several stages (Peterman et. al., 2009).

The 3-year life cycle of Lower Fraser coho salmon cohort c (see Figure 3.2 and Table 3.2 for subscripts) begins in the fall of year 0, when an initial number of spawners (S_{c-3}) (not shown in Figure 3.2) produce eggs (E) following a Ricker stock-recruit function that accounts for density dependence. Eggs hatch in the beginning of the spring of year 1 to become fry-1 ($F1$), which continue their fresh water residence becoming fry-2 ($F2$), fry 3 ($F3$), and smolts (Sm) in the summer, fall, and winter respectively. At the same time, in the beginning of the winter of year 2, a new cohort *c+1* eggs are laid by initial spawners of cohort *c-2* which will continue the same cycle, although one year behind of cohort *c*. During the spring of year 2, smolts (*Sm*) begin their migration downstream, becoming ocean-1 (*O1*). A proportion of ocean-1 fish become jacks (*Jk*) which spawn with adult spawners of cohort *c-1* while the remainder continue to ocean-2 (*O2*) until the end of the summer of year 3. All ocean-2 fish become returns (*Rt*) in the fall of year 3, and are joined by the jacks of cohort *c+1*. A harvest rate (*HR*) is applied to the returns, which allows for extracting the catch (*Ca*). The fish that survive the fishery become the spawners (*S*) that will produce the cohort *c+3*. These dynamics continue for the simulation period.

The 5-year life cycle of the ocean-type fall Chinook salmon cohort *c* (see Figure 3.3 and Table 3.3 for subscripts) begins in the fall of year 0, when an initial number of spawners (S_{c-5}) (not shown in Figure 3.3) produce eggs (*E*) following a Ricker stock-recruit function that accounts for density dependence. Eggs hatch to become emergent fry (*F*) in the spring, which continue their freshwater residence until the beginning of the summer (*Su*) for about 90 days. These become smolts (*Sm*) during the summer and finish their migration downstream by the beginning of the fall, becoming ocean-1 (*O1*). In the beginning of winter of year 2, a new cohort (*c+1*) of eggs are laid by initial spawners of the cohort *c-4*, which will continue the same cycle, although one year behind cohort c. In the summer of year 2, ocean-1 (*O1*) become ocean-2 (*O2*). In the summer of year 3, a proportion (30%) of ocean-2 (*O2*) become jacksA, which spawn in the fall with jacksB_{c-1} and Spawners (S_{c-2}). The remaining ocean-2 develop into ocean-3 (*O3*). In the summer of year 4, 60% of ocean-3 become jacksB, which spawn in the fall with jacksA_{c+1} and spawners (S_{c-1}). The remaining ocean-3 turn into ocean-4 (*O4*), as the majority of the Lower Fraser River Chinook tend to spawn near age 4 (Godfrey, Worlund, and Bilton, 1968). In the summer of year 5, all ocean-4 become ocean-5 (*O5*), joined in the Fall by the jacksA_{c+2} and $\text{jacks}B_{c+1}$ composing the Returns (*Rt*). A variable harvest rate (*HR*) is applied to the returns (*Rt*), which allows for subtracting the catch (*Ca*). The fish that survive the fishery become the spawners (*S*) that will produce the cohort *c+5*. These dynamics continue for the simulation period.

In the stage-structured life history model, sediment survivals are the percentage of fish at life stages that survive extreme sedimentation events during each season, which is explained in more detail below. The principle is to overlap the freshwater stanza in the life cycle of Chinook and coho with the potential freshwater seasonal survivals that are estimated from the time series data of SSC extracted from Chapter 2. Thus, fish at a particular life stage are those that survived both natural mortality and the mortality caused by freshwater suspended sediments. In this context, natural mortality is associated with all possible causes except fishing and sediment-related factors. In the case of spawner-egg production, density dependence mechanisms also affect the population abundance. The same principle applies to both coho and Chinook, although coho salmon present a more complex freshwater life history: 18 months, as compared to 90 days in ocean-type Chinook. The equations that model the dynamics in the life histories of coho and Chinook salmon and shown and explained in Table 3.2 and Table 3.3, respectively.

3.2.3 Spawner-Egg Production Parameter Estimation

There were no spawner-egg production data available for the Chilliwack River. Parameters were estimated using various methods and sources of information. I assumed that density dependence mechanisms operate in the spawner-egg relationship represented by a Ricker spawner-recruit function (Ricker, 1954). For coho salmon, the *a* parameter and the lognormal error term (ε) for the Ricker function of the form $E=aSe^{-bS+\varepsilon}$ were estimated using data from several naturally-spawning coho populations in southwestern British Columbia (Black Creek, Martyle Creek, Inch Creek, and Salmon River). The *b* parameter was estimated from historical abundance in the Chilliwack River based on $b=1/S_{max}$ where S_{max} is the maximum historical number of spawners in the system. I considered only 1986 onwards because of a marked change in abundance of naturally-spawning returns due to several natural and anthropogenic causes (Beamish et al., 2008). The Chinook spawner-egg parameters for the Ricker spawner-recruit function of the form $E=Se^{a(1-S/b)+\varepsilon}$ were estimated by applying the method of Liermann et al. (2010), who use the watershed size as a predictor for the management parameters. Because of the lack of accurate spawner-egg production data in the watershed, parameter values for both coho and Chinook were subject to sensitivity analyses described later in the results section. Table 3.4 and Table 3.5 are a summary of the model parameters values.

3.2.4 Estimating Stage-Specific Sediment Survivals

In the context of the stochastic life history model, an extreme sedimentation event is composed of two elements: concentration and duration, where the concentration is the mean of contiguous values of daily SSC (in mgL^{-1}) that exceeds a threshold of 25 mgL^{-1} in each sedimentation scenario, and the duration ranges from 24 to 2160 hours (1 to 90 days). The threshold of 25 mgL^{-1} was identified from literature review as the minimum value of SSC at which salmonids show some degree of physiological response to sediments. This threshold level is well documented for coho salmon (Berg, 1982; Bisson, and Bilby,. 1982; Servizi and Martens, 1992), Chinook salmon (Liber, 1992), Arctic grayling (Anderson et al., 1996; Berry et al., 2003), and other fish (EIFAC, 1964; Noggle, 1978). It is also the recommended maximum value of suspended solids concentration used in British Columbia, as the permissive limit of SSC over background levels to protect aquatic life (DFO/MELP, 1992; Ministry of Environment of British Columbia, 2009). Even though there is a direct relationship between SSC and intra-gravel sediments, the effect of SSC on buried eggs may vary (Grieg et al., 2005). Here I assume that exposure of SSC has a uniform effect on eggs and that the concentration of fine sediment in the gravel determines egg to fry survivals, as addressed in Newcombe and Jensen (1996).

3.2.5 Stochastic Event Sampling

Four seasonal databases were constructed from the collection of all extreme events present in the time series for each scenario. Subsequently, the average number of events per season was estimated by dividing the number of years in the seasonal databases (42) by the total number of extreme events in the seasonal data base. The average number of events per season also corresponds to the sample size per year to be drawn stochastically from each seasonal database. The number of events drawn stochastically from the data bases was then overlapped with the seasons of freshwater residence during each simulation year. As the sample size of seasonal extreme events contained decimals in most cases, it was later adjusted so that the number of events in a season across years equaled the actual average of events from the database. The time series of extreme events during the simulation period represent the conditions of SSC that a fish population will theoretically encounter during100 years of its population dynamics (after a 20-year burn-in period).

Once I estimated the time series of extreme events during the simulation period, I applied the dose-response model (Newcombe and Jensen, 1996) to each individual event, producing what they call the severity of ill effect. Ill effects are scores from 0 to 14 in a semi-quantitative scale of physiological responses to elevated sediments that are discriminated by life stage, particle size, and species (Figure 3.4). The life stages included eggs, juveniles (fry and smolts), and spawners affected by a particle size (defined in Newcombe and Jensen as 0.5 to 250 μm for adult salmonids and 0.5-75 100μm for juveniles and eggs/larvae), in accordance with the definition of suspended solids (<100μm) from the Ministry of Environment of British Columbia (2009). The equation for the dose-response model by Newcombe and Jensen is given by:

$$Z_{v,\gamma} = B0_{\gamma} + B1_{\gamma}(\ln \tau_{v}) + B2_{\gamma}(\ln \theta_{v}) \tag{1}$$

where $Z_{v,\gamma}$ is the severity of ill effect for each event (v) for fine particle sizes (≤75μm), and life-stage (γ), τ and θ are the concentration and duration respectively, while $B0$, $B1$, and $B2$ are the coefficients extracted directly from Newcombe and Jensen (1996) for the life stage (γ) (Table 3.6). Although Newcombe and Jensen (1996) provide only direct mortality estimates for lethal exposures (e.g. >3000 mgL^{-1} for >1 week, causing a sediment-related mortality rate of 0.2), I applied their model under the premise that sub-lethal exposures can cause indirect mortality. Thus, even if there is not direct lethal mortality associated with direct exposure of SSC at any life stage as defined by Newcombe and Jensen (1996) (Table 3.6), there is indirect mortality caused by sub-lethal effects. This mortality is assumed to be related to delayed growth, stress, decrease feeding rates, and other factors (Reynolds et al., 1989), and it is represented by the tail of the fitted mortality function (Figure 3.5), explained as follows. I translated $Z_{v,\gamma}$ into mortalities by fitting a modified exponential function with parameters μ= 11.95 and β =2.38 to the Newcombe and Jensen (1996) estimates of direct mortality (Eq. 1) using non-linear least squares, and included sub-lethal effects as a minor but essential mortality source, as explained above. The model fitted to the data takes the final form:

$$M_{v,\gamma} = \frac{e^{(Z_{v,\gamma} - \mu)/\beta}}{\beta} \tag{2}$$

where $M_{v,\gamma}$ is the mortality for each event (v), specific to each life stage (γ); μ and β represent the rate and inflexion points in the mortality function, respectively (Figure 3.5). Because of the hypothetical nature of the indirect mortality estimates at lower concentration levels, I performed a sensitivity analysis on the effects of lifestage-specific mortalities to changes in both β and μ (see section Sensitivity Analyses). Values of $M_{v,\gamma}$ were converted into seasonal survivals:

$$\delta_{t,\gamma} = \prod_{v=1}^{n_t} (1 - M_{v,\gamma}) \tag{3}$$

where $\delta_{t,\gamma}$ represents the potential freshwater seasonal survival after mortality caused by extreme sedimentation events in each scenario for each year of the simulation period, whether or not fish are present, and n_t is the number of events per season (t) (see Table 3.2 and Table 3.3). This procedure allows transformation of the values contained in the time series of extreme events during the simulation period into direct estimates of sediment-related survival.

3.2.6 Estimating Freshwater Natural Survival

The natural survival was estimated based on Bradford (1995) and it includes: egg to fry survival; egg to smolt survival; and fry to smolt survival. A drawback related to using natural survivals from the literature (which includes factors such as predation, starvation, habitat degradation, pollution, and other causes) is that, theoretically, it also includes natural sediment-related mortality. To separate the natural mortality due to factors other than natural sediments, it is necessary to obtain prior estimates of the stage-specific mortality under the "regular" conditions in the watershed, especially when increases in SSC considerably exceed background levels. In the case study, I divided the literature estimates of natural survival based on Bradford (1995) by the mean of sediment-related survivals extracted from the simulation in the scenario that represents the natural sedimentation only (see Table 3.1) for the life stages under consideration. For example, to estimate egg-to-smolt survival for all factors except sedimentation:

$$es = \frac{esl_{\gamma}}{\overline{\delta_{\gamma=E,k=2}}\;\overline{\delta_{\gamma=F,k=2}}} \tag{4}$$

where *esl* is the egg-to-smolt survival from the literature, and $\delta_{\gamma=E,k=2}$ and $\delta_{\gamma=F,k=2}$ are the mean sediment-related survivals estimated for the eggs to fry and fry to smolt stages, respectively, in the scenario that represents the natural sedimentation in the watershed (k=2). Although this approach is intended to reflect more accurate dynamics by not accounting twice for sediment-related mortality, its overall impact on the population may be negligible, depending on the natural SSC levels. In the case in which natural SSC is usually low (<25 mg L^{-1}), values of freshwater survival may be used without adjusting for sediment-related factors.

3.2.7 Monte Carlo Simulations

Both coho and Chinook populations are characterized by variability in the abundance of their annual returns and for being affected by highly stochastic natural processes in their survival (Healey, 1991; Bradford et al. 2000). I used a Monte Carlo simulation framework (Figure 3.6) to evaluate the impact of the uncertainty contained in selected parameters (Table 3.4 and Table 3.5) on the model output. In each scenario k containing 100 years of population dynamics, I ran 1000 Monte Carlo trials (the number necessary to obtain a constant mean number of spawners over the simulation period, as observed by running the simulations under different numbers of trials). In each trial, I estimated the median values of catch and spawners, in addition to obtaining several statistics for both catch and spawners, such as the mean, 0.05 and 0.95 quantiles, and the coefficient of variation (CV). For both Chinook and coho, the stochastic variables incorporated in the model included freshwater and marine survival, log-normal stock-recruitment error, the proportion of jacks that spawn every year, and harvest rates (see Table 3.4 and Table 3.5).

3.3 Results

3.3.1 Simulating Catch and Spawners for 100 years

The distribution of the populations' abundance after the Monte Carlo simulations shows mild skewness to the right for both coho and Chinook, likely a reflection of the populations' log-normal response to the log-normal error in the Ricker spawner-recruit function (Figure 3.7). The 100-year population dynamics simulation with 1000 Monte Carlo trials indicates that the trend in the population abundance can vary over time. Thus, it is possible to observe a range of population trajectories contained within the 0.05 and 0.95 quantiles due to stochasticity in selected parameters or variables (see Table 3.4 and Table 3.5). For example Figure 3.7 (*a* and *b*) show the median number of catches and their 0.05 and 0.95 quantiles for the 100-year time series of dynamics using the baseline scenario (k=0). With increasing numbers of simulations, the populations' median catch tends towards stability, and the 0.05 and 0.95 quantiles reflect the stochastic log-normal error of the Ricker spawner-egg function, normal marine mortality function, and jacking proportions.

3.3.2 Number of Events >25mg L^{-1} vs. Number of Spawners

The results demonstrate that the populations of both Chinook and coho experienced an exponential decline for a linear increase in the number of extreme sedimentation events (SSC > 25 mg/L) the fish withstood during the simulation period (Figure 3.8, Figure 3.9). This decline is a likely product of the density-dependent production. Although both populations of Chinook and coho are somewhat resilient to the exposure of SSC, continuous exposure to very large extreme sedimentation events would eventually cause the population's extinction. For example, a linear increase in the baseline time series of SSC (e.g. applying a multiplier) causes a negative exponential decline of the Chinook population until it collapses. These collapses happen when SSC reach about > 50 times the baseline SSC for Chinook and > 200 times the baseline SSC for coho (Figure 3.8 and Figure 3.9). When both populations had the same number of initial spawners at the beginning of the simulation period (e.g. 1000 spawners), the coho population remained more resistant than the Chinook population, despite its longer fresh water residence.

3.3.3 Comparison among Road Management Scenarios

In the case study, the more intense the road use levels, the higher the frequency and magnitude of extreme sedimentation events that are expected in the watershed, which result in decreased numbers of spawners and harvested fish during the simulation period (Figure 3.10 and Figure 3.11). An increased number of extreme events also led to higher variation in yearly returns (see CV in Table 3.7 and Table 3.8). The populations' decline is greatest for the scenarios of heavy road use ($k=5$), followed by the scenarios of moderate road use ($k= 4$). In regards to salmon production, the level of road use seems to be more significant than the density of roads alone. Light road use ($k= 3$) exhibits very little difference in spawning numbers compared to scenario k=1, where roads are, in theory, removed from the watershed. Nevertheless, this finding does not suggests that road density is an unimportant factor when assessing impact on aquatic species- because higher densities facilitate easier access and therefore more intense use. In addition, this study is only concerned with sedimentation effects, but many other ecological interactions occur between forest roads and aquatic species (Gucinski et al., 2001). The baseline scenario ($k=0$) shows results similar to scenarios of moderate use (Table 3.7 and Table 3.8).

3.3.4 Sensitivity Analyses

To account for uncertainty in the parameter estimates and model assumptions, I conducted sensitivity analyses of the model results to the threshold level at which fish first respond to SSC, the parameters of the fitted mortality function (Eq. 2), and the

Ricker spawner-egg production and demographic parameters. The sensitivity analysis of the Ricker spawner-recruit function parameters demonstrates that, with the exception of extreme values, these have very little effect on the model output. When the threshold of SSC at which salmonids show some degree of behavioural response is altered (i.e., values in the range between 10 to 45 mg L^{-1}), the number of spawners per scenario does too, although this response is more noticeable in coho salmon than in Chinook salmon. The reason is because coho salmon are exposed to many more extreme sedimentation events than Chinook salmon given their longer freshwater residence. Lower sediment threshold mean that salmonids would be more sensitive to *SSC*. Therefore, decreasing the thresholds (e.g., 10 mg L^{-1}) tends to produce stronger population declines, while the opposite occurs for higher threshold levels. Changes in the parameters of the sediment mortality function (Eq. 2) affect Chinook and coho by modifying the sediment-related mortality for given levels of exposures to SSC. The sensitivity of the results is stronger to parameter β, which modifies the inflexion point of the mortality function at ill-effects conducive to lethal exposures (see Figure 3.5). For details of the sensitivity analyses, see Appendix 1.

3.4 Discussion

The model presented here incorporates existing knowledge of the physiological responses of salmonids at different life stages into a stochastic life history model. The methodology provides a quantitative framework for estimating the effect of extreme sediment events in natural populations of Chinook and coho salmon in a coastal watershed of British Columbia. While the model explicitly incorporates suspended sediments as a direct source of mortality on natural populations, it also implicitly includes other natural mortality sources caused by human and environmental factors taken into account in the estimates of Bradford (1995). The model estimated returns for either Chinook and coho may differ considerably from current population assessments in the watershed of study because the model does not include all natural and anthropogenic aspects that influence the life cycle of Chinook and coho salmon- such as ocean conditions and climatic regimes, extreme natural freshwater and marine mortality, and hatchery production. Thus, the model should be understood as a tool to estimate the direct effects of suspended sediments on salmonid populations in isolation from other factors.

The suspended sediment data used to run the model were associated with different forest road management scenarios. Each scenario represents a sediment regime, which is used to estimate the population responses to SSC using a life history model of Chinook and coho salmon. Thus, the model results can be used to infer the impacts of the forest road scenarios contained in Chapter 2 on Chinook and coho production in the Chilliwack River, at least from the point of view of sedimentation generated by roads. When making the comparison among different scenarios, we must keep in mind that other important sources of non-sediment-related mortality caused specifically by forest roads are not included in the analysis, nor are some other factors that affect the relationship between fish and forestry, such as woody debris, nutrient exchange, habitat fragmentation, temperature regulation, etc. Therefore, the model likely underestimates the fish populations' response to forest road management scenarios. Under this premise, measures aimed at protecting aquatic species from the sediment produced by road networks associated with forestry activities must take into account restrictions on the use of forest roads rather than road deactivation alone.

Salmonid populations are characterized by having large variability in their annual returns (NRC, 1996). Variability in the numbers, age, and sex composition of a population is caused by stochastic physiological variables that respond to environmental conditions including survival, movement, maturation, and successful reproduction (Caswell, 2001). The uncertainty in the model results is derived from the stochasticity present in the system, which is reflected in the model parameters and variables, particularly freshwater and marine survival, jacking proportions, and harvesting. By employing techniques such as Monte Carlo simulations, fisheries scientists can achieve better natural system representations that include sources of uncertainty- such as biological responses to environmental conditions (Dorner et. al, 2009). An advantage of stochastic models is that they can account for uncertainty that, if ignored, can lead to inaccurate or overly optimistic assessments (Newmann and Lindey, 2006). The utility of stochastic stage-specific life history models that include the explicit treatment of uncertainty is increasingly being accepted in fisheries science and management (Schnute and Kronlund, 2002; Newmann and Lindey, 2006). Another way to account for uncertainty in the parameter estimates is to perform sensitivity analyses on selected parameters, which I have also done in the present study.

The results indicate that an increase in exposure to extreme sedimentation ends up with a decline in the population estimates, in agreement with previous findings (e.g. Cederholm and Reid, 1987; Birtwell and Korstrom, 2002). The non-linear decline in both coho and Chinook populations is likely due to the nature of the spawner-egg relationship. In reality, cohorts that experience strong mortality tend to compensate by producing larger offspring at lower spawning numbers (Hilborn and Walters, 1992). For example, a linear increase in the baseline time series of SSC (applying a hypothetical concentration multiplier) causes a non-linear trend in the decline until the population collapses. These collapses happen when SSC reach > 50 times the baseline SSC for Chinook and > 200 times for coho (Figure 3.8 and Figure 3.9). The coho salmon population seems to be much more resistant to increased SSC despite its much longer fresh water residence period because of its high relative abundance under baseline conditions compared to Chinook salmon and because of its higher marine survival.

Several studies associate elevated pulses of suspended sediments with salmonid population declines. For example, Tripp and Poulin (1986) suggest that high egg mortality in Haida Gwaii, BC, were attributed to debris torrents and associated stream channel scouring. In a different study, the numbers of chum salmon spawners have declined progressively since about 1980 in Carnation Creek, British Columbia. This decline is partially caused by sediment impacts associated with forestry activities in addition to other factors (Hartman and Scrivener, 1990; Tschaplinski, 2000). Grieg et al. (2005) used a set of field and laboratory experiments to investigate the relationships between fine sediment and embryonic survival. The authors found potentially harmful factors related to the presence of SSC that influence the availability of O_2- which negatively affects the survival embryos.

Not all disturbances cause negative impacts on salmonid population numbers (Gregory et. al., 1987; Schlosser, 1991; Naiman et al., 1992). Non-lethal exposure to suspended sediments may provide positive trade-offs. Moderate levels of fine sediments may benefit salmonids by contributing to increased invertebrate productivity (Everest et al., 1987), and naturally elevated turbidity can protect age-zero juvenile salmonids from excessive predation (Newcombe, 2003). However, even though elevated SSC may protect juveniles from predation, it decreases their prey capture rates substantially (Gregory, 1991). In addition, not all populations may be susceptible to SSC to the same degree, depending on their population-specific adaptations. Nonetheless, the transport and deposition of fine sediments are more frequently associated with deleterious effects on the survival of aquatic organisms than with ecological benefits.

The approach presented here is useful in medium-sized watersheds with continuous SSC and discharge data, and helps to establish a quantitative link between land-use changes in SSC and the biological consequences for salmonids. Because the model assumes that hydrologic patterns in the watershed remain stable over time, it is most applicable to regions where hydrologic conditions show recurring annual and decadal patterns, such as British Columbia and the American Pacific Northwest. In this region, hydrographs exhibit cyclical sediment patterns with relatively short sediment pulses of elevated levels of SSC (days, weeks, months) that cause disturbances on populations (Swanston, 1991). Future work may include coupling population responses to changing SSC triggered by major anthropogenic disturbances such as climate change. Recent studies in the Pacific Northwest forecast fewer mild storms in winter but a greater frequency of very severe storms due to climate change (e.g., IPCC WG I, 2007; Spittlehouse, 2008).

Suspended sediment patterns depend on the interactions among different processes such as flow generation, flow magnitude, and sediment delivery from internal and external sources; variability can occur over a wide range of spatial and temporal scales (Gomi et al., 2005). In regards to the temporal scale, given the availability of data and the relatively slow temporal response of salmonids to SSC, daily intervals of SSC data are an ideal unit of exposure. However, the model assumes no spatial variability in the distribution of

SSC, which may introduce biases, especially in channels with many tributaries and without barriers to fish movement. Scrivener and Tripp (1998) provide two principal types of response to storm conditions observed for juvenile salmonids: (a) selection of particular habitat features that offer cover within the main river channel; and (b) migration to adjacent areas such as tributaries or floodplain and side-channel areas. Migration to adjacent areas to avoid sediment events has been documented for brook trout and brown trout (Cunjak and Power, 1986), and coho salmon (Tschaplinski and Hartman, 1983; Brown and Hartman, 1988; McMahon and Hartman, 1989). These considerations must be taken into account when dealing with populations that have access to tributaries and areas of refuge when exposed to very high levels of SSC. Another limitation of the model is that it may require modification of life histories when assessing different populations. In spite of these model limitations, it is a useful tool for assessing population responses to elevated SSC in the long term; this type of assessment would be very difficult to achieve with field-based methods, given the temporal scales at which the population dynamics happen.

In conclusion, extreme sedimentation events affect populations of Chinook and coho salmon in a negative manner, although both species tend to compensate for increased in-river mortality by producing more recruits-per-spawner. The underlying mechanism of this response is a Ricker density-dependence production function representing the spawner-egg production relationship. Thus, when the populations suffer high (sediment-related) mortality, these also compensate by producing larger numbers of offspring. However, very large sediment-related mortalities allow only the production of a proportion of the recruitment experienced by the previous cohort and eventually the population will collapse under very extreme sedimentation conditions. The coho salmon population seems to withstand increased SSC better than Chinook salmon despite its much longer fresh water residence because of its relatively high abundance and better marine survival. Furthermore, the results reflect the effects of suspended sediments generated by forest roads at a population level, but do not capture all the effects of forestry activities on salmonids such as changes in the input of woody debris and water temperatures. Limitations of the model also include the need for better quality data and its inability to address spatial variability in SSC, a characteristic that, combined with salmonid behavioral adaptations to SSC, may introduce biases in the model estimates. Nonetheless, the model can potentially be used as a research tool for watershed assessment and the impact of SSC on salmonids because it addresses the deleterious effects of concentration and duration of exposure to suspended sediments, which is an ever-present problem in the watersheds of British Columbia.

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3.6 References

Alexander, C.A.D., T.M. Webb and D.R. Marmorek. 1998. The Fish Forestry Interaction Project – Management Model (FFIP-MM): Preliminary Model Description and an Application to Carnation Creek.

Alexander, G.R., Hansen, E.A. 1986. Sand bed load in a brook trout stream. North American Journal of Fisheries Management. **6**(1): 9-23.

Anderson, P.G., Taylor, B.R., and Balch, G.C. 1996. Quantifying the effects of sediment release on fish and their habitats. Can. Manuscr. Rep. Fish. Aquat. Sci. 2346.

Baker, P.F. 2009. Generalizing the multi-stage stock-production paradigm: A flexible architecture for population modeling. In Pacific Salmon and Environmental Life History Models: Advancing science for sustainable salmon in the future. E.E. Knudsen and J.H Michael (eds). American Fisheries Society, Symposium 71, Bethesda, Maryland.

Beamish, R. J., R. M. Sweeting, K. L. Lange, And C. M. Neville. 2008. Changes in the population ecology of hatchery and wild Coho salmon in the Strait of Georgia. Transactions of the American Fisheries Society **137**:503–520.

Berg, L. 1982. The effect of exposure to short-term pulses of suspended sediment on the behaviour of juvenile salmonids. In Proceedings of the Carnation Creek workshop: a ten year review. G.F. Hartman (ed). Dept. Fisheries and Oceans, Pacific Biological Station, Nanaimo, BC.

Berg, L. and T. G. Northcote. 1985. Changes in territorial, gill-flaring, and feeding behaviour in juvenile coho salmon, Oncorhynchus kisutch, following short term pulses of suspended sediment. Canadian J. Fisheries Aquatic Sciences **42**: 1410-1417.

Berry, W., Rubinstein, N., Melzian, B., and Hill, B. 2003. The biological effects of suspended and bedded sediment (SABS) in aquatic systems: a review. United States Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Laboratory. Internal report. 58p.

Birtwell, I.K. and Korstrom, J.S. 2002. Responses of aquatic organisms to sediment: preliminary findings of current research by DFO, Pacific Region, and comments relating to the Yukon Placer Authorization. Internal Fisheries and Oceans report for the review of the Yukon Placer Authorization.

Bisson, P.A., and Bilby, R.E. 1982. Avoidance of suspended sediment by juvenile coho salmon. N. Am. J. Fish. Manage. **4**: 371-374.

Bradford M.J., Myers R.A., and Irvine J.R. 2000. Reference points for coho salmon (Oncorhynchus kisutch) harvest rates and escapement goals based on freshwater production. Can. J. Fish. Aquat. Sci. **57**: 677–686

Bradford, M.J. 1995. Comparative review of Pacific salmon survival rates. Can.J. Fish. Aquat. Sci. **52**:1327-1338.

Brown, T.G., and Hartman G.F. 1988. Contribution of seasonally flooded lands and minor tributaries to the production of coho salmon in Carnation Creek, British Columbia. Transactions of the American Fisheries Society **117**:546–551.

Burgner, R.L. 1980. Some features of ocean migrations and timing of Pacific salmon. In Salmonid ecosystems of the North Pacific Ocean. W.J. McNeil and D.C. Himsworth (eds). Oregon State University Press, Corvallis, Oreg. pp. 153–164.

Caswell, H. 2001. Matrix population models, 2nd edition. Sinauer Associates, Inc., Sunderland, Massachusetts.

Cederholm, C.J., Reid, L.M. 1987. Impact of forest management on coho Salmon (Oncorhynchus kisutch) populations of the Clearwater River, Washington: a project summary. In: Salo, E.O. and Cundy, T.W. (eds). Streamside management forestry and fishery interactions. Contribution No. 57. Seattle, WA. University of Washington, Institute of Forest Resources: pp 373-397.

Chapman, D.W. 1988. Critical review of variables used to define effects of fines in redds of large salmonids. Transactions of the American Fisheries Society. **117**(1): 1-21.

Chapman, D.W. and McLeod, K.P. 1987. Development of criteria for fine sediment in the northern Rockies ecoregion. Seattle, WA: U.S. Environmental Protection Agency.

Cunjak R.A., Power, G. 1987. Cover use by stream-resident trout in winter: A field experiment. North American Journal of Fisheries Management **7**:539–544.

Department of Fisheries and Oceans (DFO). 2010a. NuSEDS V2.0 Regional Adult Salmon Escapement Database. Chilliwack River coho: 1986-2006. Accessed July 1, 2009.

Department of Fisheries and Oceans (DFO). 2010b. NuSEDS V2.0 Regional Adult Salmon Escapement Database. Chilliwack River Chinook: 1986-2006. Accessed July 1, 2009.

Department of Fisheries and Oceans, Habitat Management Division; B.C.Ministry of Environment, Lands and Parks, Integrated Management Branch (DFO/MELP). 1992. Land Development Guidelines for the Protection of Aquatic Habitat. Fisheries and Oceans Canada, Pacific and Yukon Region, Vancouver; BC, and Government of British Columbia, Victoria, BC.

Dorner, B., Peterman, R.M. and Su, Z. 2009. Evaluation of performance of alternative models of Pacific salmon in the presence of climatic change and outcome uncertainty using Monte Carlo simulations. Can. J. Fish. Aquat. Sci. **66**(12):2199-2221.

European Inland Fisheries Advisory Commission (EIFAC). 1964. Water quality criteria for European freshwater fish. Report on finely divided solids and inland fisheries. European Inland Fisheries Advisory Commission. Food and Agriculture Organization of the United Nations. Rome. EIFAC/1.

Everest, F. H., Beschta, R. L. Scrivener, J. C. Koski, K. V., Sedell, J. R., and Cederholm C. J. 1987. Fine sediment and salmonid production: a paradox. In Salo, E.; Cundy, T., (eds). In: Salo, E.O. and Cundy, T.W. (eds). Streamside management forestry and fishery interactions. Contribution No. 57. Seattle, WA. University of Washington, Institute of Forest Resources: pp 98-142.

Galbraith, R., MacIsaac E., Macdonald S., and Farrell A. 2006. The effect of suspended sediment on fertilization success in sockeye (Oncorhynchus nerka) and coho (Oncorhynchus kisutch) salmon. Canadian Journal of Fisheries and Aquatic Sciences, **63**: 2487-2494.

Gardner, J., Peterson, D.L., Wood, A., Maloney, V. 2004. Making sense of the debate about hatchery impacts: interactions between enhanced and wild salmon on Canada’s Pacific coast. Vancouver, BC: Prepared for the Pacific Fisheries Resource Conservation Council.

Godfrey, H., Worlund, D.D., Bilton, H.T. 1968. Tests on the accuracy of ageing Chinook salmon (O. tshawytscha) from their scales. J. Fish. Res. Bd Can, **25**: 1971- 1982.

Gomi, T., Moore, D., and Hassan, M.A. 2005. Suspended sediment dynamics in small forest streams of the Pacific Northwest. Journal of the American Water Resources Association (JAWRA). **41**(4):877-898.

Gregory, R. S. 1991. Foraging behaviour and perceived predation risk of juvenile Chinook salmon, Oncorhynchus tshawytscha, in turbid waters. Ph.D. Thesis, University of British Columbia, Vancouver.

Gregory, R.S. and Northcote T.G. 1993. Surface, planktonic and benthic foraging by juvenile chinook salmon (*Oncorhynchus tshawytscha*) in turbid laboratory conditions. Can. J. Fish. Aquat. Sci. **50**: 233-240

Gregory, S.V., Lamberti, G.A., Erman, D.C., Koski, K.V., Murphy, M.L, and Sedell, J.R. 1987. Influence of forest practices on aquatic production. In: Salo, E.O. and Cundy, T.W. (eds). Streamside management forestry and fishery interactions. Contribution No. 57. Seattle, WA. University of Washington, Institute of Forest Resources: pp. 233–255.

Greig, S.M., Sear, D.A., and Carling P.A. 2005. The impact of fine sediment accumulation on the survival of incubating salmon progeny: implications for sediment management. Sci Total Environ **344**:241–258

Groot, C., and Margolis, L. 1991. Pacific salmon life histories.Vancouver: University of British Columbia Press. 564 p.

Gucinski, H., Furniss, M. J., Ziemer, R.R., Brookes, M. 2001. Forest roads: a synthesis of scientific information. Gen. Tech. Rep. PNWGTR- 509. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 103 p.

Hartman, G.F., and Scrivener, J.C. 1990. Impacts of forestry practices on a coastal stream ecosystem, Carnation Creek, British Columbia. Canadian Bulletin of Fisheries and Aquatic Sciences 223. 148 p.

Harvey, B. C., and S. F. Railsback. 2009. Exploring the persistence of stream-dwelling trout populations under alternative real-world turbidity regimes with an individual-based model. Transactions of the American Fisheries Society **138**: 348-360.

Healey, M.C. 1991. Life history of Chinook salmon (Oncorhynchus tshawytscha). In Pacific salmon life histories. C. Groot and L. Margolis (eds). University of British Columbia Press, Vancouver, B.C. pp.311-394.

Heinzelmann, F. 2002. A Modeling framework to predict relative effects of forest management strategies on coastal stream channel morphology and fish habitat. Department of Forestry Resources Management. Phd. Thesis. University of British Columbia. Vancouver. B.C.

Hilborn R. 2009 Life history models for Salmon management: the challenges. In Pacific salmon and environmental life history models: advancing science for sustainable salmon in the future. E.E. Knudsen and J.H Michael (eds). American Fisheries Society, Symposium 71, Bethesda, Maryland.

Hilborn, R. and Walters C. J. 1992. Quantitative fisheries stock assessment: choice, dynamics and uncertainty. Chapman and Hall, New York. 570 p.

Intergovernmental Panel on Climate Change (IPCC)- Working Group I. 2007. Climate change 2007: The physical science basis: Summary for policymakers. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Geneva, Switzerland. Available at http://www.ipcc.ch. Accessed on Apr 2nd, 2010.

Kerr S.J. 1995. Silt, turbidity and suspended sediments in the aquatic environment: an annotated bibliography and literature review. Ontario Ministry of Natural Resources. Southern Region Science & Technology Transfer Unit Technical Report TR-008. 277 pp.

Liber, K. 1992. Effects of placer mining sediment on juvenile Chinook salmon and the eggs of Arctic grayling. Prepared for DFO, Science Branch. EVS Consultants, North Vancouver, B.C. Appendices +54p.

Liermann M.C., Sharma, R., Parken, C. K. 2010. Using accessible watershed size to predict management parameters for Chinook salmon, Oncorhynchus tshawytscha, populations with little or no spawner-recruit data: a Bayesian hierarchical modelling approach. Fisheries Management and Ecology **17**: 40–51.

MacDonald, L.H., Smart, A.W., Wissmar, R.C. 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. EPA/910/9-91-001. Seattle: U.S. Environmental Protection Agency, Region 10. 166 p.

McMahon, T.E., and Hartman G.F. 1989. Influence of cover complexity and current velocity on winter habitat use by juvenile coho salmon (Oncorhynchus kisutch). Canadian Journal of Fisheries and Aquatic Sciences **46:** 1551-1557.

Meehan, W.R., (ed). 1991. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society. Spec. Publ. 19. Bethesda, MD 751 p.

Megahan, W.F., Day, N.F., and Bliss, T.M. 1978. Landslide occurrence in the western and central northern Rocky Mountain physiographic province in Idaho. In Forest soils and land use: Proceedings of the 5th North American forest soils conference. Youngberg, C.T., (ed). Colorado State University. Fort Collins, CO. pp 116-139.

Ministry of Environment of British Columbia. 2009. Guidelines for measuring Turbidity and suspended sediment concentrations. Available at http://www.env.gov.bc.ca/wat/wq/BCguidelines/samp_strat/sampstrat.html#sediments. Accessed Jan 1st 2009.

Moore, R.D. and Wondzell S.M. 2005. Physical hydrology in the Pacific Northwest and the effects of forest harvesting - a review Journal of the American Water Resources Association **41**:753-784.

Myers, R.A. 2009. On line data and plots Oncorhynchus kisutch (Coho salmon). Available at: http://ram.biology.dal.ca/~myers/popsofCohosalmon.html Accessed Aug 23 2009.

Naiman, R.J., Beechie, T.J., Benda L.E., Bisson, P.A. MacDonald L.H., O'Connor, M.D., Oliver C., Olson, P. and Steel, E.A. 1992. Fundamentals elements of ecologically healthy watersheds in the Pacific Northwest Coastal Ecoregion. pp. 127-188. In Watershed Management: balancing sustainability and environmental change. R.J. Naiman (ed). Sprinerg- Verlag, NY.

National Research Council (NRC). 1996. Upstream: salmon and society in the Pacific Northwest. National Academy Press. Washington, DC.

Neave, F. 1964. Ocean Migrations of Pacific Salmon. J Fish Res Board Ca. **21**:1227-1244

Newcombe C.P., and Jensen J.O. 1996. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management **16**:693-727.

Newcombe, C.P.. 2003. Impact assessment model for clear water fishes exposed to excessively cloudy water. Journal of the American Water Resources Association (JAWRA) **39**(3):529-544.

Newman, K. B. and Lindley, S. T. 2006. Accounting for demographic and environmental stochasticity, observation error and parameter uncertainty in fish population dynamic models. North American J. Fisheries Management **26**:685–701.

Noggle, C.C. 1978. Behavior, physiological and lethal effects of suspended sediment on juvenile salmonids. MSc. thesis, University of Washington, Seattle, WA. 87 p.

Northcote, T.G. and G.F. Hartman (eds). 2004. Fishes and forestry: worldwide watershed interactions and management. Blackwell Science, Oxford, UK.

Pacific Salmon Commission (PSC). 2009a. Southern Coho management Plan. Treaty between the Government of Canada and the Government of the United States of America concerning Pacific Salmon. Available at www.psc.org Accessed December 23, 2009.

Pacific Salmon Commission (PSC). 2009b. Joint Chinook technical committee report 2009 annual report of the exploitation rate analysis and model calibration report TCCHINOOK (09)-3.

Peterman, R.M. Pyper, B.J., Mueter, F.J., Haeseker, S.L., Su, Z., Dorner, B. 2009. Statistical Models of Pacific Salmon that Include Environmental Variables. In Pacific salmon and environmental life history models: advancing science for sustainable salmon in the future. E.E. Knudsen and J.H Michael (eds). American Fisheries Society, Symposium 71, Bethesda, Maryland.

Pyper, B.J., Mueter, F.J. and Peterman R.M. 2005. Across-species comparisons of spatial scales of environmental effects on survival rates of northeast Pacific salmon. Trans. Am. Fish. Soc. **134:** 86–104.

R Development Core Team. 2010. A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0. Available at http://www.R-project.org.

Railsback, S.F., and Harvey, B.C. 2002. Analysis of habitat-selection rules using an individual-based model. Ecology **83**:1817– 1830.

Reice, S. R., Wissmar, R. C. and Naiman R. J. 1990. Disturbance regimes, resilience, and recovery of animal communities and habitats in lotic ecosystems. Environmental Management **14:**647–659.

Reynolds. J. B., Simmons, R. C., and Burkholder A.R. 1989. Effects of placer mining discharge on health and food of Arctic grayling. Water Resources Bulletin **25:**625-635.

Ricker, W. E. 1954. Stock and recruitment. Journal of the Fisheries Research Board of Canada **11**:559-623.

Sandercock, F.K. 1991. Life history of coho salmon (Oncorhynchus kisutch). In Pacific salmon life histories. C. Groot and L. Margolis (eds). University of British Columbia Press, Vancouver, B.C. pp. 395–446.

Scheuerell, M.D., Hilborn, R. Estimating the freshwater fomponent of essential fish habitat for Pacific salmon with the Shiraz Model. In Pacific salmon and environmental life history models: advancing science for sustainable salmon in the future. E.E. Knudsen and J.H Michael (eds). American Fisheries Society, Symposium 71, Bethesda, Maryland.

Schlosser, I.J. 1991. Stream fish ecology: A landscape perspective. BioScience **41**: 704-712.

Schnute, J.T. and Kronlund, A.R. 2002. Estimating salmon stock-recruitment relationships from catch and escapement data. Canadian Journal of Fisheries and Aquatic Sciences **59**: 433-449.

Scrivener, J.C. and Tripp, D.B. 1998. Changes of spawning gravel characteristics after forest harvesting in Queen Charlotte Islands and Carnation Creek watersheds and the apparent impacts on incubating salmonid eggs. In Hogan, D.L., P.J. Tschaplinski, and S. Chatwin (eds). B.C.Min. For., Res. Br., Victoria, B.C. Land Manage. Handb. No. 41.

Servizi, J.A., and Martens, D.W. 1992. Sublethal responses of coho salmon (Oncorhynchus kisutch) to suspended sediments. Canadian Journal of Fisheries and Aquatic Sciences **49**:1389-1395.

Shapovalov, L., and A.C. Taft. 1954. The life histories of the steelhead rainbow trout (Salmogairdneri) and silver salmon (Oncorhynchus kisutch) with special reference to Waddell Creek, California, and recommendations regarding their management. Calif. Department of Fish and Game. Fish Bulletin 98. 373 p.

Sharma, R., Cooper, A.B., and Hilborn, R. 2005. A quantitative framework for the analysis of habitat and hatchery practices on Pacific salmon. Ecological Modeling **183**:231-250.

Sigler, J.W., Bjornn, T.C., and Everest, F.H. 1984. Effect of chronic turbidity on density and growth of steelheads and coho salmon. Trans. Am. Fish. Soc. **113**: 142-150.

Smith, F.J. and Sykora J.L. 1976. Early developmental effects of lime neutralized iron hydroxide suspensions on brook trout and coho salmon. Transactions American Fisheries Society **105**: 308-312.

Spittlehouse, D.L. 2008. Climate Change, impacts, and adaptation scenarios: climate change and forest and range management in British Columbia. B.C. Min. For. Range, Res. Br., Victoria, B.C. Tech. Rep. 045. Available at www.for.gov.bc.ca/hfd/pubs/Docs/Tr/Tr045.htm. Accessed Apr 1, 2010.

Sutherland, G.D., O'Brien, D.T., Fall, S.A., Waterhouse, F.L., Harestad, A.S., & Buchanan, J.B. (eds), 2007. A Framework to Support Landscape Analyses of Habitat Supply and Effects on Populations of Forest-dwelling Species: A Case Study Based on the Northern Spotted Owl. Ministry of Forests and Range Forest Science Program. Technical Report 038.

Suttle, K.B., Power, M.E., Levine, J.M., and McNeely, C. 2004. How fine sediment in riverbeds impairs growth and survival of juvenile salmonids. Ecol. Appl. **14**:969-974.

Swanston, D.N. 1991. Natural processes. In Influences of forest and rangeland management on salmonid fishes and their habitats. W. R. Meehan (ed). American Fisheries Society Special Publication 19.

Tripp, D.B. and Poulin, V.A. 1986. The effects of logging and mass wasting on salmonid spawning habitat in streams on the Queen Charlotte Islands. Land Management Report No. 50, Victoria, British Columbia.

Trombulak, S.C. and Frissell, C.A. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology **14**(1): 18-30.

Tschaplinski, P. J. 2000. The effects of forest harvesting, fishing, climate variation, and ocean conditions on salmonid populations of Carnation Creek, Vancouver Island, British Columbia. In Sustainable fisheries management: Pacific salmon. E. E. Knudsen, C. R. Steward, D. D. MacDonald, J. E. Williams, and D. W. Reiser (eds). CRC Press, Boca Raton, FL and New York, NY. 724 p.

Tschaplinski, P. J., and Hartman, G.F. 1983. Winter distribution of juvenile coho salmon (Oncorhynchus kisutch) before and after logging in Carnation Creek, British Columbia, and some implications for overwinter survival. Canadian Journal of Fisheries and Aquatic Sciences **40**:452–461.

Whitman. R.P., Quinn, T.P. and Brannon, E.L. 1982. Influence of suspended volcanic ash on homing behavior of adult Chinook salmon. Transactions of the American Fisheries Society **111**:63-69.

3.7 Figures and Tables

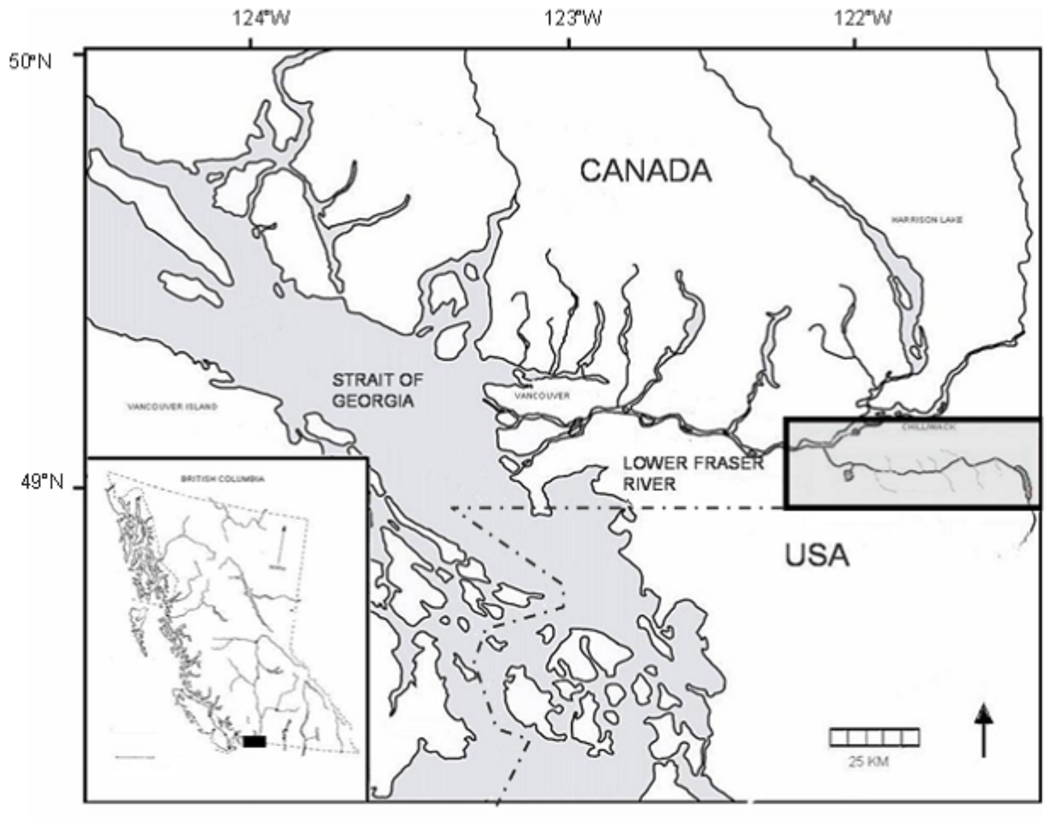


Figure 3.1. The Chilliwack River, a tributary for the Lower Fraser River

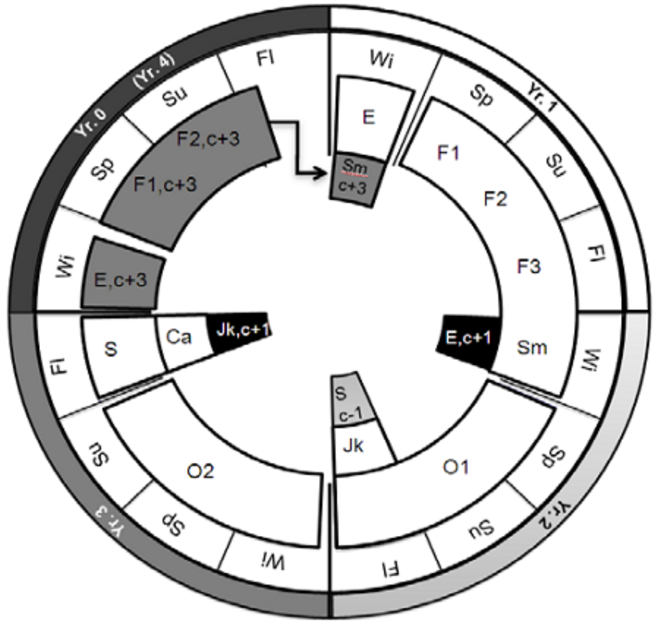


Figure 3.2. **Clockwise figure showing the 3-year life cycle of coho salmon (cohort c) in stanza form. Year is represented by *Yr.* and season subscripts are presented as follows: winter (*Wi*), spring (*Sp*), summer (*Su*), and fall (*Fl*).**

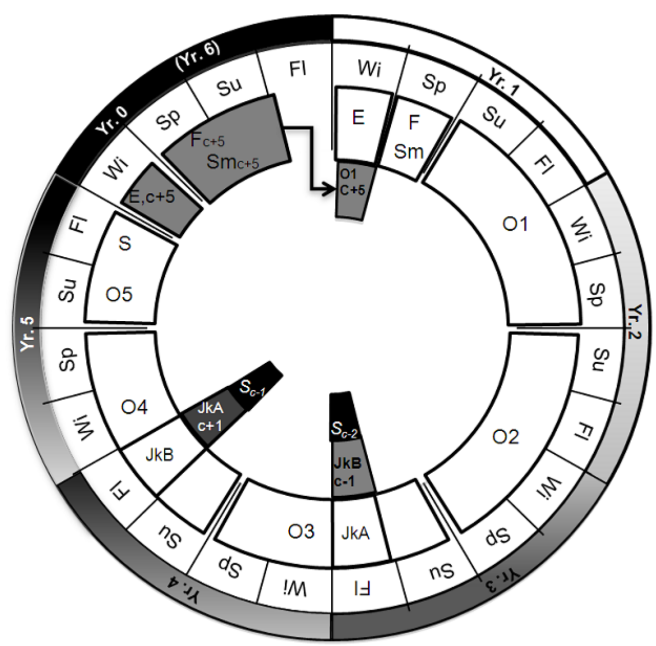


Figure 3.3. **Clockwise figure showing the 5-year life cycle of the ocean-type fall Chinook salmon (cohort c) in stanza form. Year and season subscripts are described in Figure 3.2.**

Log$_e$ mg SS/L											
12	9	10	11	11	12	13	14	14	-	-	-
11	9	9	10	11	11	12	13	14	14	-	-
10	8	9	9	10	11	11	12	13	13	14	-
9	7	8	9	9	10	11	11	12	13	13	14
8	6	7	8	9	9	10	11	11	12	13	13
7	6	6	7	8	9	9	10	11	11	12	13
6	5	6	6	7	8	9	9	10	11	11	12
5	4	5	6	6	7	8	9	9	10	11	11
4	4	4	5	6	6	7	8	8	9	10	11
3	3	4	4	5	6	6	7	8	8	9	10
2	1	3	4	4	5	6	6	7	8	8	9
1	1	2	3	4	4	5	6	6	7	8	8
0	1	1	2	3	4	4	5	6	6	7	8
	0	1	2	3	4	5	6	7	8	9	10
	Duration of Exposure (log$_e$ hours)										

0	Behavioural	No Behavioral Effects
1		Alarm Reaction
2		Abandonment of Cover
3		Avoidance Response
4	Sublethal	Short-term Reduction on Feeding sucess
5		Increased rates of coughing and respiration
6		Moderate physiological stress
7		Moderate habitat degradation, impared homing
8		Major physiol. stress, Long-term reduction of feeding success
9	Lethal and Paralethal	Reduced growth rate, delayed hatching, reduced density
10		0 to 20% mortality
11		>20 to 40% mortality
12		>40 to 60% mortality
13		>60 to 80% mortality
14		>80 to 100% mortality

Figure 3.4. Numbers represent the ill effects (*Z*), a semi-quantitative scale of behavioural, sublethal, and lethal responses to concentration and duration of exposure of suspended sediments for juvenile salmonids in log$_e$ scale. Adapted from Newcombe and Jensen (1996).

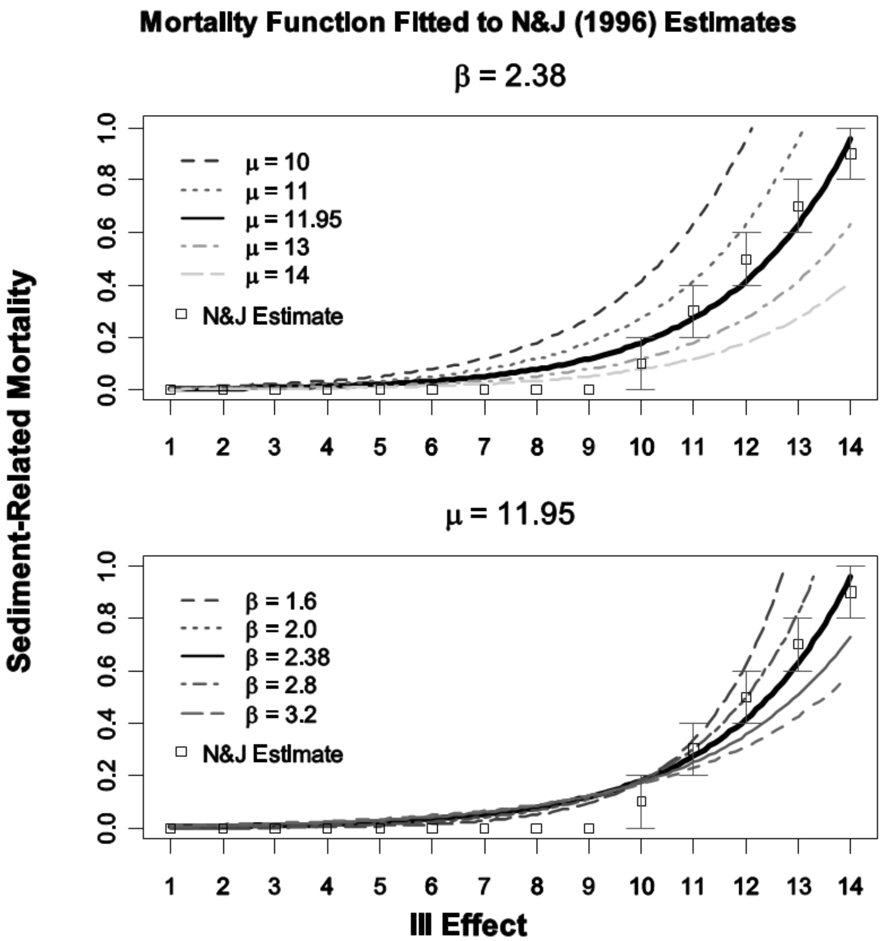


Figure 3.5. **Mortality function (Eq.2) under different μ and β parameter values overlapped to the estimates of Newcombe and Jensen (N&J) (1996) for salmonid juvenile mortality (grey shaded bars). Black thick line (μ= 11.95 β =2.38) represents the best fit to the mortality data provided by Newcombe and Jensen (1996). Additional mortality functions are displayed for different combinations of μ and β used during the sensitivity analyses.**

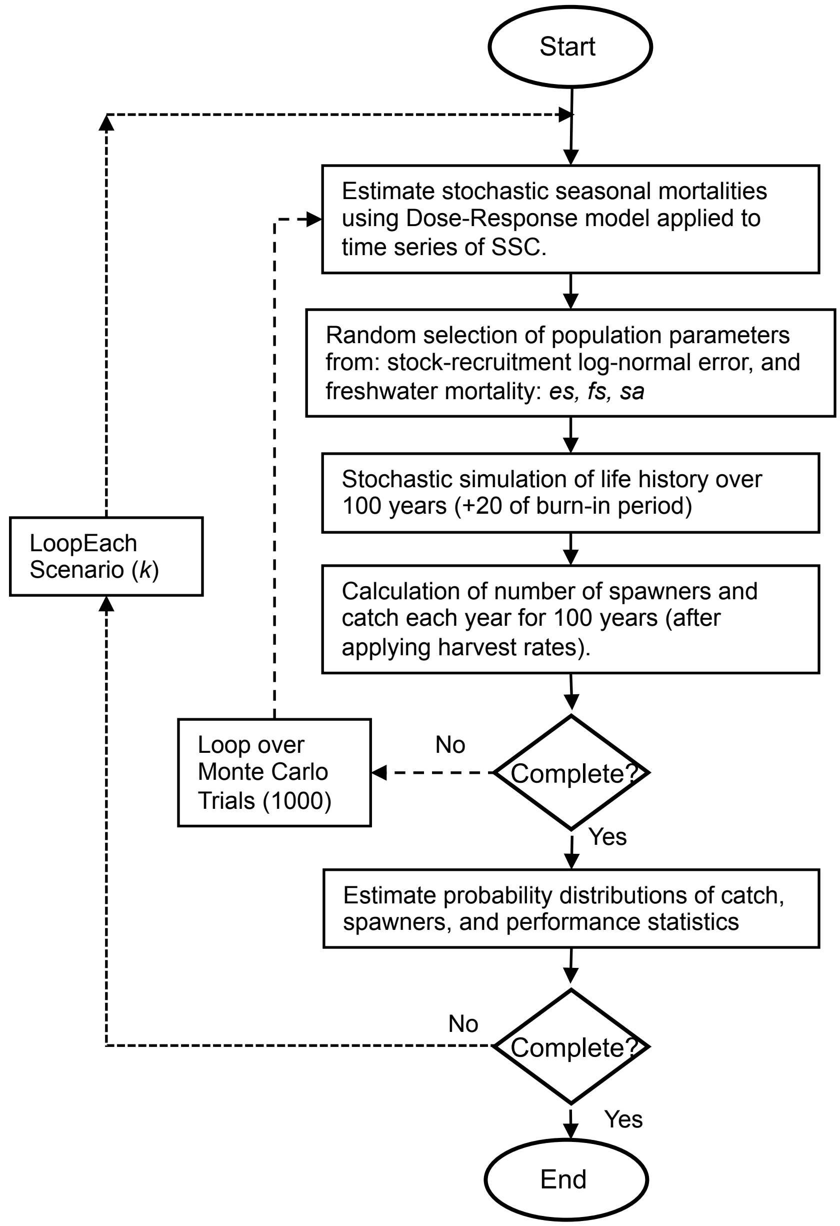
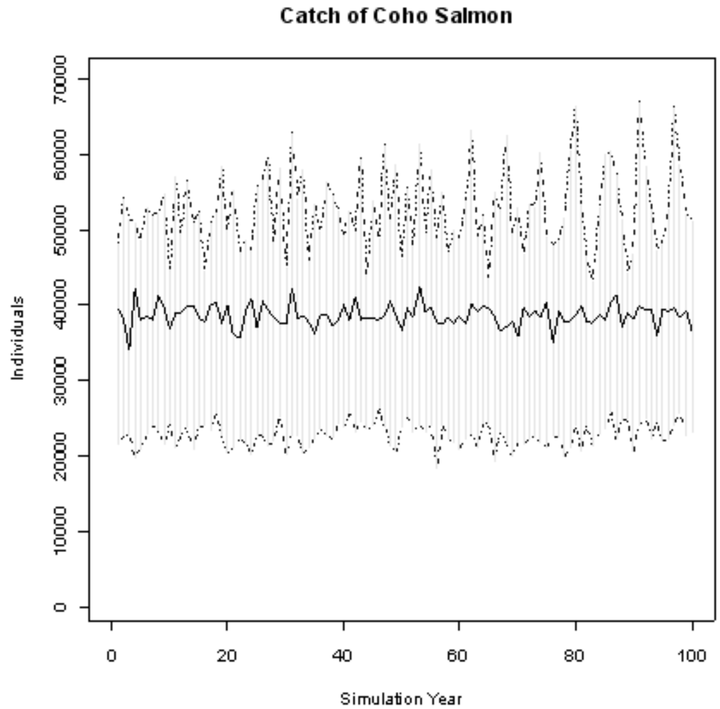


Figure 3.6. Schematic of Monte Carlo Simulations.



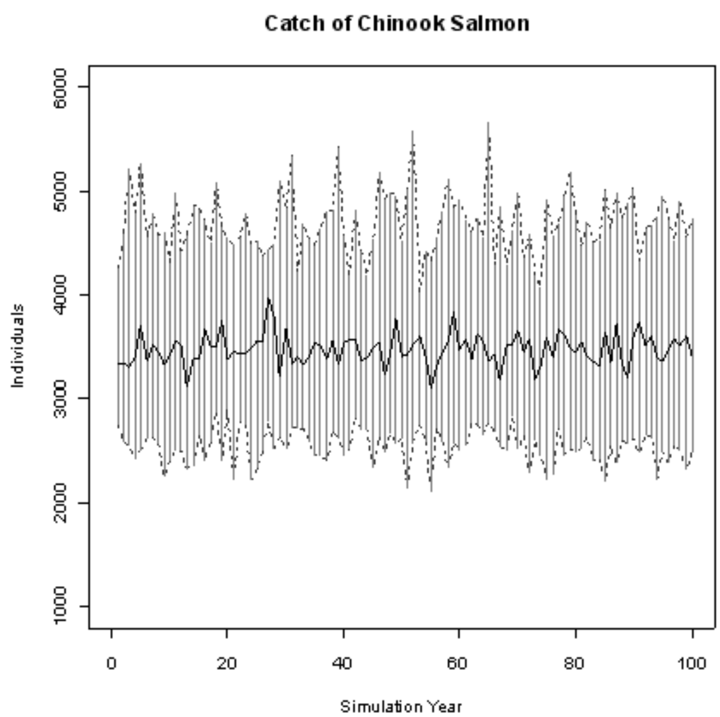


Figure 3.7. a. coho, b. Chinook. Example of the variability present in the number of fishery catches (median, 0.05 and 0.95 quantiles) each year for a hundred years in the baseline scenario (*k=0*) after including stochasticity present in selected parameters and variables.

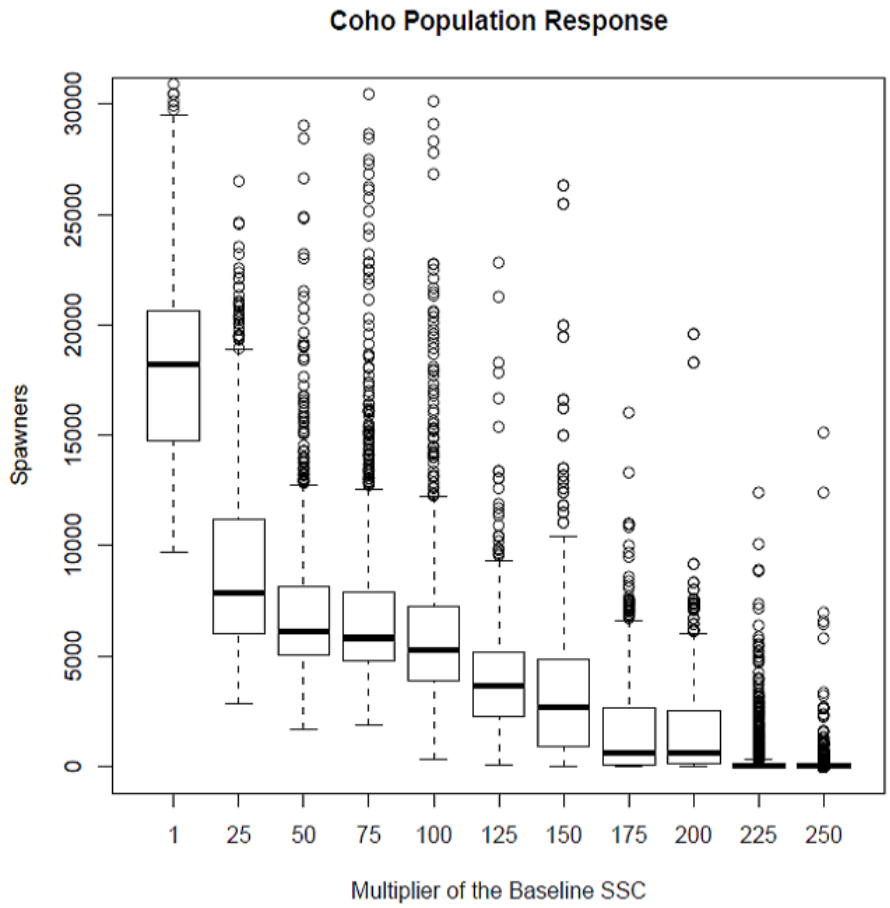


Figure 3.8 Coho population (spawners) response to increasing suspended sediment concentration events achieved by using a hypothetical concentration multiplier for the baseline time series of SSC.

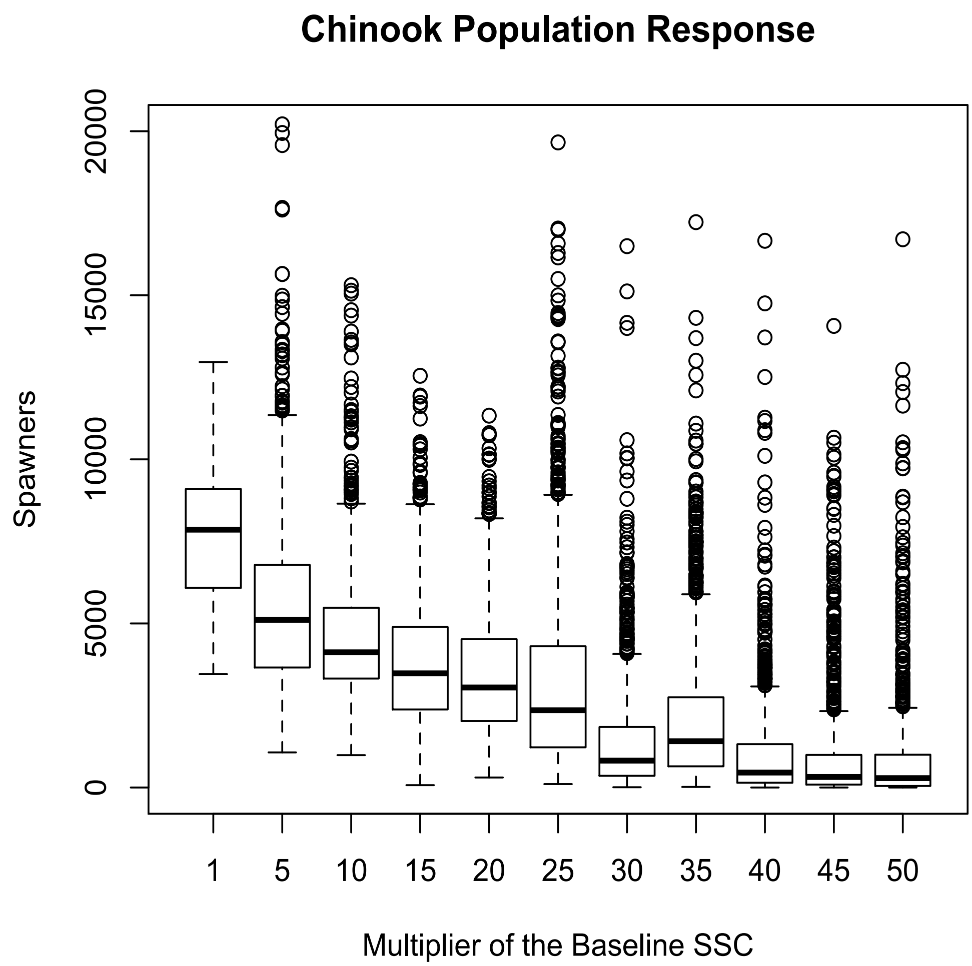


Figure 3.9. Chinook population (spawners) response to increasing suspended sediment concentration events achieved by using a hypothetical concentration multiplier for the baseline time series of SSC.

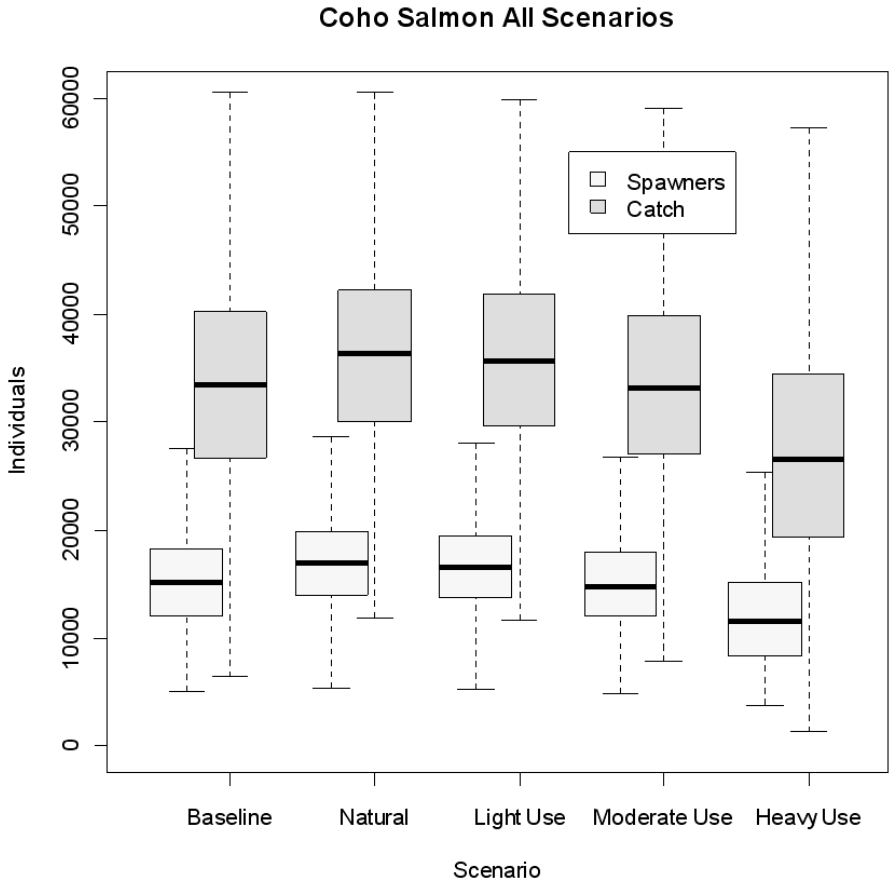


Figure 3.10. Boxplot of spawners and catch for all scenarios of forest management indicating the population abundance for coho salmon. Increased sedimentation caused by moderate and heavy use of forest roads causes marked decreases in catch and spawner numbers.

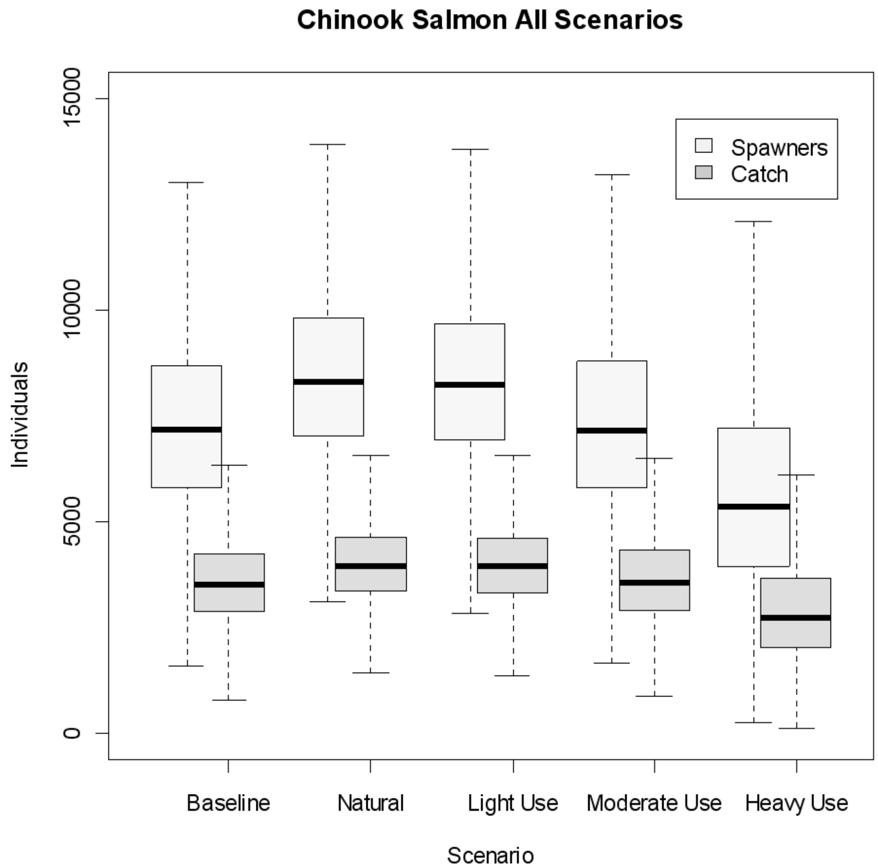


Figure 3.11. Boxplot of spawners and catch for all scenarios of forest management indicating the population abundance for Chinook salmon. Increased sedimentation caused by moderate and heavy use of forest roads causes evident decreases in catch and spawner numbers.

Table 3.1. Forest Road management Scenarios adapted from Chapter 2.

Forest Management Scenario	Description	Use level	Annual Fine Sediment Yield (tonnes year $^{-1}$)	Length at Equilibrium (Km)
Baseline	The current (as of 2006) total length of the road network in the Chilliwack River watershed (permanent and temporary roads).	Moderate	7274	541.2
No-Roads (Natural)	Assumes that all roads in the watershed have been removed. However other sources of sediments still contribute to the potential changes in SSC.	NA	NA	NA
Current Plan	Based on the current timber harvesting and road construction plan for the Chilliwack Forest District for 100 years. Implying that a minimum of 67% of Old Growth stands are reserved within long-term activity centres.	Light	806	662.7
		Moderate	8906	
		Heavy	106025	

This table indicates five different scenarios of forest road construction and use given by the use level based on the Spotted Owl management plan for the Chilliwack Forest district (Sutherland et al. 2007). Each scenario is associated with a time series of daily SSC, which is used as data for estimating the effects of extreme sedimentation events at the population level for Chinook and coho salmon. The road use level is defined in Reid (1981): Light, use only by light vehicles; Moderate, use by fewer than four logging trucks per day; and Heavy, use by more than four logging trucks per day. The AFSY (tonnes year $^{-1}$) represents the average amount of fine sediments generated by forest roads that eventually end up in the rivers and streams of the Chilliwack River watershed each year. The length at equilibrium represents the length of the road network associated with each scenario.

Table 3.2. Population Dynamics in the Life History Model for Coho Salmon

Year (*Yr.*)	Season	Season Index (*t*)	Last day of life stage (counted)	Equation for Life Stage(γ)
0	Fall	*0*	Dec 21	S_{c-3} *(initial spawner numbers)*
1	Winter	*1*	March 21	$E = ef.\delta_{t=1,\gamma=E}\ a.S_{c-3}\ e^{(-b.Sc-3)}e^{\varepsilon}$
	Spring	*2*	June 21	$F1 = E.\delta_{t=2,\gamma=J}$
	Summer	*3*	Sept 21	$F2 = F1.\delta_{t=3,\gamma=J}$
	Fall	*4*	Dec 21	$F3 = F2.\delta_{t=4,\gamma=J}$
2	Winter	*5*	March 21	$Sm = fs.F3\ .\delta_{t=5,\gamma=J}$
	Spring	*6*	June 21	$O1 = Sm*\sqrt{sa}$
	Summer	*7*	Sept 21	$O1$
	Fall	*8*	Dec 21	$Jk = O1*j$ $Rt_{c-1} = Jk + O2_{c-1}$ $Ca_{c-1} = Rt_{c-1}*HR.f(Rt_{c-1})$
3	Winter	*9*	March 21	$O1$
	Spring	*10*	June 21	$O2 = O1(1-j)*\sqrt{sa}$
	Summer	*11*	Sept 21	$O2$
	Fall	*12*	Dec 21	$Rt = O2 + Jk_{c+1}$ $Ca = Rt*HR.f(Rt)$ $S = \delta_{t=12,\gamma=S}(Rt-Ca)$
4	Winter	*13*	March 21	$E_{c+3} = ef.\delta_{t=13,\gamma=E}\ a.S\ e^{(-b.S)}e^{\varepsilon}$

This table summarizes the 3-year life cycle of coho salmon (*cohort c*) in which changes in life stages coincide with the changes in the seasons (*t*). The numbers of fish that pass from one life stage to the next are the ones that survive the natural mortality *(ef, fs, sa)* and the sediment-related mortality*(δ)*- only during the fresh water portion. Numbers of fish are counted at the end of each season (*counted*). Yearly estimates of marine survival are determined by the squared-root of the total marine survival (*sa*) given that adult coho spends two years at sea. Variables and indexes in the equations stand for:

(c) Cohort
(S) Number of spawners of cohort *c.*
($S_{c-1,2,3}$) Initial spawners, obtained from the mean historical abundance in the Chlliwack River
(E) Number of Eggs of cohort *c*.
(ef) Egg to fry survival (independent of sediments), based on from Bradford (1995)
(fs) Fry to smolt survival based on from Bradford (1995).
(sa) Smolt to adult survival, based on Bradford (1995).
(δ) Sediment-related survival specific for a season (*t*) and life stage (γ): Eggs (*E*), Juveniles(*J*), and Spawners (*S*).
(a) Ricker parameter *a*
(b) Ricker parameter *b*
(F) Number of fry at each stage
(Sm) Number of smolts
(O) Number of ocean fish of cohort c
(Jk) Number of jacks of cohort *c*
(j) Proportion of jacks of cohort *c*
(Rt) number of adult fish of cohort *c* and jacks of cohort c+1 that return before the fishery takes place
(Ca) number of fish of cohort c plus $jacks_{c-1}$ that are caught in the fishery
HR.f(Rt) Harvest rate as a function of returns. Follows harvest rule described in the Southern Coho Management Plan (2009).
(ε) Lognormal error in the stock-recruitment function estimated from local populations (Myers, 2009)

Table 3.3. Population Dynamics in the Life History Model for Chinook Salmon

Year (*Yr.)*	Season	Season Index *(t)*	Last day of life instance (counted)	Equation for Life Stage *(γ)*
0	Fall	*0*	Dec 21	S_{c-5} *(initial spawner numbers)*
1	Winter	*1*	March 21	$E = es.\delta_{t=1,\gamma=E}\ S_{c-5}\ .e^{a(1-Sc-5\ /b)}e^{\varepsilon}$
	Spring	*2*	June 21	$F = E.\delta_{t=2,\gamma=J}$
	Summer	*3*	Sept 21	$Sm = F.\delta_{t=3,\gamma=J}$ $O1 = Sm$
	Fall	*4*	Dec 21	$O1$
2	Winter	*5*	March 21	$O1$
	Spring	*6*	June 21	$O1$
	Summer	*7*	Sept 21	$O2 = O1\ \sqrt[4]{sa}$
	Fall	*8*	Dec 21	$O2$
3	Winter	*9*	March 21	$O2$
	Spring	*10*	June 21	$O2$
	Summer	*11*	Sept 21	$JkA = O2*jA$ $O3 = O2(1-jA)\ \sqrt[4]{sa}$
		12	Dec 21	$O3$
	Fall			$Rt_{c-2} = JkA + JkB_{c-1} + O5_{c-2}$ $Ca_{c-2} = Rt_{c-2}*HR.f(Rt_{c-2})$
4	Winter	*13*	March 21	$O3$
	Spring	*14*	June 21	$O3$
	Summer	*15*	Sept 21	$JkB = O3*jpB$ $O4 = O2(1-jB)\ \sqrt[4]{sa}$
	Fall	*16*	Dec 21	$O4$ $Rt_{c-1} = JkA_{c+1} + JkB + O5_{c-1}$ $Ca_{c-1} = Rt_{c-1}.HR.f(Rt_{c-1})$
5	Winter	*17*	March 21	$O4$
	Spring	*18*	June 21	$O4$
	Summer	*19*	Sept 21	$O5 = O4\sqrt[4]{sa}$
	Fall	*20*	Dec 21	$Rt = O5 + JkA_{c+2} + JkB_{c+1}$ $Ca = Rt.HR$ $\delta_{t=20,\gamma=S}\ (Rt-Ca)$
6	Winter	*21*	March 21	$E_{c+5} = es.\delta_{t=21,\gamma=E}\ S\ e^{a(1-S/b)}e^{\varepsilon}$

This table summarizes the 5-year life cycle of Chinook salmon (*cohort c*) and should be interpreted as table 3.2. The only exception is that the yearly estimates of marine survival are determined by the fourth -root of the total marine survival (*sa*), given that adult Chinook spends four years at sea. Variables and indexes in the equations stand for:

(c) Cohort
(S) Number of spawners of cohort *c.*
($S_{c-5,4,3,2,1}$) Initial spawners, obtained from the mean historical abundance in the Chilliwack River
(E) Number of Eggs of cohort *c*.
(es) Egg to smolt survival (independent of sediments), based on Bradford (1995).
(sa) Smolt to adult survival, based on from Bradford (1995).
(δ) Sediment-related survival specific for a season (*t*) and life stage (*γ*): Eggs (*E*), Juveniles(*J*), and Spawners (*S*).
(a) Ricker parameter *a*
(b) Ricker parameter *b*
(F) Number of fry at each stage
(Sm) Number of smolts
(O) Number of ocean fish per in year
(JkA) Number of jacks in the first run
(JkB) Number of jacks in the second run
(jA) Proportion of the first run of jacks
(jB) Proportion of the second run of jacks
(Rt) number of adult fish of cohort *c* and jacks of cohort c+1 that return before the fishery takes place
(Ca) number of fish of cohort c+ $jacks_{c-1}$ that are caught in the fishery
(HR) Harvest rate
(ε) Lognormal error in the stock-recruitment function estimated using the method from Liermann et. Al. (2010).

Table 3.4. Coho salmon parameters (1986 onwards)

Parameter	Meaning	Estimate	SD	Source/ Estimation Method	Included in Monte Carlo method ?	Sensitivity Analysis?
$S_{c-1, c-2, c-3}$	Initial spawners	5650	2888	NuSEDS V2.0 Regional Adult Salmon Escapement Database. Mean Spawner abundance.	No	No
Ricker a	Recruits per spawner at low stock sizes	2967.34	NA	SSQ fit to four local populations	No	Yes
Ricker b	Rate of decline of recruits-per-spawner with increasing Spawner Numbers	0.000078	NA	NuSEDS V2.0 Regional Adult Salmon Escapement Database. 1/Smax (Ricker,1965)	No	Yes
ε	Log normal-error *LN(0,SD)*	0	0.03719	SSQ fit to four local populations (Myers, 2009)	Yes	No
ef	Egg to fry survival	$e^{-1.62}$	0.08*	Based on Bradford, 1995	Yes	No
es	Egg to smolt survival	$e^{-4.20}$	0.09*	Based on Bradford, 1995	Yes	No
fs	*Fry to* smolt survival	es/ef	--	Based on Bradford, 1995	Yes	No
sa	Smolt to adult survival	$e^{-2.32}$	0.06*	Based on Bradford, 1995	Yes	No
δ	Sediment-related survival	Varies	Varies	specific for a season (*t*) and life stage (γ)	Yes	Yes
j	Proportion of jacks	0.015	0.0084	Chilliwack Hatchery historical data (pers. Com Ron Valer)	Yes	No
HR f(Rt)	Harvest rate as function of Returns in simulation year	Varies according to harvest rule: Low= 0 to 0.2 Med=0.21 to 0.4 High=0.41 to :0.65	NA	Pacific Salmon Commission (PSC) 2009a. Southern coho Management Plan.	Yes	No

*Corrected for the standard error of the estimate

Table 3.5. Chinook salmon parameters

Parameter	Meaning	Estimate	SD	Source/ Estimation Method	Included in Monte Carlo method ?	Sensitivity Analysis?
$S_{c-1, c-2, c-3, c-4, c-5}$	Initial spawners	10000	NA	Approximate value for a natural population given the watershed size (following the method of Liermann et al., 2010)	No	No
a	e^a = initial splope of the spawner-recruitment curve	1.922	NA	Estimated using watershed size as a proxy (following the method of Liermann et al., 2010)	No	Yes
b	Value at wich Recruits = Spawners in the spawner recruitment curve	12350	NA		No	Yes
ε	Log normal-error *LN(0, SD)*	0	0.29		Yes	No
es	Egg to smolt survival	$e^{-2.56}$	0.09*	Bradford, 1995	Yes	No
sa	Smolt to adult survival	$e^{-4.44}$	0.06*		Yes	No
δ	Sediment-related survival	Varies	Varies	for a season (*t*) and life stage (γ)	Yes	Yes
jA	Proportion of jacks of cohort in year 3.	30% of Ocean 2 fish	0.1**jpA*	(Pers. Comm. Ron Valer, Chilliwack hatchery)	Yes	No
jB	Proportion of jacks of cohort in year 4.	60% of Ocean 3 fish	0.1**jpB*		Yes	No
HR	Mean historical harvest rate	0.2928	0.0444	Pacific Salmon Commission (PSC) 2009.TCCHINOOK09-2	Yes	No

*Corrected for the standard error of the mean

Table 3.6. Parameter estimates adapted from Newcombe and Jensen (1996) for freshwater life histories in salmonids

Parameter		Life Stage		
		Eggs *(E*)*	Juveniles *(J)*	Adults *(S)*
B1	Intercept	3.7466	0.7262	1.6814
B2	Slope of $\log_e \theta$ (hours)	1.0946	0.7034	0.4769
B3	Slope of $\log_e \tau$ (mg L^{-1})	0.3117	0.7144	0.7565

*The data used by Newcombe and Jensen (1996) are a mix of salmonids and non-salmonid species. Non-salmonid eggs are typically exposed to SSC in the water column while the opposite occurs for salmonid eggs. However, there is a direct relationship between the amount of SSC and the amount of sediment fines that permeate the reeds (Grieg et al., 2005). In this study, I assume that exposure of SSC has a homogeneous effect on eggs and that the concentration of fine sediment in the gravel determines egg to fry survivals, as addressed by Newcombe and Jensen (1996).

Table 3.7. Mean, median, Standard deviation and coefficient of variation for spawners and catch after Monte Carlo simulations for coho salmon.

Scenario		Coho salmon							
Forest Mgmt.	Road Use	Spawners				Catch			
		Mean	Median	SD	CV%	Mean	Median	SD	CV%
Baseline	Moderate	15382	15190	4777	31	33834	33504	10455	31
No-Roads	NA	17124	16980	4822	28	36518	36327	10156	28
Current Plan	Light	16822	16582	4882	29	36079	35646	10395	29
Current Plan	Moderate	15206	14825	4688	31	33888	33164	10242	30
Current Plan	Heavy	12238	11498	5021	41	27700	26551	11575	42

Table 3.8. Mean median, Standard deviation and coefficient of variation for spawners and catch after Monte Carlo simulations for Chinook salmon.

Scenario		Chinook Salmon							
Forest Mgmt.	Road Use	Spawners				Catch			
		Mean	Median	SD	CV%	Mean	Median	SD	CV %
Baseline	Moderate	7476	7190	2520	34	3670	3536	1224	33
No-Roads	NA	8643	8281	2508	29	4111	3943	1176	29
Current Plan	Light	8484	8228	2376	28	4060	3944	1122	28
Current Plan	Moderate	7574	7139	2756	36	3768	3561	1346	36
Current Plan	Heavy	5894	5357	2870	49	3008	2745	1422	47

3.8 Appendix 1

Sensitivity Analyses

In the model, coho salmon are more sensitive than Chinook to changes in the threshold level at which fish start to show some degree of physiological response to suspended sediments (Appendix 1, Figures 1a and 1b, the radar plot helps demonstrating how the model outcome, number of spawners given different thresholds, changes in each scenario and how a particular scenario compares to others). The reason for the higher sensitivity of coho salmon compared to Chinook salmon is because the number of extreme sediment events that coho withstand in freshwater is higher due to the duration of their freshwater residence. For both Chinook and coho, lower sediment threshold levels (e.g., 10 mg SS L^{-1}) tend to be reflected in lower numbers of spawners, especially for scenarios of low and medium road use (k=1 to k=4). For the scenario of high road use (k=5), a slightly opposite behaviour is observed: higher threshold levels tend to be reflected in a small increase in the average spawner numbers. The reason is that lower threshold levels imply larger exposures to SSC, thus increasing the concentration and duration of events. In this case, the trend may be related to the magnitude of each event: a relatively large sedimentation event will cause significant mortality, which will, in turn, cause the population to "rebound" much more strongly than when the population is exposed to many small sedimentation events. This rebound behaviour is likely due to the spawner-egg production in the Ricker production function, where at lower spawners numbers, there is higher egg-per-spawner production.

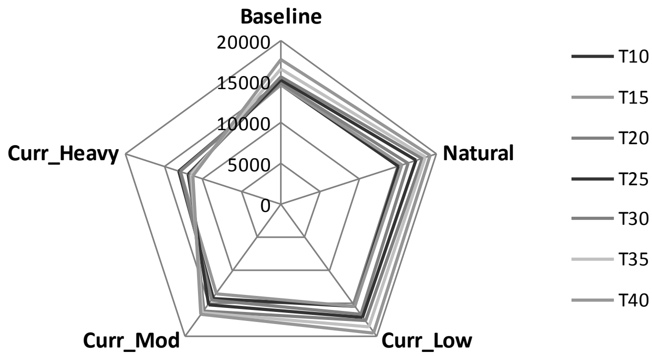


Figure 1a. **Radar plot of the model output for coho salmon representing the number of spawners for scenarios vs. selected threshold levels (10, 15, 20, 25, 30, 35, 40) at which individuals first respond to levels of SSC.**

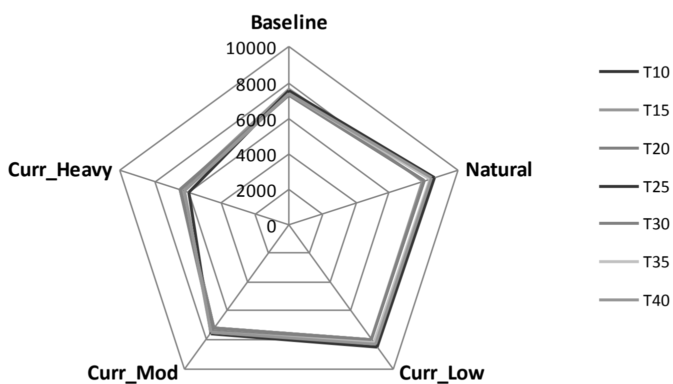


Figure 1b. **Radar plot of the model output for Chinook salmon representing the number of spawners for scenarios vs. selected threshold levels (10, 15, 20, 25, 30, 35, 40) at which individuals first respond to levels of SSC**

To account for the uncertainty contained in the mortality function derived from the estimates of Newcombe and Jensen (1996), I performed a sensitivity analysis of the parameters in Eq. 2. Ideally, in this case both the sensitivity threshold and the parameters in the mortality function should be accounted for. However, due to mathematical limitations of performing sensitivity analysis of more than two parameters, I only addressed changes in β and μ. The model output (the average number of spawners across scenarios) is more sensitive to slight variations in β than for variations of the same scale in μ (Figure 2, Appendix 1). The reason is that β is the only term in the denominator of Eq. 2; it thus influences the entire function more strongly than μ does. Changes in β determine how fast the mortality rate will increase for specific ill-effect values (see Figure 3.5).

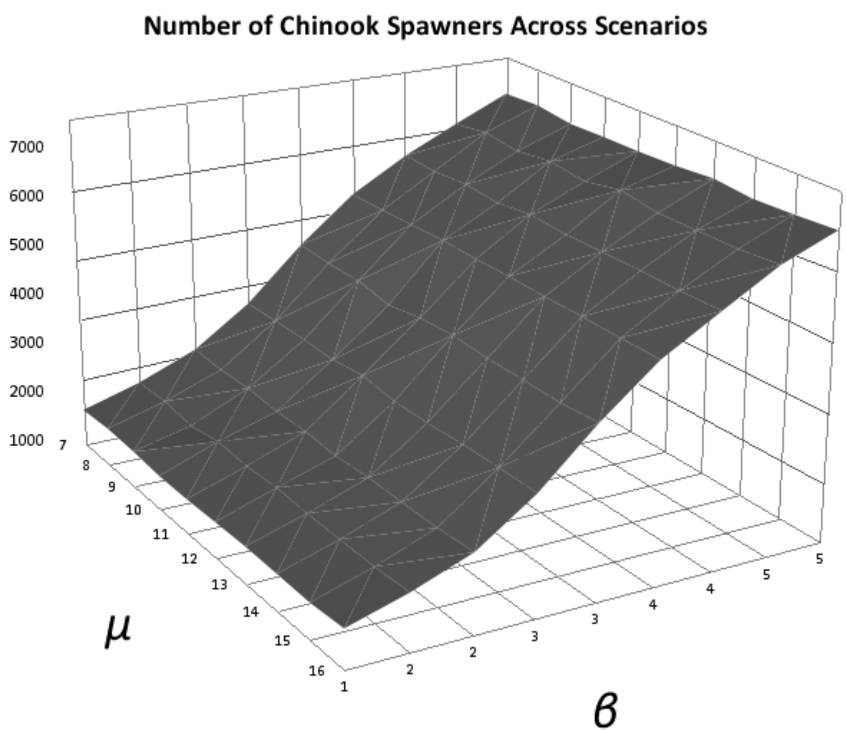


Figure 2. **3-D surface plot of the number of Chinook spawners across all scenarios (Eq. 2) for selected parameters values of μ and β. The parameter values selected for the sensitivity analyses include all the viable coefficients for the mortality function given the uncertainty contained in the estimates of Newcombe and Jensen (1996).**

As for the parameter values for the Ricker spawner-egg recruitment function for both coho and Chinook, I performed sensitivity analyses looking at the number of spawners across all scenarios. The sensitivity analyses demonstrate that, with the exception of extreme values, changes in parameter *b* do not affect the results while parameter *a* has a very small influence (Figure 3).

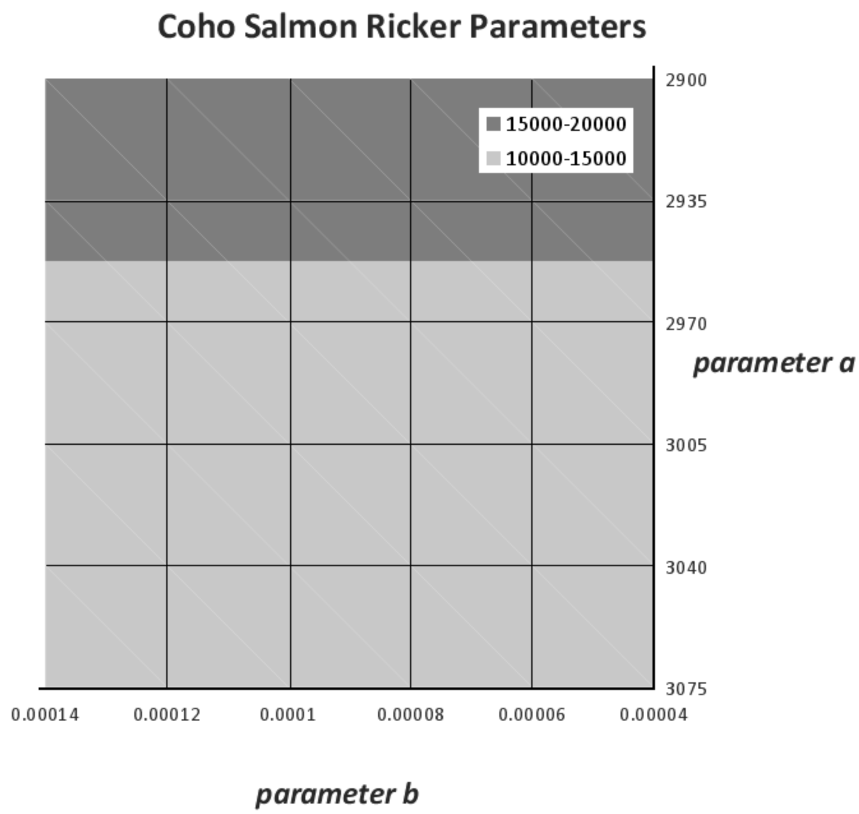


Figure 3. **The surface plot represents the number of spawners across all scenarios for different combinations of the Ricker parameters a and *b.* Plot suggests minimal sensitivity to parameter *a* and negligible sensitivity to parameter *b*.**

General Discussion

The general results indicate that the framework presented in this study has the potential to be developed into a decision support tool that allows managers to foresee potential changes in the hydrology of watersheds produced by human or climatic factors and their associated ecological effects on salmonid populations. The framework can be particulary useful when addressing the effects of climate change, especially in the absence of reliable data, as some authors suggest that climate change will increase the frequency of rainfall events that will, in turn, have a direct influence on the frequency of peak discharges. To deal with suspended sediment concentration data limitations, I have included the mixed-effects model, a tool that helps overcoming such limitations by using river discharge as a predictor variable. This model proved more accurate in extrapolating SSC than the traditional method, the sediment rating curve.

The methodology presented here uses the Chilliwack River as a case study to exemplify a coastal watershed of the lower Fraser River that is subject to a history of forest resource extraction practices and intense pressure by fisheries. When comparing the model results to other watersheds in the region, managers and researchers must keep in mind that assumptions and results can be quite different depending on factors such as the fish productive capacity and the sedimentation and discharge regimes experienced in such watersheds. Because the model assumes that hydrologic patterns in the watershed remain stable over time, it is most applicable to regions where hydrologic conditions show recurring annual and decadal patterns, such as British Columbia and the American Pacific Northwest.

Using a simulation approach to evaluate ecological trade-offs between sedimentation derived from the use of forest roads and the productivity of salmonid populations offers a viable way of quantifying the benefits of protecting watersheds and maintaining ecological integrity. This modeling approach provides the opportunity to study the Chinook and coho populations' response to the hydrological conditions on time scales that go far beyond the scope of field research. The model results suggest that increased sedimentation, whether caused by the use of forest roads or not, has negative effects on the watershed salmonid productivity. When evaluating the direct impact of forestry activities on fish production, we must keep in mind that many other interactions between the forest and fish take place in a watershed (e.g., input of large woody debris, nutrient exchange, habitat fragmentation, temperature regulation, etc). Therefore, negative impacts suggested by this study are likely an underestimation. Measures aimed at protecting aquatic species from the sediment produced by road networks associated with forestry activities should take into account restrictions on the use of forest roads. Road deactivation alone does not provide much habitat improvement for aquatic species without addressing their use.

Changes in the use of the road network have well-understood sediment production effects, and many studies have addressed the subject. In my case study, results indicate that reducing levels of suspended sediment yield caused by forest roads seems to be more effectively accomplished by limiting the use of available roads, rather than by controlling the road density alone. This finding does not suggests that road density is an unimportant factor when assessing impact on aquatic species. Simply focusing on an individual sediment source (i.e., forest roads) may not provide the information required to influence the overall sustainable management of watersheds. Nonetheless, the methodology presented here could provide insights into forestry impacts on watershed hydrological processes, and to an extent, could assist resource managers with planning effective conservation of natural resources in multiple-use watersheds.