VALUING THE WATER PURIFICATION/FILTRATION SERVICE OF TEMPERATE COASTAL RAINFORESTS IN SOUTHWESTERN BRITISH COLUMBIA

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ABSTRACT

In British Columbia, a lack of understanding exists concerning the tradeoffs between timber harvesting and maintaining ecosystem services, where losses of these services can occur as externalities from the timber harvest. As a result, the full economic potential of multiple-use watersheds frequently remains unrealized. This study provides insight into such tradeoffs by estimating the value of a change in a forest's water purification/filtration service, focusing on the quality of water as it becomes degraded from timber harvesting activities. I use an integrated economic-ecological model to quantify the economic impact of forest road induced sedimentation on raw water quality prior to its arrival at a municipal drinking water utility. With respect to road-induced sedimentation, I consider traffic volume and aggregate road length. I find that the economic value of the water purification/filtration service is more sensitive to traffic volume than aggregate road length and, therefore, should be subjected to more regulation in those watersheds that must consider the tradeoffs between the supply of clean drinking water and timber harvesting.

Keywords: Economic valuation; Ecosystem service; Water demand and supply; Hydrological and sedimentation modelling

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GLOSSARY

AFSY	Annual Fine Sediment Yield
AC	Average Cost
BAU	Business-as-usual Scenario
EAC	Equivalent Annual Cost
EMERG	Emergency Scenario
FC	Fixed Cost
ML	Million Litre
NCCW	Norrish Creek Community Watershed
NPV	Net Present Value
NTU	Nephelometric Turbidity Units
SSC	Suspended Sediment Concentration
TSA	Timber Supply Area
VC	Variable Cost

1: INTRODUCTION

1.1 General Overview

Human wellbeing is dependant on the health of the world's ecosystems. Left intact or relatively undisturbed, healthy ecosystems can contribute to production by providing many ecosystem goods and services.[1] The Millennium Ecosystem Assessment (MEA) labelled these benefits collectively as ecosystem services and, by definition, these include any human derived benefit obtained from ecosystems.[2] The purification of air and water, the supply of fish and timber, recreational enjoyment and spiritual fulfilment are but a few examples (Millennium Ecosystem Assessment, 2005a).

Unfortunately, many ecosystem services are currently at risk of degradation. Primarily to support a growing human population, healthy ecosystems are increasingly being converted to land-uses conducive to economic development, such as forestry and agriculture. Such land transformation generates economic rent and employment, but at the same time, can strain the natural delivery of other, possibly more economically valuable, ecosystem services (The World Bank, 2004). For example, timber harvesting in a community watershed is known to negatively impact the quality of raw water prior to its arrival at a water utility (Gomi et al., 2005). [3] Society must then rely on costly

[1] Ecosystem health refers to the ability of a naturally functioning ecosystem to sustain itself overtime (Costanza et al., 1997)

[2] The MEA attempts to define and relate ecosystem services to human wellbeing. Over 1,360 globally recognized scientists contributed to the analysis (Millennium Ecosystem Assessment, 2005a)

[3] A community watershed is the drainage area above the point of diversion on a stream for a water use that is for human consumption (British Columbia Ministry of Forests, Mines and Lands, 2010)

manufactured substitutes, such as a water treatment utility, to compensate for the loss of ecosystem service.

Decision makers are often faced with choosing among competing uses of the natural environment. In doing so, decision makers must assess the tradeoffs by quantifying the change in consumer and producer surplus arising from a shift from one possible land-use to another (Freeman, 2003). However, the opportunity cost of ecosystem service degradation has typically been overlooked, primarily due to an inability to adequately define and quantify the economic worth of these services (Daily, 1997; Lara et al., 2009; Millennium Ecosystem Assessment, 2005b). Nonetheless, both economists and ecologists continue to agree the value of ecosystem services must be captured to properly formulate sustainable land-use management strategies (Barbier, 2000; Costanza et al., 1997; Daily, 1997; de Groot et al., 2002; Millennium Ecosystem Assessment, 2005b).

A sustainable land-use management strategy is defined as one that maximises social welfare across all possible management scenarios. The inclusion of the economic value of ecosystem services is therefore crucial for proper results (Freeman, 2003). This study conducts an economic valuation of an ecosystem service in a multiple-use watershed, and uses the results to assess the tradeoffs among pre-defined alternative forest management scenarios. The results will therefore contribute to the sustainable management planning in a watershed that must consider tradeoffs among competing land-uses.

1.2 Problem Statement

A growing concern among British Columbia (BC) residents is the availability of current and future supply of domestic water. This concern is reflected in Metro Vancouver and Greater Victoria's decision to restrict all land-uses in their domestic source water supply watersheds, and manage solely for drinking water protection (British Columbia Ministry of Health Services, 2007). Currently, almost half of BC's population access water from multiple-use watersheds where uses such as forestry, recreation and agriculture are permitted (Atkins et al., 2003). This study evaluates the interaction between two watershed uses, forestry and municipal drinking water supply.

As stated previously, timber-harvesting practices can negatively impact the quality of raw water prior to its arrival at a water utility (Gomi et al., 2005). Currently in BC, the main focus of forestry and water quality related research is centred on sediment production (Hudson, 2001b). The main causes of induced sedimentation are erosion from forest roads (Beschta, 1978; Brown & Krygier, 1971) and forest road related landslides (Brardinoni et al., 2003). Under the right conditions, both have the potential to produce sediments in excess of what a watershed can filter out naturally and thus result in degraded water arriving at a water utility. Though sediments in lakes, rivers, and streams are natural, excessive sedimentation is not.

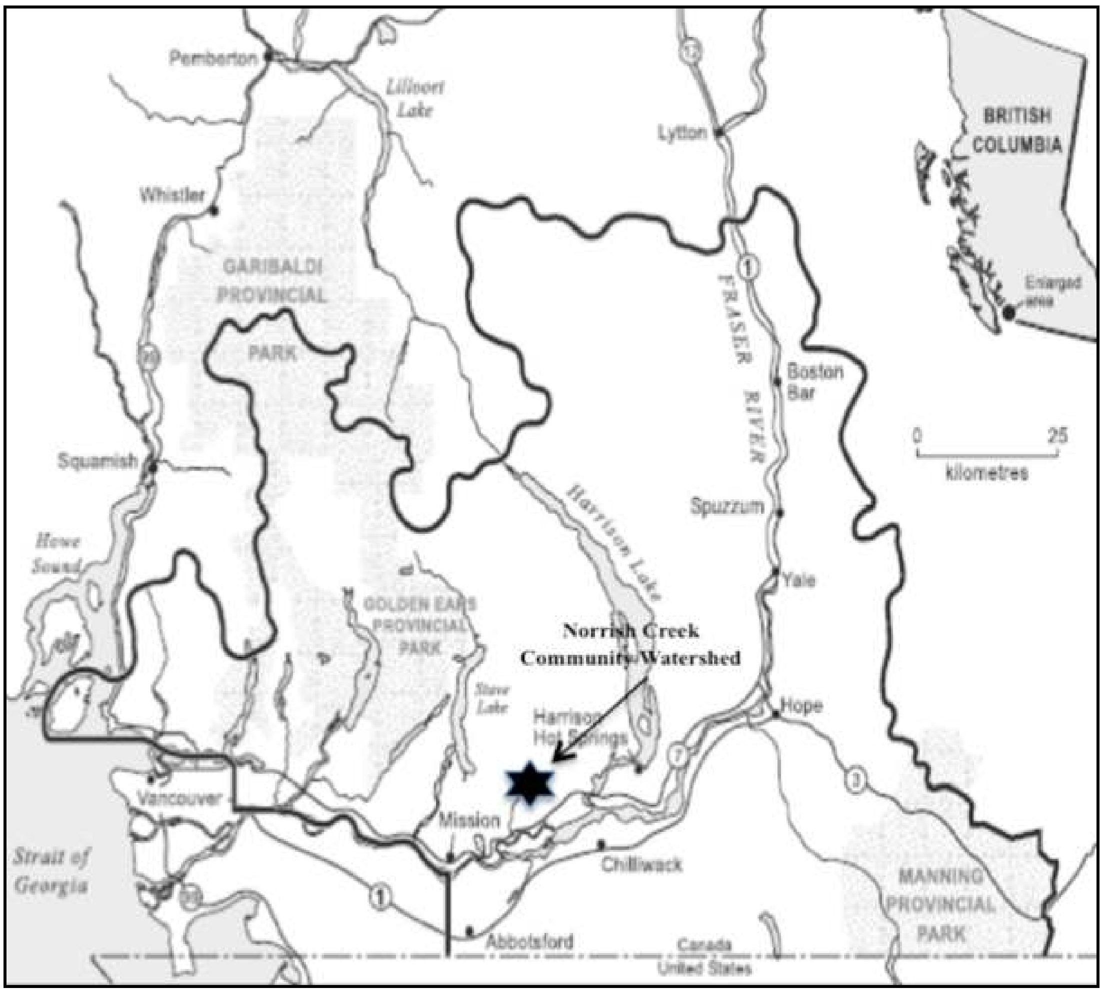
Since water quality can become degraded prior to arrival at the water utility, timber harvesting can increase the cost of water supply. For example, decreased water quality can result in increased facility maintenance, replacement and upgrades, mitigation, or other preventative and costly measures required to ensure safe drinking water (Gomi et al., 2005; Zubel, 2006). Therefore, changes in the quality of water can have an economic value (or cost, in this case).

By correctly formulating the economic valuation problem, I will quantify the economic contribution to the natural production of clean drinking water (i.e. without sedimentation) in a representative BC watershed, and account for how this value changes as timber harvesting progresses. The results of the economic valuation will contribute to the management planning in the watershed, which currently must consider the tradeoffs between the supply of clean drinking water and timber harvesting. Furthermore, the results of this study will contribute to a broader economic valuation exercise. An initial study estimated the net present value (NPV) of various forest services under alternative forest management scenarios; however, it did not include any water related watershed services (Knowler & Dust, 2008). This study will fill that gap.

1.3 Description of Study Area

The Norrish Creek Community Watershed (NCCW) provides the perfect example of a multiple-use watershed and is the study area for this research project. Located within the Fraser Timber Supply Area (TSA) in southwestern BC, the NCCW contains important commercial timber harvesting sites and is the source of domestic water for thousands of local residents. The NCCW supplies water to approximately 156,000 City of Abbotsford and the District of Mission residents (Dayton & Knight Ltd., 2006). The watershed has a long history of active logging, the impacts of which have resulted in significant decreased water quality events (Zubel, 2006). Logging roads within the watershed continue to be actively used by logging trucks and recreationists. Due to such concerns, the City of Abbotsford and District of Mission have been relying on more water treatment to ensure safe consumable water (Zubel, 2006).

The NCCW was selected for this study due to its importance for drinking water quality, geographic location, data availability, and for exhibiting the characteristics of a representative watershed in the lower Fraser TSA with an extensive logging history (Figure 1).



Source: DeGrace, n.d.

Figure 1: Location of the Norrish Creek Community Watershed in the Fraser Timber Supply Area

The NCCW drains into the north side of the Fraser River at Nicomen Slough. It is situated within the Pacific Ranges of the Coast Mountains and covers approximately 118 km^2. The community watershed encapsulates approximately 80 km^2 (above the intake). The watershed is fork-shaped with 6 sub-basins (Dickson Lake, East Norrish Creek, West Norrish Creek, Hanson Creek, and Cry Creek). In the lower Norrish area, the major tributaries are Rose Creek, Sally Creek, and Naknamura Creek (Brayshaw, 2006). The watershed lies between 250 m to 1,420 m in elevation with approximately 44% of the watershed lying within the transient snow zone (300 m - 800 m), 55% above 800 m, and the remaining 1% located below 300 m elevation. Since the 1940's, approximately 56% of the watershed has been logged equating a low average rate-of-cut of 0.93% per year (Chapman Geoscience Ltd., 2000).

1.4 Research Objectives and Questions

The main objective of this research project is to conduct an economic valuation of a water related ecosystem service in a forested watershed. In doing so, this study will contribute to the management planning in the NCCW by estimating the welfare impact of alternative forest management scenarios that account for the loss of raw water quality due to timber harvesting activities. More specifically, I aim to address the following research questions:

- What are the ecological impacts of timber harvesting on the supply of drinking water?
- What ecosystem services contribute to the supply of drinking water, focusing on water quality?

- What costs are associated with degraded raw water quality in the production of safe drinking water?
- How do changes in production costs due to raw water quality affect social welfare?

1.5 Report Organization

Chapter 2 provides a review of the economic valuation literature as it relates to watershed services. The methodology I used to conduct the economic valuation is presented in Chapter 3. Chapter 4 details the water quality time series simulation conducted for each alternative forest management scenario. Chapter 5 documents model estimates, followed by Chapter 6, which presents the final valuation results and provides a more in-depth discussion. Lastly, Chapter 7 concludes the report.

2: BACKGROUND AND LITERATURE REVIEW

This chapter reviews the economic valuation literature as it relates to watershed ecosystem services. I begin with a general overview of economic valuation. Then, I describe the ecosystem service values associated with watersheds, focusing on water related values, specifically water quality. Next, I review the common approaches to valuing water quality. Such approaches require knowledge of land-use impacts on the ecosystem service in question; therefore, I follow with a review of timber harvesting activity impacts on water quality. I conclude the chapter with a review of previous attempts at valuing changes in drinking water quality induced by the forest industry.

2.1 Economic Valuation Overview

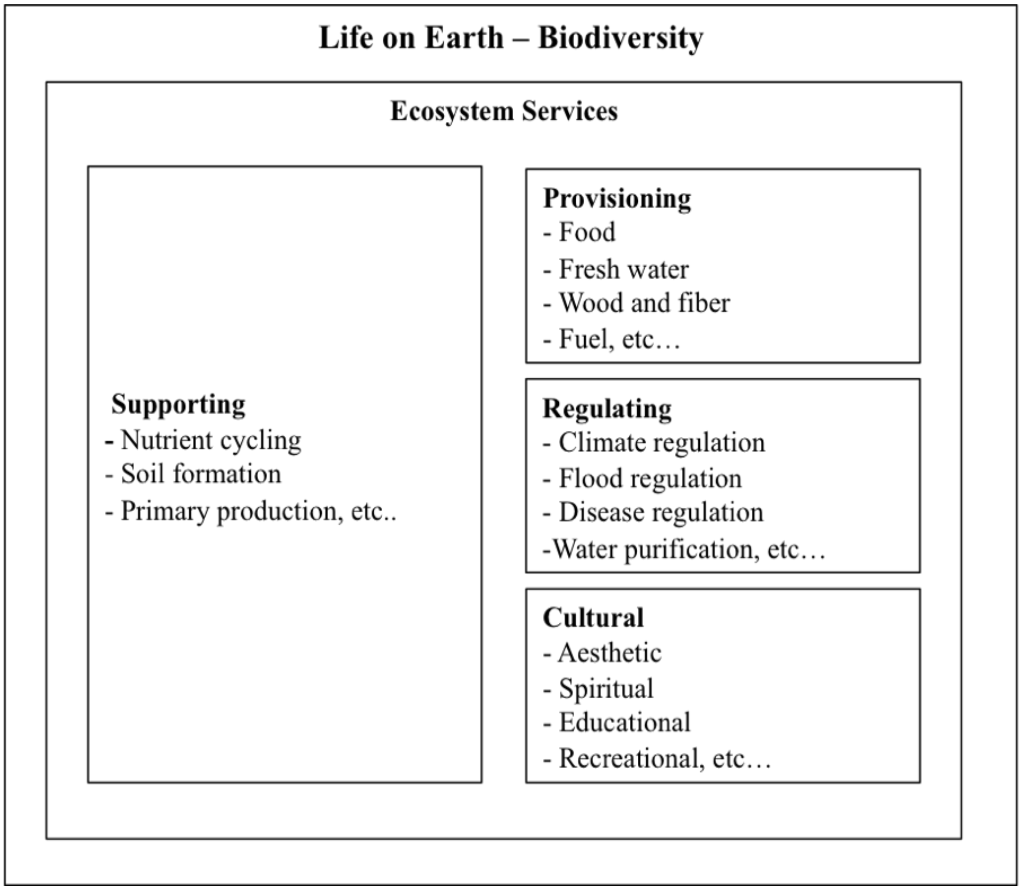
Economic valuation is a tool that can be used to quantify the utilitarian value of ecosystem services[4] (Millennium Ecosystem Assessment, 2005b). Typically, ecosystem services are not traded in any formal markets and therefore lack an ascribed monetary value (Olewiler, 2004). For example, there is no market price attached to the services that make it possible for us to breathe clean air or drink clean water. As a result, land-use decision makers often overlook the contribution ecosystem services make to human wellbeing (Lara et al., 2009; Millennium Ecosystem Assessment, 2005a).

The current economic development paradigm favours transforming ecosystems to land-uses that support commercial activity, such as forestry and agriculture.

[4] Comparatively, non-utilitarian value refers to the intrinsic value ascribed to ecosystem services, such as existence values (MEA, 2005).

Unfortunately, such land-uses can significantly degrade other ecosystem services. By assigning a monetary value to these services, economic valuation offers a means to compare the trade-offs among alternative land-uses (Bockstael et al., 2000). The ability of decision makers to properly formulate sustainable ecosystem management plans is thus significantly improved.

Many techniques have been developed to value ecosystem services lacking explicit market prices (Millennium Ecosystem Assessment, 2005b). The technique chosen depends on the type of ecosystem service in question, and specifically how it contributes to human wellbeing. Therefore, the MEA developed a framework that categorized both ecosystem services and the most common components of human wellbeing (Figure 2) (Millennium Ecosystem Assessment, 2005a).



Source: adapted from Millennium Ecosystem Assessment (2005a).

Figure 2: The Millennium Ecosystem Assessment's ecosystem service categories and examples

The four MEA ecosystem service categories include provisioning, cultural, regulating and supporting services. Provisioning services are the tangible goods produced by ecosystems, such as timber, water, and fuel. Cultural services are "closely linked to human values, identity an behaviour" (Figueroa & Pasten, 2009) and include, for example, spiritual, recreational, and aesthetic benefits. Regulating services control ecosystem processes, such as climate and water quality regulation. Lastly, supporting services, such as photosynthesis and nutrient cycling, support the functioning of other ecosystem services (Figueroa & Pasten, 2009; Millennium Ecosystem Assessment, 2005a).

The purpose of this study is to economically value a water related ecosystem service that falls within one of the four MEA ecosystem service categories. Therefore, the following section provides a description of such services as defined by the MEA.

2.2 Description of Water Related Watershed Services

A watershed is an area of land where the entire water table, above and below ground, flows to the same destination. The geographic boundary includes all lakes, streams, rivers, wetlands, ground water sources, and forested areas (United Sates Environmental Protection Agency, 2007). As such, watersheds provide a myriad of interconnected water related ecosystem services.

Due to the comprehensive number of services provided by watersheds, the MEA found it difficult to categorize water related ecosystem services as purely provisioning, supporting or regulating. For example, as a provisioning service, forested watersheds supply water for human use, such as domestic consumption, hydroelectricity and irrigation. As a supporting service, the hydrologic cycle produces the sole source of renewable fresh water. Forested watersheds enable this process to happen. Finally, as a regulating service, forested watersheds regulate water quantity and quality (Daily, 1997; Millennium Ecosystem Assessment, 2005a). Therefore, watersheds express all three of the four MEA categories.

This study focuses on the third MEA category, and specifically on a watershed's ability to regulate water quality. However, no single ecosystem service that falls within the regulating category is solely responsible for water quality. For example, soil retention, water purification/filtration, runoff protection (to name a few) all contribute to quality. Since this study evaluates the impact of timber harvesting activities on raw water quality, I will focus specifically on the purification/filtration service. This service is defined as the ability of an intact forest to retain or filter out waste products, nutrients, sediments, and pollutants that would otherwise enter a water body if, for example, it were logged (Lambert, 2003; Postel & Thompson, 2005).

2.3 Approaches to Valuing Water Quality

Economic valuation methodologies are classified as stated or revealed preference techniques. The stated preference approach uses questionnaires or surveys to determine individuals' willingness to pay (i.e. value), while revealed preferences rely on actual human behaviour, generally using the prices people pay for related goods and services (Freeman, 2003).[5]

Common stated preference methodologies to value water quality include contingent valuation and choice experiments. Contingent valuation asks individuals their willingness to pay for a specific ecosystem service by "posing hypothetical scenarios that involve some valuation of alternatives" (Farber et al., 2002). For example, Loomis et al. (2000) asked individuals what their willingness to pay would be to restore a selection of ecosystem services along a river; the water purification/filtration service was included as one. The authors found that individuals are willing to pay $21 per month for the additional rehabilitated services. Choice experiments ask respondents to choose among alternative profiles that contain specified attribute bundles among which is a payment

[5] Willingness to pay refers to the amount of money a respondent is prepared to give up in exchange for an environmental quality improvement or to avoid an expected environmental loss. In contrast, willingness to accept measures the amount respondents are prepared to accept as compensation when there is a loss in environmental quality or to forego a promised environmental improvement (Brown and Gregory, 1999).

attribute (Millennium Ecosystem Assessment, 2005b). For example, Hanley et al. (2006) created ecological status (good and fair) profiles and asked respondents their preference. The authors found that individuals are willing to pay roughly $20.17 for water quality improvements.

Common revealed preference methodologies to value water quality include the hedonic method, travel costs, avoided costs and the production function (Millennium Ecosystem Assessment, 2005b). The hedonic method assumes that environmental factors (i.e. water quality) are attributes of goods or factors of production that are traded in the market. The environmental attribute is thus reflected in the price people pay for the good or service produced (Farber et al., 2002). Leggett et al. (2000) used the hedonic method to capture the value of water quality in the market price of land. Travel cost elicits value from actual travel cost data (i.e. fuel, hotel costs, etc) (Millennium Ecosystem Assessment, 2005b). For example, Choe et al. (1996) estimated the value individuals placed on improved water quality by surveying the number of times individuals travelled to a beach before and after a water quality advisory was posted. The avoided cost approach ascribes value to an ecosystem service based on the costs avoided from environmental protection. The Catskills project is the most notable example (Postel & Thompson, 2005) and is discussed in more detail below. Lastly, the change in productivity method traces the impact of environmental change on produced goods (Millennium Ecosystem Assessment, 2005b). For example Nunez et al. (2006) estimated the economic contribution of Chilean temperate forests to the supply of clean drinking water. The authors found that during the summer months, the value per hectare of native forests was $162.40 USD compared to $61.20 USD throughout the remainder of the year.

As stated previously, the methodology chosen depends on the link between the ecosystem service and human wellbeing. Therefore, the following sections detail how the forest industry impacts raw water quality, followed by the methodology chosen to value the water purification/filtration service.

2.4 Forest Industry Impacts on Water Quality

The current focus of forestry-water quality interaction research in BC is centred on sediment production above background levels (Hudson, 2006a). Sediments entering a river or stream occur naturally or are introduced from human activity. Within a logged watershed, erosion from logging roads and logging road related landslides are two primary sources of introduced sediment (Beschta, 1978; Brardinoni et al., 2003; Brown & Krygier, 1971; Jordan, 2006; Reid, 1981; Reid & Dunne, 1984). Specifically, forestry roads are known to be the "primary harvesting-related source of fine sediments in streams" (Hudson, 2006b) and thus a significant factor when assessing forestry impacts on drinking water (Gomi et al., 2005).

Erosion from logging roads introduces fine sediment production in the following ways. First, road generated erosion is affected by the season in which the road is built, road construction techniques, maintenance and the characteristics of the road surface, as well as by cutbank height, slope, vegetative cover, and surficial and bedrock geology. Second, factors such as the amount of precipitation and the level of overland flow on a road determine the amount of sediment reaching the stream (Greater Vancouver Regional District, 1999; personal communication, Dave Dunkley, June 2009).

Road surfaces play an important role in sediment generation. Roaded watersheds may be paved in high use areas and gravel filled in other areas. Reid and Dunne (1984) found that paved roads yielded less than 1% of the sediment generated by gravel roads. If gravel road surfaces are continually in use, they provide a significant source of fine sediment. The use of the road abrades and fractures the gravel and forces it into the substrate which then drives the fine material to the surface. Once at the surface, the fine sediments are removed by flowing water that accumulates on the road and it then washes into the stream (Reid et al., 1981).

Reid and Dunne (1984) also demonstrated that "road surface erosion is extremely sensitive to traffic levels". Traffic levels are defined as the number and types of vehicles using the roads. The researchers found that heavily used forestry roads (i.e. active hauling roads) in their study area contributed 130 times the sediment generation compared to the same length of abandoned forestry roads. Wald (1975) also concluded that on average, heavily used gravel roads produce significantly more fine sediments compared to decommissioned roads.

2.5 Previous Forestry – Drinking Water Quality Valuation Attempts

Numerous studies exist that detail the impact of forest harvesting activities on water quality (e.g. Beschta, 1978; Brardinoni et al., 2003; Brown & Krygier, 1971; Jordan, 2006; Reid, 1981; Reid & Dunne, 1984). A number of studies also demonstrate correlations between water quality and the costs of water treatment (e.g. Forster & Murray, 2001; Holmes 1998). However, few studies link the above two academic fields of study, that is, to estimate the economic value of drinking water quality as it becomes degraded from timber harvesting activities. Below are a few attempts.

Ernst (2004) investigated the link between the percentage forest cover found within a drinking water watershed and the cost of water treatment. The author surveyed 40 United State water suppliers, of which 27 were used to analyze the ecology, treatment system and associated costs for each watershed. The author found that with a 10% increase in forest cover, costs decreased by roughly 20% on average. However, the percent forest cover could explain only 50% of the variation within the data. To obtain a stronger relationship, variables such as forestry management practices should be included (Ernst, 2004).

Freeman et al. (2008) used a general linear model to analyze water quality, land cover and chemical treatment costs after surveying over 60 drinking water treatment plants. The authors also found that within the United Sates, a significant relationship exists between the percentage of forest cover and water treatment costs. However, due to high variability in the data, the relationships were weak despite their significance (Freeman et al., 2008).

Postel and Thompson (2005) reviewed numerous avoided costs studies with respect to drinking water quality. The most notable is New York City's Catskills study, which concluded that the installation of the filtration plant would cost roughly $6 to $8 billion, plus $300 million to operate annually; compared to the $1 billion it would cost to purchase and sustainably manage the land. Other examples include Boston and Seattle which both opted to avoid constructing filtration plants at $180 million and $150 – $200 million respectively (Postel & Thompson, 2005).

Unlike Ernst (2004) and Freeman et al. (2008), Postel and Thompson (2005) estimated the value of water quality based on the costs avoided through watershed protection, rather than directly linking costs to timber harvesting. Other researchers contend such an approach is useful, but should not be used to infer the correct economic value associated with the ecosystem service in question. Banzhaf and Jawahar (2005) argue that the associated costs to supply a given amount of a good is not necessarily the level of benefit one would receive from the supply of that good. Therefore, it is important for the researcher to properly specify the link between physical changes in the environmental quality (i.e. water quality) to changes in production/costs (Freeman, 2003).

The above studies estimate the economic value of water quality improvements in one way or another. However, they do not directly link the level of forest activity (e.g. road use) to sediment production, and then to specific drinking water treatment costs. This study aims to do so by simulating alternative forest management scenarios, considering their impact on induced stream sedimentation, followed by an economic analysis of drinking water costs using the change in productivity approach.

3: STUDY APPROACH AND METHODS

This chapter describes the study approach, methodology used, and the necessary background information to estimate the economic value of the water purification/filtration service provided by a forested watershed. I begin with the study's general approach. I then describe the economic characteristics of a municipal water utility, followed by a description of the economic valuation model, including the appropriate welfare measure and the methodology used to estimate water demand and supply. I conclude with a brief summary of data sources.

3.1 General Approach

I constructed an integrated economic-ecological model to value the water purification/filtration service provided by a forested watershed. The ecological component consisted of a raw water quality time series simulation conducted for alternative forest management scenarios. The time series simulation is more involved and is described in Chapter 4. Each simulated time series was used to estimate a change in environmental quality, which in this study, was raw water quality. The economic component incorporated the change in raw water quality within a municipal water utility's cost function. In other words, the economic model valued the natural environment (i.e. water quality) as a factor input into the production of municipal water supply. The resulting change in supply costs was used to estimate the change in welfare, thus ascribing an economic value to the water purification/filtration service.

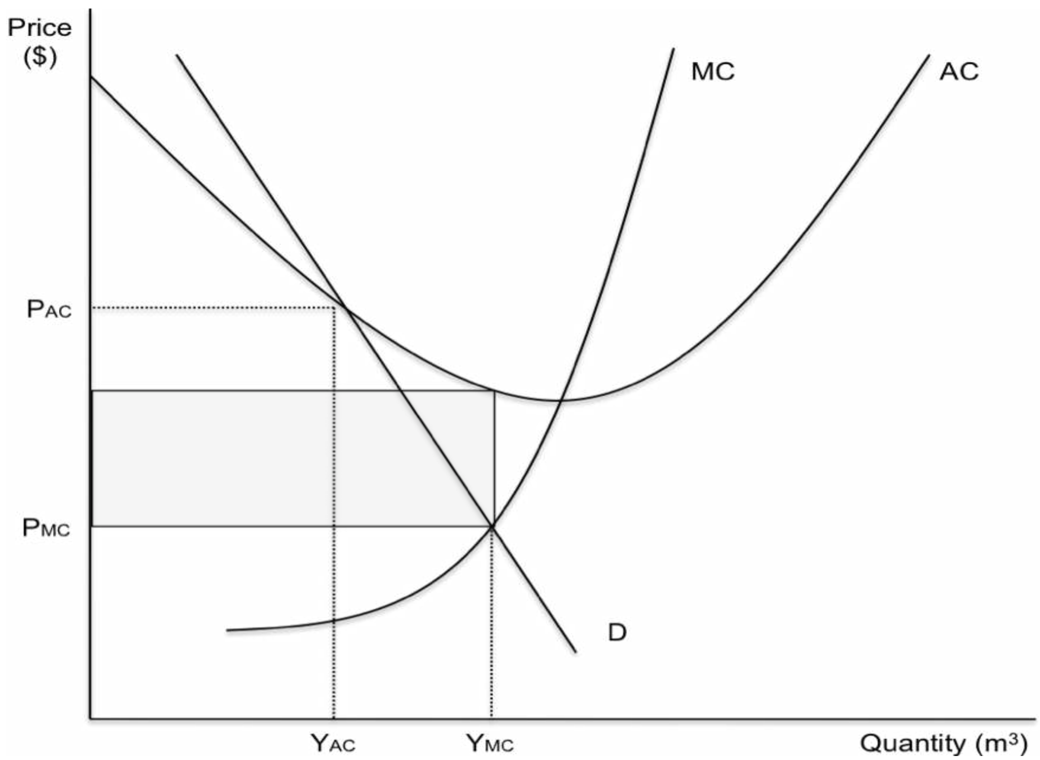
3.2 Description of a Municipal Drinking Water Utility

To determine the welfare change associated with a change in environmental quality, the economic characteristics of a municipal water utility must be known. Specifically, the economic valuation analyst must understand how the commodity's market price and quantity are determined and change as a result of a change in factor inputs (Freeman, 2003). Therefore, this section describes the economic characteristics of a typical municipal water utility that supplies treated drinking water.

When it is efficient for a single firm to supply the entire market, such as in the case of municipal drinking water supply or petroleum pipelines, a monopoly is often granted. Single, large-scale water supply facilities typically experience increasing returns to scale. As the quantity of drinking water supplied increases, the cost per unit decreases (Hosking & Preez, 2004).

However, a profit-maximizing monopolist produces output where marginal revenue equals marginal cost, and thus generates additional profit at the expense of the consumer. Not only does the monopolist not charge an efficient price, the firm also produces an inefficient level of output relative to the Pareto efficient amount, where price equals marginal cost (Varian, 2003). Operating a public drinking water utility as a monopoly is thus, socially undesirable.

To eliminate the inefficiencies, it seems logical to regulate the monopoly. The regulator would set price equal to marginal cost, thus producing an efficient level of output at a competitive price. However, drinking water utilities usually face large fixed costs and very low marginal costs; constructing and maintaining a water treatment plant is costly, but once the plant is in full operation, it costs very little to provide an additional unit of drinking water. Therefore, the utility's marginal costs could be well below its average costs, thereby creating the possibility for the firm to make negative profits if it were to be regulated. This situation is referred to as a *natural monopoly* (Figure 3). The problem it presents can be resolved by allowing the government to own and operate the utility. Ideally, the government would aim to operate at price equal to marginal cost and provide a subsidy to cover the large fixed costs (Varian, 2003).



Source: adapted from Varian (2003).

Figure 3: Losses incurred by a natural monopolist under marginal cost pricing (shown as the shaded area)

However, Brandes et al. (2010) state that most Canadian water utilities use some form of average cost pricing, not marginal cost pricing. The authors indicated that even though the economic literature highlights marginal cost pricing as the preferred theoretical methodology, that it is in fact too complex to estimate in the Canadian context. For example, the Canadian weather and distances from source to household taps are quite variable, thus producing highly variable marginal costs. The variability becomes problematic when estimating the required subsidy for the utility to break-even, which usually is required of many Canadian municipal governments. Therefore, utilities opt to price water at average costs (Barndes et al., 2010).

3.3 Economic Model

Economists value the natural environment when it is a factor input in the production of a marketed commodity. As changes in the natural environment occur, so do changes in the cost of production, which then lead to changes in the commodity's market price and quantity supplied. The resulting welfare change is what economists define as the economic value of the associated change in the natural environment (Freeman, 2003). This study estimates the economic value of the water purification/filtration service by estimating how a change in raw water quality affects a municipal water utility's cost function and then uses the average cost pricing rule to determine the welfare change. The following describes the theoretical estimation procedure.

3.3.1 Cost of Production

Knowledge of a water utility's cost function can be useful for estimating the change in producer and consumer surplus associated with a non-marginal change in raw water quality (Freeman, 2003). The cost function for a firm relying on an exogenous environmental input can be written as:

$$C = C(Q;\lambda) \tag{3.1}$$

where C is the firm's cost to supply drinking water ($/m^3), Q is the quantity of treated water supplied (m^3), and λ represents the fixed environmental input (i.e. raw water quality, which is detailed in section 3.3.3) whose value is determined exogenously to the firm. Here I assume λ is a function of timber harvesting activities, and express it as:

$$\lambda = f(l,u) \tag{3.2}$$

where l represents the aggregate length of roads (km) and u represents the fine sediment yield (tonnes/km/yr) generated by the intensity of road use and road slope within the watershed that is the source for drinking water (further detailed in Chapter 4).

The drinking water utility's total contribution to welfare (W), defined as the sum of consumer and producer surplus, is expressed as:

$$W = \int_0^Q P(Q)dQ - C(Q;\lambda) \tag{3.3}$$

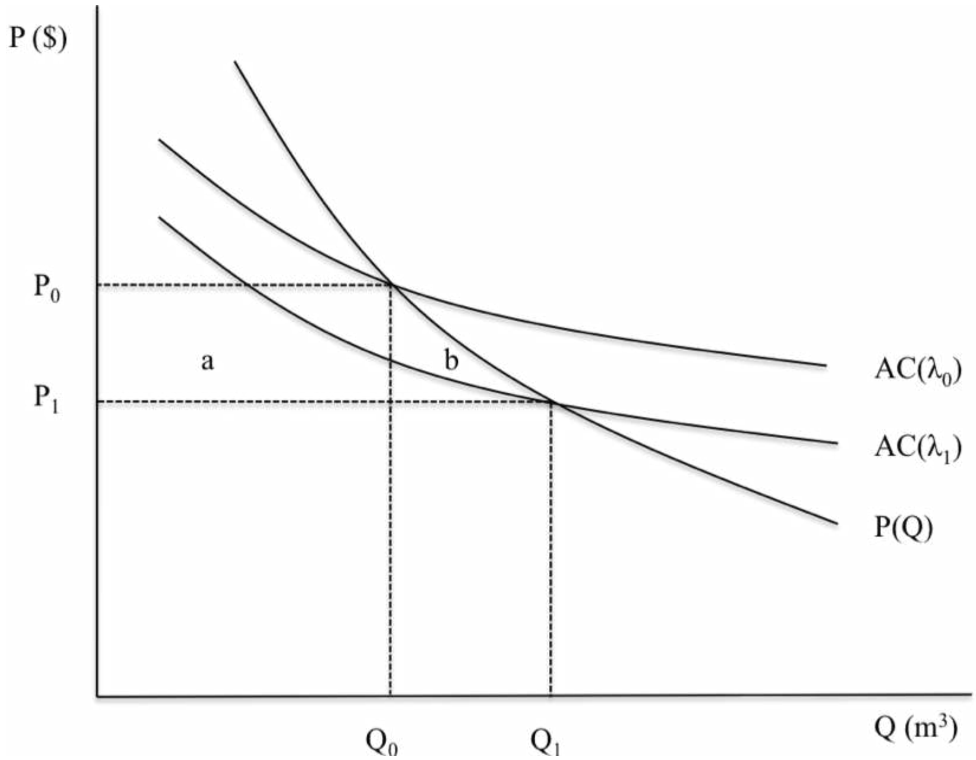
$$s.t. Q \leq \overline{Q} \text{ and } \lambda_s = f(l_s, u_s)$$

where $P(Q)$ is the inverse demand curve for treated water ($/m^3). Note that the quantity supplied cannot exceed the physical capacity of the drinking water facility infrastructure ($\overline{Q}$), and λ, as described above, represents the road characteristics for a particular forest management scenario (s). The change in producer and consumer surplus associated with a non-marginal change in raw water quality (λ) is expressed as:

$$\Delta W = \int_0^{Q_1} P(Q_1^*)dQ - C(Q_1^*;\lambda_1) - \int_0^{Q_0} P(Q_0^*)dQ - C(Q_0^*;\lambda_0) \tag{3.4}$$

where Q^* is the equilibrium quantity supplied and 0 and 1 refer to the situation before and after the environmental change. The environmental change represents the change between forest management scenarios and is further detailed in Chapter 4. In the absence of a natural monopoly, I would derive the equilibrium quantity (Q^*) for each forest management scenario by equating the demand for and the supply of treated water (Appendix A) (Varian, 2003).

However, the average cost pricing approach instead equates total revenue and total cost, which allows the water utility to break even, but dissipates any producer surplus. Changes in social welfare are, therefore, measured by changes in consumer surplus only, as producer surplus is zero. The estimated economic value of the water purification/filtration service provided by a forested watershed is thus the change in consumer surplus. The theoretical welfare measure is the area between the old ($AC(\lambda_0)$) and new ($AC(\lambda_1)$) supply curves bounded by the inverse demand curve ($P(Q)$), shown as the sum of areas a and b in Figure 4 (Ellis & Fisher, 1987; Freeman, 1991; Freeman, 2003).



Source: adapted from Freeman (2003).

Figure 4: Welfare gain from improved water quality under an average cost pricing approach

3.3.2 Water Demand and Supply Curves

To determine the welfare change due to a change in costs as outlined above, it is necessary to estimate the water demand curve and supply curves. The estimation procedure is described next.

Demand Curve

Water analysts use price elasticities from previous studies or established rules of thumb (e.g. Griffin, 2006) to estimate residential water demand. Griffin (2006) developed numeric rules of thumb for water mangers to use directly when forecasting the response of residential water use to price changes (Table 1). These rules of thumb are an effective way to include reasonable elasticity values in a water-demand forecast when more specific information is not available (Billings & Jones, 2008).

Table 1: Rule-of-thumb elasticity values for residential water use to price changes

	Marginal Price Rate Structure		
	Declining	Constant	Increasing
Base case elasticity (annual)	-0.3	-0.4	-0.5
Additions of Subtractions			
Wet/cold climate	+0.1	+0.1	+0.1
Arid West	-0.1	-0.1	-0.1

Source: adapted from Billings and Jones (2008).

Billings and Jones (2008) used the following equation to estimate quantity demanded:

$$\Delta Q = \frac{\Delta P}{P} Q \in \qquad (3.5)$$

where ΔQ is the change in quantity of treated water demanded, ΔP is the change in treated water supply price, P is the starting treated water supply price, Q is the starting quantity of treated water demanded, and $\in$ is the price elasticity of demand.[6] Furthermore, I used incremental 10% increases in price (e.g. from \$0.10 to \$0.20), which was recommended when the goal is to estimate large percent changes in price. I also assumed a constant elasticity of demand (Billings and Jones, 2008).

[6] The actual demand curve inferred by (3.5) under a constant elasticity of demand can be expressed as $Q=AP^{\in}$, where Q is quantity demanded (\$/m^3), A is an arbitrary positive constant, P is price of treated water (\$/m^3), and $\in$ is the constant price elasticity of demand (Varian, 2003).

Supply Curve

As stated previously, Canadian municipalities price treated water based on average costs. Therefore, I used the following expression to estimate the AC curve:

$$P = AC = \frac{FC + VC}{Q} \tag{3.6}$$

$$where\ VC = f(Q;\lambda)$$

where *FC* is the total fixed cost of treated water ($\$/m^3$) and *VC* is the total variable cost of treated water ($\$/m^3$). Note that the environmental quality parameter (λ) is used to estimate *VC* and its theoretical estimation is detailed in the next section. The FC and VC empirical estimations and calculations are further detailed in Chapter 5.

Furthermore, to estimate equation (3.6), I estimated the total water supply costs for a municipal water utility. However, economists only include real resource costs when the purpose is an 'economic' analysis (Florio & Vignetti, 2003). For example, debt and transfer payments do not lead to real resource use, unlike operating and capital costs. Also, to place capital costs on an annual basis, I used the Equivalent Annual Cost (EAC) methodology together with the present value capital costs (Adair, 2005). The methodology is used to derive the annualized cost over the capital asset's lifespan and is expressed as:

$$EAC = \frac{NPV}{1-(1+r)^{t}} \tag{3.7}$$

where *NPV* is the net present value of capital expenditures, *r* is the discount rate and *t* is the asset's lifespan in years (Adair, 2005).

3.3.3 Environmental Quality Parameter

The environmental quality parameter (λ) represents the number of times of a drinking water quality threshold is surpassed under a given forest management scenario. The forest management scenarios I used in the empirical analysis are described in Chapter 4. Using each forest management scenario's simulated water quality time series, I estimated the number of times per week a set water quality threshold was surpassed in the model, which generated a count data set. The probability of surpassing a set threshold in a given week was approximated using the Poisson distribution. The distribution implies that the data are individually independent and the mean and variance are equal. If overdispersion is present in the data, the Negative Binomial distribution should be used instead.[7] To determine the presence of overdispersion, I used the Pearson's Chi-Square test. Furthermore, I estimated λ using the maximum likelihood method (Crawley, 2007).

Under the assumption of a Poisson distribution, the statement for the probability that a certain number of water quality events will occur during a time period of one week is expressed as:

$$\Pr ob(K_t = k_t) = \frac{e^{-\lambda t} \lambda_t^{kt}}{k_t!}, k_t = 0,1,2,... \tag{3.8}$$

where K is a random variable (i.e. water quality readings surpassing the threshold) and k_t is the observed count of events in a given period (t). Note the restriction that k_i is a non-negative count variable (Ricci, 2005). Estimating λ using equation (3.8) allowed me to determine how changes in the probability of surpassing a water quality threshold, due to the timber harvest, affects the AC curve.

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[7] Overdispersion occurs when the mean is not equal to the variance (Crawley, 2007)

3.4 Additional Data Sources

To apply the methodology presented above, the quantity of treated water outflow (Q) and the price of treated water (P) data was required. The City of Abbotsford's Engineering Department supplied the treated water outflow data for 2008 (personal communication, Kristi Alexander, January 2010). To estimate price, the City of Abbotsford supplied their 2008-2012 Financial Plan (City of Abbotsford, 2008) and the District of Mission supplied their 2008 Annual Report (District of Mission, 2008). To estimate the environmental quality parameter, the British Columbia Ministry of Environment supplied turbidity data (personal communication, Jennifer Guay, July 2008) and the Water Survey of Canada supplied stream discharge data (detailed further in Chapter 4) (Water Survey of Canada, 2008). A full parameter glossary is presented in Chapter 5.

4: WATER QUALITY TIME SERIES SIMULATION

This chapter details the water quality time series simulation conducted for each alternative forest management scenario. Each simulated time series represents each scenario's ecological impact on water quality and was used to estimate the environmental quality change (λ) per scenario required in the economic model described in Chapter 3. I begin the chapter with an overview of the required data and raw data description. Then, I describe a three-step approach used to simulate each scenario's water quality time series. Step 1 simulated a baseline time series. Step 2 estimated each scenario's annual fine sediment yield (AFSY) generated by forest roads. Lastly, step 3 estimated each scenario's sedimentation impact on the initial water quality time series. A schematic of the approach is presented in Appendix B.[8]

4.1 Environmental Data Requirement

Water quality is determined by the amount of suspended sediments within a body of water. In most cases, suspended sediments are composed of fine sand, silt, and clay sized particles with diameters of less than 0.2 millimetres (mm) (Gomi et al., 2005). Water quality analysts measure fine sediments as either suspended sediment concentration (SSC) or as turbidity. SSC are usually expressed in milligrams of sediment per litre of water (mg/L) and require extensive stream sampling and time-consuming lab analysis. Turbidity is expressed as absolute Nephelometric Turbidity Units (NTU) and is

[8] I would like to thank Dr. Andrew Cooper, Dr. Duncan Knowler, and Andres Araujo for their significant contribution to this chapter.

measured instantly by a turbidity sensor that gauges how suspended sediment particles obstruct light transmission (Pfannkuche & Schmidt, 2003).

Municipal water utilities primarily use turbidity measurements to assess water quality. Utility management actions are directly linked to the quality of its source water supply and therefore managers require a continuous and immediate reading. For example, poor water quality at the NCCW intake will shut down the utility if it exceeds what the utility's water filters can handle (personal communication, Derrick Casey, February 2010). The City of Abbotsford and District of Mission must then rely on its backup sources for water supply. Therefore, due to the need for immediacy, turbidity measurements are preferred over SSC.

4.1.1 Raw Data

The British Columbia Ministry of Environment supplied an incomplete baseline turbidity time series. QA Environmental Consulting installed a turbidity probe at the NCCW utility's water intake in February 1998 and collected data until March 2002. The probe recorded data every 15 minutes 24 hours per day. However, the probe often malfunctioned or returned erroneous readings likely caused by debris around the optic window (QA Environmental Consulting, 2002, p21).

To provide a more realistic representation, I corrected the raw data by eliminating NTU values greater than 200 and less than or equal to 0 (personal communication, Jennifer Guay, July 2008). Furthermore, I grouped the 15-minute interval data into daily averages as such values are deemed most suitable for assessing the effects of different forest management actions on stream water quality (Chapman Geoscience Ltd., 2000;

Greater Vancouver Regional District, 1999). However, the data series remained incomplete, with missing values for average daily turbidity throughout the period 1998-2002 (Figure 5).

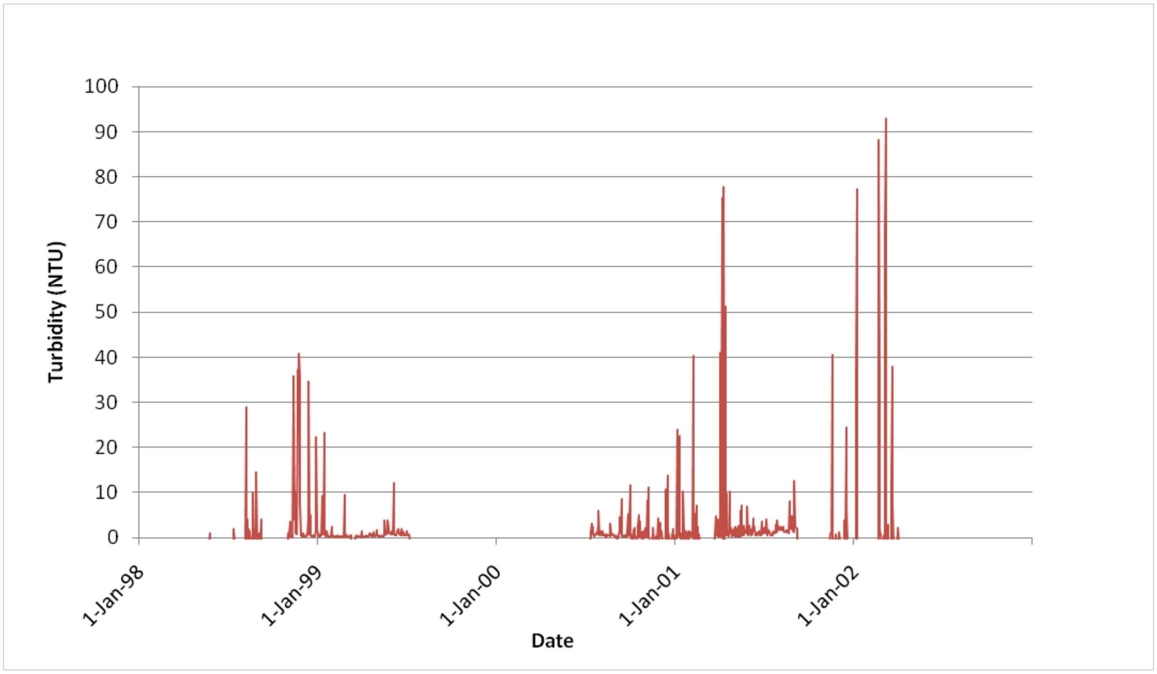

Figure 5: Norrish Creek Community Watershed daily average turbidity data (NTU)

To simulate a complete baseline time series, I required discharge data. The Water Survey of Canada provided continuous (hourly) discharge readings above Rose Creek upstream of the intake weir from 1984 to 2006. These data were available as daily averages measured in meters cubed per second (m^3/sec) (Figure 6) (Water Survey of Canada, 2008).

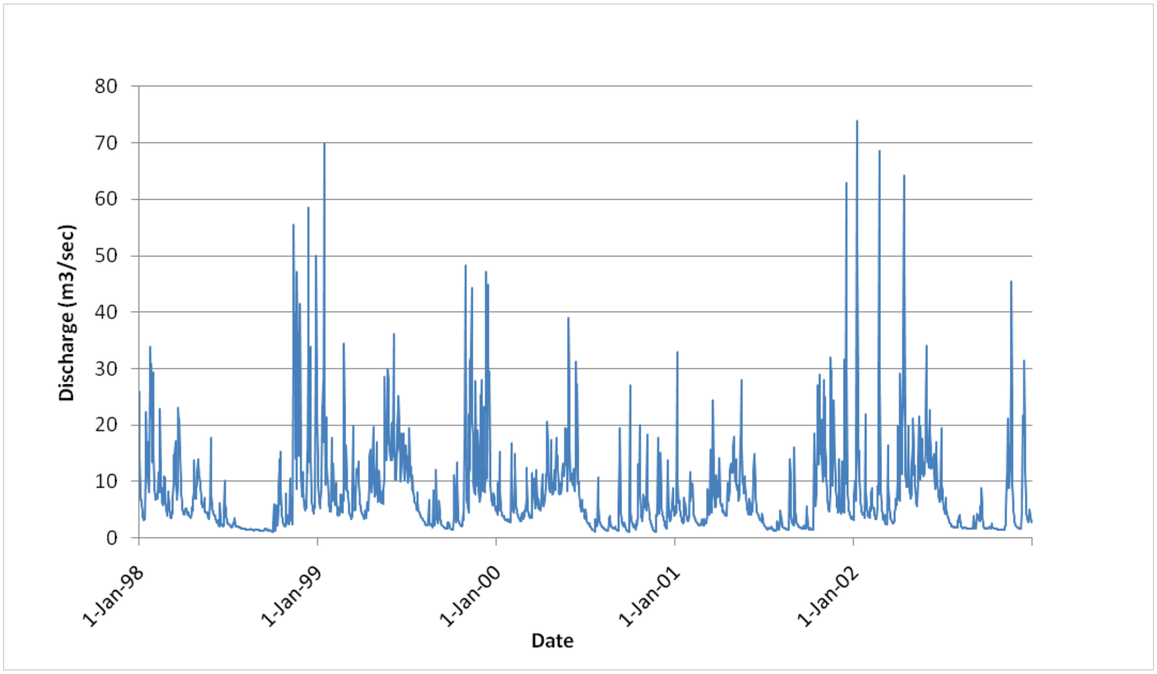

Figure 6: Norrish Creek Community Watershed daily average discharge data (m^3/sec)

4.1.2 Converting Turbidity to Suspended Sediment Concentration

The 3-step simulation that follows used SSC as the unit of water quality measurement.[9] A turbidity-SSC relationship was used to convert turbidity to SSC. Upon completion of the simulation, the SSC time series was converted back to turbidity for further analysis.

The literature suggests the most appropriate way to generate a turbidity – SSC relationship is by taking samples of both variables over a common range of water discharges and conducting a regression analysis (Birtwell et al., 2008). However, this study is a modelling exercise and thus field data was not collected. Instead, the turbidity – SSC relationship developed by Carson (2002) was used. He estimated that for an average

[9] SSC was used because of Step 2, which used Metro Vancouver's AFSY model to estimate sediment yield (measured in tonnes) generated by forest roads.

stream with turbid water that 1 NTU can be considered equivalent to 2 mg/L of suspended sediment, giving the following relationship:

$$SSC = \frac{Turbidity}{0.5} \tag{4.1}$$

The relationship was applicable to this study for two primary reasons. First Carson (2002) specifically defined suspended sediment as the fine sediment produced from surface road erosion. Second, the author used a wide range of samples from streams within coastal BC to construct the relationship (Carson, 2002; pg 10). However, it must be noted that Carson provides the disclaimer that the relationship is a gross assumption but “for order of magnitude estimations, these assumptions are reasonable and valid” (Carson, 2002; pg 10).

4.2 Step 1: Estimate Initial Water Quality Time Series

4.2.1 Approach

To simulate a complete water quality time series, a SSC-discharge (D) relationship was used. The SSC-D relationship is often called the suspended sediment-rating curve (Gomi et al., 2005; Horowitz, 2003). Regressions of the suspended sediment rating curve have been commonly used in hydrology, sedimentology, and natural resource management (De Vries & Klavers, 1994; Phillips et al., 1999; Walling & Webb, 1981). The most commonly used relation is the power law function:

$$SCC = aD^b \tag{4.2}$$

where D is discharge (m^3/s) and a and b are parameters (Ferguson, 1986; Ferguson, 1987; Porterfield, 1977).

Suspended sediment rating curves can estimate fairly accurately SSC (the dependent variable) from *D* (the independent variable) over shorter periods, but it becomes problematic when applying it to longer time series involving many years (Walling & Webb, 1988). As the discharge data supplied ranged from 1984 to 2006, there was the possibility of under predicting real concentrations due to scatter in the data and seasonal variation in the SSC-D relationship (Asselman, 2000; Fregusson, 1986; Walling & Webb, 1988). To compensate for such limitations, I used a linear mixed-effects model (Pinheiro & Bates, 2000). Using a linear mixed-effects model more accurately captures the SSC-D relationship, while accounting for inter-annual variability characteristics of hydrological time series data.

In this step, the procedure involved interpolating SSC using D as a proxy for the periods where SSC data did not exist. The resulting simulated baseline time series served as the base to evaluate the effects of forest roads on sediment generation.

The Basics of Linear Mixed-Effects Models

In this study, I employed 12 linear mixed-effects models, one for each month across all years in the data set. The reason for dividing the data set in this manner was to account for seasonal variability from month to month while acknowledging variability from year to year.[10] A linear mixed-effects model includes both fixed and random effects of the data on the fitted model. Fixed effects are associated with the variability between groups over the entire population, and the grouping is decided by the modeller (Pinheiro & Bates, 2000). The fixed effects in this study were related to the months across all years

[10] The 12 linear mixed effects models should captured seasonality well as the NCCW has clearly defined discharge regimes with abundant discharge during winter and spring months and lower discharge in the late summer and fall (Brayshaw, 1997).

in the data set (e.g. January 1998, January 1999...January 2002). Random effects are not chosen by the modeller and are associated with the variability within groups (Pinheiro & Bates, 2000). In this study, random effects were related to the variability within each month in each year (i.e. January 1998).

4.2.2 Model Formulation and Parameter Estimation

In a linear mixed-effects model applied to sedimentation, the experimental units are groups within which the observations of SSC and D are made. There are two levels of variation in the time series of SSC: groups (same month in different years) and observations nested within groups (individual SSC and D measurements in each month). Observations between levels are independent, but observations within each level are correlated because they belong to the same sub-population of months (Pinheiro & Bates, 2000).

To formulate the linear mixed effects model, I estimated different equations for the two data levels (Lai & Helser, 2004; Singer, 1996; Snijders & Bosker, 1999). The first level described the $\log_e$ transformed SSC-D (+1) relationship for each month across all of the j years where i represents the observations within each month. The data transformation was necessary to ensure normality. I used the following relationship:

$$y_{ij} = \beta_{0j} + \beta_{1j}x_{ij} + \varepsilon_{ij} \qquad \varepsilon \sim N(0,\sigma^2) \qquad (4.3)$$

where y= $\log_e$ (SSC+1), x= $\log_e$ (D+1), β_0 and β_1 are the intercept and slope respectively. The random error (ε) represents the within-month variance across groups and it is assumed to be independent and identically normally distributed with zero mean and common variance σ^2.

The second level described the variability between the same months in different years (e.g. sub-population of all Januaries in the data set). The intercept (β_{0j}) and slope (β_{1j}) were assumed to be multivariate normally distributed with mean (β_0, β_1) and covariance matrix ξ:

$$\beta_{0j} = \beta_0 + b_{0j}, \quad \beta_{1j} = \beta_1 + b_{1j} \tag{4.4}$$

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

The covariance matrix represents the between-group variance (fixed effects) and covariance for the vector of random effects b_j, which describes the variation of the SSC-D relationship in each sub-population. Substitution of equation 4.4 into equation 4.3 leads to the linear mixed-effects model:

$$y_{ij} = \beta_0 + b_{0j} + \beta_1 x_{ij} + b_1 x_{ij} + \varepsilon_{ij,} \quad \varepsilon \sim N(0, \sigma^2) \tag{4.5}$$

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

In equation 4.5, the normally distributed random variables b_{0j} and b_{1j} represent the group effects and incorporate the variability among the same month in different years. The fixed-effects coefficients β_0 and β_1 in equation 4.5 are frequently referred to as the population averages (Lai & Helser, 2004), and were used to interpolate SSC using discharge as a proxy for the periods where only discharge was available. The random errors (ε), the standard deviation of random effects for the sub-population averages (b_{0j} and b_{1j}) and the standard error of the population averages (β_0 and β_1) were not used in the interpolation procedure, but can be incorporated in future research to estimate variability and uncertainty in the predicted time series. Specifically, the random error and standard deviation of the random effects for the sub-population averages can be used to estimate variability, and the standard error of the population averages can be used to estimate uncertainty. Furthermore, the parameters of the covariance matrix (ξ) in the linear mixed-effects model were estimated using the restricted maximum likelihood method (Corbeil & Searle, 1976).

4.3 Step 2: Estimate Annual Fine Sediment Yield Generated by Forest Roads

4.3.1 Background

Paired-basin studies and sediment budgets are two primary methods used to determine sediment yields within a watershed (Reid et al., 1981). The paired-basin approach measures SSC in paired control and treatment watersheds. This method uses the measured sediment yields as an index of sediment production based on the land-use within the specific watershed. However, many limitations exist, primarily the inability to isolate individual sediment sources and assign them to natural disturbances and/or individual land-use activities. For example, if a major storm or landslide occurred, the paired-basin approach would not be able to distinguish the event from a land-use activity (Reid et al., 1981). To account for such limitations, a sediment budget approach can be used to identify the cause of sediment generation and therefore the source's long-term impact on water quality (Reid et al., 1981). This study used a sediment budget approach.

An Adapted Approach

I adapted a sediment budget study previously conducted by Metro Vancouver to estimate the AFSY from surface road erosion. Metro Vancouver used the results from Reid (1981) to estimate the AFSY from all roads in the Capilano Watershed, located in the Fraser TSA. The model estimated the AFSY under current conditions and predicted the potential contribution under different levels of road use and aggregate road length. The model was particularly useful because its predicted average AFSY only included grain sizes (i.e. fines) that caused turbidity and excluded all other sediment material such as coarse-grained materials (i.e. sand, pebbles, cobble and boulders) (Greater Vancouver Regional District, 1999; personal communication, Dave Dunkley, June 2009).

Metro Vancouver first conducted a road sedimentation study in the Seymour Watershed. The study required two years of field samples and lab analysis. The researchers obtained similar fine sediment yield results to those in Reid (1981) and Reid and Dunne (1984). Due to the similarities in the AFSY results, Metro Vancouver adapted the results of Reid (1981) to predict the AFSY for other traffic volumes in the Capilano Watershed (Greater Vancouver Regional District, 1999).

Due to the research effort and time limitations required to gather field data, I limited the scope of this section to simulation modelling. Therefore, I adapted the Metro Vancouver methodology to predict the AFSY from surface road erosion in the NCCW, at the watershed scale. The Capilano Watershed and NCCW exhibited similar watershed characteristics (Table 2) with respect to sediment generation. Therefore, adapting the methodology to the NCCW was deemed appropriate (personal communication, Dave Dunkley, June 2009).

Table 2: Comparison of the Norrish Creek Community Watershed to the Capilano Watershed

Watershed Characteristic	Capilano Watershed	Norrish Creek Community Watershed
Climate	- Maritime	- Maritime (4)
Average Annual Precipitation	- Lower Elevation = 3,159 mm - Higher Elevation = 4,500 mm (1)	- Lower Elevation = 1,860 mm - Higher Elevation = 3,490 mm (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 coarse-grained granitic rock overlain with colluvium and till (4)
Elevation	- Pacific Ranges of the Coast Mountains (2) - 155 m to 1725 m (1)	- Pacific Ranges of the Coast Mountains - 250 m to 1420 m (4)
Biogeoclimatic Ecosystem Classification	- Coastal Western Hemlock Zone (CWH) and the Mountain Hemlock Zone (MH), with a very small ara of the Alpine Tundra Zone (AT) (3)	- Coastal Western Hemlock Zone (4)

Sources: (1) Greater Vancouver Regional District, 1999; (2) Brardinoni et al., 2003; (3) B.A. Blackwell and Associates Ltd., 1999; (4) Chapman Geoscience Ltd., 2000

4.3.2 Model Formulation and Parameter Estimation

I adopted the parameters used in the Metro Vancouver model to estimate the AFSY per kilometre of road in the NCCW (Table 3). The bolded parameter values/descriptions are those specific to the NCCW for the year 2008 (i.e. baseline year for further economic analysis). Such values/descriptions are all found on average within the watershed and were selected using the best available information from available watershed reports.

Table 3: Metro Vancouver's annual fine sediment yield model parameter values and descriptions

Parameter 1: Slope of Roads	Parameter 2: Slope of Terrain	Parameter 3: Number of Streams	Parameter 4: Level of Road Use[11]
- 0-2.5 degrees - 2.5 to 7.5 degrees - 7.5 to 12.5 degrees - Greater than 12.5 degrees	- Less than 27% **- 28% to 49%** - Greater than 50%	- No ephemeral streams or major streams - One ephemeral stream but no major streams **- One major stream or more than one ephemeral stream**	- Light **- Moderate** - Heavy

Source: (Reid et al., 1981)

To estimate the AFSY per kilometre of road, the following relationship was used:

$$AFSY_n = l_n u_n LR_n DR_n \tag{4.6}$$

where $AFSY_n$ is the estimated annual fine sediment yield (tonnes/km/yr) for a given road network (n), l_n is the length of road segment (km), u_n is the sediment yield (tonnes/km/yr) based on the road segment's use level and slope, LR_n is the loss ratio and DR_n is the delivery ratio.

The loss ratio reflects diversions of sediment-laden water by obstructions in the path of flow. Reid (1981) measured the total sediment concentration at specific road culverts under different road use levels and adjusted the results to varying slopes using the Universal Soil Loss Equation (Wischmeier & Smith, 1965). The author then applied the final concentration-discharge data to predicted hydrographs (determined from precipitation data) to establish the annual total sediment yield. In the Seymour Watershed

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[11] Light: used by light vehicles only. Moderate: carry fewer than 4 logging trucks per day. Heavy: carry more than 4 logging trucks per day (Reid et al., 1981).

study, fine sediments constituted roughly 50% of the total load. Thus, Metro Vancouver adjusted Reid (1981)'s results by this factor to predict the AFSY generated by forest roads (Table 4) (Greater Vancouver Regional District, 1999).

Table 4: Metro Vancouver's annual fine sediment yield generated by forest roads based on road slope and road use level (tonnes/km/yr)

Road Slope	Light Use	Moderate Use	Heavy Use	Loss Ratio
	(tonnes/km/year)			
0 to 2.5E	0.2	2.1	25	0.7
2.5E to 7.5E	1.9	21	250	0.8
7.5E to 12.5E	5.1	57	675	0.9
>12.5E	10.5	105	1250	1.0

Source: (Reid et al., 1981)

The delivery ratio accounts for the actual portion of fine sediment that potentially enters a stream or river. Unlike sediment delivery from landslides or stream erosion, which is assumed to be 1.0 where the total mass ends up in the stream, surface road erosion is not necessarily carried (i.e. delivered) to a stream. For this study, I adapted the delivery ratios from the fine sediment delivery model used in the Metro Vancouver study (Table 5) (Greater Vancouver Regional District, 1999, prepared by June Ryder Associates Terrain Analysis).

Table 5: Metro Vancouver's annual fine sediment yield model delivery ratios

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	1.0

Source: (Reid et al., 1981)

4.4 Step 3: Estimate Forest Road Impact on Water Quality

4.4.1 Background

Forestry operations, road construction, use and decommissioning all influence peak discharge and the volume of sediment entering rivers and streams (Benda et al., 2005; Borga, 2004; Bosch & Hewlett, 1982; Jones & Grant, 1996; Macdonald et al., 2003; Rothacher, 1973; Wemple, 2003). Increases in peak discharge result in more water available to erode stream banks and mobilize sediments into streams. The removal of the forestry canopy creates more surfaces with the ability to collect snow, increasing the rates of snowmelt and modifying the runoff corridors by which water flows to the stream channel (Moore & Wondzell, 2005).

The aggregate length of roads in a watershed can influence the peak and volume of flows in several ways. Road ditches and cut-slopes alter natural sub-surface flow by interrupting it and transferring it to the surface. Surface flow reaches streams faster than sub-surface flow, and even though some road networks may actually divert water from streams, in most cases this flow will end in the nearest downhill stream (Wemple et al., 1996). Thus, roads speed up the delivery of water and sediments into streams, especially during periods of high precipitation (Jones & Grant, 1996). Road surfaces also permanently remove a portion of the forest canopy altering water and snow interception, evapotranspiration, and snowmelt processes, changing the watershed's natural hydrologic processes (Jones, 2000).

Forest Management Scenarios

To evaluate forest road induced changes in the peak and volume of suspended sediments in the NCCW, I used the individualized road construction profiles of three forest management scenarios detailed by Knowler and Dust (2008) for the Fraser Timber Supply Area (TSA) (Table 6).

Table 6: Forest management scenario descriptions

Scenario	**Description**
	Status Quo Scenario
SompCurr	67% of productive forests within LTACs are maintained as 100 years or older[12].
	Conservation-Oriented Scenarios
Suit100	100% of currently suitable Spotted Owl habitat is removed from the timber harvesting landbase.
Terr100	100% of packed territories are removed from the timber harvesting landbase regardless of whether or not the stands are currently suitable for Spotted Owl habitat[13].

Source: (Knowler & Dust, 2008)

Knowler and Dust (2008) indirectly used the road construction profiles. The authors estimated the opportunity cost of preserving old growth habitat by comparing two conservation-oriented scenarios (Suit100 and Terr100) to a status quo scenario

[12] Long term activity centres are defined as areas where spotted owls have historically been active (Knowler and Dust, 2008)

[13] Packed territories are defined as contiguous areas of habitat regardless of being suitable spotted owl habitat (Knowler and Dust, 2008)

(SOMPcurr). Specifically, the authors estimated the net present value of various forest services over a 100-year time frame. Timber harvesting was one of those services and required a 100-year road construction profile (i.e. length of new built roads required per year) in order to determine the amount of timber that could be harvested.

Each road construction profile applied to specific Landscape Units located within the Fraser TSA. The NCCW is roughly 10% of the Hatzic Landscape Unit. Therefore, I estimated the road construction output per year as 10% of the original amount shown for the Hatzic Landscape Unit (Figure 7).

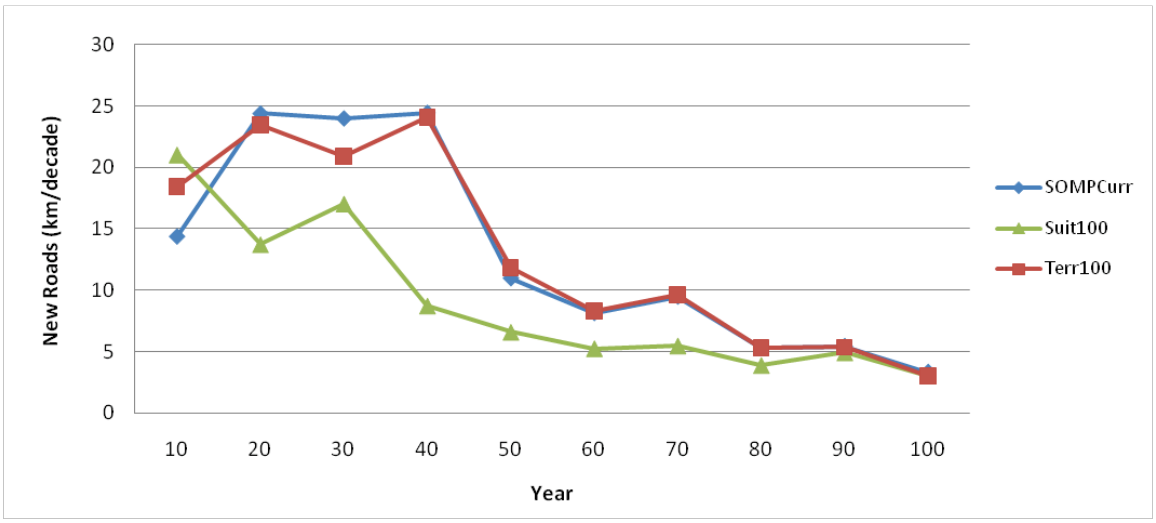


Figure 7: Norrish Creek Community Watershed 100-Year road construction profile

It must be noted that even though Terr100 is a conservation-oriented scenario in terms of volume logged at the scale of the entire Fraser Timber Supply Area (TSA), it presented a non-conservation-oriented scenario road profile within the NCCW. More roads were built within the 100-year time frame compared to the status quo. The reason for this occurrence was likely the remoteness and terrain associated with road building needed to access stands designated acceptable to harvest in place of reduced harvesting of more easily accessible old growth elsewhere in the TSA.

Simulating New Time Series

Using the forest management scenarios presented above, I developed a model to further address forest road induced changes in the peak and volume of suspended sediments. The model simulated changes in the baseline time series (i.e. the time series generated by the linear mixed effects model) by accounting for changes in the frequency of peak events and the volume of the total suspended sediments resulting from each alternative forest management scenarios 100-year road profile. The model accounted for these changes by applying the same transformation to the entire time series. Therefore, the model did not attempt to capture dynamic sedimentation processes over time as it was outside the scope of this study.

4.4.2 Model Formulation and Parameter Estimation

Daily Sediment Loads

To estimate the total daily load of suspended sediments Γ_t (tonnes/day), I used the following equation:

$$\Gamma_t = SSC_t D_t 10^{-9} \tag{4.7}$$

where SCC_t is the baseline time series produced from the linear mixed effects model (mg/L), and D_t is the discharge data measured in litres per day (L/day) (Colby, 1956). The total suspended sediment yield, Γ, measured in tonnes is therefore defined as:

$$\Gamma = \sum_{t=1}^{n} \Gamma_t \tag{4.8}$$

Error associated with summation procedures can be as small as 5% for large rivers with relatively high concentrations of suspended sediments to a very large percentage for streams with undefined seasonal sedimentation patterns and poor discharge records (Walling, 1977). However, the NCCW presented a defined seasonal trend in suspended sediments with accurate discharge records, so it can be expected that the error would be small (Brayshaw, 1997).

Separating Natural and Road Generated SSC

To separate the natural daily suspended sediment loads N_t (tonnes/day) from those generated by roads R_t (tonnes/day), I first estimated the total current suspended sediment yield generated by roads (R) measured in tonnes. I determined R by multiplying the AFSY (i.e. equation 4.6) by the total kilometres of forest roads. Second, I used the total sediment yield Γ from (4.8), which incorporated both natural and road generated sediment yields, to solve for N_t:

$$N_t = \Gamma_t (1 - \frac{R}{\Gamma}) \tag{4.9}$$

$$R_t = \Gamma_t - N_t \tag{4.10}$$

Simulating SSC due to Changes in Roads

To simulate daily SSC due to changes in forest roads, I first applied equation 4.11 to values of R_t:

$$f_t = e^{\rho \ln(R_t + 1) - 1} \tag{4.11}$$

where f_t represents the daily value of suspended sediment (tonnes/day) influenced by changes in the peak discharge for given aggregate road lengths, and ρ is the peak suspended sediment parameter. The parameter establishes the frequency of sedimentation events (i.e. peaks) given different aggregate road lenghts.

I then multiplied standardized values of f_t (f_t^*) by the road suspended sediment yield time series (R) and the volume parameter (ϕ), to obtain the simulated daily suspended sediments generated by roads (tonnes/day)

$$Rsim_t = \phi(f_t^* R_t) \tag{4.12}$$

where ϕ determines the change in volume of suspended sediments per day for a determined increase in the aggregate length and use of roads.

Lastly, I estimated the final simulated time series ($Fsim_t$) using equation (4.13):

$$Fsim_t = Rsim_t + N_t \tag{4.13}$$

where $Fsim_t$ is the final simulated time series given the amount of daily suspended sediment in the watershed due to changes in the aggregate length of roads and use of those roads (tonnes/day). In order to obtain the simulated time series in suspended sediment concentration units (mg/L), I applied the inverse summation procedure used to calculate daily loads (Colby, 1956).

Parameter Estimation

As stated previously, each forest management scenario has a 100-year road construction profile. I used each profile to estimate the peak and volume parameters, and then applied the parameters to the simulated 22-year water quality time series. To do so, I estimated the parameter values in a way that represents the amount and use of roads that are likely to exist at any given time over the 100-year profile (Appendix C). The approach assumes no temporal trends (i.e. effects from climate change) in the 100-year profile. Therefore, the water quality time series represents any 22 consecutive years within the 100-year road construction profile.[14] The following provides further detail.

Parameter ρ (Peak)

A broad range of hydrological research during the past 20 years describes changes in the peak discharge and peak suspended sediments after forestry operations such as road construction and timber harvesting (Grant et al., 1990; Jones, 2000; Jones & Grant, 1996). It is difficult to quantify peak suspended sediment due to the level of monitoring resources needed (Lewis et al., 2001). However, the frequency of peak suspended sediments given road densities can be approximated.

To estimate the peak parameter, I first estimated the percent increase (δ) in road density for each forest management scenario's 100-year road profile over the baseline year (i.e. 2008). I then used the following equation to estimate the peak parameter:

$$\rho = \frac{100 - \delta}{100} \tag{4.14}$$

where ρ is the peak parameter and δ is the percent increase in road density over the baseline year. When applied to the water quality time series (i.e. ρ values are applied to R_t.), the time series became more or less "peakier" in terms of frequency.

[14] A more involved analysis of the time series data, such as checking for stationarity, was beyond the scope of this study.

Parameter ϕ (Volume)

It is now well documented that erosion from forest roads has the potential to significantly increase the volume of sediments entering a watercourse (Beschta, 1978; Keppeler et al., 2003 Reid & Dunne, 1984). Therefore, I accounted for an increase in sediment production in the model with the volume parameter ϕ. Specifically, the volume parameter is the amount of fine sediment (tonnes/yr) that enters a water system from roads as a result of a change in road use intensity. The following relationship was used:

$$\phi = \frac{AFSY_{roaduse}}{AFSY_{baseline}} \tag{4.15}$$

where $AFSY_{roaduse}$ is each forest management scenario's AFSY generated from light, moderate and heavy use, and $AFSY_{baseline}$ is the AFSY generated from the baseline year's condition (i.e. Table 3). When applied to the water quality time series, the time series shifted up or down, representing an increase or decrease in the amount of fine sediments entering a watercourse.

4.5 Summary

In summary, this chapter detailed the methodology used to simulate each forest management scenario's water quality time series. First, I estimated a baseline water quality time series using a linear mixed effects model. Second, I estimated the AFSY generated by forest roads for each forest management scenario. Lastly, I estimated the impact of forest road sedimentation on the baseline water quality time series, thus producing individualized time series per forest management scenario. The following chapter documents the empirical model estimates and calculations.

5: MODEL ESTIMATION

This chapter details the integrated economic-ecological model's empirical estimations and calculations. I begin by documenting the water quality time series empirical estimates for each step in the time series simulation. Then, I document the economic model estimates, starting with the total water supply cost estimates for the baseline year (2008), followed by each forest management scenario's total water supply costs and resulting equilibrium water price and quantity estimates. I conclude with a summary of all parameters.

5.1 Estimating the Water Quality Time Series

As stated previously, the ecological component of the integrated economic-ecological model consisted of the water quality time series simulation. Below I document the model estimates from the simulation.

5.1.1 Step 1

In Step 1, I estimated 12 linear mixed-effects models that were used to simulate a baseline water quality time series. The parameters estimated for the model consisted of the intercept (β_0) and slope (β_1) for each sub-population, the standard deviation of the intercepts and slopes, and a residual (ε) (Table 7). The standard deviation of the intercepts and slopes indicate the variability in the distribution of monthly intercepts and slopes with respect to the estimated values of β_0 and β_1. The residual (ε), is an additive value of the difference between the observed data points and the predicted values within each month of the year across all years in the time series. As stated previously, these parameter estimates, along with the standard error of the population averages (β_0 and β_1), were not used in the interpolation procedure, but can be used in future research to incorporate variability and uncertainty in the predicted time series.

Table 7: Linear mixed-effects model estimates used to simulate suspended sediment concentration

Month	Intercept (β_0)	Slope (β_1)	SD of random intercept (b_0)	SD of random slope (b_1)	Residual (ε)	AIC
Jan	-0.15851	0.91842	1.367036	0.132734	0.801894	152.788
Feb	0.021034	0.413344	1.595821	0.546114	0.638607	97.63078
Mar	-0.41553	0.539537	0.685596	0.438108	0.333867	38.8978
Apr	-0.85935	0.718301	0.280123	0.169832	0.214536	5.326808
May	-0.79954	0.724295	0.81826	0.259825	0.473747	99.26051
Jun	-0.71993	0.73373	1.039823	0.278582	0.215316	7.867856
Jul	0.553156	0.622331	3.01E-05	9.20E-06	0.430465	71.9091
Aug	0.903857	0.658868	0.729081	5.55E-05	0.414457	98.53376
Sep	0.323839	0.796021	0.289861	1.63E-05	0.637658	86.80204
Oct	-0.03403	0.49252	7.42E-06	9.70E-09	0.576599	51.9432
Nov	-1.41542	1.324974	3.92E-05	6.30E-19	0.949771	143.2708
Dec	-1.92792	1.308752	0.143444	2.04E-05	0.488181	81.76644

The negative values in the intercept (β_0) produced negative SSC when interpolated at lower discharge values. As negative values are not possible, I approximated the resulting interpolated negative values to 0. The approximation did not affect further analysis as lower SSC (i.e. turbidity once converted for further analysis), does not have significant impacts on drinking water quality (personal communication, Derrick Casey, February 2010).

To test whether a suspended sediment rating curve (represented by a generalized least squares regression) or a linear mixed-effects model should be used to represent the SSC-D relationship, I compared each model's goodness of fit using the Akaike Information Criterion (AIC). The AIC for the linear mixed-effects model presented lower scores (i.e. better goodness of fit) for the majority of months and was therefore the preferred option. Additionally, the linear mixed-effects model better captured the trend in the observed values. When comparing the observed values in the raw data with the predicted values estimated from the linear mixed-effects model, substantial dispersion occurs around the 1:1 line (pseudo-R-squared = 0.125). The main reasons for observing this dispersion are the highly variable SSC-D relationship, which was affected by seasonal factors, and the observation error present in the raw data. Despite the low pseudo R-squared, the linear mixed-effects model is a better predictor than the usual approach (a suspended sediment rating curve) which not only gives higher AIC scores but also a lower R-squared (pseudo-R-squared = 0.106).

Using the estimated linear mixed-effects model estimates for each month, I simulated a complete baseline water quality time series (Figure 8). The raw water quality time series consisted of few extreme events (i.e. > 80 NTU) and thus extreme events tended to be under-represented in the interpolated periods. However, underestimation did not impact further analysis because this study was concerned with all events greater than 10 NTU (i.e. drinking water quality threshold).

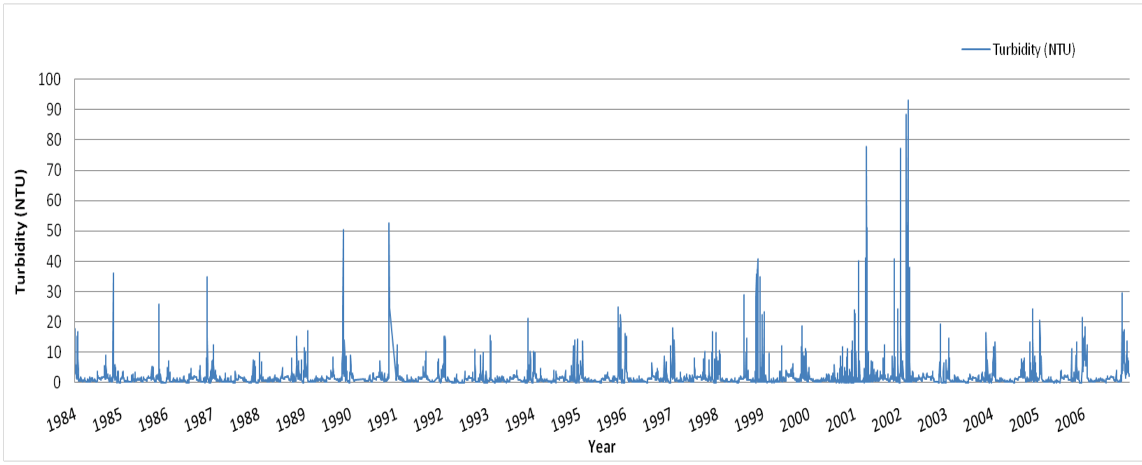


Figure 8: The Norrish Creek Community Watershed's simulated baseline water quality time series generated by 12 linear mixed-effects models (NTU)

The 12 linear mixed-effects models captured seasonality well enough, as is evident from an expanded analysis for the single year 2006 (Figure 9). As Brayshaw (1997) indicated in his study, the NCCW demonstrated a defined discharge regime characterised by generally higher discharge in the winter and spring, and lower discharge in the late summer and fall. Below normal precipitation levels were experienced during the spring of 2006 contributing to lower spring turbidity levels (Environment Canada, 2010).

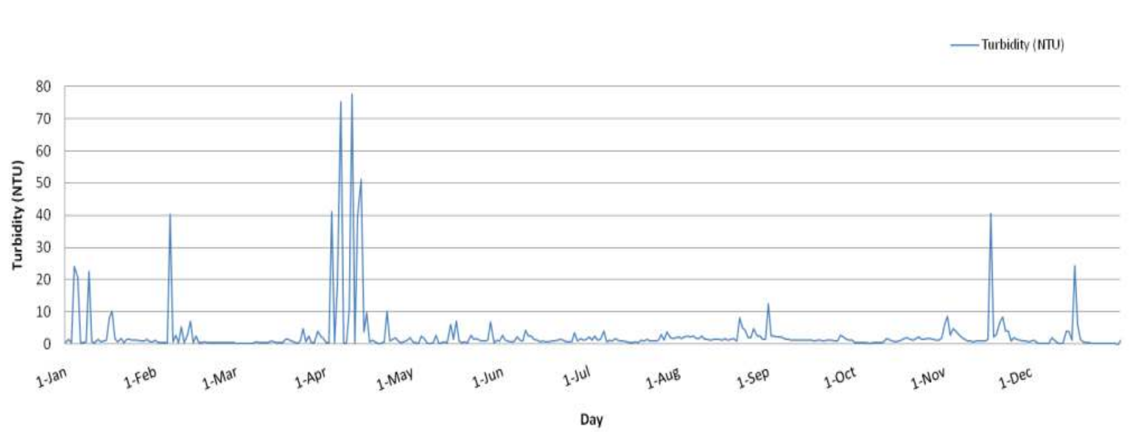


Figure 9: An expanded analysis (2006) from the Norrish Creek Community Watershed's simulated baseline water quality time series (NTU)

5.1.2 Step 2

Step 2 estimated the AFSY generated by forest roads in the NCCW. The aggregate length of roads and use of those roads drove the difference between each forest management scenario's AFSY. The aggregate length of roads in each forest management scenario was estimated using the EAC approach at a 1%, 4% and 7% decommissioning rate (Table 8) (Appendix C). A minimal variation between aggregate road lengths resulted across decommissioning rates. For example, at 1%, SOMPcurr contained 119.36 kilometres of roads compared to 122.41 kilometres at 7%. Slight variations in road length did not contribute to significant changes in AFSY (discussed below); therefore, I chose the aggregate amount of roads resulting from a 4% decommissioning rate for further analysis.

Table 8: Aggregate length of forest roads per forest management scenario in the Norrish Creek Community Watershed for varying decommissioning rates using the Equivalent Annual Cost Approach (km)

Scenario	EAC Road at 1% decommissioning rate	EAC Road at 4% decommissioning rate	EAC Road at 7% decommissioning rate
SOMPcurr	119.36	122.38	122.41
Suit100	111.66	117.53	120.68
Terr100	119.66	123.36	123.96

Both the intensity of road use and the aggregate length of roads contributed to the AFSY variation across scenarios. However, it was the level of road use that dominated the variation. For example, sedimentation levels increased significantly (approximately 1,400 tonnes) when comparing moderate use and heavy use. When comparing increases in aggregate road lengths, sedimentation levels increased only slightly (approximately 100 tonnes) (Figure 10). The sensitivity of road-induced sedimentation from road use compared to aggregate road length that resulted in this study is supported by the sedimentation literature documented in the background and literature review (Chapter 2) (Reid & Dunne, 1984; Wald, 1975).

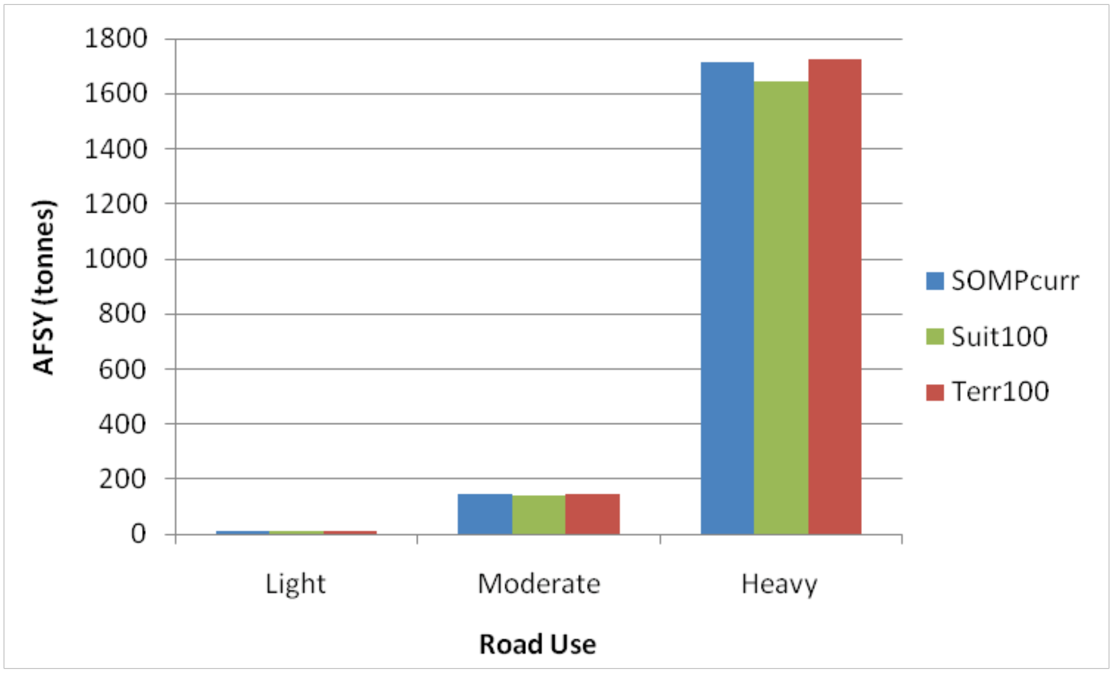


Figure 10: Annual fine sediment yield (AFSY) by forest management scenario and varying road use intensity (tonnes)

5.1.3 Step 3

Step 3 estimated the forest road impact on the baseline water quality time series. To do so required the estimation of a peak and a volume parameter (Table 9). The volume parameter shifted the time series up or down due to an increase or decrease in AFSY, while the peak parameter increased or decreased the frequency of peaks due to the existence of roads where there were no roads before (i.e. new sedimentation source). Note that a larger peak parameter value produces a less "peakier" time series.

Table 9: Peak and volume parameter estimates

Scenario	Road Use Level	Peak Parameter	Volume Parameter
SOMPcurr	Light	0.83	0.11
	Moderate	0.83	1.17
	Heavy	0.83	13.96
Suit100	Light	0.87	0.11
	Moderate	0.87	1.13
	Heavy	0.87	13.40
Terr100	Light	0.82	0.11
	Moderate	0.82	1.18
	Heavy	0.82	14.07

5.1.4 Forest Management Scenario Time Series

The water quality time series simulation produced individual time series reflecting the unique road characteristics of each forest management scenario (Figure 11). Each times series provides an important insight into the impact of road use intensity and aggregate road length on water quality, however many other road-induced sedimentation sources exist. For example, recreation on forest roads, road-induced landslides, and construction and decommissioning of roads all contribute (Carson, 2002; Gomi et al., 2005). Nevertheless, the simulated time series were used to estimate the frequency of surpassing a drinking water quality threshold described in the following section.

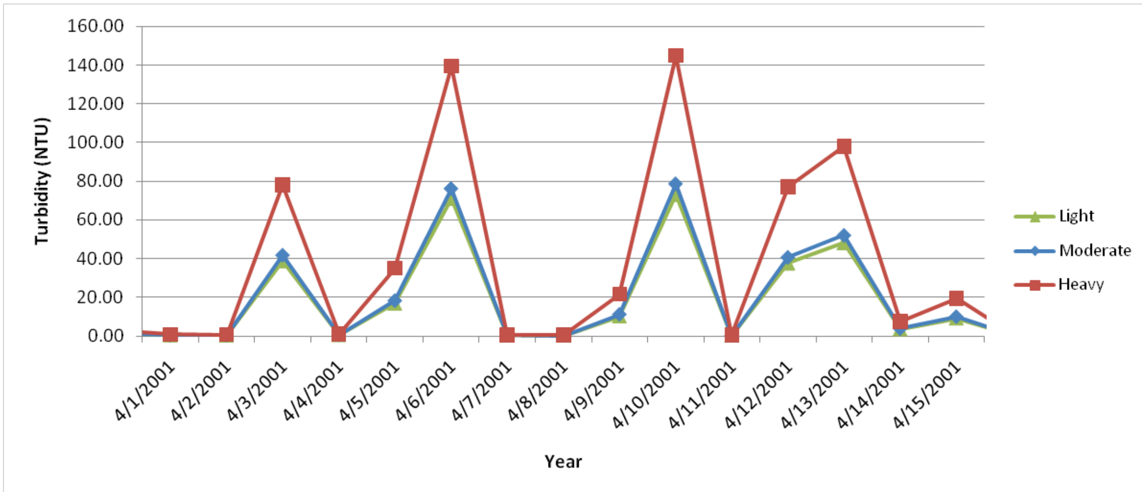


Figure 11: SOMPcurr turbidity time series for two weeks during a spring runoff event in 2001 (NTU)

5.2 Estimating the Economic Model

To estimate the change in welfare from one forest management scenario to another, equilibrium water prices and quantities must be known. Furthermore, to estimate price and quantity per forest management scenario, water supply costs must be known. The economic component of the integrated ecological-economic model estimated the water supply costs associated with exceeding a drinking water quality threshold. Below I document the model estimates and calculations.

5.2.1 Norrish Creek Community Watershed's Baseline Year Total Water Supply Costs

The NCCW supplies drinking water to the City of Abbotsford and District of Mission through one supply and distribution system. The water treatment plant is funded jointly, but the supply and distribution system is funded separately. Therefore, the City of Abbotsford and District of Mission price water separately, though both use an average cost pricing approach (personal communication, Kris Boland, May 2010; personal communication, Randy Millard, June 2010). The purpose of this study is to value the water purification/filtration service of the *entire* watershed; therefore, changes in price (required for welfare estimations; refer to Chapter 3) must be determined based on total system supply costs.

To estimate total supply costs, and thus price, I separated the fixed and variable costs for the City of Abbotsford, District of Mission and the Joint Water Supply System (Appendix D). It was necessary to separate the fixed and variable costs because of the water quality threshold analysis, which is presented in the next section. With respect to capital costs (i.e. a fixed cost), I estimated the EAC at a 1%, 4% and 7% discount rate.[15] Furthermore, treated water outflow data was needed to estimate average unit costs (i.e. price) from the total supply costs. The year 2008 was chosen for analysis due to the availability of treated water outflow for this year, which was set at 27,786,849 m^3 (i.e. equilibrium quantity for baseline year) (personal communication, Kristi Alexander, January 2010). The NCCW total water supply costs (from Table 10) were divided by the treated water outflow resulting in a calculated water price of $1.08 per m^3 at a 1% discount rate, $1.14 per m^3 at a 4% discount rate, and $1.19 per m^3 at a 7% discount rate (i.e. equilibrium prices for baseline year). These prices were then used as the starting prices to estimate the changes in price caused by the timber harvesting activities of each forest management scenario.

[15] Refer to Knowler and Dust (2008) for discussion on why 1%, 4% and 7% discount rates were used.

Table 10: Norrish Creek Community Watershed drinking water utility total water supply costs in 2008 ($)

Fixed Cost		Variable Cost	
Total @ 1%	**29,256,835**	**-**	**-**
Total @ 4%	**30,821,720**	**-**	**-**
Total @ 7%	**32,339,883**	***Total***	**798,058**

Source: (City of Abbotsford, 2008; District of Mission, 2008)

5.2.2 Estimating Total Water Supply Costs per Forest Management Scenario

Total supply costs is the sum of FC and VC. The VC per forest management scenario was estimated based on the probability and resulting costs of surpassing a drinking water quality threshold. Specifically, the NCCW system is brought offline once the water quality threshold is surpassed. Backup water sources, such as a lake (Cannell Lake) and groundwater wells, are then used to meet demand, effectively creating a business as usual (BAU) scenario and an emergency backup (EMERG) scenario (personal communication, Derrick Casey, February 2010). Therefore, the following expression was used to estimate VC:

$$VC = [C_{bau} + \lambda(C_{emerg} - C_{bau}) + VC_{other}]Q \tag{5.1}$$

where C_{bau} is the BAU cost per m^3 of treated water, C_{emerg} is the EMERG cost per m^3 of treated water, λ is the estimated average number of times per week the water quality threshold was surpassed, VC_{other} are all other variable costs, and Q is the average quantity supplied per week.

I estimated the average number of times per week the water quality threshold was surpassed using each forest management scenario's water quality time series. To do so, I counted the number of times per week the water quality threshold was surpassed, which generated a count data set. Using the count data set, I approximated the probability of surpassing a set threshold. To identify the correct model for the probability approximation, I tested the count data for overdispersion (refer to Chapter 3).

I used the Pearson's Chi-Square goodness-of-fit test for discrete distributions to determine if the data followed a Poisson distribution (Crawley, 2007). A statistically significant difference existed between the observed distribution and a Poisson distribution as indicated by the lower p-values (e.g. p-value <0.05). However, there was no compelling evidence to suggest the data did not follow a Negative Binomial distribution, as indicated by the higher p-values (e.g. p-value > 0.23) (Crawley, 2007). Therefore, I used the Negative Binomial distribution to estimate the mean times per week a drinking water quality threshold was surpassed per forest management scenario (Table 11).

Table 11: Poisson and negative binomial distribution probability estimates of the mean number of times a drinking water quality threshold is surpassed per week per forest management scenario (λ)

Scenario	Road Use	Poisson Distribution			Negative Binomial Distribution		
		Mean (λ)	***Std. Error***	***p-value***	***Mean (λ)***	***Std. Error***	***p-value***
SOMPcurr	Light	0.125104	0.010214	1.247e-11	0.125108	0.016476	0.7207
	Mod	0.146788	0.011064	2.246e-13	0.146789	0.018038	0.8009
	Heavy	0.385321	0.017926	2.2e-16	0.385323	0.035501	0.2352
Suit100	Light	0.125104	0.010214	1.247e-11	0.125108	0.016476	0.7207
	Mod	0.145120	0.011001	4.557e-13	0.145124	0.017927	0.8338
	Heavy	0.372810	0.017633	2.2e-16	0.372820	0.034652	0.3539
Terr100	Light	0.125104	0.010214	1.247e-11	0.125108	0.016476	0.7207
	Mod	0.146788	0.011064	2.246e-13	0.146789	0.018038	0.8009
	Heavy	0.391159	0.018062	2.2e-16	0.391111	0.035706	0.2717

The mean indicates the average number of times a drinking water quality threshold is surpassed per week. For example, the SOMPcurr Light Use scenario resulted with an average of 0.125108 times per week compared to SOMPcurr Heavy Use with an average of 0.385323 times per week.

Additionally, I did not test for autocorrelation in the Negative Binomial distribution, though the potential for correlation exists. For example, if a turbidity event occurs one day, it likely increases the chance of a turbidity event occurring in the following days. Therefore, it must be noted that the mean values presented above (Table 11) likely overestimate the true mean.

With respect to the BAU and EMERG cost scenarios, only energy costs were included when estimating equation (5.1). A multitude of supply costs exist with respect to the operation and maintenance of the NCCW treatment and distribution system, however, such costs are not recorded for individual water quality events (personal communication, Derrick Casey, February 2010)[16]. Energy costs were the exception, as I was able to separate the cost to operate under a BAU and an EMERG scenario. Therefore, I used energy costs to determine the change in supply costs between forest management scenarios. Furthermore, the energy costs as a result from sourcing from the backup supply were substantially higher due to the energy required to pump from the wells (Table 12).

Table 12: Norrish Creek Community Watershed business as usual and emergency scenario energy costs in 2008

	Total Supply (m^3)	**Total Cost ($)**	**Average Cost per Week ($/$m^3$)**
Norrish Creek (BAU)	22,537,880	91,277.62	0.00405
Cannell Lake and Wells (EMERG)	5,248,970	109,681.64	0.02090
Total Supply	27,786,850	200,959.26	0.00723

Finally, FC remained the same for each forest management scenario. The FC and VC estimates per forest management scenario are detailed in Table 13. Note that VC increased by approximately $120,000 when comparing light use to heavy use. The cost increase is a result of the mean number of times per week a drinking water quality was

[16] Other costs include, for example, chemical treatment costs and labour costs. These costs are relatively low compared to energy costs (personal communication, Derrick Casey, February 2010).

surpassed (i.e. 0.125108 under light use to 0.385323 under heavy use) and thus the increased use of the costly backup source.

Table 13: Total annual costs resulting from the mean count per week that a drinking water quality threshold was surpassed (from Table 11), by forest management scenario and using data for 2008

Scenario	Road Use	Total Annual Variable Cost ($/yr)	Total Annual Fixed Costs ($/yr)	Total Annual Cost ($/yr)
SOMPcurr	Light	768,196	30,821,720	31,589,916
	Mod	778,345	30,821,720	31,600,065
	Heavy	890,002	30,821,720	31,711,722
Suit100	Light	768,196	30,821,720	31,589,916
	Mod	777,566	30,821,720	31,599,286
	Heavy	884,149	30,821,720	31,705,869
Terr100	Light	768,196	30,821,720	31,589,916
	Mod	778,345	30,821,720	31,600,065
	Heavy	892,711	30,821,720	31,714,431

The NCCW's total supply costs varied little across forest management scenarios with respect to aggregate length of roads. Comparatively, the sedimentation impact from traffic volume significantly increased the total cost to supply treated drinking water (Figure 12). However, such results are likely an underestimate due to the exclusion of many other costs incurred when a drinking water quality threshold is surpassed. For example, chemical treatment and extra labour hours would be incurred. As previously stated, such costs are not recorded for individual drinking water quality events and therefore could not be included in the analysis (personal communication, Derrick Casey, February 2010).

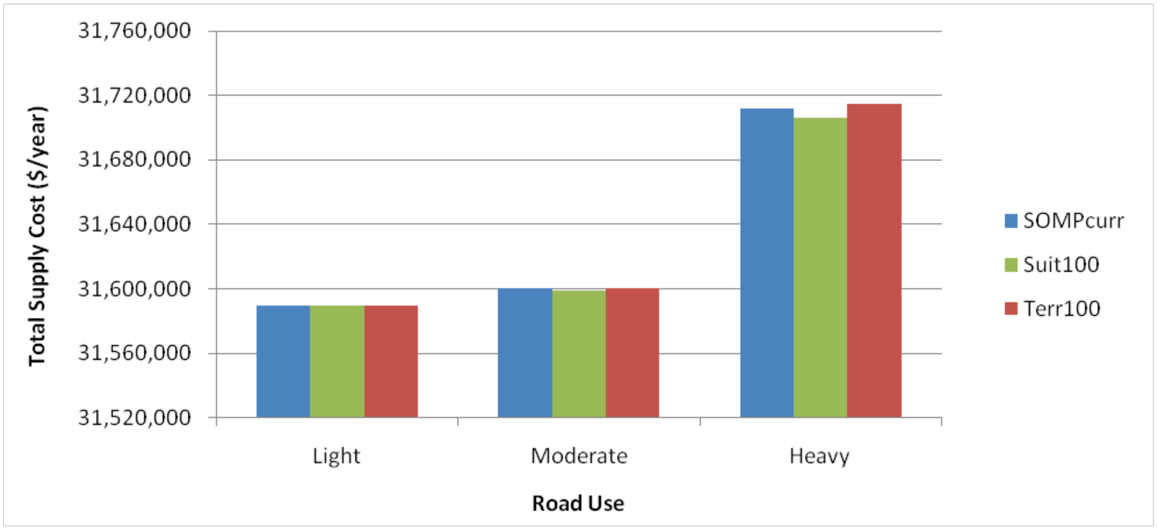


Figure 12: Total supply costs by forest management scenario and varying road use intensity at a 4% social discount rate ($/year)

5.2.3 Estimating Equilibrium Prices and Quantities per Forest Management Scenario

I used each forest management scenario's total supply costs to estimate each scenario's AC curve (i.e. supply curve) using equation (3.6). I estimated the inverse demand curve using equation (3.5). With respect to demand elasticities, both the City of Abbotsford and District of Mission charge a constant rate per volume supplied (personal communication, Kris Boland, May 2010; personal communication, Randy Millard, June 2010) and both are subject to a wet/cold climate. Therefore, the appropriate elasticity of demand is -0.3 (i.e. -0.4+0.1). Additionally, I estimated the inverse demand curve using the equilibrium price and quantity values stated in section 5.2.1. Finally, I determined each forest management scenario's equilibrium price and quantity by estimating the point at which the difference between the AC curve and demand curve was minimized (Table 14 to 16).

Table 14: SOMPcurr equilibrium prices and quantities of treated water to be used in welfare analysis

	Light Road Use		Moderate Road Use		Heavy Road Use	
Discount Rate	Price ($)	Quantity (m^3)	Price ($)	Quantity (m^3)	Price ($)	Quantity (m^3)
1%	1.08288	27,725,534	1.08340	27,721,489	1.08907	27,678,092
4%	1.13934	27,724,948	1.13986	27,721,110	1.14555	27,679,567
7%	1.19413	27,724,274	1.19465	27,720,670	1.20032	27,681,289

Table 15: Suit100 equilibrium prices and quantities of treated water to be used in welfare analysis

	Light Road Use		Moderate Road Use		Heavy Road Use	
Discount Rate	Price ($)	Quantity (m^3)	Price ($)	Quantity (m^3)	Price ($)	Quantity (m^3)
1%	1.08288	27,725,534	1.08337	27,721,563	1.08878	27,680,212
4%	1.13934	27,724,948	1.13983	27,721,202	1.14525	27,681,821
7%	1.19413	27,724,274	1.19461	27,720,779	1.20003	27,683,218

Table 16: Terr100 equilibrium prices and quantities of treated water to be used in welfare analysis

	Light Road Use		Moderate Road Use		Heavy Road Use	
Discount Rate	Price ($)	Quantity (m^3)	Price ($)	Quantity (m^3)	Price ($)	Quantity (m^3)
1%	1.08288	27,725,534	1.08340	27,721,489	1.08920	27,677,151
4%	1.13934	27,724,948	1.13986	27,721,110	1.14569	27,678,565
7%	1.19413	27,724,274	1.19465	27,720,670	1.20047	27,680,231

5.3 Summary of Study Parameters

A multitude of parameters were used in the integrated economic-ecological model. A glossary of parameters and descriptive table are presented below and include all parameters from Chapters 3 to 5 (Table 17).

Table 17: Integrated economic-ecological model parameter glossary and descriptions

Parameter	Description	Unit	Value	Data Source
P	Price of treated water	$/m^3$	Starting prices: 1% *dr*: 1.08 4% *dr:* 1.14 7% *dr*: 1.19	(City of Abbotsford, 2008; District of Mission, 2008; personal communication, Kristi Alexander, January 2010)
Q	Quantity of treated water outflow	m^3	Starting quantity: 27,786,849	(personal communication, Kristi Alexander, January 2010)
r	Discount rate		1%, 4%, 7%	(Knowler & Dust, 2008)
λ	Environmental quality parameter	Probability of surpassing the drinking water quality threshold per week	Table 14	-
l	Aggregate length of roads	km	SOMPcurr: 122.38 Suit100: 117.53 Terr100: 123.36	(Knowler & Dust, 2008)
u	Fine sediment yield generated by intensity of road use and road slope	tonnes/km/yr	Table 4	(Reid et al., 1981)
LR	Loss ratio		Table 4	(Reid et al., 1981)
DR	Delivery ratio		Table 5	(Reid et al., 1981)
∈	Treated water elasticity of demand		Table 1	(Billings & Jones, 2008)
D	Discharge	m^3/sec	Figure 6	(Water Survey of Canada, 2008)
ρ	Peak		Table 9	-

	parameter			
ϕ	Volume parameter		Table 9	-
δ	Road density	km/km^2	Base year (2008): 1.3	(Chapman Geoscience Ltd, 2000; Knowler & Dust, 2008).

6: RESULTS AND DISCUSSION

This chapter details the final valuation model results and discussion. I begin with a general overview of the welfare analysis conducted for this study. Then, I present the forest management scenario results and discuss the welfare implications of each. I conclude with a discussion of this study's contribution to watershed management in the NCCW.

6.1 Welfare Analysis

The economic value of the water purification/filtration service of the NCCW was estimated by the change in welfare resulting from implementing the conservation-oriented scenarios (Suit100 and Terr100) in place of the status quo scenario (SOMPcurr). Specifically, the welfare change measured the change in consumer surplus resulting from a change in raw water quality prior to entering the NCCW water utility. Furthermore, the change in raw water quality resulted from a change in the aggregate length and use of roads within the NCCW.

6.1.1 SOMPcurr to Suit100

Implementing Suit100 in place of SOMPcurr under most road use levels produced positive welfare results (Table 18; Figure 13). In terms of kilometre of roads, SOMPcurr has slightly more, but the difference did not significantly affect the welfare estimates. This suggests that slight changes in aggregate road length in the NCCW are of minimal concern with respect to the cost of domestic water supply. However, if a larger difference existed between the aggregate length of roads per forest management scenario, the length of roads would likely present a more noticeable impact (Jordan, 2006; Reid et al., 1981; Reid & Dunne, 1984). Varying the length of roads outside the range produced by the EAC analysis was outside the scope of this study.

Table 18: Annual change in consumer surplus as a result of adopting Suit100 in place of SOMPcurr, by level of road use (2008 $/year)

			SOMPcurr		
	Discount Rate	**Road Use**	*Light*	*Moderate*	*Heavy*
Suit100	1%	*Light*	**0**	14,391	171,243
		Moderate	-13,540	**855**	157,749
		Heavy	-163,733	-149,294	**8069**
	4%	*Light*	**0**	14,390	171,643
		Moderate	-13,514	**879**	158,173
		Heavy	-163,839	-149,404	**8339**
	7%	*Light*	**0**	14,326	171,350
		Moderate	-13,426	**903**	157,966
		Heavy	-163,774	-149,405	**8083**

Note: Estimates in the table are calculated as the sum of areas a and b in Figure 4. Bold values represent results for identical levels of road use under both scenarios.

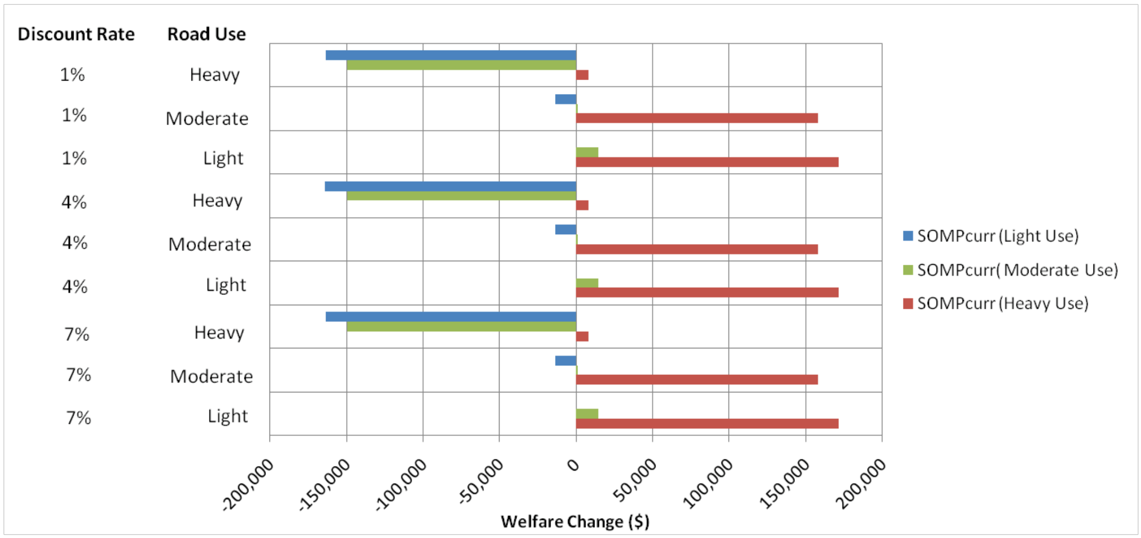


Figure 13: Annual change in consumer surplus as a result from adopting Suit100 in place of SOMPcurr by level of road use (2008 $/year)

Unlike length of roads, road use levels have a significant impact on welfare change. For example, the welfare change from SOMPcurr Heavy Use to Suit100 Light or Moderate Use produced over $150,000 in consumer surplus. In contrast, adopting Suit100 Heavy Use in place of SOMPcurr Heavy Use produced less than $10,000 in consumer surplus (Table 18; Figure 13).

Discount rates did not have a significant impact on the welfare estimates. Adopting Suit100 in place of SOMPcurr under any combination of road use level, produced welfare estimates that varied little between discount rates. For example, if NCCW managers decided to adopt Suit100 Light Use over SOMPcurr Heavy Use, and used a 7% discount rate compared to a 1% discount rate, would produce a mere $107.37 difference (i.e. $171,350.64 -$171,243.27).

Given all possible combinations of SOMPcurr and Suit100, the shift from SOMPcurr Heavy Use to Suit100 Light Use at a 4% discount rate maximized welfare. This case produced $171,643 in consumer surplus indicated by the grey area in Figure 14.

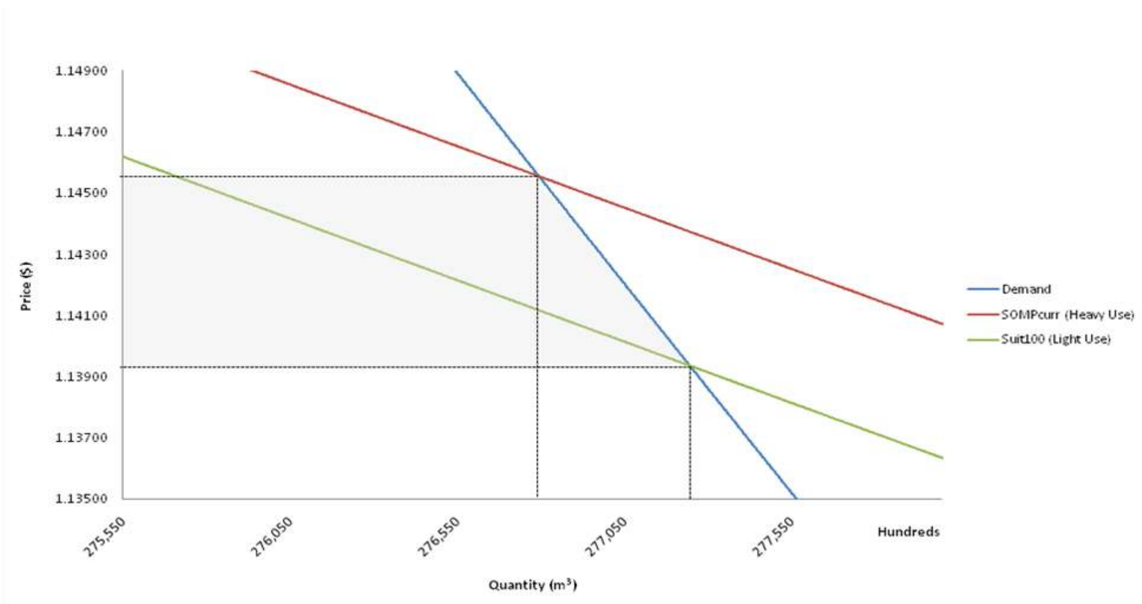


Figure 14: Consumer surplus as a result of adopting Suit100 Light Use in place of SOMPcurr Heavy Use at a 4% discount rate

Furthermore, consumer surplus remained positive among all shifts to or from heavy use. For example, it would be more desirable to continue under the SOMPcurr light or moderate use scenarios if the alternative was Suit100 Heavy Use. Therefore, the primary driver contributing to the welfare estimates was traffic volume.

6.1.2 SOMPcurr to Terr100

Unlike Suit100, Terr100 produced a larger variation between positive and negative welfare results (Table 19; Figure 15). Terr100 is a conservation oriented scenario in the Fraser TSA, however it has reverse impacts in the NCCW with respect to aggregate length of roads, which is slightly more than SOMPcurr. Like Suit100, the difference in road kilometres did not have a significant impact on welfare change. Again, this suggests the aggregate length of roads within the NCCW to be of minimal concern given the adoption of Terr100 over SOMPcurr.

Table 19: Annual change in consumer surplus as a result of adopting Terr100 in place of SOMPcurr by level of road use (2008 $/year)

			SOMPcurr		
	Discount Rate	**Road Use**	*Light*	*Moderate*	*Heavy*
Terr100	1%	*Light*	**0**	14,391	171,243
		Moderate	-14,396	**0**	156,896
		Heavy	-175,536	-161,093	**-5539**
	4%	*Light*	**0**	14,390	171,643
		Moderate	-14,394	**0**	157,296
		Heavy	-176,034	-161,595	**-5723**
	7%	*Light*	**0**	14,326	171,350
		Moderate	-14,330	**0**	157,064
		Heavy	-175,829	-161,457	**-5901**

Note: Estimates in the table are calculated as the sum of areas a and b in Figure 4. Bold values represent results for identical levels of road use under both scenarios.

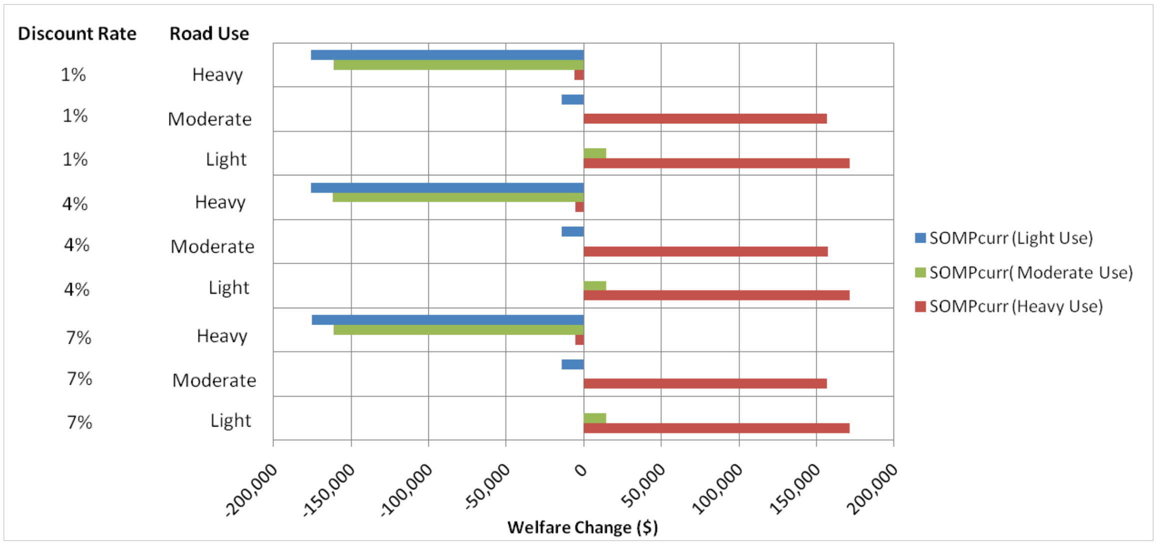


Figure 15: Annual change in consumer surplus as a result of adopting Terr100 in place of SOMPcurr by level of road use (2008 $/year)

Similar to Suit100, the road use level has a significant impact on welfare change (Table 19; Figure 15). For example, even though Terr100 contains more roads, adopting Terr100 light or moderate use in place of SOMPcurr Heavy Use produced over $150,000 in consumer surplus. This further highlights the importance to focus attention on traffic volume if resource managers must consider the tradeoffs between drinking water quality and timber harvesting in the NCCW. Additionally, no significant impact on welfare estimates existed with respect to a discount rates. Further analysis on this topic remains the same as Suit100.

Given all possible combinations of SOMPcurr and Terr100, the strategy that maximized welfare was the shift from SOMPCurr Heavy Use to Terr100 Light Use at a 4% discount rate. This case, like Suit100, also produced $171,643 in consumer surplus as indicated by the shaded area in Figure 16. The only difference in terms of positive and negative welfare results compared to Suit100, was the shift from SOMPcurr Heavy Use to Terr100 Heavy Use, which produced a negative welfare result; likely due to the increase in kilometre of roads. Therefore, like Suit100, the primary driver contributing to the welfare estimates, was again, traffic volume.

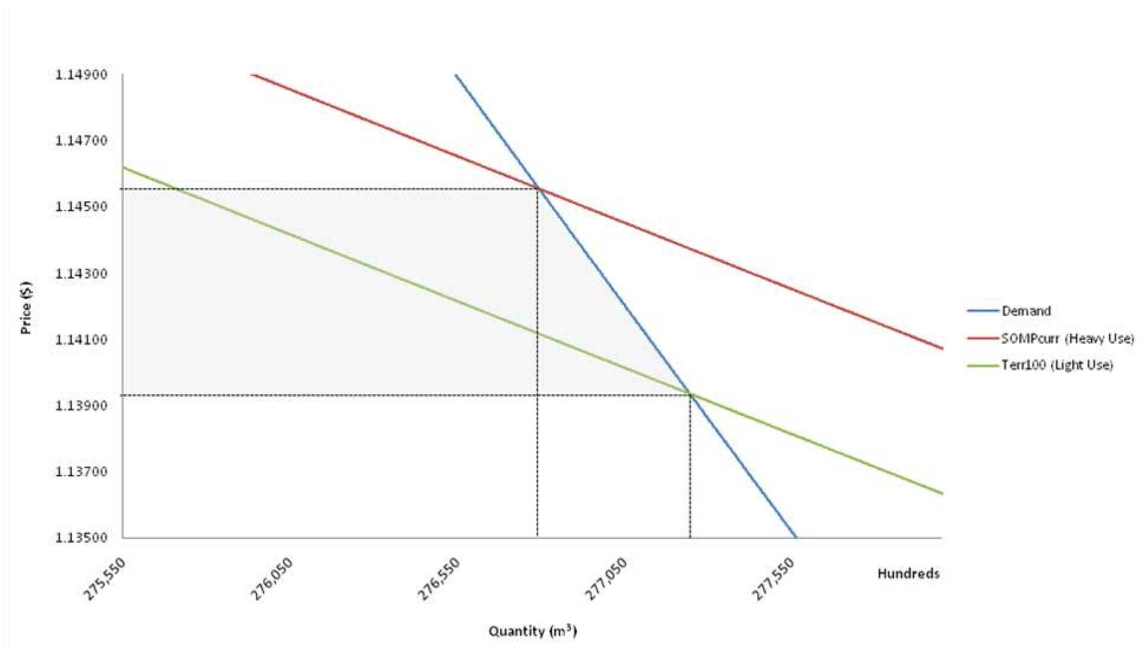


Figure 16: Consumer surplus as a result of adopting Terr100 Light Use in place of SOMPcurr Heavy Use at a 4% discount rate

6.2 Contribution to Watershed Management and Previous Research

The results of this study will contribute to the management planning of representative BC watersheds as well as further the analysis of Knowler and Dust (2008). I describe these contributions below.

The background and literature review (Chapter 2) indicated that forest roads are a primary source of fine sediments in streams (Hudson, 2006b; Jordan, 2006). This study focused on two road induced sedimentation sources; road use intensity and aggregate length. The literature specifically highlights fine sediment generation to be extremely sensitive to traffic volumes when compared to other sources including aggregate road length (Ried & Dunne, 1984; Wald, 1975) and thus should be subject to more regulation, especially during times of heavy rainfall or snowmelt when runoff would likely be higher (Jordan, 2006).

This study supports such findings in a new way. First, few studies have estimated the economic value of drinking water quality as it becomes degraded from timber harvesting activities. This study filled the gap by directly linking changes in the physical environment (i.e. from forest roads) to changes in social welfare (i.e. cost of municipal drinking water supply). In doing so, this study found that traffic volumes, especially from adopting moderate use in place of heavy use, increased consumer surplus by approximately $170,000. Comparatively, decreasing aggregate road length (e.g. SOMPcurr Heavy Use to Suit100 Heavy Use) increased consumer surplus by less than $10,000.

Furthermore, the results of this study can also be used to further the analysis of Knowler and Dust (2008). The authors suggested that further analysis was needed to estimate other benefits/costs from adopting Suit100 or Terr100 in place of SOMPcurr. Therefore, the water purification/filtration service welfare results presented here, which are presented on a per hectare basis in Table 20 and Table 21, can be inputted directly in the NPV estimate to help further their analysis.

Table 20: Annual change in consumer surplus as a result of adopting Suit100 in place of SOMPcurr by level of road use (2008 $/ha/year)

			SOMPcurr		
Suit100	**Capital Cost Discount Rate**	**Road Use**	*Light*	*Moderate*	*Heavy*
	1%	*Light*	**0.00**	1.80	21.41
		Moderate	-1.69	**0.11**	19.72
		Heavy	-20.47	-18.66	**1.01**
	4%	*Light*	**0.00**	1.80	21.46
		Moderate	-1.69	**0.11**	19.77
		Heavy	-20.48	-18.68	**1.04**
	7%	*Light*	**0.00**	1.79	21.42
		Moderate	-1.68	**0.11**	19.75
		Heavy	-20.47	-18.68	**1.01**

Note: Estimates in the table are calculated as the sum of areas a and b in Figure 4 divided by the land area of the NCCW (8,000 ha). Bold values represent results for identical levels of road use under both scenarios.

Table 21: Annual change in consumer surplus as a result of adopting Terr100 in place of SOMPcurr by level of road use (2008 $/ha/year)

			SOMPcurr		
Terr100	**Capital Cost Discount Rate**	**Road Use**	*Light*	*Moderate*	*Heavy*
	1%	*Light*	**0.00**	1.80	21.41
		Moderate	-1.80	**0.00**	19.61
		Heavy	-21.94	-20.14	**-0.69**
	4%	*Light*	**0.00**	1.80	21.46
		Moderate	-1.80	**0.00**	19.66
		Heavy	-22.00	-20.20	**-0.72**
	7%	*Light*	**0.00**	1.79	21.42
		Moderate	-1.79	**0.00**	19.63
		Heavy	-21.98	-20.18	**-0.74**

Note: Estimates in the table are calculated as the sum of areas a and b in Figure 4 divided by the land area of the NCCW (8,000 ha). Bold values represent results for identical levels of road use under both scenarios.

Lastly, it must be noted that the results presented above may involve a degree of bias. As stated earlier, biases in the estimation of the environmental quality parameter and omission of some emergency supply costs and sedimentation estimates may exist. Due to the potential for autocorrelation, the mean turbidity event per week is likely an overestimate. However, this is likely offset by the sedimentation and emergency supply cost biases. Only forest road sedimentation and energy costs were considered. Certainly other sedimentation sources and emergency supply costs exist. The underestimation of these estimates likely balanced the overestimation of the environmental quality parameter, and thus, produced a realistic consumer surplus result.

7: CONCLUSIONS

This study showed that human wellbeing is dependant on ecosystem health. I constructed an integrated economic-ecological model to value the water purification/filtration service in the NCCW. Currently, the NCCW must consider the economic tradeoffs between timber harvesting and municipal drinking water supply. The model found that the health of the water purification/filtration service was negatively compromised due to timber harvesting activities, which led to a decrease in social welfare, in terms of municipal water supply, under certain forest management scenarios.

This study also contributed to furthering the analysis of Knowler and Dust (2008). The authors estimated the NPV of various forest services, including timber harvesting, carbon sequestration, recreation values, and commercial and recreational mushroom colleting under alternative forest management scenarios in the Fraser TSA. However it did not include any water related watershed services. This study filled that gap by estimating the economic value on a per hectare basis of the water purification/filtration service for the same forest management scenarios used by Knowler and Dust (2008) (Section 7.2). Once such values are incorporated, the study by Knowler and Dust (2008) will possibly become the most comprehensive economic valuation study conducted in BC due to the breadth of values included.

Other important resource management implications drawn from this study include the applicability of a linear mixed-effects model to estimate water quality from discharge data, and the geographical scale at which AFSY was estimated. First, the linear mixed-effects model captured a more realistic SSC-D relationship due to the incorporation of inter-annual variability over a longer time periods. Second, this study simulated the total AFSY from all roads in the NCCW for the purpose of identifying the complete impact of those uses. Simply focusing on an individual sediment source (i.e. a specific road segment) may not provide the information required to influence the overall sustainable management of a multiple-use watershed like the NCCW. Therefore, studies at the watershed scale can provide a convenient unit of measurement that can capture the impacts of multiple uses.

Lastly, this study found that changes in welfare are more sensitive to changes in traffic volume when compared to changes in aggregate length of roads. However, it must be noted that testing for larger variations between aggregate road lengths were outside the scope of this study.

The integrated economic-ecological modelling approach used in this study provided insight into the welfare impacts of a multiple-use watershed. Specifically, the ecological model enabled the analysis of multiple scenarios. The economic valuation component added further rigor by estimating how a change in raw water quality affected a municipal water utility's cost function, which, unlike other economic valuation methodologies, properly specified the link between changes in raw water quality to changes in social welfare. However, there are of course, certain limitations in this study, which to interpret results correctly, should be made explicit.

The economic value of the water purification/filtration service in the NCCW was estimated by considering the sedimentation impacts from timber harvesting activities with respect to forest roads only. However, many other sedimentation sources exist within a multiple-use watershed. For example, recreation on forest roads, landslides (natural and forest road-induced), and stream bank erosion all contribute to the quality of raw water prior to its arrival at the water utility intake pipe. With respect to recreational use of forest roads, such use could substantially contribute to sedimentation due to the proven sensitivity of traffic volumes on stream-induced sedimentation. Therefore, the estimated economic value would likely be an underestimate produced by the simulation, although this may be offset by the possible overestimation of the environmental quality parameter.

Furthermore, only energy costs were considered when assessing the costs of surpassing a drinking water quality threshold. Certainly other costs exist, such as increased chemical treatment and labour, and if included, would provide a more realistic estimate of the water purification/filtration service. As mentioned previously, such costs were unavailable due to the accounting practices of the NCCW's water utility. However, the NCCW utility managers suggested such analysis (i.e. tracking costs resulting from surpassing a drinking water quality threshold) to be highly useful and are considering revising their accounting methodologies.

8: APPENDICES

Appendix A: Equilibrium quantity (Q*) solution

Following Varian (2003), the general formula for a demand with a constant elasticity is expressed as:

$$Q = AP^{\in} \tag{A.1}$$

where Q is quantity demanded ($/m^3), A is an arbitrary positive constant, P is the price of treated water ($/m^3), and $\in$ is the price elasticity of demand (Varian, 2003). Rearranging equation (B.1), the inverse demand curve is expressed as:

$$P = \frac{Q^{1/\in}}{A} \tag{A.2}$$

The supply is curve is expressed as:

$$P = AC = \frac{FC + VC}{Q} \tag{A.3}$$

where AC is average costs ($/m^3), FC is fixed costs ($/m^3), VC is variable costs ($/m^3), and Q is quantity of treated water (m^3).

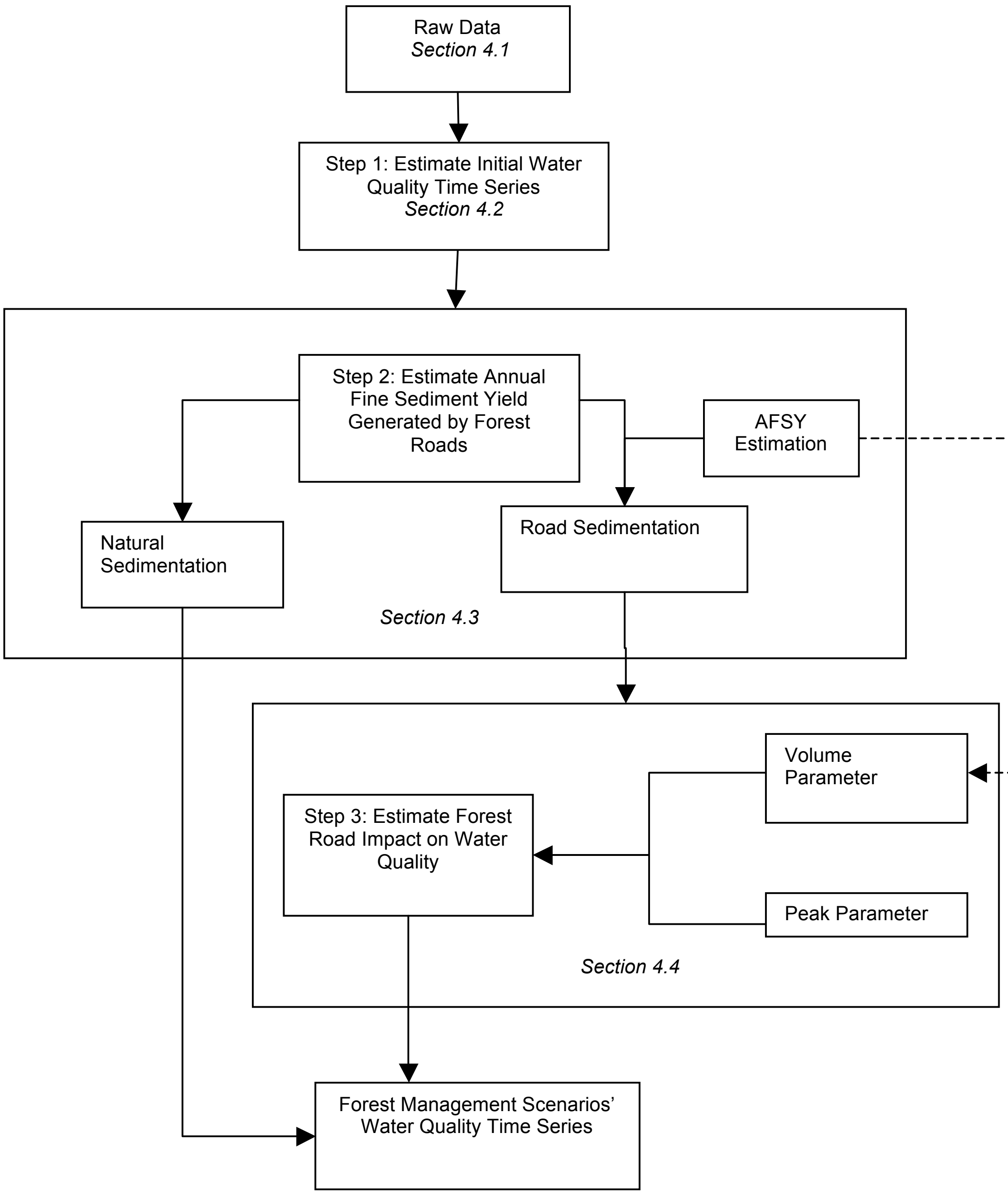
To estimate the equilibrium quantity (Q*) for each forest management scenario, I equated equations (B.2) and (B.3) as follows:

$$\frac{Q^{\frac{1}{\in}}}{A} = \frac{FC + VC}{Q} \tag{A.4}$$

Rearranging equation (B.4), the equilibrium quantity (Q*) is expressed as:

$$Q^{*} = A(FC + VC)^{\frac{1+\in}{\in}} \tag{A.5}$$

Appendix B: Conceptual water quality time series simulation approach



Appendix C: Procedure used to estimate aggregate length of roads per forest management scenario

To account for road decommissioning and thus provide for a more accurate assessment of the aggregate length of roads likely to exist at any given time over the 100-year construction profile, the Equivalent Annual Cost (EAC) approach was used. To estimate the EAC, I used the following equation (Adair, 2005):

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

where NPV is the net present value of total built roads over 100 years and *r* is the discount rate (i.e. road decommissioning rate). The approach discounts each year's new built roads and thus the discount rate is seen as the decommissioning rate. I estimated the EAC of roads at a 1% decommissioning rate, at a 4% decommissioning rate, and at a 7% decommissioning rate. The EAC aggregate length of roads was then used to estimate road density for further analysis.

Appendix D: Total water supply costs in 2008 ($) for the City of Abbotsford, District of Mission and Joint Water Supply System

Table 22: City of Abbotsford total water supply costs in 2008 ($)

Fixed Cost		**Variable Cost**	
Capital @ 1%	7,914,234	Local supply and distribution	200,000
Capital @ 4%	8,151,107	Meters	269,000
Capital @ 7%	8,391,829	-	-
Operating		-	-
- Admin	1,290,000	-	-
- Hydrants	248,000	-	-
- Meters	269,000	-	-
- Local supply and distribution	1,097,000	-	-
Total @ 1%	**10,818,234**	**-**	**-**
Total @ 4%	**11,055,107**	**-**	**-**
Total @ 7%	**11,295,829**	***Total***	**469,000**

Source: (City of Abbotsford, 2008)

Table 23: District of Mission total water supply costs in 2008 ($)

Fixed Cost		Variable Cost	
Capital @ 1%	424,184	Utilities	10,058
Capital @ 4%	428,464	-	-
Capital @ 7%	433,464	-	-
Operating		-	-
- Admin and Miscellaneous	288,228	-	-
- Contracted services	130,822	-	-
- Equipment	126,651	-	-
- Fees	270	-	-
- Insurance and professional services	2,349	-	-
- Materials and supplies	175,817	-	-
- Salaries and benefits	516,289	-	-
Total @ 1%	**1,664,610**	**-**	**-**
Total @ 4%	**1,668,890**	**-**	**-**
Total @ 7%	**1,673,890**	***Total***	**10,058**

Source: (District of Mission, 2008)

Table 24: Joint Water total water supply costs in 2008 ($)

Fixed Cost		Variable Cost	
Capital @ 1%	14,989,992	Treatment and disinfection	74,000
Capital @ 4%	16,313,723	Utilities	44,000
Capital @ 7%	17,586,165	Utilities (Norrish, Cannell, Wells)	200,960
Operating		-	-
- Lab supplies, sampling and analysis	99,000	-	-
- Inspections	158,000	-	-
- Maintenance	565,000	-	-
- General services	962,000	-	-
Total @ 1%	**16,773,992**	**-**	**-**
Total @ 4%	**18,097,723**	**-**	**-**
Total @ 7%	**19,370,165**	***Total***	**319,000**

Source: (City of Abbotsford, 2008; District of Mission, 2008)

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