Protecting Temperate Coastal Rainforests in British Columbia: Valuing Water Purification Services Using a Stochastic Production Function Approach

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Abstract

In many biodiversity rich forest areas, 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 potential benefit from an appropriate mix of activities in 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 improvement in the quality of water as the management emphasis shifts from timber harvesting to conservation for drinking water supply. We use an integrated economic-ecological model to quantify the economic impact of reduced forest road induced sedimentation on raw water quality prior to its arrival at a municipal water treatment plant. With respect to road-induced sedimentation, we consider traffic volume and aggregate road length. We find that the economic value of the water purification/filtration service is not as sensitive to traffic volume as it is to the aggregate road length. Our analysis will be helpful to forest planners who must consider the tradeoffs in forest management when timber harvesting can have harmful impacts on important ecosystem services, such as water purification/filtration.

Keywords: non-market valuation; ecosystem services; domestic water supply; public utilities; hydrological modelling; sedimentation

Introduction

Human wellbeing depends on the health of the world's ecosystems and the biodiversity the support. Left intact or relatively undisturbed, healthy ecosystems can contribute to production and wellbeing by providing many free goods and services, including those related to biodiversity.[1] 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 (World Bank, 2004). For example, timber harvesting in a watershed that also supplies water for domestic use can reduce the quality of the raw water prior to its arrival at the water treatment plant (Gomi et al., 2005). Society must then rely on additional costly processing of the raw water to compensate for the loss of the ecosystem service. Valuing the gain in economic welfare when timber harvesting ceases, the rainforest is protected and these ecosystem services are restored is the objective of this paper.

British Columbia (BC) contains some of Canada's most pristine forests and is renowned for its freshwater availability; however, it is also a major center of forest industry activity. A growing concern among BC residents is the current and future supply of domestic water within a reasonable distance of communities and urban areas. This concern is reflected in Metro Vancouver and Greater Victoria's decision to restrict land-use within the watersheds supplying their domestic water so as to safeguard these watersheds for drinking water supply (British Columbia Ministry of Health Services, 2007). Currently, almost half of BC's population accesses water from multiple-use watersheds where uses such as forestry, recreation and agriculture are permitted (Atkins et al., 2003). Interactions between two watershed uses, forestry and municipal

[1] The Millennium Ecosystem Assessment (2005a) labelled these benefits collectively as ecosystem services and, by definition; these include any human derived benefit obtained from ecosystems.

drinking water supply, are the focus of this paper. In particular, we attempt to value the ecosystem service benefits from a forested area when its use shifts from a mix of domestic water use and logging to conservation for domestic water use alone (no logging).

Valuation of the various contributions made by the natural environment to human welfare can be achieved using a variety of approaches (Freeman, 2003). The appropriate approach depends upon the specific benefit provided by the natural environment; in this case it is the ecosystem services supplied by watersheds. A well-known example used the avoided cost method to value water supply/quality benefits in the Catskills Mountains water protection project (Postel & Thompson, 2005).[2] However, some researchers contend such an approach should not be used to infer the welfare effects associated with changes in the ecosystem service in question (Banzhaf and Jawahar 2005). Instead, it is argued that researchers need to properly specify the link between physical changes in the environmental quality (i.e. water quality) and changes in production/costs (Freeman, 2003). An alternative approach that accounts for these physical links treats the natural environment as a factor input in the production function of a marketed commodity (Freeman 2003). We use an integrated ecological-economic model, together with the production function approach, to value the effects of a change in the value of the water purification/filtration service (Barbier, 2000). We estimate how a change in raw water quality

[2] Several other studies use an expressed preference approach. For example, Loomis et al. (2000) used contingent valuation and asked individuals their willingness to pay to restore a selection of ecosystem services in a riparian area, including water purification/filtration. In contrast, Hanley et al. (2006) used a choice experiment and created ecological status profiles (good and fair) and asked respondents about their preferences for water quality improvements. Other studies have used a revealed preference approach. Leggett et al. (2000) used the hedonic method to value water quality benefits captured in the market price of land, while Choe et al. (1996) used the travel cost approach to estimate the value individuals placed on improved water quality by surveying individuals travelling to a beach before and after a water quality advisory was posted. Additionally, Nunez et al. (2006) used the change in productivity method to estimate the economic contribution of Chilean temperate forests to the supply of clean drinking water.

related to increased sedimentation affects a municipal water utility's costs and then determine the associated change in economic welfare.

Sediment production is one of the key issues linking forestry activities and water quality and is the topic of extensive research (Hudson, 2001b). The main causes of induced sedimentation are erosion from forest roads (Beschta, 1978; Brown & Krygier, 1971) and landslides related to forest roads (Brardinoni et al., 2003). Under the right conditions, both have the potential to produce sediment in excess of what a watershed can filter naturally.[3] As a result, timber harvesting activities can reduce the quality of water prior to its arrival at the water treatment plant. 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).

Numerous studies 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 1988). However, few studies link the two fields of study to estimate the change in welfare as drinking water quality becomes degraded from timber harvesting activities. In one case, 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. Freeman et al. (2008) used a general linear model to analyze water quality, land cover and chemical treatment

[3] 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)

costs after surveying over 60 drinking water treatment plants. While the above studies estimate the economic value of water quality improvements, 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 the implications of these scenarios for the restoration of the ecosystem service provided by a standing forest associated with water quality.

A key consideration in carrying out our modelling was the potential for stochastic elements in our analysis. For example, sediment concentrations in watercourses depend critically on runoff conditions, which in turn rely on unpredictable rainfall patterns. To address this problem, we model the influence of sediment load on welfare as the expected number of events per week that raw water quality (i.e. turbidity) surpasses a treatment threshold, requiring the water utility to switch from routine treatment to more costly alternative water sources. We use a count data model to fit a probability distribution that describes the number of high turbidity events under differing assumptions about forest management (i.e. logging). Finally, we measure the change in welfare as the difference in consumer surplus when comparing each of these forest management scenarios. A key element in the analysis is the presence of increasing returns for most municipal water utilities and the need to consider average rather than marginal cost pricing.

In the section that follows, we provide a description of the case study area and then review the methodology used to construct our integrated ecological-economic model and to conduct the economic valuation. We then provide a brief description of our efforts to simulate a complete time series describing raw water quality in our case study watershed from the limited original data, along with the sources for other data and parameters. In particular, we describe the

estimation of the key parameter comprising the environmental influence in our domestic water supply production function. Subsequently, we present the economic cost data for the utility under conditions of normal water supply and when water quality surpasses a fixed threshold. Finally, we present our valuation results and provide a more in-depth discussion and some concluding comments.

Description of the Study Area

The Norrish Creek Community Watershed (NCCW) provides a good example of a multiple-use watershed and, as a result, was selected as our study area. Located within the Fraser Timber Supply Area (TSA) in south western BC (Figure 1), the NCCW contains important commercial timber harvesting sites and is the source of domestic water for approximately 156,000 residents of the City of Abbotsford and the District of Mission (Dayton & Knight Ltd., 2006).

The NCCW is situated within the Pacific Range of the Coast Mountains and drains into the north side of the Fraser River at Nicomen Slough. The watershed covers approximately 118 km^2 and consists of six 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 sits at an altitude of 250 m to 1420 m, with approximately 44% of its area lying within the transient snow zone (300 m to 800 m), 55% above 800 m and the remaining area located below 300 m. Since the 1940's, approximately 56% of the watershed has been logged yielding a comparatively low average annual harvest rate of 0.93% of the watershed per year (Chapman Geoscience Ltd., 2000).

Nonetheless, given its long history in the NCCW, the impacts of active logging have resulted in a significant number of events associated with reduced water quality (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).

Modeling Approach

In this section we explain the approach taken in our modeling and then describe the data we used and the various sources for this data (Figure 2). To determine the welfare change associated with a change in environmental quality affecting domestic water supply, we need to determine the economic characteristics of the municipal water utility assumed to extract and treat water from the watercourse. Large water supply facilities typically experience increasing returns to scale and, therefore, water utilities are often monopolists (Hosking & Preez, 2004). To eliminate the resulting inefficiencies, it is normally advised that the monopoly be regulated. However, drinking water utilities usually face large fixed costs and very low marginal costs so that the utility's marginal cost is likely to be below its average cost, creating a natural monopoly (Figure 3). The problem can be resolved if the water utility sets price equal to marginal cost and receives a subsidy to cover the large fixed costs (Varian, 2003). However, Brandes et al. (2010) point out that most Canadian water utilities instead use some form of average cost pricing, not marginal cost pricing, for a variety of reasons related to Canadian conditions.[4] We assumed the

[4] 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 cost (Brandes et al., 2010).

latter approach in modeling the change in economic welfare associated with a change in environmental quality affecting municipal water supply; that is, that price equals average cost.

Following Freeman (2003), the cost function in year t for a firm relying on an exogenous environmental input can be written as:

$$C_t = C(Q_t; \lambda) \tag{1}$$

where $C(..)$ is the firm's cost function to supply drinking water, Q_t is the quantity of treated water supplied in year t (m^3), and λ represents an environmental input influencing raw water quality that is determined exogenously to the utility. In our model, the environmental quality influence λ represents the mean number of times a drinking water quality threshold is surpassed within a specified period. The implications of surpassing the threshold for utility costs are described below.

We assume λ is a function of the road network characteristics within the watershed:

$$\lambda = f(l, u) \tag{2}$$

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 supplying drinking water. Of course, in watersheds where timber harvesting is taking place the road network will depend critically on these logging activities.

To determine λ we simulated water quality under various forest management scenarios and constructed a count data variable measuring the number of times per week the water quality threshold was surpassed in each simulated time series. The probability of surpassing the

threshold in a given week was fitted as a Poisson distribution applied to the count variable data.[5] Estimating the distribution for λ allowed us to determine how changes in the probability of surpassing a water quality threshold affect average costs when forest management is altered. This effect occurs because the water supply system is taken offline when the water quality threshold is surpassed and backup water sources, such as a lake and groundwater wells, are used to meet demand instead. A difference in the costs of supply arises because the costs in the offline situation using the backup sources are much higher than when no turbidity event occurs (D. Casey, personal communication, February 2010).

Under the usual conditions of profit maximization and no increasing costs, the water utility's total contribution to welfare W would be the sum of consumer and producer surplus, expressed as:

$$W_t = \int_0^Q P(Q_t)dQ - C(Q_t;\lambda_i) \tag{3}$$

$$st\ Q_t \leq \overline{Q} \text{ and } \lambda_i = f(l_i, u_i),\ \ for\, i = 0,1,2,..$$

where P is the price of treated water (\$/m^3), $P(Q)$ is the inverse demand curve for treated water (\$/m^3) and λ_i represents the environmental influence associated with the road characteristics of a given forest management scenario i. Note that the quantity of water supplied cannot exceed the physical capacity of the drinking water facility, $\overline{Q}$.

For a given year t, the change in producer and consumer surplus associated with a non-marginal change in raw water quality from λ_0 to λ_1 is expressed as:

[5] Fitting the distribution assumed that the data are individually independent and that the mean and variance are equal. To determine whether overdispersion was present, we used the Pearson's Chi-Square test and then estimated the mean for λ using maximum likelihood (Crawley, 2007). See next section.

$$\Delta W = \int_0^{Q_t^*} P(Q_t^*)dQ - C(Q_t^*;\lambda_1) - \int_0^{Q_t^*} P(Q_t^*)dQ - C(Q_t^*;\lambda_0) \tag{4}$$

where Q^* is the equilibrium quantity of water supplied and 0 and 1 refer to the situation before and after the environmental change. The environmental change results from the shift from one forest management scenario to another, i.e. from logging to no logging. In the absence of increasing returns, we would derive the equilibrium quantity of water Q^* for each forest management scenario by equating the price and marginal cost of treated water. However, the average cost pricing approach instead equates price and average cost, which allows the water utility to break even, but dissipates any producer surplus. Changes in social welfare are measured using (4) but reflect a different Q^* and a change in consumer surplus only. The welfare change can be measured as the area between the old and new average cost curves and bounded by the inverse demand curve (Ellis & Fisher, 1987; Freeman, 1991; Freeman, 2003).

Assuming constant elasticity of demand, the demand curve for water can be expressed as:

$$Q_t = \alpha P_t^{\in} \tag{5}$$

where α is an arbitrary positive constant and $\in$ is the constant own price elasticity of demand (Varian, 2003). Rearranging (5), the inverse demand curve is:

$$P(Q_t) = \frac{Q_t^{1/\in}}{\alpha} \tag{6}$$

and the supply curve is:

$$P(Q_t) = AC_t = \frac{C(Q_t;\lambda_i)}{Q_t} = \frac{F}{Q_t} + V(\lambda_i) \tag{7}$$

where F are total fixed costs and $V(\lambda_i)$ are the unit variable costs of water supplied per m^3, expressed as a function of the environmental parameter λ_i. We specify unit variable costs $V(\lambda_i)$ as:

$$V(\lambda_i) = V_{ne} + \frac{\lambda_i}{7}\left(V_e - V_{ne}\right) \tag{8}$$

where V_{ne} is the unit variable cost of treated water under the normal (non-event) conditions, V_e is the unit variable cost when there is a water quality event (threshold exceeded) and λ_i is the estimated mean number of times per week the water quality threshold is surpassed under forest management scenario i.

Substituting (8) into (7), equating (6) and (7) and rearranging, the equilibrium quantity Q^* for each forest management scenario can be determined for given values of α and the elasticity of demand by solving the resulting implicit equation:

$$\alpha F + \alpha\left[V_{ne} + \frac{\lambda_i}{7}(V_e - V_{ne})\right]Q^* - Q_t^{*\frac{1+\in}{\in}} = 0 \tag{9}$$

We fitted the demand curve for water in (6) using industry rules of thumb for the own price elasticity of demand (Billings and Jones, 2008), and by calibrating with industry data. We then solved for the equilibrium values Q^* numerically using (9). Once the unique equilibrium values Q^* were determined for each forest management scenario, we used (4) to estimate the change in welfare associated with shifting from one scenario to another, i.e. from logging to no logging.

In the empirical analysis that follows, we use our model to analyse two forest management scenarios describing the *NCCW* over a 100 year planning horizon:

a) continued logging according to the projected forest harvesting management plan (Sutherland et al., 2007), referred to as the "logging" scenario; and,

b) cessation of all logging within the *NCCW*.

We used the road construction data from the proposed harvest management plan for the *NCCW* portion of the Fraser TSA to drive our sediment modelling (Sutherland et al., 2007). The proposed road construction is 15 km per decade initially, rising to 25 km per decade for the next three decades and then it gradually declines over the remaining six decades to less than 5 km per decade by the end of the period. In addition, we allowed the level of intensity of road use to vary from low to medium to high. As a result, we assessed two distinct forest management scenarios, logging and no logging, but assessed the former at three levels of road use intensity.

Water Quality Modelling and Environmental Parameter Estimation

A key requirement for our modeling were projected water quality time series for the Norrish Creek Community Watershed (*NCCW*) under the alternative forest management scenarios. These time series were used to derive unique values for the environmental parameter λ for each management scenario.[6]

[6] We assume 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 (Gomi et al., 2005). Fine sediments are measured as either suspended sediment concentration (SSC) or as turbidity. SSC is usually expressed in milligrams of sediment per litre of water (mg/L) and requires extensive stream sampling and time-consuming lab analysis. Turbidity is expressed as absolute Nephelometric Turbidity Units (NTU) and is measured instantly by a turbidity sensor that gauges how suspended sediment particles obstruct light transmission (Pfannkuche & Schmidt, 2003).

Mixed-Effects modelling of historical water quality in the *NCCW* (over 22 years)

To simulate a complete water quality time series for the historical period 1984 to 2006, a SSC-discharge (D) relationship was estimated as a suspended sediment-rating curve (Gomi et al., 2005; Horowitz, 2003). The most commonly used relation is the power law function, expressed as $SSC = aD^b$, 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 *SSC* fairly accurately over shorter periods, but they become problematic when applied to longer time series (Walling & Webb, 1988). Given the length of our time series period (22 years), there was the possibility of under predicting actual concentrations due to the dispersion of the data and seasonal variation in the SSC-D relationship (Asselman, 2000; Ferguson, 1986; Walling & Webb, 1988). To compensate for this limitations, we used a linear mixed-effects model to estimate the *SSC*-D relationship (Pinheiro & Bates, 2000). Using a linear mixed-effects model more accurately captures the SSC-D relationship, while accounting for the inter-annual variability of hydrological time series data.

We 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.[7] 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 in the data set, while the random effects were related to the variability within each month for individual years. Using the linear mixed-effects model

[7] The 12 linear mixed effects models 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).

estimates for each month, we simulated a complete baseline water quality time series reflecting historical road and weather conditions for the 1984 to 2006 period (Figure 4).[8] Stream discharge data was provided by the Water Survey of Canada (Water Survey of Canada, 2008) and we used turbidity data from the British Columbia Ministry of Environment (personal communication, Jennifer Guay, July 2008). Further details on the use of a mixed-effects modelling approach in our study is provided in the Appendix.

Simulating water quality time series under the Logging and No Logging Scenarios

The next step in obtaining the data we needed was to develop hypothetical water quality time series describing the effects that our forest management scenarios would have on water quality (turbidity) over the 100 year planning horizon. Using the simulated 22 year historical time series described above as a baseline, we made adjustments to account for the differences in logging and road building activity under each management plan, resulting in a set of unique simulated water quality time series for each possible management scenario (no logging and logging with three levels of road use intensity). Several steps were involved in generating these time series, beginning with determining the sediment yield from the watershed under varying logging road conditions.

Paired-basin studies and sediment budgets are two primary methods used to determine sediment yield within a watershed (Reid, 1981).[9] Taking a sediment budget approach, the load of

[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 affect further analysis because this study was concerned with events associated with a much lower drinking water quality threshold of only 10 NTU. Additionally, our approach assumes no temporal trends (i.e. effects from climate change).

[9] Under the paired-basin approach, suspended sediment concentration (SSC) is measured in paired treatment and control watersheds. Differences in the measured sediment yield are used as an indicator of varying sediment production due to differences in land use within the paired watersheds. An important limitation of this approach is the inability to distinguish individual sediment sources and assign them to natural disturbances and/or individual

suspended sediments Γ_s (tonnes/day) for a given road network on day s can be expressed as the sum of the background (non-logging road) daily suspended sediment load N_s and the sediment generated by roads R_s. Since we calculate N_s as a residual we can express the relationship as:

$$N_s = \Gamma_s - R_s \tag{10}$$

Following Colby (1956), we estimated Γ_s as a function of the suspended sediment concentration *SSC*:

$$\Gamma_s = SSC_s D_s 10^{-9} \tag{11}$$

where SSC_s is the suspended sediment concentration (mg/L) for day s and D_s is the watercourse discharge data for the same day (L/day).

To estimate R_s without daily time series data, we simulated the total contribution of sediment from roads over the entire period R_T. We then used this aggregate value, together with total sediments over the period from all sources Γ_T, to prorate the estimate of daily suspended sediments from (11). Replacing R_s in (10) we get:

$$N_s = \Gamma_s (1 - \frac{R_T}{\Gamma_T}) \tag{12}$$

The aggregate suspended sediment yield Γ_T for period T is:

$$\Gamma_T = \sum_{s=1}^{T} \Gamma_s \tag{13}$$

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land-use activities. Due to this (and other) limitations, we instead used the budget approach to model sediment generation.

We adapted an earlier sediment budget study from a nearby watershed to estimate R_T as the sum of annual fine sediment yields (*AFSY*) from surface road erosion over the period (Greater Vancouver Regional District, 1999).[10] *AFSY* can be calculated from data for a given forest road network scenario as**:**

$$AFSY_i = l_i u_i LR\ DR \tag{14}$$

where $AFSY_i$ is the estimated annual fine sediment yield (tonnes/yr) for road network scenario i, l_i is the aggregate length of roads (km), u_i is the fine sediment yield (tonnes/km/yr) based on the road segment's use level and slope, *LR* is the sediment loss ratio and *DR* is the sediment delivery ratio.

We used Reid (1981) to establish the parameters for the *AFSY* relationship. The loss ratio *LR* reflects diversions of sediment-laden water by obstructions in the path of flow. Reid (1981) measured the total sediment concentration (both u and *LR*) at specific road culverts under different road use levels and calibrated the results for varying slopes using the Universal Soil Loss Equation (Wischmeier & Smith, 1965). We adjusted these results to arrive at values for u and *LR* for various assumptions describing forest roads (e.g. slope, road use intensity).[11] The delivery ratio *DR* accounts for the portion of fine sediment that eventually enters a stream or river. Reid (1981) estimated the delivery ratio at 50% for a road with a 28 to 49% slope.

Our adapted model was used to generate a time series based on the daily sediment yield from background sources N_s and the road network R_t under historic conditions of road use and

[10] *AFSY* includes only sediment grain sizes (i.e. fines) that cause turbid waters and excludes all other sediment material such as coarse-grained materials (e.g. sand, pebbles, cobble and boulders).

[11] Since we consider an average road sloped at 0 to 2.5 degrees but later allow for varying road use intensities, we used the following parameter values for the sediment yield (tonne/yr): Light road use (0.2), Moderate road use (2.1), and Heavy road use (25.0), along with a loss ratio (*LR*) of 0.7 (Reid, 1981).

aggregate road length.[12] We then adjusted this time series that is based on historical conditions to represent forest management scenario conditions over the 100-year road profile, taking into account how the frequency of peak events might change, as well as the volume of the total suspended sediments. We used the 100-year road profile from the proposed forest harvest plan in the *NCCW* to estimate separate peak and volume parameters for each forest management scenario and then applied the parameters to our 22-year water quality time series for Norrish Creek.[13]

As a first step, we simulated the influence from the construction of forest roads on peaks in the daily *SSC* under the assumption that an increasing road density would lead to more frequent water quality events but at lower peaks than with a reduced road density; in other words, an increase in road density should diminish the distance between lower and higher peaks in the water quality time series.[14] We created an adjusted peak level for daily suspended sediments under each forest management scenario using:

$$f_{si} = e^{\rho_i \ln(R_s+1)} - 1 \tag{15}$$

where f_{si} represents the adjusted daily peak level of suspended sediment (tonnes/day) and ρ_i is a parameter representing the influence of the road network in forest management scenario i on the

[12] Specifically, we assumed the average logging road in the NCCW was sloped at 0 to 2.5 degrees, terrain was sloped at 28 to 49%, the number of streams consisted of one major stream or more than one ephemeral stream, and road use was 'moderate'. See Reid (1981) for details.

[13] To make our modelling tractable, we established a constant 100 year forest road construction profile using a levelized (equivalent annual) calculation using:

$$EAC_{roads} = \frac{PV_{roads}}{1-(1+r)^t}$$

where PV_{roads} is the present value of the annual road construction kilometres over the planning horizon and r is the discount rate (Adair, 2005).

[14] Assuming the amount of sediment delivered to a stream from the watershed is constant, then a high density road network will deliver sediment to the channel many times with moderate peaks. If there are no roads, the sediment will be stored in various locations in the watershed and released in large pulses caused by debris flows, landslides, or vary large floods (J. Venditti, pers. comm..).

peak in suspended sediment load. The ρ parameter establishes the frequency of sedimentation events (i.e. peaks) given different aggregate road lengths and was estimated as:

$$\rho_i = \frac{100 - \delta_i}{100} \tag{16}$$

where δ_i is the percent increase in road density over the baseline year for forest management scenario i. Using (15) and (16), we generated a new peak suspended sediment value for each day, and for each forest management scenario, by prorating the total amount of suspended sediments delivered over the entire period (R_T) using:

$$f_{si}^{*} = \frac{e^{\rho_i \ln(R_s+1)} - 1}{\sum_{s=1}^{T}\left\{e^{\rho_i \ln(R_s+1)} - 1\right\}} = \frac{f_{si}}{\sum_{s=1}^{T} f_{si}} \tag{17}$$

Next we accounted for variations in sediment yield attributable to the intensity of road use. The volume parameter ϕ determines the change in the daily volume of suspended sediments for deviations in the aggregate length and use of roads from the baseline in 2008. The volume parameter ϕ was estimated as:

$$\phi_i = \frac{AFSY_i}{AFSY_0} \tag{18}$$

where $AFSY_i$ is the $AFSY$ generated from Light, Moderate or Heavy road use associated with forest management scenario i, and $AFSY_0$ is the $AFSY$ generated in 2008. We then used (18) to shift the historic water quality time series up or down, resulting in an increase or decrease in the amount of fine sediments entering the watercourse.

Finally, to obtain the simulated daily suspended sediments generated by roads under our logging management scenarios (R_{si}'), we multiplied the total historic volume of road induced sediments R_T by f_{si}^* and ϕ_i,:

$$R_{si}' = \phi_i f_{si}^* R_T \tag{19}$$

We then added the background sediment load for each day N_s to (19) to yield:

$$\Gamma'_{si} = R'_{si} + N_s \tag{20}$$

where Γ'_{si} is the simulated total daily suspended sediment in the watershed for forest management scenario i.[15]

Estimation of the stochastic environmental parameter λ as the mean number of water quality events per week

To estimate the mean number of times per week the water quality threshold was surpassed we used the simulated water quality time series for the logging and no logging scenarios described above. We simply counted the number of times per week the water quality threshold was surpassed and used this data to generate a count data set. Using this data set, we fitted a probability distribution describing the likelihood of surpassing the water quality threshold. Assuming the probability of surpassing the threshold is described by the Poisson distribution, the statement for the probability that a given number of water quality events will occur during a typical week is expressed as:

[15] Since municipal water utilities use turbidity measurements to assess water quality, we applied the inverse summation procedure to calculate *SSC* (mg/L) from sediment yield (Colby, 1956) and then converted *SSC* to turbidity units using a standard turbidity-SSC relationship (Carson, 2002; pg 10).

$$\text{Prob}(K_w = k_w) = \frac{e^{-\lambda_i w}\lambda_i^{kw}}{k_w!}, k_w = 0,1,2,... \tag{10}$$

where K_w is a random variable (i.e. water quality readings surpassing the threshold) and k_w is the observed count of events during week w. Note the restriction that k_w is a non-negative count variable (Ricci, 2005).

To identify the correct model to fit the probability distribution, we 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 at the 5% level (p-value < 0.05). However, there was no compelling evidence to suggest the data did not follow a Negative Binomial distribution (p-value > 0.23). Although we fitted both the Poisson and the Negative Binomial distributions to estimate the mean number of times per week the water quality threshold was surpassed for our two forest management scenarios (Table 1), we used the estimate from the Negative Binomial distribution in the ensuing valuation exercise.[16] Our estimate for the mean number of times a drinking water quality threshold would be surpassed per week under the Logging scenario with Light road use was 0.137 times per week; this compares to 0.404 times per week under the Logging scenario but with Heavy road use (Table 1).

Water Utility Costs and Data Sources

[16] Additionally, we 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 may increase the chance of a turbidity event occurring in the following days. Therefore, it must be noted that our estimated mean values for λ may overestimate the true mean.

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 (Kris Boland, personal communication, May 2010; Randy Millard, personal communication, June 2010). Since our concern is with the entire watershed, changes in average cost (price) must be based on total system supply costs.

To estimate total supply costs, we separated the fixed and variable costs for the City of Abbotsford, District of Mission and the Joint Water Supply System (Table 2). With respect to capital costs, we estimated a levelized annual value using 1%, 4% and 7% discount rates applied to the capital expenditure plan.[17] Furthermore, we used 2008 as our baseline year to determine average unit costs (i.e. price) from the total supply costs since the most recent treated water outflow data were available for this year (Kristi Alexander, personal communication, January 2010). When the NCCW's total water supply costs were divided by the treated water outflow for 2008, the resulting estimates of average costs (prices) were: \$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; these values were used as the equilibrium prices for the baseline year. Following the methodology described in Billings and Jones (2008), we then used the elasticity formula to estimate changes in the quantity of water demanded consistent with our values for the constant elasticity of demand and arbitrary increments in price.[18] The results of this exercise were used in the next section to establish the

[17] To create a levelized value for capital costs we calculated an equivalent annual cost (EAC) using:

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

where NPV is the net present value of total capital expenditures over the planning horizon and r is the discount rate (Adair, 2005).

[18] Billings and Jones (2008) used the standard equation for demand elasticity to estimate quantity demanded:

new equilibrium price and quantities under the alternative forest management regimes (logging vs. no logging).

Various costs are incurred for the operation and maintenance of the NCCW treatment and distribution system but are not recorded individually for individual water quality events except for energy costs associated with pumping, which are the most significant cost item affected by an event (Derrick Casey, personal communication, February 2010). Thus, to distinguish the normal conditions with no water quality event from event conditions, only energy costs were considered.[19] Under water quality event conditions, the energy costs were substantially higher from the backup supply source due to the much higher energy requirement to pump from wells. When comparing the Light road use scenario to Heavy road use under the Logging scenario, the mean number of times per week the drinking water quality threshold was surpassed declines, requiring less use of the costly backup source. The resulting savings in energy costs are over $120,000 per year (see Table 5).

The various economic parameters used in our modeling were obtained from a variety of sources. Using information from the City of Abbotsford and District of Mission (Kris Boland, personal communication, May 2010; Randy Millard, personal communication, June 2010), the own price elasticity of demand was established from rule of thumb estimates in Billings and Jones (2008). Other parameters included the quantity of treated water for the baseline case Q_0 and the price of treated water P. The City of Abbotsford's Engineering Department supplied the treated water outflow data for 2008, our baseline year (Kristi Alexander, personal

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$$\Delta Q = \frac{\Delta P}{P} Q \in$$

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. We used $0.10 increments in price to solve for ΔQ.

[19] Other costs include chemical treatment and labour but these costs are relatively low compared to energy costs.

communication, January 2010). The price of water was determined from data contained in the City of Abbotsford 2008-2012 Financial Plan (City of Abbotsford, 2008) and the District of Mission 2008 Annual Report (District of Mission, 2008).

Results and Discussion

This section details the valuation modelling results for the NCCW and presents a brief discussion of our study's contribution to watershed management more generally. To derive our results we first determined the equilibrium price and quantity for our logging and no logging scenarios numerically (Table 3). Once the new values for price and quantity were determined for our logging and no logging scenarios, we calculated the change in consumer surplus associated with the shift from logging scenario to the no logging scenario (Table 4). Therefore, our estimates measure the change in consumer surplus resulting from a change in raw water quality entering the *NCCW* water treatment facility. Furthermore, the change in raw water quality results from a change in the aggregate length and use of roads within the *NCCW* and the subsequent impact this has on the frequency of water quality events and, subsequently, on water system supply costs (Table 5).

Our results indicate that the shift to conservation to protect the water purification service provided by the watershed results in substantial welfare gains, running from $2178/year, at a 1% capital amortization rate and Light road use, to as much as $178,101/year, at a 1% capital amortization rate and Heavy road use (Table 4). Values for other amortization rates (4% and 7%) fall between these two values. When these consumer surplus values are expressed per hectare of watershed area, the values suggest the ecosystem service provided by the watershed is significant but depends dramatically on the level of road use; the per hectare values are $0.28/ha/year and

$22.22/ha/year at a 4% amortization rate for Light and Heavy road use, respectively (Table 4). The large welfare change associated with the full conservation of the watershed arises because of the substantial impact on the frequency of water quality events associated with the construction of roads under the current harvesting plan. Although our estimates of welfare change are influenced significantly by our intensity of road use assumptions (low/moderate/high), the rate used to amortize capital costs mattered much less; not a surprising result given that water quality events affect short run costs (energy) and have no impact on longer run capital cost, which are sunk for modelling purposes.

Our welfare change estimates are likely to be a slight underestimate due to the exclusion of 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. However, our belief is that these costs are not very large in comparison to the energy costs for pumping on which we based our calculations.

The results of this study can contribute to management planning for watersheds more generally. We indicated earlier that forest roads are a primary source of fine sediments in streams but that relatively little valuation work has been undertaken. Our study helps to advance understanding in new ways. 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).

Lastly, it should be noted that although the results presented above may involve a modest degree of bias these various sources of uncertainty may offset one another. As stated earlier, a

bias in the estimation of the environmental quality parameter due to the potential for autocorrelation, may result in overestimation of the mean number of turbidity events per week. Concomitantly, the omission of some backup water supply costs and sources of sedimentation potentially could lead to underestimation of additional costs due to an increased frequency of water quality events. Overall, we believe are estimates are reasonable.

Conclusions

The integrated economic-ecological modelling approach used in this study provided insight into the welfare impacts of changes in the use of a multiple-use watershed. The economic valuation component added further rigor by showing how a change in raw water quality affected a municipal water utility's cost function. Unlike other economic valuation methodologies, our use of the production function method explicitly specified the link between changes in raw water quality and changes in welfare. 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.

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. 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 complete estimate of the water purification/filtration service. Water system managers suggested that keeping track of detailed costs resulting from surpassing a drinking water quality threshold would be useful and are considering revising their accounting methodologies. We would support such a move at the *NCCW* and more generally at water utilities.

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Table 1

Results from fitting the Poisson and Negative Binomial Distribution for the mean number of times the water quality threshold is surpassed per week (10 NTU), by forest management scenario and level of road use

Scenario	**Road Use**	**Poisson Distribution**			**Negative Binomial Distribution**		
		Mean (λ)	***Std. Error***	***p-value***	***Mean (λ)***	***Std. Error***	***p-value***
No Logging		0.133444	0.010549	8.573e-13	0.133448	0.017439	0.6977
Logging	Light	0.136780	0.010680	1.294e-13	0.136784	0.017850	0.7668
	Mod	0.155129	0.011374	2.31e-15	0.155132	0.019257	0.8696
	Heavy	0.403669	0.018348	2.2e-16	0.403663	0.037199	0.1415

Table 2

Norrish Creek Community Watershed annual water supply costs for 2008, by interest rate used to amortize capital expenditures

Interest Rate (Capital Amortization)	Fixed Cost ($/year)	Variable Cost ($/year)	Total Cost ($/year)
1%	29,256,835	798,058	30,054,893
4%	30,821,720	798,058	31,619,778
7%	32,339,883	798,058	33,137,941

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

Note: based on production of 27,786,850 m^3 of treated water

Table 3

Equilibrium prices and quantities of treated water used in welfare analysis for the Logging and No Logging scenarios, assuming a 100 year planning horizon and by level of road use (2008 prices)

	Logging						No Logging	
	Light Road Use		Moderate Road Use		Heavy Road Use			
Interest Rate	Price ($/m^3)	Quantity (m^3)	Price ($/m^3)	Quantity (m^3)	Price ($/m^3)	Quantity (m^3)	Price ($/m^3)	Quantity (m^3)
1%	1.08	27,723,254	1.08	27,719,797	1.09	27,674,597	1.08	27,723,843
4%	1.14	27,722,542	1.14	27,719,328	1.15	27,676,763	1.14	27,723,166
7%	1.19	27,722,199	1.19	27,719,251	1.20	27,678,311	1.19	27,722,855

Table 4

Mean annual change in consumer surplus (undiscounted) assuming a 100 year planning horizon as a result of shifting from the Logging to No Logging scenario in the NCCW, by level of road use (2008 prices)

Interest Rate (Capital Amortization)	Road Use	Change in Consumer Surplus for Watershed ($/year)	Change in Consumer Surplus for Watershed ($/ha/year)
1 %	Light	2,178.76	0.27
	Moderate	14,396.17	1.80
	Heavy	178,101.59	22.26
4%	Light	2,252.15	0.28
	Moderate	14,394.36	1.80
	Heavy	177,724.16	22.22
7%	Light	2,323.35	0.29
	Moderate	14,330.46	1.79
	Heavy	178,097.41	22.26

Table 5

Mean annual water treatment costs assuming a 100 year planning horizon and by forest management scenario (2008 prices and 4% capital amortization rate)

Scenario	Road Use	Total Annual Variable Cost ($/yr)	Total Annual Fixed Costs ($/yr)	Total Annual Cost ($/yr)
Logging	Light	773,662	30,821,720	31,595,382
	Moderate	782,251	30,821,720	31,603,971
	Heavy	898,586	30,821,720	31,720,307
No Logging		772,100	30,821,720	31,593,820

Figure 1

Location of the Norrish Creek Community Watershed in the Fraser Timber Supply Area, British Columbia, Canada

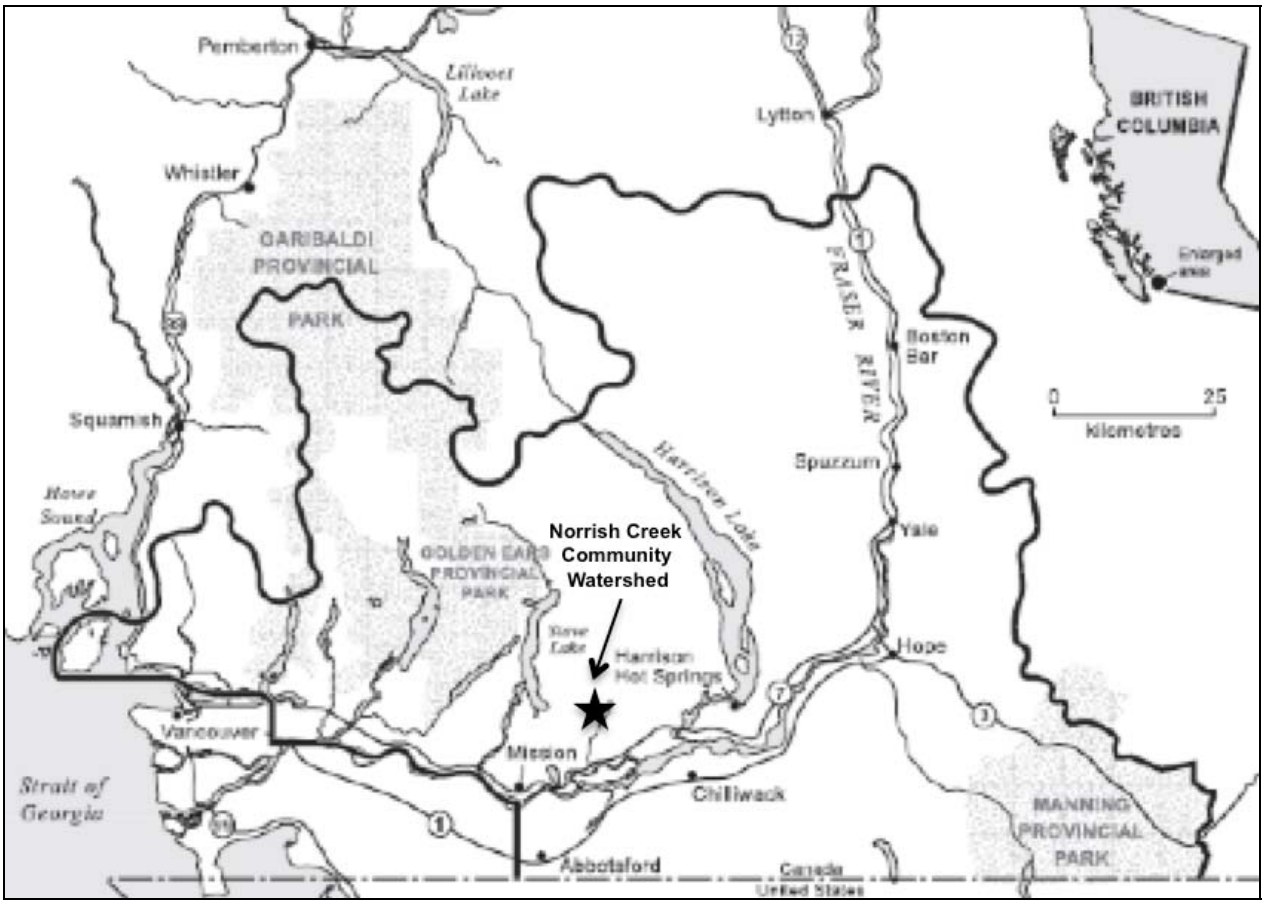


Figure 2

Conceptual approach to simulate water quality time series under alternative forest management scenarios from raw historical data (1984 to 2006)

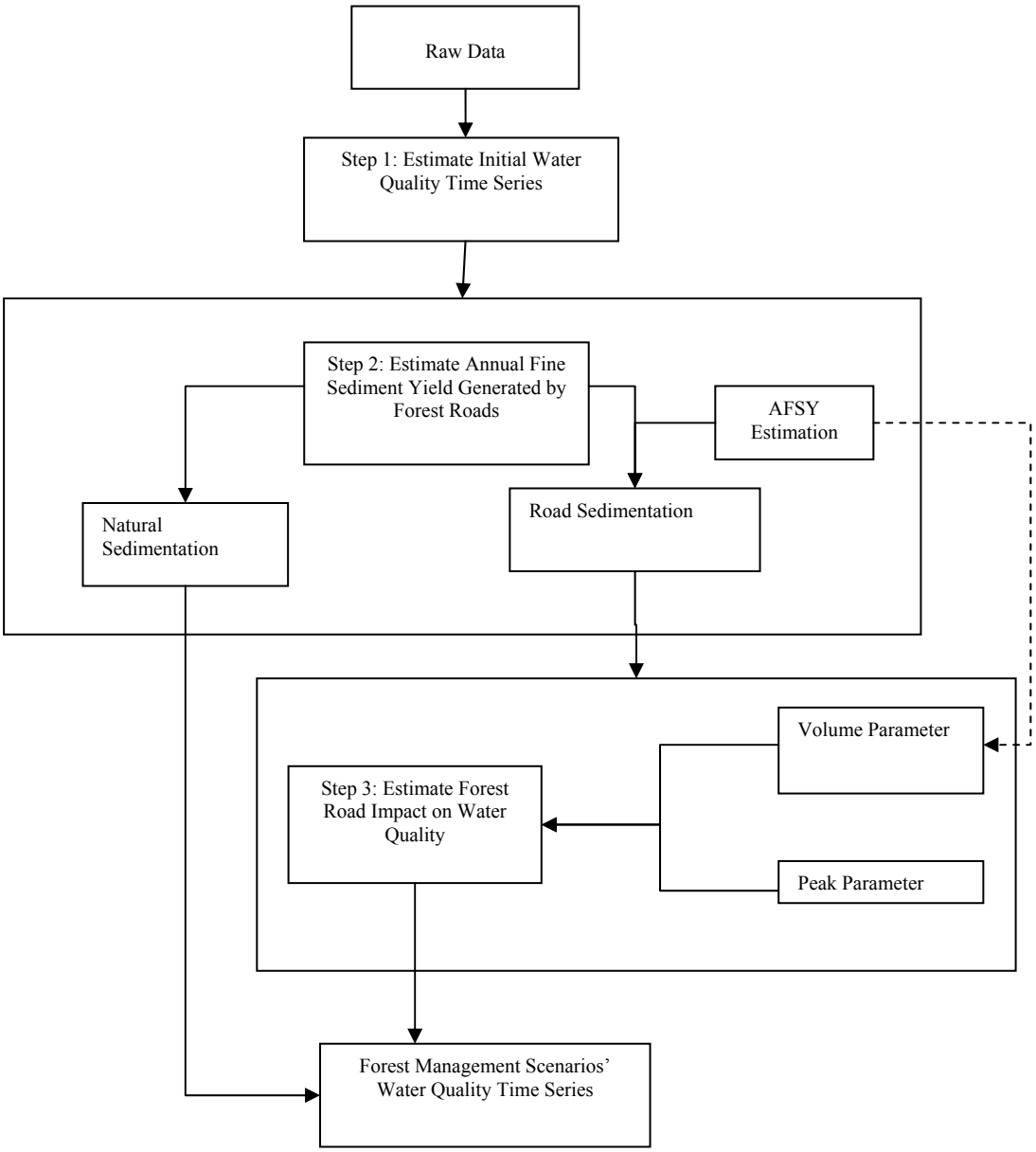
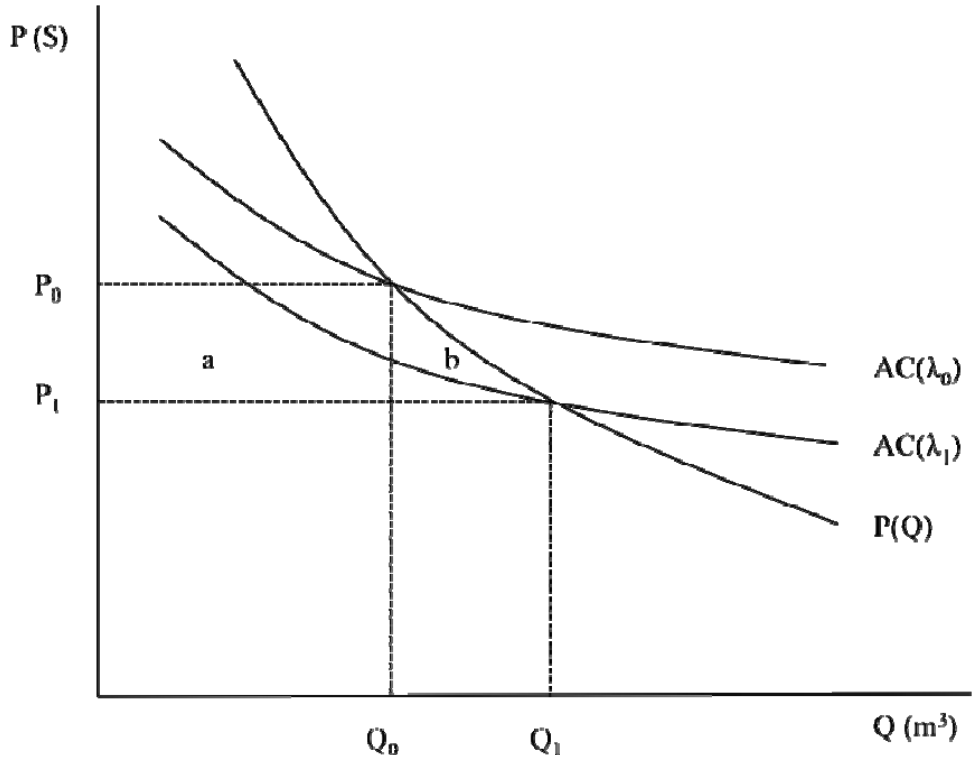


Figure 3

Welfare gain from improved raw water quality under an average cost pricing approach used by a water utility

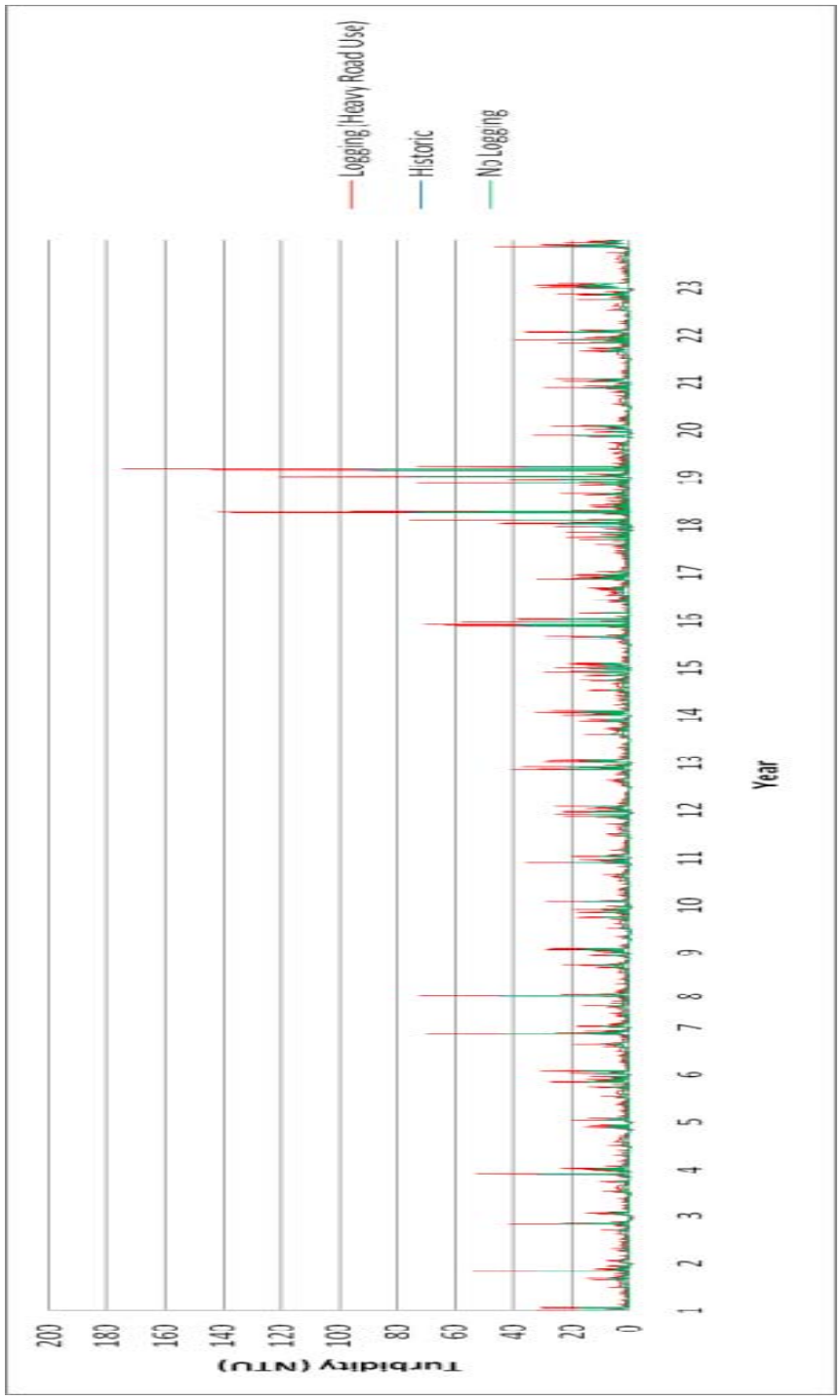


Source: adapted from Freeman (2003).

Note: welfare gain consists of consumer surplus only and comprises the sum of areas a and b

Figure 4

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



Appendix

Formulation and Parameter Estimation of a Linear Mixed Effects Model to Simulate a Time Series of Turbidity in Norrish Creek, British Columbia

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:

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

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:

$$\beta_{0j} = \beta_0 + b_{0j}, \quad \beta_{1j} = \beta_1 + b_{1j} \tag{A.2}$$

The covariance matrix $b_j = \begin{bmatrix} b_{0j} \\ b_{1j} \end{bmatrix} \sim N\left(\begin{bmatrix} 0 \\ 0 \end{bmatrix}, \xi\right)$ $\xi = \begin{bmatrix} \sigma_0^2 & \sigma_{01} \\ \sigma_{01} & \sigma_1^2 \end{bmatrix}$ 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 A.2 into equation A.1 leads to the linear mixed-effects model:

$$y_{ij} = \beta_0 + b_{0j} + \beta_1 x_{ij} + b_1 x_{ij} + \varepsilon_{ij,} \quad \varepsilon \sim N(0, \sigma^2) \tag{A.3}$$

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

In (A.3), 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 (A.3) 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). For further details on the model and parameter estimation, see Araujo et al. (2010).