A DECISION-MAKING FRAMEWORK TO IDENTIFY "TEMPERATURE-SENSITIVE STREAMS" FOR FOREST MANAGEMENT IN THE NORTH-CENTRAL INTERIOR OF BRITISH COLUMBIA

by

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Abstract

Forest harvesting is known to increase stream temperatures, which affect Pacific salmon, trout, and char (i.e., salmonids). In British Columbia (B.C.), a new *Forest and Range Practices Act* calls for the designation of "Temperature-Sensitive Streams" yet there is currently no methodology for their identification. My objective was to develop a framework for designating these streams. To do so, I analyzed data that managers would need to assess a stream's sensitivity, first compiling temperature data from 104 streams in the north-central interior of B.C. and then analyzing correlations among different measures of a thermal regime to identify a single temperature index. Next, I applied a regression tree analysis to examine the influence of a stream's watershed features and climatic setting on this index. I also used several temperature-dependent models to relate a stream's summer temperatures to a variety of salmonid responses. Finally, I used linear and Bayesian regression to analyze how forestry activities, summed across a watershed, influence stream temperatures.

To implement the proposed framework, I have four recommendations for scientists and managers. (1) Use a temperature index, such as the maximum of a 7-day average of the daily mean temperature (maximum weekly average temperature, MWAT), to characterize a seasonally variable thermal regime. (2) Use a stream's watershed features and climatic setting to identify stream-types with the most similar MWATs. (3) Identify the key temperature-driven responses of a fish community and quantitatively relate these responses to a MWAT. (4) Assess the magnitude and probability of

temperature increases from proposed forest practices, both at the local stream-scale and watershed-scale.

A stream's abiotic conditions would be considered "Temperature-Sensitive" if forestry activities are likely to increase temperatures beyond an acceptable range of variation. Furthermore, this designation would be warranted if temperatures are likely to increase beyond acceptable limits for a variety of temperature-responsive salmonid indicators. Forest practices should then be restricted to protect the abiotic conditions and biological processes in these sensitive streams. Failure to incorporate all this information into decision-making could result in the mis-identification of "Temperature-Sensitive Streams," thereby leading to unnecessary restrictions to forest harvesting or undesirable impacts to fish populations.

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Introduction

In British Columbia (B.C.), a new *Forest and Range Practices Act* calls for the designation of "Temperature-Sensitive Streams" (Province of British Columbia 2002). Currently, B.C.'s working definition considers a stream as temperature-sensitive if small temperature changes result in large changes to stream biota (E. Parkinson, Ministry of Water, Land and Air Protection, Vancouver, B.C., personal communication). This stream designation can lead to restrictions, such as a reduction in the removal of riparian vegetation, because forestry activities, such as harvesting and road-building, can lead to increases in stream temperature and deleterious effects on fish and fish habitat (reviewed by Beschta et al. 1987). However, there are no explicit methods for identifying these streams in B.C. This lack of clarity creates the potential for negative consequences to the forest industry and fish populations. Streams that are incorrectly designated as "Temperature-Sensitive" may result in unwarranted restrictions to forest harvesting. Conversely, streams that are incorrectly denoted as not "Temperature-Sensitive" may lead to forestry activities that detrimentally affect fish populations.

Understanding the relationship between land-use, fish habitat, and fish productivity is important because government management agencies focus efforts to protect fish populations by minimizing impacts on freshwater fish habitat (e.g., Department of Fisheries and Oceans 1986). Resource managers monitor fish habitat variables, in part because they are less variable, easier to measure, and more readily available than either estimates of fish abundance and productivity or the productive capacity of fish habitat. Furthermore, large interannual fluctuations in fish abundance can hinder the management and protection of fish populations because of the difficulty of relating fish abundance to the quality of the habitat (Rose 2000). Despite this challenge, long-term changes in fish abundance within the Pacific Northwest have been correlated with long-term changes in land-use activities (Bradford and Irvine 2000; Thompson and Lee 2000; Sharma and Hilborn 2001). Within B.C., impacts from land-use have contributed to the degradation of fish habitat and declines in salmon populations (Slaney et al. 1996). Forestry can affect fish and fish habitat by altering the abundance, distribution, and quality of woody debris, stream bed materials, and water in a stream (Meehan 1991). Given the potential for adverse effects and the difficulty of directly relating fish abundance to changes in the condition of fish habitat, there is a need to better understand how land-use practices influence fish habitat variables (e.g., how forestry affects stream temperature), and thereby contribute to declines in fish abundance.

Typically, resource managers minimize impacts from land-use by using one or a combination of the following approaches (Montgomery 1995). *Rules-based management* uses a set of prescriptions or "Best Management Practices (BMP)" to protect ecosystem processes and linkages (e.g., Young 2000). For instance, a BMP could state that a riparian buffer of at least 30 m should be maintained along fish-bearing streams with a channel width less than 5 m. *Results-based management* uses environmental monitoring combined with acceptable standards to avoid threshold violations and a degradation of resource condition (e.g., Nagpal et al. 1998). In this case, riparian areas could be harvested as needed with the restriction that these practices do not result in peak daily sediment loads greater than 25 mg/L, an acceptable threshold. The proposed designation of "Temperature-Sensitive Streams" in B.C. follows a results-based management approach.

Stream temperature is an excellent variable to monitor for results-based management because it meets Bauer and Ralph's (2001) criteria for selecting appropriate indicators of fish habitat condition. First, stream temperature is a biologically relevant indicator because it influences salmonid egg development and survival (Velsen 1987; Murray and McPhail 1988), as well as juvenile growth (Hokanson et al. 1977; Brett et al. 1982; Selong et al. 2001) and survival (Brett 1952; Hokanson et al. 1977; Selong et al. 2001). Water temperature can also influence fish distribution (Torgersen et al. 1999; Welsh et al. 2001; Dunham et al. 2003), abundance (Holtby 1988), and community composition (Wehrly et al. 2003), as well as macroinvertebrates (Vannote and Sweeney 1980; Pritchard et al. 1996; Hawkins et al. 1997), an important food source for salmonids. Second, we can distinguish forestry-induced changes to stream temperature (Brown and Krygier 1970; Feller 1981; Beschta and Taylor 1988; Hostetler 1991; Johnson and Jones 2000; Macdonald et al. 2003) from natural fluctuations because the processes controlling stream heating are relatively well understood. Third, we understand many of the factors that control variations in stream temperatures and can therefore manage at the appropriate time scale and stream-type (Poole and Berman 2001). Fourth, links between forestry activities, impacts on stream temperature, and effects on salmonids have been quantified (Holtby 1988; Macdonald et al. 1998). Finally, the accuracy, precision, and ease of measuring stream temperatures help to detect environmental change by reducing measurement errors and improving statistical power (Peterman 1990), as compared to measuring a variable such as survival rate of salmonids during some early life stage.

Even though designated stream temperature thresholds for salmonids differ across the Pacific Northwest (e.g., Nagpal et al. 1998; Oregon State 2002; Washington State

2003), management approaches in these jurisdictions share several common features. Numerous laboratory, field, and modeling studies have related land-use, water temperatures, and effects on fish and in some instances have been used to set temperature thresholds. That research has been used in qualitative assessments to identify temperature ranges that describe biological optimum for the species and life history stage of interest (e.g., McCullough 1999; Oliver and Fidler 2001; United States Environmental Protection Agency 2003). It has also been used in quantitative assessments of empirical or mechanistic models to predict the effects of temperature on fish (e.g., Brungs and Jones 1977; Armour 1991; Sullivan et al. 2000). Generally, there are two management approaches that are applied when setting water quality guidelines and stream temperature thresholds. The first recognizes that there is a range of water temperatures that are optimum for each species and life history stage. As a result, thresholds could be set so that temperatures do not cross the maximum or minimum of that optimum range (e.g., daily stream temperatures should not go outside a 9-to-13 °C range during egg incubation). A second consideration recognizes that there are natural fluctuations in stream temperatures due to the variation in watershed features, climatic settings, and natural disturbances among streams and across years. Consequently, there is some maximum deviation from natural conditions that is acceptable to managers; thresholds could be set within a range of these conditions (e.g., average stream temperatures should remain within 1 °C from average natural conditions).

Following these typical management approaches, a manager must define a stream's temperature-sensitivity based on how likely forestry activities are to increase temperatures beyond (1) a range of natural variation, or (2) a salmonid's optimal thermal conditions. However, there are problems with the way these two approaches can be applied to manage stream temperatures in B.C. From the perspective of a stream's biological community, sensitivity to temperature change is best viewed as a continuum without discrete thresholds that distinguish "Temperature-Sensitive Streams" from those that are not sensitive to water temperature changes. Therefore, forest managers may have difficulties defining acceptable limits to temperature changes and evaluating whether proposed forest practices will increase stream temperatures beyond those limits.

In addition, when attempting to keep temperatures within some range of natural variation, data need to be collected for long periods and across large spatial scales so that natural fluctuations among streams and across years can be discerned from anthropogenic impacts. In a management context, it is not practical to monitor temperatures for a long time in all streams that may be affected by forest harvesting. A better alternative is needed to describe a stream's baseline thermal conditions. Also, when maintaining temperatures within a salmonid's optimal thermal conditions, several indicators (e.g., juvenile growth, egg survival rate, and resistance to disease mortality) must be considered, otherwise limits set to protect a salmonid response, such as egg survival rate, with cool thermal requirements may negatively affect a response, such as juvenile growth, with warm thermal requirements. Therefore, quantitative models must be used to integrate the biological effects of a seasonally variable thermal regime and to properly evaluate trade-offs among salmonid responses and the effect of temperature increases due to forestry.

To accommodate these two management approaches and deal with their respective problems, my analyses were designed to illustrate how scientists can develop the tools and how forest managers can interpret the information needed to designate streams as "Temperature-Sensitive." First, I summarized the seasonal variability of a summer temperature profile into a single index to easily (1) analyze the factors that influence variations in thermal regimes among streams and across years, and (2) assess how a stream's thermal regime relates to several modeled salmonid responses. Second, I examined how a stream's watershed features and climatic setting explain differences in thermal regimes among streams and across years. Third, I used temperature-dependent biological models to estimate the effects of a stream's summer temperatures on several salmonid responses, specifically rainbow trout (*Oncorhynchus mykiss*) egg survival, juvenile growth, and resistance to disease mortality, as well as direct temperature mortality of rainbow and bull trout (*Salvelinus confluentus*) during the summer.

By applying the results from these three analyses, forest managers will be able to define acceptable levels of impact for a particular stream and evaluate the likelihood that proposed forest practices will increase temperatures beyond these levels. In particular, managers will be able to identify a stream's "Temperature-Sensitivity" in terms of either the abiotic conditions (i.e., thermal responses) or biological processes (i.e., salmonid responses) in a stream. These analyses help quantify the link between forestry activities and effects on fish, and support the use of stream temperature as an indicator of fish habitat condition. Understanding this linkage is also critical to forest management because of the potential for warming of streams due to climatic change.

Methods

The framework

A flow chart (Figure 1) illustrates how the information, analyses, and interpretations from this study fit into the proposed framework to identify "Temperature-Sensitive Streams." Details of these steps are provided later, but the general framework is as follows. First, I used a single year of temperature data from each of the 104 study streams to calculate 16 indices that described the different characteristics of a seasonally variable thermal regime (*Summarizing a thermal regime* in Figure 1). I then conducted pairwise comparisons of the correlations among these 16 indices to help identify a single index for my analyses and forest management.

Next, I addressed the concern regarding the management of stream temperatures within a range of natural variation by analyzing how a stream's watershed features and climatic setting influence variations in the identified temperature index (*Variation in stream temperatures*). In particular, I used regression tree analysis to stratify study streams by a few simple factors and to reduce the observed and unexplained variation in the temperature index. These results illustrate how managers can describe a range of variation in temperatures for particular stream-types, thereby helping to identify unacceptable limits to change in a stream's thermal regime and designate streams as "Temperature-Sensitive."

I also addressed the concern regarding the protection of optimum temperatures for more fish species and life history stages than the one with the coolest thermal requirements (*Thermal requirements of salmonids*) by modeling the response of four salmonid processes (e.g., egg survival, growth, resistance to disease mortality, and direct temperature mortality) to the daily temperatures in my study streams over the summer. I then used linear regression models to relate the end-of-summer model predictions to a stream's identified temperature index, and make generalizations about a stream's predicted thermal suitability. Comparisons among these relationships illustrate the positive and/or negative effects that increases in a temperature index may have on the identified salmonid responses in a stream. These comparisons could then be used by managers to identify "Temperature-Sensitive Streams" with respect to an acceptable limit to change in a stream's salmonid responses.

Finally, I examined some effects of forest harvesting and road-building on stream temperatures by measuring the watershed-scale activities in study streams (*Thermal impacts from forestry*). Specifically, I used linear regression and Bayesian regression to examine how each of four measures of watershed-scale activities can influence stream temperatures. When *Identifying "Temperature-Sensitive Streams"* for forest management, the results from all of these analyses are needed to evaluate whether new *Streams of management interest* are sensitive with respect to an acceptable (1) level of predicted impact, (2) predicted change in a thermal regime, and (3) predicted change in salmonid responses.

Study area

The 104 study streams were located within an area spanning approximately 106,000 km^2 of the upper Fraser and upper Skeena Rivers of British Columbia (Figure 2). This area lies at the northern extent of the Interior Plateau physiographic region, east of the Coast Mountains, and within the more mountainous terrain of the Nechako Plateau and Hazelton Mountains (Valentine et al. 1978). Forest ecosystems are dominated by the Sub-Boreal Spruce biogeoclimatic zone (Meidinger and Pojar 1991). Average annual air temperatures in Prince George and Smithers were similar between 1942 and 1999, with ranges from 0.8 to 5.8 °C and summer highs above 30 °C (Environment Canada 2001a). Average annual precipitation was also similar between these communities with ranges from 312 to 845 mm for the period from 1942 to 1999 (Environment Canada 2001a). Seasonal patterns of streamflow were characteristic of inland watersheds dominated by snowmelt runoff. Peak flows occurred in the spring as snowpacks melted with increasing air temperatures; flows declined through the summer to low flows over the fall and winter months (Environment Canada 2001b).

Fish communities are diverse and streams from the region support several economically and regionally significant salmonid species, notably rainbow (*Oncorhynchus mykiss*) and bull trout (*Salvelinus confluentus*), as well as chinook (*O. tshawytscha*), coho (*O. kisutch*), and sockeye salmon (*O. nerka*) (McPhail and Carveth 1993).

Forest management is the dominant land-use and an important contributor to the regional economy. The study area lay entirely within the Northern Interior Forest Region. Forest practices are guided by the provincial *Forest and Range Practices Act* and administered by four Forest Districts. This legislation has an important role in managing forestry practices and protecting streams within these Districts because 45 to 71% of the total land base is designated as Crown-owned productive forest (B.C. Ministry of Forests 2001a; 2001b; 2001c; 2002a; 2002b).

Summarizing a thermal regime

The characteristics of a thermal regime can vary among streams and across years. A thermal regime can be characterized by the annual peak of a temperature profile, frequency and duration above a specified temperature threshold, timing of maximum temperatures, or daily and seasonal patterns of heating and cooling. I selected 16 different indices of a summer temperature profile to measure these characteristics. Table 1 describes these indicators in detail.

To calculate these indices, I gathered a non-random sample of stream temperature data from 104 streams in the north-central interior. I restricted my evaluation of temperatures to the summer, from June 9 to September 15, because the warmest conditions occur during this period and warm temperatures can result in direct mortalities to salmonids. I used only one year of data from each of the 104 streams because I wanted this sample to include streams that had been exposed to contrasting climatic settings as described below. These years of stream temperature data spanned from 1990 to 2002. I could not analyze data across many years for each stream because almost 80% of sites had fewer than 3 years of stream temperature data available. Data were summarized by daily maximum, minimum, and mean temperature. To explore the correlations among these simple indices, I calculated Pearson's correlation coefficients for all pairwise comparisons among the 16 indices.

As water levels fluctuate over the summer, data recorders may become exposed to the air and measurement errors can result. I used the quality assurance criteria provided in the appendices of Lewis et al. (2000) to verify the accuracy of my stream temperature data. Erroneous daily values, which appeared as spikes and had excessively large daily fluctuations, were removed from the data set. I also inspected the plots and timing of peak summer temperatures to ensure that all data measured the rise and decline of a summer temperature profile.

For the analyses described below, I selected the maximum of a 7-day average of the daily mean temperature (MG(4) in Table 1) for four reasons. First, this maximum weekly average temperature (MWAT) is commonly used to manage stream temperatures (Nagpal et al. 1998; Oregon State 2002; United States Environmental Protection Agency 2003; Washington State 2003). Second, the MWAT was highly correlated with 9 other indices that similarly described a thermal regime's annual maximum temperature (e.g., all MG and TH indices – see *Summarizing a thermal regime* in Results). Third, the 10 indices that measured the annual peak of a temperature profile were responsive to changes in a stream's watershed features and climatic setting (e.g., drainage area, basin elevation, and air temperature – see *Variation in stream temperatures* in Results). Fourth, these 10 indices could be used to approximate the modelled biological responses in a stream (e.g., egg survival, growth, and resistance to disease mortality – see *Thermal requirements of salmonids* in Results).

Variation in stream temperatures

Defining ranges of natural variation — To address the problem of a manager normally being unable to monitor stream temperatures for a long time and across many streams to describe baseline conditions, I examined the influence of several factors on the spatial and temporal variation in a thermal regime. In particular, I used a stream's MWAT as my response variable and calculated seven watershed features and a climate index as my explanatory variables, described below (Table 2). The locations of stream temperature monitoring were mapped and spatial queries were conducted in ArcView 3.2 (Environmental Systems Research Institute (ESRI), Redlands, Ca), a geographic information system (GIS).

I then used regression tree analysis (Brieman et al. 1984) to stratify my 104 streams into groups that had the most similar thermal regimes. I used this statistical method to illustrate how managers can use a few simple explanatory variables to stratify streams into thermally distinct groupings. For each of these groupings, the variation in a MWAT index will be less than the variation in a MWAT index if all 104 streams were pooled. These groupings can then be used as a basis to classify new streams, identify an expected range of natural variation for those groupings, and help managers identify unacceptable or "Temperature-Sensitive" conditions. For example, streams with MWATs near the upper range for a given group of streams would be the most sensitive to effects from forestry because increases will likely result in temperatures greater than the observed range for that particular group. Conversely, MWATs near the middle of the range for a particular grouping would be less sensitive because increases are less likely to result in observations outside the observed range of temperatures.

Watershed features — I used seven variables to describe the watershed features associated with each stream temperature location: latitude, distance to the coast, average basin elevation, drainage area, channel orientation, biogeoclimatic zone, and surficial geology. I calculated latitude and distance from each stream to the coast using the coordinates provided with the temperature data. I measured these variables because I expected streams at higher latitudes and nearest to the coast to be exposed to cooler summer climate conditions than streams at lower latitudes and further from the coast. I then determined the point elevation at each monitoring location and the average upstream basin elevation using a 25-m resolution gridded digital elevation model (DEM). I used average upstream basin elevation in the regression tree analysis because it was more strongly correlated with stream temperatures than a point elevation, and was thought to provide a better reflection of the climate conditions that influence a watershed as water passes downstream from the headwaters. I also used the DEM to calculate the drainage area upstream from each stream site. Next, I calculated the orientation of a straight line between each monitoring location and a point 600 m upstream, and grouped each stream into one of four orientation classes (NW-NE, NE-SE, SE-SW, and SW-NW). Stream channels that had similar orientations were grouped together because it was assumed that they had similar exposures to the sun and would experience similar heating influences. I cross-referenced streams with biogeoclimatic polygons (Meidinger and Pojar 1991) to identify the riparian forest-type at each monitoring station and investigate potential differences in stream shading. Finally, I overlaid streams with a map of surficial geology (Fulton 1995) to identify the surficial materials at each monitoring station and investigate potential influences of groundwater zones on stream temperature. I suspected that streams with the most porous surficial materials would have the greatest groundwater influences and as a result, cooler stream temperatures.

Climatic setting — I used air temperatures from seven fire weather stations across the study area to examine the influences of regional and year-to-year differences in climate on stream temperatures (data provided by E. Meyer, Ministry of Forests, Victoria, B.C.). I assumed that these seven weather stations would estimate streamside climatic influences because Stefan and Preud'homme (1993) found that air temperatures measured at distant meteorological stations could be used to predict stream temperatures. To compensate for regional differences in summer air temperatures among stations and across years, I calculated annual deviations from a 13-year average summer air temperature (May 1 – August 31, 1990-2002) for each weather station. I used only one year of stream temperature data in all of my analyses because I wanted my sample to include streams that were exposed to contrasting climate conditions. Therefore, years with similar summer air temperatures represented year-types with either above-average, average, or below-average climate conditions. The value for a stream's air temperature index was then measured as the annual deviation at the nearest weather station and year in which stream temperatures were measured.

Regression trees — I used regression tree analysis (Brieman et al. 1984) to partition the MWAT variable using the eight explanatory variables described above. The regression tree algorithm was performed in S-Plus (Venables and Ripley 1999). This procedure recursively searched for values or categories within all explanatory variables to split the temperature data into two groups (termed 'nodes') and minimize the within-node variance. This splitting was repeated and a tree was generated until a minimum node size ($n = 10$) or minimum reduction in node variance (*complexity parameter* = 0.001) was reached.

I used a 10-fold cross-validation procedure to explore the relationship between tree complexity (i.e., number of terminal nodes or "Stream Temperature Classes" as termed in this study) and a measure of the tree error (i.e., uncertainty in the variables and values used for a split) (Venables and Ripley 1999). A 10-fold cross-validation procedure first split all 104 streams into 10 equally-sized groups. This procedure then generated a tree using 9 of the groups, and tested the accuracy of this tree by using it to classify streams in the tenth group. This testing was repeated 10 times, each time with a different group removed. In each trial, a measure of the accuracy of the tree, or tree error, was calculated as the number of terminal nodes increased. Measurements of tree error were then averaged across all 10 trials and plotted against tree size (i.e., number of terminal nodes). I chose a tree size that minimized the error calculated from this cross-validation procedure.

Thermal requirements of salmonids

Estimating salmonid responses — To address the problem of evaluating the acceptability of temperatures based on more than just the salmonid response with the coolest thermal requirement, I modeled the effect of daily water temperatures on several indicators of population processes in salmonids. Specifically, I used existing temperature-dependent biological models (described below) to examine the influence of summer temperatures on rainbow trout egg survival, growth, and resistance to disease mortality, as well as direct temperature-induced mortality of rainbow and bull trout (see Appendix A for equations and parameter values). Rainbow trout was the species of primary interest because (1) it is the most common salmonid in my study area, (2) the relationship between water temperature and the physiology of rainbow trout has been well studied, and (3) it is a spring spawner, which allowed me to use the available temperature data to assess effects on egg incubation. For those streams and years that had continuous records from June 9 to September 15 (31 of 104 streams), the models estimated a fish's response to a thermal regime by translating daily stream temperatures over the summer into more biologically meaningful measures of fitness (e.g., eggs survival, growth, or resistance to disease mortality).

I then examined the relationship between these measures of fitness and a MWAT to see if a simple temperature index would accurately represent a salmonid's modelled response to a thermal regime. I used linear regression to relate these response variables to the explanatory variable. The functional forms of these relationships were selected as those that qualitatively appeared to best fit the data. For the relationship between egg survival and MWAT I used an arcsine transformation to standardize variances and improve normality about the regression line because the egg survival variable was measured in proportions (Sokal and Rohlf 1995).

The simple temperature index, MWAT, adequately represented a salmonid's predicted response over the range of thermal regimes observed in the study area. Therefore, I compared the upper and lower range of MWAT values that predicted the highest biological responses (e.g., survival rate of eggs) for each of these relationships. I described this range by plotting the upper and lower MWAT values that resulted in a 5% reduction from the predicted maximum biological response (as used by Sullivan et al. 2000). Plots of these ranges helped to illustrate how managers should explicitly recognize the trade-offs among salmonid life history processes when identifying acceptable limits to temperature change. For example, a stream that is managed to protect MWATs that result in a maximum end-of-summer growth would result in a less-than-maximum proportion of eggs surviving from fertilization to hatch. By considering a number of salmonid responses, a forest manager will be better able to compare the potential positive and/or negative effects of given temperature increases due to forestry.

Model assumptions — The following models required three general assumptions. First, these models could only estimate the relative effects of temperature on the biological responses of fish. In addition to temperature, fish populations are controlled by numerous habitat variables that vary widely among streams. These models could not evaluate absolute effects because this additional habitat information was not available for all study streams. Second, I assumed that the models predicting responses for juvenile steelhead trout (*O. mykiss*), an anadromous life history form of rainbow trout, could be used to estimate responses for juvenile rainbow trout. Third, I assumed that the predicted responses, derived from lab experiments using constant temperatures, would represent responses in streams that have diurnally and seasonally variable temperatures. Even though temporally variable temperatures are known to influence rainbow trout differently than constant temperatures (Hokanson et al. 1977), this level of complexity has not been incorporated into the following models.

Egg survival rate — To estimate the effects of stream temperature on rainbow trout egg survival rate, I used the models from McLean et al.'s (1991) and Jensen et al.'s (2002) "Salmonid Incubation and Rearing Program" to first predict the number of days from fertilization to the date when 50% of eggs are expected to hatch, and then predict the proportion of fertilized eggs surviving to that median hatch date. Given that rainbow trout normally spawn during the spring months after freshet and the timing of the available temperature data, I assumed an egg fertilization date of June 9, which is reasonable for streams within my study area (H. Herunter, Fisheries and Oceans Canada, Burnaby, B.C., personal communication). McLean et al. (1991) used Schnute's (1981) development model to calculate the number of days to the median hatch date (D) at a constant water temperature (T):

$$(1) \qquad D=\left[D_1^{\,b}+\frac{\left(D_2^{\,b}-D_1^{\,b}\right)\cdot\left(1-e^{-a(T-T_1)}\right)}{\left(1-e^{-a(T_2-T_1)}\right)}\right]^{\frac{1}{b}}$$

in which a and b defined the shape of the curve, and D_1 and D_2 were the predicted development times to the minimum (T_1) and maximum (T_2) incubation temperatures from a data set. The parameterization of this equation was based on experiments using constant incubation temperatures (Velsen 1987). Because stream temperatures fluctuate over the summer, I used a daily mean temperature in equation (1) to calculate a daily contribution to incubation development (i.e., $1/D$ as used by Clark and Rose 1997). Therefore, daily fractional contributions were accumulated until incubation development reached 1, at which time the median hatch date was predicted.

Next, I used Jensen et al.'s (2002) polynomial function (derived using data from Velsen 1987) to describe the relationship between proportion of eggs surviving from fertilization to hatching (s) and constant water temperature (T):

$$(2) \qquad \begin{array}{l} s=1-\left(a+bT+cT^2\right) \quad \textit{if} \quad 0<T<18.75 \quad \textit{else} \\ s=0 \end{array}$$

in which a, b, and c determined the shape of the parabola. Equation (2) predicted egg survival rate from fertilization to hatching in a constant temperature environment. To incorporate more realistic seasonal fluctuations, I converted the proportion of eggs surviving over a life stage to a daily measure. I first assumed that the proportion of eggs surviving (s from equation (2)) over the incubation period (D from equation (1)) would follow an exponential decay function:

(3) $$s = e^{-MD}$$

in which M described the mortality of eggs. I then solved equation (3) for M, to calculate the mortality on a particular day (i) given the predicted survival (s_i) and development time (D_i) from a daily mean stream temperature:

(4) $$M_i = \frac{-\log_e(s_i)}{D_i}$$

Finally, equation (4) was used to calculate the proportion of eggs surviving (E) to the predicted date of median hatch (t) in a particular stream:

(5) $$E_t = \prod_{i=0}^{t}(1 - M_i)$$

Growth — To estimate the effects of temperature on growth of juvenile rainbow trout, I used the steelhead trout growth model described by Sullivan et al. (2000). This model's equations are too detailed to describe fully here; see Appendix A for additional details. They developed this model in two parts. The first was based on a bioenergetics model (Hanson et al. 1997); it described the relationship between consumption (C_i), temperature (T_i), weight (W_i), and food ration (R_i) on a particular day (i):

(6) $$C_i = f(T_i, W_i, R_i)$$

The second part, developed by Sullivan et al. (2000), described the relationship between specific growth rate (g_i), temperature (T_i), and consumption (C_i) on a particular day (i):

(7) $$g_i = f(T_i, C_i)$$

These relationships calculated weight (w) over the summer up to day (t) (i.e., 99 days from June 9 to September 15) using:

(8)
$$w_t = w_0 \prod_{i=0}^{t} (1 + g_i)$$

in which w_0 was the initial weight of a fish. Stream temperature can also influence the development (Pritchard et al. 1996) and community composition (Vannote and Sweeney 1980; Hawkins et al. 1997) of macroinvertebrates, an important food source for salmonids. Therefore, to investigate the relationship between growth, temperature, and food supply, I repeated the weight calculations at four food rations, 40, 60, 80, and 100% satiation.

Resistance to disease mortality — Water temperatures influence mortality from diseases by influencing a fish's immune response (Roberts 1978) and growth of some bacterial diseases (Holt et al. 1975). To examine the resistance of rainbow trout to disease mortality, I first selected two common bacterial diseases, *Flexibacter columnaris* and *Aeromonas salmonicida* because of their strong response to warmer water temperatures (Roberts 1978), their occurrence in salmonids within the Pacific Northwest (as cited by Fujihara et al. 1971; Holt et al. 1975), and the lack of information regarding the distribution of fish diseases in British Columbia (Sherri Guest, Ministry of Water, Land and Air Protection, Nanaimo B.C., personal communication). I then compiled data that related the proportion of juvenile rainbow or steelhead trout surviving to water temperature (Fryer and Pilcher 1974; Fryer et al. 1976). These data described a typical sigmoid "dose-response" relationship between mortality of exposed fish and constant water temperatures from 3.9 to 23.3 °C. Next, I used probit analysis (Finney 1971) to estimate the water temperature that resulted in 50% mortality of the exposed sample of fish (i.e., median lethal temperature – LT_{50}). I only used data from trials at 12.2, 15.0, and 17.8 °C because observations with very low or high mortality could bias the estimate of LT_{50}. Water temperatures for these trials were also well below lethal values for rainbow trout, which meant that mortalities were the result of disease exposures and not related to temperature-induced mortalities. I then created an index of resistance to disease mortality by summing the number of days that a stream's daily maximum temperature was below the median lethal temperature. Streams with a high index value and a greater number of days below the median lethal temperature had thermal conditions that were more favourable for survival of exposed fish and less favourable for these diseases.

Direct temperature mortality — I also examined the susceptibility of rainbow trout, a salmonid which can withstand one of the highest absolute temperatures, and bull trout, a salmonid which has one of the lowest absolute temperature tolerances, to mortality from warm water temperatures. I identified temperatures that resulted in 50% mortality of a test sample of fish over a 7-day test period (LT_{50}) for juvenile rainbow (Hokanson et al. 1977) and bull trout (Selong et al. 2001). Lethal temperatures for rainbow trout were considerably higher than the lethal temperatures due to the two diseases discussed previously. Therefore, I assumed that fish mortalities in these studies were related to temperature effects only and not the result of disease exposures. I then created an index of direct temperature mortality which summed the number of days over the summer that a stream's daily maximum temperature exceeded these lethal thresholds. Although important, this index did not consider the duration of exposure or the timing of high temperatures in relation to life history events because these effects could not be estimated from the available data.

Thermal impacts from forestry

Analyzing effects — To help managers assess the likelihood and magnitude of effects of forestry activities and evaluate whether proposed forestry practices will exceed acceptable limits to temperature change, I examined the influence of forest practices on stream temperatures. The effect of local stream-scale activities (e.g., riparian and upslope harvesting) on stream temperature has been well documented (e.g., Brown and Krygier 1970; Feller 1981; Johnson and Jones 2000; Macdonald et al. 2003). However, these effects could not be assessed in this study because measures of local activities were not recorded at each stream. In contrast, the effects of watershed-scale activities (e.g., density of roads and proportion of a watershed harvested) have not been as well documented and results are conflicting (e.g., Beschta and Taylor 1988; Bettinger et al. 1998; Zwieniecki and Newton 1999). These activities were described for some of my study streams and I could therefore use these measurements for my analyses.

I started this evaluation with the Stream Temperature Classes identified by the regression tree analysis. The stratification from this procedure attributed some of the variation in a temperatures index to a stream's watershed features and climatic setting, thereby improving the detection of effects of watershed-scale influences on stream temperature. I then used an existing land-use database to describe four measures of watershed-scale activities in the study drainages as described below. Next, I plotted the difference between a stream's MWAT and the average MWAT for that stream's grouping (i.e., regression tree residuals) against each of the four measures of forest development. I only evaluated these effects for a single Class of streams because I was not able to measure watershed-scale activities for more than a small subset of the study streams (see explanation that follows).

I used two approaches to examine the relationship between the regression tree residuals and each of the four measures of land-use. Linear regression was used to test the null hypothesis that there was no effect of watershed-scale activities on stream temperatures (i.e., slope of the regression was zero). Because forest harvesting activities were only expected to increase stream temperatures, I used a one-tailed statistical test. This traditional method determined the likelihood that the observed data were sampled from a population with a slope parameter of zero. We would fail to reject the null hypothesis if it was likely (e.g., a P-value greater than 0.05) that these data were sampled from a population in which the null hypothesis was true. In contrast, we would reject the null hypothesis if it was unlikely (e.g., a P-value less than 0.05) that the data were sampled from a population in which the null hypothesis was true. In this second case, we would therefore conclude that the data were sampled from a different population of streams with some fixed non-zero slope parameter.

Though common, there are two problems with this approach. First, inferences from the data are limited to interpretations about only two states of nature, even though many more may be possible. Therefore, the use of a linear regression implies that the slope parameter can be either zero or some fixed non-zero value. Second, the failure to detect a significant effect may be the result of a test with low statistical power and not because there is no effect at some specified important effect size (Peterman 1990). Power is a function of sample size, sample variance, true effect size, and the level of statistical significance. Important effects may therefore go unrecognized by decision-makers because a non-significant test with low power has resulted from a poor sampling design. Thus, I performed a retrospective power analysis for the two tests that examined the effect of roads because sample sizes were small and P-values were close to 0.05.

As an alternative I used Bayesian regression (Press 1989) because it does a better job of quantifying uncertainty in the slope parameter than a linear regression. Specifically, it uses the data to estimate the range of underlying slope values that are possible and estimates a degree of belief (i.e., posterior probability) in those values. This method also allows for results from other independent studies (i.e., prior information) to be incorporated into the analysis. I did not use additional information here; rather I used an uninformative prior probability distribution. In addition, I used this Bayesian technique due to concerns over traditional statistical methods (Johnson 1999), especially when used to make inferences for environmental decision-making (Reckhow 1994; Ellison 1996; Wade 2000).

Watershed-scale activities — I used B.C.'s Watershed Statistics database (Ministry of Sustainable Resources Management 2002) to initially summarize six measures of watershed-scale activities: (1) drainage area logged – the proportion of a watershed that had been logged (> 15 ha) or selectively logged (> 30 ha) within the previous 20 years (km^2 of logged area per km^2 of watershed area), (2) riparian area logged – the proportion of 1:20,000 scale streams in a watershed that had been logged or selectively logged to the bank (km of logged riparian area per km of stream within a watershed), (3) roads – the density of roads within a watershed (km of road per km^2 of watershed area), (4) road crossings – the density of road-stream crossings within a watershed (number of road crossings per km^2 of watershed area), (5) non-forestry land-use – the proportion of a watershed that had agricultural, urban, or mining land cover designations (km^2 of non-forestry land-use per km^2 of watershed area), and (6) fire disturbances – the proportion of each watershed that had been burned (> 30 ha) within the previous 20 years (km^2 of burned area per km^2 of watershed area). In the end, I did not use indicators of non-forestry land-use or fire disturbance because these variables were only observed at low levels in the study watersheds.

These watershed-scale influences were calculated using three GIS coverages: 1:50,000 scale watershed delineations, 1:20,000 stream and road information from Terrain Resource Information Management (TRIM), and land-use cover from Baseline Thematic Mapping (BTM). The spatial extent of roads was calculated using TRIM data from 1979 to 1988, while land-use cover was compiled using BTM data from 1990 to 1997. Measures of watershed-scale activities could only be summed over 20 years for predefined watershed polygons. In many instances, these watershed polygons did not coincide with the drainage areas associated with the stream temperature locations. Therefore, only a small subset of the stream temperature data could be analyzed for effects from upstream land-use (see last column of Table 2).

Results

Summarizing a thermal regime

I compared 16 temperature indices to help me select a suitable index for my analyses and recommend one for forest management. Indices were grouped according to the way in which they measured the annual peak of a temperature profile (*MG*), number of days that temperatures exceed a threshold (*TH*), daily fluctuation in temperatures (*DF*), seasonal rate of temperature change (*RT*), or timing of annual maximum temperatures (*TM*) over the summer. Pairwise comparisons of the correlations among these indices (Figure 3) revealed strong correlations within-groups (compare MG(1) and MG(2)) and poor correlations between-groups (compare MG(1) and DF(1)). Within-group comparisons showed that indices that describe a similar aspect of a summer thermal regime (e.g., MG and TH – measures of the annual peak of a temperature profile) were significantly ($P < 0.05$) and positively correlated ($r > 0.55$). Between-group comparisons also revealed significant ($P < 0.05$) and negative correlations ($r < -0.55$) between a rate of temperature decrease (RT(2)) and the MG and TH indices. All but one of the other between-group comparisons showed poor correlations ($-0.55 < r < 0.55$).

The strong correlations among 10 indices suggested that there was some overlap among these and that only one was needed to describe the annual peak of a temperature profile. These 10 indices were poorly correlated with most other indices, which suggested that they described different aspects of a thermal regime. I narrowed my selection to these 10 indicators on the basis of the results from other parts of this study. First, an exploratory analysis revealed that these 10 measures were the only ones that could be related reliably to the landscape and climatic factors used in the regression tree analysis. Also, our understanding of the biological effects of water temperatures is so limited that I could not find models to predict the effect of forestry-induced changes in any of the other 6 indices (DF(1), DF(2), DF(3), RT(1), RT(2), and TM(1)). I then selected a MWAT, or MG(4), because this measure is commonly used to manage stream temperatures.

Variation in stream temperatures

To be better able to manage forest practices and protect stream temperatures within a range of natural variation, the regression tree analysis helps managers because it uses the study streams' watershed features and climatic setting to identify stream-types with the most similar MWATs. As illustrated by the dendrogram in Figure 4, two landscape variables, *Drainage* and *Elevation*, and one climate variable, *Air Temp*, stratified the MWAT data into thermally distinct groupings and reduced the within-group variation by more than half (from 17 °C to a range between 4 and 7 °C). The 10-fold cross validation procedure indicated that a tree with five splits and six terminal nodes, or Stream Temperature Classes, provided the best fit to the MWAT data. The first split partitioned the temperature data into two groups with the most similar thermal regimes; streams with smaller drainage areas < 132 km^2 tend to have cooler MWATs, whereas streams with larger drainage areas ≥ 132 km^2 tend to have warmer MWATs. The splits near the top of the tree reduced the variation in a MWAT more than the lower splits. Classes I, II, V, and VI describe groups of streams that were characterized by different average basin elevations and drainage areas, whereas the watershed features describing Classes III and IV were the same. Interestingly, for streams in Classes III and IV, *Air*

Temp, the variable that estimated interannual and regional differences in summer air temperatures was important in reducing the variation in a MWAT even further.

Thermal requirements of salmonids

Forest managers must also manage forest practices and stream temperatures to protect a variety of salmonid characteristics or indicators in a stream. Data points in Figure 5 represent the end-of-season model predictions for egg survival, growth, and resistance to disease mortality plotted against the MWAT index calculated for the 31 of 104 study streams for which appropriate data were available. The solid lines represent the best-fit relationships between these observations. All regression models fit the data relatively well and had R^2 values > 0.84. These modeling results can help with decision-making because they encourage managers to explicitly compare the positive and/or negative effects of temperature increases due to forest harvesting or road-building among more salmonid response variables than simply the one with the coolest thermal requirements.

The predicted shape of the relationships between these biological responses and a MWAT index provide three main observations about the thermal requirements of salmonids. First, in spite of the differences in thermal regimes among streams, a MWAT index represented a modeled biological response relatively well because the scatter about the regression line was small. Second, the direction of a stream's biological response to temperature increases caused by forestry activities will depend on where the measured MWAT lay upon the biological response curve. A MWAT of 12 °C would predict increasing growth in body weight with increasing temperatures to about 14 °C (Figure

5b), beyond which decreasing growth would be expected. The direction of these responses will also depend on the variable of interest. For example, egg survival and growth rates are different for a range of temperatures. Finally, the MWAT values that predict the maximum of each biological response variable differed among variables. As Figure 6 illustrates, MWATs resulting in maximum growth at four food rations (40% - 14.5 °C, 60% - 15.6 °C, 80% - 16.1 °C, and 100% satiation - 16.4 °C) were higher than the MWATs predicting maximum egg survival (8.4 °C) and resistance to mortality from two diseases (*Aeromonas salmonicida* - 10.8 °C and *Flexibacter columnaris* - 12.5 °C). Furthermore, the MWAT predicting maximum growth increased, from 14.5 to 16.4 °C, with increasing food supply, from 40% to 100% satiation. The relationship between a measure of direct temperature-induced mortality and a MWAT index was not provided because, across the study area, there were few streams and days that exceeded lethal limits for rainbow and bull trout.

Thermal impacts from forestry

When evaluating a stream's "Temperature-Sensitivity" prior to forest harvesting, managers must assess the potential magnitude and likelihood of temperature increases from proposed forest practices. The analyses of the relationships between the regression tree residuals (i.e., indicator of a stream's MWAT value) for each stream and four measures of watershed-scale activities provided some insight into this assessment. The regression tree residuals represented the difference between a stream's MWAT and the average MWAT for that stream's Class. The residuals thus reflected the unexplained variation among streams around the mean across streams. Based on the results from the linear regression, there were no significant relationships (all $P > 0.05$) between the regression tree residuals in Stream Temperature Class II, and any of the four measures describing the level of forestry development in a watershed: (1) the proportion of the upstream basin logged, (2) the proportion of streams logged to the banks, (3) the density of roads within the upstream basin, and (4) the density of road crossings within the upstream basin (Figure 7). The effect of forest practices on streams from the other five Classes was not determined because watershed-scale forestry statistics were not available on enough streams to examine these relationships (sample sizes ≤ 7).

Although not statistically significant, there is a tendency for streams in Stream Temperature Class II to have warmer stream temperatures when associated with a higher density of roads (Figure 7c, $P = 0.07$) or road crossings (Figure 7d, $P = 0.25$). A retrospective power analysis revealed that these tests had insufficient power (< 80% probability) to detect small slopes at a 0.05 level of significance. For example, the relationship between the regression tree residuals and road density had a 22% and 66% chance to detect a slope of 0.5 and 1.0, respectively, while the test for an effect of the density of road crossings only had a power of 20% and 60% to detect the same slopes.

Alternatively, the marginal posterior probability distributions from the Bayesian regression offer a potentially more informative interpretation of the relationship between stream temperatures and roads (Figure 8). These distributions represent the degree of belief in different values for the slope parameter from the linear regression models presented in Figure 7c and 7d. A comparison of the two distributions in Figure 8 reveals that there is a narrower distribution and greater certainty in our belief about the true value of the slope of the relationship between road density and stream temperature (Figure 8a), than in our belief about the slope of the relationship with road crossing density (Figure 8b).

These distributions can also be used to illustrate the degree of belief in, or probability associated with, three scenarios about the 'true' relationship between stream temperature and roads; there may be (1) no true effect (i.e., slope ≤ 0), (2) a moderate effect (i.e., $0 < \text{slope} \leq 2$), or (3) an extreme effect (i.e., slope > 2 or a greater than 2 °C increase in a stream's MWAT for each km of road or road crossing per km^2 of watershed area). By summing the area under the probability distributions between these ranges, we can infer that there is a 7% probability of no true effect, a 78% probability of a moderate effect, and a 15% probability of an extreme effect of road density on stream temperatures (Figure 8a). Similarly, there is a 26% probability of no true effect, a 62% probability of a moderate effect, and a 12% probability of an extreme effect of road crossing density on stream temperatures (Figure 8b).

The independent variables reflecting human activities in the regression relationships were not all statistically independent. The proportion of the drainage area logged was correlated with the proportion of riparian area logged ($r = 0.97$), as was the density of roads and road crossings ($r = 0.93$). However, the measures of harvesting were not correlated with the measures of roads ($r = 0.25$ to 0.47).

Discussion

Important findings

The results from the Bayesian regression documenting the effect of roads and watershed-scale activities on stream temperature are important for several reasons. First, understanding this influence may be biologically important because other studies have found negative correlations between salmonid abundance and road density (Bradford and Irvine 2000; Thompson and Lee 2000; Sharma and Hilborn 2001). Forest roads affect streams and salmonids in other ways (reviewed by Furniss et al. 1991), but impacts on stream temperature have not been well documented (Herunter et al. 2003). Second, conclusions about the effects of watershed-scale influences on stream temperatures are conflicting (Beschta and Taylor 1988; Bettinger et al. 1998; Zwieniecki and Newton 1999). Therefore, it is critical from a forest management perspective to properly understand the effects of roads and forest harvesting so that forest practices can be managed at the watershed-scale and effects on fish and fish habitat can be minimized.

The empirical results presented here are consistent with field evidence from Herunter et al. (2003). In their study, temperature increases at stream crossings were more pronounced than those observed in streams passing through cutblocks. This finding implied that stream heating could be attributed, in part, to the local effect of roads as well as the local effect of riparian harvesting. They supported this interpretation by proposing that groundwater flow and exchange into the stream may have been altered by the roadbed. My results offer additional insight because I found a greater effect of road density on stream temperature than road crossing density, which suggests that the road network across an entire watershed influences stream temperatures to a greater extent than localized stream crossings. Therefore, the influence of roadbeds on stream temperatures may be driven by changes to groundwater flow and exchange across a watershed and not just the localized effects of roads.

This study also developed a new comprehensive framework to manage stream temperatures that explicitly recognizes (1) the factors that affect variation in a thermal regime among streams and across years, and (2) the trade-offs among several temperature-dependent salmonid responses. These types of information were previously available, but other management studies neglected to combine these two aspects into a management framework for setting stream temperature guidelines (Sullivan et al. 2000; Oliver and Fidler 2001; United States Environmental Protection Agency 2003).

The regression-tree and modeling results are consistent with the findings from other studies. In particular, many studies have reported a downstream warming trend or a positive relationship between stream temperatures and *Drainage Area* (Torgersen et al. 1999; Zwieniecki and Newton 1999; Lewis et al. 2000), and a negative relationship between *Basin Elevation* and stream temperatures (Isaak and Hubert 2001). The positive influence of an *Air Temperature Index* was also expected because other studies have used air temperatures to predict stream temperatures (Cluis 1972; Stefan and Preud'homme 1993). Finally, the modeling results are consistent with other studies that have related a variety of salmonid responses in a stream to simple temperature indices (Sullivan et al. 2000; Welsh et al. 2001; Dunham et al. 2003; Picard et al. 2003; Wehrly et al. 2003).

Identifying "Temperature-Sensitive Streams"

Summarizing a thermal regime — The framework developed here was based on the simple characterization of a thermal regime. This simplification was necessary because the complexity of a daily and seasonally variable temperature profile complicates our understanding of the biological responses to a stream's thermal regime and of the influence of natural phenomena and anthropogenic activities on stream temperatures. The results from the pairwise comparisons, regression tree analysis, and biological modeling helped to select a suitable index for analyses and recommend one for forest management. In particular, I proposed a MWAT index for the reasons stated previously. However, by considering a single temperature indicator, managers should recognize that they may be limited in their ability to manage changes to other aspects of a stream's summer thermal regime. This constraint could have biological implications because streams with different annual maximum temperatures and daily temperature fluctuations are generally associated with different fish communities (Wehrly et al. 2003) and macroinvertebrate groups (Vannote and Sweeney 1980).

Strong correlations among certain indices of temperature and lack of correlations among others are reasonable from a stream heating point of view. Other studies have also shown strong correlations among indices that describe the peak of a temperature profile (Lewis et al. 2000; Sullivan et al. 2000). These indices may be highly correlated because they all describe the same annual heating influence of the sun in different, but related, ways. In addition, the large amount of heat needed to warm a unit of water (i.e., specific heat capacity of water) suggests that streams must accumulate significant energy to reach an annual maximum. Therefore, different streams that have similar summer averages will also have similar MWATs because the annual cycle of the sun will ensure consistency in stream heating patterns among streams, and the amount of energy needed to heat water to a specified summer average temperature will be similar among streams.

A lack of correlation between different categories of indices may be the result of different groups measuring different physical heating and cooling processes in a stream. Indices that measure the seasonal peak of the temperature profile (i.e., MG and TH groups) may not be correlated with indices that describe the daily fluctuation in temperatures (i.e., DF group) because each reflects a different temporal pattern to stream heating. MG and TH groups reflect an annual pattern to stream heating and are influenced by processes that operate across a season (e.g., average summer climate conditions). In contrast, DF indices describe a stream heating pattern that follows the daily cycle of the sun and are influenced by changes at a shorter time-scale (e.g., daily air temperature, wind, and cloud cover conditions). Poor correlations are also observed when comparing spring heating (RT(1)) and fall cooling (RT(2)) rates of temperature change. The steepness of the stream heating and cooling shoulders of the annual temperature profile is most likely dominated by the spring and fall climate conditions. A lack of correlation between spring and fall climate processes would explain the lack of correlation between RT(1) and RT(2).

Variation in stream temperatures — When *Identifying "Temperature-Sensitive Streams"* a forest manager will need to define acceptable levels of impact relative to an expected range of variation in temperatures. The regression tree analysis will help to do this because it uses a few simple watershed and climatic descriptors to group a sample of streams into Stream Temperature Classes and reduce the variation in a MWAT (Figure 4).

A forest manager will then be able to measure these descriptors for a new *Stream of management interest* to assign it to one of the groupings and determine the likelihood of observing that stream's MWAT given the range of MWATs for that Class. New streams with MWATs near the upper range for a particular group (e.g., 90th percentile) would be the most "Temperature-Sensitive" because effects from forestry activities are likely to increase temperatures beyond the natural range of variation. Conversely, MWATs near the middle of the range (e.g., 50th percentile) would be less sensitive to negative effects from forest practices because temperatures are less likely to increase outside the natural range after forest harvesting. The difficult part of implementing this consideration comes from defining an unacceptable level of impact and the thermal conditions under which a stream will be considered "Temperature-Sensitive." Sensitivity should be viewed as a continuum, and managers should avoid defining discrete numeric thresholds or percentiles that distinguish sensitive from not sensitive.

Stream classifications and managing within ranges of natural variation are useful approaches when implemented properly. A classification scheme provides a means by which to group or stratify streams based on the similarity of a stream's physical or biological variables (e.g., Montgomery and Buffington 1998; Naiman 1998). Stratification assists with drawing conclusions relevant to management because of the associated reduction in variation. This management approach also recognizes that spatial and temporal variability are vital for ecological systems (Holling and Meffe 1996; Landres et al. 1999). In the case of stream temperature, it is recommended (Poole et al. 2004) because thermal heterogeneity helps structure stream ecosystems (Magnuson et al. 1979; Vannote et al. 1980; Hughes 1998; Wehrly et al. 2003).

Thermal requirements of salmonids — A forest manager will also need to define an acceptable level of impact relative to the thermal requirements of a variety of salmonid response variables because conditions cannot be optimum for egg survival, growth, and resistance to diseases in a single stream. The modeling results will help with this challenge because they predict the suitability of a stream's thermal regime for a variety of salmonid responses relatively well (Figure 5). Once these and similar relationships are defined, a forest manager can collect the fish species, life history, and stock status information relevant to a new *Stream of management interest*. Managers will then be able to compare whether temperature increases due to forestry activities will have a positive, negative, or no biological effect and identify acceptable levels of impact that they feel adequately protects a number of salmonid processes. For example, streams with a MWAT that predicts the maximum growth response may be considered sensitive to additional stream warming because increases in a MWAT would predict a reduced growth response, a less-than-maximum egg survival rate, and a decreased resistance to disease mortality. Again, the difficult part of determining a stream's sensitivity relative to these salmonid responses will come from defining an unacceptable level of impact and the biological conditions under which a stream will be considered "Temperature-Sensitive."

To define these unacceptable levels, managers need to make the trade-offs explicit among different salmonid responses. Trade-offs can be evaluated with life history models that relate thermal impacts at the individual-level to a population-level response (e.g., Holtby 1988), or with individual-based models that translate habitat alterations to effects at higher biological levels (as advocated by Rosenfeld 2003). Alternatively, a manager may weight the protection of thermal conditions for a biological response differently depending on the stock status or productivity of the stream. For example, if fish populations, such as bull trout, are depressed relative to historic abundances, a manager may wish to protect the thermal conditions for maximum juvenile recruitment, including high egg survival rate and low disease susceptibility, rather than higher growth rates. Also, in unproductive waters, where food rations are low because of natural or anthropogenic influences, temperature increases may provide no growth benefit, just a reduction in egg survival rate and increased disease susceptibility. Similarly, streams with high population densities may experience no growth benefits from temperature increases because competition for limited resources is high and density-dependent factors may be limiting growth. By considering a variety of salmonid responses and the local biological conditions in a stream, a manager can evaluate how different individual effects may have different limiting constraints at the population or community level. These considerations will strengthen the causal linkage between a temperature stressor and effects by allowing managers to monitor the potential direct and indirect effects on salmonids and manage for unanticipated changes to stream ecosystems (Adams et al. 2002).

Thermal impacts from forestry — When evaluating whether forest harvesting activities on new streams will exceed acceptable levels, managers need to assess the likelihood and magnitude of both local stream-scale influences and watershed-scale activities. In this study, the effect of local stream-scale activities (e.g., riparian harvesting and stream crossings) was not analyzed because the data were not available. However, I was able to analyze the effect of watershed-scale activities (e.g., proportion of the watershed logged and density of roads), and observed an important effect of roads on stream temperature.

The results from the Bayesian regression illustrate how managers can quantify the probability and magnitude of impacts from forestry activities to evaluate whether a stream should be designated as either "Temperature-Sensitive" or not. In this study, I found that there was a 93% probability that there is an effect of road density on stream temperatures. Specifically, there was a 78% probability that this effect will be moderate (i.e., $0 <$ regression slope ≤ 2), and a 15% probability that this effect will be extreme (i.e., regression slope > 2). Given three simple scenarios about the effect of roads (no effect, a moderate effect, and an extreme effect), a manager can then weight the probabilities associated with each of these against the biological consequences to salmonids to determine an appropriate stream designation. If there are extremely large biological consequences of having high road densities and warmer conditions in a stream, then managers may be more prone to designate a stream as "Temperature-Sensitive" and minimize road densities to avoid potentially extreme temperature changes.

For example, consider two streams in which the first has a MWAT near the maximum tolerable value for bull trout, and where a > 2 °C increase in a MWAT may result in local extinctions or a thermal isolation of a bull trout population. The second stream has a MWAT which is near the estimated MWAT for maximum growth of rainbow trout, and a > 2 °C increase in a MWAT will result in no appreciable impacts to the viability of this population. If the probability of observing a > 2 °C increase in a MWAT is the same for these two streams, managers would be more averse to the negative effects in the bull trout stream than they would be towards the minimal threats to rainbow trout. In other words, they would weight the biological consequences of the first stream higher than the second stream, and be more prone to designate the first stream as "Temperature-Sensitive" to restrict road densities and stream heating. A decision-making system such as this takes full advantage of the results from Bayesian analyses, and allow decision-makers to do a better job of incorporating uncertainty than most current management systems.

Challenges to defining "Sensitivity"

Defining acceptable levels of impact is the greatest challenge to implementing the proposed framework. Managers should avoid setting discrete thresholds that distinguish sensitive streams from those that are not because of difficulties in applying the regression tree and modeling results. First, the size or number of final groupings in a regression tree can increase as the number of streams used to generate the tree increases. For example, if a manager defines an unacceptable level as the 90th percentile within a Class, the number of these thresholds would increase with the number of Classes in the regression tree. As a result, more streams would be considered "Temperature-Sensitive" irrespective of the effects of forest practices. Second, the modeling results allow managers to compare the relative effects of temperature increases among a variety of indicators of salmonid processes. However, trade-offs among these indicators may not be clear; they will depend on the objectives for protection of salmonid resources in a particular stream. For example, if the objective is to maximize production of rainbow trout, a manager may manage forest practices to protect the temperatures that are most suitable for egg survival rate and resistance to disease. This objective may conflict with one intended to maximize the production of other salmonid species or protect other critical elements of the freshwater system (e.g., macroinvertebrate communities). Therefore, the modeling results do not provide information on the relative benefits of protecting temperatures for different life history stages, fish species, or biological communities, but instead quantify variables that decision makers must trade off.

Forest managers have at least two options to determine a stream's temperature-sensitivity and select the most appropriate forest practices. One option is to use discrete thresholds (e.g., 90^{th} percentile within a Class) in spite of the obvious difficulties. In this instance, managers would need to clearly identify the objectives for forest harvesting and stream protection and use a subjective, yet informed, approach to setting the appropriate levels of impact. For example, classifications would not need to be as detailed as a regression tree analysis indicates. Instead, researchers and managers would need to subjectively select the "stop splitting" rules for the regression tree and choose the appropriate classifications and range of natural variation in temperatures within which to manage forestry practices. With this option, managers would also need to weight the relative importance of different temperature-dependent responses based on the objective for protecting salmonid resources in a stream (e.g., maximize production, maximize growth, or minimize thermal barriers). They would then need to select forest practices that protected stream temperatures to best achieve that objective.

To address the idea of temperature-sensitivity as a continuum, a better alternative would be to determine a stream's sensitivity by evaluating a range of proposed forest practices using the quantitative approach, decision analysis (Keeney 1982; Cohan et al. 1984; Clemen 1996; Peterman and Anderson 1999). With this procedure, managers could identify a number of forest management options ranging from very restrictive on the most

"Temperature-Sensitive Streams" to very lenient on the least sensitive. This approach can explicitly incorporate the uncertainty in the magnitude of temperature changes, the uncertainty surrounding the biological responses of the stream, as well as the objectives for stream protection and forest harvesting. A manager can then use all of this information to rank the different management options and determine the most appropriate forestry practices. This approach removes the subjectivity from decision-making and provides a clear framework within which managers can determine a stream's sensitivity.

An additional challenge to defining sensitivity relative to a range of natural variation relates to the difficulty of correctly understanding the influence of a stream's watershed features and climatic setting on its temperatures. By combining aspects of space (all streams were from different locations) and time (streams were sampled across a number of years) in the regression tree, there is the potential for confounding effects. To illustrate this problem, streams in Classes III and IV have similar physical characteristics (e.g., drainage areas < 132 km2 and average basin elevations ≥ 1140 m), yet they do not represent the same streams. Streams in these Classes have differences in the physical features of their watersheds, as well as differences in climatic setting. Therefore, I cannot determine whether to attribute the differences in stream temperatures to fluctuations in climate or variations in some other potentially confounding watershed feature that was not measured or identified in the regression tree. One way to deal with this problem in the future would be to only use a stream's watershed features in the regression tree. Understanding the influence of changes in climate could then be incorporated using other types of analyses (e.g., time series analysis).

Limitations to managing stream temperatures

There are several sources of potential errors in these analyses. Daily air and stream temperatures, land-use information, and spatial data such as drainage area and basin elevation calculations, may include measurement errors. This type of error will increase the variation in the explanatory and dependent variables (e.g., Walters and Ludwig 1981) and may influence the results from this study.

For example, stream temperature data recorders are prone to measurement errors (see Lewis et al. 2000), which would reduce the ability (i.e., power) of linear regression to detect the effect of logging-related activities on the MWAT residuals from the regression tree. Errors in stream temperatures could also influence the selection of the variables and values used to partition the MWAT data in the regression tree in Figure 4, and increase the variation around the best-fit relationships in Figure 5.

There are also concerns with the way in which forest harvesting activities were summarized and applied in this study. First, in the data sets used here, logging activities were summarized over 20 years; no shorter time-period was available. This period may have been inappropriate because recovery of streams to unimpacted thermal conditions has been observed between seven (Feller 1981) and 20 years (as cited by Beschta and Taylor 1988) after forest harvesting. Consequently, the land-use data may be biased towards watersheds with a higher measured level of forest development than is actually the case. Second, logging activities were also summed over an entire watershed with local and more distant upstream influences weighted equally, even though local effects can have stronger effects on stream temperatures (Macdonald et al. 2003). This error suggests that the land-use data may underestimate the influence of forest harvesting and be biased low. The inability to accurately measure local and watershed-scale activities on all streams may influence the regression tree and results from the linear regression by (1) increasing the amount of unexplained variation in the box plots in Figure 4 and regression relationships in Figure 7, (2) reducing the power to detect the influence of forestry activities, and (3) creating a bias in the range of MWATs for each Class because streams do not reflect unimpacted conditions.

Applying models from other studies to predict the effect of water temperature on salmonids in the north-central interior may also be problematic. First, by ignoring the influence of other habitat variables in these models, a manager may overlook other likely constraints in a stream and recommend inappropriate management decisions. For example, a manager may wrongly conclude that forestry activities and increased stream temperatures are limiting salmonid growth when density-dependent factors are actually constraining the population. Second, salmonid responses in constant-temperature environments used in laboratory studies may differ from those observed in naturally fluctuating stream environments. Understanding the implications of this statement are difficult because there are few relevant studies and results are mixed (Hokanson et al. 1977; Thomas et al. 1986; Johnstone and Rahel 2003). Third, there may be different responses to temperature for salmon populations with distinct geographical distributions. Geographical differences have been observed in the duration of egg incubation (Macdonald et al. 1998), migratory timing (Robards and Quinn 2002), and growth (Nicieza et al. 1994) of salmonids; temperature may be an important factor contributing to this diversity. As a result of these concerns, managers need to recognize that the data points and the best-fit line between a salmonid response and a MWAT index may misrepresent the 'true' relationships because the models do not adequately quantify all of the biological and ecological conditions for salmonids in the north-central interior. Therefore, the 'true' response curves may lie to the left or right (i.e., shifted horizontally towards either cooler or warmer MWATs) or have scatter about the regression lines that is greater than illustrated by the plots in Figure 5.

Implications of climate change

Due to climatic change, air temperatures and water supplies are expected to change in Canada (Hengeveld 1990), and warmer water temperatures are forecast in the Fraser River watershed of B.C. (Morrison et al. 2002). It is also believed that air temperature increases due to climate change will increase temperatures in smaller streams of the north-central interior (Tyedmers and Ward 2001) because of the strong relationship between stream and air temperatures (Cluis 1972; Stefan and Preud'homme 1993). From an ecological perspective, warming of the freshwater environment has led to broad concerns about the potential impacts of climatic change (Carpenter et al. 1992; Hauer et al. 1997). Consequently, the management of forest practices and stream temperatures may become more difficult as average air temperatures increase with a changing climate. These difficulties will relate to two key uncertainties; the uncertainty in changes to air and stream temperatures, and the uncertainty in the responses of the freshwater ecosystem to those temperature changes.

The implications of uncertain changes in air and stream temperatures can be examined using the regression tree. In general, these results suggest that each Stream Temperature Class will have a different sensitivity to increases in air temperature because the physical processes controlling stream heating and cooling, and relative influence of air temperature, differ among Classes. For example, streams in Classes V and VI have large drainage areas, large stream channels, and discharge large volumes of water. An air temperature index did not help classify these streams because its heating influence is likely weak for large quantities of water. In contrast, streams from Class III and IV have smaller drainage areas, smaller channels, and discharge smaller quantities of water. For these streams, changes in air temperatures are more likely to result in an increase in stream temperatures because less energy is needed. Even though streams from Classes I and II had similar sized drainage areas, they have a higher average basin elevation, which suggests that they are closer to headwaters than streams in Class III and IV. Therefore, the cooling influence of headwater sources of groundwater is likely to have an overriding effect on temperatures in streams from Classes I and II.

Even though an air temperature index did not appear in four of the six Classes, increases in air temperature may still have an influence on stream temperatures for all groupings. A regression tree analysis with a larger sample would increase the number of climate and watershed features used to classify streams and provide a better indication of the relative influence of air temperatures for different types of streams. As air temperatures increase with climatic change, there will be a greater number of years in which air temperatures are above the historical average, and a greater number of streams that are placed in Classes with warmer summer air temperatures. The identification of a "Temperature-Sensitive Stream" will then depend on the predicted sensitivity of each Class to temperature increases from forestry activities (as discussed in *Identifying "Temperature-Sensitive Streams"*).

Understanding the uncertainty surrounding the response of the freshwater ecosystem to climatic change is also critical because forestry activities can confound our understanding and exacerbate the effects of climate change. Managers may assume that responses to climatic change will be consistent with our current understanding of salmonid thermal requirements. However, unpredictable responses cannot be ruled out (Healey 1990). For example, the use of models that borrow parameters from separate populations may fail to predict the 'true' effect of temperature change on populations that have acclimated to local thermal conditions. By considering the direct (e.g., changes in salmonid growth) and indirect (e.g., changes in macroinvertebrate communities) effects on salmonids, a manager will be better informed about potential thermal impacts, whether from forestry or climatic change. This type of approach is consistent with the one proposed in this study and recognizes the effect of changes in temperature on more of a stream's variables than the one with the coolest thermal requirements. The framework developed here and similar alternatives will help ensure that management responses are more adaptive and robust to a wider range of climatic change scenarios than existing ones.

Recommendations for scientists and managers

The aim of this study was to develop and illustrate a framework to help forest managers identify "Temperature-Sensitive Streams." Before this framework can be implemented, I have four recommendations for future research and analyses:

(1) Use a simple temperature index, such as a MWAT, to characterize a seasonally variable thermal regime. Other indices, such as a measure of the daily fluctuation in

stream temperatures, may also provide meaningful information about the biological suitability of a thermal regime, but they may be more difficult to relate to the watershed features and climatic setting that influence a thermal regime, and more difficult to relate to the various biological responses in a stream.

(2) Use a stream's watershed features and climatic setting to identify stream-types with the most similar thermal regimes. The sample of streams for a regression tree analysis should be large enough so that an independent group of streams can be used to validate the regression tree results, and random so that inferences about the sample can be applied to the entire population of streams within a group. This classification scheme is useful because it reduces the observed variation in a stream temperature index and provides a baseline against which to compare streams that are new to the analysis. A reduction in the observed variation is important when maintaining temperatures within a range of natural variation and testing hypotheses about the effects of forest practices on stream temperatures.

(3) Identify the key biological sensitivities and responses of a fish community, and quantitatively relate these responses to a simple temperature index. A manager can determine these relationships by using field studies (Welsh et al. 2001; Dunham et al. 2003; Picard et al. 2003; Wehrly et al. 2003), or mathematical models, as demonstrated in this and other studies (Sullivan et al. 2000). These relationships are useful because they can estimate the biological responses in a stream and allow for an easy assessment of the positive and/or negative effects of anthropogenic activities on a variety of temperature-responsive salmonid variables.

(4) Assess the magnitude and probability of temperature impacts from proposed forest practices, both at the local stream-scale and watershed-scale. Bayesian analyses, such as the one illustrated here, can explicitly quantify the magnitude and probability of temperature changes given a particular type of forestry activity. Alternatively, existing stream temperature models (see Sullivan et al. 1990) can be used to run Monte Carlo simulations and similarly quantify the probability of temperature impacts from proposed forest practices. Once quantified, these results can be incorporated into a decision-making system to help managers determine whether new streams of interest will be altered beyond acceptable levels.

By following up on these recommendations for the region(s) of interest, researchers will provide forest managers with the tools and information necessary to manage forest practices and stream temperatures. Managers will then need to consider three sources of information when *Identifying "Temperature-Sensitive Streams"* (Figure 1). First, they will need to collect and summarize stream temperature data on new *Streams of management interest*, measure a stream's watershed features and climatic setting, and identify the relevant biological information for those streams. Second, they will need to define acceptable levels of impact by evaluating sensitivity relative to the expected change in the abiotic conditions and biological processes in a stream. The regression tree results (*Variation in stream temperatures*) can be used to evaluate the abiotic conditions and to maintain temperatures within a range of natural variation. The modeling results (*Thermal requirements of salmonids*) can be used to evaluate the biological processes in a stream and to maintain temperatures for more salmonid responses than the one with the coolest thermal requirements. Finally, they will need to assess the *Thermal impacts from* *forestry* to determine the magnitude and likelihood of temperature increases from proposed forest practices. If these practices are predicted to increase temperatures beyond acceptable levels, streams should be designated as "Temperature-Sensitive," and stream protection measures should be enhanced.

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Table 1. Descriptions of the 16 indices used to characterize a summer temperature profile (June 9 to September 15) as derived from raw stream temperature data.

Stream temperature index	Descriptions
MG (1)	Annual maximum of the *daily maximum temperature*
MG (2)	Annual maximum of a 7-day average of the *daily maximum temperature*
MG (3)	Annual maximum of a 7-day average of the *daily minimum temperature*
MG (4) [a]	Annual maximum of a 7-day average of the *daily mean temperature*
MG (5)	Value representing the 95th percentile of the *daily mean temperature* over the summer
MG (6)	Median value of the *daily mean temperature* over the summer
MG (7)	Average of the *daily mean temperature* over the summer
TH (1)	Number of days the *daily maximum temperature* exceeds 19 °C over the summer
TH (2)	Number of days the *daily maximum temperature* exceeds 15 °C over the summer
TH (3)	Number of days a 7-day average of the *daily maximum temperature* (i.e., MG(2)) exceeds 18 °C over the summer
DF (1)	Maximum difference between the *daily maximum* and the *daily minimum temperature* over the summer
DF (2)	Minimum difference between the *daily maximum* and the *daily minimum temperature* over the summer
DF (3)	Summer average of the difference between the *daily maximum* and the *daily minimum temperature*
RT (1)	Average rate of increase (°C/day) from the *daily minimum temperature* on June 9 to the maximum of the *daily maximum temperature* (i.e., MG(1))
RT (2)	Average rate of decrease (°C/day) from the maximum of the *daily maximum temperature* (i.e., MG(1)) to the *daily minimum temperature* on September 15
TM (1)	Date of the maximum of the *daily maximum temperature* (i.e., MG(1))

Note: [a] In this study MG(4) is referred to as a maximum weekly average temperature (MWAT) because this usage is common in water quality guidelines

Table 2. Summary of the stream temperature index, watershed features, and climatic setting for the 104 study streams.

Region	Station identifier	Stream temperature year	Stream temperature index (°C) [a]	Easting [b]	Northing	Drainage area (km^2)	Basin elevation (m)	Air temperature index (°C) [c]	Air temperature station [d]	Distance to coast (m)	Orientation class [e]	Surficial geology class [f]	BEC zone [g]	Source [h]	Analysis of land-use [i]
Babine L.	BABI01	2000	10.82	668261	6112791	1.1	943	-0.86	Upper Fulton	266243	SW-NW	Tb	SBS	6	-
	BABI03	2000	10.02	667466	6111384	3.9	1157	-0.86	Upper Fulton	265080	SW-NW	Tb	SBS	6	-
	BABI04	2001	12.18	667176	6111437	3.8	1162	-1.55	Upper Fulton	264818	SE-SW	Tb	SBS	6	-
	BABI05	2000	11.12	668146	6112874	1.9	955	-0.86	Upper Fulton	266156	SW-NW	Tb	SBS	6	-
	BABI07	2001	11.67	668370	6113620	2.8	968	-1.55	Upper Fulton	266587	SW-NW	Tb	SBS	6	-
	BABI08	2001	11.97	628394	6149896	131.0	1096	-1.37	Nilkitkwa	243434	SW-NW	Tb	SBS	3	-
	BABL01	2002	9.24	316836	6053031	125.7	1148	-1.20	Augier Lake	291677	SE-SW	Tb	SBS	5	-
	BABL02	2002	10.92	309607	6051741	30.3	1257	-1.20	Augier Lake	284506	SW-NW	Tb	SBS	5	-
	BABL03	2000	9.96	358656	6036620	6.0	1159	-1.31	Augier Lake	320192	SW-NW	Tb	SBS	4	-
	BABL04	2000	10.53	344194	6035482	5.2	1013	-1.31	Augier Lake	307776	NE-SE	Tb	SBS	4	-
	BABL05	2000	13.54	684461	6074707	93.9	1135	-0.86	Upper Fulton	273875	SW-NW	Tb	SBS	4	Y
	BABL06	2000	12.31	654708	6106557	34.7	894	-0.86	Upper Fulton	251510	SW-NW	Tb	SBS	4	Y
	BULK02	2001	13.07	636821	6175415	3.9	1051	-1.37	Nilkitkwa	263337	NW-NE	Tb	SBS	4	-
	BULK03	2001	11.11	632016	6145058	1.7	1246	-1.37	Nilkitkwa	244436	SE-SW	Tb	ESSF	4	-
Baptiste Cr.	BAPT02	1997	11.09	350299	6081540	2.9	1190	1.22	Augier Lake	325646	NE-SE	Ra	SBS	1	-
	BAPT03	1996	9.28	350935	6081338	1.4	1202	-1.59	Augier Lake	326267	NE-SE	Tb	SBS	1	-
	BAPT04	1998	11.14	350930	6081330	1.1	1228	1.84	Augier Lake	326261	SE-SW	Tb	SBS	1	-
	BAPT07	1999	11.16	349902	6080966	3.2	1076	-0.31	Augier Lake	325214	SW-NW	Ra	SBS	1	-
	BAPT18	1997	10.54	349357	6080249	1.4	1086	1.22	Augier Lake	324627	SE-SW	Ra	SBS	1	-
Bivouac Cr.	BIVO01	1997	11.48	335559	6105552	39.6	1178	0.57	Upper Fulton	313537	SE-SW	Tv	SBS	1	-
Bulkley R.	BULK09	2000	14.27	681023	6040358	69.5	1116	-0.60	Houston	267796	NW-NE	Tb	SBS	4	Y
	BULK10	2001	12.74	682238	6028479	71.4	1088	-1.52	Houston	262561	SW-NW	Tb	SBS	3	Y
	BULK11	2001	8.29	591178	6047696	54.5	1488	-1.52	Houston	178504	SE-SW	Tv	SBS	3	-
	BULK12	2001	9.29	641796	6035559	64.9	1228	-1.52	Houston	228650	SE-SW	Tb	SBS	3	-
	BULK13	2001	15.65	616081	6073989	38.0	935	-1.55	Upper Fulton	206432	NW-NE	Tb	SBS	3	Y
	BULK14	1997	11.91	611175	6084288	75.9	1028	0.57	Upper Fulton	203796	SE-SW	Tb	SBS	3	-
Endako R.	ENDA01	2002	10.28	307211	6036055	4.2	1066	-1.20	Augier Lake	278213	NW-NE	Tb	SBS	5	-
	ENDA03	2002	11.24	308695	6032440	9.4	1005	-1.20	Augier Lake	277231	NE-SE	Tb	SBS	5	-
	ENDA05	2000	14.21	313439	6014821	2.8	1068	-1.31	Augier Lake	270767	SE-SW	Tb	SBS	4	-
Fleming Cr.	FLEM01	2002	9.68	329791	6070923	4.5	1045	-1.20	Augier Lake	304685	SW-NW	Tb	SBS	5	-
	FLEM02	2002	10.19	329829	6071978	2.4	1031	-1.20	Augier Lake	304761	SW-NW	Tb	SBS	5	-

Table 2. continued.

Region	Station identifier	Stream temperature year	Stream temperature index (°C) [a]	Easting [b]	Northing	Drainage area (km^2)	Basin elevation (m)	Air temperature index (°C) [c]	Air temperature station [d]	Distance to coast (m)	Orientation class [e]	Surficial geology class [f]	BEC zone [g]	Source [h]	Analysis of land-use [i]
Fleming Cr.	TILD02	2002	9.82	336536	6077842	4.9	1159	-1.20	Augier Lake	311720	NW-NE	Tb	SBS	5	-
	TILD03	2002	17.12	330127	6079879	171.4	1249	-1.20	Augier Lake	305462	SW-NW	Tb	SBS	5	-
	TILD04	2002	11.34	329793	6079114	24.3	1092	-1.20	Augier Lake	305080	SW-NW	Tb	SBS	5	Y
	TILD06	2002	9.77	329695	6078981	3.2	987	-1.20	Augier Lake	304975	SE-SW	Tb	SBS	5	-
	TILD07	2002	9.08	332840	6078937	2.9	971	-1.20	Augier Lake	308103	NW-NE	Tb	SBS	5	-
Forfar Cr.	FORF02	1991	13.12	342311	6102160	37.8	1308	0.29	Augier Lake	319740	SE-SW	Tb	SBS	1	-
Fulton R.	FULT01	2000	11.12	669990	6083160	2.6	968	-0.86	Upper Fulton	260976	NW-NE	Tb	SBS	6	-
	FULT03	2000	15.23	659413	6079986	326.1	1127	-0.86	Upper Fulton	250031	SE-SW	Tb	SBS	4	-
	FULT04	2000	14.56	658007	6087710	95.7	1002	-0.86	Upper Fulton	250123	NW-NE	Tb	SBS	4	-
	FULT05	2001	9.72	654184	6074711	6.4	1218	-1.55	Upper Fulton	244014	SW-NW	Tb	ESSF	4	-
	FULT06	2001	7.18	654577	6073894	19.0	1385	-1.55	Upper Fulton	244274	SW-NW	Tb	ESSF	4	-
	FULT07	2001	16.22	650841	6083501	16.0	1116	-1.55	Upper Fulton	242277	SW-NW	Tb	SBS	4	-
	FULT08	2000	13.14	664521	6080705	50.3	1056	-0.86	Upper Fulton	255174	NW-NE	Tb	SBS	4	Y
	FULT09	2000	10.14	657223	6078556	23.9	1171	-0.86	Upper Fulton	247633	SW-NW	Tb	SBS	4	-
Gates Cr.	GATE03	1995	13.39	653752	5983733	21.5	1069	0.04	Peden	211646	NE-SE	Tb	SBS	1	-
	GATE04	1994	11.73	653752	5983733	38.5	1208	0.48	Peden	211646	SE-SW	Tb	SBS	1	-
	GATE07	1994	12.84	656237	5978929	2.0	1246	0.48	Peden	210421	SE-SW	Tb	ESSF	1	-
	GATE11	1994	11.41	656862	5981496	7.5	1165	0.48	Peden	212561	SE-SW	Tb	SBS	1	-
	GATE12	1994	11.80	657250	5980130	3.2	1070	0.48	Peden	211970	SE-SW	Tb	SBS	1	-
	GATE14	1995	13.66	654080	5987312	83.1	1074	0.04	Peden	214217	SW-NW	Tb	SBS	1	Y
	GATE15	1995	11.52	656652	5980315	6.0	1202	0.04	Peden	211635	SE-SW	Tb	SBS	1	-
	GATE19	1995	11.44	653782	5982801	33.3	1243	0.04	Peden	211064	SE-SW	Tb	SBS	1	-
	GATE21	2000	10.56	653129	5982159	3.1	1070	-0.35	Peden	210152	SW-NW	Tb	SBS	6	-
Gluskie Cr.	GLUS01	1997	13.07	339420	6103611	48.9	1285	1.22	Augier Lake	317076	SE-SW	Tv	SBS	2	-
	GLUS05	1997	9.21	332285	6102236	25.0	1424	0.57	Upper Fulton	309838	SE-SW	Tv	SBS	1	-
	GLUS06	1998	10.86	335603	6102488	34.0	1342	1.84	Augier Lake	313151	SW-NW	Tv	SBS	1	-
	GLUS07	1999	9.87	333255	6102426	29.7	1379	-0.31	Augier Lake	310822	SW-NW	Tv	SBS	1	-
Kispiox R.	KISP03	1998	12.84	572473	6148644	24.4	1014	2.00	Kispiox	194321	SE-SW	Tb	ICH	4	Y
	KISP04	1999	11.36	572139	6145945	18.3	1149	-0.88	Kispiox	192679	SW-NW	Tb	ICH	4	-
	KISP05	2001	15.55	568989	6158730	122.4	672	-1.41	Kispiox	196367	NW-NE	Tb	ICH	3	Y
	KISP06	1998	11.01	536360	6176742	70.8	1055	2.00	Kispiox	177200	SW-NW	Tb	ICH	3	-
Kitsumkalum R.	KITS01	1997	12.50	507649	6087067	41.9	717	0.68	Kispiox	107495	NW-NE	Tb	CWH	3	Y
Kitwanga R.	SKEE03	1996	19.62	558508	6107015	832.6	924	-1.24	Kispiox	161798	NW-NE	Tv	ICH	3	-

Table 2. continued.

Region	Station identifier	Stream temperature year	Stream temperature index (°C) [a]	Easting [b]	Northing	Drainage area (km^2)	Basin elevation (m)	Air temperature index (°C) [c]	Air temperature station [d]	Distance to coast (m)	Orientation class [e]	Surficial geology class [f]	BEC zone [g]	Source [h]	Analysis of land-use [i]
Kynock Cr.	KYNO01	1992	12.87	346401	6096660	71.2	1251	1.17	Augier Lake	323118	SE-SW	Tv	SBS	1	-
Leo Cr.	LEOC01	1998	16.26	335377	6107809	92.5	1081	1.91	Upper Fulton	313692	NW-NE	Tv	SBS	1	Y
Middle R.	MIDR01	1998	20.54	339964	6103933	5,682.1	1058	1.84	Augier Lake	317658	NW-NE	Tb	SBS	1	-
	MIDR02	1991	18.88	350622	6093655	6,006.8	1055	0.29	Augier Lake	326982	SW-NW	Tv	SBS	2	-
Morice R.	MORI01	1997	14.38	597331	6010294	191.4	1103	0.35	Houston	186177	NW-NE	Tv	SBS	4	-
	MORI02	1999	11.68	606965	6017914	41.9	1288	-1.16	Houston	194811	NE-SE	Tv	SBS	4	-
	MORI03	1999	9.66	607995	6022093	9.5	1155	-1.16	Houston	195476	NE-SE	Tv	SBS	4	-
	MORI04	1999	10.68	606272	6012149	12.8	1144	-1.16	Houston	194766	NE-SE	Tb	SBS	4	-
	MORI05	1999	10.48	607158	6010252	9.6	987	-1.16	Houston	194442	NE-SE	Tb	SBS	4	-
	MORI07	1999	10.85	605469	6023120	2.6	1059	-1.16	Houston	192889	NW-NE	Tv	SBS	4	-
	MORI08	1999	5.36	605163	6022440	6.4	1190	-1.16	Houston	192633	SW-NW	Tv	SBS	4	-
	MORI09	1997	13.26	605494	6008380	530.6	1156	0.35	Houston	191955	SW-NW	Tb	SBS	3	-
	MORI10	1999	18.11	646293	5998398	81.2	971	-1.60	Peden	215735	SE-SW	Tb	SBS	4	-
	MORI11	1999	15.99	639824	6007737	213.1	966	-1.16	Houston	217129	SE-SW	Tb	SBS	4	-
	MORI13	1999	12.52	637535	6007790	37.3	1022	-1.16	Houston	215457	SE-SW	Tb	SBS	4	-
	MORI14	1999	11.81	638820	6016753	247.6	1192	-1.16	Houston	222383	SW-NW	Tb	SBS	4	-
	MORI15	1999	12.48	615541	6008673	19.5	1094	-1.16	Houston	199643	SW-NW	Tv	SBS	4	-
	MORI16	1999	13.22	623890	6005250	3,141.4	1197	-1.16	Houston	203590	SW-NW	Tb	SBS	3	-
	THAU01	2000	9.95	607614	6019925	1.0	985	-0.60	Houston	195272	NE-SE	Tv	SBS	6	-
	THAU05	2001	9.23	608160	6020875	1.5	1072	-1.52	Houston	195735	NE-SE	Tv	SBS	6	-
	THAU07	2001	8.96	611290	6011990	2.7	1211	-1.52	Houston	198681	NE-SE	Tv	SBS	6	-
Nadina R.	NADI03	1994	18.81	653200	5988327	902.1	1091	0.48	Peden	214210	SW-NW	Tb	SBS	1	-
Nautley R.	NAUT01	1992	21.27	395379	5994195	6,553.5	948	1.17	Augier Lake	325359	SW-NW	Tb	SBS	2	-
Parrott Cr.	PARR01	1999	21.16	664898	5997254	158.8	1024	-1.60	Peden	228892	SW-NW	Tb	SBS	4	-
	PARR03	1999	18.10	677364	5987574	395.3	980	-1.60	Peden	232093	SW-NW	Tb	SBS	4	-
Peter Alec Cr.	PETE01	1994	15.58	649186	5991428	69.3	1124	0.48	Peden	213252	SW-NW	Tb	SBS	1	-
Pinkut Cr.	PINK02	2002	10.93	324097	6032451	18.8	1112	-1.20	Augier Lake	289681	SE-SW	Tb	SBS	5	-
	PINK03	2002	15.58	331169	6026417	36.4	1095	-1.20	Augier Lake	291919	NE-SE	Tb	SBS	5	Y
	PINK04	2000	18.00	342628	6030444	803.1	1093	-1.31	Augier Lake	303574	SE-SW	Tb	SBS	4	-
	PINK05	2000	16.40	335747	6024864	28.1	1125	-1.31	Augier Lake	294736	SW-NW	Tb	SBS	4	-
Stellako R.	STEL01	1993	18.38	368631	5986427	4,016.9	968	0.26	Augier Lake	299102	SW-NW	Tb	SBS	2	-
Stuart R.	STUA02	1997	18.84	417650	6031000	14,211.9	958	1.22	Augier Lake	364872	NW-NE	fL	SBS	2	-

Table 2. continued.

Region	Station identifier	Stream temperature year	Stream temperature index (°C) [a]	Easting [b]	Northing	Drainage area (km²)	Basin elevation (m)	Air temperature index (°C) [c]	Air temperature station [d]	Distance to coast (m)	Orientation class [e]	Surficial geology class [f]	BEC zone [g]	Source [h]	Analysis of land-use [i]
Stuart R.	STUA05	1998	22.63	458993	6002045	14,863.7	951	2.31	Bear Lake	381629	SW-NW	fL	SBS	2	-
Tachie R.	TACH01	1995	18.53	379074	6073312	8,627.9	1022	0.47	Augier Lake	353895	SW-NW	Tb	SBS	2	-
	TACH02	1994	19.73	384226	6063202	10,216.8	1001	0.49	Augier Lake	356436	NW-NE	Tb	SBS	2	-
Tahtsa L.	TAHT01	1999	13.28	636141	5956550	3.8	1096	-1.60	Peden	180623	NE-SE	Tb	SBS	4	-
	TAHT02	1999	14.39	633225	5955940	21.9	1000	-1.60	Peden	178010	NW-NE	Tb	SBS	4	-
	TAHT03	1999	15.98	653963	5963657	209.1	1035	-1.60	Peden	198788	SW-NW	Tb	SBS	4	-
Zymoetz R.	ZYMO03	2001	15.11	596128	6072317	133.9	1260	-1.55	Upper Fulton	186566	NE-SE	Tb	SBS	3	-
	ZYMO04	1998	12.39	596053	6073377	51.1	1095	1.91	Upper Fulton	186700	NW-NE	Tv	SBS	3	Y
Average	n/a	n/a	12.93	n/a	6053170	777.4	1102	-0.50	n/a	253257	n/a	n/a	n/a	n/a	n/a
Maximum	n/a	n/a	22.63	n/a	6176742	14,863.7	1488	2.31	n/a	381629	n/a	n/a	n/a	n/a	n/a
Minimum	n/a	n/a	5.36	n/a	5955940	1.0	672	-1.60	n/a	107495	n/a	n/a	n/a	n/a	n/a
Standard Deviation	n/a	n/a	3.36	n/a	51704	2,568.1	129	1.10	n/a	55074	n/a	n/a	n/a	n/a	n/a

Note: [a] The *Stream temperature index* represents a maximum weekly average temperature (MG(4)) calculated from the summer temperatures for the *Stream temperature years* represented in this table.

[b] *Easting* values in the 300000 and 400000 range represent streams from UTM Zone 10, whereas values in the 500000 and 600000 range are from UTM Zone 9.

[c] *Air temperature index* values > 0 represent year-types in which a regional measure of summer air temperatures was warmer than a 13-year average, values < 0 represent year-types in which a regional measure of summer air temperatures was cooler than a 13-year average, and values near 0 represent year-types in which a regional measure of summer air temperatures was near a 13-year average.

[d] *Air temperature station* refers to the name of the climate stations used to measure summer air temperatures: Augier Lake (54.36°N, 125.52°W); Bear Lake (54.51°N, 122.69°W); Houston (54.41°N, 126.63°W); Kispiox (55.44°N,127.65°W); Nilkitkwa (55.55°N, 126.58°W); Peden (53.99°N, 126.52°W); Upper Fulton (55.03°N, 126.80°W).

[e] *Orientation class*: NW-NE (0-45°, 315-360°); NE-SE (45-135°); SE-SW (135-225°); SW-NW (225-315°).

[f] *Surficial geology class*: Tb - Till Blanket; Ra - Alpine Complexes; Tv - Till Veneer; fL - Fine grained (Glacio) Lacustrine.

[g] *Biogeoclimatic ecosystem classification (BEC) zone*: CWH - Coastal Western Hemlock; ESSF - Engelmann Spruce-Subalpine Fir; ICH - Interior Cedar-Hemlock; SBS - Sub-Boreal Spruce.

[h] *Source*: 1. Erl MacIsaac and Herb Herunter, Department of Fisheries and Oceans, Burnaby, B.C. (personal communication); 2. Terry Sowden, Department of Fisheries and Oceans, Sidney, B.C. (WATEMP database); 3. Barry Finnegan, Department of Fisheries and Oceans, Nanaimo, B.C. (personal communication); 4. Ian Sharpe, Ministry of Water, Land and Air Protection, Smithers, B.C. (personal communication); 5. Lisa Torunski, McElhanney Consulting Ltd. and Baptiste Forest Products, Smithers, B.C. (personal communication); 6. Pat Hudson, Freshwater Resources, Smithers, B.C. (personal communication).

[i] *Analysis of land-use*: Streams with "Y" indicate that they were used to analyze the effects of forestry activities on stream temperatures.

Appendix

Appendix A. Equations, parameter values, and references for the models described in the text and used to estimate various salmonid responses to summer stream temperatures from 31 study streams.

Rainbow trout egg development

(1)
$$D = \left[D_1^{\ b} + \frac{\left(D_2^{\ b} - D_1^{\ b}\right) \cdot \left(1 - e^{-a(T - T_1)}\right)}{\left(1 - e^{-a(T_2 - T_1)}\right)} \right]^{\frac{1}{b}}$$

Variables	Parameter values	References
D = number of days to date of median hatch	a = 0.4084	McLean (1991)
T = water temperature (°C)	b = 2.3614	Schnute (1981)
	D_1 = 139.3	Velsen (1987)
	D_2 = 18.3	
	T_1 = 1	
	T_2 =20	

Rainbow trout egg survival

(2)
$$s = 1 - \left(a + bT + cT^2\right) \quad if \quad 0 < T < 18.75 \quad else$$
$$s = 0$$

Variables	Parameter values	References
s = proportion of eggs surviving from fertilization to hatch	a = 0.5617	Jensen et al. (2002)
T = water temperature (°C)	b = -0.1332	Velsen (1987)
	c = 0.0083	

Appendix A. continued.

Rainbow trout juvenile growth

(6) $$C_i = f(T_i, W_i, R_i)$$

$C_i = R_i \cdot p_s \cdot C_{max}$

$p_s = p_w \cdot p_t$

$p_w = W_i^{CB}$

$p_t = \lambda_0 + \lambda_1 \cdot T_i + \lambda_2 \cdot T_i^2 + \lambda_3 \cdot T_i^3$

Variables	Parameter values	References
C_i = daily consumption rate (grams· gram body weight^{-1}·day^{-1})	R_i = 1.0, 0.8, 0.6, or 0.4	Sullivan et al. (2000)
	C_{max} = 0.16	Hanson et al. (1997)
p_s = proportional adjustment of maximum consumption rate		
p_w = proportional adjustment in consumption due to the weight of a fish	$CB = -0.275$	
W_i = daily weight of a fish (grams)		
p_t = proportional adjustment in consumption due to water temperature	λ_0 = -0.1229	
T_i = daily water temperature (°C)	λ_1 = 0.0607	
	λ_2 = 0.0055	
	λ_3 = -0.0003	

(7) $$g_i = f(T_i, C_i)$$

$g_i = X_0 + X_1 \cdot T_i + X_2 \cdot T_i^2 + X_3 \cdot C_i + X_4 \cdot C_i^2 + X_5 \cdot C_i \cdot T_i + X_6 \cdot W_{i-1}$

(8) $$w_t = w_0 \prod_{i=0}^{t} (1 + g_i)$$

Variables	Parameter values	References
g_i = daily growth rate (grams· gram body weight^{-1}·day^{-1})	X_0 = 0.00631	Sullivan et al. (2000)
T_i = daily water temperature (°C)	X_1 = -0.0007403	
	X_2 = -00003909	
	X_3 = 0.4302	
	X_4 = -1.438	
	X_5 = 0.00735	
	X_6 = -0.00517	
	W_0 = 0.5	

Appendix A. continued.

Lethal temperatures

Test conditions	LT_{50}	References
Rainbow trout juvenile mortality from *Aeromonas salmonicida*	13.5 °C	Fryer and Pilcher (1974); Fryer et al. (1976)
Rainbow trout juvenile mortality from *Flexibacter columnaris*	15.0 °C	Fryer and Pilcher (1974); Fryer et al. (1976)
Rainbow trout direct temperature mortality	25.6 °C	Hokanson et al. (1977)
Bull trout direct temperature mortality	23.5 °C	Selong et al. (2001)

Note: LT_{50} refers to the temperature resulting in 50% mortality of a test sample of fish.

Figure 1. Flow chart illustrating the proposed information, analyses, and management actions required to identify "Temperature-Sensitive Streams." Dotted boxes (·····) represent points at which information is required, the solid boxes (—) represent points at which analyses are required, and the dashed-line boxes (– -) represent points at which management actions are required.

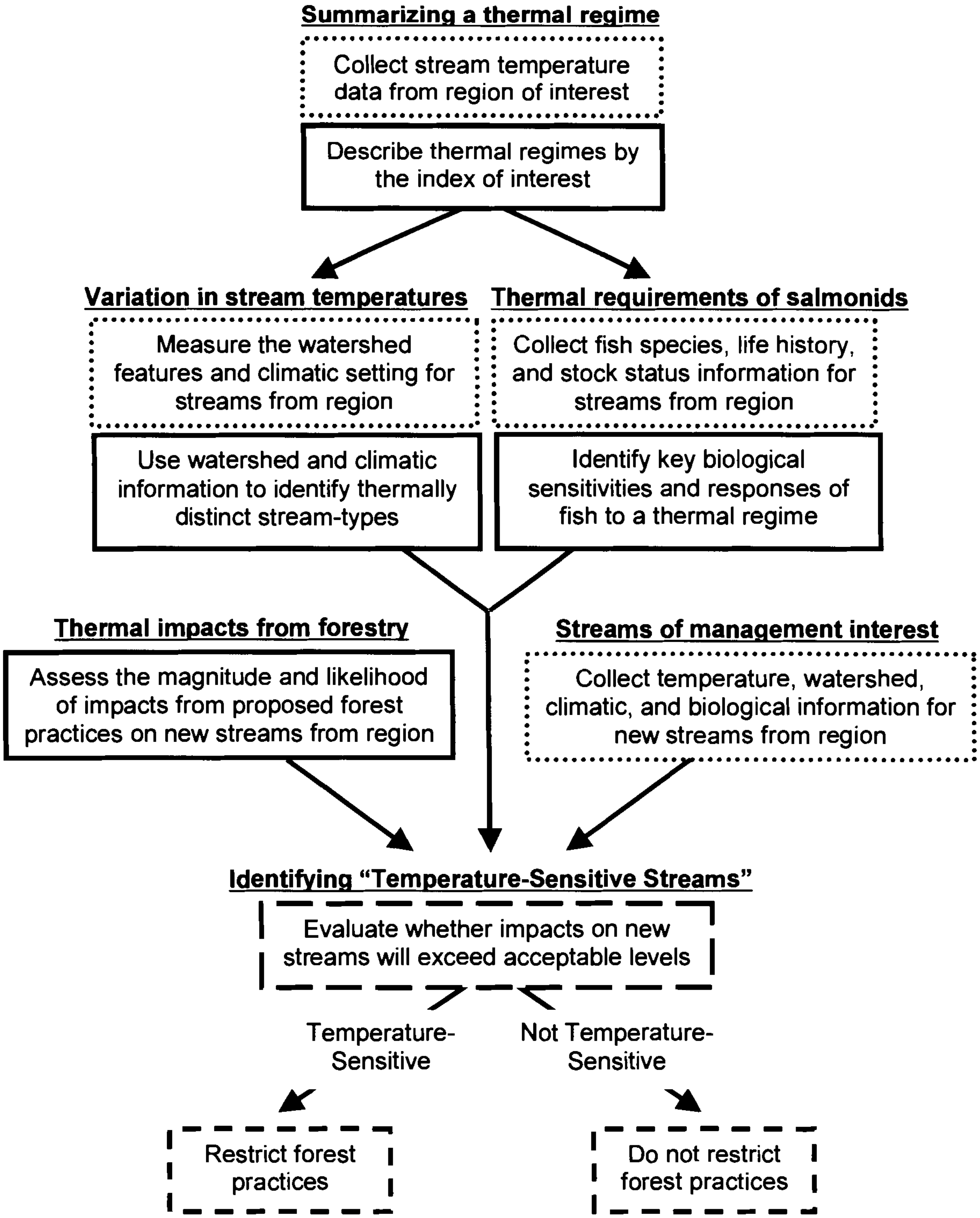


Figure 2. Study area and 104 stream temperature monitoring locations in the north-central interior of British Columbia.

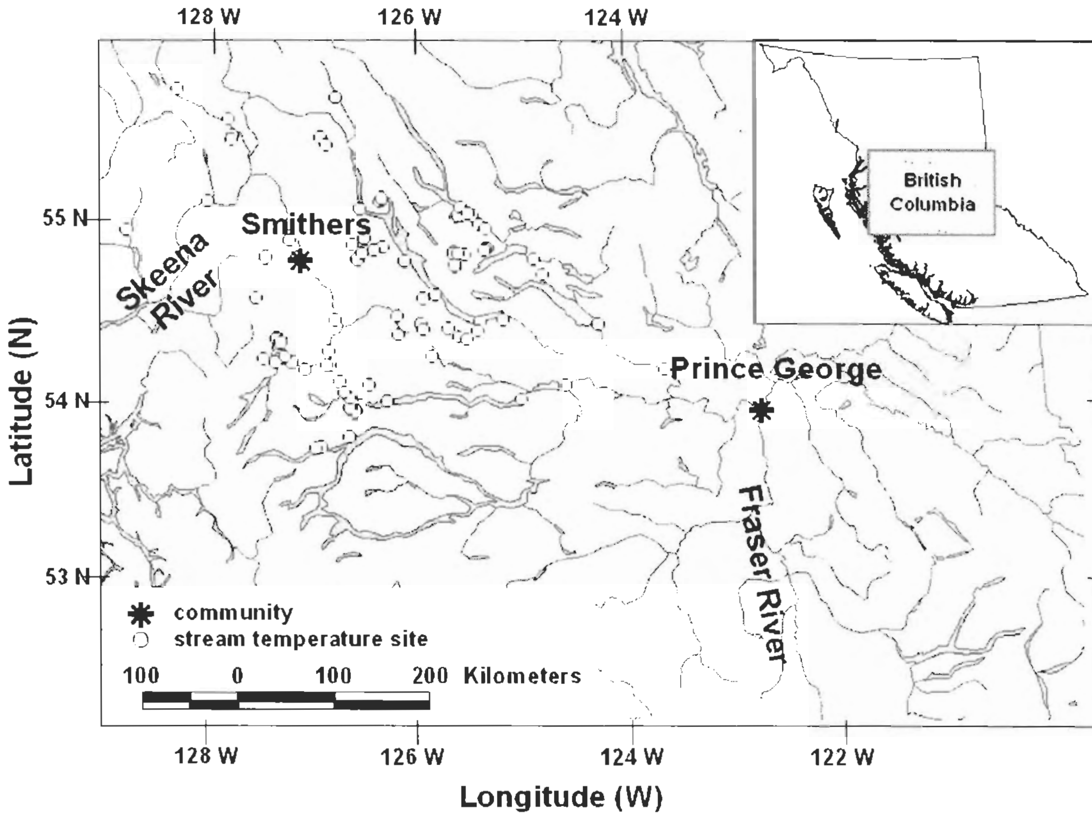


Figure 3. Pairwise comparisons of the correlations among 16 stream temperature indices. Indices are grouped according to their description of the annual peak of a temperature profile (*MG*), number of days that temperatures exceed a threshold value (*TH*), daily fluctuation in temperatures (*DF*), seasonal rate of temperature change (*RT*), or timing of annual maximum temperatures (*TM*) over the summer (June 9 to September 15). Detailed definitions of these indices are provided in Table 1. MG(4) represents the maximum weekly average temperature (MWAT) index referenced throughout this report. All correlation coefficients < -0.55 and > 0.55 are significant ($P < 0.05$).

Correlation coefficients (| r | > 0.8)

Correlation coefficients (0.55 < r < 0.8 or -0.8 < r < -0.55)

	MG(1)	MG(2)	MG(3)	MG(4)	MG(5)	MG(6)	MG(7)	TH(1)	TH(2)	TH(3)	DF(1)	DF(2)	DF(3)	RT(1)	RT(2)	TM(1)
MG(1)																
MG(2)	0.99															
MG(3)	0.87	0.90														
MG(4)	0.94	0.96	0.98													
MG(5)	0.97	0.98	0.99	0.99												
MG(6)	0.96	0.97	0.98	0.99	0.99											
MG(7)	0.97	0.98	0.99	0.99	0.99	0.99										
TH(1)	**0.77**	**0.79**	0.80	0.81	0.81	0.82	0.81									
TH(2)	0.92	0.93	0.93	0.94	0.95	0.95	0.96	0.82								
TH(3)	0.83	0.84	0.85	0.86	0.86	0.87	0.86	0.98	0.88							
DF(1)	0.51	0.47	0.25	0.35	0.35	0.33	0.34	0.30	0.32	0.32						
DF(2)	0.20	0.18	0.00	0.08	0.09	0.10	0.09	0.28	0.09	0.25	**0.55**					
DF(3)	0.41	0.37	0.12	0.23	0.23	0.22	0.23	0.26	0.22	0.27	0.86	**0.78**				
RT(1)	0.53	0.48	0.34	0.40	0.39	0.35	0.36	0.14	0.34	0.21	**0.59**	0.26	0.51			
RT(2)	**-0.64**	**-0.65**	**-0.60**	**-0.63**	**-0.62**	**-0.65**	**-0.62**	-0.82	**-0.56**	**-0.77**	-0.40	-0.42	-0.42	-0.09		
TM(1)	-0.04	-0.01	-0.01	-0.01	0.14	0.17	0.13	0.36	0.09	0.29	-0.29	-0.16	-0.33	-0.43	-0.49	

Figure 4. Dendrogram for the regression tree analysis showing the variables and values used to partition the maximum weekly average temperature (MWAT) for 104 streams from the root and intermediate nodes (ovals) to the terminal nodes (rectangles). Values within each node represent the average MWAT for that group of streams as described by the variables above that node. Sample sizes (n) are also provided for the terminal nodes. Variables are represented as, *Drainage*, the drainage area upstream of a temperature station, *Elevation*, the average elevation of the upstream basin, and *Air Temp*, a regional measure of the summer air temperatures. Box plots represent the median, interquartile range, and the 90th and 10th percentiles of the stream temperature data for each group of streams, called a Stream Temperature Class. The single horizontal line represents the average MWAT for all 104 streams.

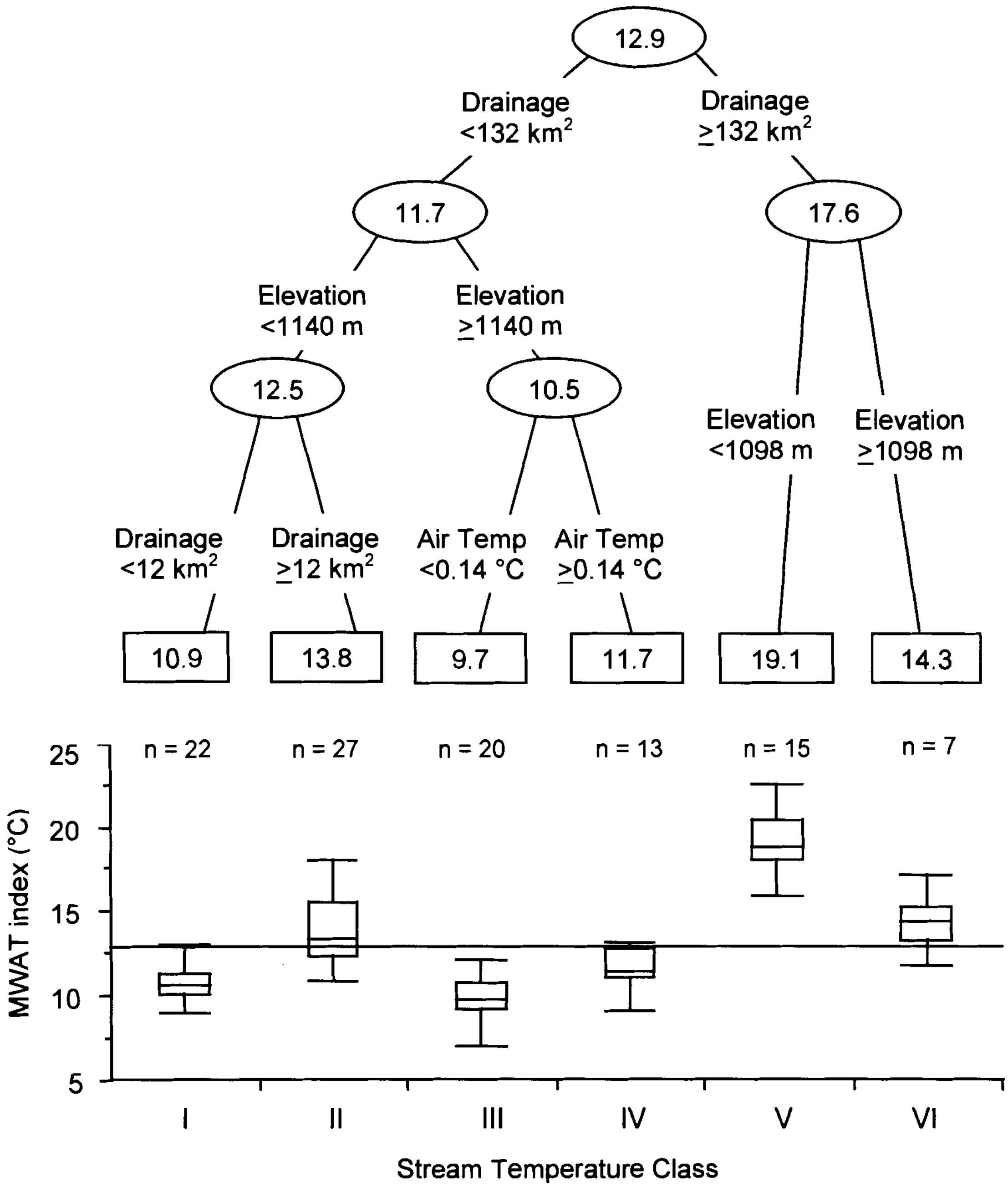
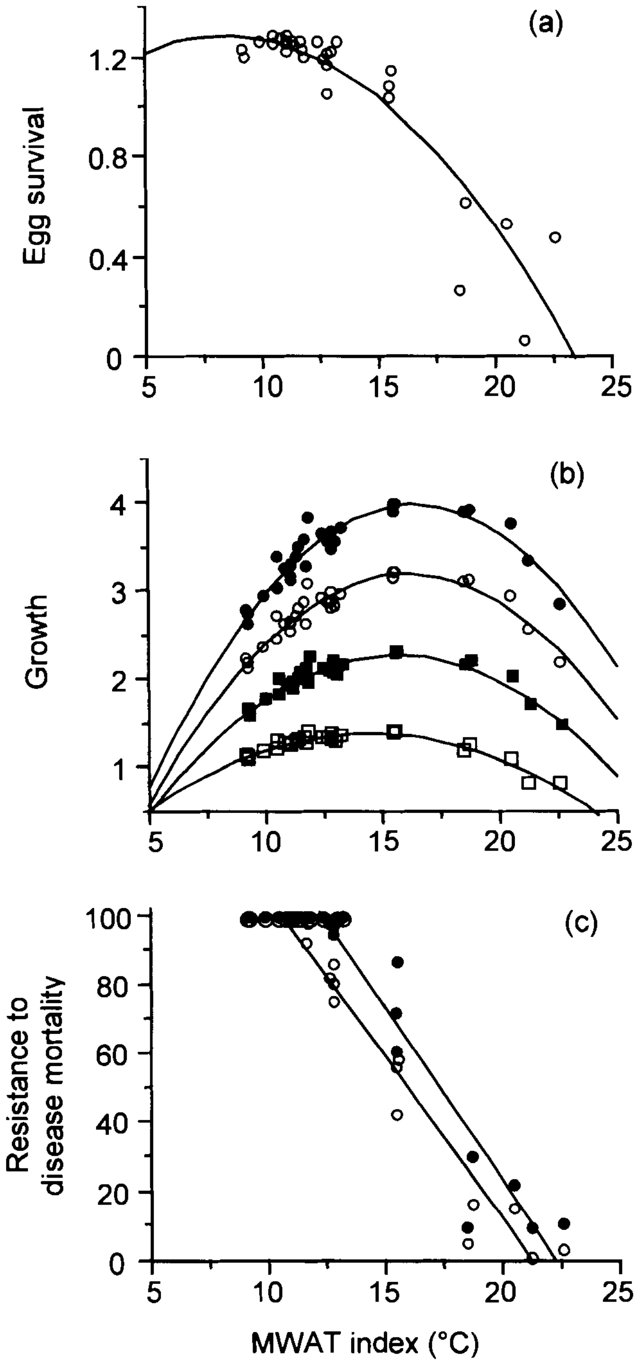


Figure 5. Relationships between a MWAT index and three modeled biological responses: (a) rainbow trout egg survival, (b) juvenile rainbow trout growth at four food rations (40% (□),60% (■),80% (○), and 100% (●) satiation), and (c) the number of days temperatures remain below the LT_{50} for two juvenile rainbow trout diseases (*Aeromonas salmonicida* (○) and *Flexibacter columnaris* (●)). Lines represent the best fit to the linear regression models presented here. R^2 values represent the proportion of the variance in the response variable explained by the fit of the regression line.



Egg survival (Arcsine(prop. of eggs surviving to date of median hatch)
$= 1.83 - 0.0510 \cdot MWAT - 0.00569 \cdot (MWAT-12.9)^2$ $R^2 = 0.85$

Growth (end-of-summer weight (g))
(●) $= 1.47 + 0.172 \cdot MWAT - 0.0248 \cdot (MWAT-12.9)^2$ $R^2 = 0.89$
(○) $= 1.21 + 0.136 \cdot MWAT - 0.0211 \cdot (MWAT-12.9)^2$ $R^2 = 0.89$
(■) $= 1.06 + 0.0869 \cdot MWAT - 0.0158 \cdot (MWAT-12.9)^2$ $R^2 = 0.90$
(□) $= 0.984 + 0.0302 \cdot MWAT - 0.00944 \cdot (MWAT-12.9)^2$ $R^2 = 0.91$

Resistance to disease mortality (number of days with summer temperatures < LT_{50})
(○)$= 200 - 9.33 \cdot MWAT$ if $MWAT > 10.8$ $R^2 = 0.94$
(●)$= 225 - 10.1 \cdot MWAT$ if $MWAT > 12.5$ $R^2 = 0.91$

Figure 6. Comparison of the MWAT values that predicted the maximum modeled responses in the egg survival, growth, and resistance to disease mortality relationships in Figure 5a, 5b, and 5c respectively. The solid squares represent the MWATs that predicted the maximum response. The upper and lower horizontal bars represent MWAT temperatures that predict a 5% reduction from the maximum response.

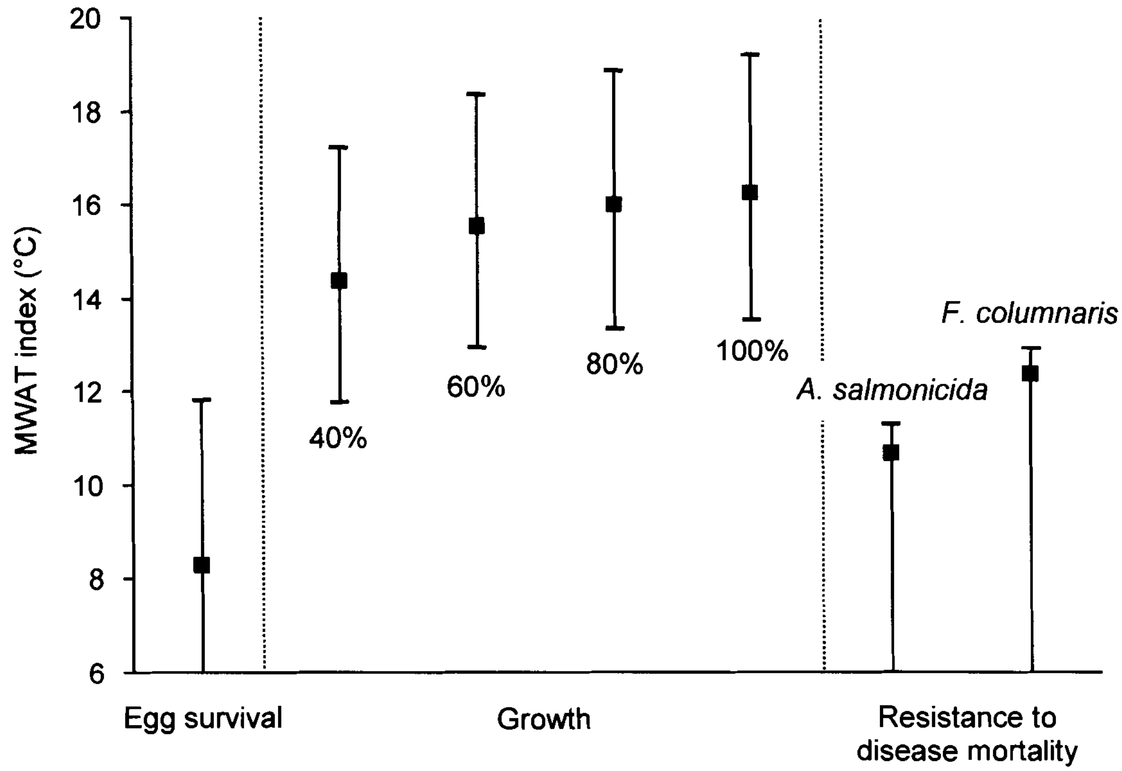


Figure 7. Relationships between the regression tree residuals from each stream and four measures of watershed-scale forestry activities: (a) the proportion of the upstream basin logged and selectively logged within the previous 20 years, (b) the proportion of 1:20,000 mapped streams logged and selectively logged to the banks within the previous 20 years, (c) the density of roads within the upstream basin, and (d) the density of road crossings within the upstream basin (i.e., crossings of the stream or its tributaries). Data are from Stream Temperature Class II; 14 streams with drainage areas ≥ 12 km^2 and < 132 km^2, and average basin elevations < 1140 m. Lines represent regression fits, correlation values (r) are Pearson's correlation coefficients, and P-values (P) represent the statistical significance from a one-tailed test.

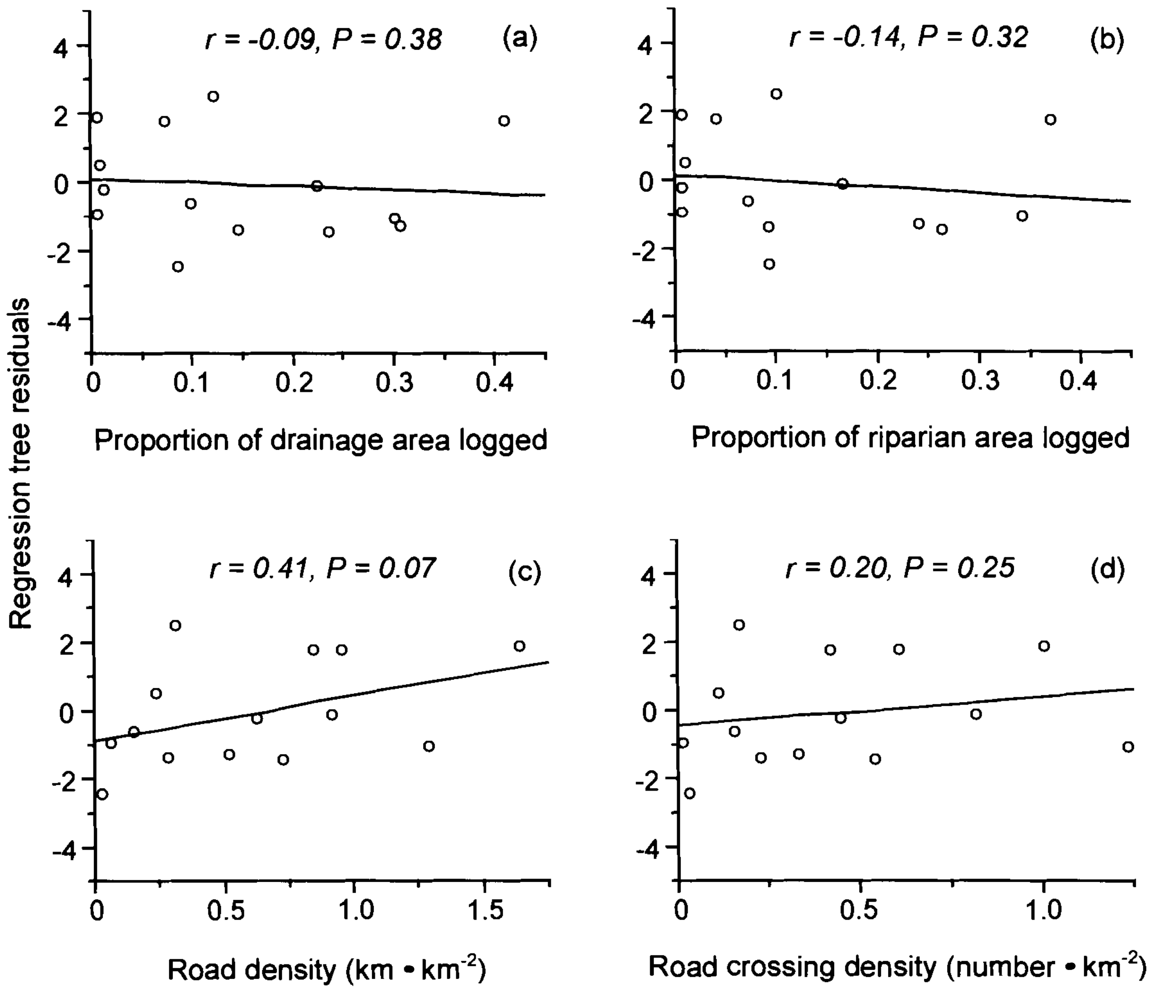


Figure 8. Marginal posterior probability distributions of the slope parameter from a Bayesian regression of the regression tree residuals and each of two measures of watershed-scale forestry activities; (a) the density of roads within the upstream basin, and (b) the density of road crossings within the upstream basin (linear regression models are presented in Figure 7c and 7d, respectively). Distributions were derived using data from streams in Stream Temperature Class II; 14 streams with drainage areas $\geq$ 12 km^2 and < 132 km^2, and average basin elevations < 1140 m. The solid vertical line emphasizes the point at which the regression slope equals zero, and the dashed vertical lines represent the 95% credibility intervals.