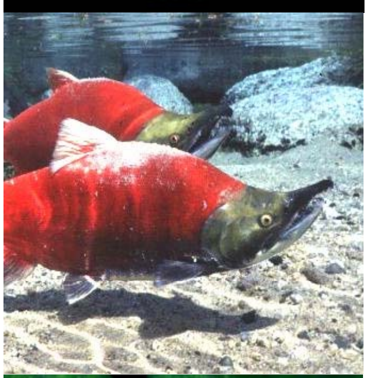
Magnitude-Duration Based Ecological Risk Assessment for Turbidity and Chronic Temperature Impacts:

Method Development and Application to Millionaire Creek



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Caveat

This report present the results of forward-looking R&D work performed by Aquatic Informatics Inc. (AI) on contract to BC MOE. As such, some of the methodologies outlined in this report remain experimental.

1. Overview

Stream water quality, and consequently the health of lotic ecosystems and salmonid populations, can be deeply impacted by watershed modifications. These include the effects of forestry, agricultural, mining, and industrial activities. Expansion of urban and suburban population and associated development, however, is quickly becoming a leading and potentially permanent cause of freshwater habitat degradation, a process that is accelerating in many regions including the British Columbia lower mainland. At the same time, public concern and demand for environmental sustainability is growing quickly. The net result of this juxtaposition of values and needs is an immediate requirement for accurate and scientifically defensible, yet easily understood and readily applied, tools for water quality monitoring, risk assessment, and watershed management.

Two ecologically highly salient water quality parameters potentially impacted by land use change are temperature and turbidity. However, existing practical tools for assessing temperature and turbidity impacts possess substantial limitations. A basic but thorny problem with setting guidelines for these parameters, and assessing levels in a given watershed for impact from of ongoing land use change, is that both are naturally highly variable in both time and space. Disentangling natural spatiotemporal variability from development impacts, and setting and enforcing appropriate regulatory criteria, can therefore be difficult from both a technical and a managerial perspective. Present tools do not provide a defensible, straightforward, and explicit mechanism for dealing with this challenge. An additional concern with current methods is the importance of including both the magnitude and the duration of exposure to elevated temperature and turbidity levels. Existing approaches do not, or only approximately, incorporate both of these controls upon total exposure to sub-optimal or hostile environmental conditions, instead focusing largely upon setting threshold temperature or turbidity values that primarily reflect magnitude considerations. Doing so is loosely akin to assessing the health consequences of smoking, without drawing an explicit distinction between smoking a single cigarette per year at a party and smoking two packs a day of the same brand. A third issue is that existing approaches used for general watershed monitoring, assessment, and management do not explicitly incorporate basic concepts from, and therefore lessons learned by, the broader risk assessment community. One obvious example is the aforementioned failure to properly recognize the importance of dose, the combination of exposure magnitude and duration. A substantial number of formal ecological risk assessments for individual rivers have certainly been performed, but these usually focus on toxicological risks, and tend to be highly site-specific, high-budget affairs that are not amenable to standard application as general watershed management tools.

Without questioning in any way the value of further basic research, none of the foregoing should be taken to imply that the scientific understanding necessary to adequately assess the excess turbidity or temperature risk associated with development or other land use change is unavailable. Rather, the practical challenge lies with integrating existing fisheries science knowledge into a formal but pragmatic framework for setting risk-based water quality objectives that can be readily applied by watershed managers. Specific requirements for such a protocol, then, are as follows. The method must:

(i) be quantitative, to permit comparison of results to some kind of numerical standard;
(ii) also generate that numerical standard;
(iii) be generally applicable;
(iv) nevertheless explicitly accommodate site-to-site variability in natural water quality background values;
(v) explicitly incorporate both magnitude and duration of exposure;
(vi) be as consistent as practicable with broader risk assessment concepts;
(vii) be based upon existing fisheries science knowledge and be scientifically defensible;
(viii) be logistically feasible to implement as a standard watershed monitoring and assessment tool, without a requirement for in-depth ecological studies on a site-by-site basis or extremely specialized technical knowledge on the part of watershed managers; and
(ix) yield a relatively straightforward result, preferably as some form of index, which clearly indicates whether an ecologically negative change in water quality conditions has occurred.

We propose here protocols for assessment and monitoring of cumulative, or chronic, risks to salmonids (in particular, coho and steelhead) from elevated water temperature, and to clear-water fish (including salmonids) from elevated turbidity. Both risk assessment methods meet all the requirements listed above and produce risk-based, site-specific water quality objectives. Of particular note is that both ultimately yield a risk quotient, RQ, analogous to that used in toxicological risk assessment, which provides a simple decision rule for watershed managers: $RQ \leq 1$ indicates an acceptable cumulative risk, whereas $RQ > 1$ indicates an unacceptable cumulative risk and, therefore, a need for further management action. That is, $RQ = 1$ is a risk-based, site-specific water quality criterion incorporating both magnitude and duration considerations.

The methods also produce clear visual portrayals of watershed conditions and, for chronic temperature risks, also lead to a convenient, three-tiered structure for risk assessment akin to that used in evaluating toxicological human health risk. Additionally, a simple but robust method for developing site-specific look-up tables for determining acceptable/unacceptable turbidity conditions for individual turbidity events is developed.

It is proposed that the methods introduced here fill an important practical gap between simple, threshold-based regulatory guidelines, which in general do not adequately capture the importance of both magnitude and duration of exposure or properly accommodate natural site-to-site variability, and detailed site-specific biophysical models, which are logistically infeasible for many or most standard environmental management applications.

Method development took place in the context of temperature and turbidity risk assessment for Millionaire Creek, Maple Ridge, British Columbia. All the methods developed were applied to Millionaire Creek, putting in place a potentially fully operational risk-based mechanism for assessing the water quality impacts of future activities in this watershed vis-à-vis chronic temperature and turbidity effects upon salmonids.

2. Assessment of Chronic Temperature Risk

2.1 Introduction

Elevated stream water temperatures present two general kinds of risks to salmonids: acute (lethal), and chronic (sub-lethal, or cumulative). Acute effects occur when fish are exposed to sufficiently high water temperatures for a sufficient amount of time to experience mortality. Chronic effects occur when fish are exposed to sufficiently high temperatures to compromise feeding, growth, disease resistance, competitive ability, predator avoidance, and migration and spawning success, primarily via bioenergetic (metabolic) pathways (see Kitchell et al., 1977; Elliott, 1981; Poole et al., 2001). Temperatures at which chronic effects occur are lower than those associated with acute risks. While chronic exposures by definition do not directly cause fish mortality over the short term, they can contribute to eventual mortality of individual fish and potentially lead to severe degradation of overall population viability (Poole et al., 2001).

Formal protocols for assessing human and ecological risks arising from toxins in the environment are well established. While much thought has been given to the fundamental science of environmental temperature effects upon salmonids, no parallel risk assessment framework has been formally developed and broadly accepted as a practical management tool for temperature risk assessment. A powerful complication with temperature risk is that meaningful and reliable, single-valued, risk-based threshold temperatures are difficult, and perhaps impossible, to develop.

Unlike most toxicological risks, water temperatures vary greatly in both space and time under fully natural conditions, even within a generally uniform hydroecological region, and thus are quite likely to be biologically sub-optimal at any given place and date in the absence of pollution. The net result is that no single threshold temperature can appropriately be set as a general watershed management standard, even for a single life stage. For example, a threshold high enough to account for naturally warm streams may leave thermal pollution in a colder river undetected, and a threshold low enough to detect thermal pollution in a cool river may flag naturally warmer rivers as being in violation (for detailed discussions, see Poole et al., 2001; Ice et al., 2004).

More fundamentally, such an approach fails to recognize the biological importance of both magnitude and duration of exposure (see Sullivan et al., 2000; Ice et al., 2004). One might attempt to circumvent this limitation through the use of summary metrics, such as the mean seven-day maximum daily temperature (e.g., Sullivan et al., 2000). However, this method incorporates magnitude-duration relationships in a manner that is highly imprecise, and its biophysical basis is non-explicit at best. Use of a single temperature threshold also seems to represent a major departure from standard ecological and human health risk assessment procedure, which is generally phrased in terms of dose, the product of exposure concentration (analogous to temperature) and duration (e.g., Caux et al., 1997). While fixed risk-based screening or remediation target concentrations are frequently encountered in toxicological risk assessment, these are based on specific exposure pathway and duration assumptions (e.g., ASTM, 1995). Conversely, analogous standard screening or remediation target temperatures likely cannot be reliably determined for natural rivers on the basis of a risk model using simple, generalized exposure duration assumptions, because natural thermal regimes exhibit such temporal and spatial variability (see above). Moreover, such an approach would at least require a formal assessment protocol for thermal risk to salmonids, explicitly incorporating both magnitude and duration of exposure.

Although concepts from toxicological risk assessment remain highly useful, and prior fisheries science examining water temperature impact is the cornerstone of any related risk assessment, alternative practical assessment protocols are therefore required to adequately monitor and manage temperature risk. Recent work has shown how acute temperature risks can be quantitatively assessed in a logistically feasible manner using a magnitude-duration curve approach (Sullivan et al., 2000; Quilty et al., 2004a). In addition, Sullivan et al. (2000) developed a growth model to assess chronic temperature risks to salmonids and considered the potential effects of both magnitude and duration. Ultimately, however, Sullivan et al. (2000) reduced the results to a risk-based temperature threshold for chronic impacts. While those thresholds are simple to implement, they do not incorporate heterogeneity in natural thermal regimes, or recognize the combined impact of magnitude and duration upon chronic thermal risk in a fully explicit manner.

The purpose of the current work is to develop a generalized method for quantitatively assessing chronic risks to salmonids from high stream water temperature, which adequately addresses the foregoing issues while remaining straightforward to implement as a practical watershed management tool. The resulting protocol is divided into two steps. Phase I yields a primarily visual assessment, and phase II provides a single but comprehensive risk index, the risk quotient (RQ), which gives a clear flag for the presence of ecologically negative changes in river thermal regime. The approach was developed for, and applied to, Millionaire Creek, British Columbia using validated and corrected 2001-2004 water temperature data and assuming coho and steelhead to be the target species for watershed management. However, this risk assessment protocol should be generally applicable to salmonids in lotic ecosystems elsewhere.

2.2 Method Development

Ecological risk assessment generally includes at least the following four elements: (i) assessment endpoint identification, (ii) effect analysis on the basis of the identified endpoint, (iii) exposure analysis on the basis of the identified endpoint, and (iv) risk characterization, integrating (ii) and (iii) (e.g., EPA, 2003). Endpoint identification and effect analysis in the context of assessing chronic temperature risks to salmonids in freshwater are discussed in sections 2.2.1, 2.2.2, and 2.2.3 below. Exposure analysis and risk characterization are discussed in sections 2.2.4 and 2.2.5. Our overall methodological emphasis is on the integration of reasonably well-established techniques and results from the fisheries science and general risk assessment communities, in order to develop a practical method for setting risk-based water quality criteria for chronic temperature impacts.

2.2.1 Growth, specific growth curves, and temperature: general

Growth is strongly sensitive to water temperature and is an effective metric for assessing the chronic impacts of water temperature upon fish (e.g., Hill and Magnuson, 1990; Burgner, 1991; Sandercock, 1991; McCullough, 1999; Ice et al., 2004; see in particular Elliott, 1981 and Sullivan et al., 2000). Even for anadromous species, which spend a relatively short part of their lives in fresh water, river temperature effects can be profound, particularly during the summer rearing period for young fish. The implication is not that maximization of growth should be regarded as management goal, which can have unexpectedly negative repercussions (see Poole et al., 2001). Rather, consistent with much previous fisheries research, we consider growth rate to be an effective general measure of the chronic biological impacts of elevated water temperature. The ultimate objective is to use such relationships to develop a risk assessment method which succinctly compares net observed impacts to those associated with the natural hydroecological regime.

We therefore need a quantitative method for relating water temperature to fish growth. This could be accomplished using a variety of sophisticated, process-based techniques; the most common of these (Jager et al., 1999; Railsback and Rose, 1999) may be bioenergetic modelling (e.g., Kitchell et al., 1977; Hill and Magnuson, 1990; Railsback and Rose, 1999). The utility of such models in a practical watershed management context, however, may be powerfully limited by logistical constraints. Environmental managers are typically responsible for many individual watersheds, and may have very limited funds for in-depth modelling (and the requisite data acquisition) for any given catchment. Some of these problems are particularly acute in regions where large numbers of small spawning streams are threatened by pervasive and accelerating human watershed modifications (e.g., the rapid urbanization occurring throughout much of Pacific coastal North America). Careful construction of a detailed, process-based watershed model of any kind (physical, chemical, or biological) is rarely feasible. To be potentially widely applicable as a practical monitoring and assessment technique, which is one of our primary goals (see Overview), a less time-, data-, and expertise-intensive approach is therefore needed. Moreover, the greater comprehensiveness of (for example) bioenergetic models also renders them less specific: it can be challenging to separate the modelled effects of different environmental parameters upon fish growth, so that the potential chronic impacts of raised temperatures may be difficult to specifically identify (see Railsback and Rose, 1999). This problem may be particularly troublesome in a risk assessment framework, where transparency is key (see effect of concern, below), and is further exacerbated by the relatively high uncertainty associated with the formulation and parameterization of full bioenergetic models (Railsback and Rose, 1999) and the potentially substantial systematic errors in their predictions (Bajer et al., 2004). Other detailed modelling approaches, such as individual-based population models, can offer some advantages over bioenergetic modelling (see, for example, Jager et al., 1999) but may require even more site-specific data acquisition and modelling to adequately calibrate (Railsback and Rose, 1999). Thus, without in any way questioning the value of sophisticated process-based models, it seems reasonable to posit that these are not the appropriate tools for ongoing, high-volume, risk-based monitoring and assessment of many-site environmental networks.

Here, we use relatively simple, empirical rules to describe the relationship between temperature and growth. The specific growth rate, g [g g^{-1} d^{-1}], gives the mass change of a fish per unit body mass per day. It is a roughly parabolic function of temperature for salmonids, reaching a maximum, g_o, at an optimal temperature, T_o, with $g < g_o$ for $T \neq T_o$. Temperature in this context is typically phrased as daily mean temperature, which is appropriate for evaluation of growth effects, diurnal temperature fluctuations notwithstanding (Sullivan et al., 2000).

2.2.2 Defining the effect of concern (EOC)

Defining the contaminant(s) of concern (COC) is a conceptually simple but important first step in ecological and human health risk assessments for contaminated sites, as it facilitates both efficiency and transparency by explicitly identifying the specific potential problem of environment management concern (e.g., ASTM, 1995). The concept is also useful for assessing non-toxicological environmental impacts. Here, we define an effect of concern (EOC), which is taken to be the chronic effect of elevated water temperatures ($T > T_o$) upon salmonid growth.

Low water temperatures can also lead to growth reductions relative to the optimum ($T < T_o$), but the primary watershed management concern for Millionaire Creek (and likely many other stream environments) is the converse. Thus, low water temperatures are not viewed as an EOC for this study. Note that due to the form of $g(T)$, low daily mean temperatures during certain days do not compensate for high daily mean temperatures during others.

2.2.3 Specific growth curves: coho and steelhead

The specific growth curve, and therefore values of g_o and T_o, vary between species. Specific growth curves are readily available for coho (*Oncorhynchus kisutch*) and steelhead (*Oncorhynchus mykiss*) (Sullivan et al., 2000). Both were used in this analysis. As insufficiencies or inaccuracies in the empirical relationships for g specified by Sullivan et al. (2000) preclude their direct use, and the EOC relates to elevated water temperatures, a polynomial fit to predicted $g(T)$ for $T \geq T_o$ as portrayed graphically by Sullivan et al. (2000) for C/C_{max} = 1 (see below) was performed using the Matlab™ function `polyfit`:

$$g = \beta_1 T^3 + \beta_2 T^2 + \beta_3 T + \beta_4 \qquad (1)$$

The sets of coefficients, β, for coho and steelhead are given in the table below; a large number of digits must be retained due to the large powers to which T is raised. For coho, g_o = 0.026 g g^{-1} d^{-1} and T_o = 16°C – 18°C, and for steelhead, g_o = 0.0315 g g^{-1} d^{-1} and T_o = 13°C – 14°C. Both specific growth curves exhibit nearly constant g over the foregoing optimal temperature ranges; for assessing the chronic effects of sub-optimally high temperatures ($T > T_o$), we set ${}^{coho}T_o = 18^oC$ and ${}^{steelhead}T_o = 14^oC$.

Polynomial coefficients for $g(T \geq T_o)$

coefficient	coho	steelhead
β_1	$-1.4855072464 \times 10^{-4}$	$-3.5031969347 \times 10^{-5}$
β_2	$8.3467908903 \times 10^{-3}$	$1.4732181572 \times 10^{-3}$
β_3	$-1.5707505176 \times 10^{-1}$	$-2.0793599899 \times 10^{-2}$
β_4	1.0152960663	$1.2980793533 \times 10^{-1}$

Salmonids present in Millionaire Creek are coho (*Oncorhynchus kisutch*), pink (*Oncorhynchus gorbuscha*), and chum (*Oncorhynchus keta*) salmon, rainbow (*Oncorhynchus mykiss*) and cutthroat (*Oncorhynchus clarki clarki*) trout, and mountain whitefish (*Prosopium williamsoni*) (see Quilty, 2001); of particular management concern are coho and chum (Quilty et al., 2004b). The coho specific growth curve above is therefore of direct importance to Millionaire Creek watershed management. Unfortunately, specific growth data appropriate to our present purposes do not seem to be readily available for chum (see Sullivan et al., 2000). While steelhead are not present in Millionaire Creek, a risk analysis for chronic temperature effects upon this species is performed due to the ready availability of steelhead specific growth curves; because steelhead is a variety of rainbow trout, which is present in Millionaire Creek; and because previously developed $g(T)$ relationships for steelhead exhibit a greater sensitivity to high temperatures relative to coho, i.e., ${}^{steelhead}T_o < {}^{coho}T_o$ (see above). The risk analysis for steelhead thus provides a potentially more conservative assessment of chronic temperature risks, where a conservative (liberal) assessment is taken to mean one which is more (less) protective of the environment, as per human and ecological risk assessment convention.

The specific growth curve also varies with percent satiation, the proportion (C/C_{max}) of the maximum food consumption for a given species (C_{max}) that is available to fish at a given location and time (C). Specifically, both optimal growth and the temperature at which it occurs increase with per capita food consumption, up to C_{max}. Available data suggest that C/C_{max} can vary widely in space and time; overall, however, little information is available regarding the value of this parameter under natural field conditions, and reliably ascertaining appropriate site-specific values under such conditions can be challenging (see Railsback and Rose, 1999; Sullivan et al., 2000). This is, of course, a problem with any method of relating fish growth to water temperature. Moreover, C and C_{max} can be influenced by T (e.g., Kitchell et al., 1977). The method could, in theory, be readily modified to accommodate time-varying satiation by expressing (1) in the form $g = g(T, C/C_{max})$, and incorporating observations of $C(t)/C_{max}$ if available, but this is rarely the case. Overall, it seems reasonable for our immediate purposes to hold C/C_{max} fixed. That is, we assume that C/C_{max} is a constant value for a given implementation of the technique so that $g = g(T)$ only as in (1).

Use of the $C/C_{max} = 1$ specific growth curve as indicated above effectively presumes that food availability is not a limiting factor upon fish growth. This is a potentially liberal assumption. Note, however, that the importance of correctly choosing C/C_{max} may be substantially reduced by the normalization involved in the calculation of a risk quotient, as discussed in due course.

2.2.4 Phase I assessment: cumulative magnitude-duration risk curves

Here we introduce magnitude-duration chronic risk curves, which are broadly analogous to the existing concept of magnitude-duration acute risk curves (see above). One significant difference, however, is that acute risk occurs when a certain temperature is exceeded continuously for a certain amount of time. In contrast, chronic risk is cumulative over the year. For example, under this framework, five days in a row of sub-optimally high temperatures have the same growth effect as five days of the same sub-optimally high temperatures interspersed with a few days of optimal temperatures.

The method first entails constructing a graph illustrating the number of days over the course of about a year during which a range of daily mean temperatures were matched or exceeded. The exact timeframe considered is irrelevant, provided that all days potentially for which $T > T_o$ are included, without gaps; for coastal British Columbia streams, this is about June - August. That is, the horizontal and vertical axes are stressor magnitude and cumulative duration at or above that magnitude, respectively. Superimposed upon this graph of observed values are a series of vertical lines, each representing a temperature corresponding to a different daily growth risk. Sullivan et al. (2002) defined percent daily growth risk, denoted *DGR* here, as:

$$DGR = \left(1 - \frac{g}{g_o}\right) \cdot 100 \qquad (2)$$

DGR = 0% indicates $T = T_o$ (optimal daily conditions, $g = g_o$, $T = T_o$) for a given day, whereas DGR = 100% indicates no daily growth. Intermediate DGR values indicate intermediate chronic growth risks. Values of the magnitude-duration scatterplot which lie to the right of the DGR = 0% line indicate days during which high-temperature chronic growth risk was incurred in the watershed. The result may be viewed as a form of the stressor-response versus cumulative exposure distribution method for ecological risk assessment (see EPA, 1998).

The procedure is implemented as follows. First, for a gap-free daily mean temperature record, T_t, $t = 1,N$, where N is the number of days of record, a plot of the cumulative number of days (duration) observed to exhibit a temperature equal to, or greater than, each observed temperature (magnitude) is constructed:

$$duration(T_i) = \left[\sum_{k=1}^{N} I(T_k \geq T_i)\right] \;\; \forall \; i = 1, N \qquad (3)$$

where I is the indicator function:

$$I = \begin{array}{c|l} 1 & T_k \geq T_i \\ 0 & otherwise \end{array} \qquad (4)$$

Second, critical daily mean water temperatures corresponding to a selected suite of DGR values are calculated; $T_{DGR=0\%}$, $T_{DGR=5\%}$, $T_{DGR=10\%}$, and $T_{DGR=20\%}$ were considered here. Specifically, (2) may be re-arranged to provide the growth rate corresponding to a particular growth risk (DGR = 0, 5, 10, or 20% in our case):

$$g = g_o\left(1 - \frac{DGR}{100}\right) \qquad (5)$$

The corresponding value of temperature may then be found by setting g in (1) to the value found using (5) and solving for T; the Matlab™ function `fsolve` was employed for the purpose here. Finally, the four resulting values of T_{DGR} are plotted over the magnitude-duration curve found using (3) to ascertain presence and degree of chronic risk associated with observed daily water temperatures.

2.2.5 Phase II assessment: risk quotient for chronic growth impacts

The foregoing method, while quantitatively based, serves primarily as a visual check of water temperature data for potential chronic growth risks. Here we develop a second approach, fully complementary to the above magnitude-duration curve method, which yields a single number that can be used as a powerful guideline for risk assessment: a chronic risk quotient. Unlike a fixed upper temperature threshold, however, the chronic risk quotient incorporates both magnitude and duration considerations, explicitly reflects bioenergetic requirements and assumptions, and is tuned to the natural temperature conditions of individual watersheds.

Specific growth rate amounts to the derivative of fish mass, normalized by initial total mass, with respect to time. Thus, cumulative proportional yearly growth is clearly the time integral of daily observed specific growth, which in turn may be evaluated as a function of daily mean temperature. We can therefore introduce a percentage total growth risk, *TGR*, analogous to *DGR* but cumulative over the year (or annual rearing season):

$$TGR = 100\% \cdot \int_{t_0}^{t_1} G(t)dt \tag{6a}$$

with:

$$G(t) = \begin{array}{ll} g_o - g(t) & \Big| \ \arg\left[t \mid T(t) > T_o\right] \\ 0 & \Big| \ \textit{otherwise} \end{array} \tag{6b}$$

where $\arg[t \mid T(t) > T_o]$ denotes values of t such that $T(t)$ is greater than the optimal value, T_o. The condition listed in (6b) ensures that days for which growth losses arise from sub-optimally cool temperatures do not influence the assessment. Equation (6) may be rewritten in discrete form, corresponding to the discrete nature of environmental sampling, as:

$$TGR = 100\% \cdot \sum_{t=t_0}^{t_1} G_t \Delta t \tag{7a}$$

with:

$$G_t = \begin{array}{l|l} g_o - g_t & \arg\left[t \mid T_t > T_o\right] \\ 0 & \textit{otherwise} \end{array} \tag{7b}$$

where the sampling interval, Δt, is 1 day and g_t is obtained on a daily basis using (1) with T set to the observed daily mean temperature, T_t.

In general, the limits of integration (6) or summation (7) must bracket the portion of the year when there is potential for $T > T_o$, as was the case for the graphical magnitude-duration approach above; failure to do so may yield risk underestimates. Beyond this, there is no restriction upon or importance to the choice of (t_o, t_1). Beginning and end of the calendar year are often convenient. Note, however, that use of this method is not limited to retrospective analyses. In summer, for instance, one could recalculate (7) on a daily, almost-real-time basis, setting t_1 to the day previous to the analysis date and making use of the previous day's observed T_t, in order to obtain an evolving measure of current chronic growth risks in the watershed.

Total growth risk, $TGR \geq 0$, gives the loss in cumulative percent yearly growth due specifically to high temperature ($T > T_o$), relative to the growth that would have occurred under optimal thermal conditions ($T = T_o$) on days of temperature exceedance. We make three important notes regarding the meaning of *TGR*. First, given the EOC defined previously, (6) and (7) are deliberately constructed such that sub-optimally low temperatures do not lead to $TGR > 0$. Second, the observed value of *TGR* depends upon the magnitude of exposure, cumulative duration of exposure, and the thermal requirements of individual species. For example, relative to steelhead, coho growth is less sensitive to thermal stress, so for a given observed temperature dataset ${}^{coho}TGR$ may be 0 whereas ${}^{steelhead}TGR$ may be > 0. Third, *TGR* is not referenced to natural watershed conditions. No river is consistently at T_o. In particular, naturally warmer rivers may consistently exhibit $T > T_o$ over some portion of the year and therefore $TGR > 0$. Consequently, a non-zero observed *TGR* value does not necessarily indicate negative environmental impacts from, for example, human watershed modification. Conversely, if the natural conditions for a particular river and species are such that, usually, $T << T_o$, even a small positive *TGR* value may indicate physically and ecologically severe changes to watershed conditions.

We now define a risk quotient, RQ, analogous to the risk or hazard quotients widely used in conventional (toxicologically oriented) ecological and human health risk assessment (e.g., ASTM, 1995; EPA, 1998), which quantifies the chronic effects of sub-optimally high water temperatures through a simple index:

$$RQ = \frac{TGR_{obs}}{TGR_{ref}} \tag{8}$$

where TGR_{obs} is the observed value of TGR in a given year, and TGR_{ref} is a reference value which describes the acceptable chronic total growth risk, preferably on the basis of observations obtained over a baseline period.

The risk quotient possesses many of the same desirable qualities as TGR. RQ reflects the combined effects of exposure magnitude, exposure duration, and species-specific temperature requirements; it is sensitive specifically and exclusively to the prescribed EOC; and it may also be implemented on a near-real-time basis for day-by-day assessment of evolving seasonal chronic risk. However, it offers two significant advantages over the use of TGR alone.

First, normalization of observed risk in a given year by a reference TGR value, obtained from the same river over a baseline period, facilitates a more appropriate and robust metric of chronic thermal risk due to watershed modification. By focusing on changes from baseline conditions, rather than absolute growth values and risks, the risk quotient adjusts (at least in part) for the following complications: (i) natural thermal conditions in the study watershed, compensating for naturally cooler or warmer streams; (ii) potentially substantial spatial heterogeneity in water temperature within a given stream (e.g., Malcolm et al., 2004), insofar as installation of monitoring equipment at a relatively warm stream location, for example, is in part compensated for by the normalization; and (iii) potential biases between the laboratory studies from which $g(T)$ relationships are derived and the hydroecological relationships experienced by fish in the field (see, for example, Sullivan et al., 2000). Likewise, framing the risk assessment in terms of comparison to a historical baseline value may reduce the importance of correctly setting site-specific C/C_{max}, as both the numerator and denominator of (8) would be similarly affected; an analogous argument might apply to possible size-dependence (e.g., Sullivan et al., 2000) of the specific growth curve. Additionally, normalization by a reference value partially generalizes the risk assessment across species, in that some information regarding between-species variability in thermal requirements is encapsulated by TGR_{ref}. The latter consideration does not obviate the desirability of performing separate analyses for each species present, at a minimum because the form of $g(T)$ varies between species; nevertheless, it may prove useful when limited data or resources are available, which is often the case in practice.

Second, the risk quotient leads to a formal, simple, and robust decision rule for risk assessment and watershed management:

$$\begin{aligned} &RQ > 1 : \textit{unacceptable risk} \\ &RQ \leq 1 : \textit{acceptable risk} \end{aligned} \tag{9}$$

Observed $RQ > 1$ thus indicates high-temperature chronic growth risks in excess of typical / acceptable levels. Note that $RQ = 0$ indicates no chronic high-temperature risk at all that year, and $0 < RQ \leq 1$ indicates that such risk was incurred but was within acceptable (e.g., natural historical) limits; only if $RQ > 1$ is there cause for management concern. The risk quotient method therefore provides a single, straightforward metric that serves as a clear flag for the presence of ecologically negative changes in stream temperature conditions. In particular, $RQ = 1$ is the site-specific, risk-based, magnitude/duration-based water quality objective.

Such changes include the effects of watershed modifications, such as those associated with urban development and other land use changes potentially capable of raising stream temperatures. However, although this method explicitly adjusts for naturally lower-than-optimal water temperatures when TRG_{ref} is based upon a historical record, observed $RQ > 1$ could conceivably arise from anomalously large natural effects, such as particularly substantial El Niño-Southern Oscillation (ENSO) events. While the corresponding RQ would remain valid, external constraints and professional judgement are therefore still required to correctly attribute a high RQ to its source(s). By the same token, however, the risk quotient can also be used as a measure of salmonid growth risks potentially associated with large-scale ocean-atmosphere circulation patterns, such as ENSO, or with longer-term climatic changes.

In general, the historical record from which TGR_{ref} is determined should be as long as possible and encompass a period of relatively stable and/or natural watershed conditions. Note, however, that an unusually long temperature record may pose slight difficulties in defining a historically-based TGR_{ref} due to decadal-scale climate nonstationarities, such as those arising from Pacific Decadal Oscillation regime shifts or climatic change, which may in turn induce nonstationarity in stream temperature data. Roughly a decade of data may be ideal, although substantially shorter or longer records would remain serviceable.

A variety of potential choices for TGR_{ref} exist. Note that lower TGR_{ref} values yield higher RQ for a given TGR_{obs} and are thus more protective of aquatic habitat. Options include the mean or median value of annual TGR_{obs} over the baseline period, the positive 68% or 95% confidence level on that historical mean, one or two standard deviations above the historical mean, an interquartile range above the historical median, or some percentile of historical yearly values. A short baseline dataset might not adequately capture natural interannual temperature variability, and the foregoing statistical summary measures could therefore be poorly constrained; in such a case, setting $TGR_{ref} = \max(TGR_{obs})$ may be sensible. If $T \leq T_o$ every day over the baseline period without exception, then (8) is mathematically undefined for the above TGR_{ref} definitions, but may still be evaluated by setting TGR_{ref} to an arbitrarily small number (0.01 is adequate). Doing so may yield very large RQ values should $T > T_o$ occur in the future, but an alarmingly large RQ would seem appropriate, from a practical management perspective, for a stream in which chronic risks have never before been incurred.

If water temperature records are available but span a period of watershed modification, it may still be feasible to use such data to set a historically based TGR_{ref} using one of the foregoing techniques. The requisite condition is that either the inferred stream temperature impact of previous development is negligible; or that the practical management goal is to ensure that future watershed modification does not degrade lotic habitat quality beyond current, albeit potentially already impacted, levels. If no usable baseline information exists, it may be necessary to employ data from nearby streams possessing similar thermal regimes as a surrogate or, where possible, to statistically reconstruct daily temperature data for the study river from predictor variables (e.g., air and adjacent stream temperatures), provided such surrogate data are available. Alternatively, TGR_{ref} may be set on the basis of broader considerations. For example, noting practical uncertainty levels associated with sampling of fish size distributions, Sullivan et al. (2000) suggested that a 10% total annual growth loss may be an acceptable risk threshold for anadromous salmonids. While likely necessary in some instances, such a fixed-loss approach is generally inferior to a baseline-derived TGR_{ref} as it requires a semi-subjective risk choice and, in particular, does not adjust the final risk index for natural local conditions or the other complications discussed above.

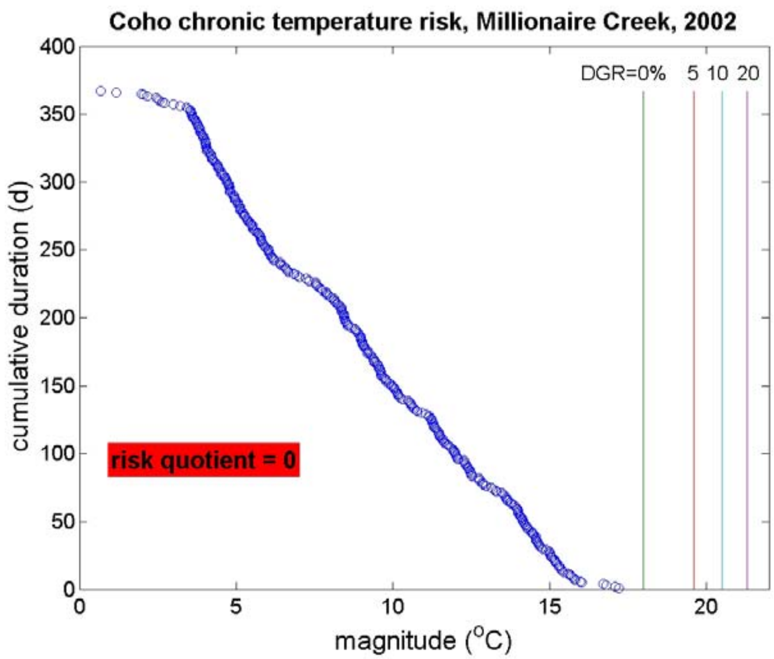
Ultimately, identification of a single, fully universal approach to setting TGR_{ref} may not be appropriate. Rather, TGR_{ref} selection might best be performed on a watershed-by-watershed basis. Key considerations include data availability; magnitude of interannual water temperature fluctuation, which reflects in part the regional hydroclimatic regime, and helps determine requisite baseline dataset size; degree of management concern; and the levels of conservatism of individual watershed stakeholders. Such subjectivity is ultimately unavoidable in watershed management, which lies at the intersection of socioeconomic policy and physical and life science. Nevertheless, the methods outlined above provide a quantitative, explicit, uniform, and scientifically sound protocol for making watershed management decisions with respect to stream temperature impacts.

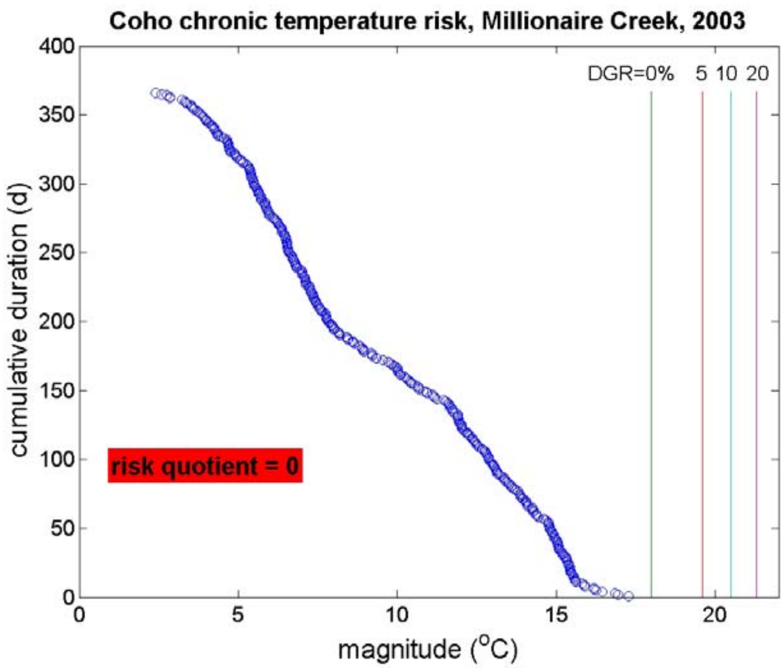
2.3 Application to Millionaire Creek

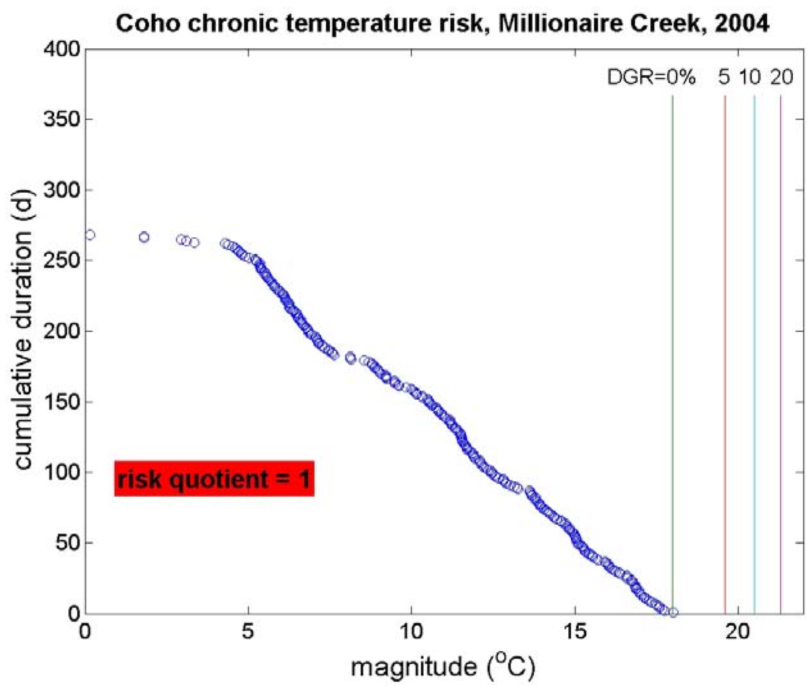
The magnitude-duration curve and risk quotient methods developed above were applied to water temperature data from Millionaire Creek on a yearly basis using a Matlab™ script written for the purpose. We employed daily mean temperatures as calculated from corrected and validated 15-minute raw data (see Appendix). The available temperature time series spans late October 2001 to late September 2004; note that the 2004 data sufficiently bracket the local range of days over which, potentially, $T > T_o$. Analyses were therefore performed for 2002, 2003, and 2004.

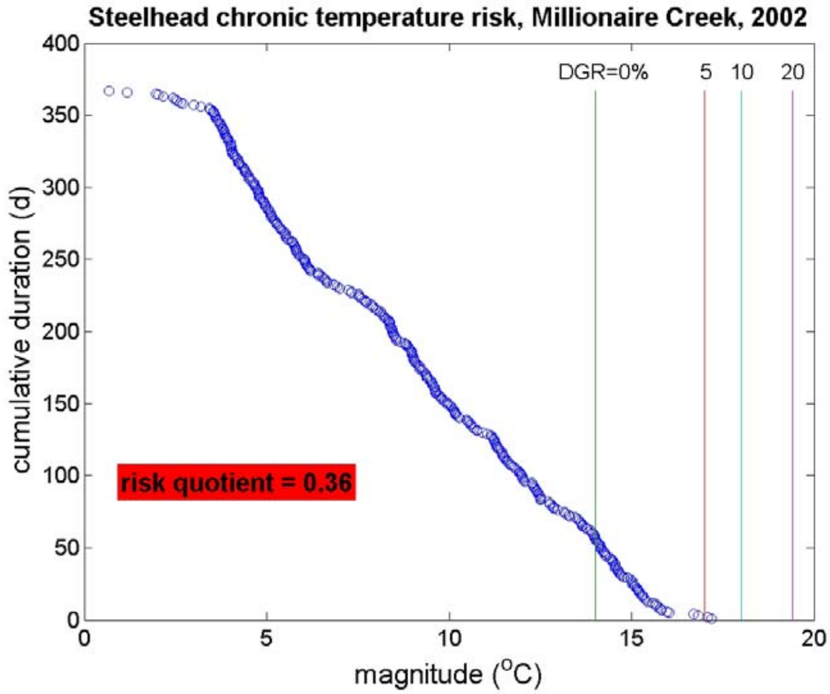
Options for setting TGR_{ref} include using historical baseline data from Millionaire Creek, historical baseline data from adjacent creeks, and a risk-based, non-site-specific reference value (see above). Unfortunately, the Millionaire Creek data are of modest duration, and while largely forested with some light urban and agricultural land use, the watershed was non-pristine over this entire three-year period. Watershed alteration increased somewhat in early 2004 with the start of a development adjacent to North Millionaire Creek, a tributary (Rod Shead, B.C. Ministry of Environment, pers. com., 2004). However, data from other creeks in the region were not deemed appropriate for use as a surrogate. Moreover, analyses using Millionaire Creek data yielded *TGR*s that were in all cases below 10%, a value that has been suggested, on the basis of sampling uncertainty rather ecological health considerations, as a potential acceptable growth loss. Thus, for Millionaire Creek, substantial deterioration in habitat quality could be incurred, relative to pre-existing conditions, without triggering $RQ > 1$ if this non-site-specific reference value is used. We therefore set TGR_{ref} to the maximum annual value found over the three summers of available Millionaire Creek data: ${}^{coho}TGR_{ref} = 0.005\%$, and ${}^{steelhead}TGR_{ref} = 7\%$. The implied management goal is to ensure that the Millionaire Creek thermal regime is not degraded beyond present, likely non-impacted to moderately impacted, conditions.

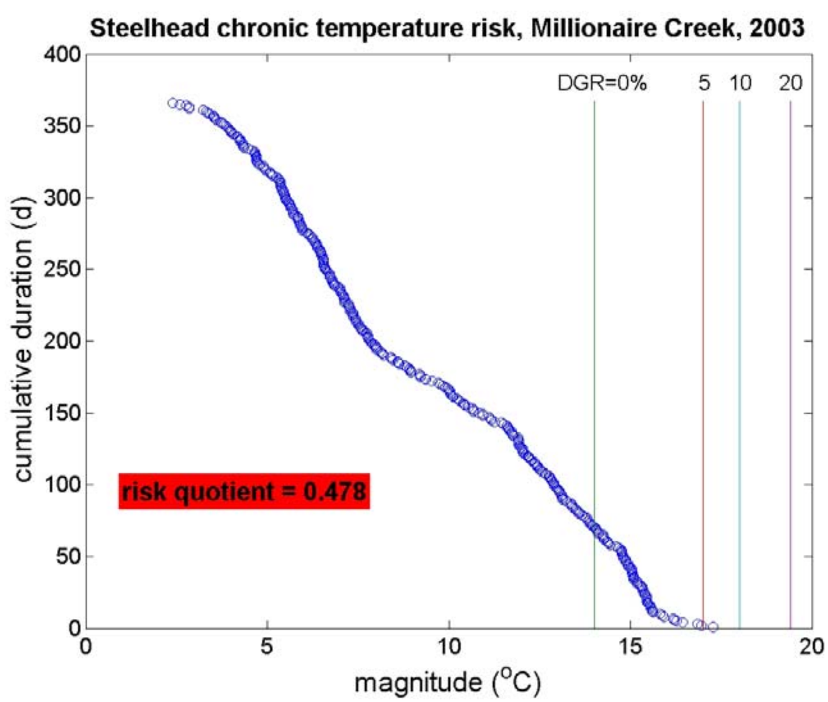
Ideally, reference values of *TGR* would be set using a long historical record and the risk assessment would then be applied to temperature data from a subsequent year, which would not be used in the evaluation of TGR_{ref}. Nevertheless, this application to Millionaire Creek provides a good illustration of the method and, in particular, yields a reasonable baseline risk against which future Millionaire Creek water temperature impacts can be compared. Results are illustrated in the following figures.

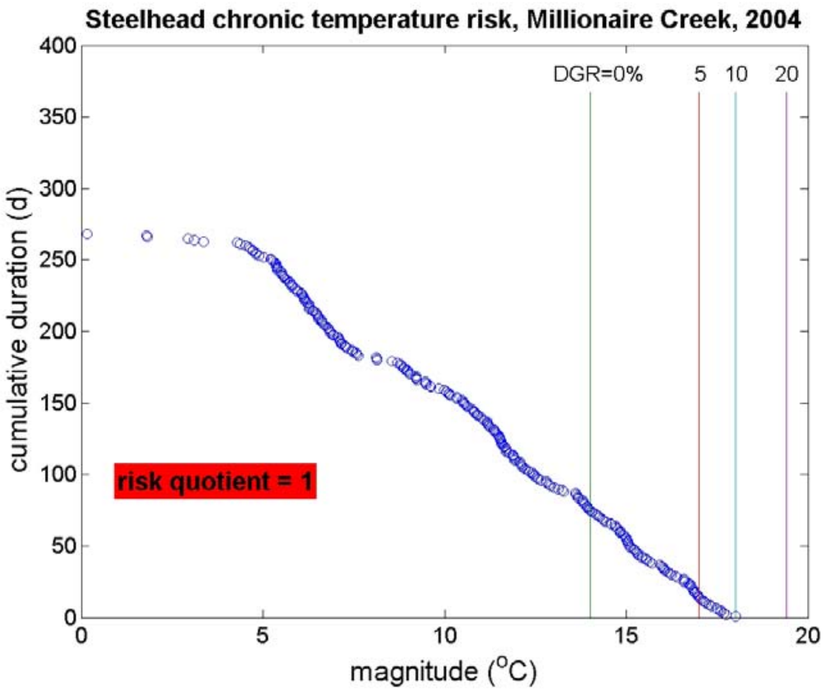



Coho chronic temperature risk, Millionaire Creek, 2003
DGR=0%
5 10 20
risk quotient = 0
cumulative duration (d)
magnitude (°C)


Coho chronic temperature risk, Millionaire Creek, 2004
DGR=0%
5 10 20
risk quotient = 1
cumulative duration (d)
magnitude (°C)

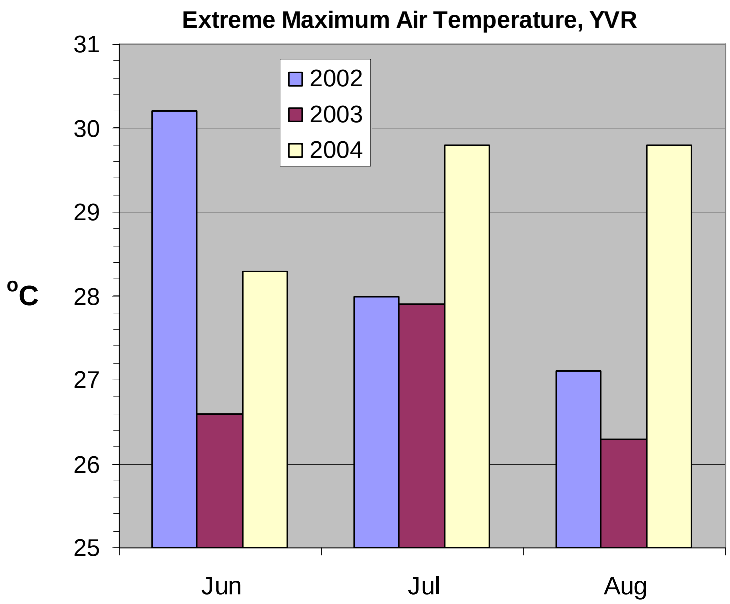

Steelhead chronic temperature risk, Millionaire Creek, 2002
DGR=0%
5
10
20
cumulative duration (d)
risk quotient = 0.36
magnitude (^oC)
0
50
100
150
200
250
300
350
400
5
10
15
20

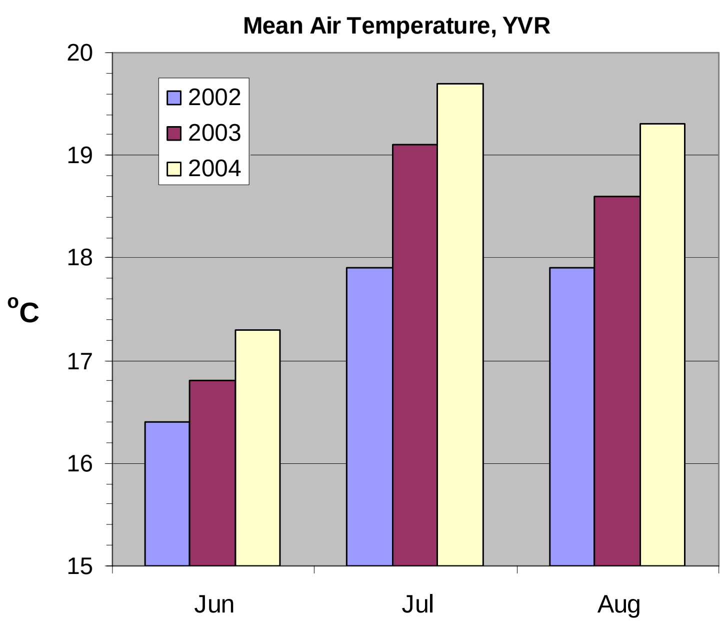

Steelhead chronic temperature risk, Millionaire Creek, 2003
DGR=0%
5
10
20
cumulative duration (d)
risk quotient = 0.478
magnitude (^oC)
0
50
100
150
200
250
300
350
400
5
10
15
20



Due to the TGR_{ref} definition applied, no *RQ* exceeds unity over the baseline period. Substantially longer exposures to higher water temperatures occurred in 2004, relative to 2002 and 2003, resulting in the highest *TGR* values observed for each species over the baseline period and thus, by construction, *RQ* = 1. Any future annual *RQ* > 1 would be indicative of unacceptable risk and be cause for management concern.

Although the higher water temperature and risk for 2004 loosely coincide with renewed development activity in the watershed, two considerations suggest that there may be little or no causal relationship in this case. First, analyses for turbidity risk (see following chapter, this report) indicate that turbidity levels, which are also sensitive to development activity, are low to moderate in 2004 relative to prior years. Second, the higher water temperature and risk values for 2004 are likely due, at least in part, to variability in climatic forcing. The nearest air temperature station for which 2002-2004 summer data are fully available at present is Vancouver International Airport (YVR; available at www.climate.weatheroffice.ec.gc.ca/ climateData/ monthlydata_e.html). We considered two monthly time series, consisting of average daily mean temperature, and monthly extreme maximum daily temperature, over the June-August period of each year. Plots are shown below. For both metrics, the summer averages (the average for a given metric over June-August of a given year) are equal between all years within one confidence interval about the mean. However, the 68% confidence band is wide due to the small number of samples used to calculate each mean (n = 3, i.e., June, July, and August), so a statistical comparison of this type offers little useful information. Graphically, however, it is readily apparent that 2004 values are substantially higher than in 2002 and 2003.





2.4 Synthesis

2.4.1 General framework for risk assessment and risk-based objectives

On the basis of the foregoing work, a three-tiered approach to risk assessment for chronic temperature impacts, analogous to that used (for example) in assessment of toxicological risks to human health (e.g., ASTM, 1995), can be proposed as follows:

Tier I Screening levels are magnitudes which, if exceeded, trigger closer scrutiny, but not necessarily regulatory or legal action. The needed sensitivity requires that levels be set relatively low. Risk-based screening levels applied to other scenarios (e.g., ASTM, 1995) require broad, standardized exposure duration assumptions which, as noted in the introductory section of this chapter, are difficult to establish in a reliable manner for stream temperature. A good choice, then, is the daily optimal growth temperature for the species of concern, T_o. If multiple species are watershed management targets, the lowest T_o should be used. If no observed daily temperature exceeds T_o, then there is no temperature-induced chronic growth risk. An exceedance is defined as any observed daily mean temperature greater than T_o, and triggers a Tier II assessment. For some species and rivers, the Tier I criterion will be naturally and regularly exceeded, requiring all ongoing monitoring, assessment, and management to be performed using a Tier II procedure.

Tier II The Tier II assessment consists of the two-phase risk assessment procedure introduced in this chapter, which explicitly incorporates exposure magnitude and duration and baseline spatiotemporal variability in stream temperature regime. Tier II assessment should ideally be performed for every salmonid species of concern. While both phases of the Tier II procedure should be completed, the bottom-line product from a management decision perspective is the risk quotient. If $RQ \leq 1$, then risk is judged to be acceptable. If $RQ > 1$, then unacceptable risk has been incurred by the river of concern and a Tier III assessment is required.

Tier III The Tier III assessment consists of reasonably attributing a Tier II exceedance to its source. This procedure might often be successfully performed using site visits, interviews, qualitative data interpretation, and other so-called “soft” approaches. In more complex or highstakes circumstances, minor to extensive additional quantitative analysis may also be necessary. This can include additional quantitative data collection and statistical and/or process-based physical modeling. Details of the Tier III assessment will in general be highly site-specific and should not be standardized, although there may be room for setting broad Tier III protocols.

The Tier II risk assessment serves as a practically feasible means for establishing risk-based, site-specific water quality objectives and, subsequently, as a basis for monitoring temperature data for compliance with these criteria. Specifically, the water quality objective for chronic temperature impacts may be defined as follows:

Quantity	**Criterion**
RQ	Must be ≤ 1

This water quality objective is intended largely as a complement to, rather than a replacement for, current temperature objectives (depending to some degree on the sophistication of current local objectives). Note that, by construction, our method does not address acute risks to fish (primarily a summertime concern, like the chronic risks considered here) or temperature impacts upon eggs and alevin (for salmonids, more typically but not universally a wintertime concern). Additional criteria are necessary, and are currently in place in some jurisdictions, for these other types of thermal risk. However, the risk quotient concept used here might also be adapted to such additional risk types. Doing so might have the potential to yield a complete suite of risk-based, site-specific water temperature objectives appropriate to all life stages and risk types.

2.4.2 Application of general framework to Millionaire Creek

For Millionaire Creek, baseline assessments indicate that Tier I levels are naturally violated for both coho and steelhead. Thus, ongoing assessment of chronic temperature risks in Millionaire Creek require continued application of Tier II procedures, using the TGR_{ref} values for steelhead and coho defined in this report using the baseline dataset. $RQ = 1$ is the risk-based water quality objective, and any future $RQ > 1$ will be cause for management concern and should trigger a Tier III assessment.

3. Assessment of Turbidity Risk

3.1 Introduction

Sediment suspended in the water column can harm fisheries resources via optical and non-optical pathways. Non-optical impacts consist of direct biophysical implications, such as gill membrane damage and reduced capacity for gas exchange, habitat degradation by blanketing of stream beds with fine sediment, and enhanced contaminant mobility. Optical impacts are associated with reductions in water clarity and light transmission. Such impacts include reductions in the volume of the photic zone and thus in primary production, initiating a negative trophic cascade throughout the ecosystem and potentially altering natural species assemblages and diversity; and harmful alteration of natural feeding efficiency, behaviour patterns, and predator-prey interactions. For recent reviews, see Caux et al. (1997), Welch et al. (1998), and Newcombe (2003).

Fluvial suspended sediment concentrations and lack of visual water clarity are increased by activities and land use changes within a watershed which potentially enhance erosion rates, including logging, road construction, mining, agriculture, and of exponentially increasing concern, urban development. There is, therefore, a strong need to monitor, assess, and manage attendant water quality changes. Due to logistical considerations, suspended sediment concentrations and water clarity are most often measured as turbidity, a semi-physical parameter defined by its own unit of measurement, the nephelometric turbidity unit or *NTU* (e.g., Welch et al., 1998). The use of a constant upper *NTU* threshold as a fixed criterion for water quality within a given regulatory jurisdiction is usually appropriate for setting drinking water standards, particularly if treatment facilities or multiple reservoirs are available, giving some flexibility to the water supply system if the threshold is violated. From a more general watershed and ecological management perspective, however, such an approach is subject to two very strong limitations. These disadvantages are closely analogous to those associated with the application of single-valued thresholds to assessment of risk associated with high water temperatures (see previous chapter, this report).

First, the turbidity of natural streams is extremely variable in both space and time (e.g., Caux et al., 1997; Welch et al., 1998). Coastal rivers of British Columbia and the U.S. Pacific northwest, for example, are typically clear-water oligotrophic streams, but can experience very high turbidity during rainstorms. Within this region, background turbidity also varies markedly with topographic and geologic characteristics of the individual watershed, and glacial rivers, for instance, carry much higher sediment loads than nival or pluvial streams. Thus, application of a constant upper threshold value is problematic.

Second, and more fundamentally, employing a single *NTU* value as an upper limit addresses only the magnitude, not the duration, of turbidity events. Moderate but sustained turbidity levels can have fisheries consequences exceeding those arising from a sharp but short-lived turbidity spike (Caux et al., 1997; Newcombe, 2003). One might attempt to circumvent this limitation using some summary metric – say, the six-hour mean or seven-day mean daily maximum *NTU* – but such an approach is highly imprecise, and its biophysical basis is non-explicit at best.

Attempts have been therefore been made to create turbidity guidelines which are more flexible than a single threshold value, and which incorporate both magnitude and duration criteria. Nevertheless, a tendency remains for expressing such guidelines in terms of threshold *NTU* values, albeit variable ones. For example, the British Columbia regulatory criteria for protection of aquatic life are as follows (almost verbatim from Singleton, 2001; see also Caux et al., 1997). (i) For clear flow periods, induced turbidity should not exceed background levels by more than 8 *NTU* during any 24-hour period (hourly sampling preferred). For sediment inputs that last between 24 hours and 30 days (daily sampling preferred), the mean turbidity should not exceed background by more than 2 *NTU*. (ii) For turbid flow periods, induced turbidity should not exceed background levels by more than 8 *NTU* at any time when background turbidity is between 8 and 80 *NTU*. When background exceeds 80 *NTU*, turbidity should not be increased by more than 10% of the measured background level at any one time. (iii) The clear and turbid flow periods are defined by the portion of the hydrograph when suspended sediment concentrations are low (taken to be less than 8 *NTU*) and relatively elevated (taken to be greater than or equal to 8 *NTU*), respectively.

Apart from its awkwardness, the foregoing approach also remains technically problematic. Dividing flows into clear-water and turbid-water regimes, each with a separate set of regulatory criteria, may be difficult in practice, particularly for smaller rivers and streams with flashy hydrologic responses. Moreover, the validity of this dual-standard model as a watershed management tool is unclear: the increased erosion potential associated with ongoing anthropogenic watershed alterations may be more likely to show up during periods with generally high flow and turbidity, so raising the tolerance level during such periods may desensitize the monitoring and assessment protocol to the very phenomena it is intended to detect. Conversely, regulatory criteria must not be set so strictly as to identify a pristine watershed as polluted. In addition, exposure-duration relationships are only approximately incorporated into the criteria, and watershed-to-watershed variability in background turbidity is not fully accounted for. A more uniformly and easily applicable, yet rigorous, method would thus be desirable.

Here, we introduce two methods for assessing turbidity risk, intended as practical watershed management tools. Both are extensions of the severity-of-ill-effect index developed by Newcombe (2003) for optical, or visual clarity, impacts. The first focuses on individual turbidity events. Specifically, the method yields a means for developing look-up tables customized to individual watersheds, which provide an easily-applied algorithm for making action-no action management decisions on an event-by-event basis as they occur. The second provides a formal risk assessment framework for evaluating the cumulative risk to fisheries health from lack of water clarity. It is phrased in terms of duration-magnitude curves and, ultimately, a risk quotient, analogous to those introduced for chronic temperature effects in the preceding chapter of this report. Both approaches are referenced to historical watershed conditions and explicitly incorporate magnitude and duration considerations.

3.2 Method Development

Endpoint identification and effect analysis in the context of assessing cumulative turbidity risks to clear water fish in freshwater are discussed above and in section 3.2.1 below. Exposure analysis and risk characterization are discussed in sections 3.2.2, 3.3.3, and 3.2.4. Additional aspects of risk characterization are further explored in 3.2.5. As in the previous chapter, our overall methodological emphasis is on the integration of reasonably well-established techniques and results from the fisheries science and general risk assessment communities, in order to develop a practical risk assessment method for chronic turbidity impacts functionally superior to those currently available.

3.2.1 The severity-of-ill-effect index

The severity-of-ill-effect index, or *SEV*, was introduced by Newcombe (2003). It assesses the impacts of water clarity losses to clear-water fish species as a function of both the magnitude and duration of turbidity events. The method was developed primarily through meta-analysis of available literature and consensus-based peer consultation. The index is given by:

$$SEV = -4.49 + 0.92[\ln(t)] - 2.59[\ln(yBD)] \tag{10}$$

where t is the elapsed time (hr) over which a particular black-disk sighting distance, yBD (m), is sustained. The black-disk sighting distance is related to turbidity by (Newcombe, 2003):

$$\ln(yBD) = 5.572012 - 0.80137\ln(NTU) \tag{11}$$

where [yBD] = cm; note different dimensions from (10). Larger SEV indicates worse effects. Newcombe (2003) proposed the following rating scheme, which has since been applied to practical exercises in watershed management (e.g., Quilty et al., 2004):

***SEV* rating criteria**

index	effect
$0 \le SEV < 0.5$	nil
$0.5 \le SEV < 3.5$	minor
$3.5 \le SEV < 8.5$	moderate
$SEV \ge 8.5$	severe

Note that the formulation of (10) can lead to negative SEV for small NTU and t. This does not imply that the corresponding turbidity event magnitude and duration are ecologically beneficial relative to zero turbidity. Newcombe (2003) implicitly applied the following cutoff to (10): $SEV \equiv 0$ if $SEV(NTU, t) < 0$. In our work, we consider a turbidity event to contribute to net risk only if $SEV \ge 0.5$ (see above table and sections 3.2.3 and 3.2.4 below).

3.2.2 Defining turbidity events

We begin by defining any period over which NTU continuously > 1 as a turbidity event. For risk assessment, however, we must also decide how to pick one out of a suite of overlapping turbidity events of different magnitude and duration. Turbidity often exhibits a temporal pattern roughly similar to that illustrated schematically in the following table:

Hypothetical turbidity data

elapsed time (hr)	*NTU*
0	0.2
1	0.5
2	1.1
3	5.8
4	20.6
5	12.4
6	4.7
7	2.4
8	1.3
9	0.9
10	0.7

Over the interval considered, we have one turbidity sub-event at >1 *NTU* for ~7 hr, another at >5 *NTU* for ~3 hr, and a third at >20 *NTU* for ~1 hr (note that in practice, we use turbidity data sampled using an automated water quality monitoring station at Δt = 15 min, allowing much finer timing, and therefore magnitude, resolution; see following sections). Picking all three sub-events would amount to triple-counting, and there is a tradeoff in net risk between magnitude and duration, so picking the longest or largest turbidity sub-event may not be appropriate. Rather, we represent turbidity impacts over this interval using the single sub-event having the largest associated *SEV* as calculated using (10) and (11). Note that the largest sub-event *SEV* may be less than 0.5, resulting in no net ecological risk (see previous table). The term "turbidity event" hereafter refers to the single turbidity sub-event thus selected.

3.2.3 Per-event real-time risk assessment

It would be very useful for watershed managers to have on hand a simple look-up table, tuned to the river of concern, which can immediately provide a robust measure of the risk associated with a turbidity event of a given magnitude and elapsed duration. Such a management tool would be particularly useful in conjunction with real-time telemetred data acquisition, which is growing increasingly common. This would permit assessment of the state of a river before an observed turbidity event is over or even before it has peaked, and thus facilitate prompt and proactive measures (such as a site visit or contacting stakeholders) if appropriate.

The look-up table we introduce here consists simply of a list of magnitudes, corresponding to a broad array of set durations, as calculated from (10) and (11) using a reference *SEV* value. The preferred method for setting the SEV_{ref} is to use a baseline turbidity dataset from the river under evaluation. A reasonable and simple approach is as follows. (i) Evaluate the *SEV* corresponding to each turbidity event over the historical record. (ii) Truncate the resulting set of *SEV* values to keep only those associated with non-nil risk ($SEV \geq 0.5$), and calculate the empirical cumulative distribution function for that subset. (iii) Use some percentile value of the resulting *SEV* distribution as a limit above which we consider risk to be unacceptable. The 90th percentile *SEV* is a reasonable choice. Further discussions regarding baseline data and reference values are provided in a subsequent section. (iv) The magnitude and duration of an observed turbidity event may then be compared against those listed on the look-up table; if the observed combination of magnitude and duration exceed those listed on the table, there is cause for concern with respect to that individual event. The advantages of this procedure are that it incorporates both magnitude and duration considerations; it adjusts the assessment for the baseline characteristics of study watershed; and it does so in a precise and fully explicit manner.

3.2.4 Cumulative risk

Published studies to date on quantitative management frameworks for assessment of ecological risks associated with lotic turbidity have not considered the cumulative impacts of multiple events. Cumulative risk may be a crucial factor in watershed health. For example, a large number of moderate-*SEV* events may ultimately have an equal or greater net ecological impact relative to one or two high-*SEV* events. We introduce here a method for assessing cumulative risk in a manner that explicitly incorporates magnitude and duration of individual events, the frequency of events, and local watershed characteristics, and collapses the resulting information into a single risk index.

3.2.4.1 Cumulative turbidity risk

We define the cumulative turbidity risk, *CTR*, as:

$$CTR = \sum_{i=1}^{N_{events}} RE_i \tag{12}$$

where N_{events} is the number of turbidity events occurring over the analysis interval, *i* indexes individual observed turbidity events, and the risk per event, RE_i, is given by:

$$RE_i = \begin{array}{l|l} SEV_i & SEV_i \geq 0.5 \\ 0 & otherwise \end{array} \tag{13}$$

so that only turbidity events with non-nil ecological impacts (see foregoing table) contribute to *CTR*. One could more conservatively set the cutoff *SEV* in (13) to 0 (or omit the cutoff altogether, but see section 3.2.1). We assume here, however, that if an individual event poses no net ecological risk, then a large number of such events also pose no risk. The analysis interval over which *CTR* is evaluated is technically arbitrary, but one year may often be an appropriate choice from a management perspective.

Unfortunately, relative to other water quality parameters often sampled using automated monitoring programs, turbidity data are prone to gaps that are difficult to reliably interpolate. Recognizing that events may have occurred during those gaps, and that the number and duration of gaps may vary substantially from one *CTR* calculation period to the next, apples-to-apples comparison of *CTR* values across analysis intervals would seem to require an adjustment for dataset size. We therefore define a linear adjustment factor as:

$$\eta = \frac{N_{full}}{N_{actual}} \tag{14}$$

and modify *CTR* accordingly:

$$^{adj}CTR = \eta \; CTR \tag{15}$$

where N_{full} is the number of data that would have been acquired over the full analysis interval at sampling interval, Δt, if no gaps had occurred; and N_{actual} is the actual number of data sampled over that interval. For example, N_{full} = 365 for an analysis interval of one year and daily sampling, so if the number of samples actually acquired was only 274, then $\eta \sim 1.33$. Thus, observed *CTR* is upscaled to accommodate the fact that a quarter-year of data, and thus a quarter-year of potential turbidity events, were missed. The underlying assumption is that *SEV* is statistically stationary over the analysis interval, so that risk over the unsampled part of the interval can be adequately represented by results from the sampled portion. For an annual analysis interval, this assumption is best satisfied when data gaps are distributed throughout the year, avoiding potential seasonal effects. For shorter (e.g., seasonal) analysis intervals, the effects of cyclostationarity are very likely negligible. Note that the adjustment procedure is primarily intended for turbidity time series collected using automated, high-frequency water quality sampling programs, and does not seem readily applicable to manual, infrequent, and/or irregular sampling.

3.2.4.2 Risk quotient

We now introduce a risk quotient, closely analogous to that defined with respect to chronic temperature risks in the preceding chapter of this report and used extensively in toxicological risk assessment:

$$RQ = \frac{CTR_{obs}}{CTR_{ref}} \tag{16}$$

where CTR_{obs} is the observed *CTR* for a given assessment interval, and CTR_{ref} is a reference *CTR* value, preferably evaluated from baseline data for the study river. The *CTR* values may be adjusted or unadjusted for data gaps (see preceding section). Further discussions regarding baseline data and reference values are provided in the subsequent section. This *RQ* definition again leads to a formal, simple, and robust decision rule for risk assessment and watershed management:

$$\begin{aligned} &RQ > 1 : \textit{unacceptable risk} \\ &RQ \leq 1 : \textit{acceptable risk} \end{aligned} \tag{17}$$

Observed $RQ > 1$ thus indicates cumulative turbidity risk in excess of typical / acceptable levels. As in the previous chapter, $RQ = 0$ indicates no risk at all over that analysis interval (although this is generally unlikely for turbidity), and $0 < RQ \leq 1$ indicates that such risk was incurred but was within acceptable (e.g., natural historical) limits. Only if $RQ > 1$ is there cause for management concern. $RQ = 1$ thus constitutes the risk-based water quality objective.

The risk quotient meets our goals of (i) explicitly incorporating information regarding the magnitude and duration of individual events (embedded in individual *SEV* values) and the frequency of events (embedded in the summation), (ii) explicitly adjusting the acceptable level of risk to baseline conditions for the individual watershed (embedded in the normalization by a historically derived CTR_{ref}), and (iii) expressing the result as a single, convenient index of risk, which serves as a clear flag for the presence of significant ecologically negative changes in stream turbidity conditions. It should also be noted that, as was the case for the methods introduced in the previous chapter, use of the turbidity *RQ* is not limited to retrospective analyses. One could recalculate (16) on a real-time and potentially automated basis, setting the analysis interval to the year-to-date, in order to obtain an evolving measure of current accumulated turbidity risk in the watershed.

Interpretation of high turbidity *RQ* is subject to considerations similar to those listed for chronic temperature *RQ*. Changes in turbidity risk include the effects of watershed modifications, such as those associated with urban development, logging, mining, road construction, and other activities and land use changes potentially capable of increasing erosion and turbidity levels. However, although this method explicitly adjusts for naturally high turbidity risks when CTR_{ref} is based upon a historical record, observed $RQ > 1$ could conceivably arise from anomalously large natural and/or external effects, such as natural mass wasting or climatic variability. While the corresponding *RQ* would remain valid, external constraints and professional judgment are therefore still required to correctly attribute a high *RQ* to its source(s). Similarly, turbidity *RQ* might also be used as a metric to investigate the potential impacts of large-scale ocean-atmosphere circulation patterns and long-term climatic change upon water quality, relative to historical conditions.

3.2.5 Establishing reference values

3.2.5.1 Reference SEV *and* CTR

Lower percentiles of an *SEV* distribution derived from historical data for the study watershed, and therefore lower SEV_{ref}, lead to a more conservative risk assessment, where we take a conservative (liberal) assessment to be one which is more (less) protective of the environment. There is some subjectivity in selecting the reference *SEV*. In our applications thus far, we have found SEV_{ref} to be only moderately sensitive to choice of critical percentile, provided that a reasonable number (likely between 85th and 95th percentile) is used; the 90th percentile value of the *SEV* distribution seems a good compromise. It is generally wise to assess the *SEV* distribution from each river considered, however, for local SEV_{ref} sensitivity to percentile choice.

Options for CTR_{ref} selection are closely analogous to those described for TGR_{ref} (preceding chapter). Note that lower CTR_{ref} yields higher *RQ* for a given CTR_{obs} and is therefore more protective of aquatic habitat. For an annual analysis interval, options include the mean or median value of annual CTR_{obs} over the baseline period, the upper 68% or 95% confidence level on that historical mean, one or two standard deviations above the historical mean, an interquartile range above the historical median, or some percentile of historical yearly values. A short baseline dataset might not adequately capture natural interannual turbidity variability, and the foregoing statistical summary measures could therefore be poorly constrained; in such a case, setting $CTR_{ref} = \max(CTR_{obs})$ may be sensible. Analogous CTR_{ref} definitions can be identified for seasonal analysis intervals, if preferred. If $SEV < 0.5$ for every event on record, then (7) is mathematically undefined for the above CTR_{ref} definitions, but may be evaluated by setting CTR_{ref} to an arbitrarily small number. However, $CTR_{obs} = 0$ is very unlikely for many or most natural streams, particularly over a yearly analysis interval, although it may be more likely in dammed rivers where downstream sediment supply is artificially limited.

Ultimately, identification of a single, fully universal method for setting SEV_{ref} and CTR_{ref} may not be appropriate. Rather, reference value selection might best be performed on a watershed-by-watershed basis. Key considerations include data availability; magnitude of interannual turbidity fluctuation, which reflects in part the regional hydroclimatic regime, and helps determine requisite baseline dataset size; degree of management concern; and the levels of conservatism of individual watershed stakeholders.

3.2.5.2 Baseline data alternatives

Alternatives to a substantial historical dataset from the study river under natural conditions are more limited for turbidity than for temperature risk assessment (see previous chapter). If turbidity records are available but span a period of watershed modification, it may still be feasible to use such data to set historically based *SEV* and *CTR* reference values, as was the case for temperature risk. The requisite condition is again that either the inferred turbidity impact of previous development is negligible; or that the practical management goal is to ensure that future watershed modification does not degrade lotic habitat quality beyond current, albeit potentially already impacted, levels.

If no usable baseline information exists, there is some potential to set reference values on the basis of other considerations, but guidelines for doing so are much weaker than was the case for chronic temperature assessments. For example, one might consider only minor turbidity impacts to be acceptable, giving $SEV_{ref} \sim 3.49$. However, this approach requires an arbitrary risk choice and, in particular, does not adjust the final risk index for natural local conditions. Moreover, this approach is not applicable to CTR_{ref}, because we would also need to define an acceptable frequency for the study river, and there appears to be no general guideline for doing so in the absence of historical site-specific data.

Turbidity can be extremely sensitive to individual watershed characteristics, such as catchment topography and geology. Particularly in a small watershed, the presence or absence of single large clay cut bank could profoundly alter the natural turbidity regime. Consequently, it is unlikely that records from adjacent streams could be used as a surrogate if insufficient baseline data is available for the study watershed. This situation may be quite different from that for assessing water temperature impacts.

3.3 Application to Millionaire Creek

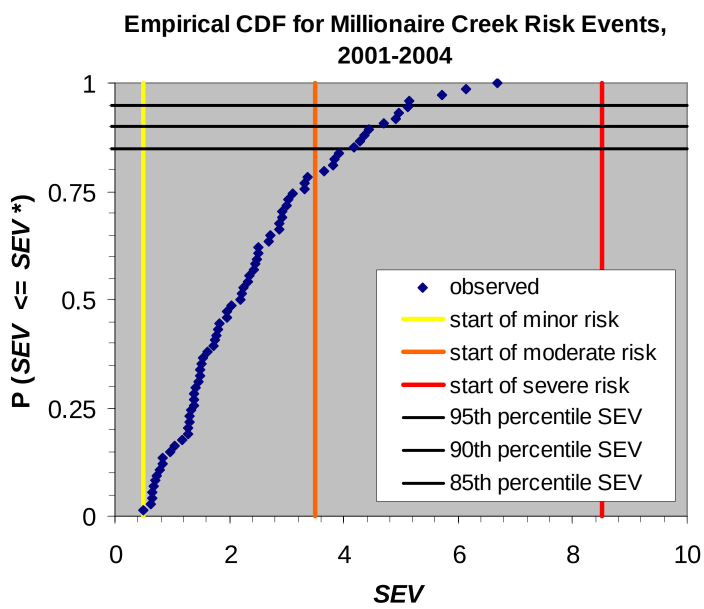
3.3.1 Data

Millionaire Creek turbidity data were available over 2001-2004 at a sampling interval of Δt = 15 min. Low-pass filtering and data validation were completed prior to analysis (see Appendix). Missing data were not interpolated. Data were parsed into calendar years, yielding partial time series for 2001 and 2004 and full time series (but with gaps) for 2002 and 2003. Events were picked according to the general procedure outlined previously in this chapter. Specifically, for time intervals over which $NTU > 1$, individual sub-events were defined by 1 *NTU* intervals, and corresponding durations were found. Of the resulting suite of sub-events, that with the largest *SEV* was selected to represent the interval. If a sub-event ended with a data gap, it was omitted from the procedure. Seasonality in the turbidity time series was found to be surprisingly low. Although turbidity events tended to be larger, longer, and more frequent during the winter rainy season, it was found that substantial events could occur at any time of year.

The full period of available Millionaire Creek turbidity data was taken to be a baseline period. For a detailed discussion of the rationale and watershed management implications of this choice, please refer to the preceding chapter.

3.3.2 Per-event real-time risk assessment

A total of 214 turbidity events were observed over the baseline period. Of these, 74 exhibited $SEV \geq 0.5$ (minor or greater risk). The empirical cumulative distribution function for the 74 non-nil risk events is shown below:



The 90th percentile value of *SEV* was ~4.6 (corresponding to a moderate risk), and taken to be SEV_{ref}. Using (10) and (11) to calculate magnitude values corresponding to this SEV_{ref} and a broad range of prescribed durations yields a risk look-up table for Millionaire Creek (continued on next page):

Maximum acceptable risk-per-event, Millionaire Creek

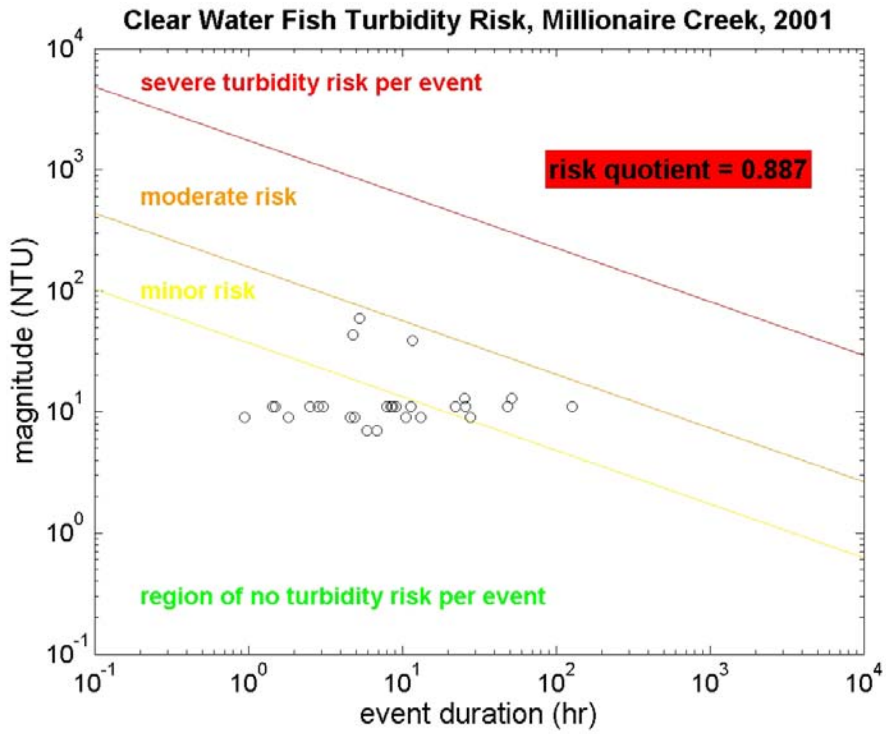
duration			magnitude
hr	**days**	**weeks**	**(*NTU*)**
0.25			497
0.5			365
1			269
2			198
4			145
8			107
12			89
24	1		66
48	2		48
96	4		36

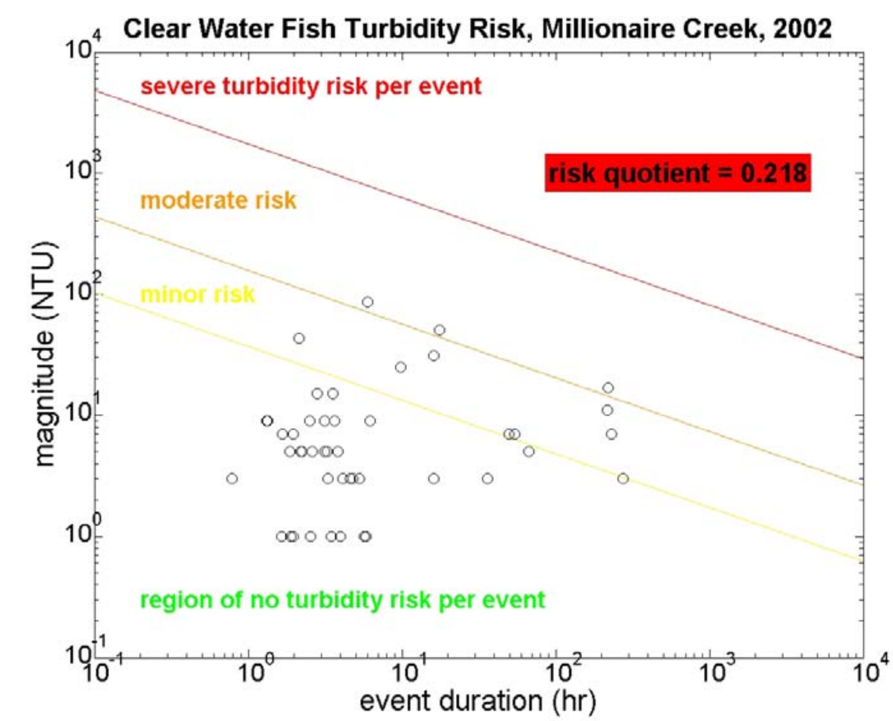
120	5		32
168	7	1	28
336	14	2	20
672	28	4	15
1344	56	8	11
2688	112	16	8.1
5376	224	32	6.0
10752	448	64	4.4

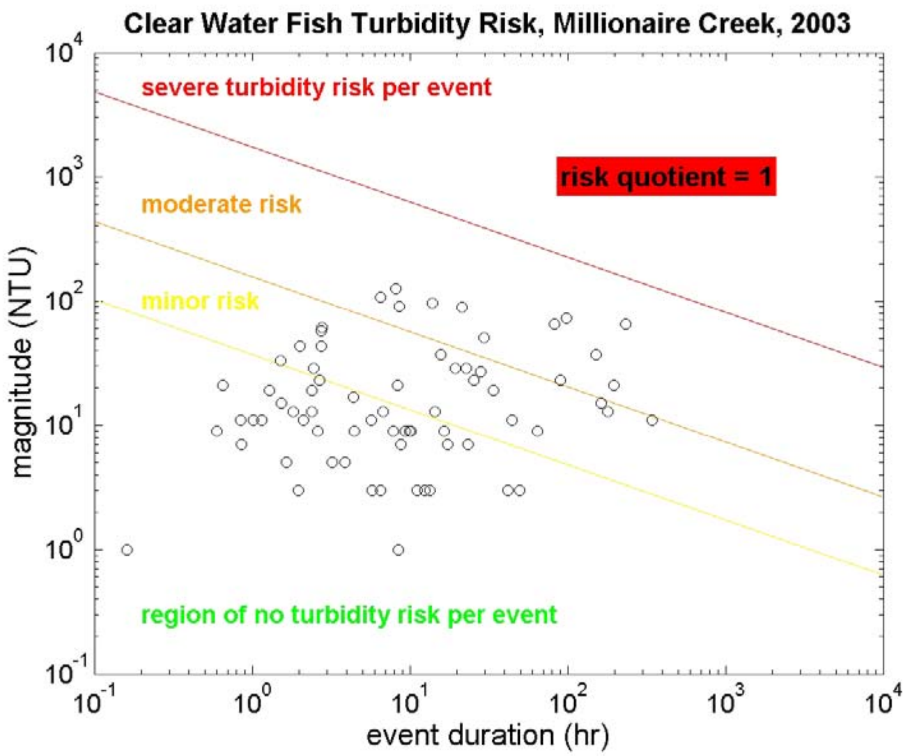
Millionaire Creek turbidity events exhibiting a combination of duration and magnitude which exceed the combinations listed on the table have an *SEV* larger than SEV_{ref}. Such events, by construction, are associated with an unacceptable ecological risk. For example, an event consisting of 23 *NTU* for three weeks would exceed the historically based risk tolerance level and be cause for management concern. Conversely, an event consisting of 200 *NTU* for 45 minutes, although larger, would not be of sufficient duration to exceed the risk tolerance level.

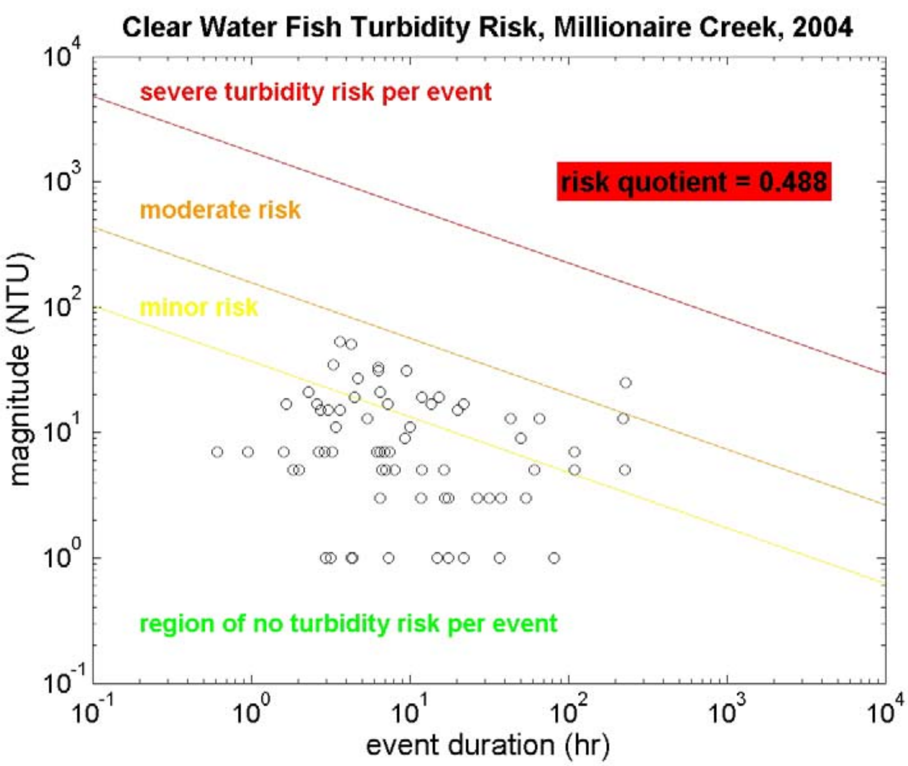
3.3.3 Cumulative risk

Cumulative risk was considered for a yearly analysis interval. Due to the relatively short historical record, CTR_{ref} was set to the maximum value observed over the baseline period (see previous chapter and section 3.2.5, this chapter), which is ~110. A linear adjustment for data gaps was applied separately for each year. Results are shown below.


Clear Water Fish Turbidity Risk, Millionaire Creek, 2001
severe turbidity risk per event
risk quotient = 0.887
moderate risk
minor risk
region of no turbidity risk per event
magnitude (NTU)
event duration (hr)


Clear Water Fish Turbidity Risk, Millionaire Creek, 2002
severe turbidity risk per event
risk quotient = 0.218
moderate risk
minor risk
region of no turbidity risk per event
magnitude (NTU)
event duration (hr)


Clear Water Fish Turbidity Risk, Millionaire Creek, 2003
severe turbidity risk per event
risk quotient = 1
moderate risk
minor risk
region of no turbidity risk per event
magnitude (NTU)
event duration (hr)


Clear Water Fish Turbidity Risk, Millionaire Creek, 2004
severe turbidity risk per event
risk quotient = 0.488
moderate risk
minor risk
region of no turbidity risk per event
magnitude (NTU)
event duration (hr)

Maximum observed *CTR* occurred in 2003; given the CTR_{ref} definition employed, by construction *RQ* = 1 for that year. Annual mean monthly total precipitation from YVR (see previous chapter) and the observed *RQ* values are illustrated in the following table:

***RQ* and Vancouver International Airport precipitation data**

year	mean precipitation (mm)	*RQ*
2001	98.2	0.887
2002	71.5	0.218
2003	92.2	1.00
2004	65.9	0.488

There is a generally good correspondence between precipitation and risk quotient. Although the sample size is small and the associated *P*-value is therefore a very modest 0.14, the correlation coefficient between the two quantities is nevertheless a substantial 0.86, explaining about 74% of the variance in *RQ*. Thus, precipitation appears to be the primary driver of interannual variability in turbidity risk for Millionaire Creek, which is fully consistent with general understanding of natural temporal variability in turbidity levels.

Interestingly, 2004 *RQ* is the second-lowest of the four years, suggesting that renewed development activity in the watershed that year (see preceding chapter) did not pose a significant turbidity risk. Any future annual *RQ* > 1 would indicate a deterioration of watershed health from its 2001-2004 state and be cause for management concern.

4. Conclusions and Recommendations

We propose risk assessment methodologies for chronic temperature and turbidity impacts which satisfy the nine requirements outlined in the Overview chapter of this report, and yield site-specific, risk-based water quality criteria. The methodologies are based upon integration of broad risk assessment concepts with the current body of fisheries science knowledge, explicitly incorporate both magnitude-duration relationships and watershed-to-watershed variability in background conditions, and are generally feasible for wide implementation as standard watershed assessment and management tools. The primary product of both protocols is a risk quotient, *RQ*, which yields a straightforward decision rule for watershed managers concerned about the potential water quality impacts of ongoing or future activities in the catchment. In addition, a method for developing catchment-specific look-up tables to establish ecologically acceptable turbidity conditions, and whether an individual turbidity event exceeds those conditions, was introduced.

All the methods were applied to Millionaire Creek, using 2002-2004 (temperature) or 2001-2004 (turbidity) water quality data as a baseline, so that acceptable/not acceptable risk conditions are referenced to deterioration of watershed conditions beyond current, non-impacted to moderately impacted, levels. From a management perspective, this choice of baseline focuses, in effect, upon holding those performing future activities in the watershed accountable for the contributions they may or may not make to water quality degradation in Millionaire Creek. Specific products implemented for Millionaire Creek include a risk assessment protocol for chronic temperature impacts, such that any future $RQ > 1$ indicates unacceptable risk to Millionaire Creek salmonids and cause for management concern and, potentially, further action with respect to that water quality parameter; a risk assessment protocol for cumulative turbidity impacts, such that any future $RQ > 1$ similarly indicates unacceptable risk to Millionaire Creek clear water fish and cause for management concern and, potentially, further action with respect to that water quality parameter; and a look-up table permitting watershed managers to assess, in real- or near-real-time, whether an individual turbidity episode constitutes a water quality condition posing unacceptable impacts upon clear water fish in Millionaire Creek. All three constitute risk-based, site-specific water quality objectives, and are in place for immediate use, with the caveat that the methodologies of course remain experimental.

The knowledge integration upon which the overall risk assessment protocols are based render them somewhat modular in nature. That is, certain technical elements, drawn from existing fisheries science knowledge and incorporated into the risk assessment and management procedures, can be updated or replaced as necessary or desirable while maintaining the protocols as a whole. We believe that the overall protocols, along with specific technical elements we have incorporated into them, as presented in this report constitute effective tools for standardized and practical watershed risk assessment and management for chronic turbidity and temperature impacts, and are generally superior to the procedures currently in place for most rivers. Nevertheless, there remains substantial room for improvement in, and adjustment or replacement of, individual technical elements. Some potential directions for future work include developing specific growth relationships over the relevant interval, $g(T > T_o)$, for additional salmonid (and potentially other) fish species; closer examination of the net risk assessment and management implications of C/C_{max} ratios, how to incorporate food availability information into the assessment without inducing an infeasible requirement for site-by-site field studies, and the practical viability of generalized assumptions in this regard; and broadening the turbidity assessment to include both optical and non-optical impacts.

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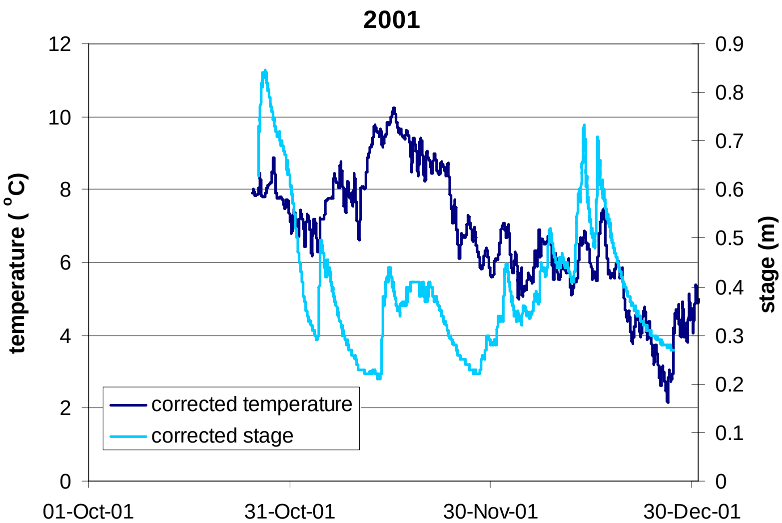
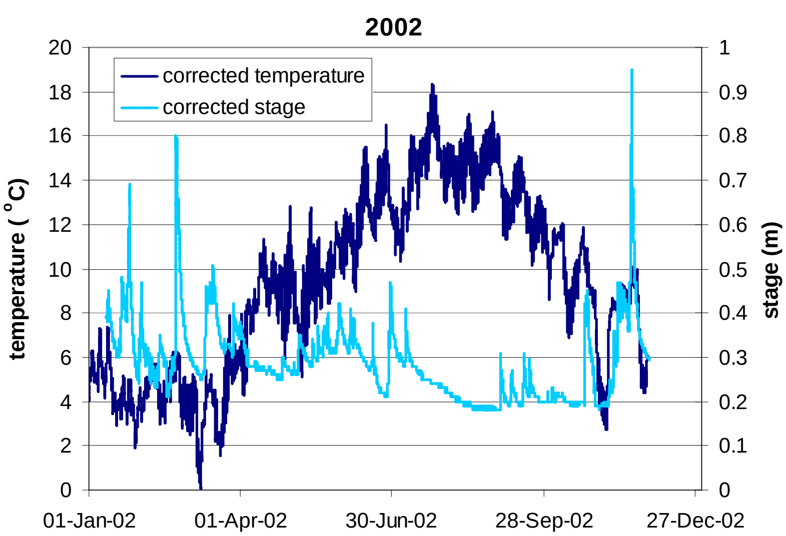
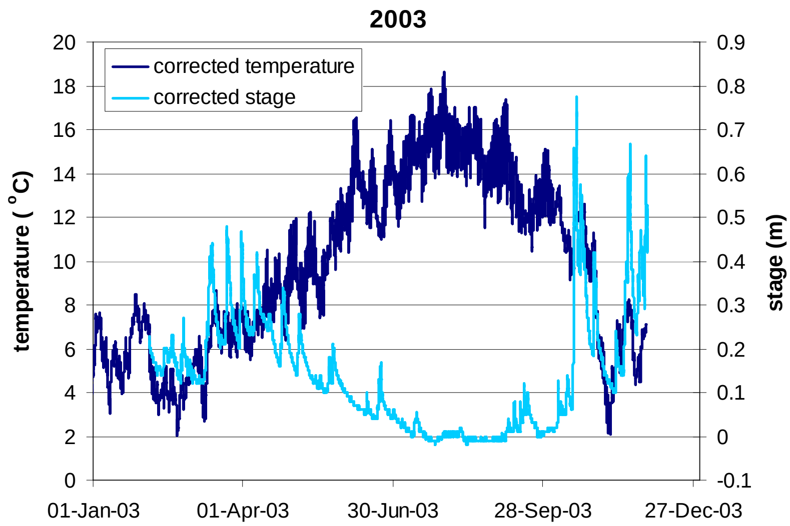
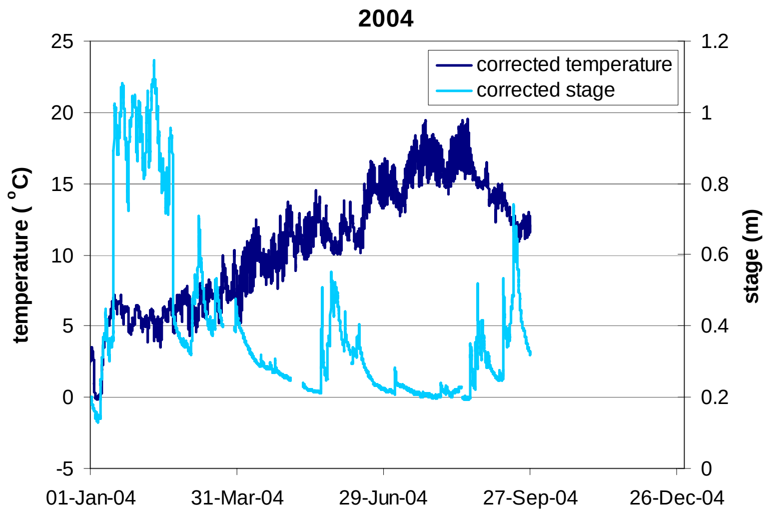
Appendix: Data Validation and Correction

Available Millionaire Creek water temperature, water level, dissolved oxygen, pH, specific conductivity, and turbidity data were validated and corrected as per Quilty et al. (2004b). The time series span October 24, 2001 through September 27, 2004. The 2001-2003 data have previously been processed (Quilty et al., 2004b), so effectively the emphasis here was upon further checks and improvements upon the quality of existing quality-controlled records. Overall, changes from Quilty et al., 2004b are minor. Validated, corrected, and when possible, gap-filled water temperature and stage data for each year are shown below.

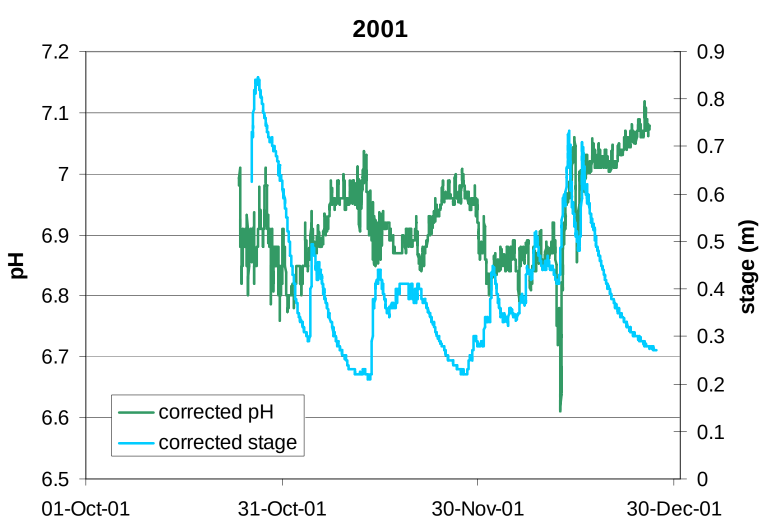


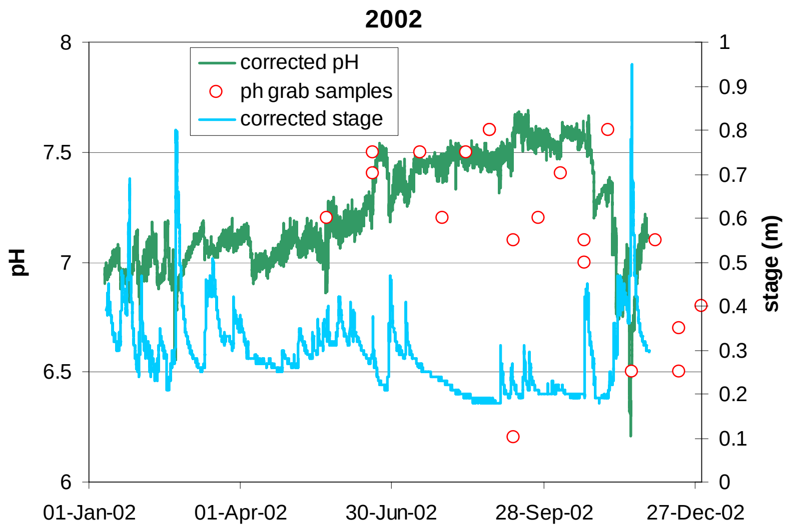


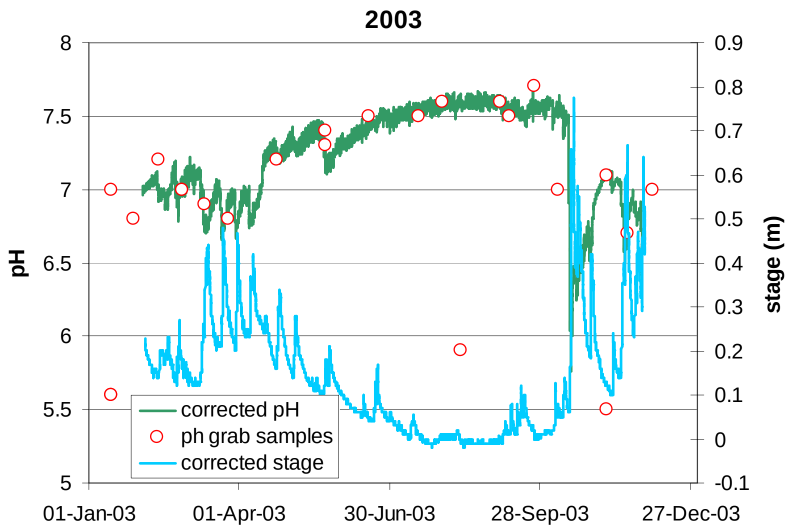


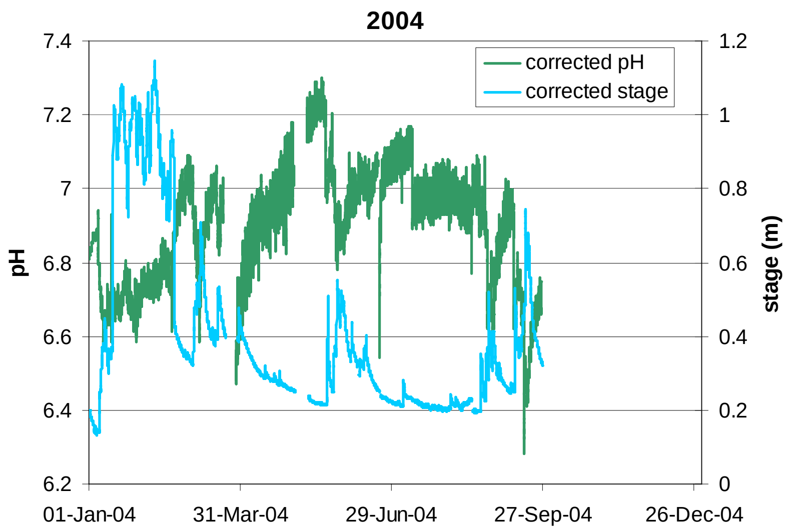


Corrected and quality-controlled pH data for each year are illustrated below, again with river stage:

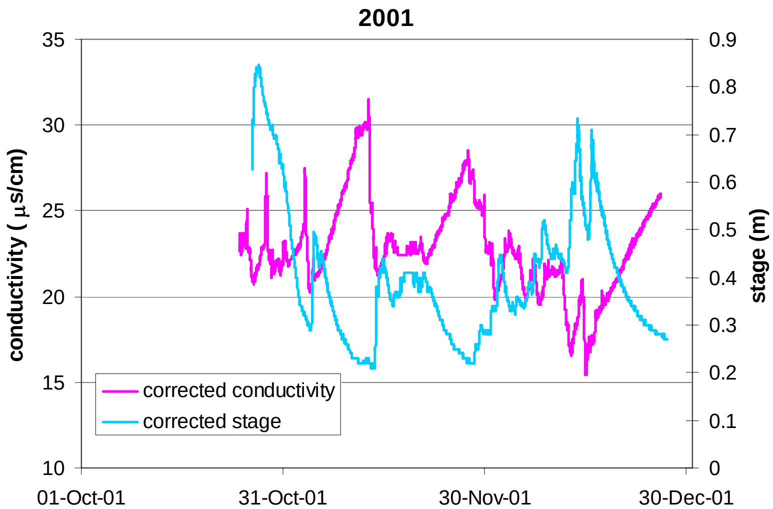


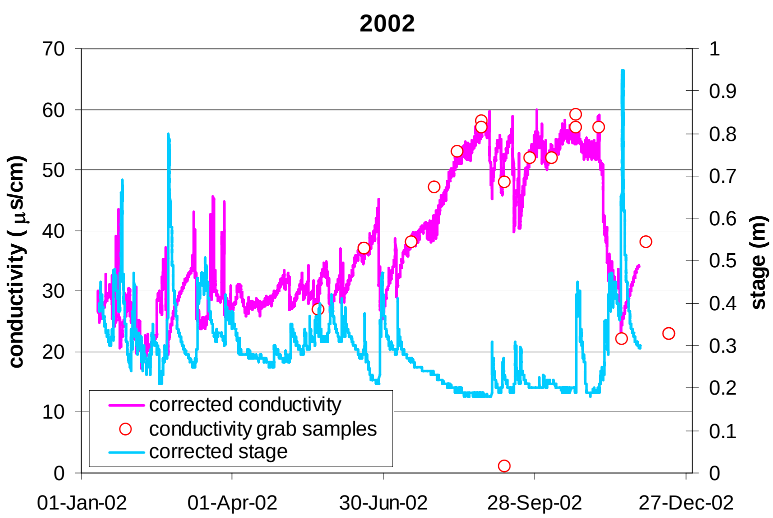

2002
corrected pH
ph grab samples
corrected stage
pH
stage (m)
8
7.5
7
6.5
6
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
01-Jan-02
01-Apr-02
30-Jun-02
28-Sep-02
27-Dec-02

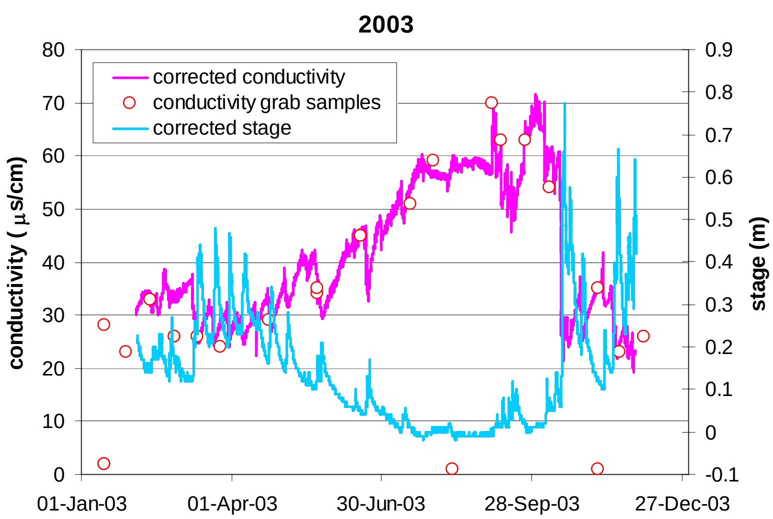

2003
corrected pH
ph grab samples
corrected stage
pH
stage (m)
8
7.5
7
6.5
6
5.5
5
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
-0.1
01-Jan-03
01-Apr-03
30-Jun-03
28-Sep-03
27-Dec-03

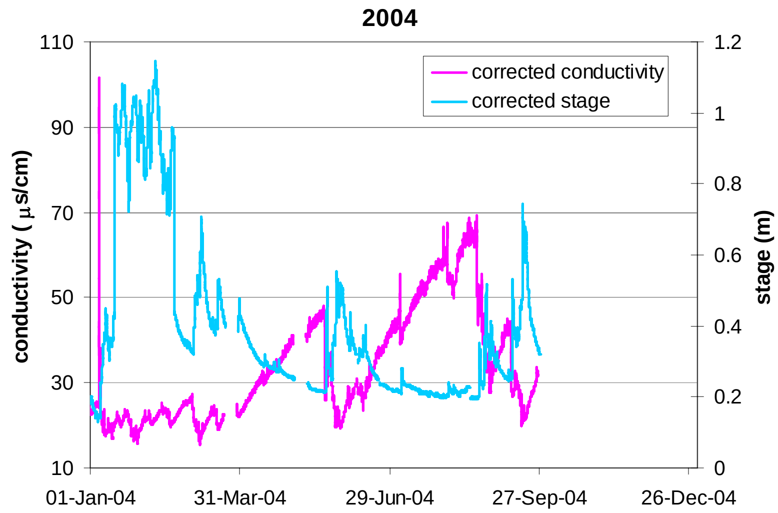

2004
corrected pH
corrected stage
pH
stage (m)
7.4
7.2
7
6.8
6.6
6.4
6.2
1.2
1
0.8
0.6
0.4
0.2
0
01-Jan-04
31-Mar-04
29-Jun-04
27-Sep-04
26-Dec-04

Validated and corrected conductivity data are shown below, together with corrected stage data:

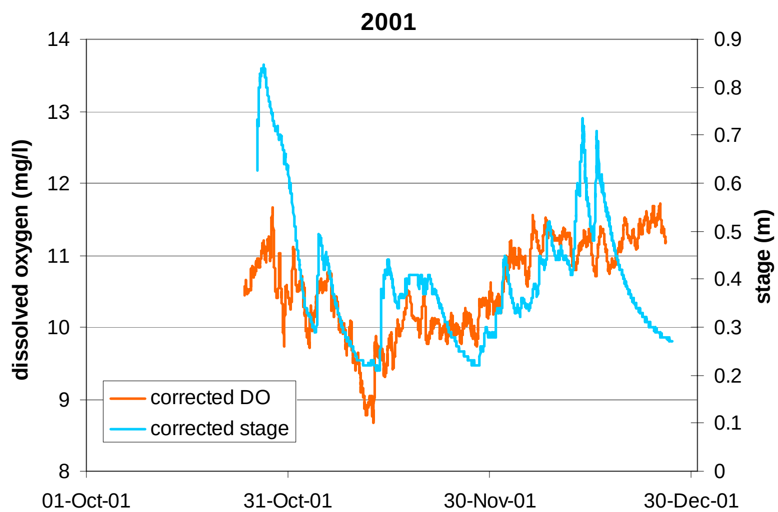


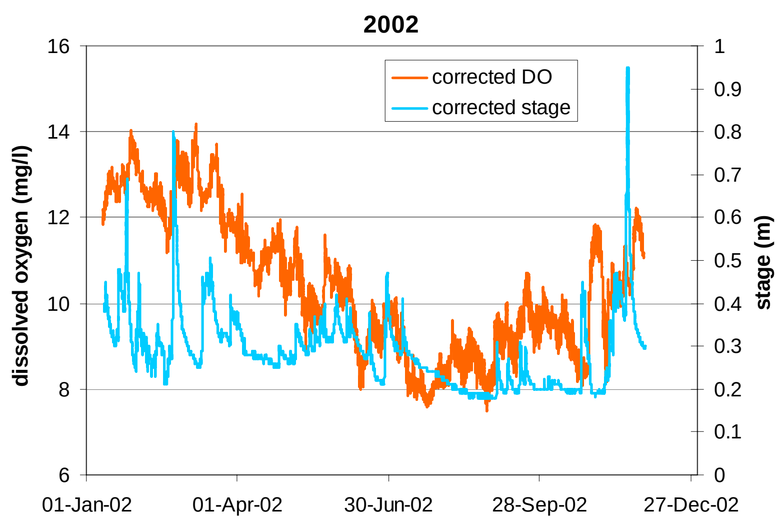


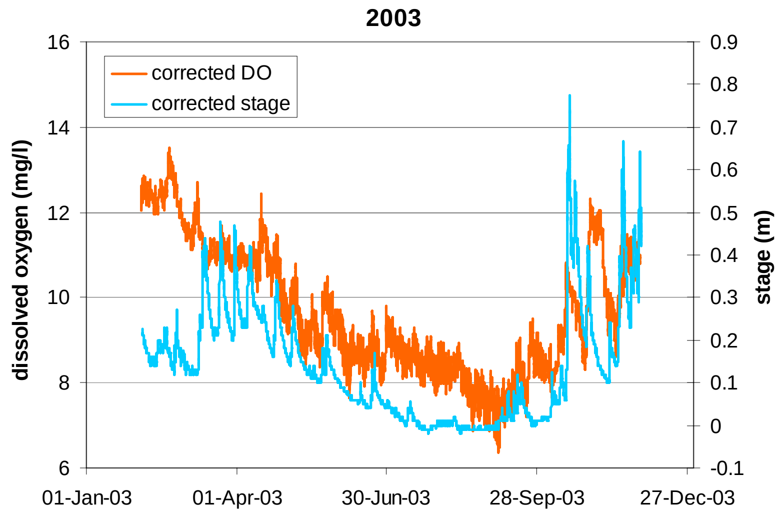


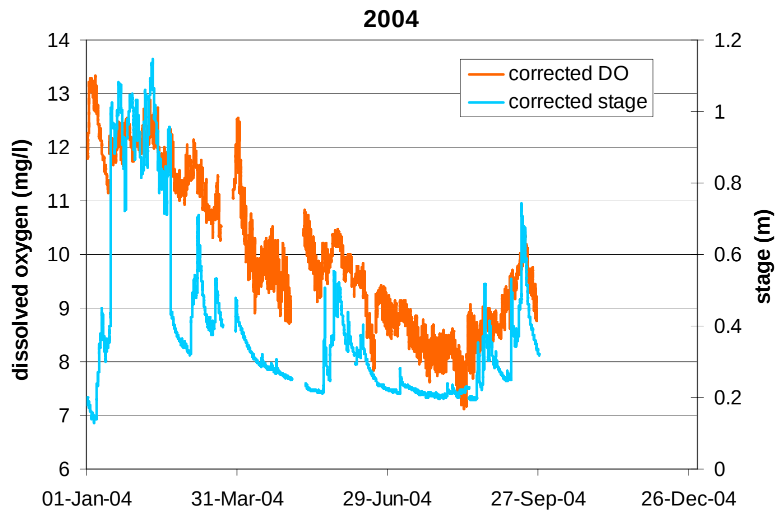


Validated and corrected dissolved oxygen and stream level data for 2001-2004 are as follows:

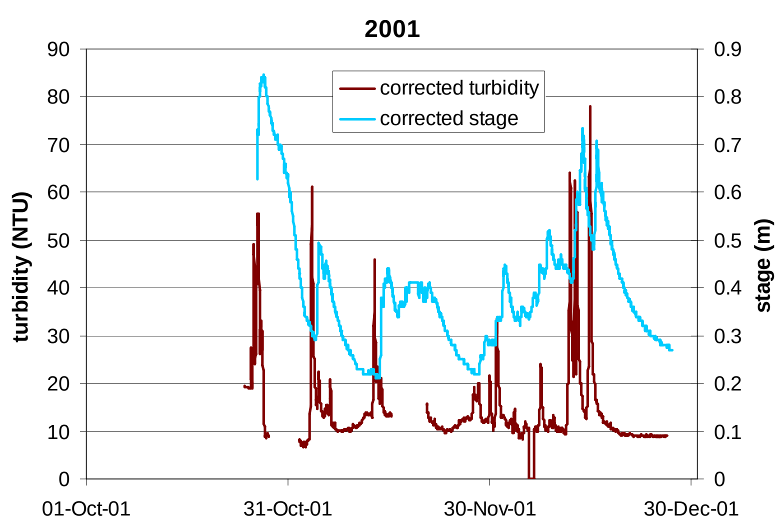


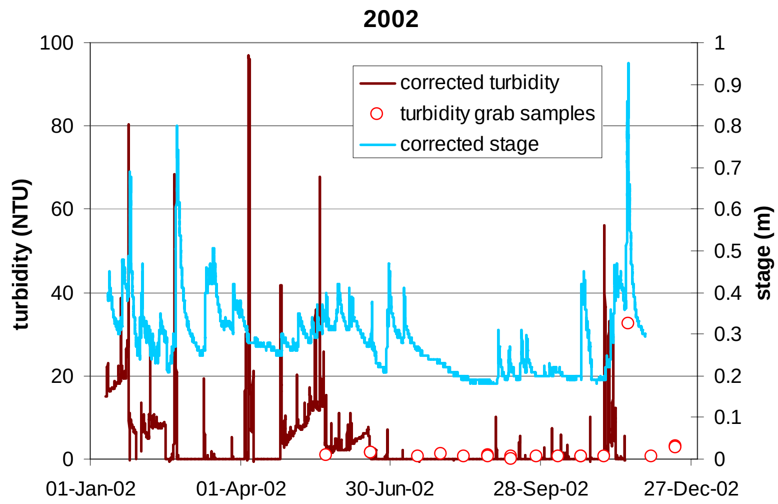


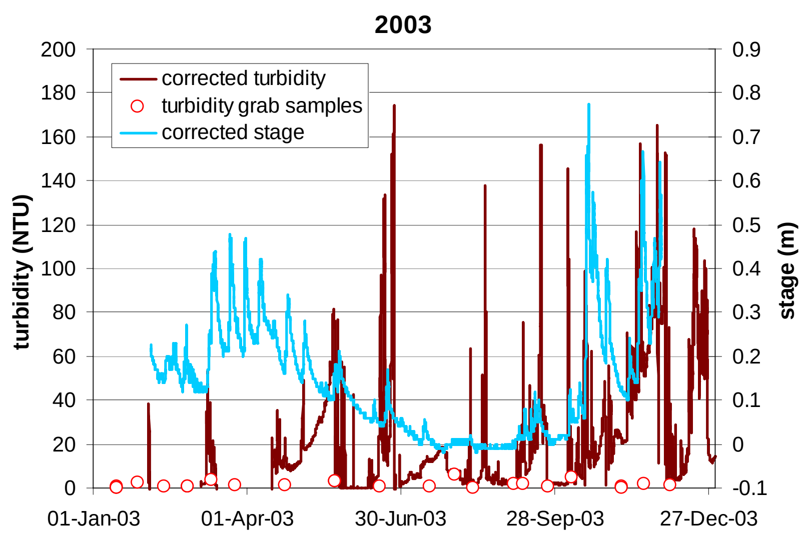


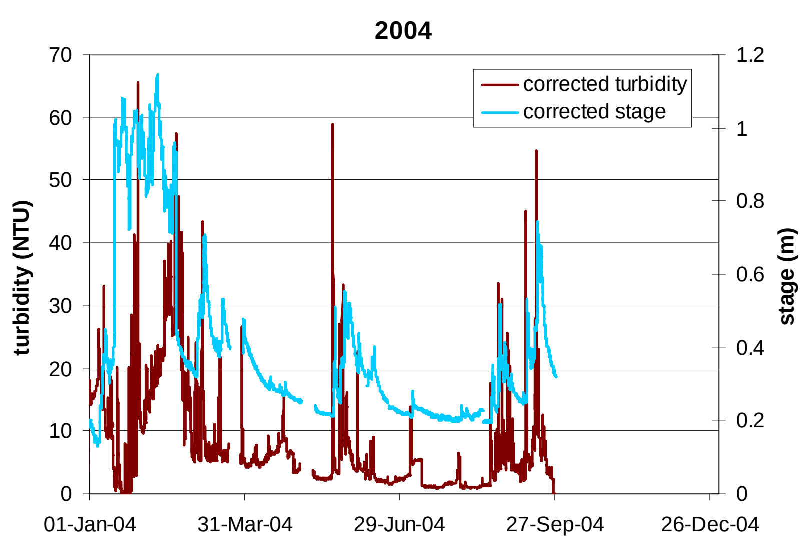


Filtered and validated turbidity time series are shown below. No attempt was made to gap-fill turbidity data (see Quilty et al., 2004b). Corrected, validated, and gap-filled stage data are again shown for comparison:




2002
turbidity (NTU)
stage (m)
corrected turbidity
turbidity grab samples
corrected stage
100
80
60
40
20
0
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
01-Jan-02
01-Apr-02
30-Jun-02
28-Sep-02
27-Dec-02


2003
turbidity (NTU)
stage (m)
corrected turbidity
turbidity grab samples
corrected stage
200
180
160
140
120
100
80
60
40
20
0
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
-0.1
01-Jan-03
01-Apr-03
30-Jun-03
28-Sep-03
27-Dec-03


2004
turbidity (NTU)
stage (m)
corrected turbidity
corrected stage
70
60
50
40
30
20
10
0
1.2
1
0.8
0.6
0.4
0.2
0
01-Jan-04
31-Mar-04
29-Jun-04
27-Sep-04
26-Dec-04