Salmonid distribution in relation to stream temperatures in Fortune Creek, British Columbia – The influence of surface water and groundwater interactions

by

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Abstract

There is concern over the impacts of increased water temperature on salmonid populations in British Columbia's (BC) Southern Interior. Groundwater influx may moderate stream temperatures and provides thermal refuge for salmonids. The objective of this study was to establish quantitative linkages among groundwater, water temperatures and juvenile coho salmon (*Oncorhynchus kisutch* Walbaum), Chinook salmon (*Oncorhynchus tshawytscha* Walbaum) and rainbow trout (*Oncorhynchus mykiss* Walbaum) in Fortune Creek, a small regulated stream in BC's Southern Interior. In addition, this study quantified whether groundwater influx could provide thermal relief to salmonids during periods of heat stress.

Salmonid enumeration and field data collection were carried out between July 2008 and June 2009. Generalized Linear Mixed Models (Poisson and logistic regression) were used to establish linkages between juvenile salmonids and physical and chemical fish habitat indicators. Thermal modeling of vertical groundwater fluxes in the streambed was conducted using the software VS2DH, which simulates fluid and conductive-convective heat transport. The relative influence of groundwater and stream discharge on stream temperatures was assessed using a conceptual energy balance approach.

Statistical modeling confirmed that the distribution of juvenile salmonids was consistently related to water temperatures. Salmonids avoided reaches where maximum daily water temperatures exceeded 22°C, which was exceeded at most of the sites studied. Relationships to water chemistry were less consistent and existed almost exclusively in combination with water temperature variables.

While groundwater flux to the stream was widespread, estimated upward fluxes were low (10^{-7} m/s) and were primarily constrained by low streambed hydraulic conductivities. The influence of advective cooling on stream temperatures in unshaded reaches was too small to provide significant thermal relief for salmonids. However, groundwater influx provided up to 88% of baseflows during the summer low flow season and was critical in maintaining flows in the creek.

Conceptual energy balance modeling confirmed that low flows in Fortune Creek limit juvenile salmonid rearing by causing water temperatures to escalate. Recommendations for maintaining suitable water temperatures for juvenile salmonids include re-vegetation of the banks to provide shade and strategically supplementing flows to increase discharge at times of high water temperature potential.

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1.0 INTRODUCTION

Streams throughout the semi-arid Southern Interior of British Columbia (BC) are particularly susceptible to high water temperatures due to a warm summer climate, naturally low summer flows, and prolonged drought periods in the summer (Walthers & Nener, 2000). Extensive agricultural activity in the valleys and an increasing population base result in high water demand on streams, that frequently leads to low streamflow volumes.

Water demand is high for irrigation of agricultural fields and orchards, watering of livestock, as well as for industrial and domestic purposes. Water users impact streams through direct surface water withdrawal and potentially also by groundwater pumping, which can lead to a lowering of the water table and a subsequent reduction in stream baseflows.

Summer water temperatures in streams throughout the Southern Interior of BC frequently approach the upper thermal limits of the salmonid species inhabiting them (Mathews et al., 2007). There is concern for the sustained existence of some salmonid populations in this region and high water temperatures are thought to be a limiting factor (Walthers & Nener, 2000). In light of this issue, various management strategies have been considered to ensure summer stream temperatures remain within ranges that can be tolerated by salmonids. A common strategy is the re-vegetation of stream banks to increase shade and reduce the solar load reaching the stream surface. Another is the release of water from upstream reservoirs, which has been demonstrated to lead to downstream cooling (Gu et al., 1998). However, this option can be difficult to implement. Water demand is high and water licenses on many interior streams are fully allocated (BC Ministry of Water, Land and Air Protection, 2002). Additional water during the low flow season comes from storage in reservoirs, which were constructed to provide water for human use. Consequently, using stored water for conservation purposes can conflict with human water uses where little water is available.

Most of the water flowing in streams originates from groundwater during baseflow conditions. Groundwater intrusion through the stream bed provides areas of stable temperature regime in summer and winter, ensures stream baseflows, provides ice-free habitat in winter, and provides nutrient input to streams (Power et al., 1999). Consequently, groundwater plays several important roles in the maintenance of fish populations in many streams.

The role of groundwater seeps in moderating stream temperatures and providing thermal refuge for temperature stressed salmonids has been discussed in the context of salmonids persisting in streams

considered unsuitably warm (Nielsen et al., 1994; Ebersole et al., 2001; Baird & Krueger, 2003; Tobias, 2006). In the BC Interior, groundwater and surface water interactions and their impact on fish habitat have been identified as an important knowledge gap for understanding how the endangered Interior Fraser Coho salmon use their available habitat and what may be the cause of decline in their abundance (Interior Fraser Coho Recovery Team, 2006).

The Pacific Salmon Foundation has expressed concern over the persistence of salmonids in Fortune Creek, a small stream in BC's Southern Interior. Fortune Creek is a regulated system that serves as a water supply for approximately 4,500 residences and various irrigation users from the City of Armstrong and Township of Spallumcheen in the North Okanagan. The creek provides important habitat for resident rainbow trout (*Oncorhynchus mykiss*), juvenile coho (*Oncorhynchus kisutch*) and Chinook salmon (*Oncorhynchus tshawytscha*). Low flows, high water temperatures, and declines in salmonid numbers have been documented in the creek, and water management in the watershed has become increasingly difficult (Seebacher et al., 2007). A scoping study has identified surface water and groundwater interactions as the highest priority issue to be examined in the creek, followed by high water temperatures and low streamflows (Seebacher et al., 2007). These conditions make Fortune Creek suitable for studying the influence of groundwater and surface water interactions on fish habitat and the potential for groundwater to provide thermal relief to salmonids.

1.1 Objectives

The ultimate goal of this research was to establish quantitative linkages among groundwater, water temperature and fish populations in Fortune Creek. To do so, the following research questions and hypotheses were investigated:

1. What are the primary limitations to salmonid rearing in Fortune Creek?
2. What is the relationship between stream water temperatures and the distribution of salmonids in small Interior streams? Null hypothesis: Stream temperatures are not significant predictors of salmonid distribution. Alternative hypothesis: Stream temperatures are significant predictors of salmonid distribution.
3. Is there a potential for upwelling groundwater to provide thermal refuge for juvenile salmonids?
4. What is the potential impact of nearby groundwater extraction or declining groundwater levels on groundwater inflows to the stream?

First, the distribution of salmonids in relation to stream temperature and other physical and chemical fish habitat indicators was analyzed to assess whether water temperature limits salmonid use of Fortune Creek. Then, the direction and magnitude of groundwater – stream water exchanges were explored along the upstream-downstream gradient. The advective cooling effect of groundwater discharge on stream temperatures was quantified.

1.2 Thesis Format

This thesis is organized in the following manner. Chapter 2 provides a review of the literature relevant to the salmonid life cycle, previous studies of salmonid responses to stream temperature, a review of statistical methods relevant to this study, controls on stream temperatures and methods of determining groundwater flow into streams. A background on the general study area and detailed descriptions of the study sites are provided in Chapter 3. Further, the methodology for field data collection and analysis are described. Chapter 4 presents the results of the field data collection and the analyses. This begins with an overview of the field data collected in the creek during 2008, followed by a statistical analysis of salmonids and stream temperature. The effects of groundwater and stream flow management on stream temperature are also assessed. Chapter 5 discusses how these results address the research questions outlined above, and includes recommended strategies for stream management. The main conclusions of this research are summarized in Chapter 6 along with suggestions for future research.

2.0 BACKGROUND

2.1 Stream Temperature

Numerous mechanisms are involved in energy exchange in a stream. These include solar radiation, evaporation/condensation (latent heat), conduction with the stream bed, conductive exchange of heat at the air/water interface (sensible heat), groundwater inflows (advection) (Figure 2.1), precipitation, as well as internal heat sources such as friction/dissipation and heat produced by chemical and biological processes (Brown, 1969; Meier et al., 2003).

Brown (1969) pioneered the use of energy balances to predict water temperatures in small streams. He found that during the day, net all-wave radiation was the predominant energy source on unshaded stretches of small streams and that it could be more than five times greater than on shaded reaches. Except at very low solar elevation angles, most incident solar radiation is absorbed in the water column or the stream bed, providing direct thermal energy to the stream (Evans et al., 1998; Johnson, 2004). Values for incoming net solar radiation reported in literature range from 367 W/m^2 to 800 W/m^2 in unshaded reaches (Brown, 1969; Evans et al., 1998; Johnson, 2004) and from 4 W/m^2 to 140 W/m^2 in shaded reaches (Brown, 1969; Meier et al., 2003; Story et al., 2003; Johnson, 2004).

Besides solar input, stream discharge may be one of the most important factors in determining stream temperatures (Boyd & Sturdevant, 1997). Low discharge volumes have been demonstrated to result in high river water temperatures, and the implementation of minimum flow requirements has been suggested to manage high stream temperatures (Sinokrot & Gulliver, 2000). The smaller the volume of water, the longer the residence time and hence the larger the influence of heating processes per unit volume. As a result, small streams are particularly vulnerable to high temperatures during low flow periods, which in the interior of BC occur when the highest air temperatures are recorded. Increasing stream discharge moderates summer water temperatures through an increase in depth (lower surface to volume ratio), increased velocity (shorter exposure to heating processes) and the thermal inertia of larger water volumes (Gu et al., 1998). The relationship is exponential, with rapid stream temperature increases as discharges decline beyond certain thresholds.
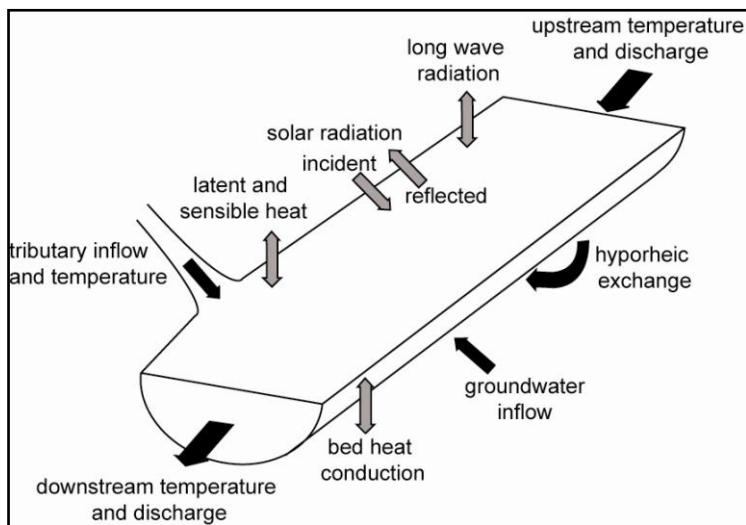
Figure 2.1: Factors controlling stream temperature (modified from Moore et al., 2005).

Brown (1969) found that latent and sensible heat exchange accounted for less than 10% of the total energy exchange in unshaded sections of small streams and thus played minor roles in determining maximum stream temperatures. He also concluded that streambed conduction was only significant in bedrock bottom streams and did not play a major role in gravel-bottom creeks due to the low thermal conductivity of the streambed. However, Johnson (2004) demonstrated that stream temperatures were much more dampened in an alluvial reach than a bedrock reach, suggesting that conduction and hyporheic exchange played an important role in regulating stream temperatures.

Energy flux due to precipitation has been considered to be localized in time and negligible within the overall heat budget, even during large rainfall events (Evans et al., 1998). The difference in air temperature between shaded and unshaded reaches and resulting differences in convective heat exchange between the stream and the air are considered to be negligible because convective heat transfer has been demonstrated to comprise a relatively small portion of the total heat flux of small streams (Johnson, 2004).

Many components of the stream heat budget have been studied extensively but few studies quantify the magnitude of advective cooling from groundwater inflows (Mellina et al., 2002; Story et al., 2003). However, stream cooling in groundwater discharge areas may not be limited to advective cooling from groundwater influx. Silliman and Booth (1993) found that streambed temperature gradients in areas of groundwater inflow are greater, resulting in increased heat loss via streambed conduction. Therefore, groundwater inflows may play an important role in moderating stream temperatures.

2.2 Groundwater - Surface Water Interactions

The importance of groundwater - surface water interactions and their effect on stream ecology have been studied since the 1960s (Sophocleous, 2002) and have been emphasized by researchers such as Hynes (1975; 1983) and Danielopol (1980). Streams that are connected to the subsurface through their hyporheic zone (Figure 2.1) cannot be regarded as separate from groundwater. Rather, the hyporheic zone is an interface between surface and groundwater systems and is highly active biologically (Gordon et al., 2004).

Streams can be gaining where groundwater flows upwards through the streambed, or losing where water flows downwards into the aquifer. The direction and amount of flow can vary greatly at different locations within the same stream. They are determined by the hydraulic gradient between the stream and the groundwater system, and the hydraulic conductivity of the streambed (Kalbus et al., 2006). Surface water seepage from streams has been studied primarily in the context of aquifer recharge (Ferguson et al., 2003; Niswonger et al., 2005) and thermal pollution of groundwater from cooling ponds (Andrews & Anderson, 1979).

Groundwater discharge to streams has been recognized for its importance in maintaining base flows (Winter, 2007), instream nutrient cycling (Hayashi & Rosenberry, 2002), stream water chemistry and metabolism (Jones & Holmes, 1996), salmonid spawning, and providing thermal refugia from temperature extremes to aquatic fauna (Power et al., 1999). Jones and Mulholland (2000) provide a synthesis of current research in groundwater – surface water interactions and their importance for biogeochemical and ecological processes in and around streams.

Measuring interactions of surface water and groundwater is not a simple task. The identification of locations of groundwater inflows to streams can be particularly difficult (Silliman & Booth, 1993). Kalbus et al. (2006) reviewed common methods that have been used for this purpose. These include the direct measurement of fluxes using seepage meters (Lee, 1977), methods based on Darcy's law (Baxter et al., 2003; Hunt et al., 2006), incremental streamflow measurements (Grapes et al., 2005), naturally occurring tracers (Malcolm et al., 2004; Wollschläger et al., 2007), injected tracers (Constantz et al., 2003), hydrograph separation (Caissie et al., 1996), thermal remote sensing (Loheide & Gorelick, 2006), distributed temperature sensing using fibre optics cables (Selker et al., 2005), and streambed heat tracer methods (Lapham, 1989).

The use of heat as a natural tracer of groundwater movement has been employed since the 1960s when Suzuki (1960) presented an analytical solution that used soil temperature profiles to estimate water

infiltration rates into the ground. Stallman (1965) generalized this approach to estimate vertical groundwater fluxes and hydraulic conductivities. The method is based on the concept that in the absence of groundwater flow, heat is transferred between the soil surface and depth purely via conduction. Bredehoeft and Papadopulos (1965) describe how the departure of a temperature profile from a purely conductive state can be used to determine the magnitude and direction of advective flows (Figure 2.2).

At a constant surface temperature, the thermal gradient becomes convex upward or downward depending on direction of groundwater movement. The amount of curvature depends on the magnitude of groundwater flow. The penetration of surface temperature fluctuations into the ground is deeper where water is flowing downward and shallower where groundwater flows upward (Figure 2.3). Bredehoeft and Papadopulos (1965) provided a solution to the one-dimensional form of the heat transport equation (Stallman, 1963) via the use of a number of type curves.
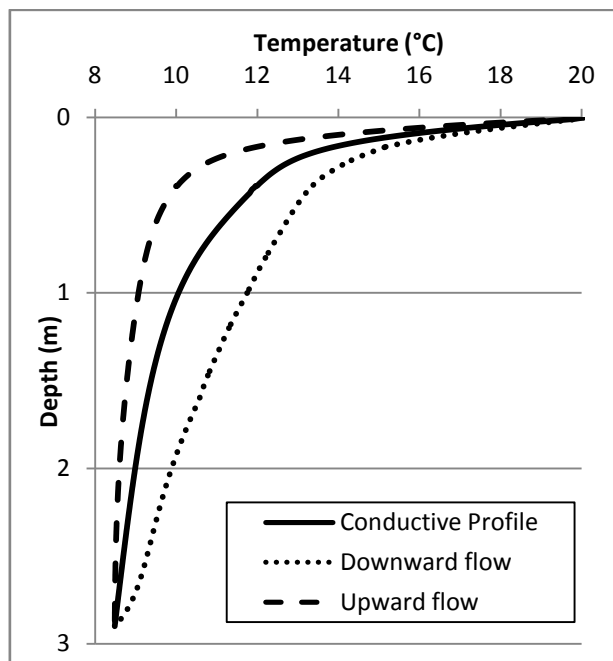
Figure 2.2: Schematic summer season stream bed temperature profile with depth under purely conductive and upward and downward groundwater flow conditions.
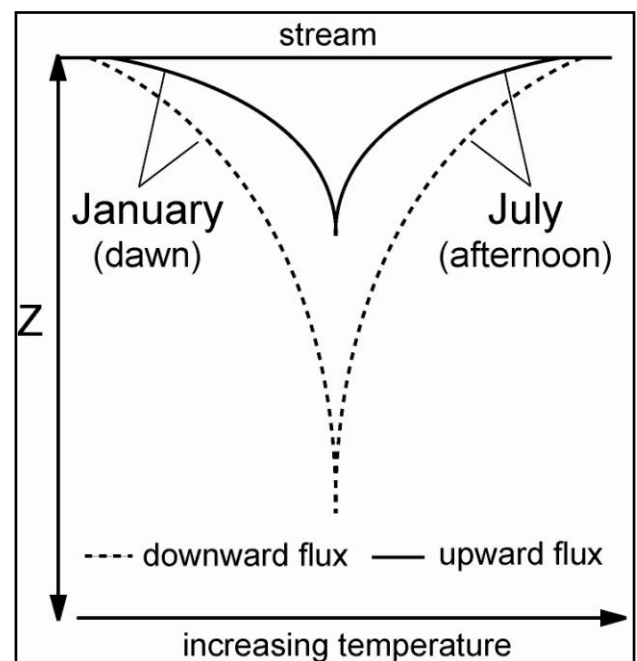
Figure 2.3: Sediment temperatures versus depth for gaining streams compared with losing streams over daily or annual cycles (modified from Constantz & Stonestrom, 2003).

The one-dimensional formulation of the differential conductive-convective heat transport equation according to Stallman (1965) is

$$\frac{K_e}{\rho c} \frac{\partial^2 T}{\partial z^2} - \frac{\rho_f\, c_f}{\rho c} \frac{\partial T}{\partial z} \cdot q_z = \frac{\partial T}{\partial t} \qquad (2.1)$$

where

K_e = effective thermal conductivity of the solid-fluid matrix (W/mK)

ρ = density of the solid-fluid matrix (kg/m^3)

c = specific heat of the solid-fluid matrix (J/kg K)

T = temperature (°C)

ρ_f = density of the fluid (kg/m^3)

c_f = specific heat of the fluid (J/kg K)

q_z = vertical darcian fluid velocity (m/s)

z = depth (m)

This equation consists of a conductive term (first term on the left) and an advective term (second term on the left) and temperature change over time on the right hand side. The solution applies under steady, uniform, one dimensional fluid flow, under isotropic and homogeneous conditions, where the temperature of the fluid is in equilibrium with the solid at all times (Stallman, 1965). The analytical solutions of Suzuki (1960) and Stallman (1965) assume sinusoidal fluctuation of temperatures on the surface and thus represent transient conditions more similar to those found in nature.

Following the work of Suzuki (1960), Bredehoeft and Papadopulos (1965), and Stallman (1965), temperature profiles were used to assess heat and groundwater flow in several basin-scale groundwater studies (Anderson, 2005). With the improvement in computational abilities, numerical solutions were developed with the ability to model two and three dimensional coupled heat and groundwater flow (Anderson, 2005).

Lapham (1989) made important modifications to the Bredehoeft and Papadopulos (1965) solution, presenting an explicit finite difference approximation of their equation, specifically applicable to stream environments. His solution allows for non-sinusoidal surface temperature fluctuations like those of a natural stream and does not assume the ambient groundwater temperature is equal to the mean stream temperature. He successfully used his equation in a computer model to estimate vertical groundwater flow and vertical hydraulic conductivities in two streams under upward and downward

flow conditions. Since then, numerous computer codes have emerged that can model the joint transport of heat and fluid.

Several analytical solutions have also been developed for use in stream environments. Silliman et al. (1995) used the time series of surface water temperatures as the upper boundary condition in a one-dimensional analytical heat transport model based on Stallman's (1965) analytical solution. The solution estimates downward seepage velocities from the propagation of temperature fluctuations from the water column to the deeper sediments. Becker et al. (2004) used the same method to estimate upward groundwater seepage into a stream. Several additional analytical solutions have used the phase shift and amplitude decrease with depth of a fluctuating temperature signal to assess seepage rates under streams and on the sea floor (Goto et al., 2005; Hatch et al., 2006; Keery et al., 2007).

The use of heat tracer methods in stream environments has gained popularity due to the low expense of instrumentation and relative ease of data collection and analysis (Stonestrom & Constantz, 2003). The method has been successfully used to identify gaining and losing stream reaches (Silliman & Booth, 1993) and to estimate groundwater flow rates into or out of streams (Lapham, 1989; Silliman et al., 1995; Constantz & Thomas, 1996; Constantz et al., 2003; Becker et al., 2004; Conant, 2004; Hatch et al., 2006; Keery et al., 2007; Schmidt et al., 2007; Essaid et al., 2008).

2.2.1 Numerical Modeling

One of the more frequently used models in surface water – groundwater interaction studies is the model VS2DH which was developed by the US Geological Survey (USGS). The code is a modification by Healy and Ronan (1996) of a solute transport model initially developed by Lappala et al. (1987).

Niswonger and Prudic (2003) describe the use of VS2DH in a stream environment to estimate streambed seepage and hydraulic conductivity of the streambed. The model is set up to predict temperatures at a preset depth in the streambed under various seepage fluxes. The simulated temperatures are then compared to measured temperatures. Where hydraulic head and streambed temperature data are available as boundary conditions, the model is calibrated by adjusting streambed hydraulic conductivities until the best fit to the observed temperature time series is obtained (Stonestrom & Constantz, 2003). In the absence of head data, groundwater flux can be directly adjusted until a good fit is obtained.

Several site-specific parameters describing the thermal and hydraulic characteristics of the streambed are required for this model. The major parameter governing seepage velocities besides head gradient is

streambed hydraulic conductivity. It is usually a parameter of interest in groundwater - surface water interaction studies and is adjusted in the modeling process until a good fit between observed and modeled temperatures is obtained.

The thermal characteristics of a streambed vary over a much smaller range among different sediment types than do hydraulic properties and do not result in very large errors in estimated flux compared to estimates derived from hydraulic properties alone (Stonestrom & Blasch, 2003). Therefore, thermal properties are frequently derived from the literature. Where little groundwater flux is present and conduction is the primary method of heat transfer in the streambed, thermal properties become more important to obtaining a good model fit than where heat transport is dominated by advection. Therefore, adjustment of the thermal parameters may be necessary where a good fit cannot be obtained by varying hydraulic conductivities alone. For example, Schmidt et al. (2007) had to adjust their thermal conductivity value obtained from direct measurement of a sediment sample to obtain a good fit for their simulated vs. observed groundwater fluxes. Generally, the influence of streambed thermal properties on modeled groundwater fluxes becomes significant when fluxes are less than 1×10^{-7} m/s (Essaid et al., 2008).

2.2.1.1 Thermal Conductivity

The streambed thermal properties that need to be estimated are the saturated bulk thermal conductivity and the heat capacity of the dry solids. Lapham (1989) discusses sediment thermal properties in details. The thermal conductivity of the fluid-solid mixture is a measure of its ability to transfer heat. It is influenced by sediment structure and packing, mineral composition of the sediment, salt content of the water, temperature of the sediment, and thermal conductivity of the sediment solids. In unsaturated sediments, thermal conductivity is also a function of water content (Lapham, 1989). Typical streambed sediment bulk thermal conductivity values range from 0.8 to 2.5 W/mK (Hatch et al., 2006). In comparison, hydraulic conductivities of streambed materials may vary over many orders of magnitude.

According to Lapham (1989), the thermal conductivity of saturated fine grained sediments ranges from 0.84 to 1.42 W/mK and that of coarse grained sediments ranges from 1.21 to 2.47 W/mK. Fine grained sediments mainly consist of silt and clay sized particles that may be predominantly clay minerals. Coarse grained sediments primarily consist of sand and gravel sized feldspathic and siliceous minerals. Organic matter has a relatively low thermal conductivity (0.25 W/mK) compared to clay minerals (2.9 W/mK) and quartz (8.4 W/mK) (Stonestrom & Blasch, 2003).

Quartz is usually abundant in coarse grained sediments, whereas micaceous or clay minerals are usually abundant in fine-grained sediments. This contributes to the higher thermal conductivity of coarse grained materials. The thermal conductivity would be expected to be comparatively low where organic material is present in the streambed. The thermal conductivity of saturated sediments varies with temperature as the thermal conductivity of water has a temperature dependence, but generally the variation is very small (Lapham, 1989). The literature reviewed in Table 2.1 indicates that bulk thermal conductivity values for streambed materials used in other studies range from 0.5 (peat) to 3.76 W/mK (gravel, sand and clayey sand).

Table 2.1: Thermal conductivity of saturated sediments from literature

Streambed Material	Value (W/mK)	Method	Source
sandy silt, sandy clay, and silty clay	1.0	literature	Hatch et al. (2006)
sand, gravel and silt	2.2	estimated from best fit in VS2DH	Essaid et al. (2008)
gravel, sticky silt	1.4		
sand, gravel, cobbles	2.0		
sand and gravel	1.8		
fine to very fine sandy silt	1.4		
medium to coarse grained sand with cobble layers	1.8		
poorly sorted gravel, sand, silt, and clay (mountain front stream)	2.3	inferred from bulk density and texture	Niswonger et al. (2005)
crushed rock, sandy, alluvial material	2.0	literature	Kalbus et al. (2009)
sand	2.0 - 2.5	literature	Stonestrom and Blasch (2003)
clay	1.5		
peat	0.5		
gravel, sand and clayey sand	3.76	estimated from best fit in VS2DH	Dowman et al. (2003)
coarse and fine sand (tightly packed)	2.9	measured	Smits et al. (2009)
coarse and fine sand (loosely packed)	2.5		
clay	1.1	unknown	Campbell and Norman (1998)
loam	1.5		
sand	2.2		
gravel rich alluvial aquifer	1.59	estimated from sediment properties	Taniguchi (1993)
mixture of clay, silt, fine sand and organics	0.92	estimated from sediment properties	Lapham (1989)
fine to coarse sand and gravel	1.93		
fine grained sediments	0.84 -1.42		
coarse grained sediments	1.21 -2.47		
unconsolidated medium and fine sand layers	1.0	literature	Constantz et al. (2003)
medium coarse sand with gravel	1.88	experimentally determined	Andrews & Anderson (1979)
fine to very fine sand	2.1		
sandy silt to sandy clayey silt	1.75		
grey sandy silt with organic matter	1.75		
varved clay with sandy seams	1.67		
fine sand	1.45 – 1.5	measured, adjusted for best model fit	Schmidt et al. (2007)
mixed sand and gravel	1.4	literature	Keery et al. (2007)
fine sand with some organic matter	1.8	literature	Anibas et al. (2009)
coarse gravel to fine silts containing substantial organic matter	0.96	literature	Silliman et al. (1995)

2.2.1.2 Volumetric Heat Capacity

VS2DH also requires the heat capacity of dry solids in the streambed. This parameter refers to the quantity of heat (Joules) that must be added to a unit volume of dry mineral grains to raise its temperature by 1°C. Heat capacity varies little among coarse and fine grained materials. Some studies report the heat capacity of mineral solids for this parameter but others report that of the dry bulk sediments. The heat capacity values for dry mineral solids expected to be encountered in a stream channel ranges from 1.1×10^6 J/m^{3}K to 2.18×10^6 J/m^{3}K (Table 2.2).

Table 2.2: Volumetric heat capacity of dry solids from literature

Streambed Material	Value (x 10^6 J/m^3 K)	Method	Source
sands and gravels, interbedded with thin layers of silt and clay, and alluvium	2.18	literature	Cox et al. (2007)
unconsolidated medium and fine sand layers	2.18	literature	Constanz et al. (2003)
sand clay silt loam	1.3 1.2 1.1	literature	Stonestrom and Blasch (2003)
coarse and fine sand	1.5	unknown	Smits et al. (2009)
sand loam clay	1.4 1.2 1.1	unknown	Campbell and Norman (1998)
sand	2.18	literature	Su et al. (2004)
silt, clay, peat, and sand	2.0	unknown	Schmidt et al. (2007)
soil	1.1	unknown	Nobel (2005)
poorly sorted gravel, sand, silt, and clay (mountain front stream)	2.0	inferred from bulk density and sediment texture	Niswonger et al. (2005)

2.2.1.3 Thermal Dispersivity

Anderson (2005) noted that the effective thermal conductivity of the solid-fluid matrix (K_e) ought to account for the effects of thermal dispersion, the transport of heat caused by groundwater velocity variations within the pore space. They suggest the following alteration to the first term on the left hand side of equation (1):

$$\frac{K_e}{\rho c} = \frac{K_o}{\rho c} + \alpha^* |q| \qquad (2.2)$$

where

K_o = effective thermal conductivity of the solid-fluid matrix (W/mK)

α^* = thermal dispersion coefficient (m)

$|q|$ = vector of darcian fluid velocity (m/s)

The first term on the right ($K_o/\rho c$) is the thermal diffusivity. The treatment of thermal dispersion and its magnitude in the context of surface water groundwater interactions is a contentious topic in the literature (Anderson, 2005). The term is analogous to solute dispersivity and can only be neglected when seepage velocity is nearly zero. Contrary to solute dispersivity, heat dispersion results in additional heat conduction as heat spreads through both solids and fluids in the streambed (Rau, 2008). Hatch et al. (2006) conducted a sensitivity analysis of modeled seepage fluxes to a range of thermal dispersivity values. The errors associated with an incorrect selection of the magnitude of thermal dispersivity increased as seepage velocities increased.

Overall, great uncertainty is associated with thermal dispersivity as it is difficult to quantify and is thought to be scale dependent. Commonly quoted values range from 1% to 10% of the length scale, which in this type of study is the distance between the lowest and uppermost temperature loggers (Keery et al., 2007).

Constantz et al. (2003) used VS2DH to assess stream water – groundwater exchanges in the Santa Clara River in California. They determined that, at a spatial scale of approximately 1 m, changing dispersivity from 0.01 m to 0.5 m made no difference in simulated temperature time series (and therefore, groundwater fluxes) when fluxes were on the order of 10^{-7} m/s. However, the change did have a significant impact when seepage fluxes were on the order of 10^{-5} m/s. Dispersivity values of 0.01 m produced the best fit in this case, and lowering them to 0.00 m did not have any effect on modeled temperatures and groundwater fluxes.

In a separate study, incorporating thermal dispersivity of 0.1 m and 0.01 m had an impact on flux estimates only where upward fluxes were greater than 10^{-6} m/s or 10^{-5} m/s, respectively (Keery et al., 2007). A range of dispersivity values from the literature is presented in Table 2.3.

Table 2.3: Thermal dispersivity values for streambed materials from literature.

Streambed Material	Value (m)	Scale (m)	Source
unconsolidated medium and fine sand layers	0.01	1.0	Constantz et al. (2003)
sandy silt, sandy clay,and silty clay	0.001	1.0	Hatch et al. (2006)
sands and gravels, interbedded thin layers of silt and clay, alluvium	0.5	3.0	Su et al. (2004)
poorly sorted gravel, sand, silt, and clay (mountain front stream)	0.1	1.5	Niswonger et al. (2005)
mixed sand and gravel	0.01 – 0.1	1.0	Keery et al. (2007)

2.3   Pacific Salmon Life History

Pacific salmon are a highly valued resource throughout the north Pacific for their economic, ecological and cultural importance. Groot and Margolis (1991) provide detailed descriptions of the life history of the anadromous Pacific salmon species. Their natural range extends from the San Francisco area of California north along the BC and Alaska coast south to Korea, but they have been widely introduced elsewhere.

At various life stages, salmonids inhabit oceans, estuaries, coastal and inland rivers and lakes. Most spawn in the fall in rivers or on lakeshores. Fry emerge in the following spring and after a period of freshwater residence in their natal or nearby non-natal streams, the fry migrate to the ocean. Freshwater residence can last from several days to several years depending on the species. Upon smoltification (adjustment to saline water), juvenile salmonids migrate to estuaries and oceans where a majority of their growth occurs. The fish mature and migrate back to the spawning grounds following a residence at sea from one to seven years (Groot & Margolis, 1991).

The freshwater residency period of juvenile coho upon emergence is generally at least one year, with migration to the ocean commencing during freshet of the following year (Sandercock, 1991). However, the occurrence of coho "nomads" has been documented extensively. These are age 0 coho fry that migrate downstream, spend their first summer rearing in estuaries, and then overwinter in freshwater (Koski, 2009). Thus, outmigration from streams of coho fry in their first summer is not uncommon.

The length of freshwater residence varies among Chinook populations. Ocean-type Chinook remain in freshwater from a few days to a few months after hatching but migrate to the estuaries within their first summer. In contrast, stream-type Chinook rear in freshwater for a year or even longer in some northern rivers (Murray & Rosenau, 1989). Major causes of mortality during freshwater residence are predation by other fish, invertebrates, and birds, as well as disease infection (Healey, 1991).

2.4   Freshwater Salmon Habitat

2.4.1   Physical Habitat

During their freshwater residence, juvenile salmonids inhabit the mainstems or tributaries of their natal streams. Chinook fry occupy riverine habitat, avoiding areas of still water and velocities above 0.3 m/s (Murphy et al., 1989). This separates their habitat from coho and other salmonids, which prefer still water areas (Murphy et al. 1989). The diet of Chinook fry consists primarily of aquatic and terrestrial

insects. They feed in the water column and on food drifting at the surface (Healey, 1991). Once they establish a territory, limited upstream or downstream movement occurs and the territory is defended against other fish (Edmundson et al., 1968; Reimers, 1968).

Coho fry emerge from their redds during freshet and readily colonize flooded areas. They rear in small creeks, backwaters and side channels where water velocities are low or still and channel gradient is less than 3%. They prefer structurally complex habitat with abundant cover (Interior Fraser Coho Recovery Team, 2006). The diet of coho is similar to that of Chinook. They prefer to capture food from the current or the surface and seldom feed off the bottom of the stream (Sandercock, 1991). Coho may migrate considerable distances to find suitable rearing habitat and defend their territory aggressively once established (Hoar, 1958).

2.4.2 Dissolved Oxygen

Salmonids require cool, clean and well-oxygenated water to survive. Turbulent flowing waters are normally saturated with oxygen due to constant mixing and resulting contact of water with the atmosphere. When water stagnates, this constant contact is no longer present and oxygen concentrations are dictated by oxygen production and use within the stream. In stagnant, eutrophic streams, dissolved oxygen (DO) may be high during the day and very low at night, when photosynthesis cannot counterbalance the loss of oxygen through respiration and decomposition (Kramer, 1987).

Low DO concentrations ultimately lead to suffocation of fish, but other adverse effects are noted with decreasing oxygen concentrations. Significant mortality in Chinook and coho salmon was observed over a 24 hr time period as DO values were reduced below 2 mg/L (Doudoroff & Shumway, 1970). Sublethal levels can negatively impact the availability of energy for locomotion, growth and reproduction of fish (Kramer, 1987). Feeding is often strongly reduced under low oxygen conditions. For instance, the growth rate and food consumption of coho were reduced when DO levels were between 4 and 5 mg/L (Herrmann, 1958).

Hypoxic waters can act as a barrier to the movement and distribution of salmonids. Salmonids in the laboratory have frequently demonstrated preference for waters with high DO and avoidance of waters with low DO (Kramer, 1987). Juvenile Chinook salmon actively avoided waters with DO values between 1.5 and 4.5 mg/L (Whitmore et al., 1960). Hallock et al. (1970) indicated that Chinook salmon would not migrate through sections of a river until DO rose to about 4.5 to 5.0 mg/L. A guideline for dangerously low DO values has been published by the BC Ministry of Environment, Lands and Parks (Truelson, 1997),

which cites the instantaneous minimum DO for all life stages of salmonids other than buried embryos or alevins as 5 mg/L.

2.4.3 pH

The detrimental effects of extreme pH levels on fish were described in detail by McKean and Nagpal (1991). Acid toxicity at low pH values can result in an inhibition of oxygen uptake and transport through the gills, disturbance of the ionoregulation mechanism, and disturbance of the acid base balance of intra- and extracellular fluids. Further, low pH values have negative effects on egg fertilization, embryo development and hatchling success. Coho salmon from six BC hatcheries showed significant physiological responses and mortalities following exposures to pH values ranging from 3.55 to 4.10 (McGeer et al., 1991). These pH levels are very low and are not naturally encountered in water bodies of the BC Interior, which on average have slightly higher pH levels than coastal streams (McKean & Nagpal, 1991). The critical low pH for rainbow trout reproduction is 5.5 (McKean & Nagpal, 1991).

pH values over 9.0 can cause a disruption of the ammonia secretion mechanism through the gills, resulting in a buildup of ammonia in the bloodstream and body tissue of fish. Where high concentrations of NH_3 are present in the water the ammonia secretion mechanism can be disrupted by small increases in pH even at ambient pH values lower than 9.0 (McKean & Nagpal, 1991). Wright and Wood (1985) found that ammonia exchange in rainbow trout is reduced by approximately 25% at a pH of 8.69 and is almost entirely inhibited at pH values of 9.54. Under such alkaline conditions, ammonia began to accumulate internally within a few hours. However, McGeer et al. (1991) found significant physiological changes in some coho salmon stocks during exposures of varying length and intensity to high pH conditions (9.4 to 10.1 over 1 to 6 days), while coho from other stocks remained unaffected. They indicated that the lack of consistent response may be a result of the sampling time, pH and the physiological parameters measured.

Acclimation may explain some of the variation in response of fish to high pH, particularly when increases happen slowly (McKean & Nagpal, 1991). Murray and Ziebell (1984) showed that rainbow trout can acclimate to pH levels of 9.5 if increases are gradual, and that under such circumstances mortality occurs at pH of 10.2. When pH was increased rapidly, mortality occurred at pH 9.5.

McKean and Nagpal (1991) conclude that the tolerance range for pH for trout is between 5 and 9 to 9.5. However, sublethal stress is likely to occur at pH levels approaching the upper and lower limit, especially in combination with other stressors.

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2.4.4 Electrical Conductivity

Guidelines for conductivity values in British Columbia do not currently exist and typical measurements for interior streams range up to 500 µS/cm (Cavanagh et al., 1998). Conductivity is easily measured in the field and is frequently used as a proxy for the concentration of Total Dissolved Solids (TDS) in water (Mackie, 2001). Weber-Scannell and Duffy (2007) reviewed the effects of TDS on aquatic organisms. They noted that TDS level is a summary measurement of dissolved materials in the water, including inorganic salts and organic matter, and does not differentiate between ions. TDS and conductivity are not a particularly sensitive measure to assess salmonid habitat quality since toxicity varies strongly depending on ionic species, fish species, and life stage (Weber-Scannell & Duffy, 2007).

Stekoll et al. (2003) determined that coho salmon fry exposed to TDS levels up to 2500 mg/L were not significantly affected by short term or chronic exposures. Similarly, Chapman et al. (2000) found no toxicity in rainbow trout fry after exposures to TDS levels over 2000 mg/L. This indicates that juvenile salmonids may be relatively insensitive to naturally occurring TDS levels. However, exposure of coho eggs to TDS above 1875 mg/L during fertilization resulted in lower hatch rates, hatching delays, and effects on subsequent growth and development of fertilized eggs (Stekoll et al., 2003). Thus, although fertilized eggs, fry and juveniles seem unaffected, high TDS levels may have significant negative effects on egg fertilization during salmonid spawning (Weber-Scannell & Duffy 2007).

Kimmel and Argent (2010) conducted a study on fish community metrics in a warm water stream impacted by mine drainage. They found an impairment of fish communities at conductivities of 3,000 to 3,500 µS/cm and TDS levels of 2,000 to 2,300 mg/L.

The absolute value of conductivity (or TDS) tolerated by fish will be related to the chemical composition of the materials making up the stream dissolved load. Aside from direct effects on fish, the relative value of conductivity within a stream can be used as a measure of the change in chemistry from one area of a stream to another. A simple approximation of the relative importance of water sources in a stream can be achieved by comparing the conductivity of different water sources such as groundwater, upstream streamwater, or discharges to surface water.

2.4.5 Temperature

Salmonids are ectotherms, meaning that their body temperature depends on the temperature of their environment. They do not have the ability to anatomically or physiologically regulate their body temperature and their metabolic rate is a function of temperature (Barton, 2007). Most fish species

have well-defined temperature preferences and tolerances that often coincide with the species-specific optimum temperature. At this temperature a maximum surplus of energy is available for growth and activity beyond the maintenance of basic bodily functions (Coutant, 1976). Salmonids are cold water species, meaning that their metabolic optimum is reached at cooler water temperatures.

Studies of temperature effects on salmonids have experienced renewed interest in light of the current debate on climate change and concerns over low streamflow volumes (McCullough et al., 2009). High water temperatures are of particular concern in the interior areas of the Pacific Northwest, and have been documented to approach or exceed upper thermal limits of salmonids in several streams in the Thompson watershed, near which Fortune Creek is located (e.g., Walthers & Nener, 2000). Climate change may leave large sections of previously utilized rearing habitat unsuitable and effectively restricts the available habitat and thus the carrying capacity.

At temperatures above the optimum, energy requirements for the maintenance of vital bodily functions exceed the amount of energy that can be obtained through respiration (Coutant, 1976). No energy is then available for growth which leads to the weakening and ultimately the demise of the fish. Water temperatures above approximately 22°C have also been shown to cause heat shock and protein damage in Atlantic salmon (Lund et al., 2002).

Water temperatures can greatly disrupt the life cycle of salmonids, even when they remain well below lethal limits, by affecting smoltification and migration timing, resistance to disease infection, ability to compete for food and habitat, predation risk, and the toxicity of various pollutants and poisons (Coutant, 1976).

Zaugg and Wagner (1973) noted that when water temperatures increased to 13°C steelhead trout migration decreased. They suggested that higher springtime temperatures lead to lower success in smoltification and interference with migration. Temperatures between 15°C and 20°C have been demonstrated to inhibit smoltification outright in steelhead and in coho salmon resulted in transitory smoltification only (Zaugg & McLain, 1976).

Water temperature has an influence on the susceptibility of salmonids to freshwater diseases (McCullough, 1999). One of the most extensively documented is the bacterial *columnaris* disease. This disease was found to be more infectious at water temperatures above 20°C and has been accused of playing a major role in the decline of Columbia River Chinook salmon (Ordal & Pacha, 1963, as cited in McCullough, 1999). Fish infected with highly virulent strains of this disease may die within 12 to 24

hours (Pacha, 1961). Sockeye infected with the disease have been found to recover when held between a mean of 12.8°C and a maximum of 13.9°C (Colgrove & Wood, 1966, as cited in McCullough, 1999).

Fryer and Pilcher (1974) and Fryer et al. (1976) investigated the effect of temperature on several important salmonid diseases, including two bacterial diseases (*Aeromonas* and *Columnaris* diseases) and infections attributable to the myxosporean parasite *Ceratomyxa Shasta* (Noble). They found that mortality resulting from all diseases was significantly increased at water temperatures between 17.8°C and 23.3°C and was very low at 3.9°C to 9.4°C. All diseases progressed more rapidly as temperatures increased. While some freshwater salmonid diseases exist at cold temperatures, many are more pathogenic at elevated temperatures. The lowest risk for bacterial disease infection of salmonids exists between approximately 12.8°C and 15°C (McCullough, 1999).

The presence of toxic substances in streams can affect the survival of juvenile salmonids. Cairns et al. (1978) extensively studied the effects of temperature on the toxicity of various chemicals. Their research covered numerous aquatic organisms including plankton, aquatic invertebrates as well as fishes. They found that when exposed to chemicals such as phenol, cyanide, copper, zinc, and chromium, temperature affects the time until death of a fish occurs and concentration of the chemical affects which temperature is best for survival. However, they did not observe a consistent response of all fish to elevated chemical concentrations and elevated temperatures. Servizi and Martens (1991) found that the tolerance of juvenile coho to suspended sediment decreased by 67% when temperatures were 18°C instead of 7°C.

Juvenile salmonids frequently enter into competitive interactions with each other, with other salmonid species, and with warm water species during the period of freshwater rearing. Several studies have provided evidence that the ability of salmonids to compete is inhibited by high temperatures (McCullough, 1999). For instance, brook trout (*Salvelinus fontinalis* Mitchill) were dominant over rainbow trout at 13°C, but not at 19°C (Cunjak & Green, 1986). At 19°C to 22°C, steelhead production was decreased by more than half when redside shiners (*Richardsonius balteatus* Richardson) were present. This was not the case at temperatures between 12°C and 15°C (Reeves et al., 1987). Similarly, Reese and Harvey (2002) noted that the presence of Sacramento Squawfish (*Ptychocheilus grandis* Ayres) reduced steelhead growth by 50% in water temperatures between 20°C and 23°C compared to 15°C to 18°C. This suggests strong temperature dependence in the competitive ability of steelhead.

Likewise, temperature has been demonstrated to affect the distribution of juvenile Chinook salmon in the presence of redside shiners at 18°C to 21°C but not at 12°C to 15°C (Hillman, 1991, as cited in Sauter

et al., 2001). Taniguchi et al. (1998) found that creek chub (*Semotilus atromaculatus* Mitchill) became competitively dominant over brook trout at 24°C and over brown trout at 26°C.

Increasing water temperatures can force juvenile salmonids to share their habitat with predatory warm water species. These may include species such as northern pikeminnow (*Ptychocheilus oregonesis* Richardson) and largemouth bass (*Micropterus salmoides* Lacépède) (McCullough, 1999). The presence of such predators is especially dangerous when temperature preferences of salmonids are exceeded, as they are not able to escape at their maximum swimming speed under such conditions (McCullough, 1999).

Temperature has indirect effects by influencing the environment in which fish live. The solubility of oxygen is strongly dependent on water temperature. Naturally occurring saturated concentrations of DO are between 11.3 (mg/L) at 10°C to 8.2 (mg/L) at 25°C at zero salinity and at sea level (Gordon et al., 2004). Salmonids generally require DO concentrations of no less than 5 mg/L (Truelson, 1997). Metabolic oxygen consumption in fish increases as water temperatures increase. For instance, the oxygen consumption of brown trout (*Salmo trutta L.*) at 30°C is almost 3.5 times higher than at 10°C (Beamish, 1964). However, the saturation levels of DO in water at 30°C are only approximately 67% of those at 10°C (Lewis, 2006). Thus, as salmonid oxygen requirements increase with temperature, the absolute amount of oxygen available to the fish decreases.

2.4.5.1 Thermal Limits

Temperature preferences and tolerances of salmonid species in their various life stages have been studied extensively in the past (e.g., Fry, 1947; Brett, 1952). At the optimum temperature, a maximum amount of energy is available for growth (Coutant, 1976), whereas the optimum growth temperature declines under low food availability (Brett et al., 1982). Thus, the temperature range occupied by salmonids in natural streams may depend on food availability and juvenile salmonids in productive rivers may thrive at higher temperatures than in less productive rivers (McCullough et al., 2009).

The accepted method for determining the preferred temperature of a species is to place fish in a habitat that is otherwise identical except for a thermal gradient and observing the temperature at which they congregate. Laboratory studies allow for the control of other factors such as light and chemical gradients as well as food abundance and predation (Brett, 1956). Although such congregations can be observed in the field, field thermal gradient studies are limited because other habitat conditions cannot be held constant.

The zero net growth temperature defines where the mortality and growth of a sample population are exactly equal (Armour, 1991). Under natural conditions in the field, the zero net growth temperature is dependent on food availability. Rearing temperatures selected by salmonids in nature would be below the zero net growth temperature to ensure a sustained yield (Hokanson et al., 1977).

Numerous studies have examined critical maximum temperatures at which mortality occurs in salmonids (Brett, 1952; Becker & Genoway, 1979; Brett et al., 1982; McGeer et al., 1991; Konecki et al., 1995a). At critically high water temperatures, mortality is a function of exposure time, final water temperature and starting acclimation temperature (McCullough, 1999). Lethal temperatures have been studied and reported in different ways which leads to some difficulty in comparing studies (Figure 2.4).

The upper incipient lethal temperature (UILT) is determined by subjecting the sample population to an abrupt change in temperature from a constant acclimation temperature and recording the exposure time until 50% mortality occurs (Becker & Genoway, 1979) (Figure 2.4a). This method originated with Fry et al. (1946, as cited in Kilgour & McCauley, 1986). Studies have had variable exposure times reaching from 1000 minutes (Brett, 1952) to 7 days (Elliott, 1981). Thermal tolerance increases with rising acclimation temperatures up to a certain point, at which tolerance cannot be increased (the upper ultimate incipient lethal temperature) (Armour, 1991).

The critical thermal maximum (CTM), introduced by Cowles and Bogert (1944) who studied reptiles, is determined by progressively changing the temperature from acclimation and recording the exposure time and temperature at which a physical disorganization response (e.g., loss of equilibrium) occurs (Becker & Genoway, 1979) (Figure 2.4b). Although the UILT and CTM methods differ in their approach, results from both methods can be compared to each other (Kilgour & McCauley, 1986).

Both methods described above work by exposing a sample population of fish to constant temperatures (UILT) or increasing temperatures at a constant rate (CTM). The validity of applying thermal thresholds derived from constant temperature experiments in a field setting has been questioned (Hokanson et al., 1977). Natural stream temperatures undergo diel and seasonal fluctuations. Temperatures fluctuate substantially and fish may be exposed to temperatures near their lethal limits repeatedly for short periods of time. Under such conditions, mortality may not result from a single exposure but from cumulative temperature stress after several cycles. Laboratory methods utilizing diurnal fluctuations may include constant temperature acclimation (Figure 2.4c) or acclimation to a diurnal regime (Figure 2.4d) different from the exposure regime.

For example, cutthroat trout (*Oncorhynchus clarki Utah* Suckley) did not survive constant temperatures above 24°C, but survived several hours at these high temperatures when they were alternated with cooler temperatures (Johnstone & Rahel, 2003). For coho reared in fluctuating temperatures, mortality was a function of exposure time to temperatures above 25°C as long as temperature did not exceed 28°C (DeHart, 1975). The acclimation temperature in a fluctuating regime is equal to a temperature somewhere between the daily mean and maximum, meaning that lethal temperatures in such a regime may be higher than at a constant temperature with an equal mean (DeHart, 1975; Hokanson et al., 1977).
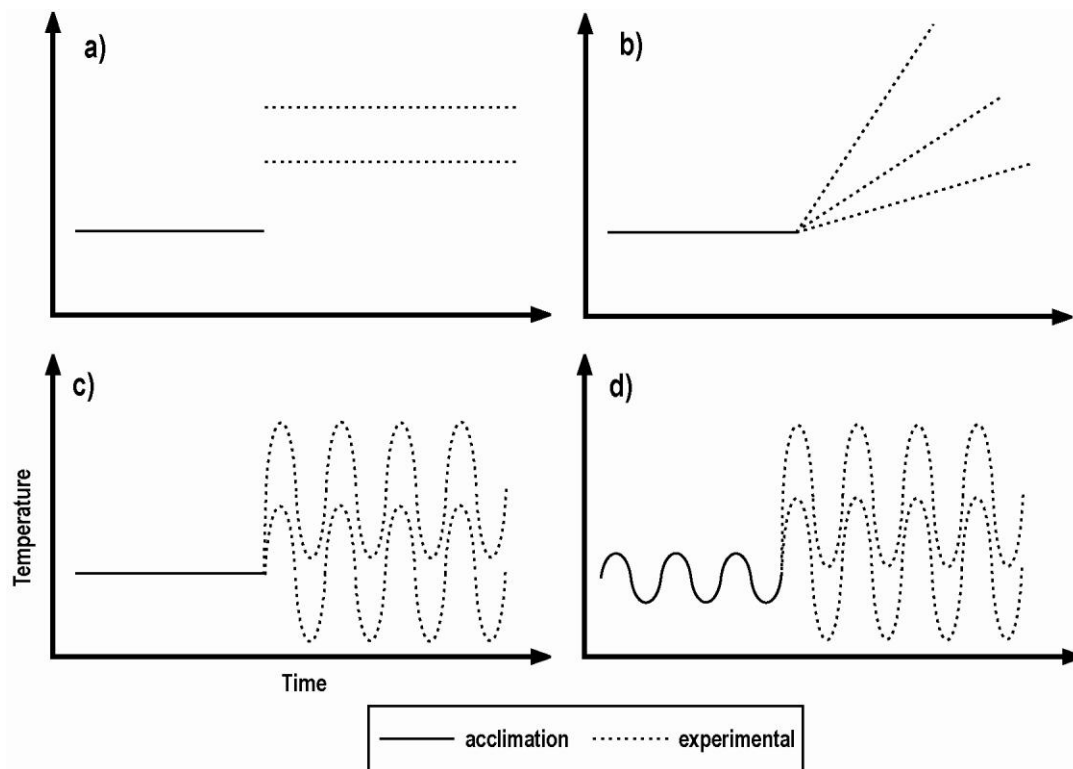
Figure 2.4: Different methods of determining thermal tolerances of salmonids including a) the UILT method, b) the CMT method, c) constant acclimation temperature and fluctuating experimental temperature and d) fluctuating acclimation and experimental temperature.

Small variations of the thermal tolerance within the same species can be explained by factors such as acclimation temperature and food availability. However, genetic differences between populations may enable some populations to withstand higher temperatures than others. For instance, Chinook salmon from a southern BC population were found to better tolerate high temperatures (22°C) than those from a northern BC population (Beacham & Withler, 1991).

While many studies using a fluctuating temperature regime provide information on daily maximum temperatures and exposure stress, the importance of daily minimum temperatures in terms of providing

respite from temperature stress has been examined in relatively few studies (McCullough et al., 2009). Schrank et al. (2003) noted that cutthroat trout did not emigrate, die or use coolwater refugia in a stream where summer temperatures reached 27°C, well above their thermal limit. They explained this by cool nighttime temperatures (10°C to 14°C below the maximum temperatures) that allowed trout to survive despite stream temperatures exceeding the 7-day UILT for several weeks. Similarly, Bjornn and Reiser (1991) reported normal growth and high densities of juvenile Chinook and steelhead in a small stream where daily temperatures frequently exceeded 24°C for short periods of time but decreased to the optimal range at night. The role and range of minimum temperatures that can enable salmonids to survive repeated short-term exposures to adverse temperatures is unclear.

The temperature thresholds discussed above are presented schematically in Figure 2.5. This research is interested in how salmonid distribution may be limited by high water temperatures. It is assumed that salmonids will avoid stream reaches with high water temperatures. The avoidance temperature is unclear but likely lies somewhere between the optimum range and lethal temperatures. Salmonids may be present in the zero growth range for short periods of time, especially where mobility is low. Avoidance temperatures in the field are frequently inferred from distribution studies (Section 2.4.5.3).
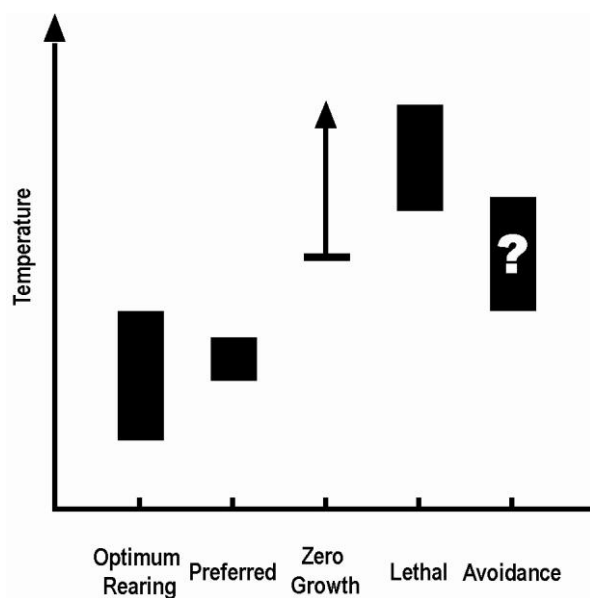
Figure 2.5: Commonly measured temperature thresholds for salmonids.

2.4.5.2 Species Specific Temperature Thresholds

A quantitative graphical summary of the salmonid temperature thresholds discussed below is presented in Figure 2.6.

2.4.5.2.1 Chinook

Temperature criteria for Chinook salmon are summarized in Table 2.4. Optimum rearing temperatures are reported to be between 10.0°C and 15.6°C by McCullough et al. (2001) and between 12.2°C and 14.8°C by Hicks (2000). These recommendations are based on a food ration of 60% of satiation which is assumed to be a typical level in natural streams. Several researchers have adjusted the constant laboratory temperatures to fluctuate so they more closely resemble natural stream temperatures. Hicks (2000) recommends that optimal growth conditions for Chinook salmon are achieved when the 7-day average maximum is between 14.2°C and 16.8°C during the peak of summer.

Marine and Cech (2004) found that juvenile Chinook can survive and grow at temperatures up to 24°C, but growth rates were decreased and smoltification was impaired compared to juveniles reared at 13°C to 16°C. Armour (1991) calculated the zero net growth temperature limits at 4.5°C and 19.1°C. Brett et al. (1982) reported that the optimum growth temperature for Chinook would be approximately 19°C at unlimited food rations. Under these conditions they were able to feed in temperatures as high as 23°C, but feeding declined to nil between 23°C and 25°C. A zero individual growth rate was observed under 60% satiation feeding at 21.4°C.

Upper thermal limits for Chinook have been determined between 23°C and 26.7°C through laboratory and field studies (Table 2.4). McCullough (1999) suggests that at high exposure temperatures, mortality is extremely sensitive to small changes in exposure time and to the acclimation temperature.

Beacham and Withler (1991) have suggested that ocean-type Chinook are better adapted to short-term exposure to high water temperature than stream-type Chinook from northern BC, as they spend a greater portion of their life in warmer coastal waters than the stream-type populations. This may mean that reported temperature criteria from literature, which are often derived from ocean-type populations, do not apply to stream-type populations as they may be even more sensitive to high temperatures. However, stream-type Chinook in the study area for this project are exposed to temperatures higher than those encountered in the ocean and the temperature limits derived from ocean-type populations are likely appropriate.

Table 2.4: Thermal limits for Chinook salmon determined from laboratory and field experiments.

Optimum rearing temperature (ºC)	Upper zero growth limit (ºC)	Upper thermal limit (ºC)	Setting/method	Source
14.8	21.4	25.1	Lab	Brett et al. (1982)
	19.1		Calculated	Armour (1991)
		23.0	Field	Baker et al. (1995)
		25.5	Field	Burck (1993)
		26.7	Lab	Snyder and Blahm (1971)

2.4.5.2.2 Coho

Temperature criteria for coho salmon are summarized in Table 2.5. Optimum rearing temperatures for coho have been reported between 9.4°C and 14.4°C (Beschta et al., 1987, as cited in Hicks, 2000). Based on his review of current literature, Hicks (2000) stated that optimum rearing temperatures for juvenile coho are between 12°C and 15°C. There is considerable variation in the preferred values for juvenile rearing reported in literature. Konecki et al. (1995b) observed it to be between 10°C and 12°C while Bell (1986, as cited in Hicks, 2000) found them to prefer cooler temperatures between 4.4°C and 9.4°C. This difference may be explained by the fact that temperature preference in coho has been demonstrated to vary between parental stocks (Konecki et al., 1995b).

Maximum growth of coho fed unlimited rations occurred at a constant temperature of 17°C (Shelbourn, 1980, as cited in Hicks, 2000). Everson (1973) (as cited in Richter & Kolmes, 2005) imposed a fluctuating temperature regime where maximum growth occurred between 12.1°C and 20.8°C (median 16.5°C). Martin et al. (1984) found that coho salmon growth was negatively impacted by high water temperatures (over 25°C) and lack of cover in a drainage affected by the eruption of Mount St. Helens. Active feeding ceased at 20.3°C (Brett, 1952; Reiser & Bjornn, 1979).

The upper thermal limits of coho as determined from literature range from 23.8°C to 29.2°C (Table 2.5). Juvenile coho mortality was correlated with large daily temperature fluctuations and high water temperatures in excess of 25°C in a study by Martin et al. (1984). They concluded that high water temperatures were the single most important factor limiting coho production. Hicks (2000) concluded that to prevent mortality of juvenile and adult coho maximum temperatures should remain below 21°C.

Table 2.5: Thermal limits for coho salmon determined from laboratory and field experiments.

Optimum rearing temperature (ºC)	Upper zero growth limit (ºC)	Upper thermal limit (ºC)	Setting/ method	Source
12.0 - 14.0	20.3°C	25.0	Laboratory	Brett (1952)
11.8 – 14.6		25.8	Unknown	Bell (1973) (as cited in Reiser & Bjornn, 1979)
		27.7 - 29.6	Laboratory	Becker and Genoway (1979)
		23.8 - 24.4	Laboratory	McGeer et al. (1991)
		27.6 - 29.2	Field	Konecki et al. (1995a)
		25.0	Laboratory	DeHart (1975)

2.4.5.2.3 Rainbow Trout

Temperature criteria for rainbow trout are summarized in Table 2.6. Various optimal temperatures for rainbow trout have been cited in literature. These range from 10°C to 16.7°C (Piper et al., 1982, as cited in Hicks, 2000) at the lower end of the temperature spectrum to 12°C to 19°C (Taylor & Barton, 1992, as cited in Hicks, 2000). If fish are fed a satiation diet, optimal temperatures for growth are even higher at 15°C to 21°C (McCauley & Pond, 1971). Hicks (2000) suggests that the most commonly cited optimum temperatures for juvenile and adult rainbow trout lie between 13°C and 16°C. In a naturally fluctuating temperature regime, the 7-day average of the daily maximum temperatures should not exceed 15°C to 18°C.

Hokanson et al. (1977) determined that the greatest growth rates in rainbow given unlimited food rations were produced at a constant temperature of 17.2°C. Growth was accelerated at mean temperatures below 17.2°C and inhibited by mean temperatures above this optimum when temperatures were allowed to fluctuate. Their results indicated that rainbow trout acclimated to some value between the mean and the maximum daily temperatures. Similar values were obtained in a study by Dockray et al. (1996) where temperature fluctuations were found to be beneficial to growth up to a maximum of 18°C when it started to inhibit growth. Other researchers found the final preferred temperature for rainbow trout to be much lower around 13°C (Ferguson, 1958). Feeding has been observed to cease at high temperatures in other juvenile salmonids but Linton et al. (1997) found that rainbow trout fed to satiation continued to feed near their thermal maximum; however, a reduction in food intake occurred above 22°C.

Numerous studies on the upper thermal limit for rainbow trout have been conducted (Table 2.6). The limit lies between 24°C and 29.4°C and may vary with oxygen concentration, fish size, and acclimation temperature. Under natural acclimation temperatures, lethal temperatures are approximately 25°C to 26°C (Bidgood & Berst, 1969; Hokanson et al., 1977). When acclimated at high temperatures, 27°C

resulted in high or complete mortality in less than 24 hours (Charlon et al., 1970, as cited in McCullough, 1999), and temperatures of 29°C to 30°C resulted in 50% mortality in periods of one to two hours (Kaya, 1978).

Unlike coho and Chinook salmon, rainbow trout are non-anadromous and spend their adult life in freshwater. Therefore, a review of thermal limits for this species must include adults. McCauley and Huggins (1979) noted that temperature tolerances and preferences of adult rainbow trout are generally lower than those of juveniles of the same species. Adult salmonids generally appear more sensitive to temperature extremes than juveniles (McCullough, 1999). High temperatures have demonstrated negative effects on egg quality and survival in the body cavity of female rainbow trout (Billard & Gillet, 1981, as cited in Bonnet et al., 2003).

Table 2.6: Thermal limits for rainbow trout determined from laboratory and field experiments.

Optimum rearing temperature (ºC)	Upper zero growth limit (ºC)	Upper thermal limit (ºC)	Setting/ method	Source
17.2	23	25.6	Laboratory	Hokanson et al. (1977)
		26.4	Laboratory	Charlon et al. (1970, as cited in McCullough, 1999)
		25.0 – 26.0	Laboratory	Bidgood and Berst (1969)
		25.0	Laboratory	Cherry et al. (1977)
		26.0	Laboratory	Stauffer et al. (1984)
		25.9	Laboratory	Threader and Houston (1983)
		26.7	Laboratory	Alabaster (1963)
		24	Laboratory	Black (1953)
		26.2	Laboratory	Kaya (1978)
		26.3	Laboratory	Grande and Andersen (1991)
		29.4	Laboratory	Lee and Rinne (1980)
		25	Field	Bowlby and Roff (1986)
		25.6	Field	Barton et al. (1985)
		24.2 – 26.3	Field	Binns and Eiserman (1979)
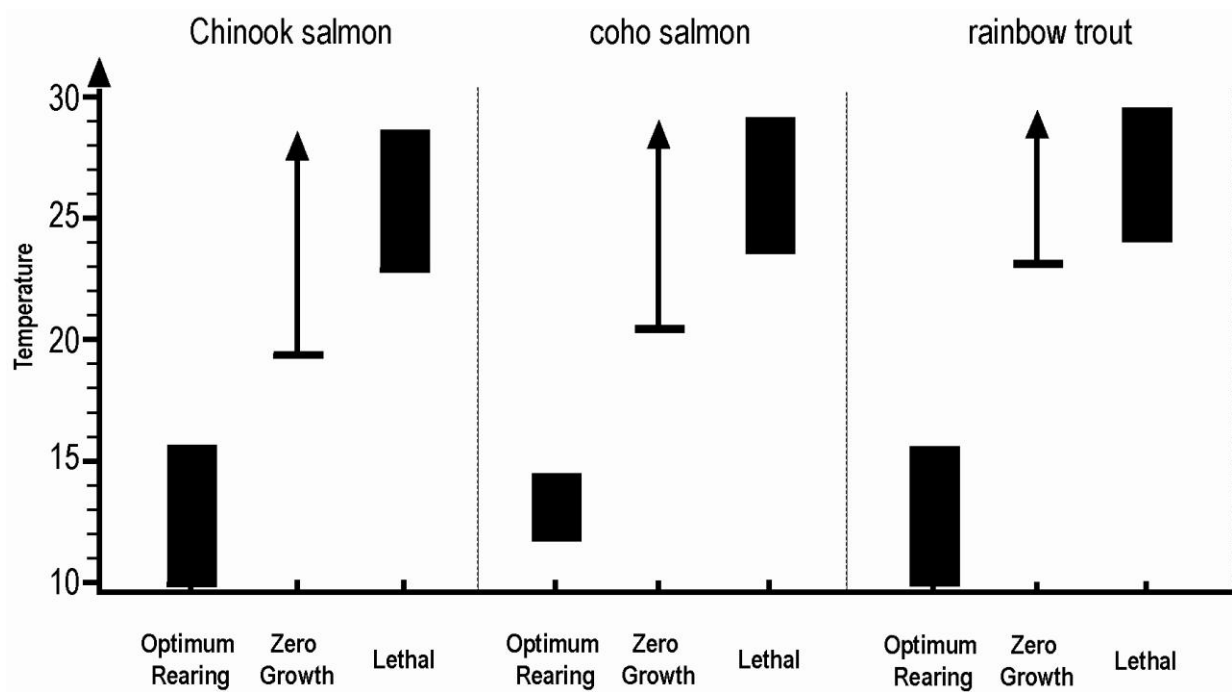
Figure 2.6: Summary of salmonid temperature thresholds from literature.

2.4.5.3 Salmon Distribution and Temperature

Most studies on thermal limits of salmonids have been conducted in the laboratory where controlled experiments are possible. However, determining temperature limits in the natural environment is invaluable because of their implications for the distribution and survival of wild salmonid populations. Researchers frequently derive temperature limits in the field by observing the distribution of fish in relation to water temperatures.

In the field, the interaction of various habitat factors can confound the effects of temperature alone, making it difficult to establish thermal limits. Despite these confounding factors, field studies provide valuable information on temperatures salmonids can tolerate under a wide range of other stressors (McCullough, 1999). They are important for determining actual habitat use by salmonids under a range of thermal conditions. This allows researchers to draw conclusions on salmonid habitat availability and carrying capacities of rivers and streams.

Studying fish distribution in the field commonly involves the collection of presence / absence data (Dunham et al., 2003) or fish density data (de la Hoz Franco & Budy, 2005). Although a range of habitat factors have been studied, temperature has frequently been shown to be related to the distribution of salmonids. Buisson et al. (2008) compared the effects of temperature versus other physical factors (e.g., stream slope, width, depth) on the distribution of 28 fish species. They found that some fish species

have a strong thermal requirement (Atlantic salmon) and are mainly sensitive to temperature and not to physical habitat variables. In a large scale study on riparian land use, environmental parameters, and trout populations in southern Ontario, Barton et al. (1985) concluded that the only variable that consistently distinguished between streams with and without trout was maximum weekly water temperature (limit 22°C). McRae and Diana (2005) determined that water temperature was the most important variable correlated to juvenile brook trout density but not brown trout density, which was instead highly correlated with percentages of gravel substrate and emergent vegetation.

Several studies have been completed on the distribution of Chinook and coho salmon in relation to water temperature. Burck et al. (1980) reported that Chinook juveniles were not found in any areas on the John Day River, Oregon, where mean weekly water temperatures were over 21°C to 22°C. Similar results were obtained by Lindsay et al. (1986) who found no juveniles in reaches where mean weekly temperatures exceeded 22.8°C. A detailed study completed on the Sixes River in Oregon found juvenile Chinook to be absent in reaches where daily maximum temperatures exceeded 23°C (Frissell et al., 1992, as cited in McCullough, 1999). The same study found that coho juveniles rarely occurred where daily maximum temperatures exceeded 21°C. Rainbow trout appeared to be slightly more tolerant of warm temperatures and were present in stream reaches up to daily maximum temperatures of 24°C.

Based on his extensive review of thermal effects on salmonids, McCullough (1999) summarizes that the extent of Chinook, coho and rainbow trout in streams is limited by mean daily water temperatures of 20°C and maximum daily water temperature of 22°C to 24°C, at which point biomass approaches zero. He concludes that these upper temperature limits to field distribution are a good conservative index beyond which the species are expected to be completely absent. Dunham et al. (2001) cautions that the observation of fish should always be interpreted in a probabilistic context, as occasionally studies report the presence of salmonids beyond these temperature limits.

2.4.5.4 Thermal Refugia

Salmonids have no anatomical or physiological means of regulating their body temperature but they demonstrate behavioral changes to avoid temperature stress. Behavioral thermoregulation allows fish to escape stressful water temperatures temporarily or permanently. On a large scale, salmonids may permanently vacate areas where water temperatures move outside their thermal tolerances. On a smaller scale, fish may move into "thermal refugia" when ambient stream temperatures become stressful. Movements to these refugia can range from less than a meter to several kilometers.

Movements to thermal refuges are generally temporary in nature with a return to the previously vacated habitat when temperatures become tolerable again.

Cold water thermal refugia comprise any areas accessible to fish that provide water temperatures within the tolerance range of the species when ambient stream temperatures elsewhere exceed the upper tolerance limits. These may include thermally stratified deep pools (Matthews & Berg, 1997; Nielsen et al., 1994), small forested headwater streams (Curry et al., 1997), cooler tributaries (Kaya et al., 1977; Kaeding, 1996; Goniea et al., 2006), and groundwater inflows (Nielsen et al., 1994; Li et al., 1994).

Apart from directly observing the movement of fish into thermal refugia, the utilization of cool water areas by fish has been demonstrated by means of temperature-sensitive radio transmitters that are implanted into individual fish. Such studies have shown that the body temperature of salmonids is frequently several degrees colder than main river temperatures, particularly when water temperatures are moving beyond the tolerance levels of the species (Berman & Quinn, 1991; Baird & Krueger, 2003; Clabough et al., 2006). This indicates that fish seek out cool areas to avoid high stream temperatures.

Nielsen et al. (1994) found that rainbow trout moved into cool-water refugia for several hours when stream temperatures exceeded 23°C. Although a small number of such studies exist, it is unclear at which temperatures the need for survival overrides the need of salmonids to maintain their territory and movement into refugia takes place. Specific thresholds likely depend on a multitude of factors such as species, stock, age, size, habitat quality, competition, predation, and the presence of various other stressors.

Groundwater inflows to streams have been considered for the potential to provide moderate water temperatures and thermal refugia for cold water fish. Several studies have reported salmonids using localized pockets of cold water created by groundwater inflows. For instance, steelhead moved into cold water pools created by the inflow of tributaries, intergravel flow and groundwater seepage when main flow temperatures reached 23°C to 28°C (Nielsen et al., 1994). Matthews and Berg (1997) noted that rainbow trout congregated in the coldest areas of a thermally stratified pool fed by groundwater seeps during periods of high ambient stream temperatures up to 28.9°C.

On a reach scale, Olsen and Young (2009) investigated whether groundwater inflow would provide areas of moderated temperatures for brown trout. They found that groundwater inflows had a small effect on water temperatures at the reach scale and were not likely to influence brown trout growth. Several cold water patches (2°C to 3°C colder than the main flow) in groundwater inflow areas were not utilized by

trout. However, trout were observed to congregate in the cold-water plume from a spring which stayed approximately 7°C colder than the maximum temperatures recorded in the main stream (23.8°C).

It is unclear whether the existence of cold water patches in high temperature streams is sufficient to sustain viable salmonid populations. Ebersole et al. (2001) investigated rainbow trout abundance in relation to thermal refugia use in a variety of streams with varying environmental conditions. They did report the use of groundwater-fed cold water patches (3°C to 8°C below ambient stream temperatures) during sustained periods of temperatures above 22°C, but failed to find a significant relationship between the frequency of cold water patches and trout density.

The importance of observing other habitat variables besides temperature is demonstrated in a study by Tobias (2006), who found that although steelhead preferentially used areas with spring-fed groundwater inflows, the most likely explanation was better habitat characteristics and not water temperature.

2.4.5.5 Relating Salmon and Temperature in the Field

Researchers have used a multitude of statistical methods to establish meaningful relationships between fish habitat variables and salmonid distributions. A very common approach is the use of regression analysis, which allows researchers to establish quantitative linkages and predict salmonid distribution based on environmental factors. Some frequently used types of regression include simple linear regression (Kaeding, 1996; de la Hoz Franco & Budy, 2005; McRae & Diana, 2005), logistic regression (Dunham et al., 2001), generalized linear regression (Clabough et al., 2006; Goniea et al., 2006), and generalized additive models (Buisson et al., 2008).

One of the key assumptions of regression analysis is the independence of individual observations, meaning that one observation cannot be influenced by the value of other observations. Ignoring this assumption can lead to serially correlated residuals. As a result, standard errors underestimate the variability in the data and p-values are artificially low. This means that null hypotheses are rejected more often than they should be. A regression model in which this autocorrelation is not accounted for may conclude that a predictor variable is significant when it actually is not. Hurlbert (1984) investigated numerous ecological field experiments using inferential statistics and cautioned that this type of "pseudoreplication" was widespread. Researchers were considering observations as independent replicates when they were clearly collected from the same experimental unit and were not independent.

Implications for regression analyses of fish distributions are that sampling sites have to be carefully selected to ensure observations are truly independent from one another. Field observations can be correlated in space (i.e., numerous sites on one stream) and time (i.e., numerous observations of the same individual or site). To avoid pseudoreplication, the ideal field study would establish sample sites on a large number of streams, and sample fish distribution at one point in time. Several studies of this type have been undertaken, but they require substantial field effort which is very costly (e.g., Welsh et al., 2001). It is difficult in such a study to find enough replicate streams while keeping other factors as constant as possible. For example, variation in temperature tolerance of salmonids from different stocks has been demonstrated. Therefore, all replicate streams would have to contain salmonids from the same stock while ensuring that physical conditions are identical.

For many researchers, large-scale studies of this kind are not feasible or desirable and many fish distribution studies are conducted on a much smaller scale. Where studies are restricted to a single stream, valuable information can be gained despite the lack of independence. However, the selection of analysis method becomes critical. In single-stream studies, researchers have to increase their number of replicates by collecting several observations from the same experimental unit, such as several observations of the same stream reach over time, and/or several observations in different areas of the same stream (Dunham et al., 2003). As a result, a hierarchical grouping structure exists in the data and none of the observations are truly independent of one another because of temporal and spatial autocorrelation. A series of statistical methods have been developed to deal with non-independent observations.

2.4.5.5.1 Generalized Linear Mixed Models

Generalized Linear Mixed Models (GLMM) account for a lack of independence by incorporating a hierarchical grouping structure of the data into the regression model through random effects. Including a factor like study site as a random effect implies that all observations from a given study site are correlated and more similar to each other than observations from other sites (Zuur et al., 2009). GLMMs can incorporate such random effects and can also accommodate non-Gaussian data such as counts or presence/absence data. Poisson regression is frequently used for count data and logistic regression is frequently used for presence/absence data (Zuur et al., 2009). Where multiple random effects exist that act independently on individuals (e.g., temporal and spatial blocks in the same design) the term "crossed" random effects is used (Bolker et al., 2008).

The univariate regression function for a Poisson GLMM with two random effects (site and week) is (Zuur et al., 2009):

$$log(\mu_{ij}) = \alpha + \beta_1 \times x_{ij} + a_i + b_j \qquad (2.3)$$

where μ_{ij}= fitted count value at site i in week j

α = intercept

β_1 = regression slope

x_1 = explanatory variable

a_i= random site effect, $a_i \sim N(0, \sigma_a^2)$

b_j=random week effect, $b_j \sim N(0, \sigma_b^2)$

The univariate regression function for the equivalent logistic GLMM is (Zuur et al., 2009):

$$log\left(\frac{p_{ij}}{1-p_{ij}}\right) = \alpha + \beta_1 \times x_{ij} + a_i + b_j \qquad (2.4)$$

where p_{ij}= probability of salmonid presence at site i in week j

According to Zuur et al. (2009), GLMMs are an area of active statistical research and model selection resembles an art rather than a science. Models can be selected either by forward or backward selection. Forward selection begins with models that include one explanatory variable and the random effects. Additional explanatory variables are included one at a time and retained if significant ($p < 0.05$). Backward selection starts with a full model including all explanatory variables and the random effects. The least significant explanatory variable (based on p-value) is removed and the model is then refit until only significant explanatory variables are left. This approach often results in a different model than the forward selection approach (Zuur et al., 2009). Backward selection is not appropriate when high multicollinearity exists between fixed effects.

The Akaike Information Criterion (AIC) can be used to compare GLMM models (Akaike, 1974). The model that best explains the data with minimal explanatory variables has the lowest AIC. This comparison is legitimate only when all models have the same random effects structure (Zuur et al., 2009).

A word of caution is necessary when discussing p-values resulting from the GLMM modeling process. The calculation of p-values and associated determination of the significance of explanatory variables is difficult when fitting GLMMs with a Poisson distribution. This is due to the uncertainty of the GLMM models used when it comes to dealing with overdispersion. By definition, the variance of a Poisson distribution is equal to its mean. Overdispersion occurs when the variance in the data is not equal to the mean (Zuur et al., 2009). Overdispersion does not change the regression slope estimates, but it does

affect the standard errors and p-values and, subsequently, which variables are included as significant or not.

Overdispersion can be accounted for via the dispersion parameter. This parameter can be calculated for each model, and the standard errors of the regression estimates for each fixed effect are then multiplied by the square root of the dispersion parameter. This increases the standard errors (and subsequently, the p-values) and the analyst can make valid inferences regarding the true significance of the fixed effects (Zuur et al., 2009). Calculation of the dispersion parameter is based on model deviance and residual degrees of freedom. The difficulty in determining dispersion parameters for GLMMs lies in a general uncertainty associated with the proper estimation of the residual degrees of freedom for random effects (Bolker et al. 2008).

2.4.5.5.2 Generalized Estimating Equations

The correlation structure implied in the GLMMs discussed above assumes that observations from one group are correlated equally to all other observations from the same group. Where data are collected over time it would be reasonable to expect that subsequent observations may be more strongly related to another than observations farther apart. Similarly, where spatial correlation exists, it would be expected that observations from sites closer to each other are more similar than those from sites farther apart.

Generalized Estimating Equations (GEE) are Generalized Linear regression models that can fit more specific correlation structures to the data. An autoregressive correlation structure implies that the correlation between observations is a function of the distance (either spatial or temporal) between observations, where observations that are closer together are more correlated than observations that are farther apart (Zuur et al., 2009).

Which correlation structure is ultimately used depends on the exact circumstances of each study and the nature of data collection. The key point is that it is critical to address correlation between observations to ensure that the assumptions underlying regression analysis are not violated, although many published studies that use regression do not specifically deal with autocorrelation.

3.0 METHODS

3.1 Study Area

Fortune Creek is a regulated system situated in the North Okanagan near Armstrong, BC (Figure 3.1). The stream is approximately 21 km long and is part of the Fraser River drainage basin (Freshwater Fisheries Society of BC, 2010). The Fortune Creek drainage area is approximately 151 km^2. The creek flows north from its headwaters near Silver Star Mountain (1,488 m) and eventually joins the Shuswap River near Enderby, BC (353 m). The average annual precipitation measured at the Armstrong North climate station from 1971 to 2000 was 488 mm (Figure 3.1). Average monthly temperatures varied from a daily minimum of -8.6°C in January to a daily maximum of 26.7°C in July and August. However, extreme temperatures reached from -37°C to 39°C (Environment Canada, 2009a).

Fortune Creek has two reservoirs at its headwaters and initially flows through mainly undisturbed forest. The gradient on the mountainside is steep and the stream flows through deeply incised gullies and waterfalls. A sand and gravel alluvial fan deposit up to 52 m thick is situated where the stream enters the valley from the bedrock mountainside (Monahan, 2006).

At the upper end of the fan deposit, the creek is deeply shaded by coniferous forest and the streambed materials consist of gravel, cobbles and boulders. The gradient is approximately 5%. The lower portion of the fan area has similar cobble and boulder sized stream materials, but lies on the same elevation as the main valley floor. The stream in this lower fan area has been extensively channelized by earlier bulldozing and bank building activity to straighten the stream, prevent flooding and armour the stream sides.

The portions of Fortune Creek farther downstream in the valley bottom floodplain have been extensively modified over the past century by surrounding agricultural land use activities (40% of the drainage area) (Rood & Hamilton, 1995). The dominant physical alterations have been the channelization, dredging and dyking of large sections of the creek for flood control purposes. These modifications have led to a streambed that is substantially lower than the floodplain in most areas. The valley bottom reaches are now characterized by high sediment loads, muddy bottoms composed of sand, silt and clay, little or no vegetative cover, channel incision, water stagnation and a general lack of habitat complexity. Some restoration work has been completed in the past between sites 2 and 5,

including tree planting, installation of cattle crossings, the addition of various flow deflectors (logs and boulders), and instream boulder placement to enhance fish habitat.
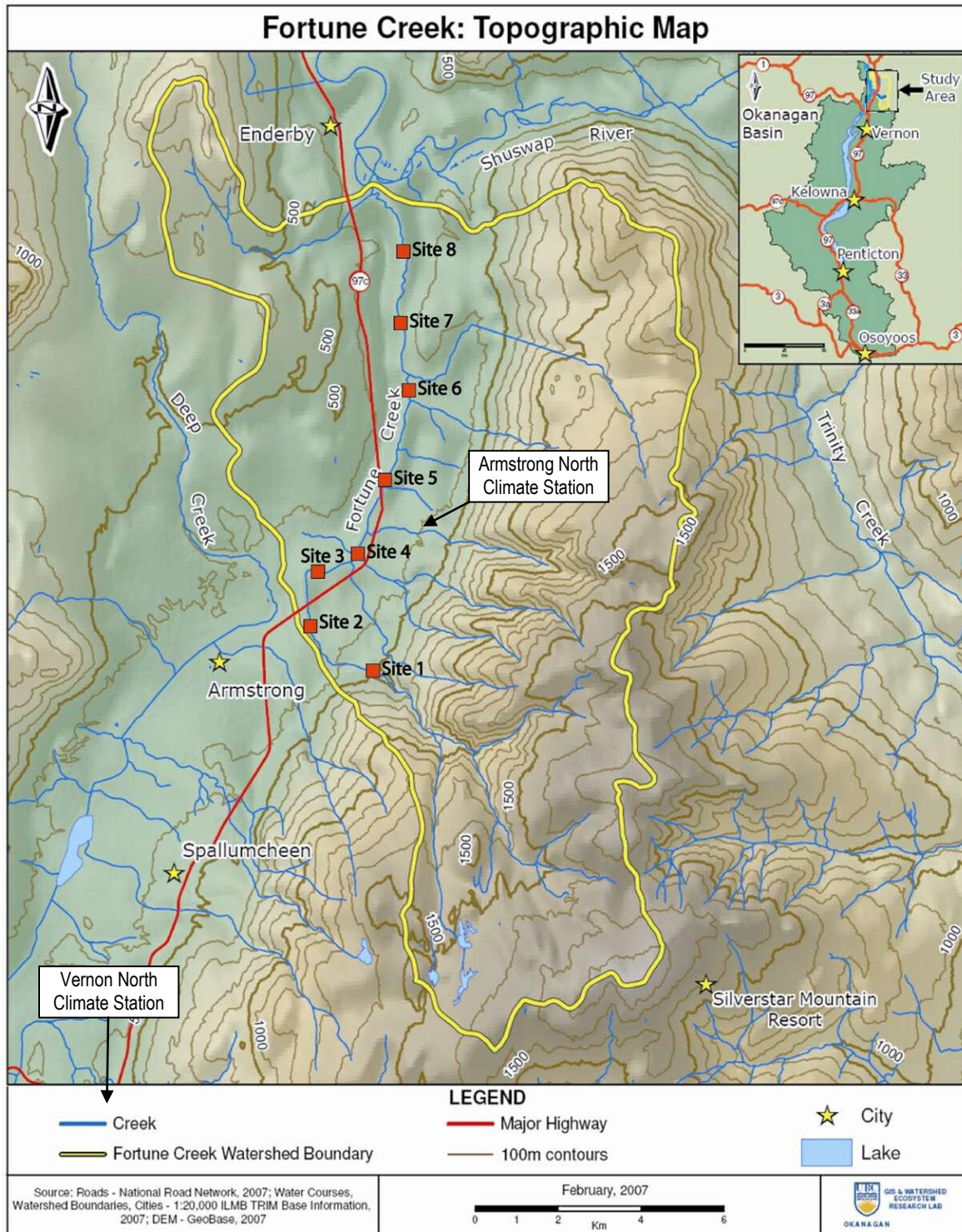
Figure 3.1: Overview map of the study area (modified from Seebacher et al., 2007).

Stream width varies from approximately 2 m at the top of the alluvial fan to 10 m near the confluence with the Shuswap River. Between site 1 and 2 (Figure 3.1) a stream reach of several hundred meters in

length dries out periodically in both summer and winter. Anecdotal information suggests that rapid water level fluctuations in this section are common (Seebacher et al., 2007). Water re-emerges downstream as the creek enters the valley-bottom floodplain. However, chemical and isotopic analysis and streamflow measurements from the North Okanagan Groundwater Characterization and Assessment Project indicate that some of the water does not re-emerge to Fortune Creek and instead enters the regional groundwater flow regime (Ping et al., 2010).

The uppermost valley bottom surficial deposits in the area consist of glaciolacustrine silts and clays. Beneath these are two pre-glacial confined aquifers (Monahan, 2006). Artesian wells exist in the upper aquifer (Spallumcheen A), indicating upward groundwater flow in the valley bottom. North of site 6, the aquifers in the Fortune Creek watershed flow northwards to the Shuswap drainage but south of site 6, they flow southwards into the adjacent Deep Creek watershed (Ping et al., 2010).

The City of Armstrong operates the two reservoirs in the headwaters of Fortune Creek. South Silver Star Lake reservoir was constructed in 1970 and North Silver Star Lake reservoir was constructed in 1992 (Summit Environmental Consultants Ltd., 2006). South Silver Star Lake water is usually retained for fire protection. Water from North Silver Star Lake is used for domestic drinking water by the City of Armstrong as well as providing summertime supply to several industrial users and to other water districts in the valley. Any excess water at the reservoirs bypasses the dams via an overflow channel and enters Fortune Creek.

The City of Armstrong has installed a water intake facility on Fortune Creek approximately 8 km downstream of the north reservoir, just before Fortune Creek leaves the bedrock hillside and enters the valley proper (Figure 3.2). During the drier parts of the year water is released via manually controllable valves at the base of the dam at the north reservoir and flows downstream to the intake facility where some of it is withdrawn. During low flow periods, up to 100% of the streamflow is diverted at this point and routed into two holding tanks situated approximately 500 m downstream. Excess water from the holding tanks is allowed to overflow back into the creek. Release of water from the upland reservoirs commences around the middle to end of June, and the valves are typically closed around the end of September or the beginning of October. Adjustments to the water released from the reservoirs are made every few weeks. Manual adjustments to the stream intake diversions are made daily. The Fortune Creek water supply provides water for approximately 4,500 users.
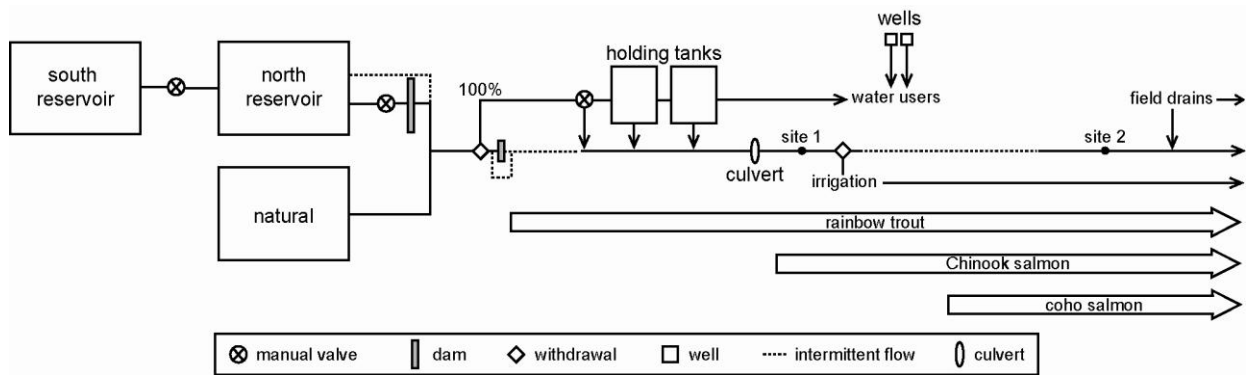
Figure 3.2: Schematic drawing of the processes and facilities located in the upper reaches of Fortune Creek.

The Fortune Creek hydrograph is typical of interior streams with snowmelt-driven high peak flows in late spring and early summer (Figure 3.3). Discharges show a distinct peak during freshet from late April to early July followed by much lower flows for the remainder of the year. Naturalized average monthly flows range from a low of 0.08 m^3/s to a high of 3.42 m^3/s. Licensed water use reaches from approximately 50% of estimated naturalized flows in the winter to approximately 65% in August (Seebacher et al., 2007).
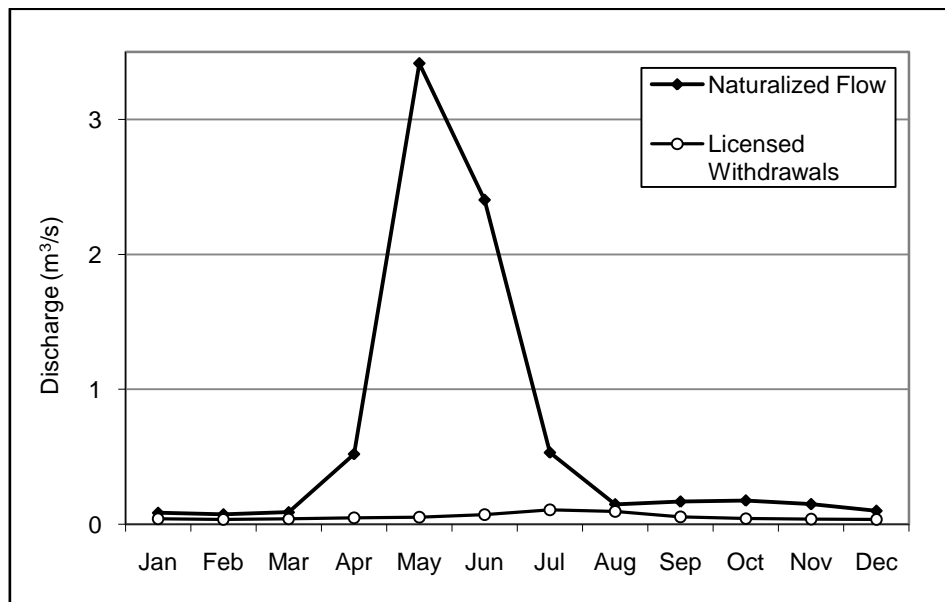
Figure 3.3: Naturalized hydrograph of mean monthly flows in Fortune Creek at Stepney Road (1960-1984) and licensed withdrawals (Seebacher et al., 2007).

Salmonid species inhabiting Fortune Creek include rainbow trout, Chinook salmon and coho salmon. Juvenile coho salmon have been documented in Fortune Creek in various fisheries inventories over the last two decades (Ross, 1994; Fortune Creek Fisheries Crew Spallumcheen Band, 1993; Seebacher et al.,

2007). Coho egg release programs were undertaken by the local fish and game club in the late 1980s and mid-1990s. However, the last report of coho spawning in the creek was in 1986 (Kurtz, 1986).

The coho salmon found in Fortune Creek are Interior Fraser Coho (Interior Fraser Coho Recovery Team, 2006). Interior Fraser Coho generally return from the ocean to spawn in freshwater in their third year. While many salmonids exhibit strong fidelity to their natal streams for spawning, Interior Fraser Coho seem to have lower fidelity to their spawning streams than coastal coho (Interior Fraser Coho Recovery Team, 2006). Escapement numbers for Interior Fraser Coho, monitored by the Department of Fisheries and Oceans Canada (DFO) over the last 30 years, have declined by about 75% (Interior Fraser Coho Recovery Team, 2006). The Interior Fraser River coho salmon is designated as Endangered by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) and is under consideration for addition to Schedule 1 of the Species at Risk Act (SARA) (COSEWIC, 2002).

Chinook salmon have been reported in Fortune Creek much less frequently and were only caught sporadically at very low numbers during surveys completed in 1986 and 1994 (Kurtz, 1986; Ross, 1994; Fortune Creek Fisheries Crew Spallumcheen Band, 1993). The Chinook salmon inhabiting Fortune Creek likely originate from spawning populations in the Shuswap River. Escapement numbers for the Lower and Middle Shuswap River have increased steadily since the mid-1970s (DFO, 2010a; 2010b) and the Fraser River Chinook population has not been designated by COSEWIC. The nearby Okanagan basin population is listed as Threatened by COSEWIC and is pending addition to Schedule 1 of SARA.

Other species documented in Fortune Creek include largescale sucker (*Catostomus macrocheilus* Girard), redside shiner and sculpin (*Cottus sp.*). Fortune Creek has been listed as one of the top ten most sensitive salmon streams in the South Thompson watershed in terms of water demand and summer and winter low flows (Rood & Hamilton, 1995).

Over the past decade, DFO has received reports of late summer and early fall fish mortality in Fortune Creek (Seebacher et al., 2007). It is unknown whether the fish perished due to high water temperatures, stranding resulting from rapid water level fluctuations, low oxygen conditions, or other causes. Discharge volumes in Fortune Creek are currently too low to sustain spawning populations of coho, Chinook and rainbow trout for most parts of the year (Seebacher et al., 2007). However, the report indicated that a better understanding of fish flow requirements in Fortune Creek is needed.

3.1.1 Study Site Selection

The entire accessible 15 km length of Fortune Creek was walked for an initial assessment in September of 2007 and March 2008. During this assessment, streamflow was measured at sixteen intermediate locations to identify gaining and losing portions of the creek. In addition, manual temperature measurements were taken at each of the sixteen locations. Additional manual surveys of the entire creek were conducted in March 2008 with a handheld temperature probe. The purpose was to locate areas of elevated temperatures that may be indicative of groundwater inflows.

The main observations during the stream walk in September 2007 were high water temperatures (>25°C), low streamflows (0.0001 m^3/s) and very slow flow velocities throughout the valley-bottom reaches of Fortune Creek. The intermittent stream section between site 1 and 2 was dry at the time of the survey and numerous dead juvenile salmonids were noted in this section, presumably due to stranding (Figure 3.4). Numerous field drains and discharge pipes flowed into Fortune Creek between site 2 and site 6 (e.g., Figure 3.5). Chemical analysis on some of the field drains conducted as part of the scoping study revealed conductivity values up to 939 µS/cm (Seebacher et al., 2007).

Surface runoff from adjacent cattle operations entered the stream in several locations. Where cattle fencing was not installed, substantial streambank degradation from cattle access was noted (Figure 3.6). Abundant instream vegetation growth and algae, suggestive of nutrient loading and eutrophication, were observed in the near stagnant waters of the lower reaches of the creek below site 4 (Figure 3.7) from July to November 2008.

Figure 3.4: Stranded juvenile salmonids in Fortune Creek.

Figure 3.5: Field drains entering Fortune Creek.

Figure 3.6: Bank degradation by cattle.   **Figure 3.7: Excessive algal growth noted near site 7.**

The manual temperature surveys of Fortune Creek conducted in March 2008 identified one stream section suspected to have significant groundwater discharge. This section extended between site 3 and site 4. Groundwater discharge was indicated by stream temperatures elevated by several degrees over the remainder of the creek.

The observations from the September 2007 and March 2008 surveys aided in the selection of the final study sites. Based on these surveys, channel morphology and hydrology, eight study reaches approximately 30 m in length were established (Figure 3.8). This was the maximum number of sites that could be sampled on a weekly basis. Site 1 was the most upstream site and all other sites were numbered in a downstream direction to site 8 near the confluence of Fortune Creek with the Shuswap River.

Detailed site descriptions are presented in Table 3.1. Site 1 was situated at the top end of the alluvial fan and was typical of a step-pool headwater stream. It was situated in a densely shaded section of stream. The streambed was composed of a heterogeneous mix of cobbles, gravel and sand, in addition to large boulders. The study reach contained several pools and some woody debris. Average and maximum flow velocities between July and August were 0.4 m/s and 1.1 m/s, respectively.

Site 2 was situated in a riffle-pool sequence in a transitional area between the lower reaches of the alluvial fan and the floodplain. The site was partially shaded by deciduous vegetation. Substrate was composed of cobbles, gravel and sand. The site contained two log-jams, one large pool and undercut banks. Average and maximum flow velocities between July and August were 0.3 m/s and 0.6 m/s, respectively. Just downstream of site 2, an artesian well flows into the creek throughout the fall, winter and spring. During the summer, the well is capped and used for irrigation.

The remainder of the sites were located in the flat valley bottom. Substrate at site 3 was composed of gravel, sand and clay. Substrate at all remaining downstream sites was composed of sand, silt and clay.

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Valley-bottom sites ranged from completely unshaded (sites 3, 5, 7 and 8) to partially shaded (less than 50%, sites 4 and 6). Flow velocities at sites 3 and 4 were on average less than 0.2 m/s between July and August and less than 0.1 m/s at the remaining valley bottom sites. The water was nearly stagnant at sites 6 to 8 for long periods during the summer.
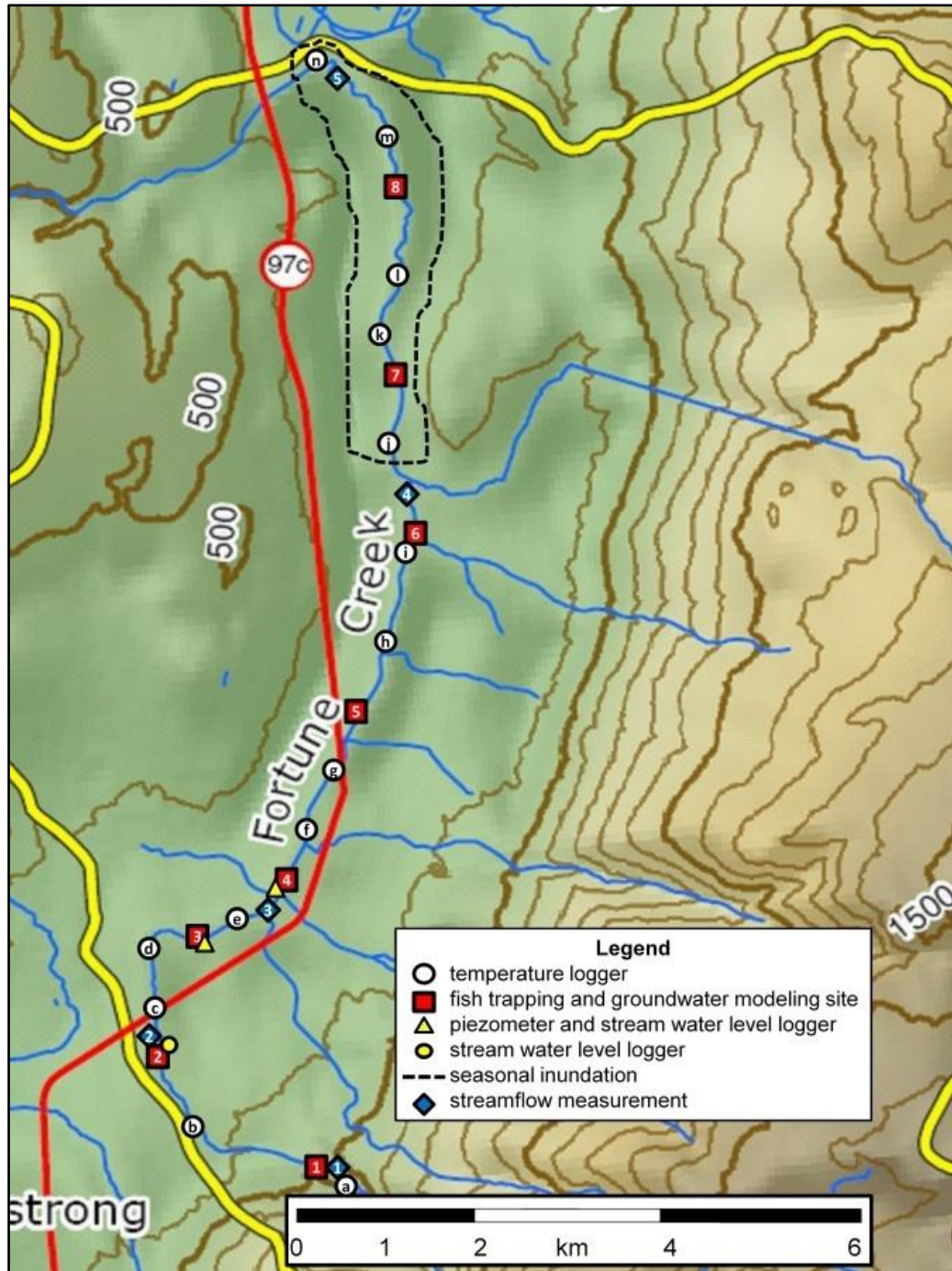
Figure 3.8: Location of study sites and instrumentation deployment on Fortune Creek (modified from Seebacher et al., 2007).

Table 3.1: Description of study sites on Fortune Creek.

Site	Location (UTM)	Elevation (m)	Gradient (%)
1	347440E 5589764N	428	5.77

Site	Location (UTM)	Elevation (m)	Gradient (%)
2	346192E 5591251N	369	0.26

Site	Location (UTM)	Elevation (m)	Gradient (%)
3	346634E 5592122N	358	< 0.5

Site	Location (UTM)	Elevation (m)	Gradient (%)
4	347471E 5592360N	356	< 0.5

Site	Location (UTM)	Elevation (m)	Gradient (%)
5	348356E 5593766N	356	< 0.5

Site	Location (UTM)	Elevation (m)	Gradient (%)
6	349275E 5595908N	354	< 0.5

Site	Location (UTM)	Elevation (m)	Gradient (%)	Site	Location (UTM)	Elevation (m)	Gradient (%)
7	349351E 5597670N	354	< 0.5	8	349634E 5599354N	354	< 0.5
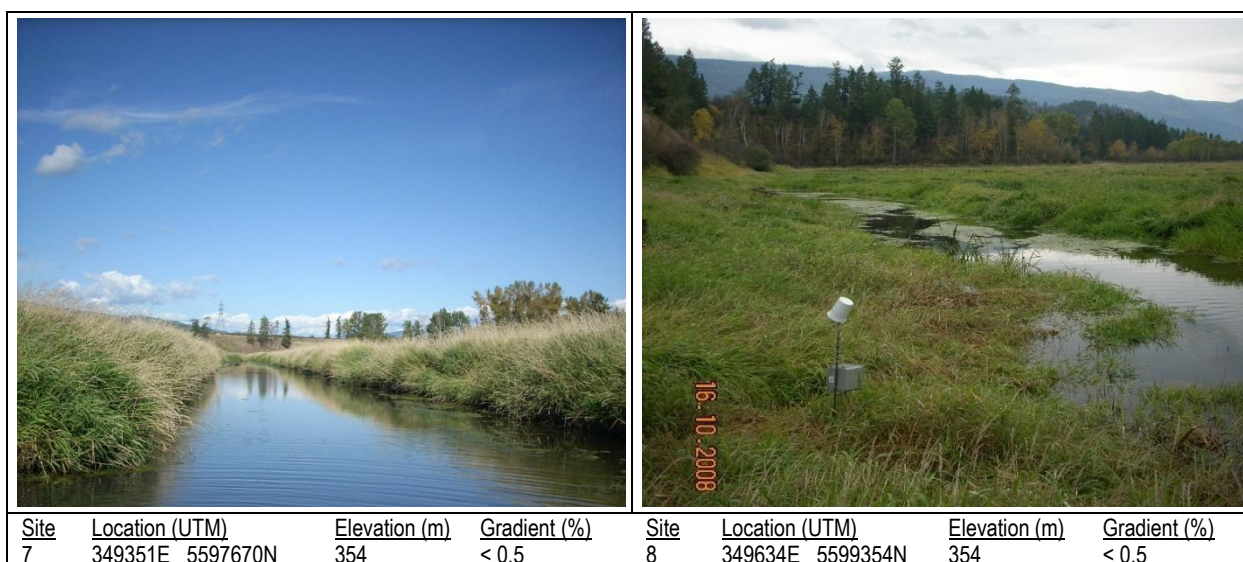
3.2   Field Data Collection

3.2.1 Streamflow

Streamflow in Fortune Creek has been measured at five sites (Figure 3.8) in a joint effort with the North Okanagan Groundwater Characterization and Assessment since early 2007 (Ping et al., 2010). Sampling during 2008 and 2009 was conducted biweekly during the summer months (April to October) and monthly during the remainder of the year. Discharge was measured using a Swoffer Model 3000 Flowmeter (Swoffer Instruments, Inc.) and a FlowTracker Handheld Acoustic Doppler Velocimeter and wading rod (Sontek/YSI).

3.2.2 Temperature

At each of the eight sites, temperature was continuously measured from June 2008 to April 2009 using thermistor sensors deployed in the air, in the water column and at 10 cm and 50 cm depth in the streambed (Figure 3.8). HOBO U12 4 Channel loggers (Onset Corporation) were used to record temperature at a 30-minute interval. Fourteen additional data loggers (Hobo pendant loggers, Onset Corporation), numbered a to n, were deployed along the full length of the creek to monitor stream temperature at intermediate locations.

All temperature loggers were deployed within 150 mm lengths of 52 mm diameter white PVC piping suspended in the mid water column. This provided shielding from direct solar input and contact with the

streambed as described by Quilty and Moore (2007). Holes were drilled into the pipe to allow for water to circulate freely. Streambed sensors were located by direct burial (10 cm) or by insertion inside a 4 foot section of sealed, water-filled copper piping pounded into the streambed (50 cm). Sensors were anchored in the stream by a piece of rebar pounded into the streambed. Air sensors were attached to a piece of rebar and shaded with opaque white plastic containers with slits for air circulation. The time periods during which temperature was recorded are presented in Table 3.2.

Table 3.2: Water temperature recording periods.

Logger Location (Figure 3.8)	Recording Period	Jun 2008	Jul	Aug	Sep	Oct	Nov	Dec	Jan 2009	Feb	Mar	Apr
a	Jul-2-08 to Apr-24-09											
1	Jul-2-08 to Apr-24-09											
b	Jul-7-08 to Apr-24-09											
2	Jun-27-08 to Apr-24-09											
c	Jun-19-08 to Apr-24-09											
d	Jun-19-08 to Apr-24-09											
3	Jun-26-08 to Apr-24-09											
e	Jun-19-08 to Apr-24-09											
4	Jun-27-08 to Apr-24-09											
f	Jun-17-08 to Jul-15-08											
g	Jun-17-08 to Apr-24-09											
5	Jul-17-08 to Apr-24-09											
h	Jun-17-08 to Apr-24-09											
i	Jun-17-08 to Apr-24-09											
6	Jul-17-08 to Apr-24-09											
j	Jun-12-08 to Apr-24-09											
7	Jul-18-08 to - Apr-24-09											
k	Jun-12-08 to Apr-24-09											
l	Jun-12-08 to Apr-24-09											
8	Jul-18-08 to - Apr-24-09											
m	Jun-12-08 to Apr-24-09											
n	Jun-12-08 to Apr-24-09											

3.2.3 Salmonid Distribution

The timing of field data collection in the spring of 2008 was delayed by high peak flows and a late freshet, resulting in high flow conditions under which trapping and instrument deployment could not be conducted at all sites in a safe manner. To ensure the safety of field crews and fish, trapping was not commenced until early July. Minnow traps were deployed at each of the eight sites once per week from July 2008 to December 2008 (Table 3.3). Trapping was continued at three-week intervals in the spring and early summer of 2009. At each sampling, seven minnow traps were deployed at each site to ensure adequate coverage of the reach. The traps were baited with fresh unsalted salmon roe inserted into perforated film canisters. They were set over night no longer than 24 hours to minimize the risk of

trapping-induced fish mortalities. All trapped fish were identified and tallied up by species. The total number of each fish species was recorded at each site.

Table 3.3: List of dates on which fish trapping was carried out.

	2008		2009
July 1	August 12	September 26	April 29
July 8	August 20	October 2	June 2
July 15	August 27	October 10	July 16
July 23	September 3	October 24	November 3
July 29	September 12	October 31	
August 06	September 19	December 11	

3.2.4 Water Quality

Manual measurements of DO, temperature, pH and conductivity were collected at each of the eight study sites during the fish counting events. Measurements were made at the time the traps were set and the next day when they were retrieved. Measurements were conducted using a Hatch HQ40D Water Quality Meter and LDO101 luminescent dissolved oxygen probe, PHC101 pH probe and CDC401 conductivity probe.

Fish cover was assessed as a physical habitat variable and included any structure that provides protection for fish from predators and floods. This included undercut banks, roots, cobbles and boulders, woody debris, aquatic and terrestrial plants, and deep pools. Cover was assessed at each site by mapping all covered areas in the stream reach and then calculating the percentage covered of the total area of the reach as outlined in Johnston and Slaney (1996). Cover was assessed once during the study period in September 2008.

3.2.5 Tissue Sampling for DNA Analysis

Tissue samples for DNA analysis were collected from Chinook salmon in late July 2008 and mid-June 2009. The sampling was conducted on request from DFO and was intended to determine whether Chinook occupying Fortune Creek are ocean or stream type Chinook. Prior to sample collection, fish were sedated in a bucket of water mixed with the sedative Aquacalm (Metomidate hydrochloride, Syndel Laboratories Ltd.) at an approximate concentration of 1.0 mg/L. Once loss of equilibrium had occurred, a small clip was collected from the caudal fin of each fish. Samples were stored in vials filled with 95% ethanol provided by the DFO molecular genetics lab in Nanaimo. After sampling, fish were held in a bucket of stream water for at least 15 minutes until they fully regained consciousness, at which

point they were released back into the stream. Tissue samples were sent to the DFO molecular genetics lab in Nanaimo for DNA analysis.

3.3   Statistical Analysis

Regression analysis was selected as a method of determining which habitat indicators were related to salmonid distribution in Fortune Creek. A total of 37 predictor variables representing a range of habitat conditions were assessed in regression models (Table 3.4). Several temperature summary values were calculated for the week preceding a fish counting event. These were chosen over in situ temperature measurements recorded manually at the time of sampling because they are more representative of average temperature conditions during each week.

To capture exposure to extreme values not well-represented by these average temperature metrics, the number of hours that stream temperatures exceeded certain temperature thresholds were calculated. The total weekly hours and the maximum weekly continuous hours over each degree Celsius from 15°C to 28°C were calculated to determine whether there was a threshold temperature at which, when exceeded for certain amounts of time, salmonids would be no longer present (Mather et al., 2008).

DO values often differed substantially between the two readings each week. Therefore, two DO variables were assessed: the lower of the two readings, and the average of the two. pH and conductivity values were averaged for each sampling run.

Table 3.4: Predictor variables included in the regression models.

Predictor Variable	Description
Fixed effects	
• absolute weekly maximum temperature	highest water temperature recorded in the 7 days prior to sampling
• average weekly maximum temperature	average of daily maximum water temperatures in the 7 days prior to sampling
• average weekly mean temperature	average of daily mean water temperatures in the 7 days prior to sampling
• average weekly minimum temperature	average of daily minimum water temperatures in the 7 days prior to sampling
• total weekly hours in exceedance of a temperature criterion (15°C to 28°C, 14 variables)	total hours above water temperature threshold in the 7 days prior to sampling
• maximum weekly continuous hours in exceedance of a temperature criterion (15°C to 28°C, 14 variables)	maximum continuous hours above temperature threshold in the 7 days prior to sampling
• cover	fish cover (%)
• pH	average pH during sampling event
• conductivity	average conductivity during sampling event
• minimum dissolved oxygen	lower of the two dissolved oxygen values recorded during sampling event
• average dissolved oxygen	average of the two dissolved oxygen values recorded during sampling event
Random effects	
• site	1 to 8
• week	1 to 8

For modeling salmonid distribution, only data collected from weeks one to eight were included in the statistical models. This captures the period from July 1 to August 20, 2008, which was also the period in which maximum temperatures were recorded in the creek. In addition, data from site 1 were excluded from the analysis of coho distribution, as the streambed gradient was larger than 3% and was considered too steep. The reason for excluding any observations past August from the analysis was that salmonids tended to remain in the territories they had established over the summer (Section 2.4.1). Movement to other areas seemed limited even when water temperatures declined in the fall, and previously vacated stream reaches were not re-populated at that time. The few salmon captured in previously vacated reaches in the fall were smolting and migrating downstream, and were not considered residents in the context of this study.

This does not apply to non-anadromous rainbow trout who occupy the creek year round. However, it was decided to analyze rainbow data from the same time period to enable comparisons among species. Rainbow trout spawned in the vicinity of site 3 and a large number of fry emerged in the middle of the study season in 2008. To prevent this from affecting the results of the statistical analysis, only 1+ year old rainbow were included in the statistical models.

The fish count data was analyzed in two ways. First, count values were used as the response variable in the regression models (i.e., number of fish captured). These regression models used the Poisson link function in a Generalized Linear Mixed Model (GLMM). Second, the counts were translated to simple presence/absence observations and analyzed using logistic regression. Each type of model was analyzed for coho, Chinook, and rainbow trout.

During preliminary data analysis, the Wald–Wolfowitz runs test was used to test for autocorrelation in the raw salmonid presence/absence data. The analysis was performed in the statistical software R (R Development Core Team, 2009) using the library *tseries* and the function *runs.test()* (Trapletti & Hornik, 2009). The results of the runs test indicated that significant ($p < 0.05$) dependencies between sites and between sample weeks were present (Table 3.5) and temporal and spatial correlation needed to be addressed in the analysis.

Table 3.5: Number of sites and weeks with significant ($p < 0.05$) Wald–Wolfowitz runs test.

Species	Number of sites with temporal autocorrelation (out of 8)	Number of weeks with spatial autocorrelation (out of 8)
Chinook	5	2
Coho	4	0
Rainbow	5	4

To control for reduced variance in the data resulting from its grouping structure, site number and week number were incorporated as random effects into the regression models. Including site and week as random effects implies that all observations from a given site (or week) are correlated equally to all other observations from the same site (or week) (Zuur et al., 2009). Site and week were included as crossed random effects (Section 2.4.5.5.1). In addition to the random effects, the predictor variables presented in Table 3.4 were included as fixed effects.

GLMMs were analyzed using the statistical software R and the function *glmer()* in the library *lme4* (Bates et al., 2008). *lme4* is one of the few R packages that allows for crossed random effects. Because of multicollinearity, forward selection was employed including only non-correlated explanatory variables in the same model. Interactions between explanatory variables were also assessed for each model. The AIC was used to compare GLMM models (Akaike, 1974). The objective of this research was not to find the best model for predicting salmonid abundance or presence, but rather to determine which variables have an important influence on salmonid distribution. Therefore, all variables that had a significant effect ($p < 0.05$) on salmonid distribution are presented in the results section regardless of whether they

were included in the final "best" model or not. The AIC values are nonetheless useful to compare model performance.

Data for all three salmonid species analysed in this paper were overdispersed. The *glmer()* function in R fixes the variance at one and thus does not account for overdispersion nor does it calculate dispersion parameters. Although a quasi-Poisson distribution, which is supposed to account for overdispersion, is available in *glmer()*, estimates of standard errors provided by the function were highly questionable at the time of analysis. Instead, dispersion parameters for each fitted model were obtained via the function *glmm.admb()* in the *glmmADMB* package written for R (Otter Research Ltd., 2008). The dispersion parameters were then used to adjust standard errors and p-values in the previously fitted *glmer()* models.

It is not currently possible to specify correlation structures other than the random effects discussed above (all observations equally related) in GLMM models through the *glmer()* function in R. More specific correlation structures can be fit in GEE models through the function *geeglm()* from the library *geepack* in R (Yan & Fine, 2004; Yan, 2002).

GEEs with a first order autoregressive correlation structure (AR1) were fitted to the data using *geeglm()* to assess whether this correlation structure would be better suited to modeling the correlation present in the data. This seemed like a good fit for the data in this research. Unfortunately, the function does not allow for more than one grouping structure as *glmer()* does. Therefore, one has to specify either the temporal correlation or the spatial correlation and cannot incorporate both in the same model. While this was not ideal for the data collected, exploratory data analysis of the raw data autocorrelation coefficients revealed that temporal autocorrelation was more prevalent than spatial autocorrelation and therefore, only temporal correlation was modeled (Table 3.5).

The fit of all regression models was assessed by inspecting the residuals of each model for patterns (indicative of autocorrelation and a violation of regression assumptions), and by calculating an autocorrelation coefficient for the residuals using the software JMP 7.0.2 (SAS Institute Inc.). Models showing residual patterns or significant autocorrelation coefficients (Ljung-Box Q-statistic $p<0.05$) were discarded. The Q-statistic is provided in the time series analysis platform in JMP and can be used to test whether the residuals from a fitted regression can be distinguished from white-noise. The overall statistical modeling process is outlined in Figure 3.9.
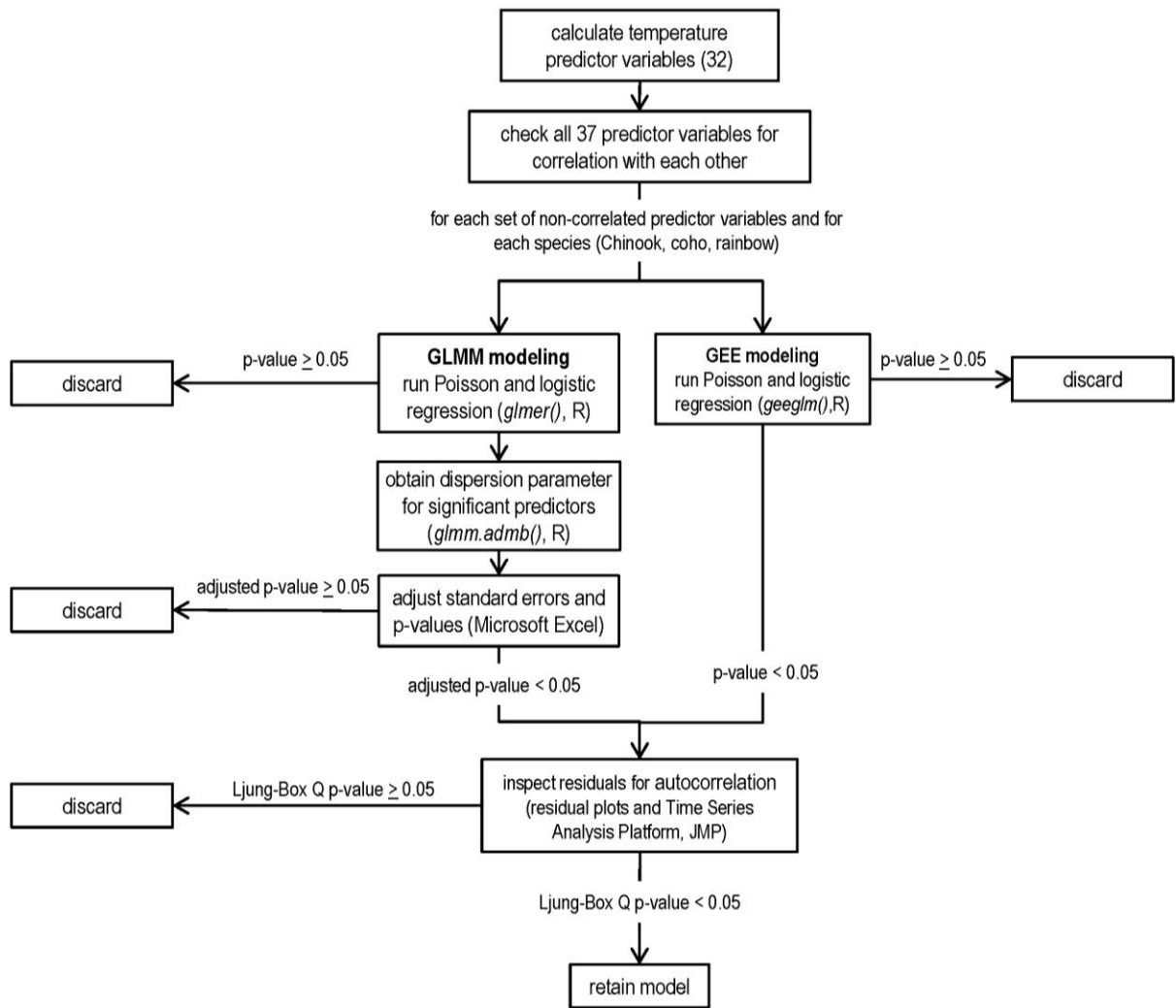
Figure 3.9: Statistical modeling process used to assess the relationship between salmonid distribution and habitat indicators.

3.4   Modeling of Groundwater - Surface Water Interactions

Two piezometers were installed to quantify groundwater discharge to Fortune Creek in the vicinity of site 3 and site 4. The piezometers were constructed of schedule 80 PVC piping with a 6 inch screened section at the base. They were manually advanced with a sledge hammer to a depth of 3 m below the creek bed. Onset HOBO U20 0–4 m Water Level USB Loggers were placed in the piezometers to record water levels and temperature every 30 minutes. Water level loggers were placed in the creek paired with the piezometers to simultaneously measure water levels in the stream (Figure 3.10). This information was used to calculate the vertical hydraulic gradient (VHG) in the streambed according to the following equation:

$$VHG = \frac{h_{piezometer} - h_{stream}}{d_{piezometer}} \qquad (3.1)$$

where $h_{piezometer}$ = head (m) measured in the piezometer

h_{stream} = head (m) measured at the stream bed (stream water level)

$d_{piezometer}$ = depth (m) of midpoint of piezometer screen below the stream bed

Slug tests on the two piezometers were performed in April 2009 to estimate hydraulic conductivity (K) of the materials in the streambed. Testing was conducted by removing a volume of water from the piezometer and recording the recovery of the head in the piezometer over time. Hydraulic conductivities were estimated from these data using the Hvorslev (1951) method. Once a value for hydraulic conductivity was established, the groundwater flux into or out of the streambed was calculated by the formula:

$$q = K * VHG \qquad (3.2)$$

where K = streambed hydraulic conductivity (m/s)

VHG = vertical hydraulic gradient

One dimensional modelling of vertical temperature profiles was performed using the software VS2DHI (Hsieh et al., 2000), which is a graphical extension of the software VS2DH (Healy & Ronan, 1996). Streambed temperatures at 50 cm depth were simulated under various flux rates and directions and then compared to measured temperatures. Groundwater flux was varied by adjusting the hydraulic conductivity of the streambed.
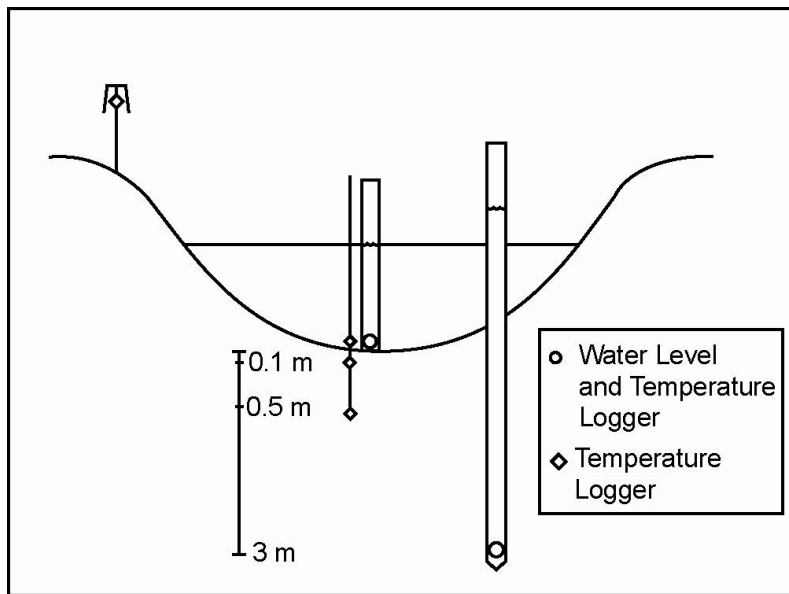
Figure 3.10: Piezometer and temperature logger configuration in Fortune Creek.

The best fit between simulated and observed temperature time series was determined by calculating the root mean square error (RMSE) from the following equation:

$$RMSE(y, c) = \sqrt{\frac{1}{n} \sum_{i=1}^{n} (c_i - y_i)^2} \qquad (3.3)$$

where   y = observed temperature series

c = simulated temperature series

n = number of observations

The flux rate that yielded the lowest RMSE between observed and simulated temperatures was selected as the best fit (Su et al., 2004).

The boundary conditions for the model were temperature and average pressure head at 3 m depth and at 10 cm depth below the streambed (Figure 3.11). Average pressure head at 3 m depth was used even though daily measurements were available because there was very little change over the simulation periods. First, simulations were run at site 3 and 4, where detailed information on pressure heads and temperatures at 3 m depth were available from the piezometers.

Subsequently, simulations were run at the remaining sites. Flow direction, hydraulic gradient and deep streambed temperatures were unknown at those sites. Niswonger and Prudic (2003) describe how under these circumstances, one dimensional modeling can be used to assess the situation as a model simulating upward flow would not be able to match an observed streambed temperature profile if downward flow existed, and vice versa. At these sites, the hydraulic gradient was assumed to be one and both upward and downward fluxes were separately simulated and compared to measured streambed temperatures.

The lower temperature boundary condition at these sites (3 m depth) was taken as the average temperature measured in the piezometers at sites 3 and 4. This was deemed acceptable due to the minor variation and similarity in temperatures between those sites over the modeling periods (0.2°C). Streambed soil properties and thermal constants used in the simulations are presented in Table 3.6. These parameters were taken from literature. For simulation purposes, streambed materials were assumed to be homogeneous and isotropic.
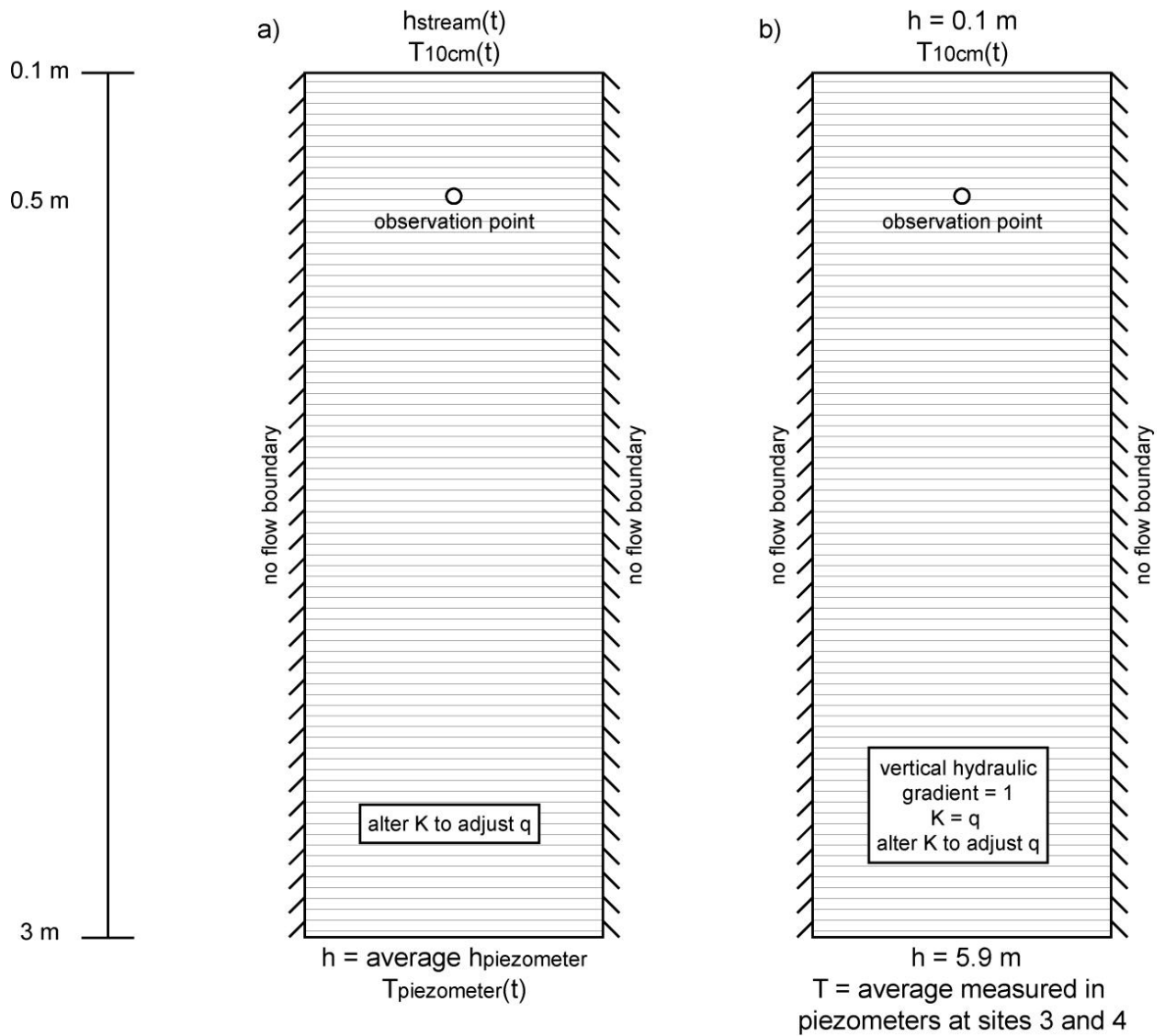
Figure 3.11: Schematic diagram of the model domain and boundary conditions for the VS2DHI streambed temperature simulations at a) sites 3 and 4 with piezometers and b) the remaining sites without piezometers.

Table 3.6: Streambed soil properties and thermal constants used in VS2DHI simulations of vertical groundwater fluxes in Fortune Creek.

Variable	Value
Porosity	0.377[a]
Heat capacity of dry solids (Cs)	2.08×10^6 J/m^3 K[b]
Heat capacity of water (Cw)	4.186×10^6 J/m^3 K[b]
Thermal conductivity	varied by site (proportion of organic vs. mineral streambed materials varied between the upstream and downstream reaches)
Longitudinal and transverse thermal dispersivity	0.01[a]
Anisotropy ratio, Kh/Kv	1[c]

a   Constantz et al. (2003)
b   Healy & Ronan (1996)
c   isotropic conditions assumed

Groundwater flux was simulated for both summer and winter time periods (Table 3.7). During winter, time periods were not the same for all sites due to logger failures at some sites.

Table 3.7: Groundwater flux simulation periods.

Site	Summer 2008	Winter 2009
1	Jul 18 – Aug 31 (45 days)	n/a
2	Jul 18 – Aug 31 (45 days)	Feb 18 – Mar 11 (22 days)
3	Jul 18 – Aug 31 (45 days)	Feb 18 – Mar 11 (22 days)
4	Jul 18 – Aug 31 (45 days)	Feb 18 – Mar 11 (22 days)
5	Jul 18 – Aug 31 (45 days)	Dec 14 – Jan 3 (21 days)
6	Jul 18 – Aug 31 (45 days)	Dec 25 – Jan 3 (10 days)
7	Jul 25 – Aug 31 (38 days)	Dec 21 – Jan 13 (11 days)
8	Jul 25 – Aug 31 (38 days)	Feb 18 – Mar 11 (22 days)

3.5 Stream Energy Balance

This research used an energy balance as a conceptual model and did not attempt to numerically model measured stream temperatures in Fortune Creek. The goal was to understand the physical processes governing stream temperature during the summer months in light of the need for management recommendations for meeting salmonid water temperature criteria in the future. The purpose was to assess whether groundwater inflow into Fortune Creek is capable of providing thermal refuge for resident salmonids or lead to a noticeable reduction in stream temperatures during periods of heat stress. The influence of discharge volumes on stream temperatures was also of interest. To fully model stream temperatures using a physical process based numerical modeling approach to energy balances, extensive meteorological data collection beyond the scope of this research would be required (Brown, 1969; Meier et al,. 2003).

To get an understanding of the relative magnitude of cooling effects from groundwater inflows and variations in discharge volume, changes in water temperature for a hypothetical 1 km long, 1 m wide stream reach under shaded and unshaded conditions were calculated (Figure 3.12). As a 1m^2 parcel of water travels along this hypothetical reach, its change in temperature over the 1 km distance is a function of energy inputs minus outputs, its volume (= depth x 1 m^2, determined by streamflow) and the time it takes to travel the distance (determined by streamflow). The change in temperature over the 1 km distance can be calculated from the following equation:

$$\Delta T = \frac{Q * \frac{1000}{velocity}}{volume * C_w} \qquad (3.4)$$

where Δ T = change in temperature (°C) over 1 km stream reach

Q = net energy flux (W/m^2)

C$_w$ = volumetric heat capacity of water (4,186,000 J/m^{3}K)

volume = depth (m) x 1 m^2 area

velocity = average measured velocity during discharge gauging (m/s)

Once conceptual modeling was completed, the results were compared to real heating rates measured in a 1 km long stream reach between site 2 and site 3 from June to August 2008.

Since several studies have found that solar radiation is the principal energy input for unshaded streams (Section 2.1), it was assumed to play a dominant role in summer daytime heating of Fortune Creek and was the only energy input considered. For unshaded conditions, the average peak net solar radiation value from a study conducted near Kamloops (approximately 100 km northwest of Fortune Creek) of 580 W/m^2 was used (Leach, 2008). For shaded conditions, a value of 50 W/m^2 was used.

The VS2DHI software provides an energy mass balance and energy flux rates at the top and bottom boundary of the simulated domain. Flux values provided apply per square meter of streambed. Advective heat flux values at the streambed provided by the simulations at site 4 were used as this was the site with the strongest upward groundwater flux and was indicative of "best case" conditions.

Discharge data measured near site 2 were used in the calculations because the channel and flows at this site were most representative of conditions between site 2 and 3, the reach used to compare estimated to real temperature changes. Since no other components of the stream energy budget were considered, this exercise provided an estimation of the heating/cooling effect specifically attributable to each of the variables, and not an estimate of true heating rates.
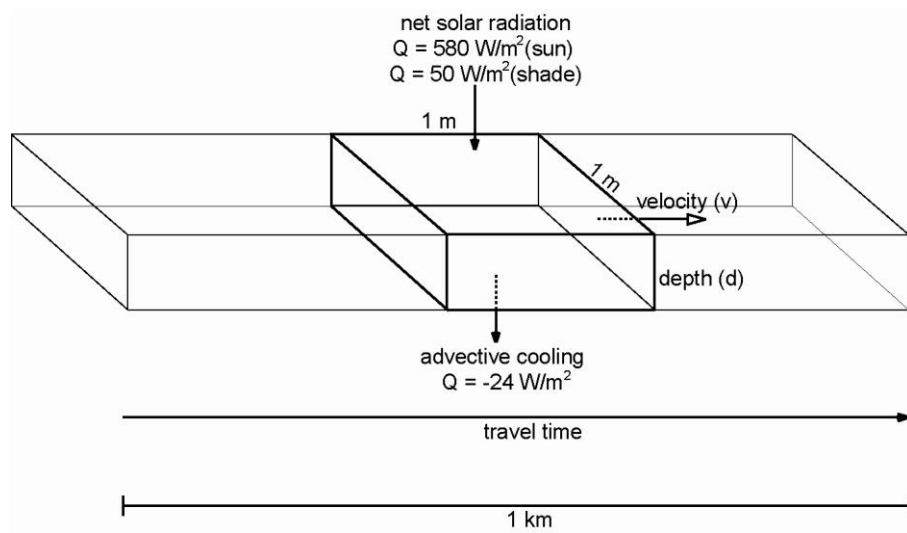
Figure 3.12: Conceptual model of stream temperature change under various streamflow conditions and groundwater inflow.

4.0 RESULTS

4.1 Field Survey Results

4.1.1 Flows and Water Levels

Streamflow measured over the 2008/2009 study season is presented in Figure 4.1. It is evident that peak flows during the 2008 season were much higher than in 2009, though still below average naturalized values. The lower reaches of Fortune Creek from site 6 downstream to the Shuswap River were inundated entirely during freshet in 2008 (Figure 3.8), likely the combined result of high peak flows and extensive backflooding from the Shuswap River. Flooding of the lower reaches did not occur in 2009. Flows from mid-July to the end of September were very low (<0.05 m^3/s) compared to naturalized flows in both years. The lowest measured flows during the summer period were 0.009 m^3/s in 2008 and 0.001 m^3/s in 2009.
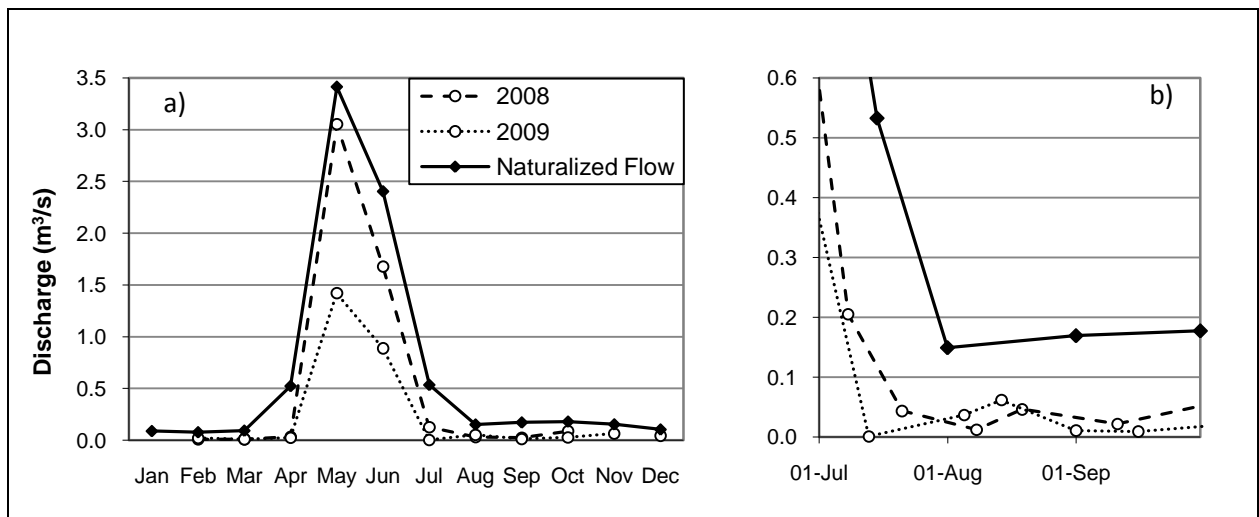
Figure 4.1: Naturalized (1960-1984 average, Seebacher et al., 2007) and measured streamflow volumes in Fortune Creek a) over the year and b) over the summer season.

Stream water levels recorded at sites 2 to 4 fluctuated substantially and rapidly (up to 5 cm in 30 minutes) in the summer of 2008 (Figure 4.2). The maximum daily fluctuation in water levels was 10 cm. Water levels at site 2 were highest because the logger was situated in a pool. Water levels generally declined from mid-July onward but several rises can be seen at each site. These followed several water releases from the headwater reservoirs that occurred over the summer. Water levels at the study sites began to rise approximately 3 days after the releases from the reservoir occurred. However, rises in water levels coincided with major precipitation events and it is therefore unknown whether rises in

water levels were the result of reservoir releases or rainfall. Released water volumes totaled approximately 25% of measured discharge in the creek during the first release in late July and approximately 300% of measured discharge during the second release in early August. Dates and release volumes are presented in Table 4.1.
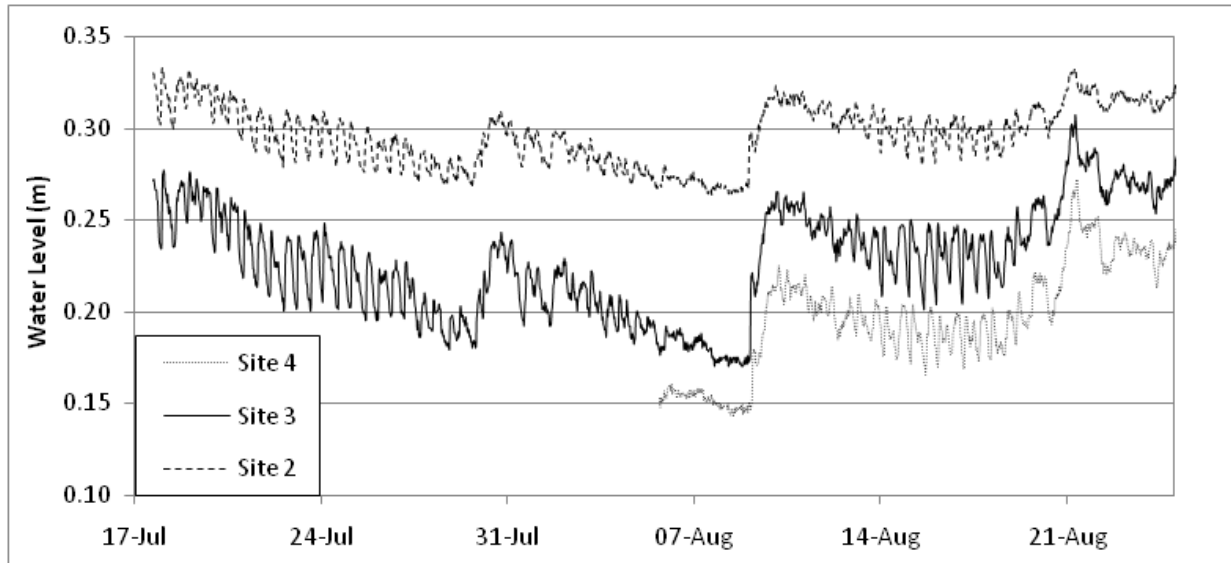
Figure 4.2: Water levels measured in Fortune Creek over the summer of 2008.

Table 4.1: Initial water release volumes provided by the City of Armstrong (Clement, personal communication, 2008) and streamflow measured in Fortune Creek over the summer of 2008.

Date	Released Volume (m^3/s)	Streamflow prior to release (m^3/s)
July 25	0.0158	0.0437
July 29	0.0079	0.0437
Aug 6	0.0315	0.0124

4.1.2 Stream and Streambed Temperatures

Water temperatures measured in Fortune Creek varied from a low of 0°C to a high of 29.5°C (Table 4.2). Temperatures measured in the air, stream, and the streambed between June 27 and August 31, 2008, are presented in Figure 4.3. Summer stream temperatures generally increased from high to low-elevation reaches. However, maximum temperatures were highest in the intermediate reaches. The highest water temperature of 29.5°C was recorded at site 3. At the adjacent downstream site 4, the highest temperature recorded was about 5°C lower at 24.1°C. Maximum values at sites further downstream were also cooler than at site 3. During winter, sites 1 and 2 were the only ones that did not freeze over and only patchy ice coverage was observed at site 4. Site 4 maintained relatively warm temperatures similar to those at site 1, whereas the remainder of the valley bottom sites was colder.

Table 4.2: Summer and winter temperatures recorded in Fortune Creek in 2008.

Site	Summer Temperatures (°C)					Winter Temperatures (°C)				
	Air	Stream	10 cm	50 cm	3 m	Air	Stream	10 cm	50 cm	3 m
1	5.7–29.4	7.0–11.0	7.1–10.5	8.1–9.6	-	-23.2–2.6	0.2–4.1	0.4–3.9	1.9–4.1	-
2	1.9–36.7	7.8–16.8	8.5–15.0	10.8–13.6	-	-31.5–8.4	0.1–3.8	0.4–3.1	1.4–3.7	-
3	-1.4–40.4	7.4–29.5	7.5–28.4	11.4–17.2	8.1–8.9	-36.4–15.7	0.0–2.9	0.1–2.9	1.6–4.2	8.5–9.4
4	0.5–40.8	7.5–24.1	8.8–20.1	10.9–13.9	8.2–8.9	-35.9–14.8	0.7–3.8	1.4–3.7	3.6–5.3	8.5–9.2
5	0.1–38.5	8.1–25.3	10.0–21.7	11.8–15.5	-	-36.7–12.0	-0.1–2.9	0.3–2.7	2.1–4.4	-
6	2.5–34.1	10.0–27.0	11.5–22.7	13.8–17.2	-	-33.1–5.5	0.0–0.2	0.2–1.0	1.8–3.8	-
7	-0.7–44.0	11.0–26.2	13.2–22.5	15.5–18.0	-	-34.8–11.1	0.0–0.2	0.2–1.0	1.8–3.9	-
8	-0.6–40.5	11.5–27.9	13.2–24.4	15.1–18.8	-	-34.9–14.7	0.0–0.3	0.6–1.2	2.5–3.6	-
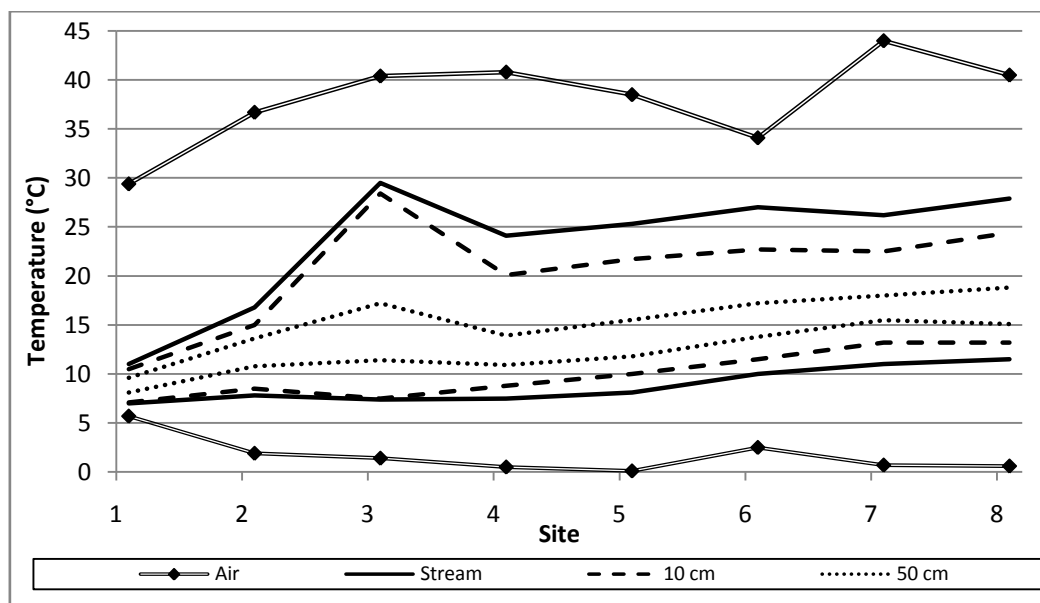
Figure 4.3: Range of air, stream and streambed temperatures recorded at the study sites between June 27 and August 31, 2008.

Streambed temperatures at 10 cm depth are presented in Figure 4.4. They ranged from 0.1°C to 28.4°C. During both summer and winter, the range of temperatures recorded at 10 cm depth between site 1 and 3 was almost identical to that recorded instream. Substrate in these reaches is relatively coarse (cobble to boulder size) and stream water can easily enter the bed. The temperature range at 10 cm depth at sites 4 to 8, where substrate is much finer, was several degrees smaller than that recorded instream during summer.

During winter, temperatures at 10 cm depth declined in a downstream direction. Temperatures at site 4 were noticeably higher than the remainder of the sites. Most notably, stream and streambed temperatures at 10 cm were substantially higher at site 4 than at the upstream site 3 during winter and lower during summer. This is contrary to the general downstream warming (summer) and cooling (winter) trend observed.
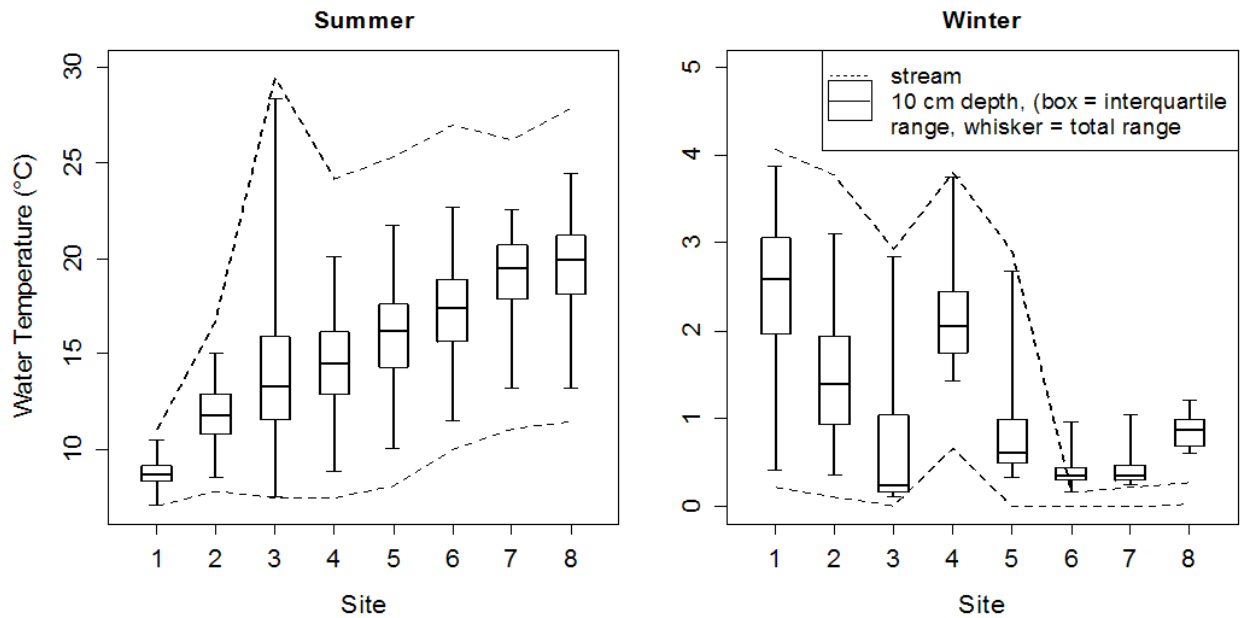
Figure 4.4: Streambed temperatures at 10 cm depth and minimum and maximum stream water temperatures in the summer (July 18 to August 31, 2008) and winter (December 14, 2008 to February 28, 2009) in Fortune Creek.

Similar patterns were observed at depths of 50 cm below the streambed (Figure 4.5). During summer, temperatures increased in a downstream direction, except for site 4, which was cooler than site 3. During winter, streambed temperatures at site 4 were substantially higher than any of the other sites, and even higher than temperatures measured in the water column during the same period. Winter streambed temperatures at sites 1 to 3 were within the range of stream temperatures indicating that they were driven primarily by stream water temperatures. Winter streambed temperatures at sites 4 to 8 were several degrees higher than temperatures in the water column.
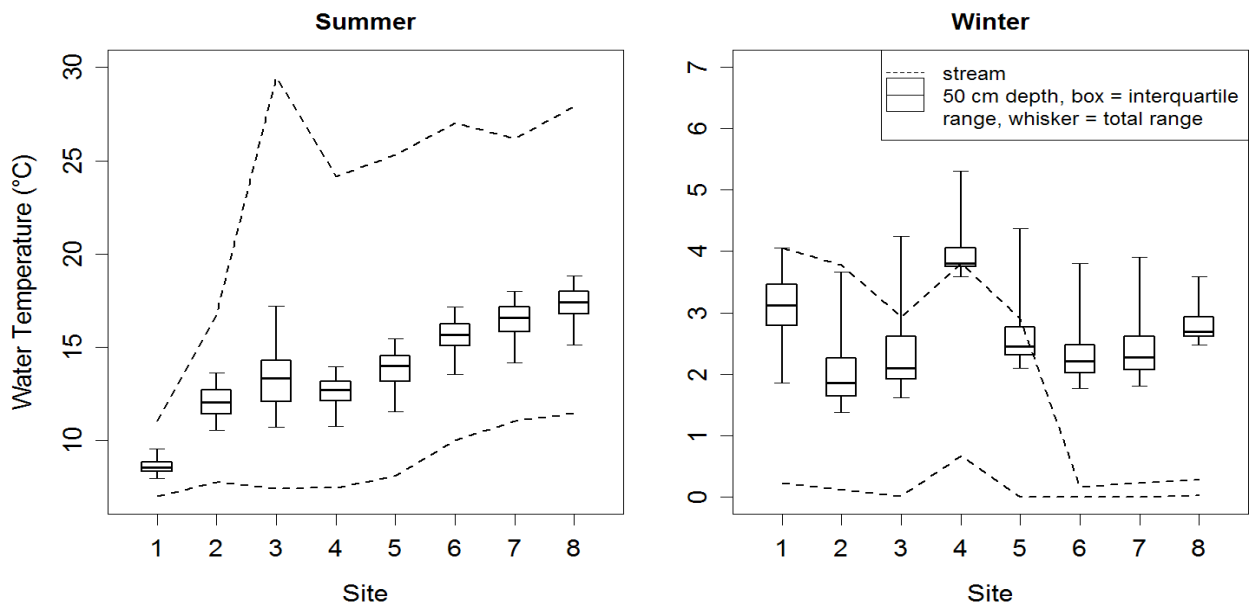
Figure 4.5: Streambed temperatures at 50 cm depth and minimum and maximum stream water temperatures in the summer (July 18 to August 31, 2008) and winter (December 14, 2008 to February 28, 2009) in Fortune Creek.

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At site 3, temperatures measured in the piezometer 3 m below the streambed ranged from a low of 8.1°C in April to a high of 9.4°C in November. At site 4, temperatures ranged from a low of 8.2°C in April to a high of 9.2°C in November. These sites were more than 1 km apart, which indicates that temperature at 3 m below the streambed fluctuates relatively little on an annual basis (1.1°C) and between locations (0.2°C). This supports the use of temperatures measured at sites 3 and 4 for the lower boundary condition at the remaining sites.

4.1.3 Streambed Hydraulic Gradient

The streambed vertical hydraulic gradient was calculated from water levels measured in the piezometers and in the stream. After installation, water levels in the piezometer at site 3 took almost 4 weeks to equilibrate (Figure 4.6). The piezometer at site 4 equilibrated in approximately 10 days. Water levels in both piezometers rose approximately 0.25 m from August/September to March/April. The dip in water levels in April was due to slug testing on the piezometers. The slower recovery of water levels at site 3 is evident in the graph (Figure 4.6).

Stream water levels at site 3 and site 4 were well below piezometer water levels throughout the study period. The vertical hydraulic gradients were on average 0.25 upward at site 3 and 0.37 upward at site 4. This indicates that the section of stream between site 3 and 4 was under groundwater discharge conditions throughout the entire study season. Streambed hydraulic conductivities calculated from the slug tests based on Hvorslev's (1951) method were 1.2×10^{-8} m/s at site 3 and 4.7×10^{-8} m/s at site 4. Resulting upward groundwater fluxes calculated from head and hydraulic conductivity measurements were 3×10^{-9} m/s and 1.8×10^{-8} m/s, respectively.
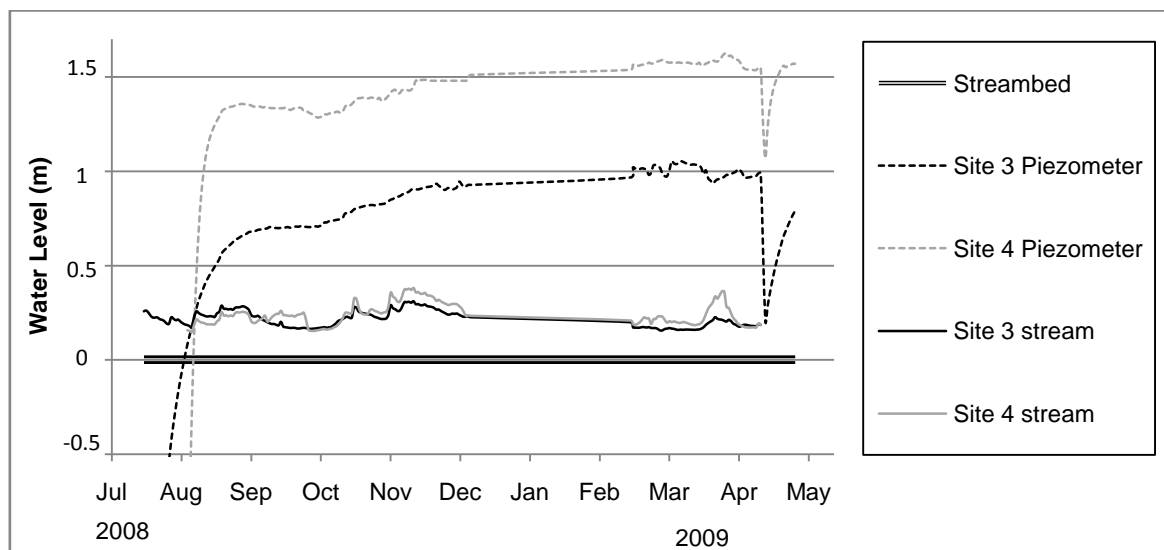
Figure 4.6: Water levels in Fortune Creek recorded at 3 m depth in the piezometers and instream.

4.1.4 Water Quality

Water chemistry parameters measured in Fortune Creek over the summer of 2008 (July 1 to September 3) are presented in Table 4.3 and Figure 4.7. DO values ranged from a low of 3.1 mg/L at site 8 to a high of 18.1 mg/L at site 7. DO values below the 5 mg/L threshold were measured at site 6 and 8 on July 7, 2008 (Section 2.4.2). Dead juvenile salmonids and adult rainbow trout were observed in the water during the low oxygen event at site 8. Figure 4.7 demonstrates that DO saturation levels near sites 1, 2 and 4 fluctuated little (90% to 110%). At sites 3 and 5 to 8, the range of saturation levels measured was much larger (up to 196%). These are the sites at which abundant algal growth and instream vegetation were noted.

Conductivity values ranged from a low of 150 µS/cm measured at site 1 to a high of 372 µS/cm measured at site 4 (Figure 4.7). Starting between site 2 and 3, various field drains and surface runoff entered the creek with a major discharge upstream of site 4. This likely contributed to the jump in conductivity values between site 3 and 4 and the generally higher conductivity values measured at all downstream sites.

pH values measured ranged from 6.23 to 9.27 (Figure 4.7). pH was more variable and on average higher (more basic) in the valley-bottom reaches (sites 3 to 8). Critically high pH values of 9.0 were exceeded at sites 3, 6 and 7 (Section 2.4.3).

Table 4.3: Water chemistry in Fortune Creek between July 1 and September 3, 2008.

Site	Maximum DO (mg/L)	Minimum DO (mg/L)	Maximum Conductivity (µS/cm)	Minimum Conductivity (µS/cm)	Maximum pH	Minimum pH
1	11.3	9.4	226	150	8.27	7.30
2	11.1	8.9	246	159	8.07	7.20
3	13.6	10.2	234	160	9.27	7.79
4	11.3	8.1	372	166	8.79	6.23
5	14.4	9.1	346	177	8.78	7.89
6	17.6	5.0	310	178	9.00	6.90
7	18.1	6.7	269	201	9.13	7.68
8	11.1	3.1	289	188	8.55	6.80
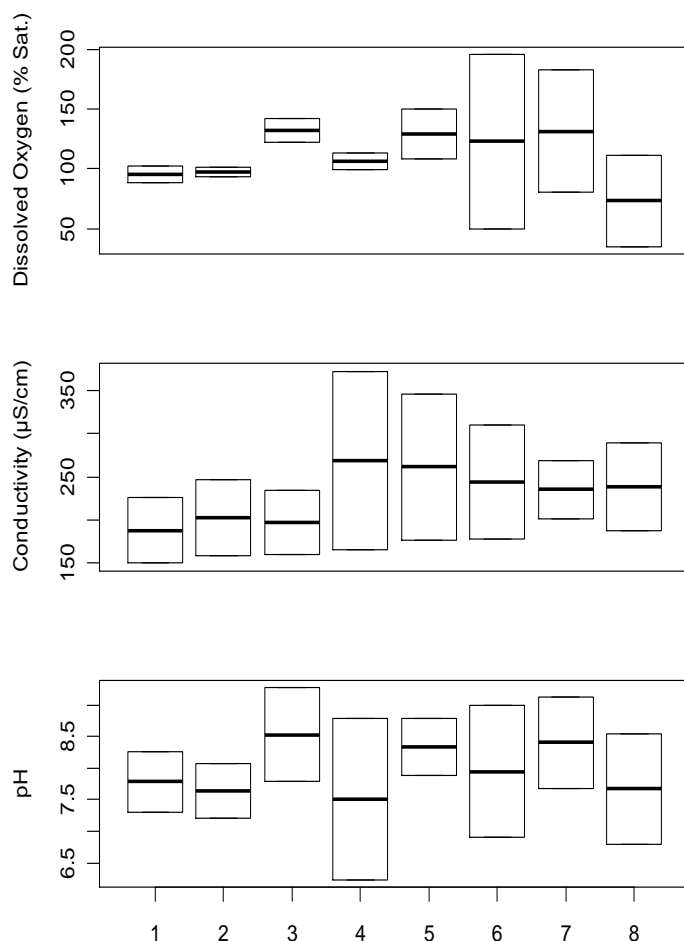
Figure 4.7: Dissolved oxygen saturation, conductivity and pH measured at the eight study sites along Fortune Creek.

4.1.5 Observed Salmonid Distribution

Visual observations prior to the onset of sampling revealed that large numbers of salmonids used the lower, seasonally inundated reaches of Fortune Creek during the freshet of 2008 (sites 6 to 8) but were absent later on when sampling commenced (July 1 2008). Peak flows were much lower in 2009 and backflooding in the lower reaches was limited. The large number of salmonids observed during the 2008 freshet was not observed in the early summer of 2009.

Fish species captured in Fortune Creek between July 2008 and November 2009 included coho salmon and Chinook salmon, rainbow trout, redside shiner, sculpin, sucker (*Catostomus sp.*), northern pikeminnow, peamouth chub (*Mylocheilus caurinus* Richardson) and carp (*Cyprinus carpio* L.). Of these, pikeminnow, peamouth chub and carp had not been previously reported in Fortune Creek.

A total of 5,397 fish were captured over the study period, 49% (2,670) of which were salmonids (Figure 4.8). Of the salmonids captured, Chinook and rainbow were slightly more abundant than coho. Fish

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captured generally ranged from approximately 10 mm to 200 mm in length. Coho and Chinook juveniles ranged from approximately 40 mm to 140 mm. All captured rainbow trout were juveniles; however several adults were observed in the creek, including individuals found within the stream reach between site 1 and the intake dam (Figure 3.2). Juvenile coho and Chinook were present in Fortune Creek year round (Figure 4.9).
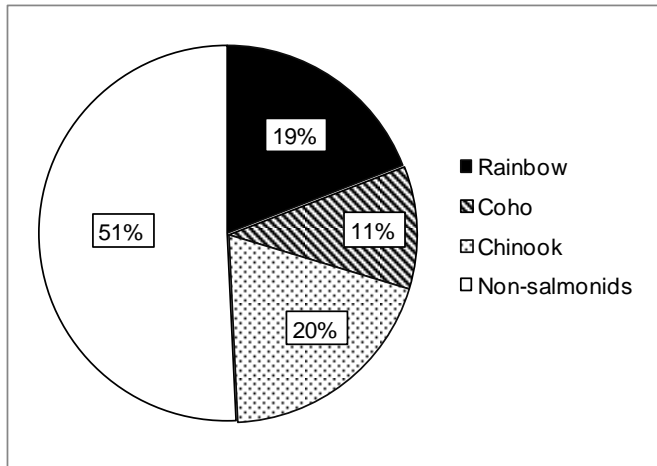
Figure 4.8: Proportion of fish species captured over the 2008/2009 study period.

Figure 4.9: Number of coho and Chinook captured in Fortune Creek over the 2008/2009 study period. Each bar represents a sampling event.

The relative abundance of the fish species captured at each study site over the entire study period is presented in Figure 4.10. Salmonid species were dominant at sites 1 through 4, making up over 99% of fish species captured. Non-salmonid species were dominant at the remaining sites, making up between 74% and 99%. The only site with substantial overlap between salmonids and non-salmonids was site 5. The most frequently captured non-salmonid species were suckers, northern pikeminnow and redside shiners, with the occasional peamouth chub, sculpin and carp trapped at the lower-elevation sites (sites 5 through 8).

Figure 4.10: Fish species recorded in Fortune Creek over the 2008/2009 study period by site.

Salmonid counts for each site are presented in Figure 4.11 for the duration of the study period. Rainbow trout were the most abundant species captured at site 1 (82%). The remainder was mainly Chinook; however they were only captured during the summer of 2008. Site 2 through 4 had a mixture of rainbow, coho and Chinook. All three species were consistently abundant at site 2, but coho and Chinook were absent from the site in late spring just prior to freshet. Large spikes in all salmonids were recorded at this site in the fall in both 2008 and 2009. Rainbow numbers were highest at site 3 (66% of fish captured).

Coho and Chinook were present at site 3 in June/July of both sampling years, but not for the remainder of the year. Large numbers of newly emerged rainbow fry were observed at the site starting in late July of 2008 and continued to be present at the site throughout the year. It is therefore suspected that rainbow spawned in the vicinity of the site in the spring of 2008. Field observations suggest that, while the fry remained at site 3 throughout the entire summer and persisted through water temperatures up to 29.5°C, they hardly grew over the study season and most were shorter than 40 mm in the late fall.

Chinook and coho were present at site 4 throughout July and August but not for the remainder of the year. Both salmonids and non-salmonids were present at site 5. Salmonid numbers were relatively low for most of the year, but spikes were observed in July 2008 (Chinook only) and September/October 2008 (coho and Chinook). Coho and Chinook were only occasionally captured in low numbers at sites 6 to 8. These observations occurred primarily in spring and fall.

The maximum daily water temperatures at which salmonids were directly observed were 25.5°C (Chinook and rainbow trout) and 26.3°C (coho).
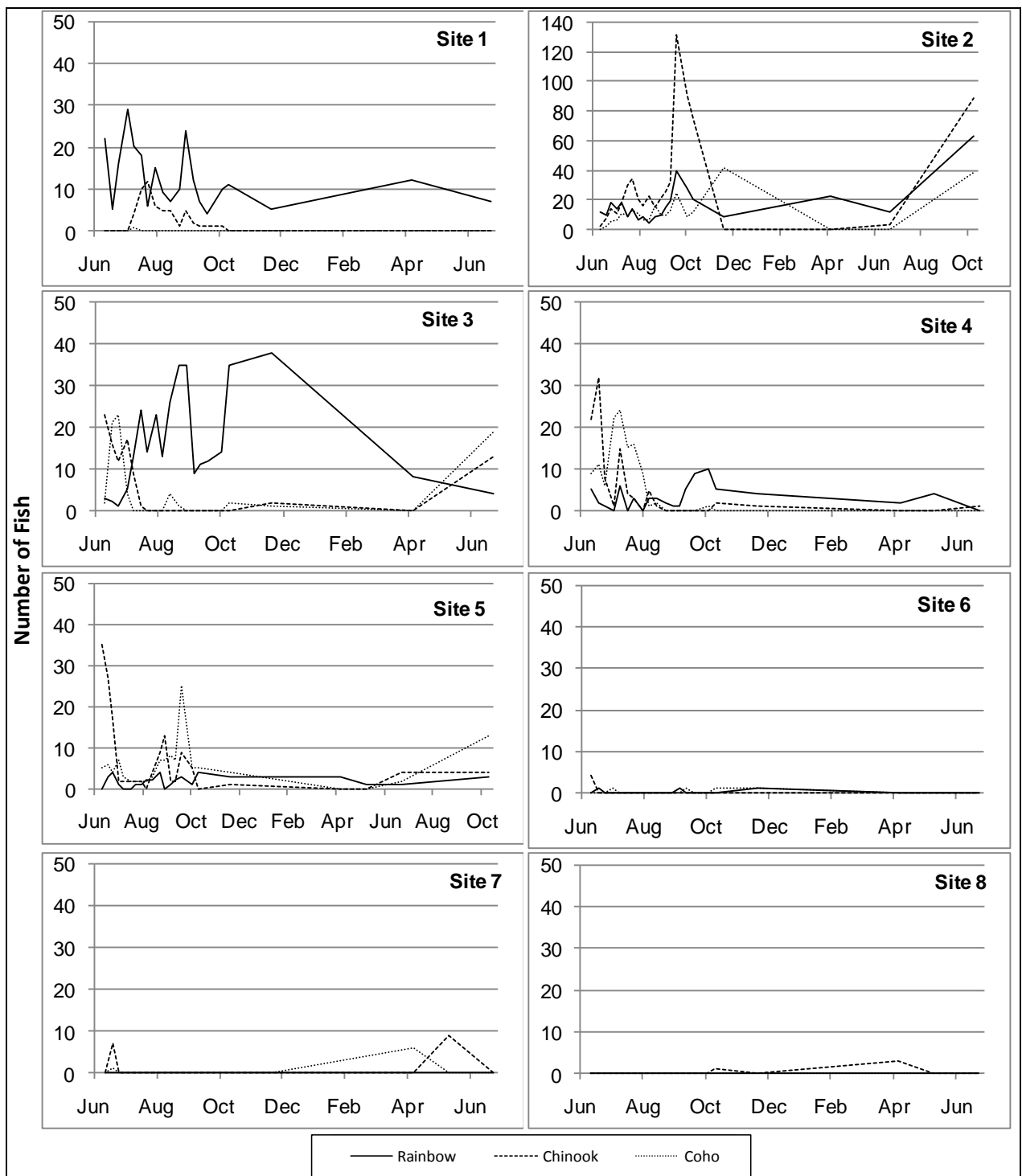
Figure 4.11: Salmonid species captured at each study site over the 2008/2009 study period.

Results of the DNA analysis indicate that most of the the Chinook present in Fortune Creek in the late summer of 2008 were genetically similar to the spawning populations in Bessette Creek, which is a stream-type population. Escapement numbers for Chinook in Bessette Creek have declined sharply in recent years (DFO, 2010c). These results indicate that juvenile stream-type Chinook utilize Fortune Creek during the summer, fall and winter seasons for rearing.

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4.2   Statistical Analysis of Salmonid Distribution

4.2.1 Poisson Regression

Salmonid distribution in relation to physical habitat indicators was statistically analyzed over an 8-week period in July and August 2008. During this time, salmonid numbers varied substantially by site but the variation was not as strong by week (Figure 4.12 , Figure 4.13, and Figure 4.14). Chinook salmon numbers were highest at site 2 and overall weekly catches generally decreased over the summer (Figure 4.12). Coho numbers were highest at site 4 but substantial numbers were also captured at sites 2 and 3 (Figure 4.13). There was no trend in overall weekly catches over July and August. Rainbow numbers were substantially higher at sites 1 and 2 than the rest of the sites, and similar to coho, there was no distinct trend in total rainbow numbers captured over the analysis period (Figure 4.14).
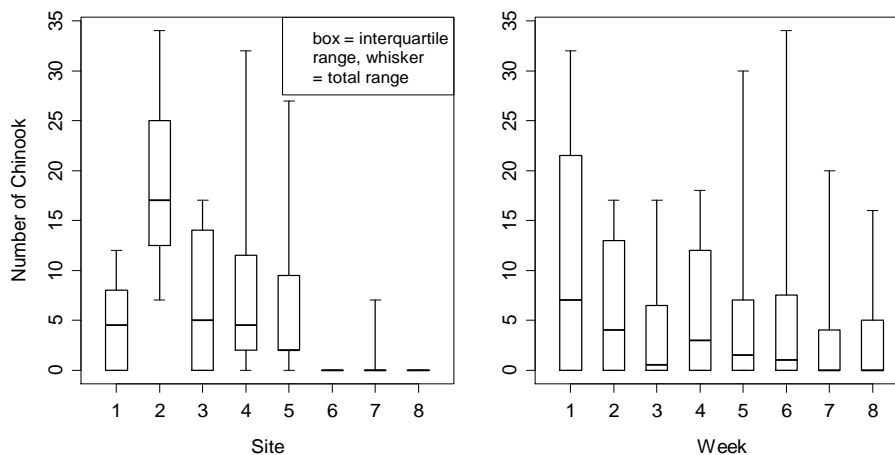
Figure 4.12: Variation in the number of Chinook captured in Fortune Creek by site and by week (July - August 2008).
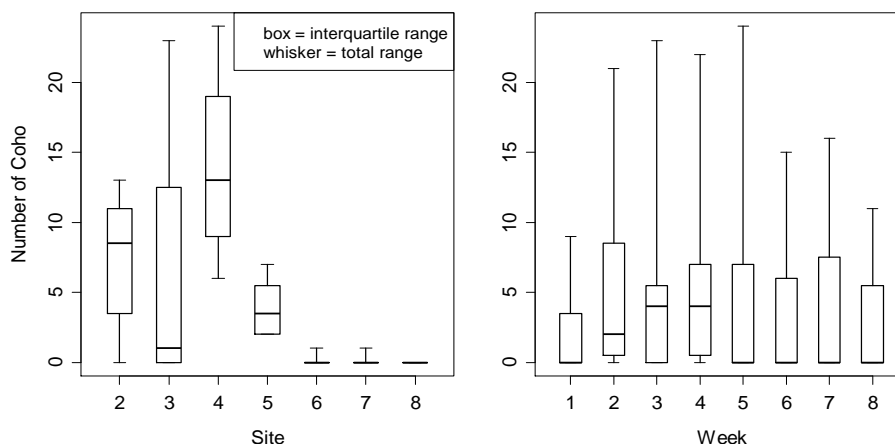
Figure 4.13: Variation in the number of coho captured in Fortune Creek by site and by week (July - August 2008).
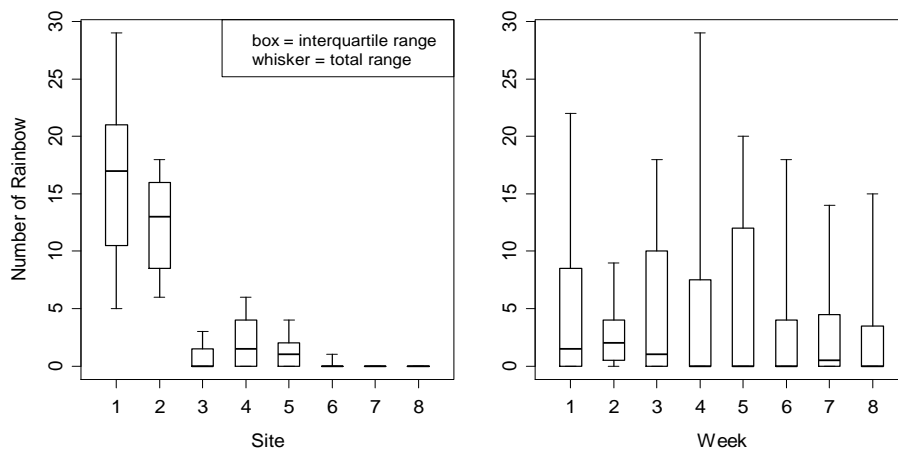
Figure 4.14: Variation in the number of rainbow trout captured in Fortune Creek by site and by week (July - August 2008).

All Poisson regression models included site and week as random effects. Results are only presented for those models that contain significant terms and where neither numerical (Ljung-Box Q p-value <0.05) nor visual analysis of the residuals revealed autocorrelation. For rainbow trout, numerical modeling errors were encountered for some of the temperature exceedance variables. The numerical errors were related to the algorithm used in the R function, which is open source software that is constantly being revised. Zuur et al. (2009) caution that model simplicity is important when conducting GLMM modeling to avoid numerical modeling errors. As a result of the errors encountered, the relationship between some of the exceedance variables and rainbow trout abundance was not analysed.

Backward model selection was extremely difficult due to a high degree of multicollinearity among most of the explanatory variables (Section 2.4.5.5.1). This was manifested in large changes in the estimated regression slopes and p-values for one variable once another was added or removed. Correlation coefficients between explanatory variables frequently exceeded 0.75. Naturally, the highest correlation existed among the temperature variables. However, correlation was also high between conductivity and most temperature variables and between pH, temperature and DO. Correlated variables were never included in the same model. Given the difficulties with backward model selection, only forward model selection was completed and most models included only one or two explanatory variables.

In most cases, a number of variable combinations yielded acceptable models for predicting salmonid counts. Where a variable was included in several valid models, the range of values for the regression slope estimate is provided. Initially, a large number of models resulted in significant explanatory variables but many were discarded once corrected for overdispersion and when inspection of the residuals revealed a lack of independence.

4.2.1.1 Chinook

Eight valid models for predicting Chinook abundance were retained after accounting for overdispersion and residual autoregression. Based on these models, six of the evaluated predictor variables are associated with Chinook abundance in Fortune Creek. The regression estimates for each predictor variable are presented in Table 4.4. At this point it should be emphasized that all temperature variables used in the regression models were weekly averages (maximum, mean and minimum), weekly totals (total exceedance hours) or the highest daily values encountered in the preceding 7-day period (absolute maximum and longest continuous exceedance hours).

Average mean and minimum temperatures were significant predictors of Chinook abundance, and abundance decreased with increasing temperatures. Total hours > 15°C were also significantly and negatively associated with Chinook abundance. Conductivity and pH were both negatively associated with Chinook abundance. Specifically, Chinook abundance decreased with increasing conductivity and increasingly basic pH. Minimum DO was negatively related to Chinook abundance. While absolute and average weekly maximum temperatures were significant predictors of Chinook counts, the models had to be discarded because of autocorrelation in the residuals.

Table 4.4: Regression estimates of Poisson mixed-modeling of Chinook salmon counts in Fortune Creek.

Variable	Regression Slope Estimate	p-value[a]
Average weekly mean temperature	-0.85	< 0.001
Average weekly minimum temperature	-1.04 to -0.98	< 0.001
Weekly total hours > 15°C	-0.03	< 0.001
Conductivity	-0.01	0.001
pH	-2.81 to -2.18	< 0.001
Minimum DO	-0.66	0.001

a   if variable present in several valid models the highest recorded p-value is reported

All final models for predicting Chinook abundance in Fortune Creek are presented in Table 4.5. They are presented in order from best to worst, based on the AIC. The best predictions were obtained from a model containing the total hours > 15°C and minimum DO (at sampling). For ease of presentation, figures are only shown for the univariate models (CHP4, CHP6, and CHP8) and these are discussed in detail (Figure 4.15). The best predictions from a univariate model were obtained from Model CHP4, which contains total hours > 15°C as a predictor variable. Despite a slightly worse model fit (higher AIC value), Model CHP6 containing mean temperature had similar predictive capabilities, as indicated by the similarity in the penalized residual sum of squares (PWRSS). Model CHP8, which contains minimum temperature as a predictor variable, had a slightly poorer fit and had weaker predictive capabilities as indicated by a higher AIC and higher PWRSS, respectively.

Table 4.5: Poisson models for predicting Chinook counts in Fortune Creek.

Model		AIC	Log Likelihood	Deviance	Deviance Reduction[a]	PWRSS[b]
CHP1	7.85 – 0.03 (Total Hrs > 15°C) – 0.66 (Min. DO) ± 5.91	156.7	-73.4	146.7	57%	82
CHP2	17.94 – 0.84 (Mean Temp.) – 0.66 (Min.DO) ± 6.78	172.4	-81.2	162.4	53%	91
CHP3	25.76 – 0.71 (Min.Temp.) – 2.18 (pH) ± 5.42	185.4	-87.7	175.4	49%	112
CHP4	2.49 – 0.03 (Total Hrs > 15°C) ± 3.5	186.6	-89.3	178.6	48%	119
CHP5	18.12 – 1.04 (Min.Temp.) – 0.65 (Min.DO) ± 6.54	200.0	-95.0	190	45%	217
CHP6	12.22 – 0.85 (Mean Temp.) ± 6.51	200.2	-96.1	192.2	44%	117
CHP7	24.95 – 0.01 (Conductivity) – 2.81 (pH) ± 7.04	217.1	-103.5	207.1	40%	136
CHP8	11.75 – 0.98 (Min.Temp.) ± 5.03	236.8	-114.4	228.8	33%	164

a   Reduction in deviance compared to a null model containing only the random effects and an intercept
b   Penalized weighted residual sum of squares

When comparing the predicted Chinook counts based on mean and minimum temperatures, it is evident that Chinook could tolerate a wider range of mean than minimum temperatures (Figure 4.15). On average, Chinook counts were not greatly affected by mean temperatures up to approximately 13°C. Once mean temperatures exceed 13°C, Chinook abundance decreased rapidly, approaching zero at mean temperatures of approximately 14°C to 15°C. However, when variation among sites and weeks was considered, this threshold could be up to 22.0°C.

Similarly, Chinook counts seemed relatively unaffected by minimum temperatures up to approximately 11°C, at which point abundance started to decline rapidly (Figure 4.15). On average, predicted Chinook abundance approached zero at minimum temperatures of approximately 12.0°C; however, considering variation among sites and weeks suggests that Chinook could be present up to a minimum temperature of approximately 17.1°C.
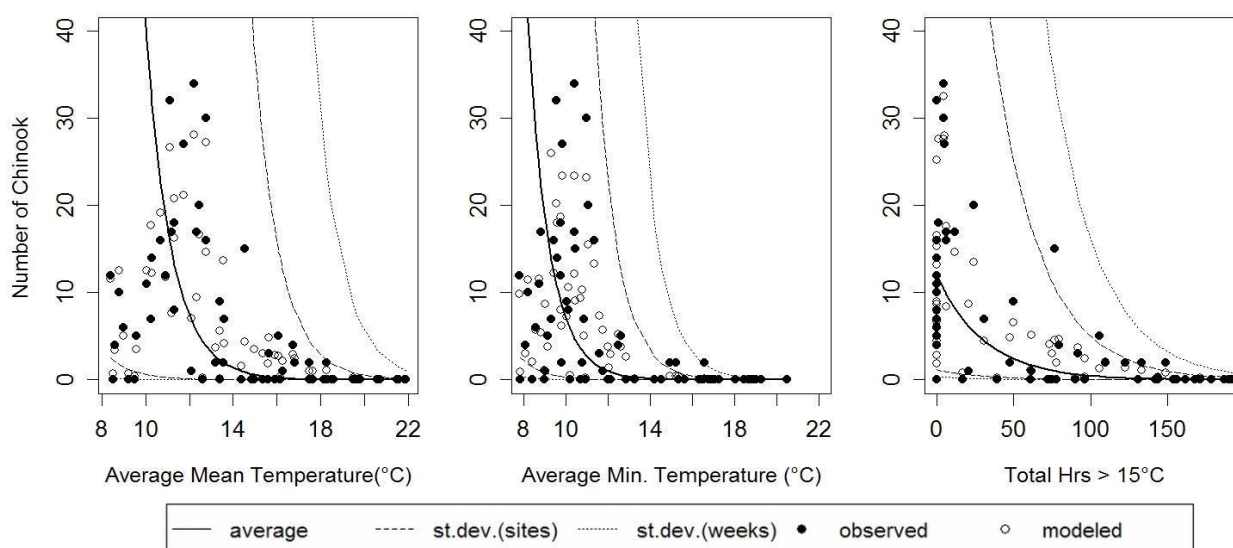
Figure 4.15: GLMM predicted Chinook abundance in Fortune Creek in relation to average minimum and mean water temperatures and total hours exceeding 15°C.

On average, Chinook counts were relatively unaffected when total hours > 15°C were below approximately 25 (Figure 4.15). Past that, Chinook numbers declined to zero at approximately 85 hours. With variation among sites and weeks considered, this number could reach up to 200 hours. The residual plot for Model CHP4 is presented in (Figure 4.16) to illustrate the lack of autocorrelation. The residuals show no clear patterns indicative of autocorrelation although the residuals from sites 6 to 8 (observations 42 to 64) are much smaller than the remainder. This is due to the fact that most observations at these lower sites were zero and therefore, residual values are very small. The second plot in (Figure 4.16) indicates that the variance is equal across the range of predicted values.
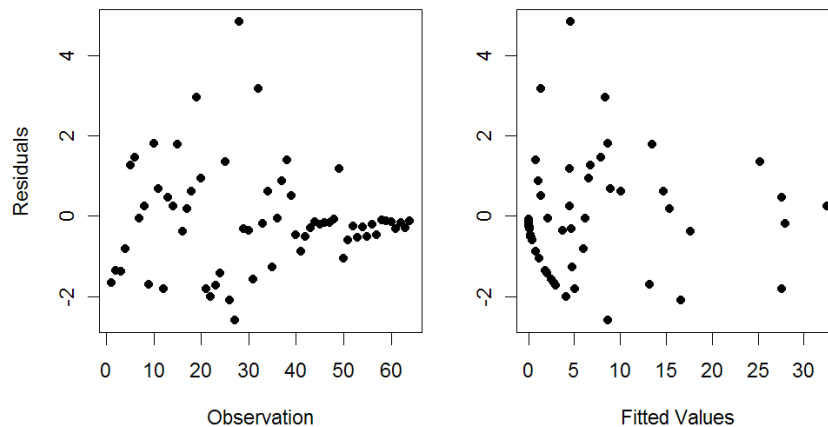
Figure 4.16: Plot of residuals for Chinook Model CHP4 (Total hrs > 15°C) by observation number and against fitted values.

4.2.1.2 Coho

After accounting for overdispersion and residual autoregression, ten valid models for predicting coho abundance were retained. These models included ten of the evaluated predictor variables and two interaction terms. Due to extensive multicollinearity, no models included more than two predictor variables. The regression slope estimates for each predictor variable and the interaction terms are presented in Table 4.6.

Absolute maximum temperature, average maximum temperature, and average mean temperatures were significant predictors of coho abundance. Coho numbers generally decreased with increasing temperatures. Coho abundance was also associated with the duration that water temperatures exceeded several temperature limits. The longer water temperatures of 21°C and 22°C were exceeded, the lower coho abundance was in Fortune Creek. Likewise, coho abundance decreased with longer continuous exposures to temperatures above 18°C and 19°C. For both total exceedance and continuous exceedance variables, the higher temperature value had a stronger effect on coho abundance.

Both average and minimum DO were significant predictors of coho abundance, but similar to Chinook, coho numbers decreased with increasing oxygen concentrations. Electrical conductivity was positively related to coho abundance.

Two interaction terms were significant predictors of coho abundance in Fortune Creek. There was a significant interaction between average maximum temperature and average DO. This interaction term was negatively related to coho abundance and indicates that the tolerance of coho to adverse temperatures varies under different DO levels (and vice versa). The second significant interaction term is between minimum temperature and pH. Neither one of the variables was significant on their own; however, their interaction term was negatively related to coho abundance.

Table 4.6: Regression estimates of Poisson mixed-modeling of coho salmon counts in Fortune Creek

Variable	Regression Slope Estimate	p-value[a]
Absolute weekly maximum temperature	-0.31	< 0.001
Average weekly maximum temperature	-0.31 to -0.29	0.008
Average weekly mean temperature	-0.45	0.003
Weekly total hours > 21°C	-0.04	0.024
Weekly total hours > 22°C	-0.07	0.018
Weekly continuous hours > 18°C	-0.14	< 0.001
Weekly continuous hours > 19°C	-0.25	< 0.001
Minimum DO	-0.65 to -0.52	0.003
Average DO	-0.87	< 0.001
Conductivity	0.02	< 0.001
Average DO * average weekly maximum temperature	-0.19	0.023
pH * average weekly minimum temperature	-0.79	0.013

a   if variable present in several models the highest recorded p-value is reported

The retained models for predicting coho abundance in Fortune Creek are presented in Table 4.7. They are presented in order from best to worst fit, based on the AIC. The best model based on AIC contains an interaction term between average maximum temperature and average DO (Model COP1). This model had the lowest AIC and the lowest PWRSS. The three univariate models selected contained absolute maximum temperature (Model COP8), continuous hours > 18°C (Model COP9), and average maximum temperature (Model COP10) as predictor variables. While these have the lowest AIC values, their predictive capabilities are better than some of the multivariate models (indicated by PWRSS).

Table 4.7: Poisson models for predicting coho counts in Fortune Creek.

Model		AIC	Log Likelihood	Deviance	Deviance Reduction[a]	PWRSS[b]
COP1	–22.76 – 0.19 (Average Max. Temp x Average DO) ± 7.40	123.0	-55.5	111	44%	52
COP2	13.40 – 0.29 (Average Max. Temp.) – 0.87 (Average DO) ± 8.07	137.5	-63.8	127.5	35%	94
COP3	–3.26 – 0.25 (Continuous Hrs > 19°C) + 0.02 (Conductivity) ± 4.03	143.1	-66.6	133.1	32%	85
COP4	–55.99 – 0.79 (Min. Temp. x pH) ± 10.98	149.9	-68.9	137.9	30%	84
COP5	12.42 – 0.45 (Mean Temp.) – 0.65 (Min. DO) ± 6.86	157.5	-73.8	147.5	25%	153
COP6	4.86 – 0.07 (Total Hrs > 22°C) – 0.52 (Min. DO) ± 7.09	158.1	-74.0	148.1	25%	141
COP7	5.41 – 0.04 (Total Hrs > 21°C) – 0.58 (Min. DO) ± 7.33	160.4	-75.2	150.4	24%	145
COP8	6.48 – 0.31 (Absolute Max. Temp.) ± 5.90	162.5	-77.2	154.4	22%	98
COP9	1.39 – 0.14 (Continuous Hrs > 18°C) ± 4.09	180.7	-86.4	172.7	12%	118
COP10	5.94 – 0.31 (Average Max. Temp.) ± 5.22	186.2	-89.1	178.2	9%	117

a   Reduction in deviance compared to a null model containing only the random effects and an intercept
b   Penalized weighted residual sum of squares

As for Chinook, figures are only presented for the univariate models of coho abundance (Figure 4.17). Absolute and average maximum temperatures were both significant predictors of coho in Fortune Creek. On average, predicted coho abundance declined to zero at an absolute maximum temperature of 21°C; however given the large amount of variation among sites and weeks, this could reach all the way up to 40°C (Model COP8, Figure 4.17).

When weekly average maximum temperatures exceeded approximately 19.2°C, predicted coho abundances became zero (Model COP10). Similar to Model COP8, there is a large amount of variation among sites and weeks, and this value can reach up to 36°C. Even though absolute and average maximum temperatures were significant predictors of coho abundance, they only led to an additional reduction in deviance of 22% and 9%, respectively, over a null model which contains only random effects and no fixed effects. In comparison, the best model, which included an interaction term of average DO and average maximum temperature led to a reduction in deviance by 44% over the null model.

Continuous hours > 18°C in the week prior to sampling were a significant predictor of coho abundance (Model COP9). On average, the predicted number of coho declined to zero when water temperatures of 18°C were exceeded for a continuous period of 10 hours. Considering variation between sites and weeks, this period may reach up to a maximum of 40 hours. This indicates that coho were relatively sensitive to continuous exposures to temperatures above 18°C. The residuals do not show any obvious patterns when plotted against observation number or fitted values (Figure 4.18).
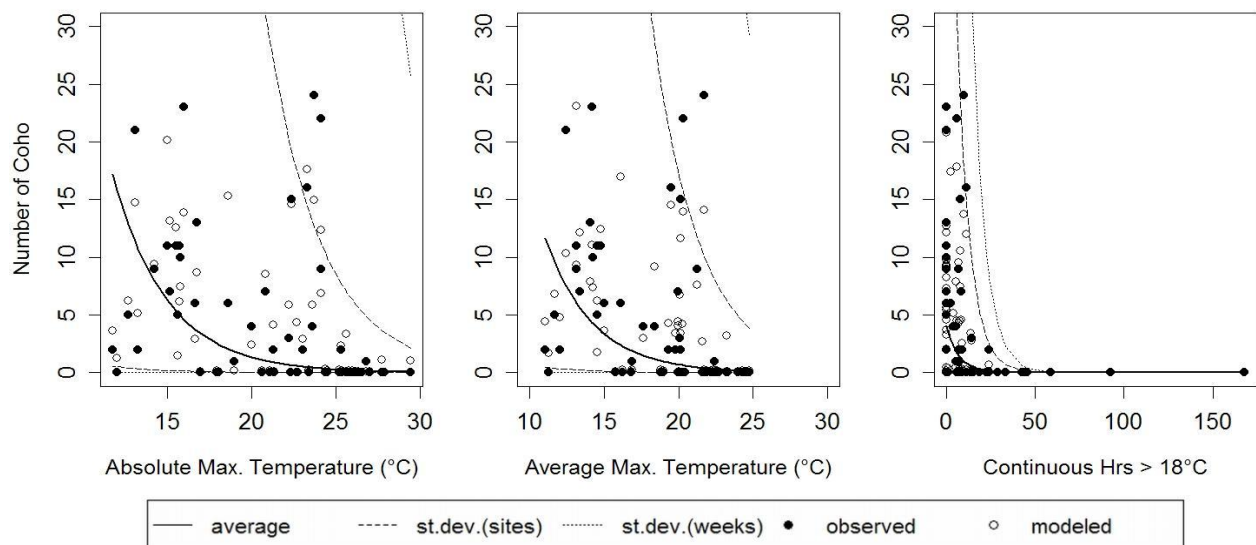
Figure 4.17: GLMM predicted coho abundance in Fortune Creek in relation to absolute maximum and average maximum water temperatures and continuous hours > 18°C.
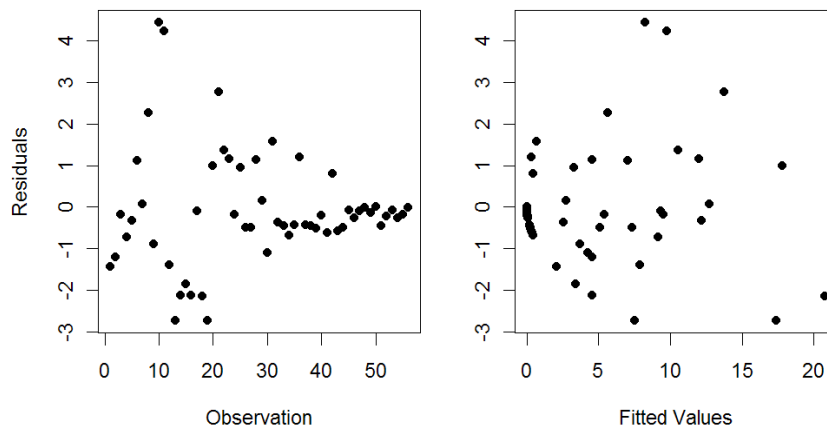
Figure 4.18: Plot of residuals for coho Model COP10 (Total hrs > 18°C) by observation number and against fitted values.

4.2.1.3 Rainbow

Only two models for predicting age 1+ rainbow trout abundance were retained after accounting for residual autocorrelation and overdispersion (Table 4.8). These were univariate models containing absolute (p = 0.035) and average weekly maximum (p = 0.032) temperatures. Both variables were negatively related to rainbow trout abundance. The regression slope estimate is slightly larger for average maximum temperature (-0.18) than absolute maximum temperature (-0.14), but both models are very similar in terms of fit.

Table 4.8: Poisson models for predicting rainbow trout abundance in Fortune Creek

	Model	AIC	Log Likelihood	Deviance	Deviance Reduction[a]	PWRSS[b]
RP1	2.44 – 0.14 (Absolute Max. Temp.) ± 3.68	118.2	-55.1	110.2	11%	74
RP2	2.78 – 0.18 (Average Max. Temp.) ± 3.55	119.3	-55.7	111.3	10%	76

a   Reduction in deviance compared to a null model containing only the random effects and an intercept
b   Penalized weighted residual sum of squares

Based on Model RP1, rainbow trout abundance approached zero when the absolute maximum temperature reached 17.5°C, but this threshold could reach up to 44°C when the substantial variation between sites and weeks was considered. For average maximum temperature, the threshold was 15.5°C, but reached up to 35°C.

Figure 4.19 indicates that the average prediction curve from either model is much lower than the observed rainbow numbers and there is a large amount of variation among sites and weeks. Adding the temperature variables only resulted in a 10% to 11% reduction in deviance over a null model, which indicates that the random effects have a much greater effect on the predictions than the temperature variables in both models. Nonetheless, both variables were significant predictors of rainbow trout abundance in Fortune Creek. Residual plots for Model RP1 (Figure 4.20) indicate that there are no obvious residual patterns.
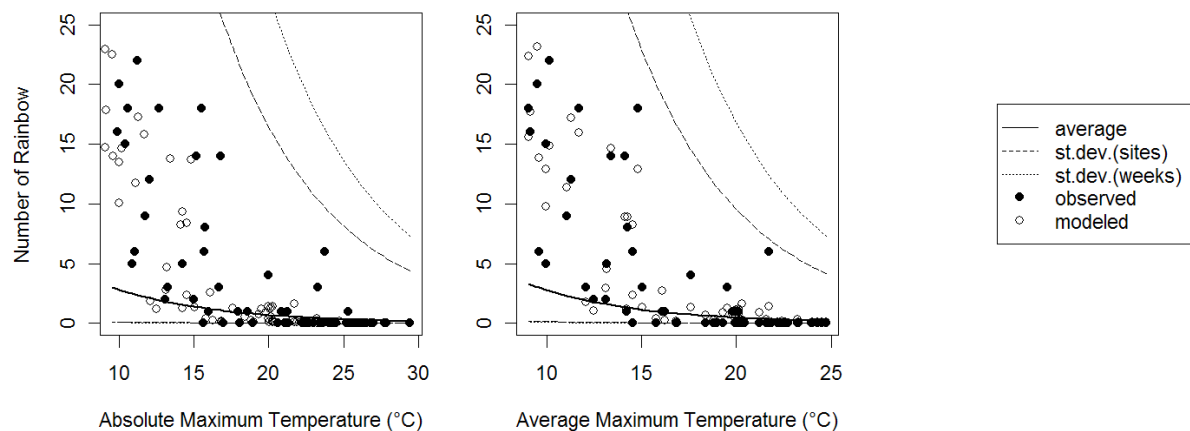
Figure 4.19: Average GLMM predicted rainbow trout abundance in Fortune Creek in relation to absolute maximum and average maximum water temperatures
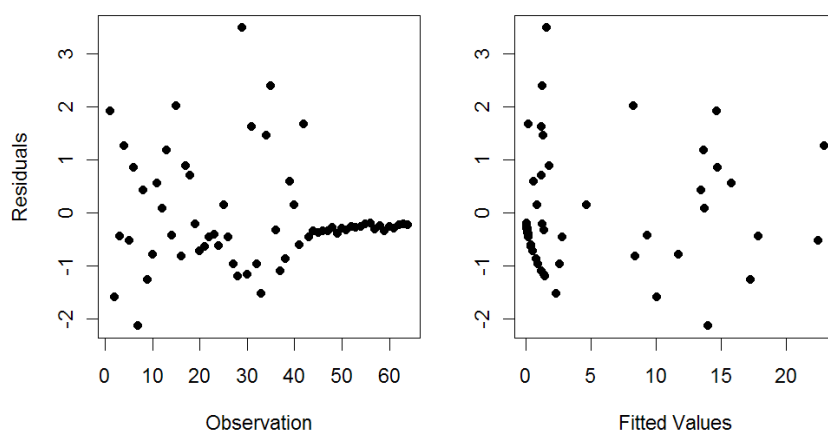
Figure 4.20: Plot of residuals for rainbow Model RP1 by observation number and against fitted values.

4.2.2 Logistic Regression

All logistic regression models included site and week as random effects. Similar to the Poisson regression models, backward selection was not conducted due to extensive multicollinearity among the explanatory variables. Many variables initially appeared to be significant predictors of salmonid presence/absence but were rejected when residual autocorrelation was detected.

4.2.2.1 Chinook

Six models for predicting the probability of Chinook presence were retained after accounting for residual autocorrelation (Table 4.9). The best model with the lowest AIC shows that the probability of Chinook presence decreases with increasing average weekly minimum temperature, and increases when more cover is available. Incorporating these explanatory variables into the model led to a 47% reduction in residual deviance compared to a model containing only an intercept and the random effects.

Table 4.9: Logistic regression models for predicting the probability of Chinook presence in Fortune Creek.

	Model (logit (p) =)	AIC	Log Likelihood	Deviance	Deviance Reduction[a]	PWRSS[b]
CHL1	15.16 – 1.84 (Min. Temp.) + 0.20 (Cover) ± 5.59	35.1	-13.5	27.1	47%	15.8
CHL2	20.76 – 1.38 (Mean Temp.) ± 6.70	36.3	-15.1	30.3	41%	15.7
CHL3	22.42 – 1.79 (Minimum Temp.) ± 9.11	38.5	-16.3	32.5	38%	15.8
CHL4	2.84 – 0.69 (Continuous Hrs > 20°C) ± 6.02	41.5	-17.7	35.5	31%	22.2
CHL5	2.08 – 0.15 (Continuous Hrs > 17°C) ± 4.39	48.6	-21.3	42.6	17%	32.8
CHL6	9.89 – 1.02 (Min. DO) ± 16.93	48.9	-21.4	42.9	16%	33.1

a   Reduction in deviance compared to a null model containing only the random effects and an intercept
b   Penalized weighted residual sum of squares

Both weekly average mean and minimum temperatures were significantly and negatively related to the probability of Chinook presence (Figure 4.21). These were also significant predictors in the Poisson regression models. On average, Chinook salmon were predicted to be absent (i.e., probability < 50%) where average mean temperatures were above 15.1°C, reaching up to mean temperatures of 19.9°C when variation between sites and weeks was considered. This is similar to the results of the Poisson regression, which predicted that Chinook abundance would approach zero at mean temperatures above 14.4°C reaching up to 22.0°C.

Model CHL3 predicted that Chinook salmon are absent where average minimum temperatures exceeded 12.5°C. However, given the variation in predicted probabilities among sites, this threshold could reach up to 17.6°C. These estimated values are similar to the Poisson regression results, which predicted Chinook abundance to approach zero at minimum temperatures of 12.0°C (up to 17.1°C).

Continuous hours > 17°C and 20°C were both significant predictors of Chinook salmon presence in Fortune Creek with decreasing probabilities the longer these thresholds were exceeded (Figure 4.22). However, Chinook were more sensitive to temperatures exceeding 20°C than 17°C, as indicated by a steeper regression slope. There was also substantially less variation among sites and weeks for the 20°C model (Model CHL4). Overall, Chinook salmon were predicted to be absent where water temperatures exceeded 17°C for more than 13.7 hours at a time (reaching up to 44.9 hours). Model CHL4 predicted Chinook to be absent where water temperatures exceeded 20°C for more than 4.1 hours at a time (reaching up to 17.6 hours). Besides cover, the only non-temperature related significant predictor was minimum DO, which was negatively related to Chinook presence (Chinook presence less likely at increasing DO levels).
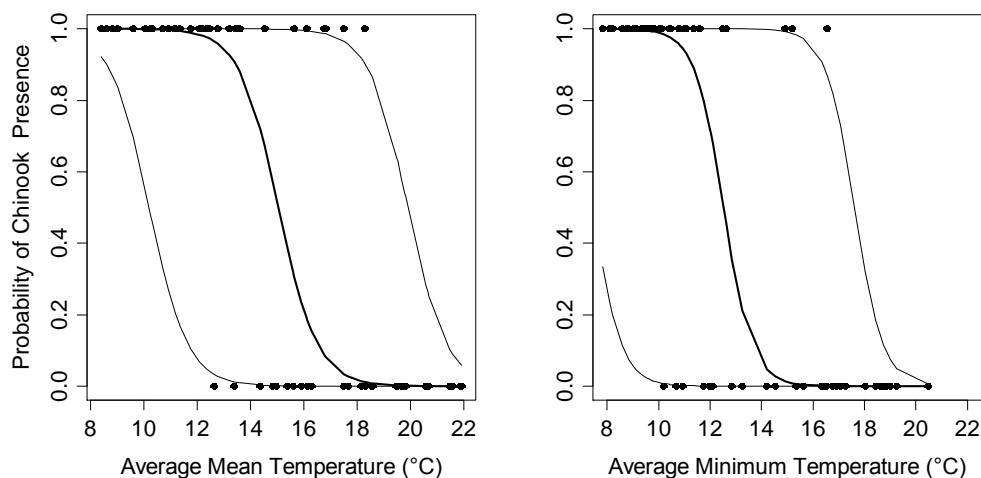
Figure 4.21: Average GLMM predicted probability of Chinook salmon presence in Fortune Creek (thick line) in relation to average mean and minimum water temperatures. Thin lines represent additional variation in probabilities between sites and weeks. The black circles represent observed Chinook presence (1) and absence (0).
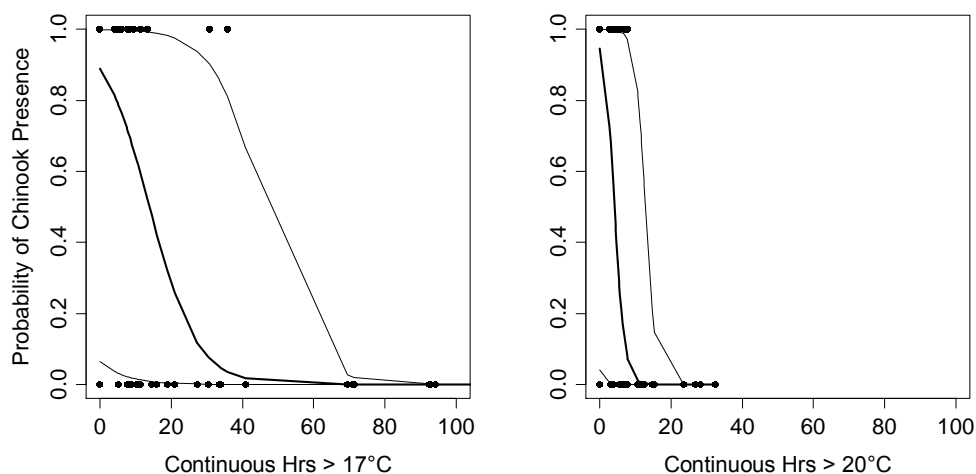
Figure 4.22: Average GLMM predicted probability of Chinook salmon presence in Fortune Creek (thick line) in relation to continuous periods of water temperatures exceeding 17°C and 20°C. The thin lines represent additional variation in probabilities between sites and weeks. The black circles represent observed Chinook presence (1) and absence (0).

4.2.2.2 Coho

Only two models for predicting the probability of coho presence in Fortune Creek were retained after discarding models with residual autocorrelation (Table 4.10). Both average and absolute maximum temperature in the week preceding sampling were negatively related to the probability of coho presence. Both variables were also significant in the Poisson regression models.

Table 4.10: Logistic regression models for predicting the probability of coho presence in Fortune Creek.

	Model (logit (p) =)	AIC	Log Likelihood	Deviance	Deviance Reduction[a]	PWRSS[b]
COL1	14.80 – 0.73 (Average Max. Temp.) ± 10.31	41.7	-16.9	33.7	21%	21.6
COL2	9.53 – 0.40 (Absolute Max. Temp.) ± 9.79	44.1	-18.1	36.1	15%	24.9

a   Reduction in deviance compared to a null model containing only the random effects and an intercept
b   Penalized weighted residual sum of squares

Overall, coho were predicted to be absent (probability < 50%) when average maximum temperatures exceeded 20.3°C (Figure 4.23). This is similar to the predicted zero-abundance threshold from the equivalent Poisson model (19.2°C, Model COP10). The upper limits for both models are also quite similar at 34.4°C (logistic) and 36°C (Poisson). The validity of these upper limits is questionable as they greatly exceed the upper thermal limits for coho reported in the literature.

Model COL2 predicts that coho were absent when absolute maximum temperatures exceeded 24.2°C, but this threshold reached up to 48°C when variation among sites and weeks was considered. The threshold of 24.2°C is several degrees larger than the estimate from the Poisson model (21°C). The

maximum limit estimated from the logistic model (48°C) was also substantially higher than from the Poisson regression (40°C). Regardless, both limits are far beyond the lethal limits reported in literature. It is evident that similar to the coho Poisson models, the logistic models were plagued by a large amount of uncertainty. The thin lines in Figure 4.23 clearly illustrate the large amount of variation between sites and sample weeks.
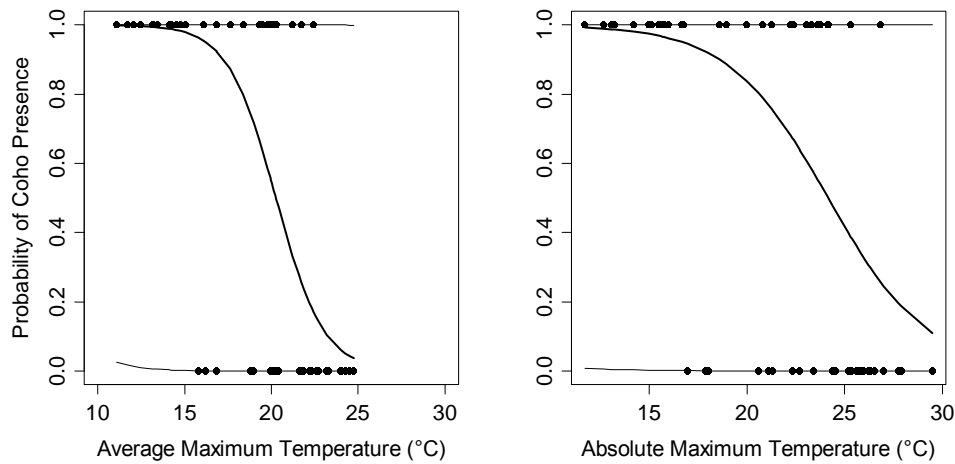
Figure 4.23: Average GLMM predicted probability of coho salmon presence in Fortune Creek (thick line) in relation to absolute and average maximum temperatures. The thin lines represent additional variation in probabilities between sites and weeks. The black circles represent observed Chinook presence (1) and absence (0).

4.2.2.3 Rainbow

Even though many variables (including most of the temperature variables) appeared significantly related to the probability of rainbow trout presence/absence, all were deemed invalid due to significant autocorrelation in the model residuals. As a result, no logistic regression models for rainbow trout were retained.

4.2.3 Generalized Estimating Equations

GEE models with AR1 autocorrelation structure yielded poor results, showing substantial residual patterns and significant autocorrelation in the residuals. This may be because the AR1 correlation structure only models autocorrelation at lag 1 (adjacent observations), while many models showed significant autocorrelation at higher lags.

4.3   Modeling of Groundwater - Surface Water Interactions

During modeling it became evident that in general, vertical fluxes seemed to determine simulated average temperatures whereas thermal conductivity seemed to influence the extent to which daily

water temperature variations in the stream penetrated into the bed. The influence of groundwater flux on streambed temperatures became negligible at fluxes below approximately 1×10^{-7} m/s (Section 2.2.1).

Model results of the simulations at all sites are presented in Table 4.11 (summer) and Table 4.12 (winter). Simulated flux directions were downward at sites 1 and 2 (streamwater flowing into the bed) and upward at sites 3 to 8 (groundwater discharging to the creek). Thermal conductivities were adjusted to obtain a good fit to the observed temperature profile, and were generally higher in the upstream areas where the streambed consisted of boulders and cobbles. In the valley-bottom reaches where the streambed consists of silt, clay and organic materials, the best fits were achieved at lower thermal conductivities.

Table 4.11: Simulated summer streambed vertical flux velocities in Fortune Creek.

Site	Vertical Flux (m/s) Summer	RMSE	Estimated Hydraulic Conductivity (m/s)	Flow Direction	Thermal Conductivity (W/mK)	Figure
1	7.7×10^{-7} to 2.2×10^{-6}	0.05	--	Down	3.5	Figure 4.28 Figure 4.29
2	3.5×10^{-6}	0.22	--	Down	2.7	Figure 4.30
3	negligible	--	--	Neutral	1.5	Figure 4.24
4	5.7×10^{-7}	0.12	1×10^{-6}	Up	1.5	Figure 4.26
5	4.5 to 6.6×10^{-7}	0.28	--	Up	1.5	Figure 4.32
6	1.1 to 2.0×10^{-7}	0.12	--	Up	1.5	Figure 4.34
7	1.1×10^{-7}	0.09	--	Up	1.0	Figure 4.36
8	2.3×10^{-7}	0.12	--	Up	1.75	Figure 4.38

Table 4.12: Simulated winter streambed vertical flux velocities in Fortune Creek.

Site	Vertical Flux (m/s)	RMSE	Estimated Hydraulic Conductivity (m/s)	Flow Direction	Thermal Conductivity (W/mK)	Figure
1	No good fit achieved	--	--	--	--	--
2	1.9×10^{-6}	0.08	--	Down	2.7	Figure 4.31
3	negligible	--	--	neutral	1.5	Figure 4.25
4	1.8×10^{-7}	0.02	4×10^{-7}	Up	1.5	Figure 4.27
5	2.0×10^{-7}	0.09	--	Up	1.5	Figure 4.33
6	5.7×10^{-7}	0.03	--	Up	1.5	Figure 4.35
7	9.5×10^{-8}	0.05	--	Up	1.0	Figure 4.37
8	1.7×10^{-7}	0.02	--	Up	1.75	Figure 4.39

Results from sites 3 and 4, where piezometer data existed, are discussed first. At site 3, streambed temperatures at 50 cm depth were quite variable through the summer modeling period, and daily fluctuations can clearly be seen in the data (Figure 4.24). The best fit was achieved at a streambed thermal conductivity of 1.5 W/mK.

Simulated summer streambed temperatures at site 3 were not particularly sensitive to modeled upward or downward groundwater fluxes. Adjusting seepage velocities upward or downward by three orders of magnitude only resulted in approximately 1°C change in simulated temperatures during warm periods. The effect was even smaller during cooler periods. The best fit to observed summer streambed temperatures at 50 cm depth was obtained when vertical groundwater flux was nearly zero (<10^{-9} m/s up or down). This is very similar to the flux value estimated from slug testing at this site (3 x 10^{-9} m/s), which suggests that site 3 was under neutral vertical flux conditions throughout the summer 2008 modeling period.

In winter, streambed temperatures at site 3 were equally unaffected by small adjustments in seepage velocities as in summer (Figure 4.25). The best fit (smallest RMSE) to the observed temperatures at 50 cm depth was obtained when groundwater flux was nearly zero, indicating that vertical flux velocities were very small during the winter 2009 modeling period.

Figure 4.24: Simulated and measured summer temperatures 50 cm below the streambed under various groundwater flow conditions at site 3.

Figure 4.25: Simulated and measured winter temperatures 50 cm below the streambed under various groundwater flow conditions at site 3.

Simulated and measured streambed temperatures at 50 cm depth at site 4 are presented in Figure 4.26. Streambed temperatures at this site were much more sensitive to groundwater seepage velocities than at site 3, although the thermal conductivity was set to the same at 1.5 W/mK. Adjusting upward or downward flux by one order of magnitude resulted in a change in simulated temperatures of approximately 1.5°C. The best fit was obtained when hydraulic conductivity at site 4 was set to 7×10^{-7} m/s, with an upward groundwater flux of 5.7×10^{-7} m/s. This is more than one order of magnitude larger than flux values estimated from slug testing (1.8×10^{-8} m/s). No acceptable fit could be obtained under simulated downward flux conditions.

The thermal data was consistent with physical data indicating that site 4 was under upward flux conditions during the 2008 summer season. These conditions continued through the winter modeling period, when upward groundwater flux was estimated at 1.8×10^{-7} m/s (Figure 4.27). As in summer, small changes in seepage velocity resulted in large differences in simulated temperatures.

Figure 4.26: Simulated and measured summer temperatures 50 cm below the streambed under various groundwater flow conditions at site 4.

Figure 4.27: Simulated and measured winter temperatures 50 cm below the streambed under various groundwater flow conditions at site 4.

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Groundwater flux simulations at the remaining sites contain a higher degree of uncertainty because the lower boundary conditions were estimated from data collected at sites 3 and 4. Simulations indicate that in the summer, downward fluxes at site 1 ranged from 7.7×10^{-7} m/s to 2.2×10^{-6} m/s. The modeling period at site 1 had to be broken up into two segments (July 18–29 and July 30–August 31) because an acceptable fit to the observed temperature profile could not be obtained at a constant flux over the entire summer modeling period (Figure 4.28 and Figure 4.29).

At site 2, downward flux was estimated at 3.5×10^{-6} m/s (Figure 4.30). Sites 1 and 2 are at the top (site 1) and bottom (site 2) of the alluvial fan that exists where Fortune Creek enters the flat valley bottom from the mountainside. The section between the two sites periodically goes dry during low flow seasons (summer and winter), which indicates that stream water may infiltrate into the streambed at the top end of the fan near site 1. The water reappears upstream of site 2 suggesting upward flow at this site; however no fit to the observed streambed temperature profile could be achieved under upward flux conditions. The simulated downward flow at site 2 was unexpected as there are a number of deep artesian wells in the vicinity of the site, indicative of an upward hydraulic gradient in the area. Piezometer installation at the site was attempted on several occasions but failed due to the cobbly streambed.

During summer, streambed fluxes were upward at sites 5 to 8, ranging from 6.6×10^{-7} m/s at site 5 to 1.1×10^{-7} m/s at sites 6 and 7. At sites 5 and 6, good fits to the observed streambed temperatures could only be achieved when fluxes were adjusted from higher to lower through the modeling period.

Simulation periods in winter differed between sites due to logger failures resulting from rodent activity and ice damage. At some sites, low temperatures led to battery failures and subsequently recorded data had to be discarded. Simulated winter fluxes were similar but slightly lower than those observed in the summer, particularly at sites 4 and 5. Estimated winter fluxes were slightly higher than in the summer at site 6. A good fit to the observed temperature profile could not be achieved at site 1.
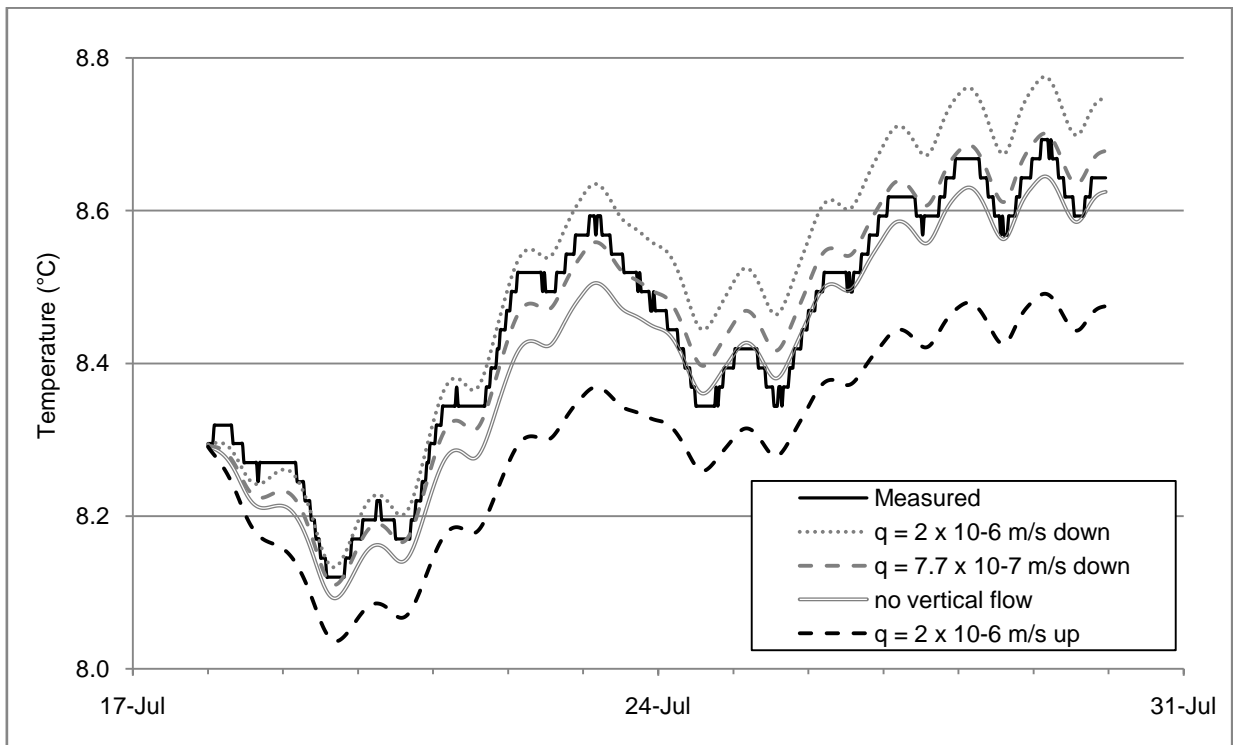
Figure 4.28: Simulated and measured summer temperatures 50 cm below the streambed under various groundwater flow conditions (Period 1) at site 1.
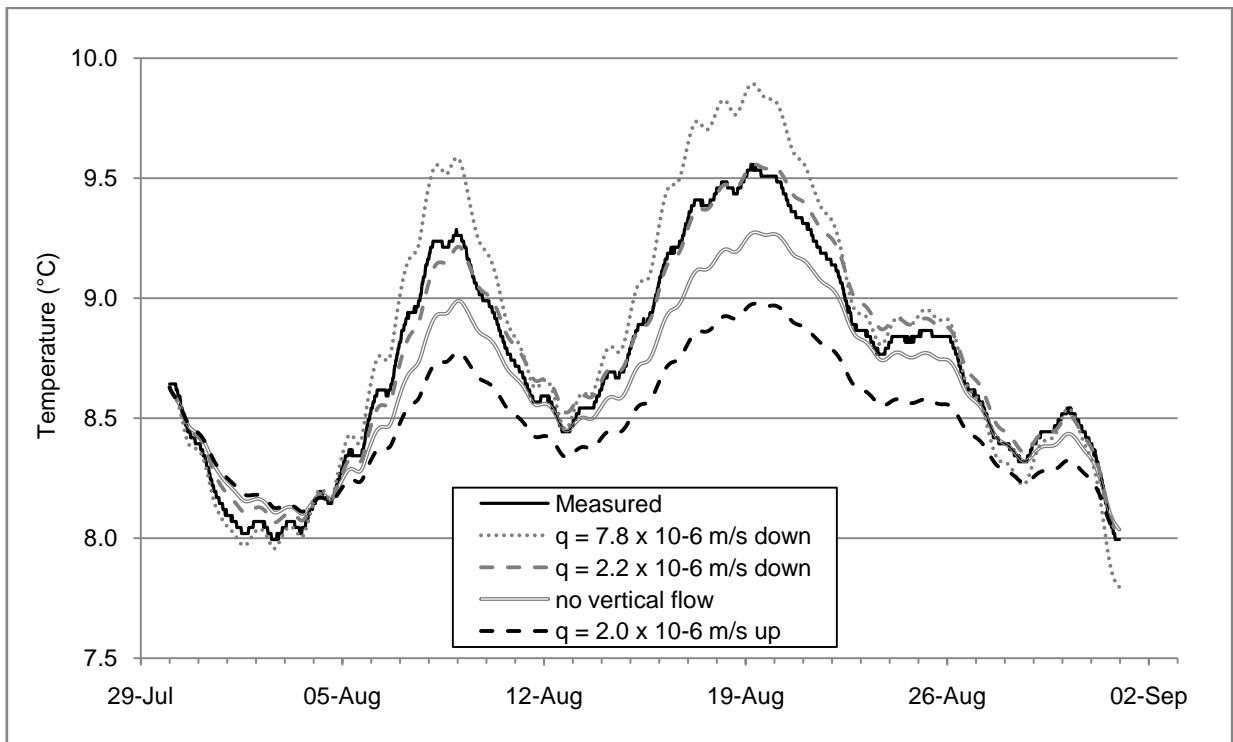
Figure 4.29: Simulated and measured summer temperatures 50 cm below the streambed under various groundwater flow conditions (Period 2) at site 1.
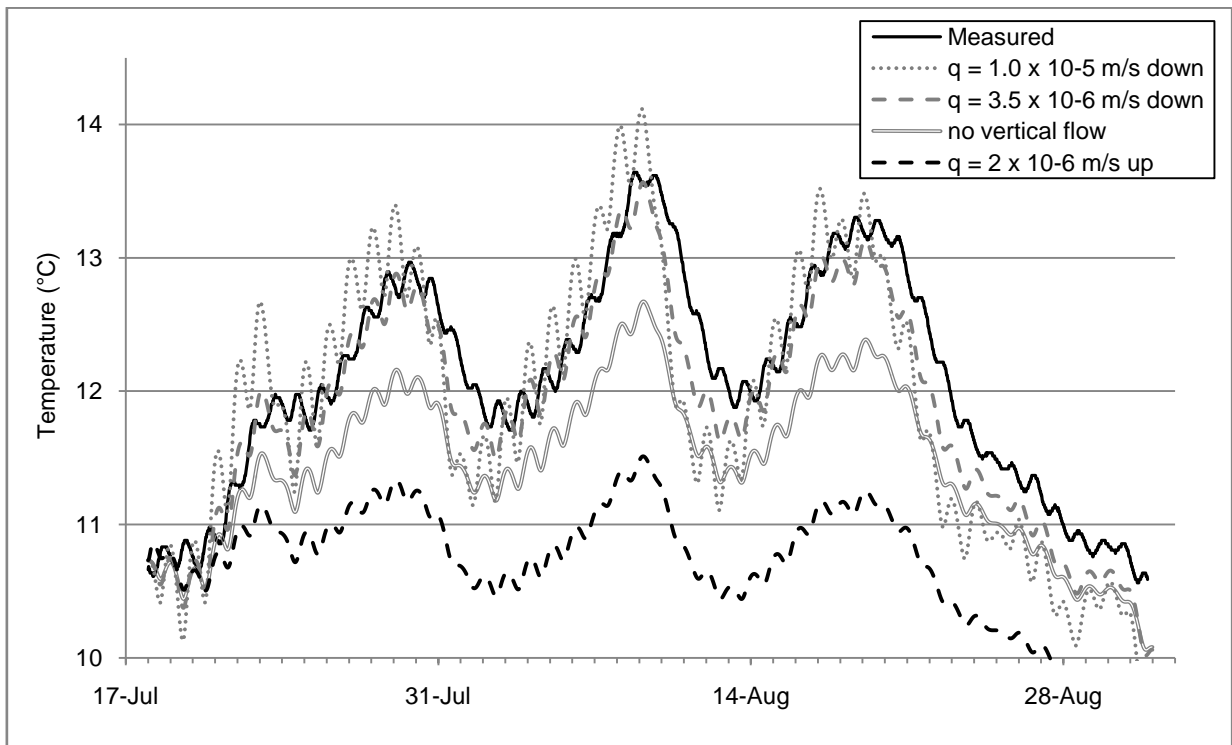
Figure 4.30: Simulated and measured summer temperatures 50 cm below the streambed under various groundwater flow conditions at site 2.
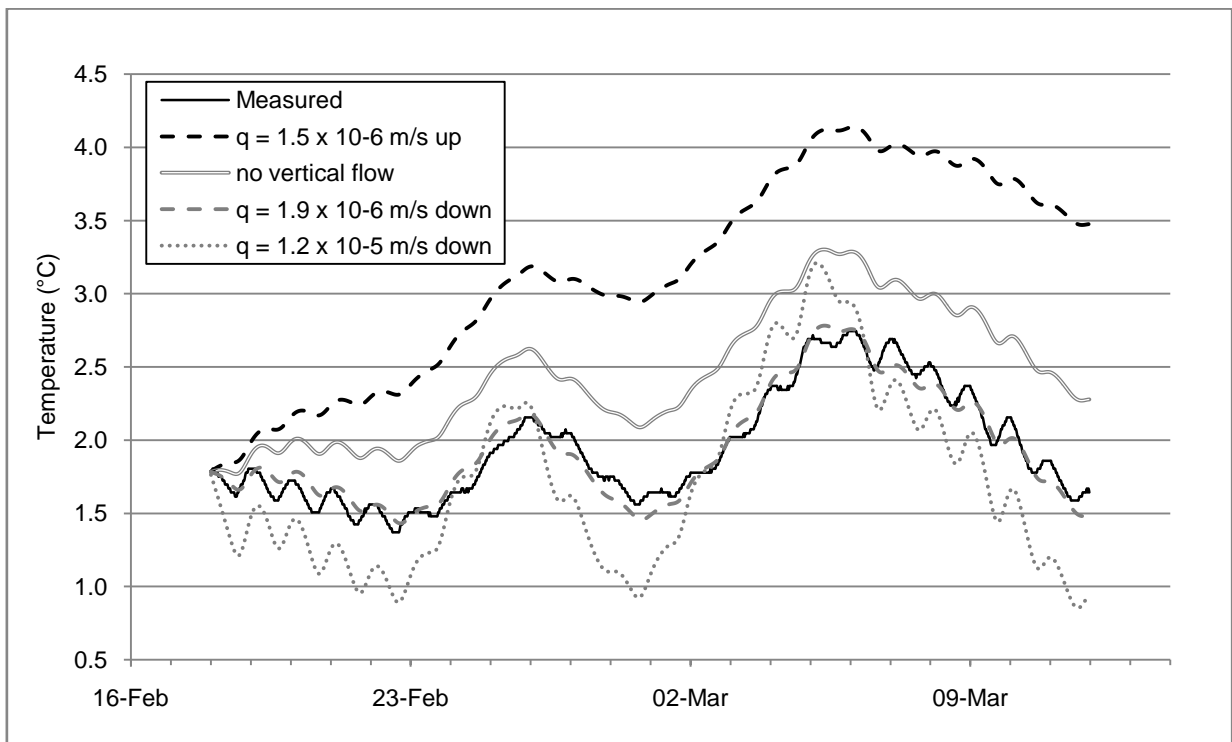
Figure 4.31: Simulated and measured winter temperatures 50 cm below the streambed under various groundwater flow conditions at site 2.
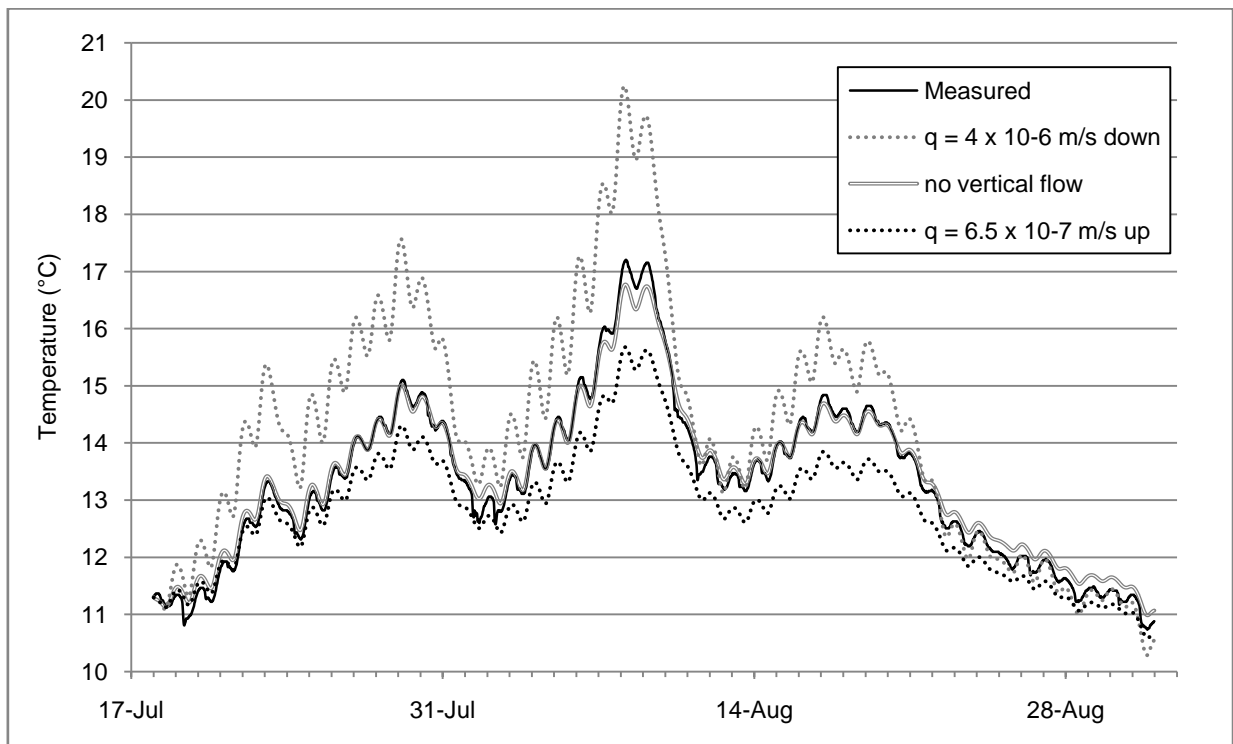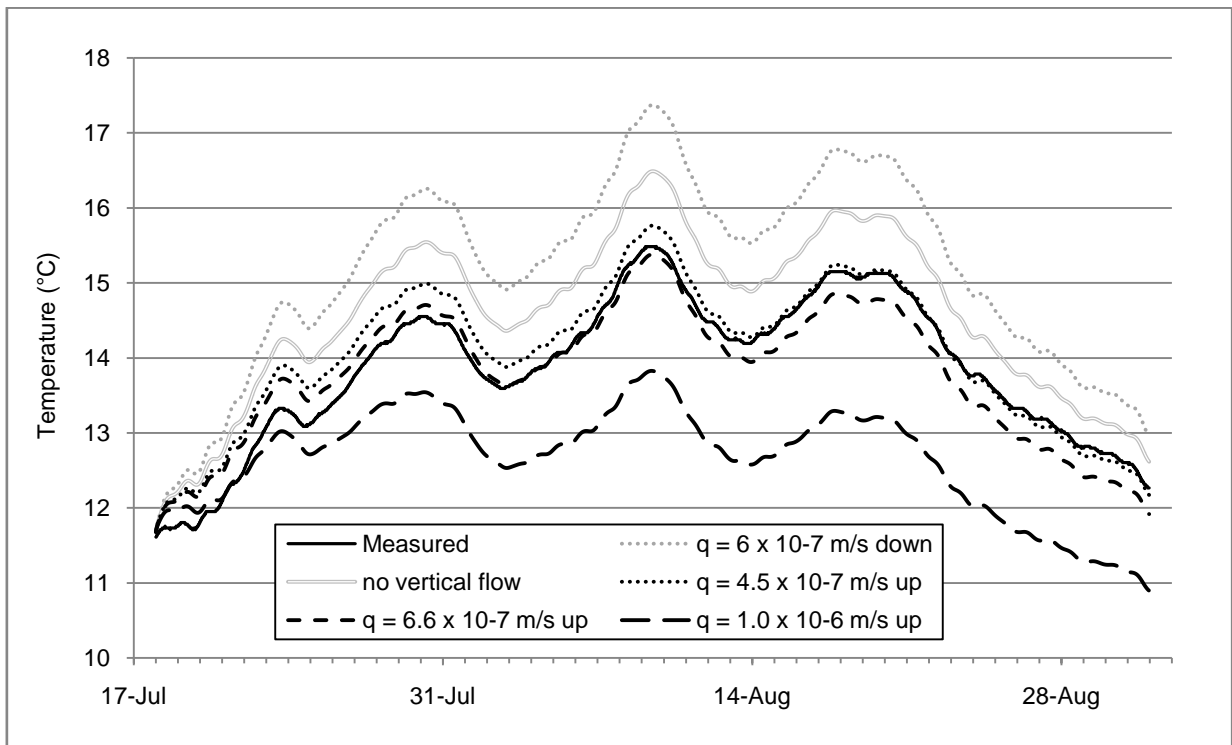
Figure 4.32: Simulated and measured summer temperatures at 50 cm below the streambed under various groundwater flow conditions at site 5.
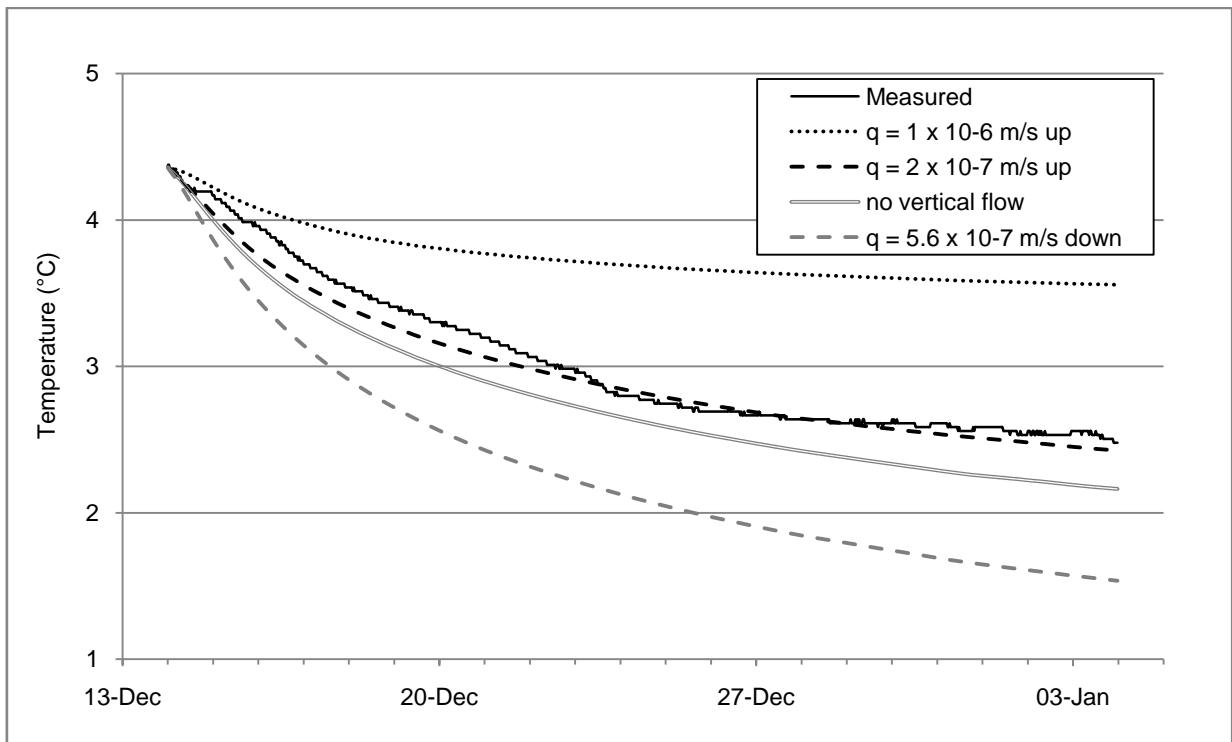
Figure 4.33: Simulated and measured winter temperatures at 50 cm below the streambed under various groundwater flow conditions at site 5.
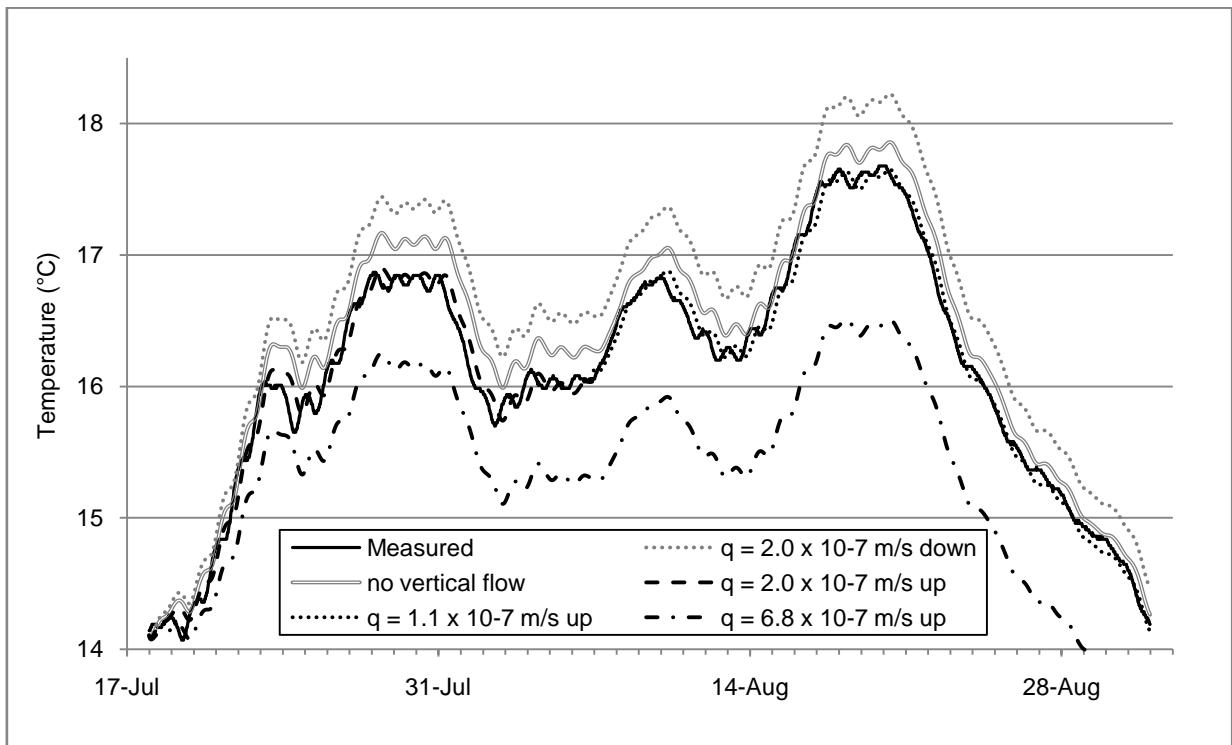
Figure 4.34: Simulated and measured summer temperatures 50 cm below the streambed under various groundwater flow conditions at site 6.
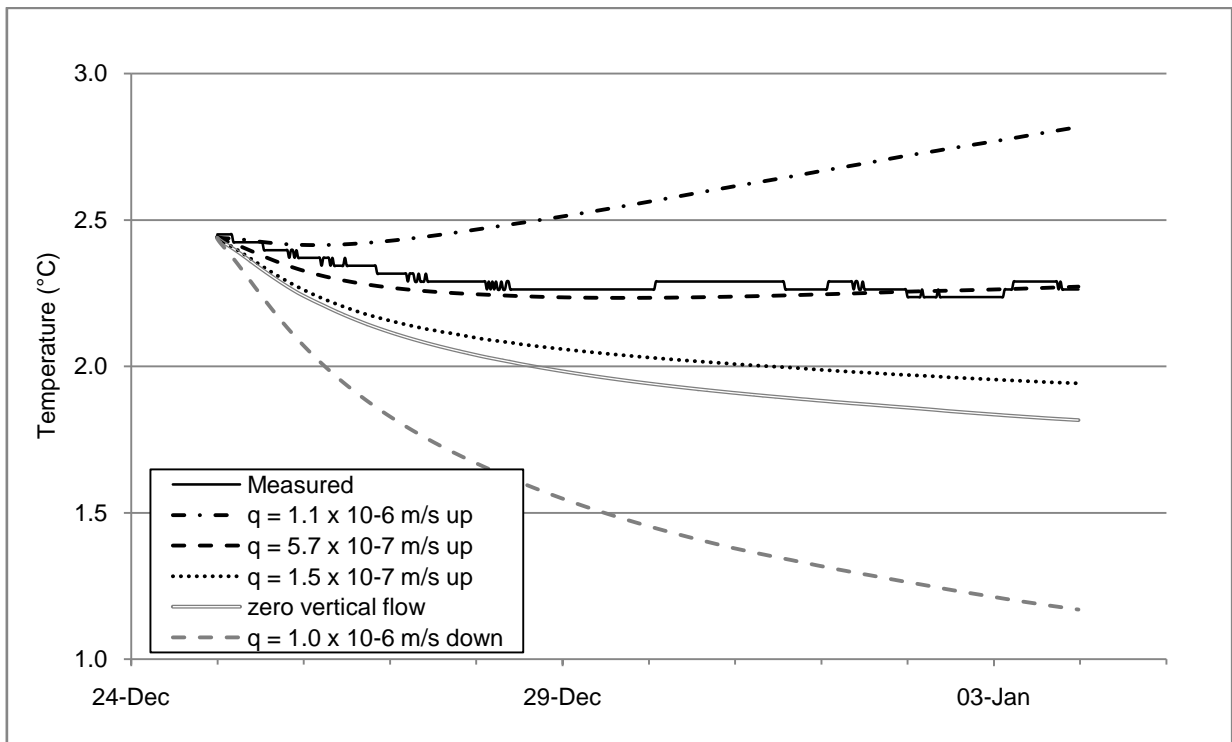
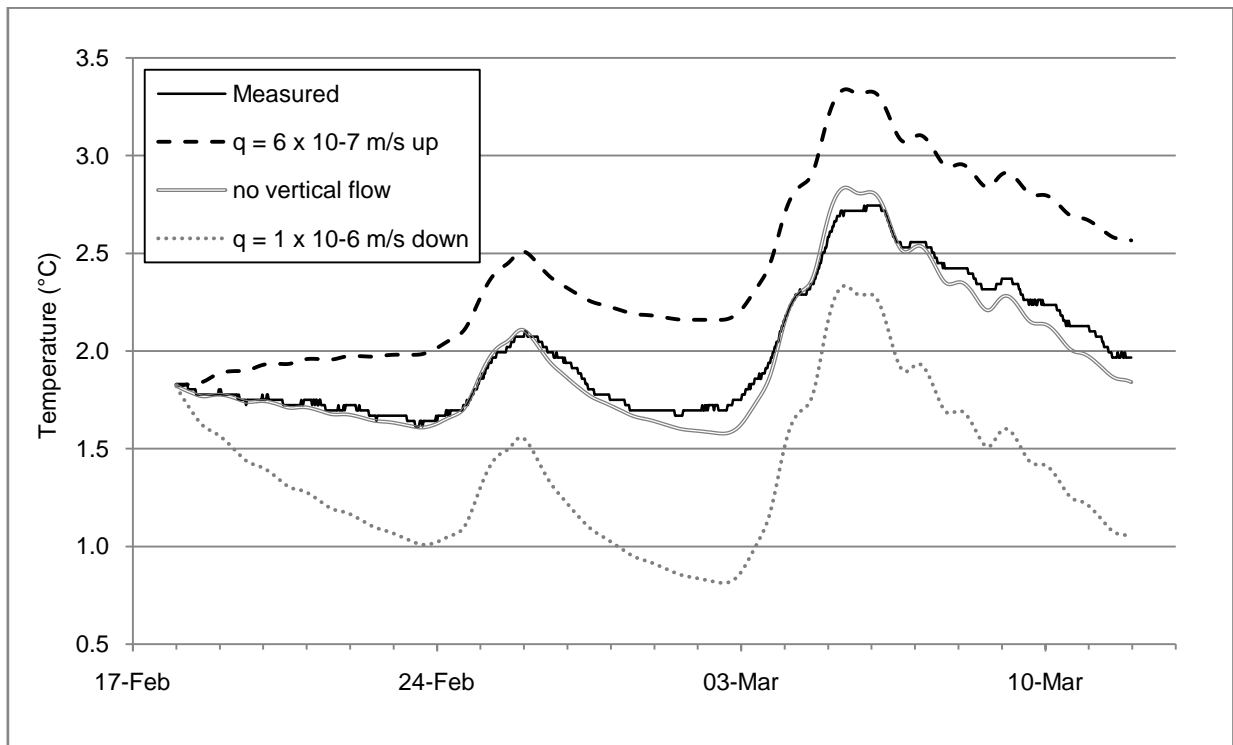**Figure 4.35: Simulated and measured winter temperatures 50 cm below the streambed under various groundwater flow conditions at site 6.**

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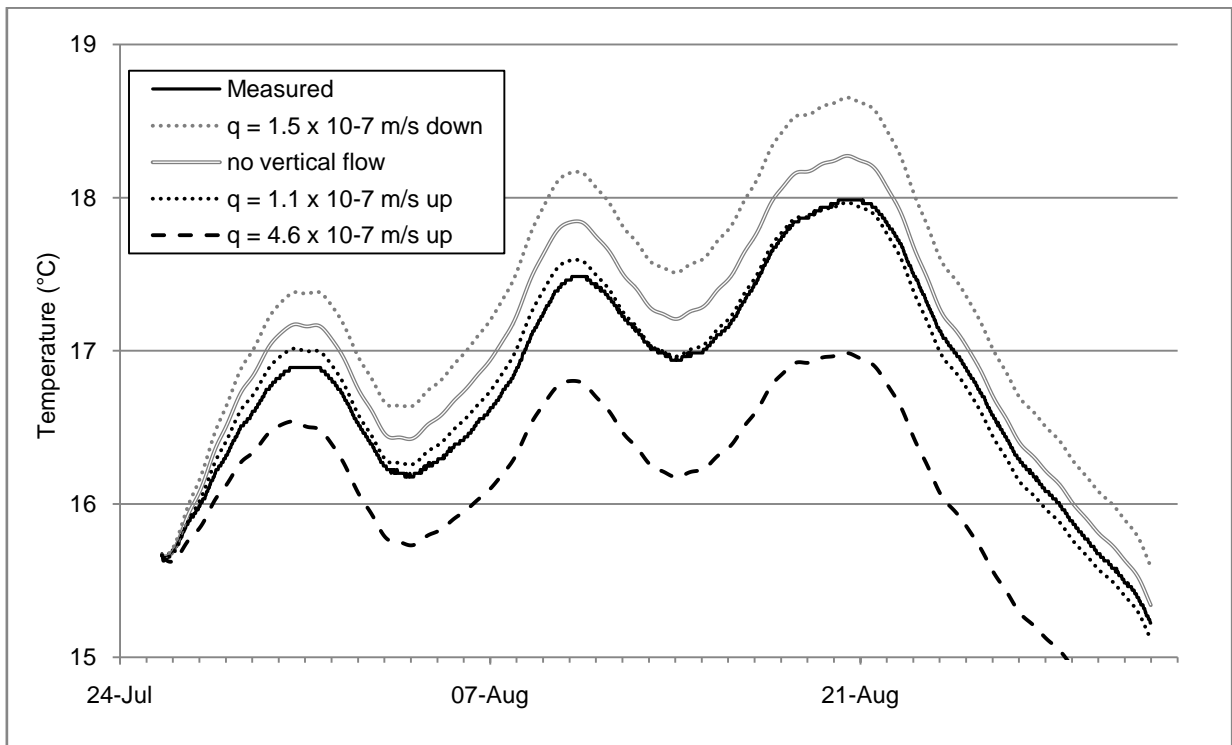
Figure 4.36: Simulated and measured summer temperatures 50 cm below the streambed under various groundwater flow conditions at site 7.
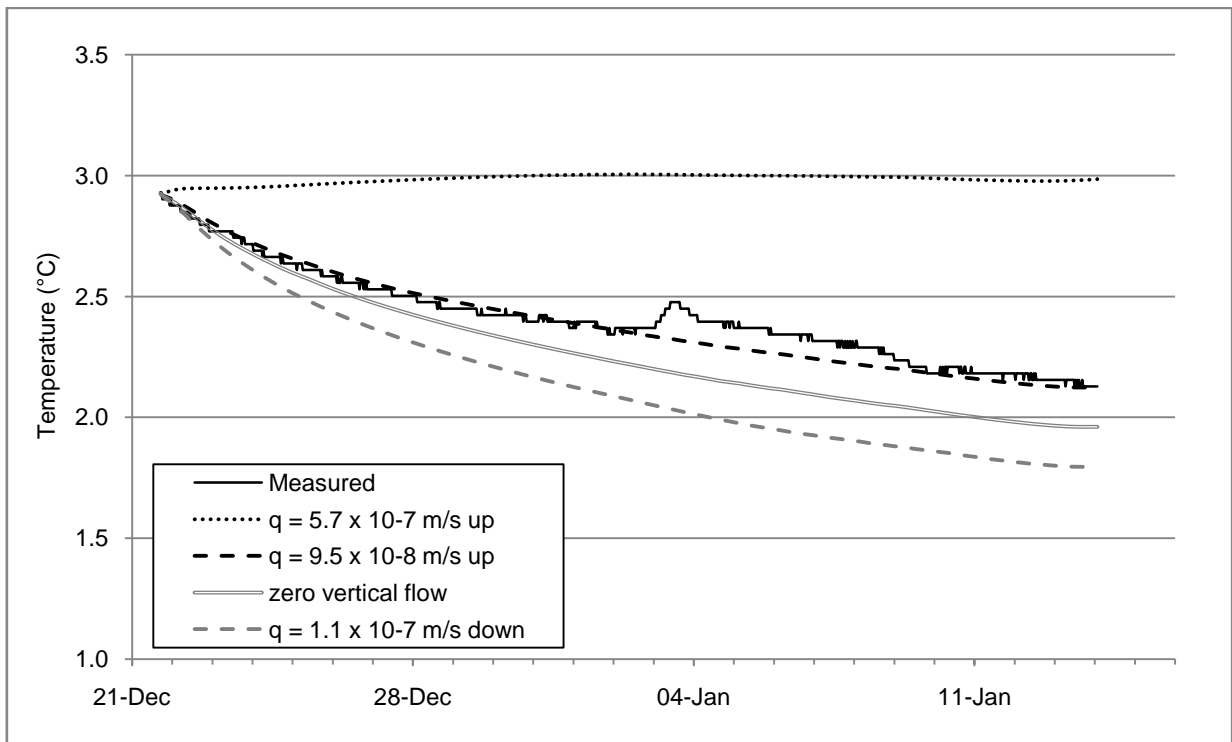
Figure 4.37: Simulated and measured winter temperatures 50 cm below the streambed under various groundwater flow conditions at site 7.
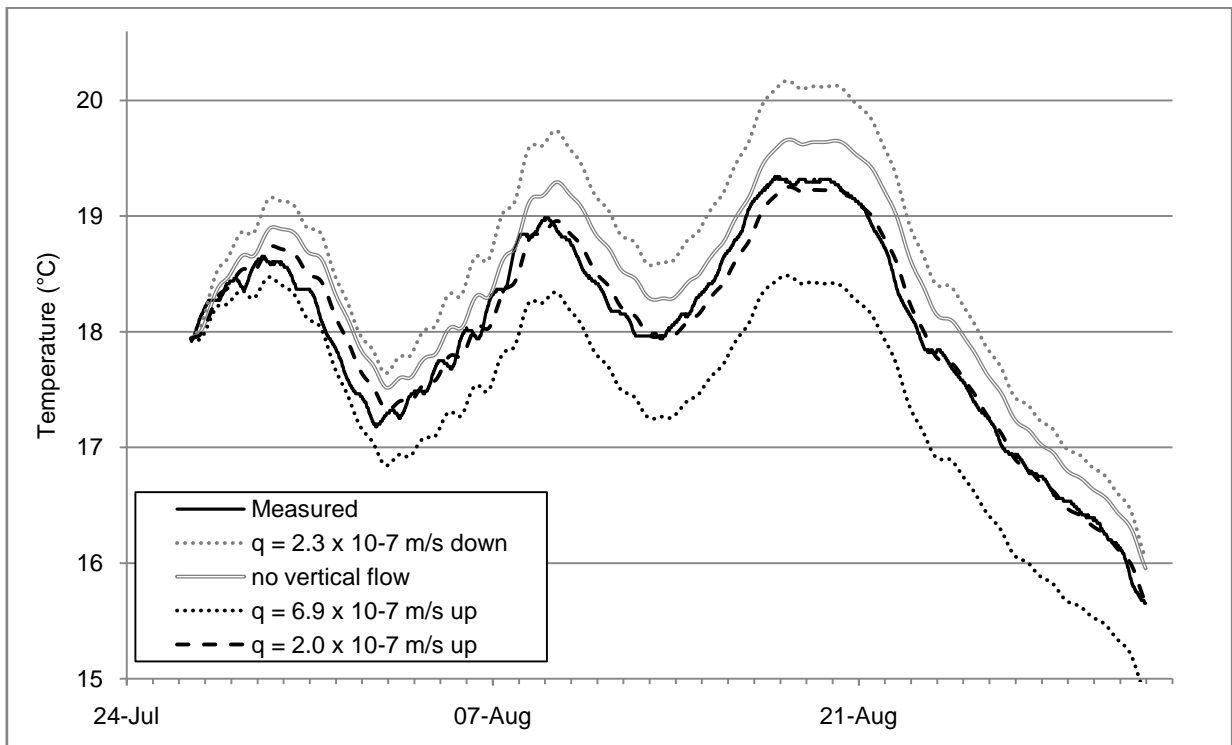
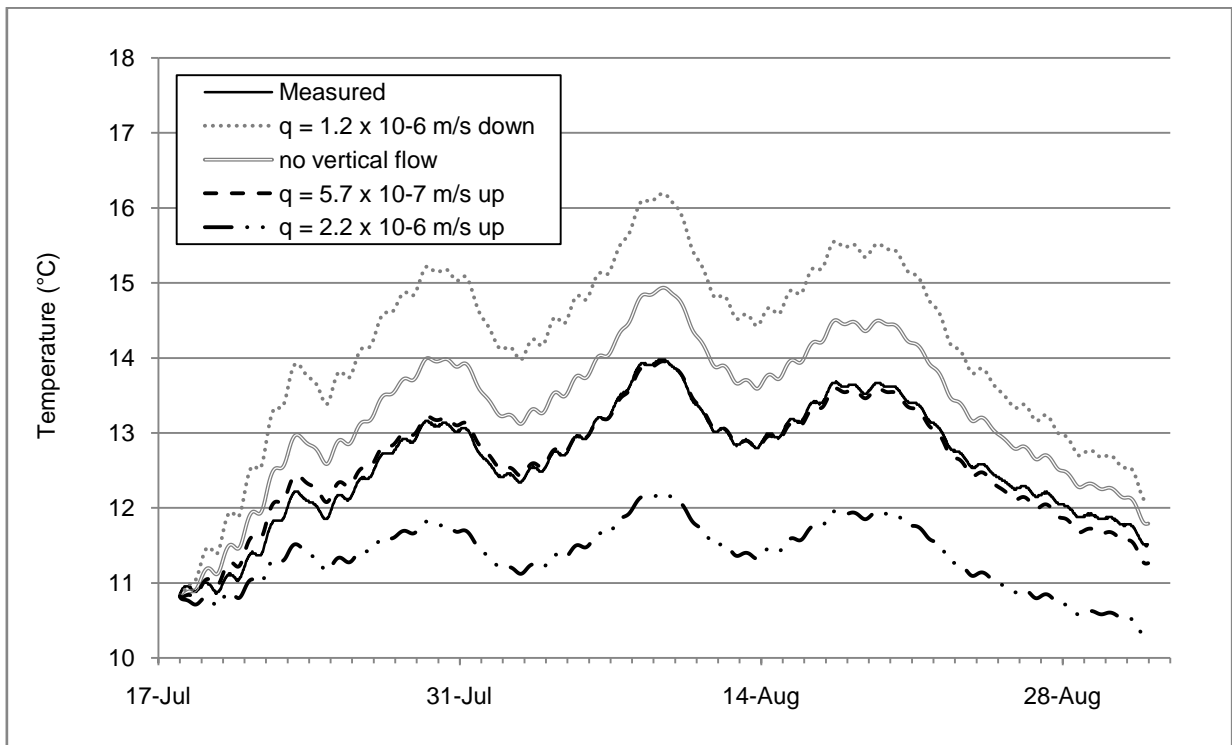**Figure 4.38: Simulated and measured summer temperatures 50 cm below the streambed under various groundwater flow conditions at site 8.**
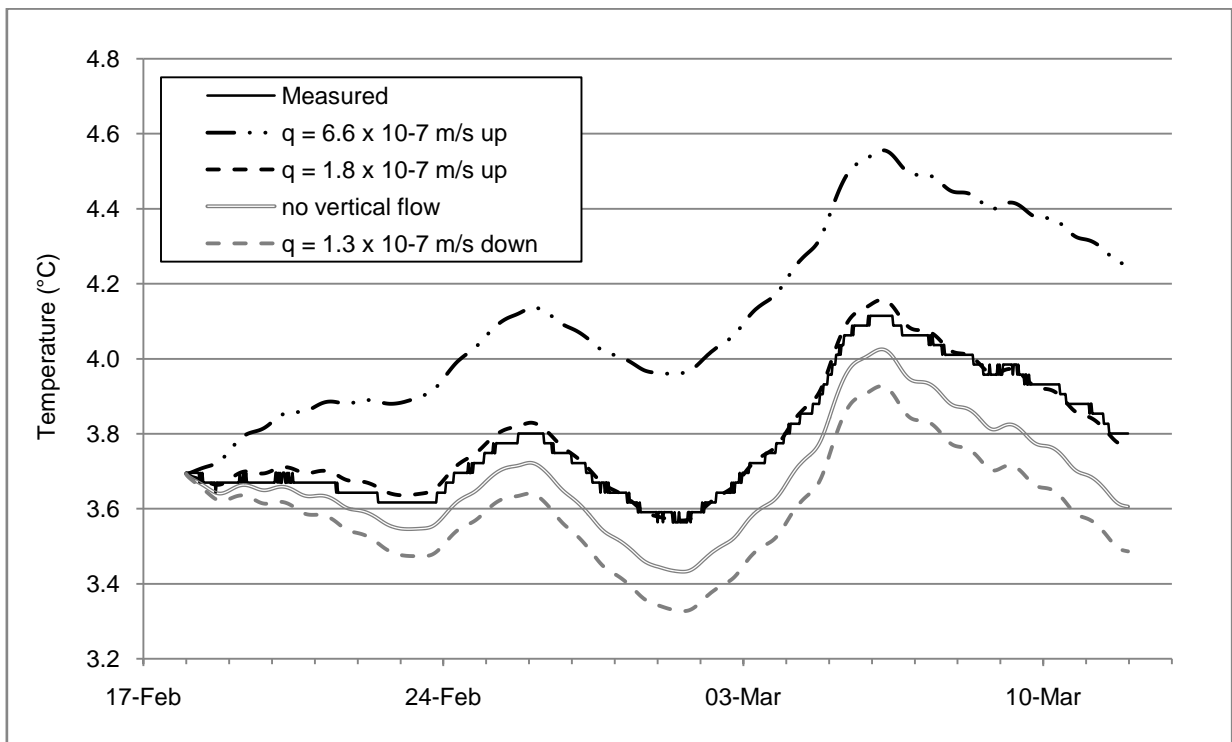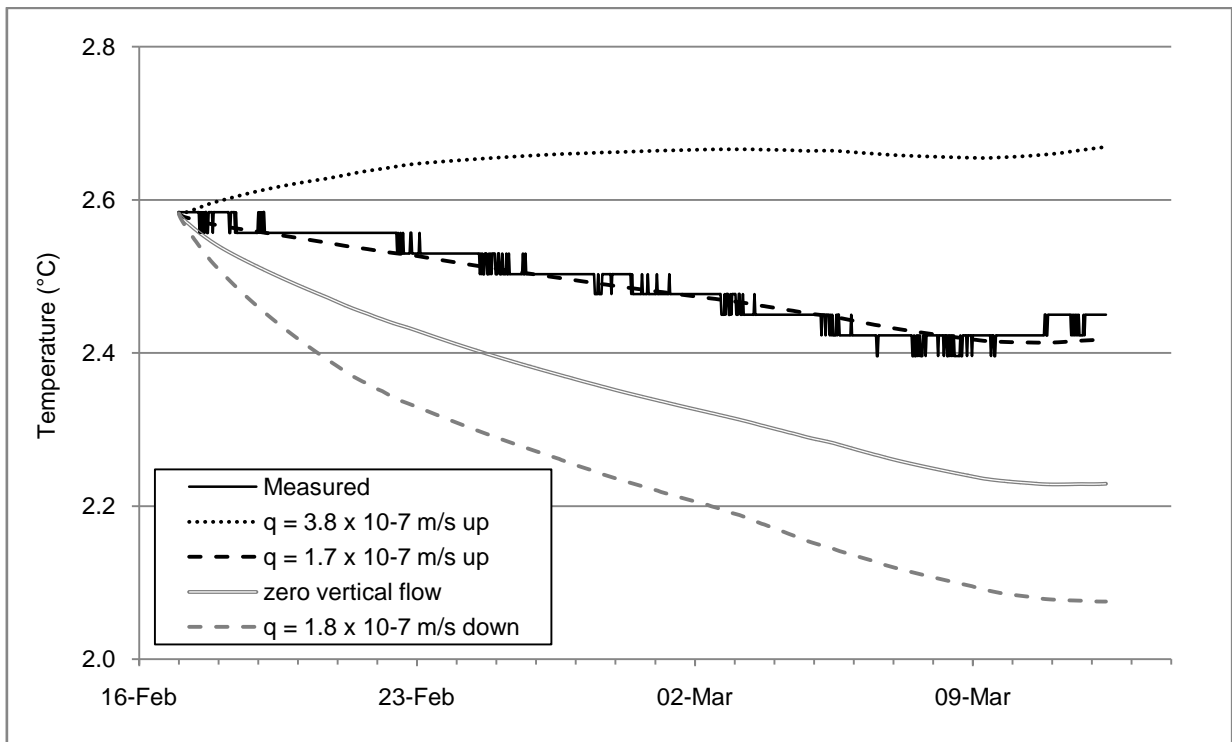

Figure 4.39: Simulated and measured winter temperatures 50 cm below the streambed under various groundwater flow conditions at site 8.

Time series of measured summer stream and streambed temperatures exemplary of downward streambed flux conditions (site 1) and upward groundwater flux conditions (site 4) are presented in Figure 4.40 and Figure 4.41, respectively. They illustrate how temperatures in the streambed are almost identical to those instream under downward flow. Under upward flow, temperatures at depth are cooler and show less variability with depth.
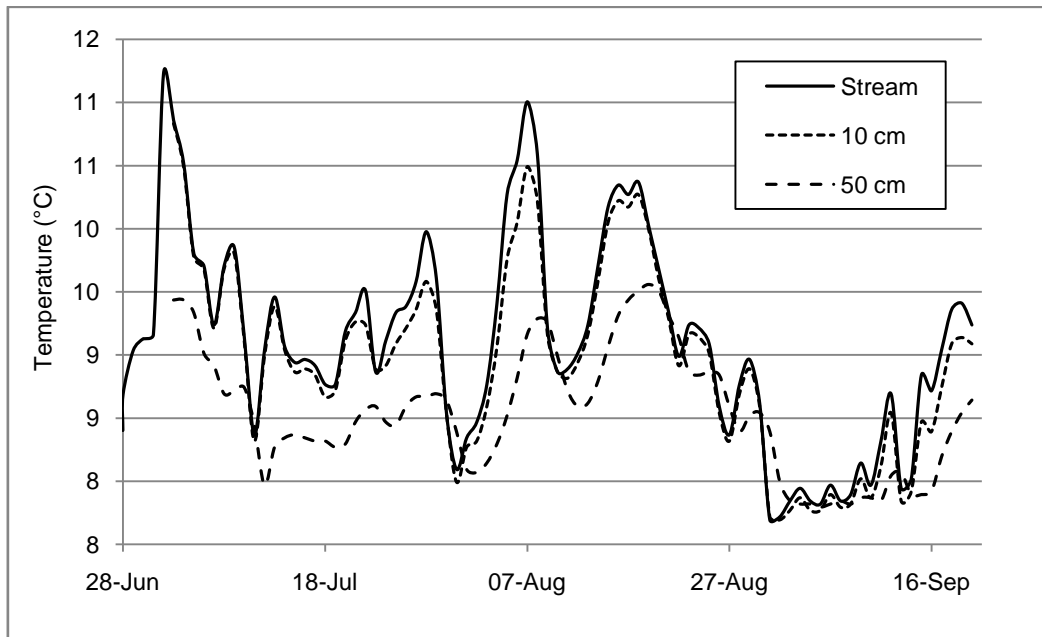
Figure 4.40: Daily maximum temperatures recorded in the stream, at 10 cm and at 50 cm depth in the streambed at site 1 over the 2008 summer.
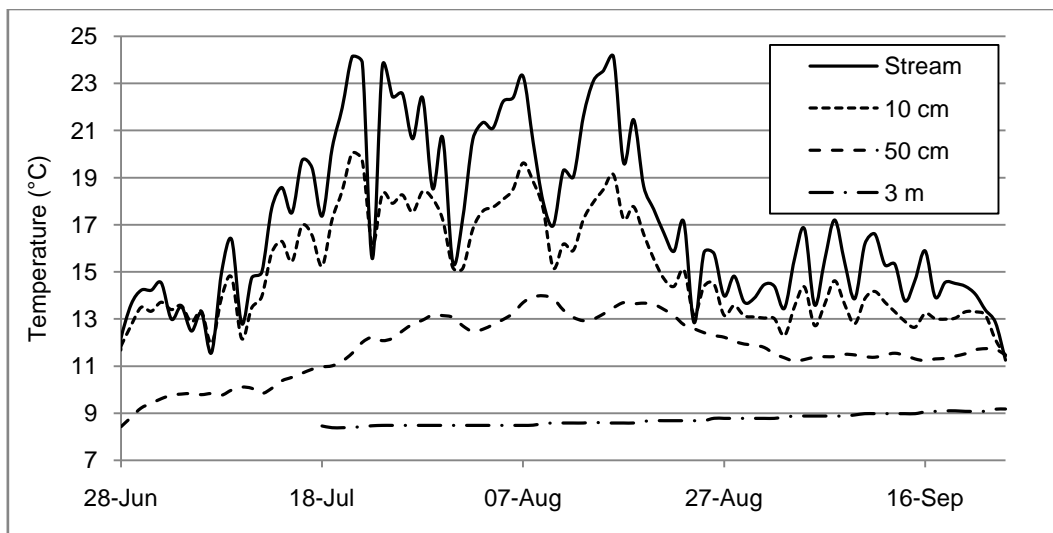
Figure 4.41: Daily maximum temperatures recorded in the stream, at 10 cm and at 50 cm depth in the streambed at site 4 over the 2008 summer.

4.4   Stream Energy Balance Calculations

Summer stream temperatures in Fortune Creek greatly increased in a downstream direction. The largest temperature increase was observed between sites 2 and 3, where the maximum measured temperature increased by 12.7°C over a 1.1 km distance (16.8°C to 29.5°C, Section 4.1.2). The stream changes from deciduous shade (site 2) to completely unshaded (site 3) along this section.

Several water releases from the headwater reservoirs occurred over the summer of 2008 (Section 4.1.1). These were followed by noticeable increases in stream water levels and reductions in stream temperatures at sites 2, 3, and 4 (Figure 4.42, Figure 4.43, and Figure 4.44). The sites are between 9.7 km and 11.7 km downstream of the reservoirs but the increases in water level and corresponding decreases in temperatures can be clearly observed. Temperatures decreased most strongly in the unshaded stream sections (site 3 and 4) where temperatures initially were higher than in the shaded section (site 2). However, rises in water levels coincided with major precipitation events and it is therefore unknown whether rises in water levels were the result of reservoir releases or rainfall.

Over the summer 2008, temperatures at site 2 remained well below 22°C (Figure 4.42), which was taken as a general distribution limit for Chinook, coho and rainbow trout in Fortune Creek based on results from the statistical analysis and literature review (Section 2.4.5.3). Daily maximum temperatures decreased about 4°C after the early August release. At site 3, temperatures frequently exceeded the 22°C threshold between mid-July and mid-August (Figure 4.43). Daily maximum temperatures decreased approximately 15°C after the early August water release. At site 4, temperatures were near or slightly above the 22°C threshold between mid-July and mid-August (Figure 4.44). Daily maximum temperatures decreased about 6°C after the early August release. However, the water releases in late July and August were followed by declines in air temperatures, making it difficult to tell whether water temperatures cooled as a result of the water releases or cooler air temperatures.
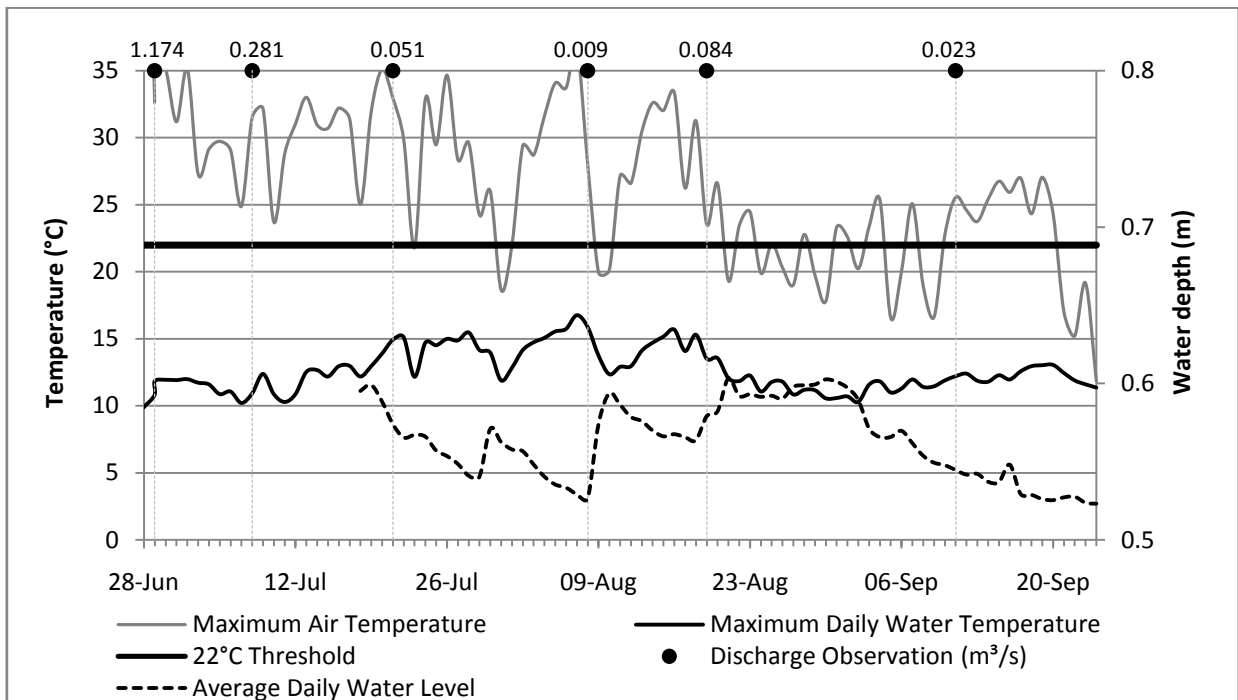
Figure 4.42: Water level, stream temperature and discharge at site 2 over the summer 2008 in Fortune Creek.
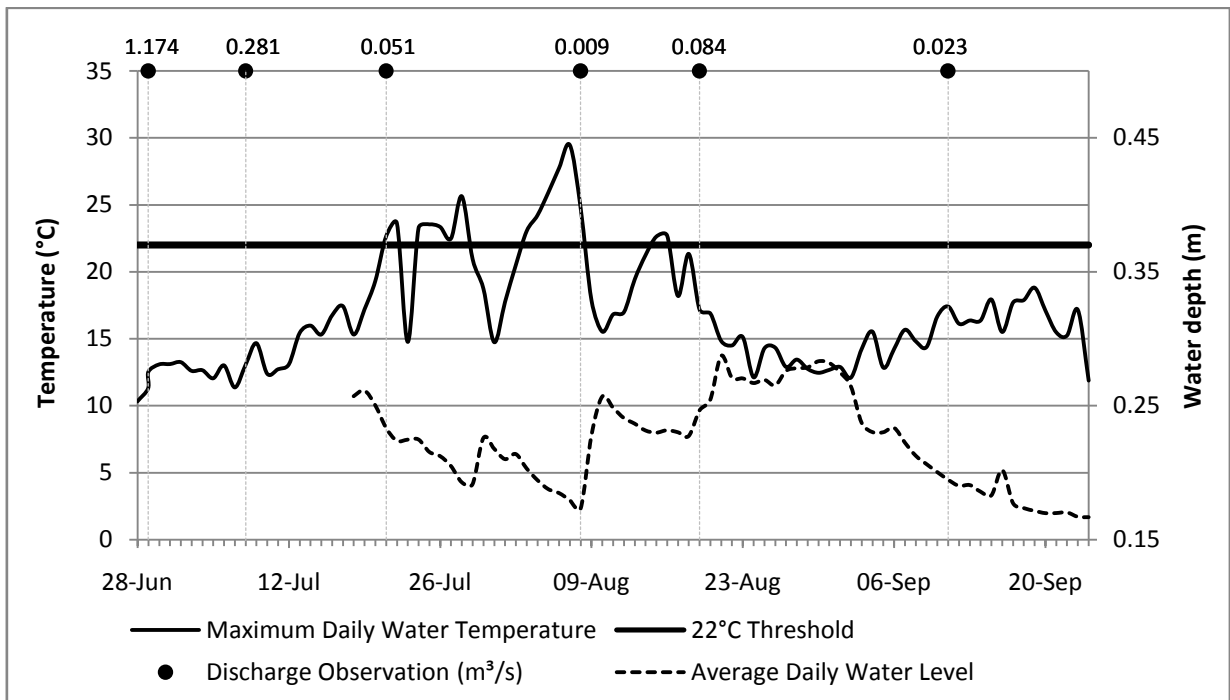
Figure 4.43: Water level, stream temperature and discharge at site 3 over the summer 2008 in Fortune Creek.
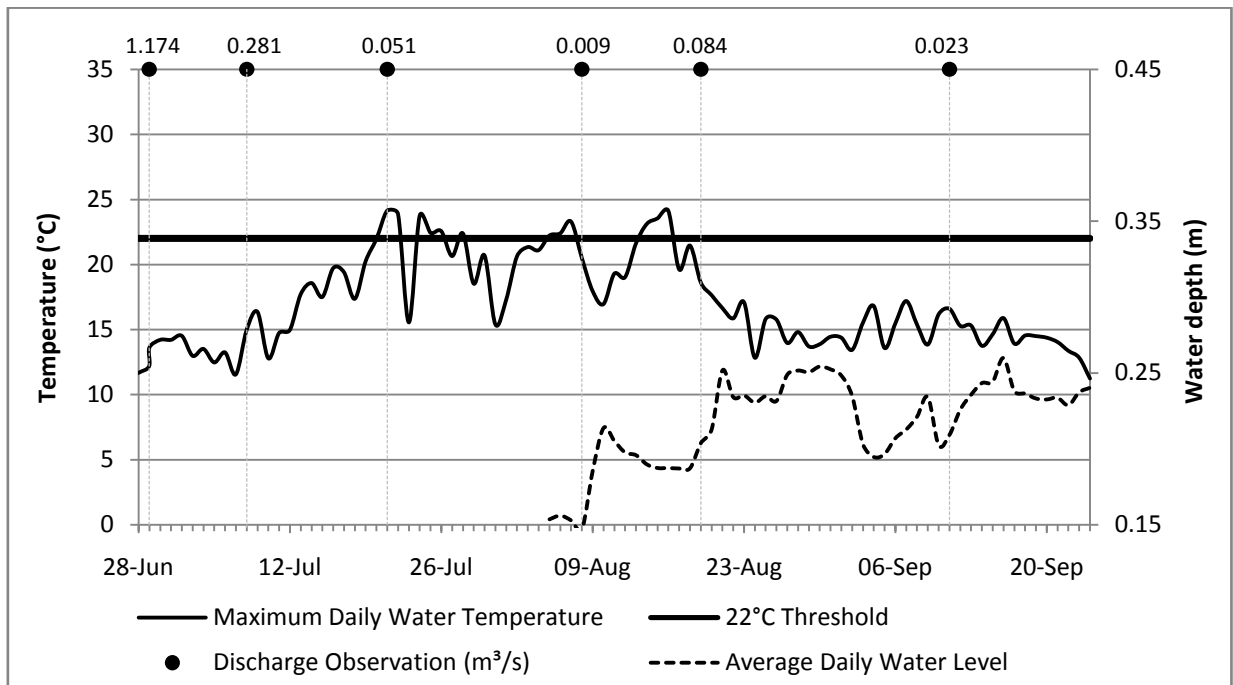
Figure 4.44: Water level, stream temperature and discharge at site 4 over the summer 2008 in Fortune Creek.

Figure 4.45 shows the stream-ground interface energy fluxes for the summer modeling period at site 4, which was one of the sites with the strongest upward groundwater flux. It shows conductive energy transfer from the stream to the streambed (heat loss) during the day and from the streambed to the stream (heat gain) at night, as well as advective cooling of the stream during day and night. Conductive daytime energy losses to the streambed ranged from 8 W/m^2 to a peak of 105 W/m^2 and were on average 47 W/m^2. Advective energy losses ranged from 9 W/m^2 to 24 W/m^2 and were on average 15 W/m^2.
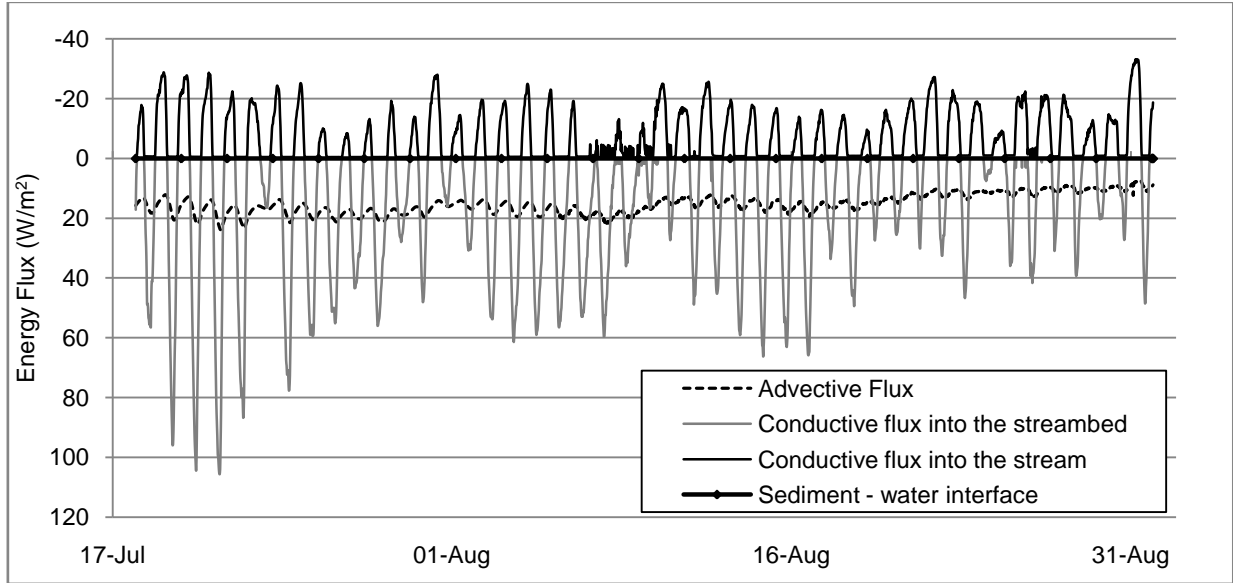
Figure 4.45: Simulated conductive and advective energy fluxes between the streambed and the creek at site 4 in Fortune Creek.

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The estimated cooling effect resulting from groundwater inflows for a hypothetical 1 km long and 1 m wide stream reach under various measured discharge conditions is listed in Table 4.13. Estimated temperature decreases are presented for the maximum groundwater flux (24 W/m^2) and the average groundwater flux recorded (15 W/m^2) over the study period. Estimated cooling effects under maximum groundwater flux ranged from 0.01°C to 0.30°C over the 1 km reach. Under average groundwater flux conditions, estimated cooling effects reached from 0.01°C to 0.19°C over the 1 km reach.

Heating from solar radiation is substantially larger than advective or conductive cooling in unshaded reaches (Table 4.13). In Fortune Creek, the calculated solar heating rate reached up to 7.31°C/km under the lowest discharge conditions and shallowest water depths. In shaded reaches, the heating rate reached up to 0.63°C/km which is approximately twice the cooling effect estimated from groundwater influx. Ultimately, net solar radiation and stream heating under shaded conditions depend on the height and density of streamside vegetation.

The heating effect of solar radiation on stream temperatures in Fortune Creek increases exponentially with decreasing discharge volumes (Figure 4.46). The change in heating rate is relatively slow at discharges above 0.1 m^3/s, but increases rapidly below that. Measured heating rates over the stream reach between site 2 and 3 were slightly higher than estimated heating rates (Figure 4.46, Table 4.13), but the relationship between discharge volume and heating rates is very similar to our simulated data. Since discharge in the reach was well below 0.1 m^3/s from mid-July to mid-August, heating rates per kilometer are upwards of 2°C for most of the summer.

Figure 4.46 illustrates the small cooling effect of groundwater inflows when compared to solar heating rates in unshaded stream reaches. However, groundwater cooling plays a larger relative role in shaded stream reaches.

Table 4.13: Temperature change from advective cooling and solar heating over a hypothetical 1 km long, 1 m wide stream reach, and observed heating rates between site 2 and 3 under various discharge volumes measured in Fortune Creek.

Date		09-Jun	25-Jun	08-Jul	21-Jul	08-Aug	19-Aug	11-Sep
Travel time (min)		49	22	37	57	44	40	57
Flow Velocity (m/s) [a]		0.37	0.78	0.45	0.29	0.38	0.43	0.29
Discharge (m³/s) [a]		3.00	1.17	0.28	0.051	0.009	0.084	0.023
Calculated temperature change per hour (°C)	maximum advective [b]	-0.02	-0.10	-0.11	-0.14	-0.41	-0.13	-0.24
	average advective [c]	-0.01	-0.06	-0.07	-0.09	-0.26	-0.08	-0.15
	net solar radiation (shaded)	0.04	0.21	0.23	0.29	0.86	0.27	0.51
	net solar radiation (unshaded)	0.42	2.42	2.63	3.33	9.98	3.12	5.87
Calculated temperature change per km (°C)	maximum advective [b]	-0.01	-0.04	-0.07	-0.13	-0.30	-0.08	-0.23
	average advective [c]	-0.01	-0.02	-0.04	-0.08	-0.19	-0.05	-0.15
	net solar radiation (shaded)	0.03	0.07	0.14	0.27	0.63	0.18	0.48
	net solar radiation (unshaded)	0.31	**0.87**	**1.61**	**3.19**	**7.31**	**2.04**	**5.62**
Measured temperature change over unshaded 1 km reach between site 2 and 3			**0.52**	**1.64**	**5.60**	**9.40**	**2.16**	**4.31**

a   measured near site 2
b   maximum measured advective heat loss (-24 W/m²)
c   average measured advective heat loss (-15 W/m²)
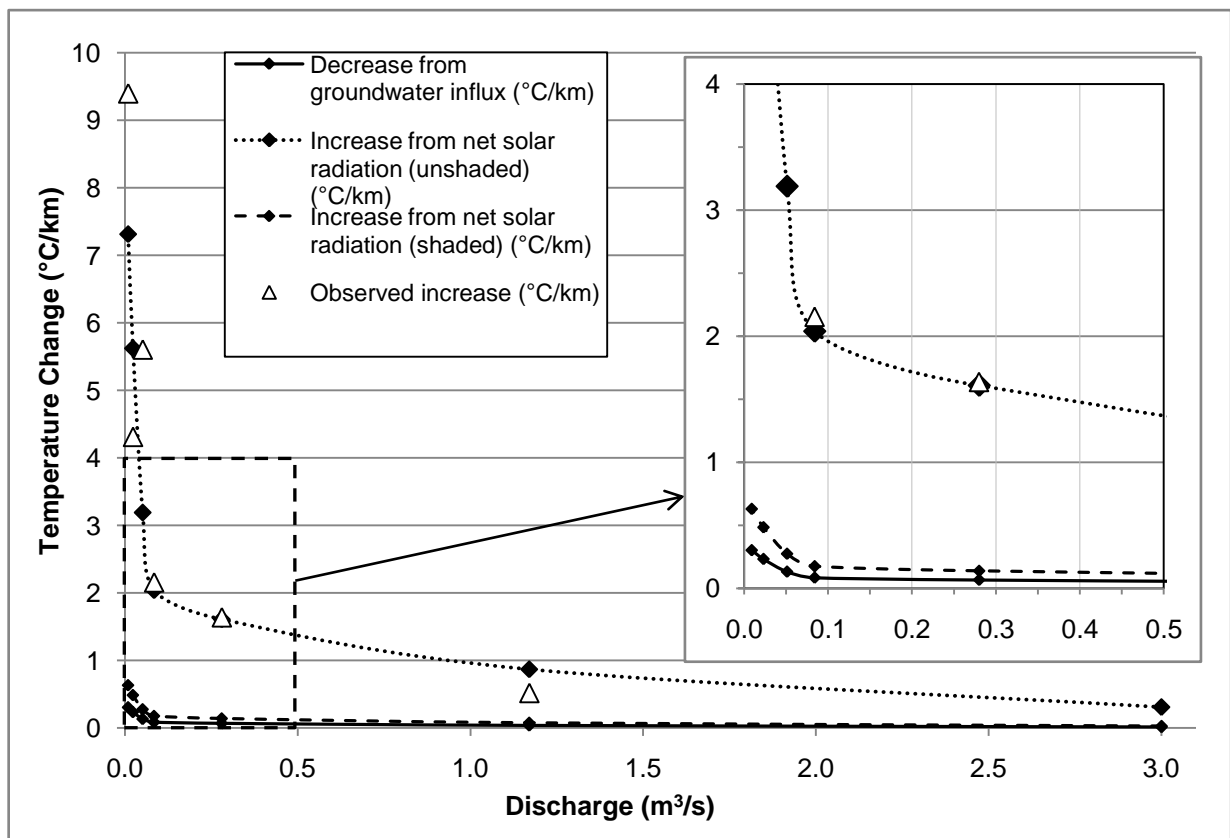
Figure 4.46: Estimated temperature change over a 1 km long stream reach due to solar radiation and groundwater inflows, and observed temperature increase between site 2 and 3 under various discharge conditions in Fortune Creek.

Although groundwater influx only has a small cooling effect on stream temperatures in unshaded reaches, groundwater contributes a substantial water volume to the total discharge of Fortune Creek. In 2008, discharge volumes at the mouth of Fortune Creek near the Shuswap River ranged from a low of 0.017 m^3/s in early August to a high of 3.879 m^3/s in early June (Figure 4.47). The average groundwater flux to the stream in the valley bottom reaches (site 3 to 8) in August was 3.3 x 10^{-7} m^3/s per square meter of streambed. If mean stream width is estimated at 4 m, the total streamflow contribution from groundwater along the 12 km section of Fortune Creek in the valley bottom was 0.015 m^3/s. This was approximately 88% of the total streamflow during the lowest flow period in August 2008.
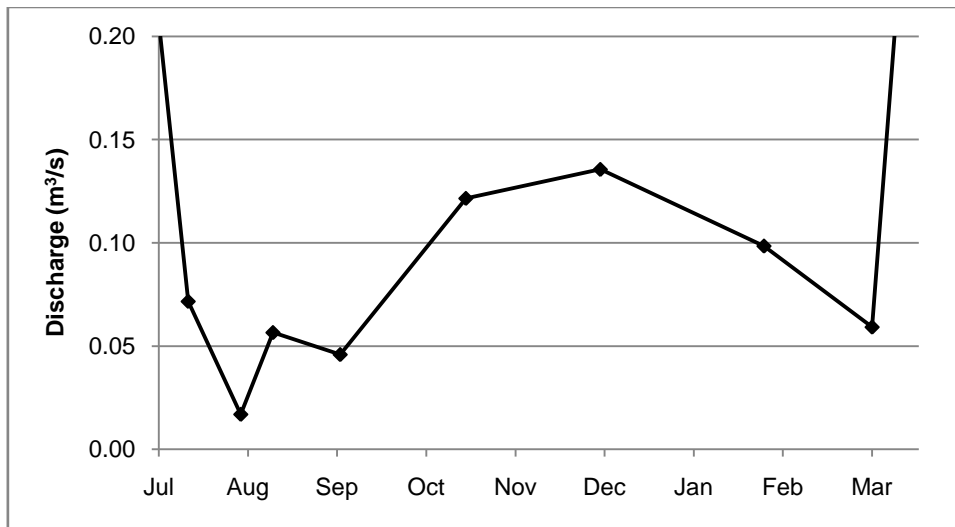
Figure 4.47: Discharge from July 2008 to March 2009 at the mouth of Fortune Creek.

5.0 DISCUSSION

Over the summer of 2008 Fortune Creek followed a general temperature and discharge pattern typical of small interior streams. Mean monthly summer air temperatures measured at the Vernon North climate station in the North Okanagan (Figure 3.1) were relatively similar to long-term average conditions in 2008 and about 2°C above average in 2009 (Figure 5.1). Data for the study period was unavailable from the Armstrong North climate station. Precipitation was below average from March to September 2008 and for most of the summer of 2009, except for April and August which were above normal (Figure 5.2). The flow pattern observed in Fortune Creek was consistent with long-term averages but discharges were well below naturalized flows (Section 4.1.1). Climatic conditions encountered over the study period were considered typical for temperatures and flows in Fortune Creek.
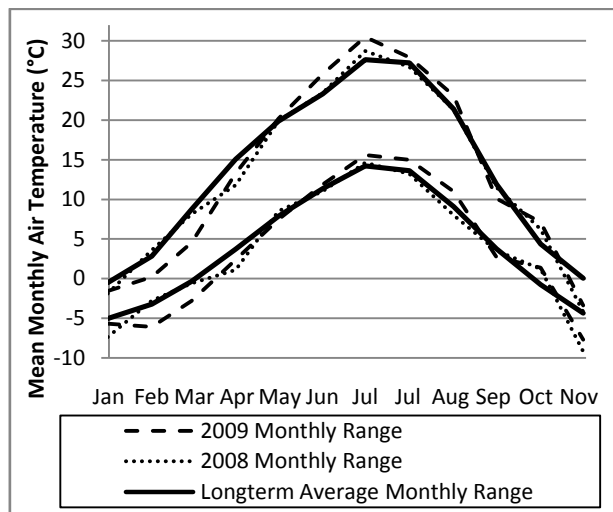
Figure 5.1: Mean monthly air temperature at the Vernon North climate station for the 2008/2009 study period and average conditions (1990 – 2006) (Environment Canada, 2009b).
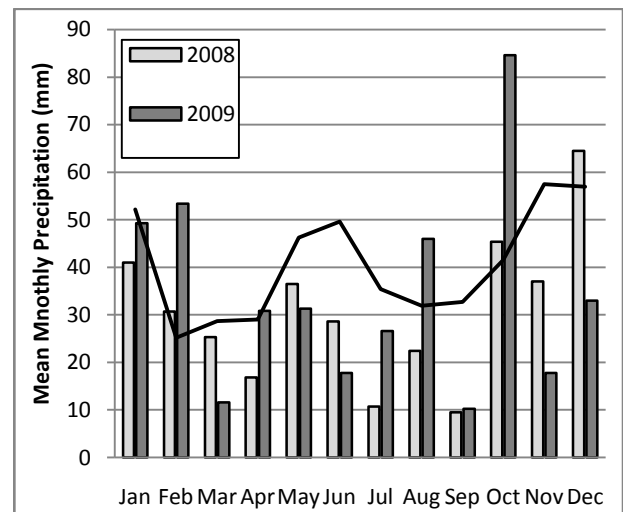
Figure 5.2: Mean monthly precipitation at the Vernon North climate station for the 2008/2009 study period and average conditions (1990 – 2006) (Environment Canada, 2009b).

Water temperatures throughout the creek were cool during the peak flows in early summer but increased dramatically during the critically low flows in July and August. During that time, the valley-bottom reaches (sites 3 to 4) where characterized by high water temperatures that exceeded salmonid temperature thresholds for extended periods of time, water stagnation and eutrophic conditions with sometimes critically low DO.

The Chinook salmon that persisted in the upper reaches throughout the year were confirmed through DNA analysis to be mainly stream-type Chinook. They appear to migrate into Fortune Creek in April and remain throughout the fall and winter, utilizing the creek year round. While the use of non-natal

streams for rearing is generally associated with ocean-type Chinook, some studies have documented this behavior for stream-type Chinook (e.g., Scrivener et al., 1994, Daum & Flannery, 2009).

The large number of salmonids observed in the backflooded valley-bottom sections during the 2008 freshet likely utilized the inundated floodplains, as floodplain rearing has been demonstrated to be advantageous for juvenile Chinook for growth and survival (Sommer et al., 2001). It is suspected that they were ocean-type Chinook that only remain in freshwater for short periods of time.

5.1   Statistical Analysis of Salmonid Distribution

Poisson and logistic regression were successfully used to identify water quality parameters related to salmonid distribution in Fortune Creek. Weekly water temperature variables were confirmed to be important predictors of salmonid distribution for all three salmonid species and generally yielded the most consistent results among species and regression types.

5.1.1 Water Temperature

Relationships between Chinook and mean and minimum temperatures were established (Models CHP6, CHP8, CHL2, and CHL3) while coho and rainbow trout distribution was linked to absolute (Models COP8, COL2, and RP1) and average maximum temperatures (Models COP10, COL1, and RP2).

For Chinook, the temperature limits for zero abundance calculated from the Poisson models and presence/absence calculated from the logistic models generally showed good agreement. The mean and minimum temperatures at zero abundance/absence were 14.4°C to 15.1°C (mean) and 12.0°C to 12.5°C (minimum). Unfortunately, upper thresholds for minimum summer temperatures are rarely reported in the literature. However, optimum rearing temperatures cited between 12.2°C and 14.8°C (Hicks 2000) are in agreement with our results, indicating that for Chinook, daily minimum temperatures should not exceed 12°C to 12.5°C.

Chinook salmon were not observed at sites where weekly mean temperatures exceeded 18.3°C. This is several degrees lower than comparison values from the John Day River in Oregon, which ranged from 21°C to 22°C (Burck et al. 1980). However, our model-predicted zero abundance temperature reached up to 22°C (Model CHP6) considering variability from the random effects, which is identical to that observed in the John Day River. While there is some variability in the results from incorporating the random effects, our temperature limits for Chinook seem in accordance with literature values.

Absolute and average maximum temperature thresholds of zero abundance for coho (21.0°C and 19.2°C) were in accordance with Hicks (2000), who recommends maintaining daily maximum water temperatures below 21°C to prevent mortality of juvenile and adult coho. Upper limits for coho presence from the logistic models were higher at 24.2°C (absolute maximum) and 20.3°C (average maximum). However, given that coho upper thermal limits reported in the literature range from 23.8°C to 29.2°C, our values are still within the commonly accepted range.

Mean temperature was a significant predictor of coho abundance in a multivariate regression model with DO (Model COP5). The same model was significant for Chinook abundance (Model CHP2), which allows for comparison between the species. At a constant DO level (8.8 mg/L), the weekly mean temperatures at which coho and Chinook abundance approached zero (14.9°C and 14.4°C, respectively) were within 0.5°C of each other, suggesting that both species' distribution is similarly related to mean temperatures in Fortune Creek.

Absolute and average maximum temperature thresholds of zero abundance for rainbow trout were several degrees lower than those for coho, at 17.5°C and 15.5°C, respectively. These values are in close agreement with Hicks (2000) who recommends that the 7-day average of daily maximum temperatures should not exceed 15°C to 18°C in a naturally fluctuating temperature regime. The average prediction curve from both rainbow trout models is much lower than the observed numbers (Figure 4.19), indicating that the model fit may not be ideal. No logistic models for predicting rainbow presence/absence were retained.

The results suggest that rainbow trout were present at lower temperatures than coho. However, the analysis excluded new rainbow trout fry that emerged during the study period at site 3 where temperatures reach 29.5°C. The lack of growth observed in these fry indicates that, while juvenile rainbow may be able to withstand high temperatures for periods of time, growth and subsequent survival are severely affected. It is unlikely that healthy populations could persist under such conditions, as the zero growth limit for rainbow trout is 23°C (Section 2.4.5.2.3).

Temperature exceedance variables were significantly related to Chinook and coho salmon. It is unclear whether rainbow abundance or presence/absence was significantly related to any of the temperature exceedance variables, as numerical errors were encountered and modeling could not be completed for these predictors.

For Chinook models, significant exceedance variables included total hours over 15°C (Poisson) and continuous hours over 17°C and 20°C (logistic). Chinook were much less tolerant of water temperatures

over 20°C than 17°C as indicated by a steeper regression slope and substantially smaller variability in the 20°C model (Figure 4.22).

Coho abundance was negatively related to continuous hours over 18°C. The zero abundance threshold (10 hrs) was similar to the absence threshold for Chinook at continuous exposures over 17°C (13.7 hrs). This suggests that coho and Chinook are sensitive to relatively short periods of water temperatures over 18°C.

Through multivariate models, coho abundance was also related to continuous hours over 19°C and total hours over 21°C and 22°C. The regression slope estimates were steeper for the continuous exceedance model, although the temperature thresholds were lower. This indicates that coho were more sensitive to short continuous exposures at lower temperatures (19°C) than repeated exposures over a week long period at higher temperatures (21°C and 22°C). This is presumably because for total exposure, water temperatures decreased to a tolerable range at night providing respite from temperature stress

The results suggest that juvenile salmonids in Fortune Creek will largely avoid stream reaches where maximum daily water temperatures exceed 22°C (Section 2.4.5.1, Figure 2.5). This temperature was exceeded at sites 3 to 8 for extended periods of time in July and August of 2008 (Figure 5.3).
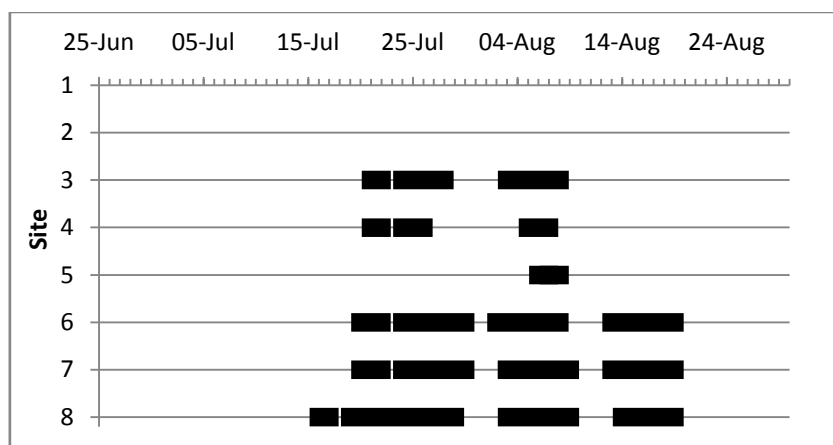
Figure 5.3: Days during 2008 on which maximum water temperatures in Fortune Creek exceeded 22°C, above which juvenile salmonids are expected to be largely absent.

5.1.2 Chemical Water Quality Parameters

While most of the chemical water quality parameters (pH, DO, conductivity) were significant predictors of Chinook and coho abundance in the Poisson regression models, they were not significant for predicting coho and Chinook presence/absence in the logistic models with the exception of DO for

Chinook (Model CHL6). No relationships could be established between rainbow trout and any of the chemical water quality parameters.

Surprisingly, both coho and Chinook counts and the probability of Chinook presence decreased with increasing minimum DO levels. This negative relationship is unlikely to be correct as it clearly conflicts with the literature. Further, salmonid mortality was directly observed at site 8 when DO levels were 3.14 mg/L in early July. The relationship is an artifact resulting from the way DO was measured.

DO was measured twice during each fish counting event, once when minnow traps were set and once the next day when they were retrieved. Thus measurements were always conducted during daylight hours when photosynthesis is active. Sampling time was usually mid-afternoon. Values were often high (up to 196% saturation, Figure 4.7) in the lower reaches of Fortune Creek where the unshaded channel was observed to be inundated with aquatic plants and algae (Section 3.1.1). These plants produce large amounts of oxygen during the day when DO readings were taken. It is suspected that DO values in those reaches declined sharply over night, but the true minimum DO was not recorded. Low overnight DO levels due to respiration and decomposition of accumulated organic carbon are characteristic of high nutrient conditions (Section 2.4.2).

In the upper reaches (sites 1 to 2) where most salmonids were captured, DO was less variable (Figure 4.7), reaching a maximum of 100% saturation. At sites 3 and 4, DO levels did exceed 100%, which was consistent with the observed higher density of instream vegetation. At these four sites, daytime DO values were often lower than in the valley bottom reaches, which explains the resulting negative relationship between DO and salmonid abundance.

Initially, the installation of DO sensors at all study sites in Fortune Creek was considered to continuously record DO values. However, this was not feasible due to concerns over cost, biofouling and drift of the sensors. Point measurements of DO are not representative of the conditions in the creek over time and a continuous record of DO values would have provided a better understanding of the range of values encountered. The daytime DO measurements are able to indicate a statistical relationship to fish abundance, but likely the true relationship is between nighttime low DO and fish abundance. Nighttime low DO and saturations greater than 100% in daylight would be correlated.

Variability in pH values generally increased in a downstream direction and pH was higher in the nutrient-loaded lower reaches of Fortune Creek (Section 4.1.4). It is likely that daytime photosynthesis of the abundant instream vegetation in those reaches contributed to this rise in pH similar to the high DO

values recorded there. Maximum pH values did reach levels reported to adversely affect salmonids (pH > 9.0) but the minimum values were within acceptable range.

pH values were significantly and negatively related to Chinook abundance only in association with minimum temperature (Model CHP3) or conductivity (Model CHP7). They were related to coho only in an interaction with minimum temperature (Model COP4). As described further below, the relationship between salmonids and pH would have been better explained by a curvilinear relationship (Section 5.1.4).

Conductivity was a significant negative predictor of Chinook salmon only in association with pH (Model CHP7) and a positive predictor of coho only in association with continuous hours > 19°C (Model COP3). Maximum conductivity values increased by approximately 60% between site 3 and 4, coinciding with the area where the strongest upward groundwater fluxes were simulated. While elevated electrical conductivities have been used to identify groundwater discharge areas in lakes (Harvey et al. 1997), it is questionable if they were indicative of groundwater input into Fortune Creek. Water chemistry in this section is likely influenced by numerous field drains with high conductivities discharging into the creek and the conductivity levels are indicative of the general degradation of water quality in a downstream direction.

The range of conductivity values encountered over the study period (226 µS/cm to 372 µS/cm) was within the range of typical values for interior streams and well below the levels at which research has demonstrated impairment of fish (3,000 µS/cm, Section 2.4.4) It is suspected that the relationships found between salmonid numbers and conductivity in this research are mainly coincidental. The negative relationship with Chinook likely exists because Chinook numbers were highest where conductivity levels were low (site 2). Coho numbers were highest at site 4 where conductivities were high, hence the positive relationship.

Cover was only significant in one logistic regression model for Chinook in association with weekly minimum temperature (Model CHL1), although it was expected to be an important predictor of salmonid abundance and presence/absence. This may be explained by the way cover was defined for this study. Any object providing shelter for salmonids from predation or swift currents was considered cover. Aquatic plants, such as water lilies, were counted equal to undercut banks and log jams in terms of their ability to provide cover. However, some of these maybe more desirable cover than others.

5.1.3 Model Performance

In general, modeled Chinook counts were relatively similar to observed counts. The temperature variables on their own were able to reduce residual deviance by an additional 33% to 48% over a null model containing only the random effects and an intercept. Adding in chemical water quality variables increased deviance reduction to 57%. This indicates that the temperature variables are quite important in explaining variability in Chinook abundance in Fortune Creek.

Modeled and observed counts for coho were generally in relatively close agreement, but the univariate models containing temperature variables only reduced residual deviance between 9% and 22% over a null model containing the random effects. Adding in the chemical water quality parameters decreased residual deviance up to 44% over the null model. This indicates that water temperatures alone were not as powerful in explaining variation in coho numbers in Fortune Creek as they were for Chinook.

The residual deviance reduction for the two rainbow trout temperature models was relatively low at 10% and 11% over the null model. This indicates that most of the variability in rainbow counts was explained by the random effects (site and week) and not the temperature variables. Additional unknown effects not measured may include competition with coho and Chinook.

Variability was much greater in the coho and rainbow models than the Chinook models. The models were generally poorly constrained and the range of predicted temperature thresholds when variability between sites and weeks was considered was high (up to 19°C). Temperatures up to 48°C were predicted for coho presence thresholds. Given that these temperatures are clearly above the upper thermal limits for coho from literature, the results raise questions regarding the reliability of predictions based on these models. In addition, such temperatures would not be encountered in natural salmon streams.

5.1.4 Limitations of Statistical Analysis

Several issues were encountered in the statistical analysis of the data. Multicollinearity between the predictor variables was widespread and may have affected regression estimates and the selection of significant parameters even though efforts were made to avoid the inclusion of correlated variables in the same model. The chemical water quality parameters were mainly significant in models together with temperature or each other and only one univariate model contained a water quality parameter (DO, Chinook Model CHL6). While this is indicative of multicollinearity, the correlation between fixed effects

in the multivariate models was always low (<0.25) and there were no large changes in regression estimates when variables were added or removed.

Graham (2003) describes how even low levels of multicollinearity can seriously impair multiple regression analysis. He offers several remedies such as sequential regression, residual regression and principal component analysis. Buisson et al. (2008) used hierarchical partitioning in a similar study which assessed the relative effect of temperature versus physical parameters in fish distribution. However, this approach does not allow for temporal and spatial autocorrelation to be addressed and was therefore unsuitable for our data.

While linear regression was used for the analysis, several predictor variables would have been better described by a non-linear relationship. For instance, salmonids occurred over a range of pH values and most observations were clustered in the range of pH 7.5 to 8.5. The observed distribution would be better described by a curvilinear relationship. However, since most variables were adequately described by linear relationships, linear modeling was chosen.

An alternative would be to use Generalized Additive Mixed Models (GAMMs), which allow for non-linear relationships between the dependent and explanatory variables by fitting a smoothing curve through the data (Zuur et al., 2009). In R, the function *amer()* in the library amer (Scheipl, 2009) analyzes GAMMs. The function is based on *glmer()* which was used in this study, and allows for the same random effects structure to be implemented.

Initial data exploration revealed a large number of zero fish counts at sites 6 through 8. To address the large number of zeros, use of the negative binomial distribution and the zero-inflated Poisson distribution instead of the Poisson distribution were considered for the regression analysis. However, there are currently no packages available in R that allow for these distributions and crossed random effects in the same model. It was determined that the excess zeros were adequately accommodated by the random effects and the Poisson regression was deemed suitable.

5.1.5 Additional Data Collection

Various other factors that were not measured during this study could have influenced the distribution of salmonids in Fortune Creek. Competition between the salmonid species could have influenced their distribution particularly in the upper reaches of Fortune Creek (sites 1 to 4). While rainbow trout were often the largest salmonids captured (up to 2+ years), Chinook were frequently larger than coho and were also more abundant. Coho may have been most numerous at site 4 not out of preference but

because they were outcompeted by Chinook and rainbow in adjacent preferred habitats (e.g., site 2). Since water temperatures were higher at site 4, this would have influenced our analysis suggesting that coho were more tolerant of higher temperatures. It is likely that competition for habitat and food also occurred between salmonids and non-salmonids in the downstream reaches where large numbers of pikeminnow, peamouth chub and redside shiners were captured.

Food availability may influence the distribution of salmonids in Fortune Creek. Aquatic invertebrates were generally very abundant throughout Fortune creek and large numbers of freshwater amphipods and other invertebrates frequently entered the minnow traps during fish counting events. Food availability was not measured as part of this study but is not suspected to play a major role in limiting salmonid distribution in Fortune Creek.

Northern pikeminnow, known to be key predators of juvenile salmonids (Friesen and Ward, 1999), were captured frequently at sites 5 to 8. It is possible that predation by pikeminnow contributed to the displacement of salmonids from the lower reaches. The pikeminnow observed in this study were too small to prey on juvenile salmonids because they were captured using minnow traps; however, it is likely that larger pikeminnow are present in the creek. The extent of predatory interactions could be assessed by capturing large pikeminnow and examining their stomach contents for evidence of ingestion of juvenile salmonids.

Several other variables would have been valuable in evaluating salmonid distribution. Discharge and flow velocities were measured at two week intervals at five sites in Fortune Creek, but this information would have been valuable at all eight study sites on a weekly basis for inclusion in the statistical models. This could have been achieved by the installation of water level loggers and the development of rating curves at each site. This would have been particularly important because the absence of coho from site 1 was thought to be related to the steep channel gradient and high flow velocities. Monetary and time restrictions prevented the measurement of discharge during the fish count events and the installation of water level gauges at all sites.

Monitoring DO continuously would have been valuable in determining true DO concentrations encountered at each site, as it is known that low oxygen events occurred and they most likely influenced the distribution of salmonids in the creek.

Additional variables characterizing the physical habitat quality might have provided insight in the evident appeal of site 2. While cover was included in the analysis as a physical fish habitat indicator, it was significant only in one model. Other variables might have included substrate size, streambed

morphology. It might have been advantageous to include an additional variable that is more representative of the quality of physical salmonid habitat available at each site. This could be the habitat unit type (e.g., pool, riffle, etc) or a measure of habitat heterogeneity (Johnston & Slaney, 1996).

Substantial numbers of juvenile salmonids were observed at the downstream sites (sites 6 to 8) in June 2008 prior to the onset of trapping. It was therefore known that these sites were utilized by juvenile salmonids in early summer, although few were captured during the July/August study period. It would have been beneficial to commence trapping earlier where possible to document some of the observations at the lower sites. Without them, the downstream sites were dominated by zero counts, which was problematic for the statistical analysis but could be accommodated by the GLMM modeling.

5.1.6 Assessment of Statistical Methodology

When modeling salmonid distribution, Poisson and logistic regression each have their advantages and disadvantages. For instance, chemical water quality parameters were not significant predictors of salmonid presence/absence (logistic models) but were important in the abundance modeling (Poisson models). This may be interpreted by considering the difference in the modeled responses in Poisson and logistic regression. The response variable in Poisson regression is a count and the response in logistic regression is the probability of occurrence of an event. While the chemical water quality parameters over the range measured in this study may not have a strong enough influence on salmonid distribution to fully exclude salmon from a site, they may contribute to habitat quality significantly enough to influence the number of salmon present. Therefore, Poisson modeling of salmonid abundance provides more information on the quality of habitat in Fortune Creek.

In contrast, when modeling salmonid distribution we expect that within the optimum temperature range, factors other than temperature determine salmonid abundance. It is when temperatures move outside of that optimum range that we would expect to see a decrease in abundance. This relationship is better explained by logistic regression, which specifies that salmonids are expected to be present when temperature conditions are suitable and that the probability of presence declines when conditions deteriorate. Thus, both regression types provide valuable insight into habitat use and limiting factors of salmonid distribution.

5.2   Groundwater - Surface Water Interaction

Streambed temperature profiles were successfully used to estimate seepage velocities in the streambed of Fortune Creek. Surface water groundwater interactions changed from downward flow at the top end of the alluvial fan (site 1) to upward flux through the downstream reaches of Fortune Creek (sites 4 to 8). Groundwater simulations in the valley bottom reaches (sites 4 through 8) performed well and indicated that the strongest groundwater discharge to Fortune Creek occurs at sites 4 and 5. Confidence is higher in the model results from the valley bottom reaches where upward fluxes existed than the uppermost two sites where flow direction was less certain.

Water levels in both piezometers were very stable throughout the summer and increased slightly in the fall. This indicates that the upward hydraulic gradient was relatively constant and did not seem to be greatly affected by groundwater pumping over the summer. Groundwater discharge in the lower reaches of Fortune Creek is primarily constrained by low streambed hydraulic conductivities (estimated at 10^{-7} m/s and 10^{-8} m/s), and it does not appear that groundwater pumping at the present rate has a large enough influence on the regional groundwater table to greatly influence flux to the creek.

Flux simulation at sites 1 and 2 was difficult as the flow direction was unknown and downward flow was suspected. Since deep bed temperatures were unavailable for these sites (no piezometer data), the temperatures measured in the piezometers at sites 3 and 4 were used for the bottom boundary condition. Given that modeled temperature profiles indicated downward flow, it was uncertain whether these temperatures were very representative of the true conditions at the sites. This introduced substantial uncertainty into the models, which was reflected in the inability to obtain a good fit between modeled and observed temperatures at site 1 in the winter. The downward flow direction at site 1 at the upper end of the alluvial fan is reasonably certain, because the stream loses water in this reach and a section just downstream of the site dries out periodically, indicating stream water infiltration into the fan.

Flow direction was more questionable at site 2. Simulated temperatures at site 2 strongly indicated downward flow, although artesian wells exist near the site and the stream appears to be gaining in this section. Given that the streambed is dominated by gravel and cobbles, it is possible that substantial longitudinal flows exist in this location, making vertical one-dimensional modeling unsuitable. Silver (2007) demonstrated that longitudinal fluxes of 10^{-6} m/s or greater led to higher than expected bed temperatures, higher amplitudes of diurnal temperature fluctuations, and quicker responses to day-to-day stream temperature changes. These effects mimic those resulting from vertical downward flow into the streambed. It is therefore difficult to say whether downward flow truly existed or whether an

upward flow temperature signal was masked by longitudinal flow. Piezometer installation near the site was attempted to measure hydraulic gradient, but failed repeatedly due to the cobbly substrate.

Piezometer water level monitoring confirmed that an upward hydraulic gradient was present at site 3. However, the best fit to the observed temperature time series was achieved under negligible fluxes (< 10^{-10} m/s). Similar to site 2, it is possible that substantial longitudinal flows affected the logger at 50 cm depth. Distinct daily fluctuations can be observed in the measured streambed temperatures (Figure 4.24). The logger was situated in the thalweg where substrate consisted of fist size cobbles. The piezometer was installed at the edge of the stream to a depth of 3 m. The tip of the piezometer extended into deposits of low hydraulic conductivity, estimated at 1.2 x 10^{-8} m/s from the slug test. This is in the range of a clay or silt (Fetter, 2001). Since much of the valley bottom is occupied by clay soils, it is likely that the layer of cobbles, gravel and sand deposited by the creek between the current dyke locations is underlain by clay soils. These would constrain upward groundwater flux while longitudinal flow through the cobbly layer along the thalweg drives the large daily temperature fluctuations and high temperatures observed at 50 cm. In summary, it is likely that longitudinal flow affected groundwater flux simulations at sites 1 to 3.

Besides longitudinal flow, another source of uncertainty in the simulations is streambed heterogeneity, particularly at the upper three sites. Substrate sizes in the upper reaches varied widely and substantial heterogeneity was expected. However, since substrate sizes in the streambed were unknown, all models assumed homogeneous conditions. It has been demonstrated that considerable uncertainty in subsurface heat distribution is introduced by even small amounts of heterogeneity in hydraulic conductivities (Ferguson, 2007).

In addition, seasonal variations in streamflow velocity leads to the deposit of alternating layers of differing grain size. Constantz et al. (2003) state that clogging layers with low hydraulic conductivities are common in streambeds, leading to a disconnection between stream and aquifer. The layering would lead to anisotropic conditions in the streambed. While the presence of clogging layers in the streambed is suspected, the exact composition of the streambed was unknown and could not be modeled. Therefore, all models assumed isotropic conditions.

The thermal dispersivity for all models was set to 0.01. Varying dispersivity was not expected to have a large influence on simulated temperatures because simulated upward fluxes at most sites were low (<10^{-6} m/s). As discussed in Section 2.2.1.3, simulated streambed temperatures are more sensitive to changes in thermal dispersivity at fluxes higher than those encountered in this study.

At sites 1 and 2, where fluxes were higher (10^{-6} m/s), varying dispersivity values had small but noticeable effects on model fit by changing the magnitude of daily temperature variations, but did not lead to a change in the estimated fluxes. Errors introduced by variations in dispersivity were considered negligible compared to those introduced by the uncertain flow direction and suspected longitudinal flow.

The streambed hydraulic conductivity and groundwater flux estimated from the slug test at site 3 were in relatively close agreement with that estimated from thermal modeling. At site 4, they were more than one order of magnitude smaller. The reason for this difference may be that the piezometer and the temperature probes were not installed in the exact same location at this site. Due to streambed heterogeneity, streambed fluxes can vary by several orders of magnitude between nearby locations (Conant, 2004). As a result, spot measurements may not be representative of flux conditions encountered elsewhere in a reach.

Both piezometers were slow to recover from the slug tests, although that at site 3 took much longer than at site 4. When the piezometers were first installed they were not developed. Clogging of the piezometer slots may have contributed to the slow response time. Developing the piezometers may have helped to flush out any sediment plugging the slots. Nonetheless, it is evident that low conductivity clay and silt deposits underly most of the valley bottom reaches and limit upward groundwater flux velocities.

At the sites where no piezometers were installed, deep temperatures from the piezometers at sites 3 and 4 were used as boundary conditions in the groundwater flow models. This was deemed acceptable as temperatures between the two piezometers varied very little (0.2°C) and it was assumed that they would vary equally little throughout the remainder of the valley bottom. While the flux direction was relatively obvious at the valley bottom sites, estimated flux magnitudes could be incorrect if the bottom temperature at the unmeasured sites was different from that at site 3 and 4. This was a more substantial problem at sites 1 and 2 where simulations indicated that downward fluxes existed and streambed materials were very different from the remaining sites.

While the installation of additional piezometers was not feasible for this study, temperature and pressure head measured in the piezometers are very important for accurate simulations in VS2DH. Where longitudinal flow is limited, measurement of deeper temperatures (e.g., 1 m depth) would be sufficient to model fluxes even where no pressure head information is available. For reasons discussed above, this is not suitable where longitudinal flow exists and the flow direction is unclear.

While the above-mentioned issues would have to be resolved for detailed quantification of groundwater fluxes throughout the creek, the purpose of this study was to assess the potential for groundwater influx to provide thermal relief for salmonids. If the results indicated more substantial groundwater heat flux, it would be worth expending greater efforts on determining exact flux values throughout the creek. However, the accuracy of the results was considered sufficient for addressing research question 3 (Section 1.1).

5.3   Stream Energy Balance

Thermally suitable habitat in which juvenile salmonids can persist throughout the summer currently only exists between site 1 and site 2 (Figure 5.3). The largest stream temperature increase (up to 12.7°C) occurs in the exposed stream section between site 2 and 3. Temperature frequently decreased between site 3 and 4, and temperature increases between site 4 and 5 were relatively small (<1.5°C). Therefore, thermally suitable habitat for juvenile salmonids could be extended from site 2 to site 5 if stream heating on the exposed reach between site 2 and 3 could be controlled. This would represent a gain of almost 4 km of habitat. All sites between site 1 and 5 were occupied by juvenile salmonids at some point during the summer of 2008 and are therefore assumed to provide suitable physical habitat conditions (Figure 4.11).

The estimated heating rates over a 1 km stream reach under unshaded conditions were very close to true heating rates between sites 2 and 3 observed in the field (Table 4.13). The maximum estimated heating rate under the lowest measured discharge in the reach (0.009 m^3/s) was 7.31°C/km. However, the true maximum temperature increase observed over the 1 km reach between site 2 and 3 was slightly higher at 9.4°C/km. Small differences like these were expected as this was a very simplified simulation that used point measurements of average depth and velocity and did not consider many other components of the stream energy budget (e.g., air temperature).

Groundwater influx to Fortune Creek was diffuse and no substantial localized cold-water discharges were identified. Given the lack of shading along most of the creek and the resulting high solar load, the estimated cooling influence of groundwater influx on stream temperatures (0.02°C to 0.41°C per km, Table 4.13) is too small to provide significant thermal relief for salmonids. Therefore, the possibility for groundwater inflows to provide thermal refuge for salmonids is considered low. However, the contribution of groundwater to total streamflow was approximately 88% during the low flow period in August. Thus, while groundwater influx has a small effect through advective cooling, it is essential in providing baseflows to Fortune Creek during the summer months.

The highest daily maximum temperatures were 16.8°C at site 2, which is well below the 22°C threshold above which salmonids are expected to be largely absent (Section 5.1.1). This means that a maximum allowable temperature increase of 5.2°C/km can occur to ensure temperature at site 3 remains below 22°C.

Figure 4.46 indicates that heating rates due to solar radiation in unshaded reaches increased greatly when flows decreased below 0.1 m³/s. Observed heating rates indicate that to ensure stream temperatures at site 3 remain below this threshold, stream discharge at site 2 would have to be at least 0.06 m³/s. Since the measured flow velocities from site 2 used in this conceptual model are likely slightly higher than average conditions in the reach, a conservative threshold would be 0.1 m³/s. This is near the naturalized discharge of Fortune Creek for August (0.15 m³/s).

Maximum estimated solar heating rates under shaded conditions were much lower (up to 0.63°C/km) than in unshaded reaches and can be more than offset by advective cooling and streambed conduction. This is well below the maximum allowable temperature increase between site 2 and 3 discussed above (5.2°C/km). It is therefore likely that shading between sites 2 and 3 could extend thermally suitable habitat for juvenile salmonids from site 1 and 2 to site 4 and 5.

This analysis indicates that the two primary options for extending thermally suitable habitat from site 2 to site 5 are to increase shading between site 2 and 3 through re-vegetation of the banks and to ensure minimum flows between 0.06 m³/s and 0.1 m³/s are maintained throughout the summer. Re-vegetation efforts between site 2 and 3 were undertaken in the 1990s, but vegetation has not yet grown tall enough to provide sufficient shading. Large stretches of the creek remain completely exposed and stream temperatures would greatly benefit from streamside vegetation reducing the solar load.

The main objective in increasing flows is twofold: 1) to increase the volume of water subjected to heating and cooling and 2) to minimize the time that the water is subjected to heating and cooling by increasing the velocity at which the water moves through the channel. Target stream discharge could be achieved by additional water releases from the headwater reservoirs, better timing of releases from the reservoirs, augmentation of flow from an additional source, or a reduction in water withdrawals. As discussed in Chapter 4, several water releases from the reservoirs in the summer of 2008 were followed by noticeable reductions in stream temperatures (Section 4.4), although it is unclear whether temperatures declined because of the releases or because of cooler air temperatures and precipitation.

Water diversions can have significant effects on stream temperatures particularly in small, low-gradient streams (Meier et al., 2003). Fortune Creek is the primary water supply for approximately 4,500 users in

addition to various irrigation water licenses. Licensed water demand on the creek is high and reached up to 92% of estimated naturalized flows in August 2008 and up to 99.3% in August 2009. If summer flows in the creek were not augmented by the headwater reservoirs, it is likely that all flows in Fortune Creek would have been utilized during those periods. While the actual amount of water withdrawn is unknown it is highly likely that the current volumes of diversion and resulting reduction in streamflows leads to a considerable increase in stream temperatures.

As an alternative to reservoir releases, streamflow could be supplemented by pumping groundwater from one of the deep confined aquifers into the creek near site 2. While groundwater pumping may seem counterproductive, a strong upward gradient exists underneath the creek and inflows are primarily constrained by the low hydraulic conductivity of the streambed and clay soils in the area, not a lack of upward gradient. Further, the hydraulic gradient under the creek and in a nearby agricultural well remained unchanged over the summer and it is not expected that one additional well would reduce groundwater levels enough to affect fluxes to the creek.

If groundwater was pumped into the creek, the target flow rate for maintaining suitable temperatures for salmonids in this reach would be lower than that discussed above, as the temperature of groundwater is lower (10°C) than that of stream water (16.8°C at site 2). Therefore, the allowable increase in stream water temperature between site 2 and 3 would be about 10°C to 12°C before exceeding the 22°C threshold for salmonids at site 3. At a 10°C/km heating rate, the target flow rate with cold groundwater supplementation would be approximately 0.02 m^3/s.

The difference between the lowest recorded flows in 2008 (0.009 m^3/s) and the groundwater target flow rate (0.02 m^3/s) is 0.011 m^3/s. If this shortfall was to be provided solely by groundwater pumping, it would require a pumping rate of at least 0.011 m^3/s (145 imperial gallons per minute) for several hours during the warmest part of the day. This would be a feasible pumping rate if a groundwater source could be established. If flow augmentation by groundwater alone is not feasible, groundwater pumping could be used in combination with other flow augmentation methods such as reservoir water releases.

Stream temperatures at sites 6 to 8 were frequently above 22°C. Estimates of heating rates in relation to discharge cannot be extrapolated to these stream reaches as the analysis is specific to the flow velocities and water depths measured in the stream reach between site 2 and 3. Flow velocities decrease to near zero in the lower reaches but the stream is generally deeper, which would likely result in a different relationship between flows and stream temperature in those reaches. The reaches between site 5 and 8 suffer from many other water quality issues (eutrophication, low DO, lack of

habitat complexity) and it is unlikely that a reduction in stream temperatures alone would make a difference in salmonid use of these reaches.

This analysis indicates that the primary options for maintaining thermally suitable habitat for juvenile salmonids in the upper half of Fortune Creek (sites 1 to 5) are to increase shading between sites 2 and 3 or to increase streamflow volumes by additional water releases from the reservoirs, a reduction in water withdrawals, or groundwater pumping. While naturally occurring groundwater influx provides little thermal relief through advective cooling, it is critical in providing baseflows and therefore reduces stream heating by increasing flows.

6.0 CONCLUSION

The objective of this study was to establish quantitative linkages among groundwater, water temperatures and salmonid populations in a small stream in the BC Southern Interior. Fortune Creek provides year-round rearing habitat for juvenile coho, stream-type Chinook salmon and juvenile and adult rainbow trout. Seasonal backflooding in the lower reaches of Fortune Creek is common in early summer and provides productive rearing habitat for salmonids originating in the Shuswap River system.

This study demonstrates that high water temperatures appear to be the primary limiting factor to the extent of juvenile coho, Chinook and rainbow trout rearing in Fortune Creek. While the upper reaches of the creek upstream of Highway 97 are shaded and remain cool throughout summer, salmonids vacated previously occupied habitats downstream of the highway when water temperatures exceeded their tolerance thresholds. Salmonids in the upper reaches where habitat is thermally suitable may be subject to rapid water level fluctuations and the risk of stranding.

Water chemistry was not consistently related to salmonid distribution and it was concluded that water temperature was the primary determinant of salmonid distribution in the creek. However, high daytime DO conditions, increasing electrical conductivity downstream of streamside discharges, and observations of abundant instream vegetation are consistent with high nutrient conditions. These begin at site 3, but become problematic below site 6. Further consideration of water quality between sites 2 and 5 would be secondary to improvement in thermal conditions.

Generalized Linear Mixed Modeling suggests that, to ensure minimum survival conditions for salmonid rearing in Fortune Creek, daily maximum temperatures should not exceed 22°C. Predicted water temperature thresholds were in good agreement with values reported in literature. The 22°C threshold was exceeded at six of the eight sites studied (3 to 8) on numerous occasions during summer. The entire valley bottom section of Fortune Creek experiences summer water temperatures that range from marginal to unsuitable for salmonid rearing in the current state. Nonetheless, these reaches are utilized by salmonids during parts of the year and have the potential for providing suitable rearing habitat if high water temperatures could be managed.

In spite of that, the creek currently supports all life stages of rainbow trout and limited populations of juvenile Chinook and coho. While the creek has good potential for providing juvenile rearing habitat, conditions are further compromised by water diversion reaching up to 99% of naturalized flows during the low flow season. Low flows in Fortune Creek currently limit rearing in two ways: by drying out some

of the shaded upper reaches that provide the best physical habitat and coolest temperatures and by causing water temperatures to escalate in the lower reaches.

Thermal groundwater modeling confirmed that stream water is infiltrating into the alluvial fan in the upper reaches of Fortune Creek and groundwater is discharging to the creek throughout the valley bottom (Section 4.3). While a strong upward hydraulic gradient exists throughout the summer, upward groundwater fluxes are low (10^{-7} m/s) and are primarily constrained by low streambed hydraulic conductivities. There was no indication that current groundwater pumping in the vicinity of Fortune Creek caused changes to the upward hydraulic gradient along the creek. Groundwater fluxes are diffuse and no localized cold water refugia were identified.

The cooling influence of groundwater influx on stream temperatures is too small to provide significant relief for salmonids suffering from temperature stress (Section 5.3). This results primarily from the solar load created by the lack of shading along most of the creek. Consequently, groundwater inflows alone will not be able to sustain salmonids in Fortune Creek. However, groundwater contributes substantially (88%) to baseflows during low flow periods, and is critical in maintaining fish habitat in the creek by providing most of the discharge during summer.

Recommendations for maintaining suitable water temperatures in Fortune Creek include re-vegetation of the banks to provide shade and strategically supplementing flows to increase discharge at times of high water temperature potential. Thermally suitable habitat for juvenile salmonids could be extended by an additional 4 km downstream of Hwy 97 by either providing substantial shading downstream of Hwy 97 or by maintaining discharge volumes above 0.06 m^3/s to 0.1 m^3/s. Options for increasing discharge include reducing water diversion, releasing water from the headwater reservoirs, augmenting from another headwater source, and pumping groundwater from the deep artesian aquifers into the creek.

6.1 Future Research

To assess the extent of cooling that mature streamside vegetation might contribute, future research should consider experimental shading of some sections of the creek to quantify the cooling influence of groundwater influx without the confounding effects of solar radiation. Given the small size of Fortune Creek and its easily accessible banks, installation of shading over sections of the creek would be relatively easy. Simultaneous measurements of solar radiation would reveal the true solar load under

shaded and unshaded conditions. This would indicate whether investment into re-vegetation efforts would be advisable or whether efforts should focus on increasing streamflows.

A better understanding of streamflow requirements for temperature management in Fortune Creek is required given that groundwater influx has been ruled out for providing thermal relief to salmonids. A detailed statistical model would be useful for relating streamflows to downstream temperatures. This model could account for confounding factors such as air temperatures and provide more accurate estimations of minimum flows required to extend thermally suitable salmonid habitat downstream.

This research has demonstrated that single-stream studies can yield useful information on how juvenile salmonids utilize stream habitats if suitable analytical methods are used. It would nonetheless be useful to extend this type of study beyond the boundaries of Fortune Creek.

While groundwater is discharging throughout the valley bottom reaches of Fortune Creek, the diffuse nature of the influx does not provide distinct cold water refugia that salmonids could access during periods of thermal stress. A gravel bottom river with higher hydraulic conductivities would be better suited for identifying localized cold water discharges. Where distinct cold groundwater inflows can be identified, the use of the cold water patches by salmonids could be monitored to assess whether they are preferred habitats during periods of temperature stress. This would allow researchers to identify temperature avoidance thresholds more precisely.

Establishing sites on several streams would further allow researchers a greater choice in analytical methods because issues such as spatial autocorrelation would not need to be addressed. This type of design would also show whether relationships between salmonids and temperature established in Fortune Creek apply elsewhere.

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