Appendix A

Salmonid distribution in relation to stream temperatures in Fortune Creek, British Columbia:

The influence of surface water and groundwater interactions

Submitted to the Pacific Salmon Foundation by

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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 anthropogenic purposes. 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 (Ebersole et al., 2001; Nielsen et al., 1994; 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 Project Objectives

The objective of this project was to establish quantitative linkages among groundwater, water temperature and fish populations in Fortune Creek. To do so, the following research questions 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?
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?

2.0 BACKGROUND

Pacific salmon are a highly valued resource throughout the north Pacific for their economic, ecological and cultural importance. 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). 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).

2.1 Freshwater Salmon 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).

2.1.1 Water Chemistry

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 and hypoxic waters can act as a barrier to the movement and distribution of salmonids. 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 embryo or alevins as 5 mg/L.

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 McKean and Nagpal (1991). Further, low pH values have negative effects on egg fertilization, embryo development and hatchling success. The tolerance range for pH for trout is between 5 and 9 to 9.5 (McKean and Nagpal, 1991). However, sublethal stress is likely to occur at pH levels approaching the upper and lower limit, especially in combination with other stressors.

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). 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.

2.1.2 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). The lowest risk for bacterial disease infection of salmonids exists between approximately 12.8°C and 15°C (McCullough, 1999).

Increasing water temperatures can force juvenile salmonids to share their habitat with predatory warm water species. These may include species such as northern pikeminnow and largemouth bass (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).

A quantitative graphical summary of salmonid temperature thresholds from literature is presented in Figure 2.1 (McGrath, 2010).

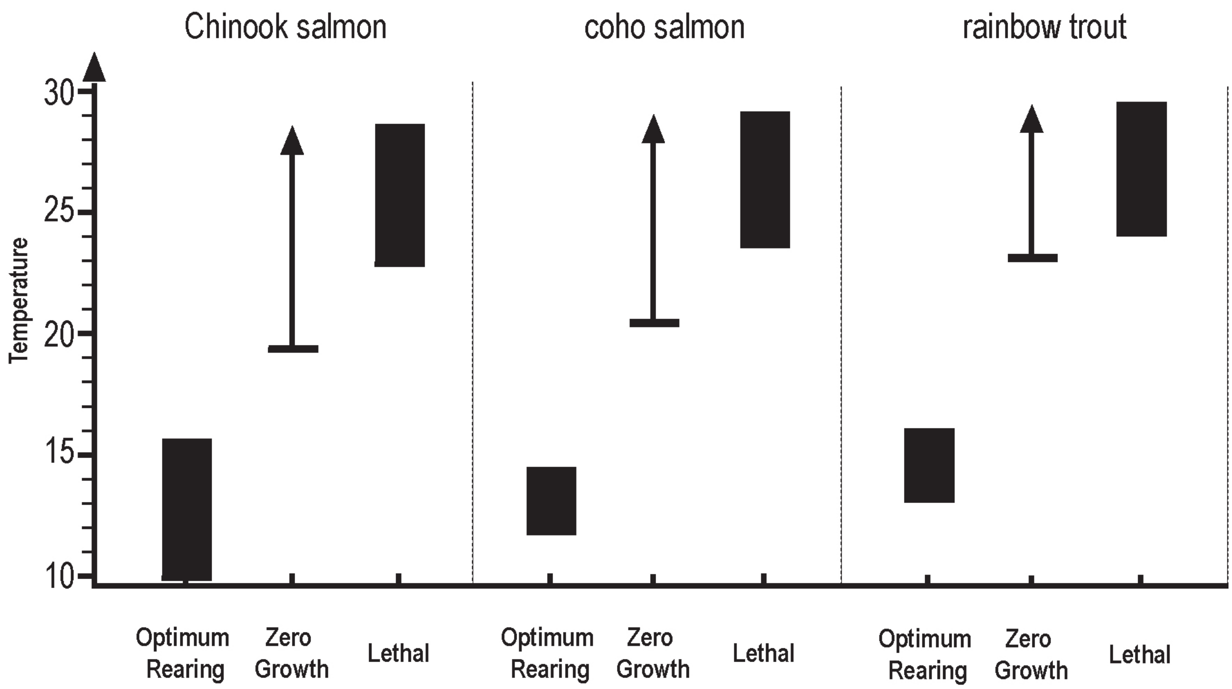


Figure 2.1: Summary of salmonid temperature thresholds from literature.

This project was interested in how salmonid distribution may be limited by high water temperatures. It was 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 (Figure 2.2). Salmonids may be present in the zero growth range for short periods of time, especially where mobility is low.

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.

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.

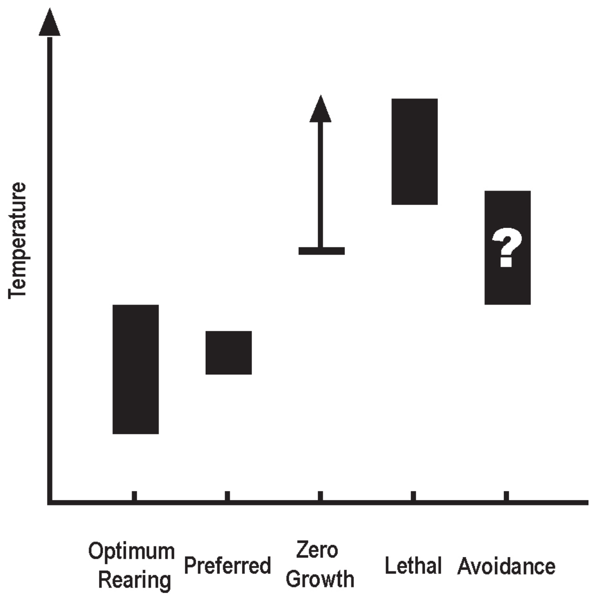


Figure 2.2: Commonly measured temperature thresholds for salmonids.

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 exceed the upper tolerance limits. 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 (Nielsen et al., 1994; Matthews and Berg, 1997).

2.2 Stream Temperature

Numerous mechanisms are involved in energy exchange in a stream. Brown (1969) found that during the day, net all-wave radiation was the predominant energy source on unshaded stretches of small streams. 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). 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.

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.

2.3 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). Streams that are connected to the subsurface through their hyporheic zone 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).

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 refugium from temperature extremes to aquatic fauna (Power et al., 1999).

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). The use of heat as a natural tracer of groundwater movement has been employed since the 1960s (Suzuki, 1960). 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. Where groundwater is flowing it transports heat, and the resulting departure of a temperature profile from a purely conductive state can be used to determine the magnitude and direction of advective flows (Bredehoeft and Papadopulos, 1965).

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.

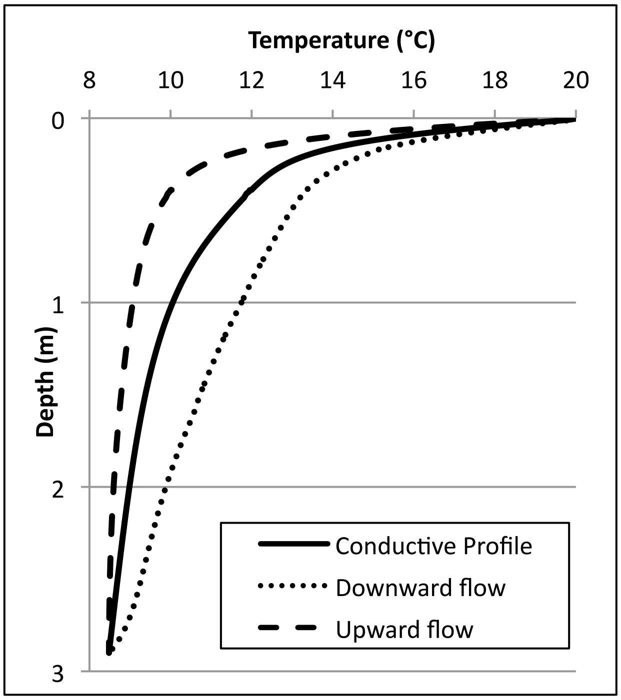


Figure 2.3: Schematic summer season stream bed temperature profile with depth under purely conductive and upward and downward groundwater flow conditions.

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; Conant Jr., 2004; Becker et al., 2004; Hatch et al., 2006; Schmidt et al., 2007; Keery et al., 2007; Essaid et al., 2008).

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) (Healy and Ronan, 1996). 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 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 between 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.

3.0 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 reached from a daily minimum of -8.6°C in January to a daily maximum of 26.7°C in July and August (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 further 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 most 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 (Figure 3.1), including tree planting, installation of cattle crossings, the addition of various flow deflectors (logs and boulders), and instream boulder placement to enhance fish habitat.

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).

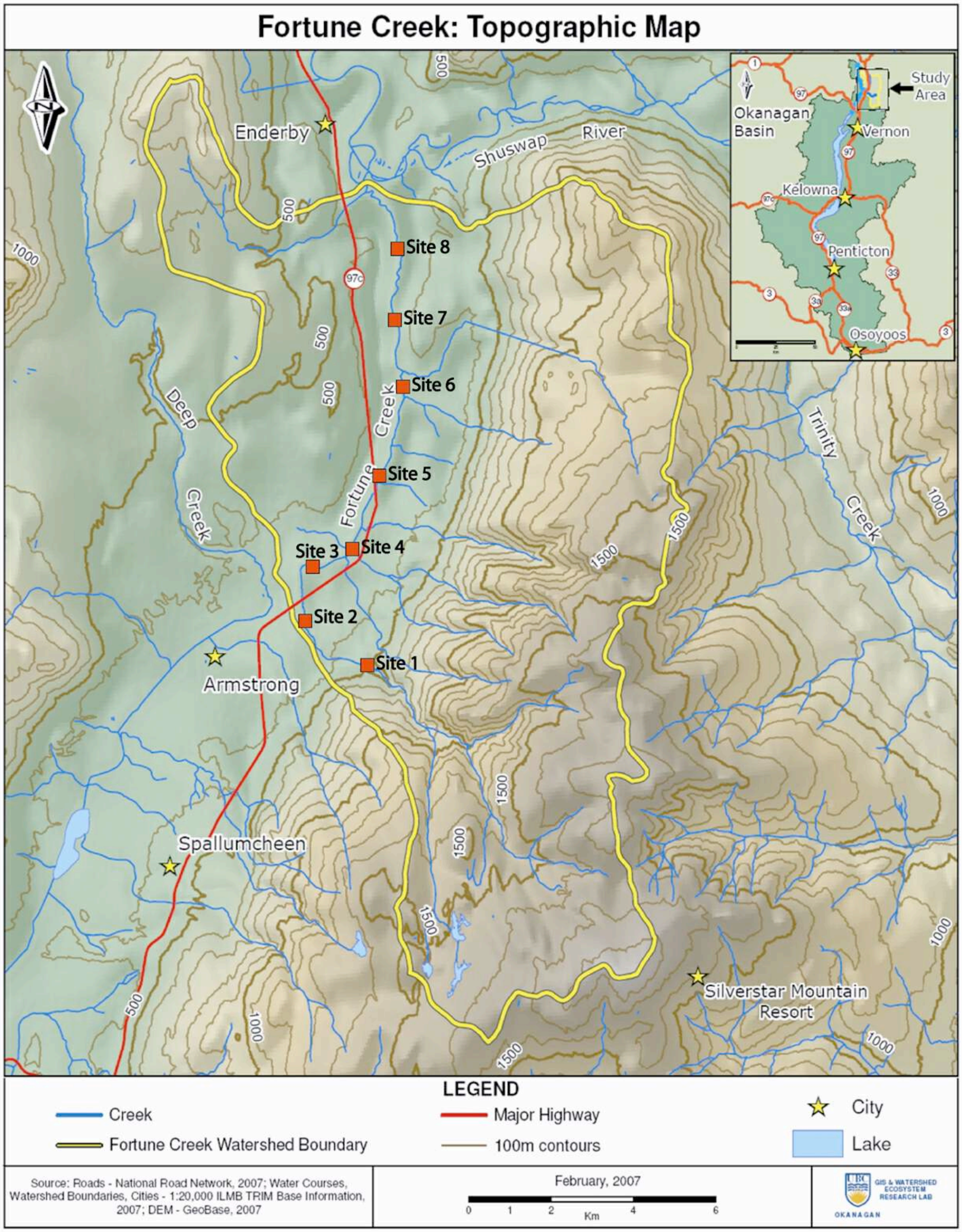


Figure 3.1: Overview map of the study area.

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 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. Two groundwater wells located closer to Armstrong provide water directly into the water distribution system when the stream water is too turbid, or when water levels within the holding tanks fall below a specified reserve capacity. The Fortune Creek water supply provides water for approximately 4,500 users.

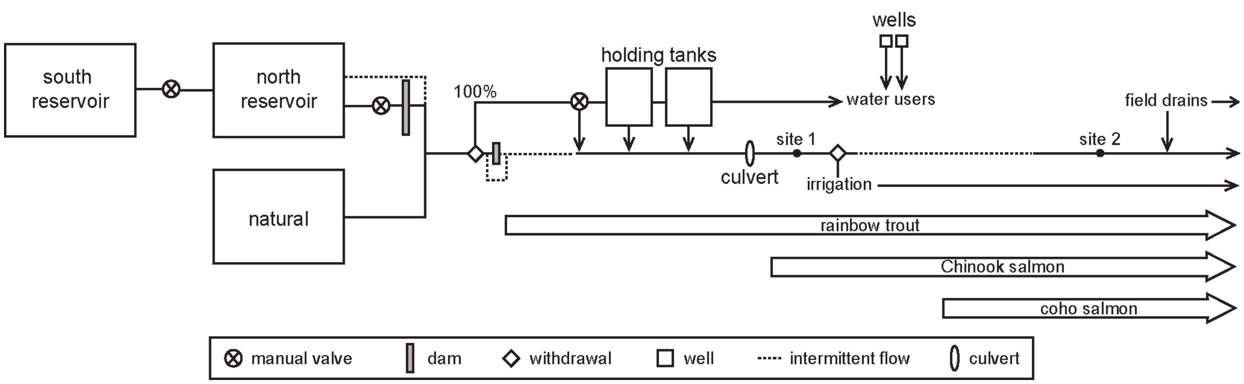


Figure 3.2: Schematic drawing of the processes and facilities located in the upper reaches of Fortune Creek.

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 15% of estimated naturalized flows in the winter to approximately 92% in August (Seebacher et al., 2007).

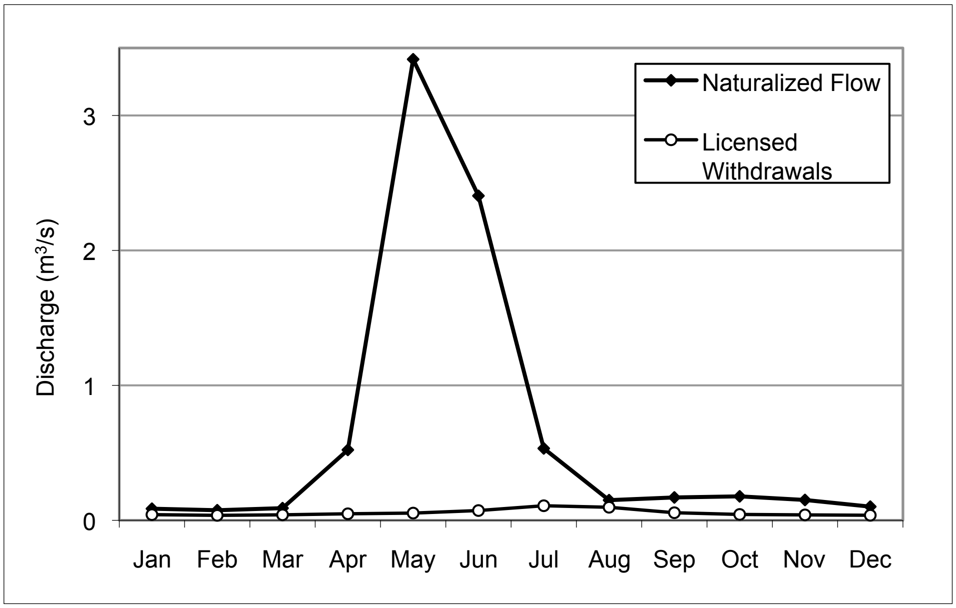


Figure 3.3: Naturalized hydrograph of mean monthly flows in Fortune Creek at Stepney Road (1960-1984) and licensed withdrawals (1998 – 2006) (Seebacher et al., 2007).

Salmonid species inhabiting Fortune Creek include rainbow trout, Chinook salmon and coho salmon. The coho salmon found in Fortune Creek are Interior Fraser 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*), redside shiner (*Richardsonius balteatus*) 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.

4.0 METHODS

4.1.1 Study Site Selection

The entire accessible 15 km length of Fortune Creek was walked for an initial assessment in September of 2007. 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 4.1). Numerous field drains and discharge pipes flowed into Fortune Creek between site 2 and site 6 (e.g. Figure 4.2). 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 4.3). 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 4.4).

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 study sites for the fish enumeration study in 2008 and 2009. Based on these surveys, channel morphology and hydrology, eight study reaches approximately 30 m in length were established (Figure 4.5). 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.

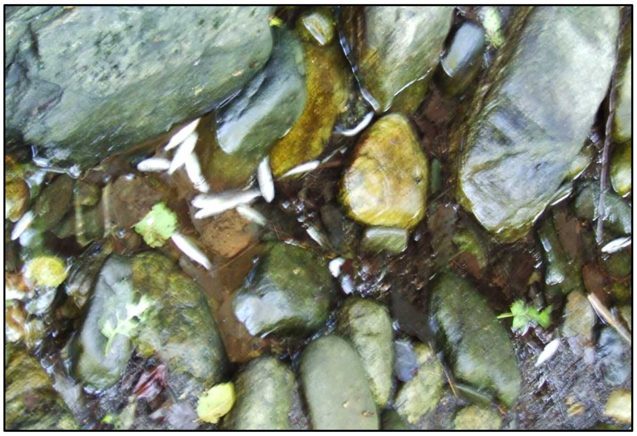


Figure 4.1: Stranded juvenile salmonids in Fortune Creek.

Figure 4.2: Field drains entering Fortune Creek.

Figure 4.3: Bank degradation by cattle.

Figure 4.4: Excessive algal growth noted near site 7.

Detailed site descriptions are presented in Table 4.1. 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 used for irrigation and does not overflow during times when it is actively pumping.

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. 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.

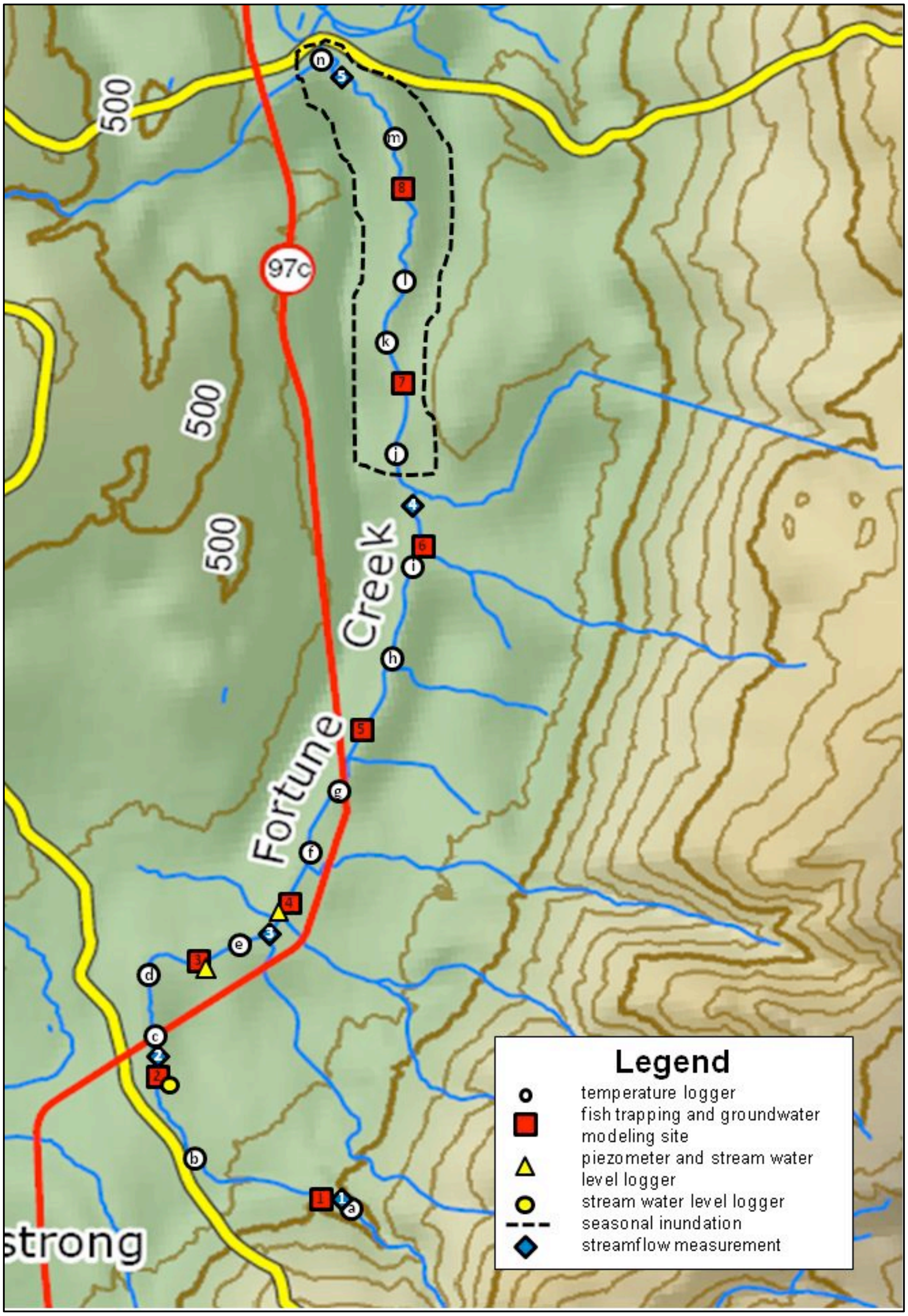


Figure 4.5: Location of study sites and instrumentation deployment on Fortune Creek.

Table 4.1: Description of study sites on Fortune Creek.

							
Site	Location (UTM)	Elevation (m)	Gradient (%)	Site	Location (UTM)	Elevation (m)	Gradient (%)
1	347440E 5589764N	428	5.77	2	346192E 5591251N	369	0.26
Site	Location (UTM)	Elevation (m)	Gradient (%)	Site	Location (UTM)	Elevation (m)	Gradient (%)
3	346634E 5592122N	358	< 0.5	4	347471E 5592360N	356	< 0.5
							
Site	Location (UTM)	Elevation (m)	Gradient (%)	Site	Location (UTM)	Elevation (m)	Gradient (%)
5	348356E 5593766N	356	< 0.5	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

4.2 Field Data Collection

4.2.1 Streamflow

Streamflow in Fortune Creek was measured at five sites (Figure 4.5) 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 the FlowTracker Handheld Acoustic Doppler Velocimeter and wading rod (Sontek/YSI). Streamflow at the Stepney Road station was monitored using an automated hydrometric station in 2009. Additional hydrometric data is contained within Appendix B.

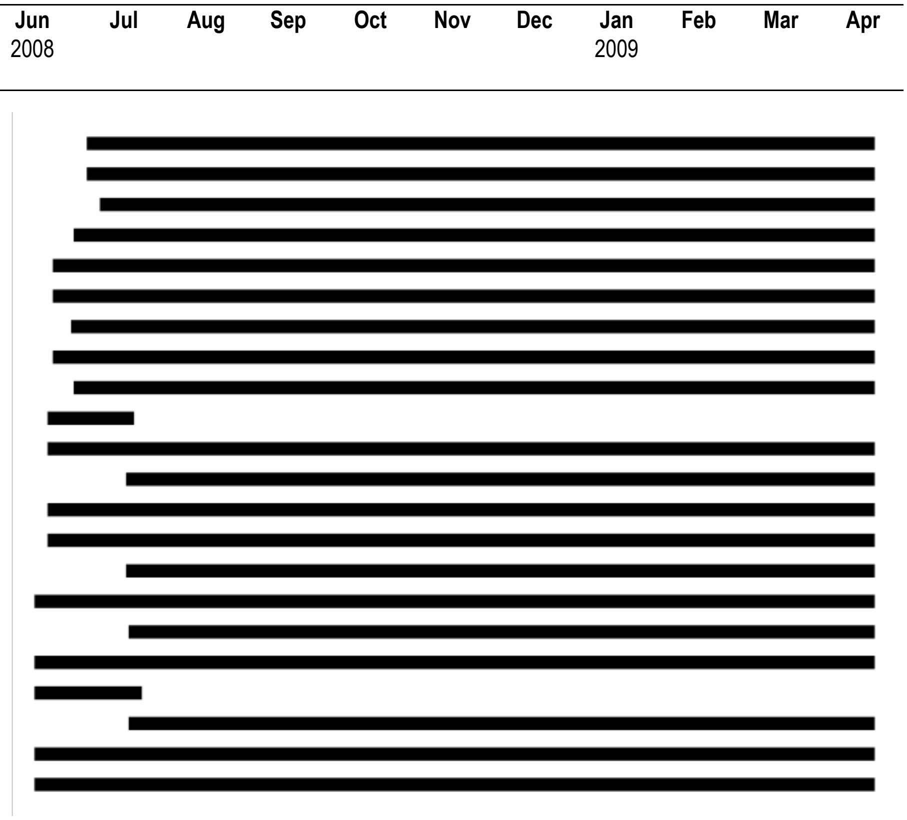
4.2.2 Temperature

At each of the eight study 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 4.5). 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 4.2.

Table 4.2: Water temperature recording periods

Logger Location (Figure 4.5)	Recording Period
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



4.2.3 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 4.3). 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 4.3: 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	

4.2.4 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.

4.2.5 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 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.

4.3 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 4.4). 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 4.4: 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. 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 between 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 4.5) and temporal and spatial correlation needed to be addressed in the analysis.

Table 4.5: 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. In addition to the random effects, the predictor variables presented in Table 4.4 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). Overdispersion in the data for all three salmonid species analyzed in this paper was accounted for by adjusting standard errors and p-values in the fitted models using dispersion parameters.

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. Further details of the statistical methods are presented in McGrath (2010).

4.4 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. Attempts were made to install further peizometers upstream of site 3, but these were frustrated by the cobbly nature of the stream bed. 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 4.6). This information was used to calculate the vertical hydraulic gradient in the streambed.

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 this data using the Hvorslev (1951) method.

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. The best fit between simulated and observed temperature time series was determined by calculating the root mean square error (RMSE). The flux rate that yielded the lowest RMSE between observed and simulated temperatures was selected as the best fit (Su et al., 2004).

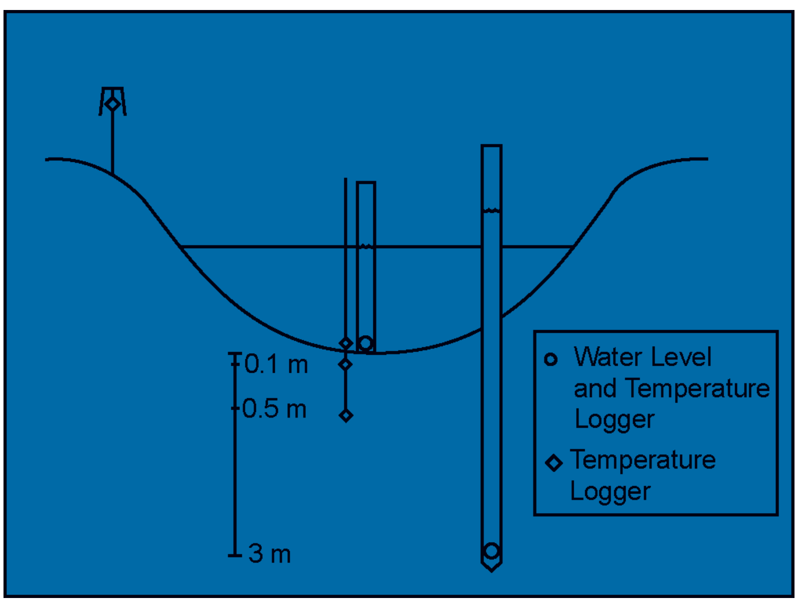


Figure 4.6: Piezometer and temperature logger configuration in Fortune Creek.

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 4.7). 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 4.6. These parameters were taken from literature. For simulation purposes, streambed materials were assumed to be homogeneous and isotropic.

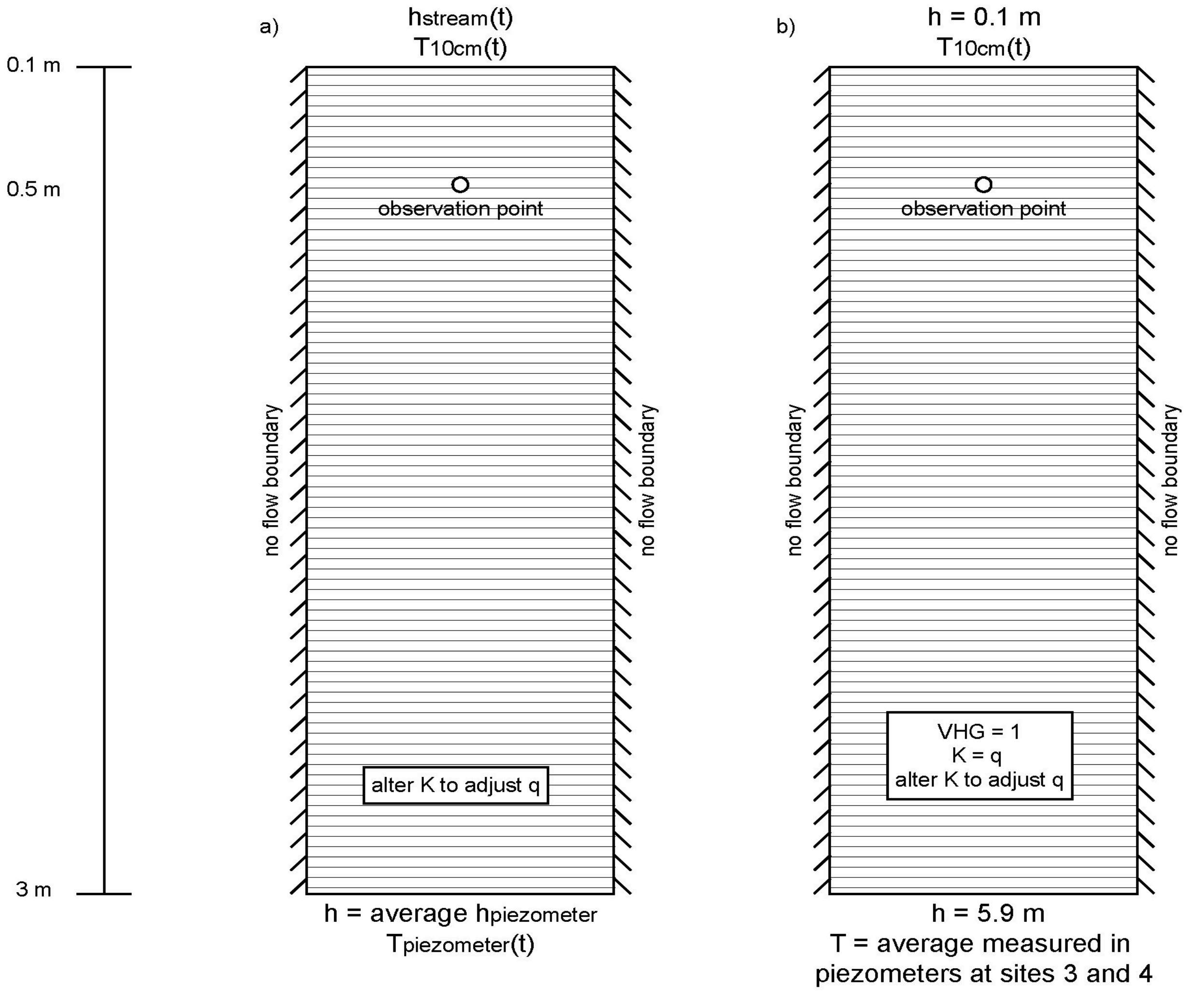


Figure 4.7: Schematic diagram of the model domain and boundary conditions for the VS2DHI streambed temperature simulations at sites a) sites 3 and 4 with piezometers and b) the remaining sites without piezometers.

Table 4.6: 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 x 10^6 J/m^3 K[b]
Heat capacity of water (Cw)	4.186 x 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 4.7). During winter, time periods were not the same for all sites due to logger failures at some sites.

Table 4.7: Groundwater flux simulation periods.

Site	Summer	Winter
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)

4.5 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 study 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 4.8). 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:

Δ ▯=▯∗1000▯▯▯▯▯▯▯▯▯▯▯▯▯▯ ∗▯▯

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 2.2), 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 based on literature values.

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.

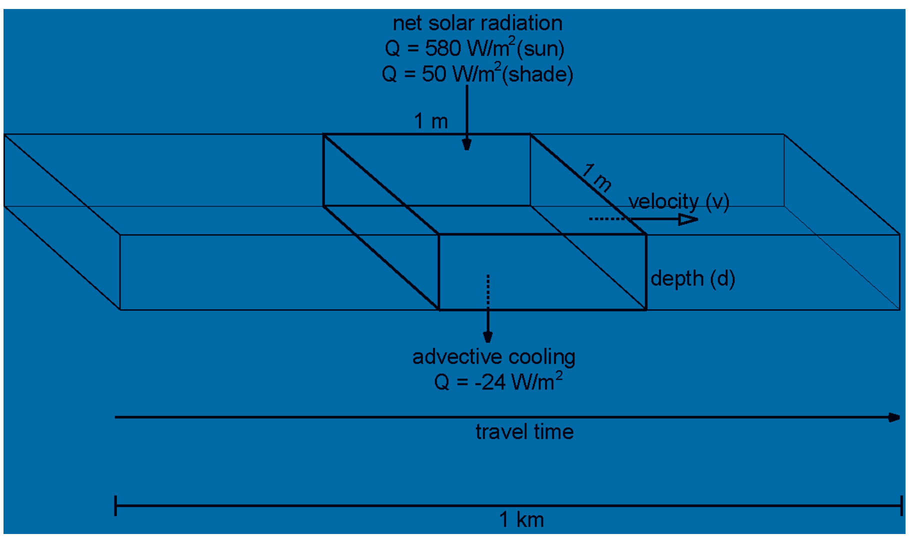


Figure 4.8: Conceptual model of stream temperature change under various streamflow conditions and groundwater inflow.

5.0 RESULTS

5.1 Field Survey Results

5.1.1 Flows and water levels

Detailed hydrometric data is contained in Appendix B. Streamflow measured over the 2008/2009 study season is presented in Figure 5.1. 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 4.5), 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 0.002 m^3/s in 2007, 0.009 m^3/s in 2008 and 0.001 m^3/s in 2009.

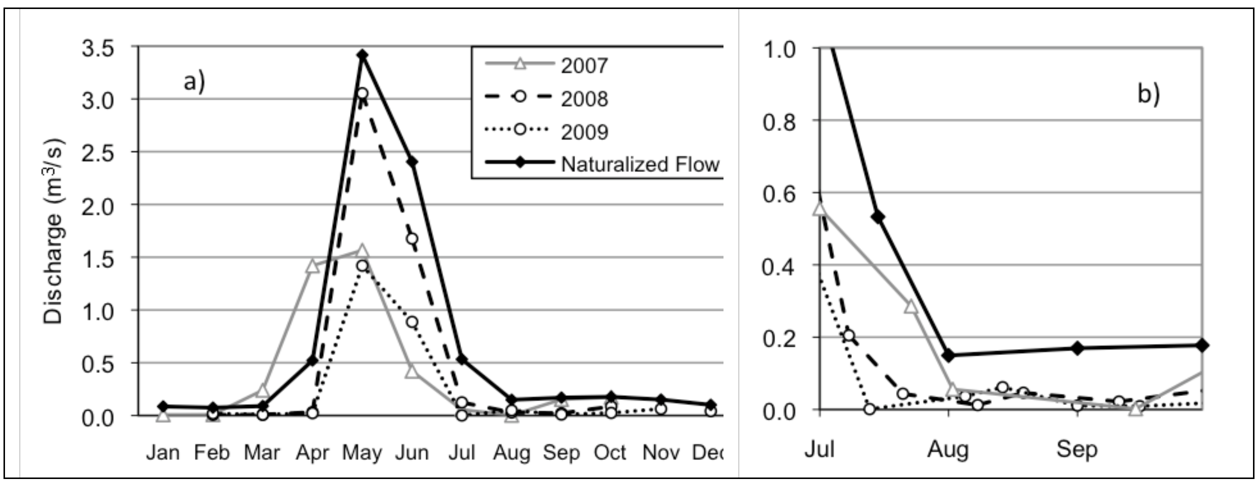


Figure 5.1: Naturalized and measured streamflow volumes in Fortune Creek a) over the year and b) over the summer season.

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 5.2). The maximum daily fluctuation in water levels was 10 cm. Water levels appear higher at site 2 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. Dates and release volumes are presented in Table 5.1.

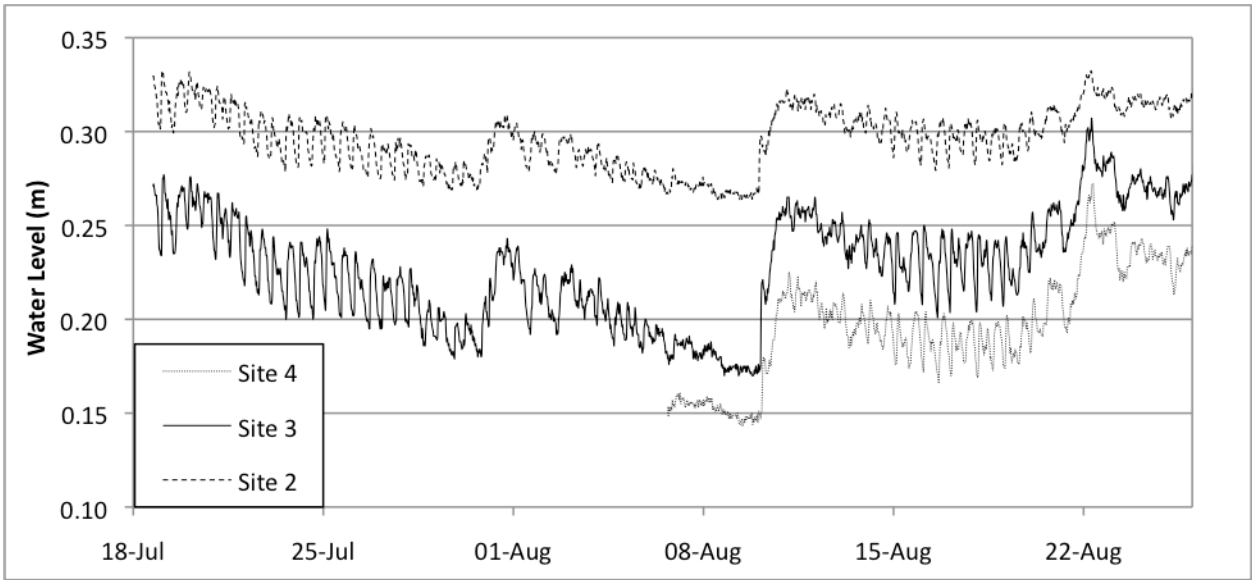


Figure 5.2: Water levels measured in the Fortune Creek over the summer of 2008.

Table 5.1: Water release volumes provided by the City of Armstrong and streamflow measured in Fortune Creek over the summer of 2008.

Date	Released Volume (m^3/s)	Measured 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

Note: Released volume indicates the initial volume of water released from the reservoir on the date of opening. Data provided by C. Clement, City of Armstrong, personal communication, 2008.

5.1.2 Stream and Streambed Temperatures

Over the summer of 2008 Fortune Creek followed a general temperature and discharge pattern typical of small interior streams. Climatic conditions encountered over the study period were considered typical for temperatures and flows in Fortune Creek. 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.

Water temperatures measured in Fortune Creek reached from a low of 0°C to a high of 29.5°C (Table 5.2). Temperatures measured in the air, stream, and the streambed between June 27 and August 31, 2008, are presented in Figure 5.3. 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. The site maintained relatively warm temperatures similar to those at site 1, whereas the remainder of the valley bottom sites was colder.

Table 5.2: Summer and winter temperatures recorded in Fortune Creek.

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	-

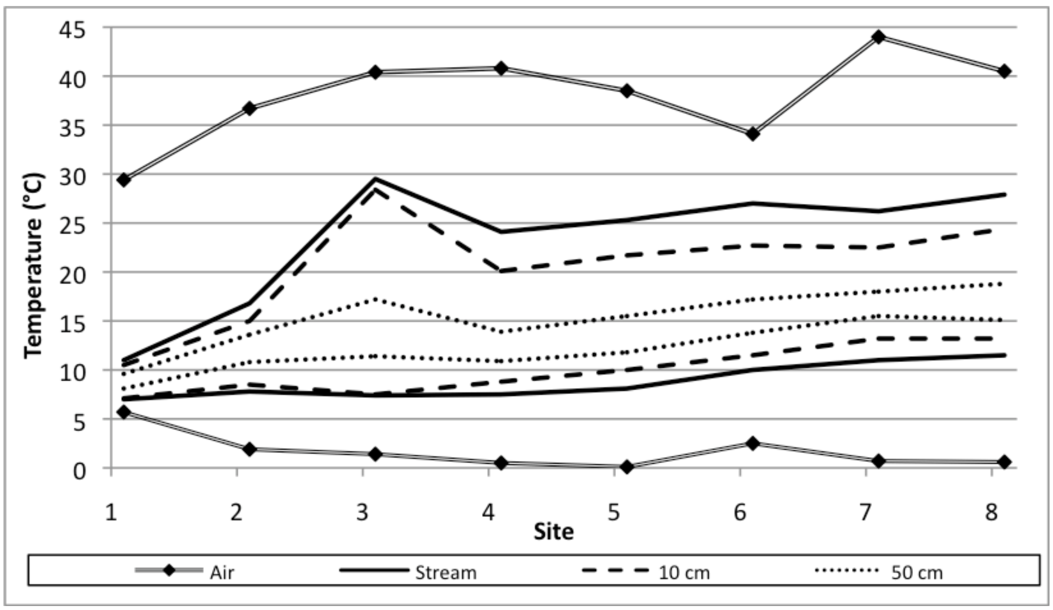


Figure 5.3: Range of air, stream and streambed temperatures recorded at the study sites between June 27 and August 31, 2008.

Streambed temperatures at 10 cm depth are presented in Figure 5.4. 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 narrower 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.

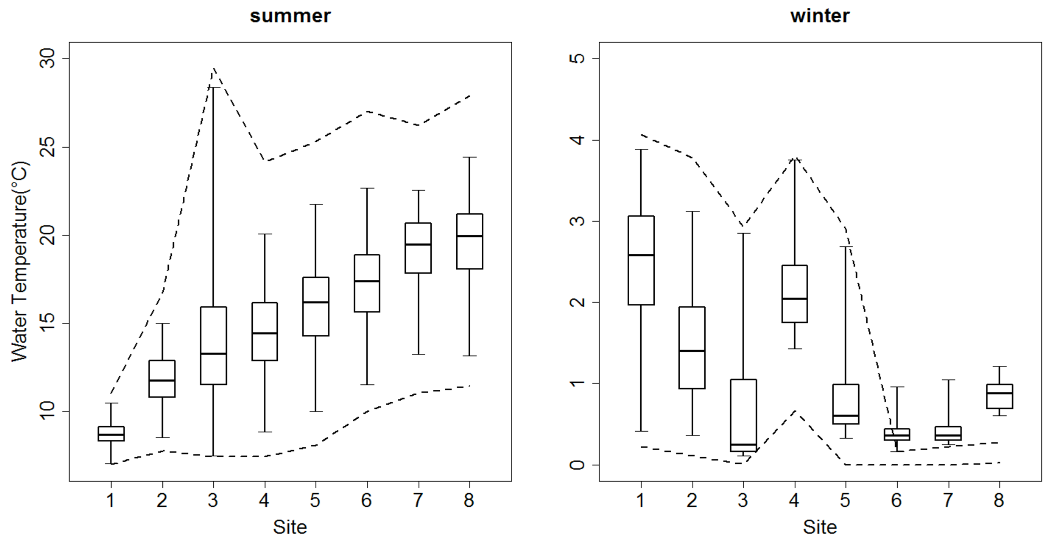


Figure 5.4: Streambed temperatures (boxes) at 10 cm in the bed and minimum and maximum stream water temperatures (dashed lines) in the summer (July 18 to August 31, 2008) and winter (December 14, 2008 to February 28, 2009) in Fortune Creek.

Similar patterns were observed at depths of 50 cm below the streambed (Figure 5.5). 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.

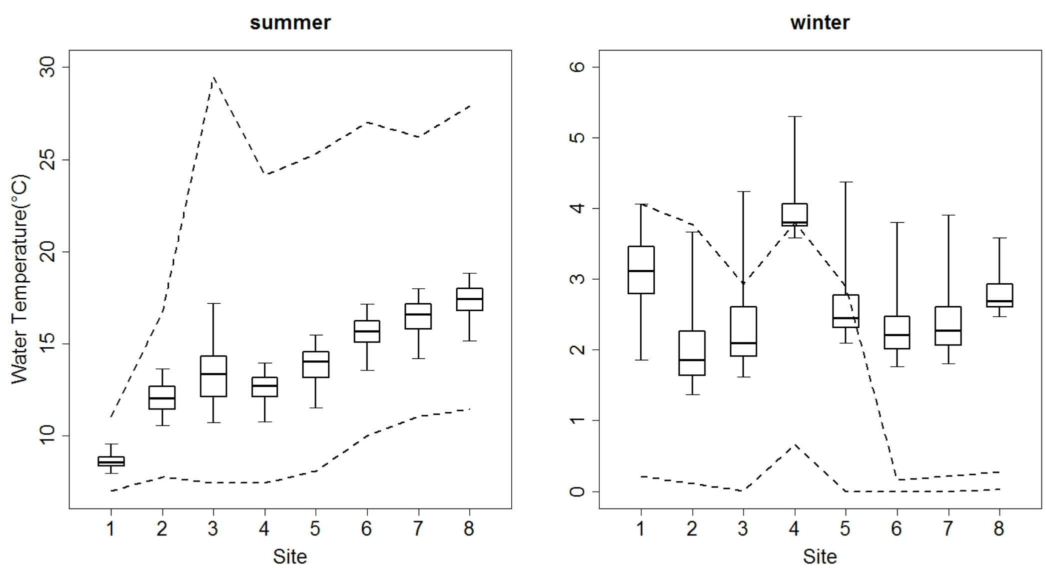


Figure 5.5: Streambed temperatures (boxes) at 50 cm in the bed and minimum and maximum stream water temperatures (dashed lines) in the summer (July 18 to August 31, 2008) and winter (December 14, 2008 to February 28, 2009) in Fortune Creek.

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.

5.1.3 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 5.6). 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 5.6).

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 x 10^{-8}m/s at site 3 and 4.7 x 10^{-8} m/s at site 4. Resulting upward groundwater fluxes calculated from head and hydraulic conductivity measurements were 3 x 10^{-9} m/s and 1.8 x 10^{-8} m/s, respectively.

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.

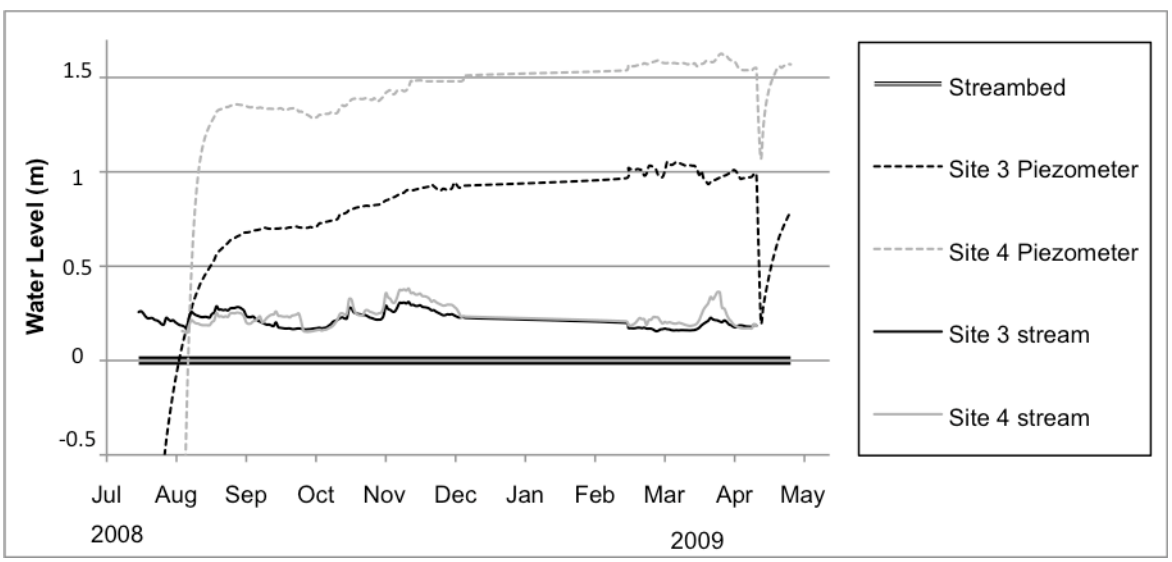


Figure 5.6: Water levels in Fortune Creek recorded at 3 m depth in the piezometers and instream

5.1.4 Water Quality

Water chemistry parameters measured in Fortune Creek over the summer of 2008 (July 1 to September 3) are presented in Figure 5.7 and Table 5.3. 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 were below the 5 mg/L threshold at three sampling times at site 8. Dead juvenile salmonids and adult rainbow trout were observed in the water during low oxygen events. Figure 5.7 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 5.7). 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 5.7). 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.

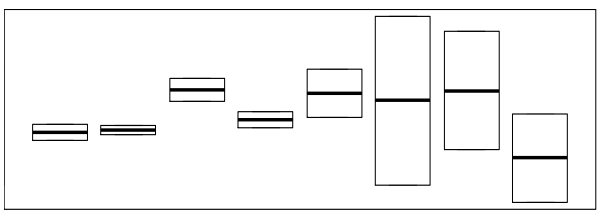


Figure 5.7: Dissolved oxygen saturation, conductivity and pH measured at the eight study sites along Fortune Creek.

Table 5.3: Water chemistry in Fortune Creek between July 1 to 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

5.1.5 Observed Salmonid Distribution

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 (*Ptychocheilus oregonesis*), peamouth chub (*Mylocheilus caurinus*) and carp (*Cyprinus carpio*). 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 5.8). Of the salmonids captured, Chinook and rainbow were slightly more abundant than coho. Fish 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 5.9).

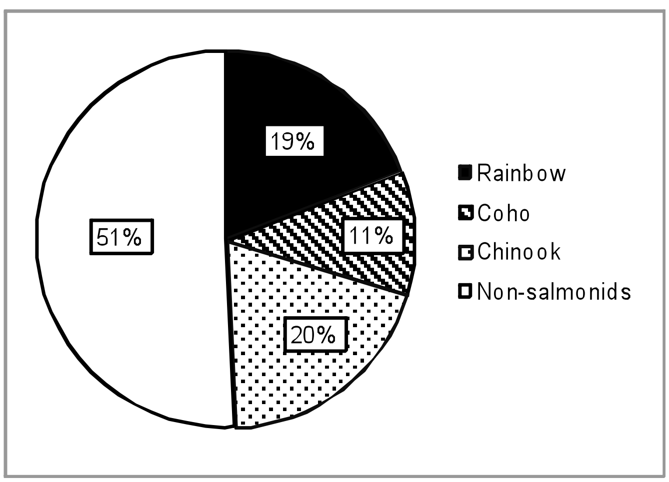


Figure 5.8: Proportion of fish species captured over the 2008/2009 study period.

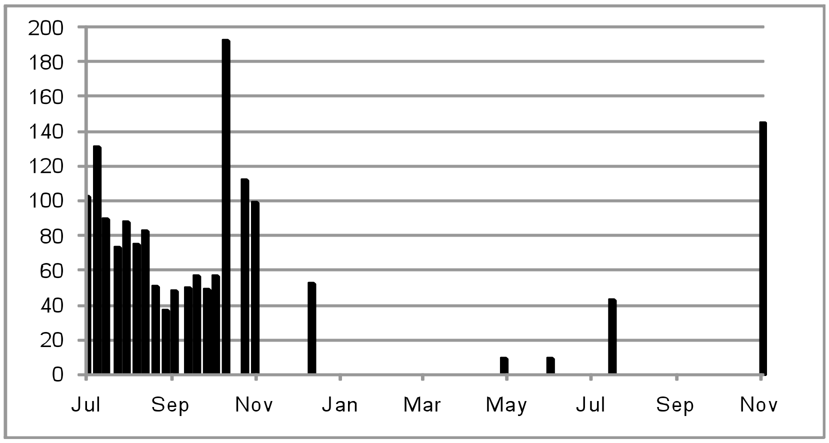


Figure 5.9: Number of coho and Chinook captured in Fortune Creek over the 2008/2009 study period. Each bar represents a sampling event.

The relative abundance of the fish species captured at each study site over the entire study period is presented in Figure 5.10. 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).

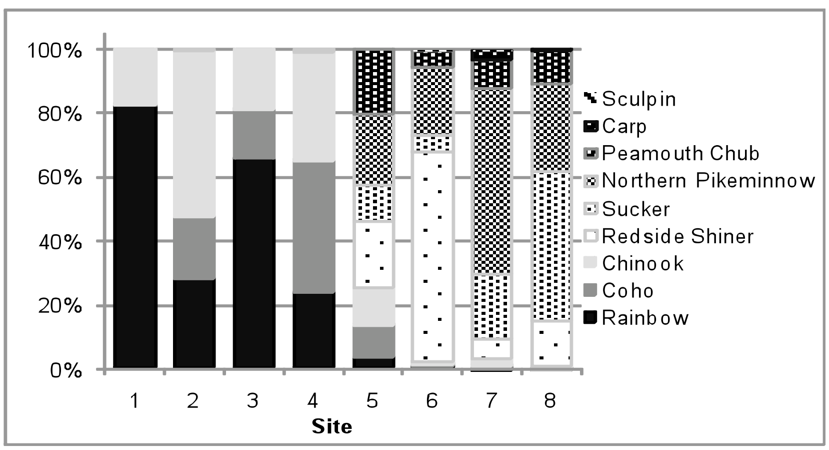


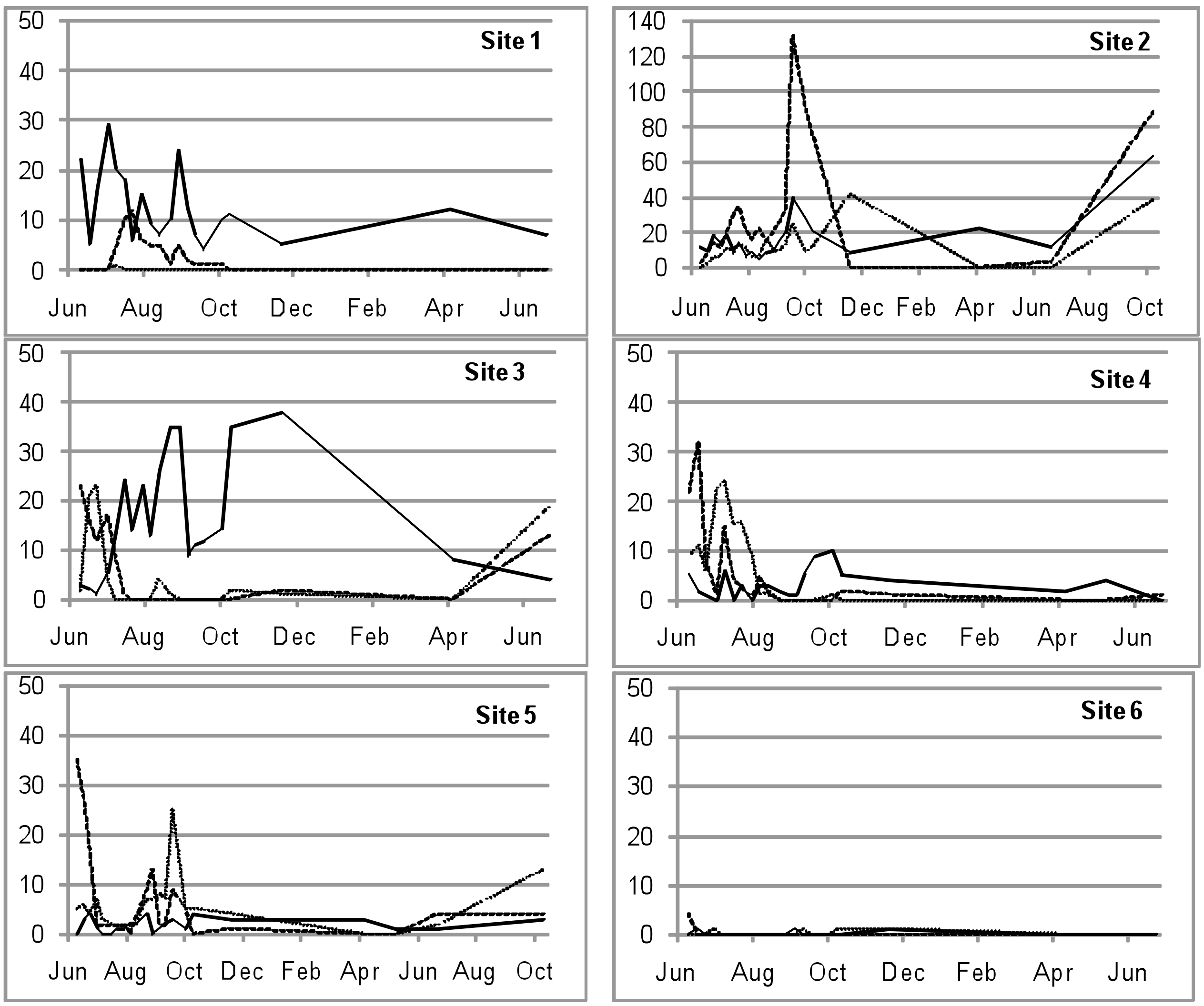
Figure 5.10: Fish species recorded in Fortune Creek over the 2008/2009 study period by site.

Salmonid counts for each site are presented in Figure 5.11 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).



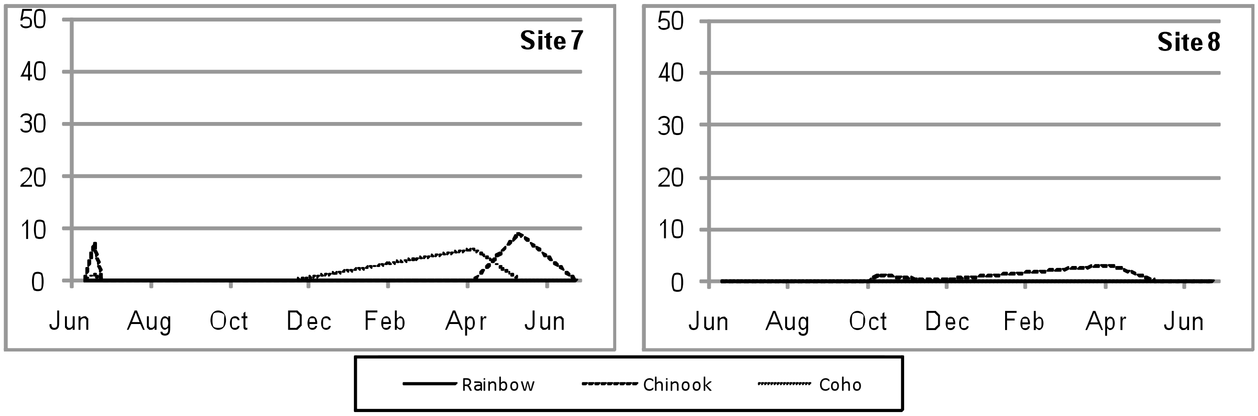


Figure 5.11: Salmonid species captured at each study site over the 2008/2009 study period.

Visual observations prior to the onset of sampling revealed that large numbers of salmonids utilized 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. 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.

The large number of salmonids observed in the backflooded valley-bottom sections during the 2008 freshet likely utilize 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.

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).

Results of the DNA analysis indicate that most of 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.

5.2 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 between species and regression types.

All 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. As a result of the errors encountered, the relationship between some of the exceedance variables and rainbow trout abundance was not analyzed.

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.

5.2.1 Chinook

5.2.1.1 Poisson Regression

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 5.4. 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. This relationship is the opposite of what would normally be expected, and is further explored below for all three species in Section 5.2.5. 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 5.4: 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 5.5. 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 5.12). 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 5.5: 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 5.12). 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 between 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 5.12). On average, predicted Chinook abundance approached zero at minimum temperatures of approximately 12.0°C, however, considering variation between sites and weeks suggests that Chinook could be present up to a minimum temperature of approximately 17.1°C.

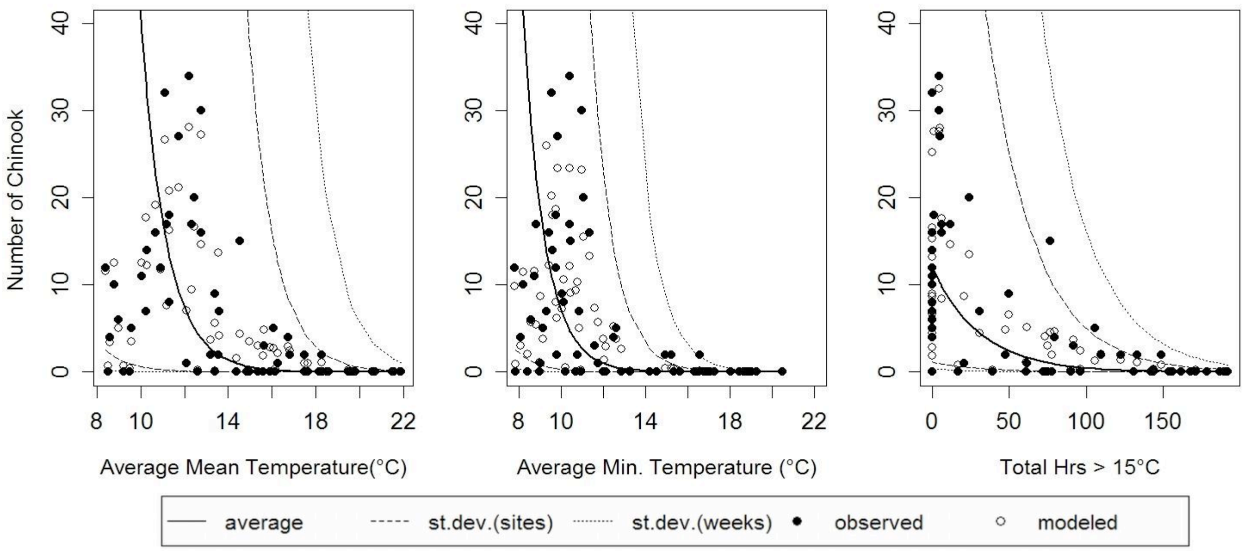


Figure 5.12: GLMM predicted Chinook abundance in Fortune Creek in relation to average minimum and mean water temperatures and total hours exceeding 15°C.

On average, Chinook counts were relatively unaffected when total hours > 15°C were below approximately 25 (Figure 5.12). Past that, Chinook numbers declined to zero at approximately 85 hours. With variation between sites and weeks considered, this number could reach up to 200 hours.

5.2.1.2 Logistic Regression

Six models for predicting the probability of Chinook presence were retained after accounting for residual autocorrelation (Table 5.6). 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 5.6: 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 5.13). 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 between 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 5.14). 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 between 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).

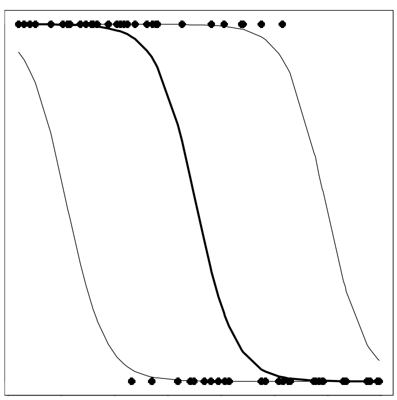


Figure 5.13: Average GLMM predicted probability of Chinook salmon presence in Fortune Creek (thick line) in relation to average mean and minimum water temperatures. Thin lines represent additional variation in probabilities between sites and weeks. The black circles represent observed Chinook presence (1) and absence (0).

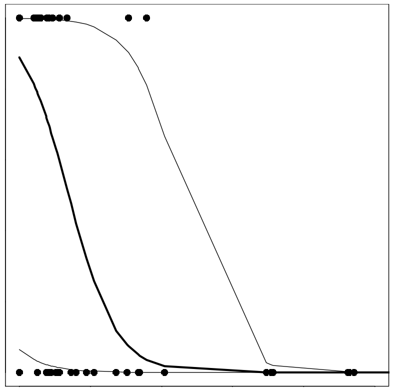


Figure 5.14: Average GLMM predicted probability of Chinook salmon presence in Fortune Creek (thick line) in relation to continuous periods of water temperatures exceeding 17°C and 20°C. The thin lines represent additional variation in probabilities between sites and weeks. The black circles represent observed Chinook presence (1) and absence (0).

5.2.2 Coho

5.2.2.1 Poisson Regression

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. The regression slope estimates for each predictor variable and the interaction terms are presented in Table 5.7.

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 5.7: 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 5.8. 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 5.8: 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 5.15). 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 between sites and weeks, this could reach all the way up to 40°C (Model COP8, Figure 5.15).

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 between 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.

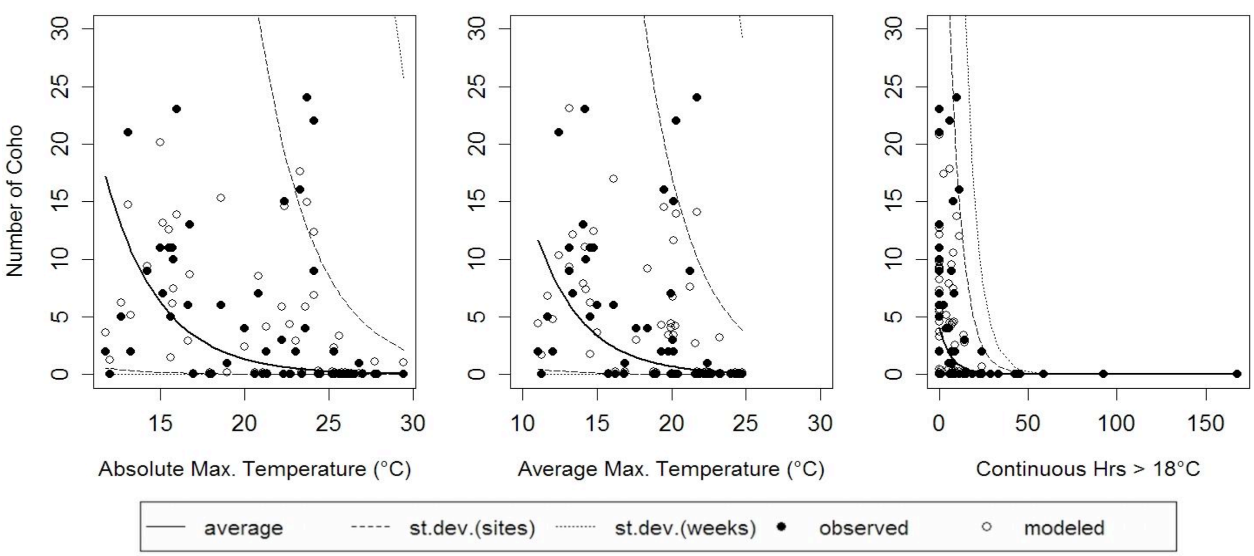


Figure 5.15: GLMM predicted coho abundance in Fortune Creek in relation to absolute maximum and average maximum water temperatures and continuous hours > 18°C.

5.2.2.2 Logistic Regression

Only two models for predicting the probability of coho presence in Fortune Creek were retained after discarding models with residual autocorrelation (Table 5.9). 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 5.9: 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 5.16). 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 between 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 5.16 clearly illustrate the large amount of variation between sites and sample weeks.

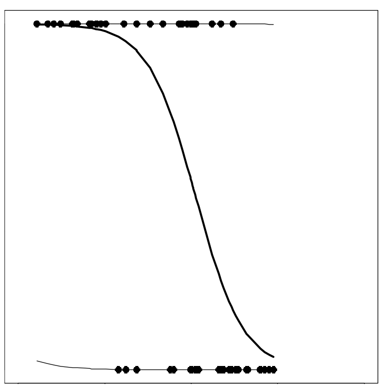


Figure 5.16: Average GLMM predicted probability of coho salmon presence in Fortune Creek (tick line) in relation to absolute and average maximum temperatures. The thin lines represent additional variation in probabilities between sites and weeks. The black circles represent observed Chinook presence (1) and absence (0).

5.2.3 Rainbow

Only two models for predicting age 1+ rainbow trout abundance were retained after accounting for residual autocorrelation and overdispersion (Table 5.10). Even though many variables (including most of the temperature variables) appeared significantly related to the probability of rainbow trout presence/absence in logistic models, all were deemed invalid due to significant autocorrelation in the model residuals. As a result, no logistic regression models for rainbow trout were retained.

The Poisson models retained 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 5.10: 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.

The results suggest that rainbow 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.

Figure 5.17 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 between 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.

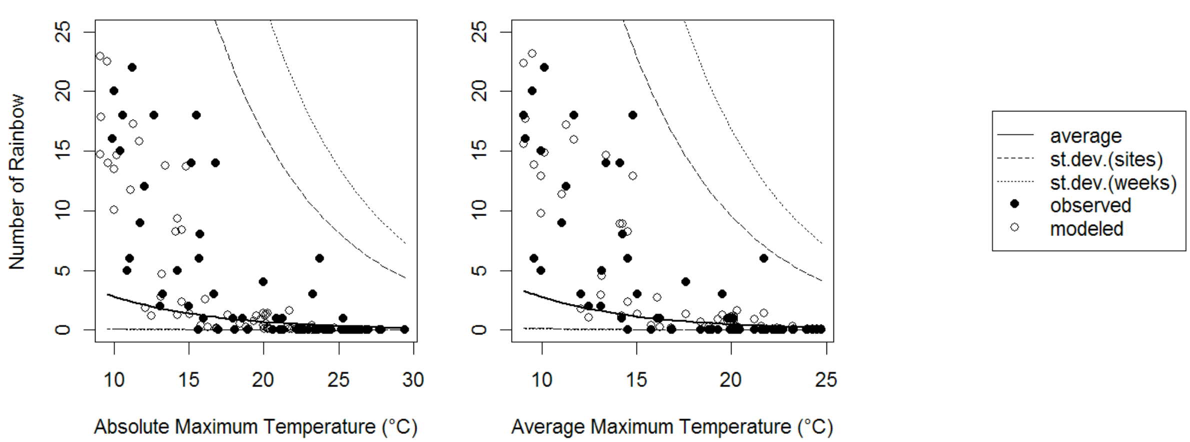


Figure 5.17: Average GLMM predicted rainbow trout abundance in Fortune Creek in relation to absolute maximum and average maximum water temperatures

5.2.4 Statistical 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 effect 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.

5.2.5 Summary of Statistical Analysis

The results of the statistical analysis suggest that juvenile salmonids in Fortune Creek will largely avoid stream reaches where maximum daily water temperatures exceed 22°C (Figure 2.2). This temperature was exceeded at sites 3 to 8 for extended periods of time in July and August of 2008 (Figure 5.18).

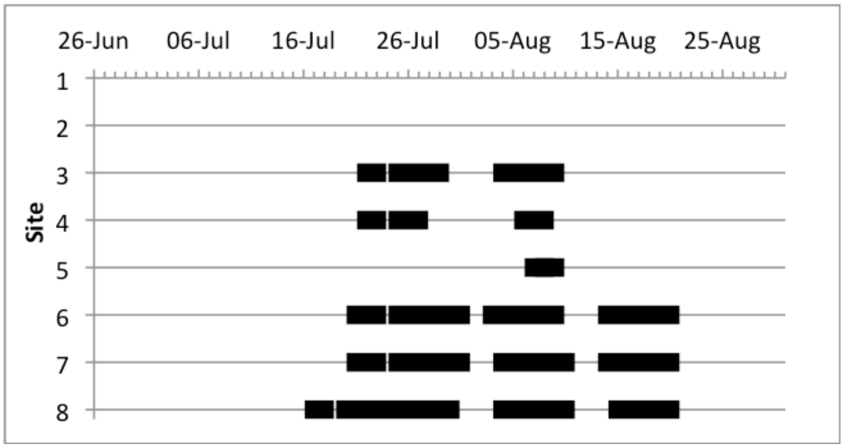


Figure 5.18: Days during 2008 on which maximum water temperatures in Fortune Creek exceeded 22°C, above which juvenile salmonids are expected to be largely absent.

While most of the chemical water quality parameters (pH, DO, conductivity) were significant predictors of Chinook and coho abundance in the 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 5.7) in the lower reaches of Fortune Creek where the unshaded channel was observed to be inundated with aquatic plants and algae (Section 4.1.1). 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 of accumulated organic carbon are characteristic of high nutrient conditions.

In the upper reaches (sites 1 to 2) where most salmonids were captured, DO was less variable (Figure 5.7), 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 permanently 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 the likely 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 5.1.4). 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 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).

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 decrease in 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. 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.

5.3 Modeling of Groundwater Surface Water Interactions

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). Model results of the simulations at all sites are presented in Table 5.11 (summer) and Table 5.12 (winter). 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 5.11: 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	
2	3.5×10^{-6}	0.22		Down	2.7	
3	negligible	--	--	Neutral	1.5	Figure 5.19
4	5.7×10^{-7}	0.12	1×10^{-6}	Up	1.5	Figure 5.21
5	4.5 to 6.6×10^{-7}	0.28		Up	1.5	
6	1.1 to 2.0×10^{-7}	0.12		Up	1.5	
7	1.1×10^{-7}	0.09		Up	1.0	
8	2.3×10^{-7}	0.12		Up	1.75	

Table 5.12: Simulated winter streambed 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	
3	negligible	--	--	neutral	1.5	Figure 5.20
4	1.8×10^{-7}	0.02	4×10^{-7}	Up	1.5	Figure 5.22
5	2.0×10^{-7}	0.09		Up	1.5	
6	5.7×10^{-7}	0.03		Up	1.5	
7	9.5×10^{-8}	0.05		Up	1.0	
8	1.7×10^{-7}	0.02		Up	1.75	

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 5.19). 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 5.20). 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.

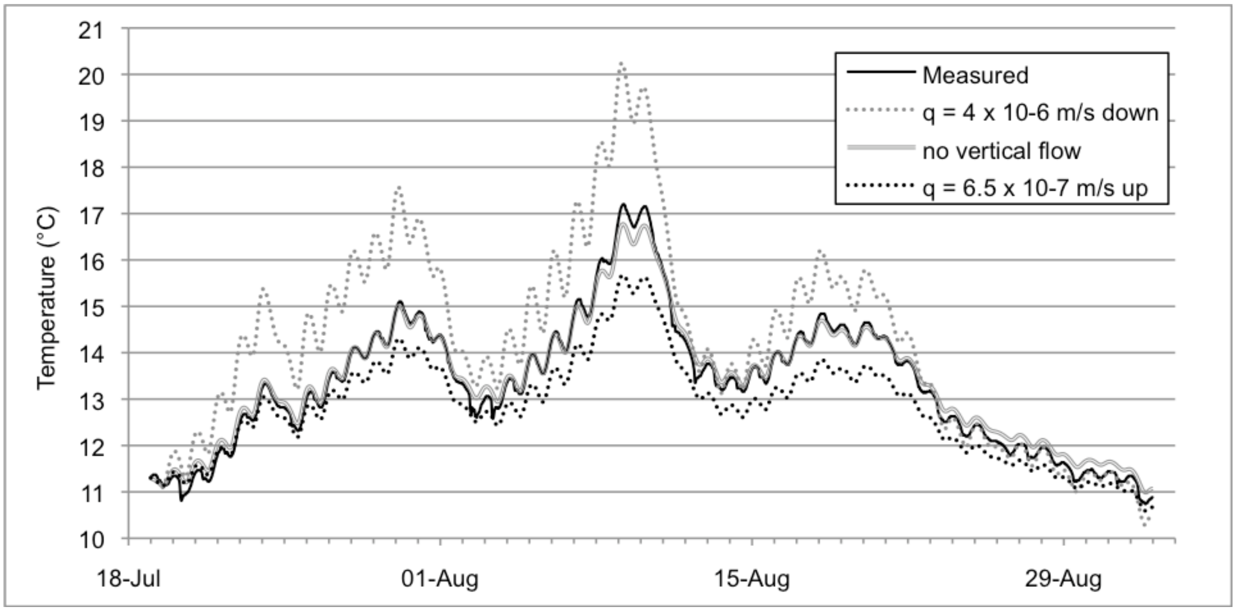


Figure 5.19: Simulated and measured summer temperatures 50 cm below the streambed under various groundwater flow conditions at site 3.

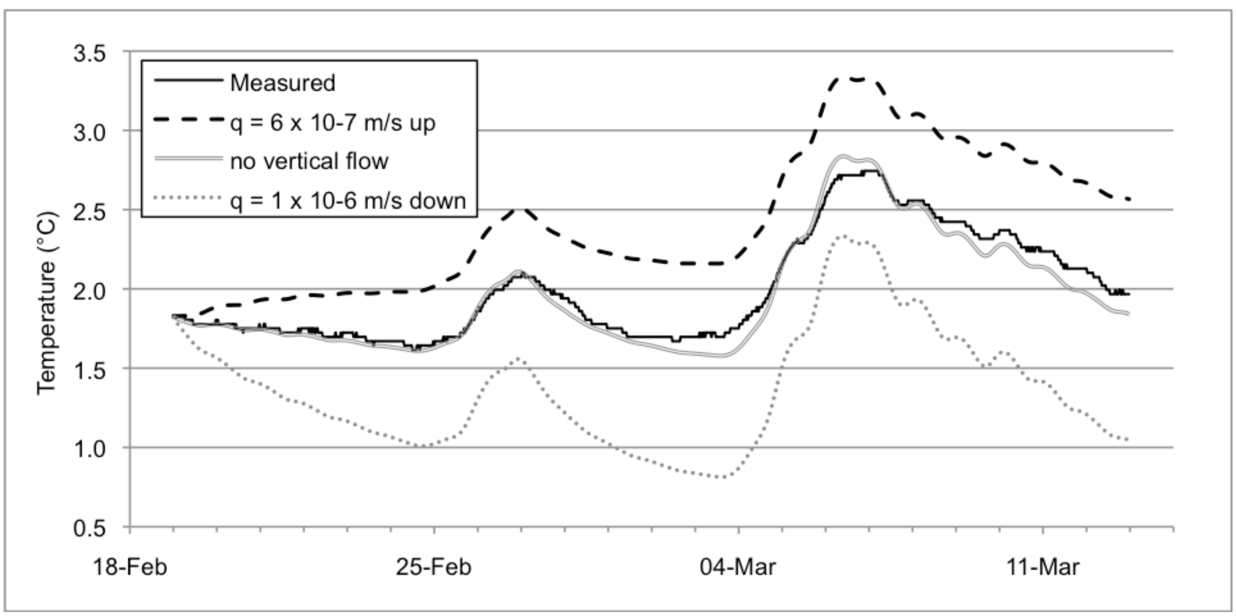


Figure 5.20: Simulated and measured winter temperatures 50 cm below the streambed under various groundwater flow conditions at site 3.

Simulated and measured streambed temperatures at 50 cm depth at site 4 are presented in Figure 5.21. 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 1×10^{-6} m/s, with an upward groundwater flux of 5.7×10^{-7} m/s. This is almost two orders 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 5.22). As in summer, small changes in seepage velocity resulted in large differences in simulated temperatures.

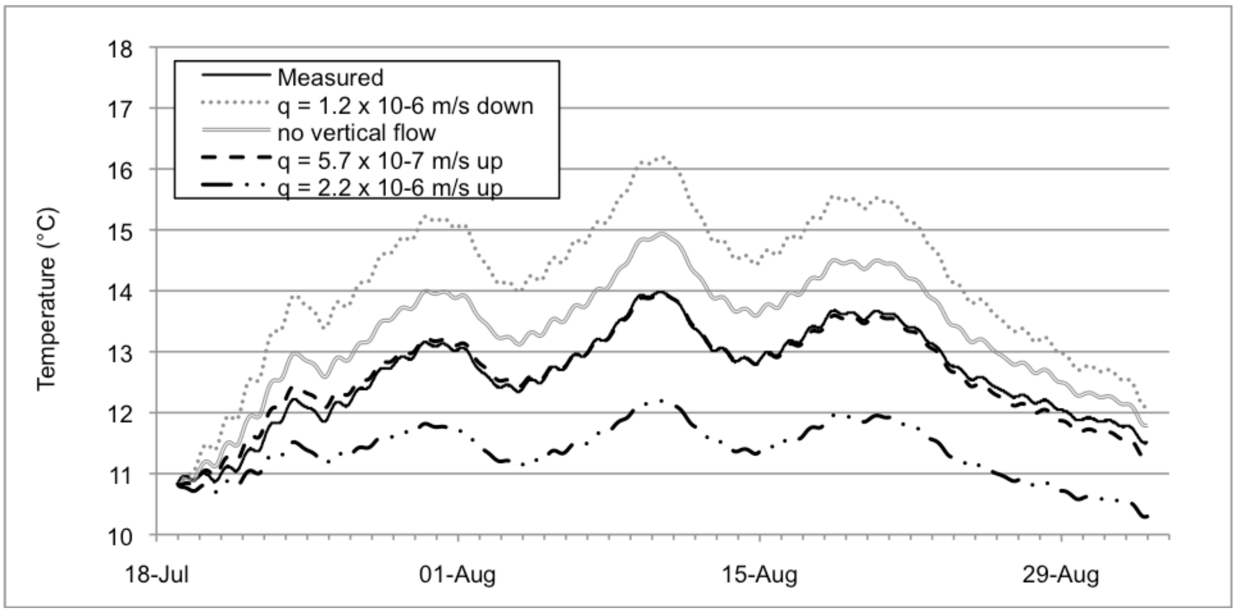


Figure 5.21: Simulated and measured summer temperatures 50 cm below the streambed under various groundwater flow conditions at site 4.

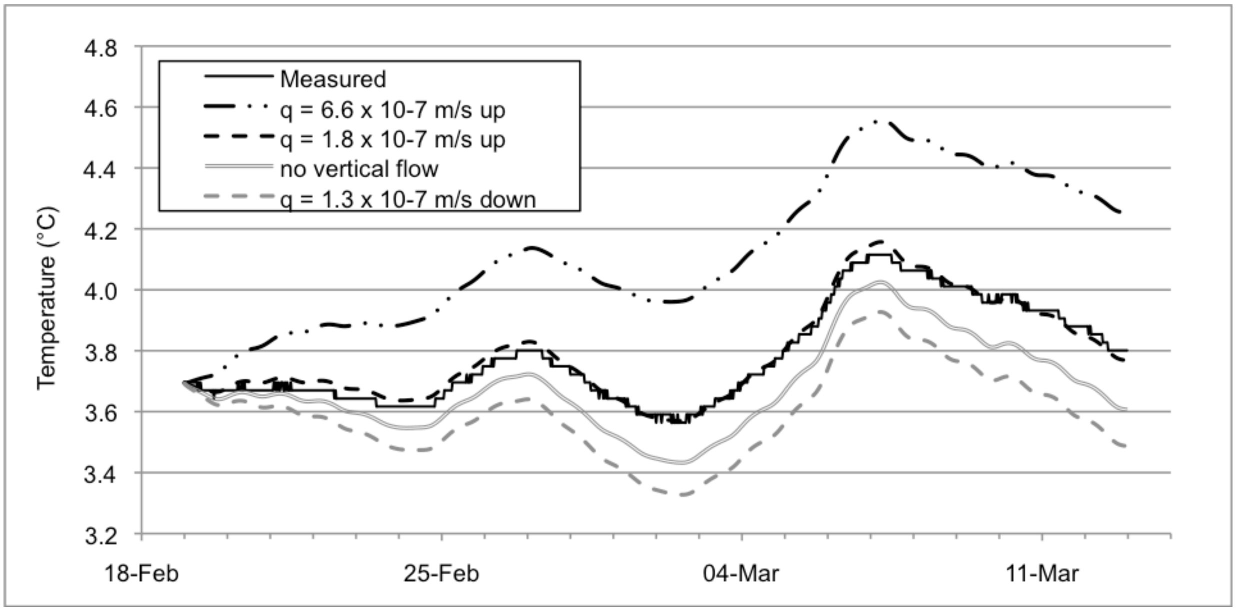


Figure 5.22: Simulated and measured winter temperatures 50 cm below the streambed under various groundwater flow conditions at site 4.

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 x 10^{-7} m/s to 2.2 x 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.

At site 2, downward flux was estimated at 3.5 x 10^{-6} m/s. 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. Attempts to install peizometers in this site were hampered by the cobble substrate

During summer, streambed fluxes were upward at sites 5 to 8, ranging from 6.6 x 10^{-7} m/s at site 5 to 1.1 x 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.

Given that the upward hydraulic gradient was strong and stable over the study season, 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.

5.4 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 00). 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 5.1.1). These were followed by noticeable increases in stream water levels and reductions in stream temperatures at sites 2, 3, and 4 (Figure 5.23 to Figure 5.25). 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 of 2008, temperatures at site 2 remained well below 22°C (Figure 5.23), which was taken as a general upper distribution limit for Chinook, coho and rainbow trout in Fortune Creek based on results from the statistical analysis and literature review. 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 5.24). 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 5.25). 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. The July 23rd release was followed by no change in water level or flow rate, the July 29th release was followed by a small increase in water levels (about 3 cm), and the August 6th release was followed by a 7 cm increase in water levels. It is unclear whether increases in flow were caused by the reservoir water releases or rainfall events that coincided with the releases.

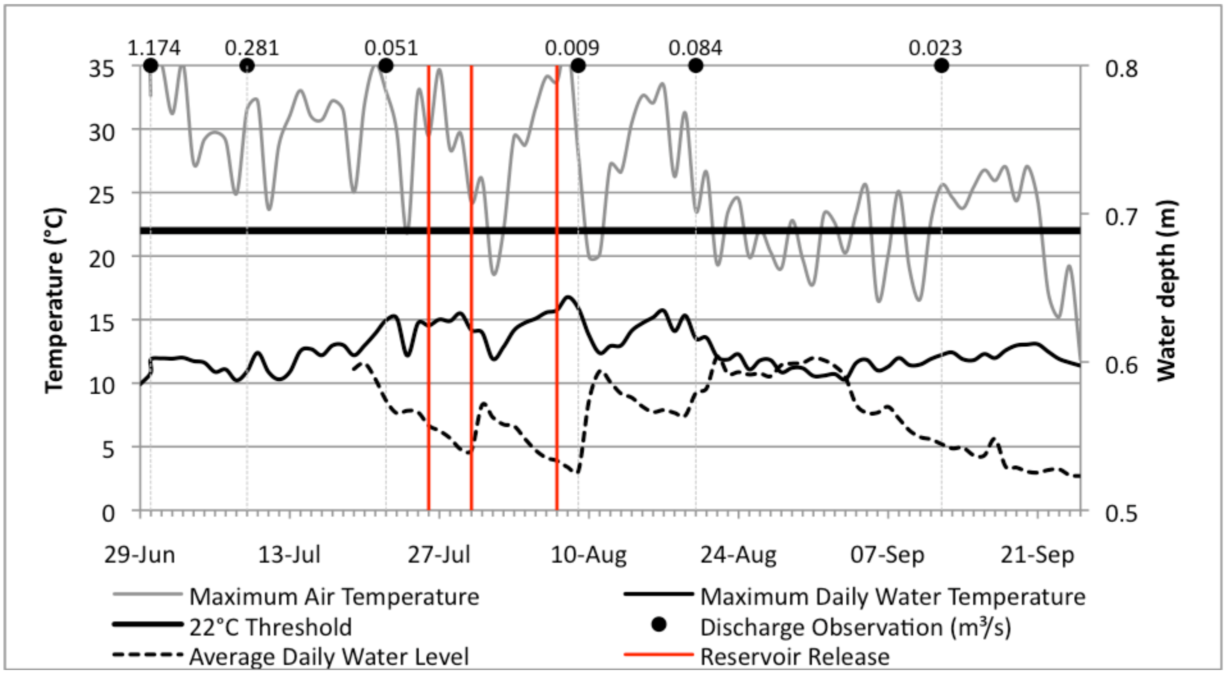


Figure 5.23: Water level, stream temperature and discharge at site 2 over the summer 2008 in Fortune Creek.

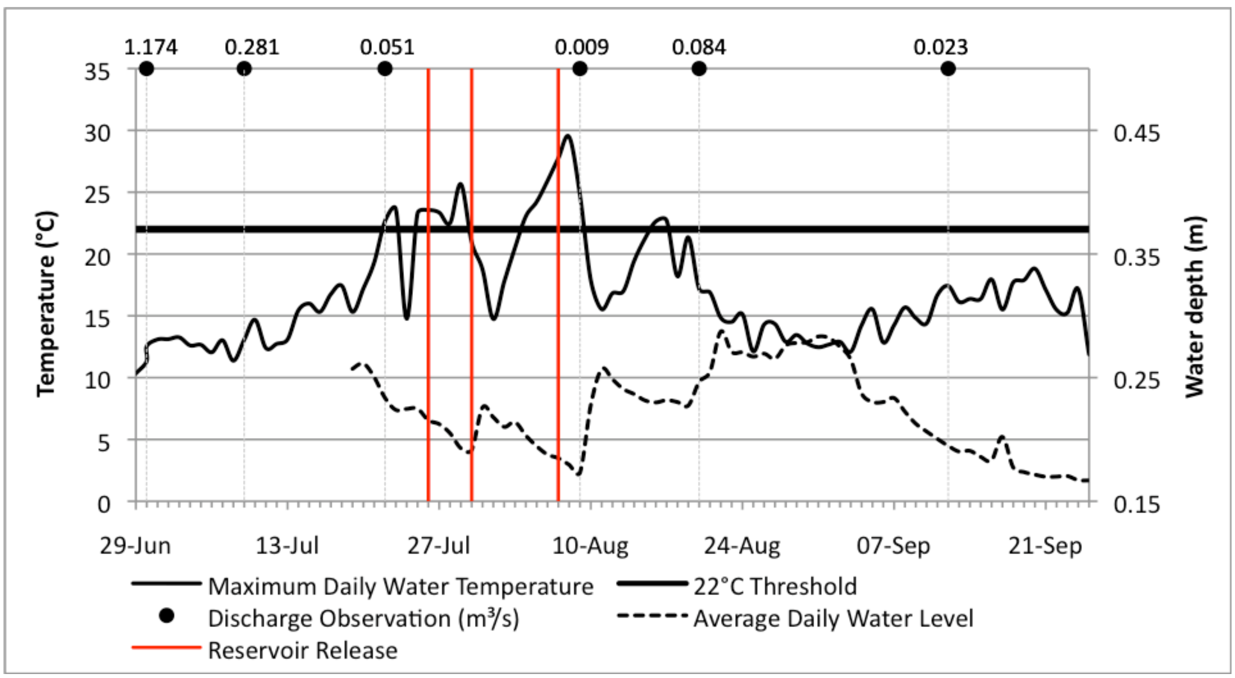


Figure 5.24: Water level, stream temperature and discharge at site 3 over the summer 2008 in Fortune Creek.

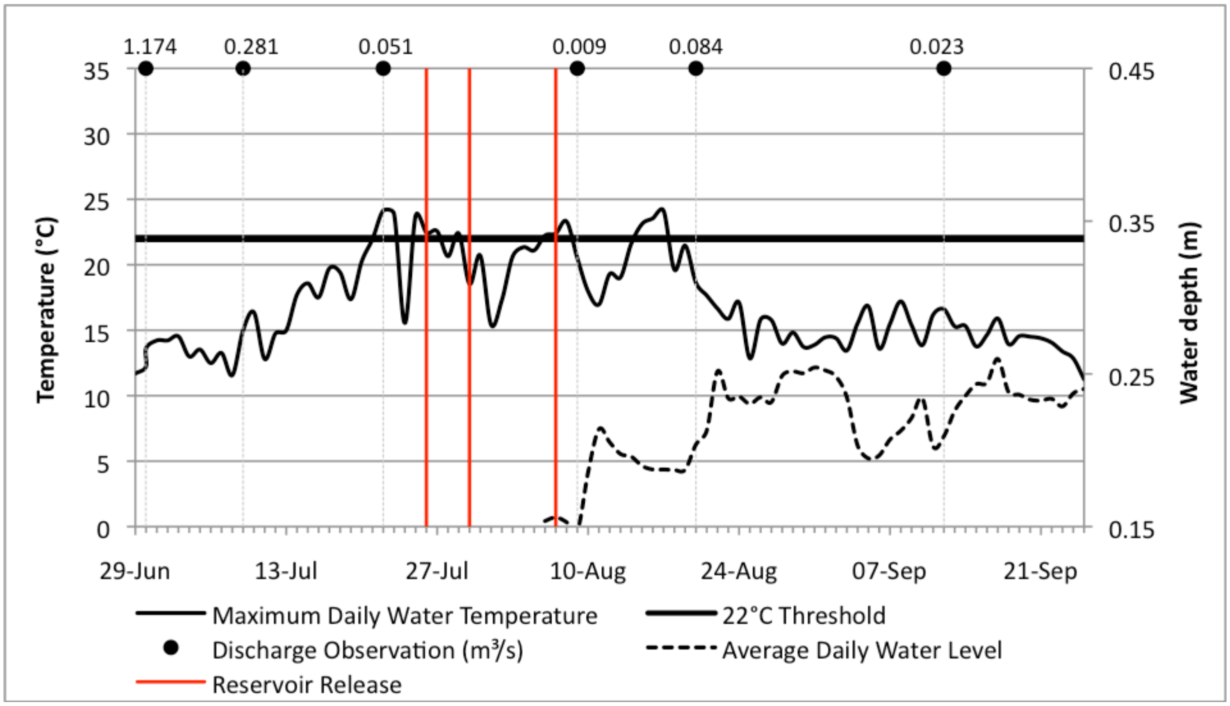


Figure 5.25: Water level, stream temperature and discharge at site 4 over the summer 2008 in Fortune Creek.

Figure 5.26 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.

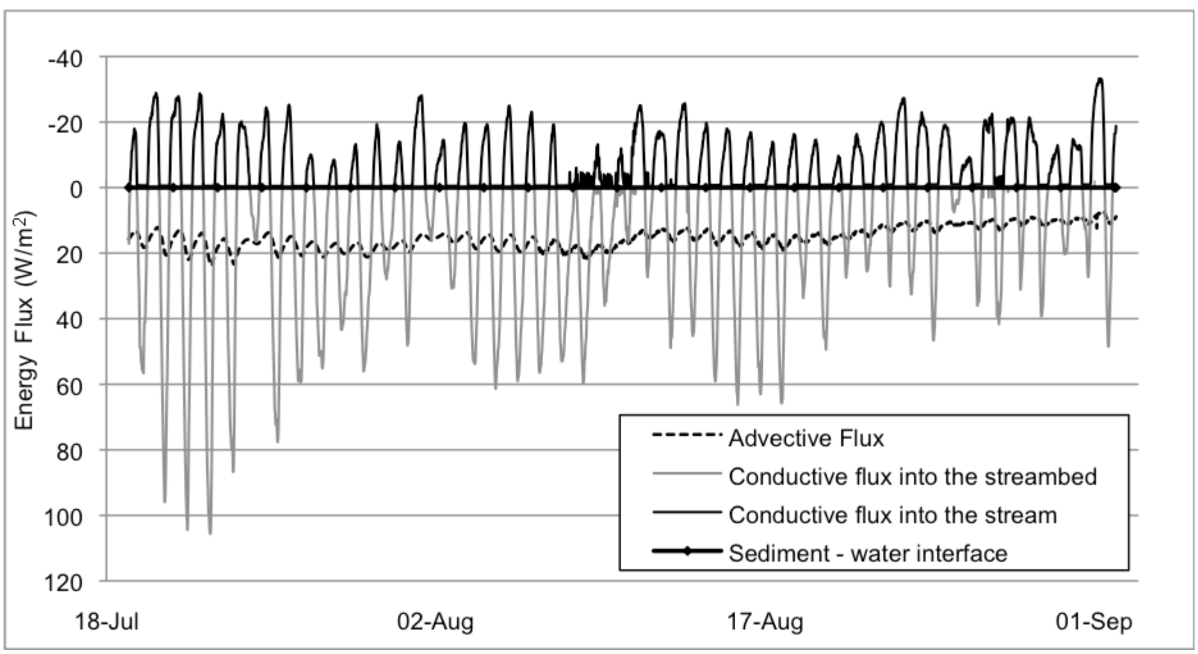


Figure 5.26: Simulated conductive and advective energy fluxes between the streambed and the creek at site 4 in Fortune Creek.

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 5.13. 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 5.13). 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 5.27). 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 5.27, Table 5.13), 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.

Table 5.13: 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 (m^3/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 5.27 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.

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 5.13). 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).

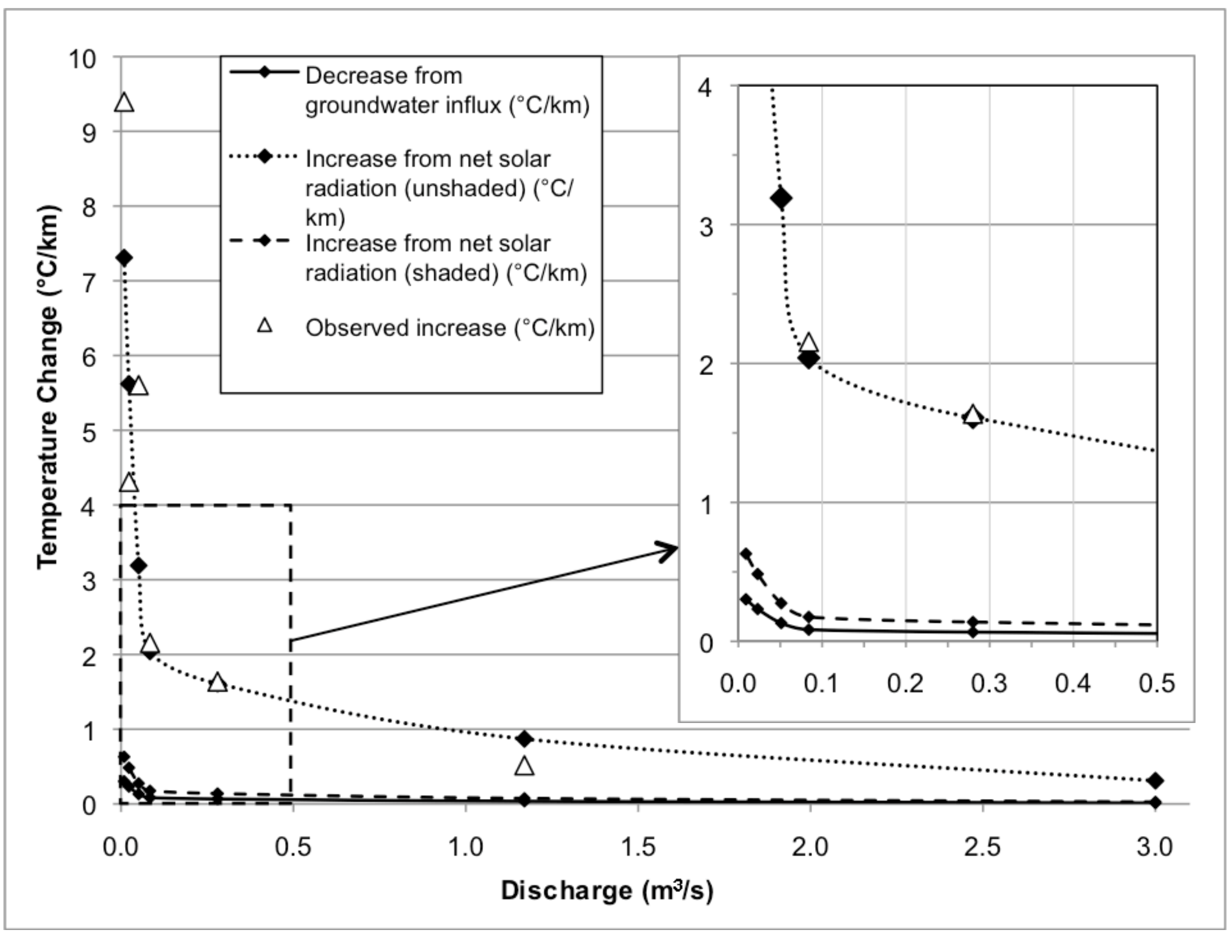


Figure 5.27: Estimated temperature change over a 1 km long stream reach due to solar radiation and groundwater inflows, and observed temperature increase between site 2 and 3 under various discharge conditions in Fortune Creek.

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 5.28). The average groundwater flux to the stream in the valley bottom reaches (site 3 to 8) in August was 3.3×10^{-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.

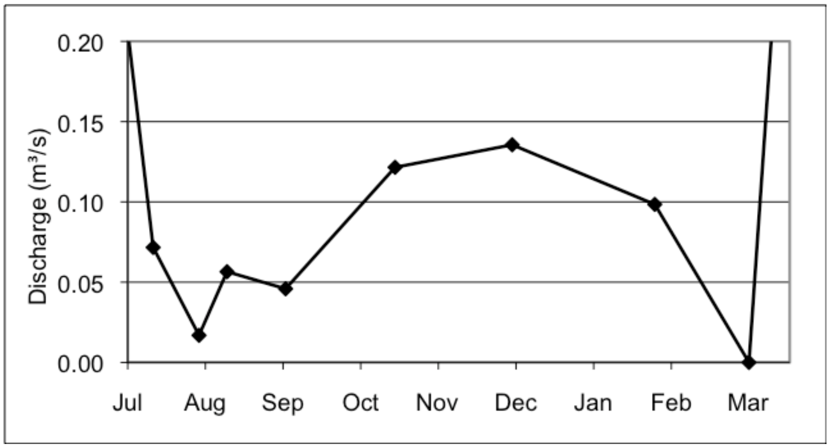


Figure 5.28: Discharge from Jul 2008 to Mar 2009 at the mouth of Fortune Creek.

6.0 Discussions and Management Recommendations

The results of the studies from 2007 to 2009 have presented the following as key issues surrounding the presence of salmonids in the creek:

- Stream temperatures within the valley floor exceed avoidance thresholds for salmonids.
- Flow rates within the creek are low, leading to stagnant conditions. Periods of concern are during the July/August summer period and the January to March winter period.
- Flow rates are highly variable both on a daily and seasonable basis, leading to fish stranding in the upper reaches of the creek.
- Stream water quality is affected by discharges to the creek and by bankside processes which lead to eutrophic conditions.

6.1 Stream Temperature and Discharge

Statistical modeling suggests that, to ensure minimum survival conditions for salmonid rearing in Fortune Creek, daily maximum temperatures should not exceed 22°C. However, temperatures below 18°C would be more suitable to ensure that salmonid production in Fortune Creek is not impaired.

Thermally suitable habitat in which juvenile salmonids can persist throughout the summer currently only exists between site 1 and 2 (Figure 5.18). 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 useable habitat. All sites between site 1 and 5 were occupied by juvenile salmonids at some points during the summer of 2008 and are therefore assumed to provide suitable physical habitat conditions (Figure 5.11).

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. 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.

Groundwater influx to Fortune Creek was diffuse and no substantial localized cold-water discharges were identified. Substrate in the valley bottom reaches is too fine to permit juvenile salmonids to seek thermal refuge by burying into sediment interstices. In the upper reaches where substrate is coarser and would provide space for juvenile salmonids, temperatures are cool so thermal refuge is not needed (site 1 and 2) or shallow bed temperatures are nearly identical to stream temperatures (site 3). 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 5.13) is too small to provide significant thermal relief for salmonids. Therefore, the possibility for groundwater discharges 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.

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 or to ensure minimum flows are maintained throughout the summer.

6.1.1 Shading

Temperatures at site 2 approach a maximum of 16.8°C during summer, which indicates that locations anywhere downstream of site 2 would likely experience periods of temperatures above the optimal temperatures for salmonid habitat. However, 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. The temperature rise from site 2 to 3 in shaded conditions would be well below the maximum allowable temperature increase (5.2°C/km). It is therefore likely that shading between site 2 and 3 could extend minimum thermally suitable habitat (<22°C) for juvenile salmonids from site 1 and 2 to site 4 and 5. Estimated heating rates apply to deeply shaded conditions and are expected to be higher under light shade.

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. It is therefore recommended to expand re-vegetation efforts to provide shading to the creek.

6.1.2 Discharge

Prior to the development of proper shading, temperatures can be moderated with appropriate discharge levels. Figure 5.27 indicates that heating rates due to solar radiation in unshaded reaches increased greatly when flows decreased below 0.1 m^3/s. Observed heating rates indicate that to ensure stream temperatures at site 3 remain below the 22°C threshold, stream discharge at site 2 would have to be at least 0.06 m^3/s. Thus, most likely range of thresholds would be between 0.06 to 0.1 m^3/s. This is near the 1960-1984 naturalized discharge of Fortune Creek for August (0.15 m^3/s) (Seebacher et al, 2007). In comparison, the measured discharges at site 1 in 2007 to 2009 range from 0.001 to 0.06 m^3/s during the summer period.

Stream temperatures at sites 6 to 8 further downstream 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 while the stream is generally deeper, which would likely result in a different relationship between flows and stream temperature in those reaches. Outside of the freshet period, the reaches between site 5 and 8 suffer from other habitat quality issues (eutrophication, low DO, lack of habitat complexity, predation by with northern pikeminnow and habitat competition with redside shiner) and it is unlikely that a reduction in stream temperatures alone would make a difference in salmonid use of these lower reaches.

The main benefit in increasing suitable levels of flow is at least threefold: 1) to increase the volume of water subjected to heating and cooling; 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; and 3) to increase total suitable habitat area. Target stream discharge could be achieved by additional water releases from headwater reservoirs, better timing of releases from the reservoirs, augmentation of flow from an additional source, or a reduction in water withdrawals.

6.1.2.1 Water Releases

Estimated naturalized flows in July and August (0.15 m^3/s) would be enough to meet the threshold range (0.06 to 0.1 m^3/s flows) recommended under current riparian conditions, assuming that the estimates derived from 1960-1984 data are not high under current climate conditions. This in-stream flow need would be reduced to a lower amount if all stream banks were fully re-vegetated. It is possible that the reservoirs intercept some of the flow that would otherwise naturally enter Fortune Creek. Allowing natural flow to regulate temperature would require a reduction in the removal of water from the creek or carefully matched augmentation with stored supply such that summer anthropogenic needs are fully supplied by stored water on an hourly basis.

Water releases from an additional upstream reservoir could ensure that instream flow needs are maintained throughout the summer. The summer period of most concern for temperature is approximately 30 days (Figure 5.18), which would require a storage volume of 260,000 m^3 to supply 0.1 m^3/s, which falls midway between the volumes stored within the two City of Armstrong Reservoirs (South Silver Star Lake: 185,000 to 246,000 m^3 and North Silver Star Lake: 487,000 m^3). Evenly controlled release of this water from mid-July to mid-August could maintain flows in the lower reaches.

In both the case of existing reservoirs or an additional reservoir, the release of water from the base of the reservoir would ensure that water entering the creek at the upper reaches would be at the lowest temperature possible.

6.1.2.2 Groundwater Pumping

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. However, more studies should be conducted to further confirm this if this option is implemented.

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. The lowest measured discharges at site 2 reached 0.001 m^3/s. At these low flow rates, the creek would be dominated by groundwater and 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 0.006 m^3/s, or approximately 90 gpm. At higher stream flow rates, the water leaving site 2 would be a mix of streamwater at 16.8°C and groundwater at 10°C, and calculations based on Figure 5.27 indicate that groundwater would need to increase slightly to 0.0065 m^3/s (100 gpm) before dropping down to a contribution of 0 m^3/s at 0.1 m^3/s. This pumping rate must be viewed as approximate, but irrigation wells in the vicinity can produce 200 gpm. The drawdown from pumping at 200 gpm for a few hours a day over a one-month period would not affect neighbouring wells.

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. These calculations are based upon temperature considerations only.

6.1.2.3 Reduction in Diversion

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 reaches up to 92% of estimated naturalized flows in August. If summer flows in the creek were not augmented by the headwater reservoirs, it is likely that all flows in Fortune Creek would be utilized during those periods. With augmentation, stagnant conditions were still observed in the lower reaches.

Determination of the flow conditions within Fortune Creek is complicated by the operational parameters of the City of Armstrong reservoir and intake systems, and the lack of flow metering above the intake. The releases of water from the upper reservoir are made by manual adjustment of a valving system on several week intervals. The reported releases of water (Table 5.1) in 2009 were 0.008 to 0.032 m^3/s. These occur during a period with average reported usage of 0.075 to 0.085 m^3/s (Ping et al, 2010, City of Armstrong data), indicating that the releases meet only part of the usage. However, Figure 5.23 indicates that the three release events (0.0158, 0.0079 and 0.0315 m^3/s) were followed by changes in measured downstream flow rates of 0, 0.03 and 0.09 m^3/s, respectively. The reservoir releases coincided with rainfall events and this makes it unclear whether the higher observed increases in flow rates were a result of underestimated reservoir releases or increases in rainfall-induced runoff. Comparison of stream flows, release rates and usage rates are complicated by the presence of the lower reservoirs near the intake which may be accumulating or drawing down during any given period.

In contrast, January to March naturalized flows are estimated to range from 0.075 m^3/s to 0.085 m^3/s (Seebacher et al, 2007), at a time of average monthly reported usage of 0.035 to 0.045 m^3/s. Flow rates measured by this study during January to March in 2007, 2008 and 2009 were generally less than 0.001 m^3/s. This seems to indicate some discrepancy between reported usage rates, estimated natural flows and observed flows. Thus, the operational parameters associated with the intake and augmentation systems are currently poorly defined. Active management of water flows for temperature management would be difficult.

Without considering the amount of water withdrawn it is highly likely that the current volumes of summer diversion and resulting reduction in streamflow leads to a considerable increase in stream temperatures. It is recommended that the City of Armstrong consider drawing a larger proportion of their water needs from their existing groundwater wells during the summer months instead of Fortune Creek.

6.2 Water Level Fluctuations

Water levels in the stream reaches on the alluvial fan fluctuate rapidly throughout the year because adjustments to the water released from the reservoirs are made every few weeks but manual adjustments to the stream intake diversions are made daily (Figure 3.2). Fluctuations of 5 to 10 cm were observed within the site 1 to 2 areas which represent the most thermally suitable habitat. Stranded salmonids were observed both up- and downstream of the culverts in all years of the study. It is recommended that water levels be managed automatically to avoid rapid daily fluctuation leading to stranding.

More extensive stranding events were noted in the fall of 2007 and 2008 associated with the seasonal closure of the reservoirs in late September. It is recommended that water release from the reservoirs be extended until fall rains bring up stream water levels (usually in October). It is recommended that the release valve is closed gradually to ensure that water levels in the creek decline slowly and the risk of stranding is minimized.

6.3 Metering of Water Usage

While total water diversion from Fortune Creek is determined by the water licenses, the exact timing of water usage is critical and currently unclear. It is recommended that metering of streamflow above the city water intake be automated to ensure accurate estimates of available water. In addition, water that enters the creek from the city holding tanks needs to be metered to get an accurate estimate of the amount and timing of diversions. Metering of flows released from the headwater reservoir is needed to provide a continuous record over time as opposed to the point estimates that are currently provided from the manual valve adjustments.

6.4 Anthropogenic Discharges

The measured dissolved oxygen data and extensive instream vegetation points towards eutrophic conditions in the lower half of the creek. The chemistry of discharges entering the creek should be investigated to get a better understanding of potential impacts to aquatic life. It is recommended that anthropogenic discharges into Fortune Creek be reduced. There are currently numerous field drains and other discharge pipes entering the creek, in addition to surface runoff from fields (organic and synthetic fertilizers) and feed lots (manure). Cattle guards should be maintained to prevent cattle from directly accessing the creek and trampling the banks.

6.5 Fish Habitat Restoration

Several restoration efforts completed in the past include flow deflectors near site 3 and instream boulder placement at site 5. Students participating in a UBC Okangan Course on Applied Fluvial Geomorphology, instructed by Dr. Leif Burge of Okanagan College, determined that the flow deflectors near site 3 improved fish habitat by increasing streambed heterogeneity compared to unrestored reaches. The restoration near site 3 involved larger structures designed to withstand high freshet runoff rates. They have led to macroscale meandering of the creek within the confines of the dykes, but have led to less smaller scale habitat development. The instream boulder placement by DFO near site 5 also led to greater streambed heterogeneity and provided fish cover. Placement of boulders also lead to faster stream flow velocities, which is beneficial in the lower reaches plagued by water stagnation.

Fish habitat restoration through boulder placement is relatively simple and it is recommended that this type of restoration be completed in other stream reaches between site 3 and 8 to improve physical habitat quality.

6.6 Future Research

It was assumed in this study that the salmonids utilizing the lower flooded reaches of Fortune Creek in the spring were ocean-type Chinook. It would be useful to compare their DNA samples to the stream-type Chinook persisting in the upper reaches throughout the year.

The numbers of fish occupying Fortune Creek was noted to be variable between 2008 and 2009, particularly during the freshet period when salmonids were abundant in the lower reaches only in 2008. It would also be useful to compare fish data collected in Fortune Creek to that from other nearby streams in the same year to understand if patterns observed in the creek were representative of regional dynamics or localized to Fortune Creek.

Northern pikeminnow, found throughout the lower reaches of Fortune Creek, are known to be predators of juvenile salmonids. Improvements to stream habitat quality brought about by extension of suitable thermal habitat downstream to site 5 and reductions in eutrophic conditions would bring salmonids further downstream, and potentially into greater contact with the pikeminnow observed in the lower reaches. Future studies could investigate predation by such non-salmonid species to understand whether it plays a major role in preventing utilization of streams by juvenile salmonids.

High spring flows in Fortune Creek occasionally lead to flooding of the lower reaches, which has led to the construction of dykes and the dredging of the creek in the past. There is continued pressure from local landowners to get permission to dredge the creek because the valley bottom reaches are aggrading and the stream bed is coming up. A study is needed to assess gravel transport in Fortune Creek, particularly the relationships of flow rates to the presence of spawning sized gravels near site 3.

This study establishes the relationship among water, temperature and fish populations in the creek, which paves good foundation for more watershed-scale research in the near future. With climate change impacts, pressure from population growth and increased forest disturbance (e.g., mountain pine beetle infestation, wildfire), water conflicts among various users are likely to become more intense. A watershed-wide study considering those variables will be critical to support the community to design management strategies for sustaining both fish and human needs.

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