Aquifer – Stream Connectivity at Various Scales: Application of Sediment – Water Interface Temperature and Vulnerability Assessments of Groundwater Dependent Streams

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Abstract

Streams with greater connectivity to an aquifer are potentially more sensitive to changes in groundwater levels and fluxes than streams with less connectivity. Aquifer-stream connectivity during the summer low flow period is of particular concern because this is a period of maximum relative groundwater contribution to stream flow volumes, which coincides with periods of peak water demands and critical aquatic habitat needs.

Field and statistical methods were used to characterize aquifer-stream connectivity and evaluate factors influencing the groundwater flux to streams at different scales during the summer low flow period. The research focused on the use of sediment-water interface temperature in combination with a range of field methods, including manual stream discharge measurements, seepage meters, and in-stream piezometers, to characterize aquifer-stream connectivity in Fishtrap and Bertrand Creeks in the Lower Fraser Valley of southwest British Columbia. A combination of field measurements aided in reducing measurement uncertainties and improved estimation of the groundwater flux. A simplified heat budget demonstrated that, despite their similar climate and geographical setting, the groundwater flux during the summer periods was higher in Fishtrap Creek than in Bertrand Creek, due to its more permeable geological substrate. Independent component analysis (ICA) combined with cross-correlation was a novel approach to temperature signal separation. ICA directly linked the extracted signals to factors in the heat budget that influence sediment-water interface temperatures within a stream reach. Surface heating from solar radiation was the dominant factor influencing the interface temperature in most years, but there is evidence that thermal exchanges took place at the water-sediment interface, and the correlation with groundwater levels indicated these heat exchanges were associated with groundwater influx. Overall, the combined approaches were able to attribute temporal and spatial variability in streamflow and sediment-water interface temperatures to relative contributions of groundwater to streams.

The understanding of aquifer-stream connectivity at different scales was applied in the development of a vulnerability framework for assessing stream vulnerability to changes in groundwater conditions. This framework can be used in support of decision making surrounding Sensitive Stream Designation in British Columbia and water allocation under the Water Sustainability Act.

Keywords: groundwater-surface water interactions, groundwater flux; low flows; sediment-water interface temperature; independent component analysis; stream vulnerability
Keep Moving Forward;
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Chapter 1.

Introduction

Groundwater and surface water are increasingly being recognized as interconnected components within the hydrologic continuum (Winter et al. 1998; Sophocleous 2002; Kalbus et al. 2006). Groundwater exchanges with streams are of particular importance, especially where the baseflow (i.e. the groundwater contribution to streamflow) constitutes a significant portion of the stream water budget at certain times of the year (e.g. during low flow periods). In many regions, flow in these “groundwater dependent streams” is sustained by groundwater contributions, and as a result can be sensitive to changes in groundwater flux during these periods (Hayashi and Rosenberry 2002; Allen et al. 2010). Low flow periods occur seasonally or during extended periods of dry weather (Sophocleous 2002). During summer, low flows are particularly critical because the warming associated with higher air temperatures and low streamflow may cause critical thresholds (flow and stream temperature) to be reached or exceeded for many aquatic species (Fleming et al. 2007; Burn et al. 2008; Cunjak et al. 2013; Moore et al. 2013). Groundwater influxes to streams have the ability to buffer flow levels and temperature effects by maintaining water levels (stream discharge) (Fleckenstein et al. 2004; Beatty et al. 2010) and providing thermal refugia (Hayashi and Rosenberry 2002; Kanno et al. 2014; Kurylyk et al. 2014). Streams with greater connectivity to the aquifer system\(^1\) (hereafter referred to as the aquifer) are potentially more sensitive to changes in groundwater levels and fluxes. Understanding the factors that influence the connectivity between aquifers and streams, and how that connectivity can be evaluated, is necessary for integrated management of water resources, and for assessing the potential impacts

\(^{1}\) An aquifer system includes both permeable units (aquifers) and less permeable units (aquitards) and thus serves as a useful paradigm for describing the geological system in relation to groundwater flow.
from climate variability, and changes in climate, land use / land cover, and groundwater pumping.

This dissertation focuses on the use of stream interface temperature in combination with various field measurements as a means to characterize aquifer – stream connectivity; identify the factors that control the relative contribution of groundwater to a stream; and evaluate the temporal and spatial variability of that connectivity. The research explores connectivity at different scales, and uses this understanding to frame stream vulnerability to groundwater-related stressors. The research focus is on the summer low flow period characteristic of a temperate climate setting.

1.1. Background

1.1.1. Aquifer – Stream Connectivity

Groundwater and streams interact over a range of spatial and temporal scales. Winter et al. (1998) identifies four categories of groundwater-stream interactions (Table 1.1 - A through D). The primary types of exchange occur when groundwater discharges to a stream (A - gaining stream), and when the aquifer is recharged by the stream (B - losing stream). In addition, groundwater may pass through a stream laterally (C - groundwater flow through), with groundwater inflow occurring on one side of the stream and outflow on the other side. Such lateral flow conditions may occur through meanders resulting in flow parallel to the main stream course (not shown). Losing streams may also be disconnected (Table 1.1 – D) from the water table by an unsaturated zone. For each case, groundwater exchange may differ along a single stream, gaining in some reaches and losing in others, and these exchanges often vary temporally (Silliman and Booth 1993; Winter et al. 1998; Sophocleous 2002; Constantz 2008)

The variability in exchanges between groundwater and a stream has been attributed to complexities at several scales. Studies have investigated the variability resulting from heterogeneity in shallow aquifer and streambed sediments (Angerman et al. 2012); bedform influences (Cardenas and Wilson 2006); and hydraulic gradients
influenced by river morphology and topography (Harvey and Bencala 1993; Cardenas 2008). These various studies demonstrate that groundwater-stream exchanges occur at different scales: from centimeters, to reach scale, to watersheds (Alexander and Caissie 2003; Conant 2004, Anderson 2005; Constantz 2008). However, due to the complexities in the groundwater-stream exchanges, it can be challenging to extend field-based data (generally collected at the reach scale or smaller) into broader understanding of the processes driving the groundwater-surface water interactions at a watershed scale.
Table 1.1. Types of groundwater-stream interactions (adapted from Winter et al. 1998).

<table>
<thead>
<tr>
<th>A - Gaining stream:</th>
</tr>
</thead>
<tbody>
<tr>
<td>Groundwater discharges into the stream. The water table in the vicinity of the stream is higher than the water level in the stream.</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>B - Losing stream:</th>
</tr>
</thead>
<tbody>
<tr>
<td>Stream water recharges the aquifer. The water table in the vicinity of the stream is lower than the water level in the stream.</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>C - Groundwater flow through:</th>
</tr>
</thead>
<tbody>
<tr>
<td>The groundwater flow direction is approx. perpendicular to the stream (or segment) and groundwater flows into one side of the stream, and outflows on the other side. The water table is higher on the upgradient side of the stream.</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>D - Disconnected stream:</th>
</tr>
</thead>
<tbody>
<tr>
<td>The water table is disconnected from the losing stream by an unsaturated zone. The water table is below the water level in the stream; however, localized mounding of the water table may occur in the vicinity of the stream.</td>
</tr>
</tbody>
</table>

1.1.2. Assessing connectivity

Methods used to evaluate the connectivity between the aquifer and the stream are varied, and range from field-based methods to integrative approaches. The field based methods include direct methods for measuring exchanges between groundwater and surface water; indirect methods; and combinations of both (Cey et al. 1998; Essaid et al. 2008; Rosenberry and LaBaugh 2008). Direct methods include seepage meters,
piezometers, and stream flow measurements (Boulton 1993; Baxter et al. 2003; Kalbus et al. 2006; Rosenberry 2008). Indirect methods include indicators such as water chemistry and mixing properties, or tracers such as heat (Stonestrom and Constantz 2003; Anderson 2005; Malcolm et al. 2005; Constantz 2008). The combination of methods can range from first-order assessments of the aquifer-stream system using mapping tools, to analytical solutions, to numerical modelling solutions.

Field measurements and integrative methods represent a spectrum of evaluation methods for groundwater-stream connectivity investigations. The integrative methods, such as numerical modelling, incorporate the field data, where available. Many studies have emphasized the benefits of using multiple methods to better quantify groundwater-stream interactions (Cey et al. 1998; Becker et al. 2004; Conant 2004; Kalbus et al. 2006; Brodie et al. 2009). Groundwater-stream exchanges often occur over a range of scales and in heterogeneous conditions; therefore, using a combination of methods is advantageous for understanding interactions in complex environments. A combination of methods reduces uncertainty, which may arise from limitations of the methods themselves, as well as errors and uncertainty in the available data. Use of multiple methods aids in linking information about the groundwater and the stream, and these linkages are necessary for assessment of connectivity. Lack of long term records for groundwater and streams can impede assessments of connectivity, and therefore the use of proxies, such as water temperature, can be incorporated within the spectrum of investigative methods.

Temperature is considered a robust and easily measured parameter for heat tracing and assessing groundwater interactions with streams (Anderson 2005; Caissie 2006; Hatch et al. 2006; Brewer 2013; Rau et al. 2014). At depths several metres below land surface, at which there is practically no annual fluctuation in ground temperature, groundwater temperatures remain relatively stable throughout the year, with values often similar to the mean annual air temperature (Alexander and Caissie 2003; Constantz 2008; Brewer 2013). Diurnal and seasonal groundwater temperature fluctuations are less pronounced than the fluctuations in stream water, which responds with similar patterns to air temperature, in response to solar radiation (Johnson and Jones 2000; Johnson 2003; Moore et al. 2005). As a result of the relatively stable temperature of
groundwater, groundwater influxes to streams can moderate the surface water temperature fluctuations, and the resulting attenuation of stream temperatures makes temperature variations suitable as a proxy for identifying relative magnitude of groundwater fluxes to streams and as a tracer for exchanges with groundwater (Silliman and Booth 1993; Becker et al. 2004; Conant 2004; Anderson 2005; Constantz 2008; Krause et al. 2012; Caissie et al. 2014). In addition, temperature varies both spatially and temporally, and this variability can inform on the timing and magnitude of groundwater-surface water exchanges. Previous studies have shown that streambed interface temperature, in particular, is variable due to focused (Krause et al. 2012; Briggs et al. 2013), or diffuse groundwater discharge (Lowry et al. 2007). Therefore, interface temperature can potentially be used to study connectivity over a wide range of groundwater flux conditions.

1.1.3. Aquifer – Stream System Types

A useful approach for understanding the broader scale connectivity between an aquifer and a stream was proposed by Allen et al. (2010) who analyzed groundwater level responses in relation to streamflows in various temperate mountainous settings (British Columbia, Canada). They demonstrated that the coupled response of aquifers and streams can be reasonably predicted by considering the hydroclimatology and the “aquifer-stream system” type. The seasonal timing of the aquifer-stream system response depends on the hydroclimatology of the region; rainfall-dominated (pluvial) or snowmelt-dominated (nival) (Figure 1.1). A hybrid (mixture of rain and snow) hydroclimatology is also possible (not shown). These aquifer-stream system types were defined based on classifying different aquifer types (Wei et al. 2009). The magnitude and timing of the recharge and discharge response of the aquifer-stream system was shown to depend not only on the storage and permeability characteristics of the aquifer, as might be anticipated, but also on the aquifer-stream system type, which is broadly classified as diffuse recharge-driven or stream-driven (Allen et al. 2010).

1. Diffuse Recharge-Driven – the aquifer is recharged solely by precipitation and groundwater discharges to streams throughout the year as baseflow. These systems are commonly associated with first order streams.
2. Stream-Driven - groundwater flow to and from streams is bi-directional, and varies seasonally depending on stream stage. These aquifer-stream systems are found in association with major streams/rivers.

![Figure 1.1](image1.png)  
**Figure 1.1.** Framework for classifying the responses in aquifer-stream systems, showing end members of the hydroclimatic regimes (from Allen et al. 2010 with permission).

In the diffuse recharge-driven system, precipitation across the watershed provides the recharge to the aquifer, and the groundwater discharges to the streams, forming the baseflow (Figure 1.2) (Allen et al. 2010). This process occurs in both hydroclimatic regimes, and is often continuous through the year, although the amount of discharge varies seasonally. During the annual summer low flow period, the streams are sustained primarily by groundwater discharge, with some minor contributions from storm events.

![Figure 1.2](image2.png)  
**Figure 1.2.** A diffuse recharge-driven system in which precipitation (blue arrows) falls across the watershed, recharging the aquifer, and ultimately discharging to the stream (from Allen et al. 2014 with permission).
In a streamflow-driven system, streamflow originates from different sources at different times during the year, and these sources vary depending on the hydroclimatology. During the spring freshet in snowmelt-dominated hydroclimatic regimes, the stream discharge is dominated by snowmelt runoff. The snowmelt contribution may originate from remote areas of the watershed (allogenic source); therefore, the streamflow during the freshet depends on conditions elsewhere. Because stream discharge is high during the freshet, the stream stage will also likely be high, and may exceed the elevation of the water table in a valley aquifer. Therefore, during the freshet, the stream recharges the aquifer adjacent to the stream (Figure 1.3). This aquifer recharge mechanism is relatively short lived (less than a month in studied aquifers; Scibek et al., 2007). Following the freshet, when stream discharge reduces, the flow direction within the aquifer reverses and groundwater recharges the stream. Local precipitation (either rainfall or snowmelt) may also recharge the aquifer throughout the year, as shown by the blue arrows in Figure 1.3. During the summer low flow period, the main contribution to the stream is local groundwater discharge. The same processes occur in rainfall-dominated hydroclimatic regimes, with the exception that snowmelt is not a driver of streamflow.

Figure 1.3. A streamflow driven system, with the snowmelt-derived streamflow recharging the aquifer adjacent to the stream (left). The blue arrows indicate precipitation across the aquifer contributing locally to the recharge. During the summer low flow period (right), groundwater discharge to the stream is driven by the local groundwater flow, which depends on diffuse recharge to the aquifer (modified from Allen et al. 2014 with permission).

In both aquifer-stream system types, streamflow is often sustained by the groundwater discharge during the summer low flow period. First order streams will be the most dependent on groundwater discharge from the local aquifers during the low flow season. Higher order streams, because they accumulate discharge from lower
order streams up-gradient and thus are often larger, generally are less dependent on discharge from local aquifers. Therefore, the sensitivity of a stream to stressors in the aquifer (e.g. pumping) will depend on the stream discharge during the low flow season and the proportion of the discharge that derives from local groundwater discharge.

1.1.4. Stressors and stream vulnerability

Groundwater pumping in the vicinity of the stream is one of the most important stressors on an aquifer-stream system. Pumping can impact the various types of groundwater-stream interactions shown in Table 1.1, with the exception of the disconnected stream, which is not directly impacted by pumping of groundwater\(^2\).

The potential effects of groundwater pumping on streamflow in an unconfined aquifer are presented in Table 1.2 (A through D). In a non-pumping situation (A), the shallow groundwater flows from recharge areas at higher elevation and discharges into the stream. The installation of a groundwater well, pumping at some rate (Q1) (B), draws groundwater from a capture zone around the well, and creates a cone of depression\(^3\). In this scenario, the pumping well intercepts a portion of the groundwater that would have otherwise discharged to the stream, leading to some lowering of the stream level. Under conditions of reduced recharge to the aquifer (C), perhaps due to climate change or land use change, the water table may be lower over a large area. Pumping under reduced recharge conditions would exacerbate the drawdown effect. At a higher pumping rate (Q2) (D), the water table is lowered further (compared to at Q1), and water may be drawn from the stream.

\(^2\) If the disconnection is temporary under natural conditions; prolonged pumping may lead to more permanent disconnection.

\(^3\) A cone of depression in an unconfined aquifer is a lowering of the water table in the vicinity of the well.
Table 1.2. Groundwater pumping effects on streamflow (adapted from Winter et al. 1998).

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>A - No pumping:</strong></td>
<td>Natural condition in a gaining stream with groundwater flowing from recharge areas at higher elevation and discharging to the gaining stream.</td>
</tr>
<tr>
<td><strong>B - Pumping at a rate Q1 from a well near the stream:</strong></td>
<td>The pumping well draws water from a zone around the well (capture zone) and may intercept a portion of the groundwater that would have discharged to the stream.</td>
</tr>
<tr>
<td><strong>C - Pumping at Q1 with lower recharge:</strong></td>
<td>The lower recharge leads to a lower water table, exacerbating the effects of drawdown.</td>
</tr>
<tr>
<td><strong>D - Pumping at a higher rate Q2 from a well near the stream:</strong></td>
<td>The pumping well draws more water from a larger capture zone. Pumping can lower the water table in the vicinity of the stream, and water may be drawn from the stream to meet the higher pumping demand.</td>
</tr>
</tbody>
</table>

The size of the cone of depression in an unconfined aquifer depends on the aquifer transmissivity (T) and specific yield (Sy). When T and Sy are low, the water table may lower substantially in a localized area around the well during pumping. In contrast, when T and Sy are high, for the same pumping rate, there is less drawdown near the well, and drawdown is distributed over a larger area. An aquifer with high T and Sy values is productive and also generally well connected to the stream; therefore, if a well
Pumping from a confined aquifer in the vicinity of a stream may or may not have an impact on stream levels. Pumping causes a zone of depressurization (a lowering of the hydraulic head) in the vicinity of the well, which causes groundwater to move towards the well in much the same way as in an unconfined aquifer (Alley et al. 1999; Barlow and Leake 2012). The size of the zone of depressurization depends on the T and the storativity (S)\(^4\) of the confined aquifer. While confined aquifers are oftentimes disconnected from surface water bodies, lowering of the hydraulic head in a confined aquifer can induce leakage from overlying unconfined aquifers, resulting in a lowering of the water table, and therefore, an indirect impact on stream level as discussed above for unconfined aquifers. Connection between a stream and a confined aquifer may occur if the stream is incised into the confining layer, or if the aquifer outcrops along the stream channel.

Other stressors include potential changes in the timing and amount of precipitation as a result of climate change. Such changes have the potential to lead to more extreme summer low flow events, and to extend the length of the summer low flow period in many streams (Déry et al. 2009). Climate variability and climate change also have the potential to impact aquifer recharge (e.g. Allen et al. 2004) as well as lead to increased water resource demands (e.g. Cohen et al. 2004), which in turn will lead to changes in groundwater conditions. Changes in land use/land cover (urbanization, timber harvesting, etc.) also impact recharge (Arnell 2002).

\(^4\) S is much smaller than Sy; therefore, the same amount of pumping will result in a much larger cone of depression in a confined aquifers compared to an unconfined aquifer.
Understanding the likely response of streams to changes in the groundwater conditions is important for management of water resources, and for evaluating the potential impacts from the various stressors described above, particularly pumping. However, generalized frameworks for evaluating the vulnerability of streams to changes in the aquifer are currently lacking. For jurisdictions like British Columbia, which for the first time will be licensing groundwater under the new Water Sustainability Act (Bill 18: WSA, 2014) consideration of the impacts to streams due to groundwater pumping is of critical importance. To develop such a framework, however, would require information on groundwater – stream interactions for different geographical regions. In most regions, data are often available for other applications, such as aquifer characteristics compiled for aquifer inventory, surface water data, and fish habitat metrics. These data available may not be ideal for the representation of groundwater – stream interactions; however; they represent a valuable resource that can be repurposed for evaluation of stream vulnerability.

1.2. Purpose and Objectives

The purpose of this thesis is to extend the understanding of aquifer – stream connectivity, specifically to characterize how the groundwater influx varies spatially over a range of scales (reach scale to watershed scale), and temporally (hourly to interannually). The research focuses on the summer low flow period because this is a period of particular ecological importance as summer low flows have the potential to negatively impact aquatic health and thereby fish ecology. The summer period also coincides with peak water demands from both the groundwater and streams, compounding low flow impacts to groundwater-dependent streams.
This research aims to test the hypothesis that the connectivity between groundwater and streams, and the factors that influence that connectivity can be explained at different scales using streambed interface temperature in combination with various field measurements. To test the main hypothesis, the following objectives of this research will be met:

- evaluating different field measurement methods for directly measuring groundwater influx into a stream;
- demonstrating that stream interface temperatures represent a mixture of temperature signals related to different heat exchanges across a reach and these signals can be separated into the individual components.
- demonstrating that differences in streambed interface temperature are related to differences in the groundwater flux;
- demonstrating that stream interface temperature (and hence groundwater flux) varies over a range of spatial and temporal scales.

A secondary objective of this research is to frame stream vulnerability to groundwater-related stressors using the understanding of groundwater-stream connectivity at different scales.

The study sites selected for the primary objective were Fishtrap and Bertrand Creeks, which drain the Abbotsford-Sumas aquifer in the Lower Fraser Valley of southwest British Columbia (BC). These particular streams were selected because, while the climate and topographic relief are the same, and the watersheds are of similar size, previous studies have shown the creeks have mixed interactions with groundwater making them suitable to assess connectivity between the groundwater and streams (Johanson 1988; Pearson 2004; Scibek 2005; Avery-Gomm et al. 2014). The streams are known to provide habitat for the Nooksack Dace (*Rhinichthys cataractae* spp.) and Salish Sucker (*Catostomus* spp.), both federally listed as endangered species in Canada (Pearson 2004).

For the secondary objective to evaluate stream vulnerability, nine sites across BC were selected, including Fishtrap and Bertrand Creeks. The nine sites were chosen to represent different aquifer-stream settings and to incorporate one stream designated as a “Sensitive Stream” under the BC *Fish Protection Act* (Bill 25: FPA, 1997).
1.3. **Scope of Work**

The following work was undertaken in order to complete the research objectives:

- Instrument and collect field data from the primary field sites to quantify relative exchanges between groundwater flux and stream flow.
- Compare summer groundwater contribution between the study streams using a heat budget approach and field-scale measurements.
- Develop empirical and conceptual models for the study sites that can be used to compare the relative contributions of groundwater to the streams, and identify factors influencing groundwater – surface water interactions at the study sites.
- Apply Independent Component Analysis (ICA) as a signal separation technique for analysis of spatial and temporal variability in temperatures at the streambed interface.
- Develop a vulnerability framework applicable to groundwater dependent streams in British Columbia, which can be used to designate groundwater-sensitive streams and evaluate consequences of changing groundwater conditions on summer low flows.
- Test the criteria established in the vulnerability framework using nine case study locations from British Columbia to represent a range of hydroclimatology and regional settings across the province.
- Further test the assessment structure in the framework using a numerical model available for the Fishtrap and Bertrand Creeks.
1.4. Thesis Overview

The thesis is comprised of seven chapters.

The first chapters provide a general introduction to the main concepts and the scope of the research. Chapter 1 provides a background on groundwater – stream interactions and defines the thesis objectives and the scope of work. Chapter 2 provides an overview of the main study sites, Fishtrap and Bertrand Creeks, as well as the field methodology. The background and methodology for the ICA analysis are detailed in Chapter 3.

Chapters 4 and 5 were prepared as a journal papers and submitted for publication. Chapter 6 was prepared as a guidance document and submitted to the BC Ministry of Environment. Each chapter is described in more detail below.

Chapter 4: Comparing the Groundwater Contribution in Two Groundwater-Fed Streams Using a Combination of Methods

This paper compares different methods for estimating the relative contribution of groundwater in Fishtrap and Bertrand Creeks using a combination of heat budget, regression, and field methods. This paper has been published in Canadian Journal of Water Resources, Special Issue on Groundwater-Surface Water Interactions, authored by M. A. Middleton, D. M. Allen, and P. H. Whitfield. While this paper was co-authored for publication, I completed the research and writing, with D. M. Allen, and P. H. Whitfield providing technical input, guidance, and editing.

Chapter 5: Independent Component Analysis of Local-Scale Temporal Variability in the Streambed Interface Temperature

This chapter examines the temporal variability in interface temperatures over four summer periods in Fishtrap Creek. The chapter presents the Independent Component Analysis (ICA) method to separate the observed temperature signals into input components temporally over the four summer periods. This paper was
published in Water Resources Research, authored by M. A. Middleton, P. H. Whitfield, and D. M. Allen. I completed the research and writing, with P. H. Whitfield and D. M. Allen providing technical input, guidance, and editing.

Chapter 6: Vulnerability Assessment for Groundwater Dependent Streams

This chapter comprises a document that presents multi-step assessment method for evaluating the vulnerability of a groundwater dependent stream to changes in the aquifer. The chapter was submitted as guidance document to the BC Ministry of Environment with the following difference: the background section is not presented within this chapter; rather it is presented in Chapter 1. This manuscript is co-authored by M. A. Middleton and D. M. Allen. I completed the research and writing, with D. M. Allen providing technical input, guidance, and editing.

Chapter 7 provides conclusions based on the collection of chapters 3 and 6 and manuscripts in Chapters 4, 5, presented in this thesis.

Additional work completed during the course of this research is presented in appendices.
Chapter 2.

Study Sites and Field Methodology

2.1. Study Sites

2.1.1. Rationale for Site Selection

The two study sites for the field component of this research are Fishtrap and Bertrand Creeks, located in the Abbotsford-Sumas aquifer in the east-central region of the Lower Fraser Valley (LFV) in southwest British Columbia, Canada (Figure 2.1). Fishtrap and Bertrand Creeks provide ideal settings for studying aquifer-stream connectivity for several reasons. These two transboundary creeks have been the subject of numerous studies due to concerns over agricultural contamination, water rights and development, and fisheries conservation efforts (e.g. Johansen 1998; Mitchell et al. 2000; Pearson 2007; Pruneda et al. 2010; Scibek and Allen 2005; Starzyk 2012; Avery-Gomm et al. 2014;). The hydrogeology of the aquifer has been extensively studied and a groundwater model has been developed for the aquifer (Scibek and Allen 2005). This model has been used to assess climate change impacts on groundwater recharge (Scibek and Allen 2005), simulate nitrate transport (Chesnaux et al. 2011), and inform further modelling efforts on groundwater-surface water interactions (Pruneda 2007; Starzyk 2012). Separate work on this area has also been completed on watershed basin management (Johanson 1988), surface water processes (Cox et al. 2005; Starzyk 2012), and stream ecosystem restoration strategies (COSEWIC 2002; Patton 2003; Pearson 2004; Pearson 2007; Avery-Gomm et al. 2014). In addition, while there are similarities between the watersheds in topographic relief and climate, there are local variations in stream morphology, bed material and riparian cover that likely influence aquifer-stream connectivity.
The streams are known to provide spawning habitat for salmonid species, as well as habitat for the Nooksack Dace (*Rhinichthys cataractae* spp.) and the Salish Sucker (*Catostomus* spp.), both of which are listed as endangered species in Canada (Pearson 2004). These streams provide some of the only habitat in Canada for these species, both of which have limited ranges, and recovery strategies have been developed for both streams (Pearson 2004). Low flows, attributed in part to surface and groundwater withdrawals, have been identified as a threat to fish populations in these streams (COSEWIC 2002; Avery-Gomm et al. 2014). Extending the understanding of groundwater-surface water interactions has been identified as a data gap in the management strategies for habitat and population conservation, coupled with the need to balance demands for water resources in a rapidly developing region.

Finally, the area comprises portions of the agricultural land reserve (ALR), which restricts urban development and rezoning of large tracts of land into urban development. The entire area is under pressure from development and was described in the 1990s as having population and industrial growth that is among the fastest growing in North America (Boyle et al. 1997). These development pressures are continuing and are placing strain on the water resources in the area, throughout both the Canada and the USA portions of the aquifer.
2.1.2. The Abbotsford-Sumas Aquifer

The Abbotsford-Sumas aquifer is approximately 160 km$^2$ and is a transboundary aquifer, spanning the Canada-USA border; approximately half of which is located BC, while the balance is within northwest Washington State (WA) (Scibek and Allen 2005) (Figure 2.1). The aquifer is comprised of Quaternary aged glacial deposits, which overlie a thick Tertiary aged sedimentary sequence of low permeability bedrock that overlies granitic rock basement (Scibek and Allen 2005). The most extensive units in the area being studied are the Fort Langley Formation and the Sumas Drift (Figure 2.2) (Armstrong 1981, 1984; Johansen 1988, Clague 1994). Lesser amounts of Salish Sediments also occur discontinuously in the area. The Fort Langley Formation comprises pebbly silt in a clay glaciomarine layer. This unit has been interpreted as continuous at depth, forming a confining layer. Sumas Drift deposits include glaciofluvial sand and gravel deposits, with intermittent lenses of till. This unit becomes coarser in the Abbotsford area, encompassing what is called the Abbotsford outwash. The sand and
gravel deposits of the Sumas Drift, including the Abbotsford outwash, are the units that comprise the Abbotsford-Sumas aquifer, and in the area of Abbotsford, overlie the Fort Langley Formation (Johanson 1988, Scibek and Allen 2005).

<table>
<thead>
<tr>
<th>Age (ka)</th>
<th>Period</th>
<th>Geologic Climate Unit</th>
<th>Stratigraphic Unit</th>
</tr>
</thead>
<tbody>
<tr>
<td>10</td>
<td>Holocene</td>
<td>Postglacial</td>
<td>Salish Sediments</td>
</tr>
<tr>
<td>11</td>
<td>Late Wisconsin</td>
<td>Fraser Glaciation</td>
<td>Sumas Drift</td>
</tr>
<tr>
<td>13</td>
<td></td>
<td>Sumas Stade</td>
<td>Ft. Langley Formation</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Everson Interstade</td>
<td></td>
</tr>
</tbody>
</table>

**Figure 2.2.** Late Quaternary events and stratigraphic deposits in the Fishtrap and Bertrand Creek watersheds. (modified from Clague 1994 and Armstrong 1984).

The aquifer is predominantly unconfined, but becomes confined in a small area to the southeast (Figure 2.1). Where unconfined, it is generally 15-25 m thick, with a range in thickness from 5 to 70 m (Scibek and Allen 2005). Depth to groundwater generally varies between 2 and 10 m below ground surface (Starzyk 2012). Mean hydraulic conductivity has been estimated as 80 m/day, with a range from 2 to 2380 m/day. The median lateral velocity of the groundwater has been estimated at 0.75 m/day (Mitchell et al. 2003). Groundwater flow is generally to the southwest, but is strongly influenced by local topographic variations, as well as by stream interactions (Johanson 1988, Scibek and Allen 2005). The aquifer is highly productive (Scibek and Allen 2005). It supplies irrigation water to the agricultural areas, and municipal water to approximately 100,000 people in Canada and 10,000 people in the USA (Mitchell et al. 2000).

The climate of the Abbotsford-Sumas aquifer area is characterized by maritime influences, with moderate temperatures and high annual precipitation rates. The temperatures throughout the year are moderated by the close proximity to the Pacific Ocean, and during November through May, the prevailing westerly winds transport precipitation laden clouds inland. The annual average precipitation is 1500 mm/yr, with less than 100 mm as snow (Figure 2.3). Approximately 70% of precipitation falls in the
period between October and May, and only six percent of the precipitation falls in July and August (Wernick et al. 1998, Zebarth et al. 1998, Berka et al. 2001, Environment Canada 2007).

Soils over the aquifer are generally thin (approximately <0.7 m thick) with permeability that exceeds precipitation rates (Mitchell et al. 2003). Thus, diffuse recharge to the aquifer by precipitation is rapid (Scibek and Allen 2005). Water levels in the aquifer fluctuate seasonally (Figure 2.2) as well as annually as a result of variations in precipitation. The average annual variation in the aquifer is 2 m, with a maximum variation of 3 m (Scibek and Allen 2005). The minimum groundwater levels occur during the summer period, and lag precipitation and stream flow by approximate one to three months (Figure 2.3). The minimum stream flow for the study streams occurs during the summer period, defined in this study as the time period of July through September, and pattern of stream discharge follows the precipitation patterns closely.

Figure 2.3. Mean monthly precipitation from the Abbotsford International Airport (Station 1100030), with the mean monthly discharge for Fishtrap and Bertrand Creeks, and groundwater levels from Environment Canada (ABB01) and BC Ministry of Environment (#299) observation wells (see Figure 2.7 for locations).
2.1.3. **Fishtrap and Bertrand Creek Watersheds**

The Fishtrap and Bertrand watersheds are of similar size, have similar climate and topographic relief, and the flow regimes are similarly rainfall driven. The area of Fishtrap Creek watershed is approximately 37 km$^2$ and Bertrand Creek is approximately 51 km$^2$. Situated between the two streams is the Pepin Brook watershed, which is a tributary to Fishtrap Creek, south of the International Border (Figure 2.4). Previous studies have indicated that groundwater contributions are important for maintaining summer streamflows, and that groundwater contributions may be variable between the streams (Johanson 1988; Pearson 2004; Berg and Allen 2007; Starzyk 2012). The two streams were selected for this study, in part because while there are many similarities between the watersheds, the dissimilarities in groundwater contribution are not well understood and indicate differences within the watersheds that are influencing aquifer-stream connectivity.

![Legend](image)

![Figure 2.4. Generalized surficial geology of the Canadian portion of Fishtrap and Bertrand Creek watersheds (outlined in black). Pepin Brook is situated between the two study watersheds. South of the International border, Pepin Brook is a tributary to Fishtrap Creek.](image)

The surficial geology in the Fishtrap Creek watershed (Figure 2.4) is dominated by coarse grained Sumas Drift, with some fine grained Ft. Langley Formation sediments
in the upper reaches and Salish Sediments occurring discontinuously near the stream channel (Johanson 1988; Scibek and Allen 2005). The geological substrate in the Bertrand Creek watershed has greater amounts of Ft. Langley Formation throughout (Figure 2.4) and, in contrast to Fishtrap Creek, has lesser amounts of the coarser Sumas Drift unit near the stream channel. In the lower reaches of Bertrand Creek, the surficial geology is heterogeneous, with occurrences of both coarse and fine grained material (Berg 2006).

Land use data (BC Ministry of Agriculture and Lands 2006) are available for Fishtrap Creek and the upper reaches of Bertrand Creek. Fishtrap Creek watershed is dominated by agricultural land-use (shown in green in Figure 2.5), with some industrial use around the Abbotsford International Airport at the center of the watershed and in the upper reaches (brown areas in Figure 2.5). The northwest section of the watershed is listed as unclassified (grey areas in Figure 2.5) but is primarily residential and commercial. In the lower reaches of Fishtrap, larger tracts of land are identified as agricultural land use. At the Canada-USA border, (site F1 in Figure 2.5) the stream reach is a sequence of pools and runs. The bed material is dominantly sand and fines, with trace to minor amounts of gravel, and abundant organic material (leaf litter and small woody debris). The land use surrounding the F1 site is agricultural, dominated by berry crops, and the riparian cover is limited to seasonal invasive Canary reed grasses present within the channel as in-stream vegetation, which becomes denser throughout the summer season.

Limited land use data are available for Bertrand Creek; however, the creek flows through more mixed development, with a combination of industrial, agricultural, and residential land use (blue areas in Figure 2.5) throughout the watershed. In the lower reaches of Bertrand, there are smaller rural residential land parcels that are classified as agricultural. At the B1 site (Figure 2.5), Bertrand is riffle and run prevalent, and the bed material is dominated by well sorted fine gravel and coarse sand. The thickness of the bed material ranges between 0-30 cm, and overlies clayey silt. At this location, the land use is rural residential, with some partial clearing along the south bank for road right-of-way. Overall, the riparian cover is partial from shrubs and deciduous tree canopy along the channel margins overhanging the channel.
To supplement the land use map for areas where there are no data, or where parcels are unclassified, the paved areas (Scibek 2005) were also mapped (Figure 2.6). In both watersheds, paved areas correspond to the industrial areas, and to the residential and commercial areas in the northwest section of the Fishtrap watershed.

Figure 2.5. Generalized land use map for the study watersheds. Green indicates agricultural land use, brown indicates industrial, blue is residential, and grey is unclassified. The dark grey area in the western portion of the Bertrand watershed indicates areas for which data are not available.
Figure 2.6. The paved areas in Fishtrap and Bertrand Creek watersheds shown in black. Paved areas indicate areas of development, consistent with land use classes for industrial, commercial, or residential.
2.2. Data Sources

Locations for regional data sources are shown in Figure 2.7. Climate data were obtained from the Environment Canada Abbotsford International Airport (Station 1100030) and from the US Geological Survey (USGS) hydrometric station (Station 12212390) at Bertrand Creek. Hydrometric stations are located at the Canada-USA border at both Fishtrap and Bertrand Creeks. Stream discharge data were obtained from the USGS hydrometric station at Bertrand Creek, and from the Water Survey of Canada (WSC) hydrometric station (Station 08MH153) at Fishtrap Creek. Hourly groundwater level data were obtained from BC Ministry of Environment’s observation wells #2, 299, and 301. Hourly groundwater level data and groundwater temperature were available from Environment Canada Observation Wells FT1, FT5, and ABB01.

Figure 2.7. The location of the climate and hydrometric and hydrogeological data sources within the study region and locations of field study sites in Fishtrap and Bertrand Creeks.

2.3. Field Methodology

Field measurements were conducted at two scales: 1) regional scale along the length of the Canadian portion of Fishtrap Creek (Figure 2.7), and 2) local reach-scale at both Fishtrap and Bertrand Creeks at the international border (Figure 2.8). The regional measurements were made at thirteen locations (F1 through F13) throughout the Fishtrap
Creek watershed (Figure 2.7). The local scale measurements were made within single reaches of Fishtrap and Bertrand Creeks (F1 and B1 – Figure 2.7).

Field observations and measurements were completed over five summer periods at Fishtrap and Bertrand Creek, beginning in July 2008, until October 2012. The sediment-water interface temperatures (described below) were recorded over five summer periods at the local scale sites, and at the regional sites, over two summer periods (2010-2011). The seepage meter and in-stream piezometer readings were taken over three summer periods at the local sites (Figure 2.8) (2009-2011), and over two summers at sites F2 and F3 (2009-2010). The stream discharge recordings were collected at site F1 over three summer periods (2009-2011), and two summer periods at sites B1, F2, and F3. At the remainder of the regional sites, the stream discharge was measured in 2009. The water chemistry samples were collected at sites F1, B1, F2, and F3 in the 2010 summer period, in April 2011 following a rainfall event, and from F1, F2, F3, and FT1 in July 2011. Summary data are tabulated in Appendix A.

2.3.1. Streambed Temperatures

Sediment-Water Interface Temperature Transects

In each stream at the local scale, a transect of TidbiT® v2 Temp loggers (UTBI-001) was installed to record temperature hourly at the sediment-water interface. The loggers locations were selected to be distributed across representative sections of channel, including ranges of bed material types, bed forms, and riparian cover (where present). Loggers were first installed in July 2008, and data were recorded until September 2012.

The loggers were installed at the local scale study sites F1 and B1 (Figure 2.8). At site F1, 15 dataloggers were initially deployed, and 19 at B1 (Figure 2.7). The loggers were attached to rebar to prevent movement (Figure 2.9), and positioned directly, unshielded, on the streambed (Figure 2.10) to monitor temperatures at the interface between the streambed and the water column. At both sites, loggers were distributed in a transect perpendicular to flow at the downstream end of the site, and the remaining loggers were distributed at approximately even distances longitudinally along the
channel bed (Figure 2.8). The distribution of loggers was designed to capture differences in sediment-water interface temperature along the flow direction, across the channel width, and over a representative distribution of channel characteristics (bed material, flow patterns, and riparian cover). At the Fishtrap Creek site, one datalogger (#12 – Figure 2.8) was lost in October 2008, another was removed in July 2011 due to a low battery (#14), and one had a battery failure after October 2011 (#9). In Bertrand Creek, securing the loggers to withstand the high flows in the winter was difficult and consequently, a large number of loggers were lost over the 2008/2009 winter period. Lost dataloggers were replaced for the 2009 summer period, but were removed from the stream prior to the higher flows in the winter. The number of loggers was reduced from 19 in 2008 and 2009 to 3 in 2010, and further reduced to 2 in 2011 due to failure of one unit. In 2012, the remaining loggers were lost due to shifting bedload, and water temperature data from the USGS hydrometric station were used. A table of the dataloggers units lost or decommissioned over the study period is presented in Appendix B. Summary descriptions of each datalogger location are also presented in Appendix A.

The Tidbit temperature loggers have an accuracy of ±0.2°C and a resolution of 0.02°C. The manufacturing specifications indicate an annual drift of up to 0.1°C/year. Therefore, calibration of the dataloggers was verified in a temperature bath prior to deployment and at the end of the sampling period. As detailed in Appendix B, a linear drift correction, specific to each datalogger, was applied to the data. The mean drift over the five years was 0.04°C/year, less than the maximum 0.5°C suggested by the specifications.
Figure 2.8. Distribution of temperature loggers, piezometers, seepage meters, and discharge measurement transects for local scale measurements at a) Bertrand Creek (B1) and b) Fishtrap Creek (F1).
Figure 2.9. TidbiT® temperature logger (orange disk) affixed to rebar prior to installation in Fishtrap Creek.

Figure 2.10. TidbiT® temperature logger installed at the sediment-water interface (channel bed) in Bertrand Creek.

At the Fishtrap Creek regional sites (Figure 2.7), two temperature dataloggers were installed at each location from F2 through F13. The dataloggers were installed using the same method as at the local scale sites, and were positioned longitudinally in the channel in similar conditions (bed material, position within the channel, channel
morphology, and riparian cover). Two dataloggers were used at each site to reduce uncertainty in the recorded values and to provide better coverage in case of equipment damage or loss.

**Streambed Temperature Profiles**

Temperature loggers were installed within the streambed sediments at four locations F1, F2, F3, and B1 in an effort to profile the temperature within the streambed sediments. The loggers were attached to a single rebar at the bed surface and at depths of approximately 0.1 m, and 0.2 m in the bed. To install the loggers within the subsurface, the bed sediments were excavated with a posthole auger, and the loggers placed at the respective depths. At each location, two installations were made, one upstream and one downstream. The loggers remained in the sediments from August 2010 to September 2011. Unfortunately, the TidbiT® v2 Temp loggers suffered a high optical reading failure rate likely due to abrasion in the sediments, and most data could not be retrieved. The depth profile at site F3 between August 21, 2010 and September 9, 2011 was the only profile with retrievable data. The summary data for the temperature depth profile at F3, and plots of the upstream and downstream profiles are provided in Appendix A.

2.3.2. **Flux Measurements**

Groundwater flux into each stream was measured at the local scale sites using three methods; seepage meter measurements, in-stream piezometers, and stream discharge measurements (see Figure 2.8 for locations). At the regional sites on Fishtrap Creek, flux measurements were also made at site F2 in the lower reaches of Fishtrap and at site F3 near the center of the watershed (see Figure 2.7 for locations). At the remainder of the regional sites, only stream discharge measurements were made along with the sediment-water interface temperature measurements.

**Seepage**

Seepage meters have been used effectively to quantify fluid exchange across the sediment-water interface (Alexander and Caissie 2003; Kalbus et al. 2006; Essaid et al. 2008; Rosenberry 2008). The seepage meter apparatus is a bottomless cylinder inserted
into the channel sediments and connected to a lightweight bag that either collects water seeping into the stream (gaining) or draining out of the stream (losing) (Kalbus et al. 2006, Rosenberry 2008). As groundwater fluxes occur across the bed interface, the volume and rate of the water entering or draining the bag are used to calculate the flux rate. For use in flowing water, modifications have been suggested, such as containing the collection bag in a stilling well, and using a low profile design for shallow flowing water (Rosenberry 2008). To compensate for resistance to flow within the apparatus and frictional flow losses, which result in underestimation of the flux rate, a resistance correction factor of 1.05 is normally applied to seepage flux measurements (Rosenberry and LaBaugh 2008). Seepage meters have limitations because they rely on the assumptions that all the groundwater flux is vertical into the channel and that the seepage flux collected entirely represents the groundwater flux and not a mixture with circulating hyporheic flow. Calculated seepage meter fluxes have limitations in space and time, because measurements are localized and represent point measurements within the channel and do not capture the variability in flux that may exist across the stream channel.

Two seepage meter designs were used in this study due to the different stream morphologies and flow depths. In Fishtrap Creek, a higher profile seepage meter was used at sites F1 and F2, which have deep water and fine bed sediments (Figure 2.11), while a low profile design was constructed for use in the shallower flow and coarser grained bed material in Bertrand Creek at site B1 (Figure 2.12). The same stilling well (container) was used for both streams, and was positioned on its side in the shallow water (Figure 2.13), or upright in the deeper water (Figure 2.11). Seepage meter construction and operation are detailed in Appendix C. The seepage meter was allowed to equilibrate following installation, often for a period of 24 hours, and measurements were recorded approximately every 30 minutes, between and 8 and 24 times in a day. The seepage meter data are summarized in Appendix A.
Figure 2.11. The seepage meter designed for use in deeper water, installed at site F1 in Fishtrap Creek. The seepage meter is inserted approximately 8 cm into the bed sediments, and vented to the atmosphere. The seepage collection bag is positioned within the upright stilling well on the right side of the photo.

Figure 2.12. The seepage meter designed for use in shallow water, installed at site B1 in Bertrand Creek. The seepage meter is set approximately 4 cm into the bed material.
Figure 2.13. The seepage collection bag in the stilling well (container) in shallow water. The stilling well is positioned in a low flow section of the stream, parallel to the flow and perpendicular to the seepage meter. The seepage meter is installed adjacent to the nested in-stream piezometers, seen on the left side of the photo.
**In-Stream Piezometers**

Shallow in-stream piezometers were installed in Fishtrap and Bertrand Creeks as nested pairs to monitor water levels to estimate the vertical hydraulic gradient. The piezometers were constructed of stainless steel with solid tips, and holes drilled at even intervals around the bottom 0.10 m (Figure 2.14). To prevent plugging of the holes by fine sediments while still allowing water to flow freely into the piezometers, heavy duty scour pads were inserted inside the piezometers as filter packs. The piezometers were installed by pounding them into the substrate to depths of approximately 1.0 m and 0.5 m, at locations adjacent to the stream bank (Figures 2.14 and 2.15) at sites F1, F2, F3 and B1 (Figure 2.7). The position of the nested piezometers adjacent to the stream bank was intended to evaluate the vertical hydraulic gradient at the stream, as well as limit potential damage from high flow events and vandalism.

Measurements of the water levels in the nested piezometers were made using a water level tape with millimeter increments (Figure 2.15), recorded approximately at 30 minute intervals, and repeated between 6 to 25 times per day. The vertical hydraulic gradient was calculated using the vertical depth separation of the center of the screened interval for reference. The flux was calculated using the estimated vertical hydraulic conductivity (Kz) of the sediments (based on Scibek and Allen 2005). Data for the flux are summarized in Appendix A.

Uncertainty in the water levels measured in the piezometers is ±0.035 m, and is the result of uncertainties in the depth of installation as well as the measurement methods. Similar to the seepage meters, the use of piezometer assumes the flux into the channel is vertical. The piezometers also have the limitation that they represent the localized flux at the point of the nested pair. In addition, the calculated flux is based on the hydraulic conductivity of the bed material, and assumes no change in conductivity with depth.
Figure 2.14. Installation of an in-stream piezometer in Bertrand Creek, adjacent to the stream bank. The seepage meter apparatus is visible in the foreground.
Figure 2.15. Measuring the water level in the nested in-stream piezometers in Bertrand Creek.

Discharge

Stream discharge transects, also called seepage runs, use measured differences in discharge over a length of stream to estimate groundwater flux as the difference in the discharge (Harvey and Wagner 2000). Changes in the stream discharge along the length of the channel are a result of groundwater flux if there are no tributaries or outflows between transects and the measurements are completed over a short time window such as within the same hour. Previous work measuring stream discharge in the Bertrand and Fishtrap Creek watersheds by Berg (2006) showed that diurnal fluctuations in stream discharge can increase the uncertainty in measurements over longer periods throughout a day. As well, the previous work also found that uncertainty in the measurements can potentially increase under low flow conditions. In an attempt to reduce the uncertainty in the stream discharge readings, the measurements were repeated a minimum two times at each transect to obtain a mean value for the discharge (Berg and Allen 2007).
Stream discharge was measured at the local scale sites and the regional sites (Figure 2.7). The discharge measurements were made using a Scientific Instruments 1250 Mini “pygmy” current meter which is designed for use in small, shallow streams, and for low flow conditions. The method used to calculate the flow was the sixth-tenths depth method, and the velocity at that point assumed to approximate the average velocity in that section of the channel (Dingman 2002). The streams were sampled using the velocity-area method, in which the width, depth, and velocity are measured and used with the velocity to calculate the discharge. The width of the channel was measured for each cross-section, and divided into ten evenly spaced sections. The flow measurement was taken at the mid-point of each of these vertical sections to represent an approximate average of the depth and average velocity in the panel (Figure 2.16).

The optimal operating conditions for the meter range between approximately 0.08 to 0.9 m/s, and require a minimum flow depth of 7 cm (Rickly Hydrological 2016). Overall uncertainty in the stream discharge was calculated from the combined uncertainty in the velocity measurements and the area calculation. The accuracy of the meter is reported to range from 3.5% up to 10% at the extent at the optimal range, and in this study, 6.5% was used (Nelson et al. 2010). Measurement uncertainty from the tape readings were 0.05 m, and were added to the velocity uncertainty.

Figure 2.16. Stream discharge measurement using the velocity-area method. The pygmy meter is mounted to the wading rod, and the flow velocity is measured at six-tenths the depth of the stream.
2.3.3. Stream Water Chemistry

Groundwater exchanges with streams can result in changes in the surface water quality as a result of mixing of water with different physical and chemical properties. The mixing can result in variations in pH, dissolved oxygen (DO), electrical conductivity (EC), and dissolved constituents. Differences in the stream water chemistry were anticipated to be indicators of variations in relative contributions of groundwater flux. Field parameters (water temperature, pH, DO and EC) were measured concurrently with the collection of the water samples at locations F1, F2, F3 and B1 (Figure 2.7) both from the stream water and the water collected from the seepage meter (pairs of samples), as well as groundwater from observation wells FT5-25 and FT1-24 (Figure 2.7).

The pairs of water samples from the stream and the seepage collection bag were collected at approximately 30 minute intervals, between 4 and 16 times per day during measurements of seepage at each site. All samples were field filtered to remove suspended sediments using a 0.47 µm Whatman Cellulose nitrate Membrane Filter and a MityvacII hand pump. Samples were collected in plastic bottles, and for each pair, one sample was unpreserved and the other was acidified with 5 mL of 1% nitric acid to a pH <2. Water collected from the groundwater well FT1-24 was collected using 16 mm (5/8") high density polyethylene (HDPE) Watera® tubing and a foot valve. The water was hand pumped from the well from a depth of approximately 7 m. The volume purged was calculated as the minimum of ten times the saturated thickness (approximately 30 L) and the pH and electrical conductivity values were stable. Samples were passed through a 0.45 µm in-line filter and preserved similarly to the paired samples.

Samples were analyzed for total alkalinity as HCO$_3^-$, nitrate (NO$_3^-$), sulfate (SO$_4^{2-}$), chloride, sodium, fluoride, and bromide as well as minor dissolved constituents. The sample from well FT5-25 was provided by Environment Canada personnel during regular sampling of the well and was collected from a depth of approximately 7.5 m. The water chemistry data are tabulated in Appendix A.

Total alkalinity as bicarbonate was analyzed by colorimetric titration using a modified Hanna alkalinity test kit (HI 4811). The titrations were completed with 5 mL of filtered unpreserved sample, 2 drops of bromcresol green indicator, and 0.02N HCl.
reagent, which was added drop-wise to change the blue solution to yellow. For each sample, two titrations were completed, and the average used to calculate the total alkalinity. Total alkalinity as bicarbonate was calculated from volume of titrant added as given in Equation 2.1.

\[
\text{Total Alkalinity} \left( \frac{\text{mg}}{\text{L}} \text{HCO}_3^- \right) = 0.02 N \text{HCl(mL)} \times 244
\]  

Dissolved constituents were measured at Simon Fraser University. Anions (F-, Cl-, Br-, NO3-, PO4-3, SO4-2) were measured with the Dionex Ion Chromatography System (ICS), and major (Na, K, Ca, Mg) and minor (Al, As, B, Ba, Fe, Li, Mn, Mo, Si, Sr, Zn) elements are measured by the Jobin-Yvon Ultima 2 Inductively Coupled Plasma Atomic Emission Spectrometer (ICP-AES).

All meters were field calibrated daily. The pH calibration was a two-point calibration using pH 4 and 7 solutions. EC was calibrated to a 1413 μS solution. Temperature and pH were recorded with a Hanna Instruments portable pH meter (HI 9023). EC was measured with a Hanna Instruments portable conductivity meter (HI 9033). DO was measured with the Oakton portable DO meter (WD-35640-06). To record the stream values, the probes were positioned in the stream, within the main flow path where possible to measure mixed flowing stream water. Recordings of the seepage water parameters were done after the paired samples were collected for filtration, and the observations were recorded in a 1L plastic graduated cylinder.

2.3.4. Groundwater Monitoring

A YSI (6290 model) multi-parameter datalogger was deployed in Environment Canada’s monitoring well FT1-24 (Figure 2.17), adjacent to Fishtrap Creek at the International Border (Figure 2.7). Well FT1-24 is one of a set of nested wells, and is the intermediate depth of ~7m (24 feet) below ground surface. The YSI logger was installed at a depth of 5.9 m below the top of casing. The datalogger collected hourly data of groundwater temperature, depth, DO, specific conductance, and nitrate (NO₃⁻N). Due to hardware and software malfunctions throughout the period of deployment, the data
records have numerous gaps and hourly measurements were recorded over inconsistent periods of time, ranging from less than one day to two months.

Figure 2.17. Environment Canada observation well FT1, comprising a nested installation of three wells. The groundwater data for this study was collected from FT1-24, which has a depth of 24' below ground surface.

2.3.5. Riparian Mapping

Delineating riparian zones (Figure 2.18) was completed to assist in interpretation of the regional sediment-water interface temperature monitoring. The riparian zone comprises the vegetated area adjacent to a creek. This area is the transition zone from creek to land, and is characterized by hydrophilic plants. The diverse vegetation adds various nutrients to the creek and shades the water. Consequently, the chemistry and temperature of the creek may change temporally and spatially. For example, a taller, denser, and wider span of plants can increase shade and the amount of nutrients available for the creek.
The riparian mapping of Fishtrap Creek was performed in two stages. First, the riparian zone was outlined on orthophotos, which are listed in Appendix D. ArcMap 9.3 was used to delineate polygons around the potential riparian vegetation (Figure 2.18) using the reach breaks previously characterized by Pearson (2004), and further constrained within the area defined by slope breaks determined from the orthophotos. The polygons encompass the vegetation bordering the creek, up to a maximum of 50 m on either side of the channel, which is the Riparian Management Area (RMA) defined within the BC Forest Practices Code Riparian Management Planning Guidebook (1995) for a fish-bearing stream 5-20 m wide. This distance was applied as a conservative estimate of stream width to encompass all reaches Fishtrap Creek and provide a standardized approach, although the stream width is less than 5 m in many sections of the upper reaches.

Following the mapping, field reconnaissance and verification was performed to confirm mapped polygons and to determine vegetation type. For the purposes of this research, riparian vegetation was separated into four categories based on dominant vegetation type: grasses, shrubs, deciduous trees, and coniferous trees. Field
verification was undertaken along portions of Fishtrap Creek and its tributaries that were visible from the road or reaches on private property where access was granted. At each observed location, the extent of the riparian vegetation on both sides of the creek was measured were access was possible, or estimated visually from the nearest observation point. Select photos from field reconnaissance are included in Appendix D, with a summary table of the reach orthophotos used, and the reach break attribute table from ArcMap.
Chapter 3.

Exploratory Data Analysis and Independent Component Analysis

Sediment-water interface temperature data from the local scale study sites (Fishtrap Creek (F1) and Bertrand Creek (B1); Figure 3.1) were summarized with descriptive statistics, and then evaluated using an exploratory data analysis approach, followed by Independent Component Analysis (ICA). Exploratory methods were used to evaluate the hourly sediment-water interface temperature data for general patterns, both spatially and temporally. Exploratory data analyses were performed in R (R Development Core Team 2011). Further statistical tests were completed to determine if the hourly temperatures were normally distributed at each location, and to test if the variability in the sediment-water interface temperatures was statistically significant at the local scale sites. Detailed analyses of the sediment-water interface temperatures employed ICA to separate the observed sediment-water interface temperatures into the component signals. A background on ICA is presented in this chapter, and a summary of the approach and detailed results are discussed in Chapter 5.

Sediment-water interface temperature data from the regional scale study sites (Fishtrap Creek Watershed; Figure 3.1) were summarized with descriptive statistics, and the spatial and temporal patterns were evaluated using boxplots.
3.1. Descriptive Statistics

The hourly sediment-water interface temperatures for both the full period of record and for the summer low flow periods (July to September) at the local scale sites (Figure 3.1) are summarized in Table 3.1. Fishtrap (F1) included data from 15 loggers, and Bertrand (B1) for 19 loggers. Figure 3.2 shows the mean daily mean, minimum, and maximum sediment-water interface temperatures for the period of record, with the summer period indicated by the thick lines.

Figure 3.1. Locations of the local and regional scale study sites in the Canadian portion of the Bertrand and Fishtrap Creek watersheds (outlined in black).
Table 3.1.  Summary of hourly sediment-water interface temperatures (°C) for Fishtrap (F1) and Bertrand (B1) local scale sites. Annual and summer (July - September) statistics are shown.

<table>
<thead>
<tr>
<th>Site</th>
<th>Period</th>
<th>Median</th>
<th>Mean</th>
<th>Min</th>
<th>Max</th>
<th>Mean Range</th>
<th>n (obs.)</th>
<th>Standard Deviation</th>
</tr>
</thead>
<tbody>
<tr>
<td>F1</td>
<td>Annual (July 2008- Oct 2012)</td>
<td>10.33</td>
<td>10.35</td>
<td>0.39</td>
<td>20.05</td>
<td>1.55</td>
<td>510172</td>
<td>± 0.44</td>
</tr>
<tr>
<td></td>
<td>Summer (2008-2012)</td>
<td>13.89</td>
<td>13.77</td>
<td>10.16</td>
<td>20.05</td>
<td>1.39</td>
<td>135496</td>
<td>± 0.39</td>
</tr>
<tr>
<td>B1</td>
<td>Annual (July 2008- Oct 2011)</td>
<td>10.52</td>
<td>10.53</td>
<td>0.02</td>
<td>23.95</td>
<td>1.19</td>
<td>193585</td>
<td>± 0.83</td>
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<tr>
<td></td>
<td>Summer (2008-2011)</td>
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<td>15.76</td>
<td>8.74</td>
<td>23.95</td>
<td>0.73</td>
<td>83630</td>
<td>± 0.30</td>
</tr>
</tbody>
</table>

Figure 3.2. Mean daily sediment-water interface temperatures for a) Fishtrap (F1) and b) Bertrand (B1) Creek local scale sites. The figure shows the daily mean (blue), the maximum (red), and minimum (green) temperatures. The summer periods (July – September) are indicated by the thicker lines.
3.2. Histograms

Histograms of the hourly data were produced for each datalogger using the `truehist` function in the MASS package in R. The histograms were compared for both the full period of record and for the summer periods, and were plotted with even distribution of break points for the bins. These histograms were produced for both the F1 and B1 local scale sites. Examples of annual and summer histograms for Fishtrap Creek and Bertrand Creek are shown in Figure 3.3 for two datalogger locations (F1-4 and B1-17). The remaining histograms for Fishtrap and Bertrand Creeks are included in Appendix E.

The histograms reflect the different temperature distributions at different locations, both annually and during the summers. For some locations, the annual sediment-water interface temperature distribution is bimodal, with the division between the clusters slightly above 11°C (Figure 3.3). A similar pattern has been reported for soil moisture, which is described as having “preferred states” of wet or dry, and the system switches between those states, resulting in the bimodal pattern (Kampf 2011). Sediment-water interface temperature may be responding in a similar manner, whereby the interface temperature has “preferred state” of warm or cold; the threshold in this system is near 11°C. This temperature corresponds to the mean annual groundwater temperature which may be influencing the “preferred state”. When only the summer data are plotted, the stream temperatures no longer have a bimodal distribution (Figure 3.2), indicating that the break in the temperature distribution is seasonally controlled. Many of the dataloggers have skewed distributions, both seasonally and annually.
Figure 3.3. Examples of histograms for the data for all years for the annual period (top row) and the summer period (bottom row) for locations a) F1, Location 9 and b) B1, Location 17. The locations are shown in Figure 3.1.

3.3. Cumulative Distribution Functions

To further compare the temperature data and examine the data for outliers and skewedness, the hourly data were plotted as Quantile-Quantile (QQ) plots for each datalogger location using the `qqnorm` function in R. The QQ plots display the data against theoretical distribution line; thus QQ plots can be used as a diagnostic tool to observe normality and skewness. Normally distributed data will plot along a straight line. At F1, the data have an approximate sigmoidal form with tails deviating from the normal line at both ends, indicating non-normal distribution (Figure 3.4). The QQ plots for all the datalogger locations are provided in Appendix C.
Figure 3.4. QQ plot of the hourly sediment-water interface temperature for datalogger Location 8 at F1.

To further understand the QQ plots, and look for spatial variability in the temperature distributions across each site, as well as to compare between the sites, empirical cumulative distribution functions (ecdf) were plotted using the `ecdf` function in R. The cumulative distribution functions for the hourly summer period data for each datalogger for each site are plotted together in Figure 3.5. The distributions are similar; there are no outliers. Moreover, the range in summer sediment-water interface temperature at F1 is smaller than at B1.
Figure 3.5. Empirical cumulative distribution functions for the hourly summer period sediment-water interface temperatures at each of the temperature loggers for F1 and B1. Each color line represents a different datalogger.

In addition to the spatial variability in the hourly sediment-water interface temperatures at each site (Figure 3.5), the distributions also varied temporally at both sites, with variations occurring between the summer periods. An example of difference between summers is shown for F1 in Figure 3.6, which shows differences in the temperature range and distribution for the summer periods of 2008 and 2009. The full suite of cumulative distribution function plots for individual summer months, as well as those compiled for the summer periods at F1 and F2, are provided in Appendix E.

Figure 3.6. Empirical cumulative distribution function for the 2008 and 2009 summer periods for the hourly sediment-water interface temperatures for all the dataloggers at Fishtrap Creek.
3.4. Wilcoxon Rank Sum Test

The ecdf of the sediment-water interface temperatures (Figure 3.4) show spatial variability in the temperatures within each local site, and the histograms and the QQ plots indicate that temperatures at each location may not be normally distributed. To evaluate if both the hourly and mean daily sediment-water interface temperature differed significantly from the site mean, the temperatures were evaluated using Wilcoxon rank sum test using the `wilcox.test` function in R. The Wilcoxon test is a non parametric test and is used to test the null hypothesis that the median temperatures at each datalogger location are not significantly different from the site mean at a significance level of $p < 0.05$ (Crawley 2005). The Wilcoxon rank sum statistic ($W$) is the sum of the ranks for the observations from one group, the smallest group if they are different sizes. The p-value is obtained from tables of values and compared to the standard significance level $\alpha = 0.05$. The null hypothesis is that the distributions of the groups are equal; therefore, the probability of a value from one group (A) exceeding a value from the second group (B) is equal to the probability of a value from B exceeding a value from A.

The Wilcoxon rank sum test was completed as a two-tailed paired test, comparing the hourly and daily sediment-water interface temperature for each datalogger location to the mean of site. An assumption of this method is that each member of the paired test is independent; however, by comparing each location temperature to the mean of all the temperatures, the members of the pair are no longer independent. It is acknowledged that the approach of comparing one series to a mean calculated with the inclusion of all the times series violates the assumption of independence. The outcome of Wilcoxon tests completed in this manner was used as a first approximation to test for significance and further analysis on the statistical significance was completed to evaluate the results. The p values for the hourly and daily temperatures are presented in Table 3.2. The results show that for both hourly and mean daily temperatures at several datalogger locations, in both streams, the p-values are $<0.05$, and therefore the null hypothesis that the temperatures at those locations have the same distribution as the site mean temperature is rejected.
Table 3.2. Wilcoxon rank sum test p-values for the hourly and daily sediment-water interface temperatures for Fishtrap and Bertrand Creeks. Data logger locations with p-values <0.05 are indicated by *. Data logger locations are shown in Figure 3.1.

<table>
<thead>
<tr>
<th>Datalogger Location</th>
<th>Fishtrap Creek</th>
<th>Bertrand Creek</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Hourly</td>
<td>Daily</td>
</tr>
<tr>
<td>1</td>
<td>0.106</td>
<td>*</td>
</tr>
<tr>
<td>2</td>
<td>0.668</td>
<td>*</td>
</tr>
<tr>
<td>3</td>
<td>*</td>
<td>*</td>
</tr>
<tr>
<td>4</td>
<td>0.143</td>
<td>*</td>
</tr>
<tr>
<td>5</td>
<td>*</td>
<td>*</td>
</tr>
<tr>
<td>6</td>
<td>*</td>
<td>*</td>
</tr>
<tr>
<td>7</td>
<td>0.627</td>
<td>*</td>
</tr>
<tr>
<td>8</td>
<td>*</td>
<td>*</td>
</tr>
<tr>
<td>9</td>
<td>*</td>
<td>*</td>
</tr>
<tr>
<td>10</td>
<td>0.905</td>
<td>*</td>
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<tr>
<td>11</td>
<td>*</td>
<td>*</td>
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<tr>
<td>12</td>
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<td>13</td>
<td>*</td>
<td>*</td>
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<tr>
<td>14</td>
<td>*</td>
<td>0.348</td>
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<tr>
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<td>*</td>
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<td>16</td>
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<tr>
<td>17</td>
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<tr>
<td>18</td>
<td></td>
<td>*</td>
</tr>
<tr>
<td>19</td>
<td></td>
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</tr>
</tbody>
</table>
3.5. Boxplots

Boxplots were used to summarize the sediment-water interface temperatures both spatially and temporally to identify significant differences in the interface temperatures within each local site. This method was used to validate and expand on the results of the Wilcoxon rank-sum test (Section 3.4). Notched boxplots were produced using the `boxplot` function in R. The boxplots show the median sediment-water interface temperature at center of the notch, and the extent of the notch indicates the 95th percent confidence interval of the median (Crawley 2005). The top of the box is the 75th percentile value, and the bottom of the box is the 25th percentile, encompassing the interquartile range. The maximum and minimum values are represented by the whiskers at each end of the box, and any outliers greater than 1.5 times the quartiles are shown as dots. The notches can be used to visualize significant differences in the sample medians. The notch represents the 95% confidence interval of the sample median, and as a general test, if the notches do not overlap, the samples have significantly different median values (Chambers et al. 1983; Crawley 2005). Variable widths were used for all of the boxplots, with the width of each box proportional to the square root of the number of temperature recordings at each location. Mean monthly sediment-water interface temperatures were compiled and presented as a regular (un-notched) boxplots for comparison between the local scale sites.

Examples of the mean hourly and mean daily sediment-water interface temperatures for F1 and B1 are shown in Figures 3.7 and 3.8, respectively. August data are used as an example of summer period temperatures as the most data were available for all dataloggers for August. Each boxplot shows all available data for the period of July 2008 to October 2012 for F1, and July 2008 to September 2011 for B1. The notched boxplots show spatial variability in sediment-water interface temperatures both hourly and as a daily mean, as shown in the August plots. Similar to the Wilcoxon test results (Table 3.2), significant differences are observed at the hourly scale, and these become more discernible in the mean daily values. Overall, there are fewer outliers for the daily data than for the hourly data as expected.

The boxplots show site-wide differences between the two streams; for example, in August there are generally higher temperatures and temperature ranges at B1 relative to F1. B1 has a much greater number of outliers in August compared to F1; however,
during other months there is no consistent pattern in outliers between the creeks. Note, however, that the distribution of outliers is affected by the number of temperature recordings at each location. At B1, for example, data were collected seasonally or for shorter periods of time than at F1 (see number of observations in Table 3.1). Therefore, caution must be exercised when interpreting the outlier data.

Figure 3.7. Notched boxplots of the mean hourly and daily August sediment-water interface temperature for the 15 dataloggers at F1 (locations are shown in Figure 3.1).

Figure 3.8. Notched boxplots of the mean hourly and daily August sediment-water interface temperature for the 19 dataloggers at B1 (locations are shown in Figure 3.1).
The Wilcoxon test only compared each location to the site mean, while the boxplots provide a visual representation of the sediment-water interface temperatures as well as the variation by location, allowing for all locations to be compared to each other. Also, the notched boxplots of the temperatures for each month show significant differences throughout the year, which provides temporal information not included in the Wilcoxon test. Hourly and daily boxplots for all months are included in Appendix C.

The mean monthly sediment-water interface temperatures for each datalogger location at F1 and B1 are shown in Figures 3.9 and 3.10, respectively, using August again as an example. The apparent larger range in mean monthly interface temperatures at B1 Locations 17 and 18 is in part due to data being collected from those locations for four summers, while the remaining locations have fewer data points. The significant differences at the monthly scale are consistent with those observed at the hourly and daily scales. The mean monthly boxplots for both F1 and B1 are included in Appendix E.

![Boxplot of the mean monthly August sediment-water interface temperatures at F1 for 15 dataloggers (locations shown in Figure 3.1).](image)
Overall, there were significant differences in the mean hourly, daily, and monthly sediment-water interface temperatures across each local scale site. The locations with consistently significant differences in temperature had warmer temperature in the winter and cooler temperatures in the summer (e.g., Locations F1 -4 and -9, B1-5 and -17), which suggests a localized influx of groundwater that results in temperature attenuation (cooling in summer) near those datalogger locations.

Side-by-side boxplots were used to compare the distributions of the mean monthly temperatures at F1 and B1; these are plotted with the mean groundwater temperature in Figure 3.11. Statistically significant differences in the temperatures are observed during all months of the year. At both sites, the interquartile ranges were variable throughout the months; the smallest mean monthly range in temperature occurred in late spring (April – May) and the largest range in temperature occurred in winter (Jan – March). At B1, the range was also large in early summer (June – July). The mean monthly temperatures for F1 had a smaller range over the year, remaining warmer in winter and cooler in summer, relative to B1. At F1, the largest number of outliers occurred in July. Generally, the outliers at both sites occurred during periods of warmer interface temperatures, with outliers in the lower temperature range. In the fall and winter, the outliers were mainly in the upper range of temperature. The temperature distribution at F1 is closer to the mean daily groundwater temperature throughout the
year. The temperatures at F1 are attenuated annually relative to B1, and remain warmer in the winter and cooler in the summer period. Overall, the F1 temperature patterns are characteristic of a stream that is receiving greater relative groundwater contributions (Caissie 2006).

![Figure 3.11. Mean monthly sediment-water interface temperatures for F1 and B1. The horizontal line shows the mean daily groundwater temperature (10.9°C) annually, with the grey band showing the standard deviation in the daily groundwater temperature (±0.4°C).](image)

3.6. Regional Scale Boxplots

Notched boxplots were used to evaluate spatial differences in sediment-water interface temperatures through Fishtrap Creek watershed. The locations of the 13 regional sites are shown in Figure 3.1. The number of dataloggers at each regional site ranged from 1 datalogger at location F13 to 15 at location F1 (the full list is provided in Appendix A). The period of record available for each location ranged from 24 months (F13) to 51 months (F1). To obtain a single temperature value representing the sediment-water interface temperature at each location, the mean daily temperature was calculated from hourly temperature from all the available dataloggers at each location.
Boxplots were produced to show the mean daily values for each location by month in order to standardize the data for the different number of months over which the data were recorded. Figure 3.12 provides an example of boxplots for the regional sites, showing the mean daily August temperatures. The flow direction is from right (upper watershed) to left (F1 at the International Border), and the green and yellow boxplots indicate sites located along tributaries to the mainstem (blue boxes). The boxplot shows that overall; temperatures in the upper watershed (sites F3 to F11) increase with distance downstream, with Enns Brook (yellow) having the highest temperatures. Enns Brook had beaver activity along some reaches, damming the flow, as well as a small lake near the headwaters. Both features created stagnant water, likely increasing the temperature of the water originating in the upper reaches of Enns Brook. Flow from the Waetcher Creek tributary (green) had significantly lower temperatures than the upstream reaches of Fishtrap Creek. The furthest upstream site on Waetcher Creek had a temperature and interquartile range significantly lower than the other locations in the Fishtrap Creek watershed, and was close to the mean summer groundwater temperature (10.7 ± 0.2° C). Downstream of the inputs from Waetcher Creek, the temperatures along the Fishtrap mainstem (sites F1 to F5) remained relatively stable with distance downstream.
The higher sediment-water interface temperatures along all the upper reaches are likely a result of decreased connectivity with groundwater in areas associated with finer grained surficial material (see Chapter 2). Waetcher Creek water temperature in the upper reaches was similar to the groundwater temperature, suggesting a strong connectivity with groundwater at that location. Below the confluence of the Waetcher Creek and Fishtrap Creek mainstem (sites F1 to F5), the sediment-water interface temperatures are relatively stable with distance, and this corresponds to sections of the watershed characterized by coarser grained surficial units (Chapter 2). This indicates temperature buffering is occurring, likely from groundwater influx. The temperature differences along the length of Fishtrap Creek are less pronounced outside of the summer period, and during winter months the sediment-water interface temperatures tend to increase in the downstream direction, indicating temperature attenuation from warmer groundwater influx. The results of the exploratory analysis and the spatial...
variations in sediment-water interface temperatures for the regional sites are discussed in more detail in Chapter 6. The boxplots for the regional scale sites for all months are provided in Appendix E.

### 3.7. Independent Component Analysis

To evaluate the significant differences in the summer sediment-water interface temperature spatially and temporally at the local scale sites, independent component analysis (ICA) was employed to compare how temperature signals differed between the summers. The results of the ICA are discussed in detail in Chapter 5. This section provides an overview of the ICA method.

Independent component analysis is a statistically-based, signal processing technique that can be used to separate independent source signals from an input of mixed time series signals (Comon 1994; Whitfield et al. 1999; Hyvarinen and Oja 2000; Naik and Kumar 2011; Hyvarinen 2012). ICA requires no prior knowledge of the mixing process and is thus one of the most common forms of blind signal separation. The basis of ICA, summarized in Figure 3.12, is that recording devices, such as temperature dataloggers, record mixed signals (x). These mixed signals are products of source signals (s) and some mixing matrix (A), where both s and A are unknown. The classic explanation of ICA is described by the “cocktail party” problem, wherein conversations from a number of simultaneous conversations “observed” by a number of microphones are separated into the individual speech signals. Figure 3.13 shows an overview of ICA, the goal of which is to obtain an estimate of the independent source signals (s), using the recorded signals (x). The source signals (s) can be estimated from the mixed signals (x) and an un-mixing matrix (W) which is defined as where $W=A^{-1}$. With the ICA method, the only observed variable is the mixed signal, $x$, and there is no prior information input on the original source signals or the mixing matrix, $A$. This absence of input information is the aspect of the method that is considered “blind”. As a simplification of the process in this application, the ICA equations do not incorporate any noise components, or time lag in the recordings.
ICA requires three main simple assumptions of the statistical properties (Hyvarinen and Oja 2000; Naik and Kumar 2011; Hyvarinen 2012), and these are discussed in further detail in Chapter 5:

1) source signals are statistically independent at each time instant,
2) source components, $s$, have non-Gaussian distributions; and
3) the mixing matrix, $A$, is square and invertible.

The first assumption is that that source signals, at each time instant, are statistically independent, and as such, the information of any one source signal does not provide information about any other signal. Hyvarinen and Oja (2000) noted that in many cases this assumption is not unrealistic and in practice does not need to be precisely true. The fundamental assumption in ICA that separates it from other methods is the requirement that the source components, $s$, have non-Gaussian distributions (Comon 1994; Hyvarinen 1999b; Hyvarinen and Oja 2000; Naik and Kumar 2011; Hyvarinen 2012). The mixing matrix, $A$, cannot be determined for Gaussian independent components because the sum of two or more Gaussian independent components will have a distribution that is itself Gaussian, and therefore cannot be distinguished from the individual sources. When two or more Gaussian signals are summed, the joint density is symmetric and does not provide necessary information on the directions of the columns in the mixing matrix. While being the fundamental assumption in ICA, the non-Gaussian requirement is not excessively restrictive because it is possible for one component to be Gaussian, and also in practice most independent sources are in fact non-Gaussian.
(Hyvarinen, 1999b; Hyvarinen and Oja, 2000; Naik and Kumar, 2011). The final assumption is that mixing matrix, $A$, is square and invertible (Naik and Kumar 2011; Hyvarinen 2012). This assumption must be met in order to achieve the estimation of the un-mixing matrix, and subsequent estimation of the source signals. In addition to the assumptions on the source signals, there are three effects of mixing related to the assumptions. Signal mixtures are not independent because they share the same source signals, although the source signals themselves are independent. The mixtures are a sum of two independent random variables, and as such, have a distribution closer to Gaussian than the two original variables alone. Lastly, the mixing of signals increases the complexity of the signal and that complexity will be greater than that of the simplest source signal.

By meeting the three assumptions and associated conditions and understanding the effects of mixing, the independent source signals and the mixing matrix can be estimated using ICA, with two ambiguities remaining (Hyvarinen and Oja 2000; Naik and Kumar 2011; Hyvarinen 2012). The first is a magnitude and scaling ambiguity, in which the true variance of the independent components cannot be determined. Since both the source signals and the mixing matrix are unknown, any scalar multiplier in one of the sources could be cancelled by dividing through in the corresponding column of the mixing matrix by the same scalar. Therefore, the solution is to assume that each source has a unit variance; however, this leaves ambiguity in the sign, as the sources could be multiplied by $-1$ without impact on the results. The other ambiguity is that the order of the independent sources cannot be determined, because of both the source signals and the mixing matrix being unknown. With both terms being unknown, the order of terms can be freely changed and no restrictions or conditions are imposed on the sources during separation, leaving the order indistinguishable, and each permutation is equally valid.

In this study, the ICA was used to perform blind signal separation of sediment-water interface temperatures using the FastICA algorithm in R (Marchini et al. 2010). ICA signal separation was used for all available dataloggers (up to 15) at site F1 (Figure 3.1) for sediment-water interface temperatures for four consecutive summer periods. This section provides a summary of the application of ICA in this study, and the methods and results are presented in more detail in Chapter 5. The ICA was used to evaluate
temporal variability in the sediment-water interface temperatures, and relate those to processes that influence sediment-water interface temperature.

The first step in using ICA for this study was to determine the number of components to extract from the mixed signals. The number of components that can be extracted are any number equal to or less than the number of recordings (datalogger locations). As the number of extracted components increases, the signals become non-unique and difficult to distinguish and interpret. While ICA requires no prior knowledge of the mixing processes in order to separate the signals, this study found that a reasonable conceptual model was warranted to better constrain the analysis process and to interpret the results. To determine the number of components to extract, the temperature signals were evaluated against variables that were known, or identified as potentially important, in the conceptual model. The important factors influencing the sediment-water interface temperature at this site were identified from a simplified heat budget as the solar radiation, background stream temperature flowing into the reach, and groundwater flux. The first variable representing these factors was the estimated sediment-water interface temperature, which represents the incoming solar radiation. It was calculated through the empirical relationship between air and sediment-water interface temperature (see Chapter 4 for details). The second variable is stream discharge, representing the heat flux into the stream reach from upstream. The third variable, representing groundwater flux, was groundwater level because groundwater temperature remained relatively constant over all the summer periods, while the groundwater level fluctuated during and between the summer periods. For this reason groundwater level is a more appropriate variable to represent groundwater flux into the reach.

For each of the four summer periods, three ICA signals were extracted using FastICA, and compared to the relevant variables using cross-correlation. An example of the ICA extracted signals, and the cross-correlation analysis is shown in Figure 3.14, which presents the results for the 2009 summer period. The variables for comparison in the cross-correlation were a) the estimated sediment-water interface temperature, b) stream discharge, c) 2-day smoothed stream discharge, and/or d) groundwater level. In Figure 3.14, the top row shows plots of each variable for the summer period. The column on the far left shows the mean sediment-water interface temperature at the top, and the
three extracted components below it. For comparison, the cross-correlation plots are shown below each variable.

The use of the cross-correlation also provides a means to overcome the ICA ambiguities. The cross-correlation is used to determine if the correlation is significant (greater than 0.1 and exceeding the calculated statistical significance values on the plot), and the lag time of the correlation. The sign of the correlation is not relevant in determining either of those factors. The order of the extracted signals was manually established after the signals were classified, and the signal correlating with the estimated sediment-water interface temperature was determined. The order applied was to establish that signal 1 was the signal most strongly correlated to the estimated interface temperature. The order established had no impact on the results or interpretation of the ICA, but was a means to simplify the discussion.

The use of ICA in this study differed from the analytical approaches that are classically used to examine stream temperatures, which consider the source signals and relevant noise and bias components (air temperature, water temperature, groundwater temperature, noise and sensor drift). By combining ICA with cross-correlation analysis, it was possible to directly relate the extracted independent signals to variables that can influence sediment-water interface temperature through inputs to the heat budget within a stream reach.
Figure 3.14. Results of the extracted ICA components and the cross-correlation analysis with the variables contributing to the heat exchanges. The extracted ICA components are shown in the left panel (pink) below the mean stream temperature recorded by the dataloggers at the sediment-water interface. The cross-correlations for each ICA component are shown in red, with the correlated variable shown in the top row of each column. The horizontal dashed lines in the correlation plots mark the 0.1 correlation value, above which the correlations were considered significant.
Chapter 4.

Comparing the Groundwater Contribution in Two Groundwater – Fed Streams Using a Combination of Methods

4.1. Introduction

Groundwater is an important component of streamflow (baseflow) throughout the year in many streams, but during the summer low flow period, stream flow is often sustained by groundwater influxes (Winter et al. 1998; Smakhtin 2001; Hatch et al 2006). Summer is also a critical period for aquatic habitat, because the warming associated with higher air temperature and low streamflow may cause critical thresholds (flow and stream temperature) to be reached for many aquatic species (Nelitz et al. 2007; Burn et al. 2008b; Cunjak et al. 2013). Groundwater influxes to streams, however, have the ability to buffer these impacts by: 1) maintaining water levels (stream discharge) (Fleckenstein et al. 2004; Beatty et al. 2010), 2) providing thermal refugia (Hayashi and Rosenberry 2002; Caissie 2006; Kanno et al. 2014; Kurylyk et al, 2014), and 3) supplying nutrients and inorganic matter (Power et al. 1999), each being important to aquatic health. Therefore, understanding what factors influence the degree to which groundwater interacts with streams is important for management of streams, riparian habitats, and water use (Brewer 2013; Kanno et al. 2014).

A variety of methods can be used to characterize groundwater-surface water interactions at different spatiotemporal scales. These include direct methods for measuring exchanges between groundwater and surface water; indirect methods; and combinations of both (Cey et al. 1998; Jones and Mulholland 2000; Essaid et al. 2008; Rosenberry and LaBaugh 2008). Direct methods include seepage meters, piezometers, and stream flow measurements (Boulton 1993; Baxter et al. 2003; Rosenberry 2008). Indirect methods include indicators such as water chemistry and mixing properties, or
tracers such as heat (Stonestrom and Constantz 2003; Anderson 2005; Malcolm et al.
2005; Constantz 2008). These various field methods have limitations, and the choice of
field method(s) utilized must be appropriate for the spatial and temporal scale of the
investigation (Cey et al. 1998; Kalbus et al. 2006).

Temperature is considered a robust and easily measured parameter to assess
groundwater interactions with surface water (Anderson 2005; Webb et al. 2008). At
depths below a few metres, at which there is practically no annual fluctuation in ground
temperature, groundwater temperatures remain relatively stable throughout the year,
with values 1-2°C higher than mean annual air temperature, at least in the absence of
deep seasonal snowpack (Meisner et al. 1988; Zhang 2005). The fluctuations in
groundwater temperatures are also much less pronounced than the diurnal and
seasonal fluctuations in surface water temperatures, which are known to respond
strongly to patterns in air temperatures and solar radiation (Johnson and Jones 2000;
Johnson 2003; Moore et al. 2005; Caissie 2006). Stream temperature variations have
been used as a proxy for identifying groundwater fluxes to streams, and in combination
with other field methods have been employed as a tracer to quantify water exchanges
(Silliman and Booth 1993; Stonestrom and Constantz 2003; Becker et al. 2004; Conant
2004; Anderson 2005; Kalbus et al. 2006; Schmidt et al. 2006; Arrigoni et al. 2008;
Constantz 2008; Hebert et al. 2011; Caissie et al. 2014). For streams showing
attenuated temperature variations both diurnally and seasonally, temperature has been
used to identify the relative flux of groundwater to the channel, and relative contributions
are useful for site comparisons (Stonestrom and Constantz 2003; Moore et al. 2005;
Caissie 2006).

Both regression and stochastic approaches have been used to predict the
thermal regime of a stream using air temperature as a predictor given the correlation
between air temperature and stream temperature (e.g. Stefan and Preud’homme 1993).
Deterministic models have also been used (e.g. Caissie et al. 2014). For example, to aid
in understanding of fluxes, a simple heat budget for a stream can be calculated from the
combination of energy fluxes at the water-air interface and the water-bed interface
(Sinokrot and Stefan 1993; Webb and Zhang 1997; Evans et al. 1998; Hannah et al.
2004; Xin and Kinouchi 2013). Differences between the heat budget for the air-water
interface and the calculated heat transfer to the streambed can be useful in identifying gaps in the heat balance from processes taking place at the sediment-water interface. The relationship between air and water temperature can further be used for relative comparisons of groundwater-surface water interactions at similar sites (Caissie 2006).

The purpose of this study is to compare different methods for estimating the relative contribution of groundwater to two groundwater-fed streams, Fishtrap and Bertrand Creeks, which drain the Abbotsford-Sumas aquifer in the Lower Fraser Valley of southwest British Columbia (BC) (Figure 4.1). Previous studies have suggested the groundwater contribution in Fishtrap Creek is greater than in Bertrand Creek (Johanson 1988; Pearson 2004; Berg and Allen 2007), but to date, no comprehensive studies have been undertaken to compare the groundwater influxes into each stream. These particular streams were selected because, while the climate and topographic relief are similar, and the watersheds are of similar size, the geological substrate differs, with coarser grained surficial geology in the Fishtrap Creek watershed. Therefore, one hypothesis for the higher groundwater contribution to Fishtrap Creek is the more permeable geological substrate.

Figure 4.1. The Bertrand and Fishtrap Creek study sites in British Columbia (Canada). The study sites are located within the Canadian portion of the watersheds (dark grey).
This chapter employs a simplified heat budget approach in combination with direct and indirect measurements of groundwater flux to compare the relative groundwater contributions along two reaches, one in Fishtrap Creek and one in Bertrand Creek. Field data were collected over five summer periods (July through September). In addition, other high quality datasets are available (streamflow, climate, and groundwater data) to support the analysis. The chapter also explores some of the challenges and uncertainties in making measurements in streams with low flows.

4.2. The Study Area

The surficial geology is composed of Quaternary glacial sediments overlying Tertiary bedrock (Scibek and Allen 2005). The dominant units in the vicinity of the study area are the Fort Langley Formation and the Sumas Drift, and lesser amounts of Salish Sediments (Figure 4.2). The Fort Langley Formation is a glaciomarine unit, comprising pebbly silt in clay, which has been interpreted as a confining layer (Mitchell et al. 2003). Sumas Drift comprises glaciofluvial sands and gravels, with discontinuous lenses of till and sandy till. The sand and gravel of the Sumas Drift are the sediments that host the Abbotsford-Sumas aquifer (~160 km² in area). Salish Sediments occur in isolated locations in the study area, and comprise fluvial, lacustrine, and colluvial sediments (Johanson 1988; Scibek and Allen 2005).
Fishtrap Creek watershed is approximately 37 km², and Bertrand Creek watershed is approximately 51 km² (Pruneda 2007). Within each watershed there are differences in the surficial geology (Figure 4.2) and land use. Fishtrap Creek flows over predominantly Salish Sediments and Sumas Drift, with lesser amounts of fine grained Ft. Langley Formation in the upper reaches (Johanson 1988; Scibek and Allen 2005). Fishtrap Creek bed material at the study site is dominantly sand and fines, with trace to minor amounts of gravel, and abundant organic material (leaf litter and small woody debris). Fishtrap Creek watershed has agricultural use, dominantly blueberry fields, in the lower watershed, with some industrial and urban land in the upper reaches. Riparian shade is limited to seasonal invasive reed grasses present within the channel as in-stream vegetation, which becomes thicker throughout the summer season.

In contrast, the surficial sediments in the Bertrand Creek watershed are dominantly Ft. Langley Formation, with lesser amounts of Sumas Drift and Salish sediments near the stream channel (Johanson 1988; Scibek and Allen 2005). Near the study site on Bertrand Creek, the surficial material has intermittent sections of both coarse and fine grained material. The bed material is dominated by well sorted fine
gravel and coarse sand, and the thickness of the bed material ranges between 0-30 cm, overlying clayey silt. Bertrand Creek flows through mixed development, dominantly rural residential in the study area, with some industrial and urban development in the upper reaches and, at the study site, land use and cover is rural residential with some partial riparian cover from shrubs and deciduous tree canopy along the channel margins.

4.2.1. Climate and Hydrology

The climate of the Abbotsford area is characterized as humid and temperate. Mean daily temperature at the Abbotsford International Airport (Station 1100030) ranges from 2.6°C in January to 17.7°C in August, with an annual mean of 10°C (Environment Canada 2014). The daily mean maximum temperature reaches close to 24°C in July and August. Annual average precipitation is 1573 mm/yr, with approximately 70% occurring between October and May, and only 64 cm falling as snow (Zebarth et al. 1998; Berka et al. 2001; Environment Canada 2014).

Fishtrap and Bertrand Creeks originate in the Abbotsford uplands and flow in a southerly direction to the Nooksack River in Washington, USA (Figure 4.1). Both streams have flow regimes that are controlled by precipitation and interaction with groundwater (Berg and Allen 2007). The flow regime is driven by direct precipitation, and stream discharge lags precipitation by a few days, with minimum flows occurring in August, shortly after the minimum precipitation in July and August (Berg and Allen 2007). The streams provide habitat for salmon and two fish species listed as endangered in Canada, the Nooksack Dace and the Salish Sucker (Pearson 2004), and summer low flows have threatened fish populations through loss of habitat, increased predation, and decreased water quality (Avery-Gomm et al. 2014). These streams have been identified as having potentially variable interaction with groundwater both between them, and along their lengths (Johanson 1988; Starzyk 2012).

Groundwater recharge to the Abbotsford-Sumas Aquifer is primarily by rainfall, and groundwater levels fluctuate seasonally and inter-annually (Graham et al. 2014). Groundwater flow in the aquifer is generally southwest, but is controlled locally by differences in topography, geology, and interactions with surface water. Groundwater
levels generally lag precipitation by approximately three months and the annual variation in water levels is approximately 2 m (Scibek and Allen 2005). The variability of interaction with groundwater in the vicinity of the streams (Johanson 1988) is thought to be related to differences in surficial geologic units underlying the streams.

4.3. Methodology

4.3.1. Climate and Hydrology

The nearest climate station to Fishtrap Creek is located at the Abbotsford International Airport approximately 3.6 km away from the site (Figure 4.3). Air temperature from this station was used to represent Fishtrap Creek. The stream gauging station on Bertrand Creek, operated by the US Geological Survey (USGS), (Figure 4.3) also records air temperature (and water temperature) using a Design Analysis® H-377 temperature sensor with an accuracy of ± 0.2°C. Air temperature from this station was used in this study to represent Bertrand Creek. Precipitation for both sites was based on that measured at the Abbotsford Airport. Based on regional precipitation patterns, the annual precipitation at Bertrand Creek is estimated to be within 100 mm/year of the amounts recorded at the Abbotsford Airport, which is only ~11 km away (Scibek and Allen 2005).

For Fishtrap Creek, continuous stream discharge data were obtained at 15-minute intervals from the nearby Water Survey of Canada (WSC) gauging station (Station 08MH153) (Figure 4.3). For Bertrand Creek, continuous stream discharge data were obtained at 15-minute intervals from the USGS gauging station (Station 12212390) located approximately 35 m downstream of the lower end of the study site (Figure 4.3).

Near Fishtrap Creek, hourly groundwater levels were recorded at Environment Canada’s monitoring well ABB01, located approximately 660 m from the Fishtrap Creek study site (Figure 4.3). Hourly groundwater level data for Bertrand Creek were obtained from the BC Ministry of Environment’s observation well #301, which is located 7 km northeast of the study site. Hourly groundwater temperatures were only available from
The temperature was recorded 8 m below ground surface (mbgs), a few metres below the average water table depth.

4.3.2. Interface Temperature

The interface temperatures in this study are defined as water temperatures recorded at the surface of the streambed, using temperature loggers affixed at the water-bed interface. In July 2008, 15 temperature dataloggers were installed along a 40 m long reach of Fishtrap Creek, and 19 temperature loggers were installed along a 70 m long reach in Bertrand Creek (Figure 4.3). Interface temperatures were recorded by TidbiT® v2 Temp loggers (UTBI-001) secured, unshielded, at the water-sediment interface. The loggers have an accuracy of ± 0.2°C and a resolution of 0.02°C. The loggers were factory calibrated, and calibration was verified prior to deployment in 2008 and upon retrieval of the loggers. The instrument drift was within the maximum specified by the manufacturer, and a linear drift correction was applied to each individual record over the period of deployment.

At the downstream end of both sites, dataloggers were distributed in two transects, perpendicular to the channel, to monitor a cross-section of the streambed (Figure 4.3). The remaining dataloggers were installed longitudinally along the channel bed at approximately evenly distributed spacing, in order to capture differences along the flow direction. Temperature was recorded hourly for the period of July 2008 to October 2012 in Fishtrap Creek, and from July 2008 to October 2011 in Bertrand Creek. In Bertrand Creek, the number of dataloggers decreased from 19 in 2008 and 2009, to three in 2010, and two in 2011 due to equipment loss (Figure 4.3). Note that for 2012, logger data were not available and, therefore, water temperature data from the USGS gauging station were used.
Figure 4.3. a) The distribution of instrumentation at the Bertrand Creek study site. The USGS gauging station is located adjacent to the international border approximately 35 m downstream of the study reach. b) The distribution of instrumentation at the Fishtrap Creek study site. The WSC gauging station is located adjacent to the international border at the downstream end of the study reach.
4.3.3. **Heat Budget**

The purpose of the simplified heat budgets calculated in this study is to examine the dominant potential drivers of the stream temperature at a broad scale. The heat budgets for Fishtrap and Bertrand Creeks were estimated using mean daily interface temperatures measured across the site, and daily climate data from the Abbotsford Airport. Surface heat budgets were calculated at a daily time step for the water-air interface to identify climatic factors influencing the stream temperature, and to indirectly isolate differences in the heat budget that may originate at the water-bed interface. The simplified heat budget was calculated based on the methods described in Sinokrot and Stefan (1993) and Xin and Kinouchi (2013). The total heat budget \( (H_T) \) is represented by the heat exchange at the air-water interface \( (H_A) \) and at the water-bed interface \( (H_B) \) (Sinokrot and Stefan 1993):

\[
H_T = H_A + H_B
\]  

(4.1)

If all the heat exchanges occur at the air-water interface, then \( H_T = H_A \). Any differences can be accounted for by exchanges at the water-bed interface \( (H_B) \). Without instrumentation at each interface, this simplified approach can be used to identify broad patterns and differences with the heat budgets.

The main components in the net heat exchange at the air-water interface \( (H_A) \) (Figure 4.4) following Sinokrot and Stefan (1993), Evans et al. (1998), and Xin and Kinouchi (2013) are:

\[
H_A = H_{is}(1-\alpha) - H_l - H_e - H_c
\]

(4.2)

where \( H_{is} \) is the incident solar radiation at the water surface, and \( \alpha \) is the albedo of the water surface (0.06). Incident solar radiation was estimated using the solar position and radiation calculator available from the State of Washington Department of Ecology (2014). The Bird Clear Sky model for direct radiation incident upon a horizontal surface was used, which is based on the latitude and elevation of each site (Bird and Hulstrom 1981). The albedo in Equation 4.2 \( (\alpha) \) follows the work by Xin and Kinouchi (2013), in which they applied an albedo of the water surface of 0.06. As Fishtrap Creek has limited
vegetated canopy shading, and Bertrand Creek has intermittent deciduous riparian shading, an intermediate albedo value of 0.06 for water bodies between 0° and 60° latitude, and for a leafy deciduous canopy was used (Ankstrom 1925; Henderson-Sellers and Wilson 1983; Betts and Ball 1997). Despite the differences in vegetation at both sites, the surface heat budget did not include a shade factor as no data were available to quantify the seasonal differences. This is a limitation of the approach that could be explored if more quantitative vegetation data were available.

**Figure 4.4.** A schematic of the total heat budget ($H_T$), with heat exchange across the water-air interface ($H_A$) acting across the grey surface representing the top of the stream flow. The heat exchange across the water-bed interface is $H_B$.

The net longwave radiation ($H_l$) is given by (Sinokrot and Stefan 1993):

$$H_l = \sigma (\varepsilon_w T_s^4 - \varepsilon_a T_a^4)$$

(4.3)

where $T_s$ is the mean daily stream temperature (degrees Kelvin) measured by the temperature loggers at the streambed interface across each site, $T_a$ is the mean daily air temperature (degrees Kelvin), $\varepsilon_w$ and $\varepsilon_a$ are the emissivities of the water surface (0.97) and atmosphere (Equation 4.4), respectively, and $\sigma$ is the Stefan-Bolzmann constant (4.9 x 1015 TJ m$^{-2}$ day$^{-1}$ K$^{-4}$) (Xin and Kinouchi 2013):
The net longwave radiation was calculated as a lumped parameter because data for the fraction of sunshine hours and cloud cover ratio were not available.

The evaporative heat flux \( (H_e) \) is calculated by Equation 4.5 (Sinokrot and Stefan 1993; Xin and Kinouchi 2013):

\[
H_e = \rho w \lambda Wftn(e_s - e_a)
\]

where \( \rho_w \) is the density of water \((\text{kg m}^{-3})\) determined from the mean steam temperature from Martin and McCutcheon (1998) in Equation 4.6, and \( \lambda \) is latent heat of vaporization of water \((\text{TJ kg}^{-1})\) given by Equation 4.7.

\[
\rho_w = 1000 \times \left[1 - \frac{T_s + 288.9414}{508929.2 + (T_s + 68.12963)} \times (T_s + 3.9862)^2 \right] \quad (4.6)
\]

\[
\lambda = (2499 - 2.36T_s) \times 10^{-9} \quad (4.7)
\]

where \( e_s \) is the saturated vapor pressure at the water surface temperature \((\text{hPa})\) (Equation 4.8), and \( e_a \) is the vapor pressure of the air \((\text{hPa})\) (Equation 4.9) and are calculated following Allen et al. (1998):

\[
e_s = 0.618 \exp \left(\frac{17.22 + T_{a,\text{max}}}{T_{a,\text{max}} + 273.3} \right) \quad (4.8)
\]

\[
e_a = \left(\frac{\text{RH}}{100}\right) \times e_s \quad (4.9)
\]

In Equation 4.10, the term \( Wftn \) is the wind speed function for wind velocity at a height \((z)\) above the water surface \((\text{m/s})\) (Gulliver and Stefan 1986; Sinokrot and Stefan 1993; Xin and Kinouchi 2013):

\[
Wftn = 9.34 \times 10^{-5} \times (\Delta \theta_e)^{1/3} + 8.528 \times 10^{-5} \times u_2 \quad (4.10)
\]
The wind speed $u_2$ from the climate station is converted to the wind speed at a height of 2 m (Equation 4.11) following the FAO Penman-Monteith method (Allen et al. 1998).

$$u_2 = u_h \frac{4.87}{\ln (67.8 \ h - 5.42)}$$  \hspace{1cm} (4.11)

where $u_h$ is the wind speed measured at the climate station (m s$^{-1}$) at a height $h$, which for this study is 10 m.

The wind speed function calculation also relies on the calculation of the difference in the virtual temperature ($\Delta \Theta_v$), which is a function of air and water temperatures:

$$\Delta \Theta_v = T_s \left(1 + \frac{0.378e_l}{P_a}\right) - T_a \left(1 + \frac{0.378e_a}{P_a}\right)$$  \hspace{1cm} (4.12)

where $P_a$ is the atmospheric pressure recorded at the climate station.

The sensitive heat flux ($H_c$) is calculated following Sinokrot and Stefan (1993) and Xin and Kinouchi (2013):

$$H_c = 0.61 \rho_w \lambda \frac{P_a}{1000} (T_s - T_a)$$  \hspace{1cm} (4.13)

At the study sites, there are no tributaries contributing to the heat budget, and heat fluxes due to precipitation, biological, and chemical fluxes are assumed to be negligible (Evans et al. 1998). Heat fluxes at the water-bed interface ($H_B$) may comprise bed conduction, advection, and fluid friction. In the absence of direct measurement of these individual parameters, these various heat flux components are lumped into a single water-bed interface heat flux component $H_B$.

### 4.3.4. Air – Water Temperature Relationship

Stream temperature responds to changing atmospheric conditions and solar radiation inputs. Air and water temperatures are influenced by solar radiation and
therefore in the absence of detailed climate data, air temperature is often used as a surrogate for solar radiation (Smith 1981; Mohseni and Stefan 1999; Johnson 2003). Due to thermal inertia, the temperature response at the bed interface is dampened and delayed relative to air temperature (Edinger et al. 1968; Sinokrot and Stefan 1993; Stefan and Preud'homme 1993; Bogan et al. 2003). To evaluate heat transfer from the air-water surface to the water-streambed interface, a response time calculation is required (Sinokrot and Stefan 1993; Stefan and Preud'homme 1993). The time constant, \( \delta \), for the stream temperature response follows Sinokrot and Stefan (1993):

\[
\delta = K (\rho c_p d)^{-1}
\]  

(4.14)

where \( K \) is the net surface heat transfer coefficient (Wm\(^{-2}\) °C\(^{-1}\)) - a value of 30 was used based on Sinokrot and Stefan (1993) - \( \rho c_p \) is the specific heat (kcal m\(^{-3}\) °C\(^{-1}\)), and the mean water depth across the site \( d \) (m). To apply to the daily mean temperatures, the time constant is rounded to the nearest whole day. In this study, the time constant was calculated using the mean depth of each stream over the five summers assuming a uniformly mixed water column. Using Equation 4.14, the time constant to relate the air temperature to the stream temperature was 30 hours for Fishtrap Creek, and 14 hours for Bertrand Creek. To facilitate calculations, the lag time for air temperature for both study sites was rounded to 1 day (24 hours).

### 4.3.5. Flux Measurements

Groundwater flux into each stream was also measured. In an attempt to reduce uncertainty, three methods were used at each site: 1) one set of nested piezometers to measure the vertical hydraulic gradient, 2) a seepage meter to measure seepage flux across the channel bed, and 3) stream gauging along two transects to measure stream discharge both upstream and downstream to estimate inputs of groundwater over a length of stream (Mellina et al. 2002; Moore et al. 2005; Rosenberry 2008). Field measurement locations for in-stream nested piezometers, seepage meters, and stream discharge measurement transects are shown in Figure 4.3.
**In-stream Piezometers**

The in-stream piezometer water level measurements provide an estimate of vertical hydraulic gradient between the deep and shallow piezometers. Shallow in-stream piezometers were installed as nested pairs in both Fishtrap and Bertrand Creeks. The piezometers were constructed of stainless steel with solid tips, and installed to depths of approximately 1.0 m and 0.5 m into the streambed adjacent to the channel bank. In the small streams, the position of the piezometers near the bank was considered appropriate to capture the vertical hydraulic gradient as well as provide some protection from high flow events and vandalism. Measurements were made approximately every 30 minutes, with between 6 to 25 measurements per day. To estimate the flux, estimates of vertical hydraulic conductivity ($K_z$) of the surficial geology units at each study site were used (values from Scibek and Allen 2005). $K_z$ for Sumas Drift in the vicinity of Fishtrap Creek is 0.56 m/d, and for the Ft. Langley Formation stoney clays underlying Bertrand Creek, $K_z$ is estimated as 0.20 m/d. Uncertainty in the piezometer readings is ± 0.035 m from the combined uncertainty in the installation depths and the individual measurements. The use of piezometers has the limitation that measurements are localized and represent the vertical flux at the point of the nested pair, and may not represent the site wide conditions.

**Seepage Meters**

Seepage meters can quantify fluid exchange across the sediment-water interface (Alexander and Caissie 2003; Kalbus et al. 2006; Essaid et al. 2008; Rosenberry 2008). The seepage meter consists of a bottomless cylinder set into the channel sediments and vented to a light-weight bag (Kalbus et al. 2006; Rosenberry 2008). The groundwater seeps into the surface water and enters the seepage meter, filling the bag, and the volume and rate of the water entering the bag are used to calculate the flux rate. The seepage meter in this study was modified for use in a flowing stream by giving it a low profile along the channel bed, and using a stilling box to protect the seepage collection bag from horizontal flow (Rosenberry 2008). Seepage meter measurements have limitations because the measurements are localized and represent point estimates of flux, while flux varies across the stream channel in space and time (Cey et al. 1998; Krause et al. 2007). Also the use of a seepage meter assumes that all flux into the meter
is vertical, and that there is no component of hyporheic circulation entering the seepage meter. Despite this, the seepage meter has been used effectively in many studies to measure the flux, which normalizes the area covered by the meter, allowing the values recorded to be compared to other methods (Rosenberry and LaBaugh 2008).

The seepage meter was installed in approximately the same location in the stream in each of the three summer periods. Seepage measurement locations were selected to be representative of the channel section, and suitable for instrument installation. In Fishtrap Creek, the seepage meter was set approximately 8-10 cm into the fine grained channel sediments; however, due to the low stream stage in Bertrand Creek, a lower profile seepage meter was used, and it was inserted into the sands and gravels of the streambed. Seepage was monitored for two to three days each summer in 2009, 2010, and 2011. Following an equilibration period of usually 24 hours after installation, the seepage flow into the collection bag was measured approximately every 30 minutes, with measurements repeated between 8 and 24 times per day. Uncertainty in the seepage meter readings is estimated at ± 2 cm/day. Uncertainty was calculated as the sum of the uncertainty in the measured diameter of the seepage meter and uncertainty in volume measurements over time.

**Stream Discharge**

Stream discharge transects can be used to estimate groundwater flux along the length of a channel by measuring the volume lost or gained over a distance. If there are no tributaries, or withdrawals along that length, then changes in stream discharge will be the result of groundwater fluxes. Stream discharge measurements were made using a Scientific Instruments 1250 Mini “pygmy” current meter. The pygmy meter is specially designed for use in small, shallow streams; the range of operation is approximately 0.08 to 0.9 m/s. The accuracy of the pygmy meter is given as generally 3.5%, but may approach 10% near the extent of its operating range, and therefore, for this study, 6.5% was used (Nelson et al. 2010). The streams were sampled using the 0.6 depth velocity-area method, in which the width, depth, and velocity are measured across ten vertical panels and used to calculate the stream discharge. The measurement uncertainty from the width and depth measurements for the area calculation across the section of channel is 0.05 m, which is added to the velocity uncertainty. The stream discharge was
measured at upstream and downstream locations at each site, relative to the seepage meter, for comparison with the seepage volumes. Diurnal fluctuations in stream discharge and uncertainty in measurements under low flow conditions potentially increase the uncertainty in the discharge measurements over the period of a day. Each velocity transect measurement was completed over approximately 30 minutes, and the velocity measurements were repeated a minimum of two times at both the upstream and downstream transects to obtain a mean value and to increase confidence in the readings (Berg and Allen 2007).

4.4. Results

4.4.1. Climate, Stream Discharge and Groundwater Level

Figure 4.5 shows the mean daily precipitation (top panel), stream discharge (middle panel) and groundwater elevation (bottom panel) for the five year period (2008-2012). Annually, precipitation ranged from 1231.6 mm/year (2008) to 1536.1 mm/year (2012) with an average of 1422.5 ± 119.8 mm/year (Table 1). Over the summers, the mean daily precipitation was 1.8 mm, with an average summer total of 168 mm, and a range of 63 mm (2013) to 252 mm (2010) (Table 1).

The mean annual stream discharge in Fishtrap Creek (0.9 ± 0.9 m/s) was similar to the Bertrand Creek discharge (1.19 ± 2.2 m³/s) (Figure 4.5 – middle panel; Table 1). Although the annual stream discharge patterns are similar, Fishtrap Creek has fewer extremes during both the high and low flow periods, as shown by the maximum and minimum values relative to the mean (Figure 4.5 – middle panel). Bertrand Creek stream discharge is flashier; more high flow events are observed outside the summer periods. During the summers, the mean daily stream discharge in Fishtrap Creek was 0.20 ± 0.3 m³/s, while in Bertrand Creek it was 0.1 ± 0.2 m³/s (Table 1).

The mean annual groundwater elevation in ABB01 was 43.8 ± 0.4 masl (ground surface elevation is 45.7 masl), and 80.6 ± 0.3 masl (ground surface elevation is 98.9 masl) in Obs. Well #301 (Table 1). During the summers, the groundwater elevation at ABB01 was 43.4 ± 0.2 masl, and at Obs. Well #301 it was 80.6 ± 0.3 masl (Table 4.1).
Overall, the groundwater levels mimic the stream discharge; however, there is a greater lag at Obs. Well #301 (Figure 4.5 – bottom panel).
Table 4.1. Summary of mean daily data for Fishtrap and Bertrand Creeks for the five summer periods and annually (2008-2012). The summer and annual means for each parameter are shown with the standard deviations.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Details</th>
<th>Summer Period*</th>
<th>Summer mean</th>
<th>Annual mean</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>2008 2009 2010 2011 2012</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Air temperature (°C)</td>
<td>Fishtrap Creek</td>
<td>16.7 18 17.4 17.3 17.1</td>
<td>17.3±2.9</td>
<td>10.2±6.3</td>
</tr>
<tr>
<td></td>
<td>Bertrand Creek</td>
<td>15.7 17 16.2 16.1 16.8</td>
<td>16.4±2.7</td>
<td>10.2±4.9</td>
</tr>
<tr>
<td>Precipitation (mm)</td>
<td>Mean daily</td>
<td>2 1.7 2.7 2 0.7</td>
<td>1.8±5.5</td>
<td>3.9±7.7</td>
</tr>
<tr>
<td></td>
<td>Summer total</td>
<td>186.5 155 252.4 182.3 63.1</td>
<td>167.9±68.6</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Annual total</td>
<td>1231.6 1387.3 1495 1462.5 1536.1</td>
<td>1422.5</td>
<td>119.84</td>
</tr>
<tr>
<td>Stream discharge (m³/s)</td>
<td>Fishtrap Creek</td>
<td>0.18 0.09 0.23 0.28 N/A</td>
<td>0.2±0.3</td>
<td>0.9±0.9</td>
</tr>
<tr>
<td></td>
<td>Bertrand Creek</td>
<td>0.1 0.05 0.18 0.1 0.11</td>
<td>0.1±0.2</td>
<td>1.1±2.2</td>
</tr>
<tr>
<td>Streambed temperature (°C)</td>
<td>Fishtrap Creek</td>
<td>13.8 14.6 14.1 13.3 13</td>
<td>13.8±1.3</td>
<td>10.3±0.4</td>
</tr>
<tr>
<td></td>
<td>Bertrand Creek</td>
<td>15 15.6 16 14.4 15.9</td>
<td>15.6±1.8</td>
<td>10.5±0.6</td>
</tr>
<tr>
<td>Groundwater level (elevation - masl)</td>
<td>Env Can. ABB01</td>
<td>N/A 43.2 43.4 43.5 43.4</td>
<td>43.4±0.2</td>
<td>43.8±0.4</td>
</tr>
<tr>
<td></td>
<td>BC MoE Obs Well 301</td>
<td>80.6 80.5 80.5 80.7 80.9</td>
<td>80.6±0.3</td>
<td>81.3±0.7</td>
</tr>
<tr>
<td>Groundwater temperature (°C)</td>
<td>Env Can. ABB01</td>
<td>N/A 10.6 10.9 10.5 10.7</td>
<td>10.7±0.2</td>
<td>10.9±0.4</td>
</tr>
</tbody>
</table>

N/A Not available
1 Datalogger n=3
2 Datalogger n=2
3 Stream temperature obtained from the USGS gauging station.
* Summer period is July - September
Figure 4.5. The mean, maximum, and minimum daily precipitation (top panel), stream discharge (middle panel), and groundwater elevation (bottom panel) over the 5 year period (2008-2012) for Fishtrap Creek (left column) and Bertrand Creek (right columns).

4.4.2. Stream – Bed Interface Temperature

Figure 4.6 shows the annual mean daily air temperature at Fishtrap (represented by the Abbotsford Airport) and at Bertrand Creek (represented by the USGS gauging station) (top panel). Over the five summers, the mean daily air temperature at Fishtrap
Creek was 17.3 ± 2.9°C, while at Bertrand Creek it was 16.4 ± 2.7°C (Table 1). Thus, air temperatures at Fishtrap are roughly 1°C higher, on average, compared to Bertrand Creek. The lower temperature at Bertrand Creek may be due to the stream setting where the sensor is located, which is characterized by more riparian vegetation and shading relative to the open field setting at the Abbotsford Airport and Fishtrap Creek.

In Fishtrap Creek, the mean interface temperature over the five summer periods (Figure 4.6 – middle panel) was 13.8 ± 1.3°C, while in Bertrand Creek it was 15.6 ± 1.8°C, a 1.8°C difference. It is noted that there was sub-reach scale variability in the interface temperatures at both sites. Here, the mean daily interface temperature across the site is discussed so as to allow a comparison between the two sites. The spatial (along the reach) and temporal (consecutive summer periods) variations in Fishtrap are explored elsewhere (Middleton, Whitfield, and Allen 2015). In addition to being lower, the interface temperatures in Fishtrap Creek have a smaller annual range than in Bertrand Creek (Figure 4.6 - middle panel).

The annual mean daily groundwater temperature at ABB01 (representing both sites) is 10.9 ± 0.4°C (Table 1). The mean annual groundwater temperature is close to the mean annual air temperature (10.2°C). Groundwater temperatures are also relatively constant inter-annually, as shown by the maximum and minimum curves in Figure 4.6. Over the five summers, the mean groundwater temperature was 10.7 ± 0.2°C.
4.4.3. Heat Budget and Regression Analysis

The simple daily surface heat budgets for Fishtrap and Bertrand Creeks were computed for the five year study period, and an example for 2009 is shown in Figure 4.7.
The surface heat budget for both streams indicates that, annually, incoming solar radiation dominates the net radiation. At a daily time scale, evaporative heat flux events are noticeable. Precipitation events do not appear to be an important component, either during the event or during the absence of precipitation.

Figure 4.7. Daily surface heat budget and precipitation for 2009 for a) Fishtrap Creek, and b) Bertrand Creek.
The surface heat budget indicates that solar radiation dominates the heat flux at the air-water interface; however, the interface temperature response to solar radiation is delayed and dampened relative to air temperature due to the thermal inertia of water (Sinokrot and Stefan 1993; Bogan et al. 2003). The linear relationship between the air temperature and the one-day (24 hour) lagged interface temperature was used to determine the variance in interface temperature that is explained by air temperature. The linear relationship is plotted in Figure 4.8, for all days with non-negative temperatures over the five year period. The one-day lagged air temperature explains more variance in interface temperature in Bertrand Creek ($R^2 = 0.92$) than in Fishtrap Creek ($R^2 = 0.86$). The intercept of the linear regression line is also higher for Fishtrap Creek (5.15°C) than for Bertrand Creek (2.70°C). The strength of the regression shows that that incoming solar radiation inputs dominate the heat exchanges in the stream, but indicates there are differences between the two streams.

Using observations from the summer periods only (Figure 4.9), the slope of the regression lines for both streams is lower than for the annual data, indicating that the interface temperatures increase at a lower rate relative to air temperature during the summer. The conductive heat flux across the streambed depends on the amplitude of the daily water temperature cycle, and because the amplitude is greater during the summer, streambed conductance would be more prominent, leading to a lower regression slope. However, the magnitude of the decrease in the regression slope during the summer as compared to annually was not the same for each stream. Langan et al. (2001) showed that the relationship between air and water temperatures is strongest during the summer when radiation inputs are highest and low flow occurrences are most common. From the regression relationship, the one-day lagged air temperature explains only 31% of the variance in interface temperature in Fishtrap Creek, while in Bertrand Creek it explains 76% of the variance. In Fishtrap Creek, factors other than air temperature are contributing to the remaining 69% of the variance in the summer interface temperatures. At temperatures above 20°C evaporative cooling can alter the linear relationship between air and water temperature (Mohseni and Stefan 1999; Kelleher et al. 2012). However, evaporative cooling is not considered significant as stream temperatures during the summer generally remain below 20°C. Furthermore,
The surface heat budget indicates that the evaporative heat flux remains relatively constant over the summer (Figure 4.7).

**Figure 4.8.** Mean daily one-day lagged air temperature and interface temperature for all non-negative air temperature days from 2008 to 2012 for a) Fishtrap Creek and, b) Bertrand Creek.
Figure 4.9. Mean daily one-day lagged air temperature and interface temperature for the summer periods (July through September) from 2008 to 2012 for a) Fishtrap Creek, and b) Bertrand Creek.

Figure 4.10 compares the interface and air temperature for both sites, for both the annual and summer periods. The slope and intercept of the linear regression has been shown to be related to the interaction between streams and the groundwater inputs (Caissie 2006). Caissie (2006) reports that streams not dominated by groundwater inputs generally have steeper slopes than groundwater dominated streams, and comparisons of the regression lines for different streams can be indicative of relative
contributions from groundwater. The regression line for Fishtrap Creek has a lower slope and higher intercept than Bertrand Creek, both for the annual data and the summer period (Figure 4.10). The slopes for both streams for both summer periods decrease during the summer, indicating that interface temperatures in the summer are moderated, relative to air temperature. The comparison in Figure 4.10 clearly shows that this moderating influence is stronger in Fishtrap Creek during both comparison periods.

![Figure 4.10. Comparison of the linear regressions Fishtrap and Bertrand Creek for all non-negative temperature days (solid lines), and for summer period data only (dashed lines).]

4.4.4. Flux Measurements

The hydraulic gradients measured in the in-stream piezometers at both sites varied during the day, between consecutive days, and between the summer periods. Table 2 reports the vertical hydraulic gradients and the calculated flux, both as mean daily values. A positive value for the flux (negative hydraulic gradient) indicates an upward flow. In Fishtrap Creek, daily flux was positive for all but two days, and on these two days, the hydraulic gradients were very low. Over the five years, the average flux measured by the piezometers in Fishtrap was $11.4 \pm 9.5$ cm/day. In Bertrand Creek, the hydraulic gradients were very low and the flux direction variable; therefore, the results at
this site are highly uncertain. The average flux over the five years was -0.12 ± 0.88 cm/day.

Seepage measured with the seepage meters similarly fluctuated throughout the day, between consecutive days, and between the summer periods at both sites. The final values reported in Table 2 are the mean daily seepage flux; the reported values include a resistance correction factor of 1.05, following Rosenberry and LaBaugh (2008). The resistance correction factor is applied to compensate for resistance to flow within the meter and frictional flow losses that lead to underestimation of the flux rate. The seepage values for both streams are positive, which indicate an upward flux of groundwater into the stream. In Fishtrap Creek, the mean daily upward flux over the five year period was 23.5 ± 13.6 cm/day, and this seepage rate is of the same order of magnitude as the flux measured by the piezometers. In Bertrand Creek, the seepage flux was 25.3 ± 5.3 cm/day.

The seepage increased in Fishtrap Creek over the summer measurement period in both 2009 and 2010. In Bertrand Creek, the seepage was more variable, with a decrease in seepage over the period of measurement in 2009 and 2010, and an increase in 2011. However, with only a maximum of three sampling days per summer, it is not known if these trends are significant.

The stream discharge transects at both Fishtrap and Bertrand Creeks showed small volume decreases per length of stream, but all the values are within the range of uncertainty (Table 4.2) so it is unclear whether indeed there is a reach scale loss of stream water, which would be inconsistent with the flux and seepage measurements. As a result, the volume lost or gained between the stream discharge transects could not be determined with any confidence.
Table 4.2. Summary of mean daily field data from the seepage meters, in-stream piezometers, and discharge transects collected at Fishtrap and Bertrand Creeks for the five summer periods (2008-2012). The number of measurements is shown in brackets beside each daily value.

<table>
<thead>
<tr>
<th>Date</th>
<th>Vertical hydraulic gradient</th>
<th>Piezometer flux (cm/day)</th>
<th>Seepage flux (cm/day)</th>
<th>Change in downstream discharge (m³/s/m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>7/16/2009</td>
<td>-0.26±7E-3 (9)</td>
<td>14.49</td>
<td>11.6±2.6 (8)</td>
<td>-1.5E-3 ±6.8E-3</td>
</tr>
<tr>
<td>7/20/2009</td>
<td>-0.34±0.1 (25)</td>
<td>19.24</td>
<td>22.4±4.7 (24)</td>
<td></td>
</tr>
<tr>
<td>8/6/2009</td>
<td>-0.23±4E-2 (15)</td>
<td>13.02</td>
<td>42.5±20.6 (14)</td>
<td></td>
</tr>
<tr>
<td>8/16/2010</td>
<td>0.02±8E-3 (6)</td>
<td>-1.13</td>
<td>17.6±3.5 (5)</td>
<td></td>
</tr>
<tr>
<td>8/17/2010</td>
<td>0.03±2E-2 (16)</td>
<td>-1.70</td>
<td>17.3±4.6 (17)</td>
<td>-9.1E-4±8.2E-3</td>
</tr>
<tr>
<td>8/25/2010</td>
<td>-0.01±0.2 (12)</td>
<td>0.57</td>
<td>50.4±13.0 (12)</td>
<td></td>
</tr>
<tr>
<td>8/11/2011</td>
<td>-0.29±1E-2 (13)</td>
<td>16.41</td>
<td>20.5±9.1 (14)</td>
<td></td>
</tr>
<tr>
<td>8/12/2011</td>
<td>-0.37±3E-2 (19)</td>
<td>20.94</td>
<td>14.1±4.2 (18)</td>
<td>-3.0E-3±8.2E-3</td>
</tr>
<tr>
<td>8/17/2011</td>
<td>-0.36±5E-3 (10)</td>
<td>20.38</td>
<td>15.0±4.1 (11)</td>
<td></td>
</tr>
<tr>
<td>Average</td>
<td>0.20±0.2</td>
<td>11.36±9.46</td>
<td>23.5±13.6</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Date</th>
<th>Vertical hydraulic gradient</th>
<th>Piezometer flux (cm/day)</th>
<th>Seepage flux (cm/day)</th>
<th>Change in downstream discharge (m³/s/m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>7/17/2009</td>
<td>-E-3±4E-2 (14)</td>
<td>-0.02</td>
<td>25.0±9.9 (13)</td>
<td>-6.6E-4±3.2E-3</td>
</tr>
<tr>
<td>8/5/2009</td>
<td>-0.02±5E-3 (15)</td>
<td>0.40</td>
<td>18.1±6.4 (13)</td>
<td></td>
</tr>
<tr>
<td>8/18/2010</td>
<td>0.09±1E-2 (12)</td>
<td>-1.78</td>
<td>23.9±8.0 (15)</td>
<td>-2.9E-4±7.3E-3</td>
</tr>
<tr>
<td>8/24/2010</td>
<td>0.04±2E-3 (14)</td>
<td>-0.79</td>
<td>25.5±10.3 (13)</td>
<td></td>
</tr>
<tr>
<td>8/13/2011</td>
<td>-0.01± 7E-3(14)</td>
<td>0.20</td>
<td>21.9±7.9 (17)</td>
<td></td>
</tr>
<tr>
<td>8/14/2011</td>
<td>-0.02± 5E-3(14)</td>
<td>0.40</td>
<td>27.6±10.3 (14)</td>
<td></td>
</tr>
<tr>
<td>8/15/2011</td>
<td>-0.04± 3E-3(18)</td>
<td>0.79</td>
<td>35.3±15.3 (15)</td>
<td></td>
</tr>
<tr>
<td>Average</td>
<td>-0.01±.04</td>
<td>-0.12±0.88</td>
<td>25.3±5.3</td>
<td></td>
</tr>
</tbody>
</table>

4.5. Discussion

Overall, the annual mean daily stream discharge and stream temperature at Fishtrap Creek are similar to Bertrand Creek; however; the main differences occur at the extremes. Bertrand Creek discharge and temperatures have a larger annual range between the peaks and the lows. The stream temperatures in Fishtrap Creek are attenuated relative to the air temperature, and the minimum temperatures remain closer to the mean temperatures than in Bertrand Creek.
The surface heat budgets for the two streams (Figure 4.7) show that Fishtrap Creek has higher net radiation than Bertrand Creek throughout the year, peaking during the summer, and more variability over the annual period. This is a reflection of the different air and stream temperatures datasets for each site, which are used to calculate the evaporative, convective, and longwave radiation. Although the net radiation is higher and more variable in Fishtrap Creek relative to Bertrand Creek, during the summer periods, stream temperatures in Fishtrap Creek are lower (Figure 4.5), and are less variable than in Bertrand Creek. This suggests that there is a heat sink in Fishtrap Creek that cannot be accounted for in the heat budget at the air-water interface. The cooler summer interface temperatures and higher stream discharge in Fishtrap Creek (Figures 4.5 and 4.6), relative to Bertrand Creek, suggest a greater contribution of groundwater to the summer stream flow in Fishtrap Creek. The comparison of the linear regression of the one-day lagged air temperature and stream temperature (Figure 4.10) indicates that Fishtrap Creek receives greater groundwater contribution throughout the year, relative to Bertrand Creek, and that the relative proportion of groundwater to the stream flow increases during the summer periods. The influx of groundwater at a relatively constant temperature, near 11°C over the summer, acts to attenuate interface temperature variations and moderates water temperature fluctuations relative to changing atmospheric conditions.

The field measurement data from seepage meters and in-stream piezometers loosely support the surface heat budget and interface temperature measurements. In Fishtrap Creek, the seepage meter and piezometers both indicated an upward flux of water into the stream, and the results are of the same order of magnitude over the five year period, 11 and 23 cm/day, respectively. In Bertrand Creek, the hydraulic gradient was very low. This, in combination with the presence of lower Kz material around Bertrand Creek, leads to a very low vertical groundwater flux compared to Fishtrap Creek. The seepage meter and piezometer results at Bertrand Creek differed by an order of magnitude, -0.12 cm/day and 25 cm/day, respectively. The higher seepage values can be explained by the placement of the low profile seepage meter. The seepage measured by the seepage meter may have been derived from circulating hyporheic flows in the sands and gravels, or perhaps from leakage through the sands and gravels into the seepage meter. Therefore, it is likely that the seepage rates are
over-estimated for Bertrand Creek. Unfortunately, the stream discharge measurements at both sites could not be used due to measurement difficulties at low flow. In Fishtrap Creek, slow stream velocity and in-stream vegetation impeded measurements, and at Bertrand Creek, shallow water depth and slow velocity were factors impeding accurate stream discharge measurements. These factors have been shown to increase the level of uncertainty and variability in stream discharge measurements over similar stream lengths (Berg and Allen 2007).

Considering the heat balance results, the interface temperatures, and the in-stream measurements at both sites, groundwater flux into Fishtrap Creek is higher than at Bertrand Creek. The main difference between these two sites is the surficial geology. The lower reaches of Fishtrap Creek are in the higher permeability coarse sands of the Sumas Drift, while the lower reaches of Bertrand Creek are in dominantly fine-grained Ft. Langley Formation. Topographically, the streams reaches are similar, but a much lower vertical hydraulic gradient was measured in Bertrand Creek compared to Fishtrap Creek, suggesting that there is limited potential for vertical movement of groundwater into the stream at Bertrand Creek. Combined with the low vertical Kz at Bertrand Creek, the overall groundwater flux is much reduced. Starzyk (2012) found negligible change in stream discharge in the same reach of Bertrand Creek, and attributed this to a large component of groundwater flowing horizontally under this portion of the stream due to topographic controls. Starzyk (2012) also concluded that in the absence of strongly upwelling groundwater, hyporheic flow creates local-scale circulation along the streambed. This suggests that regional groundwater flow patterns may play an important role in determining local exchanges between groundwater and surface water. Future work should focus on incorporating knowledge of the regional groundwater flow in order to assess the local connectivity between groundwater and surface water.

4.6. Conclusions

In many streams, the relative proportion of groundwater contributing to stream flow is generally highest during the summer low flow period, and groundwater sustains the flow. Groundwater also has a fairly stable temperature year-round so it can buffer the effects of summer heating on stream temperatures. For these reasons, it is important to
understand controls on groundwater influxes to streams, particularly during the summer when critical thresholds for streamflow and stream temperature may be reached. However, data on groundwater-surface water interactions are often sparse and application of field methods can be challenging during periods of low flow. This study used a variety of methods to evaluate the relative contributions of groundwater to two groundwater-fed streams.

While a simplified surface heat budget was used due to lack of detailed energy flux data, it was useful for understanding heat exchanges at the field sites. Specifically, the surface heat budget indicates that, annually, incoming solar radiation dominates the net radiation. At a daily time scale, evaporative heat flux events are noticeable, but precipitation events do not appear to be an important component, either during the event or during the absence of precipitation. A regression analysis between interface temperature and air temperature showed that that incoming solar radiation inputs dominate the heat exchanges in the stream, but indicates there are differences between the two streams annually and during the summer. The strength of the regression in Fishtrap Creek is particularly poor in the summer, when factors other than air temperature contribute 69% of the variance in the summer interface temperatures. These results suggest that there is a heat sink at the interface in Fishtrap Creek, which is interpreted as a groundwater influx.

In-stream piezometers, supported by seepage meter measurements, were most effective at Fishtrap Creek for evaluating the groundwater influx. At Bertrand Creek, the combined effect of a low vertical hydraulic gradient and possible interference of hyporheic flow with the seepage meters, led to inconclusive flux measurements. Manual stream discharge measurements down the stream reaches were found to be inconclusive due to the magnitude of uncertainty in the stream discharge during the low flow measurements.

What is particularly interesting is that while the two streams share similar climate and topography, the geological substrate is different. The more permeable substrate surrounding Fishtrap Creek, in combination with steeper vertical gradient, result in a
greater groundwater influx. This greater influx will act to sustain streamflow and buffer increased stream temperatures associated with low summer flows.
Chapter 5.

Independent Component Analysis of Local-Scale Temporal Variability in Sediment – Water Interface Temperature

5.1. Introduction

Stream temperature is a key parameter for assessing water quality and the overall health of aquatic ecosystems (Caissie 1991; Winter et al. 1998; Poole and Berman 2001; Hatch et al. 2006; Hannah et al. 2008; Cunjak et al. 2013; Rau et al. 2014). The temperature of a stream influences biological and chemical processes, the life-histories of aquatic species, and community processes and structure (Power et al. 1999; Alexander and Caissie 2003; Benyahya et al. 2007; Velasco-Cruz et al. 2012). Stream temperature, however, has a complex response to a variety of processes, particularly interactions between the water and the environment through exchanges across the water surface and the sediment-water interface (Johnson and Jones 2000; Hannah et al. 2004; Moore et al. 2005; Caissie 2006).

Most variations in stream water temperature (e.g., diel, daily and seasonal) occur as the result of heating and cooling of the river by outside sources, which are strongly influenced by meteorological and geophysical conditions (e.g., Webb and Zhang, 1997; Evans et al., 1998; Bogan et al., 2003; Moore et al., 2005). As such, regression and stochastic models have been used to predict the thermal regime of a surface water body using air temperature as a predictor (Stefan and Preud’homme 1993; Mohseni et al. 1998; Benyaha et al. 2007). Deterministic models have also been used to quantify heat fluxes across the sediment - water interface (e.g. Caissie et al., 2014).
Water exchanges between the stream and the groundwater system are of particular importance. From a thermal perspective, groundwater flux can be considered to have both diffuse and localized effects. Groundwater temperatures are relatively constant throughout the year, and groundwater influxes (whether diffuse or localized) buffer the temperature fluctuations in the stream (Alexander and Caissie, 2003; Constantz, 2008; Brewer, 2013). Localized groundwater influxes (e.g., seeps, springs, alcoves, and hyporheic discharge) create thermal anomalies that can provide microhabitats (thermal refugia) for cold-water fish and other aquatic species (Brunke and Gonser 1997; Alexander and Caissie 2003; Brewer 2013; Briggs et al. 2013; Kurylyk et al. 2014). These anomalies can have a temperature difference of only 1-2°C, and still be biologically important (Caissie 2006; Velasco-Cruz et al. 2012). Summer low flow periods are particularly critical for aquatic health because stream flow is at a minimum and the stream temperatures typically reach the annual maximum (Fleming et al. 2007; Brewer 2013; Moore et al. 2013). Therefore, during summer in the Pacific Northwest, when precipitation inputs are minimal, the contributions of groundwater become increasingly important to maintain suitable flow and thermal conditions for aquatic life (Smakhtin 2001; Hatch et al. 2006; Mayer 2012; Briggs et al. 2013; Kurylyk et al. 2014).

Temperatures measured within the stream water column, the streambed, and at the sediment-water interface have been identified as valuable tracers for understanding groundwater-surface water interactions, which often vary both spatially and temporally (Evans and Petts 1997; Evans et al. 1998; Conant 2004; Anderson 2005; Hatch et al. 2006; Constantz 2008; Rau et al. 2014). Variations in the sediment-water interface temperature, in particular, can be attributed to differences in exchanges between the stream water and groundwater (Krause et al. 2012). For example, streams with a connection to groundwater can seasonally become gaining streams during the low flow period (Silliman and Booth 1993; Winter et al. 1998; Sophocleous 2007; Constantz 2008). Understanding of the spatial variability in groundwater contributions to streamflow may be gained by mapping streambed temperatures (e.g. Conant, 2004), while time series analysis can be used to determine fluxes between streams and groundwater (e.g., Hatch et al., 2006; Rau et al. 2010; Irvine et al., 2015). The influence of multi-dimensional flows (e.g. hyporheic, diffuse groundwater discharge, etc.) and the high degree of spatial heterogeneity in streambed and aquifer hydraulic properties strongly
influences the temperatures (and fluxes), making analysis and interpretation challenging (Irvine et al., 2015). Thus, there is value in examining temperature information from multiple time series.

This chapter examines temporal variability in sediment-water interface temperature recorded at the reach scale over four summer periods (July through September) in a coastal stream. Independent Component Analysis (ICA) is employed as a statistical method to separate the observed signals into the independent components in order to compare how the signals differed between the four years. Independent Component Analysis (ICA) has many applications for signal separation. Classic applications of ICA include audio signal processing, separation of biomedical signals such as electrocardiogram components, and image processing (Funaro et al. 2003; Mitiandoudis and Davies 2003; Ungureanu et al. 2004). More recent applications of ICA have extended the method into climate analysis, modeling, forecasting, and hydrologic time series analysis (Aires et al. 2000; Westra et al. 2007; Moradkhani and Meier 2010). Much of the ICA literature related to climate and environment research has focused on problems at spatial scales ranging from global (e.g., global climate models) to basin and watershed scales. The temporal scales considered in these studies employ periods of record that are appropriate for the spatial scale; for example, decadal oscillations, and monthly or seasonal variations. Here ICA is used to examine time series of daily sediment-water interface temperatures observed at a spatial scale of metres.

5.2. Study Area

The study site is a reach of Fishtrap Creek, located in the Lower Fraser Valley of southwestern British Columbia (BC) (Figure 5.1). Fishtrap Creek watershed is situated within the regional Abbotsford-Sumas aquifer. This particular reach was selected because it is a gaining reach during the summer (Johanson 1988; Berg and Allen 2007). There is no forest canopy or overhanging vegetation, only seasonal grasses present as in-stream vegetation, so the direct and indirect effects of shading on stream temperature are avoided (Middleton, Allen, and Whitfield 2015). The reach is well suited for making comparisons of temperature patterns between years because the channel characteristics, such as summer water depth, bed material, and stream vegetation
cover, are stable. This location has other data sources, including a Water Survey of Canada gauging station at the downstream end of the site (Fishtrap Creek at International Boundary 08MH153), and a climate station at the nearby Abbotsford International Airport (Climate ID 1100030).

Figure 5.1. The location of the Fishtrap Creek study site in British Columbia, Canada within the Abbotsford-Sumas aquifer which spans the international border (Canada-USA). Abbotsford International Airport is the location of the Environment Canada climate station.

The climate of Fishtrap Creek watershed is Maritime and dominated by moderate temperatures with high annual precipitation rates. The temperatures throughout the year are moderated by the close proximity to the Pacific Ocean. The annual average precipitation is 1500 mm/yr, with little snow; less than 100 mm as water. Approximately 70% of precipitation falls in the period between October and May, and only 6% of the precipitation falls in July and August (Wernick et al. 1998; Zebarth et al. 1998; Berka et al. 2001; Environment Canada 2007).

Fishtrap Creek watershed is ~47 km² and originates at relatively low elevation and relief (slightly above mean sea level). The flow regime is driven by rainfall and interaction with the groundwater (Johanson 1988; Berg and Allen 2007). The flow regime is pluvial and runoff mimics the timing of the precipitation, with a time lag of only a few days. The lowest streamflows generally occur during August (Berg and Allen 2007).
During this period, groundwater sustains the streamflow, as there are no other major inputs to the streams because precipitation levels are at a minimum (Johanson 1988; Pearson 2004). Average monthly groundwater levels have an approximately 1.5 month lag relative to stream discharge (Berg and Allen 2007).

5.3. Methodology

5.3.1. Data Acquisition and Pre-processing

Fifteen (15) TidbiT® v2 Temp loggers (UTBI-001) were installed over a distance of approximately 40 m in a reach of Fishtrap Creek (Figure 5.2). The loggers were placed directly on the streambed to observe temperatures at the sediment-water interface and were attached to rebar to prevent movement. The loggers have an accuracy of ±0.2°C and a resolution of 0.02°C. Data were collected hourly; however, all analysis reported here are based upon daily averages. Loggers were first installed in July 2008, and the final data reported here are from October 2011. One datalogger (#12; see Figure 5.2) was lost following October 2008, and another was removed in July 2011 due to a low battery (#14).

The manufacturing specifications for the Tidbit temperature loggers indicate an annual drift of up to 0.1°C/year. The calibration of the dataloggers was verified in a temperature bath prior to deployment and at the end of the sampling period. A logger-specific linear drift correction was applied to the data. The mean drift over the four years was 0.04°C/year, and none of the dataloggers exceeded the maximum 0.5°C suggested by the specifications.
Dataloggers at the downstream end of the site (Figure 5.2) were distributed in two transects perpendicular to the channel to monitor a cross-section of the streambed. The remaining dataloggers were installed up the channel at approximately evenly distributed distances to capture differences along the direction of flow. This design was intended to capture the range of spatial variability of temperature that might exist by employing a spot measurement strategy commonly used by fisheries biologists. Figure 5.2 also shows the location of the 19 m deep BC Ministry of Environment Observation Well #2, which records daily groundwater levels, and the 8 m deep Environment Canada...
Observation Well ABB01, which records hourly groundwater temperatures. Unfortunately, no single observation well recorded both groundwater level and groundwater temperature over the period of the study.

The multiple time series files from each datalogger within each year were joined, time / date formats standardized, and quality assurance / quality control checks on the data performed using Aquarius v. 3.0.75.1 (Aquatic Informatics Inc., 2012). Data gaps up to several hours occurred during downloading events when dataloggers were removed from the stream. These minor gaps were filled using polynomial interpolation which was found to provide the best fit for hourly data gaps. The infilling of these gaps has little effect on the daily temperature series used here.

The analyses were performed in R (R Development Core Team 2011) using the contributed packages lubridate (Grolemund and Wickham 2011) for converting data and time from dataloggers, xts (Ryan and Ulrich 2011) for aggregating data to daily time steps, and fastICA (Marchini et al. 2010).

5.3.2. Heat Balance Model

Stream temperature is controlled by fluxes of heat energy that act on the water course, including a combination of radiation, conduction, convection, and advection (Webb 1996; Webb and Zhang 1997; Evans and Petts 1997; Hannah et al. 2004). The heat balance in the stream is the combination of energy fluxes at the water-air interface and the sediment-water interface. Dividing the system into interfaces can be useful for isolating the factors influencing the stream temperature which act to add or remove heat from the system (Evans et al. 1998; Hannah et al. 2004). The daily heat balance was calculated for the water-air interface to isolate climatic factors that may be influencing how the stream temperature changes as water flows from upstream to downstream through the reach. We needed to confirm that solar radiation dominated the heat balance and that other effects (e.g. precipitation events) were negligible.

The simplified heat balance for the site was calculated based on the following equation, following the methods described in Sinokrot and Stefan (1993), Evans et al.
(1998) and Xin and Kinouchi (2013) for the exchange of thermal energy across the air-water interface:

\[ H_{net} = H_{is}(1-\alpha) - H_l - H_e - H_c \]  

(5.1)

where \( H_{net} \) is the net heat exchange at the air-water interface, \( H_{is} \) is the incident solar radiation, \( \alpha \) is the albedo of the stream surface, \( H_l \) is the net longwave radiation, \( H_e \) is the evaporative heat transfer, and \( H_c \) is the convective (sensible) heat transfer. Daily incident solar radiation (\( H_{is} \)) was estimated using the solar position and radiation calculator developed by the State of Washington Department of Ecology (2014). The Bird Clear Sky model for direct radiation incident upon a horizontal surface was used, which is based on the latitude and elevation of each site (Bird and Hulstrom 1981). A value of 0.06 was used for \( \alpha \), following the approach by Xin and Kinouchi, (2013). The output is a daily estimate of the solar radiation; note that there is no correction for the conditions in the atmosphere except as exhibited in the air temperatures.

The net longwave radiation (\( H_l \)) was calculated using the daily mean stream and air temperatures and emissivities of the water surface and atmosphere (Xin and Kinouchi 2013). The evaporative heat flux (\( H_e \)) was calculated using the air temperature, relative humidity, and wind speed from the climate station following Xin and Kinouchi (2013) and Sinokrot and Stefan (1993). The convective heat flux (\( H_c \)) was calculated using the air and mean stream temperatures and the air pressure (Sinokrot and Stefan 1993; Xin and Kinouchi 2013). The equations and methods for calculation of the heat budget are presented in more detail in Middleton, Allen, and Whitfield (2015).

Heat budget plots were examined for variations in the net radiation, and to identify heat budget components that were dominating those variations. Plotting the daily heat balance for short windows of the summer period (Figure 5.3) allowed for comparison of the components that dominate the heat balance under different precipitation conditions as precipitation was not directly considered in the simplified heat budget. Figure 5.3a shows a two week period with precipitation, while Figure 5.3b shows a period without rainfall. The absence of variation in the heat budget components during the wet period indicates that precipitation events do not appear to impact the daily
heat balance for this site. The period with precipitation was examined specifically to evaluate if there were any lags in the daily heat balance as a result of precipitation events. For both conditions, incoming solar radiation dominates. Net longwave radiation and convective heat fluxes remained relatively unchanged in the daily heat balance, and therefore are not considered to be influencing factors for individual sediment-water interface temperature measurement locations. However, evaporative fluxes were reflected in the net heat flux for the stream, indicating evaporative processes can be important to the overall heat balance at a daily timescale. Figure 5.3 shows that variations in the evaporative heat flux are reflected inversely in the net radiation, with no time lag. The strong dependence of the heat balance on solar radiation means that, at this site, the air temperature can be used as a predictor of the sediment-water interface temperature to evaluate the influence of solar radiation on the interface temperatures. The use of air temperature as a surrogate for solar radiation in the absence of detailed heat flux data has been demonstrated in the literature as a reasonable approach (e.g. Smith, 1981; Mohseni and Stefan, 1999).
Figure 5.3. Daily heat balance components showing examples from 2009 of early summer periods with a) a period of rainfall and b) a period without rain.

5.3.3. Estimated Sediment – Water Interface Temperature

Both air and water temperatures respond to changes in incoming solar radiation; however, water temperature responses are delayed and damped due to the thermal inertia of water. Given that the sediment-water interface temperature was expected to correlate with solar radiation to some degree, a daily time series was needed for cross-correlation analysis with the ICA extracted components (as discussed later).
Water temperature at the sediment-water interface can be estimated from air temperature through an empirical linear relationship (Stefan and Preud'homme 1993). Comparisons between linear relationships and higher order polynomial relationships have found that a linear relationship is appropriate for estimating moderate (0°C to 20°C) water temperatures, which is the range of Fishtrap Creek summer temperatures (Mohseni et al. 1998; Mohseni and Stefan 1999; Kelleher et al. 2012). To relate the air temperature to sediment-water interface temperature, it is assumed that the stream is well mixed, uniform, and free flowing.

The time lag between the air and sediment-water interface temperatures for the study site was calculated as 30 hours. The lag period was found using the net surface heat transfer coefficient for heat transfer between the atmosphere and the water surface, and a mean summer water depth at the study site of 0.78 m (see Middleton, Allen, and Whitfield 2015). The value of the heat transfer coefficient of 30 Watts/m²/°C is consistent for this stream depth based on the time lag estimate given in Sinokrot and Stefan (1993) for summer values. For the calculation of sediment-water interface temperature at a daily time step, a lag of 1 day was used for simplicity. Thus, the lag is slightly lower (24 hours) compared to the calculated lag (30 hours) – see Discussion for the implications. The mean daily sediment-water interface temperature was calculated from the mean of all the interface loggers across the site. Using daily mean air temperatures with only positive values over the period of record, the linear relationship between daily sediment-water interface temperature ($T_s$) and daily air temperature ($T_a$), at a lag of one day, was determined empirically for the site as:

$$T_s(t) = 5.15 + 0.49 \times T_a(t-1\text{day})$$

where $T$ is in °C.

5.3.4. ICA

Independent component analysis is a statistically-based, signal processing technique that can be used to separate independent source components from an input of mixed signals that are time series (Comon 1994; Whitfield et al. 1999; Hyvarinen and Oja 2000; Naik and Kumar 2011; Hyvarinen 2012). The classic explanation of ICA is the
"cocktail party problem"; sample data from a number of microphones that ‘observe’ people talking simultaneously in a room are separated into individual speech signals. ICA finds the independent components by maximizing the statistical independence of the estimated components. There are different ways to define independence, and this choice governs the form of the ICA algorithm. The two broadest definitions of independence for ICA are minimization of mutual information and maximization of non-Gaussianity. The non-Gaussianity family of ICA algorithms, which include FastICA, are motivated by the central limit theory. Typical algorithms for ICA use centering (subtract the mean to create a zero mean signal), whitening (usually with the eigenvalue decomposition), and dimensionality reduction. These preprocessing steps are used in order to simplify and reduce the complexity of the problem for the actual iterative algorithm. Whitening ensures that all dimensions are treated equally a priori before the algorithm is run. ICA cannot identify the actual number of source signals, a uniquely correct ordering of the source signals, nor the proper scaling (including sign) of the source signals. ICA requires no prior knowledge of the mixing process and thus is one of the most common forms of blind signal separation.

The basis of ICA is that recording devices, such as temperature dataloggers, record mixed signals (x). These mixed signals are products of source signals (s) and some mixing matrix (A), where both s and A are unknown:

\[ x = As \tag{5.3} \]

The goal of ICA is to obtain an estimate of the independent source components (s), using the recorded signals (x). The source components (s) can be estimated from the mixed signals (x) and an un-mixing matrix (W):

\[ s = Wx \tag{5.4} \]

where \( W = A^{-1} \). The only observed variable is the mixed signal, x, and there is no prior input information on the original source signals or the mixing matrix, A. This absence of input information is the aspect of the method that is considered “blind”. To simplify the process, the equations do not consider any noise components, or time lag in the recordings.
Several ICA algorithms have been discussed in the literature and one of the most common is the FastICA method (Hyvarinen 1997; Hyvarinen and Oja 1997; Hyvarinen 1999a; Hyvarinen and Oja 2000; Naik and Kumar 2011). Fast ICA is a fixed point algorithm that uses higher order statistics for estimating the independent source components. The FastICA method (Marchini et al. 2010) performs centering and the pre-whitening, in addition to the ICA component extraction. The ICA separation of mixed signals is based on two assumptions and three effects of mixing source signals. The assumptions are a) the source signals are independent of each other, and b) the values in each source signal have non-Gaussian distributions. The three effects of mixing are:

1. The source signals are independent; however, their signal mixtures are not. This is because the signal mixtures share the same source signals.

2. According to the Central Limit Theorem, the distribution of a sum of independent random variables tends towards a Gaussian distribution. Thus, a sum of two independent random variables usually has a distribution that is closer to Gaussian than any of the two original variables. Here we consider the value of each signal as the random variable.

3. The temporal complexity of any signal mixture is greater than that of its simplest constituent source signal.

If the components extracted from a set of mixtures are independent (like source signals), or have non-Gaussian histograms (like source signals), or have low complexity (like source signals), then they must be source signals. One can drop the independence assumption and separate mutually correlated signals, thus, statistically "dependent" signals. Whitfield et al. (1999) demonstrated that ICA separation worked well in the presence of Gaussian noise. Some authors have noted that there is no guarantee that any particular algorithm can capture the individual source signals if its components are a nonlinear mixtures (Chawla 2009). In such cases, ICA does not ensure separation, and emphasizes very large indeterminacies (Jutten et al. 2004). While the transfer of heat into the groundwater is nonlinear, and at some time scales the temperature of the groundwater and the surface water could be correlated, the difference in statistical memory of these two sources suggests that the temperature of the sources can be considered locally independent at a daily time step. Surface water temperature is linked to air temperature relatively directly, as explained above, and groundwater temperature is similarly non-linearly driven at a longer time scale and considerably dampened. The
groundwater temperature thus varies only a small amount over a summer. Since the process we are interested in is the flux of water from different sources, that mixing, and hence the signal mixing, is linear which is sufficient to meet our objective of comparing daily signals between years of groundwater heat fluxes. In our study, we take a further step of conducting a cross-correlation analysis with variables that are expected to be related to the various extracted components, in order to limit ambiguities, and to identify the main hydrological processes linked to the components.

Two ambiguities exist in the ICA output components (Hyvarinen and Oja 2000; Naik and Kumar 2011; Hyvarinen 2012). The first is a magnitude and scaling ambiguity, in which the true variance of the independent components cannot be determined. The second is that the order of the estimated sources cannot be determined. Both ambiguities result because the source signals and the mixing matrix are unknown. Thus, no restrictions or conditions are imposed on the sources during separation, leaving the order indistinguishable, and each permutation equally valid.

In this study, ICA was used to perform a blind separation of the component signals contained in the records for each summer for all available temperature loggers. The number of available recorded signals was fifteen in 2008, fourteen in 2009 and 2010, and thirteen in 2011. ICA was run for each year using all available loggers as input signals. Any number of components may be extracted; however, as the number of extracted components increases, they become non-unique and it becomes difficult to distinguish between them. The strategy, therefore, was to consider what variables likely contribute to the heat budget of the stream reach, as determined by the simplified heat budget. At this site, solar radiation, stream inflow, groundwater exchange were considered the dominant variables. Therefore, three components were extracted.

5.3.5. Cross-correlation Analysis

The extracted temperature components were then compared to three main variables using the cross-correlation with the ICA components (estimated streambed temperature, stream discharge, and groundwater level). Unsmoothed and smoothed (2-day moving average) stream discharge were considered. The smoothed stream
discharge was used to strengthen the cross-correlation results by removing the short-term influence of precipitation events on stream discharge. The groundwater temperature remained relatively constant over all the summer periods, while the groundwater level fluctuated during and between the summer periods, and for this reason groundwater level is a more appropriate variable to test with cross-correlation. These variables are considered most likely to relate directly to three contributing variables described above. Cross-correlation provides a value that indicates the strength of the relationship between the variables, and also any lags between the two variables. The sign of the lag indicates which variable leads in the correlation, with a positive lag indicating that the x variable lags the y, and a negative lag the reverse. The estimated sediment-water interface temperature signal is the x variable in all cross-correlations. The lag is useful in evaluating the timing in the interface temperature responses.

5.4. Results

A linear model of daily mean air temperature and the one-day lagged mean measured sediment-water interface temperature for all non-negative temperatures in the four year period was found to explain more than 86 percent of the variance in interface temperature (Figure 5.4a). In the individual years, the relationship was consistent, with the air temperature explaining between 82 to 89 percent of the variance. The regression slope (0.50) and the positive intercept (5.1°C) are characteristic of streams with groundwater contributions, which moderate seasonal temperature fluctuations (Smith, 1981; Caissie, 2006). Over the summer period only, the linear model explains only 31 percent of the variance over the four summers (Figure 5.4b), suggesting that other factors dominantly contribute to the remaining variance. In individual summers the air temperature explained between 18 percent (2011) and 57 percent (2009) of the variance in the sediment-water interface temperature. In 2008 and 2010, the variance explained was 25 and 23 percent respectively. The lower regression slope (0.25) indicates that the interface temperature increases less relative to the increase in air temperature during summer compared to annually. As this reach is physically uniform, and evaporative fluxes do not dominate the heat budget over the summer period, the observed variability in sediment-water interface temperatures that are not explained by the estimated
sediment-water interface temperature can be attributed to variations in the groundwater flux or other local-scale drivers. The extraction method presented next seeks to understand these additional components over each summer.

Figure 5.4. a) The relationship between daily mean air temperature and mean observed one-day lagged sediment-water interface temperature for days with all non-negative air temperatures for the four year study period, including all data for the period of July 2008 to October 2011. b) The relationship between daily mean air temperature and mean observed one-day lagged sediment-water interface temperature for only the summer periods from 2008 through 2011.

\begin{align*}
    y &= 0.4955x + 5.132 \\
    R^2 &= 0.866
\end{align*}

\begin{align*}
    y &= 0.2478x + 9.633 \\
    R^2 &= 0.328
\end{align*}
Two important hydrologic variables in this reach of Fishtrap Creek are stream discharge and groundwater level. The mean daily summer stream discharge was 0.20 m$^3$/s (±0.25 m$^3$/s), but stream discharge generally decreased over the summer period, and had peaks associated with precipitation events throughout the summer. Figure 5.5a shows mean daily stream discharge over the 2008 to 2011 period. Groundwater levels in observation well #2 for the period from 2008 to 2011 are shown in Figure 5.5b. There is a definite seasonal pattern of groundwater levels and some year to year variation; highest groundwater levels were observed in 2011. The mean daily summer groundwater level was 12.8 m (±0.73 m) below ground surface, with a mean recession of 1.44 m. The summer periods consistently show a decline in groundwater levels. Precipitation amounts were low during the summer periods, with maximum summer rainfall events ranging from 16 mm in 2012 to 45.5 mm in 2010; the mean daily summer precipitation was 1.8 mm (±5.5 mm).
5.4.1. Sediment – water Interface Temperatures

Figure 5.6 shows the mean sediment-water interface temperatures (from all loggers across the site), air temperature, groundwater temperature, stream discharge, groundwater level, and precipitation over the four summer periods (July through September). In all years, the mean sediment-water interface temperature is lower than the air temperature with a pattern that closely follows the fluctuations in air temperature, but with a dampened amplitude in the temperature range as would be expected if incoming solar radiation were the greatest contributor of heat to streams (Stefan and Preud’homme 1993; Webb and Zhang 1997; Hannah et al. 2008). There are also
generally consistent peaks in air and sediment-water interface temperatures over the summers. The air and the sediment-water interface temperatures in 2011 had the smallest standard deviation and range of all of the four years. In the summer periods, the mean daily air temperature was 17.3°C (±2.8°C), the mean daily sediment-water interface temperature was 13.6°C (±1.0°C), and the mean daily groundwater temperature remained relatively constant with a mean of 10.6°C (±0.3°C). The lag between precipitation and discharge of up to one day is also evident in Figure 5.6.
Figure 5.6. Daily air temperatures, estimated sediment-water interface temperatures, mean daily sediment-water interface temperatures, daily precipitation, stream discharge (all on primary y-axis), and groundwater levels (secondary y-axis) for the summer periods of 2008-2011. The estimated sediment-water interface temperatures were calculated using Eq. 5.2.
To illustrate the ICA methodology, we will focus on the results for the summer of 2008; the other years were approached identically. The observed daily mean sediment-water interface temperatures for each of the 15 dataloggers are shown in Figure 5.7. These constitute the input signals that were used in the analysis. As mentioned, three components were extracted using ICA for each summer period. The complete set of ICA components extracted for summer 2008 is shown in Figure 5.8, along with the estimated sediment-water interface temperature and stream discharge for comparison. Groundwater level is not plotted in Figure 5.8; groundwater level declines throughout the summer as shown in Figure 5.5.

Figure 5.7. The fifteen observed daily mean temperatures at the sediment-water interface for 2008. L.1 to L.15 relate to the site locations shown in Figure 5.2.
For each of the four summer periods, cross-correlation analysis was conducted to classify the three extracted components. For classification we considered whether each extracted component (1, 2, 3) correlated with a) estimated sediment-water interface temperature, b) stream discharge, and/or c) groundwater level. Component 1 was most strongly correlated with the estimated sediment-water interface temperature measured over that particular summer. Correlation values were > 0.1 (considered significant). The patterns for components 2 and 3 differed between the summers.

As an example, Figure 5.9 shows the ICA components for 2008, with the various cross-correlation results. The top row is the plot of the time series for each variable, with the cross-correlations with each ICA component signal shown below. The mean sediment-water interface temperature and the ICA components are shown in the left column, followed by correlations with the estimated one-day lagged estimated sediment-water interface temperature in the second column. The third and fourth columns show the cross-correlations with discharge (unsmoothed discharge, then smoothed with a 2-
day moving average). The final column shows cross-correlation with the groundwater level. For completeness, the cross-correlation results are shown for 2009, 2010 and 2011 in Figures 5.10 – 5.12.
Figure 5.9.  Results for summer 2008 of the ICA components and the cross-correlation with the variables contributing to the heat exchanges. The extracted ICA components are shown in the left panel (pink) below the mean stream temperature recorded by the dataloggers at the sediment-water interface. The cross-correlations for each ICA component are shown in red, with the correlated variable shown in the top row of each column. The horizontal dashed lines in the correlation plots mark the 0.1 correlation value, above which the correlations were considered significant.
Figure 5.10. Results for summer 2009 of the ICA components and the cross-correlation with the variables contributing to the heat exchanges. The extracted ICA components are shown in the left panel (pink) below the mean stream temperature recorded by the dataloggers at the sediment-water interface. The cross-correlations for each ICA component are shown in red, with the correlated variable shown in the top row of each column. The horizontal dashed lines in the correlation plots mark the 0.1 correlation value, above which the correlations were considered significant.
Figure 5.11. Results for summer 2010 of the ICA components and the cross-correlation with the variables contributing to the heat exchanges. The extracted ICA components are shown in the left panel (pink) below the mean stream temperature recorded by the dataloggers at the sediment-water interface. The cross-correlations for each ICA component are shown in red, with the correlated variable shown in the top row of each column. The horizontal dashed lines in the correlation plots mark the 0.1 correlation value, above which the correlations were considered significant.
Figure 5.12. Results for summer 2011 of the ICA components and the cross-correlation with the variables contributing to the heat exchanges. The extracted ICA components are shown in the left panel (pink) below the mean stream temperature recorded by the dataloggers at the sediment-water interface. The cross-correlations for each ICA component are shown in red, with the correlated variable shown in the top row of each column. The horizontal dashed lines in the correlation plots mark the 0.1 correlation value, above which the correlations were considered significant.
Because climate conditions varied from summer to summer, there are twelve unique ICA components; three in each of the four summer periods. Visual comparison of these components in Figure 5.9 (2008) through Figure 5.12 (2011) demonstrates the variability between the years (discussed later). Table 5.1 provides a summary of the results of the cross-correlation tests for each of the summer periods. Correlations were considered significant when they were both greater than 0.1 and exceeded the calculated statistical significance values on the cross-correlation plot. In Table 5.1, components are listed in order of strongest correlation (e.g. in 2009 groundwater level correlates strongly with component 2 and to a lesser degree with component 1).

Table 5.1. A summary of the ICA components and the correlations with predictor variables. The table lists the ICA component number (1 through 3). Each component correlates with one or more of the variables listed in the top row. The results for 2008 are bolded for comparison with Figure 5.9. Correlations were considered significant when they were both greater than 0.1 and exceeded the calculated statistical significance values on the cross-correlation plot. Components are listed in order of strongest correlation.

<table>
<thead>
<tr>
<th>Summer</th>
<th>Estimated Sediment-Water Interface Temp.</th>
<th>Discharge (un-smoothed)</th>
<th>Discharge (2-day moving average)</th>
<th>Groundwater Level</th>
</tr>
</thead>
<tbody>
<tr>
<td>2008</td>
<td>1, 2</td>
<td>2, 3</td>
<td>3, 2</td>
<td>3, 1</td>
</tr>
<tr>
<td>2009</td>
<td>1</td>
<td>3</td>
<td>3</td>
<td>2, 1</td>
</tr>
<tr>
<td>2010</td>
<td>3, 2, 1</td>
<td>1</td>
<td>1</td>
<td>1, 2, 3</td>
</tr>
<tr>
<td>2011</td>
<td>1</td>
<td>2, 1</td>
<td>2, 1</td>
<td>2, 3</td>
</tr>
</tbody>
</table>

### 5.5. Discussion

ICA differs from the analytical approaches that are classically used to examine stream temperatures. Those approaches consider the source signals and relevant noise and bias components (air temperature, water temperature, groundwater temperature, noise and sensor drift). We do not consider these components directly. Rather, using ICA, we extracted independent signals from the observed time series and correlated those signals to variables that can influence sediment-water interface temperature through inputs to the heat budget over the distance of the stream reach. These variables
include the solar radiation, represented by the estimated sediment-water interface temperature; the heat of the incoming stream flow, changes to which are represented by the stream discharge; and heat transfer due to groundwater whereby changes in the groundwater levels reflect the potential of the groundwater to contribute to the stream flow and thus influence the sediment-water interface temperature.

To interpret the ICA and cross-correlation results, Table 5.2 summarizes the climate and hydrological processes in each summer that are captured across the top row in Figures 5.9 through 5.12. The table ranks each parameter (where (1) is highest and (4) is lowest) and provides generalized comments about responses.
Table 5.2. Overview of the climate and hydrological observations over the four year period. Rank is shown in ( ) with (1) the highest and (4) the lowest.

<table>
<thead>
<tr>
<th>Summer</th>
<th>Mean / Max Air Temp. (°C)</th>
<th>Mean / Range Sediment-Water Interface Temp. (°C)</th>
<th>Total Precip. (mm)</th>
<th>Median Discharge (m³/s)</th>
<th>Max / Range Groundwater Level / (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2008</td>
<td>16.8 (4) / 23.6 (3)</td>
<td>13.8 (3) / 5.2 (2)</td>
<td>186 (2) most in August</td>
<td>0.11 (2)</td>
<td>13.6 (3) / 1.0 (4) Declined throughout the summer</td>
</tr>
<tr>
<td>2009</td>
<td>18.1 (1) / 28.8 (1)</td>
<td>14.6 (4) / 8.1 (1)</td>
<td>155 (4)</td>
<td>0.05 (4)</td>
<td>14.0 (1) / 1.4(3) Declined throughout the summer</td>
</tr>
<tr>
<td>2010</td>
<td>17.3 (2) / 25.9 (2)</td>
<td>14.1 (2) / 4.7 (3)</td>
<td>252 (1) most in September</td>
<td>0.1 (3)</td>
<td>13.6 (2) / 1.4 (2) Declined from July to Aug. with little variation in Sept.</td>
</tr>
<tr>
<td>2011</td>
<td>17.3 (3) / 22.1(4) (1)</td>
<td>13.3 (4) / 3.6 (4)</td>
<td>182 (3) rainfall events were mainly in early July and late September</td>
<td>0.2 (1)</td>
<td>12.8 (4) / 1.9 (1) Declined throughout the summer</td>
</tr>
</tbody>
</table>

The estimated sediment-water interface temperature varied each summer. In all summers except 2010, component 1 was strongly correlated with estimated interface temperature. This suggests that surface heating from solar radiation is the dominant factor influencing the sediment-water interface temperature in most years. In summer 2008, however, there was a negative lag of one day in the correlation between the ICA component and the estimated sediment-water interface temperature (Figure 5.9).
negative lag in the temperature correlation suggests that in the cool periods, such as in 2008, the one-day calculated lag for thermal energy transfer from solar radiation may be an over estimation of the lag. In the other summer periods, the correlation with estimated sediment-water interface temperature was at zero lag time, indicating that the one-day calculated lag was representative of the heat transfer rate.

Stream discharge differed between the four summers, as might be expected due to variations in precipitation. However, no consistent patterns emerged when comparing the cross-correlation results between summers. In contrast, there is a distinct decline in local groundwater levels each summer, although the amount of decline varied between the summers. This indicates that groundwater is discharging and contributing to the surface flow, but that the magnitude of groundwater discharge decreases over the summer and is variable between summers. The groundwater may contribute a diffuse influence on the sediment-water interface temperature distributed along the stream channel, or groundwater influxes may be localized, particularly when hyporheic flows are present due to variations in bed and reach morphology, among other factors. However, because all the loggers were combined in this study (i.e. ICA was run for all available loggers each year), the diffuse or localized nature of the groundwater contribution could not be distinguished. A spatial analysis using the ICA method, comparing signals among dataloggers situated at different locations, may provide greater insight into the spatial variability of groundwater influxes.

In each of the four summer periods, the ICA components correlate to at least one of the three heat contributing variables (Table 5.1). In most years, however, more than one component correlated with a particular heat contributing variable. While independence of the variables is an assumption of ICA, these results suggest that the variables influencing the sediment-water interface temperatures are not entirely independent. The correlation of the components (Table 5.1) is related to the trends in the variables listed in Table 2. In cool wet years, such as 2008, the stream temperatures are lower, while the discharge and groundwater levels are higher. In summer 2008, three components correlated to multiple variables (Table 1), indicating that when variations in groundwater levels are low, the variations in the temperature signals and groundwater contributions are more difficult to separate. In 2009, the streamflow and groundwater
levels were among the lowest of the summer periods, the air and stream temperatures reached their highest values. In 2009, fewer components were correlated to more than one variable (Table 5.1). Summer 2010 had moderate rankings in all variables, but had the highest precipitation, which occurred mainly in September resulting in the higher stream flows. The mean sediment-water interface temperature in 2010 also had the highest range of values. The components also correlated with a mix of variables in 2010, with only component 1 correlating with discharge, but all components correlating with estimated sediment-water interface temperature and groundwater level. In 2011, the stream discharge and groundwater levels were high, and the air and stream temperatures were low. The lowest range in stream temperature occurred in 2011. One component signal (component 2) in 2011 correlated to both discharge and groundwater levels (Table 1), indicating that variation in temperature contributions from these variables are more difficult to separate when they are both high.

Overall, in all summers the extracted components were correlated with more than one variable. However, in summers with lower stream discharge and greater stream temperature ranges the contributing variables were more easily separated. This inability to completely separate the components and relate them to specific variables is no doubt a product of the fact that the variables we considered might be nonlinear combinations (e.g. the interaction between air temperature and streamflow). Cloudy conditions affect air temperature, increase the probability of precipitation, and subsequently discharge and groundwater level. We suspect that the separations in Table 1 reflect this complex interrelationship. Thus, ICA has limitations in natural settings where, for example, climate influences multiple processes and interactions between processes exist.

Nevertheless, some broad observations can be made based on the ICA results, which enhance our understanding of the system. Specifically, thermal exchanges appear to be taking place in addition to the air-water interface. These exchanges also take place at the sediment-water interface, and the correlation with groundwater levels indicates these heat exchanges are associated with groundwater inflow. The results are not surprising given that Fishtrap Creek has been described as a groundwater-fed stream (Berg and Allen 2007). This study provides stronger evidence that in some years (e.g. 2009) the sediment-water interface temperature is highly influenced by groundwater
inflows across the site. Based on the spatial variability of the component signals (results not shown), the locations of groundwater inflow are variably distributed across this site, indicating that this reach is influenced by a combination of focused and diffuse groundwater discharge.

Other studies have similarly reported temporal variability of groundwater inflows (e.g., Constantz, 1998; Wroblicky et al., 1998; Keery et al., 2007). The inflow of groundwater to streams was reported in these studies to be a complex process, with scale-dependent variability occurring both spatially and temporally. Temporally, variability can range from diurnal to interannual as shown in this study. Here, we have demonstrated the use of ICA in blind separation of mixed signals from temperature loggers at the sediment-water interface and assessed how those component signals can be used to identify important heat transfer processes. While there were some ambiguities in the extracted signals, likely due to non-independence of the temperature signal components, the use of cross-correlation helped to reduce these ambiguities. Without cross-correlation, it was challenging to associate a particular extracted component with a particular variable, with the exception of the estimated sediment-water interface temperature, which was both visually similar to component 1 and often had a high correlation with it in most years.

The value of ICA is that temperature signals from multiple dataloggers can be evaluated against known, or suspected, variables. A priori knowledge of these variables (here estimated sediment-water interface temperature, stream discharge and groundwater level) helped to determine the number of components for extraction. In previous iterations of this work, we used ICA somewhat blindly, and extracted several components. Through experimentation, we ultimately settled on three to reflect the dominant processes at the site. Other studies may benefit from this approach, but a reasonable conceptual model of the site is warranted in order to focus the analysis.

Finally, as mentioned above, other applications of ICA in the hydrological sciences should be explored. In particular, given the presence of multi-dimensional flows such as those due to hyporheic flows, and the high degree of spatial heterogeneity in
streambed and aquifer hydraulic properties that may influence the temperatures (and fluxes), a spatial analysis using ICA may prove particularly useful at some sites.

5.6. Conclusions

The study focused on the summer period of July to September, when streamflow in the studied coastal stream is low and the relative contribution of groundwater to streamflow is often the highest for the year. Sediment-water interface temperatures in this small 40 m reach of Fishtrap Creek are controlled by multiple processes. The dominant process is the transfer of thermal energy from the atmosphere to the stream and then to the streambed in each of the four summers. Stream discharge and groundwater contributions influence the observed sediment-water interface temperatures and their importance varied between the four summers reported here. The contributions of these processes are complex, varying between and within the summer periods. It is demonstrated that components of observed sediment-water interface temperatures extracted using ICA can be used to interpret information from multiple sediment-water interface temperature sensors.

The timing and magnitude of discharge in the summer periods as well as annual groundwater levels are important factors in the distribution of the temperature components across the stream reach. Separation of the temperature components is more apparent during summers with lower flows, and greater stream temperature ranges. While solar radiation is the dominant thermal contribution to the reach, observed sediment-water interface temperatures are modified by streamflow variations and groundwater inputs.
Chapter 6.

Vulnerability Assessment for Groundwater Dependent Streams

6.1. Introduction

This chapter proposes a new methodology “Vulnerability Assessment for Groundwater Dependent Streams” (hereafter Stream Vulnerability Assessment). It is a multi-step, risk-based approach aimed at evaluating the vulnerability of a groundwater dependent stream to changes in the aquifer system. There is a particular emphasis on the summer low flow period, because it is during this time that streams can be sensitive to changes in the aquifer system; however, in principle the methodology can be used to assess stream vulnerability year round.

Understanding the likely response of streams to changes in the groundwater levels is important for integrated management of water quantity, quantity particularly in relation to groundwater pumping and its impact on stream flow. In many streams in the province of British Columbia (BC), stream flow during the annual summer low flow period\(^1\) is sustained by groundwater inputs (baseflow); as a result, such “groundwater dependent streams” can be sensitive to lower groundwater fluxes during this period. Streams with greater connectivity to the aquifer system\(^2\) may respond to changes in groundwater levels and fluxes, especially during the summer low flow period (Allen et al. 2010). Decreases in the timing and amount of precipitation as a result of climate change

\(^1\) In British Columbia, many streams also have a winter low flow period during which the streams may also be sensitive to changes in groundwater flux; however, this document is focused only on the summer period.

\(^2\) The “aquifer system” includes aquifers and aquitards, although the connection with a stream will be primarily through the more permeable geological units which are characterized as aquifers.
are projected for many areas of the province, and this has the potential to lead directly or indirectly to more extreme summer low flow events, and to extend the length of the summer low flow period in many streams (Déry et al. 2009). Climate variability and climate change have the potential to impact recharge conditions (e.g. Allen et al. 2004) as well as lead to increased water resource demands (e.g. Cohen et al. 2004), which in turn will lead to changes in groundwater conditions. Changes in land use/land cover (urbanization, timber harvesting, etc.) also impact recharge (Arnell 2002).

This chapter also discusses how the Stream Vulnerability Assessment can be incorporated into a Risk Assessment / Risk Management Framework, to include indicators relevant to groundwater dependent streams and how these indicators may be used to inform decision making. The Vulnerability Assessment is envisioned to provide critical information for the Sensitive Stream Designation in BC. Under the *Fish Protection Act*, (Bill 25: FPA, 1997: Section 6(2)) a stream is designated a “sensitive stream” when it “contributes to the population of fish whose sustainability is at risk because of inadequate flow of water within the stream or degradation of fish habitat.” Under this regulation, a stream designated as sensitive will have mitigation measures and recovery plans in place to ensure sufficient water quantity for fish survival. The Sensitive Stream Designation is addressed in the *Fish Protection Act* (Bill 25: FPA, 1997) The *Water Sustainability Act* also addresses surface water and groundwater use including provision for environmental flows (Bill 18: WSA, 2014). The vulnerability assessment method presented in this document aims to contribute to the definition of a sensitive stream by including groundwater sensitive streams. A groundwater sensitive stream is herein defined as “a stream that is groundwater dependent and vulnerable to changes in the aquifer system, and is likely to have measurable impacts in water quantity to potential stressors.”
6.2. Vulnerability Assessment for Groundwater Dependent Streams: Overview

The Vulnerability Assessment involves a three level assessment procedure shown schematically in Figure 6.1:

![Vulnerability Assessment Method for Groundwater Dependent Streams](image)

*Figure 6.1. Overview of the levels of assessment for Stream Vulnerability.*
6.2.1. Level I Assessment

A Level I Assessment evaluates the potential vulnerability of a stream based on the hydrologic setting and the level of development of the aquifer. The assessment is qualitative and relies on existing publicly available information for classified aquifers in BC. A Level I Assessment is intentioned for screening or prioritizing purposes or for provincial level classification of stream-aquifer connectivity in diverse settings. If a Level I Assessment determines the stream is potentially vulnerable, then a Level II Assessment would be undertaken.

6.2.2. Level II Assessment

A Level II Assessment rates the vulnerability of the stream relative to other streams, specifically the degree of connectivity between the stream and the aquifer, and the stressor(s) that act on the system. This Level II Assessment is semi-quantitative and is intended for establishing water management guidelines or policies in aquifers where the stream is connected to the aquifer system. If a Level II Assessment determines the stream is vulnerable, a Level III assessment would be undertaken to quantify the potential impact to the stream from the stressor(s).

6.2.3. Level III Assessment

A Level III Assessment considers the impacts to the stream from groundwater-related stressors. Level III Assessments are quantitative in that they incorporate data analysis requiring more specific information about how the stream-aquifer system functions as well as the magnitude of the stressors acting on the system. Level III Assessments are site specific and intended for such activities as drought preparedness, groundwater licensing, planning of subdivisions, etc. The assessment aims to demonstrate the likely impact on a stream due to stressors acting on the aquifer system. The assessment could include, for example, the impact of groundwater pumping or impacts due to changes in recharge rates caused by land use/land cover changes or climate change/climate variability.
6.3. Level I Assessment: Potential Stream Vulnerability

A Level I Assessment evaluates the potential vulnerability of a stream based on the hydrologic setting and the level of development of the aquifer in the area of interest. The main objective of a Level I Assessment is to assess whether the stream is potentially connected to the aquifer, and whether the aquifer can produce adequate quantities of water to meet the current demand. A Level I Assessment is intended for screening or prioritizing purposes, or for provincial level classification of stream-aquifer connectivity in diverse settings.

A Level I Assessment uses publicly available information, where possible, on aquifers and streams. Spatial data can be accessed from iMapBC (http://maps.gov.bc.ca/ess/sv/imapbc/).

6.3.1. Step 1: The Hydrologic Setting

Step 1 of the Level I Assessment establishes whether the stream intersects the aquifer. The BC Ministry of Environment maintains an inventory of aquifers in the province. Aquifer polygons have been mapped, and their attributes (outline, vulnerability, among other parameters available for specific aquifers) characterized. All aquatic-related features (streams, rivers, lakes, wetlands, etc.) are also mapped at a 1:50,000 scale for the province (BC Watershed Atlas - http://www.env.gov.bc.ca/fish/watershed_atlas_maps/). If the stream intersects an aquifer, then there is a potential for that stream to interact with the aquifer. Some examples are provided below.

Figure 6.2 shows the aquifer polygons in the Fraser Valley with a stream map layer based on the BC Watershed Atlas (Stream Routes 50k). Figure 6.3 shows an enlarged portion of Figure 6.2. In both figures, most of the streams intersect an aquifer polygon. Not all aquifers are at the surface, so these figures identify only potential interactions.
Figure 6.2. Aquifer polygons (black outline) shown with streams (blue) in the Fraser Valley.

Figure 6.3. Zoomed view of aquifer polygons (black outline) with streams (blue) in the Fraser Valley.
Figure 6.4 shows the aquifer polygons in the Fraser Valley with the map layer of streams designated as Sensitive Streams in the *Fish Protection Act*. In catchments with sensitive streams, a Level II Assessment is recommended.

![Figure 6.4](image)

**Other Considerations**

In some cases, there may be insufficient information available on iMapBC to undertake step 1 of the Level I Assessment and it is recommended a hydrogeologist be consulted in those situations.

**Not all streams intersect aquifers.** For example, confined aquifers typically lie at depth below the surface, and while they are mapped and appear to be at surface based on the aquifer polygons, further investigation would be needed to determine if an aquifer is unconfined or confined. Also, a number of aquifers mapped as confined aquifers are semi-confined by semi-permeable and/or discontinuous confining units. Those aquifers mapped as confined, therefore, should be investigated to determine potential for connectivity with streams.
Not all aquifers are mapped. While some 1,000 aquifers have been mapped to date in the province, not all aquifers have been mapped. If aquifer polygons are not shown for the area of interest, then further investigation is needed to obtain the data necessary to complete a Level I Assessment.

Aquifer polygons in bedrock regions typically do not extend beyond areas with water wells. In bedrock regions (areas with no substantial accumulation of surficial materials), aquifer polygons typically extend no further than the area with existing water wells. This is because the aquifer inventory focuses on developed areas. However, even a single well placed outside an aquifer boundary in bedrock has the potential to be connected to a stream. If aquifer polygons in bedrock do not extend to the area of interest, then further investigation is needed.

6.3.2. Step 2: The Level of Development

Step 2 of the Level I Assessment is based on the BC Aquifer Classification System (ACS) (Kreye and Wei 1994). The system 1) classifies aquifers on the basis of their level of development and vulnerability to contamination, and 2) provides ranking values for aquifers using hydrogeologic and water use criteria. While designed primarily for assessing the vulnerability of the aquifer to contamination from surface activities, the development component is readily adapted to assess stream vulnerability as it relies on information on the aquifer properties.

The Level of Development is a relative and subjective term. However, it enables comparison of the amount of groundwater withdrawn from an aquifer (demand) to the aquifer’s inferred ability to supply groundwater for use (productivity) (Berandinucci and Ronneseth 2002). In the context of stream interaction, aquifers with a higher level of development are more likely to impact streamflow than those with a low level of development, other factors being equal. The Level of Development is assessed based on 1) aquifer productivity and 2) groundwater demand. Aquifer Productivity and Groundwater Demand have been assessed for over 1,000 aquifers in BC as part of the aquifer inventory process (BC Ministry of Environment, 2014).
Aquifer Productivity

Aquifer Productivity describes the rate of groundwater flow from wells and springs and the abundance of groundwater in an aquifer. Indicators of productivity (e.g., aquifer material, reported well yields, specific capacity of wells, and transmissivity of the aquifer) are used to infer potential water availability of the aquifer (Table 6.1). For example, Kreye and Wei (1994) assign an indicator of 1 to a low productivity aquifer, 2 for moderate, and 3 for high (Table 6.1). Figure 6.5 shows an example of aquifers in the Fraser Valley for which productivity has been estimated.

Table 6.1. Productivity classes (from Kreye and Wei, 1994).

<table>
<thead>
<tr>
<th>Indicators of Productivity</th>
<th>Low (1)</th>
<th>Moderate (2)</th>
<th>High (3)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aquifer Material</td>
<td>-Silt and sand</td>
<td>Sand</td>
<td>Sand and gravel</td>
</tr>
<tr>
<td></td>
<td>-Fractured bedrock</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Well Yield (L/s)</td>
<td>&lt; 0.3</td>
<td>0.3 – 3.0</td>
<td>&gt; 3.0</td>
</tr>
<tr>
<td>Specific Capacity (L/s.m)</td>
<td>&lt; 0.4</td>
<td>0.4 – 4</td>
<td>&gt; 4</td>
</tr>
<tr>
<td>Transmissivity (m²/s)</td>
<td>&lt; 5.0E-4</td>
<td>5.0E-4 - 5.0E-3</td>
<td>&gt; 5.0E-3</td>
</tr>
</tbody>
</table>
Figure 6.5. Productivity (High (3), Moderate (2), Low (1)) of aquifers in the Fraser Valley with an overlay of sensitive streams (pink).

**Groundwater Demand**

The Groundwater Demand provides information on the groundwater use. Well use and actual withdrawal rates are usually not available. Therefore, groundwater demand is generally assessed subjectively based on domestic well density per map quadrant, the number and type of production wells, and general knowledge of well use and land use in the area (Table 6.2). Figure 6.6 shows an example of aquifers in the Fraser Valley for which demand has been estimated.
Table 6.2. Demand classes (modified from Kreye and Wei, 1994).

<table>
<thead>
<tr>
<th>Demand Classes</th>
<th>Low (1)</th>
<th>Moderate (2)</th>
<th>High (3)¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>Well Density Descriptor</td>
<td>Low</td>
<td>Moderate</td>
<td>High</td>
</tr>
<tr>
<td>Number of Wells</td>
<td>&lt; 10</td>
<td>10-50</td>
<td>&gt;50</td>
</tr>
<tr>
<td>Well Density (wells/km²)</td>
<td>&lt; 4</td>
<td>4-20</td>
<td>&gt;20</td>
</tr>
</tbody>
</table>

¹The high and very high categories from Kreye and Wei (1994) have been combined in this table. Aquifers categorized as very high well density in the aquifer database are simply classified high in this table.

Figure 6.6. Groundwater Demand High, Moderate, Low) in the Fraser Valley with an overlay of sensitive streams (pink).

Level of Development

The Level of Development is based on Table 6.3. Three levels of development are designated: heavy I; moderate II; or light, III. Given the same demand, the Level of
Development of an aquifer of low productivity would be considered higher than an aquifer of higher productivity. Thus, the rating would be increased as shown in Table 6.3.

<table>
<thead>
<tr>
<th>Demand</th>
<th>Productivity</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low (1)</td>
<td>III</td>
</tr>
<tr>
<td>Moderate (2)</td>
<td>II-III</td>
</tr>
<tr>
<td>High (3)</td>
<td>III</td>
</tr>
</tbody>
</table>

Table 6.3. Level of Development risk matrix.

Other Considerations

Where an aquifer has not yet been classified, the Aquifer Productivity and Level of Demand can be assessed using the tables above following the methodology by Berardinucci and Ronneseth (2002). Indicators of Aquifer Productivity and Level of Demand can be estimated using information on wells in the BC WELLS database, which is also linked to iMapBC.

6.3.3. Final Level I Assessment Criteria

Table 6.4 lists the criteria that are used to determine the action required depending on whether the stream intersects the aquifer in the area of interest and Level of Development within the aquifer in the area of interest. The Level of Development and the description in Table 6.4 are modified from Berardinucci and Ronneseth (2002).
Table 6.4. Level of Development within an aquifer and action required.

<table>
<thead>
<tr>
<th>Level of Development</th>
<th>Description</th>
<th>Action Required</th>
</tr>
</thead>
<tbody>
<tr>
<td>Light III</td>
<td>No stream intersects the aquifer in the area of interest. Thus, there is a low potential for connection between the stream and the aquifer. Demand for water is light relative to water availability.</td>
<td>No further action required</td>
</tr>
<tr>
<td>Moderate II</td>
<td>A stream either passes through the aquifer or borders the aquifer. Thus, there is a moderate potential for connection between the stream and the aquifer, particularly in areas very close to the stream. Demand for water is moderate relative to water availability.</td>
<td>Proceed to Level II Assessment</td>
</tr>
<tr>
<td>Heavy I</td>
<td>A sensitive stream either passes through the aquifer or borders the aquifer and/or there is a high potential for connection between the stream and the aquifer. Demand for water is high relative to water availability.</td>
<td>Proceed to Level II Assessment</td>
</tr>
</tbody>
</table>

6.4. Level II Assessment: Rating Stream Vulnerability

6.4.1. Overview

A Level II Assessment results in a rating of stream vulnerability. It is a semi-quantitative assessment process that relies on understanding of the physical system (the aquifer - stream system), and what current stressors act on the system. A Level II Assessment rates the stream vulnerability relative to other streams, specifically the degree of connectivity between the stream and aquifer, and is intended for establishing water management guidelines or policies in aquifers where the stream is connected to the aquifer system.

The Stream Vulnerability (SV) is the combination of the Stream Susceptibility (SS) and the Hazard (H) (Figure 6.7) and is rated in a matrix. The stream susceptibility evaluates the potential for the stream to be influenced by stressors acting on the aquifer system. It
is based on the aquifer characteristics and the recharge to the aquifer system. The hazards represent the current stressors to the aquifer, specifically pumping, which may translate into potential changes to the stream.

![Flow chart outlining the components of stream vulnerability.](image)

### 6.4.2. Stream Susceptibility (SS)

The stream susceptibility (Equation 6.1) represents the natural hydrogeological system, characterized by the aquifer setting, the aquifer properties, the nature of the interconnection between the aquifer system and the stream, and the recharge characteristics:

\[
\text{Stream Susceptibility (SS)} = \text{Aquifer Characteristics (A)} \times \text{Recharge Ratio \((Q_s/Q_r)\)} 
\]

#### Aquifer Characteristics (A)

Wei et al. (2009) summarize the characteristics of the major aquifer types in the province. The aquifer characteristics reflect the aquifer setting, the origin and type of geologic deposit, the degree of confinement, and the potential hydraulic connection with surface water. There are twelve aquifer types, eight in unconsolidated sand and gravel settings, and four in bedrock (Table 6.5).

Each aquifer type is defined primarily on geological and hydrological properties, as well as on practical considerations, such as data availability (Wei et al. 2009). The main geologic factors are the origin and type of the geologic deposit that comprise an
aquifer (e.g., sand and gravel aquifer forming a delta at the mouth of a river, or a plutonic granitic fractured bedrock aquifer). The origin and type of geologic deposit often governs an aquifer’s hydraulic properties, such as the nature of the porous medium (porous sand and gravel, or fractured bedrock) and its ability to transmit and store water. The degree of aquifer confinement represents the hydraulic separation of aquifers from each other and from surface waters. Aquifers can be unconfined, discontinuously confined, partially confined, or confined. Unconfined aquifers have the highest likelihood of being connected to the stream because there is no low permeability layer separating the aquifer from the stream. Deep confined aquifers are the least likely to be connected to streams.

Table 6.5 rates each aquifer type according its likely connection with a stream. The rating scheme is also shown in Figure 6.8. A direct hydraulic connection can be disadvantageous to streamflow because pumping could induce infiltration of surface water into the aquifers, and thereby remove water from the stream.

Table 6.5. Aquifer types and key hydrogeological characteristics (from Wei et al. 2009) with the assigned Aquifer Characteristics (A) ratings assigned through consultation with BC Ministry of Environment.

<table>
<thead>
<tr>
<th>Aquifer type</th>
<th>Confined - unconfined</th>
<th>Connection with streams</th>
<th>Rating¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Aquifers of fluvial or glaciofluvial origin along river valley bottoms</td>
<td>Unconfined</td>
<td>Commonly connected but stream size buffers impact</td>
<td>4</td>
</tr>
<tr>
<td>a. aquifers along low gradient, higher order rivers</td>
<td>Unconfined</td>
<td>Commonly connected but stream size buffers impact</td>
<td>4</td>
</tr>
<tr>
<td>b. aquifers along generally higher gradient, moderate order rivers</td>
<td>Unconfined</td>
<td>Commonly connected</td>
<td>10</td>
</tr>
<tr>
<td>c. aquifers along lower order streams; limited aquifer thickness and lateral extent</td>
<td>Unconfined</td>
<td>Commonly connected</td>
<td>10</td>
</tr>
<tr>
<td>2. Deltaic (sand and gravel) aquifers</td>
<td>Unconfined</td>
<td>Commonly connected</td>
<td>10</td>
</tr>
<tr>
<td>3. Alluvial, colluvial (sand and gravel) fan aquifers</td>
<td>Unconfined</td>
<td>Commonly connected near the stream</td>
<td>8</td>
</tr>
</tbody>
</table>
### Aquifer type

<table>
<thead>
<tr>
<th>Aquifer type</th>
<th>Confined - unconfined</th>
<th>Connection with streams</th>
<th>Rating¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>4. Aquifers of glacial or pre-glacial origin</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>a. Outwash and ice-contact sand and gravel aquifers (glacio-fluvial)</td>
<td>Unconfined</td>
<td>Commonly connected near the stream</td>
<td>8</td>
</tr>
<tr>
<td>b. Aquifers of glacial or pre-glacial origin</td>
<td>Mostly confined</td>
<td>Possibly connected if unconfined</td>
<td>4</td>
</tr>
<tr>
<td>c. Confined aquifers of glacio-marine origin</td>
<td>Confined</td>
<td>Unlikely to be connected</td>
<td>4</td>
</tr>
<tr>
<td>5. Sedimentary rock aquifers</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>a. fractured sedimentary rock aquifers</td>
<td>Unconfined near surface</td>
<td>Possibly connected near the stream</td>
<td>3</td>
</tr>
<tr>
<td>b. karstic limestone aquifers</td>
<td>Unconfined near surface</td>
<td>Likely connected</td>
<td>5</td>
</tr>
<tr>
<td>6. Crystalline rock aquifers</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>a. flat-lying or gently-dipping volcanic flow aquifers</td>
<td>Unconfined near surface</td>
<td>Likely connected</td>
<td>5</td>
</tr>
<tr>
<td>b. fractured igneous intrusive, metamorphic, fractured volcanic or metavolcanic aquifers</td>
<td>Unconfined near surface</td>
<td>Possibly connected near the stream</td>
<td>3</td>
</tr>
</tbody>
</table>

¹The ratings were determined based on expert knowledge of aquifer types in British Columbia. Intermediate rating values could be assigned based on local hydrogeological conditions.
Figure 6.8. Overview of Aquifer Characteristics rating.
Recharge to Stream \( (Q_R) \)

The recharge component applied to assess the reliance of the stream during the summer low flow period on locally-derived aquifer recharge. Annual recharge is assessed based on the area contributing groundwater discharge to the stream, and is compared to baseflow to estimate the ability of the discharge amount to sustain the streamflow.

When considering the annual summer low flow period, the annual recharge to the aquifer is a critical factor. Aquifer recharge, however, is difficult to quantify due to large uncertainties in the various water balance components. Equation 6.2 shows a typical annual water balance equation for an aquifer system.

\[
R = (P + Q_{in} + GW_{in}) - (AET + Q_{out} + GW_{out}) \pm \Delta S_G \pm \Delta S_S
\]  

(6.2)

where \( R \) is the recharge, \( P \) is precipitation, \( Q_{in} \) is the surface water inflow (influent water bodies), \( GW_{in} \) is the groundwater inflow to the area (from adjacent areas and return flows from irrigation), \( AET \) is actual evapotranspiration, \( Q_{out} \) is the surface water outflow (effluent water bodies), \( GW_{out} \) is the groundwater outflow from the area (to adjacent areas and pumping), and \( \Delta S_G \pm \Delta S_S \) are the changes in storage for each of groundwater and surface water (typically assumed to be zero on an annual basis). Quantification of each component of the water balance equation for the aquifer system requires data and a sound understanding of the system. For this reason, a full water balance assessment would require a Level III Assessment.

For a Level II Assessment, the water balance of the aquifer is approximated as follows:

\[
R = P - PET = Q_R
\]  

(6.3)

This water balance equation is a gross simplification of the aquifer system, but it provides a first order approximation of the potential recharge \( (R) \) and hence the discharge to the stream from the aquifer system \( (Q_R) \). Equation 6.3 assumes the aquifer drains to a stream and that all the recharge to the aquifer discharges to the stream \( (R = \)
Q_r). It assumes no pumping. It assumes that if there is any groundwater inflow from adjacent areas, that this groundwater leaves the aquifer through adjacent areas. It also assumes that there are no gains to the aquifer from the stream.

Precipitation is measured at many locations in Canada, and annual estimates are available as precipitation normals over a recent 30 year period (Environment Canada 2007). BC station data are also available from the Data Portal maintained by the Pacific Climate Impacts Consortium (http://www.pacificclimate.org/data/bc-station-data). In addition to the Environment Canada stations, data for many non-federal climate stations are available. Near real-time climate data are available. The quality of the data used for a Level II Assessment must be balanced with the availability of data (e.g. period of record, seasonal availability) and proximity to the area of interest. For example a close proximity climate station with high quality data, but at high elevation may not be appropriate for evaluating a valley bottom aquifer due to orographic effects and greater snowfall, such that a climate station with poorer quality data but at a representative elevation might be more appropriate to use.

Estimates of actual evapotranspiration (AET) are often not available due to limited measurements. For this reason, the potential evapotranspiration (PET) is used in this assessment. PET, however, generally overestimates AET because it does not consider the available water. The PET values can be derived using the FAO Penman-Monteith method, which is considered one of the more comprehensive PET estimation methods (Hess 1996, Herrera-Pantoja and Hiscock 2008). The FAO Penman-Monteith method for reference crop evapotranspiration requires air temperature, wind speed, radiation, and humidity (Allen et al. 1998). The full suite of these parameters may not be readily available at some climate stations; therefore, it is possible to estimate PET from using a simplified approach that requires the daily solar radiation (SR) and maximum air temperature (T_max) (Equation 6.4) (Cohen et al. 2004).

\[-3.26 + 0.201T_{max} + 0.058SR = PET\] (6.4)

Solar radiation can be calculated for the days of the year using the solar position and radiation calculator (Washington State Department of Ecology, 2014), using longitude/latitude and elevation.
When summed over the year, PET can exceed precipitation, specifically in arid or semi-arid areas. For this reason, R was calculated daily. If precipitation occurred on a particular day, a recharge amount was computed according to Equation 6.3. If there was no precipitation, then R was assumed to be zero. This approach likely overestimates R, because soil moisture is able to evaporate and plants are able to transpire even on days it does not rain; however, for a Level II Assessment, recharge calculated in this way is a first approximation.

Using values for P (mm/yr) and the calculated values of PET (mm/year) for the aquifer area (m²), R or Q_R (m³/year) is estimated. The aquifer area corresponds to the area contributing to streamflow measured at a gauging station (see below). For simplicity, the aquifer area can be considered the same as the watershed or catchment area. This definition assumes that all the recharge within the watershed exits the watershed via the stream. Any deep groundwater flow is neglected.

**Summer Streamflow (Q_s)**

Q_R represents the volume of groundwater that discharges to the stream on an annual basis as baseflow. Ideally, the baseflow would be calculated from the same period of record as the climate normals. While there are hydrograph separation techniques that can be used to estimate the baseflow, which varies seasonally, the approach used here is to calculate the average summer streamflow, Q_s, (from July to September) over the period of record. In actuality, the summer streamflow will include the baseflow as well as storm runoff from rain events, and so may overestimate summer baseflow. But, countering this is the fact that summer baseflow is less than the average annual baseflow. Therefore, summer streamflow (Q_s) is considered a reasonable approximation to baseflow.

**Recharge Ratio (Q_s/Q_R)**

The Recharge ratio Q_s / Q_R represents the proportion of the summer streamflow that derives from groundwater recharge. There are three main outcomes for this ratio: 1) If the summer streamflow is fully dependent on groundwater recharge, the ratio will be one, and the stream would be considered sensitive to the amount of recharge in the aquifer. 2) If Q_s is larger than Q_R, then streamflow likely derives from an area remote to
the aquifer, such that the streamflow is augmented by upstream contributions. 3) if \( Q_S \) is smaller than \( Q_R \), then what small contributions of recharge to the streamflow there are must be significant, and the stream is considered sensitive. The rating scheme for Recharge Ratio is shown in Table 6.6. The maximum and minimum ratings were determined from the highest and lowest likely recharge ratios expected in British Columbia. The intermediate values were assigned according to order of magnitude changes in the recharge ratio to best capture the observed ranges during testing of the method.

Table 6.6. Recharge Ratio (\( Q_S/Q_R \)), and the assigned ratings.

<table>
<thead>
<tr>
<th>Ratio (( Q_S/Q_R ))</th>
<th>Rating</th>
</tr>
</thead>
<tbody>
<tr>
<td>&gt; 1000</td>
<td>1</td>
</tr>
<tr>
<td>&gt; 100</td>
<td>2</td>
</tr>
<tr>
<td>&gt; 10</td>
<td>3</td>
</tr>
<tr>
<td>1.0 - 9.9</td>
<td>4</td>
</tr>
<tr>
<td>0.1 - 0.9</td>
<td>5</td>
</tr>
<tr>
<td>0.01 - 0.09</td>
<td>6</td>
</tr>
<tr>
<td>0.001 - 0.009</td>
<td>7</td>
</tr>
<tr>
<td>0.0001 - 0.0009</td>
<td>8</td>
</tr>
<tr>
<td>0.00001 - 0.00009</td>
<td>9</td>
</tr>
<tr>
<td>&lt; 0.00001</td>
<td>10</td>
</tr>
</tbody>
</table>

**Other Considerations**

Climate varies spatially. In large watersheds or in watersheds with elevation changes, the precipitation (P) and temperature can be quite different from one area to another. For example, measurements of P at valley bottom climate stations generally underestimate P at higher elevation due to orographic effects. If climate is known to vary spatially, the climate data should be interpolated or zoned appropriately to estimate recharge (R).

Climate varies interannually. Precipitation and temperature vary from year to year, and at longer time scales due to climate oscillations such as the El Nino Southern Oscillation (ENSO), the Pacific Decadal Oscillation (PDO), among others, and is expected to have effects on groundwater and watershed hydrology in BC (Fleming and
Quilty 2006; Merritt et al. 2006; Scibek and Allen 2006a; Pike et al. 2010). Recharge calculations could incorporate potential climate variability by using historic records where available. Recharge calculated using historic high and low values rather than annual averages would provide a means to assess the sensitivity of recharge to climate variability in the assessment. Many methods are discussed in the literature for estimating climate change impacts on groundwater recharge; however, adopting these approaches is non-trivial and would best be carried out under a Level III Assessment. Some areas of BC have climate change impacts on recharge assessed and these estimates of future recharge could be used in a Level III Assessment. (e.g. Scibek and Allen 2006 (Grand Forks); Toews et al. 2009 (Oliver); Foster and Allen 2015 (Cowichan Watershed). In areas where climate change impacts are identified as a factor contributing to high stream susceptibility, a Level III Assessment is recommended to address site specific outcomes.

Summer streamflow (Q_S) varies interannually. For similar reasons as above for recharge, a range of Q_S values corresponding to the same years used to estimate the range of Q_R values (as above) could be used.

The rating for the recharge component of stream sensitivity could be based on other criteria. For example, if the instream flow needs (Q_{INF}) for a particular stream are known, Q_R could be compared to Q_{INF}.

Other measures of the relative importance of groundwater recharge to streamflow include the baseflow index (BFI), which is defined as the ratio of annual baseflow in a river to the total annual runoff. BFI values for different streams could be compared in different areas of the province and these values used to rate different streams. The baseflow of the stream can also be estimated using simple hydrograph separation techniques rather than Q_S, and use of an alternate method is required if the low flow period being evaluated is not the summer period.

**Stream Susceptibility (SS) Rating**

The Stream Susceptibility (SS) rating is calculated as the product of the Aquifer Characteristics (A) rating and the Recharge Ratio (Q_S/Q_R) rating. The overall SS rating
will range from 3 – 100. The rating scheme for Stream Susceptibility is shown in Table 6.7.

Table 6.7. Stream Susceptibility (SS), and the assigned ratings.

<table>
<thead>
<tr>
<th>Aquifer Characteristics (A)</th>
<th>Recharge Ratio (Qd/Qr)</th>
<th>Low (1-3)</th>
<th>Medium (4-7)</th>
<th>High (8-10)</th>
</tr>
</thead>
</table>

\(^1\)Ranges for Aquifer Characteristics (A) are continuous in this table, but in Table 6.5 they are not continuous.
**Hazard (H)**

The Hazard (H) component of stream vulnerability represents the primary stressor to the aquifer system, i.e., pumping (Equation 6.5). For the purpose of a Level II Assessment, the H component is considered representative of current conditions. Future stressors are evaluated in a Level III Assessment, and could include land use/land cover changes, climate variability and climate change, which can lower the net recharge, and increased groundwater extraction.

H represents the magnitude and likelihood that the hazards that may change the water quantity in the stream:

\[ \text{Hazard (H) = Groundwater Pumping Magnitude} \times \text{Likelihood of Impact} \]  

The Groundwater Pumping Magnitude is assessed based on the volumetric pumping rate. The Likelihood of Impact is based on the ratio of the pumping volume to the recharge to the stream. The volumetric pumping rate is assessed for either the area of aquifer polygon, or the area of the stream watershed.

The annual volume of groundwater pumped is then compared to the Recharge to stream \( (Q_R) \), as calculated in Equation 6.3. If \( Q_P \) is equal to, or greater than, \( Q_R \), the pumping is very likely impacting the streamflow quantity and represents a hazard. If \( Q_P \) is less than the \( Q_R \), pumping may not be impacting the stream; however, the magnitude of the ratio between the two components provides an indication of the condition of the system.

**Other Considerations**

For establishing water management guidelines or policies, a sensitivity analysis should be conducted whereby the effect on the assessment results are compared for different Hazard magnitudes (increasing the number of wells in the zone). This is necessary for two reasons: 1) The Province estimates that perhaps only 50% of wells are recorded in the WELLS database; therefore, the number of active wells may be significantly underestimated; and 2) The assessment would better reflect how sensitive
the results are for current conditions. If there is a noticeable change in H rating, then the system is particularly sensitive to the number of wells.

The Level II Assessment utilizes the actual pumping rate (if known) or the estimated yield of the well, when reported in the WELLS database. If no information is available from the WELLS database on estimated well yield, then the well can be assumed to be pumped at the domestic rate of 2,270 L/day, which is defined within the BC Well Protection Toolkit as the estimated water use per household (BC Ministry of Environment 2004). If possible, all wells in the area of interest should be included. A door-to-door survey may be required to identify the well location and the pumping rate.

The Level II Assessment assumes that the pumping rate is constant. Seasonal changes in water use are not accounted for. The Level II Assessment also assumes all the groundwater pumped is removed from the aquifer. Return flows (e.g. irrigation and septic fields) are not accounted for.

**Hazard (H) Rating**

The Hazard (H) rating is derived directly from the $Q_p/Q_R$ ratios. Table 6.8 shows the Hazard (H) rating for a range of $Q_p/Q_R$ ratios. Intermediate ratings are scaled accordingly.

<table>
<thead>
<tr>
<th>Ratio ($Q_p/Q_R$)</th>
<th>H Rating</th>
</tr>
</thead>
<tbody>
<tr>
<td>&lt; 0.19</td>
<td>Low (1)</td>
</tr>
<tr>
<td>0.2 – 0.39</td>
<td>Low (2)</td>
</tr>
<tr>
<td>0.4 – 0.59</td>
<td>Medium (4)</td>
</tr>
<tr>
<td>0.6 – 0.79</td>
<td>Medium (6)</td>
</tr>
<tr>
<td>0.8 – 0.99</td>
<td>High (8)</td>
</tr>
<tr>
<td>&gt; 1</td>
<td>High (10)</td>
</tr>
</tbody>
</table>

**6.4.3. Final Level II Assessment for Stream Vulnerability (SV)**

The Stream Vulnerability (SV) ratings can range from low to high, based on the Stream Susceptibility rating and the Hazard rating in Tables 6.7 and 6.8, respectively.
Table 6.9 shows the Stream Vulnerability ratings as a matrix, which captures both components of the assessment.

Table 6.9. Stream Vulnerability (SV) matrix.

<table>
<thead>
<tr>
<th>Hazard</th>
<th>Stream Susceptibility</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low</td>
<td>Low</td>
</tr>
<tr>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td>High</td>
<td>Medium</td>
</tr>
</tbody>
</table>

Table 6.10 describes the whether or not further assessment is required based on the stream vulnerability rating.

Table 6.10. Stream Vulnerability (SV) rating and assessment required.

<table>
<thead>
<tr>
<th>Stream Vulnerability Rating</th>
<th>Description</th>
<th>Action Required</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low</td>
<td>The stream is currently of low vulnerability.</td>
<td>No further action required unless there is a significant change to the water demand. A Level II Re-Assessment would then be required.</td>
</tr>
<tr>
<td>Medium</td>
<td>The stream is currently of moderate vulnerability.</td>
<td>No further action required unless there are changes to the water demand or the recharge conditions. A Level II Re-Assessment would then be required.</td>
</tr>
<tr>
<td>High</td>
<td>The stream is currently of high vulnerability.</td>
<td>Proceed to Level III Assessment</td>
</tr>
</tbody>
</table>
6.4.4. Example Level II Assessment

Level II Assessments were completed for nine streams in BC to represent different aquifer-stream settings in the province (Figure 6.9).

- **Fishtrap and Bertrand Creeks** - These creeks drain the Abbotsford aquifer in the Lower Fraser Valley in the Abbotsford aquifer. The aquifer-stream system is diffuse recharge-driven in a rainfall dominated hydroclimatic regime. Topographic relief is low. The aquifer is comprised of sands and gravels.

- **The Kettle River Section at Grand Forks** – The Kettle River meanders through the Grand Forks valley in south-central BC. The river originates at high elevation remotely to the valley. The aquifer-stream system is stream-driven in a snowmelt-dominated hydroclimatic regime. Topographic relief in the valley is low. The aquifer is comprised of sands and gravels.

- **Daves Creek** – The creek is situated in Okanagan Basin. The aquifer-stream system is diffuse recharge-driven in a snowmelt-dominated hydroclimatic regime. Topography is steep. The aquifer is comprised of bedrock.

- **Upper Mission Creek** – The creek is situated in Okanagan Basin. The aquifer-stream system is stream-driven in a snowmelt-dominated hydroclimatic regime. In this section of Mission Creek, topography is moderately steep. The stream incises a mostly confined aquifer comprised of sands and gravels.

- **Fulford Creek** – The creek is situated on Salt Spring Island and has been designated as a Sensitive Stream. The aquifer is diffuse recharge-driven in a rainfall-dominated hydroclimatic regime. Topography is moderately steep. The aquifer is comprised of bedrock.

- **Cowichan River section in the lower Cowichan Valley** – The river is situated in Cowichan Valley, Vancouver Island. The river originates at high elevation in the valley. The aquifer-stream system is stream-driven in a snowmelt-dominated hydroclimatic regime. Two aquifers were assessed for comparison:
  - Aquifer 179 – Topography is moderately steep. Well records for the aquifer indicate 62 known wells. The aquifer is comprised of sands and gravels.
  - Aquifer 186 – Topography is low. Well records for the aquifer indicate 222 known wells. The aquifer is comprised of sands and gravels.

- **Kiskatinaw River section** – The Kiskatinaw River bounds the west and northwest side of the aquifer. The aquifer-stream system is stream-driven in a snowmelt-dominated hydroclimatic regime. Topography is low. The aquifer is comprised of bedrock.
For each example, climate data were obtained from the nearest Environment Canada climate station. Daily recharge was estimated from daily precipitation minus PET for days where precipitation occurred. With the exception of the Grand Forks River example, the watershed area upstream of the gauge was used for calculating $Q_R$. $Q_S$ was calculated over the period of record for the nearest stream gauge downstream. The summary of the steps of the Level II Assessment are presented in Table 6.10.

Fishtrap and Bertrand Creeks are very similar in their physical settings, locations, and size (same Aquifer Characteristics rating); however, the lower discharge and higher pumping volume at Bertrand Creek leads to a higher overall Stream Vulnerability.

The Cowichan River aquifers are very similar in their physical settings and have the same Aquifer Characteristic ratings and Stream Susceptibility; however, the volume pumped from the aquifer is greater in Aquifer 186 and leads to a high Stream Vulnerability, compared with a low rating for Aquifer 179.
Fulford Creek is designated as a Sensitive Stream under the Fish Protection Act, but was rated as having a low Stream Vulnerability in this assessment. However, surface water extraction volumes, and fish population status and habitat conditions are other key components to the Sensitive Stream Designation and are not included in this assessment method, which is focuses on sensitivity to changes in groundwater.

A Level III Assessment would be recommended for Fishtrap Creek, Bertrand Creek, Kettle River at the Grand Forks aquifer, Daves Creek, Mission Creek, and Cowichan River Aquifer 186; all of which have high Stream Vulnerability ratings. The Stream Vulnerabilities are low for Fulford Creek, Cowichan River Aquifer 179 and the Kiskatinaw River, and no further action is required unless there is a change in the recharge or volume pumped, at which time a Level II Re-assessment would be required.
## Table 6.11. Example of Level II Assessments for nine streams.

<table>
<thead>
<tr>
<th></th>
<th>Fishtrap Creek</th>
<th>Bertrand Creek</th>
<th>Kettle River</th>
<th>Daves Creek</th>
<th>Mission Creek</th>
<th>Fulford Creek</th>
<th>Cowichan River</th>
<th>Kiskatinaw River</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aquifer #</td>
<td>015</td>
<td>015</td>
<td>158</td>
<td>473</td>
<td>461</td>
<td>722-723</td>
<td>179</td>
<td>186</td>
</tr>
<tr>
<td>Type</td>
<td>4A</td>
<td>4A</td>
<td>1B</td>
<td>6B</td>
<td>4B</td>
<td>6B</td>
<td>1B</td>
<td>IA</td>
</tr>
<tr>
<td>Aquifer Characteristics Rating (A)</td>
<td>8</td>
<td>8</td>
<td>10</td>
<td>3</td>
<td>4</td>
<td>3</td>
<td>10</td>
<td>10</td>
</tr>
<tr>
<td>Precipitation (mm/yr)&lt;sup&gt;a&lt;/sup&gt;</td>
<td>1619.5</td>
<td>1619.5</td>
<td>552.7</td>
<td>313</td>
<td>414.82</td>
<td>997.7</td>
<td>1379.1</td>
<td>1379.1</td>
</tr>
<tr>
<td>PET (mm/yr)&lt;sup&gt;b&lt;/sup&gt;</td>
<td>479.5</td>
<td>478.5</td>
<td>579.3</td>
<td>313</td>
<td>538.3</td>
<td>424.1</td>
<td>499</td>
<td>499</td>
</tr>
<tr>
<td>Recharge (mm/yr)&lt;sup&gt;c&lt;/sup&gt;</td>
<td>1563.3</td>
<td>1518.3</td>
<td>445.2</td>
<td>566.9</td>
<td>326.3</td>
<td>941.8</td>
<td>1305.2</td>
<td>1305.2</td>
</tr>
<tr>
<td>Area of watershed/aquifer (km²)</td>
<td>37.0</td>
<td>51.0</td>
<td>38.8</td>
<td>37.2</td>
<td>15.1</td>
<td>21.1</td>
<td>7.6</td>
<td>16.9</td>
</tr>
<tr>
<td>Q₀ (m³/yr)&lt;sup&gt;d&lt;/sup&gt;</td>
<td>57.8</td>
<td>77.4</td>
<td>41.3</td>
<td>21.1</td>
<td>4.93</td>
<td>19.9</td>
<td>9.9</td>
<td>22.1</td>
</tr>
<tr>
<td>Q₀ (m³/yr)&lt;sup&gt;d&lt;/sup&gt; *10&lt;sup&gt;6&lt;/sup&gt;</td>
<td>1.80</td>
<td>0.29</td>
<td>401.</td>
<td>88.</td>
<td>34.8</td>
<td>4.65</td>
<td>683.</td>
<td>683.</td>
</tr>
<tr>
<td>Q₀/Q₀</td>
<td>0.031</td>
<td>0.004</td>
<td>23.214</td>
<td>0.042</td>
<td>7.063</td>
<td>0.234</td>
<td>68854</td>
<td>30.964</td>
</tr>
<tr>
<td>Recharge Ratio Rating (Q₀/Q₀)</td>
<td>6</td>
<td>7</td>
<td>3</td>
<td>6</td>
<td>4</td>
<td>5</td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>Stream Susceptibility (SS)</td>
<td>48</td>
<td>56</td>
<td>30</td>
<td>18</td>
<td>16</td>
<td>15</td>
<td>30</td>
<td>30</td>
</tr>
<tr>
<td>Q₀ (m³/yr)&lt;sup&gt;e&lt;/sup&gt;</td>
<td>240.</td>
<td>122.</td>
<td>200</td>
<td>128</td>
<td>6.42</td>
<td>8.78</td>
<td>3.54</td>
<td>308</td>
</tr>
<tr>
<td>n (number of wells)</td>
<td>856</td>
<td>828</td>
<td>611</td>
<td>188</td>
<td>57</td>
<td>204</td>
<td>62</td>
<td>222</td>
</tr>
<tr>
<td>Q₀/Q₀</td>
<td>4.15</td>
<td>1.58</td>
<td>11.58</td>
<td>6.07</td>
<td>1.30</td>
<td>0.44</td>
<td>0.36</td>
<td>13.96</td>
</tr>
<tr>
<td>Hazard Rating (H)</td>
<td>8</td>
<td>8</td>
<td>10</td>
<td>8</td>
<td>8</td>
<td>4</td>
<td>2</td>
<td>10</td>
</tr>
<tr>
<td>Stream Vulnerability (SV)</td>
<td>High</td>
<td>High</td>
<td>High</td>
<td>High</td>
<td>High</td>
<td>Low</td>
<td>Low</td>
<td>High</td>
</tr>
</tbody>
</table>

<sup>a</sup> The climate data were for the period spanning 1990-2002 based on availability;
<sup>b</sup> Estimated from Equation 6.5;
<sup>c</sup> Recharge calculated only for days when precipitation occurred;
<sup>d</sup> Stream discharge data were for the summer periods (July – Sept.) spanning 1980 to 2012 based on availability.
Other Considerations

The Level II Assessment may require some assumptions or simplifications, based on the availability of data or the complexity of the aquifer-stream system. In completing the Level II Assessment for the nine example streams, some of complexities were encountered. These are described below. The list is not exhaustive, but rather acknowledges some of the challenges.

- Some streams intersect multiple aquifers.
  - For Fulford Creek, both aquifers (Aquifers 722 & 723) are classified as bedrock. These aquifers were merged into a single shapefile in ArcGIS for this assessment.
  - For Fishtrap and Bertrand Creeks, the dominant aquifer (Aquifer 015) was selected as the aquifer of interest.
  - For the Kiskatinaw River, the aquifer with the greatest potential connectivity to the stream was selected (Aquifer 593). Here, some of the aquifer polygons were unconfined, while others were confined or located at depth.

- The recharge area for the aquifer-stream area can be defined by the watershed area or the aquifer area.
  - The contributing area for a low order stream can be readily defined as the watershed area (e.g. Daves Creek).
  - For higher order streams, and for systems where the flow originates remotely (such as the Kettle River), the aquifer area will be the more appropriate area to use.
  - For bedrock aquifers, it is important to note that the mapped aquifer boundaries are defined according to whether wells are present or not. The bedrock extends beyond the mapped boundary. Therefore, the contributing recharge area must be carefully assessed based on available data (e.g. topography, geology).

- Some large aquifers may be bounded by multiple streams (e.g. Kiskatinaw River). In these situations, recharge to the aquifer does not discharge to a single stream. Therefore, the $Q_S/Q_R$ ratio calculated in this assessment is likely underestimated and would require adjustment in the Stream Vulnerability calculation.

- Data periods for the climate and the stream flow should be selected for the same time span for comparison. For these example assessments, ten-year periods were used when available.

- To calculate the recharge, the latitude, longitude, and elevation are required as a single point. For the calculations, a middle point in the aquifer or watershed was selected.
6.5. Level III Assessment - Stream Impact

A Level III Assessment aims to quantify the impacts to the stream from groundwater-related stressors. The Level III Assessment evaluates streams that have been identified as having a potential connectivity with the surrounding aquifer and where the aquifer productivity may be insufficient to meet the current demand (Level I Assessment), and where the stream is determined to be highly vulnerable to stressors (Level II Assessment). Level III Assessments are quantitative in that they require more specific information about how the stream-aquifer system functions as well as the magnitude of the stressors acting on the system. The stressors can include, for example, groundwater pumping from wells in zones adjacent to a susceptible stream and/or changes in recharge due to land use/land cover change, climate variability or climate change. Level III Assessments are site specific and are intended for such activities as drought preparedness, groundwater licensing, planning of subdivisions, etc.

The main objective of a Level III Assessment is to demonstrate the likely impact on a stream due to stressors acting on the aquifer system. Therefore, quantitative assessment tools are needed. Such tools can range from simple analytical methods to sophisticated numerical hydrogeological models. The range of possible tools is quite large. Two examples are used to demonstrate how certain tools could be used for Summer Low Flow Impact Assessment: 1) a simple method based on the fixed radius capture zone of a single well, and 2) a numerical groundwater flow model of an aquifer.

6.5.1. Well Scale: Well Capture Zone Analysis

Delineation of well capture zones is a critical component of any well vulnerability study. In the simplest of terms, a well capture zone visually represents the area (or volume) of aquifer that the well receives its water from (Hemmer and Beach 1997; BC Ministry of Environment 2004). By defining a well capture zone, decisions can be made to mitigate risk to the well, for example, eliminating hazardous land use activities within the capture zone area that may result in the well becoming contaminated. In the context of interactions with streams, well capture zones identify if the well likely receives water from the stream as it pumps. Capture zones normally are constructed for different periods (e.g. 60 day capture zone, 5 year capture zone, etc.) to reflect the time of travel.
time of the water (or contaminant). For the purpose of a Level III Assessment, the time of travel is equivalent to the summer low flow period (in days).

Three methods of increasing complexity can be used to estimate well capture zones: 1) calculated fixed radius, 2) analytical calculations, and 3) numerical modeling. These methods can be used to assess if there are impacts potentially occurring at the interface of the stream and the capture zone, but do not provide information regarding the magnitude of the impact. To determine the magnitude of the impact, a numerical modeling approach would be needed.

A novel application of the calculated fixed radius capture zone analysis is described below. It extends the traditional capture zone method specifically for quantifying the maximum pumping rate that could be accommodated by a well situated close to a stream if the summer low flow period is longer than normal.
**Calculated Fixed Radius Capture Zone**

The calculated fixed radius capture zone method (CFR) is a simplified approach to calculate the radius of the circular groundwater contribution area related to a pumping well (Equation 6.6). The CFR method equates the volume pumped to the volume of a cylinder (Figure 6.10), and from this, the radius of the capture zone ($R_o$) can be calculated based on the method described by Hemmer and Beach (1997):

$$ R_o = FS \sqrt{\frac{Qt}{\pi n H}} $$  \hspace{1cm} (6.6)

where:

- $FS$ is the factor of safety to related to uncertainty in the parameters ($FS = 1.3$ when all parameters are known, and $FS=1.5$ when one or more parameters are not known). The factor of safety is unitless.
- $Q$ is the pumped flow rate ($\text{m}^3/\text{d}$);
- $t$ is the time of travel for the period of the annual summer low flow period (days);
- $\pi$ is $\pi = 3.1416$;
- $n$ is the aquifer system porosity (unitless);
- $H$ is the screened length of the well (m).

![Figure 6.10. A schematic of the fixed capture radius method, showing the pumping well, at the center of the cylindrical capture zone with a radius of $R_o$.](image-url)
The purpose of finding \( R_o \) is to determine if the capture zone from a pumping well situated a distance \( D \) from a stream, pumping at a constant rate \( Q \) for the duration of the annual summer low flow period \( t \), will intersect, and thereby divert water from, an adjacent stream. The ratio \( R_o/D \) indicates the position of the capture zone relative to the stream (Figure 6.11).

If \( \frac{R_o}{D} \ll 1 \) then there is no impact to the stream;

If \( \frac{R_o}{D} = 1 \) then there is an exact impact which is the point at which the effect of the capture zone, and thus the impact, just reaches the stream; and

If \( \frac{R_o}{D} \gg 1 \) then impacts are likely significant.

The ratio \( R_o/D \) changes with the pumping rate, and this can be plotted to determine the maximum pumping rate for a well in a zone adjacent to a stream. Within the calculation of \( R_o \) there is uncertainty in the aquifer properties, namely heterogeneity, and there may be uncertainty in the well parameters also, and these are incorporated into the equation through the Factor of Safety.

Figure 6.11. A pumping well is positioned at a distance, \( D \), from the stream. The dashed circles represent circular capture zones with different values of \( R_o \).

The ratio \( R_o/D \) is related to both the pumping rate \( Q \), and the time of pumping \( t \) which is the length of the low flow period in days \( t \). There is growing concern that climate change may result in an extended period of more extreme low flows (Moore et al. 2007). This would increase the number of low flow days, such that for a well pumping at a constant rate \( Q \), \( R_o \) would be larger. Therefore, climate change could increase the impacts to streams from pumping wells in adjacent zones.
To illustrate the impacts of an extended summer low flow period, Figure 6.12 shows the ratio $R_o/D$ as a function of the pumping rate ($Q$) for a simple system. In this system, the pumping well is completed in an unconsolidated aquifer, with an estimated porosity of 0.25. The well has a screened interval of 3 m and is positioned 30 m away from the stream. The factor of safety applied to the calculation for $R_o$ is 1.5, given the uncertainty in porosity. The scenario shown in Figure 6.12 shows the results for two summer low flow periods: 77 days, and 122 days.

Based on Figure 6.12, the maximum pumping rate for the low flow period of 77 days is 12.2 m$^3$/day. However, if the pumping time is increased to 122 days, the safe pumping rate drops to a maximum of 7.7 m$^3$/day. Therefore, as the length of the low flow period increases the pumping rate should be decreased or there will likely be impacts to the stream.

Figure 6.12. Ratio $R_o/D$ as a function of pumping rate ($Q$) in a well adjacent to a stream. The blue line indicates a low flow period of 77 days with safe pumping rate of 12.2 m$^3$/d, and the orange line represents 122 days, with a safe pumping rate of 7.7 m$^3$/d.
6.5.2. Aquifer Scale Case Study

A Level III Assessment may be carried out to quantify the magnitude of impact due to one or a combination of stressors. At the scale of an aquifer or watershed, the assessment is likely complex. One option for this type of assessment is to develop a numerical model to quantify the magnitude of impact from the stressors.

Two of the nine streams presented in the example of Level I and Level II Assessments (Table 6.11) are Fishtrap and Bertrand Creeks. The Level II Assessment indicated that both streams are rated as High Stream Vulnerability (Table 6.11). The result of this rating is a recommendation for the completion of a Level III Assessment to quantify the impact to the streams from the groundwater pumping. This section presents an example of a Level III Assessment for the Fishtrap and Bertrand Creek watersheds in the Lower Fraser Valley using an existing groundwater flow model for the Abbotsford-Sumas aquifer and incorporating field indicators. In this example, the Level III Assessment is used to show:

1. Annual recharge in the model is compared to recharge estimated in Level II Assessment (Table 6.11);
2. Zone Budget results for the two streams presently under non-pumping conditions. Using the Zone Budget results, the baseflow simulated in the model is compared to observed summer discharge in the Level II Assessment; All wells within each watershed are turned on in the model and the Zone Budget results under pumping conditions are compared to non-pumping conditions to quantify the impact of pumping to each stream;
3. Zone Budget results are compared for a selection of pumping conditions to evaluate how different configurations of pumping wells impact the zone budgets for each watershed;
4. Field indicator measurements are presented as a method to monitor the aquifer-stream connectivity, and indicators are used as a tool to support the findings of the Zone Budget results; and,
5. Results of the regional field monitoring are used to provide an example of integration of indicators and a Level III Assessment into a risk management framework.

The Numerical Groundwater Flow Model

A regional steady state numerical groundwater model was developed for the Abbotsford-Sumas aquifer by Scibek and Allen (2005). The model was constructed in Visual MODFLOW (Scibek and Allen, 2005; Scibek, 2005) and has subsequently been
used in several studies (with some modifications\textsuperscript{7}), including groundwater-surface water interactions (Pruneda et al. 2010), nitrate transport (Chesnaux et al. 2007), and potential impacts of climate change on groundwater (Scibek and Allen 2006). The streams are delineated as main-stem segments and ephemeral channel segments. The boundary conditions used to represent the streams are a combination of river boundary conditions and drain boundary conditions. For those portions of the streams with river boundary conditions assigned, the head is maintained at a fixed level intended to represent summer baseflow conditions (Scibek and Allen 2005). Those portions of the streams assigned as drain boundaries have no fixed head – they simply act as drains. The recharge is applied as a monthly average and is spatially distributed based on the zonation of precipitation, and the soil cover, slope and vegetation type. Recharge was modeled separately using the US EPA code HELP (see Scibek and Allen, 2005 for details). Recharge is net recharge and incorporates evapotranspiration. The model was originally calibrated to observed historical static heads in existing domestic wells (Scibek and Allen 2005). Figure 6.13 shows the model area, stream main-stem and ephemeral segments, the watersheds, and the pumping wells within each watershed. Details concerning the model development are given in Scibek and Allen (2005).

\textsuperscript{7} Modifications include minor domain adjustments to truncate the model at the Nooksak River in Washington; implementing time varying recharge for transient models; converting constant head boundary conditions for streams to river boundary condition; and other minor adjustments specific to the various studies.
Figure 6.13. MODFLOW model of the Abbotsford-Sumas aquifer. The model area is shown in white. The green area represents de-activated cells. The Canadian portion of the Fishtrap and Bertrand watersheds are outlined and, for the purpose of this case study, terminate along the southern extent at the International border (yellow line). Stream main-stem segments are shown in blue, and ephemeral segments are red. The pink points are the pumping wells within each watershed.

Zone Budget in MODFLOW uses the MODFLOW simulation results to calculate water budgets through all the layers in the model. For the Level III Vulnerability Assessment, the zone budget zones are defined to focus entirely on the Canadian portion of the Fishtrap and Bertrand Creek watersheds. Figure 6.14 shows an example of the zone budget zones for layer 1 of the model. In total, eight zones were defined and are detailed in Table 6.12. The area outside the watershed zones was assigned as a single zone (Zone 1) through all model layers. Each watershed was assigned a zone based on the watershed Canadian boundaries (Zone 2 for Bertrand and Zone 9 for Fishtrap, through all model layers). Within each watershed, zones were applied to specific cells assigned as drain and river boundaries along main channel reaches for each stream. The ephemeral reaches of the stream were also assigned separate zones, because those are considered most likely to have the lowest flow conditions in the summer periods. The constant head boundaries associated with the lakes in the Fishtrap Creek watershed were assigned to Zone 7.
All simulations were transient – January to December with monthly stress periods to reflect monthly recharge variability. Because the simulation was transient, an initial condition had to be specified. This initial condition was head distribution derived from a steady-state simulation that used the mean annual recharge. Monthly water balance outputs were summed annually. The mass balance for flow for each simulation ranged from 0 to 1% for all simulations, except for a single step with a value of 18%, with an average discrepancy of 0.15% indicating that a satisfactory water balance closure had been achieved for the model as a whole.

Figure 6.14. Zone Budget zones for the Level III Assessment, showing layer 1 in the model. Green is de-activated cells, white (Zone 1) is the background zone representing the model area outside of the watersheds. The variably coloured zones along the stream segments and lakes correspond to the zones described in Table 6.12.

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8 Each month comprised a stress period. A cumulative monthly water balance was calculated by summing the water balance output for all time steps in each stress period.
Table 6.12. The MODFLOW Zone Budget zones for a Level III Vulnerability Assessment of Fishtrap and Bertrand Creek watersheds.

<table>
<thead>
<tr>
<th>Zone ID</th>
<th>Area Represented</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Aquifer area outside of Fishtrap and Bertrand Creek Watersheds</td>
</tr>
<tr>
<td>2</td>
<td>Bertrand Creek Watershed (blue in Figure 6.14)</td>
</tr>
<tr>
<td>3</td>
<td>Fishtrap Creek Watershed (grey in Figure 6.14)</td>
</tr>
<tr>
<td>4</td>
<td>Bertrand Creek main reaches</td>
</tr>
<tr>
<td>5</td>
<td>Fishtrap Creek main reaches</td>
</tr>
<tr>
<td>6</td>
<td>Bertrand Creek ephemeral reaches</td>
</tr>
<tr>
<td>7</td>
<td>Constant head boundaries (lakes) (yellow in Figure 6.14)</td>
</tr>
</tbody>
</table>

Comparing Annual Recharge and Baseflow under Non-Pumping Conditions

The estimated annual recharge and baseflow volumes under non-pumping conditions are compared for the Level II and Level III Assessments (Table 6.13). The results are discussed in the following subsections.

Table 6.13. Comparison of the annual recharge and baseflow estimated for the Fishtrap and Bertrand Creek watersheds in the Level II and Level III Assessments.

<table>
<thead>
<tr>
<th>Watershed</th>
<th>Assessment Level</th>
<th>Recharge (QR) (m³/yr) *10⁶</th>
<th>Baseflow (QS) (m³/yr) *10⁶</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fishtrap Creek</td>
<td>Level II</td>
<td>57.8</td>
<td>0.29</td>
</tr>
<tr>
<td></td>
<td>Level III</td>
<td>39.3</td>
<td>2.90</td>
</tr>
<tr>
<td>Bertrand Creek</td>
<td>Level II</td>
<td>77.4</td>
<td>0.11</td>
</tr>
<tr>
<td></td>
<td>Level III</td>
<td>41.4</td>
<td>2.48</td>
</tr>
</tbody>
</table>

Annual Recharge

The estimated annual recharge in the Level II Assessment is slightly lower, but comparable to the annual recharge in the Level III Assessment (Table 6.13, Figure 6.15). Both methods use the same watershed area, but the approach differs. The annual recharge estimate in the Level II Assessment is calculated using a simplified water balance approach. The input parameters are generalized regional daily climate inputs (precipitation, solar radiation, and maximum air temperature) derived from the best available data for the area from the nearest climate station. The recharge is not
spatially distributed and does not account for ground or near surface geological conditions, such as vegetation type, soil, slope or aquifer material. In contrast, the applied monthly recharge in the Abbotsford-Sumas model was modelled spatially using the US EPA HELP code, which accounts for the spatial distribution in these various parameters.

![Bar chart](image)

**Figure 6.15.** Recharge values estimated for the Fishtrap and Bertrand Creek watersheds in the Level II and III Assessments.

**Baseflow**

The baseflow estimates for both creeks in the Level III Assessment method are higher than in the Level II Assessment estimates by approximately one order of magnitude (Table 6.13, Figure 6.16). In the Level II Assessment, the estimated baseflow is the average observed summer discharge (from July to September – 3 months) measured at the respective gauging stations at the international border. There may be summer stormflow or bank storage contributions or contributions from irrigation return flow, and pumping impacts from activities within the watershed. In the Level III Assessment, baseflow is estimated as sum of the Zone Budget flux out of the river and drain boundary condition cells along the stream segments north of the border, over the summer period (model time steps - days 182 to 274). In the model, the calculated baseflow relies on simplifications and assumptions inherent in modelling, and is
inherently uncertain. First, the river boundary conditions used to represent some portions of the streams fix (hold constant) the head (stage) values throughout the simulation. This is likely the greatest limitation of the model. Second, the model is designed as a groundwater flow model; therefore, the flux through the river and drain cells functions to moderate groundwater levels. There is no contribution from stormflow or bank storage or irrigation return flow (no irrigation applied in the model) – the only source is groundwater. A benefit of the model is that the model domain is not limited to the actual watershed and there may be contributions from adjoining watersheds, and non-pumping conditions can be simulated.

![Baseflow](image)

**Figure 6.16.** Baseflow estimates for Fishtrap and Bertrand Creek from the Level II and Level III Assessments. The baseflow estimate in the Level II Assessment is the observed summer discharge, and in the Level III Assessment, the baseflow is estimated from the outflow cells corresponding to river and drain cells along the stream segments.

**Comparing Pumping Impacts to the Stream Zones**

The impacts of pumping on the streams were evaluated by simulating a series of pumping scenarios for each watershed and comparing these to the non-pumping scenario. For the pumping models, the wells with known or estimated pumping rates used in the Level II Assessment were added to the model. Wells were added as groups; first, for each watershed individually, then in combination, and finally the remaining wells
across the entire model area. The rationale for the estimated pumping rates is provided in the Example Level II Assessment Section. Zone Budget was used to calculate the annual fluxes into and out of the various zones during non-pumping and pumping conditions.

The Zone Budget results are shown for Fishtrap and Bertrand Creek watersheds in Figures 6.17 and 6.18, respectively. Each figure compares the volume of water out of the zones (the watershed itself, the main-stem and the ephemeral streams) by boundary type for the non-pumping and pumping scenarios. The water flow occurs through the river and drain cells or to other zones (for both the non-pumping and pumping scenarios), and out through wells for the pumping scenarios. In both watersheds, there is an increase in the total volume out of the watershed zone during pumping, which can be attributed to the volume pumped (red columns in Figures 6.17 and 6.18). As a consequence, the volume of water out of the stream segments is lower during pumping. In Fishtrap Creek (Figure 6.17) the volume of water out of the main-stem decreased by approximately 27%, and in the ephemeral segments by 37% relative to non-pumping conditions. In Bertrand Creek (Figure 6.18), the decrease in outflow along the main-stem channel was approximately 25%, and in the ephemeral segments 31%. Some water was also lost directly from Bertrand Creek main-stem by pumping (Figure 6.18).

This information can be directly applied to address water quantity management objectives for the watersheds, and for risk management. For example the estimated decrease in flow could be compared to in-stream flow for fisheries needs to ensure thresholds are not exceeded. For risk management, a sensitivity analysis in the model could be run to determine groundwater management zones in riparian areas and well set back distances for groundwater pumping to manage the decrease in flows.
Figure 6.17. Comparison of the Zone Budget results for the non-pumping (left) and pumping (right) scenarios for Fishtrap Creek. The annual volumes shown represent the flow out of the model for the watershed zone, and the stream segments.
Figure 6.18. Comparison of the Zone Budget results for the non-pumping (left) and pumping (right) scenarios for Fishtrap Creek. The annual volumes shown represent the flow out of the model for the watershed zone, and the stream segments.
Comparing Impacts for Non-Pumping and Pumping

To compare the impact of pumping on the system as a whole, the model was run with Zone Budget to evaluate the changes in the water balance in each watershed (Fishtrap and Bertrand) for each of the following scenarios.

1. No pumping in either watershed;
2. Wells activated only in the specified watershed;
3. Wells activated only in the adjacent watershed;
4. Wells activated in both watersheds; and,
5. All wells activated in the model area.

The results of these simulations are discussed in detail in the Appendix F. The results show that pumping within the specified watershed is important for the water budget of each stream, as expected. However, the effects of pumping from a more widespread area such as the adjacent watershed, or cumulative impacts from pumping over the larger aquifer area have minimal additional impact on the water balance for each stream. These results suggest that data acquisition for the groundwater can be focused on the area of assessment, defined as either the watershed, or the aquifer intersecting the groundwater-dependent stream. In this case study, the streams are located within a larger regional aquifer setting; however, the watershed boundary was found to be an appropriate area for delineating the vulnerability assessment for the groundwater dependent stream.

Regional Monitoring: Field Indicators

The results of the numerical modelling were compared with field data collected at the regional sites, throughout Fishtrap Creek watershed and at the Bertrand Creek site (see Chapter 2 for locations and methodology). The data collected regionally include manual discharge measurements, sediment-water interface temperatures, and water chemistry parameters. The field methodology is presented in Chapter 2, and the data are tabulated in Appendix A for each collection period in 2009, 2010, and 2012. The regional data are used here support interpretations of aquifer-stream connectivity and zones of potential groundwater contributions (baseflow). The results are also described in the context of supporting the Zone Budget results from the numerical model. For each parameter discussed for the regional monitoring, the most spatially complete data are presented in Figures 6.18 through Figure 6.23. Unfortunately, data for these parameters
were not all represented within the same periods and therefore are presented over the period of 2009 to 2012. The data from the different summer periods were compared to regional monitoring data collected as part of other work (Berg 2006), and the patterns observed were similar and therefore comparing data between the summer periods seemed to be a reasonable approach.

The spatial distribution of the mean daily sediment-water interface temperatures measured at all regional study locations is shown in Figure 6.19. The temperatures ranged from 13.1°C (F13) to 20.5°C (F10). The temperature in Fishtrap Creek generally decreases in the downstream direction, with the highest temperatures recorded at sites F10 and F11 (Enns Brook) and the lowest temperature at F13 (Waetcher Creek tributary). Along the mainstem, the water temperatures decrease downstream towards F1 reflecting both the influence of cooler water from Waetcher Creek and an increased contribution of cooler groundwater (mean annual temperature 10.9±0.4°C at ABB01) to baseflow through this section of stream. The temperature at B1 is comparable to moderate values in Fishtrap Creek, similar to F3.

Figure 6.19. Mean daily sediment-water interface temperatures recorded August 20, 2009 at the regional sites. Shown also are the locations of two observation wells, ABB01 and FT5-25 (in pink). Values are reported in Appendix A.

Manual stream discharge measurements were made at seven regional locations (Figure 6.20), and are reported as the mean of repeated measurements (refer to Chapter 2 for detailed methodology). The discharge in Fishtrap Creek generally increases with distance downstream, consistent with the findings of Berg and Allen.
(2007). The lowest discharge was along the length of Waetcher Creek (F6, F7, and F13). The largest stream volumes were measured in the lower reaches of Fishtrap (F1 and F2), indicating increased groundwater contribution supporting the flow volumes. The upper reach sites (F3 and F11) had low to moderate discharge volumes, similar to B1.

Figure 6.20. Manual stream discharge values measured during the 2009 summer period. Values are reported in Appendix A.

Total alkalinity (as HCO$_3^-$) was measured by titration from single grab samples from four stream locations and observation well FT5-25 (7.6 m deep) adjacent to F2 in 2010 (Figure 6.21). Groundwater typically has a higher total alkalinity than surface water given that HCO$_3^-$ is produced in the soil zone and through dissolution of various minerals. The alkalinity from the groundwater well, however, was 34 mg/L, and measured values in 2011 from observation well FT1-24 adjacent to F1 (38 and 41 mg/L) were consistent. Not only is the alkalinity low for groundwater, but it is also lower than the stream values, which ranged from 84 to 96 mg/L in Fishtrap Creek and 70 mg/L at B1. The highest alkalinity values were at F2 and F3, while the alkalinity at F1 was lower.

The potential decrease in alkalinity downstream from F2 (as suggested by the low value at F1) may be related to the declining influence of Waetcher Creek. Alkalinity was not measured in Waetcher Creek in this study, but it was 102 mg/L in August 2005 (Berg and Allen 2007). These authors noted that Waetcher Creek had higher alkalinity than Fishtrap Creek and suggested that Waetcher Creek inflow influenced the alkalinity values for some distance downstream of the confluence. Also consistent with this earlier work, the alkalinity at B1 was the lowest measured in this study. Given the influence of
Waetcher Creek alkalinity on Fishtrap Creek alkalinity, and the generally low groundwater alkalinity, this parameter is not considered particularly valuable for detecting groundwater contributions to baseflow at this particular study location.

![Map of the study area with Alkalinity values](image)

**Figure 6.21.** Alkalinity values for water samples collected in summer 2010 from regional sites and groundwater observation wells (FT1-24 and FT5-25 shown in pink but using the same legend scale. Groundwater was collected at a depth of approximately 7 m below ground elevation. Values are reported in Appendix A.

Electrical conductivity (EC) was measured at seven regional sites on October 11, 2012, at the end of the summer period (Figure 6.22). EC was recorded as single spot measurements at each location. EC can be used as a tracer for baseflow contribution to streams as the EC of groundwater is often higher than surface water (Cox et al. 2007; Vogt et al. 2010). The EC at the regional sites were highest at Enns Brook (F11). These high values may be a result of beaver ponding, and the presence of a small wetland in the upper reaches, both of which create sources of stagnant turbid water that could undergo concentration of dissolved solids through evaporation, thus leading to a higher EC. The EC values along Fishtrap Creek were variable, with moderate values through the central portion of the watershed, and higher values at F1. Given that the sediment-water interface temperature and discharge indicate increased groundwater contributions in this section of stream near F1, and that mean daily groundwater EC measured in the FT1-24 well was 468 μS/cm, the increase in EC at F1 is likely due to groundwater influx, but this parameter is not a strong indicator in this study. The lowest observed value (191 μS/cm) was recorded at B1.
pH values were also as single spot measurements recorded at the same seven regional locations in 2012 as the EC (Figure 6.23). The changes in pH values were subtle, and generally followed the same pattern as the EC values. pH in the groundwater well FT1-24 was monitored hourly and recorded intermittently between July 2008 and July 2011 (refer to Chapter 2 for details), and the mean daily pH was 6.3, which is lower than the stream pH values. Overall, the pH values are difficult to interpret in this study area given the complexity of inflows from various creeks and the similarity of the groundwater pH to that of the streams.
Figure 6.23. pH recorded at regional sites at the end of the summer period (October 11, 2012) and the mean daily pH recorded at observation well FT1-24 (pink). Values are reported in Appendix A.

Many of the differences individual spot measurements at the regional scale are subtle and may not provide much information in isolation; however, when considered together, the measurements provide indications of potential aquifer-stream connectivity. The regional differences observed in this study are also consistent with regional measurements collected in 2005 (Berg 2006) which supports the interpretations. Of the regional measurements, the stream discharge and sediment-water interface temperature measurements provide clearer indication of aquifer-stream connectivity relative to the other parameters measured. Overall, the combined results of the regional field measurements suggest:

- tributary inflow from Waetcher Creek appears to significantly moderate water temperature (discussed in Chapter 3) in the mainstem channel despite the contribution of a small proportion to the overall discharge of the mainstem flow in Fishtrap Creek;

- a greater groundwater contribution to baseflow along Waetcher Creek are suggested by the patterns of stream discharge and sediment-water interface temperatures. Inflows from this tributary to the main channel appear to influence the water chemistry downstream of the confluence;

- a greater groundwater contribution to baseflow through the lower reaches of Fishtrap, downstream of F5, as indicated by moderated sediment-water interface temperatures;

- likely a greater groundwater contribution in Fishtrap relative to Bertrand.

- similarities in the regional measurements between sites B1 and F3, which have similar surficial geology, and potentially similar aquifer-stream connectivity.
• influences (higher temperature and EC) from stagnant water from beaver activity and the headwater wetland at site F11 suggesting that these results should be viewed with caution.

The spot measurements of the regional field parameters are highly variable, both spatially and temporally, and the regional measurements reported here do not consider uncertainty. The representativeness of these data is uncertain and the interpretation is limited by the small sample sizes. This section provided a descriptive approach to summarizing potential patterns in the regional measurements in an effort to demonstrate how field indicators could be applied to aquifer-stream connectivity studies, and for comparison with Vulnerability Assessment results. To add value to monitoring of groundwater dependent streams and Vulnerability Assessments, use of field indicators would require rigorous field campaigns. In this study, detailed field methods were limited to sediment-water interface temperature monitoring, however similar programs that incorporated other parameters could also be useful in these studies.

Comparing Regional Field Measurements to Vulnerability Assessment Results

The Level I and II Assessments for Bertrand and Fishtrap Creeks took a generalized watershed approach for evaluating the vulnerability of the streams. For the Level III Assessment, the Zone Budget results from the numerical model refined the results by considering zones categorized into main channel and ephemeral sections. For example, in Fishtrap watershed, the ephemeral stream zones delineated in Zone Budget were primarily in the upper reaches of the watershed, and the tributaries: Waetcher Creek and the west segments of Enns Brook (Figure 6.13), corresponding to field locations F8, F9, F12 and F13. The regional field data was included to support the Zone Budget results by identifying stream sections with potentially greater groundwater contributions, both along the sections corresponding to defined zones in Zone Budget, and also within the zone segments. The regional temperature and chemistry data similarly suggest greater groundwater contribution to baseflow along Waetcher Creek and the upper reaches in the west portion of the watershed.

At the watershed scale, both the Level II and Level III Assessments estimated a larger volume of baseflow in Fishtrap Creek relative to Bertrand Creek. The regional field data, primarily the rigorous streambed-water interface temperature monitoring, also
show a likely greater groundwater contribution in Fishtrap relative to Bertrand. However, the regional field results suggest greater groundwater contributions to baseflow through the lower reaches of Fishtrap, downstream of F5, and similarities between the temperature and water chemistry between B1 and F3. These areas and patterns of groundwater contribution were not detected in the Zone Budget, and provide an example of how field indicators could be used to refine Vulnerability Assessment for site specific requirements and identify stream segments with higher potential vulnerability.

**Other Considerations**

Regardless of the approach used in a Level III Assessment, there will be uncertainty in the predicted impact. These uncertainties can arise due to a poor conceptual understanding of how the system functions, a lack of information on the properties of the system (e.g. the hydraulic properties), and/or a poor understanding of the stressors themselves (e.g., changes in recharge under future climate change). For these reasons, it is important to recognize sources of uncertainty in a Level III Assessment and convey the implications of uncertainties in a transparent manner.

Changes to recharge can result from changes in land use/land cover, climate variability and climate change. Several studies have been conducted in BC concerning the potential responses of aquifer systems to changes in recharge (e.g., Scibek and Allen, 2006a; Scibek and Allen 2006b; Scibek et al. 2007; Toews and Allen 2009; Allen et al. 2011). Uncertainties in future climate projections, downscaling methods, current recharge estimates and aquifer dynamics complicate the ability to predict how climate change, or indeed climate variability, might influence recharge.

Current best practices for estimating aquifer responses to climate change recommend numerical hydrogeological modeling (Holman et al. 2012). For similar reasons, changes in recharge due to land use/land cover change also would require numerical hydrogeological modeling.

### 6.6. Risk Assessment / Risk Management Framework

The results of Level II and Level III Assessments can be integrated into a risk assessment / risk management framework (Figure 6.24). In addition to assessing stream
vulnerability, the consequence of the impacts to the stream must be assessed so as to assess the risk to the stream as summarized below. A full risk assessment in combination with monitoring (using specific indicators for the stream) could be utilized to develop integrated management plans for water quantity planning in aquifer-stream systems. These plans could include measures such as well set-back distances or buffer zones along sensitive streams, and integrated planning for abstraction volumes.

Figure 6.24. Flow chart for stream vulnerability within a risk assessment framework.

6.6.1. Risk to a Stream

The risk to a stream resulting from a specific hazard is defined in Equation 6.8, and the components of the risk framework are shown in Figure 6.25.

\[
Risk (R) = Stream \text{ Vulnerability} (SV) * Loss (L) \tag{6.8}
\]

The Level II Assessment results in a rating for Stream Vulnerability. The second component of the risk equation is the Loss (L), which is the consequence of a change to stream flow quantity resulting from stressors in the aquifer system (such as pumping). This loss component can represent a range of impacts, including economic losses from
changes in flow conditions, or restoration costs, human health impacts from changes to drinking water sources, and ecological losses. The ecological loss component may arise from loss of ecosystem health, or loss of the valuation assigned to the ecosystem service. For example, a loss of ecosystem health could relate to insufficient flow to sustain fish habitat, as is currently defined for Coho salmon (*Oncorhynchus kisutch*) under the Sensitive Stream Designation in the *Fish Protection Act*. The value assigned to this loss component could be based on presence or absence of this loss arising from loss of stream flow. The loss may be related to costs in the mitigation strategies, such as relocation of migrating salmon, loss of water use due to surface water license restrictions, or changes in groundwater use such as relocation of pumping wells outside the boundary of the active zone of the stream.

6.6.2. Risk Management Example

As an example of risk management, the Zone Budget results for the Fishtrap and Bertrand Watersheds indicated that stream discharge would decrease in both watersheds as result of groundwater pumping, with likely greater decreases in the ephemeral stream sections. These results suggest that the ephemeral stream segments receive greater groundwater contributions on an annual basis, and likely in the summer periods. Figure 6.25 shows the measured discharge in summer 2009 at various regional sites along with the estimated percent decrease in discharge due to pumping based on the Zone Budget results. The stream discharge was measured while an unknown number of wells were pumped in the watershed. Therefore pumping impacts to the streamflow are likely incorporated in the measurements; however, these discharge values were used in the absence of available non-pumping discharge measurements in order to provide an example how field indicators can be combined with the Level III Assessment for risk management. The ephemeral streams have much lower discharge than the main stem of Fishtrap Creek. The Zone Budget results estimated decreases in overall discharge in the streams as a result of pumping (Figure 6.25), with the greater impact in the groundwater-dependent ephemeral sections. Given that the regional field data show that the stream temperatures and discharge from the ephemeral reaches influence the downstream stream sections (Figures 6.19 and 6.20), it is likely that decreases in baseflow due to pumping will not only reduce the stream flow, but will also result in reduced buffering of high summer stream temperatures. Decreases in stream...
flow and increases in water temperature can be associated with decreased aquatic habitat suitability, and thus could be quantified as loss of ecosystem health. As shown in Figure 6.24, the field indicators (here discharge) can be used to monitor the status of the stream and groundwater contributions for management of the risk.

Figure 6.25. Stream discharge measured regionally in summer 2009 (light blue) with the percent decrease in summer baseflow estimated in the Zone Budget results (black).
Chapter 7.

Conclusions

In this study, field and statistical methods were used to characterize aquifer-stream connectivity primarily during the summer low flow period, utilizing two streams in the Lower Fraser Valley of southwest British Columbia (BC) as case studies. The research aimed to demonstrate that the connectivity between groundwater and streams, and the factors that influence that connectivity, can be explained at different scales using sediment-water interface temperature in combination with various field measurements. The research focused on the summer low flow period because this is a period of particular ecological importance as both decreased flows and elevated stream temperatures during summer low flows have the potential to negatively impact aquatic health and thereby fish ecology.

The two watersheds were selected for comparison because they have many similarities, including similar watershed size, climate and topography, and they are within the same aquifer setting. As such, they are ideally suited for comparing and contrasting the various factors that might influence aquifer – stream connectivity in space and time. Both streams have been identified in previous work as being groundwater-fed streams (Johanson 1988, Pearson 2004, Berg and Allen 2007, Starzyk 2012). While the previous studies indicated differences in the magnitudes of the groundwater contributions to each stream, no detailed studies had been completed to quantitatively compare the groundwater inputs. The various methods employed in this study were applied to evaluate the connectivity at different spatial and temporal scales, and to identify the factors influencing groundwater influx.

Several field methods, in combination with a simplified heat budget, were used to estimate the relative contribution of groundwater to Fishtrap and Bertrand Creeks at respective local sites, as well as more regionally in Fishtrap Creek Watershed. The results were combined to develop empirical and conceptual models of the study sites,
which in turn aided in understanding the factors influencing the aquifer – stream connectivity in the two watersheds. Using the conceptual models, independent component analysis (ICA) was used in combination cross-correlation to evaluate factors influencing groundwater flux to the stream at the local scale using sediment-water interface temperatures.

The understanding of the connectivity between groundwater and streams at the different scales was used to develop an assessment framework for stream vulnerability that can be applied to groundwater-dependent streams provincially to classify steams that are groundwater sensitive. Classification of streams that are likely to have changes in quantity resulting from groundwater stressors is important for management of potential impacts from climate variability, and changes in climate, land use / land cover, and groundwater use.

7.1. Aquifer - Stream Connectivity

The field sites lacked detailed energy flux data; however, in order to interpret the sediment-water interface temperatures, it was necessary to develop a conceptual model of the energy flux and estimate the energy exchanges. A simplified heat budget for both sites was developed using basic principles of energy exchanges, latitude and elevation, and climate data from the nearest climate station. The heat budget approach, applied with a regression analysis of one-day lagged air temperature and sediment-water temperature was successfully applied to understand heat exchanges within the streams and identify differences between the sites. The results showed that incoming solar radiation inputs dominate the heat exchanges in both streams, but indicated there are differences between the two streams both annually and during the summer periods despite their similar settings. The results also indicated that heat exchanges were occurring at each site that could not be accounted for at the water-air interface, and therefore were occurring across the water-streambed interface.

Field measurements taken during the summer low flow periods were compared with the heat budget results for the local field sites. The field measurements were a combination of sediment-water interface temperatures and flux measurements comprising manual stream discharge measurements, seepage meter measurements,
and in-stream piezometery. The data collected using this spectrum of methods were used in combination with groundwater level and temperature data from nearby groundwater monitoring wells to evaluate connectivity. The sediment-water interface temperatures and stream discharge measurements from gauging stations were applied to directly compare the relative groundwater contributions between the sites. The cooler summer interface temperatures and higher stream discharge in Fishtrap Creek, relative to Bertrand Creek, suggested a greater contribution of groundwater to the summer stream flow in Fishtrap Creek, and this supported the heat budget results which identified a heat sink at the streambed interface in Fishtrap Creek.

As part of the field component of this research, the various approaches for measuring the groundwater flux were compared to the results of the heat balance analysis. In part, the field measurements of flux supported the heat budget results; however, due to high uncertainty in some of the field methods, there were varied levels of success with the field methods when applied during the summer low flow periods. The in-stream piezometers and seepage meter measurements were more consistent in Fishtrap Creek than in Bertrand Creek. In Fishtrap Creek, the various methods indicated an influx of groundwater, within the same order of magnitude. In Bertrand Creek, the hydraulic gradient was low, and the field conditions (lower vertical hydraulic conductivity, shallow flow depths, and coarse bed material) were less favorable for the use of the seepage meter and piezometers. In Bertrand Creek, the direct flux measurements by these methods differed on average by an order of magnitude, and the direction of flux was inconsistent. The degree of uncertainty in the manual stream discharge measurements at both sites rendered the results of this method inconclusive for determining changes in flow volume from groundwater flux. Therefore, it was found that the seepage meter results were more successful in the finer grained bed substrate and deeper stream flows at Fishtrap Creek. The coarse grained bed material and shallow flow in Bertrand Creek was a difficult environment for installing the seepage meter, and flux into the meter was interpreted as being dominated by hyporheic flows.

To further explore aquifer - stream connectivity at a regional scale, the sediment-water interface temperatures were measured at various sites in the Fishtrap Creek watershed. At the regional scale, sediment-water temperature measurements were an effective method to trace relative groundwater contributions to the stream. Moderated
temperatures in the lower reaches of the watershed showed diffuse groundwater influx to different sections of the stream and this indicated increased groundwater contributions through the lower reaches of the stream where the stream flowed through more permeable substrate.

At the local scale, a network of temperature dataloggers recorded sediment-water interface temperatures over a short distance of the stream bed. The temperatures recorded at each location are mixed signals representing inputs from different components of the heat budget at the site. ICA was used to separate the temperature signals into individual components and evaluate how the components varied between summer periods. Independent signals were extracted from the recorded mixed signal time series. Using the conceptual model developed from the heat budget and the field measurements, the signals extracted using ICA were related to the factors contributing to the heat budget across the stream reach with cross-correlation. The correlation with groundwater levels indicated some heat exchanges were associated with groundwater inflow. The ICA signal separation technique provided evidence that the influence of groundwater influx on the interface temperature was temporally variable as the groundwater levels, and subsequently the relative contribution of flux to the stream, varied between the summer periods. This demonstrates the stream flow and sediment-water interface temperature can be sensitive to changes in groundwater conditions with time.

A general summary of the two streams would suggest that overall, they are very comparable, and would be expected to have a similar connectivity with groundwater. The steams are located within the bounds of the same aquifer, have watershed sizes of same order of magnitude, have similar stream lengths, and originate at similar elevations. Due to their close proximity, the streams also share similar climate and topography. However, the aquifer - stream connectivity component of this research showed there are significant differences in the sediment-water interface temperatures between the streams at a local scale, and also within the Fishtrap Creek watershed. Previous studies indicated differing interactions with groundwater between the streams, and sediment-water interface temperatures, and measured field parameters confirmed greater relative contribution of groundwater to Fishtrap Creek during the summer period.
The conceptual model of the sites identified differences in the geological substrate in the vicinity of either stream, as well as differences in the heat budgets. The field methods confirmed variations in the groundwater flux to the streams, and related the sediment-water interface temperatures to processes influencing the stream temperature through inputs to the heat budget. Therefore, it was shown that the more permeable substrate surrounding Fishtrap Creek, in combination with steeper groundwater gradient toward the stream, result in a greater groundwater influx.

To evaluate aquifer-stream connectivity, complex datasets were not required but basic information and principles of heat and flow exchanges were sufficient when guided by the use of a conceptual model of the system. This finding is important because in many instances, data applicable to aquifer-stream connectivity are not directly available; however, data collected for other studies (i.e. fish habitat metrics such as water temperature) can potentially be repurposed for evaluating aquifer-stream systems and groundwater resources. The use of sediment-water interface temperature measurements and signal separation using ICA was particularly valuable for developing a better understanding of heat transfer across the sediment-water interface. The application of ICA to sediment-water temperature differs from more commonly applied analytical approaches, which consider the source signals, and the noise and bias components. This research did not directly consider those components of the signal, rather it demonstrated the use of ICA for blind signal separation of mixed signals, and how those components can be linked to important heat transfer processes including groundwater influx.

### 7.2. Stream Vulnerability

The understanding of aquifer-stream connectivity at different scales was applied in the development of a vulnerability framework for the assessment of groundwater-dependent streams. The framework combines groundwater-related stressors with the stream susceptibility to determine stream vulnerability to changes in groundwater conditions, using a multi-step risk-based approach.

The vulnerability framework was developed with the intent to contribute to the existing “sensitive stream” designation defined within *Fish Protection Act*, Section 6(2),...
by expanding the definition of the sensitive stream to include groundwater-dependent streams. As part of the vulnerability assessment, a groundwater-dependent stream is defined as a stream whose flow sustained by groundwater inputs (baseflow) during the annual summer low flow period, and as such is vulnerable to changes in the aquifer system. As a result, these streams are likely to have measurable impacts in water quantity resulting from potential groundwater stressors.

The vulnerability assessment involves a three-level procedure that proceeds step-wise, incorporating increasing levels of complexity, and can be completed using publicly available data in British Columbia. The Level I Assessment is a screening tool that combines the hydrologic setting and the level of development through a desktop survey of the mapped aquifer and intersecting stream(s). The Level II Assessment incorporates the aquifer-stream connectivity concept, and requires an understanding of the aquifer-stream system as well as the stressors acting on the system. The outcome of the Level II Assessment is stream vulnerability which is a product of the stream susceptibility and hazard (stressor). Stream susceptibility is assessed based on recharge to the aquifer and the aquifer characteristics, which are metrics used to evaluate the ability of the aquifer to sustain streamflow during baseflow conditions. The vulnerability ranking is an evaluation of the potential for the stream flow to be influenced by current stressors in the aquifer, which are defined as the hazards. If a stream is found to be vulnerable, a Level III Assessment is undertaken with the objective of quantifying the potential impact to the stream. The Level III Assessments are site specific and integrate information on the aquifer-stream system with the magnitude of the stressors acting on that system. To quantify the magnitude of the stressors, quantitative tools are required; the options range from simple analytical methods through to numerical modelling, and the selection of the tool will be dependent on the site specific stressors being addressed, as well as available data and technical resources (e.g. numerical modelling capabilities).

The Stream Vulnerability Assessment Levels I and II were tested on a selection of streams province-wide to demonstrate the application of the assessment method. Nine streams across BC were evaluated, representing different aquifer-stream settings, including a designated “sensitive stream” and Fishtrap and Bertrand Creeks studied in this research. The aquifer settings include bedrock aquifers and unconsolidated
The assessment method was applied using publicly available map and data resources, and performed well despite numerous assumptions and simplifications required as a result of data limitations or complexities in the aquifer-stream system. The greatest simplification in the method is in the estimation of recharge to the stream which is based on a water balance approach that provides a first-order approximation of recharge. Thus, indirectly, recharge to the stream is estimated as discharge from the aquifer system along the length of the stream segments. This method assumes that all recharge to the aquifer will discharge to the stream, and further requires assumptions on the appropriate bounds of the aquifer-stream system in the assessment. Also, the recharge considers only potential evapotranspiration in the assessment. Of the nine test cases, the Level II Assessment indicated six of the streams had a Stream Vulnerability ranking of high, and a Level III Assessment was recommended for those systems. The remaining three systems were ranked low vulnerability, and no further assessment would be required unless there is a change in the recharge to the system, or an increase in the pumped volume from the aquifer, at which point a Level II Re-assessment would be required.

A case study of a Level III Assessment was completed for Fishtrap and Bertrand Creeks, both of which were found to have a high Stream Vulnerability in the Level II Assessment. The Level III Assessment was completed using an existing transient groundwater flow model for the Abbotsford-Sumas aquifer (Scibek and Allen, 2005). Zone Budget in MODFLOW was used to quantify impacts of pumping on the streams for the Level III Assessment, as well as to validate the results of the recharge and baseflow estimations in the Level II Assessment. To evaluate the impact to the streams from pumping, the wells within each watershed were activated, and volume changes from the stream boundaries were compared to the non-pumping model results. When pumping was activated in the model in each watershed, the volume of water out of the mainstem stream segments decreased by approximately 25%, indicating that pumping was intercepting groundwater that had been discharging to the stream, as well as drawing water out of the stream segments. In both watersheds, the ephemeral stream segments had the greatest decline water volume (approximately 30%) when pumping was active, indicating that generally, ephemeral stream channels are most vulnerable to changes in groundwater conditions.
The numerical model used to complete the Level III Assessment for Fishtrap and Bertrand Creeks was also used to compare result of the recharge and baseflow with the Level II Assessment and to validate the assumption that aquifer-stream area can be defined by the aquifer area, or if the aquifer is large, the watershed area. For both case study streams, the aquifer area is larger than the watershed; therefore, following the methodology in the Level II Assessment, the watershed area was defined as the area of potential impact. To verify that this is a reasonable assumption, five pumping scenarios were completed using the model to evaluate impacts to the streams from cumulative effects of pumping over larger areas within the aquifer. The results of this analysis indicated that impacts to the streams occurred primarily when pumping occurred in within each individual watershed. Therefore, the area of interest can be defined as either by the watershed boundary or the aquifer intersecting the groundwater-dependent stream. The estimated annual recharge in both Assessment Levels compared well, which demonstrated that despite the broad simplifications in the Level II Assessment, the method provides a reasonable approach to annual recharge estimation. This is significant because often the data required for comprehensive recharge calculations are limited, requiring simplification of the process as defined in the Level II Assessment procedure. The baseflow estimate from the model was higher than the observed streamflow that was used in the Level II Assessment. The Level II baseflow estimates were derived from measured streamflow within each watershed, and incorporated impacts from unknown volumes of pumping within each watershed. The estimates in the model have greater uncertainty than observed values, because simplifications and assumptions are inherent in numerical modelling, and those translated into uncertainty in the outputs. Also, in this case study, the model used was a groundwater flow model, and therefore stream segments function to moderate groundwater levels, and not directly model surface water processes. However, the model provides a method to estimate baseflow under non-pumping conditions, which is a useful benchmark for risk management. The results of the regional field monitoring were used in this case study to provide an example of integrating indicators and a Level III Assessment into a risk management framework.
7.3. Recommendations and Future Work

The field component of this study demonstrated some of the challenges of evaluating aquifer-stream connectivity, namely uncertainty in measurements. The uncertainty in some of the flux measurements rendered their results inconclusive; however, collectively the methods provided useful information. It is recommended that use of the manual discharge measurements be collected over longer stream distances in order to detect larger flow volume changes with distance. Also, at the study sites, the use of the seepage meters was found to be limited to ideal site conditions during low flow periods (i.e. deeper water). In shallow water, or where channel morphology generates significant hyporheic circulation, seepage meters measurements should be supported by secondary flux measurements, such as in-stream piezometers. To improve the conceptual understanding of aquifer-stream systems, it is recommended that conceptual models include knowledge on regional groundwater flow as well as localized processes such as hyporheic flows.

For future studies, incorporation of multiple indicators beyond direct flux measurements is recommended to strengthen the interpretation of aquifer-stream connectivity. Lack of data is a limitation for many aquifer-stream connectivity studies. However, there are large datasets related to fish habitat assessments, pre-feasibility studies, and reclamation projects, etc. that may prove particularly useful for better understanding aquifer-stream connectivity. Many of these studies collect spot measurements or produce short term datasets (e.g. water temperature, climate, flow measurements) similar to the data used in this study. These datasets could be repurposed to fill data gaps or at least provide a starting point for studies on aquifer-stream systems, and provide direction for more focused and rigorous field programs. Therefore, a recommended next step is to evaluate the use of existing water quality data sets to compliment the range of methods used in this study.

The site-specific nature of the Level III Assessment will inherently have uncertainty in the predicted impact to the stream from changes in the groundwater, regardless of the method used. The sources of the uncertainty may stem from lack of available data or quality of the data that are available; poor conceptual understanding of the aquifer-stream connectivity or aquifer system; lack of understanding of existing or future stressors (e.g. pumping rates or changes in recharge under future climate...
change). Similar to the field measurements, it is important in the vulnerability assessment method to identify the sources, and magnitude if possible, of the uncertainty, and to convey the implications of these uncertainties through the assessment process. If detailed information is required to better estimate baseflow and model the aquifer – stream connectivity, a coupled groundwater-surface water model that is capable of incorporating bi-directional exchanges between the groundwater and stream is recommended.

This research presented a vulnerability assessment method that is relevant to a broad range of water resources management applications. It is envisioned that this method could be applied to address water quantity management objectives, and be incorporated into watershed risk management. A risk assessment would allow for the consequence of the impacts to the stream (from the Level III Assessment) to be assessed. This would require assessing loss, which is the consequence of a change in streamflow resulting from groundwater stressors. The definition of loss would be specific to the management objective. Loss can range from economic loss due to change in water availability, restoration costs for habitat loss, loss of ecological function, or changes to ecosystem health and services. Given the specific nature of loss assessment, this component was not included within the scope of this research.

To meet management objectives, the results of the risk assessment would be integrated with a monitoring protocol, using specific indicators, or combinations of indicators, for assessing the aquifer-stream connectivity and streamflow quantity. The indicators would be used to monitor the status of the system, and to guide the implementation management decisions as required to maintain an acceptable level of risk as part of a dynamic management strategy. The risk could be managed through integrated aquifer-stream management plans potentially including administration of water use licences, pumping management such as well set-back distances, buffer zones along sensitive streams, and integrated planning for abstraction volumes.

7.4. Summary

The methods developed in this research were applied to advance the understanding of aquifer - stream connectivity at different spatial and temporal scales,
particularly during summer low flows. The research showed that use of a combination of field methods and a heat budget analysis was critical for developing conceptual models the aquifer - stream connectivity in the two study watersheds. Within the context of understanding aquifer - stream connectivity, temporal and spatial variability in streamflow and sediment-water interface temperatures was attributed to variations in relative contributions of groundwater to streams. To understand this variability in sediment-water interface temperatures across a site, this research explored the use of ICA as technique for blind signal separation. The use of ICA, combined with cross-correlation of was a novel approach, which directly linked the extracted signals to factors in the heat budget that influence sediment-water interface temperatures. This research applied the multi-scale understanding of aquifer-stream connectivity to the development of a risk-based vulnerability assessment method aimed at evaluating vulnerability of a groundwater dependent stream to changes in the aquifer system.
References


Aquatic Informatics Inc. 2012. *Aquarius time series software (v. 3.0.75.1)*. Vancouver, Canada: Aquatic Informatics Inc.


BC Ministry of Agriculture and Lands. 2006. GIS land use data for the Abbotsford aquifer.


Appendix A.

Field Data Summary

Streambed Temperature Depth Profile

The temperature depth profile at site F3 (location shown on Figure 2.7) recorded temperature within the streambed at an upstream and a downstream location, separated by distance of 6.25 m. The temperatures were recorded hourly with TidbiT® v2 Temp loggers (UTBI-001) between August 21, 2010 and Sept 9, 2011, and are summarized in Table A1 along with a description of the general material type at the depth of each installation. The temperature profiles at the upstream and downstream locations are plotted in Figures A1 and A2.

Table A1. Summary of the temperature profiles within the sediment at F3.

<table>
<thead>
<tr>
<th></th>
<th>Upstream Profile</th>
<th>Downstream Profile</th>
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</thead>
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<tr>
<td></td>
<td>Depth 0.1 m</td>
<td>Depth 0.19 m</td>
</tr>
<tr>
<td>Mean (°C)</td>
<td>10.58</td>
<td>10.59</td>
</tr>
<tr>
<td>Min (°C)</td>
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<td>1.78</td>
</tr>
<tr>
<td>Max (°C)</td>
<td>19.08</td>
<td>17.94</td>
</tr>
<tr>
<td>Sediment Description</td>
<td>Med-coarse sand, some med gravel</td>
<td>Med-coarse sand, some med gravel</td>
</tr>
</tbody>
</table>
Figure A1. The temperature profile at the upstream (US) location at site F3, with temperature loggers installed at depths of 0.1 m and 0.19 m.

Figure A2. The temperature profile at the downstream (DS) location at F3, with temperature loggers installed at depths of 0.1 m and 0.2 m.
Flux Measurements

Field data for flux measurements from the seepage, piezometer and manual discharge measurements are summarized in Table A2.

Table A2. Summary of the flux measurements from the seepage meters, the nested in-stream piezometers, and the manual discharge measurements. The standard deviation (SD) is given for the seepage measurements and \( n \) is the number of measurements recorded at approximately 30 minute intervals. For the piezometers, negative values indicate an upward flux into the channel.

<table>
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<tr>
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<th>Piezometer (dh/dl)</th>
<th>Manual Discharge (m³/s)</th>
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Regional Measurements

Water chemistry samples were collected with physiochemical and stream discharge measurements at the regional scale sites in the Fishtrap Creek watershed and at site B1. These data were collected on separate occasions in 2009, 2010, and 2012, and tabulated below (Table A3) with the UTM coordinates of each regional site and two groundwater observation wells (FT1-24 and FT5-25) from which alkalinity samples were obtained.
<table>
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<th>Site ID</th>
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<th>UTM Zone 10 Easting</th>
<th>Date</th>
<th>Manual Discharge (m³/s)</th>
<th>EC (μS/cm)</th>
<th>pH</th>
<th>Spot Temp (°C)</th>
<th>DO (mg/L)</th>
<th>Alkalinity (mg/L HCO₃⁻)</th>
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<tr>
<td>F1 1</td>
<td>Right bank: Reed grass. Left bank: Reed grass and blackberries. No in-stream vegetation.</td>
<td>Gravel and sand, medium-coarse gravel, fine-coarse sand, some fines, trace cobbles, some organics.</td>
<td>Downstream end of pool in a series of pools.</td>
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</tr>
<tr>
<td>2</td>
<td>Left Bank: 70% grass cover.</td>
<td>Gravel and sand, medium-coarse gravel, fine-coarse sand, some fines, trace cobbles, some organics.</td>
<td>Side of pool.</td>
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<tr>
<td>3</td>
<td>100% reed grass ~0.7 m into channel.</td>
<td>2-3 cm thick layer of organics and fines. Below organics, sand and gravel. Fine-coarse sand, fine-medium gravel, trace to some coarse gravel.</td>
<td>Side of upstream end of run.</td>
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<tr>
<td>4</td>
<td>Left Bank: 100% reed grass ~0.8 m into channel.</td>
<td>1-2 cm organics. Sand, fine-coarse, some fine-medium gravel.</td>
<td>Upstream side of pool.</td>
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</tr>
<tr>
<td>5</td>
<td>Right bank: 100% reeds ~0.8 m into channel.</td>
<td>Fines and organics, some small woody debris.</td>
<td>Center upstream end of a run.</td>
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<tr>
<td>6</td>
<td>Originally ~100% reed grass in the channel, but cleared sections so was ~60% covered after clearing.</td>
<td>Gravel and sand, fine-coarse gravel, fine-coarse sand, trace cobbles, trace-some organics and fines.</td>
<td>Run.</td>
<td></td>
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</tr>
<tr>
<td>7</td>
<td>Originally ~100% reed grass in the channel, but cleared sections so was ~60% covered after clearing</td>
<td>Sand, gravelly. Fine-coarse sand, fine-medium gravel, trace coarse gravel, trace organics.</td>
<td>Run.</td>
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<tr>
<td>8</td>
<td>100% reed grass ~0.3 m into the channel.</td>
<td>Sand, gravelly. Fine-coarse sand, fine-medium gravel, trace coarse gravel, trace organics.</td>
<td>Run.</td>
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<tr>
<td>9</td>
<td>Channel ~80% reed grass. Datalogger in open section of channel, with ~10% cover.</td>
<td>Sand and gravel. Fine-coarse sand, fine-medium gravel, trace-some coarse gravel, trace fines.</td>
<td>Run.</td>
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<td>Bed Sediments</td>
<td>Morphology</td>
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<tr>
<td>11</td>
<td>Channel ~70% reed grass. Datalogger in open section of channel, with ~20-30% cover.</td>
<td>Gravel, sandy. Fine-coarse gravel, fine-coarse sand, some organics, trace cobble.</td>
<td>Run.</td>
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<td>12</td>
<td>Channel ~80% reed grass. Datalogger in open section of channel, with ~40% cover.</td>
<td>Gravel, sandy. Fine-coarse gravel, fine-coarse sand, some organics, trace cobble.</td>
<td>Outer bend of run, at the downstream end of possible bar, island, or large mass of reed grass.</td>
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</tr>
<tr>
<td>13</td>
<td>100% reed cover, channel is overall ~90% reeds.</td>
<td>Gravel and sand, medium-coarse gravel, some fine-coarse sand, some organics</td>
<td>Top of run.</td>
<td></td>
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<td></td>
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</tr>
<tr>
<td>14</td>
<td>100% reed cover.</td>
<td>Gravel and sand, medium-coarse gravel, some fine-coarse sand, some organics</td>
<td>Right bank of pool.</td>
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<td></td>
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</tr>
<tr>
<td>15</td>
<td>100% reed cover. Reeds ~0.6 m into channel.</td>
<td>At datalogger location, silt and clay, organic root masses. Channel center: sand, silt and fine-medium gravel. Very soft.</td>
<td>Deep pool.</td>
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### F2

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<th>Riparian Vegetation</th>
<th>Bed Sediments</th>
<th>Morphology</th>
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</thead>
<tbody>
<tr>
<td>1</td>
<td>Right bank: 1.5-2 m reed grass, bounded by shrubs to a height of 3 m. Left bank: 3.5 m reed grass, bounded by shrubs, red osier dogwood, alder, birch, blackberry. ~75-85% reeds in the channel.</td>
<td>Sand, some fine-medium gravel, some silt, some fines and organics. Very soft sediments.</td>
<td>Upstream end of pool.</td>
</tr>
<tr>
<td>2</td>
<td>Right bank: 1.5-2 m reed grass, bounded by shrubs to a height of 3 m. Left bank: 3.5 m reed grass, bounded by shrubs, red osier dogwood, alder, birch, blackberry. ~15% reed cover.</td>
<td>Gravel and sand, medium-coarse gravel, with some coarse. Fine-coarse sand, some cobbles, some small woody debris/organics. Trace fines. Soft sediments.</td>
<td>Run.</td>
</tr>
<tr>
<td>Location</td>
<td>Riparian Vegetation</td>
<td>Bed Sediments</td>
<td>Morphology</td>
</tr>
<tr>
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</tr>
<tr>
<td>3</td>
<td>Right bank: 1.5-2 m reed grass, bounded by shrubs to a height of 3 m. Left bank: 3.5 m reed grass, bounded by shrubs, red osier dogwood, alder, birch, blackberry. In-stream vegetation ~75% reed grass.</td>
<td>Sand and gravel. Medium-coarse sand, some fine sand. Fine-coarse gravel, trace to some cobbles.</td>
<td>Run.</td>
</tr>
<tr>
<td>4</td>
<td>Right bank: 1.5-2 m reed grass, bounded by shrubs to a height of 3 m. Left bank: 3.5 m reed grass, bounded by shrubs, red osier dogwood, alder, birch, blackberry. In-stream vegetation 80% reed grass. Datalogger ~15% covered.</td>
<td>Sand, gravelly. Fine-coarse sand, fine gravel, some medium gravel, trace silt.</td>
<td>Run.</td>
</tr>
<tr>
<td>5</td>
<td>Right bank: 1.5-2 m reed grass, bounded by shrubs to a height of 3 m. Left bank: 3.5 m reed grass, bounded by shrubs, red osier dogwood, alder, birch, blackberry. 100% reed cover</td>
<td>Sand, gravelly. Fine-coarse sand, fine gravel, some medium gravel, trace silt.</td>
<td>Run.</td>
</tr>
<tr>
<td>6</td>
<td>Right bank: 1.5-2 m reed grass, bounded by shrubs to a height of 3 m. Left bank: 3.5 m reed grass, bounded by shrubs, red osier dogwood, alder, birch, blackberry. In-stream no cover in center at datalogger location.</td>
<td>Sand, gravelly. Fine-coarse sand, fine gravel, some medium gravel, trace silt.</td>
<td>Run.</td>
</tr>
<tr>
<td>B1</td>
<td>10% canopy cover, willow, some high maple. Mostly uncovered on upstream side.</td>
<td>Sand, gravelly. Medium-coarse sand, some fine. Medium-coarse gravel, some cobble, trace boulders (rip-rap), trace fines.</td>
<td>Riffle, side.</td>
</tr>
<tr>
<td>2</td>
<td>~15% canopy cover, willow.</td>
<td>Sand, fine-medium, some coarse. Some fine-medium gravel, trace-some fines, some small woody debris.</td>
<td>Riffle, side.</td>
</tr>
<tr>
<td>Location</td>
<td>Riparian Vegetation</td>
<td>Bed Sediments</td>
<td>Morphology</td>
</tr>
<tr>
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</tr>
<tr>
<td>5</td>
<td>Shrubs, 80-85% canopy cover.</td>
<td>Gravel and sand. Fine-medium gravel, fine-coarse sand, abundant small woody debris, some fines and organics.</td>
<td>Run/pool, off-center.</td>
</tr>
<tr>
<td>7</td>
<td>Deciduous vegetation, broad leaf maple. 75% canopy cover from high canopy.</td>
<td>Sand. Fine-coarse sand, trace to some fine-medium gravel, some cobbles (rip-rap), some fines and organics.</td>
<td>Run/pool, center.</td>
</tr>
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<td>9</td>
<td>Deciduous riparian vegetation: willow. 75% canopy cover.</td>
<td>Sand. Fine-medium sand, organics. ~2 cm thick organics and small woody debris.</td>
<td>Edge of pool.</td>
</tr>
<tr>
<td>Location</td>
<td>Riparian Vegetation</td>
<td>Bed Sediments</td>
<td>Morphology</td>
</tr>
<tr>
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<tr>
<td>12</td>
<td>Left bank, rip-rap slope to road cut. Right bank deciduous riparian vegetation: willow and alder. 5% canopy cover.</td>
<td>Gravel and sand. Medium-coarse gravel, fine-coarse sand.</td>
<td>Run.</td>
</tr>
<tr>
<td>14</td>
<td>Shrubs: blackberries. 20-30% canopy cover</td>
<td>Sand. Fine-coarse sand, trace to some fine-medium gravel, trace fines.</td>
<td>Downstream end of riffle.</td>
</tr>
<tr>
<td>16</td>
<td>Right bank: Deciduous, cottonwood, willow, red osier, policemens helmet. Left bank: reed grass, blackberry, 1.5m to alder. ~20% canopy cover</td>
<td>Gravel, sandy. Med-coarse gravel, some fine gravel. Medium-coarse sand, some fine, trace organics.</td>
<td>Pool center.</td>
</tr>
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Appendix B.

Tidbit Temperature Logger Calibration Verification

The streambed interface and profile temperatures were recorded using Tidbit® v2 Temp loggers (UTBI-001). The specifications of the temperature sensors are provided in Table B1. Tidbit loggers are factory calibrated to ±0.2°C. To verify the accuracy of the factory calibration and provide a baseline for drift corrections, the water temperature measurements from the Tidbit dataloggers was compared in to a YSI datalogger (Model 6920) in 2008, and in 2012 values were compared against HOBO® U20 Water Level Data Logger (Model U20-001-02). The specifications of all the data loggers are presented in Table B1. The goal of the verification was to test the accuracy of the Tidbit loggers over their full recommended range of operation (0°C to 50°C), and to test for drift over the period of deployment between 2008 and 2012.

Table B1. Equipment specifications for the YSI and Tidbit data loggers.

<table>
<thead>
<tr>
<th></th>
<th>YSI 6920 SONDE</th>
<th>Tidbit v2 Temp (UTBI-001)</th>
<th>HOBO U20 Water Level Data Logger (U20-001-02)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Operation Range (°C)</td>
<td>- 5° to + 45°</td>
<td>- 20 to + 70°</td>
<td>- 20° to + 50°</td>
</tr>
<tr>
<td>Accuracy (°C)</td>
<td>± 0.15°</td>
<td>± 0.2° from 0° to 50 °C</td>
<td>± 0.37° at 2°</td>
</tr>
<tr>
<td>Resolution (°C)</td>
<td>0.01°</td>
<td>0.02° at 25°</td>
<td>0.1 at 20°</td>
</tr>
<tr>
<td>Response Time</td>
<td>5 minutes in water</td>
<td>3.5 minutes in water</td>
<td></td>
</tr>
<tr>
<td>Drift</td>
<td>0.1° per year</td>
<td>0.1° per year</td>
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The Tidbit loggers used in the field program for this research are listed in Table B2, which shows the location at which they were deployed for the streambed interface temperature monitoring (Refer to Chapter 2, Figure 2.6 for locations). Table B3 shows the locations of Tidbit loggers that were deployed for temperature profiling within the streambed sediments. Tables B2 and B3 report the deployment and retrieval dates for the data, the final status of the logger, and replacement Tidbits where applicable.

Table B2. Serial numbers, locations, data periods, and status of Tidbit temperature loggers installed for streambed interface temperature monitoring.

<table>
<thead>
<tr>
<th>Tidbit Serial Number</th>
<th>Location</th>
<th>Deployment Date</th>
<th>End of data</th>
<th>Status</th>
<th>Replace Serial Number</th>
<th>Deployment Date</th>
<th>End of data</th>
<th>Status</th>
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Table B3. Serial numbers, locations, data periods, and status of Tidbit temperature loggers installed for profiling streambed temperature within the streambed sediments. The depth of installation (cm) is indicated in the location (e.g. F2-6-10 was installed at location F2-6 at a depth of 10 cm).

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Verification Methodology

2008

A verification test was conducted in 2008 on forty (40) Tidbit temperature loggers prior to installation in the streams. The test was conducted in two phases; a warming test on July 1, 2008 (Test 2008-1), and a cooling test on July 6, 2008 (Test 2008-3). For both phases, the forty (40) Tidbit loggers were placed in a bathtub, with the YSI 6290 datalogger (Figure B1). The Tidbit loggers and the YSI were time synchronized and set to log temperature data at 5 minute intervals. During the test, ice or water was added to the tub two minutes increments following the logging interval. The two minute time lag was to avoid a temperature shift during the logging interval. Following the addition of the ice or water, the water in which the dataloggers were submerged was circulated to the water in the bath, and distribute the temperature. The temperature in the bath during the mixing was monitored in real time using the YSI logger.
Figure B1. Setup of the YSI logger (center) surrounded by the Tidbit Loggers (orange) in the tub, prior to the addition of water.

In the warming test (Test 2008-1) on July 1, 2008 at 15:00, the loggers were submerged in cold tap water (approximately 3°C), and allowed to equilibrate for thirty minutes. Hot tap water was then added to raise the temperature to approximately 15°C. Additional hot water was added in 10 minute increments until the water temperature was raised to approximately 49°C. The water was then allowed to cool, undisturbed, to room temperature over a period of 5.5 hours. The test was completed on July 1, 2008 at 22:15, and took a total of 7 hour and 15 minutes, which corresponds to 87 measurements at 5-minute intervals.

The cooling test (2008-3) was initiated at 10:00 on July 6, 2008. The dataloggers were submerged in maximum temperature tap water (~50°C) and allowed to cool, undisturbed for 5.5 hours to approximately 25°C. Ice was then added step wise to the water bath, gradually reducing the temperature to 0.09°C. The ice water was then allowed to warm undisturbed overnight for 12 hours, during which time it warmed to 1.7°C. The test was completed on July 7, 2008 at 05:25, for a total of 19 hours and 25 minutes, corresponding to 233 measurements at 5-minute intervals.

For both tests, the data for each of the forty Tidbit temperature loggers was plotted against the measurements of the YSI, an example of each test is presented in Figures B2 and B3. These data were then used to calculate the any offset of the Tidbit logger temperature recordings from the YSI recorded temperatures and to verify if the average difference between the recorded temperatures were within the range of uncertainty of ±0.35°C, which is the combined accuracy of both loggers, as listed by the manufacturers (TableB1).

For both verification tests, the difference between the Tidbit datalogger and the YSI was compared for the duration of the test and additionally, the period of undisturbed cooling was examined as a period of interest. An example of the warming test (2008-1) with the cooling period is shown in Figure B2, using Tidbit unit 1283600. For the warming test, the period of undisturbed cooling was the period beginning at the maximum temperature until the end of the test, circled in orange in Figure B2. For the cooling test, the period of undisturbed cooling was the initial temperature drop (circled in orange in Figure B3), from July 6, 2008 at 10:05 through 13:130. These two periods of the data were examined specifically as they likely best represent natural conditions with gradual
changes in the temperature. However, by comparing to the entire periods of record, the response to perturbations in the system can also be examined.

Figure B2. Temperatures recorded by Tidbit unit 1283600 and the YSI datalogger during the warming test. The temperature steps during the warming portion of the test correspond to addition of warm water to the temperature bath. The circled section (orange) indicates the period of undisturbed cooling.
Figure B3. Temperatures recorded by Tidbit unit 1283600 and the YSI datalogger during the cooling test. The temperature steps below 25°C reflect the intervals at which ice was added to the tub water. The circled section (orange) indicates the period of undisturbed cooling.

The temperature recorded by the YSI was subtracted from the Tidbit logger value at each logged interval and the mean difference for each period was calculated. The mean differences are presented in Table B4.

Table B4. 2008 verification mean data, with the YSI values subtracted from the Tidbit temperature values. Maximum values are indicated by the bolded values.

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Calibration verification was completed on forty-three (43) Tidbit temperature loggers following retrieval from the field sites. The Tidbit loggers and the HOBO Water Level Data Logger, which also records temperature, were time synchronized and set to log on 2 minute intervals, and placed in a portable cooler (Figure B4). The test was a combined cooling and warming test, recorded continuously between December 2 2012, 19:30, and December 5, 2012, 09:24. The test was designed to observe the synchronicity between the Tidbit loggers and the HOBO logger during both periods of undisturbed warming and cooling, as well as over periods of rapid change.

The test was initiated at 19:30 on December 2, 2012, and five minutes after the logging was started, cubed ice was added to the cooler, covering the dataloggers. The dataloggers were allowed to sit in the ice for approximately 5 minutes, and then covered with cold tap water. The dataloggers and the cold water were allowed to warm, undisturbed, for 42 hours and 42 minutes, until the water had warmed to approximately 15°C. The dataloggers were then transferred to a plastic bucket containing hot tap water, near 50°C. The water in the bucket was allowed to cool undisturbed for 6 hours and 30 minutes, until the water was approximately 25°C. To further lower the temperature below 25°C and increase the rate of cooling, cubed ice was added to the container. The water temperature was then allowed to warm, undisturbed, for 12 hours and 42 minutes to approximately 16°C, at which point the test was ended and all dataloggers removed from the water bath.

For both tests, the data for each data logger was plotted against the measurements of the HOBO logger, an example of each is shown in Figure B6. These data were then used to calculate the offset of the Tidbit logger from the HOBO values to assess if the average value was within the range of ±0.57°C, which is the combined accuracy of both loggers, as listed by the manufacturer (Table B1).
For the 2012 verification test, the difference between the Tidbit datalogger and the HOBO loggers was examined for the entire period, as well as for the periods of undisturbed warming and cooling, both circled in Figure B5 using Tidbit unit 1283600 as an example. The period of undisturbed warming was from December 2, 2012 beginning at 21:20 to December 4, 2012, ending at 14:10. The period of undisturbed cooling is from December 4, 2012, over the time period of 14:22 to 20:50. These two segments of the data were used to best represent natural conditions with gradual changes in the temperature. However by comparing to the entire periods of record, the response to perturbations in the system can also be examined.

[Image: Figure B5. Plot of the Tidbit temperature and HOBO loggers for the 2012 verification test. The circled sections (orange) indicate the period of undisturbed warming and cooling.]

The temperature recorded by the HOBO datalogger was subtracted from the Tidbit temperature values, and the mean difference for each period was calculated. The mean differences are presented in Table B5.
Table B5. 2012 verification data showing the mean differences between the temperatures recorded using the HOBO logger subtracted from the Tidbit temperature values. The mean and maximum values were repeated for the full data set, and with outlier values (bolded) from serial number 1297697 removed.

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<th>2012 Verification All Cooling Average Difference (Tidbit-HOBO)</th>
<th>2012 Verification Full Period Average Difference (Tidbit-HOBO)</th>
<th>2012 Verification Main Warming Period Average Difference (Tidbit-HOBO)</th>
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Results

The results for the 2008 verification indicate there was some offset between the YSI logger and the Tidbit loggers during the stepped phases of each test (Figure B2), but during the periods of undisturbed warming or cooling, there appears to be a strong correlation between the recorded temperatures (Figures B2 and B3). The offset during the stepped temperature changes in both sets of tests may be a result of the 5 minute response time of the Tidbit loggers, and the 3.5 minute response time of the HOBO logger. The response time may have overlapped the logging intervals during the tests and resulted in an apparent offset during the stepped test periods.

For the warming test for test 2008-1 (Figure B2), there were nine values that exceeded the 0.4°C accuracy of the loggers (orange data - Figure B6), and of those two values (Serial numbers 1283611 and 1297697) were greater than 0.5°C from the YSI recorded values. Most of the mean differences between the Tidbit loggers and the YSI were below 0°C, and as shown in Figure B2 and B3, most of those can be attributed to the offset during the steps and the water mixing. The second calibration verification in 2008 was the cooling test (2008-2), and the mean differences are shown as the blue data points in Figure B6. During undisturbed cooling period (blue data - Figure B6), only four values exceeded the 0.4°C accuracy rating of the datalogger. All of these values were positive, and do not coincide to the same dataloggers units with exceedances in the previous test. One value is an outlier (circled in Figure B6), with a negative difference > -0.2°C (Serial number 1297697) and while it does not exceed the equipment accuracy, it is observed to be an outlier in both verification test. Over the full period of warming and cooling in 2012, (Figure B7) unit 1297697 has a mean difference from the HOBO logger of -4.9°C and is clearly an outlier.
In comparing both verification tests from 2008 (Figure B6), the warming test (orange) had more scatter in the mean differences, and the differences were greater with the Tidbit loggers recording below the YSI logger more often. For the cooling test (blue), in which the loggers were allowed to cool with fewer perturbations in the water temperature, the differences between the loggers were less. There is only logger (Serial number 1297697) that consistently recorded values lower than the YSI, and appears to be consistently recording nearly 0.4°C below the other loggers.
The results for the 2012 verification test (Figure B7) shows that the data loggers follow a similar pattern to those tested in 2008, with one outlier (Serial number 1297697) that has a difference of nearly 5°C below the mean of the other forty-two dataloggers. With the outlier removed, there are no data loggers that exceed the threshold of accuracy listed in the specifications (Table B1)

**Drift Correction**

There were twenty-three (23) dataloggers that were tested in both 2008 and 2012 (Table B5). The results of the mean difference between the Tidbit loggers and the comparison logger are plotted in Figure B8, showing that for the two tests, the results from 2012 show more scatter and generally are shifted to more positive values. None of the values exceeded the accuracy threshold, with the exception of the previously discussed outlier (Serial number 1297697) which is not included Figure B8. The increased scatter and upward shift in the temperature differences in 2012 suggests there was drift over the four years of deployment.
The manufacturing specifications indicate a annual drift of 0.1°C/year, which would correspond to a shift of up to 0.4°C over the four year period. There was a small positive drift, but only four data loggers have mean difference as high as the potential ±0.4°C (Table B5 and Figure B8). There is potential that comparing to the YSI logger in 2008 and the HOBO logger in 2012 may have introduced some apparent shifts in the verification results. However, the occurrence of mixed positive and negative shifts suggests that the use of different secondary loggers did not introduce any systematic shifts.
Table B6. The data for dataloggers tested in both the 2008 and 2012 verification tests. The difference between the 2012 and 2008 mean differences, which indicates potential drift, is included.

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</tr>
</tbody>
</table>

*1297697 is an outlier, not included in the calculations of the mean values.
Using Aquarius software - Aquatic Infomatics, v. 3.0.75.1 (Aquatic Informatics Inc. 2012), a linear drift correction was applied to the Tidbit temperature loggers. For the twenty-three dataloggers with verification results from both 2008 and 2012 listed in Table B5 (not including 1297697), the drift correction applied was logger specific, using the drift calculated as the difference between the 2008 and 2012 temperatures as shown in Table B6. For the dataloggers installed following the 2008 verification period which did not have a baseline verification test, a standard drift correction of 0.04°C/year was applied. This value was determined from the mean annual drift in Table B6 rather than the manufacturing specifications, because the tests indicated the drift was less than the maximum specified.

Further examination of the outlier (serial number 1297697) in 2008 (Figure B9) for the cooling test shows the temperature matches closely the values recorded by the YSI logger within the specifications of the equipment, however, the Tidbit logger consistently records lower temperature, with a mean difference of -0.39°C from the YSI (Table B4). The verification test in 2012 (Figure B10) shows that Tidbit logger 1297697 has a uniform offset from the temperatures recorded by the HOBO logger. Comparison of the field data recorded by Tidbit loggers positioned in close proximity to 1297697 indicates the drift occurred gradually over the four years. The mean difference between this Tidbit logger (red line in Figure B10) and the HOBO logger (blue line in Figure B10) during the 2012 verification test was -4.92°C. A uniform correction factor of +4.9°C was applied to the Tidbit unit 1297697 values (green line in Figure B11). With the correction factor applied, unit 1297697 values matched the HOBO values with a mean difference from the HOBO logger of only -0.02°C. A linear drift correction was applied to Tidbit unit 1297697 of 1.13°C/year.
Figure B9. The verification curve for Tidbit logger 1297697 for the 2008-2 (cooling) test. The temperatures recorded by the Tidbit logger match the YSI values closely with a mean difference for the verification tests of -0.39°C. As with all of the Tidbit loggers, the greatest offset occurs during periods with the greatest rates of change.

Figure B10. Temperatures recorded using Tidbit logger unit 1297697 and the HOBO logger for the entire 2012 verification test. The plots show a uniform negative offset of -4.92°C between the Tidbit and HOBO loggers. Application of a uniform correction factor of +4.9° restores the match between the recorded temperatures.
Conclusion

In conclusion, the 2008 verification tests show the Tidbit loggers correlate well to the YSI recorded temperatures, within the factory calibration specifications and expected accuracy ranges for thirty-nine of the forty loggers. The correlation appeared strongest when the loggers were left undisturbed for a period of time, and that may be an artifact of the five-minute response time of the Tidbit loggers, which was the length of time for the recording interval. Tidbit logger with serial number 1297697 was consistently recording values below those of the YSI logger.

The 2012 verification tests also show reasonable correlation between the Tidbit loggers and the HOBO data logger. Consistent with 2008, one Tidbit logger (unit 1297697) was an outlier, recording values 4.9°C below the HOBO logger. By applying a correction factor of +4.9°C, a good match between the two data series was restored.

Of the forty loggers tested in 2008, and the forty-three in 2012, the tests were repeated in both years on only twenty-three loggers, as some dataloggers were lost in the field, and others were added to the field program without a baseline verification test, but were necessary to replace missing units, or for different phase of the research. The results of the verification tests were used to apply linear drift corrections to the Tidbit loggers, based on the results for the individual loggers where available, or using the mean value of 0.04°C/year determined from the twenty-three available loggers.
Appendix C.

Seepage Meter Design

Overview

Seepage meters were initially designed for use in lakes (Lee 1977), and with modifications have been used effectively in streams and flowing water to quantify fluid exchange across the streambed interface (Alexander and Caissie 2003; Kalbus et al. 2006; Essaid et al. 2008; Rosenberry and LaBaugh 2008). Seepage meters for use at the Fishtrap and Bertrand Creeks study sites were designed following Lee (1977) with modifications based on Rosenberry (2008). The general design of a seepage meter is a bottomless cylinder, set into the streambed sediments, and connected to a light weight collection bag that is protected from the pressure of the flowing water in a stilling well (Kalbus et al. 2006; Rosenberry 2008). The seepage meter is allowed to equilibrate in the sediment following installation, and is vented to the atmosphere to allow gasses and pressure in the meter to equilibrate. Once the measurements are initiated, water collects in the light weight bag in a gaining stream, or drains a known volume over time in the case of a losing stream. The seepage meter collection method provides an estimate of seepage flux over time at the location the seepage meter is installed.

At the field sites in this research, the summer water levels and flow conditions are different between the study locations, and therefore, two seepage meters with different heights were designed for this field study. The seepage meter designed for use in deep water and finer sediments was the base of a plastic tub, inverted to insert the open end into the bed sediments. The meter had a diameter of 0.44 m, and a height of 0.24 m (Figure C1). The seepage meter designed for use in shallow water was a low profile design, constructed using a plastic garbage can lid and had a diameter of 0.5 m, and a total height of 0.12 m (Figure C2). The seepage meters were vented to the atmosphere to allow equilibration of the gases and pressure inside the meter, and the venting occurred through an open tube attached to rebar above the water surface (Figures C1 and C2). The seepage collection bag for both seepage meter designs were placed in a stilling well (Figures C3 and C4) to protect the collection bag from the force of the flowing water. Flowing water has the potential to apply external force to the collection bag, create resistance in the bag to the influx of seepage water (gaining stream) or apply force that may increase the rate of drainage into the seepage meter (losing stream). The stilling well is positioned parallel to the seepage meter within the flow path in order to prevent head difference between the seepage meter and the collection bag from influencing the flow rate. The stilling well was a large plastic garbage can with holes drilled at regularly spaced intervals along the side wall to allow passive exchange of water with the channel, and was positioned upright in the deeper water (Figure C3), or on its side in the shallow water (Figure C4).
Figure C1. The seepage meter designed for use in the deeper flow conditions. The seepage meter is installed to a depth of 6-8 cm into the bed sediments, and vented to the atmosphere using the plastic tubing attached to the rebar.
Figure C2. The seepage meter designed for use in shallow flow, constructed from the lid of a plastic garbage can. The meter is inserted 4 cm into the bed sediments, and is vented to the atmosphere through the plastic tubing attached to the rebar. The unit is weighted down on the channel bed for stability during equilibration, and to prevent agitation from the flow.
Figure C3. Installation of the seepage meter in the deeper water was completed using snorkelling equipment. The seepage collection bag is visible in the stilling well (right side of the photo), positioned in the channel parallel to the meter.
Figure C4. The seepage meter collection bag in the stilling well in shallow water. The protection container is positioned on its side, parallel in the channel to the seepage meter. The air was removed from the bag as much as possible to allow it to passively fill with seepage water.

Design Specifications

The seepage meter designed for deeper water was fitted with a barbed brass nozzle at the top of the apparatus for attachment of the vent tube, and a second barbed brass nozzle on the side of the meter, near the top, for attachment of the tubing connected to the collection bag (Figure C1). For the lower profile meter, both brass nozzles were fitted at the top of the meter to accommodate the low profile design (Figure C2). On both seepage meters, the fittings were sealed with silicone, and fitted with clear plastic tubing. The vent tubing (4 mm inside diameter) was open the full length and was secured above the water surface to rebar. The water from the seepage meter would rise inside the vent tube to a level approximately even with the stream water surface when the unit was full equilibrated. The seepage water drained through 6 mm (inside diameter) stiff plastic tubing. During meter operation, the plastic tubing was connected to a second piece of
tubing via an inline valve which connected to the tubing with a barbed brass fitting. The second length of tubing was approximately 0.3 m before the seepage collection bag (Figure C4). The 0.3 m segment of tubing after the inline valve was open-ended to allow attachment to the seepage meter fittings. The collection bags used were double lined, 3 or 4 L plastic wine bags. The bags had their original valves removed and the openings were refitted with a high density polyethylene (HDPE) insert. The HDPE was fitted with a rubber O-ring, and adhered to a barbed brass fitting, over which the plastic tubing from the inline valve was attached (Figure C4) during the meter operation.

To assemble the seepage collection bag apparatus in the stilling well, the tubing from the meter was inserted at the base of the stilling well, either in the upright position in deep water, or on its side in the shallow water, and any air in the tubing was drained prior to attaching it to the second section via the inline valve. The tubing was weighted to the streambed, if necessary, to keep all parts of the collection apparatus below the stream water surface and this weighting was generally only necessary in the shallow flows. As much air as possible was removed from the collection bag by compressing it, and it was attached to the end of the two pieces of plastic tubing (Figure C4). The bag was smoothed as much as possible prior to the start of measurements, to ensure the bag sat evenly in the water, and that the bag opening was not blocked by bag material (Figure C3). The bag was left undisturbed in the stilling well for the duration of the collection period once the valve was opened.

**Installation Specifications**

The seepage meter installed in the deeper flow was inserted slowly into the bed sediments, evenly to a depth of approximately 6-8 cm into the channel sediments. Installation required between 1 to 2.5 hours, using snorkelling equipment (Figure C3). The installation of the lower profile meter generally took less than an hour, and the meter was inserted 4 cm into the bed sediments which was the maximum depth, based on the height of the sidewalls. The meters were inserted slowly into the channel bed by rotating the meters and applying pressure to as evenly as possible. The goal was to minimize the disturbance to the bed sediments during the installation process. For the larger seepage meter, the depth of installation of 6-8 cm was considered optimal in order to achieve a seal around the base of the meter, and secure the meter firmly, without creating a disturbance in the sediments that blocked or altered flow paths.

The seepage meters were allowed to equilibrate following installation, to allow equilibration of the gasses and pressure within the meter, as well as to allow fine sediments around the base that may have been disturbed during installation to settle. Following the equilibration period, the contact between the meter and the bed was checked for scouring or uneven contact. The meter installation would be adjusted if necessary if the installation was uneven, or relocated in the case that scouring had occurred. Scouring indicated a failed installation in which stream water was able to short circuit directly into the seepage meter. The equilibration period in the streams was between 1 to 6 hours, at a minimum, and was determined when the fluid level in the vent tube was at the same elevation as the stream water. In most cases, the seepage meters were allowed to equilibrate overnight prior to operation. Following the assembly of the collection bag apparatus in the stilling well, the inline valve would be opened, and the collection bag allowed to passively fill (or drain) over a timed interval. In the instance of a
losing stream, the collection bag had a known volume of stream water added at the start of the measurements. Where possible, the bag and the stilling well were positioned in shade to prevent expansion of the bag or heating of the collection water due to direct sunlight.

**Data Collection**

Measurements of seepage rates were collected over time intervals between 30 to 60 minutes each. The measurements were repeated between 8 and 24 times per day. To measure the seepage rate, the inline valve was turned off, and the bag disconnected from the tubing and the change in water volume measured using a 1L plastic graduated cylinder. To calculate the flux, the volume change over time (mL/min) was divided by the area of the seepage meter (m²) and reported in cm/day. The final flux values reported were calculated to include a resistance factor of 1.05 applied to the measured flux values. The resistance factor was included to compensate for underestimation of the flux rates due to frictional forces and resistance to flow through the seepage meter and into the collection bag (Rosenberry and LaBaugh 2008).
Appendix D.

Riparian Mapping

Riparian mapping was completed regionally in the Fishtrap Creek watershed. The first stage of the mapping was delineation of the polygons of five classes of riparian vegetation from orthophotos listed in Table D1. Table D2 is the reach break attribute table from Arcmap listing the reach breaks and vegetation classification.

Table D1. Orthophotos used for delineation of riparian polygons in the Fishtrap Creek watershed.

<table>
<thead>
<tr>
<th>Watershed</th>
<th>Roll Tag</th>
<th>Date Taken</th>
<th>Air Photo Frames</th>
<th>Scale</th>
<th>Operation Tag</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fishtrap</td>
<td>30BCC96085</td>
<td>23-Jul-96</td>
<td>47, 48, 149-151</td>
<td>1:15000</td>
<td>C-009-FI-96</td>
</tr>
<tr>
<td>Fishtrap</td>
<td>15BCB95045</td>
<td>28-Jun-95</td>
<td>32,33</td>
<td>1:50000</td>
<td>B-018-FI-95</td>
</tr>
<tr>
<td>Fishtrap</td>
<td>15BCB91157</td>
<td>09-Sep-91</td>
<td>188</td>
<td>1:40000</td>
<td>B-164-FI-91</td>
</tr>
<tr>
<td>Fishtrap</td>
<td>30BC88007</td>
<td>07-Jul-88</td>
<td>62, 102, 139-141</td>
<td>1:15000</td>
<td>B-159-A-88</td>
</tr>
</tbody>
</table>

Table D2. Fishtrap Creek riparian vegetation reach breaks and associated vegetation class.

<table>
<thead>
<tr>
<th>ReachID</th>
<th>Riparian Class</th>
</tr>
</thead>
<tbody>
<tr>
<td>FTRV1</td>
<td>Grass</td>
</tr>
<tr>
<td>FTRV2</td>
<td>Shrub</td>
</tr>
<tr>
<td>FTRV3</td>
<td>Shrub</td>
</tr>
<tr>
<td>FTRV4</td>
<td>Deciduous</td>
</tr>
<tr>
<td>FTRV5</td>
<td>Shrub</td>
</tr>
<tr>
<td>FTRV6</td>
<td>Deciduous</td>
</tr>
<tr>
<td>FTRV7</td>
<td>Deciduous</td>
</tr>
<tr>
<td>FTRV8</td>
<td>Coniferous</td>
</tr>
<tr>
<td>FTRV9</td>
<td>Shrub</td>
</tr>
<tr>
<td>FTRV10</td>
<td>Shrub</td>
</tr>
<tr>
<td>FTRV11</td>
<td>Lake/Wetland</td>
</tr>
<tr>
<td>FTRV12</td>
<td>Deciduous</td>
</tr>
<tr>
<td>FTRV13</td>
<td>Deciduous</td>
</tr>
<tr>
<td>FTRV14</td>
<td>Deciduous</td>
</tr>
</tbody>
</table>
The riparian mapping was verified in the field by ground-truthing the polygons classified from the orthophotos and reach breaks in ArcMap (ArcMap 9.3). Select field photos showing examples of the riparian classes throughout the watersheds, and highlighting the characteristics of the regional sites are provided in Figures D1 through D20. The regional study locations are shown on Figure 2.7, and the photos are arranged by location sequence up the stream from the lower reaches towards the upper reaches.

**Regional Riparian Ground-Truthing**

![View north at site F1. Riparian vegetation is dominated by grasses. Blue line indicates the approximate center of the channel.](image)

*Figure D1.* View north at site F1. Riparian vegetation is dominated by grasses. Blue line indicates the approximate center of the channel.
Figure D2. View north from the bridge at site F2. Riparian vegetation is dominated by grasses. In the background, a seepage meter sample is being measured, adjacent to the in-stream piezometers. The tape measure in the foreground is the location of one of the manual stream discharge transects.
Figure D3. Site F4, riparian class is dominated by grasses, with minimal deciduous trees.

Figure D4. Site F5 dominated by grasses in the riparian zone.
Figure D5. Riparian vegetation between sites F5 and F6, showing the riparian zone dominated by grasses, with deciduous trees bordering the extent of the riparian zone.

Figure D6. Deciduous riparian vegetation at site F6 on Waetcher Creek.
Figure D7. Site F7 on Waetcher Creek, with a mix of deciduous trees and shrubs. The reach overall is dominated by deciduous vegetation. The pink ribbon indicates the location of a Tidbit temperature logger.
Figure D8. Site F13 in the upper reaches of Waetcher Creek. Riparian vegetation is deciduous vegetation. The pink ribbon indicates the location of a Tidbit temperature logger.
Figure D9. Mixed coniferous and deciduous riparian vegetation along Waetcher Creek tributary near the origin, north of F13.
Figure D10. Regional site F8 mixed shrubs and deciduous vegetation, overall classified as dominated by shrubs. The pink ribbon indicates the location of a Tidbit temperature logger.
Figure D11. View south at regional site F8 mixed shrubs and deciduous vegetation, overall classified as dominated by shrubs.
Figure D12. Deciduous vegetation, mixed with shrubs, at site F9.
Figure D13. Riparian vegetation north of site F8, mix of coniferous and deciduous trees, with some shrubs, classed as conifer dominant.
Figure D14. Riparian vegetation at site F12 in the upper reaches of Fishtrap Creek, near the origin of the flow of Fishtrap Creek. This section is a mix of coniferous and deciduous, and is classed as coniferous dominant.
Figure D15. Site F3 with deciduous vegetation overhanging the channel. The in-stream peizometers and seepage meter installation are visible in this photo, and a Tidbit temperature logger is afixed to the rebar in the foreground.
Figure D16. Enns brook tributary north of site F3, view is to the south, towards F3. This section was classed as dominant deciduous riparian vegetation.
Figure D17. Enns brook tributary, view north from the reach break north of Figure D16. This is a wetland-type area, with grasses, shrubs, and deciduous vegetation. This section was classed lake/wetland.

Figure D18. Riparian shrubs as site F10 (Enns Brook). This location was impounded by a beaver dam during portions of the study.
Figure D19. Deciduous riparian vegetation at site F11 (Enns Brook), with shrubs in the zone immediately adjacent to the stream.

Figure D20. Lake/wetland area in Fishtrap Creek Park in the upper reaches of the watershed.
Appendix E.

Exploratory Data Analysis Figures

Histograms

Hourly histograms of the data for each local scale sediment-water interface temperature logger location are provided in the figures below. The figures are arranged to show side-by-side histograms of the hourly temperature data for the full period of record and the summer low flow period for each location.

*Fishtrap Creek – F1*
Bertrand Creek – B1

Bertrand Location 1_AllData_Hourly

Bertrand Location 1_LowFlow_Hourly

Bertrand Location 2_AllData_Hourly

Bertrand Location 2_LowFlow_Hourly
QQ Plots

Hourly sediment-water interface temperature data were plotted as Quantile-Quantile (QQ) plots for each datalogger location at sites F1 and B1. The QQ plots are plotted with all available data for each location.

*Fishtrap Creek – F1*
Bertrand Creek – B1

Bertrand Location 1

Bertrand Location 2

Bertrand Location 3

Bertrand Location 4
Hourly sediment-water interface temperature data were plotted as empirical cumulative distribution functions. (ecdf) plots. The first figure shows the ecdf plot for each datalogger location plotted together for both F1 and B1.
The following ecdf plots show the hourly temperature data for each site, plotted by separate summer months and compiled as summer periods for each year.

**Fishtrap Creek – F1**
Bertrand Creek – B1

Bertrand Jul 2008

Bertrand Jul 2009

Bertrand Jul 2010

Bertrand Jul 2011
Boxplots

Notched boxplots of the sediment-water interface temperatures at each data logger location are included below, showing both the hourly and mean daily temperature distributions. The boxplots of the summer period months are indicated by yellow boxes for both hourly and daily plots.

Hourly boxplots

Fishtrap – F1
Bertrand Hourly - Oct

Temperature °C

Datalogger Location

Bertrand Hourly - Nov

Temperature °C

Datalogger Location
Daily boxplots

Fishtrap – F1
Fishtrap Daily- Apr

Fishtrap Daily- May
Bertrand – B1

Bertrand Daily - Jan

Temperature °C

Data logger Location

Loc. 1  Loc. 4  Loc. 7  Loc. 10  Loc. 13  Loc. 16  Loc. 19
Appendix F.

Level III Assessment - Comparing Impacts for Non-Pumping and Pumping for Fishtrap and Bertrand Creek Watersheds

Non-Pumping Impacts to the Watershed Zones

The Zone Budget results for the non-pumping conditions for Fishtrap and Bertrand Creek watersheds are shown in Figures F1 and F2, respectively. The results are divided into the main Zone Budget zones: the watershed zone, the main-stem stream zones, and the ephemeral stream zones. The results show the main water inputs to the watershed zones is recharge with some exchange between zones. The exchanges between the zones for each watershed are dominated by exchange between the watershed zones (zones 2 and 9) and the main aquifer zone (zone 1), rather than with the main stem and ephemeral zones. Water leaving the watershed zones, goes partially into storage and partially as zone transfers (discussed in more detail below). The Zone Budget results for the stream segments (Figures F1 and F2) indicate that water exchange from other zones, primarily from the broader watershed zone, is the main input of water, and water leaves the stream main stem zone via the river and drain cells which comprise these two zones. For the ephemeral stream segments, the drain cells are the primary outflow zones, which is to be expected given that drains are the boundary condition for most of the ephemeral reaches.

The gain in storage over the year is an interesting outcome and a consequence of the initial head distribution (steady-state heads) for the transient simulation. An examination of the heads time series suggests that heads gradually increase over the simulation time period, consistent with water being added to storage. Upon closer examination of the detailed water balance, water is made available to a zone from storage or is lost from the zone to storage. In Fishtrap and Bertrand watersheds, more water is supplied to the zone from storage during the summer months, and in the winter more water goes into storage. This is an expected outcome. Over the long term, roughly the same amount of water should go into and out of storage on an annual basis. However, for this single simulation year, both watershed zones lose more water to storage (Figures F1 and F2). Thus, the annual water budget is not reflecting conditions that might be expected over the long term. The model is not in dynamic equilibrium for this single year simulation. Multiple years could be run to achieve a better dynamic equilibrium, and the first few years of output removed for analysis of outputs. This is referred to as model spin up. The water balance would then be examined over the latter portion of the simulation period. Consequently, the overall annual water balance results should be viewed with caution. Over the longer term, the amount of water entering storage should equal the amount leaving storage. This suggests that of the amount of recharge added to the watershed, most of the water will exit to other zones, resulting in the green bars (zone transfer out of the watershed in Figures F1 and F2) comprising the only outflow component. Notwithstanding the storage problem, changes in the water balance results between the pumping and non-pumping conditions are likely fairly representative. If water does not
come from storage to supply pumping, then water would come from elsewhere. Therefore, the results are conservative.

**Figure F1.** Zone Budget results for Fishtrap Creek for non-pumping conditions.

**Figure F2.** Zone Budget results for Bertrand Creek for non-pumping conditions.

**Pumping Impacts to the Watershed Zones**

The results for Fishtrap and Bertrand Creeks for the various pumping scenarios are presented in Table F1. The table includes the non-pumping scenario in the first column. The reported values represent the annual volume of water leaving the two watersheds through the wells, river and drain zones, as well as outflow to other zones.
Table F1. Zone Budget results showing the annual water volume out of the Fishtrap and Bertrand Creek watersheds zones.

<table>
<thead>
<tr>
<th></th>
<th>No Pumping (m³/yr)</th>
<th>Pumping Single Watershed (m³/yr)</th>
<th>Pumping Other Watershed (m³/yr)</th>
<th>Pumping Both Watersheds (m³/yr)</th>
<th>Pumping All Model Wells (m³/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fishtrap Creek</td>
<td>3.46E+07</td>
<td>4.33E+07</td>
<td>1.12E+07</td>
<td>2.69E+07</td>
<td>2.69E+07</td>
</tr>
<tr>
<td>Bertrand Creek</td>
<td>3.31E+07</td>
<td>3.56E-07</td>
<td>2.83E+07</td>
<td>3.58E+07</td>
<td>3.58E+07</td>
</tr>
</tbody>
</table>

The pumping scenarios show that for each stream, the strongest influence is the result of pumping within the respective watershed, manifested by an increase in the water volume out of the watershed (Table F1; Figures F3 and F7). There is an apparent decrease in the volume of water out of each watershed through the river, drain, wells, and to other zones during the pumping scenarios when wells in other model areas are activated. This is due diversion of available water to the pumped wells, and a subsequent decrease in water volume into the individual watersheds.

Cumulative impacts from pumping outside the watershed result in a decrease in the water volume out of the watershed (Table F1) and the volumes are the same if the pumping is occurring in the two watersheds (Figures F5 and F6) or aquifer-wide (Figures F9 and F10). Therefore, the watershed boundary (in this study) provides an appropriate boundary to monitor pumping when there is pumping across a wider area.

When pumping occurs only in the adjacent watershed (Figure F4 and F8), there is a larger apparent decrease in the water volume out of the watershed. This is because available water in the model is being drawn to the adjacent watershed, and there is no pumping in the specified watershed drawing water from other zones. There is the potential that pumping in each watershed, when isolated, will have influence on an area larger than the watershed boundary. The impacts to adjacent watersheds can be direct as well, as seen in the Bertrand watershed, where a minor volume of water from the ephemeral stream segments is lost to wells when pumping is activated in the Fishtrap Creek watershed (Figure F8). These ephemeral stream reaches are higher order segments of the stream, which are closer to the boundary of the watershed, and therefore some segments are located closer to the pumping wells adjoining watershed. Consequently, pumping in adjacent watersheds only can have the combined effect of intercepting water that would have entered the watershed, and directly drawing water away. For the scenario with all of the pumping wells across the model area activated, the results indicate decreases in total water volume out of each watershed on the order of approximately 10⁵ m³/year, relative to the non-pumping scenario.
Figure F3. Zone Budget results for Fishtrap Creek for wells pumping only in the Fishtrap Creek watershed.

Figure F4. Zone Budget results for Fishtrap Creek for wells pumping only in the adjacent Bertrand Creek watershed.
Figure F5. Zone Budget results for Fishtrap Creek for wells pumping in both Fishtrap and Bertrand Creek watersheds.

Figure F6. Zone Budget results for Fishtrap Creek for all wells pumping throughout the model domain.
Figure F7. Zone Budget results for Bertrand Creek for wells pumping only in the Bertrand Creek watershed.

Figure F8. Zone Budget results for Bertrand Creek for wells pumping only in the adjacent Fishtrap Creek watershed.
Figure F9. Zone Budget results for Bertrand Creek for wells pumping in both Fishtrap and Bertrand Creek watersheds.

Figure F10. Zone Budget results for Bertrand Creek for all wells pumping throughout the model domain.