

USE OF STREAM RESPONSE FUNCTIONS AND STELLA SOFTWARE TO DETERMINE
IMPACTS OF REPLACING SURFACE WATER DIVERSIONS WITH GROUNDWATER
PUMPING WITHDRAWALS ON INSTREAM FLOWS WITHIN THE BERTRAND CREEK
AND FISHTRAP CREEK WATERSHEDS, WASHINGTON STATE, USA

By

ERIK BRIAN PRUNEDA

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the requirements for the degree of

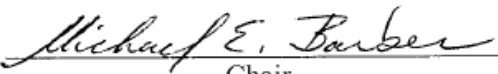
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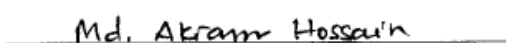
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Department of Civil and Environmental Engineering

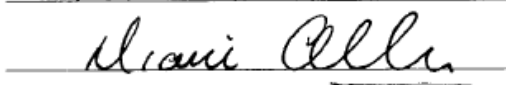
DECEMBER 2007

To the Faculty of Washington State University:

The members of the Committee appointed to examine the
dissertation/thesis of ERIK BRIAN PRUNEDA find it satisfactory and recommend that it be
accepted.


Chair


Md. Akram Hossain


Miami Allen



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Abstract

by Erik Brian Pruneda, M.S.
Washington State University
December 2007

Chair: Michael E. Barber

A numerical groundwater model was used to study the impacts of replacing surface water diversions with groundwater pumping wells within the Bertrand and Fishtrap watersheds, Whatcom County, Washington, USA. A regional steady-state groundwater flow model was calibrated to locally-observed conditions, using stream flow measurements, groundwater level data, and streambed hydraulic conductivity collected as part of this study. Stream response functions were calculated for individual wells placed at varying distances from the streams, to determine the impact these replacement wells might have on the instream flows of Bertrand and Fishtrap Creeks. Response ratios ranged from 0 to 1.0, with high ratios occurring in close proximity to the creeks and within areas of high aquifer hydraulic conductivity. Response ratios less than 1.0 indicate that groundwater pumping wells will have less of an impact on stream flow than taking the same amount of water directly from a surface water diversion. On the basis of this modeling study, replacing surface water diversions with groundwater pumping withdrawals may be a viable alternative for increasing summer stream flows.

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1. Introduction

Consumption of water for municipal and irrigation uses can have adverse impacts on minimum instream flows necessary for ecosystem health. In the Pacific Northwest, this problem is most acute during summer and early fall months when dry weather and increased demands combine to create severe water shortages in many streams (Adelsman, 2003). Moreover, the problem is not just limited to surface water diversions, as many alluvial well withdrawals cause significant decreases in streams flows through surface and groundwater interaction (Winter *et al.*, 1998).

Recent instream flow studies on two watersheds in Northwest Washington (Bertrand and Fishtrap) have indicated that summer flows are too low to support desired salmon uses (Kemblowski *et al.*, 2002; WAC 173-501-030, 1985). An innovative way to manage water demand is needed to help alleviate this problem. One proposed alternative involves replacing surface water diversions with groundwater pumping withdrawals. The desire is to have the lag between the time of groundwater withdrawal and its adverse effect on the stream extend into the winter rainy season when runoff has begun to increase stream flow and recharge has begun to replenish the aquifer. While removing surface water diversions will keep the previously-diverted water in the stream, the overall net effect on stream flow will depend on the location of replacement wells and the complex interaction of aquifer and streambed properties.

A regional steady-state MODFLOW groundwater model was previously developed by Scibek and Allen (2005) for use in two studies of the Abbotsford-Sumas Aquifer: to identify the potential impacts of climate change on groundwater (Scibek and Allen, 2006) and to simulate nitrate transport within the aquifer (Allen *et al.*, 2007). However, because of the regional scale, the model contained insufficient localized information to accurately examine groundwater and surface water interactions for specific stream reaches. The objective of this project was to

incorporate local information into the Scibek and Allen groundwater model to determine the impact on stream flow in Bertrand and Fishtrap Creeks by the exchange of surface water diversions for groundwater pumping wells. The model was adapted in this study using seepage analyses data from Bertrand and Fishtrap Creeks, local groundwater and surface water elevations, streambed hydraulic conductivities, and additional groundwater pumping well locations and rates of extraction. Because MODFLOW can be difficult to understand and operate for non-specialists, the simulated stream response functions were incorporated into a groundwater and surface water interaction tool. The interaction tool was created in the Stella environment, and allows the user to simulate the effects on the instream flows of Bertrand and Fishtrap Creeks through exchanging a surface water diversion for a single replacement groundwater well of the same withdrawal rate, without the need to run the MODFLOW groundwater model. Because a steady-state groundwater model was used, the stream flow responses represent a worst-case scenario because the zones of influence of the pumping wells are at a maximum under steady-state conditions.

The project site is situated within the Abbotsford-Sumas, trans-national aquifer, which extends from southern British Columbia, Canada southward into northern Washington, USA (Figure 3). The study area specifically encompasses the Bertrand and Fishtrap watersheds within Whatcom County near Lynden, Washington (Figure 1). Approximately 46% of the Bertrand watershed, 50.2 km² (19.4 mi²), and 39% of the Fishtrap watershed, 37.3 km² (14.4 mi²), are within the United States, with the remaining areas extending into Canada. Pepin Creek also begins in Canada and is a significant contributor of water to Fishtrap Creek year around. Bertrand Creek is a naturally-formed meandering stream, whereas Fishtrap Creek has been channelized in many places to accommodate agriculture and reduce flooding. The predominant land use within the

U.S. for both watersheds is agricultural, with the town of Lynden (population 9,020) being the only urbanized area in the region. The area has warm, dry summers and mild, wet winters, receiving on average 88.9 cm (35 in) of precipitation per year, with 18% falling during the months of June through September, and 82% during the months of October through May (McKenzie, 2007). The Abbotsford-Sumas Aquifer underlies the study area and is composed primarily of non-stratified silts, clays, sands, and gravels (Culhane, 1993). The vertical extent of the aquifer ranges from 7.6 m (25 ft) thick near Blaine, WA (western edge) and 22.8 m (75 ft) thick near Sumas, WA (eastern edge), while the tertiary bedrock surface underneath the study area is approximately 213.36 m (700 ft) below the ground surface (Scibek and Allen, 2005).



Figure 1. Location of Lynden, Washington with outlines of Bertrand and Fishtrap watersheds.

2. Previous Studies

Welch *et al.* (1996) conducted a pilot low-flow investigation on several small tributaries and along the main stem of the Nooksack River during the summer of 1995. The purpose of the investigation was to collect concurrent stream flow, groundwater level, and precipitation data. Bertrand and Fishtrap Creeks were found to be gaining reaches from the USA-Canada border to their termini at the Nooksack River, while Pepin Creek was found to be a losing reach. Recorded groundwater levels correlated well with daily precipitation measurements.

Cox *et al.* (2005) conducted a groundwater and surface water interaction study for streams in the lower Nooksack River basin of Whatcom County, Washington. A network of nine in-stream piezometers was installed in Fishtrap Creek to measure the local vertical hydraulic gradients between the stream and underlying aquifer. The magnitudes of the vertical hydraulic gradients were found to be higher during the winter rain season, November to April, and were lower during the summer and early fall dry season, June to September. Vertical hydraulic gradients were generally upward during their study period indicating discharging groundwater, except for one piezometer located within the town of Lynden. The gradient measurements at this well were consistently negative indicating stream flow recharging groundwater (Cox *et al.*, 2005). Upon analyzing individual storm events, Cox *et al.* (2005) stated that, “surface-water and ground-water levels respond rapidly to precipitation events, and periods of negative vertical hydraulic gradients occur during peak streamflows, but typically are of short duration.”

Culhane (1993) calculated theoretical stream depletion rates expected under various pumping scenarios within the Abbotsford-Sumas Aquifer. The Jenkins analytical model (Jenkins, 1968a, 1968b) was used to calculate the rate of stream depletion caused by nearby wells during and after pumping. The main assumptions of the Jenkins analytical model are: an isotropic and

homogeneous aquifer, a pumping well that is open to the full saturated thickness of the aquifer, transmissivity does not change with time, and the pumping rate is steady for the entire pumping period (Jenkins, 1968b). Transmissivity data were estimated for the aquifer from well specific capacity data. Pumping rates and durations were used in various combinations for this analysis. While the goal of the study was to determine a critical distance for separating wells from nearby streams in order to minimize stream depletion, a single critical distance was not found to be scientifically defensible due to the limitations of the Jenkins model and the variety of factors that cause stream depletion by pumping wells. The assumptions of the Jenkins model do not hold true for the Abbotsford-Sumas Aquifer. The aquifer is not isotropic and homogeneous, pumping wells are not commonly open to the full saturated thickness nor are they pumped continuously at a steady rate, and because the aquifer is unconfined the transmissivity changes over time. Analytical models can only provide a rough estimate of stream depletion, whereas a properly developed numerical model can provide a much more accurate prediction.

A regional groundwater flow model was previously created for the Abbotsford-Sumas Aquifer using Visual MODFLOW 4.2 by Scibek and Allen (2005). The model was calibrated to steady-state conditions representative of average August conditions (i.e., stream base flow, groundwater levels). The lithostratigraphy for the region was mapped using over 2000 well lithologic logs in combination with surficial geology maps and depositional models. Hydrostratigraphic units were defined based on the lithostratigraphy and available estimates of hydraulic properties, and the various units were assigned representative hydraulic conductivity, porosity, and storativity values (Scibek and Allen, 2005). Ten layers were used to represent the aquifer; each of which was comprised of varying hydraulic conductivity zones. Hydraulic conductivity zones within the study area for layer 1 are shown in Figure 2. Additional layers can be found in Appendix C.

Model boundary conditions were based on both physical and hydrologic features. The lower model boundary corresponds to the bedrock surface, as the bedrock is considered relatively impermeable. The model domain extends slightly beyond the Abbotsford-Sumas aquifer proper, as illustrated in Figure 3, in order to adequately represent the physical and hydrologic features that can serve as appropriate model boundary conditions. These include regional surface water divides to the west and north, and bedrock outcrops to the east. Surface water divides are thought to approximate the regional groundwater divides as the aquifer is largely unconfined. Other model boundary conditions include water bodies with specified heads and ditches, corresponding to the major rivers that receive this drainage (i.e., the Nooksack and the Sumas Rivers), and the numerous streams that drain the aquifer, respectively. Head values for specified head features were determined using a combination of survey data and topographic information as described by Scibek and Allen (2005). Finally, recharge was modeled using the HELP software developed by the US Environmental Protection Agency, and mapped spatially across the aquifer, taking into consideration the range of soil media types, shallow aquifer permeability and depth to water-table (Scibek and Allen, 2006).

In collaboration with Simon Fraser University, I used this model in this study to examine the complex interactions between the surface water and groundwater in the Bertrand Creek and Fishtrap Creek watersheds. However, to accomplish the objectives, additional local information would be needed.

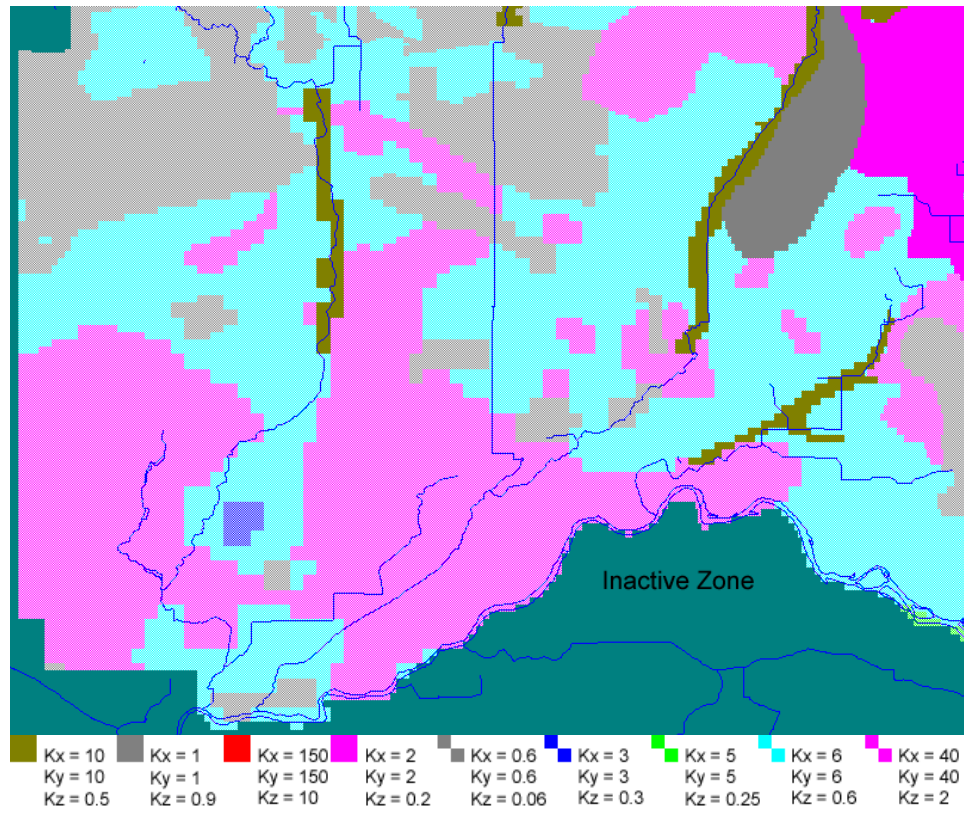


Figure 2. Hydraulic conductivity zones for model layer 1 within the study area. Units are in m/d.

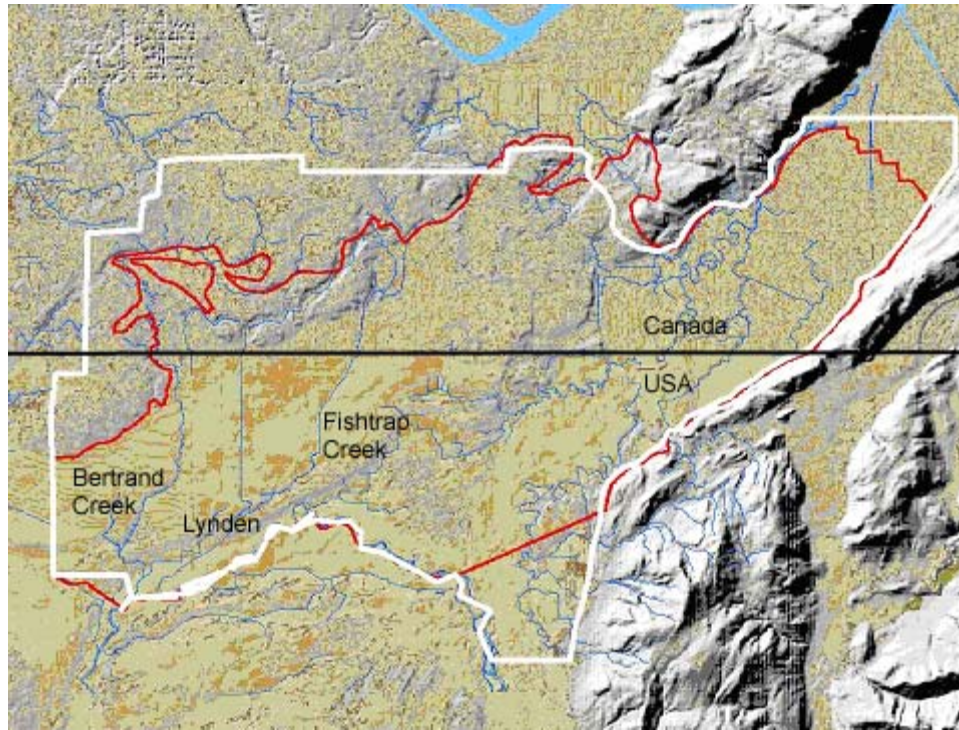


Figure 3. Horizontal extent of the model domain, white line, and boundary of Abbotsford-Sumas Aquifer, red line (Scibek and Allen, 2005).

3. Field Investigation

The field investigation included seepage analyses of both Bertrand and Fishtrap Creeks and their tributaries, streambed hydraulic conductivity measurements for both Bertrand and Fishtrap Creeks, monitoring of static groundwater levels in selected wells near each stream, and monitoring of stages of each stream.

The seepage analyses were conducted in July 2006 during low-flow conditions. Measurements were taken using a Pygmy or AA current meter following standard USGS procedures (Buchanan and Somers, 1969). Velocity and area data for each location were input into a spreadsheet for discharge estimation (Thomas Cichosz, State of Washington Water Research Center, 2007, personal communication). According to the Oregon Water Resources Department, the accuracy of a stream flow measurement is considered to be good if the value is within $\pm 10\%$ of the true

stream flow. The locations of flow measurements are shown in Figure 4 and data in Tables 1 and 2.

Bertrand Creek was found to be gaining water from the aquifer up to site B-4 where presumably, surface water diversions used for irrigation purposes cause the stream flow to steadily decline (Figure 5). Fishtrap Creek was also found to be a gaining stream, except for site F-3, where a loss in discharge relative to site F-2 was found (Figure 5). This loss of water seems to be consistent with the results of the Cox *et al.* (2005) study.

Estimation of surface water diversions was necessary in order to compare the field flow values to the predicted values by the MODFLOW Zone Budget analysis within the numerical groundwater model. Location and quantities of surface water rights were available in the form of a GIS database created by the Public Utilities District 1 Water Rights Team for the Water Resource Inventory Area (WRIA) 1 Watershed Management Project. These data were used in conjunction with local knowledge from Henry Bierlink, Administrator of the Bertrand Watershed Improvement District, and observations during the field investigation, to determine locations and quantities of surface water diversions for Bertrand and Fishtrap Creeks. These estimates were added to the measured field values to obtain “corrected” flows for Bertrand Creek only (Table 1). Observation during field work as well as analysis of the water right database suggest that minimal surface diversions are in use for Fishtrap Creek and, as a result, flow values were unchanged from the field measurements. Upon accounting for the surface water diversions in Bertrand Creek, virtually all locations were found to be gaining water from the aquifer (Figure 6).

Table 1. Estimated discharge for Bertrand Creek.

July 25–26, 2006 Calculated Discharge Data				
Site Description	Approximate River Mile	Measured Discharge (cfs)	Estimated Surface Water Withdrawals (cfs)	Corrected Discharge (cfs)
(B–1) Bertrand Creek on Carlsen Property	8.61	0.8	0.00	0.8
(B–2) Bertrand Creek on Steensma Property	6.74	0.8	0.61	1.4
(B–3) Bertrand Creek at Berthusen Park	5.10	2.5	0.39	3.5
(B–4) Bertrand Creek at Loomis Trail Rd.	4.06	4.3	0.00	5.3
(B–5) Bertrand Creek upstream of Mcklelland Creek	3.17	3.2	0.83	5.1
(B–D1) McClelland Creek	–	0.1	0.93	1.0
(B–6) Bertrand Creek south of West Branch	1.82	1.4	3.31	7.4
(B–7) Bertrand Creek at Rathbone Rd.	1.03	0.4	1.50	8.0
(B–8) Bertrand Creek South of Rathbone Rd.	0.62	0.7	0.13	8.4

Table 2. Estimated discharge for Fishtrap Creek.

July 20–21, 2006 Calculated Discharge Data		
Site Description	Approximate River Mile	Measured Discharge (cfs)
(F–1) Fishtrap Creek at Assink Rd.	7.45	4.3
(F–2) Fishtrap Creek at Badger Rd.	6.27	4.9
(F–3) Fishtrap Creek at REC Park on Bender Rd.	5.34	3.9
(F–D1) Bender Ditch	–	0.5
(F–4) Fishtrap Creek at Lynden Park on Depot Rd.	4.68	5.0
(F–D2) Depot Ditch	–	0.3
(F–5) Fishtrap Creek upstream of Double Ditch	3.32	6.4
(F–D3) Pepin Creek upstream of Fishtrap Creek	–	1.3
(F–6) Fishtrap Creek at Kok Rd.	2.72	9.8
(F–7) Fishtrap Creek at Flynn Rd.	1.75	9.3
(F–8) Fishtrap Creek at River Rd.	0.23	9.1

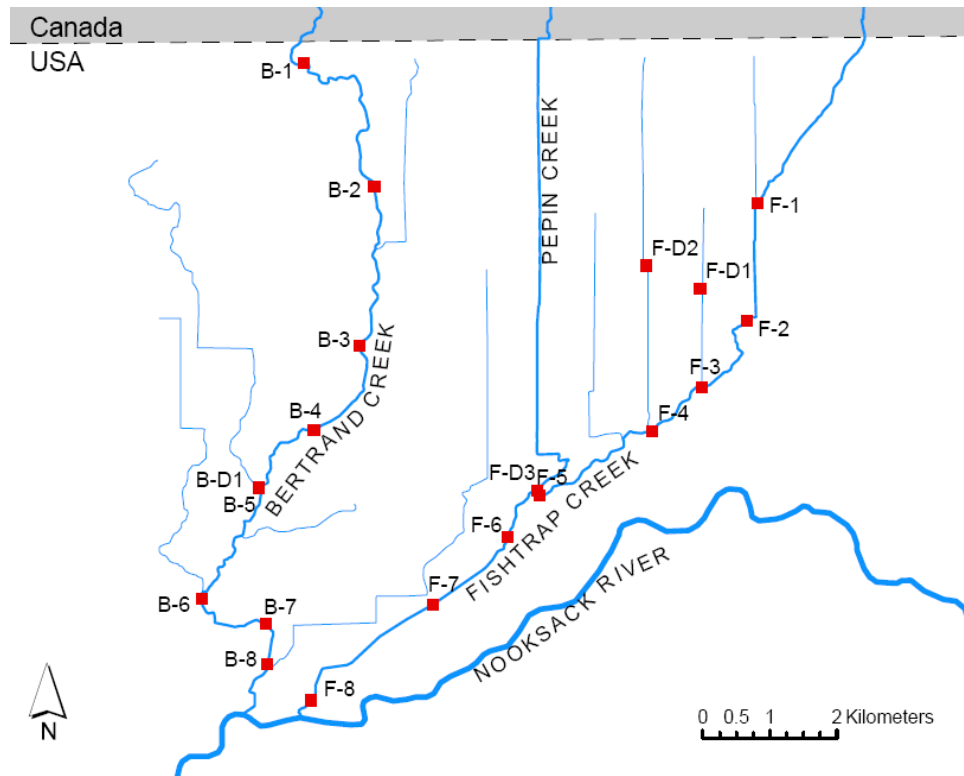


Figure 4. Location of flow measurements.

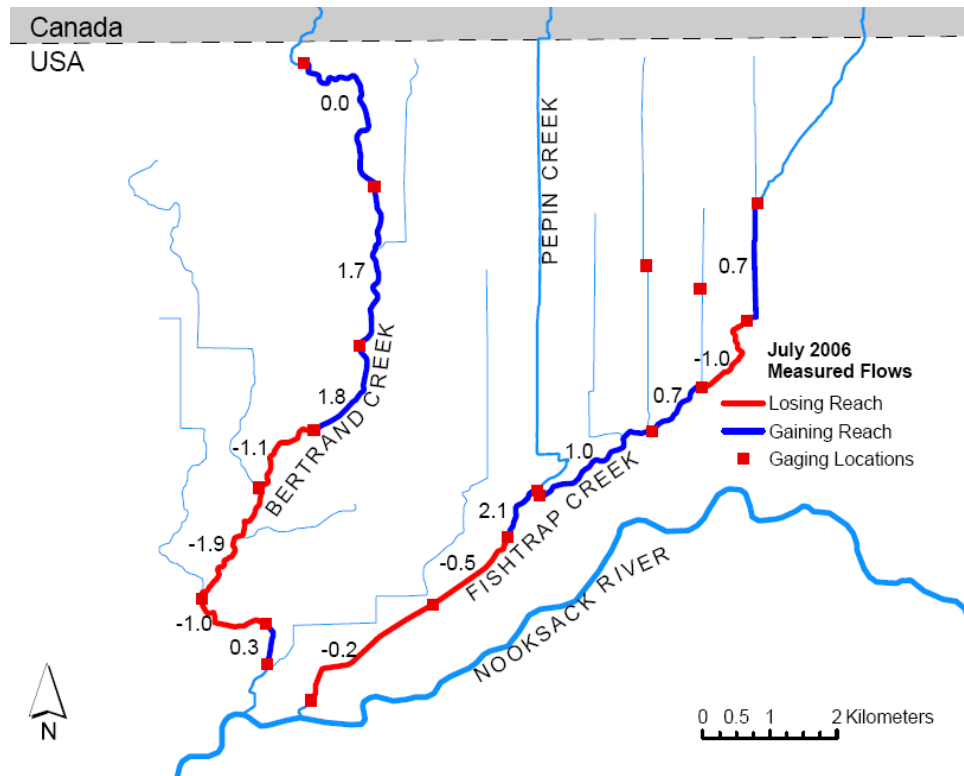


Figure 5. Gaining and losing reaches as measured in July 2006 labeled with the amount gained from or lost to the aquifer in cubic feet per second (cfs).

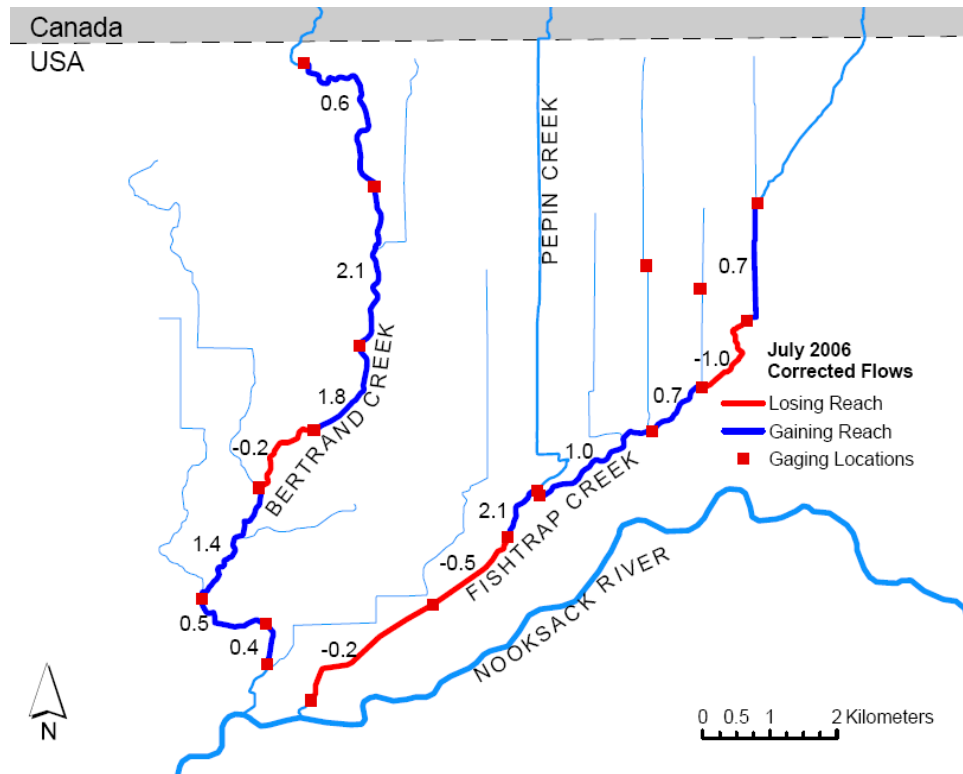


Figure 6. Corrected gaining and losing reaches for July 2006 labeled with the amount gained from or lost to the aquifer in cubic feet per second (cfs).

Instream slug tests were conducted in July 2006 to determine the hydraulic conductivity of the streambed sediments (Figure 7). Measurements were taken at 0.5 meter and 1.0 meter depths below the streambed. A full discussion of the instream slug tests and methodology can be found in McKenzie (2007). The conductivities derived from these tests (Table 3) were used to estimate the conductance values for the model boundary conditions that are used to represent the streams.

Static groundwater elevations were monitored once every hour by use of Onset Hobo Water Level Logger pressure transducers and were used to calibrate the groundwater flow model. Six wells were monitored (Figure 8), each of which was surveyed by Whatcom County to determine its elevation. All observation wells were not pumped for the duration of the monitoring (July 2006 to July 2007) with the exception of B-3, which was pumped during the summer of 2007. Plots of the observation well data are provided in Appendix A.

Surface water levels in each stream were monitored once every hour using Global Water pressure transducers and were used in conjunction with the monitored static groundwater elevations to determine lag times between monitored wells and stream. Two sites were chosen for installation of the pressure transducers; one in Bertrand Creek and the other in Fishtrap Creek (Figure 8). Plots of surface water level data are provided in Appendix A.

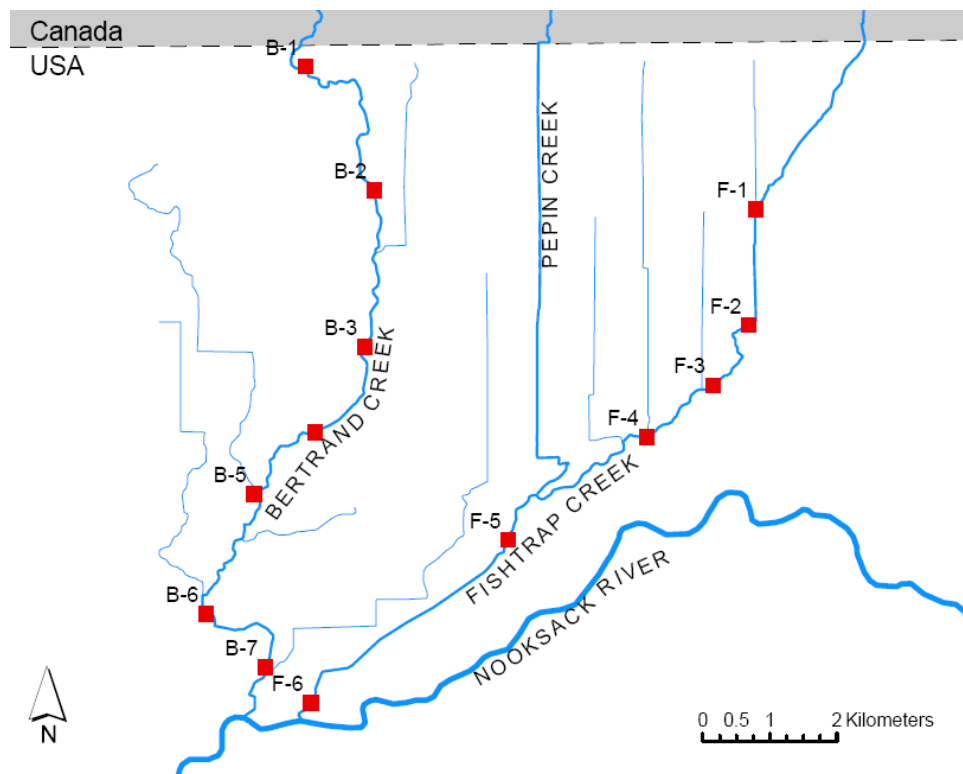


Figure 7. Location of instream slug tests.

Table 3. Streambed hydraulic conductivity values used in the groundwater model. Site names correspond to Figure 7.

Site Name	Hydraulic Conductivity, K (m/d)
B-1	7.50E-01
B-2	1.42E+02
B-3	6.08E+01
B-4	1.67E+01
B-5	5.80E+01
B-6	5.76E+01
B-7	7.88E+00
F-1	1.70E+01
F-2	6.30E+00
F-3	4.78E+00
F-4	3.03E+01
F-5	8.94E+01
F-6	1.16E+01

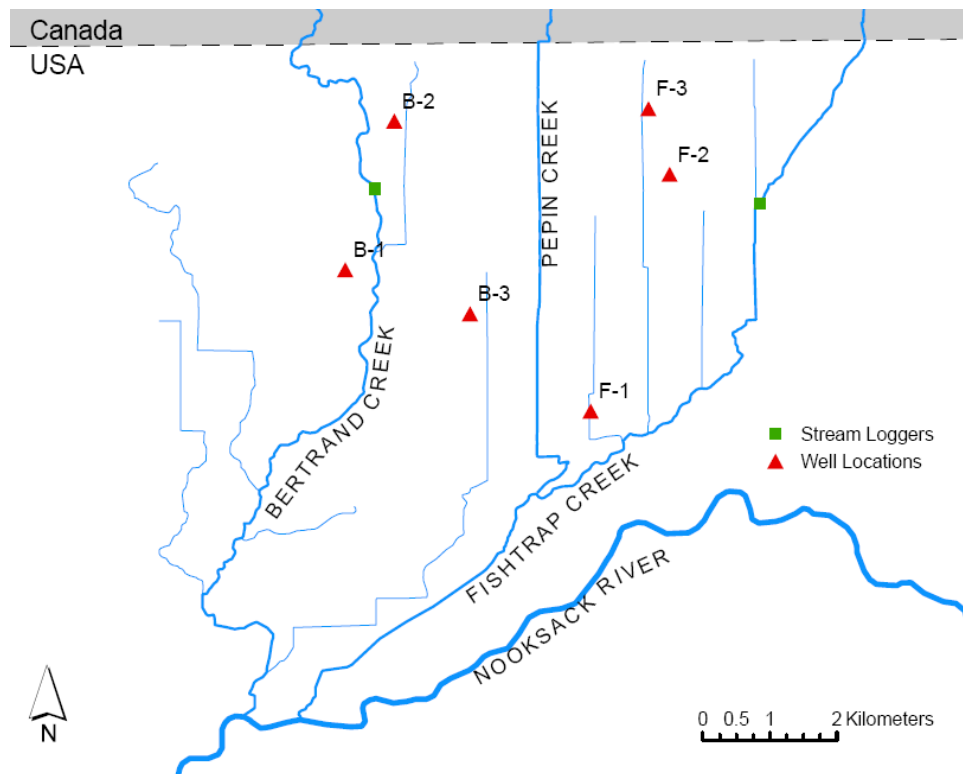


Figure 8. Location of groundwater and surface water monitoring sites.

4. Groundwater Flow Model

Field measurements of stream flows, recorded groundwater and surface water elevations, and stream bed hydraulic conductivities, as well as additional groundwater pumping well locations and rates of extraction were added to the SFU numerical groundwater model or compared to the Zone Budget results to ensure the best possible local calibration for the Bertrand Creek and Fishtrap Creek watersheds.

4.1. Model Boundary Conditions

4.1.1 River Boundary Condition

Surface waters may either contribute water to the groundwater system or extract water by acting as groundwater discharge zones (Figure 9) (Waterloo Hydrogeologic Inc., 2006). The original SFU model represented all flowing rivers, streams, or creeks as specified-head boundaries or drains. Specified-head boundaries allow for perfect hydraulic connections between the surface water and the underlying aquifer, meaning if the head in the aquifer is below that of the specified-head boundary, a limitless amount of water can be transferred into the groundwater system. These boundary conditions were used because of the coarse-grained nature of the aquifer materials and the lack of measured streambed conductivity. However, in order to better simulate water exchange between the streams and the aquifer and to make use of the available streambed conductivity data, the boundary conditions for Bertrand, Fishtrap, and Pepin Creeks were changed to River boundary conditions.

The MODFLOW River package simulates the interaction between groundwater and surface water via a seepage layer separating the surface water body from the groundwater system (Waterloo Hydrogeologic Inc., 2006). In addition to providing the surface water elevation, each cell modeled as a river allows for a conductance term. The conductance of a river cell represents

the ability of the seepage layer to transmit water between the surface water body and the groundwater (Waterloo Hydrogeologic Inc., 2006), and is defined as:

$$C = \frac{K \cdot L \cdot W}{B} \quad (1)$$

where C is the conductance of seepage layer (m^2/d), K is the hydraulic conductivity of stream bed sediments (m/d), L is the length of river reach through model cell (m), W is the width of river reach in model cell (m), and B is the thickness of stream bed sediments (m).

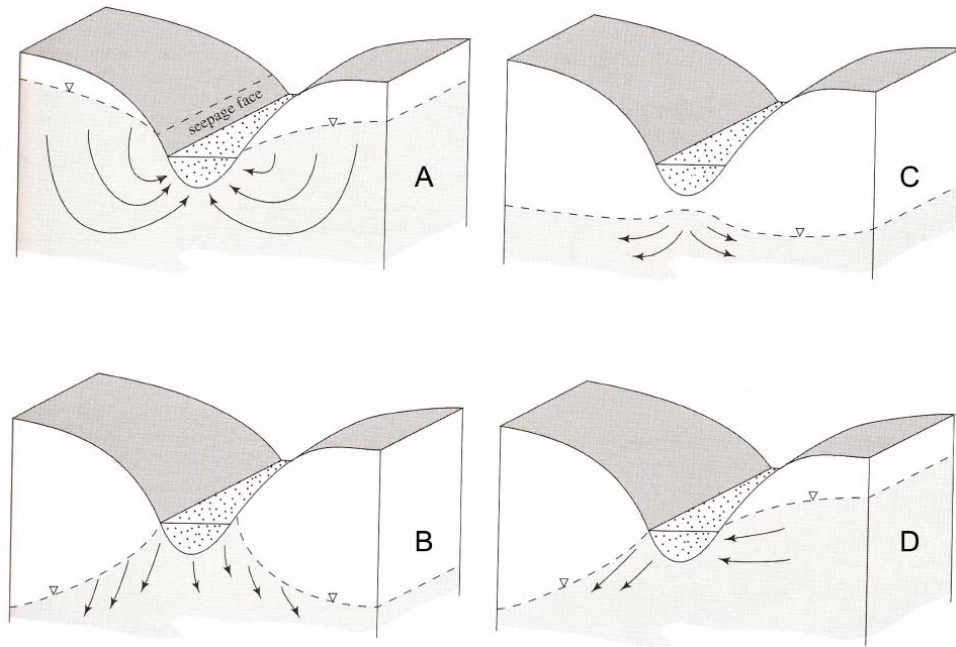


Figure 9. Four scenarios of stream-aquifer interaction (Dingman, 2002). (A) Gaining stream connected to the aquifer. (B) Losing stream connected to the aquifer. (C) Losing stream perched above the aquifer. (D). Gaining and Losing stream connected to the aquifer.

It is important to note that the stream-aquifer interaction is independent of the location of the stream reach within the MODFLOW cell, and that the level of water in the stream is uniform over the reach, and constant over each stress period (Rumbaugh *et al.*, 1996). A defined river reach does not take the place of a MODFLOW grid cell, in other words, the entire grid cell is not

given a head value equal to the defined stream elevation. Instead, the river reach is contained within the MODFLOW grid cell that has a top elevation greater than and bottom elevation less than the defined bottom elevation of the river's seepage layer. Figures 10 and 11 demonstrate gaining and losing river reaches in a MODFLOW grid cell. At the beginning of each computational iteration, terms representing river seepage are added to the flow equation for each MODFLOW grid cell containing a river reach (Rumbaugh *et al.*, 1996). Depending on the elevation of bottom elevation of the seepage layer, either equation (2) or equation (3) is used to determine the amount of water seepage to or from the river reach (Rumbaugh *et al.*, 1996):

$$\begin{aligned} QRIV &= CRIV(HRIV - h), h > RBOT \\ QRIV &= CRIV(HRIV - RBOT), h < RBOT \end{aligned} \quad (2) \text{ and } (3)$$

where $QRIV$ is the flow between stream and aquifer (m^3/d), $CRIV$ is the hydraulic conductance of seepage layer (m^2/d), $HRIV$ is the head in the stream (m), h is the head in the MODFLOW grid cell (m), and $RBOT$ is the bottom elevation of the seepage layer (m).

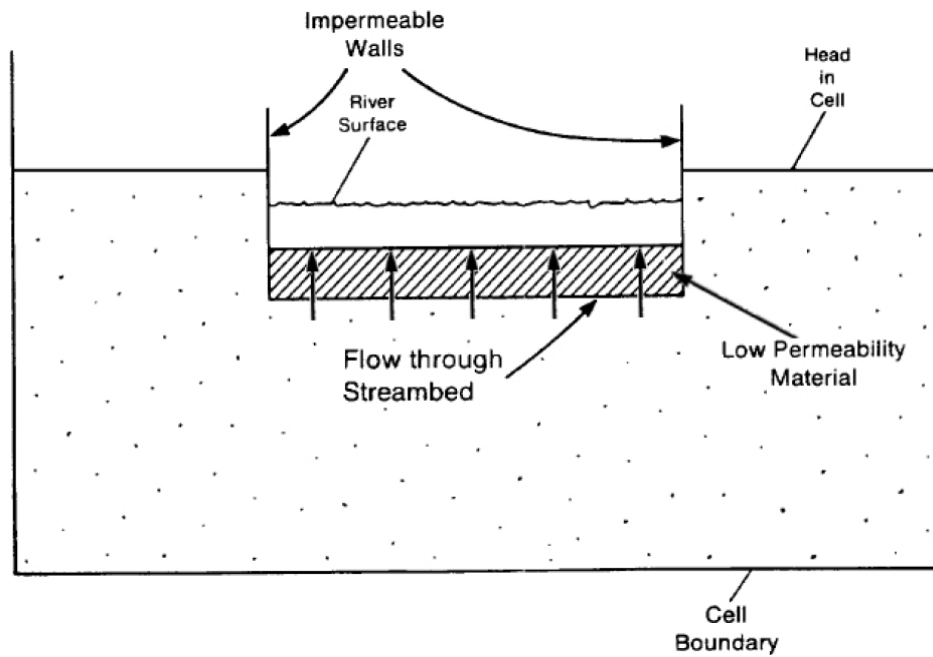


Figure 10. Modified representation of a gaining river reach within a MODFLOW cell (Rumbaugh *et al.*, 1996).

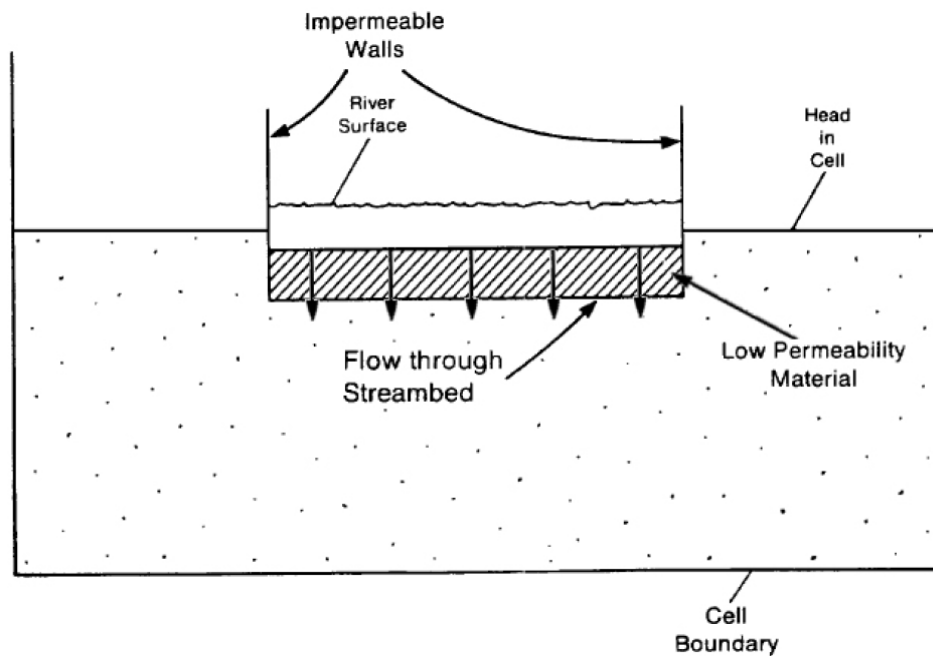


Figure 11. Modified representation of a losing river reach within a MODFLOW cell (Rumbaugh *et al.*, 1996).

Each creek was divided into sections: seven in Bertrand Creek, six in Fishtrap Creek, and one in Pepin Creek (Figure 12). Each section was assigned a conductance value based on the nearest measurement of streambed hydraulic conductivity (Table 4). All river cells north of the first site in both Bertrand and Fishtrap Creeks were given the same conductance value as the first river section in each creek.

The following assumptions were made in the calculation of each conductance term. It was not feasible to determine the exact river length through each model grid cell. Therefore, the length of the river through a cell was approximated as the average of the cell height and width for that section. If a cell was defined with a width of 100 m and a height of 50 m, then the approximated river length through that cell would have been 75 m. The width of the river through a cell was assumed to be the same width as measured during the flow measurement nearest each site for the instream slug test. The thickness of the stream sediments was assumed to be 1.0 m, the maximum depth of the instream slug tests. A sediment thickness of 0.75 m was assigned if the calculated hydraulic conductivity of the 0.5 m slug test was lower than that of the 1.0 m slug test.

In addition to surveying the well elevations, Whatcom County created benchmarks on two bridges in Bertrand Creek and two in Fishtrap Creek. These benchmarks allowed for manual recording of the surface water elevation throughout the year. To assure that the modeled surface water elevations for Bertrand and Fishtrap Creeks were representative of August low-flow conditions, the specified heads originally assigned in the model were compared with the lowest recorded surveyed surface water elevations. It was found that the original modeled river heads were too low. The heads were subsequently increased accordingly to match the surveyed values (Table 5).

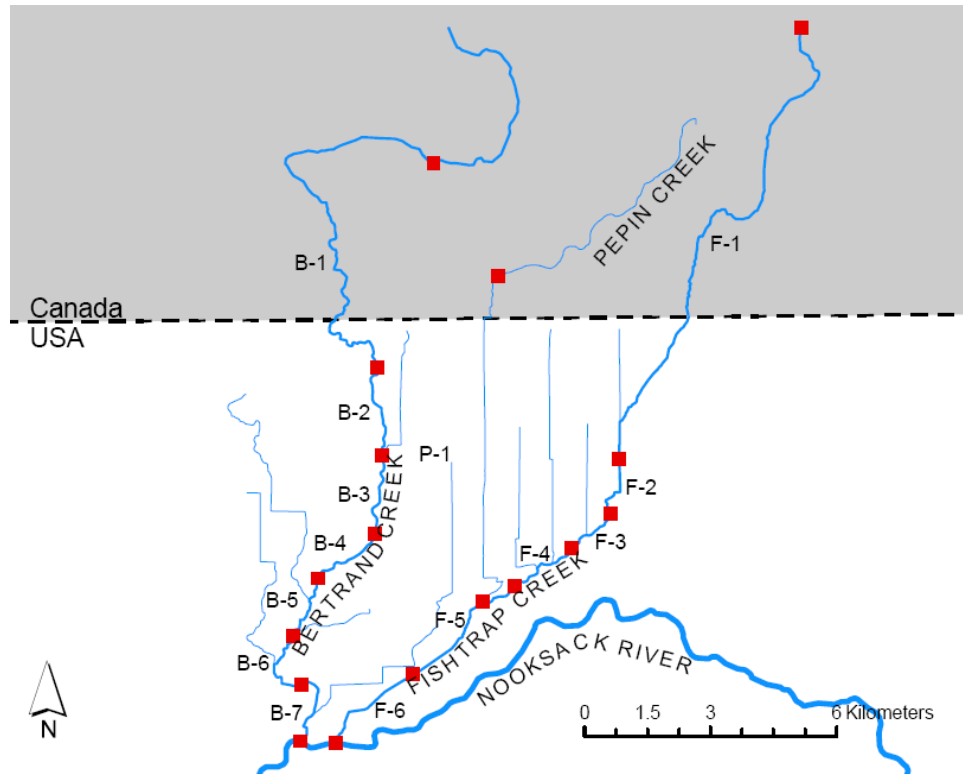


Figure 12. Start and end locations of river sections with unique conductance values.

Table 4. Conductance values for each river section.

Site Name	Hydraulic Conductivity, K (m/d)	Stream Width (m)	Stream Length (m)	Sediment Thickness (m)	Conductance (m ² /d)
B-1	7.50E-01	4.5	75	0.75	3.38E+02
B-2	1.42E+02	3.3	100	1.00	4.70E+04
B-3	6.08E+01	3.7	100	1.00	2.25E+04
B-4	1.67E+01	3.2	100	1.00	5.35E+03
B-5	5.80E+01	3.0	100	0.75	2.32E+04
B-6	5.76E+01	3.0	100	1.00	1.73E+04
B-7	7.88E+00	4.3	100	0.75	4.52E+03
F-1	1.70E+01	5.5	50	1.00	4.66E+03
F-2	6.30E+00	4.3	75	0.75	2.71E+03
F-3	4.78E+00	4.5	75	1.00	1.61E+03
F-4	3.03E+01	6.7	100	1.00	2.03E+04
F-5	8.94E+01	4.8	100	1.00	4.29E+04
F-6	1.16E+01	4.0	100	1.00	4.64E+03
P-1	8.94E+01	2.0	100	1.00	1.79E+04

Table 5. River head changes for each river section.

River Section	Head Change (m)
B-1	2.17
B-2	2.09
B-3	2.01
B-4	1.93
B-5	1.85
B-6	1.77
B-7	1.69
F-1	1.38
F-2	1.32
F-3	1.25
F-4	1.18
F-5	1.12
F-6	1.05
P-1	-0.36

4.1.2. Specified-Head Boundary

The Nooksack River was modeled as a specified-head boundary condition. This boundary condition was used to fix the head value for a given model cell. Because the condition does not include a conductance term, an infinite amount of water can be allowed to enter or leave the system (Waterloo Hydrogeologic Inc., 2006).

Upon raising the heads within Bertrand and Fishtrap Creeks, the Nooksack River head values were found to be too low. Through examination of two USGS gaging stations on the Nooksack River, one upstream at North Cedarville, WA and one downstream at Ferndale, WA, the specified-head cells representing the Nooksack River within the model were raised by 2.5 m. With this increase, the heads at the terminus of Bertrand and Fishtrap Creeks match up better with the Nooksack River constant heads.

4.1.3. Drains

Small creeks and ditches were modeled as drain boundary conditions in the original SFU model. Drain cells are given only two values: bed elevation and conductance. Drains do not

affect the flow model unless the groundwater table rises above a defined drain elevation. During the low-flow period for which the steady-state model is based, most drains are not in contact with the groundwater table because their bed elevations are above the groundwater table under August conditions. These drains function under transient conditions. Drains surrounding Bertrand and Fishtrap Creeks were modified from their original conductance values of $100 \text{ m}^2/\text{d}$ and given values similar to those found in the nearest Bertrand or Fishtrap Creek river section.

4.1.4. Pumping Wells

The original SFU model only included selected pumping wells from the Washington State Department of Ecology's well log database. When combined, the wells within the Bertrand Watershed Improvement District (WID) totaled a pumping rate of 138 liter/s (4.88 cfs). According to Wubbena *et al.* (2004), 2995 hectares (7,400 acres) within the Bertrand WID require approximately 1379 l/s (48.71 cfs) of groundwater during the month of July for irrigation purposes. In order to properly simulate the local conditions, all groundwater rights, certificates, and claims were added to the model.

A water right database developed by the Public Utilities District 1 Water Rights Team for the Water Resource Inventory Area (WRIA) 1 Watershed Management Project was used to import groundwater rights, certificates, and claims into the groundwater model. Because the amount of pumping does not generally match the written water right at all times of the year, all water rights were scaled month-to-month to match the estimated groundwater irrigation use for the Bertrand Watershed Improvement District as determined by Hydrologic Services Company (Table 6) (Wubbena *et al.*, 2004). The water rights, certificates, and claims within the Bertrand Watershed Improvement District were scaled to match those groundwater irrigation use rates for each month of the year. Those monthly factors (Table 7) were then applied to the remaining water rights,

certificates, and claims within the Fishtrap watershed. Pumping rates for the steady-state model were taken to be those of July, the month with the greatest pumping rates.

Table 6. Estimated instantaneous crop water use in WID (Wubben *et al.*, 2004).

Month	Surface Water Irrigation Use (cfs)			Ground Water Irrigation Use (cfs)			Total Irrigation Use (cfs)		
	Sprinkler	Drip	Total	Sprinkler	Drip	Total	Sprinkler	Drip	Total
Jan	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Feb	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Mar	0.00	0.00	0.00	0.10	0.00	0.10	0.10	0.00	0.10
Apr	0.07	0.00	0.07	4.67	0.00	4.67	4.74	0.00	4.74
May	0.51	2.46	2.97	16.54	2.55	19.09	17.05	5.01	22.06
Jun	0.83	4.36	5.19	23.65	4.52	28.17	24.48	8.88	33.36
Jul	1.60	6.10	7.70	42.38	6.33	48.71	43.98	12.43	56.41
Aug	1.28	3.84	5.12	34.50	3.99	38.49	35.78	7.83	43.61
Sep	0.34	0.52	0.86	11.66	0.54	12.20	12.00	1.06	13.06
Oct	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nov	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Dec	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table 7. Monthly factors applied to all water-right wells.

Month	Scaling Factor
Jan	0.000
Feb	0.000
Mar	0.001
Apr	0.053
May	0.217
Jun	0.321
Jul	0.554
Aug	0.438
Sep	0.139
Oct	0.000
Nov	0.000
Dec	0.000

4.1.5. Recharge

Recharge rates were defined in the SFU model based on spatial distributions of a number of factors: type of soil cover, aquifer material, and water-table depth. Average mean annual climate

data were used in running the HELP (Hydrologic Evaluation of Landfill Performance, US EPA) model for one-dimensional soil columns. The resulting recharge results were mapped spatially and used as input to the groundwater flow model. A full discussion of the recharge mapping and methodology can be found in Scibek and Allen (2006).

To make sure that average annual recharge values would be acceptable when using 2006 field data to adapt the model, annual precipitation for 2006 at Clearbrook, WA was compared to the normal precipitation observed since the year 1919. For 2006 a total of 113.9 cm (44.86 inches) of rain was recorded, which amounted to only 2.3 cm (0.91 inches) less than normal (NCDC, 2007a). This departure from normal was not significant enough to warrant a reduction of the recharge values previously defined by Scibek and Allen.

4.1.6. Observation Wells

In addition to the more than 1000 existing observation wells input in the model by SFU, the six observation wells described previously in section 3, along with a number of USGS wells (Appendix B), were added to the groundwater model to ensure that the model correctly predicted the water-table elevations.

4.1.7. Zone Budget

Within MODFLOW, Zone Budget was used to calculate sub-regional water budgets for specific sections of Bertrand and Fishtrap Creeks, along with Pepin Creek and other major drains. For each sub-regional water budget (zone), the cell by cell budget results produced by MODFLOW are tabulated by Zone Budget (Waterloo Hydrogeologic Inc., 2006). A total of 20 zones were created between locations of measured stream flow; eight in Bertrand Creek, eight in Fishtrap Creek, one in Pepin Creek, and three for major drains (Figure 13). Only cells that were

defined as river or drain boundaries were included in a zone. Beginning with known flows from Environment Canada gaging stations at the USA-Canada border for Bertrand, Fishtrap, and Pepin Creeks, the predicted gains and losses from each river reach or zone were added to or subtracted from the known flow and compared to our measured flow values to determine the accuracy or validity of the model. The stream routing package within Visual MODFLOW would have accounted for the flows automatically, but the river package was chosen in order to preserve the original surface water head values developed by SFU.

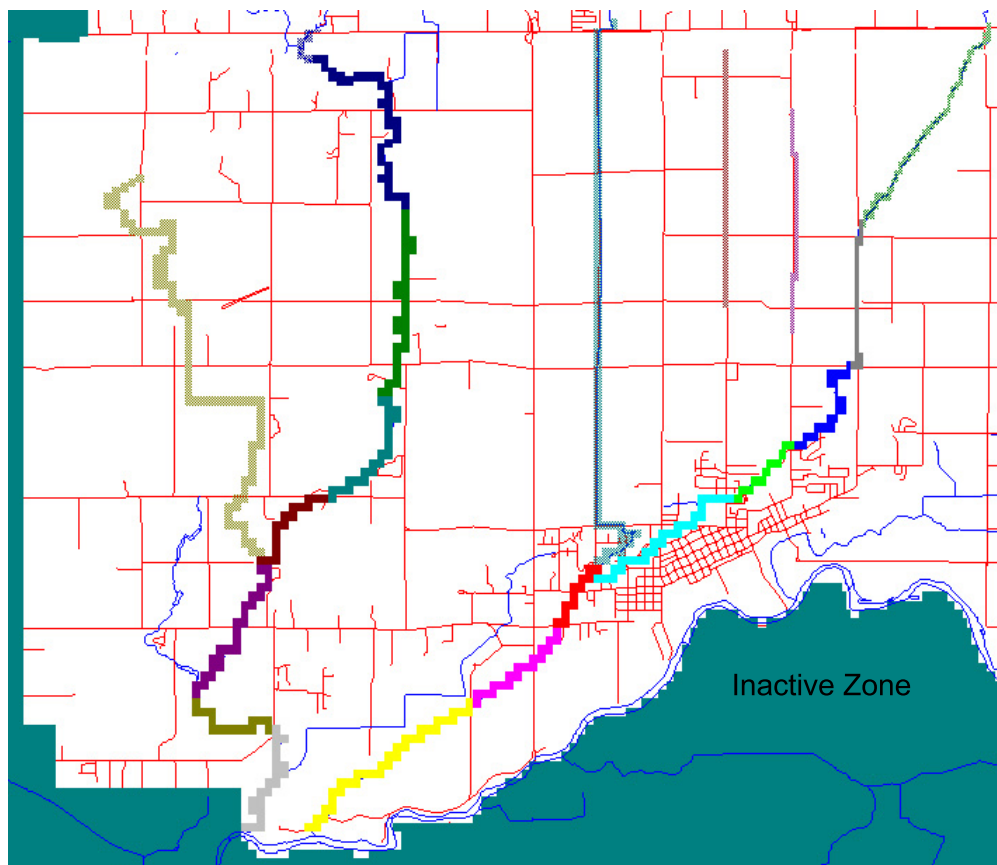


Figure 13. Location of color-coded zone budget zones. 20 in total.

4.2. Local Model Calibration

Throughout the entire model domain, the calibration of observed to measured static water levels (Figure 14) yielded a Root Mean Squared (RMS) error of 10.0 m, with a normalized RMS error of 8.7% and a residual mean error of 3.5 m. The calibration statistics were found to be similar to those of the original model developed by Scibek and Allen (2005), which were regarded to be reasonable given the scale of the model and the number of observations (Scibek and Allen, 2005).

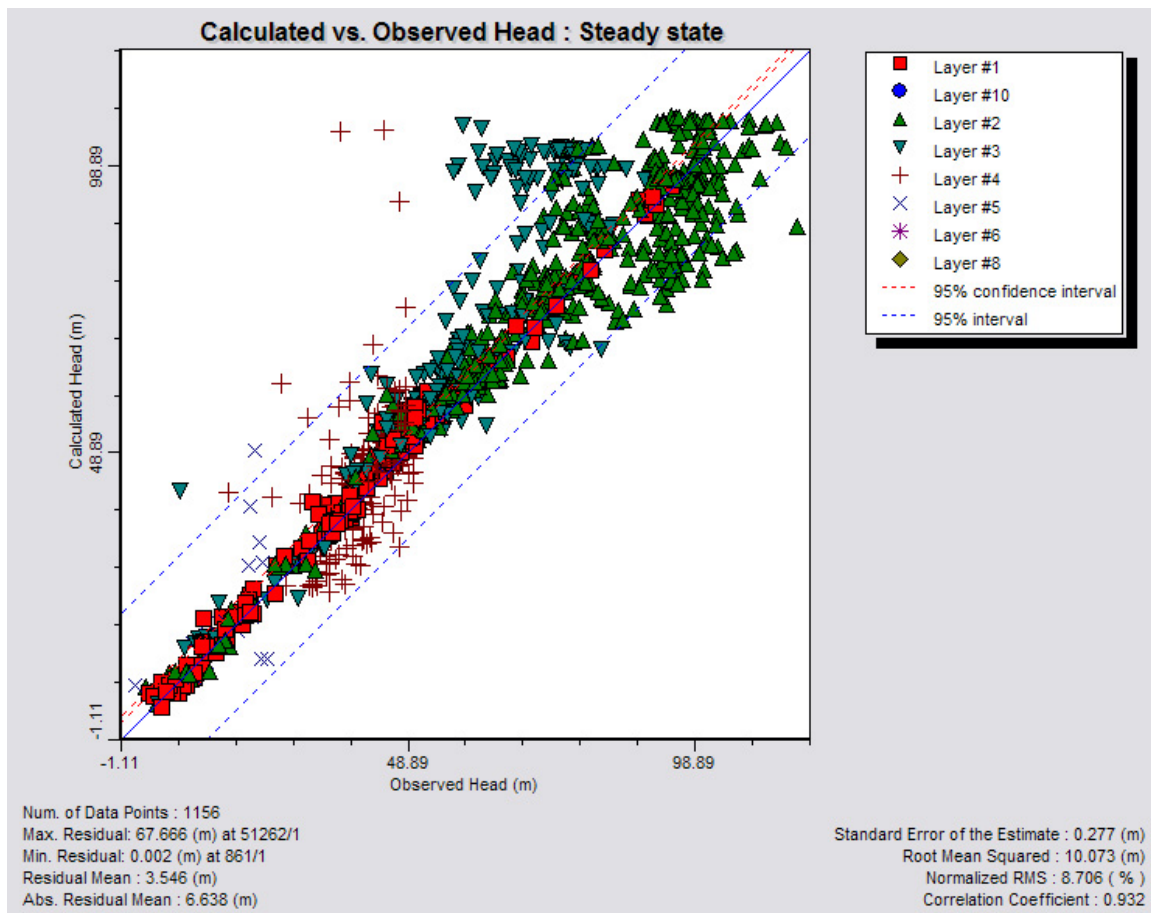


Figure 14. Visual MODFLOW 4.2 output of measured to observed static water levels and statistics for the entire model domain.

Locally the calibration results point to some discrepancies between “corrected” and modeled stream flows. A comparison of the “corrected” creek flows and modeled flows for Bertrand Creek (Figure 15) and measured flows and modeled flows for Fishtrap Creek (Figure 16) revealed that the model over-predicted the stream flow in the area of station B-2, and slightly over-predicted stream flow in the upper reaches of Fishtrap Creek; however, the model closely matched the “corrected” flows in the lower reaches of Bertrand Creek and the measured flows of Fishtrap Creek. The overestimation of river leakage in the upper reaches of both creeks may be due to non-permitted wells which are unaccounted for, uncertainty in stream elevations, or the hydraulic conductivity of the aquifer materials may be too low within those areas. A comparison of the observed heads and the modeled heads within the study area yielded better statistics than the overall model, with a RMS error of 3.1 m, a normalized RMS error of 5.4% and a residual mean error of 1.8 m.

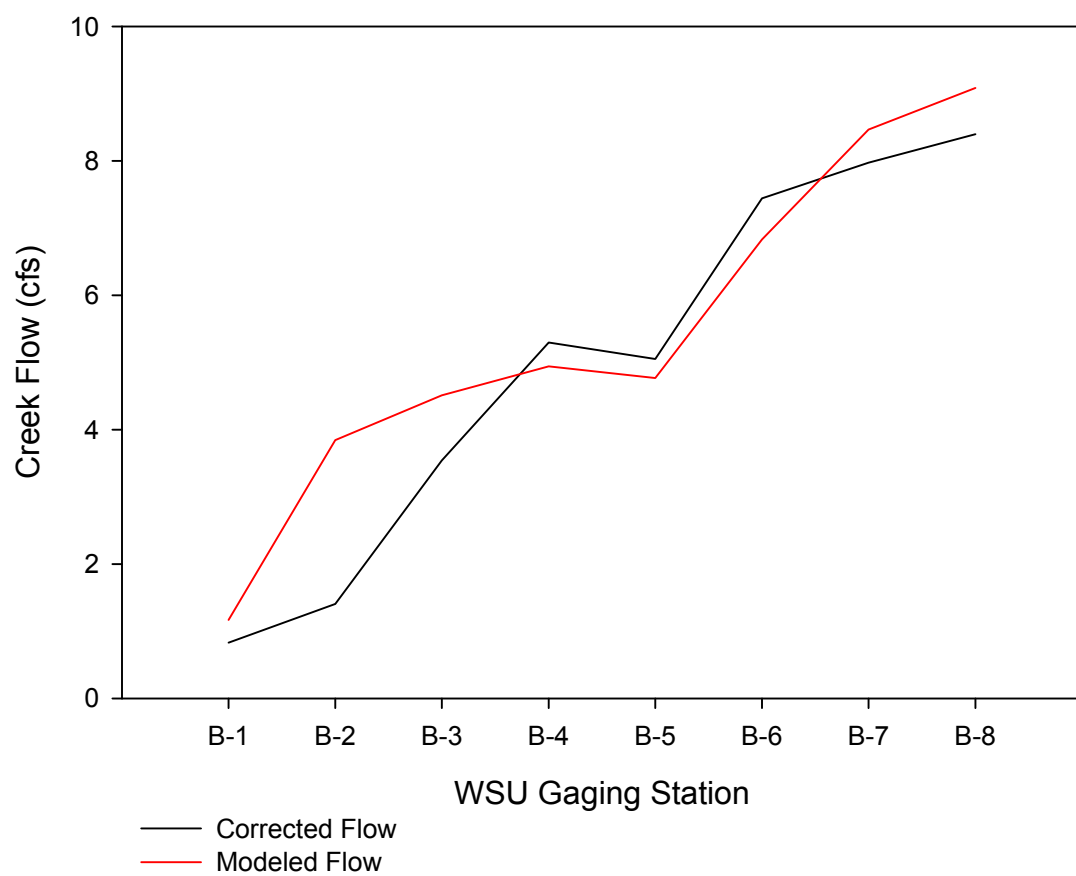


Figure 15. A comparison of corrected and modeled stream flows for Bertrand Creek.

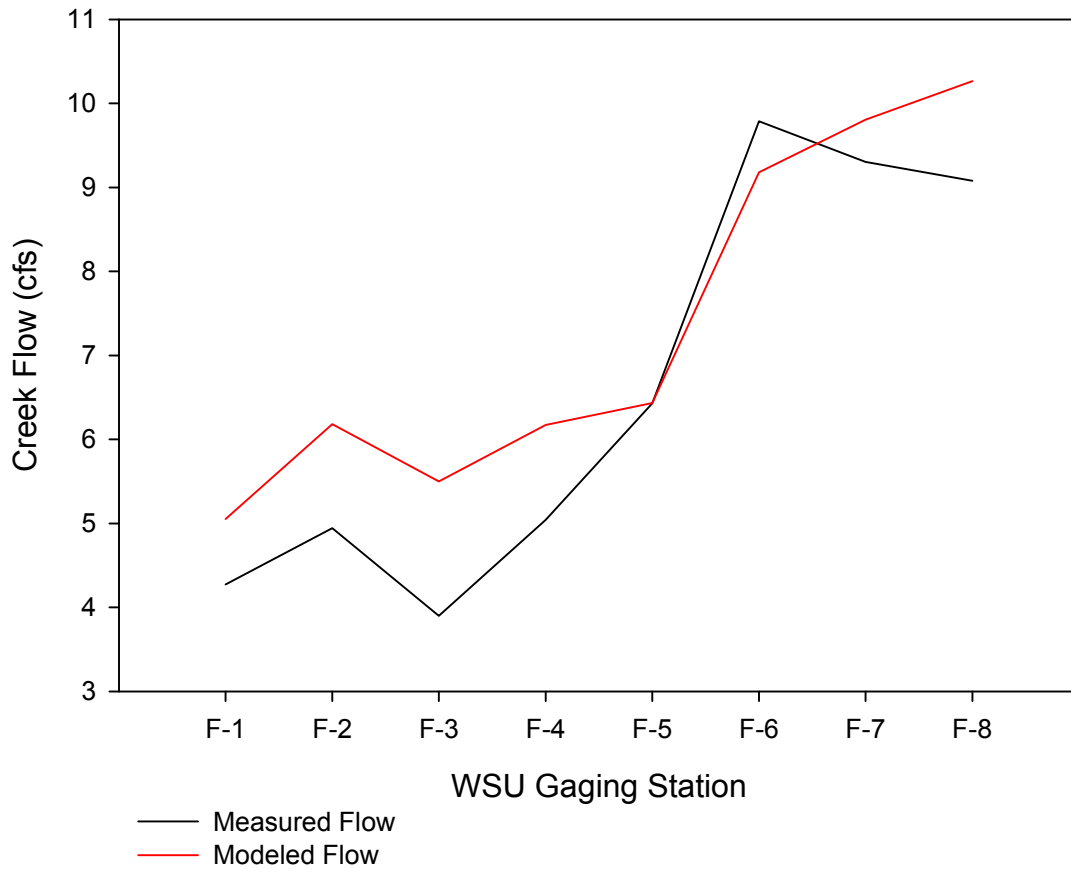


Figure 16. A comparison of measured and modeled stream flows for Fishtrap Creek.

4.2. Procedure for Determining Impact on Streams Due to Groundwater Pumping

Barlow *et al.* (2003) and Cosgrove and Johnson (2004) created groundwater models using MODFLOW to study the local groundwater and surface water interactions for the Hunt-Annaquatucket-Pettaquamscutt stream-aquifer, Rhode Island, and the Snake River Plain, Idaho, respectively. The goal in both studies was to create stream response functions that would allow them to quantify the impacts of groundwater pumping on surface water flows.

Using a response-matrix technique, Barlow *et al.* (2003) coupled a numerical groundwater model and optimization techniques to maximize total groundwater withdrawal from the Hunt-

Annaquatucket-Pettaquamscutt stream-aquifer of central Rhode Island, during the summer months of July, August and September, while maintaining desired stream flows. Response functions were generated for 14 public water supply wells and 2 hypothetical wells. Barlow *et al.* (2003) assumed the rate of stream flow depletion at a constraint site to be a linear function of the pumping rate of each groundwater well. By assuming linearity, the concept of superposition allowed for individual stream flow depletions caused by each well to be summed together at a constraint site to derive a total stream flow depletion.

Cosgrove and Johnson (2004) modified an existing single layer, unconfined, transient MODFLOW groundwater model for the Snake River Plain Aquifer, Idaho, for use with the MODRSP code to generate response functions. The unconfined groundwater model was converted to a confined system, to conform to the MODRSP requirement of modeling a linear system. Transient response functions for 51 river cells were generated for each of the numerical model cells using 150 four-month stress periods representing 50 years. The response functions are a result of a unit stress applied only during the first stress period and, as a result, they represent the propagation of the effects of that unit stress over time. Making use of the transient response functions, a spreadsheet was developed that allows the user to enter water use scenarios and determine the impact to surface water resources.

For this study, response functions were manually created for each of 346 hypothetical well locations (Figure 17). Pumping wells were added to the calibrated steady-state groundwater flow model, one at a time, and the stream flow impacts were recorded for each through the use of Zone Budget. Each pumping well was given a screen interval of 9–13 m below the ground surface, and because response functions are typically based on a unit stress, the wells were assigned a pumping rate of 1 cubic foot per second (cfs) (Cosgrove and Johnson, 2004). For each

well location, a response ratio, ranging from 0.0 to 1.0, was determined for Bertrand Creek and Fishtrap Creek as the change in modeled stream flow at each creek's terminus with the Nooksack River divided by the pumping rate. As in Barlow *et al.* (2003) and Cosgrove and Johnson (2004) it was assumed that the rate of stream flow depletion at each constraint site is a linear function of the pumping rate of each groundwater well. Due to the unconfined nature of the groundwater model, the decline in water level was assumed to be very small such that linearity could be approximated. Response ratios less than 1.0 indicate that groundwater pumping wells would have less of an impact on stream flow than taking the same amount of water directly from a surface water diversion.

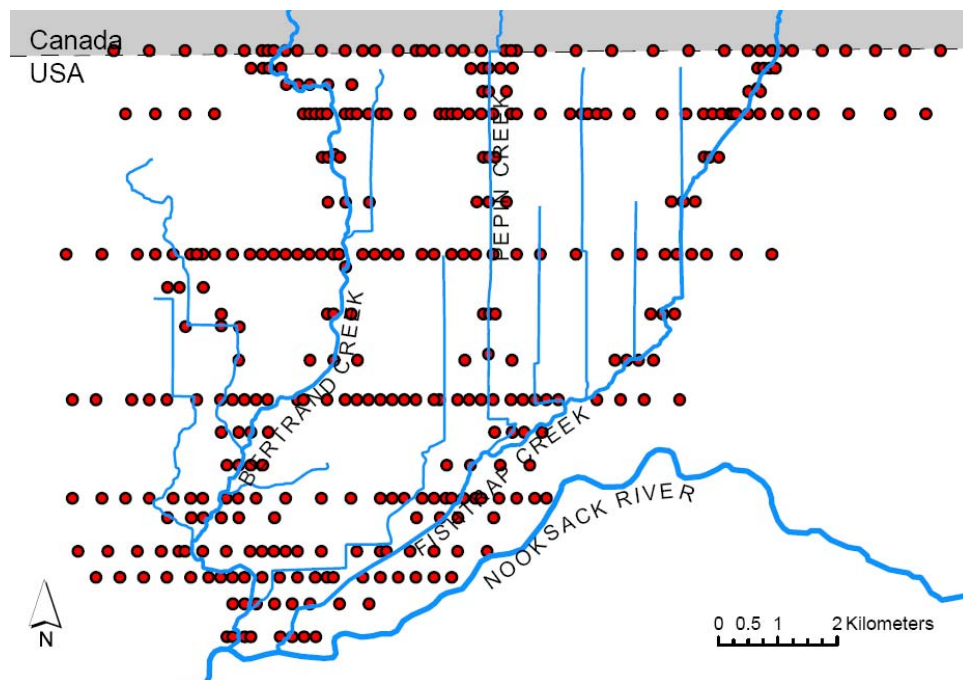


Figure 17. Modeled well locations for determination of response ratios.

Raster maps of the response ratios with a 100 m cell size were created for each stream using natural neighbor interpolation in ArcGIS. Natural neighbor interpolation uses a subset of data

points that surround a query point and applies weights to them based on proportionate areas in order to interpolate a value (Sibson, 1981).

The mapped response zones were used to create a groundwater and surface water interaction tool, whereby the user can replace a surface water diversion with a groundwater pumping well of the same withdrawal rate, and determine the stream flow impact to Bertrand and Fishtrap Creeks at their terminus with the Nooksack River. STELLA version 9.0.3 by isee Systems was chosen as the modeling environment for the interaction tool. The users can choose between 4 regions of interest within the study area, and then select one of four sub-regions within that chosen region. Upon choosing a sub-region, the user can easily locate the location of a surface water diversion, and determine the best location for a replacement groundwater well. Overlain on the sub-region maps are the mapped response function zones for Bertrand and Fishtrap Creeks. The user then identifies the Bertrand Creek response zone and the Fishtrap Creek response zone for which the desired replacement groundwater well falls, and enters a value for the surface water withdrawal rate to be replaced by the groundwater well. Using the response functions and the provided user input, the STELLA model compares the stream flow values for Bertrand and Fishtrap Creeks for each of a surface water replacement well and a surface water diversion. The impact to each creek is determined as the difference between those two sets of flow values. The user may, through trial-and-error, select the best option, but must personally keep track of the stream flow impacts for each case. The STELLA model is not capable of storing the locations of surface water diversions or the replacement groundwater pumping wells; it is only capable of determining the stream flow impacts for a single surface water replacement well. Screen captures of the STELLA model are shown in Appendix D.

A limitation of the approach is that a groundwater flow model was used, rather than a coupled surface water-groundwater model. As such, the boundary conditions used to represent the streams are fixed (specified heads). Thus, the impact on stream discharge cannot be simulated, i.e., while the stream discharge can vary in the simulations, this change in discharge is not reflected as a shift in stream stage. However, because we are replacing a surface water diversion for a single groundwater pumping well of the same withdrawal rate, the impact to stream stage is considered to be minimal.

5. Results

5.1 Groundwater Level and Climate

The recorded groundwater elevations for Bertrand observation well # 1 appear to correlate with local precipitation events (Figure 18). Hourly precipitation data were acquired from the Public Agricultural Weather System (PAWS) weather station in Lynden, Washington, and Bellingham International Airport weather data were used to fill in missing data from the PAWS weather station. During the summer months, the slope of the cumulative precipitation curve is small, and during the winter months the slope is large and appears to account for groundwater elevation changes. Also, there appears to be little lag time between Bertrand observation well # 1 and the nearby upstream Bertrand Creek stream logger (Figure 19). These observations suggest that the groundwater and surface water are well connected. Additional observation well plots can be found in Appendix A.

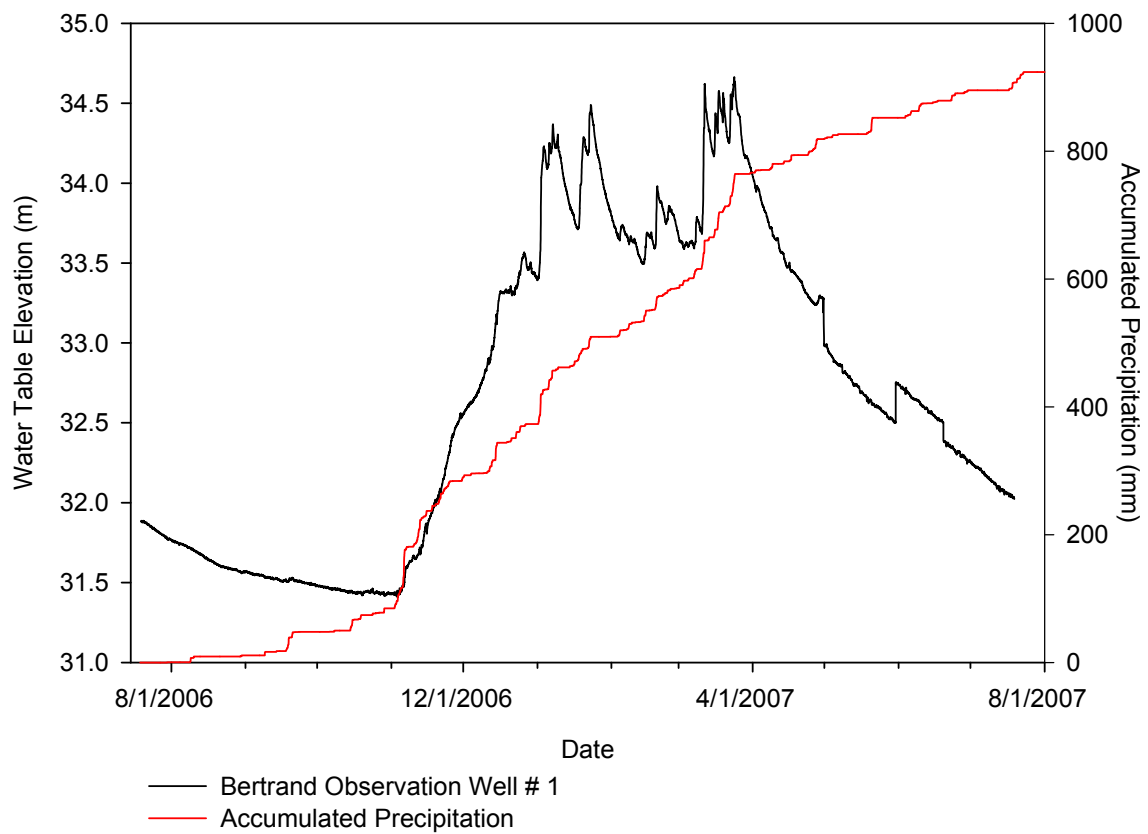


Figure 18. Water-table elevation for Bertrand observation well # 1 and local cumulative precipitation (WAWN, 2007 and NCDC, 2007b).

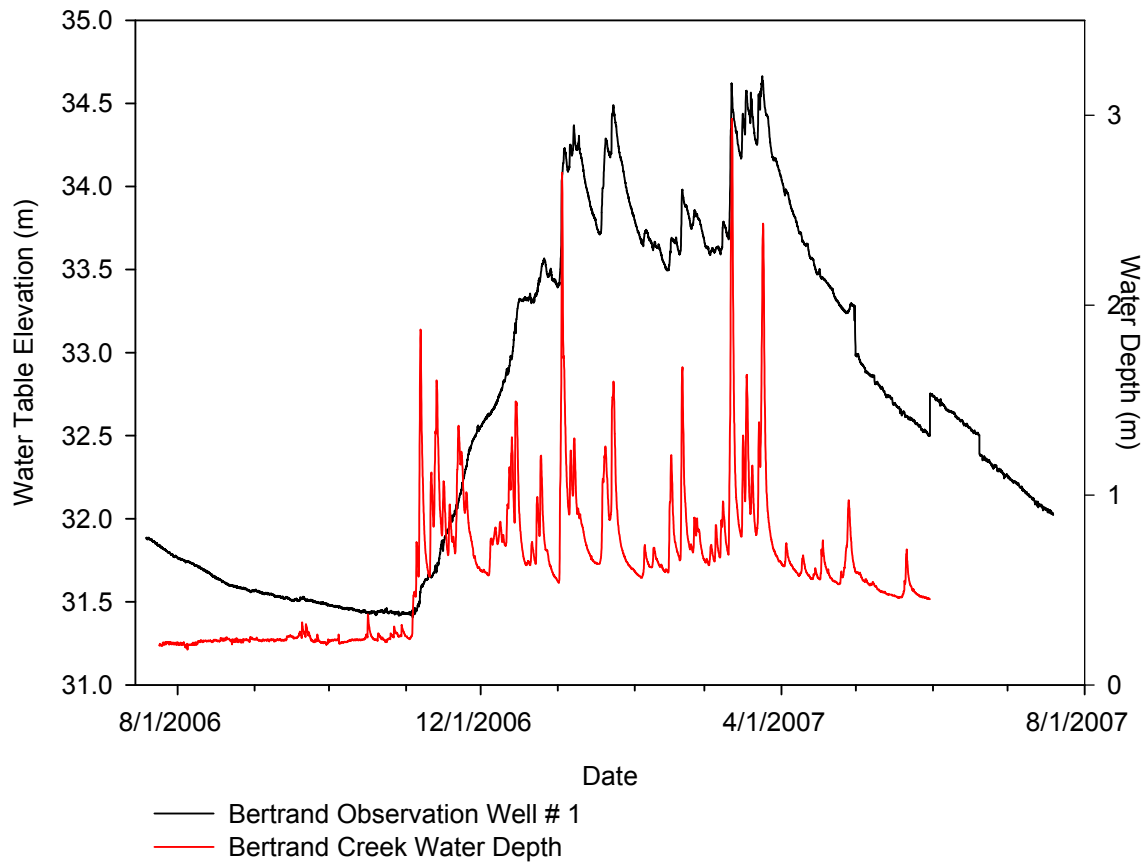


Figure 19. Water-table elevation for Bertrand observation well # 1 and Bertrand Creek water depth.

5.2 Response Functions

Maps of the well response functions for each of Bertrand and Fishtap Creek are shown in Figures 20 and 21. Because wells were placed by rows, the areas between the modeled rows are lacking data and, as a result, the interpolation does not accurately predict stream flow impacts due to pumping wells within those areas. In future work, more wells should be modeled along each creek in order to fill gaps of missing response ratio data.

Pumping wells placed east of Fishtrap Creek have almost no discernable impact on Bertrand Creek. Similarly, pumping wells placed west of Bertrand Creek have almost no discernable impact on Fishtrap Creek. However, groundwater pumping wells located in the area between Bertrand and Fishtrap Creeks can impact stream flows in both creeks. There are reaches where groundwater pumping has less of an impact on the stream flow on one side of the creek as opposed to the other. To better illustrate the overall response functions, a combined response ratio interpolation map was created by adding the Bertrand Creek and Fishtrap Creek responses for each well location (Figure 22).

Variations in the spatial distribution of response ratios appear to be correlated with spatial variations in the hydraulic conductivity within the model layer containing the screened interval of the pumping well (Figure 23). Surface water replacement wells placed within zones with high hydraulic conductivity values seem to produce greater responses to the instream flows of Bertrand and Fishtrap Creeks.

Table 8 presents the Bertrand and Fishtrap Creek flow responses, respectively, as the percent area of their corresponding watershed. 79% of the Bertrand Creek watershed and 79% of the Fishtrap Creek watershed have response ratios less than 0.5. However, because groundwater movement occurs across watershed boundaries, stream flow impacts to Fishtrap Creek can occur from wells within the Bertrand Creek watershed. A surface water replacement well should not be allowed to benefit one creek while harming the other. Table 9 presents the combined flow response for Bertrand and Fishtrap Creeks as the percent area of their corresponding watershed. 67% of the Bertrand Creek watershed and 79% of the Fishtrap Creek watershed have combined response ratios less than 0.5, indicating very favorable exchange opportunities. However, it might not be economically practical for farmers to replace their surface water diversion for a

groundwater withdrawal if they have to construct a lengthy pipeline in order to get the water to their field. Consequently, stream flow responses from wells located within narrow bands of both streams were specifically examined. Of the area within a 0.5 mile band of Bertrand Creek, 57% has a combined flow response ratio less than 0.5, and within a 1.0 mile band, 64% has a combined flow response ratio less than 0.5 (Table 10). Of the area within a 0.5 mile band of Fishtrap Creek, 70% has a combined flow response ratio less than 0.5, and within a 1.0 mile band, 77% has a combined flow response ratio less than 0.5 (Table 11).

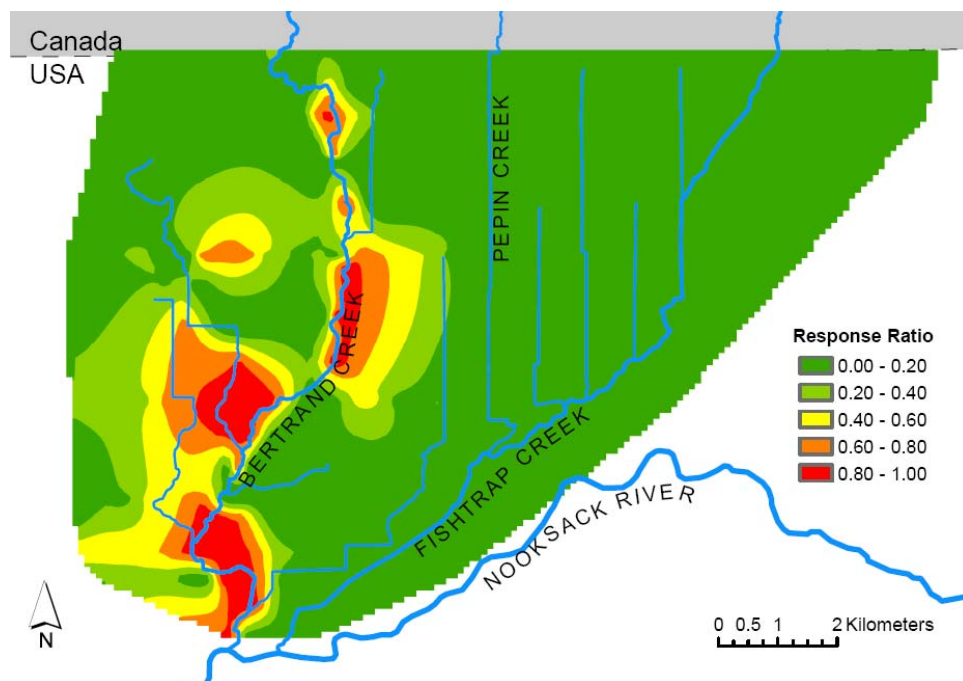


Figure 20. Raster map of Bertrand Creek response ratios.

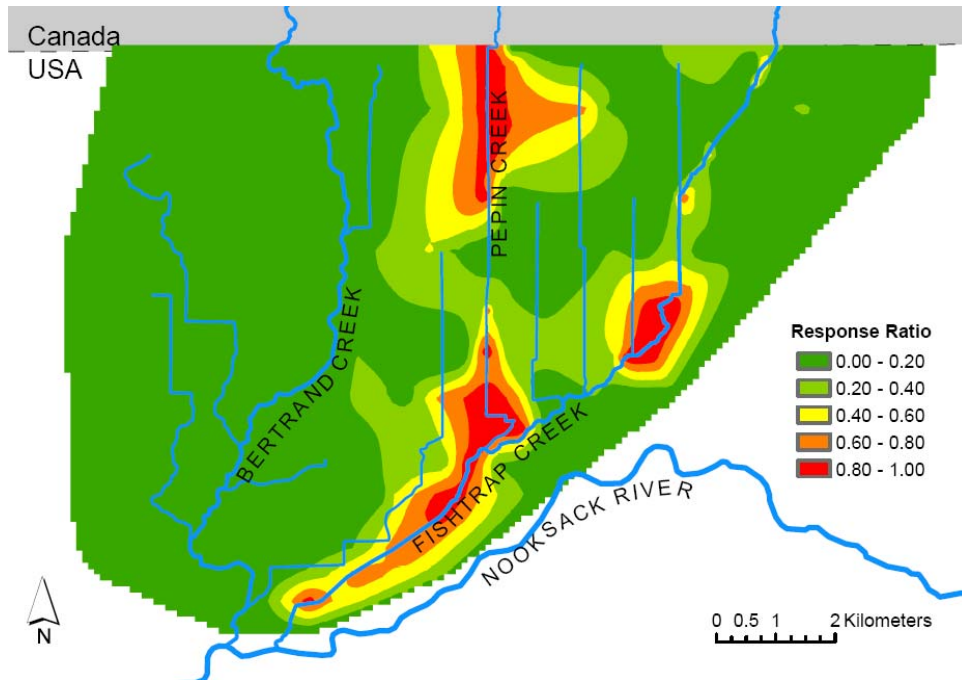


Figure 21. Raster map of Fishtrap Creek response ratios.

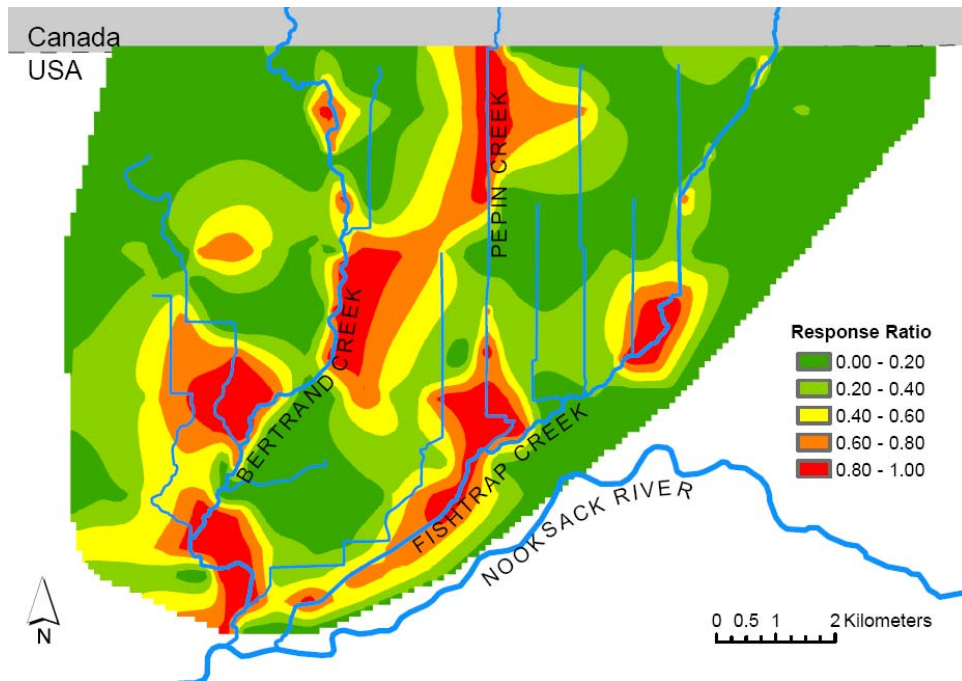


Figure 22. Combined raster interpolation of Bertrand and Fishtrap Creek response ratios using a natural neighbor technique.

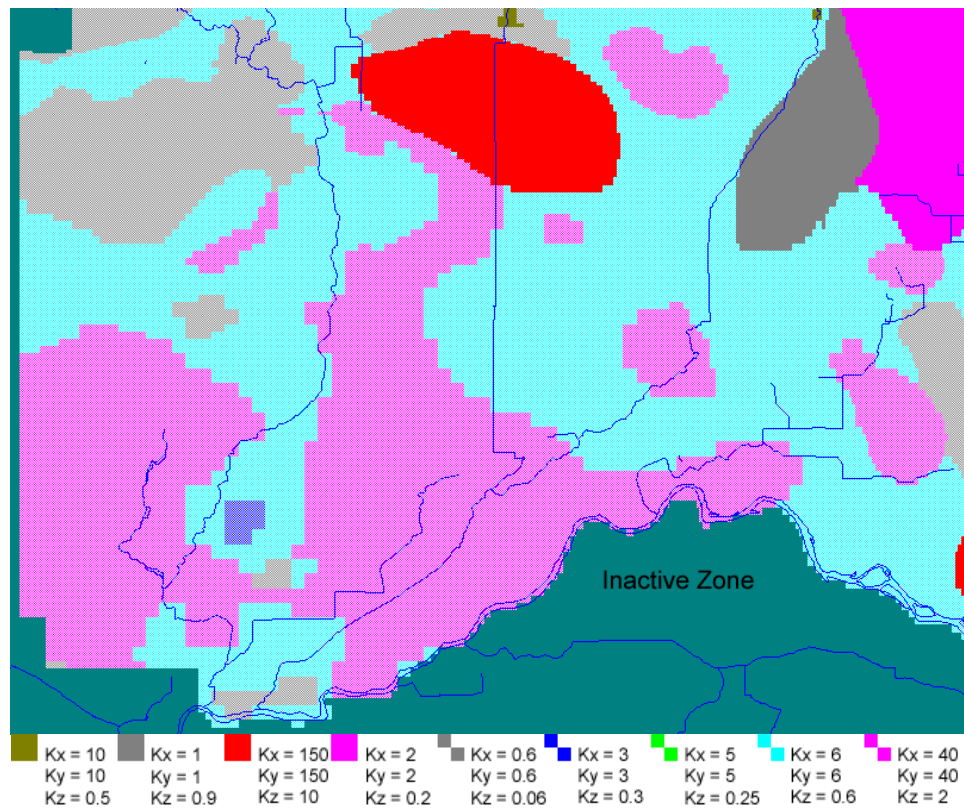


Figure 23. Hydraulic conductivity zones for model layer 3. Units are in meters/day.

Table 8. Bertrand and Fishtrap Creek flow responses, respectively, as the percent area of their corresponding watershed.

Response Zone	Bertrand Flow Response for Wells within Bertrand Creek Watershed (%)	Fishtrap Flow Response for Wells within Fishtrap Creek Watershed (%)
0.0–0.1	38.99%	23.03%
0.1–0.2	16.73%	27.41%
0.2–0.3	9.81%	15.12%
0.3–0.4	6.88%	8.07%
0.4–0.5	6.26%	5.33%
0.5–0.6	6.65%	5.41%
0.6–0.7	4.95%	4.94%
0.7–0.8	3.89%	4.08%
0.8–0.9	3.72%	4.27%
0.9–1.0	2.12%	2.33%

Table 9. Bertrand and Fishtrap Creek combined flow response as the percent area of their corresponding watershed.

Response Zone	Combined Flow Response for Wells within Bertrand Creek Watershed (%)	Combined Flow Response for Wells within Fishtrap Creek Watershed (%)
0.0–0.1	20.18%	21.95%
0.1–0.2	14.51%	27.64%
0.2–0.3	11.99%	15.73%
0.3–0.4	11.08%	8.19%
0.4–0.5	9.02%	5.13%
0.5–0.6	9.35%	5.47%
0.6–0.7	7.75%	4.99%
0.7–0.8	6.98%	3.97%
0.8–0.9	5.70%	4.44%
0.9–1.0	3.45%	2.50%

Table 10. Bertrand and Fishtrap Creek combined flow response as the percent area within 0.5 and 1.0 miles of Bertrand Creek.

Response Zone	Combined Flow Response for Wells within 0.5 mi of Bertrand Creek (%)	Combined Flow Response for Wells within 1.0 mi of Bertrand Creek (%)
0.0–0.1	17.87%	17.87%
0.1–0.2	14.12%	14.87%
0.2–0.3	11.35%	12.35%
0.3–0.4	7.40%	9.96%
0.4–0.5	6.07%	8.79%
0.5–0.6	9.23%	10.20%
0.6–0.7	7.65%	7.67%
0.7–0.8	7.55%	7.22%
0.8–0.9	10.32%	6.53%
0.9–1.0	8.44%	4.54%

Table 11. Bertrand and Fishtrap Creek combined flow response as the percent area within 0.5 and 1.0 miles of Fishtrap Creek.

Response Zone	Combined Flow Response for Wells within 0.5 mi of Fishtrap Creek (%)	Combined Flow Response for Wells within 1.0 mi of Fishtrap Creek (%)
0.0–0.1	19.53%	21.10%
0.1–0.2	19.35%	24.38%
0.2–0.3	13.16%	15.12%
0.3–0.4	10.33%	10.24%
0.4–0.5	7.95%	6.04%
0.5–0.6	8.35%	5.83%
0.6–0.7	7.36%	5.26%
0.7–0.8	5.48%	4.49%
0.8–0.9	4.67%	4.18%
0.9–1.0	3.82%	3.35%

6. Conclusions

Groundwater and surface water interactions are prominent within the Bertrand and Fishtrap Creek watersheds based on measured responses of stream flow and groundwater levels as well as modeling results. Summer low flows in these streams are currently at levels to jeopardize fish habitat. Hence, an innovative conjunctive management scheme is needed. This study investigated the replacement of surface water diversions with groundwater withdrawals using a numerical model MODFLOW. Response ratios, calculated from the modeled change in stream flow divided by the pumping rate, were used to assess the impact on stream flow of exchanging a surface water diversion with a groundwater pumping well, based on the steady-state groundwater flow model. Resulting response ratios ranged from 0 to 1.0, with 0 representing no impact on the stream and 1 representing an impact equivalent to that of a surface water withdrawal of the same pumping rate. The model demonstrated that the greatest values occurred in close proximity to the creeks and in areas with high hydraulic conductivity.

Simulation results suggested that replacing surface water diversions with groundwater pumping wells may be a viable alternative for improving summer stream flows. It is clear that pumping wells do impact Bertrand and Fishtrap Creek flows, but if placed within zones of a low response ratio, less of an impact would be felt than removing an equivalent amount of water directly from the stream. Within a one mile band, 64% of Bertrand Creek had combined response ratios less than 0.5. While within a one mile band of Fishtrap Creek, 77% of that area had combined response ratios less than 0.5, indicating very favorable exchange opportunities for both creeks.

Because MODFLOW is difficult to understand and operate for non-specialists, response functions were created and, in conjunction with STELLA software, a visually-pleasing and easy-to-use interface was created through which users can learn about groundwater and surface water interactions within the study area. The STELLA model provides a quick and easy estimation of the stream flow impacts to Bertrand and Fishtrap Creeks without the need to re-run the MODFLOW groundwater model.

There seems to be a strong connection between a given response ratio and the hydraulic conductivity of the model layer containing the screened interval of the pumping well. Therefore, the response zones may only be as reliable as the hydraulic conductivity values assigned by Scibek and Allen (2005). In the future, additional work to quantify these values would be beneficial.

While the additional field data incorporated into the original groundwater flow model provided improved local detail, calibration results suggest that additional research and data collection could be used to improve model calibration locally. Specifically, accurate knowledge of how much water is being withdrawn from both creeks for irrigation and how much groundwater is

being pumped in the surrounding area is lacking. Knowledge of this information would help to improve the calibration of the groundwater model. Furthermore, additional well monitoring stations are needed to observe the groundwater table elevation throughout the year.

Transient effects were also not studied in this project. In order to determine precisely when the effects of a pumped well will impact a nearby stream, a transient model is needed. The steady-state model predicted how much the stream will be impacted given an infinite amount of time and, therefore, represents a worst-case scenario. A transient model can indicate when and by how much a pumped well will impact a stream assuming sufficient pumping data are available. A transient version of the groundwater flow model is available from SFU, and if updated according to this study, could be used to investigate transient conditions.

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Appendix A

Additional Field Investigation Data

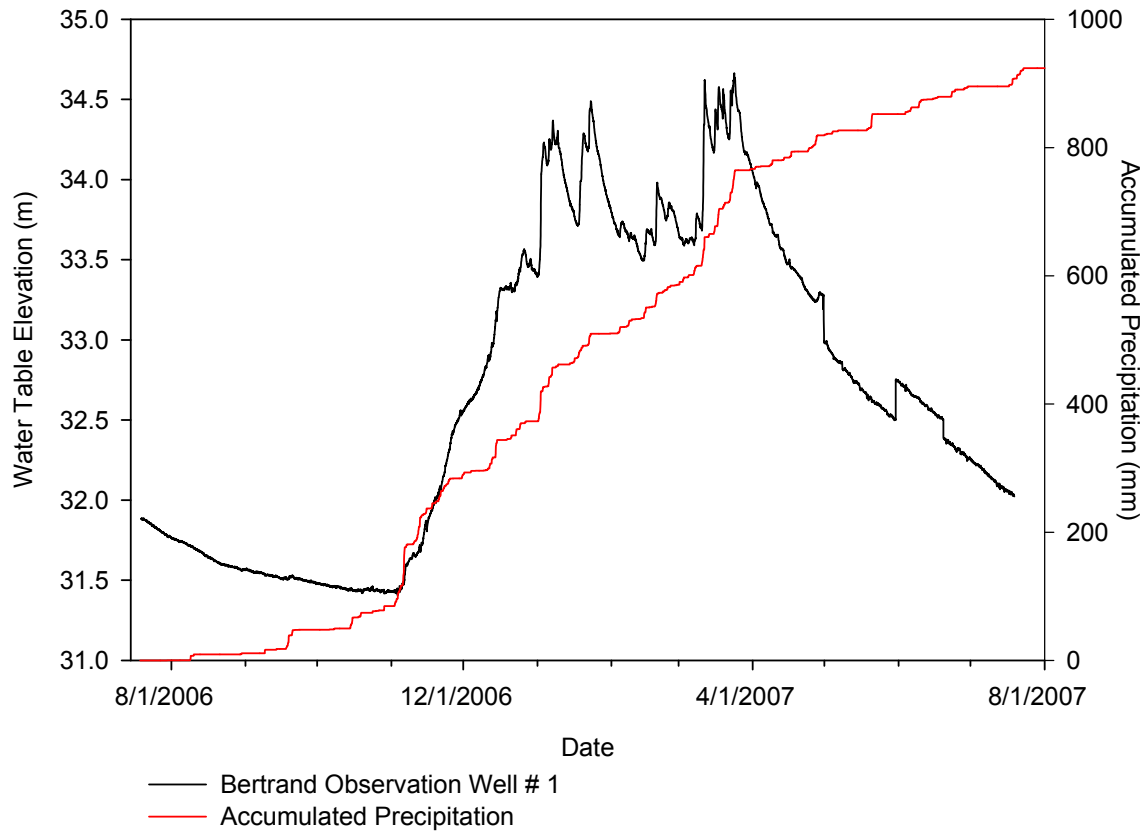


Figure A1. Water-table elevation for Bertrand observation well # 1 and local precipitation (WAWN, 2007 and NCDC, 2007b).

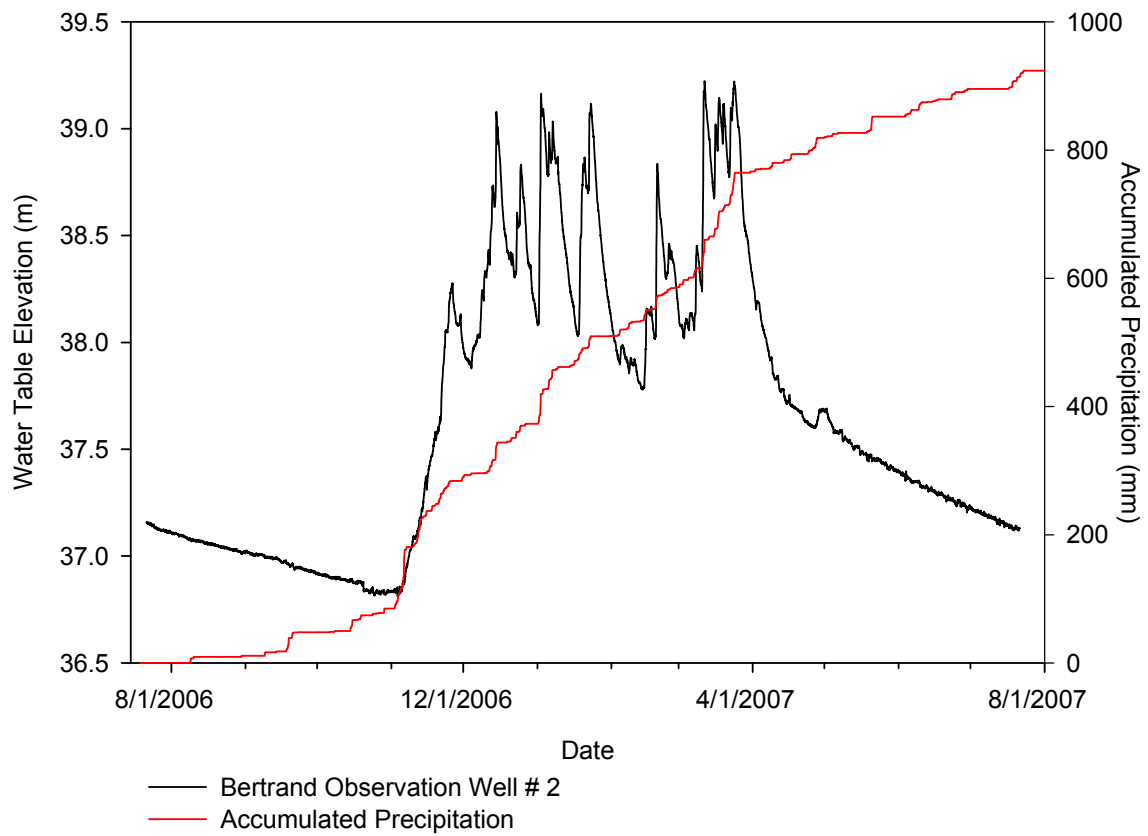


Figure A2. Water-table elevation for Bertrand observation well # 2 and local precipitation (WAWN, 2007 and NCDC, 2007b).

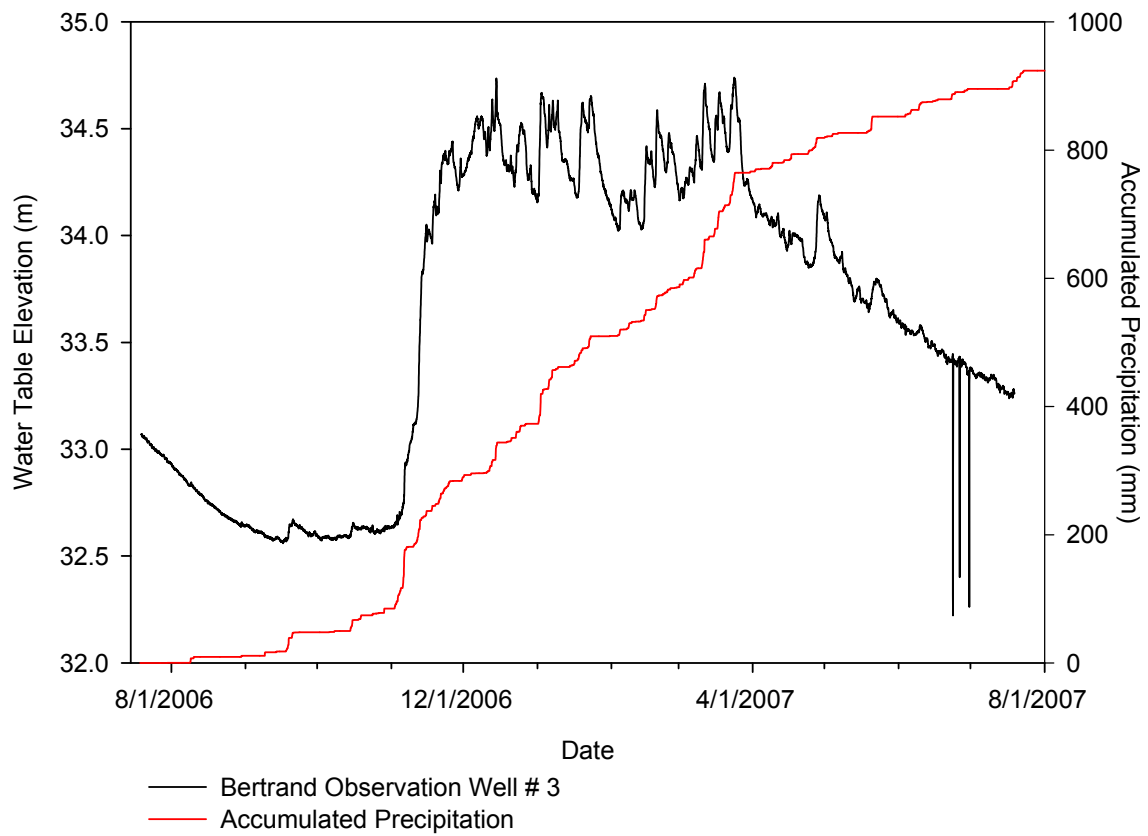


Figure A3. Water-table elevation for Bertrand observation well # 3 and local precipitation (WAWN, 2007 and NCDC, 2007b).

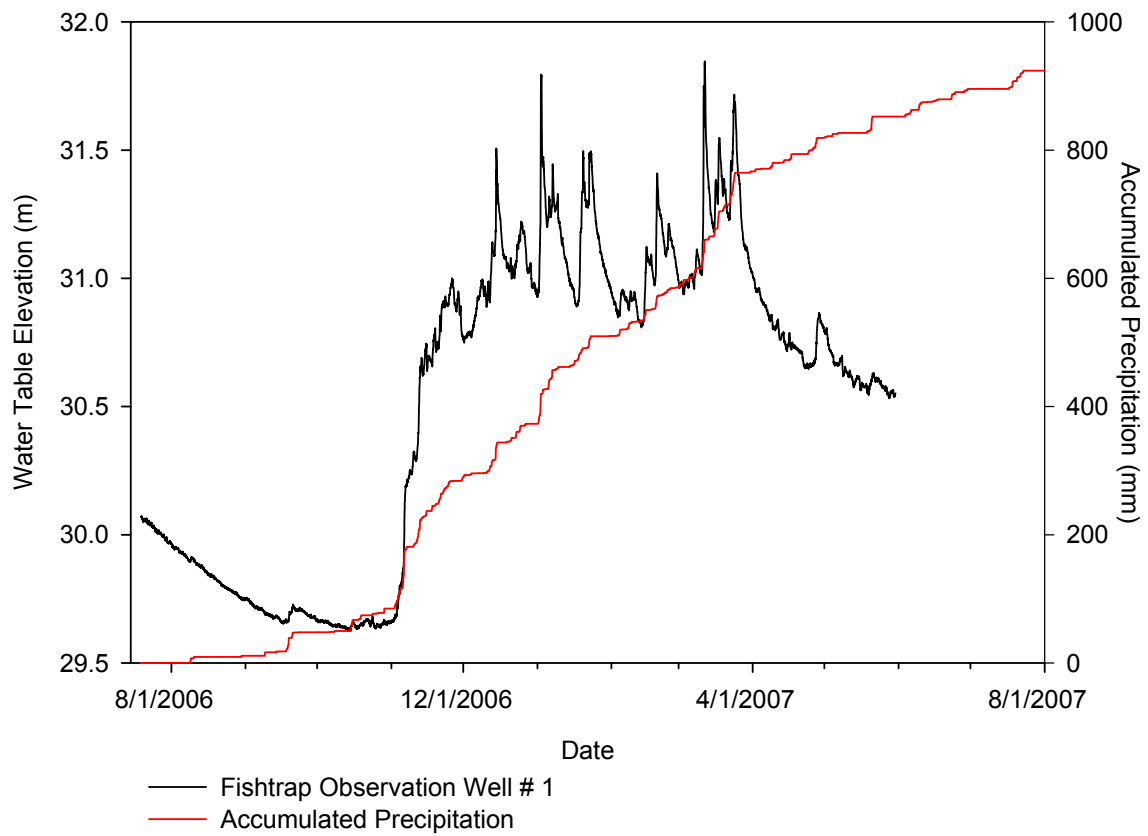


Figure A4. Water-table elevation for Fishtrap observation well # 1 and local precipitation (WAWN, 2007 and NCDC, 2007b).

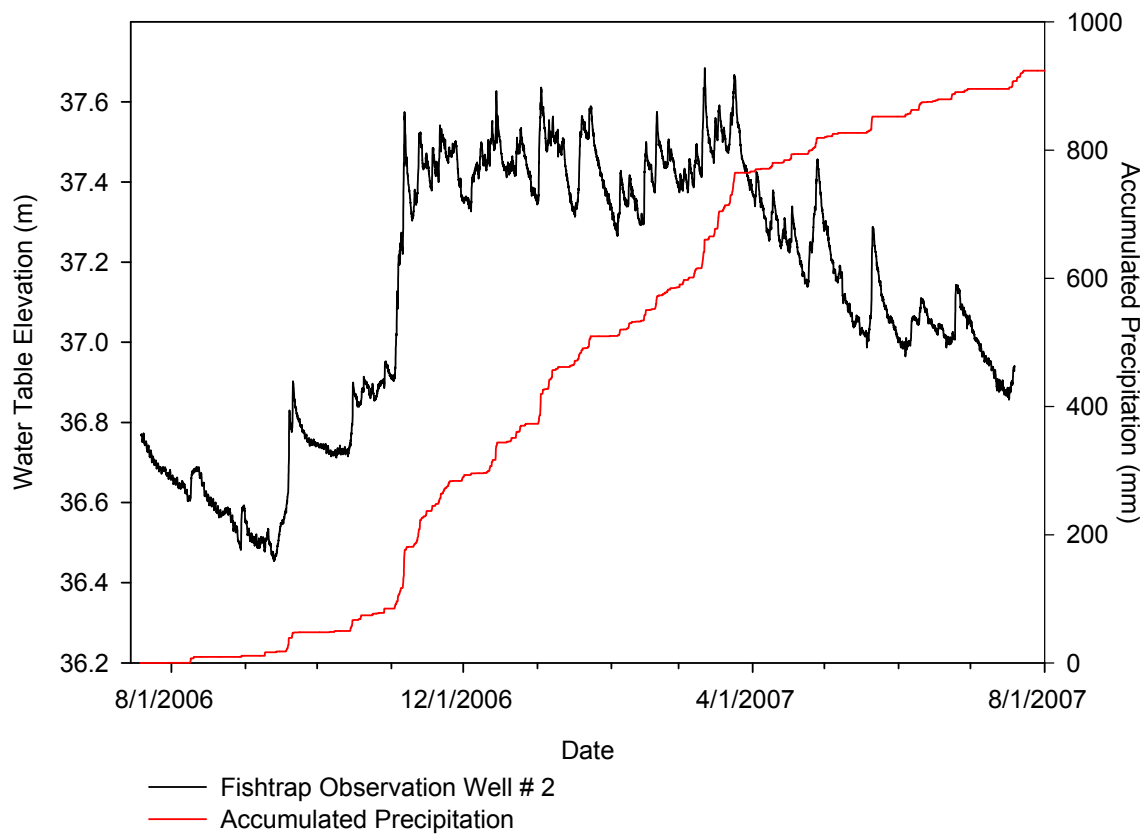


Figure A5. Water-table elevation for Fishtrap observation well # 2 and local precipitation (WAWN, 2007 and NCDC, 2007b).

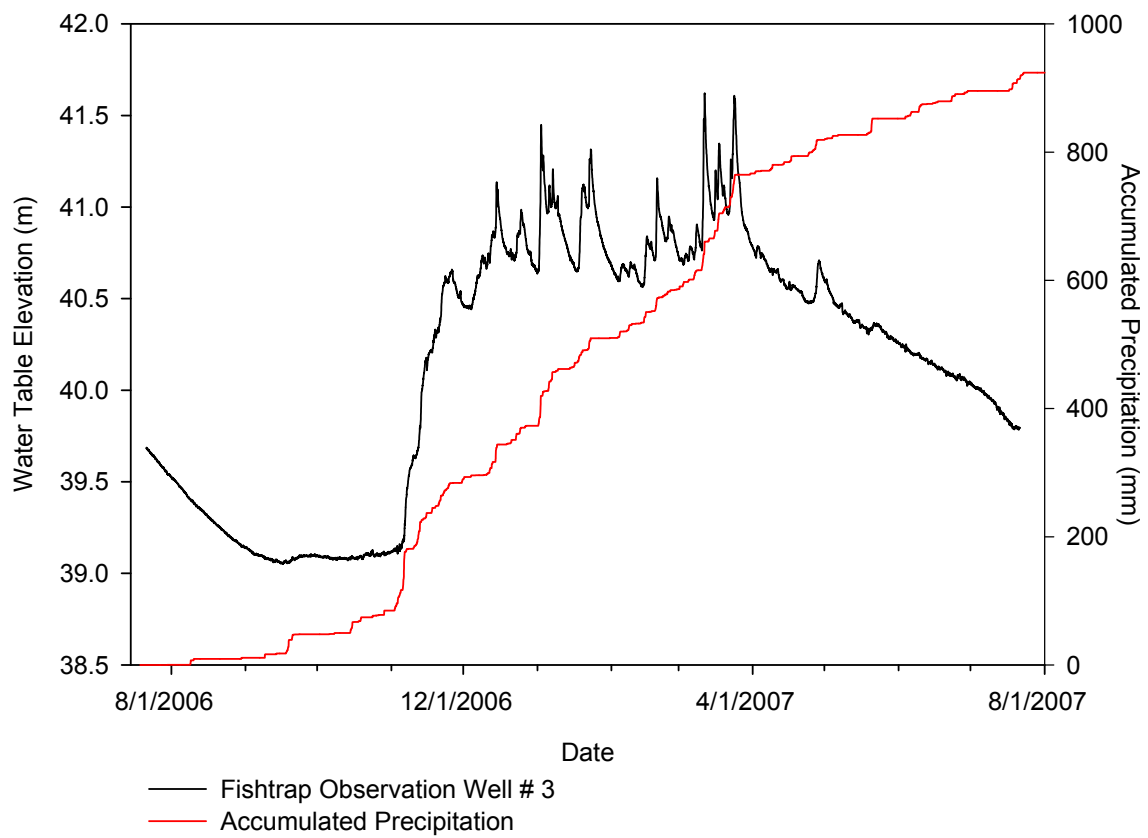


Figure A6. Water-table elevation for Fishtrap observation well # 3 and local precipitation (WAWN, 2007 and NCDC, 2007b).

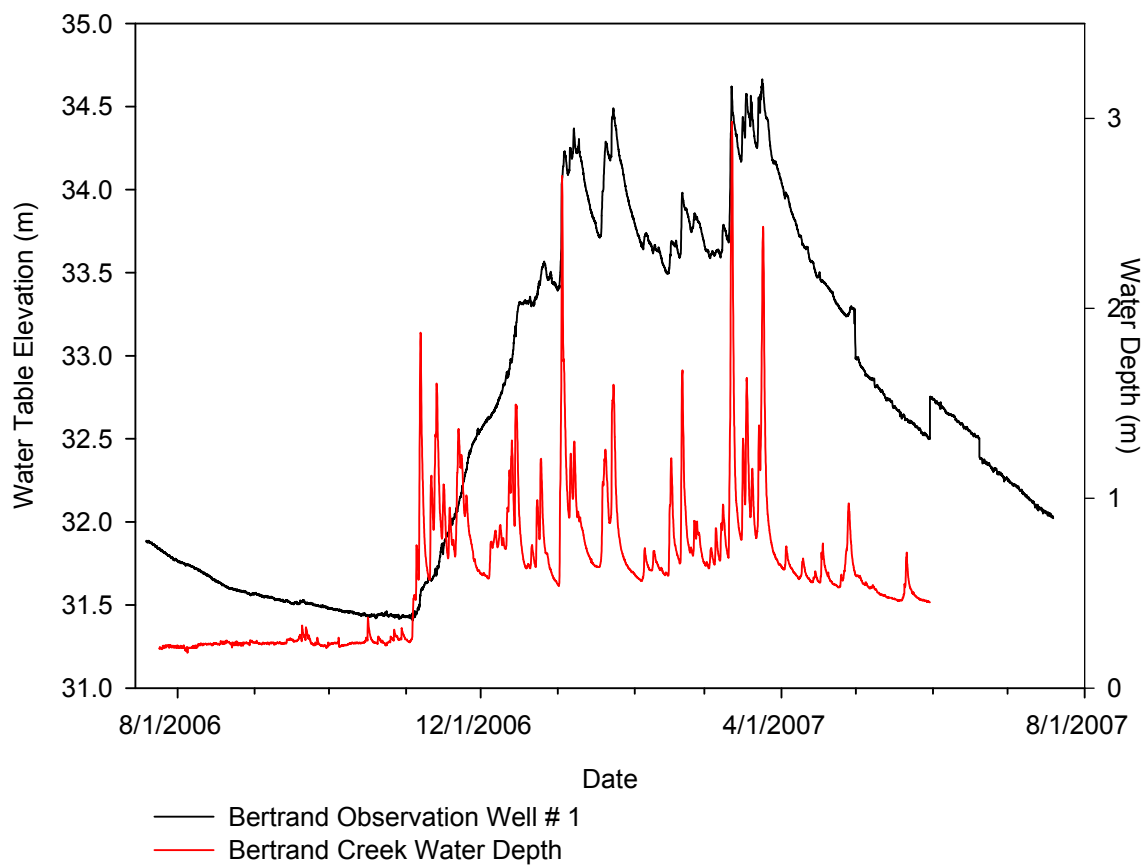


Figure A7. Water-table elevation for Bertrand observation well # 1 and Bertrand Creek water depth.

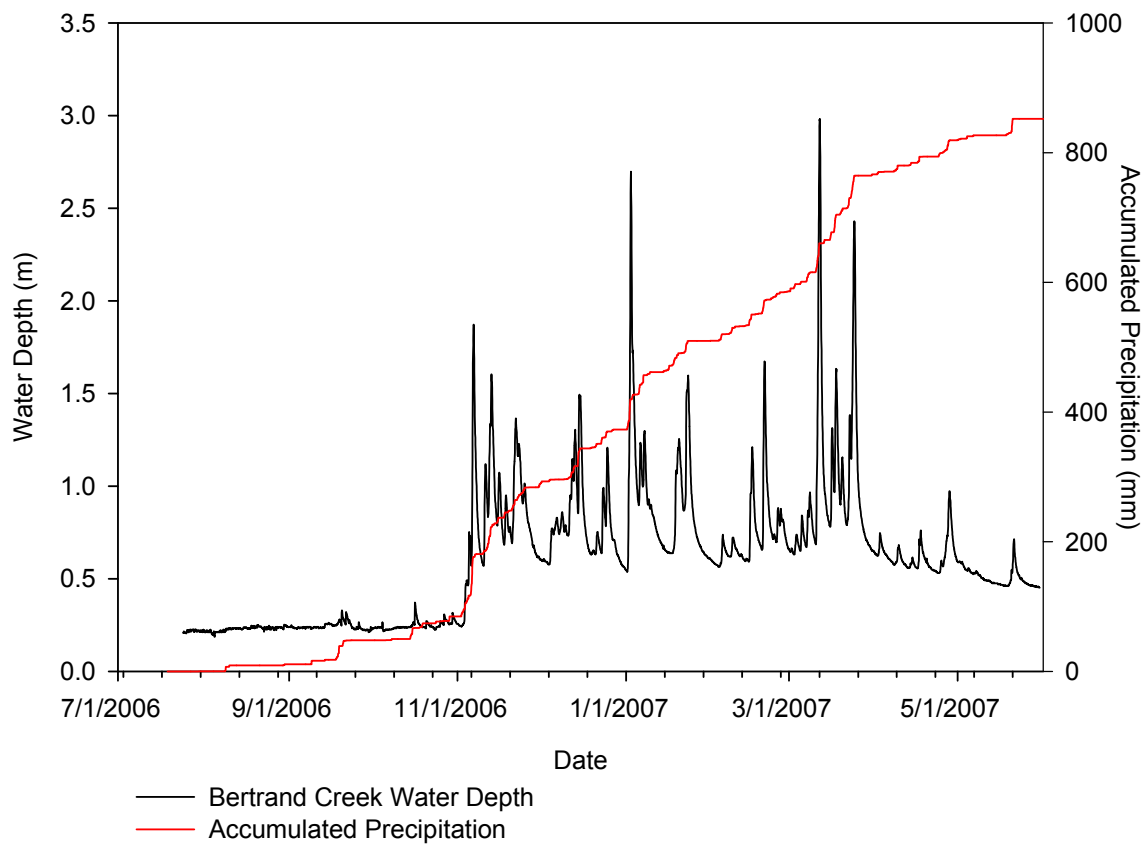


Figure A8. Bertrand Creek water depth and local precipitation (WAWN, 2007 and NCDC, 2007b).

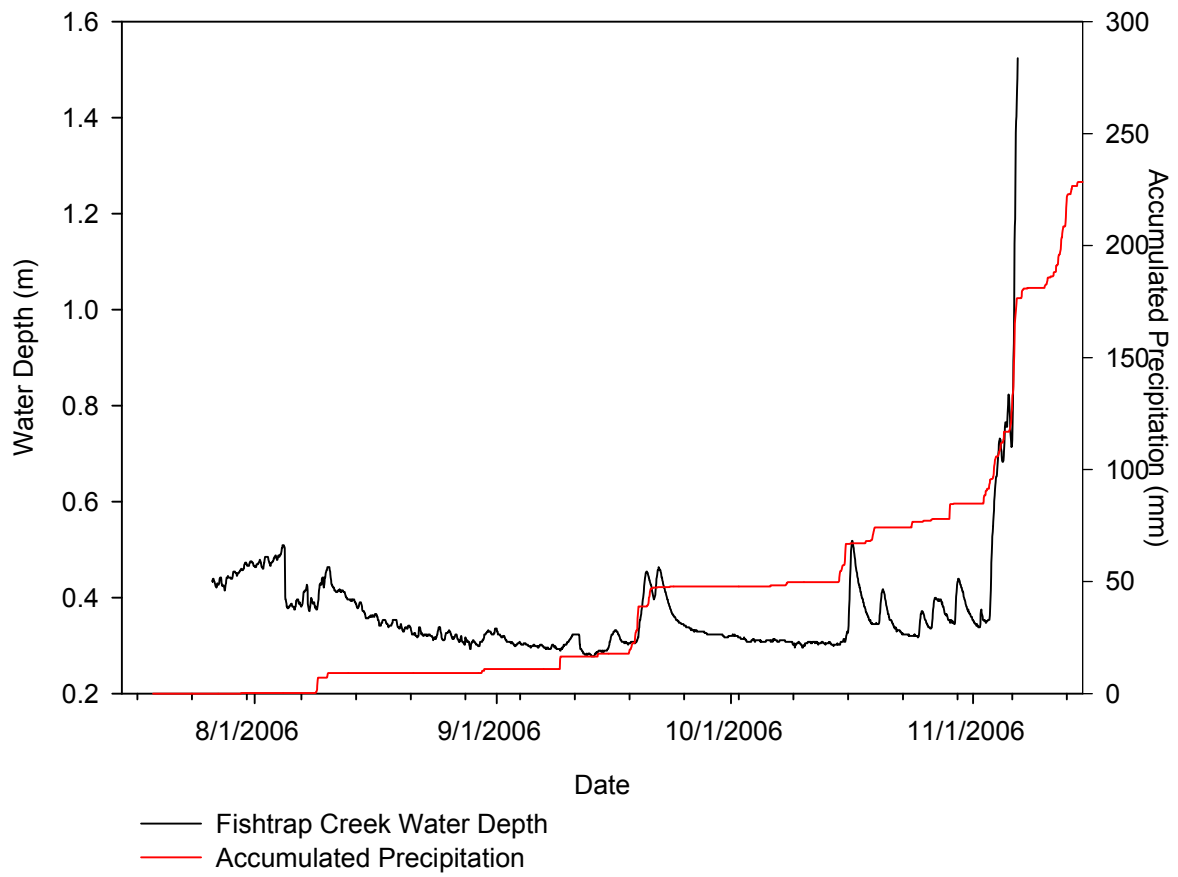


Figure A9. Fishtrap Creek water depth and local precipitation (WAWN, 2007 and NCDC, 2007b).

Appendix B

Additional Model Boundary Condition Data

Table B1. USGS observation well data (Kimbrough *et al.*, 2004 and 2005).

USGS Well Number	Latitude (DMS)	Longitude (DMS)	Date (mm-dd-yyyy)	Water Table Elevation (ft)
485607122321401	48°56'02.24"	122°32'18.33"	5/1/1987	39.00
			8/15/1990	39.44
			10/17/1990	39.88
			11/14/1990	42.79
			12/18/1990	43.38
			1/18/1991	43.76
			2/20/1991	42.79
			3/14/1991	42.16
			3/24/1991	41.50
			5/21/1991	41.05
			6/26/1991	39.90
			7/18/1991	40.05
			8/13/1991	39.80
			9/25/1991	39.47
			10/23/1991	39.52
			9/3/2004	39.72
			8/29/2005	39.53
485700122313401	48°56'59.00"	122°31'39.34"	5/3/1989	73.30
			8/14/1990	70.85
			9/18/2002	70.07
			9/3/2004	69.92
			8/29/2005	70.32
			3/7/2006	74.13
485725122242701	48°57'23.95"	122°24'32.09"	10/22/1997	87.07
			10/22/1997	87.02
			11/4/1997	86.16
			11/11/1997	86.16
			4/14/1998	87.74
			6/17/1998	87.74
			6/17/1998	87.74
			8/21/2006	85.83
485746122250301	48°57'46"	122°25'03"	8/4/1989	101.00
			7/23/1990	105.40
			8/27/1990	104.07
			10/17/1990	103.76
			11/14/1990	105.29
			12/17/1990	110.55
			1/19/1991	111.50
			2/21/1991	111.22
			3/14/1991	110.96

USGS Well Number	Latitude (DMS)	Longitude (DMS)	Date (mm-dd-yyyy)	Water Table Elevation (ft)
			4/23/1991	109.82
			5/21/1991	108.61
			6/25/1991	107.36
			7/17/1991	105.90
			8/22/1991	104.58
			9/26/1991	103.93
			10/23/1991	103.93
485749122250301	48°57'48.08"	122°25'08.25"	7/27/1994	106.00
			4/16/1998	106.00
			9/3/2004	107.36
			8/29/2005	108.73
			8/29/2006	107.35
485751122241601	48°57'50.88"	122°24'21.39"	10/23/1997	103.16
			11/5/1997	102.51
			11/11/1997	102.51
			9/20/2001	101.51
			9/2/2003	101.27
			9/3/2004	100.81
			8/29/2005	102.33
			3/7/2006	108.25
			8/17/2006	101.76
485751122304601	48°57'51"	122°30'46"	10/13/2004	92.41
			10/13/2004	92.41
			10/14/2004	92.41
			8/29/2005	93.46
			3/7/2006	98.18
			8/10/2006	93.33
485755122253901	48°57'54.40"	122°25'42.60"	10/21/1997	107.08
			10/28/1997	107.02
			11/5/1997	107.61
			11/12/1997	107.61
			4/15/1998	107.45
			9/20/2001	105.92
			9/2/2003	105.05
			9/3/2004	105.46
			8/29/2005	105.64
			3/7/2006	107.57
485817122244701	48°58'17.63"	122°24'40.00"	8/16/2006	105.45
			2/10/1998	122.00
			9/3/2004	112.88
485843122242301	48°58'43.19"	122°24'28.64"	8/30/2006	113.70
			10/23/1997	124.16
			11/5/1997	124.81
			11/11/1997	124.81
			9/21/2001	121.56
485843122242302	48°58'43.19"	122°24'28.73"	8/21/2006	120.29
			10/23/1997	124.21

USGS Well Number	Latitude (DMS)	Longitude (DMS)	Date (mm-dd-yyyy)	Water Table Elevation (ft)
			11/5/1997	124.86
			11/5/1997	124.86
			11/11/1997	124.86
			9/21/2001	121.49
			8/21/2006	120.33
485917122241901	48°59'16.23"	122°24'24.30"	10/24/1997	129.09
			10/27/1997	128.89
			11/10/1997	130.01
			9/21/2001	126.35
			8/28/2002	125.13
			8/28/2002	125.13
			9/2/2003	124.30
			9/2/2004	124.37
			8/29/2005	125.62
			3/7/2006	133.04
			8/24/2006	124.73
485932122241601	48°59'31.14"	122°24'22.93"	4/27/1988	134.00
			7/12/1990	134.05
			8/30/1990	132.24
			10/18/1990	132.24
			11/14/1990	137.11
			12/18/1990	139.96
			1/17/1991	139.88
			2/20/1991	139.85
			3/14/1991	139.12
			4/23/1991	137.81
			5/21/1991	136.48
			6/25/1991	135.30
			7/17/1991	132.92
			8/21/1991	132.22
			9/29/1991	132.95
			10/23/1991	132.58
485934122305901	48°59'34"	122°30'59"	2/22/1983	132.00
			12/18/1990	131.20
			1/18/1991	130.84
			2/20/1991	131.18
			3/14/1991	130.98
			4/23/1991	130.80
			6/6/1991	130.24
			6/25/1991	129.98
			7/17/1991	129.47
			8/22/1991	128.99
			9/25/1991	129.26
			10/23/1991	129.15
485936122322901	48°59'36"	122°32'29"	6/9/1980	195.00
			8/10/1990	189.48
			11/16/1990	190.15

USGS Well Number	Latitude (DMS)	Longitude (DMS)	Date (mm-dd-yyyy)	Water Table Elevation (ft)
			2/20/1991	191.07
			3/14/1991	191.18
			5/21/1991	190.82
			6/25/1991	189.59
			7/17/1991	190.10
			8/22/1991	189.50
			9/25/1991	189.90
			10/23/1991	190.01

Appendix C

Numerical Ground-Water Model Hydraulic Conductivity Zones

Shown below are the hydraulic conductivity zones defined by Scibek and Allen (2005) within the study area for model layers 2-5.

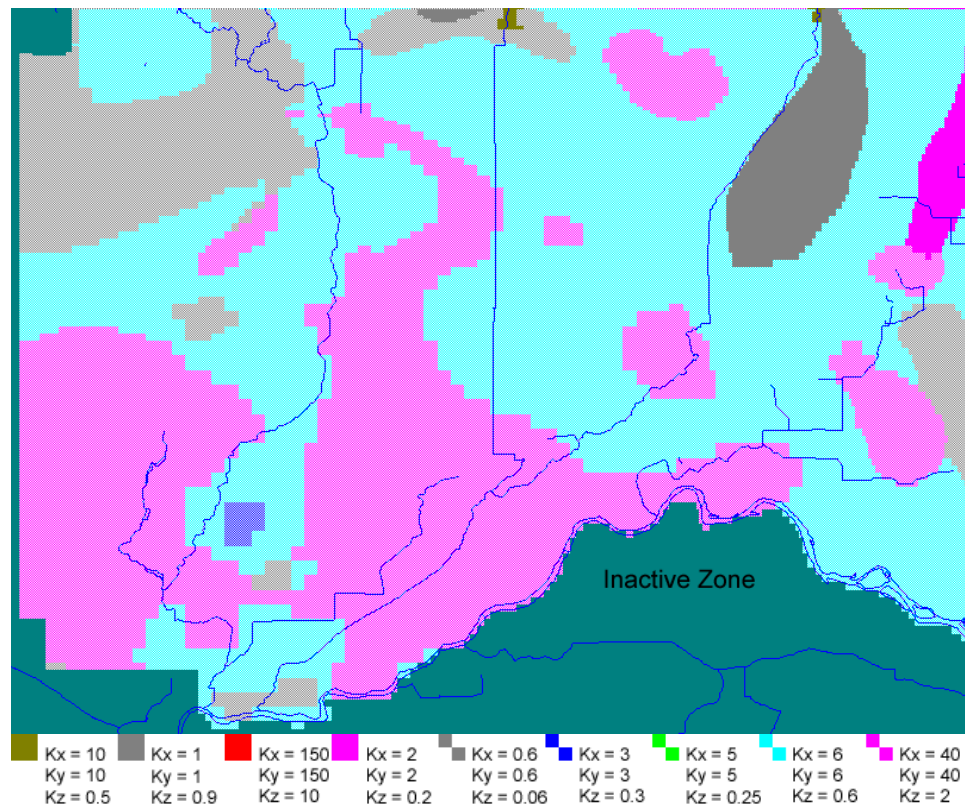


Figure C1. Hydraulic conductivity zones for model layer 2. Units are in m/d.

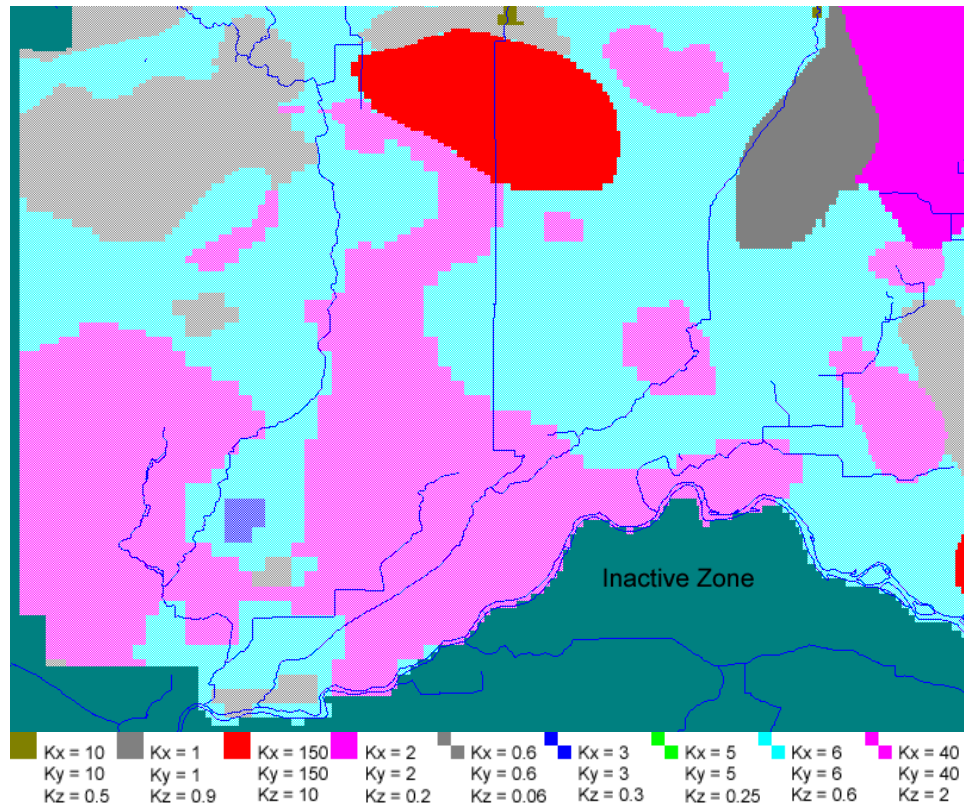


Figure C2. Hydraulic conductivity zones for model layer 3. Units are in m/d.

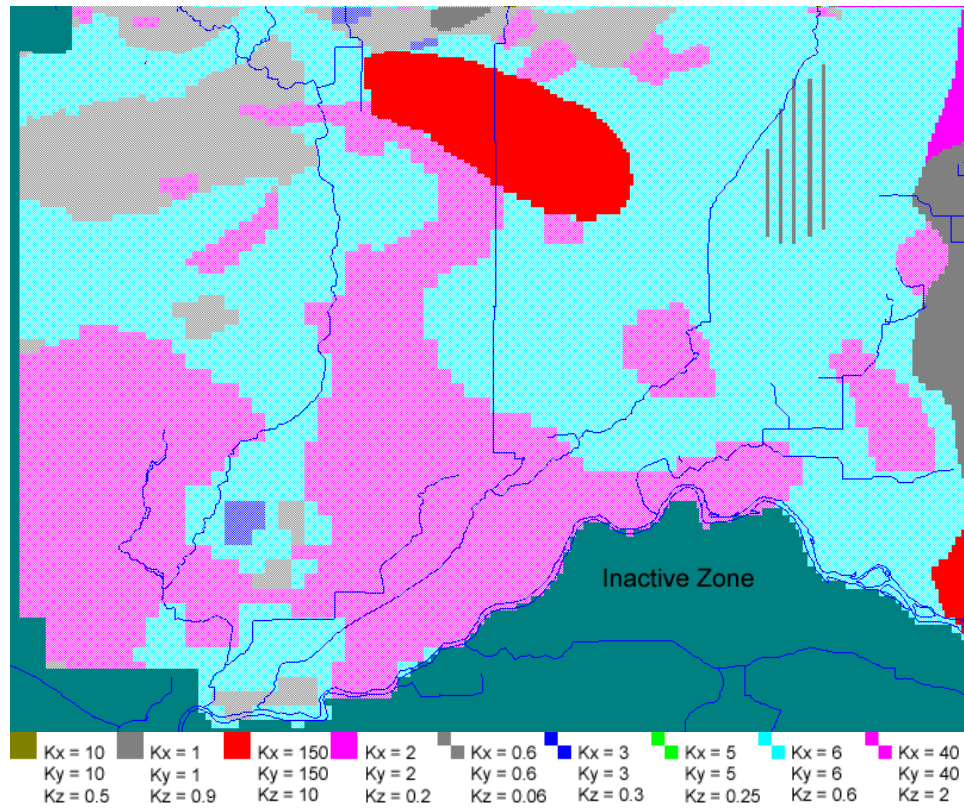


Figure C3. Hydraulic conductivity zones for model layer 4. Units are in m/d.

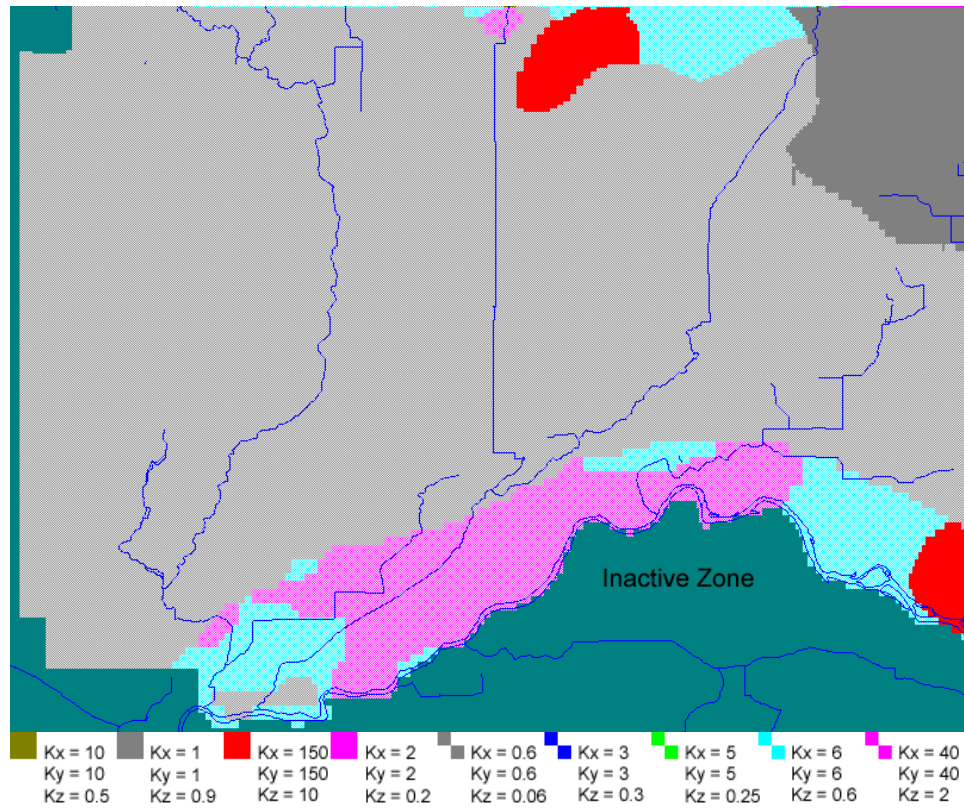


Figure C4. Hydraulic conductivity zones for model layer 5. Units are in m/d.

Appendix D

STELLA Model Screen Captures

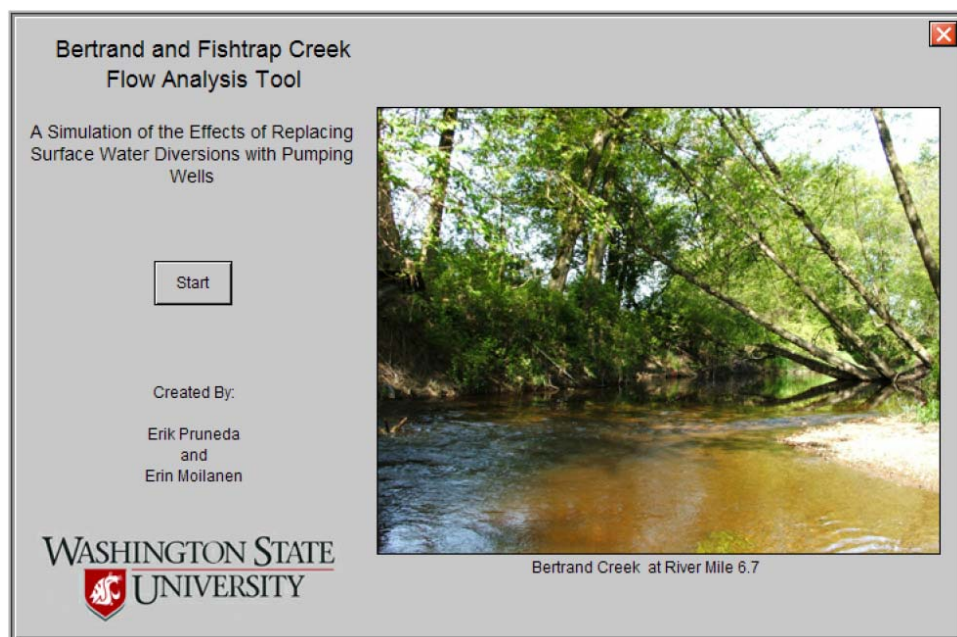


Figure D1. Screen capture of STELLA model title screen.

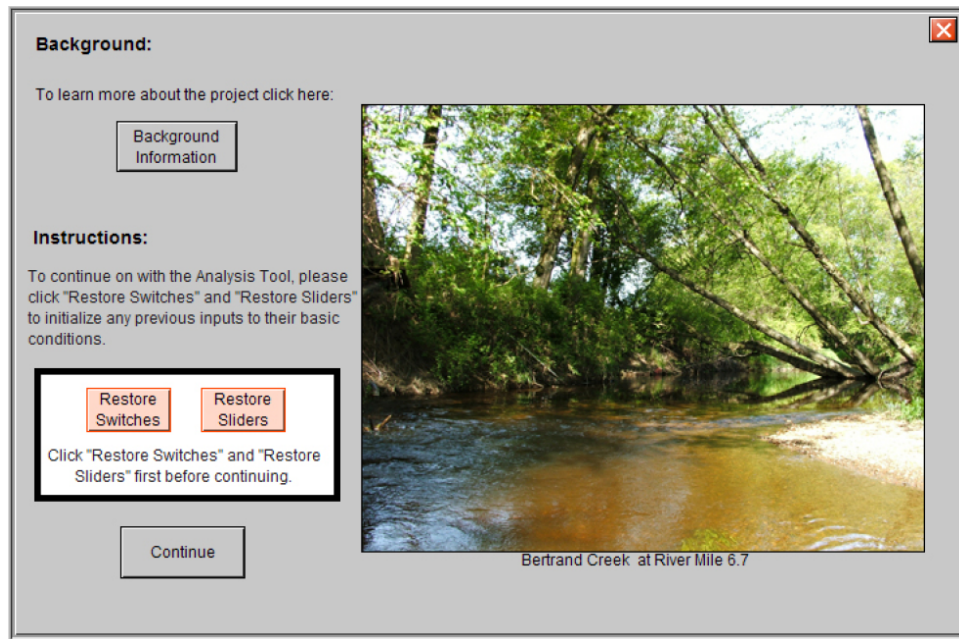


Figure D2. Screen capture of STELLA model. User can view background information, and restore switches and sliders to default values.

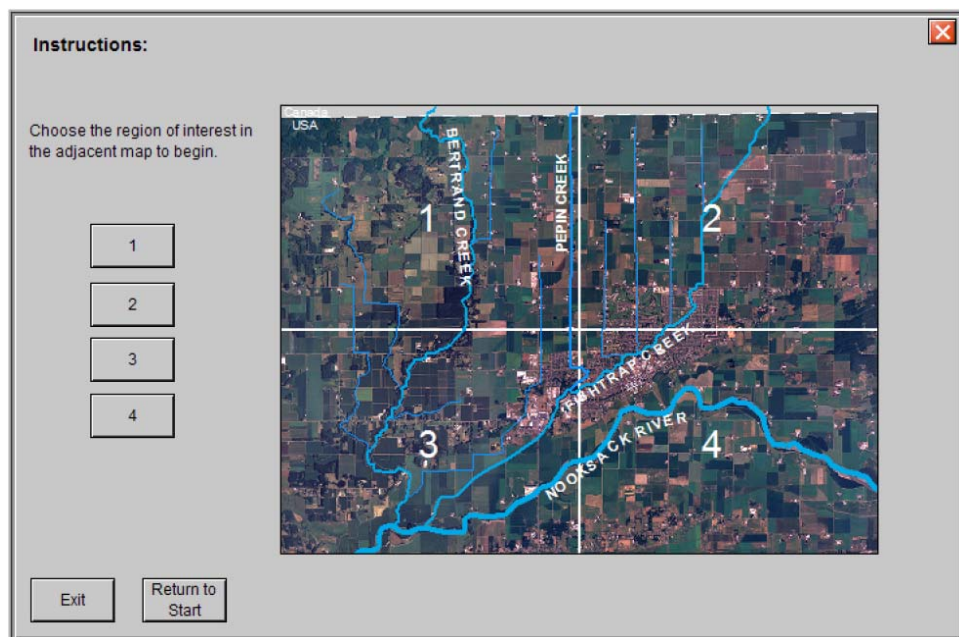


Figure D3. Screen capture of STELLA model. User is asked to choose a region of interest.

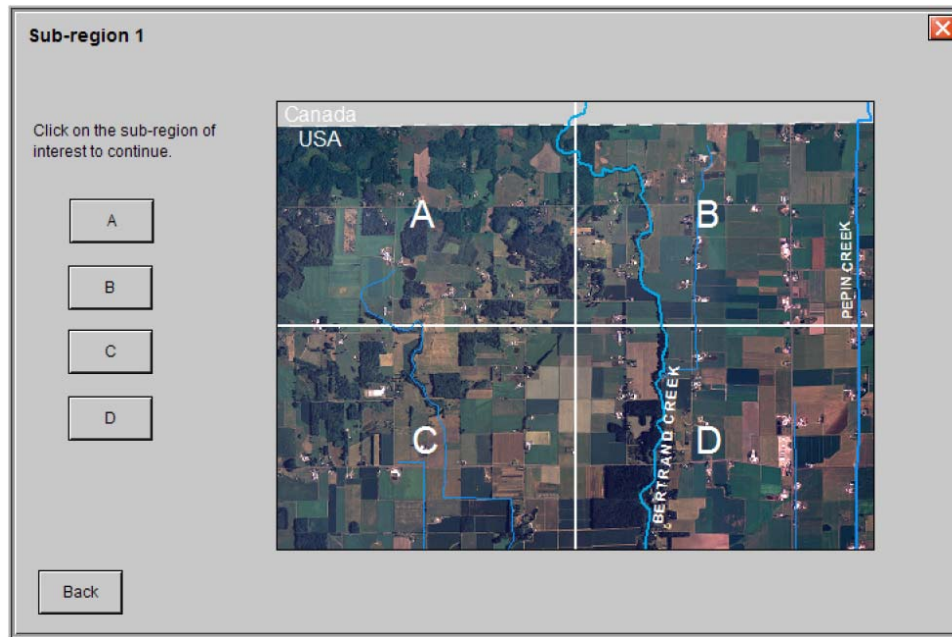


Figure D4. Screen capture of STELLA model. User is asked to choose a sub-region of interest.

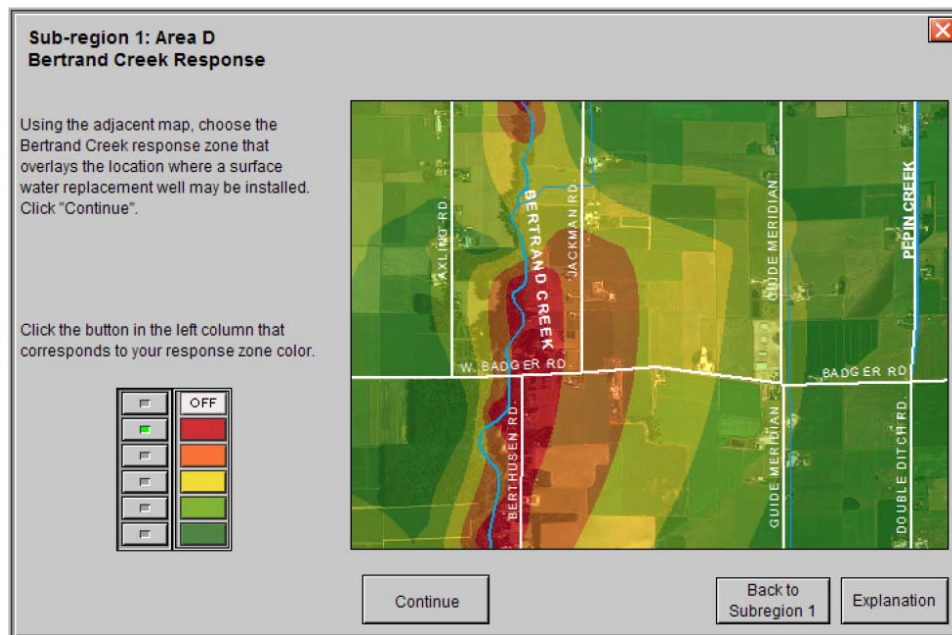


Figure D5. Screen capture of STELLA model. User is asked to choose the Bertrand Creek response zone where a surface water replacement well is to be placed.

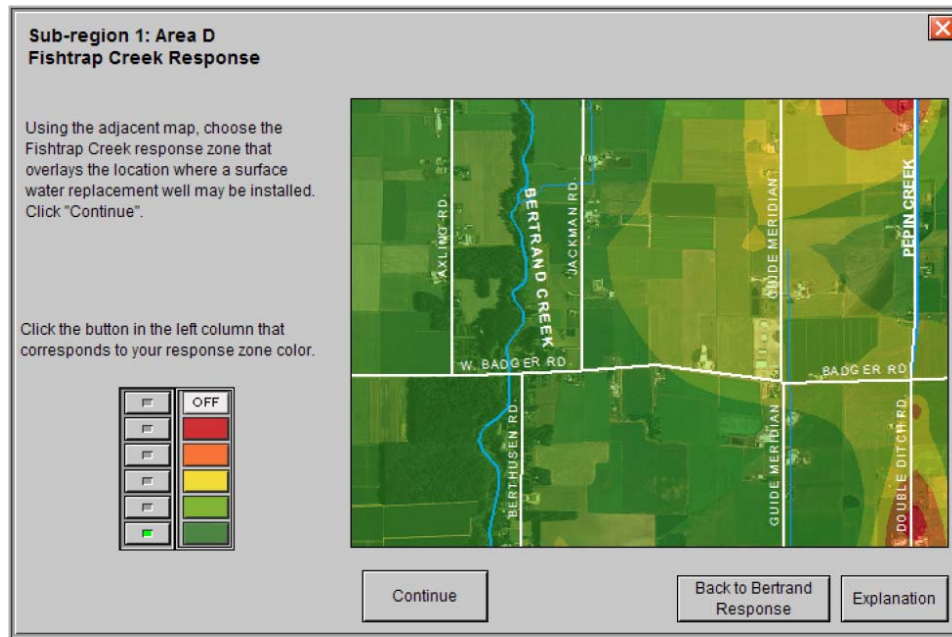


Figure D6. Screen capture of STELLA model. User is asked to choose the Fishtrap Creek response zone where a surface water replacement well is to be placed.

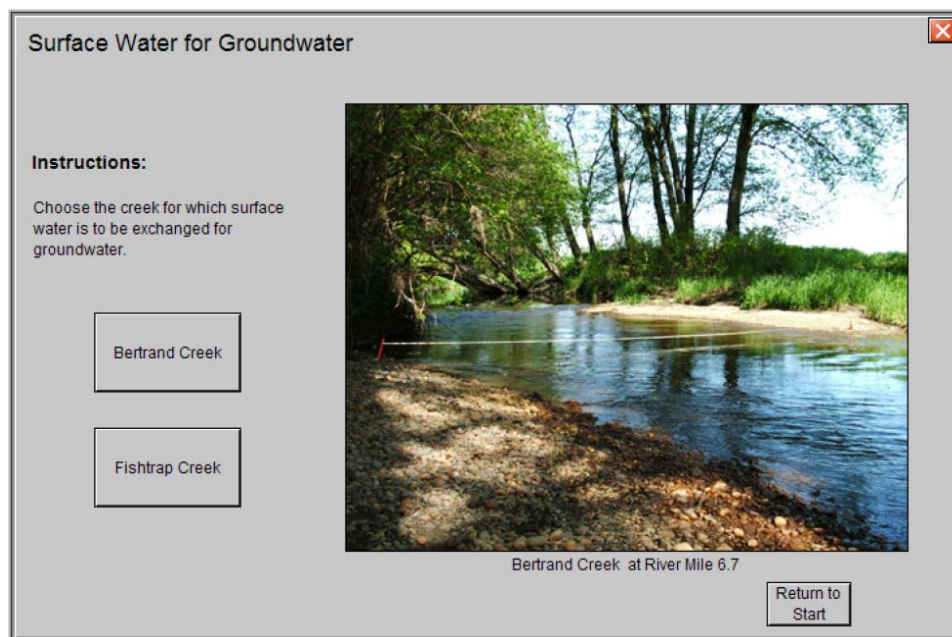


Figure D7. Screen capture of STELLA model. User is asked to select the creek for which surfacewater is to be exchanged for groundwater.

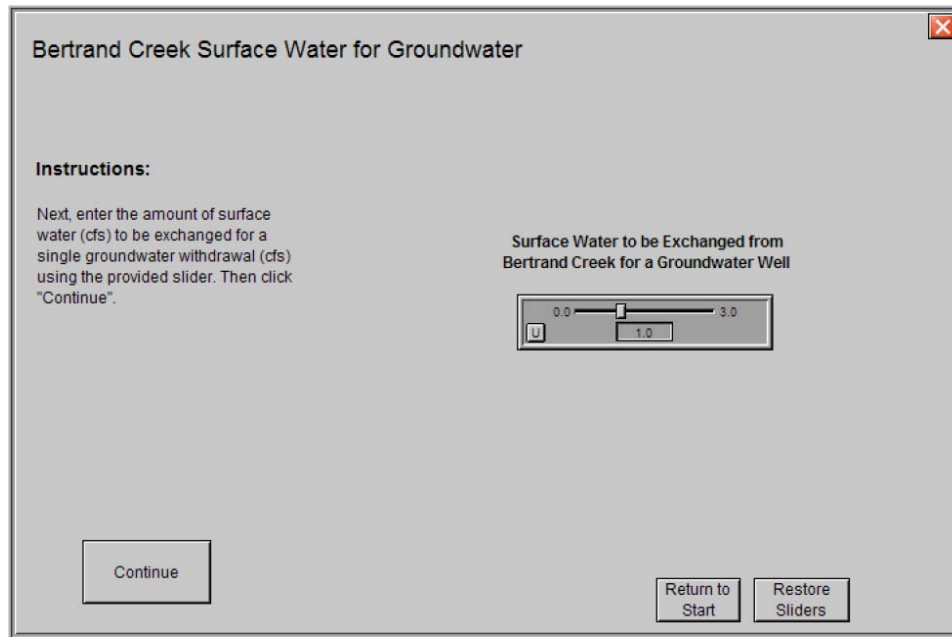


Figure D8. Screen capture of STELLA model. User is asked to select the amount of surface-water diversion to be exchanged for groundwater withdrawal.

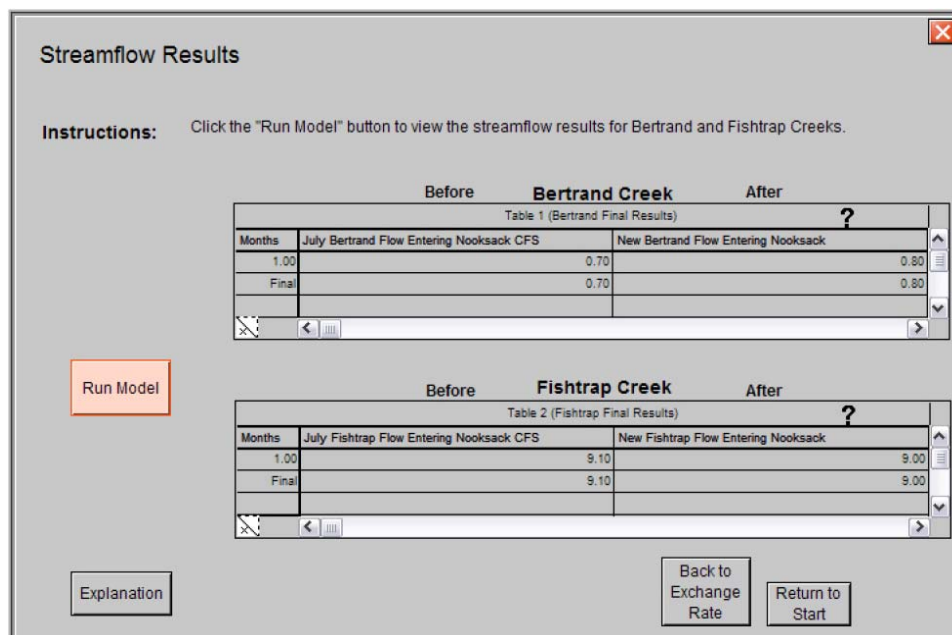


Figure D9. Screen capture of STELLA model. User is allowed to run the model and view the results.