Use of stream response functions to determine impacts of replacing surface-water use with groundwater withdrawals

Erik B. Pruneda · Michael E. Barber · Diana M. Allen · Joan Q. Wu

Abstract A regional-scale numerical groundwater model is used to study the impacts of replacing surface-water use with groundwater wells to improve low-flow stream conditions for endangered species within the Bertrand and Fishtrap watersheds, southern British Columbia, Canada and Washington, USA. Stream response functions ranging from 0 to 1.0 were calculated for individual wells placed within a steady-state groundwater flow model at varying distances from the streams to determine the impact that these replacement wells, operating under sustained pumping rates, would have on summer instream flows. Lower response ratios indicate groundwater pumping will have less of an impact on streamflow than taking an equivalent amount of water directly from a surface-water source. Results show that replacing surface-water use with groundwater withdrawals may be a viable alternative for increasing summer streamflows. Assuming combined response factors should be ≤0.5 for irrigators to undergo the expense of installing new wells, ~57% of the land area within 0.8km of Bertrand Creek would be suitable for replacement wells. Similarly, 70% of the land area within 0.8km of Fishtrap Creek was found to be appropriate. A visual analysis tool was developed using STELLA to allow stakeholders to quickly evaluate the impact associated with moving their water right.

Keywords Groundwater/surface-water relations · Stream response functions · Water-resources conservation · Numerical modeling · Canada · USA

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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, USA and southern British Columbia, Canada, 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 (e.g., Adelsman 2003). Recent instream flow studies on two watersheds in northwest Washington (Bertrand and Fishtrap Creeks) found that summer flows are too low to support desired salmon uses (WAC 173-501-030 1985; Kemblowski et al. 2002). The problem is not limited to surface-water use, as many groundwater withdrawals from wells completed within the shallow alluvial aquifer nearby the streams cause significant decreases in streamflow through surface water and groundwater interaction (Winter et al. 1998; Hantush 2005).

An innovative way to manage water demand is needed to help alleviate the problem in this and similar watersheds. One proposed alternative involves replacing surface-water use with groundwater withdrawals. While removing surface-water use will keep the previously used water in the stream, the overall net effect on streamflow will depend on the location of replacement wells and aquifer and streambed properties. Essentially, groundwater pumping wells should be far enough away from the streams so as not to have a negative effect on streamflow, either within some specified time period (i.e., a transient response) or indefinitely (i.e., steady-state response).

Understanding the interaction between surface water and groundwater is the key to accurately predicting the likelihood of success for this alternative. Numerous approaches have been used to investigate surface water

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and groundwater interaction ranging from analytical approximations (Cooper and Rorabaugh 1963; Jenkins 1968a; Moench and Barlow 2000) to combined analytical and numerical solutions (Leake et al. 2008a) to entirely numerical solutions (Pinder and Sauer 1971; Saquillace 1996; Chen and Chen 2003; Leake and Reeves 2008; Leake et al. 2008b) with the strengths and weaknesses of both approaches being discussed in the literature (Sharp 1977; Perkins and Koussis 1996). The methodology used in this study combines a steady-state numerical MODFLOW model (Harbaugh et al. 2000) implemented within Visual MODFLOW version 4.1 (Waterloo Hydrogeologic Inc. 2006) with the linear response theory first proposed by Morel-Seytoux (1975) to develop stream response functions.

The effects of a groundwater withdrawal propagate radially until they reach a boundary condition such as a no-flow or recharge boundary. As the effects reach a recharge boundary such as a hydraulically connected stream, the result is either a decrease in stream gain or an increase in stream loss. As noted by Jenkins (1968a), in infinite time (at steady state), the full aquifer withdrawal will be drawn from a hydraulically connected recharge boundary. In a large developed groundwater system, evaluation of the impacts of multiple groundwater stresses on stream reaches or aquifer water levels over long periods of time can be a complicated process. Response functions can be used to describe the spatial and temporal propagation of such impacts.

Response functions can be generated using either analytical techniques or a numerical model. Analytical techniques are typically subject to restrictive or simplifying assumptions (Cosgrove and Johnson 2004). For example, Jenkins (1968a) assumes a straight, fully penetrating stream in a homogeneous aquifer to determine response functions of stream depletion by wells. Generating response functions using a numerical groundwater model enables the representation of complex system heterogeneities and anisotropies. For example, Barlow et al. (2003) and Cosgrove and Johnson (2004) demonstrated the use of stream response functions within a groundwater flow model to study the local groundwater and surface-water interactions. The goal in both of these studies was to create stream response functions that would allow the impacts of groundwater pumping on surface-water flows to be quantified.

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 two 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 (Harbaugh et al. 2000) groundwater model for the Snake River Plain Aquifer, Idaho, for use with the MODRSP code (Maddock and Lacher 1991) 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 4-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.

Leake and Reeves (2008) describe a procedure and considerations for using groundwater flow models to construct maps that illustrate the distribution and timing of capture of natural discharge due to pumping. Three types of maps described in their paper include: (1) transient capture from all head-dependent flow boundaries, (2) transient capture from a particular head-dependent flow boundary, and (3) ultimate steady-state capture from a particular head-dependent flow boundary. Leake et al. (2008b) show spatial distributions of total change in inflow and outflow from withdrawal or injection for select times of interest for the Sierra Vista Subwatershed and the Sonora, Mexico, portion of the Upper San Pedro Basin in the USA. Maps of transient capture of discharge to all MODFLOW head-dependent flow boundaries were constructed. The mapped areal distributions show the effect of a single well in terms of the ratio of the change in boundary flow rate to rate of withdrawal or injection by the well. To the extent that the system responds linearly to groundwater withdrawal or injection, fractional responses in the mapped distributions can be used to quantify response for any withdrawal or injection rate.

The overall objective of this research was to determine the impacts of exchanging surface-water sources for groundwater wells on streamflows in Bertrand and Fishtrap Creeks. A regional-scale steady-state groundwater flow model, developed previously for the aquifer (Scibek and Allen 2005) was used. Because of its regional scale, the groundwater flow model contained insufficient localized information to accurately examine groundwater and surface-water interactions for specific stream reaches. Therefore, the model was refined locally using measured seepage 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. Stream response functions were then determined by sequentially adding groundwater extraction wells to the model at increasing distances from the stream to determine the net effect on the steady-state stream water budget. A steady-state model was used to facilitate the development of a decision support system implemented in STELLA (isee Systems 2007) for evaluating

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the impacts. Specifically, the simulated stream response functions were incorporated into a STELLA model that allows the user to simulate the effects on the instream flows of Bertrand and Fishtrap Creeks through exchanging a surface-water source for a single replacement groundwater well of the same withdrawal rate, without the need to run the groundwater flow model.

Background

The study site is situated within the Abbotsford-Sumas aquifer, which extends from southern British Columbia, Canada, southward into northern Washington, USA (Fig. 1). The aquifer is the largest aquifer in the region, covering an area of approximately 161 km^2, and is roughly bisected by the Canada-USA border. The aquifer is also highly productive, and provides water supply for nearly 10,000 people in the USA (towns of Sumas, Lynden, Ferndale, Everson and scattered agricultural establishments) and 100,000 in Canada, mostly in the city of Abbotsford, but also in the township of Langley (Mitchell et al. 2003). In the 1990s, the region was described as having population and industrial growth that was among the fastest in North America (Boyle et al. 1997). These development pressures are continuing and place strain on the water resources in the area, throughout both the Canada and the USA portions of the aquifer.

The Abbotsford-Sumas aquifer is located on a broad outwash plain, which is elevated above the adjacent river floodplains (Fig. 1); topographic relief is roughly 150 m. The uplands are centered on the city of Abbotsford, BC and extend westward through Langley, BC and south to Lynden, WA. The largest valley is Sumas Valley, which runs north-east to south-west from the city of Sumas to the city of Chilliwack (Fig. 1), and contains the lower drainage of the Sumas River. The Sumas River flows to the northeast and picks up a significant baseflow component from aquifer discharge on its eastern side. To the south is the Nooksack River, which flows west and then south. Most of the surface and groundwater flow from the Abbotsford-Sumas aquifer ends up in the Nooksack River. To the north is the Fraser River, which does not receive any significant discharge from this aquifer as it lies to the north of the topographic high and groundwater divide.

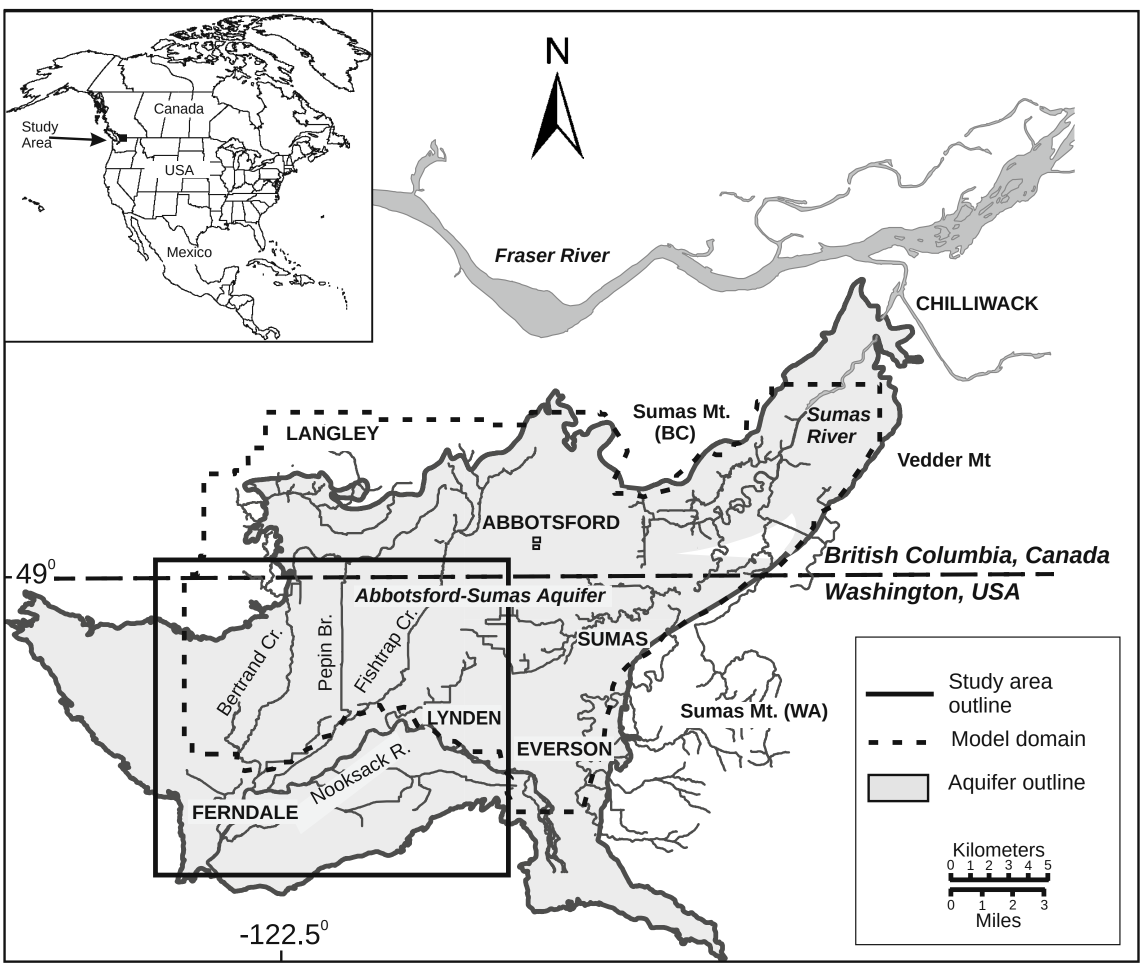

Fig. 1 Location of study area in southwestern British Columbia, Canada, and northwestern Washington State, USA. The international border is shown. The Abbotsford-Sumas aquifer is *shaded*, and the outline of the groundwater model domain is shown by a *dashed line*. The model domain encompasses the Fishtrap and Bertrand Creek watersheds, which drain to the *Nooksak River*

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The climate in this coastal region is humid and temperate, with warm, dry summers and mild, wet winters. Mean annual precipitation ranges from 1,000 to 2,100 mm (N–S gradient) and falls mostly as rain. Roughly 18% of precipitation falls during the months of June through September, and 82% during the months of October through May (McKenzie 2007). Recharge to the aquifer (900–1,100 mm) is primarily from direct precipitation (Scibek and Allen 2006).

The aquifer is mostly unconfined (it becomes confined under Sumas Valley) and is comprised of Quaternary age coarse-grained sediments of glaciofluvial drift origin, referred to as the Sumas Drift (11,000–10,000 BP) (Armstrong et al. 1965). The Sumas Drift consists of diamictons (lodgement and flow tills), thick and well-sorted glaciofluvial outwash (uncompacted) sands and gravels (advance and recessional), glaciolacustrine sediments, and ice-contact sediments deposited during the Sumas Stade of the Fraser Glaciation (Armstrong et al. 1965). It also contains lenses of till. The thickness of Sumas Drift can be up to 65 m, and it is thickest in the northeast where glacial terminal moraine deposits are found. The aquifer is underlain by an extensive glaciomarine deposit, the Fort Langley Formation/Everson, which outcrops in the uplands to the northwest (Langley Uplands) and southeast (Sumas Valley). The Tertiary age bedrock surface underlying the unconsolidated deposits is approximately 210 m below the ground surface (Scibek and Allen 2005). Soils over the aquifer are generally thin (approximately <0.7 m thick) with permeability that exceeds precipitation rates (Mitchell et al. 2003).

Recharge to the aquifer is predominantly by direct precipitation (Scibek and Allen 2005). The average annual variation in water levels in the aquifer is 2 m, with a maximum variation estimated by Scibek and Allen (2005) as 3 m. There is approximately a 3-month lag period between minimum precipitation and minimum water table levels. Groundwater flows regionally from north to south, although there are some local variations (Scibek and Allen 2005), particularly in the vicinity of the streams that drain the aquifer.

The research focuses on two main streams that drain the aquifer, namely Fishtrap Creek and Bertrand Creek. Situated between these two streams is Pepin Brook, which is a tributary to Fishtrap Creek south of the international border. Combined, the watersheds cover approximately 100 km^2. Approximately 46% of the Bertrand watershed (50.2 km^2) and 39% of the Fishtrap watershed (37.3 km^2) are within the USA, with the remaining areas extending into Canada. Bertrand Creek is a naturally formed, meandering stream, whereas Fishtrap Creek has been channelized in many places to accommodate agriculture and to reduce flooding. The predominant land use within both countries for both watersheds is agricultural.

Fishtrap and Bertrand Creeks originate at relatively low elevation (slightly above mean sea level) and the flow regime is driven by precipitation and interaction with the groundwater (Berg and Allen 2007). The flow regime mimics the timing of the precipitation, with a time lag of approximately 1 month. Peak flow occurs between October and May, corresponding to the period of highest precipitation. Minimum precipitation occurs in July and August, and the lowest streamflows, or even dry conditions, occur during August, shortly after the minimum precipitation (Berg and Allen 2007).

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 to collect concurrent streamflow, 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. Cox et al. (2005) carried out a groundwater and surface-water interaction study using a network of nine in-stream piezometers 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 lower during the summer and early fall dry season (June to September). Vertical hydraulic gradients were generally upward, indicating discharging groundwater, except for one piezometer located within the town of Lynden where consistently negative gradients indicated that the streamflow was recharging the groundwater (Cox et al. 2005).

Culhane (1993) calculated theoretical stream depletion rates expected under various pumping scenarios within the Abbotsford-Sumas aquifer using the Jenkins analytical model (Jenkins 1968a,b). While the goal of that study was to determine a critical distance for separating wells from nearby streams to minimize stream depletion, a single critical distance was not found to be scientifically defensible due to the isotropic and homogeneous constraints of the Jenkins model and a variety of other assumptions necessary for the analytical solution, e.g., the pumping wells are not commonly open to the full saturated thickness. For these reasons, a numerical approach was adopted in this study.

Materials and methods

Field investigations

The field investigations included (1) flow measurements for both Bertrand and Fishtrap Creeks and their tributaries, (2) streambed hydraulic conductivity measurements for both Bertrand and Fishtrap Creeks, and (3) monitoring of static groundwater levels in selected wells near each stream and stages of each stream.

Streamflow measurements

Streamflow measurements 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 cross-sectional area were used to estimate discharge. The locations of the flow measurements are shown in

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Fig. 2 and the data are summarized in Table 1. Net stream loss (or gain) was estimated by taking the difference in flow between stations. Kilometer markers are shown on Table 1 relative to the confluence of each stream with the Nooksak River (zero km). Based on the hydrogeology of the area, it is expected that streamflow would increase down-gradient due to discharge of groundwater into the streams. However, as evident in column 3 of Table 1, stream discharge decreases at some stations relative to the nearest up-gradient station. To correct for surface-water use along the different stream segments, and thus obtain stream discharge values that could be compared to the steady-state model, the locations and quantities of surface-water rights were obtained. These data 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 use for Bertrand and Fishtrap creeks. The estimates were added to the measured field values to obtain "corrected" flows for Bertrand Creek (Table 1). Observation as well as analysis of the water right database suggested that there was minimal surface-water use for Fishtrap Creek at the time the flow measurements were made and, as a result, flow values were unchanged from the field measurements (grey shading in Table 1).

Estimated gains (or losses) in stream discharge from (or to) groundwater (column 6 in Table 1) were estimated for each site by subtracting the corrected discharge at the nearest up-gradient measurement site from the corrected value at that site (column 5 values in Table 1). In some cases (e.g., F-3), this resulted in a negative value, suggesting a loss of streamflow. Upon accounting for the surface-water use in Bertrand Creek, all locations were found to be gaining water from the aquifer. Fishtrap Creek was found to be primarily gaining; exceptions included sites F-3, F-7 and F-8. These losses seem to be consistent with the results of the Cox et al. (2005) study. Because Fishtrap Creek had higher streamflows than Bertrand Creek, accounting for small individual surface-water use would have made minimal changes in streamflow.

Streambed hydraulic conductivity

Slug tests were conducted in July 2006 to estimate hydraulic conductivities of the streambed sediments (Fig. 3). A detailed description of the slug tests methodology and results can be found in McKenzie (2007). Measurements were taken at 0.5-m and 1.0-m depths below the streambed at two stations per sample location. The hydraulic conductivities derived from these tests (Table 2) were used to estimate the conductance values needed to represent as input to the River boundary conditions in the groundwater model (as discussed later).

It is noted that there is likely significant heterogeneity of the streambed sediments owing to the heterogeneity of the surficial sediments throughout the study region. The slug test values are likely very site-specific, and indeed a slug testing method may be of too small a scale to adequately characterize the streambed materials and provide reasonable estimates of the streambed conduc-

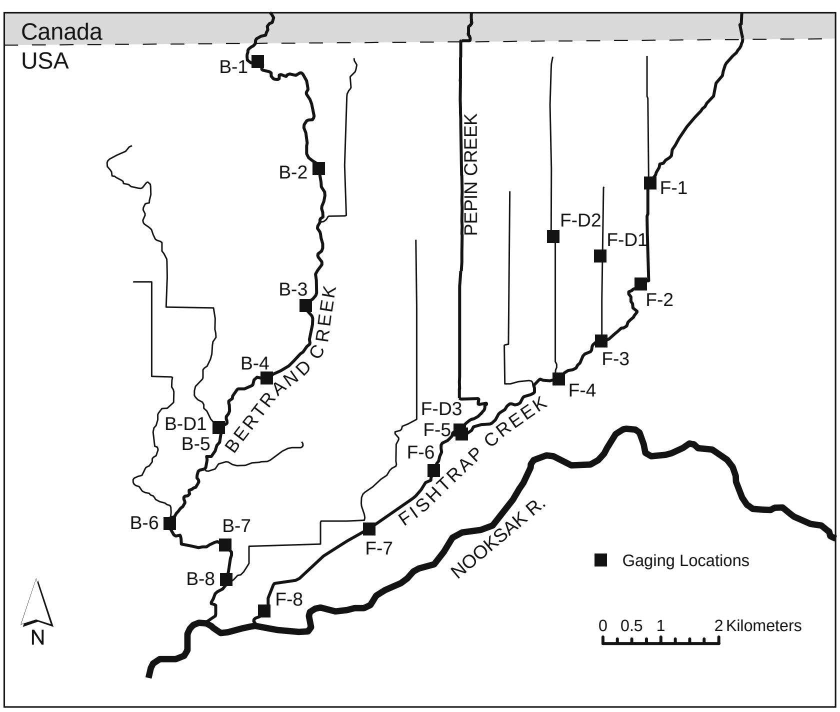

Fig. 2 Location of streamflow gaging sites

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Table 1 Estimated discharge for Bertrand and Fishtrap Creeks during July 2006 Flow Analysis

Site	River marker (km from confluence with Nooksak River)	Measured discharge (l/s)	Estimated surface-water use (l/s)	Discharge corrected for cumulative use (l/s)	Estimated gain in streamflow due to groundwater seepage relative to up-gradient site (l/s)[a]
Bertrand Creek sites					
B-1	13.9	23.5	0.0	23.5	-
B-2	10.8	22.4	23.2	45.6	22.1
B-3	8.2	71.6	16.7	111.5	65.9
B-4	6.5	121.5	0.0	161.4	49.9
B-5	5.1	91.2	35.1	166.2	4.8
B-D1	-	2.5	39.3	41.8	-
B-6	2.9	38.8	140.5	179.3	13.1[b]
B-7	1.7	11.6	63.4	329.8	150.5
B-8	1.0	19.8	5.7	343.7	13.9
Fishtrap Creek sites					
F-1	12.0	120.9			-
F-2	10.1	139.9			17.0
F-3	8.6	110.4			−28.4
F-D1	-	14.2			-
F-4	7.5	141.6			31.2[b]
F-D2	-	8.5			-
F-5	5.3	181.2			39.6[b]
F-D3	-	36.8			-
F-6	4.4	277.5			96.3[b]
F-7	2.8	263.3			−14.2
F-8	0.4	257.7			−5.6

[a] Negative numbers indicate a loss in streamflow from the up-gradient site

[b] Additional water at confluences with tributaries is not considered. The net loss or gain is calculated relative to the up-gradient site without tributary contributions

tance needed for a larger scale model. Ideally, a variety of techniques could be used in combination to estimate these values, including (as done here) slug tests, seepage measurements, grain size analysis, and estimation through model calibration. Our approach was to use the slug test derived results and verify (or modify) these through model calibration.

Groundwater and surface water monitoring sites

Static groundwater elevations were monitored hourly in six wells (Fig. 3) using Onset Hobo Water Level Logger pressure transducers. These values were used to calibrate the groundwater flow model along with historical well record data. Two surface water sites were chosen for installation of the pressure transducers: one in Bertrand

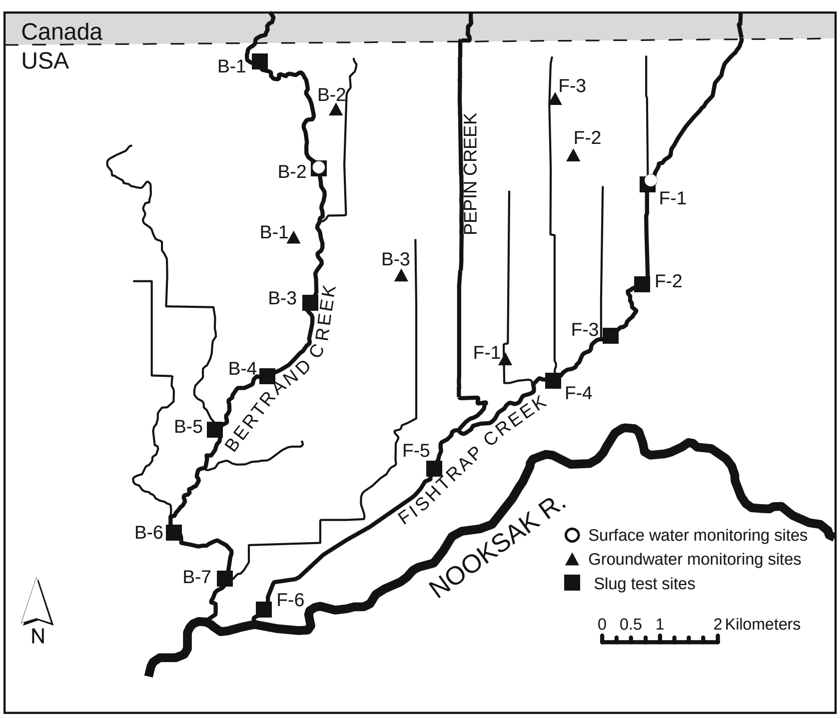

Fig. 3 Location of slug test sites, and groundwater and surface-water monitoring sites

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Table 2 Input parameters for the River boundary condition for each river section. Hydraulic conductivity values were derived from slug tests. See Fig. 3 for location of sites

Section Name	Hydraulic conductivity K from slug tests (m/day)	Stream width (m)	Stream length (m)	Sediment thickness (m)	Conductance (m^2/day)
B-1	7.50E-01	4.5	75	0.75	3.4E+02
B-2	1.42E+02	3.3	100	1.00	4.7E+04
B-3	6.08E+01	3.7	100	1.00	2.3E+04
B-4	1.67E+01	3.2	100	1.00	5.4E+03
B-5	5.80E+01	3.0	100	0.75	2.3E+04
B-6	5.76E+01	3.0	100	1.00	1.7E+04
B-7	7.88E+00	4.3	100	0.75	4.5E+03
F-1	1.70E+01	5.5	50	1.00	4.7E+03
F-2	6.30E+00	4.3	75	0.75	2.7E+03
F-3	4.78E+00	4.5	75	1.00	1.6E+03
F-4	3.03E+01	6.7	100	1.00	2.0E+04
F-5	8.94E+01	4.8	100	1.00	4.3E+04
F-6	1.16E+01	4.0	100	1.00	4.6E+03

Creek (B-2) and the other in Fishtrap Creek (F-1; Fig. 3). Surface-water levels in each stream were monitored hourly 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 (data reported in Pruneda 2007).

Groundwater modeling

The regional steady-state groundwater model, developed originally by Scibek and Allen (2005), was used in two previous studies of the Abbotsford-Sumas aquifer: (1) to identify the potential impacts of climate change on groundwater (Scibek and Allen 2006) and (2) to simulate nitrate transport within the aquifer (Allen et al. 2007). Visual MODFLOW version 3.0 was used (Waterloo Hydrogeologic Inc. 2000). The following sections provide an overview of the original model, highlighting what changes were made to the boundary conditions in this study.

Geological framework

The lithostratigraphy for the region was mapped using over 2,000 well lithologic logs in combination with surficial geology maps and depositional models (Scibek and Allen 2005). Hydrostratigraphic units were defined based on the lithostratigraphy and available estimates of hydraulic properties from pumping tests; data were related specifically to the aquifer material encountered at the well screen (Cox and Kahle 1999). Four main hydrostratigraphic units were mapped: Sumas Drift (sandy), Sumas Drift (gravelly), silt (mostly silt), and clay or till (mostly Ft. Langley formation and similar). Mean values of hydraulic conductivity K (based on the geometric mean) were: 105 ± 4 m/day for the Sumas Drift (gravelly), 57 ± 4 m/day for Sumas Drift (sandy), 51 ± 2 m/day for silt, and 19 ± 3 m/day for clay/till. K was surprisingly high in both Sumas Drift (sandy) and silty units, and somewhat high for clay/till unit relative to what might be expected for this material type, suggesting that there may be pockets or lenses of highly permeable material within an overall less permeable unit. This heterogeneity can be expected to result in complex groundwater paths at both regional and local scales. Ten layers were used to represent the aquifer, each of which was comprised of varying hydraulic conductivity zones based on the hydrostratigraphic units. The various units were then assigned representative hydraulic conductivity, porosity, and storativity values, which were later adjusted slightly during model calibration (Scibek and Allen 2005). In order to calibrate the model, Scibek and Allen (2005) incorporated pockets of flow till in the Sumas Drift unit in the Abbotsford uplands. These flow tills are observed on surficial geology maps, but had not been included in the original conceptual geological model described above. With these materials included, the model could predict the existence of kettle lakes at the observed elevations, whereas the original model (with just a sand or gravel Sumas Drift unit) could not, and the water table was greatly underestimated compared to observed in the uplands.

Model domain

The lower model boundary corresponds to the bedrock surface, which is assumed impermeable relative to the overlying sediments. The bedrock surface was mapped using deep borehole data, existing bedrock contour maps (Hamilton and Ricketts 1994), valley wall profiles, and extrapolated cross-sections through the study area (Scibek and Allen 2005). The model domain at surface extends slightly beyond the Abbotsford-Sumas aquifer, as illustrated in Fig. 1, in order to adequately capture the physical and hydrologic features that can serve as appropriate model boundaries. These include regional surface water divides to the west and north, and the bedrock outcrops to the east. Surface water divides are thought to approximate the regional groundwater divides as the aquifer is largely unconfined.

Surface-water boundary conditions

Boundary conditions related to surface-water features in the original model (Scibek and Allen 2005) included specified heads and drains corresponding, respectively, to the major rivers (i.e., the Nooksack and the Sumas

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Rivers) and small lakes, and the numerous streams that drain the aquifer. Values for specified-head features were determined using a combination of survey data and topographic information as described by Scibek and Allen (2005). Drain conductance values were assigned a uniform value of 100 m^2/day, due to a lack of measured values.

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 River package in MODFLOW simulates the interaction between groundwater and surface water via a seepage layer, with each cell modeled as a river reach assigned a user-specified conductance term defined as (Harbaugh et al. 2000):

$$C = \frac{K \cdot L \cdot W}{B} \tag{1}$$

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

Each stream was divided into sections or groups: seven in Bertrand Creek, six in Fishtrap Creek, and one in Pepin Creek. Each section was assigned a conductance value based on the nearest measurement of streambed hydraulic conductivity (Table 3), the physical properties of the stream as described in Eq. 1, and seepage analysis data. The widths (W) of river reaches were assumed to be constant for each river group and were based on the nearest streamflow measurement. The lengths (L) of river reaches were approximated per river group as the average of the model cell height and width within each group. The thickness of the stream sediments (B) was assumed to be 1.0 m, the maximum depth of the 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, because it was assumed that the deeper slug test penetrated the aquifer media beneath the streambed and, thus, was measuring aquifer hydraulic conductivity as opposed to stream bed conductivity. 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 stream.

Within MODFLOW, water seepage to or from the stream is determined at each computational iteration. Depending on the bottom elevation of the seepage layer, either Eq. 2 or Eq. 3 is used (Harbaugh et al. 2000):

$$Q_{RIV} = C \cdot (H_{RIV} - h), \quad h > RBOT \tag{2}$$

$$Q_{RIV} = C \cdot (H_{RIV} - RBOT), \quad h < RBOT \tag{3}$$

where Q_{RIV} is the flow between stream and aquifer (m^3/day), C is the hydraulic conductance of seepage layer from Eq. 1 (m^2/day), H_{RIV} is the head in the stream (m), h is the head in the grid cell (m), and RBOT is the bottom elevation of the seepage layer (m). Use of Eq. 2 leads to groundwater discharge to the stream, and Eq. 3 leads to steamflow leakage to the aquifer. Heads in streams (H_{RIV}) were set equal to the surveyed values under August low-flow conditions. This condition effectively eliminates stormflow effects as precipitation is generally very low during the late summer and most (if not all) of streamflow derives from groundwater. Consequently, the model represents the average annual groundwater state. RBOT assumes a 1-m streambed thickness as used for the calculation of conductance.

The Nooksack River at the south end of the study area was modeled as a specified-head boundary condition using observed stage data from two USGS gaging stations on the Nooksack River, one upstream at North Cedarville, WA and one downstream at Ferndale, WA. Small creeks and ditches were modeled as drain boundary conditions. Drains surrounding Bertrand and Fishtrap Creeks were given conductance values similar to those found in the nearest Bertrand or Fishtrap Creek river section. 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.

Recharge

Recharge was modeled separately using the HELP software developed by the US Environmental Protection

Table 3 Bertrand and Fishtrap Creek flow responses individually and collectively 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 (%)	Combined flow response for wells within Bertrand Creek Watershed (%)	Combined flow response for wells within Fishtrap Creek Watershed (%)
0.0–0.1	39.0	23.0	20.2	22.0
0.1–0.2	16.7	27.4	14.5	27.6
0.2–0.3	9.8	15.1	12.0	15.7
0.3–0.4	6.9	8.1	11.1	8.2
0.4–0.5	6.3	5.3	9.0	5.1
0.5–0.6	6.7	5.4	9.4	5.5
0.6–0.7	5.0	4.9	7.8	5.0
0.7–0.8	3.9	4.1	7.0	4.0
0.8–0.9	3.7	4.3	5.7	4.4
0.9–1.0	2.1	2.3	3.5	2.5

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Agency (Schroeder et al. 1994). For this reason, recharge was not considered a calibration variable. HELP solves a series of soil–water balance equations for a layered column of material using a time series of meteorological data as input to the top of the model. HELP accounts for the effects of surface storage, runoff, evapotranspiration, snowmelt, infiltration, vegetation growth, soil moisture storage, lateral subsurface drainage, unsaturated vertical drainage, and leakage through soil. Detailed descriptions of all inputs and equations can be found in the supporting documentation for HELP (Schroeder et al. 1994). Slope can be incorporated (quasi two-dimensional); however, the simulation is one-dimensional. For recharge simulations, the base of the column is set equal to the depth of the water table, and leakage simulated through the bottom of the soil column is considered representative of direct recharge to the groundwater system. To account for spatial variability, recharge was simulated for different recharge zones (Scibek and Allen 2006); each zone represented unique combinations of soil media type, shallow aquifer permeability, and depth to water table. In total, 64 different recharge zones were modeled. Recharge was then applied to the top active layer of the groundwater flow model.

To assure that the average annual recharge values in the model (based on 30-year historical climate data) are representative of 2006 field data, annual precipitation for 2006 at Clearbrook, WA (approximately 14 km from the study site; National Climate Data Centre 2007a) was compared to the normal precipitation observed since the year 1919 (National Climate Data Centre 2007b). For 2006, a total of 1,139 mm of rain was recorded, which amounted to only 23 mm less than normal. This departure from normal was insignificant, and therefore no attempt was made to adjust the recharge values previously defined by Scibek and Allen (2006).

Pumping wells

The original model by Scibek and Allen (2005) included only 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 l/s. According to Wubbena et al. (2004), approximately 3,000 ha within the Bertrand WID require approximately 1,379 l/s of groundwater during the month of July for irrigation purposes. Similar demands were assumed to be applicable in August.

To determine pumping rates, 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 estimated amount of pumping did not match the sum of the permitted water rights, all water rights were scaled equally to match the estimated groundwater irrigation use for the Bertrand Watershed Improvement District as determined by Hydrologic Services Company (Wubbena et al. 2004).

Model calibration

In addition to the more than 1,000 existing observation wells input in the original model by Scibek and Allen (2006), six local monitoring wells (Fig. 3), along with a number of USGS wells, were added to the groundwater model to serve as head observation points. For the entire model domain, the calibration of observed to measured static water levels 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 of Scibek and Allen (2005), which were considered to be reasonable given the large scale of the model and limited number of observations. Moreover, the spatial distribution of residuals was generally good, although there were pockets where simulated heads were all higher or all lower than observed values.

Zone Budget (Harbaugh 1990) was used in Visual MODFLOW to calculate sub-regional water budgets for different zones in the model and to verify the calibration. A total of 20 zones were created between locations of measured streamflow: eight in Bertrand Creek, eight in Fishtrap Creek, one in Pepin Creek, and three for major drains. Only cells that were defined as river or drain boundaries were included in a zone. For each sub-regional water budget zone, the cell-by-cell budget results were tabulated. 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 to obtain a "corrected" flow (Table 1 values) and compared to our measured flow values to determine the accuracy of the model. It is noted that the Stream package within Visual MODFLOW would have accounted for the flows automatically, but the River package was chosen in order to preserve the surface water head values in the original model.

Locally, the calibration results pointed to some discrepancies between "corrected" and modeled streamflows. A comparison of the "corrected" and modeled flows for Bertrand Creek (Fig. 4) and measured[1] and modeled flows for Fishtrap Creek (Fig. 4) revealed that the model over-predicts streamflow in the area of site B-2, and slightly over-predicts streamflow in the upper reaches of Fishtrap Creek; however, the model closely matches the "corrected" flows in the lower reaches of Bertrand Creek and the measured flows of Fishtrap Creek. A comparison of the observed and modeled hydraulic heads within the local study area (in the aquifer nearby Bertrand and Fishtrap Creeks) yielded better statistics than the overall regional 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.

[1] Fishtrap Creek flows were not corrected as discussed previously.

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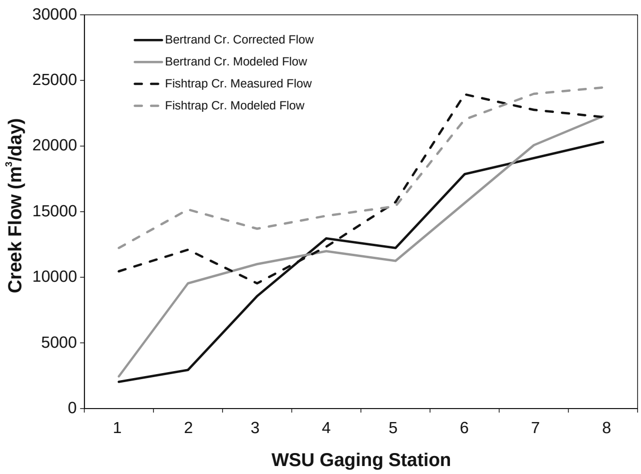

Fig. 4 A comparison of corrected and modeled streamflows for Bertrand and Fishtrap Creeks

Response functions

Response functions were manually created for each of 346 hypothetical well locations (Fig. 5). Pumping wells were added to the calibrated steady-state groundwater flow model, one at a time, and the streamflow impacts were obtained 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 28.3 l/s—this is equivalent to 1 cubic foot per second (cfs).

For each well location, a response ratio ranging from 0.0 (no impact on stream) to 1.0 (completely taken from stream) was determined for Bertrand Creek and Fishtrap Creek as the change in modeled streamflow 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 the rate of streamflow depletion at each constraint site was 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. Since a steady-state groundwater model was used, the streamflow responses represent a worst-case scenario, because the zones of influence of the pumping wells are at a maximum under steady-state conditions.

Raster maps of the response ratios with a 100-m cell size were created for each stream using natural neighbor interpolation in ArcGIS (ESRI 2007). 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).

Groundwater-surface water interaction tool

The mapped response zones were used to create a groundwater and surface-water interaction tool, whereby the user can replace surface-water use with a groundwater pumping well of the same withdrawal rate, and determine the streamflow impact for Bertrand and Fishtrap Creeks at their terminus with the Nooksack River. STELLA version 9.0.3 by isee Systems (2007) was chosen as the modeling environment for the interaction tool. The users can choose between four regions of interest within the study area, and then select one of four sub-regions within that chosen region (Fig. 6). Upon choosing a sub-region, the user can easily locate the location of a surface-water intake, 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 identifies the Bertrand Creek response zone and the Fishtrap Creek response zone for which the desired replacement groundwater well is located, and enters a value for the surface-water withdrawal rate to be replaced by the groundwater well. Using the response functions

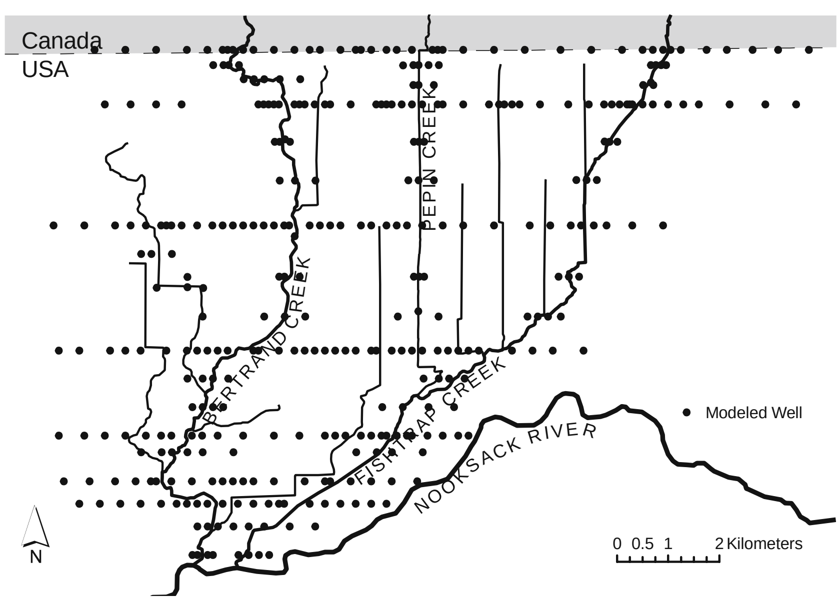

Fig. 5 Modeled well locations for determination of response ratios

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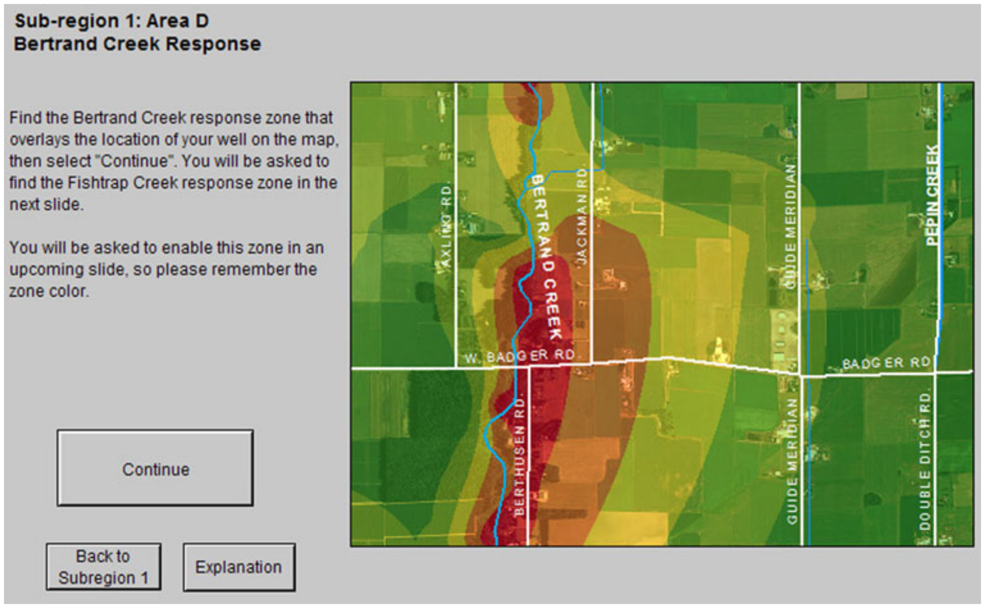

Fig. 6 Screen shot of the STELLA interactive tool. *Colors* represent ranges of response functions: *red* (0.8–1.0), *orange* (0.6–0.8), *yellow* (0.4–0.6), *light green* (0.2–0.4), *green* (0–0.2)

and the provided user input, the STELLA model compares the streamflow values for Bertrand and Fishtrap creeks for each of a surface-water replacement well and a surface-water use. The impact on 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.

Results

Maps of the response functions for each of Bertrand Creek and Fishtrap Creek are shown in Figs. 7 and 8, respectively, based on an independent modeling assessment of each. Response functions for each well location ranged from 0 to 1.0; these were contoured and classed into five categories. Table 3 presents the Bertrand and Fishtrap Creek flow responses, respectively, as the percent area of their corresponding watershed. Seventy-nine percent of the Bertrand Creek watershed and 79% of the Fishtrap Creek watershed have response ratios less than 0.5.

Because a steady-state model was used to generate the response functions, it is important to consider what the other sources of water are to each pumping well.

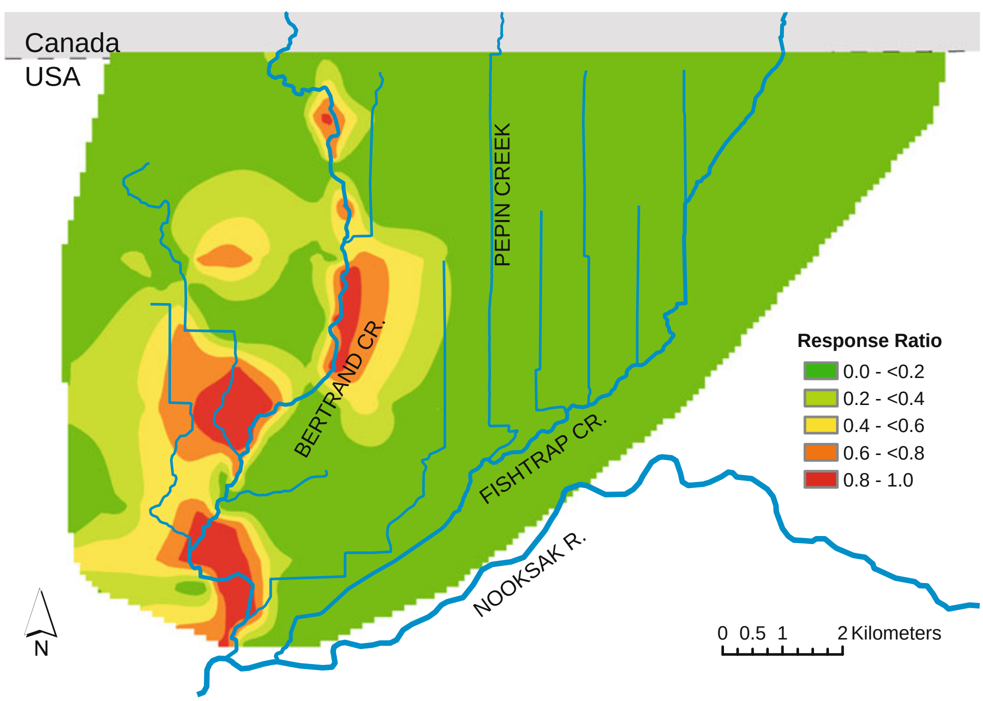

Fig. 7 Raster map of Bertrand Creek response ratios

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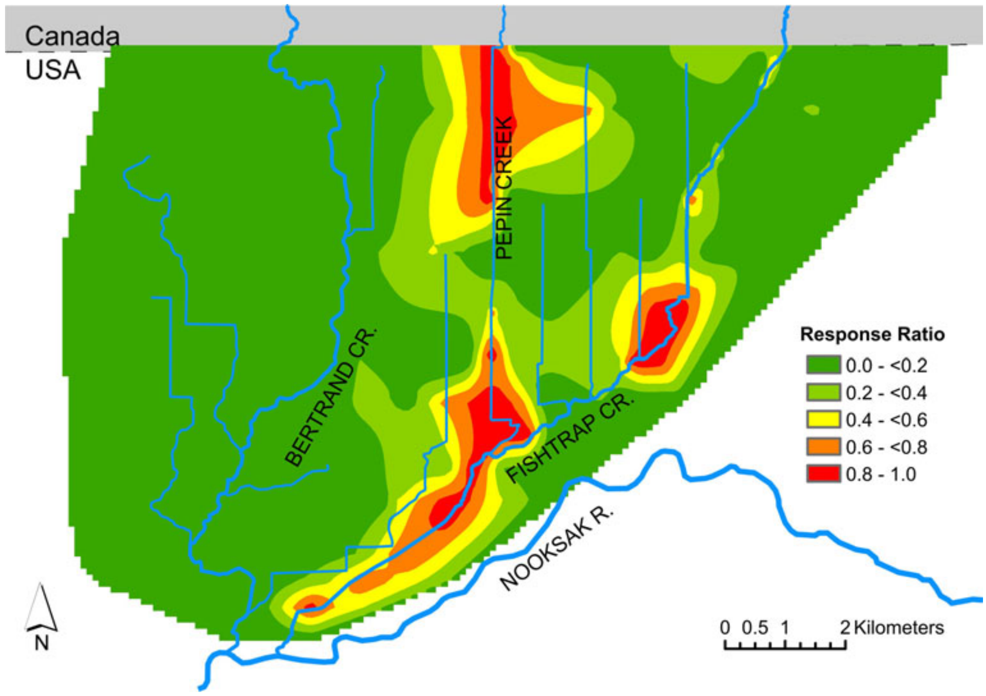

Fig. 8 Raster map of Fishtrap Creek response ratios

There are four potential sources of water to each pumping well simulated: (1) streamflow-from one of the three main streams simulated as River boundaries, (2) seepage from the drain cells used to represent the smaller tributary streams, (3) seepage from the Nooksak River, which was defined as a specified head boundary condition, and (4) recharge applied to the top surface of the model (mean annual recharge). As noted previously, during the low-flow period, most drains cells are not in contact with the groundwater table because their bed elevations are above the groundwater table under August conditions. Therefore, generally, the small streams do not act as a source of water to the wells. For wells in proximity to the Nooksak River, some of the water may derive from this source, but as the impact on the main streams was of interest, this source was not considered. Therefore, apart from water derived from the major streams, the only other source is recharge. The calculated response functions are non-uniformly distributed as shown in Figs. 7 and 8. For areas with low response ratios such as the lower area between Bertrand and Fishtrap Creeks just above the confluence with the Nooksak, the wells are far enough from either creek to draw water from them, and recharge is the source. For wells near the Nooksak River below Fishtrap Creek, some water derives from that river.

The results suggest that pumping wells placed east of Fishtrap Creek essentially have no discernable impact on Bertrand Creek. Similarly, pumping wells placed west of Bertrand Creek have almost no discernable impact on Fishtrap Creek. Because groundwater movement occurs across the watershed boundaries, groundwater pumping wells located in the area between Bertrand and Fishtrap Creeks can impact streamflows in both creeks. Also, there are reaches where groundwater pumping has less impact on the streamflow on one side of the creek as opposed to the other. From a practical perspective, a surface-water replacement well should not be allowed to benefit one creek while harming 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 (Fig. 9). Table 3 also shows the combined flow response for Bertrand and Fishtrap Creeks as the percent area of their corresponding watershed. Sixty-seven percent of the Bertrand Creek watershed and 79% of the Fishtrap Creek watershed have combined response ratios less than 0.5, indicating highly favorable exchange opportunities.

Despite the favorable conditions for replacement of surface water use by groundwater pumping wells in over 77% of the study area, internal testing of the STELLA interface suggested that it might not be economically practical for farmers to replace their surface-water source for a groundwater withdrawal if they have to construct a lengthy pipeline to transport water to their field. Consequently, streamflow responses from wells located within narrow bands of both streams were specifically examined. Of the area within a 0.8-km band of Bertrand Creek, 57% has a combined flow response ratio less than 0.5, and within a 1.6-km band, 64% has a combined flow response ratio less than 0.5 (Table 4). Of the area within a 0.8-km band of Fishtrap Creek, 70% has a combined flow response ratio less than 0.5, and within a 1.6-km band, 77% has a combined flow response ratio less than 0.5 (Table 4).

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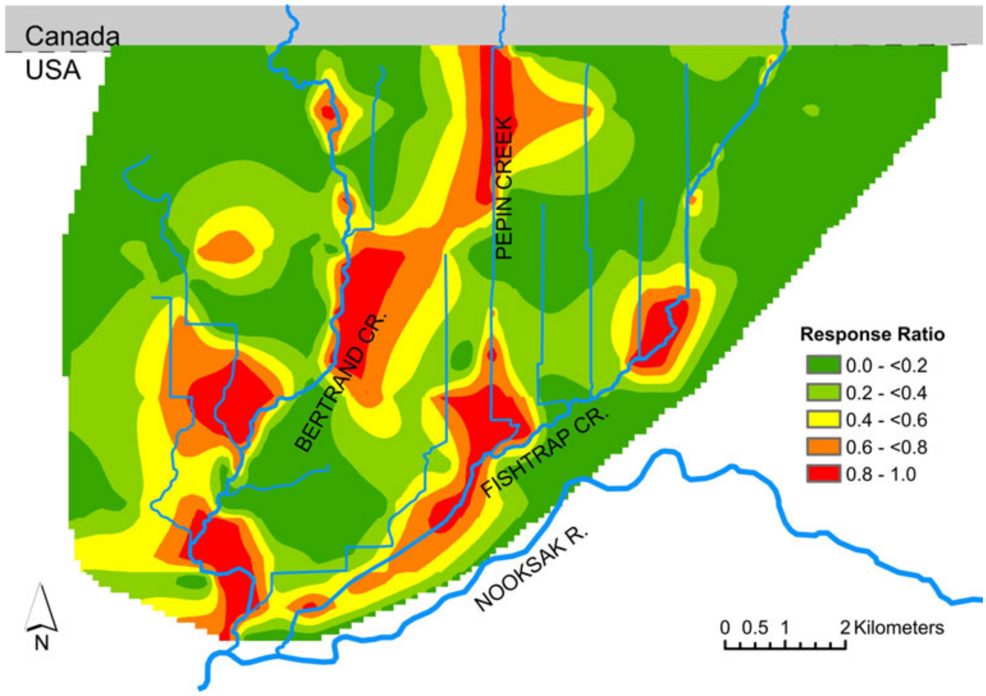

Fig. 9 Combined raster interpolation of Bertrand and Fishtrap Creek response ratios using a natural neighbor technique

Discussion

Simulation results suggested that replacing surface-water sources with groundwater pumping wells may be a viable alternative for improving summer streamflows. It is clear that pumping wells do impact Bertrand and Fishtrap Creek flows, but if placed within zones of a low response ratio, less impact would occur than removing an equivalent amount of water directly from the stream.

For each stream (Figs. 7 and 8) and for the combined stream network (Fig. 9), areas where the response ratio is high (i.e., above 0.6) are proximal to the stream. However, the response ratios are not uniform along the stream length as might be expected if they were solely a function of distance to the stream and the pumping rate. Rather, 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. Thus, surface-water replacement wells placed within zones with high hydraulic conductivity values will likely produce greater responses to the instream flows of Bertrand and Fishtrap creeks.

The fact that the response ratios are not uniform along the stream length lends support for the use of a numerical groundwater flow model over an analytical model for this type of analysis. Analytical models cannot adequately capture the heterogeneity of the aquifer materials nor the range of stream bed conductance values and stream physical properties. Numerical models generally have this ability provided there is sufficient information with which to construct the model.

While the numerical groundwater flow model used in this study was constructed at the regional scale, it does capture a reasonable amount of heterogeneity in both spatial recharge and geology (Scibek and Allen 2005) as

Table 4 Bertrand and Fishtrap Creek combined flow response as the percent area within 0.8 km and 1.6 km of Bertrand and Fishtrap Creeks

Response zone	Combined flow response for wells within 0.8 km of Bertrand Creek (%)	Combined flow response for wells within 1.6 km of Bertrand Creek (%)	Combined flow response for wells within 0.8 km of Fishtrap Creek (%)	Combined flow Response for wells within 1.6 km of Fishtrap Creek (%)
0.0–0.1	17.9	17.9	19.5	21.1
0.1–0.2	14.1	14.9	19.4	24.4
0.2–0.3	11.4	12.4	13.2	15.1
0.3–0.4	7.4	10.0	10.3	10.2
0.4–0.5	6.1	8.6	8.0	6.0
0.5–0.6	9.2	10.2	8.4	5.8
0.6–0.7	7.7	7.7	7.4	5.3
0.7–0.8	7.6	7.2	5.5	4.5
0.8–0.9	10.3	6.5	4.7	4.2
0.9–1.0	8.4	4.5	3.8	3.4

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evidenced by the good calibration results both for stream flow and hydraulic head in the aquifer. While the additional field data incorporated into the original groundwater flow model provided improved local detail, calibration results suggested that additional research and data collection could be used to further improve model calibration locally. Specifically, it is suggested that the overestimation of river leakage in the upper reaches of both creeks may be due to non-permitted wells that were unaccounted for, uncertainty in stream elevations, lower hydraulic conductivity of the aquifer materials in those areas, or a combination of these factors.

Finally, there are three main limitations to this study. First, 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 crucial and is currently lacking. This situation is not uncommon in most watersheds and points to the need for ongoing accounting of water use. Second, transient effects were not evaluated in this study despite the fact that a transient model has been developed and used for climate change impacts assessment (Scibek and Allen 2006). A transient groundwater model would have provided greater information on lag times between pumping and stream impacts; however, this route was not taken partly due to the paucity transient calibration data-additional long-term monitoring wells are needed to improve on transient model calibration-and partly because it would be difficult to implement transient response functions into a STELLA model. Our choice of a steady-state model essentially provides a maximum impact of pumping wells at a time of the year when streamflow would be most impacted.

Third, and perhaps most importantly, a groundwater flow code was used rather than a coupled groundwater-surface water code, such as MIKE SHE (DHI 2009). Such codes, although highly parameterized, have the potential to simulate the exchange of water between the surface water system and the groundwater system more accurately than a groundwater flow code. In MODFLOW, the river or specified head boundary conditions, as were used in the model for this study, assume that the head in the stream will remain at some specified level for the duration of the simulation, regardless of how much water is extracted from the surrounding aquifer. Clearly, this could represent a significant limitation to determining stream response functions, particularly under low-flow conditions. However, because simulations represented the replacement of surface water use with groundwater, there would be no change in the stream width or depth (i.e., there is no net change in streamflow). However, if stream response functions are used to assess new groundwater abstraction, then if the pumping rates are relatively low and the effect of individual wells pumping does not lower the stream stage appreciably, the approach is reasonable. Where the approach will begin to fall apart is when the cumulative effects of pumping are considered and where these cumulative effects result in a lowering of the stream stage. This, of course, is more likely the real situation and one that demands a more rigorous coupled surface-water/groundwater model.

Finally, the STELLA interface was not rigorously tested in this study. This interface was developed specifically for Whatcom County as a means to assess what the potential impact on streamflow would be if surface-water sources were replaced with equivalent groundwater extraction wells. Nonetheless, decision support systems for water managers clearly offer a means to make informed decisions without the need for expert knowledge. For problems involving groundwater and surface water, such tools have the potential to be very valuable if the supporting model outcomes have themselves been reasonably determined through scientific methods.

Conclusions

Groundwater and surface-water interactions are prominent within the Bertrand and Fishtrap Creek watersheds based on measured responses of streamflow and groundwater levels as well as modeling results. Summer low flows in these streams are currently at levels to jeopardize endangered and threatened fish habitat. Hence, an innovative conjunctive management scheme is needed.

This study investigated the replacement of surface-water sources with groundwater withdrawals using a numerical groundwater flow model. Response ratios, calculated from the modeled change in streamflow divided by the pumping rate, were used to assess the impact on streamflow of exchanging a surface-water source with a groundwater pumping well, based on the groundwater flow model under steady state. Resulting response ratios ranged from 0 to 1, with 0 representing no impact on the stream and 1 representing an impact equivalent to that of a surface-water withdrawal at 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. For areas with low response functions, the balance of water derived from precipitation recharge or, for wells near the Nooksak River, from that river source.

Simulation results suggest that replacing surface-water sources with groundwater pumping wells may be a viable alternative for improving summer streamflows. It is clear that pumping wells do impact Bertrand and Fishtrap Creek flows, but if placed within zones of a low response ratio, less impact would occur than removing an equivalent amount of water directly from the stream. Within a 1.6-km distance on either side of the stream, 64% of Bertrand Creek had combined response ratios less than 0.5, while within the same distance, 77% of Fishtrap Creek had combined response ratios less than 0.5, indicating highly favorable exchange opportunities for both creeks.

Because MODFLOW is difficult to understand and operate for non-specialists, response functions were created and, by using the STELLA software, a user-friendly 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 streamflow impacts on

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Bertrand and Fishtrap Creeks without the need to re-run the groundwater flow model.

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