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Hydrological Evolution of a Small Watershed Over Nearly a Century of Pumping

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ABSTRACT

Understanding how watersheds evolve and change hydrologically under decades of pumping can help inform sustainable watershed management. This study involved simulating the hydrological evolution of Bertrand Creek Watershed (BCW) in the Lower Fraser Valley in British Columbia, Canada, over nearly a century of groundwater pumping to assess the long-term impacts of steadily increasing pumping on subsurface storage, aquifer-river exchanges and streamflow. A non-pumping simulation was used to isolate pumping effects from natural climate variability. The results reveal that changes in subsurface storage in BCW were driven primarily by climate (water availability), with pumping exerting a secondary but detectable influence. Droughts and high pumping rates interacted, producing storage losses up to 670% more than under normal conditions, while storage gains were rare, occurring only in very wet years and reaching up to 270% more than normal. Across the entire watershed, groundwater discharge and overland flow to the river declined by an average of 6% and 8%, respectively, under pumping conditions. These reductions caused simulated streamflow at two stream gauge locations to decrease by about 11% and 4% per year, respectively, consistent with observed streamflow depletion. River to groundwater exchange is low and remained the same in both simulations. A critical shift in the hydrological regime occurred in the 1970s, when multiple large-capacity municipal wells became operational. In recent years, these wells diverted nearly 4700 m^3/day of water that otherwise would have discharged to the stream and accounted for 6%–7% of the simulated streamflow depletion.

1 | Introduction

Watersheds are fundamental to life on Earth, sustaining ecosystems and supporting human societies through drinking water, agriculture, industry, and recreation (Jordan and Benson 2015). In recent decades, many watersheds have seen an increased demand for freshwater, driven by urbanization and economic development (UNESCO 2015). While surface water is often the primary source of this water, groundwater is increasingly used (Gleeson et al. 2012; Bierkens and Wada 2019). The effects of pumping on aquifers, streamflow, and groundwater–surface water (GW–SW) interactions are well-documented (Barlow and Leake 2012; Konikow and Leake 2014; Zipper et al. 2019; Konikow and Bredehoeft 2020). The literature is abundant on topics such as the impacts of pumping on aquifers and streams (Wada et al. 2010; Mukherjee et al. 2018; Peters et al. 2022; Mieno et al. 2024), the quantification of streamflow and groundwater depletion (Ahmed and Abdelmohsen 2018; Zipper et al. 2022), the sensitivity of streams and aquifers to pumping (Bierkens et al. 2021; Hall et al. 2017; Mitra et al. 2019), and the mechanisms involved in streamflow and groundwater depletion (Starzyk 2012; Tripathi et al. 2021). Climate change impacts on these processes have also been explored (Ferguson and Maxwell 2012;

Ouyang et al. 2024; Ruzzante and Gleeson 2025). Banerjee and Ganguly (2023) reviewed the research advances on GW–SW interactions, detailing the scales and mechanisms of these interactions, as well as the methods used by researchers to study them. Other studies have highlighted a number of factors that influence the GW–SW interactions including streambed width and conductivity, aquifer hydraulic conductivity, and thickness, and the geometry and position of streams (Storey et al. 2003; Tripathi et al. 2021). Mahmoodzadeh et al. (2025) and Wang et al. (2011) examined the influence of pumping on these interactions. In essence, GW–SW interactions have emerged as a critical focus of water research and policy. This is evident in the increasing recognition by governments worldwide (e.g., DHI Water Policy and UNEP-DHI Centre for Water and Environment 2009; Scherer et al. 2017; Cantor et al. 2018) of the importance of managing surface water and groundwater as a single, integrated resource.

In their review of the literature on modelling GW–SW interactions, Barthel and Banzhaf (2016) noted that there is a great demand for integrated solutions relating to water resources and for scientifically sound approaches to managing and using water resources at the watershed scale. However, while there are numerous modelling studies on the impacts of pumping in watersheds globally (Karki et al. 2021; Lee et al. 2023; Mahmoodzadeh et al. 2025), many of these employed strictly groundwater flow models, which treat streams as boundary conditions. Relatively few studies have modelled the impacts of pumping at a watershed scale, and to our knowledge, no studies have simulated the long-term hydrologic evolution of a watershed under historical pumping conditions. Moreover, most previous studies simulate pumping by introducing all wells simultaneously at a single start date. While useful, this approach is neither realistic nor historically accurate, since wells are typically added incrementally over many years.

In this study, we develop an interpretive integrated hydrological model to: (1) trace the gradual increase in pumping over a near century; (2) evaluate how cumulative pumping has progressively altered the watershed's hydrological regime over time; and (3) identify potential tipping points that pinpoint the specific cumulative pumping volume or year (or decade) at which the watershed began to show significant impacts. A non-pumping simulation is also run to isolate pumping effects from natural climate variability.

2 | Data and Methods

2.1 | Study Area

In British Columbia (BC), Canada, groundwater use for residential, municipal, and agricultural purposes has increased in recent decades, leading to aquifer stress in some regions (Forstner 2018; Forstner and Gleeson 2019). Bertrand Creek Watershed (BCW), a lowland catchment which straddles the Canada—USA border between British Columbia and Washington State (Figure 1), is particularly affected. BC Aquifer 33—one of the aquifers in the province with a large rate of decline (Environmental Reporting BC 2024)—lies within this watershed. Groundwater development in the watershed has expanded rapidly, with the number of wells increasing from approximately 40 in 1949 to more than 1200 today.

Bertrand Creek Watershed covers an area of 109 km^2. Elevations range from 9 to 139 m above sea level (masl). BCW is characterized by a temperate oceanic climate (Beck et al. 2018). Mean annual temperature is 10.8°C, with summer highs near 25°C and winter lows just above 0°C. The watershed is rainfall-dominated, receiving an average annual precipitation of 1500 mm, falling primarily as rain between November and January, with minimal snow accumulation (Nott 2024). The summer months (June–September) are typically dry, with potential evapotranspiration (PET) exceeding precipitation (Halstead 1986). Groundwater levels follow seasonal precipitation trends, with fluctuations of 3–4 m, peaking in winter and declining into the early fall.

Bertrand Creek, about 34.8 km long, is the main stream in the watershed; it discharges into the Nooksack River in Washington. Its tributaries include multiple small streams, which are largely ephemeral (Kerr Wood Leidal Associates Ltd. 2009; Nott and Allen 2020). Streamflow in Bertrand Creek is highly event-driven, peaking from autumn to spring and often dropping to very low or no flow during the dry summer months (Nott 2024). Groundwater pumping may exacerbate summer low flows and contribute to elevated stream temperatures (Risley et al. 2010), which may be detrimental to Bertrand Creek's multiple fish species, including the endangered Salish sucker and Nooksack dace (Pearson 2004; Fisheries and Oceans Canada 2020).

Land use within BCW is diverse. Aldergrove, within the Township of Langley (ToL), is the main urban centre, while large portions of the watershed remain forested (Starzyk 2012). Agriculture is the predominant land use outside urban areas, consisting largely of berry production, poultry and mushroom farming, and pasture grazing for small livestock operations (GrowBC, n.d.).

Eleven mapped aquifers underlie BCW (Table S1). Water supply to the Aldergrove urban area is provided by seven municipal groundwater wells that withdraw groundwater mostly from the semi-confined Aldergrove AB aquifer (BC Aquifer 27). Outside the urban area, water supply to residential, agricultural, and commercial properties is supplied by privately owned wells (Starzyk 2012).

2.2 | Model Development

The software MIKE SHE (Danish Hydraulic Institute (DHI), 2025) was used for this study. MIKE SHE is an integrated catchment hydrological modelling software designed to simulate surface water and groundwater interactions in complex systems (DHI 2025). MIKE SHE models are forced using daily climate data (precipitation, temperature, and potential evapotranspiration) and can simulate surface overland processes, unsaturated zone flow, evapotranspiration and saturated zone (i.e., groundwater) flow (Golmohammadi et al. 2014). MIKE SHE uses a loosely coupled scheme (Barthel and Banzhaf 2016) requiring rivers, lakes, and other channels to be simulated first using MIKE 1D (formerly MIKE HYDRO) (DHI 2022) with the output

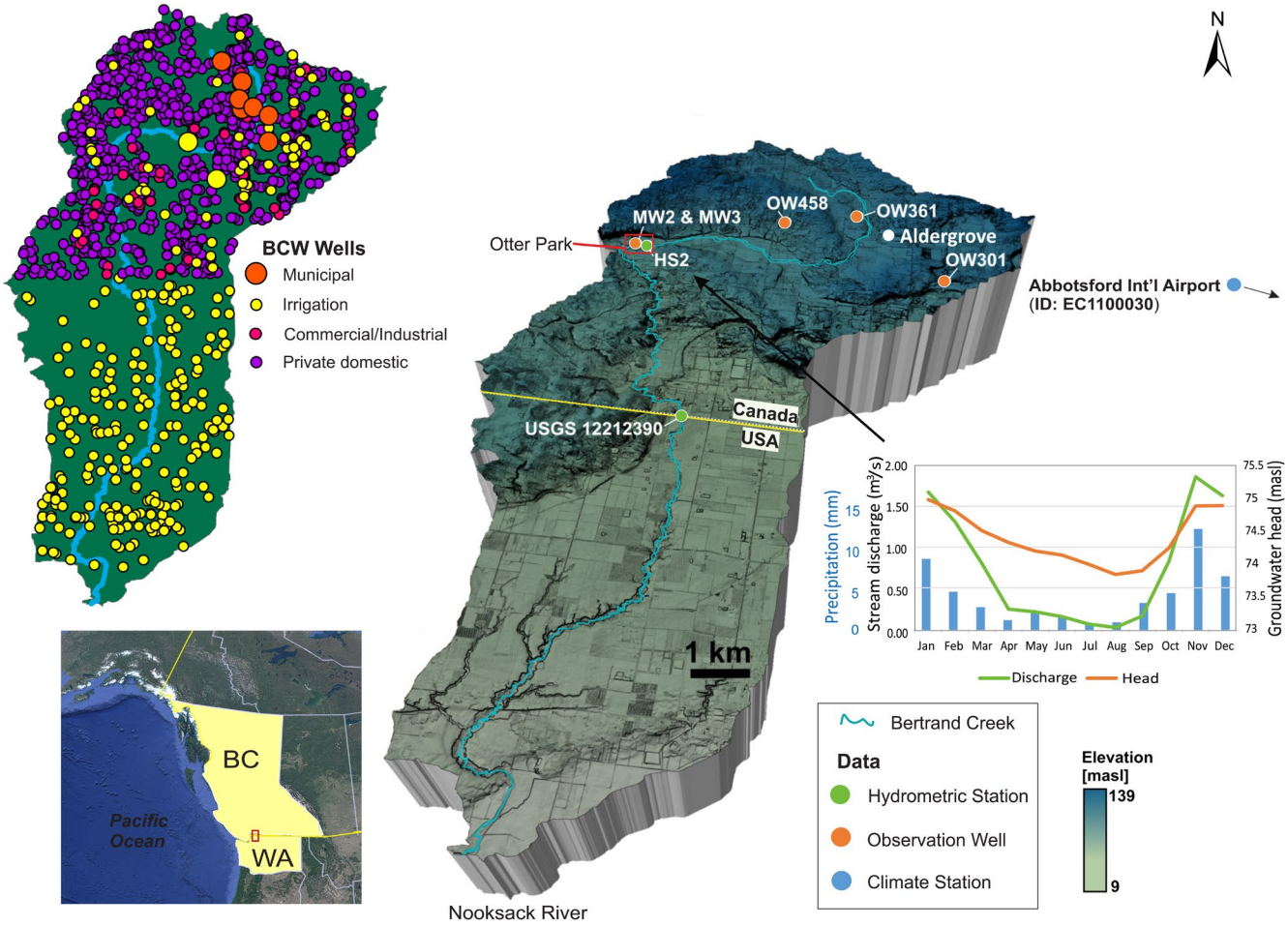

FIGURE 1 | Bertrand Creek Watershed at the Canada-USA border spanning British Columbia (BC) and Washington State (WA) (after Nott 2024). Shown are the provincial observation wells OW 458, 361 and 301, Otter Park wells MW3 and MW2, and stream gauges HS2 at Otter Park and USGS (12212390) at the Canada-USA border. The Abbotsford Airport climate station (Climate ID: 1100032 or CA001100040) is located in the City of Abbotsford, almost 9 km to the east of the City of Aldergrove in the Township of Langley. At the far right is a graph showing average stream discharge and groundwater head at Otter Park, along with precipitation at the Abbotsford Airport station, all for the period 2020–2021. Wells in the watershed and their uses are shown in the upper left; larger dots represent wells with pumping rates greater than 300 m^3/day; this includes two irrigation wells and seven municipal wells in Aldergrove.

forming the input of MIKE SHE. The coupling between MIKE 1D and MIKE SHE is made via river links. The river link network is created based on specified coupling reaches. While the entire river system is always included in the hydraulic model, MIKE SHE will only exchange water with the coupling reaches. A variety of timeseries and spatial outputs are available. A water balance can be generated at a variety of spatial and temporal scales using the water balance utility.

Nott (2024) developed the original hydrological model of BCW for the purpose of exploring naturalized watershed responses to extreme climate, thus stream diversions and groundwater pumping were not incorporated into the model. The model was vertically discretized into nine computational layers with varying thickness (10–45 m) totalling 218 m, representing the surficial geology of the watershed. Each computational layer has spatially varying geological units, based on previous groundwater modelling studies using MODFLOW by Allen et al. (2020) and Scibek and Allen (2005) (Table 1). Isotropic hydraulic conductivity values were initially set to those used in the earlier MODFLOW models, while specific yield (S_y) and specific storage (S_s) were obtained from values used by Scibek and Allen (2005) and supplemented by literature values.

TABLE 1 | Geological units and assigned hydraulic properties for the BCW model.

Unit name	K (m/s)	S_y	S_s
Fort Langley Fm. (clay)	8.7E−07	0.07	7.7E−04
Salish sediments (peat)	7.8E−05	0.44	6.0E−03
Sumas drift (till)	8.7E−06	0.07	7.7E−04
Sumas drift (glaciofluvial)	1.8E−04	0.04	3.0E−06
Sumas drift (glaciolacustrine)	2.4E−06	0.15	1.7E−04
Sumas drift (gravel)	1.8E−04	0.04	3.0E−06
Sumas drift (sand)	1.8E−04	0.04	3.0E−06

Two types of boundary conditions were specified. A constant head of 8.0 masl was assigned to layers 1 and 2 at the watershed

TABLE 2 | Pumping rates assigned to unlicensed wells and well counts according to well use category.

Well use	Assigned pumping rate (m^3/day)	Number of wells
BC private domestic wells	2.00	948
BC irrigation wells	118.06	31
BC industrial/ commercial wells	2.55	54
BC municipal wells	Variable (see Table 3)	7
WA irrigation wells	285.12	232

outlet to represent the average annual stream stage of the Nooksack River. Meanwhile, the entire catchment boundary and the base of the model were assigned as zero-flux (no-flow) boundaries.

An overview of the original model development and calibration is provided in Appendix S1. Herein, modifications specific to the purpose of this study, particularly climate data, pumping wells, and simulation settings, are described in detail.

2.2.1 | Simulation Scenarios

Two main simulations were run. The first is the non-pumping simulation, run from January 1, 1937 to September 30, 2023 to establish baseline or natural watershed conditions without any groundwater pumping. The second simulation included pumping over the same time period.

Model spin up refers to the time the model takes to stabilize and give results that are realistic (less sensitive to initial forcing conditions) enough to analyze (Seck et al. 2015). In this study, the period between 1937 and 1947 (the first 10 years of the simulation) was identified as the spin up time.

2.2.2 | Climate

The simulation began on January 1, 1937. Precipitation and temperature data were sourced from NOAA's National Centers for Environmental Information (NCEI). Three datasets from two nearby stations were identified: the earliest from the Abbotsford Upper Sumas station (GHCND:CA001100040), with data from 1935 to 1946; and two datasets from the Abbotsford A station (Abbotsford International Airport) (GHCND:CA001100030 or Environment Canada ID: 1100032), covering 1944 to 2012 and 2012 to 2024, respectively. Although records date back to 1935, data between 1935 and 1936 are incomplete. Therefore, 1937 is the earliest year with complete precipitation and temperature data.

PET was calculated using the FAO56 Penman-Monteith method (Ruzmetov 2021). This method requires minimum temperature (T_{min}), maximum temperature (T_{max}), latitude, elevation, average wind speed, and an estimate of dew point temperature relative to T_{min}.

TABLE 3 | Average volumes of groundwater pumped from all Aldergrove water supply wells between 2009 and 2022 (2017 for well #3) in m^3/day.

Well name	Average volume pumped between 2009 and 2022 (m^3/day)
Aldergrove #3	388.90[a]
Aldergrove #4	332.42
Aldergrove #6	826.50
Aldergrove #7	791.60
Aldergrove #8	1227.57
Aldergrove #9	773.54
Aldergrove #10	1313.11

[a]Pumping ceased in 2017.

2.2.3 | Pumping Wells

Well information, including well locations, well uses, pumping rates, well and screen depths, and drilling years were gathered for all wells within the watershed. Information on Washington wells was obtained from the Washington water rights map (https://appswr.ecology.wa.gov/WaterRightsMap) as well as from supplementary data from an earlier modelling study (Scibek and Allen 2005). Information on BC wells, including well use, was obtained from the Provincial Groundwater Wells and Aquifers database (GWELLS) (https://apps.nrs.gov.bc.ca/gwells). Pumping rates for unlicensed wells were estimated based on their intended use (Table 2).

For private domestic wells (PDWs), a rate of 2 m^3/day (0.000023 m^3/s) was used, as this is the "deemed right" in the *Water Sustainability Act* for domestic purposes (Section 22, *Water Sustainability Act, Province of British Columbia, 2016*). For licensed irrigation and commercial/industrial wells, the maximum pumping rates specified in their licences were assigned. For unlicensed irrigation wells, the average of the licensed wells' rates (118 m^3/day or 0.0014 m^3/s) was used. For unlicensed industrial and commercial wells, the pumping rate was estimated to be 2.55 m^3/day (0.00003 m^3/s), obtained by taking the average of 2.7 m^3/day (average of metered industrial properties in the ToL) and 2.4 m^3/day (average of metered commercial properties in the ToL) (Simpson et al. 2014). Not all wells registered in GWELLS have known intended uses. For wells with unknown uses, they were assumed to be domestic, as this is the most common well use, and were assigned the "deemed right" rate of 2 m^3/day.

Municipal wells in the watershed are all located near Aldergrove. Their individual annual pumping rates were obtained from water quality reports published by the ToL (https://www.tol.ca/en/the-township/water-quality-report.aspx). Average annual volume from 2009 to 2022 was used as the pumping rates for each well (Table 3). Well #3 was decommissioned in 2017, so its average is from 2009 to 2017.

On the Washington side, total groundwater pumping volumes were based on estimates from Bandaragoda et al. (2012),

who conducted a water budget modelling study for the lower Nooksack sub-basin and its watersheds. According to the study, between April and September (irrigation season), an average of 27.39 cubic feet per second (cfs) or 0.776 m^3/s of groundwater, used almost entirely for irrigation and livestock purposes, is pumped out of BCW. All Washington wells in this study are irrigation or stock water wells (all wells recorded in the Washington Water Rights Map within the BCW boundaries are irrigation/stock water wells). To distribute the total volume, each well was assigned a uniform pumping rate of approximately 0.0033 m^3/s (285.12 m^3/day).

Screen depths were specified for all wells. For most BC wells, screen depth information was directly available in the GWELLS database. In cases where screen depths were not available (including for US wells), it was assumed that their screen bottom was 1 m above the well bottom elevation and the screen top was 3 m above the well bottom elevation, resulting in a 2 m screened interval for the pumping well. This approach is based on a general observation that the bottom part of the well is typically the most productive unit.

Additional details, such as well locations (coordinates) and drilling years, were also obtained from the GWELLS database and Water Rights Map.

Separate pumping rate files were prepared based on well use. All wells under a particular well-use category were included in a separate file and set to begin pumping on the year the well was drilled. The wells were pumped continuously at a constant rate until the end of the simulation, except those decommissioned before 2023, such as Aldergrove well #3. This method attempts to replicate the gradual historical increase in pumping that occurred in the watershed between 1937 and 2023.

During the pumping simulation, all wells pumped at a constant rate, except irrigation wells, which typically only pump in the summer. For irrigation wells, pumping rate files were created to specify pumping only from April 1 to September 30, which is the irrigation season and the time period stated for pumping in most BC irrigation well licences.

A total of 1272 wells were used in the pumping simulation, with 1041 wells on the BC side of the watershed and 232 wells on the Washington side of the watershed (Table 4).

2.2.4 | Additional Analyses

Besides the two main simulations, additional simulations were run to test the model's sensitivity to various key parameter changes (sensitivity analysis), as well as another pumping simulation without the municipal wells to isolate the impact of municipal pumping on the watershed.

Cross-correlation analyses (CCAs) were also conducted in RStudio (version 2024.04.2) to examine relationships between precipitation minus evapotranspiration (P-ET) and subsurface storage change (SSC), as well as between P-ET and streamflow. The objective of these analyses was to further clarify the influence of climate on the watershed's evolution by determining the response times of subsurface storage and streamflow to changes in climate and water availability. Details on each additional simulation and analysis, and the results generated from them are provided in Appendix S1.

TABLE 4 | Number of wells present in the watershed by time period.

Time period	Number of wells installed	Cumulative number of wells
Pre-1950	41	41
1950s	206	247
1960s	66	313
1970s	372	685
1980s	157	842
1990s	166	1008
2000s	93	1101
2010s	117	1218
2020–2023	54	1272

3 | Results

3.1 | Simulated Water Balances

The generated total water balance (fluxes) for the entire watershed includes precipitation (P), evapotranspiration (ET), overland flow (OL) to river, groundwater to river discharge (GW to River), river to groundwater (River to GW), and other water balance components. ET, OL to River, and recharge are the main processes through which precipitation is partitioned. Recharge is the amount of precipitation that infiltrates into the saturated zone (SZ). It is not part of the generated total water balance; instead, it is obtained from the SZ (or unsaturated zone (UZ)) water balance, which is extracted separately. River to GW and GW to River represent exchanges between the river and aquifer, the direction of which is determined by groundwater levels; when the groundwater head is greater than the stream stage, the aquifer discharges into the stream, creating a gaining stream. When the stream stage is higher, the stream recharges the aquifer, creating a losing stream (Winter 1999). A stream can be both losing and gaining on different reaches and at different times (Frye 2013; Mahmoodzadeh et al. 2025). Subsurface storage change (SSC) describes the gain (+) or loss (−) of water from the subsurface (both UZ and SZ).

The water balance output was generated every 720 h or 30 days. Results were summarized by water year (October 1 to September 30) and then averaged by decade (e.g., yearly average from the 1940 water year to the 1949 water year represents the 1940s) to highlight long-term trends over the simulation period. As noted earlier, the period between 1937 and 1947 (the first 10 years of the simulation) is the spin up time. Though this period is still included in the graphs and tables, it is excluded from the analysis and discussion. Similarly, only the relevant water balance components are analyzed and discussed in this paper.

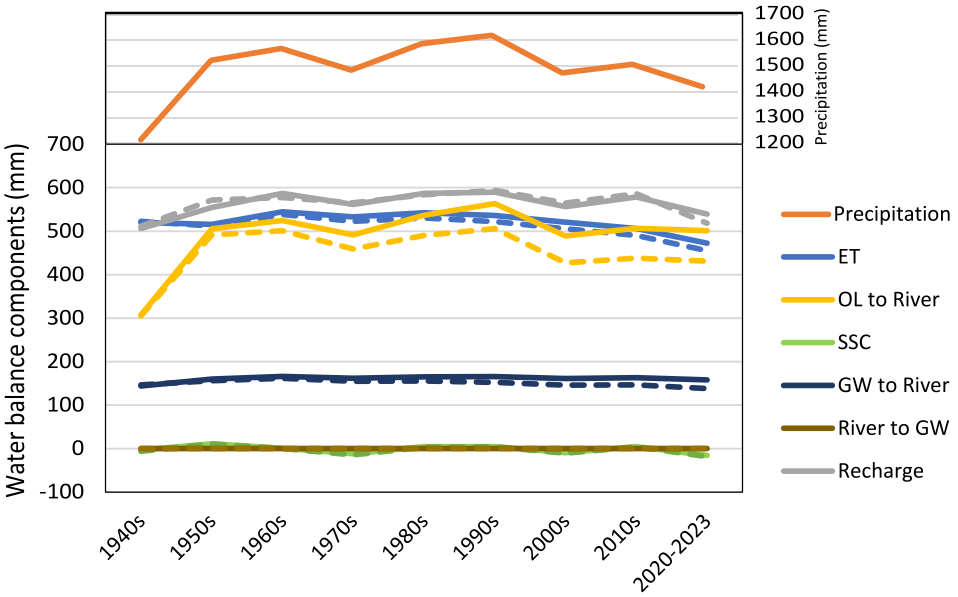

FIGURE 2 | Average annual water balance components (mm) in the non-pumping (solid lines) and pumping (dashed lines) simulations grouped by decade. Each value represents the mean of the water years within that decade. Note that for most of the 1940s, the model is spinning up.

Figure 2 shows the decadal water balance results, with those for the non-pumping simulation shown in solid lines, and those for the pumping simulation in dashed lines. Tables showing the results in more detail are provided in Appendix S1. MIKE SHE reports the water balance components as storage depths in millimetres (mm). To convert the depths to volumes, they were multiplied by the area of BCW (109 km^2).

BCW received an average precipitation of 1488 mm per water year. In the non-pumping simulation, an average of 35% or 521 mm was lost to ET, 33% or 492 mm as OL to River, and 38% or 564 mm formed recharge to the SZ (Table S7). In the pumping simulation, an average of 34% or 510 mm was lost to ET, 30% or 449 mm as OL to River, and 38% or 572 mm was recharge to the SZ (Table S8).

Under non-pumping conditions, groundwater contributed significantly to streamflow at Bertrand Creek, at an average of 160 mm per water year, while the stream contributed barely 0.30 mm/year to groundwater. SSC was generally within ±10 mm, except for some years in the 1950s, 1970s and 2020s, which had fairly large gains or losses. The average SSC across all water years in the simulation was −1.6 mm, translating to a cumulative loss of 138 mm over the 86 years simulated.

The incorporation of pumping wells introduced significant changes in the water balance. With an increasing number of wells added to the simulation every year, the pumping amount increased rapidly, from an average of 10.57 mm or 1 152 130 m^3 per water year in the 1940s to almost 140 mm or 15 260 000 m^3 per water year between 2020 and 2023. As a result, GW to River discharge decreased to an average of 150 mm per water year (from 160 mm/year) in the pumping simulation, and OL to River decreased from 492 to 450 mm/year. It also resulted in a more negative SSC, especially in decades that already had negative SSCs. The average SSC in the pumping simulation was −2.91 mm, indicating a cumulative loss of 249 mm over the simulated period. However, River to GW remained the same, at 0.30 mm/year.

Finally, the model errors are mostly negative, with an average of −8.8 mm for the non-pumping simulation and an average of −9.9 mm for the pumping simulation, suggesting unaccounted water loss in both simulations.

3.2 | Comparisons: Non-Pumping Versus Pumping Simulation

3.2.1 | Subsurface Storage Changes

The average SSC across all water years (excluding the spin up period) in the non-pumping simulation was −1.4, and −2.5 mm in the pumping simulation; an additional 1.1 mm (121 512 m^3) or approximately a 79% increase in storage loss attributable to pumping. Figure 3a shows the difference between the pumping SSC and the non-pumping SSC, which isolates the storage change attributable to pumping. Most of the differences are negative, especially the largest ones, indicating greater subsurface storage losses due to pumping, as expected. Positive changes tend to occur in wet years, particularly following a dry year. A positive change means that there is a greater increase in the recovery of the water level (i.e., a greater increase in water going into storage) under pumping conditions compared to non-pumping conditions.

SSC under non-pumping and pumping conditions are also consistent with P-ET, as shown in Figure 3b. For most years, the difference between the non-pumping and pumping SSC is small. Greater negative storage changes are observed after 1970, when a cumulative pumping amount of about 60.7 mm (or 6 616 300 m^3 across the watershed) was reached. The years with the largest negative changes (1998, 2015 and 2023) coincide with below average P-ET values. The latter 2 years (2015 and 2023) were classified as drought years in BC (https://droughtportal.gov.bc.ca/pages/historical-drought-levels). Average SSC for non-drought years is 26.6 mm, and in these drought years (1998, 2015 and 2023), the simulated SSCs were −152, −59 and −45 mm,

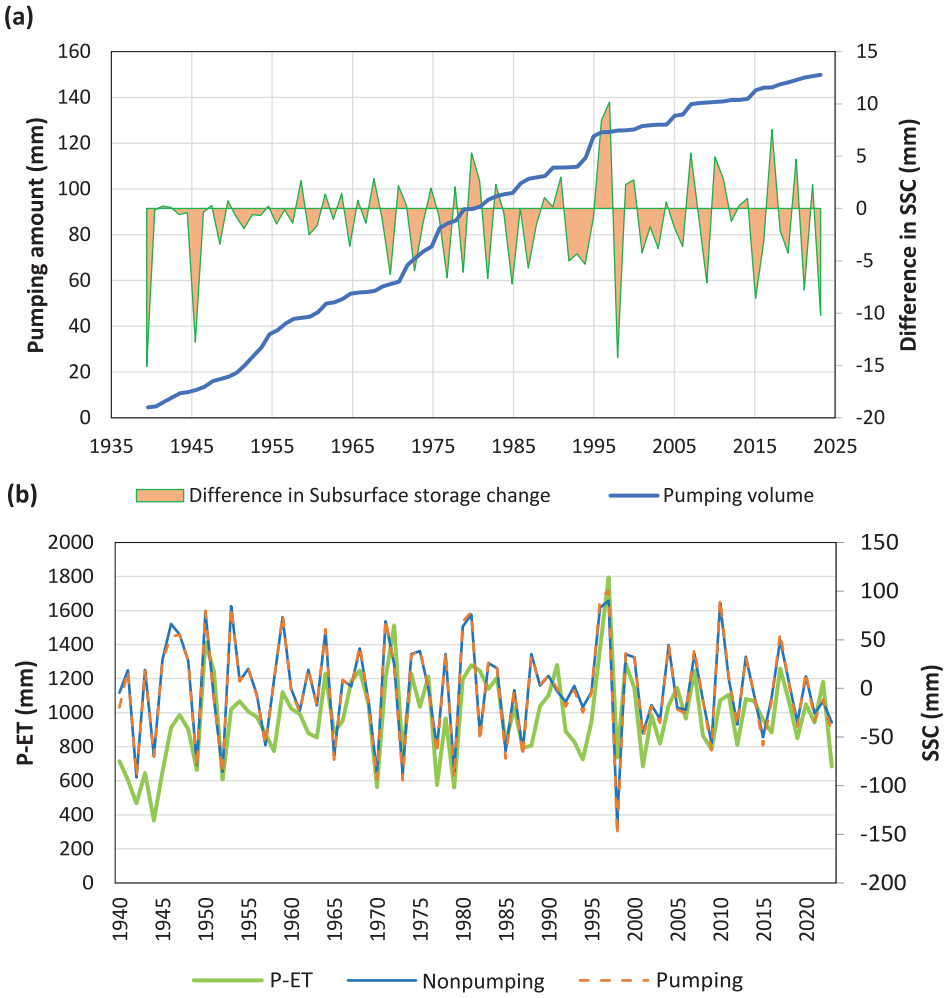

FIGURE 3 | (a) Annual differences in SSC between the pumping and non-pumping scenarios, shown alongside cumulative pumping amount (mm); (b) comparison of annual P-ET with SSC under pumping and non-pumping scenarios. Note that 1937 to 1947 is the model spin up period.

respectively, indicating storage deficits of −71.6 mm to as high as −178.6 mm (269%–673% greater losses compared to non-drought conditions).

Other years with significant, though less severe, declines align with the introduction of large-capacity (often municipal) wells, and sometimes also droughts. The year 1977 recorded a below-average P-ET of 575 mm, but in addition to this, two large-capacity municipal wells (combined pumping rate of 1124 m^3/day) drilled in that year resulted in a subsurface storage loss of 6.6 mm (719 400 m^3). Meanwhile, the year 1985 recorded a P-ET of 820 mm (lower than the average of 978 mm, but higher than that of 1977), but the addition of a municipal well with a pumping rate of 1227 m^3/day still led to a storage loss of 72, or 7 mm (784 800 m^3) more than without pumping.

In contrast, the few positive changes occurred in years following a year with below-average P-ET (dry years) or in exceptionally wet years. This likely occurs because dry years combined with pumping produce greater water level declines, creating additional available subsurface storage. During the following wet year, the increased storage capacity allows more infiltration and recharge, resulting in a larger increase in subsurface storage in the pumping simulation relative to the non-pumping scenario. For example, the greatest positive change (SSC = 100 mm) was in 1997, which had the highest rainfall of all the simulated years, at 2403 mm (P-ET of 1795 mm). The year 2017 was also a very wet year, with 1740 mm of rainfall and a P-ET of 1259 mm; however, the previous year, 2016, had a low P-ET of 884 mm. In addition to this, the Aldergrove well #3, which had been pumping at a rate of 388 m^3/day since 1972, was shut down in 2017. These wet years (1997 and 2017) showed additional storage gains of 276% (73.4 mm) and 114% (30 mm) respectively, relative to the non-drought average 26.6 mm.

These results suggest that climate variability and pumping interact to shape subsurface storage changes, with climate playing the dominant role and pumping having a secondary but detectable effect on annual storage fluctuations.

3.2.2 | Aquifer-River Exchanges

Aquifer-river exchanges differed significantly between the pumping and non-pumping simulations. Groundwater discharging from the aquifer into the river (forming baseflow) decreased steadily since 1960, averaging about 6% per water year, and reaching a 13% decrease between 2020 and 2023 (Figure 4a,c). Due to cumulative pumping over the entire simulation, almost

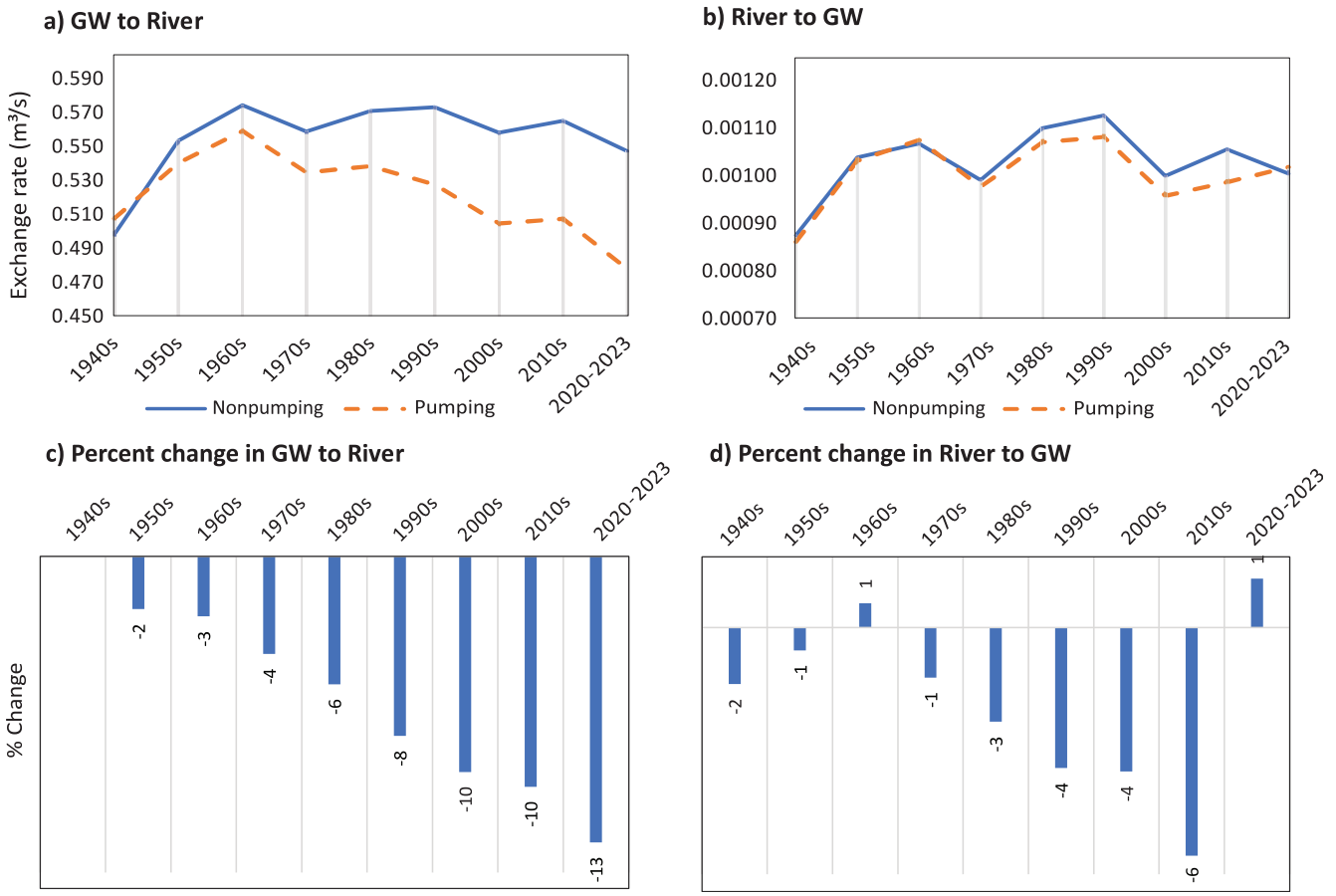

FIGURE 4 | Comparing (a) GW to River, and (b) River to GW exchange rates over the watershed between the pumping and non-pumping simulations, shown by decade. (c) and (d) are the corresponding percent changes in GW to River and River to GW exchange volumes, respectively. Note that most of the 1940s is model spin up time.

2.2 MCM/year or 6000 m^3/day less water is being discharged to the stream in recent times (2020–2023) compared to non-pumping conditions.

On the other hand, River to GW volumes showed a more variable pattern (Figure 4b,d). Although pumping was expected to increase induced recharge from the river, this was not the case. In early decades, similar volumes were exchanged from the river to the aquifer between the pumping and non-pumping simulations, but in the 1970s, the exchange volumes in the pumping simulation started to decrease slightly below those of the non-pumping simulation, reaching a 6% decrease in the 2010s, before increasing slightly again in the 2020s (for annual graphs, see Figures S3 and S4). One likely explanation for the loss of exchange volume between the river and the aquifer is the head differences. As baseflow declines significantly due to pumping, stream stage also declines, potentially reducing or eliminating the head gradient needed for river water to flow into the aquifer. In essence, the river may no longer be high enough to drive infiltration into the groundwater system.

The SZ to River gridded output shows the amount of groundwater being discharged along the stream. As noted by Nott (2024), Bertrand Creek is predominantly gaining and receives groundwater from adjacent aquifers along most of its length. However, no exchanges are simulated in the middle section of the stream, likely due to the presence of a till layer underlying this section in model layers 1 and 2. Figure 5 compares the non-pumping and pumping GW to River exchanges for the end of the 2022 water year (2nd October 2022) and illustrates the reduction in groundwater discharge throughout the stream due to pumping. Overall, less water is discharged to Bertrand Creek under pumping conditions; however, the greatest losses are observed in the southern part of the stream, as well as in the northern reach near Aldergrove.

3.2.3 | Overland Flow Changes

OL to River shows the greatest pumping-induced decrease of all water balance components simulated. It shows an average reduction of approximately 8% per water year, increasing to about 14% during the final 4 years of the simulation (2020–2023) (Figure 6b), even surpassing the decrease in GW to River exchange. Between 2020 and 2023, an estimated 70 mm/year (21 000 m^3/day) of additional overland flow would have been generated if there were no pumping.

The most likely driver of this reduction in OL to River is decreased seepage. Seepage is common in the areas around Bertrand Creek, since the water table is shallow and intersects the surface. Pumping lowers the water table, thereby reducing the amount of seepage that contributes to overland flow. Using the seepage flow gridded output in MIKE SHE, the impact of

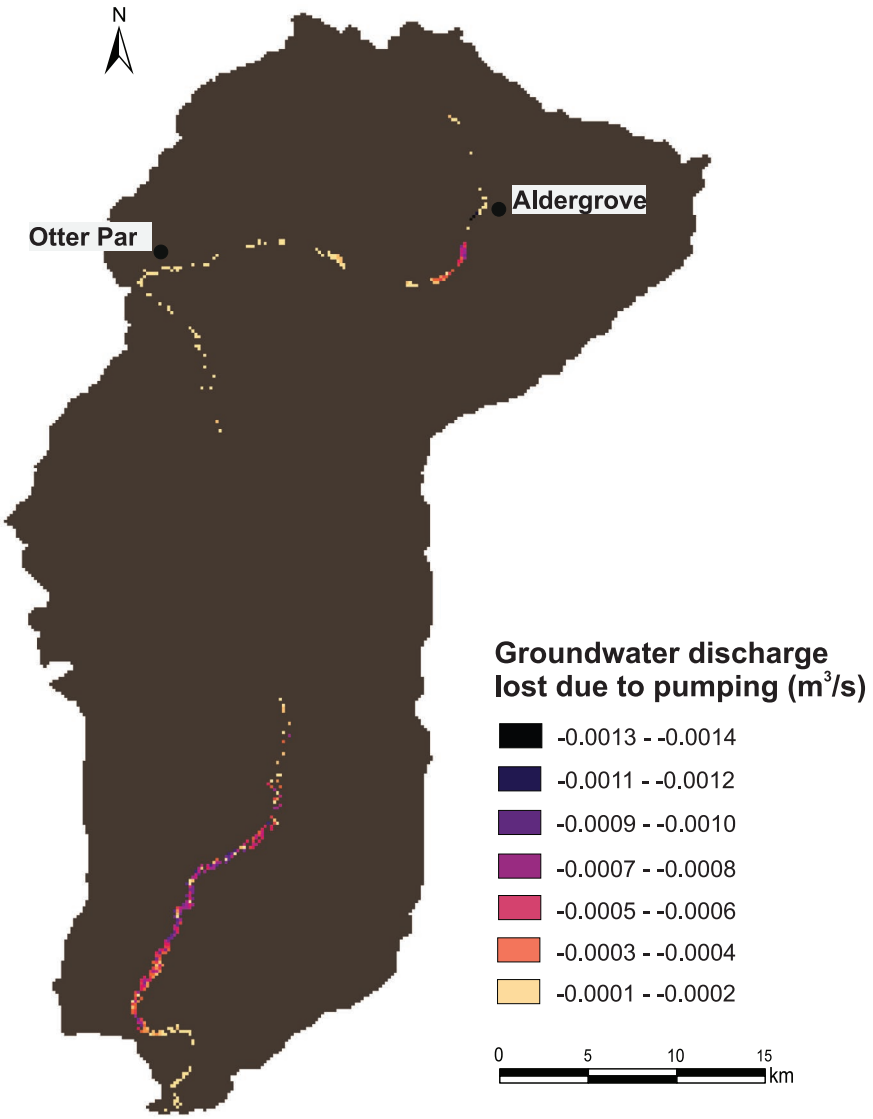

FIGURE 5 | Reduction in groundwater discharge to Bertrand Creek in the pumping simulation.

pumping on seepage generation can be quantified. Figure 6c compares the non-pumping and pumping SZ to OL seepage for January 1, 2023, and as expected, the map shows a reduction in seepage generated in the pumping simulation. Losses in seepage are typically in the range of 20–50 mm/day and are primarily concentrated in the southern portion of the watershed. A small number of locations show greater losses, up to 128 mm/day, while a few isolated areas show slight increases in seepage.

3.2.4 | Streamflow Changes

Simulated stream discharge at the locations of two gauges shows a progressive decline under pumping, beginning after the 1970s (Figure 7). For yearly graphs, see Figures S5 and S6. This is consistent with earlier results that groundwater and overland flow contributions to streamflow decreased in the pumping simulation. However, streamflow at the two gauges is not equally affected. Otter Park HS2 shows a much greater decrease, averaging about 11% per water year, and reaching 27% in the 2010s and 2020s. Meanwhile, USGS shows a more modest decrease, with an average of 4% per water year, and reaching 9% in the 2010s and 2020s. The likely cause of the modest response at USGS is discussed in Section 4.3.

4 | Discussion

4.1 | Pumping and Climate: Combined Impacts on Subsurface Storage, GW-River Exchanges, Overland Flow and Streamflow

Pumping in BCW has had a clear, immediate and cumulative impact on the hydrologic functioning of the watershed, particularly on GW-River exchanges, streamflow, and, to some degree, subsurface storage.

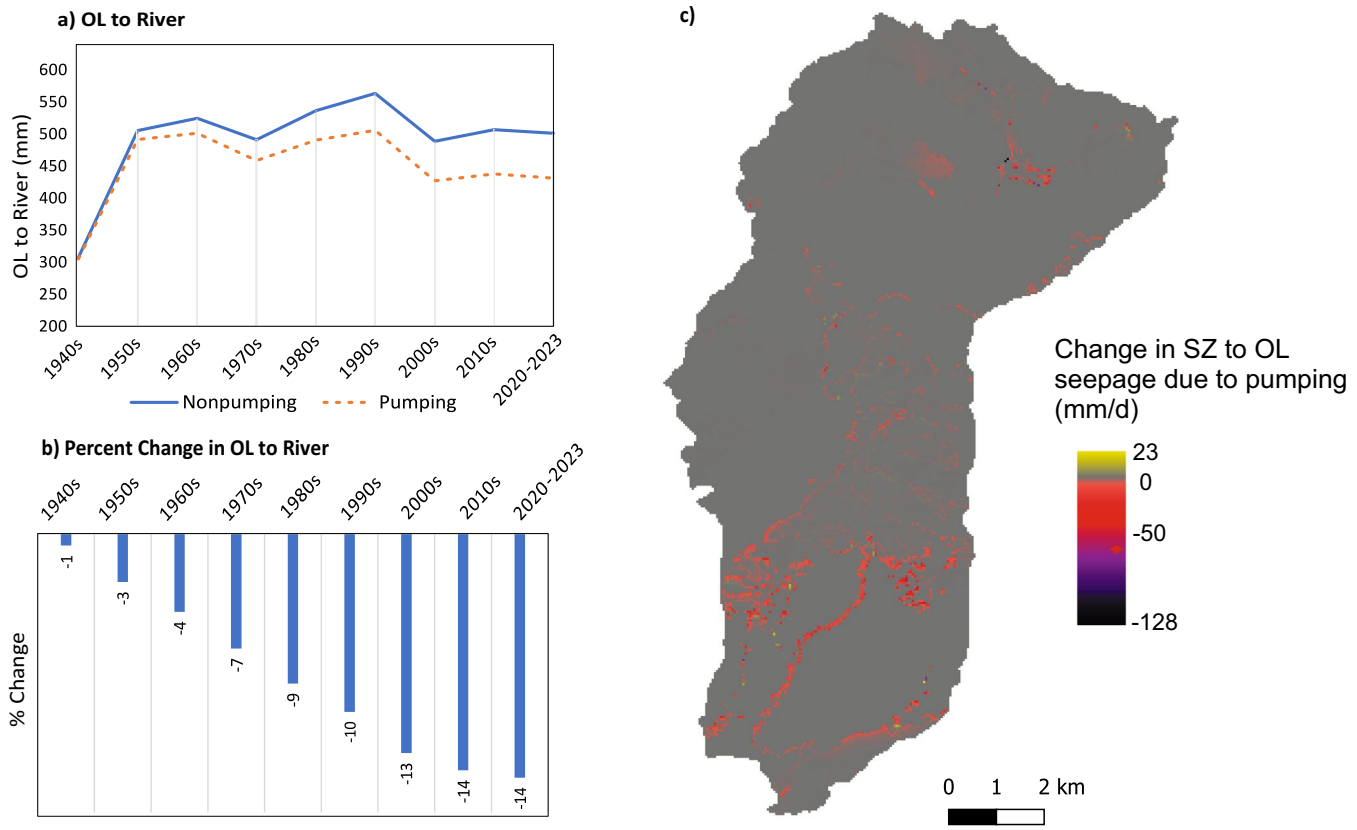

FIGURE 6 | (a) OL to River in the non-pumping and pumping simulations, (b) percent change in OL to River volumes, (c) change in seepage generated to OL due to pumping (mm/day).

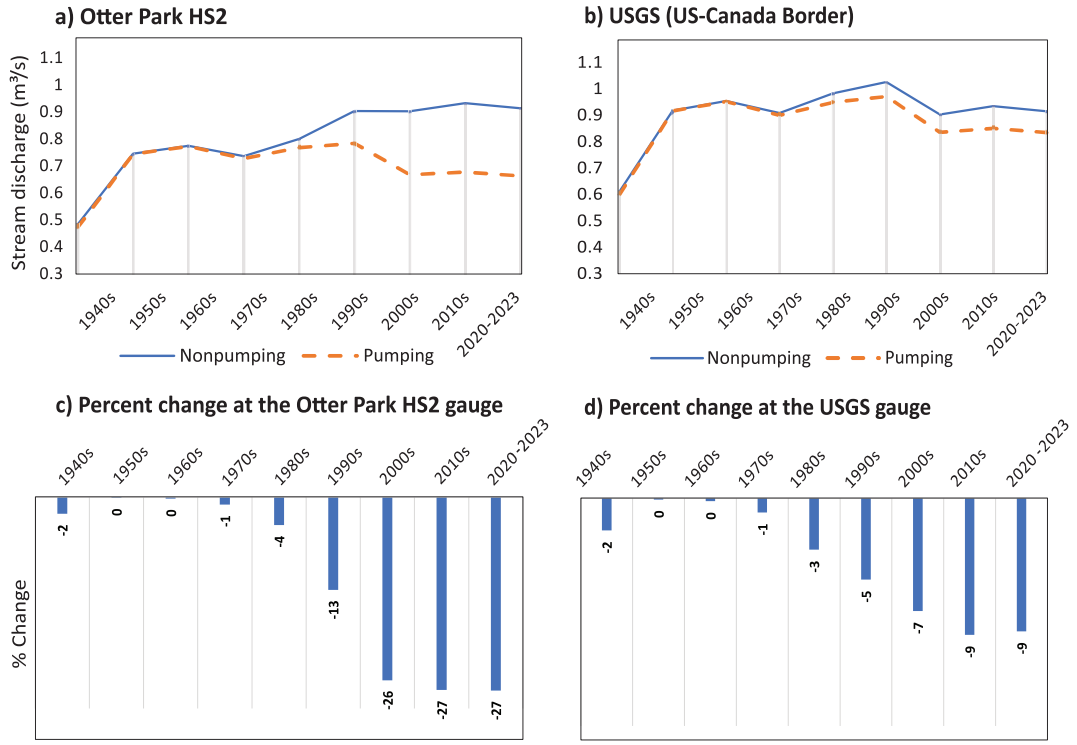

FIGURE 7 | Comparison of simulated average annual stream discharge between pumping and non-pumping simulations, grouped by decade, at (a) the Otter Park HS2 stream gauge and (b) the USGS border stream gauge (12212390). (c) and (d) are the corresponding percent changes at the Otter Park HS2 and USGS stations, respectively. See Figure 1 for gauge locations. Note that most of the 1940s is model spin up time.

Based on the simulation results, subsurface storage, though affected by pumping, is mostly controlled by climate and does not appear to be depleting significantly in response to pumping (Figure 8). Although there is an approximately 79% increase in storage loss in the pumping simulation, this translates to just over 1 mm (120 000 m^3) more water being lost per year in the entire watershed over the simulated time period compared to the non-pumping simulation. For comparison, historical groundwater levels were used to calculate SSC over the simulation period. Groundwater levels were measured at each well after drilling and are reported as static water levels (SWLs) in GWELLS. The average SWL was computed for each water year. Calculated SSC was determined by multiplying the SWL changes by a S_y of 0.008. This S_y value is lower in magnitude than the S_y values assigned to the units in the model (Table 1). It was selected because it gave the best visual match in the amplitudes of the simulated and calculated SSC values. Changes are plotted relative to the average 1970 SWL, giving a zero intercept for the trendline.

Figure 8a compares the simulated SSC to a calculated SSC based on historical groundwater levels measured since the 1970s in all pumping wells on the BC side of the watershed. The calculated SSC shows a slight increasing trend, while the simulated SSC has a very slight negative trend. Figure 8b only compares wells in Aquifer 33. This aquifer was isolated because it has been identified as having a large rate of decline, as reflected here, likely due to pumping of multiple municipal wells (note that the municipal wells in Aquifer 33 are not within BCW like the Aldergrove wells are). Aquifer 27, when isolated, also has a negative trend in calculated SSC (Figure 8c). However, even when Aquifer 33 and Aquifer 27 wells are removed, there remains a positive trend in calculated SSC (Figure 8d). Aquifer 27 and Aquifer 33 are important here because almost 60% of all BC wells in the watershed were drilled in these two aquifers alone.

The likely explanation for the calculated SSC within the watershed increasing over time is the cessation of pumping of municipal wells that lie outside the BCW. As noted in Table 3, Aldergrove #3 (Aquifer 27) was shut down in 2017 and the well was turned off in the model. But four other wells in the portion of Aquifer 33 outside BCW were shut down between 2019 and 2020. Groundwater levels in observation wells 361 and 458 have increased by 4 and 3 m, respectively, since 2015.

Drought conditions exacerbated the effects of pumping, leading to between 270% and 670% greater storage deficits, while exceptionally wet years resulted in 114%–276% increases in subsurface storage, although a reduction in pumping also contributed to an increase in subsurface storage. Similar results were found by Seo et al. (2018) who concluded that groundwater storage in the Haw River Basin in North Carolina was more sensitive to climatic conditions than to pumping volume, and that groundwater and streamflow depletion worsen under severe drought conditions. Other modelling studies report variable outcomes. Thatch et al. (2020) and Ouyang et al. (2024) found that in California's San Joaquin basin and the Mississippi Embayment, respectively, groundwater pumping had a greater influence over drought/climate change in driving storage losses and streamflow depletion. Meanwhile, Ferguson and Maxwell (2012) concluded that climate and pumping have equivalent impacts on groundwater storage and streamflow in the southern Great Plains, USA. The relationship between streamflow, climate and pumping is further explored in later paragraphs. As climate change worsens and dry conditions become more frequent and severe (Canadian Climate Institute 2025), subsurface storage is expected to experience greater stress, possibly triggering groundwater depletion (Thatch et al. 2020).

Over time, as pumping continues, the groundwater system adjusts and compensates for reduced storage by inducing recharge and/or decreasing discharge to the stream (Theis 1940; Wilson 1993). GW to River exchange declined by an average of 8% each water year since 1970 when pumping increased dramatically, and in the final 4 years of the simulation (2020–2023) decreased by 13% (see Figure 4c). In contrast, the River to GW exchange saw a 0% change under pumping conditions compared to non-pumping conditions, with an average of 0.30 mm of water lost to aquifers in the non-pumping simulation, and 0.297 mm in the pumping simulation. However, the distribution shifted; the amounts were roughly similar in the earlier decades, but during the 1970s the pumping scenario showed even less River to GW loss than the non-pumping case, before rising again in the 2020s (see Figure 4d).

Overland flow also declined in the pumping simulation, even exceeding the magnitude of decline in groundwater discharge and SSC, with an average decrease of 8% per water year and up to 14% in 2020–2023 (see Figure 6b). This translates to approximately 21 000 m^3/day less overland flow in the 2020s in the pumping simulation. Seo et al. (2018) similarly found substantial reductions in overland flow due to pumping in their simulation of the Haw River Basin. In their case, however, the decline was driven by increased infiltration, as opposed to decreased seepage, which is the case in BCW.

Since groundwater discharge and overland flow both contribute to streamflow, a reduction in these components can result in reduced streamflow. The simulation results suggest that the reduction in GW to River discharge and OL to River resulted in reduced year-round streamflow at two stream stations. The simulated streamflow timeseries at Otter Park HS2 showed a decline of up to 27% in the 2010s and 2020s in the pumping simulation compared to non-pumping conditions. At the USGS gauge, there was a decrease of 9% in the 2010s and 2020s in the pumping simulation compared to the non-pumping scenario.

These simulated streamflow trends are consistent with observed streamflow measurements at the USGS station (which has a longer-term record than Otter Park HS2), when low flows are isolated. Figure 9 shows the lowest observed discharge and simulated discharge (pumping simulation) in each water year since 1984. In Figure 9b, both timeseries were normalized by subtracting their respective means. Although the model overestimates low flows, as discussed further in Section 4.3, the two timeseries both show a declining trend and similar patterns in highs and lows, with the simulated streamflow timeseries showing greater amplitude.

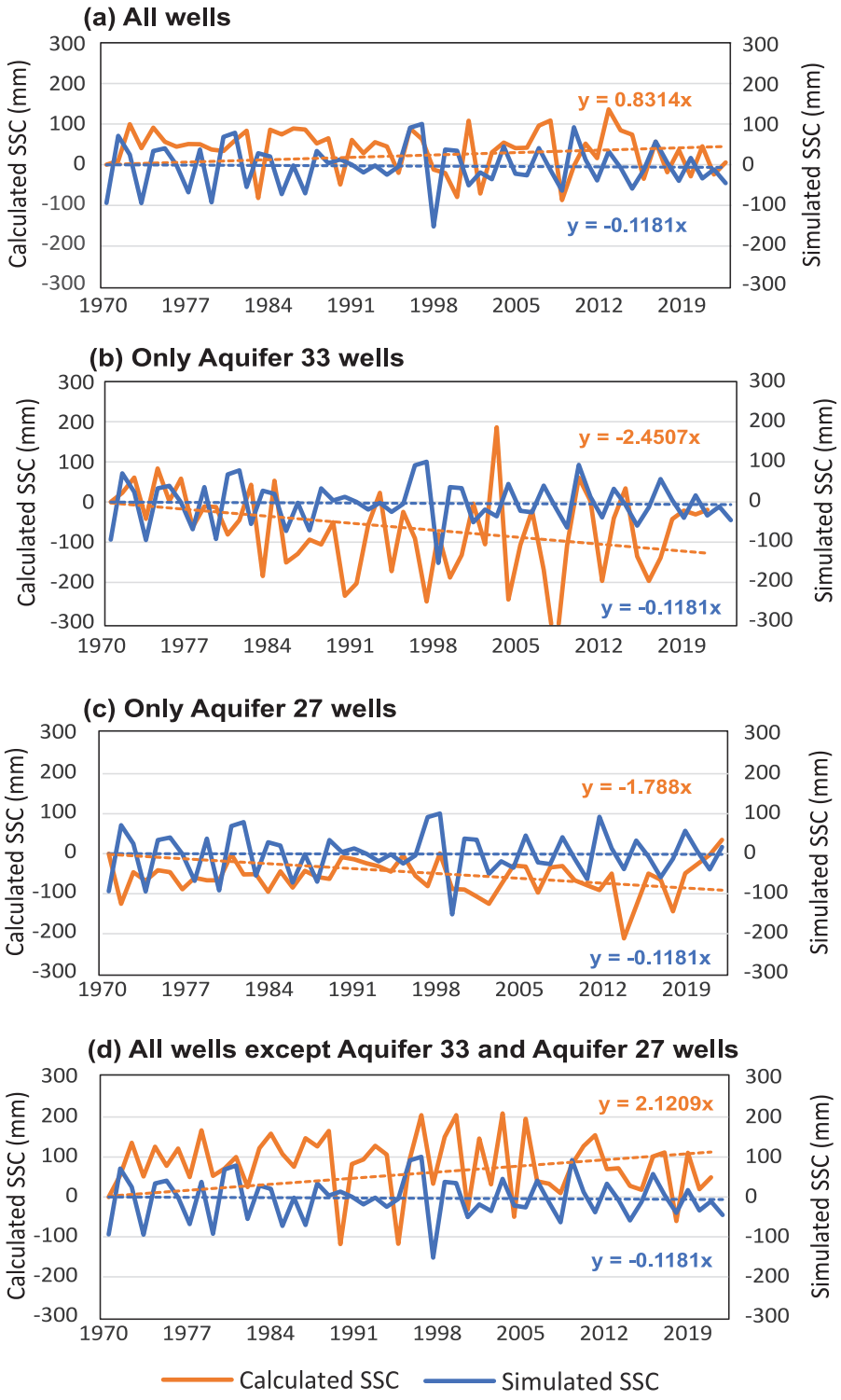

FIGURE 8 | Comparison of simulated SSC and SSC calculated using recorded static water levels in (a) all wells in the watershed, (b) only Aquifer 33 wells, (c) only Aquifer 27 wells, and (d) all wells in the watershed except Aquifer 33 and Aquifer 27 wells, between 1970 and 2023. Note that the static water levels in Aquifers 33 and 27 were multiplied by the same specific yield value despite them being confined aquifers. This is to enable comparison with the wells in other aquifers. The trendlines have a zero intercept.

There is evidence of climate influence on streamflow. Three of the 5 years with the lowest flows were drought years, with below-average P-ET. Similarly, three of the 5 years with the highest streamflow were in very wet years with P-ET values exceeding 1200 mm. The CCA supports this as it shows a strong correlation ($r = 0.77$) between P-ET and streamflow (Figures S7 and S8).

Overall, Bertrand Creek remained a primarily gaining stream in both the non-pumping and pumping scenarios. Though these results represent the watershed scale as a whole; smaller reaches of Bertrand Creek may actually be losing, as noted by Starzyk (2012), Allen et al. (2020), and Nott (2024). The modelling results also suggest that streamflow depletion from pumping is caused by a combination of reduced overland flow

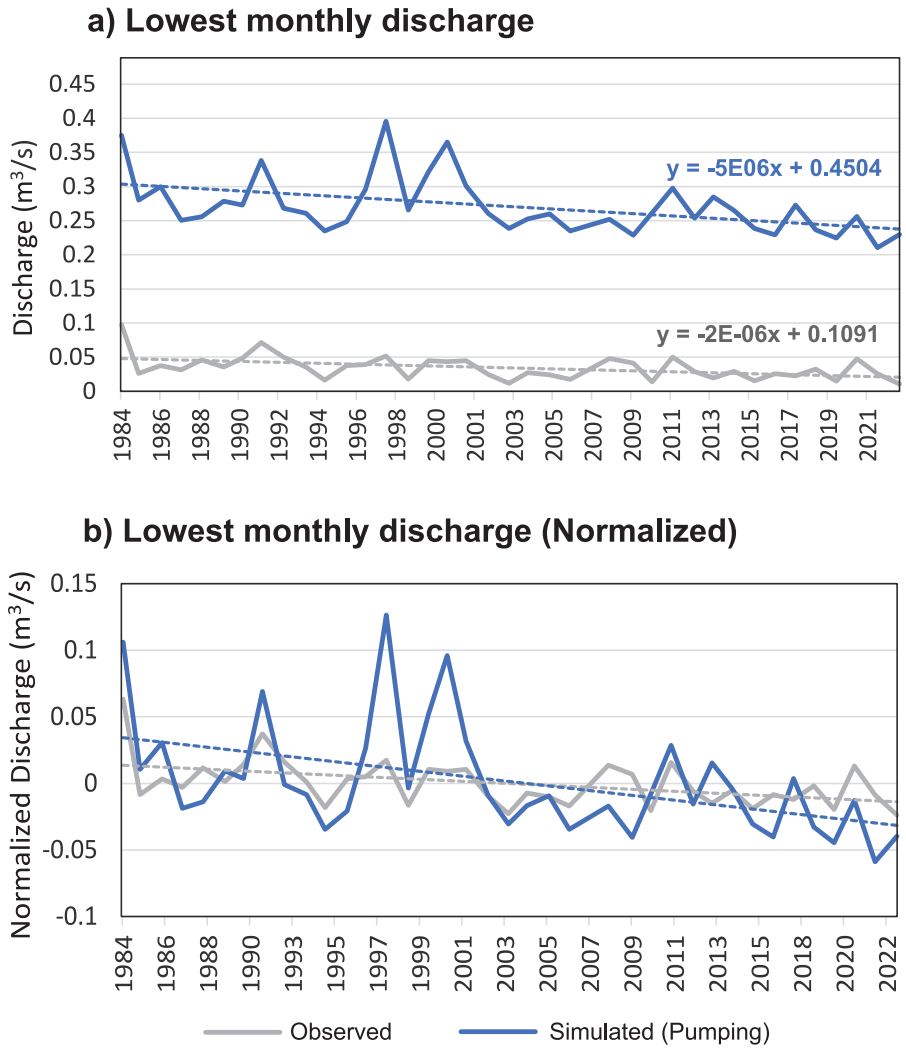

FIGURE 9 | (a) Observed and simulated (pumping) lowest monthly stream discharge at the USGS station (ID: 12212390) for each year between 1984 and 2022. (b) Observed and simulated lowest monthly stream discharge normalized by subtracting the mean from each value. The line equations for the timeseries in (a) are shown in colours corresponding to their respective datasets.

(i.e., decreased OL to River), and reduced groundwater discharge (i.e., decreased GW to River). Induced recharge (River to GW) did not play a role, as it remained constant in the two scenarios.

Streamflow depletion has been discussed and studied extensively in works by Mair and Fares (2010), Barlow and Leake (2012), Condon and Maxwell (2019) and Zipper et al. (2019), as well as in other BC watersheds (Najafi et al. 2017; Allen et al. 2020; Mahmoodzadeh et al. 2025). Just as subsurface storage losses are worsened in droughts, streamflow depletion also appears to be more severe in dry years (Najafi et al. 2017; Lapides et al. 2023; Ruzzante and Gleeson 2025).

4.2 | Tipping Points

The gradual, yearly introduction of wells in the pumping simulation allows for the identification of potential tipping points, that is, periods when pumping volumes become large enough to a drive significant deviation in watershed response between the pumping and non-pumping simulations. Tipping points generally coincide with the year that the first large wells began pumping. For instance, the streamflow timeseries (Figure 7a,b) and River to GW timeseries (Figure 4b) show significant deviations beginning in the 1970s, when the first municipal wells were drilled. In fact, more wells were drilled in the 1970s than in any other decade. Interestingly, the GW to River and OL to River timeseries shows almost immediate declines in response to pumping; however, this does not translate to visibly depleting streamflow until the 1970s. So, although domestic and irrigation pumping had been occurring since the beginning of the simulation, the introduction of municipal wells in the 1970s (alongside additional domestic and irrigation wells), marked a tipping point in the watershed.

Without pumping the municipal wells, simulated GW to River exchange was on average 4811 m^3/day less than the non-pumping simulation in 2020–2023, instead of almost 6000 m^3/day less in the same period with the municipal wells (Figure S13). Overland flow declined by about 59 mm/year (17 500 m^3/day) in the simulation without municipal wells, compared to over 70 mm/year (21 000 m^3/day) with the municipal wells pumping (Figure S14). Also, stream discharge at the USGS station declined by only

3% at the end of the simulation (compared to 9% when the municipal wells are present), while stream discharge at the Otter Park gauge declined by just over 20% at the end of the simulation (compared to 27% when the wells are present) (Figure S15). The implication is that the Aldergrove municipal wells alone are responsible for capturing almost 1200 m^3/day of groundwater, and 3500 m^3/day of overland flow that would have discharged to the stream between 2020 and 2023, and between 6% and 7% of the streamflow depletion observed in the same time period. These additional losses matter most in low flow seasons, when even a few hundred m^3/day can be critical for aquatic organisms (Benejam et al. 2010; Rolls and Bond 2017).

These results mirror findings by Zipper et al. (2019) that a small number of wells (10% of wells in the studied watershed) contributed disproportionately to streamflow depletion in the watershed. Starzyk (2012) also noted the contribution of the municipal wells to streamflow depletion in Bertrand Creek. But besides the seven municipal wells, there are also two irrigation wells with similarly large volumes; one pumping 449 m^3/day drilled in the 1980s and the other pumping 510 m^3/day drilled in the 1990s (see locations in Figure 1). Collectively, these nine large capacity wells, just 0.7% of all wells in the simulation, pump 6300 m^3/day out of the watershed (14% of total daily pumping in the watershed).

4.3 | Limitations, Uncertainty and Implications

While the BCW model was useful for examining trends, drivers, and hydrologic dynamics in the watershed, a key limitation is the inconsistency between some simulated and observed values. When calibrating the original non-pumping watershed model, Nott (2024) determined that the simulated stream discharge underestimated peak flows and tended to overestimate low flows but replicated the flashiness and seasonality of the observed streamflow. Nott theorized that this was due to stream diversions and groundwater pumping not being accounted for in the model. Thus, in this study, it was assumed that accounting for pumping in the model would correct the calibration issues identified by Nott. This was not the case. When compared to observed measurements, the model still underestimates peak flows and overestimates low flows at the USGS and Otter Park gauging stations (Figures S9 and S10).

Understanding the reason for the unsatisfactory simulated stream discharge requires consideration of deep groundwater flow. Previous groundwater flow modelling at a regional scale identified a significant component of deep groundwater flow that discharges into the Nooksack River to the south (Scibek and Allen 2005). In the current MIKE SHE model, however, the watershed boundary is a no-flow boundary, apart from a fixed head of 8.0 m in single cells beneath Bertrand Creek in layers 1 and 2. This effectively traps groundwater that would otherwise have flowed south into the Nooksack. The groundwater is forced to flow upward, creating seepage areas that ultimately generate overland flow that discharges into Bertrand Creek. This can be seen in the seepage map in Figure 6c, which shows most of the seepage in the model happening in the south. The fact that all the groundwater is forced to exit precisely at the watershed outlet causes the heads to rise and, consequently, the stage under low flow conditions to be higher than observed, especially at locations closer to the south. Hence the USGS gauge showed less of a reduction in discharge than HS2.

Although the no-flow watershed boundary causes simulated low flows to be overestimated by preventing deep groundwater from flowing out of the watershed, the relative impacts of pumping remain robust because the same boundary condition was applied consistently in both pumping and non-pumping simulations. Therefore, the model still reasonably simulates the relative response of the system to pumping.

Use of a model boundary defined by surface water drainage (i.e., a watershed boundary) also influenced the groundwater head calibration results. The groundwater head is overestimated at provincial observation wells (361 and 458) by about 10 m, while the wells at Otter Park are only slightly underestimated (Figure S11). The two wells at Otter Park (MW2 and MW3; Figure 1) are completed in an unconfined sand and gravel aquifer (Aquifer 15). Underestimation of the heads can be accounted for by local heterogeneity that is not accounted for in the model. The two provincial observation wells (361 and 458), however, are completed in a confined sand and gravel aquifer (Aquifer 27) that underlies a local topographic high. Regional groundwater flow is radially outwards with notable local flow deflections towards topographic lows associated with surface water drainage, in this case, the Salmon River to the north (Scibek and Allen 2005). Therefore, the footprint of this aquifer extends past the Bertrand Creek Watershed boundary. Consequently, it is unsurprising that the hydraulic heads in these observation wells are overestimated as the aquifer is effectively cut off by the model watershed boundary. Overall, however, groundwater heads across the watershed are well calibrated (see Figure S12). The historical SWLs measured in wells at the end of drilling were compared to the hydraulic heads in layer 2 at the end of the simulation (layer 2 generally contains the water table). The normalized root mean square (NRMS) error is 13.6% and the R^2 is 0.97, indicating very good calibration.

Both simulations also show computational errors of approximately −9 to −10 mm, indicating that the model lost a non-negligible amount of water during the simulation. Some portion of this lost water may represent subsurface storage or river-aquifer exchange that was not fully captured by the model. Therefore, the model cannot be considered fully accurate in a strict physical sense. The aforementioned errors, omissions, and simplifications likely contributed to the discrepancies between the simulated and observed values.

Sensitivity analysis results show that the model simulations are sensitive to hydraulic conductivity (both $K_{vertical}$ and $K_{horizontal}$) and vertical hydraulic conductivity ($K_{vertical}$). In particular, SSC is most sensitive to $K_{vertical}$, but not to K, and OL to River is most sensitive to K, but not to $K_{vertical}$, while recharge, River to GW and GW to River, appear to be equally sensitive to $K_{vertical}$ and K. On the other hand, all of the listed components showed little sensitivity to S_y (see Figures S16–S18).

The increase in pumping in BCW likely would have been accompanied by changes in land cover; however, land cover was

held constant in the model. This simplification may reduce the accuracy of the results, as it does not account for hydrological changes associated with land use change. For example, Miri et al. (2026) reported that forest cover over the Hopington C aquifer declined from 71.7% in 2000 to 42.6% in 2020 due to conversion to agricultural land. Such deforestation would have altered hydrological processes such as ET, OL to River, and recharge independently of pumping. To evaluate the potential impacts of land cover on hydrology, an additional simulation was conducted in which all current agricultural land (~40% of the watershed area) was converted to forest. Under this scenario, ET increased by 8%, while OL to River and groundwater recharge decreased by 3% and 6%, respectively. This suggests that the historical transition from forest to agricultural land would have had the opposite effect: reducing ET and increasing OL to River and recharge. Therefore, land cover change could have significantly influenced the watershed's hydrology, and its exclusion from the model may lead to an underestimation of the combined impacts of forest loss and groundwater extraction.

The wells' pumping rates are another source of uncertainty, for two reasons. First, most wells were assigned pumping rates based on the well use. However, in reality, the well may pump significantly more or less than the rate assigned. Wells with unknown uses were assumed to be domestic, though some may serve other purposes or be inactive. On the Washington side, only irrigation/stock water wells were included due to limited information. This likely underestimates total withdrawals since other types of wells may also be in use but were not accounted for. Consequently, groundwater pumping across the watershed is probably greater than the model represents. Second, constant pumping rates were assigned. In reality, pumping rates vary over time and are likely influenced by climatic conditions. In dry years, for example, pumping for irrigation and municipal supply likely increases to meet higher crop and household water demands (Apurv et al. 2017). Keeping pumping rates constant results in an underestimation of the combined effects of pumping and climate variability. Therefore, dry years likely experience greater subsurface and streamflow declines than simulated in this study.

Irrigation return flow was not simulated in this model, although it has been shown to contribute significantly to streamflow (Kendy and Bredehoeft 2006; Bailey 2025).

In addition, only the main channel, Bertrand Creek, was represented in the model, while its tributaries were omitted. This simplification may have affected the simulated streamflow, as water that would naturally be dispersed among several channels is instead concentrated within the main creek. This could also partly explain the higher simulated low flows compared to observed. In addition, Nott (2024), documented 44 active surface water licences in BCW diverting nearly 500000 m^3/year, although this accounts for only 3.2% of the total amount of diverted surface water and groundwater combined. Therefore, surface water diversions in the watershed were not accounted for in the model.

Despite these limitations, this study of the hydrologic evolution of BCW offers several benefits with implications that extend beyond the watershed. By reconstructing past changes in subsurface storage, GW-SW exchanges, and streamflow, this study highlights the timing, causes, and consequences of watershed change. It helps to distinguish the relative roles of climate variability and human activities like pumping and identifies key tipping points. Beyond contributing to scientific understanding, such historical reconstructions provide a foundation for anticipating future responses under continued stress, supporting evidence-based strategies for sustainable water management, and strengthening aquatic habitats, especially in regions with endangered species (Horne et al. 2017; Juracek et al. 2017; Zipper et al. 2022, 2024).

5 | Conclusions

This modelling study involved two primary simulations: a baseline, non-pumping simulation and a pumping simulation designed to recreate the hydrologic evolution of BCW under nearly a century of groundwater pumping. By comparing the two simulations, we assessed the long-term impacts of steadily increasing pumping on the watershed, specifically on the subsurface storage, GW-SW exchanges, and streamflow.

The results indicate that changes in subsurface storage have been shaped by interactions between climate and pumping, with climate and water availability (P-ET) playing the dominant role, and pumping having a secondary but detectable effect. The subsurface storage showed the greatest losses in dry years, often amplified by increased (municipal) pumping, while gains were rare and only occurred in exceptionally wet years and/or when large wells were shut down.

Aquifer-river exchanges were also impacted by pumping. GW to River volumes declined by about 6% each year on average, with roughly 6000 m^3/day less water being released to the stream under pumping conditions in 2020–2023 compared to without pumping. Conversely, River to GW fluxes initially resembled the non-pumping scenario but eventually declined, possibly due to the stream stage dropping below the water table. Overland flow also declined in the pumping simulation, by an average of 8% each water year, driven by a reduction in groundwater seepage to the surface. Due to decreased GW to River discharge and overland flow, streamflow at the Otter Park and USGS stations decreased by about 11% and 4% per year, respectively. However, the decrease at the USGS station is likely underestimated due to the limitation of using a watershed boundary for simulating surface water and groundwater flow. Deep groundwater actually discharges along the Nooksack River, but is forced to discharge into Bertrand Creek at its outlet, causing more seepage and higher low flows in the southern portion of the model. This limitation is an important one for future modelling studies attempting to simulate an integrated hydrological system using a surface watershed as a focus rather than a groundwatershed.

Our analysis also highlights the 1970s as a pivotal decade in BCW's history. This decade saw the introduction of multiple large-capacity municipal wells, which proved to be the main trigger for subsurface storage and streamflow depletion. These municipal wells are responsible for capturing almost 4700 m^3/day of water (both GW to River and OL to River) that would have discharged to the stream in the final 4 years and contributing 6%–7% of the streamflow depletion observed in that same period.

Finally, despite the model's limitations, it adequately recreated BCW's pumping history and enabled us to study and understand its hydrological evolution from the 1930s. This understanding is crucial for explaining current hydrological conditions and can also help inform sustainable watershed management moving forward.

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Data Availability Statement

The data that support the findings of this study are openly available in HydroShare at http://www.hydroshare.org/resource/d4bca931868340e5900747a5f6fade2c.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section. **Appendix S1:** Supplementary data.