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Constraining groundwater discharge in a large watershed: Integrated isotopic, hydraulic, and thermal data from the Canadian shield

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[1] The objective of this study is to evaluate the pattern and rate of groundwater discharge in a large, regulated fractured rock watershed using novel and standard methods that are independent of base flow recession. Understanding the rate and pattern of groundwater discharge to surface water bodies is critical for watershed budgets, as a proxy for recharge rates, and for protecting the ecological integrity of lake and river ecosystems. The Tay River is a low-gradient, warm-water river that flows over exposed and fractured bedrock or a thin veneer of coarse-grained sediments. Natural conservative (δ^2H, δ^{18}O, Cl, and specific conductance), radioactive (^{222}Rn), and thermal tracers are integrated with streamflow measurements and a steady state advective model to delimit the discharge locations and quantify the discharge fluxes to lakes, wetlands, creeks, and the Tay River. The groundwater discharge rates to most surface water body types are low, indicating that the groundwater and surface water system may be largely decoupled in this watershed compared to watersheds underlain by porous media. Groundwater discharge is distributed across the watershed rather than localized around lineaments or high-density zones of exposed brittle fractures. The results improve our understanding of the rate, localization, and conceptualization of discharge in a large, fractured rock watershed. Applying hydraulic, isotopic, or chemical hydrograph separation techniques would be difficult because the groundwater discharge "signal" is small compared to the "background" surface water inflows or volumes of the surface water bodies. Although this study focuses on a large watershed underlain by fractured bedrock, the methodology developed is transferable to any large regulated or unregulated watershed. The low groundwater discharge rates have significant implications for the ecology, sustainability, and management of large, crystalline watersheds.

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1. Introduction

[2] Quantifying the rate and pattern of groundwater discharge to surface water bodies is vital for developing watershed budgets, constraining recharge rates, and protecting the ecological integrity of lake and river ecosystems [*Hayashi and Rosenberry*, 2002; *Sophocleous*, 2002]. In watersheds underlain by porous media aquifers, groundwater and surface water are understood as intricately coupled systems with complex local-scale hyporheic exchange patterns and larger-scale gaining or losing stream sections [*Winter*, 1999; *Alley et al.*, 2002; *Sophocleous*, 2002]. Discharge in fractured rock watersheds has been examined previously [*Stephenson et al.*, 1992; *Rosenberry and Winter*, 1993; *Thorne and Gascoyne*, 1993; *Oxtobee and Novakowski*, 2002; *Cook et al.*, 2006; *Fan et al.*, 2007; *Praamsma et al.*, 2009] but significant questions remain about the rate, localization and conceptualization of discharge at a watershed scale. Additionally, groundwater and surface water may not be as intricately coupled due to the low permeability of fractured rock. Discharge and base flow have been examined in fractured rock watersheds using methods developed in porous media settings [*Mau and Winter*, 1997; *Risser et al.*, 2005, 2009]. Other studies examine and conceptualize discharge at discrete features such as faults, fracture zones, bedding planes or lineaments [*Stephenson et al.*, 1992; *Oxtobee and Novakowski*, 2002; *Fan et al.*, 2007; *Praamsma et al.*, 2009].

[3] Groundwater recharge, base flow and other important watershed characteristics are often estimated from hydrograph separation [*Rorabaugh*, 1964; *Moore*, 1992; *Rutledge and Daniel*, 1994; *Mau and Winter*, 1997; *Neff et al.*, 2005; *Risser et al.*, 2005, 2009]. Hydrograph or base flow methods are only applicable to gauged and unregulated watersheds with substantial groundwater discharge [*Fetter*, 2001]. Many medium to large rivers are, however, regulated by

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dams or other control structures [*Nilsson et al.*, 2005] and groundwater discharge in some rivers may be insignificant compared to streamflow. New methods for evaluating groundwater discharge in watersheds affected by dams or where groundwater discharge is minimal are essential. New and refined methods could lead to better prediction of low-flow conditions in gauged and ungauged watersheds and better characterization of groundwater-surface water interactions [*Kalbus et al.*, 2006; *Soulsby and Tetzlaff*, 2008; *Tetzlaff and Soulsby*, 2008].

[4] Natural tracers can provide important constraints on groundwater discharge rates and patterns. Radon (^{222}Rn) is a radioactive gas which is an excellent tracer of groundwater discharging to surface water bodies [*Rogers*, 1958; *Ellins et al.*, 1990; *Genereux et al.*, 1993; *Cook et al.*, 2006; *Charette et al.*, 2008; *Cook et al.*, 2008; *Stellato et al.*, 2008]. Radon accumulates in groundwater due to the radioactive decay of uranium and radium in aquifer materials and activities in groundwater are typically 1–2 orders of magnitude larger than surface water bodies, where radon is lost due to air-water exchange and radioactive decay. Temperature, specific conductance, major ions (e.g. chloride) and stable isotopes (δ^2H, δ^{18}O) can also be useful indicators of groundwater discharge to surface water bodies [*Lee*, 1985; *Krabbenhoft et al.*, 1990; *Harvey et al.*, 1997; *Constantz*, 1998; *Becker et al.*, 2004; *Conant*, 2004; *Cox et al.*, 2007; *Lowry et al.*, 2007]. Natural tracers have been previously used to estimate discharge rates and patterns but generally studies examine a single water body type (lake or river).

[5] The objective of this study is to evaluate the patterns and rates of groundwater discharge in a large, regulated fractured rock watershed using novel and standard methods that are independent of base flow recession. Natural conservative (δ^2H, δ^{18}O, Cl, and specific conductance), radioactive (^{222}Rn), and thermal tracers are integrated with streamflow measurements and a steady state advective model to delimit the discharge locations and quantify the discharge fluxes. Low gradient watersheds in crystalline fractured rock settings, such as the Canadian Shield, are often dominated by lakes and wetlands [*Farvolden et al.*, 1988; *Burn et al.*, 2008]. We examine multiple types of water bodies (lake, wetland, river and creek) using multiple methods for each. This study focuses on a large watershed underlain by fractured bedrock although the methodology developed is transferable to any large watershed. Since the base flow rate in this watershed is uncertain we use the term "low-flow" rather than "base flow" [*Smakhtin*, 2001; *Burn et al.*, 2008]. Also for clarity "discharge" refers to groundwater discharge whereas "streamflow" refers to the rate of water flow in a creek or river.

2. Regional Hydrology

[6] The study site is located in rural eastern Ontario, Canada in the ~900 km^2 Tay River watershed (Figure 1a) which is much larger than most previous hydrologic studies in the Canadian Shield [e.g. *Peters et al.*, 1995; *Devito et al.*, 1996; *Branfireun and Roulet*, 1998; *Buttle et al.*, 2001; *Spence and Woo*, 2003; *Buttle et al.*, 2004]. Previous studies in small watersheds focused on runoff and streamflow generation, surface water storage and surface-subsurface connectivity and emphasize the importance of the distribution of soil thickness. Groundwater discharge is limited where soil is minimal and perennial streams only develop in drainage areas >0.25–0.5 km^2 [*Buttle et al.*, 2004; *Steedman et al.*, 2004]. Previous research in a 2 km section of the Tay River (around SW1 and SW2 on Figure 1b) indicated that discharge to this section of the river was limited and standard isotope storm hydrograph methods are not appropriate because the Tay River is dominated by surface water flow [*Praamsma et al.*, 2009].

[7] The topography throughout the watershed is undulating with over 3000 mapped permanent surface water features. Three cold-bottomed lakes support trout populations and other lakes support warm water fish species. Four wetlands in the study area contain biodiversity that is designated provincially significant (Rideau Valley Conservation Authority, unpublished data). The humid climate is characterized by an average annual precipitation of 0.95 m (30 years of data from Environment Canada Station 6104027 in Kemptville, ON augmented with 3 years of onsite data). Precipitation is distributed relatively uniformly through out the year with typical summer precipitation of 0.08 to 0.1 m per month.

[8] Much of the watershed has minimal soil over Precambrian crystalline or flat-lying Paleozoic sedimentary units [*Gleeson and Novakowski*, 2009]. The headwaters and upper Tay River watershed are underlain by crystalline rocks and have large lake and wetland areas with small interconnecting creeks such as Uens and Eagle Creek. The water bodies of the upper watershed all flow into Bobs Lake which is regulated by the Bolingbroke dam. The Tay River begins below the Bolingbroke dam and is divided herein into the upper and lower Tay River by Christie Lake. The lower Tay flows over exposed crystalline bedrock and sedimentary units. Other tributary creeks such as Grants and Ruddsdale flow over sedimentary units and flow into the lower Tay River. Two small, unnamed creeks examined in detail in this study are herein called "Lineament Creek" and "Cameron Creek" because they cross the Christie Lake lineament [*Gleeson and Novakowski*, 2009] and Cameron Side Road, respectively.

[9] The Tay River is typically 5–10 m wide and less than 1 m deep and gauged at the Bolingbroke dam, Bowes Road and the town of Perth (Figure 1b). The only significant surface water abstraction from the Tay River is by an industrial plant near Bowes Road that removes <0.5% of the streamflow. Elsewhere, the sparse rural population predominantly uses groundwater supply. Examining the impact of water abstractions is outside of the scope of this research. Low-flow conditions for the Tay River typically occur in late July to early September [unpublished data from the Rideau Valley Conservation Authority] due to increased evapotranspiration. The surface water bodies typically have lower specific conductance (50–700 μs/cm) and warmer temperatures (15–25°C) in the summer, relative to groundwater in the fractured bedrock aquifers which typically have a specific conductance and temperature of 500–1500 μs/cm and 8–12°C, respectively.

3. Theory and Methodology

3.1. Approach

[10] Streamflow measurements are the most direct method for determining if rivers are gaining or losing along a reach

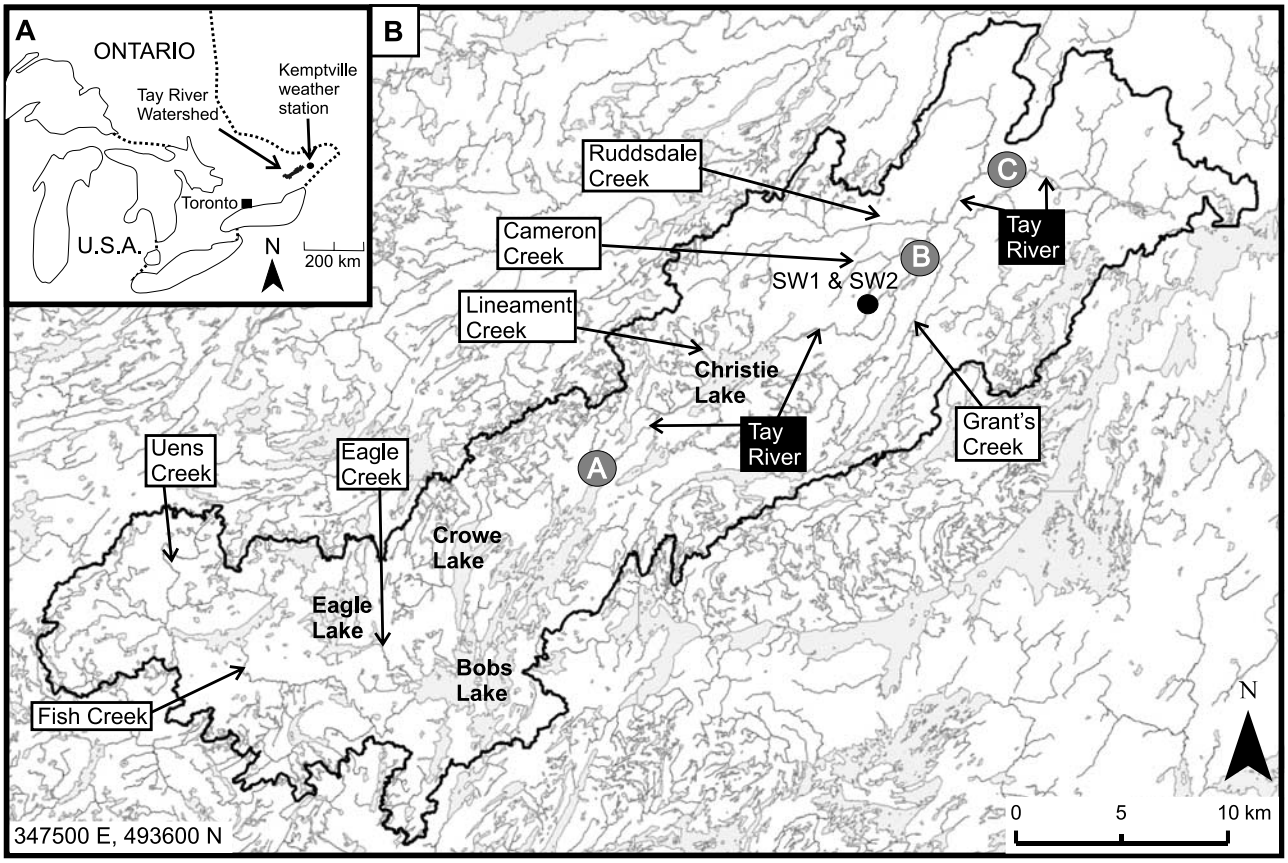

Figure 1. (a) Tay River watershed study location in Ontario. (b) The complex network of surface water features in the Tay River watershed. The watershed boundary is the black line. The creeks, large lakes, and surface water sampling locations (SW1 and SW2) by *Praamsma et al.* [2009] are shown for reference. The streamflow of the Tay River is measured at (A) the Bolingbroke Dam, (B) Bowes Road, and (C) the town of Perth. See *Gleeson and Novakowski* [2009] for the location of lineaments identified in Landsat and digital elevation model imagery.

but they are not as useful for constraining groundwater discharge to lakes and wetlands, due to larger uncertainties in other water budget terms such as evapotranspiration [*Winter*, 1981]. Therefore streamflow measurements are integrated with chemical, isotopic and thermal tracers to delimit the groundwater discharge locations and quantify the groundwater discharge fluxes in lakes, wetlands, creeks and at various locations along the Tay River. Discharge is identified by exploiting the chemical, isotopic and thermal differences between groundwater and surface waters, which are accentuated in the low-flow summer months due to warmer temperatures and evaporation in the surface water bodies. Chloride, radon and streamflow data are used to quantify discharge rates. Stable isotope, specific conductance and thermal data are used to qualitatively evaluate discharge patterns due to uncertainty in fractionation factors and limited observation of specific conductivity and temperature anomalies.

[11] Twenty-one representative lakes and wetlands were sampled for δ^2H, δ^{18}O, Cl and specific conductance in May and August of 2006 and 2007 to evaluate the relative influence of evaporation and groundwater discharge. Groundwater consistently plots close to the local meteoric water line [*Praamsma et al.*, 2009] whereas during the summer months open water bodies fractionate along a distinct isotopic trajectory with a slope of ~5 on the plot of δ^2H versus δ^{18}O [*Gonfiantini*, 1986; *Clark and Fritz*, 1997]. Surface water features that are influenced by groundwater discharge would plot along a mixing line between evaporated rainfall and groundwater, although as the isotopic composition of groundwater would be expected to be similar to mean annual rainfall, this mixing line might be difficult to distinguish from the aforementioned evaporation line. Radon activities and chloride concentrations in groundwater are significantly greater than in surface water, and so elevated activities or concentrations of these tracers provide indications of groundwater discharge. Furthermore, because of the short half-life of radon and its propensity to be lost to the atmosphere, high activities of radon provide evidence of relatively recent groundwater discharge. In contrast, the residence time of chloride in lakes and wetlands can be much longer, and high chloride concentrations can also be due to evaporative enrichment. ^{222}Rn activities and Cl concentrations measured during one week in August 2008 are used to quantify groundwater discharge rates using a steady state advective model described in section 3.2.

[12] Specific conductance, temperature and ^{222}Rn activities were continuously measured along a transect of the Tay

River and Christie Lake. The Tay River was also sampled for δ^2H, δ^{18}O, Cl and specific conductance in May and August of 2006 and 2007 and this data is interpreted qualitatively. Additionally, streamflow rates for three gauging stations located on the Tay River are compared to determine if the Tay River is gaining with distance downstream. A number of creeks were sampled at multiple access points along their reach for δ^2H, δ^{18}O, Cl, specific conductance and ^{222}Rn to qualitatively identify groundwater discharge patterns. Streamflows were also manually measured at the multiple access points to determine if streamflow increases with distance downstream.

3.2. Steady State Advective Model

[13] A steady state advective model is developed to estimate the rates of groundwater discharge and surface water inflow to lakes and wetlands in a large watershed (Figure 2). The model represents the flux of groundwater and surface water in the days before sampling due to the short half-life of ^{222}Rn. Since the residence time of chloride is much greater than radon, the importance of the steady state assumption for chloride is evaluated below in the uncertainty analysis. For steady state conditions the water budget of a lake or wetland system can be expressed:

$$\frac{\partial V}{\partial t} = I_s + I_g + PA - EA - Q = 0 \tag{1}$$

where V is the water volume (m^3), I_s is the surface water inflow rate (m^3/day), I_g is the groundwater discharge rate (m^3/day), Q is the combined surface water and groundwater outflow rate (m^3/day), P is the rate of direct precipitation to the water surface (m/day), E is the evaporation rate from the water surface (m/day), A is the surface water area (m^2) and t is time. Similarly, the conservative solute balance of a lake or wetland system can be expressed:

$$\frac{\partial cV}{\partial t} = I_s c_s + I_g c_g + PAc_p - kAc - Qc - \lambda Vc = 0 \tag{2}$$

where k is the gas exchange velocity (m/day), λ is the radioactive decay constant (day^{-1}) and c, c_s, c_g and c_p are the solute concentration (mg/L) or activity (Bq/L) of the surface water, surface water inflow, groundwater discharge and precipitation, respectively. Equations (1) and (2) can be combined and solved as:

$$c = \frac{I_s c_s + I_g c_g + PAc_p}{I_s + I_g - EA + PA + kA + \lambda V} \tag{3}$$

Gas exchange and radioactive decay are negligible for a conservative ionic tracer, such as chloride. Therefore for chloride, equation (3) is simplified:

$$c_{Cl} = \frac{I_s c_{sCl} + I_g c_{gCl} + PAc_{pCl}}{I_s + I_g - EA + PA} \tag{4}$$

Radon is an unreactive, radioactive gas with a negligible activity in the atmosphere and precipitation. The only source is radioactive decay of uranium and radium in aquifer materials and surface water inflow (c_s) if an upstream water body has significant radon activity. Therefore equation (3) can also be expressed as

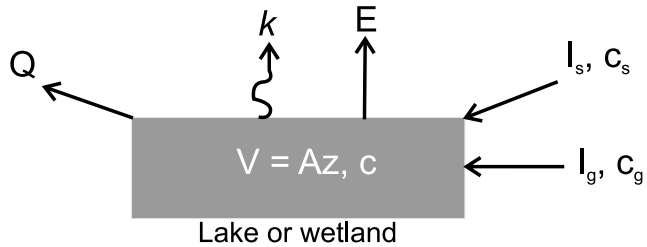

Figure 2. Schematic of the steady state advective model with the variables defined in the text.

$$c_{Rn} = \frac{I_s c_{sRn} + I_g c_{gRn}}{I_s + I_g - EA + PA + kA + \lambda V} \tag{5}$$

For this equation we assume that radium activities in the lake and the diffusive radon flux from the lake sediments are negligible, which will usually be the case. Using this assumption, a maximum groundwater discharge flux is calculated like other methods used in this study (see section 5.1).

[14] Equations (4) and (5) can be simultaneously solved for I_g and I_s using measurements of c_{Cl}, c_{sCl}, c_{gCl}, c_{pCl}, c_{Rn}, c_{sRn}, c_{gRn}, A and V, estimates of P, E and k, and λ = 0.18 day^{-1}. Chloride concentrations in the surface water inflows (c_{sCl}) were assigned from measurements of the inflowing creek, the upstream lake if the inflowing creek was inaccessible or the concentration in precipitation (c_{pCl} = 0.1 mg/L) if the lake or wetland does not have significant surface water inflow (i.e., headwater lake). Chloride precipitation values (c_{pCl}) are from a meteorological station 190 km north of the study area [*NATChem*, 2008]. For lakes and wetlands with radon activities below detection, the detection limit was used as the radon activity (c_{Rn}) for the steady state advective model which provides a maximum rate of groundwater discharge (I_g). Groundwater radon activities (c_{gRn}) and chloride concentrations (c_{gCl}) were measured throughout the watershed but primarily near the hay field research site as discussed in section 3.3. The surface area and volume of the lake or wetland is extracted from a provincial GIS database of surface water bodies or bathymetric surveys of the smaller water bodies and are considered <20% uncertain [*Winter*, 1981]. Monthly precipitation rate (P = 0.002 m/d) and lake evaporation rate (E = 0.005 m/d) was measured at a weather station ~50 km from the watershed (Figure 1a) and are considered <50% uncertain [*Winter*, 1981]. Lake evaporation is calculated using the observed daily values of pan evaporative water loss, the mean temperatures of the water in the pan and of the nearby air, and the total wind run over the pan. A gas exchange velocity of k = 0.16 m/d was calculated for a wetland in South Australia by *Cook et al.* [2008] using an injection of SF_6 tracers, and we have adopted this value for our model. This gas exchange velocity is also within the range of values derived for radon from other hydraulic settings [*Wanninkhof et al.*, 1990; *Corbett et al.*, 1997; *Kluge et al.*, 2007] and is considered <20% uncertain. Using a similar model, *Cook et al.* [2008] showed that the solution for groundwater discharge (I_g) is much less sensitive to evapo-

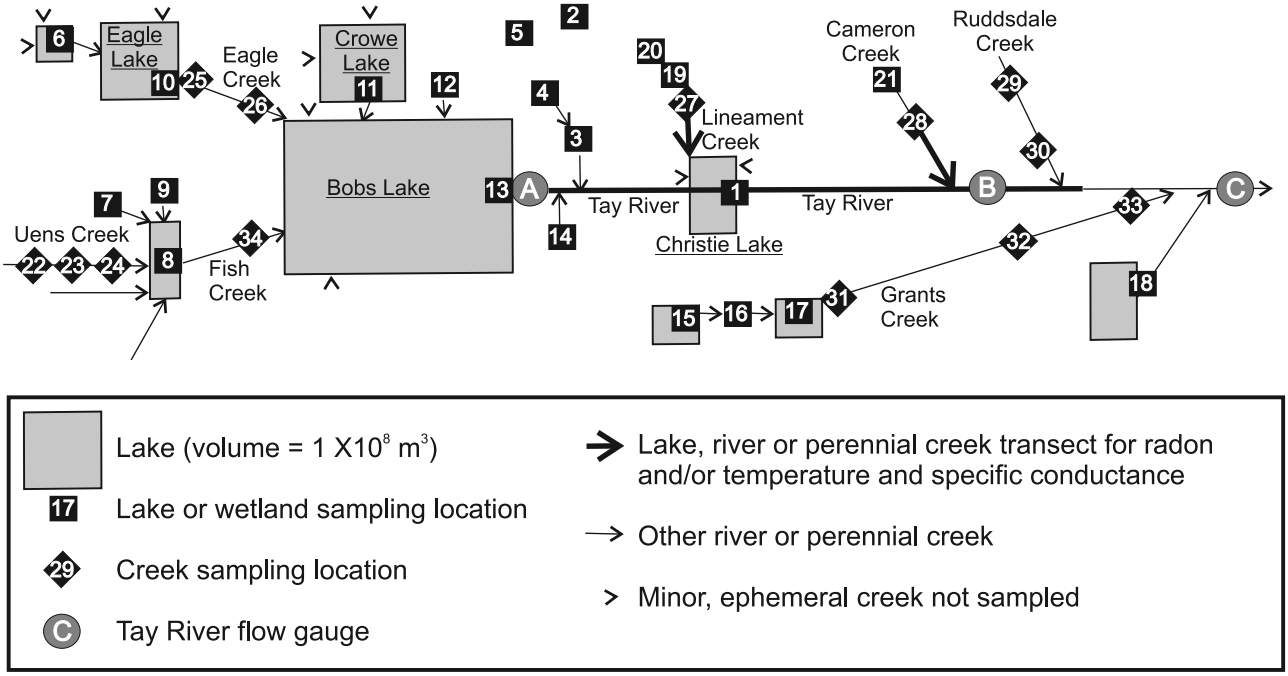

Figure 3. Schematic of surface water bodies sampled during the study. See Table 1 for the geographic name of each numbered sample location. The lakes and wetlands are scaled by volume. Water bodies that are smaller than the size of the label ($\sim 1 \times 10^7$ m^3) are only labeled. Both perennial and ephemeral creeks are shown. The Tay River is gauged at (A) the Bolingbroke dam, (B) Bowes Road, and (C) the town of Perth.

ration rate and gas exchange rate than to groundwater radon activity (c_{gRn}).

[15] The steady state advective model is considered a screening level tool that is useful for analyzing synoptic chloride and radon data gathered from a large watershed. The model is appropriate for watershed-scale quantification of discharge (see section 5.1) or constraining recharge patterns to focus detailed studies but would be inappropriate for analyzing data from a single water body [*Corbett et al.*, 1997; *Kluge et al.*, 2007; *Cook et al.*, 2008] because of the inherent assumptions. The model accounts for differences in evaporation, gas exchange and radioactive decay in different surface water bodies using estimates of area and volume. Significant assumptions of the model are that (1) the chloride concentration and radon activity are at steady state in the surface water body; (2) the surface water body is well mixed (i.e., the collected surface water samples are representative) and (3) representative groundwater chloride concentrations and radon activities are available. The validity of the assumptions is discussed briefly here and evaluated more fully in a sensitivity analysis (section 4.2) where each input parameter is varied over the expected range of potential uncertainty. Since radon has a very short half-life the primary concern with the steady state assumption is chloride which has a longer residence time in lakes. Therefore the appropriateness of the steady state assumption is tested by examining the sensitivity of I_g to uncertainty in measured lake chloride concentrations.

[16] Radon activities measured repeatedly in Christie Lake were consistent both spatially and temporally suggesting a representative activity was measured. But activities in other larger, deeper water bodies could be heterogeneous both vertically and horizontally due to thermal stratification and incomplete wind mixing, respectively [*Kluge et al.*, 2007]. Radon activities in the lake documented by *Kluge et al.* [2007] were highest in the thermocline and lower in the epilimnion (due to gas exchange) and hypolimnion (due to limited groundwater water discharge at depth). In this study, the stratified lakes were sampled from the epilimnion where groundwater discharge is generally focused [*Winter*, 1978; *Kluge et al.*, 2007]. In shallow wetlands mixing may also be limited [*Cook et al.*, 2008]. Groundwater radon activities in crystalline aquifers are highly variable at the kilometer scale [*Folger et al.*, 1996; *Veeger and Ruderman*, 1998; *Wood et al.*, 2004] suggesting it may be difficult to estimate activities in the vicinity of the water body. The model sensitivity to uncertainty in radon activities is evaluated in section 4.2.

3.3. Field and Laboratory Methods

[17] Field work was conducted with varying temporal resolution in the summers 2005, 2006, 2007 and 2008 during low-flow conditions in late July–August. Sampling locations of the lakes, wetlands, creeks and along the Tay River are represented on Figure 3. One location at or near the outlet of each lake or wetland was sampled. The creeks and the Tay River were sampled near the midway point of the reach. Multiple samples with depth or across the reach were not collected for the creeks or Tay River because they are generally very shallow (<0.5 m) and well mixed [*Praamsma et al.*, 2009].

[18] In terrestrial hydraulic systems, radon is typically discretely sampled and analyzed using liquid scintillation methods [*Rogers*, 1958; *Cook et al.*, 2006, 2008]. Recently, continuous, real-time, *in situ* radon measurement developed by the oceanographic community [*Burnett et al.*, 2001] have

been used to measure radon activities in a lake [*Kluge et al.*, 2007]. A commercial radon-in-air detector (RAD7®) is outfitted with an air-water exchanger using the "RAD-AQUA®" methodology. Surface water is pumped continuously into the air-water exchanger and the activity of ^{222}Rn-in-air (which equilibrated with the surface water) is calculated by measurement of the α-emitting daughters ^{214}Po and ^{218}Po. Radon-in-water activities are calculated from radon-in-air activities using the temperature dependence of the air-water phase equilibrium of radon [*Burnett et al.*, 2001]. Detection limits are the activities which can be counted with a precision of ±100% at the 95% confidence level [*EPA*, 2002]. Radon activities are measured over 10 minute intervals with a detection limit of 0.02 Bq/L. For the Tay River, the radon was measured continuously by canoeing slowly with the radon measuring apparatus. For individual lakes, wetlands and creeks three 10 minute intervals after the air and water equilibrated are integrated to lower the detection limit to 0.01 Bq/L [*Kluge et al.*, 2007]. Groundwater samples were collected from residential wells throughout the watershed and from 25 multi-level piezometers completed at 5–50 m below ground surface near SW1 and SW2 on Figure 1 [*Gleeson et al.*, 2007; *Levison and Novakowski*, 2009; *Praamsma et al.*, 2009]. Groundwater samples were collected after purging for three well volumes and radon activity was analyzed using the Rad-H_2O methodology [*Kluge et al.*, 2007] with a typical uncertainty of ±5%.

[19] A temperature and specific conductance probe was manually dragged along the bottom of a 25 km long reach of the Tay River and four creeks [*Lee*, 1985; *Harvey et al.*, 1997]. The probe is considered accurate to ±0.15°C and ±1 μs/cm for temperature and specific conductance, respectively. The probe was directly inserted into open bedrock fractures or other potential discharge features in the middle of the reach as well as near the banks. During each transect temperature and specific conductance was logged at 1 second intervals for a total of more than 14500 individual temperature and specific conductance readings. Differential temperature and specific conductance values were calculated (daily mean minus individual value) and are reported below such that measurements from different days and locations are directly comparable. Potential groundwater discharge locations are identified by a brief negative temperature excursions and/or positive specific conductance excursions. During the transects, the type of river bed (soil type or rock type) as well as the fracture density in the exposed bedrock river bottom were also mapped.

[20] The streamflow of the Tay River was measured hourly and integrated into daily mean streamflow at the three gauging stations by the Rideau Valley Conservation Authority. Streamflow was also measured manually in monthly surveys and are considered accurate to ±5% for all levels of the rating curve by the National Water Research Institute of Canada. Streamflow measured at different locations along the Tay River are compared to determine if the Tay River is gaining or losing. Additionally, streamflow measurements using a pygmy Price AA flowmeter in four creeks were made at multiple locations along their reach during low-flow conditions following the *Hinton* [2005] method. Uncertainties accrue due to a number of factors including the number of verticals the cross section is divided into, the length of measurement time and the flow velocity. Random and systematic uncertainties in the individual streamflow measurements were calculated to estimate potential error [*Hinton*, 2005].

[21] The stable isotopic analyses were completed at the Queen's Facility for Isotope Research using Finnigan MAT 252 and ThermoFinnigan Delta Plus XP mass spectrometers for ^{2}H and ^{18}O, respectively. Isotope values are expressed in δ units (‰, parts per mil) relative to Vienna Standard Mean Ocean Water (VSMOW). Analytical error was approximately ±1‰ for ^{2}H and ±0.1‰ for ^{18}O. Surface waters were analyzed for major and minor elements using ion chromatography and inductively coupled plasma mass spectroscopy but only chloride concentrations are reported here.

4. Results

4.1. Stable Isotopes and Chloride

[22] The lakes and wetlands of the Tay River watershed plot along a well-defined evaporative trajectory with an equation $\delta^2H = 4.7\delta^{18}O + 22.2 \times 10^{-3}$ (Figure 4). The lakes and wetlands have a range of isotopic values (i.e., −4.2 to −9.5‰ δ^{18}O in August 2006) and a range of differences between spring and summer values of 0.3 to 2.4‰ δ^{18}O. The differences between spring and summer isotopic values are a product of evaporative enrichment which increase isotopic values and/or groundwater discharge which causes a decrease in isotopic value as surface waters are mixed with isotopically depleted groundwater.

[23] The creeks plot along an evaporative trend with a $\delta^2H/\delta^{18}O$ slope of ~4.7 like the lakes and wetlands (Figure 4a). The relative influence of groundwater discharge versus evaporation can be directly evaluated using stable isotopes by sampling the creeks at multiple locations along their reach on the same day (Figure 4b). Cameron Creek plots directly in the groundwater field (Figures 4a and 4b) suggesting that the rate of groundwater discharge is high relative to the evaporation rate. The isotopic values of Ruddsdale Creek decrease with distance downstream, suggesting significant groundwater inflow within the sampled reach. Upper Uens Creek has a similar trend suggesting this headwater stream is groundwater dependent. However, lower Uens Creek and Eagle Creek have stagnant stretches and isotopic data suggests that evaporation is a more significant process in these systems (Figure 4b). Grants Creek and Lineament Creek do not show a significant downstream isotopic trend, suggesting that groundwater discharge and evaporative loss represent only a relatively small proportion of the streamflow on the sampled reach.

[24] The influence of evaporation versus groundwater discharge for lakes and wetlands is also distinguishable when either stable isotope (δ^2H or δ^{18}O) is plotted against chloride. Figure 4c illustrates δ^{18}O versus chloride as an example. Chloride is a conservative ionic tracer that increases due to evaporation in surface waters or water-rock interactions in groundwater. A minority of shallow wetlands are significantly evaporated and a minority of creeks may be groundwater dependent (Figure 4c). However, most lakes and creeks are not significantly affected by either evaporation or groundwater discharge. Using stable isotopes in conjunction with chloride it is possible to qualitatively determine the relative influence of evaporation and groundwater discharge.

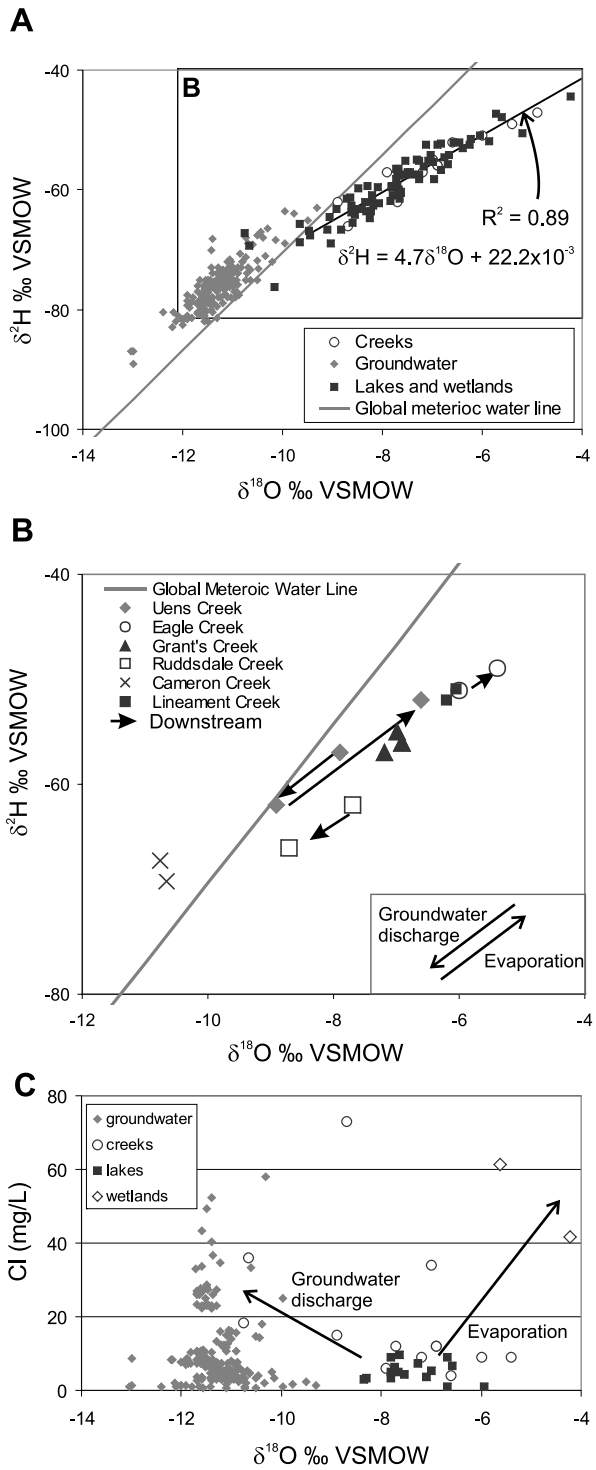

Figure 4. (a) Groundwater and surface water stable isotope data from the Tay River watershed. (b) Low-flow stable isotope value along the different creeks, sampled on the same day. The laboratory uncertainty bars are equivalent to the size of the symbol. The arrows indicate sampling locations downstream from previous samples. Streams that are influenced by groundwater discharge or evaporation are differentiated by the direction of isotopic shift. (c) Chloride versus $\delta^{18}O$ differentiates the influence of groundwater discharge or evaporation for lakes, wetlands, and creeks. Groundwater data by *Praamsma et al.* [2009] and *Levison and Novakowski* [2009].

4.2. Radon and the Steady State Advective Model

[25] The majority of the surface water bodies in the Tay River watershed have insignificant radon activities whereas groundwater activities are 2–3 orders of magnitude larger (Figure 5a). The median and upper quartile activities for lakes and wetlands are below detection at 0.01 Bq/L. Radon activities in Christie Lake were measured repeatedly over one week and in a transect across the lake (Figure 6). The activities varied within standard deviation both spatially and temporally suggesting the epilimnion of the lake is well mixed. For creeks the upper quartile is 0.05 Bq/L but the median is below detection like the lakes and wetlands. The largest value for the wetlands and creeks is the Cameron Creek headwater and reach, respectively (Figure 5a).

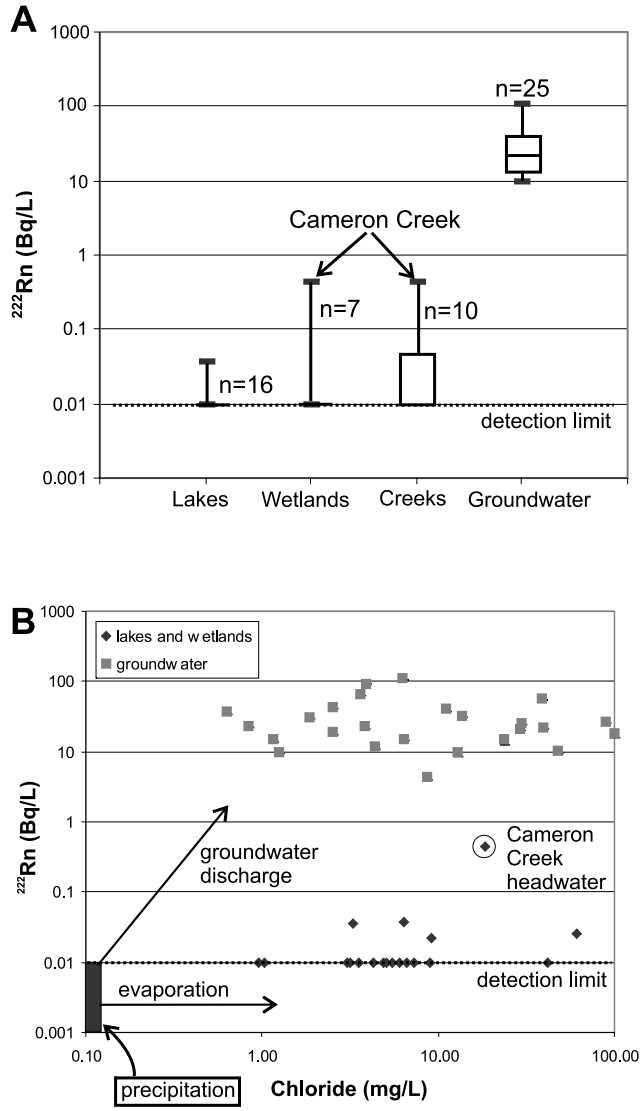

Figure 5. (a) Box-and-whisker plot of ^{222}Rn activities in lakes, wetlands, creeks, and groundwater. The upper quartile of lakes and wetlands is below detection. The headwater of Cameron Creek is a wetland where the maximum value for all wetlands was detected. Groundwater activities are one to two orders of magnitude larger. (b) ^{222}Rn activities versus chloride concentrations which is the primary input data for the steady state advective model.

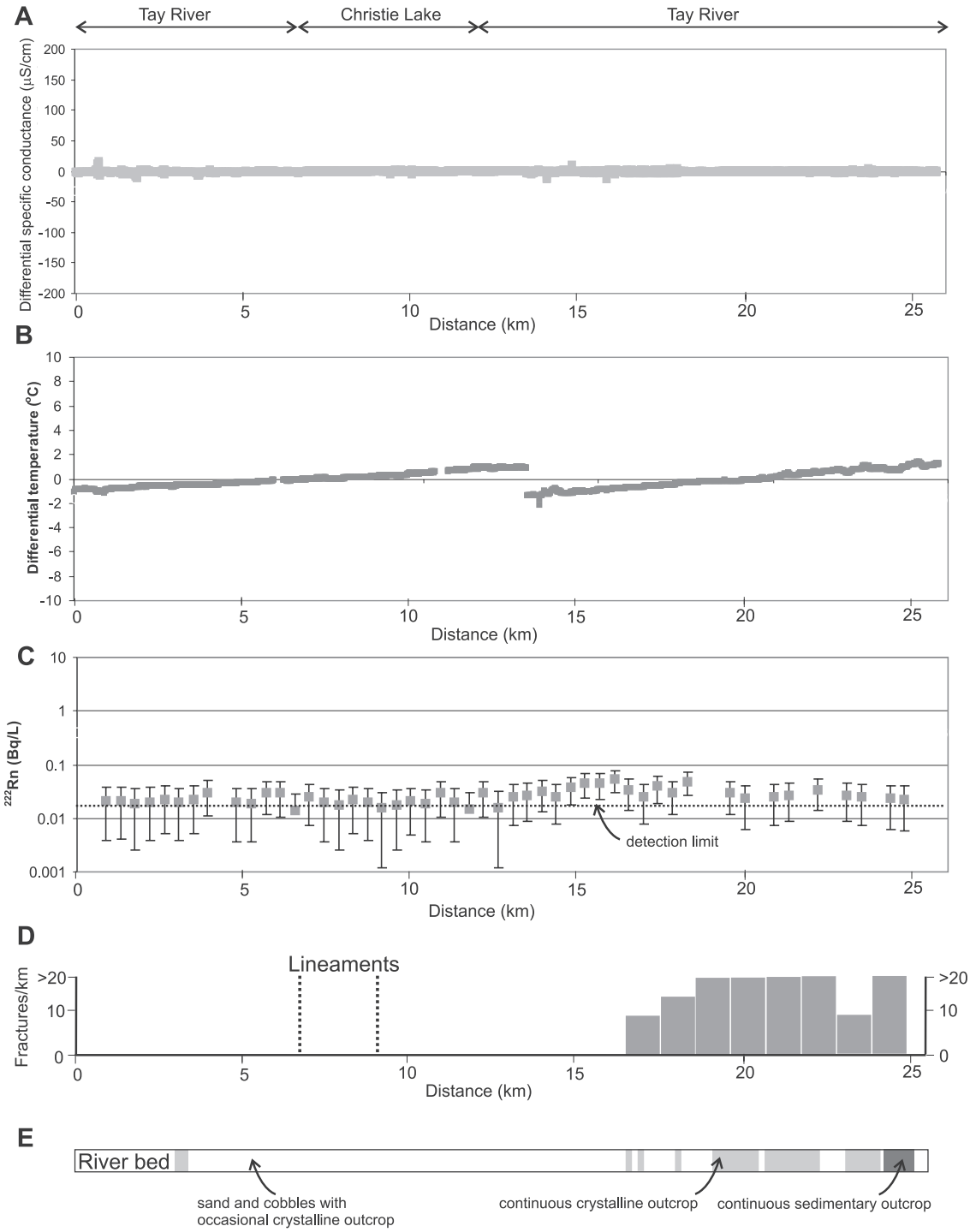

Figure 6. Specific conductance, temperature, ^{222}Rn, fracture density, and river bed data from a 25-km-long transect of the Tay River and Christie Lake. Groundwater discharge is identified by high specific conductance, high ^{222}Rn activities, and low temperature. Differential (a) specific conductance and (b) temperature is the difference between individual data points and the daily mean so that data from different days are comparable. Transects were completed over 2 days, and the water temperatures fluctuate diurnally. The apparent offset in temperature around km 13 is due to the difference between morning and afternoon temperatures. (c) ^{222}Rn activities from the Tay River. (d) Fracture density, lineament location, and (e) river bed type are plotted for reference.

Groundwater radon activities range from 9.8 to 112.1 Bq/L with a median value of 22.9 Bq/L.

[26] Radon activities in the Tay River are consistently low, at or near the detection limit, and not correlated with lineament location or density of fractures in the exposed bedrock river bottom (Figure 6). Radon activities in the individual creeks support the stable isotope results (Table 1). Cameron Creek has the highest radon activity (0.444 ± 0.022 Bq/L)

Table 1. Geochemical Data and Results of Steady State Advective Model

Sample	Sample Location	Mean Depth (m)	Area (km^2)	Volume (1×10^6 m^3)	$10^3\delta^{18}O$ VSMOW	$10^3\delta^2H$ VSMOW	$10^3\delta^{18}O$ VSMOW	$10^3\delta^2H$ VSMOW	Cond. (μS/cm)	Cl (mg/L)	^{222}Rn (Bq/L)	±	I_g (m^3/day)	I_s (m^3/day)	I_g/V	I_g/I_s
Lakes and wetlands					5/18/06		7/31/06			7/29–31/08						
1	Christie Lake	8.5	7	60	−8.2	−68	−7.6	−61	154	5	0.025	0.001	13,065	321,400	2.2E−04	4.1E−02
2	Unnamed wetland	1.2	0.1	0.1	−8.8	−73	−5.6	−54	386	61	<0.01	–	10	296	9.7E−05	3.3E−02
3	Davern Lake	15	0.5	8	−7.0	−64	−6.9	−58	259	5	<0.01	–	621	25,498	8.3E−05	2.4E−02
4	Little Silver Lake	6	0.6	4	−8.3	−70	−7.7	−63	147	5	<0.01	–	325	2403	9.0E−05	1.4E−01
5	Unnamed wetland	1.3	0.1	0.1	−7.6	−66	−4.2	−50	215	42	<0.01	–	11	297	1.1E−04	3.8E−02
6	Leggat Lake	10	1.8	18	−5.9	−58	−5.7	−53	70	1	<0.01	–	1559	26,611	8.7E−05	5.9E−02
7	Miller Lake	4	0.3	1.2	−8.6	−69	−7.3	−61	87	3	0.038	0.005	443	2048	3.7E−04	2.2E−01
8	Long Lake	11.5	3.4	39	−8.7	−67	−7.5	−63	116	3	0.036	0.005	11,880	40,765	3.0E−04	2.9E−01
9	Abbot Lake	2	0.1	0.2	−8.5	−66	−6.5	−58	140	4	0.022	0.006	51	461	2.6E−04	1.1E−01
10	Eagle Lake	15	6.4	96	−6.8	−58	−6.2	−58	134	7	<0.01	–	8018	29,333	8.4E−05	2.7E−01
11	Crowe Lake	25	4.4	110	−8.1	−66	−7.7	−64	149	3	<0.01	–	9018	36,353	8.2E−05	2.5E−01
12	Sucker Lake	2	0.3	0.6	−8.9	−69	−6.8	−61	164	1	<0.01	–	69	2291	1.1E−04	3.0E−02
13	Bob's Lake	20	29.5	590	−7.7	−65	−7.4	−63	145	4	<0.01	–	48,730	560,688	8.3E−05	8.7E−02
14	Farren Lake	8.3	1.7	14	−7.1	−59	−6.3	−59	156	9	<0.01	–	1225	5647	8.7E−05	2.2E−01
15	Big Crosby Lake	12	2.2	26	−8.1	−68	−7.7	−63	145	5	<0.01	–	2236	8974	8.5E−05	2.5E−01
16	Little Crosby Lake	9	0.6	5.4	−8.2	−69	−7.2	−62	147	6	<0.01	–	467	2291	8.6E−05	2.0E−01
17	Pike Lake	8.4	3.3	28	−7.7	−67	−7.7	−64	175	9	0.010	0.007	2387	35,738	8.6E−05	6.7E−02
18	Otty Lake	9	6.7	60	−8.5	−70	−7.3	−61	229	7	<0.01	–	5210	24,299	8.6E−05	2.1E−01
19	TW14 Lower wetland	1.5	0.01	0.02	–	–	−6.0	−57	–	–	–	–	–	–	–	–
20	TW14 Upper wetland	1.7	0.0001	0.0002	–	–	−7.0	−60	–	–	–	–	–	–	–	–
21	Cameron Creek headwater	1.1	0.0001	0.0001	–	–	−9.5	−73	403	18	0.451	0.022	0.61	0.03	6.1E−03	2.0E+01
Tay River		–	–	–	−8.2	−62	−7.2	−62	135	–	0.02–0.05	–				
Creeks							7/31/07			7/29–31/08						
22	Uens Creek (upper)	–	–	–	–	–	−7.9	−57	281	4	0.052	0.005				
23	Uens Creek (middle)	–	–	–	–	–	−8.9	−62	180	15	–	–				
24	Uens Creek (lower)	–	–	–	–	–	−6.6	−52	181	6	–	–				
25	Eagle Creek (upper)	–	–	–	–	–	−6.0	−51	144	9	<0.01	–				
26	Eagle Creek (lower)	–	–	–	–	–	−5.4	−49	146	9	<0.01	–				
27	Lineament Creek	–	–	–	–	–	−6.0	−51	125	–	<0.01	–				
28	Cameron Creek	–	–	–	–	–	−11.5	−81	616	36	0.444	0.022				
29	Ruddsdale Creek (upper)	–	–	–	–	–	−7.7	−62	382	12	0.012	0.002				
30	Ruddsdale Creek (lower)	–	–	–	–	–	−8.7	−66	696	73	0.041	0.003				
31	Grant's Creek (upper)	–	–	–	–	–	−7.2	−57	185	9	<0.01	–				
32	Grant's Creek (middle)	–	–	–	–	–	−6.9	−56	198	12	<0.01	–				
33	Grant's Creek (lower)	–	–	–	–	–	−7.0	−55	231	34	<0.01	–				
34	Fish Creek	–	–	–	–	–	–	–	181	3	<0.01	–				

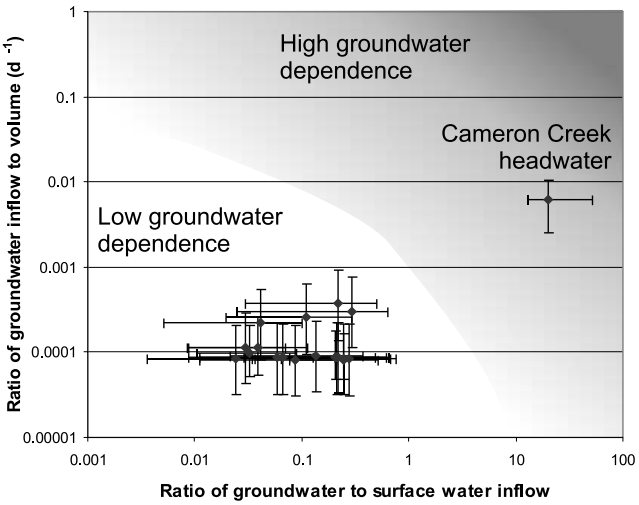

Figure 7. Results of the steady state advective model indicate that most lakes and wetlands are not dependent on groundwater except the headwaters of Cameron Creek. Error bars, range of values calculated using different groundwater ^{222}Rn activities.

suggesting it is primarily groundwater discharge. Radon activities in Ruddsdale Creek increase downstream from 0.012 ± 0.002 to 0.041 ± 0.003 Bq/L and Uens Creek also has measurable radon activities of 0.052 ± 0.005 Bq/L, suggesting these creeks have a groundwater component. Eagle Creek, Lineament Creek and Grants Creek are all below detection for radon activities, consistent with stable isotopic results.

[27] Figure 5b compares radon activities and chloride concentrations. The headwater of Cameron Creek is an outlier between the groundwater and surface water populations. Estimates of groundwater discharge were made using the steady state advective model described in section 3.2. Median groundwater values of 22.9 Bq/L and 10 mg/L were assumed for radon and chloride, respectively. Table 1 compiles lake area and volume as well as radon activities and chloride concentrations and the resulting estimates of surface water inflow (I_s) and groundwater discharge (I_g). Groundwater dependence is quantified by the ratio of groundwater discharge to volume of the wetland (I_g/V) and the ratio of groundwater to surface water inflow (I_g/I_s). Results indicate that all of the lakes and wetlands have low groundwater dependence, except the headwaters of Cameron Creek (Figure 7). The minimum, median and maximum ratio of I_g/I_s for individual lakes and wetlands are 2%, 14% and 29%, respectively. The groundwater/surface water inflow ratio for most lakes and wetlands is <20% suggesting that the Tay River watershed is generally not a groundwater-dependent system.

[28] The sensitivity of I_g/I_s and I_g/V ratios to a range of the groundwater ^{222}Rn activities (lower to upper quartile from the box-and-whisker plot in Figure 5a) was evaluated. This range is equivalent to an uncertainty of ±50% of the median groundwater ^{222}Rn activity. Figure 7 indicates that the uncertainty in groundwater ^{222}Rn activities does not affect the interpretation of groundwater dependence. Watershed-scale groundwater discharge (I_g) is most sensitive to groundwater radon activity (c_{gRn}), surface water radon activity (c_{Rn}), volume (V), and area (A) although volume and area are less uncertain than radon activities (Figure 8). Each parameter was varied over the expected range of potential uncertainty. Surface water radon activities were varied by ±50% which is the approximate range of heterogeneity documented in a thermally stratified lake [*Kluge et al.*, 2007] and a shallow wetland with limited mixing [*Cook et al.*, 2008]. Section 3.2 outlines the uncertainty for other parameters. Groundwater discharge is less sensitive to gas exchange velocity (k), evaporation (E) and chloride concentrations in groundwater (c_{gCl}) and surface water (c_{Cl}). The lack of sensitivity to surface water chloride concentration (c_{Cl}) suggests that model results are not highly sensitive to the steady state assumption (see section 3.2). The sensitivity to groundwater (c_{gRn}) and surface water (c_{Rn}) radon activity suggests that the more important assumptions are a well-mixed surface water body and a representative groundwater radon activities (see section 5.2). Since the model is sensitive to radon activities, the sensitivity of watershed-scale groundwater discharge and surface water inflows to groundwater ^{222}Rn activities was evaluated in more detail (Table 2). The groundwater discharge (I_g) varies significantly but importantly, the ratio of groundwater discharge to surface water inflow (I_g/I_s) remains low and is relatively insensitive to groundwater radon activities suggesting the lakes and wetlands of the Tay River watershed are not groundwater dependent.

4.3. Temperature and Specific Conductance Transects

[29] Discrete groundwater discharge locations were mapped using high-resolution transects with a temperature and specific conductance probe of the Tay River and Christie Lake (Figure 6) as well as Lineament Creek and Cameron Creek (Figure 8). The transect in Figure 6 was completed over two days and the water temperatures fluctuate diurnally. The apparent offset in temperature around km 13 in Figure 6b is due to the difference between morning and afternoon temperatures. For the upper and lower Tay River and Christie Lake, no temperature and specific conductance anomalies

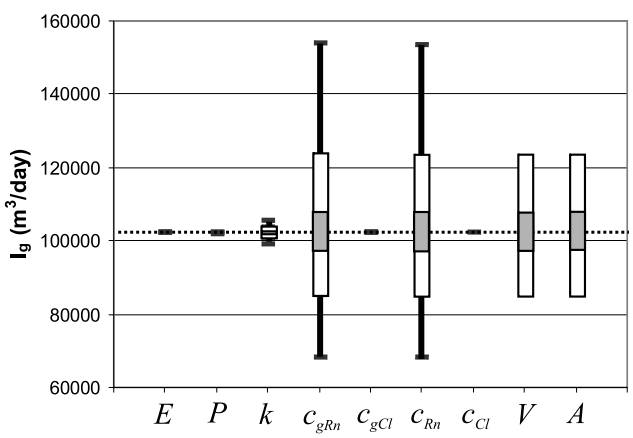

Figure 8. Sensitivity analysis of the total groundwater discharge for the sampled lakes using the steady state advective model. Each parameter is varied over the expected range of potential uncertainty. Gray and white rectangles depict ranges in I_g from ±5% and ±20% uncertainty in each parameter, respectively, while vertical lines represent ±50% uncertainty. The broken line represents the value of ~102,000 m^3/day estimated using the measured values from Table 1 as described in the text.

Table 2. Total Groundwater Discharge and Surface Water Inflow Rates for the 21 Sampled Lakes and Wetlands From the Steady Steady Advective Model With Variable Groundwater ^{222}Rn Activities

c_{gRn}	I_g (m^3/day)	I_s (m^3/day)	I_g/I_s	I_g/V (day^{-1})
36 Bq/L (upper quartile)	64,157	805,441	0.08	0.00006
23 Bq/L (median)	102,305	1,118,595	0.09	0.00010
15 Bq/L (lower quartile)	156,717	1,177,372	0.13	0.00016

were detected over a 25 km transect representing over 10,500 individual temperature and specific conductance measurements. Figure 6 also compiles type of river bed, lineaments that cross the river and fracture density in the exposed bedrock river bottom for the 25 km transect of the Tay River and Christie Lake. From the detailed transects of specific conductance, temperature and radon activities it is clear that discharge is not localized at lineaments or in zones of exposed, high-density fracturing.

[30] Lineament and Cameron Creeks show larger temperature and specific conductance anomalies, likely due to their lower streamflow and smaller sizes (Figure 9). In Cameron Creek, a significant positive specific conductance anomaly and negative temperature anomaly was found near the beginning of the transect indicating localized groundwater discharge. The reach of Cameron Creek examined was downstream of the headwater reach of Cameron Creek which was not accessible. In Lineament Creek a positive specific conductance anomaly and negative temperature anomaly was not found.

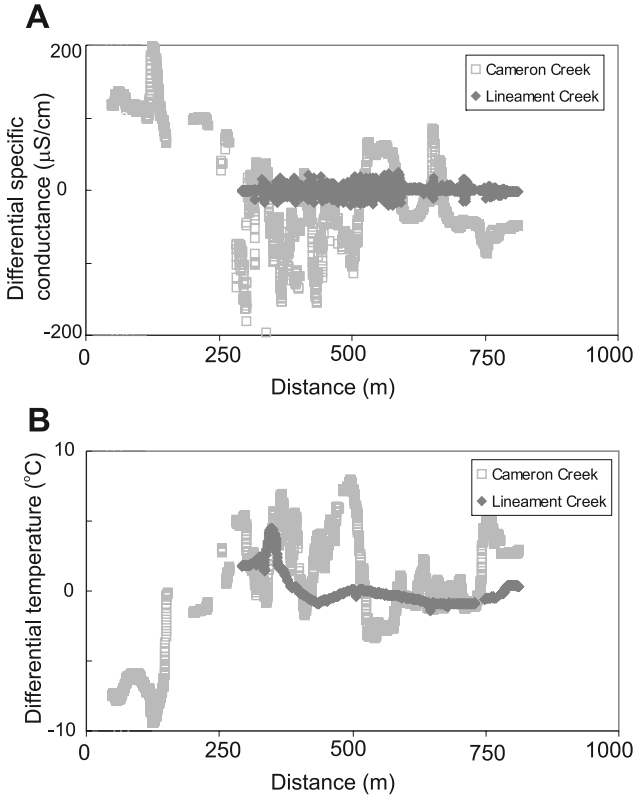

Figure 9. Transects of Cameron Creek and Lineament Creek showing (a) differential specific conductance and (b) differential temperature. Differential values are the mean of the whole transect minus the individual measurement. The upper part of Cameron Creek was not accessible.

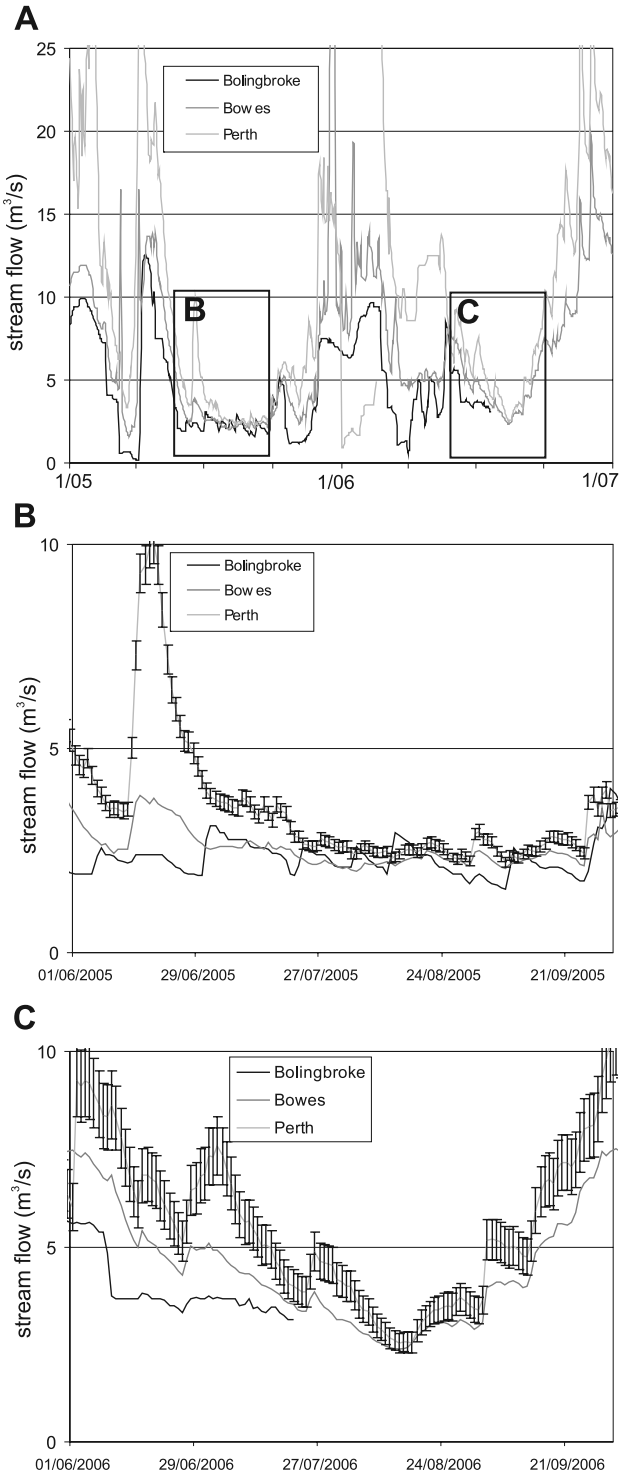

Figure 10. Tay River streamflow measured at three gauging stations by the Rideau Valley Conservation Authority. Error bars on Perth station is the measurement uncertainty (±5%), which indicate that, during low-flow conditions, river streamflow does not increase downstream.

4.4. Streamflow Measurements

[31] Streamflow from the three different stations along the Tay River (Figure 3) are compared to determine if the Tay River is gaining during low-flow conditions in 2005 and 2006 (Figure 10). The furthest downstream station at the

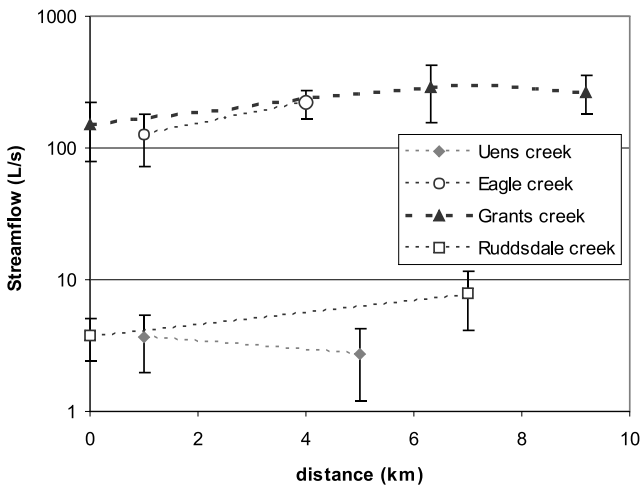

Figure 11. Low-flow streamflow measurement of minor creeks in the Tay River watershed in August 2007. The streamflow rate of Cameron Creek was too low to measure.

town of Perth is plotted with ±5% measurement uncertainty as error bars. This shows that during low-flow conditions there is no measurable increase in streamflow downstream in the Tay River (Figure 10), even though minor tributary streams also contribute to the river streamflow (Figure 3).

[32] The total random and systematic uncertainty in the streamflow measurement of the four minor creeks was 26–48% due primarily to the low-flow velocities which cause significant uncertainties in the rotations/minute of the flowmeter. The downstream measurements are within the error uncertainty of the upstream measurement for all four streams (Figure 11). Groundwater discharge conditions are difficult to detect due to the large uncertainties in streamflow measurements.

5. Discussion

5.1. Estimating Groundwater Discharge at the Watershed Scale

[33] A maximal groundwater discharge estimate for the Tay River watershed during low-flow conditions is calculated using a variety of methods to quantify discharge patterns and rates. Different methods were applied to the different types of surface water bodies and multiple methods were used for each type of water body to corroborate results from other methods. The total groundwater discharge at the watershed scale during low-flow conditions is estimated by summing the approximate discharge from each component:

$$Q_{watershed} = Q_{lakes} + Q_{river} + Q_{creeks} \quad (6)$$

where Q is groundwater discharge (m^3/day) and $Q_{watershed}$, Q_{lakes}, Q_{river} and Q_{creeks} are the groundwater discharge to the total watershed, to the lakes and wetlands, to the Tay River, and to the creeks, respectively. Groundwater discharge via evapotranspiration is not quantified during this study. The groundwater discharge patterns and rates for each type of surface water body (lakes and wetlands, the Tay River and the creeks) are discussed in order to sum the total $Q_{watershed}$.

[34] Multiple geochemical indicators (Figures 4 and 5) suggest that groundwater discharge to lakes and wetlands is systematically limited with the exception of the headwaters of Cameron Creek. Without a dense network of flowmeters [*Lee*, 1977; *Taniguchi et al.*, 2002] directly measuring distributed groundwater discharge in the lakes and wetlands is impossible. Qualitative analysis of the stable isotopes alone cannot differentiate between the relative influence of evaporation and groundwater discharge in lakes and wetlands (Figure 4a). However, stable isotopes in conjunction with chloride concentrations reveal the patterns of groundwater discharge and the relative influence of evaporation and groundwater discharge but are not used to quantify actual fluxes. The chemistry of most lakes and creeks are not significantly changed by either evaporation or groundwater discharge (Figure 4c). Low radon activities in surface water bodies and low I_g/I_s ratios (Table 1) support the interpretation of limited groundwater discharge to most lakes and wetlands. Additionally, the temperature and specific conductance transect in Christie Lake did not identify any significant thermochemical anomalies (Figure 6).

[35] Results of the steady state advective model suggest that the groundwater discharge to the 21 sampled lakes and wetlands is ~102,000 m^3/d (Table 2). The sampled lakes and wetlands represent 76 % of the lakes and wetlands in the Tay River watershed by volume. For the remainder of the lakes and wetlands an approximate discharge rate is calculated assuming a ratio of groundwater discharge to volume ratio of 0.0001 (Figure 7). The remainder of the lakes and wetlands therefore likely contribute ~31,000 m^3/d for a watershed total Q_{lakes} of ~133,000 m^3/d. It should be emphasized that the steady state advective model is considered a screening level tool that provides estimates of groundwater and surface water inflows. In many cases these are maximal estimates because the radon activities were below detection. The assumptions discussed in section 3.2 are also important caveats.

[36] Daily flow measurements (Figure 10) and detailed transects of specific conductance, temperature and radon activities (Figure 6) all indicate that discharge to the Tay River in the 25 km reach examined in this study is not significant. Potential uncertainty in the flow measurements (±5%) indicate that Q_{river} is <0.13 m^3/s or <11,000 m^3/d.

[37] Discharge patterns to individual creeks evaluated using stable isotopes (Figure 4b) and radon activities (Table 1) are internally consistent. Cameron Creek has high radon activity and specific conductance as well as stable isotopic values suggesting it is primarily groundwater discharge. Radon activities and specific conductance in Ruddsdale Creek increase downstream concurrent with an isotopic shift indicating groundwater discharge. Uens Creek has measurable radon activities and an isotopic shift indicating groundwater discharge. Eagle Creek, Lineament Creek and Grants Creek are all below detection for radon activities, consistent with stable isotopic results. Temperature and specific conductance transects of Cameron Creek and Lineament Creek support these interpretations (Figure 9). Unfortunately, the significant uncertainty in streamflow measurements of the creeks (Figure 11) limits the usefulness of this data for quantifying groundwater discharge rates in the creeks. Instead, streamflow measurements can be used as the maximum potential groundwater discharge rate for the creeks in which multiple geochemical indicators reveal groundwater discharge (Cameron Creek, <0.002 m^3/s streamflow; Uens

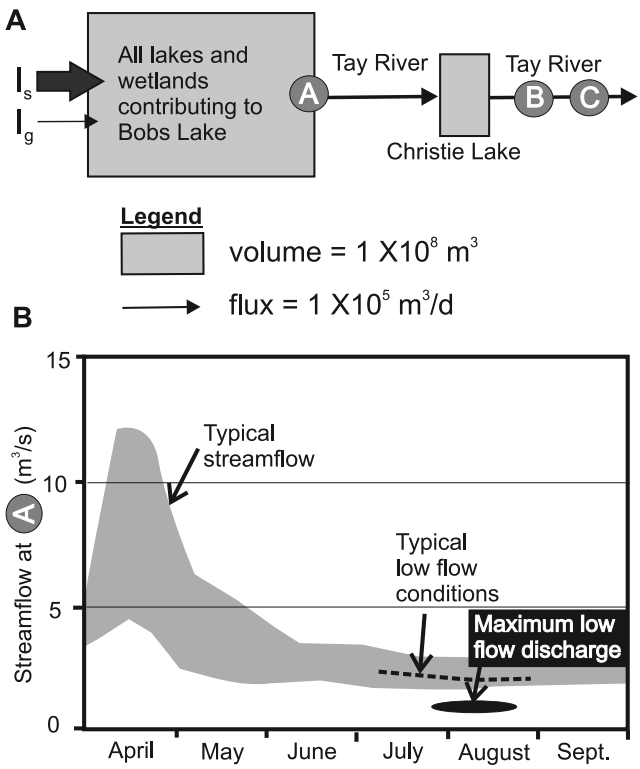

Figure 12. (a) Comparison of the streamflow of the Tay River to the contribution of surface water (I_s) and groundwater (I_g) to lakes and wetlands that are the source of the Tay River. The thickness of the line is scaled to the flux and the size of the surface water body is scaled to the volume (Table 1). The contribution of groundwater to Christie Lake and the Tay River is not shown because it is insignificant relative to the depicted fluxes. (b) Typical streamflow of the Tay River, including low-flow conditions, measured at Bolingbroke Dam (compiled from 5 years of data from the Rideau Valley Conservation Authority). The maximum low-flow discharge for the contributing area of Bob's Lake is compiled from Table 1. Applying base flow recession techniques to the Tay River during low-flow conditions would grossly overestimate groundwater discharge.

Creek, 0.003 m^3/s; and Ruddsdale Creek, 0.008 m^3/s). Other creeks not examined in this study have insignificant streamflow compared to Q_{lakes} or Q_{river}. Therefore the total Q_{creeks} is <0.013 m^3/s or <1,100 m^3/d.

[38] The overall relationship of discharge rates to the different types of water bodies is therefore $Q_{lakes} > Q_{river} > Q_{creeks}$. Discharge to lakes and wetlands, which is distributed over a very large surface area, is therefore the most important for constraining $Q_{watershed}$. Individual estimates that are summed in the steady state advective model are maximal values since many lakes were below detection limits for radon. But higher values are possible if lower groundwater radon activities are considered (Table 2 and Figure 8). The maximum groundwater discharge for the watershed ($Q_{watershed}$) is less than ~144,100 m^3/d, assuming the median groundwater radon activity for the steady state advective model. The impact of this discharge rate is discussed in section 5.3. The low-flow discharge rates and patterns in the Tay River watershed could be influenced by the higher surface water levels due to regulation structures such as the Bolingbroke dam. High water levels could lead to lower hydraulic gradients in the groundwater which would lower discharge. However, since the water table is generally near the surface throughout the year, discharge is more likely controlled by bedrock permeability than hydraulic gradients.

5.2. Comparison to Other Groundwater Discharge Rates

[39] Groundwater discharge rates in the Tay River watershed can be compared to other hydrologic settings by calculating an areal discharge flux (cm/d). The average areal discharge flux (total I_g divided by total lake and wetland area) for the 21 lakes and wetlands examined using the steady state advective model is 0.15 cm/d. The maximum areal discharge flux measured is 0.61 cm/day in the Cameron Creek headwater. In other hydrologic settings, average areal discharge fluxes to lakes and wetlands have been estimated using radon to be 0.30–0.74 cm/d [*Corbett et al.*, 1997], 0.36 cm/d [*Kluge et al.*, 2007] and 0.22–0.39 cm/d [*Cook et al.*, 2008]. In the topographically subdued continental shelf of Louisiana, *McCoy et al.* [2007] recently documented low rates of submarine groundwater water discharge (0.01–0.14 cm/d) which were corroborated with a regional groundwater model [*Thompson et al.*, 2007]. Therefore the maximum areal discharge flux observed in the Tay River watershed is consistent with estimates from other lakes and wetlands. The average areal discharge flux to the lakes and wetlands of the Tay River watershed is ~2 times lower than estimates from other lakes and wetlands but are consistent with estimates of submarine groundwater discharge in topographically subdued areas.

[40] Low-flow groundwater discharge rates from the Tay River watershed can be compared to other watersheds with similar geology and climate by normalizing discharge rate to precipitation rate (unitless) as part of a water budget, although water budgets usually contain significant uncertainties [*Winter*, 1981]. This approach is only reasonable for humid areas where the monthly precipitation rate is relatively consistent. For the Tay River watershed the discharge/precipitation ratio (total I_g divided by product of the watershed area and precipitation rate) is 4%. Mirror Lake, New Hampshire is a small, well-characterized watershed also underlain by fractured crystalline rock and variable soil thickness. *Rosenberry and Winter* [1993] estimated a 4% discharge/precipitation ratio for bedrock discharge in the Mirror Lake water budget. WE-38 is a small watershed in Pennsylvania is also underlain by fractured crystalline rock and variable soil thickness. Low-flow rates in WE-38 are 34 L/s [*Gburek and Folmar*, 1999] which equates to a discharge/precipitation ratio of 14%. In contrast, groundwater discharge rates are often estimated to be 15–50% of precipitation rates in porous media watersheds [*Arnold and Allen*, 1996; *Corbett et al.*, 1997]. Therefore groundwater discharge from fractured bedrock normalized to precipitation rate may be relatively low compared to porous media watersheds. For some fractured rock watersheds such as the Tay River and Mirror Lake, the rate of bedrock groundwater discharge is a relatively insignificant

part of the water budget compared to the residual of the water budget [*Rosenberry and Winter*, 1993].

5.3. Groundwater Discharge in Fractured Bedrock Watersheds

[41] The small areal discharge flux, low discharge/precipitation ratio and low I_g/I_s ratio (Figure 7 and Table 2) all suggest that the Tay River watershed is a surface water dominated system. Yet during low-flow conditions the groundwater discharge rate may be 20–40% of the Tay River streamflow suggesting that groundwater discharge may be volumetrically supporting streamflow low-flow conditions. Figure 12 illustrates how surface water influx (I_s) dominates over groundwater discharge (I_g) but that groundwater discharge can be a volumetrically appreciable component of streamflow. This apparent contradiction is due to the importance of surface water storage in the watershed. The Tay River and the flux of groundwater (I_g) to the lakes and wetlands that contribute to Bob's Lake in Figure 12 can be viewed as two minor fluxes compared to the reservoir volume or the substantial surface water influx (I_s). The importance of storage to watershed dynamics is underscored by the fact that at low streamflow the Tay River would take 2000–5000 days to drain the volume of the lakes and wetlands, depending on the low-flow rate.

[42] The low rates of groundwater discharge may impact our understanding of fractured bedrock watershed processes. The low groundwater discharge rates suggest that the groundwater and surface water system may be largely decoupled in this watershed compared to watersheds underlain by porous media. The low discharge rate is consistent with the low rate of groundwater recharge to the fractured bedrock aquifer [*Milloy*, 2007]. The low rates of groundwater discharge are consistent with previous studies of small watersheds in the Canadian Shield that indicate that groundwater discharge is limited where soil is minimal [*Buttle et al.*, 2004; *Steedman et al.*, 2004].

5.4. Groundwater Discharge Methods in a Large Watershed

[43] Groundwater discharge rates or base flow are often used as a proxy for groundwater recharge [*Rorabaugh*, 1964; *Rutledge and Daniel*, 1994; *Mau and Winter*, 1997; *Risser et al.*, 2005, 2009] but in regulated or lake-dominated watersheds the assumption that low-flow streamflow equals recharge can be problematic. Therefore we developed a mixture of novel and standard field methods and calculations to determine discharge patterns and rates in a large watershed independent of base flow recession. The methodology is transferable to any large watershed study even though this study focuses on a large, regulated watershed underlain by fractured bedrock. Here we make recommendations that might streamline the design of future research projects. One caveat is that the methods used in this study were implemented to constrain groundwater discharge but other study areas may have surface water bodies with sections that are gaining from groundwater discharge while other sections are losing. For example, comparing different streamflow measurements along a reach integrates both discharge and recharge fluxes. In these settings with complex groundwater-surface water interactions, each method must be implemented carefully.

[44] Temperature and specific conductance transects can be a useful and affordable tool to use, especially during reconnaissance, to identify significant groundwater discharge points (i.e., Cameron Creek). Similarly, synoptic sampling for stable isotopes and bulk chemistry can identify overall pattern in groundwater discharge versus evaporation or actual groundwater discharge in creeks sampled along their reach (Figure 4). Radon activities alone and concurrent with chloride measurement were used in the novel steady state advective model that was essential to quantifying groundwater and surface water inflow rates. The accuracy of this method is limited by the dependence on well-mixed surface water bodies and representative groundwater radon activities. These limitations should be considered when planning future applications of the steady state advective model. Manual flow measurements of creeks with a low velocity can be misleading due to the large error (Figure 11). Installing a permanent stream gauge could reduce these uncertainties.

6. Conclusions and Implications

[45] In this study we evaluate the pattern and rate of groundwater discharge in a regulated watershed using methods that are independent of base flow recession. Natural conservative (δ^2H, δ^{18}O, Cl, and specific conductance), radioactive (^{222}Rn), and thermal tracers are integrated with flow measurements to delimit the discharge locations and quantify the discharge fluxes to lakes, wetlands, creeks and the Tay River. The results improve our understanding of the rate, localization and conceptualization of discharge in a large, fractured rock watershed:

[46] 1. The groundwater discharge rate to the Tay River watershed is low. Surface water inflow to lakes and wetlands is up to an order of magnitude larger than groundwater discharge. Groundwater discharge to the Tay River is not geochemically, thermally or hydraulically detectable. A few creeks in the watershed have a groundwater component but the streamflow of these creeks is a minor fraction (<0.1%) the overall watershed budget. The low permeability of the bedrock aquifer likely limits the rate of groundwater discharge.

[47] 2. Groundwater discharge is not localized around lineaments or high-density zones of exposed brittle fractures. Instead, groundwater discharge seems to be distributed throughout the watershed except in the case of Cameron Creek which is a zone of localized groundwater discharge that was not predicted a priori from lineament or fracture mapping. Therefore groundwater discharge in the Tay River watershed is best conceptualized as a distributed, minimal flux. Groundwater discharge not being localized at lineament is consistent with a recent reinterpretation of lineaments as watershed-scale hydraulic barriers [*Gleeson and Novakowski*, 2009].

[48] 3. Distributed discharge is difficult to measure with physical methods, therefore geochemical methods which can integrate larger areas are more effective. Multiple complimentary methods are essential, especially in watersheds that are hydraulically complex (i.e., multiple surface water body types). A suite of methods are useful for corroborating results and because a single methods does not work for all types of water bodies.

[49] 4. This study focuses on a large watershed underlain by fractured bedrock although the methodology developed is transferable to any large watershed. This suite of methods can constrain groundwater discharge rates in regulated or unregulated watersheds which is increasingly important since dams control many of the medium to large rivers in the world [*Nilsson et al.*, 2005]. The developed steady state advective model provides important constraints on groundwater discharge and surface water inflows to the lakes and wetlands. The field data are relatively easy to acquire making it a useful screening-level tool.

[50] The low groundwater discharge rates have significant implications for the ecology, sustainability and management of large, crystalline watersheds which are common in North America, northern Europe and tropical shield regions in South America and Africa. Low flows are integral to sustaining cold-temperature fish species and other aquatic ecology [*Hayashi and Rosenberry*, 2002; *Sophocleous*, 2002]. Low-flows also have important socioeconomic impacts such as water supply, recreation and reservoir operation [*Burn et al.*, 2008]. Prediction of low-flow conditions in ungauged basins remains a challenge and an important management concern [*Burn et al.*, 2008; *Spence et al.*, 2008]. Most attempts to predict low-flow conditions in ungauged basins focus on hydrologic, geomorphic, physiographic and geological comparisons of basins. Reexamining basins using the suite of isotopic and geochemical methods described in this paper may enable better prediction of low flows in ungauged basins [*Soulsby and Tetzlaff*, 2008].

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