GROUNDWATER INUNDATION OF COASTAL ONSITE WASTEWATER TREATMENT SYSTEMS: INVESTIGATING PRESENT AND FUTURE IMPACTS TO COASTAL WATERS

by

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ABSTRACT

Coastal groundwater is a critical freshwater resource that supports human life, while coastal surface water supports both aquatic life and coastal industries (e.g., tourism, aquaculture). The quality of these critical groundwater and surface water resources is threatened by anthropogenic perturbations such as increased groundwater extraction and pollution, as well as climate change forcing such as sea-level rise. Onsite wastewater treatment systems (OWTS), which are common for wastewater disposal, can become inundated by rising groundwater tables as a result of rising sea levels, resulting in decreased performance of the OWTS. Contaminants that are not attenuated before reaching the elevated groundwater table can migrate through the coastal aquifer and be delivered to coastal waters via submarine groundwater discharge. In this thesis, we use field techniques (piezometers, seepage meters, radon analysis) to characterize the groundwater flow and submarine groundwater discharge at a popular public beach in Nova Scotia, Canada that is lined with cottages using OWTS. Fieldwork is combined with water quality sampling of coastal surface water and submarine groundwater discharge, with a focus on comparing the effectiveness of a novel viral tracer of human fecal contamination (crAssphage) with classic bacterial indicators. The effects of climate change (changing recharge and sea-level rise) on groundwater table elevations and the saltwater-freshwater interface across the same study site are assessed using a coupled groundwater flow and solute (salt) transport model, SEAWAT.

Increased use of OWTS during the summer cottage season at the study site coincided with widespread detections of crAssphage in submarine groundwater discharge (4/4 samples) and coastal surface waters (3/8 samples). Conversely, classical fecal pollution indicators based on bacterial targets were sparsely detected in the samples in the coastal environment (2/12 *E. coli* samples, 0/12 HF183 samples), likely due to greater attenuation of bacterial contaminants within the subsurface environment. Results from this first application of crAssphage in coastal groundwater contribute to a growing body of research reporting the application of this emerging tracer in various environments impacted by sewage pollution sources.

Results from the SEAWAT modelling indicate that as many as 9% of OWTS in this small but densely populated coastal watershed are either inundated or completely flooded by groundwater. This number could grow to 27% of OWTS for the climate change scenario with the highest recharge and sea-level rise. The location of the modeled saltwater-freshwater interface was also tracked in the model to investigate the potential salinization of groundwater resources used for drinking water supply. The modeled interface moved landward by ≤20 m and proved to be less of a concern than OWTS inundation, except for shoreline dwellings. This research contributes to an increasing number of groundwater modelling studies focusing on the impacts of sea-level rise on coastal subsurface infrastructure and provides important insight for rural coastal communities reliant on OWTS and fresh groundwater for drinking water.

The results of this thesis are used to locate at-risk OWTS across the province, and areas of high OWTS use, high potential sea-level rise, and low elevation are highlighted. These communities in particular should consider the implications of climate change for OWTS vulnerability, but also consider employing novel tracers such as crAssphage to provide early detection of low levels of surface water contamination from OWTS.

LIST OF ABBREVIATIONS USED

CFU	Colony Forming Unit
E. coli	*Escherichia coli*
ECCC	Environment and Climate Change Canada
h	Hydraulic head
h_f	Height of groundwater table above mean sea level
K	Hydraulic conductivity
k	Intrinsic permeability
K_g	Kilograms
L	Length
mm	Millimeters
m	Meters
M	Mass
mg	Milligrams
MSL	Mean Sea Level
N	Nitrogen
OWTS	Onsite Wastewater Treatment Systems
P	Phosphorus
p	Pressure
q	Darcy flux
PCR	Polymerase Chain Reaction

rRNA	Ribosomal Ribonucleic Acid
s	Seconds
SGD	Submarine Groundwater Discharge
SLR	Sea-Level Rise
STA	Soil Treatment Area
T	Time
μ	Dynamic viscosity
ρ	Density
z	Depth of the saltwater-freshwater interface below mean sea level

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I want to thank my parents for their unconditional love, support, and encouragement in whatever it is I want to do with my life. The latitude and safety net you have given me to experiment with my jobs, passions, and interests has made my life full. A special thanks to my brother for inspiring me to be the smartest version of myself. You're probably still right about most things, but I could certainly win an argument about variable density groundwater flow.

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CHAPTER 1 – INTRODUCTION

1.1 – Coastal Hydrogeology

1.1.1 – Critical but Vulnerable Coastal Ecosystems

Coastal groundwater and surface water systems are environmentally critical and economically significant as they support global human populations, industry, transportation, and trade (Moore et al., 2010; Michael et al., 2017; Luijendijk et al., 2020; Babu et al., 2021). Five times the number of people per area live within 100 km of the coast compared to inland, with an estimated 1.4 billion people living in the 'low elevation coastal zone' by the year 2100 (Neumann et al., 2015). Tourism is amongst the world's largest industries, with coastal destinations being the largest segment within the tourism industry (Philips & Jones, 2021). Aquaculture, another significant industry reliant on coastlines, is worth nearly $200 billion annually and is growing (Global Industry Analytics, 2022). Maintaining coastal water quality is integral to allow these populations and industries to continue to thrive.

Coastal communities often rely on groundwater from coastal aquifers for domestic and industrial purposes, making coastal groundwater a critical resource (Polemio & Walreavens, 2019). The migration of human populations towards the coast coupled with overuse of water resources has resulted in a "coastal groundwater squeeze", as potable water resources are threatened from above and below (Michael et al., 2017). Groundwater pumping, contamination from the land surface (e.g., fertilizer from farms, wastewater inputs to groundwater), saltwater intrusion, and land subsidence from urban expansion are just a few of the threats to coastal groundwater (Polemio & Walreavens, 2019). There are also feedbacks between coastal groundwater quantity and quality issues as groundwater extraction may pose a greater saltwater intrusion threat to coastal communities globally than sea-level rise (SLR) (Ferguson & Gleeson, 2012).

Enhanced understanding and monitoring of coastal groundwater dynamics and ocean-aquifer interactions is critical to sustaining coastal communities long term (Panthi et al., 2022). Interactions between the ocean and coastal aquifers are dynamic and complex, making climate change impacts difficult to predict. This confluence of forcings and feedbacks can be thought of as an aquifer-ocean-climate nexus, as changes in any one individual component impact the other two. For example, as discussed in detail in this thesis, climate change can result in changes to groundwater recharge and sea level, which in turn can influence the delicate balance between fresh and saline groundwater along the coast (Werner & Simmons, 2009; Oppenhimer et al., 2019; Jiao & Post, 2019, pp. 283 - 293). Coastal groundwater flow, groundwater–surface water interactions, and contaminant inputs and transport dynamics, are all important components when evaluating coastal groundwater systems and their sensitivity to future climate conditions.

1.1.2 – Thesis Overview

This thesis examines coastal ecosystem vulnerability with a focus on direct anthropogenic and climatic forcings on coastal groundwater and surface water systems. Coastal pollution from onsite wastewater treatment systems (OWTS) is explored, along with the implications of OWTS performance as a function of groundwater inundation, and subsequent impacts to coastal water quality. The impacts of climate change on coastal hydrogeology are examined through numerical modelling, and related back to OWTS performance, focusing specifically on the impacts of SLR on groundwater table elevations and OWTS inundation. Detailed objectives are stated in section 1.4 of this chapter. Field investigations, water quality sampling and analysis, hydraulic monitoring, data analysis, and numerical modelling methods are employed to gather data and formulate results and interpretations.

Chapter 1 is divided into four sections, three of which review critical background literature and related theory, and one of which outlines the thesis objectives. Chapters 2 and 3 are separate but related investigations, each formatted as individual journal papers. Chapter 2 highlights coastal contamination from OWTS, employing groundwater and surface water quality sampling to investigate the presence of established and novel indicators of human fecal pollution. Chapter 3 is a groundwater modelling study focusing on the impacts of SLR on groundwater table rise and OWTS inundation and makes recommendations for future OWTS installation standards. Chapter 4 concludes by linking these two papers and presenting a risk map of the province-wide coastal OWTS susceptibility to SLR.

1.1.3 – The Saltwater–Freshwater Interface and Variable-Density Flow Dynamics

Density differences between saltwater from the ocean and freshwater from the terrestrial subsurface result in the intrusion of saline groundwater beneath fresh groundwater along the coast in the form of a ‘wedge’ of saltwater dipping beneath freshwater (Fig. 1). The transition zone between the two domains is known as the saltwater–freshwater interface (Werner et al., 2013; Jiao & Post, 2019). As a result, fresh coastal groundwater effectively 'floats' on salty groundwater, and responds to perturbations from the ocean and the terrestrial land surface (Heiss & Michael, 2014). The characteristics and behaviors of the saltwater-freshwater interface are dependent on a variety of complex variables (e.g., beach slope, aquifer properties, wave action, tides), making field studies challenging and modelling studies, which assume idealized environments, common (Werner et al., 2013).

Estimating groundwater flow in this environment requires the consideration of density differences between saltwater and freshwater. Classic formulations of groundwater flow such as Darcy's law (eq. 1) must be augmented to account for the density differences between seawater and freshwater (eq. 2):

$$q = -K\ (dh\ /dl) \qquad (1)$$

$$q = -\kappa/u\ (\nabla p - \rho g) \qquad (2)$$

where q is the Darcy velocity [L/T], K is the hydraulic conductivity [L/T], dh/dl is the change in hydraulic head over a unit length, κ is the intrinsic permeability [L^2], u is the dynamic viscosity of the fluid [$M/(L \times T)$], ρ is the fluid density [M/L^3], p is pressure [$M/L/T^2$] and is g gravitational acceleration [L/T^2]. In eq. (2), both the fluid density and the viscosity are influenced by the groundwater salinity. The application of such equations to solve groundwater flow problems is complex because the mass conservation for equations representing groundwater flow (Darcy's law) and solute transport (Fick's law with advection included) are written as interdependent partial differential equations (Jiao & Post, 2019. pp. 35 – 36). The solute concentration influences the flow dynamics, and the Darcy flux influences the advective solute transport.

Determining the location of the saltwater–freshwater interface is often a critical first step in coastal groundwater management (Werner et al., 2013). For a homogeneous, unconfined coastal aquifer at steady-state and hydrostatic conditions (Fig. 1), the depth of the saltwater-freshwater interface beneath mean sea level (MSL), z [L], can be approximated as 35 to 40 times the elevation difference between the water table and the ocean, as stated by the Ghyben – Herzberg principle (Ghyben & Drabbe, 1889; Herzberg, 1901). This is due to density differences between saltwater and freshwater:

$$z = \frac{\rho_f}{\rho_s - \rho_f} h_f \tag{3}$$

where ρ_f and ρ_s are densities [M/L^3] for freshwater and saltwater respectively, and h_f is the height [L] of the freshwater (e.g., water table) above the MSL datum (Figure 1). This relationship can be simplified by substituting approximate densities for ρ_f and ρ_s as 1000 kg/m^3 and 1025 kg/m^3 respectively.

$$z = -40h_f \tag{4}$$

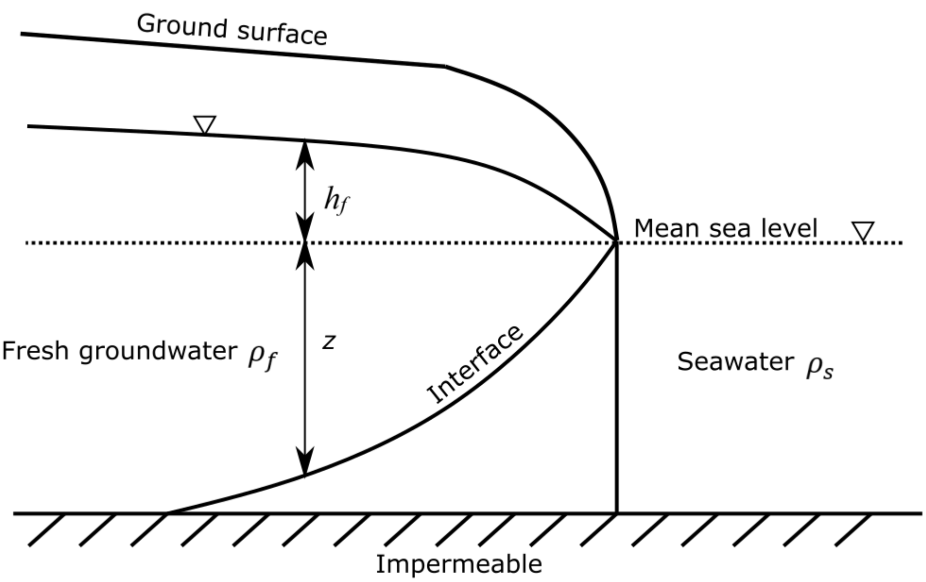

Figure 1 A not-to-scale representation of the Ghyben-Herzberg principle used to estimate the depth to the saltwater-freshwater interface (z) based on the fresh groundwater table elevation above mean sea level (h_f) (adapted from Jiao & Post 2019).

This formula is extended by Hubbert (1940) (Eq. 5); however, it is less practical for field use as the pressure heads of both fresh and saltwater are required:

$$z = \left(\frac{\rho_s}{\rho_f - \rho_s}\right) * h_s - \left(\frac{\rho_f}{\rho_f - \rho_s}\right) * h_f \tag{5}$$

where h_s and h_f are saltwater and freshwater head, respectively. Extended derivations of Hubbert's formula for a variety of field scenarios are outlined by Jiao & Post (2019, pp. 54 – 56), but become increasingly complex with various coastal aquifer conditions.

The saltwater-freshwater interface is not truly sharp as presented in Figure 1; rather there is a finite transition zone between saline and fresh groundwater (Abarca & Clement, 2009; Werner et al., 2013). The measured width of this transition zone, often termed a 'mixing zone', differs between field and laboratory settings. While idealized models with estimated dispersion, diffusion, and density coefficients often approximate narrow mixing zones on the order of a few meters or less (Goswami & Clement, 2007; Abarca & Clement, 2009), mixing zones in the field can range by orders of magnitude from meters to kilometers (Price et al., 2003; Paster et al., 2006). This misalignment is due in large part to geologic heterogeneities and marine forcing operating at a range of temporal scales (e.g., tides, sea-level rise). Variations in grain size and the effect of stratification combine to widen the dispersion zone to various degrees, depending on size and scale (Dagan & Zeitoun, 1998).

1.1.4 – Submarine Groundwater Discharge

1.1.4.1 – Mechanisms for Subterranean Groundwater Discharge

Understanding the processes driving the movement and exchange of fresh and saline groundwater in a coastal setting is critical for managing groundwater resources, especially those threatened by seawater intrusion and climate change (Werner et al., 2013). Submarine groundwater discharge (SGD) is defined as the total combined (fresh and circulated seawater) flow of groundwater from the seabed to the coastal ocean (Burnett et al., 2003b) (Fig. 2). Moore (2010) compiled over 100 studies identifying SGD as a primary source of nutrients and bacteria to coastal regions. Santos et al. (2012) reviewed the drivers of SGD and identified 12 different forces, with terrestrial hydraulic gradients being the only one resulting in the net flow of water towards the ocean. However, the remaining 11, including seasonal aquifer changes on land, tidal pumping, and wave setup, all represent a source of new or recycled nutrients to seawater. Therefore, not all SGD is freshwater derived from onshore recharge. Seawater that has infiltrated the sediment and mixed with groundwater via waves, tides, and storms can discharge back to the ocean as circulated SGD (Burnett et al., 2003b). Density differences between freshwater and seawater play an important role in driving circulation near the saltwater–freshwater interface (Cooper, 1959), which also contributes to SGD (Smith, 2004). Recharge, tides, waves, and evaporation, have all been identified as further drivers influencing water movement and SGD along the saltwater–freshwater interface (Robinson et al., 2018).

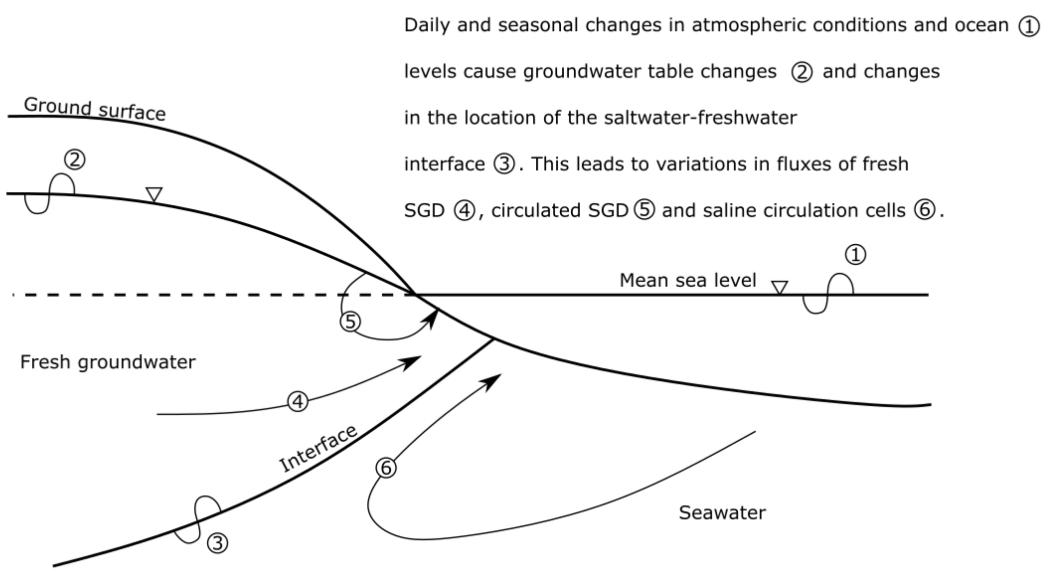

Figure 2 Coastal groundwater forcings driving various forms of SGD (adapted from Taniguchi et al., 2019).

In the simplest case, SGD involves groundwater flowing towards the sea in accordance with the gradient created from higher hydraulic head in the onshore groundwater compared to that of the ocean (Moore et al., 2010). Topography driven SGD is greatest near the shoreline where water depths are shallower (Taniguchi et al., 2002), primarily within the first tens of meters of the high tide waterline (Sawyer et al., 2016). The flux of near-shore SGD is also a function of geologic properties, and groundwater recharge along a coastline (Taniguchi et al., 2002).

SGD rates are temporally variable, in part due to dynamic sea levels (Fig. 2). Seepage meter investigations and synthesis (e.g. Taniguchi et al., 2019; Lee et al.,1977) show SGD flux variations in accordance with the induced semi-diurnal tidal gradients (high SGD flux at low tide, lower at high tide). Predictably, semi-monthly tidal variations from the spring and neap tides also cause variations in SGD flux, and annual variations such as decreased water table elevations during the dry season can influence SGD dynamics (Kim and Hwang, 2002; Michael et al., 2005). Episodic rainfall events also impact SGD rates, with rainfall increasing SGD, as well as influencing the position of the saltwater-freshwater interface (Yu et al., 2017).

1.1.4.2 – Contaminant Transport via Submarine Groundwater Discharge

Contaminant transport is a critical consideration in the assessment of SGD flux and origin (Sawyer et al., 2016). For example, nitrogen, phosphorus, bacteria, heavy metals, and viruses can all originate from near-shore OWTS and migrate to coastal waters via SGD (LaPointe et al., 1990; Slomp & Van Cappelan, 2004; de Sieyes et al., 2011; Yau et al., 2014). This can lead to negative outcomes for coastal waters including eutrophication via enhanced nutrient inputs (de Sieyes et al., 2011), and impacts to human health from SGD-derived fecal microorganisms in beach waters (Yau et al., 2014). The degree to which these contaminants reach coastal waters from their point of origin is dependent on their flux into the subsurface and downgradient contaminant attenuation (sorption, straining) and degradation (elimination, die-off) processes (Stevik et al., 1999; Stevik et al., 2004). Contaminant transport can also be impacted by porous media properties, including media temperature, organic matter content, and moisture content (Stevik et al., 2004).

1.1.4.3 – Measurement of Submarine Groundwater Discharge

As SGD can be a diffuse flux, potentially occurring across vast extents of coastline, measurement at both local and global scales is difficult (Burnett et al., 2006). Methods for measuring SGD involve direct measurement of SGD to the coast using seepage meters, as well as indirect measurement using radioisotopes (Lee et al., 1977; Kwon et al., 2014) or temperature (Tamborski et al., 2015; Bejannin et al., 2017; KarisAllen et al., 2022) as tracers. Seepage meters are inexpensive and simple to use, providing a direct measurement of SGD at a local scale (Lee, 1977). However, they are limited by design, as they only capture direct SGD at a single point, and SGD can exhibit pronounced spatial variability (Taniguchi et al., 2002). Methods of detecting radon and radium *in-situ* have led to the development of using radioisotopes as a groundwater tracer to estimate SGD at the shoreline scale (Moore, 2007; Burnett et al., 2003c). Although more complex and requiring more interpretation than seepage meters, these radioisotope approaches provide a relatively reliable estimate of SGD, and are now commonly employed (Kwon et al., 2014).

1.1.5 – Impact of Sea-Level Rise on Groundwater Table Elevation

The impacts of sea-level rise (SLR) on coastal groundwater systems are dependent on aquifer confinement and hydraulic properties, as well as groundwater recharge. Hence SLR impacts are highly dependent on the hydrogeologic conditions (Jiao & Post, 2019, pp. 283 – 293). The lateral movement of the mean tide water mark due to SLR is dependent on shoreline slope, with shallow sloping beaches being inundated much further inland compared to steep cliffs for a given SLR (Jiao & Post, 2019, pp. 290 – 292). Figure 3 illustrates the impacts of SLR on a coastal unconfined aquifer. As sea levels rise, the saltwater-freshwater interface moves landward and upward, and the groundwater table is accordingly forced upwards. This groundwater table movement saturates previously unsaturated space in the subsurface. This phenomenon has been confirmed by examining the impact of tides and longer-term processes on groundwater table elevations (Ataie-Ashtiani et al., 2013). For example, a 50-year study of well water levels in Cape Cod Massachusetts by McCobb and Weiskel (2003) found that a 2.1 mm/yr rise in well water levels largely aligned with local SLR rates of 2.5 mm/yr over the same 50-year period.

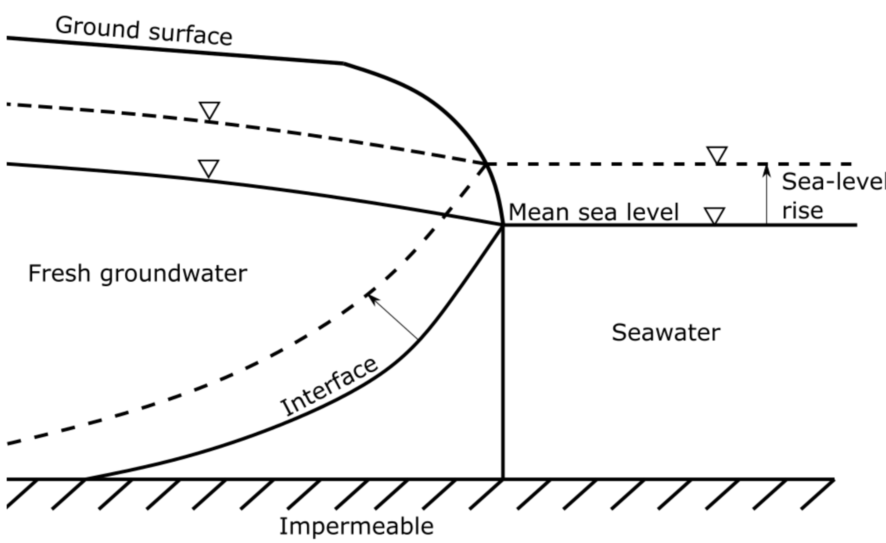

Figure 3 The impact of sea-level rise on groundwater table elevation in an unconfined aquifer and the movement of the saltwater-freshwater interface (adapted from Jiao & Post 2019).

Elevated groundwater tables have resulted in documented impacts on surface and subsurface infrastructure for coastal communities (Habel et al., 2017; Cox et al., 2019). Rotzoll & Fletcher

(2013) found that 1 m of SLR along an urbanized coastal area in Honolulu Hawaii, caused flooding across 10% of the region, more than double the projection for marine inundation alone. Direct seawater surface flooding is often secondary to the groundwater response to SLR (Befus et al., 2020). The projected migration of hundreds of millions of people towards the coast over the next 40 years (Neumann et al., 2015), will be compounded with groundwater inundation caused by SLR, affecting water resources, infrastructure, and ecosystems (Befus et al., 2020).

1.1.6 – Seawater Intrusion Driven by Sea-Level Rise

Seawater intrusion refers to the landward, subsurface movement of seawater, which is often caused by anthropogenic forcing (e.g., pumping) or climate driven changes in sea-level (Werner et al., 2013). As sea levels rise, the inland movement of the saltwater-freshwater interface must occur to account for the increase in saltwater volume and pressure (Fig. 2). Landward movement of the saltwater-freshwater interface can also occur due to decreases in recharge leading to a lowered freshwater pressure and volume (Werner & Simmons, 2009).

Approximations of the movement of the saltwater-freshwater interface in response to SLR can be developed using either numerical or analytical approaches (Werner et al., 2013). Analytical solutions provide first-order approximations as they neglect transient effects as well as the transition zone between saltwater and freshwater, often overpredicting the landward shift of the toe of the saltwater-freshwater wedge (Werner et al., 2010). However, timescales for SLR are known to be gradual, and interfacial mixing at the saltwater–freshwater interface changes these approximations (Bear, 1985). Using a numerical modeling approach, Werner et al. (2010) found that the time to reach a steady-state saltwater-freshwater interface following 1 m of SLR ranged from decades to centuries, depending on which quantitative indicator was used to assess 'steady state'. This highlights the difficulty in making these predictions, and the transient but inertial nature of these dynamic environments.

Determining whether a coastal groundwater system is flux controlled or head controlled when predicting the movement of the saltwater-freshwater interface with SLR is a critical consideration (Werner & Simmons, 2009). A flux-controlled system is one in which groundwater discharge from the coastal aquifer to the ocean is persistent despite changes in sea level (Werner & Simmons, 2009). For this situation, the rise in the groundwater table elevation is equal to that of the rise in sea-level, and hydraulic gradients across the system are maintained. In a head-controlled system, surface hydraulic controls such as lakes, rivers, and surface flooding (i.e., water table at the land surface) maintain the inland aquifer head even as SLR persists. For example on study found in a flux-controlled system, with a given SLR up to 1.5 m, the calculated position of the salt wedge toe moved landward linearly on the scale of 10's of meters depending on environmental characteristics such as recharge and hydraulic conductivity (Werner & Simmons, 2009). However, in the same study, the head-controlled system experienced exponential landward movement of the interface due to rising sea levels, with movement potentially on the order of 100s of meters (Werner & Simmons, 2009).

Groundwater withdrawal from coastal aquifers can also draw the saltwater-freshwater interface landward, in some cases resulting in salinization of groundwater wells due to upconing of saltwater beneath the well (Hussain et al., 2019), especially when combined with SLR (Abd-Elhamid & Javadi, 2011). Predicting when and where this will occur is an important problem for coastal groundwater managers. There are many factors that influence the degree to which saltwater intrusion into groundwater wells occurs including aquifer properties, groundwater pumping rates and duration, local hydrology such as recharge rates and regional flow, marine forcing, and contaminant transport characteristics such as dispersive forces and sorption characteristics (Werner et al., 2013). These plethora of factors and drivers influencing saltwater intrusion makes numerical modeling a critical tool in groundwater resource management.

1.2 – Onsite Wastewater Treatment Systems and Coastal Groundwater Interactions

1.2.1 – Form and Function of Onsite Wastewater Treatment Systems

Access to safe drinking water and proper wastewater sanitation, have been demonstrated to be closely correlated with the economic growth and development of countries worldwide (Shafik, 1994). Onsite wastewater treatment systems, or more commonly septic systems, are one of the primary mechanisms for wastewater treatment in the developed world, with an estimated 23% of the United States reliant on some form of an OWTS to treat their wastewater (USEPA, 2002). Locally, 45% of Nova Scotia residents rely on OWTS (Nova Scotia Environment, 2011). Typical OWTS consist of a septic tank and disposal field. The septic tank allows for the settling of solids from household drains. What remains is septic tank effluent, which is transported to a distribution field of a coarse-grained porous media such as gravel, allowing for the even distribution of septic tank effluent across a soil treatment area (STA). This allows for the mechanical attenuation and degradation of pollutants (Siegrist et al., 2000; Province of Nova Scotia, 2013).

In Nova Scotia, a minimum of 1 m vertical distance of unsaturated STA is required between the bottom of the distribution trench and either the water table or the top of an impermeable layer such as bedrock (Fig. 3) (Province of Nova Scotia., 2013). This separation distance is intended to promote a zone of unsaturated flow, which in turn promotes the filtration, sorption, and transformation of wastewater contaminants such as phosphorus, nitrogen, and fecal microorganisms (Van Cuyk et al., 2001).

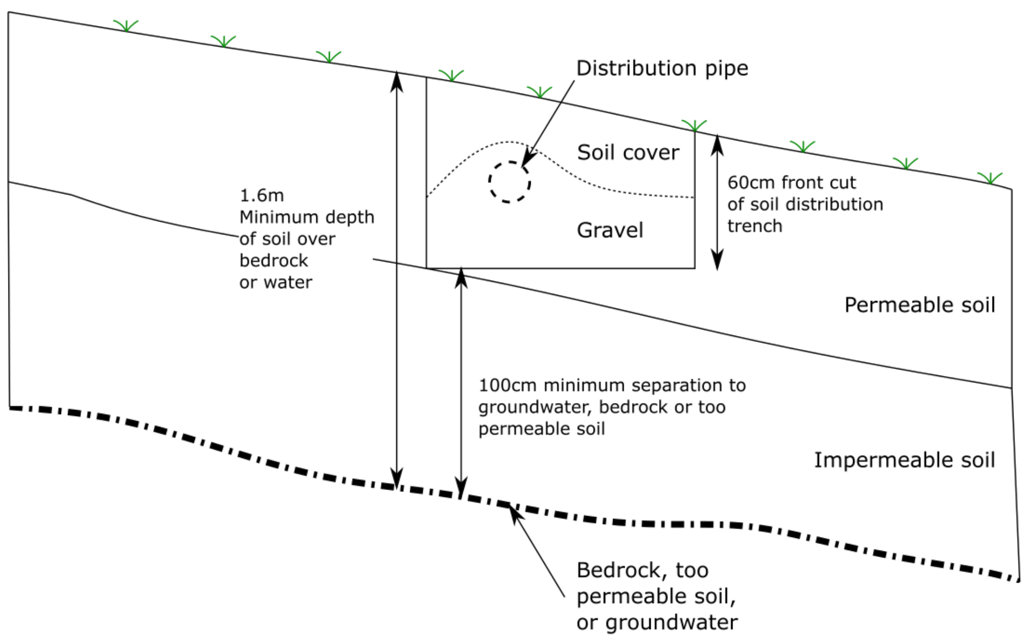

Figure 4 Typical design components and general depth requirements for OWTS in Nova Scotia (adapted from Nova Scotia Onsite Wastewater Treatment Technical Guidelines 2013).

When performing properly, OWTS are effective at bacteria removal and environmental protection. Stevik et al. (2004) identified the main processes leading to the retention and degradation of bacteria in porous media. Retention involves straining, which is the physical blocking of bacteria in pore spaces and is dependent on factors such as grain size and bacteria size, as well as adsorption, which is a function of pH, temperature, and physical characteristics of the porous media itself. Bacterial degradation (or elimination) occurs due to abiotic factors such as moisture content, pH, and temperature, as well as biotic factors including the presence of predatory microorganisms (Jamieson et al., 2002). These mechanisms combine to effectively treat septic tank effluent so it can be safely discharged into groundwater. Stevik et al. (1999) demonstrated the effective removal of *E. coli* (a common bacterial indicator of water quality) in vertical sand filters 80 cm in height. The majority of the *E. coli* was removed (via attenuation or degradation) in the upper 12 cm of the column, although results varied with hydraulic dosing rate and filter material.

1.2.2 – Pollutants of Concern from OWTS

OWTS have a demonstrated impact on coastal water quality (De Sieyes et al., 2008; Schneeberger et al., 2015), in some cases impacting public health (Yau et al., 2014). Pathogens such as viruses and bacteria released from OWTS can degrade water quality along the coast (Schneeberger et al., 2015). Nitrogen and phosphorus from OWTS can cause eutrophication of coastlines and harmful algal blooms (De Sieyes et al., 2008). These contaminants have unique transport characteristics as discussed below.

The primary controlling factors of pathogen transport from OWTS to groundwater are attenuation (straining, sorption) and survival (die-off) (Stevik et al., 2004). In general, bacteria are readily attenuated in the STA of OWTS due to their larger size and lower survivability outside the body (Gerba et al., 1975). While large bacteria are readily strained in an OWTS STA, viruses are generally not attenuated in this manner owing to their order of magnitude smaller size (Gerba, 1975). Conversely, the net negative charge on viruses, as well as bacteria, typically enhances sorption of these pathogens to soil particles in the STA (Tefenkji, 2007).

Unlike many of the key wastewater solutes, viruses and bacteria can be harmful to human health at very low concentrations, approaching 20 or less individual organisms (Hunt & Johnson, 2017). Further, pathogens have the benefit of replication, enabling their concentrations to potentially increase downstream of their source under proper conditions (Hunt & Johnson, 2017). Unsaturated zones, such as the OWTS STA, limit pathogen transport, but can act as a 'holding reservoir' where pathogens could remain and survive for extended periods of time (Tafuri &

Selvakumar, 2002). The re-saturation of these 'holding zones', via large precipitation events, or groundwater table rise, results in the re-mobilization and rapid discharge of this buildup of harmful pathogenic material (McCarthy and McKay, 2004; Bradford et al., 2003). Although pathogen diffusion is very low compared to the diffusion of solutes in the soil matrix, pathogen transport in macropore flow spaces has been found to be much more rapid (McKay et al., 1993; McKay, 2011). This low diffusion also limits a pathogen's ability to re-enter the matrix pore space from a fracture (McKay et al. 1993; McKay 2011).

Straining is the most important attenuation mechanism for bacteria in OWTS. The process of straining is physical and occurs when the colloid is larger in diameter than the pore space it is attempting to pass through (McDowell-Boyer et al., 1986). Pore openings can constrict with continued colloid attachment, resulting in a compounding effect where previously large pore spaces become small enough to filter smaller colloids. The process has a significant impact on flow as well. Continuous straining leads to decreased groundwater flow, barring any increase in applied pressure (Bradford et al., 2003). Straining during regional groundwater flow results in smaller diameter viruses occurring more frequently in groundwater wells compared to larger diameter bacteria (Wu et al., 2011).

Nitrogen and phosphorus treatment in OWTS STA is complex, as transformations of both elements in different forms are dependent on temperature, moisture content, pH, and other environmental factors (Lusk et al., 2017). Ammonification, volatilization, immobilization, nitrification, and denitrification can all occur in the STA at different times, impacting the volume and form of nitrogen entering the groundwater (Lusk et al., 2017). Phosphorus typically enters the STA as orthophosphate and organic phosphorus, but can be reversibly precipitated in many forms, sorbed to soil grains, or mineralized. In both cases, N and P can be transported to coastal surface water via SGD and cause eutrophication (De Sieyes et al., 2008).

1.2.3 – Impact of SLR on OWTS Performance

As noted in section 1.1.4, SLR can cause a subsequent rise in groundwater table elevations, leading to the inundation of subsurface infrastructure such as OWTS, specifically the STA. As noted in section 1.2.2, moisture content is an important variable impacting the ability of the STA to attenuate pathogens and nutrients. Evaluating the impact of SLR on groundwater table elevation is thus a growing research topic worldwide.

Cox et al. (2019) determined sea-level rise and increased precipitation to be the two primary factors leading to an increase in groundwater table elevation in communities in southern Rhode Island, which are largely dependent on OWTS for wastewater treatment. Results suggested locations that have currently functioning OWTS may, in the future, experience groundwater inundation of the STA, and compromised OWTS treatment. Cooper et al. (2016) examined the performance of soil mesocosms, replicating the STA of an OWTS, at removing contaminants from OWTS effluent under both present climate and future climate change (increasing temperature and groundwater table elevation) scenarios. They found a decrease in the removal of both phosphorus and fecal coliforms under the 'climate change' scenario, brought on by the increasing temperatures and water table elevations. Wetter soils were postulated to increase survival rates of fecal coliforms in the soil mesocosms and limit their attachment to soil surfaces. Although higher temperatures have been shown in the past to decrease the survival rate of bacteria such as *E. coli* (Gerba et al., 1975; Morales et al., 2015), moisture conditions were found to have a dominant impact on survival rate. Finally, modelling of groundwater response to SLR has shown groundwater inundation of OWTS can occur at the same rate as marine inundation in certain hydrogeological settings (Manda et al., 2015), highlighting the importance of both factors when evaluating risk.

Due to the inherent difficulties associated with long-term field studies of OWTS, modelling is an important tool for predicting and improving filter performance. Software programs like Hydrus are used to model bacteria transport in variably saturated porous media, and modelled OWTS performance is well documented across various studies (Pang & Šimůnek, 2006; Šimůnek et al., 2008; Morales et al., 2015). Studies have also focused on the inundation of the OWTS STA itself. Hydrus is commonly used to model the variable saturation and contaminant transport dynamics within the OWTS STA (Sinclair et al., 2014; Morales et al., 2015; Hayward et al., 2019). Column experiments are also useful in predicting how environmental changes in the STA (e.g., moisture content, temperature), will impact pathogen transport (Bradford et al., 2006; Gargiulo et al., 2008; Cooper et al., 2019). Generally, these experiments indicate that increasing moisture contents leads to more mobile contaminants within the soil column.

In addition to enhanced pathogen transport in saturated, or re-saturated environments, pathogen transport can be increased in lower temperature soils. Moynihan et al. (2015) found cool, moist environments to be ideal for pathogen survival, and *E. coli* populations have been found to be highest following rainfall events (Hagedorn et al., 1978). Despite survival rates being highest in saturated soils, *E. coli* can persist in soils with as little as 0.08g H_2O per gram of dry soil (Fremaux et al., 2008), making this a particularly problematic pathogen when considering buildup and saturation in an OWTS. It should be noted that other studies have found the opposite effects of moisture on *E. coli* survival (van Elsas et al., 2011a; Williams et al., 2015), and further research is warranted. The literature widely supports pathogen survival increasing at lower temperatures (Alegbeleye et al., 2020), a fact that may assist in the removal of pathogens alongside a warming climate. However, the predicted temperature increases over the next 100 years will not likely have enough of an impact to offset the rapid increase in moisture conditions due to SLR. A more likely scenario is that inundation of filtration zones will result in conditions more suited for pathogen transport, causing the rapid contamination of groundwater sources near impacted OWTS systems (Fig. 5).

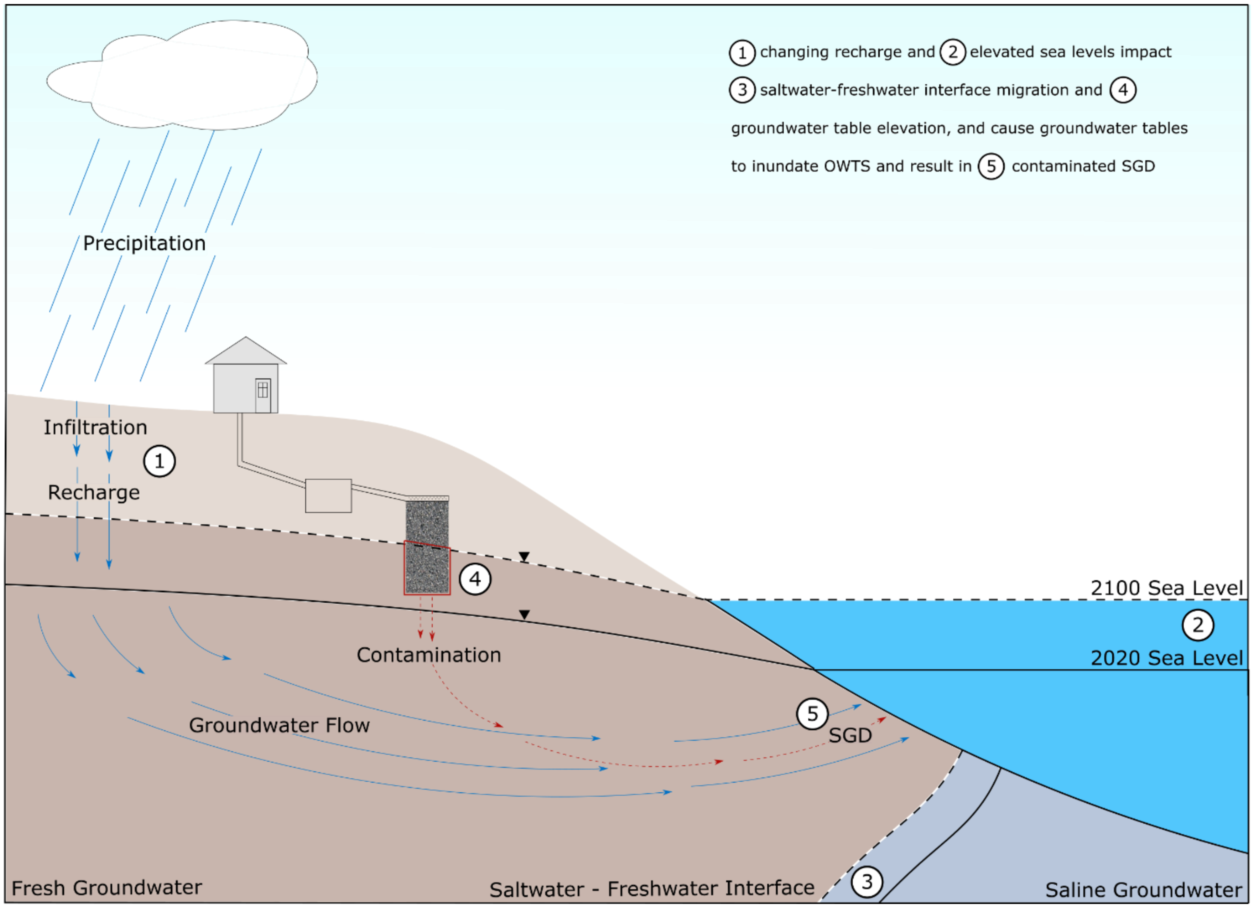

Figure 5 Conceptual diagram showing the impacts of SLR on groundwater table elevations, and implications for contaminant transport to coastal surface waters via SGD (figure from Threndyle et al., 2022).

1.2.4 – Coastal Groundwater Detection and Monitoring

Pathogenic *E. coli* is an important contaminant because, like many other pathogens, it can be harmful for human consumption, even at very low concentrations (Hunt & Johnson, 2017). Drinking water, when contaminated with pathogenic *E. coli* strains, causes illness and in some cases death, as demonstrated by the Walkerton Ontario crisis in 2000, where seven deaths and thousands of illnesses resulted from an *E. coli* (O157:H7) outbreak in the municipal drinking water (O'Conner, 2002). Due to its ability to persist across a large range of environments, its occurrence in almost all animal feces, and inexpensive sampling detection, *E. coli* has historically been considered the best biological indicator of drinking water quality when monitoring for public health protection (Edberg et al., 2000).

E. coli has some limitations as an indicator organism, as smaller contaminants, such as viruses, are more mobile and thus less likely to be attenuated in OWTS STA (Gerba, 1975). Viruses have been shown to impact coastal water quality, including human health and aquaculture (Walker & Mohan, 2009). Recent advances in detection have allowed for increased monitoring for viruses during water quality surveys (Dutilh et al., 2014); however, coastal assessments are still limited, and identifying a contaminant source can be difficult. Identifying contaminant fluxes and sources from rivers (point sources) discharging to the ocean is relatively simple compared to the distributed flux of contaminants from diffusive SGD (a non-point source). As a result, improved methods for detection of microbial contaminants, which aid in identifying both sources and fluxes, are important.

CrAssphage, an enteric virus that grows in the human gut and is shed with waste, has attracted recent attention as a fecal indictor virus that has high specificity to humans (Stachler et al., 2017). Wastewater treatment plant receiving water bodies, groundwater, and coastal surface water, have all been evaluated for the presence of crAssphage (Balleste et al., 2019; Kongprajug et al., 2019; Sala-Comorera et al., 2021). CrAssphage is a useful indicator as its specificity to humans can allow for the determination of source as well as the flux of contaminants. As viral pollution concerns increase, the ability to monitor and detect for viruses and low levels of pollution are critical, creating widespread interest in the application of crAssphage across a range of aquatic environments.

1.3 – SLR Impacts on OWTS and Other Subsurface Infrastructure

1.3.1 – OWTS Focused SLR Studies

Predicting the impact that SLR could have on OWTS inundation is an important step in protecting coastal surface and groundwater quality. Numerical modelling software to simulate coupled groundwater flow and solute transport permits the evaluation of impacts that predicted forcings (such as SLR or changing recharge) may have on groundwater (Langevin et al., 2004).

MODFLOW is used commonly to evaluate the impacts of SLR on coastal groundwater table elevations, including studies based in Hawaii, New Hampshire, and California in recent years (Habel et al., 2017; Knott et al., 2019; Befus et al., 2020). Like all hydrologic investigations, simplifying assumptions were built into these studies. Most notably, MODFLOW is a uniform density code, and these three prior studies (Habel et al., 2017; Knott et al., 2019; Befus et al., 2020) were run with steady-state conditions imposed. As described in section 1.1.3, groundwater flow dynamics in coastal aquifers can be strongly influenced by the density differences in fresh and saline groundwater, and thus variable-density groundwater models should ideally be employed when studying ocean-aquifer interactions. Also, transient dynamics may be critical to understand in the context of low-frequency perturbations such as SLR. All three studies predict widespread subsurface inundation dependent on SLR severity, which highlights the risks climate change poses to subsurface infrastructure.

1.3.2 – SLR Research in Nova Scotia

Nova Scotia has some of the highest rates of SLR globally (James et al., 2021), and nearly half of the province relies on OWTS to treat their household wastewater (Nova Scotia Environment, 2011). Because of this, low-lying coastal areas of the province (especially those not serviced by municipal wastewater treatment) are uniquely threatened by inundation of OWTS due to SLR. Provincial studies have assessed the impacts of SLR on coastal communities in specific locations and province wide, although no prior studies in Nova Scotia (or more generally in Canada) have considered the impacts of SLR on OWTS performance.

Analytical solutions and GIS-based vulnerability mapping have highlighted areas of the province that are susceptible to saltwater intrusion based on factors such as surface slope, groundwater usage, and population density (Beebe, 2011; Kennedy, 2012). A GIS-based approach to predicting SLR and storm surge impacts on infrastructure flooding in Yarmouth, N.S. was also completed, with potential impacts to many services including electricity and water treatment by the year 2025 (Muise et al., 2012). Similar studies have assessed the risk of surface flooding in Bridgewater, N.S. and Annapolis Royal, N.S. and revealed decreased return periods for significant storm events with rising mean sea levels, resulting in more intense and frequent surface flooding of coastal communities (Webster, 2010; Webster et al., 2014).

1.4 – Thesis Objectives

Nova Scotia's expansive coastline is home to both recreational (e.g., swimming, boating, surfing) and industrial (e.g., aquaculture, fishing) activities, both of which rely on favourable coastal water quality. This thesis is comprised of two distinct but connected papers on the subjects of climate change, OWTS vulnerability, and impacts on coastal water quality, with the general objective of assessing coastal OWTS vulnerability, and thus coastal water quality vulnerability, to climate change.

Chapter 2 is the first paper, ‘CrAssphage as an indicator of groundwater–borne pollution in coastal ecosystems’, which focuses on water quality impacts from OWTS using a novel tracer in a new environment to enhance our understanding of coastal pollution sources. The objectives of this research were to understand SGD fluxes and contaminant transport from OWTS at the study site with the aid of a novel wastewater tracer. Chapter 3 is the second paper, ‘Future inundation of coastal on-site wastewater treatment systems in a region with pronounced sea-level rise’, which focuses on groundwater table inundation of OWTS under various climate change (SLR, changing recharge) scenarios. The specific objectives of this research were to first create a 3D groundwater flow and solute transport model to simulate the impacts of SLR and changing recharge on groundwater table elevations in the watershed encompassing the study site. GIS software was then used to interpret the modeling results and identify OWTS systems inundated by rising groundwater tables. Additionally, migration of the saltwater-freshwater interface was evaluated under these same climate change scenarios. Together, these papers make up the primary body of work for my thesis. Brief introduction (chapter 1) and conclusion (chapter 4) sections tie these concepts together and make general recommendations for the province.

CHAPTER 2 – CRASSPHAGE AS AN INDICATOR OF GROUNDWATER BORNE POLLUTION IN COASTAL ECOSYSTEMS

*As published in *Environmental Research Communications*:

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2.1 – Abstract

Novel approaches for monitoring coastal water quality changes and identifying associated contaminant source(s) are of growing importance as climate change and population redistribution to coastal zones continue to impact coastal systems. CrAssphage, a virus found in the human gut and shed with fecal matter, is currently gaining popularity as an indicator of human fecal contamination in surface water and groundwater. Here we demonstrate that DNA assays targeting crAssphage genetic fragments can be used to detect pollution from nearshore onsite wastewater treatment systems discharging to the ocean via submarine groundwater discharge. We integrated this novel viral monitoring tool into a field study that characterized the physical hydrogeology (hydraulic gradients, hydraulic conductivity, and seepage fluxes) and surface water and groundwater quality at a study site on the north shore of Nova Scotia, Canada. Increased use of onsite wastewater treatment systems during the summer cottage season coincided with widespread detections of crAssphage in submarine groundwater discharge (4/4 samples) and coastal surface waters (3/8 samples). Conversely, classical fecal pollution indicators based on bacterial targets (*Escherichia coli* and human-specific *Bacteroidales* genetic marker (HF183)) were sparsely detected in the samples in the coastal environment (2/12 *E. coli* samples, 0/12 HF183 samples), likely due to greater attenuation of bacterial contaminants within the subsurface environments. Results from this first application of crAssphage in coastal groundwater contribute to a growing body of research reporting the application of this emerging tracer in various environments impacted by sewage pollution sources.

Keywords: CrAssphage, Onsite wastewater treatment, Submarine groundwater discharge, Coastal hydrogeology, Groundwater-surface water interactions, Coastal pollution

2.2 – Introduction

Coastal groundwater and surface water quality is of growing importance given recent and projected increases in coastal population density (Mallin et al., 2000; Neumann et al., 2015), with concomitant industrial activity and contaminant loading (e.g., Howarth, 2008; Michael et al., 2017). Coastal waters provide important societal and economic services for local populations and tourists alike (Volker & Kistemann, 2011; DeFlorio-Barker et al., 2018); however, these services are threatened by coastal pollution. For example, recreational activities in contaminated waters can lead to disease outbreak and in severe cases, death (Graciaa et al., 2018). Similarly, fisheries and aquaculture operations can be impacted by the presence of pathogenic microorganisms, resulting in a loss of economic activity in the case of harvesting restrictions, or a range of negative health outcomes if products are consumed (Howarth, 2008; Malham et al., 2014; Vikas & Dwarakish, 2015).

Coastal groundwater systems are also changing due to seawater intrusion induced by rising seas and increased coastal flooding (Werner et al., 2013; Ketabchi et al., 2016). In unconfined coastal aquifers, fresh groundwater 'floats' above the denser saline groundwater located in a salt wedge (Figure 6). These freshwater zones in coastal aquifers often serve as important water supplies but are threatened by rising sea levels (Sawyer et al., 2016; Michael et al., 2017; McKenzie et al., 2021). Changing coastal groundwater dynamics will also impact contaminant transport processes and negatively impact potable coastal groundwater supplies and the marine ecosystems receiving groundwater discharge (LeMonte et al., 2017; Guo et al., 2020). Submarine groundwater discharge (SGD) is defined as the flow of fresh and circulated saline groundwater (e.g., LeRoux et al., 2021) from the seabed to the coastal ocean, regardless of fluid composition or driving force (Burnett et al., 2003). Coastal aquifers and associated SGD provide a conduit for chemical and biological contaminants to enter coastal surface waters (Boehm et al., 2004; Moore, 2010; Sawyer et al., 2016; Robinson et al., 2018; Ruiz-Gonzalez et al., 2021). One potential source of contaminants in coastal settings is Onsite Wastewater Treatment Systems (OWTS), a decentralized form of domestic wastewater management that involves the discharge of partially treated wastewater effluent into the subsurface environment. This effluent subsequently enters groundwater systems, and, in the case of coastal aquifers, can be transported to the ocean via SGD (de Sieyes et al., 2008; Izbicki et al., 2012). Coastal OWTS are particularly vulnerable to climate change and coastal storms, as elevated groundwater tables resulting from sea-level rise (SLR), high spring tides, or storm surges will impact OWTS performance (Cooper et al., 2016; Cox et al., 2019). In general, improper OWTS design can result in the introduction of untreated or partially treated wastewater effluent to groundwater systems and receiving surface water bodies (Figure 6).

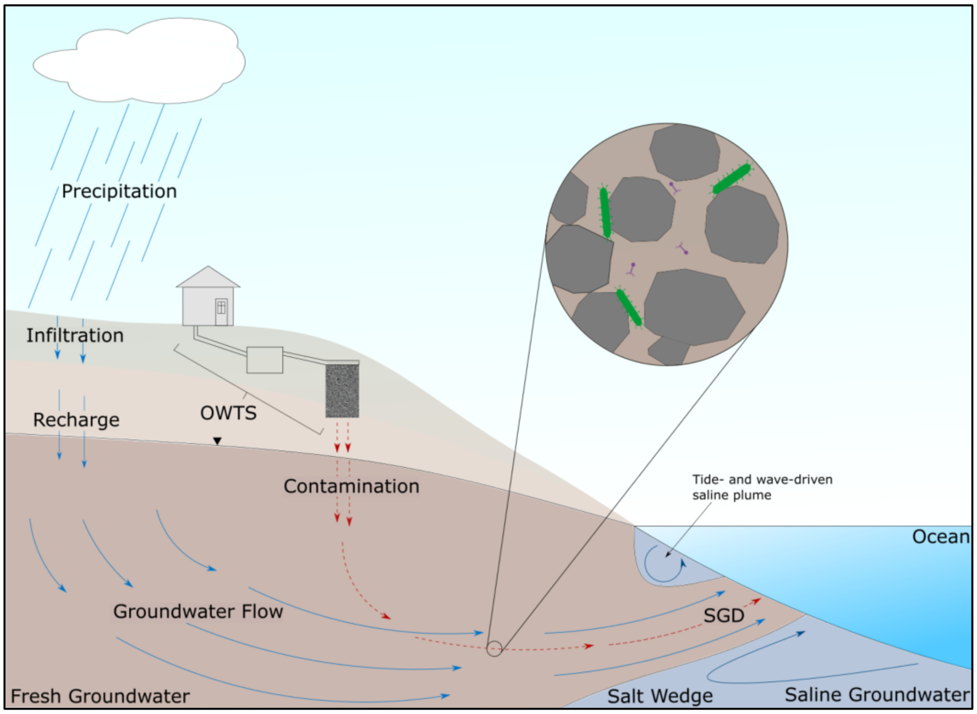

Figure 6 Conceptual diagram showing an OWTS discharging effluent wastewater into groundwater, which is subsequently discharged into the ocean via SGD. Image inset shows green bacteria being strained and attenuated, while smaller and more mobile purple viruses pass through the pore space.

OWTS are a potential source of pathogenic microorganisms (e.g., bacteria, viruses, protozoa) that could be transmitted to people using coastal waters. Historically, bacteria such as *Escherichia coli* (*E. coli*) have been used as indicators of fecal pollution in both groundwater and surface water due to, among other factors, their relatively inexpensive detection methods and general occurrence across human and animal feces (Edberg et al., 2000; Health Canada, 2012). Recent advances in molecular technologies, such as quantitative polymerase chain reaction (qPCR), have also led to the development of new tools for assessing sources of fecal contamination. The HF183 qPCR assay, for example, targets a *Bacteroides* 16S ribosomal ribonucleic acid (rRNA) gene marker, that is specific to humans and is applied to measure human fecal pollution in the environment (Haugland et al., 2010). However, there are limitations with using bacteria-based monitoring targets for assessing pathogen transport from groundwater-derived fecal contamination sources. For example, the size and shape of enteric bacteria, such as *E. coli,* typically leads to greater attenuation in porous media due to physical straining (Figure 6, inset), resulting in effective removal of fecal bacteria within an OWTS (Stevik et al., 1999; O'Luanaigh et al., 2012). Viruses, in particular, have different transport and attenuation characteristics than bacteria, due to their small size and greater persistence in groundwater environments (Morrison et al., 2020, Pang et al., 2021). Several studies have documented the presence of enteric viruses in groundwater systems in the absence of conventional enteric bacteria indicators (Fout et al., 2017; Morrison et al., 2020; Pang et al., 2021).

The recent discovery of a bacteriophage, crAssphage, that is associated with bacteria inhabiting the human intestinal tract, has provided a practical virus-based fecal indicator (Balleste et al., 2019; Kongprajug et al., 2019; Sala-Comorera et al., 2021). First discovered in 2013, crAssphage is a novel viral marker that is host-specific and shed with human fecal material in high quantities (Dutilh et al., 2014). The specificity of the crAssphage used in this study (CPQ-056; Stachler et al., 2017) has been assessed in several geographies, with specificities ranging from 87 to 98% (Stachler et al., 2017; Ahmed et al., 2018; Gyawali et al., 2021). Cross-reactions with dog, cat, gull, and poultry fecal samples have been reported, but they are infrequent and concentrations of the genetic marker in these samples have been orders of magnitude lower than those observed in human fecal samples. CrAssphage assays have also been shown to be highly sensitive, typically detected in 100% of raw sewage samples, with concentrations ranging from 7 – 9 log copies/L (Ahmed et al., 2018; Gyawali et al., 2021). These studies have also demonstrated that concentrations are comparable to concentrations of widely used bacterial markers (e.g., HF183), and orders of magnitude higher than other enteric viruses, such as F-RNA phage and norovirus. Gyawali et al. (2021) also found that the presence of crAssphage in shellfish samples was highly predictive of norovirus contamination. Past crAssphage studies have primarily focused on pollution sources released directly to surface waters, and have largely demonstrated congruence with bacteria-based indicators (Balleste et al., 2019; Sala-Comorera et al., 2021). Morrison et al. (2020) were the first to assess crAssphage as a monitoring tool in subsurface environments and concluded that it is a promising tool; however, they did not directly compare crAssphage to conventional bacteria-based indicators.

The study of crAssphage is an emerging research topic with only very recent applications in groundwater and surface water. However, to our knowledge, no prior studies have investigated the efficacy of crAssphage to detect human fecal pollution in SGD, or directly compared crAssphage to bacteria-based indicators in settings where the fecal pollution is primarily derived from groundwater sources. Such conditions represent the environmental settings where this monitoring tool could potentially have the most impact and value. This represents a critical knowledge gap given the establishment of SGD as a major research theme in hydrology, hydrogeology, and oceanography (Burnett et al., 2003; 2006; Moore, 2010; Robinson et al., 2018; Taniguchi et al., 2019; Alorda-Kleinglass et al., 2021), with implications for coastal water quality and ecosystem health. The objectives of this study are to (1) evaluate the presence of human fecal contamination in a coastal system impacted by OWTS using crAssphage, HF183 and *E. coli* as fecal pollution markers and (2) compare the performance and relative sensitivities of these markers both before and during cottaging season (associated with elevated levels of fecal pollution from discharging OWTS).

2.3 – Methods

2.3.1 – Study Site

This study was conducted at a provincial park and public beach in northern Nova Scotia, Canada, that is surrounded by private cottages (Figure 7). The local climate is typical for the Canadian Maritimes with average summer (June – August) air temperatures of approximately 19 °C and mean annual precipitation of 969 mm (Government of Canada, 2021, ECCC Station 8205774). Use of the beach and the surrounding private dwellings is concentrated in the summer months. The provincial park and all dwellings in the immediate area use OWTS for domestic wastewater management. The underlying geology of the site consists primarily of interbedded sandstone and siltstone units overlain by glacial tills (Hennigar, 1972). Recharge to the underlying bedrock aquifer is estimated to be between 180 – 220 mm/year based on the baseflow from this watershed (Kennedy et al., 2010). Groundwater monitoring and sampling took place in the surficial aquifer, which is primarily sand along the beach and sandy clay in the upland beginning approximately 20 m beyond the high tide mark. The maximum tidal range recorded by our tidal logger (Figure 8) was approximately 2.50 m. The nearshore bathymetry is characterized by a shallow slope (e.g., < 5 m depth 1 km from shore).

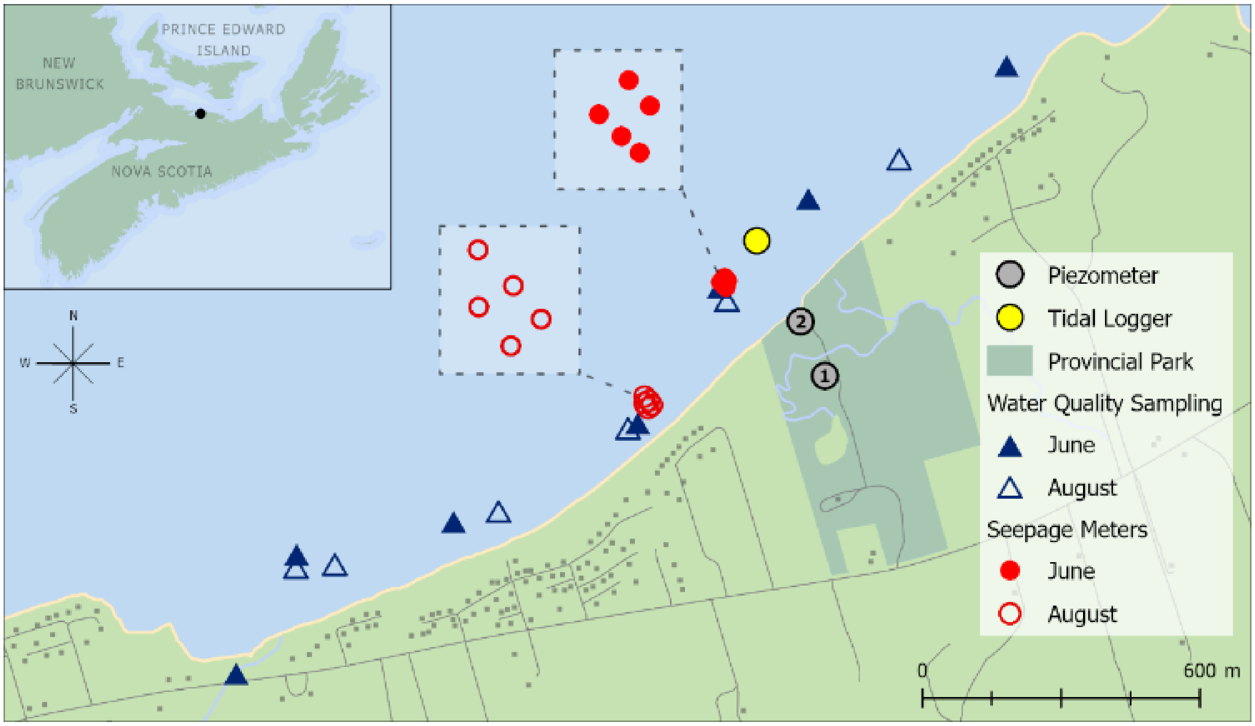

Figure 7 Map of the study site in Nova Scotia, Canada, with residential properties indicated by grey squares. East and West streams (blue lines) referenced in Table 1 can be seen in the bottom left (West) and middle (East, within the public park) of the map.

2.3.2 – Hydrogeology Field Work and Data Analysis

The site was instrumented in the fall of 2020 with a transect of piezometers and a tidal logger (Figure 7). Piezometers were installed in the surficial aquifer using a backpack drill to depths of 2-3 m. Pressure sensors (Solinst, Canada and Onset, U.S.A.) were deployed to record pressure every 30 minutes in each piezometer, as well as the tidal signal just offshore (Figures 7, 8). Ambient air pressure was recorded in a provincial monitoring well located approximately 10 km from the study site.

The hydraulic diffusivity of the unconfined sand aquifer along the beach was estimated using the tidal amplitude and the resultant damped groundwater tidal response in piezometer 2 approximately 160 m from the mean water edge (Eqs. 8 and 9 of Nielsen, 1990). Hydraulic conductivity was calculated from hydraulic diffusivity by estimating the beach sand aquifer depth to be ~7 meters based on the upper end of sand thickness recorded in Nova Scotia shoreline well logs and by assuming perfect drainage of the sediment, such that specific yield equaled sand porosity (0.25). Average linear groundwater velocity (v) and travel time (t) in the beach aquifer (which represents the groundwater flow path from the cottages located along the beach edge (Figure 7), was then estimated using the calculated hydraulic conductivity (*K*), the mean hydraulic gradient between the beach piezometer and the coast (*i*), the cross-shore beach length (L), and the sand porosity (ε):

$$v = \frac{-Ki}{\varepsilon} \text{ and } t = \frac{L}{v} \quad (6)$$

Seepage meters (Lee, 1997; Duque et al., 2020) were installed approximately 200 m from the high-tide waterline, such that they were just submerged during low tide (A1 in appendix). Volumetric measurements of water collected in the seepage meter bags over recorded time intervals yielded the SGD flux of water from the seabed to the ocean. Water quality samples were collected from the seepage bags approximately every 12 hours during the first field campaign in June of 2021, and approximately every 24 hours during the second field campaign in August 2021 (Peeler et al., 2006).

2.3.3 – Sampling and Water Quality Analysis

Water samples from the seepage meters, surface water at several locations along the coastline, and streams draining into the study area (Figure 7) were analyzed for chemical and microbiological parameters. Sampling was completed on single days during separate, weeklong field campaigns: the first on June 16, 2021, and the second on August 31, 2021, close to the end of the cottaging season. In both sampling campaigns, six open seawater samples were retrieved, as well as samples from the outlets of the nearby creeks (one in June and two in August). Additionally, four water quality samples were retrieved from the water collected in the seepage meter bags. During the August campaign only, two samples were collected from the water immediately surrounding the seepage meters, to compare SGD water quality with that of open seawater in the immediate area. As radon is enriched in groundwater compared to surface water (Hoehn and Gunten, 1989), radon analysis was completed on the sampled water using a RAD 7 (DURRIDGE Company Inc., MA, U.S.A.), and electrical conductivity readings of the collected water and surrounding seawater were recorded using a calibrated Conductivity Plus meter (Herron Instruments Inc., Dundas, ON).

Microbial parameters included *E. coli*, HF183, and crAssphage. Samples were collected in sterilized 1 L Nalgene collection bottles (Thermo Fisher Scientific, Waltham, MA, USA), fully submerged to 5cm depth, capped underwater, and kept on ice packs while being transported back to the laboratory at Dalhousie University in Halifax, Nova Scotia for immediate analysis. Seepage meter bags were emptied into similar 1L Nalgene collection bottles and transported in the same manner as surface water samples. For *E. coli* enumeration, 100 mL samples were analyzed using the membrane filtration method with mcoliblue-24 selective growth media (Hach, 1999). For HF183 and crAssphage markers, 500 mL sample volumes were filtered (0.45 µM pore size, 47 mm diameter, Millipore, Inc., Bedford, MA, USA), and DNA was extracted from the filters using a DNeasy PowerSoil Pro kit (Qiagen Inc., Toronto, Ontario, Canada). The concentration and purity of genomic DNA was first measured by ultraviolet absorbance spectrophotometry at 260/280 nm and 260/230 nm (Implen NanoPhotometer™, Implen, München, Germany). The qPCR assays for HF183 (Haugland et al., 2010) and crAssphage (Stachler et al., 2017) were conducted on a Bio-Rad CFX96 Touch™ Real-Time PCR detection system (Bio-Rad, Hercules, CA, USA). The limits of detection (LOD) of HF183 and crAssphage markers are 1.1 Log copies/100 mL and 2.83 Log copies/100 mL, respectively.

2.4 – Results and Discussion

2.4.1 – Groundwater Field Data and Analysis

The hydraulic heads obtained from the piezometers and tidal logger are presented in Figure 8. The ratio of the mean tidal range in the beach piezometer (0.50 m) to the mean tidal range in the strait (2.41 m) is 0.21, which yielded an aquifer diffusivity of 2400 m^2/hour and a corresponding K (based on storage and thickness values described earlier) of 87 m/hr. Based on equation (1), the tidally averaged hydraulic gradient (0.0018) between the beach piezometer and the tidal logger, and the mean cross-shore beach dimensions (200 m), an approximate travel time through the beach aquifer of 13 days was calculated. As many cottages line the beach (Figure 7), this represents the travel time from the nearest OWTS to the strait.

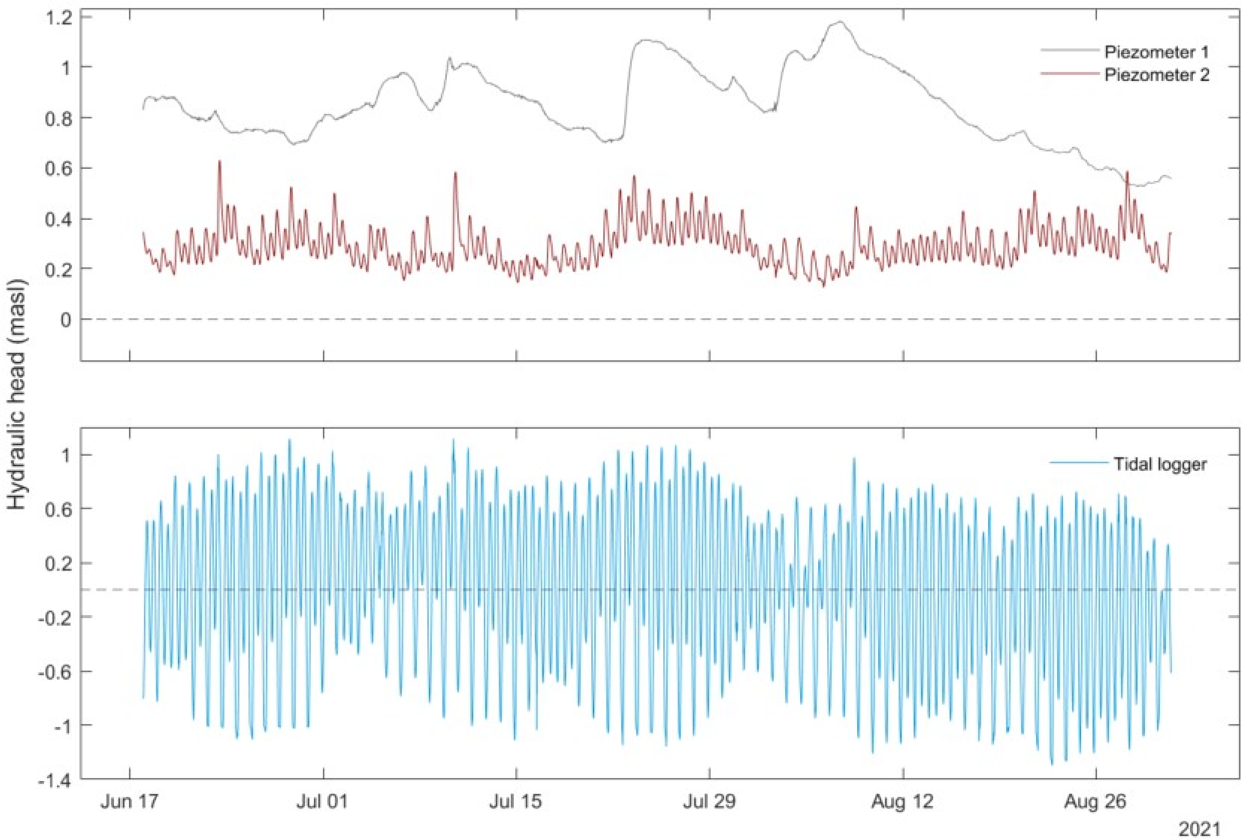

Figure 8 Groundwater (top, piezometer locations indicated in Figure 7) and surface water (bottom, strait) head data from the study site during the study period.

Our seepage meters indicated an upwelling SGD flux between 2.0×10^{-4} meters/day and 4.2×10^{-3} meters/day, which is lower than in many previous studies (e.g., Taniguchi et al., 2002). The electrical conductivity of the water collected in the seepage meter bags indicated lower salt contents (31,881 us/cm, on average) compared to the surrounding seawater (45,200 us/cm), which suggests that the SGD was made up of both fresh groundwater discharge and circulated seawater. Water samples for radon analysis were taken from a seepage meter bag, as well as the surrounding seawater to test for elevated radon concentrations, a common indicator of water with a groundwater origin (Burnett & Dulaiova, 2003). The seepage meter samples had a mean radon concentration of 77 Bq/m^3, while the seawater had a lower concentration of 24.6 Bq/m^3, which supports our assumption that the water collected in the seepage meters is in part derived from the terrestrial aquifer and thus could be impacted by nearby OWTS.

2.4.2 – Coastal Water Quality Analysis

Results of both the June and August sampling campaigns for the three fecal indicators (*E. coli*, HF183, and crAssphage) are presented in Table 1. *E. coli* was absent during the June sampling campaign, except for the West Stream and a single seepage meter in very low quantities (1 CFU/100ml), while HF183 and crAssphage were not detected. The elevated presence of *E. coli* in the creek could be the result of upstream agricultural practices, including observed livestock farming. The August sampling event revealed elevated *E. coli* levels in two of the four seepage meters, as well as presence in both the East and West stream, albeit in lower quantities compared to June. HF183 was absent in all sampling events, excluding the August sampling of the two streams

Table 1	Results from the water quality sampling during both the June and August sampling campaigns. Bolded values indicate microbial targets exceeding detection limits.

June 2021	*E. coli* (CFU/100 mL)	HF183 marker (Log copies/100 mL)	crAssphage marker (Log copies/100 mL)
Beach A1[1]	<1	<1.1	<2.83
Beach A2	**1**	<1.1	<2.83
Beach A3	<1	<1.1	<2.83
Beach A4	<1	<1.1	<2.83
Beach A5	<1	<1.1	<2.83
Beach A6	<1	<1.1	<2.83
West Stream	**240**	<1.1	<2.83
Seepage A1[2]	<1	<1.1	<2.83
Seepage A2	<1	<1.1	<2.83
Seepage A3	<1	<1.1	<2.83
Seepage A4	<1	<1.1	<2.83

August 2021	*E. coli* (CFU/100 mL)	HF183 marker (Log copies/100 mL)	crAssphage marker (Log copies/100 mL)
Beach B1	<1	<1.1	**2.85**
Beach B2	**3**	<1.1	**2.86**
Beach B3	<1	<1.1	<2.83
Beach B4	**6**	<1.1	**2.85**
Beach B5	<1	<1.1	<2.83
Beach B6	<1	<1.1	<2.83
Seepage B1	<1	<1.1	**2.85**
Seepage B2	<1	<1.1	**3.78**
Seepage B3	<1	<1.1	**3.75**
Seepage B4	<1	<1.1	**2.89**
Seawater 1[3]	<1	<1.1	<2.83
Seawater 2	<1	<1.1	<2.83
East Stream[4]	**8**	**1.28**	**3.05**
West Stream	**11**	**1.97**	**2.98**

[1] 'Beach' samples reference open seawater, [2] 'Seepage' samples were drawn directly from the water

The absence of both *E. coli* (2/11 detections in June, 4/14 detections in August) and HF183 (0/11 detections in June, 2/14 detections in August) markers in the majority of the samples in the coastal waters and streams (Table 1) is likely the result of the porous media successfully filtering these contaminants via straining and adsorption, as has been demonstrated for bacteria in many column experiments and field investigations (Bradford et al., 2006; Jiang et al., 2007). CrAssphage was undetected during the June sampling event compared to its detection in nine of fourteen samples during August sampling (Table 1). CrAssphage was present in three of six open seawater samples (maximum concentration of 2.86 log copies/100mL), and in all four seepage meter samples (maximum concentration of 3.78 log copies/100mL), suggesting that groundwater was a source of this enteric virus. The presence of crAssphage in August, particularly in the groundwater (seepage samples), compared to the complete absence in June (Table 1) suggests that increased OWTS usage during the summer contributes to wastewater loading to the shoreline via SGD pathways. Septic systems in this area consist of a septic tank that stores waste for 3-4 days, before discharge to a soil absorption field. Effluent discharge would be limited to the summer months when the septic system is being loaded. The ~13-day groundwater travel time (Section 3.1) is short enough to result in different coastal water quality conditions before and after cottage season based on SGD-borne contamination from OWTS. This indicates that contamination could occur seasonally, with a spike each year during cottage season when the OWTS see increased usage.

2.4.3 – CrAssphage as a Monitoring Tool

OWTS are used widely for domestic wastewater treatment in rural settings, many of which are also coastal. In the United States, more than one in five homes utilize OWTS for their wastewater treatment (EPA, 2013). In Nova Scotia, OWTS are used by nearly half of the province’s 1 million residents (Nova Scotia Environment, 2011). Many of these OWTS are coastal and potentially at risk of subsurface inundation due to sea-level rise (James et al., 2015). Globally, coastlines are widely used for recreation as well as aquaculture, both of which are sensitive to contaminant loading from SGD (Ghermandi & Nunes, 2013). In this study, we demonstrate the efficacy of crAssphage as a monitoring tool for fecal contamination derived from diffuse, groundwater-derived pollution sources. Early detection of human fecal contaminants in coastal waters enables preemptive action in identifying sources and initiating pollution mitigation programs.

2.5 – Summary and Conclusion

Our findings add to a growing body of literature highlighting crAssphage as a useful indicator of fecal pollution. Specifically, we demonstrate its efficacy for studying coastal pollution from SGD. We highlight the ability of crAssphage to detect fecal pollution from OWTS in a coastal setting, while demonstrating the impacts of seasonal contaminant loading, likely resulting from increased use of cottages reliant on OWTS for wastewater disposal. While fecal contamination at many coastal sites may be low today, enhanced rates of SLR will impact OWTS performance and lead to elevated contaminant levels in the future. Further studies could attempt to relate crAssphage concentrations to the presence and levels of other pathogenic viruses and related human health outcomes. We suggest that crAssphage could be an important tool for detecting viral contamination of coastal recreational waters impacted by OWTS.

2.6 – Acknowledgements

Funding for this project was provided through NSERC Discovery Grants to R. Jamieson and B. Kurylyk, and the Atlantic Ecosystems Initiative program of Environment and Climate Change Canada. Support was also provided through the Canada Research Chairs Program (Jamieson and Kurylyk) and a Geological Society of America Graduate Student Research Grant and Killam and Nova Scotia Graduate Scholarships to R. Threndyle. G. Kennedy (Nova Scotia Geological Survey) is thanked for providing assistance with geological interpretation as well as barometric pressure data.

CHAPTER 3 - FUTURE INUNDATION OF COASTAL ON-SITE WASTEWATER TREATMENT SYSTEMS IN A REGION WITH PRONOUNCED SEA-LEVEL RISE

*In review with *Journal of Hydrology*:

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3.1 – Abstract

An often-overlooked impact of rising sea levels is the concurrent rise in coastal groundwater tables, which can result in subsurface inundation of below-ground infrastructure. This is an important consideration for rural coastal communities, many of which rely on on-site wastewater treatment systems (OWTS). Subsurface inundation and flooding of OWTS leads to treatment performance issues, potentially resulting in enhanced contaminant transport to coastal groundwater resources and the ocean via submarine groundwater discharge. We use SEAWAT to develop a variable-density groundwater flow model for the groundwater system underlying a densely populated rural community on the north shore of Nova Scotia, Canada. We then impose various climate change scenarios (sea-level rise, changes in recharge) to evaluate the risk of OWTS inundation from rising groundwater. We find that under current conditions as many as 9% of OWTS in this small but densely populated watershed are either inundated or completely flooded. This number that could grow to 27% of OWTS for the climate change scenario with the highest recharge and sea-level rise. As a secondary objective, we track the location of the modeled saltwater-freshwater interface, and investigate the potential salinization of groundwater resources used for drinking water supply. The modeled interface moved landward by ≤20 m and proved to be less of a concern than OWTS inundation, except for shoreline dwellings. We also use the modelling results to develop adaptation recommendations for OWTS installation regulations. This research contributes to an increasing number of groundwater modelling studies focusing on the impacts of sea-level rise on coastal subsurface infrastructure and provides important insight for rural coastal communities.

Keywords – Groundwater table rise: Climate change impacts on groundwater; Subsurface infrastructure inundation; Submarine groundwater discharge; Coastal contamination; Saltwater intrusion

3.2 – Introduction

Accelerating sea-level rise (SLR) is threatening coastal communities globally (Kulp & Strauss, 2019; Oppenheimer et al., 2019). For example, in much of the US and Canada, sea levels are predicted to rise more than 1m by 2100 (Oppenheimer et al., 2019; James et al., 2021), which poses new threats to coastal infrastructure and freshwater resources. While the impacts of SLR on land surface conditions (e.g., coastal flooding and erosion) are widely recognized, less public attention has been placed on the unseen impacts of SLR on groundwater quantity and quality (Michael et al., 2017). Currently, more than 10% (625 million people) of the global population live in low-elevation coastal zones, an estimate projected to nearly double by 2100 (Neumann et al., 2015), lending heighted importance to research focused on SLR and the impacts to groundwater and coastal water quality.

In coastal subsurface environments, fresh groundwater meets saline groundwater along a saltwater-freshwater interface where less dense fresh groundwater 'floats' above a wedge of denser saline groundwater (Glover R.E. 1959; Werner et al., 2013). The slope, shape, and approximate location of this interface depends on aquifer conditions, tides, climate, and groundwater management (Werner & Simmons 2009; Kuan et al., 2012; Ketabchi et al., 2016). This interface may move inland due to climatic or anthropogenic perturbations (e.g., SLR or groundwater abstraction) in a process known as saltwater intrusion (Werner et al., 2013). Also, meteoric groundwater flows to the ocean as a component of submarine groundwater discharge (SGD) (Taniguchi et al., 2002; Burnett et al., 2006), and can carry terrestrial contaminants that pollute coastal zones (Sawyer et al., 2016). Consequently, there are both seaward (contaminated SGD) and landward (saltwater intrusion) contamination processes that occur along the subsurface coastal transition zone, and contaminant transport in both directions can be impacted by SLR (Werner et al., 2013; Befus et al., 2020).

The impacts of rising sea levels on coastal aquifers have been synthesized in review articles, with a predominant focus on saltwater intrusion processes (e.g., Werner et al., 2013; Ketabchi et al., 2016). For coastal aquifers with an unsaturated zone, as sea level rises, the groundwater table rises in unison, which decreases the unsaturated zone thickness (Werner & Simmons 2009; Bjerklie et al., 2012; Rotzoll & Fletcher 2013). Water tables rising above desired levels is referred to as groundwater flooding, which is of concern in areas with dense coastal populations, as subsurface infrastructure (e.g., foundations, sewer networks, water distribution pipes, drains, septic systems) may be impacted (e.g., flooding, corrosion), resulting in costly damages (de Almeida & Mostafavi 2016; Habel et al., 2017; Knott et al., 2017; McKenzie et al., 2021). Unusually high tides and storm surges can compound the effects of SLR, propagating elevated sea levels through coastal aquifers and temporarily elevating groundwater tables by tens of centimeters (Mao et al., 2006). In general, coastal groundwater flooding can be caused by superimposed climate (SLR, changes in recharge) and anthropogenic (coastal zone development) changes that exacerbate existing threats to coastal infrastructure.

On-site wastewater treatment systems (OWTS), often referred to as septic systems, are uniquely threatened by SLR (Cooper et al., 2015; Cox et al., 2019). OWTS are common in rural or underdeveloped areas where municipal wastewater treatment services do not exist. They function by distributing primary treated wastewater to a subsurface disposal field comprised of filter sand or native soils. Contaminants are treated by physical, chemical and biological processes as the effluent flows through the disposal field. Adequate attenuation of many contaminants of concern relies on some prescribed depth of unsaturated flow through the sand/soil materials (Cogger et al., 1988; Lusk et al., 2017). For example, when an OWTS becomes partially inundated, there is a corresponding decrease in the removal of many microbial colloids (such as *E. coli*), which are typically strained in the air portion of pore space of soils during unsaturated flow (Morales et al., 2015; Cox et al., 2019). Modern OWTS design standards (e.g., Nova Scotia Environment, 2017) specify minimum separation distances between groundwater tables and the bottom of the OWTS drain field (typically on the order of 1m) to ensure unsaturated flow. However, these guidelines do not account for potential groundwater table rise due to SLR, and typically do not apply to systems installed before the adoption of modern design standards. In addition to long-term, persistent threats imposed by SLR, OWTS in low-elevation coastal zones have been shown to become temporarily inundated during storm surges and high tides (Cox et al., 2020).

Previous studies have modelled the projected impacts of SLR on groundwater table elevation, notably along the east and west coasts of the United States (Cox et al., 2019; Befus et al., 2020), and bench-scale experiments have been conducted to evaluate the impact of elevated water tables on OWTS performance (Cooper et al., 2016). Modelling studies have evaluated colloid filtration under changing climate scenarios (Morales et al., 2015), while analytical studies and numerical simulations have been used to predict ground surface inundation with projected SLR (Rotzoll & Fletcher 2013; McKenzie et al., 2021). In general, much of the past SLR modelling research has focused on ground surface inundation, and only a few studies have employed the use of groundwater flow models to predict groundwater table dynamics under changing marine conditions. To our knowledge, no 3D variable-density modelling studies have evaluated the impact of SLR on OWTS inundation anywhere globally. Further, no studies focused on SLR and OWTS performance have been conducted in Canada, which has the world's longest coastline and is characterized by some of the highest projected SLR rates globally (Atlantic Canada, James et al., 2021) and high rates (~50% in some provinces) of OWTS use (Nova Scotia Environment, 2013). In this study, we used a variable-density groundwater flow and solute (salt) transport model to investigate the impact of SLR on groundwater table elevation and OWTS inundation in a rural but densely populated coastal community in Atlantic Canada that relies on both OWTS and residential water supply wells. Our primary objectives were to (1) evaluate the impact of various SLR and groundwater recharge scenarios on groundwater table elevation at our study site, (2) assess OWTS inundation under these same corresponding groundwater table elevations based on the distribution of residential dwellings, and (3) propose new considerations for OWTS guidelines to proactively design future systems with climate change in mind. As a secondary objective, we also investigate potential saltwater intrusion (subsurface saltwater-freshwater interface movement) due to climate change and consider the implications for groundwater resources that support this community.

3.3 – Study Site

Our study site is located on the north shore of the province of Nova Scotia, Canada, approximately 150 km north of the city of Halifax (Fig. 1). This area primarily consists of farmland, forests, a provincial park, and rural residential properties (Fig. 1) (Nova Scotia Department of Natural Resources and Renewables, 2021). The park includes an upland area, a forested area, and a wetland complex nearer the ocean. The region is underlain by sedimentary bedrock that is part of the Pictou Group and includes the Cape John and Tatamagouche formations (Ryan, 1985). These formations consist of primarily interbedded sandstones and mudstones, with groundwater flow taking place via intergranular flow, and along fractures and bedding planes (Hennigar, 1968; Rivard et al., 2008). Overburden material consists primarily of

0.6 to 10 metres of coarse- to fine-grained glacial tills. The shoreline along the Northumberland Strait is a public sandy beach frequented during the summer months. The beach has a tidal range of 2.5 m and is a relatively low-energy wave environment.

The climate is typical of the Canadian Maritimes, with mean daily air temperatures ranging from 19 °C during the summer to -3 °C in the winter (Government of Canada, 2021, ECCC Station 8205774). The region receives close to 1000 mm of precipitation annually, with recharge to the underlying bedrock estimated to range between 180 – 220 mm/year based on analysis of baseflow data (Kennedy et al., 2010; Government of Canada, 2021, ECCC Station 8205774). The provincial park is surrounded by private cottages and residential properties, all of which rely on OWTS for wastewater treatment and groundwater wells for drinking water (Fig. 1). A recent provincial GIS-based analysis classified this region as at a mid-high risk of saltwater intrusion (Kennedy et al., 2012), but no process-based modeling of coastal groundwater processes has been conducted for this site. Field observations and GIS analysis indicate that many of these properties are within meters of the current high-water mark, suggesting their OWTS could already be threatened during high tides and storm surges. Current OWTS standards for Nova Scotia (Nova Scotia Environment, 2017) indicate that many of these systems should be raised or at-grade systems based on soil permeability and water table position. However, some of the older properties in the area were constructed prior to 1975 when on-site sewage disposal system regulations first came into effect (Province of Nova Scotia, 1975). Previous research using conventional and novel (crAssphage) fecal indicators measured in submarine groundwater discharge and in coastal waters indicated that OWTS at this site are likely already contributing contaminants into the ocean (Threndyle et al., 2022).

3.4 – Methods and Data

3.4.1 – Data Sources

Data for model development and calibration were collected from provincial online databases, regional reports, and multiple field campaigns in the study area. The horizontal extents of the model domain are informed by watershed divides, which were assumed to represent groundwater divides, and the vertical extent is based on the maximum depth of 34 available borehole logs from the Nova Scotia Groundwater Atlas (Nova Scotia Department of Natural Resources and

Renewables, 2021). Ground surface elevation (<1 m horizontal resolution, 0.1 m vertical resolution) was downloaded from the GeoNova Provincial LiDAR database (digital elevation model) and was combined with offshore ocean floor elevation data from the Canadian Hydrographic Service's Non-Navigational bathymetric database (Nova Scotia GeoNova, 2020; Fisheries and Oceans Canada, 2021). OWTS locations were inferred from residential property locations downloaded from the Nova Scotia Geographic Data Directory (Nova Scotia GeoNova**,** 2018). Groundwater head and sea-level data (e.g., horizontal hydraulic gradients and tidal fluctuations) were recorded during multiple field campaigns and through continuous data logging by four piezometers and a tidal logger (Fig. 9) as described in more detail in Threndyle et al. (2022). Hydrogeological parameters used in the model were estimated from literature, including local sources (Hennigar, 1968), and adjusted (within constraints) during model calibration.

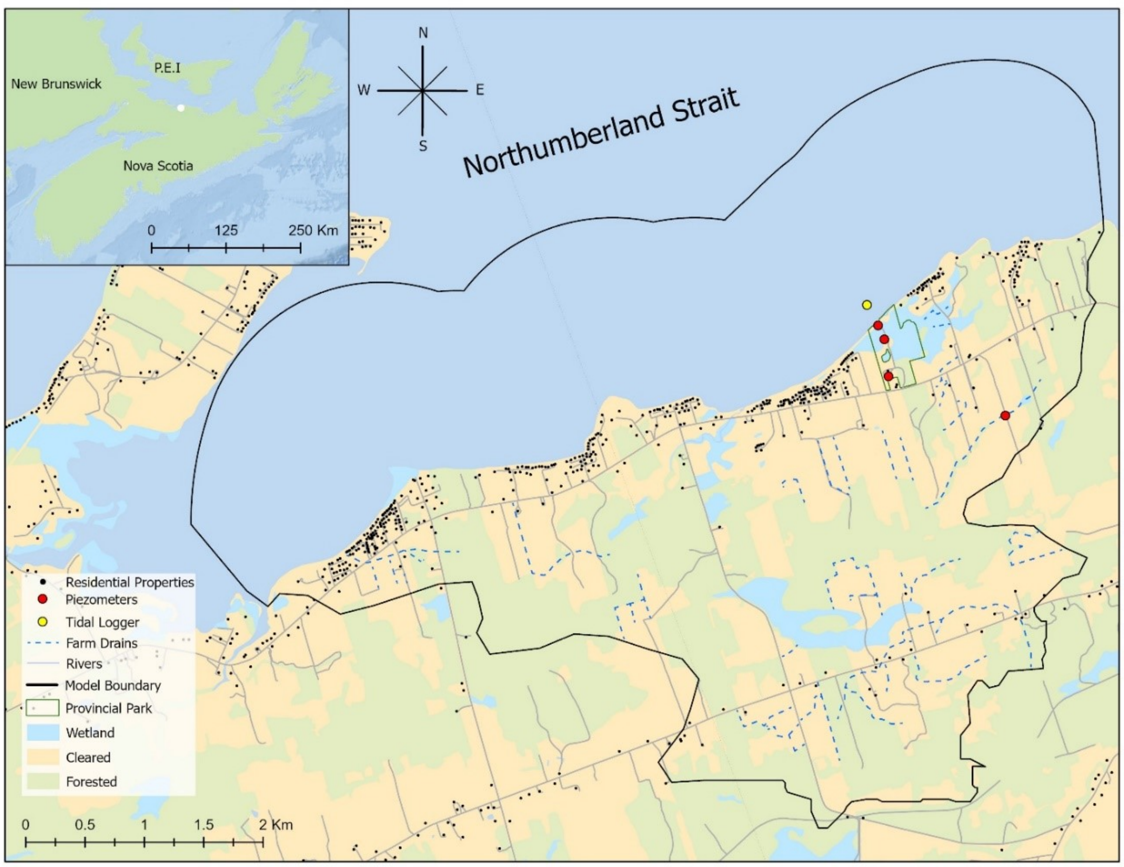

Figure 9 Map of the study site and modelling domain extents highlighting residential properties (and thus OWTS locations), piezometer and tidal logger locations, as well as surface hydrology features. The inset shows the location (white circle) within the Canadian Maritimes.

3.4.2 – Overview of Numerical Model of Groundwater Flow and Solute Transport

The Visual MODFLOW Flex interface (Waterloo Hydrogeologic, Waterloo ON. Canada) was used to develop and run the groundwater flow simulations using the variable-density coupled groundwater flow and solute transport code SEAWAT (Guo & Langevin 2002). Visual MODFLOW Flex provides a graphical user interface to parameterize a gridded model domain on a cell-by-cell basis and assign boundary and initial conditions. SEAWAT effectively links MODFLOW (groundwater flow) and MT3DMS (solute transport, Bedekar et al., 2016) and is commonly applied in coastal groundwater studies (Langevin et al., 2009) to account for the coupled solute (salt) transport, variable-density flow, and related saltwater–freshwater dynamics that occur in the coastal zone. The groundwater flow engine employed by this version of SEAWAT is MODFLOW-2000, which is a saturated zone model. The elevation of the groundwater table is taken as the hydraulic head in the uppermost active grid cell. MODFLOW has been employed to evaluate coastal groundwater table dynamics in previous studies (Habel et al., 2017; Sukop et al., 2018; Befus et al., 2020). We go beyond many prior coastal groundwater table studies by also including SEAWAT given our desire to represent the saltwater-freshwater interface and its influence on coastal groundwater discharge patterns under present and future climate conditions.

3.4.3 – Geological Model

Well logs from the Nova Scotia Groundwater Atlas were used in combination with field data to create a representative geologic model using Leapfrog software (Seequent Limited, Christchurch, New Zealand). This tool uses imported borehole data and interpolates contact surfaces between associated units before creating 3D volumes representing the different bedrock units. Modelled stratigraphy is shown in Figure 10. The surficial geology is represented in the model as fine-grained glacial till, which was also observed during our piezometer installation. The underlying sandstone and shale units were interpreted from borehole data from the Nova Scotia Groundwater Atlas (Nova Scotia Department of Mines, 2021). Borehole information was simplified to generate a reasonable number of units to accurately represent groundwater flow in the model.

3.4.4 – Model Discretization

Leapfrog was also used to discretize the model domain, which was imported into Visual MODFLOW Flex. Our grid consists of cells that are 20m wide (longshore) by 40m long (cross shore), fining down to 20m wide by 10m long near the saltwater-freshwater interface. Grid refinement around the interface was required to enable model convergence where solute concentration gradients were high. Our model has 14 layers, spaced closer together near the ground surface (~3m vertical thickness) and further apart at depth (~6m thickness). The model base is ~63m below sea level, which corresponds to the depth of the deepest geologic information in the available well logs.

3.4.5 – Boundary Conditions

The model sides and bottom boundaries were set as no-flow conditions (Fig. 10), in accordance with expected groundwater divides along the sides (regional watershed) and predominantly horizontal flow at depth (Anderson et al., 2002). A recharge boundary condition with constant freshwater flux was applied across the land surface of the domain, with the recharge flux initially informed by reported baseflow data (Kennedy et al., 2010). Drain boundaries were assigned to swamps, streams, and agricultural drains across the site. The swamps and streams had leakance values of 50 m/day, and the agricultural drains had leakance values of 25 m/day. The top layer of the portion of the model representing the ocean was assigned a constant head and constant concentration boundary, corresponding to measured mean sea level at the site and seawater concentration (32,000 mg/L of dissolved salts), respectively (Hussain M.S., 2015).

When modelling climate change scenarios, the ocean boundary condition (specified head) was raised at a constant rate between 2020 and 2100 such that the 2100 sea level was in accordance with ensemble SLR projections downscaled to the site (James et al., 2021). SLR rates for RCP4.5 and 8.5 at the 5th and 95th percentile were used to inform the SLR rates imposed in the modeled climate scenarios (Table 2). In two runs (R4.1 and R4.2) with the highest SLR, the recharge boundary condition was also adjusted in equal intervals every 20 years. Future recharge rates were considered to investigate the sensitivity of the groundwater table dynamics to paired changes in sea level and recharge. Future recharge rates were obtained from future precipitation projections (Lines et al., 2006) by invoking the simplifying assumption that the recharge/precipitation ratio remains constant. As future recharge estimates are highly variable and uncertain even in terms of the direction of change (Kurylyk & MacQuarrie, 2013), we have modelled a high and low recharge scenario with identical magnitudes but opposite signs for the recharge change. The influence of tides was ignored in this study, as our purpose was to study long-term changes in groundwater table elevations and the potential impacts to fresh submarine groundwater discharge flowing under OWTS, rather than recirculated submarine groundwater discharge (LeRoux et al., 2021).

Table 2 Numerical model runs and associated boundary conditions

Run Name	**Sea-Level Rise Scenario**	**2100 Modelled SLR (cm)**	**Annual SLR (cm/year)**	**Recharge (mm/year)**
R0	Current Conditions	NA	NA	165
R1	RCP 4.5 5th Percentile	40	0.5	165
R2	RCP 4.5 95th Percentile	100	1.25	165
R3	RCP 8.5 5th Percentile	60	0.75	165
R4	RCP 8.5 95th Percentile	120	1.5	165
R4.1	RCP 8.5 95th Percentile	120	1.5	165 → 204
R4.2	RCP 8.5 95th Percentile	120	1.5	165 → 127

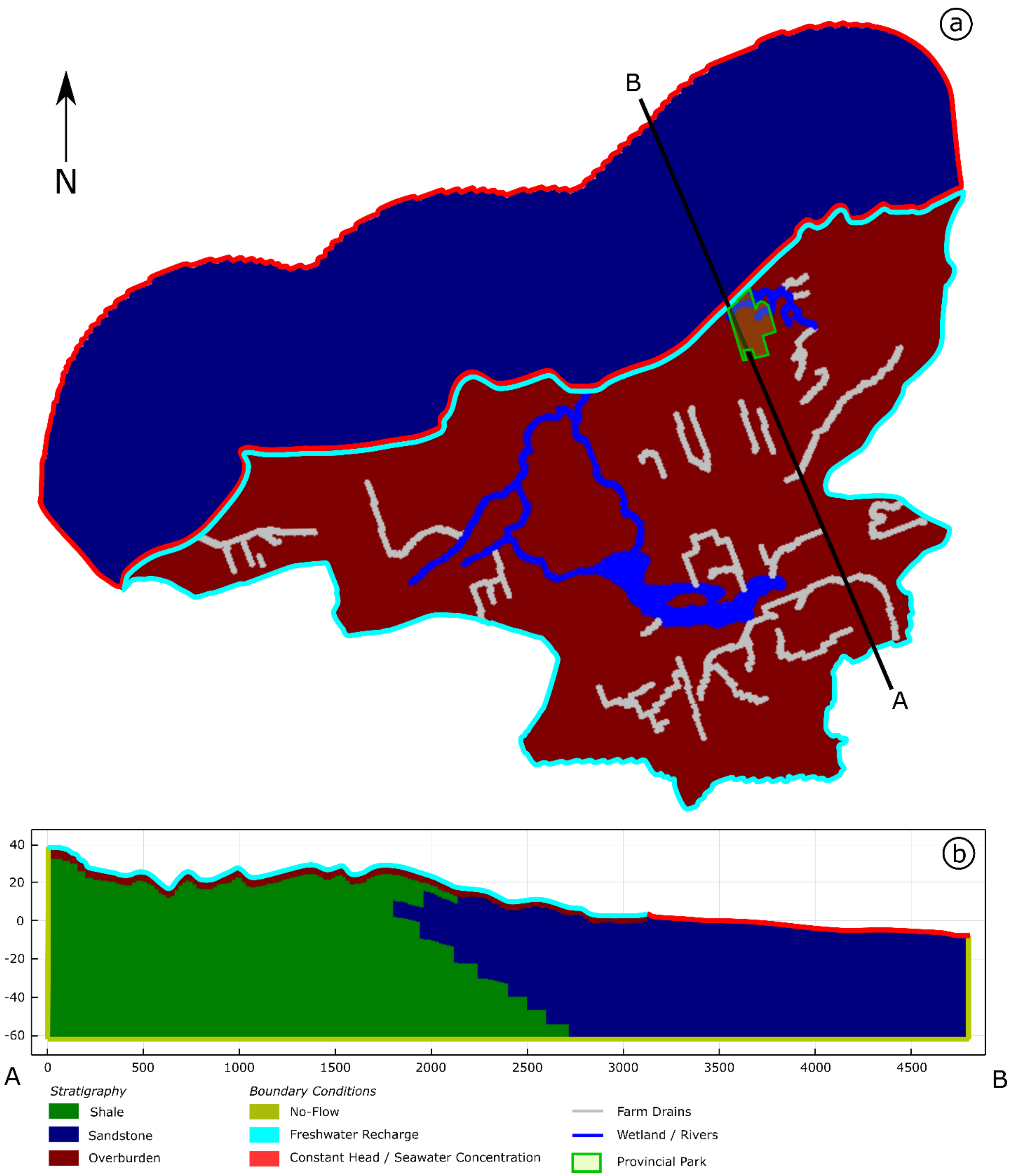

Figure 10 Plan view (a) and profile view (b) of model domain highlighting stratigraphy and boundary conditions with the lower legend applying to both panels. In panel (a), the freshwater recharge and constant head boundary conditions are shown as an outline of the surface area they are applied to in the model.

3.4.6 – Initial Conditions

Initial groundwater heads were assigned to match observed mean annual heads in the installed piezometers (Fig. 9). A model simulation of 50 years was run repeatedly to allow groundwater flow to reach steady state, and model parameters were continuously adjusted until calculated and observed head conditions agreed. Hydraulic conductivity of the overburden, shale, and sandstone units were initially estimated based on literature values (Rivard et al., 2008) and field measurements, and then adjusted during calibration. Rivers, swamps, and farms drains, which were delineated from aerial imagery of the site, were included in the model to better represent surface drainage. The default recharge boundary condition (165 mm/year, Table 1) was determined by adjusting the value in small increments to help match calculated to observed head data.

Following calibration, a 50-year model 'spin-up' period was employed to allow the saltwater-freshwater interface to reach steady state before any climate change forcing was introduced. A block of saltwater (32,000 mg/L) was imposed vertically below the constant head sea-level boundary to the bottom of the model domain to decrease the time needed for the saltwater–freshwater interface to reach steady state conditions. Attempts were made to calibrate the wedge location to collected, local time-domain EM (WalkTEM, ABEM) geophysical data following Pavlovskii et al. (2022), but interpretation was challenging due to the coarse resolution of the geophysical soundings as well as the stratigraphic changes that obscured resistivity changes due to porewater salinity.

3.4.7 – Transient Runs and Post-Processing

Calibrated, steady-state groundwater head and salt concentration distributions were used as initial conditions to run six climate change scenarios (Table 2). Groundwater table elevation data for each model grid cell was exported from Visual MODFLOW Flex at various timesteps, and imported into ArcGIS Pro to evaluate the impact on residential properties and associated OWTS. The model ground surface raster was subtracted from the groundwater table elevation raster to reveal the depth to the groundwater table and the areas of ground surface flooding for each of the climate change scenarios (Fig. 11). There are 472 septic systems in the modelling domain, one for each property ID reported by the province. Inundation for each septic system was evaluated based on the depth to groundwater in the grid cell corresponding to the location of the septic system. We assumed the bottom of the drain field (unsaturated zone) of each septic system was 1.6m below ground surface, as is required by provincial guidelines for septic systems to ensure adequate filtration of wastewater (Nova Scotia Environment, 2013). Any location where the simulated water table was within 1.6m of the ground surface was considered 'inundated', and any location where the simulated groundwater table was above the ground surface was considered 'flooded'.

Seawater (TDS) concentration, including the location of the saltwater–freshwater interface, was visualised using postprocessing capabilities in Visual MODFLOW Flex. A 50% concentration line (16,000 mg/L) was plotted along a designated column of cells for calibrated initial conditions and climate chance scenarios, and saltwater intrusion was investigated and quantified by the landward movement of this modeled interface (iso-concentration line).

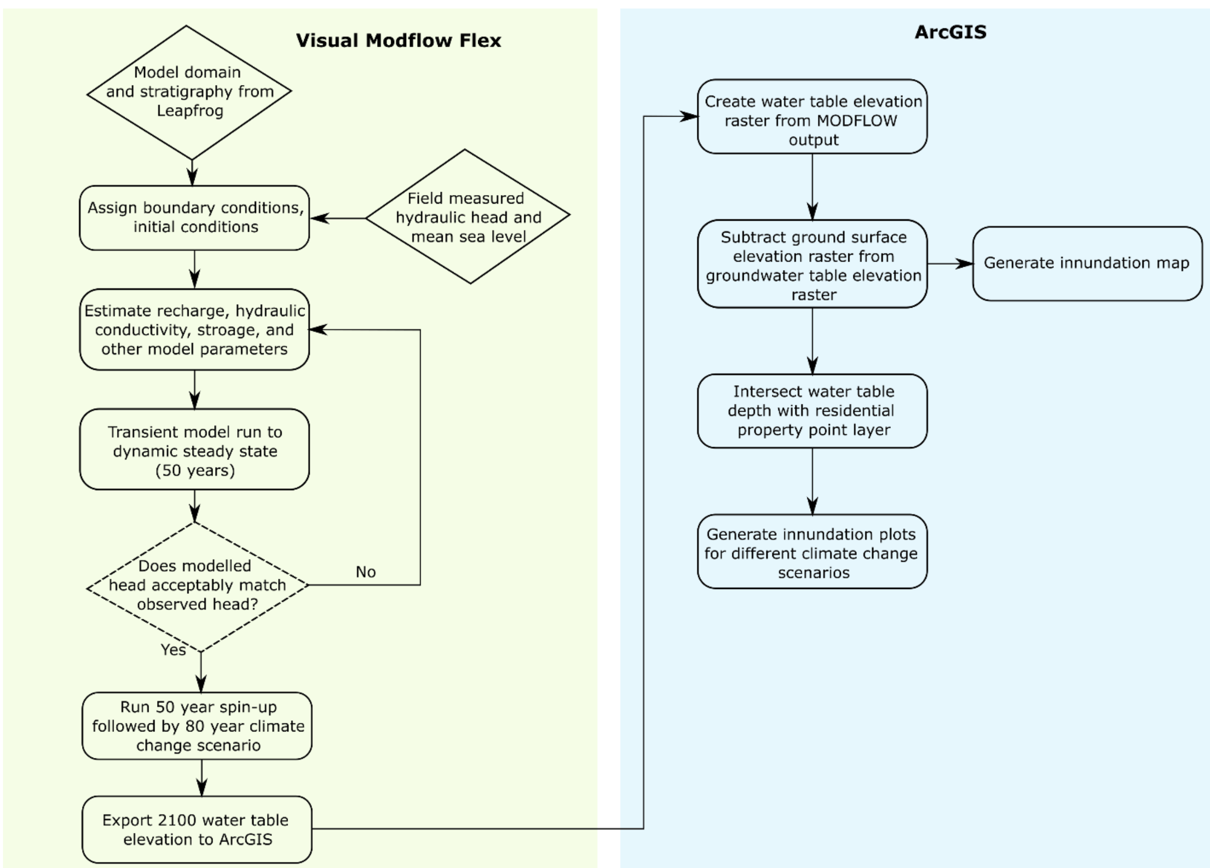

Figure 11 Flow chart indicating the workflow for data processing in the numerical model interface (left) and ArcGIS (right) to quantify the impacts of SLR and changing recharge on groundwater table elevation and OWTS performance

3.5 – Results

3.5.1 – Numerical Modelling Calibration Results

Several iterations of calibration (Fig. 11) were required to achieve agreement between measured and observed heads across the domain. Calibrated values for recharge, storage, drains/river leakance, and hydraulic conductivity of the different geologic units are shown in Table 3. Hydraulic conductivity of the overburden and shale units, and recharge were the most sensitive parameters, and as such, were the primary focus of model calibration. A plot of the observed vs. calculated head values is shown later in Figure 14d. While calibration points are limited, modelled head in the piezometers nearest to the beach are in closest agreement with observed head, lending confidence to our calibration in the areas most susceptible to SLR.

Our calibrated recharge of 165mm/year was in general agreement with baseflow-derived estimates of 180 mm/year (Kennedy et al., 2010). The final hydraulic conductivity of the shale layer is orders of magnitude higher than a typical shale (e.g., Freeze & Cherry, 1979); however, there is documented, local evidence of primary groundwater flow occurring along bedding planes and between bedrock fractures (Gibb & McMullin, 1980). Further, the geology of the region is characterized by interbedded sandstone and shale units (Hennigar, 1968; Ryan, 1985). This interbedded geology was excluded from the model (Fig. 10) to limit complexity, but its net effect is likely represented in the higher calibrated hydraulic conductivity of the shale unit. The conductivity of the overburden layer reflected the soils present in the provincial park where piezometers were installed and was also adjusted during calibration, with the calibrated value (2.0×10^{-6} m/s) typical of glacial till for the region (Rivard et al., 2008). Based on observations of borehole cuttings during well installation, the overburden was predominantly silty clay with varying amounts of sand, which is in agreement with provincial surficial geology maps (Stea & Flinck 1984).

Table 3 Calibrated boundary conditions and flow parameters

Parameter	Value
Recharge	165 mm/year
Specific storage (all units)	1×10^{-6} /m
Farm drain leakance	25 m/day
River / wetland leakance	50 m/day
Overburden *K**	2×10^{-6} m/s
Sandstone *K*	1.5×10^{-5} m/s
Shale *K*	2.5×10^{-6} m/s

**K* is hydraulic conductivity. *See geologic model (Fig. 10) for location of stratigraphic units. Beach sand was not considered in the model because (1) the layer is relatively thin and (2) it did not underlie any OWTS systems which were the focus of the study.*

3.5.2 – Impact of SLR on Groundwater Table Elevation and OWTS

The calibrated, steady-state salinity and head distributions were perturbed by the climate scenarios to investigate impacts to the water table elevation and saltwater-freshwater interface. Figure 12 shows areas of OWTS inundation and ground surface flooding across the modelling domain for the calibrated ’current conditions’ (R0) and the R1, R3, R4, R4.1, and R4.2 scenarios (Table 2). Areas where groundwater flooding occurs under current conditions include the wetlands and low-lying areas adjacent to the river channels across the domain, which all have imposed drain boundaries. Each climate scenario with higher SLR rates results in progressively more surface flooding and near-surface inundation of OWTS across the domain (Fig. 12). While much of the impacted OWTS are along the coast, the high and low recharge scenarios (R4.1, R4.2) also impact the upland OWTS systems south of the wetland (Fig. 12e,f). This model result is expected as SLR would have a reduced effect on groundwater table elevations further from the coast, while elevated recharge would affect groundwater tables across the entire domain.

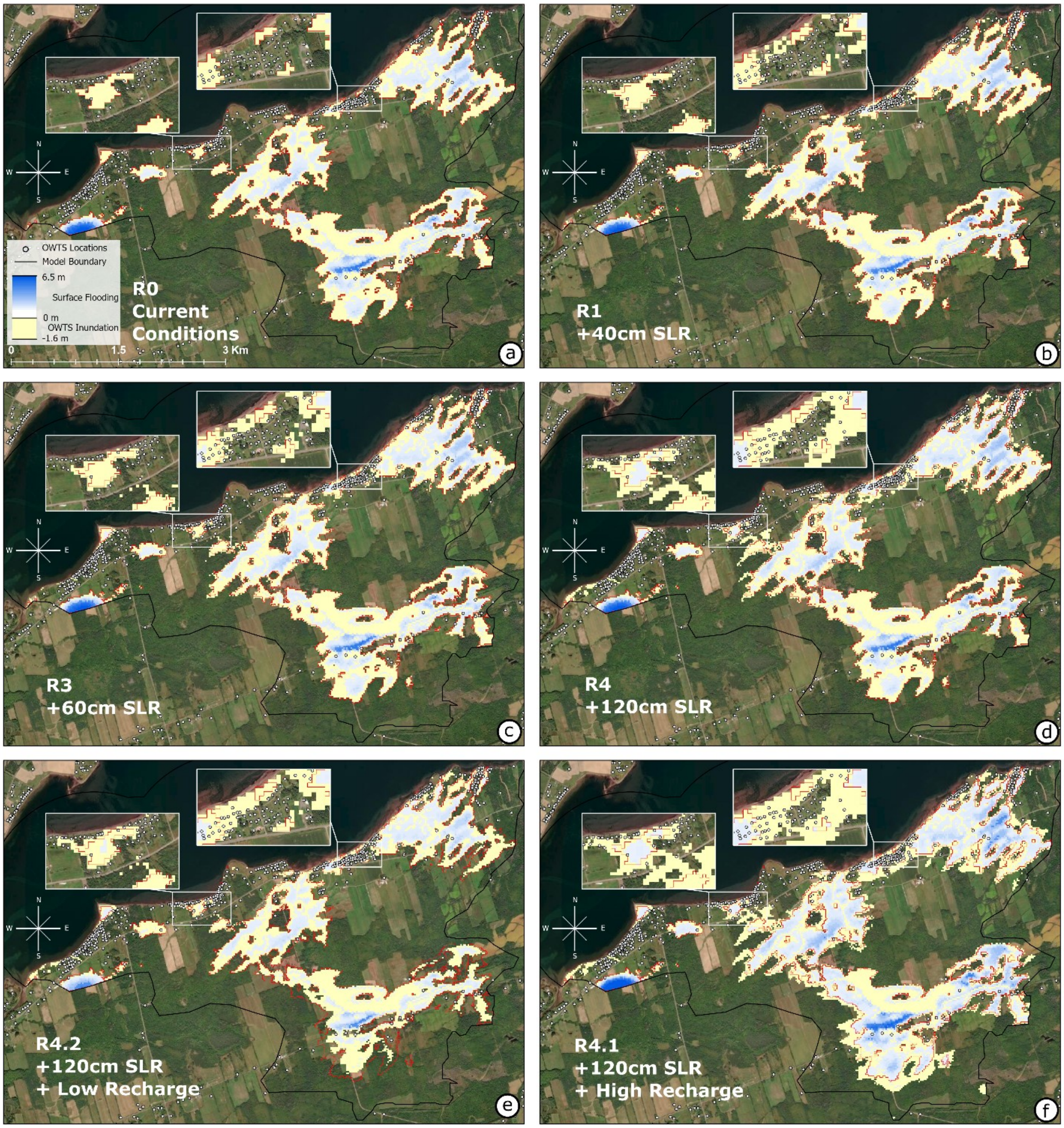

Figure 12 OWTS and ground surface inundation maps for SEAWAT modeling results for (a) current conditions and various climate change scenarios: (b) +40 cm by 2100, (c) +60 cm by 2100, (d) +120 cm by 2100, (e) +120 cm and low recharge by 2100, (f) +120 cm and high recharge by 2100. Yellow shading indicates the water table is within 1.6 m below the ground surface, and blue shading indicate the water table is above ground surface. Red line, which is visible when zoomed in, indicates the inundation extent under current conditions as point of comparison for the climate change scenarios.

While Figure 12 illustrates the general spatial pattern of flooding, Figure 13 (a – c) presents the holistic impacts of SLR on the OWTS across the domain. The grouped bar chart (Fig. 13a) shows that each progressive increase in mean sea level leads to more OWTS being inundated and a greater extent of flooding. For example, under current conditions, the model and subsequent GIS workflow (Fig. 11) suggest 40 OWTS have their filters inundated, while 3 OWTS are flooded. With 100cm of SLR and recharge left unchanged, 83 filters are inundated and 16 are flooded, and with 120cm of SLR and higher recharge, 99 filters are inundated and 29 are flooded (Fig. 13a). In addition, scenarios with higher SLR and recharge shift the distribution of the elevation difference between the land surface and groundwater table (Fig. 13b), which indicates a transition to more inundated and flooded OWTS with progressive SLR and enhanced recharge. Figure 13c shows OWTS status (colors) and difference between the water table and surface elevations vs. the ground surface elevation of all properties in the modelling domain following 120cm of SLR and high recharge, demonstrating unsurprisingly that lower elevation OWTS are at higher risk of inundation.

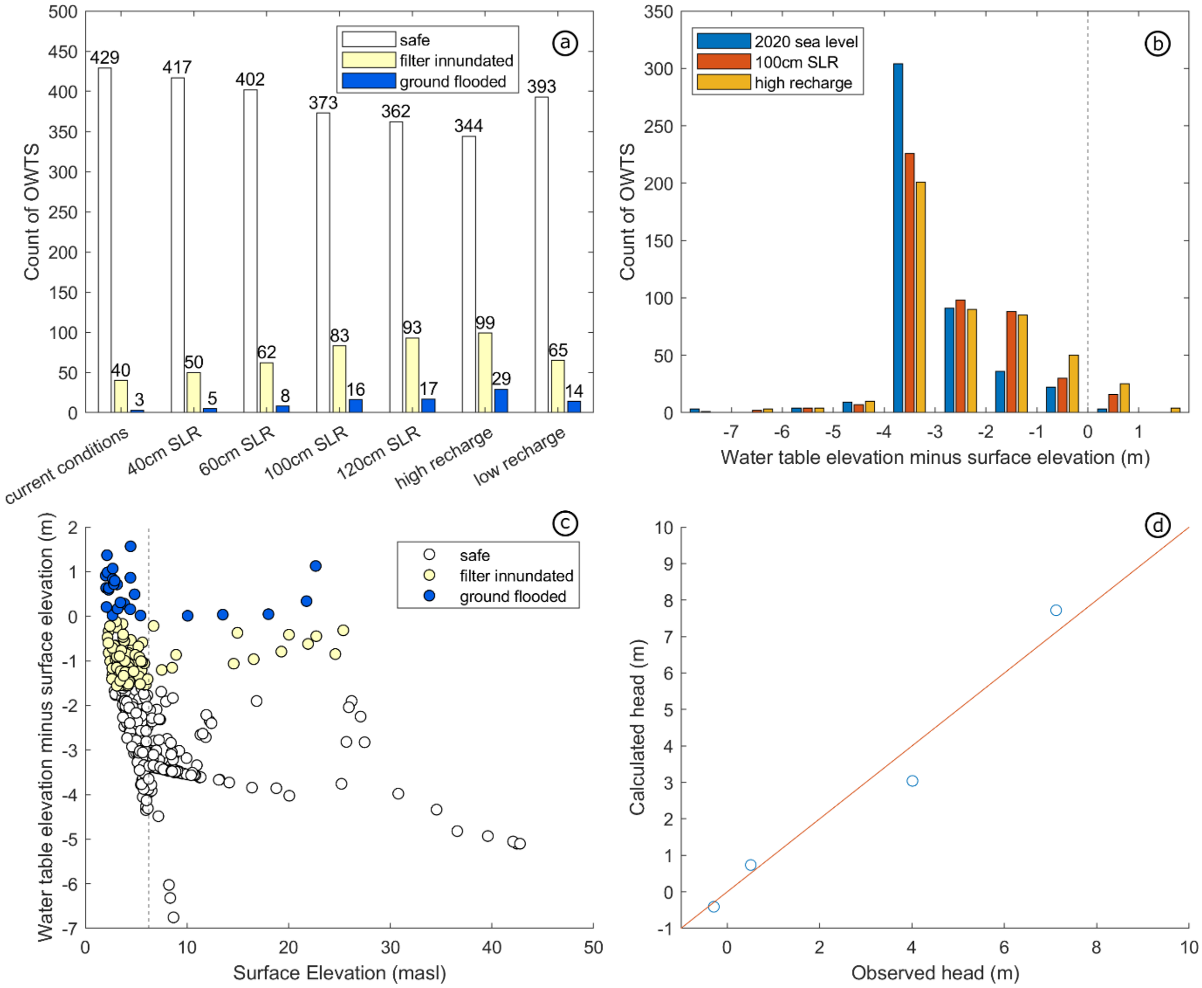

Figure 13 (a) Grouped bar chart showing counts of OWTS that are safe, inundated, and flooded for each climate change scenario. (b) Grouped histogram for three different climate change scenarios showing depth to groundwater table for the OWTS. (c) Plot of water table elevation minus surface elevation vs. surface elevation for all properties in the modelling domain showing OWTS inundation status following 120 cm of SLR and high recharge. (d) Model calibrated vs. field observed groundwater head in site piezometers.

3.5.3 – Saltwater Intrusion Due to SLR and Changing Recharge

The saltwater-freshwater interface moved landward in response to SLR and reduced recharge and moved seaward in response to increased recharge. Figure 14 shows the saltwater–freshwater interface under current conditions (2020 mean sea level and 165 mm/year recharge), and for 2100 for each climate change scenario (Table 1), where each interface line represents a salt concentration that is 50% that of seawater (16,000 mg/L). The high recharge scenario (R4.1) has a significant impact on the movement of the wedge, with increasing SGD from the enhanced recharge overcoming the impacts of SLR and driving the salt wedge >10 m seaward compared to current conditions. The low recharge (R4.2) scenario had the opposite effect, with lowered recharge and 120 cm of SLR moving the saltwater–freshwater interface ~20 m landward. Each progressively higher SLR scenario (R1 – R4) resulted in additional landward movement of the interface at the ground surface, and variable conditions at the interface toe. Although only the location of the 50% interface is shown, not all water to the left of the interface is 100% fresh, and 100% saline to the right, rather a gradient is present. Further, the impact of tidal pumping was not modelled here, which would alter the salinity distribution, particularly close to the surface along the coastal interface.

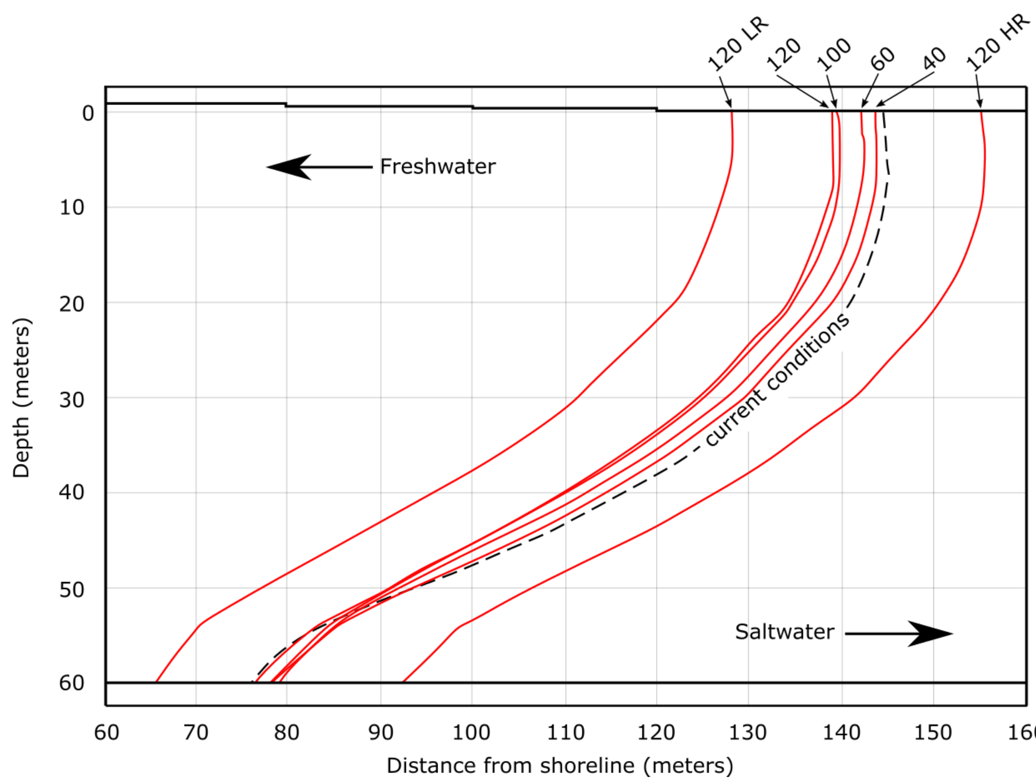

Figure 14 The 50% (16,000 mg/L) location of the saltwater–freshwater interface for current conditions and climate change scenarios (numbers on line labels indicate SLR amount in cm; LR = lower recharge and HR = higher recharge).

3.6 – Discussion

3.6.1 – Implications of SLR on Groundwater Table Elevation and OWTS

The modelling results demonstrate that progressive SLR has an impact on the number and degree to which OWTS in the region are inundated by elevated groundwater tables (Fig. 12, 13). Even under current conditions, model results suggest that as many as 43 systems in the region are compromised though filter inundation or surface flooding (Fig. 13a), which could explain elevated levels of fecal indicator bacteria and enteric viruses detected in samples of coastal water and SGD at this site in 2021 (Threndyle et al., 2022). This aligns with studies by Cox et al., (2019) and McKenzie et al., (2021), highlighting presently compromised OWTS and other subsurface infrastructure, and linking this inundation to enhanced seaward contaminant transport. OWTS inundation and ground surface flooding increases steadily with each SLR scenario, with up to 110 compromised systems for 120 cm of SLR. Inundation maps (Fig. 12) and plots (Fig. 13c) reveal that much of this inundation occurs at low elevation and near the ocean, as well as within the vicinity of the provincial park (Fig. 9) which intersects a tidally influenced wetland. This is in agreement with SLR and groundwater table modelling results by Habel et al. (2017) that pointed to inundation in low-lying and near-shore areas with progressive SLR.

The influence of recharge on groundwater table elevations was also investigated. The high recharge, high SLR scenario (R4.1) compromises 128 OWTS across the domain, and significantly impacts the flooding locations in the model upland compared to the scenarios modelling strictly SLR (Fig. 12d vs. 12f, 13a). By contrast, the low recharge scenario with high SLR produced a significant decrease in upland inundation, even compared to current conditions (Fig. 12a vs. 12e). There are still, however, more OWTS compromised in the 120 cm SLR plus low recharge scenario (R4.2), than in the 60 cm SLR scenario (R3), with the compromised OWTS existing mostly along the shoreline. This indicates that recharge changes have an important, but different impact on OWTS inundation compared to SLR. The results also demonstrate that decreased recharge will not fully counteract the impact of rising sea levels on subsurface inundation and flooding, at least for the limited recharge scenarios considered herein. Alternatively, a series of high recharge years in the absence of SLR could still have significant impacts on groundwater table elevations and thus OWTS inundation across the entire domain. These findings expand our understanding from previous studies examining the impacts of SLR on groundwater table elevation (Habel et al., 2017; Befus et al., 2020) and suggest that recharge should be considered in future studies, as incorporated in some recent studies examining the compound effects of SLR, tides, and rainfall (Sukop et al., 2018).

Figure 15 presents a conceptual model of the impacts of SLR and changing recharge on groundwater elevations and the associated impacts to OWTS and SGD-borne coastal contamination. The results of our modeling indicate that an increased number of OWTS in our study area will be impacted by these processes. Although not explicitly studied here, the ability of a soil filtration system to remove contaminants is strongly influenced by moisture content (i.e., degree of saturation) (Cooper et al., 2016; Morales et al., 2016). Inundated disposal fields limit the adsorption and straining of many colloids, mobilizing these contaminants towards coastal waters through SGD (McKenzie et al., 2021). The removal of other wastewater constituents (e.g., nitrogen) could also be impacted, but will likely vary depending on severity and timing of groundwater inundation (Morales et al., 2016). As the households in the study area also rely on groundwater for their domestic water supply, contaminants that reach the saturated zone could be intercepted by drinking water wells, impacting domestic water quality for residents.

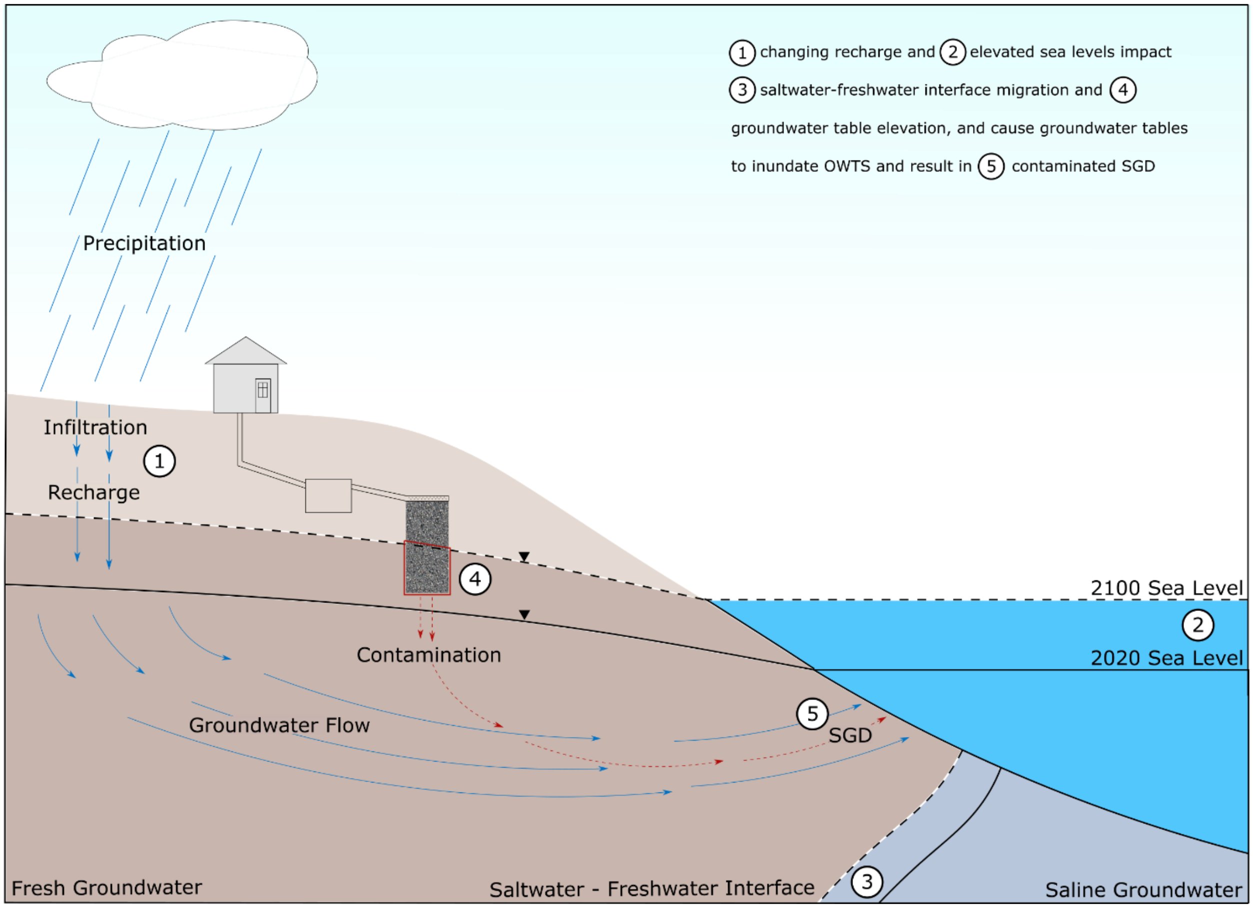

Figure 15 Conceptual diagram showing OWTS inundation resulting from climate change processes (adapted from Threndyle et al., 2022).

3.6.2 – Impacts of SLR and Changing Recharge on the Saltwater-Freshwater Interface

The locations of the saltwater–freshwater interface under present-day conditions and climate change scenarios are shown in Figure 14. Movement of the interface for the highest SLR scenario (120 cm by 2100) occurs primarily at the groundwater table. The relatively minor movement of the salt wedge in response to SLR is because this aquifer is a ‘recharge-limited’ aquifer (Michael et al., 2013), as the water table can rise with the sea level and maintain the same hydraulic gradient if recharge is unchanged (Werner and Simmons, 2009). Recharge, however, does have a significant impact on the interface location. Increases and decreases in recharge impact the flux of SGD towards the ocean, which in turn influences the position of the interface

(Green and MacQuarrie, 2014). The high recharge scenario (R4.1) results in the furthest seaward interface, even when coupled with the highest SLR. In this way, recharge and SLR act as opposing forces on the interface position.

The contrasting impact of SLR and recharge on both groundwater table and interface movement (saltwater intrusion) has implications for evaluating what a 'best case' or 'worst case' scenario may be. From a OWTS contaminant transport perspective, *high* recharge and high SLR would elevate the groundwater table the most and inundate the maximum number of filters, driving a high volume of contaminants from OWTS seaward. In contrast, from a saltwater intrusion perspective, scenarios with *low* recharge and high SLR would result in the most landward interface movement and a greater risk of saltwater intrusion into domestic water supply wells. While our model results indicate that the interface location is sufficiently distanced from the shoreline so as not to impact the salinity of drinking water resources (Fig. 14), the salinity distribution could be impacted by both tidal mixing and pumping of residential wells which were not considered in the model. It is also important to note that the impacts of saltwater intrusion and OWTS-generated groundwater contamination may be related because porewater salinity can influence biogeochemical processes and contaminant mobility. For example, the introduction of more saltwater into the subsurface could increase the adsorption of viral and bacterial colloids, as increased ionic strength can supress the electrical double layer surrounding porous media (Knappett et al., 2008).

3.6.3 – Model Limitations

Model calibration was limited by the number of monitoring wells installed across the study site. Our calibration points were concentrated mainly within the provincial park (Fig. 9) as options for drilling on private property were limited, and access to unpumped private domestic wells was limited. Calibration of groundwater table elevation in the upland was facilitated by comparing visual observations of low-lying and wet areas from field trips and satellite imagery, with flooded areas in our model outputs.

We did not model groundwater pumping from the estimated 472 homes in the study site, which use between 680 – 1360 L/day of groundwater (Nova Scotia Environment and Labour, 2004). This pumping could both lower the groundwater table and bring the saltwater-freshwater interface landward; however, as these homes also rely on septic systems for waste disposal, it is reasonable to assume that this water would be returned to the ground at roughly the same rate it is extracted. Furthermore, the pumped volume from the homes is only 6.8% of recharge across the model domain (% may be higher locally) even during the limited summer occupancy, based on mean values for household water usage in this area.

Groundwater inundation in the model occurs when groundwater rises above the land surface. SEAWAT however does not simulate overland flow, runoff, ponding, evapotranspiration, or unsaturated flow. As a result, estimates of groundwater table elevation were limited to flow within the saturated zone.

3.6.4 – Implications for Guidelines and Coastal OWTS

Our modelling suggests that 43 OWTS (9%) in the region are compromised under current conditions, a number that may increase up to 128 OWTS (27%) with SLR and recharge increases (Fig. 13a). These findings, coupled with evidence of present fecal pollution from OWTS in the region (Threndyle et al., 2022), suggest that water quality along the coastline, which is also near a popular provincial park, could be further degraded in the future.

The modelling results have direct application to provincial OWTS guidelines. Although re-design of current systems is likely challenging, areas that are demonstrated to be at increased risk of inundation due to SLR should be considered in guidelines for future OWTS replacement and new system installation. In some cases, conventional disposal fields may not be feasible, and homeowners will need to use alternative treatment systems such as mechanically aerated treatment units which have additional operation and maintenance requirements. Figure 13c (dashed vertical line) demonstrates that at our study site households where the ground surface is within approximately 6.25m of current local mean sea level could be at increased risk of OWTS inundation. Under climate change scenario R4.1, 41% of properties below 6.25m of local mean sea level have their OWTS inundated, which is true for only 9% of properties above 6.25m. OWTS guidelines could be revised for this region to include increased vertical separation distances between the bottom of disposal fields and the groundwater table. Similar recommendations could be made for property and infrastructure development in the region. Basements, foundations, water supply, and other below-grade utilities should consider how elevated groundwater tables from rising sea levels would impact their long-term viability (de Almeida & Mostafavi., 2016; Knott et al., 2017; Hummel et al., 2018).

Finally, these recommendations should be considered elsewhere where high SLR and low-lying, densely populated cottage areas intersect. These protections are relatively simple, yet effective safeguards against key impacts of climate change. Enclosed or semi-enclosed coastal water bodies used for swimming, fishing, and aquaculture are of particular interest as less mixing and more population density enhances the chances of contamination from SGD-derived pollution, a phenomenon which could be accelerated with SLR.

3.6 – Conclusions

As climate change increasingly threatens coastal communities and water resources, our ability to assess the potential impacts that climate change will have on key infrastructure (e.g., OWTS) is increasingly important. Our study uses numerical modelling to demonstrate that as many as 27% of OWTS in our study region could be inundated or flooded under 120cm of SLR and high recharge conditions (vs. 9% under current conditions). We also show how recharge changes impact the spatial distribution of OWTS inundation, having greater effect than SLR further from the coastline. Finally, we propose that the installation of OWTS at properties in our study area where the surface elevation is within 6.25 m of local mean sea level should consider design modifications to increase the vertical distance between the bottom of their OWTS drain fields and the water table. These findings have applications outside of the study site, particularly where high SLR rates are projected along low-elevation coastal communities that rely on OWTS for wastewater treatment, and where groundwater and surface water quality are important to community health and recreational use of the area. In these zones, OWTS design guidelines may need to incorporate local predictions of potential water table changes under climate change scenarios.

The model results also highlight the contrasting risks to coastal groundwater from either salinization or OWTS. For example, despite the significant impacts to the water table elevation and associated OWTS performance, the results reveal that this aquifer is relatively resilient to climate change from a saltwater intrusion perspective, as the saltwater-freshwater interface movement due to the climate change scenarios is limited to ≤ 20 m and generally remains offshore. This aquifer is used for drinking water supply, and in general, increasing groundwater pumping would draw the water table down and protect against OWTS contamination of SGD, but could result in lateral saltwater intrusion. Also, changes in precipitation and resultant recharge would likely have counteracting effects on OWTS performance and salinization, with increased precipitation leading to more SGD and pushing the saltwater-freshwater interface seaward, but simultaneously elevating the water table and potentially inundating OWTS. Future studies could focus on coupled groundwater flow and contaminant transport modelling, evaluating the impact of contaminant transport from inundated OWTS filters towards recreational and drinking water supplies. Critical contaminant pathways for OWTS contaminant transport should be determined to identify potential areas of rapid contaminant transport. Finally, continued monitoring of shoreline water quality in vulnerable regions will be useful for understanding contaminant sources and adaptations required to limit future contamination.

3.7 – Acknowledgements

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CHAPTER 4 – CONCLUSION

This thesis highlights important risks that climate change pose to OWTS along low-elevation coastlines threatened by high rates of sea-level rise or increasing recharge. We show crAssphage to be an indicator of pollution from OWTS, where classic fecal indicators fail to be source specific or sensitive. We also employed SEAWAT to highlight the risks that SLR and enhanced recharge could have on the future subsurface inundation of OWTS, showing elevated risk of inundation with climate change for properties within 6.25 meters of MSL.

In Chapter 2 we demonstrate that crAssphage is a novel indicator of fecal pollution derived from OWTS and could be useful in pollution monitoring as a more sensitive and specific indicator of viral pollutants from anthropogenic sources. This sensitive indicator would enable earlier detection of pollution and help inform adaptation policies for coastal water management before waters are critically contaminated. As climate-driven changes in coastal ecosystems (e.g., SLR, storm surges, erosion) and population migration towards coastlines both place enhanced stress on coastal water quality, viral indicators are important complements to standard measures of water quality.

The numerical modeling in Chapter 3 that simulated future inundation of coastal OWTS due to climate change provides insight into which OWTS are most at risk with climate change, and can be used to help inform adaptation strategies. For example, the findings presented in Chapter 3 that homes reliant on OWTS and within 6.25m of MSL are at greater risk of their OWTS being inundated from SLR can be used to identify vulnerable areas within the province. An initial attempt to identify these potentially vulnerable areas in Nova Scotia is shown in Figure 16 Vulnerable regions highlighted in the map were selected visually by scanning the shoreline for communities with a high density of properties within 6.25 m of mean sea level, and not serviced by municipal wastewater treatment. This map qualitatively highlights areas of the province that are particularly susceptible to the impacts of SLR on OWTS, which in turn could impact coastal water quality in these areas.

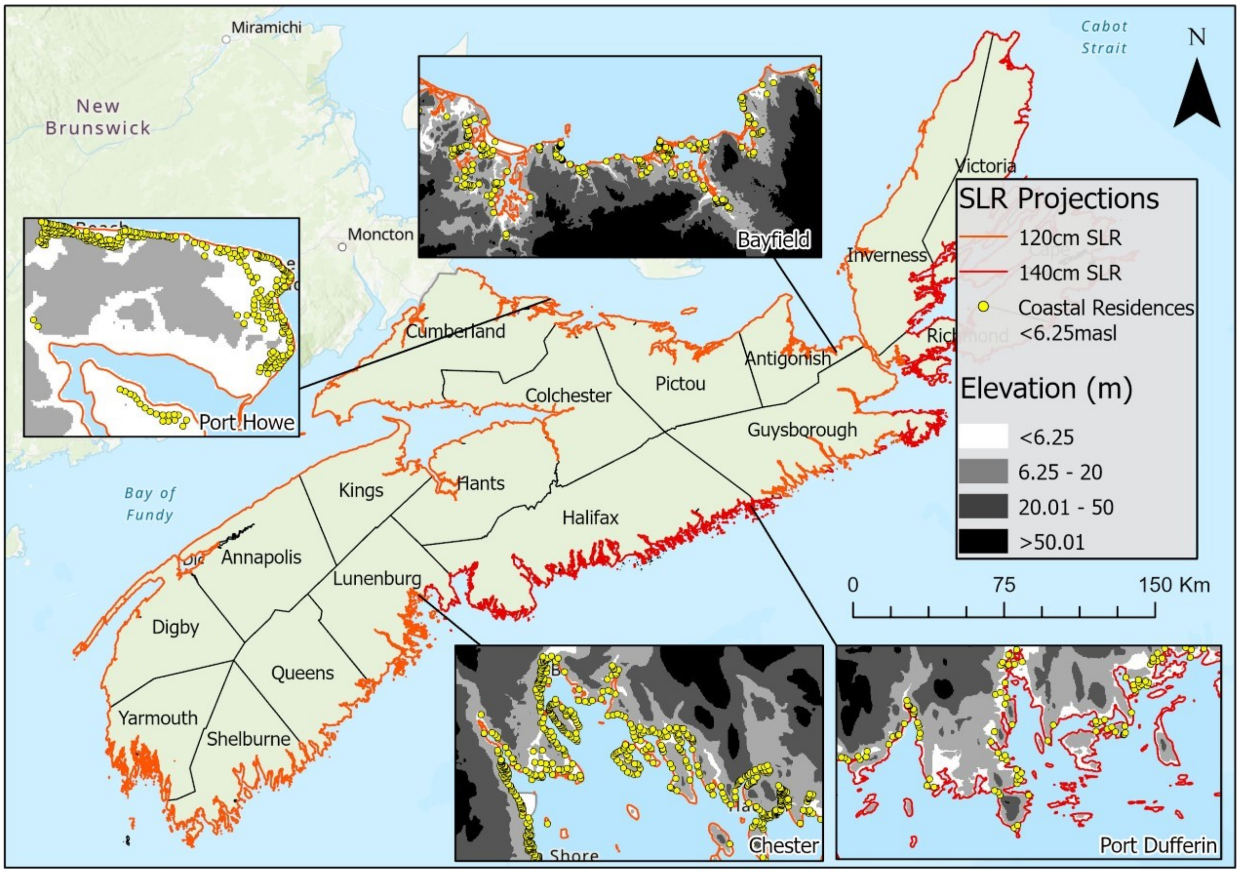

Figure 16 Map of Nova Scotia showing estimated SLR under the RCP 8.5 95th percentile scenario. Map insets show densely populated areas of the low-elevation coastline most susceptible to OWTS inundation with SLR.

There are, however, a variety of considerations aside from elevation that influence the risk OWTS inundation poses to coastal surface and groundwater quality. For example, regions of the south shoreline that experience greater mixing from storm activity and exposure to the Atlantic Ocean would rapidly dissipate contamination from SGD. Also, sites along the mega-tidal Bay of Fundy would also likely experience considerable mixing. The surficial geology that OWTS effluent flows through also plays a critical role in groundwater flow pathways and contaminant transport dynamics. Finer grained soils could result in longer travel times of effluent, resulting in enhanced retention and elimination (Stevik et al., 2004). Other hydrogeological factors that play a role in the water table response to SLR, namely, flux controlled vs. head controlled systems, are presented in Section 1.1.6.

Three directions for future work are recommended. First, additional studies validating the use of crAssphage in coastal surface and groundwater to detect faecal contamination should continue.

The preliminary research described in this thesis highlights a potentially important tool, but future work to relate crAssphage levels to detections of actual pathogens (viruses and bacteria) will be important for this indicator to become an effective method of water quality monitoring. Second, coupled groundwater flow and contaminant transport (from the OWTS filters themselves) models will allow for an enhanced evaluation of risk to coastal water quality. Preferential flow pathways will likely play a significant role in contaminant transport through SGD to coastal surface water. Identifying these pathways for at-risk locations will be an important step in informing adaptation strategies. Localized surficial geology and storm surge dynamics will also be important parameters in future groundwater flow models to accurately understand travel times of contamination and potential degree and frequency of inundation. Finally, a robust quantitative province-wide assessment of shorelines vulnerable to OWTS-derived contamination, including additional vulnerability criteria such as geology and shoreline mixing, would provide more accurate identification of at-risk coastal surface waters.

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APPENDIX

A 1 Field photographs of installed seepage meter (left) and seepage meter bag full of water from SGD.

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